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5 Development of the Water Quality Index (WQI) to Assess Effects of Basin-wide Land-use Alteration on Coastal Marshes of the Laurentian Great Lakes

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CONTENTS

5.1 INTRODUCTION

Development and use of biological indicators to monitor the status and trends of aquatic ecosystems such as streams and rivers (Karr 1981, 1991; Wichert 1995; Wichert and Rapport 1998), lakes (Hughes and Noss 1992) and inland freshwater wetlands (Keddy et al. 1993; Adamus et al. 2001; Weigel 2003) have become routine for many environmental agencies throughout the world (e.g. van Dam et al. 1998; U.S. EPA 2002). The ecological basis for using biological indicators is that the community of plants and animals will reflect the overall condition or quality of the habitat. Suter (2001) has criticized that the goal of this form of environmental monitoring, which summarizes the overall effects of both natural and human-induced disturbances, is problematic because it does not address the causal relationship between the disturbance and the indicator, or in terms of environmental risk assessment, the measures of effect and the assessment endpoint, respectively.

In many cases, the presumed human-induced disturbance is the associated increase in nutrient and sediment loading from conversion of forests in the watershed into agricultural and urban land (e.g. Field et al. 1996; Müller et al. 1998; Dodson and Lillies 2001; Wang et al. 2001). In other cases, however, site-level disturbances such as proximity to roads and highways (Nelson and Booth 2002; Eyles et al. 2003; Ourso

TABLE 5.1.

Summary of relationships between stressors and indicators for coastal wetlands presented in chronological order according to numbers that accompany arrows in Fig. 1. Italics indicate that the relationship is not supported by scientific study.

Number	Proposed relationship	Supporting literature
1	Significant inverse relationship between late summer percent cover of emergent vegetation and mean annual water level of Great Lake	Williams and Lyon 1997; Chow-Fraser et al. 1998
	Species-specific response to deep and shallow water will give rise to predictable changes in community composition and structure	Wilcox and Meeker 1992; Thiet 2002
1a	Recent low water levels encourage establishment of exotic aquatic plants	Hudon 1997; Hudon et al. 2000
2	Seiche effects will affect nutrient dynamics of exposed wetlands	Sager et al. 1985;
	Seasonally disconnected systems through dykes or natural beach barriers can significantly alter water quality within wetlands	McLaughlin and Harris 1990; Keough et al. 1999
3	Significant positive correlation between biomass of benthic algae and water-quality degradation	McCormick et al. 2001; McNair and Chow-Fraser 2003
4	Culturally enriched sediment will affect the species composition and richness of submergent vegetation	Smith et al. 2002; Tracy et al. 2003.
5	Culturally enriched sediment will affect the species composition and richness of emergent vegetation	Chow-Fraser et al. 1998; Miao et al. 2000.
6	Water clarity (depth) determines the extent of submergent vegetation colonization	Hudon et al. 2000
	Water quality determines the species richness of submergent macrophytes	Crosbie and Chow-Fraser 1999; Lougheed et al. 2001
7	Benthic algae negatively affects the species richness of submergent macrophytes	McNair and Chow-Fraser 2003
8	Significant positive association between submergent aquatic vegetation and wetland-dependent fish	Randall et al. 1996; Chow-Fraser et al. 1998; Wei et al. 2003
9	Water quality affects the diversity and species richness of fish	Brazner 1997; Seilheimer and Chow-Fraser, unpub. data
10	Loss of emergent vegetation will lead to eventual loss of submergent vegetation	Engel and Nichols 1994; Chow-Fraser et al. 1998

11	Community of submergent vegetation will affect zoobenthos (zooplankton and benthic invertebrates) species richness	Chow-Fraser 1998; Burton et al. 1999; Lougheed and Chow-Fraser 2002
12	Community of emergent vegetation will affect distribution of zoobenthos (zooplankton and benthic invertebrates)	McLaughlin and Harris 1990; Chow-Fraser et al. 1998; Euliss et al. 1999
13	Establishment of exotic species (e.g. common carp, zebra mussels, Eurasian milfoil, purple loosestrife, and common reed) can negatively affect the community dynamics of native plants and animals	Brady et al. 1995; Chow-Fraser et al. 1998; Boylen et al. 1999; Mills et al. 1999; Blossey et al. 2001; Tewksbury et al. 2002; Bartsch et al. 2003; Hall et al. 2003; Nalepa et al. 2003; Wilcox et al. 2003
14	There is a significant positive correlation between biomass of carp and water turbidity	Lougheed et al. 1998; Chow-Fraser 1999
14a	Removal of carp benefits species richness of submergent vegetation	Lougheed et al. 2003; Angeler et al. 2003
15	Significant interactions exist between wetland fish and benthic invertebrates	Kohler et al. 1999; Batzer et al. 2000
16	Boating activities will contribute to degraded water quality	---
16a	Boating activities will negatively affect the integrity of the fish and benthic community	Backhurst and Cole 2000; Arlinghaus et al. 2002; Penczak et al. 2002
17	Point source discharge will contribute to water-quality impairment	Chow-Fraser 1998 and many others
18	Shoreline modification will reduce available fish habitat	Randall and Minns 2002; Garland et al. 2002
19	Roadway/highway runoff contribute to water-quality impairment in urban wetlands; amount of paved surfaces was best correlate (negative) with fish diversity and IBI	Chow-Fraser et al. 1996; Wang et al. 2001
20	Riparian condition (presence of buffer strip or location of golf courses/cottages/residential property) can alter water quality independently of basin-wide land use	Lammert and Allan 1999; Meador and Goldstein 2003; Houlahan and Findlay 2003
21	Basin-wide land-use affects water quality in streams and wetlands	Harding et al. 1998; Crosbie and Chow-Fraser 1999; Lougheed et al. 2001

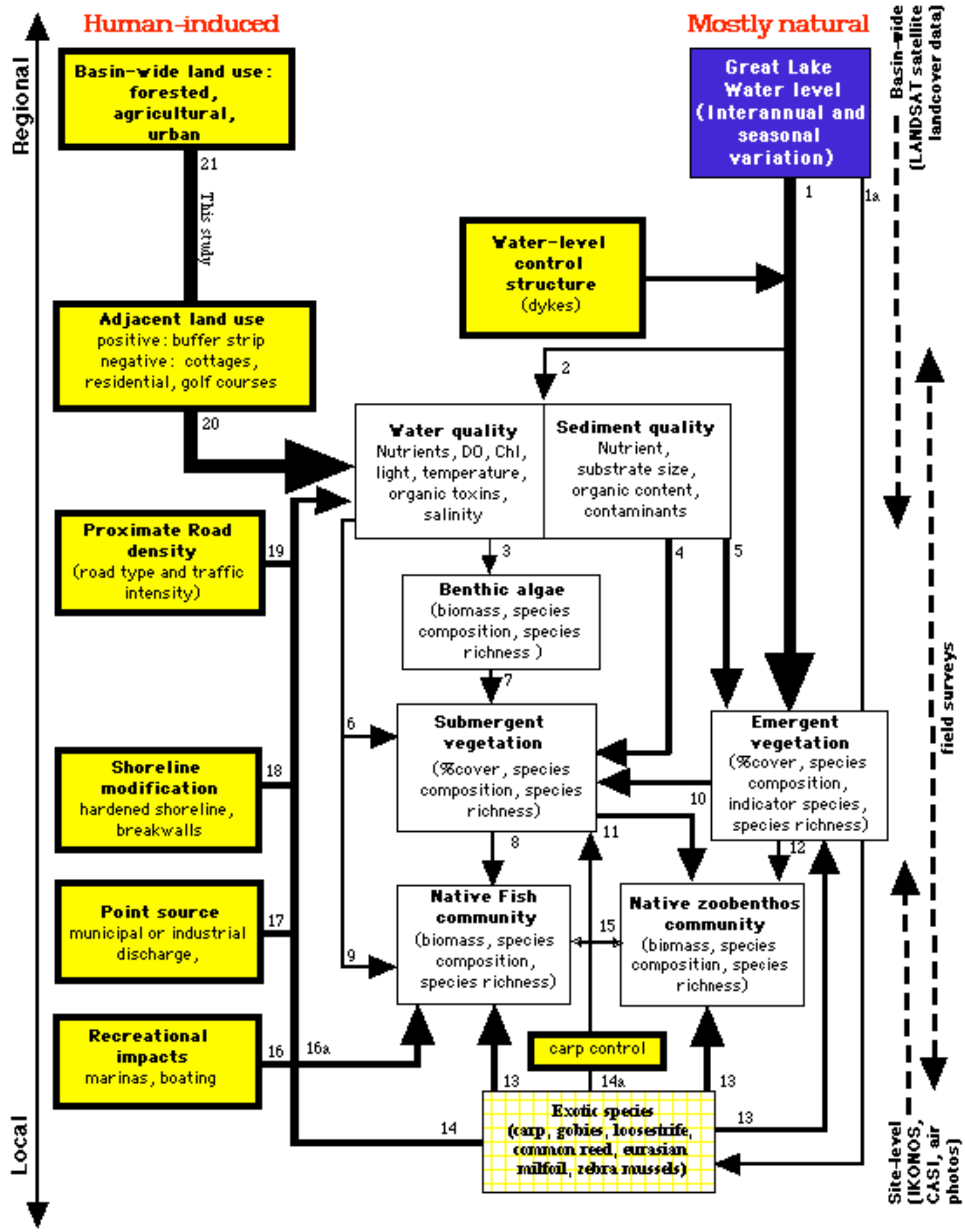

Figure 5.1 Relationship diagram linking stressors (shaded boxes with thickened edges) to indicators (clear boxes). See Table 1 for explanation of relationships and supporting literature.

and Frenzel 2003) or distance to forest cover (Houlahan and Findlay 2003), riparian condition and shoreline development (e.g. Lammert and Allan 1999; Meador and Goldstein 2003), as well as recreational impact (Kashian and Burton 2000; Lewis et al. 2002; Penczak et al. 2002) can have overriding effects on the biotic community in the absence of changes in basin-wide land uses.

For freshwater wetlands, natural disturbances such as inter-annual changes in water levels (Keddy and Reznicek 1986; Wilcox et al. 2002), site-to-site variation in ambient temperature (Anderson and Vondracek 1999; Tangen et al. 2003), and in-stream hydrologic variability (Poff and Ward 1989; Poff and Allan 1995) may also be more influential on the composition of the flora and fauna than proportion of developed land in the catchment. Finally, remedial actions such as carp exclusion (Lougheed et al. 2003), treatment of sewage effluent prior to being discharged into wetlands (Chow-Fraser et al. 1998) and water-level management through dyking (McLaughlin and Harris 1990; Thiet 2002) can induce changes in the plant community that are completely unrelated to natural disturbances or land-use changes. While all these stressor categories can potentially produce changes in the biotic community of impacted relative to least-impacted (or "reference") sites, few if any of these biotic indices can distinguish among disturbance types (Suter 2001; Wilcox et al. 2002). In addition, a biotic index for two reference sites can vary because of natural variability (e.g. hydrologic regime), or an impacted site may have a biotic score that is similar to that of a reference site because of good riparian condition resulting from implementation of best-management practices (e.g. Wang et al. 2003). Indeed, without knowing the relationship between an environmental condition (stressor) and the biological response, it is difficult to select appropriate reference areas for a particular study.

