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HYDROLOGIC VARIABILITY AND THE APPLICATION OF INDEX OF BIOTIC INTEGRITY METRICS TO WETLANDS: A GREAT LAKES EVALUATION

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Abstract: Interest by land-management and regulatory agencies in using biological indicators to detect wetland degradation, coupled with ongoing use of this approach to assess water quality in streams, led to the desire to develop and evaluate an Index of Biotic Integrity (IBI) for wetlands that could be used to categorize the level of degradation. We undertook this challenge with data from coastal wetlands of the Great Lakes, which have been degraded by a variety of human disturbances. We studied six barrier beach wetlands in western Lake Superior, six drowned-river-mouth wetlands along the eastern shore of Lake Michigan, and six open shoreline wetlands in Saginaw Bay of Lake Huron. Plant, fish, and invertebrate communities were sampled in each wetland. The resulting data were assessed in various forms against gradients of human disturbance to identify potential metrics that could be used in IBI development. Our results suggested that the metrics proposed as potential components of an IBI for barrier beach wetlands of Lake Superior held promise. The metrics for Lake Michigan drowned-river-mouth wetlands were inconsistent in identifying gradients of disturbance; those for Lake Huron open embayment wetlands were yet more inconsistent. Despite the potential displayed by the Lake Superior results within the year sampled, we concluded that an IBI for use in Great Lakes wetlands would not be valid unless separate scoring ranges were derived for each of several sequences of water-level histories. Variability in lake levels from year to year can produce variability in data and affect the reproducibility of data collected, primarily due to extreme changes in plant communities and the faunal habitat they provide. Substantially different results could be obtained in the same wetland in different years as a result of the response to lake-level change, with no change in the level of human disturbance. Additional problems included limited numbers of comparable sites, potential lack of undisturbed reference sites, and variable effects of different disturbance types. We also evaluated our conclusions with respect to hydrologic variability and other major natural disturbances affecting wetlands in other regions. We concluded that after segregation of wetland types by geographic, geomorphic, and hydrologic features, a functional IBI may be possible for wetlands with relatively stable hydrology. However, an IBI for wetlands with unpredictable yet recurring influences of climate-induced, long-term high water periods, droughts, or drought-related fires or weather-related catastrophic floods or high winds (hurricanes) would also require differing scales of measurement for years that differ in the length of time since the last major natural disturbance. A site-specific, detailed ecological analysis of biological indicators may indeed be of value in determining the quality or status of wetlands, but we recommend that IBI scores not be used unless the scoring ranges are calibrated for the specific hydrologic history pre-dating any sampling year.

Key Words: biological indicators, fish, Great Lakes, human disturbance, hydrologic variability, Index of Biotic Integrity (IBI), invertebrates, Lake Michigan, Lake Superior, plants, water-level fluctuations, wetlands

INTRODUCTION

Methods for assessing the quality of wetlands have received much attention in recent years. Developed specifically for wetlands, the functional assessment approach was popularized by the Wetland Evaluation Technique (WET) (Adamus 1983, Adamus et al. 1987). More recently, the Hydrogeomorphic approach (HGM) was introduced, which classifies wetlands by type and incorporates physical and biological sampling of reference sites to assess their hydrologic, biogeochemical, plant community maintenance, and faunal community habitat maintenance functions (Brinson 1993, Brinson and Rheinhardt 1996). Alternatively, biological assessment techniques in which biological organisms are used as indicators of environmental health or stress have been developed for a variety of aquatic environments (e.g., Karr 1981, Plafkin et al. 1989, Rankin 1989, Adamus and Brandt 1990, Rosenberg and Resh 1993, Kramer 1994, Lovett Doust et al. 1994, Butterworth et al. 1995, Davis and Simon 1995), and some are presently being adapted for use in wetlands. Adaptation has not been straightforward, however, because wetland environments can differ from other aquatic environments, both in the response to hydrologic changes and in the importance of plant communities as faunal habitat. Therefore, any such attempted adaptation must be tested successfully before being implemented.

The biological assessment approach receiving the most attention focuses on biological integrity, which was defined by Karr and Dudley (1981) as the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region. Biological integrity is assessed using the Index of Biological Integrity (IBI), a method developed by Karr (1981) in which fish communities were used to assess water quality in streams of the midwestern United States (Karr 1981, 1991, Karr et al. 1986). The method has since been extended to other aquatic ecosystems elsewhere (e.g., Steedman 1988, Minns et al. 1994, Deegan et al. 1997). A fish IBI is developed by sampling fish communities in reference streams and additional degraded streams that span a gradient of human disturbance, as determined by some external measure of water quality or human influence (e.g., extent of logging, agriculture, or impervious surfaces in the watershed). Potential metrics, or measurements of fish-community attributes, are then tested against the disturbance gradient using dose-response curves. Metrics that demonstrate a clear response to increasing disturbance are selected for incorporation into the IBI. Scores of 1, 3, or 5 are assigned to ranges of values for each metric (e.g., *Proportion of Individuals as Piscivores*: $<1\% = 1$, $1–5\% = 3$, $>5\% = 5$), and the sums of scores for all metrics are used to categorize stream quality as very poor, poor, fair, good, or excellent. Finally, additional gradients of disturbance or additional sites not included in IBI development are used to test and validate the IBI (Karr et al. 1986).

The apparent success of the IBI approach using fish in lotic environments created an interest among land-management and regulatory agencies in the Great Lakes region and elsewhere for developing a wetland IBI using fish, invertebrate, plant, or other biotic communities (Keough and Griffin 1994, Minns et al. 1994, Bertram and Stadler-Salt 1999, Danielson 1999). Such a system could be used to identify degraded wetlands, compare degraded wetlands with natural wetlands, and potentially characterize specific problems associated with sources of degradation, thus allowing wetland degradation to be recognized in early stages when mitigation can prevent severe impacts. Development of criteria and standards by which to evaluate the presence/absence or extent of degradation not only would serve in prevention and mitigation but would greatly assist inventory and restoration activities.

We collected data that could be used to evaluate the potential for a Great Lakes wetland IBI prior to the more recent call for IBI development. In a study initiated in 1993 by the U.S. Environmental Protection Agency, we sampled plant, fish, and invertebrate communities in selected wetlands of lakes Superior, Michigan, and Huron for the purpose of assessing methods to evaluate biological integrity. These wetlands have been degraded by disturbance from a variety of human activities, including regulation of lake levels, ditching, residential and industrial development, marina development, shoreline protection, road and bridge construction, landfills, and contamination by biological and chemical waste discharges (Maynard and Wilcox 1997). This paper does not develop nor present a wetland IBI for implementation; rather, it reports on the evaluation of our data to address several objectives that seem critical in wetland IBI development: 1) determine if this approach is valid for Great Lakes wetlands; 2) explore the limitations that constrain this approach caused by the effects of natural disturbance, variability in disturbance types among wetlands, restricted numbers of comparable wetlands, and potential lack of pristine reference sites; 3) propose alternative approaches, and 4) extend our findings to other wetland types.

STUDY AREAS

Wetlands of the Great Lakes occur in a number of geomorphic settings that provide at least some protection from wave attack (Maynard and Wilcox 1997), including barrier beach embayments, drowned river mouths, open embayments, protected embayments, and shallow sloping beaches. All are affected greatly by water-level changes that occur at varying magnitudes, frequencies, timing, and duration. The effects on plant communities and faunal habitat differ in response to fluctuations at hourly, seasonal, annual, and various multiple-year frequencies (Wilcox 1995, Maynard and Wilcox 1997, Keough et al. 1999). However, the longer-term, multiple-year fluctuation patterns (33 years in lakes Michigan and Huron; Baedke and Thompson 2000) create the greatest change and drive the maintenance of wetland diversity. High lake levels periodically eliminate canopy-dominating emergent plants and invading upland and woody species; under the right conditions, aquatic communities may also expand (Figure 1). When water levels recede, less competitive species are able to grow from dormant seeds and propagules, complete at least one life cycle, and replenish the seed bank before being replaced through competitive interactions. Extreme low water levels often expose large areas of mudflat and significantly reduce the extent of open water and aquatic communities; this also may result in a large expansion of emergent plant communities and more invasion of upland and woody species (Figure 1). Habitat for wetland fauna, especially invertebrates and fish that require standing water and make use of the various physical attributes of wetland plants, undergoes substantial changes between years as the plant communities change in response to water level (McDonald 1955, Harris et al. 1981, Farney and Bookhout 1982, Keddy and Reznicek 1986, Wilcox 1995, Maynard and Wilcox 1997, Keough et al. 1999).

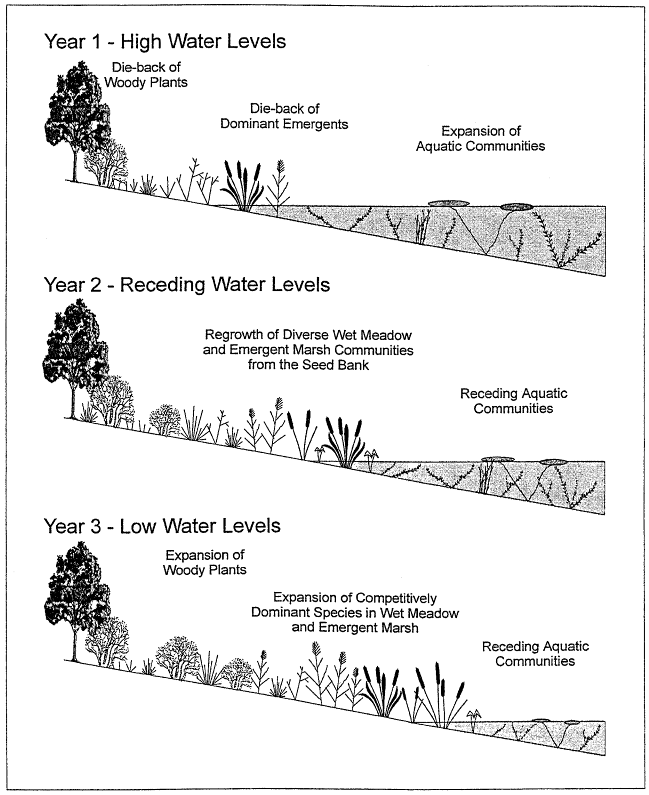

Figure 1. Simplified diagram of the effects of water-level fluctuations on coastal wetland plant communities of the Great Lakes (from Maynard and Wilcox 1997).

Six wetland study sites subject to the above conditions were selected in each of lakes Superior, Michigan, and Huron. The sites were selected based on geomorphic features (Brinson 1993, Brinson and Rheinhardt 1996), rather than biological characteristics, to ensure that evaluation of biological indicators was not biased by preconceived expectations. Site descriptions provided below were prepared following field data collection. Geographical differences between lakes, differences in water-level histories between Lake Superior and lakes Michigan/Huron (one lake hydrologically), and differences in geomorphic setting of wetland types in each lake dictated that biological attributes be evaluated separately for each wetland type/lake in attempts to develop metrics. These differences also severely limited the number of comparable sites that could be studied and precluded replication. Designation of the least-disturbed wetland among the Lake Michigan sites as a pristine reference site was tentative, and all of our Lake Huron sites proved to be disturbed, a fact later corroborated by Burton et al. (1999) and Stanley (2000). Since useful data were not

