Percentage land use in the watershed determines the water and sediment quality of 22 marshes in the Great Lakes basin

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Abstract: Data from 22 Ontario marshes were used to test the hypothesis that distribution of forested, agricultural, and urban land in the watershed determines the water and sediment quality of Great Lakes wetlands. The first three components of the principal components analysis explained 82% of the overall variation. PC1 ordinated wetlands along a trophic gradient; species richness of submergent vegetation decreased with PC1 scores. PC2 reflected the content of inorganic solids and phosphorus in sediment and the ionic strength of the water. Both PC1 and PC2 scores were positively correlated with percent agricultural land, whereas PC1 scores were negatively correlated with forested land. Correlation between PC1 and agricultural land improved when best-management practices were considered. Accounting for common carp (*Cyprinus carpio*) disturbance did not confound the relationship between land use and water quality. PC3, driven by soluble reactive phosphorus and nitrate nitrogen concentration in the water, was not correlated with land use. Concentrations of polycyclic aromatic hydrocarbons and Metolachlor were correlated with urban and agricultural land, respectively, and may be useful as land use surrogates. Watershed management favouring the retention of forested land, or creation of buffer strips to trap agricultural runoff in the drainage basin, should help maintain aquatic plant diversity in coastal wetlands.

Résumé : Nous avons utilisé les données de 22 marais de l'Ontario afin de vérifier l'hypothèse selon laquelle la qualité de l'eau et des sédiments des milieux humides des Grands Lacs dépend de la répartition des terres boisées, agricoles et urbaines dans le bassin hydrographique. Les trois premières composantes de l'analyse des composantes principales expliquaient 82 % de la variation totale. La composante principale 1 (CP1) plaçait en ordonnée les milieux humides le long d'un gradient trophique; la richesse des espèces végétales partiellement submergées diminuait avec les valeurs de la CP1. La CP2 correspondait à la teneur en matières solides inorganiques et en phosphore des sédiments et à la force ionique de l'eau. Les valeurs de la CP1 et de la CP2 présentaient une corrélation positive avec le pourcentage de terres agricoles, tandis que les valeurs de la CP1 présentaient une corrélation négative avec les terres boisées. La corrélation entre la CP1 et les terres agricoles était meilleure lorsque l'on considérait les meilleures pratiques de gestion. Le fait de prendre en compte la perturbation de la carpe n'a pas troublé le rapport entre l'utilisation des terres et la qualité de l'eau. La CP3, commandée par la concentration de phosphore réactif soluble et de l'azote des nitrates dans l'eau, n'était pas corrélée à l'utilisation des terres. Les concentrations d'hydrocarbures aromatiques polycycliques et de métolachlore étaient corrélées respectivement avec les terres urbaines et les terres agricoles, et peuvent être utiles comme substituts de l'utilisation des terres. La gestion des bassins hydrographiques favorisant la conservation des terres boisées, ou la création de bandes tampons pour piéger les eaux de lessivage des terres cultivées dans le bassin hydrographique, devrait permettre le maintien de la diversité des plantes aquatiques dans les milieux humides côtiers.

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Introduction

It has been recognized for over two decades that different amounts and types of nutrients exported from agricultural, urbanized, and forested landscapes have a direct effect on the trophic quality of downstream rivers and lakes (Beaulac and Reckhow 1982; Peterjohn and Correll 1984; Nelson et al. 1996). These studies have focussed on phosphorus and nitrogen, since they are known to limit the growth of algae in lakes. Using export coefficients associated with different land use categories, investigators have been able to predict or hindcast changes in lake water nutrient concentrations in response to projected or past land use alterations (e.g., Frink 1991; Johnes et al. 1996; Mattikalli and Richards 1996).

By comparison, the impact of different land uses on the water quality of wetlands has seldom been documented, and seldom for more than one watershed at a time (Ehrenfeld 1983; Smith et al. 1991). The tendency to use nutrient levels alone to indicate trophic quality may be problematic for wetlands, since both submersed aquatic vegetation and algae are primary producers, and macrophytes are much more limited by light availability (Hough et al. 1989; Lougheed et al. 1998; Chow-Fraser 1998) and substrate type (Barko et al.

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Table 1. Summary of general descriptions of wetlands in the study.

Wetland	Latitude, longitude	Wetland size (km^2)	Watershed size (km^2)	Wetland type	Hydrologic character	Common carp disturbance
1. 15 Mile Creek Marsh	43°10′00″, 79°19′00″	0.31	63	Riverine	No flow	Yes
2. Big Creek Marsh	42°57′20″, 80°26′50″	0.61	200	Riverine	No flow	Yes
3. Centreville Creek Marsh	43°54′04″, 79°50′00″	0.13	35	Riverine	Observed flow	
4. Christie Lake Marsh	44°47′00″, 76°28′00″	1.13	66	Lacustrine	No flow	
5. Cootes Paradise Marsh	43°16′00″, 79°55′00″	2.5	268	Riverine	No flow	Yes
6. Harris Lake Marsh	45°42′00″, 80°82′00″	20	>1000	Lacustrine	No flow	
7. Hay Bay Marsh	44°10′30″, 76°55′30″	13.33	225	Riverine	No flow	Yes
8. Holiday Marsh	42°02′05″, 83°03′00″	11.77	64	Riverine	No flow	Yes
9. Humber River Marsh	43°38′00″, 79°29′00″	0.26	667	Riverine	No flow	Yes
10. Joe’s Lake Marsh	45°08′00″, 76°41′00″	2.04	268	Lacustrine	No flow	
11. Jordan Harbour Marsh	43°11′00″, 79°23′00″	1.72	311	Riverine	No flow	Yes
12. Martindale Pond Marsh	43°10′07″, 79°16′00″	0.55	29	Riverine	No flow	Yes
13. Presqu’ile Marsh	44°00′00″, 77°43′00″	9.92	22.2	Lacustrine	No flow	
14. Sawguin Marsh	44°06′00″, 77°23′00″	20.93	91	Lacustrine	No flow	Yes
15. Second Marsh	43°52′00″, 79°51′00″	1.06	85	Lacustrine	No flow	Yes
16. Shebeshekong River Marsh	45°24′30″, 80°19′00″	1.08	107	Riverine	Moderate flow	
17. Stump Lake Marsh	44°56′48″, 76°38′12″	0.93	1060	Lacustrine	No flow	
18. Sutton Pond Marsh	42°50′00″, 80°18′00″	0.10	85	Riverine	No flow	Yes
19. Tay River Marsh	44°52′45″, 76°10′30″	5.53	669	Riverine	No flow	
20. Tobies Bay Marsh	44°51′00″, 79°47′00″	1.94	9	Riverine	No flow	
21. Turkey Creek Marsh	42°14′08″, 83°05′07″	0.32	5	Riverine	No flow	Yes
22. Waterford Pond Marsh	42°56′10″, 80°18′45″	2.39	71	Lacustrine	No flow	Yes

