

## DEVELOPMENT AND USE OF A ZOOPLANKTON INDEX OF WETLAND QUALITY IN THE LAURENTIAN GREAT LAKES BASIN

VANESSA L. LOUGHEED<sup>1</sup> AND PATRICIA CHOW-FRASER

McMaster University, Department of Biology, Hamilton, Ontario, Canada L8S 4K1

**Abstract.** Recent interest in biological monitoring as an ecosystem assessment tool has stimulated the development of a number of biotic indices designed to aid in the evaluation of ecosystem integrity; however, zooplankton have rarely been included in biomonitoring schemes. We developed a wetland zooplankton index (WZI) based on water quality and zooplankton associations with aquatic vegetation (emergent, submergent, and floating-leaf) that could be used to assess wetland quality, in particular in marshes of the Laurentian Great Lakes basin. Seventy coastal and inland marshes were sampled during 1995–2000; these ranged from pristine, macrophyte-dominated systems, to highly degraded systems containing only a fringe of emergent vegetation. The index was developed based on the results of a partial canonical correspondence analysis (pCCA), which indicated that plant-associated taxa such as chydorid and macrothricid cladocerans were common in high-quality wetlands, while more open-water, pollution-tolerant taxa (e.g., *Brachionus*, *Moina*) dominated degraded wetlands. The WZI was found to be more useful than indices of diversity ( $H'$ , species richness) and measures of community structure (mean cladoceran size, total abundance) for indicating wetland quality. Furthermore, an independent test of the WZI in a coastal wetland of the Great Lakes, Cootes Paradise Marsh, correctly detected moderate improvements in water quality following carp exclusion. Since wetlands used in this study covered a wide environmental and geographic range, the index should be broadly applicable to wetlands in the Laurentian Great Lakes basin, while further research is required to confirm its suitability in other regions and other vegetated habitats.

**Key words:** biomonitoring index; Laurentian Great Lakes basin; partial canonical correspondence analysis (pCCA); restoration; wetland zooplankton index (WZI); wetlands; zooplankton.

### INTRODUCTION

The utilization of biological monitoring in assessing ecosystem integrity has received more and more attention in recent years. A growing number of studies have reported on the use of fish (e.g., Karr 1981, Minns et al. 1994), macroinvertebrates (e.g., Lenat 1993, Burton et al. 1999), diatoms (e.g., Kelly and Whitton 1995), and periphyton (e.g., McCormick and Stevenson 1998) to detect habitat degradation in a variety of geographic locations and ecosystem types. In the Laurentian Great Lakes, the development and selection of indicators of ecosystem health was one of the primary challenges given to the scientific community during the 1996 State of the Lakes Ecosystem Conference (SOLEC; a biennial conference hosted by the Canadian and U.S. governments to examine issues concerning the Laurentian Great Lakes) (Bertram and Statler-Salt 1999).

Wetlands are ecologically and economically important ecosystems that are being rapidly lost and degraded. Whillans (1982) estimated that heavily settled areas of the lower Great Lakes have lost 75% of their wetland area, with some areas experiencing 100% loss. Many

of the remaining wetlands have become degraded by exotic benthivorous common carp (*Cyprinus carpio*) and changes in land use patterns over the past several decades, both of which have increased water turbidity and nutrient concentrations (Whillans 1996, Chow-Fraser 1998, 1999). To track these changes in ecosystem quality and quantity, managers require simple, robust indicators that can be applied across the basin by many environmental agencies. We propose to use zooplankton, and their association with macrophytes and relevant water quality variables, to indicate the quality of marshes in the Great Lakes basin.

Although zooplankton are known to respond quickly to environmental conditions (Schindler 1987), only a few attempts have been made to use the zooplankton community to indicate quality of aquatic ecosystems (Gannon and Stemberger 1978, Sládeček 1983, Stemberger and Lazorchak 1994, Gaiser and Lang 1998). Our rationale for choosing zooplankton was three-fold. First, there are well-documented plant-associated zooplankton taxa whose presence in a system is highly dependent on the presence of submersed macrophytes (Quade 1969, Paterson 1993). Hence, the taxonomic composition of the zooplankton community should reflect the presence and distribution of submergent plants in the wetland. Second, zooplankton taxa often have different preferences for trophic state (Berzins and Bertilsson 1989, Berzins and Pejler 1989) and water clarity

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<sup>1</sup> Present address: Department of Zoology, Michigan State University, East Lansing, Michigan 48824-1115 USA. E-mail: loughee2@msu.edu