Many coastal marshes of the Laurentian Great Lakes, especially those occurring in Lakes Erie and Ontario show obvious signs of degradation because of poor water quality (Smith et al. 1991; Chow-Fraser and Albert 1999; Thoma 1999). Their great ecological, hydrological, educational and recreational value has prompted governments at all levels to make them a priority both for monitoring and impact assessment. To this end, various agency-wide research programs have been implemented in both Canada and the U.S. to develop appropriate environmental indicators that can be applied widely throughout the Great Lakes basin. Because stressors responsible for ecosystem impairment occur at spatial scales that span local to regional extents, a number of indicators (assessment endpoints) can be measured for coastal wetlands, ranging from information measured at the site level to those sensed remotely (airborne or satellite) (Figure 5.1). These stressors can be considered naturally-occurring (represented by box with black background) or human-induced (represented by boxes with grey background), and can vary in terms of how they relate to a number of abiotic and biotic indicators (boxes with clear background).

The scope of this paper only permits me to discuss a subset of all relationships (arrows) in Figure 5.1, but the specific details of all proposed relationships together with supporting literature (numbers accompanying arrows) are summarized in Table 5.1. Water-quality impairment for many coastal wetlands of the lower Great Lakes has been attributed to nutrient and sediment inputs from agricultural and urban landscapes (#21; refer to Figure 5.1 and Table 5.1), although for some marshes, point-source pollution from municipal or industrial waste-treatment facilities (#17) and carp bioturbation (#13) have played an equally important role. Regardless of the pollution source, however, the resulting eutrophic and turbid conditions generally lead to a higher biomass of benthic algae (#3), which can reduce the species richness of submergent plants (#6, #7), and which can in turn affect the species richness, species composition and size structure of higher trophic levels (i.e. zooplankton, benthic invertebrates and fish (#8, #11 and #15). A direct link should exist between basin-wide land use (e.g. percentage forested, agricultural and urban land) and water-quality conditions in coastal wetlands (Figure 5.1; Table 5.1), although this assumption has not been tested rigorously at a lake-wide scale of all five Great Lakes. In this paper, I will use water-quality data collected from 110 widely distributed wetland complexes (146 wetland-years; Figure 5.2) to develop a "Water Quality Index" (WQI) that can be used to directly test this assumption. Specifically, I will investigate if WQI scores can be statistically related to proportion of forested and altered land in wetland catchments. I will show how WQI scores can be used to assess the quality of wetlands in cross-sectional (many wetlands across the basin at one time) as well as longitudinal studies (how Cootes Paradise has changed over an 8-year period (1994-2001) in response to a carp-exclusion program). For a subset of the wetlands, I will also show how the WQI compares with published IBI ranks derived from benthic macroinvertebrate data for wetlands in Lake Huron (Burton et al. 1999) and from fish, plant and macroinvertebrate data for wetlands in Lakes Superior, Michigan and Huron (Wilcox et al. 2002). Finally, I will demonstrate the relationship between water quality (WQI scores) and higher trophic levels, including biomass of benthic algae (McNair and Chow-Fraser 2003), and species richness of submergent plants (Lougheed et al.

2001). By directly linking biotic indicators to WQI and percentage land use, I will show that the WQI is a reliable indicator of human-induced land use alterations, and that it should be used as an independent means of assessing wetland quality when developing biological indicators.

5.2 METHODS

5.2.1 Study Sites

The database used to develop the WQI includes water-quality information from 110 wetlands located throughout all five Great Lakes (Figure 5.2; Appendix 1). Almost all data included here come from samples collected between 1998 and 2002 (early June to end of August), except those for Cootes Paradise which represent years before (1994) and after implementation of a carp-exclusion program (1998, 2000, and 2001; see Lougheed and Chow-Fraser 2002; Lougheed et al. 2004). In total, there were 53 sites from the lower lakes and connecting channels (including the St. Lawrence River (upstream of Cornwall, Ontario, Canada), Lake Ontario, Niagara River, Lake Erie and Lake St. Clair), and 57 from the upper lakes (including Georgian Bay, Lakes Huron, Michigan and Superior). Wetlands were not randomly selected, but were chosen to represent most of the ecologically important eco-reaches identified in Chow-Fraser and Albert (1999) and to ensure that the database included a very large range of land-use and water-quality conditions.

5.2.2 Field Methods

Water samples used for analysis of planktonic algae, primary nutrients and suspended solids were collected in a standardized manner from an open-water site located at least 10 m from the edge of the aquatic vegetation; in certain wetlands, submergent vegetation was present throughout and in those cases, deeper areas that

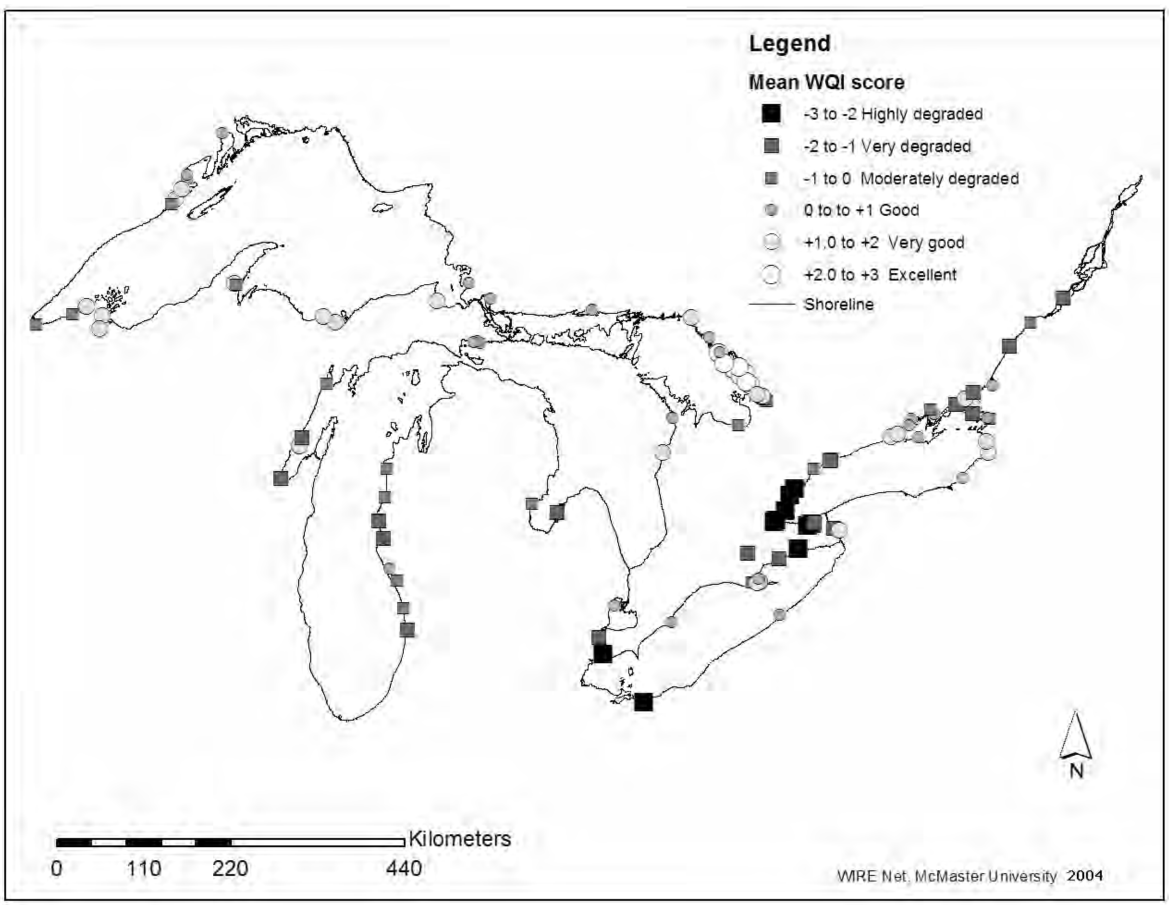

Figure 5.2 Location of study sites in this study.

had minimal submergent vegetation were sampled. This ensured that samples were not contaminated with benthic algae (either epiphytic or periphytic). All water samples were collected with a clean 1-L van Dorn bottle deployed at mid-depth in the open-water areas. Because water depths in the various wetlands ranged from 20 cm to 5.5 m (mean of 1.1m), it was not always necessary or possible to use the Van Dorn sampler; in those instances, water was collected by simply plunging a clean 1-L beaker upside down into the water and quickly inverting it to collect water, taking care that the sediment and plants were not disturbed in the process. Aliquots of this water were immediately measured in triplicate for water turbidity (TURB) with a Hach Portalab turbidimeter. Samples were also stored in 1-L brown (for planktonic chlorophyll-a) or clear acid-washed (for nutrients and suspended solids) polyethylene bottles, and kept in a cooler until they were processed that evening either in the field or at a laboratory.

5.2.3 Physical Parameters

Temperature (TEMP), conductivity (COND), pH, and dissolved oxygen (DO) were measured with several *in situ* probes during the study period. Prior to 2000, we used a Hydrolab H20 equipped with a Scout monitor (Hydrolab, Austin, Texas, USA); during 2000 and 2001, we used a Hydrolab Minisonde multi-parameter probe and Surveyor monitor (Hydrolab, Austin, Texas, USA); and in 2002, we used a YSI 6600 multi-parameter probe with YSI 650 display (YSI, Yellow Springs, Ohio, USA). During 2001, we conducted a side-by-side comparison of all three instruments, and found no significant differences with respect to any of the above parameters. Regardless of the instrument used, all sensors were calibrated as indicated by the manufacturers at the beginning of multi-day field trips (up to 8 days). The remoteness of many of our sites from the University laboratory (where calibrations were carried out) precluded daily calibrations during these multi-day sampling trips, even though this would have been desirable. The time of day at which these physical measurements were taken varied from site to site; the earliest measurements were taken close to 09:00 and the latest were taken close to 20:00. Differences in sampling times did not generally affect the parameters of interest, except for DO, which could vary from <4 mg/L in early morning to >10 mg/L in mid-day in very eutrophic sites (Chow-Fraser, unpub. data). Coordinates reported for sites sampled prior to 2000 were obtained from published sources (Crosbie and Chow-Fraser 1999 and Lougheed and Chow-Fraser 2002), whereas after 2000, all sites were georeferenced with either a Trimble GPS unit (4- to 5-m accuracy) attached to the Hydrolab Surveyor or a Garmin GPS unit (4- to 6-m accuracy), which was attached to the YSI 650 display.

