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Application of the Wetland Fish Index to Northern Great Lakes Marshes with Emphasis on Georgian Bay Coastal Wetlands

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ABSTRACT. *The wetland fish index (WFI), a published indicator of wetland condition that ranks wetlands based on tolerance of fish species to degraded water-quality conditions, had been developed with data from 40 wetlands located exclusively in the southern portion of the Great Lakes basin (Erie, Ontario, and Michigan). No data had been included from wetlands of the northern Great Lakes (Superior and Huron) and especially those of eastern and northern Georgian Bay, where many wetlands are still unaffected by human activities. We demonstrate why application of the WFI for the lower lakes (WFI_{Lower}) can yield biased scores when applied to data for upper lakes wetlands. We then develop a basin-wide index to include data from 60 other coastal wetlands located in the northern portion of the basin, including 32 from Georgian Bay. Inclusion of northern sites in development of a basin-wide WFI (WFI_{Basin}) still produced index scores that were positively correlated with water-quality conditions as indicated by water quality index scores. We explain why use of the basin-wide WFI is better than one developed specifically for upper lakes (WFI_{Upper}). Overall, WFI_{Basin} scores were higher in the northern lakes (Superior 3.49, Georgian Bay 3.67, Huron 3.62) than in the southern lakes (Michigan 3.33, Erie 3.12, Ontario 3.09). WFI scores are only minimally affected by inter-annual variation, which allows for its use for long-term monitoring. We recommend that the WFIBasin be used when managers need to manage at a scale across the entire Great Lakes basin.*

INDEX WORDS: *Coastal wetlands, bioindicator, fish, Great Lakes, water quality.*

INTRODUCTION

Located on the interface between terrestrial and aquatic ecosystems, Great Lake coastal wetlands are productive habitats that are vital for the existence of diverse fish communities. Jude and Pappas (1992) identified more than 80 fish species that use wetlands at some time in their life-cycle and a large proportion are dependent on wetlands. In the Great Lakes, fish preferentially use wetlands over other more common but less productive habitats (e.g., bedrock, beaches; Wei *et al.* 2004). Agricultural and urban development are accompanied by high nutrient and sediment load to wetlands, leading to increased algal production and water turbidity, and ultimately leading to decreases in macrophyte abundance and diversity (Chow-Fraser 1998, McNair and Chow-Fraser 2003). There is a documented relationship between degradation of water quality and fish habitat quality, which impacts the fish community. In heavily populated areas of the Great Lakes basin, near large urban centers and agriculturally developed regions of Lakes Ontario, Erie, and Michigan, there has been accelerated loss of coastal wetland fish habitat (Seilheimer and Chow-Fraser 2006, Brazner and Beals 1997).

Ecological indices have been developed as a method for ranking habitats (e.g., streams or wetlands), for tracking change through time, or to diagnose cause of change (e.g., Dale and Beyeler 2001, Niemi and McDonald 2004, Seilheimer and Chow-Fraser 2006). Indices provide a simple method to present and analyze very complicated and complex ecological and community data. Fish, commonly used to monitor ecological integrity in streams (Karr 1981) and lakes (Drake and Valley 2005), have begun to be used in the Great Lakes (Minns *et*

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al. 1994, Seilheimer and Chow-Fraser 2006). Recent work has focused on development of indicators specifically for Great Lakes coastal wetlands using water quality (Chow-Fraser 2006), zooplankton (Lougheed and Chow-Fraser 2002), and fish (Uzarski *et al.* 2005, Seilheimer and Chow-Fraser 2006). The wetland fish index developed by Seilheimer and Chow-Fraser (2006) ranks wetlands based on species-specific tolerance to degradation. As a comparison, the index of biotic integrity (IBI) is a common type of ecological index (Karr 1981, 1991) that has also been used to quantify ecosystem health. Unlike the WFI, the IBI relies on a combination of fish community metrics (e.g., percent exotic species, number of sensitive species) that have not been derived from empirical tolerance of fish species to environmental conditions (Suter 2001). Since the WFI was originally developed with data from wetlands located in the southern Great Lakes (Michigan, Erie, and Ontario), it was primarily intended for use in southern sites, and should not be applied to northern sites without validation.

Our first objective in this study was to determine if the WFI developed in the lower Great Lakes is appropriate for use in the upper Great Lakes without the addition of fish species found in northern lakes. We modified the WFI (Seilheimer and Chow-Fraser 2006) with the inclusion of wetlands and species from the northern Great Lakes (Superior, Georgian Bay, and Huron). Second, we investigated the relationship between the WFI with water-quality conditions. For the WFI to be used as an indicator of wetland condition, it should be strongly and positively correlated with water quality (i.e., environmental condition). We also predicted that differences in the water quality between the unimpacted northern wetlands compared with those in the impacted southern sites will be reflected in WFI scores. Finally, we provided two comparisons of WFI scores: on multiple year variation in eight wetlands, and on electofishing and fyke nets for fish collection. The examples of WFI use represent common challenges that are likely to be experienced by users of the WFI and will provide guidance and a starting point for future index validation and research.

METHODS

Study Sites

We used environmental and fish data collected from 100 coastal wetlands located in all five Great Lakes, including Georgian Bay in Lake Huron. Georgian Bay will be treated as a separate waterbody because it has distinctive environmental conditions, such as geology, land use, climate (Fuller *et al.* 1995). There were a total of 60 wetlands from northern or upper Great Lakes: 15 in Superior, 32 in Georgian Bay, and 13 in Huron (Fig. 1; see inset for Georgian Bay). There were a total of 40 wetlands from the southern or lower lakes: 24 in Ontario, eight in Erie, and eight in Michigan (Fig. 1).

Field Sampling

We measured a set of important variables for fish habitat in each wetland, which included water-quality parameters, substrate type, and other physical attributes. Wetlands were visited between early June and late August during 2001–2005. Water samples were collected at least 10 m from the edge of the emergent aquatic vegetation for analysis of planktonic algae, primary nutrients, and suspended solids. To minimize contamination from benthic algae in certain wetlands where submergent vegetation was present throughout, we sampled in the deeper areas with little submergent vegetation. Water samples were collected with a 1-L Van Dorn bottle deployed at mid-depth and dispensed into clean, acid-washed Nalgene bottles for nutrient analyses and into brown 1-L bottles for chlorophyll analyses. All samples were frozen until analysis (usually within 3 months of collection) with standard APHA water quality processing methods (1992) as outlined in Chow-Fraser (2006).

Temperature, conductivity, dissolved oxygen, and turbidity were measured with a Hydrolab Minisonde multiparameter probe attached to a surveyor display (Hydrolab, Austin, TX) during 2000–2001. A YSI 6600 multiparameter probe with two optical sensors (turbidity and chlorophyll) and a YSI 650 display (YSI, Yellow Springs, OH) was used in 2002–2005. We conducted a side-by-side comparison of both the Hydrolab Minisonde and YSI 6600, and found no significant ($P > 0.05$) deviations with respect to any of the parameters, except for turbidity, which was not used in the analysis. We georeferenced the sites with a handheld global positioning system unit (4–6 m accuracy). For the fish community survey, we used three paired fyke nets (two pairs of large nets [13 and 4 mm bar mesh, 4.25 m length, 1 m × 1.25 m front opening] and one pair of small nets [4 mm bar mesh, 2.1 m length, 0.5 m × 1.0 m front opening]) set parallel to the emergent zone at the 1 and 0.5 m depth contour, respectively.