1991) than by nutrient concentration in the water column. Hence, studies that do not consider nonalgal turbidity and sediment quality may have limited value for understanding land use impacts on the trophic quality of wetland communities. In the Great Lakes basin, in-marsh effects such as sediment resuspension by the common carp (*Cyprinus carpio*) are also known to contribute significantly to water turbidity (e.g., Whillans 1996; Lougheed et al. 1998; Chow-Fraser 1998) and may confound any external effects from non-point-sources in the watershed.

This study was initiated to determine the impact of land use on water and sediment quality of coastal marshes in the Great Lakes basin because these wetlands have been lost at an alarmingly high rate following European settlement (Whillans 1982). This loss has occurred in spite of their great value in water storage and groundwater recharge, flood prevention, and sediment and nutrient filtration and their vital ecological role in maintaining biodiversity and providing habitat for plants, birds, mammals, fish, and invertebrates (Maynard and Wilcox 1996).

Until recent decades, most of these wetlands were lost through conversion to agricultural land and (or) use for urban development. At present, however, wetlands are more at risk from non-point-source pollution and in-marsh stresses such as carp disturbance and water level regulation than from dredging, infilling, or draining (Keddy and Reznicek 1986; Maynard and Wilcox 1996). If governments are to seriously devise strategies to protect and restore the remaining wetlands, scientists must begin to provide some basic understanding of factors that impair water quality of wetlands, since deterioration in water quality has been consistently linked to marsh degradation (Crowder and Painter 1991; Chow-Fraser et al. 1998).

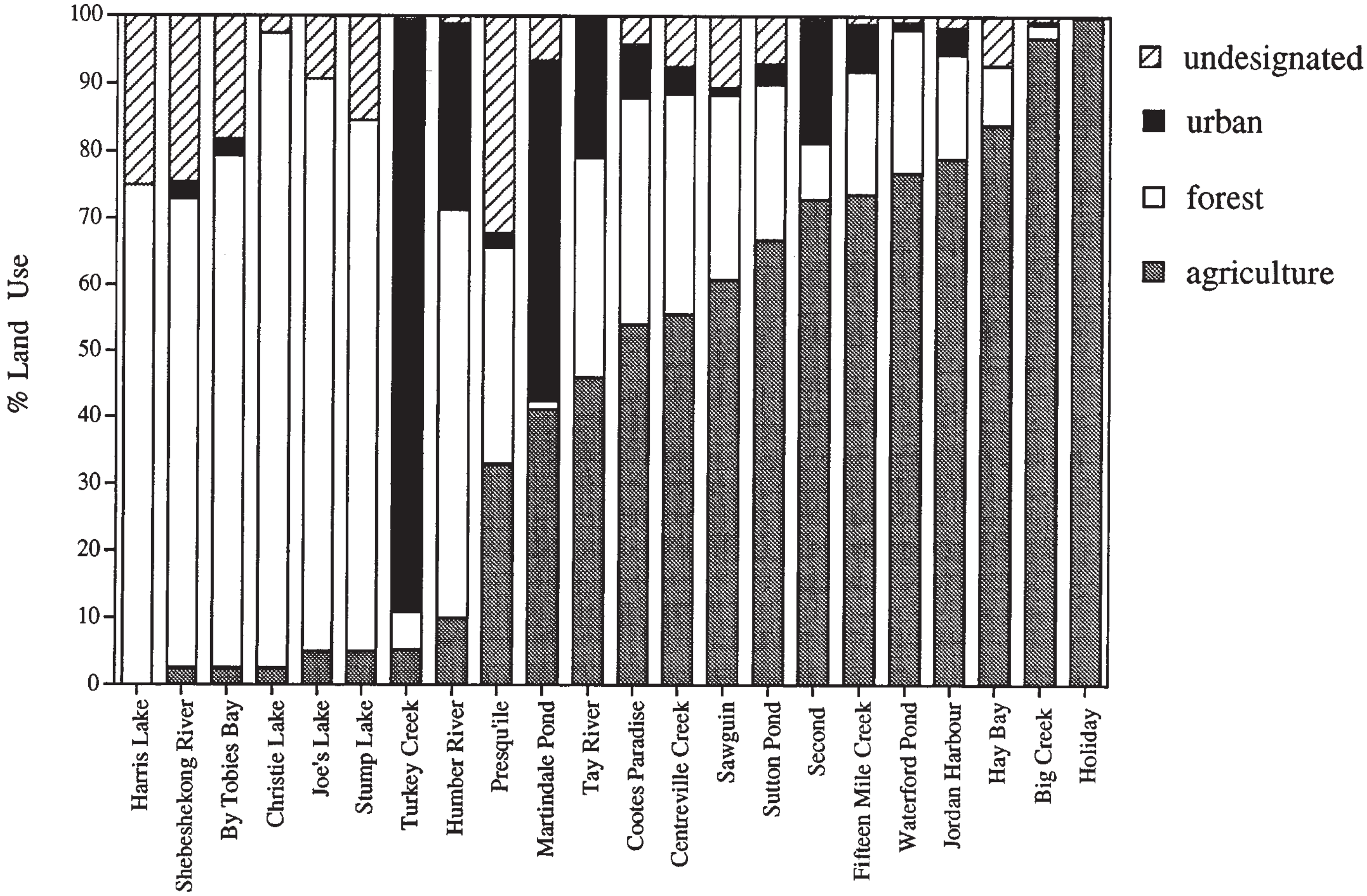
In this paper, we present seldom-documented water and sediment quality information for 22 Ontario marshes in the Great Lakes basin. We used principal components analysis (PCA) to create a multidimentional index of trophic quality for each wetland to test the relationship between this index and the relative distribution of forested, agricultural, and urban land in respective watersheds. We also simultaneously consider the impact of land use both in the presence and absence of carp disturbance to evaluate the relative importance of the latter on water quality in wetlands. Since current land use information is often difficult to acquire, we investigate the potential for using organic contaminants specific to agricultural or urban landscapes to predict land use. The level of polycyclic aromatic hydrocarbons (PAHs) in wetland sediment was used to predict the proportion of urbanized land in our watersheds, since PAHs are common pollutants associated with roadway runoff (Maltby et al. 1995). As a predictor of the proportion of agricultural land, we used the concentration of Metolachlor in upstream creeks and rivers that feed into our wetlands, since this herbicide is typically associated with agricultural runoff (Gaynor et al. 1995). Finally, we relate water quality impairment associated with basin development to changes in the species richness of submersed aquatic vegetation and thus provide a direct link between land use and one possible measure of ecosystem health in coastal wetlands.

