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Use and Development of the Wetland Macrophyte Index to Detect Water Quality Impairment in Fish Habitat of Great Lakes Coastal Marshes

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ABSTRACT. *Indices have been developed with invertebrates, fish, and water quality parameters to detect the impact of human disturbance on coastal wetlands, but a macrophyte index of fish habitat for the Great Lakes does not currently exist. Because wetland macrophytes are directly influenced by water quality, any impairment in wetland quality should be reflected by taxonomic composition of the aquatic plant community. We developed a wetland macrophyte index (WMI) with plant presence/absence data for 127 coastal wetlands (154 wetland-years) from all five Great Lakes, using results of a canonical correspondence analysis (CCA) to ordinate plant species along a water quality gradient (CCA axis 1). We validated the WMI with data collected before and after the implementation of remedial actions plans (RAPs) in Sturgeon Bay (Severn Sound) and Cootes Paradise Marsh. Consistent with predictions, WMI scores for Sturgeon Bay were significantly higher after the implementation of the RAP. Historical data from Cootes Paradise Marsh were used to track the declining condition of the plant community from the 1940s to 1990s. Subsequently, when remedial actions had been implemented in 1997, the calculated WMI scores showed improvement, but when the presence of exotic species (WMIadj) was accounted for, improvements in ecological integrity of the aquatic-plant community were no longer evident. We show how WMI scores can be used by environmental agencies to assess the historic, current, and future ecological status of wetland ecosystems in two Canadian national parks, Point Pelee National Park (PPNP) and Fathom Five National Marine Park (FFNMP).*

INDEX WORDS: *Coastal wetlands, environmental index, water quality, macrophyte, Great Lakes, wetland macrophyte index.*

INTRODUCTION

Coastal wetlands provide critical spawning and nursery habitat for the Great Lakes fish community (Jude and Pappas 1992, Wei *et al.* 2004) as well as valuable habitat for both migratory and nesting birds (Maynard and Wilcox 1996). Approximately 60 to 80% of the coastal wetlands of the Great Lakes have been lost since the arrival of European settlers (Smith *et al.* 1991, Ball *et al.* 2003). The rapid rate of wetland loss and associated services makes it imperative that high-quality sites be identified and conserved before they are further degraded and/or destroyed. To achieve this goal, managers of environmental agencies must be provided with appropriate indicators of ecosystem health which could be used in routine monitoring.

Wetland degradation in the Great Lakes basin has been attributed to a variety of human disturbances, including increased loading of nutrients and sediment from agricultural and urban development (Chow-Fraser 1998, Crosbie and Chow-Fraser 1999, Lougheed *et al.* 2001), introduction of invasive species (Lougheed *et al.* 1998), and shoreline development and recreational activities (Chow-Fraser 2006). The extent to which these factors contribute to marsh degradation depends on the type of wetland. For example, coastal marshes located at the mouth of rivers and estuaries are susceptible to altered land uses in their watersheds, and many in Lakes Ontario and Erie have become turbid, eutrophic systems which limit species composition of submergent macrophytes (Lougheed *et al.* 2001, McNair and Chow-Fraser 2003). Changes in the submergent community are known to affect com-

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munities of zooplankton (Lougheed and Chow-Fraser 2002), benthic invertebrates (Kashian and Burton 2000, Kostuk 2006), and fish (Minns *et al.* 1994, Seilheimer and Chow-Fraser 2006). Because water clarity and nutrient levels in coastal marshes have overriding influence on subsequent trophic levels, Chow-Fraser (2006) developed the water quality index (WQI) to measure the degree of degradation attributable to human activities. This index includes six categories that range from highly degraded (index score of –3) to excellent (index score of +3) and has been used successfully to rank 110 wetlands throughout the Great Lakes shoreline according to their degree of water quality impairment (Chow-Fraser 2006). WQI scores were significantly correlated with the proportion of altered (agricultural and urban) land in watersheds, and this has been confirmed as a major determinant of water quality conditions for other Great Lakes coastal ecosystems (Danz *et al.* 2005).

Despite effectiveness of the WQI as a monitoring tool, the effort required to measure all 12 water quality parameters (i.e., physical characteristics, various forms of major nutrients, suspended solids, and chlorophyll concentrations), renders it unlikely to be adopted by most environmental agencies. This is a major reason for the recent development of biotic indices using zooplankton (wetland zooplankton index [WZI]) (Lougheed and Chow-Fraser 2002), periphytic algae (McNair and Chow-Fraser 2003), benthic invertebrates (Kostuk 2006), and fish (wetland fish index [WFI]) (Seilheimer and Chow-Fraser 2006). Interest in developing biotic indices for wetlands has also been evident elsewhere (Cardinale *et al.* 1998, van Dam *et al.* 1998, Kashian and Burton 2000, Wilcox *et al.* 2002, Tangen *et al.* 2003, Uzarski *et al.* 2004).

Even though the relationship between water quality and aquatic vegetation in coastal wetlands of the Great Lakes has been well-studied (Lougheed *et al.* 2001, McNair and Chow-Fraser 2003, McNair 2006), no basin-wide biotic index of anthropogenic disturbance based on aquatic wetland plants for the Great Lakes has emerged. This is surprising considering the number of advantages in using plants as a biotic indicator. First, because wetland plants are essentially non-motile, their distribution can be georeferenced on each sampling occasion and changes in distribution can be tracked over time. Second, compared with fish surveys that require either an electrofishing boat or series of paired fyke nets (Seilheimer and Chow-Fraser 2006), plant surveys can be accomplished without specialized and expensive equipment, and with only one or two trained personnel in waders and/or canoe. Unlike fish and zoobenthos surveys that require overnight traps, most plant surveys can be completed in a day. Additionally, results are available immediately with limited need for further processing such as surveys for macroinvertebrates, zooplankton, or periphyton.

Indices have been developed with wetland plants that focus on upland vegetation such as wet meadow and emergent species related to bird and mammal habitat (e.g., Wilcox *et al.* 2002, Albert and Minc 2004), but with less emphasis on the submergent, floating, and aquatic emergent species which are better related to fish habitat. The development of a cost-effective index that can be used to indicate the degree of anthropogenic impact and the resultant influence on fish habitat can be an important contribution for conservation and management.

Here, we show how aquatic vegetation can be used in the WMI to indicate human-induced degradation of coastal marshes in all five Great Lakes. The methodology for the development of the WMI is based on previous papers that relate zooplankton (Lougheed and Chow-Fraser 2002) and fish (Seilheimer and Chow-Fraser 2006) to environmental variables using canonical correspondence analysis (CCA). The use of CCA to develop plant indices is prevalent in Europe (e.g., Dodkins *et al.* 2005), but has not been widely used in North America. The WMI assumes aquatic plants (all species growing obligately in flooded areas but excluding those typically associated with wet meadows) will respond directly (through competition for light and nutrients) or indirectly (through food-web interactions) to changes in water quality conditions. We show that response to the degree of water quality impairment is reflected in the taxonomic composition of the aquatic plant community. We validate the WMI by choosing two sites that have undergone rehabilitation as part of a Great Lakes remedial action plan (RAP) program (Cootes Paradise Marsh in the Hamilton Harbour RAP and Sturgeon Bay in the Severn Sound RAP, [Hartig 1993]), and for which there exist plant species lists corresponding to conditions before and after RAP initiatives. We also show how the WMI can be used to build a monitoring program to track changes in quality of wetland habitat in two Canadian national parks which vary greatly in their current ecological status. By testing the usefulness of the WMI, we will show that the WMI is versatile, easy to use, and a sensitive indicator of anthropogenic impact on coastal wetlands.

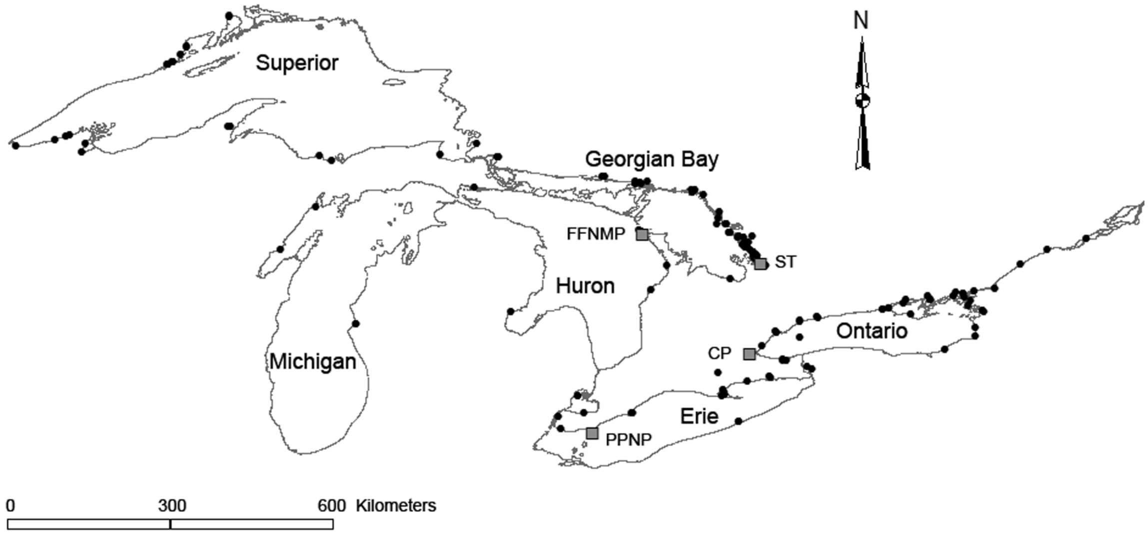

FIG. 1. Location of 176 wetland years used in the application of the WMI. Location of the four study sites used for validation of the WMI are indicated by square symbols. FFNMP = Fathom Five National Marine Park, ST = Sturgeon Bay, CP = Cootes Paradise, and PPNP = Point Pelee National Park.

METHODS AND MATERIALS

Description of Study Sites

RAP sites

The first RAP site is Cootes Paradise Marsh, which is located on the western end of Lake Ontario. It is subjected to multiple stressors (Wei and Chow-Fraser 2005), including urban run-off, nutrient and sediment inputs from the Dundas Sewage Treatment Plant (see Fig. 1) (Chow-Fraser *et al.* 1998), and feeding and spawning activity of benthivorous common carp (*Cyprinus carpio*) (Lougheed *et al.* 1998). Several initiatives were implemented to restore Cootes Paradise Marsh as part of the Hamilton Harbour RAP; a revegetation program began in 1994 (Chow-Fraser 1999b), followed by a marsh-wide carp exclusion program which began in the spring of 1997 (Lougheed *et al.* 2004, Chow-Fraser 2005). Chow-Fraser (2006) demonstrated that water quality in the marsh improved from the “Highly Degraded” category to the “Very Degraded” category from 1994 to 1998. By 2002, conditions were approaching “Moderately Degraded;” however, because the high nutrient and sediment concentration in the marsh is associated with both external and internal loading (Chow-Fraser 1999a, Kelton and Chow-Fraser 2005), it is unlikely further improvements could be expected without additional human interventions.

The second site is Sturgeon Bay, which is one of several bays included for remedial actions as part of the Severn Sound RAP. It is located in the southern end of Georgian Bay and was identified by the International Joint Commission as an area of concern (AOC) because of excessive algal growth (see Fig. 1). A RAP was implemented in 1989 to reduce nutrient inputs into Severn Sound from eight surrounding sewage treatment plants (Sherman 2002). The Victoria Harbour Pollution Control Plant, which empties into Sturgeon Bay, began its operation as a tertiary facility in 1985. Environmental conditions have improved sufficiently that in 2003, the Severn Sound RAP was delisted.

Canadian national parks

The two Canadian national parks in this study are Fathom Five National Marine Park (FFNMP) and the Point Pelee National Park (PPNP). Fathom Five National Marine Park is located at the northern tip of the Bruce Peninsula, at the junction between Lake Huron and Georgian Bay. At this site development of coastal wetlands is extremely limited because of exposure to wind and wave action. The park includes several remote wetlands on Cove and Russel islands, which receive relatively little human disturbance, compared with several on the mainland near the town of Tobermory (see Fig. 1). Because FFNMP has become one of the most popular destinations in Canada for freshwater recreational diving, increased tourism and cottage development in the area is threatening the integrity of the fragile coastal wetlands, especially those located on the mainland.