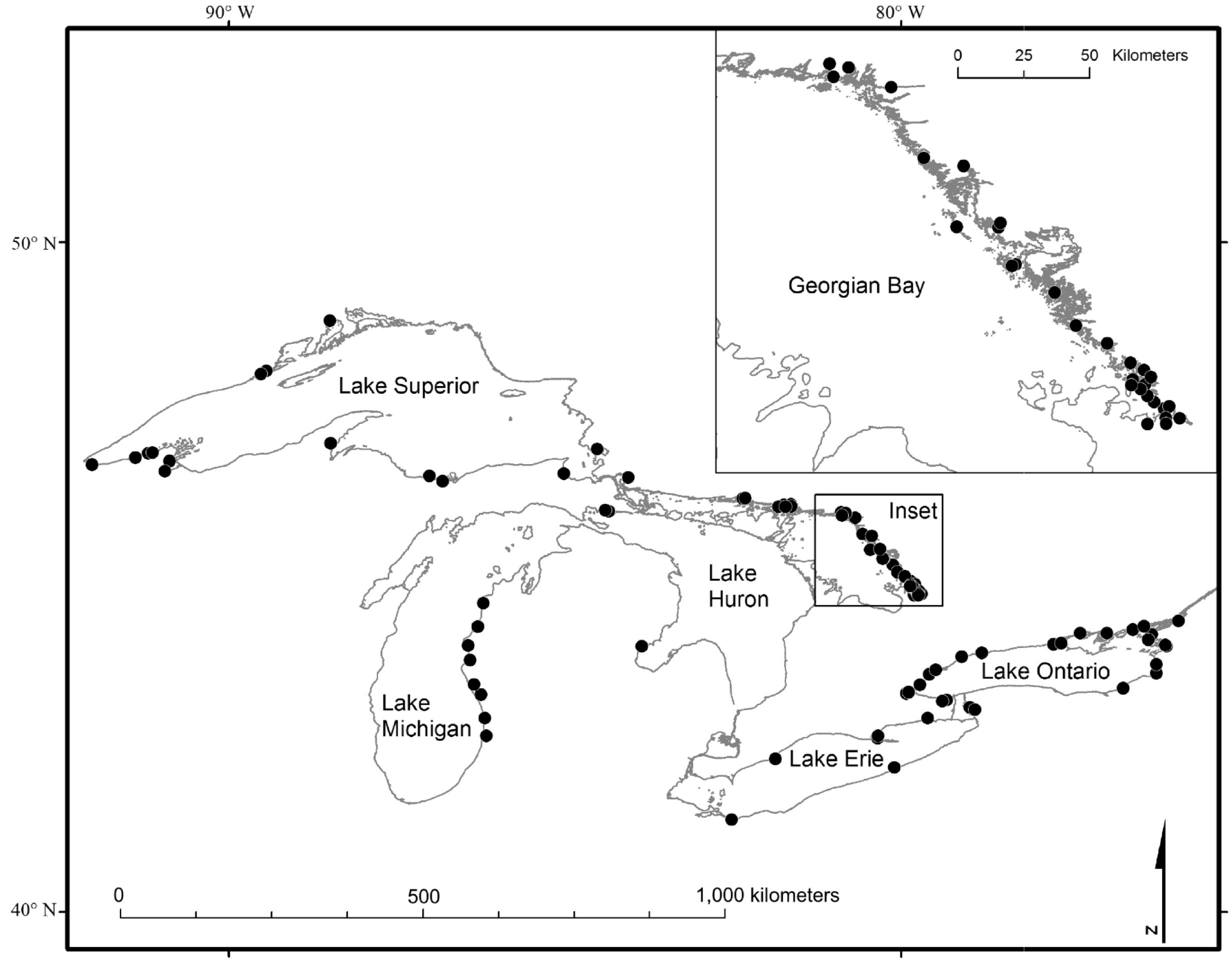

FIG. 1. Location of 100 Laurentian Great Lake wetlands that were sampled for water quality and fish from 2001 to 2005.

The paired nets were positioned face-to-face and connected with a 7 m lead, while 2.5 m wings were set off the front openings at a 45° angle. Nets were set within submergent vegetation, unless there was too little vegetation or when appropriate depths were not available, and then the nets were set near the emergent vegetation. After 24 hours, fish present in the nets were measured for total length to the nearest mm, enumerated, and identified according to Scott and Crossman (1998), and then released at the site. Fish data were pooled for the three pairs of nets at each wetland.

Parallel sampling between fyke nets and electrofishing was conducted in 31 wetlands. We used a 5 m Smith-Root (Vancouver, WA) electrofishing boat with a 7.5 kW generator powered pulsator (GPP). Fish were collected during three daytime transects that corresponded to the location of each of the three, paired fyke nets. Transects were sampled for a total of 300–400 shock seconds (i.e., total time of 1,000 to 1,200 seconds of shocking per wetland). We used direct current at a frequency of 60 hertz to produce an output of 3 to 7 amps. To standardize our fishing effort, the voltage was adjusted to maintain power output of 2000 watts. WFI scores were then calculated for the fyke nets and the electrofishing separately in each wetland and compared for differences.

Statistical Analysis

CANOCO 4.5 (ter Braak and Smilauer 1998) was used to run partial canonical correspondence analysis (pCCA) as in Seilheimer and Chow-Fraser (2006). Seasonal variation in the data set was adjusted by performing a partial CCA (pCCA), rather than a regular CCA, with day of the year as a covariable (ter Braak and Verdonschot 1995). Prior to conducting the CCA, we used the detrended correspondence analysis (DCA) to verify that the species had a unimodal distribution on an artificial gradient (i.e., length of the gradient was greater than 4.0 standard deviation units; ter Braak and Smilauer 1998). All environmental variables were log10-transformed and standardized to have zero mean and unit variance. Species abundances (pooled data from three fyke nets) were log10-transformed before analysis, and rare species (i.e., those occurring in only one wetland) were excluded from this analysis to allow for the index to focus on species that are more likely to be encountered by other investigators. Statistical significance of the canonical axes was determined with Monte Carlo permutations under the full model, which provides the maximum amount of power compared to the reduced model (500 random permutations; ter Braak and Smilauer 1998) for presence/absence (PA) and abundance (AB) data separately. All other statistical analyses (e.g., Tukey-Kramer honestly significant difference [HSD], analysis of variance [ANOVA], one-way ANOVA) were performed with SAS JMP IN (version 5.1, SAS Institute Inc., Cary, NC). Differences in water quality between wetlands were tested with t-test and Tukey-Kramer. The Tukey-Kramer test was used to examine differences between predicted and observed WFI scores. We used ANOVA to test for a significant relationship between WQI and WFI. A one-way ANOVA was used to compare WFI scores between different gear types. An alpha of 0.05 was used to determine statistical significance in all cases.

Index Development

The wetland fish index (WFI_{Lower}) was developed by Seilheimer and Chow-Fraser (2006) with environmental and fish data from 40 coastal marshes located primarily in Lakes Michigan, Erie, and Ontario. To assess the need for a new WFI that could be applied to sites in the upper Great Lakes, we first calculated observed WFI scores by using the U and T values from Table 3 in Seilheimer and Chow-Fraser (2006), and applying them to the fish community data we collected from 57 upper lakes sites. We then used the linear regression between WFI_{Lower} and WQI scores (p-value < 0.01 and a $R^2 = 0.64$, see Fig. 3a of Seilheimer and Chow-Fraser 2006):

$$WFI_{Predicted} = 3.2829 + 0.3619*WQI \qquad (1)$$

where WQI is any measured WQI score and $WFI_{Predicted}$ is a score derived from the relationship between WFI_{Lower} and WQI (Seilheimer and Chow-Fraser 2006). From this regression we were able to derive predicted WFI scores from WQI scores generated independently from corresponding environmental data for the 57 northern sites. WQI scores were calculated from 12 water-quality parameters, including primary nutrients (e.g., total phosphorus), physical (e.g., temperature), and chemical parameters (e.g., conductivity, see Chow-Fraser 2006 for details). The WQI scores ranged from –3, which is indicative of the most impacted conditions, to +3, which is indicative of the most undisturbed sites.

We also attempted to develop a separate WFI using only wetlands in the upper Great Lakes. We used multivariate statistics as outlined previously (Seilheimer and Chow-Fraser 2006) to ordinate fish in relation to environmental data (same as Seilheimer and Chow-Fraser 2006 and this study, see Table 1) for 60 wetlands in the northern Great Lakes (Lake Superior, Georgian Bay, and Lake Huron) to develop a WFI_{Upper}. The analysis is not included here because the relationship between water quality and fish occurrence did not produce any significant pCCA axes (P = 0.15). Based on these findings, we believe that development of a WFI_{Basin} would be a suitable alternative, since the underlying water quality degradation gradient for sites in the upper lakes is too weak to support development of a separate index. Additionally, we felt that an index that could be applied to all five Great Lakes would be useful for the type of cross-lake comparisons being requested by resource managers to assess the conditions of coastal wetlands across the basin (Bertram and Stadler-Salt 2000).

Development of the WFI for the entire Great Lake basin was based on observed trends in the pCCA (WFI_{Basin}; Fig. 2). This analysis was expanded to 61 fish species from the 41 used by Seilheimer and Chow-Fraser (2006) to include species that are primarily from the upper lakes (see bold entries in Table 2). It is important to note that species that were encountered in only a single wetland were excluded from the development of the WFI_{Basin}. Species that occurred in two or more wetlands would then have at least two sets of environmental conditions associated with them for the multivariate analysis. By using this cut-off, we were still able to include species that could be excluded from analy-

TABLE 1. Summary of environmental conditions (mean, median, and range) for 100 wetlands organized by Great Lake. Environmental variables include: water quality index (WQI), temperature (Temp.), conductivity (Cond.), total phosphorus (TP), soluble reactive phosphorus (SRP), chlorophyll a (Chl a), total nitrate nitrogen (TNN), total ammonia (TAN), total organic suspended solids (TOSS), and total inorganic suspended solids (TISS). P-values correspond to t-test used to compare data between southern (Ontario, Erie, and Michigan) and northern sites (Superior, Georgian Bay, and Huron).

Lake		Superior	Georgian Bay	Huron	Michigan	Erie	Ontario	P-values
Wetlands	n	15	32	13	8	8	24	
WQI	Mean	0.63	1.08	1.06	–0.41	–0.28	–0.43	< 0.01
	Min	–0.16	–0.64	–0.07	–1.36	–2.42	–2.31	
	Max	2.13	2.25	2.03	0.15	1.01	1.28	
Temp. (°C)	Mean	17.8	22.9	23.1	24.4	24.9	23.9	<0.01
	Min	12.5	14.4	21.4	20.1	21.1	14.9	
	Max	24.2	28.6	24.6	27.5	31.5	28.7	
Cond. ($\mu S \cdot cm^{-1}$)	Mean	135	155	156	382	434	463	< 0.01
	Min	56	48	69	149	224	91	
	Max	267	383	361	584	770	1658	
TP ($\mu g \cdot L^{-1}$)	Mean	38.0	24.4	30.8	80.6	74.1	112.0	< 0.01
	Min	8.8	5.4	6.7	12.4	19.3	13.3	
	Max	76.8	63.5	140.9	154.0	398.0	333.0	
SRP ($\mu g \cdot L^{-1}$)	Mean	9.4	7.1	12.4	4.7	13.7	17.6	0.063
	Min	2.6	0.1	1.0	1.4	0.5	0.7	
	Max	19.7	22.8	83.9	12.2	67.2	94.1	
Chl *a* ($\mu g \cdot L$)	Mean	4.1	2.7	2.2	17.7	18.4	14.2	< 0.01
	Min	0.1	0.4	0.5	0.5	0.8	0.3	
	Max	11.4	9.1	7.6	90.9	100.3	95.3	
TNN ($mg \cdot L^{-1}$)	Mean	0.026	0.018	0.029	0.024	0.168	0.093	< 0.01
	Min	0.001	0.001	0.001	0.001	0.001	0.001	
	Max	0.1	0.10	0.11	0.06	0.64	0.56	
TAN ($mg \cdot L^{-1}$)	Mean	0.30	0.22	0.22	0.45	0.40	0.33	0.031
	Min	0.02	0.001	0.01	0.17	0.001	0.07	
	Max	0.93	0.57	0.67	0.87	1.27	1.03	
TOSS ($mg \cdot L^{-1}$)	Mean	3.6	2.8	2.1	7.9	6.9	21.7	0.031
	Min	0.8	0.1	0.2	1.7	1.8	0.1	
	Max	6.5	16.6	9.0	14.4	24.1	331.5	
TISS ($mg \cdot L^{-1}$)	Mean	5.1	3.6	4.8	6.3	16.5	26.5	0.060
	Min	0.01	0.001	0.001	0.01	0.3	0.3	
	Max	13.7	16.1	18.5	13.6	67.2	339.0	

*WQI categories: –3 to –2, highly degraded; –2 to –1, very degraded; –1 to 0, moderately degraded; 0 to 1, good; 1 to 2, very good; 2 to 3, excellent.

ses and will provide future researchers with the opportunity to confirm their placement in the index.

