GREAT LAKES COASTAL WETLANDS MONITORING AND ASSESMENT TECHNIQUES

By
KRISTINA KOSTUK, B.Sc.

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AUTHOR:	**Kristina Kostuk, B.Sc. (McMaster University)**
SUPERVISOR:	**Dr. Patricia Chow-Fraser**
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GENERAL ABSTRACT

The overall objective of this study is to contribute to general knowledge on bioassessment of Great Lakes coastal wetlands. Coastal wetlands (also referred to as marshes) are unique systems that experience day-to-day changes due to storms, high winds, and rapid changes in barometric pressure, exposing the shorelines to wave conditions; in addition to this annual and seasonal water level flucations contribute to this distinctive ecosystem.

The first chapter examines the influence of gear type and sampling protocol on fish catch data that are used to calculate biotic indices of wetland quality in Lake Huron. We surveyed fish communities in coastal wetlands of eastern Georgian Bay and Long Point Bay, Lake Erie, to determine biases associated with different gear types and sampling protocols. Parallel data collected from 26 wetlands were used to compare species richness obtained by two standardized protocols: fyke nets (set for 24-h parallel to shore) and boat electrofishing (1500 shock seconds during the day). We found differences between sampling protocols with respect to abundances and type of fish caught. Despite this difference, Wetland Fish Index (WFI; Seilheimer and Chow-Fraser 2006) scores derived from data obtained by the two gear types did not differ significantly. By contrast, when data for 6 exposed sites dominated by Scirpus were compared separately, we found significantly higher WFI scores associated with fyke net data compared with electrofishing data, and these differences were sufficiently large that they should not be ignored. We conclude that both methods can be used interchangeably in

routine ecological assessments, as long as methods are used within areas of dense submergent vegetation.

The second chapter used zoobenthos as a bio-indicator of wetland quality. "Zoobenthos" used in this study refers to the invertebrate primary and secondary consumers that are found associated with the sediment-water interface, and includes some of the zooplankton (copepods, cladocerans), which are found floating in the water column and many of the benthic invertebrates that reside on top of the sediment or that emerge from the sediment during the 24-h incubation period. It does not include any of the macroinvertebrates that live in emergent vegetation or that glide on the surface tension of the water. We determined that both water quality and aquatic macrophytes significantly influenced the distribution of zoobenthos. However, we also found that exposure also affected the type of invertebrates found in wetlands, regardless of water-quality conditions. We developed 26 metrics that could be used by wetland managers to assess wetland quality based solely on taxonomic composition of zoobenthos.

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GENERAL INTRODUCTION

Wetlands

According to the U.S. Environmental Protection Agency (2002), wetlands are areas where the water table occurs above or near the land surface for at least part of the year, and when open water is present, the depth must not exceed 2 m, whether it is stagnant or slow moving. These areas have hydric soils because of the presence of excessive amounts of water, and selects for the establishment of water tolerant (hydrophytic) plants. The Ontario Ministry of Natural Resources recognizes four types of wetlands: marsh, swamps, bogs and fens (Ontario Ministry of Natural Resources 1993) based on the hydrology, and type of vegetation found in the wetland. According to Environment Canada (1993), wetlands are considered one of the most productive ecosystems on earth, rich in wildlife and provide critical habitat for fish, especially when they occur along the coast of large lakes. Other services they provide to us include enhancement of water quality, erosion and flooding control, and a source of aesthetic and recreational enjoyment.

The Laurentian Great Lakes and their coastal wetlands

The Laurentian Great Lakes basin is found in one Canadian province and eight U.S. states, each of the Lakes are among the world's 12 largest lakes. It contains the largest freshwater system on earth (18%); only the polar ice caps contain a larger amount. Despite being part of the same hydrologic system, each of the five Great Lakes is large and unique. Lake Ontario (19,530 km^2) the smallest of the five, Lake Erie (25,745 km^2), the southernmost lake, Lake Huron (59,595 km^2) the second largest and includes Georgian Bay (15,500 km^2) which is often referred to as the 6th Great Lake, Lake Michigan (58,015 km^2) the only one entirely in the U.S. and Lake Superior (82,415 km^2) the northernmost lake and the largest of the five. The geographical expansion of the basin provides a variety of habitat types that range from the Canadian Shield (granite bedrock) in the northern and northwest portions of the basin, and sedimentary rocks in the southern and eastern portions of the basin.

About 10,000 years ago the first human inhabitants of the Great Lakes basin arrived. It wasn't until the 16th century, when the first European settlers came, that the landscape changed irrevocably. Large portions of the watershed were logged for building houses and ships, to provide land for agriculture, towns and cities, and eventually canals were built to enhance transportation, and this led to the establishment of non-native invasive species. During the1900s, human population growth in the basin increased dramatically, particularly in the southern lakes, Erie and Ontario, and Lake Michigan. To feed this increasing population, more land was cleared, and more intensive agriculture was practiced, and more cities were built. Almost all of these activities took place near the shoreline, close to the water source. As a result, there has been a loss of up to 90% of baseline wetlands, particularly in heavily populated areas (Jude and Pappas 1992, Dahl 1990). In southern Ontario alone, over 14,000 km^2 (61%) of original wetlands have been converted since European settlement (Jude and Pappas 1992).

Why are coastal wetlands important?

A total of 216,743 hectares (535,584 acres) of coastal wetlands have been identified in the Great Lakes basin (including St. Lawrence River) (State of the Great Lakes 2005). These coastal wetlands play a vital role in part of or all of their life cycle for many wildlife and plant communities. Many species including fish, invertebrates, amphibians, birds, and mammals, utilize the diverse habitats found in these wetlands for spawning grounds, nurseries, feeding areas, breeding and migration, etc (Maynard and Wilcox 1997, Jude and Pappas 1992).

The Great Lakes basin supports a diversity of coastal wetland types including lacustrine, riverine, and estuarine systems, which are all under the hydrologic influence of the Great Lakes (Chow-Fraser and Albert 1999). Some wetlands are found in protected embayments (often found in the lower lakes) while others are found on exposed shorelines of the lake (many found in Georgian Bay and the North Channel). The different geomorphologic classes are influenced by wave exposure, storm surges and lake level change, all of which affect the type of vegetation found there (Albert and Minc, 2001). Lougheed *et al.* (2001) also found that deterioration in water quality through cultural eutrophication can decrease the species richness of the submergent plant communities, and this change in habitat structure and productivity can greatly influence how fish and wildlife will use the wetland.

Utilization of coastal wetlands by fish and wildlife has been actively researched over the past decade (e.g. Jude and Pappas 1992; Brazner 1997; Tanner *et al.* 2004). The high primary production and the correspondingly high diversity of invertebrate prey is one of the main reasons why wetlands provide great spawning and nursery habitats for the Great Lakes fish community (e.g. Wiley *et al.* 1984; Jude and Pappas 1992). That is why diversity in aquatic vegetation tends to be associated with more diverse fish assemblages that include many prey species of minnows, shiners, and sunfish, as well as predators such as northern pike, largemouth bass and yellow perch (Savino and Stein 1982, Stephenson 1990, Brazner 1997, Tanner *et al.* 2004, Jude and Pappas 1992).

Thesis objectives

Ecological issues regarding coastal wetland ecosystems remain understudied despite the importance of this habitat to the overall health of the Great Lakes ecosystem. In the 1996 State of the Lakes Conference, Maynard and Wilcox (1996) noted a dire need for development of standardized methods that can be used by environmental agencies in routine monitoring programs to assess the ecological health of the remaining coastal wetlands, and to track their changes through time. Since then, there have been proposals to use benthic invertebrates as an indicator (e.g. Kashian and Burton 2000, Wilcox *et al.* 2002) but these studies have been limited to small stretches of a single Great Lakes shoreline. Lougheed and Chow-Fraser (2002) carried out their research throughout the Great Lakes basin to develop an indicator based on zooplankton, and this large-scale synoptic approach yielded some general insights into ecological functions of coastal wetlands that have eluded investigators with a much narrower geographic focus.

For my thesis, I have chosen to write two chapters pertaining to development of ecological indicators, while adopting the large-scale approach. For my first chapter, I will examine the influence of gear type and sampling protocol on fish catch data that are used to calculate biotic indices of wetland quality in Lake Huron. According to Chow-Fraser et al. (2006), there is a gear-specific bias on the species richness of catch data associated with wetland quality for wetlands in Lakes Erie and Ontario, when two commonly employed gears were compared. The gears in question are fyke nets (FN; set overnight), and boat electrofishing (EB; conducted only during the daytime). One objective of Chapter 1 is to confirm this trend for Lake Huron, especially for wetlands in eastern and northern Georgian Bay and the North Channel, where there is only a small range in wetland quality (Chow-Fraser 2006). A second objective is to compare data generated by the two gear types to determine if there are consistent and predictable differences that could be attributed to common sampling protocols that are used by different agencies. My third objective for Chapter 1 is to determine if data collected by standardized protocols for each gear type would result in significant differences in Wetland Fish Index (WFI; Seilheimer and Chow-Fraser 2006). The rationale is that significant differences in catch data stemming from differences in gear used can be ignored if they lead to similar conclusions regarding the quality of the wetland. For my second chapter, I will test the feasibility of using zoobenthos (within areas of dense submergent vegetation).

For the second chapter, I will investigate the feasibility of using zoobenthos as a bio-indicator of wetland quality. “Zoobenthos” used in this study refers to the invertebrate primary and secondary consumers that are found associated with the sediment-water interface, and includes some of the *zoo*plankton (copepods, cladocerans), which are found floating in the water column and many of the *benthic* invertebrates that reside on top of the sediment or that emerge from the sediment during the 24-h incubation period. I will use samples that have been collected throughout the Great Lakes shoreline to determine how exposure disturbance from wind and wave action, and the impact of water-quality impairment affect the type of invertebrates found in wetlands.

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Chapter 1: Influence of gear type and sampling protocol on fish catch data for calculating biotic indices of wetland quality in Lake Huron

By

Kristina Kostuk

and

Patricia Chow-Fraser

ABSTRACT

We surveyed fish communities in coastal wetlands of eastern Georgian Bay and Long Point Bay, Lake Erie, to determine biases associated with different gear types and sampling protocols. Parallel data collected from 26 wetlands were used to compare species richness obtained by two standardized protocols: fyke nets (set for 24-h parallel to shore) and boat electrofishing (1500 shock seconds during the day). We found differences between sampling protocols with respect to abundances and type of fish caught. Despite this difference, Wetland Fish Index (WFI) scores derived from data obtained by the two gear types did not differ significantly. By contrast, when data for 6 exposed sites dominated by *Scirpus* were compared separately, we found significantly higher WFI scores associated with fyke net data compared with electrofishing data, and these differences were sufficiently large that they should not be ignored. We also conducted parallel studies to determine the effect of different sampling protocols on catch data. We found no significant differences in either species richness or WFI scores when 'parallel' vs. 'perpendicular' sets of fyke net data were compared. In addition, we found that results from daytime boat electrofishing were significantly influenced by the amount of submergent vegetation at the site, the manner in which electricity was applied (point versus transects), and the level of effort expended (i.e. total shock seconds applied). We conclude that both methods can be used interchangeably in routine ecological assessments, as long as methods are used within areas of dense submergent vegetation.

INTRODUCTION

Standardization of fish sampling has been stated as an issue for sampling streams, rivers, wetlands and lakes (Bonar and Hubert 2002), and reflects the importance of accurate assessments of fish biomass and population size in fisheries research and management (Bayley and Austen 2002). Thus, the management objective of many environmental agencies is to characterize both the species and size of the fish community within their jurisdictions. Since every gear that has ever been invented is selective to a certain degree, it is important to know exactly how different gear types influence catch for specified species and size of fish and for given habitats. Hardin and Conner (1992) has also attributed differences to the level of experience of the sampling crew.

A variety of sampling gears have been employed to sample fish communities. These include electrofishing, often used to characterize sportfish (e.g. Hardin and Conner 1992), and gill nets, which are commonly used to assess fish communities in lakes and reservoirs (e.g. Krueger *et al.* 1998). Often, more than one environmental agency has the shared responsibility for assessing fish populations in large watersheds. Data sharing allows different programs, environmental agencies and researchers with limited funds to build an appropriately large database to properly manage these large watersheds (Cao *et al*. 2005). However, this requires consistent choice of gear as well as the way they are deployed, to ensure that data can be validly compared and analyzed statistically (Bonar and Hubert 2002; McGeoch and Gaston 2002).

Currently, fisheries managers across the Great Lakes region use both active and passive gears. Active gears include seining and electrofishing (boat or back-pack); these capture techniques are those that include capture of fish by sieving them from the water by means of mesh panels or bags (Hayes 1989) or passing a field of electricity through the water that causes a muscle response from the fish (Couchman 2003). Passive gears include the use of hoop nets, minnow traps, or fyke nets to capture fish by entanglement or entrapment devices that are not actively moved by man or machine (Hubert 1989). Even when the same gear is used, however, results may differ depending on the protocols employed. Protocols are often based on familiarity with a method, budget, and sampler availability (Resh and McElravy 1993, Carter and Resh 2001, Bonar and Hubert 2002). Differences in protocols can affect both the size and species being caught (Claramunt *et al.* 2005), which can severely limit the ability of agencies to pool their data for co-management purposes. Therefore, a study that considers the effectiveness of different sampling gear must also consider differences in how the gears are commonly used by the constituents.

Ever since the introduction of the fish-based Index of Biotic Integrity (IBI) (Karr 1981), other indices have been developed for streams (Simon and Lyons 1995), Great Lakes Areas of Concern (Minns *et al.* 1994, Simon and Lyons 1995), as well as coastal wetlands of the Laurentian Great Lakes (Wetland Fish Index (WFI); Seilheimer and Chow-Fraser 2006). These tools offer a relatively quick and effective way for managers to assess habitat quality on a routine basis, and to track changes to increased human disturbance, or to remedial actions (Seilheimer and Chow-Fraser 2006; Seilheimer *et al.*, in submission). Reliability of these indices depend greatly on the type of information used, which in turn depend on the gears and protocols used to collect the fish information. Nevertheless, the extent to which differences in sampling protocols can contribute to conclusions drawn from such indices has seldom been addressed in the literature.

There are three main objectives of this study. First, we want to compare results generated by two commonly employed gears for surveying fish assemblages in coastal wetlands, to determine if there are consistent and predictable differences associated with different protocols for each gear type. The two gears involved are fyke nets, a passive-sampling device, and boat electrofishing, an active-sampling device, both of which have been used extensively by environmental agencies and researchers to sample in the Great Lakes basin (Chow-Fraser *et al.* 2006). The protocols used with each gear are known to vary among agencies. For instance, fyke nets can be set up in parallel (e.g. Brazner 1997, Seilheimer and Chow-Fraser 2006) or perpendicular (e.g. Wilcox *et al.* 2002) orientations to the shoreline, while the electrofishing can be conducted at specific points for a given duration or continuously along a transect of given lengths (Brousseau *et al.* 2003). Since the extent of differences associated with these different protocols have not yet been determined in parallel trials, there is little basis for differentiating among datasets on the basis of sampling protocol.

Our second objective is to determine if data collected by standardized protocols for each gear type would result in significant differences in Wetland Fish Index scores (Seilheimer and Chow-Fraser, 2006). The rationale is that significant differences in catch data stemming from differences in gear used can be ignored if they lead to similar conclusions regarding the quality of the wetland. Finally, we want to compare results of this study to a similar study that had been conducted by Chow-Fraser *et al.* (2006) on wetlands primarily in Lakes Ontario and Erie. Wetlands sampled in the present study are primarily located in Lake Huron, particularly in eastern Georgian Bay and the North Channel, and are known to be in relatively good quality compared with those in the lower lakes (Chow-Fraser 2006). We want to determine if the gear-specific bias in taxon richness associated with wetland quality (Chow-Fraser et al. 2006) can be confirmed for a dataset that does not have as large a range in wetland quality.

METHODS

Fyke Net versus Electrofishing Comparison

Study Sites

Forty Great Lakes coastal marshes were sampled from 2004 through 2005 on Lakes Huron and Georgian Bay (Figure 1-1); all 40 were sampled by electrofishing boat and 26 by fyke nets (Table 1-1). Study sites ranged in wetland quality based on the Wetland Water Quality Index (WQI) (Chow-Fraser 2006; Table 1-1). Wetlands in the study ranged from excellent quality to moderately degraded.

Sampling Technique

Fyke Nets - Two sets of large nets (height x width=3' x 4' rectangular opening; 10' long; one net 1/2" and the other 3/16" nylon mesh), one small set (3' x 1' rectangular opening; 3/16" nylon mesh for both) were placed in submergent vegetation communities. If no submergent communities were available then nets were set along emergent vegetation.

Large nets were set at depths from 0.5m to 1m depths and small fyke net were set in water from 0.25m to 0.5m deep.

All nets were set parallel to the shoreline, with the wings (3' x 10'; 3/16"mesh) extending out at 45º angles from the net opening, with one side along the emergent plant communities and the other side extending out towards the open water. The 25' lead (3/16" mesh) connected the paired nets. Nets, wings and lead were secured in place using six 10" steel tubing. These nets remained in the water for a 24-hour period (allowing for both daytime and overnight). Fish were removed from the FN and placed into a plastic container filled with water for identification and measurement (see Data Collection and Recording). The location (latitude and longitude) of all the fyke net sets was recorded with a Garmin Etrex Summit hand-held global positioning system.

Electrofishing **–** Electrofishing was conducted with a Smith-Root SR-16EB electrofisher with a Mercury 60 hp 4-stroke outboard motor and 7.5 GPP generator. Two round, 1-meter diameter LPA-6 anode arrays were extended on a pair booms mounted on opposite sides of the bow of the boat at 25° from the center. Electrofishing settings were 60 pulses per second DC current, with a power output maintained at 2000 Watts (400-1000 Volts, 2.5-7 Amperes). Boat speed was maintained at idle, allowing netters to obtain stunned fish. Transects were typically within 1m of shorline/emgergent vegetation, within submergent vegetation (wherever possible) at 0.5 to 1 m in depth. Normally three transects (at larger sites, four transects were) were completed at an average of 300 to 500 shock seconds, for approximate total of 1000 to 1500 shock seconds. At some sites, less effort was expended because of the small size of the wetland. Each transect was on average 240 m in length. During sampling, one person retrieved fish at the bow of the boat, and boat operator maintained settings and retrieved any missed stunned fish. All netted fish were placed into a live-well on board. Upon completion of each transect, fish were identified and measured (see Data Collection and Recording), and released at the site. See Table 2 for settings of individual wetlands. All transects locations and lengths were recorded using a Garmin Extrex Summit hand-held global positioning system.

Data Collection and Recording

Fish were all identified to species, except for some young-of-the-year (YOY) species (e.g. *Lepomis spp*.), counted, and measured to the nearest millimeter (mm). If more than 15 individuals of a species were present then 15 randomly chosen individuals were measured. In some cases counts for a species may be divided into two size classes for convenience, i.e. YOY (less than 90 mm) and juvenile/adult (greater than 90 mm) (e.g. Yellow Perch, *Perca flavens*). In such cases, 10 randomly chosen individuals were measured. Measured fish were released unharmed at the capture site. Biomass was estimated with length-weight regressions (Schneider et al. 2000).

Functional Taxa Categories

Fish species were placed into six categories based on their life stage (Chow-Fraser *et al.* 2006). The categories included: Piscivorous, Carnivorous (mainly insects and other invertebrates in diet), Omnivorous (consuming algae and zooplankton), Benthivorous (primarily benthic invertebrates and other organisms that reside in the sediment), Herbivorous (mainly algae and plant material,) and Planktivorous (eating primarily zooplankton).

Comparison of fyke-net orientation

Study Sites

This portion of the study took place in three coastal wetlands during the summer of 2005 (Figure 1-2). Two of the sites are located in southern Georgian Bay on the north and south side of Green Island (Figure 2-inset). The Green Island north (GI-N) site was *Scirpus*- dominated with no defined shoreline, and had mats of submergent vegetation scattered in various densities. By comparison, the Green Island south (GI-S) had a defined *Scirpus*-shoreline, with dense submergent vegetation extending out from the edge of the shoreline. At Long Point Provincial Park (LP-PP) in Lake Erie, sampling took place near the neighboring sand spit (Figure 1-2). The site contains a mix of *Scirpus* and *Typha*, with a moderate to dense cover of submergent vegetation.

Fyke nets were set in pairs, with their mouth openings facing each other, and connected by "leads" (Figure 1-3a). The nets also had "wings" that extended outwards at 45° angle to help corral fish towards the opening along the lead. These nets can be set in an orientation that is parallel (Figure 1-3b) (e.g. Brazner 1997, Seilheimer and Chow-Fraser 2006) or perpendicular (Figure 1-3c) (e.g. Wilcox *et al.* 2002) to the shoreline. Sampling always occurred over 3 days. On Day 1, nets were set parallel (same protocol used as fyke net versus electrofishing study) to the shoreline, on Day 2, two nets were processed and re-set in a perpendicular orientation (in the same spot) to the shoreline, and fish captured in these nets were processed on Day 3. At Long Point Provincial Park there was a seiche that occurred during the night leading into the third day. This resulted in an increase of water levels by approximately 1 foot and the large mesh of one of the large nets collapsed.

Comparison of electrofishing techniques

Study Sites

The Electrofishing study occurred in two coastal wetlands in southern Georgian Bay during the summer of 2005. The two sites were located on the north and south side of Green Island (see fyke net comparison for full description, Figure 1-2-inset).

Point Sample **–** Point sampling was conducted at Green Island-South at 4 different depths (0.5m, 1m, 1.5m, 2.0m) with 5 points at each depth (Figure 1-2a). Electrofishing settings were set at 60 pulses per second DC current, with a power maintained at approximately 2000 to 2400 Watts (500 Volts at 50-60%, 8 Amperes). 20 points were established along a line. When we reached each point, we stopped the boat and delivered shocks for 10 seconds. Therefore, a total of 200 shock seconds were delivered with this protocol.

Transects - Transects were taken at the Green Island-South and Green Island-North sites, parallel to the shoreline, at depths that varied from 0.50 m to 1.5 m. The percent cover of vegetation along the transect was recorded as follows: 0 = 0%; 1 = 1 to 19%; 2 = 20 to 70%; and 3 = >70%; adapted from Brousseau *et al.* 2003). Electrofishing settings were set at 60 pulses per second DC current, with power maintained at approximately 2000 to 2800 Watts (500 Volts at 60-70%, 6 to 8 Amperes).

Water quality analyses

All water sampling and measurements of physical and chemical parameters were conducted inside a canoe or boat (depending on depth of the water). We measured temperature, pH, specific conductivity and dissolved oxygen concentration *in situ* with several a YSI™ 6600 probe with 650 display (YSI, Yellow Springs, Ohio, USA). All sensors for the instruments were calibrated on a weekly basis. Sampling was always conducted during daylight hours (generally between 0900 and 2000). Geographic coordinates of the sites were taken with either a Trimble™ GPS (4-5 m accuracy) or a Garmin™ Etrex GPS (4-6 m accuracy).

Water was collected for nutrient and turbidity analysis in 1-L van Dorn bottles at mid-depth in water outside the submergent plant zone. Water for nutrient analysis was dispensed into clean Nalgene™ bottles (acid washed and rinsed with deionized water). Water for chlorophyll analysis was dispensed into opaque Nalgene™ bottles. All samples were stored on ice in a cooler and were analyzed later that day at the field lab. Water for the turbidity analysis was collected in an identical manner and was measured in the canoe, with a Hach™ 2100 Portalab. Methods used for processing samples in the field and the laboratory have been documented in detail elsewhere (Chow-Fraser 2006).

Statistical Analyses

Statistical analyses were performed with SAS JMP IN 5.1 software, and included ANOVA, Wilcoxon-signed rank test, and linear regression analysis. For graphs displaying abundance and biomass, data were first $\log_{10}$-transformed to normalize the data.

Wetland Fish Index

The Wetland Fish Index (WFI) was developed by Seilheimer and Chow-Fraser (2006) to assess the degree of human disturbance associated with water-quality impairment in coastal wetlands of the Great Lakes. The index used either presence/absence (WFI-PA) or abundance (WFI-AB) fish data. We generated WFI scores for data collected in each of 26 wetlands to determine the effect of gear type on biotic indices.

Water Quality Index

We used the 12-parameter equation in Chow-Fraser (2006) to calculate WQI scores for each of the 26 wetlands that had been fished with both gears, and these allowed us to assign a status to the wetlands as follows:

WQI Score	Category
+3 to +2	Excellent
+2 to +1	Very good
+1 to 0	Good
0 to -1	Moderately degraded
-1 to -2	Very degraded
-2 to -3	Highly degraded

RESULTS

Fyke Net versus Electrofishing Boat Comparison

A total of 10,398 fish, representing 45 species and 16 families, were caught between the two gear types from the 26 wetlands (Table 1-3). FN yielded consistently higher catch per unit effort for both abundance and biomass data; 71.9% of the total catch could be attributed to FN, compared with 28.1% for EB. There were marginal differences in the total number of species and functional taxa recovered between the two gears (FN 42 and 46 and EB 40 and 43). Common species included bluntnose minnow, brown bullhead, pumpkinseed, rockbass, blackchin shiner, and juvenile yellow perch (Table 1-4). No significant differences were shown with respect to total richness between the two gears (12.9 versus 11.2 for FN and EB; Table 3).

We then sorted the data into functional feeding categories to further compare the two methods. FN caught significantly larger fish in the piscivore and carnivore categories, both with respect to mean weight and length, and the same was true for benthivore length (Figure 1-4a and b). There were no significant differences between gear types for the omnivore and planktivore categories (likely due to smaller numbers caught).

Data for each wetland were compared once the fish were sorted by functional taxa (Table 1-5). FN tended to catch a significantly higher number of species compared with EB (Wilcoxon Sign Test; P=0.021). For 11 of the sites, FN captured at least twice as many unique species (Moose Bay, Jumbo Bay-04, Wardrope Island-04, Ojibway Bay, Wardrope Island-05, Garden Channel, Robert's Bay, Green Island-04, Green Island-05, Charles Inlet, Boom Camp Bay). By comparison, there were only 5 sites where twice as many unique species were captured with EB (Iroquois Island, Dogfish Bay, Dead Horse Bay, Vincent's Bunk, Hog Bay).

Overall, both gears were efficient at catching all functional groups in individual wetlands, with the exception of 6 *Scirpus*-dominated sites, where there was a bias towards the FN method (Moose Bay, Jumbo Bay, Wardrobe Island, Ojibway Bay, Garden Channel, and Robert's Bay. In these *Scirpus*-dominated systems, FN always caught a significantly greater number of fish compared with EB (mean of 12.5 versus 7.9 for FN and EB; Wilcoxon sign test, P=0.004).

We calculated WFI scores for each of the 26 site-years for both techniques. Despite the apparent difference in catch data for the two methods, there were no significant differences between mean WFI scores associated with either presence/absence or abundance data (Paired T-test; P=0.20 and P=0.240; Table 1-6). However, when we focused only on the 6 *Scirpus*-dominated sites, we found significant differences between methods for both presence/absence and abundance data (Wilcoxon Signed Rank Test; $P<0.05$; Table 1-6).

Comparison of fyke net orientations

A total of 1,595 fish were caught; parallel setting caught 62% of the total, while the perpendicular setting caught the remaining 38% (Table 1-7). However, there were no significant differences between the numbers of species recovered between the two orientations (Table 1-8), despite the disparity in total number of species caught (Table 1-7). We were also unable to find significant differences between orientations with respect to WFI scores calculated with either presence/absence or abundance data.

Comparison of electrofishing sampling protocols

Point versus Transects Samples

A total of 1543 shock seconds were applied while conducting the three transects. Transect 1 was conducted at the edge of the *Typha* and *Scirpus* (0.75 m to 1.5 m), Transect 2 was conducted within *Scirpus* and moderate densities of submergent vegetation (0.5 m to 1 m), while Transect 3 was conducted outside the *Scirpus* edge within moderate densities of submergent vegetation (1.5 m) (Table 1-9 and Figure 1-2). Approximately 500 shock seconds were delivered at each transect, with a total of 200 shock seconds being applied at the end of the 20-point line sampling (Table 1-9 and Figure 1-2). Each of the points were found within the same areas of the 3 transects described above.

