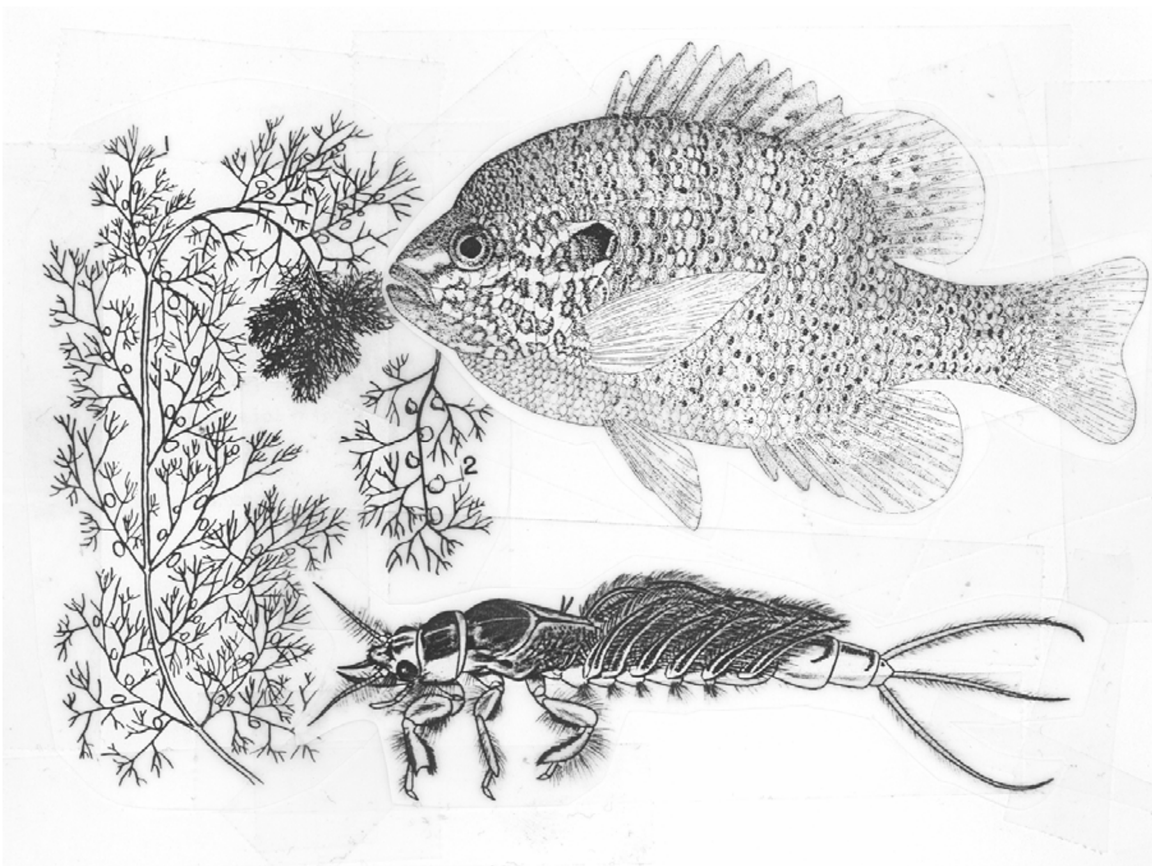
Biological Services Program & Division of Ecological Services
Bloomington Field Office
June 2004

Evaluation of Great Lakes Nearshore Coastal Wetlands: with Emphasis on Development of Watershed Biotic Indicators and Status



Fish and Wildlife Service

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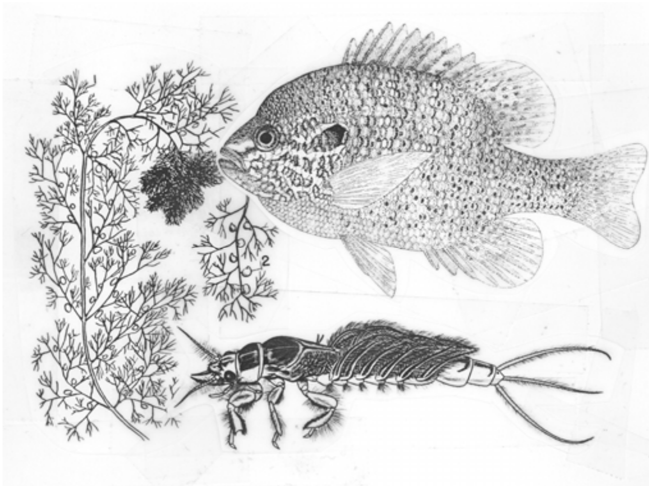

Fish and Wildlife Service
U.S. Department of the Interior

EXECUTIVE SUMMARY **15 June 2004**

Evaluation of Great Lakes Nearshore Coastal Wetlands: with Emphasis on Development of Watershed Biotic Indicators and Status

Interagency Agreement: from the U.S. Environmental Protection Agency/ORD for Region 5 awarded to the U.S. Department of the Interior/ Fish and Wildlife Service/ Bloomington Field Office, Thomas P. Simon, Principal Investigator

Project Period: January 2000 –June 2004

Background

Severe degradation and destruction of coastal wetlands along the Great Lakes has reduced what were once expansive wetlands to only a remnant. The loss of coastal wetlands has formed an immense void for aquatic life found in the transition between terrestrial and aquatic habitats. These habitats are important spawning, nursery, and feeding areas for a majority of Great Lakes' fish species during a portion of their life cycle. The U.S. Fish and Wildlife Service in collaboration with the U.S. Environmental Protection Agency and other Great Lake state, federal, and local agencies are completing a project that developed environmental indicators for assessment of remaining Great Lake coastal wetlands.

The fiscal year priorities for the International Joint Commission and the focus of the State of the Lakes Ecosystem Conference (SOLEC) included the establishment of environmental indicators for nearshore habitats and coastal wetlands of the Great Lakes. Development of reference conditions for the Great Lakes was an imposing task when confronted by the need to account for differences among Great Lakes, wetland types, and size dimensions. The Great Lake coastal wetlands are affected by filling and disturbance, urbanization, and are heavily impaired by a variety of land uses. Coastal wetlands possess substantial spatial heterogeneity and are vulnerable to contaminants from industrial sources, atmospheric deposition, non-point sources, erosion, and invasion by alien species.

Efforts to characterize Great Lakes biocriteria and establish regional reference conditions have generally concentrated on tributary waters within several states (Simon 1991; Lyons 1992; Thoma 1999) or have included limited numbers of sites (Wilcox et al. 1999; Burton et al. 1999). Greater portions of the Great Lakes nearshore and coastal wetlands have not been sampled, thus restricting a more global spatial depiction. Our effort was to develop sampling procedures, conduct pilot studies to determine spatial relevance of indicator development, and produce reference conditions for fish, plants, and macroinvertebrates. This effort is a preliminary study that has answered many questions regarding the quality of coastal wetlands throughout the Great Lakes. Our efforts will impact future data needs for the establishment of reference conditions, numerical biocriteria development, and State/Provincial-Regional monitoring and trend assessment.

Objectives

The main goal of this project was to establish reference conditions for Great Lakes coastal wetlands and develop and test multimetric indices of biological integrity for wetland plants, invertebrates, and fish assemblages. Specifically, objectives included the following: 1) establish consistent reference condition expectations and calibrate multimetric indices for fish assemblages in various wetland types of the four Great Lakes; 2) evaluate the differences in biological community expectations due to spatial heterogeneity of the coastal wetlands (among ecoregions, eco-reaches and latitudinal differences); 3) determine whether methods for biological integrity assessments in the Great Lakes can be applied similarly across political boundaries; 4) assess the current condition of biological integrity of the nearshore coastal resources of the Great Lakes; and 5) refine and test indices of biological integrity for fish, aquatic plants, and aquatic invertebrates in specific Great Lakes coastal wetland type, e.g. drowned river mouth wetlands in Lake Michigan.

Approach

The design strategy used was a tessellated, random-stratified sampling design that was used to obtain an unbiased selection of wetlands that could be used to estimate the range of wetland conditions in their “true” abundance. Fish assemblages within all the Great Lakes and connecting channels were used as the primary indicator to provide an unbiased validation of pending reference conditions for fish, wetland plants, and invertebrates. Site selection began with the National Wetland Inventory (NWI) electronic layers for each Great Lake for the United States shoreline. In addition, State specific inventories for Wisconsin and Michigan provided information from all shorelines. The final site compilation included a review of most published literature and ground-truthed evaluation of specific sites. This point coverage included information from Herdendorf et al. (1981) and Albert and Chow-Fraser (1999). A large change from previous wetland efforts was to change the paradigm from recognizing each coastal wetland as unique, to grouping them based on hydrogeomorphic class. The three hydrogeomorphic classes described by Keough et al. (1999) were used to place remaining wetlands into the proper classification. In addition, review of maps and site visits added additional coastal wetlands to the data base beyond that found in the literature. These coastal wetlands were assigned unique numbers in the 700’s so that further wetland scientists would recognize the addition. The coastal wetland site data base was used for site selection using the USEPA-Corvallis Environmental Monitoring and Assessment Program (EMAP) algorithm. Sites selected for sampling in each Great Lake was selected based on a stratification of wetland type for each Great Lake and weighted equally based on wetlands size.

The project consisted of two parts, a methods comparison and a regional comparison. During 2000, a pilot project was initiated in Lake Michigan, which was used to develop sampling approaches for fish, macroinvertebrates, and aquatic plant assemblages. Comparative sampling at 20 drowned river mouth wetlands utilized existing methods. Fish methodology comparisons included fyke netting (Brazner 1997; Wilcox et al. 1999) and boat electrofishing (Simon 1998) within a 500 m sampling distance. Day and night electrofishing was conducted to determine differences in catch. Fyke nets were set so that 8 nets were positioned perpendicular to shore so that the lead ends extended into Lake Michigan. Side leads were positioned so that they formed a contiguous boundary parallel to shore. Nets were fished for 24-hr and included two small and two large pairs of fyke nets with differential experimental mesh sizes. Data were analyzed based on wetted stream width so that wetlands were placed into small, medium, and large wetland categories. No differences were seen in large and medium sized drowned river mouth wetland fish communities based on number of species or relative abundance. Either methods produced a similar catch for evaluation of coastal wetland fish assemblages. Significant differences were observed in small coastal wetlands. The electrofishing method captured significantly more species and more individuals than did the fyke nets. This was probably a result of the low water depths during 2000 and the inability to locate depths greater than 0.5 m in many small wetlands to set the small fyke nets. Macroinvertebrate comparative sampling included activity traps (Wilcox et al. 1999) and sweep net sampling (Burton et al.1999). Activity traps were positioned in the dominant habitat types so that two pair of traps were set for 24-hr. Sweep net sampling was conducted within the 500 m sampling reach and included 20 sampling efforts divided equally among the dominant habitat types as identified by Burton et al. (1999). Macroinvertebrate sampling showed that either method was equally adept at collecting a representative sample of the aquatic community, however, the two methods differed in the portions of the community sampled. The choice of sampling approaches is left to the discretion of the scientist and we have produced calibrations for both methods for Lake Michigan. Aquatic plant assemblage indicators included a quantitative transect approach (Albert et al. 1987) and a qualitative rapid approach (Simon et al. 2001). Transects were perpendicular to shore and a random number of quadrants were selected along the grid to evaluate plant assemblages. The two approaches each showed some advantages. Quantitative sampling methods produced information about specific quadrants that could be used for trend assessment to evaluate changes in community structure and function, while the qualitative approach provided a rapid screening tool. The qualitative method consistently found greater diversity of species than the quantitative method.

During 2001, an evaluation of the status of Great Lakes coastal wetlands focused on sampling in every Great Lake and connecting channel. Work was completed by 32 partners representing state, federal, and local agencies, academicians, and industrial sectors. Sites were selected using the tessellated, random-stratified design. Data from this phase of the project was used to develop multimetric indices of biological integrity for fish assemblages. Drowned river mouth wetlands in each of the Great Lakes and connecting channels were surveyed. Sampling included 62 wetlands in Lake Michigan, 45 in Lake Superior, 23 in Lake Huron, 19 in Lake Erie, 23 in the St. Clair-Detroit River system, and 13 in the Niagara River. An additional 475 collections representing over 150 sites in the Niagara and St. Lawrence Rivers and the Lake Ontario open lake embayments were provided by the New York Department of Environmental Conservation. This information was the impetus for developing an index for Lake Ontario, the Niagara and St. Lawrence Rivers. During the analysis of the data from each lake comparisons were made between existing multimetric indices including Lake Erie (Thoma 1999) and Lake Ontario (Minns et al. 1994). Lake Michigan coastal wetlands were compared to an index developed for southern Lake Michigan (Simon 1998). Fish assemblages showed significant differences between lakes, however, nonsignificant differences with respect to Ecoregion and wetland size. Only a few metrics required calibration to account for scale differences based on wetted width. Multimetric indices were developed for Lake Superior, Lake Michigan, Lake Huron, and the St. Clair system. Connecting channel indices were calibrated for the St. Clair and Detroit Rivers and the Niagara and St. Lawrence Rivers. The IBI for Lake Erie (Thoma 1999) was verified using drowned river mouth wetland data from the western and central basins. Additional papers describe our rationale and methods for sampling and basis for multimetric development.

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2 Identification and Classification of Coastal Wetlands of the Laurentian Great Lakes

Mary F. Moffett, Ronda L. Dufour, & Thomas P. Simon

CONTENTS

2.1 INTRODUCTION

HGM emphasis

A consistent classification scheme, collection and assessment methodology for biota associated with Great lakes coastal wetlands has not been developed. In addition, the biological status of wetlands within and across wetland classes has not been documented. Coastal wetlands are a vital link between the open lake and terrestrial communities (Bedford 1990; Wilcox 1995) providing, some part of the fish life cycle, important adult, spawning, or nursery habitat (Chubb and Liston 1986; Jude and Pappas 1992; Klarer and Millie 1992). Maynard and Wilcox (1997) reviewed the distribution and status of coastal wetlands, summarized wetland loss, the types of natural and human-induced stressors that affected these wetlands, and their significant biological features. Minc and Albert (1997) classified the Great Lakes coastal wetlands floristically along a regional geographic basis.

The purpose of this study is to describe the distribution of wetlands in the Laurentian Great Lakes and document the rationale and approach used for implementing a large-scale assessment project in coastal wetlands of the Great Lakes. This paper focuses on the application of the classification approach to coastal wetland assessment and describes the development and implementation of the approach across the entire Great Lakes. This paper includes a list of all of the Great Lake wetlands (including point coverage information for further assessment), a review of site selection criteria and stratification (providing a review of wetland classification selection), and identification of drowned river mouth coastal wetlands.

Our preliminary evaluation of the condition of Lake Michigan drowned river mouth wetlands addressed three hypotheses: 1) larger wetlands may indicate larger size rivers; 2) larger wetlands may be "least-impacted" because of less edge effect, and 3) larger wetlands would be fewer in number and much more difficult to assess, thus requiring a greater number of samples within the wetland (T.P. Simon, unpublished data).

2.1.1 DEFINITION OF COASTAL WETLANDS

For purposes of this study, coastal wetlands of the Great Lakes are defined as "…lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. Wetlands must have one or more of the following three attributes: 1) at least periodically, the land supports predominantly hydrophytes; 2) the substrate is predominantly undrained hydric soil; and 3) the substrate is non-soil and is saturated with water or covered by shallow water at some time during the growing season of the year. Wetlands may be considered to extend lakeward to the water depth of 2 m, using the historic low and high water levels or the greatest extent of wetland vegetation. Hydrologic connections with one of the Great Lakes may extend upstream along rivers since exchanges caused by seiches and longer-period lake-level fluctuations influence riverine wetlands. Wetlands under substantial hydrologic influence from Great Lakes waters may be considered coastal wetlands." This is a modification of Cowardin et al. (1979) that was presented by Keough and Griffin (1994).

2.1.2 RATIONALE FOR THE CLASSIFICATION OF COASTAL WETLANDS

2.2 METHODS

2.2.1 STUDY AREA

2.2.2 SURVEY DESIGN FOR COASTAL WETLANDS OF THE GREAT LAKES

2.4 RESULTS AND DISCUSSION

2.3.1 LAKE SUPERIOR

2.3.2 LAKE MICHIGAN

2.3.3 LAKE HURON

2.3.4 LAKE ST. CLAIR AND CONNECTING CHANNELS

2.3.5 LAKE ERIE

2.3.6 LAKE ONTARIO

2.4 CONCLUSIONS

ACKNOWLEDGEMENTS

REFERENCES

3 Effects of Ecoregion on the Structure and Function of Coastal Wetland Fish Assemblages: Lake Michigan during a Low Water Year

Thomas P. Simon, Paul M. Stewart, Ronda L. Dufour

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3.1 INTRODUCTION

Studies have used ecological regions, termed “ecoregions” as regional frameworks (Omernik 1987) to examine variability in biological assemblages as they relate to land form factors (Hawkes et al. 1986; Lyons 1989; Hughes et al. 1994). Ecological land classification is a process of delineating and classifying ecologically distinctive areas of the earth’s surface (Wilken 1986; Omernik 1995). These classifications serve as regional frameworks, which provide a valuable estimate of areas of homogenous resource types. These areas can be used to estimate water body conditions and determine the range of biological communities that occur among various types of water resources representative of an area. Thus, water bodies across an ecoregional gradient would be expected to look more similar to each other than to those in adjacent ecoregions.

Regional frameworks, such as ecoregions, are important factors for the determination of reference conditions for establishing biological criteria expectations (Hughes 1995; Simon 1999; Thoma 1999). Without regional boundaries, which suggest changes in land forms, biological assemblage structure would be assumed to be continuous and without differences across North America. However, changes in zoogeography are an adaptive function of these changing land form patterns (Strange 1999). Several studies have evaluated biological reference condition expectations within the Great Lakes basin, which have resulted in models for Lake Ontario (Minns et al. 1994), Lake Erie (Thoma 1999), and northern Lake Michigan, Superior, and Huron (Wilcox et al. 1999). In addition, basin expectations for Lake Michigan have been determined using regional frameworks in Wisconsin (Lyons 1989; 1992), Illinois (Hite and Bertrand 1989), Indiana (Simon 1991; Simon 1998; Simon and Stewart 1999; Simon et al. 2000), and Michigan (Seelbach and Wiley 1997). These regional frameworks are used to derive estimates of water resource integrity as a reflection of expected differences in biological indicators as a function of land form changes.

Coastal wetlands of the Great Lakes have not been evaluated with regard to patterns in species diversity across large-scale regional gradients. Previous studies have not assumed that regional difference, scale differences, or placement would influence biological assemblages in coastal wetlands in the Great Lakes (Minns et al. 1994; Wilcox et al. 1999; Thoma 1999). Generally, previous studies had drowned river mouth wetland reference sites scattered across multiple ecoregions. Limited studies have focused in select portions of the Great Lakes and have not evaluated patterns across larger scales, but rather have focused on human impacts, microhabitat differences, and exotic species introductions (Brazner 1997; Brazner and Beals 1997; Brazner et al. 1998).

For purposes of this study, a definition of coastal wetlands of the Great Lakes is a modification of Cowardin et al (1979) presented by Keough and Griffin (1992). Coastal wetlands are “lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. Wetlands must have one or more of the following three attributes: 1) at least periodically, the land supports predominantly hydrophytes; 2) the substrate is predominantly undrained hydric soil; and 3) the substrate is non-soil and is saturated with water or covered by shallow water at some time during the growing season of the year. Wetlands may be considered to extend lakeward to the water depth of 2 m, using the historic low and high water levels or the greatest extent of wetland vegetation. Hydrologic connections with one of the Great Lakes may extend upstream along rivers since exchanges caused by seiches and longer-period lake-level fluctuations influence riverine wetlands. Wetlands under substantial hydrologic influence from Great Lakes waters may be considered coastal wetlands."

This study evaluates ecoregional influences on fish assemblage structure in Lake Michigan coastal wetlands during a low-water year in order to assist in the calibration of biological multimetric indices using a reference condition approach (Simon 1999). Wilcox et al. (1999), during the formulation of their reference conditions for drowned river mouth wetlands during normal water years in Lake Michigan, suggested that fish assemblage structure would not change across ecoregions. We randomly selected a variety of drowned river mouth wetlands across the five Lake Michigan ecoregions to test their prediction during a low-water year. Our hypothesis was that fish assemblage structure and function would change in Lake Michigan drowned river mouth coastal wetlands among ecoregions.

3.2 METHODS

3.2.1 STUDY AREA

The study area includes drowned river mouth and flooded estuary wetlands in Lake Michigan (Fig. 3.1).   We chose to pilot this effort in Lake Michigan because it potentially included the greatest clinal effect among the Great Lakes, possessed the greatest diversity of ecoregions, and was entirely located within the United States. Keough et al. (1999) used a hydrogeomorphic classification model to develop a classification system for Great Lake coastal wetlands. Their hydrogeomorphic classification model is based on physical, hydrologic, biological, and chemical features. Wetlands are classified into open, drowned river, and protected coastal wetlands. Drowned river mouth and flooded estuaries are among the most common hydrogeomorphic wetland type in Lake Michigan (Herdendorf et al. 1981; Chow-Fraser and Albert 1998; U.S. Fish and Wildlife Service, unpublished data). The coastal wetlands of the Great Lakes have been extensively changed by settlement during the last 250 years resulting in an estimated 80% loss of wetlands. Lake Michigan drowned river mouth and flooded estuary wetlands number less than 200 wetlands (T.P. Simon, unpublished data). The greatest number remaining in Lake Michigan are concentrated in northern portions of the lake. The lake is surrounded by a variety of land uses ranging from urban and residential to some of the most pristine habitats that remain in the Great Lakes basin. Lake Michigan has five ecoregions that comprise the margins of the nearshore habitats (Omernik 1987). These ecoregions include the Central Corn Belt Plain (CCBP), Southeastern Wisconsin Till Plain (SEWTP), Central Hardwood Forest (CHF), Northern Lakes and Forest (NLF), and Southern Michigan-Northern Indiana Till Plain (SMNITP). These ecoregions show differences in land form, soils, potential natural vegetation, and land use (Table 3.1).

3.2.2 STUDY DESIGN

A random stratified probability design (Overton et al. 1991) was used to assess the fish assemblages of drowned river mouth wetlands in Lake Michigan. This study design included the random selection of 23 wetlands from among the 123 remaining wetlands in Lake Michigan. Great Lake coastal wetland drowned river mouth and flooded estuary sites were selected using a Geographic Information System (GIS) point coverage data layer compiled from published literature (Herdendorf et al. 1981; Minc 1997a, b; Albert et al. 1987, 1988, 1989; Minc and Albert 1998; Chow-Fraser and Albert 1998), topographic map review, expert peer review, and ground-truthed reconnaissance (Simon 2000). We were unable to use National Wetland

Inventory data layers to select coastal wetlands because of inconsistencies with how wetlands were classified along the various Great Lake shorelines (Simon et al., this volume).

After considerable review, a point coverage data layer was prepared that initially included all wetlands identified as coastal wetlands (U.S. Fish and Wildlife Service, unpublished data). This data layer was subsequently reviewed to include only those wetlands that were within the first contour level as each of the Great Lakes. This data coverage was provided to the U.S. Environmental Protection Agency, Western Ecology Division, Corvallis, OR, for site selection using the Environmental Monitoring and Assessment Program (EMAP) algorithm. Sites were selected using a random stratified design to ensure that five wetlands from each ecoregion in Lake Michigan were selected using the Reach File 3.0 data layer.

The final selection of sites was stratified by wetland size to ensure that we would have a mixture of wetland sizes representing small (< 40 ha), medium (40-400 ha), and large (>400 ha) drowned river mouth and flooded estuary wetlands for each of the Lake Michigan ecoregions. Since the majority of wetlands remaining in the Great Lakes are less than 40 ha, we weighted them so that were selected from each of the small and medium size classes and a single large one from each ecoregion. With the exception of the CCBP, all other ecoregions had sufficient small, medium, and large wetlands. Large and medium sized wetlands are rare in the CCBP ecoregion because of large-scale land use disturbance, so only a single drowned river mouth or flooded estuary wetland existed from the medium-sized category. Final sites selected were field reconnaissanced to ensure that sites had sufficient water for fish assemblages to occur, possessed wetland vegetation, and were under the direct physical influence of Lake Michigan. These randomly selected sites represented a variety of environmental conditions from severely degraded wetlands with limited wetland function to high quality "least-impacted" wetlands (Mitsch and Gosselink 1993).

3.2.3 SAMPLING EFFORT, DATA COLLECTION, AND REACH SELECTION

Fish assemblage structure and function in Lake Michigan was determined from daytime inventories conducted between July and August 2000 (Simon 2000). Representative community sampling was required to characterize sites and determine patterns in ecoregion effects based on fish assemblage dynamics. Twenty-three drowned river mouth and flooded estuary wetland sites (Table 3.2) were evaluated using two collection techniques and methods (Simon 2000). Representative samples were collected to document species diversity and relative abundance for each site (Hocutt et al. 1974). All habitats within a sample area were surveyed relative to their frequency of occurrence.

Drowned river mouth wetlands and flooded estuary wetlands included three size categories of wetlands that possessed wetted stream channel widths from 3.0 to 65 m. Longitudinal sampling distances were 35 times the average stream wetted width, with a minimum sampling distance of 150 m and a maximum distance of 500 m. The field crew leader chose representative coastal wetland habitat when arriving on-site. These reflected marsh-type habitat and were positioned so that a variety of habitat types could be sampled among emergent, submergent, and floating vegetation. In large wetland complexes, multiple sampling zones were included to capture site heterogeneity. In small wadeable wetlands (less than 10 m across) sampling was done using a Wisconsin battery backpack electroshocker using settings that optimized voltage and amperage consistent with the manufacturer's specifications in an upstream, serpentine manner. For stream widths greater than 10 m or those that were non-wadeable, we used a DC boat-mounted electrofishing unit consisting of a Smith Root 3,500-watt generator with a ball-mounted electrosphere serving as the anode. The boat electrofishing unit was fished with pulsed DC output and a bow-mounted electrosphere. Boat-mounted electrofishing collections were performed for a minimum of 1800 seconds by sampling parallel to shore for a maximum distance of 500 m. All habitats encountered were sampled including emergent, submergent, and floating vegetation, woody debris, shallow littoral areas, adjacent backwater, and depressions. Surf and emergent vegetation zones were sampled by dragging the electrosphere in the water along the shoreline and through the vegetation. Stunned fish were netted and placed into a live well for identification, enumeration, and batch weighing. Fish were inspected for gross external deformities, eroded fins, lesions, and tumors (DELT) and released. Smaller specimens and vouchers were preserved in 10% formalin and returned to the laboratory for processing and identification. Preserved specimens were identified in the laboratory using standard taxonomic references (Gerking 1955, Smith 1979, and Becker 1983).

3.2.4 DATA ANALYSIS

An evaluation of community differences between sites used a cluster analysis (Ward Method) on a matrix of similarity coefficients (Jaccard's percent similarity). These results were represented in a Bray-Curtis dendrogram based on the similarity of site fish community composition using Primer® software (Clarke and Warwick 1994). Non-metric Multi-Dimensional Scaling (NMDS) was also used to evaluate the structure of fish assemblages among ecoregions in Lake Michigan. NMDS is flexible in the choice of similarity indices and consequently allows a much greater range of analyses with respect to incorporating rare species. Unlike Principal Component Analysis (PCA) and canonical analysis (CA), NMDS does not assume linear and unimodal forms of species response. NMDS contains no explicit assumption about the form of the response and may therefore be the most appropriate technique to use (ter Braak 1987).

Cao et al. (2000) reviewed the objectives of using rare species in biological assessment studies and found that rare species often revealed subtle patterns that may not always be observed when using only abundant species. Since the purpose of this study was to evaluate large-scale patterns in Lake Michigan fish assemblages for determining ecoregional effects, rare species were included in our analysis. Cao et al. (2000) suggested that in order to use the information that rare species provides, a multivariate technique needs to weight them sufficiently. Cao et al. (2001) suggested that data transformations or standardization could increase the weight of rare species, as does the use of similarity measures. Presence/absence transformations are a means of weighting rare species equal to those that are abundant. This method is often used and in many cases improves interpretation of species assemblage ordinations.

3.3 RESULTS

3.3.1 SPECIES DISTRIBUTION AND RELATIVE ABUNDANCE

Fifty-nine species were present in drowned river mouth wetlands among the five Lake Michigan ecoregions (Table 3.3). The most diverse ecoregion was the SMNITP (35 species), followed by the SEWTP (33 species), CHF (32 species), NLF (27 species), and CCBP (17 species). Of the total species, only 13.6 percent were found among all ecoregions. Species with ubiquitous distributions included central mudminnow (*Umbra limi*), carp (*Cyprinus carpio*), spottail shiner (*Notropis hudsonius*), sand shiner *(N. lubidundus*), bluntnose minnow

(*Pimephales notatus*), pumpkinseed (*Lepomis gibbosus*), largemouth bass (*Micropterus salmoides*), and yellow perch (*Perca flavescens*). Species unique to the CCBP ecoregion included goldfish (*Carassius auratus*), striped bass hybrid *(Morone saxatilis x chrysops*), and black crappie (*Pomoxis nigromaculatus*), while northern brook lamprey (*Ichthyomyzon fossor*), rosyface shiner (*Notropis rubellus*), lake chubsucker *(Erimyzon sucetta*), silver redhorse (*Moxostoma anisurum*), and greater redhorse (*M. valenciennesi*) were unique to the SMNITP ecoregion. Least brook lamprey (*Lampetra appendix*), hornyhead chub (*Nocomis biguttatus*), longnose dace (*Rhinichthys cataractae*), and nine-spine stickleback (*Pungitius pungitius*) were only collected in the NLF ecoregion. Brown trout (*Salmo trutta*) and mimic shiner (*Notropis volucellus*) were the only unique species collected in the CHF, while northern redbelly dace (*Phoxinus eos*), channel catfish (*Ictalurus punctatus*), white perch *(Morone americana*), and walleye (*Stizostedion vitreum*) were collected only from the SEWTP ecoregion.

Species similarity between adjacent ecoregions were not very high (Table 3.4). The strongest associations were among the northern ecoregions with Jaccard's percent similarity between the SEWTP and CHF (0.537), CHF and NLF (0.564), and CHF and SMNITP (0.468) equaling almost half or more of the shared faunas. As clinal differences in ecoregions increased between north-south and east-west Lake Michigan, differences between group membership of species was observed. Rainbow trout (*Oncorhynchus mykiss*), black bullhead (*Ameiurus melas*), and mottled sculpin (*Cottus bairdi*) were collected from drowned river mouth wetlands in the northern ecoregions (NLF, SMNITP, and CHF), sculpin, and darters were present in the SMNITP and NLF ecoregions, while coolwater species were found in the CHF and SEWTP ecoregions. The assemblage structure of the CCBP ecoregion was reflective of warmwater fish assemblages that also had some overlap with fish assemblages from the SMNITP ecoregion.

Species found in the CCBP were either ubiquitously found in the remainder of Lake Michigan or were associated with species assemblages found in the southern portion of the SMNITP ecoregion. Three species, i.e., banded topminnow (*Fundulus diaphanus*), brook stickleback (*Culaea inconstans*), and blackside darter (*Percina maculata*), were collected from the NLF and CHF ecoregions.

3.3.2 Patterns in multivariate analyses

Cluster analysis showed that site cluster membership was determined to some extent by ecoregion (Fig. 3.2). The first cluster separation showed that Portage Creek retained very little semblance to drowned river mouth coastal wetlands in Lake Michigan. The other cluster was a combination of CCBP streams and SMNITP streams. Wetlands from the NLF and CHF clustered together as did streams from the SEWTP and large river sites from the SMNITP.

NMDS analysis showed that the NLF drowned river mouth wetlands grouped together in the bottom left, while CCBP wetlands grouped on the right (Fig. 3.3). Wetlands from the SEWTP and SMNITP were grouped in the center of the graph, while CHF wetlands grouped throughout the center of the graph.

3.4 DISCUSSION

3.4.1 Effects of Ecoregion based on patterns in multivariate analysis

The effect of ecoregion membership is an important consideration in establishing reference condition expections for multimetric index development (Hughes 1995; Simon 1999). The associations among adjacent ecoregions showed that fish assemblages among the SEWTP and CHF ecoregions were more similar than those between the SEWTP and CCBP (Fig. 3.4; Table 3.4), while the CHF was most similar to the NLF and SMNITP ecoregions. This may be explained because the CHF occurs on both the eastern and western shores of Lake Michigan. No other ecoregion occurs adjacent to so many other ecoregions as does the CHF. Even though they are not contiguous, the SMNITP and SEWTP (Jaccard's similarity index = 0.457) showed a close relationship between fish assemblages. The two ecoregions occupy the same general latitude in Lake Michigan but are located on opposite shores. This suggests that a latitudinal (clinal) gradient may exist for fish assemblages on both sides of Lake Michigan.

3.4.2 Effects of rare species

Cao et al. (2000) reviewed the problems of including rare species in multivariate analysis based on questions related to biological assessments. Community pattern analysis is always based on similarity matrices (Green 1980; Legendre and Legendre 1998), and the various multivariate techniques differ in how they reveal the potential data structure of the matrix. Similarity measures differ greatly in weighting abundant and rare species and the choice of similarity measures and data transformation can greatly influence the results of the analysis.

Faith and Norris (1989) applied Bray-Curtis measure-based hybrid multidimensional scaling (HMDS) to river benthic data. They observed that the analysis of rare taxa alone revealed significant correlations between the HMDS and many physical-chemical variables, particularly water quality variables, but many of these significant correlations disappeared after rare species were removed. Yant and Karr (1984) noted that the inclusion of less abundant and rare species improved the temporal stability of fish community descriptions in terms of ranked correlation coefficients among species, i.e., the procedure can reduce the background noise of potential ecological impact. Pusey et al. (1998) found that adding rare species by collecting larger samples gave a more reliable multivariate description of fish assemblage structure than excluding them by the use of small samples.

When a particular analysis involves large spatial scales, with strong natural environmental gradients or severe impacts, rare species may not be important and analyses based on abundant species may be sufficient (Webb et al. 1967; Austin and Greig-Smith 1968; Day et al. 1971). The distribution of abundant species appears to be the result of major environmental processes or gradients. However, the inclusion of rare species may assist in recognizing unique communities (Goodall 1954), the discontinuity of an environmental gradient, or outlier samples. Outliers can have significant effects on ordination analysis, but no side effects on cluster analysis (Belbin and McDonald 1993).

3.5 CONCLUSIONS

Development of biological criteria for the Great Lakes Coastal wetlands requires that the effects of site placement be considered for developing reference conditions. Reference condition development is based on a regional framework, which is necessary in the calibration of a multimetric index. A random stratified sampling design was used to evaluate the structure and function of fish assemblages in coastal wetlands of Lake Michigan during 2000. Lake Michigan possesses five ecoregions including the SMNITP, NLF, CHF, SEWTP, and CCBP. A cluster analysis based on fish assemblages collected from 23 riverine coastal wetlands, following Keough et al.'s hydrogeomorphic classification, showed patterns in site membership that reflected ecoregional and latitudinal differences. Salmonid fishes were present in the SMNITP and NLF ecoregions, while coolwater species were found in the CHF and SEWTP Ecoregions. The assemblage structure of the CCBP Ecoregion was reflective of warmwater fish assemblages that also had some intermediate relationships with the CHF and SEWTP. Calibration of reference conditions for low-water year assessment in Lake Michigan should consider assemblage structure and function based on ecoregions.

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Table 3.1.

Comparison of characteristics of the five ecoregions of Lake Michigan (based on Omernik 1987). CCBP = Central Corn Belt Plain, SEWTP = Southeastern Wisconsin Till Plain, CHF = Central Hardwood Forest, NLF = Northern Lakes and Forest, and SMNITP = Southern Michigan-Northern Indiana Till Plain.

Ecoregion	Land-surface form	Potential natural vegetation	Land use	Soils
Central Corn Belt Plain	Smooth plains	Mosiac of bluestem Prairie (bluestem, panic, Indiangrass) and oak/hickory	Cropland	Mollisols, Brunizem/Humic Gley (dark-colored soils developed under prairie Vegetation
Southeastern Wisconsin Till Plains	Irregular plains (10-50% covered by standing water	Maple/basswood, oak savanna (oak, bluestem), bluestem prairie (bluestem, panic, and indiangrass)	Cropland	Gray-Brown Podzolic, Udalfs
North Central Hardwood Forests	Irregular plains	Maple/basswood, northern hardwoods, (maple, birch, beech, hemlock)	Cropland with pasture, woodland, and forest	Gray-Brown Podzolic
Northern Lakes and	Smooth to irregular	Great Lakes spruce/fir,	Forest and woodland	Podzolic (Gray-Brown

Forests	plains, plains with hills, tablelands with considerable reflief	Great Lakes pine, northern hardwoods (maple, birch, fir hemlock)	mostly ungrazed	Podzolic, Podzol, and Brown Podzolic)
Southern Michigan/ Northern Indiana Till Plains	Irregular plains	Oak/hickory, beech/ maple	Cropland with pasture, woodland, and forest	Gray-Brown Podzolic

TABLE 3.2.

Wetland number, name, latitude and longitude, size (ha), and ecoregion membership for drowned river mouth wetlands sampled in Lake Michigan during 2000. Ecoregion Codes: NLF = Northern Lakes and Forest; NCHF = Northern Central Hardwood Forest; SMNITP = Southern Michigan Northern Indiana Till Plain; CCBP = Central Corn Belt Plain; SWTP = Southeastern Wisconsin Till Plain

Number	Wetland	Lattitude	Longitude	Area (ha)	Ecoregion
5	Carp Lake River	45.741	-84.833	11.7	NLF
56	Hog Island	45.74	-85.69	6.07	NLG
75	Arcadia Lake Wetland	44.489	-86.225	145	NCHF
80	Manistee River Wetland	44.258	-86.25	3706	NCHF
98	Bass Lake Wetland #2	43.811	-86.414	55	SMNITP
100	Pentwater River Wetland	43.758	-86.404	110	SMNITP
105	White River Wetland	43.45	-86.289	1579.7	SMNITP
113	Little Pigeon River	43.965	-86.215	17	SMNITP
114	Pigeon River Wetland	42.903	-86.182	36.4	SMNITP
129	Dunes Creek	41.65	-87.11	0.4	CCBP
167	Grand Calumet River Mouth Wetland	41.647	-87.558	2.8	CCBP
174	Dead River	42.443	-87.811	40.4	CCBP
191	Kewaunee River Wetland #2	44.475	-87.514	145.7	SWTP
253	Keyes Creek Wetland	44.831	-87.572	28.3	SWTP
258	Fox River	44.535	-88.017	12.1	SWTP
262	Dead Horse Bay Wetland #1	44.61	-88.02	8.1	NCHF
274	Little Tail Point	44.68	-88.	64.7	NCHF
283	Oconto River	44.883	-87.85	283.4	NCHF
299	Portage Creek	45 7	-87.083	526.3	NLF
305	Days River	45.883	-87.	23.4	NLF
405	Brevort Area Wetland	46.018	-85.033	202.4	NLF
524	East Twin River	44.158	-87.57	80.9	SWTP

TABLE 3.3.
Presence of wetland fish species among five Lake Michigan ecoregions. CCBP = Central Corn Belt Plain, SEWTP = Southeastern Wisconsin Till Plain, CHF = Central Hardwood Forest, SMNITP = Southern Michigan-Northern Indiana Till Plain, SEWTP = Southeastern Wisconsin Till Plain, and NLF = Northern Lakes and Forest.

Species	CCBP	CHF	SMNITP	SEWTP	NLF
Northern brook lamprey, *Ichthyomyzon fossor*			X		
American brook lamprey, *Lampetra appendix*					X
Bowfin, *Amia calva*	X	X	X		
Alewife, *Alosa psuedoharengus*		X		X	X
Gizzard shad, *Dorosoma cepedianum*			X	X	
Rainbow trout, *Oncorhynchus mykiss*		X	X		X
Brown trout, *Salmo trutta*		X			
Central mudminnow, *Umbra limi*	X	X	X	X	X
Grass pickerel, *Esox americanus*	X		X		
Northern pike, *E. lucius*	X	X			X
Goldfish, *Carassius auratus*	X				
Spotfin shiner, *Cyprinella spiloptera*		X	X	X	X
Carp, *Cyprinus carpio*	X	X	X	X	X
Common shiner, *Luxilus cornutus*			X	X	X
Hornyhead chub, *Nocomis biguttatus*					X
Golden shiner, *Notemigonus crysoleucas*	X	X	X	X	
Emerald shiner, *Notropis atherinoides*		X	X	X	
Spottail shiner, *N. hudsonius*	X	X	X	X	X
Sand shiner, *N. ludibundus*	X	X	X	X	X
Rosyface shiner, *N. rubellus*					
Mimic shiner, *N. volucellus*		X			
Northern redbelly dace, *Phoxinus eos*				X	

Species	CCBP	CHF	SMNITP	SEWTP	NLF
Bluntnose minnow, *Pimephales notatus*		X	X	X	X
Fathead minnow, *P. promelas*		X		X	
Longnose dace, *Rhinichthys cataractae*					X
Creek chub, *Semotilus atromaculatus*		X	X		X
Quillback, *Carpiodes cyprinus*		X		X	
Longnose sucker, *Catostomus catostomus*			X		
White sucker, *C. commersoni*		X	X	X	X
Lake chubsucker, *Erimyzon sucetta*			X		
Silver redhorse, *Moxostoma anisurum*			X		
Golden redhorse, *M. erythrurum*		X	X		
Black bullhead, *Ameiurus melas*		X		X	X
Yellow bullhead, *A. natalis*		X	X		
Brown bullhead, *A. nebulosus*			X	X	
Channel catfish, *Ictalurus punctatus*				X	
Tadpole madtom, *Noturus gyrinus*	X			X	
Banded topminnow, *Fundulus diaphanus*		X			X
Trout-perch, *Percopsis omiscomaycus*		X		X	
Brook stickleback, *Culaea inconstans*		X			X
Ninespine stickleback, *Pungitius pungitius*					X
White perch, *Morone americana*				X	
Striped bass hybrid, *Morone saxatilis x chrysops*	X				
Rock bass, *Ambloplites rupestris*		X	X	X	X
Green sunfish, *Lepomis cyanellus*	X	X	X	X	
Pumpkinseed, *L. gibbosus*	X	X	X	X	
Warmouth, *L. gulosus*		X	X	X	
Bluegill, *L. macrochirus*	X		X		
Smallmouth bass, *Micropterus dolomieu*		X	X	X	X
Largemouth bass, *M. salmoides*	X	X	X	X	X

Species	CCBP	CHF	SMNITP	SEWTP	NLF
Black crappie, *Pomoxis nigromaculatus*	X				
Least darter, *Etheostoma microperca*			X		
Johnny darter, *Etheostoma nigrum*		X	X	X	X
Yellow perch, *Perca flavescens*	X	X	X	X	X
Logperch, *Percina caprodes*		X		X	X
Blackside darter, *P. maculata*		X			X
Walleye, *Stizostedion vitreum*				X	
Banded sculpin, *Cottus bairdi*		X	X		X
Total Number of Species	17	32	35	33	27

TABLE 3.4.
Jaccard's similarity index scores among drowned river mouth wetlands in five ecoregions of Lake Michigan. CCBP = Central Corn Belt Plain, SEWTP = Southeastern Wisconsin Till Plain, CHF = Central Hardwood Forest, NLF = Northern Lakes and Forest, and SMNITP = Southern Michigan-Northern Indiana Till Plain.

	Jaccard's Percent Similarity				
Ecoregion	CCBP	SEWTP	CHF	NLF	SMNITP
CCBP	--				
SEWTP	0.263	--			
CHF	0.275	0.537	--		
NLF	0.189	0.372	0.564	--	
SMNITP	0.225	0.457	0.468	0.326	--

Figure Captions

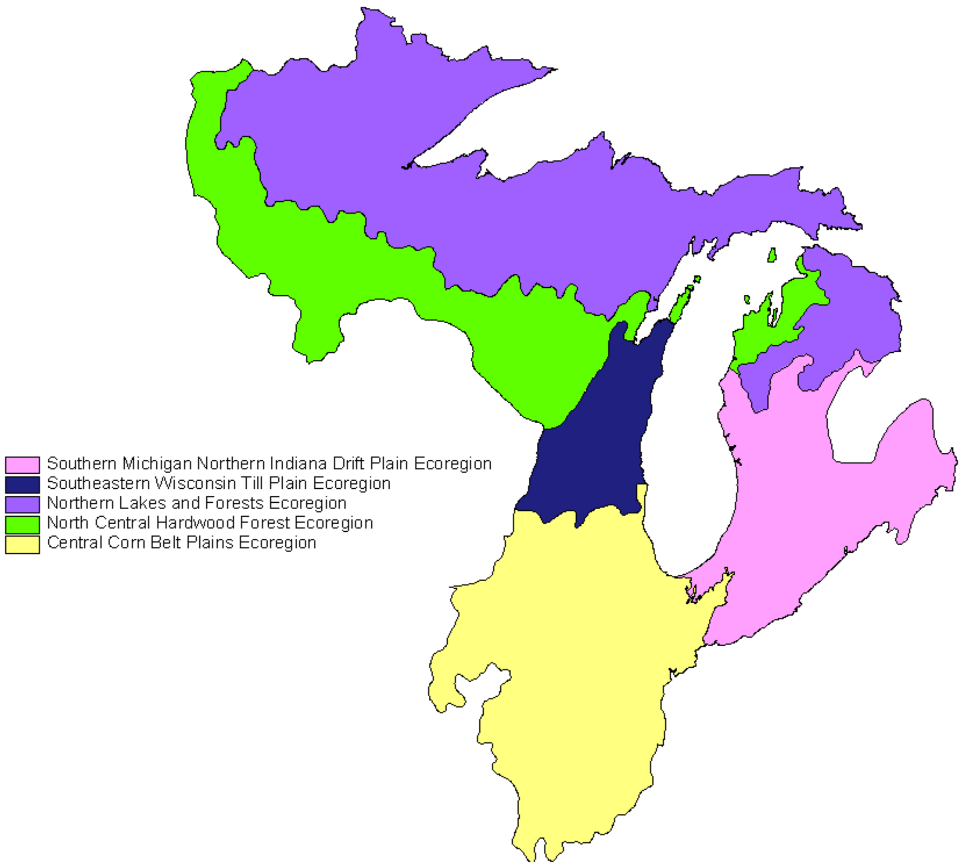
Fig. 3.1. Level III Ecoregions of Lake Michigan (based on Omernik 1987).

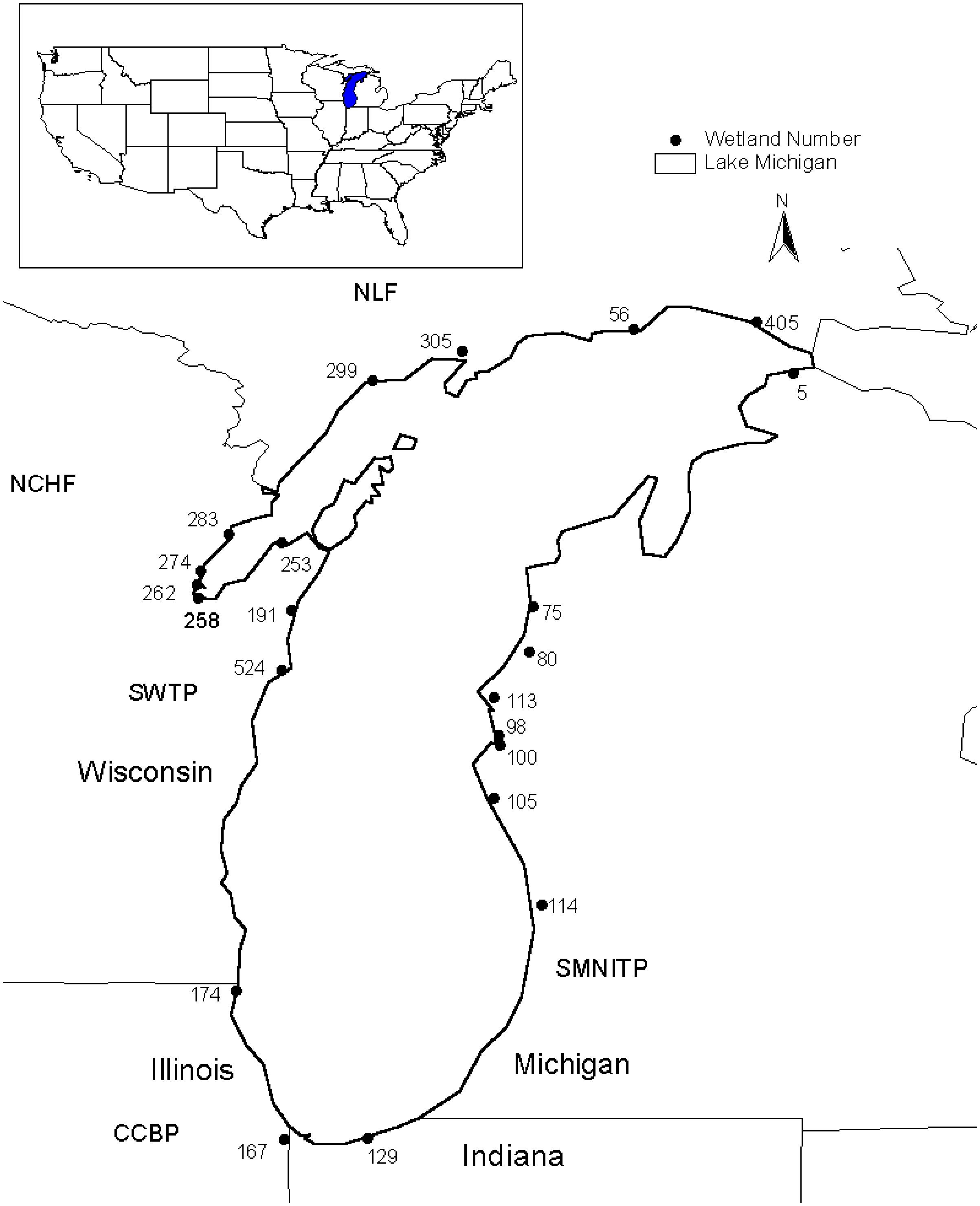
Fig. 3.2. Distribution of drowned river mouth coastal wetlands of the Great Lakes size-stratified for each of the five Lake Michigan ecoregions.

Fig. 3.3. Bray-Curtis Cluster analysis using fish assemblages collected from 23 drowned river mouth wetlands stratified by size and representing five Lake Michigan ecoregions.

Fig. 3.4 Non-dimensional multi-dimensional scaling analysis using fish assemblages collected from 23 drowned river mouth wetlands stratified by size for five Lake Michigan ecoregions.

Fig. 3.1

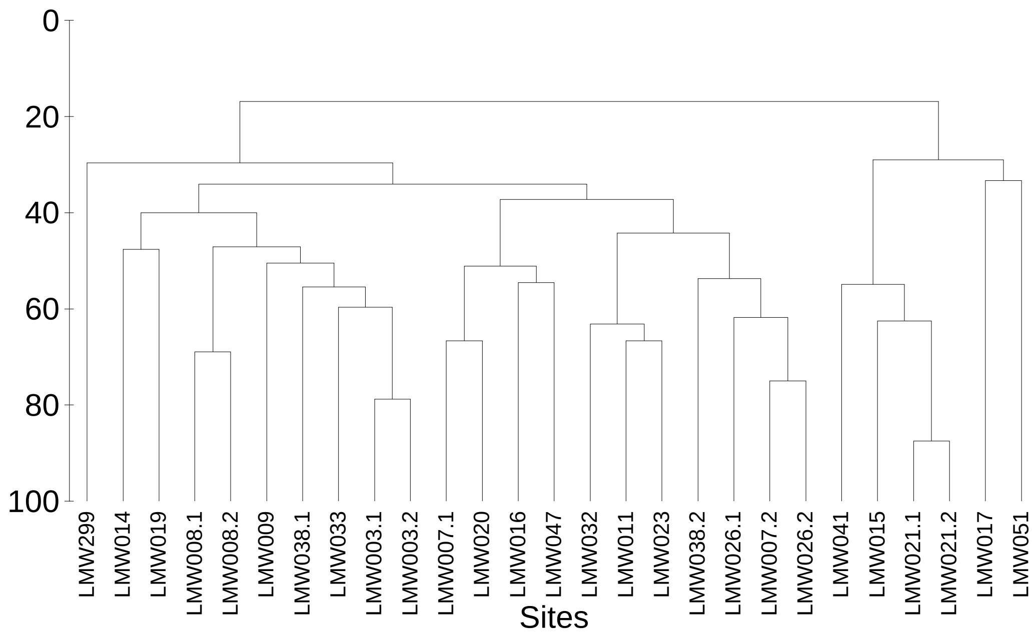

Southern Michigan Northern Indiana Drift Plain Ecoregion
Southeastern Wisconsin Till Plain Ecoregion
Northern Lakes and Forests Ecoregion
North Central Hardwood Forest Ecoregion
Central Corn Belt Plains Ecoregion



Wetland Number
Lake Michigan
N
NLF
NCHF
SWTP
Wisconsin
Illinois
CCBP
SMNITP
Michigan
Indiana
305
299
56
405
5
283
274
262
258
253
191
524
75
80
113
98
100
105
114
174
167
129

Fig. 3.2

Fig. 3.3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20
40
60
80
100
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LMW019
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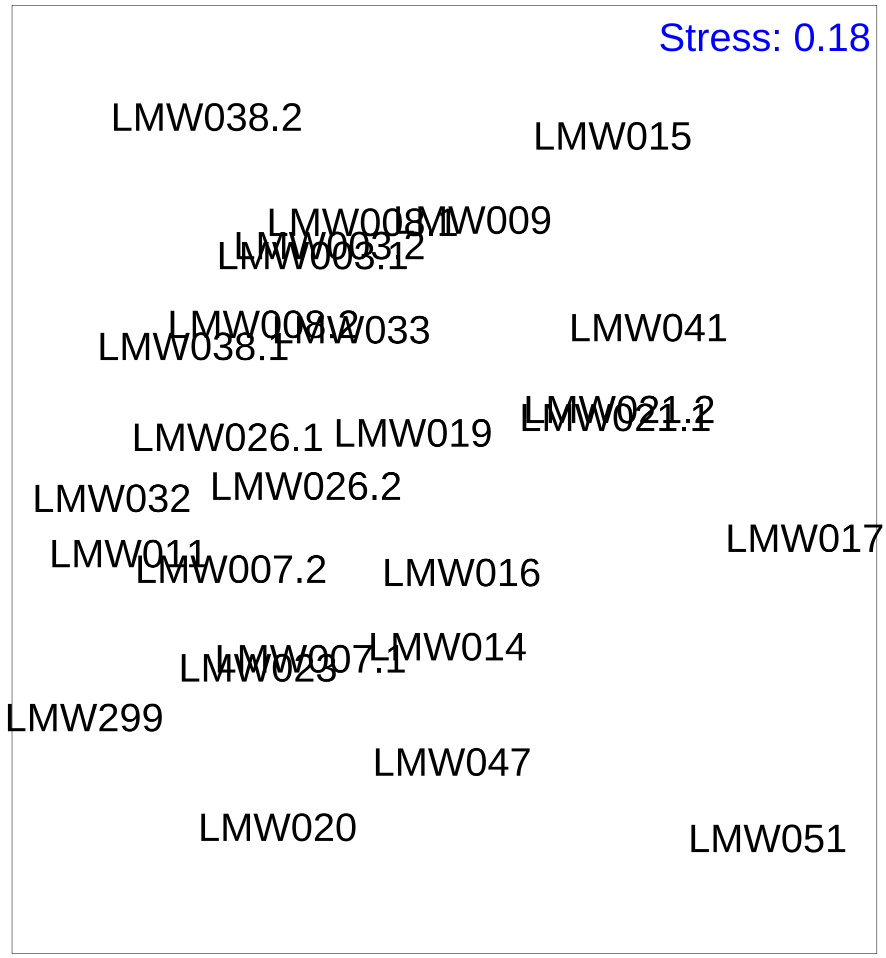
LMW003.1
LMW003.2
LMW007.1
LMW020
LMW016
LMW047
LMW032
LMW011
LMW023
LMW038.2
LMW026.1
LMW007.2
LMW026.2
LMW041
LMW015
LMW021.1
LMW021.2
LMW017
LMW051
Sites

Fig. 3.4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LMW015
LMW008.1
LMW009
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LMW008.2
LMW033
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LMW038.1
LMW021.2
LMW021.1
LMW026.1
LMW019
LMW032
LMW026.2
LMW017
LMW011
LMW007.2
LMW016
LMW007.1
LMW014
LMW023
LMW299
LMW047
LMW020
LMW051

Plants as Indicators for Lake Michigan's Great Lakes Coastal Riverine Wetland Health

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Abstract

In this paper we explore the development of plant-based indicators for Lake Michigan's coastal riverine wetlands, evaluating the correlation of aquatic macrophyte diversity and coverage to a land use gradient.   This study focuses on development of multiple plant metrics combined into an IBI score for an entire site, with sampling conducted along short transects at sites identified by USEPA. Floristic Quality Assessment (Herman et al., 1996) was evaluated as an alternative approach for comparing wetland sites.

Land cover data was calculated for the watershed of each site, using the 1990 National Land Cover Database. More localized land cover was also calculated for an area of two miles (3.33 km) in diameter surrounding the centroid of each sampling site.

Sixty-one metrics were evaluated, most for the separate marsh zones, submergent, emergent, and wet meadow. Of these potential metrics, 25 were included in the final IBI.

The averaged IBI scores for the sampling sites ranged from 3.96 to 1.96, while the sum of all metrics ranges from 99 to 49. IBI scores were compared to land use for each site. The relationship between residential and urban land use was strongest, with an R^2 of 0.71 for local land use and an R^2 of 0.26 for the entire watershed. The relationship between natural land use and IBI score was strong, while the relationship between agricultural land use and IBI score was weak.   Local land use is much more strongly related to plant response in coastal wetlands than watershed-wide land use.   Comparisons of FQI scores with IBI scores showed a relatively strong correlation (R^2 = 0.71), indicating that FQI might serve as a rapid alternative to multi-metric IBIs.

The strength of metric and IBI development would probably improve with further stratification of sampling sites. Dimensions of variability within the sampling sites that complicate their comparison include 1) stream size, 2) stream velocity, 3) geomorphic setting, and 4) sampling location.

Separate metrics for structural zones (wet meadow, emergent marsh, and submergent marsh) provide more ecological information than a single set of plant-based metrics for the entire wetland. Submergent plants were less effective metrics because of low water conditions.

Keywords: Laurentian Great Lakes, biological integrity, riverine wetlands, aquatic macrophytes

Introduction

For more than two decades, aquatic biologists have been seeking to develop biotic indicators of ecosystem health that display a sensitive and consistent response to specific anthropogenic stresses over a wide geographic region. Most of these studies have focused on fish and invertebrates to develop widely applicable measures of stream health (Karr, 1981; Karr and Chu, 1997). The search for faunal indicators has expanded into Great Lakes coastal wetlands, where fish and invertebrates remained major focal groups (Burton et al., 1999; Kashian and Burton, 2000). There has also been an increasing interest in recent years to explore the use of plants as indicators of aquatic ecosystem health in both inland and Great Lakes coastal wetlands (Stewart, 1995; Gernes and Helgen, 1999; Stewart et al., 1999; Simon et al., 2001).

To date, most of the published literature on plant IBIs for Great Lakes coastal wetlands has focused on southern Lake Michigan (Stewart et al., 1999; Simon et al., 2001), where high levels of wetland degradation and southern latitude does not represent the full range of natural floristic variation or the full range of wetland quality encountered within the Great Lakes region. There are, however, recent studies on wetlands in other portions of the Great Lakes, where levels of wetland degradation are not consistently as extreme as those found along southern Lake Michigan, including wetlands along Lake Huron's shoreline (Tepley, et al. 2003). In this study we evaluate plants, especially aquatic macrophytes, as potential indicators of Great Lakes riverine wetland health at representative sites along the entire Lake Michigan shoreline.

In this study we focus on the development of multiple plant metrics that are combined into an IBI score for an entire site. Sampling is conducted along short transects, with plant coverage values, water depth, and substrate characteristics described at each randomly located sampling point. Sampling was conducted at sites identified by USEPA. USEPA also identified the sampling transects, which were used except where those transects could not be located.

Methods

Nineteen wetland sites were sampled near the mouths of streams entering Lake Michigan. Sites were identified using a probability-based random selection of Lake Michigan coastal wetlands to allow assessment of regional estimates of condition with associated measures of uncertainty (Detenbeck et al., in press, Simon et al., in press).   To achieve spatial balance across the region, a random tessellation stratified design with multidensity categories was used (Stevens and Olsen, 1999).

Plant sampling was conducted at 5 sampling points along short transects, with sampling conducted in each of the marsh zones: submergent, emergent, and wet meadow. Points along the emergent and wet meadow zones were typically 25 meters apart, unless the small size of the wetland required shorting the distance between sampling points. The exact location of the sampling quadrat was randomly selected. Coverage values were estimated for each plant species present within the 0.5 m^2 sampling frame. Water depth, organic sediment depth, mineral soil texture, and sechi readings were taken at each sampling point. For the submergent zone within the stream, vegetation and physical parameters were sampled at one meter from each bank, in the center of the stream, and midway between the stream center and the near-bank sampling points, providing 5 submergent sampling points per transect. Sechi depths were taken only when the bottom was not visible. The cross sectional area of the sampled streams was also approximated, multiplying the stream width by the maximum stream depth, where maximum stream depth was from the submergent transect samples.

Metric and IBI development. Mean coverage values were calculated for each plant species by marsh zone. These mean values were the basis for identifying and calculating potential metrics. The scores for all sites were calculated to evaluate breaking points for all potential metrics, with roughly the 1/3 lowest scores being scored 1, the middle 1/3 being scored 3, and the upper 1/3 scored 5 (See Figures 1-3).   Potential metrics were eliminated if they did not provide information that could divide the wetlands into three quality classes.

