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Comparative study of ecological indices for assessing human-induced disturbance in coastal wetlands of the Laurentian Great Lakes

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ARTICLE INFO

Article history:
Received 4 June 2007
Received in revised form
10 December 2007
Accepted 8 February 2008

Keywords:
Laurentian Great Lakes
Wetlands
Bioassessment
Water quality
Macrophytes
Zooplankton
Fish

ABSTRACT

Several ecological indices have been developed to evaluate the wetland quality in the Laurentian Great Lakes. One index, the water quality index (WQI) can be widely applied to wetlands and produces accurate measurements of wetland condition. The WQI measures the degree of water quality degradation as a result of nutrient enrichment and road runoff. The wetland fish index (WFI), wetland zooplankton index (WZI), and the wetland macrophyte index (WMI), are all derived from the statistical relationships of biotic communities along a gradient of deteriorating water quality. Compared to the WQI, these indices are less labor-intensive, cost less, and have the potential to produce immediate results. We tested the relative sensitivity of each biotic index for 32 Great Lakes wetlands relative to the WQI and to each other. The WMI ($r^2 = 0.84$) and WFI ($r^2 = 0.75$) had significant positive relationships ($P < 0.0001$) with the WQI in a linear and polynomial fashion. Slopes of the WMI and WFI were similar when comparing the polynomial regressions (ANCOVA; $P = 0.117$) but intercepts were significantly different ($P = 0.004$). The WZI had a positive relationship with the WQI in degraded wetlands and a negative relationship in minimally impacted wetlands. The strengths and weaknesses of each index can be explained by the interactions among fish, zooplankton, aquatic plants and water chemistry. The distribution of different species indicative of low and high quality in each index provides insight into the relative wetland community composition in different parts of the Great Lakes and helps to explain the differences in index scores when different organisms are used. Our findings suggest that the WMI and WFI produce comparable results but the WZI should not be used in the minimally impacted wetlands without further study.

1. Introduction

Wetland ecosystems provide important ecological and economic services for the Great Lakes (e.g. support biotic diversity, nutrient retention, flood protection, Mitsch and Gosselink, 2000). Basin-wide land use changes have caused impairment of water quality and the outright loss of wetland habitat (Smith et al., 1991; Chow-Fraser et al., 1998). Populated areas of Lake Ontario have had an average loss of 75% of their wetland area, while the areas of highest development have had 100% losses (Whillans, 1982). Degradation of wetlands via degraded water quality causes a cascade of changes in the biotic communities, where the altered macrophyte community results in impacted zooplankton and fish communities.

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1470-160X/$ – see front matter
doi:10.1016/j.ecolind.2008.02.001

Zooplankton are dependent upon submergent vegetation as a refugia from planktivorous fish (Timms and Moss, 1984; Lougheed and Chow-Fraser, 2002), so the loss of submergent vegetation can lead to a shift in the zooplankton assemblage from turbidity intolerant, epiphytic grazers to turbidity tolerant species that thrive in open water (Lougheed and Chow-Fraser, 1998). There is also a well documented relationship between fish species and diverse and structurally complex submergent vegetation community (Casselman and Lewis, 1996; Brazner and Beals, 1997; Weaver et al., 1997). The association between wetland biotic communities and water quality can be documented and then the biotic community alone can be used to quantify the condition of a wetland using while still reflecting the water quality aspects.

Indicators of ecological change are important tools for managers, allow them to have a stable and comparable method for tracking how wetland condition changes as degraded sites are actively restored and as a means to measure and define endpoints for success. The extent of degradation among sites should be measured so that an objective system may be used to identify wetlands that would benefit from restorative efforts. Accountability and efficiency are vital to restoration efforts because of the large cost associated with them: 187 habitat restoration projects completed in Canadian Great Lake Areas of Concern cost a total of \$80.3 million (Canadian dollars, International Joint Commission, 2003). Wetlands that have not been impacted by human activities can also be identified and protected with indicators. The protection of the aquatic habitats before they can be degraded is likely to be more beneficial to the health of the ecosystems and, ultimately, may prove to be more cost effective than habitat restoration.

Within the Great Lakes, few indicators of wetland condition have been developed and those in existence have never been directly compared. Chow-Fraser (2006) developed the water quality index (WQI), derived from analysis of 110 wetlands located throughout the Great Lakes, using measurement of 12 environmental parameters (e.g. nutrients, suspended solids, etc.) to gauge water quality impairment within the wetlands. The WQI proved to be a significant and robust measurement of land use alteration (shift from forested to agriculture or urban) and a sensitive indicator of human-induced impairment of water quality. Water quality-based indices are limited because they are costly, time consuming, and require specialized equipment. Water quality-based indices, like the WQI, also cannot be applied to historical data sets where they are available because the necessary parameters are seldom measured or recorded in previous surveys of wetlands.

Biotic indicators have been developed because of the limitations associated with chemical indicators (e.g. WQI). Although biotic indices have been extensively used in other systems for plants (prairie wetlands: DeKeyser et al., 2003), invertebrates (streams: Kerans and Karr, 1994), and fish (streams: Karr, 1991; Wang et al., 1997), the development of biotic indices in the Great Lakes coastal wetlands has only recently begun in earnest (plants: Albert and Minc, 2004; macroinvertebrates: Kashian and Burton, 2000; fish: Uzarski et al., 2005). Three biological indicators have been developed specifically for Great Lake coastal marshes: the wetland macrophyte index (WMI; Croft and Chow-Fraser, 2007), the wetland zooplankton index (WZI; Lougheed and Chow-Fraser, 2002), and the wetland fish index (WFI; Seilheimer and Chow-Fraser, 2006, 2007). Developed as surrogates of the WQI, these indicators relate changes in a subset of the variables used to develop the WQI with changes in each biotic community. These indicators are desirable substitutes for the WQI because they can be more easily implemented in wetlands and are more cost effective than the WQI. However, the relative sensitivity of each index to changes in water quality is not known, but needs to be completed, as was done elsewhere (periphyton, macroinvertebrates, and fish, Griffith et al., 2005). Unlike the WQI, each biotic index can be used on historical species lists, which are more likely to be available than extensive water quality.

