GREAT LAKES FISHERY COMMISSION

2003 Project Completion Report[1]

Ecosystem-based assessment of fish habitat in coastal wetlands of the Great Lakes

by:

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Completion Report
September 30, 2003

PROJECT TITLE:

Ecosystem-based assessment of fish habitat in coastal wetlands of the Great Lakes

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* Their results were reported in detail in September 2002, and are only summarized here.

This proposal has two main objectives:

1) We will assess the type and quality of fish habitat in 40 Great Lakes coastal wetlands using direct gradient analyses, which will statistically relate fish distribution to a set of key physical, chemical and biological variables. These results will be used to identify variables to be entered into multivariate regression models to predict habitat for appropriate fish assemblages based on environmental attributes. For a subset of these wetlands (20-25), we will sample the benthic invertebrate and zooplankton communities to assess the properties of lower trophic levels for indicating changes in habitat quality resulting from anthropogenic disturbance (e.g. conversion of forested land for urban and agricultural development in the wetland watershed, evidence of urban encroachment, proximity to roadways, etc.). We will then validate the accuracy and utility of the multivariate models using data from an independent set of 10 wetlands.

2) We plan to conduct a pilot study to determine if key habitat features (e.g. macrophyte cover, substrate type, water clarity) can be monitored remotely (through airborne or spaceborne multispectral imagery) to provide an automated and comprehensive means of tracking habitat loss at large spatial scales.

RATIONALE AND RELEVANCE:

This proposal responds to the Great Lakes Fishery Commission's RFP to develop a consistent method of fish habitat classification and inventory that can be readily utilized by environmental management agencies to track human-induced habitat losses across the basin. It will provide a statistical basis for classifying fish habitat using biophysical and chemical variables, some of which may be obtained remotely. This proposal will complement ongoing efforts (e.g. Wetland Inventory for Research and Education Network (WIRE Net) to develop an internet-accessible, Geographic-Information-Systems-(GIS)-based, binational inventory of coastal wetlands for the Great Lakes basin, a goal that will have overarching benefits for the GLFC and many other environmental management agencies.

PROCEDURES:

1. We will conduct an intensive field survey of 50 wetlands over three sampling seasons beginning the summer of 2000. These wetlands will be selected from established "eco-reaches" (distinct shoreline segments delineated by Chow-Fraser and Albert (1999) for the 1998 State of the Lake Conference) of all five Great Lakes. For each wetland, we will measure a set of variables that are known to be important abiotic and biotic forcing factors in the distribution of fish communities in coastal wetlands and shallow lakes (Randall et al. 1993; Romo et al. 1996; Brazner and Beals 1997; Chow-Fraser 1998; Chow-Fraser et al. 1998). We will use appropriate gear to sample juvenile and adult fish and determine the presence/absence and abundance (where possible) of dominant fish taxa for each wetland.

2. We will use a direct gradient analysis pioneered by ter Braak (1986) to relate community composition of fish taxa to variation in the biophysico-chemical environment for 40 of these wetlands. This technique is an extension of correspondence analysis (reciprocal averaging), in which ordination axes are extracted as linear combinations of environmental variables. Thus, variation in the fish assemblages can be directly related to environmental variables which may be quantitative or nominal. Similar analyses will be carried out for zooplankton and macroinvertebrate communities in a subset of 20-25 wetlands; these lower trophic levels have been shown to be good indicators of anthropogenic disturbance in natural ecosystems (e.g. Attayde and Bozelli 1998; Burton et al. 1999). Multivariate models of habitat for appropriate fish assemblages will be constructed using multiple linear regression analysis. These models will be validated with data from an independent dataset of 10 additional wetlands.

3. For a limited number of wetlands, a pilot study will be conducted to determine if macrophyte cover, water clarity, and substrate type can be monitored remotely.

DELIVERABLES:

1. We will provide a database of 50 coastal wetlands located in representative eco-reaches of all five Great Lakes that will contain information on species list and relative abundances of dominant fish along with a suite of above-named biophyisco-chemical variables (all sampled and processed with standardized techniques to make them directly comparable). We will develop predictive models of habitat for appropriate fish assemblages using the suite of environmental variables from 40 of these wetlands and validate the models with data from the 10 remaining. For up to 25 of these wetlands, we will also collect information on the communities of plant-associated zooplankton and macroinvertebrate to assess their properties for indicating habitat quality.

2. We will develop a pilot program combining field sampling with remote-sensing technology to evaluate the potential of using remotely sensed data to determine subtle changes in the floristic community, substrate type and water clarity of a few wetlands over several-year intervals.

3. Three students will obtain post-graduate degrees in an *interdisciplinary* environment involving field sampling of all five Great Lakes, and the use of remote-sensing, an increasingly important ecological tool for ecosystem research.

LIST OF DELIVERABLES REPORTED

1. *We will sample 50 costal wetlands from representative "eco-reach" segments for a suite of biological, physical, and chemical components, along with a list of fish species (adults and juveniles) and relative abundances of the dominant taxa*.

METHODS

We conducted a field survey of 60 wetlands over three sampling seasons beginning the summer of 2000; four wetlands were visited in two consecutive seasons for a total of 64 wetland-years (Appendix 1; Fig. 1). These wetlands were selected from established "eco-reaches" along the shoreline of all five Great Lakes (after Chow-Fraser and Albert (1999); Wei et al. 2003).

Comparison between fyke nets and boat electrofishing

A subset of eleven wetlands were used to conduct a gear-comparison study in which data collected with fyke nets were compared with data collected by electrofishing boat at the same time and in the same locations. This study was funded through another grant but we present the data here because of its relevance to interpreting data from this study, and we have therefore included the manuscript as an appendix (Appendix 2). We compared sampling biases associated with these two methods because both methods have been and continue to be used to survey fish communities in coastal wetlands of the Great Lakes. Hence, it was important to determine possible biases that may exist in order to make data comparable across studies.

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) although on average, FN caught 75% whereas EB only captured 68% of the species encountered in respective wetlands.

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.

Justification and choice of fishing gear and protocol

The results of the above comparison clearly indicated that neither method alone could fully represent the entire community of fish, and that the extent of bias depended on wetland quality. However, given that there is neither sufficient time nor money to carry out a survey with both methods, we had to decide to use one or the other method to develop the model. We chose to use fyke nets to sample all of the wetlands for two reasons. First, there was no electrofishing boat available for this project and we would not have been able to complete the sampling of 50+ wetlands within three sampling seasons were we to depend solely on the assistance of the different agencies. Secondly, there was no agreement among agencies as to the best protocol to standardize sampling effort across the wide range of conditions encountered in this study. The catch per unit effort varied depending on knowledge and expertise of the operator from each agency, as well as type of shocking unit used. While one agency used total shock seconds to standardize the sampling (OMNR; limiting to 1000 shock seconds), another standardized by the minimum amount of transect area covered (RBG; 100-m^2 transect (50 m x 2 m)), and the third sampled representative habitats within wetlands, and the extent of this area varied depending on wetland quality (UFWS). These different approaches highlight the current lack of standardization in the use of boat electrofishing boat for sampling coastal wetlands, and the need for a definitive study to determine the most appropriate protocol. On the other hand, use of fyke nets had already been standardized (Brazner and Beals 1997; Brazner et al. 2000) and we were able to apply this protocol to sampling all wetlands with a standardized fishing period of 24 h. We consulted J. Brazner (USEPA, Duluth, Mn) throughout the planning and design stages of this project to ensure that we collected data that could be used in cross-study comparison. We even ensured that the nets used in this study came from the same manufacturer as that used by EPA.

Field Sampling with fyke nets

All fish data used to develop and validate the model were collected with three pairs of fykenets, two pairs of large (13-mm and 4-mm bar mesh, 5-m length, 1.1 X 1.4-front opening) and one pair of small (4-mm bar mesh, 2.1-m length, 0.5 X 1.0-m front opening) nets, that were set parallel to the emergent zone at the 1-m and 0.5-m depth contour, respectively. The paired nets were positioned face-to-face with a 10-m lead connecting them, while 3-m wings were set off the front openings at a 45°angle. To the extent possible, all nets were set within submergent vegetation, but when there were too little vegetation or when appropriate depths were not available, the nets were set near the emergent vegetation. After twenty-four hours, fish present in the nets were sized, enumerated and identified according to Scott and Crossman (1998). Unknown fish (i.e. small minnows) were placed in bags, kept frozen, and identified later in the lab. All other fish were released live. The summary of fish found in the 60 wetlands (64 wetland years) is included in Table 1. Abundance refers to the total catch of fish recovered in all 6 fyke nets in each wetland.

Measurement of habitat variables

For each wetland, we also measured a set of important variables for fish habitat, which included water-quality parameters, substrate type, and other physical attributes. Wetlands were visited between early-June and late August during 2000-2002 inclusive. Water samples were collected from an open water site located at least 10 m from the edge of the aquatic vegetation for analysis of planktonic algae, primary nutrients and suspended solids; in certain wetlands, submergent vegetation was present throughout and in those cases, we sampled in the deeper areas that had very little submergent vegetation. This ensured that we would not contaminate the samples with benthic algae (either epiphytic or periphytic). Water depth at the open water site varied between 20 cm at the shallowest marshes to 4.1 m (mean of 1.28 m) because of variation among sites and differences in water level among years. Water samples were collected with a 1-L Van Dorn bottle deployed at mid-depth of the wetland, and dispensed into clean Nalgene bottles (acid-washed and rinsed with deionized water) for nutrient analyses. Samples intended for chlorophyll analyses were placed in opaque Nalgene bottles that were simply cleaned and rinsed with deionized water. All sample bottles were kept in coolers and processed in the field or in the laboratory within 6 hours of collection, after which sample jars were kept frozen in a freezer until analysis (usually within 3 months of collection).

Temperature (TEMP), conductivity (COND) and dissolved oxygen (DO) and turbidity (TURB) were measured with a Hydrolab Minisonde multi-parameter probe attached to a Surveyor display (Hydrolab, Austin, Texas, USA) during 2000-2001 whereas a YSI 6600 multi-parameter probe with two optical sensors (turbidity and chlorophyll) and a YSI 650 display (YSI, Yellow Springs, Ohio, USA) were used in 2002. At the end of the 2001 field season, we conducted a side-by-side comparison of both the Hydrolab Minisonde and YSI 6600, and found no significant deviations with respect to any of the parameters, except for turbidity. We therefore chose to use the YSI 6600 for routine monitoring in 2002 because it was much easier to use. However, to permit valid comparisons across years, turbidity data collected by these probes were not used in statistical analyses. Instead, we used the Hach 2100 Portalab to measure turbidities in triplicate from water samples collected with the Van Dorn sampler (see above). All sites were georeferenced with a Gramin GPS unit (4- to 6-m accuracy), which was attached to the YSI 650 display.

Laboratory processing and analyses

Samples for chlorophyll-a content of phytoplankton were first filtered through 0.45-µm GF/C filters, then stored frozen in tin foil until analysis. At the time of analysis, frozen filters were unwrapped and placed in 10 mL of 90% reagent-grade acetone for 12-24 h (APHA 1992).

Samples were centrifuged, and chlorophyll-a content was determined by measuring absorbance with a Milton Roy 301 spectrophotometer before and after acidification (to account for phaeophytin pigments). Chlorophyll samples reported in this study were all measured in triplicate. Following digestion in potassium persulfate in an autoclave, samples for total phosphorus (TP) were measured in triplicate according to the molybdenum blue method of Murphy and Riley (1962). Samples for soluble reactive phosphorus (SRP) were first passed through 0.45 µm-filters before molybdenum blue analysis, without digestion. Total Kjeldahl nitrogen (TKN), total nitrate nitrogen (TNN) and total-ammonia nitrogen (TAN) were measured with Hach protocols and reagents (Hach Company 1989) using a Hach DR2000 spectrophotometer (Hach, Loveland, Colorado, U.S.A.). Total nitrogen (TN) was calculated by addition of TKN and TNN.

Water samples for total suspended solids (TSS) determination were filtered through pre-weighed GF/C filters and frozen until processing. Filters were first dried at 100 °C for 1 h, dried in a dessicator with calcium sulphate for another hour, and then weighed to determine TSS. Loss on ignition was determined after combustion at 550 °C for 20 min followed by drying in the dessicator for an hour. Weight of the combusted filter was assumed to be total inorganic suspended solids (TISS), whereas difference in the weight of the filter before and after combustion was total organic suspended solids (TOSS).

2. *We will develop predictive models of habitat for appropriate fish assemblages using the suite of environmental variables from 40 of these wetlands and validate the models with data from the 10 remaining. For up to 25 of these wetlands, we will also collect information on the communities of plant-associated zooplankton and macroinvertebrate to assess their properties for indicating habitat quality.*

As stated in the procedures of our original proposal, we used a direct gradient analysis pioneered by ter Braak (1986) to relate community composition of fish taxa to variation in the biophysico-chemical environment for 40 of these wetlands (43 wetland years). CANOCO 4.5 (ter Braak and Smilauer 1998) was used to run the Canonical Correspondence Analysis (CCA). The fish species and wetlands were ordinated by creating synthetic axes that best fit the gradients of the environmental data (ter Braak and Verdonschot 1995). Prior to conducting the CCA, we used the Detrended Correspondence Analysis (DCA) to verify that the species were in fact distributed unimodally over the environmental gradient (i.e. length of the gradient was greater than 4.0). All environmental variables and species abundances (i.e. total fish caught in the 3-paired fyke nets during the 24-h incubation) were $\log_{10}$-transformed and standardized to have zero mean and unit variance. Rare species (i.e. those occurring in only one wetland) were excluded from this analysis. We used Monte Carlo permutations (499 random permutations; ter Braak and Smilauer 1998) to determine the significance of the canonical axes, and analyzed presence-absence (PA) and abundance (AB) data separately. Shannon-Wiener diversity and Simpson's Evenness values were calculated with MVSP (version 3.1, Kovach Computing Services, Ithaca, New York, USA).

To develop the WFI, each species was assigned U and T values according to the following formula (Kelly and Whitton 1995, Lougheed and Chow-Fraser 2002):

$$WFI = \frac{\sum_{i=1}^{n} Y_i T_i U_i}{\sum_{i=1}^{n} Y_i T_i}$$

where: Y_i =the presence or $\log_{10}$abundance of species *i*
T_i =value from 1-3, indicating niche breadth
U_i =value from 1-5, indicating tolerance of degradation

Development of Wetland Fish Index

In this project, fish and environmental data were collected from 60 wetlands located throughout the shoreline of all five Great Lakes (Fig. 1). Because of the large latitudinal gradient in our dataset, there were large differences in degree-days and wetland geomorphology that could confound any effect of human disturbance on the quality of fish habitat in wetlands. To reduce the effect of the large latitudinal gradient, we used 45° North Latitude to arbitrarily divide the dataset into two groups. One group contained 43 wetland-years and consisted of sites that occurred primarily along the shoreline of Lakes Ontario, Lake Erie, Niagara River, Lake Michigan and Lake Huron; the other group contained 21 wetland-years, which were located in Lakes Superior, Huron and northern Lake Michigan. The group of southern wetlands is expected to experience higher development pressure compared with the northern group, and this effect should not be confounded by differences in bedrock geology and climate since they share many similarities in this regard (Fig. 2; after Wei 2002). We will use the larger group to develop the index, and will validate the index with data from the smaller dataset.

In total, 80 fish taxa were identified and enumerated in this study (Table 1). Of the 15 environmental variables that were initially entered into the CCA, six (SRP, TNN, TKN, TN, TSS, pH) were eliminated because they were redundant (according to value of the inflation factor). The final list of variables retained in the CCA were: TP, TAN, TISS, TOSS, CHL, TEMP, COND, DO, and longitude. The **U** and **T** values (see Methods) were assigned based on patterns that emerged from the CCA, which related the statistical mode for each species along the synthetic axes (Fig. 3 a to d). The first axis explained most of the variation in the species abundance data (26.3%; P=0.002) and was significantly correlated with several environmental vectors: COND (r=0.7693), TAN (r=0.6870), TP (r=0.6141), CHL (r=0.6040), and TISS (r=0.5234) (Fig. 3a). CCA axis 2 explained another 21.3%, and was highly correlated with the longitude of the wetland (LONG; r=-0.8914). This second axis primarily ordinated wetlands from west (positive) to east (negative end), and is likely a reflection of lake-to-lake differences in wetland development.

Both types of fish data (i.e. abundance and presence/absence; see Table 1 for species codes) were ordinated according to their statistical mode along synthetic axis 1 (Fig 3c and d, respectively). Those species associated with positive values tended to be very tolerant of degraded conditions, examples of which are white bass *Morone americana* (*MOCH*) and white crappie *Pomoxis annularis* (*POAN*)), whereas species associated with negative values were intolerant of pollution and tended to have a preference for environments with low nutrients, TSS, COND, and Chl, examples of which are longnose gar *Lepisosteus osseeus* (*LEOS*) and the banded killifish *Fundulus diaphanus* (*FUDI*). Although much of the variation in fish distribution was due to species-specific tolerance of habitat degradation (i.e. the 1st axis), some variation could be attributed to the location of wetlands (i.e. 2nd axis). For example, species such as johnny darter *Etheostoma nigrum* (*ETNI*) and grass pickerel *Esox americanus vermiculatus* (*ESAV*) which were associated with the positive end of CCA 2, more commonly associated with the wetlands of Lake Michigan.

Because the first axis appeared to ordinate wetlands according to degree of water-quality degradation, we used CCA axis 1 to derive values for "**U**" and "**T**". A centroid is the center of a cluster of species scores, and we used the placement of this centroid along the synthetic degradation axis to indicate the species' "**U**" value. Thus, both PECA and HYHA were each given a "**U**" value of 1 because their centroids were located near the positive end of axis 1 (Fig. 3c and d), which indicates they are very tolerant of degraded conditions (Table 2). By contrast, a species such as blackchin shiner *Notropis heterodon* (*NOHN*) and the longnose gar (*LEOS*) were each given a "**U**" value of 5 because their centroids were placed near the negative end of the 1st axis, which indicates they are very intolerant of environmental degradation. All of the remaining species were assigned intermediate values (2, 3 or 4) according to the position of the species centroids along CCA axis 1. The weighted standard deviations were assumed to indicate niche breadth and were used to assign the "**T**" values. These values ranged from 1 to 3, where 1 indicated a wide niche, 2 an intermediate niche, and 3 a narrow niche. Species occurring in ≤ 5% of total wetlands were given a "**T**" value of 1 since we do not know if the species truly has a narrow niche or if the fishing method we used was ineffective in sampling the species in question. All species were assigned "**U**" and "**T**" values using the above approach and are presented separately for abundance and presence-absence data in Table 2.