We caught 8 times more fish and twice as many species with the transect protocol compared with the point-protocol (Table 1-9). However, regardless of the protocol used, we could not find any significant difference between mean lengths and weights (t-test; P-0.18 and 0.55, respectively). These data were sorted by species and compared again (Table 1-10). The Transect protocol was much more effective than the Point protocol in capturing the full range of species, with 11 species being recovered exclusively by transect, compared with only one being recovered exclusively by the Point protocol. Given these differences between protocols, it was not surprising to see that the Transect protocol yielded WFI scores that were significantly higher than those for the Point protocol (Wilcoxon signed rank test, $n=3$; $P<0.05$) (Table 1-11).

Effect of sampling effort

We conducted a study at the Green Island-North site to determine the relationship between catch and degree of electrofishing effort (Table 1-12). A total of 4600 shock seconds (7 transects with durations ranging from 300 to1500 shock second) were delivered in 5 continuous trials. These transects occurred in water ranging in depths from 0.5 m to 1.5 m, in sparse to moderate submergent vegetation along the shoreline and within the *Scirpus* bed. As sampling effort increased, cumulative catch and biomass increased linearly (Figure 1-5a). By comparison, the number of conventional and functional taxa increased non-linearly in an asymptotic fashion, and began to level off at 2000 shock seconds (Figure 5b). Despite an increase in number of fish caught, there were no effective differences in WFI scores corresponding to 1500 to 4600 shock seconds for either presence/absence and abundance data (Table 1-12).

There were some note-worthy differences in the relationship between cumulative catch and sampling effort for several fish species (Figure 1-6). For instance, slopes of the regression for three most common species, pumpkinseed, yellow perch, and bluntnose minnow, were much higher than those for the less commonly encountered species such as Iowa darter, johnny darter and logperch. Therefore, the reason we see a disproportionately high representation of the pumpkinseed and yellow perch in our dataset is probably because they are caught more effectively per unit effort. By comparison, there was no effect of sampling effort on catch of longnose gar.

There were similar positive correlations between richness, catch and biomass with sampling effort (total shock seconds) for 40 wetlands we surveyed with the electrofishing boat (Figure 1-7a to c; Table 1-2). We found significant linear relationships for all three parameters when all data were included ($P<0.05$), although clearly, there was a great deal of unexplained variation (r^2 values varied from a high of 0.37 to a low of 0.10). By accounting for differences in site characteristics, we may improve these relationships. For instance, data for the 6 *Scirpus*-dominated sites did not appear to vary with electrofishing effort (solid squares in Figure 1-7), possibly because these sites are highly exposed and have very sparse submergent vegetation. Therefore, we carried out further analyses to determine how data obtained by the two different gear types are influenced by variation in submergent plant density.

Effect of plant density and plant type

We pooled the data from the 40 wetlands that we electrofished to examine the effect of plant density and plant type on fish catch (Figure 1-8). To enable valid comparisons, we first standardized the data according to fishing effort (divided the data by the number of shock seconds). The amount of vegetation encountered in each of 3 transects conducted in the 40 wetlands were assigned a value of 0 to 3 as described in the Methods. The wetlands were also sorted according to site characteristics, where sites dominated by *Scirpus* sp. with very little submersed aquatic vegetation (usually highly exposed) formed a group we referred to as "Scirpus", while all others formed the group we referred to as "Regular" (Figure 1-8). Both standardized richness and standardized catch were significantly higher (T-test, P=0.0029 and 0.0113, respectively) for the "Regular" compared with the "Scirpus" sites (Fig. 1-8 a and c, respectively). We also found significant differences in standardized richness and catch (Figure 1-8b and d, respectively) as a function of plant-density category (Figures 1-9c and d); richness and catch were highest in transects with >70% cover of submergent vegetation, and lowest in transects with little submergent vegetation (< 20% cover) (ANOVA; P=0.0004 and 0.0007, respectively).

Effect of water quality on fyke nets and electrofishing boat

Our final objective was to compare results of this study to data collected from coastal marshes located in the two lower Great Lakes (Chow-Fraser *et al.* 2006). We combined data from the 26 wetlands in eastern Georgian Bay and the North Channel (this study; open symbols in Figure 1-9) with those in 11 wetlands of Lakes Ontario and Erie (Chow-Fraser *et al.* 2006; closed symbols in Figure 1-9) to evaluate the relationship between proportion of total catch and WQI score (Figure 1-9; see WQI scores in Table 1). Consistent with findings of Chow-Fraser *et al*. (2006), we found that the amount of total taxa decreased significantly as the value of WQI scores increased for data collected by FN (Figure 1-9a), whereas the reverse was true for data collected by EB (Figure 1-9b). When data from this study was analyzed on their own, we found no effect of wetland quality for either EB or FN data (Figure 1-10).

DISCUSSION

Few studies have been conducted with as large a geographic coverage as this (40 wetlands located throughout eastern Georgian Bay and the North Channel), and in which the performance of two gears are compared in parallel over 2 field seasons. This study, together with Chow-Fraser *et al.*'s (2006), with complementary data for the two lower lakes, provide the most comprehensive comparison of FN and EB data for coastal marshes and other shallow habitats along the Great Lakes shoreline. Consistent with observations noted for the lower lakes, FN was associated with much higher total catch (72% and 66% of total abundance and biomass, respectively; Table 1-3), while EB was associated with slightly longer fish, especially with respect to piscivores and carnivores (Table 1-3 and Figure 1-4). The two gears also differed with respect to capture of unique species (Table 1-5). FN was clearly more efficient at capturing a range of functional taxa, and this was especially true when we sampled the *Scirpus*-dominated sites.

Therefore, we do not recommend using EB to survey exposed wetlands that are dominated by bulrush.

Despite these differences, we found no significant differences between gears when WFI scores were generated from the survey data. This finding must be reassuring to environmental agencies that need to share data with other jurisdictions to manage large watersheds. Agencies increasingly rely on the use of multimetric indices (e.g. Stream IBI, Karr 1981; Area of Concern IBI, Minns *et al.* 1994) to monitor ecosystem health of aquatic ecosystems. In study, we focused on the WFI (Seilheimer and Chow-Fraser (2006) because this index can be used with either abundance/catch data or presence/absence data, and has been used successfully to rank wetlands according to degree of water-quality impairment throughout the lower Great Lakes. We did not have sufficient time to expand our evaluation to include other indices, but we hope that future studies will be conducted to determine how indicators such as the IBI will perform when different gears are used.

Although we found a significant effect of fyke-net orientation on total catch (62 vs 38% for parallel vs. perpendicular orientations, respectively; Table 1-7), the number of recoverable taxa was similar. In practice, we know that sites with a defined shoreline and distinct vegetation zonations (e.g. Green Island-South) are more efficiently sampled with nets set parallel to shore, and this is borne out by the observation that 6 more species were caught with the parallel over the perpendicular set-up (Table 1-8). This may be attributed to diurnal migration of fish from littoral to pelagic areas for food (Gauthier and Boisclair 1997; Lewin *et al.* 2004) and schools of smaller fish that are seeking refuge from predators (e.g. yellow perch in Lake Mendota; Weaver 1993). The parallel orientation takes advantage of this horizontal movement, and may be more successful at herding fish into the net than perpendicular set-ups that are primarily targeting fish moving parallel to shore. Nevertheless, these two set-ups performed equally when data were used to generate WFI scores (Table 1-8), and there is no justification for rejecting data on the basis of fyke-net orientations for assessment purposes.

In this study, we carefully standardized the power output of the electrofishing boat to minimize bias and variation between point sample surveys and transect surveys. We found that point-sample surveys greatly underestimated the fish population, both with respect to total catch and species richness, especially for fish in the carnivore category (Tables 1-9 and 1-10). Compared with the point method, the transect method covered a greater area of the wetland and resulted in a much greater catch (126 vs. 16 for transect and point method, respectively), which resulted in significantly higher WFI scores (Table 1-11). Ironically, the sampling effort required (i.e. total time and labour) to conduct the point method was extremely high compared with the transect method. Garner (1997) found that point shocking gave similar results as seining in catching short fish in riverine habitats, but Brousseau *et al.* (2003) published similar results to what we found in this study. Therefore, we do not recommend using this protocol in coastal wetlands for general community assessments, although it may be suitable for surveys that target a particular species.

It is clear that total catch will depend on the amount of electrofishing effort (total shock seconds applied) (Figure 1-5a). Given this, it is important that interagency collaborations establish standardized minimal effort. Anagermeier and Smogor (1995) and Meador (2005) have discussed the importance of determining a suitable level of sampling effort. In Chow-Fraser *et al.*'s (2006) study, three different agencies were involved, and each chose a different protocol. In discussions with biologists from various agencies throughout the Great Lakes states and province, there was little consensus on the suitable level of effort required to sample coastal wetlands (levels from <100 to >5000 shock seconds have been mentioned to us). Based on our results, we propose 1500 shock seconds as an appropriate level of effort since total species richness was underestimated with effort <1000 shock seconds, and effort >2000 shock seconds resulted in diminishing returns (Figure 1-5b; Table 1-12).

We did not expect to see the species-specific responses to increased sampling effort as demonstrated in Figure 6. Total catch of the most commonly encountered species (e.g. pumpkinseed) increased disproportionately with total shock seconds compared with the rarer species (e.g. log perch and longnose gar). Despite these differences in total catch, we found that the WFI score corresponding to 1500 shock seconds was very similar to those generated by much higher sampling effort, and this reinforces our proposal to choose 1500 as the standard.

The importance of submergent plants as a component of fish habitat was clearly demonstrated in this study. Vegetated habitats in coastal wetlands provide refuge, nesting areas, nursery habitat, and abundance of prey items for both forage and sport fish (Wiley *et al.* 1984, Stephenson 1990, Jude and Pappas 1992, Brazner 1997, Tanner *et al.* 2004, Jacobus and Ivon 2005). Quite simply, the denser the macrophyte, the more abundant the fish (Weaver *et al.* 1997, Killgore *et al.* 1989, Chubb and Luston 1986), and the easier it is for EB to accurately survey the community (Figure 1-8). Others have suggested that very dense vegetation is less attractive to fish, and that more fish can be found in sites with moderate vegetation (Brazner and Beals 1997; Jacobus and Ivan 2005). We did not find this to be the case, since the highest capture by EB was associated with the highest density category (Figure 1-8), and this is in agreement with Killgore *et al.* (1989), who found similar results in the Potomac River, Virginia.

For sites dominated by *Scirpus*, FN was more efficient at capturing the full range of species than EB (Table 1-5). These sites tended to be highly exposed, where *Scirpus* is the only emergent plant, and where submergent vegetation is scarce. In these sites, fish must move frequently in search of food, or to avoid being eaten by piscivores, and under these circumstances, passive gear such as FN appears to be more effective. At the other extreme, vegetation can be so dense that fish movement is prohibited (Jacobus & Ivon 2005; Tanner et al. 2004), and when this happens, EB can be more effective than FN. This was demonstrated by data from Dogfish and Deadhorse, two sites with extremely dense vegetation, and where EB caught a higher number of functional taxa than did FN (Table 1-5). To increase the efficiency of FN at these sites, nets may have to be set over multiple days. to provide a proper representation of the fish community. This view of how performance of passive gear is affected by plant density agrees with the observations of Killgore *et al.* (1989), who found that pop nets caught a significantly higher number of fish at intermediate than at high plant densities.

Chow-Fraser *et al.* (2006) found a bias associated with survey method when wetlands varied across a pollution gradient. Fyke nets were more efficient in degraded conditions, while electrofishing was better in high quality sites. When we combined our data with theirs, we found this trend to be upheld (Figure 1-9). Had we considered only data from Georgian Bay and the North Channel, there would not have been a significant relationship (Figure 1-10), and this demonstrates the danger in drawing conclusions when only a portion of the degradation gradient is available.

GENERAL CONCLUSIONS

Many studies have concluded that more than one technique may be required to properly sample fish communities with a range of individuals from juveniles to adults (Conrow *et al.* 1990, Van Snik Gray *et al.* 2005*,* Jackson and Harvey 1997; Chow-Fraser *et al.* 2006). We agree with this sentiment, particularly if the goal is to fully characterize the species assemblage. It is clear from this study that some species will be missed when only one gear is used. However, if the goal is to use the survey data to assess habitat quality, then EB and FN would give similar results, especially when an index such as the WFI is used, an index that relies on the presence of ecological analogues to indicate the degree of water-quality degradation. FN data were not affected by the orientation of the net to shore (parallel vs. perpendicular), although more species were captured when nets were set parallel to shore at the site with a defined shoreline and a distinct vegetation zonations. When surveying with EB, the transect method was better than the point method, both with respect to total number of species recovered and the total catch. Total catch associated with EB also depended on the density of the submersed aquatic vegetation and the total effort expended (shock seconds delivered). Overall we recommend the use of fyke nets to sample coastal wetlands, as it is a more affordable option in both purchasing and training of staff, and although it requires more time to set-up and retrieve the fish, fewer factors can affect the results.

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Table 1-1. Summary statistics for wetlands sampled with fyke nets and electrofishing boat in this study. Asterisks indicate that the sites are dominated by *Scirpus* sp.

Wetland name	Wetland code	Date sampled	WQI score	Wetland quality category	Type of gear used	
					FN	EF
Boom Camp	BC	08/10/2004	0.895	Good	X	X
Charles Inlet	CI	07/07/2004	1.185	Very good	X	X
Cow Island	CO	08/14/2005	1.317	Very good		X
Dead Horse Bay	DH	08/09/2005	1.360	Very good	X	X
Dogfish Bay	DF	08/13/2005	1.363	Very good	X	X
Garden Channel*	GC	06/23/2004	1.618	Very good	X	X
Gooseneck Bay	GN	06/22/2004	1.463	Very good		X
Green Island North	GI-N04	06/02/2004	1.380	Very good	X	X
Green Island North	GI-N05	06/09/2005	1.380	Very good	X	X
Green Island South	GI-S	06/09/2005	1.380	Very good	X	X
Hay Bay 1	HB1	07/05/2005	1.454	Very good	X	X
Hog Bay	HG	06/07/2004	0.717	Good	X	X
Iroquois Bay	IQ	08/05/2004	1.841	Very good	X	X
Isle of Pine	IP	07/08/2004	1.849	Very good		X
Jumbo Bay*	JB-04	08/08/2004	1.845	Very good	X	X
Jumbo Bay*	JB-05	08/13/2005	1.774	Very good	X	X
Kirk Creek	KC	08/07/2004	1.227	Very good		X
Longuissa Bay	LG	06/28/2004	2.232	Excellent	X	X
Matchedash Bay	MB	07/27/2004	-0.167	Moderately degraded	X	X
Moose Bay*	ME	06/14/2004	1.847	Very good	X	X
Moreau Bay	MO	06/16/2004	1.168	Very good	X	X
Musky Bay	MS	06/03/2004	1.229	Very good		X
Naiscoot North 1	NN1	07/06/2004	1.200	Very good		X
Naiscoot North 2	NN2	07/06/2004	1.368	Very good		X
Naiscoot South	NS	07/07/2004	1.355	Very good		X
North Bay	NB	06/15/2005	0.432	Good	X	X
Oak Bay	OB	06/08/2004	1.124	Very good	X	X
Ojibway*	OJ	06/16/2005	1.796	Very good	X	X
Port Rawson East	PWE	06/23/2004	1.657	Very good		X
Port Rawson West	PWW	06/23/2004	1.785	Very good		X
Quarry Island	QI	06/01/2004	1.109	Very good		X
Ragged Bight	RG	07/05/2005	2.038	Excellent		X
Robert's Bay*	RB	06/01/2004	1.443	Very good	X	X
Sturgeon Bay South	SG	07/26/2004	0.636	Good	X	X
Treasure Bay	TB	06/13/2005	1.782	Very good	X	X
Treasure Bay Inner	TBI	06/15/2005	---	---		X
Vincent's Bunk	VB	08/06/2004	1.270	Very good	X	X
Wardrope Island*	WI-04	08/04/2004	1.824	Very good	X	X
Wardrope Island*	WI-05	08/09/2005	1.748	Very good	X	X
Woods Bay	WO	06/15/2004	1.451	Very good		X

Table 1-2. Summary statistics for wetlands sampled with electrofishing boat in this study. Asterisks indicate that the sites are dominated by *Scirpus* sp.

Wetland name	Total length of transect	Number of transects	Total shock seconds	Fish species richness	Total catch	Total Biomass
Boom Camp	521	3	940	7	193	1598
Charles Inlet	621	4	1145	11	37	3753
Cow Island	1,398	4	2065	17	261	4853
Dead Horse Bay	886	3	1250	14	236	15521
Dogfish Bay	706	3	1445	15	235	5280
Garden Channel*	940	3	991	14	68	164
Gooseneck Bay	---	3	930	6	62	325
Green Island North	---	3	1194	12	62	7964
Green Island North	1,145	4	1514	15	99	8150
Green Island South	1,109	3	1543	15	127	7136
Hay Bay 1	317	1	500	13	65	581
Hog Bay	---	3	887	10	228	13214
Iroquois Bay	853	4	1200	14	130	1845
Isle of Pine	---	3	922	11	157	890
Jumbo Bay*	1,119	3	1000	7	101	4067
Jumbo Bay*	913	4	1960	11	23	2310
Kirk Creek	---	4	1804	19	136	14745
Longuissa Bay	588	3	925	11	116	6847
Matchedash Bay	715	4	989	14	155	5904

Moose Bay*	---	4	1102	9	91	3813
Moreau Bay	686	3	908	13	124	6266
Musky Bay	---	3	868	14	95	20646
Naiscoot North 1	---	3	909	12	184	399
Naiscoot North 2	---	3	902	16	219	3856
Naiscoot South	---	4	1056	14	54	4376
North Bay	961	3	1500	14	175	7119
Oak Bay	---	3	1008	14	69	11696
Ojibway*	469	2	704	5	54	2219
Port Rawson East	---	3	810	10	251	3023
Port Rawson West	---	3	860	8	46	4237
Quarry Island	---	3	900	8	73	3030
Ragged Bight	119	1	135	4	7	22
Robert's Bay*	940	2	488	8	42	1577
Sturgeon Bay South	902	3	1006	11	185	1844
Treasure Bay	1,397	4	2040	16	140	20304
Treasure Bay Inner	999	3	1560	15	330	5943
Vincent's Bunk	744	3	1000	11	101	914
Wardrope Island*	642	3	1000	4	18	23
Wardrope Island*	699	2	1000	4	22	5032
Woods Bay	---	3	1001	9	141	164

Table 1-3. Summary statistics of fish collected by fyke nets (FN) and by electrofishing boat (EB). Numbers in brackets are the SE.

	Survey Method		
Parameter	**Both methods**	**FN only**	**EB only**
# fish caught	10398	7478	2920
% all fish caught	---	71.9	28.1
Biomass of fish (kg)	204,198.59	133,818	65,844.66
% all fish biomass	---	65.5	32.2
Number of species recovered	45	42	40
% total species recovered	---	93.3	88.9
Number of functional taxa recovered	48	46	43
% total functional taxa recovered	---	95.8	89.6
Mean fish length (cm)	92.6	91.9	93.9
	(± 1.12)	(± 1.43)	(± 1.79)
Mean fish weight (g)	67.0	69.6	62.2
	(± 4.68)	(± 6.25)	(± 6.68)
Mean species richness per wetland	17.0	12.9	11.2
	(± 0.74)	(± 0.68)	(± 0.72)
Mean number of functional taxa per wetland	18.4	13.7	12.2
	(± 0.79)	(± 0.65)	(± 0.84)
Mean number fish per wetland	399.9	287.6	112.3
	(± 35.45)	(± 36.19)	(± 12.92)

Table 1-4. Comparison of total number of species caught by fyke nets (FN) versus electrofishing boat (EB).

			Number of Specimens			Number of Wetlands		
Family	**Common Name**	**Scientific name**	**Both**	**FN**	**EB**	**Both**	**FN**	**EB**
Benthivore								
Castostomidae	White Sucker	*Catostomus commersonii*	22	3	19	10	3	9
Castostomidae	Shorthead Redhorse	*Moxostoma macrolepidotum*	28	27	1	2	2	1
Cottidae	Mottled Sculpin	*Cottus bairdii*	2	2	0	1	1	0
Cyprinidae	Common Carp	*Cyprinus carpio*	6	4	2	4	2	2
Cyprinidae	Bluntnose Minnow	*Pimephales notatus*	696	501	195	23	22	22
Gobiidae	Round Goby	*Neogobius melanostomus*	3	2	1	2	2	1
Ictaluridae	Brown Bullhead	*Ameiurus nebulosus*	248	164	84	23	22	16
Ictaluridae	Bullhead (juvenile)	*Icalurus sp.*	2	1	1	2	1	1
Ictaluridae	Tadpole Madtom	*Noturus gyrinus*	28	28	0	7	7	0
Percidae	Johnny Darter	*Etheostoma nigr*	43	8	35	11	4	11
Umbridae	Central Mudminnow	*Umbra limi*	10	1	9	2	1	1
Carnivore								
Atherinidae	Brook Silverside	*Labidesthes sicculus*	52	10	42	5	7	10
Centrarchidae	Rockbass	*Ambloplites rupestris*	330	317	13	25	24	10
Centrarchidae	Pumpkinseed	*Lepomis gibbosus*	1779	1279	500	26	25	20
Centrarchidae	Bluegill	*Lepomis macrochirus*	333	274	59	4	4	4
Centrarchidae	Longear Sunfish	*Lepomis megalotis*	74	55	19	7	7	3
Centrarchidae	Sunfish	*Lepomis sp.*	242	192	50	17	17	8
Centrarchidae	Smallmouth Bass (0-20mm)	*Micropterus dolomieu*	19	15	4	6	5	3
Centrarchidae	Largemouth Bass (30-70mm)	*Micropterus salmoides*	267	153	114	14	13	12
Centrarchidae	Black Crappie (0-160mm)	*Pomoxis nigromaculatus*	69	52	17	9	9	4
Cyprinidae	Spotfin Shiner	*Cyprinella spiloptera*	7	4	3	3	2	1
Cyprinidae	Blackchin Shiner	*Notropis heterodon*	235	129	106	22	14	13
Cyprinidae	Blacknose Shiner	*Notropis heterolepis*	249	209	40	10	11	6
Fundulida	Banded Killifish	*Fundulus diaphanus*	59	31	28	12	8	6
Gasterosteidae	Brook Stickleback	*Culaea inconstans*	1	1	0	1	1	0

Gasterosteidae	Ninespine Stickleback	*Pungitius pungitius*	1	1	0	1	1	0
Moronidae	White Perch	*Morone americana*	8	1	7	1	1	1
Moronidae	White Bass	*Morone chrysops*	1	0	1	1	0	1
Percidae	Iowa Darter	*Etheostoma exile*	36	3	33	13	2	12
Percidae	Yellow Perch (1-150mm)	*Perca flavescen*	763	277	486	26	23	23
Percidae	Logperch	*Percina caprodes*	26	6	20	8	4	5
Omnivore								
Cyprinidae	Common Shiner	*Luxilus cornutus*	129	45	84	10	5	9
Cyprinidae	Golden Shiner	*Notemigonus crysoleucas*	123	66	57	14	10	8
Cyprinidae	Spottail Shiner	*Notropis hudsonius*	99	22	77	17	8	22
Cyprinidae	Shiner (juvenile)	Notropis sp.	7	0	7	2	0	2
Cyprinidae	Sand Shiner	*Notropis stramineus*	42	9	33	4	2	2
Cyprinidae	Mimic Shiner	*Notropis volucellus*	212	161	51	15	12	8
Cyprinidae	Northern Redbelly Dace	*Phoxinus eos*	8	6	2	3	3	1
Piscivore								
Amiidae	Bowfin	*Amia calva*	76	59	17	18	18	8
Centrarchidae	Smallmouth Bass (20+mm/Adults)	*Micropterus dolomieu*	17	8	9	8	6	6
Centrarchidae	Largemouth Bass (Adult)	*Micropterus salmoides*	71	27	44	16	11	12
Centrarchidae	Black Crappie (+160mm)	*Pomoxis nigromaculatus*	7	5	2	4	4	1
Esocidae	Northern Pike (Adult)	*Esox lucius*	26	16	10	11	8	4
Lepisosteidae	Longnose Gar	*Lepisosteus osseus*	37	23	14	19	10	10
Percidae	Yellow Perch (+150mm)	*Perca flavescens*	54	23	31	19	11	15
Plankivore								
Clupeidae	Alewife	*Alosa pseudoharengus*	2	1	1	2	1	1
Cyprinidae	Emerald Shiner	*Notropis atherinoides*	20	15	5	4	3	3
Esocidae	Muskellunge	*Esox masquinongy*	2	2	0	2	1	1

Table 1-5. Comparison of number of functional taxa captured with fyke net (FN) versus electrofishing boat (EB). Asterisks indicate *Scirpus*-dominated sites. Numbers in bracket are abundances.

	Number of functional taxa captured					
Wetland	**Total**	**FN& EB**	**FN**	**EB**	**Only FN**	**Only EB**
Longuissa Bay	**16**	**7**	**12** (324)	**10** (113)	**5**	**4**
Moose Bay*	**15**	**6**	**12** (204)	**9** (86)	**6**	**3**
Jumbo Bay-04*	**17**	**2**	**12** (297)	**7** (101)	**10**	**5**
Iroquois Island	**15**	**8**	**9** (339)	**14** (130)	**1**	**6**
Wardrope Island-04*	**13**	**2**	**12** (147)	**4** (36)	**10**	**1**
Ojibway Bay*	**11**	**4**	**10** (546)	**5** (54)	**6**	**1**
Treasure Bay	**19**	**12**	**15** (235)	**16** (140)	**3**	**4**
Jumbo Bay-05*	**16**	**7**	**13** (203)	**11** (23)	**5**	**4**
Wardrope Island-05*	**10**	**4**	**10** (488)	**4** (22)	**6**	**0**
Garden Channel*	**18**	**12**	**16** (854)	**13** (66)	**4**	**2**
Hay Bay	**18**	**9**	**14** (202)	**13** (65)	**5**	**4**
Robert's Bay*	**16**	**7**	**15** (166)	**8** (42)	**8**	**1**
Green Island-04	**18**	**8**	**14** (211)	**12** (62)	**6**	**3**
Green Island-05	**24**	**11**	**20** (286)	**15** (127)	**9**	**4**
Green Island South	**24**	**11**	**19** (208)	**15** (127)	**7**	**4**
Dogfish Bay	**17**	**6**	**8** (166)	**15** (235)	**2**	**8**
Dead Horse Bay	**15**	**7**	**8** (158)	**14** (236)	**1**	**7**
Vincent's Bunk	**13**	**6**	**8** (431)	**10** (101)	**2**	**5**
Charles Inlet	**18**	**9**	**16** (251)	**11** (36)	**7**	**2**
Moreau Bay	**23**	**7**	**17** (716)	**13** (124)	**10**	**6**
Oak Bay	**23**	**5**	**15**	**14**	**9**	**9**

			(56)	(69)		
Boom Camp Bay	**15**	**4**	**12** (233)	**7** (193)	**8**	**3**
Hog Bay	**12**	**6**	**14** (165)	**13** (228)	**1**	**5**
Sturgeon Bay	**19**	**6**	**14** (297)	**11** (185)	**8**	**5**
North Bay	**17**	**10**	**13** (160)	**14** (175)	**4**	**3**
Matchedash Bay	**22**	**11**	**17** (135)	**14** (155)	**6**	**5**

Table 1-6. Comparison of the mean Wetland Fish Index scores calculated with presence-absence (WFI-PA) and abundance (WFI-AB) data collected with fyke net (FN) and electrofishing boat (EB). Numbers in bracket are SE. There were no significant differences between means for the two methods when all data were compared (n=26; P>0.20; paired t-test) but there were significant differences when data for Scirpus-dominated sites were compared (n=8; P<0.05; Wilcoxon-signed rank test).

Survey Method	WFI PA	WFI AB
All 26 site-years		
FN	3.76 (0.041)	3.70 (0.061)
EB	3.70 (0.059)	3.65 (0.059)
6 Scirpus sites (8 site-years)		
FN	3.91 (0.033)	3.96 (0.066)
EB	3.72 (0.159)	3.82 (0.119)

Table 1-7. Comparison of summary statistics for fish collected in fyke nets that had been set in two different orientations to the shoreline. Numbers in brackets are SE.

Parameter	Parallel	Perpendicular
Number of fish caught	993	602
% all fish caught for both orientations	62.3	37.7
Biomass of fish (kg)	31,726	26,598
% all fish biomass for both orientations	54.4	45.6
Number of species recovered	28	26
% total species recovered from both orientations	87.5	81.25
Number of functional taxa recovered	32	28
% total functional taxa recovered from both orientations	88.9	77.8
Mean fish length (cm)	90.1	96.5
	(± 2.94)	(±3.81)
Mean fish weight (g)	49.8	53.7
	(± 8.92)	(± 11.20)
Mean species richness per wetland	19.7	16.0
	(± 0.33)	(± 1.73)
Mean number of functional taxa per wetland	21.0	17.3
	(± 1.00)	(± 1.20)
Mean number fish per wetland	212.3	165.0
	(± 29.90)	(± 22.72)

Table 1-8. Comparison of the number of species collected with parallel versus perpendicular orientation in three wetlands. P-values correspond to Wilcoxon signed rank test used to compare differences between means for the two orientations.