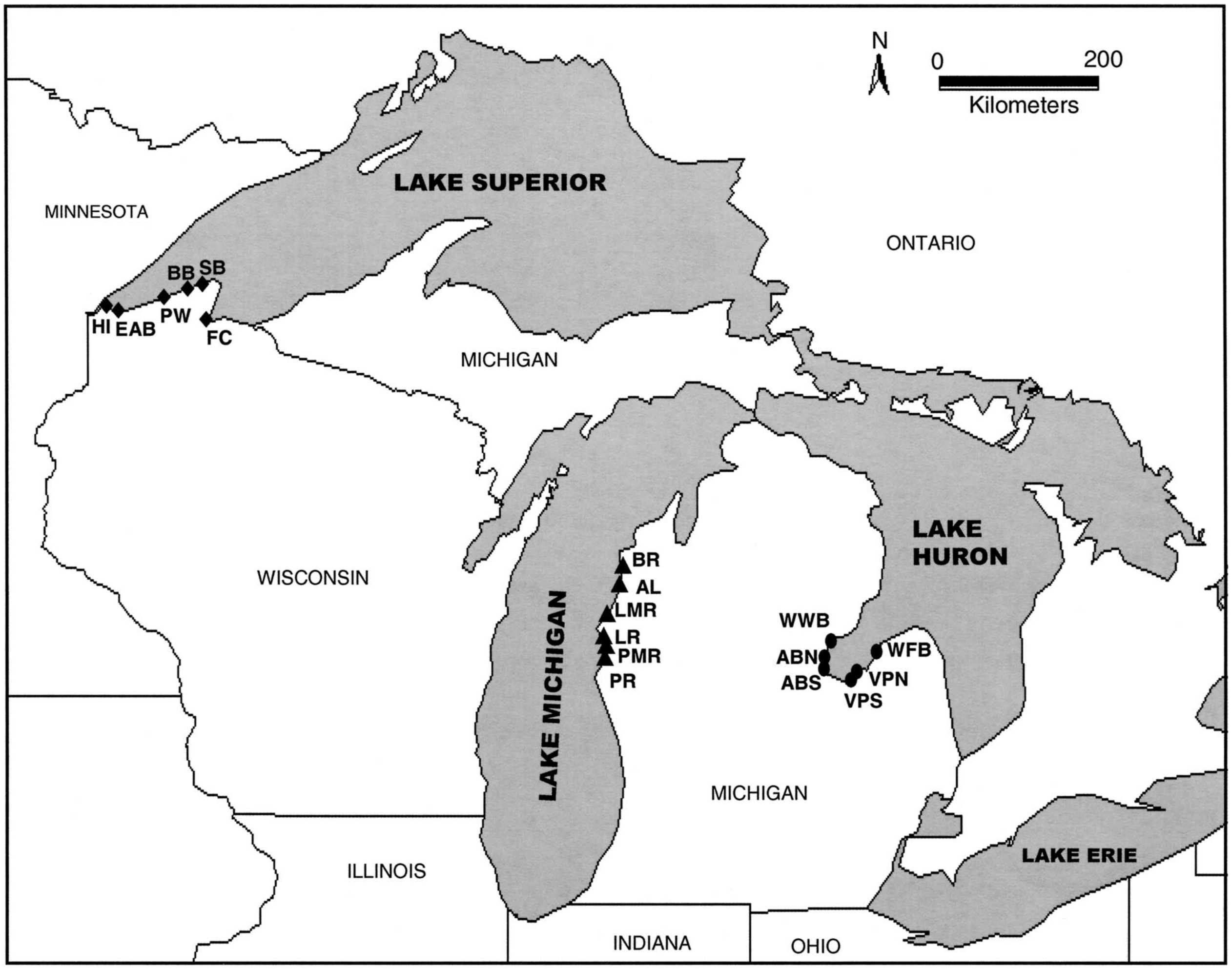

Figure 2. Map of wetland study sites in lakes Superior, Michigan, and Huron used to evaluate the potential for development of biological indicators of human disturbance. Lake Superior sites are Hog Island (HI), East Allouez Bay (EAB), Port Wing (PW), Bark Bay (BB), Siskiwit Bay (SB), and Fish Creek (FC). Lake Michigan sites are Pentwater River (PR), Pere Marquette River (PMR), Lincoln River (LR), Little Manistee River (LMR), Arcadia Lake (AL), and Betsie River (BR). Lake Huron sites are Wigwam Bay (WWB), Alameda Beach North (ABN), Alameda Beach South (ABS), Vanderbilt Park South (VPS), Vanderbilt Park North (VPN), and Wild Fowl Bay (WFB).

forthcoming, we do not present further information on the Lake Huron sites.

Lake Superior

Numerous embayments along the south shore of western Lake Superior in Wisconsin (Figure 2) are protected from wave attack by barrier beaches along their lakeward margins. Openings in the barriers connect the wetlands to the lake, and wetland hydrology is thus controlled by Lake Superior water-level changes. Inflowing streams connect the wetlands to larger watersheds. Much of the region is forested, with some agricultural land use. However, the city of Superior and Superior-Duluth Harbor border the western end of the study area, and the city of Ashland borders the eastern end. Six barrier beach wetlands were selected from a pool of 11 potential sites. Restricted access through private property removed four of the potential sites from the pool. From the remaining seven sites, six were selected that had varying degrees of apparent anthropogenic disturbance and watershed characteristics. Descriptions of these sites are presented in Table 1 in order of increasing percent forested watershed.

Lake Michigan

Numerous rivers discharge into Lake Michigan along its eastern shore in the state of Michigan (Figure 2). Isostatic rebound at the outlet of the lake (Lake Huron outlet at Port Huron, Michigan) has caused most of the river mouths to become flooded, forming

Table 1. Description of barrier-beach-protected wetlands in Lake Superior and drowned-river-mouth wetlands in Lake Michigan used as study sites, including area, percent forested watershed, wetland sediments types (decomposed peat, sand, silt, gravel), and potential sources of degradation.

Site	Area (ha)	% Forested	Sediment	Potential Source of Degradation
LAKE SUPERIOR				
Hog Island (HI)	32	20	d. peat, silt	urban, industrial, industrial landfill, pier, railroad bed, on-site oil refinery wastes, 1 stream
Fish Creek (FC)	244	55	silt sand in streams	agriculture, urban, timber harvest, residential, former fish farm, turbidity, 3 roads, bridges at 2 outlets, 2 streams
East Allouez Bay (EAB)	113	60	d. peat, silt	residential, turbidity, 1 stream
Port Wing (PW)	24	80	d. peat sand in stream	agriculture, residential, sewage treatment plant, marina, campground, 2 roads, 1 stream
Bark Bay (BB)	149	80	d. peat, sand & silt in streams	min. agriculture, scattered houses, boat launch, 1 road, 2 streams
Siskiwit Bay (SB)	56	85	d. peat sand in creek	scattered houses, 3 roads, 1 internal creek
LAKE MICHIGAN				
Lincoln River (LR)	30	30	d. peat, sand, silt, gravel	residential, agriculture, mute swans, 0.1 km road crossing, 1 bridge, 1 river
Pentwater River (PR)	51	40	d. peat, sand, gravel, silt	agriculture, 1986 dam break 6 km upstream on s. branch, mute swans, 0.6 km road crossing, 1 bridge, 1 river (2 branches)
Arcadia Lake (AL)	170	50	d. peat, sand, silt	agriculture, cattle grazing, mute swans, 6 ditches, turbidity, 1.0 km road crossing, 2 culverts, 3 streams
Pere Marquette River (PMR)	204	60	sand, gravel, d. peat	agriculture, residential, industry, industrial landfill, 3 upriver towns, 0.6 km road crossing, 2 bridges, 1 river (2 branches)
Betsie River (BR)	162	70	sand, gravel, d. peat	residential, agriculture, 15 ditches, mute swans, 0.4 km road crossing, 0.4 km railroad crossing, 2 bridges, 1 river
Little Manistee River (LMR)	33	95	sand, d. peat	industry, residential, industrial landfill, large ditch and berm, 1 river

small lakes with wetlands in their upper reaches; they are described as drowned river mouths. Road crossings separate most of the wetlands from the adjoining small lake, and downriver water flow can be restricted under relatively narrow bridges during peak flow periods. However, wetland hydrology is largely determined by Lake Michigan water levels, including seiches that can cause short-term flow reversals. The upstream limits of each wetland were placed where a gradient in water-surface elevation began and wetland water levels were no longer determined by Lake Michigan. Despite the lake-controlled hydrology, ditches are present in some of the wetlands as a result of failed attempts to drain them. Rivers in the middle third of the east shore have watersheds containing agricultural, forested, municipal, industrial, and residential land uses. Six drowned-river-mouth wetlands were selected from the pool of seven potential sites in this region; the remaining site (Big Manistee River) was not selected because it is much larger than the other rivers and not readily comparable to them. The type and amount of disturbance varied among the selected study sites, which are described in Table 1 in order of increasing percent forested watershed.

METHODS

Plant, fish, and invertebrate communities in wetlands at the Lake Superior sites were sampled from 20 July to 20 August 1993. Lake Michigan sites were sampled from 10 July to 18 August 1995. Sampling in 1994 was conducted at the Lake Huron sites.

Vegetation Sampling

Recent existing or new aerial photographs (Lake Superior: 1988 color infrared, 1:24000 enlarged to 1:6000 or 1:12000; Lake Michigan: 1987 black and white infrared, 1:15840 enlarged to 1:7920) were used to map major vegetation types in each wetland, with groundtruthing to verify vegetation types and to identify changes in boundaries since the date of the photography. The area of each vegetation type in each wetland was determined by planimetry. The plant communities were characterized by sampling randomly placed 1-m × 1-m quadrats in each vegetation type. The number of sampled quadrats differed between lakes and vegetation types based on relative size and on variability of the plant communities. At Lake Superior sites, 15 quadrats were sampled in submersed/floating and the most prominent emergent vegetation types, and 10 quadrats were sampled in the other vegetation types. At Lake Michigan sites, 30 quadrats were sampled in the most prominent emergent vegetation type, 20 quadrats in submersed/floating vegetation, and 10 quadrats in other vegetation types. All taxa present were identified to the lowest taxonomic level possible, and percent cover within each quadrat was estimated visually. Nomenclature for all sites follows Voss (1972, 1985, 1996).

Fish Sampling

Fish communities were sampled using two sets of fyke nets (eight total) that were placed in the morning and retrieved the following morning. At each wetland, the nets were fished for two consecutive days then moved to different locations for two additional days. One set consisted of two 91-cm × 91-cm-frame and two 45-cm × 45-cm-frame nets, with both 0.48-cm and 1.27-cm standard knotted mesh. The large frame nets were placed facing the shore in water 1 m deep or greater with 6- to 15-m leads perpendicular to and reaching shore and 3-m wings extending to each side. The small frame nets were placed similarly in water less than 1 m deep. Sets at all wetlands in a given lake were placed in locations of similar geomorphic structure. Sampling was conducted in similar plant communities if present at those locations, but sampling was not biased by seeking similar plant communities in different geomorphic settings. After collection, all fish were anesthetized with MS-222, identified, counted, measured for length, and released. Nomenclature for all sites follows Robins et al. (1991).

Invertebrate Sampling

To reduce effort in sorting specimens from sediments and detritus, invertebrate communities were sampled using funnel traps (Swanson 1978) in daily sets. Paired clear plastic funnels attached to collection vessels were mounted in vertical and horizontal positions from rods anchored in the sediments. Two pairs of traps were set, moved, and sampled in association with each of the two fyke net sets. Generally, they were placed in submersed aquatic beds, short emergent marsh, and tall emergent marsh vegetation types in standing water. After a 24-h period, invertebrates were removed, placed in labeled jars with preservative, and returned to the laboratory for identification to genus and enumeration. Nomenclature for all sites follows Brooks (1959), Wilson and Yeatman (1959), Balcer et al. (1984), Pennak (1989), Thorp and Covich (1991), Merritt and Cummins (1996), or Hudson et al. (1998), depending on Class of organisms.

In addition, adult caddisflies (Trichoptera) were sampled using ultraviolet blacklight traps placed in overnight sets at each site for two nights (Armitage et al. 2001). The traps consist of an Eveready 9450 flashlight containing an F6T5-BLB blacklight tube and a small plastic pan partially filled with 85% ethanol. Because of the low luminosity of the bulb and placement of the lights in locations with limited long-distance visibility, the traps draw insects from only a limited area within a wetland and generally will not draw insects from other habitats. Caddisflies from each collection were placed in jars, picked, sorted, and identified to species level. Nomenclature for all sites follows Morse (1993).

Attribute Evaluation, Potential Metric Selection and Testing, and Evaluation of IBI Development

Data for each wetland type (and thus each lake) were analyzed and evaluated separately, resulting in two sets of data with six wetlands in each set. Plant data were evaluated using measures of species richness and composition, community composition, and community health at both the vegetation-type mapping and quadrat-sampling scale. At the mapping scale, percent of wetland in individual mapped vegetation types and sum of percent wetland in invasive vegetation types were calculated. At the quadrat-sampling scale, the following attributes were evaluated: total number of taxa, number of native taxa, and number of invasive taxa in each specific vegetation type; total number of taxa, number of native taxa, and number of invasive taxa across all vegetation types; mean percent cover of various dominant taxa and individual invasive taxa in specific vegetation types; mean percent cover of all submersed aquatic plants and individual turbidity-tolerant taxa in submersed aquatic vegetation types (SAV); sum of mean percent cover of all invasive taxa in specific vegetation types and of all turbidity-tolerant taxa in SAV vegetation types; percent of all taxa that are obligate wetland plants; and Floristic Quality Index (FQI) scores for each wetland based on the list of all taxa identified in each wetland and calculated as

$$\mathrm{FQI} = \bar{\mathrm{C}} * \sqrt{\mathrm{n}}$$

where $\bar{\mathrm{C}}$ is the mean coefficient of conservatism ($\bar{\mathrm{C}} = \Sigma\ \mathrm{C/n}$) and n = total number of plant taxa. The method and values for C are derived from Herman et al. (1996).