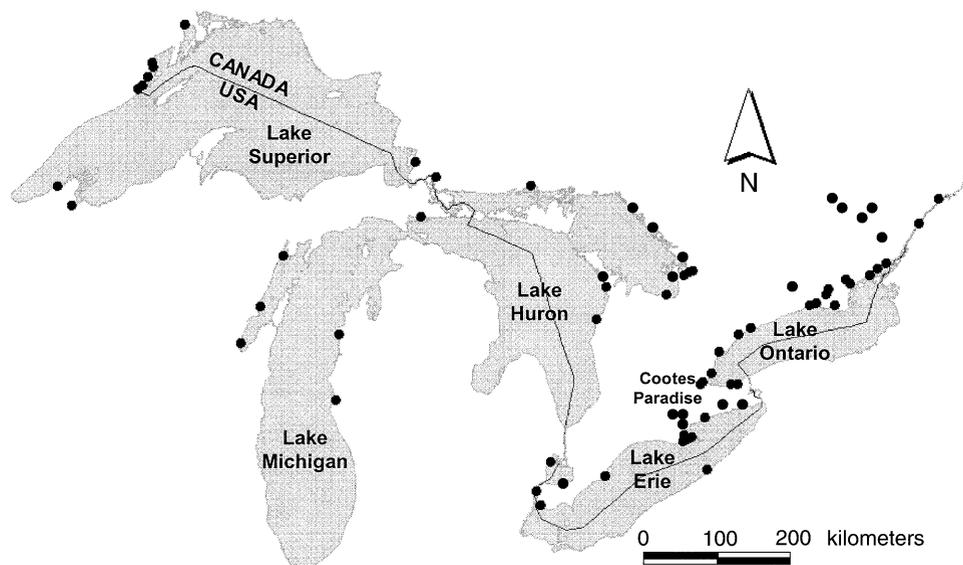


FIG. 1. Map of Great Lakes region showing the location of 70 wetlands sampled during 1995–2000, including the location of Cootes Paradise Marsh (Lake Ontario).

(Lougheed and Chow-Fraser 1998), and therefore species replacement will also occur with water quality degradation. Finally, since the species richness of submersed macrophytes declines as marshes become eutrophic and degraded (Crosbie and Chow-Fraser 1999; Lougheed et al. 2001), we hypothesize that the zooplankton community should reflect changes in the plant community as we proceed along the trophic–degradation gradient.

In general, biotic indices integrate the sensitivity of several taxonomic groups to environmental conditions (e.g., acidity, degree of degradation) into a single index value. These indices are particularly useful in situations where data must be statistically analyzed and conveyed to nonbiologists. In particular, a biotic index that could track changes over the short term (i.e., several years) would be a useful tool for evaluating restoration techniques such as the recent and costly exclusion of carp from Great Lakes coastal wetlands (e.g., Cootes Paradise Marsh in Lake Ontario, Metzgers Marsh in Lake Erie) (Lougheed et al. 1998, Wilcox and Whillans 1999). A common criticism is that these indices may oversimplify effects on the community in question, especially when organisms are grouped into higher taxonomic groups, where responses of individual members may vary greatly (Sládeček 1983, Gaiser and Lang 1998). However, we feel that these integrative indices are superior to measures that result in loss of species-specific information (e.g., diversity) or those that rely only on a select number of taxa (e.g., indicator species).

The objectives of this study were to develop a wetland zooplankton index (WZI) based on water quality and observed associations of zooplankton with aquatic vegetation that can be used to assess wetland quality; to compare this biotic index with commonly used in-

dicators of community quality such as the Shannon-Wiener species diversity index; and to evaluate the ability of the WZI to assess the response of a degraded coastal wetland of Lake Ontario (Cootes Paradise Marsh) to carp exclusion as a restoration technique.

#### METHODS

Water and zooplankton samples were collected from 70 marshes in the Great Lakes basin during 1995–2000. Wetlands ranged from St. Lawrence River sites within Ontario, down to the Windsor–Detroit area and Lake St. Clair, then up to Lake Superior (Fig. 1). Fifty-four of these were coastal marshes (lying within 2 km of Great Lakes shoreline or connecting channels, but not separated hydrologically from the lakes due to dams or waterfalls) of the upper (Lake Huron, Lake Michigan, and Lake Superior) and lower (Lake Ontario, Lake Erie) lakes, while the remaining 16 were inland wetlands located within the Great Lakes–St. Lawrence River basin. Coastal wetlands are subjected to unique hydrological phenomena relative to more protected inland systems; however, we included both coastal and inland systems in this study to (1) broaden our disturbance gradient since almost all coastal wetlands in southern Ontario have been disturbed to a certain degree, and (2) ensure the results were broadly applicable to both coastal and inland wetlands.

All wetlands were visited at least once in midsummer (mid-June–end of July), while a subset were also visited in early summer (May–mid-June) and/or late summer (August). Zooplankton samples collected in midsummer constituted two-thirds of all samples included in this study. To ensure consistent measurements of both water quality and zooplankton communities, both of which have been shown to be affected by storm

events (Krieger and Klarer 1991, Chow-Fraser 1999), wetlands were generally visited no sooner than 48 h following a rain event.

We characterized the plankton communities within four different habitat types in each wetland: open water 10 m from the outermost edge of aquatic plants (OPEN 10 m), open water 3 m from aquatic plants (OPEN 3 m), within submergent plant beds (>5 stems/m<sup>2</sup>; SUB) and within or immediately adjacent to emergent aquatic vegetation (EM). When submergent or emergent beds were absent or inaccessible, plankton below floating aquatics were sampled (FLTG). The species richness of submergent plants in each wetland was determined as described in Lougheed et al. (1998).