Site	**Parameter**	**Orientation**		**P-value**
		Parallel	**Perpendicular**	
Green Island South	Number of species	19	13	--
Green Island North		20	19	--
Long Point		20	16	--
	Mean	***19.67***	***16.0***	***0.13***
Green Island South	WFI-PA scores	3.87	3.57	--
Green Island North		3.70	3.95	--
Long Point		3.74	3.71	--
	Mean	***3.74***	***3.77***	***0.88***
Green Island South	WFI-PA scores	3.79	3.68	--
Green Island North		3.67	3.73	--
Long Point		3.71	3.71	--
	Mean	***3.71***	***3.72***	***0.77***

Table 1-9. Comparison of summary statistics for fish collected with electrofishing boat during point- and transect-sampling. Numbers in brackets are the SE. There were no significant differences between mean lengths and weights (t-test; P=0.18 and 0.55, respectively). Approximately the same size of wetland area was covered with both sampling protocols.

	Survey Method	
Parameter	**Point Sample**	**Transect**
Total shock seconds delivered	200	1543
Number of fish caught	16	126
Biomass of fish (kg)	1488	7127
Number of species recovered	6	14
Number of functional taxa recovered	6	16
Mean fish length (cm)	77.9	105.1
	(± 17.38)	(± 6.50)
Mean fish weight (g)	93.0	57.0
	(± 47.70)	(± 20.59)

Table 1-10. Comparison of total number of individuals per species recovered by each survey method.

	Survey Method	
Taxa	**Point Sample**	**Transect**
Carnivore		
Blackchin shiner	-	6
Brook silverside	-	3
Iowa darter	-	3
Largemouth bass	-	3
Logperch	1	4
Pumpkinseed	2	33
Rockbass	-	2
Spotfin shiner	2	-
Yellow perch	-	39
Piscivore		
Bowfin	-	2
Largemouth bass	1	2
Yellow perch	-	2
Benthivore		
Bluntnose minnow	6	17
Brown bullhead	-	3
Johnny darter	4	6
Round goby	-	1
Omnivore		
Golden shiner	-	1

Table 1-11. Comparison of species caught, and associated Wetland Fish Index scores determined from presence-absence (WFI-PA) and abundance data (WFI-AB) for the two electrofishing protocols. All fishing was conducted in Green Island South in June 2005. Numbers in bracket are SE. Asterisks indicate that means are statistically significant between protocols (Wilcoxon signed rank test, n=3, P<0.05).

Site	Point sampling	Transect sampling
Total shock seconds	200	1500
Number of species	6	14
WFI-PA score*	2.96 (0.04)	3.58 (0.06)
WFI-AB score*	3.23 (0.13)	3.66 (0.08)

Table 1-12. Comparison of species caught, and associated Wetland Fish Index scores determined from presence-absence (WFI-PA) and abundance data (WFI-AB) for an increased effort in electrofishing transects. All fishing was conducted in Green Island North in June 2005.

Shock Seconds	WFI PA	WFI AB	No. of Taxa
1000	3.79	3.80	11
1500	3.71	3.79	15
2100	3.69	3.73	19
3600	3.68	3.73	20
4600	3.71	3.79	21

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Figure 1-9. Proportion of total taxa as a function of WQI score for a) FN and b) EB data. All wetlands in this study are open squares, and those for Lakes Erie and Ontario (taken from Chow-Fraser et al. 2006) are closed squares.

Figure 1-10. Proportion of total taxa as a function of WQI score for EB (closed) and FN (open) data plotted for all Lake Huron/Georgian Bay sites. Neither of the two regression lines have slopes that are significantly different from zero (P>0.05).

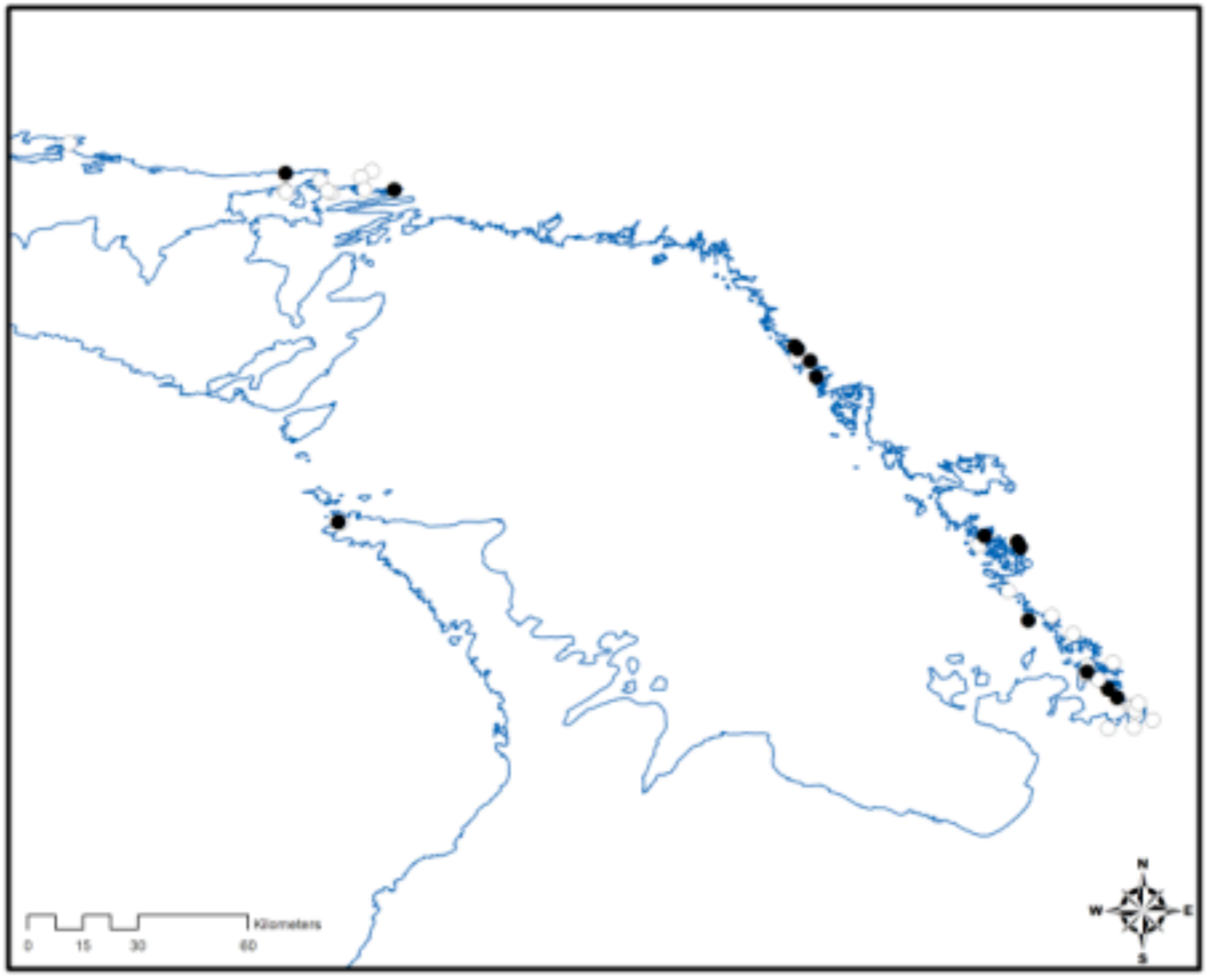


Figure 1-1. Map of wetland locations for comparing FN and EB data (n=26) and comparing the effect of plant densities on electrofishing effort (n=40).

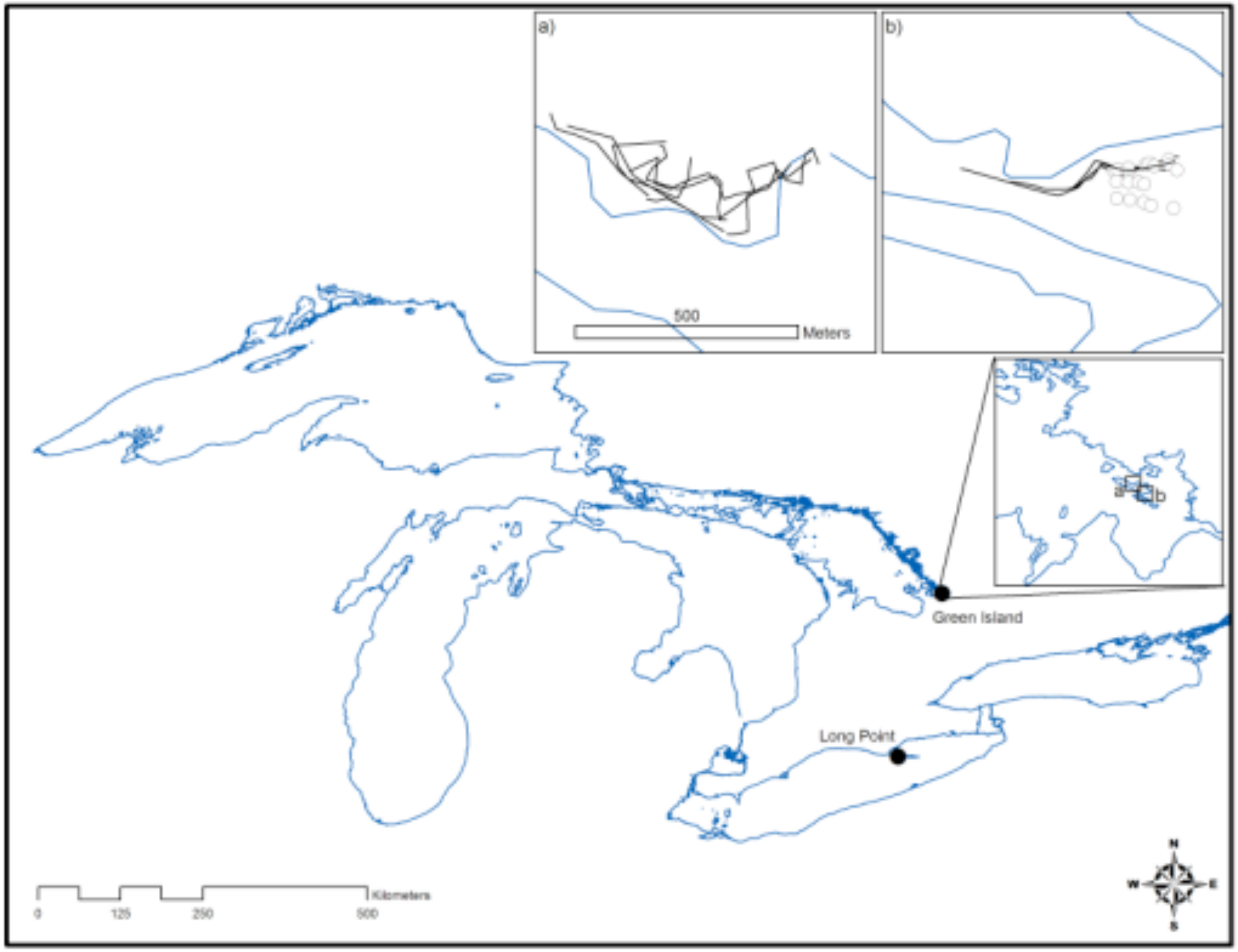
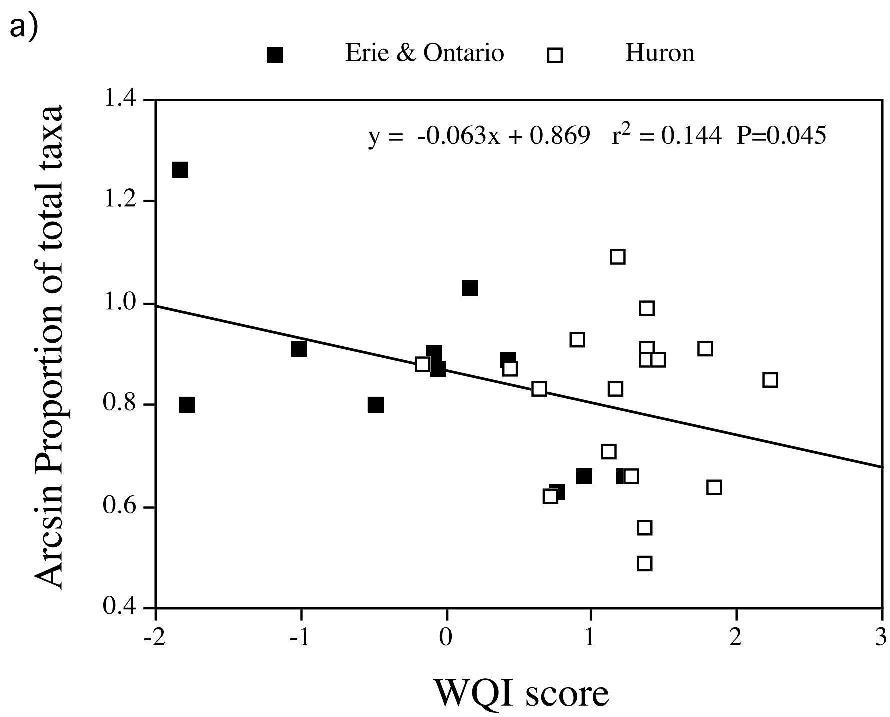


Figure 1-2. Map of wetland locations in the fyke net comparison, a) Georgian Bay Green Island, b) Lake Erie, Long Point Provincial Park.



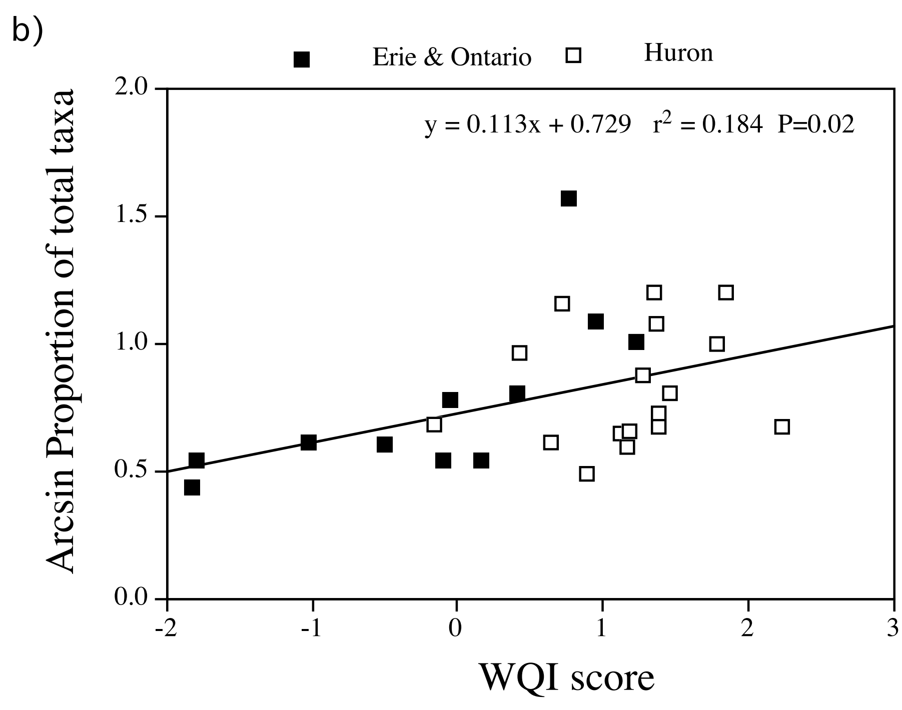


Figure 1-9. Proportion of total taxa as a function of WQI score for a) FN and b) EB data. All wetlands in this study are open squares, and those for Lakes Erie and Ontario (taken from Chow-Fraser *et al.* 2006) are closed squares.

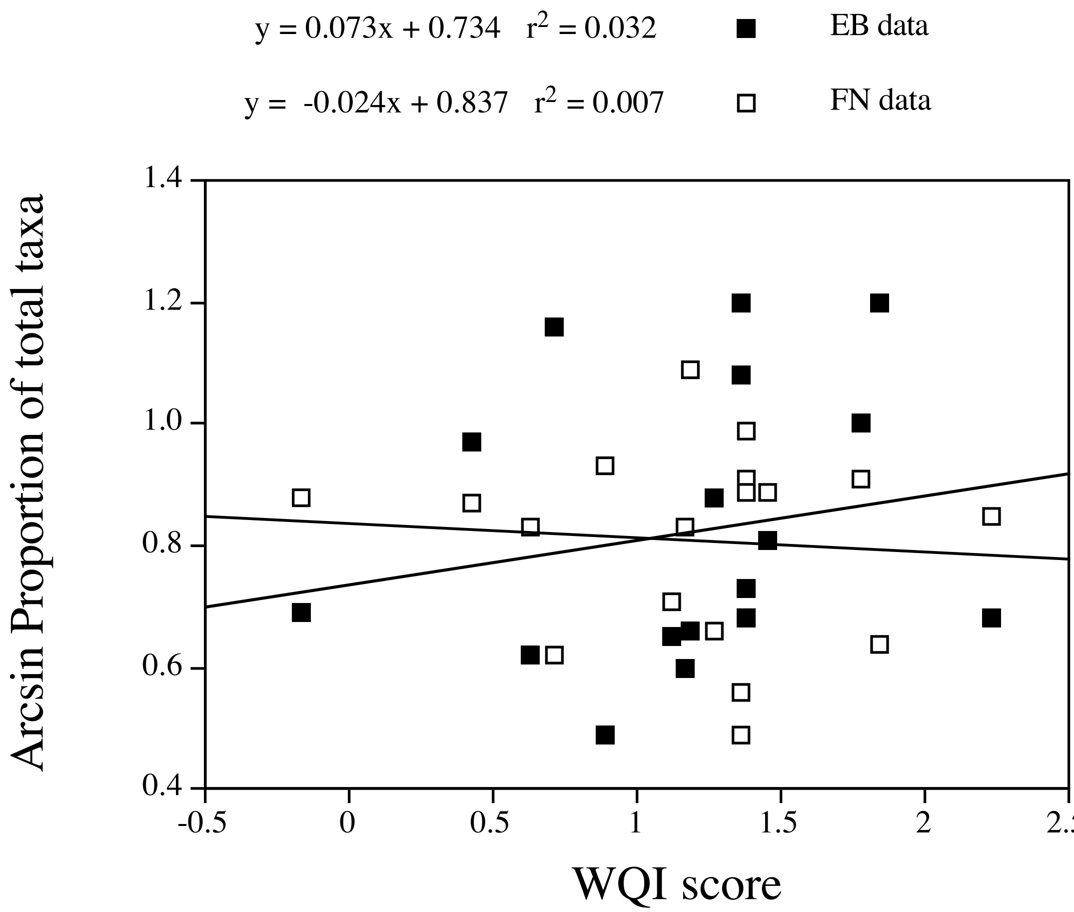


Figure 1-10. Proportion of total taxa as a function of WQI score for EB (closed) and FN (open) data plotted for all Lake Huron/Georgian Bay sites. Neither of the two regression lines have slopes that are significantly different from zero.

Chapter 2: Use of zoobenthos to indicate human-induced disturbance and degree of exposure in Great Lakes Coastal Wetlands

Kristina Kostuk

and

Patricia Chow-Fraser

significantly related to degree of exposure and/or human-induced disturbance.

INTRODUCTION

Coastal wetlands of the Laurentian Great Lakes have been greatly affected by human activities, particularly those in the settled regions of Lakes Erie, Ontario and Michigan over the past 200 years. As urban sprawl increases over the next decade in response to rising human populations, there will be further stress on these coastal ecosystems. Already two-thirds of these wetlands that were present prior to European settlement, have been lost due to dredging, dyking or infilling (Mitsch and Bouchard 1998), and this loss is tragic because coastal wetlands provide exceptional value for both human and non-human communities (Maynard and Wilcox 1997). There is also strong evidence that conversion of forested to agricultural and urban land in wetland watersheds have caused remainder of the coastal marshes to become degraded, further compromising their ecological functions (Crosbie and Chow-Fraser 1999; Chow-Fraser 2006).

When coastal marshes become enriched and degraded by urban and agricultural runoff, the ecosystem components undergo highly visible, predictable changes (Chow-Fraser 1998; Lougheed *et al.* 2001; McNair 2006; Chow-Fraser 2006). When wetlands are undisturbed by human activities, the water is oligotrophic and clear, with a very diverse mix of submersed aquatic vegetation (SAV), interspersed with assemblages of floating plants. As wetlands become mesotrophic, the richness of SAV may be maintained, but certain taxa that thrive in nutrient-rich waters tend to dominate, and this leads to marshes with very dense macrophytic communities, consisting of heavy growth of SAV and floating plants. With increased nutrient loading, however, the macrophytes become shaded out by planktonic algae, and this leads to eutrophic, open-water areas

ABSTRACT

Benthic invertebrates were collected from 55 coastal wetlands of Lakes Superior, Huron, Erie and Ontario from 2001 to 2005, inclusive. Based on the Water Quality Index (WQI; Chow-Fraser 2006), which indicates the degree of human-induced disturbance in wetlands, these sites range in quality from highly degraded to virtually undisturbed. We followed the approach that had been used successfully by others to develop an index of wetland quality, by using Canonical Correspondence Analysis (CCA) to ordinate the distribution of zoobenthos species along a water-quality gradient (i.e. nutrient and turbidity gradient associated with altered land uses in watersheds). Unlike past studies, however, the zoobenthos data were not unimodally distributed and did not respond strongly to an underlying water-quality gradient. Because of the demonstrated importance of plant zones in structuring benthic invertebrates in other coastal marshes in Lakes Michigan and Huron, we decided to use Cluster Analysis to first group wetlands according to both plant and water-quality information, and then use zoobenthos abundance data in a Discriminant Function Analysis (DFA) to determine the discriminatory power of the zoobenthos data. The Cluster Analysis identified five groups that were separated on the basis of water-quality degradation, and degree of exposure disturbance associated with Great Lake. Zoobenthos data were able to discriminate among the 5 groups with 80% of the sites correctly matched (Wilks' Lambda P=0.048, Hotelling-Lawley P=0.035; Roy's Max Root P0.001), while the first three canonical axes explained 93% of the total variation. We used results of the DFA to develop potential metrics of wetland quality by identifying zoobenthos taxa that were

with little or no submergent vegetation, with only a fringe of emergent vegetation near the shore, and floating plants in more protected areas.

This transition from a clear-water, macrophyte-dominated, species-rich SAV system to a turbid, phytoplankton-dominated, species-poor SAV system is known to be accompanied by predictable changes in the zooplankton and fish community (Chow-Fraser *et al.* 1998; Chow-Fraser 1998). The associated change in zooplankton and fish assemblages with deterioration in water quality is the basis for the Wetland Zooplankton Index (WZI; Lougheed and Chow-Fraser 2002) and the Wetland Fish Index (WFI; Seilheimer and Chow-Fraser 2006), respectively. The WZI has been used successfully to track improvements in habitat quality as wetlands become restored, while the WFI has been used to differentiate among habitat quality of known human-induced disturbance. These, along with a family of other biotic indicators (e.g. Index of Biotic Integrity, Minns *et al.* 1994; Burton *et al.* 1999; Kashian and Burton 2000) are being actively developed to track the health of coastal wetlands throughout the Laurentian Great Lake.

Use of biotic indicators to assess ecosystem health has been commonplace for other aquatic ecosystems for many years, and for streams, use of macroinvertebrates dates back three decades (Hillsenoff 1977, 1987, 1988, Karr 1991, Lenat 1993). Invertebrates are good indicators of both short-term and long-term environmental conditions because they are relatively immobile and exhibit a range of sensitivity and tolerance to pollutants. In the Great Lakes, invertebrates have been studied primarily in open and near-shore areas, and despite its importance in the aquatic food web, there has been limited research on this group in coastal wetlands (Krieger 1992). Kashian and

Burton (2000) were the first to develop an Index of Ecological Integrity for coastal wetlands based on benthic macroinvertebrates, but the geographic extent was limited to sites in northern Lake Huron. Even though Uzarski *et al.* (2004) expanded the geographic coverage to include wetlands of northern Lake Michigan, there is still a need to develop an index for benthic invertebrates that can be applied widely throughout the Great Lakes, in a manner similar to that of the WZI and the WFI.

The primary goal of this study is to apply the multivariate statistical approach used in development of the WFI and WZI to build an ecological index with benthic invertebrate data, that could be applied widely throughout the Great Lakes shoreline to allow for large-scale comparisons at regular intervals for the State of the Lake Ecosystem Conferences. We wanted the index to be developed from data collected by standardized protocols, that include a level of taxonomic resolution matching the abilities of most technicians in environmental agencies charged with the task of monitoring wetlands.

MATERIALS AND METHODS

Materials

Definition of "Zoobenthos"

The word "zoobenthos" used in this study refers to the invertebrate primary and secondary consumers caught in funnel traps (see Methods). The community of animals includes some of the *zoo*plankton (copepods, cladocerans), which are found floating in the water column and many of the *benthic* invertebrates that reside on top of the sediment or that emerge from the sediment during the 24-h incubation period. This method does not sample any of the macroinvertebrates that live in emergent or vegetation or that glide on the surface tension of the water.

Why use Zoobenthos?

Benthic invertebrates have been useful in biomonitoring as there are both intolerant organisms, and those that are tolerant of poor water quality and can live and even thrive in degraded environments. Knowledge of these taxonomic groups as indicators of pollution has been successfully applied in streams and rivers (Hillsenoff 1977, 1987, 1988, Lenat 1993). In the Great Lakes, invertebrates have been studied primarily in open and near-shore areas. Despite the important role invertebrates play in the food web, there has been limited research on this group in coastal wetlands (Krieger1992). In 2002, Lougheed and Chow-Fraser demonstrated that zooplankton could be used as environmental indicators of wetland quality in 60+ coastal wetlands, while Kashian and Burton (2000) developed an Index of Ecological Integrity for coastal wetlands in northern Lake Huron based solely on benthic macroinvertebrates. Recently, Uzarski *et al.* (2004) validated an invertebrate index of biotic integrity for Lakes Huron and Michigan fringing wetlands in low lake level years.

Study Sites

We assumed that the documented gradient in water-quality degradation evident in previous studies (e.g. Lougheed and Chow-Fraser 2002; McNair 2006; Seilheimer and Chow-Fraser 2006) would have a major influence on the distribution of zoobenthos.

Therefore, we chose wetlands based on their geographic location as well as degree of water-quality impairment, as indicated by their Water Quality Index (WQI) score (Chow-Fraser 2006). In total, 55 coastal marshes from Lakes Erie, Ontario, Huron (Georgian Bay/North Channel), and Superior were included (Fig. 2-1; Table 2-1).

Eight wetlands were located on the shoreline of Lake Erie, and included some of the most agriculturally disturbed areas of the Great Lakes basin (Chow-Fraser and Albert 1999). Except for the two marshes located on the sandspit at Long Point (Provincial Park) and Turkey Point, these were a mixture of primarily protected estuarine, lacustrine and riverine systems. Of the eighteen Lake Ontario wetlands, only a few were located on the U.S. shoreline. Sites located on the western portion of the lake were very degraded due primarily to runoff from large urban centers (Greater Hamilton and Greater Toronto). By comparison, wetlands located in eastern Lake Ontario had minimal impact from urbanization, and only a few sites had moderate impact from low-intensity agricultural activities (i.e. grazing).

Wetlands included equal numbers of exposed (shoreline exposed to wind and wave action) and protected sites, but were largely estuarine, with a few lacustrine and riverine. All six wetlands from Lake Huron are highly exposed lacustrine sites, with the five Canadian sites located on the tip of the Bruce Peninsula in Fathom Five National Marine Park. Several of these are located on unpopulated islands and are therefore subject to minimal human disturbance, unlike the one U.S. site, which is moderately degraded by human activities. There were twenty sites from eastern and northern Georgian Bay. Except for those in the southeastern Georgian Bay (Severn Sound and

Honey Harbour area), most of the wetlands are in very good condition, and show very little evidence of human impact. They are predominantly lacustrine, and include some that are protected and some that are highly exposed. Since much of the Canadian portion of Lake Superior is exposed and windswept, there are only a handful of coastal wetlands (Chow-Fraser and Albert 1999). We included two exposed lacustrine and one protected estuarine systems.