In the above calculations, *Typha* sp., *Lythrum salicaria* L., *Phragmites australis* (Cav.) Steudel, *Phalaris arundinacea* L., *Cornus stolonifera* Michaux, *Salix exigua* Nutt., *Salix lucida* Muhl., and *Alnus rugosa* (Duroi) Sprengel were considered invasive taxa, depending on the vegetation type in which they occurred. *Ceratophyllum demersum* L., *Elodea canadensis* Michaux, *Heteranthera dubia* (Jacq.) MacM., *Myriophyllum spicatum* L., *Ranunculus longirostris* Godron, *Najas minor* All., *Potamogeton pectinatus* L., *P. crispus* L., *P. pusillus* L., and *P. foliosus* Raf. were considered turbidity-tolerant taxa (Adamus and Brandt 1990).

Fish data were evaluated using many of the IBI attributes commonly described by others for use in streams (e.g., Karr 1981, 1991, Karr et al. 1986) and nearshore waters (Minns et al. 1994, Deegan et al. 1997). Measures of species richness and composition included number and percent native species, non-native species, sensitive species, tolerant species, centrarchids, and native cyprinids (Scott and Crossman 1973, Trautman 1981, Becker 1983). Measures of trophic composition included number and percent of individuals as omnivores, insectivores, planktivores, and piscivores (Scott and Crossman 1973, Trautman 1981, Becker 1983). Measures of fish abundance included total catch, total native catch, and number and percent non-native species in total catch. Measures of diversity included the Shannon Index (Shannon and Weaver 1949).

Invertebrate data were evaluated using attributes selected from those recommended by others for use in streams and wetlands (e.g., DeShon 1995, Burton et al. 1999). Density measures included number of individuals in each of the following groups: Amphipoda, Chironomidae, Cladocera, Copepoda, Corixidae, Crustacea (non-benthic), Diptera, Ephemeroptera, Gastropoda, Hemiptera, Isopoda, mites, Mollusca, Odonata, Oligochaeta, Ostracoda, Trichoptera, and Turbellaria. Relative abundance measures included percent of individuals in each of the above groups. Diversity measures included adult Trichoptera species richness, Cladocera genera richness, Copepoda genera richness, Evenness, Shannon Index, Shannon Index of Cladocera, Shannon Index of Copepoda, and total taxa richness (not lower than genera). Number of rare or uncommon adult Trichoptera species (based on the literature and the knowledge and experiences of author BJA from collecting adult caddisflies in the Great Lakes Region for over 15 years) was evaluated as a sensitivity measure. Invertebrate collections in funnel traps at individual sites were highly variable; therefore, medians were used rather than means to evaluate attributes and develop potential metrics because the influence of outliers is reduced when determining a central tendency (Burton et al. 1999).

As recommended by Karr et al. (1986), Karr (1991), and USEPA (1998), the above measures were evaluated graphically, using dose-response curves in which the values for a biological attribute are plotted against a gradient of human disturbance generally described by some measure of local or watershed conditions. Because this study included three types of biota that likely respond to different types of disturbance, we attempted to generate dose-response curves that tested attributes against a number of gradients. Gradients of disturbance in watersheds included percent forested, percent in agriculture, percent urban, and combinations of those land uses. Gradients of local conditions included proximity to discharges from known contaminant sources, measured light attenuation, and sediment type. Since fish and invertebrate communities respond to differences in the habitat provided by degraded vs. non-degraded plant communities, we also tested fish and invertebrate attributes against the FQI scores for each wetland, the sum of mean percent cover of all turbidity-tolerant taxa in SAV vegetation types, the percent of wetland in SAV and floating vegetation types, and the sum of all plant IBI metric scores recommended for potential use.

Dose-response curves that demonstrated an obvious response to increasing human disturbance in various forms when displayed graphically were used to select potential metrics, although some metrics were tentatively selected for evaluation despite obvious outlier data if the outliers could be explained readily. We attempted to maintain similarities among lakes in metrics selected for each biological community. However, we sometimes selected a metric for one lake but eliminated it for another where it showed no response (e.g., *Number of Native Cyprinid Species* for Lake Superior but not Lake Michigan sites). In some cases, we fully recognized that individual potential metrics were tentative at best, but we followed through with attempted IBI development to allow evaluation of the results.

From the graphic display of the dose-response curves, scores of 1, 3, or 5 were assigned to various ranges of attribute data, generally according to natural breaks in the data, to reflect poor conditions, moderate conditions, and minimally impacted conditions, respectively (Karr 1981, Karr et al. 1986, USEPA 1998, Karr and Chu 1999). The ranges for scoring the potential metrics were developed separately for each lake. The sums of metric scores for plant, fish, and invertebrate data were tallied separately for each wetland in each lake. The sums for plant, fish, and invertebrate metrics for each wetland were then added to derive a tentative wetland IBI score for comparison with other wetlands of the same type in the same lake.

IBI development requires testing and validation, which may be accomplished by 1) splitting the data, using half for development and half for testing, 2) collecting data from additional sites for testing, or 3) testing against more than one gradient of human disturbance (USEPA 1998). The limited number of comparable wetlands for each wetland type/lake precluded the first two options; we therefore tested metrics against multiple gradients of human disturbance as described previously.

After metrics are developed and tested, the next step is to use them to determine the biological integrity of study sites. Karr et al. (1986) noted that individual metrics are functions of the underlying biotic integrity of the study sites, but biotic integrity is not a function of the metrics. Thus, collective IBI scores, not individual metric scores, were used to compare sites. Because we had collective scores for plant, fish, and invertebrate communities, we chose to rate our sites based on each community individually and also across all three community types. However, the individual community scores may be intercorrelated, so we caution that patterns present in each taxonomic group should be examined individually before they are combined.

Karr et al. (1986) used the sum of individual metric scores to classify sites into six quality classes (excellent, good, fair, poor, very poor, and no fish), each with a specific scoring range. However, their scoring ranges are not continuous and thus allow for intermediate classifications (e.g., fair-poor), as described in Karr (1981). We found that such intermediate classifications were useful in comparing our wetland study sites. We lacked extensive data sets necessary to develop independent scoring ranges specific to each biological community or wetland type in the three lakes. Therefore, the scoring ranges for each integrity class were modified slightly from equivalent ranges used by Karr (1981) and Karr et al. (1986) and were based on the greatest number of points possible, which differed between community types and lakes. These scoring ranges and integrity classes must be recognized as tentative, at best, because this was an IBI metric evaluation exercise, and no confirmed and tested IBI is being proposed.

RESULTS

The extensive data sets covering three community types were organized by lake. We first present the means used to rank wetland sites along a disturbance gradient, then present summary data and attempted metric evaluations for plants, fish, and invertebrates, then attemped IBI scores and wetland ranks for each site followed by the derived wetland biotic integrity class. Finally, we present the results of metric testing.

Lake Superior

The percent of watershed that is forested provided an estimate of disturbance for the barrier beach wetlands in western Lake Superior that was consistent with observed degradation. It ranked the sites from least to most disturbed in the order Siskiwit Bay, Bark Bay, Port Wing, East Allouez Bay, Fish Creek, and Hog Island. Other disturbance gradients that were evaluated to examine characteristics of habitat quality for specific groups of organisms (e.g., plant metric *Sum of Mean Percent Cover of Turbidity-Tolerant Taxa in SAV Vegetation Type* representing a gradient for fish or invertebrate habitat) often overlapped with this gradient but reordered some of the four better sites or the two worst sites. However, we chose to adhere to percent watershed forested as the only gradient used in preparing dose-response curves and obtained results that made ecological sense when outlier data were explained.

Plant Community Data and Potential Metrics. Vegetation in the barrier-beach-protected wetlands is generally composed of a floating sedge mat dominated by *Carex lacustris* Willd. or *C. lasiocarpa* Ehrh.; emergent communities with prominent species such as *Sparganium eurycarpum* Engelm., *Typha angustifolia* L., *Potentilla palustris* (L.) Scop., and *Eleocharis smallii* Britton; shrub communities dominated by species such as *Alnus rugosa* and *Myrica gale* L.; and submersed aquatic and/or floating leaf communities with prominent taxa such as *Ceratophyllum demersum*, *Elodea canadensis*, *Potamogeton* sp., and *Nuphar variegata* Durand.

Two mapping-scale attributes related to abundance and condition were selected for potential metric development, both of which gave scores of 1 to the most disturbed sites, Fish Creek and Hog Island (Table 2). *Percent Wetland in Sedge Vegetation Type* compares the relative area of each wetland that remains in the dominant natural emergent vegetation for these wetlands; it should decrease with disturbance. *Percent Wetland in Invasive Vegetation Types* compares the relative area of vegetation types not naturally present in these wetlands and should increase with disturbance. The only outlier of note in these two metrics was the somewhat low percent sedge vegetation type in Siskiwit Bay, where perimeter areas containing sedges but otherwise dominated by *Chamaedaphne calyculata* (L.) Moench or *Myrica gale* were mapped as native shrub-dominated vegetation type.

Table 2. Data, scores, scoring ranges, and metric scores for plant community attributes selected as potential IBI metrics for use in barrier beach wetlands in western Lake Superior. Sites in order of decreasing percent of forested watershed are Siskiwit Bay (SB), Bark Bay (BB), Port Wing (PW), East Allouez Bay (EAB), Fish Creek (FC), and Hog Island (HI).

Attribute	SB	BB	PW	EAB	FC	HI	Scoring Range	Metric Score
Percent Wetland in Sedge Veg. Type	50	76	74	59	18	28	>65	5
Score	3	5	5	3	1	1	40–65	3
							<40	1
Percent Wetland in Invasive Veg. Types	1	5	2	16	44	35	<10	5
Score	5	5	5	3	1	1	10–30	3
							>30	1
Percent Wetland Obligate Species	91	95	90	95	84	75	>89	5
Score	5	5	5	5	3	1	80–89	3
							<80	1
Floristic Quality Index (FQI)	61.4	59.7	49.4	44.2	27.1	18.5	>55	5
Score	5	5	3	3	1	1	30–55	3
							<30	1
Number Native Taxa	66	66	55	49	49	32	>60	5
Score	5	5	3	3	3	1	40–60	3
							<40	1
Sum of Mean Percent Cover Turbidity-Tolerant							<10	5
Taxa in SAV Veg. Type	0	23	2	0	26	82	10–40	3
Score	5	3	5	5	3	1	>40	1
Sum of Mean Percent Cover Invasive Taxa in							<3	5
Sedge Veg. Type	1	0	1	4	13	10	3–7	3
Score	5	5	5	3	1	1	>7	1

Table 3. Data, scores, scoring ranges, and metric scores for fish community attributes selected as potential IBI metrics for use in barrier beach wetlands in western Lake Superior. Sites in order of decreasing percent of forested watershed are Siskiwit Bay (SB), Bark Bay (BB), Port Wing (PW), East Allouez Bay (EAB), Fish Creek (FC), and Hog Island (HI).

Attribute	SB	BB	PW	EAB	FC	HI	Scoring Range	Metric Score
Number Native Species	7	13	13	15	8	8	>14	5
Score	1	3	3	5	1	1	11–14	3
							<11	1
Number Native Cyprinid Species	2	4	2	3	2	0	>3	5
Score	3	5	3	3	3	1	2–3	3
							<2	1
Percent Sensitive Species	13	36	29	37	27	13	>32	5
Score	1	5	3	5	3	1	20–32	3
							<20	1
Percent Non-Native Species	13	14	7	21	27	25	<10	5
Score	3	3	5	1	1	1	10–18	3
							>18	1
Percent Individuals as Piscivores	25	4	8	19	10	1	>15	5
Score	5	1	3	5	3	1	6–15	3
							<6	1
Number Native Individuals	97	280	145	818	30	147	>200	5
Score	3	5	3	5	1	3	60–200	3
							<60	1
Percent Individuals Non-Native	0	2	1	20	9	13	<5	5
Score	5	5	5	1	3	3	5–15	3
							>15	1

At the quadrat-sampling scale, five attributes showed potential as metrics, also scoring Fish Creek and Hog Island low (Table 2). *Percent of Taxa as Obligate Wetland Plants* is a measure of tolerance that should decrease with disturbance. *FQI Score* is a measure of both tolerance/sensitivity and species richness and composition; it should decrease with disturbance. *Number of Native Taxa* is a measure of species richness that should decrease with disturbance. *Sum of Mean Percent Cover of Turbidity-Tolerant Taxa in SAV Vegetation Type* and *Sum of Mean Percent Cover of Invasive Taxa in the Sedge Vegetation Type* are measures of species composition that should increase with disturbance. The only outlier for these metrics was *Sum of Mean Percent Cover of Turbidity-Tolerant Taxa in SAV Vegetation Type* at Bark Bay, which resulted from quadrats sampled in the eastern part of the wetland that is influenced by a stream originating on a clay till plain and has naturally increased turbidity.