Floristic Quality Assessment. Floristic Quality Assessment (Herman et al., 1996) was evaluated as an alternative approach to an IBI for comparing wetland sites. Floristic Quality Assessments, rather than identifying several different plant metrics, provides a Coefficient of Conservatism for each plant species, based on the breadth of habitats in which a species can grow. A species with narrow habitat typically has a high Coefficient of Conservatism, while weedy, widely distributed species are assigned a low coefficient. Coefficients of conservatism ranks range from a high score of 10 to a low score of zero for exotic (introduced) species.

Land cover. Land cover data was calculated by USEPA for the watershed of each site, using the 1990 National Land Cover Database (Vogelmann et al., 2001). A further, more localized calculation of the land cover was created by the lead author for an area of two miles in diameter surrounding the approximate centroid of each of the 19 sampling sites. For these more localized land cover measurements, only the water area of the streams were included; the Great Lakes area within the two-mile circle was removed from the calculations.

Stream cross-sectional area. Stream cross-sectional areas were calculated by multiplying the maximum depth and the stream width at the vegetation transect. For most streams multiple stream cross-sections were sampled, but only a single value was recorded for the cross-sectional area tabular comparison.

Results

Metrics and IBI scores. Sixty-one metrics were evaluated, most for the separate marsh zones, submergent, emergent, and wet meadow. Of these potential metrics, 25 were included in the final IBI (Table 1), while 36 were rejected (Table 2), either because they provided little information or because they were highly redundant with other metrics. Additional metrics based on aerial photo evaluation of the marsh rather than plant sampling were originally suggested, but these were eliminated, focusing metric development for this study only on field-collected data.

The averaged IBI scores for the riverine sampling sites ranged from a high score of 3.96 to a low score of 1.96 (Table 3). The sum of all metrics ranges from a high of 99 to a low of 49.

Metrics were calculated for individual zones due to differences in land use impact on each zone. By far the most actively managed zone of coastal marshes is the wet meadow zone, where agricultural management, drainage, and other activities have been initiated in the past and continue to be initiated.

The submergent zone has also been heavily impacted, either by dredging or by upstream alterations, such as airports, golf courses, and sewage treatment plants. The direct impact of these remote activities is more difficult to evaluate than the direct activities.

IBI scores were compared to land use for each site (Figures 4-7). These comparisons included comparison of IBI scores to land use, both for the entire watershed (Figures 4 and 5) and for an area within a mile radius of the site (Figures 6 and 7); we called the latter *local* land use. IBI scores were compared to the percent of agricultural, percent of urban and residential, and percent of natural habitat. The highest R^2 values were for local land use, with maximum R^2 values of 0.71 (Figure 6), while for the entire watershed, the maximum R^2 value was 0.26 (Figures 4). These R^2 values were much lower before the three smallest, and most anomalous sites, Dead Horse Bay, Dunes Creek, and Thomas Slough , were removed from the analysis. Land use comparisons to IBI scores were made to percent agriculture (not shown), percent residential and urban (Figures 4 and 6 , and to percent natural (Figures 5 and 7). The relationship between residential and urban land use was strongest, with an R^2 of 0.71 for local land use (Figure 6) and an R^2 of 0.26 for the entire watershed (Figure 4). The relationship between natural land use and the IBI score remained strong, 0.62 for the local land use (Figure 7) and 0.10 for watershed-wide land use (Figure 5). The relationship between agricultural land use and IBI score was weak for both local and watershed land use.

Floristic Quality Assessment (FQI). FQI scores were calculated to determine if they could provide the same information as that provided by the IBI score, with the assumption that the FQI sampling and analytic approach would be more rapid. FQI scores were calculated for both local and watershed land use. The relationship between local residential and urban land use and FQI scores were strongest, with an R^2 of 0.52, while the relationship between FQI and watershed-wide land use had an R^2 of only 0.17. Correlations between local natural land use and FQI scores remained relatively high, with an R^2 of 0.49, while for the watershed-wide land use it was only 0.02. Comparisons of FQI scores with IBI scores showed a relatively strong correlation, with an R^2 of 0.71, indicating that FQI might be viewed as a rapid alternative to development of multi-metric IBIs.

Stream cross-section. Stream cross-sections were also calculated for the 19 streams. The cross-sectional area of the 19 streams ranged from less than 0.4 m^2 to 105 m^2. For most of the streams there was nothing to indicated that the stream's natural cross-section had been altered. However, a small number of sites exhibited signs of significant alteration to the stream's basin. These highly altered sites included Dead Horse Bay and East Fox River, both occupying dredged channels. Thomas Slough also one of the smallest streams sampled, appeared to have been channelized near the upper sampling site, where the majority of plants were weedy exotics. Little Pigeon Creek's flow was restricted by narrow culverts at two points, resulting in reduced flow from the stream and a much broader channel (approximately 60 meters wide) than would have been expected from the creek's small drainage basin. Stream size relationships to IBI scores were not clear, but three of the smallest sites, Dead Horse Bay, Thomas Slough, and Dunes Creek had IBI scores that showed the weakest relationship to land use.

Discussion

Patterson and Whillans (1985) have identified three major classes of stresses to Great Lakes wetlands: (1) ecological structural breakdown, (2) hydrologic flow modification, and (3) water quality degradation. For each of these major classes, we briefly identify specific stresses affecting Great Lakes wetlands, and propose plant-based metrics.

Ecological structural breakdown. Physical modification and elimination of coastal wetlands resulted from a broad range of activities that hardened the shoreline or altered the sediments of a wetland; these activities including dredging, filling, diking, rip-rapping shoreline, and many others. The loss of coastal wetlands can be most readily documented by comparing early maps or aerial photos to recent maps and photos. Examination of aerial photos is a commonly used method to document changes in wetland extent resulting from human modification of wetlands, such as along western Lake Erie and Lake St. Clair (Jaworski and Raphael, 1976), Green Bay (Bosley, 1978, Harris et al., 1981), and northeastern Lake Michigan (Lyon et al., 1986). Initially several metrics based on wetland photo interpretation were proposed for this study, but these metrics were eliminated from the IBI and the focus was instead placed on field-collected data.

Field-based studies have identified several exotic plant species that respond rapidly to physically modified wetlands and thus are potentially good indicators of disturbance. The more wide-spread of these include *Phragmites australis*, *Phalaris arundinacea*, and *Lythrum salicaria* in the wet meadow or emergent marsh zones, along with *Myriophyllum spicatum* and *Potamogeton crispus* in the submergent marsh zone. These exotics often form dense monotypic stands that can exclude the native flora. With no natural predators, exotics often replace the native flora, but provide few benefits of the native flora to the fauna.

Some researchers have suggested that the number of exotic plant species at a wetland site is a good indicator of the level of site degradation (Gerne and Helgen, 1999; Stewart et al., 1999; Simon et al., 2001), but regional marsh analysis does not strongly support this assumption (Albert and Minc, 2003 manuscript). Instead, the number of exotic species typically covaries with wetland size. In only a few highly degraded wetlands were the number of exotic species high.

Some exotic species, such as dandelion (*Taraxicum officionale*) or common plantain (*Plantago major*), are not aggressive in coastal wetlands, and the presence of several of these might thus be of little ecological consequence. Their presence is less of a concern than the presence of more aggressive species, which tend to have higher coverage values.

Based on Great Lakes-wide wetland studies, the total coverage of exotic plants appears to more accurately assess the present condition of a wetland than the number of exotic species (Minc, 1997a). Wetlands within highly modified urban or industrial environments are often dominated by exotic species, even though the number of exotic species at these highly disturbed sites may be relatively low.

In this study, several metrics were based on the presence of exotic aquatic macrophyte species. Metrics based on either the number or coverage of native or exotic species were calculated by marsh zone, and include metrics 2, 3, 12, 13, 19, 20 (Table 1). Metrics 2, 12, and 19 focus on the number of exotics, while coverage metrics (3, 13, 20) provide us with important measures of how extensive the spread of exotics is within the wetland. Metrics based on either number of exotics or coverage of exotics alone were less effective at separating high and low quality sites than expected.

Overall diversity of native species has also been identified as an important measure of wetland structural intactness by many researchers, with the assumption that higher diversity occurs in structurally intact wetlands, while highly modified wetlands are assumed to have lower native diversity. In our dataset we developed metrics based both on native species richness (Table 3, metrics 1, 4, 6, 8, 10, 11, 14, 15, 17, 18, 21, 23) and native species coverage (Table 3, metrics 5, 7, 9, 16). We found that the positive relationship between high levels of native species diversity and low levels of human land use generally held. Overall diversity of the wet meadow zone (metric 1) and emergent zone (metric 10), were generally high for sites with the highest IBI scores and low for sites with low IBI scores (Table 3).

Hydrologic flow modification. Hydrologic flow modification is potentially an important change in riverine wetlands, where several types of alteration can occur, including upstream dam construction, channelization, reduction of channel cross-section due to culverts or bridges, diking of portions of the wetland, and hardening of stream shorelines. All of these flow modifications were present along the streams in this study.

Effective metrics of hydrologic flow modification were not identified in this study. Potential metrics for evaluating flow alterations were measurements of the amount (coverage) of submergent and floating aquatic plant cover (Table 2, rejected metrics 21, 25, 30, 34). The assumption was that low coverage of submergent plants indicated degraded sites. However these metrics did not prove effective. Many of our streams lacked submergent and floating vegetation within the stream channel. In some cases lack of vegetation was the result of rapid stream flow, rather than degradation resulting from hydrologic flow modification. Instead, lack of of submergent and floating vegetation was characteristic of many of the larger streams that had unstable bottom sediment. For example, both the Manistee and Brevoort rivers supported very low levels of submergent vegetation, but this absence could not be used as an effective metric. Streams that had been partially restricted by culverts or bridges, for example Little Pigeon Creek and Pigeon River, were among the most diverse sites for submergent and floating plants sampled; their high divesity of submergent and floating vegetation may have been partially the result of the modification of flow conditions, partially the result of moderate levels of nutrient enrichment from sewage plants.

The effect of channelization was also not consistent on submergent and floating vegetation, as demonstrated by four channelized streams, Thomas Slough, Dead Horse Bay, East Fox River, and Portage Creek; all but East Fox River had moderate diversity of submergents and floating species. East Fox had no submergent or floating species present. But even in these four sites channelization was compounded with other forms of management, including mining, sewage treatment, and nutrient enrichment from an airport runway.

Water quality degradation. Two forms of water quality degradation are known to effect aquatic plants within wetlands, nutrient enrichment and sedimentation. These will be discussed separately, but they often occur jointly and many indicator species are shared.

Nutrient enrichment. The effect of increased nutrient loading on aquatic plants is well documented from lakes and streams throughout the northern hemisphere (Kimbel, 1982; Niemeier and Hubert, 1986; Rorslett et al., 1986; Scheffer et al., 1992; Toivonen and Huttunen, 1995). Two common forms of nutrient enrichment along the Lake Michigan shoreline include introduction of sewage effluent or agricultural animal wastes, and the introduction of fine-textured mineral soils (siltation) and fertilizers from agricultural activities. The effects of these is often not easily separated.

Several species of Great Lakes aquatic macrophytes respond with increased growth when organic nutrients are added to wetlands, including common submergent species such as *Myriophyllum spicatum*, *Potamogeton crispus*, *Potamogeton pectinatus*, *Elodea canadensis*, and *Ceratophyllum demersum* (Kimbel, 1982; Rorslett et al., 1986; Scheffer et al., 1992; Toivonen and Huttunen, 1995), and emergents like *Typha* spp. and *Phragmites australis* (Niemeier and Hubert, 1986; Srivastava et al., 1995). There is evidence that nodding smartweed (*Polygonum lapathifolium*) responds similarly, producing large monocultures on recently exposed, nutrient rich sediments (Minc, 1997a; Minc, 1997b; Minc and Albert, 1998). Other species responding to nutrient enrichment include blue-green algae and several floating-leaved plant species, especially species of *Lemna*, *Spirodela*, and *Wolfia* (Tubea et al., 1981).

Although mean submergent cover values can reach 40% in the relatively clean waters of the upper

Great Lakes, nutrient-responding species are typically poorly represented, with mean coverage values of < 10% (Minc, 1997a). Within the upper Great Lakes, however, nutrient loading can locally impact wetland quality, resulting in increased coverage values for submergent aquatic plants. This local response was documented at Cedarville in northern Lake Huron, an area with low levels of residential development. At Cedarville, low levels of effluent from a sewage plant twice annually in May-June and September (Kashian and Burton, 2000), resulted in enhanced growth of nutrient-responsive submergent and floating plants, including *Elodea canadensis*, *Ceratophyllum demersum*, *Ranunculus longirostris*, *Myriophyllum spicatum*, *Lemna trisulca*, and *Lemna minor*, along with a significant algal bloom (based on sampling by author, 1998-2002).

Our metrics 24 and 25 (Table 1) are nutrient-loading metrics. Metric 24 is based on *Myriophyllum spicatum*, *Potamogeton crispus*, *Potamogeton pectinatus*, *Elodea canadensis*, and *Ceratophyllum demersum*, while metric 25 includes filamentous algae and *Lemna*, *Spirodela*, and *Wolfia*.
A nutrient-loading metric based on *Typha* spp. in the emergent and wet meadow zone proved ineffective for the sites sampled and was rejected (Table 2, metrics 16 and 17).

Fifteen of the nineteen sampling sites sampled in the present study contained nutrient-loving submergent plants. Of these sites, only two have relatively high levels of nutrient-loving plants; one of these, Thomas Slough, flows from a large forested wetland, not a typical source of nutrient enrichment. The other, Little Pigeon River, has a sewage-plant outflow near the sampling site, verifying the high nutrient metric. Several other sites with low levels of nutrient-loving plants have either golf courses or sewage plants nearby and it is unclear why the submergent flora does not show a response to these added nutrients. Of the four sites have no nutrient-loving plants; two of these were high nutrient streams with high levels of turbidity that limited submergent plant growth, while a third was a clear, fast-flowing northern stream with almost no submergent vegetation. The fourth was Portage Creek, a stream with a sewage-treatment plant and a major airport at its headwaters, typical sources for high levels of nutrients. While Portage Creek supported no nutrient-loving plants, 30 meters below the creek in Portage Bay there are extremely high coverages of nutrient-loving plants. Overall, the nutrient metric was less effective than expected.

Sedimentation. Heavy sedimentation from agricultural land use in Great Lakes watersheds is a major source of wetland degradation, especially in Green Bay. An important factor resulting in further turbidity is the presence of another exotic species, common carp (*Cyprinus carpio* L.), which resuspends fine sediments both when it breeds and feeds (Anderson, 1950; Chow-Fraser, 1998; Crivelli, 1983; Sager et al., 1998). High turbidity, with light penetration of only a few centimeters, is inadequate for most aquatic macrophytes and algae to photosynthesize and survive (Carter and Rybicki, 1985). The deposition of thick sediments can also result in loss of seed germination for both emergent and submergent aquatic plants (Barko et al., 1986). Submergent species in the turbidity tolerant category include *Potamogeton pectinatus, P. crispus, P. foliosus, P. pusillus, Ceratophyllum demersum, Elodea canadensis, Heteranthera dubia, Ranunculus longirostris,* and *Myriophyllum spicatum* (Stuckey, 1989; van Dijk and van Vierssen, 1991). In this study, metric 24 (Table 1) contains the major indicators of both high nutrient levels and high turbidity; an additional metric for high turbidity also including *Potamogeton. foliosus, P. pusillus, Heteranthera dubia*, and *Ranunculus longirostris* was not introduced. In our sample sites, the effectiveness of metric 24 was mixed. Only a small number of sites had turbid waters, Grand Calumet, Kewaunee, East Twin, and East Fox. All of these except Grand Calumet lacked submergent vegetation or had it limited to the extreme margins of the streams, where water depth was typically less than 20 cm. The Grand Calumet, among the most turbid streams sampled, supported *Potamogeton pectinatus*, a submergent plant recognized as tolerant of high turbidity, and *Lemna minor*, a floating species, therefore unaffected by turbidity.

One weakness of a turbidity metric is that a highly turbid stream may contain no submergent plant species. Using our present metric for nutrient enrichment and turbidity tolerant species, lack of submergent species in a highly turbid stream would result in a score of five, a false score. For this metric to be accurate in turbid streams, it must have a qualifier based on the turbidity reading, not on the plant coverage. A similar false reading could be expected in streams like Dunes Creek, where coverage values of floating duckweed were high enough to possibly compete with submergent vegetation; lack of submergents at this site resulted in a high score for this metric which was likely a false score as well. An effective metric for both nutrient loading and turbidity would be quite complex because of the combination of several factors that determine presence or absence of submergent and floating vegetation.

In streams with high levels of sediment load, sediment is often deposited near stream mouths within the wetlands. In the wet meadow zone, deposition of thick sediments over the surface favors a suite of aggressive colonizing species that include aggressive native annuals (*Polygonum lapathafolium*, *Bidens cernua*, *Impatiens capensis*, *Leersia orizoides*, and *Rorippa palustris*) and a host of exotics (particularly *Lythrum salicaria*, *Phragmites australis*, and *Phalaris arundinacea*). All of these species, including the native species respond not only to human-caused sedimentation, but also to exposures of organic-rich bottom sediments resulting from drops in water level associated with natural Great Lakes interannual water-level fluctuations. This rapid expansion onto recently exposed bottom sediment along the shoreline was seen during our sampling in 2000 as water levels dropped. A metric based on native annuals is not effective for any of the marsh zones is not a predictably accurate metric for any zone of Great Lakes wetlands because of these characteristic rapid water level drops.

Zonal metrics. In most Great Lakes coastal wetlands, distinct zones termed wet meadow, emergent, and submergent marsh can be identified in the field or on aerial photographs. While it is important to develop metrics by zone, it is also important to understand that water levels in Great Lakes coastal wetlands fluctuate significantly from year to year, resulting in the establishment of species from any zone to establish in another zone when conditions are right. When water levels are high, it is not uncommon for floating and submergent plants to grow in the flooded wet meadow zone. And, when the water levels drop considerably, emergent plants can establish on exposed mudflats that were previously dominated by submergent plants. For this study, metrics for plants from adjacent zones were evaluated (Table 2, metrics 1, 2, 4, 5, 6, 9, 11, 13, 14, 18, 22, 23, 26-29, 31, 32, 35, 36), but the low number of sites containing these “cross-over” species limited their value for lake-wide metric development. Had sampling been conducted a year later, more wet meadow species would likely have established in the dried down portions of the emergent and submergent marsh zones and might have served as effective metrics. During periods of rising water level, metrics for submergent and floating species in either the wet meadow or emergent zone may also prove effective.

Successional metrics. Successional groups of aquatic macrophyte species did not prove to be effective metrics. Annual plants have often viewed as indicative of disturbed wetland sites (Gernes and Helgen, 1999, Stewart et al., 1999), but as was previously discussed within the sedimentation section, low water levels in year 2000 resulted in an abundance of annuals on exposed bottom sediments at many sites, regardless of land management history. Native annuals that were especially common on these moist mudflats were *Bidens cernuus* and *Leersia oryzoides*. These and other native annuals do not necessarily indicate degradation, but rather water-level fluctuation, whether it be natural or human induced. For this reason, a metric based on dominance by annuals was not included in the study.

Structural metrics. The number of meaningful metrics utilizing structural species groups were restricted because of the low water conditions during the year 2000 sampling. In higher water conditions, submergent and floating plant species are often common in protected emergent marsh zone, and might be used as metrics in this zone for both for exotic and native species. In 2000, there were almost no emergent zones encountered with standing water, thus limiting the usefulness of emergent zone metrics that included submergent plant metrics. Instead, the moist substrates of the emergent marsh zone and portions of the submergent marsh were dominated by early successional emergent species.

Based on earlier sampling in other geomorphic coastal wetland types, some native submergent species, such as *Potamogeton* and *Utricularia* species, can be common in both the flooded wet meadow and emergent zones (Minc, 1997a; Minc, 1997b. These two submergent genera along with the genus *Myriophyllum*, were tested as potential metrics for the submergent and emergent zones (Table 2, metrics 20-30). The low levels of submergent species present in sampling transects during the 2000 sampling season resulted in none of these potentials metrics qualifying as effective. In future years, when water levels rise, development of these submergent metrics will likely prove to be critical for monitoring conditions in the submergent zone.

T**otal IBI scores.** Total scores for each site result from the summing of all metrics (Table 3). The

IBIs are relatively effective at ranking the sites relative to local land use (Figure 4 and Figure 5), i.e., within a mile radius of the site. The highest R^2 values, 0.71, are for the relationship between local residential and urban land use and IBI score (Figure 4), followed by the relationship between local overall human land use, with an R^2 of 0.62. Using watershed level land use data from 1990 satellite imagery is less effective for accurate ranking of the IBI scores relative to human land use at a site (Figure 6 and Figure 7). Using wateshed scale land use data, combined and urban land use still has the strongest relationship to IBI scores, with an R^2 of 0.26 (Figure 6), followed by total human land use, with an R^2 of 0.10 (Figure 7). Obviously local land use is much more strongly related to plant response in coastal wetlands than watershed-wide land use.

The effectiveness of the total IBI scores for evaluating levels of human land use is dependant on removing three outliers, Dune Creek, Dead Horse Bay, and Thomas Slough, from the analyses. Two of these sites, Dead Horse Bay and Dunes Creek, are clearly quite different from the other sites sampled. Dead Horse Bay is a very small drainage ditch and Dunes Creek is an isolated, stagnant swale over a mile from Lake Michigan. Thomas Slough, shows on both topographic maps and satellite imagery as natural shrub and wetland forest land, but both the submergent and emergent vegetation contain highest levels of exotic plants.

Developing a more effective riverine IBI. The strength of metric and IBI development would probably improve with further stratification of sampling sites. There are several dimensions of variability within the sampling sites that complicate their comparison. These are 1) stream size, 2) stream velocity, 3) geomorphic setting, and 4) sampling location.

Stream size. The streams included within this study range in size from 0.4 m^2 (2.4 m wide) to 105 m^2 (70 m wide) in cross-sectional area. The smallest stream is actually a drainage ditch, and the zones associated with this ditch are almost completely determined by Lake Michigan, with little influence from the ditch. The largest stream’s dynamics and water quality control the vegetation in its channel and along its banks to a much greater degree. Separating the nineteen sites into 3 or 4 distinct size classes would likely result in a better understanding of the vegetation relationships to other factors. To do this effectively would require sampling additional sites in each of these size classes.

Stream velocity. The velocity of the streams sampled also varies considerably. Several of the streams are slow and meandering, cutting through fine-textured till plain. These streams can be expected to support much more abundant submergent vegetation than fast flowing streams. Among the slow flowing streams with abundant submergent vegetation were Arcadia River, Keyes Creek, and Pigeon River. Other slow flowing streams were highly turbid due to agricultural land use, and for this reason did not support submergent vegetation, except in extremely shallow water along the banks. Two sites had almost no flow, Little Pigeon Creek and Dunes Creek, the former because small culverts converted the stream to conditions more pond-like, and the latter occupied a swale behind a sand dune and separated from the lake by more than 1.5 miles, resulting in similarly pond-like conditions with almost no influence from Lake Michigan. In contrast to these slow flowing streams, some of the streams were fast flowing with sandy bottoms and unstable, moving sediment; examples include Dead, Brevoort, White, Days, and Manistee rivers. While all of these streams were clear, moving sand in the channel probably limited vegetation establishment on the Brevoort, White, and Manistee; on these streams vegetation only occurs in very localized shallow zones along the stream margins. Submergent vegetation was rare enough in most of these streams to preclude development of meaningful submergent metrics.

Geomorphic setting. The original proposal for sampling Lake Michigan streams was to sample drowned river mouths. The streams and ditches sampled probably represent several different geomorphic settings, and therefore can be expected to have different physical characteristics and associated plant communities. Several sites are classic drowned river mouths, including Pigeon, Manistee, White, and Pentwater rivers and Little Pigeon Creek. Another set of sites are associated with dune and swale complexes, including Brevoort River, Dead River, and Dunes Creek. Brevoort River and Dead River are similar, except that very different snowmelt conditions resulting from latitude differences may result in much more extreme seasonal sediment movement in the Brevoort River channel, scouring away most submergent vegetation. Many of the other small streams do not fit into a shared geomorphic class..

Sampling location. The distance from the Great Lake varies greatly from site to site. This can greatly affect the level of water level connection to the Great Lakes. Sampling was relatively distant from the lake on the Kewanee River, East Twin River, and at many of the true drowned river mouths (Pentwater, White, and Manistee rivers). In contrast, sampling was very close to the lake at Days River and many of the smaller streams, providing very different dynamics of change.

Conclusions:

1. Metrics were developed by structural zone. Separate metrics for wet meadow, emergent marsh, and submergent marsh provide more ecological information than a single set of plant-based metrics for the entire wetland due to differences in land use stress by zone and because each zone has a different set of characteristic species.
2. Submergent plants were less effective metrics in 2000 because of low water conditions. For high water conditions in future, development of a set of metrics based more heavily on submergent plants will be necessary.
3. Correlation of IBI scores were stronger to the more localized 1-mile radius land use than to the watershed-scale land use. Refinement of the scale of land use data may improve the effectiveness of coastal wetland IBIs. For riverine systems, modifiers may be required for streams with high levels of agriculture or presence of sewage treatment plants or golf courses upstream from wetland sites.
4. Correlation of IBIs to land use was strengthened by eliminating outlier streams, those that were very small and very isolated from the lake. Further stratification of streams by geomorphic types and size classes may result in greatly improved IBIs.

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Table Captions:

Table 1. Plant Metrics.

Table 2. Rejected Plant Metrics.

Table 3. Summary of metrics and IBI scores for 19 Lake Michigan Great Lakes riverine wetlands.

Figure Captions:

Figure 1. Metric 1: Native Species Richness of the Wet Meadow – total number of native species recorded in sampling plots.

Figure 2. Metric 2: Exotic Species Richness of the Wet Meadow – total number of exotic species recorded in sampling plots.

Figure 3. Metric 19: Exotic Species Richness of the Submergent Marsh – total number of exotic species recorded in submergent marsh sampling plots.

Figure 4. Relationship of IBI score to % local (within two miles of sampling transect) residential and urban development. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

Figure 5. Relationship of IBI score to % local (within two miles of sampling transect) human development. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

Figure 6. Relationship of IBI score to % residential and urban development within the watershed. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

Figure 7. Relationship of IBI score to % total human development within the watershed. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

Table 1. Plant Metrics.

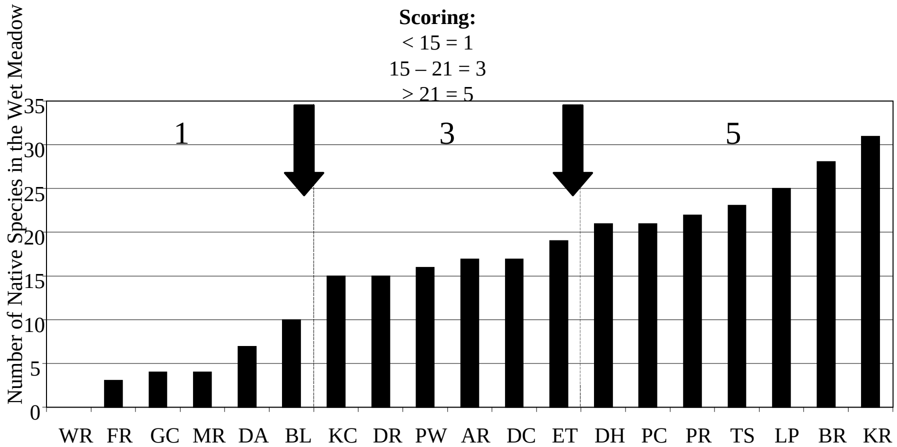
Metric #	Metric
1	Native Species Richness of the Wet Meadow
2	Exotic Species Richness of the Wet Meadow
3	Exotic Species Abundance in the Wet Meadow
4	Number of native sedge species in the wet meadow
5	Coverage of native sedge species in the wet meadow
6	Number of native grass species in the wet meadow
7	Coverage of native grass species in the wet meadow
8	Number of native *Juncus* species in the wet meadow
9	Coverage of native graminoid species in the wet meadow
10	Native species richness of the emergent marsh
11	Richness of native emergent species in the emergent marsh
12	Exotic species richness of the emergent marsh
13	Exotic species abundance in the emergent marsh
14	Number of native *Juncus* species in the emergent marsh
15	Number of native graminoid species in the emergent marsh
16	Coverage of native graminoid species in the emergent marsh
17	Native species richness of the submergent marsh
18	Submergent/floating native species richness of the submergent marsh
19	Exotic species richness of the submergent marsh
20	Exotic species abundance of the submergent marsh
21	Number of native *Potamogeton* species in the submergent marsh
22	Coverage of floating unattached species in the submergent marsh
23	Richness of native emergent species in submergent and emergent marsh
24	Nutrient loading: total coverage of nutrient-loving submergent species
25	Nutrient loading: total coverage of nutrient-loving floating species

Table 2. Rejected Plant Metrics.

Metric #	Metric
1-reject	Number of native sedge species in the submergent marsh
2-reject	Coverage of native sedge species in the submergent marsh
3-reject	Coverage of native sedge species in the emergent marsh
4-reject	Number of native sedge species in the submergent marsh
5-reject	Number of native grass species in the submergent marsh
6-reject	Coverage of native grass species in the submergent marsh
7-reject	Coverage of native grass species in the emergent marsh
8-reject	Number of native grass species in the emergent marsh
9-reject	Number of native *Juncus* species in the submergent marsh
10-reject	Number of native *Juncus* species in the wet meadow
11-reject	Coverage of native *Juncus* species in the submergent marsh
12-reject	Coverage of native *Juncus* species in the emergent marsh
13-reject	Number of native graminoid species in the submergent marsh
14-reject	Coverage of native graminoid species in the submergent marsh
15-reject	Number of native graminoid species in the wet meadow
16-reject	Coverage of *Typha* spp. in wet meadow
17-reject	Coverage of *Typha* spp. in emergent marsh
18-reject	Submergent/floating native species richness of the emergent marsh
19-reject	Number of floating unattached species in the submergent marsh
20-reject	Number of native *Utricularia* species in the submergent marsh
21-reject	Coverage of native *Utricularia* species in the submergent marsh
22-reject	Coverage of native *Utricularia* species in the emergent marsh
23-reject	Number of native *Utricularia* species in the emergent marsh
24-reject	Number of native *Myriophyllum* species in the submergent marsh
25-reject	Coverage of native *Myriophyllum* species in the submergent marsh
26-reject	Coverage of native *Myriophyllum* species in the emergent marsh
27-reject	Number of native *Myriophyllum* species in the emergent marsh
28-reject	Coverage of native *Potamogeton* species in the emergent marsh
29-reject	Number of native *Potamogeton* species in the emergent marsh
30-reject	Coverage of native *Potamogeton* species in the submergent marsh
31-reject	Total coverage of nutrient-loving submergent species in the emergent marsh
32-reject	Nutrient loading: total coverage of nutrient-loving floating species in the emergent marsh
33-reject	Coverage of floating unattached species as a % of total plant coverage in submergent marsh
34-reject	Coverage of floating unattached species as % of submerg. plant cover in submergent marsh
35-reject	Coverage of native floating unattached species in the emergent marsh
36-reject	Number of native floating unattached species in the emergent marsh

Figure1.

Metric 1: Native Species Richness of the Wet Meadow – total number of native species recorded in sam



AR = Arcadia River
BL = Bass Lake
BR = Brevort River
DA = Days River
DC = Dunes Creek
DH = Dead Horse EPA
ET = E Twin River
FR = Fox River
GC = Grand Calumet Il.
KC = Keyes Creek
KR = Kewaunee River
LP = Little Pigeon Creek
MR = Manistee River
PC = Portage Creek
PR = Pigeon River X
PW = Pentwater
TS = Thomas Slough
WR = White River

* Italics indicate
the zone was r

Figure 2.

Metric 2: Exotic Species Richness of the Wet Meadow – total number of exotic species recorded in sam

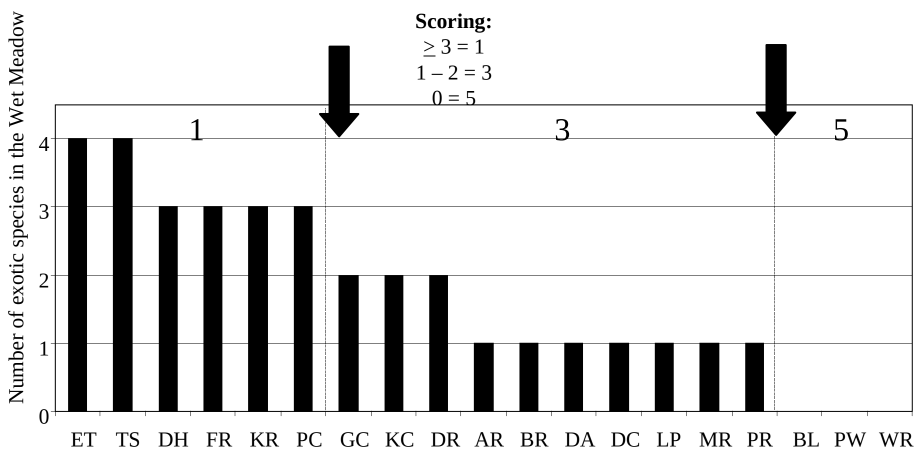


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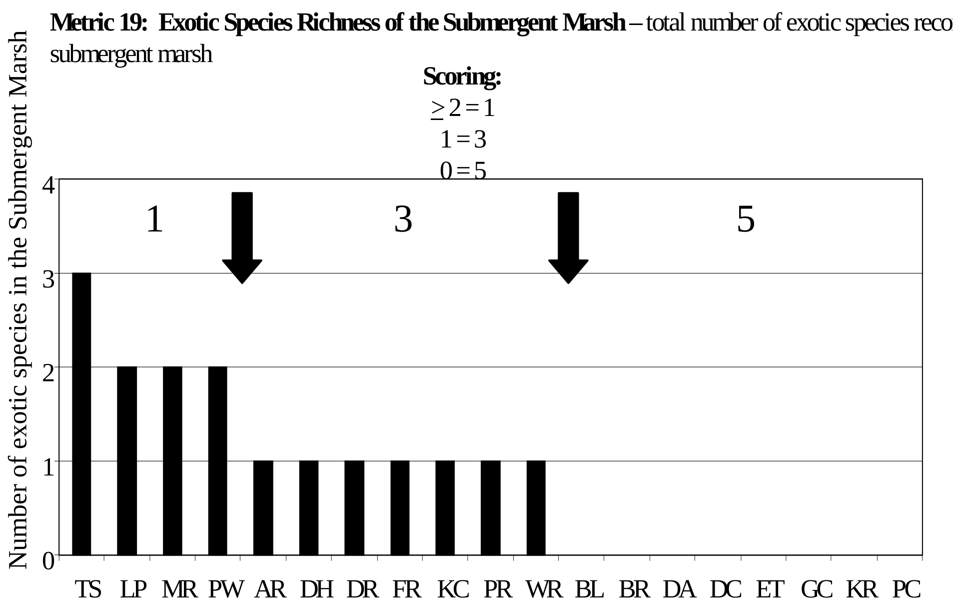
Figure 3



AR = Arcadia River ET = E Twin River MR = Manistee River
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BR = Brevort River GC = Grand Calumet Il. PR = Pigeon River X
DA = Days River KC = Keyes Creek PW = Pentwater
DC = Dunes Creek KR = Kewaunee River TS = Thomas Slough
DH = Dead Horse EPA LP = Little Pigeon Creek WR = White River

Table 3. Summary of metrics and IBI scores for 19 Lake Michigan Great Lakes riverine wetlands.

	Wet Meadow									Emergent Marsh							Submergent Marsh						Submergent & Emergent Marsh				
Site	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	Sum	Average
Pigeon River X	5	5	3	3	5	5	5	5	5	5	5	5	5	3	5	3	3	3	3	3	1	3	5	3	3	99	3.96
Days River	1	5	1	3	5	1	3	1	3	5	5	3	5	5	5	5	3	5	5	5	1	5	5	3	5	93	3.72
Dead River	5	3	3	1	1	5	5	3	3	5	5	1	3	3	5	3	5	5	3	1	5	5	5	1	5	89	3.56
Bass Lake	1	5	5	3	3	1	1	1	1	3	3	5	5	1	3	3	5	5	5	5	5	5	3	3	5	85	3.40
Brevoort River	5	5	3	5	5	5	3	5	3	1	1	5	5	1	1	1	1	1	5	5	1	5	1	5	5	83	3.32
Pentwater	3	5	5	3	3	1	3	1	3	5	5	1	3	3	5	5	3	3	1	3	3	3	5	3	5	83	3.32
Arcadia River	3	5	3	3	5	3	3	1	5	3	5	3	3	3	3	3	3	3	3	1	3	3	3	1	5	79	3.16
Kewaunee River	5	1	3	5	5	3	5	5	5	1	1	1	1	3	3	5	1	1	5	5	1	3	1	5	5	79	3.16
Keyes Creek	3	3	3	3	3	3	5	3	5	3	3	5	5	1	1	1	3	5	3	3	1	5	1	3	5	79	3.16
Little Pigeon Ck.	5	5	3	5	5	1	1	1	3	3	3	3	3	1	3	3	5	5	1	1	5	5	3	1	5	79	3.16
Manistee River	1	5	1	3	3	1	3	1	3	5	5	1	1	5	5	5	5	5	1	3	3	3	5	1	5	79	3.16
Portage Creek	5	1	1	5	3	5	3	3	3	1	1	3	1	1	1	1	5	3	5	5	5	5	3	5	5	79	3.16
White River	1	5	5	5	5	1	1	1	1	5	5	1	3	3	5	5	1	1	3	3	1	5	5	3	3	77	3.08
Dead Horse EPA	5	1	3	5	3	3	5	3	3	1	1	5	5	1	3	5	1	3	3	1	3	1	1	1	1	67	2.68
E Twin River	3	1	3	3	5	1	5	1	5	1	1	5	5	1	1	1	3	1	5	5	1	1	1	1	5	65	2.60
Thomas Slough	5	1	1	3	5	3	3	1	1	1	1	5	5	1	1	1	5	3	1	1	3	5	3	1	5	65	2.60
Dunes Creek	3	5	3	3	5	1	1	1	1	3	3	3	3	1	3	3	1	1	5	5	1	5	1	1	1	63	2.52
Fox River	1	1	1	5	5	1	1	1	1	1	1	1	1	1	3	3	1	1	3	3	1	5	1	5	5	53	2.12
Grand Calumet	1	3	1	5	5	1	1	1	1	1	1	3	1	1	1	1	1	3	5	5	1	1	1	3	1	49	1.96
																							Maximum			99	3.96
																							Minimum			49	1.96
																							Median			79	3.16
																							Mean			77.6	3.10

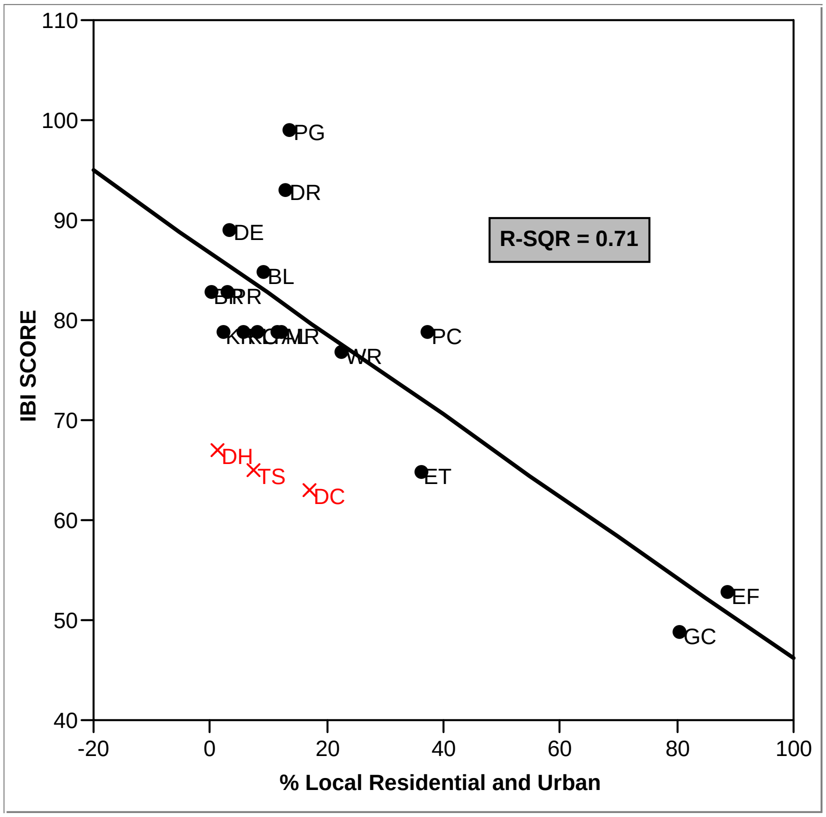

Figure 4. Relationship of IBI score to % local (within two miles of sampling transect) residential and urban development. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

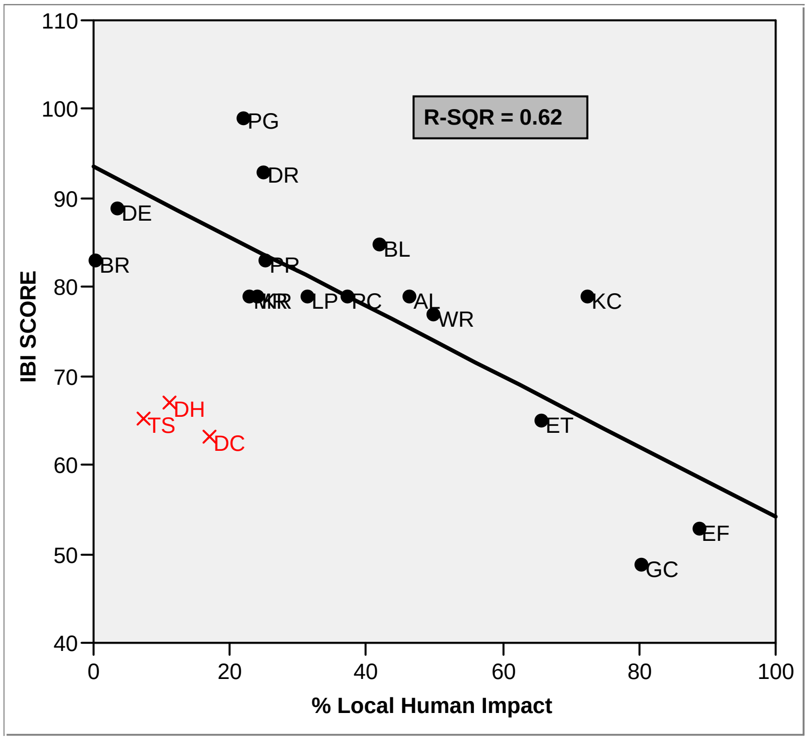

Figure 5. Relationship of IBI score to % local (within two miles of sampling transect) human development. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

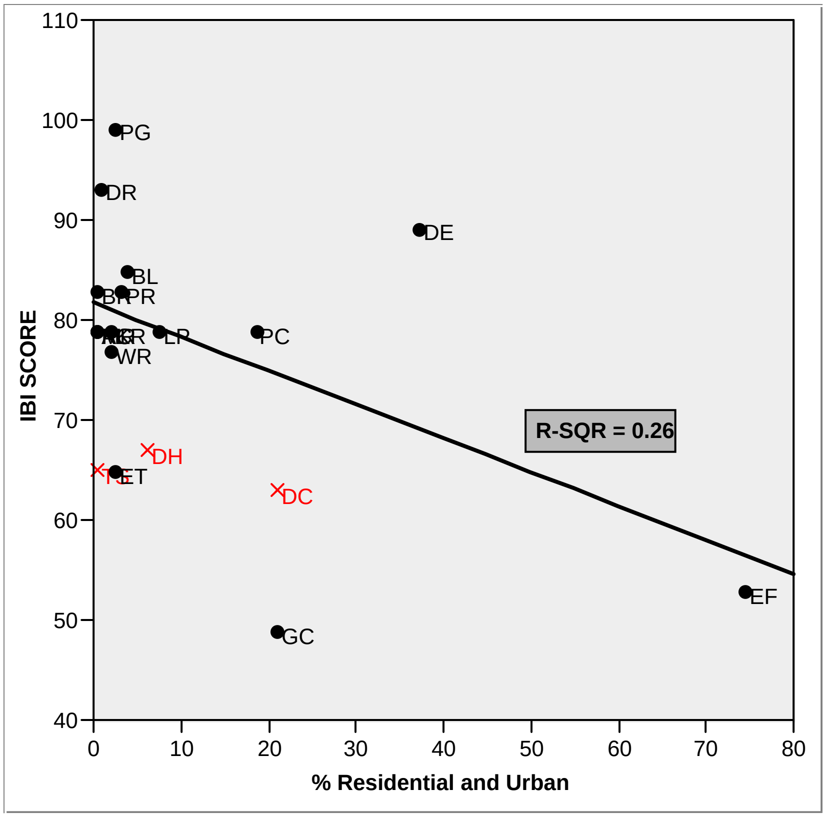

Figure 6. Relationship of IBI score to % residential and urban development within the watershed. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

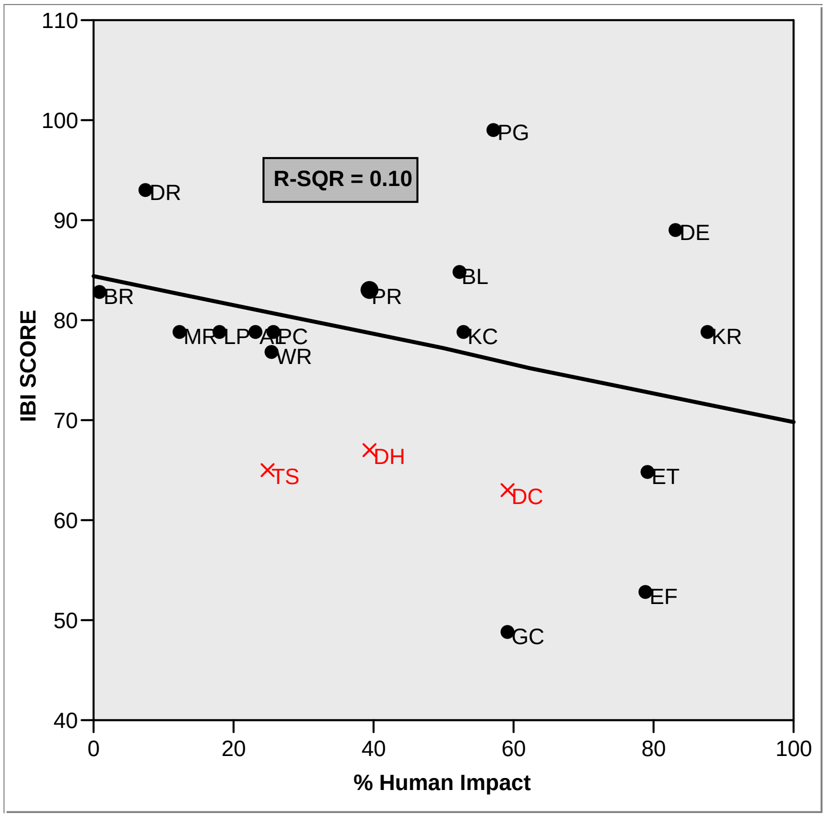

Figure 7. Relationship of IBI score to % total human development within the watershed. Cases marked in gray (x's) represent extremely disturbed sites removed from the analysis.

7 A Plant Index of Biotic Integrity for Drowned River Mouth Coastal Wetlands of Lake Michigan

Paul E. Rothrock & Thomas P. Simon

CONTENTS

7.1 INTRODUCTION

Monitoring and assessment programs are being formulated to provide information on the structure and function of biological indicators for Great Lakes coastal wetlands. This has required the development of new indicators and assessment tools (Simon 2000). Development of macrophyte indicators for coastal wetlands of the Great Lakes has required a paradigm shift from previous research programs. This shift has challenged the concept that all wetlands are unique (Chow-Fraser and Albert 1998) and substituted recognition that wetlands can be clustered into three basic hydrogeomorphic classes (Keough et al., 1999).

Wetland biological assemblages have evolved in harsh, changing environments, where water fluctuations, seiche, and turbidity changes have caused significant cycles in wetland patterns (Wilcox ,1995). Likewise, land use changes around the Great Lakes have experienced shoreline development, toxic impacts, and constructed industrial and impervious structures (i.e., confined disposal facilities [CDFs]) in areas that were once large expanses of wetlands (Stewart et al., 2003). Wilhelm et al. (2003) evaluated the condition of these CDFs around the Great Lakes in order to determine recovery trajectories. Thus, fragmentation and edge effects have become significant concerns. Finally, since so few wetlands remain, it is imperative that a variety of wetland sizes, wetland conditions, and drainage areas be included in any calibration to provide an ecological dose-response curve to test candidate metrics (Karr and Chu, 1999).

The development of biological indicators for primary producers in the Great Lakes has only recently begun (Stewart et al., 1999). Wetland indicators using plant assemblages have been developed for use in Wisconsin (Nichols et al., 2000), Ohio

(Mack, 2001), Minnesota (Gerness, 19xx), and the northern plains (DeKeyser et al, 2003), but no indicators have been developed for coastal wetlands. Plant indices of biotic integrity (PIBIs) using plant assemblages have been established for riverine and palustrine wetlands in southern Lake Michigan (Simon et al., 2001) and inland lacustrine wetlands in the Lake Michigan drainage (Rothrock et al., in review). As plant assemblage indicators are developed for drowned river mouth coastal wetlands of the Great Lakes, current indicators may need modification from each of these systems and require a separate calibration.

The purpose of this study was to determine whether the riverine, palustrine, or lacustrine wetland indices could be used as they are currently developed (Simon et al., 2001; Rothrock et al., in review) or adapted for coastal wetlands of the Great Lakes. We developed our pilot project in Lake Michigan for several reasons: 1) Lake Michigan is entirely within the United States and enabled sampling on both shorelines, 2) the orientation of Lake Michigan provides a snapshot of the full extent of latitudinal differences within the entire Great Lakes, 3) previous reference calibrations, tolerance, and metrics have been developed in Lake Michigan, thus, testing of these indices required sampling in Lake Michigan to enable similar comparisons. Finally, we provide a new calibration for wetland plant assemblages in Lake Michigan using a random probability design and use an ecological dose response paradigm.

7.2 MATERIALS AND METHODS

7.2.1 STUDY SITES AND STUDY DESIGN

Fifteen Lake Michigan drowned river mouth wetlands were randomly chosen as study sites using a tessellated, stratified design incorporating ecoregions and wetland size (Fig.

7.1). The sites encompassed the five EPA Level III Ecoregions that surround Lake Michigan (Northern Lakes and Forests, North Central Hardwood Forests, Southeastern Wisconsin Till Plains, Central Corn Belt Plains, and Southern Michigan/ Northern Indiana Drift Plains[Omernik 1987]). These wetlands covered a broad range of quality including several severely degraded by industrial activity and several deemed least-impacted by human activity. Stream channel width and wetland size also varied greatly across the suite of wetlands and within each Ecoregion. Overall, channel widths ranged from 3 to 65 m.

7.2.2 SAMPLING STRATEGY AND COLLECTION METHODS

Qualitative plant sampling techniques were used to evaluate plant assemblages. Sampling was done by surveying a distance of up to 35 times the channel width along the shore in all vegetative zones. The sampling intent was to perform a representative qualitative survey, not an exhaustive census, and was targeted at biological diversity and relative abundance estimates (Simon et al., 2001). All species of wetland obligate and facultative plants were identified and an abundance rating (1-Observed, 2-Rare, 3-Rare/Common, 4-Common, 5-Very Common, 6- Abundant) assigned to each species. Abundance categories represented the number of individuals of a plant species at a site; “observed" was assigned when only one individual of a species is found; "rare" was assigned when a plant species was found two to four times at a site; "rare/common" was assigned when the plant species was found more than four instances, but was never a common component of the community at a site; "common" species were those that were easily located at a site; "very common" species were slightly dominant at a site, and comprised up to about 25% of the community at a site; and "abundant" species were those that dominated a site, and comprised from 25% to almost 100% of the plant community. Identifications were done in the field and unknowns were identified using appropriate floristic manuals.

In addition to the qualitative plant sampling, each site was assigned a quality rank, from 0 to 10, based upon the best professional judgment (BPJ) of two independent observers. The BPJ rankings by independent observers consistently differed by 2 or less and had a Spearman $r^2 = 0.81$ ($p < 0.0003$). These average BPJ estimates for each site provided one benchmark for the testing of metric hypotheses.

7.2.3 TESTING OF METRICS

We used cluster analysis to demonstrate whether wetland types were so exceptionally distinct as to require multiple PIBI tools for assessing Lake Michigan river mouth wetlands. Between-site similarity of the wetland plant communities was evaluated by clustering and by ordination techniques. For clustering analysis, the species X cover matrix was converted to a distance matrix and subjected to an unweighted pair-group cluster analysis.

Our previous PIBI's consisted of 11 to 12 metrics covering 4 or 5 function categories. We retested the metrics from the palustrine and riverine PIBI (Simon et al., 2001) and lacustrine PIBI (Rothrock et al., in review) as well as new metrics across four function categories. Metric hypotheses were examined graphically against estimates of habitat quality to determine if the patterns found fit expectations. Further quantitative testing was then performed by means of Spearman correlations. Correlations between potential metrics were calculated and those with $r^2 = 0.80$ and above were considered redundant. The scoring for each PIBI metric follows Karr et al. (1986). In short, each metric was scaled against river width to detect possible factor ceiling-distributions and the data were then trisected. A score of 5 was assigned to the least impacted or reference condition wetlands, 3 to the middle grouping that shows deviation from reference conditions, and 1 to the lowest quality, most impacted sites.

We used Swink and Wilhelm's (1994) coefficient of conservatism (CC) to classify plants as either sensitive or tolerant. Plants with highest scores (8—10) are sensitive while tolerant plants have low scores (0—2). Plants with high CC values are not necessarily rare in the flora nor are plants with low CC values necessarily common. In each case, plants are essentially defined on the basis of ecological behavior. Values have been formalized for plants of northeastern Illinois (Swink and Wilhelm, 1994), Michigan (Herman et al., 1996), and recently for Wisconsin (Bernthal, T.W. 2003).

7.3 RESULTS AND DISCUSSION

7.3.1 FLORA AND COMMUNITY SIMILARITY

The drowned river mouth wetland communities of Lake Michigan have a diverse flora. The 15 coastal wetland sites in this study supported about 225 species from over 60 families (Rothrock et al., chapter 10). Among the largest families are Cyperaceae (32 species), Potamogetonaceae and Poaceae (14 species), and Asteraceae (12 species). The physiognomy of these species is also broad. There were over 25 species of submergents, 8 floating-leaved species, and 160 emergent and 25 woody species. Of the species observed, 1/3 were considered sensitive (i.e., had coefficients of conservatism ranging between 8--10), while 20% were exotic species or tolerant species (i.e., had coefficients ranginf between 0--2). Only two species that we encountered are considered threatened or endangered at the state level (Michigan State University Extension 2000). Given the large number of sensitive species observed, i.e., species characteristic of intact natural communities, we do not consider these sites to have experienced sufficient history of disturbance to extirpate the rarest elements. Rather, we suggest that this paucity of endangered or threatened species indicates that these productive Lake Michigan wetland habitats historically lacked numbers of rare species (Moore et al. 1989) and that they currently remain expansive enough to support sustainable populations of indigenous species.

Cluster analyses and ordination of the 15 coastal wetland sites did not reveal any unexpected heterogeneity of wetland communities (Fig. 7.2). Neither of the ordination methods showed any compelling necessity for developing multiple PIBIs within the Lake Michigan basin. High quality wetlands located in different Ecoregions readily clustered with others, as seen with Dead River (IL) and Hog Island Creek (MI); as well as between Dunes Creek (IN) and Pigeon River (MI). The cluster formed by White River (MI), Arcadia (MI), Kenauwee River, (WI), and Little Tail Creek (WI) not only included different ecoregions but also different river channel widths and associated watershed areas that differed strongly in scale (Fig. 7.2). The narrowest channel, Little Tail Creek, measured only 3 m in width compared to 39 m for Kenauwee River. Although the two most degraded sites, i.e., Grand Calumet River and Fox River, had low species diversity, they fell within the general clustering of wetland sites.

7.3.2 METRICS FOR SPECIES RICHNESS AND COMPOSITION

The number of species, a common measure of species diversity, has found wide usage in animal IBI's (Karr and Chu, 1999) and in recent PIBI efforts (Simon et al., 2001;

Rothrock et al., in review). The metric hypothesis postulates that the number of plant species would increase with biotic integrity due to reductions in chemical and physical disturbances (Ehrenfeld and Schneider, 1990; Jurik et al., 1994; Findlay and Houlahan, 1997). We accepted this hypothesis because the most degraded sites had 10—33 species, while three high quality sites had between 60—79 species. Conversely, it is important to note that the two sites, Dead River and Hog Island Creek, which were given the highest ranking of site quality based upon best professional judgment, only had 44—48 species. In this study, the response of total species richness to changes in habitat quality was a non-linear, resulting in a low Spearman correlation (Table 7.2). We attribute the anomalous behavior of these two sites to unusual stream morphology. These waterways incise through rather sandy substrates resulting in relatively narrow and U-shaped channels with restricted wetland areas. Nonetheless, these sites scored high for most metrics, especially metrics that evaluate species quality. Despite the limited wetland area, the total species number observed in Dead River and Hog Island Creek was high enough to achieve a 5 (least impacted score).

Two groups of emergent graminoids, sedges (Cyperaceae) and rushes (Juncaceae) are an important component of temperate and cold temperate wetland communities (Heywood, 1978). Simon et al. (2001) found that a greater number of *Carex* species, a large and significant genus of Cyperaceae, were associated with high quality riverine wetlands. We found that same relationship to apply to drowned river mouth coastal wetlands. In addition, members of the genus *Juncus* have the potential, either alone or in combination with members of the sedge family, to act as a metric of habitat quality. We tested the metric for total sedge-rush species, expecting an increase in the number of species as quality increased. A significant relationship was demonstrated ($p = 0.03$, Table 7.2): worst sites had as few as 0—2 species compared to 15 or more in best quality sites (Fig. 7.3). As with total number of species (Fig. 7.3A), no relationship was evident between river width and the range of values for this metric (Fig. 7.3B).

PIBI's for lacustrine and palustrine sites (Simon et al., 2001; Rothrock et al., in review) used the metric "number of emergent species" in place of a one based upon graminoids alone. For coastal wetlands we found that a high correlation ($r^2 = 0.97$) existed between the number of emergent species and total number of species, rendering this potential metric redundant.

Waters of the Great Lakes have a high natural diversity of submergent species, especially those belonging to the genus *Potamogeton* (Voss, 1972; Wiegleb, 1988). Previous work (Simon et al., 2001) indicated that the number of submergent species would increase with habitat quality in riverine habitat, but too few *Potamogeton* species were present to provide a useful metric. In the lacustrine setting (Rothrock et al., in review), although both metrics were predictive of habitat quality, the total number of submergent species was tightly constrained than the number of *Potamogeton* species alone. For Lake Michigan drowned river mouth wetlands, the generalized metric, number of submergent species, provided a strong indicator of habitat quality (Table 7.2), which had a low correlation with other potential metrics(r^2 mostly $<< 0.5$). Least impacted sites often had 9 or more submergent species, while degraded sites ranged from 0 to 3. Surprising, the severely degraded Grand Calumet River site supported 3 species, although the most abundant among these was the weedy exotic *Potamogeton crispus*.

Finally, in constructing PIBI's for palustrine, riverine, and lacustrine settings, the number or percent of floating species and perennial species were potential measures of biotic integrity. Floating species, useful in lacustrine PIBI, proved too few in number in drowned river mouth coastal wetlands. The number of perennial species showed some relationship to biotic integrity, but was highly correlated ($r^2 = 0.99$) with the total number of species.

7.3.3 SPECIES TOLERANCE AND SENSITIVITY

Sensitive species, those species associated with less impacted natural communities, are expected to be among the first to disappear under conditions of declining biological integrity (Karr 1981; Simon et al. 2001). On the other hand, tolerant and exotic species can grow under a wide range of habitat conditions, such as high rates of sedimentation (Dittmar and Neely 1999), and would tend to increase with increasing degradation and disturbance (Karr 1981; Magee et al. 1999). The expected behavior of sensitive and tolerant/exotic species was supported by results from the 15 wetland sites. The percent sensitive species increased significantly ($p = 0.008$) with increasing habitat quality, while degraded habitat showed an even stronger relationship ($p < 0.0001$) with the percent of tolerant and exotic species (Table 7.2). Least-impacted wetlands had at least 20% sensitive species and less than 30% tolerant species (Table 7.1). In calibrating these metrics, no scaling against river width was required (Fig. 7.2).

7.3.4 GUILD STRUCTURE

Root (1967) coined the term "guild" to describe groups of functionally similar species in a community. The concept of trophic guilds has wide usage in zoological literature, including IBI's for fish and macroinvertebrate communities (Karr 1981; Kerans and Karr 1994; Karr and Chu 1999). For PIBI's, guild identification has relied upon a broad range of attributes, such as obligate wetland species; woody, emergent, floating-leaved, and submergent species; pioneer and weed species; and tolerant and sensitive species (Simon et al. 2001; Rothrock et al. in review). Measuring the integrity of these guilds is achieved by estimating either the number, percent of species within the guild, or the overall relative abundance of guild members.

Obligate wetland plants are species occurring in wetlands with an estimated probability of greater than 99% under natural conditions (Reed 1988). The number of species in this guild is expected to decline with changes in hydrology due to ditches or dredging (Ehrenfeld and Schneider 1991), with nutrient enrichment and pollution from septic and industrial effluents (Moore et al. 1989), and with loss of aerial extend due to wetland filling (Camargo 1997). Results from Lake Michigan coastal wetland sites supported this hypothesis (Table 7.2; Fig. 7.2) since the most degraded wetlands had fewer than 12 obligate species compared to over 30 in those considered least-impacted (Table 7.1).