Methods

All sites in this study were visited during the summer (June to August) between 2001 and 2005. Wetlands were sampled for water quality and physical conditions within a day of the zoobenthos and plants being surveyed.

Water Quality Sampling and WQI score

All water samples were collected with a 1-L van Dorn bottle at mid-depth in open water away from submersed or floating vegetation. Samples collected were subsequently used for determination of nutrients, chlorophyll and suspended solids as indicated in Chow-Fraser (2006). Physical parameters (temperature, pH, specific conductivity, dissolved oxygen and turbidity were measured with a Hydrolab Minisonde multiparameter probe during 2000 to 2001 and then with an YSI 6600 multiparameter probe from 2002-2005). Parallel trials were carried out in 2001 to ensure comparability of data between instruments. All Water Quality Index (WQI) scores were determined with the 12-parameter equation presented in Chow-Fraser (2006). Chow-Fraser interpreted the WQI scores as follows:

WQI Score	Wetland Quality
+2 to +3	Excellent
+1 to +2	Very good
0 to +1	Good
-1 to 0	Moderately degraded
-2 to -1	Very degraded
-3 to -2	Highly degraded

Zoobenthos Sampling

Invertebrate communities were sampled with funnel traps, which consisted of three inverted plastic funnels (19 cm, covering a surface of 0.028m^2), each of which was attached via a short fitted plastic tubing to a 620-mL Nalgene bottle. The three funnels were oriented in a triangle and held in place with a sheet of Plexiglas. The Plexiglas had three holes through which each of the tubing protruded, and which were then attached to the Nalgene bottle. Two funnel traps (n=6 bottles) were deployed in vegetated areas whenever possible, and if there is no submersed vegetation, they were placed on sediment surface close to emergent stands or near the roots of floating plants. Funnel traps were left for up to 24 h, after which the bottles were unscrewed from the funnel-Plexiglas. Within 2-3 hours of collection, contents of the bottles were filtered through 63-µm Nitex screen. All of the filtrate were backwashed into storage bottles, and preserved in 4% formalin until they were processed in the laboratory.

Vegetation Sampling

We sampled the plant community by identifying all plant taxa surveyed in arbitrarily placed quadrats (approximately 20 quadrats/wetland). Since the primary focus was submergent and floating communities, we only performed a survey of the emergent community located near the funnel traps, rather than performing a representative survey of the entire emergent community. Hence, plants typically classified as a "wet meadow" species were excluded from this survey, as this was not deemed to be a component of the aquatic habitat. All taxa were identified at least to genus, and in most cases, to species (when flowering parts were present) with the help of Newmaster et al. (1997) and Chadde (2002).

Zoobenthos Processing

In the laboratory, zoobenthos were initially transferred from formalin into 70% ethanol before further processing with the aid of a dissecting microscope (up to 40X magnification). Specimens from four to six bottles were sorted and then identified, depending on the size (and therefore distribution) of the organism in question. For example, smaller organisms such as zooplankton were identified and enumerated in four bottles, whereas large insects that were found in good-quality sites were counted from all six. Thorp and Covich (1991), Merritt and Cummins (1996), and Pennak (1989) were used to identify the zoobenthos to the lowest operational taxonomic unit (usually genus or species, but some groups only to family or order (i.e. Oligochaeta)). Chironomidae larvae were identified to sub-family or tribe. Because the number of bottles processed varied from site to site, we had to first standardize the data by calculating mean number per bottle (funnel trap) before entering them into the multivariate analyses.

Statistical Analyses

The Cluster Analysis (CLA), Discriminant Function Analysis (DFA) and the logistic regression analyses were performed with SAS JMP IN 5.1 software (Cary, N.C). We also used CANOCO 4.5 to run the canonical correspondence analysis and PC ORD 4 to perform the Nonmetric Multidimensional Scaling. CLA was used to classify groups of objects judged to be similar based on multiple variables (James and McCulloch 1990) and maximizing within-group similarity (McGarigal *et al.* 2000). We used this analysis to first group wetlands by habitat characteristics such as plant community composition and water quality. According to West (1986), when groups are defined by such ecological factors, it may be difficult to discriminate among sites because of overlapping distributions of habitat characteristics. "Based on the variables used to describe the original groups, discriminant analysis creates new distributions (hybrid distributions between the defined groups." (Webster and Burrough 1974, referenced in West 1986). Therefore, by applying Discriminant Function Analysis (DFA) to the site variables, we can determine the predicted group membership and the probability that the sites are found in the assigned group based on chance alone. Hence, DFA was used to determine the probability that the zoobenthos data could be used to discriminate among clusters identified in the CLA.

RESULTS

Initial Ordination Results

The canonical correspondence analysis (CCA) included a variety of environmental parameters: total ammonia nitrogen, total nitrate nitrogen, total and soluble reactive phosphorus, ambient temperature, specific conductivity, planktonic chlorophyll, dissolved oxygen, and total suspended solids, as well as taxa richness of submergents, floating, and emergent plants. Although we found that the first two axes were significant (p=0.05, p=0.002), the associated eignevalue was only 0.05, and according to ter Braak and Verdonschot (1995), only eigenvalues >0.30 indicate a strong underlying gradient. Furthermore, results of the Detrended Correspondence Analysis indicated that the data were not strongly unimodal, and it was therefore inappropriate for us to continue with a CCA. Subjecting the zoobenthos data to Nonmetric Multidimensional Scaling (NMS) yielded similarly disappointing results that indicated absence of a single strong underlying gradient.

We concluded that a different approach from that used by Lougheed and Chow-Fraser (2002) and Seilheimer and Chow-Fraser (2006) had to be used to examine the environmental factors that structured the zoobenthos data. We decided to take a two-stage approach, in which habitat characteristics (water quality and plant community information) are first entered into a Cluster Analysis (CLA) to classify wetlands into groups, and then a Discriminant Function Analysis is performed to determine if zoobenthos data can be used to discriminate among the groups.

Cluster Analysis

In the previous multivariate analyses (i.e. CCA, NMS), we included most of the water-chemistry and physical variables associated with water quality of wetlands, and only a few variables relating to the plant communities. This over-emphasis on water-quality variables produced inconclusive results, and the reason for this may be related to the greater influence that plant communities may have on the distribution of macroinvertebrates (Krieger 1992, Burton *et al.* 1999). Therefore, we decided to use only one water-quality indicator, the Water Quality Index (WQI) score, which integrates information from 12 water-quality variables (Chow-Fraser 2006). To better represent the plant community, we included the number of *Typha* and *Scirpus* species, all species of floating vegetation, as well as the number of submersed aquatic vegetation (SAV) within five functional groups, sorted according to their tolerance of water-quality impairment (Croft and Chow-Fraser, unpublished data).

The CLA classified the 55 wetlands into five groups (Fig. 2-2). **Group A** consisted of 12 sites (Fig. 2-3; Table 2-2) that were all found in the lower lakes (Lakes Erie and Ontario), except for one Lake Superior wetland, Chippewa Marsh, which is located in the City of Thunder Bay. Most of the sites were protected estuarine systems, with only one lacustrine and one riverine. Most of the associated WQI scores were indicative of highly degraded to very degraded conditions; however, the WQI score for Chippewa Marsh indicated that it was in good condition, although it had an impaired submergent plant community. Typical emergent species included *Typha angustifolia* (Narrow-leaved cattail), *Polygonum amphibium* (Water smartweed), and the invasive exotic *Lythrum salicaria* (Purple loosestrife). The floating-leaved species include *Nymphaea odorata* (White water lily) and *Lemna minor* (Small duckweed) while the SAV species were limited to *Potamogeton sp* (Slender pondweed sp.) and *Stuckenia pectinata* (Sago pondweed) both of which are known to tolerate degraded conditions.

Group B consisted of 11 sites, of which 9 were found in the lower lakes and 2 in southern Georgian Bay (Fig. 2-3). The sites were a mix of primarily protected estuarine, lacustrine and riverine systems, while one of the sites was a sandspit. These sites had a range of WQI scores, indicating very degraded to very good conditions, although majority were in the moderately degraded category, and showed visible signs of degradation (Fig. 2-2; Table 2-2). Typical emergent taxa included *Typha angustifolia* (Narrow-leaved cattail), the exotic *Lythrum salicaria* (Purple loosestrife), and *Pontederia cordata* (Pickerelweed). The floating-leaved species included an exotic species *Hydrocharis morsus-ranae* (Frog bit), *Nuphar variegata* (Yellow pond lily), and *Nymphaea odorata* (White water lily). The SAV community included many species such as *Ceratophyllum demersum* (Coontail), *Chara sp*. (Muskgrass), *Elodea canadensis* (Canada waterweed), *Myriophyllum sibiricum* (Northern water milfoil), *Najas flexilis* (Slender naiad), *Potamogeton richardsonii* (Richardson's pondweed), several slender forms of *Potamogeton*, *P. zosteriformis* (Flat-stemmed pondweed), *Ranunculus longirostris* (Stiff water crowfoot), *Stuckenia pectinata* (Sago pondweed), *Utricularia vulgaris* (Common bladderwort), *Vallisneria americana* (Wild celery) as well as two exotic species, *Potamogeton crispus* (Curly pondweed) and *Myriophyllum spicatum* (Eurasian water milfoil).

Group C consisted of 10 sites, of which 7 were found in the lower lakes, and 3 in southern Georgian Bay (Fig. 2-3). The sites were primarily lacustrine with only one riverine system, and most of them were exposed. One of the sites was a sand spit. Associated WQI scores were mostsly indicative of good to very good conditions, with the exception of Wigwam Bay, which was classified as moderately degraded (Table 2-2). The emeregent taxa was represented by *Pontederia cordata* (Pickerelweed), while the floating-leaved species were represented by *Nuphar variegata* (Yellow pond lily), and *Nymphaea odorata* (White water lily). The SAV community was commonly made up of *Ceratophyllum demersum* (Coontail), *Chara sp*. (Muskgrass), *Elodea canadensis* (Canada waterweed), *Najas flexilis* (Slender naiad), *Potamogeton richardsonii* (Richardson's pondweed), several slender species of *Potamogeton*, *P. zosteriformis* (Flat-stemmed pondweed), and *Vallisneria americana* (Wild celery), as well as the exotic species, *Myriophyllum spicatum* (Eurasian water milfoil), *Potamogeton crispus* (Curly pondweed).

Group D consisted of 12 sites, with 11 located in the upper lakes and only 1 found in L. Ontario (Fig. 2-3). The sites were predominantly lacustrine, with only one estuarine. All except one was highly exposed to Georgian Bay. Associated WQI scores were indicative of good to excellent quality (Tables 2-1 and 2-2). The emergent zone was represented by *Scirpus acutus* (Hardstem bulrush) and *Eleocharis smallii* (Creeping spike-rush). The dominant SAV species were *Chara sp.* (Muskgrass), *Elodea canadensis* (Canada waterweed), *Najas flexilis* (Slender naiad), *Potamogeton gramineus* (Variable-leaved pondweed), *Potamogeton richardsonii* (Richardson's pondweed), slender species of *Potamogeton.* and *Vallisneria americana* (Wild celery). Floating-leaved species were rarely found at these sites, presumably because of the high degree of exposure to the wind and wave action of Georgian Bay (Table 2).

Group E consisted of 10 sites, which were all found in Georgian Bay/North Channel (Fig. 2-3), all lacustrine systems. Except for 2, these wetlands were very protected. Associated WQI scores indicated they were in very good to excellent condition (Table 2-1 & 2-2). The emergent community was represented by *Scirpus acutus* (Hardstem bulrush), *S. validus* (Softstem bulrush), *Sagittaria sp.* (Arrowhead), *Pontederia cordata* (Pickerelweed), *Eleocharis smallii* (Creeping spike-rush), and *Eriocaulon aquaticum* (Pipewort). Floating-leaved species included *Brasenia schreberi* (Watershield), *Nuphar variegata* (Yellow pond lily), and *Nymphaea odorata* (White water lily). The dominant SAV taxa were *Chara sp.* (Muskgrass), E*lodea canadensis* (Canada waterweed), *Bidens beckii* (Water marigold), *Myriophyllum sibiricum* (Northern water milfoil), *Najas flexilis* (Slender naiad), *Potamogeton amplifolius* (Large-leaved pondweed), *P. gramineus* (Variable-leaved pondweed), *P. natans* (Floating-leaved pondweed), *P. richardsonii* (Richardson's pondweed), *P. robbinsii* (Fern pondweed), slender species of *Potamogeton, P. zosteriformis* (Flat-stemmed pondweed), *Scirpus subterminalis* (Water bulrush), *Vallisneria americana* (Wild celery), and *Zizania sp.* (Wild rice).

Discriminant Function Analysis

Next, we performed a Discriminant Function Analysis (DFA) to see if zoobenthos community composition could be used to discriminate among the 5 groups (A to E) that were obtained based on the water quality and plant characteristics in the cluster analysis. We carried out some preliminary analyses to determine the appropriate level of taxonomic resolution to use, and found 30 taxa to yield meaningful results (see Table 2-3). Platyhelminthes did not need to be brought below phylum, whereas Chironomidae had to be identified to sub-family to discriminate among the groups. A significant discrimination of the 5 groups was achieved when these 30 taxa were used (Fig. 4;. Wilks' Lambda p=0.048, Pillai's Traxe p=0.077, Hotelling-Lawley p=0.035, Roy's Max Root p=0.001). The 1st and 2nd axes explained 78.8% of the variation, while the 3rd canonical axis explained an additional 14%. Of the 55 wetlands, only 11 were misclassified (20%) (Table 2-4). By superimposing the centroids of the 30 zoobenthos taxa onto the bi-plot of the 5 clusters, we were able to visually associate the benthic taxa to one of the 5 groups and have presented these in Table 3.

In general, the number of specimens in the very eutrophic wetlands of **Group A** was relatively high, with a mean of 3,266 (ranging from 66 to 26,780). In all, 16 taxa were identified, but only 4 of these were dominant (found in >50% of the sites sampled). Indicators of Group A included the microcrustaceans Cladocera, Copepoda and Ostracoda. Dipteran sub-families that were more exclusively associated with degraded wetlands in Group A included Chrionomini and Orthocladinae dipterans (Table 2-3).

By comparison, there were generally fewer specimens in the mesotrophic wetlands of **Group B**, with a mean of 1,049 (ranging from 158 to 3,156); however, the taxon-richness was high, with a total of 29 taxa represented, of which 15 were dominant. The dominant Insects included the family of dragonflies, Anisoptera, and the trichopteran, Leptoceridae, as well as the order of Collembola. Crustaceans included the amphipod, Gammaridae. There were three gastropod families, two Limnophila (Physidae and Planorbidae) and one Mesogastropoda (Hydrobiidae). Two taxa, Hirudinea and Collembola, were found in low numbers, but were generally confined to Groups B and C, and are therefore good indicators of these mesotrophic wetlands (Table 2-3).

The mean number of zoobenthos speciments in **Group C** was similar to that of **Group B**, 1,082 (ranging from 349 to 1,995), and were distributed among 29 taxa. These wetlands ranged in quality from mesotrophic to oligotrophic. Insects such as Caenidae, in the mayfly family, Zygoptera in the damselfly family, trichopterans other than Leptoceridae and Hydroptillidae, and families in the order of Lepidoptera were well represented in Group B. In addition, the Hydrachnida, the family of Gastropods Lymnaeidae, and oligochaete worms were also indicators of this group (Table 2-3).

There were comparatively few specimens in the exposed, oligotrophic sites of **Group D**, with a mean of only 289 (ranging from 19 to 703); nevertheless, a total of 25 taxa were represented, and 9 of these were common. In addition to many of those found in the other groups (inclulding Microcrustacea, Hydrachnida, Hyalellidae, Tanytarsini, and Oligochaeta), the dipteran,Tanypodinae and Ceratopogonidae were found almost exclusively at these sites (Table 2-3).

The protected oligotrophic wetlands in **Group E** had relatively few specimens per sample, with a mean of 513 (ranging from 123 to 1360); however, they belonged to 27 taxa, making these sites the most species rich. Seven taxa could be considered indicators, including insects of the order Diptera (Tanytarsini), Ephemeroptera (Baetidae), Trichoptera (Hydroptillidae), Hemiptera and Coleoptera. In addition, the amphipod, Hyalellidae, and the gastropod, Valvatidae, were also found associated with this group (Table 2-3). These benthic invertebrates were found exclusively in high-quality sites, and rarely appeared in degraded wetlands.

Benthic invertebrate associations with plant communities

In addition to the multivariate analyses, we also conducted logistic regression analyses to determine significant relationships between the presence/absence of zoobenthos and characteristics of the aquatic plant communities for all wetlands (Table 2-5). There were many significant positive relationships between animal taxa and the richness of SAV in wetlands, including all 5 families of gastropods. There were also increased occurrence of dipterans, ephemeropterans and odonates, as well as Platyhelminthes and Hydrachnida as the community of submergent plants became more diverse. The likelihood of Hirudinea, Collembola and Anisoptera being present in wetlands increased as the number of floating species increased. With increase in the number of *Scirpus* species at a site, both Hyallelidae and Baetidae were more likely to be present. This contrasts the situation where Gammaridae were negatively associated with *Scirpus* diversity. Oligochaetes showed a negative relationship with the number of

Typha species found in wetlands, whereas Hirudinea demonstrated a positive relationship.

DISCUSSION

Coastal wetlands are an important part of the Laurentian Great Lakes ecosystem. They provide a diversity of habitats for a variety of wildlife and plant communities; they also act as buffers from chemical and other inputs from surrounding watersheds (Mitsch 1992). As primary consumers and detritivores, zoobenthos are some of the most abundant animals in these wetlands. The ability of various taxa to colonize and thrive in a wetland is dependent on the quality of its water and sediment (Chow-Fraser *et al.* 1998), and as such, this makes them a potentially good indicator of wetland quality, and by implication, the richness of the submergent plant community (Chow-Fraser *et al.* 1998; Lougheed *et al.* 2001; McNair and Chow-Fraser 2003). However, it has also been demonstrated by Burton *et al.* (1999) and Kashian and Burton (2000) that the distribution of macroinvertebrates also depend on the type of emergent plant communities present in a wetland. For example, these authors found it more useful to ordinate their macroinvertebrate communities according to the local presence of *Typha*, *Scirpus*, and lilies rather than water-quality characteristics. Therefore, unlike the ecological indices developed for zooplankton (Wetland Zooplankton Index; Lougheed and Chow-Fraser 2002) and fish (Wetland Fish Index; Seilheimer and Chow-Fraser 2006), which are based on the association between animals and water-quality characteristics of wetlands alone, an index involving zoobenthos may need to consider more than one underlying disturbance gradient.

Results of the CCA were initially disappointing because we had expected to use the same basic approach employed by Lougheed and Chow-Fraser (2002) and Seilheimer and Chow-Fraser (2006), but they are consistent with what would be expected if there is more than one strong underlying factor governing the distribution of zoobenthos. However, when we used a combination of water-quality information together with plant-community characteristics in the cluster analysis, we were able to produce five meaningful clusters (Fig. 2-2), that separated the highly degraded wetlands of the lower lakes (Group A) from most of the others (Fig. 2-3). Consistent with the Intermediate Disturbance Hypothesis (Connell 1978), sites with mesotrophic conditions, Groups B and C, which have either good or moderately degraded conditions according to the WQI, were associated with the highest species richness and diversity. The composition of zoobenthos in these groups clearly differentiated them from Groups A (eutrophic), and from D and E (both oligotrophic) (Fig. 2-4).

The close alignment of Groups A and D in Fig. 2-4, which are the eutrophic and exposed oligotrophic sites, respectively, is largely because both have very sparse representation of submergent taxa, but for different reasons. Members of Group A are the highly degraded sites with turbid water, and high algal biomass that tend to out compete SAV for light in the water column. By comparison, Group D wetlands are oligotrophic, but have few submersed species because they are highly exposed to wind and wave action (Table 2-2). These results show that exposure is an important variable that governs the distribution of zoobenthos.

The close alignment of Groups D and E in Fig. 4 likely reflect the greater dominance of *Scirpus* sp. in these sites primarily located in Georgian Bay and the North Channel of Lake Huron (Fig. 2-3). It is also noteworthy that only a few exotic species were present in the plant communities of these two groups, whereas many more were found in Groups B and C. Of the five, Group A had the lowest SAV richness, and this may be linked to the lower oxygen levels experienced in degraded sites in Peshtigo River near Green Bay, Lake Michigan that was reported by MacKenzie *et al.* (2004).

Development of potential metrics to indicate ecological conditions

Over the past decade, several investigators have proposed metrics for an Index of Biotic Integrity for coastal Great Lakes marshes (Burton *et al.* 1999; Wilcox *et al.* 2002), which are intended to detect impairment from multiple stressors (Plafkin *et al.* 1989). One of the more common metrics used is the **total abundance of organisms** in a sample, which has been found to vary directly with the trophic status of wetlands. Our results are consistent with this general trend, and therefore we propose this as a potential metric (Table 2-6). In our case, however, the largest proportion of organisms enumerated were microcrustaceans (Cladocera, Copepoda, and Ostracoda), which were present in high numbers in all 5 groups (Figure 2-6a) and therefore we propose that another useful potential metric is **total abundance excluding microcrustaceans** (Table 2-6), especially since an index based on cladoceran species already exist (Wetland Zooplankton Index;

Lougheed and Chow-Fraser 2002). We propose that high values of this metric should indicate mesotrophic conditions.

Proportion of amphipods in samples has been used as a metric of ecosystem health in both lotic and wetland systems. We found that the **% Amphipoda** generally peaked in mesotrophic sites (intermediate disturbance); however, when we brought the identification to the family level, we could distinguish mesotrophic (higher **% Gammaridae**) from oligotrophic sites (higher **% Hyalellidae)** (Table 2-6). This proposition is consistent with literature on streams (Lenat 1993; Hilsenhoff 1998) and coastal wetlands of Lakes Huron and Michigan (Kashian and Burton 2000; Uzarski *et al.* 2004). In general, Gammaridae were not found in any of the exposed oligotrophic systems (Group D), and may be used as an indicator of exposure.

Ephemeropterans were the dominant insect group in majority of the wetland sites, which included Baetidae and Caenidae. Kashian and Burton (2000) found that the number of ephemeropterans were significantly higher in the reference wetland than in the impacted wetland. In this study, however, we found it useful to identify these to family because **% Baetidae** increased as sites became more oligotrophic, while **% Caenidae** peaked in mesotrophic sites (Table 2-6).

Although dipterans were ubiquitous, and were found in all sites regardless of quality, **% dipterans** generally increased with eutrophic conditions. There were twice as many dipterans in Groups A and B than in Groups C to E. The Chironomidae family could be further identified to sub-families to provide greater resolving power among the 5 groups. **% Chironomini and Orthocladinae** tended to increase with eutrophic conditions, and this is consistent with Kashian and Burton (2000) who found highest abundances of Chironomini and Orthocladinae associated with impacted sites and were most dominant with plant-associated samples in two coastal wetlands of Lake Huron. Johnson *et al.* (1987) also found an increase in numbers of Chironomid taxa, as conditions became more eutrophic in some Georgian Bay wetlands. By contrast, **% Tanytarsini and Tanypondinae** tended to decrease with eutrophy, and increase with oligotrophy (highest in Groups D and E). Similar results were observed by Kashian (1998), where both Tanypondinae and Tanytarsini were more dominant in the reference wetlands. We also found that highly exposed sites had the highest proportion of Tanytarsini, and these may be good indicators of exposure (Table 2-6).

Another useful metric is **% Non-Dipteran Insects**, which tended to decrease as wetlands became eutrophic. Group A (eutrophic sites) had very few insects other than dipterans, Group B (mesotrophic sites) had more non-dipteran insects, but Groups D and E (exposed and non-exposed oligotrophic sites) had the highest proportion (Figure 2-6c).

Previous indicators have tended to exclude gastropods as a metric. Often, Mollusca has been used in conjunction with another taxon such as Crustacea (Burton *et al.* 1999; Uzarski *et al.* 2004). In this study, however, we found that **% Gastropods** indicated mesotrophic conditions, and when we brought this down to the level of families, **% Physidae** and **% Planorbidae** peaked in mesotrophic sites, and neither were found in eutrophic conditions. **% Hydrobiidae, % Lymnaeidae**, and **% Valvatidae** all increased with oligotrophic conditions, and in particular, the latter two families were found exclusively in high-quality sites. These are in agreement with stream indices, in which gastropods that have shells with an operculum and opening usually on the right (i.e. Lymnaeidae, Hydrobiidae, and Valvatidae) are assumed to be pollution sensitive, whereas those without an operculum with opening usually on the left (Physidae and Planorbidae) are assumed to be pollution tolerant (Citizen's Environment Watch Data Manual 2005).

According to the stream literature, annelid worms are known to be pollution tolerant. We found that Oligochaetes were distributed widely throughout the 55 wetlands regardless of quality. However, **% Oligochaetes** tended to increase in poor-quality conditions, the average abundances in Group A being 4 times higher than those in Group E (oligotrophic sites), while the **% Hirudnea** peaked in sites with mesotrophic conditions (Table 2-6).

% Odonata, which included both sub-orders Zygoptera (Damselfly) and Anisoptera (Dragonfly), were highest in mesotrophic sites. The latter was not present in any exposed sites, and therefore, its absence could be used as evidence of wind and wave action. Similar results have been observed in Lake Huron wetlands, where Odonata decreased as sites became more disturbed (Kashian 1998; Burton *et al.* 1999), although both are also known to be somewhat pollution tolerant in stream systems (Citizen's Environment Watch Data Manual 2005). Therefore, designation as an indicator of mesotrophic conditions is appropriate (Table 2-6).

Trichopterans were another dominant insect group in our samples. In two northern Lake Huron wetlands, Trichoptera were found to be significantly higher in the unimpacted sites (both plant and sediment communities) compared with the impacted wetland (Kashian and Burton 2000). Consistent with the literature, we found that Trichoptera was absent from all degraded sites. We found that identifying Trichoptera to the family level greatly improved its resolving power. **% Hydroptilidae** increased with oligotrophic conditions (twice as high in oligotrophic compared with mesotrophic sites) and were adapted to highly exposed sites, whereas **% Leptoceridae** was twice as high in mesotrophic compared with oligotrophic sites. All remaining trichopteran families (**% Trichoptera-others**) could be grouped and the proportion of these tended to be highest in mesotrophic conditions (Table 2-6).

Even though Hemiptera was not found as commonly as the other insect groups in our samples, **% Hemiptera** tended to increase in oligotrophic sites. Similarly, Platyhelminthes were also scarce, but despite this, **% Platyhelminthes** tended to be highest at mesotrophic conditions, and this may be useful as an indicator.

As general indicators of degree of exposure, we found that **% Ephemeroptera and Trichoptera** (as a metric) was twice as high in exposed, oligotrophic conditions compared with mesotrophic and protected, oligotrophic sites. Other indices, especially those used for streams, have also found these two orders (and occasionally Odonata) to be useful indicators of reference sites (Jones et al. 2005).

CONCLUSIONS

Our study suggests that zoobenthos can be used as indicators of quality of a wetland. Previous studies that employed benthic invertebrates as indicators of wetland condition have shown some success (e.g. Burton *et al.* 1999 and Uzarski *et al.* 2004), but have been limited by the size of their geographic coverage. We have used 55 study sites that occur throughout the Great Lakes basin, representing a variety of conditions with respect to water quality, plant community types, as well as degree of exposure. Our initial multivariate analysis was inconclusive because only zoobenthos species and water-quality parameters were involved, and we had more than water-quality impairment as a strong underlying gradient. However, the combined Cluster Analysis and Discriminate Function Analysis yielded very useful information to identify potential metrics of both water-quality degradation and exposure disturbance in the sense described by Wei and Chow-Fraser (in submission).

Use of biological organisms to characterize the quality of aquatic habitats has become a common tool in environmental assessment and management. Based on metrics in other bioassessment papers (Jones *et al.* 2005, Kearns and Karr 1994, Burton *et al.* 1999, etc) as well as results from our own analyses, we have proposed 26 metrics that could be applied to Great Lakes coastal wetlands (Table 2-6). These metrics should be able to indicate both the degree of human-induced disturbance and exposure disturbance due to wind and wave action. With further testing to confirm the value of these metrics, we aim to eventually develop a suitable index of ecological conditions for coastal wetlands, an index that could be added to the toolbox along with the Wetland Zooplankton Index (Lougheed and Chow-Fraser 2002), the Wetland Fish Index (Seilheimer and Chow-Fraser 2006) and the Wetland Macrophyte Index (Croft and Chow-Fraser, in submission).