The first axis of the pCCA was strongly correlated with water-quality degradation and the species and wetlands were ordinated across the degradation gradient; as a result, we used pCCA axis 1 to derive values for optimum and tolerance (hereafter referred to as U and T, respectively; ter Braak and Verdonschot 1995). A centroid is the center of a cluster of species scores and, hence, we used the placement of this centroid along the synthetic degradation axis to indicate the species' U value. Each species was assigned a weight that corresponded to its position on pCCA axis 1, where 1 indicated most tolerant to degradation and 5 was most intolerant to degradation. The weighted standard deviations of the species scores on pCCA axis 1 were used to indicate niche breadth (ter Braak and Smilauer 1998) and were used to assign the T values, where 1 indicated a wide niche breadth and 3 indicated a narrow niche breadth. Species having narrow niche breadths were indicative of specific

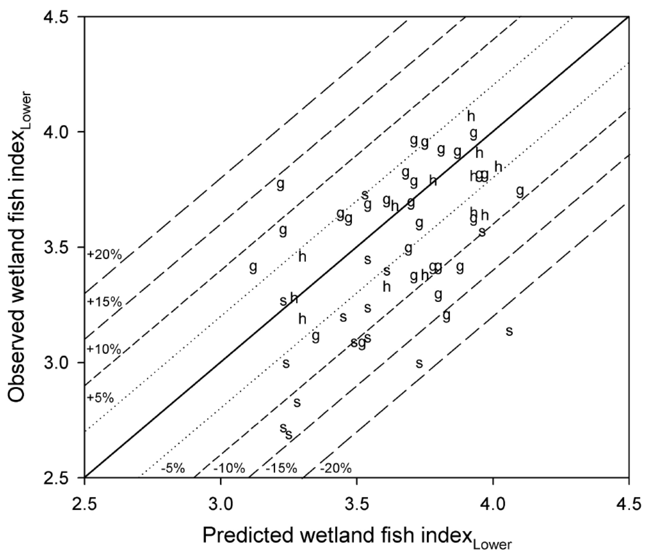

FIG. 2. Observed wetland fish index scores, calculated by applying U and T values to fish community data according to Seilheimer and Chow-Fraser (2006), vs. predicted wetland fish index scores estimated from Seilheimer and Chow-Fraser's (2006) documented relationship between wetland fish index and water quality index scores (see Chow-Fraser 2006). Data are for 57 wetlands in the northern Great Lakes: Lake Superior (s), Georgian Bay (g), and Lake Huron (h). The solid line is a line indicating observed = predicted, while the dashed lines delineate ±5%, ±10%, ±15%, and ±20% above and below the line of unity, respectively.

environmental conditions and were more useful as indicator species. To develop the WFI, each species was assigned U and T values according to the following equation (Lougheed and Chow-Fraser 2002, Seilheimer and Chow-Fraser 2006):

$$WFI = \frac{\sum_{i=1}^{n} Y_i T_i U_i}{\sum_{i=1}^{n} Y_i T_i} \tag{2}$$

where Y_i is the presence or log10 abundance (log[x + 1]) of species i, T_i is the value from one to three (indicating niche breadth), and U_i is the value from one to five (indicating tolerance of degradation). WFI_{Basin} (PA) refers to scores calculated using presence/absence data, and WFI_{Basin} (AB) refers to scores when log10 abundance (log[x + 1]) data were used. Note that only the presence of a species could contribute to the calculation of the WFI score, and that the score itself was not directly influenced by absence of a particular species. The 12-variable equation from Chow-Fraser (2006) was used to calculate WQI scores for all wetlands in this study, using data collected at the same time that fish were collected.

Exotic Species Correction

The exotic species correction for the WFI_{Basin} score is a way to adjust the WFI_{Basin} to account for wetland disturbance associated with the presence of exotic species (listed in Table 3) in the fish community. To account for the presence of exotic fish species, we subtracted the square root of the proportion of exotic species in each wetland from the corresponding WFI_{Basin} score calculated from PA data. For abundance data, the correction was equal to the square root of the proportion of exotic individuals in a wetland, and this quantity was subtracted from the WFI_{Basin} score. However, WFI scores had a minimum value of 1, even if the exotic species correction resulted in a lower value. We will refer to these modified basin-wide scores as WFI_{Basin} (PA_{ex}) and WFI_{Basin} (AB_{ex}) for the PA and AB data, respectively. This correction for proportion of exotic species was originally conceived for the wetland macrophyte index (WMI; Croft and Chow-Fraser 2007, this issue).

RESULTS

Environmental Conditions

Wetlands in this study ranged from oligotrophic to hypereutrophic based on nutrient concentrations (Table 1), and we detected broad regional differences that could be separated into northern sites (wetlands of Superior, Georgian Bay, Huron) vs. southern lakes (wetlands of Michigan, Erie, Ontario) (t-test; $P < 0.05$) (Table 1). The northern lakes were associated with higher mean WQI scores, which corresponded to an overall "good" status for Lake Superior and "very good" for Lake Huron and Georgian Bay. By comparison, mean WQI scores were classified as "moderately degraded" for each of Lakes Ontario, Erie, and Michigan, although WQI scores for several Erie and Ontario wetlands were considered in "good" condition (1.01 and 1.28, respectively; Table 1). Temperatures were coolest in Lake Superior wetlands, with a mean value of 17.8°C compared with temperatures of 22.9°C and 24.9°C for Georgian Bay and Erie wetlands (Tukey-Kramer; $P < 0.0001$). Conductivity

TABLE 2. Species occurrence and total catch for all fish species collected in 100 Great Lake coastal wetlands from 2001 to 2005, which were included in the development of the basin-wide wetland fish index (WFIBasin). Species are sorted by number of wetlands captured from species found in the most wetlands (e.g., 94 of 100 wetlands) to the species found in a single wetland.

Common name		Species occurrence	Total abundance
Pumpkinseed	*Lepomis gibbosus*	94	6,274
Rock bass	*Ambloplites rupestris*	83	855
Yellow perch	*Perca flavescens*	81	1,374
Brown bullhead	*Ameiurus nebulosus*	80	4,649
Largemouth bass	*Micropterus salmoides*	64	3,354
Bluntnose minnow	*Pimephales notatus*	56	3,396
Bowfin	*Amia calva*	49	158
Spottail shiner	*Notropis hudsonius*	39	3,774
Bluegill	*Lepomis macrochirus*	37	1,891
Northern pike	*Esox lucius*	36	75
Banded killifish	*Fundulus diaphanus*	35	585
Golden shiner	*Notemigonus crysoleucas*	34	463
Blacknose shiner	*Notropis heterolepis*	33	1,409
Tadpole madtom	*Noturus gyrinus*	31	187
Smallmouth bass	*Micropterus dolomieu*	31	183
Common carp	*Cyprinus carpio*	30	363
Johnny darter	*Etheostoma nigrum*	27	151
Black crappie	*Pomoxis nigromaculatus*	26	297
Emerald shiner	*Notropis atherinoides*	24	588
Blackchin shiner	*Notropis heterondon*	24	291
Longnose gar	*Lepisoteus osseus*	24	62
White sucker	*Catostomus commersoni*	23	1,548
Mimic shiner	*Notropis volucellus*	23	898
Common shiner	*Luxilus cornutus*	19	238
Logperch	*Percina caprodes*	14	62
White crappie	*Pomoxis annularis*	13	234
Brook silverside	*Labidesthes sicculus*	12	21
Fathead minnow	*Pimephales promelas*	11	329
Spotfin shiner	*Cyprinella spilopterus*	10	198
Longear sunfish	*Lepomis megalotis*	10	181
Alewife	*Alosa pseudoharengus*	10	154
Central mudmonnow	*Umbra limi*	10	20
Gizzard shad	*Dorosoma cepedianum*	9	454
Black bullhead	*Ameiurus melas*	9	139
Brook stickleback	*Culaea inconstans*	9	123
Trout-perch	*Percopsis omiscomaycus*	8	1,466
Threespine stickleback	*Gasterosteus aculeatus*	8	526
White perch	*Morone americana*	8	182
Ninespine stickleback	*Pungitius pungitius*	7	143
Northern redbelly dace	*Phoxinus eos*	7	120
Shorthead redhorse	*Moxostoma macrolepidotum*	7	54
Iowa darter	*Etheostoma exile*	6	41
Rainbow smelt	*Osmerus mordax*	5	147
Green sunfish	*Lepomis cyanellus*	5	140
Channel catfish	*Icalurus punctatus*	5	32
Sand shiner	*Notropis stramineus*	4	70
Mottled sculpin	*Cottus bairdi*	4	12
Brassy minnow	*Hybognathus hankinsoni*	3	35
Freshwater drum	*Aplodinotus grunniens*	3	6
Longnose sucker	*Catostomus catostomus*	3	6

TABLE 2. Continued.