Pioneer species are characteristic of early successional stages or invade bare substrates (Whittaker 1993); where, according to Grime's (1977) primary plant strategies model, competition and stress are low to moderate. Pioneer species include many annual and biennial herbs and may be part of a persistent seedbank (van der Valk 1981) that germinates following disturbance or sedimentation (Dittmar and Neely 1999). Competitive weed species, in contrast, may perform poorly under these same conditions (Jurik et al. 1994). Instead weed species are perennials or woody plants with a strong power of vegetative spread (e.g., clonal dominants such as *Typha angustifolia* and *T. latifolia*), rapid growth rates and competitive ability (e.g. *Lythrum salicaria*), and phenotypic plasticity (Hill 1977; van der Valk 1981; Bazzaz 1986). Late successional communities are expected to have a minor presence of pioneer species, while the percent pioneer species should be higher in a community experiencing physical disturbance. Similarly, the percent weed species should be higher in disturbance communities, especially when experiencing nutrient enrichment (Weiher et al. 1996). Our sites supported this trend for Lake Michigan drowned river mouth wetlands (Table 7.2; Fig. 7.3). The most impacted wetlands, such as of Grand Calumet, Fox, and Kewaunee Rivers, had over 30% of both weed and pioneer species (Table 7.1). In calibrating these guild structure metrics, none required scaling against river width (Fig. 7.3).

Emergent species provided diagnostic guild metrics for palustrine and riverine wetlands (Simon et al. 2001). In these communities, the percent emergent species was lower in reference wetlands than in degraded sites. We tested percent emergents and relative abundance of emergents as guild metrics for drowned river mouth wetlands. Both metrics proved to be non-predictive ($r^2 < 0.02$, $p = 0.96$).

In the lake PIBI, the relative abundance of woody species was a non-redundant guild-based metric (Rothrock et al. in review). We tested this same metrics for drowned river mouth coastal wetlands, as well as the relative abundance of submergents, various emergent cohorts, and sensitive and tolerant species. While the relative abundance of sensitive and tolerant species had significant responses with biological integrity, they were also correlated (Spearman r^2 = 0.80 or more) with other metrics. The relative abundance of woody species, which decreased under reference conditions in lacustrine wetlands, did not respond in drowned river mouth communities (Spearman r^2 = -0.16, p = 0.5). The relative abundance of native submergents showed potential as a signature of environmental quality, usually having greater than 8% representation in least impacted sites. Several degraded sites completely lacked submergent plants; however, one of our most severely degraded sites, Grand Calumet River, had a relative abundance of native submergents of 15%. The high percentage we observed was seen when comparing a modest submergent plant community against total community abundance derived from only a few, prolific weedy species. To resolve this problem, we tested an alternative metric, average cover of native submergents, and found a consistent response signature to habitat quality (r^2 = 0.75; p = 0.001).

7.3.5 VEGETATION ABUNDANCE

Low abundance values may be due to general diminution of vegetative cover, but typically are the result of vegetative dominance of a few weedy species (Farnsworth & Ellis 2001). Mean relative abundance may be indicative of degraded wetland habitats as seen, for example, in lacustrine and palustrine settings (Simon et al. 2001; Rothrock et al. in review). In the Lake Michigan coastal wetlands this potential metric increased with habitat quality, but correlated with total species richness ($r^2 > 0.85$, $p < 0.0001$) and was not used in the PIBI due to redundancy.

Although average abundances failed our validation test as a metric, abundance can resolve issues of species dominance or evenness across the wetland community. Abundance in reference wetlands is expected to follow a log normal curve, i.e. many species have low to moderate abundance and a few have higher abundance. The contrasting degraded sites tend to have a limited number of low abundance species and high dominance by one or a few species. We used the variance of relative abundances as a simple means of capturing dominance (Table 7.2; Fig. 7.3). The variance in degraded sites was generally high (>1.67), an indication of high dominance by a few species, while the variance of least-impacted sites was low (<1.33).

Exotic species are known for their negative impact on habitat quality (McKnight 1993). Habitat degradation creates conditions favorable for invasion and high relative abundance of exotic species (Morin et al. 1989; David 1999; Galatowitsch et al. 1999). As expected, least impacted drowned river mouth wetland sites had relative abundance of exotics less than 10%, while severely degraded sites had higher relative abundances greater than 20% (Table 7.2; Fig. 7.3). Neither dominance nor abundance of exotics required scaling with river width (Fig. 7.3).

7.3.6 PLANT INDEX OF BIOTIC INTEGRITY

The Floristic Quality Index (FQI) assesses the diversity and quality of a plant community and has found wide acceptance in the Midwestern USA. The FQI is based upon the number of species observed in a habitat and the average quality of those species as determined through application of the CC concept. We calculated the FQI's for each of the 15 drowned river mouth wetland sites and compared them to our total PIBI values (Table 7.3). Since both FQI and PIBI are attempting to specifically estimate vegetation quality and since the PIBI even relies to a limited extent upon the same criterion (namely CC values) used in the FQI, it was expected that the two indices would have a high correlation (Spearman $r^2 = 0.88$, $p < 0.0001$). PIBI scores in the good to excellent range (PIBI = 44 or more) had FQI values above 36.9 (Table 3). By contrast, poor to very poor sites (PIBI = 31 or less) had FQI values of less than 27. Further validation and calibration of the PIBI is needed through assessment of additional drowned river mouth wetland sites and to test its general applicability to other Great Lake basins.

Nonetheless, the comparison between PIBI and FQI suggests that PIBI can be a potent rapid assessment tool for wetland habitats. At the same time, given the current availability of FQI as a measure of habitat quality, what advantage is provided by a PIBI? We suggest that a PIBI provides additional information about site quality in the form of response signatures. In addition to the final PIBI value, 11 sub-scores are available that detail specific aspects of community function that either meet standards or diverge from reference conditions. For example, fair quality sites may have weedy species and exotics, compromises that could be readily observed in low sub-scores, and yet support an overall richness of species and a richness of submerged species in particular. To highlight a specific case among our study sites, the Manistee River wetland scored negatively for the dominance metric, due to significant patches of *Alnus* trees and *Phalaris arundinacea* and *Circium arvense* in portions of this large wetland. A comparable PIBI score was measured at the White River site; however, in this case, the deficient metric was a low percentage of sensitive species due to the overall lower species richness. A third fair site, Portage Creek had a dearth of submergent plants, perhaps due to channel dredging.

Although PIBI and FQI have high correlations, it is of interest to note that the two sites with the highest PIBI scores, i.e., Dead River and Hog Island Creek, did not achieve the highest FQI score. In these cases, FQI, which only evaluates species number and quality, neglected relevant measures of community structure such as the abundance of exotic and submergent species guilds.

Since the Lake Michigan sites were part of a larger study of drowned river mouth coastal wetlands we evaluated predicted quality with QHEI (qualitative habitat evaluation index) scores (Table 7.3). The correlation between QHEI and PIBI was low (r^2 = 21, p = 0.45). This shows that habitat quality as measured for invertebrate and fish assemblages does not correlated with habitat quality in plant assemblages. This shows that plants are not responding to the same environmental signs as animal assemblages.

Fish and macroinvertebrate indices of biotic integrity have enjoyed wide usage over the part several decades. Our recent investigation of plant IBI's in the Midwestern USA Great Lake region has demonstrated the feasibility of making similar rapid, multimetric quality assessments based upon vegetation. The development of PIBI has entailed a diversity of metrics and, an overview of the metrics included in PIBI, to date, may provide a pattern for devising PIBI for additional habitat types (Table 7.4). Some metrics have been of value across a range of wetland habits. These include number of species overall and of submergent and emergent species; tolerant and sensitive species (either as number or percent of species); pioneer species; overall vegetation abundance or dominance; and abundance of exotics. A few had specific application; floating leaved species were diverse in the lacustrine setting as were woody species. Similarly, the cover of native submergent species showed relative worth well in the drowned river mouth coastal wetlands.

The individual deformity and anomaly metric used in animal IBI's to identify the lowest levels of biological integrity (Karr 1981; Karr et al. 1986), may lack a clear response signature in PIBI's. Symptoms of toxicity that might be observed during rapid assessment could include growth reduction, small leaves, necrotic, chlorotic or discolored leaves, and early leaf fall (Adamus et al. 2001). In our experience, individual plant condition, even in our most degraded sites, remained visually excellent. Among the Lake Michigan drowned river mouth sites, two were particularly degraded. The Grand Calumet site supported an abundance of a few tolerant species with no visible deterioration of individual health despite murky water quality and sediments capable of emitting hydrocarbons. Likewise, the second most degraded site on the Fox River, Wisconsin, lacked observable submergent species. With the exception of a large stand of purple loosestrife undergoing experimental treatment with *Galerucella* beetles, river margin emergent species had abundant, vigorous biomass.

In conclusion, it is clear from the comparison of PIBI metrics (Table 7.4) that a working prototype for PIBI's is emerging. Nonetheless, as with animal IBI's, validation of metrics, as well as their calibration, is necessary before applying a PIBI to a new community type. In addition, we would observe that more information is needed on the inter-annual stability of vegetation quality. The Great Lakes are known to undergo meaningful, natural changes in water level. Water level changes in Lake Michigan are sometimes measured at 1 m or more over a several year period (Environment Canada, 2003). As a result, habitat quality measures may obtain lowered values during the several years of transition and could occur in high quality as well as degraded sites. In this study the entire suite of sites were visited within the same growing season, thus avoiding this confounding factor. However, for purposes of monitoring plant community quality around Lake Michigan, reference wetlands need to be identified.

7.4 CONCLUSIONS

Indices of Biotic Integrity (IBI) have been developed for a variety of animal assemblages

and aquatic habitat types. The use of macrophytes as indicators of wetland quality, especially in the form of plant IBI's, are in a formative stages of development. Fifteen drowned river mouth coastal wetlands in Lake Michigan, diverse in size, quality, and ecoregion location, were semi-quantitatively sampled. Eleven metrics, divided in 4 function categories (species richness and composition, species tolerance, guild structure, and vegetative abundance) had strong response signatures and low autocorrelation. Final PIBI scores, ranging from 17 to 53, were strongly correlated with the familiar floristic quality index (FQI) but did not show significant cluster or ordination relationships with either Ecoregions or wetland size. Specific wetland examples suggest that although the PIBI represents a rapid assessment technique, it can provide more information about vegetation quality than the FQI.

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TABLE 7.1

Calibration of plant index of biological integrity (PIBI) for submerged rivermouth wetlands of Lake Michigan. The ranges for the various IBI scores are based upon trisection of the metrics presented in figure 7.2.

	Scoring		
Attribute	1 (worst)	3	5 (best)
I. Species richness and composition			
1. total number of species	0-21	22-43	>43
2. number of sedge-rush species	0-5	6-11	>11
3. number of submergent species	0-3	4-7	>7
II. Species tolerance			
1. percent sensitive species	0-10	10-20	>20
2. percent tolerant and exotic species	>40	30-40	<30
III. Guild structure			
1. number of obligate wet species	0-16	16-30	>30
2. average cover of native submergent species	0-0.8	0.8-1.6	>1.6
3. percent pioneer species	>36	31-36	<31
4. percent weed species	>22	11-22	<11
IV. Vegetation Abundance			
1. dominance (variance)	<0.62	0.62-0.77	>0.77
2. relative abundance of exotics	>18	9-18	<9

TABLE 7.2.

Spearman rank correlation coefficients and significance level between proposed plant metrics and qualitative estimates of habitat quality (best professional judgment).

Metric	Hypothesized Change with Increasing Quality	r^2	p
I. Species Richness and Composition			
1. total number species	Increase	0.33	0.23
2. number of sedge-rush species	Increase	0.56	0.03
3. number of submergent species	Increase	0.69	0.005
II. Species tolerance			
1. per cent sensitive species	Increase	0.65	0.008
2. percent of tolerant and exotic species	Decrease	-0.90	<0.0001
III. Guild Structure			
1. number of obligate species	Increase	0.55	0.03
2. average cover of native submergent species	Increase	0.75	0.001
3. percent pioneer species	Decrease	-0.61	0.02
4. percent weed species	Decrease	-0.78	0.007
IV. Vegetation Abundance			
1. dominance (variance)	Decrease	-0.81	0.0003
2. relative abundance of exotics	Decrease	-0.68	0.006

TABLE 7.3

Submerged rivermouth sites of coastal Lake Michigan: ecoregion, floristic quality index (FQI) and plant index of biotic integrity (PIBI). CCBP = central corn belt plains; NCHF = north central hardwood forests; NLF = northern lakes and forests; SMNITP = southern Michigan/ northern Indiana till plains; SWTP = southeastern Wisconsin Till Plains.

Site	Ecoregion	QHEI	FQI	PIBI
Dead River, IL	CCBP	78	37.8	53
Hog Island, MI	NLF	50	36.9	51
Pigeon River, MI	SMNITP	66	45.3	51
Dunes Creek, IN	CCBP	71	47.5	51
Days River, MI	NLF	51	41.2	49
Arcadia, MI	NCHF	43	37.3	47
Little Tail Point, WI	NCHF	55	33.8	43
Keyes Creek, WI	NCHF	72	33.1	41
Pentwater River, MI	SMNITP	54	32.9	37
Manistee River, MI	NCHF	62	35.3	37
Portage Creek, MI	NLF	57	35.5	33
White River, MI	SMNITP	65	24.4	33
Kewaunee River, WI	SWTP	72	26.8	31
Grand Calumet, IL	CCBP	48	8.5	17
Fox River. WI	SWTP	49	17.2	17

TABLE 7.4.

Comparison of metrics used in plant indices of biotic integrity: lake IBI (Rothrock et al. in review), palustrine IBI and riverine IBI (Simon et al., 2001), and drowned rivermouth IBI. Yes = used, Variation = used in modified form, No = not used, Yes* = used but not calibrated.

Metric	Lake IBI	Palustrine IBI	Riverine IBI	Rivermouth IBI
I. Species richness and composition				
Total number of species	Yes	Yes	Yes	Yes
Number of emergent species	Yes	Yes	Variation	Variation
Number of floating leaved species	Yes	No	No	No
Number of submergent species	Yes	Variation	Yes	Yes
Number of perennial species	No	Yes	Yes	No
II. Species Tolerance				
Sensitive species	Number	Number	Number	Percent
Tolerant and exotic species	Percent	Number	Number	Percent
III. Guild Structure				
Obligate species	Relative abundance	Percent	Percent	Number
Emergent species	No	Percent	Percent	No
Submergent species	No	No	No	Avg. Cover
Pioneer species	Relative abundance	Number	Number	Percent

Weed species	No	Number	Number	Percent
Woody Species	Relative abundance	No	No	No
IV. Abundance				
Mean relative abundance, mean cover, dominance	Mean cover	Mean relative abundance	Mean relative abundance	Dominance
Exotics (relative abundance)	Yes	No	No	Yes
V. Individual condition				
Percent taxa with deformities or anomalies	No	Yes*	Yes*	No

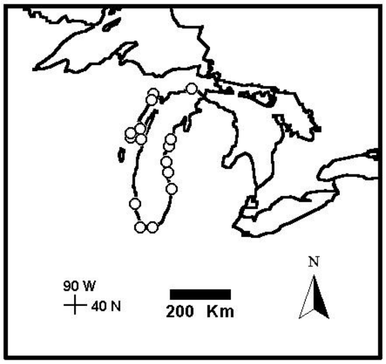
Figure Captions

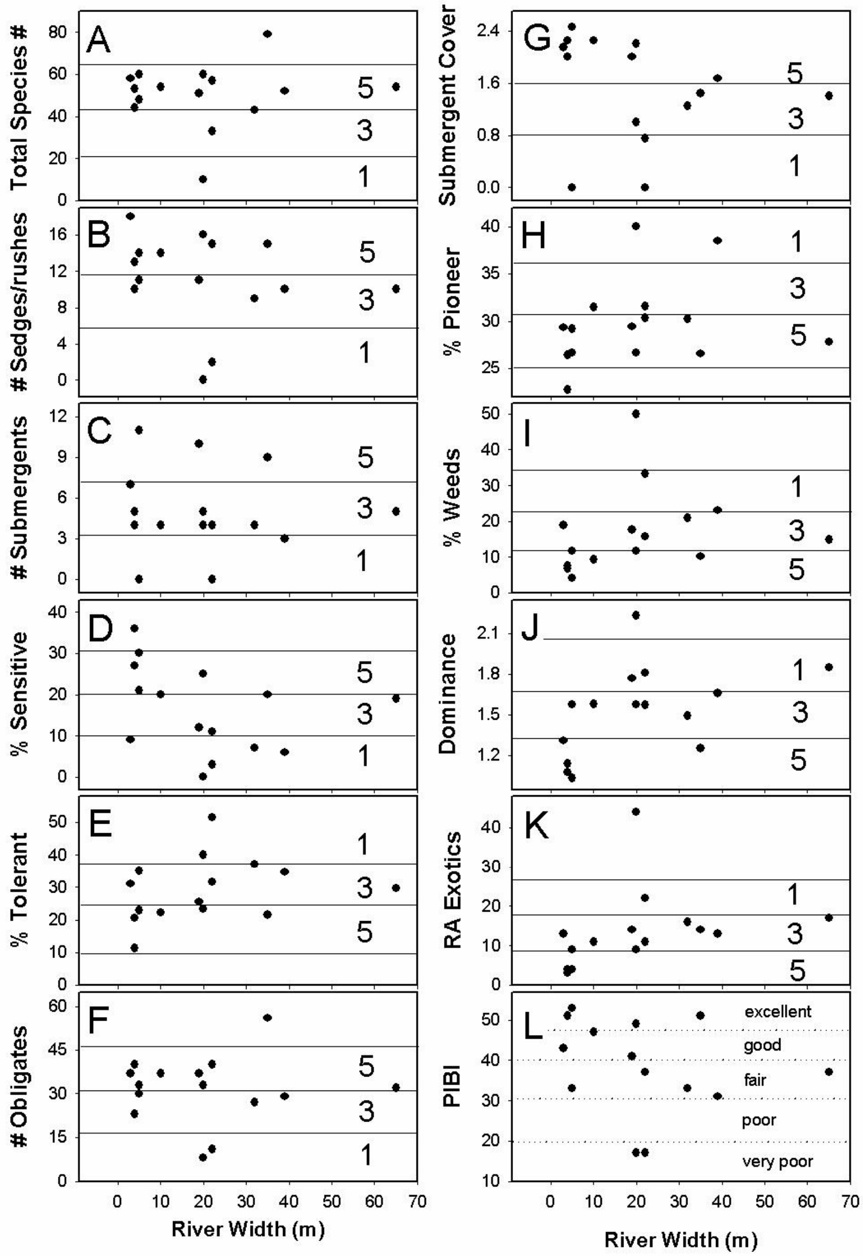
Figure 7.1. Map showing the 15 drowned river mouth coastal wetland sites sampled in Lake Michigan.

Figure 2. Cluster analysis of 15 Lake Michigan drowned river mouth coastal wetlands. Only taxa found in more than two samples were used in the analysis.

Figure 3. Metrics of drowned river mouth coastal wetland plant community used in assessment of biotic integrity.

Fig. 7.1





A
B
C
D
E
F
G
H
I
J
K
L
Total Species #
Sedges/rushes
Submergents
% Sensitive
% Tolerant
# Obligates
Submergent Cover
% Pioneer
% Weeds
Dominance
RA Exotics
PIBI
excellent
good
fair
poor
very poor
River Width (m)
River Width (m)

8 Comparison of Ecoregional Discriminative Ability and Collection Efficacy of D-net versus Activity Trap Sampling of Macroinvertebrate Assemblages in Great Lake Coastal Wetlands

Paul M. Stewart and Thomas P. Simon

CONTENTS

8.1 INTRODUCTION

The ecological integrity of Great Lakes Coastal Wetlands has been severely reduced due to development and other anthropogenic stressors (Brazner 1997; Kashian and Burton 2000). Coastal wetlands have been severely impacted with many either completely disappearing, been reduced in size, or loss of function (Simon et al., in prep a, b). These water resources have been identified as extremely important for wildlife including bird habitat, fish spawning, and fish nursery habitat (Wilcox, 1995). In addition, the benthic macroinvertebrate assemblages located there are critical food resources for fish and are important assimilators of energy in the trophic web (Burton et al. 1999; Kashian and Burton 2000).

There are five ecoregions in Lake Michigan covering generally a north-south gradient. Simon et al. (this volume, chapter 3) did not show an ecoregional effect for fish assemblages in Lake Michigan. In order to adequately develop macroinvertebrate reference conditions for the Great Lakes, the effect of ecoregion needs to be examined.

To properly assess the biological integrity of an area, there should be careful attention to adequate sampling (Resh 1979; Kerans et al. 1992; Turner and Trexler 1997) and processing (Ettinger 1984; Courtemanch 1996) of macroinvertebrate communities. Each type of sampling method has its own advantages and disadvantages (Turner and Trexler 1997). There are clear advantages to sampling both natural and artificial substrates (Cairns 1982), however, each type of artificial substrate sampler selects a subset of all species available, and in many cases, selects for certain groups over others. Evaluations of those assemblages collected by each type of sampling device will determine the type sampler most appropriate for the needs of each investigation.

Our overall study objective was to assess the biological expectations of the Great Lakes and develop ecoregion reference conditions stratified for each Great Lake, with the exception of Lake Ontario. The present paper evaluates the efficacy of two types of commonly used natural substrate macroinvertebrate sampling methods used in the Great Lakes coastal wetlands. The comparison of d-net (sweep) sampling and activity traps is based on sampling efficacy in both the numbers and kinds of species found. In addition, this research will determine if the two sampling procedures can adequately discriminate ecoregion differences among macroinvertebrate assemblages using both collection methods, and will discern if one method more clearly discriminates ecoregion over the other. Our hypothesis is that both sampling methods will sample the wetlands adequately, but that different subsets of the available assemblage will be collected by each method. In addition, we expect that one method may more clearly discriminate ecoregions over the other, but that both will serve to separate ecoregions by examination of the macroinvertebrate community.

8.2 METHODS

8.2.1 STUDY AREA

In order to compare sampling methodology, 23 drowned river mouth wetlands (Figure 8.1, Table 8.1) (Keough et al. 1999) in Lake Michigan were used as part of a pilot study designed to help select sampling strategies for a more extensive project. These drowned river mouth wetlands were randomly chosen based on stratification so that wetlands were located in each of the five ecoregions of Lake Michigan (Omernik 1987). By stratifying sampling in wetlands among the five ecoregions, samples were collected from widely divergent habitat types and wetland sizes with a wide range of anthropogenic disturbances. See Simon et al. (in press. a) for the rationale behind site selection.

8.2.2 SAMPLING METHODS

8.2.2.1 Field sampling procedures

Each method was replicated based on information contained in Burton et al. (1999) and Wilcox et al. (1999). The details of the method were described in Simon (2000). Macroinvertebrate comparative sampling included activity traps (Swanson 1978, Wilcox et al. 1999) and d-net sampling (Burton et al. 1999). Activity traps were positioned in the dominant habitat types such that five pairs of traps were set for 24 hours. The use of activity traps has been adequately described (Wilcox et al. 1999). The activity trap is a screen mesh cylinder that is formed into the shape of a minnow trap. Two of these traps were sunk to the bottom and attached to a rod that was inserted into the sediment, so that the orientation of both traps was horizontal and parallel to shore. Five of these pairs were set in and among the aquatic vegetation for 24 hours and retrieved the next day. Upon retrieval, the contents of the bottle were poured through a 500-micron sieve and all trap samples were composited for a site, preserved in 70% ethyl alcohol, and transported to the laboratory.

Sweep (d-net) sampling was done in a 500-m sampling zone and included all habitat types found within an area in proportion to its abundance in the reach. No more than 10% of the effort was expended in sampling sand or fine substrates, even when the dominant substrate type was fine substrates. Each wetland unit was sampled using a d-net (600-µm mesh) that comprised 20 sampling efforts or subsamples (each made of two sweeps with the net). These 20 efforts represented the dominant substrate and habitat types at the site. Each sampling effort was composited in the zone and emptied into a 500-µm mesh sieve bucket for preliminary cleansing (filtration of siltand sand), then deposited into wide-mouthed jars and preserved with 70% ethyl alcohol.

8.2.2.2 Laboratory processing, identification, and data reduction

In the laboratory, each composited sample was washed and placed in a grid pan with a hundred 50 mm x 50 mm squares. Subsamples were randomly chosen using a random number table, picked so that no organisms remained in each square, and sorted until at least 300 individual invertebrates were found. Records were kept of the number of grids picked and sorted. At the conclusion of the 300 organism sorting, a 5-minute pick of large and rare invertebrates was done and stored separately. All organisms were identified using standard taxonomic literature.

Organisms were identified to the lowest practical taxonomic level using standard literature (Snider 1967, Simpson and Bode 1980, Weiderholm 1983, Wiggins 1995, Merritt and Cummins 1996). Chironomids were identified to subfamily or tribe.

8.2.3 ANALYSES

Univariate structural indices (species richness, Shannon-Wiener diversity, abundance, major group results) from each sampling method were compared using a Student's t-test or one-way analysis of variance followed by a Tukey's multiple comparison test. In addition, Pearson correlation was used to examine the relationship between structural index results from d-net and activity trap sampling. A Mann-Whitney U test was used to compare univariate indices across ecoregions. Cluster analysis and non-metric multidimensional scaling was used to discriminate among ecoregions based on the two sampling methods.

8.3 RESULTS

8.3.1 NUMBER OF SPECIES, SHANNON-WIENER DIVERSITY, AND NUMBER OF INDIVIDUALS

The number of species collected by activity trap and d-net sampling from the 23 sites were correlated ($r = 0.421$, $p = 0.045$), but the number of species collected by d-net sampling was significantly higher (13 to 54 taxa) with d-net sampling than activity trap sampling (9 to 39 taxa) ($p = 0.001$). Shannon-Wiener diversity was also correlated between the two sampling methods ($r = 0.463$, $p = 0.026$) but again significantly higher for the d-net sampling (1.20 to 3.26) over the activity trap sampling (0.27 to 2.77) ($p = 0.012$). In picking and sorting, attempts are made to pick at least 300 individuals, if less than 300 individuals are found, the entire sample needs to be picked. Both d-net and activity trap sampling resulted in about 12 sites with over 300 macroinvertebrate individuals picked, but only the activity trap method had sites (two) where the entire sample picked resulted in fewer than 100 individuals collected. The general consensus among benthic biologists is that many of the metrics break down when fewer than 100 individuals are included (Kerans et al. 1992; Barbour and Gerritsen 1996; Courtemanch 1996; Vinson and Hawkins 1996).

8.3.2 Higher level classifications (family and order)

The relative abundance and number of taxa occurrences (number of taxa X encounters) is presented (Table 8.3). As expected, the efficiency of sampling for one sampling method for individual taxon are different, yet what was surprising was that in many taxa, both sampling methods appeared to work consistently. For example, there were 1179 Hemiptera individuals from the activity traps and 651 from the d-net sampling. There were 77 occurrences from the activity trap and 62 occurrences from the d-net sampling. Diptera individuals and occurrences followed an opposite trend, but both methods appeared to collect this Order adequately. Some differences in collection efficacy occurred as well. For example, Branchiobdellidae were fairly common in the activity traps but were not collected by the d-net sampling method. Bivalvia were collected more efficiently with d-net than by activity trap.

8.3.3 Ecoregional differences among d-net and activity trap sampling (multidimensional scaling)

Non-metric mutltidimensional scaling (MDS) of macroinvertebrate data is presented from the five ecoregions by activity trap sampling (Figure 8.2). This figure demonstrates no ecoregional affiliation among the samples as shown using presence/absence data only. Relative abundance, additional data transformations, and cluster analysis showed a similar lack of ecoregional pattern. A similar lack of pattern was shown using presence/absence data from the d-net samples as well (Figure 8.3). Only samples collected from the Northern Lakes and Forests Ecoregion showed any clustering by MDS.

8.3.4 MULTI-DIMENSIONAL SCALING OF ACTIVITY TRAP AND D-NET\ SAMPLES

No differences were seen among the distribution of activity trap samples when compared with d-net samples (Figure 8.4). This figure identifies samples as either an activity trap or a d-net sample and there was no separation between the two types.

8.4 DISCUSSION

8.4.1 NUMBER OF SPECIES, SHANNON-WIENER DIVERSITY, AND NUMBER OF INDIVIDUALS

Both the number of taxa and Shannon-Wiener diversity of activity trap and d-net samples were correlated. In most cases, number of taxa and diversity were higher in the d-net samples. Thus, the d-net method would be more informative than the activity trap method. In addition, the d-net samples usually had more individuals in the sample and no instances of less than 100 individuals found in any of the total samples. Based on these observations, it appears that the d-net method might be preferred slightly, but there is not really any significant difference between the two methods.

Another consideration when selecting methods is that the activity trap method is generally used in conjunction with the fyke net sampling for fish assemblages. Simon et al. (2001) found that either electrofishing or fyke netting will produce similar results for moderate and large drowned river mouth wetlands, but that electrofishing was a superior method in small wetland classes. Thus, activity traps may not be a preferred method to use in small-drowned river mouth coastal wetlands since fyke nets would not be a preferred method.

8.4.2 HIGHER LEVEL CLASSIFICATIONS (FAMILY AND ORDER)

Both methods behaved relatively consistently regarding major groups. As expected, the activity trap sampled one group better than other groups. Our initial hypothesis was that the d-net would collect passive-dispersing organisms better, while the activity traps would collect predators and other organisms with diel movement patterns better than the d-net. However, no real pattern emerged to allow a preference of one type sampling method over another. We speculate that this may due to the selectivity of the operator and the experience of the crew in obtaining the specimens from a variety of habitats. Similar results were found in the wetlands of the Everglades, where selective gear found some differences among groups but was not significantly different to cause one gear (activity traps and d-nets) to be considered better than another (Turner and Trexler 1997).

8.4.3 ECOREGIONAL DIFFERENCES BETWEEN D-NET AND ACTIVITY TRAP SAMPLING (MULTIDIMENSIONAL SCALING)

Neither method allowed consistent and obvious discrimination of ecoregions by macroinvertebrate assemblages. Thus, one could use one method or the other with neither method strongly preferred. The lack of ecoregion differences is not unusual among macroinvertebrates. Unlike fish, invertebrates have a broad dispersal range that is not determined by watersheds. Several studies did not find a relationship with ecoregion for macroinvertebrates including Ohio (Ohio EPA 1989), southeastern United States

(Feminella 2000), Wisconsin (Weigel 2003), and Oregon (Sickle and Hughes 2000). This aspect simplifies the calibration of an index since a single index will be calibrated for lake-wide application, rather than ecoregion specific calibrations for various portions of the basin.

8.4.4 MULTI-DIMENSIONAL SCALING OF ACTIVITY TRAP AND D-NET SAMPLES: PROCESSING REQUIREMENTS

Sampling of macroinvertebrates using the d-net method takes from 30 to 45 minutes at each site using one person. The activity trap requires about 15 minutes to set a series of traps at a site, once the fyke-nets are set, and an additional fifteen minutes to retrieve the trap and preserve the sample. In addition, activity traps require a return trip to the site, which can take hours and would require an overnight stay in the area in order to retrieve the trap 24 h later. This is acceptable if the investigation also uses fyke-nets for fish sampling, but not if the only reason for a return to the site is for activity trap retrieval.

There are other practical considerations for selecting one method over another, and selection is certainly objective dependent (Resh et al. 1995). For example, the d-net method enables the field staff to collect a single sample within a site during a single visit. The logistic consequences of this method enables the field staff to continue collecting samples rather than being limited by the extent that they can be away from the first collection site, i.e., the staff must travel back to the first site that activity traps were set. Thus, staff may not want to venture more than 80-160 km from the first site. This can be problematic in coastal wetlands when the sites are often clumped, but distances between clusters of wetlands can be more than 300 km.

While the activity trap also requires some picking and cleaning time, the d-net sample is characterized by a large mass of debris that must be removed and the macroinvertebrates picked from the sample. Relative laboratory time for this process is in most cases less than 1-2 h per activity trap sample and up to 6 h for each d-net sample. The processing time for the d-net sample is minimized if there are personnel devoted to the picking and sorting process. These technicians can then provide quality assurance for each other's picked sample and greatly accelerate processing speed.

8.5 CONCLUSIONS

Few differences were observed between d-net and activity trap methods in adequately sampling drowned river mouth wetlands of Great Lake coastal wetlands. Both the number of taxa and Shannon-Wiener diversity of activity trap and d-net samples were correlated. Both methods behaved relatively consistently regarding major groups. As expected, the activity trap sampled some groups better than others. Our initial hypothesis was that the d-net would collect passive dispersing organisms better, while the activity traps would collect predators and other organisms with diel movement patterns better than the d-net. However, no real pattern emerged to allow a preference of one type sampling method over another. In most cases, number of taxa and diversity were higher in the d-net samples. Neither method allowed consistent and obvious discrimination of ecoregions by macroinvertebrate assemblages. Thus, one could use one method or the other with neither method strongly preferred. Finally, logistic constraints may preclude the use of activity traps since they require a minimum of a 24 h set to collect samples. The rapid use of the d-net has many advantages since it enables crews to collect a greater number of samples per day and maximizes crew efficiency, since there is not any requirement to return to a site to retrieve samplers.

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TABLE 8.1.

Wetland number (refers to number on Figure 8.1), name, latitude and longitude, size (ha), and ecoregion membership for drowned river mouth wetlands sampled in Lake Michigan during 2000. Ecoregion Codes: NLF = Northern Lakes and Forest; NCHF = Northern Central Hardwood Forest; SMNITP = Southern Michigan Northern Indiana Till Plain; CCBP = Central Corn Belt Plain; SWTP = Southeastern Wisconsin Till Plain.

Number	Wetland	Lattitude	Longitude	Area (ha)	Ecoregion
5	Carp Lake River	45.741	-84.833	11.7	NLF
56	Hog Island	45.74	-85.69	6.07	NLF
75	Arcadia Lake Wetland	44.489	-86.225	145	NCHF
80	Manistee River Wetland	44.258	-86.25	3706	NCHF
98	Bass Lake Wetland #2	43.811	-86.414	55	SMNITP
100	Pentwater River Wetland	43.758	-86.404	110	SMNITP
105	White River Wetland	43.45	-86.289	1579.7	SMNITP
113	Little Pigeon River	43.965	-86.215	17	SMNITP
114	Pigeon River Wetland	42.903	-86.182	36.4	SMNITP
129	Dunes Creek	41.65	-87.11	0.4	CCBP
167	Grand Calumet River Mouth Wetland	41.647	-87.558	2.8	CCBP
174	Dead River	42.443	-87.811	40.4	CCBP
191	Kewaunee River Wetland #2	44.475	-87.514	145.7	SWTP
253	Keyes Creek Wetland	44.831	-87.572	28.3	SWTP
258	Fox River	44.535	-88.017	12.1	SWTP
262	Dead Horse Bay Wetland #1	44.61	-88.02	8.1	NCHF
274	Little Tail Point	44.68	-88.	64.7	NCHF
283	Thomas Slough	44.883	-87.85	283.4	NCHF
299	Portage Creek	45 7	-87.083	526.3	NLF
305	Days River	45.883	-87.	23.4	NLF
405	Brevort Area Wetland	46.018	-85.033	202.4	NLF
524	East Twin River	44.158	-87.57	80.9	SWTP

TABLE 8.2.

Number of taxa, individuals counted, and Shannon-Wiener diversity found using d-net sampling and activity traps at 23 coastal wetland sites.

Site Name	D-Net Sampling			Activity Trap Sampling		
	Taxa	Individuals	H'	Taxa	Individuals	H'
Arcadia Lake	38	294	2.48	31	190	2.55
Dead River	31	312	1.89	23	297	1.43
Little Pigeon	28	333	1.98	27	203	2.53
Manistee River	41	372	2.60	33	225	2.36
Carp Lake River	53	317	3.26	25	229	2.13
Little Tail Point	44	317	3.04	36	235	2.50
Portage Creek	20	172	1.80	11	66	1.54
Grand Calumet River	13	284	1.62	9	317	0.71
Pigeon River	31	296	1.85	18	106	2.18
Brevort River	55	273	2.91	19	395	0.85
Days River	42	296	2.95	15	113	2.01
Dunes Creek	29	323	2.05	33	302	2.22
Fox River	21	136	2.01	12	301	0.82
Thomas Slough	28	310	2.32	39	305	2.77
East Twin River	23	270	1.59	21	310	1.35
White River	50	334	2.87	34	306	1.96
Pentwater Marsh 1	28	321	2.09	16	310	1.36
Pentwater Marsh 2	21	230	1.37	30	337	2.27
Bass Lake	27	144	2.32	20	223	1.70
Dead Horse Bay	23	364	1.55	22	189	1.79
Keyes Creek	44	304	2.84	34	322	2.74
Kewaunee River	17	255	1.20	10	335	0.27
Hog Island	54	329	2.91	21	72	2.50

TABLE 8.3.

Comparison of macroinvertebrate taxa collected by activity trap and d-net sampling. Columns are number of individuals and number of species occurrences (number of species X occurrence) from Lake Michigan drowned river mouth coastal wetlands.

	Number of Individuals		**Number of Species Occurrences**	
TAXA	Trap	d-net	Trap	d-net
TURBELLARIA	28	1	5	1
CNIDARIA	5	1	2	1
OLIGOCHAETA	419	158	29	30
BRANCHIOBDELLIDAE	127	0	5	0
HIRUDINEA	6	4	5	4
BRANCHIURA	1	0	1	0
COPEPODA	31	79	6	13
OSTRACODA	191	55	10	11
CLADOCERA	130	528	16	26
ISOPODA	106	226	14	16
AMPHIPODA	1684	1376	51	47
DECAPODA	34	7	9	4
ARANEAE	4	4	3	3
ACARI	83	100	17	25
COLLEMBOLA	30	64	7	9
EPHEMEROPTERA	320	584	42	57
ODONATA	65	162	21	38
PLECOPTERA	2	3	1	3
HEMIPTERA	1179	651	77	62
TRICHOPTERA	231	260	17	37
LEPIDOPTERA	1	7	1	4
MEGALOPTERA	0	1	0	1
COLEOPTERA	123	212	60	70
DIPTERA	942	1825	115	281
GASTROPODA	328	459	55	45
BIVALVIA	1	21	1	9

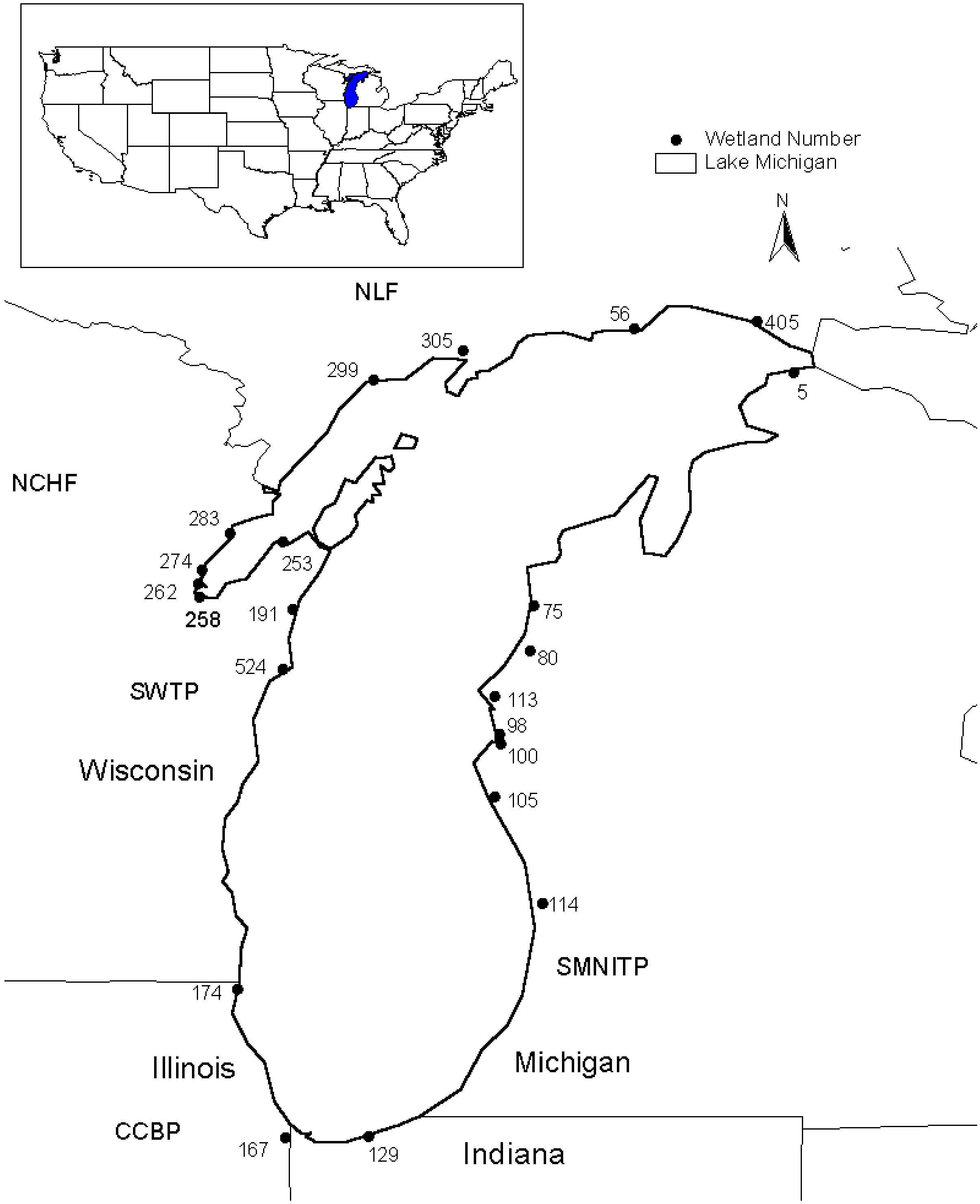
Figure Captions

Figure 8.1. Distribution of drowned river mouth coastal wetlands of the Great Lakes for each of the five Lake Michigan ecoregions.

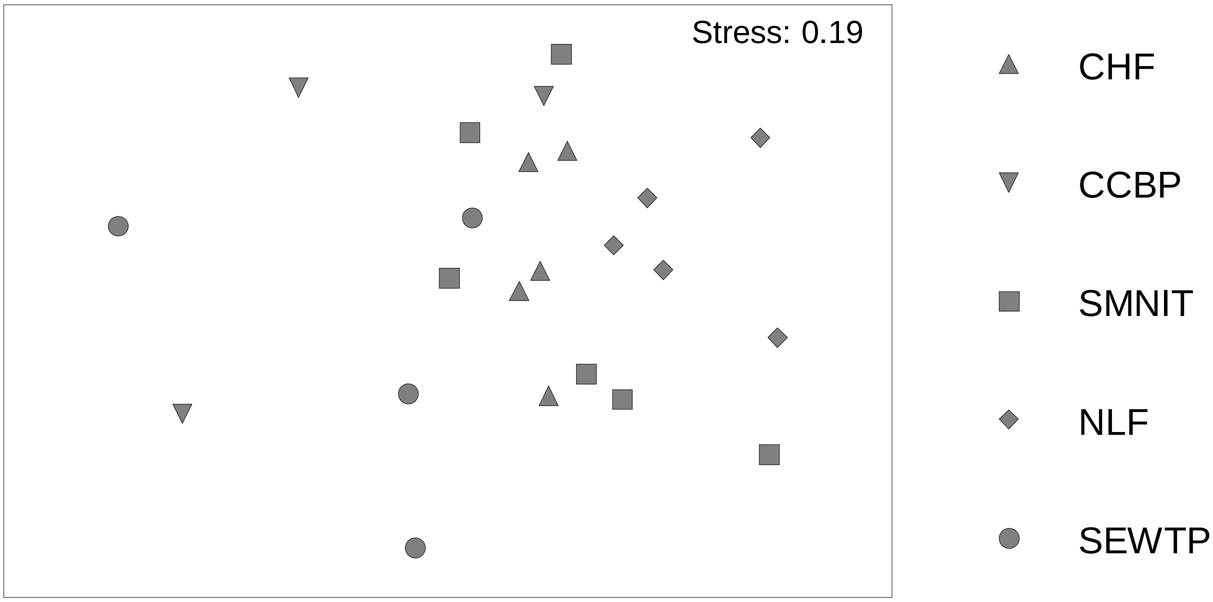
Figure 8.2. Non-metric multidimensional scaling of Lake Michigan activity trap samples using presence/absence data (symbols refer to ecoregions as identified on Figure 8.1 and caption to Table 8.1).

Figure 8.3. Non-metric multidimensional scaling of Lake Michigan d-net samples using presence/absence data (symbols refer to ecoregions as identified on Figure 8.1 and caption to Table 8.1).

Figure 8.4. Non-metric multidimensional scaling of Lake Michigan activity trap and d-net net samples combined.


Wetland Number
Lake Michigan
N
NLF
NCHF
SWTP
SMNITP
CCBP
Wisconsin
Illinois
Michigan
Indiana
56
405
305
299
5
283
274
262
258
253
191
524
75
80
113
98
100
105
114
174
167
129


Stress: 0.22
CHF
CCBP
SMNITP
NLF
SEWTP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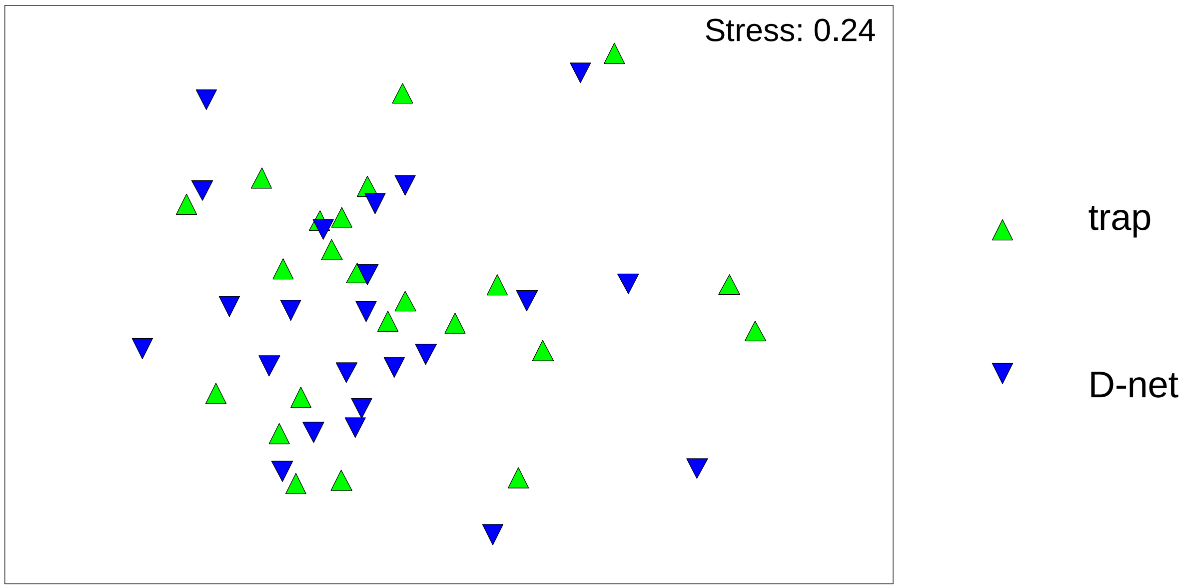
CHF
CCBP
SMNIT
NLF
SEWTP


Stress: 0.24
trap
D-net

12 Classification of Freshwater Fish Species of the Great Lakes Basin for Use in the Development of Indices of Biological Integrity for Coastal Wetlands

Thomas P. Simon, Ronda L. Dufour, Roger F. Thoma, Douglas Carlson

CONTENTS

12.1 INTRODUCTION

The index of biotic integrity (IBI) has evolved into a family of indices that require calibration to properly function (Karr et al., 1986; Simon, 1999a; Simon, 2000). This calibration requires considerably information with respect to zoogeographic, regional, and biological interaction constraints (Karr, 1981; Simon, 2000). Each component of this information is necessary to develop and validate candidate metrics for inclusion in any modification of the IBI. This summarized literature information is required to classify every fish species into a series of structural and functional metrics, which are used to develop an Index of Biotic Integrity (IBI). Species are classified according to their native or introduced origin, trophic or feeding status, aquatic habitats, reproductive guild, and tolerance to environmental degradation (Fausch et al., 1990; Schlosser, 1990; Halliwell et al. 1999).

The intent of this chapter is to summarize information on the origin, current distribution patterns, trophic guilds, characteristic macrohabitats, fish species tolerance to environmental degradation, obligate status as lake inhabiting species, and reproductive guilds. We recognize that there are significant differences between species assemblages across the Great Lakes basin, from east-west clines, i.e., Lake Ontario to Lake Michigan and north-south clines, i.e., Lake Erie to Lake Superior. These differences in species richness, distribution, and colonization potential reflect the current condition of Great Lakes coastal wetlands.

The impact of non-indigenous and exotic fish species has had a dramatic effect on the biological integrity of Great Lakes coastal wetlands (Mills et al., 1993; Simon and Moy, 2000). Karr and Dudley (1985) defined biological integrity as “the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region". Thus, aquatic communities have been used as indicators of condition to determine the sensitive measure of site-specific condition (Karr, 1987). As we observe increases in the non-indigenous and exotic species at a site, this can be considered "biological pollution" (McKnight, 1993) and a loss of biological integrity.

This classification of freshwater fish provides a synthesis of current information to further advance IBI development in the Great Lakes basin.

12.1.1 Overview of the Index of Biotic Integrity for Coastal Wetlands

The IBI was originally developed for small wadeable streams of the Midwestern United States (Karr, 1981; Karr et al., 1986). The need for a rapid assessment tool that can combine several hierarchial levels, i.e., ecosystem, community, population, individual, into a single composite number has required that the original "stream" version be modified repeatedly for use in other areas (Simon and Lyons, 1995; Hughes and Oberdorf, 1999), resource types (Lyons et al., 2001; Emery et al., 2003) and among other indicator assemblages (DeShon 1995; Simon et al. 2001; Bryce et al. 2002).

Fish assemblages in wetlands have received very little attention as indicators of environmental quality (Simon et al., chapter 17) and only a few papers have attempted to develop an assessment index for coastal wetlands. Simon et al. (chapter 17) described the types of studies that have been conducted for assessing coastal wetlands of the Great Lakes and placed them into three categories: 1) long-term ecological monitoring, 2) method comparison, and 3) early life history use. Limited information from these studies would be sufficient to calibrate an index of biotic integrity for Great Lakes coastal wetlands.

The first attempt to calibrate an index for coastal wetlands was in southern Lake Michigan (Simon 1998; Simon and Stewart 1998). Simon (1998) and Simon and Stewart (1998) developed and tested an index of biotic integrity for fish assemblages in protected wetlands using the hydrogeomorphic classification of Keough et al. (1999). Thoma (1999) developed an IBI for nearshore and coastal wetlands of Lake Erie. Thoma coined the term “lacustuaries” for coastal wetland areas where the riverine and lake interfaced. This calibration is validated elsewhere in this document (Thoma and Simon, chapter 23). Simon et al. (2001) further modified the protected wetland index for vernal ponds and small panne wetlands using fish, amphibians, and crayfish assemblages. Neither of the protected wetland calibrations was adequate for application to drowned river mouth situations throughout the Great Lakes. However, they are suitable for the regions from which they were calibrated and developed.

12.2 METHODOLOGY

We evaluated fish distribution, ecology, and tolerance information from the states bordering the Great Lakes and evaluated classifications found in Illinois (Hite and Bertrand, 1989), Ohio (Ohio EPA,1989: Thoma, 1999), Indiana (Simon, 1991), Wisconsin (Lyons, 1992), Minnesota (Niemela and Fiest, 2000), Michigan (Michigan DNR, 1989), and New York (Halliwell et al., 1999). This information was checked against species occurrence accounts found in Lee et al. (1980).

12.2.1 INFORMATION SOURCES

Natural history and distribution information for 124 species of freshwater fish from a variety of published and unpublished records from museums, state ichthyology texts (Hubbs and Lagler, 1959; Scott and Crossman, 1973; Smith 1979; Gerking 1945; Lee et al. 1980), and natural resource agencies is compiled for each of the Great Lakes (Tables 12.1-12.2). Fish macrohabitat information and environmental tolerance information are derived from state biological criteria development (Hite and Bertrand, 1989; Ohio EPA,1989; Michigan DNR, 1989; Simon, 1991; Lyons, 1992; Halliwell et al., 1999; Thoma, 1999; Niemela and Fiest, 2000), while trophic and feeding ecology (Goldstein and Simon, 1999) and reproductive guilds (Simon, 1999b) were based on extensive published literature reviews. A compilation of all species of fish reproducing in the Great Lakes, including landlocked forms of marine origin, such as introduced (e.g., rainbow smelt, alewife, white perch) and naturally occurring forms (e.g., sticklebacks and killifish). Numerous fish species have been historically reported from the Great Lakes, whose occurrence is either not recent, episodic, or of short duration (e.g., extirpated, exotic, or estuarine) were not included in our list (Table 12.1-12.2).

12.3 FISH SPECIES DISTRIBUTIONS

Species that are considered native in one of the Great Lakes (e.g., sea lamprey *Petromyzon marinus*, white perch *Morone americana*, Atlantic salmon *Salmo salar*), but considered non-native elsewhere must be considered in calibrating several of the IBI metrics. For example, the total number of species and relative abundance metrics both remove non-indigenous and exotic species from the species lists. Proportional metrics are compiled for Lake Erie without non-indigenous and exotic species (Thoma 1999), while for some applications non-indigenous and exotic species are important indicators at the lowest extremes of biological integrity.

Exotic species that should be excluded from all coastal wetland calibrations of the Great Lakes would include three-spine stickleback *Gasterosteus aculeatus*, round goby *Neogobius melanostomus*, grass carp *Ctenopharyngodon idella*, goldfish *Carassius auratus*, rudd *Scardinius erythrophthalmus*, common carp *Cyprinus carpio*, Caspian carps genus *Hypophthalmichthys* spp., ruffe *Gymnocephalus cernuus*, and tubenose goby *M* . These species have been introduced by ballast water release (Mills et al., 1993; Simon and Vondruska, 1991; Pratt et al., 1992; Simon et al. 1998; Simon et al. 2002), intentional stocking (Cooper, 1987), or as fisheries management tools.

12.3.1 Lake Ontario

Halliwell et al. (1999) described the status of native and non-indigenous species for the Atlantic Coast. Species that are considered non-indigenous or exotic to Lake Ontario include green sunfish *Lepomis cyanellus*, margined madtom *Noturus insignis*, and three-spine stickleback *Gasterosteus aculeatus* (Table 12.1). On the contrary, sea lamprey, rainbow smelt, alewife, Atlantic salmon, and white perch would be considered native to Lake Ontario. The spread of these species into the rest of the Great Lakes was facilitated by the construction of the Welland Canal.

12.3.2 LAKE ERIE AND LAKE ST. CLAIR

Thoma (1999) described a variety of species that would be considered non-indigenous to Lake Erie and St. Clair (Table 12.2). The list includes sea lamprey, shortnose gar *Lepisosteus platostomus*, goldeye *Hiodon alosoides*, skipjack herring *Alosa chrysochloris*, alewife *Alosa psuedoharengus*, gizzard shad *Dorosoma cepedianum*, and threadfin shad *D. petenense*. In addition, all of the Pacific salmon genus *Oncorhynchus*, brown trout *Salmo trutta*, rainbow smelt *Osmerus mordax*, bigmouth buffalo *Ictiobus cyprinellus*, smallmouth buffalo *I. bubalus*, river carpsucker *Carpiodes carpio*, highfin carpsucker *Carpiodes velifer*, ghost shiner *Notropis buchanani*, American eel *Anguilla rostrata*, eastern banded killifish *Fundulus diaphanus diaphanus*, western mosquitofish *Gambusia affinis*, striped bass *Morone saxatilis*, white perch *Morone americana*, orangespotted sunfish *Lepomis humilis*, and three-spine stickleback *Gasterosteus aculeatus*. In addition, a species that was not on Thoma's list include walleye *Sander vitreus*. The native walleye to Lake Erie was the blue pike *S. v. glaceum*. This subspecies is thought to be extirpated as a result of overfishing. In order to fill this niche gap, the current genetic form *S. v. vitreus* is from stock from the Mississippi River.

12.3.3 LAKE HURON

The ruffe occurs in northern Lake Huron (A. Bowen, personal communication), while the round goby and tubenose goby occur in the St. Clair River and the lower portions of Lake Huron near Port Huron, Michigan (Jude et al., 1995). Species mentioned as well as

Pacific salmon, *Oncorhynchus* spp., occur in the Lake, as does carp, goldfish, sea lamprey, white perch, alewife, threadfin shad, and gizzard shad.

12.3.4 LAKE MICHIGAN

Page and Laird (1993) and Simon et al. (1998) described the status of non-indigenous species in the Lake Michigan basin. Currently, there are 20 species of freshwater fish that are either considered exotic or non-indigenous.

12.3.5 LAKE SUPERIOR

The invasion of round goby, ruffe, and sea lamprey has caused widespread changes in the nearshore assemblage of Lake Superior. The stocking of Pacific salmonids has created an artificial assemblage of exotic and nonindigenous species.

12.4 NATIVE FISH SPECIES

Native species have been referred to by a variety of terms, i.e., resident, resident indigenous, resident naturalized, or native. The term “*resident*” species is a natural or adaptive assemblage of fish (Karr and Dudley, 1985) represented by several age classes that populate similar aquatic environments (Ricklefs, 1990). *Resident indigenous* species is equivalent to native species. These are species that are naturally occurring native species populating suitable aquatic habitats, e.g., brook trout, slimy sculpin, and longnose dace in coldwaters; brown bullhead, pumpkinseed, and golden shiner in warmwater habitats. *Resident naturalized* species are well-established non-native species populating suitable habitats; i.e., exotic brown trout in coldwaters, smallmouth bass in coolwaters.

Non-resident transient species are non-populating fish species found to occur in unsuitable aquatic habitats; e.g., rainbow trout in the Grand Calumet River during select times of the year. *Nonresident stocked* species are nonpopulating fish species that are introduced for a recreational fishery only; i.e., salmonid stocks.

Species were designated as native, non-indigenous, or exotic for each of the Great Lakes (Table 12.1 and 12.2). Species native or alien status was determined from published literature for each of the Great Lakes (Mather, 1886; Evermann and Kendall, 1902a, b; Bean, 1903; Greeley 1927 to 1940; Greeley and Bishop, 1932; Greeley and Greene, 1931; George, 1981; Smith, 1985; Bouton 1994; Whittier et al. 1997) and were based on distributions contained in Lee et al. (1980). Native species status was incorporated into several new metrics including the number or percentage of Great Lakes obligate species, percent exotic or non-indigenous species, deletion of exotic and non-indigenous species from functional percentage metrics, and removal from relative abundance estimates (Thoma 1999; Simon et al., chapters 18-24).

12.5 INTRODUCED FISH SPECIES

Introduced fish species found in the Great Lakes include exotic and European species (e.g., common carp, goldfish, brown trout, rudd, round goby, tubenose goby, ruffe, and oriental weatherfish). Rudd was originally introduced by the baitfish industry and are known from the Grand Calumet River in Indiana (Simon et al. 1998).

Pacific salmon, i.e., coho salmon *Oncorhynchus kisutch*, rainbow trout *O. mykiss*, chinook salmon *O. tshawytscha*, and pink salmon *O. gorbusha*, have all been introduced into the Great Lakes. These species were stocked for recreation and to control the alewife populations that had invaded through the Welland Canal. In addition, brown trout *Salmo trutta* have also been intentionally stocked in the Great Lakes and in many of the tributary streams they are well established and self-sustaining.

12.6 NATURALIZED FISH SPECIES

Several fish species were introduced as either sportfish or foodfish throughout the Great Lakes over a century ago (Smith, 1985). Naturalized species are those species that are sufficiently established that they are capable of continuing as a self-sustaining population without supplemental stocking. In the northeastern Great Lakes, species such as smallmouth bass *Micropterus dolomieu*, largemouth bass *M. salmoides*, rock bass *Ambloplites rupestris*, bluegill *Lepomis macrochirus*, black crappie *Pomoxis nigromaculatus*, yellow bullhead *Ameiurus natalis*, and the exotic brown trout are all widespread and often common to abundant (Halliwell et al. 1999).

12.7 INDEX OF BIOTIC INTEGRITY CATEGORIES

To develop indices of biological integrity, it is necessary to assign each fish species to a series of guild categories (Karr et al. 1986; Simon 1999a). Karr et al. (1986) originally designed the IBI to represent compositional guilds, trophic guilds, and tolerance guilds. Since then many researchers have included macrohabitat guilds, reproductive guilds, and other structural guilds as replacement metrics (Simon and Lyons, 1995; Hughes and Oberdorf, 1999).

12.7.1 TROPHIC GUILDS

Karr et al. (1986) originally recognized fish species from four feeding categories: herbivores, omnivores, insectivores, and piscivores. As the IBI was modified to areas outside of the Midwestern United States, problems with definitions of omnivores and top level piscivores led to confusion on how to classify species (Goldstein and Simon, 1999).

Gerking (1994) formulated the structure for defining all species worldwide into a series of five trophic categories, i.e., herbivores, detritivores, planktivores, invertivores, and carnivores. Goldstein and Simon (1999) used this approach to formulate the current trophic classification scheme for North American species and included trophic subclass and trophic mode categories to more narrowly define species feeding ecology. This is the approach we have utilized to define species feeding ecology for Great Lakes coastal wetlands.

12.7.2 MACROHABITAT GUILD CLASSIFICATION

Fish species have been classified along two macrohabitat variables based on waterbody type and temperature regime (i.e., cool, cold, warmwater). Some species are not restricted to a single macrohabitat class, while all species tend to prefer an optimum range of environmental conditions within this range of aquatic habitats (Eaton et al., 1995).

We have used macrohabitat class concepts to classify species among two lake classes, whether they are obligate Great Lakes species or whether they are obligate lake habitat species. Obligate Great Lake species are those signature species that when one refers to the Great Lakes one immediately considers, e.g., lake sturgeon *Acipenser fulvescens*, lake trout *Salvelinus namaycush*, coregonids genus *Coregonus.* Obligate lake species are those taxa that predominate in lentic conditions. These families include a variety of species including brown bullhead, Iowa darter *Etheostoma exile*, yellow perch *Perca flavescens*, warmouth *Lepomis gulosus*. We have not chosen to place species into thermophilic guilds since we believe that our calibrations for each individual Great Lake takes this into consideration.