All water samples were collected from the middle of the water column at the OPEN 3-m site. Because of the large site-to-site, year-to-year, and seasonal variation in water levels, which are characteristic of Great Lakes coastal marshes (Maynard and Wilcox 1997, Chow-Fraser 1999), water depths in this study range 5–260 cm, depending on wetland site, time of year, and the sampling year in question. Water samples were analyzed according to standard methods (American Public Health Association 1992) for total phosphorus (TP), total nitrogen (TN) (sum of total kjeldahl nitrogen [TKN] and total nitrate nitrogen [TNN]), and total suspended solids (TSS). Following digestion by potassium persulfate, TP was analyzed according to Murphy and Riley (1962) and measured on a Milton Roy spectrophotometer (Thermo Spectronic, Rochester, New York, USA). Nitrogen analyses (TKN and TNN) were performed using Hach protocols and Hach reagents (Hach Company 1989) and measured on a Hach DR2000 spectrophotometer (Hach, Loveland, Colorado, USA). Planktonic chlorophyll-*a* (CHL-*a*) was filtered onto Whatman GF/C filters (Whatman, Clifton, New Jersey, USA) and extracted using 90% acetone with a one-hour extraction period. Absorbance measurements were made with a Milton Roy spectrophotometer, and results were corrected for phaeopigments by acidification. Temperature, pH, dissolved oxygen, and conductivity were determined using a H2O Hydrolab multiprobe and Scout monitor (Hydrolab, Austin, Texas, USA).

All zooplankton samples were collected from the middle of the water column at each of the habitat site types (where available). Zooplankton at open-water sites were collected using a five-liter Schindler-Patalas trap (Schindler 1969), while zooplankton in vegetation were collected using a one-liter beaker inverted three times into adjacent areas of vegetation for a total sample volume of three liters. All samples were filtered through 64- $\mu$ m mesh Nitex screen (Sefar America, Depew, New York, USA), back washed into 60-mL bottles and preserved in 4% sugar-formalin. Samples were thoroughly mixed and subsampled to obtain at  $\geq 100$  animals. Abundances and lengths of all animals present in the subsamples were recorded, but biomass could

not be determined because of the absence of published length-mass relationships for many of the taxa encountered. Copepods were identified to suborder only (i.e., cyclopoids, calanoids, harpacticoids), and therefore we report on their abundances, but exclude them from all subsequent, more species-specific analyses. Rotifer identification was based on Stemberger (1979), whereas crustacean identification was based on Balcer et al. (1984), Pennak (1989), and Thorp and Covich (1991).

All statistical analyses were performed using SAS.Jmp software (version 3.1.5, SAS Institute, Cary, North Carolina, USA), with the exception of canonical correspondence analysis, which was performed using CANOCO 4.0 (ter Braak and Smilauer 1998), and the Shannon-Wiener diversity index, which was calculated using MVSP (version 3.1, Kovach Computing Services, Ithaca, New York, USA).

#### *Multivariate statistical analyses*

Canonical correspondence analysis (CCA) is a useful technique when species data have been collected over a suitably large habitat range to display a unimodal relationship to the measured environmental variables (Jongman et al. 1995). It has been used by many authors to relate the distribution of zooplankton taxa to their environment (e.g., Hessen et al. 1995, Romo et al. 1996, Attayde and Bozelli 1998). Canonical correspondence analysis maximizes the separation of species optima along synthetic axes representing the primary environmental gradients in the data set. To confirm that CCA was an appropriate model for this data set, we first entered the species abundance data into a detrended correspondence analysis (CANOCO 4.0, not shown), which revealed sufficiently large gradient lengths (>4 SD) to indicate that zooplankton distributions in this study showed a clear unimodal response along the environmental gradient (ter Braak and Smilauer 1998). Then, because samples were collected over a four-month period, we performed a partial CCA (pCCA) instead of a regular CCA, which eliminated background variation due to the covariable time (i.e., day of the year; Jongman et al. 1995, ter Braak and Verdonschot 1995).

Environmental parameters entered into the pCCA analysis included continuous variables such as: TP, TN, TSS, chlorophyll-*a* (CHL-*a*), temperature (TEMP), dissolved oxygen (DO), pH, conductivity (COND), and latitude (LATITUDE; measured as decimal degrees), along with more qualitative variables that described habitat type: SUB, EM, FLTG (within submergent, emergent, and floating-leaf macrophyte beds, respectively), OPEN 3 m, and OPEN 10 m (open water 3 m and 10 m away from macrophyte beds, respectively). All continuous environmental variables were log<sub>10</sub>-transformed (excluding pH) to approximate normal distributions and standardized to zero mean and unit variance.