5.2.4 Field Processing

Sample processing usually took place within six hours of collection. Water for nutrient analyses were dispensed into clean, acid-washed Nalgene bottles that had been first rinsed with deionized water. They were then kept frozen until analysis (usually within three months of collection). Samples for chlorophyll-a content of phytoplankton (CHL) were first filtered through 0.45-μm GF/C filters, and then stored frozen in tin foil. Parallel samples for total suspended solids (TSS) were similarly filtered through pre-weighed filters, then placed in clean small petri plates, sealed and put in a freezer.

5.2.5 Sample Analyses

At the time of analysis, frozen filters designated for CHL analyses were unwrapped from foil, placed in 10 mL of 90% reagent-grade acetone, and kept in the freezer from 4 to 24h (APHA 1992). Samples were then centrifuged, and chlorophyll-a content was determined by measuring absorbance with a Milton Roy 301 spectrophotometer before and after acidification (to account for phaeophytin pigments). Chlorophyll samples reported in this study were all measured in triplicate, and final concentrations were calculated as described in Chow-Fraser et al. (1994). Following digestion in potassium persulfate in an autoclave, samples for total phosphorus (TP) were measured in triplicate according to the molybdenum blue method of Murphy and Riley (1962). Samples for soluble reactive phosphorus (SRP) were first passed through 0.45 μm-filters before molybdenum blue analysis, without digestion. Total Kjeldahl nitrogen (TKN), total nitrate nitrogen (TNN) and total-ammonia nitrogen (TAN) (measured on the day of water collection) were processed and analyzed with Hach protocols and reagents (Hach Company 1989) with a Hach DR2000 spectrophotometer (Hach, Loveland, Colorado, U.S.A.). Total nitrogen (TN) was calculated by addition of TKN and TNN.

Filters designated for TSS analyses were taken out of the freezer and first dried at 100°C for 1 h, then dried in a dessicator with calcium sulphate for another hour, before they were weighed to determine TSS. Loss on ignition was determined after combustion at 550°C for 20 min, followed by drying in the dessicator for an hour. Weight of the combusted filter was assumed to be total inorganic suspended solids (TISS), whereas difference in the weight of the filter before and after combustion was total organic suspended solids (TOSS).

5.2.6 Land Use Delineation

I was able to obtain basin-wide land-use information for a subset of the 110 wetlands in two ways. First, I obtained information from published sources (Crosbie and Chow-Fraser 1999; Kashian and Burton 2000; Wilcox et al. 2002). Secondly, I delineated watersheds from topographic maps (1:50,000 and 1:24,000, respectively for Canadian and U.S. wetlands, respectively), and overlaid corresponding land-use information as described in Crosbie and Chow-Fraser (1999). I may have introduced an unknown and probably inconsistent error across the dataset because land-use maps for most of the Canadian wetlands date back to the mid-to-late 1980s, whereas the sampling was carried out in the late 1990s and early 2000s. I do not know the extent to which this bias applies to the U.S. wetlands since published land-use information had not been fully described in these papers. This error was unavoidable because updated land-cover data that can be applied to the entire Great Lakes basin do not currently exist. Since published sources did not uniformly report proportions of all land uses, I was only able to obtain land-use information classified into the three categories (i.e. forested, agricultural or urban) for 45 wetlands. However, when I was willing to examine forested versus altered (lumping agricultural and urban categories together), I was able to calculate proportion of altered land (PROPALT) for 74, and proportion of forested land (PROPFOR) for 81 sites, and this greatly increased the statistical power of my analyses.

5.2.7 Statistical Procedures

I used SAS JMP for the Macintosh (version 4.04; SAS Institute Inc., Cary, North Carolina) to conduct all the univariate analyses (paired

TABLE 5.2.

Summary of water- and sediment-quality variables originally considered in the principal components analysis, and those that were finally included in the development of the WaterQuality Index (WQI).

Variable	Included in final WQI model
Depth (cm)	No
Turbidity (TURB; NTU)	Yes
Temperature (TEMP, °C)	Yes
pH	Yes
Conductivity (COND; µS/cm)	Yes
Dissolved oxygen (DO; mg/L)	No
Chlorophyll-*a* (CHL; µg/L)	Yes
Total suspended solids (TSS; mg/L)	Yes
Total inorganic suspended solids (TISS; mg/L)	Yes
Total organic suspended solids (TOSS; mg/L)	No
Total phosphorus (TP; µg/L)	Yes
Total dissolved phosphorus (TDP; µg/L)	No
Soluble reactive phosphorus (SRP; µg/L)	Yes
Total kjeldahl nitrogen (TKN; mg/L)	No
Total ammonium nitrogen (TAN; µg/L)	Yes
Total nitrate nitrogen (TNN; mg/L)	Yes
Total nitrogen (TN; mg/L)	Yes
TN:TP ratio	No
Sediment total phosphorus (TP_{sed}; µg/g)	No
% Inorganic sediment ($TISS_{sed}$)	No
% Organic sediment ($TOSS_{sed}$)	No

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t-test, ANOVA and Tukey-Kramer multiple comparisons and linear regression analyses). I also used it to conduct the Principal Components Analysis (PCA), an ordination technique which is an extension of fitting straight lines and planes (or axis) through many variables by least-squares regression (Jongman et al.1995), which became the basis of the Water Quality Index. Twelve of 21 variables (see Table 5.2) from 146 wetland-years were included in the development of the index. All data were $\log_{10}$-transformed to standardize the data to a mean of zero and a standard deviation of one to eliminate scale biases. Since twelve variables were entered into the PCA, twelve possible axes were fitted. Each was fitted to the data sequentially to account for as much variability as possible at successive steps, and each axis was orthogonal to the preceding axis and independent of all others. The WQI score was calculated as the weighted sum of the site score from all twelve axes. I used a stepwise multiple linear regression procedure to build predictive models so that WQI scores can be generated from various combinations of water-quality variables. I also calculated a Pearson correlation coefficient to determine if there is a statistically significant relationship between WQI scores and PROPFOR and PROPALT. The proportions were first arcsin-transformed, and Dutilleul's (1993) correction had to be applied to the data to correct for spatial autocorrelation of both variables (Fortin and Payette 2002; Legendre et al. 2002; Wei et al. 2003). Spatial autocorrelation may be measured by Moran's I (1950) or Geary's C (1954). A correlogram is a graph of autocorrelation values plotted against distance classes and is analyzed mostly by looking at its shape.

5.3 RESULTS

From an original list of 21 variables explored in preliminary analyses, 12 were finally included for development of the Water Quality Index (WQI; (Table 5.2). The final list of variables was chosen based on ease of measurement (and presumed availability in routine monitoring programs), and their potential for pointing

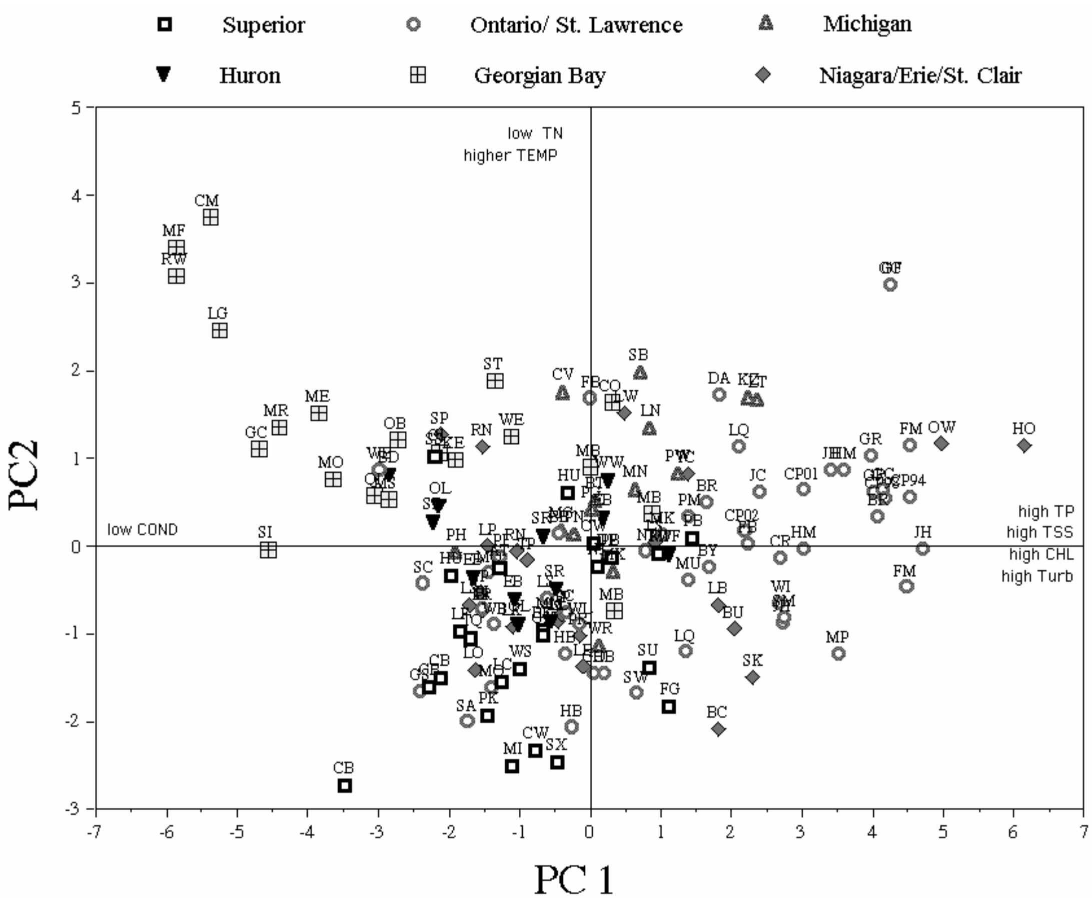

Figure 5.3 Plot of PC2 vs PC1 for 146 wetland years in this study.

TABLE 5.3.
Summary of eigenvalues produced by PCA using standardized values of 12 water-quality variables for 146 wetland-years.

PC axis	Eigenvalue	Percent explained	Cumulative Percent explained
1	5.6039	46.699	46.699
2	1.5204	12.670	59.369
3	1.1218	9.348	68.718
4	0.9541	7.951	76.669
5	0.7172	5.977	82.645
6	0.6255	5.212	87.858
7	0.4412	3.677	91.535
8	0.3340	2.783	94.317
9	0.2726	2.272	96.589
10	0.2191	1.826	98.415
11	0.1264	1.053	99.468
12	0.0638	0.532	100.000

TABLE 5.4.
Summary of correlation coefficients between principal components (PC) scores and environmental variables and loadings for each parameter in respective PC axes.