12.7.3 FISH SPECIES TOLERANCE CLASSIFICATIONS

The recognition of stress factors and the use of fish assemblages as indicators of environmental degradation have increased the discriminate capacity of the IBI to detect varying degrees of stress (Fausch et al., 1990). Three general classes of fish species tolerance to environmental perturbations are used: intolerant, intermediate or moderate, and tolerant.

Typically, a variety of fish tolerance classifications are available in the literature and they seem to contradict each other, however, we have chosen to use different classifications for each of the various Great Lakes (Karr et al. 1986; Ohio EPA 1989; Whittier et al., 1987; Plafkin et al., 1989). To presume that the same species will respond in the same predictable manner across different parts of the country to the same habitat disturbance is not necessarily what has been observed (Simon, 2003). We recognize that the ichthyologists that are working in these regions know more about the sensitivities of the species in their area, thus differences in tolerances may be a reflection of changing sensitivities on the periphery of a species distribution or the interactions with other sympatric species (Halliwell et al., 1999). Thus, we have chosen to use the following classifications for use in the Great Lakes: Lake Ontario, St. Lawrence River, and Niagara River (Halliwell et al. 1999); Lake Erie (Thoma, 1999), Lake St. Clair, Lake Michigan, and St. Clair River (Simon, 1991), and Lake Superior, Lake Huron, and St. Marys River (Lyons, 1994).

12.7.4 REPRODUCTIVE GUILD CLASSIFICATIONS

Reproductive guild classification are a new "generation" group of replacement metrics that have been used to substitute the original hybrid and sometimes top carnivore metrics. The reproductive guild classification was based mainly on form and function in early developmental intervals, on preferred spawning grounds, and on features of reproductive behavior (Kryzhanovsky, 1949; Balon 1975, 1978, 1981, 1985). Simon (1999) compiled a complete listing of reproductive guilds for Midwestern fish species and properly classified them based on spawning habitat placement, larval development, and reproductive behavior. We have chosen to use this classification for application to Great Lakes coastal wetlands.

12.8 CONCLUSIONS

The need to consistently classify species among a variety of guild memberships is a fundamental component of the IBI (Karr et al. 1986; Simon and Lyons, 1995). This paper describes the rationale and the selection process for classifying the 124 species of freshwater fish in the Great Lakes into native origin, macrohabitat, trophic guilds, tolerance, and reproductive guilds. This information came from years of literature review and searching to determine the proper placement of species.

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13 Development of an Activity Trap Macroinvertebrate Index of Biotic Integrity for Lake Michigan Coastal Wetlands

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CONTENTS

13.1 INTRODUCTION

The need for a Lake Michigan Index of Biotic Integrity for coastal wetlands has been established as part of a major funding effort by U.S. Environmental Protection Agency to the U.S. Fish and Wildlife Service. We have established that there are no ecoregional differences among ecoregions in Lake Michigan for fish (Simon et al., in prep b) and macroinvertebrate assemblages (Stewart and Simon , chapter 16, this volume). Therefore, we will combine all sites into one lake-wide index. Designing an effective sampling regime using appropriate technology requires consideration of the taxa to be expected and the labor requirements for each type sampler (Turner and Trexler, 1997). This first-year sampling for the overall project was to determine if activity trap or d-net (sweep) net sampling was to be used for all of this four Great Lake study (Simon, 2000; Stewart and Simon, chapter 15, this volume). We found that while there were some differences among macroinvertebrate groups in their preference for one type of sampling over another, for the most part, both activity traps and d-nets sampled macroinvertebrate assemblages adequately and either can be used to calibrate an index of biotic integrity (Stewart and Simon, chapter16, this volume). The main reason to choose one macroinvertebrate method over the other would be whether you are using a fyke-net or electroshocking for fish sampling. Use of the activity trap and fyke net requires setting the traps for 24 hours and returning to the site for retrieval. In situations where long distances are involved (such as establishing regional biocriteria for the Great Lakes), we decided that remaining in the vicinity for retrieval was too constrictive as the study required distance of up to 300 km between sites. However, the fact remains that some agencies prefer fyke-nets and along with them activity traps may be a useful alternative.

Multimetric indices have been well demonstrated and used in a variety of situations. Those using macroinvertebrates have not been well developed in Great Lakes wetlands with the exception of two papers from northern Lake Huron that tested potential metrics (Kashian and Burton, 2000) and the development of a preliminary index for Lake Huron (Burton et al. 1999). In addition, Lougheed and Chow-Fraser (2002) developed a zooplankton index of wetland quality in the Great Lakes. Lougheed and Chow-Fraser (2002) found their index, based on water quality and zooplankton associations with water quality through multivariate analyses to be more useful than diversity indices and measures of community structure. The objectives of this paper are to develop a Lake Michigan Index of Biotic Integrity for macroinvertebrate assemblages using activity traps. While only 22 wetlands were sampled for a total of 23 samples, we decided that would be enough to roughly sketch out the index. Further development and testing of the index will require additional sampling, which is beyond the scope of this preliminary study. In addition, we explored a much greater list of potential metrics than have been previously used for development of macroinvertebrate indices thereby more closely following the intentions of the original IBI developed for fish.

13.2 METHODS

13.2.1 STUDY AREA

In order to compare sampling methodology, 22 drowned river mouth wetlands (Figure 13.1, Table 13.1) (Keough et al. 1999) in Lake Michigan were used as part of a pilot study designed to help select sampling strategies for a more extensive project. These drowned river mouth wetlands were randomly chosen based on stratification so that wetlands were located in each of the five ecoregions of Lake Michigan (Omernik 1987). By stratifying sampling in wetlands among the five ecoregions, samples were collected from widely divergent habitat types and wetland sizes with a wide range of anthropogenic disturbances. See Simon et al. (in prep. a) for the rationale behind site selection.

13.2.2 SAMPLING METHODS

13.2.2.1 Field sampling procedures

Macroinvertebrate sampling was with activity traps (Swanson 1978, Wilcox et al. 1999) positioned in the dominant habitat types such that five pairs of traps were set for 24 hours. Basically, the activity trap is a screen mesh cylinder 640 micron that is formed into the shape of a minnow trap. Two of these traps were sunk to the bottom and attached to a rod that was inserted into the sediment, so that the orientation of both traps was horizontal and parallel to shore. Five of these pairs were set in and among the aquatic vegetation for 24 hours and retrieved the next day. Upon retrieval, the contents of the bottle were poured through a 500-µ sieve and all trap samples were composited for a site, preserved in 70% ethyl alcohol, and transported to the laboratory.

13.2.2.2 Laboratory processing, identification, and data reduction

In the laboratory, each sample was washed and placed in a grid pan with a hundred 50 mm x 50 mm squares. Subsamples were randomly chosen using a random number table, picked so that no organisms remained in each square, and sorted until at least 300 individual invertebrates were found. Records were kept of the number of grids picked and sorted for a later determination of estimated abundance. At the conclusion of the 300 organism sorting, a 5-minute pick of large and rare invertebrates was done and stored separately. Organisms were identified to the lowest practical taxonomic level using standard literature (Snider 1967, Simpson and Bode 1980, Weiderholm 1983, Pennak 1989, Wiggins 1995, Merritt and Cummins 1996). Chironomids were identified to subfamily or tribe.

13.2.2.3 ANALYSES

Over 100 candidate metrics were computed in the following major categories: structure (such as number of species), functional feeding groups, health including tolerance and sensitivity of taxa, number of life stages, and habitat groups. These were made to follow Karr's original intention to make a multimetric index of several levels of the assemblage. This intention has not been fulfilled in most macroinvertebrate indices since most use only number of taxa, functional feeding groups, and several indices. As metrics, we urge the discontinued use of available indices such as Shannon-Wiener, Hilsonhoff biotic index, and others. These are summations of community level data and can and should be used on their own to describe a community and not used as metrics and added to a multimetric index. The use of a summation index in a summation index probably invalidates any inherent meaning in the indices and should be avoided.

When faced with over 100 potential metrics, one needs to eliminate metrics through a series of steps, so that you are left with 10 to 12 metrics at the end of the selection process. The steps that we used are as follows: 1) Remove those metrics that mostly have missing data. If over 50% of the sites have no data, then that site is eliminated. 2) Next, perform a range test. If all the site variation is contained within a very small range, for example, one vs two Ephemeroptera species, then that metric should be eliminated. 3) Perform a box plot of the data and a test of skewness. By examining these two tests, you see what the data looks like in the box plot. When the data is very much skewed to one side or the other it would not make a good metric and is eliminated. 4) Examine the metric to determine if there is a relationship with an anthropomorphically-related pattern, such as site quality. If no relationship exists, then that metric should be eliminated. In this study, we relied on site quality assessments made by two papers in this volume. The plant IBI (PIBI) by Albert et al., (chapter XX, this volume) and the fish IBI by Simon et al. (Chapter 19, this volume), along with our professional judgement regarding the three best and worst sites based on multiple visits to the sites.

After metric elimination, each metric was plotted against wetland wetted width. The final metric step was to standardize the data so that the metrics are on the same scale so that they can be added. In other words, you can't add 43% with 17 species to make a multimetric index. A maximum metric line (MML) is drawn parallel to the x-axis at the highest measurement recorded for each metric. Finally, the data below the MML line is trisected, and each third of the graph formed is scored a one, three, or five as the case may be. The five is assigned to the best third of the sites, the three to the next third, and a one to the worst third. In some cases, the metric is reverse scored, for example, the number of tolerant taxa increases with the lower quality of the wetland. A score of zero was assigned when none of a positive scoring metric existed. The final multimetric index (M-IBI) is formed by adding all of the individual metric scores for each site.

13.3 RESULTS

Over 100 potential metrics were calculated (Table 13.2). The reduction steps described above reduced this number down to 10 metrics (Figure 13.2). These included representatives from species composition metrics, tolerance vs. sensitive metrics, life history, and trophic guilds. Macroinvertebrate IBI scores computed for individual wetlands ranged from 40 to 7 with the Dead River and one of the Pentwater Marsh sites both scoring 40, the highest score received by any of the sites. The Grand Calumet River wetland scored 7, and the Fox River wetland scored an 8, these were the two lowest scored sites.

Narrative categories (poor, fair, good, exceptional) for the M-IBI were scored as poor (0-20), fair (21-30), good (31-40), and exceptional (41-50). The poor and good narrative category each comprised 26% of the wetland sites. The fair category made up 47% of the wetlands sites and no wetland sites were scored in the excellent range. Sites located near the dividing lines for each narrative category should be resampled before final scores are determined. Both replicates for the Pentwater Marsh wetland scored as fair.

13.4 DISCUSSION

Several of the metrics that we selected for the M-IBI were also suggested by previous work (Kashian and Burton 2000). These included number of EPT taxa and the percent individuals as predators . As explained above and because of their erratic response to several antropogenic disturbances, supported by Kashian and Burton (2000), and suggested by Karr and Chu (1997), we did not attempt to use any diversity indices.

Our inclusion of number of tolerant taxa and the percent individuals as sensitive taxa, is a departure from related studies. Extensive surveys of the literature identified a higher proportion of tolerant versus sensitive taxa in our data set making this a desirable and informative metric. Our final metric, percent univoltine abundance was included as a means to point out that life history data for many more invertebrates is needed to fully develop Great Lake wetland indices.

Since 22 sites were sampled by macroinvertebrate activity traps, this index is ready for additional confirmation and continued validation. In the absence of further work, we suggest using this index with caution. No sites scored in the excellent range. This was not unexpected as most sites in Lake Michigan have been or are being degraded to some extent.

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Table 13.1.

Wetland number (refers to number on Figure 13.1), name, latitude and longitude, size (ha), and ecoregion membership for drowned river mouth wetlands sampled in Lake Michigan during 2000. Ecoregion Codes: NLF = Northern Lakes and Forest; NCHF = Northern Central Hardwood Forest; SMNITP = Southern Michigan Northern Indiana Till Plain; CCBP = Central Corn Belt Plain; SWTP = Southeastern Wisconsin Till Plain

Number	Wetland	Lattitude	Longitude	Area (ha)	Ecoregion
5	Carp Lake River	45.741	-84.833	11.7	NLF
56	Hog Island	45.74	-85.69	6.07	NLF
75	Arcadia Lake Wetland	44.489	-86.225	145	NCHF
80	Manistee River Wetland	44.258	-86.25	3706	NCHF
98	Bass Lake Wetland #2	43.811	-86.414	55	SMNITP
100	Pentwater River Wetland	43.758	-86.404	110	SMNITP
105	White River Wetland	43.45	-86.289	1579.7	SMNITP
113	Little Pigeon River	43.965	-86.215	17	SMNITP
114	Pigeon River Wetland	42.903	-86.182	36.4	SMNITP
129	Dunes Creek	41.65	-87.11	0.4	CCBP
167	Grand Calumet River Mouth Wetland	41.647	-87.558	2.8	CCBP
174	Dead River	42.443	-87.811	40.4	CCBP
191	Kewaunee River Wetland #2	44.475	-87.514	145.7	SWTP
253	Keyes Creek Wetland	44.831	-87.572	28.3	SWTP
258	Fox River	44.535	-88.017	12.1	SWTP
262	Dead Horse Bay Wetland #1	44.61	-88.02	8.1	NCHF
274	Little Tail Point	44.68	-88.	64.7	NCHF
283	Thomas Slough	44.883	-87.85	283.4	NCHF
299	Portage Creek	45 7	-87.083	526.3	NLF
305	Days River	45.883	-87.	23.4	NLF
405	Brevort Area Wetland	46.018	-85.033	202.4	NLF
524	East Twin River	44.158	-87.57	80.9	SWTP

Table 13.2.

Candidate macroinvertebrate metrics with description (m=data from main sample; l/r=data from large/rare sample). Eliminations were based on high percentage of zeros = Z, failing the range test = R, failure to produce a balanced box plot = BP, highly skewed data = S, or failure to significantly separate the three best from the three worst sites (G/B) based on a combination of professional judgement, IBI and PIBI scores from companion papers in this volume.

Metric	Description	Acc./Rej.	
Number of Taxa	**Number of unique taxa (m,l/r)**	**Accepted**	**Sp. Comp**
Abundance/sample	Number of organisms per sample (m)	G/B (Alt.)	Abund.
# Ephemeroptera Taxa	Number of Ephemeroptera taxa (m,l/r)	BP,S	
# Trichoptera Taxa	Number of Trichoptera taxa (m,l/r)	R	
# Plecoptera Taxa	Number of Plecoptera taxa (m,l/r)	Z	
# Diptera Taxa	Number of Diptera taxa (m,l/r)	G/B	
# Odonata Taxa	Number of Odonata taxa (m,l/r)	BP,S,Z,R	
% Oligochaeta	Percent Oligochaeta abundance (m)	BP/S,Z	
% Odonata	Percent Odonata abundance (m)	BP,S,Z	
% Ephemeroptera	Percent Ephemeroptera abundance (m)	S	
% Tricoptera	Percent Trichoptera abundance (m)	BP,S,Z	
% Plecoptera	Percent Plecoptera abundance (m)	Z	
% Coleoptera	Percent Coleoptera abundance (m)	G/B	
% Diptera	Percent Diptera abundance (m)	BP,S	
% non-chironomid Diptera	Percent non-chir. Diptera abund. (m)	BP,S	
% Chironomids	Percent Chironomids abundance (m)	BP,S	
% Ind. as Crust. & Moll.	**Percent Crust. and Moll. abund. (m)**	**Accepted**	**Sp. Comp**
% Gastropoda	Percent Gastropoda abundance (m)	S	
% Ind. as Amphipoda	**Percent Amphipoda abundance (m)**	**Accepted**	**Sp. Comp**
% Isopoda	Percent Isopoda abundance (m)	S	
# Shredders – Taxa	Number of Shredder taxa (m,l/r)	G/B	
# Scrapers – Taxa	Number of Scraper taxa (m,l/r)	BP	
# Gathers – Taxa	Number of Gatherer taxa (m,l/r)	G/B	
# Filterers – Taxa	Number of Filterer taxa (m,l/r)	R	
# Predators – Taxa	Number of Predator taxa (m,l/r)	G/B	
# Omnivores – Taxa	Number of Omnivore taxa (m,l/r)	R	
# Parasites – Taxa	Number of Parasite taxa (m,l/r)	R	

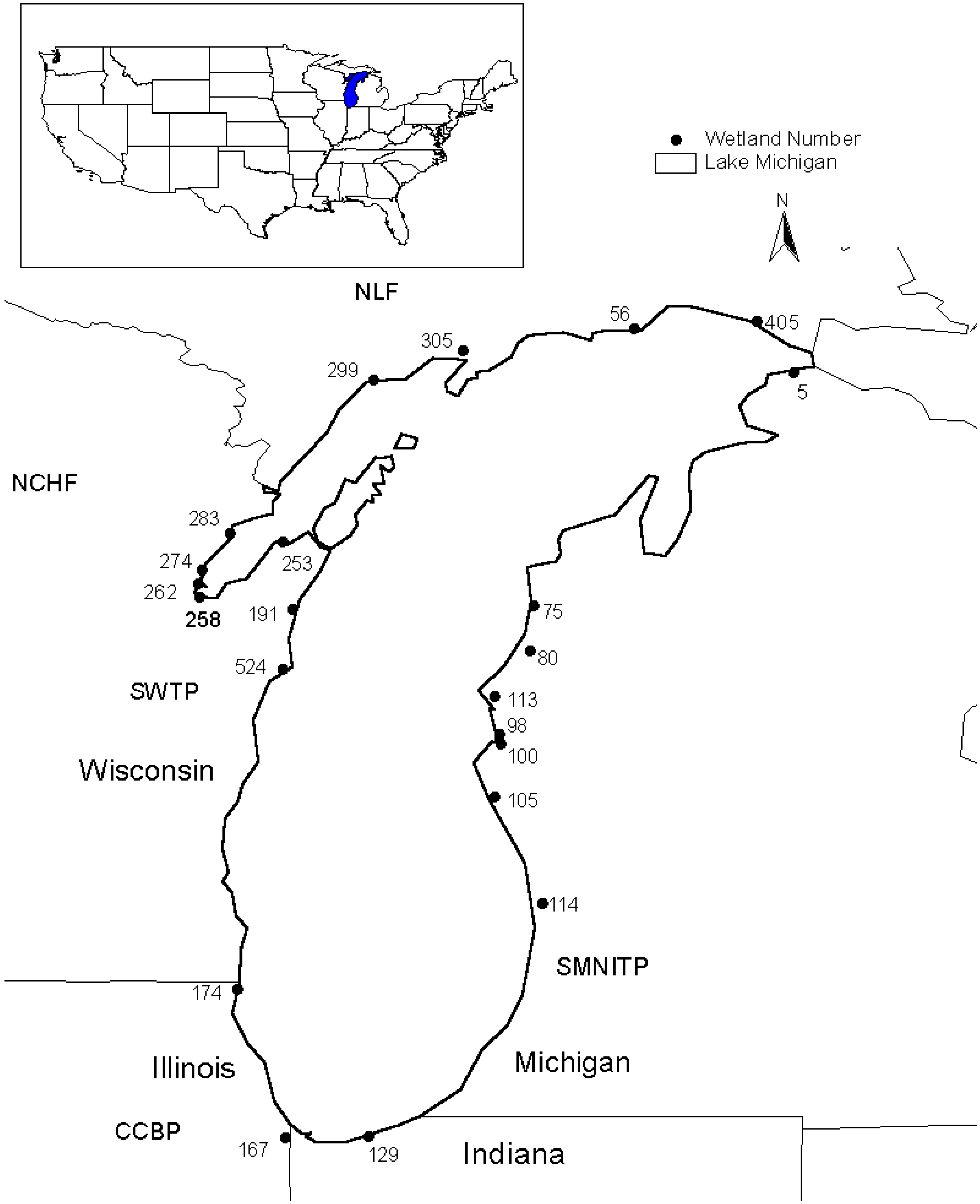
# Piercers – Taxa	Number of Piercer taxa (m,l/r)	G/B	
# Shredders – Individuals	Number of Shredder individuals (m,l/r)	S	
# Scrapers – Individuals	Number of Scraper individuals (m,l/r)	S	
# Gathers – Individuals	Number of Gatherer individuals (m,l/r)	G/B	
# Filterers – Individuals	Number of Filterer individuals (m,l/r)	BP,S	
# Predators – Individuals	Number of Predator individuals (m,l/r)	BP,S	
# Omnivores – Individuals	Number of Omnivore individuals (m,l/r)	Z	
# Parasites – Individuals	Number of Parasite individuals (m,l/r)	Z	
# Piercers – Individuals	Number of Piercer individuals (m,l/r)	S	
% Shredders	Percent Shredder abundance (m)	S	
% Scrapers	Percent Scraper abundance (m)	S	
% Gatherers	Percent Gatherer abundance (m)	G/B	
% Filterers	Percent Filterer abundance (m)	S,Z	
% Predators	**Percent Predator abundance (m)**	**Accepted**	?????????
% Omnivores	Percent Omnivore abundance (m)	Z	
% Parasites	Percent Parasite abundance (m)	Z	
% Piercers	Percent Piercer abundance (m)	S	
# of EPT Taxa (EPT)	**No. Eph.., Trich., and Plec. taxa (m,l/r)**	**Accepted**	**Sp. Comp**
# of Tolerant Taxa	**No. Tolerance value 7-10 taxa (m)**	**Accepted**	**Tol/sens**
# Sensitive Taxa	No. Tolerance value 0-3 taxa (m)	BP	
# Tolerant Individuals 7-10	Number Tolerance value 7-10 ind. (m)	S	
# Sensitive Individuals 0-3	Number Tolerance value 0-3 ind. (m)	S	
% Tolerant	Percent Tolerant individuals 7-10 (m)	S	
% Ind. as Sens. Taxa	**Percent Tolerant individuals 0-3 (m)**	**Accepted**	**Tol/sens**
# Multi – Taxa	Number of Multivoltine taxa (m,l/r)	Z	
# Uni – Taxa	Number of Univoltine taxa (m,l/r)	G/B	
# Bi – Taxa	Number of Bivoltine taxa (m,l/r)	Z	
# Semi – Taxa	Number of Semivoltine taxa (m,l/r)	Z	
# Mero – Taxa	Number of Merovoltine taxa (m,l/r)	Z	
# Bi-Tri – Taxa	Number of Bi-Trivoltine taxa (m,l/r)	Z	
# Uni-Semi – Taxa	Number of Uni-Semivoltine taxa (m,l/r)	Z	
# Uni-Bi/Bi-Uni – Taxa	Number of Uni-Bi/Bi-Uni taxa (m,l/r)	Z	
# Semi-Mero- Taxa	Number of Semi-Mero individuals (m,l/r)	Z	
# Multi – Individuals	Number of Multi individuals (m,l/r)	Z	
# Uni – Individuals	Number of Univoltine individuals (m,l/r)	S	
# Bi – Individuals	Number of Bivoltine individuals (m,l/r)	Z	
# Semi – Individuals	Number of Semi individuals (m,l/r)	Z	
# Metro – Individuals	Number of Mero individuals (m,l/r)	Z	
# Bi-Tri – Individuals	Number of Bi-Tri individuals (m,l/r)	Z	
# Uni-Semi – Individuals	Number of Uni-Semi individuals (m,l/r)	Z	
# Uni-Bi/Bi-Uni - Ind.	Number of Uni-Bi/Bi-Uni ind. (m,l/r)	Z	
# Semi-Mero – Ind.	Number of Semi-Mero individuals (m,l/r)	Z	
%Multi Individuals	Percent Multivoltine abundance (m)	Z	
%Uni Individuals	**Percent Univoltine abundance (m)**	**Accepted**	**Life hist**
%Bi Individuals	Percent Bivoltine abundance (m)	Z	

%Semi Individuals	Percent Semivoltine abundance (m)	Z	
%Metro Individuals	Percent Metrovoltine abundance (m)	Z	
%Bi-Tri Individuals	Percent Bi-Trivoltine abundance (m)	Z	
%Uni-Semi Individuals	Percent Uni-Semivoltine abundance (m)	Z	
%Uni-Bi/Bi-Uni Ind.	Percent Uni-Bi/Bi-Uni abundance (m)	Z	
%Semi-Mero Individuals	Percent Semi-Merovoltine abundance (m)	Z	
# Clingers - Taxa	**Number of Clinger taxa (m,l/r)**	**Altern.**	**Habitat**
# Burrowers – Taxa	Number of Burrower taxa (m,l/r)	S	
# Crawlers – Taxa	Number of Crawler taxa (m,l/r)	BP	
# Swimmers – Taxa	Number of Swimmer taxa (m,l/r)	G/B	
# Sprawlers – Taxa	Number of Sprawler taxa (m,l/r)	B/G	
# Climbers – Taxa	**Number of Climber taxa (m,l/r)**	**Accepted**	**Habitat**
# Skaters – Taxa	Number of Skater taxa (m,l/r)	Z	
# Planktonic – Taxa	Number of Planktonic taxa (m,l/r)	Z	
# Clingers – Individuals	Number of Clinger individuals (m,l/r)	BP,S	
# Burrowers – Individuals	Number of Burrower individuals (m,l/r)	S	
# Crawlers – Individuals	**Number of Crawler individuals (m,l/r)**	**Altern.**	**Habitat**
# Swimmers – Individuals	Number of Swimmer individuals (m,l/r)	S	
# Sprawlers – Individuals	Number of Sprawler individuals (m,l/r)	BP,S	
# Climbers – Individuals	Number of Climber individuals (m,l/r)	S	
# Skaters – Individuals	Number of Skater individuals (m,l/r)	Z	
# Planktonic – Individuals	Number of Planktonic individuals (m,l/r)	Z	
%Clingers	Percent Clinger abundance (m)	S	
%Burrowers	Percent Burrower abundance (m)	S	
%Crawlers	**Percent Crawler abundance (m)**	**Accepted**	**Habitat**
%Swimmers	Percent Swimmer abundance (m)	S	
%Sprawlers	Percent Sprawler abundance (m)	S	
%Climbers	Percent Climber abundance (m)	S	
%Skaters	Percent Skater abundance (m)	Z	
%Planktonic	Percent Planktonic abundance (m)	Z	

Figure Captions

Figure 13.1. Distribution of drowned river mouth coastal wetlands of the Great Lakes for each of the five Lake Michigan ecoregions.

Figure 13.2. Scatterplots of metrics (y-axis) against wetland wetted width; lines differentiate scoring criteria (bolded numbers).


Wetland Number
Lake Michigan
N
NLF
NCHF
SWTP
Wisconsin
Illinois
CCBP
SMNITP
Michigan
Indiana
56
405
305
299
5
283
274
262
258
253
191
524
75
80
113
98
100
105
114
174
167
129

17 Fish Communities as Indicators of Condition in Great Lakes Coastal Wetlands with Emphasis on Development of an Index of Biotic Integrity

Thomas P. Simon, Roger F. Thoma, Paul M. Stewart, Douglas M. Carlson, Frank Stone, Anjanette Bowen, & Michael Hoff

CONTENTS

17.1 INTRODUCTION

Communities have changed drastically due to anthropomorphic modifications (Simon *et al*,. 2001). In most cases, these changes were made prior to our being able to properly access them thus establish a “baseline” condition (Grootjans & van Diggelen, 1995; Vitousek *et al.*, 1997). Those interested in assessing the status of a region are forced to use an alternative approach, which entails establishment of a “least-impacted” condition based on the best estimates of desirable ecological conditions representing the best remaining “natural” conditions of a region (Davis & Simon, 1995).

The greatest challenge facing the development of reference conditions for Great Lakes coastal wetlands is the loss of pristine or natural wetlands. The use of least-impacted or minimally impacted sites will enable the development of a wetland model that will provide subtle attributes of past conditions. Furthermore, this model of wetland condition can be used to restore and maintain the chemical, physical, and biological integrity of coastal wetlands.

The purpose of this paper is to describe the rationale and proposed metrics for development of a fish assessment index for wetland indicators. In addition, we intend to provide an overview of coastal wetland studies as they pertain to fish assemblage research; describe the rationale and implementation of a multimetric index development for drowned river mouth coastal wetlands; and describe some results and approaches using examples from Lake Erie, Lake Ontario, the St. Lawrence and Niagara Rivers. Lastly, we present a case study using biological indicators whose objectives were to determine assemblage attributes, calibrate reference conditions in the coastal wetlands, and propose a method for development and testing of the index.

17.2 METHODS

17.2.1 STUDY AREA AND DESIGN STRATEGY

Using a similar rationale to that developed by Karr *et al.*, (1986), Simon (1998), Simon & Stewart (1998), and Simon (1999a), random probability design sampling in drowned river mouth wetlands was done to assess biological assemblages. Biological indicators evaluated for this study include fish communities along the Great Lakes and connecting channels (Figure 17.1). Additional targeted data was collected from Lake Erie, Lake Michigan, Lake Superior, Lake Huron, Lake Ontario, and the St. Lawrence, Niagara, and St. Clair Rivers.

Site selection was based on a variety of chemical, physical, and other biological indicators (Stewart *et al.*, 1997; Gillespie *et al.*, 1998; Simon, 1998a; Simon & Stewart, 1998; Simon & Stewart, 1999; Stewart *et al.*, 1999a; Stewart *et al.*, 1999b; Stewart *et al.*, 1999c; Thoma 1999; Stewart et al., 2002; Thoma & Simon 2002). Sites selected ranged from very poor to the best sites remaining among extant drowned river mouth wetlands. In addition, a data set of 475 targeted sites from least-impacted protected and open lake embayment sites to some of the most degraded sites in the Great Lakes region were included to validate the study approach.

To achieve spatial balance across this region, a random tessellation stratified design with multidensity categories was used (Stevens and Olsen 1999). One density variable defined ecoregions (Omernik 1987, http://www.epa.gov/wed/pages/ecoregions/level_iii.htm). A second variable defined three size classes that were based on approximate wetland size (Simon et al. in press).

The Great Lakes coastal wetland REMAP project's primary objective included the development of biological indicators and calibration of reference conditions. The project required that all wetlands were represented in the dataset in order for an accurate random draw to be done that would be representative of the resource. Wetland site selection was made using their algorithm for weighting, selection, and oversampling.

The proposed list of sites was subjected to field verification and reconnaissance prior to final selection (Simon et al. in press). Sites were excluded if any of the following criteria were met: (1) streams were dry or not sampleable, (2) sites were located outside the first contour level with the lake or were across road crossings, (3) sites were located near beaver dams or other impassable landforms, or (4) streams had so little water that it was inhabitable for fish. During the two years of field sampling, 145 random wetlands were sampled (Figure 17.1), as well as, an additional 475 targeted wetland sites. In addition, field crews revisited sites and replicate data were collected for quality control estimates.

17.2.2 COLLECTION STRATEGY, APPROACH, AND METHODS

Sampling methods followed Simon (2000) which describes field procedures for collection of fish assemblages. Fish assemblages were sampled using DC-pulsed electrofishing equipment depending on site conditions. Crews used either a Smith-Root back-pack electrofisher, tote barge, or boat-mounted DC shocker with a 3,500 watt Honda generator and pulse-box for varying the amperage. Sampling was done upstream using the appropriate gear. Each zone consisted of sampling 35 times the wetted stream width for a minimum distance of 150 m and a maximum of 500 m. Sampling all available habitats was done at each site so that a minimum of 900 s were fished in the smallest stream and a minimum of 1800 s using boat electrofishing gear. A single netter was on the bow of the boat and netted all fish using a 3-mm standard mesh net. Fish were netted and placed into a live-well until the completion of the sample zone.

At the completion of the zone, fish were sorted by species, measured for minimum and maximum total length, batch weighed, counted, inspected for deformities, eroded fins, lesions, and tumors, and released. Voucher specimens of each species were retained and all small specimens (i.e., darters, madtoms, and sculpins) were preserved and brought back to the laboratory for identification. Fish were identified using standard references including Gerking (1955), Smith (1979), Trautman (1981), and Becker (1983).

17.2.3 Metric assessment and criteria for acceptance

We followed the indicator development rationale used in Ohio for developing coastal wetland indicators (Figure 17.2). This approach uses a development data set and a test data set to validate newly formed indices. Candidate metrics were either accepted or rejected on the basis of several criteria. Initially, draftsman plots (each metric plotted against the remaining metrics in a scatter plot) were examined for linearity, skewness, and kurtosis. Candidate metrics were correlated with each other using a Spearman's correlation and metrics were rejected from further analysis if they had a correlation coefficient (rho) > 0.85. If two metrics correlated with each other, the one that was correlated with additional metrics or had the highest correlation coefficient with additional metrics was removed. After removing correlated candidate metrics, a subset of sites was chosen to represent impaired conditions (i.e., sites that had a high potential for human perturbation). A site was considered an impaired site if: (1) dissolved oxygen was less than 3.5 mg/l, (2) reach-scale landuse at 0-5, 0-30, or 30-100 m from the stream edge was greater than 25% disturbed landuse (sum of pasture, feedlot, cropland, developed), or (3) basin-scale landuse included greater than 40% disturbed landuse (sum of agriculture, industrial, commercial, urban, and residential landuse). Impaired sites were used for comparison of metric scores against references sites (sites not classified as impaired) using a Mann-Whitney *U* test. This rank-based test was used as a distribution-free alternative to the independent-samples *t*-test because it does not require normally distributed data. Candidate metrics were rejected if the reference site values were not significantly different from the impaired site values. Finally, an examination of the boxplots (impaired site values versus reference site values) was used to reject candidate metrics if: (1) the mean taxa richness or mean abundance was too low (i.e., richness <5 taxa or abundance <10%), or (2) the interquartile range of the impaired or reference sites overlapped with the mean of the other (Barbour et al. 1996).

The final suite of metrics selected was normalized to develop a multimetric index. For each metric, data points (sites) were divided into three categories using criteria lines. Based on previous studies of environmental dose-response relationships, most metric values were expected to decrease with increasing perturbation. For these metrics a score was given to each category: a score of one indicated that values were below the 25^{th} percentile of the reference conditions, a score of three indicated that values were within the middle interquartile range of reference conditions, and a score of five indicated that the value was greater than the 75^{th} percentile of the reference conditions (Barbour et al. 1996). Metric values that were expected to increase with perturbation were scored in the opposite direction. Scatter plots of metric scores (y-axis) versus a spatial variable (x-axis; e.g., stream order or basin area) were used to visually represent scoring criteria lines

(Fausch et al. 1984). The division lines for a metric with no linear relationship with the spatial variable (i.e., a slope not significantly different than zero) were drawn parallel to the x-axis of the scatter plot. If the slope was significantly different from zero, the regression line was used to guide division lines. Scoring criteria were delineated by a best-fit regression line with 25% of the reference sites below and above for the lower and upper division lines.

Based on this three-tiered system, metrics were assigned a normalized numeric score that could be summed to derive a total index score. This index score was evaluated in a similar fashion to its individual component metrics. The same set of impaired sites was used for comparison of index scores with the reference sites using a Mann-Whitney *U*-test. The index was rejected if the reference condition values were not significantly different from the impaired site values ($p>0.05$). Boxplots of index scores were also examined for any overlap in the interquartile range or mean of the scores of impaired versus reference sites.

Narrative scores were assigned to the index scores using a four-tiered approach to increase the interpretive power of the index; other studies show that multimetric indices can reliably discern four scoring categories, however, fish assemblage indicators have typically used three categories (Lenat 1988, Barbour et al. 1996). Scoring criteria for the index scores were delineated by percentiles: values less than the 15^{th} percentile were given a narrative score of “poor”, values between the 15^{th} and 50^{th} percentiles were considered “fair”, values between the 50^{th} and 85^{th} percentiles were considered “good”, and values above the 85^{th} percentile were considered “exceptional”.

17.3 RESULTS

For development of the open lake and drowned river mouth coastal wetland index, we reviewed the current literature regarding fish assemblage structure and function, life history, and tolerance literature (Simon, 1999b; Goldstein & Simon 1999; Halliwell et al., 1999; Lyons et al. 2001) for species from the Great Lakes. Generally, 40-62 fish attributes were examined among six main categories in selecting the final metrics that were incorporated into separate multimetric indices for drowned river mouth and open lake wetlands. These metrics were developed into a series of wetland indices of biological integrity (IBI) for each Great Lake based on the hydrogeomorphic model. Simon et al. (chapter 3) found that ecoregion membership was not as important a forcing factor in evaluating species richness in Lake Michigan coastal wetlands as lake membership (Chapters 18-24).

Metrics were classified into one of six categories including species richness and composition, tolerance, trophic ecology, reproductive guild, relative abundance and individual condition. Community structure, key indicator species, and group membership was the focus of the structural metrics such as sucker (Catostomidae) or minnow (Cyprinidae) species; and number of centrarchid species (including black basses). Functional metrics included sensitivity and tolerance measures and trophic ecology; i.e., percent carnivore, insectivore, detritivore. Several guild metrics were designed to evaluate stressors from low water levels, i.e, percent pioneer species (Smith 1972), and obligate wetland species (Simon 1998) as substitute metrics. Ratings in Simon (1998) and Simon et al. (2000) were used to classify species based on a wetland affinity. Abundance was a measure of relative abundance estimated from the number of individuals captured during the 35 times the wetted width sampling. Individual condition incorporating presence of disease measured the lowest extremes of biotic integrity. Site scoring criteria followed that of Karr *et al*., (1986) wherein three levels were based on a trisection of the data and assigned a “5, 3, or 1” depending on the metric and where a site fell on the scatterplot.

Data from coastal wetlands of Lake Ontario open lake embayments, and drowned river mouth wetlands of the Niagara and St. Lawrence Rivers show that limited differences exist between these two wetland hydrogeomorphic types. Different metrics that are being considered as substitutes for coastal wetland indices include the substitution of functional guild metrics. For example, reproductive habitats in coastal wetlands are not typically hard-bottomed surfaces, thus the inclusion of simple lithophilic spawning species may be inappropriate (Simon 1999b). Rather, the expected condition is towards a phytophilic reproductive mode. Fish assemblage information from Lake Ontario compared to the Niagara River and St. Lawrence River show that limited differences between open lake and drowned river mouth assemblages exist for phytophilic species expectations (Figure 17.3A).

Another category that may require adjustment for coastal wetlands is the tolerant species metric. Karr et al. (1986) originally designed the tolerance category for streams to include only the percent occurrence of green sunfish (*Lepomis cyanellus*). This metric has been modified in most IBIs because green sunfish are not usually present in systems other than streams. Adjustments have included the addition of other species to the list of tolerant taxa so that the rationale for the metric has been maintained. One of the metrics we have evaluated is the percent individuals as planktivores. Planktivorous species include a wide range of species that consume a resource that is patchy and include both animal and plant resources. This is similar to fish previously defined as omnivores, so we discontinued using the omnivore definition for this study. The feeding of organisms on 25% animal and 25% plant material is the definition of an *omnivore*, which is considered to increase in degraded habitats.   Data from Lake Ontario suggest that phytoplankton tend to be dominant in the western areas of the Lake in areas that are influenced by urban impacts (D.M. Carlson, unpublished data).   Our results show that the percent individuals as plantkivores does increase in the western basin of Lake Ontario (Figure 17.3B), with the exception of the highest percentages at Lake Mile 270**..**Thus, changes in fish assemblage structure and guild membership can provide indicators of condition in coastal wetlands of the Great Lakes**.**

17.4 DISCUSSION

17.4.1 BENEFITS OF FISH AS ENVIRONMENTAL INDICATORS OF COASTAL WETLANDS

Biological organisms provide an important direct measure of wetland condition. Since many species live their entire lives in a single wetland and other species must spend a portion of their life in wetlands to complete their life history, wetlands serve a necessary role in fish annual recruitment. Biological indicator development benefits from information on species ecology and life history requirements (Simon 1999b). In addition, the public’s perception that living organisms are valuable, particularly fish adds to their importance as indicators. As assemblages change with thermal, habitat, and clinal differences between the Great Lakes, so can the level of anthropogenic disturbance be used to determine a dose-response curve of influence (Karr and Chu 1999).

17.4.2 HISTORICAL PERSPECTIVES OF FISH WETLAND RESEARCH

Three basic research programs have evolved in Great Lakes coastal wetlands. Most previous coastal wetland studies have been a single wetland that has been monitored for multiple years. Other studies have concentrated on assessment procedures and methodological issues (Brazner & Beals 1997; Brazner 1997; Weaver et al. 1993; Thoma 1999). A third category of wetland studies has focused on the function of wetlands including the use of them as reproduction and nursery habitats for Great Lake fish (Goodyear et al. 1982; Chubb & Liston 1986).

17.4.2.1 ADVANTAGES OF LONG-TERM STUDIES AT SINGLE SITES

Ecological studies of a single site include the documentation of species composition, change, and inherent natural variation during diverse hydrologic cycles. Single wetlands have been extensively monitored in the Great Lakes, however, many include State and Federal lands and Great Lakes contaminant “hotspots” (Table 1). These sites enable good representation of site-specific variability as an important consideration for development of indicators. Variability can be evaluated using single sites over many years or can be documented with many sites sampled over a shorter duration of time (Gammon & Simon 2000). Single sites provide representative understanding of area and the types of species using habitats, which enables interpretation of site specific conditions for similar wetland types in that lake. The disadvantage of single site monitoring is that there is little extrapolation potential to entire lakewide conditions since too few sites are available. Also, few or a single site cannot adequately capture the full range of reference condition, so the sites cannot be used to establish reference conditions for larger scale studies. Despite the wetland being data rich for a single site, there is limited potential for establishing a database of site conditions. Likewise, targeted approaches cannot be used generalize lakewide condition of fish assemblages based on single site assessment.

17.4.2.2 COMPARISON OF FISH COLLECTION METHODS

Perhaps, one of the most controversial aspects of coastal wetland management has been the application of methodology. Gear efficiency is perhaps one of the largest stumbling blocks in study design and execution since the comparison of data between studies makes gear compatibility a crucial issue. The development of indicators, can be done and the reference condition calibrated for any gear type and any sample method. As long as the narrative classification of the site assessment provides a similar answer, add comma the methods for collecting the data are not as important. However, this will prohibit the comparison of raw species diversity and abundances. Thus, within our coastal wetland project we chose electrofishing methods for drowned river mouth wetlands, while seining and fyke nets may be the method of choice for protected and open lake embayment wetland types. Night fishing may be required for open lake embayments (Thoma 1999), while preliminary results show that day sampling can be conducted in the drowned river mouths and protected wetlands (Thoma 1999; Simon, unpublished data). The largest differences were observed in the number of species in small drowned river mouth wetland streams. These two results were statistically significant in a comparison of 35 drowned river mouth wetlands in Lake Michigan.

Several studies have compared the efficiency of gear and the sample sampling duration. Thoma (1999) compared gill nets, hoop nets, seining, electrofishing (day vs. night) in targeted lacustuaries of various sizes in Lake Erie. Thoma found that electrofishing was the single best method and did not require multiple gear applications to collect a representative sample form Lake Erie drowned river mouth coastal wetlands. Brazner (1997) compared fyke nets and minnow traps in Green Bay coastal wetlands. Recent studies by Brazner and Tanner (2001) in Lake Superior compared fyke nets and electrofishing in large targeted wetlands. In Lake Michigan, Wilcox et al (1999) compared six large wetlands on the eastern shore, while Simon *et al.* (2001) compared fyke nets and day and night electrofishing in random drowned river mouth wetlands of various sizes.

Interestingly, Brazner and Tanner (2001) found that fyke nets captured a greater number of fish but a similar species list using the two methods. Likewise, Simon *et al*. (2001) also found a similar species list but more similarity in numbers. Simon *et al.* also found that fyke nets were limiting in the very shallow areas along the edge of the lake, often sites of greatest diversity of the small fishes. There may have been a large gear selection bias based on the preferred collection method and investigator experience with that method (P.M. Stewart, pers. observ.).

17.4.2.3 COASTAL WETLANDS AS NURSERY HABITATS

Coastal wetlands of the Great Lakes are important spawning and nursery habitats for fish (Goodyear et al. 1982). Jude and colleagues in numerous publications described the ecology of the southeastern shore of Lake Michigan ichthyoplankton communities in the vicinity of two thermal stations were described in numerous publications. These studies by the University of Michigan for nearly a decade described the seasonality, relative abundance, and habitat use of the nearshore environment and coastal wetlands. Herdendorf and colleagues at the Ohio State University Stone Laboratory on Bass Island documented the ecology of Lake Erie and Lake St. Clair estuaries. Perhaps, one of the most definitive studies was one conducted by Chub and Liston (1986) where they described the ecology of ichthyoplankton assemblage near the Ludington coastal wetlands. All studies have shown that the association of wetland plant assemblages and fish assemblages are necessary for the development and recruitment of healthy and stable populations of Great Lakes fish.

17.4.3 COASTAL WETLAND INDICATOR DEVELOPMENT

Biological patterns validating the biological integrity of wetland communities have generally not been analyzed . Numerous higher level biological indicators have been incorporated into multimetric indices for assessing the condition of wetland resource quality (McKenzie *et al.*, 1992; Davis & Simon, 1995; Simon, 1998; Simon, 1999a; Simon & Stewart, 1998; Simon, 2000a; Simon et al. 2000). Initial efforts to use wetland fish assemblages as biological indicators have been encouraging. Thoma (1999) developed indicators for Lake Erie coastal wetlands, while Simon (1998) and Simon et al.

(2000) developed indicators for protected coastal wetlands of southern Lake Michigan. The use of biological assemblages as direct measures of biological integrity has been well established in the literature, however, coastal wetland studies have not focused on this area of research.

Despite the uniqueness and large differences in resource types between the Great Lakes coastal wetlands and streams and lakes, these resource types share many similarities. Fish assemblage structure and function may not deviate as much as previously considered, thus the large amount of existing research on lakes and streams will greatly advance assessment of Great Lakes coastal wetlands. The stream literature will be directly applicable to the drowned river wetlands, while lake assessments literature is applicable to protected and open lake embayment wetlands.

Our study used hydrogeomorphic wetland classification (Keough et al. 1999) to group wetlands into broad types. Many classification systems exist for wetlands, however, the relevance of these classifications are unknown as to how aquatic organisms view these systems. These were stratified by lake and wetland size to evaluate differences in drowned river mouth and flooded estuary systems (Simon et al. in press). As metric selection was completed, IBIs were calibrated for each wetland class within each lake taking into consideration placement, stream size, and drainage area. The final process for coastal wetland assessment was the use of targeted wetland sites to validate and test the newly created wetland IBIs. These targeted sites represented a variety of impact types and land uses and were collected by our research team. These sites will be used to determine the relevance of the development data set of sites collected during the random probability sampling.

17.5 CONCLUSIONS

The development and assessment of Great Lakes coastal wetlands has benefited greatly by the extensive amounts of prior research conducted in places such as Green Bay, Metzger Marsh, and Pentwater Marsh. As the basic underpinnings of indicator development are explored and evaluated, issues such as methodology, inherent natural variation, and wetland structure and function become foremost in the minds of those implementing these procedures. The approach is based on the experience of stream and lake assessment indicator research, which we have modified these to be consistent with our current understanding of Great Lake coastal wetland assemblage structure.

Development of indices of biotic integrity for Great Lake coastal wetlands is necessary to quantify the extent of degradation. This approach is based on the development of a reference condition based on least-impacted wetlands and evaluating the dose-response to stressors. The current study used hydrogeomorphic wetland classification (Keough et al. 1999) to group wetlands into broad types. These were stratified by lake and wetland size to evaluate differences in drowned river mouth and flooded estuary systems. As metric selection was completed, IBIs were calibrated for each wetland class within each lake taking into consideration placement, stream size, and drainage area. Another aspect of this study was the use of targeted test wetland sites to validate and test the newly created wetland IBIs. These targeted sites represented a variety of impact types and land uses. As further implementation of coastal wetland indicators evolves, modifications will ultimately be necessary. This study is a preliminary attempt to document the steps taken to develop Great Lake coastal wetland indicators.

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TABLE 1.
List of site categories and some examples of single site coastal wetland monitoring studies conducted in the Great Lakes.

Site Category	Coastal Wetland
National &State Parks	Pentwater Marsh, Indiana Dunes National Lakeshore (Lake Michigan) Metzger Marsh, Old Woman Creek, Conneaut Creek (Lake Erie) St. Louis River estuary (Lake Superior)
Areas of Concern	Grand Calumet River (Lake Michigan) St. Louis River estuary (Lake Superior) Hamilton Harbour (Lake Ontario)
Natural Resource Damage Assessments	Grand Calumet River & Green Bay (Lake Michigan) Cuyahoga River, Maumee River, Black River (Lake Erie).
Confined Disposal Facilities	Times Beach, Detroit CDF, & Cleveland CDF (Lake Erie) Milwaukee, Alsip CDF (Lake Michigan) Saginaw CDF (Lake Huron)

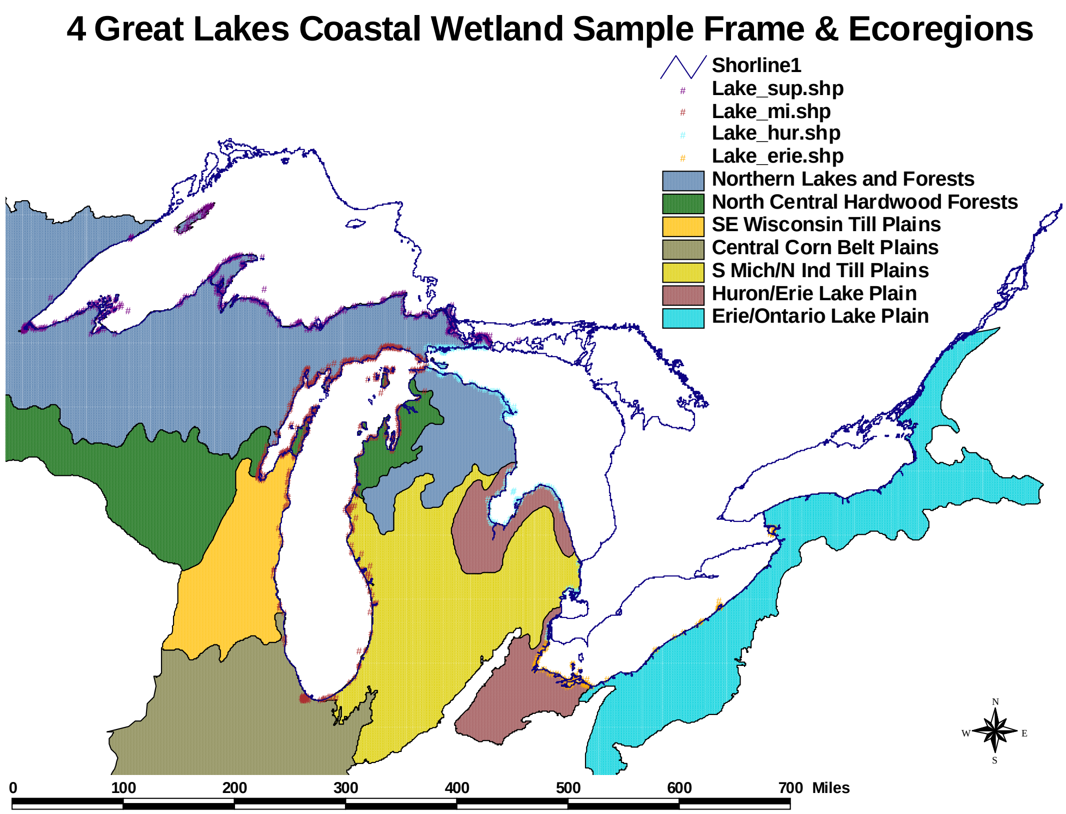
Figure captions

17.1 Sample locations and Ecoregions of Great Lakes coastal wetlands.

17.2 Biological indicator development rationale used for the Great Lakes Coastal wetland project based on Yoder & Rankin (1995).

17.3 Graphs showing the relationship between lake mile and reproductive and tolerance metrics developed for the Lake Ontario open lake embayments and Niagra and St. Lawrence River drowned river mouth wetlands. A. percent phytophilic species, B. percent planktivores.

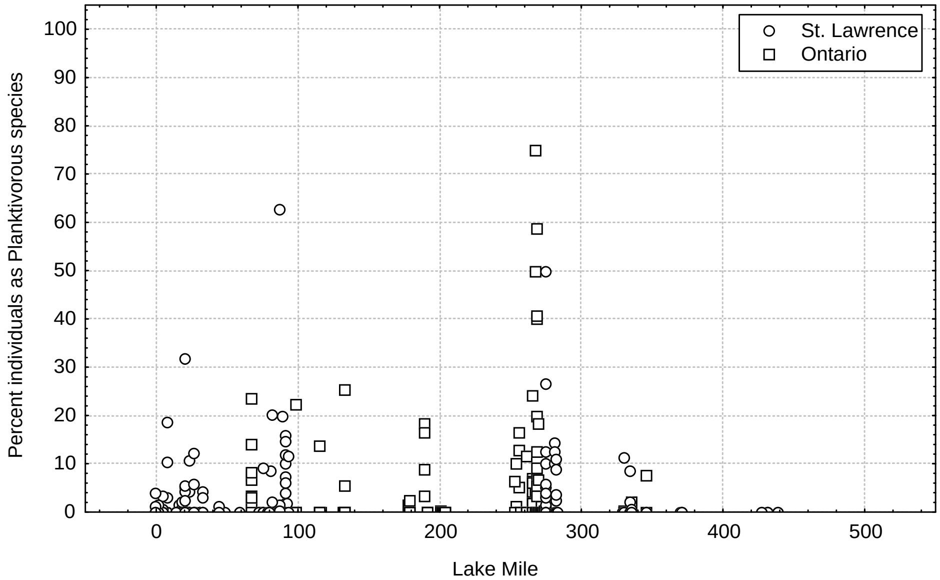
Fig 17.1.



4 Great Lakes Coastal Wetland Sample Frame & Ecoregions
Shorline1
Lake_sup.shp
Lake_mi.shp
Lake_hur.shp
Lake_erie.shp
Northern Lakes and Forests
North Central Hardwood Forests
SE Wisconsin Till Plains
Central Corn Belt Plains
S Mich/N Ind Till Plains
Huron/Erie Lake Plain
Erie/Ontario Lake Plain
0
100
200
300
400
500
600
700 Miles
N
W
E
S

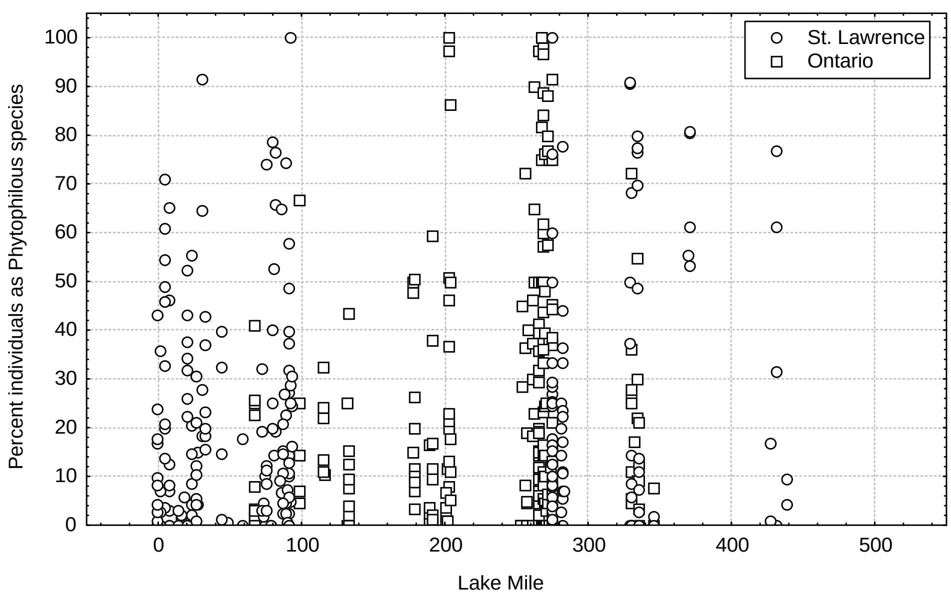
Fig. 17.3

A



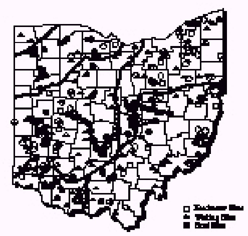
St. Lawrence
Ontario
Percent individuals as Planktivorous species
Lake Mile
0
10
20
30
40
50
60
70
80
90
100
0
100
200
300
400
500

B

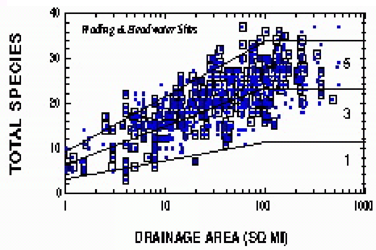


St. Lawrence
Ontario
Percent individuals as Phytophilous species
Lake Mile
0
10
20
30
40
50
60
70
80
90
100
0
100
200
300
400
500

Ohio IBI Calibration & Biocriteria Derivation



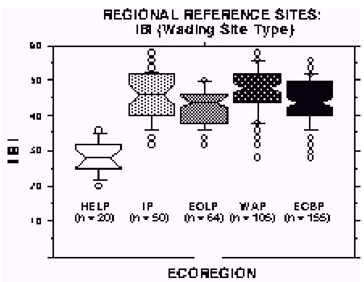
I. Select & sample reference sites



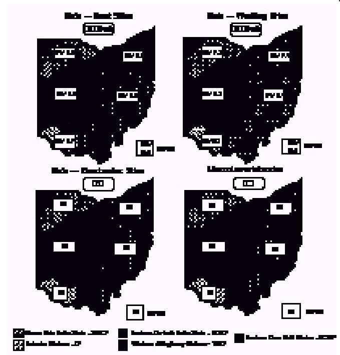
II. Calibration of IBI metrics

Metric	5	3	1
Number of Species	Varies x Drainage Area		
No. of Darter Spp.	Varies x Drainage Area		
No. of Sunfish Spp.	>3	2-3	<2
No. of Sucker Spp.	Varies x Drainage Area		
Intolerant Species			
>100 sq. mi.	>5	3-5	<3
<100 sq. mi.	Varies x Drainage Area		
%Tolerant Species	Varies x Drainage Area		
%Omnivores	<19	19-34	>34
%Insectivores			
<30 sq. mi.	Varies x Drainage Area		
>30 sq. mi.	>55	26-55	<26
%Top Carnivores	>5	1-5	<1
%Simple Lithophils	Varies x Drainage Area		
%DELT Anomalies	>1.3	0.5-1.3	<0.5
Relative Abundance	>750	200-750	<200

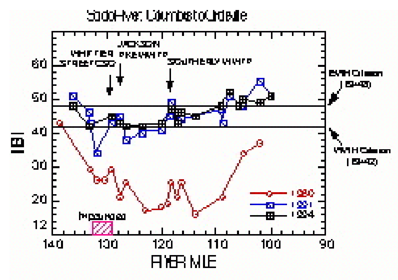
III. Calibrated IBI modified for Ohio waters



IV. Establish patterns/expectations



V. Derive numeric bio-criteria: Codify in WQS



VI. Numeric biocriteria used in coastal wetlands

18 Development of an Index of Biotic Integrity for Drowned River Mouth Coastal Wetlands of Lake Superior

Thomas P. Simon & Ronda L. Dufour

CONTENTS

18.1 INTRODUCTION

A primary goal of environmental assessment is to measure anthropogenic impacts on ecosystem health (Karr et al. 1986; Davis and Simon 1995; Karr and Chu 1999; Simon 1999a). The variability associated with measurement must be capable of distinguishing between natural variability and the amount due to human disturbance. Regionalization and customization of indices are critical components of current methods to assess and regulate the condition of the aquatic ecosystem (Karr 1991). The use of natural, ecologically relevant regions and properly calibrated indices helps managers stratify natural spatial variation among aquatic systems, as well as, determine the variation in stressor response (Simon 2003).

The increasing use of biological indicators has improved assessments of environmental condition relative to assessments based only on physicochemical indicators. The development of biologically based multimetric indices has grown rapidly since Karr (1981) developed the first index for small, Midwestern streams (Simon 1999a). Multimetric indices integrate information over many biological attributes combining them into a numerical index that is scaled to reflect the ecological health of the community. The IBI is a family of multimetric indices that has gained wide acceptance and use (Simon 2000a). Karr et al. (1986) used reach specific information from fish assemblages to assess ecological health. These indices have been developed for other regions (Miller et al. 1988; Simon and Lyons 1995) and continents (Hughes and Oberdorff 1999; Angermeier and Davideanu 2004), other indicator organisms (Kerans and Karr 1994; Barbour et al. 1996; Simon et al. 2001; Fore 2003), and other resource types (Simon 1998; Simon et al. 2001; Whittier et al. 1999). Additional development is necessary for new regions and resource types so that resource managers and biologists can accurately assess the ecological health of water of the United States.

Strengths of the IBI and multimetric indices are the broad ecological basis that information is integrated. For example, multiple levels of ecological organization are included in metrics that summarize the health of individual organisms, populations, and community. Also, metrics are included that integrate taxonomic composition (e.g., number of species), while others describe the function (e.g., percent abundance of detritivores) of communities. This approach integrates information from a broad spectrum of human disturbance that is detectable beyond usual physicochemical measures of water quality, and includes a variety of other disturbances from habitat modification, flow, energy transfer, and biotic interactions (Yoder and Rankin 1999).

The substitution of metrics to represent the most important attributes of a region or resource type is a required step in indicator development (Fausch et al. 1984; Angermeier and Karr 1986). Many metrics have been substituted for the original IBI metrics as investigators have expanded into different regions and resource types (Simon and Lyons 1995). Such flexibility increases the ability of multimetric indices to detect biological responses from stressors. Most region-specific adaptations of IBIs typically have been based on expert knowledge, but recent applications increasingly rely on empirical relations to select metrics (Hughes et al. 1998; Angermeier et al. 2000; McCormick et al. 2001).