Materials and methods

Study sites

Twenty-two wetlands, covering a broad spectrum from highly disturbed to pristine conditions, were sampled for a suite of water quality variables and vegetation in 1995 and 1996 (Table 1). Wet-

Fig. 1. Relative distribution of different land uses in the watersheds of the 22 wetlands in this study.



land sites were chosen based on the amount of urban, agricultural, and forested land in their watershed to ensure a sufficient gradient of disturbance (Fig. 1). A survey of the submergent plant community was conducted in each wetland at the time of sampling, as described in Lougheed et al. (1998). All emergent and submergent aquatic vegetation was identified in the field to taxa and samples of representative species were collected and identified in the laboratory to species. It was not our intent to conduct a comprehensive plant survey of each wetland but to choose a site within the wetland that had a structurally diverse plant community consisting, where possible, of emergent, submergent, and floating taxa interspersed with open water. Hence, only plant taxa in the vicinity of the area sampled for water and sediment were critically evaluated. This area totalled a minimum of 100 m^2 in each of the 22 wetlands, but in some cases more than 400 m^2. Emergent taxa encountered in this survey included species of *Typha*, *Scirpus*, *Pontederia*, *Sparganium*, *Decodon*, and *Sagittaria*; submergent taxa included species of *Vallisneria*, *Potamogeton*, *Vallisneria*, *Utricularia*, *Ceratophylum*, *Najas*, and *Elodea* (Crosbie 1997).

Field collection and observations

At each site, we collected water and sediment samples twice during the summer in either 1995 or 1996. Water and sediment samples were collected from an open-water location at least 3 m away from a stand of emergent vegetation, except in the case of Centreville Creek Marsh where the sampling station was restricted to a vegetated tributary less than 1 m wide. In situ measurements of pH and specific conductance were recorded with a Hydrolab H20 sonde equipped with a Scout2 monitor. The Hydrolab was calibrated for conductivity, pH, and dissolved oxygen with known standards prior to each sampling trip. Turbidity values were collected with a Hach turbidimeter (model 2100P) that was calibrated biweekly with known standards. Secchi depth transparencies were recorded in triplicate with a 20-cm disc from the shaded side of the sampling vessel. To control for differences due to weather conditions, we excluded data collected during rain events. Therefore, all samples were collected following at least 72 h of dry weather. The presence of carp at each site was verified visually during our sampling and (or) from wetland evaluation reports by the Ontario Ministry of Natural Resources. These data are purely qualitative, as they represent only the presence–absence of carp, but we know from other studies that the presence of carp can alter water clarity over a wide range of fish biomass (Lougheed et al. 1998).

Nutrient, chlorophyll, and suspended solids analyses

On each sampling occasion, we collected water samples from the middle of the water column with a horizontal 1-L Van Dorn bottle. Water samples for nutrient analyses were collected in clean, acid-washed Nalgene bottles and stored at 5°C until our return to the laboratory. In the laboratory, soluble reactive phosphorus (SRP) samples were filtered immediately through a 0.45-μm GF/C filter and kept frozen until analysis. Whole-water samples for total phosphorus (TP) and total nitrate-nitrogen (NO_3-N) were also frozen until analysis. Whole-water samples for total ammonia-nitrogen (NH_4-N) were acidified to pH 2.0 and stored at 5°C until analysis. All samples were processed usually within 6 h of collection, and all analyses were conducted within 3 months of collection.

Prior to analysis, a standard curve for phosphorus was generated from standards of known values. TP and SRP samples were analyzed in triplicate according to Murphy and Riley (1962), with TP samples digested according to standard methods (American Public Health Association 1992) prior to phosphorus determination. All absorbance readings were read with a Milton Roy Spec 301 spectrophotometer. We followed EPA-approved Hach methods and used Hach reagents (Hach Company 1989) to measure NH_4-N and NO_3-N concentrations in duplicate; absorbances were read with a Hach DR2000 spectrophotometer and converted to concentrations using regularly calibrated standard curves. The accuracy of all nutrient analyses were within 10% of actual concentrations.

Whole-water samples for chlorophyll *a* (CHL) analysis were collected in clean, opaque Nalgene bottles and stored at 5°C until our return to the laboratory where they were immediately filtered through 0.45-μm GF/C filters. Filters were folded, wrapped in aluminum foil, and kept frozen for up to 3 months before they were analyzed. CHL samples were extracted in 90% reagent-grade acetone (in the freezer), acidified, and analyzed for CHL pre- and post-acidification with a Spec 301 spectrophotometer. Equation 1 of Speziale et al. (1984) was used to calculate CHL concentrations, and all values reported here have been corrected for phaeopigments.

Samples for total suspended solids (TSS) were first filtered through preweighed GF/C filters and then frozen for less than 3 months. They were dried at 100°C for 1 h, placed in a desiccator for 1 h, and then weighed to the nearest 0.0001 g with an OHAUS AS 120 balance. Filters were subsequently combusted at 550°C for 20 min and desiccated for 1 h to determine ash-free weights or total inorganic suspended solids (TISS). Total organic suspended solids (TOSS) was calculated as the difference between TSS and TISS.