	Variance explained (%)	Environmental Variable	Loading	r-value	P-value
PC1	46.69	TP	0.37056	0.877	0.0001
		TURB	0.36175	0.856	0.0001
		TSS	0.34646	0.820	0.0001
		TISS	0.33031	0.782	0.0001
		CHL	0.32217	0.763	0.0001
		COND	0.31942	0.756	0.0001
PC2	12.67	pH	0.44920	0.554	0.0001
		TEMP	0.44180	0.545	0.0001
		TN	-0.37226	-0.459	0.0001
PC3	9.35	TEMP	0.50468	0.535	0.0001
		pH	0.39073	0.414	0.0001
		COND	0.36816	0.390	0.0001
Cumulative	68.71	----		----	----

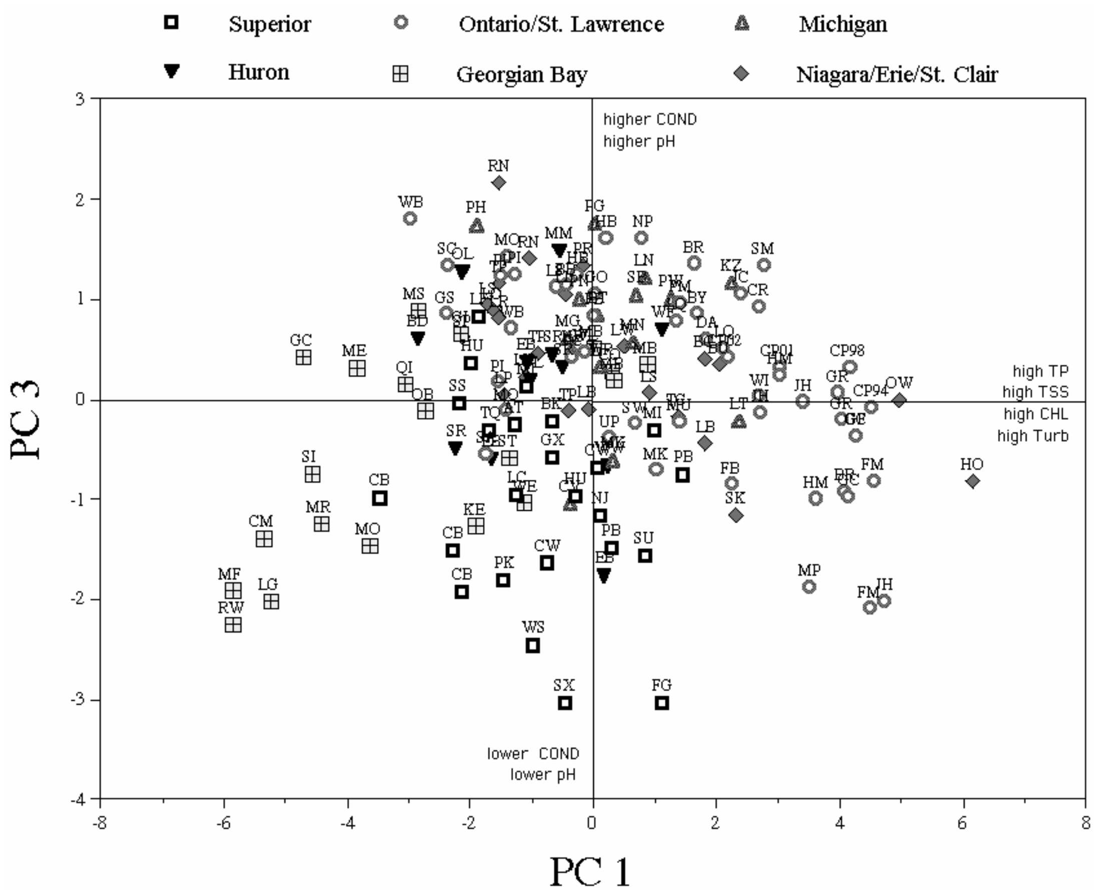

Figure 5.4 Plot of PC3 vs PC1 for 146 wetland years in this study.

out unique information when included. Dissolved oxygen (DO) was initially included, but was subsequently dropped because this was the only variable that was extremely sensitive to the time of day at which it was measured (see Methods) and since it was not possible to standardize the time of sampling for all sites, a single reading could not be used to represent the full range of DO conditions encountered at that site. A more meaningful indicator of oxygen conditions may be a daily mean calculated from continous hourly measurements (Chow-Fraser, unpub. data).

The twelve variables from 146 wetland-years were entered into the Principal Components Analysis. Although the PCA fits as many axes as there are variables, the first four axes explained 76% of all the variation in the dataset (Table 5.3). The first axis, which explained 47% of the total variation, ordinated wetlands according to the degree of water-quality impairment, since it was highly correlated with, TP, TURB, TSS, ISS, CHL and COND (Table 5.4). A plot of the PC1 scores against respective PC2 scores for all wetland-years is a good way to show the ordination results (Figure 5.3). The most degraded wetlands, those that were highly turbid, nutrient-rich, and had high water conductivity (e.g. Old Woman Creek (OW) and Holiday Marsh (HO)) were located at the far right on Axis 1, while the least-impacted wetlands, those that had clear water, low nutrient and low water conductivity (e.g. Cloud Bay (CB) in Lake Superior and Port Rawson (RW), Longuissa Bay (LG) and Sandy Island (SI) in Georgian Bay) were located at the opposite end (Figure 5.3). The second axis, which accounted for an additional 13% of the variation, was significantly correlated with temperature, pH and nitrogen concentrations (Table 5.4; Figure 5.3), which reflected in part the large geographic distribution of wetlands throughout the five Great Lakes and their associated differences in bedrock geology and latitude. The negative correlation between TN and PC2 is primarily driven by the nutrient-poor sites in Georgian Bay (Moon River Falls (MF) and Cormican Bay (CM)). The third axis was significantly correlated with COND and pH (Table 5.3; Figure 5.4); the Georgian Bay sites with very soft water (Port Rawson (RW), Longuissa

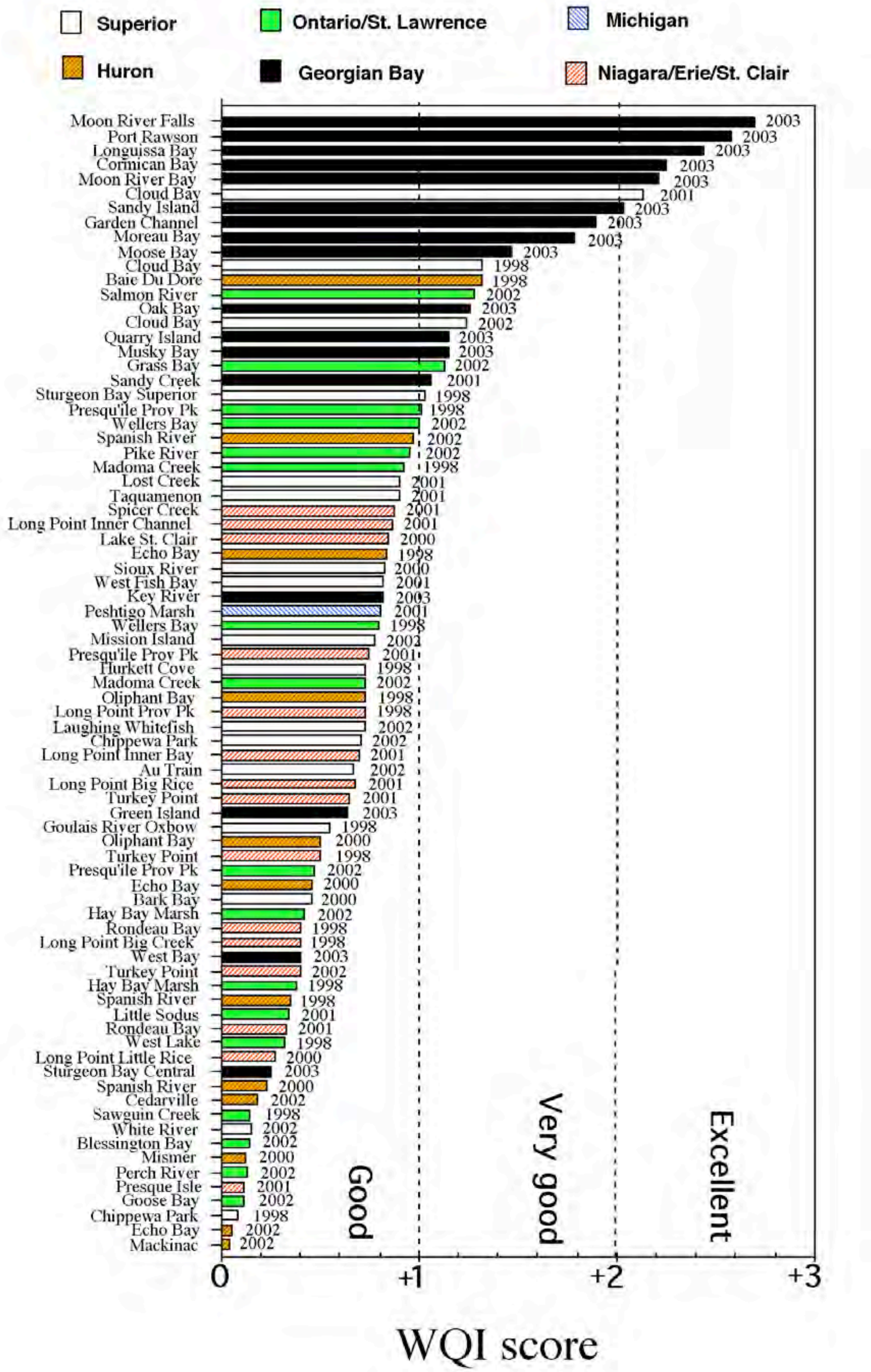

Figure 5.5 Wetlands with positive WQI scores are presented in descending order of rank. Years associated with the WQI scores appear to the right of the bars.

Bay (LG), Cormican Bay (CM) were clustered away from the more alkaline sites that occur in Lake Ontario and Erie (Figure 5.4).

Both PC1 and PC2 were strongly correlated with nutrients and suspended solids (Figure 5.3), and together explained almost 60% of the total variation. These axes effectively ordinated wetlands according to degree of water-quality impairment, regardless of lake of origin. For instance, sites designated by the IJC within Areas of Concern (AOC) (International Joint Commission 2003) in Lake Ontario (Jordan Harbour (JH), Niagara River RAP; Cootes Paradise (CP) and Grindstone Creek (GC; GF), Hamilton Harbour RAP; and Humber River (HM), Toronto and Region RAP) were positioned far to right of the less impacted wetlands in eastern Lake Ontario (Salmon River (SA); Sandy Creek (SC); Weller's Bay (WB)). Similarly, the degraded wetlands that occur in western Lake Erie (Holiday Marsh (HO) and Old Woman Creek (OW)) occur far to the right of those high-quality wetlands of Long Point Marsh complex (LO, LP, TP). Even though most of the Georgian Bay wetlands were very good quality, AOC sites (Collingwood (CO) and Matchedash Bay (MB)) were positioned to the extreme right of the group. Although there was not as great a range of water-quality impairment for the Lake Superior wetlands, there were similar trends with degraded sites (Mission Island (MI) and Pine Bay (PB)) occurring to the right of pristine sites such as Cloud Bay (CB).