ACKNOWLEDGEMENTS

This project was funded by research grants from The Great Lakes Fishery Commission (2000-2002) and the Canada-Ontario Agreement (2003-2005). Both field and lab assistance were provided by a number of undergraduate and graduate students, including: A. Aiken, M. Croft, M. Cvetkovic, T. Docherty, J. Labuda, K. de Mutsert, S. McNair, J. Midwood, L. O'Reilly, H. Pankhurst, D. Quinn, B. Radix, R. Razavi, B. Reich, C. Robinson, T. Seilheimer, M. Strack, A. Wei. We would also thank the Georgian Bay Association, especially Mary Muter for use of the Georgian Baykeeper from 2003-2005, and for GBA volunteers for providing accommodation and inspiration for the project.

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Table 2-1. Summary of location of wetlands, lake of origin, and the associated Water Quality Index Score (WQI; Chow-Fraser 2006) for study sites. Letters in bracket correspond to codes in Fig. 2.

Wetland name	Lake	WQI score	Wetland quality category
Batchwana (BW)	Superior	1.87	Very Good
Boat Passage (BG)	Huron	1.65	Very Good
Boom Camp (BC)	Georgian Bay/North Channel	0.90	Good
Bronte Creek (BR)	Ontario	-0.98	Moderately Degraded
Charles's Inlet (CI)	Georgian Bay/North Channel	1.18	Very Good
Chippewa Park (CW)	Superior	0.70	Good
Cloud Bay (CB)	Superior	2.13	Excellent
Cootes Paradise (CP)	Ontario	-1.56	Very Degraded
Cove Island North (CI)	Huron	2.36	Excellent
Credit River (CR)	Ontario	-1.48	Very Degraded
Darlington (DA)	Ontario	-1.02	Very Degraded
Echo Bay (EB)	Georgian Bay/North Channel	0.05	Good
Fifteen Mile Creek (FM)	Ontario	-1.99	Very Degraded
Frenchman's Bay (FB)	Ontario	-0.29	Moderately Degraded
Garden Channel (GC)	Georgian Bay/North Channel	1.62	Very Good
Goose Bay (GO)	Ontario	0.11	Good
Grand River (GR)	Erie	-1.88	Very Degraded
Grass Bay (GS)	Ontario	1.13	Very Good
Green Island (GI)	Georgian Bay/North Channel	1.38	Very Good
Hay Bay 1 (HB1)	Huron	1.45	Very Good
Hay Bay 2 (HB2)	Huron	0.86	Good
Hay Bay Marsh (HB)	Ontario	0.45	Good
Hog Bay (HG)	Georgian Bay/North Channel	0.72	Good
Humber River (HM)	Ontario	-1.42	Very Degraded
Iroquois Island (IQ)	Georgian Bay/North Channel	1.84	Very Good
Jordan Harbour (JH)	Ontario	-1.95	Very Degraded
Jumbo Bay (JB)	Georgian Bay/North Channel	1.84	Very Good
Lily Pond (LY)	Georgian Bay/North Channel	-0.46	Moderately Degraded
Little Cataraqui Creek (LQ)	Ontario	-1.28	Very Degraded
Little Sodus (LS)	Ontario	0.33	Good
Long Point Prov Park (LPK)	Erie	0.72	Good
Longuissa Bay (LG)	Georgian Bay/North Channel	2.23	Excellent
Matchedash Bay (MB)	Georgian Bay/North Channel	-0.17	Moderately Degraded
Moose Bay (ME)	Georgian Bay/North Channel	1.85	Very Good
Moreau Bay (MO)	Georgian Bay/North Channel	1.17	Very Good
Mud Bay (MD)	Ontario	-0.72	Moderately Degraded
Oak Bay (OB)	Georgian Bay/North Channel	1.12	Very Good
Old Woman Creek (OWC)	Erie	-2.42	Highly Degraded
Perch River (PF)	Ontario	0.13	Good

Presque Isle (PR)	Erie	0.01	Good
Presqu'ile Prov Pk (PI)	Ontario	0.47	Good
Robert's Bay (RB)	Georgian Bay/North Channel	1.44	Very Good
Rondeau (RN)	Erie	0.41	Good
Russell Island West (RUW)	Huron	2.32	Excellent
Salmon River (SA)	Ontario	1.28	Very Good
Sanctuary Pond (SN)	Erie	-2.20	Highly Degraded
Sandy Creek (SC)	Ontario	1.06	Very Good
Spicer Creek (SP)	Erie	1.01	Very Good
Sturgeon Bay South (SG)	Georgian Bay/North Channel	0.64	Good
Tadenac Bay (TD)	Georgian Bay/North Channel	1.79	Very Good
Treasure Bay (TB)	Georgian Bay/North Channel	1.78	Very Good
Turkey Point (TP)	Erie	0.64	Good
Vincent's Bunk (VB)	Georgian Bay/North Channel	1.27	Very Good
Wardrope Island (WI)	Georgian Bay/North Channel	1.82	Very Good
Wigwam Bay (WW)	Huron	-0.07	Moderately Degraded

Table 2-2. Summary of the type of water-quality conditions and composition of plant communities associated with the five groups identified in the cluster analysis. (see Fig. 2).

	Group				
Variable	**A**	**B**	**C**	**D**	**E**
Number of wetlands included	12	11	10	12	10
Geographic location	Protected estuarine primarily in lower lakes	Mixed site types, mostly from lower lakes	Primarily lacustrine found throughout	Predominantly lacustrine, exposed, with majority from upper lakes	All lacustrine protected, and located in upper lakes
WQI score	-1.37 (-2.42-0.70)	0.13 (-1.28-1.28)	0.68 (-0.07-1.45)	1.51 (0.05-2.36)	1.61 (1.12-2.23)
Mean number of aquatic plant taxa	8.17 (1–18)	24.73 (16-36)	14.30 (8-22)	12.92 (4-25)	25.10 (12-38)
Mean number of submergent taxa	2.83 (0-7)	14.82 (11-21)	10.20 (7-15)	8.50 (2-17)	15.60 (6-20)
Mean number of floating taxa	2.42 (1-5)	3.91 (1-8)	2.00 (1-4)	0.67 (0-2)	2.90 (1-6)
Mean number of *Scirpus* species	0.33 (0-1)	0.73 (0-3)	0.50 (0-2)	1.33 (0-3)	1.80 (0-4
Mean number of *Typha* species	1.00 (1-2)	1.36 (1-2)	0.10 (0-1)	0.42 (0-1)	0.60 (0-2)

Table 2-3. List of the taxa found in samples (excludes rare taxa) and their affiliation with 5 major groups identified in the cluster analysis (see Fig. 2). Bold indicates taxonomic level used for group comparisons and in the discriminant analysis.

Phylum	Class/ sub-class	Order	Family	Sub-family	Group Affiliation
Arthropoda	Insecta	Diptera	Chironomidae	**Chironomini**	A
Arthropoda	Insecta	Diptera	Chironomidae	**Orthocladinae**	A
Arthropoda	Insecta	Diptera	Chironomidae	**Tanypodinae**	D
Arthropoda	Insecta	Diptera	Chironomidae	**Tanytarsini**	E
Arthropoda	Insecta	Diptera	**Ceratopogonidae**		D
Arthropoda	Insecta	Ephemeroptera	**Baetidae**		E
Arthropoda	Insecta	Ephemeroptera	**Caenidae**		C
Arthropoda	Insecta	Odonata	**Zygoptera**		C
Arthropoda	Insecta	Odonata	**Anisoptera**		B
Arthropoda	Insecta	Trichoptera	**Hydroptillidae**		E
Arthropoda	Insecta	Trichoptera	**Leptoceridae**		B
Arthropoda	Insecta	Trichoptera	**Others**		C
Arthropoda	Insecta	**Hemiptera**			E
Arthropoda	Insecta	**Lepidoptera**			C
Arthropoda	Insecta	**Coleoptera**			E
Arthropoda	Insecta	**Collembola**			B
Arthropoda	Crustacea	Amphipoda	**Gammaridae**		B
Arthropoda	Crustacea	Amphipoda	**Hyalellidae**		E
Arthropoda	Crustacea	**Cladocera**			A
Arthropoda	Crustacea	**Copepoda**			A
Arthropoda	Arachnida	**Hydrachnida**			C
Arthropoda	Crustacea	**Ostracoda**			A
Mollusca	Gastropoda	Limnophila	**Physidae**		B
Mollusca	Gastropoda	Limnophila	**Planorbidae**		B
Mollusca	Gastropoda	Limnophila	**Lymnaeidae**		C
Mollusca	Gastropoda	Mesogastropoda	**Hydrobiidae**		B
Mollusca	Gastropoda	Mesogastropoda	**Valvatidae**		E
Annelida	**Hirudinea**				B
Annelida	**Oligochaeta**				C
Platyhelminthes					B

Table 2-4. Correlation Matrix results using discriminant analysis. Bold numbers indicated the correctly classified sites.

Group		**A**	**B**	**C**	**D**	**E**	**Total**
Wetland Condition		Eutrophic	Mesotrophic	Mesotrophic	Exposed (Oligotrophic)	Oligotrophic	
	A	**9**	0	0	1	2	12
	B	0	**10**	0	1	0	11
Actual rows by predicted columns	C	0	0	**9**	1	0	10
	D	1	0	1	**7**	3	12
	E	0	0	0	1	**9**	10

Table 2-5. Summary of relationship between zoobenthos taxa and plants in this study. "+" means there is a significant positive relationship; "-" means there is a significant negative relationship as indicated by a logistic regression analysis (P<0.05).

Taxa		Relationship with plants			
		Submergents	Floating	*Scirpus*	*Typha*
Crustacea	Gammaridae			-	
	Hyalellidae	+		+	
Gastropoda	Physidae	+			
	Planorbidae	+			
	Lymnaeidae	+			
	Hydrobiidae	+			
	Valvatidae	+			
Annelida	Hirudinea		+		+
	Oligochaeta				-
Insecta	Collembola		+		
	Diptera:				
	Tanypondinae	+			
	Tanytarsini	+			
	Ceratopogonidae	+			
	Ephemeroptera:				
	Baetidae	+		+	
	Odonata:				
	Anisoptera	+	+		
Platyhelminthes		+			
Hydrachnida		+			

Table 2-6. Potential metrics for use in Great Lakes Coastal wetlands. Composition excludes micro-crustaceans. (Note: The percents use Total abundance excluding microcrustaceans).

Potential Metric	Indication
Total abundance	Increases with eutrophic conditions
Total abundance excluding microcrustaceans	Highest at sites with mesotrophic conditions
% Amphipoda	Highest at sites with mesotrophic conditions
% Gammaridae	Increases with eutrophic conditions, and extremely intolerant of exposure.
% Hyalellidae	Increases with oligotrophic conditions
% Chironomidae and Orthocladinae	Increases with eutrophic conditions
% Tanytarsini and Tanypondinae	Increases with oligotrophic conditions and sites that are highly exposed
% Baetidae	Increases with oligotrophic conditions
% Caenidae	Highest at sites with mesotrophic conditions
% Gastropods	Highest at sites with mesotrophic conditions
% Physidae	Highest at sites with mesotrophic conditions
% Planorbidae	Highest at sites with mesotrophic conditions
% Lymnaeidae	Increases with oligotrophic conditions
% Hydrobiidae	Increases with oligotrophic conditions
% Valvatidae	Increases with oligotrophic conditions but no in exposed sites
% Hirudinea	Highest at sites with mesotrophic conditions
% Oligochaeta	Increases with eutrophic conditions
% Odonata	Highest at sites with mesotrophic conditions—Anisoptera not present in exposed sites
% Dipterans	Increases with eutrophic conditions
% Non-Dipteran Insects	Increases with oligotrophic conditions
% Hydroptilidae	Increases with oligotrophic conditions; adapted to highly exposed sites
% Leptoceridae	Highest at sites with mesotrophic conditions
% Trichoptera - others	Highest at sites with mesotrophic conditions
% Hemiptera	Increases with oligotrophic conditions
% Platyhelminthes	Highest at sites with mesotrophic conditions
% Ephemeroptera and Trichoptera	Increases with sites that are highly exposed

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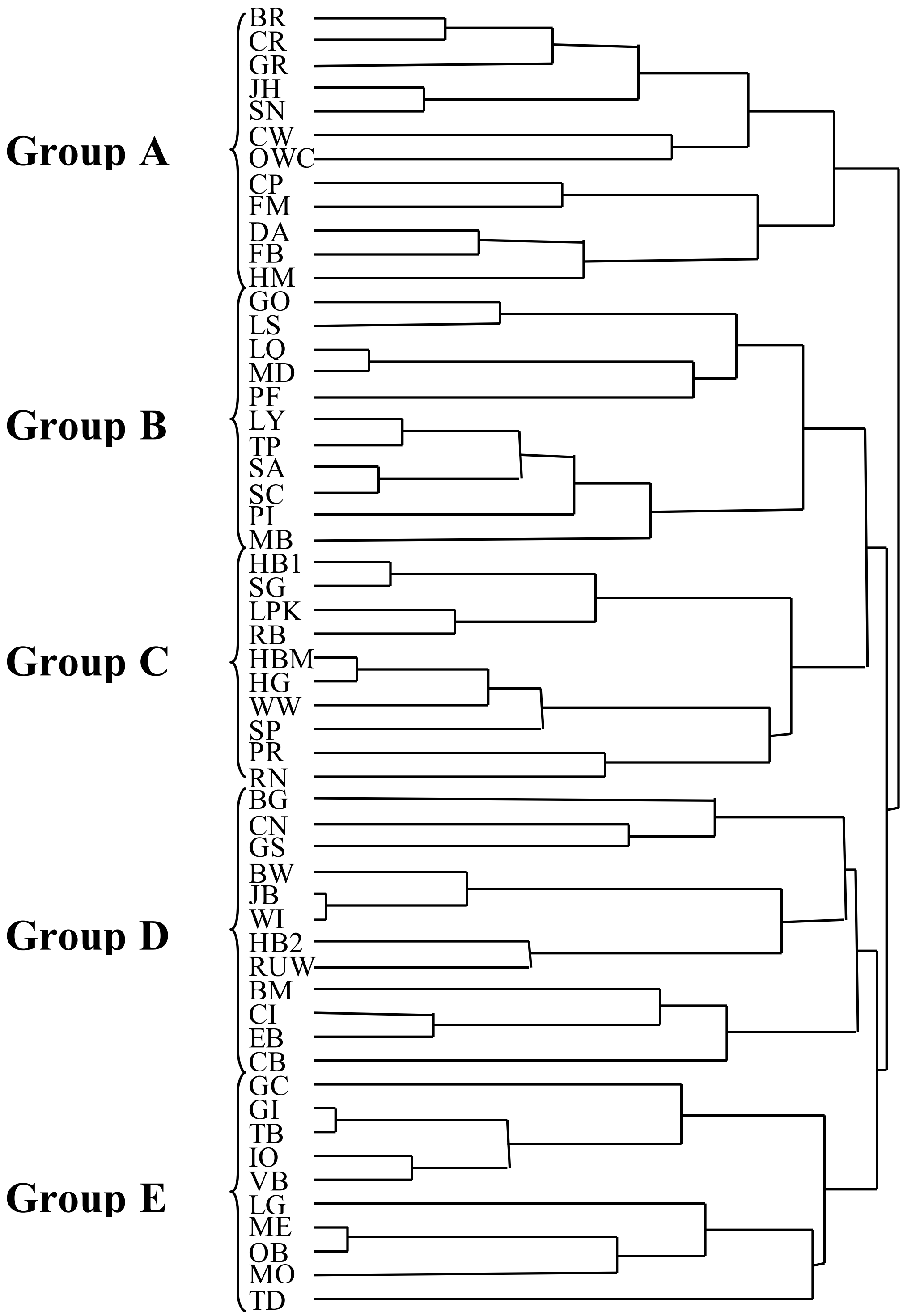

Figure 2-2. Results of a Ward's Cluster Analysis

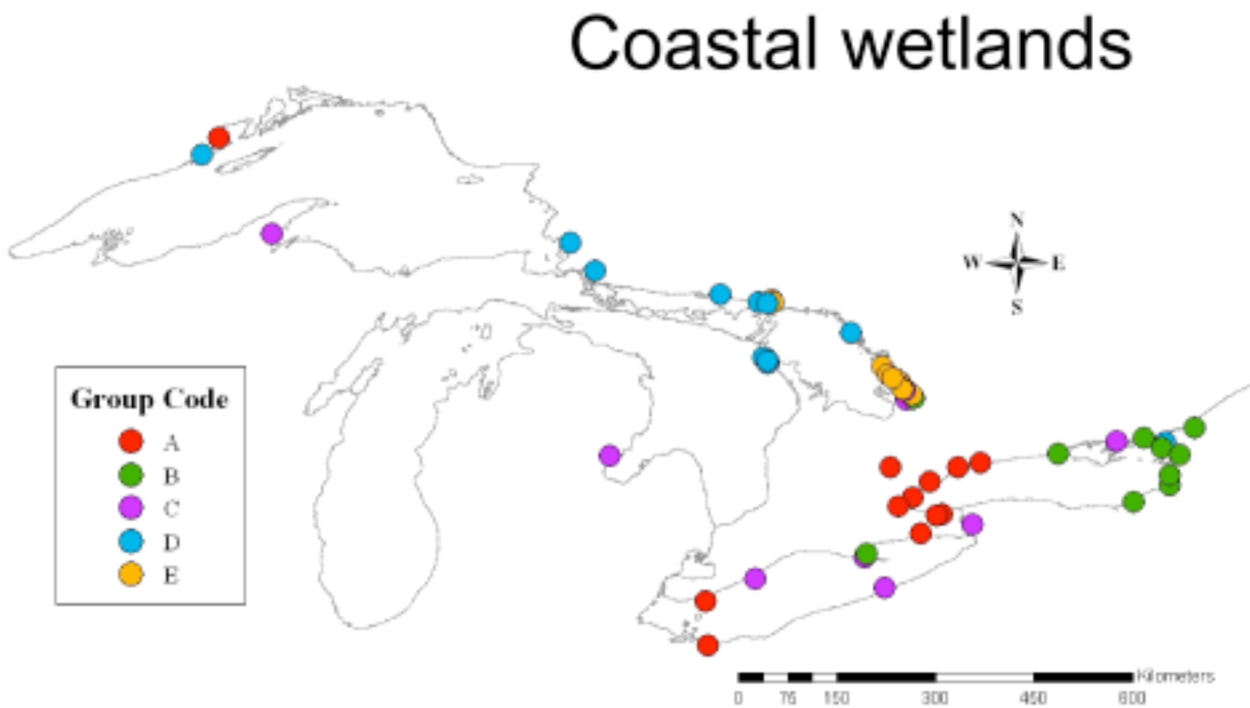

Figure 2-3. Location of study sites based on cluster analyses groupings.

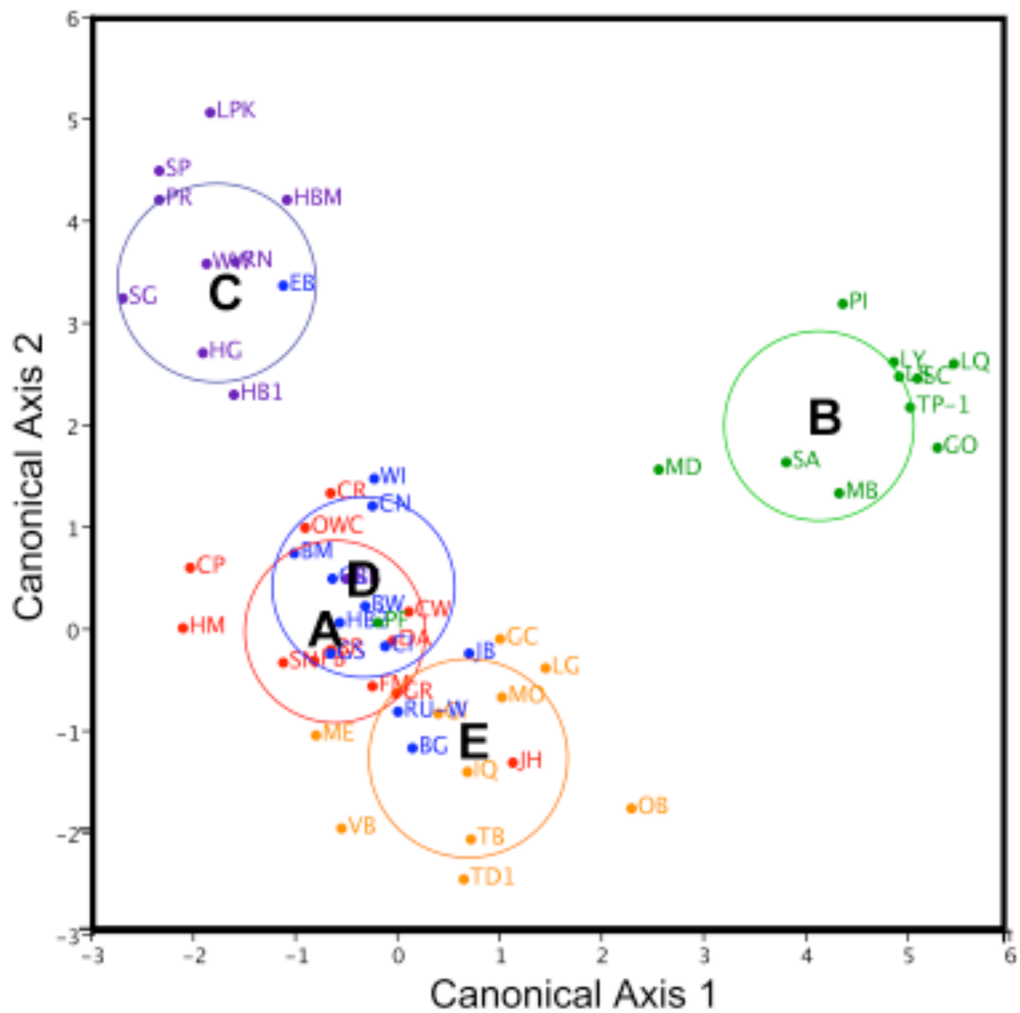

Figure 2-4. Discriminant analysis results of the 55 wetland sites, based on the 5 cluster groups.

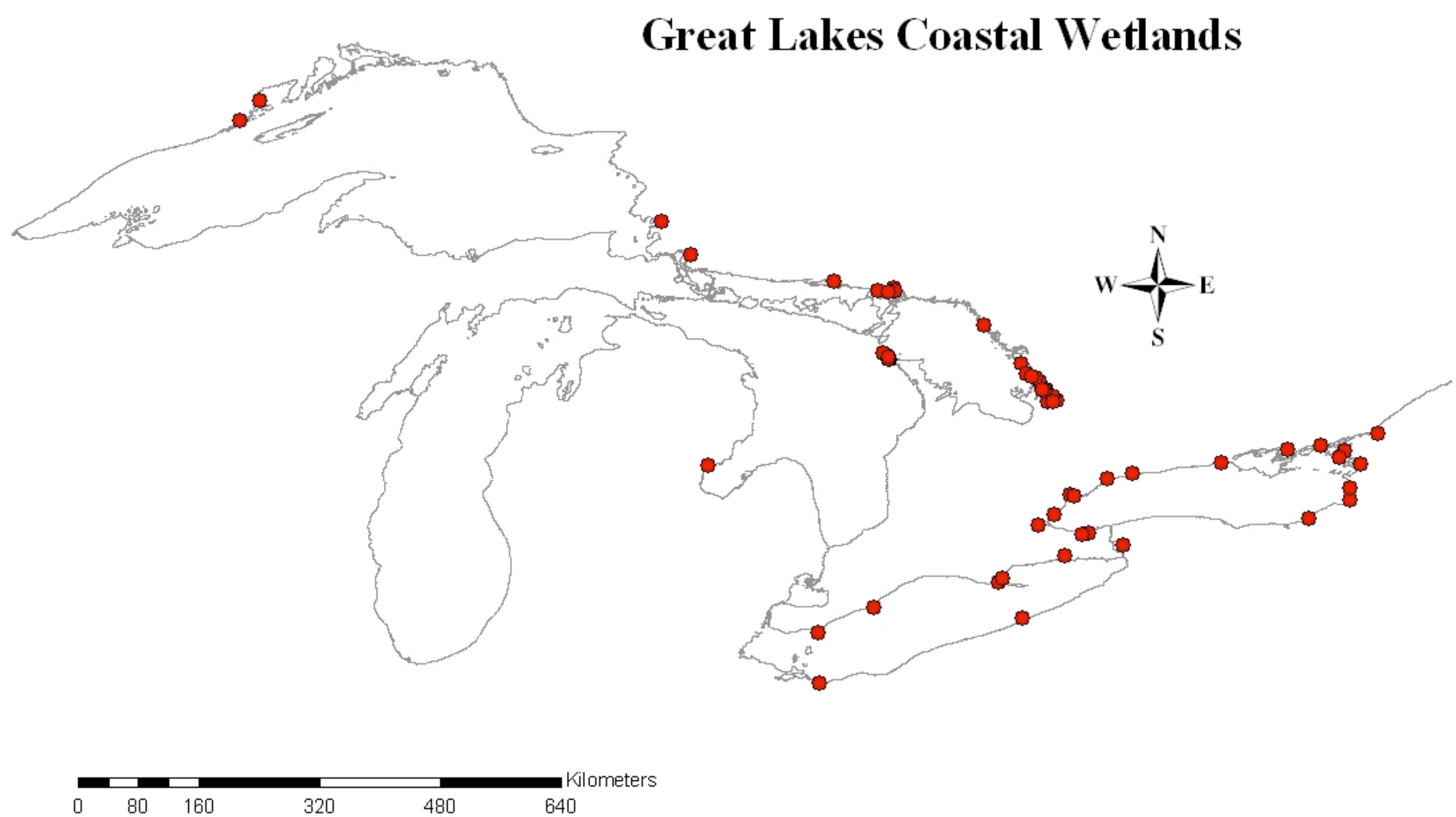

Figure 2-1. Location of study sites around the Great Lakes shoreline.

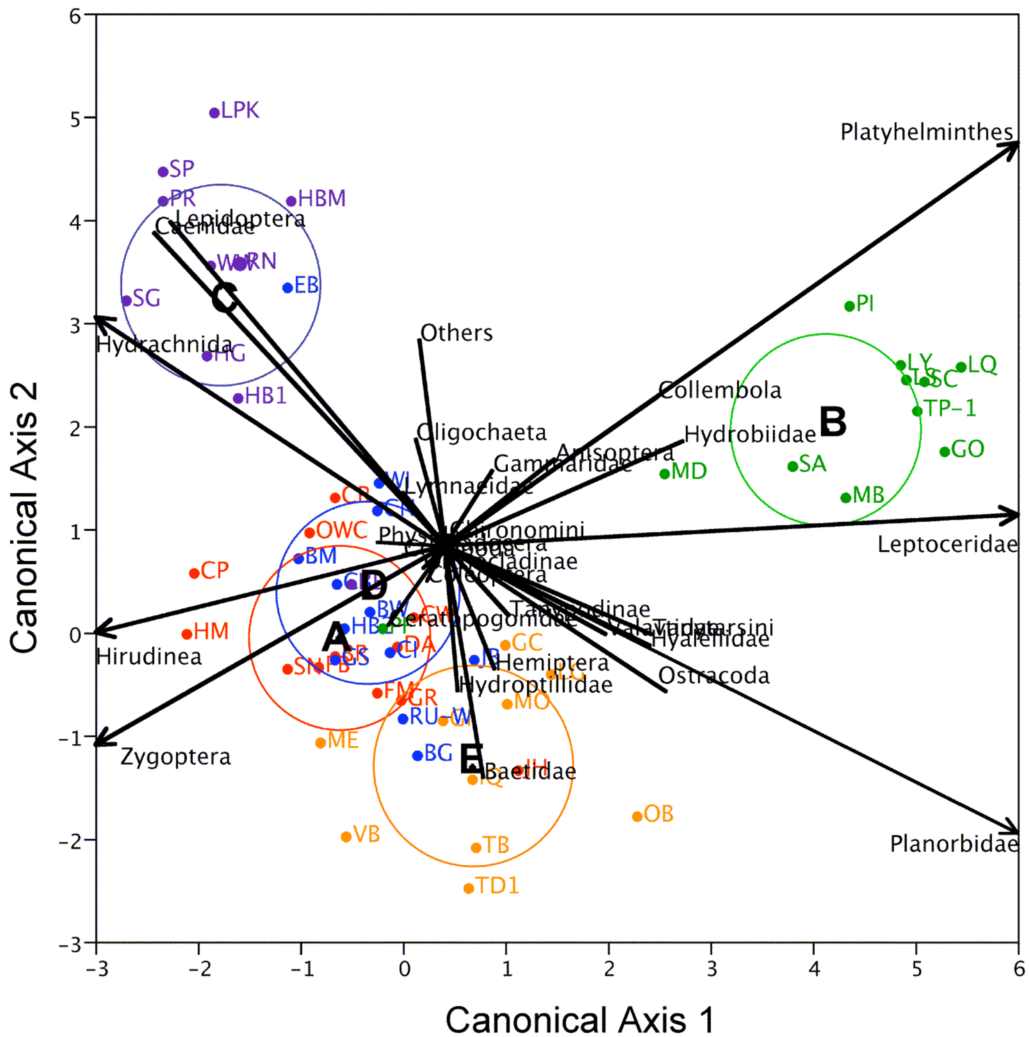

Figure 2-5. Discriminant analysis results of the 55 wetlands (in their cluster groups) with the 30 taxa.