The Point Pelee National Park is located on the north shore of Lake Erie, near the town of Leamington, and includes an island which has distinction of being the southernmost point of Canada. The half a dozen wetlands of PPNP are separated by a sand barrier from Lake Erie, but are hydrologically connected to the lake when high water levels cause the barrier to breach. Since the creation of the park in 1918, PPNP has been a popular destination for tourists and cottagers, as well as being a vital stopover for migratory birds and waterfowl. The park has been impacted by agricultural run-off to the north, and from the 1940s to the 1970s was severely affected by activities of many tourists and cottagers. Over the past three decades, managers of PPNP have gradually removed all cottages and administrative buildings, and greatly reduced vehicular use and camping within the park, to permit the wetlands and other natural areas to recover.

Description of Overall Field Monitoring Programs

Over 200 wetland-years had been sampled between 1996 and 2005 specifically for development of ecological indicators for Great Lakes coastal wetlands (e.g., Lougheed and Chow-Fraser 2002, Chow-Fraser 2006, Seilheimer and Chow-Fraser 2006). These wetlands cover environmental conditions ranging from hypereutrophic Lake Erie sites that are agriculturally impacted, to wetlands in remote islands of Georgian Bay and several that are located in nature reserves of eastern Georgian Bay with limited human impact over a century. A total of 154 wetland-years with a complete suite of water quality variables sampled between 1998 and 2005 were chosen for development of the WMI (Fig. 1).

Field Sampling for Development of the WMI

All water sampling and measurements of physical and chemical parameters were conducted from a canoe or boat (depending on depth of the water). We measured temperature, pH, specific conductivity, and dissolved oxygen concentration *in situ* with several types of probes. A Hydrolab™ H20 equipped with a Scout monitor (Hydrolab, Austin, Texas) was used prior to 2000; during 2000 and 2001, a Hydrolab™ Minisonde and Surveyor (Hydrolab, Austin, Texas) was used, and from 2002-2005 a YSI™ 6600 probe with 650 display was used (YSI, Yellow Springs, Ohio). A comparison was done among all three probes in 2001, and we found no significant differences for any of the above parameters (Chow-Fraser 2006). All sensors for the instruments were calibrated on a weekly basis. Daily calibration was not feasible due to the remoteness of many of our sampling sites. Sampling was always conducted during daylight hours (generally between 0900 and 2000). After 2000, coordinates of the sites were taken with either a Trimble™ GPS (4–5 m accuracy) or a Garmin™ Etrex GPS (4–6 m accuracy). For sites sampled prior to 2000, coordinates were taken from published sources (Crosbie and Chow-Fraser 1999, Lougheed and Chow-Fraser 2002).

Water was collected for nutrient and turbidity analysis in 1-L van Dorn bottles at mid-depth in water outside the submergent plant zone. Water for nutrient analysis was dispensed into clean Nalgene™ bottles (acid-washed and rinsed with deionized water), while those for chlorophyll analysis was dispensed into opaque Nalgene™ bottles. All samples were stored on ice in a cooler and were analyzed later that day. Turbidity samples were collected in an identical manner and were measured in the canoe, with a Hach™ 2100 Portalab. Methods used for processing samples in the field and the laboratory have been documented in detail elsewhere (Lougheed *et al.* 1998; Chow-Fraser 1999a, 2006). The final list of nutrient and suspended-solids variables included were: pH, total phosphorus (TP), organic suspended solids (OSS), inorganic suspended solids (ISS), total nitrate-nitrogen (TNN), total ammonia nitrogen (TAN), conductivity (COND), and chlorophyll (Chl).

Aquatic Plant Surveys

Development of the WMI

Plant data used for development of the WMI were collected between 1998 and 2005, although the majority were collected between 2000 and 2005. On each sampling occasion in a wetland, the aquatic plant community was surveyed as follows (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 the majority of these were surveyed with quadrats (0.75 m × 0.75 m) from a canoe or boat, within the vicinity where fyke nets had been set contemporaneously to survey the fish community. Depending on the size and complexity of the wetland, these surveys would take from 20 min to several hrs to complete. Generally, 10 to 15 quadrats would be sampled in each wetland and only the occurrence of species was noted—i.e., we did not estimate percent cover of particular species within the quadrats. At least ten quadrats were sampled in each wetland, and after that point the sampling would cease when no new species were found in two consecutive quadrats. 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. All plant taxa were identified with Crow and Hellquist (2000) and Chaade (2002).

Comparison of submergent plant communities in Sturgeon Bay

Percent cover of submersed aquatic plants was observed at eight stations in Sturgeon Bay in late Sept 04. All observations were noted by divers in the water within an area of approximately 55 m^2 (diameter of a circle with the length of a 16 ft canoe). The sites were chosen to represent different habitats within the bay based on a previous report (K. Sherman, pers. comm.).

Application of the WMI to wetlands

The WMI was developed with 154 wetland-years (127 wetlands) for which we had both plant presence/absence data and water quality information. The wetlands that were used for validation (Cootes Paradise Marsh and Sturgeon Bay) and direct application (Fathom Five and Point Pelee) were excluded from the CCA to eliminate direct overlap between sites used for development and validation (details of these data below). Following development of the WMI, we applied the index to data that were available for 176 wetland-years (135 wetlands) for which we could calculate both WMI and WQI scores since we wanted to determine the relationship between these indices. Data for this portion of our study came from wetlands that had been sampled between 1996 and 2005 according to procedures described previously. This larger database included the 154 wetland-years used in the development of the WMI, as well as 22 that had been excluded for a variety of reasons. For instance, some had been specifically excluded because they were case studies we wanted to use to validate the WMI and others because the plants had been surveyed by inexperienced people. Application of the WMI to such a heterogeneous dataset allowed us to assess the robustness of the WMI across all environmental conditions, including differences in water-level scenarios across years, and differences due to lake-basin origin.

The historical data used to calculate the WMI values for Cootes Paradise Marsh were acquired from the Royal Botanical Gardens for 1946, 1973, 1993, and 2003 (Chow-Fraser 1998, Rothfels *et al.* 2004), while data for 1994, 1996, and 2002 were collected by Chow-Fraser (pers. comm.). No WQI values could be calculated for Cootes Paradise prior to 1993 because relevant water quality information was lacking. Sturgeon Bay was sampled for plants in 1988 by K. Sherman *(Severn Sound Environmental, Midland, ON, pers. comm.)* and eight of those sites were re-sampled in 2004 by Chow-Fraser (pers. comm.) with the same protocols as outlined above. Similar to the Cootes study, no WQI values could be calculated for the 1988 period because of the absence of appropriate data.

The WMI was calculated for various sites in PPNP and FFNMP from presence/absence data surveyed between late June and mid August 2005, and WQI scores were determined for these sites based on water quality information collected at the time of the plant surveys. We were not able to obtain detailed survey data from historic periods for either national park to compare with 2005 scores.

Multivariate statistical analyses

Canonical correspondence analysis (CCA) is a useful tool in ecological analysis because it produces synthetic axes that maximally separate the niches of the various species (ter Braak and Verdonschot 1995). CCA uses both species and environmental data with the premise that each species thrives under specific environmental conditions. CANOCO™ 4.5 (ter Braak and Smilauer 1998) was used to run both the detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA). Detrended correspondence analysis was initially used to verify that the species data had unimodal distributions across the environmental (water quality) gradient. CCA was used to ordinate the species along the environmental gradient, where the ordination is constrained by the environmental variables. Environmental variables were standardized to a mean of 0 and a standard deviation of 1. Since CCA has the tendency to over-emphasize rare species, taxa that occurred in fewer than three wetlands were excluded (ter Braak and Smilauer 1998). The significance of the axes was determined with the full model (499 permutations) and Monte Carlo permutations. Biplot scaling was used and the scaling was focused on interspecies distances. Points in the ordination plot were based on LC (linear combination) scores (biological data is described in relation to the environmental variables), which is the standard method in CANOCO 4.5.

Because different multivariate procedures may produce variable results, we also used another common multivariate technique called nonmetric multidimensional scaling (NMS) (PC ORD™ 4.0) to perform an ordination on the species dataset to see whether the two statistical approaches would yield consistent results. Unlike CCA, NMS uses only species information to align the data according to an underlying gradient. SAS JMP IN 5.1 (SAS Institute, Cary, North Carolina) was used for all other statistical analyses including: T-tests, ANCOVA, and ANOVA.

Geographic information system

ArcMap 8.2 (ESRI copyright 2002) was used to produce maps and ArcView 3.2 (ESRI copyright 1992–1999) was used to determine distances from sampling points to shore.

RESULTS AND DISCUSSION

The 127 (154 wetland-years) wetlands used for the development of the WMI were located throughout the five Great Lakes, on both the U.S. and Canadian shoreline (Fig. 1, Appendix 1). Although ecological conditions vary widely throughout the basin, there is a general trend towards more polluted wetlands being associated with the two lower lakes, Erie and Ontario, where there is greater agricultural and urban development (USEPA and GC 1995). For comparison, we calculated mean, median, minimum, and maximum values for each environmental variable on a lake-by-lake basis (Table 1). The most pristine wetlands in the remote areas of eastern Georgian Bay and the North Channel were associated with the lowest concentrations of nutrients and suspended solids (mean turbidity of 2.87 NTU and mean inorganic suspended solids of 2.11 mg/L). These values are significantly lower than corresponding values for Lakes Erie and Ontario, where average levels of nutrients were an order of magnitude higher, and suspended solids concentrations were seven to nine times higher (see Table 1). Similarly, mean CHL in Georgian Bay wetlands were ten-fold lower than those in Lakes Erie and Ontario (2.28 versus 24.82 and 16.37 μg/L, respectively). Mean conductivity (μS/cm) levels ranged from 126 in Georgian Bay to a high of 388 and 470 for Lakes Erie and Ontario, where many wetlands are urbanized and receive large volumes of highway runoff (e.g., Eyles *et al.* 2003). By comparison, pH values for all five Great Lakes were generally circumneutral, with median values of 7.61 for Lake Superior to slightly more alkaline conditions of 8.27 for Lake Michigan. Most wetlands in Georgian Bay were un-impacted and, hence, mean WQI score was relatively high (1.37, signifying very good conditions) and that for Lake Erie was relatively low (–0.35, signifying moderately degraded conditions). It is also important to note that there were no wetlands in the "excellent" category for either Lake Ontario or Lake Erie.

Development of the WMI

As a first step in the development of the WMI, we carried out a CCA with environmental and plant data from the large dataset which included 154 wetland-years (Fig. 1). Altogether, 11 environmental variables were entered into the analysis, including latitude, longitude, turbidity, conductivity, pH, ISS, OSS, TAN, TNN, TP, and CHL. Indicators of wetland degradation such as high nutrient levels and high turbidity were found to ordinate along the first synthetic CCA axis. CCA axis 1 explained 40.2% ($P = 0.002$) of the variance while axis 2 explained an additional 16.2% ($P < 0.01$; Figs. 2 and 3). The "cumulative percent variance" is a percent of the total explained variance of the species-environment relation, and should not be interpreted as the amount of variation of the community which is explained by the environmental variables. Axis 1 was highly correlated with COND ($r = 0.83$), TP ($r = 0.57$), TAN ($r = 0.52$), and latitude ($r = 0.46$), while axis 2 was highly correlated with longitude ($r = .66$) and latitude ($r = 0.43$). The cumulative eigenvalue of 3.52 indicated a good separation of species in our dataset.

Figures 2 and 3 are biplots of CCA axis 1 against CCA axis 2 for the 154 wetland-years in this study. The location of a species or a site along an axis is referred to as its "centroid," and the spatial associa-

TABLE 1. Summary of environmental variables for the 154 wetlands used for the CCA, where Turb = Turbidity, ISS = Inorganic Suspended Solids, OSS = Organic Suspended Solids, Chla = Chlorophyll, Cond = Conductivity, TP = Total Phosporus, TAN = Total Ammonia Nitrogen, TNN = Total Nitrate Nitrogen. Total plant taxa refers to all emergent, submergent, and floating plants, and # exotic taxa refers to only floating and submergent taxa. See text for explanation of WQI and the WMI scores.