Common name		Species occurrence	Total abundance
Slimy sculpin	*Cottus cognatus*	3	6
Muskellunge	*Esox masquinongy*	3	6
Walleye	*Sander vitreus*	3	4
Creek chub	*Semotilus atromaculatus*	3	4
Least darter	*Etheostoma microperca*	2	162
White bass	*Morone chrysops*	2	37
Silver redhorse	*Moxostoma anisurum*	2	14
Pearl dace	*Margariscus margarita*	2	10
Round whitefish	*Prosopium cylindraceum*	2	9
Goldfish	*Carassius auratus*	2	6
Redfin pickerel	*Esox americanus*	2	3
Total		100	38,219

Note: Eleven species were excluded from the analysis because they only occurred in a single wetland: American eel *Anguilla rostrata,* pirate perch *Aphredoderus sayanus,* lake chub *Couesius plumbeus,* lake chubsucker *Erimyzon sucetta,* round goby *Apollonia melanostomus* (exotic), whitemouth shiner *Notropis alborus,* pugnose shiner *Notropis anogenus,* rainbow trout *Oncorhynchus mykiss,* sea lamprey *Petromyzon marinus* (exotic), longnose dace *Rhinichthys cataractae,* and rudd *Scardinius erythrophthalmus* (exotic).

was higher in the southern lakes (Michigan 382 $\mu S \cdot cm^{-1}$; Erie 343 $\mu S \cdot cm^{-1}$; Ontario 463 $\mu S \cdot cm^{-1}$) compared with northern lakes (Superior 135 $\mu S \cdot cm^{-1}$; Georgian Bay 155 $\mu S \cdot cm^{-1}$; Huron 156 $\mu S \cdot cm^{-1}$). Total phosphorus, total ammonia, and total nitrate were significantly lower in the northern lakes than in southern lakes (Table 1). Generally, wetlands of the north had lower concentrations of total suspended solids, compared with those of the south (Table 1).

Application of WFI_{Lower} to Northern Sites

Given these broad differences in environmental conditions between the northern and southern wetlands, we first assessed the applicability of the WFI_{Lower} to wetlands of the upper lakes. Data from 57 sites of the upper lakes were included in a comparison of observed WFI_{Lower} scores (calculated by applying U and T values from Table 3 of Seilheimer and Chow-Fraser 2006) and predicted scores (calculated by using the relationship between WFI_{Lower} and WQI for the lower Great Lakes) (Fig. 2). Only 41% (n = 24) of the wetlands had predicted WFI scores that fell within 5% (i.e., ±0.20 units) above or below the expected WFI_{Lower} score based on water-quality conditions alone. Few wetlands scored higher than expected; only seven wetlands had observed WFI_{Lower} scores between 5 to 15% greater than expected value (i.e., 0.20 and 0.60 units), with the largest positive deviation from expected occurring with Matchedash Bay in Georgian Bay. However, the majority of the observed WFI_{Lower} scores (36 of 57 total wetlands) had lower than expected values; 40% (n = 23) scored 5 to 15% (i.e., 0.2 to 0.6 units) lower than expected, while 1% (n = 3) had scores that were as much as 15 to 25% (i.e., 0.60 to 1.0 units) below expected (Fig. 2). Observed scores associated with Lake Superior wetlands deviated most from expected scores, where mean difference between predicted and observed WFI_{Lower} scores was significantly greater (–0.35 below expected; Tukey-Kramer HSD P = 0.01) than that for Georgian Bay (–0.06 below expected). This is most likely a reflection of the number of species present in Lake Superior wetlands that had not been included in the WFI_{Lower} (an average of three species per site, as compared with one species per site for Georgian Bay, and one species per site for Lake Huron) (Tukey-Kramer HSD P < 0.01).

Fish Assemblages for Wetlands in all Five Great Lakes

Given the poor performance of the original WFI_{Lower}, and that our initial analyses revealed the environmental gradient was insufficient to develop an index specifically for upper lake wetlands, a new basin-wide index was developed from fish assem-

TABLE 3. Water-quality optimum (U) and tolerance (T) values for 61 fish species from presence-absence (PA) or abundance (AB) data. Species not included in Seilheimer and Chow-Fraser (2006) are shown in bold. * Indicates species exotic to the Great Lakes watershed, and † indicates that species is native to Lake Ontario but not the other Great Lakes (Mills* et al. *1993).

			PA		AB		
Species #	Family	Common name	U	T	U	T	Exotic
1	Amiidae	Bowfin	4	2	4	2	
2	Atherinopsidae	Brook silverside	4	2	4	2	
3	**Catostomidae**	**Longnose sucker**	**5**	**3**	**5**	**3**	
4		White sucker	3	1	3	2	
5		**Silver redhorse**	**5**	**3**	**5**	**3**	
6		**Shorthead redhorse**	**4**	**2**	**4**	**2**	
7	Centrarchidae	Rock bass	4	1	4	2	
8		Green sunfish	1	1	1	1	
9		Pumpkinseed	3	2	3	2	
10		Bluegill	3	1	3	1	
11		**Longear sunfish**	**4**	**3**	**4**	**3**	
12		Smallmouth bass	4	2	4	2	
13		Largemouth bass	3	2	3	2	
14		White crappie	1	1	1	1	
15		Black crappie	3	2	3	2	
16	Clupeidae	Alewife	2	2	1	2	*
17		Gizzard shad	1	2	1	2	
18	**Cottidae**	**Mottled sculpin**	**4**	**3**	**4**	**3**	
19		**Slimy sculpin**	**4**	**2**	**4**	**2**	
20	Cyprinidae	Goldfish	1	2	1	2	*
21		Spotfin shiner	2	1	1	1	
22		Common carp	2	1	1	1	*
23		Brassy minnow	1	2	1	2	
24		**Common shiner**	**4**	**3**	**4**	**3**	
25		**Pearl dace**	**4**	**3**	**4**	**3**	
26		Golden shiner	3	2	3	2	
27		Emerald shiner	3	2	3	2	
28		Blacknose shiner	4	2	4	2	
29		Blackchin shiner	5	3	5	3	
30		Spottail shiner	2	1	2	1	
31		**Sand shiner**	**3**	**1**	**3**	**1**	
32		Mimic shiner	5	3	5	3	
33		**Northern redbelly dace**	**5**	**3**	**5**	**3**	
34		Bluntnose minnow	3	1	4	2	
35		Fathead minnow	2	1	2	1	
36		**Creek chub**	**3**	**1**	**3**	**1**	
37	Esocidae	Redfin pickerel	4	3	4	3	
38		Northern pike	4	2	4	2	
39		Muskellunge	4	3	4	3	
40	Fundulidae	Banded killifish	4	3	4	3	
41	Gasterosteidae	Brook stickleback	4	2	4	2	
42		Threespine stickleback	2	2	2	1	†
43		**Ninespine stickleback**	**4**	**3**	**4**	**3**	
44	Ictaluridae	Black bullhead	3	2	3	2	
45		Brown bullhead	3	1	2	1	
46		Channel catfish	1	2	1	2	
47		Tadpole madtom	4	2	4	2	
48	Lepisosteidae	Longnose gar	5	3	5	3	
49	Moronidae	White perch	1	1	1	2	*
50		White bass	1	1	1	1	

TABLE 3. Continued.

			PA		AB		
Species #	Family	Common name	U	T	U	T	Exotic
51	**Osmeridae**	**Rainbow smelt**	**4**	**3**	**4**	**3**	*
52	**Percidae**	**Iowa darter**	**5**	**3**	**4**	**3**	
53		**Least darter**	**4**	**3**	**5**	**3**	
54		Johnny darter	3	2	3	2	
55		Yellow perch	3	2	3	2	
56		Logperch	3	2	4	2	
57		Walleye	4	3	4	3	
58	**Percopsidae**	**Trout-perch**	**4**	**3**	**4**	**2**	
59	**Salmonidae**	**Round whitefish**	**4**	**3**	**4**	**3**	
60	Sciaenidae	Freshwater drum	1	2	1	2	
61	Umbridae	Central mudminnow	4	2	4	2	

blages associated with wetlands in all five Great Lakes. Our sampling efforts included a total of 38,219 individual fish, from 19 families, and 61 species (captured with fyke nets in all Great Lakes). These data are used here to develop the basin-wide WFI (Table 2). Six species were present in more than half the sites and represented 52% of the total number of individuals. These included three sunfish (pumpkinseed *Lepomis gibbosus*, rock bass *Ambloplites rupestris*, and largemouth bass *Micropterus salmoides*), yellow perch *Perca flavescens*, brown bullhead *Ameiurus nebulosus*, and bluntnose minnow *Pimephales notatus* (Table 2). More than 75% of the total catch was accounted for by 18 taxa present in 25 or more sites (Table 2). Eleven of the remaining species, including three species exotic to the Great Lakes, were captured in only a single site, and were excluded from the multivariate analyses (see note in Table 2).