The primary objective of this study is to determine the relationship between each of the WMI, WZI, and WFI to the WQI for 32 wetlands located throughout the Great Lakes shoreline. First, we hypothesize that in most situations the biotic indices will perform as appropriate substitutes for the WQI but that the gradient underlying the development of each index is important in its application to new wetlands. Secondly, we will identify the advantages and disadvantages of each index as an indicator of water quality conditions and ecological basis for the conclusions. We will formally test the following hypotheses; (1) the WMI and WFI are more sensitive indicators of water quality than is the WZI because they were developed from a broader environmental gradient and (2) weaknesses incurred by the WZI will be due to both the absence of appropriate habitat for zooplankton, and predation pressure from planktivorous fish. Finally, we will make recommendations for the appropriate use of these indices for assessing water quality conditions in Great Lakes marshes.

2. Materials and methods

2.1. Study sites

The database used to conduct this comparison was assembled from previously published studies of Chow-Fraser (2006), Croft and Chow-Fraser (2007), Lougheed and Chow-Fraser (2002), and Seilheimer and Chow-Fraser (2006, 2007). Although these combined studies included over 200 coastal marshes in total, only 32 met the requirement for inclusion in this study: sampled at each individual wetland within a consecutive 2-year period for water quality, macrophytes, zooplankton, and fish (Fig. 1; Table 1). The 32 sites within the database were sampled during the time period of 2001–2005. The wetlands ranged from a high degree of anthropogenic impact (eutrophic; high nutrients and suspended solids) to minimal anthropogenic impact (oligotrophic; low nutrients and suspended solids). Unfortunately, there were no data from Lake Michigan wetlands, and hence only sites from the four of five Great Lakes are represented in this study (Fig. 1). The northern Great Lakes region had greater representation in the study, with 21 total wetlands (Superior (4 wetlands), Georgian Bay (16 wetlands), Huron (1 wetland)) compared with 11 from the southern region (Erie (4 wetlands) and Ontario (7 wetlands); Fig. 1).

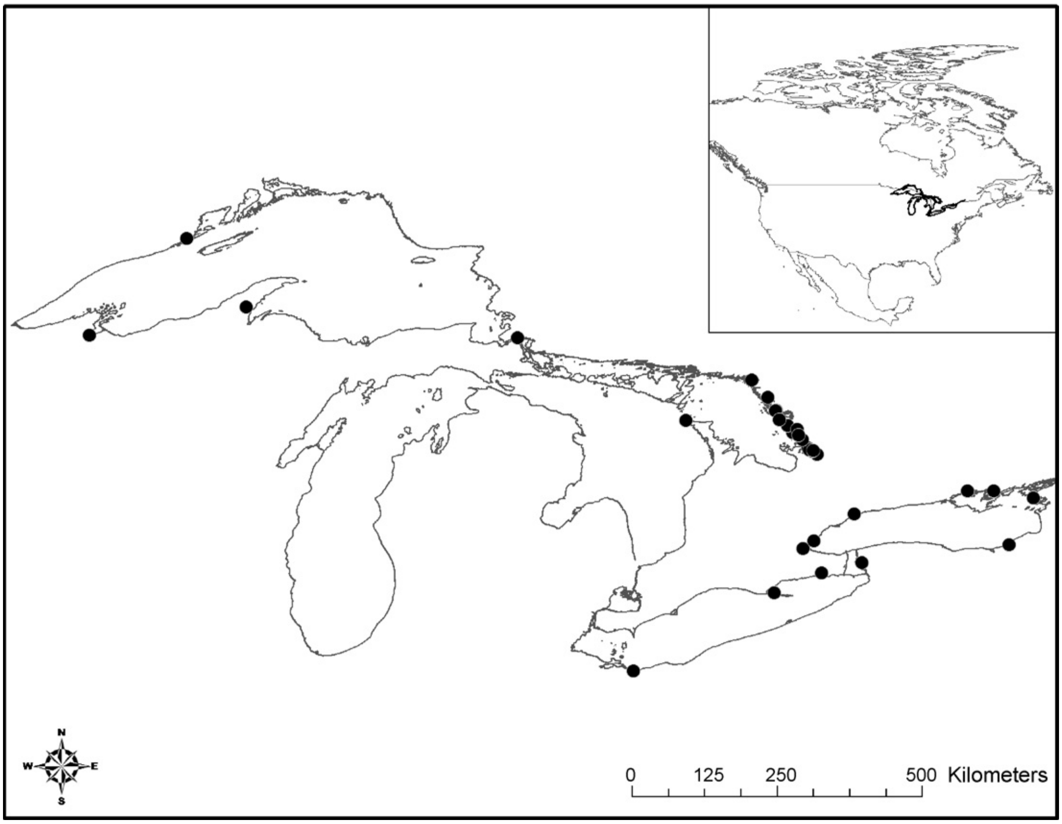

Fig. 1 – Location of 32 wetlands sampled for fish, zooplankton, and plants in the Great Lakes basin between the years of 2001 and 2005 (inset in upper right shows Great Lakes location in North America).

2.2. *Sampling*

Water quality data were collected at mid-depth and at least 10 m from the edge of the aquatic vegetation at each wetland and processed with methods outlined in Chow-Fraser (2006). Physico-chemical variables (e.g. temperature, conductivity, pH) were measured *in situ* with a YSI 6600 multi-parameter probe and a YSI 650 display (YSI, Yellow Springs, OH, USA). Water samples were collected and frozen for nutrient, suspended solids, and chlorophyll *a* and processed using standard methods (Chow-Fraser, 2006). Sites were sampled for zooplankton and fish as outlined by Lougheed and Chow-Fraser (2002) and Seilheimer and Chow-Fraser (2006, 2007), respectively. Zooplankton were collected at mid-depth in a vegetation-free zone of the wetland with a 5 L Schindler-Patalas zooplankton trap. Fish were collected with three sets of 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. Fish were identified to species after 24 h and returned to the wetland. The aquatic-plant community was surveyed usually between late June and late August. In wadeable water, emergent plants would be surveyed by walking along random transects parallel to the shoreline within the flooded zone. Some submergent taxa could be identified within these transects, but majority of these were surveyed with quadrats (0.75 m × 0.75 m) from a canoe or boat, within the vicinity of fyke nets that had been set contemporaneously to survey the fish community. In deeper water (>0.5 m), a rake would be used to collect plants that could not be identified from the canoe. Depending on the size and complexity of the wetland, these surveys would take from 20 min to several hours to complete; generally, 10–15 quadrats would be sampled in each wetland and only the occurrence of species was noted. The focus of the survey was to identify submergent, emergent and floating plant taxa that serve as fish habitat; therefore, species associated with wet meadow were largely ignored.

