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Testing a Fish Index of Biotic Integrity for Responses to Different Stressors in Great Lakes Coastal Wetlands

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ABSTRACT. *Fish community composition often varies across ecoregions and hydrogeomorphic types within ecoregions. We evaluated two indices of biotic integrity (IBIs) developed for fish in Great Lakes coastal wetlands dominated (> 50% cover) by* Typha *(cattail) and* Schoenoplectus *(formerly* Scirpus*) (bulrush) vegetation. Thirty-three coastal wetlands dominated by either* Typha *or* Schoenoplectus *vegetation were sampled using fyke nets set overnight. These sites were selected to span anthropogenic disturbance gradients based on population density, road density, urban development, point-source pollution, and agricultural inputs (nutrients, sediments), measured using a GIS-based analysis of Great Lakes coastal land use. Sites subject to low levels of anthropogenic influence had high IBI scores. The* Typha*-specific IBI showed a marginally significant negative correlation with population density and residential development ($r = -0.54$, $p < 0.05$; $n = 19$). The* Schoenoplectus*-specific IBI negatively correlated most strongly with nutrient and chemical inputs associated with agricultural activity and point-source pollution ($r = -0.66$ and -0.53, respectively; $p < 0.01$; $n = 30$). However, some relationships between IBI and disturbance scores were non-linear and likely exhibit a threshold relationship, particularly for* Schoenoplectus *dominant sites. Once a certain level of disturbance has been exceeded, a sharp change in fish community's composition and function occurs which is symptomatic of a degraded site. The IBI indices appear to indicate effects of some, but not all classes of anthropogenic disturbance on fish communities.*

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Calibrating these measures against specific stress gradients allows one to interpret the sources of impairment, and thereby use the measures beyond a simple identification of impaired sites.

INDEX WORDS: *Great Lakes, coastal wetland,* Typha, Schoenoplectus, *fish IBI, anthropogenic stressors.*

INTRODUCTION

Wetlands are integral links between lakes, rivers, and surrounding land areas and provide unique habitats for birds, amphibians, fish, and benthic invertebrates. Despite their importance, wetlands continue to be lost at an alarming rate. In the U.S., 60,000 acres (25,000 ha) of wetlands are lost each year (U.S. Environmental Protection Agency [U.S. EPA] 2004). Almost two-thirds of southern Ontario Canada wetlands have been lost or severely degraded as a result of agricultural run off, development, toxic inputs, non-indigenous species invasion, and/or water level regulation (Environment Canada 2002). Measuring the biotic integrity of Great Lakes wetlands has been a primary focus of many researchers and managers since the Great Lakes Water Quality Agreement in 1972 and the subsequent partnership between U.S. EPA and Environment Canada through the biennial State of the Lake Ecosystem Conference (SOLEC) meetings. However, it is difficult to attribute degradation of natural wetland ecosystems to specific causes because many different stresses impinge on the systems simultaneously.

The multimetric index of biotic integrity (IBI) approach has been widely used to assess the ecological condition of streams across the U.S. (Karr 1981, Karr *et al*. 1986, Fausch *et al*. 1984, Fausch and Lyons 1990, Leonard and Orth 1986, Mundahl and Simon 1999). An important step in the application and development of environmental indicators is to test the index with an independently collected dataset to see if the index consistently and correctly classifies the biological condition of sites along one or more gradients of human disturbance (Simon and Lyons 1995). To test the IBI, the attributes of the new sites (e.g., ecoregion of origin, stream order) must also fall within the range of the reference sites used to develop the index.

Several studies have developed and applied fish IBIs to regions of the Great Lakes. Minns *et al*. (1994) developed an IBI for three Great Lakes areas of concern (AOCs). Their IBI demonstrated that fish assemblages are affected by water quality and macrophyte cover. The components of the fish assemblages especially affected by human disturbance were increases in nonindigenous species and reduced piscivore abundance. In a study quantifying biotic and abiotic factors affecting fish assemblages in Green Bay of Lake Michigan, Brazner and Beals (1997) noted that macrophyte coverage was among the most influential factor structuring fish communities. They reported that increased fish species richness and abundance were often correlated with increased macrophyte species richness and density.

Using correspondence analysis and non-metric multidimensional scaling, Uzarski *et al.* (2005) determined that fish community composition in Great Lakes wetlands could be predicted more effectively by stratifying with respect to plant zones rather than by ecoregion, Great Lake, or wetland hydrogeomorphic type (but see Brazner *et al.* 2007). Because vegetation cover can change in response to natural water level fluctuations, fish community composition may change independently of anthropogenic disturbance at a site, thus confounding interpretations of the IBI applied in coastal wetlands. Wilcox *et al*. (2002) evaluated the use of fish IBI for wetlands of Lakes Superior, Michigan, and Huron, but concluded that lagged responses to inter-annual water level fluctuations would result in changes to wetland vegetation species composition that could invalidate their fish IBI scores. In response, Uzarski *et al*. (2005) developed separate fish IBIs for two major macrophyte zones, *Typha*-dominated areas and *Schoenoplectus* (formerly *Scirpus*)-dominated areas in Great Lakes wetlands.

An ecoregion-specific fish IBI was derived for the northern Great Lakes (NGL) ecoregion (Bhagat 2005). However, a fish IBI for the wetlands in the Lake Erie and Ontario plains ecoregion was not feasible because too few of the IBI metrics varied meaningfully across disturbance gradients. The absence of minimally-disturbed reference sites in the lower Lake Erie basin may have contributed greatly to this result (Bhagat 2005). However, the data collected during that study were suitable to evaluate plant-zone based IBIs developed by Uzarski *et al.* (2005).