Phosphorus and inorganic content of sediment

Sediment samples were collected from the top 5 cm of sediment with either an Eckman grab sampler or a 5-cm-diameter Plexiglas tube equipped with a plunger. Sediment samples were dried at 100°C, ground to achieve homogeneity, and fired at 550°C in a muffle furnace to calculate the amount of inorganic material contained within the sediment. The TP in the sediment was analyzed according to Andersen (1976). Between 0.15 and 0.2 g of this material was weighed and boiled in 25 mL of 1 M HCl for 15 min and the extract was diluted to 100 mL. After neutralization with 10 M NaOH, a 20-mL aliquot was analyzed for TP content of the sediment according to the protocol for phosphorus analyses (above).

Analyses of trace organics by immunoassay techniques

PAHs and Metolachlor are organic contaminants that are commonly present in disturbed environments. Their presence can be detected by conventional gas chromatography (GC) methods that are expensive and labour-intensive. Recently, immunoassay methods have been used as a screening tool, as they are more cost-effective and less time-consuming than GC methods. Immunoassay techniques and conventional GC/mass spectrometry methods have been compared in the literature and were found to yield comparable data for both PAHs and other herbicides in laboratory and field samples (Zaruk et al. 1995). Direct comparisons between GC and immunoassay results for a subset of these wetlands show that samples measured by immunoassays tend to overestimate contaminant concentrations provided by GC methods; however, since this error is consistent throughout the range of concentrations tested, valid comparisons can be made among data obtained by the same technique (Crosbie 1997). Since immunoassay results have compared favourably with those of traditional GC methods, we chose to use the less expensive method in this study with 4% of the samples analyzed by GC methods as a confirmatory measure.

PAH

Samples were collected from 10 of the 22 wetlands, sampled in 1995, to determine the PAH content in sediment. Analyses were performed on these select wetlands due to budget and time constraints. The top 5 cm of sediment was collected using a Plexiglas tube (5 cm inside diameter) equipped with a plunger. These samples were kept at 5°C until analyses and placed in a dehydrator for 24 h at 35°C to remove excess water. Subsamples were stored in a desiccator until analysis by immunoassay. We used the Millipore EnviroGard PAH in Soil Plate Kit (Millipore Intertech) to extract and analyze for total PAHs (range from 1.0 to 10 $ng \cdot g^{-1}$ in soils). The enzyme-linked immunoassay technique utilizes polyclonal antibodies coated to the walls of microwells in test kits and an enzyme conjugate that competes for binding sites with the contaminant of interest in the collected sample. The amount of the target compounds in the sample is thus inversely proportional to the amount of colour produced by the reaction. The coloured products were measured spectrophotometrically on a microplate reader fitted with a 450-nm filter. Duplicates, spikes, and blanks were run on the kit to measure variation, recovery, and contamination, as recommended by the manufacturer.

Metolachlor concentration in water

Surface samples were collected in washed and solvent rinsed glassware with teflon- or foil-lined lids from 14 different marshes during June and July in 1996. They were stored in the dark in amber bottles at 5°C until analysis by immunoassay. We used Quantix Metolachlor 1.0 immunoassay kits (Idetek Inc., Sunnyvale, Calif.) to measure the Metolachlor concentrations and followed instructions recommended by the manufacturer. Samples were measured spectrophotometrically at 650 nm, and concentrations were determined from a calibrated standard curve.

Land use delineation

Relevant maps from the 1 : 50 000 National Topographic Series were used to delineate the wetland watersheds, except for those in the Niagara region, where 1 : 25 000 land use maps that included spot elevations were used to delineate the watersheds. The watershed was delineated from contour lines of the highest elevation and (or) equidistant points between two tributaries of adjacent watersheds. Forested, agricultural, and urban land within the watersheds was determined from Ontario's Agricultural Land Use Series maps (majority being current to the mid-1980s) and supplemented by information from aerial surveys taken during 1996 in cases where land use maps were incomplete (i.e., Cootes Paradise Marsh, Christie Lake Marsh, Joe's Lake Marsh, Stump Lake Marsh, Hay Bay Marsh, Sawguin Marsh, Tay River Marsh, and Presqu'ile Marsh). Such aerial surveys confirmed that the land use data available were accurate for the purposes of this comparative study. Triplicate planimetry measurements were used to quantify the size of the wetland, watershed, and each of the three land use types. Where applicable, presence of buffer strips surrounding wetlands was also confirmed from aerial surveys and land use maps.

Statistical analysis

All statistical analyses were performed using SAS Jmp software for the Macintosh (SAS Institute Inc., Cary, N.C.). Statistical tests included paired *t* tests, correlation analysis, PCA, and general linear regression analysis. To achieve normality, data were either log transformed or arcsine transformed to normalize the variances. We used standardized residuals to identify outliers in the linear regression analyses, since they can have a strong influence on estimates of the fitted parameter. This exercise allowed us to eliminate Big Creek and Holiday Marsh (wetlands 2 and 8, respectively, in Table 1) from the regression of PC1 scores against percent agricultural land.

Results

The size of wetlands varied, ranging from 0.10 to over

20 km^2, with corresponding watersheds ranging from very small (5 km^2) to extremely large (1060 km^2) (Table 1). In addition, these sites represented both riverine and lacustrine wetlands. Water flow through the wetland at the sampling site was generally very slow (or not observable), except for Centreville Creek Marsh where the observed water flow was greater than 2 m·s^{-1}. This riverine wetland was also unique in that it was heavily vegetated with emergent plants only, with no submergent plants or open-water areas that were found in all of the other 21 wetlands. Slightly more than half of these wetlands are known to be stressed by carp disturbance, although the relative importance of this stressor was not evaluated quantitatively (e.g., biomass, mean density). The fact that there are different types of wetlands covering a large range of ecological conditions should strengthen the utility of this study and make our conclusions more generally applicable.