2.3. *Calculation of the WQI, WMI, WZI, and WFI*

WQI scores were calculated with the 12-variable equation from Chow-Fraser (2006; Eq. (1)). The equation used to generate WQI scores for all wetlands is as follows:

$$\begin{aligned} \mathrm{WQI} = {} & +10.0239684 - 0.3154965 \times \log \mathrm{TURB} - 0.3656606 \\ & \times \log \mathrm{TSS} - 0.3554498 \times \log \mathrm{ISS} - 0.3760789 \\ & \times \log \mathrm{TP} - 0.1876029 \times \log \mathrm{SRP} - 0.0732574 \\ & \times \log \mathrm{TAN} - 0.2016657 \times \log \mathrm{TNN} - 0.2276255 \\ & \times \log \mathrm{TN} - 0.5711395 \times \log \mathrm{COND} - 1.1659027 \\ & \times \log \mathrm{TEMP} - 4.3562126 \times \log \mathrm{pH} - 0.2287166 \\ & \times \log \mathrm{CHL} \end{aligned} \tag{1}$$

Table 1 – Summary of 32 wetlands sampled for fish, zooplankton, and plants between the years of 2001 and 2005 used for this study, included are index scores for the WQI, WMI, WFI, and WZI

Wetland	Location	Year	Water quality	WQI	WMI	WZI	WFI	Latitude	Longitude
Old Woman Creek	Erie	2001	HD	−2.42	1.00	2.32	1.88	41.38	−82.51
Grindstone Creek	Ontario	2002	HD	−2.31	1.00	2.07	2.36	43.28	−79.88
Grand River	Erie	2001	VD	−1.88	1.25	3.33	2.40	42.90	−79.60
Bronte Creek	Ontario	2002	MD	−0.98	1.45	2.70	2.80	43.39	−79.71
Mud Bay	Ontario	2002	MD	−0.72	2.05	3.07	3.20	44.06	−76.31
Buckhorn	Erie	2001	MD	−0.72	2.27	2.86	3.08	43.05	−78.97
Matchedash Bay	Georgian Bay	2002	MD	−0.65	2.44	4.11	3.16	44.73	−79.66
Frenchman's Bay	Ontario	2001	MD	−0.29	2.06	2.89	2.89	43.81	−79.09
Matchedash Bay	Georgian Bay	2003	MD	−0.15	2.45	4.17	3.58	44.73	−79.66
Blessington Bay	Ontario	2002	GD	0.14	2.44	3.53	3.38	44.16	−77.33
Little Sodus	Ontario	2001	GD	0.33	2.03	3.64	3.30	43.33	−76.69
Hay Bay Marsh	Ontario	2002	GD	0.45	2.44	4.24	3.53	44.16	−76.93
Sturgeon Central	Georgian Bay	2003	GD	0.52	3.42	3.96	3.25	45.61	−80.43
West Bay	Georgian Bay	2003	GD	0.56	3.50	3.40	3.68	45.42	−80.30
Key River	Georgian Bay	2003	GD	0.66	3.22	3.20	3.09	45.88	−80.67
Long Point	Erie	2001	GD	0.70	3.00	4.06	3.57	42.59	−80.33
West Fish Creek	Superior	2001	GD	0.70	2.75	4.10	3.24	46.58	−90.94
Green Island	Georgian Bay	2003	GD	0.91	3.04	2.99	3.71	44.78	−79.74
Pike River	Superior	2002	VG	1.01	3.00	3.10	3.07	47.01	−88.51
Quarry Island	Georgian Bay	2003	VG	1.11	3.48	3.06	3.84	44.83	−79.81
Musky Bay	Georgian Bay	2003	VG	1.15	3.48	2.95	3.73	44.81	−79.78
Oak Bay	Georgian Bay	2003	VG	1.23	2.98	3.11	3.61	44.79	−79.73
Moose Bay	Georgian Bay	2003	VG	1.35	3.31	3.07	3.82	45.07	−80.05
Cormican Bay	Georgian Bay	2003	VG	1.86	N/A	3.08	3.75	45.40	−80.31
Batchawana Bay	Superior	2004	VG	1.88	3.75	2.56	3.70	46.54	−84.31
Garden Channel	Georgian Bay	2003	VG	1.89	3.61	3.01	4.00	45.18	−80.12
Sandy Island	Georgian Bay	2003	EL	2.01	3.87	2.94	4.12	45.26	−80.25
Cloud Bay	Superior	2001	EL	2.14	3.38	4.40	3.74	48.08	−89.44
Moon River Bay	Georgian Bay	2003	EL	2.26	3.63	3.02	4.00	45.12	−79.97
Russell Island West	Huron	2005	EL	2.32	3.00	N/A	3.92	45.26	−81.70
Longuissa Bay	Georgian Bay	2003	EL	2.41	3.51	3.02	3.55	44.96	−79.89
Tadenac Lake	Georgian Bay	2005	EL	2.79	3.84	N/A	3.56	45.03	−79.95

Water quality column codes indicate: HD, highly degraded; VD, very degraded; MD, moderately degraded; GD, good; VG, very good; and EL, excellent.

The 12 environmental variables included in the calculation of the WQI were: physical (turbidity (TURB), total inorganic suspended solids (ISS), total suspended solids (TSS), and temperature (TEMP)), chemical (conductivity (COND) and pH (pH)), nutrient (total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), total ammonia nitrogen (TAN), and total nitrate nitrogen (TNN)), and chlorophyll *a* concentration (CHL). The formula represents the weighted sum of all Principal Components calculated using a Principal Component Analysis that included the 12 environmental variables in the formula and explained 100% of the variation in the dataset of 146 wetland samples (Chow-Fraser, 2006). The index scores range from −3 (indicative of the most impacted condition) to +3 (indicative of the most undisturbed site). For descriptive purposes, Chow-Fraser (2006) identified six categories to explain the relative condition of the wetlands: ≤2, "highly degraded"; −2 to −1, "very degraded"; −1 to 0, "moderately degraded"; 0 to +1, "good"; +1 to +2, "very good"; and >+2, "excellent", which we will use throughout the text.