Most researchers have used agriculture and other altered land uses as the primary disturbances affecting fish communities (Brazner and Beals 1997, Crosbie and Chow-Fraser 1999). However, land use is not the only disturbance affecting the Great Lakes basin. The recent Great Lakes Environmental Indicators program (GLEI) quantified disturbance gradients consisting of six classes of anthropogenic disturbance across the U.S. Great Lakes coastline (Danz *et al*. 2005). Using GLEI's stratified-random sites allowed us to evaluate changes in the composition of fish assemblages and IBI scores across these disturbance gradients. We arrayed data from two published fish IBIs (Uzarski *et al.* 2005) across the GLEI disturbance gradients to determine specific anthropogenic disturbances that most strongly influence IBI scores. Identifying which specific stressors produce the greatest biological responses allows the indicators to infer not only condition, but where condition is poor, to diagnose potential causes of impairment (Danz *et al.* 2005, Danz *et al.* 2007). We examined the performance of Uzarski *et al*.'s (2005) fish IBIs for *Typha* and *Schoenoplectus* across the broad suite of GLEI anthropogenic stressors. Further, we applied Uzarski *et al*.'s (2005) fish IBIs to the GLEI fish assembled data. Lastly, we assessed the overall fish community response to various anthropogenic stressors.

METHODS

Site Selection

As part of a larger GLEI effort to identify indicators of condition along the Great Lakes coastal margin (Niemi *et al*. 2006), Great Lakes coastal wetland sites were selected based on a stratified random design that included a balanced effort across five Great Lakes, three hydrogeomorphic wetland types, and six anthropogenic disturbance gradients (Danz *et al*. 2005, Danz *et al*. 2007). To account for hydrogeomorphic differences (i.e., the influence of watershed versus lake), coastal wetlands were classified as lacustrine, protected, or river-influenced (Keough *et al.* 1999). Six categories of human disturbance and one category of environmental variation (soils) were measured across the U.S. side of the Great Lakes and summarized at the segment-shed scale (Danz *et al*. 2005). A segment-shed is defined as the coastal land and the adjoining drainage of a second-order or higher tributary stream. The coastal land area attributed to a segment-shed extends from the river mouth to the mid-point between the stream and the adjacent second-order or higher stream mouths on either side (Danz *et al*. 2005). The six categories of human disturbance in the GLEI study were: 1) nutrient, sediment, and chemical inputs associated with agricultural activity; 2) atmospheric deposition; 3) degree of natural land cover alteration; 4) human population density and development; 5) point source pollution; and 6) shoreline modification. Agricultural inputs largely were comprised of nitrogen (N) and phosphorus (P) inputs from fertilizers, pesticides, and suspended sediment loads. A soils gradient was also determined to reflect the variation in natural landforms across the basin. Stressors were summarized through a set of sequential principal components analyses performed on the seven sets of stressor variables to reduce the overall number of variables. Methods are described in detail by Danz *et al*. (2005). Eighty-two wetland sample sites were selected using a stratified-random procedure with geomorphic types as strata from segment-sheds clustered by disturbance type and disturbance intensity. Sites were sampled in 2002 and 2003.

Fish Sampling

Fish communities were sampled using two large (1.25-cm mesh) and two small fyke nets (0.5-cm mesh) set overnight at each wetland. Each fyke net array was placed lead-to-lead (leads parallel to shore), with the wings set at 45° angles (Brazner and Beals 1997). One set of large and one set of small nets were placed near each of the two dominant shoreline habitats at a wetland (e.g., sandy beach, vegetated bank, muddy bank, rocky shoreline, etc.). Fish assemblage composition (numbers of individuals of each species) was noted at each fyke net the next day, and the catch was standardized by net size (small versus. large) and catch per unit effort (total number of individuals/net effort based on set time). Fish of indeterminate identity were euthanized in clove oil, preserved in 2.5:1 v/v ethanol: formalin, and taken to the lab for identification. Physicochemical variables (temperature, dissolved oxygen concentration, conductivity, and pH) were measured at each net using a multi-probe meter (Yellow Springs Instruments Inc., model 556 MPS). Water clarity at each net was measured using a Secchi disk and transparency/turbidity tube (a plastic tube with a Secchi disk pattern at its base). Dominant and subdominant genera of emergent, submergent, and floating vegetation (cover and density) were also noted at each net per site, as were physical variables including depth, slope, substrate characteristics, and surrounding land use.