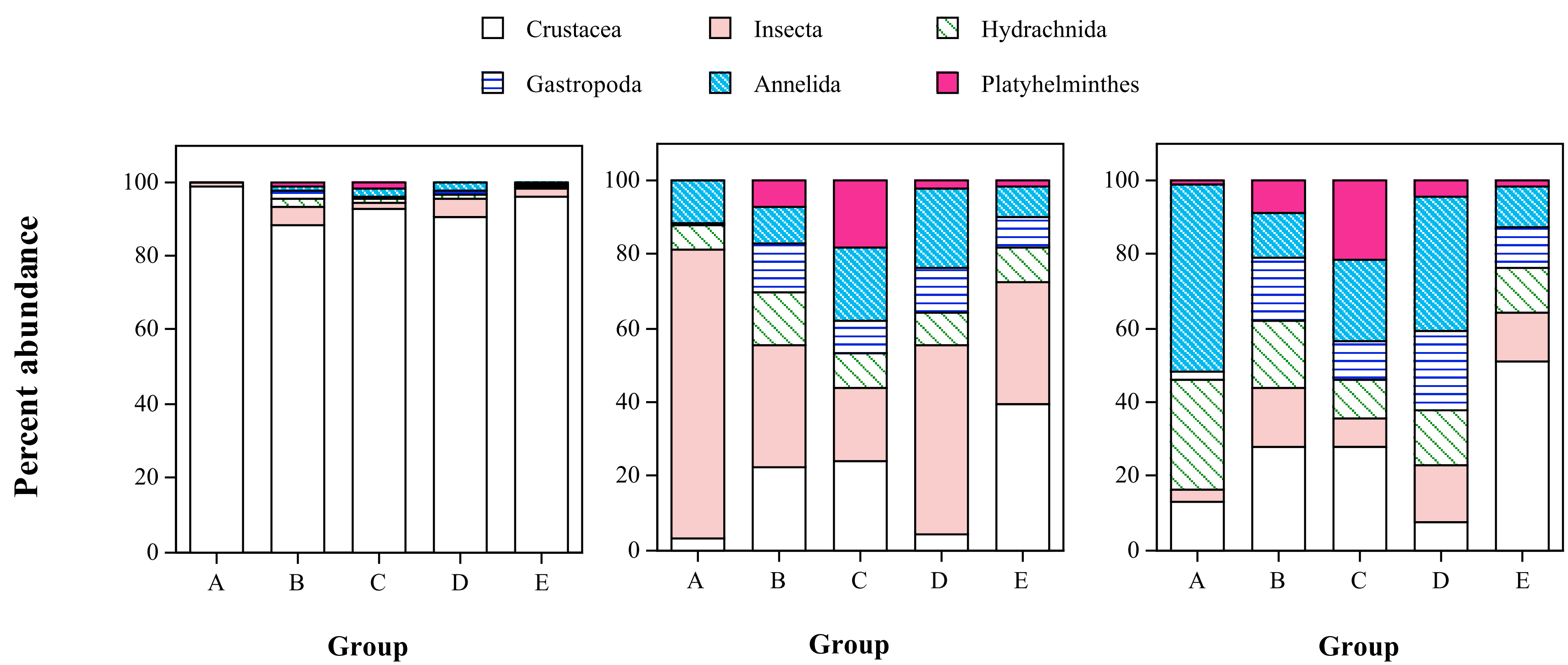

Figure 2-6. Zoobenthos Community Composition Structure a) includes all taxa, b) Crustacea excludes microcrustaceans (Cladocera, Copepoda, Ostracoda) only including Amphipoda, and c) excludes microcrustaceans and dipterans (Chironomidae and Ceratopogonidae).

Appendix

Effect of wetland quality on sampling bias associated with two fish survey methods for coastal wetlands of the lower Great Lakes

[*]**Chow-Fraser, P., Kostuk, K., Seilheimer,T**.
McMaster University,
Biology Department,1280 Main St. West
Hamilton, ON, Canada L8S 4K1

Weimer, M.,
U.S. Fish & Wildlife Service,
Lower G.L. Fishery Resources Office,
405 N. French Rd., Suite 120A, Amherst, NY, 14228

MacDougall, T.
Ontario Ministry of Natural Resources,
Lake Erie Management Unit, P.O. Box 429,
1 Passmore Street, Port Dover, ON, N0A 1N0

and

Theÿsmeÿer, T
Royal Botanical Gardens, Burlington, ON, L8N 3H8.

*Corresponding author: email: chowfras@mcmaster.ca

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Summary

We compared sampling biases associated with two different methods (24-h fyke nets [FN] versus daytime boat electrofishing [EB]) that are commonly used to survey fish communities in coastal wetlands of the Great Lakes. During June and July of 2001 and 2002, we employed both methods to survey the fish community in eleven coastal marshes of Lakes Erie and Ontario that ranged from very degraded to excellent quality based on the Water Quality Index (WQI; scores range from -3 to +3 where a value of –3 indicates the most degraded wetland and +3 indicates the highest quality. Of the 9592 fish (totaling 218.5 kg), FN surveys accounted for 88% and 58% of the total number and biomass, respectively. Regardless of wetland quality, there was a consistently higher catch associated with FN, with an average of 770.2 (± 382.8 SE) for FN versus 101.81 (± 17.85 SE) for EB. However, the average size of the fish caught by EB was almost twice as long (122.3 ± 2.83 cm) as that caught by FN (63.6 ± 0.56 cm), and had a weight that was four times greater (85.8 ± 9.48 g versus 17.2 ± 1.05 g for EB and FN, respectively). There were no significant differences with respect to the total number of species encountered per wetland (11.2 ± 0.58 versus 12.9 ± 0.99 for EB and FN, respectively); on average, FN caught 75% of the species encountered whereas EB captured 68%.

When data were sorted according to six functional feeding categories (piscivores, benthivores, omnivores, carnivores, herbivores, planktivores), we found a significant effect of fishing method on distributions among the six categories (P=0.0001; Chi-square); further analysis of the data by wetland revealed significant effect of the method

for all wetlands except the two most degraded. Eight species were recovered exclusively by EB and all occurred in relatively low numbers (<6 individuals/ species in all wetlands). By comparison, there were ten species that were captured exclusively by FN, and four were present in relatively high numbers (up to 279 individuals in one wetland). Overall, EB appeared to systematically catch larger (with respect to both size and weight) benthivores, planktivores, carnivores, and herbivores. The number of species-functional groups recovered by FN in wetlands decreased significantly (P=0.02) with WQI score, whereas that recovered by EB increased significantly (P=0.03) with WQI score. In a similar manner, the percent of total species-functional groups recovered by FN decreased significantly whereas that recovered by EB increased significantly with WQI score (P=0.03 and 0.004, respectively). Therefore, sampling bias associated with fishing method was dependent on wetland quality, a factor that should be taken into consideration in the design of large-scale sampling programs when both gear types are used, and when data from basin-wide surveys involving both gear types and sampling protocols are compared.

INTRODUCTION

Coastal wetlands provide important spawning and nursery habitat for many fishes of the Great Lakes (Jude and Pappas 1992) and have been the target of extensive restoration and conservation efforts in Canada over the past decade (Environment Canada and U.S. Environmental Protection Agency 1999). The ecology of these coastal wetlands are known to be strongly influenced by land-use characteristics of their watersheds (Crosbie and Chow-Fraser 1999; Lougheed et al. 2001; Thoma 1999); in heavily settled regions of the Great Lakes basin, many of the coastal wetlands have been severely degraded by increased sediment and nutrient loading from agricultural and urban runoff (Maynard and Wilcox 1997). Consequently, the current status of many of the wetlands in Lakes Erie and Ontario are highly variable, ranging from severely degraded coastal marshes of western Lake Ontario and Erie, to relatively undisturbed ones of eastern Lake Ontario (Chow-Fraser 2005). To properly assess their current status and to track changes in wetlands through time, ecologists must develop robust habitat assessment tools that can be used repeatedly and that can be applied widely across all environmental conditions and physiographic regions, similar to those that exist for other aquatic ecosystems (e.g. Munné et al. 2003).

A variety of sampling gear and protocols have been used in the literature to characterize the fish communities of Great Lakes coastal wetlands, and these include *passive-capture* gears such as gill nets, trap nets, and fyke nets, as well as *active-capture* gears such as beach seines, trawls, plankton nets and electroshockers (backpack or boat electrofishing) (e.g. Chubb and Liston 1986; Stephenson 1990; Jude and Pappas 1992; Leslie and Timmins 1992; Brazner 1997). Passive gear involves the capture of fish through an entrapment device or entanglement, in which the fish come into the gear on their own and are trapped (Hubert 1989). A good example of passive gear is the fyke net, which are most effective when they are set in pairs parallel to shore in coastal wetlands (Brazner 1997). These modified hoop nets have two wings, and a lead that connect their mouth opening. When fish swim away or into shore, they are guided into the funnel by wings and the lead. In contrast, electrofishing is an active method, since it is used to seek out fish where they occur at the time of sampling. The electrofishing unit creates an electrical field that momentarily stuns the fish and causes it to float to the surface so that it can be picked up by dip nets for processing (Reynolds 1989). The current density must be neither too low nor too high, else the fish would either escape or die, respectively.

The goal of this study is to investigate sampling biases associated with two different sampling protocols (24-h fyke nets versus daytime boat electrofishing), both of which are currently used by researchers to develop indicators of habitat quality for coastal wetlands of the Great Lakes basin (Great Lakes Coastal Wetland Consortium; http://www.glc.org/wetlands). We wanted to compare differences with respect to the taxonomic affiliation, mode of feeding, size and number of fish caught by the two different methods. The feeding mode was of particular interest to us because fish communities tend to change from one dominated by piscivores to one dominated by benthivores and planktivores as wetlands become degraded (e.g. Chow-Fraser et al. 1998), and if sampling bias reflected differences in feeding mode of the fish, then wetland quality would be an important factor to consider. Hence, we examined the bias associated with these two gear types as a function of wetland quality. Our results will provide a scientific basis to set criteria for proper cross-study comparisons, and to guide development of meaningful long-term, basin-wide monitoring programs.

METHODS

Study Sites

During the summer of 2001 and 2002, we used two methods (see description below) to survey fish communities in eleven coastal wetlands of Lake Erie and Ontario (Table 1; Figure 1). Study sites were chosen to represent a range of wetland quality, based on Chow-Fraser's (2005) Wetland Water Quality Index (WQI), which classified 146 wetlands into six categories (excellent, very good, good, moderately degraded, very degraded and highly degraded), based on a suite of physico-chemical, nutrient, and water clarity variables. Five wetlands in this study had been classified as being in good or very good condition, while six had been classified as being moderately to highly degraded (Table 1).

Fish Sampling Methods

Data for this study were collected in collaboration among four different research groups/agencies. All fyke nets were set and processed by McMaster University, whereas fishing with electrofishing boat was performed by three different agencies, using slightly different protocols as indicated in Table 1. We purposely involved different agencies around the basin that are responsible for routine fish surveys so that our database would be a realistic reflection of the type of data that would be made available for basin-wide comparisons. We recognize that this type of collaborative sampling would introduce errors due to differences in protocols, effort and sampling gear, but we feel that the trends that emerge from such a heterogeneous database would be statistically robust and thus widely applicable. The main goal of this study was to identify possible biases associated with each method rather than to determine which of these gear types or protocols performed better overall.

Fyke nets (FN)

One to three pairs of fyke nets were deployed in each wetland (see Table 1 for types and numbers of nets used at each site). The large nets (3 m long; 0.9 m x 1.2 m rectangular front openings; 1.27 cm for one net and 0.19 cm nylon mesh for the other) had five 76 cm stainless steel rings forming two throats that led to a cod end, and were deployed in approximately one meter of water. In contrast, the small nets (1.5 m long; 0.9 m x 0.3 m rectangular front openings; 0.19 cm nylon mesh for both nets) could only be deployed where water depths were shallow (< 0.5 m). Wings (0.9 m x 3 meters; 0.19 cm mesh) on each side of small and large nets were oriented at a 45° angle from the front opening. For many of these, fyke nets (large or small) were joined with 7.6 m leads (0.19 cm nylon mesh). Regardless of size and number of nets used, all nets were set in pairs parallel to shore, and staked into place with six pieces of 3 m steel conduit. Parallel set-up along the shoreline was chosen over perpendicular, based on recommendations of J. Brazner (U.S. Environmental Protection Agency, Duluth, Minnesota, personal communication). To prevent death due to suffocation of air-breathing species such as turtles, ducks and small mammals, 1000mL nalgene bottles were placed at the cod end to provide an air pocket.

Fyke nets were left to capture fish for approximately 24 h in each wetland, after which all fish that were present in the nets were removed and identified to species (according to Scott and Crossman 1998) and then released. Unknown species (especially small fish) were anesthetized, labeled, and then kept frozen until they could be identified at a later date. Their lengths were measured and later used with length-weight regressions (Schneider et al. 2000) to generate biomass estimates. When certain species were too abundant to process individually, they were grouped into size classes (small and large) and a suitable subset was measured and the average lengths were applied to the sub-groups. To the extent possible, wetland fishing occurred in areas that best represented the distribution of habitat and variation in conditions. Criteria included appropriate depth, and proximity to emergent vegetation and the presence of submergent vegetation; however, this was not always possible, especially in degraded wetlands where there were little or no submergent vegetation present during the fishing surveys.

Electrofishing boat (EB)

Usually within a day or two of sampling a wetland with fyke nets, we surveyed the same location in the wetland with an electrofishing boat. Characteristics of depth, presence/type of aquatic vegetation, and general substrate type were similar to those for FN. The actual fishing was carried out by three different agencies: U.S. Fish and Wildlife Service (USFWS) at Amherst, New York, Ontario Ministry of Natural Resources (OMNR) at Port Dover, Ontario, and Royal Botanical Gardens (RBG) at Burlington, Ontario (see Table 1). In all cases, the EB was conducted during daylight hours. The specific protocols used by each agency will be outlined in detail below. Total effort in shock-seconds for each wetland is given in Table 1. In all cases, fish were processed in the manner similar to that described above for fyke net fishing. Afterwards, all fish were returned to the site of capture and released.

USFWS (Amherst, NY)

Electrofishing was conducted using a 15-foot (4.6 m) jonboat outfitted with a Smith-Root 2.5 GPP electrofishing system and a 15-hp outboard motor. The boat had a single boom-mounted anode, consisting of a 36-inch (91 cm) diameter collapsible umbrella-style array, with the boat hull acting as the cathode. The anode boom was positioned at an angle of approximately 20° left of boat centerline to accommodate close-shoreline sampling. Electrofishing settings were typically 120 pulses per second DC current, with output range of 6-8 amperes GPP, powered by a 5.5 horsepower gas-powered generator. In wetlands with lower conductivity (<130 μS), output range was often limited to 4-6 amperes GPP. Boat speed was approximately 1-m • sec^{-1}, depending upon wind direction, presence of vegetation, and flow rate (if any). Shocking was conducted in linear transects, typically parallel to shore, targeting depths of approximately 1 to 1.5 m in depth. Several transects (minimum of 300 shock sec per transect) were conducted in each wetland. Effective width of area shocked was approximately 2-3 m, centered around the submerged anode (umbrella array). During sampling, one person was stationed at the bow of the boat with a long-handled fiberglass dip net to retrieve fish, while the boat operator conducted additional fish netting, as needed. All fish shocked during transects were netted and placed into a live-well on board for identification to species level and measurement (total length to the nearest mm). Any stunned fish missed during the initial pass were netted while driving back over the length of the original transect (without deploying electrofishing equipment). During 2002 sampling, a DC-powered trolling motor was used for better control of the boat, and to minimize potential disturbance to fish. In general, transparency was relatively high, but in more turbid wetlands, it was potentially more difficult to spot and retrieve stunned fish. Presence of dense aquatic vegetation posed an additional problem, as fish would sometimes become entangled in plants below the surface and were difficult to retrieve. Smaller fish (larvae, juveniles, and some cyprinids), and ictalurids (all sizes), appeared more likely to be missed as a result of sampling in heavy vegetation.

OMNR (Port Dover, ON)

Ontario Ministry of Natural Resources used a 6 m centre-console boat (Smith-Root SR-20) equipped with a Smith Root GPP 7.5 electrofisher. Dual cable-drop anodes were extended on 1.5 m booms from the bow of the boat at an approximate angle of 30° from the centreline. The boat hull acted as the cathode (anode/cathode ratio 1:10 maximum). The area to be sampled was shocked with pulsed (60 pulses/sec) DC current, correcting voltage and %-range settings to maintain a power output of 4000-5000 Watts (typically 400-500 Volts and 10 Amperes). Two people retrieved fish with 3-m long dip nets. Boat speed was maintained at a slow idle, backtracking over areas where the netters failed to obtain all stunned fish on the first pass. Effort was limited to 1,000 shock sec, covering an approximate area of 5-7,000 m^2. All fish captured were placed into an aerated live-well and allowed to recover before sampling.

RBG (Burlington, ON)

Royal Botanical Gardens used an 5.5 m flat-bottom Grumman. During electrofishing, propulsion was provided by a Minn Kota 2 hp electric trolling motor, to avoid disturbing the fish. The electrofisher was the Smith-Root GPP 5.0 portable electrofishing unit with a 9 hp generator, a tote barge, and a 6 m anode line and anode. The anode used a 30-cm diameter anode ring. The area to be sampled was shocked with a series of point shocks (500 Volts, 6 Amperes; 60 pulses/sec). The crew consisted of 3-4 members, with one crew member operating the anode, while the others netted the stunned fish. All fish netted in a transect were placed in a live-well. Effort varied for the number of shock seconds per wetland, but always covered a minimum of one 100-m^2 transect (50 m x 2 m).

Determination of Functional Feeding Categories

We consulted Scott and Crossman (1998) to determine if the species and life stage of the fish in question was primarily piscivorous, carnivorous (mainly insects and other invertebrates in diet), omnivorous (consuming algae and zooplankton), benthivorous (primarily benthic invertebrates and other organisms that reside in the sediment), herbivorous (mainly algae and plant material) or planktivorous (eating primarily zooplankton). Hence, within one species, the juveniles may be carnivorous, whereas the adults would be piscivorous (e.g. largemouth bass).

Statistical Analysis

All data manipulation, cross-tabulation analyses, ANOVA, non-parametric (Wilcoxon sign test) and linear regression analysis were performed with SAS JMP 4.04 on a Macintosh™ computer. We first ensured that the variables were not spatially autocorrelated (using S-plus in Arcview) before we used the Chi-square goodness-of-fit test to determine if gear type had a significant effect on the distribution of functional feeding categories in the eleven wetlands.

RESULTS

We caught 9,592 fish, representing 47 species, totalling approximately 220 kg in the eleven wetlands (Table 2; Figure 2). The 47 species were further sorted according to functional feeding categories (piscivores, carnivores, omnivores, planktivores, benthivores, and herbivores) to yield a total of 55 species-functional groups (henceforth referred to as functional taxa) that accounted for both taxonomic affiliation and diet at the different life stages of the organism. Fyke net accounted for a disproportionate amount of the total catch and biomass (88% and 58%, respectively), and a larger proportion of the total species and functional taxa encountered (85 and 84% versus 77 and 73% for FN and EB, respectively). Despite significant differences between catch data for the two methods (Wilcoxon Sign Test; P=0.0004), the average species richness per wetland was similar (12 versus 12.9 for EB and FN, respectively). However, there was a systematic bias towards larger fish (two-way ANOVA; P<0.0001) in the EB relative to FN surveys (85.8 vs 17.2 g and 122.3 vs 63.6 cm , respectively; Table 2).

Species that were encountered frequently (more than 100 occurrences in the wetlands combined) in these surveys included white perch, pumpkinseed, bluegills, juvenile largemouth bass, adult brown bullhead, yellow perch, blacknose shiner, alewife, sunfish and adult gizzard shad (Figure 2). Of the 55 functional taxa, six were ubiquitous, found in eight or more of the eleven wetlands when catch data from either gear type were considered (Table 3). These included rockbass, pumpkinseed, bluegill, juvenile and adult yellow perch, and brown bullhead. Except for juvenile yellow perch, FN recovered twice as many fish as did EB. There were similar disparities in the number of fish recovered for juvenile largemouth bass, white perch, and bullheads.

We compared how the two methods represented overall species richness in each wetland (Table 2). The average number of species and functional taxa recovered for both methods combined were 17.1 and 19, respectively. There were no significant differences between the mean number of species for EB and FN (11.3 versus 12.9; Wilcoxon sign test; P=0.19), nor between the number of functional taxa for either method (mean of 12.1 versus 14.2 for EB and FN, respectively; Wilcoxon sign test; P=0.14; Table 2). However, when we accounted for differences in wetland quality, we found a predictable bias associated with the two gear types. The number of functional taxa captured in wetlands by FN decreased significantly with WQI score (see Table 1) whereas that captured by EB increased significantly with WQI scores (Figure 3a). Therefore, there was a systematic bias towards more species being recovered by fyke net surveys in the poor-quality wetlands, and towards more species being caught by electrofishing boat in good-quality wetlands. These relationships were confirmed when we regressed the corresponding percentages against WQI scores (Figure 3b).

We also wanted to determine if there were sampling bias in the size of fish caught by the two methods once we accounted for differences in functional feeding groups. Functional category and gear type each had a significant effect on the mean length and mean size of fish caught, and there was also a significant interaction between these two factors (two-way ANOVA with interaction; $P<0.0001$ for all effect tests). Mean weight and length of benthivores, planktivores, carnivores and herbivores were significantly larger for fish caught by EB (Figure 4a and b), whereas corresponding size of omnivores were significantly larger in FN surveys. However, there was no significant difference in the size of piscivore caught by the two sampling gear, either in regards to the mean length or mean weight.

We sorted the data by functional feeding category to further examine sampling bias associated with the two gear types within wetlands. Catch data for the eleven wetlands are presented in Figure 5. The general tendency for FN to catch a larger number of fish was confirmed. Another obvious feature in this comparison is the distinct absence of planktivores and herbivores in the good-quality and moderately degraded wetlands (WQI scores < 0.1); only the very degraded wetlands (WQI scores > 0.1) had fish in this functional feeding group. General trends for the corresponding biomass data were very similar (Figure 6).

To properly test the hypothesis that there were no significant differences in fish distribution among the feeding categories that could be attributed to sampling methods used, we carried out a categorical analysis (log-likelihood ratio in Chi-square goodness-of-fit test) after first verifying that the data were not spatially autocorrelated. The results were highly significant ($P<0.0001$), confirming an effect of gear type on the distribution of fish in the six functional categories. We then performed Chi-square tests for individual wetlands to determine if all wetlands were similarly affected. To make these tests valid, we had to reduce the number of categories to three (piscivores, benthivores and others) to avoid empty cells. In all cases except for the most degraded sites (Grand River and Grindstone Creek), we found a significant effect of sampling gear on the fish distributions (Table 5).

We summarized all taxa that were recovered exclusively by one gear type in this survey. There were eight taxa recovered exclusively by EB, compared with ten by FN (Table 6). Consistent with previous trends, FN tended to catch comparatively more of the smaller individuals. All taxa recovered by EB occurred in relatively low numbers (< 6), whereas several of those caught by FN occurred in greater numbers (up to 279 individuals). Because grass pickerel had been recovered exclusively in five of the eleven wetlands by EB, we suggest that FN is not effective at sampling this taxa. Using the same reasoning, EB appears to be ineffective for sampling tadpole madtom, since this taxa was caught exclusively by FN in four of the eleven wetlands, presumably because it is a very small fish that would be difficult to catch with EB. Nevertheless, most of the other species listed in Table 6 occurred in low numbers (1 or 2 individuals) except for juvenile bullheads and white crappie.

We also compared the performance of the two sampling gear on a species-by-species basis; to ease comparison, data were presented according to the six functional feeding categories. Except for rock bass, both EB and FN were similar in their ability to capture carnivorous species across the full spectrum of wetland conditions (Figure 7). In most cases, the higher catch-per-unit effort associated with the FN method relative to EB was evident for carnivores, but this could not be said generally for the other feeding categories (Figure 8 and 9). For piscivores, however, EB was better at capturing largemouth bass and northern pike but did not appear to be as effective as FN in capturing yellow perch in degraded wetlands (Figure 8). Both techniques appeared to be equally effective in sampling benthivores (Figure 8). The main observation regarding omnivores was that FN was better at capturing these species in the degraded sites, whereas EB appeared to be better at the good-quality sites, especially for golden shiner (Figure 9). Both planktivores and herbivores were present only in the more disturbed wetlands, and whereas the former were caught with both gear types without any obvious bias, EB appeared to be better at capturing gizzard shad (Figure 9).

Discussion

A variety of methods have been used to assess fish communities of Great Lakes coastal wetlands. In this study, we compared the performance of two very common methods, paired fyke nets (FN) set for 24-h, and electrofishing boat (EB) performed during the daytime. In the eleven wetlands sampled in this survey, FN recovered significantly more fish than EB per effort, and this was generally true when the data were sorted according to species or to functional feeding categories (Tables 3 and 5). However, the EB method generally caught larger fish (Table 2); mean weight and length of benthivores, planktivores, carnivores and herbivores caught in EB surveys were significantly larger than those caught in FN surveys (Figure 4a and b). A more important finding is that the quality of wetland affected the number of functional taxa captured in the wetland. As wetlands became more degraded (i.e., WQI score decreased), the number of functional taxa recovered by FN increased (P=0.02), whereas that recovered by EB decreased (P=0.03) (Figure 3a). These trends were upheld when we standardized the data as a percent of total functional taxa and performed the regression again (P=0.03 and 0.004 for FN and EB, respectively) (Figure 3b). Therefore, sampling bias associated with gear type was dependent on wetland quality, and when this difference was ignored, there were no significant differences in the number of species (mean of 11.3 versus 12.9 for EB and FN, respectively) or functional taxa (mean of 12.1 versus 14.2 for EB and FN, respectively) associated with the two methods (Table 2).

Differences in capture efficiency observed in this study can be attributed to differences in specific features of the gear and how they operate in the wetlands. All else being equal, both the size of the frame and size of mesh used in the fyke nets will affect fish size (Hubert 1989; Shoup et al. 2003). Therefore, surveys that include both large and small (sometimes referred to as mini-fyke nets) nets would catch fish with overall smaller mean size. On the other hand, the EB will tend to select for larger fish since the total body voltage increases with length, and small fish are not as easily stunned as large fish for a given voltage. As well, larger fish are more visible to the operator and may be preferentially removed from the water column during the transect (Reynolds 1989; Wiley and Tsai 1983). That we used both small and large fyke nets in 8 of 11 wetlands (Table 1) may explain why the overall size of fish caught by FN was significantly smaller than that caught by EB. This tendency for EB to capture bigger fish has been well documented in other studies (e.g. Bohlin et al. 1989; Copp 1989).

The apparent shift in the fish community along the degradation gradient from one in which carnivores and piscivores dominated in the better quality wetlands (low WQI scores) to one in which planktivores and herbivores dominated in the poor-quality sites (Figures 5 and 6) is consistent with documented changes in aquatic food-webs associated with wetland degradation in Cootes Paradise Marsh, a Lake Ontario coastal wetland that became degraded by cultural eutrophication over the course of 6 decades (Chow-Fraser et al. 1998). During the 1940s, when the marsh had been extensively vegetated, piscivores such as northern pike and largemouth bass and other sunfishes dominated, and there had been many shiner species as well as rock bass that fed on the abundant insects and other invertebrates associated with macrophytes. However, as the marsh became degraded from sewage effluent over the course of the next three decades, the macrophyte community declined while the algal community proliferated and became dominated by several nitrogen-fixing blue-green species as well as filamentous and colonial green algae that formed blooms throughout the summer. The fish community that dominated this degraded state during the 1970 and 1980s consisted mainly of benthivores such as common carp and brown bullheads, planktivores such as alewife that migrated seasonally into the marsh, and gizzard shad, a herbivore that fed on the plentiful algae in the marsh (Chow-Fraser et al. 1998).