The biotic index scores were calculated with presence data and species-specific *U* and *T* values for macrophytes, zooplankton, and fish species (Lougheed and Chow-Fraser, 2002; Seilheimer and Chow-Fraser, 2006, 2007) according to Eq. (2):

$$\text{index score} = \frac{\sum_{i=1}^{n} Y_i T_i U_i}{\sum_{i=1}^{n} Y_i T_i} \qquad (2)$$

where Y_i is the presence (present species = 1) or log 10 abundance ($\log(x+1)$) of species *i*, T_i is the value from 1 to 3 (indicating niche breadth), and U_i is the value from 1 to 5 (indicating tolerance of degradation). Individual species have been assigned *U* and *T* values based on the relationship between their occurrence and environmental variables, which was quantified using the Canonical Correspondence Analysis, multivariate analysis (ter Braak and Verdonschot, 1995; see Appendices A–C in supplementary information for published *U* and *T* values). The three biotic indices considered in this report were developed from large datasets that included a large number of wetlands in all of the Great Lakes (WMI: 154 wetland samples, 5 Great Lakes (Croft and Chow-Fraser, 2007); WZI: 70 wetland samples, 5 Great Lakes (Lougheed and Chow-Fraser, 2002; WFI: 100 wetland samples, 5 Great Lakes), Table 2). The *U* value is assigned to each species based on its

Table 2 – Total number of wetland-sites and their locations in the Great Lakes basin corresponding to the development of the water quality index (Chow-Fraser, 2006), wetland macrophyte index (Croft and Chow-Fraser, 2007), wetland zooplankton index (Lougheed and Chow-Fraser, 2002), and wetland fish index (Seilheimer and Chow-Fraser, 2007)

	Distribution of wetlands by lake			
	Water quality index	Wetland macrophyte index	Wetland zooplankton index	Wetland fish index
Index development				
Wetlands (*n*)	110	127	70	100
Lake Superior	18	21	7	15
Georgian Bay	18	44	7	32
Lake Huron	8	18	4	13
Lake Michigan	14	5	5	8
Lake Erie	17	26	8	8
Lake Ontario	35	40	23	24
Inland marshes	0	0	16	0

centroid (i.e., the center of a cluster of species scores in an ordination), along the synthetic degradation axis. Each species was assigned a weight that corresponded to its position on the axis of degradation, where 1 indicated most tolerant of degradation and 5 was most intolerant of degradation. The weighted standard deviations of the species scores (ter Braak and Smilauer, 1998) on the axis of degradation were used to indicate niche breadth and then used to assign the *T* values, where 1 indicated a wide niche breadth (e.g. found in a wide range of wetland conditions) and 3 indicated a narrow niche breadth (e.g. only found in a narrow range of wetland condition). Species having narrow niche breadths were indicative of specific environmental conditions and were more useful as indicator species. The score for the WMI, WZI, and WFI in each wetland ranges from 1 to 5, where scores of 1 are characteristic of poor wetland condition (e.g. high anthropogenic impacts), while scores of 5 are associated with excellent wetland condition (e.g. minimal anthropogenic impacts).

2.4. *Comparison of the WQI, WMI, WZI, and WFI*

Pairwise correlations and regression analyses (linear and polynomial) were conducted separately for the WMI, WZI, and WFI against the WQI, with the software program SAS JMP (version 5.1, SAS Institute Inc., Cary, NC). Similar regression analysis was used in the development of each index and there is a strong, significant relationship between increasing index scores and higher water quality (WMI: $r^2 = 0.58$, $P < 0.0001$ (Croft and Chow-Fraser, 2007); WZI: $r^2 = 0.25$, $P < 0.0001$ (Lougheed and Chow-Fraser, 2002); WFI: $r^2 = 0.66$, $P < 0.0001$ (Seilheimer and Chow-Fraser, 2007)). SAS JMP was also used to compare the slopes of the linear and polynomial regressions between the WMI and WFI with the analysis of covariance (ANCOVA). The alpha level for significance of all statistical tests was set at $P = 0.05$.

3. Results

3.1. *Index overlay*

The 32 wetlands in this study range from low WQI scores indicating ‘‘highly degraded’’ condition (e.g. high nutrients and suspended solids) to high WQI scores indicating ‘‘excellent’’ conditions (e.g. low nutrients and suspended solids; Table 1). WQI scores were higher in the northern than the southern wetlands (Fig. 1; Table 1), with the highest-quality wetlands primarily found in the northern Lakes (Superior, Georgian Bay, and Huron); only a single wetland from the northern Lakes had a negative WQI score (Matchedash Bay, 2002, 2003; Table 1). By comparison, four wetlands from Lake Erie and Lake Ontario had WQI scores corresponding to ‘‘good’’ quality (0–1), but the remaining sites in the south were considered degraded (Table 1). The regression of WZI against WQI was not significant ($r^2 = 0.04$, $P = 0.29$) but both regressions of WMI and the WFI against WQI were significant ($r^2 = 0.84$, $P < 0.0001$; $r^2 = 0.75$, $P < 0.0001$, respectively). An analysis of covariance indicated that the slopes of the two regression lines for WMI and WFI were significantly different (0.57 and 0.33, respectively, $P < 0.0001$; Fig. 2a).

3.2. *Relationship between WMI and WQI*

The WMI scores ranged from 1.00 (lowest possible score) to a high of 3.87 for the 32 wetlands in this study (Table 1). WMI scores increased linearly with WQI until a threshold of 2.0, where WMI values began to plateau. Therefore, we re-ran the regression with a polynomial fit and found that it explained slightly more variation ($r^2 = 0.86$) than did the linear regression ($r^2 = 0.84$; Fig. 2a). In degraded wetlands (those with WQI scores <0), macrophyte species richness tended to be lower than that in minimally disturbed wetlands (those with WQI > 0; Fig. 3). Wetlands with intermediate quality (those with WQI scores between 1 and 2) generally had a higher number of submergent species than that in wetlands of ‘‘excellent’’ quality.