Index Development

An ordination of the 61 fish species with eight environmental variables produced significant correlations between species and variables associated with environmental degradation for both PA (Fig. 3) and AB data (not presented here because of strong similarities in the results). The first two axes explain 55.2% of the variation in the presence data, and 56.8% of the variation in abundance data. The pCCA results produced a significant CCA axis 1 and all canonical axes ($P < 0.01$). The first axis of the pCCA was strongly correlated with environmental conditions (PA: 0.906, AB: 0.896), where the positive end of pCCA axis 1 was associated with species normally found in degraded conditions (e.g., high primary nutrient concentration, suspended solids) while the negative end was associated with species that are intolerant of water-quality impairment (Fig. 3). The environmental variables that were highly correlated with the first pCCA axis

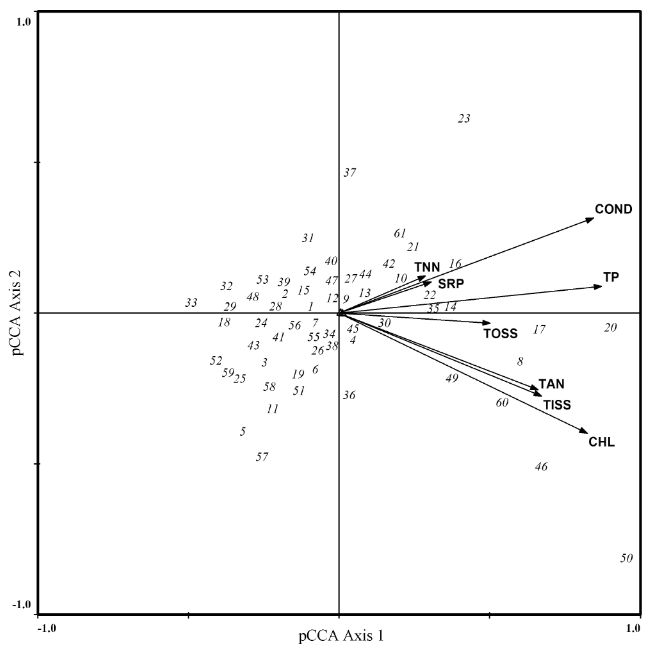

FIG. 3. Ordination biplot of 61 fish species (see Table 3 for species numbers) from partial canonical correspondence analysis (pCCA) of 100 Great Lakes wetlands with 8 environmental variables (conductivity COND, total nitrate nitrogen TNN, soluble reactive phosphorus SRP, total phosphorus TP, total organic suspended solids TOSS, total inorganic suspended solids TISS, total ammonia nitrogen TAN, and chlorophyll CHL).

were total phosphorus (TP, r = 0.81), conductivity (COND, r = 0.79), chlorophyll (CHL, r = 0.76), total inorganic suspended solids (TISS, r = 0.63), and total ammonia (TAN, r = 0.61). The remaining three environmental variables were only weakly correlated with the first pCCA axis but the correlation was greater with the first axis than with the second. These were total organic suspended solids (TOSS, r = 0.47), soluble reactive phosphorus (SRP, r = 0.29), and total nitrate (TNN, r = 0.27).

The position of a species on the pCCA biplot is a reflection of the environmental conditions where it was found. This location can be interpreted as representing the species' affinity for degraded vs. unimpacted habitat. For example, species located on the positive end of the ordination axis should be tolerant of high levels of nutrients, chlorophyll, conductivity, and suspended solids (i.e. disturbed sites) and were assigned U values of 1 (Fig. 3), and these include (numbers in parenthesis in Fig. 3): goldfish, *Carassius auratus* (20), gizzard shad *Dorosoma cepedianum* (17), green sunfish *Lepomis cyanellus* (8), freshwater drum *Aplodinotus grunniens* (60), and channel catfish *Icalurus punctatus* (46) (Table 3, Fig. 3). Species in the middle of the ordination are associated with intermediate water quality conditions, tend to be distributed over a wide range of environmental conditions, and were assigned a U value of 3. These are common, ubiquitous taxa, and include: pumpkinseed *Lepomis gibbonus* (9), smallmouth bass *Micropterus dolomieu* (12), brown bullhead *Ameiurus nebulosus* (45), bluntnose minnow *Pimephales notatus* (34), and yellow perch *Perca flavescens* (55) (Table 2, Fig. 3).

By contrast, species found on the negative end of the ordination axis tend to occur in water quality conditions with low levels of conductivity, nutrients, chlorophyll, and suspended solids, and were assigned a U value of 5. These include blackchin shiner *Notropis heterondon* (29), mimic shiner *Notropis volucellus* (32), longnose gar *Lepisosteus osseus* (48), northern redbelly dace *Phoxinus eos* (33), and Iowa darter, *Etheostoma exile* (52) (Fig. 3). Most species that had not been included in $\text{WFI}_{\text{Lower}}$ (bold names in Table 3) had centroids located on the negative end of the pCCA biplot (Fig. 3). One northern species (creek chub *Semotilus atromaculatus* [36[) had a centroid located on the positive end of the ordination, suggesting it is an indicator of degraded conditions at northern sites. This is consistent with its occurrence in degraded wetlands in the lower lakes. All remaining species were assigned intermediate U values (2 or 5) according to the position of the species centroids along pCCA axis 1.

WFI Performance

There were similar trends in frequency distribution of U values for the $\text{WFI}_{\text{Basin}}$ derived from PA (Fig. 4 a–f) and AB data (Fig. 4 g–l). Occurrence of species with a U value of 5 is more common in the northern lakes than the southern, especially for the basin-wide index ($\text{WFI}_{\text{Basin}}$ (PA), Fig. 4 a–c). There were few occurrences where U equaled 5 for wetlands in the lower lakes (Fig. 4e), but many instances where U equaled 1 (see Fig. 4). When data for all lakes were combined, the most common U-value was the intermediate score of 3.

We found a significant positive regression relationship when $\text{WFI}_{\text{Basin}}$ scores were plotted against WQI scores for each wetland (Fig. 5 a–d). Wetlands with unimpacted water quality had higher $\text{WFI}_{\text{Basin}}$ scores for both PA (slope = 0.31; $r^2 = 0.66$; $P < 0.01$; Fig. 5a) and AB data (slope = 0.35; $r^2 = 0.61$; $P < 0.01$; Fig. 5b). When presence of exotic species was accounted for using our correction factor (i.e., square root of proportion of exotic species), the relationship between WFI and WQI remained statistically significant but the slope increased for both the PA (slope = 0.41; $r^2 = 0.70$; $P < 0.01$; Fig. 5c) and AB regressions (slope = 0.44; $R^2 = 0.51$; $P < 0.01$; Fig. 5d). However, correction for the presence of exotic species only improved the explanatory power of the relationship between WQI and $\text{WFI}_{\text{Basin}}$ (PA_{ex}).

Application of the WFI

We compared data for eight wetlands located in four of the five Great Lakes where we had sampled in multiple years to assess the performance of the $\text{WFI}_{\text{Basin}}$ (Fig. 6a). Because annual variation was similar for both the $\text{WFI}_{\text{Basin}}$ (PA) and $\text{WFI}_{\text{Basin}}$ (AB) scores, we only include results for $\text{WFI}_{\text{Basin}}$ (PA). The largest inter-annual variation in WFI scores occurred in two wetlands, Matchedash Bay in Georgian Bay (0.29 units) and Turkey Point in Lake Erie (0.35 units) (Fig. 6a). By comparison, there was little inter-annual variation associated with corresponding WQI scores (Fig. 6b). The largest change in WQI scores between years was evidenced in Cloud Bay, although there were no corresponding departures for WFI scores. Despite differences in the performance of the two indices,

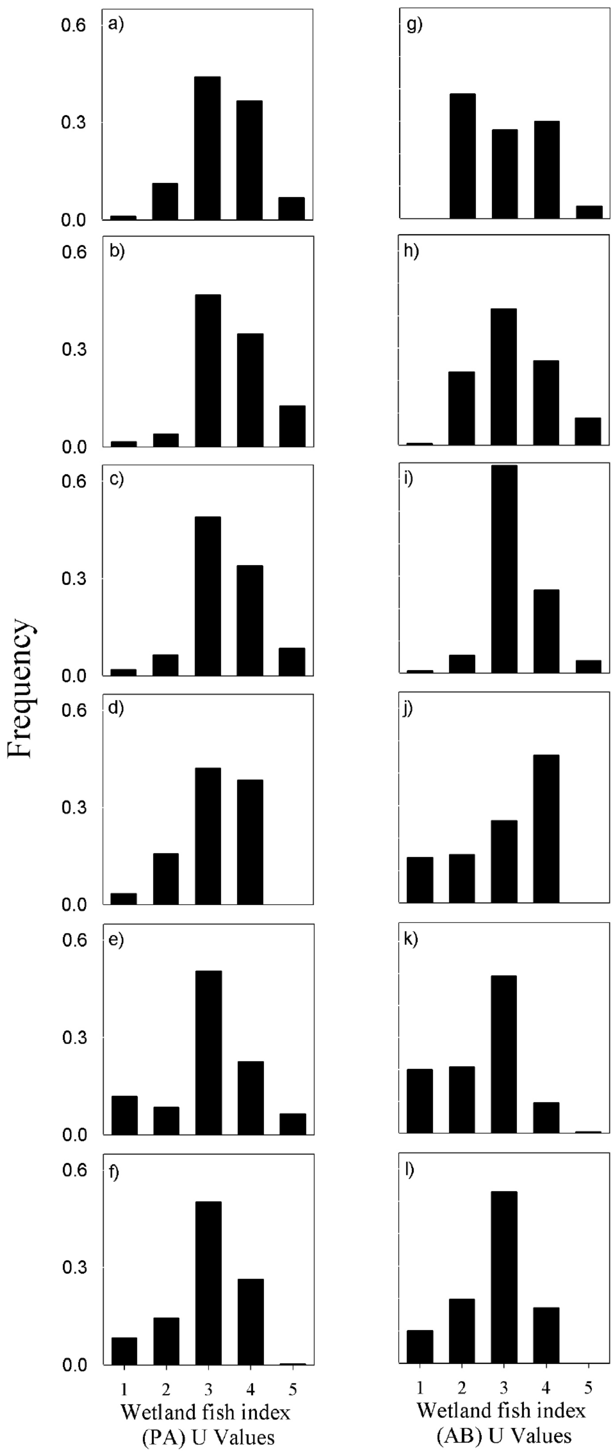

FIG. 4. Frequency of occurrence of wetland fish index (PA [for presence data]) species U values (a–f) and wetland fish index (AB [for abundance data]) species U values (g–l) for Lake Superior (a and g), Georgian Bay (b and h), Lake Huron (c and i), Lake Michigan (d and j), Lake Erie (e and k), and Lake Ontario (f and l).

there was generally agreement among scores from multiple years within indices, and there does not appear to be consistency in how scores deviated from lake to lake.