5.3.1 Derivation of the Water Quality Index

In the initial stages of developing the Water Quality Index, I only included the first four axes, since they together explained 76% of the total variability. A reviewer of an earlier draft of this paper argued that I needed to include the first seven, which together explained 90% of the total variation. In the end, I opted to include all twelve axes, rather than risk losing any amount of useful information. Therefore, the Water Quality Index (WQI) score for any wetland was the weighted sum of all PC site scores (i.e. all twelve axes). That is, I multiplied the wetland score associated with a particular PC axis with the proportion of variation explained by the corresponding eigenvalue (i.e. PC1 * 0.46699; PC2 * 0.1267., etc; see Table 5.3), and summing the products for all twelve PC axes for each of the 146 wetland-years.

The highest score (interpreted as being the most pristine) calculated in this dataset was less than +3 (see Figure 5.5), while the lowest score was greater than –3 (interpreted as being the most degraded) (see Figure 5.6). For ease of interpretation, I arbitrarily divided the scale into six categories as follows:

WQI Score	Category
+3 to +2	Excellent
+2 to +1	Very good
+1 to 0	Good
0 to -1	Moderately degraded
-1 to -2	Very degraded
-2 to -3	Highly degraded

There is no theoretical reason for choosing six categories. It should be considered a starting point, to be revised when a better scheme emerges. Wetlands originating from all five Great Lakes were represented in the "Good" to "Excellent" categories (all positive scores; Figure 5.5), as well as the "Moderately degraded" to "Highly degraded" categories (all negative scores; Figure 5.6). Although there were disproportionately more Georgian Bay wetlands in the good categories (solid bars), the index was able to identify the AOCs (Collingwood and Matchedash Bay (Severn Sound AOC)) as being "Moderately degraded". By comparison, almost all of the "Very degraded" and "Highly degraded" sites were from Lakes Erie and Ontario, while two wetlands were from Lake Michigan (blue bars), Long Tail Point in Green Bay and Kalamazoo River, which are both associated with AOCs (Figure 5.6).

5.3.2 Using the WQI

The index was effective in tracking the improved health of Cootes Paradise Marsh, over the course of a marsh-wide carp exclusion program as part of the Hamilton Harbour Remedial Action Plan (Lougheed et al. 2004). In

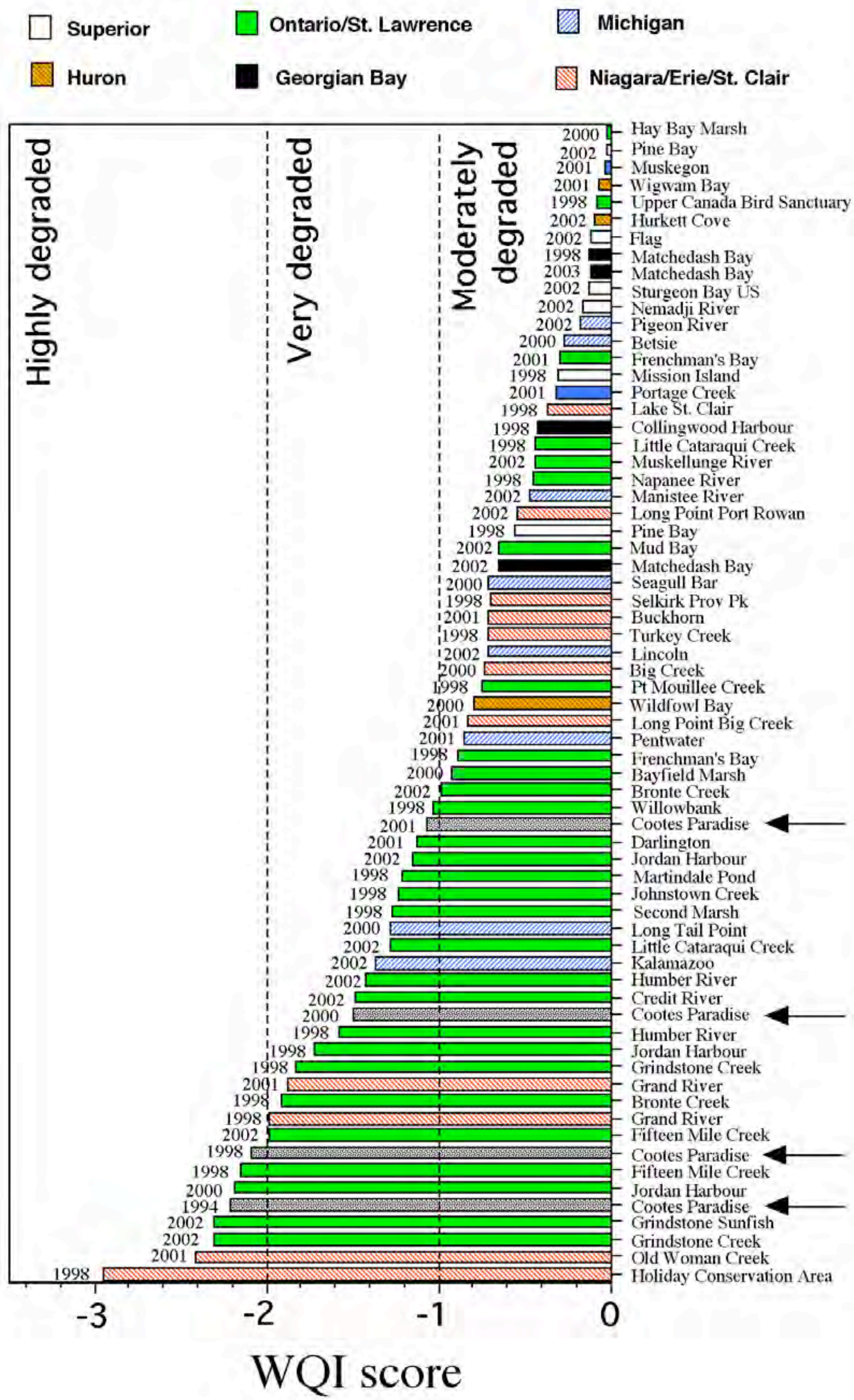

Figure 5.6 Wetlands with negative WQI scores are presented in descending order of rank. Bars corresponding to Cootes Paradise Marsh (Lake Ontario) are indicated by arrows. Years associated with the WQI scores appear to the left of the bars.

TABLE 5.5.
Comparison of ranks between 1998 and 2000,2001 or 2002 as determined by WQI scores.

Type of Change	Wetland	Rank in 1998	Rank in 2000, 2001 or 2002
None	Cloud Bay	Very good	Very good
	Rondeau Bay	Good	Good
	Spanish River	Good	Good
	Turkey Point	Good	Good
	Wellers Bay	Good	Good
	Chippewa Park	Good	Good
	Echo Bay	Good	Good
	Madoma Creek	Good	Good
	Presqu'ile Provincial Park	Good	Good
	Hay Bay Marsh	Good	Good
	Oliphant Bay	Good	Good
	Long Point Big Creek	Moderately degraded	Moderately degraded
	Matchedash Bay	Moderately degraded	Moderately degraded
	Frenchman's Bay	Moderately degraded	Moderately degraded
	Pine Bay	Moderately degraded	Moderately degraded
	Grand River Marsh (Dunville)	Very degraded	Very degraded
	Humber River	Very degraded	Very degraded
	Hurkett Cove	Good	Moderately degraded
Worsened	Grindstone Creek	Very degraded	Highly degraded
	Little Cataraqui Creek	Good	Very degraded
Improved	Fifteen Mile Creek	Highly degraded	Very degraded
	Bronte Creek	Very degraded	Moderately degraded
	Mission Island	Moderately degraded	Good
	Cootes Paradise Marsh	Highly degraded	Very degraded

1994, three years prior to the carp exclusion, the marsh had a very low WQI score, sharing the rank of "Highly degraded" with a handful of other wetlands (Old Woman Creek, Holiday Conservation Area, Grindstone Creek, Jordan Harbour, and Fifteen Mile Creek) (Figure 5.6). In 1998, one year following carp exclusion, the WQI score improved slightly, although it was still found within the "Highly degraded" category. In 2000 and 2001, the corresponding WQI scores continued to increase and placed the marsh in the "Very degraded" category. A key point is that these improvements in water quality have been accompanied by improved health of the zooplankton (Lougheed and Chow-Fraser 2002), plant, and fish communities (Lougheed et al. 2004).

To determine if WQI scores varied significantly between years for 23 other wetlands, I used both a paired t-test and a Wilcoxon sign-rank test to compare 1998 WQI scores with those from 2000, 2001 or 2002. There were no significant differences between years (two-tailed t test: $P=0.6338$, correlation=0.82439; Wilcoxon test: $P=0.746$), which indicated to me that data were reasonably well replicated through the five years. It confirmed the robustness of the indicator because the index is based on data collected during a single visit each summer. Of the 23 wetlands, 18 retained the same status, two worsened, while three improved, although it is not clear if these were due to management actions or to natural variation (Table 5.5). Nevertheless, if only 13% of the

TABLE 5.6.
Summary of regression equations to predict WQI scores

Eq. #	Variables in model	Associated r^2-value	Predictive equation
1	*All 12 variables: TURB, TSS, ISS, TP* **SRP, TAN, TNN, TN, COND, TEMP, pH, CHL**	1.00	+10.0239684 - 0.3154965 * log TURB -0.3656606 * logTSS -0.3554498 * log ISS -0.3760789 * log TP -0.1876029 * log SRP -0.0732574 * log TAN -0.2016657 * log TNN -0.2276255 * log TN -0.5711395 * log COND -1.1659027 * log TEMP -4.3562126 * log pH -0.2287166 * log CHL
2	**TURB, TSS, TP, COND** **TN**	**0.965**	+5.2427978 -0.298509 * log TURB -0.865436 * log TSS -0.626229 * log TP -0.818190 * log COND -0.330760 * log TN
3	**TURB, COND, TEMP, pH, TP, TN, CHL**	**0.964**	**+10.753047** **-0.946098* log TURB** **-0.837294 * log COND** **-1.319621 * log TEMP** **-4.604864 * log pH** **-0.387189 * log TP** **-0.353713 * log TN** -0.337888 * log CHL
4	**TP, TN, SRP, TNN, TAN, TSS, CHL**	0.963	+3.8311461 -0.629834 * log TP -0.271059 * log TN -0.083724 * log SRP -0.211261 * log TNN -0.119190 * log TAN -0.995406 * log TSS -0.243290 * log CHL
5	**TURB, COND, TEMP, pH, SRP, TNN, TAN**	0.947	+11.88597 -1.147966 * log TURB -1.048255 * log COND -2.308968 * log TEMP -4.653771 * log pH -0.278112 * log SRP -0.324002 * log TNN -0.116383 * log TAN

6	**TURB, COND, TEMP, pH, TP, TN**	0.947	**+11.590154 -1.073765 * log TURB -0.916011 * log COND -1.684796 * log TEMP -4.677050 * log pH -0.599127 * log TP -0.306512 * log TN**
7	**TURB, COND, TEMP, pH**	0.898	+9.2663224 -1.367148 * log TURB -1.577380 * log COND -1.628048 * log TEMP -2.371337 * log pH
8	**TP, TN, COND, CHL**	0.867	+5.2333056 -0.832012 * log TP -0.313032 * log TN -0.982628 * log COND -0.583014 * log CHL
9	**TP, TAN, TNN, TN, CHL**	0.853	+3.5161294 -0.985870 * log TP -0.195332 * log TAN -0.261192 * log TNN -0.171508 * log TN -0.599259 * log CHL

wetlands examined showed an improvement since 1998, it is probably reasonable to conclude that the positive trend in Cootes Paradise since 1994 reflects the effects of remedial actions rather than sampling error.