The purpose of this study is to assess drowned river mouth coastal wetlands of Lake Superior. No multimetric indicator effort has been developed previously for Lake Superior, nor have any metrics with area-wide utility been identified. Goldstein et al

(1994) and Niemela et al. (1999) developed a fish IBI for the adjacent Red River of the North drainage. Mundahl and Simon (1999) developed a coldwater index for stream tributaries into Lake Superior, as did Lyons et al. (1996). Lyons (1992) also calibrated a warmwater index of biological integrity for northern Wisconsin. Butcher et al. (2003) developed a benthic macroinvertebrate index for the Northern Lakes and Forest Ecoregion. For an index to be useful in this area, appropriate care must be taken to consider historical impacts that resulted from mining and silviculture (Exl and Simon 2001; Simon and Exl 2003). The purpose of this contribution is to 1) define reference conditions, 2) select metrics and analyzing the relationships between these metrics and human impacts on water and substrate quality, and 3) set metric scoring criteria.

23.2 METHODS

23.2.1 STUDY AREA, SITE SELECTION, AND SAMPLE DESIGN

Lake Superior is the largest of the Great Lakes and has the largest surface area of any freshwater lake in the world. It contains almost 12,504.55 cubic km of water, an amount equivalent to all the other Great Lakes plus three times Lake Erie. Lake Superior has an average depth about 149.7 m and is the coldest and deepest (maximum depth 398.8 m) of the Great Lakes. The length of the lake is about 563.3 km from west to east, and 257.5 km north to south, with a shoreline about 7,251.7 km. The drainage basin, totaling 127,700 square km, encompasses parts of Michigan, Minnesota, Wisconsin and Ontario. Most of the Superior basin is sparsely populated, and heavily forested, with little agriculture because of a cool climate and poor soils. The area is included entirely in the Northern Lakes and Forest Ecoregion (Omernik 1987).

Lake Superior is distinguished by its relatively pristine character, heavily forested watershed and high water quality. Lake Superior is the only lake that contains stretches of the arctic-alpine communities, which are found in isolated areas along Lake Superior's northern shore. Lake Superior is sparsely populated with less than 2 percent of the entire Great Lakes population within its basin; the shoreline has been substantially modified in certain areas; particularly in Michigan's Upper Peninsula where urban areas have replaced wetlands, especially at the mouths of rivers. Over 90 percent of the Lake Superior basin is forested; with agricultural (2.3%), urban and public/recreational land uses make up the remainder (USEPA, BASINS version 2). The importance of agricultural lands in the Lake Superior basin is limited due to small cropland acreage and a shorter growing season. The shoreline is relatively undeveloped compared to the other Great Lakes. On the United States shore, much of the eastern shoreline as well as important tracts in the western basin are under federal or state ownership. Over 90 percent of the northern shoreline is owned by the Canadian Crown. These government-owned lands are generally classified as "public" and recreational lands. Urban land use in the basin is concentrated in the two largest urban areas of Duluth-Superior and Thunder Bay. Residential lands are clustered in these urban areas, but shoreline areas are increasingly being subdivided for potential residential development as demands continue for lake-adjacent cottage homes.

18.2.2 COLLECTION

Fish were collected using daytime DC boat electrofishing, tote barge, or backpack units. Electrofishing was conducted on a single shoreline over a linear distance of 500 m or 35 times the wetted width (minimum distance 150 m) using a serpentine travel route within the zone to incorporate all available habitat types. Simon and Sanders (1999) found that 500 m was sufficient distance to capture representative numbers of species to characterize biological integrity but not biological diversity in great waters. This is similar to results obtained from Lake Michigan coastal wetlands where IBI scores did not change with distances increasing beyond 500 m (T.P. Simon, unpublished data). Fish were collected at 37 sites in Lake Superior using a Smith Root (350-V, 8-A) electrofishing unit deployed in either a 4.2 m johnboat (for non-wadeable sites) or using either a 2.4 m tote barge (wadeable wetted widths >4 and < 15 m) or a Smith-Root back-pack electrofishing unit (wadeable sites with wetted widths < 4 m). Amperage was maintained by varying pulse widths according to individual site conditions. We varied the pulse width to obtain 4-6-A output for at least 1800 s. Because boat electrofishing was most effective when deployed within 15 m of shoreline (i.e., at depths less than 2 m), sampling was conducted only under stable, low-flow conditions at a stage level within 1 m of normal water depths. Unlike coastal wetlands in the other Great Lakes, secchi depths were always at least 0.3 m. A single netter was used on the bow and another person maneuvered the boat and was responsible for any fish that surfaced behind the boat. Every attempt was made to capture all fish observed using 4.7 mm mesh dipnets. Captured fish were placed into an onboard, aerated live well for later processing. The capture of any young-of-the-year individuals less than 25 mm TL was not included in the results. At the completion of the reach, fish were identified to species, counted, and inspected for deformities, eroded fins, lesions, and tumor (DELT) anomalies (Sanders et al. 1999). All fish were released except for small species (e.g., minnows, darters, and madtoms), which were retained for laboratory identification using regional fish references (Smith 1979; Trautman 1981; Becker 1983).

18.2.3 Metric development

The Lake Superior coastal wetland system is unique among the remaining Great Lakes since it possesses sites representative of pristine conditions and many remaining coastal wetlands are representative of reference sites (Hughes et al. 1986; Hughes 1995). Some coastal wetlands have sites that have been permanently altered (i.e., hydrologic and channel modifications associated with riparian corridors), such as the Duluth area wetland complex. . Wetland sites were randomly chosen by US Environmental Protection Agency so that equal numbers of small, medium, and large wetlands were sampled (Simon et al., in press). Reconnaissance and sampling of each wetland was based on the following criteria: 1) they had remnants of wetland function, including wetland vegetation; 2) they contained water depths sufficient to provide permanent habitat for fish assemblages, and 3) they had typical habitat conditions representative of the area. Metric scoring was conducted on a dataset of 37 drowned river mouth coastal wetland sites, with the exception of sites from the Duluth Harbor Area of Concern were eliminated from the dataset and used as test sites for validating the index. Sources of disturbance in the electrofishing zone (e.g., shipping activity, docks or mooring sites, navigation traffic wash area, and artificial structures such as piped or other metal debris in the water) were present and used to evaluate the modified IBI for Lake Superior.

All species collected were classified into various taxonomic, tolerance, feeding, and reproductive guilds (Simon et al., chapter 12) using regional references (Eddy and Underhill 1974; Becker 1983; Simon 1999b) and consultation with professional ichthyologists and fisheries biologists. We evaluated indices developed for interior tributaries of Lake Superior (Lyons 1992; Lyons et al. 1996; Mundahl and Simon 1999) and determined that they could not be used to assess coastal wetlands of Lake Superior drowned river mouth coastal wetlands, since chosen metrics would not be reflective of community attributes in coastal wetlands. This would have devalued the quality of the Superior coastal wetland system. We developed a set of 62 candidate metrics incorporating the original metrics described by Karr (1981), modifications suggested by Miller et al. (1988), Simon and Lyons (1995), Goldstein and Simon (1999), Simon (1999b), Thoma (1999), and Hughes and Oberdorff (1999) and new metrics developed specifically for this study (including various combinations of species that were designated in various guilds). Metrics chosen for the Lake Superior IBI focus on six areas of fish assemblage structure and function: species richness, pollution tolerance, breeding habits, feeding habits, fish health, and abundance. The metrics were chosen to reflect biological and habitat integrity, trophic complexity, and future restoration and recovery efforts.

Candidate metrics were evaluated for scoring range, variability, responsiveness, and redundancy following Hughes et al. (1998), McCormick et al. (2001), Emery et al. (2003), and Simon et al. (see chapters 19--24). Metrics were rejected if they failed a range test (i.e., raw values were between 0 and 2 species or were otherwise too small to provide a range of response to disturbance).

18.2.4 STATISTICS

We tested the responsiveness of the remaining candidate metrics using Spearman correlations and scatter plots compared to physical habitat structure and water quality.

Metrics were retained if they reflected the predicted response to physical habitat and water quality variables with significant correlations ($r > 0.15$; $P < 0.001$)(Hughes et al. 1998). Redundancy among metrics was tested using a high Pearson's correlation ($r > 0.75$). One of the redundant pair of metrics was rejected so that the most representative metric for the Lake Superior system fish assemblage was retained (Table 18.2). We tested the response of the Lake Superior coastal wetland IBI using Spearman correlations of metrics and habitat and water quality variables (Table 18.3) and a plot of least-impacted (AI) and test coastal (C) wetlands (Fig. 18.3).

We evaluated patterns in metric performance using linear regressions of the species richness metrics on wetland stream width, which we used as a surrogate for watershed area. If differences were observed, then the slope of the metric line would be adjusted; differences were seen between the number of native species, number of benthic species, and percent individuals as sensitive species and wetted wetland width. These lines were drawn to show differences in expectation. However, to maintain the current biological community standards, we used the maximum value for observed species richness (interpreted as the y-intecept) for the maximum observed line (MOL) for scoring species richness metrics (Emery et al. 2003), instead of the 95th percentile (Fausch et al. 1984). The MOL was drawn through the data and parallel to the regression line. The area below the MOL was evenly trisected into regions providing scores of 1, 3, or 5 (Emery et al. 2003; Simon et al., see chapters 19-24).

Schooling species were excluded that could affect the responsiveness of percent metrics (Thoma 1999). Species such as gizzard shad and emerald shiner, which can occur unpredictably and in large numbers (Simon and Emery 1995; Simon and Sanders

1999) were removed from proportion metric calculations. Both species are included in species richness metrics. Each percent metric was scored following the methods described by Fausch et al. (1984), so that data for each metric was plotted and a line drawn at the 95th percentile; the area beneath the line was then trisected into regions representing scores of 1, 3, and 5. In cases where fewer than 50 individuals were collected (after removing gizzard shad and emerald shiners, tolerant species, nonindigenous species, and hybrids), all proportional metrics were scored as 1 (Yoder and Rankin 1995). In the event that no individuals in a particular metric category were collected, the metric was scored as 0.

18.3 RESULTS

18.3.1 REJECTION RATES OF CANDIDATE METRICS

We selected 12 metrics, each of which was significantly correlated ($P < 0.0001$, $r > 0.2$) with one or more habitat or chemical variables; however, four metrics were unable to differentiate between impaired and the remaining coastal wetlands, and from these we calculated the Lake Superior drowned river mouth coastal wetland IBI (Table 18.2). We rejected 20 metrics because they failed our range test, 20 metrics because they were redundant with other metrics, and 14 metrics because they were not responsive to anthropogenic disturbance (Table 18.1). The number of native species, number of individuals (subtracting nonindigenous, exotic, and hybrid individuals), percent individuals as phytophilic species, and percent individuals as pioneer species were retained as metrics were retained despite the inability to discriminate between impaired and remaining wetlands. These particular metrics showed similar results between both test sites and representative coastal wetlands, attributed to the high quality remaining among Lake Superior drowned river mouth coastal wetlands.

23.3.2 METRIC DESCRIPTIONS OF DROWNED RIVER MOUTH COASTAL WETLANDS IN LAKE SUPERIOR

The number of native species is a modification of the original IBI metric (number of species; Karr 1981). The Lake Superior system has experienced alien species invasion rates similar to the other Great Lakes, however, alien species are currently isolated in specific coastal wetlands (Simon and Vondruska 1991; Pratt et al. 1992). The increase of nonindigenous species in the Great Lakes has made efforts to assess biological integrity problematic. We exclude nonindigenous species and hybrids so that the number of species metric increases with increasing biological integrity. The loss of coastal wetlands and urbanization has resulted in a depauperate fauna as seen in the other Great Lakes compared to historical conditions. The number of native species was dependent on wetted wetland width; thus, calibration showed a gradient response (Fig. 18.2A). The number of native species was correlated with clean sand and submerged aquatic vegetation and with good water clarity, cooler temperatures and more available cover (Table 18.3). Native species declined with degraded water quality (based on turbidity and conductivity), and at wetland sites with excessive fines or clay, highly embedded substrates, and lacking aquatic macrophytes.

The number of benthic species was modified from Karr’s (1981) metric (the number of darter species), since darters are not dominant members of the drowned river mouth coastal wetlands of Lake Superior. The number of benthic species varied with wetted wetland width (Fig. 18.2B) and required calibration modification. The number of benthic species was correlated with submerged vegetation, coarse substrates, and negatively correlated with silt and embedded substrates (Table 18.3). Number of benthic insectivores increases with increasing biological integrity.

Number of centrarchid species was modified from Karr's (1981) metric (the number of sunfish species) to include the black basses (*Micropterus* spp.). The black bass are dominant centrarchids in the Great Lakes, especially in drowned river mouth coastal wetland pool habitat. The number of centrarchid species did not change significantly with wetland width or lake mile. It increased at deeper sites with coarse substrates and habitat complexity (Table 18.3). Centrarchid species richness declined with increased turbidity and water temperature. This metric should decline with the loss of biological integrity of pool habitat.

Percent individuals as lake-habitat species was a replacement richness metric (number of sucker species), focusing on native lake species diversity (Simon and Lyons 1995; Hughes and Oberdorff 1999). This species guild is expected to be present in lentic habitat conditions. Lack of these species indicates a decline in biological integrity. Changes in riparian habitats, which constrain floodplain systems in urban areas, and the loss of Great Lake species result in a depauperate fauna. The percent individuals as lake habitat species was correlated at sites with clean sand, submerged aquatic vegetation, and good water clarity, cooler temperatures, and more available cover (Table 18.3). Lake habitat species declined with degraded water quality (dissolved oxygen), and at wetland sites with excessive fines or clay, highly embedded substrates, and lacking submerged aquatic macrophytes.

Percent sensitive species distinguishes areas of highest quality since species that are especially sensitive to anthropogenic stressors are the first to be eliminated and the last to return to a site. This metric differs from the intolerant species metric by including those species defined as highly- and moderately-intolerant based on taxa classification from Wisconsin (Lyons 1992; Simon et al., chapter 12). The species included in the sensitive list include only species that are highly sensitive to habitat disturbance, toxins, and thermal and nutrient stressors. Species that are sensitive to only one type of stressor are not included (Simon et al., chapter 12). The number of sensitive species showed a response to wetted wetland width (Fig. 18.2F). The number of sensitive species decreased significantly with degraded water quality, and at sites with increased sand, fines, and highly embedded substrates (Table 18.3). This metric reflected the highest levels of biological integrity and is expected to increase with improved water and habitat quality.

Percent individuals as tolerant species represent the worst conditions in the Great Lakes. The percent tolerant species metric should increase with declining biological integrity. Thus, tolerant species represent individuals that increase in abundance with increased anthropogenic disturbance. We used the designation of species from Wisconsin (Lyons 1992) to calibrate this metric. The percent tolerant species increased with degraded water quality (increased turbidity and low dissolved oxygen) (Table 18.3).

Percent detritivores replaces the percent omnivore metric of Karr et al. (1986). The percent detritivore metric will increase with decreasing biological integrity and represent an increase in fine organic particulate matter. Species identified as detritvores possess a long coiled gut that enables them to eat vegetation, and has a black peritoneum, which aids in the digestion of food. The percent omnivores did not discriminate between species that switched between food types or were behaviorally plastic in feeding ecology as a result of disturbance (Goldstein and Simon 1999). The percentage of detritivores increased with increasing percentages of sand and fine substrates and higher water temperature (Table 18.3). The percent individuals as detritivores increased as habitat quality declined and the abundance of ultrafine particulate organic matter increased.

Percent benthic insectivore species was modified from Karr's (1981) metric (percent insectivorous cyprinids). Minnows (family Cyprinidae) are not a dominant component of coastal wetland, drowned river mouth fish assemblages; however, they are important indicators of high quality systems. The minnow metric was replaced with the percent benthic insectivores, a niche equivalent metric, so that the same rationale as Karr's (1981) original metric was retained. This guild of species includes darters (family Percidae), round-bodied suckers (genera *Moxostoma, Minytrema, Erimyzon*), madtoms and bullheads (genera *Noturus* and *Ameiurus*), and several benthic minnow species, such as longnose dace (*Rhinichthys cataractae*) and blacknose dace (*Rhinichthys atratulus*)(Simon et al., chapter 12). Percent benthic insectivore species metric did not change significantly with wetland width (Fig. 18.2H). The metric increased at deeper sites with coarse substrates and habitat complexity (Table 18.3). Benthic insectivore species richness declined with increased turbidity, and water temperature. This metric should decline with the loss of biological integrity.

The percent individuals as carnivores was not retained nor Karr's (1981) percent top carnivore metric. The metric was not sensitive to disturbance or wetted wetland width probably because of the diverse carnivore populations remaining in Lake Superior coastal wetlands. We substituted the percent pioneer species (Simon et al., chapter 12) to reflect problems with water quantity in coastal wetlands. Pioneer species are representative of areas that have experienced water withdrawal or low water conditions. Species that are designated as pioneer species are the first to invade these area when water returns, but are incapable of competing with stable populations as a dominate assemblage within the fish assemblage. Smith (1971) indicated that pioneer species are indicators of temporally unavailable or stressed habitats. Species such as fathead minnow (*Pimephales promelas*), bluntnose minnow (*P. notatus*), lake chubsucker (*Erimyzon sucetta*), and johnny darter (*Etheostoma nigrum*) are classified as pioneers. The percent pioneer species in Great Lake coastal wetlands increased with decreased depth, highly embedded substrates, and with increased water temperature (Table 18.3). We expect the percent pioneer species metric to increase with unstable habitats.

Percent phytophils represents reproductive guilds that are sensitive to substrate disturbance and degradation (Simon 1999b; Thoma 1999; Emery et al. 2003). The abundant submerged and emergent aquatic vegetation in Lake Superior drowned river mouth coastal wetlands is an important spawning habitat. Although this metric was not significantly correlated with our degraded test sites, we consider the loss of submergent vegetation to be an important indicator (Simon et al. 2001) that will affect fish species function. We expect the decrease of phytophils with the loss of biological integrity. Percent phytophils was positively correlated with increased sand, fine substrates, and percent submerged vegetation.

Percent deformities, eroded fins, lesions, and tumor (DELT) anomalies measures the effects of contaminants, diet, and overcrowding (Sanders et al., 1999). Karr (1981) considered a high percentage of disease to be a reflection of the lowest extremes of biological integrity. These anomalies are absent or occur infrequently in areas with high water quality, but their occurrence increases at impacted sites (Baumann et al. 1987; Sanders et al. 1999; MacDonald et al. 2002). This metric will capture future degradation or impacts specifically associated with point- and non-point-source pollution (Karr 1981; Thoma 1999). Despite the rarity of DELT anomalies, we retained this metric since we observed a significant correlation between test sites and other remaining wetlands (Table 18.3). Percent DELT anomalies were correlated with increased turbidity and conductivity, and low dissolved oxygen (Table 18.3).

We used CPUE based on application of a standard sampling technique, which is a modification of Karr's (1981) number of individuals metric. An increase in abundance reflects greater biological integrity, although nutrients can exaggerate the productivity of a reach by causing an increase in abundance (Thoma and Simon 2003). Specific taxa often respond to increased stimulation in a predictable manner. These increases have been accounted for in our CPUE metric by removing species designated as tolerant, nonindigenous, and hybrids (Simon et al., chapter 12).

18.3.3 INDEX SCORING AND RESPONSIVENESS

We generated scoring criteria for each of the 12 metrics (Table 18.2; Fig. 18.2). Metrics were not significantly correlated with stream width, with the exception of three metrics; number native species, number benthic species, and percent sensitive species. Significant differences were observed between test impaired sites (AI) and remaining sites (C) for 10 metrics and IBI score (Table 18.4). Several metrics were not able to show a difference between impaired and remaining coastal wetland conditions including, number of benthic species and CPUE (Table 18.4). These metrics were skewed since these metrics reflect the widespread quality in the system. Lake Superior is an oligotrophic system and will contain fewer numbers of individuals and the number of benthic species showed similar levels between both degraded and nondegraded coastal wetlands. The nonsignificant result in the number of benthic species is due to many sites exhibiting excellent quality for this metric (Fig. 18.2B). The sum of the scores for the 12 metrics resulted in Lake Superior coastal wetland IBI scores ranged from 20 to 48 (mean $\pm$ SD, 36.9 $\pm$ 6.3). The potential range is 0—60. We were able to identify fish assemblage variables that were strongly correlated with degraded substrate quality and water quality variables that reflected anthropogenic disturbance. In our analyses, the strongest correlations between metrics and environmental variables were between those measures that described water clarity, submerged vegetation, and substrate quality.

18.4 DISCUSSION

The reason for defining calibration parameters for newly developed indices of biotic integrity is to minimize geographic variation in natural variability and maximize information content in expected variability resulting from anthropogenic disturbance (Suter et al. 2002; Simon 2003). Properly calibrated indices enhance the precision and accuracy of assessments by minimizing confounding effects of natural variability (Fore et al. 1994; Karr and Chu 1999). The presence of reference sites representing minimally disturbed conditions has affected our choice of metrics and the calibration process for Lake Superior drowned river mouth coastal wetlands. The significant response of 10 metrics to anthropogenic disturbance (measured by habitat and water quality) in Lake

Superior coastal wetlands shows that most metrics in this index will be important indicators of biological integrity (Karr and Chu 1999).

Karr and Chu (1999) introduced the ecological dose response curve concept so that indices could be developed with fewer data points. This concept was a vital component of the study design for the Great Lakes coastal wetlands. By using randomly selected wetlands we did not introduce bias associated with targeted sampling (Wilcox et al. 1999; Burton et al.1999), or diminish sample sizes that previous wetland indices used for calibration (Wilcox et al. 1999; Burton et al. 1999; Kashian and Burton 2000), nor violated basic assumptions and premises of the IBI (Burton et al. 1999; Wilcox et al. 1999) creating nonsensical multimetric indices that do not have value in applied biological monitoring.

Our approach was to include a variety of wetland sizes in our calibration process; thus, enabling index calibration for the full spectrum of drowned river mouth coastal wetlands, which has not been done to date in studies of coastal wetlands. Most previous coastal wetland studies have focused on the largest remaining coastal wetlands and reject coastal wetlands that do not fit a preconceived model (Wilcox et al. 1999; Brazner 1997). We have included a wide variety of coastal wetlands in this study because close to 80% of wetlands have been lost and most remaining wetlands have been disturbed in some manner. Wetlands that retain any remaining wetland function were included in our sample universe since our long-term goal is restoration of Great Lake coastal wetlands. Without a model of coastal wetland quality for preliminary consideration, it will be nearly impossible to restore coastal wetland function.

Each of the coastal wetland calibrations for the Great Lakes and connecting channels (see chapters 19-24) distinguished between high- and low-quality sites; thus illustrating the utility in assessing remaining coastal wetlands. We developed fish assemblage metrics that represent the diversity, structure and function of native fish assemblages, and provide restoration endpoints for fish assemblage conditions. Without preliminary models of biological integrity for coastal wetlands, we will continue to see rampant degradation (Karr et al. 1995).

Our initial goal was to establish a single index of integrity for the Great Lakes: however, the regional differences resulting from differences in climate, zoogeography, watershed and basin characteristics, and land use issues required separate calibrations for each lake and the various connecting channels (Simon et al., chapter 17). The results of this research describes an approach for determining least-impacted conditions and provides a set of fish assemblage metrics that will be useful in establishing reference conditions in the Lake Superior drowned river mouth coastal wetlands. Our efforts should be valuable to state and provincial agencies that currently developing biologically based environmental indicators. The benefit of this project was that the management agencies responsible for the actual implementation participated in the development of the process and research. The spatial scale and magnitude of implementing this study would have been beyond the capability of any single management agency. The cooperation among all of the management agencies was essential in covering areas that included disproportionate spatial scales and greater disturbance gradients. By investing in the development of a biologically based indicator a management agency can enhance communication of environmentally sensitive areas to the public and regulate more efficiently since the lake-wide differences observed in the Great Lakes are put on an even scale for regulation (Yoder and Rankin 1995, 1999).

18.5 CONCLUSIONS

Regionally customized biological indices are increasing in popularity to assess environmental quality. We used multivariate statistical measures to calibrate and select regionally sensitive metrics for drowned river mouth coastal wetlands of Lake Superior; and we used an ecological dose-response relationship to validate the index based on test wetlands. We developed an index to assess the condition of fish assemblages from 37 coastal wetland sites distributed throughout the Lake Superior basin. Representative samples of fish assemblages were sampled during 2001 using standardized daytime electrofishing techniques. We evaluated 62 candidate metrics based on attributes of fish assemblage structure and function so that a multimetric index of health could be regionally calibrated. We examined spatial (by stream width) variability of these metrics and assessed their responsiveness to anthropogenic disturbances, specifically effluents, turbidity, and highly embedded substrates. The resulting Lake Superior IBI is comprised of 12 metrics selected for their predictable response to anthropogenic disturbance or reflection of desirable features of a restored Great Lakes coastal wetland based on reference site quality. We modified all of Karr's original index of biotic integrity metrics. Four metrics (the number of native species; number of centrarchid species; percent sensitive species; percent individuals with deformities, eroded fins, lesions, and tumors) were modified from metrics originally designed by Karr. We designed three metrics to replace original Karr metrics (number of benthic species; percent lake habitat species; percent individuals as tolerant species) so that similar rationale would be retained in the index. We also incorporated two trophic metrics (percent individuals as detritivores and benthic insectivores), one metric based on catch per unit effort, one guild metric (percent pioneer species), and one metric based on reproductive mode (percent individuals as phytophilous spawning fish species). The Lake Superior IBI declined significantly when anthropogenic effects of substrate and water quality were present. The approaches used in this study are an important addition to an arsenal of tools necessary to restore and protect coastal wetlands in the Great Lakes. The application of these tools for other unique resource types would be directly transferable to marine coastal wetlands, bayous and embayments of the Gulf of Mexico, worldwide application to large lake systems such as the East African lakes, other sensitive resources including arctic and desert ecosystems, and Great Rivers worldwide. Additional research on the temporal stability of the index will enhance the reliability of the IBI; however, its use will be a significant improvement over current physiochemical protocols.

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Figure Captions

18.1 Drowned river mouth coastal wetlands associated with Lake Superior.

18.2. Metric expectations and scoring relationships for thirteen metrics used to assess biological integrity of drowned river mouth coastal wetlands of Lake Michigan. A. Number of species minus exotic and non-native species, B. Number of benthic species, C. Number of centrarchid species, D. Percent individuals as lake habitat species, E. Percent individuals as sensitive species, F. Percent individuals as tolerant species (Lake Superior), G. Percent individuals as detritivore species, H. Percent individuals as benthic invertivores, I. Percent individuals as pioneer species, J. Number of individuals minus exotic and non-native species, K. Percent individuals as phytophils, and L. Percent individuals with DELT anomalies.

18.3 Validation of an Index of Biotic integrity for Lake Superior coastal wetlands showing relationships between least impacted (AI) and test impaired condition (C) drowned river mouth coastal wetlands.

Fig. 18.1

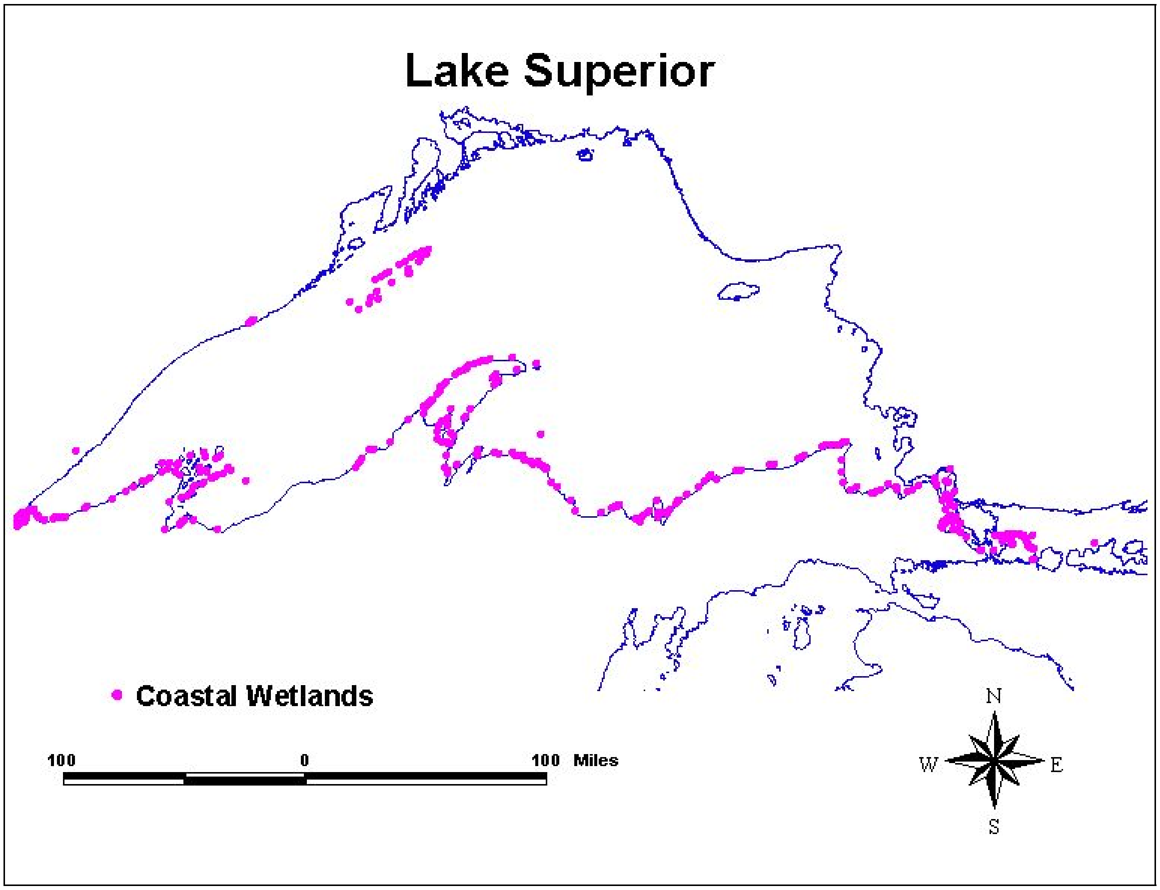

Fig. 18.2

Fig. 18.3

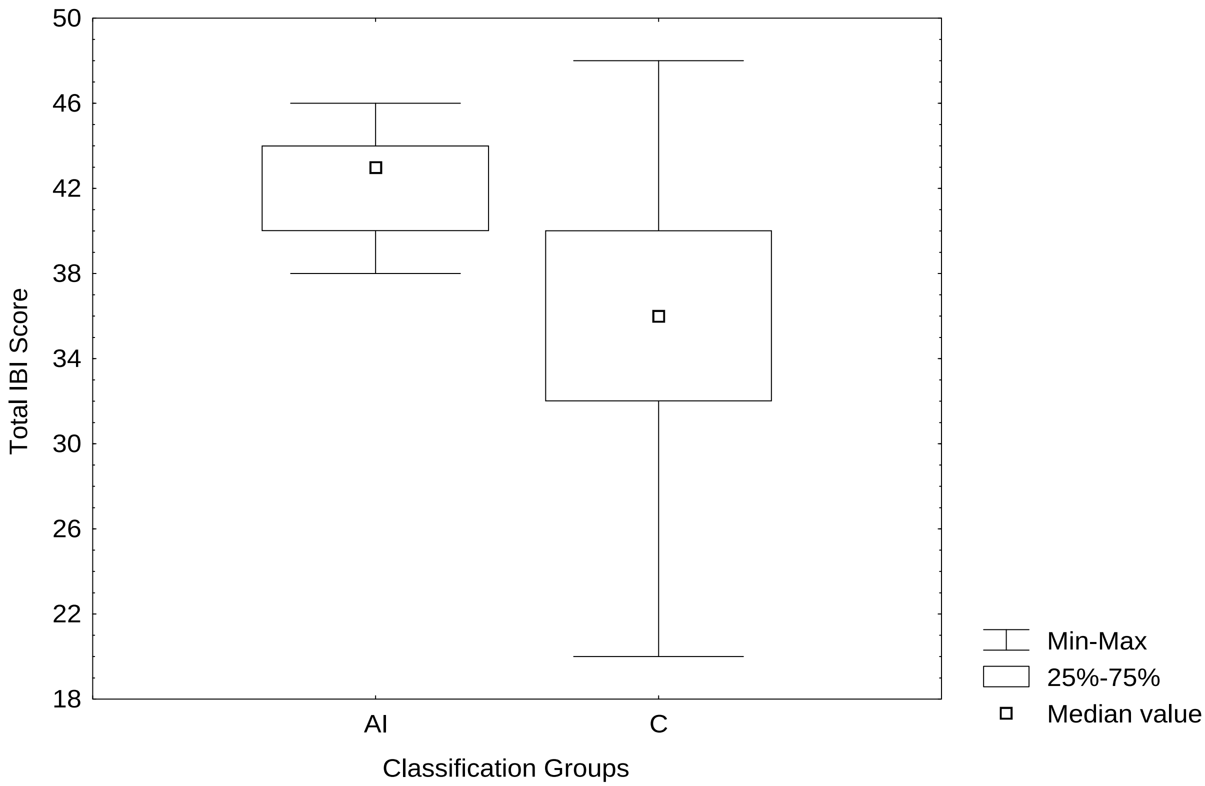
50
46
42
38
34
30
26
22
18
Total IBI Score
AI
C
Classification Groups
Min-Max
25%-75%
Median value

TABLE 18.1

METRICS REJECTED IN THE EVALUATION PROCESS, BY REASON FOR REJECTION. LISTS 1 AND 2 COMPRISE GROUPS OF SPECIES CREATED FOR TEST PURPOSES; SEE TEXT FOR DESCRIPTION OF OTHER SPECIES GROUPS. IND = METRIC BASED ON INDIVIDUALS; BOLD METRICS USED IN FINAL INDEX.

Failed Range Test	Failed redundancy test	Failed responsiveness test
Number darter species	Number of sunfish species	Catch per unit effort (list 1)
Number darters, madtoms, sculpin	Catch per unit effort (list 2)	Percent carnivores
Percent Great Lakes species	Number intolerant species (ind; list 1)	Number tolerant species (list 2)
Number Obligate Great Lakes species	Number of intolerant species (ind.; list 2)	Number lake habitat species
Number sucker species	Number tolerant species (ind.)	Percent tolerant species (list 1; biomass)
Percent great-river species (biomass)	Number lake habitat species	**Number native species**
Number catfish and sucker species	Number benthic invertivores	Percent intolerant species
Percent hybrids (ind.)	Percent top piscivores	Percent insectivores
Number hybrids	Percent deep-bodied suckers (ind.)	**Percent phytophils**
Percent sensitive species (ind.)	CPUE	Number salmonid species
Number of DELT anomalies	Number minnow species	Percent salmonid species (ind.)
Number round-bodied suckers	Percent omnivores (biomass; list O)	Percent simple lithophils
Number deep-bodied suckers species	Percent omnivores (biomass; list O)	**Number individuals minus nonindigenous**
Number catfish and sucker species	Percent nonindigenous	**Percent pioneer species**
Percent green sunfish (ind.)	Percent omnivores (ind.; list 1)	
Percent sucker biomass	Percent omnivores (ind.; list 2)	
Number of planktivores	Number sensitive species	
Percent planktivores (ind.)	Percent tolerant species (list 1)	
Percent round-bodied suckers (ind.)	Number of piscivores (list 1)	
Percent round-bodied suckers (biomass)	Number of piscivores (list 2)	

TABLE 18.2
Metrics and scoring criteria for a Lake Superior index of biotic integrity for drowned river mouth coastal wetlands.

	Expectations		
Species Richness and Composition	1	3	5
Number of native species	Varies with wetted wetland width		(Fig. 18.2A)
Number benthic species	Varies with wetted wetland width		(Fig. 18.2B)
Number of centrarchid species	≤ 1	2-3	≥ 4 (Fig. 18.2C)
Percent individuals as lake habitat species	≤ 33%	>33-66%	>66% (Fig.23.2D)
Tolerance and Sensitivity			
Percent individuals sensitive species (LS)	Varies with wetted wetland width		(Fig. 18.2E)
Percent individuals tolerant species (LS)	>66%	>33-66%	≤33% (Fig. 18.2F)
Trophic guilds			
Percent individuals as detritivores,	>60%	30-60%	<30% (Fig. 18.2G)
Percent individuals benthic insectivore species	<33	33-66%	>66% (Fig. 18.2H)
Percent individuals as pioneer species	>40%	20-40%	<20% (Fig. 18.2I)
Abundance, condition, reproduction, and naturalness			
CPUE	≤275	275-500	>500 (Fig. 18.2J)
Percent individuals as phytophils	≤28%	>28-56%	≥56 (Fig. 18.2K)
Percent individuals with DELT anomalies	> 1.4%	>0.7-1.4%	≤0.7%(Fig. 23.2M)

TABLE 18.3

SPEARMAN CORRELATIONS OF FISH ASSEMBLAGE METRICS AND LAKE SUPERIOR IBI SCORES WITH HABITAT AND WATER QUALITY VARIABLES. HABITAT AND WATER QUALITY DATA WERE AVAILABLE FOR 45 SITES. ALL CORRELATIONS ARE SIGNIFICANT AT THE 0.0001 LEVEL.

	Variable													
Metric	**Mean depth**	**% coarse**	**% sand**	**% fines**	**% wood**	**% sub veg**	**% emg veg**	**% float**	**% highly embed**	**Secchi depth**	**Diss olved oxygen**	**Tem- per- ature**	**pH**	**Cond- uctivity**
Number species	0.45	0.54	0.22	-0.28	0.17	0.32	0.17		-0.44	0.22		-0.26		0.23
Number benthic	0.21	0.32	0.24	-0.47	0.15	0.17			-0.41			-0.24		
Number centrarchid	0.45	0.45	0.22	-0.34	0.36	0.32	0.32	0.19	-0.34	0.21		-0.24		0.23
% lake habitat species	0.32	0.22	0.21	-0.28	0.23	0.32	0.17	0.19	-0.44	0.21		-0.27		0.25
% sensitive species	0.32	0.51	0.23	-0.48		0.22			-0.21	0.28	0.33	0.27		0.20
% tolerant species	0.21	0.29		-0.21	0.21									0.23
% detritivores	-0.36		0.24	0.42						0.24	0.23			
% benthic insectivores	0.35	0.32	-0.23	-0.38	0.19				-0.24			-0.24		0.17
% pioneer species	-0.23			0.21							-0.28			
% phytophils	0.29		0.18	-0.21		0.56	0.34		-0.39	-0.39		-0.22		
% DELT anomalies				-0.29					-0.25	-0.19	-0.24			
CPUE									-0.27					
IBI	0.45	0.54	0.24	-0.34		0.23			-0.44	0.21	0.23	-0.26	0.15	0.23

TABLE 18.4

Descriptive statistics of index of biotic integrity metrics for drowned river mouth coastal wetlands in Lake Superior and significance between impaired test sites and remaining wetlands (p = 0.10).

Attribute	Mean	SD	Range	r(p-value)
Number of species minus exotic and non-native species	10.76	3.44	4-17	.36 (.030)
Number of benthic species	1.73	1.04	0-4	.19 (.269)
Number of centrarchid species	2.08	1.53	0-5	.55 (.000)
Percent individuals as lake habitat species	40.43	28.55	0-97.0	.51 (.001)
Percent individuals as sensitive species	11.04	15.75	0-83.6	.44 (.007)
Percent individuals as tolerant species (Lake Superior)	41.84	27.28	0-99.4	-.35 (.034)
Percent individuals as detritivores	25.37	28.19	0-88.7	-.39 (.017)
Percent individuals as benthic invertivores	55.68	31.70	.65-95.9	.49 (.002)
Percent individuals as pioneer species	6.92	12.69	0-59.6	-.33 (.043)
Number of individuals minus exotic and non-native species	172.86	150.17	15-792	.11 (.503)
Percent individuals as phytophils	14.52	19.47	0-82	.36 (.031)
Percent individuals with DELT anomalies	.45	.98	0-5.5	.48 (.003)
IBI				.47 (.003)

19 Development of an Index of Biotic Integrity for Fish Assemblages in Drowned River Mouth Coastal Wetlands of Lake Michigan

Thomas P. Simon & Ronda L. Dufour

CONTENTS

19.1 INTRODUCTION

Limited information is available on coastal wetland fish assemblages in Lake Michigan, thus the ecology of fish, anthropogenic impacts, and the use of coastal wetlands are poorly known (Jude and Pappas 1992; Krieger et al. 1992, Whillians 1992). Previous biological assessments of Lake Michigan coastal wetlands have focused on Green Bay and Areas of Concern (Brazner and Beals 1997; Brazner 1997; Simon and Moy 2000). Physical changes in the extent of coastal wetland coverage shows that the southern bay and western shore of Green Bay have been reduced by 60-75% (Bosley 1978). In addition, contaminants and pollutants have affected the fish assemblages due in part to the high concentration of pulp and paper mills (Epstein et al. 1974, Bertrand et al. 1976), industrial development (Simon et al. 2003), and urban and residential expansion. Similar trends are apparent for the southern portion of the basin (Simon 1998, Simon and Stewart 1998, 1999; Simon and Moy 2000). Remaining coastal wetlands have been highly impacted and show reduced diversity in fish assemblage structure and function (Simon and Stewart 1999).

Lake Michigan, like most of the Great Lakes, has experienced substantial changes as a result of exotic invasion (Jude et al. 1995, Simon et al. 1998), wetland loss and degradation (Herdendorf et al. 1982), and toxic chemical contamination (Simon et al. 2003, Stewart et al. 2003). These biological and chemical impacts have had a dramatic effect on remaining coastal wetlands in Lake Michigan coastal wetlands and nursery habitats for fishes (Goodyear et al., 1982).

Previously, Lake Michigan had xx indigenous fish species (Bailey and Smith 1981), but this has changed dramatically during the last century (Smith 1968, 1972). The lake trout fishery collapse was evident by the 1960's along with the extirpation of four species of deepwater cisco (Van Oosten et al. 1946). An invasion of exotic species into Lake Michigan has changed the ecology of the system causing widespread changes in trophic dynamics and ecological function (Simon et al. 2000).

The index of biotic integrity (IBI) was developed for assessing fish assemblages of small streams (Karr, 1981; Karr et al., 1986). The IBI is a family of multimetric indices that has been modified and regional calibrated for application in various resource types and among different indicator groups (Simon 2000). No application has been developed for Lake Michigan coastal wetland fish assemblages, with the exception of protected wetlands hydrogeomorphic (HGM) classification completed for southern portions of Lake Michigan (Simon 1998). A regional reference condition includes coastal wetland calibrations for Indiana (Simon 1991) and Wisconsin (Lyons 1992). Limited efforts to date to formulate similar indices have been done for the States of Michigan and Illinois. The purpose of this study was to develop an IBI for drowned river mouth coastal wetlands for Lake Michigan in order to assess the status and condition of remaining coastal wetlands. This project was conducted in drowned river mouth wetlands following wetland definitions by Keough et al. (1999).

19.1.1 OVERVIEW OF IBI APPLICATIONS FOR LAKE MICHIGAN

Several wetland applications of the IBI have been developed for Lake Michigan including indices for fish (Simon 1991, 1998, Lyons 1992, Simon et al. 2001, 2002), aquatic plant (Simon et al. 2001), and macroinvertebrates (Butcher et al. 2000a,b). Simon (1991) and Lyons (1992) developed calibrations for streams in Indiana and Wisconsin. They included among their sites wetland streams that were tributary to Lake

Michigan. These indices equated high quality streams with the presence of emergent and submergent wetland vegetation, natural riparian corridors, and native fish assemblages. These long-term efforts in Wisconsin and Indiana have resulted in an intensive long-term database for drowned river mouth wetlands that were compared to other riverine applications (T.P. Simon, unpublished data). Simon (1998) developed an application for palustrine wetlands, which included protected wetlands, pannes, and small dunal ponds along Illinois and Indiana. This calibration was validated by a study of the Grand Calumet Lagoons, which showed response of the index to anthropogenic stressors in a predictable manner (Simon and Stewart 1998). In addition, Simon et al. (2000) developed a modified index for vernal ponds and small palustrine wetlands using three indicator groups. Reference conditions were developed in the Indiana Dunes National Lakeshore, Miller Woods study unit, and included fish, amphibians, and crayfish as indicators.

Simon et al. (2001) developed an index of biological integrity for wetland plant assemblages for southern Lake Michigan. Two applications were developed for riverine and palustrine wetlands, which included drowned river mouth and protected wetland HGM types. Further modification of this index is described by Rothrock and Simon (this volume, Chapter 8).

Lastly, an index of integrity was developed for macroinvertebrate assemblages for streams of the Northern Lakes and Forest Ecoregion (NLF) (Butcher et al. 2001a, b). This index included wetland streams that were tributary to Lake Michigan. Stewart et al. (this volume, chapter 13) and Stewart and Simon (this volume, chapter 14) developed calibrations for different gear types. Calibrations for activity traps and D-net collection methods showed that different portions of the invertebrate assemblage were sampled depending on the gear type used.

19.2 METHODS

19.2.1 STUDY AREA, SITE SELECTION, AND SAMPLE DESIGN

Lake Michigan is the second largest Great Lake by volume with just under 4,918.5 cubic km of water. It is the only Great Lake entirely within the United States and possesses more than 2,574.5 km of shoreline. The lake is approximately 189.9 km wide and 494.1 km long. The average lake depth is 85.3-m and the maximum depth is 276.9-m. Lake Michigan possesses two climatic and land use divisions, a northern tier in the colder, upper Great Lakes, while its more temperate southern basin contains the Milwaukee and Chicago metropolitan areas. The Lake Michigan drainage basin is about twice as large as the 57,756.7 square km of water surface area and includes portions of Illinois, Indiana, Michigan and Wisconsin. Lake Michigan is hydrologically inseparable from Lake Huron, joined by the wide Straits of Mackinac.

In order to develop an IBI, with similar rationale to that developed for other regions of the world (Karr, 1981; Karr et al. 1986; Hughes and Oberdorf 1999), drowned river mouth wetlands were surveyed along all shoreline of Lake Michigan (Fig. 19.1). Sixty-two drowned river mouth coastal wetlands were selected using a random selection process (Table 19.1). The Northwestern shore of Lake Michigan includes a large series of wetland complexes that extends from Port Washington, Wisconsin, to St. Ignace, Michigan; and southeast from Waukegan, Illinois along the western shore of Michigan to Mackinaw City, Michigan, (Herdendorf et al. 1982). Few pristine wetlands remain in

Lake Michigan, thus our sites are known to represent a wide range of habitat quality types from "good" to "very poor" quality coastal wetlands.

Sample areas were randomly selected and equally weighted so that sufficient numbers of small, medium, and large wetlands were surveyed to provide the most diverse fish collections (Simon et al. in press). Locations within the wetlands were targeted to ensure that sampling occurred in the most diverse and natural remaining habitat within the randomly selected wetland. This rapid assessment approach ensured that a representative sample of the fish species was collected.

19.2.2 COLLECTION

Electrofishing surveys were conducted during day periods in drowned river mouth coastal wetlands using a variety of gear types based on wetland wetted widths at 62 drowned river mouth coastal wetlands (Table 19.1). Daytime fishing provided a similar species catch compared to night electrofishing, but eliminated the influence of lake species that were transients into the coastal wetlands at night (T.P. Simon, unpublished data). We chose to sample exclusively during the day so that safety and logistic issues would be minimized.

In non-wadeable drowned river mouth wetlands, a Smith-Root boat-mounted electrofishing unit was used at depths less than 2 m for maximum distances of 500 m and minimum times of 1800 s (Simon 2000b). Site specific changes in the frequency and pulse width were made at each site, but typically each unit supplied 300-500 volts and 4-6 amps of DC current. For wetland widths >3.4 m – 10 m, a Smith Root tote barge electrofishing unit was used. Electrofishing in small wetlands with wetted widths less than 3.3 m were surveyed using a Smith-Root generator backpack system for about 15-45 minutes. The minimum sampling distance was 150 m, but sites represented 35 times the wetted width.

Fish were netted using dipnets with 4.7 mm mesh and placed into an aerated live well until the completion of the sample distance. The boat operator was responsible for the operation of the boat and also the capture of any individuals that surfaced behind the boat. Every effort was made to capture fish that were observed behind the boat. All fish were identified to species, measured for minimum and maximum length by species, counted, batch weighted, and inspected for deformities, eroded fins, lesions, and tumor (DELT) anomalies. A voucher specimen or photograph was retained of each species collected and small specimens of minnows and other non-game species were preserved for later analysis in the laboratory using Smith (1979), Trautman (1981), and Becker (1983).

19.2.3 Metric development

Metrics among the five main classification categories were based on more than 50 characteristics of fish communities in selecting multimetric indices for coastal wetlands of Lake Michigan (Table 19.2). All metrics are plotted against the wetted width of the wetland so that comparisons between similar sized wetlands could be made (Fig. 19.2). Classification criteria for fish species in Lake Michigan were developed by reviewing fish assemblage structure and function literature, published life history, and tolerance information (Simon et al., Chapter 12).

Structural metrics incorporated community structure, key indicator species, and compositional group membership attributes. Functional metrics included sensitivity and tolerance metrics, percent individuals based on different trophic ecology, macrohabitat specialists, and reproductive guilds. Relative abundance was based on the number of fish collected within a given sampling zone based on the collection protocol (Simon 2000b). Scoring criteria for this calibration follows Karr et al. (1986), which uses three levels based on a trisection of the data (Table 19.3). For a metric to score a "5" the attribute needs to be representative of the reference condition, a score of "3" shows deviation from the reference condition, and a score of "1" suggests the metric is significantly different from the reference condition (Karr 1981).

We evaluated 50 metrics for suitability and eliminated many based on a range test, colinearity, skewness, and statistical correlations to a measure of disturbance (Hughes et al., 19xx). We substituted metrics using the same rationale as Karr et al. (1986), which resulted in 12 metrics chosen for this application. We validated this modification of the IBI by comparing our rating scores to varying measures of environmental perturbation. We calculated the IBI score using data from samples collected between June and September 2000-2001.

Standards of quality for validating the IBI were considered from1) a subset of drowned river mouth wetlands demonstrating minimum and maximum degradation based on water quality monitoring, and 2) a comparison of percentages of wetland cover types and respective basins' percentages with land use/cover types and density of roads using Geographic Information System data layers provided in BASINS.

19.2.4 STATISTICS

Patterns in species composition, group membership, and functional percentages were scaled against wetted wetland width to determine if a statistically significant relationship existed. Scoring lines trisected the data so that the maximum observed line (MOL) included a trisection of the data beneath the highest observed point (Emery et al., 2003). The MOL approach was used rather than the Maximum Species Richness line approach since we believe that few high quality coastal wetlands remain in Lake Michigan. Thus, we wanted to ensure that we did not overestimate the quality of any of the coastal wetlands by rating them too high.   Metric hypotheses were made *a priori* and qualitatively examined to determine if the patterns found fit these expectations based on a range test.

Spearman correlation ($p < 0.05$) were used to examine the relationship between wetland qualities among a "best remaining" group of wetlands and an "impacted" set of wetlands (Conover, 1971).

19.3 RESULTS AND DISCUSSION

19.3.1 SPECIES COMPOSITION

Fish communities in Lake Michigan drowned river mouth coastal wetlands included 85 species represented by 78 native taxa. Nineteen species occurred at fewer than 4.8% of the sites, and eleven of these were rare species including, Northern brook lamprey *Ichthyomyzon fossor,* chestnut lamprey *I. castaneus,* silver lamprey *I. unicispis*, finescale dace *Phoxinus neogeus*, rosyface shiner *Notropis rubellus*, lake chub *Couseius plumbeus*, blacknose shiner *Notropis heterolepis,* lake chubsucker *Erimyzon sucetta*, burbot *Lota lota,* least darter *Etheostoma microperca,* and spoonhead sculpin *Cottus ricei*,.

Several species, which are indicators of high water quality, have increased their range within Lake Michigan including greater redhorse *Moxostoma valenciennesi,* blacknose shiner *Notropis heterodon,* finescale dace *Phoxinus neogeus,* and northern redbelly dace *Phoxinus eos* compared to historical information from 20 years earlier (Becker 1979; Goodyear et al. 1982).

Among the ubiquitous species in these shallow water areas were seven species that occurred at 50% of the sites, pumpkinseed *Lepomis gibbosus,* largemouth bass *Micropterus salmoides,* johnny darter *Etheostoma nigrum,* bluntnose minnow *Pimephales notatus,* white sucker *Catostomus commersoni,* rock bass *Ambloplites rupestris,* and central mudminnow *Umbra limi*. Three of these seven species are considered tolerant to extreme levels of environmental degradation.

Species represented by the greatest number of individuals included bluntnose minnow, johnny darter, banded killifish *Fundulus diaphanus*, and pumpkinseed. Species that dominated by weight included carp *Cyprinus carpio,* white sucker, largemouth bass, and pumpkinseed.

19.3.2 STRUCTURAL ATTRIBUTES OF DROWNED RIVER MOUTH COASTAL WETLANDS IN LAKE MICHIGAN

Drowned river mouth coastal wetlands of Lake Michigan showed a statistically significant relationship between number (ANOVA, t = -2.16207, p = 0.0338) and percent individuals as lake habitat species (ANOVA, t = -2.2268, p = 0.0290) and the percent individuals as exotic and non-indigenous species (ANOVA, t = 2.8667, p = 0.0053) (Table 19.2). The relationship between wetlands and lake habitat species showed a decline in the number and percentage of individuals. Likewise, the relationship between wetlands and the percent individuals as exotic and non-indigenous species showed an increasing relationship. Both of these patterns show that drowned river mouth coastal wetlands are trending towards degraded conditions. Only the number of centrachid species showed a statistically significant relationship with wetland wetted stream width (Table 19.3). This result was similar to other Great Lake coastal wetlands (Simon and Dufour, chapter 21). Small stream coastal wetlands did not have a high number of centrarchid species. Centrarchid species are important components of fish assemblages in drowned river mouth wetlands within Lake Michigan. The number of centrarchid species ranged between zero and 7 centrarchid species at a site (Fig. 19.2B). We expected to find a greater number of centrarchid species with high quality wetlands. The significant relationship we observed may be a result of the low water year during which this study was conducted (Table 19.3). We anticipate that the low water levels will not reduce the power of this metric for smaller wetland streams when water levels reach normal conditions.

The number of lake habitat species was substituted for the number of darter species since we anticipated finding increasing numbers of lake habitat species with recovery of the Great Lakes system (Fig. 19.2D). Lakes habitat species are species expected in lentic habitat types; however, due to the largescale degradation basin-wide this metric is under attaining for most of the wetlands surveyed (Simon et al., chapter 12). Either the percent individuals as lake habitat species or the number of lake habitat species could have been selected. We chose the number of lake habitat species to be consistent with Karr’s (1981) original metric. This metric showed a significant relationship between current conditions and our expected response. The range in lake habitat species was between zero and 11 species.

19.3.3 SPECIES TOLERANCE AND SENSITIVITY

Regional descriptions of sensitivity were completed by Simon (1991) based on Ohio Environmental Protection Agency (1989) and other sources. The classification placed fish into broad categories of sensivity. Simon (1991) classified 17 of 85 (20%) species occurring in Lake Michigan as intolerant, while 13 (15.3%) species were considered tolerant. Karr et al. (1986) warned against classifying too many species as sensitive so that this metric can serve as an early warning to declining conditions. Karr et al. (1986) recommended that less than 10% of the fauna be considered sensitive. Despite the higher percentage of intolerant species in this study, we recognize that the distribution of these species in the Lake Michigan ecosystem will cause the classification for any portion of the lake to be closer to Karr et al.'s recommended number. We hypothesized that sensitive species will increase with biological integrity. Our results showed that drowned river mouth coastal wetlands in Lake Michigan ranged between zero and 93.5 percent sensitive individuals. The West Mile Creek wetland (93.5%) and Stony Point Area (Thompson Creek)(79%) wetlands showed the highest percentage of individuals as sensitive species (Fig. 19.2E).

The increase of tolerant species is an indicator of degraded conditions and is inversely correlated with biological integrity. Simon (1991) classified 13 species as tolerant to environmental disturbance. We did not anticipate that there should be any relationship with lake position or stream size for this metric (Fig. 19.2F), since tolerant species should be no more abundant at any given location within Lake Michigan. Thus, our results show that the range of tolerant species is equally distributed among the entire lake. The percent individuals as tolerant species ranged between 0 and 100%.

19.3.4 FUNCTIONAL ATTRIBUTES OF FISH ASSEMBLAGES

We evaluated several trophic guild categories for their ability to explain the biological integrity of drowned river mouth coastal wetlands. We followed Goldstein and Simon (1999) in the assignment of trophic guilds for Lake Michigan species. In addition, we hypothesized that the percent individuals as insectivores and carnivores would increase with biological integrity, while percent individuals as detritivores would decrease with biological integrity. The percent individuals as detritivores replaced the percent individuals as omnivores metric. Species such as gizzard shad *Dorosoma cepedianum* were combined with carp *Cyprinus carpio* since both met the original definition of an omnivore. This metric is inversely scored so higher percent individuals indicate degradation. Our results showed that ranges were between zero and 86.5 percent (Fig. 19.2G).

Neither the percent individuals as insectivore or carnivore metrics showed a relationship with wetted wetland width (Table 19.3; Fig. 19.2H and 19.2I, respectively). We did not need to modify the scoring criteria for percent individuals as carnivores (Simon and Dufour 1998) since inflated percent individuals as carnivores was not observed. High numbers of carnivores is an unstable situation, which cannot be sustained for long periods of time; however, Lake Michigan carnivores never exceeded 30% of the catch for any reach (Fig. 19.2I). Thus, we recognized that higher percent individuals as carnivores occur above what is seen in streams, but is similar to what is found in Great Rivers (Emery et al., 2003).

19.3.5 INDIVIDUAL HEALTH, CONDITION, AND ABUNDANCE

We did not change Karr et al.'s (1986) application of the relative abundance of individual fish collected at a site (Fig. 19.2J), and retained the percent individuals with deformities, eroded fins, lesions, and tumor (DELT) anomalies (Sanders et al., 1999). These two metrics were calibrated to represent coastal wetland conditions in Lake Michigan. A substitute metric was adopted for the percent individuals as hybrids. This metric was replaced with the percent individuals as phytophilous spawning species since we observed that high quality open lake coastal wetland habitat had a variety of submergent, emergent, and floating vascular plant species. We followed Simon (1999) in the placement of species into this guild assignment (Simon et al., Chapt. 12). Our expectation was that percent individuals as phytophilous spawning species would increase with biological integrity. Our results showed that the percent individuals as phytophilous spawning species ranged from 0 to 77.7 percent (Fig. 19.2K).

19.3.6 INDEX VALIDATION

We used two approaches for validating an index of biotic integrity for drowned river mouth coastal wetlands of Lake Michigan. Wetlands were selected that represented the extremes of conditions and we evaluated these "least impacted" and "impaired" wetland fish assemblages using our newly calibrated index. Patterns in the IBI scores among these two groups of wetlands determined the sensitivity of the modified index to different perturbations (Fig. 19.3). We saw a significant difference between impaired sites and all remaining locations. These remaining sites represented qualities ranging between "poor" and "fair-good".

In addition, we evaluated statistical relationships between each metric and the wetted wetland width. By evaluating patterns between metrics and wetland size we effectively evaluated differences attributed to large scale land use, ecoregions, and tributary influences (Table 19.3). The only metric that showed a statistically significant relationship with wetland wetted stream width was the number of centrarchid species. This result is most likely a result of the low water year that sampling was conducted. Although we think that the loss of centrarchid species in these systems is a negative result and a loss of biological integrity, we suspect that the number of centrarchid species will increase during normal water years.

19.5 CONCLUSIONS

Fish community assessments based on a modified Index of Biotic Integrity (IBI) for drowned river mouth coastal wetlands enabled us to compare the environmental degradation between areas in Lake Michigan. Electrofishing catches at 62 sites were collected between June and September 2000-2001 providing a ranges of values between “good-fair” and “very poor” wetlands. We evaluated over 60 metrics using a range test, skewness, colinearity, and correlation analysis to select 12 metrics. These metrics were validated using sites that were classified as impaired during the collections compared to the remaining sites.