Another example of WFI use is a comparison of wetlands where we collected fish data with fyke nets and electrofishing because different researchers will likely use different methods of fish collection. Within a subset of 31 wetlands, we compared WFI_{Basin} scores calculated with fish information collected in three sets of paired fyke nets (FN) with those calculated from three transects of boat electrofishing (EF). Mean scores were only slightly higher for FN compared with EF data (Fig. 7), but differences were not significant for Lake Erie, Lake Ontario, Georgian Bay, and Lake Huron (one-way ANOVA; $P = 0.55$, $P = 0.56$, $P = 0.46$, $P = 0.76$, respectively). We also observed lake-to-lake variation in the number of fish species captured by different fishing methods. There were no significant differences between capture methods for Lake Erie and Lake Huron wetlands (Erie: 10 vs. 14 species per wetland for FN and EF, respectively) (one-way ANOVA $P = 0.06$) (Huron: 11 vs. 10 species) (one-way ANOVA $P = 0.79$). However, significantly more species were captured with FN in Georgian Bay and Lake Ontario wetlands (Georgian Bay: 15 vs. 11 species) (one-way ANOVA $P < 0.01$) and (Lake Ontario: 13.9 vs. 10.4 species) (one-way ANOVA $P = 0.016$), Fig. 7).

DISCUSSION

This study presents an application of one of the few fish-based ecological indices for use in Great Lake coastal wetlands. This index is the only one that has been developed with data from all five Great Lakes, including eastern and northern Georgian Bay, and the North Channel. Like its predecessor, the wetland fish index (WFI_{Lower}, Seilheimer and Chow-Fraser 2006), the basin-wide index we developed produced scores that were highly correlated with WQI scores. This also confirms the relationship of fish presence-absence or abundance with overall water-quality conditions in Great Lake coastal wetlands. Our correction for the proportion of exotic species present in a wetland increased the range of possible WFI_{Basin} scores and increased the potential sensitivity of the index to detect changes in wetland condition associated with invasive species. This modified index should be useful as a rapid method for the comparison of intra- and inter-

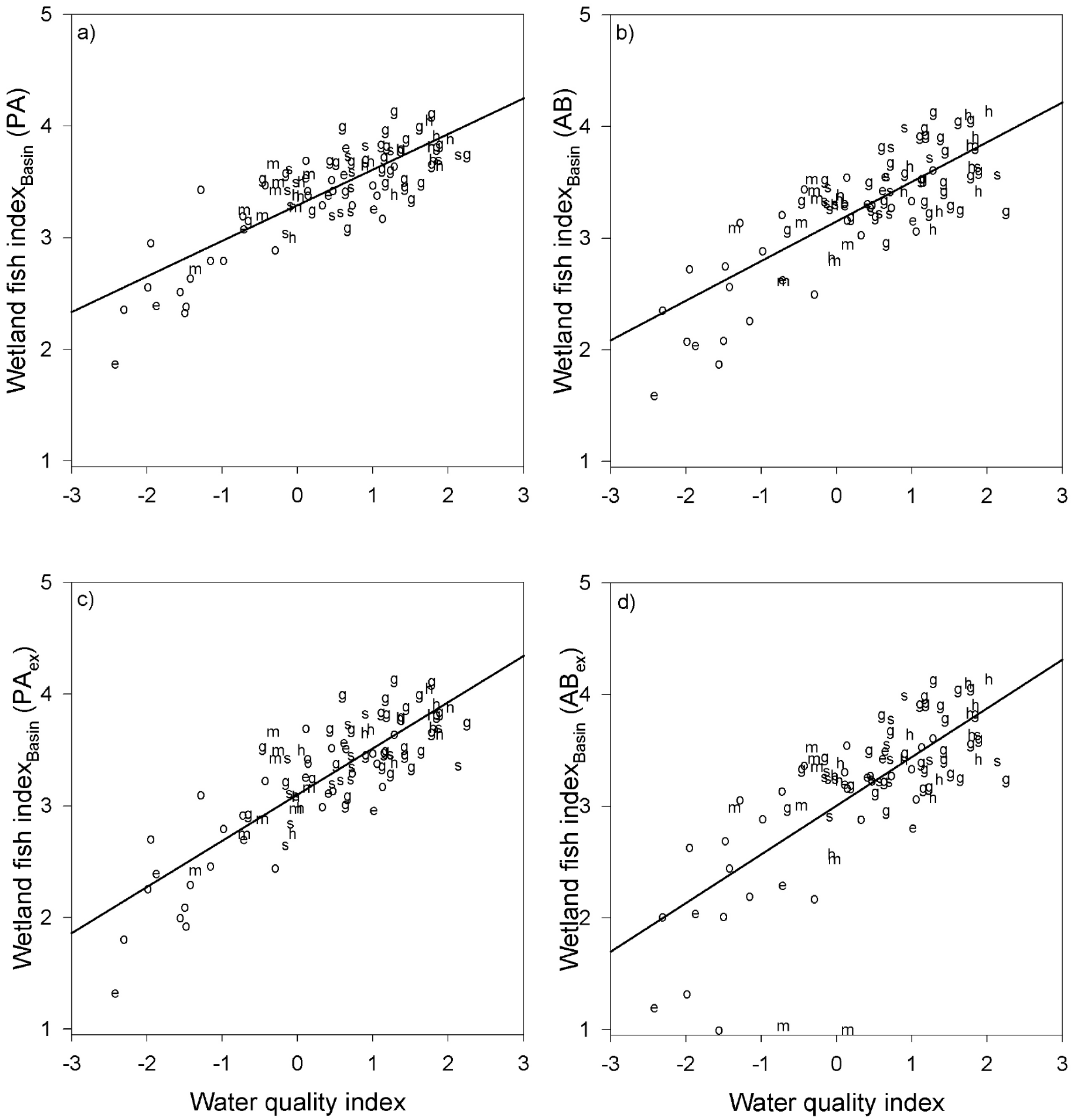

FIG. 5. Regression of wetland fish index scores against water quality index scores. Wetland fish index scores were calculated with a) presence data only, b) abundance data, c) presence data corrected for exotic species, and d) abundance data corrected for exotic species. Letter codes indicate Great Lake origin of the wetland (e = Erie, g = Georgian Bay, h = Huron, m = Michigan, o = Ontario, and s = Superior).

wetland variation and for tracking changes in wetlands through time.

The development of ecological indicators greatly increased as the public's understanding of human impacts on aquatic systems grew along with the computational power of modern technology. Niemi and McDonald (2004) define ecological indicators as measurable characteristics of structure, composition, or function of ecological systems. In this context, the WFI_{Basin} is a measurable indicator of fish species composition in coastal wetlands but also considers ecosystem "function" because environmental variables (water quality) are incorporated into the index. Alternatively, the WFI_{Basin} is considered a "state indicator" in the context of the State of the Lakes Ecosystem Conference (Shear *et al.* 2003).