3.3. *Relationship between WZI and WQI*

Linear regression of the WZI scores against WQI scores did not result in a significant relationship ($r^2 = 0.04$; $P > 0.29$; Fig. 2a). However, a significant relationship was obtained when this regression was conducted using a polynomial fit ($r^2 = 0.29$; $P = 0.01$; Fig. 2b). There was a positive relationship between WZI and WQI scores in degraded wetlands (WQI < 0) but a negative relationship for the minimally impacted sites (WQI > 1). Therefore, both degraded and undisturbed sites (with respect to water quality conditions) were associated with

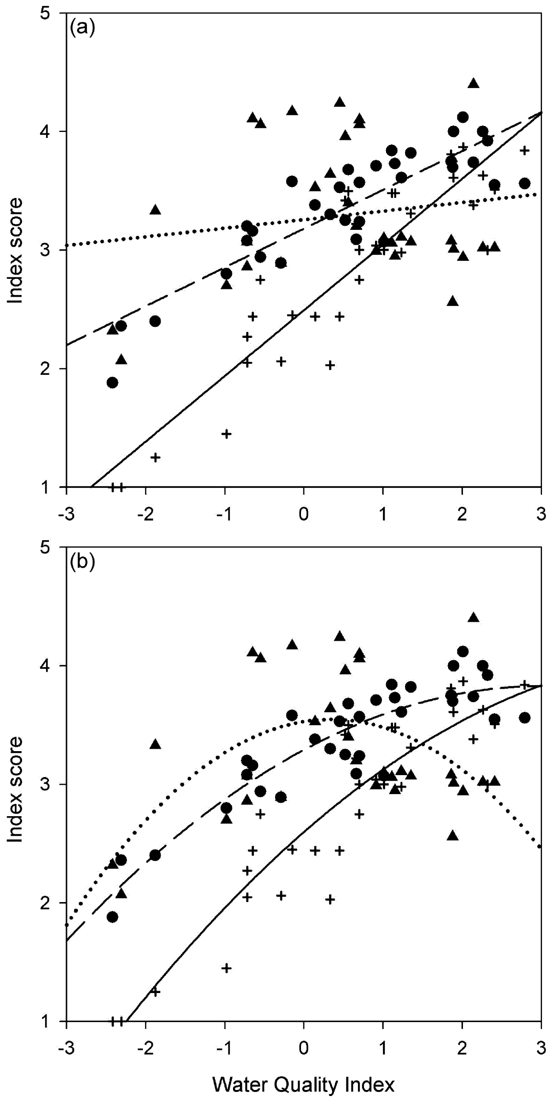

Fig. 2 – Overlay plot of the WMI (crosses with solid regression), WZI (triangles with dotted regression), and WFI (circles with dashed regression) against WQI for 32 wetlands sampled between the years of 2001 and 2005. (a) Regression of the WMI, WZI, and WFI with the WQI with linear fits (WMI, r^2 = 0.84, P < 0.0001; WZI, r^2 = 0.04, P > 0.29; WFI, r^2 = 0.75, P < 0.0001). (b) Regression of the WMI, WZI, and WFI with the WQI with polynomial fits (WMI, r^2 = 0.86, P < 0.0001; WZI, r^2 = 0.28, P = 0.01; WFI, r^2 = 0.81, P < 0.0001).

low WZI scores, while sites of intermediate quality had the highest scores.

3.4. *Relationship between WFI and WQI*

There was a more defined threshold in the relationship between WFI and WQI (Fig. 2b) than that which had been demonstrated for the WMI and WQI relationship (Fig. 2b). A polynomial regression produced a significantly better fit (r^2 = 0.81; $P < 0.0001$) and explained more of the residual variation than did the linear regression analysis (r^2 = 0.75, $P < 0.0001$; Fig. 1). Compared with the WMI–WQI relationship, there was a narrower range of WFI scores across the entire spectrum of WQI values, ranging from a low of 1.88 to a maximum of 4.12. This narrow range may be the result of index scores being suppressed within the category of best quality wetlands, while they have been inflated within the category of poor-quality sites (Table 1).

3.5. *WMI versus WFI scores as indicators of water quality*

Generally, the relationships between WMI (Fig. 2b) and WFI (Fig. 2b) with WQI are remarkably similar. The main difference is that WMI scores associated with the degraded sites are lower, which results in a larger range of WMI versus WFI scores over the same range of WQI values. A comparison of the polynomial regressions of the WMI and WFI with WQI indicated that the slopes were not significantly different (ANCOVA; P = 0.11) but the y-intercepts of the WMI was significantly lower than that of the WFI (P = 0.004). We probed further to uncover the reason for this disparity. It is clear that in highly degraded wetlands, there was little or no submergent species, and only a few floating or emergent taxa, all of which were indicative of polluted conditions (Fig. 3; U value of 1 (solid bars) in Fig. 4a). By contrast, there was a relatively large number of fish species in these same degraded sites (Fig. 4b), and although there had been more species of fish indicative of poor water quality (U values of 1 and 2), there had also been a relatively large number of species that were less pollution tolerant (U value of 3 or 4). This had the effect of inflating the WFI score in the highly degraded sites, relative to WMI scores. In fact, there was a consistent distribution of fish species with U value of 3 (diagonal bars in Fig. 4b) across the 6 water quality categories, and this also had the effect of lowering WFI scores in the "very good" and "excellent" wetland categories.

3.6. *WZI score as indicator of water quality*

The relationship between WZI and WQI was non-linear, with the highest WZI scores associated with wetlands in the "good" category (WQI score between 0 and 1). A high score in those wetlands with intermediate disturbance is the result of a large number of zooplankton with U value of 5 (most intolerant of pollution) and a T value of 3 (narrow niche breadth) (Fig. 5). There were fewer zooplankton in this category in both the highly degraded sites and in the minimally degraded sites. The reason for the absence of high-quality zooplankton in degraded sites is attributed to the absence of submergent plants, since most of the plankton in this group (e.g. *Simocephalus*, *Sida* (Appendix B in supplementary information)) are obligatorily associated with submersed aquatic vegetation. This same reason cannot be invoked to explain the low number of high-quality zooplankton in the pristine sites, since macrophyte species richness is relatively high in this category (Fig. 3). A more likely explanation is that there is a disproportionately high number of planktivorous fish in these high-quality wetlands (Fig. 6a). We speculate that planktivory by nursery fish in these sites is responsible for the lack of "high-quality" zooplankton in the pristine sites. As a comparison, we also plotted the abundance of carnivorous species (those