Lake	Parameter	WQI score	WMI score	Turb (NTU)	ISS (mg/L)	OSS (mg/L)	Chl a (µg/L)	pH	COND (µS/cm)	TP (µg/L)	TAN (mg/L)	TNN (mg/L)	Total plant taxa	# exotic taxa
Ontario	**Mean**	**–0.30**	**1.92**	**12.75**	**19.44**	**16.05**	**16.37**		**470**	**121.20**	**0.08**	**0.31**	**14**	**1**
(n = 41)	Median	–0.29	2.00	5.22	2.40	4.60	9.20	7.80	349	97.67	0.04	0.25	12	1
	Min	–2.31	1.00	1.00	0.01	0.10	0.50	6.85	91	18.40	0.00	0.00	1	0
	Max	1.28	2.78	110.7	339.00	331.45	95.3	8.86	1658	407.00	0.55	1.03	36	4
Erie	**Mean**	**–0.35**	**2.06**	**22.61**	**15.82**	**7.58**	**24.82**		**388**	**117.10**	**0.11**	**0.54**	**13**	**1**
(n = 24)	Median	–0.01	2.19	5.04	4.94	4.37	5.39	7.84	289	57.13	0.06	0.22	12	1
	Min	–2.86	1.00	1.27	0.26	1.20	0.57	6.90	185	19.77	0.00	0.10	2	0
	Max	0.73	3.00	226.30	116.80	67.00	360.7	8.74	860	569.20	0.64	3.20	28	3
Huron	**Mean**	**0.85**	**2.81**	**5.58**	**4.11**	**2.34**	**2.21**		**155**	**37.70**	**0.02**	**0.27**	**9**	**0**
(n = 14)	Median	0.69	3.00	2.54	2.50	1.85	1.09	8.20	139	25.82	0.01	0.23	8	0
	Min	–0.41	1.58	0.69	0.53	0.21	0.12	7.48	69	3.12	0.00	0.00	3	0
	Max	2.36	3.42	32.40	18.46	6.00	9.94	8.78	239	117.10	0.07	0.63	18	2
Michigan	**Mean**	**–0.06**	**2.48**	**8.77**	**5.43**	**4.37**	**11.49**		**357**	**52.30**	**0.05**	**0.69**	**14**	**1**
(n = 5)	Median	–0.30	2.41	6.17	5.93	5.40	4.78	8.27	361	52.30	0.05	0.87	16	1
	Min	–0.55	2.32	1.66	0.01	2.07	1.90	7.50	254	45.84	0.02	0.07	8	1
	Max	0.91	2.75	21.30	13.68	5.78	43.48	8.31	408	61.41	0.09	1.37	17	2
Georgian	**Mean**	**1.37**	**3.47**	**2.87**	**2.11**	**2.59**	**2.28**		**126**	**19.90**	**0.01**	**0.21**	**23**	**0**
Bay	Median	1.43	3.52	1.92	0.50	2.00	1.50	8.06	112	18.69	0.01	0.23	25	0
(n = 49)	Min	–0.65	1.56	0.53	0.01	0.11	0.08	7.38	18	1.23	0.00	0.00	5	0
	Max	2.79	4.10	23.00	16.13	16.60	9.05	8.87	439	63.50	0.10	0.57	38	2
Superior	**Mean**	**0.53**	**2.82**	**10.75**	**5.33**	**3.82**	**4.00**		**141**	**42.30**	**0.03**	**0.29**	**16**	**0**
(n = 21)	Median	0.67	3.00	6.12	2.39	3.60	3.98	7.61	134	39.50	0.02	0.15	17	0
	Min	–0.55	1.50	2.72	0.02	0.80	0.14	7.03	56	17.90	0.00	0.00	2	0
	Max	2.13	3.38	43.00	18.80	6.50	11.36	8.52	267	76.80	0.11	0.93	31	0

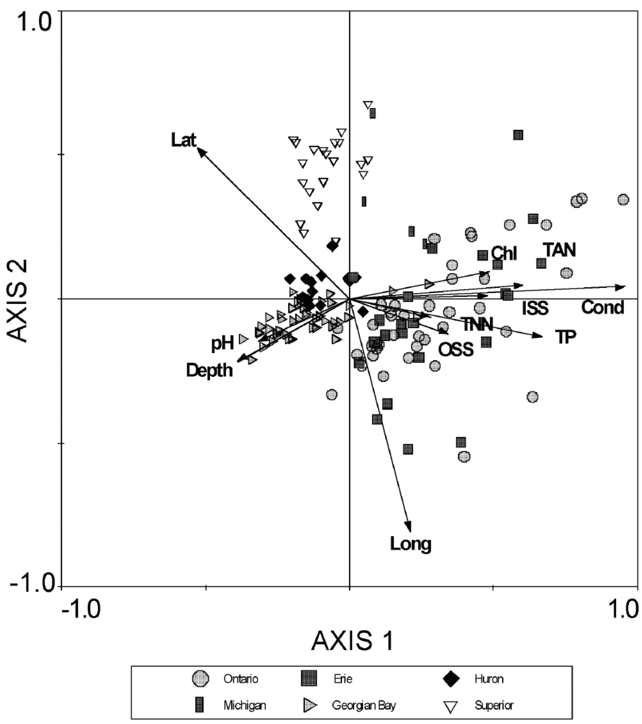

FIG. 2. Bi-plot of CCA Axis 1 versus CCA Axis 2. Vectors for the 11 environmental variables are shown (lines with arrows emanating from the origin). The strength of the correlation of the environmental variable with the axis is a direct function of the length of the vector and how close it is to the axis. Points represent the 154 wetland-years used in the CCA. Wetlands were grouped after the CCA by lake, for ease of interpretation. Wetlands on the right side of the plot have higher nutrient and turbidity levels.

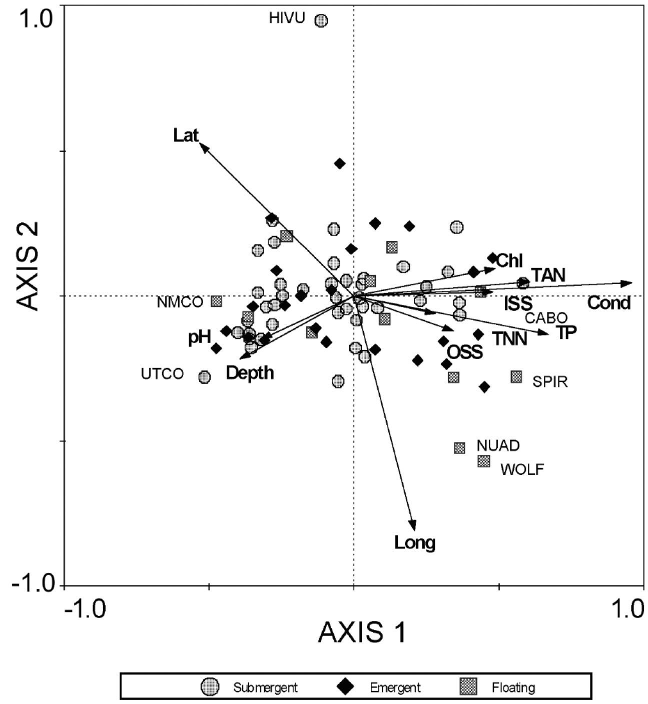

FIG. 3. Bi-plot of CCA Axis 1 versus CCA Axis 2. Vectors for the 11 environmental variables are shown (lines with arrows emanating from the origin). Some species centroids are identified with a four-letter code (see Table 2 for explanation of codes). Species were grouped by habitat. Species found on the left of the plot are intolerant of degradation.

tion of plant and site centroids provide useful information. For instance, centroids located on the right side of Figure 2 correspond to sites that tend to have high turbidity and nutrient levels and are considered to be impacted and degraded, whereas those on the left side correspond to sites that tend to have low nutrient and turbidity levels and are considered un-impacted and healthy. In the same way, corresponding species centroids located on the left side of Figure 3 are less tolerant of eutrophic and turbid water, and include species such as horned bladderwort *Utricularia cornuta* (UTCO) and floating heart *Nymphoides cordata* (NMCO). By contrast, species such as fanwort *Cabomba* sp. (CABO) and greater duckweed *Spirodela polyrhiza* (SPIR), which are found on the right side in Figure 3, are more tolerant of degradation. There is also a latitudinal gradient evident, because species found in the upper left quadrant were almost exclusively associated with Lake Superior (e.g., mare's-tail *Hippuris vulgaris* [HIVU]), while species found in the lower right quadrant were more likely to be associated with Lake Ontario (e.g., watermeal *Wolfia* sp. [WOLF] and yellow water lily *Nuphar advena* [NUAD]).

We adopted the general formula used by others (Lougheed and Chow-Fraser 2002, Seilheimer and Chow-Fraser 2006) to generate the WMI score. The two parameters, known as the optimum (U-value) and the tolerance (T-value), are related as follows:

$$WMI = \left(\frac{\sum_{i=1}^{n} YiTiUi}{\sum_{i=1}^{n} YiTi} \right) \quad (1)$$

TABLE 2. Summary of U and T values for all taxa included in this study, organized according to habitat type (emergent, floating, and submergent). Common names and species codes are also included for convenience. U value indicates the tolerance of a species to degradation (1 = very tolerant, 5 = very intolerant) and T value indicates the niche breadth (1 = broad niche, 3 = narrow niche). % occurrence indicates the percentage of wetlands (n = 176) in which the species in question occurred. *denotes that the species is not native to North America. Some species may be found in more than one group (e.g., emergent and floating) depending on the season.

Code	Taxon	Common name	U value	T value	% occurrence
Emergent					
ELAC	*Eleocharis acicularis*	Needle spikerush	4	3	9.1
ELSM	*Eleocharis smallii*	Marsh spike rush	4	2	32.9
EQFL	*Equisetum fluviatile*	Water horsetail	4	2	6.8
ERAQ	*Eriocaulon aquaticum*	Pipewort	5	3	17.6
LYSA	*Lythrum salicaria*	Purple loosestrife*	1	1	21.6
PLAM	*Polygonum amphibium*	Water smartweed	1	1	8.0
PLSP	*Polygonum* sp.	Smartweed	1	1	4.5
POCO	*Pontederia cordata*	Pickerelweed	3	2	48.3
SGCU	*Sagittaria cuneata*	Small arrowhead	3	1	9.7
SGLA	*Sagittaria latifolia*	Broad arrowhead	2	1	33.6
SGSP	*Sagittaria* sp.	Arrowhead species	2	1	6.8
SCAC	*Scirpus acutus*	Hardstem bulrush	4	2	30
SCAM	*Scirpus americanus*	Three-square bulrush	5	3	5.1
SCSP	*Scirpus* sp.	Bulrush	4	1	31.8
SCVA	*Scirpus validus*	Softstem bulrush	4	1	21.6
SPAD	*Sparganium androcladum*	Branched bureed	4	3	2.3
SPAN	*Sparganium angustifolium*	Narrow-leaf burreed	5	1	1.7
SPCL	*Sparganium chlorocarpum*	Greenfruit burreed	2	2	2.3
SPEM	*Sparganium emersum*	Unbranched burreed	1	2	2.5
SPEU	*Sparganium eurycarpum*	Giant burreed	3	2	10.8
SPSP	*Sparganium* sp.	Burreed	2	2	15.3
TYAN	*Typha angustifolia*	Narrow-leaf cattail*	1	1	21.0
TYLA	*Typha latifolia*	Broadleaf cattail	3	2	16.5
TYSP	*Typha* sp.	Cattail	1	1	23.3
TYXG	*Typha × glauca*	Hybrid cattail*	1	2	7.4
UTCO	*Utricularia cornuta*	Horned bladderwort	5	3	1.7
Floating					
BRSC	*Brasenia schreberi*	Water shield	4	1	21
EICR	*Eichhornia crassipes*	Water hyacinth*	1	1	0.6
HYMO	*Hydrocharis morsus-ranae*	Frogbit*	1	2	11.4
LEMI	*Lemna minor*	Lesser duckweed	1	1	11.4
LETR	*Lemna trisulca*	Ivy duckweed	2	2	7.4
NELU	*Nelumbo lutea*	American lotus	1	1	1.2
NUAD	*Nuphar advena*	Spatterdock	1	3	4.5
NUVA	*Nuphar variegata*	Common yellow pond lily	2	1	56.7
NYOD	*Nymphaea odorata*	Fragrant water lily (white)	2	1	66.5
NMCO	*Nymphoides cordata*	Little floating hearts	5	3	2.8
PIST	*Pistia stratiotes*	Water lettuce*	1	1	0.6
PONA	*Potamogeton natans*	Broad-leaved pondweed	2	1	30.7
SPFL	*Sparganium fluctuans*	Floating burreed	4	2	17.6
SPIR	*Spirodela polyrhiza*	Greater duckweed	1	1	5.1
TRNA	*Trapa natans*	Water chestnut*	1	1	0.6
WOLF	*Wolffia* sp.	Watermeal*	1	2	1.7

TABLE 2. Continued.