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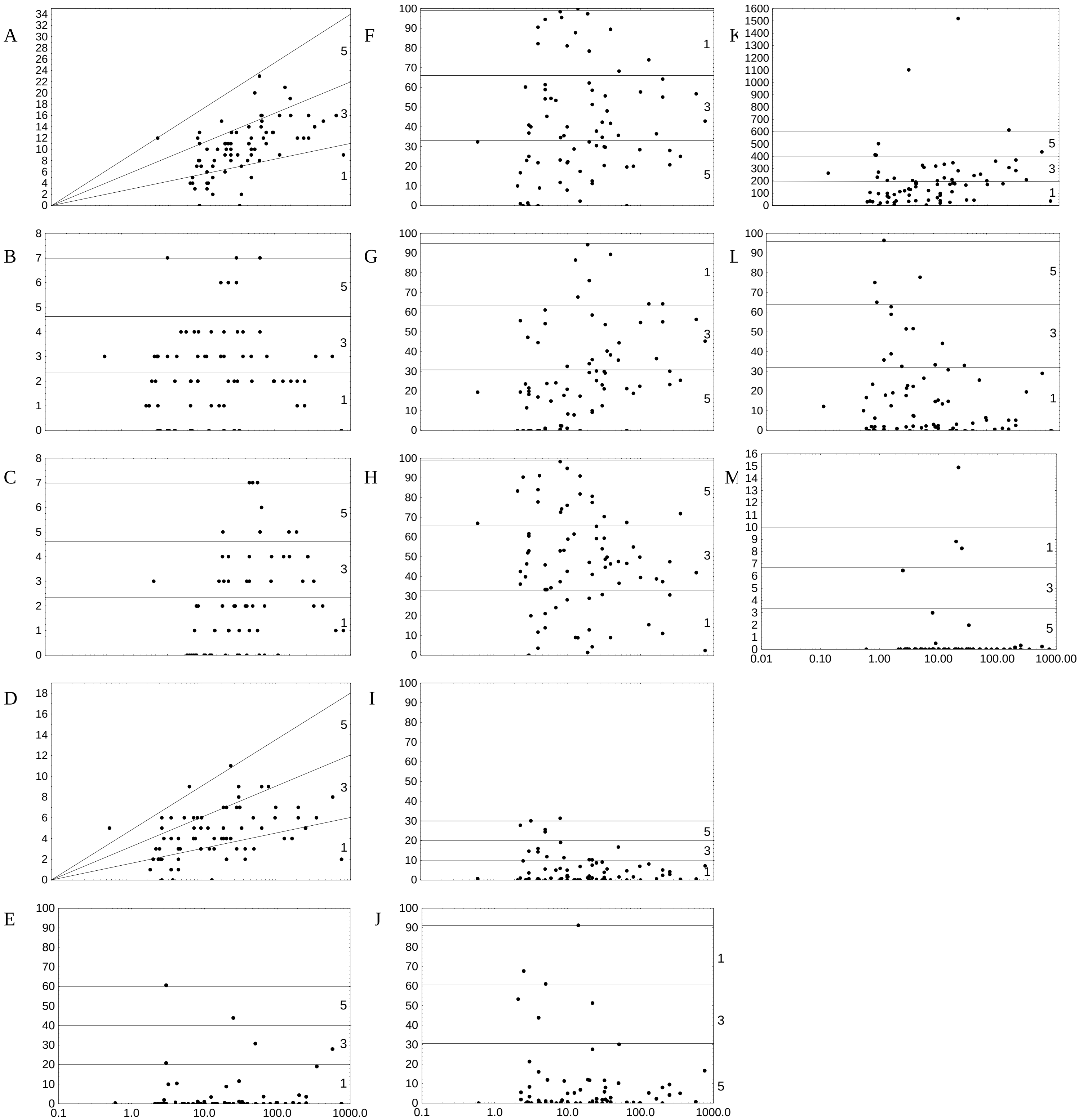
Figure Captions

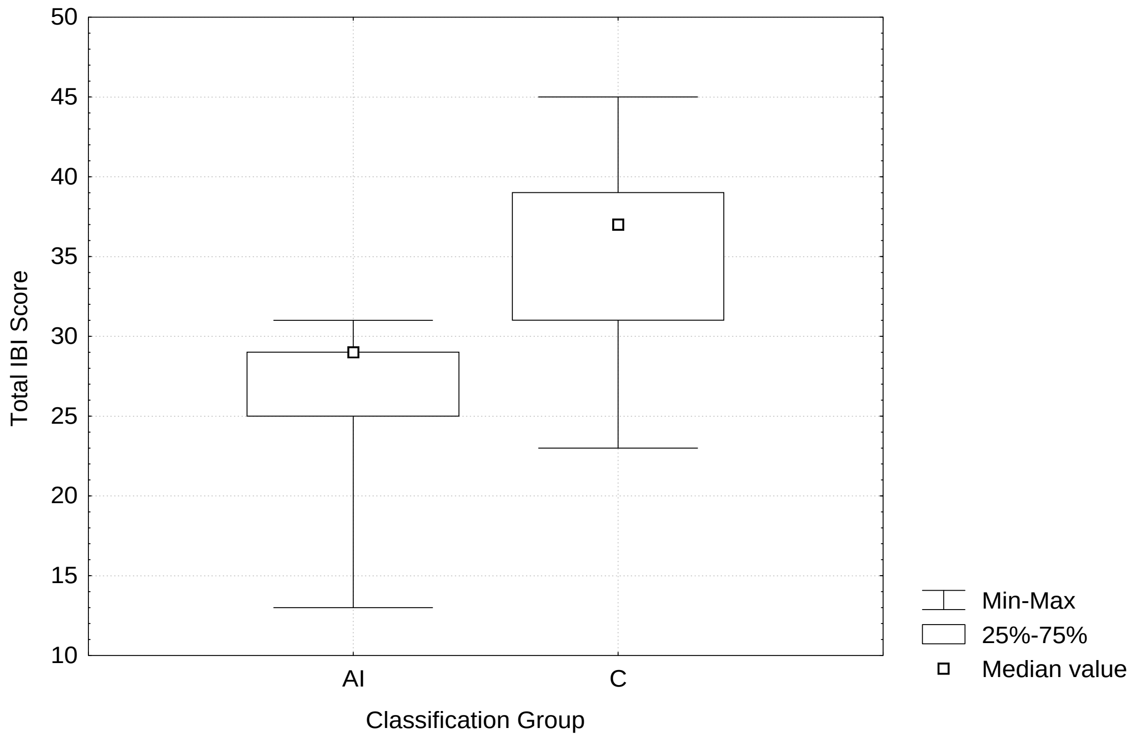
19.1 Distribution of drowned river mouth coastal wetlands in Lake Michigan sampled during this study between June and September 2000-2001.

19.2. Metric expectations and scoring relationships for thirteen metrics used to assess biological integrity of drowned river mouth coastal wetlands of Lake Michigan. A. Number of species, B. Number of centrarchid species, C. Percent individuals as obligate Great Lakes species, D. Number of lake habitat species, E. Percent individuals as intolerant species (MI), F. Percent individuals as tolerant species (MI), G. Percent individuals as detritivores, H. Percent individuals as insectivore species, I. Percent individuals as carnivores, J. Number of individuals, K. Percent individuals as phytophils, L. Percent individuals with DELT anomalies, and M. Percent individuals as exotic and non-native species.

21.3 Validation of a modified Index of Biotic integrity for Lake Michigan showing relationships between “sampled” (AI) and “impaired” (C) drowned river mouth coastal wetlands in Lake Michigan.



WETTED WETLAND WIDTH (m)



50
45
40
35
30
25
20
15
10
Total IBI Score
AI
C
Classification Group
Min-Max
25%-75%
Median value

TABLE 19.1.

List of coastal wetlands found in Lake Michigan including latitude and longitude coordinates, wetland size, and width for 62 Lake Michigan coastal wetland sites. * = identified as a drowned river mouth coastal wetland following HGM classification of Keough et al. (1999).

Number	Name	Latitude	Longitude	Area (ha)	Width (m)
5	*Carp Lake River Area Wetland	45.7417	-84.8333	11.7	9.4
14	*McGeach Creek Wetland	45.2944	-85.3111	216.2	2.7,3
28	*Traverse City Area Wetland #2 (Mitchell Creek)	44.7417	-85.5500	35.2	5.3
46	*Jordan River Wetland	45.6508	-85.5028	27.5	2.8
56	High Island Wetland #6	45.9800	-85.6944	6.1	4.1
75	*Arcadia Lake Wetland (Bowens Creek)	44.4889	-86.2250	145.7	10
80	*Manistee River Wetland	44.2583	-86.2500	3706.9	65
87	*Big Sable River Wetland	44.0861	-86.3625	141.7	98
95	*Pere Marquette River Wetland	43.9167	-86.3333	2532.8	25
96	*Kibby Creek Area Wetland	43.8417	-86.4167	7.7	3.2
98	*Bass Lake Wetland #2 (unnamed stream)	43.8111	-86.4139	55.0	8
100	*Pentwater River Wetland	43.7583	-86.4042	110.1	22
105	*White River Wetland	43.4700	-86.2886	1579.8	32
110	*Muskegon River Wetland	43.2778	-86.1556	2450.2	200;50
113	*Little Pigeon River	44.0100	-86.2156	17.0	4.5
114	*Pigeon River Wetland	42.9031	-86.1817	36.4	35
119	*Black River Wetland (at Southhaven, MI)	42.4097	-86.2700	27.5	10
127	*Galien River Wetland	41.8042	-86.7311	178.5	20
129	Dune Acres Wetland #1	41.6450	-87.1083	0.4	4
165	*O'Brien Lock and Dam Area Wetland #4	41.6506	-87.5622	8.9	130
167	*Grand Calumet River Mouth Wetland #1	41.6472	-87.5583	2.8	22
169	*Grand Calumet River Mouth Wetland #3	41.6383	-87.5450	1.2	14
174	*Illinois Beach State Park Wetland (Dead River)	42.4464	-87.8111	1174.9	5
191	*Kewaunee River Wetland #2	44.4750	-87.5139	145.7	39
192	*Threemile Creek Wetland	44.5750	-87.4556	64.8	3
202	*Baileys Harbor-Ephraim Swamp (Reibolt Creek)	45.1028	-87.1056	2044.5	7
207	*Rowleys Bay Wetland #1 (Mink River)	45.2250	-87.0389	101.2	25
253	*Keyes Creek Wetland	44.8306	-87.5722	28.3	19
258	*Fox River Wetland	44.5361	-87.9800	12.1	759
259	*Atkinson Marsh (Duck Creek)	44.5500	-88.0000	194.3	51
262	Dead Horse Bay Wetland #1	44.6100	-88.0200	8.1	3.9
274	Little Tail Point Wetland #1	44.6750	-87.9970	64.8	3
280	*Pensaukee River Wetland	44.8167	-87.9083	198.4	30
283	*Oconto Marsh (Oconto River)	44.8833	-87.8500	3793.5	8.4
286	*Menominee River Wetland	45.1056	-87.6089	1.6	200

290	*Cedar River Wetland	45.4000	-87.3333	519.4	80
299	*Portage Marsh (Portage Creek)	45.7000	-87.1056	527.1	4.5
305	*Days River Wetland	45.8600	-86.9833	23.5	20.3
308	*Whitefish River Wetland #1	45.9097	-86.9431	7.7	165
323	*Martin Creek Wetland	45.7917	-86.8111	63.2	4.2
328	Ogontz Bay Wetland #2	45.8417	-86.7611	704.5	13
330	*Upper Big Bay De Noc Wetland (Fishdam River)	45.9222	-86.5500	3777.7	10.2
347	*Stony Point Area Wetland (Thompson Creek)	45.9472	-86.2944	1763.2	4
374	*Point Patterson Wetland (Hudson Creek)	45.9889	-85.6583	597.1	2.5
384	*West Mile Creek Wetland	46.1056	-85.4306	13.0	2.9
385	*Mattix Creek Wetland	46.1000	-85.3861	594.7	15
392	*Black River Bay Wetland #3 (upper MI)	46.0944	-85.3389	2.0	15
405	*Brevort Area Wetland (Brevort River)	46.0175	-85.0333	2.4	14.5
409	*Pointe Aux Chenes Marshes (Point Aux Chenes River)	45.9167	-84.8500	1193.9	12.4
500	*Lincoln River	43.9833	-86.4333	12.1	18
505	*Pestigo River Wetland	44.9917	-87.6833	445.3	100
507	*Kalamazoo River	42.7000	-86.2500	172.0	575;350
510	*Inwood Creek Wetland	45.2944	-85.3417	74.9	3
512	*Good Harbor Bay #2 (Shalda Creek)	44.9500	-85.8833	66.8	6
516	*Black Creek (at Mona Lake)	43.1833	-86.2228	4.0	10
517	*Grand River	43.0500	-86.2333	1012.1	250
519	*Macatawa River	42.7972	-86.1167	81.0	8.1
524	*East Twin River	44.1583	-87.5700	81.0	33
526	*Black River (WI)	43.6967	-87.7047	10.1	10
532	*Gierke Creek	45.8500	-86.3417	30.4	2.3
536	*Seiners Creek	45.9833	-85.8333	81.0	2.1
539	*Little Fishdam River	45.9000	-86.5667	1012.1	8

TABLE 19.2

Significance of select metric attributes considered for inclusion in a drowned river mouth coastal wetland IBI for Lake Michigan.

Attribute	t-value	df	p
Number of species minus exotic and non-native species	-1.11467	74	.268598
Number of sunfish species	-0.52501	74	.958271
Number of Centrarchid species	0.325406	74	.745792
Number of minnow species	-0.313011	74	.755153
Percent individuals as Salmonid species	-0.692700	74	.490666
Number or benthic species	-1.50885	74	.135595
Number of sensitive species	-1.46552	74	.147016
Percent individuals as tolerant species	2.708150	74	.008399
Percent individuals as tolerant species (Lake Michigan)	3.172873	74	.002199
Number of lake habitat species	-2.16207	74	.033845
Percent individuals as lake habitat species	-2.22681	74	.029003
Number of obligate Great Lakes species	-0.721434	74	.472916
Percent individuals as obligate Great Lakes species	-1.49590	74	.138931
Percent individuals as insectivore species	-2.56387	74	.012380
Percent individuals as carnivore species	0.187784	74	.851560
Percent individuals as pioneering species	-0.209521	74	.834618
Percent individuals as detritivore species	1.44368	74	.153049
Percent individuals as simple lithophil species	-1.06849	74	.288772
Number of phytophil species	-0.644368	74	.521329
Percent individuals as phytophil species	1.420635	74	.159623
Number of benthic invertivore species	-1.42435	74	.158549
Percent individuals as benthic invertivore species	-1.09569	74	.276769
Percent individuals as exotic and non-native species	2.866743	74	.005398
Number of individuals minus exotic and non-native species	-0.825273	74	.411869
Percent individuals with DELT anomalies	3.628570	74	.000521

TABLE 19.3

Metrics and scoring criteria for an index of biotic integrity for drowned river mouth coastal wetlands of Lake Michigan.

	Expectations		
Species Richness and Composition	1	3	5
Number of species	Varies with wetted width (Fig. 19.2A)		
Number of centrarchid species	<2	3-4	>5 (Fig. 19.2B)
Percent individuals as obligate Great Lakes species	<2	3-4	>5 (Fig. 19.2C
Number of lake habitat species	Varies with wetted width (Fig. 19.2D)		
Tolerance and Sensitivity			
Percent individuals as intolerant species (MI),	<20%	20-40%	>40% (Fig. 19.2E)
Percent individuals as tolerant species (MI),	>67%	34-66%	<33% (Fig. 19.2F)
Trophic guilds			
Percent individuals as detritivores,	>67%	34-66%	<33% (Fig. 19.2G)
Percent individuals as insectivore species	>67%	34-66%	<33% (Fig. 19.2H)
Percent individuals as carnivores	<10%	10-20	>20% (Fig. 19.2I)
Abundance, condition, reproduction, and naturalness			
Percent individuals as exotic and non-native species	>62%	33-61%	<32% (Fig. 19.2J)
Number of individuals	<200	201-400	>401 (Fig. 19.2K)
Percent individuals as phytophils	>67%	34-66%	<33% (Fig. 19.2L)
Percent individuals with DELT anomalies	> 7%	3.3-6.9%	<3.2% (Fig. 19.2M)

TABLE 19.3

Statistical relationships between index of biotic integrity metrics for Drowned River Mouth Coastal Wetlands of Lake Michigan and wetland wetted width (m).

Attribute	Mean	SD	Range	r(p-value)
Number of species minus exotic and non-native	9.75	4.32	1-18	.14 (.510)
Number of benthic species	1.75	1.59	0-5	-.10 (.639)
Number of centrarchid species	1.58	1.32	0-4	.67 (.0001)
Percent individuals as lake habitat species	34.39	29.37	0-91	.04 (.846)
Percent individuals as intolerant species (Lake Erie)	27.6	32.7	0-100	-.12 (.566)
Percent individuals as tolerant species	47.85	29.87	7-100	-.005 (.979)
Percent individuals as detritivores	10.98	14.48	0-55.7	.29 (.177)
Percent individuals as insectivores	63.38	28.02	11.4-100	-.28 (.189)
Percent individuals as pioneer species	17.89	22.35	0-75.5	-.17 (.418)
Number of individuals minus exotic and non-native	2.75	1.59	1-5	-.11 (.609)
Percent individuals as phytophils	41.15	32.56	1.3-100	-.07 (.739)
Percent individuals with DELT anomalies	4.33	1.40	1-5	.03 (.873)

20 Development of an Index of Biotic Integrity for Coastal Wetlands of Great Lake Connecting Channels: with Emphasis on the St. Lawrence and Niagara Rivers

Thomas P. Simon, Douglas M. Carlson, & Ronda L. Dufour

CONTENTS

20.1 INTRODUCTION

Protecting the biological integrity of aquatic ecosystem health is a fundamental goal of the multinational agreement for the Great Lakes (UGLCC 1988; International Joint Commission, 1989). To achieve this goal requires the development of scientifically sound protocols for assessing biotic condition, including study designs, sampling methods, and analytical tools. However, limited information is available on coastal wetland fish assemblages (Krieger et al. 1992), thus the ecology of fish, anthropogenic impacts, and the use of coastal wetlands are poorly known (Jude and Pappas 1992; Whillians 1992). Few federal or state agencies have developed quantitiative criteria for assessing the biotic status of water bodies (Southerland and Stribling 1995). Rather, physicochemical measures of condition focus on the success of pollution abatement programs rather than the biological assemblage endpoints (Sparks 1995). The continued loss of biological integrity in large river ecosystems epitomizes this situation (Karr et al. 1985a).

Large floodplain rivers include the connecting channels of the Great Lakes. They are distinctive in terms of their ecological function and rate at which humans have modified them. Great Lake connecting channels are subject to a variety of stressors including the altering of flow regimes (Ward and Stanford 1989), pollution and land use practices that alter water quality and temperatures, and intensive urbanization and wetland destruction that disrupts the connectivity of the floodplain (Bayley 1995). In Great Lake connecting channels, the disturbance of natural hydrologic and sediment regimes is evident in channelization (Braaten and Guy 1999), inundation and embayment of backwaters and tributaries (Stalnaker et al. 1989), isolation and loss of wetlands, water withdrawals for irrigation and industrial uses, and excessive loading of fine sediment via land use in their catchments (Berkman and Rabeni 1987; Carlson and Muth 1989; Ebel et al. 1989; Poff et al. 1997). Flow regulation has multidimensional effects on the ecological structure and function of rivers including the reduction of native species and the increased transport of nonnative species. As a result, the assessment of biological integrity for Great Lake connecting channels should show substantial impairment from the cumulative stressors from the Great Lakes watershed.

Connecting channels of the Great Lakes have a unique set of difficulties associated with assessing their biotic condition, which is similar to Great River ecosystems. Foremost, size and spatial scale over which remaining coastal wetland habitats and biota are distributed is a primary issue. Scale has important implications for defining reference conditions and sampling biotic assemblages. Unlike smaller water bodies, which are generally replicated over a given region, connecting channels are usually unique, at least within the jurisdiction of a typical (e.g., state or province) management agency. This lack of comparable replication limits the development of region-specific reference conditions, which commonly provide a basis for biotic assessments (Hughes 1995), and forces a disproportionate reliance on historical accounts and expert judgment to define assessment benchmarks. This difficulty is intensified by the virtual absence of only slightly modified reaches from most Great Rivers; thus, even psuedoreplicate reference reaches are largely unavailable for comparison. Consequently, unless historical accounts are very explicit, which is rare, attributing observed patterns of variation (physiochemical or biological) to natural as opposed to anthropogenic sources might be arbitrary. Nevertheless, biological benchmarks can be defined on the basis of a general understanding of the ecology of riverine species, historical faunal conditions, and by comparing the assemblage structure and function at anthropogenically impacted sites with those from relatively unimpacted sites. As a starting point, this can substantially improve the environmental assessment of Great Lake connecting channels.

The biotic assemblages of large waterbodies are difficult to sample thoroughly. Fish sampling protocols for small streams commonly apply uniform sampling effort to the entire volume of multiple habitat units (e.g., riffles and pools), which collectively provides a "sample" (McCormick et al. 2001). In contrast, there are no sampling technologies that can thoroughly sample a single habitat unit of a Great River, let alone be uniformly applicable to multiple unit types. All available sampling gears have strong biases with respect to taxa, habitat morphology, or water conditions (e.g., clarity, temperature, and conductivity). Even if thorough sampling were technologically feasible, the cost (monetary and biotic) of sampling a major portion of the fishes in a Great Lake connecting channel would be generally prohibitive. Thus, biotic assessments of large rivers are necessarily based on relatively small samples with strong, but often predictable, biases.

Over the past two decades, multimetric indices (Karr et al. 1986; Karr and Chu 1999) have been developed in many areas to serve this function. These tools typically integrate information on many attributes of a biotic community (one attribute per metric) into a numerical index scaled to reflect the ecological health of the community. Strength of this approach is its broad ecological foundation, with individual metrics representing a variety of taxonomic and functional composition attributes of the biotic community. This enables detection of a broader array of human impacts than is possible using only physiochemical measurements of water quality (Yoder and Smith 1999). However, the sensitivity and general applicability of multimetric indices are contingent on appropriate customization during their development. In particular the component metrics and their scoring criteria should reflect system-specific attributes of natural biotic communities and the system-specific responses of those communities to human impacts. For example, the IBI is considered a family of multimetric indices that have had numerous substitutions of individual metrics depending on different ecosystems (Simon and Lyons 1995; Simon 2000b). This flexibility strengthens the ability of multimetric indices to accurately measure environmental degradation.

Species that are native to Great Lakes connecting channels have life history traits that enable them to survive and reproduce in a highly fluctuating environment (Dettmers et al. 2001). Sampling considerations (Simon and Sanders 1999), metric development and testing (Simon 1992, Simon and Emery 1995; Simon and Stahl 1998; Emery et al. 1999), and the variability of index of biotic integrity (IBI) metrics (Gammon and Simon 2000) complicates the assessment of Great River fish assemblages. Reash (1999) described the factors and unique features of Great Rivers that complicates metric development for Great River bioassessment. The unique nature of Great Rivers and lack of comparable size hinder development of a reference condition based on a reference site approach (Hughes et al. 1986; Hughes 1995). Recent studies have addressed the development of biological indicators for assessing the condition and ecological health of great rivers (Hickman and McDonough 1996; McDonough and Hickman 1999; Simon and Sanders 1999; Lyons et al. 2001; Emery et al. 2003; LaViolette 2003). The purpose of this contribution is to develop an assessment tool that would detect impairment from known sources of impact and assess the biological condition of the aquatic resources of the coastal wetlands of Great Lake connecting channels. We attempt to include metrics that represent measures of habitat protection, antidegradation, and ecosystem restoration in the Great Lakes. We describe three major steps in the development process: 1) defining reference conditions, 2) selecting metrics and analyzing the relationships between these metrics and human impacts on water and substrate quality, and 3) setting metric scoring criteria.

20.2 METHODS

20.2.1 STUDY AREA, SITE SELECTION, AND SAMPLE DESIGN

The St. Mary's, Straits of Mackinaw, St. Clair, Detroit, Niagara, and St. Lawrence Rivers are a series of connecting channels between the Great Lakes and their outlet to the Atlantic Ocean (Figure 20.1). These connecting channels cross five ecoregions (Northern Lakes and Forest, Southern Michigan Northern Indiana Till Plain, Huron-Erie Lake Plain, and Northeastern Highlands [Omernik 1987]). Nearly 10% of the U.S. population, more than 26 million people, resides in the coastal counties surrounding the Great Lakes. Population centers include Detroit, Chicago, Cleveland, Buffalo, and Montreal. Commercial navigation transported 165.5 million net tons of dry-bulk cargo on the Great Lakes during the 2001 navigation season. The Great Lakes system, is regulated by 19 locks from the St. Lawrence Seaway to the St. Mary's River and including the St. Lawrence River above Iroquois Dam, has a total shore of about 9,559 nm, a total water surface area of about 24,600,000 ha. With the opening of the St. Lawrence River in 1959, the system provides access by ocean-going deep-draft vessels to the industrial and agricultural heartland of North America. From the Strait of Belle Isle at the mouth of the

Gulf of St. Lawrence, the distance via the St. Lawrence River to Duluth, MN, at the head of Lake Superior is about 2,003 nm. The channels are maintained at a depth of 8.2 m.

Trautman (1981) relates accounts from early settlers along the Great Lakes describing abundant aquatic vegetation, principally of wild rice, and bottoms of clean sand, gravel, boulders, and woody debris. The cutting of forests, giving way to farms, villages and cities; creation of canals, railroads, and other thoroughfares; and the establishment of manufacturing areas creating industrial and municipal wastes caused increases in turbidity, siltation, dissolved solids, chlorides, nitrates, and sulfates. These land use changes caused the change of the fish fauna from one requiring clear and vegetated waters to one dominated by species tolerant of turbidity and bottoms composed of clay and silt.

From 1990 to 2002, the New York Department of Environmental Conservation sampled 178 sites along the St Lawrence and Niagara Rivers and U.S. Fish and Wildlife Service sampled an additional 23 coastal wetlands over the entire 9,559 nm length of the connecting channels. Each river reach incorporated the predominant habitat types with a coastal wetland, ranging from small, shallow, sand shorelines with no cover to extensive vegetated cover areas with variable depths. Samples were collected during the summer and fall (from early June until late September) when the rivers are at stable low to moderate flows.

Physical habitat data were collected from each river reach during 2000-2001. Depth and substrate composition and visual estimates of near shore area containing emergent, submergent, and floating vegetation; placement in the wetland (e.g., left or right edge and distance to open water); riparian land use and proximity of riparian human disturbances (e.g., roads, buildings, industry, and agriculture) were recorded. Water quality data (pH, temperature, dissolved oxygen, conductivity, and Secchi depth) were measured at a single point in each of the 23 wetland area sampled.

20.2.2 COLLECTION

Fish were collected using daytime DC boat electrofishing. Electrofishing was conducted on a single shoreline over a linear distance of about 500 m using a serpentine travel route within the zone to incorporate all available habitat types (Gammon 1998; Simon and Sanders 1999). Simon and Sanders found that 500 m was sufficient distance to capture representative numbers of species to characterize biological integrity but not biological diversity. Fish were collected at 178 sites in the St. Lawrence and Niagara Rivers using a Smith Root (350-V, 5-A) electrofishing unit deployed in a 5.5 m boat. Amperage was maintained by varying pulse widths according to individual site conditions. We varied the pulse width to obtain 5-A output for at least 1800 s. Because boat electrofishing was most effective when deployed within 15 m of shoreline (i.e., at depths less than 2 m), sampling was conducted only under stable, low-flow conditions at a stage level within 1 m of normal water depths and when Secchi depths were at least 0.3 m. Every attempt was made to capture all fish observed using 4.7 mm mesh dipnets. A second effort was made with a 15.4 m seine, which was set at depths of 2 m with a boat and pulled toward shore. Captured fish were placed into an onboard, aerated live well for later processing. The capture of any young-of-the-year individuals less than 25 mm TL was not included in the results. At the completion of the reach, fish were identified to species, counted, and inspected for deformities, eroded fins, lesions, and tumor (DELT) anomalies (Sanders et al. 1999). All fish were released except for small species (e.g., minnows, darters, and madtoms), which were retained for laboratory identification using regional fish references (Smith 1985).

20.2.3 METRIC DEVELOPMENT

The Great Lakes connecting channels lack reference sites representative of pristine conditions. In establishing criteria reflective of least-impacted conditions, we recognized that most of the changes in the connecting channels are permanent alterations of the system (i.e., hydrologic and channel modifications associated with dams). Metric scoring was conducted on a dataset of 178 sites. We selected these sites according to the following criteria: 1) they were at least 5 km upstream from the restricted areas in the vicinity of Moses Saunders Dam in the St. Lawrence or near Niagara Falls; 2) they were at least 1 km downstream from any point source discharge; and 3) they had typical habitat conditions representative of the area. We eliminated sites with sources of disturbance in the electrofishing zone (e.g., boating acitivity, docks or mooring sites, navigation traffic wash area, and artificial structures such as derelict warfs or other metal debris in the water).

All species collected were classified into various taxonomic, tolerance, feeding, and reproductive guilds (Simon et al., chapter 12) using regional references (Trautman 1981; Smith 1979; Becker 1983; Simon 1999a; Halliwell et al. 1999) and consultation with professional ichthyologists and fisheries biologists. We developed a set of 59 candidate metrics incorporating the original metrics described by Karr (1981), modifications suggested by Miller et al. (1988), Simon and Lyons (1995), and Hughes and Oberdorff (1999) and new metrics developed specifically for this study (including various combinations of species that were designated in various guilds). The metrics chosen for the connecting channel IBI focus on six areas of fish assemblage structure and function: species richness, pollution tolerance, breeding habits, feeding habits, fish health, and abundance. The metrics were chosen to reflect biological and habitat integrity, trophic complexity, and future restoration and recovery.

The evaluation process followed Hughes et al. (1998), McCormick et al. (2001), and Emery et al. (2003) in that we examined each candidate metric for its scoring range, variability, responsiveness, and redundancy. Metrics were rejected if they failed a range test (i.e., raw values were between 0 and 2 species or were otherwise too small to provide a range of response to disturbance). We used Spearman correlations and scatter plots to test the responsiveness of the remaining candidate metrics compared to physical habitat structure and water quality. We retained metrics with significant correlations ($r > 0.15$; $P < 0.001$) for which scatter plots reflected the predicted responses to physical habitat and water quality variables (Hughes et al. 1998). Redundancy was tested among metrics and rejected one metric of any pair with a high Pearson's correlation ($r > 0.75$). In such cases, we consulted regional fish references, professional ichthyologists and fisheries biologists and retained the metric more representative of connecting channel fish assemblage than of other systems. We retained some metrics, such as the percent individuals as intolerant species (a smaller subset of sensitive species), percent individuals as detritivores and carnivores, the number of DELT anomalies, and percent individuals as nonindigenous species, because we believed that they reflected historical conditions or they constitute important measures of recovery or represent direct measures of individual health of biological pollution. We tested the response of the IBI using a plot of least-impacted and test coastal wetlands (Fig. 20.2).

We performed linear regressions of the species richness metrics on river mile, which we used as a surrogate for watershed area (Fig. 20.4). Historical records and surveys showed that seven species have been extirpated from the Great Lakes and many others have declined due to human impacts (Mills et al., 1993). To account for these historical changes in fish assemblage structure, we used the maximum value for observed species richness (interpreted as the y-intecept) for the maximum observed line (MOL) for scoring species richness metrics instead of the 95th percentile (Fausch et al. 1984). The MOL was drawn through the data and parallel to the regression line. The area below the MOL was evenly trisected into regions providing scores of 1, 3, or 5 (Emery et al. 2003).

Large numbers of individuals of some schooling species can affect the responsiveness of percent metrics (Thoma 1999). Since gizzard shad and emerald shiner can occur unpredictably and in large numbers (Simon and Emery 1995; Simon and Sanders 1999), we excluded them from the calculations of percentile metrics; however, both species are included in species richness metrics. Each percent metric was scored following the methods described by Fausch et al. (1984), so that data for each metric was plotted against river mile and a line drawn at the 95th percentile; the area beneath the line was then trisected into regions representing scores of 1, 3, and 5. In cases where fewer than 50 individuals were collected (after removing gizzard shad and emerald shiners, tolerant species, nonindigenous species, and hybrids), all proportional metrics were scored as 1 (Yoder and Rankin 1995). In the event that no individuals in a particular metric category were collected, the metric was scored as 0.

20.2.4 Statistics

Spearman correlation ($p < 0.05$) were used to examine the relationship between wetland qualities among a “best remaining” group of wetlands and an “impacted” set of wetlands (Conover, 1971).

20.3 RESULTS

20.3.1 Rejection rates of candidate metrics

We rejected 11 metrics because they failed our range test, 19 metrics because they were redundant with other metrics, and 15 metrics because they were not responsive to anthropogenic disturbance (Table 20.1). Three of the final metrics selected for consideration failed the signal-to-noise test. We selected 13 metrics, each of which was significantly correlated ($P < 0.0001$, $r > 0.2$) with one or more of the habitat or chemical variables, and from these we calculated the connecting channels IBI (Table 20.2).

20.3.2 Metric descriptions of drowned river mouth coastal wetlands in Connecting channels

Native-species richness was modified from Karr’s (1981) species richness metric. It focuses on native-species diversity (Simon and Lyons 1995; Hughes and Oberdorff 1999) by excluding nonindigenous species and hybrids that indicate a decline in biological integrity. The number of native species decreases with declining biological integrity. Changes in gradient, constrained floodplain systems in urban areas, and the loss of Great Lake species are accompanied by a more depauperate fauna. The number of native species was greater at sites with clean sand and submerged aquatic vegetation and with good water clarity, cooler temperatures and more available cover. Native species declined with degraded water quality and at wetland sites with excessive fines or clay, highly embedded substrates, and lacking aquatic macrophytes (Table 20.2).

The number of centrarchid species was modified from Karr's (1981) metric (the number of sunfish species) to include the black basses (*Micropterus* spp.), which are the dominant centrarchids in Great Lake connecting channel coastal wetland pool habitats. The number of centrarchid species did not change significantly with wetland width or lake mile. It increased at deeper sites with coarse substrates and habitat complexity. Centrarchid species richness declined with increased turbidity and water temperature. This metric should decline with the loss of biological integrity of pool habitat.

The number of minnow species replaces the number of sucker (Catostomidae) species. Suckers are not a major component of the Great Lakes connecting channel coastal wetlands. This metric was replaced with the number of minnow (Cyprinidae) species. Minnows represent a wide range of sensitivities and species richness. The metric excludes from the count the number of exotic minnows, which can be artificially inflated in degraded habitats. The number of minnow species was significantly correlated with the presence of submerged vegetation, woody cover, and negatively correlated with elevated temperature, an abundance of sand and fines, and degraded abiotic conditions (Table 20.2). We expected the number of minnow species to decline with increased disturbance (Karr 1981).

The number of benthic invertivores represents a guild of fish species that are expected to dominate in benthic habitats of Great Lake coastal wetlands and to decline with the loss of associated sediment quality. These species include a variety of suckers, darters, catfish, and minnows that represent a wide range of environmental quality. The number of benthic insectivores was correlated with submerged vegetation, coarse substrates, and negatively correlated with silt and embedded substrates. We expect the number of benthic insectivores to increase with increasing biological integrity.

The percent individuals as intolerant species distinguishes areas of highest quality. Species that are especially sensitive to anthropogenic stressors are the first to be eliminated and the last to return to a site. Only species that are highly sensitive to habitat disturbance, toxins, and thermal and nutrient stressors are included in this metric. Species that are sensitive to only one type of stressor are not included (Simon et al., chapter 12). Sensitivity metrics are based on criteria established by Halliwell et al. (1999) for the Northeastern United States. Karr et al. (1986) warned that designating too many species as intolerant would prevent this metric from discriminating among the highest-quality areas and recommended that a maximum of 10% of the fauna be included in this classification. Our list contains more species than recommended, although several of these species (e.g., lake sturgeon and cisco) have not been collected in the connecting channels using electrofishing or seining techniques. The total number of intolerant species decreased significantly with degraded water quality (Emery et al. 2002) and at sites with increased sand, fines, and highly embedded substrates (Table 20.2). This metric reflected the highest levels of biological integrity and was expected to increase with improved water and habitat quality, but failed the redundancy test (Table 24.1).

Percent individuals as tolerant species represents the worst conditions in the Great Lakes. Fish assemblage patterns associated with Areas of Concern, degraded harbor and bays, and widespread water quality degradation are still seen in the most impaired areas. Tolerant species represent an increased proportion of coastal wetland fish assemblages at reaches. The percent individuals as tolerant species increased with degraded water quality (increased turbidity and low dissolved oxygen). We expected the percent individuals as tolerant species to increase with increased disturbance.

Percent detritivores replaced the percent omnivores metrics of Karr et al. (1986) because the original metric did not discriminate between species that switched between food types or were behaviorally plastic in feeding ecology as a result of disturbance (Goldstein and Simon 1999). The percentage of detritivores increased with increasing percentages of sand and fine substrates and higher water temperature (Table 20.2). The percent individuals as detritivores increased as habitat quality declined and the abundance of ultrafine particulate organic matter increased.

Percent individuals as insectivores was modified from Karr's (1981) proportion of cyprinid insectivores metric to measure the proportion of specialized sight feeders in the assemblage (Goldstein and Simon 1999; Simon et al., chapter 12). A scarcity of insectivorous fish species may reflect disturbance that has reduced the production of benthic insects. The percent individuals as insectivores ranged from 0 to 100% and decreased with turbidity. It was highest at sites with increased depth and coarse substrates and declined at sites with silt and higher temperatures (Table 20.2). We expected the percent individuals as insectivores to decline with increased disturbance.

Percent individuals as carnivores was modified from Karr's (1981) precent top carnivore metric. Top carnivores represent the top of the aquatic food web and should include only those species that exclusively feed on vertebrates or crayfish as adults (Simon et al., chapter 12). Species that switch among prey items during ontogeny (e.g., smallmouth bass) are included, but adult species that eat both macroinvertebrates and fish (e.g., green sunfish) were excluded. The percent individuals as top carnivores in Great Lake connecting channel coastal wetlands increased with increased depth and woody cover, but declined with increased water temperature (Table 20.2). We expect the percent individuals as top carnivores to decrease with habitat degradation.

Percent simple lithophil and percent phytophils represents reproductive guilds that are sensitive to substrate disturbance and degradation (Simon 1999b; Thoma 1999; Emery et al. 2003). Simple lithophils increased in the Niagara River, but decreased in the St. Lawrence River, presumably as a result of coarse substrates become less common. The percent simple lithophils was negatively correlated with the increase of sand and fine substrates, while percent phytophils was positively correlated with increased sand and fine substrates. Thus, we use the percent simple lithophils calibrated for the Niagara River and the percent phytophils calibrated for the St. Lawrence River. We expect the decrease of both lithophils and phytophils with the loss of biological integrity.

Percent nonindigenous individuals measures the impact that exotic, nonindigenous, and alien species and hybrids have in reducing the biological integrity of Great Lakes connecting channels. Nonindigenous species increase at degraded sites because the behavioral and ecological mechanisms of species segregation are disrupted (Courtney and Stauffer 1984; Simon and Moy 2000). The percent individuals as nonindigenous species was significantly correlated with increased turbidity (Table 20.2). We include this metric in the Great Lakes connecting channel coastal wetland IBI to document the increased impacts of biological pollution in the Great Lakes.

The number of DELT anomalies measures the effects of contaminants, diet, and overcrowding (Sanders et al., 1999). We chose to use the percent DELT anomalies to be consistent with Karr (1981) and Thoma (1999). Karr (1981) considered a high percentage of disease to be a reflection of the lowest extremes of biological integrity. These anomalies are absent or occur infrequently in areas with high water quality, but their occurrence increases at impacted sites (Baumann et al. 1987; Sanders et al. 1999; Simon and Moy 2000). Despite the rarity of DELT anomalies, we retained this metric to capture any future degradation or impacts specifically associated with point- and non-point-source pollution. The percent individuals with DELT anomalies was correlated with increased turbidity and conductivity, and low dissolved oxygen (Table 20.2). The frequency of individuals with DELT anomalies in these samples was very low (< 0.1%).

Our CPUE metric, was modified from Karr's (1981) number of individuals metric. The number of fish is a measure of productivity, however, since it is difficult to obtain a quantitative measure of fish abundance in Great Rivers (Emery et al. 2003), we used CPUE based on application of a standard sampling technique. An increase in abundance reflects greater biological integrity, although nutrients can exaggerate the productivity of a reach by causing an increase in abundance. Specific taxa often respond to increased stimulation in a predictable manner. These increases have been accounted for in our CPUE metric by removing species designated as tolerants, nonindigenous, and hybrids (Simon et al., chapter 12).

20.3.3 INDEX SCORING AND RESPONSIVENESS

We generated scoring criteria for each of the 13 metrics (Table 20.3). Most metrics were not significantly correlated with river mile, with the exception of number of intolerant species (a rejected metric), percent individuals as nonindigenous species, and number of benthic insectivores (a rejected metrics), percent individuals as insectivores and detritivores, so we did not need to adjust the regression equations (Table 20.4). The five metrics that were significantly correlated with river mile were responding to differences in environmental qualities in specific areas. We expect some skewness in these metrics since they should not represent a universal condition. The sum of the scores of the 13 metrics resulted in connecting channel IBI scores that ranged from 20 to 40 (mean $\pm$ SD, 32.4 $\pm$ 4.92) for the Niagara River, and from 16 to 48 (mean $\pm$ SD, 33.0 $\pm$ 5.84) for the St. Lawrence River. The potential range is 0—65. The mean IBI scores showed a pattern of low IBI scores at shallow sites with sand and fine substrates (ANOVA; $p < 0.05$) and highest at sites with submergent vegetation, clear water, cool temperatures, and complex habitat cover (Table 20.2). Application of these or similar metrics to the catch data in these rivers allowed a comparison of river segments (Carlson et al. chapter 25), and to another IBI developed for further downstream segments where point source impacts were identified (LaViolette 2003).

20.4 Discussion

Fish are especially effective indicators of the condition of aquatic systems because of the diverse morphological, ecological, and evolutionary adaptations to their natural habitats (Karr et al. 1986; Fausch et al. 1990; Simon and Lyons 1995). Human disturbance of streams and landscapes alter key attributes of aquatic assemblages, primarily water quality, habitat, energy flow, and biological interactions (Karr and Dudley 1981). We were able to identify fish assemblage variables that were strongly correlated with degraded substrate quality and water quality variables that reflected anthropogenic disturbance. In our analyses, the strongest correlations between metrics and environmental variables were between those measures that described water clarity, submerged vegetation, and substrate quality. Nine metrics that we expected to be sensitive to disturbance decreased with degraded substrate quality. Three metrics that we expected to be relatively insensitive to disturbance increased with increased turbidity.

This approach was adapted from the Ohio River Great River index developed by Emery et al. (2003). Emery et al. (2003) indicated that the identification of least-impacted sites, particularly the criterion for a minimum distance from point source discharges and hydrologic modifications, should be used in any Great River system. We adopted different assemblage classifications to represent local adaptations of fish assemblages. We chose to include metrics that represented a wide range of management objectives including the response of past conditions, metrics that will respond to future water quality improvement or degradation, and metrics that represent ecosystem restoration.

Efforts to test the response of these Great Lake coastal wetland indices include tests of nutrient loading (Thoma 1999), impacts associated with a industrial discharges from steel mill point sources (Stewart et al. 2003, Simon et al. 2003), and impacts of nonindigenous species (Simon et al. 1998, Simon and Moy 2000). Clearly, the lack of reference sites representing minimally disturbed conditions has affected our choice of metrics and the calibration process. The homogenization of habitat and water quality in the Great Lakes coastal wetlands has caused a loss of biological diversity (Schlosser 1991; Karr et al. 1985) that will be difficult to restore. The regulation of these systems by dams , the introduction of alien species (Simon et al. 1998; Mills et al. 1993), loss of sensitive species (Simon and Moy 2000), and habitat fragmentation (Dynesius and Nilsson 1994; Ward and Stanford 1995) have imperiled the aquatic assemblage of the

Great Lakes. However, despite pervasive and persistent disturbance throughout the system, we were able to identify least-impacted sites that had little evidence of poor water quality or degraded habitat. These sites were pooled to provide a model of biological integrity for the connecting channel coastal wetlands based on subtle attributes of a least-impacted assemblage remaining among these sites.

The results of this research describes an approach for determining least-impacted conditions and provides a set of fish assemblage metrics that will be used to establish conditions in Great Lakes connecting channel coastal wetlands. The selection of sites that were not influenced by major point source discharges, reduced the impacts of human disturbance on our sample reaches. We developed fish assemblage metrics that represent the diversity, structure and function of native fish assemblages, restoration endpoints for presettlement fish assemblage conditions, and the impacts associated with the introduction of nonindigenous species.

20.5 CONCLUSIONS

The use of fish communities to assess environmental condition has been developed for application in Great Rivers, but not for Great Lake connecting channel coastal wetlands. We developed an index to assess the condition of fish assemblages from 178 sites based on 475 collections using seining and electrofishing techniques in the St. Lawrence and Niagara Rivers. Representative samples of fish assemblages were sampled from 1990 to 2002 using standardized daytime electrofishing and seining techniques. We evaluated 59 candidate metrics based on attributes of fish assemblage structure and function to develop a multimetric index of health. We examined spatial (by river mile) and temporal variability of these metrics and assessed their responsiveness to anthropogenic disturbances, specifically turbidity, and highly embedded substrates. The resulting connecting channel IBI is comprised of 13 metrics selected for their predictable response to anthropogenic disturbance or reflection of desireable features of a restored Great Lakes coastal wetland. We retained one metric (the number of intolerant species) from Karr's original index of biotic integrity. Three metrics (the number of native species; number of centrarchid species; percent individuals with deformities, eroded fins, lesions, and tumors) were modified from metrics originally designed by Karr. Three metrics were designed for this study (number of benthic species, number of minnow species, and percent individuals as tolerant species). We also incorporated three trophic metrics (percent individuals as detritivores, insectivores, and carnivores), one metric based on catch per unit effort, one reproductive metric (percent phytophils for St. Lawrence River and percent simple lithophils is substituted for the Niagara River), and one metric based on the percent individuals as nonindigenous fish species. The connecting channel IBI declined significantly where anthropogenic effects on substrate and water quality were prevalent. Additional research on the temporal stability of the index will enhance the reliability of the IBI, its use will be a significant improvement over current physiochemical protocols.

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Figure Captions

20.1 Connecting channels and coastal wetlands associated with remaining drowned river mouth wetlands.

20.2. Metric expectations and scoring relationships for thirteen metrics used to assess biological integrity of drowned river mouth coastal wetlands of the St. Lawrence and Niagara Rivers. NE = Northeast tolerance fish classification (Halliwell et al. 1999). A. Number of species, B. Number of centrarchid species, C. Number of minnow species, D. Number of benthic invertivore species, E. Percent individuals as intolerant species (NE), F. Percent individuals as tolerant species (NE), G. Percent individuals as detritivores, H. Percent individuals as insectivore species, I. Percent individuals as carnivores, J. Number of individuals (CPUE), K. Percent individuals as phytophils, L. Percent individuals as simple lithophils, and M. Percent individuals as exotic and non-native species.

Fig. 20.1

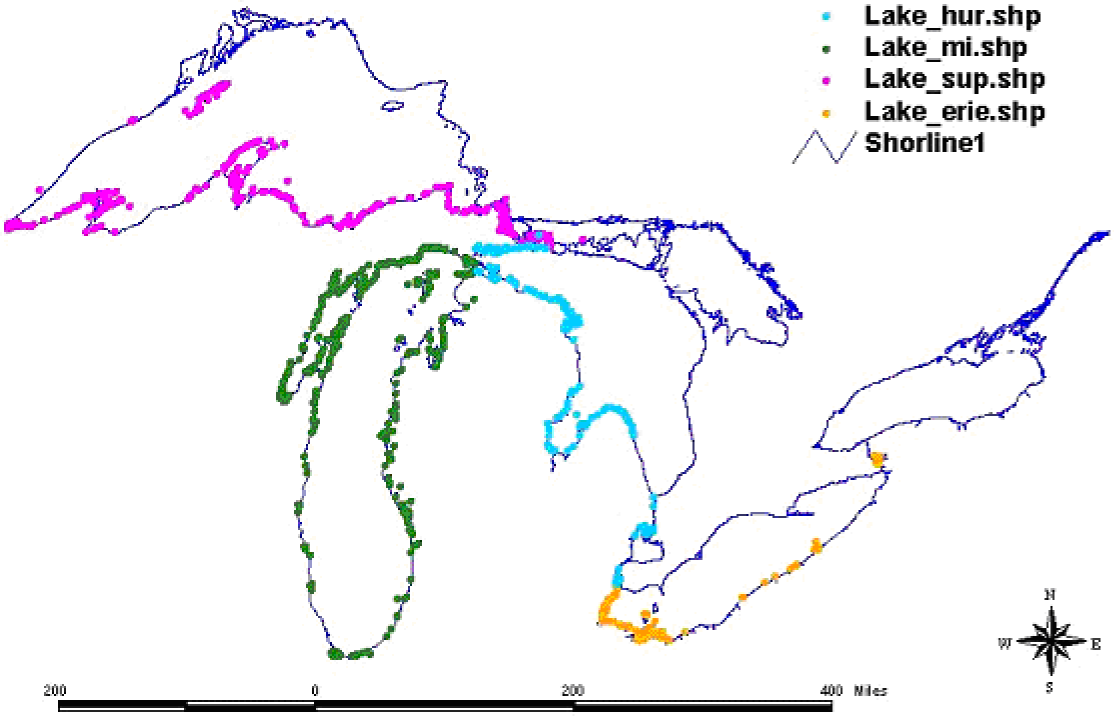
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Fig. 20.2

TABLE 20.1

METRICS REJECTED IN THE EVALUATION PROCESS, BY REASON FOR REJECTION. LISTS 1 AND 2 COMPRISE GROUPS OF SPECIES CREATED FOR TEST PURPOSES; SEE TEXT FOR DESCRIPTION OF OTHER SPECIES GROUPS. BIOMASS METRICS BASED ON 23 WETLANDS FROM 2002; IND = INDIVIDUALS.

Failed Range Test	Failed redundancy test	Failed responsiveness test
Number darter species	Number of sunfish species	Catch per unit effort (list 1)
Number darters, madtoms, sculpin	Number round-bodied suckers	Catch per unit effort (list 2)
Number salmonid species	Number intolerant species (ind; list 1)	Percent lake habitat species
Percent salmonid species (ind.)	Number intolerant species (ind.; list 2)	Number lake habitat species
Number sucker species	Number tolerant species (ind.)	Percent pioneer species (ind.)
Percent great-river species (biomass)	Percent round-bodied suckers (ind.)	Percent Great Lakes species
Percent hybrids (ind.)	Number deep-bodied suckers species	Number Obligate Great Lakes species
Number hybrids	Percent deep-bodied suckers (ind.)	Percent round-bodied suckers (biomass)
Percent sensitive species (ind.)	Percent green sunfish (ind.)	Percent sucker biomass
Number sensitive species	Number of benthic species	Percent tolerant species (list 1)
Percent DELT anomalies (ind.)	Percent benthic species (ind.)	Percent tolerant species (list 1; biomass)
	Percent omnivores (biomass; list O)	Number of planktivores
	Percent omnivores (biomass; list O)	Percent planktivores (ind.)
	Percent omnivores (ind.; list 1)	Percent tolerant species
	Percent omnivores (ind.; list 2)	Percent top piscivores
	Number catfish and sucker species	
	Number of piscivores (list 1)	
	Number of piscivores (list 2)	
	CPUE	

TABLE 20.2
SPEARMAN CORRELATIONS OF FISH ASSEMBLAGE METRICS AND CONNECTING CHANNEL IBI SCORES WITH HABITAT AND WATER QUALITY VARIABLES. HABITAT AND WATER QUALITY DATA WERE AVAILABLE FOR 23 SITES. ALL CORRELATIONS ARE SIGNIFICANT AT THE 0.0001 LEVEL.

	Variable													
Metric	**Mean depth**	**% coarse**	**% sand**	**% fines**	**% wood**	**% sub veg**	**% emg veg**	**% float**	**% highly embed**	**Secchi depth**	**Diss olved oxygen**	**Tem- per- ature**	**pH**	**Cond- uctivity**
Number species	0.43	0.47	0.28	-0.37	0.22	0.26	0.21		-0.49	0.21		-0.24		0.29
Number centrarchid	0.44	0.47	0.21	-0.41	0.37	0.32	0.28	0.18	-0.46	0.17		-0.25		0.23
Number minnows	0.18	0.21	0.22	-0.34	0.23	0.37		0.17	-0.56	0.23		-0.34		0.32
Number benthic invert.	0.19	0.27	0.23	-0.45	0.17	0.23			-0.47			-0.26		
% intolerant species	0.36	0.56	0.32	-0.58		0.21			-0.18	0.28	0.23	0.17	0.18	0.20
% tolerant species	0.17	0.19		-0.19	0.25		0.26							0.19
% detritivores		-0.26		0.27					0.18	-0.28		0.21		
% insectivores	0.25	0.28	-0.17	-0.34	0.23				-0.22			-0.28		0.19
% carnivores	0.23				0.21							-0.28		
% phytophils	0.21		0.18	-0.27		0.58	0.32		-0.45	-0.39		-0.21		
% lithophils	0.24	0.45	-0.18	-0.43					-0.56			-0.23		
% nonindigenous species			-0.21	-0.32	0.29					-0.28		-0.17		
% DELT anomalies				-0.27					-0.22	-0.17	-0.21			
CPUE									-0.30					
IBI	0.38	0.47	0.28	-0.37		0.18			-0.49	0.28	0.21	-0.26	0.15	0.23

TABLE 20.3

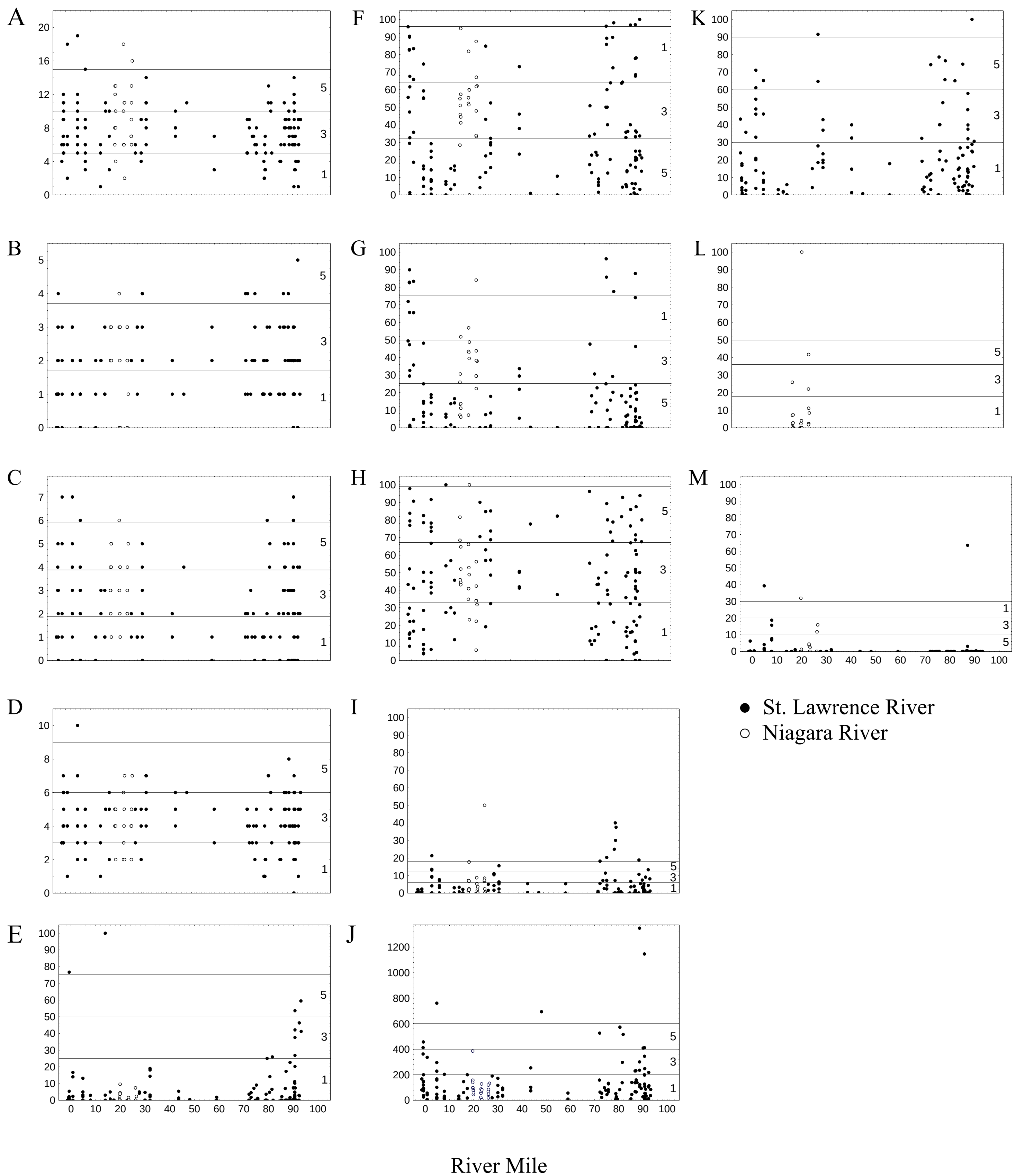
Metrics and scoring criteria for Great Lake coastal wetland index of biotic integrity for drowned river mouth coastal wetlands. NE = Northeast tolerance classification (Halliwell et al. 1999).

	Expectations		
Species Richness and Composition	1	3	5
Number of species	≤ 5	6-10	≥11 (Fig. 20.2A)
Number of centrarchid species	≤1	2-3	≥4 (Fig. 20.2B)
Number of minnow species	≤1	2-3	≥4 (Fig. 20.2C
Number of benthic invertivore species	≤3	4-6	≥7 (Fig.20.2D)
Tolerance and Sensitivity			
Percent individuals as intolerant species (NE)	<25%	26-50%	>51% (Fig. 20.2E)
Percent individuals as tolerant species (NE)	>67%	34-66%	<33% (Fig. 20.2F)
Trophic guilds			
Percent individuals as detritivores	>50%	26-50%	<25% (Fig. 20.2G)
Percent individuals as insectivore species	<33%	34-66%	>67% (Fig. 20.2H)
Percent individuals as carnivores	<6%	7-12%	>12% (Fig. 20.2I)
Abundance, condition, reproduction, and naturalness			
Number of individuals	≤200	201-400	≥401 (Fig. 20.2J)
Percent individuals as phytophils (St. Lawrence)	≤30%	31-59%	≥60% (Fig. 20.2K)
Percent individuals as lithophils (Niagara)	<18%	18-36%	>36% (Fig. 20.2L)
Percent individuals as exotic and non-native species	>20%	10-20%	<10% (Fig. 20.2M)
Percent individuals with DELT anomalies	≥ 2.6%	0.1-2.5%	≤0.1%

TABLE 20.4

Descriptive statistics of index of biotic integrity metrics for Great Lakes connecting channel drowned river mouth coastal wetlands and significance with river mile.

Attribute	**Mean**	**SD**	**Range**	r (p-value)
Number of species	7.76	3.35	1-19	-0.05 (0.573)
Number of centrarchid species	1.90	1.10	0-5	0.11 (0.166)
Number of minnow species	2.04	1.40	0-6	-0.05 (0.504)
Number of benthic invertivore species	4.16	1.65	0-10	-0.02 (0.762)
Percent individuals as intolerant speies (NE)	5.86	13.93	0-100	0.19 (0.017)
Percent individuals as tolerant species (NE)	35.40	29.12	0-100	-0.07 (0.414)
Percent individuals as detritivores	17.67	23.82	0-96.2	**-0.23 (0.004)**
Percent individuals as insectivore species	44.96	26.91	0-100	-0.01 (0.887)
Percent individuals as carnivores	4.30	7.58	0-50	0.11 (0.179)
Number of individuals	132.20	184.39	0-1,349	0.00 (0.954)
Percent individuals as phytophils (St. Lawrence)	20.00	22.45	0-100	0.13 (0.132)
Percent individuals as simple lithophils (Niagara)	11.05	22.42	0-100	0.19 (0.406)
Percent individuals as exotic and non-native species	1.56	6.92	0-63.52	-0.22 (0.005)


A
B
C
D
E
F
G
H
I
J
K
L
M
● St. Lawrence River
○ Niagara River
River Mile

St Lawrence and Niagra Rjiver

A. Number of species
B. Number of centrarchid species
C. Number of minnow species
D. Number of benthic invertivore species
E. Percent individuals as intolerant species (NE)
F. Percent individuals as tolerant species (NE)
G. Percent individuals as detritivore species
H. Percent individuals as insectivore species
I. Percent individuals as carnivore species
J. Number of individuals
K. Percent individuals as phytophil species (St Lawrence)
L. Percent individuals as lithophil species (Niagra)
M. Percent individuals as exotic and non-native species

21 Development of an Index of Biotic Integrity for Fish Assemblages in Drowned River Mouth Coastal Wetlands of Lake Huron

Thomas P. Simon & Ronda L. Dufour

CONTENTS

21.1 INTRODUCTION

The Lake Huron ecosystem has been virtually neglected with regards to nearshore fish community assemblage studies (Berst and Spangler 1973, Goodyear 1982, Munawar et al. 1995). Lake Huron has experienced substantial changes as a result of exotic invasion (Keller et al. 1987, Jude et al. 1995, Busiahn and McClain 1995), wetland loss and degradation (Wilcox 1995), toxic chemical contamination (Thomas 1973, Allender and Green 1976, Rossman 1995), and the increase in chemical, physical, and biological impacts to Saginaw Bay habitats (Bredin and Goudy 1995). This has had a dramatic effect on remaining coastal wetlands in Lake Huron embayments and nursery habitats for fishes (Goodyear et al., 1982).

The loss of biological diversity in Lake Huron has shown that 76 fish species were indigenous (Bailey and Smith 1981), but has changed dramatically during the last century (Smith 1968, 1972; Berst and Spangler 1973). The dramatic changes in the Lake Huron fish community were the result of excessive fishing (Van Oosten et al. 1946, Berst and Spangler 1973; Eshenroder et al. 1992), habitat deterioration (Keller et al. 1987), and introduction of exotic species (Berst and Spangler 1973, Coble et al. 1990). The collapse was evident by the 1960’s with the loss of the lake trout fishery, demise of the burbot populations, and the extirpation of four species of deepwater cisco. As a result, large-scale ecosystem management objectives for fish community (Ebener et al., 1995) have been developed. These actions have resulted in the improvement of coregonids, yellow perch, walleye and channel catfish populations and successful reproduction of lake trout in small numbers (Edsall et al. 1992, Ebener et al. 1995).

The index of biotic integrity (IBI) was developed for assessing fish assemblages of small streams (Karr, 1981; Karr et al., 1986), but no application has been developed for Lake Huron fish assemblages. The purpose of this study was to develop an IBI for drowned river mouth coastal wetlands for Lake Huron on the United States shoreline in order to assess the status and condition of remaining coastal wetlands. This project was conducted in drowned river mouth wetlands following wetland definitions by Keough et al. (1999).

21.1.1 OVERVIEW OF IBI APPLICATIONS FOR LAKE HURON

No application of the IBI has been developed for Lake Huron. Intensive long-term databases do not exist for the nearshore littoral waters of Lake Huron, however, open lake and off-shore habitats does exist (U.S. Geological Survey, Biological Resources Division, Great Lakes Science Center). The majority of fish studies in Lake Huron have focused on the changes in the assemblages of the top predator species (Van Oosten et al. 1946, Eshenroder et al. 1987, Coble et al. 1990). Species indicators have been used to increase fisheries population targets for assessing Lake Huron productivity (Ebener et al. 1995).

21.2 METHODS

21.2.1 STUDY AREA, SITE SELECTION, AND SAMPLE DESIGN

Lake Huron is the second largest of the Great Lakes with a surface area of 59,500 km^2 (Herdendorf et al. 1982). It possesses the largest catchment area of all the Great Lakes (Hough, 1958), and is considered oligotrophic. The lake basin contains a number of islands, shoals, and troughs associated with glacial scour of the Paleozoic rock (Beeton and Saylor 1995). Lake Huron occupies 31% of its basin and has extensive shoreline.

The mean depth of Saginaw Bay is shallow, with large areas less than 10 m (Beeton and Saylor 1995).