5.3.3 Predictive Models to Generate WQI Scores

Given that the WQI scores were calculated by summing PCA scores, I carried out a series of stepwise multiple regressions to derive predictive equations with which others could generate WQI scores from raw data (Table 5.6). Besides the 12-variable model that describes the total variation in WQI scores (Eq. 1), there are a number of predictive equations that only require five to seven variables that are commonly collected in routine monitoring programs by environmental agencies. Since the r^2-values associated with Eq. 2 to 6 inclusive are uniformly high (0.947 to 0.965), they should generate WQI scores that are comparable to each other. I have included Eq. 7 and 8, even though the associated r^2-values are lower (0.90 and 0.85, respectively) because the parameters involved are commonly available.

5.3.4 Relationship Between WQI and Basin-Wide Land Use

To determine if WQI scores are significantly related to basin-wide land use, I conducted a Pearson correlation analysis. The land use variables I used were PROPFOR and PROPALT, which correspond to the proportion of forested and altered land (combination of both urban and agricultural land), respectively. Because the data were suspected to be spatially autocorrelated, spatial correlograms were used to identify the scales of variation in WQI and the land-use data. The distribution of Moran's I indicate that the range of influence of autocorrelation for both independent variables were similar, at three distance units, while the water quality data were spatially autocorrelated at one distance unit, with one distance unit being approximately 1.3 decimal degrees. Since the data were spatially autocorrelated, Dutilleul's (1993) correction had to be applied to the data prior to the correlation analysis (Wei et al. 2004). WQI scores were significantly correlated with both arcsin PROPFOR ($n=81$, $r=0.59049$, $P=0.04763$) and arcsin PROPALT ($n=64$, $r=-0.66026$, $P=0.02295$). Had I not corrected for the spatial autocorrelation, the unadjusted P-values associated with the correlation coefficients for both would have been <0.0001 and the corresponding r-values would have been 0.64 and 0.72, respectively.

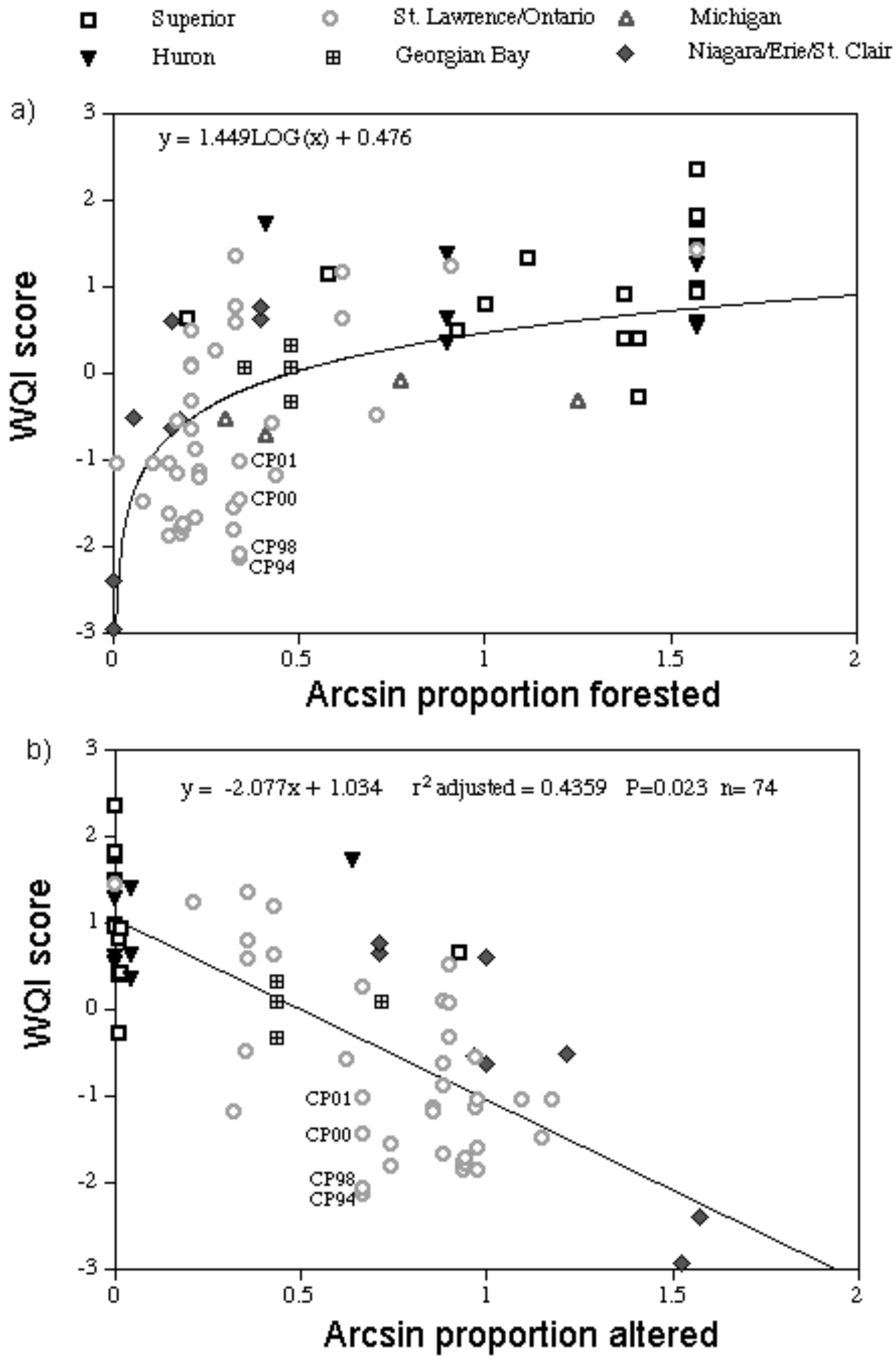

Figure 5.7 Relationship between WQI scores and a) arcsin proportion forested land (n=81) and b) arcsin proportion altered land (n=74). Data for Cootes Paradise (CP) from 1994 to 2001 are indicated.

Since both relationships were significant even after correcting for spatial autocorrelation, I plotted WQI scores against arcsin PROPFOR and arcsin PROPALT to provide a means of predicting water-quality conditions from basin-wide land use information. Rather than fitting a straight line through the PROPFOR data, I obtained a better fit with a logarithmic regression (Figure 5.7a). If we assume that only negative WQI scores indicate degraded conditions, then the minimum amount of forested land in the watersheds should not drop below arcsin proportion of 0.50 (interpolated from the y-axis to the x-axis), which is approximately 48%. By comparison, the influence of developed land on water quality appears to be linear, indicating that there is no threshold effect (Figure 5.7b). Data for Cootes Paradise are indicated in both panels, and confirm that management actions can effect substantial changes in water-quality conditions, irrespective of basin-wide changes in land use. Improved riparian conditions through provision of buffer strips (e.g. Snyder et al. 2003) probably account for some of the high WQI scores above the best-fit line in Figure 5.7a and b, and future research should be devoted to this line of inquiry.

I regrouped the data into four categories based on dominant land use in the wetland catchment: mainly forested, mixed land-use development, mainly urban, or mainly agricultural (Figure 5.8). The was an uneven distribution among the four categories, with many more agricultural watersheds than any of the other types. Not surprisingly, mean WQI scores for the mostly undeveloped watersheds (forested) were highest, and the mean was not significantly different from that of wetlands with mixed development (where the combined percentage of altered land did not exceed 50%). This is consistent with the earlier observation that good water-quality conditions tended to be maintained as long as the percentage of undeveloped land in the watershed remained above 50% (Figure 5.7a). Wetlands in primarily agricultural watersheds yielded the lowest mean WQI score (-1.009), and this was significantly lower than the primarily urbanized wetlands, which had a mean WQI score of –0.2062.

5.3.5 Comparison of WQI with Other Indices

With the recent interest in indices development, I was able to assemble published information to compare with my ranks for 9 of the wetlands (Table 5.7). Five of these sites were ranked in more or less the same way in the published source as they were in this study (Bark Bay, Pentwater River, Mackinaw Bay, Mismer Bay and Wild Fowl Bay). However, for four of the wetlands, there were some notable differences. The greatest deviation was found for Matchedash Bay, which had been ranked by Minns et al. (1994) as being 68% "Good" and 24% "Fair", but which was ranked in this study as being moderately degraded in all three years of sampling (1998, 2002, and 2003). One reason for this disparity may be that Minns et al's data had been collected over a decade earlier (1990), and environmental conditions in the marsh have since deteriorated. An alternate explanation is that the two studies do not share the same reference point, and hence, conditions considered moderately degraded in this study had been deemed to be good in theirs.

There were three other discrepancies in Table 5.7 that should be pointed out. The two Lake Michigan wetlands, Betsie and Lincoln River, were ranked as "Good" in Wilcox et al.'s study, but were ranked as "Moderately degraded" in this study. Another difference is that Betsie and Lincoln had identical total IBI scores (82; Table 9 in Wilcox et al. 2002), whereas in this study, Betsie had a substantially higher WQI score than did Lincoln (Figure 5.6). Unfortunately, without more site-specific information, it is virtually impossible to determine which index produced the more accurate assessment.

The last wetland I will mention is Cedarville, which had been identified

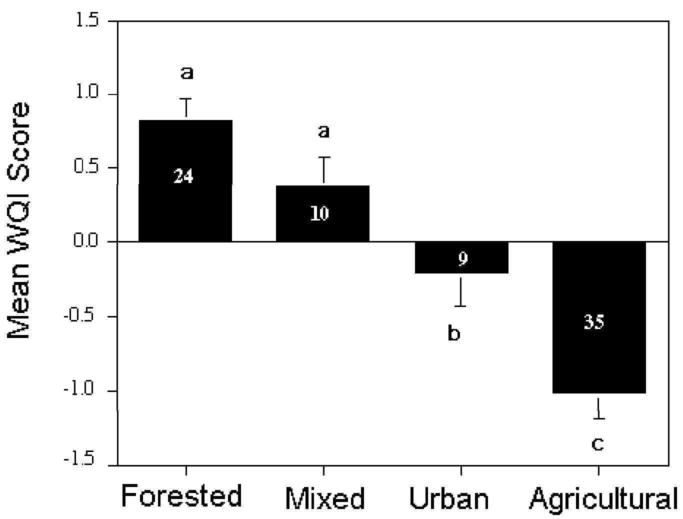

Figure 5.8 Comparison of mean WQI scores for various land-use categories. Similar letters indicate that means are statistically homogeneous (ANOVA: P<0.0001; Tukey-Kramer multiple comparisons P>0.05). Numbers indicate the number of wetlands in each category.