Code	Taxon	Common name	U value	T value	% occurrence
Submergent					
BIBE	*Bidens beckii*	Beck's marsh marigold	4	2	22.7
CABO	*Cabomba*	Fanwort	1	1	4.5
CASP	*Callitriche* sp.	Water starwort	4	2	10.2
CEDE	*Ceratophyllum demersum*	Coontail	1	1	45.5
CHSP	*Chara* sp.	Muskgrass	3	2	55.1
ELCA	*Elodea canadensis*	Canadian waterweed	2	1	63.6
HIVU	*Hippuris vulgaris*	Mare's tail	3	3	1.7
ISSP	*Isoetes* sp.	Quillwort	4	3	12.5
LODO	*Lobelia dortmanna*	Water lobelia	5	2	6.3
MYAL	*Myriophyllum alterniflorum*	Alternate water-milfoil	5	3	7.4
MYFA	*Myriophyllum farwellii*	Farwell's water-milfoil	3	1	0.6
MYHE	*Myriophyllum heterophyllum*	Two-leaf water-milfoil	3	2	8.0
MYSI	*Myriophyllum sibiricum*	Common water-milfoil	3	2	35.8
MYSC	*Myriophyllum spicatum*	Eurasian water-milfoil*	1	1	30.7
MYTE	*Myriophyllum tenellum*	Slender water-milfoil	4	3	8.5
MYVE	*Myriophyllum verticillatum*	Whorled water-milfoil	4	1	0.6
MYSP	*Myriophyllum* sp.	Water-milfoil	1	1	30.1
NAFL	*Najas flexilis*	Slender water nymph	3	2	51.7
NEAQ	*Neobeckia aquatica*	North american lake-cress	5	3	1.1
NISP	*Nitella* sp.	Stonewort	3	1	13.1
POAM	*Potamogeton amplifolius*	Large-leaved pondweed	4	2	25.0
POCR	*Potamogeton crispus*	Curly-leaf pondweed*	1	1	25.6
POEP	*Potamogeton epiphydrus*	Ribbon-leaf pondweed	4	3	10.8
POFO	*Potamogeton foliosus*	Leafy pondweed	2	1	0.6
POFR	*Potamogeton friesii*	Fries' pondweed	2	1	1.1
POGR	*Potamogeton gramineus*	Variable pondweed	4	2	29.5
POIL	*Potamogeton illinoensis*	Illinois pondweed	3	2	8.0
POOB	*Potamogeton obtusifolius*	Bluntleaf pondweed	2	1	0.6
PO SLEN	*Potamogeton pusillus*	"Slender" pondweed	2	1	2.3
PORI	*Potamogeton richardsonii*	Clasping-leaved pondweed	3	2	64.8
PORO	*Potamogeton robbinsii*	Fern-leaf pondweed	4	2	25.0
POSP	*Potamogeton* sp.	Pondweed	1	2	21.0
POSR	*Potamogeton spirillus*	Northern snailseed pondweed	5	2	14.2
POVA	*Potamogeton vaseyi*	Vaseyi pondweed	2	1	0.6
POZO	*Potamogeton zosteriformis*	Flat-stemmed pondweed	3	1	38.1
RALO	*Ranunculus longirostris*	Buttercup, crowfoot	2	1	16.5
RASP	*Ranunculus* sp.	Crowfoot	2	1	1.1
SGGR	*Sagittaria graminea*	Grassy arrowhead	4	3	5.7
SCSU	*Scirpus subterminalis*	Water bulrush	5	2	13.6
SPON	Fresh water sponges	Sponges	5	3	9.7
STPE	*Stuckenia pectinata*	Sago pondweed	1	1	37.5
STVA	*Stuckenia vaginata*	Sheathed pondweed	2	1	0.6
UTGE	*Utricularia geminiscapa*	Hidden fruit bladderwort	5	3	1.1
UTGI	*Utricularia gibba*	Humped bladderwort	5	2	1.1
UTIN	*Utricularia intermedia*	Flat-leaved bladderwort	3	2	5.1
UTMI	*Utricularia minor*	Lesser bladderwort	5	2	1.7
UTPU	*Utricularia purpurea*	Purple bladderwort	5	2	1.7
UTSP	*Utricularia* sp.	Bladderwort	3	2	4.0
UTVU	*Utricularia vulgaris*	Common bladderwort	3	2	30.0
VAAM	*Vallisneria americana*	Tape grass, eel grass	3	1	64.2
ZIPA	*Zizania* sp.	Wild rice	4	2	30.1
ZODU	*Zosterella dubia*	Water stargrass	2	2	5.7

Where:

Y_i = if the species is present, this value is 1; if absent, it is 0

T_i = value from 1–3 or niche breadth of species *i*

U_i = value from 1–5, tolerance of species *i* to degradation

We used the position of the centroid along the CCA axis 1 to determine the U-value for that species. The species centroids were ranked and sorted into five groups, and each group was assigned a value from 1 to 5, with equitable distribution of species in each group. Species with centroids that had high positive values (located to the extreme right of CCA axis 1) were given a U-value of 1 because they were associated with sites that had high nutrient and suspended solids concentrations, whereas those with high negative values (located to the extreme left of axis 1) were given a U-value of 5 because they were associated with sites with very low concentrations (Fig. 3) (Table 2). All centroids located between these two extremes assumed intermediate values, depending on their location along the first axis. It is useful to think of the U-value as being an index of the species' tolerance of (or sensitivity to) degraded water quality, which a value of 1 indicates most tolerant and a value of 5 indicates least tolerant.

The T-value was an indication of niche breadth for each species, and was estimated from the standard deviation of the species scores from the CCA print-out. The standard deviations of the species scores were first sorted in descending order (similar to the method used to assign U-values), and species with a broad niche (large standard deviation) were assigned a T-value of 1, whereas species with a narrow niche (small standard deviation) were assigned a value of 3.

Results of the NMS revealed that the plant species were ordinated by a strong underlying gradient which is consistent with the environmental gradient obtained by the CCA. We wanted to be sure that there was no other gradient that was governing the distribution of species (e.g., sediment composition). We found a highly significant correlation between NMS axis 2 and the CCA axis 1 scores (degradation axis) ($r^2 = 0.82$, $P < 0.01$). When NMS axis 2 scores were sorted by magnitude and grouped into five categories in a similar fashion as had been done for the CCA axis 1 scores (equivalent to U-values), we found almost complete overlap between NMS and CCA groupings. Therefore, similar results from the NMS analysis confirms that there is only one strong underlying environmental gradient.

U- and T- Values for Macrophyte Taxa

U- and T-values were determined for a total of 94 taxa—26 emergent, 16 floating, and 52 submergent (Table 2). There were 15 taxa that were only identified to genus, and U- and T-values assigned to these were determined in different ways. Plants such as muskgrass (*Chara*), stonewort (*Nitella*), and quillwort (*Isoetes*), which we could not readily identify to species, were treated as a single taxon. On the other hand, pondweed (*Potamogeton*), milfoil (*Myriophyllum*), and bladderwort (*Utricularia*) had species with a wide range of U- and T-values, and we could not simply assign them an "average" value. Instead, we gave the most conservative values associated with the genus. Hence, the coarser the identification, the lower the WMI score. We felt that this was less objectionable than omitting the entry, and as long as there is consistent treatment within a study, the resulting scores should be directly comparable. Therefore, the experience and knowledge of the person conducting the plant survey could affect the value of the WMI score, although we do not yet have empirical evidence of such a bias.

Three of the 26 emergent species could be considered indicative of excellent conditions (U-value of 5), and these included pipewort (*Eriocaulon aquaticum*), three-square bulrush (*Scirpus americanus*), and horned bladderwort (*Utricularia cornuta*), which were almost always found in the high-quality sites; except for three-square bulrush, they all tended to have a narrow niche breadth. Indicators of good conditions (U-value of 4) included branched burreed (*Sparganium androcladum*), softstem bulrush (*Scirpus validus*), hardstem bulrush (*Scirpus acutus*), water horsetail (*Equisetum fluviatile*), and two species of spike rush (*Eleocharis acicularis* and *E. smallii*). Species we found to be indicative of degraded water quality (U-value of 1) were dominant in polluted sites and included purple loosestrife (*Lythrum salicaria*), smartweed (*Polygonum amphibium*), the two non-native cattail species (*T. angustifolia*, the putative hybrid *T.×glauca*), and the unbranched burreed (*Sparganium emersum*). Several species could be considered "neutral" in that they were cosmopolitan and seemed to be tolerant of many different conditions. These included pickerelweed (*Pontederia cordata*), small arrowhead (*Sagittaria cuneata*), the giant burreed (*Sparganium eurycarpum*), and native broadleaf cattail (*Typha latifolia*). They were among the most common species of emergent plants encountered (Table 2).

Most of the floating species were able to withstand elevated levels of nutrients and turbidity. Because photosynthesis takes place above the water surface, turbidity in the water column was not a limiting factor for growth of these species. Free-floating species like the duckweeds and watermeal (*Lemna minor, Lemna trisulca,* and *Wolffia* sp.) must obtain all nutrients from the water column since they are not rooted to the sediment. Subsequently, they tended to be found in locations impacted by urban and agricultural runoff or sewage inputs. Of the 16 floating species, only two were found in high quality wetlands; the floating heart (*Nymphoides cordata*) (U-value 5) and water shield (*Brasenia schreberi*) (U-value 4). Both species require high water clarity and low turbidity, and are consequently only found in wetlands with little or no human impact. Since floating heart has a narrow niche breadth (T-value of 3), it can only grow in relatively undisturbed sites, whereas the water shield has a broader niche (T-value of 1) and can be found in a wide range of conditions. Both the common yellow pond lily (*Nuphar variegata*) and the fragrant white water lily (*Nymphaea odorata*) are widespread throughout the Great Lakes (56.7 and 66.5% occurrence). Both species were tolerant of relatively degraded conditions, and were given a U-value of 2, and were also given a T-value of 1, based on their ubiquitous distribution. The four exotic floating species, water hyacinth (*Eichornia crassipes*), frogbit (*Hydrocharis morsus-ranae*), water lettuce (*Pistia stratiotes*), and water chestnut (*Trapa natans*) were all assigned a U-value of 1 because of their ability to invade new habitat and outcompete native species.

Submersed aquatic species differ from the floating and emergent species because they spend most, or all parts of their life cycle (e.g., coontail (*Ceratophyllum demersum*) Philbrick and Les 1996) within the water column. They have different growth forms thought to be adaptations for living below the water surface, where light availability is often a limiting factor for growth (Middleboe and Markager 1997). Chambers and Kalff (1987) compared the growth of three species with different growth forms under different combinations of sediment nutrient and light conditions. They found that biomass of the slow-growing, low-lying species, fern-leaf pondweed (*Potamogeton robbinsii*) was entirely dependent on light levels at the sediment surface, and did not require high nutrient levels since the plant does not have large stems or leaves. By contrast, the biomass of the tall erect pondweed (white-stem pondweed, *Potamogeton praelongus*), which sends its leafy branches to the surface, was primarily determined by sediment fertility. It requires ample nutrients to grow the large number of leaves. The biomass of the rosette species, tape grass (*Vallisneria americana*) was dependent on both sediment irradiance and the sediment fertility since its growth form is intermediate between these two. Even though freshwater sponges are not green plants *per se*, we have included them in this major group, because their distribution is largely governed by water quality. They are sessile, low-growing forms that require good light penetration to support algal photosynthesis. Hence, their presence in a wetland is evidence of pristine water quality conditions (Lauer *et al.* 2001).

We could explain the U-values (Table 2) assigned to certain species based on their growth form. For instance, the delicate rosette-forming species such as water lobelia (*Lobelia dortmanna*) and quillwort (*Isoetes* sp.) were assigned high values of 5 and 4, respectively. They only occurred in undisturbed wetlands (Georgian Bay and Lake Superior), where there is generally good light penetration to the sediment and relatively infertile sediments (Farmer 1989, Middleboe and Markager 1997). In contrast, species that grow quickly and form canopies near the water surface, such as the invasive Eurasian milfoil (*Myriophyllum spicatum*) and sago pondweed (*Stukenia pectinatus*) were both assigned U-value of 1. These species can tolerate and thrive in wetlands with high levels of turbidity and nutrients (Chambers and Kalff 1985, Lougheed *et al.* 2001). Species such as tape grass (*Vallisneria americana*), clasping-leaved pondweed (*Potamogeton richardsonii*), and slender water nymph (*Najas flexilis*) were assigned U-value of 3, which appropriately reflected their intermediate growth forms (Hudon *et al.* 2000).