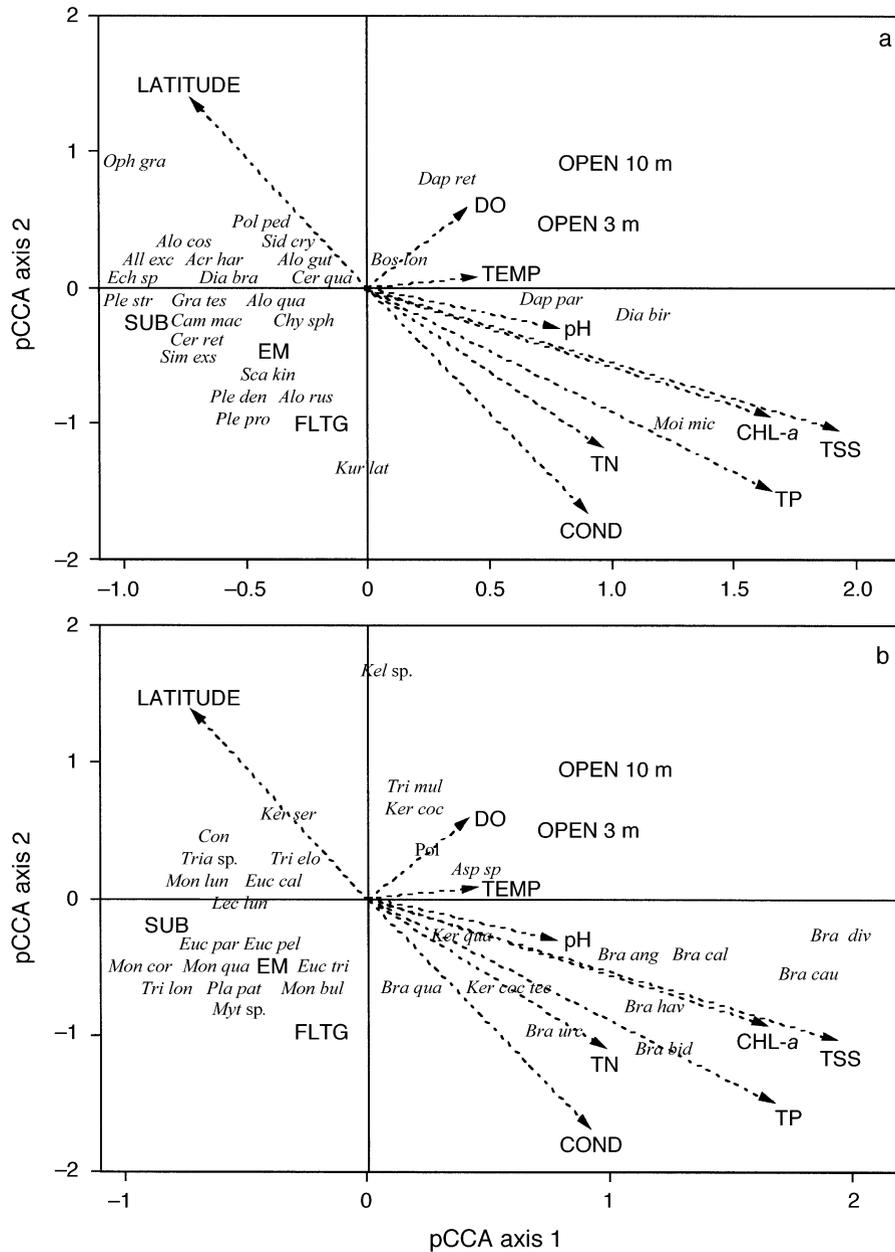


FIG. 2. Bi-plot of the partial canonical correspondence analysis (pCCA; axis 1 vs. axis 2) separated into (a) cladoceran, and (b) rotifer taxa. A complete list of cladocerans and rotifers observed in the 70 wetland sites appears in the Appendix.

*Bosmina longirostris*, and three chydorids (*Camptocercus macrurus*, *Chydorus sphaericus*, *Pleuroxus denticulatus*). There were few differences in the zooplankton communities observed in coastal vs. inland wetlands. Seven rare rotifers and seven rare cladocerans were found only in inland sites, while only three taxa (*Diaphanosoma birgei*, *Moina micrura*, *Brachionus caudatus*) that could be classified as common were absent from inland wetlands (see the Appendix).

Relative to the open water sites, the vegetated sites

tended to have greater species richness, abundances, Shannon-Wiener diversity ( $H'$ ), and larger mean size of cladocerans (Table 2). In particular, there was disproportionate representation of larger taxa in vegetated sites including Chydoridae, Daphnidae (but not *Daphnia*), Sididae, and Macrothricidae. By comparison, rotifers and *Bosmina* formed a greater proportion of organisms encountered at open-water sites. Both mature and immature forms of copepods tended to occur with greater abundances within the vegetation.

TABLE 1. Means and ranges (and in-text abbreviations) of environmental variables observed at 70 wetlands in the Great Lakes basin.

Variable	Mean	Range
Total phosphorus (TP; $\mu\text{g/L}$ )	117	13–959
Total nitrogen (TN; $\mu\text{g/L}$ )	2 911	572–13 000
Total suspended solids (TSS; $\text{mg/L}$ )	25.0	0.1–283
Chlorophyll- <i>a</i> (CHL- <i>a</i> ; $\mu\text{g/L}$ )	29.7	0–396
Temperature (TEMP; $^{\circ}\text{C}$ )	22.5	11.5–30
Dissolved oxygen (DO; $\text{mg/L}$ )	8.43	1.5–20
pH	7.6	6.0–9.4
Conductivity (COND; $\mu\text{S/cm}$ )	400	38–1387

#### Partial canonical correspondence analysis (pCCA)

Bi-plots of the first two pCCA axes are found in Fig. 2; for clarity, data corresponding to rotifers and cladocerans are shown in separate bi-plots (Fig. 2a, b). These figures explain 52% of the variation in zooplankton distribution along the first two synthetic environmental axes. The significance of each of the environmental variables in explaining the distribution of zooplankton species was assessed using the forward selection procedure of CANOCO. All variables were significant at the  $P < 0.05$  level. The most important predictors of zooplankton distribution, as indicated by their correlation with pCCA axis 1 were TSS (0.534), TP (0.531), CHL-*a* (0.445) and the presence of submergent vegetation (SUB) ( $-0.364$ ), or open water 3

m away (0.347). Because all water quality data were collected at the OPEN 3-m site, this axis does not reflect differences in water quality between open water and vegetated sites in any individual wetland; instead, it indicates that heavily vegetated wetlands have clearer water and a unique zooplankton community, whereas wetlands with sparse macrophyte distribution tend to be more turbid, and are associated with taxa that are more typically found in limnetic ecosystems. By comparison, the second axis of the pCCA, which distinguished urbanized wetlands, which occur primarily in southern Ontario, from more pristine northerly wetlands, was highly correlated with LATITUDE (0.435), COND ( $-0.521$ ), and TP ( $-0.374$ ).