Comparison of the WFI with other indices of habitat quality

To evaluate the accuracy of the WFI for predicting habitat quality, we compared them to corresponding scores of the Wetland Quality Index (WQI), an independently derived index based on water-quality data collected from 110 wetlands (146 wetland-years) located throughout the five Great Lakes (Chow-Fraser 2003). This index ranges from –3, which is indicative of the most impacted conditions to +3, which is indicative of the most undisturbed sites. WFI scores for the 43 wetland-years were plotted against corresponding WQI scores for both the abundance and presence-absence data (Fig. 4a and b, respectively; open symbols only). Both sets of data yielded highly significant regression equations, although more of the variation was explained for the WFI (PA) than for the WFI (AB) data (r^2=0.54 versus r^2=0.40, respectively). The fact that most of the Lake Michigan data (open triangles) were clustered above the best-fit line in both regression models suggest that a better model may be developed for Lake Michigan data and this should be pursued.

We also compared the relationship between other indices and WQI scores. For this comparison, we had to exclude data from 8 wetlands in Lake Michigan that did not have appropriate data for calculating all the indices under consideration, leaving us with only 35 wetland-years that correspond to Lakes Ontario, Erie and the Niagara River. Both the WFI (AB) and WFI (PA) were significantly related to WQI scores and removal of the Michigan data improved the associated r^2-values, from 0.40 to 0.72 (Fig. 5a) and from 0.54 to 0.66 (Fig. 5b), respectively. By comparison, we found no significant relationship between WQI and the Shannon-Wiener diversity index (Fig. 5c; P=0.56) nor Simpson's Evenness index (Fig. 5d; P=0.72). We also calculated IBI values using Minns et al.'s (1994) 12 metrics, which included numbers of native species and native cyprinids, percent biomass of trophic structure, and abundance of non-native species. The total site scores could range from 0 (lowest quality sites) to 120 (highest quality sites). Our calculated IBI scores ranged from 44.59 to 106.23 with a mean of 86.48, but when we regressed these against corresponding WQI scores, we found no significant relationship (Fig. 5f; P=0.24). On the other hand, we found a significant negative relationship between species richness and WQI scores (Fig 5e; P=0.0201), which if taken at face value, would mean that the less disturbed wetlands have fewer species in the fish community, and this is difficult to explain.

Validation of the WFI

To validate the WFI for assessing habitat quality of wetlands in the upper lakes, we plotted WFI scores against corresponding WQI scores for the 23 sites located north of the 45ºN latitude, that had not been included in the development of the WFI. To ease comparison, we superimposed the data for validation (closed symbols and asterisk) directly over those used for developing the model in Fig 4 (open symbols). The Lake Huron data (closed inverse triangle) fell close to the best-fit line, suggesting that the index could be applied to Lake Huron without modification. Data for Lake Superior, however, had to be analyzed separately by country of origin; most of the U.S. wetlands (closed circles) were located close to the best-fit line, but those for the Canadian wetlands were located well below the regression line, indicating that a separate model should probably be developed for these wetlands.

We conducted a further examination by comparing the observed WFI scores (calculated with U and T values from Table 2), against predicted WFI scores derived from the regression between WFI and WQI for the model sites (Fig 6a and b). This comparison points out clearly that based on water-quality information alone, WFI scores are higher than expected for all Michigan sites, and this is particularly noticeable for WFI (AB) scores (Fig 6a). Of the five Canadian sites located on the Lake Superior shoreline, four of these had observed scores that were extremely low compared with predicted scores based on corresponding WQI. It was interesting that the data point for Chippewa Park ("CH" in Fig. 6a), a relatively degraded urban wetland was closest to the line of unity. On the other hand, Cloud Bay ("CB" in Fig 6a and b), a relatively undisturbed wetland by visual inspection, showed the largest deviation from expected for both WFI (AB) and WFI (PA) scores.

Discussion of the WFI model

The WFI was developed based on quantifiable relationships between fish distribution and associated water quality conditions for 40 coastal wetlands located primarily in the lower Great Lakes. According to Chow-Fraser's (2003) WQI scores calculated for these sites, wetlands in this study included sites that were highly degraded due to nutrient and sediment enrichment, as well as those that were excellent quality and relatively undisturbed by human impact. When compared with other diversity indices such as Shannon-Wiener's H', Simpson's evenness, species richness, and Minns et al.'s (1994) IBI for coastal wetlands, only the WFI (calculated with either the presence-absence or abundance data) was found to vary significantly with WQI (Fig. 5a and b, respectively), demonstrating that the WFI is more useful than the others as an indicator of anthropogenic impact. Species diversity and richness are probably insensitive to degradation, because turnover in the fish community resulting from habitat degradation do not necessarily lead to changes in species diversity or richness (e.g. Chow-Fraser et al. 1998). As well, a large number of tolerant species could result in a higher diversity score for degraded sites, and this may explain why species richness was negatively correlated with WQI score (Fig. 5e). We were surprised that the coastal IBI developed by Minns et al. (1994) was insensitive to changes in wetland quality as indicated by WQI, and we speculate that the reason for this may be the limited number of wetlands (n=5) and range of habitat types used to develop the coastal IBI.

When we generated WFI scores for the independent dataset that included wetlands located in the upper Great Lakes, we found good agreement for wetlands of Lake Huron and U.S. wetlands of Lake Superior. However, there appeared to be unacceptable departure for many of the Canadian wetlands of Lake Superior (asterisks in Fig 4 a and b). On close examination, we found that on average, greater than 25% of the species from the Lake Superior wetlands had been excluded from Table 2, because they do not generally occur south of the 45° N latitude (Fig. 7). Since there are a large number of species that only occur in the lower lakes (whose absence in the upper lakes do not necessarily reflect patterns of human disturbance), their absence in a Lake Superior wetland would result in a reduced WFI score, and this would lead to the erroneous conclusion that the wetland in question was degraded when it may actually be very high quality. A case in point is Cloud Bay (CB in Fig. 6), a wetland that ranks as extremely high quality in other respects (i.e. WQI, Chow-Fraser 2003; periphyton biomass, McNair and Chow-Fraser 2003), and which is also known to support a diverse warm-water fishery (Chow-Fraser, unpub. data). However, because only 9 of the 21 species present in the wetland has been indexed, the associated WFI score (based either on abundance or presence-absence data) is extremely low. Therefore, a separate index should probably be created for these Upper-lake wetlands, to more adequately reflect differences in fish assemblages.

That wetland degradation leads to changes in the fish community has already been documented for degraded wetlands such as Cootes Paradise Marsh (Chow-Fraser et al. 1998), and forms a scientifically defensible basis for formulation of the WFI. Degradation causes declines in plant diversity and biomass, which negatively affects the fish species. Young fish find shelter from larger piscivorous fish in the macrophytes. Macrophytes also provide habitat for invertebrates, which are then eaten by the fish. Turbidity also leads to decreased macrophyte abundance and can affect the fish by making feeding more difficult and time-consuming (Jude and Pappas 1992). Once the habitat becomes degraded, exotic invaders such as common carp assert their dominance, promoting higher turbidity and causing further deterioration in the macrophyte community (Chow-Fraser 1999). The WFI is effective in linking this shift from species that are intolerant of pollution to more tolerant species across the degradation gradient.

Although the **"U"** and "**T**" values assigned to the species were based on statistical relationships between fish occurrence and water quality, they are consistent with what is known about the ecology of the species. For instance, two species identified as highly intolerant of degradation (U=5; Table 2) are the longnose gar (*LEOS*) and blackchin shiner (*NOHN*). The longnose gar is a species that is heavily dependent on aquatic vegetation throughout its life cycle: during spawning, at the larval stage, and as an adult (Scott and Crossman 1998). Similarly, the blackchin shiner prefers cool, clear waters and quickly disappears when turbidity increases and the amount of submerged vegetation decreases (Trautman 1981). By comparison, we classified both yellow perch *Perca flavescens* (*PEFL*) and brown bullhead *Ameiurus nebulosus* (AMNE) as moderately tolerant of degradation (U=3; Table 2). The fact that both species were distributed widely within our database (present in over 60% of the wetlands; see Table 1), fits with the conventional wisdom that these species can spawn in a variety of habitat types, but prefer moderate vegetation (Scott and Crossman 1998). In contrast, white perch have been found in eutrophic areas of Lake Champlain, where there is elevated turbidity and chlorophyll (Hawes and Parrish 2003), while white crappie are also known to be turbidity tolerant (more so than black crappie; Scott and Crossman 1998); these documented observations therefore support our assigning these latter species a low "**U**" value of 1 (Table 2).

One advantage of the WFI over the WZI and WQI is that it can provide more rapid assessments. Although more specialized and sometimes expensive equipment is required to assess the fish community, virtually all the data can be processed immediately in the field with moderate training. By comparison, both the zooplankton and water-quality samples require specialized equipment, as well as lab processing, analyses and technical expertise, that can add months to the assessment process.

The use of presence-absence or abundance data produced very similar results. However, the WFI (PA) may have more widespread applicability because it is not gear-dependent. Chow-Fraser et al. (2003) consistently caught more fish with fyke nets than with boat electrofishing across a large gradient of wetland quality. Where different sampling gear are involved in fish collection, the data may have to be standardized prior to application of the WFI (AB). Another advantage of the WFI (PA) is that the index may be used with historic species lists that do not have accompanying fish abundances. It is also relatively stable through the season, since a few spawning adults and many juveniles will lead to the same score, and therefore restrictions on time of sampling may be relaxed. Presence-absence data also seem to produce a better relationship between WFI and WQI, since the associated r^2-value is higher for WFI (PA) (r^2=0.54) than for WFI (AB) (r^2=0.40) (Figure 4b and a, respectively). In general, observed WFI (PA) scores (Fig. 6) were more consistent with predicted values compared with corresponding data for WFI (AB), suggesting that the former is not affected by either regional or seasonal differences in the fish data.

Chow-Fraser et al. (2003; Appendix 2) found that there were sampling biases associated with different fish survey methods. Since we used only one method (24-h fykenets, FN) to derive the WFI, we wanted to see if the index would change significantly if another common method, daytime boat electrofishing (EB), were used to collect the fish data. Data from the 11 wetlands used in Chow-Fraser et al.'s study were assembled for this comparison. We performed paired t-test on WFI scores calculated from data collected with each survey method. Mean WFI (PA) calculated from FN and EB were 3.03 ($\pm$ 0.174 S.E.) and 3.00 ($\pm$ 0.169 S.E.), respectively, with a mean difference of 0.035, but these differences were not significant for either the one-tailed or two-tailed tests (FN>EB, P=0.3799; FN<EB, P=0.6201; FN≠EB, P=0.7599). Similarly, when WFI (AB) were compared for FN and EB data, mean scores were 3.18 ($\pm$ 0.215 SE) and 3.13 ($\pm$ 0.186 SE), respectively, with a mean difference of 0.048, and

again there were no significant differences for either the one-tailed or two-tailed tests (FN>EB, P=0.3504; FN<EB, P=0.6469; FN≠EB, P=0.7007). Therefore, even though sampling biases are statistically associated with the different fish survey methods, the WFI is robust and insensitive to differences in collection methods, and this is very important condition for widespread adoption.

The WFI needs to be tested on other systems to determine if it can be applied to other types of wetlands outside of the Great Lakes basin. Although we do not see any problems in applying WFI to coastal wetlands located south of the 45° N latitude, we caution against applying this to wetlands north of the boundary, recognizing the real possibility that a more appropriate index may developed for wetlands of the upper lakes in the future.

Development of indicators using zooplankton

We have developed and published a similar Wetland Zooplankton Index using data collected in this study together with existing data from a previous project (Lougheed and Chow-Fraser 2002; Appendix 3). We developed a wetland zooplankton index (WZI) based on water-quality and zooplankton associations with aquatic vegetation (emergent, submergent and floating-leaf) that could be used to assess wetland quality throughout the basin of the Laurentian Great Lakes. Seventy coastal and inland marshes were sampled between 1995 and 2000; these ranged from pristine, macrophyte-dominated systems, to highly degraded systems containing only a fringe of emergent vegetation. The index was developed based on the results of a partial canonical correspondence analysis (pCCA), which indicated that plant-associated taxa such as chydorid and macrothricid cladocerans were common in high-quality wetlands while more open-water, pollution-tolerant taxa (e.g., *Brachionus*, *Moina*) dominated degraded wetlands. The WZI was found to be more useful than indices of diversity (H', species richness) and measures of community structure (mean cladoceran size, total abundance) for indicating wetland quality. Furthermore, an independent test of the WZI in a coastal wetland of the Great Lakes, Cootes Paradise Marsh, correctly detected moderate improvements in water quality following carp exclusion. Since wetlands used in this study covered a wide environmental and geographic range, the index should be broadly applicable to wetlands in the Laurentian Great Lakes basin, while further research is required to confirm its suitability in other regions and other vegetated habitats.

Development of indicators using macroinvertebrates

This study also investigated the development of a convenient indicator of wetland quality using the total dry weight of maacroinvertebrates sampled in a standardized manner using funnel traps. This is based on the observation that changes in water quality and habitat structure, as a result of wetland degradation, have cascading effects on the food web (Chow-Fraser 1998). Conditions that predominate in poor quality wetlands, such as poor water clarity, low oxygen and low macrophyte growth, have been proven to be harmful to piscivorous fish (Chow-Fraser et al. 1998). Piscivores, such as the northern pike *(Escox luscious)*, are primarily cool water fish that thrive in environments with clear water and large areas of macrophyte cover (Brönmark and Weisner 1992). Casselman and Lewis (1996) observed the decline of pike populations with increasing water temperatures and cultural eutrophication of wetlands in the Great Lakes Basin. By comparison, benthivorous fish, such as the common carp (*Cyprinus* *carpis)* and gizzard shad *(Dorosoma cepedianum)* are able to thrive in eutrophic conditions (Chow-Fraser et al. 1998) and their feeding and spawning behaviours contribute to further degradation of wetlands by destroying submergent vegetation and re-suspending nutrients and sediments (Crivelli 1983; Northcote 1988; Lougheed and Chow-Fraser 1998).

If zoobenthos (combination of the zooplankton and macroinvertebrates that reside at the sediment-water interface) are a primary food source for benthivore and planktivorous fish, then changes in fish assemblages due to wetland degradation should have a considerable impact on zoobenthic populations. The decline in piscivore populations evident in degraded wetlands should relax the predation-pressure on planktivores and benthivores, and the resulting increase in planktivore and benthivore densities should exert greater pressure on zooplankton and benthic macroinvertebrate communities. Therefore, a shift from a piscivore-dominated system to one dominated by benthivores and planktivores should result in a decline in the standing stock of zoobenthos (Brönmark and Weisner 1992; Saint-Jacques et al. 2000). We therefore hypothesized that zoobenthos biomass would increase as wetland quality improves, and that measuring zoobenthos biomass could be a time- and cost-effective indicator of wetland quality in the Great Lakes. To test this, we examined the relationship between zoobenthos biomass and WQI for a subset of the wetlands in the database.

Sampling for zoobenthos was carried out during the summers of 2001 and 2002 from 18 wetlands (19 wetland-years; see Table 3). They were collected with a set of funnel traps placed on top of the sediment in the vicinity where fyke nets were set in the wetlands. The funnel trap consisted of a fiberglass board with three attached funnels, each with a diameter of 19 cm (covering a surface of 0.028 m^2). Tubing connected a 620 ml bottle to each of the funnels. Two sets of funnel traps (n=6 bottles) were left in each wetland for 24 hours. After collection, samples were filtered through a 63-µm mesh filter in the field and preserved in a 10% sugar-formalin solution for further processing in the laboratory. A total of six zoobenthos samples were collected from each site.

In the laboratory, zoobenthos samples were transferred to an ethanol solution for processing. For each site, zoobenthos from four bottles were sorted from the debris with the aid of a binocular dissecting microscope. Two other bottles were set aside for later identification (not used in this study). In some of the good-quality wetlands, all six bottles were analyzed to reduce the sample variance because random occurrence of single large insect greatly influenced biomass values, compared to the much smaller organisms found in degraded wetlands. After the sorting process was complete, the sample was rinsed with water and dried in an oven at 60ºC for 24 hours. The dry weight of the zoobenthos was measured using an Ohaus microbalance (± 0.01 mg) and expressed as $g \cdot m^{-2}$.

Consistent with our hypothesis, we found a highly significant positive relationship between zoobenthos biomass and WQI scores for the 19 wetland-years (Fig. 8). The results of this study demonstrated that zoobenthos biomass can be a useful indicator of wetland quality. The higher zoobenthos biomass reflects the benefits that macrophyte cover confers on the macroinvertebrates of good-quality sites, and the fact that piscivores, rather than benthivorees and planktivores dominate.

Development of multiple regression models

To produce meaningful multiple regression models for species assemblages, we first conducted a correlation analysis to determine which of the 16 most common taxa (present in at least 20 of the study sites) in Table 1 could be grouped together for further analyses. Statistically significant correlations emerged for two main groups. Group 1 included white sucker, bowfin, spottail shiner, brown bullhead and common carp which were commonly found in many of the disturbed wetlands (Table 4a), while Group 2 included banded killifish, rockbass, pumpkinseed, bluegill, largemouth bass, golden shiner, yellow perch, bluntnose shiner, and emerald shiner, which were commonly found in many protected and relatively unimpacted embayments in the lower Great Lakes (Table 4b). Abundances of these empirically-derived groupings were then regressed against the following environmental variables in stepwise multiple regression analyses.