Reasons other than their growth form may be invoked to support why Canadian waterweed (*Elodea canadensis*) and coontail (*Ceratophyllum demersum*) were assigned relatively low U- values (2 and 1 respectively). Coontail lacks roots, and therefore assimilates nutrients directly from the water column and can accumulate excess nitrogen early in the season (Mjelde and Faafeng 1996). Hence, it does not tend to be found in undisturbed sites, but is instead dominant in polluted wetlands.

Canadian waterweed, on the other hand, is a species that is apparently tolerant of shade stress and a good competitor of Eurasian milfoil (*Myriophyllum spicatum*) (Abernethy *et al.* 1996), but does have a root system. This is consistent with its U- value of 2. Muskgrass (*Chara* sp.) can remain green throughout the winter, is a fast colonizer, and can tolerate relatively low light levels, persisting at depths lower than those corresponding to angiosperms in clear lakes (Blindow 1992). However, in turbid systems, charophytes are light-limited, and cannot compete effectively for light against the canopy-forming species such as Eurasian milfoil. Hence, the statistically derived U-value of 3 reflects these intermediate characteristics.

Comparison of WMI Scores

The WMI score for a wetland can theoretically range from a minimum of 1.00 to a maximum of 5.00. Only four wetlands had the lowest score of 1.00, and none of the wetlands had the maximum score of 5.00. Wetlands with the minimum score included Old Woman Creek, Tremblay Beach, Little Cataraqui Creek, and Grindstone Creek (see Appendix 1), which are all degraded wetlands in Lakes Erie and Ontario. All have turbid, nutrient-rich water and all macrophyte species present were associated with the lowest U- and T- value of 1. None of the wetlands had a maximum WMI score of 5.00 because both specialist species that require good water quality (U-value of 5 and T-value of 3), as well as generalist species that can tolerate a wide range of water quality conditions (e.g., U-value of 3 and T-value of 1) can be found in pristine wetlands. The maximum WMI score in this study was 4.10 (Tadenac Bay, see Appendix 1), which is located in a fish and wildlife sanctuary in eastern Georgian Bay, which has been managed with minimal human disturbance since the late 1900s.

Accounting for Presence of Exotic Species

In developing the WMI, our principal goal was to use the plant community to reflect water quality conditions, such as water clarity and nutrient concentrations, because these are the usual human-induced impacts on wetland environments. However, we recognize some changes in the species composition of the plant community are not reflective of altered water quality, but nevertheless reflect human-induced disturbance (e.g., recreational/boating activities). McNair (2006) has shown that impacted coastal marshes tend to be more susceptible to exotic invasions than are un-impacted wetlands, and over time, native species in human-disturbed sites can lose ground to exotic species (e.g., Wei 2007). Therefore, we accounted for the presence of exotic species by adding an additional term to the right of Equation 1 as follows:

$$WMIadj = \left(\frac{\sum_{i=1}^{n} YiTiUi}{\sum_{i=1}^{n} YiTi} \right) - \sqrt[2]{Ex} \qquad (2)$$

where "Ex" equals the proportion of floating and submergent taxa that are exotic (i.e., non-native), and we called this the adjusted the WMI (WMIadj).

Relationship Between WQI and the WMI

To aid interpretation, we calculated a range of WMI scores roughly equivalent to the six categories of water quality conditions used by Chow-Fraser (2006). These can be used qualitatively to indicate overall wetland conditions when no water quality information is available. For example, wetlands with WMI scores < 2.5 can be considered impaired (moderately to highly degraded conditions) and may require restoration and other management interventions. This list contains many wetlands in Lakes Ontario and Erie that have been targeted for restoration as part of the Great Lakes remedial action plans (RAP) (Cootes Paradise Marsh, Grand River, Humber River, and wetlands of Bay of Quinte), as well as wetlands that were part of the Severn Sound RAP (Matchedash Bay and Sturgeon Bay) (see Appendix 1). By contrast, wetlands with scores > 2.5 can be considered in "good" to "excellent" condition, and do not show negative signs of human disturbance. This list includes wetlands from all five Great Lakes, although the majority are in eastern Georgian Bay, Huron, and Superior.

To quantify the extent to which WMI scores accurately reflected water quality conditions, we regressed the WMI scores against corresponding WQI scores for 176 wetland-years from our large database which had both water quality and plant information (Fig. 4a). We found a highly significant linear relationship between the two indices ($r^2 = 0.57$, $P < 0.01$), indicating good correspondence between the presence/absence of plants and water quality conditions. As indicated in Methods, 22 wetlands had been excluded from WMI develop-

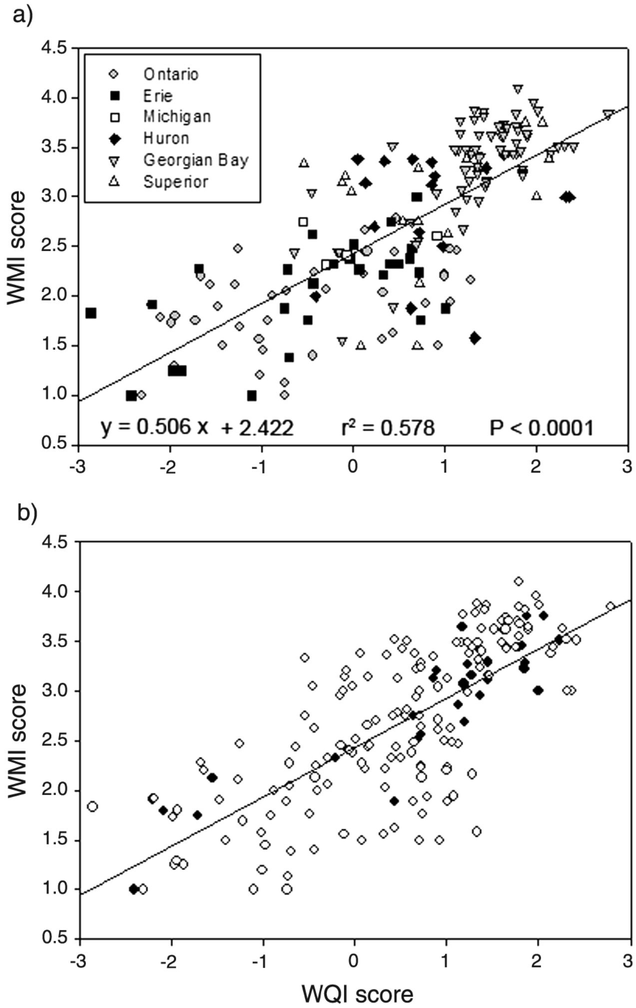

FIG. 4. Relationship between the WMI score and WQI score for 176 wetland-years a) wetlands grouped by lake, and b) open circles represent 154 wetland-years used for both the development of the WMI and for application of the WMI, closed circles represent the 22 wetland-years that were used only for the application of the WMI.

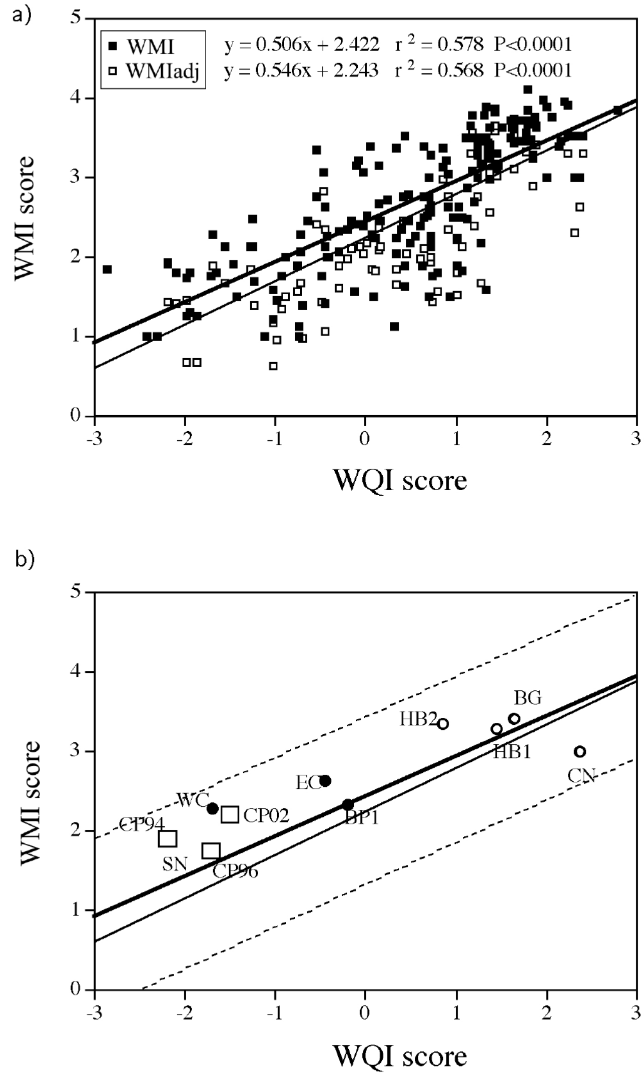

FIG. 5. a) Plot of the WMI (closed squares) and the WMIadj (open squares) vs WQI score for 176 wetland-years. b) Plot of the WMI vs WQI score corresponding to Cootes Paradise Marsh (open squares), and wetlands of Point Pelee National Park (closed circles) and Fathom Five National Marine Park (open circles). The regression lines correspond directly to those in Figure 5a. Dashed lines represent the 95% confidence intervals.

ment (Fig. 4b) for various reasons. The distribution of these wetlands is clearly within the range of data used for the WMI development, indicating that this index is robust. Data associated with the two lower lakes and their connecting channels (Erie/St. Clair, Ontario, and Niagara) tended to have WQI scores ≤ 1 and WMI scores ≤ 3, and none of the wetlands in Lakes Erie and Ontario were in the "excellent" category (WQI score ≥ 2). By contrast, over half of the wetlands of Georgian Bay, and many of those in Lake Huron were in the "very good" to "excellent" categories (WQI score ≥ 1), and these designations were matched by significantly higher WMI scores (3.33 versus 1.92 and 2.12 for Georgian Bay and Lakes Ontario and Erie, respectively; ANOVA, $P < 0.01$, $n = 176$).

We compared the relationship between the WMI (thick line in Fig. 5a) and the adjusted WMI (thin line in Fig. 5a) with WQI scores. Lack of significant interaction between type of WMI score and the WQI (P = 0.43, ANCOVA) indicated that slopes for the two regression equations were statistically homogenous; we therefore compared the two intercepts and found a numerically small but statistically significant (P < 0.01) difference (2.51 versus 2.67 for the WMI and the WMIadj, respectively). This is empirical evidence that the proportion of exotic species in the macrophyte community has a measurable effect on the WMI score, and should probably be incorporated as a metric when comparing wetlands across the Great Lakes basin.

Validation of the WMI: Cootes Paradise Marsh, Hamilton Harbour RAP

Long-term changes in the biotic community (plants, fish, and zoobenthic invertebrates) of Cootes Paradise Marsh have been well-documented (Chow-Fraser 1998, Lougheed *et al.* 2001, Lougheed *et al.* 2004, Chow-Fraser 2005). Compared with this wealth of biotic information, historic data on water quality conditions in the marsh are scarce to non-existent (Chow-Fraser *et al.* 1998, Chow-Fraser 1999a). From previous studies, we know that the marsh ecosystem of Cootes Paradise had been severely stressed by urban and agricultural development in the watershed, sustained high water levels, and bioturbation from a large population of common carp, an exotic species (Lougheed *et al.* 1998, Lougheed and Chow-Fraser 2001, Lougheed *et al.* 2004, Chow-Fraser 2005, Wei 2007). One of the most visible losses through the decades has been a decline in percentage cover of emergent vegetation from over 85% in 1934 to < 15% in the last two decades (Chow-Fraser *et al.* 1998). With the decrease in areal cover of cattails, there has been a concomitant decrease in species richness and diversity of submersed aquatic plants. One of the primary goals of the remedial actions was to restore the submersed plant community through a carp exclusion program which began in 1997 (Lougheed *et al.* 2004). The reason for choosing carp removal as a restoration strategy is that the spawning (Lougheed *et al.* 1998) and feeding (Chow-Fraser 1999a) activities of common carp can account for up to 35–40% of the turbidity in Cootes Paradise Marsh.