In order to develop an IBI, with similar rationale to that developed for other regions of the world (Karr, 1981; Karr et al. 1986; Hughes and Oberdorf 1999), drowned river mouth wetlands were surveyed along the United States shoreline of Lake Huron (Fig. 21.1). Twenty drowned river mouth coastal wetlands were selected using a random selection process (Table 21.1). The western shore of Lake Huron is a large series of wetland complexes that extends from Meade Island to South Thunder Bay and from Au Sable Point to the White Rock Area complex (Wilcox 1995). Few pristine wetlands remain in Lake Huron, thus our sites are known to represent a wide range of habitat quality types from “good” to “poor” quality coastal wetlands.

Sample areas were randomly selected and equally weighted so that sufficient numbers of small, medium, and large wetlands were surveyed to provide the most diverse fish collections. Locations within the wetlands were targeted to ensure that sampling occurred in the most diverse and natural remaining habitat within the randomly selected wetland. This rapid assessment approach ensured that a representative sample of the fish species was collected.

21.2.2 Collection

Daytime electrofishing surveys were conducted in drowned river mouth coastal wetlands using a variety of gear types based on wetland wetted widths at 23 drowned river mouth coastal wetlands (Table 21.1). Daytime fishing provided a similar species catch compared to night electrofishing, but eliminated the influence of lake species that were transients into the coastal wetlands at night. Electrofishing in small wetlands with wetted widths less than 3.3 m were surveyed using a Smith-Root generator backpack system for about 15-45 minutes. The minimum sampling distance was 150 m, but sites represented 35 times the wetted width. In larger wetted widths (>3.4 m – 10 m), a Smith Root tote barge electrofishing unit was used, while in non-wadeable drowned river mouth wetlands a Smith-Root boat-mounted electrofishing unit was used at depths less than 2 m for maximum distances of 500 m and minimum times of 1800 s (Simon 2000). Site specific changes in the frequency and pulse width were made at each site, but typically each unit supplied 300-500 volts and 4-6 amps of DC current.

All observed fish were netted using 4.7 mm mesh dipnets and placed into an on-board holding tank until the completion of the reach. Fish were identified to species, measured for minimum and maximum length by species, counted, batch weighted, and inspected for deformities, eroded fins, lesions, and tumor (DELT) anomalies. A voucher specimen or photograph was retained of each species collected and small specimens of minnows and other non-game species were preserved for later analysis in the laboratory using Smith (1979), Trautman (1981), and Becker (1983).

21.2.3 Metric development

Classification criteria for fish species collected from drowned river mouth coastal wetlands for Lake Huron were developed by reviewing fish assemblage structure and function literature, published life history, and tolerance information (Simon et al., Chapter 12). We evaluated more than 50 characteristics of fish communities in selecting metrics among the five main classification categories that were incorporated into multimetric indices for coastal wetlands of Lake Huron (Table 21.2). All metrics are plotted against the wetted width of the wetland. This enabled compatible comparisons between similar sized wetlands (Fig. 21.2).

Structural metrics incorporated community structure, key indicator species, and compositional group membership attributes. Functional metrics included sensitivity and tolerance metrics, percent individuals based on different trophic ecology, macrohabitat specialists, and reproductive guilds. Relative abundance was based on the number of fish collected within a given sampling zone based on the collection protocol (Simon 2000). Scoring criteria for this calibration follows Karr et al. (1986), which uses three levels based on a trisection of the data (Table 21.3). For a metric to score a “5” the attribute needs to be representative of the reference condition, a score of “3” shows deviation from the reference condition, and a score of “1” suggests the metric is significantly different from the reference condition (Karr 1981).

In order to validate this modification of the IBI, we chose to compare our rating scores to varying measures of environmental perturbation. We calculated the IBI score using data from samples collected between June and September 2002. We evaluated 50 metrics for suitability and eliminated many based on a range test, colinearity, skewness, and statistical correlations to a measure of disturbance (Hughes et al., 19xx). We judiciously kept the same rationale as Karr et al. (1986) when substituting metrics, which resulted in the 12 metrics that were chosen for this application.

Standards of quality for validating the IBI were considered from:1) a subset of drowned river mouth wetlands demonstrating minimum and maximum degradation based on water quality monitoring, and 2) a comparison of percentages of wetland cover types and respective basins' percentages with landuse/cover types and density of roads using a Geographic Information System.

21.2.4 STATISTICS

Patterns in species composition, group membership, and functional percentages were scaled against wetted wetland width to determine if a linear relationship existed. Scoring lines were drawn to trisect the data such that the maximum observed line (MOL) included a trisection of the data beneath the highest observed point (Emery et al., 2003). We chose the MOL approach rather than the Maximum Species Richness line approach since we believe that few high quality coastal wetlands remain in Lake Huron. Thus, we wanted to ensure that we did not overestimate the quality of any of the coastal wetlands by rating them too high. Metric hypotheses were made *a priori* and qualitatively examined to determine if the patterns found fit these expectations based on a range test.

Spearman correlation ($p < 0.05$) were used to examine the relationship between wetland qualities among a "best remaining" group of wetlands and an "impacted" set of wetlands (Conover, 1971).

21.3 RESULTS AND DISCUSSION

21.3.1 SPECIES COMPOSITION

The fish communities of Lake Huron drowned river mouth coastal wetlands were only partially sampled, but catches of 56 total species included 52 native taxa. Sixteen taxa occurred at fewer than 4.3% of the sites (Table 21.3), and eight of these, slimy sculpin *Cottus cognatus*, finescale dace *Phoxinus neogeus,* American brook lamprey *Lampetra appendix,* burbot *Lota lota*, hornyhead chub *Nocomis biguttatus,* river shiner *Notropis* *blennius,* tadpole madtom *Noturus gyrinus,* and stonecat *Noturus flavus* were rare species. None of these species are classified as endangered or threatened in Michigan (Michigan DNR URL: http://www.state.mi.us/orr/emi/admincode.asp?AdminCode=Single&Admin_Num=29901021&Dpt=NR&RngHigh=). Several species have been collected that are indicators of high water quality including blacknose shiner *Notropis heterodon*, blackchin shiner *Notropis heterolepis,* finescale dace *Phoxinus neogeus,* and pearl dace *Margarisus margarita* compared to historical information from 20 years earlier (Goodyear et al. 1982). Among the ubiquitous species in these shallow water areas were nine taxa that occurred at 50% of the sites, white sucker *Catostomus commersoni,* brook stickleback *Culaea inconstans*, johnny darter *Etheostoma nigrum*, bluntnose minnow *Pimephales notatus*, banded killifish *Fundulus diaphanus*, rock bass *Ambloplites rupestris*, and central mudminnow *Umbra limi*. Four of these nine taxa are considered tolerant to extreme levels of environmental degradation.

21.3.2 STRUCTURAL ATTRIBUTES OF DROWNED RIVER MOUTH COASTAL WETLANDS IN LAKE HURON

Drowned river mouth coastal wetlands of Lake Huron showed that the number of native species is one of the most important diversity indices (Table 21.1). We hypothesized that the number of species would increase with biological integrity. The number of species ranged from one to 18 species at a single embayment. Only the number of centrachid species showed a statistically significant relationship with wetland wetted stream width (Table 21.3). Small wetland wetted stream widths did not have a high number of centrarchid species. We substituted the remaining compositional metrics including the number of centrarchid species (replacement for number of sunfish species), percent individuals as lake habitat species (replacement for number of darter species), and number of benthic species (replacement for number of sucker species).

The number of benthic species is a group of fish that are found along the bottom and littoral habitats of coastal wetlands (Simon et al., Chapter 12). This guild is expected to increase with increasing biological integrity. Lake Huron drowned river mouth coastal wetlands ranged between zero and 5 species (Fig. 21.2B).

Centrarchid species are important components of fish assemblages in drowned river mouth wetlands within Lake Huron. The number of centrarchid species ranged between zero and 4 centrarchid species at a site (Fig. 21.2C). We expected to find a greater number of centrarchid species with high quality wetlands. The low water year during which this study was conducted may account for the significant relationship we observed in the number of centrarchids with stream wetted width (Table 21.3). We anticipate that this situation will improve when water levels reach normal levels.

The percent individuals as lake habitat species was substituted for the number of darter species since we anticipated finding increasing numbers of species with recovery of the Great Lakes system (Fig. 21.2D). This metric is currently serving as a “reality check” within our IBI, since we expect to find keystone Great Lakes species, however, due to the largescale degradation basin-wide this metric is underattaining for most of the wetlands we surveyed. The range in lake habitat species was between zero and 91 percent.

21.3.3 Species tolerance and sensitivity

Regional descriptions of sensitivity were completed by Ohio Environmental Protection Agency (1989), which classified fishes into broad categories of sensivity. Ohio EPA (1989) classified 13 of 56 (23.2%) species occurring in Lake Huron as intolerant, while 12 (21.4%) species occurring in Lake Huron were considered tolerant. Karr et al. (1986) warned against classifying too many species as intolerant so that this metric can serve as an early warning to declining conditions. Karr et al. (1986) recommended that less than 10% of the fauna be considered sensitive. Despite the higher percentage of intolerant species in this study, we recognize that the distribution of these species in the Lake Huron ecosystem will cause the classification for any portion of the lake to be closer to Karr et al.'s recommended number. We hypothesized that intolerant species will increase with biological integrity. Our results showed that drowned river mouth coastal wetlands in Lake Huron ranged between zero and 100 percent. The Nayanguing Point (100%) and South Thunder Bay (95.7%) wetlands showed the highest percentages of intolerant species (Fig. 21.2E).

The increase of tolerant species is an indicator of degraded conditions and is inversely correlated with biological integrity. Ohio EPA (1989) classified 12 species as tolerant to environmental disturbance. We did not anticipate that there should be any relationship with lake position or stream size for this metric (Fig. 21.2F), since tolerant species should be no more abundant at any given location within Lake Huron. Thus, our results show that the range of tolerant species is equally distributed among the entire lake. The percent individuals as tolerant species ranged between 7 and 100%.

21.3.4 FUNCTIONAL ATTRIBUTES OF FISH ASSEMBLAGES

We evaluated several trophic guild categories for their ability to explain the biological integrity of drowned river mouth coastal wetlands. We followed Goldstein and Simon (1999) in the assignment of trophic guilds for Lake Huron species. In addition, we hypothesized that the percent individuals as insectivores and carnivores would increase with biological integrity, while percent individuals as detritivores would decrease with biological integrity. The percent individuals as detritivores replaced the percent individuals as omnivores metric. Species such as gizzard shad *Dorosoma cepedianum* were combined with carp *Cyprinus carpio* since both met the original definition of an omnivore. This metric is inversely scored so higher percent individuals indicate degradation. Our results showed that ranges were between zero and 55.7 percent (Fig. 21.2G).

Neither the percent individuals as insectivore or carnivore metrics showed a relationship with wetted wetland width (Table 21.3; Fig. 21.2H and 21.2I, respectively). We did not need to modify the scoring criteria for percent individuals as carnivores (Simon and Dufour 1998) since inflated percent individuals as carnivores was not observed. High numbers of carnivores is an unstable situation, which cannot be sustained for long periods of time; however, Lake Huron carnivores never exceeded 50% of the catch for any reach. Thus, we recognized that higher percent individuals as carnivores occur above what is seen in streams, but is similar to what is found in Great Rivers (Emery et al., 2003). The percent individuals as carnivores ranged between zero and 28.7% (Fig. 21.2I).

21.3.5 INDIVIDUAL HEALTH, CONDITION, AND ABUNDANCE

We did not change Karr et al.'s (1986) application of the relative abundance of individual fish collected at a site (Fig. 21.2J), and retained the percent individuals with deformities, eroded fins, lesions, and tumor (DELT) anomalies (Sanders et al., 1999) following Karr's original criteria. However, we did adopt a substitute metric for percent individuals as hybrids. We chose to replace this metric with the percent individuals as phytophilous spawning species since we observed that high quality open lake coastal wetland habitat had a variety of submergent, emergent, and floating vascular plant species. We followed Simon (1999) in the placement of species into this guild assignment (Simon et al., Chapt. 12). Our expectation was that percent individuals as phytophilous spawning species would increase with biological integrity. Our results showed that the percent individuals as phytophilous spawning species ranged from 1.3 to 100 percent (Fig. 21.2K).

21.3.6 INDEX VALIDATION

We used two approaches for validating an index of biotic integrity for drowned river mouth coastal wetlands of Lake Huron. Wetlands were selected that represented the extremes of conditions and we evaluated these "least impacted" and "impaired" wetland fish assemblages using our newly calibrated index. We evaluated patterns in the IBI scores among these two groups of wetlands in order to determine the sensitivity of the modified index to different perturbations (Fig. 21.3). We saw a significant difference between impaired sites and all remaining locations. These remaining sites represented qualities ranging between "fair" and "good".

In addition, we evaluated statistical relationships between each metric and the wetted wetland width. By evaluating patterns between metrics and wetland size we effectively evaluated differences attributed to large scale land use, ecoregions, and tributary influences (Table 21.3). The only metric that showed a statistically significant relationship with wetland wetted stream width was the number of centrarchid species. This result is most likely a result of the low water year that sampling was conducted. Although we think that the loss of centrarchid species in these systems is a negative result and a sign of the loss of biological integrity, we suspect that the number of centrarchid species will increase during normal water years.

21.5 CONCLUSIONS

Fish community assessments based on a modified Index of Biotic Integrity (IBI) for drowned river mouth coastal wetlands enabled us to compare the environmental degradation between areas of Michigan's Lake Huron. Electrofishing catches at 23 sites were collected between June and September 2002 providing a ranges of values between "good" and "very poor" wetlands. We evaluated over 60 metrics using a range test, skewness, colinearity, and correlation analysis to select 12 metrics. These metrics were validated using sites that were classified as impaired during the collections compared to the remaining sites.

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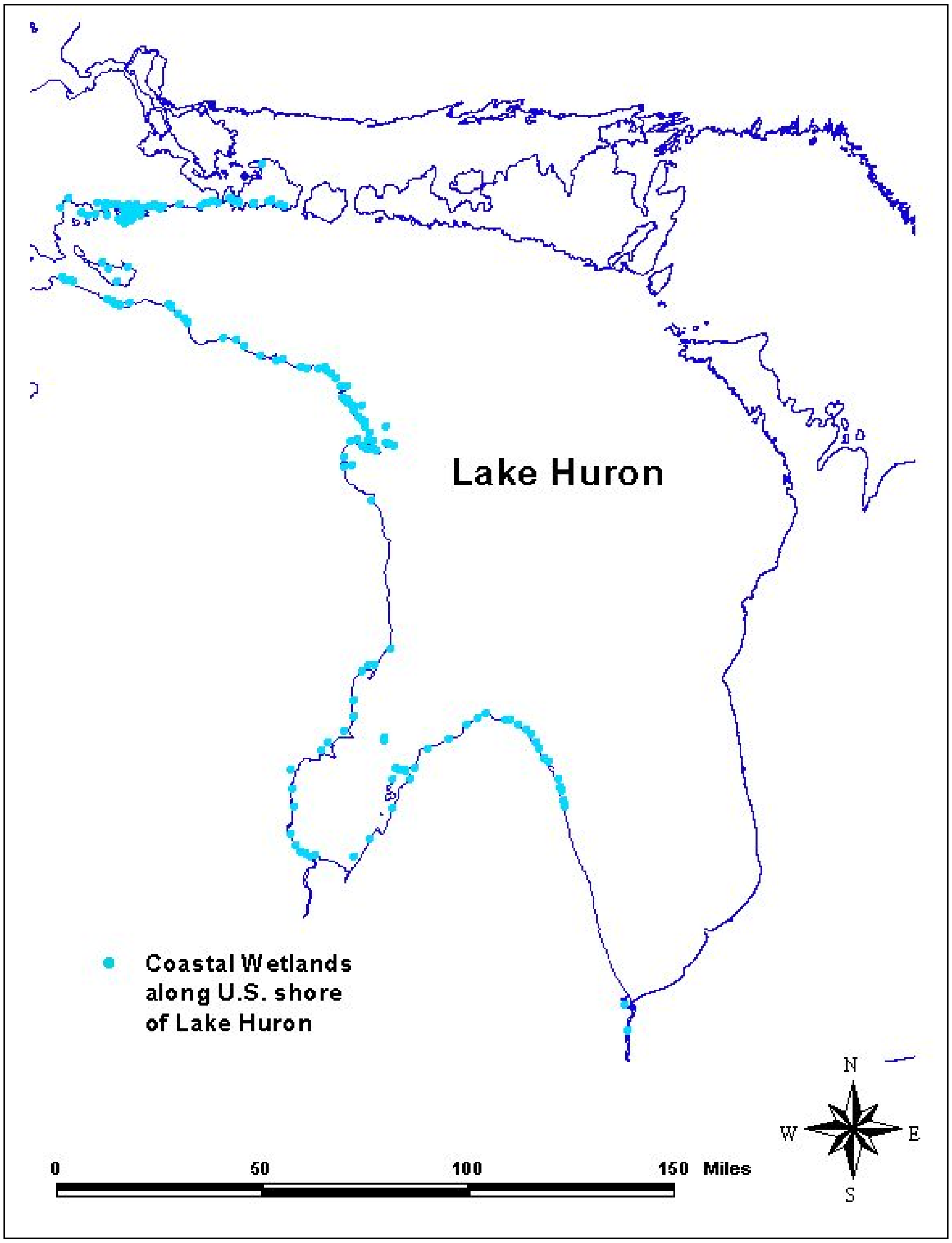
Figure Captions

21.1 Distribution of drowned river mouth coastal wetlands in Lake Huron sampled during this study between June and September 2002.

21.2. Metric expectations and scoring relationships for thirteen metrics used to assess biological integrity of drowned river mouth coastal wetlands of Lake Huron. A. Number of species, B. Number of centrarchid species, C. Percent individuals as obligate Great Lakes species, D. Number of lake habitat species, E. Percent individuals as intolerant species (NE), F. Percent individuals as tolerant species (NE), G. Percent individuals as detritivores, H. Percent individuals as insectivore species, I. Percent individuals as carnivores, J. Number of individuals, K. Percent individuals as phytophils, L. Percent individuals with DELT anomalies, and M. Percent individuals as exotic and non-native species.

21.3 Validation of a modified Index of Biotic integrity for Lake Huron showing relationships between “sampled” (AI) and “impaired” (C) drowned river mouth coastal wetlands in Lake Huron.


Lake Huron
Coastal Wetlands along U.S. shore of Lake Huron
0
50
100
150 Miles
N
W
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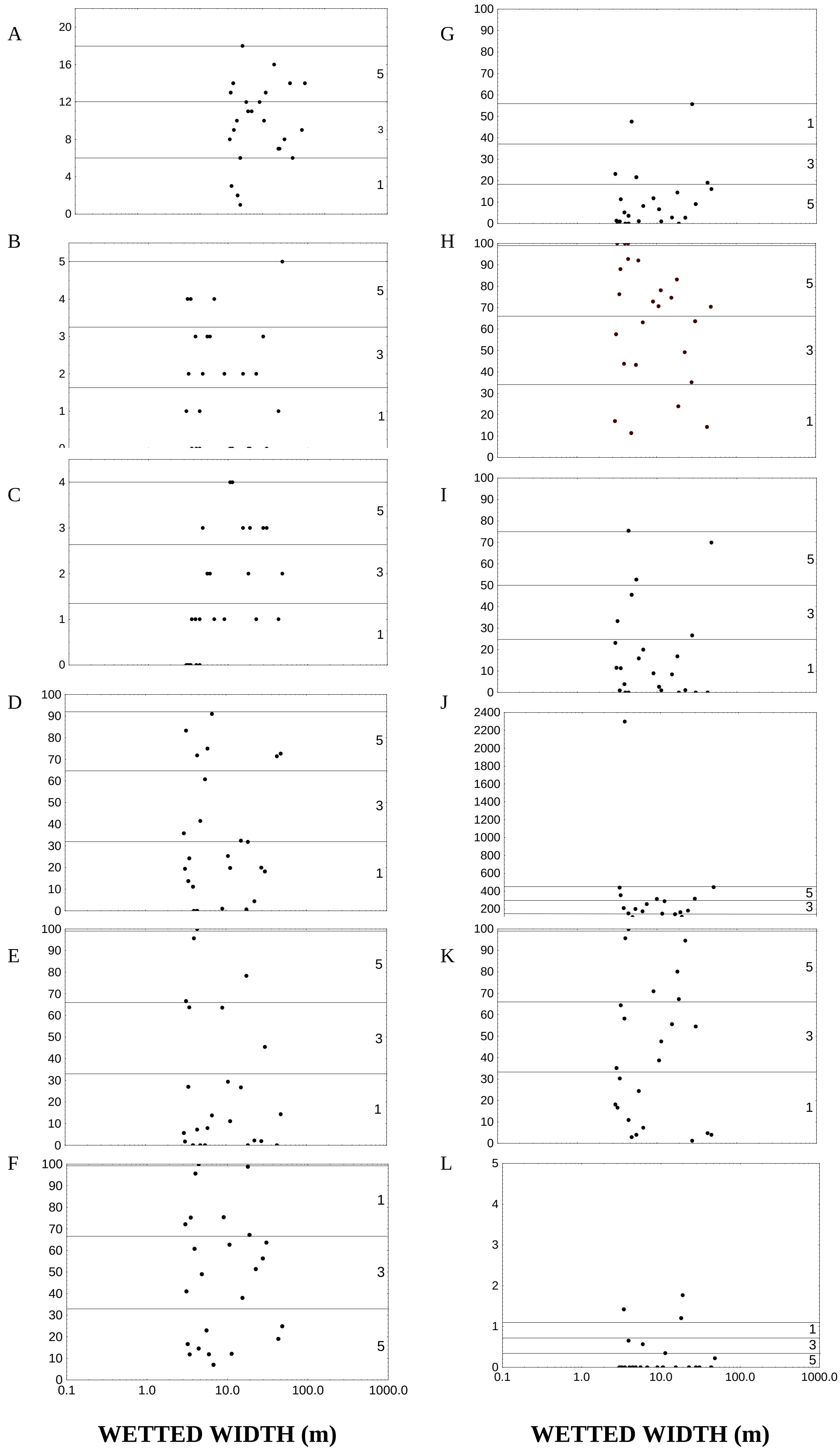
A
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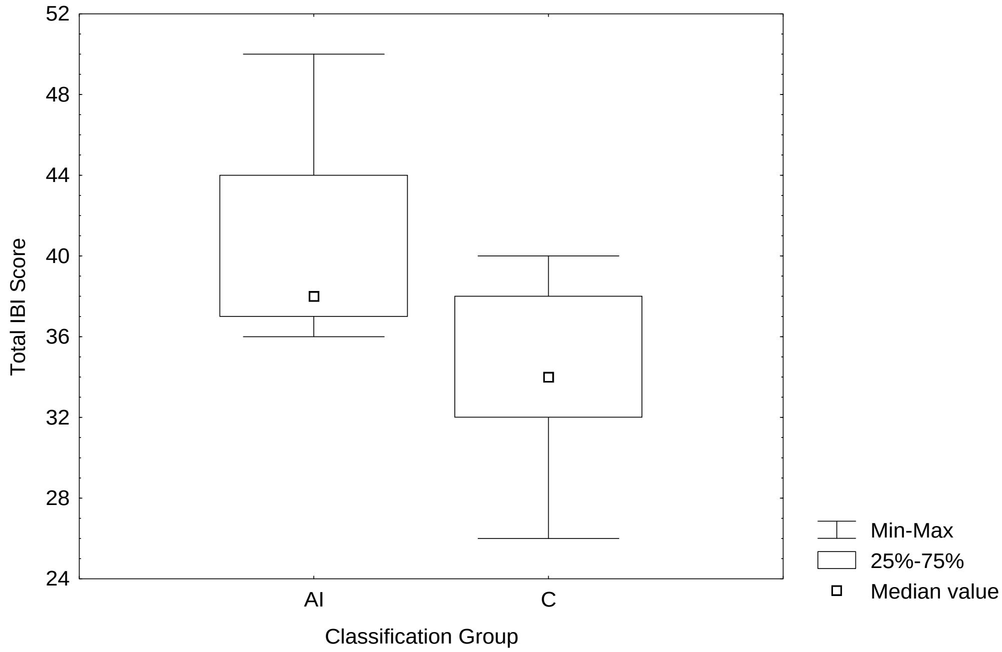
52
48
44
40
36
32
28
24
Total IBI Score
AI
C
Classification Group
Min-Max
25%-75%
Median value

TABLE 21.1.

List of drowned river mouth coastal wetlands including Herdendorf number (Herdendorf et al. 1981), wetland size, wetted width of sites, and geographic coordinates for sites sampled in Lake Huron during 2002.

Wetland Name	Herdendorf Number	Wetted width(m)	Size	Latitude	Longitude
Willow River wetland Huron Co: Port Austin Twp.	35.1 35.2	73 272	15.4 27.6	44 02 11	82 49 55
Munuscony River wetland	49	1,205	43	46 12 32	84 15 26
East Saginaw Bay wetland Tuscola Co.: Akron Twp	50	16,730	18.8	43. 67 67	83.5699
Linwood Area wetland Bay Co: Kawkawlin Twp.	56	85	10.6	43 44 01	83 57 04
Nayanguing Point wetland Bay Co.: Pinconning Twp.	57	2,135	4.4	43 49 48	83 55 05
Schnitzelbank Creek wetland Iosco Co;, Baldwin Twp	59	2,640	3.5	440313	833844
South Thunder Bay wetland Alpena Co: Alcona Twp.	66	7,241	4.0	44 51 35	83 19 14
Whitefish Bay wetland Alpena Co: Alpena Twp	74	180	3.0	45 04 36	83 22 00
El Cajon Beach # 1 wetland Alpena Co:, Alpena Twp	83	45	9.0	45 05 50	83 18 33
Grass Creek wetland Alpena Co: Alpena Twp.	85	25	22.6	45 07.37	83 18.58
Bell River wetland Presque Isle Co:, Presque Isle Twp	98	335	30.5	45 15 29	83 24 47
Thompson Harbor (Grand Lake outlet wetland) Presque Isle Co.: Krakow Twp	105	82	11.3	45 20 26	83 34 15
St. Martins Bay wetland Mackinac Co	140	4,109	3.2	45 59 53	84 30 26
Steel Creek wetland Mackinac Co.: Clark Twp.	146	375	3.9	46 00 38	84 28 20
Mackinac Creek wetland Mackinac Co: Clark Twp	149	25	3.1	46 00 21	84 24 56
Mackinac Bay wetland Mackinac Co: Clark Twp	150	35	4.4	45 59 54	84 23 31

Flowers Creek wetland Mackinac Co, Clark Twp	170	40	5.5	4559.66	84 19.11
Scotty Bay (Beaver Tail Creek) wetland Mackinac Co, Clark Twp	176	40	5.9	45 58 46	84 12 09
Albany Creek wetland Chippewa Co:, De Tour Twp.	178	25	6.7	45 58 00	84 04 35
St. Vital Bay #4 Chippewa Co, De Tour Twp	182	12	3.4	45 58 00	83 59 14
Carp River wetland Mackinac Co	700		48.0	46 01 30	84 41 45
Wigwam Bay wetland (Pine River) Arenac Co: Standish Twp	707		18.0	43 58 30	83 51 17

TABLE 21.2

Significance of select metric attributes considered for inclusion in a drowned river mouth coastal wetland IBI for Lake Huron.

Attribute	t-value	df	p
Number of species minus exotic and non-native species	1.718242	22	.099796
Number of darter, madtom, sculpin species	0.892805	22	.381624
Number of sunfish species	0.949223	22	.352819
Number of centrarchid species	2.077575	22	.049627
Number of benthic species	1.031754	22	.313397
Number of sensitive species	0.562400	22	.579530
Percent individuals as sensitive species	0.043664	22	.965566
Percent individuals as intolerant species (Lake Erie)	-0.661010	22	.515469
Percent individuals as tolerant species	-0.778881	22	.444347
Percent individuals as tolerant species (Lake Erie)	-0.048116	22	.962058
Number of lake habitat species	0.801791	22	.431250
Percent individuals as lake habitat species	1.000232	22	.328073
Number of obligate Great Lakes species	1.752491	22	.093617
Percent individuals as obligate Great Lakes species	-0.950119	22	.352374
Percent individuals as insectivores	0.103935	22	.918162
Percent individuals as carnivores	-0.183995	22	.855703
Percent individuals as pioneer species	1.572982	22	.129995
Percent individuals as detritivores	0.122521	22	.903598
Percent individuals as simple lithophils	-0.588356	22	.562285
Number of phytophil species	0.378970	22	.708343
Percent individuals as phytophil species	-1.04840	22	.305838
Number of benthic invertivores	1.031754	22	.313397
Percent individuals as benthic invertivores	0.554179	22	.585046
Percent individuals as exotic and non-native species	-0.388871	22	.701109
Percent individuals with DELT anomalies	-0.866205	22	.395727
Number of individuals minus exotic and non-native species	-0.345841	22	.732747

TABLE 21.3

Statistical relationships between index of biotic integrity metrics for Drowned River Mouth Coastal Wetlands of Lake Huron and wetted wetland width.

Attribute	Mean	SD	Range	r(p-value)
Number of species minus exotic and non-native	9.75	4.32	1-18	.14 (.510)
Number of benthic species	1.75	1.59	0-5	-.10 (.639)
Number of centrarchid species	1.58	1.32	0-4	.67 (.0001)
Percent individuals as lake habitat species	34.39	29.37	0-91	.04 (.846)
Percent individuals as intolerant species (Lake Erie)	27.6	32.7	0-100	-.12 (.566)
Percent individuals as tolerant species	47.85	29.87	7-100	-.005 (.979)
Percent individuals as detritivores	10.98	14.48	0-55.7	.29 (.177)
Percent individuals as insectivores	63.38	28.02	11.4-100	-.28 (.189)
Percent individuals as pioneer species	17.89	22.35	0-75.5	-.17 (.418)
Number of individuals minus exotic and non-native	2.75	1.59	1-5	-.11 (.609)
Percent individuals as phytophils	41.15	32.56	1.3-100	-.07 (.739)
Percent individuals with DELT anomalies	4.33	1.40	1-5	.03 (.873)

22 Validation of an Index of Biotic Integrity for Fish Assemblages in Drowned River Mouth Wetlands of Lake Erie

Thomas P. Simon, Roger F. Thoma, & Paul M. Stewart

CONTENTS

22.1 INTRODUCTION

The Lake Erie shoreline is a dichotomy of industrial and natural areas and has experienced substantial change as a result of industrialization (Leach 1999, Rasul et al. 1999), cultural eutrophication (Ryan et al. 1999), and exotic species invasions (MacIsaac 1999). These alterations have impaired the condition of most remaining coastal wetlands. The diking of many of the drowned river mouth wetlands, (Herdendorff et al. 1982; Thoma 1999) employed to preserve remaining plant communities, has also contributed to impairments.

Little is known concerning the effects of various land uses and contaminants on fish community structure, function, and aquatic ecosystem health and integrity in these wetlands. Few studies have been conducted in Lake Erie specifically evaluating wetland fish communities though Thoma (1999) evaluated all major Ohio tributaries to Lake Erie using fish community condition as measured by the IBI. Thoma and Simon (2003) evaluated patterns in nutrient impacts on fish assemblages along the Lake Erie shoreline and found that increases in specific groups of fish species correlated with phosphorus loads in the western basin.

We investigated the response of Thoma's coastal wetland drowned river mouth index of biotic integrity for wetlands exposed to various types of industrial land uses. We document community characteristics in near-field and far-field exposure to industrialized areas, evaluate fish indicators of recovery and disturbance, and validate the index of biotic integrity for Lake Erie (Thoma 1999).

22.1.1 Overview of IBI application for Lake Erie

Thoma (1999) developed an index of biotic integrity for nearshore and lacustuary applications for Lake Erie. The term “lacustuary” is defined as a transition zone in a river that flows into a freshwater lake and is the portion of river affected by the water level of the lake. Lacustuaries begin where lotic conditions end in the river (last riffle) and end where the lake proper begins (Thoma 1999). These transition areas are also known as drowned river mouth coastal wetlands following Keough et al. (1999).

Thoma (1999) used data from 125 sites (593 individual collections) to develop a modified index of biotic integrity based on collections from 1993 to 1996. Thoma used ”wetland bay like” sites from Sandusky Bay, East Harbor State Park, and Presque Isle, PA, in part, to develop his lacustuary application. Sites were sampled from the mouth, head, and midsection of each lacustuary. None of the reference sites used for development of the index came from wetlands along the western Lake Erie (Michigan) shore.

22.2 METHODS

22.2.1 Study area, site selection, and sample design

Lake Erie is the southernmost and shallowest of the five Laurentian Great Lakes with a surface area of 25,657 km^2, total volume of 484 km^3, and a maximum depth of 64 m (Bolsenga & Herdendorf 1993). Lake Erie possesses three distinct basins, formed as a result of glacial advances. The shallow western basin (mean depth 7.4 m) is separated from the central basin (mean depth 18.4 m) by a rocky island chain. The central basin is separated from the eastern basin by the Pennsylvania Ridge, a low, wide, submerged sand and gravel ridge (mean depth 24.4 m, maximum depth is 64 m).

The western shore of Lake Erie was once an immense wetland area previously known as the Black Swamp, that included Northwestern Ohio, Northeastern Indiana, and Southeastern Michigan, including the Huron-Erie Lake Plain along the Maumee River. This area has forever been altered by the draining of the Black Swamp and creation of an extensive series of channelized ditches (Cambell 1979). As a result of the low-gradient nature of many streams, drowned river mouth coastal wetlands along Lake Erie were susceptible to invasion by alien species. To reduce the impact of these alien invaders dikes have been constructed along many Lake Erie coastal wetlands to prevent passage into these wetlands by common carp (*Cyprinus carpio*). This also resulted in the elimination of native fish from diked wetlands. Undiked wetland areas have been impacted to some degree by the actions of carp and excess loadings of sediment and nutrients (Ohio Lake Erie Commission 1998, Thoma 1999).

We tested a modified IBI for Lake Erie following the methods and protocols developed by Thoma (1999), by surveying drowned river mouth wetlands along the United States shoreline of Lake Erie in Michigan and Ohio (Fig. 22.1), no drowned river mouth coastal wetlands remain in Pennsylvania. Fourteen drowned river mouth coastal wetlands were selected using a random selection process (Table 22.1). The last remaining drowned river mouth wetlands in Lake Erie along the United States shoreline are located only in the western and central basins. Many of these western and central basin wetlands have been diked (Herdendorf et al. 1982). Few pristine wetlands remain in Lake Erie, thus our sites tend to represent a wide range of habitat quality associated with the industrial and urban land uses surrounding these areas.

Sampled wetlands were randomly selected and equally weighted so that sufficient numbers of small, medium, and large wetlands were surveyed (Simon et al. in press). Locations within each wetland were targeted so that the most diverse and natural remaining habitat within the randomly selected wetland was sampled.

22.2.2 Community Collection

Fish were collected from Lake Erie drowned river mouth coastal wetlands from 14 sites (Fig. 22.1). Reaches were selected so that they represented the entire population of available drowned river mouth coastal wetlands types in Lake Erie. Reaches along the Michigan shoreline bracketed the effects of Detroit and Toledo, while Ohio wetlands were primarily east of Toledo and extending around Sandusky Bay. Based on Geographic Information Analysis, the land use associated with these wetlands is primarily urban. We used BASINS (USEPA version 2) to analyze land use information associated with each of the 14 wetland sites.

Species composition and relative abundance data were gathered through electrofishing surveys. Daytime electrofishing surveys were conducted in drowned river mouth coastal wetlands using a 5.8-m modified V-hull jonboat. A Smith-Root pulsator was used to apply between 240-340 V providing 5 to 6 amps using a 7000-W generator set for 60 pulses per sec. Sampling distances of 500 m and minimum times of 1800 sec. were employed. Anodes were two separately charged electrospheres 1-m in circumference. Two 3-m articulated booms supported by distal floats were positioned about 2.1-m in front of the boat. The booms were positioned at 20-degree angles from the centerline to the port and starboard sides. This arrangement enabled the two electrospheres to be about 4.3-m apart when deployed.

Thoma (1999) found that daytime fishing provided a similar species catch and abundance compared to night electrofishing in lacustuaries, but eliminated the influence of lake species that were transients into the coastal wetlands at night. Electrofishing in small wetlands with wetted widths less than 3.3 m were surveyed using a Smith-Root generator backpack system for about 15-45 minutes shocking time. The minimum sampling distance was 150 m, sites represented a minimum 35 times the wetted width. In larger wetted widths (>3.4 m – 10 m), a Smith Root tote barge electrofishing unit was used, while in non-wadeable drowned river mouth wetlands a Smith-Root boat-mounted electrofishing unit was used at depths less than 2 m for maximum distances of 500 m and minimum times of 1800 sec. (Simon 2000).

For boat sites, each reach sampled was about 500 m, which allowed for a complete habitat cycle within 1-m of shore (Thoma 1999). The amount of fishing time at each station was dependent on habitat complexity and ranged from 2000 to 5000 sec. The greater the number of fish captured and the greater the habitat complexity of the shoreline, the more time spent in a reach. A crew of three individuals undertook all electrofishing efforts. One person was positioned on the bow and was the principal netter; a second person was mid-ship and served as an assistant to collect any fish that the principal netter missed. The third person operated the boat, pulsator, and collected any fish that surfaced at the stern.

All observed fish were netted using 4.7 mm mesh dipnets and placed into an on-board holding tank until the completion of the reach. Fish were identified to species, measured for minimum and maximum length by species, counted, batch weighed, and inspected for deformities, eroded fins, lesions, and tumor (DELT) anomalies (Sanders et al. 1999). A voucher specimen was retained of each species collected and small specimens of minnows and other non-game species were preserved for later analysis in the laboratory using Trautman (1981).

22.2.3 Metric development

Classification criteria for fish species collected from drowned river mouth coastal wetlands for Lake Erie followed Thoma (1999). We evaluated metric response for each of the 12 lacustuary metrics selected by Thoma to determine response to anthropogenic disturbance among Lake Erie wetland types (Table 22.2). All metrics are scored using the criteria developed by Thoma, which he plotted against the percent lacustuary based on drainage area.

In order to validate this modification of the IBI, we chose to compare our rating scores to varying measures of environmental perturbation based on urban land use. Urban land use was determined from BASINS (USEPA version 2) and placed into three categories of use. Least-impacted land use had less than 30% urban land use, far-field urban land use had less than 50% urban land use, and near field sites had greater than 50% urban land use. We calculated the IBI score using data from samples collected between June and September 2001. We evaluated results based on the 14 sites that were randomly selected from the wetlands of Lake Erie. Standards for validating the IBI were based on a subset of drowned river mouth wetlands that represented a continuum of degradation based on water quality monitoring, and near- and far-field comparisons of fish assemblage structure based on distance from urban land uses.

22.2.4 STATISTICS

Community differences between least-impacted, near- and far-field reaches were examined by performing a non-parametric Kruskal-Wallis test to validate the lacustuary Lake Erie IBI followed by a Mann-Whitney U-test (Conover, 1971) used to evaluate longitudinal near- and far-field effects of the industrial landuse. All results are reported at a significance level of $p < 0.10$ (Zar, 1984). A cluster analysis was performed on the matrix of similarity coefficients and these were represented in a dendrogram based on the similarity of site fish community composition (Clarke and Warwick, 1994). Assessment of similarity was done using Bray-Curtis similarity using both a square root transformation using the group average technique and a presence absence transformation also using group average. Results were nearly identical so only the square root transformation dendrogram was used. Descriptions of standard diversity, biotic, and similarity statistics were calculated following Washington (1984).

22.3 RESULTS

22.3.1 FISH COMMUNITY ASSESSMENT, SPECIES DISTRIBUTION, AND RELATIVE ABUNDANCE

Thirty-eight fish species were collected during our investigation of drowned river mouth wetlands in Lake Erie. Banded killifish (*Fundulus diaphanus*) were collected from the Rockwood Road wetland (Table 22.5 wetland #2, LE 93). No rare species were collected during this study, however, species such as bowfin (*Amia calva)*, trout-perch (*Percopsis omiscomaycus*), orangespotted sunfish (*Lepomis humilis*), northern pike *(Esox lucius*), brook silverside (*Labidesthes sicculus*), grass pickerel (*Esox americanus*), blackside darter (*Percina maculate*), tubenose goby (*Proterorhinus marmoratus*), green sunfish

(*Lepomis cyanellus*), and black crappie (*Pomoxis nigromaculatus*) were only collected from a single wetland during the study. Dominant species included common carp, goldfish (*Carassius auratus*), pumpkinseed (*Lepomis gibbosus*), and bluegill (*L. macrochirus*). These species were found at greater than 60% of the wetlands.

Species diversity (Mann-Whitney U-test statistic = 0.835, p = 0.3608) and relative abundance (Mann-Whitney U-test statistic = 0.0205, p = 0.8863), as estimated by CPUE, did not differ significantly between "least impacted" non-urban reaches, near-field, and far-field urban reaches (Table 22.3). The most disturbed areas were those in the Sandusky Bay wetland area (LE 8 through LE 13), while wetlands downstream of the Detroit River mouth along the Michigan shoreline scored as the best remaining sites in Lake Erie and were considered "least impacted".

Otter Creek wetland (LE79) had no fish collected after repeated sampling attempts. La Plaissance Creek wetland (LE 80) and Crane Creek wetland (LE 705) had the greatest number of fish species (15 in each wetland). Along with Otter Creek wetland with no fish collected, Plumbrook wetland (LE 33) had only four species collected during the sampling event. Crane Creek wetland and Old Woman Creek National Estuary had the greatest number of fish caught during the study, while Plumbrook had only 9 fish collected. Species diversity, both Shannon-Wiener and Simpson's, was lowest in Old Woman's Creek Hemming Ditch (LE 37). Both diversity indices were highest at Cherry Isle closely followed by Crane Creek..

The cluster analysis of similarity of Lake Erie drowned river mouth coastal wetlands based on fish species abundance showed that there were four main groupings of sites (Fig. 22.2). Otter Creek and Plumbrook wetlands clustered separately from all of other wetlands. Otter Creek was sampled twice and no fish were collected during either sampling event. Plumbrook Creek had only four species and nine individuals. The next cluster was comprised of two smaller clusters that grouped sites into Western Basin and Central Basin sites.

22.3.2 COMMUNITY STRUCTURE AND FUNCTION

Land use patterns resulting from anthropogenic disturbance correlated with fish community structure and function between near- and far-field urban drowned river mouth coastal wetlands (Table 22.4). Near-field stations differed significantly (Kruskal-Wallis test statistic = 5.8157, $p < 0.01$) from far-field and least-impacted stations in the decline of phytophilic spawning species. The number of sunfish (Kruskal-Wallis test statistic = 2.74282, $p < 0.10$) and percent individuals as phytophilic species (Kruskal-Wallis test statistic = 3.5119, $p < 0.10$) differed significantly between coastal wetlands in Western and Central Lake Erie. The lack of submerged vegetation is a result of anthropogenic disturbance (Ohio Lake Erie Commission 1998, Simon et al. 2001, Thoma 1999, Trautman 1981).

Patterns in non-dominant fish collected from Lake Erie coastal wetlands showed that sensitive facultative wetland species, such as grass pickerel, northern pike, and spotted sucker were generally more numerous in the Western basin than in the Central basin (Table 22.5). The two esocids are sight-feeding predators that require macrophytes to hide and ambush prey, while the spotted sucker is a sensitive species that is found among the submerged vegetation and clear waters (Trautman 1981). The disturbance of riparian wetlands by filling or addition of contaminants and increased nutrients from runoff reduces visibility. In contrast, bluntnose minnow (*Pimephales notatus*) and brown bullhead (*Ameiurus nebulosus*) abundance were significantly different (Mann-Whitney U-test statistic = 4.3815, p = < 0.10) between the Central and Western basin. These species are considered tolerant to contaminants and inhabit a wide range of environmental disturbance (Ohio EPA 1989). Other ubiquitous, tolerant species such as common carp, goldfish, green sunfish, and white sucker did not differ statistically in abundance between Central and Western Lake Erie. The abundance of sensitive fish species, such as rock bass (*Ambloplites rupestris*), grass pickerel, northern pike, brook silverside (*Labidesthes sicculus*), spotted sucker (*Minytrema melanops*), and blackside darter (*Percina maculata*), differed significantly (Mann-Whitney U-test statistic = 5.2834, p = < 0.10) between the Western and Central basins of Lake Erie.

22.3.3 BIOLOGICAL INTEGRITY

The index of biotic integrity (IBI) is a multimetric index that has been modified to evaluate fish community structure and function based on calibration for drowned river mouth coastal wetlands for Lake Erie. The IBI has been calibrated for lacustuaries in Ohio and Pennsylvania (Thoma 1999) and uses some characteristics of warmwater streams from Karr's original index (Karr 1981; Karr et al. 1986).

Based on our assessment of drowned river mouth coastal wetlands in Lake Erie, we found fish community function to range between 'fair' and 'no fish' after repeated sampling depending on location. The Western Basin coastal wetlands scored the best with assessments of 'fair' (IBI score = 36). However, the number of sensitive species and the percent individuals as phytophilic species were below expectations. Several of the metrics met expectations for similar-sized reference wetlands. The 'least impacted' and far-field sites received reference condition scores in 73.3% of the instances, while near-field sites received reference scores 26.7% of the time. The percent individuals as tolerant species, non-indigenous species, phytophilic species, omnivores, and carnivores; and the number of cyprinid species were the only metrics that attained scores equivalent to reference conditions. Coastal wetlands that were in the near-field reaches were assessed as 'very poor' (IBI score = 12 – 18) or 'poor-very poor' (IBI score = 24-26), while far-field sites rated were poor-very poor (IBI score = 26).

22.3.4 Individual condition

The occurrence of deformities, eroded fins, lesions, and tumors (DELT) represents the lowest extremes of biological integrity. Sanders et al. (1999) found higher incidence of DELT anomalies in the presence of contaminants. High percent incidence of DELT anomalies were observed at Old Woman Creek (DELT anomalies = 3.97%), East Bay (DELT = 5.56%), Big Island wetland (DELT = 2.63%), Crane Creek wetland (DELT = 0.53%), La Plaisance wetland (DELT = 2.86%), and Cherry Isle wetland (DELT = 4.40%). Karr (1981) suggested that DELT incidence levels greater than 0.1% are above background environmental conditions and demonstrate a problem. Simon (1998) in a study of southern Lake Michigan coastal wetlands did not observe DELT anomalies at 50 least-impacted wetlands.

22.3.5 Index validation

Comparison between coastal wetland placement within the Western and Central Basins of Lake Erie; land use among least-impacted, near- and far-field wetlands surrounded by urban and industrial land uses; and differences in fish community structure and function showed differences in patterns of non-dominant species. We found that species assemblages were similar between the Western and Central Basins. Dominant exotic species such as common carp and goldfish were ubiquitously distributed in the basin and were not significantly different, however, bluntnose minnow and brown bullhead showed differences in abundance. Likewise, sensitive species were virtually absent from Lake Erie, while some remnant populations were observed in single wetlands throughout the study area.

Relative abundance and distribution patterns of non-dominant species suggested a clear difference between the Western and Central Basin communities (Table 22.5). Facultative wetland species, such as grass pickerel, northern pike, and spotted sucker were present in the Western Basin, but were either rare or absent from the Central Basin. The tolerant bluntnose minnow and brown bullhead were nearly absent from the Western Basin but dominant in the Central Basin wetlands.

The similarity analysis showed that sites were more similar within Basin than between basins (Fig. 22.2). Coastal wetland site cluster fidelity remained within basin, which suggests that either due to the proximity of many of the remaining wetlands in small areas they experience similar impacts or the large scale land use impacts in Lake Erie are masking subtle attribute-related differences in reference condition remaining for drowned river mouth coastal wetlands.

The index of biotic integrity developed by Thoma (1999) was an accurate predictor of biological quality in drowned river mouth coastal wetlands of Lake Erie (Table 22.4). The Western Basin coastal wetlands had either the highest IBI scores or standard diversity, evenness or dominance indices. The Western Basin had the highest number of species, Shannon-Weaver diversity, H max rated as fair and the lowest evenness, Simpson dominance, and CPUE. The Central Basin had the lowest index values and the IBI declined an entire assessment class, differing from the least-impacted reaches by as much as 36 IBI points.

The IBI provided an accurate description of the coastal wetlands of Lake Erie. A good index of biotic integrity has the ability to discriminate between areas with different disturbance levels. None of the wetland reaches sampled were reference condition quality, although several of the metrics showed characteristics of a least impacted wetland. Wetland reaches in the Central Basin rated between 'no fish' and 'very-poor', due to extensive modifications associated with urban land use impacts. Impacts associated within these areas include steel manufacturing and changes in land use resulting in increased turbidity. The Lake Erie lacustuary IBI was able to discriminate between near- and far-field reaches and between far-field and least impacted sites in the Western Basin.

22.4 CONCLUSIONS

An index of biotic integrity for Lake Erie lacustuaries was validated using randomly selected drowned river mouth coastal wetlands in Lake Erie along the United States shorelines in Michigan and Ohio. Three different techniques were used to validate and test the assessment scores derived from the Lake Erie index developed by Thoma (1999). Sites were evaluated based on placement in the Western and Central Basins of Lake Erie, responsiveness was evaluated based on near- and far-field affects from urban land uses, and fish community distribution information was evaluated using non-dominant species distributions among the coastal wetland reaches.

Statistical differences were observed between coastal wetlands in the Central and Western Basins of Lake Erie based on species guild classification. Tolerant species were more dominant in the Central basin, and sensitive and facultative wetland species had higher relative abundance and distribution in the Western Basin. The number of sunfish species and the percent individuals as phytophilic species differed significantly between the Western and Central basins. Cluster fidelity showed that fish communities were generally more similar within Basin than between Basins. IBI scores were able to discriminate between impaired near-field urban coastal wetlands and far-field coastal wetlands. IBI scores assessed sites as ranging between 'no fish' and 'poor' classifications.

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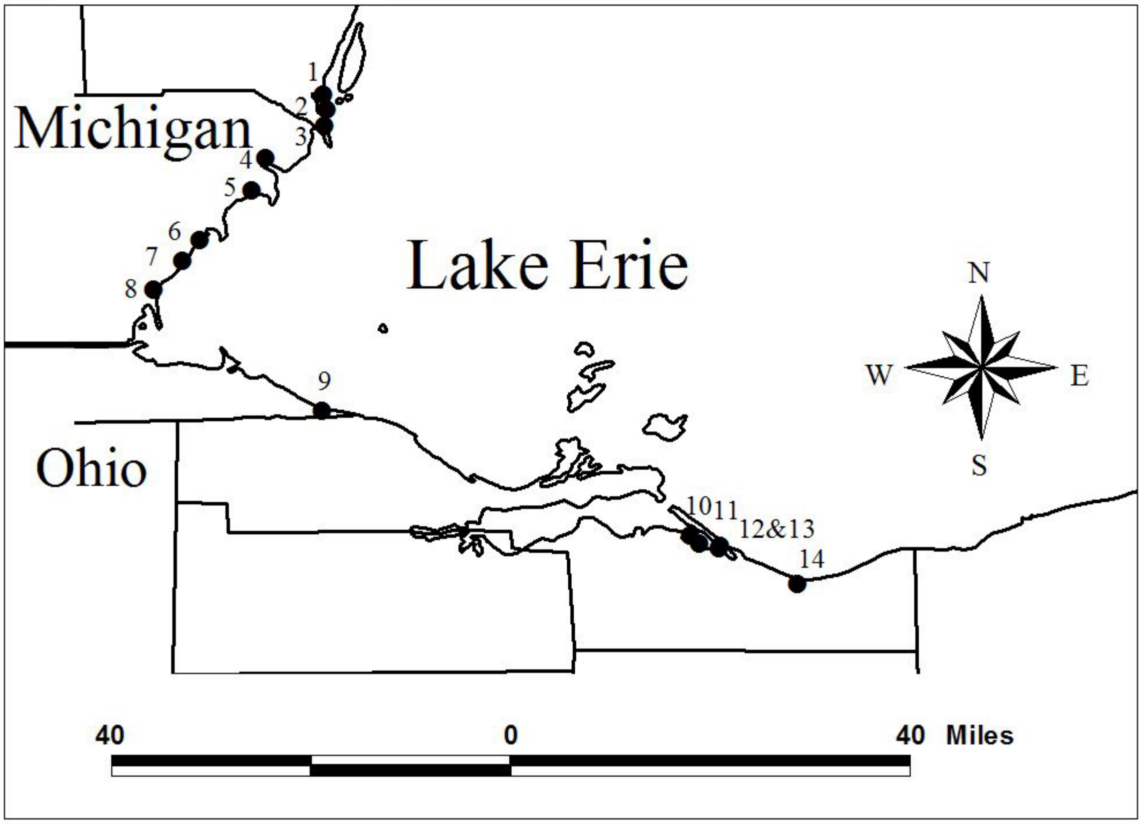
Figure Captions

22.1 Distribution of drowned river mouth coastal wetlands sampled between June and September 2001 in Lake Erie.

22.2. Coefficients of similarity between fish community composition at coastal wetlands in the Western and Central Basins of Lake Erie.

Fig. 22.1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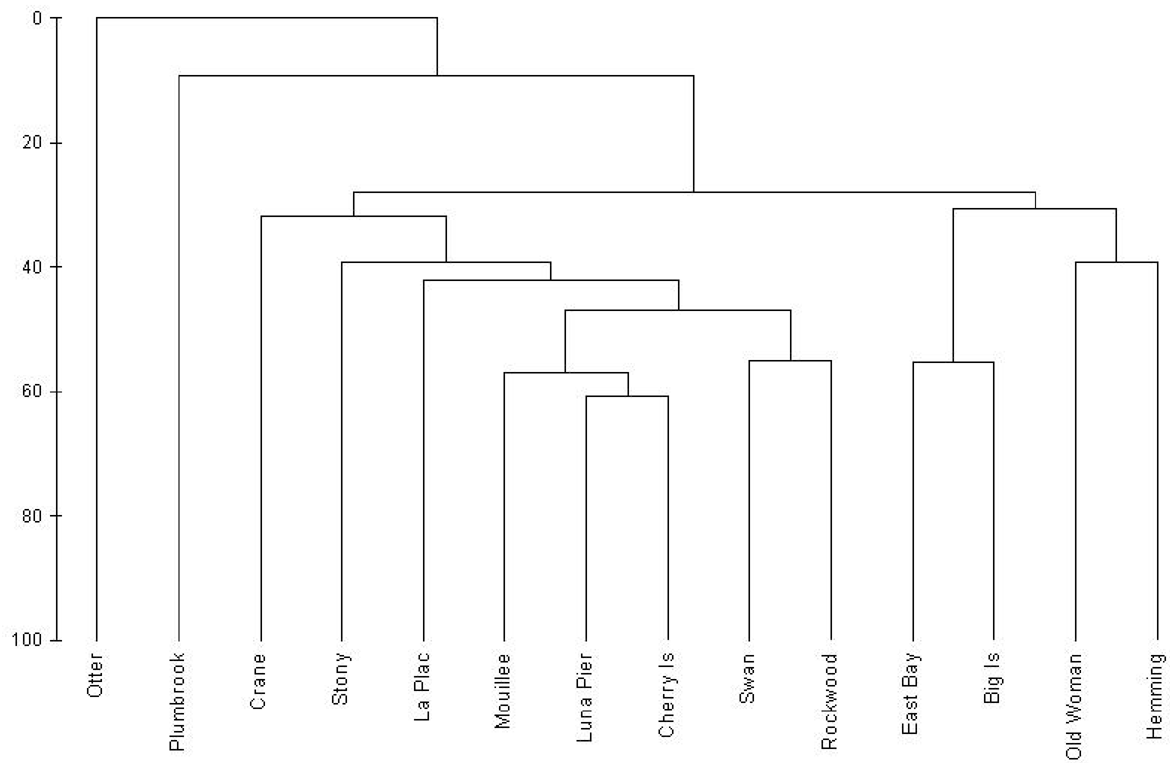
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Plumbrook
Crane
Stony
La Plac
Mouillee
Luna Pier
Cherry Is
Swan
Rockwood
East Bay
Big Is
Old Woman
Hemming

TABLE 22.1.

List of drowned river mouth coastal wetlands including Herdendorf number (Herdendorf et al. 1981), wetland size, drainage area (mi^2), and geographic coordinates for sites sampled in Lake Erie during 2001.

Wetland Name/no.	Herdendorf Number	Wetland Hectares	Drainage Area (km^2)	Latitude	Longitude
Cherry Isle wetland /1 MI: Wayne Co: Brownstown Twp	94	1.20	62.2	42.08389	83.20083
Rockwood Road wetland/2 MI: Wayne Co: Brownstown Twp	93	1.80	49.2	42.0625	83.19556
Mouillee Marsh wetland/3 MI: Monroe Co: Berlin Twp	92	1.67	108.8	42.03889	83.19972
Swan Creek Area wetland/4 MI: Bay Co: Kawkawlin Twp.	90	3.06	95.8	41.99278	83.28472
Stony Creek Area wetland/5 MI: Monroe Co: Frenchtown Twp	86	0.25	134.7	41.94528	83.30444
La Plaisance Creek wetland/6 MI: Monroe Co: Monroe Twp	80	0.23	137.3	41.87333	83.37972
Otter Creek wetland/7 MI: Monroe Co: La Salle Twp	79	1.65	77.7	41.84444	83.40444
Luna Pier wetland/8 MI: Monroe Co: Erie Twp	77	0.55	44.0	41.80194	83.44722
Crane Creek wetland/9 OH: Lucas Co: Howard/Benton Twp.	63-64	0.70	77.7	41.62700	83.20200
Big Island wetland/10 OH: Erie Co: Perkins Twp	38	1.84	259.0	41.44639	82.67000
Hemming Ditch wetland/11 OH: Erie Co: Huron/Perkins Twp	37	1.77	7.8	41.43500	82.65611
East Bay wetland /12 OH: Erie Co: Huron Twp	35	0.01	N/A	41.43139	82.62556
Plumbrook wetland/13 OH: Erie Co: Huron Twp.	33	0.45	5.2	41.42861	82.62778
Old Woman wetland /14 OH: Erie Co: Berlin Twp	26	1.58	129.5	41.37556	82.51389

TABLE 22.2

Metrics and scoring criteria for an index of biotic integrity for lacustuaries (drowned river mouth coastal wetlands) of Lake Erie (Thoma 1999).

	Lacustuary Expectations		
Species Richness and Composition	1	3	5
Total number of species	≤ 7	8-15	≥ 16
Number of sunfish species	≤ 3	4-6	≥ 7
Percent individuals as phytophilic species	≤ 10.9%	11-20%	≥20.1%
Number of benthic species	≤ 2	3-5	≥ 6
Number of cyprinid species	≤ 2	3-4	≥ 5
Tolerance			
Number of intolerant species	≤ 2	3-5	≥ 6
Percent individuals as tolerant species	≤ 10.9%	11-22%	≥ 22.1%
Trophic Guilds			
Percent individuals as omnivore species	≤ 18.9%	19-38%	≥ 38.1%
Percent individuals as top carnivore species	≤ 8.9%	9-18%	≥ 18.1%
Behavior Guilds			
Percent individuals as non-indigenous species	≤ 8.9%	9-14%	≥ 14.1%
Abundance			
Relative number of individuals	≤ 449	450-925	≥ 926
Individual Health and Condition			
Percent individuals with DELT anomalies	≥ 2.6%	0.1-2.5%	≤ 0.1%

TABLE 22.3

Biotic diversity, evenness, and index of biotic integrity scores for drowned river mouth coastal wetlands in Lake Erie Numbers below (1-14) are the same as in Figure 22.1.

Attribute	Western Basin								Central Basin					
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**
No. of species	14	8	9	11	7	15	0	11	15	7	8	6	4	5
CPUE	91	173	34	151	34	70	0	104	376	38	82	72	9	126
N	99	173	68	163	39	131	0	151	422	38	79	72	9	364
Species richness	2.83	1.36	1.90	1.96	1.64	2.87	0	1.99	2.32	1.65	1.60	1.17	1.37	0.68
Evenness	0.81	0.88	0.61	0.68	0.80	0.66	N/A	0.78	0.78	0.64	0.50	0.58	0.88	0.54
Shannon-Wiener ($\log_{10}$)	0.92	0.79	0.58	0.71	0.68	0.78	0	0.81	0.92	0.54	0.45	0.45	0.53	0.38
Simpson diversity ($1-\lambda$)	0.84	0.82	0.64	0.74	0.74	0.73	0	0.80	0.85	0.63	0.46	0.51	0.75	0.49
IBI	26	26	18	30	30	36	0	26	24	16	18	20	24	12
IBI assessment	Poor-Very Poor	Poor-Very Poor	Very Poor	Poor	Poor	Fair-Poor	No Fish	Poor-Very Poor	Poor-Very Poor	Very Poor	Very Poor	Very Poor	Poor-Very Poor	Very Poor

TABLE 22.4

Mean, standard deviation, and range of fish community characteristics from drowned river mouth coastal wetlands in Lake Erie. Statistical significance is based on differences between Black Marsh wetlands near Detroit and Toledo and Sandusky Bay wetlands near Sandusky and Cleveland.

Character	Least-impacted			Near-field (Urban)			Far-field (Urban)			p-value
	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range	
Total number of species	8	4.2	4-14	5.6	2.5	2-10	5.0	5	0-10	0.4792
Number of sunfish species	2.5	0.58	2-3	2.14	1.95	1-5	3.3	3.06	0-6	0.1561
Percent individuals as phytophilic species	42.8	29.1	5.9-76.5	11.7	13.8	0-33.3	26.8	24.7	0-48.6	**0.0328**
Number of benthic species	0.5	0.58	0-1	0	0	0	0	0	0	0.2149
Number of cyprinid species	3.75	0.96	3-5	3.14	0.90	2-4	3.0	3.0	0-6	0.5412
Number of intolerant species	0.25	0.50	0-1	0	0	0	0.33	0.58	0-1	0.1429
Percent individuals as tolerant species	30.7	34.9	7.1-82.4	56.6	27.1	23.6-99.2	33.2	31.1	0-61.5	0.1871
Percent individuals as omnivore species	29.9	33.4	7.1-79.4	53.0	26.0	16.7-91.3	29.7	26.7	0-51.6	0.2029
Percent individuals as top carnivore species	15.7	24.9	1.3-52.9	4.1	3.8	0-9.6	8.7	7.8	0-15.0	0.2287
Percent individuals as non-indigenous species	27.7	29.4	4.3-70.6	50.6	31.7	0-91.3	28.5	25.1	0-47.3	0.2675

Relative number of individuals	72.3	55.2	34-151	115.3	121.4	9-376	132.0	58.0	91-173	0.6773
Percent individuals with DELT anomalies	0.71	1.43	0-2.9	1.81	2.67	0-5.6	1.47	2.53	0-4.4	0.2848

TABLE 22.5

Relative abundance and site-specific distributions of non-dominant fish species in drowned river mouth coastal wetlands in the Western and Central Basins of Lake Erie.