To reduce clutter, we have plotted only the common (occurring in  $>10\%$  of wetlands) rotifer and cladoceran taxa on the bi-plots, using the first three letters of the genus and species as labels (Fig. 2). Species of the same rotifer genera tended to show similar environmental preferences along pCCA axis 1 (e.g., *Monostyla* in Fig. 2b) as did species of cladocerans (e.g., *Alona* in Fig. 2a). Wetlands that had clear water with substantial submersed aquatic vegetation tended to be associated with specific rotifer genera. For example, all eight species of *Monostyla* encountered in this study tended to occur within submersed macrophyte beds. Similarly, nine of 11 species of *Lecane*, six of seven species of *Euchlanis*, all five species of *Lepadella*, four of five species of *Trichocerca*, and less speciose taxa

TABLE 2. Comparison of means (1 SE) of zooplankton community structure at five habitat types.

Structure	OPEN 10 m	OPEN 3 m	EM	SUB	FLTG	Stats ( $P < 0.05$ )
<i>N</i>	55	105	94	68	10	
Species richness	7.78 (0.38)	8.82 (0.35)	11.61 (0.44)	12.71 (0.57)	12 (1.35)	S, E $> 10, 3$ ; F $> 10$
<i>H'</i>	1.08 (0.05)	1.17 (0.05)	1.48 (0.06)	1.58 (0.06)	1.50 (0.18)	S, E $> 10, 3$
Mean cladoceran size ( $\mu\text{m}$ )	291 (35)	388 (22)	522 (24)	526 (27)	457 (71)	S, E $> 10, 3$
Abundance (no. animals/L)						
Rotifers	492 (106)	251 (50)	195 (35)	257 (48)	221 (121)	NS
Cladocerans	138 (27)	157 (42)	603 (234)	211 (38)	328 (160)	S, E $> 3$
Copepods and copepodids	54 (18)	41 (9)	134 (26)	220 (40)	220 (81)	F, S, E $> 10, 3$ ; S $> E$
Nauplii	121 (25)	149 (39)	183 (37)	235 (50)	460 (253)	S $> 10, 3$
Abundance (%; excluding copepods)						
Rotifers	57 (5)	59 (3)	44 (3)	50 (3)	39 (9)	E $< 3$
Chydoridae	4 (3)	6 (2)	15 (2)	22 (2)	28 (6)	F, E, S, $> 3, 10$ ; S $> E$
Daphnidae	5 (3)	7 (2)	16 (2)	11 (2)	13 (5)	E $> 3, 10$
Sididae	1 (1)	2 (1)	3 (1)	3 (1)	5 (2)	NS
Macrothricidae	0	0	1 (1)	1 (1)	1 (1)	NS
Bosmididae	31 (4)	23 (3)	19 (3)	11 (3)	10 (8)	S $< 10, 3$
Other cladocerans	4 (0)	1 (0)	2 (0)	2 (0)	3 (0)	NS
WZI	2.85 (0.07)	3.10 (0.06)	3.50 (0.06)	3.77 (0.06)	3.49 (0.26)	S, E $> 3, 10$
WZI <sub>P-A</sub>	3.16 (0.08)	3.54 (0.07)	3.95 (0.06)	4.20 (0.04)	3.77 (0.23)	S, E $> 3, 10$ ; F $> 10$

Notes: Statistical comparisons (Stats) are the results of an analysis of variance followed by Tukey-Kramer multiple comparisons ( $P < 0.05$ ). Habitat sites are defined as follows: OPEN 10 m, open water 10 m from the outermost edge of aquatic plants; OPEN 3 m, open water 3 m from aquatic plants; EM, within or immediately adjacent to emergent aquatic vegetation; SUB, within submergent plant beds ( $>5$  stems/ $\text{m}^2$ ), FLTG, plankton below floating aquatics were sampled when submergent or emergent beds were absent or inaccessible. The habitat sites are abbreviated as 10, 3, E, S, and F, respectively, in the "Stats" column.

such as *Platyias* sp. and *Mytilina* sp. were commonly found associated with submergent vegetation. Species of cladocerans that were well represented in heavily vegetated wetlands included 25 of 28 species of chydorids encountered in this study, seven of eight species of macrothricids, *Simocephalus* sp., *Polyphemus pediculus*, *Sida crystallina*, *Diaphanosoma brachyurum*, *Scapholeberis kingi*, and *Ceriodaphnia* sp. By comparison, there were only a few taxa tolerant of highly degraded, sparsely vegetated wetlands, and these included all 10 species of *Brachionus*, *Filinia*, *Asplanchna*, *Polyarthra* and six of seven species of *Keratella*; cladoceran species that were found consistently in degraded wetlands included both species of *Moina*, all four species of *Daphnia*, and the bosminids, *Bosmina longirostris* and *Eubosmina coregoni*.