Surveys of the plant community conducted in 1994 revealed only four species, but in 2002, 5 years following remedial actions (carp removal), this number had increased to seven. However, these numbers are still low compared to the historic high of 15 during the 1940s (Chow-Fraser *et al.* 1998). Using the plant species list, we calculated the WMI scores to track changes before and after the RAP implementation. In Figure 6, we show a steady decline in the WMI scores through the five decades prior to RAP implementation which mirrored the decline in species richness already noted. WMI scores corresponding to the two data points following carp exclusion (2002 and 2003) were much higher, and this confirms the measurable improvement in water quality of Cootes Paradise noted by Chow-Fraser (2006) based on WQI scores. Because both water quality and plant information were available for 1994, 1996, and 2002, we calculated pairs of the WMI and WQI scores, and superimposed these (open squares in Fig. 5b) on the regression line for the 176 wetlands. This comparison indicates that the WMI can be applied to track the restoration of Cootes Paradise Marsh because all of the Cootes data fell within the 95% confidence intervals of the regression line.

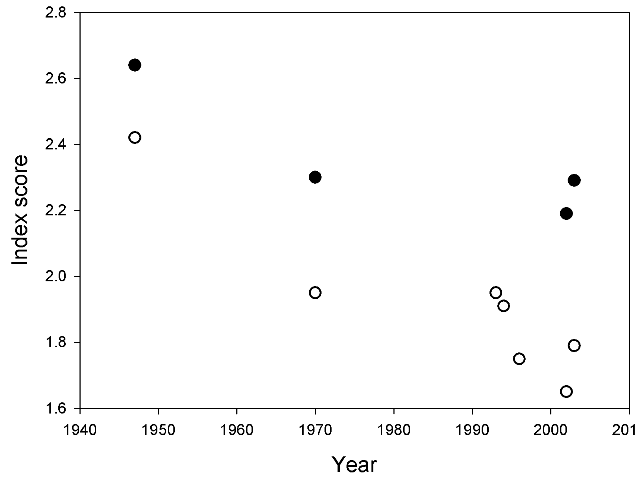

FIG. 6. Relationship between the WMI (solid symbol) and the WMIadj (open symbol) for Cootes Paradise Marsh from 1946 to 2003. The WMI and WMIadj scores during the 1990s had the same value.

Even though the 2002 and 2003 WMI scores were higher than in 1996, the increase was due to the presence of several exotic species (e.g., Eurasian milfoil [*Myriophyllum spicatum*] and the water lettuce [*Pistia stratoites*]), both of which are invasive (Cofrancesco 1998, Gordon 1998). Another non-native species, *Potamogeton crispus* was also found in 2003, but it had been observed in the 1946 and 1972 surveys. We accounted for the pres-

TABLE 3. Summary of percent composition of submersed aquatic plant species found at eight stations in Sturgeon Bay in 1988 and 2004. No significant difference between Shannon-Weiner diversity index for 1988 and 2004 (T-test, P = 0.52).

	Percent composition of species in 1988								Percent composition of species in 2004							
Taxon and variable	532	534	536	537	539	541	542	543	532	534	536	537	539	541	542	543
Ceratophyllum demersum	5	15	5	5	5	5	5	5	0	1	0	1	15	1	10	0
Chara sp.	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	1
Elodea canadensis	10	30	25	5	25	0	65	30	1	10	1	30	35	1	10	0
Freshwater Sponge	0	0	0	0	0	0	0	0	0	0	0	0	10	0	10	0
Zosterella dubia	1	1	0	1	0	1	1	1	0	1	30	10	0	0	30	1
Bidens beckii	1	0	1	0	0	0	1	1	1	0	25	0	10	0	1	0
Myriophyllum sibiricum	0	1	1	1	1	1	5	1	30	25	30	25	25	0	65	25
Myriophyllum spicatum	60	30	30	75	65	5	10	35	0	0	0	10	0	0	0	0
Najas flexilis	1	1	1	1	1	20	0	5	0	1	35	0	0	0	0	0
Nuphar variegata	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0
Potamogeton amplifolius	0	0	0	0	0	0	0	0	1	10	1	1	1	0	0	0
Potamogeton crispus	0	1	1	0	0	1	1	0	0	0	0	0	0	0	0	0
Potamogeton foliosus	1	1	0	1	1	0	0	0	0	0	0	0	0	0	0	1
Potamogeton praelongus	0	1	1	0	1	1	1	0	0	0	1	0	1	0	10	0
Potamogeton richardsonii	1	10	1	1	5	1	5	5	1	10	1	1	1	1	10	1
Potamogeton robbinsii	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0
Potomogteon zosteriformis	0	0	0	0	0	0	0	0	1	0	1	1	15	1	15	0
Ranunculus longirostris	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Vallisneria americana	15	30	20	5	0	60	0	15	55	55	0	65	0	0	35	30
Shannon-Weiner diversity index	0.52	0.83	0.65	0.41	0.48	0.59	0.50	0.84	0.40	0.65	0.72	0.64	0.82	0.08	1.08	0.39
Number of species	9	12	11	10	8	9	9	12	7	8	9	10	9	4	10	6

ence of these exotics by calculating the WMIadj score for each year and found the values had decreased to pre-RAP levels calculated for the early1990s (see Fig. 6, open circles). Therefore, although the trend in the WMI indicated an overall improvement in water quality, the trend in the WMIadj revealed the ecosystem health of the wetland continues to be poor.

Validation of the WMI: Sturgeon Bay, Severn Sound RAP

Unlike Cootes Paradise Marsh, few published data exist that can be used to track changes in the environmental quality of Sturgeon Bay before and after the RAP. Sherman (2002) reported historical total phosphorus concentrations for Sturgeon Bay, which were obtained from Environment Canada. The relationship between increased nutrients (especially phosphorus) leading to increased phytoplankton growth, resulting in increased turbidity and decreased submergent aquatic vegetation, has been well established in the literature (Hough *et al.* 1989, Crowder and Painter 1991, Golterman 1995, Lougheed *et al.* 2001, McNair and Chow-Fraser 2003). Nicholls *et al.* (1988) reported high total phosphorus levels in the vicinity of Sturgeon Bay in Severn Sound between 1973–1982, which resulted in high densities of phytoplankton that negatively impacted the submersed aquatic vegetation. When data were grouped before and after 1985 (when the sewage treatment plant was built in Victoria Harbour), we found the post-1985 mean (1986 to 2003) was significantly lower than that for the pre-1985 period (1970 to 1984) (19.50 μg/L and 16.26 μg/L respectively; t-test, $P < 0.03$), indicating overall water quality conditions have indeed improved.

Based on the reduction in TP concentrations, we expected to find corresponding improvements in WMI scores. The most comprehensive plant survey conducted in Sturgeon Bay prior to implementation of the Severn Sound RAP was carried out in 1988 by Sherman (pers. comm.). He collected plant species information at 15 sites in Sturgeon Bay in 1988; we visited eight of these sites in 2004, and collected similar information (Table 3). Two striking changes took place over the 16 years. Eurasian

TABLE 4. Location and description of human impacts on wetlands of the two Canadian national parks

National park	Wetland	Date sampled	Latitude	Longitude	Type of impact
PPNP	Sanctuary Pond (SN)	20 Jun 05	41.98032	82.54189	Sewage, agricultural run-off, common carp (*Cyprinus carpio)*
PPNP	West Cranberry Pond (WC)	21 Jun 05	41.97453	82.51620	common carp (*Cyprinus carpio)*
PPNP	East Cranberry Pond (EC)	21 Jun 05	41.97153	82.50759	common carp (*Cyprinus carpio)*
PPNP	Big Pond 1 (BP)	20 Jun 05	41.96565	82.52061	common carp (*Cyprinus carpio)*
FFNMP	Cove North (CN)	7 Jul 05	45.31340	81.76227	No human impact
FFNMP	Boat Passage (BG)	6 Jul 05	45.28953	81.71899	Boat channel
FFNMP	Hay Bay 1 (HB1)	4 Jul 05	45.24089	81.68385	Public beach, high cottage density
FFNMP	Hay Bay 2 (HB2)	7 Jul 05	45.23341	81.69424	Low cottage density

milfoil (*Myriophyllum spicatum)* had almost disappeared from Sturgeon Bay by 2004, even though it had been a dominant component in at least three stations during 1988, and was known to be a dominant species in two earlier surveys of the entire bay (1980 and 1982 according to K. Sherman (pers. comm.). It appeared to have been displaced by a conspecific (and likely competitor), the common milfoil (*M. sibiricum*), which was found abundantly in at least half of the sites during 2004, despite its rare occurrence during the 1988 survey. Several species were also found in greater abundance in 2004, including flat-stemmed pondweed (*P. zosteriformis*), large-leaved pondweed (*P. amplifolius*), white-stemmed pondweed (*P. praelongus*), and water marigold (*B. beckii*). Also noteworthy is the complete disappearance of the exotic species, curly-leaf pondweed (*P. crispus*), coupled with the establishment of the freshwater sponge at two sites.

We compared the species richness of plants between the two surveys, and found on average there were 10 ± 1.5 species observed per site in 1988, compared with only 7.87 ± 2.1 in 2004, and this difference was statistically significant (Paired T-test; $P < 0.04$) (Table 3). However, comparison of the Shannon-Weiner index scores revealed no significant difference between years (Paired T-test; $P = 0.83$) (Table 4). We then generated the WMIadj scores for each site using the 1988 and 2004 data (Fig. 7), and found a significant improvement in the WMI scores over the 16 years (Paired t-test; $P < 0.01$). Although the extent of improvement seemed to depend on the total distance separating a particular site from the sewage outflow pipe, we found no statistical evidence to support this conclusion. There was a general trend towards larger improvements for sites located closest to the sewage outflow (Table 3). We also investigated whether the extent of improvement from 1988 to 2004 could be related to distance from shore, but this also proved inconclusive (T-test; $P = 0.38$). Hence, variability associated with extent of improvement from site to site could not be attributed to either distance from sewage outflow pipe or to shoreline impacts.

Use of the WMI in Routine Monitoring Programs

Two national parks were used as case studies to demonstrate the usefulness of the WMI in routine monitoring. Within each park there were a series of wetlands, some more impacted than others with respect to past, current, and potential human-induced disturbance (see Site Description above). There were four wetlands in FFNMP: Boat Passage, Hay Bay 1, Hay Bay 2, and Cove North (see Fig. 5b). Likewise, there were four wetlands in PPNP: West Cranberry, East Cranberry, Sanctuary Pond, and Big Pond 1 (Table 4).

All wetlands in these two parks had been surveyed for plants from a canoe once during the summer of 2005 (see Table 4). To approximate the effort most likely afforded by environmental agencies, wetland surveys were carried out by two people and did not exceed half a day. Since Cove North and Boat Passage are both located on Cove Island,

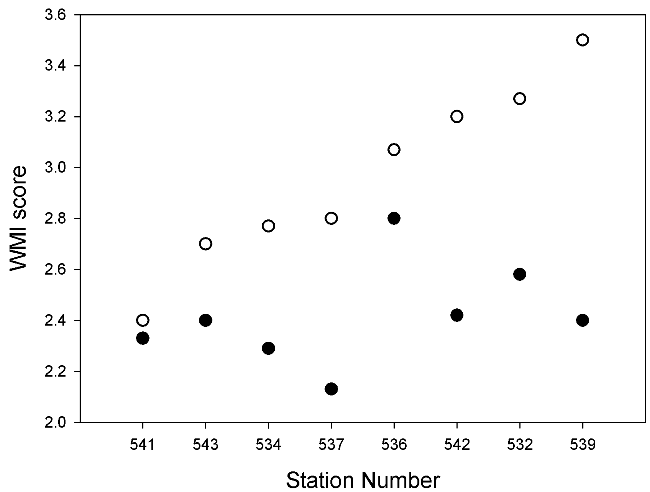

FIG. 7. Relationship between the WMIadj scores calculated for eight stations in Sturgeon Bay between 1988 (closed circles), prior to RAP delisting and 2004 (open circles), following RAP delisting. There were significant differences between the two time periods (Paired t-test; P = 0.0012).

which is relatively free of human impact other than recreational boating, these sites should be associated with high WMI scores. By comparison, Hay Bay 1 and 2 are located on the mainland and are vulnerable to sediment and nutrient enrichment resulting from cottage development and recreational activities. We therefore expected the WMI scores to be highest in Cove North and lowest in Hay Bay 1, which is known to support the highest level of human use.