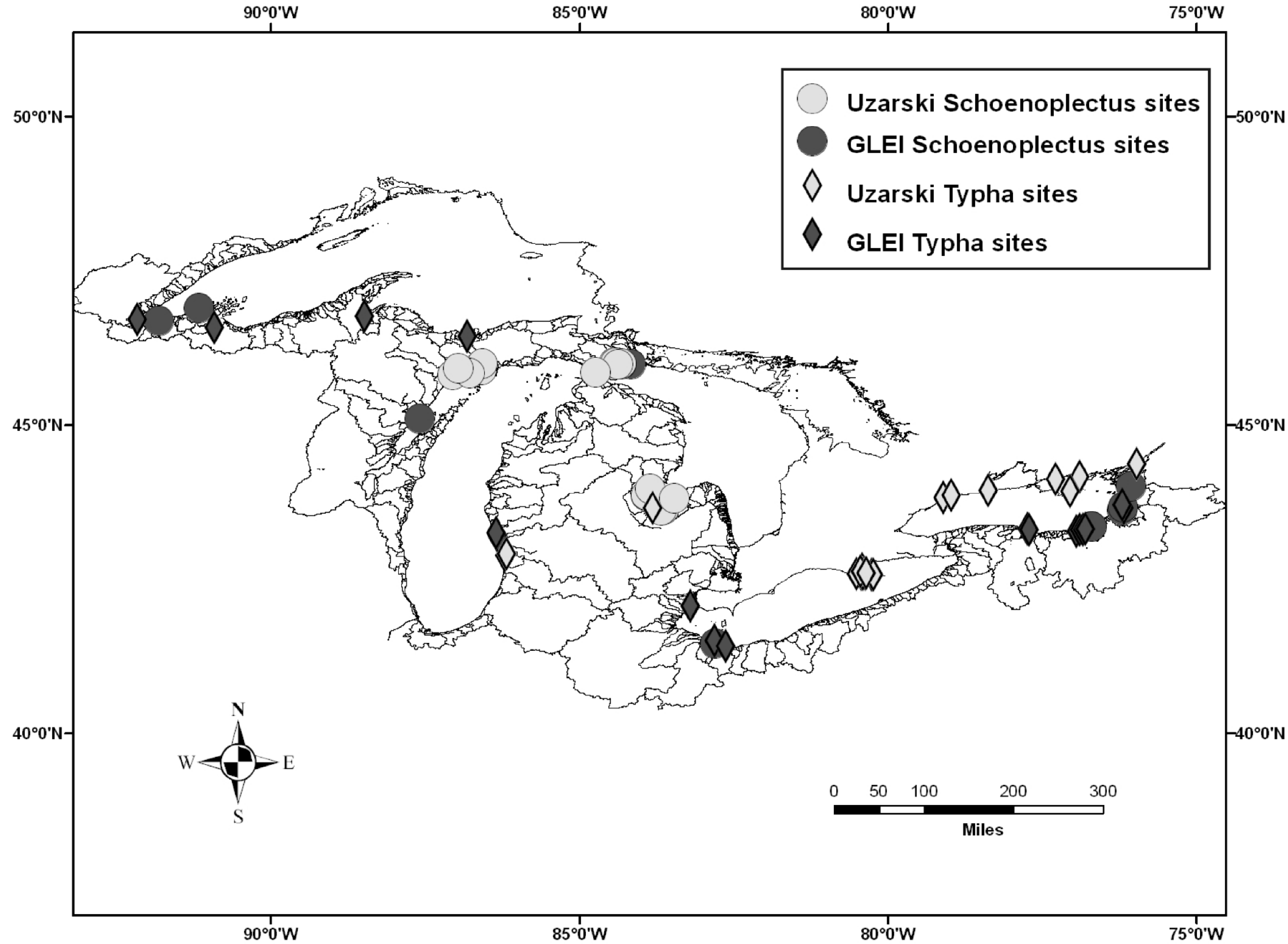

FIG. 1. Map of the Great Lakes and GLEI segment-sheds showing general locations of GLEI and Uzarski* Schoenoplectus*- and GLEI and Uzarski* Typha*-dominant sites. Symbols may represent several closely situated wetlands.

IBI Application

Thirty-six wetlands in which *Typha* or *Schoenoplectus* were the dominant macrophytes by areal cover were used to test the fish IBIs developed by Uzarski *et al.* (2005). They included sites containing several vegetation types, but nets had been set in areas visually identified as having nearly a mono-dominant vegetation type. In our study, vegetation density and cover were noted for a 10-m radius around each net. In all, 23 sites met the criterion of having dominant *Typha* cover, and 13 sites met the criterion for dominant *Schoenoplectus* vegetation across the entire U.S. Great Lakes shoreline (Fig. 1).

Typha IBI (T-IBI) and *Schoenoplectus* IBI (S-IBI) metrics were calculated for data collected from the GLEI sites using Uzarski *et al.*'s (2005) trophic guild designations for fish species. For species not listed in Uzarski *et al*. (2005), we used the trophic guild designations of the U.S. EPA (Barbour *et al.* 1999).

Spearman rank correlation coefficients were calculated to determine if there were any monotonic relationships between IBI scores and each of the six stressor variables measured at segment-shed scales. Because we expected to find negative correlations between IBI scores and anthropogenic stress scores, we used a one-tailed test of significance adjusted with a step-down modified Bonferroni correction for multiple correlation tests (Jaccard and Wan 1996).

If the Uzarski *et al.* (2005) T- IBI and S-IBI con-

TABLE 1. Site locations and IBI scores for Typha- and Schoenoplectus-dominant sites sampled through the GLEI project.