- Wetland Zooplankton Index (WZI) (see Lougheed and Chow-Fraser 2002)
- species richness of submergent taxa (# submergents)
- percent of inorganic minerals in suspended sediment (% minerals)
- water turbidity (TURB; NTU)
- concentration of total suspended solids (TSS; mg/L)
- temperature (TEMP; °C)
- conductance (COND; µS/cm)
- pH
- concentration of total phosphorus (TP; µg/L)
- concentration of soluble reactive phosphorus (SRP; µg/L)
- concentration of planktonic chlorophyll α (CHL; µg/L)
- concentration of total nitrate nitrogen (TNN; mg/L)
- concentration of total ammonia nitrogen (TAN; µg/L)
- concentration of total nitrogen (TN; mg/L)

Abundances of all fish classified into Groups 1 and 2 were entered into stepwise multiple regression analyses. Only Group 2 data produced a statistically significant 3-variable model, and the total amount of explained variance was relatively low (r^2=0.41; Table 5). The fish in this grouping appeared to favour relatively productive wetlands with sandy substrate and low water turbidity, conditions that typify fish habitat in many protected embayments and riverine wetlands (e.g. eastern Lake Ontario) with abundant submergent vegetation. Since these models only accounted for a small proportion of the total fish community (approximately 18%), all the fish were reclassified accordingly to taxocene (after Jude and Pappas 1992; Wei et al. 2003) and functional taxa (i.e. benthivores, carnivores, omnivores, herbivores, piscivores and planktivores and re-entered into multiple regressions. Regressions of the data sorted by taxocenes produced no significant multiple linear regressions. Of the 6 functional groups, statistically valid models emerged only for carnivores, benthivores and piscivores (Table 5). Compared with the earlier model, the amount of explained variance was unacceptably low for the benthivores and carnivores (r^2=0.15 and r^2=0.14, respectively), although the piscivore model had five variables that accounted for 80% of the total variation. This model indicated that piscivores were highest in wetlands with high WZI scores (indicating the presence of large, plant-associated zooplankton), species-rich submergent community, and low water turbidity. The positive association with TNN and TSS are difficult to interpret, but may indicate that moderately enriched wetlands can accommodate more piscivores.

We also conducted species-specific regressions using this same set of environmental variables (Table 6). In total, 24 of the 80 species produced significant multiple-regression models. Rather than using the accompanying equations directly to predict fish habitat, we intend to use the results to help us identify the most important variables, which would then be used to formulate ecologically meaningful predictive models for some of the species in Table 6. These predictive models could be developed for black bullhead, tadpole madtom, logperch, white bass, rockbass, bluegill, longnose gar and golden shiner (based on their relatively high r^2-values). We have also found Table 6 useful for verifying the general importance of certain water-quality parameters (e.g. SRP, CHL, pH, turbidity and % minerals) for the wetland-associated taxa.

3. *We will develop a pilot program combining field sampling with remote-sensing technology to evaluate the potential of using remotely sensed data to determine subtle changes in the floristic community, substrate type and water clarity of a few wetlands over several-year intervals*

This objective was attempted through David Lusch's project conducted by his Ph.D student, Brian Becker (see Appendix 4 for his completion report). During the late summer of 1999, the Center For Remote Sensing & GIS (CRS&GIS) at Michigan State University (MSU) contracted with 3DI, Inc. to acquire high-resolution (i.e. one meter), hyperspectral imagery (i.e. 20 very narrow spectral bands) with their AISA instrument over three coastal-wetland transects in the Les Cheneaux Islands along the northwestern coast of Lake Huron. Extensive botanical and biological data have been collected in these areas for several years by T. Burton (MSU) and D. Albert (MNFI) along established vegetative transects that encompass highly impacted, moderately impacted, and "pristine" coastal wetlands. Georeferencing control targets and spectral calibration panels were captured in the imagery, and baseline botanical characterization was completed just prior to image acquisition.

A portion of this grant in partnership with others (e.g. CRS&GIS, Ducks Unlimited) contracted with ITRES, Ltd. to acquire digital, georeferenced, airborne, hyperspectral imagery with their CASI-II instrument over the Wildfowl Bay Islands in Saginaw Bay. The resulting images (1 meter spatial resolution) contain twenty, 10-nanometer-wide spectral bands strategically located throughout the visible and near-infrared portions of the spectrum. Extensive in-situ, reflectance spectra and sub-meter GPS locations were acquired for the dominant plant genera and substrates identified at both wetland areas (Les Cheneaux Islands – B. Becker and D. Lusch; Wildfowl Bay – B. Becker and D. Albert). Extensive solar-elevation correction of these reference spectra was accomplished during the course of the study. Using the specified full-width, half-maximum (FWHM) spectral limits of the AISA and CASI-II bands, the field-acquired hyperspectral signatures of select genera and bare substrates were aggregated into synthetic, imager-specific radiances. These signatures were compared to actual imager radiance patterns. Training site statistics were generated from these Regions-of-Interest (ROIs) in order to inform a variety of image classification approaches.

By the end of the funding period in 2001, D. Lusch provided preliminary results that indicated several vegetative communities were separable: sedges (Carex spp.); cattails (Typha spp.); bulrushes (Scirpus spp.); rooted submergents as a class [e.g. eelgrass (Vallisneria spp.) and curly-leaf pondweed (Potamogeton spp.)] and floating-leaf vascular plants as a group such as water lilies (Nuphar spp.and Nymphaea spp.). Additionally, there appears to be a useful reflectance continuum related to standing biomass within the emergents, especially Typha spp., but to a lesser degree with Scirpus spp. also. The results of this research indicate that the IKONOS band centers are less than optimal with respect to the differentiation of coastal wetland vegetation (August/September imagery). SAM classification results using four “optimized” bands out-performed classifications utilizing IKONOS band centers by nearly 10 percent. This spectral limitation in combination with the 4-meter spatial resolution of multispectral IKONOS imagery limits its applicability only to broad vegetative classifications where detailed spatial patterns are not the focus. This research represents the first step in evaluating the effectiveness of applying high-resolution, narrow-band imagery to the detailed mapping of coastal wetlands in the Great Lakes region.

<u>Graduate students and post-doctoral fellows supported through this grant:</u>

As part of this overall program, D. McGoldrick, under the supervision of D. Barton, University of Waterloo, conducted a detailed investigation of the littoral food web of Little Rice Bay, in Long Point, Lake Erie, Ontario. Using a combination of stomach-content analysis and stable isotope analysis, he showed that invertebrates were an important dietary component of fish in the wetland, and that carbon from aquatic macrophyte was a significant portion of the diets of secondary producers through the detritus pool and microbial loop. His study provides strong evidence that the aquatic macrophytes contribute energetically to wetland food webs once decomposed as detritus and are not just structurally important as physical cover and refugia for wetland fauna.

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Seilheimer, T. (M.Sc. transferred to Ph.D in progress). Topic: Development and use of the Wetland Fish Index to assess the quality of the fish habitat in coastal wetlands of the lower Great Lakes.

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Table 1. Breakdown of the 40,015 fish caught with fyke nets in this study.

Species Code	Common Name	Total abundance	Occurrence in wetlands	Occurrence in wetlands (%)	% of total caught
ALPS	Alewife *Alosa pseudoharengus*	153	9	13.6	0.3824
AMCA	Bowfin *Amia calva*	82	24	36.4	0.2049
AMNE	Brown bullhead *Ameiurus nebulosus*	2243	42	63.6	5.6054
AMRU	Rock bass *Ambloplites rupestris*	395	44	66.7	0.9871
ANRO	American eel *Anguilla rostrata*	1	1	1.5	0.0025
APGR	Freshwater drum *Aplodinotus grunniens*	6	3	4.5	0.0150
APQU	Fourspine stickleback *Apeltes quadracus*	17	2	3.0	0.0425
APSA	Pirate perch *Aphredoderus sayanus*	2	1	1.5	0.0050
CAAU	Goldfish *Carassius auratus*	6	2	3.0	0.0150
CACA	Longnose sucker *Catostomas catostomus*	6	3	4.5	0.0150
CACO	White sucker *C. commersoni*	1551	18	27.3	3.8760
COBA	Mottled sculpin *Cottus bairdi*	5	2	3.0	0.0125
COCO	Slimy sculpin *C. cognatus*	6	3	4.5	0.0150
COPL	Lake chub *Couesius plumbeus*	3	1	1.5	0.0075
CUIN	Brook stickleback *Culaea inconstans*	122	7	10.6	0.3049
CYCA	Common carp *Cyprinus carpio*	334	27	40.9	0.8347
DOCE	Gizzard shad *Dorosoma cepedianum*	455	10	15.2	1.1371
ERSU	Lake chubsucker *Erimyzon sucetta*	1	1	1.5	0.0025
ESAA	Redfin pickerel *Esox americanus americanus*	1	1	1.5	0.0025
ESAV	Grass pickerel *E. a. vermiculatus*	3	2	3.0	0.0075
ESLU	Northern pike *E. lucius*	54	25	37.9	0.1349
ESMA	Muskellunge *E. masquinongy*	5	2	3.0	0.0125
ETEX	Iowa darter *Etheostoma exile*	14	2	3.0	0.0350
ETMI	Least darter *E. microperca*	162	2	3.0	0.4048
ETNI	Johnny darter *E. nigrrum*	141	18	27.3	0.3524
FUDI	Banded killifish *Fundulus diaphanus*	474	23	34.8	1.1846
GAAC	Threespine stickleback *Gasteroteus aculeatus*	545	9	13.6	1.3620
GYCE	Ruffe *Gymnocephalus cernuus*	10	2	3.0	0.0250

HYHA	Brassy minnow *Hybognathus hankinsoni*	35	3	4.5	0.0875
ICME	Black bullhead *Ameiurus melas*	5143	11	16.7	12.8527
ICPU	Channel catfish *Ictalurus punctatus*	32	5	7.6	0.0800
LASI	Brook silverside *Labidesthes sicculus*	3	3	4.5	0.0075
LECY	Green sunfish *Lepomis cyanellus*	140	5	7.6	0.3499
LEGI	Pumpkinseed *L. gibbosus*	2799	53	80.3	6.9949
LEMA	Bluegill *L. macrochirus*	1836	35	53.0	4.5883
LEMI	Redear sunfish *L. microlophus*	5	1	1.5	0.0125
LEOC	Spotted gar *Lepisosteus oculatus*	2	1	1.5	0.0050
LEOS	Longnose gar *L. osseus*	6	5	7.6	0.0150
MIDO	Smallmouth bass *Micropterus dolomieu*	74	12	18.2	0.1849
MISA	Largemouth bass *M. salmoides*	2293	33	50.0	5.7304
MOAM	White perch *Morone americana*	4232	7	10.6	10.5760
MOAN	Silver redhorse *Moxostoma anisurum*	14	2	3.0	0.0350
MOCH	White bass *Morone chrysops*	40	3	4.5	0.1000
MOMA	Shorthead redhorse *Moxostoma macrolepidotum*	4	4	6.1	0.0100
NEME	Round goby *Neogobius melanostomus*	3	1	1.5	0.0075
NOAL	Whitemouth shiner *Notropis alborus*	138	1	1.5	0.3449
NOAN	Pugnose shiner *N. anogenus*	3	1	1.5	0.0075
NOAT	Emerald shiner *N. atherinoides*	575	22	33.3	1.4370
NOCO	Common shiner *Luxilus cornutus*	155	7	10.6	0.3874
NOCR	Golden shiner *Notemigonus crysoleucas*	349	23	34.8	0.8722
NOGY	Tadpole madtom *Noturus gyrinus*	38	19	28.8	0.0950
NOHE	Blacknose shiner *Notropis heterolepis*	822	20	30.3	2.0542
NOHN	Blackchin shiner *N. heterondon*	7	3	4.5	0.0175
NOHU	Spottail shiner *N. hudsonius*	3087	32	48.5	7.7146
NOSP	Spotfin shiner *Cyprinella spiloptera*	190	8	12.1	0.4748
NOST	Sand shiner *N. stamineus*	61	2	3.0	0.1524
NOVO	Mimic shiner *N. volucellus*	83	6	9.1	0.2074
ONMY	Rainbow trout *Oncorhynchus*	2	2	3.0	0.0050

	mykiss				
ONTS	Chinook salmon *O. tshawytscha*	1	1	1.5	0.0025
OSMO	Rainbow smelt *Osmerus mordax*	147	5	7.6	0.3674
PECA	Logperch *Percina caprodes*	55	9	13.6	0.1374
PEFL	Yellow perch *Perca flavescens*	889	45	68.2	2.2217
PEMA	Sea lamprey *Petromyzon marinus*	1	1	1.5	0.0025
PEOM	Trout-perch *Percopsis omiscomaycus*	1464	7	10.6	3.6586
PHEO	Northern redbelly dace *Phoxinus eos*	93	2	3.0	0.2324
PINO	Bluntnose minnow *Pimephales notatus*	1514	26	39.4	3.7836
PIPR	Fathead minnow *P. promelas*	329	11	16.7	0.8222
POAN	White crappie *Pomoxis annularis*	238	11	16.7	0.5948
PONI	Black crappie *P. nigromaculatus*	176	12	18.2	0.4398
PRCY	Round whitefish *Prosopium cylindraceum*	9	2	3.0	0.0225
PUPU	Ninespine stickleback *Pungitius pungitius*	149	7	10.6	0.3724
RHCA	Longnose dace *Rhinichthys cataractae*	38	1	1.5	0.0950
SCER	Rudd *Scardinius erythrophthalmus*	2	1	1.5	0.0050
SEAT	Creek chub *Semotilus atromaculatus*	4	3	4.5	0.0100
SEMA	Pearl dace *Margariscus margarita*	43	3	4.5	0.1075
STVI	Walleye *Stizostedion vitreum*	3	2	3.0	0.0075
UMLI	Central mudminnow *Umbra limi*	21	10	15.2	0.0525
UNCY	Cyprinid *Cyprinidae*	20	5	7.6	0.0500
UNIC	Bullhead *Ameiurus spp.*	3780	18	27.3	9.4465
UNLE	Sunfish *Lepomis spp.*	2071	31	47.0	5.1756

Table 2. **"U" and "T"** values for fish species based on presence/absence data (P/A) and abundances (AB)

Code	**Common Name**	**P/A**		**AB**	
		U	**T**	**U**	**T**
NOHN	Blackchin shiner	5	1*	5	1*
NOVO	Mimic shiner	5	1*	5	1*
ESAV	Grass pickerel	5	1*	5	1*
NOHE	Blacknose shiner	4	2	5	3
ETNI	Johnny darter	4	2	5	3
UMLI	Central mudminnow	4	2	5	2
LEOS	Longnose gar	5	3	4	3
AMCA	Bowfin	4	2	4	2
ESMA	Muskellunge	4	1*	4	1*
FUDI	Banded killifish	4	3	4	2
NOGY	Tadpole madtom	4	2	4	2
AMRU	Rockbass	3	1	4	2
MISA	Largemouth bass	3	2	4	2
ESLU	Northern pike	3	1	4	1
LEGI	Pumpkinseed	3	1	3	1
LEMA	Bluegill	3	1	3	1
MIDO	Smallmouth bass	3	2	3	2
NOCR	Blacknose shiner	3	1	3	1
AMNE	Brown bullhead	3	1	3	1
ICME	Black bullhead	3	1	3	1
PEFL	Yellow perch	3	1	3	1
CYCA	Common carp	2	1	3	1
PONI	Black crappie	4	1	2	1
NOAT	Emerald shiner	3	1	2	1
LASI	Brook silverside	2	2	2	2
LECY	Green sunfish	2	1	2	1
DOCE	Gizzard shad	2	2	2	2
NOHU	Spottail shiner	2	1	2	1
NOSP	Spotfin shiner	2	1	2	1
PINO	Bluntnose minnow	2	1	2	1
ICPU	Channel catfish	1	2	2	2
APGR	Freshwater drum	1	1	2	1
ALPS	Alewife	2	1	1	1
PIPR	Fathead minnow	2	1	1	1
CACO	White sucker	1	1	1	1
POAN	White crappie	1	2	1	2
CAAU	Goldfish	1	1	1	3
HYHA	Brassy minnow	1	1*	1	1*
GAAC	Three-spine stickleback	1	1	1	2
MOAM	White perch	1	1	1	2
MOCH	White bass	1	3	1	3
PECA	Logperch	1	1*	1	1*

* species occurring in < 5% of wetlands were automatically assigned a T value of 1

Table 3. Description of the 18 wetlands (19 wetland-years) sampled during the summers of 2001 and 2002 for zoobenthos

Wetland	**Lake**	**Sample Date**
Cloud Bay	Superior	08-Jun-02
Echo Bay	Huron	10-Jul-02
Wigwam	Huron	13-Jul-01
Old Woman Creek	Erie	09-Aug-02
Rondeau	Erie	23-Jul-01
Turkey Point	Erie	16-Jul-01
Grand River	Erie	05-Jul-01
Spicer Creek	Niagara	09-Aug-01
Jordan Harbour	Ontario	09-Aug-02
Fifteen Mile	Ontario	23-May-02
Cootes Paradise Marsh	Ontario	11-Jul-01
Frenchman's	Ontario	18-Jun-02
Frenchman's	Ontario	02-Aug-01
Presquile	Ontario	08-Aug-01
Little Cataraqui	Ontario	24-Jul-02
Madoma	Ontario	25-Jul-02
Mud	Ontario	25-Jun-02
Sandy Creek	Ontario	18-Jul-01
Little Sodus	Ontario	20-Jul-01

Table 4. Matrices summarizing two groups of species that were significantly correlated with each other based on abundances (Pearson's correlation; P<0.05 except those indicated by an asterisk, that have values between 0.05 and 0.10). Only species that occurred in at least 20 wetlands were included in the correlation analysis (see Appendix 1).

a)

	White sucker	Bowfin	Spottail shiner	Brown bullhead	Common carp
White sucker					
Bowfin	**0.359**				
Spottail shiner	**0.925**				
Brown bullhead			* **0.295**		
Common carp				**0.563**	

b)

	Rock-bass	Banded killifish	Pumpkin-seed	Bluegill	Large-Mouth bass	Golden shiner	Yellow perch	Blutnose shiner	Emerald shiner
Rockbass									
Banded killifish	**0.377**								
Pumpkinseed									
Bluegill			**0.512**						
Largemouth bass	* **0.288**								
Golden shiner			**0.310**	* **0.283**					
Yellow perch					**0.664**				
Bluntnose shiner	* **0.258**	**0.402**	* **0.253**			**0.761**			
Emerald shiner						**0.819**			

Table 5. Summary of stepwise multiple regression analyses for fish abundance against environmental variables. Group 1 and 2 refer to the two groups that emerged from the correlation analysis (Table 4). Benthivores, carnivores, and piscivores refer to life stage of fish whose diets include primarily benthic organisms, insects and other invertebrates, and fish, respectively. "+ve" and "-ve" refers to the sign associated with the regression coefficients for each of the significant variables. See text for explanation of abbreviations of the independent variables. Variables that were entered into the regression model but which did not contribute significantly were pH and temperature.