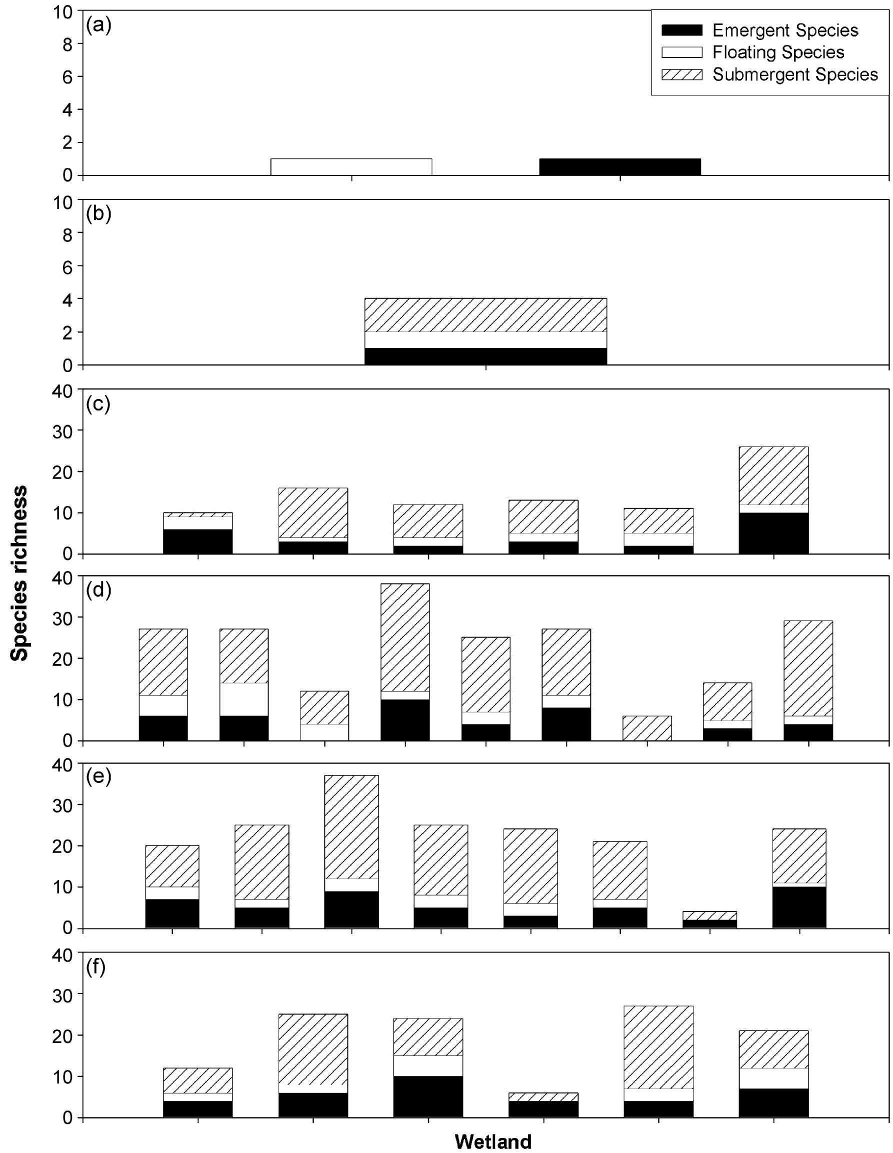

Fig. 3 – Species richness of emergent (solid), floating (open), and submergent (diagonal shading) plant species for 32 wetland-sites sampled between the years of 2001 and 2005 spanning 6 intervals of water quality conditions: (a) highly degraded, (b) very degraded, (c) moderately degraded, (d) good, (e) very good, and (f) excellent. Each bar represents a single wetland and are listed in the order they appear in Table 1, by water quality category.

that feed primarily on aquatic invertebrates and other animals) and found that they occurred in similarly high numbers in most of the water quality categories (Fig. 6b).

The total number of wetlands included in the development of these indices, and the distribution of sites along the Great Lakes shoreline may also affect the overall utility of the three indices. The WMI was developed from a relatively large group of coastal marshes from all five Great Lakes (127 sites; Table 2) and this may explain its better performance compared with the WFI (100 sites) and the WZI (70 sites). An additional problem with the WZI was the inclusion of a relatively large number of inland wetlands (Table 2), that had not been included in the development of the other indicators, and the lack of representation from Georgian Bay wetlands that are known to be of ''very good'' and ''excellent'' quality (Chow-Fraser, 2006).

4. Discussion

The direct comparison of the WFI, WZI, and WMI in a parallel dataset is an important step in identifying the strengths and weaknesses of each index within a range of water quality conditions (Griffith et al., 2005). This analysis made it apparent that different aspects of the plant, zooplankton, and fish ecology affected the relationship between the index scores and WQI scores. This will help wetland and aquatic managers choose an appropriate substitute for the WQI, based on their specific needs and initial water quality conditions. We have also shown that when developing biotic indices, it is important to have a large environmental gradient. Indices developed over a large environmental gradient and over a large spatial scale will also be more useful when comparing wetlands at the Great Lakes drainage basin level (Meador et al., 2003).