PCA

We first determined if data collected from the two sampling trips in each year could be combined. Since we found no significant differences between sampling dates for any of the 14 variables (paired *t* tests; $P > 0.05$), we merged data for the two trips and calculated averages for each variable (Table 2). PCA was subsequently used to determine the best explanatory variables in the water quality and sediment quality data set. The first three axes together explained 82% of the overall variation in the data set (Table 3). PC1, which accounted for over half of the variation, was positively and highly correlated with environmental variables that were essentially measures of particulate content in the water column (Fig. 2). These included water turbidity and concentrations of TSS, TP, and planktonic CHL. TP and total NH_4-N were also significantly correlated with PC1 scores, even though it is not a reflection of the suspended particles in the water. PC2 explained 18% of the variance and was positively correlated with the inorganic content of the sediment but was negatively correlated with its TP content. Wetlands that had high positive PC2 scores tended to have higher pH and specific conductance. PC3, which only accounted for 13% of the variance, was negatively correlated with SRP concentration but positively correlated with total NO_3-N concentration in the water column.

Taken together, the PCA scores reflected the degree of water and sediment quality impairment in the wetland: sites with high positive scores for all three components were turbid systems, enriched with phosphorus and nitrogen, with correspondingly high CHL in the water column, and lacking phosphorus and organic content in the sediment. By contrast, wetlands associated with low and negative scores were characterized by clear water, low nutrient concentrations, and correspondingly low CHL, with sediments rich in both organic content and phosphorus (Fig. 2).

Submergent vegetation

In a concurrent study, Lougheed et al. (1998) surveyed the number of submergent plant species in each of these wetlands. We found an inverse relationship between PC1 scores and the number of species of submersed aquatic vegetation (Fig. 3). Marshes with low water turbidity and nutrients supported a more species-rich community of submergent plants. Centreville Creek Marsh (wetland 3, Table 1), which did not support submergent plants, was excluded from this analysis. By comparison, the species richness of submergent plants did not vary with PC2 scores; hence, neither sediment fertility nor organic content appears to be a determinant of submergent diversity in these wetlands.

Land use and water and sediment quality

To test the hypothesis that the distribution of different land use types in the watershed affected the trophic quality of wetlands, we regressed scores of the first three principal components against the proportion of agricultural, forested, and urban land. We found three significant relationships: (*i*) PC1 scores varied inversely with the proportion of forested land (PC1 score = –4.415 arcsin FOR + 1.325; r^2 = 0.40, $P < 0.001$), (*ii*) PC1 scores varied directly with the proportion of agricultural land (PC1 score = 2.15 arcsin AGR – 2.43; r^2 = 0.43, P = 0.0017; Fig. 4*a*), and (*iii*) PC2 scores were negatively related to the proportion of forested land (PC2 score = –2.888 arcsin FOR + 0.471; r^2 = 0.42, $P < 0.001$). Accordingly, wetlands whose watersheds were dominated by agricultural areas tended to be turbid and nutrient rich, while those within forested landscapes were clear and nutrient poor. Unfortunately, owing to the small number of urban wetlands in this study, the absence of a significant relationship between principal components scores and percentage urban land is inconclusive.

Peterjohn and Correll (1984) demonstrated that forested buffer strips can reduce the nitrogen and phosphorus loads by up to 70% if they are located near the edge of a riparian system. We accounted for the ameliorating effects of such buffer strips within agricultural areas by applying a conservative reduction factor of 0.25 to sites that had forested buffer zones along one or more of their tributaries (Centreville Creek Marsh (wetland 3), Hay Bay Marsh (wetland 7), Sawguin Marsh (wetland 14), and Waterford Marsh (wetland 22)) and a factor of 0.50 to sites that had a large proportion of forested land (Presqu’ile Marsh (wetland 13) and Tay River Marsh (wetland 19)). When we compared the unmodified index of agricultural land with the modified index, we found that regression of PC1 scores against the latter improved significantly (r^2 increased from 0.43 to 0.68, respectively; Fig. 4). This comparison suggests that the existence of forested areas in agriculturally dominant watersheds may lead to measurable differences in the water quality of downstream wetlands.

Common carp disturbance

We had the opportunity to broadly evaluate the effect of carp disturbance on the trophic quality of wetlands as indicated by PC1 scores. Study sites that had been invaded by carp were well represented (solid squares in Fig. 4). Many of the wetlands known to be affected by carp disturbance were in fact associated with medium to high PC1 scores; however, there were at least two marshes (Hay Bay (wetland 7) and Sawguin (wetland 14)) that also had relatively low scores. Therefore, the presence of carp in the wetland had an inconsistent effect on water quality and did not confound the relationship between water quality and land use.

Table 2. Summary of key water quality and sediment quality data for wetlands identified by PCA.