#### Development of wetland zooplankton index (WZI)

Before calculating the WZI, we reduced the total number of taxa in our data set by grouping species of the same genus according to their location on the first axis of the pCCA bi-plot (Table 3). Grouping taxa together has a practical benefit, because some taxa (e.g., chydorids) are difficult to key to species. An index based on genera would require less training and time than one requiring species-specific identification. There were, however, some taxa with unique habitat requirements that could not be grouped together. Although most of the chydorids were similarly intolerant of degraded conditions and occurred far to the left of the origin (0,0) on pCCA axis 1 (Fig. 2a), several (e.g., *Kurzia*, *Monospilus*, and *Leydigia*; Table 3) were more tolerant of degraded conditions, and accordingly their positions in the bi-plot occurred more centrally or further to the right-hand side of pCCA axis 1. These formed a separate group (Table 3).

Optimum ( $U$ ; center of distribution) and tolerance ( $T$ ; range of distribution) values were determined for each taxon based on a pCCA using these collapsed taxonomic groupings (Table 3). The location of a taxon on pCCA axis 1 represented its center of distribution along this primary synthetic axis formed by several key environmental variables. Details of the assignment of  $U$  and  $T$  are outlined in the *Methods*. In general, taxa tolerant of degraded conditions ( $U = 1-2$ ) had broader tolerance ( $T = 1$ ), while those less tolerant ( $U = 4-5$ ) often had narrower tolerances ( $T = 3$ ) (Table 3).

We calculated WZI values for zooplankton communities in each of the habitat types in the 70 wetlands. Aquatic vegetation was an important determinant of the WZI, since WZI values from sites containing submergent and/or emergent vegetation were significantly higher than those associated with open water (Table 2). In fact, the WZI increased significantly with the species richness of submerged macrophytes observed in each of these wetlands (Fig. 3;  $r^2 = 0.32$ ,  $P < 0.0001$ ), leveling off at  $\sim 10$  species of submergent plants and a WZI value of 4. This is not unexpected, given the

TABLE 3. Optimum ( $U$ ) and tolerance ( $T$ ) of zooplankton taxa.

Taxon/species	$U$	$T$
Rotifers		
<i>Anuropsis</i> sp.	3	1†
<i>Ascomorpha</i> sp.	1	1†
<i>Asplanchna</i> sp.	2	1
<i>Brachionus</i> sp.	2	1
<i>Cephalodella</i> sp.	3	1†
<i>Collotheca</i> sp.	5	2
<i>Conochiloides</i>	4	2
<i>Euchlanis</i> sp.	4	2
<i>Filinia</i> sp.	1	1
<i>Gastropus</i> sp.	2	1†
<i>Hexarthra</i> sp.	1	1†
<i>Kellicotia</i> sp.	3	3
<i>Keratella</i> sp.	3	1
<i>Lecane</i> sp.	5	2
<i>Lepadella</i> sp.	4	2
<i>Lophocaris</i> sp.	2	1†
<i>Macrochaetus</i> sp.	4	1†
<i>Monostyla</i> sp.	5	2
<i>Mytilina</i> sp.	5	3
<i>Notholca</i> sp.	3	1†
<i>Platyias</i> sp.	4	2
<i>Ploesoma</i> sp.	4	2
<i>Polyarthra</i> sp.	3	1
<i>Pompholyx</i> sp.	1	1†
<i>Scaridium</i> sp.	5	1†
<i>Testudinella</i> sp.	4	2
<i>Trichocerca</i> sp.	4	2
<i>Trichotria</i> sp.	5	2
Chydoridae		
<i>Kurzia latissima</i> ‡	4	2
<i>Leydigia leydigit</i> ‡	3	3
<i>Monospilus dispar</i> ‡	1	1†
<i>Monospilus dispar</i> ‡	2	1†
Macrothricidae		
<i>Illiocryptus sordidus</i> ‡	5	3
<i>Illiocryptus sordidus</i> ‡	1	1†
Daphnidae		
<i>Ceriodaphnia</i> sp.	4	2
<i>Daphnia</i> sp.	2	2
<i>Megafenestra</i> sp.	2	1†
<i>Scapholeberis kingi</i>	4	2
<i>Simocephalus</i> sp.	5	3
Sididae		
<i>Diaphanosoma birgei</i>	1	2
<i>D. brachyurum</i>	5	2
<i>Latona parviremis</i>	4	1†
<i>Latonopsis occidentalis</i>	5	1†
<i>Sida crystallina</i>	5	3
Bosminidae		
<i>Bosmina longirostris</i>	3	1
Additional species		
<i>Polyphemus pediculus</i>	5	3
<i>Leptodora kindti</i>	1	1†
<i>Moina</i> sp.	1	1
<i>Holopedium gibberum</i>	3	1†

† Taxa that occurred in <10% of wetlands were automatically given a tolerance value of 1.

‡ These species represent exceptions to the Chydoridae and Macrothricidae groupings.

significant relationship between submergent plants species richness and wetland water quality (Crosbie and Chow-Fraser 1999; Lougheed et al. 2001).