Categories	**WZI score**	**# submergent**	**% minerals**	**Log TURB**	**Log TSS**	**Log COND**	**Log TP**	**Log SRP**	**Log CHL**	**Log TNN**	**Log TAN**	**Log TN**	**r^2**	**P-value**
Group 2			+ve	-ve			+ve					+ve	0.406	<0.001
Benthivores					+ve	-ve							0.146	0.0193
Carnivores					-ve						+ve		0.140	0.0232
Piscivores	+ve	+ve		-ve	+ve					+ve			0.803	0.0003

Table 6. Summary of multiple stepwise regression analyses for individual fish species. "+ve" or "-ve" refers to the sign associated with the regression coefficients for each significant variable of the stepwise regression. See text for explanation of abbreviations of the independent variables.

Species	**Wetlands included in model**	**WZI score**	**# submergents**	**% minerals**	**Log TURB**	**Log TSS**	**Log TEMP**	**Log COND**	**pH**	**Log TP**	**Log SRP**	**Log CHL**	**Log TNN**	**Log TAN**	**Log TN**	**r^2**	**P-value**
Bowfin	53			+ve	-ve	-ve	+ve				+ve	-ve				0.279	0.0154
Brown bullhead	53			+ve	-ve						+ve		-ve	+ve		0.203	0.0514
Black bullhead	18	-ve					-ve	+ve					-ve			0.566	0.0207
Shorthead redhorse	49		-ve				-ve	-ve	+ve					+ve		0.234	0.0371
Tadpole madtom	18	-ve	+ve	+ve	-ve			+ve		-ve	+ve	-ve	+ve	+ve		0.929	0.0039
Blackchin shiner	52									+ve	-ve	-ve				0.180	0.0221
Immature bullheads	52					+ve	+ve	-ve								0.239	0.0042
Logperch	18	-ve	+ve						-ve			-ve	+ve			0.624	0.0231
White bass	18	-ve	+ve	-ve	-ve											0.932	<0.0001
Rockbass	18	+ve		+ve	-ve				-ve			+ve			-ve	0.841	0.0007
Pumpkinseed	52							+ve							-ve	0.107	0.0623
Bluegill	18	+ve							+ve		-ve	+ve			-ve	0.783	0.0011
Longnose gar	18	+ve		+ve	+ve									-ve		0.649	0.0067
Largemouth bass	52				-ve										+ve	0.208	0.0033
Golden shiner	18	+ve	+ve	+ve	+ve			+ve				-ve		-ve		0.851	0.0018
Immature cyprinids	49		-ve	-ve			-ve							-ve		0.1862	0.0548
9-spine stickleback	53			-ve	+ve				-ve		-ve	-ve				0.2279	0.0282
4-spine stickleback	52								-ve		-ve	-ve				0.173	0.0264
Longnose sucker	49		-ve	-ve		-ve	-ve		-ve	+ve	-ve			-ve		0.276	0.0861
White sucker	52								-ve		-ve	-ve				0.167	0.0308
Mottled sculpin	52								-ve		-ve	-ve				0.016	0.0359
Brook stickleback	53			-ve	+ve				-ve		-ve	-ve				0.208	0.0461
White perch	52			+ve	-ve								-ve			0.371	0.0002
Trout-perch	52						+ve	-ve						+ve		0.235	0.0047
Summary	*24*	*8*	*7*	*12*	*11*	*3*	*7*	*7*	*10*	*3*	*11*	*12*	*5*	*8*	*4*		

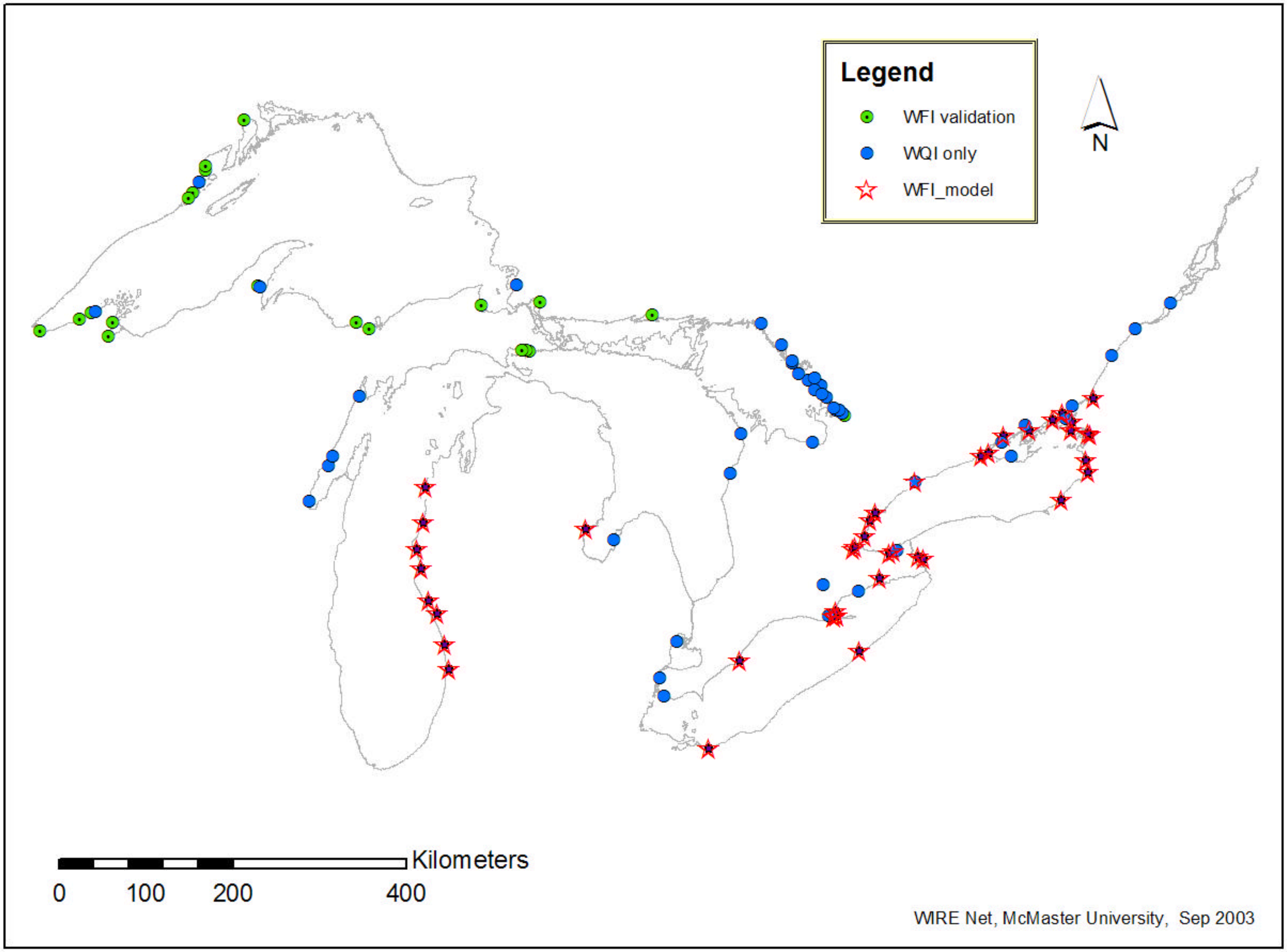

Fig. 1. Location of study sites around the shoreline of the Great Lakes. All sites were included in the development of the WQI, whereas only sites indicated by the green circle and the red star were used in the development and validation of the WFI, respectively.

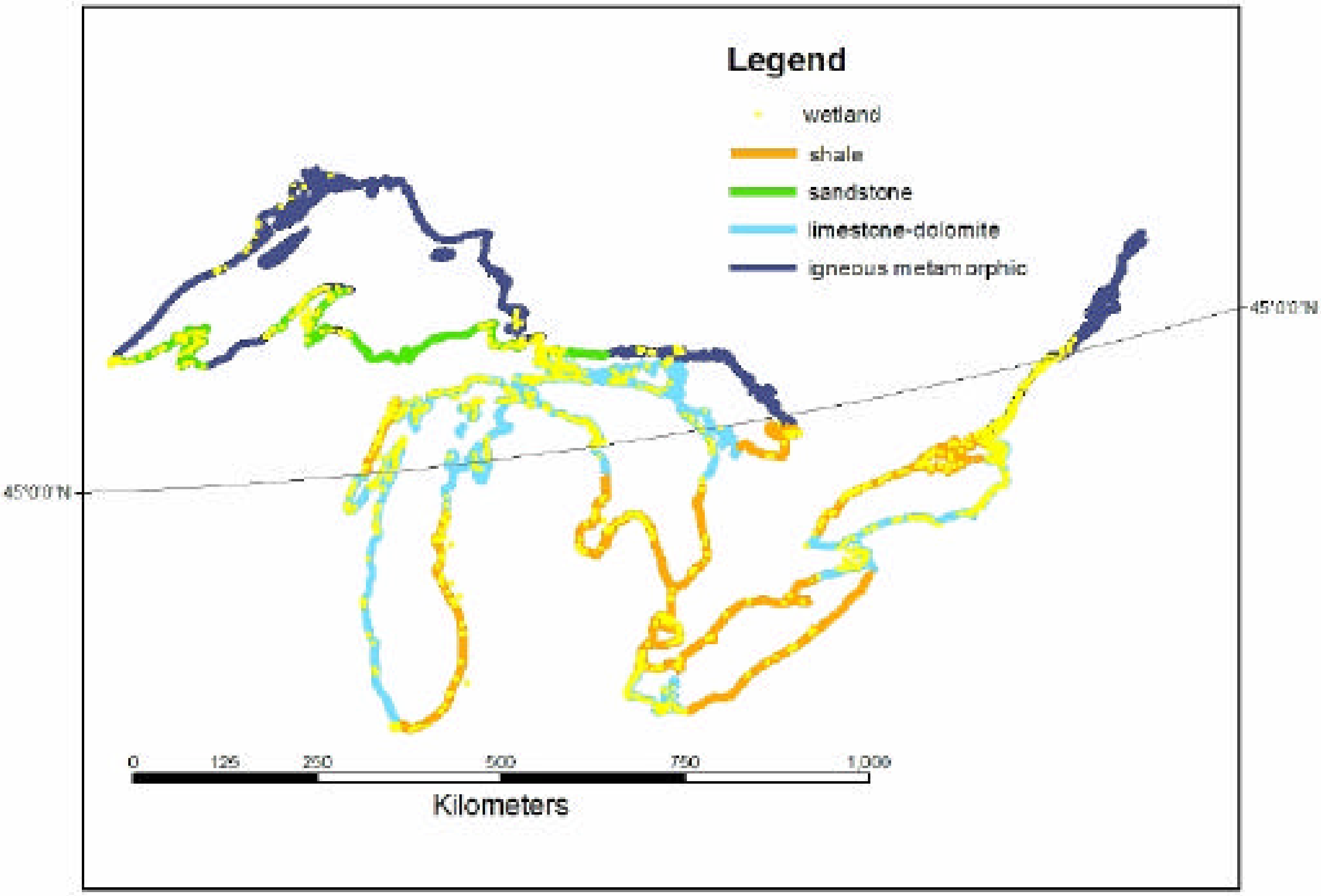

Fig. 2. Location of wetlands (yellow) superimposed over primary bedrock types along shorelines of the Great Lakes (redrawn from Wei 2002). Data for bedrock were digitized from a figure in Minc (1997) and location of wetlands were taken from McMaster University's WIRE Net database (Chow-Fraser and Albert 1999).

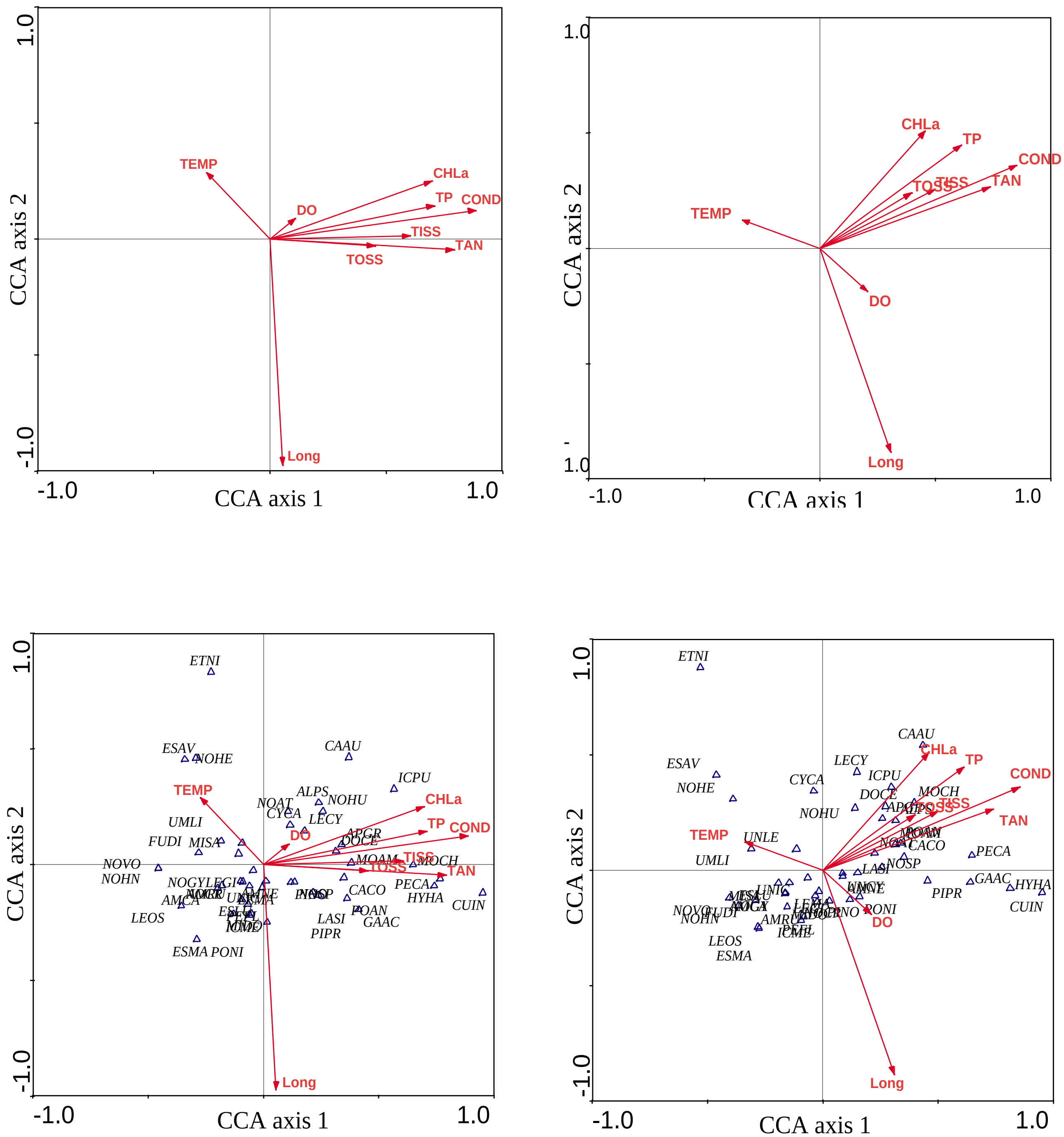

Fig. 3 Environmental vectors from CCA with a) presence-absence data and b) $\log_{10}$ abundance data. Biplot of canonical axis 1 vs. canonical axis 2 using c) presence-absence data and d) $\log_{10}$ abundance data.

a)

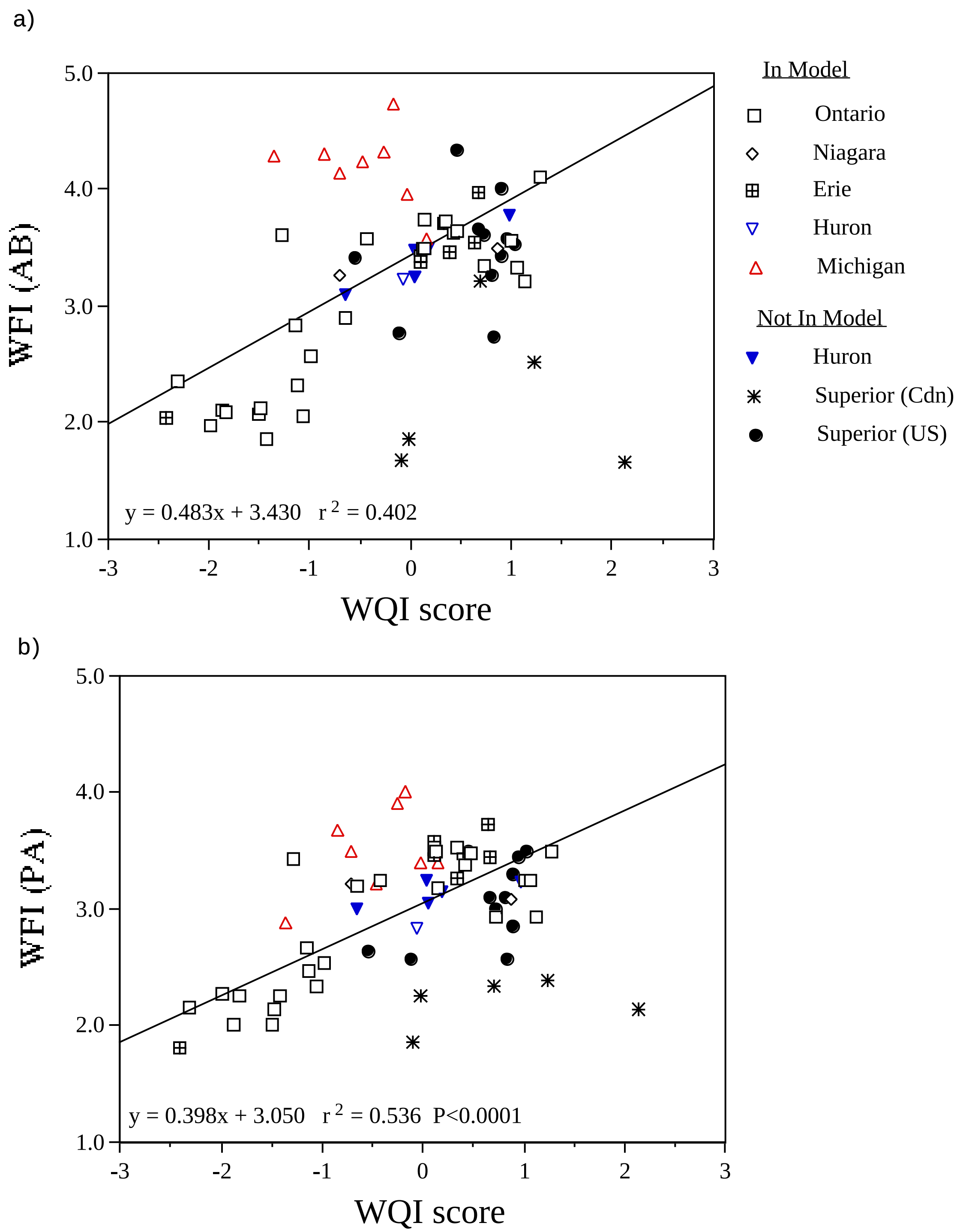

Fig. 4. Plot of a)WFI (AB) and b) WFI (PA) scores against corresponding WQI scores for all 64 wetland-years. Regression line and statistics correspond to open symbols only, which are the 43 wetland-years used to develop the WFI. All closed symbols and the asterisk correspond to sites that occurred north of 45ºN.