Wetland	Turbidity (NTU)	Light extinction coefficient	TSS ($mg \cdot L^{-1}$)	TISS ($mg \cdot L^{-1}$)	TOSS ($mg \cdot L^{-1}$)	TP ($\mu g \cdot L^{-1}$)	SRP ($\mu g \cdot L^{-1}$)	CHL ($\mu g \cdot L^{-1}$)	TAN ($mg \cdot L^{-1}$)	TNN ($mg \cdot L^{-1}$)	Specific conductance ($\mu S \cdot cm^{-1}$)	pH	$Inorg_{sed}$ ($mg \cdot g^{-1}$)	TP_{sed} ($mg \cdot g^{-1}$)
1	80.7	7.60	124	112	12	345	20	24	0.195	0.450	741	7.5	0.94	0.72
2	4.1	2.47	11.2	5.2	6	76	9.2	7.3	0.061	1.760	554	7.6	0.81	0.60
3	3.8	2.45	4.6	1.5	3.1	66	13	2.6	0.083	0.050	599	7.6	0.64	1.41
4	2.6	2.37	8.8	4.2	4.6	31	2.8	6.3	0.026	0.080	156	6.7	0.59	1.47
5	59.0	6.14	51	37	14	191	24	24	0.355	0.262	715	7.3	0.82	0.82
6	1.8	2.31	3.8	1.0	2.8	81	4.4	2	0.355	0.050	39	6.0	0.44	1.92
7	3.0	2.40	3.7	1.3	2.4	62	22	7.7	0.066	0.046	552	6.7	0.68	0.57
8	265	19.94	248	160	88	770	24	247	0.296	1.419	601	7.5	0.80	0.66
9	67.8	6.74	57	37	20	214	11	29	0.221	0.140	735	7.5	0.95	0.55
10	1.6	2.30	5.4	2.7	2.7	23	2	6.0	0.082	0	215	6.9	0.41	2.00
11	44.0	5.14	42	30	12	360	88	61	0.094	0.221	618	8.1	0.93	0.69
12	44.1	5.15	75	50	25	339	116	45	0.160	0.466	452	7.1	0.92	0.75
13	1.5	2.29	4.8	0.7	4.1	54	1.9	2.4	0.024	0.030	336	7.1	0.79	0.99
14	4.1	2.47	6.5	1.8	4.7	76	15.4	15	0.115	0.174	390	6.9	0.28	1.99
15	14.8	3.19	24	15	9	407	107	67	0.035	0.433	865	7.8	0.71	1.01
16	6.1	2.6	9.5	4.2	5.3	69	1.5	6.1	0.063	0.335	49	6.1	0.81	0.41
17	2.1	2.3	9.3	6.2	3.1	42	3.6	5.7	0.060	0.004	190	6.5	0.24	2.17
18	13.4	3.09	42	30	12	80	2.6	18	0.067	1.733	529	8.2	0.94	0.39
19	1.4	2.29	6.2	3.2	3.0	50	6.6	4.6	0.049	0.065	195	7.2	0.26	2.04
20	2.8	2.38	7.4	1.0	6.4	139	2.9	22	0.040	0.294	100	6.3	0.36	1.03
21	11.7	2.98	20.8	14.0	7.8	61	12.4	21	0.041	0.048	479	8.3	0.82	0.88
22	23.0	3.74	52	29	23	129	7.3	25	0.265	0.450	465	7.8	0.75	0.85

Note: Values are averages of two sampling trips. TSS, total suspended solids; TISS, total inorganic suspended solids; TOSS, total organic suspended solids; TP, total phosphorus; SRP, soluble reactive phosphorus; CHL, chlorophyll *a*; TAN, total NH_4-N; TNN, total NO_3-N; $Inorg_{sed}$, proportion inorganic content in sediment; TP_{sed}, total phosphorus in sediment. Wetland numbers are given in Table 1.

Table 3. Correlation coefficients between principal components scores and environmental variables.

	Variance explained (%)	Environmental variable	*r*	*P*
PC1	51	Water turbidity	0.94	0.0001
		TSS	0.94	0.0001
		TOSS	0.93	0.0001
		TISS	0.91	0.0001
		TP	0.91	0.0001
		CHL	0.89	0.0001
		TAN	0.74	0.0001
PC2	18	$Inorg_{sed}$	0.84	0.0001
		TP_{sed}	–0.82	0.0001
		pH	0.68	0.0001
		Conductance	0.66	0.0001
PC3	13	SRP	–0.61	0.004
		TNN	0.60	0.004

Fig. 3. Number of submersed aquatic plant taxa plotted against PC1 scores for each of the wetland. Centreville Creek Marsh (wetland 3) has been excluded from this analysis (see text).

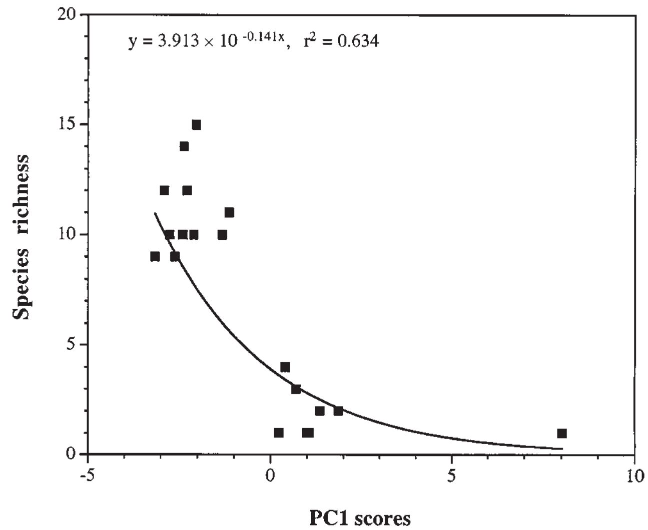
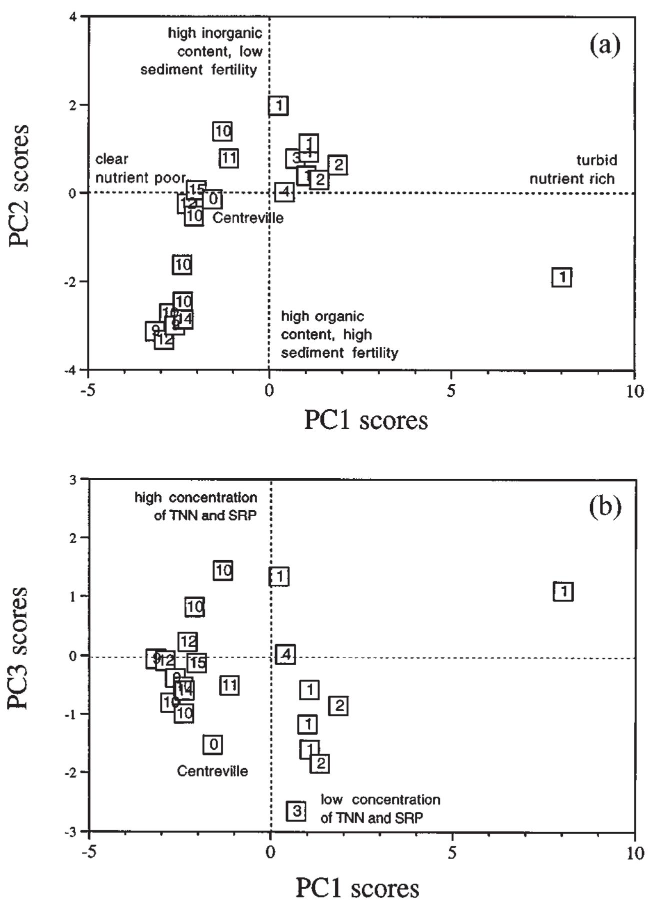


Fig. 2. Plots of (*a*) PC2 scores and (*b*) PC3 scores versus PC1 scores. Numbers inside the squares are the number of submersed aquatic plant taxa encountered during our surveys. The data point corresponding to Centreville Creek Marsh (wetland 3) is labelled in each panel.