Fish Community Data and Potential Metrics. The barrier-beach-protected wetlands generally support fish communities with prominent species such as northern pike (*Esox lucius* L.), yellow perch (*Perca flavescens* Mitchill), white bass (*Morone chrysops* Raf.), rock bass (*Ambloplites rupestris* Raf.), black bullhead (*Ameiurus melas* Raf.), white sucker (*Catostomus commersoni* Lacepede), shorthead redhorse (*Moxostoma macrolepidotum* Lesueur), spottail shiner (*Notropis hudsonius* Clinton), emerald shiner (*N. atherinoides* Raf.), common carp (*Cyprinus carpio* L.), blackstripe topminnow (*Fundulus notatus* Raf.), and central mudminnow (*Umbra limi* Kirtland).

General trends were found for a number of the attributes evaluated; however, outliers were common. Four attributes showed potential as species richness/composition metrics (Table 3). *Number of Native Species*, *Number of Native Cyprinid Species*, and *Percent Sensitive Species* should decrease with disturbance. *Percent Non-Native Species* should increase with disturbance. Siskiwit Bay was an outlier for these attributes because it has habitat limitations due to lack of an inflowing stream and relatively little SAV vegetation type. East Allouez Bay is also an outlier, perhaps due to its proximity to the larger Allouez Bay and its population of fish.

Percent Individuals as Piscivores was the only attribute that could be considered for use as a trophic composition metric (Table 3); it should decrease with disturbance. However, this metric is less accurate in ranking all sites in the proper order, partly because Bark Bay data showed a low percent of piscivore numbers despite having numerous large northern pike.

Two attributes might be useful as fish abundance metrics (Table 3). *Number of Native Individuals*

Table 4. Data, scores, scoring ranges, and metric scores for invertebrate community attributes selected as potential IBI metrics for use in barrier beach wetlands in western Lake Superior. Sites in order of decreasing percent of forested watershed are Siskiwit Bay (SB), Bark Bay (BB), Port Wing (PW), East Allouez Bay (EAB), Fish Creek (FC), and Hog Island (HI).

Attribute	SB	BB	PW	EAB	FC	HI	Scoring Range	Metric Score
Median Number Taxa (not lower than genera)	21	17	21	15	13	14.5	>19	5
Score	5	3	5	1	1	1	16–19	3
							<16	1
Number Adult Trichoptera Species	21	26	18	24	13	5	>20	5
Score	5	5	3	5	3	1	10–20	3
							<10	1
Number Uncommon Adult Trichoptera Species	4	7	5	3	1	0	>5	5
Score	3	5	3	3	1	1	3–5	3
							<3	1
Median Cladocera Genera Richness	5	5.5	7	5	4	3.5	>6	5
Score	3	3	5	3	1	1	5–6	3
							<5	1
Median Number Individual Cladocera	1750	1300	550	1564	500	488	>1500	5
Score	5	3	1	5	1	1	1000–1500	3
							<1000	1
Median Number Individual Crustacea	2484	1891	1390	2073	1713	1331	>2250	5
Score	5	3	1	5	3	1	1500–2250	3
							<1500	1
Median Percent Individual Mites	<0.1	0	2.5	0.2	3.4	4.1	<1.5	5
Score	5	5	3	5	1	1	1.5–3.0	3
							>3.0	1

should decrease with disturbance. However, Siskiwit Bay rated low because of habitat limitations described above, and Hog Island had greater numbers of native individuals as a result of capturing numerous small black bullheads that showed visible signs of being in poor condition. *Percent of Individuals as Non-Native* should increase with disturbance. The only outlier for this metric was East Allouez Bay, which resulted from capture of large numbers of white bass.

Invertebrate Community Data and Potential Metrics. The number of organisms collected per funnel trap ranged from 86 to 11,887 per night at the six Lake Superior sites. Sixty-three total taxa were collected, and 16 taxa occurred in over 50% of the samples. Catches were dominated by planktonic and benthic crustacea and four epibenthic taxa–ostracods (seed shrimps), *Gammarus* (amphipods or sideswimmers), Hydracarina (water mites), and corixids (water boatmen). Hog Island provided large catches of ostracods and water mites; catches at Fish Creek were low except for the cladoceran *Chydorus*. Catches at East Allouez Bay contained large numbers of euplanktonic taxa such as cladocerans *Daphnia*, *Bosmina*, and *Diaphanosoma* and copepods *Diaptomus* and *Mesocyclops*. Siskiwit Bay, Bark Bay, and Port Wing had the greatest overall taxa richness and large catches of cladocerans *Simocephalus* and *Ceriodaphnia*.

Collections of the adult aquatic insect community were dominated by caddisflies (Trichoptera) and midges (Chironomidae). For caddisflies, the leptocerids, polycentropodids, and limnephilids were most diverse (in descending order). Only two species, *Oecetis inconspicua* (Walker) and *Phylocentropus placidus* (Banks), were found at four or more of the six sites. Uncommon caddisfly taxa identified from these wetlands included *Platycentropus amicus* (Hagen), *Oxyethira ecornuta* Morton, *Triaenodes nox* Ross, and *Polycentropus melanae* (Ross).

Four attributes showed potential as species richness/composition metrics (Table 4). *Median Number of Taxa* (not lower than genera), *Number of Adult Trichoptera Species, Number of Uncommon Adult Trichoptera Species,* and *Median Cladocera Genera Richness* should all decrease with disturbance as a result of factors such as altered water or sediment chemistry, altered food source, and altered amount and structure of protective habitat provided by plant communities. All ranked Fish Creek and Hog Island lower than the other four sites but again displayed several outliers.

Three attributes reflecting abundance might serve as useful metrics (Table 4). *Median Number of Individuals as Cladocera* and *Median Number of Individuals as Crustacea* should decrease with disturbance if wa-

Table 5. Sums of plant, fish, and invertebrate potential metric scores and associated biotic integrity classifications for barrier beach wetlands in western Lake Superior and tentative total IBI scores and wetland integrity classifications based on sums of scores from plant, fish, and invertebrate communities. The sites, in order of decreasing percent of forested watershed, are Siskiwit Bay (SB), Bark Bay (BB), Port Wing (PW), East Allouez Bay (EAB), Fish Creek (FC), and Hog Island (HI). Integrity classes are Excellent (E), Good (G), Fair (F), Poor (P), and Very Poor (VP).

	Lake Superior Site					
Metric or Biotic Integrity Type	SB	BB	PW	EAB	FC	HI
Sum of plant metric scores	33	33	31	25	13	7
Plant biotic integrity class	E	E	G–E	F	VP	VP
Sum of fish metric scores	21	27	25	25	15	11
Fish biotic integrity class	P–F	F–G	F	F	VP–P	VP
Sum of invertebrate metric scores	31	27	21	27	11	7
Invertebrate biotic integrity class	G–E	F–G	P–F	F–G	VP	VP
Total IBI score (plant + fish + invertebrate)	85	87	77	77	39	25
Wetland biotic integrity class	G	G	F–G	F–G	VP	VP

ter/sediment chemistry, food source, or habitat are altered. They could also decrease as a result of hydrologic flashiness caused by increased runoff from the watershed or alteration of the hydrologic connection between the wetland and lake, which might allow large numbers to be flushed from the wetland occasionally. *Median Percent Individuals as Mites* should increase as disturbance reduces the protective structure of plant communities and provides better access to prey for these predators. These metrics also generally ranked Fish Creek and Hog Island lower than the other sites, with two exceptions. Low values for *Median Number of Individuals as Cladocera* and *Median Number of Individuals as Crustacea* in Port Wing samples were likely due to flashiness of water flow related to widening of the hydrologic connection with the lake at the marina. In addition, East Allouez Bay ranked unexpectedly high in the abundance metrics, again perhaps related to its contiguity with the larger bay.

Tentative IBI Scores and Wetland Ranks. Although the attempted dose-response curves for some potential metrics required explanation of outlier data, some overall patterns became evident. The sums of potential plant community metrics for Lake Superior barrier-beach wetlands ranked the sites in the order Siskiwit Bay/Bark Bay, Port Wing, East Allouez Bay, Fish Creek, and Hog Island (Table 5), which was the predicted order. As expected, Fish Creek and Hog Island scored well below the other four sites. The sums of potential fish community metrics ranked the sites in the order Bark Bay, Port Wing/East Allouez Bay, Siskiwit Bay, Fish Creek, and Hog Island (Table 5). Siskiwit Bay ranked lower than expected because of natural limitations in fish habitat. East Allouez Bay ranked higher than expected, perhaps because its connection with greater Allouez Bay provides potential access by more species and more individuals. Fish Creek and Hog Island again scored well below the other four sites. The sums of potential invertebrate community metrics ranked the sites in the order Siskiwit Bay, Bark Bay/East Allouez Bay, Port Wing, Fish Creek, and Hog Island (Table 5), again nearly matching the predicted order but suggesting that disturbance factors at Port Wing have a greater effect on invertebrates than on plants or fish.

Tentative total IBI scores based on the sums of all plant, fish, and invertebrate potential metric scores ranked the sites in order of increasing degradation as Bark Bay (87), Siskiwit Bay (85), Port Wing and East Allouez Bay (both 77), Fish Creek (39), and Hog Island (25) (Table 5). This ranking is generally consistent with the gradient of disturbance predicted by percent of watershed that is forested. However, if IBI scores are plotted against percent forested watershed, a linear relationship does not result, suggesting that other disturbance factors not related to deforestation or development in the watershed are influencing the attributes included as potential metrics in the IBI.

Biological Integrity of Study Sites. Because seven potential metrics were tentatively selected for each of the three types of biological communities sampled in Lake Superior wetlands, the maximum possible IBI score was 35 (5×7) for each community and 105 (3×35) for the sum of the three communities. Scoring ranges for the biological communities were indexed as follows: Excellent (33–35), Good (28–30), Fair (22–25), Poor (17–20), and Very Poor ($\leq$13). Therefore, scoring ranges for the sum of the three communities were Excellent (99–105), Good (84–90), Fair (66–75), Poor (51–60), and Very Poor ($\leq$39).

The sums of potential plant community metrics classified Siskiwit Bay and Bark Bay as Excellent, Port Wing as Good-Excellent, East Allouez Bay as Fair, and Fish Creek and Hog Island as Very Poor (Table

5). The sums of potential fish community metrics classified Bark Bay as Fair-Good, Port Wing and East Allouez Bay as Fair, Siskiwit Bay as Poor-Fair, and Fish Creek as Very Poor-Poor, and Hog Island as Very Poor (Table 5). The sums of potential invertebrate community metrics classified Siskiwit Bay as Good-Excellent, Bark Bay and East Allouez Bay as Fair-Good, Port Wing as Poor-Fair, and Fish Creek and Hog Island as Very Poor (Table 5). Classification according to the sum of potential metrics for the three community types rated Siskiwit Bay and Bark Bay as Good, Port Wing and East Allouez Bay as Fair-Good, and Fish Creek and Hog Island as Very Poor (Table 5).

Lake Michigan

Many of the human disturbances affecting the drowned-river-mouth wetlands of Lake Michigan were localized, thus rendering disturbance gradients based on watershed features ineffective for many attributes. Therefore, we were forced to define gradients based on other features; the results were not promising. We developed a general ranking of the six sites based on the number and severity of local disturbance sources. That order, from least to most disturbed, is Lincoln River, Betsie River and Arcadia Lake, Pentwater River, Little Manistee River, and Pere Marquette River. Our data suggest that least-disturbed Lincoln River is not pristine; however, no other reference wetlands of comparable type are available. Data are presented in tables in the order given above, but the dose response was not determined based on that order for all attributes. Certain specific attributes demonstrated correlations with other features of the sites and were tested with dose-response curves based on those characteristics. For example, variability in sediment type from sand to decomposed peat and silt ranked the sites in the order Pere Marquette River, Betsie River, Little Manistee River, Pentwater and Lincoln rivers, and Arcadia Lake, which was used to test *Percent Tolerant Species* of fish. Yet other attributes were related to specific disturbances at individual sites rather than a gradient of similar disturbances; they were also retained as potential metrics. An example is *Percent Wetland in Sedge Vegetation Type*, an attribute in which the Pentwater River site was affected by gouging of the wetland following failure of the upstream dam. We reiterate here that many of the potential metrics described do not demonstrate strong dose responses; however, they are the best choices suggested by our data and might warrant further evaluation.

Plant Community Data and Potential Metrics. Vegetation in the drowned-river-mouth wetlands generally consists of sedge/grass meadows dominated by *Carex stricta* Lam. and *Calamagrostis canadensis* (Michaux) Beauv., short emergent communities dominated by *Sparganium eurycarpum* and *Sagittaria latifolia* Willd., with *Eleocharis smallii* and *Pontederia cordata* L. also prevalent; tall emergent communities dominated by *Typha angustifolia*; and floating leaf and SAV communities containing *Nuphar variegata*, *Nymphaea odorata* Aiton, *Ceratophyllum demersum*, *Potamogeton* sp., *Elodea canadensis*, and *Myriophyllum spicatum*.