Attribute	**Western Basin**								**Central Basin**					
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**
Ambloplites rupestris	1			1		1			1					
Ameiurus nebulosus			1							2	3	5		10
Amia calva										3				
Esox americanus			1											
Esox lucius						2								
Fundulus diaphanus		9												
Labidesthes sicculus						1								
Lepomis humilis								92						
Minytrema melanops					3	6								
Neogobius melanostomus			5					19						
Perca flavescens		1			2			12			2			
Percina maculata			1											
Percopsis omiscomaycus								1						
Pimephales notatus			3	1		2			62					
Pomoxis nigromaculatus	1													
Proterorhinus marmoratus					10									

23 Development of an Index of Biotic Integrity for Coastal Wetlands of the St. Clair River Estuary, Lake St. Clair, and Detroit River

Thomas P. Simon, Ronda L. Dufour, & Paul M. Stewart

CONTENTS

23.1 INTRODUCTION

Large floodplain rivers, including the connecting channels of the Great Lakes, have received little attention as assessment objectives have been established to evaluate ecological health in North America (Karr et al. 1985; Krieger et al. 1992; Emery et al. 2003; Simon et al., chapter 24). The loss of large floodplain wetlands and the rate at which humans have modified remaining systems make Great Lake connecting channels an endangered resource. These stressors permanently alter these areas as pollution and land use practices, intensive urbanization, and wetland destruction disrupts the connectivity of the floodplain (Herdendorf et al. 1986; Ward and Stanford 1989; Bayley 1995). As a result, the assessment of biological integrity for Great Lake connecting channels show substantial impairment from the cumulative stressors generated from within the Great Lakes basin.

The St. Clair system, including the St. Clair River, Lake St. Clair, and Detroit River, has a unique set of problems associated with assessing their biotic condition. The Delta occurring at the mouth of the St. Clair River, which forms the St. Clair Flats, is perhaps one of the largest and most unique remaining wetland habitats in the Great Lakes. Scale has important implications for defining reference conditions and sampling biotic assemblages. Drowned river mouth coastal wetlands in Lake Erie, which would possibly be the most similar wetland system, have been severely disturbed and lack comparable replication (Munawar et al. 1999; Thoma 1999; Thoma and Simon 2003). This lack of replication cause's problems in finding representative reaches by the virtual absence of only slightly modified reaches; thus, even psuedoreplicate reference reaches are unavailable for comparison. Nevertheless, cumulative biological benchmarks can be defined from a model of “least-impacted” fish assemblage conditions based on the ecology of reach specific factors that are remnants of historical faunal conditions and the current assemblage structure and function at relatively unimpacted or “best remaining” sites. As a starting point, this can substantially improve the environmental assessment of the St. Clair system and can be modified as restoration goals are furthered.

The IBI is considered a family of multimetric indices that have had numerous substitutions of individual metrics depending on different ecosystems (Simon and Lyons 1995; Simon 2000b). The sensitivity and general applicability of multimetric indices are contingent on appropriate customization during their development. In particular, component metrics and their scoring criteria should reflect system-specific attributes of natural biotic communities and responses of those communities to human impacts (Hughes 1995). This flexibility strengthens the ability of multimetric indices to accurately measure environmental degradation in a variety of environments and assemblages.

The purpose of this chapter is to develop an assessment tool that would detect impairment from known sources of impact and assess the biological condition of the aquatic resources of the coastal wetlands of the St. Clair system. We attempt to include metrics that represent measures of habitat protection, antidegradation, and ecosystem restoration in the Great Lakes. We follow three major steps in the development process as outlined by Simon et al. (chapter 20) including, 1) defining reference conditions, 2) selecting metrics and analyzing the relationships between these metrics and human impacts on water and substrate quality, and 3) setting metric scoring criteria.

23.2 METHODS

23.2.1 STUDY AREA, SITE SELECTION, AND SAMPLE DESIGN

The St. Clair River, Lake St. Clair, and the Detroit River form a significant series of connecting channels between Lake Huron and Lake Erie (Figure 23.1). The St. Clair River forms the outlet from Lake Huron and flows south about 64 km where it forms an extensive delta containing numerous channels and wetlands, known as the Lake St. Clair Flats. Lake St. Clair is heart-shaped, with a maximum natural depth of 6.5 m, a maximum length of 43 km, a maximum width of 40 km, and an area of 1,115 km^2 (Edsall et al. 1988). An 8.3 m deep navigation channel bisects the lake along a northeast-southwest direction between the St. Clair cutoff in the St. Clair River Delta to the head of the Detroit River. The area crosses two ecoregions (Southern Michigan Northern Indiana Till Plain and Huron-Erie Lake Plain [Omernik 1987]).

Land use in the area surrounding the St. Clair system shows distinct contrasts with Ontario's "chemical valley", which occurs along the upper St. Clair River and along the Port Huron shoreline, to the agriculture and wetland land use along the Ontario shore, and the nearly 100% urbanized shoreline of Michigan. Permanent residential homes account for about 30 km of lake and 42 km of river shoreline (BASINS, USEPA version 2). The St. Clair River contributes 98% of the water to Lake St. Clair. Water retention in Lake St. Clair ranges from 2—30 days (mean 9 days) depending on wind and flow conditions (Schwab and Clites 1986).

Raphael (1987) indicated that the St. Clair system was initially settled during prehistoric periods and provided mineral and natural resources, transportation, and food resources for native populations. Wetlands produced wild rice and sweet grass, which was important to Chippewa and Ottawa nations for income (Jones 1935). Cultural alteration of the environment occurred during the mid- to late-1800's when the Swamp Acts of 1850 stimulated wetland alteration. By 1873 the land between Detroit and the Clinton River had been converted into agriculture (Herdendorf et al. 1986) and about half of Harsens Island was diked . By the mid-1970's, most of the Michigan shoreline of Lake St. Clair was urban, but the upper St. Clair River was considered rural to semi-rural. In Ontario, the lower St. Clair River and lake shoreline remained agricultural and the majority of the St. Clair Flats is included within the Walpole Chippewa Nation Reservation.

During 2001, the U.S. Fish and Wildlife Service and Troy University sampled 21 drowned river mouth wetland sites along the St Clair River, Lake St. Clair Flats, and Detroit River. Each 500-m zone incorporated the predominant habitat types within a coastal wetland, ranging from small, shallow, sand shorelines with no cover to extensive vegetated cover areas with variable depths. Samples were collected during the summer and fall (from early June until late September) when the river and lake are at stable with low to moderate flows.

Physical habitat data were collected from each 500-m zone. Depth and substrate composition and visual estimates of in-channel area containing emergent, submergent, and floating vegetation; placement in the extensive Delta wetland; riparian land use and occurrence and proximity of riparian human disturbances (e.g., roads, buildings, industry, and agriculture); and bank stability were recorded. Water quality data (pH, temperature, dissolved oxygen, conductivity, and turbidity) were measured at a single point in each wetland area sampled.

23.2.2 Collection

Fish were collected using daytime DC boat electrofishing. Electrofishing was conducted on a single shoreline over a linear distance of 500 m or 35 times the wetted width (minimum distance 150 m) using a serpentine travel route within the zone to incorporate all available habitat types (Gammon 1998; Simon and Sanders 1999). Simon and Sanders found that 500 m was sufficient distance to capture representative numbers of species to characterize biological integrity but not biological diversity. Fish were collected at 21 sites in the St. Clair, Lake St. Clair, and Detroit Rivers using a Smith Root (350-V, 8-A) electrofishing unit deployed in a 4.2 m johnboat. Amperage was maintained by varying pulse widths according to individual site conditions. We varied the pulse width to obtain 6-A output for at least 1800 s. Because boat electrofishing was most effective when deployed within 15 m of shoreline (i.e., at depths less than 2 m), sampling was conducted only under stable, low-flow conditions at a stage level within 1 m of normal water depths and when visibility was at least 0.3 m. Every attempt was made to capture all fish observed using 4.7 mm mesh dipnets. Captured fish were placed into an onboard, aerated live well for later processing. The capture of any young-of-the-year individuals less than 25 mm TL was not included in the results. At the completion of the reach, fish were identified to species, counted, and inspected for deformities, eroded fins, lesions, and tumor (DELT) anomalies (Sanders et al. 1999). All fish were released except for small species (e.g., minnows, darters, and madtoms), which were retained for laboratory identification using regional fish references (Smith 1979; Trautman 1981; Becker 1983).

23.2.3 METRIC DEVELOPMENT

The St. Clair system lacks reference sites representative of pristine conditions and remaining coastal wetlands have been permanently altered (i.e., hydrologic and channel modifications associated with riparian corridors). Metric scoring was conducted on a dataset of 21 drowned river mouth coastal wetland sites. Wetland sites were randomly chosen by US Environmental Protection Agency so that equal numbers of small, medium, and large wetlands were sampled (Simon et al., in press). Reconnaissance and sampling of each wetland was based on the following criteria: 1) they had remnants of wetland function, including wetland vegetation; 2) they contained water depths sufficient to provide permanent habitat for fish assemblages, and 3) they had typical habitat conditions representative of the area. We eliminated three sites with sources of disturbance in the electrofishing zone (e.g., boating activity, docks or mooring sites, navigation traffic wash area, and artificial structures such as piped or other metal debris in the water) that were used as test data to evaluate the modified IBI.

All species collected were classified into various taxonomic, tolerance, feeding, and reproductive guilds (Simon et al., chapter 12) using regional references (Trautman 1981; Smith 1979; Becker 1983; Simon 1999b) and consultation with professional ichthyologists and fisheries biologists. We evaluated an index developed for Lake Erie and determined that it could not be used to assess Lake St. Clair drowned river mouth coastal wetlands, since maximum expectations were below achievable conditions already present in Lake St. Clair. This would have inflated the quality of the St. Clair system. We developed a set of 54 candidate metrics incorporating the original metrics described by Karr (1981), modifications suggested by Miller et al. (1988), Simon and Lyons

(1995), Goldstein and Simon (1999), Simon (1999b), Thoma (1999), and Hughes and Oberdorff (1999) and new metrics developed specifically for this study (including various combinations of species that were designated in various guilds). Metrics chosen for the St. Clair system IBI focus on six areas of fish assemblage structure and function: species richness, pollution tolerance, breeding habits, feeding habits, fish health, and abundance. The metrics were chosen to reflect biological and habitat integrity, trophic complexity, and future restoration and recovery efforts.

Candidate metrics were evaluated for scoring range, variability, responsiveness, and redundancy following Hughes et al. (1998), McCormick et al. (2001), Emery et al. (2003), and Simon et al. (chapter 20). Metrics were rejected if they failed a range test (i.e., raw values were between 0 and 2 species or were otherwise too small to provide a range of response to disturbance).

23.2.4 Statistics

We used Spearman correlations and scatter plots to test the responsiveness of the remaining candidate metrics compared to physical habitat structure and water quality. Metrics with significant correlations ($r > 0.15$; $P < 0.001$) that reflected the predicted responses to physical habitat and water quality variables were retained (Hughes et al. 1998). Redundancy among metrics was tested and we rejected one metric of any pair with a high Pearson's correlation ($r > 0.75$). We retained the metric that was more representative of the St. Clair system fish assemblage than of other systems (Fig. 23.2). We tested the response of the St. Clair system IBI using a plot of least-impacted (C) and test coastal (AI) wetlands (Fig. 23.3).

We performed linear regressions of the species richness metrics on wetland stream width, which we used as a surrogate for watershed area. We did not observe any difference in expectation with wetted wetland width. However, to account for known historical changes in fish assemblage structure, we used the maximum value for observed species richness (interpreted as the y-intecept) for the maximum observed line (MOL) for scoring species richness metrics instead of the 95th percentile (Fausch et al. 1984). The MOL was drawn through the data and parallel to the regression line. The area below the MOL was evenly trisected into regions providing scores of 1, 3, or 5 (Emery et al. 2003; Simon et al., chapter 20).

We excluded schooling species that could affect the responsiveness of percent metrics (Thoma 1999), such as gizzard shad and emerald shiner, which can occur unpredictably and in large numbers (Simon and Emery 1995; Simon and Sanders 1999; Simon et al., chapter 20). These species were excluded from percentile metric calculations; however, both species are included in species richness metrics. Each percent metric was scored following the methods described by Fausch et al. (1984), so that data for each metric was plotted and a line drawn at the 95th percentile; the area beneath the line was then trisected into regions representing scores of 1, 3, and 5. In cases where fewer than 10 individuals were collected (after removing gizzard shad and emerald shiners, tolerant species, nonindigenous species, and hybrids), all proportional metrics were scored as 1 (Yoder and Rankin 1995). In the event that no individuals in a particular metric category were collected, the metric was scored as 0.

23.3 RESULTS AND DISCUSSION

We selected 13 metrics, each of which was significantly correlated ($P < 0.0001$, $r > 0.2$) with one or more habitat or chemical variables, and from these we calculated the St. Clair system IBI (Table 23.2). We rejected 11 metrics because they failed our range test, 20 metrics because they were redundant with other metrics, and 10 metrics because they were not responsive to anthropogenic disturbance (Table 23.1).

23.3.2 METRIC DESCRIPTIONS OF DROWNED RIVER MOUTH COASTAL WETLANDS IN THE ST. CLAIR SYSTEM

Percent individuals as lake-habitat species was an added species richness metric. It focuses on native lake species diversity (Simon and Lyons 1995; Hughes and Oberdorff 1999) by focusing on species expected to be present in lentic habitat conditions. Lack of these species indicates a decline in biological integrity. Changes in riparian habitats, which constrain floodplain systems in urban areas, and the loss of Great Lake species result in a depauperate fauna. The percent individuals as lake habitat species was greater at sites with clean sand (Spearman correlation = 0.22, $P > 0.0001$) and submerged aquatic vegetation (Spearman correlation = 0.52, $P > 0.0001$) and with good water clarity (Spearman correlation = 0.44, $P > 0.0001$), cooler temperatures (Spearman correlation = -0.38, $P > 0.0001$) and more available cover (Spearman correlation = 0.44, $P > 0.0001$). Lake habitat species declined with degraded water quality (Spearman correlation = -0.42, $P > 0.0001$) and at wetland sites with excessive fines or clay (Spearman correlation = -0.34, $P > 0.0001$), highly embedded substrates (Spearman correlation = -0.27, $P > 0.0001$), and lacking aquatic macrophytes (Spearman correlation = -0.46, $P > 0.0001$).

The number of benthic insectivore species was modified from Karr's (1981) metric (the number of darter species). Darters (family Percidae) are not a dominant component of Great Lake fish assemblages; however are important indicators of high quality systems. The darter metric was replaced with the number of benthic insectivores, a niche equivalent metric, to provide the same rationale as Karr's (1981) original concept. The benthic insectivore metric includes darters (family Percidae), round-bodied suckers (genera *Moxostoma, Minytrema, Erimyzon*), madtoms and bullheads (genera *Noturus* and *Ameiurus*), and several benthic minnow species, such as longnose dace (*Rhinichthys cataractae*) and blacknose dace (*Rhinichthys atratulus*)(Simon et al., chapter 12). The number of benthic insectivore species metric did not change significantly with wetland width. The metric increased at deeper sites (Spearman correlation = 0.21, $P > 0.0001$) with coarse substrates (Spearman correlation = 0.33, $P > 0.0001$) and habitat complexity (Spearman correlation = 0.28, $P > 0.0001$). Benthic insectivore species richness declined with increased turbidity (Spearmann correlation = -0.22, $P > 0.0001$), and water temperature (Spearmann correlation = -0.34, P > 0.0001). This metric should decline with the loss of biological integrity.

The number of sensitive species distinguishes areas of highest quality. Species that are especially sensitive to anthropogenic stressors are the first to be eliminated and the last to return to a site once the stressor is removed. This metric differs from the intolerant species metric by including those species defined as highly intolerant and moderately intolerant (Thoma 1999). The species included in the sensitive list includes only species that are highly sensitive to habitat disturbance, toxins, and thermal and nutrient stressors. Species that are sensitive to only one type of stressor , e.g. low dissolved oxygen, are not included (Simon et al., chapter 12). Sensitivity metrics are based on criteria established by Thoma (1999) for Lake Erie. The number of sensitive species decreased significantly with degraded water quality (Spearman correlation = -0.34, $P > 0.0001$) (Simon and Emery 1995) and at sites with increased sand (Spearman correlation = -0.21, $P > 0.0001$), fines (Spearman correlation = -0.34, $P > 0.0001$), and highly embedded substrates (Spearman correlation = -0.39, $P > 0.0001$). This metric reflected the highest levels of biological integrity and is expected to increase with improved water and habitat quality. We used the Lake Erie list so that when Lake Erie recovers to Lake St. Clair levels, the two indices could be united.

Percent individuals as tolerant species represent the worst conditions in the Great Lakes. Tolerant species represent species that increase in abundance with the loss of biological integrity. The percent individuals as tolerant species increased with degraded water quality (increased turbidity [Spearman correlation = 0.34, $P > 0.0001$] and low dissolved oxygen [Spearman correlation = 0.47, $P > 0.0001$]). We expect the percent individuals as tolerant species to increase with increased disturbance. We used the Lake Erie designation of species (Thoma 1999) to calibrate this metric. Restoration goals in Lake Erie will enable this metric to plot progress in attaining restoration expectations.

23.3.3 INDEX SCORING AND RESPONSIVENESS

We generated scoring criteria for each of the 13 metrics (Table 23.2). Metrics were not significantly correlated with stream width; however, significant differences were observed between test impaired sites (AI) and remaining sites (C) for eight metrics and IBI score (Table 23.3; Fig. 23.3). Although we did not include the percent individuals as nonindigenous species in this St. Clair system IBI, we did additional analysis in the event that Lake Erie recovers to Lake St. Clair levels. The percent individuals as nonindigenous species metric showed a significant difference between impaired (IA) and remaining (C) coastal wetlands. Several metrics were not able to show a difference between impaired and remaining coastal wetland conditions including, percent individuals as lake habitat species, number of sensitive species, percent individuals as tolerant species, percent individuals as insectivores and carnivores (Table 23.3). These metrics were responding to differences in environmental qualities in specific areas. We expected some skewness in the lake habitat, sensitive, and tolerance metrics since none of our coastal wetlands in the St. Clair system were pristine. These metrics tend to show the widespread degradation in the system and should collectively be an important indicator when improvements are observed. The nonsignificant result in the abundance of insectivores and carnivores is due to the majority of sites showing excellent numbers of insectivores, but few sites with excellent percentages of carnivores (Fig. 23.2 I, J). The sum of the scores of the 13 metrics resulted in St. Clair system IBI scores that ranged from 33 to 47 (mean $\pm$ SD, 39.7 $\pm$ 4.6). The potential range is 0—65. The IBI score was able to distinguish between sites with anthropogenic disturbance and remaining wetland sites (Fig. 23.3). The mean IBI scores showed a pattern of scores corresponding with sand and fine substrates (Spearman correlation = 0.32, $p < 0.05$) and highest IBI scores were at sites with submergent vegetation (Spearman correlation = 0.32, $p > 0.0001$), clear water (Spearman correlation = 0.22, $p > 0.0001$), cool temperatures (Spearman correlation = 0.27, $p > 0.0001$), and complex habitat cover (Spearman correlation = 0.36, $p > 0.0001$). We were able to identify fish assemblage variables that were strongly correlated with degraded substrate quality and water quality variables that reflected anthropogenic disturbance. In our analyses, the strongest correlations between metrics and environmental variables were between those measures that described water clarity, submerged vegetation, and substrate quality.

The lack of reference sites representing minimally disturbed conditions has affected our choice of metrics and the calibration process. The homogenization of habitat and water quality in the Great Lakes coastal wetlands has caused a loss of biological diversity (Schlosser 1991; Karr et al. 1985) that will be difficult to restore. The introduction of alien species (Jude and Pappas 1992; Mills et al. 1993; MacInnis and Corkum 2000) and loss of wetlands and habitat fragmentation (Edsall et al. 1988; Herdendorf et al. 1986) have imperiled the aquatic assemblage of the St. Clair system. However, despite pervasive and persistent toxic contaminants and urban disturbance throughout the system, we were able to identify least-impacted sites that had little evidence of poor water quality or degraded habitat. Coastal wetlands in the Detroit River including Humbug Marsh and the Grosse Isle Wetlands showed subtle attributes of remaining reference condition. By constructing a model of biological integrity for the St. Clair system, we pooled coastal wetlands fish assemblage characters from a variety of sites showing least-impacted assemblage attributes still remain among sites. No single site could be considered a reference site, but cumulatively we believe that biological integrity remains in the St. Clair system.

We developed fish assemblage metrics that represent the diversity, structure and function of native fish assemblages, and provided restoration endpoints for fish assemblage conditions. These select metrics will influence management decisions related to toxics control, control of nonindigenous species, and restoration activities (UGLCC

1988). The results of this research describes an approach for determining least-impacted conditions and provides fish assemblage metrics that will be useful in establishing reference conditions in the St. Clair system of the Great Lakes coastal wetlands.

23.5 CONCLUSIONS

An index was developed to assess the condition of fish assemblages from 21 sites based in the St. Clair River, Lake St. Clair, and Detroit Rivers. Representative samples of fish assemblages were sampled during 2001 using standardized daytime electrofishing techniques. Fifty-four candidate metrics were evaluated based on attributes of fish assemblage structure and function to develop a multimetric index of health. We examined spatial (by stream width) variability of these metrics and assessed their responsiveness to anthropogenic disturbances, specifically effluents, turbidity, and highly embedded substrates. The resulting St. Clair system IBI is comprised of 13 metrics selected for their predictable response to anthropogenic disturbance or reflection of desirable features of a restored Great Lakes coastal wetland. All of Karr's original index of biotic integrity metrics were modified. Four metrics (the number of native species; number of centrarchid species; number of sensitive species, percent individuals with deformities, eroded fins, lesions, and tumors) were modified from metrics originally designed by Karr. Four metrics were designed to replace original Karr metrics (number of minnow species, percent lake habitat species, percent individuals as tolerant species, and number of benthic invertivore species) so that similar rationale would be retained in the index. Three trophic metrics were incorporated into the index (percent individuals as detritivores, insectivores, and carnivores), one metric based on catch per unit effort, and one metric based on reproductive mode (percent individuals as phytophilous spawning fish species). The St. Clair system IBI declined significantly where anthropogenic effects on substrate and water quality were prevalent. Additional research on the temporal stability of the index will enhance the reliability of the IBI, its use will be a significant improvement over current physiochemical protocols.

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Figure Captions

23.1 Drowned river mouth coastal wetlands associated with the St. Clair River, Lake St. Clair Flats, and Detroit River.

23.2. Metric expectations and scoring relationships for thirteen metrics used to assess biological integrity of drowned river mouth coastal wetlands of Lake Michigan. A. Number of species, B. Number of centrarchid species, C. Percent individuals as obligate Great Lakes species, D. Number of lake habitat species, E. Percent individuals as intolerant species (LE), F. Percent individuals as tolerant species (LE), G. Percent individuals as detritivores, H. Percent individuals as insectivore species, I. Percent individuals as carnivores, J. Number of individuals, K. Percent individuals as phytophils, L. Percent individuals with DELT anomalies, and M. Percent individuals as exotic and non-native species.

23.3 Validation of an Index of Biotic integrity for the St. Clair system showing relationships between test “impaired” (AI) and remaining (C) drowned river mouth coastal wetlands.

Fig. 23.1

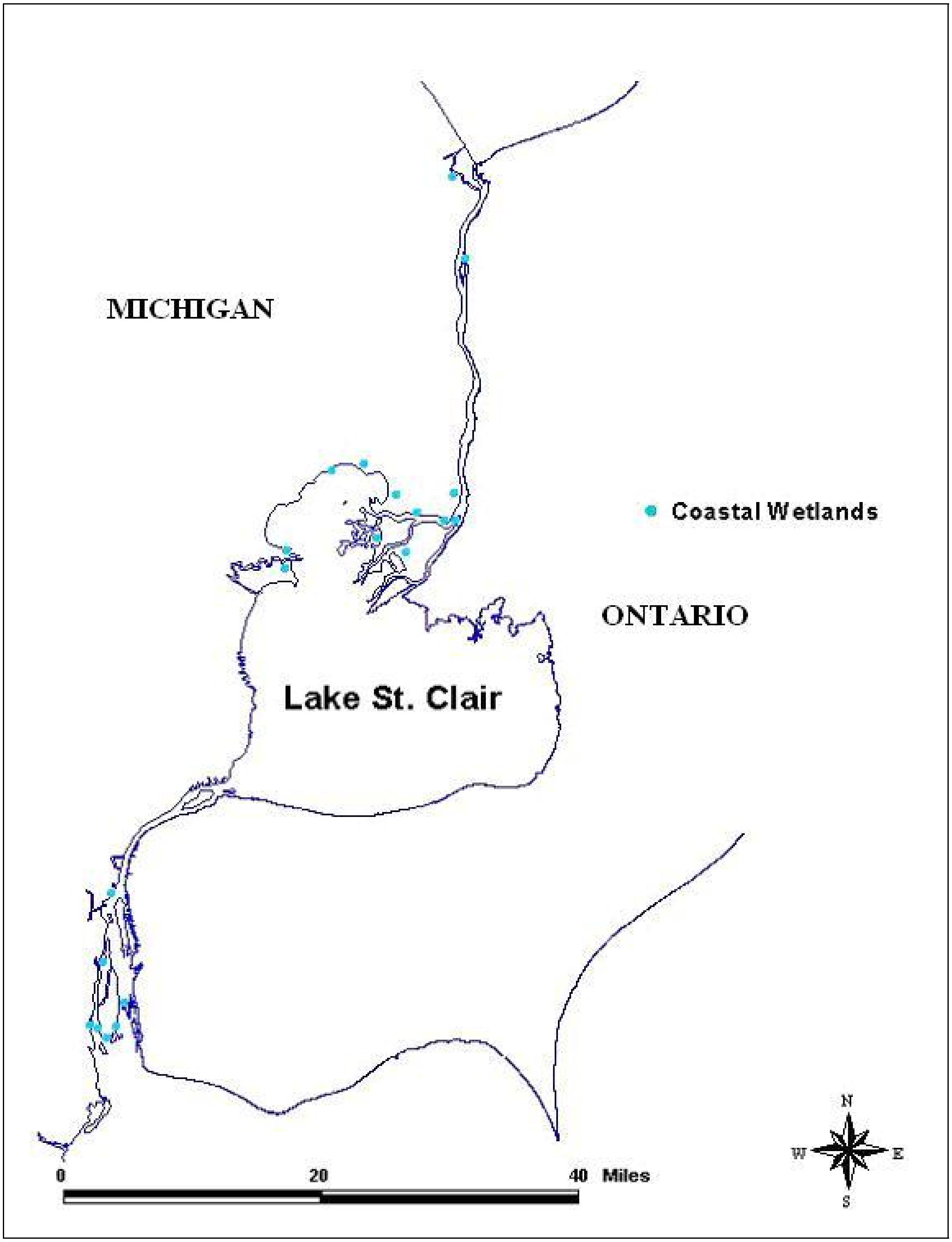

MICHIGAN
Coastal Wetlands
ONTARIO
Lake St. Clair
0
20
40 Miles
N
W
E
S

Fig. 23.2

Fig. 23.3

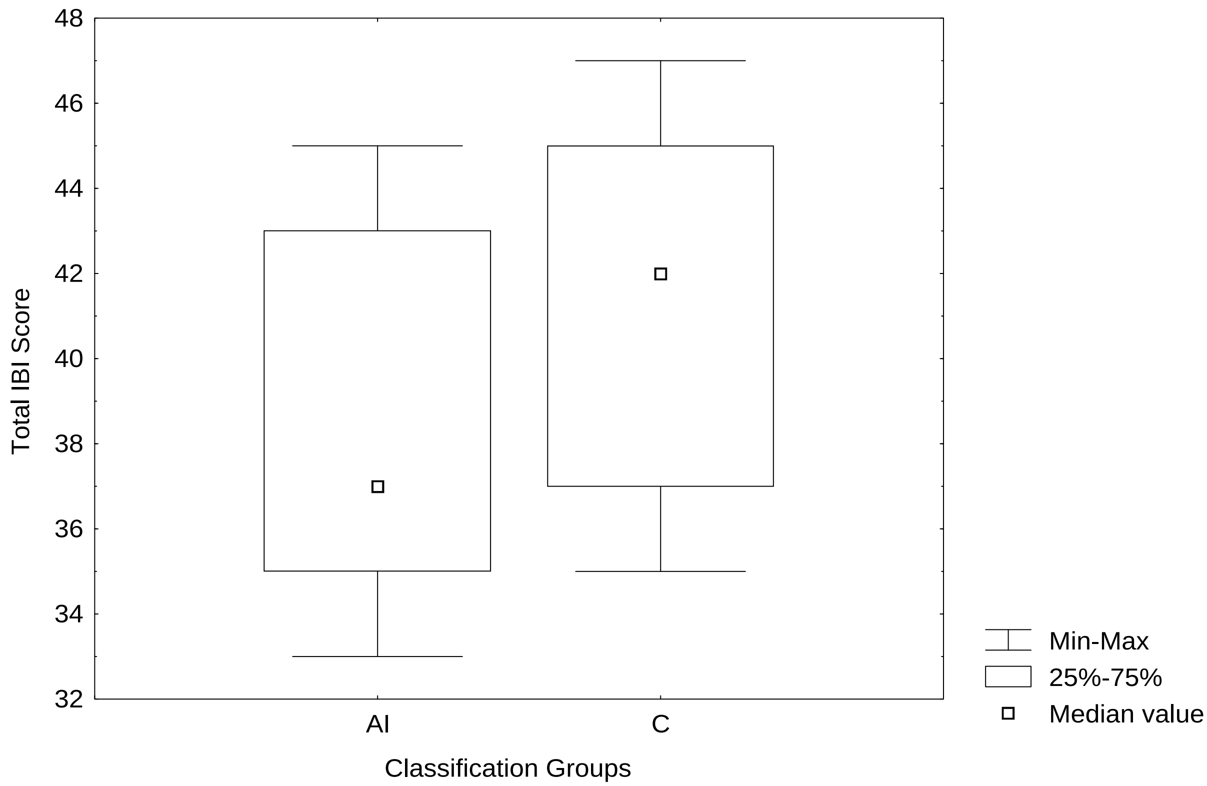

48
46
44
42
40
38
36
34
32
Total IBI Score
AI
C
Classification Groups
Min-Max
25%-75%
Median value

TABLE 23.1

METRICS REJECTED IN THE EVALUATION PROCESS, BY REASON FOR REJECTION. LISTS 1 AND 2 COMPRISE GROUPS OF SPECIES CREATED FOR TEST PURPOSES; SEE TEXT FOR DESCRIPTION OF OTHER SPECIES GROUPS.

Failed Range Test	Failed redundancy test	Failed responsiveness test
Number darter species	Number of sunfish species	Catch per unit effort (list 1)
Number darters, madtoms, sculpin	Catch per unit effort (list 2)	Percent top piscivores
Number salmonid species	Number intolerant species (ind; list 1)	Number tolerant species (list 2)
Percent salmonid species (ind.)	Number of intolerant species (ind.; list 2)	Percent tolerant species (list 1)
Number sucker species	Number tolerant species (ind.)	Percent tolerant species (list 1; biomass)
Percent great-river species (biomass)	Percent round-bodied suckers (ind.)	Percent Great Lakes species
Percent hybrids (ind.)	Number deep-bodied suckers species	Number Obligate Great Lakes species
Number hybrids	Percent deep-bodied suckers (ind.)	Percent round-bodied suckers (biomass)
Percent sensitive species (ind.)	Percent green sunfish (ind.)	Percent sucker biomass
Number of DELT anomalies	Percent benthic species (ind.)	Percent pioneer species (ind.)
Number round-bodied suckers	Percent omnivores (biomass; list O)	
	Percent omnivores (biomass; list O)	
	Percent omnivores (ind.; list 1)	
	Percent omnivores (ind.; list 2)	
	Number catfish and sucker species	
	Number of piscivores (list 1)	
	Number of piscivores (list 2)	
	CPUE	
	Number of planktivores	
	Percent planktivores (ind.)	

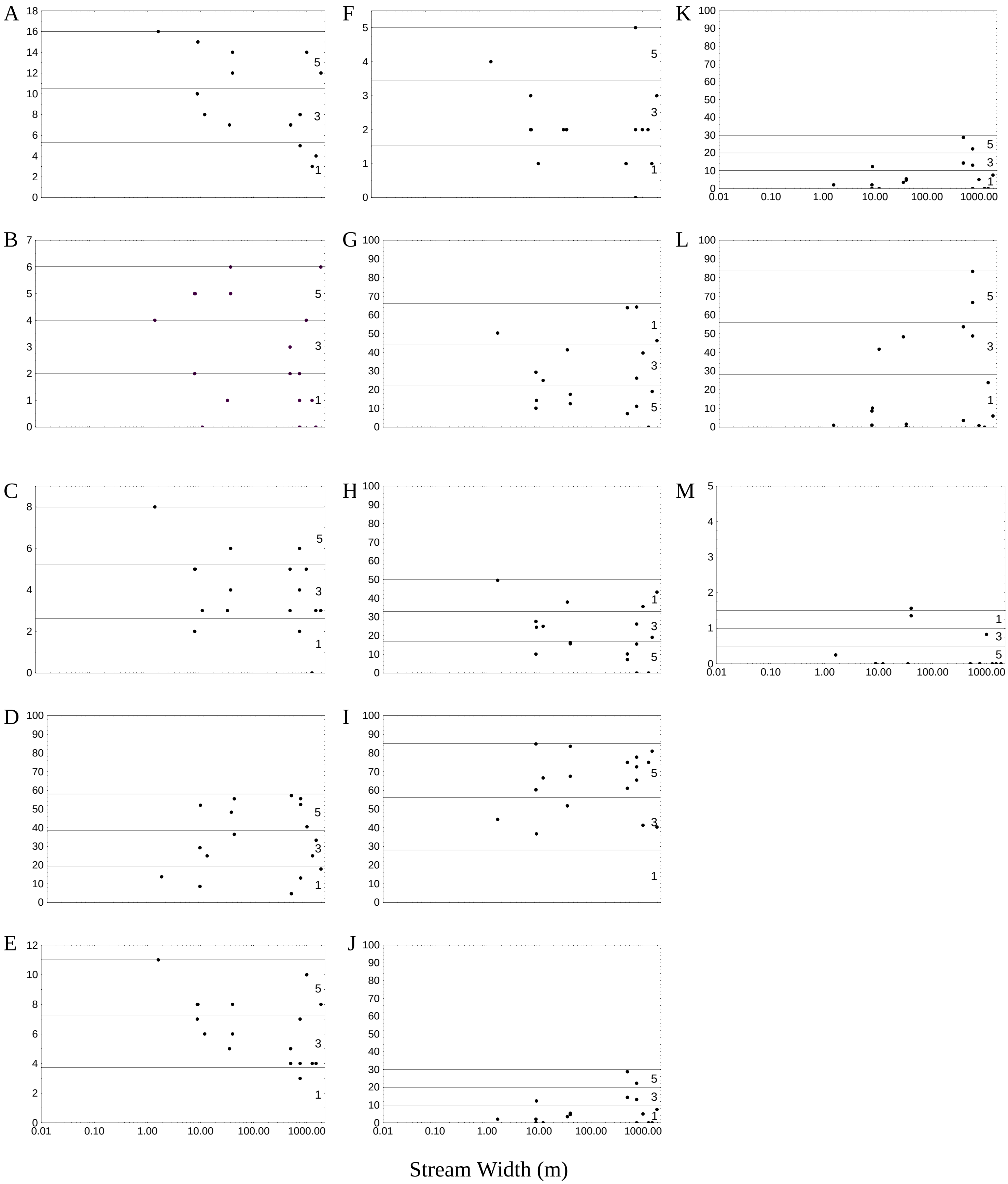
TABLE 23.2
Metrics and scoring criteria for Great Lake coastal wetland index of biotic integrity for drowned river mouth coastal wetlands of the St. Clair River Estuary, Lake St. Clair, and Detroit River.

	Expectations		
Species Richness and Composition	1	3	5
Number of native species	≤ 5	6-10	≥11 (Fig. 23.2A)
Number of centrarchid species	≤2	3-4	≥5 (Fig. 23.2B)
Number of minnow species	≤2	3-5	≥6 (Fig. 23.2C
Percent individuals as lake habitat species	≤20%	>20-40%	>40% (Fig.23.2D)
Number of benthic invertivore species	≤3	4-7	≥ 8 (Fig. 23.2E)
Tolerance and Sensitivity			
Number of sensitive species (LE)	≤1	2-3	≥4 (Fig. 23.2F)
Percent individuals as tolerant species (LE)	>44%	>22-44%	<22% (Fig. 23.2G)
Trophic guilds			
Percent individuals as detritivores,	>36%	18-36%	<18% (Fig. 23.2H)
Percent individuals as insectivore species	<28	28-59%	>59% (Fig. 23.2I)
Percent individuals as carnivores	<10%	10-20%	>20% (Fig. 23.2J)
Abundance, condition, reproduction, and naturalness			
Number of individuals	≤10	10-20	>20 (Fig. 23.2K)
Percent individuals as phytophils	≤29%	>29-58%	≥58 (Fig. 23.2L)
Percent individuals with DELT anomalies	> 1.0%	>0.5-1.0%	≤0.5%(Fig. 23.2M)

TABLE 23.3

Descriptive statistics of index of biotic integrity metrics for the St. Clair River Estuary, Lake St. Clair, and Detroit River drowned river mouth coastal wetlands and significance between impaired test sites and remaining wetlands ($p = 0.10$).

Attribute	t-value	df	p
Number of species minus exotic and non-native species	-6.57562	15	.000009
Number of centrarchid species	-6.14640	15	.000019
Percent individuals as lake habitat species	0.354077	15	.728209
Number of minnow species	-1.77518	15	.096157
Number of benthic invertivores	-5.25852	15	.000096
Number of sensitive species	-1.48522	15	.158197
Percent individuals as tolerant species (Lake Erie)	0.117603	15	.907942
Percent individuals as detritivores	-1.86936	15	.081222
Percent individuals as insectivores	1.673384	15	.114972
Percent individuals as carnivores	1.035312	15	.316927
Percent individuals as phytophils	3.785217	15	.001797
Number of individuals	-2.20774	15	.043251
Number of individuals minus exotic and non-native species	-2.10161	15	.052895
Percent individuals with DELT anomalies	-2.28618	15	.037204


A
B
C
D
E
F
G
H
I
J
K
L
M
0.01
0.10
1.00
10.00
100.00
1000.00
5
3
1
Stream Width (m)

Lake St Clair

A. Number of species minus exotic and non-native species
B. Number of centrarchid species
C. Number of minnow species
D. Percent individuals as lake habitat species
E. Number of benthic invertivore species
F. Number of sensitive species
G. Percent individuals as tolerant species (Lake Erie)
H. Percent individuals as detritivore species
I. Percent individuals as insectivore species
J. Percent individuals as carnivore species
K. Number of individuals – exotic and non-native species
L. Percent individuals as phytophil species
M. Percent individuals with DELT anomalies

24 Development, Modification, and Validation of an Index of Biotic Integrity for Fish Assemblages in Open Lake Coastal Wetlands of Lake Ontario

Douglas Carlson, Thomas P. Simon, Ronda L. Dufour

CONTENTS

24.1 INTRODUCTION

The Lake Ontario ecosystem has experienced substantial changes as a result of exotic invasion (Zaranko et al., 1997; Lozano et al., 2001; Benoit et al., 2002), erosion and water quality degradation (Chow-Fraser et al., 1998), urban pollution (Li and McAteer, 2000; McMaster, 2001), toxic chemical contamination (Brueggeman and Halfon, 1997; Munawar et al., 1999), and the increase in chemical, physical, and biological impacts to habitat (Busch and Lary, 1996). This has had a dramatic effect on remaining coastal wetlands in Lake Ontario embayments and nursery habitats for fishes (Goodyear et al., 1982; Crabtree and Ringler, 2001).

The loss of biological diversity in Lake Ontario has shown that 14 fish species have been extirpated from the lake (Cudmore-Vokey and Crossman, 2000). As a result, large-scale ecosystem management objectives for fish community (Stewart et al., 1999) and persistent toxic chemicals (Thompson et al., 1999) have been developed. These actions included the reduction in stocking rates of top-predator salmonines (Brandt et al., 1996) and drafting of the Great Lakes Water Quality Agreement and revisions, which have reduced anthropogenic disturbances (Czapla et al., 1995).

The index of biotic integrity (IBI) was developed for assessing fish assemblages of small streams (Karr, 1981; Karr et al., 1986). We modified the stream IBI for open lake coastal wetlands of the Lake Ontario portion of the Great Lakes on the United States shoreline in order to assess the status and condition of remaining coastal wetlands. Our project was primarily conducted in drowned river mouth wetlands (Keough et al., 1999), however, this modification is for an index in lentic waters in the open lake wetlands.

24.1.1 Overview of IBI applications for Lake Ontario

Intensive long-term databases that value the contributions of all species are rare for most of the Great Lakes, however, a database for Lake Ontario is more robust for open lake and off-shore habitats (Casselman et al., 1999). Several indicators have been used and developed to increase biological indices for assessing Lake Ontario wetlands. Two indicators include fish and macroinvertebrates, however, for the purpose of this paper we are limiting our attention to fish assemblages.

Minns et al. (1994) developed a modified index to evaluate the Hamilton Harbour Area of Concern in Lake Ontario. The index was designed to evaluate a grossly polluted harbor area, which had some high quality coastal wetland habitat remaining. Minns et al. (1994) used fewer than twelve metrics in that application and followed a dose-response application in their calibration. Minns et al. (1994) did not follow Karr et al.'s (1986) rationale for metric selection and replacement. However, the metrics seem to be adequate for establishing differences in condition between the target and reference condition of the Hamilton Harbor Area.

24.2 METHODS

24.2.1 Study area, site selection, and sample design

In order to develop an IBI, with similar rationale to that developed for other regions of the world (Karr, 1981; Karr et al. 1986; Simon, 1998; Simon and Stewart, 1998; Hughes and Oberdorf, 1999), open lake (embayment) wetlands were examined along the United States shoreline of Lake Ontario (Fig. 24.1). The southern shore of Lake Ontario is a large series of wetland complexes that extends from Niagara Falls to the St. Lawrence

River. We used a variety of indicators and land use information from the Geographic Information System (GIS) for evaluating patterns in our IBI results. We concluded that our sites are known to represent a wide range of habitat quality from "good" to "very poor" quality open lake coastal wetlands.

Sample areas were targeted and selected to provide the most diverse fish collections in the large bays. We used a rapid assessment approach so that our objective was to obtain a representative sample of the fish species present. Seine sites had aquatic plants of moderate density and minimal deposits of soft silt. A typical sampling effort within a bay included one visit with 1-2 seine hauls at about five sites. Also, eight of the bays did have second visits with more seining, and 14 of the areas (Table 24.1) also had electrofishing for about 15-45 minutes at each area.

24.2.2 COLLECTION

Daytime seine nets and electrofishing gear were used during 1993-1998 to catch fish in weed bays extending from the Niagara River in the west, through Lake Ontario and to the east where the Grasse River flows into the St. Lawrence River (Figure 24.1). The bag seines included one 15.2 x 1.8 m with 6 mm mesh (throughout), one 15.2 x 1.8 m with 4.5 mm mesh (3 mm mesh bag) and one 7.6 x 1.2 m with 6 mm mesh (3 mm mesh bag). There were no obvious differences in the ability of these seines to catch adult fish, but some areas were better suited to the different net lengths. Electrofishing boats included a 18 ft and a 12 ft boat, both equipped with a Smith Root variable voltage control box providing DC current at about 6 amps. In some shallow areas, a backpack electrofishing unit (designed by the NYSDEC Electronics Unit) was also used. Electrofishing and seines were used to captured fish at 633 sites in 1993-2002.

Fish were netted, placed into a live well until the completion of the zone. All fish were then sorted and counted, with some individuals set aside as vouchers. All vouchers are stored at the New York State Museum at Albany. Common and scientific names of fishes conform with Robins et al. (1991).

24.2.3 METRIC DEVELOPMENT

To develop an IBI for open lake coastal wetlands of Lake Ontario, we reviewed fish assemblage structure and function literature, published life history, and tolerance information (Simon et al., Chapter 12). We evaluated more than 20 characteristics of fish communities in selecting the 12 metrics among the five main categories that were incorporated into multimetric indices for open lake coastal wetlands of Lake Ontario (Table 24.1). When selecting a scalar for the x-axis of our metric plots, we evaluated the use of drainage area, surface area, and distance along the Lake Ontario shoreline from a discrete point. We chose to use the distance from the mouth of the Niagara River as our discrete point since this landmark would facilitate use when our Canadian colleagues calibrate the index for the northern shore (Fig. 24.1).

Structural metrics incorporated community structure, key indicator species, and compositional group membership attributes. Functional metrics included sensitivity and tolerance metrics, percent individuals based on different trophic ecology, macrohabitat specialists, and reproductive guilds. Relative abundance was based on the number of fish collected within a given sampling zone based on the collection protocol. Scoring criteria for this calibration follows Karr et al. (1986), which uses three levels based on a trisection of the data. For a metric to score a "5" the attribute needs to be representative of the reference condition, a score of "3" shows deviation from the reference condition, and a score of "1" suggests the metric is significantly different from the reference condition.

In order to validate this modification of the IBI, we chose to compare our rating scores to varying measures of environmental perturbation. We calculated the IBI score using data from 486 samples collected between 1993-2000 and validated the index with data from 144 samples collected between 2001-02. We evaluated 60 metrics for suitability and eliminated many based on a range test, colinearity, skewness, and statistical correlations to a measure of disturbance (Hughes et al., 1998). We judiciously kept the same rationale as Karr et al. (1986) when substituting metrics, which resulted in the 12 metrics that were chosen for this application.

Unfortunately there were not chemical standards that enabled the measure of disturbance to be quantified, as described by Smogor and Angermeier (1999). Instead, standards of quality for validating the IBI were considered from:1) a subset of bays demonstrating minimum and maximum degradation based on water quality monitoring (Kishbaugh, Div. Water, Monroe Co.), and 2) a comparison of Lake Ontario bays where we calculated the percentages of the bay areas with wetland cover types and calculated the respective basins' percentages with landuse/cover types and density of roads using a Geographic Information System.

24.2.4 Statistics

Patterns in species composition, group membership, and functional percentages were scaled against Lake Mile from the mouth of the Niagara River to determine if a linear relationship existed. Scoring lines were drawn to trisect the data such that the maximum observed line (MOL) included a trisection of the data beneath the highest observed point (Emery et al., 2003). We chose the MOL approach rather than the Maximum Species Richness line approach since we believe that few high quality embayments remain in Lake Ontario. Thus, we wanted to ensure that we did not overestimate the quality of any of the coastal wetlands by rating them too high. Metric hypotheses were made *a priori* and qualitatively examined to determine if the patterns found fit these expectations based on a range test.

Spearman correlation ($p < 0.05$) were used to examine the relationship between wetland qualities among a "best remaining" group of wetlands and an "impacted" set of wetlands (Conover, 1971).

24.3 RESULTS

24.3.1 Species Composition

The fish communities of Lake Ontario open lake coastal wetlands were only partially sampled, but catches of 66 total species including 59 native taxa. Nineteen taxa occurred at fewer than 0.6% of the sites (Table 3), and four of these, redfin shiner *Lythrurus umbratilis*, black redhorse *Moxostoma duquesnei*, western pirate perch *Aphredoderus sayanus*, and eastern sand darter *Ammocrypta pellucida* were rare species; classified as imperiled in New York. Only the lake chubsucker *Erimyzon sucetta*, was missing compared to historical information from 70 years earlier. Lake chubsucker has been classified as extirpated from New York. It was encouraging that several rare species were still caught in substantial numbers at a few areas. Among the ubiquitous species in these shallow water areas were seven taxa that occurred at 33% of the sites, bluntnose minnow *Pimephales notatus*, golden shiner *Notemigonus crysoleucas*, banded killifish *Fundulus diaphanus*, rock bass *Ambloplites rupestris*, pumpkinseed *Lepomis gibbosus*, largemouth bass *Micropterus salmoides*, and yellow perch *Perca flavescens*.

24.3.2 STRUCTURAL ATTRIBUTES OF OPEN LAKE COASTAL WETLANDS

The structural attributes of open lake coastal wetlands of Lake Ontario showed that the number of species is one of the most widely used diversity indices. We hypothesized that the number of species would increase with biological integrity. The number of species ranged from one at several embayments to 19 species at a single embayment (Table 24.2). None of the twelve metrics showed a relationship with lake mile distance from the Niagara River (Fig. 24.2). We substituted the remaining compositional metrics including the number of centrarchid species (replacement for number of sunfish species), percent individuals as obligate Great Lakes species (replacement for number of darter species), and number of lake habitat species (replacement for number of sucker species). Centrarchid species are important components of fish assemblages in open lake wetlands within Lake Ontario. We expected to find a greater number of centrarchid species with high quality wetlands. The number of centrarchid species ranged between zero and 5 centrarchid species at a site.

The percent individuals as obligate Great Lakes species was substituted for the number of darter species since we anticipated finding increasing numbers of obligate

Great Lakes species with recovery of the Great Lakes system (Fig. 24.2B). This metric is currently serving as a “reality check” within our IBI, since we expect to find keystone Great Lakes species, however, due to the largescale degradation basin-wide this metric is underattaining for most of the wetlands we surveyed with the exception of the embayments found in the central basin of Lake Ontario (Fig. 24.2C). The range in percentage of obligate Great Lakes species was between zero and 90 percent.

The number of lake habitat species is a group of fish that are consistently found to be common in lakes (Simon et al., Chapter 12). This guild is expected to increase with increasing biological integrity. Lake Ontario embayments ranged between zero and 6 species (Fig. 24.2D).

24.3.3 SPECIES TOLERANCE AND SENSITIVITY

Regional descriptions of sensitivity were completed by Halliwell et al. (1999), which classified Northeastern fishes into broad categories of sensivity. Halliwell et al. (1999) classified 41 of 125 (32.8%) species as intolerant, while 29 (23.2%) species were considered tolerant. Karr et al. (1986) warned against classifying too many species as intolerant so that this metric can serve as an early warning to declining conditions. Karr et al. (1986) recommended that less than 10% of the fauna be considered sensitive. Despite the higher percentage of intolerant species in this study, we recognize that the distribution of these species in the Lake Ontario ecosystem will cause the classification for any portion of the lake to be closer to Karr et al.’s recommended number. We hypothesized that intolerant species will increase with biological integrity. Our results showed that open lake coastal wetlands in Lake Ontario ranged between zero and 98 percent. The embayments in the central basin showed the highest percentages of intolerant species (Fig. 24.2E).

Abundance of tolerant species is an indicator of degraded conditions and is inversely correlated with biological integrity. Halliwell et al. (1999) classified 29 species as tolerant to environmental disturbance. We did not anticipate that there should be any relationship with lake mile for this metric (Fig. 24.2F), since tolerant species should be no more abundant at any given location within Lake Ontario. Thus, our results show that the range of tolerant species is equally distributed among the entire lake.

24.3.4 FUNCTIONAL ATTRIBUTES OF FISH ASSEMBLAGES

We evaluated several trophic guild categories for their ability to explain the biological integrity of open lake coastal wetlands. We followed Goldstein and Simon (1999) in the assignment of trophic guilds to Lake Ontario species. In addition, we hypothesized that the percent individuals as insectivores and carnivores would increase with biological integrity, while percent individuals as detritivores would decrease with biological integrity. The percent individuals as detritivores replaced the percent individuals as omnivores metric. The definition of omnivores was too broad and caused confusion among biologists in the assignment of species. This caused species such as gizzard shad *Dorosoma cepedianum*, to be combined with carp *Cyprinus carpio*. Since this metric is inversely scored so that higher percent individuals indicate degradation, our results showed that ranges were between zero and 90 percent (Fig. 24.2G).

Neither the percent individuals as insectivore or carnivore metrics showed a relationship with lake mile (Fig. 24.2H and 24.2I, respectively). We modified the scoring criteria for percent individuals as carnivores following Simon and Dufour (1998). We recognized that inflated percent individuals as carnivores is an unstable situation, which cannot be sustained for long periods of time. This inverted pyramid structure would cause the collapse of a community if these high percent individuals of carnivores are found. Thus, we recognized that higher percent individuals as carnivores occur above what is seen in streams, but is similar to what is found in Great Rivers (Emery et al., 2003). By dividing the percent area into six equal parts we recognize that both low and extremely high numbers of carnivores is not optimal for sustaining coastal wetland fish assemblages (Fig. 24.2I).

24.3.5 Individual health, condition, and abundance

We did not change Karr et al.'s (1986) application of the relative abundance of individual fish collected at a site (Fig. 24.2J), and retained the percent individuals with deformities, eroded fins, lesions, and tumor (DELT) anomalies (Sanders et al., 1999) following Karr's original criteria. However, we did adopt a substitute metric for percent individuals as hybrids. We chose to replace this metric with the percent individuals as phytophilous spawning species since we observed that high quality open lake coastal wetland habitat had a variety of submergent, emergent, and floating vascular plant species. We followed Simon (1999) in the placement of species into this guild assignment (Simon et al., Chapt. 12). Our expectation was that percent individuals as phytophilous spawning species would increase with biological integrity. Our results showed that the percent individuals as phytophilous spawning species ranged from zero to 100 percent (Fig. 24.2K). Lastly, we include a metric that evaluates the percent individuals that are non-native or nonindigenous species. We expected that the percent individuals as non-indigenous or exotic species would increase with degraded conditions, thus the metric is inversely scored (Fig. 24.2L).

24.4 DISCUSSION

24.4.1 INDEX VALIDATION

We used two approaches for validating an index of biotic integrity for Open Lake coastal wetlands of Lake Ontario. Wetlands were selected that represented the extremes of conditions and we evaluated these "least impacted" and "impaired" wetland fish assemblages using our newly calibrated index. We evaluated patterns in the IBI scores among these two groups of wetlands in order to determine the sensitivity of the modified index to different perturbations.

In addition, we evaluated statistical relationships between each metric and the distance from the mouth of the Niagara River. By evaluating patterns between metrics and distance we effectively evaluated differences attributed to large scale land use, ecoregions, and tributary influences (Table 24.2). We did not see a significant relationship for any of the metrics with distance from the Niagara River.

We also evaluated landuse patterns for every bay and compiled information based on historical and present information to evaluate a trend assessment of this data (see Chapt. 25, Carlson et al.). Carlson et al. (Chapt 25) used data from 144 samples collected between 2001-2002 to validate the newly modified index.

24.5 CONCLUSIONS

Fish community assessments based on a modified Index of Biotic Integrity (IBI) for open lake coastal wetlands enabled us to compare the environmental degradation between bays of different regions of New York's Lake Ontario. Seining and electrofishing catches at 486 sites were collected between 1996-2000 provided a ranges of values between "good" and "very poor" wetlands. We evaluated over 60 metrics using a range test, skewness, colinearity, and correlation analysis to select 12 metrics. These metrics were validated using another 144 additional samples collected between 2001-2002. In addition, standards of quality came from a subset of bays that demonstrated minimum and maximum anthropogenic degradation.

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Figure Captions

1 Distribution of Lake Ontario open lake coastal wetlands that were sampled during this study between 1996-2002.

2. Metric expectations and scoring relationships for thirteen metrics used to assess biological integrity of Open Lake coastal wetlands of Lake Ontario. A. Number of species, B. Number of centrarchid species, C. Percent individuals as obligate Great Lakes species, D. Number of lake habitat species, E. Percent individuals as intolerant species (NE), F. Percent individuals as tolerant species (NE), G. Percent individuals as detritivores, H. Percent individuals as insectivore species, I. Percent individuals as carnivores, J. Number of individuals, K. Percent individuals as phytophils, L. Percent individuals with DELT anomalies, and M. Percent individuals as exotic and non-native species.

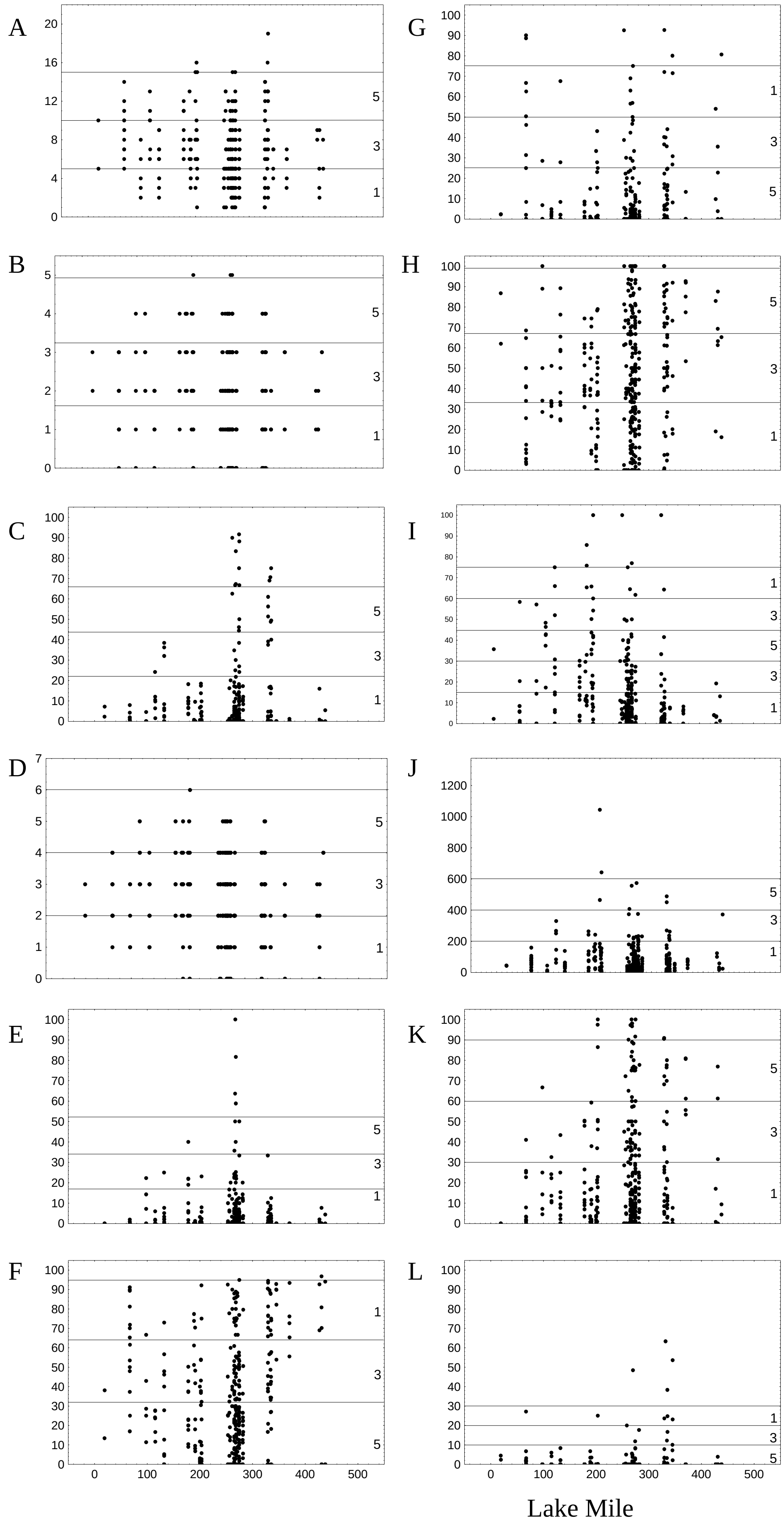
3 Depiction of Index of Biotic integrity relationships between “least-impacted” and “impaired” Open Lake coastal wetlands along Lake Ontario.

Fig 1.

TABLE 22.2

Statistical relationships between index of biotic integrity metrics for Open Lake coastal wetlands of Lake Ontario and distance from the mouth of the Niagara River.

Attributes	Mean	SD	Range	r (p-value)
Number of species	6.60	3.34	1-19	-0.13 (0.018)
Number of Centrarchid species	1.93	1.18	0-5	-0.12 (0.024)
Percent individuals as obligate great lakes species	8.08	16.90	0-91.7	0.02 (0.652)
Number of lake habitat species	2.48	1.25	0-6	-0.14 (0.004)
Percent individuals as intolerant species (NE)	4.67	11.18	0-100	-0.02 (0.678)
Percent individuals as tolerant species (NE)	35.39	28.16	0-96.8	0.13 (0.020)
Percent individuals as detritivores	9.27	18.54	0-92.6	0.02 (0.677)
Percent individuals as insectivore species	45.52	27.90	0-100	0.13 (0.018)
Percent individuals as carnivores	14.14	19.02	0-100	-0.29 (0.000)
Number of individuals	81.12	109.79	0-1043	-0.11 (0.044)
Percent individuals as phytophils	23.74	26.52	0-100	0.09 (0.094)
Percent individuals with DELT anomalies	Not Measured			
Percent individuals as exotic and non-native species	1.68	6.73	0-63.3	0.02 (0.652)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Lake Mile

Lake Ontario

A. Number of species
B. Number of centrarchid species
C. Percent individuals as obligate Great Lakes species
D. Number of lake habitat species
E. Percent individuals as intolerant species (NE)
F. Percent individuals as tolerant species (NE)
G. Percent individuals as detritivores
H. Percent individuals as insectivore species
I. Percent individuals as carnivores
J. Number of individuals
K. Percent individuals as phytophils
L. Percent individuals with DELT anomalies
M. Percent individuals as exotic and non-native species