Land use surrogates

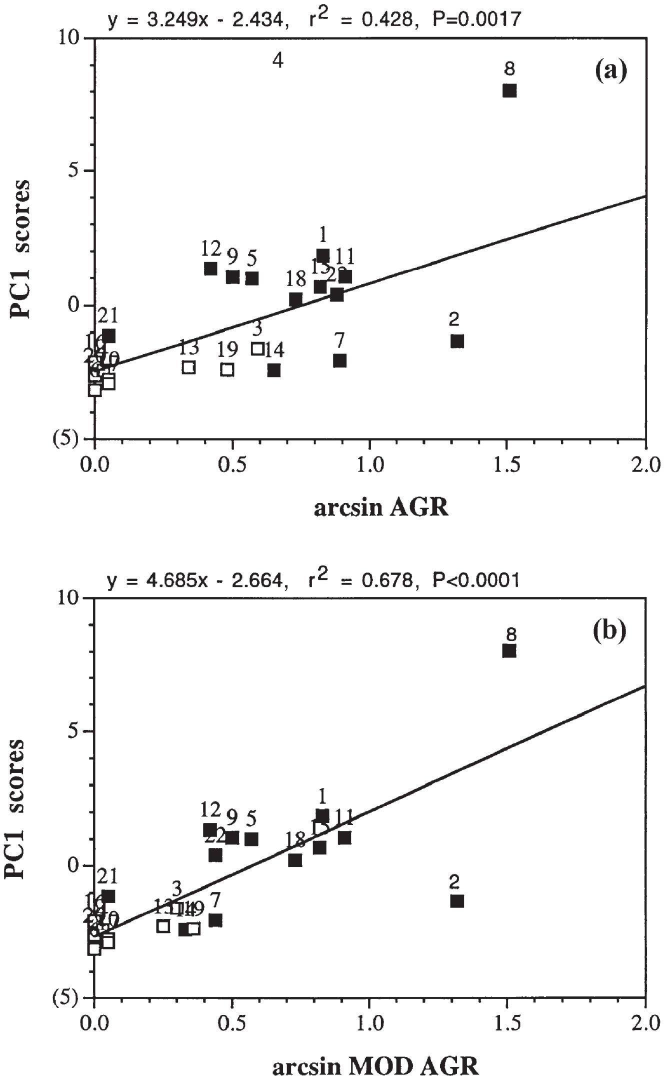
We had some difficulty obtaining current land use information for all sites in this study. This, together with the large amount of time and labour required to calculate percent land use (especially in the absence of digitized land use information), seriously limited our ability to study wetlands on a basin-wide scale. Therefore, we explored the potential for using easily measured field parameters specific to agricultural or urban activities as land use surrogates for future investigations. The parameters that we chose to test were the level of total PAHs in wetland sediment and the concentration of Metolachlor in creek and river water feeding into the wetlands. We reasoned that PAHs are common pollutants associated with roadway runoff (Maltby et al. 1995) and should therefore be an effective indicator of percent urbanized land in watersheds. Metolachlor concentration was chosen to indicate percent agricultural land because it is a commonly used herbicide in Ontario and has been found consistently in agricultural runoff (Gaynor et al. 1995).

The total level of PAHs in the sediment significantly explained close to 70% of the variation in urbanized land, whereas $\log_{10}$-transformed Metolachlor concentration explained 80% of the variation in percent agricultural land (Fig. 5). Therefore, these measures of organic contaminants in sediment and water demonstrate their potential usefulness as land use surrogates. When we regressed PC1 against log-transformed Metolachlor concentrations, we found a highly significant relationship (r^2 = 0.77, P = 0.0001, n = 16) that confirmed the utility of this variable as a surrogate of percent agricultural land. Although these results are promising, the sample size in this study was very small, and more data need to be collected in the future to determine the general applicability of these results to other wetlands.

Discussion

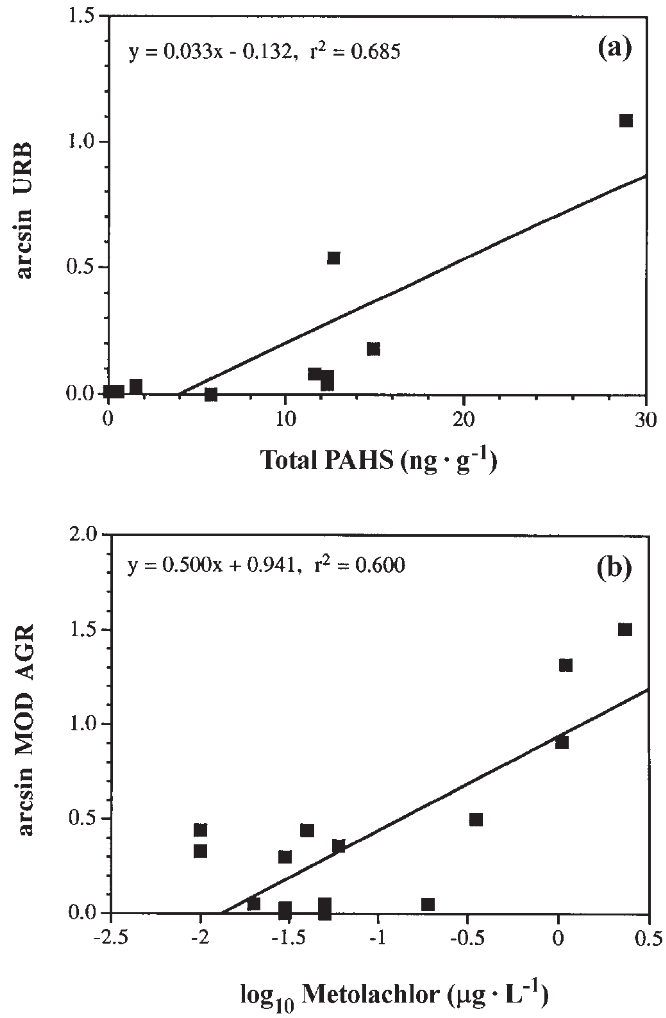
There are few published studies that characterize the ecological relationships of Great Lakes coastal wetlands on a