Unlike the coastal wetlands of FFNMP, those of PPNP are not hydrologically connected to a Great Lake, because a barrier-beach on the east side of the park prevents complete mixing of the pond water with Lake Erie water. Known breaching events occurred in 1972, 1975, 1983, 1986, and 1989 (Chow-Fraser, pers. comm.). Hence, during years when the barrier is breached, the marsh elevation approximates that of Lake Erie, and during these breaching events, less nutrient-rich water of Lake Erie tends to dilute the pond water, while benthivorous fish such as common carp and bullheads can invade from the lake (Beak Consultants 1988). The unique hydrology of these ponds should result in better water quality in East Cranberry Pond and Big Pond (which are more vulnerable to these breaching events), while Sanctuary Pond is expected to have the most degraded conditions because it has been hydrologically disconnected from the rest of the ponds, as well as from Lake Erie, for several decades (Chow-Fraser, pers. comm.).

We found a general increase in WMI scores with improvement in corresponding WQI scores, and when these were superimposed on the regression line for the 176 wetland-years, all data points were bracketed by the 95% confidence intervals of the regression line (Fig. 5b). The lowest WMI score was associated with Sanctuary Pond, but the highest score was associated with Boat Passage rather than Cove North. We attribute the lower WMI in Cove North to disturbance resulting from its geomorphology and its exposure to high wind and wave action, which are factors that negatively affect the establishment of submersed aquatic vegetation.

GENERAL DISCUSSION

Two multivariate analyses (CCA and NMS) were used in this study to derive an index that utilizes presence/absence of wetland plants to indicate the water quality conditions of 154 coastal wetlands (Figs. 1 and 2). The similarity of results from both analyses provides confidence in the assignment of U and T values for the 94 taxa in Table 2. Using data from 176 wetlands throughout the five Great Lakes, we established a highly significant relationship between the WMI and WQI scores (Fig. 4), and this supports our assumption that plants are indeed good indicators of water quality conditions in wetlands. The dependence of submergent plant colonization on nutrient and turbidity levels has also been documented by others (Chambers and Kalff 1987, Hough *et al.* 1989, Barko *et al.* 1991, Golterman 1995, Tracy *et al.* 2003). Within the Great Lakes basin, we are satisfied that the WMI can be used to rank the degree of human-induced disturbance among a wide range of coastal wetlands. However, we do acknowledge the disproportionate representation of wetlands within the Canadian portion of the shoreline (due to logistical constraints), and we recommend further testing in U.S. wetlands.

The WMI was validated with historical data from Cootes Paradise Marsh and Sturgeon Bay. For Cootes Paradise Marsh, there had been sufficient water quality information to directly compare conditions before and after RAP implementations. Chow-Fraser (2005) found a significant improvement in all water-clarity variables (extinction coefficient, Secchi depth transparency, water turbidity, and the concentration of total inorganic suspended solids) following carp exclusion at the two open-water sites (LT1 and LT5; Table 1 in Chow-Fraser, 2005). For instance, at LT1, water turbidity dropped 40%, from 72.2 (mean of data for 1993–1996 inclusive) to 43.6 NTU (mean of data for 1997–2001 inclusive); a similar magnitude in reduction was noted for LT5. Chow-Fraser (2006) reported a corresponding increase in WQI scores of >30%, from –2.20 in 1994, to –2.09 in 1998, to –1.50 in 2000. We emphasize that the increased WMI values after 2000 were partly attributed to the presence of two invasive exotic species (Eurasian milfoil and water lettuce), and therefore, the overall health of the marsh is still compromised.

Lundholm and Simser (1999) indicated that the lack of a seed bank is not a contributing factor to the return of submergent species because the majority of species historically found in Cootes Paradise Marsh are perennials which reproduce vegetatively. But it is unknown how long rhizomes and turions can persist in the sediment while they wait for favourable conditions. Prolonged periods of unfavourable conditions may prevent species from rebounding when conditions improve. Only one species found in Cootes Paradise in 1997 was annual (horned pondweed (*Zanichellia palustris*), but this pioneer species did not become abundant, probably because other more invasive and aggressive species such as *Myriophyllum spicatum*, *Pistia stratoites,* and *Potamogeton crispus* benefited disproportionately from improved conditions. *Hysteresis* is the inability of an ecosystem to rebound to its previous state once the external forcing function (e.g., phosphorus enrichment) has been removed, and this has been well documented for shallow lakes in Europe (e.g., Janse *et al.* 1998, Van Nes *et al.* 2002, Zhang *et al.* 2003). This is likely the reason for the retarded improvement in the plant community of Cootes Paradise following decrease in water turbidity resulting from carp exclusion. Zhang *et al.* (2003) suggested a shift back to clear water, macrophyte-dominated systems will be prevented in hypereutrophic shallow systems because of high release of phosphorus accumulated in the sediment, which is consistent with findings of Kelton and Chow-Fraser (2005) for Cootes Paradise Marsh.

One advantage of the WMI over the WQI is that there is historic plant information to calculate the former, but insufficient water quality information in historic databases to calculate the latter. This was certainly true for Sturgeon Bay, for which there were many gaps in the historic water quality database. It was difficult to find anything other than TP concentrations, and given the high interannual variation, we included data from 30 years to demonstrate a significant change before and after the operation of the sewage treatment plant in Victoria Harbour. By comparison, calculation of the WMI only required data from 2 years, 1988 prior to the initiation of the Severn Sound RAP, and 2004, a year following the delisting of the RAP. Even though the 1988 plant survey had not been conducted with the WMI in mind, these data were used to generate a set of the WMI scores with relative ease (Table 3). We were able to replicate the 1988 sampling protocol during the 2004 survey to generate a corresponding set of the WMI scores (Fig. 7). Mean scores associated with the 2004 survey were significantly higher than those associated with the 1988 survey (2.96 versus 2.42, respectively), thus independently confirming the results of the TP comparisons. Had macrophyte data been available from the early 1980s, we would probably have seen a greater improvement in the WMI values since the 1988 survey took place 3 years following the upgrade of the wastewater treatment facility in Victoria Harbour.

Another advantage to using macrophytes is that the plant community integrates effects of many factors that act concurrently on the assemblage over a long period of time (Wei 2006). Thus, routine monitoring programs such as those required by the PPNP and FFNMP could use the WMI as a relatively cost-effective way to screen ecosystems for evidence of human disturbance, and follow up with a more targeted and intensive sampling for water quality conditions. We have demonstrated that the relationship between the WMI and WQI is robust (Fig. 5b), and over a wide variety of conditions.

Wilcox *et al.* (2002) found their IBI was influenced by low water levels experienced during the single year they collected the data, and suggested that such an IBI should only be applied to data collected under similar water-level conditions. Since the WMI was developed with data collected over 9 years (1996–2005) from all five Great Lakes, any biases due to water-level effects should have been accounted for. We allowed the attributes of individual species and groups of species to indicate wetland quality, rather than relying on total species diversity, or total number of native taxa encountered during a particular visit. Accordingly, we saw a better correlation between the WMI scores and degree of human disturbance as measured by WQI scores.

Wilcox *et al.* (2002) also suggested that separate indices be made for each lake and each geomorphic type. We feel that such a lake-by-lake approach puts too great an emphasis on the influence of micro-climatic and geomorphic factors, and would lead to truncated gradients of human disturbance in the less populated regions of Lakes Huron and Superior. We demonstrated that the response of the common plant species to levels of suspended solids and nutrients is similar across all five Great Lakes. The higher WMI scores associated with eastern Georgian Bay, and lower scores with Lakes Erie and Ontario are primarily reflective of the degree of human impact, not regional differences in geology or climate. When wetlands of Georgian Bay were subjected to disturbance from agricultural and recreational activities (e.g., Lily Pond in Honey Harbour, or Matchedash Bay of Severn Sound), they acquired plant species indicative of human-induced disturbance encountered in wetlands of the two lower lakes.

Another major difference between the WMI and previous indices (e.g., IBI of Simon *et al.* 2001, Wilcox *et al.* 2002, Minc and Albert 2004) is that our index focuses on species related to fish habitat (submergent, floating, and emergent) or taxa found in the open-water areas of wetlands, whereas the others focused on the emergent and wet-meadow communities. For example, Simon *et al.* surveyed the wetland from the floodplain (wet meadow) to the littoral zone, and included areas that could not have been accessed by fish, and consequently, there were relatively small number of submergent and floating taxa (14 and 4, respectively) compared with ours (50 and 16, respectively; see Table 2) We feel that the WMI is a more appropriate indicator of fish habitat in wetlands, whereas the other indices may be better indicators of bird habitat. If this is true, then a holistic view of wetland health would require the use of both types of indices, or development of an integrated index that includes equal treatment of all the vegetation zones.

As with other indices that rely on accurate identification of plants to the species level, the expertise of the person conducting the plant survey may have influence on the final WMI score. We are now conducting a study to empirically determine the extent to which the level of expertise of the technician will affect wetland scores, and this should provide guidance for agencies that require the use of volunteers in their monitoring programs. The advantage of using plants for a volunteer monitoring program compared to other volunteer monitoring programs (Marsh Monitoring Program, Environment Canada) is that it is relatively easy to learn to identify the plant species included in the WMI, and voucher samples can be preserved for a brief period of time until they can be correctly identified by an expert.

We found both the WMI and the WMIadj useful for monitoring wetlands, and this is in agreement with the IBI of Wilcox *et al.* (2002). We recommend the use of the WMI to track effects of selected pollutants in wetlands since it is a sensitive indicator of water quality conditions; however, the adjusted WMI should be used if there is an additional objective of determining the ecological health of the wetland, and to account for the impact of invasive exotic species. For example, in parks such as FFNMP, where current water quality conditions are still excellent, increased human disturbance through increased boat traffic is more likely to threaten the biodiversity of native species rather than water quality. Parks Canada would benefit from tracking the negative impact of exotic invaders on ecosystem integrity through monitoring changes in WMIadj scores. Managers need to decide which index (WMI or WMIadj) is more appropriate for their location and application.

The WMI was developed specifically for coastal systems that have a hydrological linkage to a large lake or bay. The lower than expected WMI score associated with Cove North in FFNMP (Fig. 5b) indicates that this index may be modified to account for exposure disturbance due to wave and wind action. We may also be able to identify species that are more indicative of exposure or tolerant of different water-level scenarios. Application of the WMI to systems no longer hydrologically connected to the Great Lakes may also lead to lower than expected scores. This should be the focus of future study.

One main objective of this study was to develop an index that can be used by environmental agencies with limited resources and personnel, but who must choose indicators that are easy to implement, cost-effective, and sensitive to annual changes in wetland conditions. We believe WMI is an index that can be used to track the impact of human-induced disturbances and its effect on fish habitat in coastal wetlands. Once plant sampling for the WMI is completed (usually within a few hours for most wetlands), U and T values (Table 2) can be applied to the data to calculate a WMI score and can then be related back to the degree of water quality impairment of fish habitat (Table 3). An additional application is that historic species lists can be used to generate WMI scores to track long-term changes in wetlands. There are a limited number of high-quality wetlands along the Great Lakes shoreline, and many of these exist in eastern and northern Georgian Bay (Chow-Fraser 2006). These wetlands provide vital spawning and nursery habitat for fish. We believe WMI can be one of the useful biotic assessment tools available for use by both volunteers and government personnel to monitor the health of coastal wetlands.

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APPENDIX 1.0. List of 154 wetland-years and their locations (latitude and longitude) used for the development of the WMI.