Site name	Latitude	Longitude	Plant zone	Lake	IBI score
Dwights Point	46.7	–92.17	*Typha*	Superior	20
Prentice Park	46.58	–90.91	*Typha*	Superior	19
Baraga	46.76	–88.48	*Typha*	Superior	19
Au Train	46.44	–86.82	*Typha*	Superior	18
Mona Lake	43.17	–86.3	*Typha*	Michigan	26
Muskegon Lake	43.24	–86.35	*Typha*	Michigan	25
Port Sheldon	42.88	–86.21	*Typha*	Michigan	19
Little Muddy Creek	41.5	–82.8	*Typha*	Erie	37
Plum Brook	41.42	–82.62	*Typha*	Erie	19
Cherry Island	42.06	–83.19	*Typha*	Erie	6
South Sandy Creek PW	43.71	–76.2	*Typha*	Ontario	42
South Sandy Creek RW	43.71	–76.2	*Typha*	Ontario	40
Sodus Creek	43.27	–76.93	*Typha*	Ontario	37
Mudge Creek	43.29	–76.89	*Typha*	Ontario	31
Blind Creek	43.65	–76.16	*Typha*	Ontario	29
Braddock Bay	43.31	–77.72	*Typha*	Ontario	26
Red Creek	43.31	–76.79	*Typha*	Ontario	25
Wolcott Creek	43.3	–76.84	*Typha*	Ontario	22
Buttonwood Creek	43.3	–77.71	*Typha*	Ontario	22
Middle River	46.68	–91.82	*Schoenoplectus*	Superior	58
Clover	46.88	–91.17	*Schoenoplectus*	Superior	35
Menominee River	45.09	–87.59	*Schoenoplectus*	Michigan	42
McKay Creek CW	45.99	–84.34	*Schoenoplectus*	Huron	46
Bear Lake	45.97	–84.16	*Schoenoplectus*	Huron	46
McKay Creek RW	45.99	–84.34	*Schoenoplectus*	Huron	46
Pinconning	43.85	–83.92	*Schoenoplectus*	Huron	32
Little Pickerel Creek	41.46	–82.79	*Schoenoplectus*	Erie	36
Deer Tick Creek	43.61	–76.19	*Schoenoplectus*	Ontario	58
Skinner Creek	43.67	–76.18	*Schoenoplectus*	Ontario	46
Sterling Creek PW1	43.35	–76.68	*Schoenoplectus*	Ontario	44
Sterling Creek PW2	43.35	–76.68	*Schoenoplectus*	Ontario	31
Black River	43.99	–76.06	*Schoenoplectus*	Ontario	29

tained robust metrics that responded to all six types of anthropogenic disturbance we would expect to find a significant negative correlation between the IBI scores and the stressor scores for each of the stressor variables. To assess the relationship between GLEI stressors and the fish data collected by Uzarski *et al*. (2005), we overlaid cartographic coordinates for each site sampled on the U.S. side of the lakes by Uzarski *et al*. (2005; hereafter called "Uzarski sites") on a map of segment-sheds. Uzarski sites were given GLEI stressor scores by assigning the sample site cartographic coordinates to a corresponding segment-shed.

The combined association of S-IBI scores from Uzarski *et al.* (2005) *Schoenoplectus*-dominant sites and GLEI *Schoenoplectus*-dominant sites with the six GLEI stressor scores was also determined using Spearman rank correlation. Because the stressor information was available only for sites located on the U.S. coastline, we compared Uzarski *et al.* (2005) data with GLEI data at *Schoenoplectus*-dominant sites and two *Typha*-dominant sites. All statistical analyses and graphical interpretations were performed using Statistica® software package, version 6.0 (StatSoft Inc. 2003).

RESULTS

Fifty-three fish species (3,045 individuals) in *Typha*-dominant sites and 45 fish species (2,026 individuals) in *Schoenoplectus*-dominant sites were collected and identified at GLEI wetlands (Appendix A). Most taxa collected were common to both GLEI sites and Uzarski sites. Only two species found by Uzarski *et al*. (2005) were not encountered at any of the GLEI sites (*Notropis anogenus*

TABLE 2. Spearman correlations between* Typha *(n = 21) IBI scores and* Schoenoplectus *(n = 13) IBI scores (based on GLEI fish catches), Uzarski* Schoenoplectus *dominant sites (n = 17) and values of 6 anthropogenic stressors measured at segment-shed levels. Significant correlations (modified Bonferroni adjusted) are denoted by an asterisk (*).

Stressor	*Typha* IBI (GLEI sites)	*Schoenoplectus* IBI (GLEI sites)	*Schoenoplectus* IBI (Uzarski sites)
Agriculture	–0.07	–0.73*	–0.71*
Land cover	–0.17	0.34	–0.24
Population density	–0.54*§	–0.24	0.03
Point source discharge	–0.16	–0.68*	–0.52
Atmospheric deposition	0.65	–0.06	0.43
Shoreline modification	–0.25	–0.14	0.49

§ includes 2 *Typha*-dominated sites from Uzarski *et al.* (2005)

Forbes [pugnose shiner] and *Lepisosteus osseus* L. [longnose gar]). Thus, the scores for these two species were zero.

The most abundant species captured in both *Typha*- and *Schoenoplectus*-dominant sites were *Catostomus commersoni* Lacepède (white sucker), *Perca flavescens* Mitchill (yellow perch), *Lepomis macrochirus* Rafinesque (bluegill sunfish), and *Lepomis gibbosus* L. (pumpkinseed sunfish) (Appendix A). Some species were unique to each dominant plant zone type. For instance, *Alosa pseudoharengus* Wilson (alewife) were found only in *Typha*-dominant sites, whereas *Luxilus cornutus* Mitchill (common shiner) and *Notropis bifrenatus* Cope (bridled shiner) were restricted to *Schoenoplectus*-dominant sites (Appendix A). The T-IBI scores for *Typha*-dominant sites ranged from 6–42, with the highest possible score being 61 (Table 1). The S-IBI scores for *Schoenoplectus*-dominant sites ranged from 29–58, with the highest possible IBI score being 72 (Table 1).