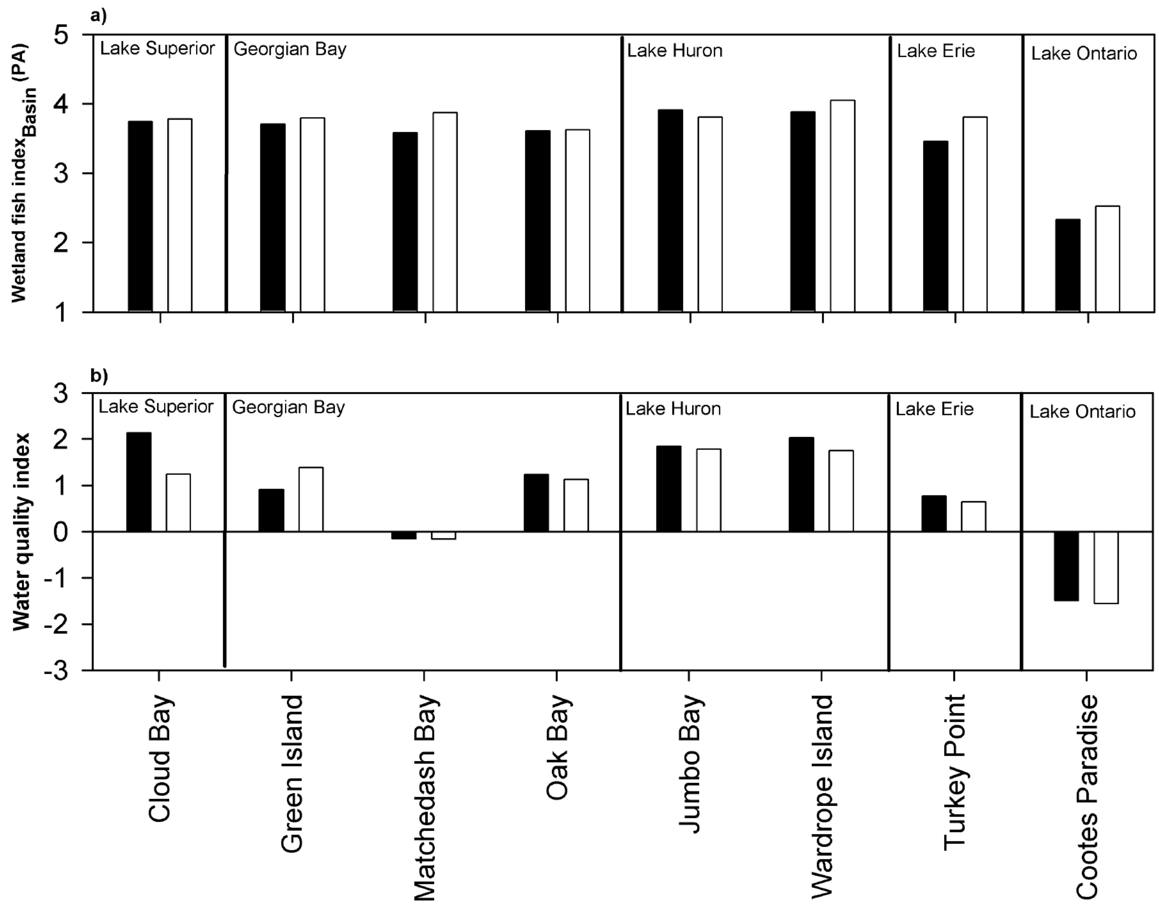

FIG. 6. Comparison of a) wetland fish index (PA) scores and b) water quality index scores for consecutive years (year one, black bars ■ and year two, white bars □) in wetlands (Cloud Bay 2001–2002, Green Island 2003–2004, Matchedash Bay 2003–2004, Oak Bay 2003–2004, Jumbo Bay 2004–2005, Wardrope Island 2004–2005, Turkey Point 2001–2002, and Cootes Paradise 2001–2002).

Fish are a suitable group of organisms for monitoring aquatic condition because they have a documented relationship with environmental perturbations (i.e., increased trophic status and degraded fish community, Lee *et al.* 1991). Variation in environmental condition cause changes in the fish community, and those changes can then be extracted into ecological indicators to quantify environmental impacts (Karr 1981, Seilheimer and Chow-Fraser 2006). Simon (1991) identified four categories of attributes of fish that make them suitable for bioassessment: accuracy, visibility, ease of use, and interpretation. Fish provide an accurate assessment of environmental condition because they have large geographical ranges, are long lived, and have a broad spectrum of tolerances to degradation (Simon 1991). They are also important to the public compared with other groups of organisms and have historically been used in association with aquatic habitats (e.g., "fishable" waters goal in Clean Water Act, Simon 1991). Finally, fish are well known taxonomically and their life-history, distribution, and tolerance to perturbation are also well-documented (Simon 1991).

The development of indicators of wetland condition needs to be based on a firm ecological basis. We have used water quality variables to guide our development of the index and have created a robust indicator of wetland condition. The relationship between fish communities and the environment has been documented in many cases: fish communities in streams (Brazner *et al.* 2005), fish communities in wetlands (Brazner 1997, Brazner and Beals 1997), and fish communities in lakes (Tonn and

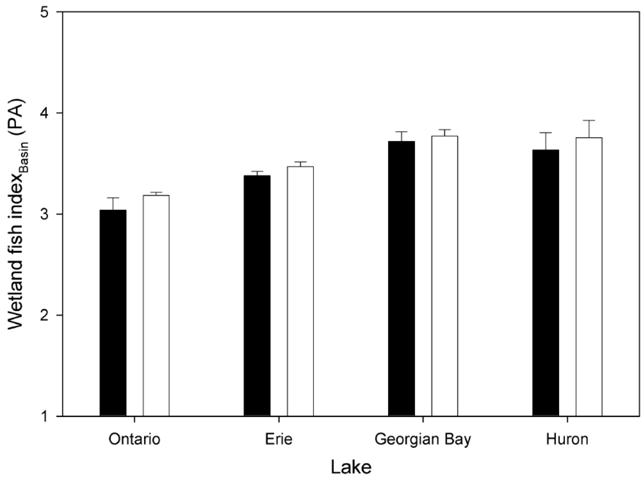

FIG. 7. Mean wetland fish index$_{Basin}$ (PA) scores and standard error calculated with data collected with boat electrofishing (black bars ■) and with fyke nets (white bars □) in Lake Ontario (n = 8), Lake Erie (n = 2), Georgian Bay (n = 12), and Lake Huron (n = 9).

Magnuson 1982). The changes in fish communities are often linked to changes in water quality, which have detrimental effects on fish habitat, such as the loss of vegetation from increased sedimentation and nutrient concentrations (Brouwer *et al.* 2002) or impairment of fish community in response to increases in the trophic state of aquatic habitats through nutrient loading (Lee *et al.* 1991).

There are many overlaps between multimetric and multivariate indices, including focus on biological endpoints, assessment of changes due to anthropogenic disruption of habitats, and application in habitat conservation (Karr 2000). There are also differences between the methods for which Karr and Chu (1999) criticized the multivariate techniques, including the assumption of a normal distribution, exclusion of rare species, and the assumption that the maximum variance in data has biological significance. While these criticisms may be true of some multivariate indices, the WFI$_{Basin}$ is not limited by these disadvantages. First, the fish species dataset had a unimodal distribution, which was tested with detrended correspondence analysis and is a requirement for the canonical correspondence analysis (ter Braak and Smilauer 1998, ter Braak and Verdonschot 1995). Second, we included species that Karr and Chu (1999) consider to be rare (< 10% sites) because these species have value as indicators of degraded water quality. Only species that were found in a single site were excluded but future research may provide insight into the use of these excluded species in the WFI. Third, we were able to confirm the placement of species as indicators of condition based on knowledge about habitat and their tolerance to disturbance (Scott and Crossman 1998, Seilheimer and Chow-Fraser 2006).

We presented differences between the development of the WFI$_{Basin}$ and other indices as well as implications for their use. First, the WFI$_{Basin}$ does not distinguish between wetland type, as do other indices: IBI metrics for *Typha*-dominated wetlands and *Scirpus*-dominated wetlands (Uzarski *et al.* 2005), littoral habitats in areas of concern (Minns *et al.* 1994), and vernal pools (Simon *et al.* 2000). Second, the WFI$_{Basin}$ can be used on wetlands in all five Great Lakes in the U.S. and Canada. Indices, like the IBI, are often limited to specific geographic regions and need to be modified for use outside these regions such as the dunal, palustrine wetlands on the southern shore of Lake Michigan (Simon 1998); or southern Great Lake wetlands in Canada (Uzarski *et al.* 2005). Additionally, the WFI$_{Basin}$ could potentially be used on other habitat types, such as shallow littoral zones of lakes, as long as the habitats contain similar species as Great Lakes coastal marshes. The WFI$_{Basin}$ should be broadly applicable to multiple types of wetlands because of the mix of wetland types and the broad range of wetlands used in its development.

Interannual variation in WFI$_{Basin}$ scores was not large and implies that the index will be a powerful tool for monitoring and for comparing scores calculated for wetlands sampled in different years. Although Karr and Chu (1999) criticize the use of data from multiple time periods in multivariate analyses, we believe that the variation included in the index development makes the WFI$_{Basin}$ more robust. Because there is a strong relationship between water quality and the WFI$_{Basin}$, we would expect that if WQI scores did not vary between years, then WFI$_{Basin}$ should not vary significantly either. This was confirmed by our analysis. Annual stability in the fish community and IBI scores have been reported by others (IBI, Lyons 2006), indicating that indices should be stable throughout the year. Wilcox *et al.* (2002) determined that IBIs could not easily be developed for Great Lake wetlands without an understanding of the impacts of variation in water level on individual wetlands over time. To confirm our findings, additional wetlands need to be examined over multiple years when there is minimal interannual variation in water quality, and to quantify the effects of variations in water levels on the index.

Bias associated with fishing gear types can greatly influence comparisons of aquatic habitats, especially when meaningful community information is desired for habitat restoration research (Jackson and Harvey 1997). Chow-Fraser *et al.* (2006) found that fyke nets tend to catch significantly more fish than boat electrofishing, although both methods captured a similar number of species per wetland. Different fishing collection methods have variable capture efficiency (Claramunt *et al.* 2005), which can result in contradictory results and incorrect conclusions (Jackson and Harvey 1996) based on gear used or sampling intensity (Jackson and Harvey 1997). Multimetric indicators are especially vulnerable to bias because many of the metrics use community factors, such as percent of total catch from feeding guilds or percent of total biomass from specific species (Simon 1999). Some researchers have recommended more use of presence-absence data to account for variation in fish collection bias (Jackson and Harvey 1997). The WFI_{Basin} is especially suitable as an ecological indicator because it can be calculated with presence-absence data and with more quantitative fish data even though water quality data are not available. Seilheimer and Chow-Fraser (2006) used the WFI_{Basin} on a published fish dataset from Green Bay, Lake Michigan (Brazner and Beals 1997) and were able to distinguish groups of wetlands with the WFI_{Lower} that were in agreement with Brazner and Beal's (1997) classifications. For best results, we recommend consistent sampling gear and fishing effort be used when collecting fish data for use with the WFI_{Basin}.