To compare the WZI with more standard measures of zooplankton community structure, we correlated it and four other indices (Shannon-Wiener diversity index

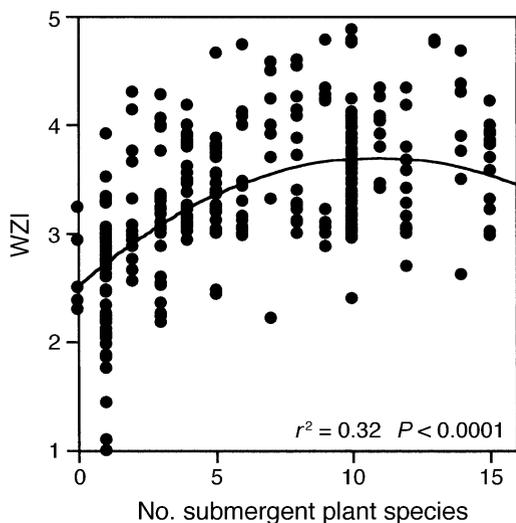


FIG. 3. Relationship between wetland zooplankton index (WZI) values and submergent plant species richness for 70 wetlands in the Great Lakes basin.

( $H'$ ), species richness, total zooplankton abundance, mean cladoceran size) with principal components analysis (PCA) axis 1 as the independent variable. The WZI correlated more highly with PCA axis 1 (Fig. 4a;  $r^2 = 0.25$ ,  $P < 0.05$ ), than did Shannon-Wiener diversity index,  $H'$  (Fig. 4c;  $P > 0.05$ ),  $\log_{10}$ (zooplankton abundance) (Fig. 4d;  $r^2 = 0.13$ ,  $P < 0.05$ ), mean size of cladocerans (Fig. 4e;  $P > 0.05$ ), or species richness (Fig. 4f;  $P > 0.05$ ). The WZI also consistently explained more variation in regressions with variables such as total phosphorus (TP;  $r^2 = 0.18$ ,  $P < 0.05$ ), CHL- $a$  ( $r^2 = 0.20$ ,  $P < 0.05$ ), and total suspended solids (TSS;  $r^2 = 0.19$ ,  $P < 0.05$ ), compared with the other measures.

We also used this principal components axis to compare the WZI calculated based on abundances vs. the index values calculated from presence/absence data ( $WZI_{P-A}$ ; Fig. 4b). The slopes of the lines were not statistically different (ANCOVA;  $P > 0.05$ ), but comparison of the intercepts indicated that the  $WZI_{P-A}$  values were  $\sim 0.4$  units greater than the corresponding WZI at any given PCA axis 1 value (intercept = 3.74 and 3.32, respectively; ANCOVA;  $P < 0.05$ ).

#### Case study: Cootes Paradise

Lougheed and Chow-Fraser (*in press*) described how water clarity increased and submersed macrophytes became established in shallow, sheltered, vegetated areas of Cootes Paradise Marsh (1997 and 1998) after most of the large carp were excluded in the spring of 1997. These changes at the vegetated site were concomitant with a significant reduction in edible algal biomass and increased representation of large zooplankton grazers and substrate-associated cladocerans. By contrast, open-water areas in the marsh remained turbid and de-

void of vegetation and only exhibited temporary or nonsignificant changes in water clarity; there were also fewer changes in the zooplankton community at the open-water sites.

We calculated WZI and  $WZI_{P-A}$  values for three different sites in the marsh (i.e., vegetated, lagoon, and open-water; see *Methods*) for each of two years prior to carp exclusion (1993 and 1994) and two years following carp exclusion (1997 and 1998). Fig. 5 compares the  $WZI_{P-A}$  values calculated for each site-year combination in Cootes Paradise to other Lake Ontario coastal marshes. Trends were no different if the WZI was used as opposed to the  $WZI_{P-A}$ . There were no consistent significant differences seen between both pre- and postexclusion years at the open water sites, although the  $WZI_{P-A}$  tended to be higher in 1998 than in one (lagoon) or both (open-water) pre-exclusion years. By comparison, the mean for the vegetated site increased significantly in both years following exclusion (ANOVA; Tukey-Kramer,  $P < 0.05$ ). Despite these significant increases, however, the average summer index values did not generally exceed the mean for all other Lake Ontario wetlands and were still considerably lower than the  $WZI_{P-A}$  value observed at a nearby reference wetland (Fig. 5).

While the WZI correctly detected moderate improvements in water quality following carp exclusion, there were no significant differences observed at any of the sites between pre- and postexclusion years with respect to species richness, cladoceran body length or rotifer and cladoceran abundance. In fact, contrary to expectations, the Shannon-Wiener index of diversity was significantly reduced at the vegetated site following carp exclusion (ANOVA; Tukey-Kramer,  $P < 0.05$ ).

#### DISCUSSION

We developed an index based on water quality, macrophyte habitat, and distribution of zooplankton that can be used to evaluate wetland quality across the Great Lakes basin. This index design method could be extrapolated to other systems and thus aid in tracking restoration and degradation of vegetated aquatic habitats. Because the database we used to develop this index included a broad range of wetland quality (Table 1), wetland types (i.e., lacustrine, riverine, palustrine), and geomorphological settings (e.g., open shoreline, river delta, barrier beach) (Maynard and Wilcox 1997, Chow-Fraser and Albert 1999), we are confident that it will have wide applicability.