Wetland	Year	Lake	WMI score	WMIadj score	Latitude	Longitude
Big Creek (Teeterville)	1996	Erie	2.13	2.13	45.95550	80.44719
East Cranberry	2005	Erie	2.62	2.34	41.97153	82.50759
Grand River	1998	Erie	1.25	0.67	42.88390	79.57220
Grand River	2001	Erie	1.25	0.67	42.90000	79.60000
Holiday Conservation	1996	Erie	1.83	1.83	42.03335	83.05000
Long Point Big Rice	2001	Erie	2.38	2.03	42.58930	80.33550
Long Point Inner Bay	2001	Erie	2.38	1.97	42.59650	80.34180
Long Point Inner Channel	2001	Erie	3.00	2.59	42.59130	80.33550
Long Point Little Rice	2000	Erie	2.22	1.83	42.58930	80.33550
Long Point Prov Park	1998	Erie	2.24	1.93	42.58333	80.38333
Presque Isle	2000	Erie	2.52	2.17	42.15900	80.09850
Redhead Pond	2005	Erie	2.27	1.84	41.95378	82.50657
Rondeau Bay	1998	Erie	2.33	2.33	42.30070	81.85530
Rondeau Bay	2001	Erie	2.75	2.45	42.28800	81.86700
Sanctuary Pond	2005	Erie	1.92	1.42	41.98032	82.54189
Selkirk Prov Park	1998	Erie	1.38	0.97	42.81667	79.95000
Turkey Creek	1996	Erie	1.88	1.56	42.23556	83.08528
Turkey Point	1998	Erie	2.17	2.17	42.66860	80.35320
Turkey Point	2002	Erie	2.48	2.09	42.63359	80.34170
West Cranberry	2005	Erie	2.28	1.87	41.97453	82.51620
Cormican Bay	2003	Georgian Bay	3.82	3.82	45.40765	80.31288
Cow Island	2005	Georgian Bay	3.78	2.78	46.09859	81.81942
David’s Bay	2004	Georgian Bay	3.48	3.48	45.04750	80.00380
Dead Horse	2005	Georgian Bay	3.23	3.23	46.10463	81.60802
Dogfish Bay	2005	Georgian Bay	3.28	3.05	46.08091	81.73593
French River Main	2005	Georgian Bay	3.73	3.52	45.96796	80.88779
Ganyon Bay	2005	Georgian Bay	3.86	3.64	44.92052	79.81763
Garden Channel	2003	Georgian Bay	3.61	3.61	45.18628	80.12147
Gooseneck	2004	Georgian Bay	3.15	3.15	45.20688	80.10749
Green Island	2003	Georgian Bay	3.04	2.76	44.78833	79.74403
Green Island	2004	Georgian Bay	3.40	3.16	44.78862	79.74900
Herman’s Bay	2004	Georgian Bay	3.62	3.62	45.08638	79.99758
Herman’s Bay	2005	Georgian Bay	3.71	3.50	45.21824	79.86969
Hockey Stick Bay	2005	Georgian Bay	3.81	3.58	44.94461	79.86297
Hole in the Wall	2005	Georgian Bay	3.63	3.63	45.52182	80.43831
Ingersoll Bay	2005	Georgian Bay	3.84	3.84	45.28132	80.25588
Jumbo Bay	2005	Georgian Bay	3.71	3.71	46.05244	81.81858
Key River	2003	Georgian Bay	3.22	2.99	45.88742	80.67858
Lily Pond	2005	Georgian Bay	3.05	2.82	44.87037	79.81478
Longuissa Bay	2003	Georgian Bay	3.51	3.30	44.96723	79.89152
Matchedash Bay	1998	Georgian Bay	1.56	1.56	44.73333	79.66667
Matchedash Bay	2002	Georgian Bay	2.44	2.44	44.73353	79.66683
Matchedash Bay	2003	Georgian Bay	2.45	2.10	44.75520	79.69648
Moon River Bay	2003	Georgian Bay	3.63	3.36	45.12053	79.97500
Moon River Falls	2003	Georgian Bay	3.52	3.52	45.10733	79.92995
Moose Bay	2003	Georgian Bay	3.31	3.31	45.07210	80.04958
Moreau Bay	2003	Georgian Bay	3.70	3.70	45.01092	79.94572
Musky Bay	2003	Georgian Bay	3.48	3.30	44.81040	79.78265
Musky Bay	2004	Georgian Bay	3.48	3.30	44.81040	79.78265
Ni Bay	2005	Georgian Bay	3.44	3.44	45.40924	80.45599
North Bay	2005	Georgian Bay	3.52	3.52	44.89638	79.79377

(Continued)

APPENDIX 1.0. Continued.

Wetland	Year	Lake	WMI score	WMIadj score	Latitude	Longitude
Oak Bay	2003	Georgian Bay	2.86	2.86	44.79630	79.73158
Ojibway Bay	2005	Georgian Bay	3.67	3.44	44.88758	79.85585
Otter Creek	2005	Georgian Bay	3.77	3.56	45.95403	80.82421
Port Rawson	2003	Georgian Bay	3.44	3.44	45.19512	80.02350
Quarry Island	2003	Georgian Bay	3.48	3.48	44.83400	79.80968
Quarry Island	2004	Georgian Bay	3.48	3.48	44.83217	79.80550
Sandy Island	2003	Georgian Bay	3.87	3.87	45.26865	80.25065
Sandy Island West	2005	Georgian Bay	3.64	3.40	45.27659	80.26755
Sturgeon Central	2003	Georgian Bay	3.42	3.23	45.61782	80.43260
Tadenac Bay	2004	Georgian Bay	3.88	3.88	45.13742	79.99287
Tadenac Bay 1	2005	Georgian Bay	4.10	4.10	45.03444	79.99145
Tadenac Bay 2	2005	Georgian Bay	3.96	3.86	45.03916	79.98792
Tadenac Lake	2005	Georgian Bay	3.84	3.84	45.03437	79.95509
Treasure Bay	2005	Georgian Bay	3.55	3.32	44.86854	79.86049
Waldon’s Pond	2005	Georgian Bay	3.62	3.62	45.92294	80.87577
Wardrope Island	2005	Georgian Bay	3.44	3.46	46.05486	81.71651
West Bay	2003	Georgian Bay	3.50	3.50	45.42228	80.30727
Baie du Dore	1998	Huron	1.58	1.58	44.33670	81.55570
Boat Passage	2005	Huron	3.42	3.10	45.28953	81.71899
Collingwood Harbour	1998	Huron	2.00	2.00	44.50920	80.23260
Cove Island North	2005	Huron	3.00	2.62	45.31340	81.76227
Echo Bay	1998	Huron	1.88	1.88	46.49453	84.07597
Echo Bay	2000	Huron	3.38	3.38	46.49453	84.07597
Echo Bay	2002	Huron	3.38	3.38	46.49460	84.05500
Hay Bay 2	2005	Huron	3.35	2.97	45.23341	81.69424
Mismer	2000	Huron	3.14	3.14	46.00510	84.46060
Oliphant Bay	1998	Huron	2.64	2.64	44.73131	81.28203
Russell Island West	2005	Huron	3.00	2.29	45.26458	81.70412
Spanish River	1998	Huron	3.36	3.36	46.18339	82.35000
Spanish River	2000	Huron	2.70	2.70	46.17845	82.34585
Spanish River	2002	Huron	2.50	2.17	46.18339	82.31691
Lake St. Clair	1999	Lake St. Clair	1.76	1.43	44.58333	82.76667
Lake St. Clair	2000	Lake St. Clair	1.76	1.43	44.58333	82.76667
Tremblay Beach	1998	Lake St. Clair	1.00	1.00	42.30000	82.65000
Pentwater Marsh	2000	Michigan	2.32	1.87	43.76280	86.40780
Pentwater Marsh	2001	Michigan	2.32	1.87	43.76280	86.40780
Peshtigo	2001	Michigan	2.61	2.33	44.98400	87.66070
Portage Creek	2001	Michigan	2.75	2.40	45.70620	87.08000
Wigwam Bay	2001	Michigan	2.41	2.13	43.97020	83.85430
Buckhorn	2001	Niagara	2.27	2.27	43.05630	78.97120
Spicer Creek	2001	Niagara	1.88	1.52	43.02338	78.89677
Bayfield Marsh	2000	Ontario	1.75	1.34	44.19758	76.36500
Blessington Bay	2002	Ontario	2.44	2.07	44.16700	77.33300
Bronte Creek	2002	Ontario	1.45	0.95	43.39340	79.71546
Credit River	2002	Ontario	1.90	1.90	43.55007	79.08358
Darlington	2001	Ontario	1.20	0.62	43.87300	78.79700
Fifteen Mile Creek	2002	Ontario	1.73	1.44	43.16693	79.31668
Frenchman’s Bay	1998	Ontario	2.00	1.50	43.82240	79.09490
Frenchman’s Bay	2001	Ontario	2.06	1.59	43.81233	79.09467
Goose Bay	2002	Ontario	2.22	1.82	44.35005	75.86671
Grass Bay	2002	Ontario	2.46	2.46	44.15018	76.26681
Grindstone Creek	2002	Ontario	1.00	1.00	43.28333	79.88333
Hay Bay Marsh	1996	Ontario	2.23	2.23	44.16675	76.93335

APPENDIX 1.0. Continued.

Wetland	Year	Lake	WMI score	WMIadj score	Latitude	Longitude
Hay Bay Marsh	2000	Ontario	2.45	2.11	44.16675	76.93335
Hay Bay Marsh	2002	Ontario	2.44	2.04	44.16675	76.93335
Humber River	1996	Ontario	1.80	1.80	43.64280	79.48860
Humber River	2002	Ontario	1.50	1.50	43.61673	79.48333
Johnstown Creek	1998	Ontario	1.69	1.38	44.73300	76.46700
Jordan Harbour	1996	Ontario	1.80	1.80	43.17930	79.37340
Jordan Harbour	2002	Ontario	1.29	1.29	43.15014	79.38333
Little Cataraqui Creek	1998	Ontario	1.00	1.00	44.28110	76.51630
Little Cataraqui Creek	2002	Ontario	2.11	1.84	44.21667	76.55000
Little Sodus	2001	Ontario	2.03	1.65	43.33942	76.69447
Madoma Creek	1998	Ontario	1.50	1.50	44.26667	76.38333
Madoma Creek	2002	Ontario	2.23	1.99	44.26667	76.38333
Mud Bay	2002	Ontario	2.05	1.66	44.06682	76.31672
Muskellunge River	2002	Ontario	2.24	1.99	43.96682	76.05010
Napanee River	1998	Ontario	1.40	1.05	44.23333	76.98333
Perch River	2002	Ontario	2.66	2.35	43.98361	76.06688
Presqu'ile Prov Pk	1998	Ontario	1.81	1.81	44.00000	77.73060
Presqu'ile Prov Pk	2002	Ontario	2.78	2.44	44.00000	77.73060
Salmon River	2002	Ontario	2.16	1.66	48.56667	76.20004
Sandy Creek	2002	Ontario	2.48	2.11	43.70089	76.19647
Sawguin Creek	1996	Ontario	1.62	1.62	44.10000	77.38333
Second Marsh	1995	Ontario	2.47	2.11	43.87500	78.81320
Weller's Bay	1998	Ontario	1.80	1.56	44.01679	77.61670
Wellers Bay	2002	Ontario	2.20	1.79	44.01679	77.61670
West Lake	1998	Ontario	1.11	1.11	43.93333	72.28333
Pt. Mouillee	1998	St. Lawrence	1.13	1.13	45.16667	74.36667
Upper Canada Bird Sanctuary	1998	St. Lawrence	2.40	2.40	44.98300	75.00000
Willowbank Marsh	1998	St. Lawrence	1.57	1.16	44.31667	76.21667
Au Train	2002	Superior	2.94	2.94	46.43334	86.81681
Bark Bay	2000	Superior	3.13	3.13	46.85042	91.19819
Chippewa Creek	1998	Superior	1.50	1.50	48.33870	89.21570
Chippewa Park	2002	Superior	1.50	1.50	48.31700	89.20000
Cloud Bay	2001	Superior	3.38	3.38	48.08120	89.44370
Cloud Bay	2002	Superior	3.38	3.38	48.08280	89.43720
Flag	2002	Superior	3.14	3.14	46.78667	91.38778
Goulais River Oxbow	1998	Superior	2.25	2.25	46.71667	84.41667
Hurkett Cove	1998	Superior	2.13	2.13	48.83300	88.50000
Hurkett Cove	2002	Superior	3.21	3.21	48.83080	88.49470
Laughing Whitefish	2002	Superior	3.23	3.01	46.51675	87.01688
Lost Creek	2001	Superior	3.28	3.28	46.85861	91.13583
Nemadji River	2002	Superior	2.96	2.96	46.68353	92.03340
Pike River	2002	Superior	3.00	3.12	47.01676	88.51679
Pine Bay	1998	Superior	3.05	3.05	48.03360	89.52320
Pine Bay	2001	Superior	3.33	3.33	48.03330	89.51950
Sioux River	2000	Superior	2.81	2.81	46.73430	90.87790
Sturgeon Bay Slough	2002	Superior	3.00	3.00	47.00024	88.48348
Sturgeon Bay Superior	1998	Superior	2.63	2.63	48.19020	89.31160
Taquamenon River	2002	Superior	2.71	2.71	46.55010	85.01691
West Fish Creek	2001	Superior	2.75	2.75	46.58420	90.94610