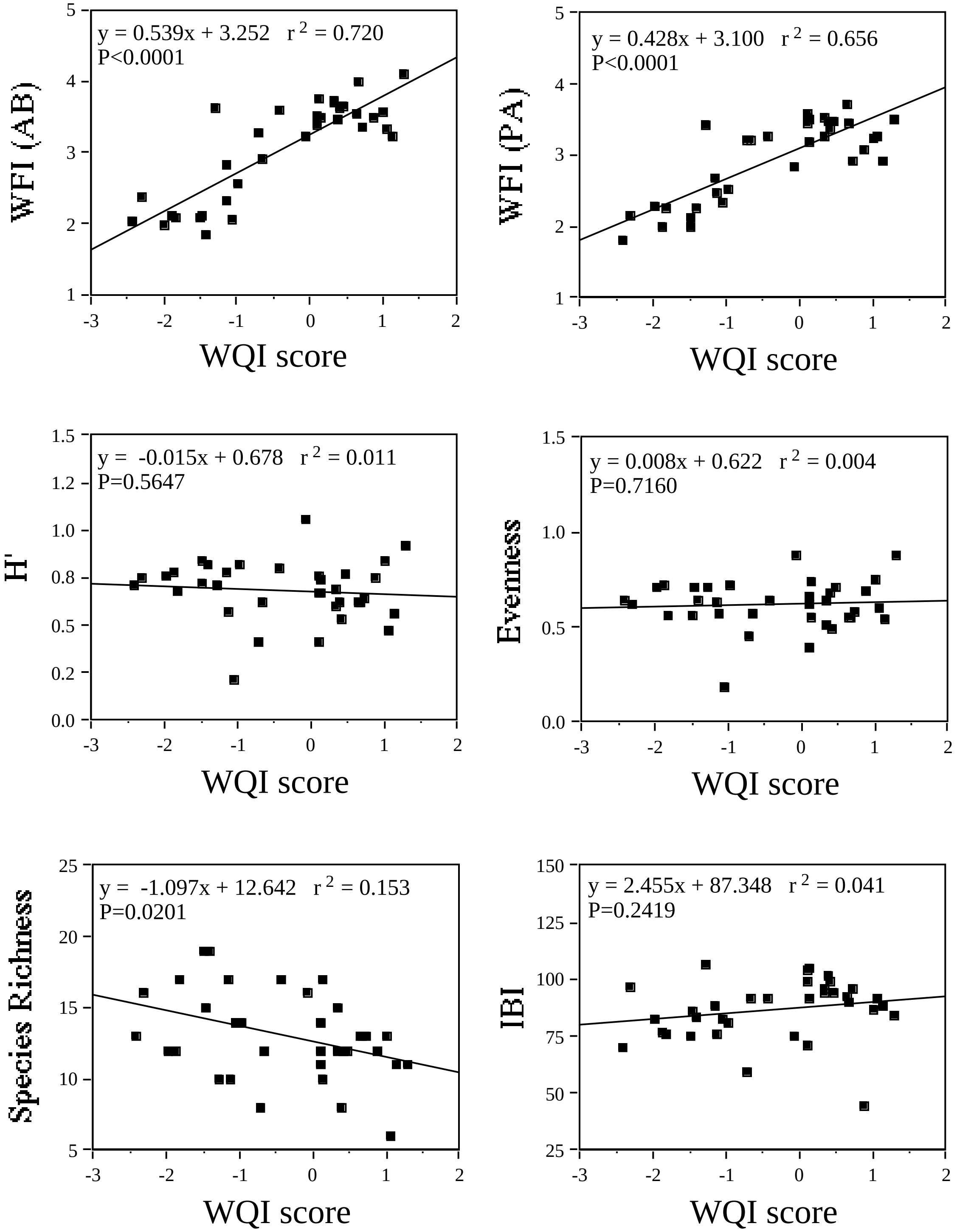

Fig. 5. Plot of a)WFI (AB), b) WFI (PA), c) Shannon-Wiener's H', d) Simpson's evenness, e) species richness and f) IBI scores versus corresponding WQI scores for 35 wetlands.

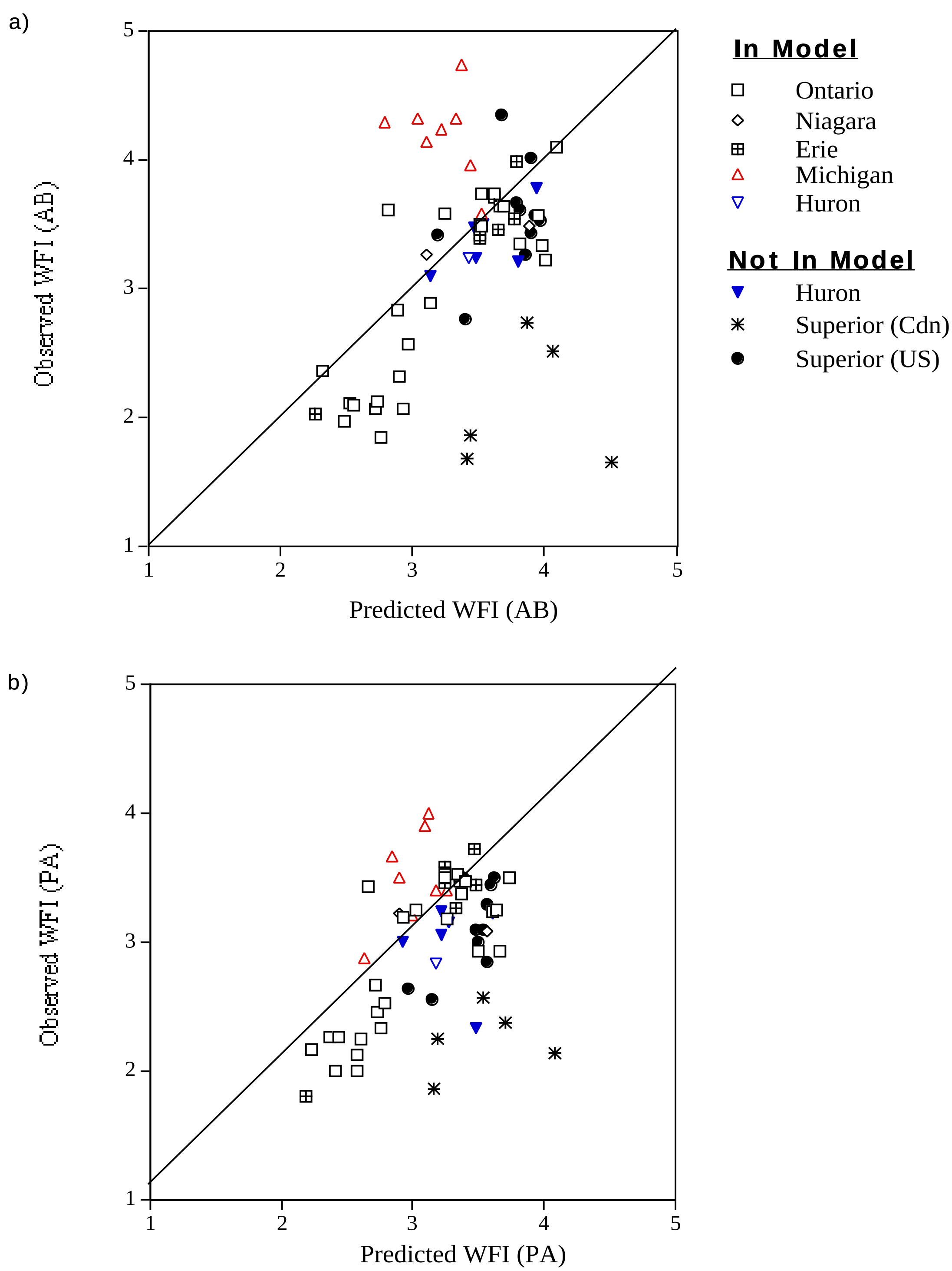

Fig. 6. Plot of observed WFI scores (calculated using **U** and **T** values from Table 2) versus predicted scores (based on WQI values and best-fit line in Fig. 4a and b) for a) abundance data and b) presence-absence data.

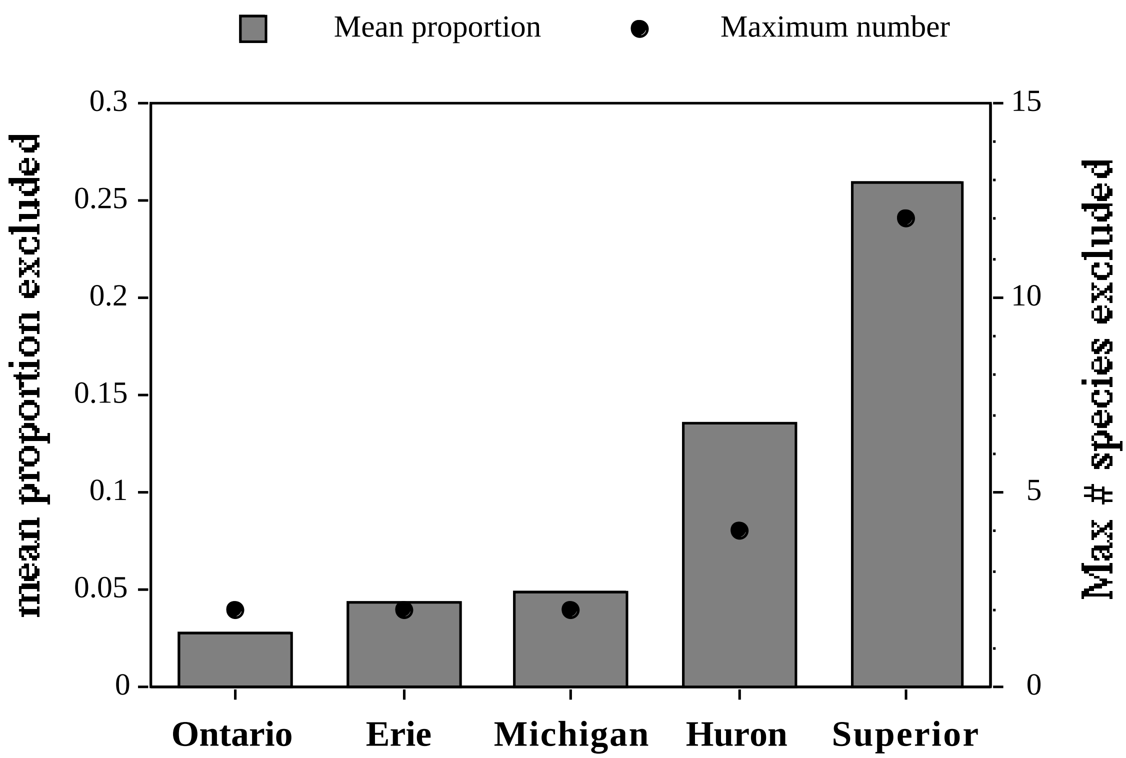

Figure 7. Differences in the proportion of species in wetlands and maximum number of species excluded in the development of the WFI, sorted by Great Lake.

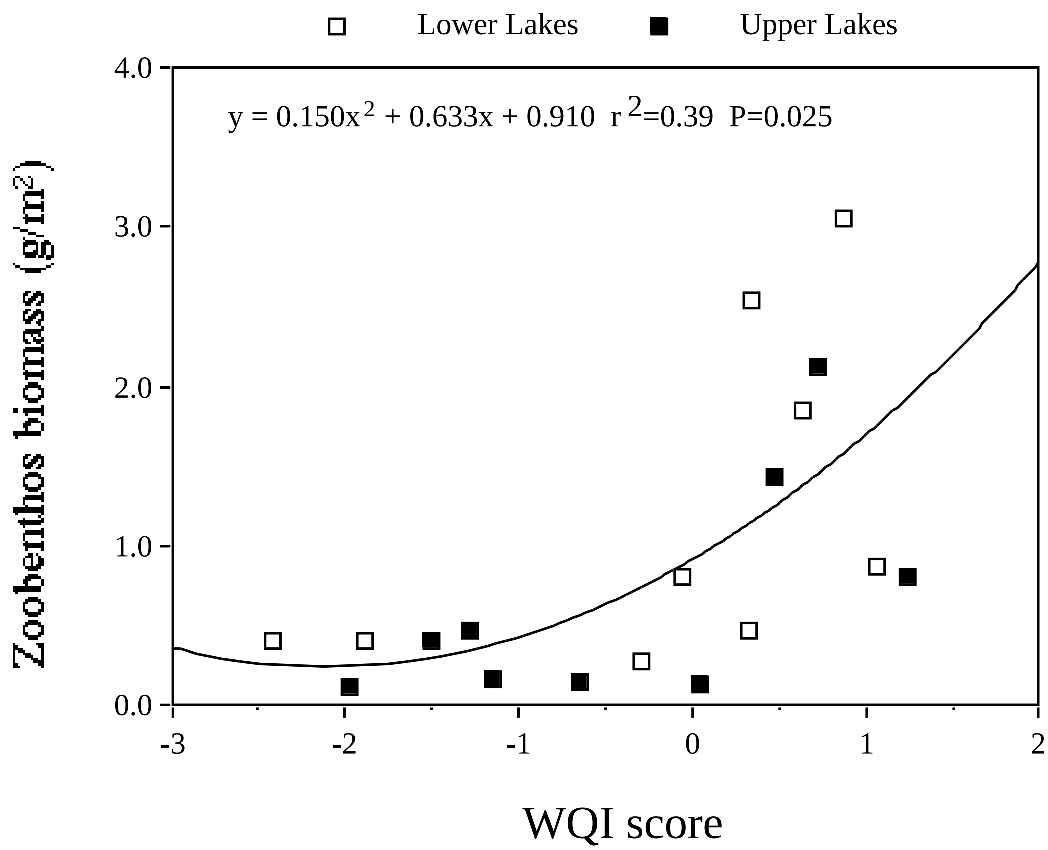

Figure 8. Relationship between biomass of zoobenthos and WQI scores for 19 wetland-years.

List of Appendices

Appendix 1:
Summary of wetland locations in this study. Asterisks indicate wetlands that were not included in the development of the Wetland Fish Index.

Appendix 2:
Chow-Fraser, P., Kostuk, K. Seilheimer, T., Weimer, M. MacDougall, T. and Theysmeyer, T. 2003. Effect of wetland quality on sampling bias associated with two fish survey methods for coastal wetlands of the lower Great Lakes. (Prepared for publication in "Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators" Eds Simon, T.P., Stewart, P.M., Munawar, M. and Edsall, T.A.)

Appendix 3:
Lougheed, V.L. and Chow-Fraser, P. 2002. Development and use of a zooplankton index of wetland quality in the Laurentian Great Lakes basin. Ecol. Applic. 12: 474-486.

Appendix 4:
Becker, B.L, and Lusch, D.P. 2001. A Classification-Based Assessment of the Optimal Spatial and Spectral Resolution of Coastal Wetland Imagery, Completion report to GLFC, Michigan State University, Center For Remote Sensing and GISci, East Lansing, Michigan.

Appendix 1. Summary of wetland locations in this study. Asterisks indicate wetlands that were not included in the development of the Wetland Fish Index.

Lake	Wetland	Code	Latitude	Longitude	Year
Erie	Grand River	GR	42.9000	79.6000	2001
Erie	Long Point	LP	42.5893	80.3355	2001
Erie	Presque Isle	PRa	42.1590	80.0985	2000
Erie	Presque Isle	PRb	42.1590	80.0985	2001
Erie	Old Woman Creek	OW	41.3822	82.5145	2002
Erie	Rondeau Bay	RN	42.2880	81.8670	2001
Erie	Turkey Point	TP01	42.6336	80.3417	2001
Erie	Turkey Point	TP02	42.6336	80.3417	2002
Huron	Wigwam Bay	WW	43.9702	83.8543	2001
Huron *	Cedarville	CV	45.9834	84.3501	2002
Huron *	Echo Bay	EB	46.4945	84.0760	2002
Huron *	Mackinac Bay	MC	46.0001	84.4001	2002
Huron *	Matchedash Bay	MB	44.7333	79.6668	2002
Huron *	Spanish River	SR	46.1834	82.3500	2002
Michigan	Betsie	BT	44.6129	86.2142	2001
Michigan	Kalamazoo	KZ	42.6335	86.1668	2001
Michigan	Lincoln	LN	43.9800	86.4400	2001
Michigan	Muskegon	MG	43.2501	86.2501	2001
Michigan	Pentwater	PW	43.7628	86.4078	2001
Michigan	Pigeon River	PN	42.8997	86.1883	2001
Michigan	White River	WR	43.4002	86.3500	2001
Michigan	Manistee River	MN	44.2622	86.2958	2001
Niagara	Buckhorn	BU	43.0563	78.9712	2001
Niagara	Spicer Creek	SP	43.0234	78.8968	2001
Ontario	Blessington Bay	BB	44.1670	77.3330	2001
Ontario	Bronte Creek	BR	43.3833	79.7002	2002
Ontario	Cootes Paradise	CP01	43.2667	79.9167	2001
Ontario	Cootes Paradise	CP02	43.2667	79.9167	2002
Ontario	Credit River	CR	43.5500	79.5800	2002
Ontario	Darlington	DA	43.8730	78.7970	2002
Ontario	Fifteen Mile Creek	FM	43.1669	79.3167	2002
Ontario	Goose Bay	GO	44.3501	75.8667	2002
Ontario	Grass Bay	GS	44.1502	76.2668	2002
Ontario	Grindstone Creek	GC	43.2932	79.8839	2002
Ontario	Grindstone Sunfish Pnd	GF	43.2833	79.8833	2002
Ontario	Hay Bay Marsh	HB	44.1667	76.9334	2002
Ontario	Humber River	HM	43.6167	79.4833	2002
Ontario	Jordan Harbour	JH	43.1501	79.3833	2002
Ontario	Little Cataraqui Creek	LQ	44.2167	76.5500	2002
Ontario	Little Sodus	LS	43.3394	76.6945	2001
Ontario	Madoma Creek	MO	44.2667	76.3833	2002

Ontario	Mud Bay	MU	44.0668	76.3167	2002
Ontario	Muskellunge River	MK	43.9668	76.0501	2002
Ontario	Perch River	PC	43.9836	76.0669	2002
Ontario	Presqu'ile Prov Pk	PI	44.0000	77.7306	2002
Ontario	Salmon River	SA	43.5683	76.2022	2002
Ontario	Sandy Creek	SC	43.7009	76.1965	2001
Ontario	Wellers Bay	WB	44.0168	77.6167	2002
Superior *	Au Train	AT	46.4333	86.8168	2001
Superior *	Bark Bay	BK	46.8504	91.1982	2001
Superior *	Chippewa Park	CW	48.3170	89.2000	2002
Superior *	Cloud Bay	CB01	48.0828	89.4372	2001
Superior *	Cloud Bay	CB02	48.0828	89.4372	2002
Superior *	Flag	FG	46.7867	91.3878	2001
Superior *	Hurkett Cove	HU	48.8330	88.5000	2002
Superior *	Laughing Whitefish	LF	46.5168	87.0169	2001
Superior *	Lost Creek	LC	46.8586	91.1358	2001
Superior *	Nemadji River	NJ	46.6835	92.0334	2001
Superior *	Pike River	PK	47.0168	88.5168	2001
Superior *	Pine Bay	PB	48.0333	89.5195	2001
Superior *	Sioux River	SX	46.7343	90.8779	2001
Superior *	Sturgeon Bay Superior	SU	48.2000	89.3167	2001
Superior *	Taquamenon River	TQ	46.5501	85.0169	2001
Superior *	West Fish Bay	WS	46.5842	90.9461	2001

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

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ABSTRACT

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 Geat 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) although on average, FN caught 75% whereas EB only captured 68% of the species encountered in respective wetlands.