TABLE 5.7.
Comparison of ranks determined by various indices for subset of coastal marshes in this study. Ranks that are in bold indicate that ranks assigned to wetlands differed between studies.

Lake	Wetland	Source	Index used	Rank
Superior	**Bark Bay**	**Wilcox et al. 2002**	**Plant, Fish, Invertebrate IBI**	**Good**
		This study	**WQI**	**Good**
Michigan	**Betsie River**	**Wilcox et al. 2002**	**Plant, Fish, Invertebrate IBI**	**Good**
		This study	**WQI**	**Moderately degraded**
	Lincoln River	**Wilcox et al. 2002**	**Plant, Fish, Invertebrate IBI**	**Good**
		This study	**WQI**	**Moderately degraded**
	Pentwater River	**Wilcox et al. 2002**	**Plant, Fish, Invertebrate IBI**	**Poor**
		This study	**WQI**	**Poor**
Huron	**Cedarville Bay**	**Burton et al. 1999**	**Invertebrate IBI**	**Moderately degraded**
		This study	**WQI**	**Good**
	Mackinaw Bay	**Burton et al. 1999**	**Invertebrate IBI**	**Mildly impacted**
		This study	**WQI**	**Good/Moderately degraded**
	Mismer Bay	**Burton et al.1999**	**Invertebrate IBI**	**Reference Condition**
		This study	**WQI**	**Good**
	Wild Fowl Bay	**Burton et al. 1999**	**Invertebrate IBI**	**Mildly impacted**
		This study	**WQI**	**Moderately degraded**
Georgian Bay	**Matchedash Bay**	**Minns et al. 1994**	**Fish IBI**	**Good-Fair**
		This study	**WQI**	**Moderately degraded**

as being "Moderately degraded" by Burton et al.'s (1999) benthic invertebrate IBI. However, on the basis of its good water quality, I placed it in the "Good" category, above Mismer Marsh (Figure 5.6), which had been identified as having "reference conditions" (Table 5 in Burton et al. 1999). When I re-examined the data, it was clear that Cedarville Bay had very low water turbidity and nutrient concentrations, and only moderate levels of water conductivity. None of these characteristics seem to reflect the disturbances listed in Burton et al.'s (1999) Table 1, which included sewage lagoon discharge, urban runoff and marina traffic. It is quite possible that the marina traffic in the bay had some overriding negative effect on the biotic community that made it possible to have a relatively high WQI score and a low IBI score at the same time.

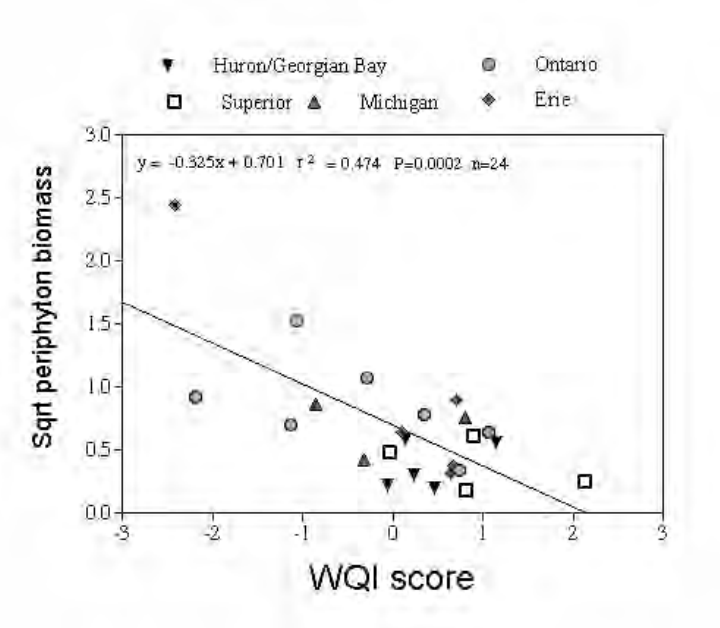

Figure 5.9 Relationship between square-root of periphyton biomass (µg/cm2/d) (from McNair and Chow-Fraser 2003) versus WQI score for 24 wetlands in this study.

Having shown a direct link between basin-wide land use and the Water Quality Index, I then investigated the relationship between WQI and wetland biota. Recently, McNair and Chow-Fraser (2003) showed that periphyton biomass (µg/cm^2/d) (grown on artificial substrate for a standardized period) varied as a function of water-quality conditions in 24 wetlands. I regressed these data against corresponding WQI scores generated in this study, and found a significant negative relationship (r=0.688; P=0.0002) that appeared to be upheld for all five Great Lakes (Figure 5.9). Next, I assembled information to calculate the species richness of submergent vegetation for a large number of wetlands in this database (n=131 wetland-years) and regressed these against WQI scores. For this analysis, I divided the wetlands into two groups based on their location north or south of the 45°N Latitude, to better reflect differences in climate and basin geochemistry that can affect plant distribution patterns (McNair and Chow-Fraser, unpub. data). In both cases, species richness of submergent plants varied significantly with WQI score, although the relationship for southern wetlands had higher r^2-value (Figure 5.10a) than that for the northern wetlands (Figure 5.10b). This means that wetlands in the north have fewer submergent species than those in the south, even if water-quality conditions were identical. In fact, majority of the Georgian Bay wetlands, especially those in the "Excellent" category (Figure 5.5), have surprisingly few species compared with wetlands that were considered to be degraded within the same geographic region (e.g. Sturgeon Bay Central; ST in Figure 5.10b).

5.4 DISCUSSION

One of the major goals of this paper is to develop an index of wetland quality that is 1) cost-effective, 2) sufficiently robust to be applied to a wide range of environmental conditions and eco-regions, and 3) sufficiently sensitive to detect incremental losses and gains at the site level. In the development of the Water Quality Index, I tried to use variables that are routinely monitored by environmental agencies so that existing monitoring programs could provide data for calculating WQI scores. The choice of turbidity over Secchi depth or light extinction requires an explanation since the former may not currently be measured by environmental agencies. I chose turbidity over Secchi depth because the latter is an insensitive measure of under-water irradiance in low water-level scenarios, such as those currently experienced by pristine wetlands of Georgian Bay. Secchi depth was also insensitive to improved water clarity associated with carp exclusion in Cootes Paradise Marsh because of the drop in water depth following the biomanipulation (Chow-Fraser 2004). Light extinction coefficient has a similar depth constraint, is more time-consuming to collect, and requires the use of expensive equipment. As is true for any index, parameters included in an initial derivation can be deleted and other more useful parameters added in subsequent versions. Therefore, the WQI may need to be modified to account for the effect of water colour, especially when assessing dystrophic wetlands of upper Georgian Bay, where reduced light availability does not necessarily reflect anthropogenic impact.

One of the major criticisms levied against the use of biotic indices to detect the effect of human disturbance in coastal wetlands is that the target communities (fish, macroinvertebrates, plants) are often simultaneously responding to other natural stressors such as water level in addition to anthropogenic impacts (see Figure 5.1), and hence, the resulting IBI scores may lead to erroneous conclusions (Wilcox et al. 2002). In the case of the Laurentian Great Lakes, differences in regional climatic conditions (Minc 1997) may also mask any effect of human disturbance when the indicator is applied throughout the basin (Seilheimer and Chow-Fraser unpub. data). Here, I am not arguing against the use of biological indicators per se, but rather I argue for more effort to be put towards developing indicators (chemical, physical or biological) that can be linked to a well-defined stressor, rather than to developing indicators of overall biological/community health (Goldstein et al. 2002).

The strength of a large-scale approach such as that used in this study is that managers have an opportunity to scale and rank wetlands in their jurisdiction against those that occur throughout the Great Lakes basin. But even if managers responsible for wetlands in Lake Erie and Ontario are not interested in comparing their wetlands with those in Lake Superior, they can use the WQI to rank wetlands on a lake-by-lake basis. Wetlands sampled in the two lower Great Lakes were found in all except the "excellent" category (Figure 5.11), and this confirms the general applicability of the WQI on both a regional and basin-wide scale. If they need to establish a more locally relevant ranking system, managers could use biotic indices such as the Wetland Zooplankton Index (Lougheed and Chow-Fraser 2002) to resolve small changes in food-webs resulting from site-level impacts that are not related to landscape-level alterations. Further effort should be devoted to documenting the relationship between biotic indices and the water-quality index so that individual metrics that are diagnostic of water-quality impairment can be identified.

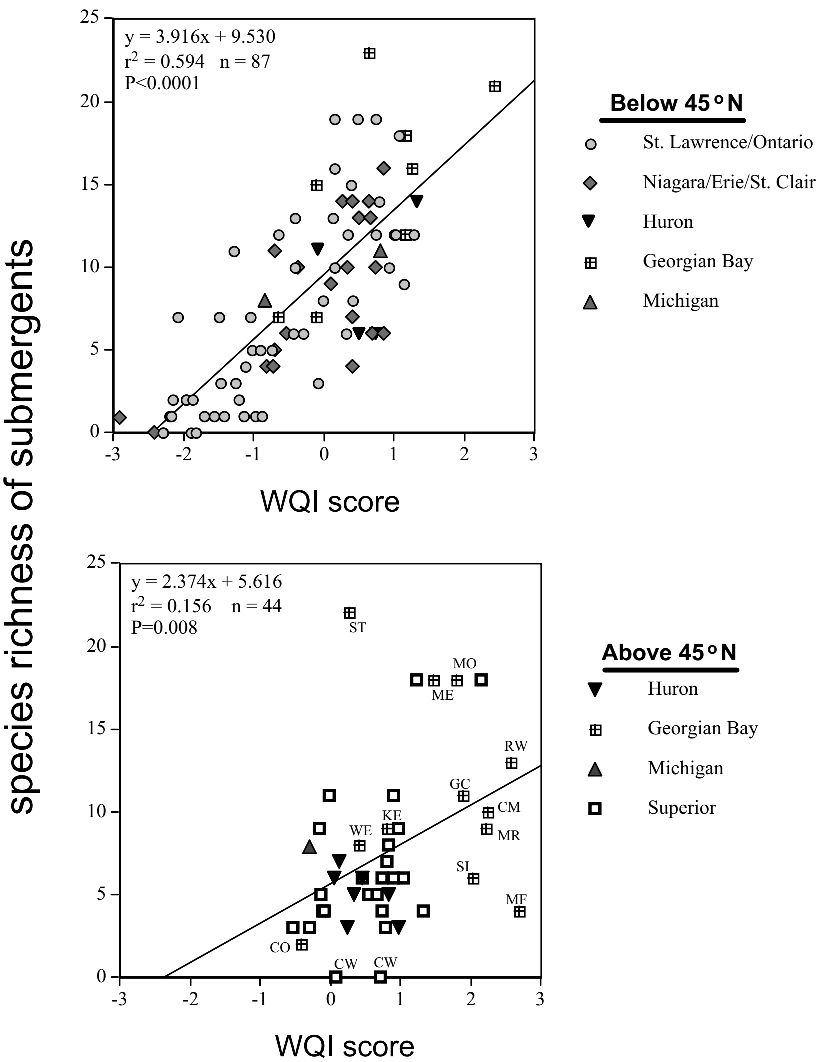

Figure 5.10 Relationship between species richness of submergent aquatic vegetation and WQI scores for two groups of wetlands, according to their location south or north of the 45°N Latitude.