Wetlands in our database fall on a continuum: at one extreme are turbid, eutrophic, and algae-rich systems with few, if any, submersed macrophytes; at the other extreme are high-quality wetlands that are relatively clear, oligotrophic with abundant and diverse submergent macrophyte beds (Table 1; Lougheed et al. 2001). This gradient in wetland quality was accompanied by a predictable change in the taxonomic composition of the zooplankton community, as indicated by the pCCA,

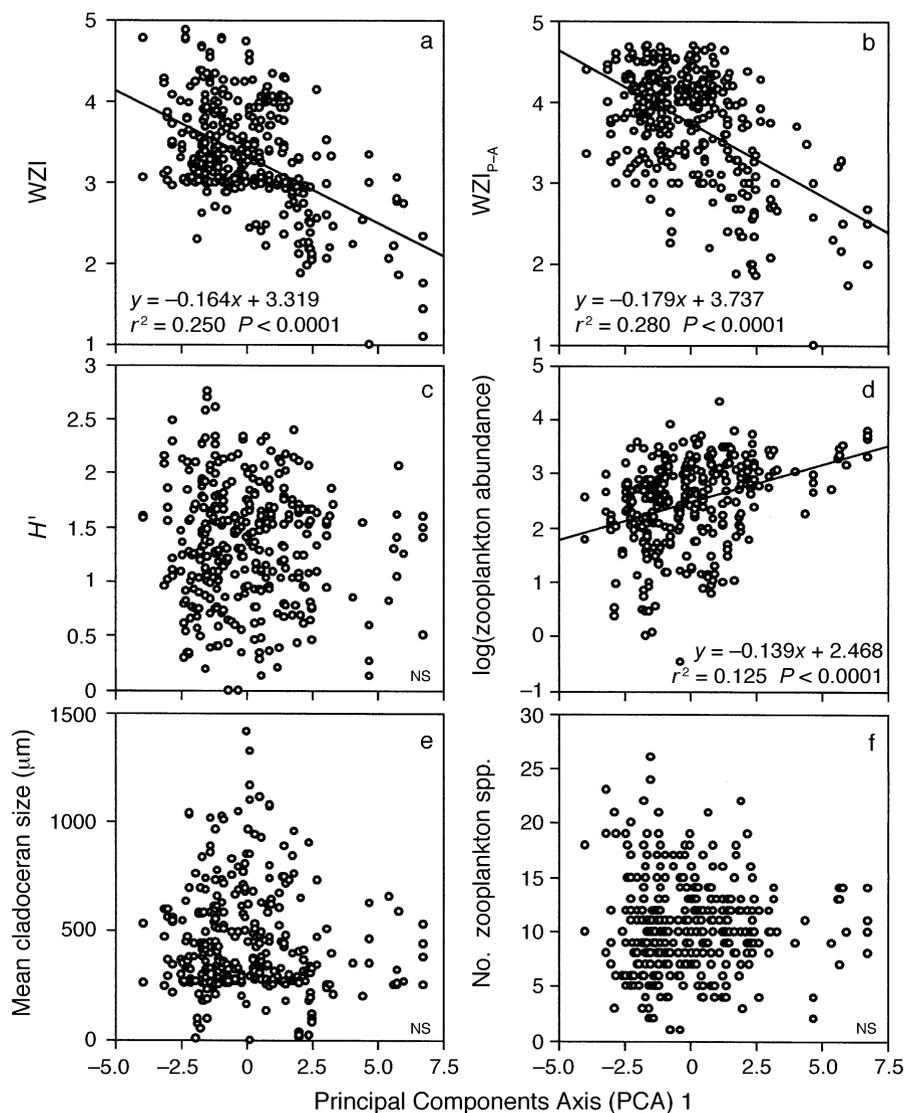


FIG. 4. Role of principal components axis 1 in explaining variation in several measures of zooplankton community structure: (a) WZI, (b)  $WZI_{P-A}$ , (c)  $H'$ , (d)  $\log(\text{total zooplankton abundance})$ , (e) mean size of cladocerans ( $\mu\text{m}$ ), and (f) species richness of rotifers and cladocerans.

which showed that plant-associated taxa dominated high-quality wetlands while more pollution-tolerant taxa dominated degraded wetlands. Results from the few published studies on zooplankton in Great Lakes wetlands (Krieger 1992) are consistent with our findings. For example, Krieger and Klarer (1991) and Lougheed and Chow-Fraser (1998) also found *Diaplanosoma birgei*, *Moina micrura*, and Bosminids occurring in degraded wetlands of the lower lakes, while other authors found plant-associated cladocerans such as macrothricids, chydorids, *Simocephalus*, and *Sida* occurring in relatively pristine Lake Huron (Cardinale et al. 1998, Gathman et al. 1999) and Lake Erie wetlands (Campbell 1993, Botts 1999). Similar species

trends have also been reported in prairie pothole wetlands (Gaiser and Lang 1998).

The 1987 revision to the Great Lakes Water Quality Agreement (GLWQA) acknowledges the importance of biological systems in the monitoring and assessment of the Great Lakes ecosystem and endorses an ecosystem approach. The selection of appropriate indicators of ecosystem health for coastal wetlands of the Great Lakes is required to accurately illustrate that changes in environmental quality are accompanied by ecological changes. The wetland zooplankton index (WZI) is a more useful indicator of wetland degradation and restoration than indices of diversity ( $H'$ , species richness) and measures of community structure (mean size,







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#### APPENDIX

A complete list of cladocerans and rotifers observed at 70 wetlands in the Great Lakes basin is available in ESA's Electronic Data Archive: *Ecological Archives* A012-004-A1.