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 2000). 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); 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 2003). To properly assess their current status and to track changes 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 (Hartwell 1998).

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* gear such as gill nets, trap nets, and fyke nets, as well as *active- capture* gear 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 1994; 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 active, since it is used to seek out fish where they occur. 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 (2003) 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 condtion, 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 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.

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 (10' long; 3' x 4' rectangular front openings; 1/2" for one net and 3/16" nylon mesh for the other) had five 30" stainless steel rings forming two throats that led to a cod end, and were deployed in deeper areas (approximately 1-m depth). In contrast, the small nets (3' x 1' rectangular front openings; 3/16" nylon mesh for both nets) could only be deployed where water depths were shallow (< 0.5 m). Wings (3' x 10'; 3/16" 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 25' leads (3/16" 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 10' steel tubing.

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 substarte 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, NY, Ontario Ministry of Natural Resources (OMNR) at Port Dover, ON, and Royal Botanical Gardens (RBG) at Burlington, ON (see Table 1). In all cases, the EB was conducted during daylight hours. The specific protocols used both 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 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" 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 $\cdot$ 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 meters in depth. Several transects (minimum of 300 shock seconds each in duration) 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)

OMNR used a 20 ft. 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 1000 shock seconds, covering an approximate area of 5-7000 m^2. All fish captured were placed into an aerated live-well and allowed to recover before sampling.

RBG (Burlington, ON)

RBG used an 18-ft flat-bottom Grumman, powered by a 25 hp Johnson outboard engine. 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 20-ft 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 Macintoshä 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 9592 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. FN 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 (2-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 very often (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 very ubiquitous, found in 8 or more of the 11 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 when data for both methods were 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), indicating that the bias associated with the two sampling methods varied according to wetland quality.

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 (2-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 6 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 species that were recovered exclusively by one gear type in this survey. There were 8 species recovered exclusively by EB, compared with 10 by FN (Table 6). Consistent with previous trends, FN tended to catch comparatively more of the smaller individuals. All of the taxa recovered by EB occurred in relatively low numbers (< 6), whereas several of those caught by FN occurred in very large 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 species. Using the same reasoning, EB appears to be ineffective for sampling tadpole madtom, since this species 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 only 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 the 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, as well as gizzard shad, a herbivore that fed on the plentiful algae in the marsh.

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 (black crappie and largemouth bass) and ambush predators (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. Corcoran 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 hours required to capture the fish, as well as the amount of time required to set the nets and process the fish. 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 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), and time of day (e.g. Paragamian 1989) and time of season (Dumont and Dennis 1997) may affect capture rates of certain species.

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	# 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.

		Survey method	
Parameter	**All fish**	**EB**	**FN**
# fish caught	9592	1120	8472
% all fish caught	---	11.7	88.3
Biomass of fish (kg)	218.5	92.7	125.8
% all fish biomass	---	42.4	57.6
# species recovered	47	36	40
% total species recovered	---	76.6	85.1
# 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 # 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 species encountered during Electrofishing Boat (EB) and Fyke net (FN) surveys in this study.

Family	Common name	Scientific name	Number of Specimens			Number of Wetlands		
			Both	EB	FN	Both	EB	FN
Carnivore								
Anjuillidae	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 gibbonus*	1110	220	890	10	11	10
Centrarchidae	Bluegill	*Lepomis macrochirus*	682	49	633	7	7	10
Centrarchidae	Sunfish	*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)	*Pomoxis annularis*	47	0	47	0	0	3
Centrarchidae	Black Crappie (0-160mm)	*Pomoxis nigromaculatus*	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
Easterosreidae	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	*Lepisoteus osseus*	2	1	1	0	1	1
Percichthyidae	White Perch (Young)	*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

Percichthyidae	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	*Stizostedion vitreum*	2	2	0	0	2	0
Benthivore								
Catostomidae	White Sucker	*Catostomus commersoni*	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	*Icalurus melas*	2	2	0	0	2	0
Ictaluridae	Bullhead (juvenile)	*Icalurus 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)	*Notropis sp.*	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.

			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.

Wetland	Others		Piscivore		Benthivore		Total		Prob
	EB	**FN**	**EB**	**FN**	**EB**	**FN**	**EB**	**FN**	
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 species 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

Figure 1

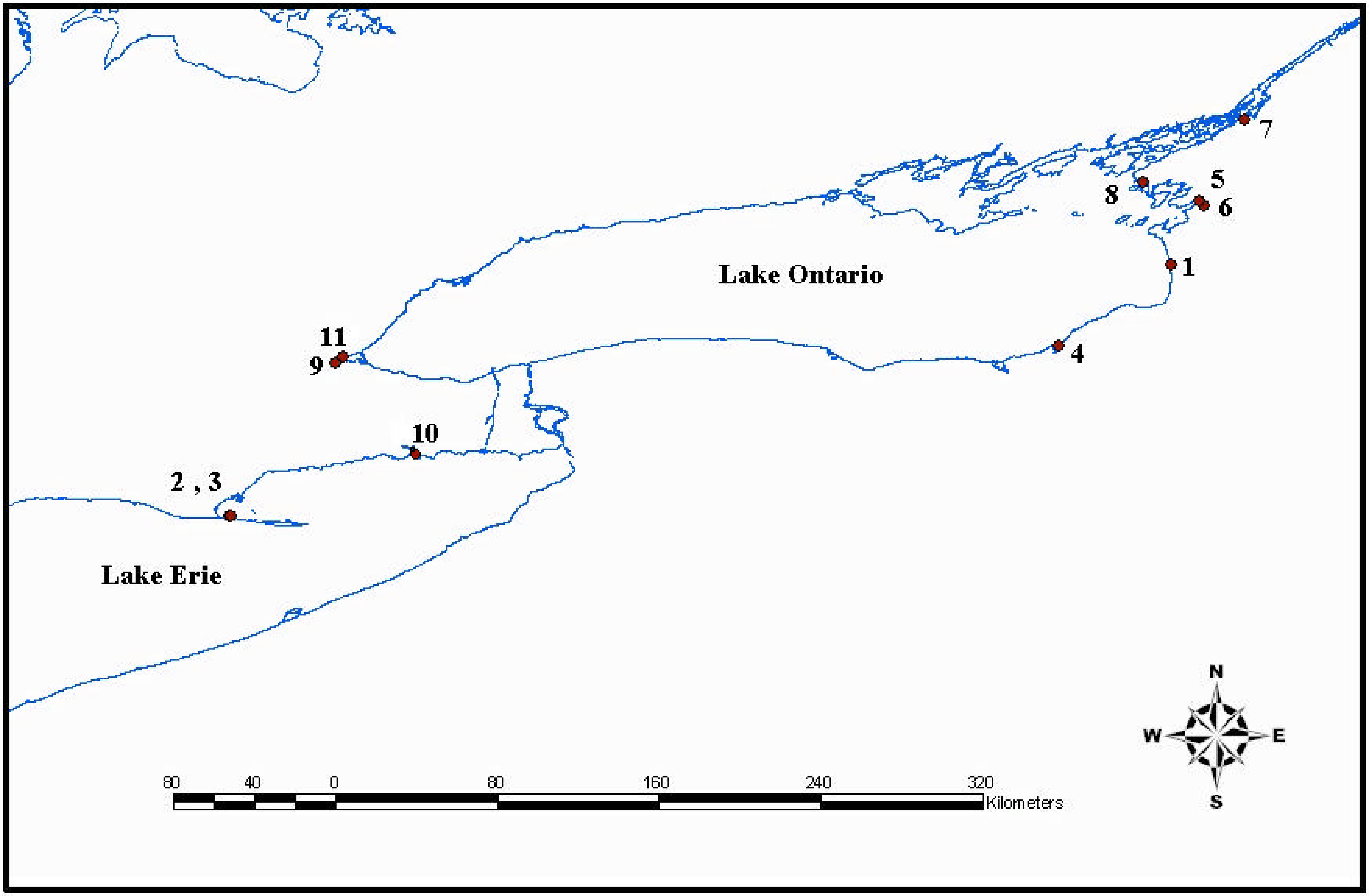

Lake Ontario
Lake Erie
1
2 , 3
4
5
6
7
8
9
10
11
N
S
E
W
80
40
0
80
160
240
320
Kilometers

Figure 2.

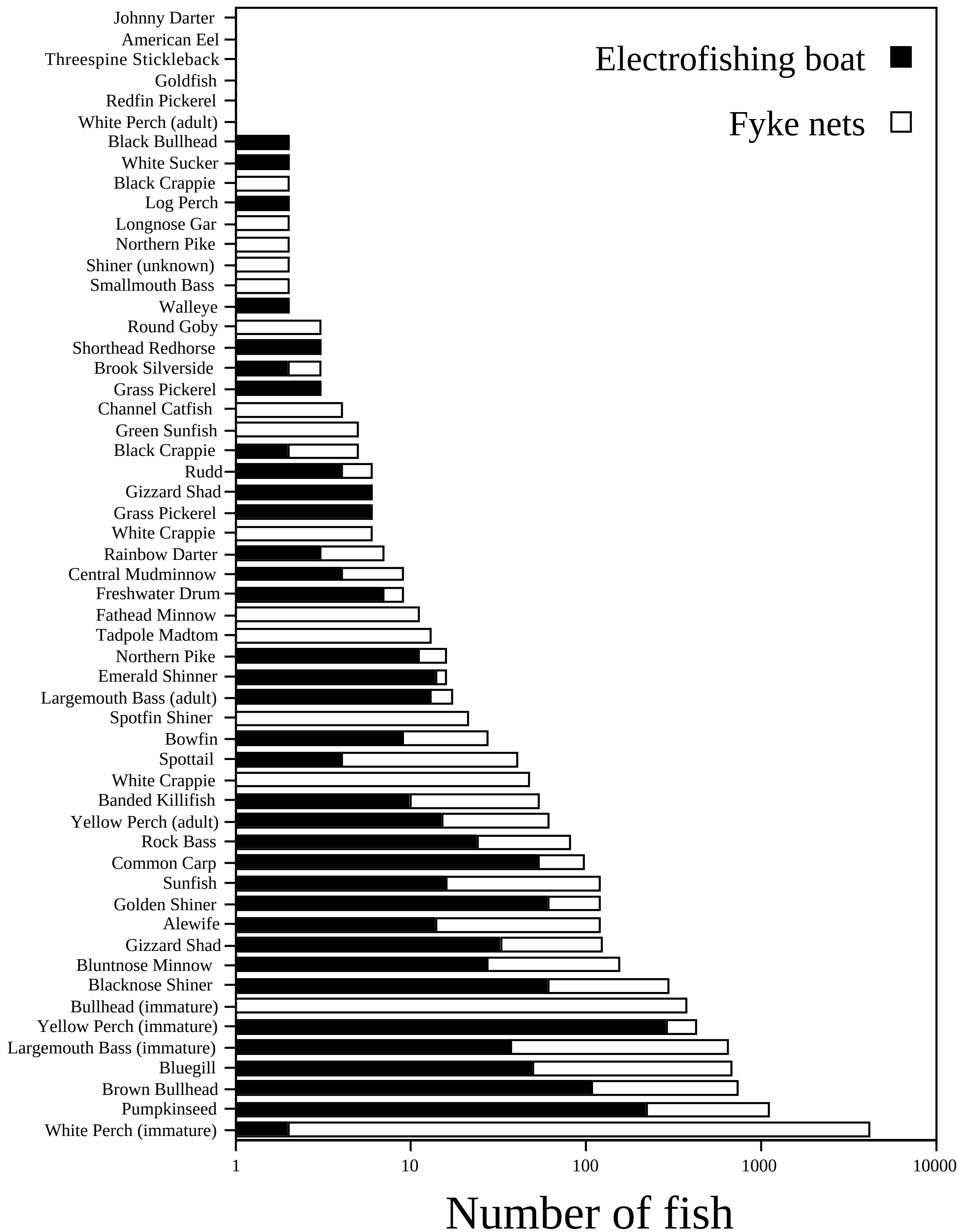
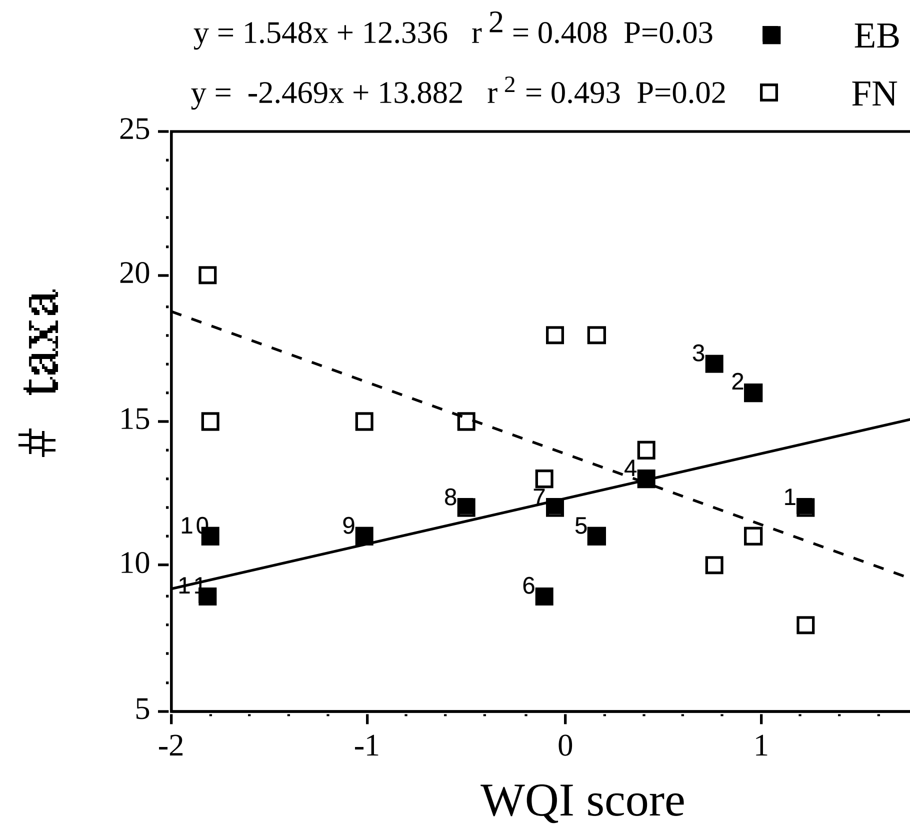

Johnny Darter
American Eel
Threespine Stickleback
Goldfish
Redfin Pickerel
White Perch (adult)
Black Bullhead
White Sucker
Black Crappie
Log Perch
Longnose Gar
Northern Pike
Shiner (unknown)
Smallmouth Bass
Walleye
Round Goby
Shorthead Redhorse
Brook Silverside
Grass Pickerel
Channel Catfish
Green Sunfish
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 (immature)
Yellow Perch (immature)
Largemouth Bass (immature)
Bluegill
Brown Bullhead
Pumpkinseed
White Perch (immature)
Electrofishing boat
Fyke nets
1
10
100
1000
10000
Number of fish

Figure 3.

a)


y = 1.548x + 12.336 r 2 = 0.408 P=0.03
EB
y = -2.469x + 13.882 r 2 = 0.493 P=0.02
FN
taxa
WQI score
25
20
15
10
5
-2
-1
0
1
2
1
2
3
4
5
6
7
8
9
10
11

b)

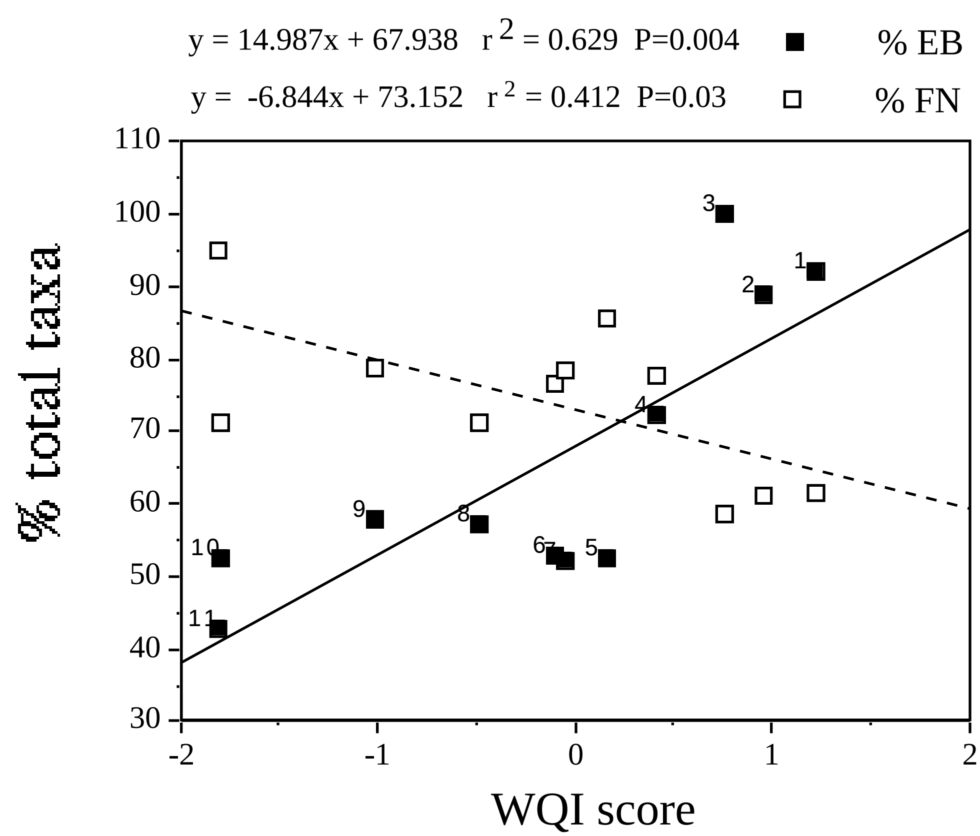

y = 14.987x + 67.938 r 2 = 0.629 P=0.004
% EB
y = -6.844x + 73.152 r 2 = 0.412 P=0.03
% FN
% total taxa
WQI score
110
100
90
80
70
60
50
40
30
-2
-1
0
1
2
1
2
3
4
5
6
7
8
9
10
11

Figure 4.