Many recent investigators have identified the conversion of forested land in watersheds to be a primary cause of water-quality impairment in freshwater ecosystems (Field et al. 1996; Müller et al. 1998; Crosbie and Chow-Fraser 1999; Nelson and Booth 2002;

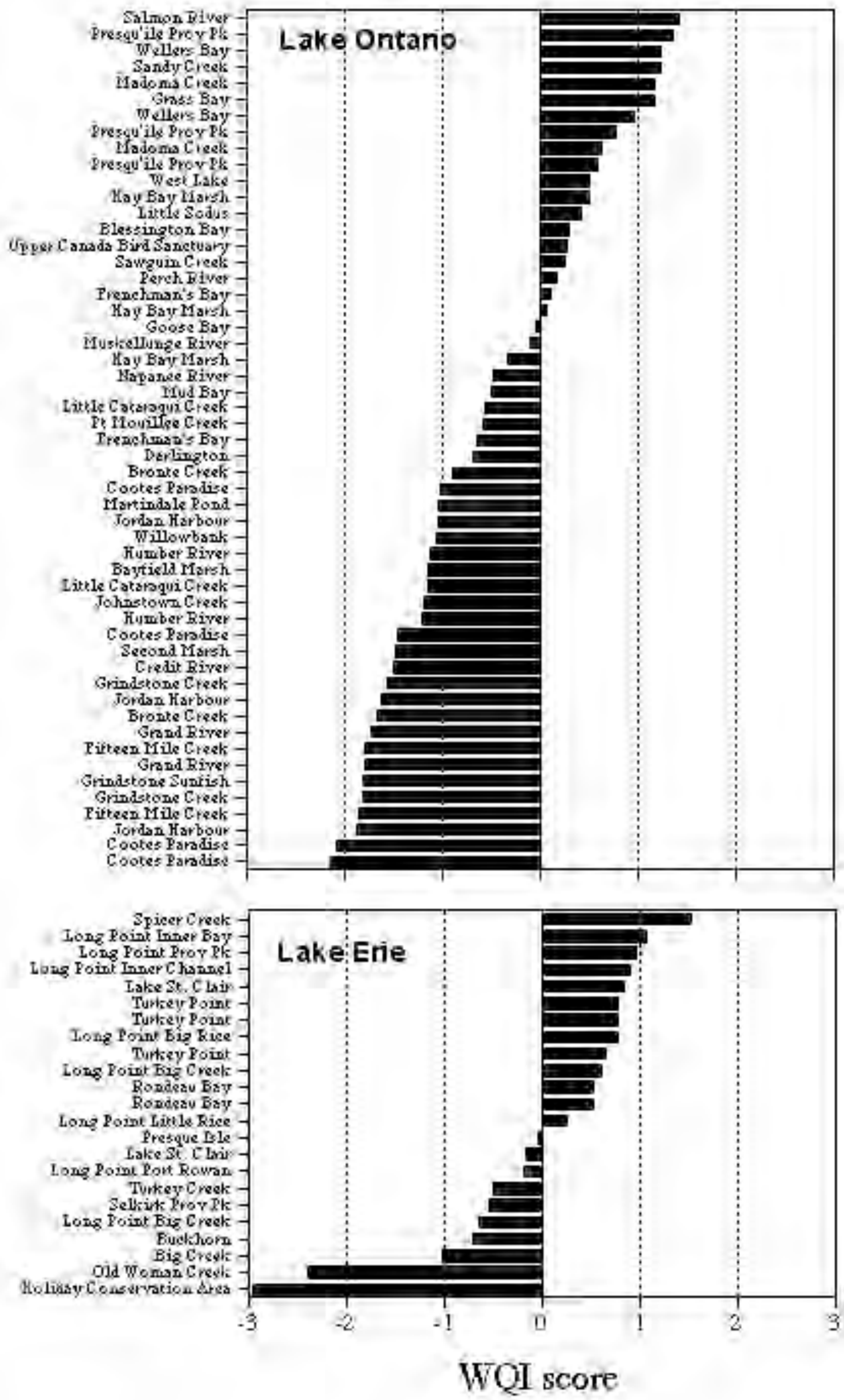

Figure 5.11 WQI scores of Lakes Ontario and Erie in descending order of quality.

Houlahan and Findlay 2003). Others have noted that naturalized shorelines and presence of buffer strips can offset the delivery of nutrients and sediments into streams from altered landscapes (Lammert and Allan 1998; Meador and Goldstein 2003). This study is one of the largest efforts in linking water-quality impairment to basin-wide land use for Great Lakes coastal wetlands. Although there were potentially 110 wetlands that could have been included in the analysis, there was incomplete land-cover information and only 81 wetlands were included in the final correlation analysis of proportion of forested land (Figure 5.7a) and 74 for proportion of altered land (Figure 5.7b). The lack of current land-cover data for both Canada and the United States also meant that outdated land-use maps had to be used in many instances, and this probably contributed to the considerable scatter in Figure 5.7. Despite these errors, however, the relationships between water-quality condition and land use were robust, and could be applied to all five Great Lakes. I want to emphasize the importance of using appropriate spatial scale to examine the effects of basin-wide land use because on a lake-by-lake basis, land-use effects would not have been significant for any of the upper lakes (L. Michigan, Huron and Superior) due to the restricted range in land use type. One of the management implications of these results is that watersheds should maintain at least 50% forested land to ensure that water quality does not become degraded (Figure 5.7a). On the other hand, the linear decline in WQI scores with increasing proportion of altered land suggests that the same deleterious effect on water quality would apply regardless of the proportion of land that is already developed (Figure 5.7b). Future studies should focus on determining possible thresholds of impervious areas such as that found for Alaskan streams (Ourso and Frenzel's 2003). The data should also be re-examined with higher-resolution spatial information to determine ameliorating effects of buffer strips on downstream water quality in these coastal wetlands, and to differentiate among effects of different agricultural enterprises (soya versus corn, dairy production versus intensive hog farming, etc.).

This study also provided a rare opportunity to examine how water quality changed over the course of a restoration project, in the absence of substantial changes in basin-wide land use. In additon to nutrient and sediment enrichment from non-point sources, Cootes Paradise Marsh had also been degraded by a number of other stressors, including disturbance by the common carp (*Cyprinus carpio*), wind and wave resuspension, and discharge from a wastewater treatment facility (Chow-Fraser et al. 1998). Since carp disturbance was deemed to be one of the primary causes of marsh degradation (Chow-Fraser 1998), the restoration plan included a large-scale carp exclusion that eventually removed 90% of the carp in the marsh (Lougheed et al. 2004). Prior to the carp exclusion in1997 that removed close to 90% of the carp, the WQI score for Cootes Paradise indicated that the marsh was "Highly degraded" (CP94) (Figure 5.7). The first year after the biomanipulation, water-quality conditions improved only slightly (CP98), but within the next four years (CP00 and CP02), the WQI score had increased from –2.20 to –1.06, indicating that the marsh was approaching the "moderately degraded" state. This type of water-quality improvement was accompanied by an increase in the species richness of the submergent community from 1 in 1994 to 7 in 1998. Unfortunately, when water levels remained low during the summer of 1999, many of the submergent species died back, and the emergent community has colonized much of the submergent habitat since that time (Chow-Fraser 2004). This points out the difficulty in using biotic indicators to track remedial actions in the presence of overriding effects of natural stressors such as extreme interannual water-level fluctuation.

Like all primary producers, one of the main determinants of macrophyte growth is availability of primary nutrients. Since aquatic plants obtain nutrients from both sediments and the water column, the nutrient content of the water can help determine the species composition and productivity of the wetland (Wisheu *et al*., 1992), especially when the substrate is naturally impoverished due to basin geology. In some of the Georgian Bay wetlands, where water is dystrophic, the extremely low TP and TN concentrations may limit the diversity of the plant community to only a few highly competitive species. When these wetlands occur in recreational lakes, increased nutrient loading from human activities, however, would tend to increase the species diversity and richness of the aquatic-plant community. This may explain why there were many more submergent species in Sturgeon Bay (ST) compared with Moon River (MR) or Sandy Island (SI) (Figure 5.9b) since the former is a heavily used recreational lake with a well developed cottage community and various campgrounds along the shoreline, whereas the latter two are essentially undeveloped. This apparent inverse relationship between species richness and WQI scores for the Georgian Bay wetlands is consistent with the observed increase in diversity of vascular plants along an upstream-downstream nutrient gradient of a weakly mineralized stream (Thiébaut and Muller 1998) and lakes with low alkalinity (Vestergaard et al. 2000). Thiébaut and Muller (1998) also demonstrated that the species composition of the downstream sites changed from one indicative of oligotrophic to one indicative of eutrophic conditions.

The WQI proved to be a valuable indicator of basin-wide land-use effects. It has the power to rank wetlands according to water quality across the entire Great Lakes basin, and was able to produce results that were generally consistent with published biotic indicators. It is sufficiently sensitive to track changes within a site over the course of a marsh restoration project, and permitted a direct link between improved water quality and removal of carp from Cootes Paradise Marsh. The index is also robust, because it was well replicated between years for 18 of 23 wetlands examined between 1998 and 2002. Water Quality Index scores can be generated from a variety of multiple-regression equations, involving as few as five field variables. I believe the WQI is an effective indicator of human-induced land-use alterations, and suggest that future investigations use this abiotic indicator to help develop their biotic indices.

5.5 CONCLUSIONS

Many factors contribute to water-quality impairment in coastal wetlands of the Great Lakes. Among these are non-point source inputs of sediment and nutrient from agricultural and urban runoff, point-source pollution from municipal or industrial waste-treatment facilities, and carp bioturbation. Regardless of the pollution source, the resulting eutrophic and turbid conditions generally lead to a higher biomass of benthic algae, which can reduce the species richness of submergent plants, and which can in turn affect the species richness, species composition and size structure of higher trophic levels. In this paper, I use water-quality data collected from 110 widely distributed wetland complexes (146 wetland-years) to develop a "Water Quality Index" (WQI). The WQI scores were then statistically related to proportion of forested and altered land in wetland catchments and these scores were used to rank the degree of water-quality impairment in all 110 wetlands across the Great Lakes basin, and to track changes in Cootes Paradise Marsh over an 8-year period (1994-2001) before and after a carp-exclusion program. For a subset of wetlands, WQI scores compared well with published IBI ranks derived from benthic macroinvertebrate, plant and fish data. There was a significant positive association between water quality (WQI scores) and higher trophic levels, including biomass of benthic algae and species richness of submergent plants. By directly linking biotic indicators to WQI and percentage land use, I show that the WQI is a reliable indicator of human-induced land use alterations, and should provide an independent and objective means of assessing anthropogenic impacts when developing indices of biotic integrity.

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