A possible explanation for the differential effect of wetland quality on the capture efficiency of the two fishing methods (Figures 3a and b), is that EB is better at capturing the sedentary, territorial, or less active species (Hubert 1989; Holland and Peters 1992) such as nest guarders (e.g., black crappie and largemouth bass) and ambush predators (e.g., northern pike) that tend to be associated with the well vegetated shallow environments in good-quality wetlands (Scott and Crossman 1998). This is because the electrofishing boat can cover a large sampling area and thereby increase encounter probability for these individuals within macrophyte beds. We speculate that in poor-quality wetlands, where both submergent and emergent vegetation are scarce and the shallow waters warm up during the day, the fish must migrate to the cooler, deeper water where they are not easily sampled by EB (e.g., northern pike and yellow perch in Figure 8). Under these degraded conditions, then, FN would be more effective because the nets could trap the fish when they migrate back inshore during the evening. Pierce et al. (2001) found that bluegills and yellow perch were caught in significantly higher numbers at night than during the day in their EB surveys. Hence, for fish that exhibit horizontal migration patterns, EB must be carried out at night to eliminate this bias. In general, fyke nets appear to be better at capturing species that school and that undergo migration between the offshore and inshore (e.g., golden shiner, Figure 9).

Another reason that may explain the differential performance of FN versus EB along the degradation gradient (Figure 3a and b) is that species that tolerate conditions in degraded wetlands are smaller (e.g., brown bullhead, shiners and gizzard shad) and are therefore not readily captured by EB as explained earlier. High turbidity normally associated with degraded wetlands can also obscure fish retrieval and this has been cited as a drawback of EB when compared with other gear such as a drop net or a pop net when sampling in vegetation (Dewey 1992). Reynolds (1989) has also noted that the fright response of fish is greater in areas with little submerged vegetation (e.g. in more degraded sites), although this response is dampened at night.

We found that capture efficiency of the two methods was affected by the life stage of some fish. For instance, we obtained greater catches with FN for juvenile largemouth bass (Figure 7) while greater catches were obtained with EB for mature individuals (Figure 8). Reynolds and Simpson (1978) also found that the capture efficiency of electrofishing techniques increased as size of largemouth bass increased, and warned that electrofishing may seriously underestimate the number of young bass.

Besides differences in capture efficiencies, each method has its own advantages and disadvantages. Fyke nets are easy to handle, require relatively little training to operate properly (Hubert 1989), and do not depend on the use of a boat, even though access to a boat can be an asset. Nets can be set in very shallow habitats (as low as 0.3 to 0.5 m), and water characteristics do not limit their effectiveness (e.g., turbidity, temperature, conductivity etc.). They can be set at anytime during the day and used throughout the ice-free season. When used properly, fyke nets will not generally harm the fish they capture (Holland and Peters 1992). On the other hand, there are a number of disadvantages. An often-cited drawback is the 24-h required to capture the fish, as well as the amount of time required to set the nets. Secondly, the gear cannot be deployed in water much deeper than 2 m. When non-target animals, such as muskrats or turtles, are inadvertently caught, they may eat some of the catch or else chew holes in the net that would allow the fish to escape.

A major advantage of using boat electrofishing in routine survey is the amount of time and labour saved per unit area (Pugh and Schramm 1998). It has been used in a wide variety of habitats, including rivers, lakes and wetlands, and can be effective for sampling large systems. However, EB requires intensive training and is expensive to purchase and to maintain. Results of the sampling may also be dependent on operator experience and the field protocol (due to the variation among agencies in this study) used as well as the degree of disturbance of the wetland (Hardin and Conner 1992). Capture efficiency can be influenced by the type of fish (e.g., bony fish conduct current more readily that cartilaginous fishes). Habitat characteristics, such as water temperature, water transparency, and dissolved oxygen concentration can also influence the efficiency of the catch (Reynolds 1989). Lastly, as was evident in this study, the type of vegetation present (Hardin and Connor 1992), time of day (e.g., Paragamian 1989) and time of season (Dumont and Dennis 1997) may all affect capture rates of certain species.

One obvious limitation of this study was involvement of different EB protocols by three different agencies, which affected the level of confidence in our conclusions. We emphasize the need for further studies involving a comparison of gear in which both the EB and FN protocols are standardized. Since FN sampling always preceded EB sampling in this study, it is possible that this systematic bias may have led to artificially lower fish abundances, and this possibility should be formally addressed in a future study.

On its own, neither EB nor FN was able to capture all of the species that both techniques could recover in any of the eleven wetlands (Table 4). Nevertheless, on average FN was able to catch a higher proportion of the total captured within each wetland (mean of 74 % vs. 66% for FN and EB, respectively). It is clear that when time and labour pool are available, both FN and EB should be used to survey the fish community of wetlands, a recommendation that was echoed by Fago (1998) when he compared the performance of mini fyke nets with a combination of electrofishing and small-mesh seine in Wisconsin lakes. However, when only one method can be employed, the choice should reflect the overall quality of the wetland as well as the local distribution of aquatic plants. As we have demonstrated in this study, the particular dynamics in good quality wetlands tend to make EB the preferred method, whereas degraded wetlands seem to be more effectively sampled by FN.

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Table 1. Details of fish surveys conducted in each of the study sites. WQI scores and corresponding wetland quality category are from Chow-Fraser (2003). "EB" refers to the total shock time delivered by electrofishing boat. Names in bracket below wetland names indicate the agency responsible for electrofishing.

Dates	ID #	Wetland	WQI	Wetland quality	No. of fyke nets		EB Time (sec)
					Large	Small	
7/18/01	1	Sandy Creek (USFWS Amherst)	1.226	Very good	2	1*	823
6/26/01	2	Long Point Prov Park (OMNR Port Dover)	0.954	Good	1	0	1000
6/26/01	3	Long Point Big Rice Bay (OMNR Port Dover)	0.760	Good	1	0	1000
7/19/01	4	Little Sodus Bay (USFWS Amherst)	0.417	Good	2	1*	1151
6/27/02	5	Perch River (USFWS Amherst)	0.162	Good	2*	1*	1116
6/26/02	6	Goose Bay (USFWS Amherst)	-0.050	Moderately degraded	2*	1*	942
6/27/02	7	Muskellunge River (USFWS Amherst)	-0.097	Moderately degraded	2*	1*	1204
6/25/02	8	Mud Bay (UWFWS Amherst)	-0.492	Moderately degraded	2*	1*	699
7/09/02	9	Cootes Paradise Marsh (RBG)	-1.019	Very degraded	2*	1*	1098
7/08/01	10	Grand River (OMNR Port Dover)	-1.791	Very degraded	2	0	1000
7/12/02	11	Grindstone Creek (RBG)	-1.813	Very degraded	2*	1*	517

* paired nets joined with leads

Table 2. Comparison of summary statistics for fish collected in wetlands in this study using the two fish survey methods (EB = Boat electrofishing; FN = Fyke nets). Where applicable, numbers in bracket indicate the SE.

Parameter	All fish	Survey method EB	Survey method FN
No. of fish caught	9,592	1,120	8,472
% all fish caught	---	11.7	88.3
Biomass of fish (kg)	218.5	92.7	125.8
% all fish biomass	---	42.4	57.6
No. of species recovered	47	36	40
% total species recovered	---	76.6	85.1
No. functional taxa recovered	55	40	46
% total functional taxa recovered	---	72.7	83.6
Mean fish weight (g)	25.19 (±1.46)	85.82 (± 9.48)	17.17 (± 1.05)
Mean fish length (cm)	70.5 (± 0.62)	122.3 (± 2.83)	63.6 (± 0.56)
Mean species richness per wetland	17.1* (±0.93)	11.2 (±0.58)	12.9 (± 0.99)
Mean number of functional taxa per wetland	19.0* (±0.84)	12.1 (±0.76)	14.2 (±1.10)
Mean no. fish per wetland	872.0 (± 384.92)	101.8 (± 17.85)	770.2 (± 382.80)

* This number refers to the mean number recovered for wetlands regardless of survey method.

Table 3. Total number of taxa encountered during Electrofishing Boat (EB) and Fyke net (FN) surveys in this study.

			Number of Specimens			Number of Wetlands		
Family	**Common name**	**Scientific name**	**Both**	**EB**	**FN**	**Both**	**EB**	**FN**
Carnivore								
Anguillidae	American Eel	*Anguilla rostrata*	1	0	1	0	0	1
Atherinidae	Brook Silverside	*Labidesthes sicculus*	3	2	1	0	2	1
Centrarchidae	Rockbass	*Ambloplites rupestris*	80	24	56	4	4	8
Centrarchidae	Green Sunfish	*Lepomis cyanellus*	5	0	5	0	0	1
Centrarchidae	Pumpkinseed	*Lepomis gibbosus*	1110	220	890	10	11	10
Centrarchidae	Bluegill	*Lepomis macrochirus*	682	49	633	7	7	10
Centrarchidae	Sunfish (juvenile)	*Lepomis sp.*	118	16	102	2	2	5
Centrarchidae	Largemouth Bass (30-70mm)	*Micropterus salmoides*	637	37	600	4	6	5
Centrarchidae	White Crappie (young-of-year)	*Pomoxis annularis*	47	0	47	0	0	3
Centrarchidae	Black Crappie (0-160mm)	*Pomoxis nigromaculat*	2	0	2	0	0	1
Cyprinidae	Blacknose Shiner	*Notropis heterolepis*	299	61	238	2	2	4
Cyprinidae	Spotfin Shiner	*Notropis spilopterus*	21	1	17	0	1	2
Gasterosteidae	Threespine Stickleback	*Gasterosteus aculeatus*	1	0	1	0	0	1
Esocidae	Grass Pickerel (0-100mm)	*Esox a. vermiculatus*	3	3	0	0	3	0
Esocidae	Northern Pike (larval/+50mm)	*Esox lucius*	2	0	2	0	0	1
Fundulida	Banded Killifish	*Fundulus diaphanus*	53	10	43	4	5	6
Lepisosteidae	Longnose Gar	*Lepisosteus osseus*	2	1	1	0	1	1
Moronidae	White Perch (young-of-year)	*Morone americana*	4102	2	4100	1	2	2
Percidae	Logperch	*Percina caprodes*	2	2	0	0	1	0
Percidae	Yellow Perch (1-150mm)	*Perca flavescens*	423	287	136	8	10	9
Piscivore								
Amiidae	Bowfin	*Amia calva*	27	9	18	4	5	7
Centrarchidae	Smallmouth Bass (20+mm/Adults)	*Micropterus dolomieu*	2	0	2	0	0	2
Centrarchidae	Largemouth Bass (Adult)	*Micropterus salmoides*	17	13	4	1	5	4
Centrarchidae	White Crappie (+152mm)	*Pomoxis annularis*	6	0	6	0	0	2
Centrarchidae	Black Crappie (+160mm)	*Pomoxis nigromaculatus*	5	2	3	0	2	1
Esocidae	Redfin Pickerel	*Esox a. americanus*	1	0	1	0	0	1
Esocidae	Grass Pickerel (+100mm)	*Esox a. vermiculatus*	6	6	0	0	4	0

Esocidae	Northern Pike (Adult)	*Esox lucius*	15	11	5	2	4	4
Moronidae	White Perch (Adult +178mm)	*Morone americana*	1	0	1	0	0	1
Percidae	Yellow Perch (+150mm)	*Perca flavescens*	60	15	46	4	4	8
Percidae	Walleye	*Sander vitreus*	2	2	0	0	2	0
Benthivore								
Catostomidae	White Sucker	*Catastomus commersonii*	2	2	0	0	1	0
Catostomidae	Shorthead Redhorse	*Moxostoma macrolepidotum*	3	3	0	0	1	0
Cyprinidae	Common Carp	*Cyprinus carpio*	94	53	45	2	4	3
Cyprinidae	Bluntnose Minnow	*Pimephales notatus*	155	27	129	3	4	7
Cyprinidae	Rudd	*Scardinius erythrophthalmus*	6	4	2	0	2	1
Gobiidae	Round Goby	*Neoglobius melanostomus*	3	0	3	0	0	1
Ictaluridae	Brown Bullhead	*Ameiurus nebulosus*	735	107	628	9	9	11
Ictaluridae	Black Bullhead	*Ameiurus melas*	2	2	0	0	2	0
Ictaluridae	Bullhead (juvenile)	*Ameiurus sp.*	368	0	368	0	0	2
Ictaluridae	Channel Catfish	*Icalurus punctatus*	4	1	3	0	1	2
Ictaluridae	Tadpole Madtom	*Noturus gyrinus*	13	0	13	0	0	4
Percidae	Rainbow Darter	*Etheostoma caeruleum*	7	3	4	0	1	1
Percidae	Johnny Darter	*Etheostoma nigrum*	1	0	1	0	0	1
Sciaenidae	Freshwater Drum	*Aplodinotus grunniens*	9	7	2	1	2	1
Umbridae	Central Mudminnow	*Umbra limi*	9	4	5	1	3	2
Omnivore								
Cyprinidae	Goldfish	*Carassius auratus*	1	1	0	0	1	0
Cyprinidae	Golden Shiner	*Notemigonus crysoleucas*	120	61	59	3	7	5
Cyprinidae	Spottail Shiner	*Notropis hudsonius*	40	4	36	3	3	4
Cyprinidae	Shiner (juvenile)	*Cyprinid*	2	0	2	0	0	1
Cyprinidae	Fathead Minnow	*Pimephales promelas*	11	1	10	1	1	3
Planktivore								
Clupeidae	Alewife	*Alosa pseudoharengus*	120	14	106	1	2	3
Clupeidae	Gizzard Shad (0-20mm)	*Dorosoma cepedianum*	6	6	0	0	1	0
Cyprinidae	Emerald Shiner	*Notropis atherinoides*	16	14	2	0	1	1
Herbivore								
Clupeidae	Gizzard Shad (+20mm)	*Dorosoma cepedianum*	123	33	90	1	3	1

Table 4. Comparison of numbers of functional taxa captured during Electrofishing Boat (EB) and/or Fykenet (FN) surveys. “Total” refers to the total number of taxa encountered regardless of method; “EB and FN” refers to the number of taxa that were caught by both EB and FN; “EB” and “FN” refer to the number of taxa recovered by each of the methods. “Only EB” and “Only FN” refer to the number of exclusive taxa that were captured by EB or FN. Numbers in italics are the total number of fish caught with each method. Wetlands are presented in order of WQI scores.

			Number of functional taxa captured by				
Wetland	**Lake**	**Total**	**EB and FN**	**EB**	**FN**	**Only EB**	**Only FN**
Sandy Creek #1	Ontario	**13** *465*	**6**	**11** *76*	**8** *389*	**7**	**2**
Long Pt Prov Pk #2	Erie	**18** *357*	**9**	**16** *157*	**11** *200*	**7**	**2**
Long Pt Big Rice #3	Erie	**17** *910*	**10**	**17** *197*	**10** *54*	**7**	**0**
Little Sodus #4	Ontario	**18** *415*	**9**	**13** *127*	**14** *288*	**4**	**5**
Perch River #5	Ontario	**21** *580*	**8**	**11** *70*	**18** *510*	**3**	**10**
Goose Bay #6	Ontario	**17** *335*	**5**	**12** *108*	**13** *227*	**8**	**8**
Muskellunge River #7	Ontario	**23** *261*	**7**	**12** *76*	**18** *185*	**5**	**11**
Mud Bay #8	Ontario	**21** *441*	**6**	**12** *56*	**15** *385*	**6**	**9**
Cootes Paradise #9	Ontario	**19** *4631*	**7**	**11** *121*	**15** *4510*	**4**	**8**
Grand River #10	Erie	**21** *127*	**3**	**11** *59*	**15** *68*	**8**	**10**
Grindstone Creek #11	Ontario	**21** *1070*	**8**	**9** *29*	**20** *1041*	**1**	**12**

Table 5. Summary of Chi-square statistics for functional groups. $P<0.05$ indicates that there is a significant bias in gear type used.

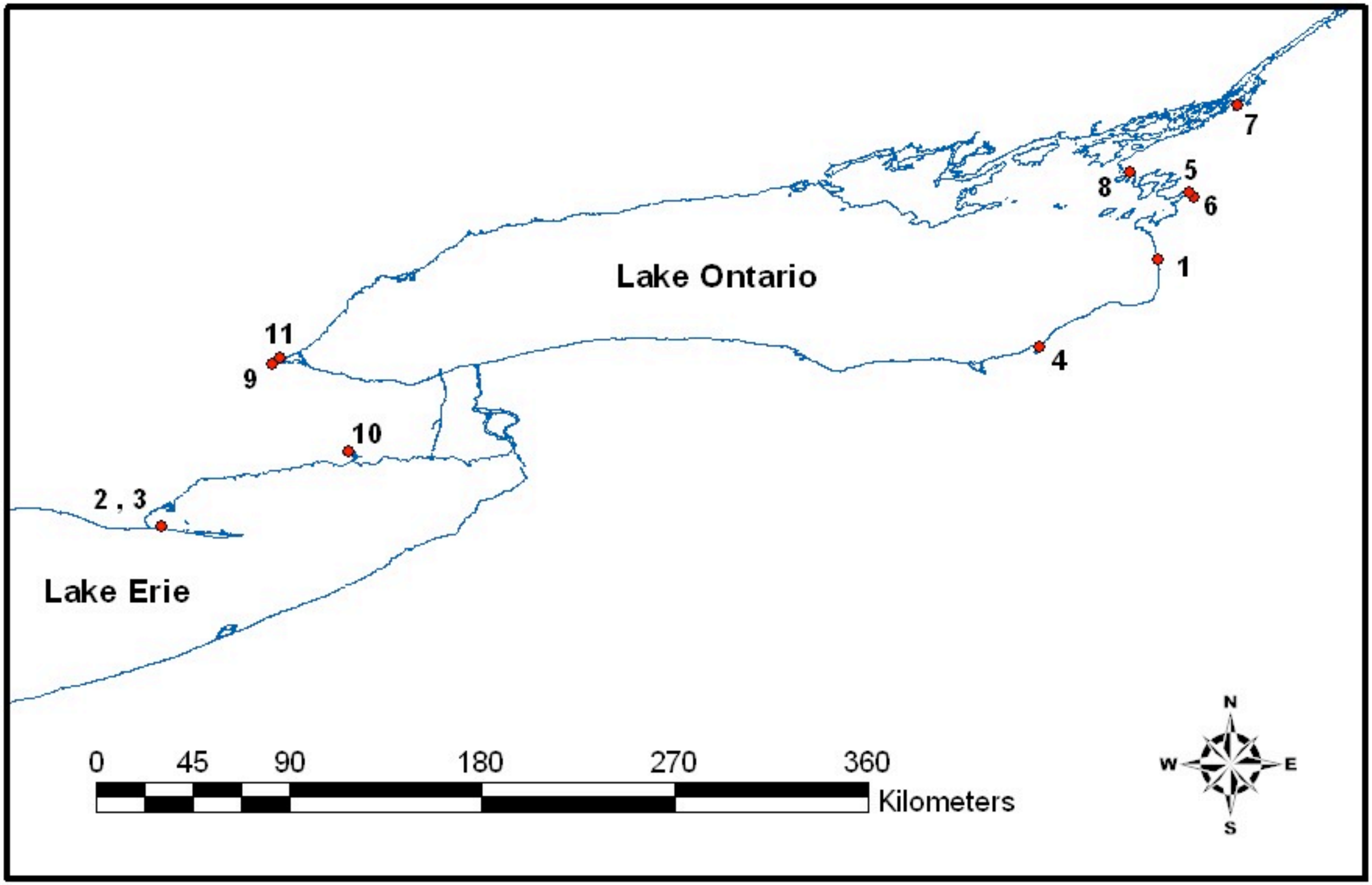
	Others		Piscivore		Benthivore		Total		
Wetland	**EB**	**FN**	**EB**	**FN**	**EB**	**FN**	**EB**	**FN**	**Prob**
Sandy Creek	66	379	4	4	6	6	76	389	0.0016
Long Point Prov Pk	138	188	12	3	7	9	157	200	0.0140
Long Point Rice Bay	210	658	12	8	19	3	241	669	<0.0001
Little Sodus Bay	115	248	9	11	3	29	127	288	0.0055
Perch River	39	161	2	3	29	346	70	510	<0.0001
Goose Bay	98	196	5	5	5	26	108	227	0.0586
Muskellunge River	45	52	7	10	24	123	76	185	<0.0001
Mud Bay	35	297	1	26	20	62	56	385	0.0020
Cootes Paradise Marsh	57	4443	0	6	64	61	121	4510	<0.0001
Grand River	31	47	6	7	22	14	59	68	0.1035
Grindstone Creek	28	924	0	3	1	115	29	1041	0.3020

Table 6. Summary of taxa recovered exclusively by one gear type in this survey. Numbers are the individuals captured in each wetland. EB = electrofishing boat; FN = fyke nets.

		Wetland ID											
Species	**Method**	**#1**	**#2**	**#3**	**#4**	**#5**	**#6**	**#7**	**#8**	**#9**	**#10**	**#11**	**Total**
Black Bullhead	EF	-	**1**	**1**	-	-	-	-	-	-	-	-	**2**
Freshwater Drum	EF	-	-	**1**	-	-	-	-	-	-	**6**	-	**7**
Goldfish	EF	-	-	-	-	-	-	-	-	**1**	-	-	**1**
Grass Pickerel	EF	**3**	**2**	**2**	**1**	-	**1**	-	-	-	-	-	**9**
Logperch	EF	-	-	-	-	-	-	-	-	-	**2**	-	**2**
Shorthead Redhorse	EF	-	-	-	-	-	-	-	**3**	-	-	-	**3**
Walleye	EF	-	-	-	-	-	-	-	**1**	-	**1**	-	**2**
White Sucker	EF	-	-	-	-	-	-	-	**2**	-	-	-	**2**
American Eel	FN	-	-	-	-	-	-	-	**1**	-	-	-	**1**
Bullhead (juvenile)	FN	-	-	-	-	**279**	-	**89**	-	-	-	-	**368**
Green Sunfish	FN	-	-	-	-	-	-	-	-	-	-	**5**	**5**
Johnny Darter	FN	-	-	-	-	**1**	-	-	-	-	-	-	**1**
Redfin Pickerel	FN	**1**	-	-	-	-	-	-	-	-	-	-	**1**
Round Goby	FN	-	-	-	-	-	-	-	-	-	-	**3**	**3**
Smallmouth Bass	FN	-	-	-	-	-	**1**	-	-	**1**	-	-	**2**
Tadpole Madtom	FN	-	**2**	-	-	**2**	**1**	**8**	-	-	-	-	**13**
Threespine Stickleback	FN	-	-	-	-	**1**	-	-	-	-	-	-	**1**
White Crappie	FN	-	-	-	-	-	-	-	-	**2**	**26**	**25**	**53**

List of Figures

Fig. 1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Lake Erie
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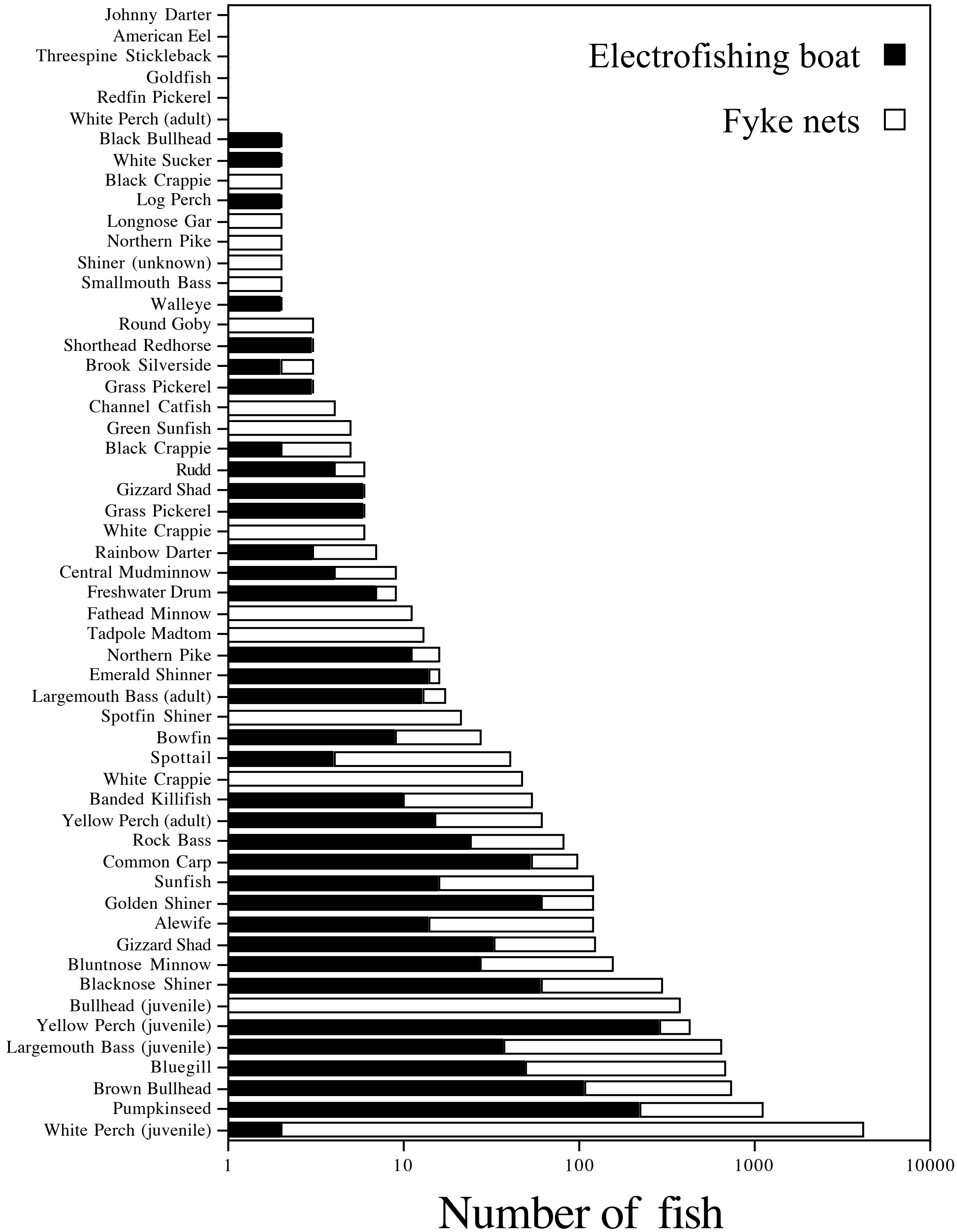
2 , 3
4
5
6
7
8
9
10
11
0
45
90
180
270
360
Kilometers
N
S
E
W