The mapping-scale attribute *Percent Wetland in Sedge Vegetation Type* was retained as a potential metric, with a reasonable dose-response curve comparing sites according to specific disturbances (Table 6). This attribute has obvious ecological implications in these wetlands where the dominant emergent vegetation type is being reduced in area and replaced by other community types. However, *Percent Wetland in Invasive Vegetation Types* does not show the same pattern (and was not retained) because not all of the replacement types are considered invasive. At the quadrat-sampling scale, four attributes showed potential as metrics (Table 6). *FQI Score* was highest at the least disturbed Lincoln River wetland and lowest at the most disturbed Pere Marquette wetland. *Number of Native Taxa* was much lower at the Pere Marquette wetland also, but Little Manistee River was an outlier with additional taxa present because landfilling, channelization, and construction of a berm created more variety in habitats. *Sum of Mean Percent Cover of Turbidity-Tolerant Taxa in SAV Vegetation Type* showed the Pere Marquette River site as an outlier because sediments beneath the SAV and floating leaf vegetation types are mostly sand, lack a silt component, and create little turbidity. Arcadia Lake was also an outlier because much of the SAV community had been heavily grazed by mute swans (*Cygnus olor* Gmelin). *Sum of Mean Percent Cover of Invasive Taxa in Sedge Vegetation Type* was less rigorous when viewed across a gradient of disturbance, but two sites affected by localized disturbance (cattle grazing at Arcadia Lake and channel construction at Little Manistee River) scored lower than the other sites. Lower-than-expected cover values (and higher scores) at Pere Marquette River and Pentwater River reflect the localized distribution pattern of invasive emergent plants at the sites; much of the invasion has occurred in areas affected by the landfill or gouging following the dam break, not in the middle of the extant sedge/grass meadow.

Fish Community Data and Potential Metrics. The fish communities in the drowned-river-mouth wetlands included northern pike; numerous bowfin (*Amia calva* L.), rock bass, yellow perch, black crappie (*Pomoxis*

Table 6. Data, scores, scoring ranges, and metric scores for plant community attributes selected as potential IBI metrics for use in drowned-river-mouth wetlands in middle Lake Michigan. Sites in order of increasing disturbance from multiples sources are Lincoln River (LR), Betsie River (BR), Arcadia Lake (AL), Pentwater River (PR), Little Manistee River (LMR), and Pere Marquette River (PMR). Different rank orders for evaluating some attributes are described in footnotes.

Attribute	LR	BR	AL	PR	LMR	PMR	Scoring Range	Metric Score
Rank Order[1]	b	c	a	f	e	d	>60	5
Percent Wetland in Sedge Veg. Type	49	42	89	33	41	50	40–60	3
Score	3	3	5	1	3	3	<40	1
Floristic Quality Index (FQI)	31.0	29.2	28.0	28.0	29.2	25.5	>30	5
Score	5	3	3	3	3	1	27–30	3
							<27	1
Number Native Taxa	59	65	55	60	68	44	>62	5
Score	3	5	3	3	5	1	50–62	3
							<50	1
Sum of Mean Percent Cover Turbidity-Tolerant Taxa							<35	5
in SAV Veg. Type	30	37	23	57	54	40	35–45	3
Score	5	3	5	1	1	3	>45	1
Rank Order[2]	a	b	f	d	e	c	<10	5
Mean % Cover Invasive Taxa in Sedge Veg. Type	12	9	26	8	15	9	10–20	3
Score	3	5	1	5	3	5	>20	1

[1] Ranked a–f according to unique basin shape (AL) and increasing levels of localized disturbance [landfills (PMR and LMR), channel construction (LMR), and dam break (PR)].
[2] Ranked a–f according to increasing levels of localized disturbance [landfills (PMR and LMR), dam break (PR), channel construction (LMR), and cattle grazing (AL)].

nigromaculatus Lesueur), bluegill (*Lepomis macrochirus* Raf.), and common carp; and large numbers of pumpkinseed (*Lepomis gibbosus* L.), largemouth bass (*Micropterus salmoides* Lacepede), bluntnose minnow (*Notropis simus* Cope), and brown bullhead (*Ameiurus nebulosus* Lesueur). Of special note were several thousand young-of-year common carp collected at the Pere Marquette River site.

The *Shannon Index* value for fish in the Pere Marquette River wetland was extremely low due to the large numbers of common carp (Table 7). The Little Manistee River wetland also scored low for this attribute, and the sites with less disturbance scored higher, thus making it a potential diversity metric. Pentwater River was an outlier because no large schools of bluntnose minnows or pumpkinseeds were captured at this site. *Number of Native Species* and *Percent Tolerant Species* are potential species richness/composition metrics, although they make sense only when compared against specific gradients of wetland size and type of sediment, respectively.

Four attributes seemed to have potential as trophic composition metrics. *Percent Individuals as Piscivores, . . . Planktivores, . . . Omnivores,* and *. . . Insectivores* are interrelated, but *Percent Individuals as Omnivores* generally increases with disturbance while the others decrease (Table 7). The catch from the Betsie River wetland was an outlier for *Percent Individuals as Piscivores* because this site had only small areas of SAV habitat and contained relatively few northern pike and largemouth bass, which perhaps partially explains the large numbers of young-of-year insectivorous bluntnose minnows (see *Percent Individuals as Insectivores,* Table 7). Abundance attribute *Number of Native Individuals* decreased with disturbance.

Invertebrate Community Data and Potential Metrics. Funnel traps collected 126 to 21,497 invertebrates per night at the Lake Michigan wetland sites. A total of 63 taxa were recorded; 15 of them occurred in more than 50% of the samples. Like Lake Superior sites, catches in these wetlands were dominated by planktonic and benthic crustacea and epibenthic ostracods, *Gammarus*, Hydracarina, and corixids. However, they had fewer *Diaphanosoma* and *Simocephalus* and a greater prevalence of caddisflies, mayflies, turbellarians, and snails. Pere Marquette River wetland had the greatest numbers of *Simocephalus*; Arcadia Lake had the most caddisflies; and Pentwater River had the most *Gammarus*. Little Manistee River wetland had the greatest abundance of euplanktonic forms, including littoral cladocerans (*Ceriodaphnia*), limnetic cyclopoid copepods (*Tropocyclops*), and *Diaphanosoma*, and low numbers of corixids. Betsie River had greater numbers of three benthic cladocerans —*Alona*, *Camptocercus*, and *Chydorus*. Lincoln River wetland was

Table 7. Data, scores, scoring ranges, and metric scores for fish community attributes selected as potential IBI metrics for use in drowned-river-mouth wetlands in middle Lake Michigan. Sites in order of increasing disturbance from multiples sources are Lincoln River (LR), Betsie River (BR), Arcadia Lake (AL), Pentwater River (PR), Little Manistee River (LMR), and Pere Marquette River (PMR). Different rank orders for evaluating some attributes are described in footnotes.

Attribute	LR	BR	AL	PR	LMR	PMR	Scoring Range	Metric Score
Shannon Index	1.65	1.92	1.71	2.20	1.42	0.45	>2	5
Score	3	5	3	5	3	1	1–2	3
							<1	1
Rank Order[1]	f	c	b	d	e	a	>19	5
Number Native Species	17	20	20	19	17	20	18–19	3
Score	1	5	5	3	1	5	<18	1
Rank Order[2]	e	b	f	d	c	a	<33	5
Percent Tolerant Species	35	31	39	36	37	26	33–36	3
Score	3	5	1	3	1	5	>36	1
Percent Individuals as Piscivores	40	5	57	30	27	3	>35	5
Score	5	1	5	3	3	1	15–35	3
							<15	1
Percent Individuals as Planktivores	1.6	0.4	0.4	0.5	0	0.3	>1.0	5
Score	5	3	3	3	1	3	0.2–1.0	3
							<0.2	1
Percent Individuals as Omnivores	49	47	41	65	73	96	<55	5
Score	5	5	5	3	3	1	55–80	3
							>80	1
Percent Individuals as Insectivores	10	48	1	5	<1	1	>20	5
Score	3	5	1	3	1	1	3–20	3
							<3	1
Number Native Individuals	2346	644	525	439	600	233	>800	5
Score	5	3	3	3	3	1	350–800	3
							<350	1

[1] Ranked a–f according to decreasing wetland size as potential for habitat availability.
[2] Ranked a–f according to increasing amounts of decomposed peat and silt in sediments.

characterized by large numbers of cladocerans *Sida* and *Diaphanosoma* and the copepod *Acanthocyclops*.

The adult aquatic insect collections were dominated by caddisflies and midges. Leptocerid, hydroptilid, limnephilid, and polycentropodid caddisflies were most diverse (in descending order). Common caddisflies identified from all six wetlands include the hydroptilids *Agraylea multipunctata* Curtis, *Hydroptila waubesiana* Betten, and *Oxyethira pallida* (Banks); the leptocerids *Ceracela tarsipunctata* (Vorhies), *Oecetis cinerascens* (Hagen), *Oecetis inconspicua*, and *Triaenodes tardus* Milne; and the polycentropodid *Polycentropus cinereus* Hagen. There were few uncommon caddisfly species.

Two diversity attributes showed potential as metrics for invertebrate communities. *Median Shannon Index* generally decreased with disturbance, as did *Median Shannon Index of Cladocera* (Table 8). However, the Little Manistee River wetland was an outlier in both attributes, as well as several others. This site is exposed to limnetic waters of Manistee Lake, which likely increased the numbers and catches of euplanktonic forms.

Species richness/composition attributes *Median Number of Taxa* (not lower than genera), *Number of Adult Trichoptera Species,* and *Median Cladocera Genera Richness* also generally decreased with disturbance and were selected as potential metrics (Table 8). The Little Manistee River wetland was again an outlier for *Median Number of Taxa* and *Median Cladocera Genera Richness;* the Arcadia Lake wetland was an outlier for *Number of Adult Trichoptera Species*, perhaps because there is less diversity in wetland vegetation types.

Median Number of Individuals as Cladocera and *Median Number of Individuals as Crustacea* also decreased with disturbance and were selected as potenial abundance metrics (Table 8). The Little Manistee River wetland was an outlier in these metrics also.

Tentative IBI Scores and Wetland Ranks. Some of the dose-response curves for Lake Michigan drowned-

Table 8. Data, scores, scoring ranges, and metric scores for invertebrate community attributes selected as potential IBI metrics for use in drowned-river-mouth wetlands in middle Lake Michigan. Sites in order of increasing disturbance from multiples sources are Lincoln River (LR), Betsie River (BR), Arcadia Lake (AL), Pentwater River (PR), Little Manistee River (LMR), and Pere Marquette River (PMR).

Attribute	LR	BR	AL	PR	LMR	PMR	Scoring Range	Metric Score
Median Shannon Index	0.81	0.81	0.84	0.70	0.78	0.63	>0.80	5
Score	5	5	5	3	3	1	0.65–0.80	3
							<0.65	1
Median Shannon Index of Cladocera	0.42	0.42	0.17	0	0.26	0	>0.30	5
Score	5	5	3	1	3	1	0.10–0.30	3
							<0.10	1
Median Number Taxa (not lower than genera)	18.5	18.5	19.5	14.5	17	13.5	>19	5
Score	5	5	5	1	3	1	16–19	3
							<16	1
Number Adult Trichoptera Species	33	24	18	24	19	21	>20	5
Score	5	3	1	3	1	1	10–20	3
							<10	1
Median Cladocera Genera Richness	4	5.5	2.5	1	3	1	>6	5
Score	5	5	3	1	3	1	5–6	3
							<5	1
Median Number Individual Cladocera	313	563	175	25	2225	25	>1500	5
Score	3	5	3	1	5	1	1000–1500	3
							<1000	1
Median Number Individual Crustacea	1703	1282	1163	1034	4172	822	>2250	5
Score	5	3	3	3	5	1	1500–2250	3
							<1500	1

river-mouth wetlands show metrics with potential, but the overall results are not promising. We followed through with IBI development anyway to allow the results to be evaluated. The sums of plant community metrics for these sites ranked them in the order Lincoln River/Betsie River, Arcadia Lake, Little Manistee River, and Pentwater River/Pere Marquette River (Table 9). The sums of fish community metrics ranked them as Betsie River, Lincoln River, Arcadia Lake/Pentwater River, Pere Marquette River, and Little Manistee River (Table 9). The sums of invertebrate community metrics ranked the sites in the order Lincoln River, Betsie River, Arcadia Lake/Little Manistee River, Pentwater River, and Pere Marquette River (Table 9). Despite reversals in ranking of some sites due to certain metrics that seemed to respond to site-specific disturbances, these rankings showed trends that were in general agreement with the gradient of disturbance based on number and severity of disturbance sources, with Lincoln River and Betsie River usually best and Pere

Table 9. Sums of plant, fish, and invertebrate potential metric scores and associated biotic integrity classifications for drowned-river-mouth wetlands in middle Lake Michigan and tentative total IBI scores and wetland integrity classifications based on sums of scores from plant, fish, and invertebrate communities. The sites, in order of decreasing disturbance from multiples sources, are Lincoln River (LR), Betsie River (BR), Arcadia Lake (AL), Pentwater River (PR), Little Manistee River (LMR), and Pere Marquette River (PMR). Integrity classes are Excellent (E), Good (G), Fair (F), Poor (P), and Very Poor (VP).