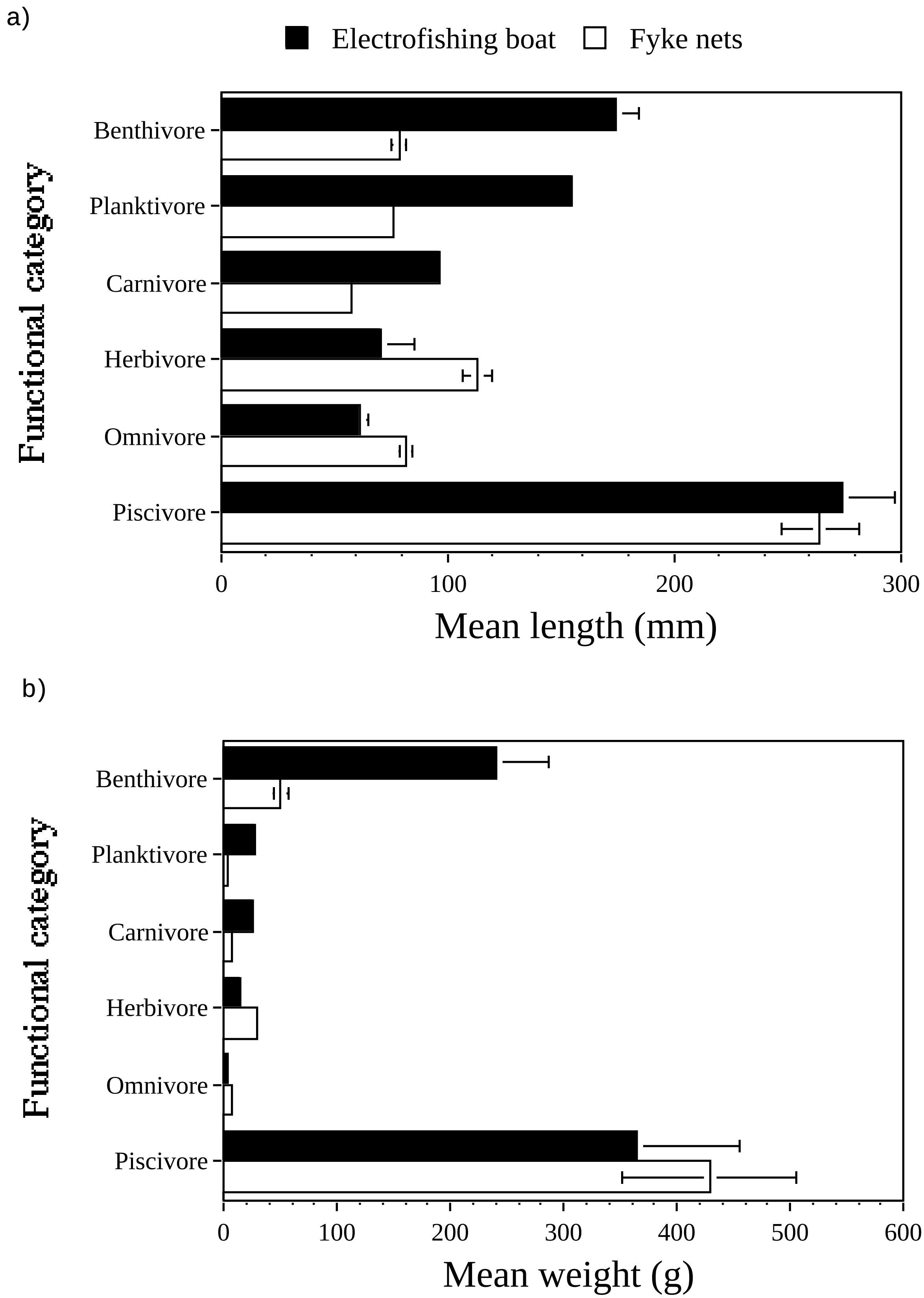
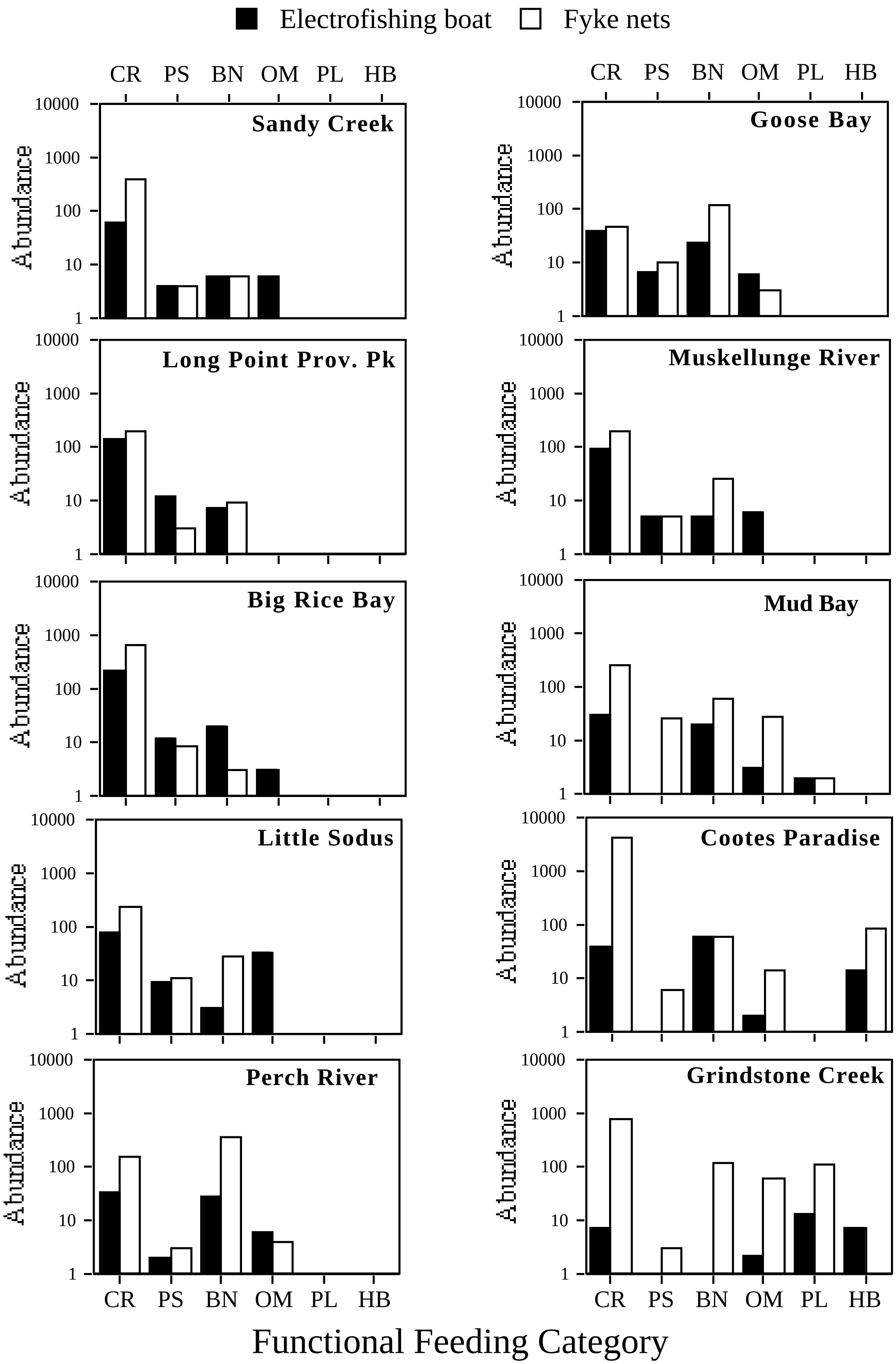

a)
Electrofishing boat
Fyke nets
Functional category
Benthivore
Planktivore
Carnivore
Herbivore
Omnivore
Piscivore
0
100
200
300
Mean length (mm)
b)
Functional category
Benthivore
Planktivore
Carnivore
Herbivore
Omnivore
Piscivore
0
100
200
300
400
500
600
Mean weight (g)

Figure 5.



Electrofishing boat
Fyke nets
CR PS BN OM PL HB
Abundance
10000
1000
100
10
1
Sandy Creek
Goose Bay
Long Point Prov. Pk
Muskellunge River
Big Rice Bay
Mud Bay
Little Sodus
Cootes Paradise
Perch River
Grindstone Creek
Functional Feeding Category

Figure 6

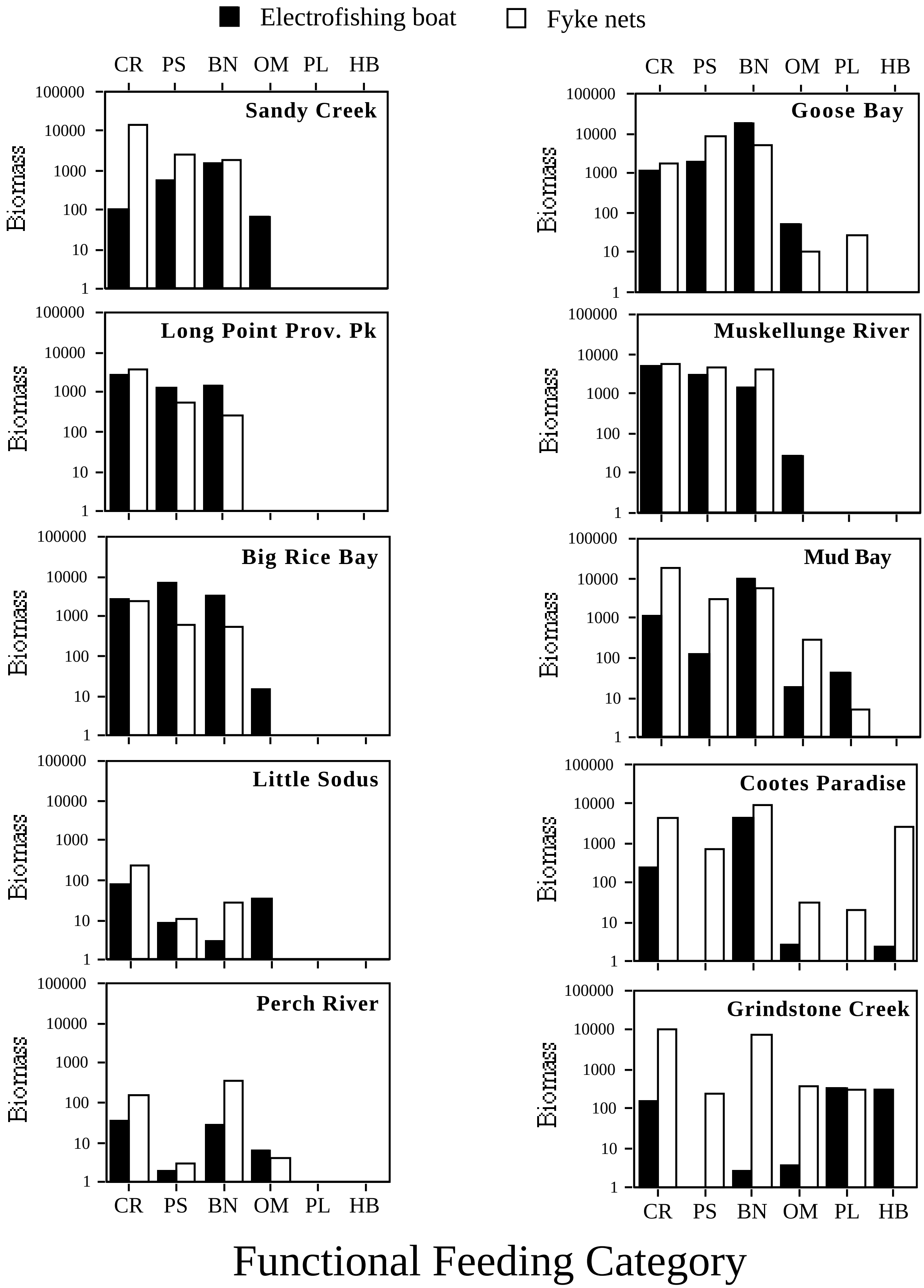

Electrofishing boat
Fyke nets
Sandy Creek
Goose Bay
Long Point Prov. Pk
Muskellunge River
Big Rice Bay
Mud Bay
Little Sodus
Cootes Paradise
Perch River
Grindstone Creek
Biomass
100000
10000
1000
100
10
1
CR PS BN OM PL HB
Functional Feeding Category