Values of T-IBI scores declined as the population density and development score increased (r_s = –0.52, n = 19). That trend, however was not statistically significant (p = 0.021 > modified-Bonferroni adjusted critical α of 0.02). However, when the two *Typha*-dominant Uzarski sites on the U.S. side of the Great Lakes were included, the larger sample size increased the overall power of the test, and the correlation of the T-IBI scores with population density and development became significant (r_s = –0.54, p < 0.02, n = 21; Table 2). Scatter plots of the variables showed that values of T-IBI declined as a linear function of the population density and development but increased as atmospheric deposition scores increased (Fig. 2). (Since we applied

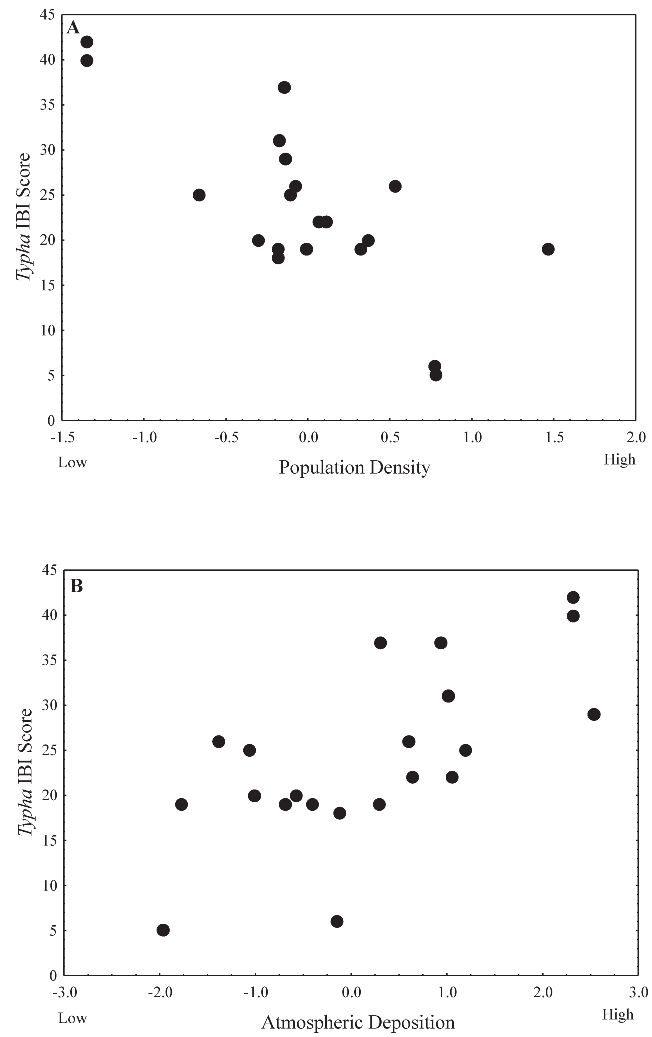

FIG. 2. Relationship between* Typha *IBI and (A) population density stress score (r_s = –0.54, p < 0.02, n = 21), and (B) atmospheric deposition (r_s = 0.65, p > 0.03, n = 21) for GLEI sites.

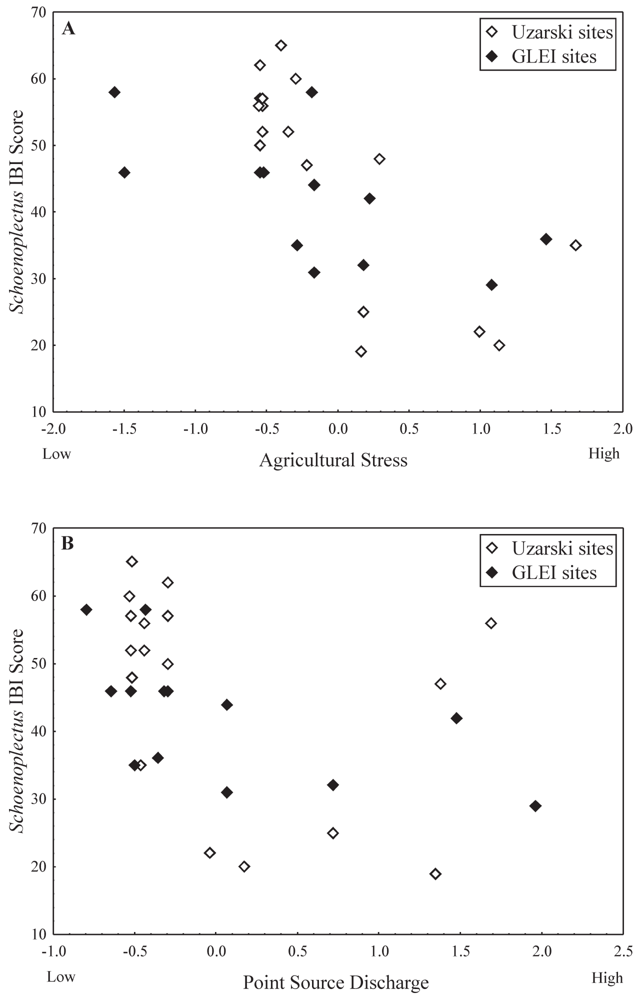

FIG. 3. Relationship between combined GLEI and Uzarski **Schoenoplectus**-***dominant sites and (A) agricultural stress (r_s = -0.66, p < 0.001, n = 30) and (B) point source pollution (r_s = –0.52, p < 0.005, n = 30).***

one-tailed tests, this strong positive correlation did not falsify the null hypothesis.)