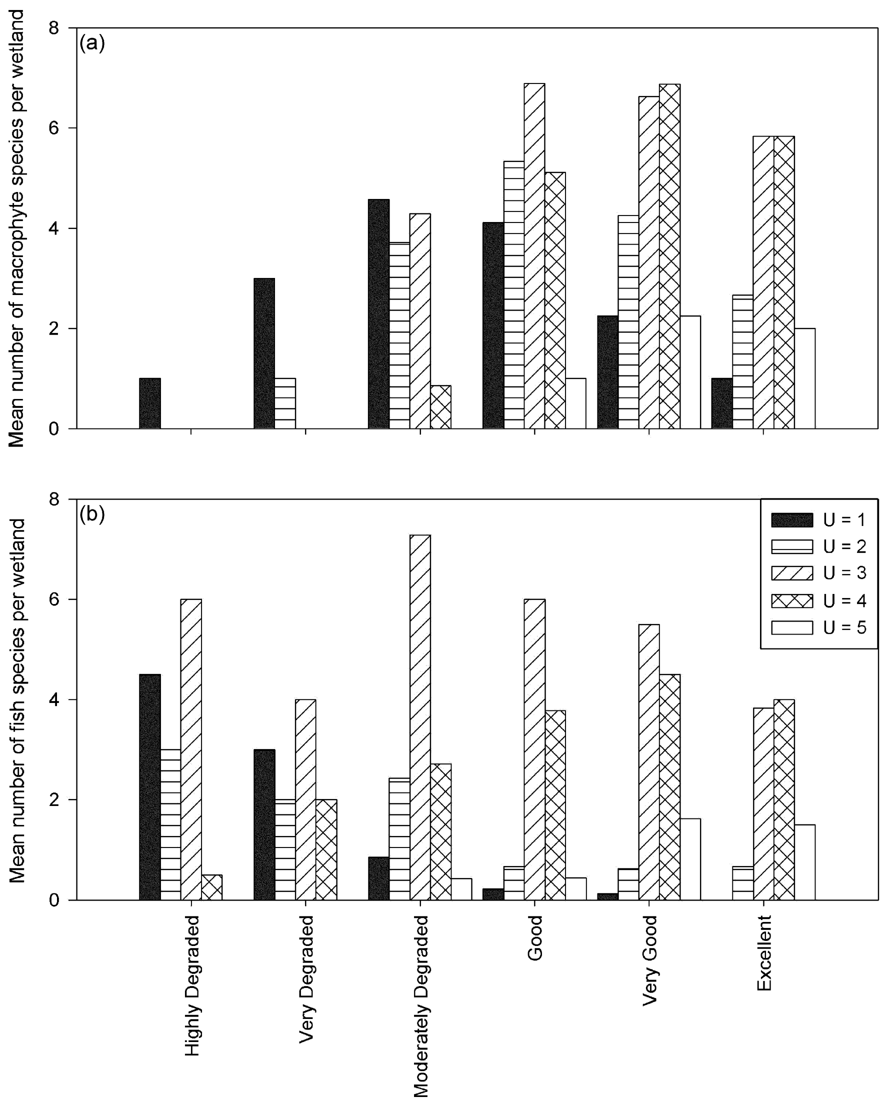

Fig. 4 – Mean number of species of (a) macrophytes and (b) fish in six water quality groups within five *U* values (1 = solid bar; 2 = horizontal line bar; 3 = diagonal line bar; 4 = crosshatched bar; and 5 = open bar), where *U* value of 1 correspond to species that are most tolerant to degraded conditions and *U* value of 5 correspond to species that are least tolerant of degraded conditions.

The WMI, WZI, and WFI are considered ''state indicators'' in the context of the SOLEC State of the Lakes Ecosystem Conference indicator program (Shear et al., 2003). State indicators do not specify the anthropogenic stressor, rather they focus on the cumulative environmental condition and are measured in relation to the entire suite of stressors (e.g. land use inputs, road development). Indicators may be divided into abiotic and biotic categories depending on the characteristics being measured. Abiotic variables, such as primary nutrients, can be easily and repeatedly measured, which is useful in monitoring programs. Total phosphorus has been used as an indicator of improved environmental condition as a result of the remediation of nutrient loads from watersheds of the Great Lakes basin (Neilson et al., 2003). Abiotic indices are sensitive to short-term changes in water quality, which is strongly linked to land use (Chow-Fraser, 2006). Often biological data are preferred by investigators and biotic indices are used for ecological monitoring. Relative to abiotic indices, biotic indices can tell more about longer-term conditions (i.e., water quality changes on short-term but species assemblages exhibit the cumulative impacts of stressors on a habitat over time). Thus a site may return to excellent water quality but lack a healthy biotic community because of other factors (i.e., contaminants not measured by WQI or changes in habitat that adversely affect community composition). Biotic monitoring has a further advantage because it can be performed at lower costs than other types of monitoring (Karr, 1991).

Plants are useful indicators of wetland condition and water quality (Miller et al., 2006) with great potential for use as indicators in Great Lakes coastal marshes (Albert and Minc, 2004; Croft and Chow-Fraser, 2007). The WMI was a more sensitive indicator of water quality impairment in the most degraded wetlands. It was associated with the lowest variance, and provided successful discrimination between

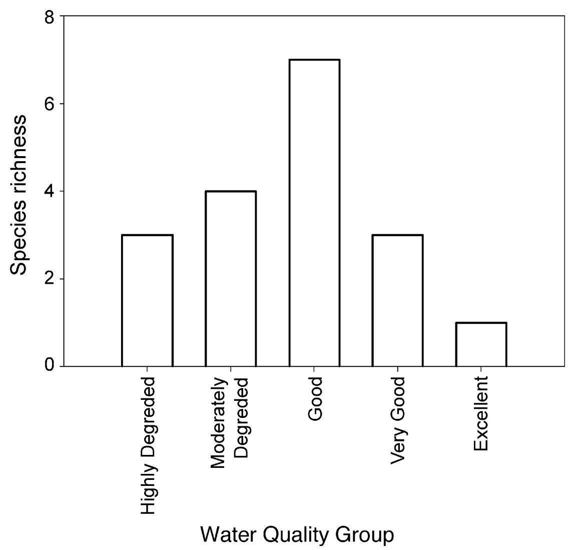

Fig. 5 – Mean number of different zooplankton species associated with excellent water quality conditions (*U* value = 5, and *T* value = 3), as defined by the WZI, across 5 different intervals of water quality for 31 wetland-sites sampled between the years of 2001 and 2005.

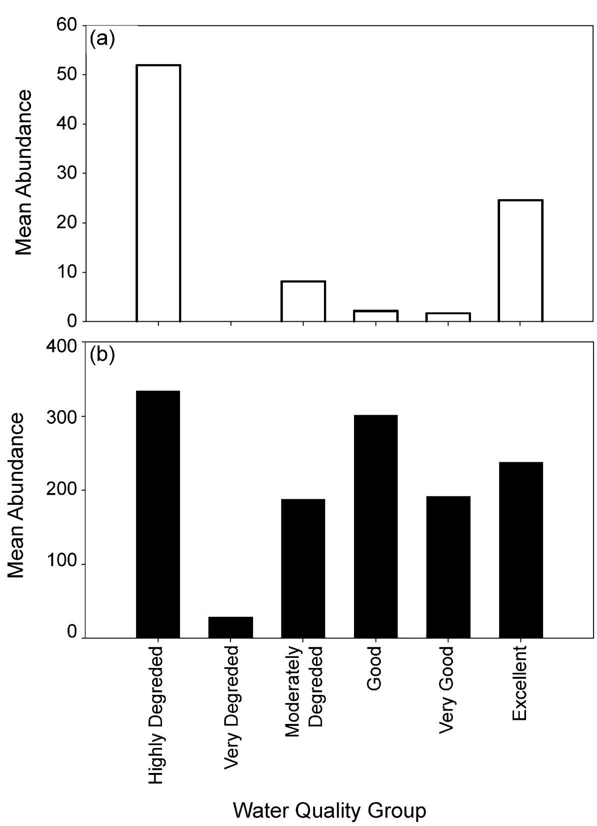

Fig. 6 – Mean abundance of (a) planktivorous fish species and (b) carnivorous fish species found in 6 different intervals of water quality for 32 sampled wetlands between the years of 2001 and 2005.

degraded and unimpaired wetlands. Its better performance as an indicator of environmental conditions may be because submergent plants are sessile and cannot migrate to more favorable habitat; hence, they are more reflective of the turbidity (Mahaney et al., 2004a,b) in the water than are zooplankton and fish. Sensitivity of the WMI compared to the other two indices may also be related to the comparatively large size of the database used to develop the index (154 wetlands distributed throughout the 5 Great Lakes) in comparison to the smaller databases used to develop the WZI (70 wetlands primarily in Lake Ontario with additional inland marshes) and the WFI (100 wetlands in all 5 Great Lakes).