	Lake Michigan Site					
Metric or Biotic Integrity Type	LR	BR	AL	PR	LMR	PMR
Sum of plant metric scores	19	19	17	13	15	13
Plant biotic integrity class	F–G	F–G	F	P	P–F	P
Sum of fish metric scores	30	32	26	26	16	18
Fish biotic integrity class	F–G	G	F	F	VP–P	VP–P
Sum of invertebrate metric scores	33	31	23	13	23	7
Invertebrate biotic integrity class	E	G–E	F	VP	F	VP
Total IBI score (plant + fish + invertebrate)	82	82	66	53	54	38
Wetland biotic integrity class	G	G	F	P	P	VP–P

Marquette River usually worst. However, they suggest that fish communities in the Little Manistee River wetlands are more impacted by disturbance than are the plant and invertebrate communities.

Tentative total IBI scores based on the sum of all plant, fish, and invertebrate metric scores ranked the sites in the order Lincoln River/Betsie River (82), Arcadia Lake (66), Little Manistee River (54), Pentwater River (53), and Pere Marquette River (38) (Table 9). Again, this ranking is nearly consistent with the gradient of disturbance based on number and severity of disturbance sources, but supporting data for it are not strong.

Biological Integrity of Study Sites. Five potential metrics were selected for plant communities, eight for fish communities, and seven for invertebrate communities sampled in Lake Michigan wetlands. Therefore, the maximum possible IBI scores for these communities were 25 (5×5), 40 (5×8), and 35 (5×7), respectively, and 105 for the sum of the three communities. Scoring ranges for the plant communities were indexed as Excellent (23–25), Good (20–21), Fair (16–18), Poor (12–14), and Very Poor ($\leq$9). Scoring ranges for fish communities were Excellent (38–40), Good (32–35), Fair (26–29), Poor (20–23), and Very Poor ($\leq$15). For invertebrate communities, the scoring ranges were Excellent (33–35), Good (28–30), Fair (22–25), Poor (17–20), and Very Poor ($\leq$13). The scoring ranges for the sum of the three communities were Excellent (94–100), Good (80–86), Fair (63–71), Poor (49–57), and Very Poor ($\leq$37).

The sums of potential plant community metrics classified Lincoln River and Betsie River as Fair-Good, Arcadia Lake as Fair, Little Manistee River as Poor-Fair, and Pentwater River and Pere Marquette River as Poor (Table 9). The sums of potential fish community metrics classified Betsie River as good, Lincoln River as Fair-Good, Arcadia Lake and Pentwater River as Fair, and Little Manistee River and Pere Marquette River as Very Poor-Poor (Table 9). The sums of potential invertebrate community metrics classified Lincoln River as Excellent, Betsie River as Good-Excellent, Arcadia Lake and Little Manistee River as Fair, and Pentwater River and Pere Marquette River as Very Poor (Table 9). Classification according to the sum of potential metrics for the three community types rated Lincoln River and Betsie River as Good, Arcadia Lake as Fair, Pentwater River and Little Manistee River as Poor, and Pere Marquette River as Very Poor-Poor (Table 9).

Testing of Metrics

The potential metrics for use in wetlands of the three lakes were tested by comparing the results to additional gradients of disturbance. Metrics for the Lake Superior sites developed against a gradient of percent forested watershed were tested against a gradient of percent urbanized watershed. The Siskiwit Bay and Bark Bay watersheds are about 1% urbanized; Port Wing is about 3% urbanized; East Allouez Bay and Fish Creek are about 10% urbanized; and Hog Island is more than 70% urbanized. In general, most potential metrics for Lake Superior wetlands tested with results similar to those for percent forested watershed, as might be expected since amounts of the two land-use classes are correlated. However, there were exceptions. Outliers identified using the percent forested watershed gradient also appeared in testing against percent urbanized watershed; Fish Creek also scored much lower than expected in many metrics (e.g., *Percent Wetland in Sedge Vegetation Type*, *Mean Percent Cover of Invasive Taxa in Sedge Vegetation Type*, *Percent Non-Native Species* (fish), *Number of Native Individuals* (fish), and *Median Number of Taxa* (invertebrates)). Lands in the Fish Creek watershed that are not forested or urbanized include active agricultural use, such as dairy farming and row crops. The remainder of the Siskiwit Bay, Bark Bay, Port Wing, and East Allouez Bay watersheds include more fallow pastures and fields. We could not derive meaningful numbers for percent agricultural watershed, but the low scores for Fish Creek are likely related to that land use.

We also tested the metrics for fish and invertebrate communities of the Lake Superior sites against habitat gradients defined by the plant FQI values and by the sum of the plant metric scores. These gradients produced results similar to percent forested watershed.

The potential metrics for Lake Michigan wetlands developed against a general ranking of sites affected by multiple local disturbance factors did not test successfully against any additional measures of disturbance based on watershed characteristics (percent forested, percent urbanized, percent agriculture). Plant FQI and sum of the plant metric scores served as habitat gradients that showed some correlation to fish community metrics *Percent Individuals as Omnivores* and *Number of Native Individuals* plus invertebrate community metrics *Median Shannon Index*, *Median Shannon Index of Cladocera*, and *Median Number of Taxa*.

DISCUSSION

Validity of an IBI for Great Lakes Wetlands

At first glance, our results suggest that the metrics proposed as potential components of an IBI for barrier beach wetlands of Lake Superior might be developed further to provide a means to evaluate wetland integrity. The metrics for Lake Michigan drowned-river-mouth wetlands may not be suitable for an IBI. As stated previously, those for Lake Huron open shoreline wetlands held little promise and were not explored further. However, sampling methods were similar at each lake, and there was overlap in taxa encountered. Why was the outcome different between lakes and wetland types? Can a valid IBI for the Lake Superior (or Lake Michigan) wetlands be developed?

Karr et al. (1986) compared IBI outcomes against six criteria defined by Herricks and Schaeffer (1985) as necessary for valid biomonitoring programs. Our wetland IBI attempts for Great Lakes wetlands satisfied some of those criteria but failed others. 1. *The measure must be biological*–satisfied. 2. *The measure must be interpretable at several trophic levels or provide a connection to other organisms not directly involved in the monitoring*–satisfied. The metrics specifically measured not only different trophic levels of fish but also different trophic levels of invertebrates that often provide food for fish and plant communities that provide habitat for both fish and invertebrates. 3. *The measure must be sensitive to the environmental conditions being monitored*–satisfied. The metrics showed apparent responses to multiple types of disturbance, including chemical contamination, landfills, hydrologic alteration of outflowing rivers, sediment loading, industrial and residental shoreline development, flood pulse from upstream dam failure, ditching, and cattle grazing. 4. *The response range of the measure must be suitable for the intended application*–satisfied. Metrics were included that identified responses to extreme disturbances, such as dam failure and chemical contamination, yet seemed responsive to incremental disturbances such as ditching. 5. *The measure must be reproducible and precise within defined and acceptable limits for data collected over space and time*–not satisfied. As will be discussed later, the magnitude of natural lake-level change from year to year could yield results that are not reproducible. 6. *The variability of the measure must be low*–not satisfied. Again, extreme natural variability in lake levels represents great variability in wetland hydrology, produces great variability in plant communities, and results in great variability in habitat for fish and invertebrate communities.

Potential for Lake Superior Barrier Beach Wetland IBI. Although most potential plant community metrics for the Lake Superior sites proved to be consistent with the predicted order based on site disturbance, our measurements were taken in a single year and represent conditions resulting from a date-specific lake-level history. Growing season water levels in Lake Superior reached a high of about 183.84 m IGLD85 (International Great Lakes Datum 1985) in August 1985 and 1986, dropped to a seasonal high ranging from 183.26 to 183.53 m from 1988 to 1991, and were about 183.60 m during the July–August 1993 sampling season (Figure 3a). Both composition and structure of wetland plant communities in the Great Lakes are highly dependent on water-level changes (Keddy and Reznicek 1986, Wilcox et al. 1993, Wilcox 1995, Kowalski and Wilcox 1999, Wilcox and Whillans 1999). The plant communities we sampled in 1993 had responded to extremely high lake levels in 1985–86 that likely killed many emergent species, a half-meter drop in water level that likely exposed sediments and elicited germination of emergent plants from the seed bank, and a subsequent quarter-meter rise in water level that likely sustained many of the newly-established emergents and permitted some intermingling of submersed or floating plants (Wilcox 1995, Wilcox and Meeker 1995).

Even with no changes in the extent of human disturbance to these wetlands through time, if we had sampled in 1986 or 1990 (or any of a number of years before and after 1985–86), at least portions of the plant community data could have differed substantially from those that we obtained, as evidenced by previous work (Keddy and Reznicek 1986, Wilcox et al. 1993, Wilcox 1995, Kowalski and Wilcox 1999). Despite our lack of comparable data for those years, we must then conclude that our plant community measures are likely not reproducible over time and have enough inherent variability to invalidate the scoring ranges for many of the metrics unless subsequent sampling was conducted in a year with similar lake-level history and similar plant communities. Because wetland plant communities are a major component of the habitat supporting wetland fish and invertebrate communities, questions regarding reproducibility and variability in fish and invertebrate data over time suggest that scoring ranges for those metrics are also not valid for use except in a year with similar lake-level history. Within the year sampled, however, the Lake Superior barrier-beach wetland metrics are useful and seem to recognize impacts from known human disturbances, as well as some natural variability among sites. For example, fish sampled in the chemically contaminated Hog Island wetland were lethargic and often discolored as if bleached; scores for most fish metrics at Hog Island were very low. Siskiwit Bay wetlands were considered least disturbed, but low scores for several fish metrics reflect reduced habitat due to natural site characteristics.

Although scoring ranges for many metrics may not be valid in years with different water-level histories, overall testing of potential metrics proved more successful for Lake Superior wetlands than for the other lakes. A possible reason for this result is that much of

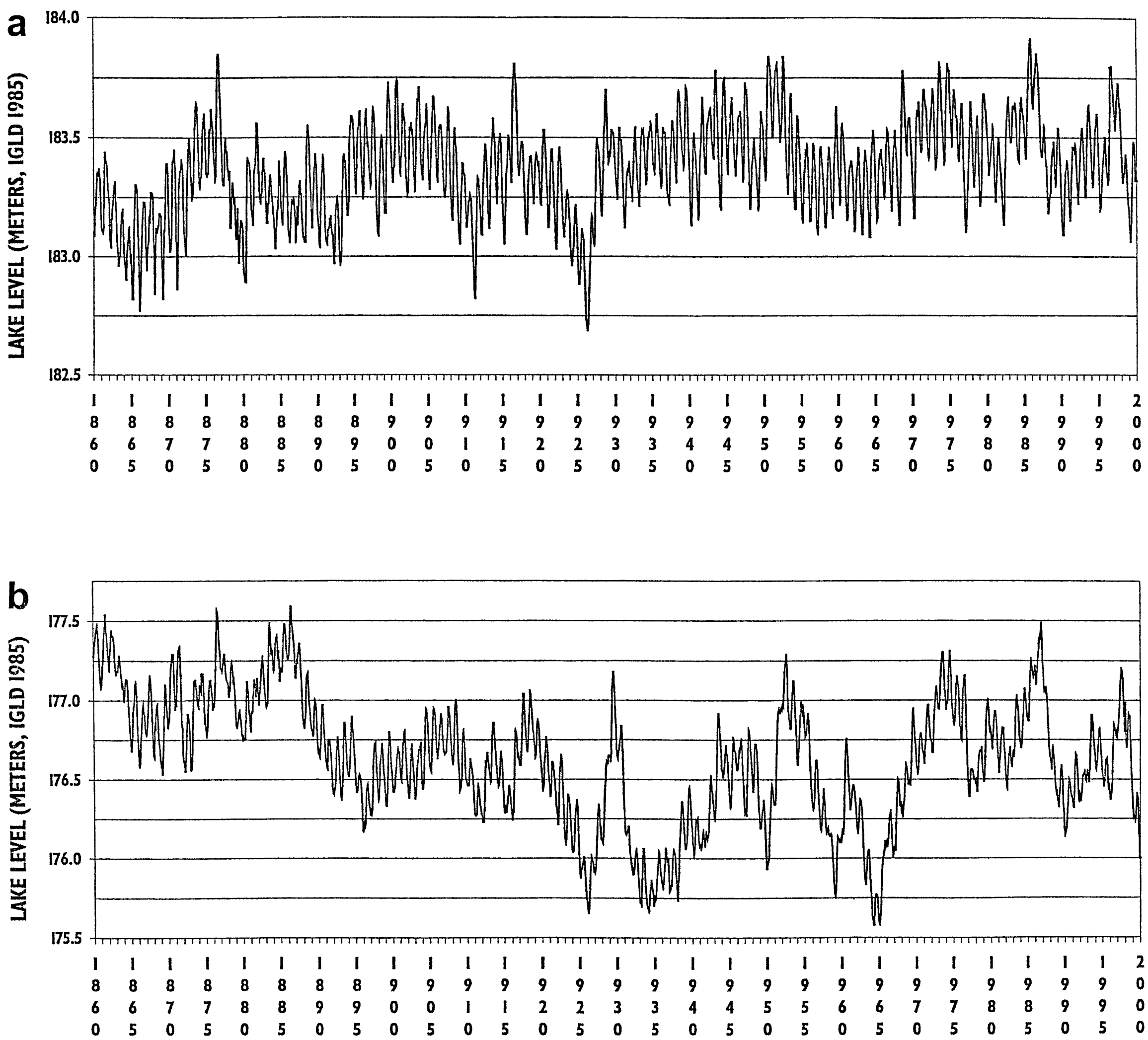

Figure 3. Water-level history from 1860 to 2000 of a) Lake Superior and b) lakes Michigan-Huron.