The S-IBI was significantly negatively correlated with agriculture (r_s = –0.73, p < 0.02, n = 13) and with point-source pollution (r_s = –0.68, p < 0.02, n = 13) (Table 2). The S-IBI scores appeared to exhibit a threshold response with respect to agricultural input scores (Fig. 3A). The S-IBI score was variable but was > 43 when agricultural stress values were –0.5 or less, whereas at stress scores > 0.5, there were no IBI scores > 35. Similarly, the pattern of decreasing S-IBI scores with increasing point source discharge was more consistent with a threshold than with a linear relationship (Fig. 3B). At a point source discharge score of 0.4 or greater, there were no S-IBI scores > 45.

As a further step in the IBI analysis, the S-IBI values obtained using GLEI data were combined with those of Uzarski *et al*. (2005) for *Schoenoplectus*-dominant sites. Uzarski's S-IBI scores declined significantly with increasing agricultural inputs (r_s = –0.71, p < 0.02, n = 17) but not with increasing point-source pollution stress (r_s = –0.52, p = 0.07, n = 17) after modified Bonferroni corrections (Table 2). When we combined this data set with the GLEI data set, there was a significant negative correlation between S-IBI scores and agricultural stress (r_s = –0.66, p < 0.001, n = 30), and between S-IBI scores and point source pollution (r_s = –0.52, p < 0.005, n = 30). The pattern of variation with respect to both agricultural stress and point-source discharge was consistent between Uzarski *et al.* (2005) sites and GLEI sites (Fig. 3). However, the GLEI sites covered a broader range of the stress gradient. The combined data magnified the apparent threshold nature of the IBI score-stress gradient response.

DISCUSSION

Minns *et al.* (1994) developed a littoral fish IBI for three AOCs in Lake Ontario and Lake Huron. Their IBI exhibited low variability between sites sampled repeatedly, and significant positive correlations with submergent vegetation density at the sites sampled. Thoma (1999) then used metrics derived by Minns *et al.* (1994) and successfully modified metrics by Karr (1981) to develop a fish IBI for the nearshore sites of Lake Erie. Such efforts using fish led researchers to believe that the IBI approach could be used to assess the condition of wetland systems. However, Wilcox *et al.* (2002) concluded that a fish IBI approach was not appropriate for Great Lakes coastal wetlands because interannual water level fluctuations would obscure fish-habitat associations. Nevertheless, Uzarski *et al*. (2005) found consistent associations between fish community composition and macrophyte zones.

The *Typha*-dominant sites sampled by Uzarski *et al.* (2005) were mainly located in Lakes Erie and Ontario, which conformed with expectations, since *Typha* sp. tends to dominate emergent marshes in lower latitudes (Lougheed *et al.* 2001), as well as marshes which are nutrient enriched (Uzarski *et al.* 2005). We identified *Typha*-dominant sites along the entire Great Lakes coastline, which allowed us to test the efficacy of the T-IBI of Uzarski *et al.* (2005) across the U.S. side of the Great Lakes basin. The T-IBI scores derived from the combined data sets showed similar trends of declining IBI scores with increasing anthropogenic stress. However, the GLEI sampling program captured a broader range of overall anthropogenic stresses (especially at the reference end of the scale) than did the study of Uzarski *et al.* (2005) (Fig. 1). Furthermore, combining data sets provided a sufficiently large sample size for patterns seen in the separate surveys to be judged statistically significant.

Typha sp. can tolerate hydrological fluctuations (Galatowitsch *et al.* 2000, Waters and Shay 1990, Waters and Shay 1992) and remain abundant despite nutrient enrichment resulting from agriculture and urbanization (Galatowitsch *et al.* 1999, 2000). Janisch and Molstad (2004) compared the density of hydrophytic vegetation between disturbed and undisturbed lands. They found *Typha* to be among the dominant species in over 60% of data points in areas subject to agricultural and non-agricultural landscape modification. We expected that the fish assemblages associated with *Typha*-dominant wetlands would be tolerant of agricultural stress, which was borne out by lack of any correlation between the T-IBI and agriculture. The T-IBI did decline in relation to increasing population and development. Fish assemblages in *Typha* wetlands may be responding to loss of other emergent and/or submergent vegetation taxa. The highest T-IBI scores were primarily in Lake Ontario and Lake Erie, which typically had high agricultural stress scores and relatively lower population density and development. This suggests that *Typha* dominance in a wetland is an indicator of disturbance, and the magnitude and direction of the T-IBI might be a diagnostic indicator for this type of stress.

Atmospheric deposition, which exhibits a strong geographic gradient increasing from west to east, was positively correlated with T-IBI. The finding of an increase in IBI scores across a broad range of anthropogenic stress is counterintuitive. The positive correlation is likely attributable to the lack of concentrated input from the individual parameters making up atmospheric deposition, such as calcium, chloride, magnesium, and sulphate (Danz *et al.* 2005). Moreover, atmospheric deposition stress included measures of nitrogen input from the atmosphere directly into streams. Because *Typha* sp. remain abundant in wetlands disturbed by agricultural stress (Galatowitsch *et al.* 1999, 2000), the impact of indirect nutrient inputs from the atmosphere possibly would not adversely affect *Typha* sp. cover or the fish community composition in such wetlands.