Although it was adequate for wetlands ranging from highly degraded to good quality, the WZI is probably an unsuitable index for the minimally impacted sites found in Lake Superior, Georgian Bay, and northern Lake Huron. Since a low WZI score could be associated with either a highly degraded or an unimpaired site, the WZI would have to be modified before it can be applied to the "very good" or "excellent" wetlands. New assignment of *U* and *T* values may have to be given to zooplankton species and additional indicator species may have to be identified. The relationship between the density and coverage of aquatic plants may also be helpful for assessing the importance of aquatic macrophytes and zooplankton because only plant species richness was used in this study (rather than biomass or relative abundance). Planktivorous fish species such as the emerald shiner *Notropis atherinoides* and rainbow smelt *Osmerus mordax* were captured in high numbers in the "excellent" wetlands. These species are efficient predators of zooplankton (Scott and Crossman, 1998) and would not be inhibited by turbidity because the "excellent" wetlands had high water clarity. The "highly degraded" wetlands also had a high number of planktivores, which were primarily alewife *Alosa pseudoharengus*, a species that feed primarily on zooplankton (Becker, 1983). Additional studies need to be done to confirm the impacts of planktivorous fish predation on the performance of the WZI, and these may include exclusion experiments, and/or measurements of stomach contents for the relevant species of interest.

The most degraded wetlands had WFI scores that were higher than would be expected based on the WMI–WQI relationship. One difference between the WMI and WFI is that while there tends to be lower species richness of aquatic plants in degraded sites, the species richness of fish is usually high across all wetland conditions, even though the species composition of the fish community is changed (Chow-Fraser et al., 1998). The "highly degraded" and "very degraded" wetlands were associated with the most fish with corresponding *U* values of 1 and 2, and this contrasts the situation where "moderately degraded" wetlands had high number of species with *U* values of 1 and 2. Both WMI and WFI scores associated with wetlands in the "excellent" category leveled off, and this may have been due to the diluting effect of a relatively large number of species with intermediate *U* values. Species with a *U* value of 3 (Appendix C in supplementary information) are cosmopolitan species, like the yellow perch *Perca flavescens* and brown bullhead *Ameiurus nebulosus* that are found over a large range of environmental conditions (Brazner and Beals, 1997). It appears that the upper range of WMI and WFI scores are limited by the presence of ubiquitous species that can thrive in a wide range of environmental conditions including both degraded and un-degraded sites. When comparing the polynomial regressions of the WMI and WFI, we found that the slopes were statistically homogeneous, and thus the relationship between the two indices and the WQI is directly comparable.

These indices may be used by managers for ranking and comparing wetlands in a single area or larger basin. The WFI has been successfully used to differentiate sampling locations where there was a varying degree of impacts from an urbanized watershed within a single wetland using the seasonal fish community (Seilheimer et al., 2007). Further study is required but we expect that these biotic indices will also be useful for tracking changes in wetlands over time, which will be useful for identifying the large-scale impacts of climate change and other basin-wide stressors. Finally, these indices may be used for assessing historic wetland condition where existing datasets exist. To our knowledge, this is the first study to compare multiple biotic indices using data collected in the same wetlands with a full suite of water quality variables to determine how comparable the indices are to each other. All three biotic indicators included in this study were developed using a multivariate technique that used the response of species to water quality, with makes the indices useful proxies for water quality and wetland condition.

The logistics of implementing each of these indicators should also be taken into account by ecosystem managers before deciding which indicator would best suit their specific needs. The most cost effective indicator of the three is the WMI, which requires a minimal amount of equipment (e.g. boat), access to the wetland, and a technician trained in identifying plant species, which can be done with minimal training (U.S. EPA, 2002). The wetland fish index is more costly because of the cost of specialized nets for capturing the fish, but identification of the fish species can be readily accomplished with published resources (Becker, 1983; Scott and Crossman, 1998). The most expensive and time consuming of the three biological indicators is the WZI, which requires specialized equipment for the capture of the zooplankton (e.g. Schindler-Patalas zooplankton trap), preservatives (until it can be transported back to a lab), and a dissecting microscope for species identification. The most time effective indicator is the WMI, where field sampling and calculation of an index score can all be accomplished within a day of surveying. The WFI takes more time to implement because the fyke nets are typically set for at least 24 h before fish identification. The WZI is the most time consuming of all three indices because zooplankton samples must be transported back to a lab where they can be properly identified under a microscope, whereas macrophytes and fish are identified in the field. Although some are more cost/time effective than others, all three biological indicators are more feasible when compared to the WQI that requires multiple probes, technicians, costly lab equipment, and hours of sampling and laboratory processing.

5. Conclusions

Selection of appropriate indicators of water quality conditions will depend on a number of logistical considerations as well as existence of base-line information. For degraded sites, the WFI, WZI, or WMI all appear to be accurate indicators of water quality. The WZI is not appropriate for use in high-quality sites in its current version, but may be further developed and modified with inclusion of data from undisturbed sites. Since macrophytes are not mobile whereas fish are, the former may be better for discriminating among high-quality wetlands. If used under appropriate situations, all three indices are suitable indicators of water quality conditions, although the WZI is both more costly and time consuming to use than the other two. We therefore recommend the use of the WMI and/or the WFI in Great Lake coastal wetlands because they both successfully differentiate wetlands based on a large gradient of water quality conditions, and because historic species list of plants and fish tend to be more available than are zooplankton data.

Acknowledgements

We would like to acknowledge the assistance of Claire Robinson, Melanie Croft, Kristina Kostuk, Sheila McNair, and Vanessa Lougheed. Funding for this research was provided to PCF by the Great Lakes Fishery Commission and the Ontario Ministry of Natural Resources. Assistance from the Georgian Bay Association members was also essential to the research. Funds were also provided to TSS through the Norman S. Baldwin Fishery Science Scholarship from the International Association of Great Lakes Research and GLFC.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolind.2008.02.001.

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