Fig. 2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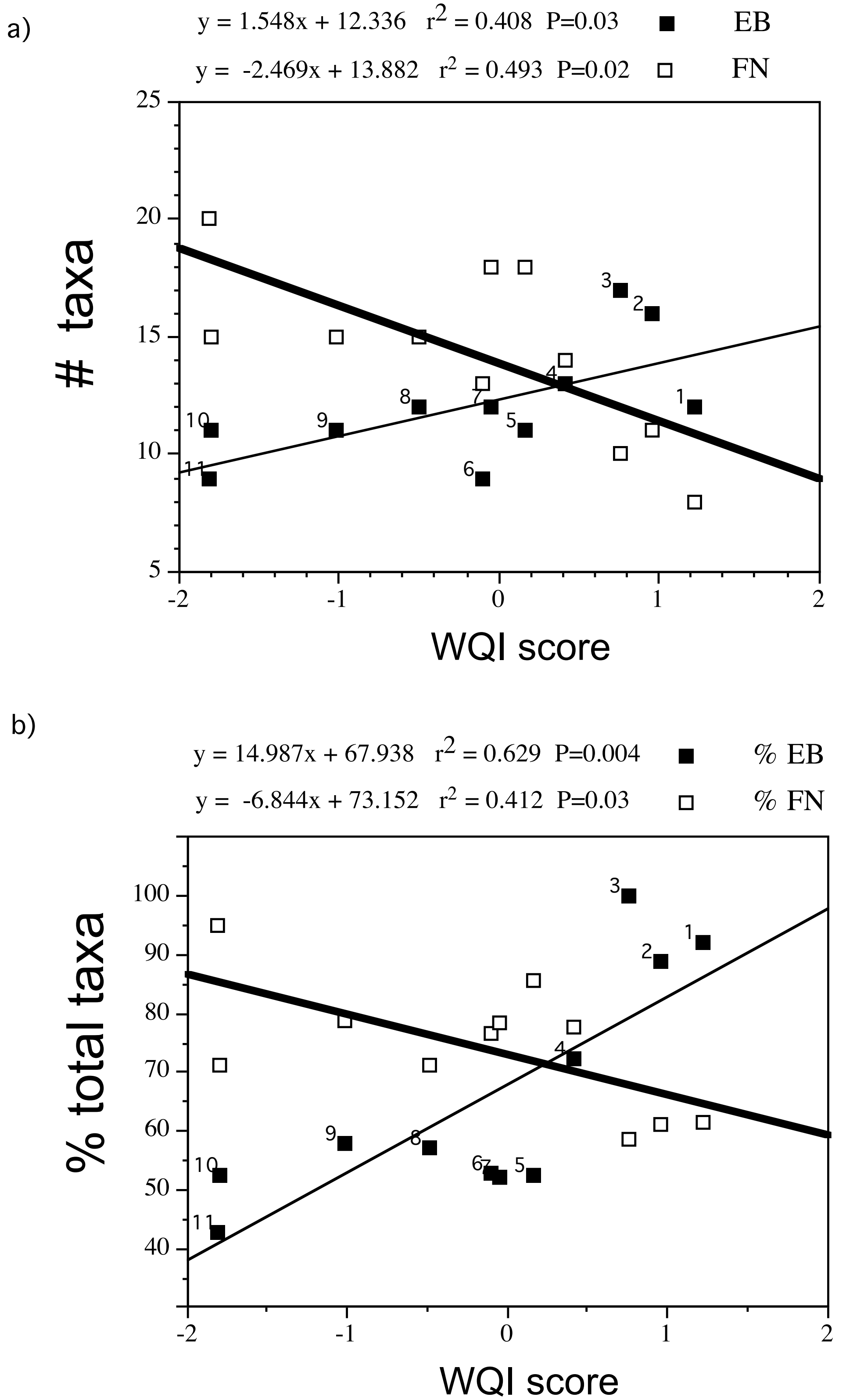
American Eel
Threespine Stickleback
Goldfish
Redfin Pickerel
White Perch (adult)
Black Bullhead
White Sucker
Black Crappie
Log Perch
Longnose Gar
Northern Pike
Shiner (unknown)
Smallmouth Bass
Walleye
Round Goby
Shorthead Redhorse
Brook Silverside
Grass Pickerel
Channel Catfish
Green Sunfish
Black Crappie
Rudd
Gizzard Shad
Grass Pickerel
White Crappie
Rainbow Darter
Central Mudminnow
Freshwater Drum
Fathead Minnow
Tadpole Madtom
Northern Pike
Emerald Shinner
Largemouth Bass (adult)
Spotfin Shiner
Bowfin
Spottail
White Crappie
Banded Killifish
Yellow Perch (adult)
Rock Bass
Common Carp
Sunfish
Golden Shiner
Alewife
Gizzard Shad
Bluntnose Minnow
Blacknose Shiner
Bullhead (juvenile)
Yellow Perch (juvenile)
Largemouth Bass (juvenile)
Bluegill
Brown Bullhead
Pumpkinseed
White Perch (juvenile)
Electrofishing boat
Fyke nets
1
10
100
1000
10000
Number of fish

Fig. 3


a)
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y = -2.469x + 13.882 r2 = 0.493 P=0.02 FN
# taxa
WQI score
b)
y = 14.987x + 67.938 r2 = 0.629 P=0.004 % EB
y = -6.844x + 73.152 r2 = 0.412 P=0.03 % FN
% total taxa
WQI score

Fig. 4

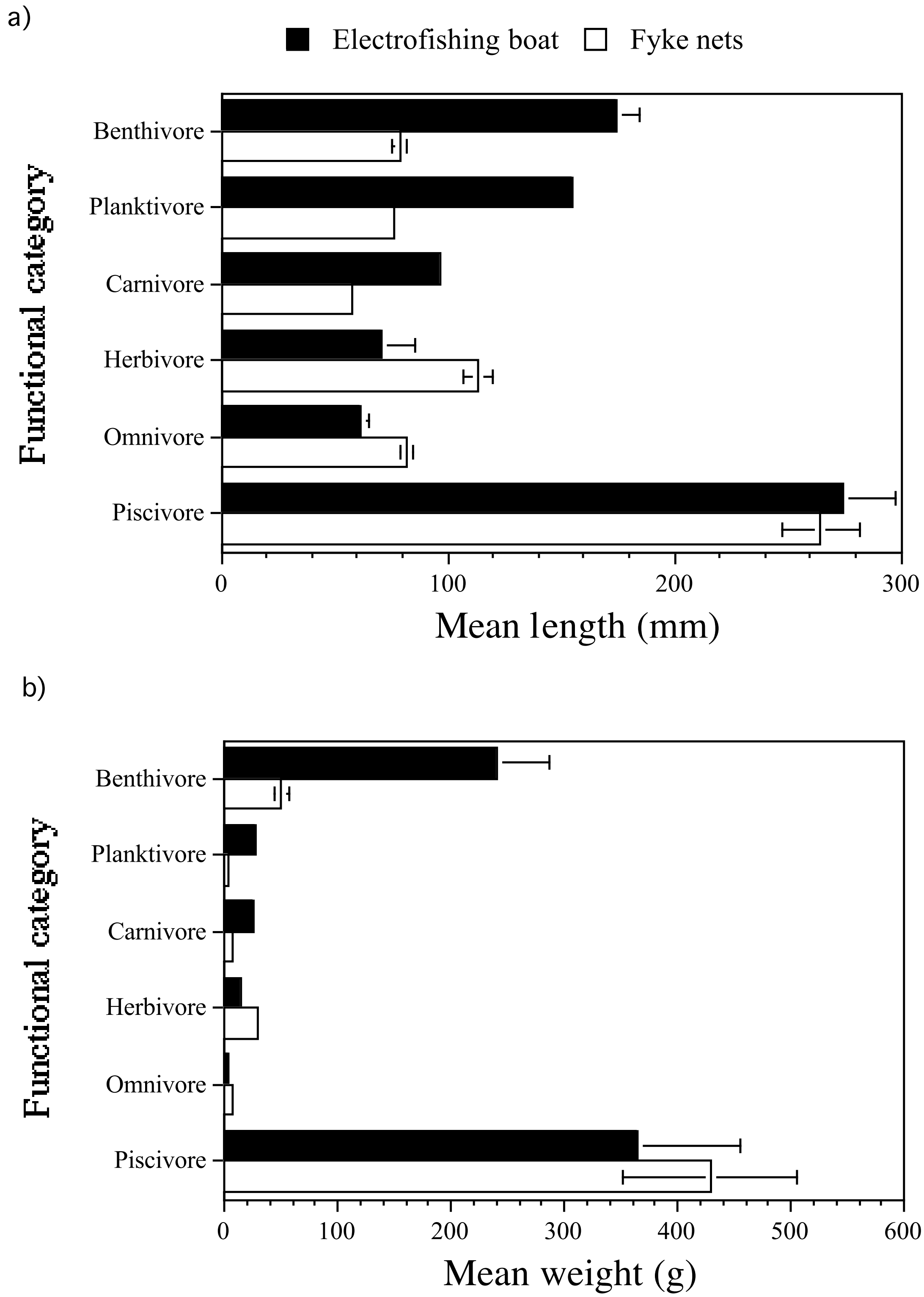

a)
Electrofishing boat
Fyke nets
Functional category
Benthivore
Planktivore
Carnivore
Herbivore
Omnivore
Piscivore
0
100
200
300
Mean length (mm)
b)
Functional category
Benthivore
Planktivore
Carnivore
Herbivore
Omnivore
Piscivore
0
100
200
300
400
500
600
Mean weight (g)

Fig. 5

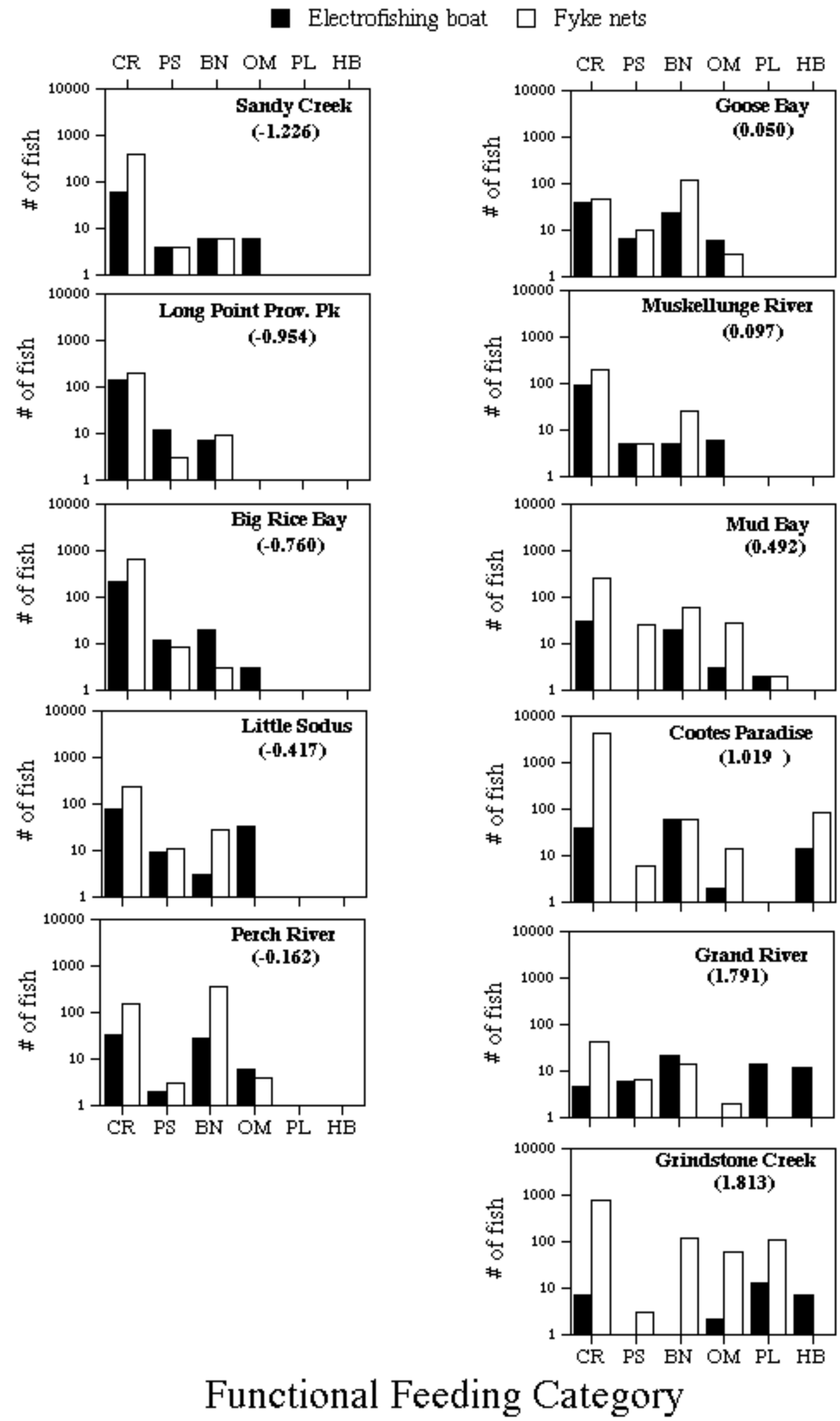
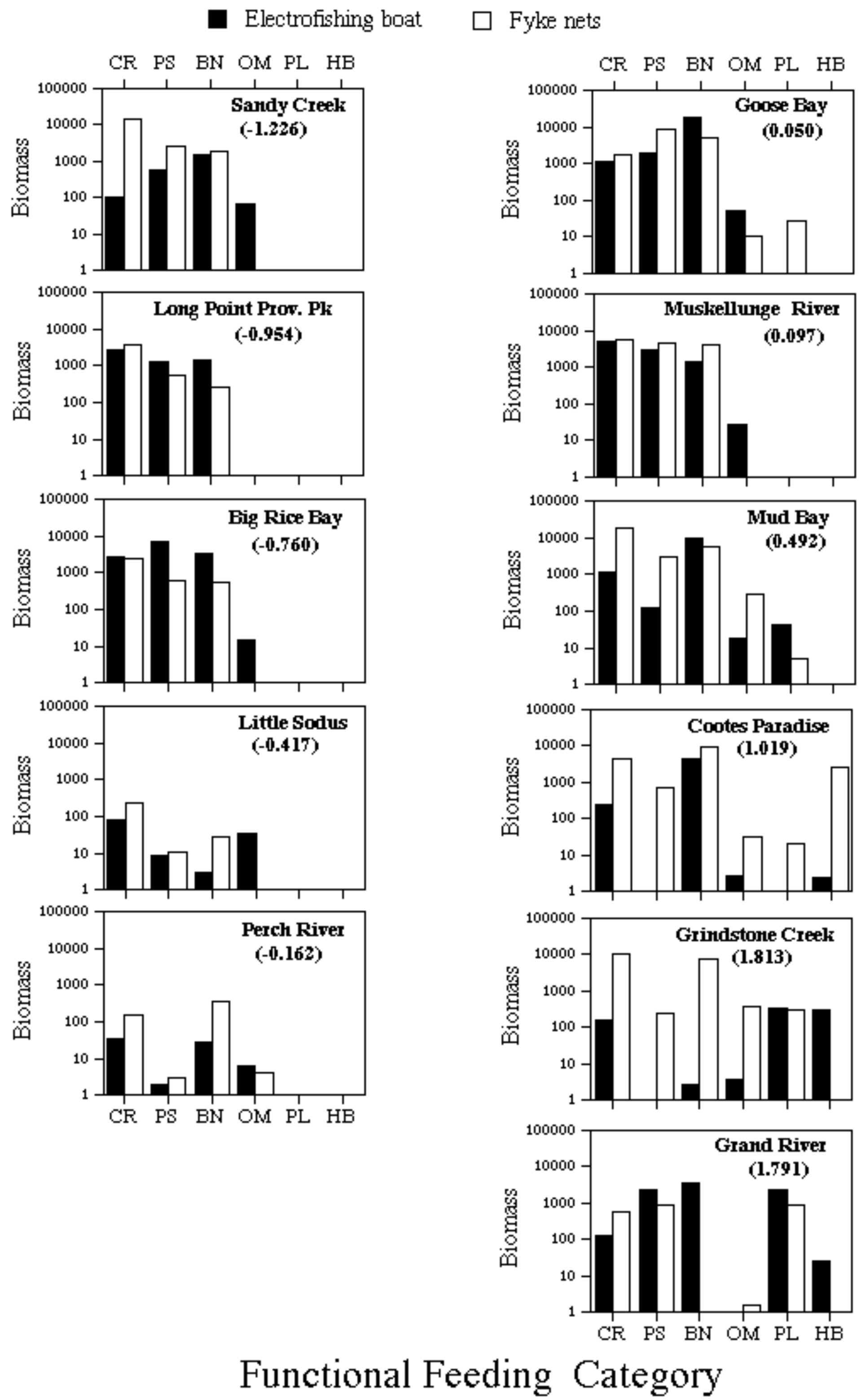

Electrofishing boat
Fyke nets
CR PS BN OM PL HB
# of fish
Sandy Creek (-1.226)
Long Point Prov. Pk (-0.954)
Big Rice Bay (-0.760)
Little Sodus (-0.417)
Perch River (-0.162)
Goose Bay (0.050)
Muskellunge River (0.097)
Mud Bay (0.492)
Cootes Paradise (1.019)
Grand River (1.791)
Grindstone Creek (1.813)
Functional Feeding Category

Fig. 6



Electrofishing boat
Fyke nets
CR PS BN OM PL HB
Biomass
Sandy Creek (-1.226)
Long Point Prov. Pk (-0.954)
Big Rice Bay (-0.760)
Little Sodus (-0.417)
Perch River (-0.162)
Goose Bay (0.050)
Muskellunge River (0.097)
Mud Bay (0.492)
Cootes Paradise (1.019)
Grindstone Creek (1.813)
Grand River (1.791)
Functional Feeding Category

Fig. 7

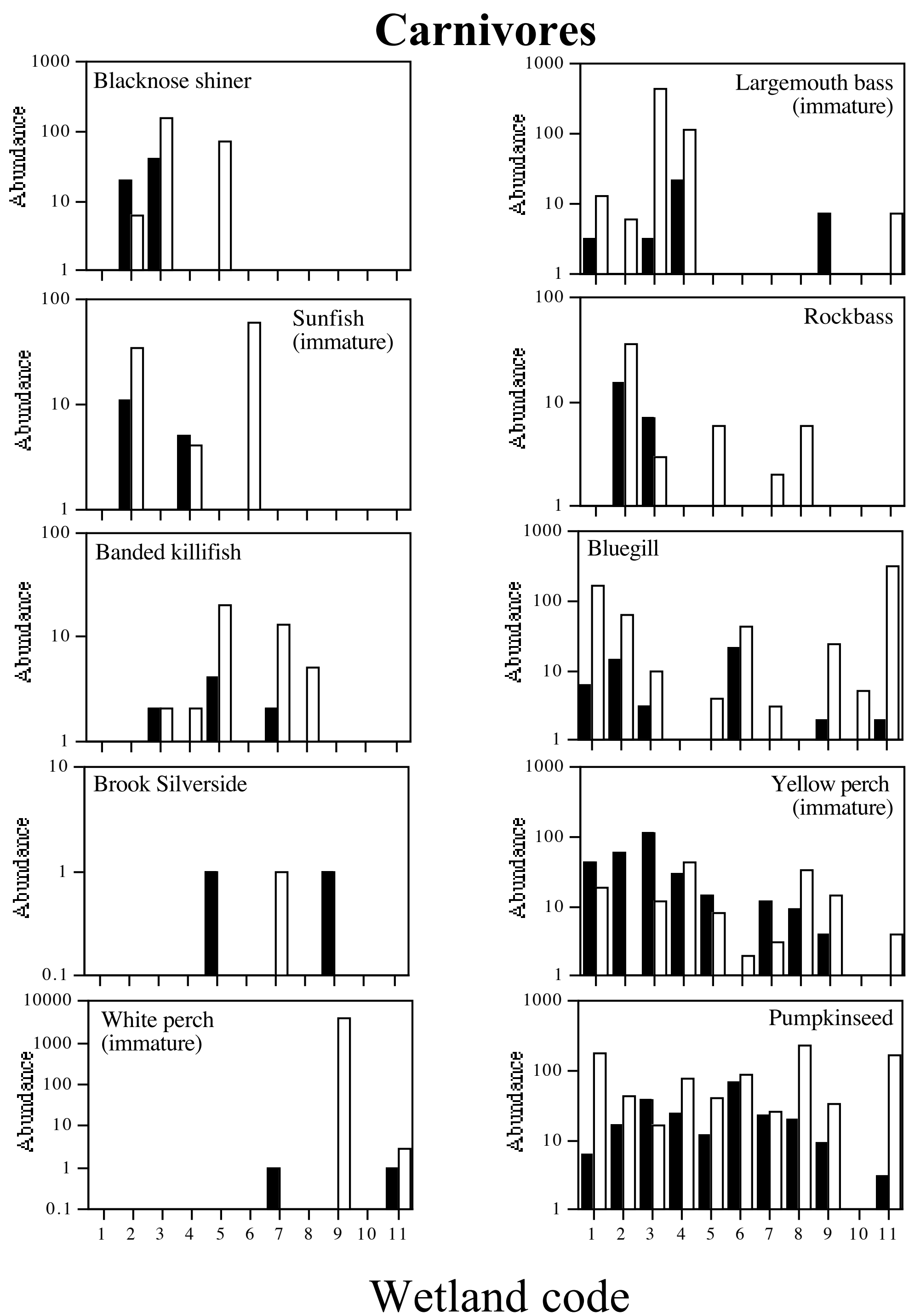

Carnivores
Blacknose shiner
Largemouth bass (immature)
Sunfish (immature)
Rockbass
Banded killifish
Bluegill
Brook Silverside
Yellow perch (immature)
White perch (immature)
Pumpkinseed
Abundance
1 2 3 4 5 6 7 8 9 10 11
Wetland code

Fig. 8

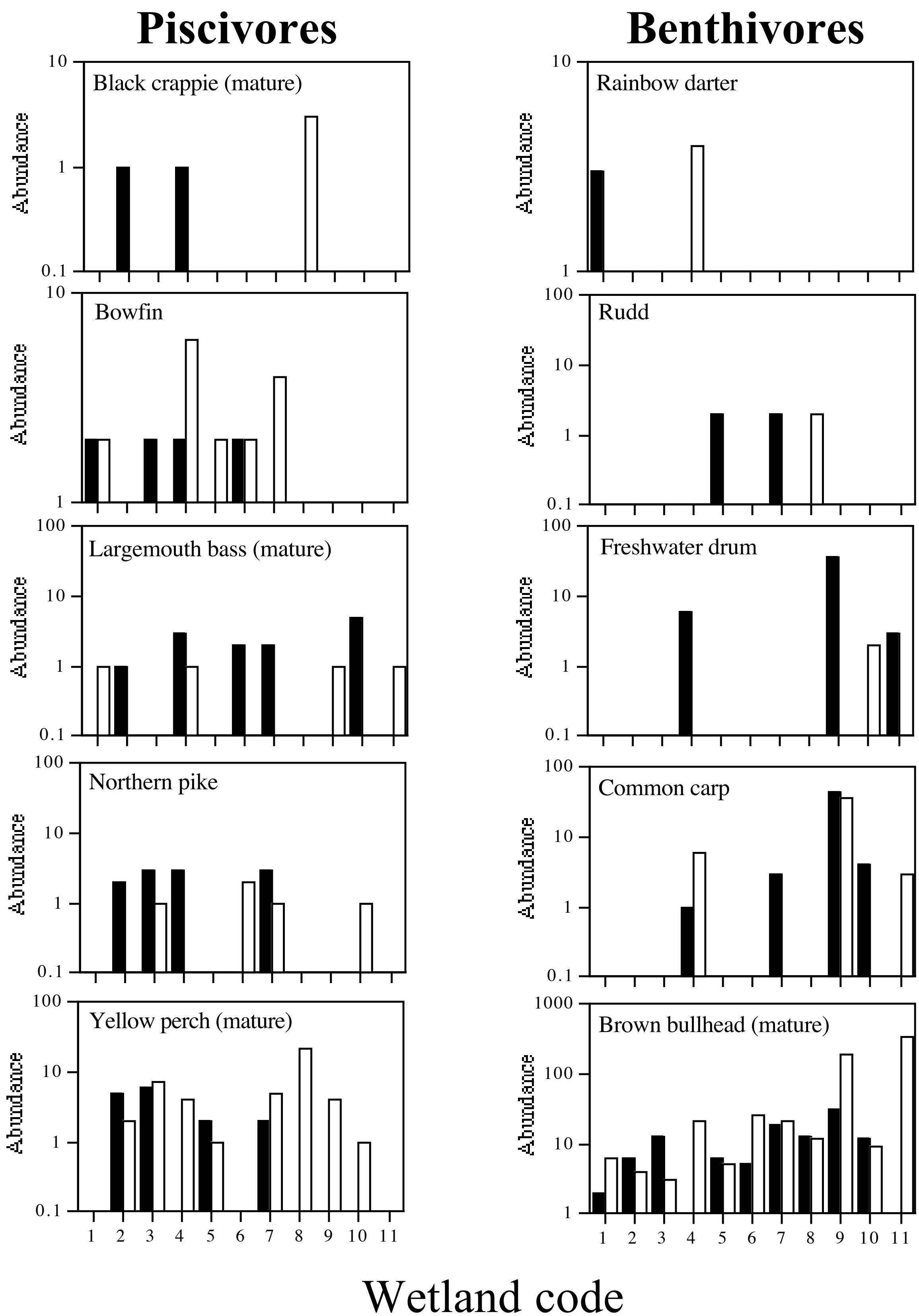
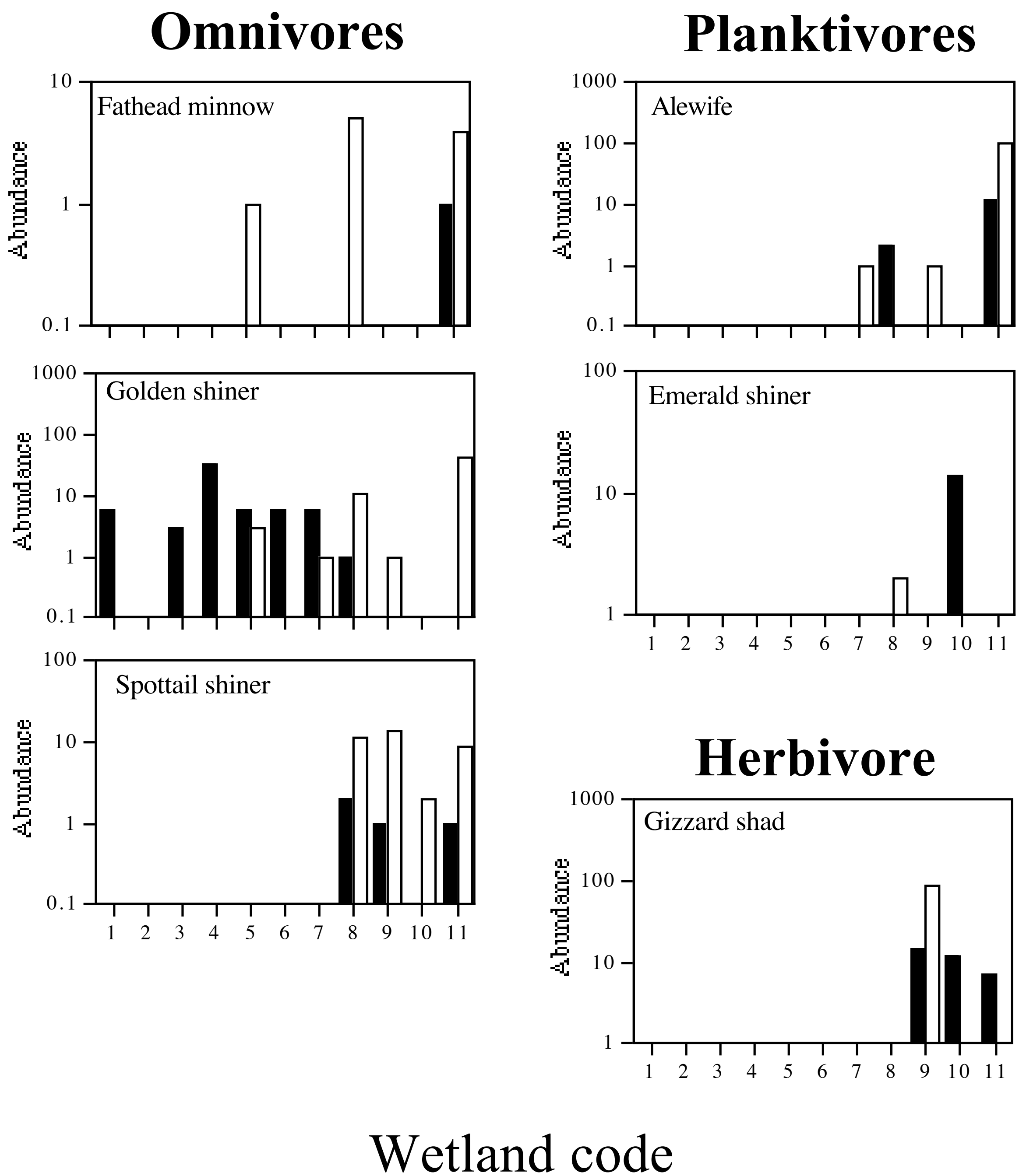

Piscivores
Benthivores
Black crappie (mature)
Rainbow darter
Bowfin
Rudd
Largemouth bass (mature)
Freshwater drum
Northern pike
Common carp
Yellow perch (mature)
Brown bullhead (mature)
Abundance
Wetland code

Fig. 9


Omnivores
Planktivores
Fathead minnow
Golden shiner
Spottail shiner
Alewife
Emerald shiner
Herbivore
Gizzard shad
Abundance
1000
100
10
1
0.1
1 2 3 4 5 6 7 8 9 10 11
Wetland code