Whereas Uzarski *Typha* sites were primarily restricted to the lower Great Lakes, their *Schoenoplectus*-dominant sites were located largely in the upper Lake Michigan and Lake Huron region. Wetlands dominated by *Schoenoplectus* are usually found in higher latitudes and in forested watersheds (Lougheed *et al.* 2001). Although Minc (1997) reported that *Schoenoplectus*, along with *Eleocharis* and *Isoetes,* are among the emergent species that are largely absent from the lower lakes, almost half (6 of 13) of our GLEI *Schoenoplectus*-dominant sites were from Lakes Erie and Ontario. Thus, we could test the applicability of the S-IBI to the wetlands of all of the Great Lakes wetlands. In contrast to the T-IBI, which did not vary across the agricultural stress gradient, the S-IBI scores were significantly negatively correlated with both agricultural inputs and point-source discharge stressors. Day *et al.* (1988) and Minc (1997) observed that species such as *Schoenoplectus americanus* and *Eleocharis smalii* are better adapted to shorter growing seasons and lower substrate fertility than are *Typha latifolia* and *Sparganium eurycarpum*, which tend to dominate highly fertile areas. The significant negative correlation between S-IBI and agricultural stress suggests that fish species in *Schoenoplectus*-dominant wetlands are most sensitive to N and P loadings. The point-source pollution stressor is a combined measure of 79 variables, including the number of facilities discharging polycyclic aromatic hydrocarbons (PAHs) into streams, as well as measures of N and P inputs from non-agricultural sources (Danz *et al.* 2005). The dominant stressors at sites exhibiting low S-IBI scores are atmospheric deposition and point-source discharge, which again indicates that fish communities in *Schoenoplectus*-dominant wetlands are sensitive to direct pollutant discharges.

The S-IBI-agricultural input stress relationship resulting from the combined dataset suggests that a threshold rather than a linear relationship best describes this fish IBI response to increases in agricultural stress. Tomal (2006) assessed various regression approaches to describe these data. He found that piecewise quantile regression, which expressed the data as two disjunct sets of data, produced a significantly lower residual sum of squares than linear, nonlinear, or piecewise linear least squares regression fits to these data. These results imply that there are levels of disturbance beyond which fish assemblages exhibit marked changes in species composition and an altered trophic structure, resulting for instance, in a decrease in the number of piscivores and insectivorous cyprinids, most often indicative of degraded sites (Karr 1981). The threshold for agricultural stress was noted at a principal components (PC) score of approximately 0.5, which corresponds to a total N input value of 244 $kg/km^2/y$ and a total phosphorus input value of 14 $kg/km^2/y$ from agricultural sources (N. Danz unpublished data). Similarly, the threshold for point-source pollution was noted at a PC score of 0.4, which translates to a total nitrogen input of 194 $kg/km^2/y$ and a total phosphorus input of 15 $kg/km^2/y$ from non-agricultural sources. These threshold values can potentially provide guidance to managers regarding the limits of pollution in a habitat, at which major and perhaps irreversible changes can be expected in the fish assemblage composition. In addition, these data provide supplementary information of value to managers seeking to understand the causes of impairment at sites that do not meet designated use criteria.

Indices of biotic integrity were designed as a tool to assess the status of a water body relative to a reference condition, and are an appropriate tool for that purpose. Diagnosing causes of water quality impairment is an important component of the Great Lakes Water Quality Agreement of the governments of Canada and the U.S., and the U.S. Federal Clean Water Act. Our results address one of the shortcomings of the IBI approach: a single value representing ecological condition does not identify the cause of impairment. Diagnosing causes of impairment requires delineation of IBI scores into component metrics and analysis of stress-component metric relationships. Since the success of mitigation and restoration efforts depends on alleviating the sources of impairment, a tool that can both assess condition and diagnose sources of stress can potentially provide cost and time savings for resource managers.

CONCLUSIONS

Uzarski *et al*. (2005) developed fish IBIs based on different plant species dominating Great Lakes coastal wetlands. They calibrated their fish IBIs against generalized anthropogenic disturbance attributed to urbanization and agriculture. Our results suggest that the Uzarski *et al*. (2005) fish IBIs for different dominant wetland plant species do respond to some of the dominant anthropogenic stressors in Great Lakes coastal wetlands. The *Typha* fish IBI shows a significant negative response to population density and development, and may provide some diagnostic capabilities for high nutrient input. The *Schoenoplectus* fish IBI appears to be low beyond a threshold level of agricultural inputs and point-source pollution. This fish IBI may also provide a diagnostic response to increased nutrient concentrations. As *Typha*- and *Schoenoplectus*-dominant wetlands are typically longitudinally separated, the fish IBIs associated with wetland type may provide relatively sensitive indicators for increasing nutrient inputs. This study emphasizes the need for carefully measured, disturbance-specific stressor data for use in an IBI for assessing biotic integrity in Great Lakes coastal wetlands.

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APPENDIX A. Trophic guild (Tr_guild) and catch per unit effort (CPUE) of all fish species collected in* Typha- *and* Schoenoplectus-*dominant wetlands of all five Great Lakes. Trophic guilds are piscivores (PISC), omnivores (OMN), insectivores (INS), and herbivores (HERB).