Figure 7

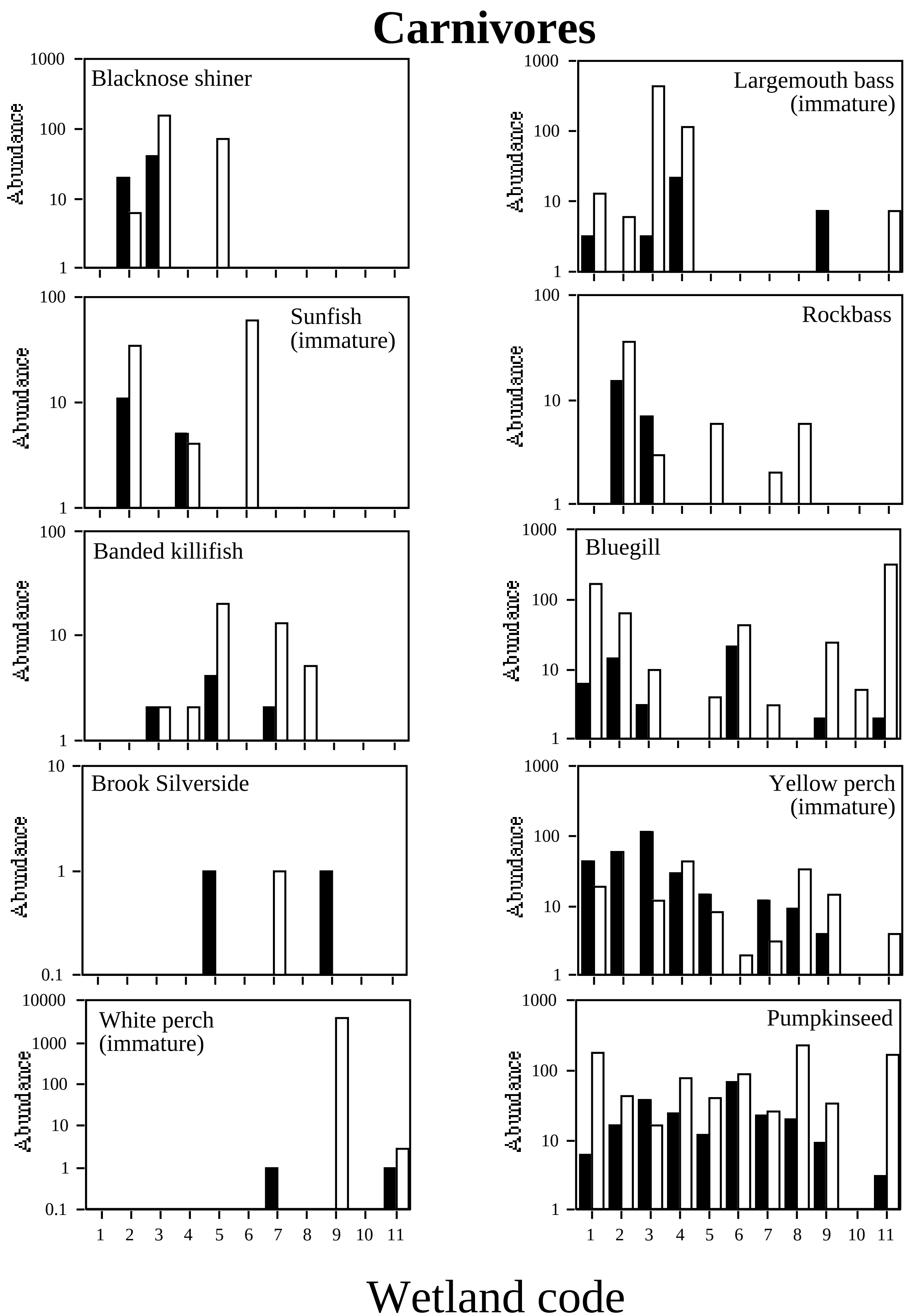
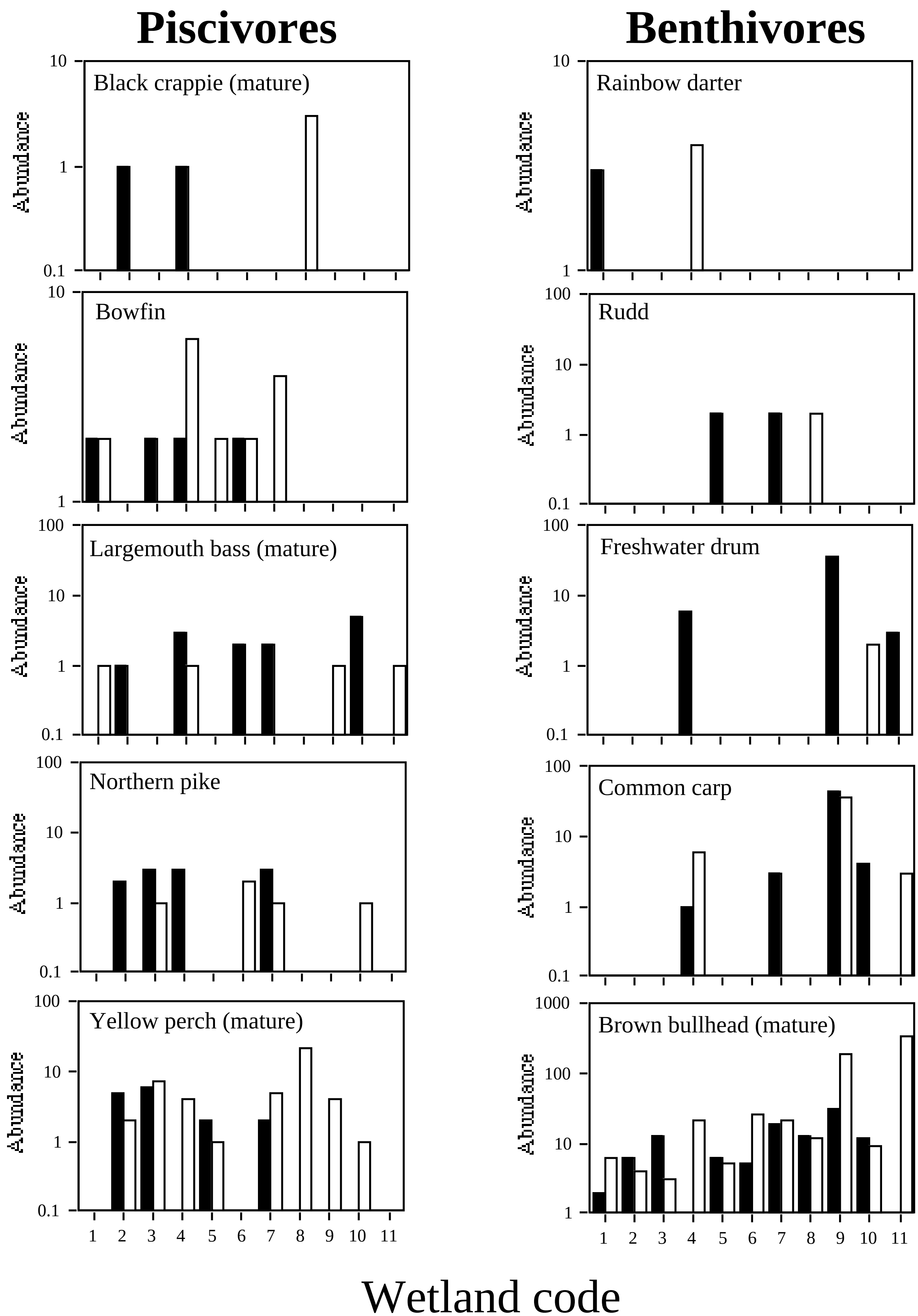
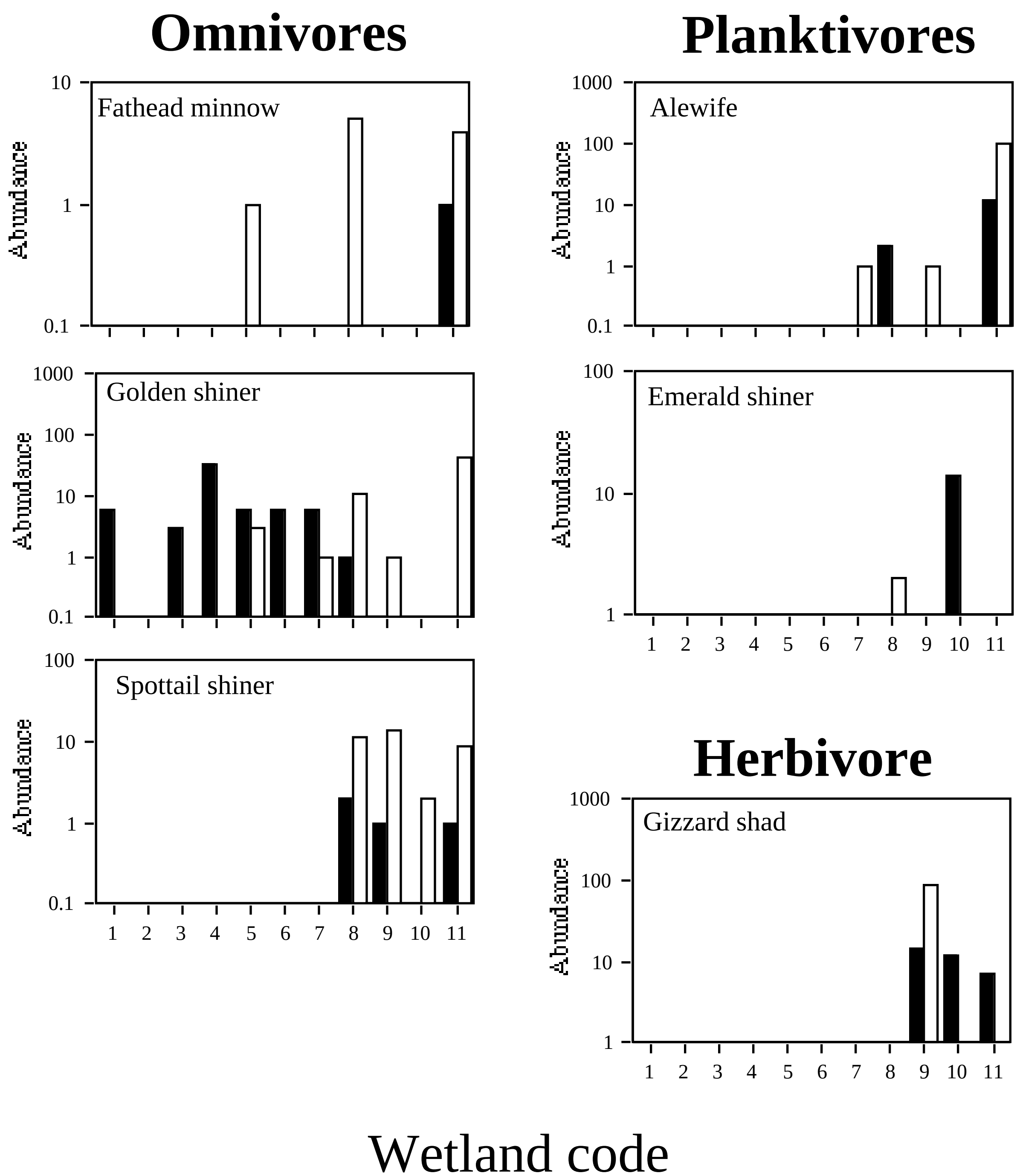

Carnivores
Blacknose shiner
Largemouth bass (immature)
Sunfish (immature)
Rockbass
Banded killifish
Bluegill
Brook Silverside
Yellow perch (immature)
White perch (immature)
Pumpkinseed
Abundance
1000
100
10
1
0.1
10000
1 2 3 4 5 6 7 8 9 10 11
Wetland code

Figure 8



Piscivores
Benthivores
Black crappie (mature)
Bowfin
Largemouth bass (mature)
Northern pike
Yellow perch (mature)
Rainbow darter
Rudd
Freshwater drum
Common carp
Brown bullhead (mature)
Abundance
0.1
1
10
100
1000
1 2 3 4 5 6 7 8 9 10 11
Wetland code

Figure 9


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

Appendix 3:

Lougheed, V.L. and Chow-Fraser, P. 2002. Development and use of a zooplankton index of wetland quality in the Laurentian Great Lakes basin. Ecol. Applic. 12: 474-486.

Appendix 4:

Becker, B.L, and Lusch, D.P. 2001. A Classification-Based Assessment of the Optimal Spatial and Spectral Resolution of Coastal Wetland Imagery, Completion report to GLFC, Michigan State University, Center For Remote Sensing and GISci, East Lansing, Michigan.

A Classification-Based Assessment of the Optimal Spatial and Spectral Resolution of Coastal Wetland Imagery

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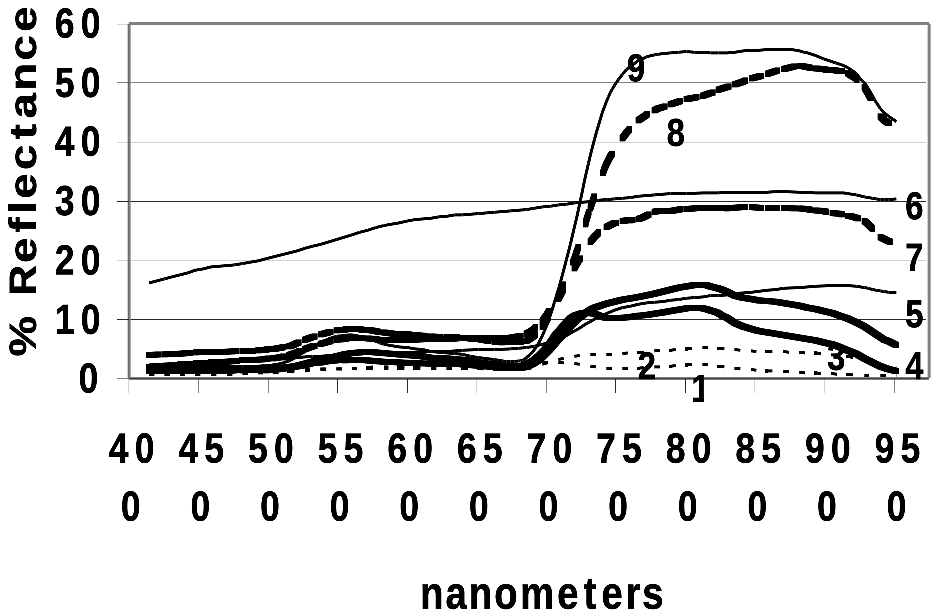
Great Lakes wetlands are increasingly being recognized as vital ecosystem components that provide valuable functions such as sediment retention, fish and wildlife habitat, and nutrient removal. Aerial photography has traditionally provided a cost-effective means to inventory and monitor coastal wetlands, but is limited by its broad spectral sensitivity and non-digital format. Airborne sensor advancements have now made the acquisition of digital imagery with high spatial and spectral resolution a reality.

In this investigation, two Lake Huron coastal wetlands, were selected (each from a distinct eco-region) over which, digital, airborne imagery (AISA or CASI-II) was acquired. The 1-meter images contain approximately twenty, 10-nanometer-wide spectral bands strategically located throughout the visible and near infrared. The 4-meter hyperspectral imagery contains 48 contiguous bands across the visible and short-wavelength near infrared. Extensive, in-situ reflectance spectra (using an SE-590 spectroradiometer manufactured by Spectron Engineering) and sub-meter GPS locations (using a Trimble Pro XRS with OmniStar RT_DGPS service enabled) were acquired for the dominant botanical and substrate classes field-delineated at each location. The spectra shown in Figure 1 represent a gradient of botanical signatures ranging from un-vegetated shallow water (1) to dense, herbaceous vegetation (9).



(1 = un-vegetated shallow water; 2 = sparse submergent vegetation in shallow water; 3= submergent vegetation in shallow water; 4 = emergent/submergent vegetation in shallow water; 5 = necrotic vegetation; 6 = gravel; 7, 8 and 9 = high biomass, emergent vegetation).

Figure 1. Nine Representative Spectra

The radiance from a target dominated by water produces a suppressed signature (e.g. curves 1, 2, 3 and 4). Water-dominated targets display a minimal amount of reflectance throughout the NIR region since water absorbs these wavelengths. Signature 2 represents a target that is predominantly water, but enough plant biomass was present (*Najas flexilis*) to increase NIR reflectance above that of a substrate-only, shallow-water target (1).

Eelgrass/musk-grass (*Vallisneria americana/Chara sp.*) beds (3) covered much of the shallow, inundated regions of Horseshoe Bay during the 2000 and 2001 growing seasons. This signature was representative of shallow areas predominantly covered by submergent vegetation. Indicative of this spectral class is the double-hump pattern between 700 and 850nm. Note that signature 4 displays a similar shape, although its double-hump is less suppressed in the NIR wavelengths due to the presence of both emergent and submergent biomass. Also note that reflectance in the green region (i.e. 560 nm) is slightly elevated due to the presence of more green biomass at or near the surface. An area composed of emergent *Sagittaria rigida* and submergent *Heteranthera dubia* was the source for signature 4.

Signatures 5 and 6 were displayed with similar line types because they exhibited relatively flat reflectance curves. Gravel (6) displays relatively constant reflectance across the visible and NIR wavelengths. Signature 5 is representative of areas dominated by necrotic, emergent vegetation (*Typha sp./Scirpus sp.*). Note that it too displays relatively constant reflectance throughout the visible and NIR region, although there is still evidence of plant biomass as indicated by the steep increase in NIR reflectance beginning just beyond 700nm.

Signatures 8 and 9 represent areas that are completely vegetated. Signature 9 exhibits the characteristic peak-and-valley reflectance pattern of high-biomass, green vegetation. This signature (*Impatians capensis*) represents one extreme of the botanical gradient, and is distinguished by the steepness and length of its red-edge reflectance (690-750nm). Signature 8 represents the dominant, tall-stature emergent vegetation found throughout the Great Lakes (*Typha sp./Scirpus sp*.). This signature is strikingly different from 5, (same genera, mostly non-living, brown biomass), indicative of its level of greenness. Signature 8 differs from signature 9 in three key areas: 1) flatter reflectance pattern across the visible domain; 2) lack of a pronounced red absorption feature (690-700nm); and 3) a more moderate transition into the high reflectance of the NIR. All of these traits are indicative of a significant spectral contribution from substrate, necrotic/senescent biomass, or both.

All of the in-situ spectral signatures were subjected to Principal Components and 2nd Derivative analyses in order to identify the most botanically explanative image bands. Seven non-overlapping bands (presented in order of relative importance: 685.5, 731.5, 916.7, 812.3, 514.9, 560.1 and 425.4 nanometers) were identified as the best performing bands with respect to classification performance. The removal of 41 of the original 48 bands only resulted in a 13.73% drop in classification performance. This result is important because obtaining 7-band, digital imagery, rather than 48-band, hyperspectral imagery, allows finer spatial resolutions to be acquired.

Three image-based investigations were implemented to evaluate the ability of three commonly-available classification algorithms (ISODATA, Spectral Angle Mapper and Maximum-Likelihood) to differentiate botanical regions-of-interest. Of the three algorithms tested, the Maximum-Likelihood classifier best differentiated the regions-of-interest (89% percent correctly classified) in both study sites. Note that, full-scene accuracy assessment was not conducted. Covariance-based PCA rotation consistently enhanced the performance of the Maximum-Likelihood classifier.

Two additional investigations were completed in order to assess classification changes associated with the *independent* manipulation of both spatial and spectral resolution. These best performing, 7 bands listed above were further manipulated to evaluate the impact of bandwidth (measured as FWHM). Somewhat surprisingly, bandwidth played only a minor role with respect to overall classification resiliency. Classifications generated from width-broadened imagery were equal to, or slightly better than, those generated from unaltered imagery (this was tested for FWHM. ranging from 5.7 nm to 35.4 nm). This conclusion only holds true until adjacent bands begin to overlap, at which point serious degradation of classification accuracy occurs.

A spatial resolution of 2 meters or less was determined to be the most appropriate in Great Lakes coastal wetland environments. Degrading the original 1-meter imagery to a 2-meter, functional pixel size caused approximately 30% of all pixels to be classified differently, compared to the 1-meter evaluation standard. At the 4-meter pixel size, nearly 58% of the pixels were classified differently. Assuming that the evaluation standard represents an accurate depiction of the botanical community, the majority of these pixels would represent classification errors. Virtually all of the pixels were classified differently as a result of increasing the functional pixel size to 8 meters. In fact, the negative Kappa

Coefficient calculated for the 8-meter classification indicates that it is less “accurate” than one generated randomly.

It was previously established (ITRES, 2000) that a spatial resolution of 5 meters or less is appropriate for wetland mapping strategies. This research indicates that even 2-meter pixels fail to adequately capture (23% and 30% change) the spatial mosaic of plant species found in Horseshoe Bay. Less heterogeneous wetland plant communities could probably be classified adequately with 4-meter imagery, but pixel sizes above this level are not recommended.

The results of this research indicate that the IKONOS band centers are less than optimal with respect to the differentiation of coastal wetland vegetation (August/September imagery). SAM classification results using four “optimized” bands out-performed classifications utilizing IKONOS band centers by nearly 10 percent. This spectral limitation in combination with the 4-meter spatial resolution of multispectral IKONOS imagery limits its applicability only to broad vegetative classifications where detailed spatial patterns are not the focus.

This research represents the first step in evaluating the effectiveness of applying high-resolution, narrow-band imagery to the detailed mapping of coastal wetlands in the Great Lakes region.

This research was also generously supported by Mr. Robb Macleod of the Midwestern Ducks Unlimited office who purchased the CASI-II imagery of Horse Shoe Bay. The full close-out report to the Great Lakes Fisheries Commission for this research segment is the dissertation by Brian L. Becker (2002), a copy of which has been supplied to the Commission.

References

ITRES Research Limited, 2000. Draft report – *Airborne Imaging Systems for Coastal Applications: An Update*. For Joint Airborne Lidar Bathymetry Technical Center of Expertise, Mobile, Alabama.

Becker, Brian L. 2002. A Classification-Based Assessment of the Optimal Spatial and Spectral Resolution of Coastal Wetland Imagery. Ph.D. Dissertation, Department of Geography, Major advisor: Dr. David P. Lusch. East Lansing, Michigan: Michigan State University. 232 p.