the emergent vegetation is floating sedge mat, and that portion of the plant community data may be immune to many of the effects of water-level change. *Percent Wetland in Sedge Vegetation Type* would likely not be affected by changes in lake level, while *Percent Wetland Obligate Species, FQI,* and *Number of Native Taxa* might be less variable and more reproducible because the floating mats support many of the taxa used to calculate those metrics. The scope of expected year-to-year variability in plant communities may, therefore, be more heavily dependent on a smaller component of the wetland vegetation (submersed, floating-leaf, and non-floating-mat emergent communities that are affected by high or low water levels), and overall variability in data would be reduced. Reduced variability in plant habitat may also translate to reduced variability in some aspects of fish and invertebrate community data. Lake Superior also lies at the upstream end of the Great Lakes system, and its hydrology is not affected by that of the other lakes and their watersheds; the amplitude and year-to-year variability of water-level changes is naturally less than the downstream lakes. Lake Superior water levels are also regulated, which further reduces amplitude and variability.

Potential for Lake Michigan Drowned-River-Mouth Wetland IBI. Problems in metric development for Lake Michigan sites involved lack of a single, well-defined disturbance gradient, which made dose-response analyses difficult, and concerns that the least-disturbed site was not pristine. In addition, the amplitude of water-level changes during the growing season at the Lake Michigan wetland sites spans a greater range than in Lake Superior, and year-to-year variability is greater also. Water levels in Lake Michigan reached a growing season high of 177.39 m (IGLD85) in July–August 1986, dropped to growing season highs ranging from 176.35 to 176.70 m from 1988 to 1992, rose to 176.91 m in 1993, and were about 176.63 m during the 1995 sampling season (Figure 3b). Therefore, after the extremely high water levels of 1986, wetland plant communities responded to a drop of nearly a meter and then an additional rise and fall of lesser magnitude. Again, we conclude that many of the plant community measures are likely not reproducible over time and are too variable to allow the scoring ranges for many of the metrics to be used unless sampling is conducted in a year with similar lake-level history.

As with Lake Superior sites, the scoring ranges for fish and invertebrate metrics for the Lake Michigan wetlands are probably not valid for multiple years due to variability and lack of reproducibility in the data. This assumption is supported by field observations at drowned-river-mouth wetlands made in early and late summer 1999, when water levels were more than 30 cm lower than during the 1995 sampling season (Figure 3b). In early summer, remnants of floating-leaf and submersed plants were evident in areas similar to those where fish and invertebrates had been sampled in 1995. However, the remnants were completely exposed on mudflats; standing water in some wetlands was largely restricted to the original course of the river channel through the wetland. The fish and invertebrate habitat provided by macrophytes was not available and could not be sampled in standing water. Fyke nets and funnel traps placed in the macrophyte-free channels where water was available to allow sampling would have captured considerably different arrays of species than collected in 1995. In early summer, emergent plant species were also beginning to grow from the exposed seed bank. By late summer, some areas of mudflat supported numerous emergent species with variable leaf shapes and heights ranging from several centimeters to greater than one meter. When reflooded in future years, these areas will likely recolonize with floating-leaf and submersed plants interspersed among the emergent plants. The rejuvenated habitat could then support fish and invertebrate communities that differ from both 1995 and 1999.

Despite the difficulty in defining metrics for drowned-river-mouth wetlands of Lake Michigan due to the localized nature of the sources of disturbance and failure of those metrics to test successfully against other disturbance gradients, some of the 1995 results reflect known human disturbances, which suggests some utility as indicators of wetland degradation. For example, the Pere Marquette River wetland had the lowest total for sum of fish metric scores, and a northern pike captured near the former industrial landfill at the site displayed a 5-cm-diameter tumor. Although the Pere Marquette River is considered a high quality trout stream, the fish metrics suggested that fish habitat was degraded in the downriver wetland section.

Potential plant community metrics for Lake Michigan showed less promise than those for Lake Superior. The major emergent vegetation type in the Lake Michigan drowned-river-mouth wetlands was sedge/grass meadow dominated by *Carex stricta* and *Calamagrostis canadensis*. Both species grew primarily on tussocks formed by *Carex stricta* (tussock sedge) that serve as an adaptation for survival in hydrologically variable environments. This adaptation did not make those two species nor the vegetation type immune from the effects of lake-level change, as may be the case for floating mats at the Lake Superior sites, but it likely reduced variability and increased reproducibility for the metric *Percent Wetland in Sedge Vegetation Type.*

Limitations of Biological Disturbance Indicators for Great Lakes Wetlands

Our mixed results for potential metrics among wetland types in different lakes point to limitations in IBI development for Great Lakes wetlands. For several reasons, they also suggest caution in general use of biological indicators of wetland degradation in the Great Lakes.

Study-Site Limitations. Different lake-level histories require that wetlands of each lake be evaluated separately (lakes Michigan and Huron are one lake in this respect). Differences in exposure to wave attack, differences in sediment transport and deposition, and differences in localized hydrology, such as presence of tributary streams, require that wetlands of different geomorphic types (ILERSB 1981, Maynard and Wilcox 1997) be evaluated separately (Brinson 1993, Brinson and Rheinhardt 1996, Karr and Chu 1997, 1999). Even wetlands of the same type in the same lake may require separate evaluation based on latitudinal differences. For example, despite separation of the Pentwater River wetland and Betsie River wetland to the north (Figure 2) by only 95 km, some caddisfly species reached the limits of their ranges across that latitudinal gradient. In addition, cattail marsh rather than sedge/grass meadow is the dominant emergent vegetation type in the numerous drowned-river-mouth wetlands that extend 180 km further south from the Pentwater River. The southerly wetlands would require evaluation by standards different from those for the sites we studied.

The restrictions described above limit the number of Great Lakes wetlands in any group that can be used to develop and test IBI metrics or other standards for evaluating biological indicators. Karr (pers. comm.) recommends that multiple sites across a gradient of human disturbance be evaluated during development of IBI metrics, including multiple reference sites. Suggested follow-up field testing of the metrics requires yet additional sites spanning a gradient of disturbance. The pools of comparable sites with the same geomorphic setting in the Great Lakes are limited in size; few of them can be described as portraying reference conditions. Thus, our data collection faced restrictions both in numbers of sites and in availability of reference conditions. Such would be the case throughout much of the Great Lakes.

Hydrologic Limitations. Even if sufficient numbers of sites and reference wetlands were available for developing and testing an IBI for Great Lakes wetlands according to Karr's recommendations, future use of the IBI would be limited to reevaluation of those same sites because they define the entire pool of comparable wetlands. However, repeat sampling in years with different histories of past lake levels (i.e., number of years since last high or low lake level) would not yield similar results, even if the level of human disturbance remained constant, because plant communities and the habitat they provide respond dramatically to lake-level changes (Wilcox and Meeker 1991, 1992, 1995, Wilcox et al. 1993, Wilcox and Whillans 1999); the scoring ranges for metrics would thus be invalid. A valid IBI could be developed only if separate scoring ranges were derived for each of several water-level histories (e.g., extreme high water year, low water year following extreme high water year, mid-level year three years after high/low water sequence, sixth year after high/low water sequence, twelfth year after high/low water sequence with intervening mid-range high water year, etc.). The variety of possible sequences of lake levels makes development of such a series of scoring ranges impractical. Given the apparent quasi-periodic, 33-yr lake-level behavior described by Thompson and Baedke (1997) and Baedke and Thompson (2000) for lakes Michigan and Huron over the past 3000 years, at least thirty years of sampling might be required to develop an appropriate series of scoring ranges. Such a complicated system would likely attract few users.

Disturbance-Type Limitations. Unlike streams, which transport water-quality characteristics downstream in a cumulative fashion, wetlands or portions of wetlands often lack directional flow of water. Impacts of water-quality degradation are thus more likely to be localized than in streams. A sampling program that failed to collect data within the affected area around a local disturbance may produce erroneous or misleading results because water flow might not direct the impacts to adjacent areas that were sampled. The various types of disturbance to wetlands also differ greatly in their effects on biological organisms (e.g., chemical contamination vs. ditching); the array of metrics must be capable of detecting each disturbance type. However, sums of metric scores in the overall IBI may not reflect even strong negative impacts of a single type of disturbance affecting only one group of organisms because only one metric in the IBI might be sensitive to it. One option that could be considered for wetland types subject to multiple and varied disturbances would be to divide individual wetlands into smaller segments that could be evaluated separately. However, that approach has inherent problems in determining where to place the dividing lines because, unlike streams, many of the wetlands have irregular, non-linear shapes.

Potential for Misuse or Abuse. Karr et al. (1986) recognized the potential for an IBI to be misapplied, and they noted several cautions that extend beyond proper sampling design and methodology. 1) Management decisions based on an IBI must be made with the guidance of a biologist familiar with the organisms and habitat. 2) IBI results require human interpretation to avoid drawing erroneous conclusions based solely on numerical calculations provided by a computer. 3) Management actions should be made at the watershed level rather than organism level to attain long-term success. 4) Management decisions should be based on biotic integrity class rather than specific IBI scores.

Given the inherent variability in plant community data from Great Lakes wetlands resulting from natural lake-level variation (potentially changing from emergent marsh to no emergent plants to a different emergent community in successive years) and the resultant effects on fish and invertebrate community data, we recognize the potential for management errors and intentional abuses associated with attempts to use IBI results such as those presented in this paper. 1) In repeat sampling of a specific wetland, the total IBI score and resultant wetland biotic integrity class may decrease due to physical habitat changes associated with recent lake-level history (Wilcox et al. 1993, Wilcox and Meeker 1995, Kowalski and Wilcox 1999). Without proper interpretation and guidance, the IBI results may cause a manager to undertake unnecessary remediation actions, such as dike construction, harvest or other controls on undesired species, or stocking/replanting of desired species, at public expense and to the overall detriment of the wetland. 2) A developer may choose to conduct an IBI evaluation of a wetland in a year when metric scores, total IBI score, and biotic integrity class will be low due to recent lake-level history. With information in hand that depicts the wetland as having little biological value, the developer may succeed in obtaining a permit to develop the site. In both of these examples, the IBI would have been used inappropriately. In any setting, an IBI or other biological indicator should not be considered a quick solution that precludes the need for a manager to understand the resource.

Potentials for Use of Biological Disturbance Indicators for Great Lakes Wetlands

In meeting the interests of managers and others, under what circumstances might biological indicators be used in Great Lakes wetlands? Karr (1991) described appropriate uses of IBI scores to 1) evaluate current conditions at a site, 2) determine trends over time, 3) compare sites sampled simultaneously, and 4) if possible, identify causes of local degradation, as well as to 5) track the results of management actions (J. Karr, pers. comm.) We contend that lake-level variations restrict use of biological indicators in the form of an IBI to determine trends over time to comparisons made on sampling data collected during periods with similar water-level histories. The preliminary invertebrate-based IBI for Lake Huron wetlands developed by Burton et al. (1999) shows promise and may prove useful when calibrated for all of the various water-level-history options. It has been tested during low lake levels but must still be tested during or after years with extremely high water levels that eliminate most if not all emergent vegetation in which they sampled. Multiple sites might be compared simultaneously if the type of human disturbance is not localized or is similar among sites. Evaluating current conditions and identifying causes of degradation are clearly possible uses for biological indicators if baseline expectations can be identified—a process that inherently involves comparing multiple sites simultaneously. However, the results of our evaluation of a multitude of attributes for metric development, as well as our experiences in Great Lakes wetlands, suggest that in-depth ecological interpretation of changes in collected data at individual sites over time may be a more reliable approach than calculation of IBI scores for assessing wetland degradation because human disturbances to these wetlands are often of different types and are also localized.