Common name	Species	Tr_ Guild	*Typha* Sites CPUE	*Schoenoplectus* sites CPUE
Bowfin	*Amia calva* L.	PISC	78	5
Mooneye	*Hiodon tergisus* Lesueur	INS	0.5	0
Alewife	*Alosa pseudoharengus* Wilson	INS	1.5	0
Gizzard shad	*Dorosoma cepedianum* Lesueur	OMN	18.25	0
Central stoneroller	*Campostoma anomalum* Rafinesque	HERB	0.5	0
Finescale dace	*Phoxinus neogaeus* Cope	INS	0.5	0
Longnose dace	*Rhinichthys cataractae*	INS	0.5	0
Goldfish	*Carassius auratus* L.	OMN	12.25	2
Common carp	*Cyprinus carpio* L.	OMN	7.25	38.9
Brassy minnow	*Hybognathus hankinsoni* Hubbs	OMN	0	36.7
Silvery minnow	*Hybognathus nuchalis* Agassiz	HERB	0	0.5
Creek chub	*Semotilus atromaculatus* Mitchill	INS	1	0.5
Horneyhead chub	*Nocomis biguttatus* Kirtland	INS	1	3.5
Common shiner	*Luxilus cornutus* Mitchill	INS	0	4
Blackchin shiner	*Notropis heterodon* Cope	OMN	2.5	0
Blacknose shiner	*Notropis heterolepis* Eigenmann and Eigenmann	INS	9	45
Spottail shiner	*Notropis hudsonius* Clinton	INS	12.25	5.5
Spotfin shiner	*Cyprinella spiloptera* Cope	INS	4.25	0
Sand shiner	*Notropis stramineus* Cope	INS	0	0.5
Mimic shiner	*Notropis volucellus* Cope	INS	0	98
Bluntnose minnow	*Pimephales notatus* Rafinesque	OMN	25.5	11
Fathead minnow	*Pimephales promelas* Rafinesque	OMN	1.75	7.5
Northern River shiner	*Notropis blennius* Girard	INS	0.5	0
Golden shiner	*Notemigonus crysoleucas* Mitchill	OMN	101.7	10.5
Emerald shiner	*Notropis atherinoides* Rafinesque	INS	56	9
Bridled shiner	*Notropis bifrenatus* Cope	INS	0	7
River carpsucker	*Carpiodes carpio* Rafinesque	OMN	0	6
White sucker	*Catostomus commersoni* Lacepède	OMN	254.75	662.5

APPENDIX A. (Continued).

Common name	Species	Tr_ Guild	*Typha* Sites CPUE	*Schoenoplectus* sites CPUE
Bigmouth buffalo	*Ictiobus cyprinellus* Valenciennes	INS	0.5	0
Spotted sucker	*Minytrema melanops* Rafinesque	INS	0.50	0
Silver redhorse	*Moxostoma anisurum* Rafinesque	INS	0.75	0
Shorthead redhorse	*Moxostoma macrolepidotum* Lesueur	INS	0	0.5
Yellow bullhead	*Ameiurus natalis* Lesueur	INS	4	5.1
Tadpole madtom	*Noturus gyrinus* Mitchill	INS	1	14
Brown bullhead	*Ameiurus nebulosus* Lesueur	INS	247.8	5
Black bullhead	*Ameiurus melas* Rafinesque	INS	28.5	123
Grass pickerel	*Esox americanus vermiculatus* Lesueur	PISC	1	0
Northern pike	*Esox lucius* L.	PISC	0.75	2
Muskellunge	*Esox masquinongy* Mitchill	PISC	0	3.5
Central mudminnow	*Umbra limi* Kirtland	INS	2.5	7.5
Troutperch	*Percopsis omiscomaycus* Walbaum	INS	0	4.5
Banded killifish	*Fundulus diaphanus diaphanus* Lesueur	INS	10.75	4.5
Brook stickleback	*Culaea inconstans* Kirtland	INS	0.5	0
Threespine stickleback	*Gasterosteus aculeatus aculeatus* L.	INS	46.5	0
Slimy sculpin	*Cottus cognatus* Richardson	INS	0.5	0
White perch	*Morone Americana* Gmelin	PISC	0.5	0
White bass	*Morone chrysops* Rafinesque	PISC	1	0
Northern rock bass	*Ambloplites rupestris* Rafinesque	PISC	19.5	85.7
Warmouth	*Chaenobryttus gulosus* Cuvier	PISC	6.5	0
Green sunfish	*Lepomis cyanellus* Rafinesque	INS	4.5	3.5
Pumpkinseed sunfish	*Lepomis gibbosus* L.	INS	617.1	79.75
Bluegill sunfish	*Lepomis macrochirus* Rafinesque	INS	926.7	90.75
Smallmouth bass	*Micropterus dolomieu* Lacepède	PISC	0.75	17.6
Largemouth bass	*Micropterus salmoides* Lacepède	PISC	172.7	16.75
White crappie	*Pomoxis annularis* Rafinesque	PISC	13.4	1.5
Black crappie	*Pomoxis nigromaculatus* Lesueur	PISC	24.75	4
Johnny darter	*Etheostoma nigrum* Rafinesque	INS	7.25	12.5
Eurasian ruffe	*Gymnocephalus cernuus* L.	INS	1.5	3
Yellow Perch	*Perca flavescens* Mitchill	INS	107.75	504
Logperch	*Percina caprodes* Rafinesque	INS	0	75.5
River darter	*Percina shumardi* Girard	INS	0	0.5
Walleye	*Sander vitreus* Mitchill	PISC	0	1
Round goby	*Neogobius melanostomus* Pallas	OMN	6	0
Freshwater drum	*Aplodinotus grunniens* Rafinesque	INS	0.5	0