The WFI_{Basin} can be used as an alternative to developing new regional IBIs for the Great Lake coastal wetlands. The WFI_{Basin} is complementary with recent efforts in stream ecology to assign "tolerance indicator values" to individual species based on occurrence and water quality derived from a large geographic scale (i.e., lower 48 states, Meador and Carlisle 2007). To date, there has not been an IBI produced for the entire Great Lakes basin. The most recent IBI developed by Uzarski *et al.* (2005) remains untested for basin-wide comparisons. The Uzarski *et al*. (2005) IBI has a regional component, because there are two sets of metrics that have been derived for *Typha* communities and *Scirpus* communities, which results in a division of south Great Lakes (*Typha*) and northern Great Lakes (*Scirpus*; Croft and Chow-Fraser 2007, this issue). Because the WFI_{Basin} does not distinguish between wetland types or regions of the Great Lakes, it can be used by managers for comparison of wetland condition and can provide a benchmark by which regional IBIs can be compared.

The WFI_{Basin} is a rapid method for determining the condition of Great Lake coastal wetlands. This study represents the development of the first comprehensive, fish-based index of wetland condition developed for the entire Great Lakes basin and has broad applications to wetland management. The WFI_{Basin} is derived from the relationship between the biotic and environmental variables and has a significant relationship with water quality. We also derived a correction for the proportion of exotic species in a wetland, which can be used to account for the detrimental effects that invasive species have on aquatic habitats. Finally, the WFI_{Basin} does not significantly vary over multiple years or when different fishing gears are used. Hence, the index can be applied to existing datasets and allows for greater flexibility in its future use.

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REFERENCES

APHA (American Public Health Association). 1992. *Standard methods for examination of water and wastewater 18th edition.* Washington, District of Columbia: American Water Works Association, and Water Environment Federation.

Bertram, P., and Stadler-Salt, N., eds. 2000. *Selection of Indicators for Great Lakes basin ecosystem health.* Version 4. U.S. Environmental Protection Agency and Environment Canada. <www.on.ec.gc.ca/solec/indicators2000-e.html >

Brazner, J.C. 1997. Regional, habitat, and human development influences on coastal wetland and beach fish assemblages in Green Bay, Lake Michigan. *J. Great Lakes Res.* 23:36–51.

______, and Beals, E.W. 1997. Patterns in fish assemblage from coastal wetland and beach habitats in Green Bay, Lake Michigan: a multivariate analysis of abiotic and biotic forcing factors. *Can. J. Fish. Aquat. Sci.* 54:1743–1761.

______, Tanner, D.K., Detenbeck, N.E., Batterman, S.L., Stark, S.L., Jagger, L.A., and Snarski, V.M. 2005. Landscape character and fish assemblage structure and function in Western Lake Superior streams: general relationships and identification of thresholds. *Environ. Manage.* 33:855–875

Brouwer, E., Bobbink, R., and Roelofs, J.G.M. 2002. Restoration of aquatic macrophyte vegetation in acidified and eutrophied softwater lakes: an overview. *Aquat. Bot.* 73:405–431.

Chow-Fraser, P. 1998. A conceptual ecological model to aid restoration of Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario, Canada. *Wetland Ecol. Manage*. 6:43–57.

______. 2006. Development of the wetland water quality index for assessing the quality of Great Lakes coastal wetlands. In *Coastal wetlands of the Laurentian Great lakes: health, habitat and indicators,* T.P. Simon and P.M. Stewart, eds., pp. 137–166. Bloomington, Indiana: Indiana Biological Survey.

______, Kostuk, K., Seilheimer, T., Weimer, M., MacDougall, T., and Theÿsmeÿer, T. 2006. Effect of wetland quality on sampling bias associated with two fish survey methods for coastal wetlands of the lower Great Lakes. In *Coastal wetlands of the Laurentian Great Lakes: health, habitat and indicators,* T.P. Simon and P.M. Stewart, eds., pp. 233–256. Bloomington, Indiana: Indiana Biological Survey.

Claramunt, R.M., Shoup, D.E., and Wahl, D.H. 2005. Comparison of push nets and tow nets for sampling larval fish with implications for assessing littoral habitat utilization. *N. Am. J. Fish. Manage.* 25:86–92.

Croft, M.V., and Chow-Fraser, P. 2007. Use and development of the wetland macrophyte index to detect water-quality impairment in fish habitat of Great Lakes coastal marshes. *J. Great Lakes Res.* 33 (Special Issue 3):172–197.

Dale, V.H., and Beyeler, S.C. 2001. Challenges in the development and use of ecological indicators. *Ecol. Indic.* 1:3–10.

Drake, M.T., and Valley, R.D. 2005. Validation and application of a fish-based index of biotic integrity for small central Minnesota lakes. *N. Am. J. Fish. Manage.* 25:1095–1111.

Fuller, K., Shear, H., and Wittig, J., eds. 1995. *The Great Lakes: an environmental atlas and resource book. 3rd edition*. Toronto, Canada: Government of Canada.

Jackson, D.A., and Harvey, H.H. 1996. Gradual reduction and extinction of fish populations in acid lakes. *Water Air Soil Pollut.* 85:389–394.

______, and Harvey, H.H. 1997. Qualitative and quantitative sampling of lake fish communities. *Can. J. Fish. Aquat. Sci.* 54:2807–2813.

Jude, D.J., and Pappas, J. 1992. Fish utilization of Great Lakes coastal wetlands. *J. Great. Lakes Res.* 18:651–672.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6:21–27.

______. 1991. Biological integrity: a long-neglected aspect of water resource management. *Ecol. Appl.* 1:66–84.

______. 2000. Health, integrity, and biological assessment: The importance of whole things. In *Ecological integrity: integrating environment, conservation, and health,* D. Pimentel, L. Westra, and R.F. Noss, eds., pp. 209–226. Washington, District of Columbia: Island Press.

______, and Chu, E.W. 1999. *Restoring life in running waters: better biological monitoring.* Washington, District of Columbia: Island Press.

Lee, G.F., Jones, P.E., and Jones, R.A. 1991. Effects of eutrophication on fisheries. *Rev. Aquat. Sci.* 5:287–305.

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

Lyons, J. 2006. A fish-based index of biotic integrity to assess intermittent headwater streams in Wisconsin, USA. *Environ. Monit. Assess.* 122:239–258.

McNair, S.A., and Chow-Fraser, P. 2003. Change in biomass of benthic and planktonic algae along a disturbance gradient for 24 Great Lakes coastal wetlands. *Can. J. Fish. Aquat. Sci.* 60:676–689.

Meador, M.R., and Carlisle, D.M. 2007. Quantifying tolerance indicator values for common stream fish species of the United States. *Ecol. Indic.* 7:329–338.

Mills, E.L., Leach, J.H., Carlton, J.T., and Secor, C.L. 1993. Exotic species in the Great Lakes: a history of biotic crises and anthropogenic introductions. *J. Great Lakes Res.* 19:1–57.

Minns, C.K., Cairns, V.W., Randall, R.G., and Moore, J.E. 1994. An index of biotic integrity (IBI) for fish assemblages in the littoral zone of Great Lakes' areas of concern. *Can. J. Fish. Aquat. Sci.* 51:1804–1822.

Niemi, G.J., and McDonald, M.E. 2004. Application of ecological indicators. *Annu. Rev. Ecol. Evol. Syst.* 35:89–111.

Scott, W.B., and Crossman, E.J. 1998. *Freshwater fishes of Canada. 2nd ed*. Bull. Fish. Res. Board Can. No. 184.

Seilheimer, T.S., and Chow-Fraser, P. 2006. Development and use of the wetland fish index to assess the quality of coastal wetlands in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 63:354–366.

Shear, H., Stadler-Salt, N., Bertram, P., and Horvatin, P. 2003. The development and implementation of indicators of ecosystem health in the Great Lakes basin. *Environ. Monit. Assess*. 88:119–152.

Simon, T.P. 1991. *Development of index of biotic integrity expectations for the ecoregions of Indiana. I. Central Corn Belt Plain*. Chicago, IL. U.S. Environmental Protection Agency EPA/905/9-91/025.

______. 1998. Modification of an index of biotic integrity and development of reference condition expectations for dunal, palustrine wetland fish communities along the southern shore of Lake Michigan. *Aquat. Ecosys. Health Manage.* 1:49–62.

______. 1999. *Assessing the suitability and biological integrity of water resources using fish communities*. Boca Raton, Florida: CRC Press.

______, Jankowski, R., and Morris, C. 2000. Modification of an index of biotic integrity for assessing vernal ponds and small palustrine wetlands using fish, crayfish, and amphibian assemblages along southern Lake Michigan. *Aquat. Ecosys. Health Manage.* 3:407–418.

Suter, G.W. 2001. Applicability of indicator monitoring to ecological risk assessment. *Ecol. Indic.* 1:101–112.

ter Braak, C.J.F., and Smilauer, P. 1998. *CANOCO reference manual and user's guide to CANOCO for windows: software for canonical community ordination*. Version 4. Ithaca, New York: Microcomputer Power.

______, and Verdonschot, P.F.M. 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. *Aquat. Sci.* 57:255–289.

Tonn, W.M., and Magnuson, J.J. 1982. Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. *Ecology* 63:1149–1166.

Uzarski, D.G., Burton, T.M., Cooper, M.J., Ingram, J.W., and Timmermans, S.T.A. 2005. Fish habitat use within and across wetland classes in coastal wetlands of the five Great Lakes: development of a fish-based index of biotic integrity. *J. Great Lakes. Res.* 31:171–187.

Wei, A., Chow-Fraser, P., and Albert, D. 2004. Influence of shoreline features on fish distribution in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 61:1113–1123.

Wilcox, D.A., Meeker, J.E., Hudson, P.L., Armitage, B.J., Black, M.G., and Uzarski, D.G. 2002. Hydrologic variability and the application of index of biotic integrity metrics to wetlands: A Great Lakes evaluation. *Wetlands* 22:588–615.

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