Effective ecological interpretation of biological indicators requires collection of adequate data at a scale that is also reasonable from the perspective of a field biologist. To assist in achieving that goal, the U.S. Environmental Protection Agency funded development of a monitoring program for Great Lakes coastal wetlands in mid-2000 that made substantial use of the results presented here and added bird, amphibian (MMP 1997), and abiotic indicators (GLC 2000). The field methods we employed consumed five days per study site. Modifications to the sampling regime that focus more closely on the indicators that proved useful could reduce the field effort to as few as three days per site without loss of critical ecological information. A consortium of personnel from federal, state, and provincial agencies from the U.S. and Canada, as well as non-governmental organizations and universities, has evaluated a suite of potential indicators, established preliminary protocols for uniform data collection, will test those protocols in pilot field studies, and make subsequent recommendations to modify the protocols as necessary (http://www.glc.org/monitoring/wetlands/).

Implications for Other Wetland IBIs

General Concerns Regarding the IBI Approach. Broad evidence suggests that a fish IBI is a legitimate, useful tool for evaluating the quality of streams and certain other aquatic systems (e.g., Karr 1981, 1991, Karr et al. 1986, Steedman 1988, Minns et al. 1994, Deegan et al. 1997). Efforts are also under way to develop IBIs for other groups of organisms (e.g., DeShon 1995, Rosen 1995, Fore et al. 1996). Advantages of the IBI approach summarized by Karr (1991) include its quantitative nature, use of reference sites, integration of temporal and spatial dynamics, no loss of information when metric scores are added, and incorporation of professional judgment. The IBI approach is not without criticism, however.

Suter (1993) detailed a number of problems with the IBI concept, although Simon and Lyons (1995) sought to refute them, and Karr and Chu (1999) also addressed several of them. One of Suter's criticisms was *post hoc* justification of indices—the health of an ecosystem is poor because the index score is low and the score is low because it should, by definition, be low for unhealthy ecosystems. Karr and Chu (1999) dismissed concern over the circular nature of this problem by noting that disturbance gradients are determined *a priori*, while Simon and Lyons (1995) stated that not only are metric responses predicted *a priori*, they are justified if actual data show clear patterns. We interpret this concern of Suter in a different manner.

In IBI development, metric responses are indeed predicted *a priori*, but the process of including a metric in an IBI is an *a posteriori* decision dependent on both the data collected and the independent ranking of relative disturbance to the sites. There is no assurance that the independent ranking adequately assesses the type of disturbance causing biological change. In many cases, especially in wetlands, the disturbance and resultant metric response occur at a local scale. Although assessment data collected in the IBI process are meant to reflect that response at both local and watershed scales, the gradients upon which IBIs are built are often defined at the watershed scale. An independent ranking of sites according to a measure such as percent of watershed that is forested or covered by impervious surfaces may not reflect accurately each of the variety of impacts of the individual disturbance factors that are present; secondary factors may cause the biological response (Karr and Chu 2000).

If ranking of sites can differ depending on the type of disturbance, some potential metrics with obvious ecological meaning may be excluded, and others with lesser meaning may be included in IBI development unless multiple gradients are used. If not handled properly, this could be construed as unintentional preselection based on the ranking system selected. The problem can be addressed by basing disturbance gradients for individual metrics on different local or watershed characteristics that are appropriate for the type of disturbance, as we attempted with our Lake Michigan sites. However, the rank order of sites could then differ for various metrics and complicate the IBI process. In any case, as Karr and Chu (2000) note, IBI results should be inspected with the intent of explaining outlier data.

With respect to wetlands, we believe that there are a variety of disturbance types at both the localized and watershed level that can impact different parts of the biological community in different ways. The IBI-development process (Karr 1981, 1991, Karr et al. 1986) dictates that the only metrics included are those that meet the expectations of the independent measure. Although Karr (1991) recognized the potential need to modify, adapt, or replace IBI metrics under some circumstances if the changes were supported by field data, we support the approach that ecological interpretation should dictate that certain obvious metrics (such as *Percent Sensitive Species* (fish) or *Percent Wetland in Invasive Vegetation Types*) be included in an IBI. If they fail to produce meaningful dose-response curves, then the gradient used to generate the curves should be reevaluated.

Overriding Influence of Hydrology and Extreme Disturbance Events in Wetlands. Karr and others (Karr et al. 1986, Karr 1991) acknowledged that natural hydrologic variability due to climate can influence the outcome of IBI analyses and that such natural variability has not been assessed adequately. Our effort is one attempt to address that concern. However, unlike most streams for which IBI development has been successful, plant communities provide much of the habitat in wetlands, and plant communities can change dramatically in response to hydrologic change. Water-level changes in the Great Lakes are a direct result of climatic influence (Fraser et al. 1990). Our results, observations, and previous work (Wilcox et al. 1993, Wilcox 1995, Wilcox and Meeker 1995, Kowalski and Wilcox 1999) indicate that the natural variability in biological communities introduced by water-level change restricts development of an IBI for Great Lakes wetlands to strict qualifying hydrologic conditions. However, this conclusion should also extend to other wetlands with wide hydrologic variability. Because wetlands supplied by surface water generally have greater variability in water levels than those receiving a relatively constant supply of ground water (Winter 2000), they are least likely to be candidates for development of a meaningful IBI.

A good example of wetlands that are subject to extreme surface-water hydrologic variability is the prairie potholes of the United States and Canada. Temperature extremes, isolated thunderstorms, and evaporation caused by strong winds can change surface-water hydrology conditions quickly and frequently (Euliss et al. 1999). Ground-water hydrology can be very complex also, with individual wetlands within close proximity to each other serving discharge, recharge, or flow-through functions (Winter 1989). Interactions between ground water and surface water are likewise complex (Winter and Rosenberry 1995a, b), and during extreme droughts, even discharge wetlands can dry out as a result of excessive evapotranspiration (LaBaugh et al. 1996). Wet and dry climatic cycles lasting 10 to 20 years (Duvick and Blasing 1981, Karl and Riebsame 1984) can cause wetlands to be flooded or remain completely dry for extended periods of time (Euliss et al. 1999). Wetland vegetation responds to these cycles in dramatic fashion (van der Valk and Davis 1978). Invertebrate communities respond not only to the changes in habitat provided by wetland plants but also to salinity changes resulting from differential precipitation, evaporation, and ground-water supply (LaBaugh et al. 1996, Euliss et al. 1999). A functional IBI for prairie pothole wetlands may be possible; however, we suggest that these wetlands would have to be segregated based on their ground-water discharge, recharge, or flow-through function. Similar to Great Lakes wetlands, an IBI would also require differing scales of measurement for years with different water-level histories to reflect the resultant changes in plant communities and the habitat they provide.

There are several additional examples of wetlands for which IBI development might be restricted due to natural variability in habitat provided by plant communities. The biota of wetlands on the floodplains of rivers, including large, regulated rivers of midwest North America (Spink and Rogers 1996, Galat et al. 1998, Sparks and Spink 1998, Sparks et al. 1998, Yin 1998), rivers of semi-arid western North America (Friedman et al. 1996, Scott et al. 1997, Stromberg et al. 1997, Auble and Scott 1998, Osterkamp 1998, Rood et al. 1998), bottomland hardwood forests of the southeastern United States (Shelford 1954, Pride et al. 1966, Brinson et al. 1981, Clark and Benforado 1981, Wharton et al. 1982), and tidal freshwater marshes (Odum et al. 1984), may be more responsive to periodic catastrophic events such as major floods or ice floes (Brinson et al. 1981) and the short-, medium-, and/or long-term effects of the flood pulse (Junk et al. 1989, Middleton 2002) than they are to human disturbance. Therefore, development of a functional IBI for riparian wetlands might also require differing scales of measurement for years that differ in the length of time since the last major flood.

Wetlands near the ocean coast, including riparian wetlands (Hook et al. 1991, Putz and Sharitz 1991, Guntenspergen and Vairin 1996, Michener et al. 1997, Ramsey et al. 1997), barrier island wetlands (Guntenspergen and Vairin 1996), saltwater, brackish, and freshwater marshes (Chabreck and Palmisano 1973, Ramsey et al. 1994, Guntenspergen et al. 1995, Jackson et al. 1995, Guntenspergen and Vairin 1996, Michener et al. 1997), mangrove swamps (Craighead and Gilbert 1962, Lugo and Snedaker 1974, Roth 1992, Smith et al. 1994, Wanless et al. 1994, Doyle et al. 1995, Michener et al. 1997), and the extensive wetlands of the Everglades in Florida (Craighead and Gilbert 1962, Duever et al. 1994, Gunderson 1994, Loope et al. 1994, Roman et al. 1994), may also be affected in catastrophic fashion by winds and salinity increases related to hurricanes. A greater natural hydrologic disturbance that can effect major changes in Everglades plant communities is periodic drought and resulting natural fires (Loveless 1959, Duever et al. 1994, Gunderson 1994, Gunderson and Snyder 1994). Each of these natural disturbances or various combinations of them in different sequences and over different time intervals could produce substantially different results if data were collected identically in different years, even if there was no change in human disturbance. So, again, we caution that an IBI should factor in the length of time since the last disturbance by formulating different measurement scales. If multiple natural disturbances occur, the complexity of such a system could make it impractical.

Certain other wetland types might be better candidates for less-complicated IBIs. Deepwater swamps not prone to flooding from rivers, inland marshes supplied by ground water, vernal pools not subject to wide variability in hydroperiod, and tidal marshes in regions not threatened by periodic hurricanes are potential candidates for IBI development. Even wetlands on regulated reservoirs could be assessed by an IBI if regulation was consistent from year to year. Other candidate wetlands include wet meadows or wet prairies in the proper setting and some expansive peatlands. However, fens often differ in ground-water source, resultant water chemistry, and associated vegetation (Boelter and Verry 1977, Vitt and Chee 1990, Gorham and Janssens 1992, Vitt et al. 1995), so a fen IBI might require segregation of wetlands based on inherent water chemistry differences. Water levels in ombrotrophic bogs are dictated by precipitation and evapotranspiration; therefore, climatic cycles that result in extended, naturally occurring drought or flood conditions could alter vegetation if the bog mat is grounded (Schwintzer 1978, Kratz and DeWitt 1986), again invoking the hydrologic restriction to IBI development.

Wetlands vs. Streams—Conclusions on IBI Development

The discussion above does not include all wetland types in all settings. However, it serves to make the point that an IBI for use in wetlands may be less straightforward than the fish or invertebrate IBIs for streams presented by others. Although fluvial systems from streams to rivers are subject to extreme natural disturbances such as floods, droughts, and large storms, much of the habitat for fish and invertebrates is in the form of abiotic cobble, rocks, etc. That habitat may be rearranged by extreme natural disturbance but generally is not destroyed; thus, an IBI may be useful in characterizing quality of the aquatic environment (Karr 1981, 1991, Karr and Dudley 1981, Karr et al. 1986) without the complications caused by drastic alterations in habitat. Many plant communities in wetlands, including those in fluvial systems, are altered substantially by extreme natural disturbances, especially those involving or related to hydrology. Periodically, those disturbance events can produce major alterations of plant communities during short-to-long time spans. Successional processes then continue to induce changes, oftentimes until the occurrence of the next extreme disturbance event. Thus, plant communities in hydrologically variable wetlands generally are not stable through time, even in reference sites. In addition, the localized nature of other natural disturbances can also result in differences in disturbance impacts to plant communities among wetlands of the same type within the same region. Therefore, we conclude that an IBI based on plant communities would be reasonable only for wetlands with relatively stable hydrology and lacking other recurring major natural disturbances unless scoring ranges were recalibrated and specific metrics selected that reflected the dramatic response to these natural events.

Unlike most streams, much of the habitat for fish and invertebrate communities in wetlands is provided by the complex structural character of the plant communities, which can change through time without change in the level of human-induced disturbance. Although fish and invertebrates are mobile and may move to preferred habitat provided by specific plant communities that have been reduced in size, moved, or rearranged by a major disturbance event, sampling only in those areas introduces bias and does not provide a true measure of the character of the fish or invertebrate community of the wetland as a whole. Therefore, we also conclude that a wetland IBI based on fish or invertebrate communities would be subject to the same limitations as an IBI based on wetland plants. A site-specific, detailed ecological analysis of biological indicators may indeed be of value in determining the quality or status of wetlands, but we recommend that IBI scores not be used unless the scoring ranges are calibrated for the specific hydrologic history pre-dating any sampling year.

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