Fig. 4. PC1 scores versus (*a*) arcsine-transformed percent agricultural land (AGR) in the watershed and (*b*) arcsine-transformed percent agricultural land (MOD AGR) after applying a reduction factor of 0.5 for wetlands that had large forested buffer zones (Presqu'ile Marsh (wetland 13) and Tay River Marsh (wetland 19)) and a factor of 0.25 to sites that only had forested strips along one or more of their tributaries (Centreville Creek Marsh (wetland 3), Hay Bay Marsh (wetland 7), Sawguin Marsh (wetland 14), and Waterford Marsh (wetland 22)). Least squares regression analysis did not include the outliers Big Creek (wetland 2) and Holiday Marsh (wetland 8). Solid and open squares correspond to wetlands that are accessible and inaccessible to common carp, respectively.



basin-wide scale. A notable exception is Smith et al. (1991)'s study of 160 Canadian coastal wetlands located mainly in the lower lakes and the St. Lawrence River. They showed that patterns of variation in wetland and site types, soil composition, dissolved solids concentration, vegetation complexity, and the presence of rare flora and fauna could be attributed to the geology of different lake basins. Findlay and Houlahan (1997) found that the species richness of some mammal and herpetile taxa was directly related to the proportion of forested land within a 2-km radius of wetlands

Fig. 5. (*a*) Arcsine-transformed percentage urbanized land (URB) versus total PAHs in wetland sediment and (*b*) arcsine-transformed percentage agricultural land (MOD AGR) modified to account for the ameliorating effects of buffer strips (see legend to Fig. 4) versus log-transformed Metolachlor concentration in the water.



and that removal of forested land threatened the biodiversity of these wetlands. Our study shows that the biodiversity of aquatic plants is similarly affected by altered land use. Apart from these few studies, large-scale investigations that examine the impact of land use changes on Great Lakes wetlands are lacking, even though such indirect impacts may cause greater damage to wetlands than can direct impacts such as drainage and infilling (Keddy 1983).

Although we found no comparable studies from other geographic regions that focussed on freshwater coastal wetlands, there are a few published studies that examined the distribution of aquatic macrophytes along ecological or disturbance gradients in inland wetlands and lakes. Moore et al. (1989) showed that infertile wetlands in north-central Ontario had higher species richness and many more rare species than fertile wetlands. Our observations in these wetlands echo their opinion that infertile wetlands are more desirable for conservation. Current wetland evaluation systems do not gain an adequate measure of fertility, which appears to be essential in determining diversity. Toivonen and Huttunen (1995), who sampled 57 small lakes in southern Finland, also found that the distribution of submerged aquatic vegetation was principally determined by trophic state and water transparency. However, the highest species diversity of submergents was found in clear, mesoeutrophic lakes, which were intermediate in trophic state between oligotrophic, brown-water lakes (humic stained) and hypereutrophic lakes. Expansion of our data set to include bogs (humic stained) may have yielded a similar trend.

By contrast, Morgan and Philipp (1986), who worked on streams in the New Jersey Pine Barrens, found a slight increase in the number of aquatic macrophytes in nitrogen-enriched streams following four decades of agricultural and urban development. In addition, there was evidence of species replacement: the native flora that were tolerant of nitrogen-limited environments had become outcompeted by species normally found outside the Barrens that tolerated phosphorus-limited conditions. Thus, it is important to consider both the species richness and the species composition of wetland ecosystems when evaluating the possible impact of land use changes. The restricted sample size of wetlands in this study did not permit us to analyze changes in species composition with trophic quality, but this topic is being investigated elsewhere with a larger data set (V.L. Lougheed et al., unpublished data).

In our study, concentrations of phosphorus, nitrogen, and inorganic suspended solids in wetlands increased predictably as agriculture became the dominant land use in their respective watersheds. These results are entirely consistent with previous studies on stream ecosystems, where 50–80% of the variation in nutrient and suspended particulate concentrations was related to the amount of agricultural land in the watershed (Osborne and Wiley 1988; Johnson et al. 1997). Holiday Marsh (wetland 8 in Fig. 4) exemplified this damage from agricultural runoff to an extreme and is probably why it was revealed as an outlier in the analysis of residuals. It had a mean water turbidity of 265 NTU, compared with 80 NTU for Big Creek, the wetland with the next highest water turbidity. The low relief in the watershed is quite unique to the region, and consequently, the entire watershed is farmed (Fig. 1). The type of crops grown include corn, soybean, and winter wheat, which usually requires tilling and extensive use of fertilizer, the cumulative negative effects of which are amply evident in the extremely degraded state of Holiday Marsh today. Therefore, Holiday Marsh should only be considered a statistical outlier in this study, but should be viewed as representing the extreme in water turbidity for all coastal marshes. In fact, if Holiday Marsh were included in the regression analyses, the r^2 values would increase substantially from 0.43 to 0.61 in Fig. 4*a* and from 0.68 to 0.81 in Fig. 4*b*, without significant alteration of the slopes (ANCOVA; $P > 0.5$ in both cases).

By comparison, Big Creek Marsh (wetland 2 in Fig. 4) is probably truly anomalous, since it had relatively good water and sediment quality, even though it is located in a heavily agricultural watershed and is also subject to carp disturbance (Table 1; Fig. 1). Inclusion of Big Creek to the regression analyses lowered the r^2 values from 0.43 to 0.30 in Fig. 4*a* and from 0.68 to 0.42 in Fig. 4*b*. One possible explanation for this departure is that the substrate in the Big Creek watershed is primarily sandy, whereas soils in other heavily farmed wetlands are dominated by clay and silt. The larger and heavier sand particles are known to settle out of the water column more rapidly than clay and silt particles, and this may explain why water turbidity in Big Creek Marsh was consistently low. Clay particles also tend to be ladened with adsorbed phosphorus, whereas sand particles do not (Golterman 1995), and this difference may account for the low phosphorus concentrations in Big Creek Marsh compared with other wetlands. In contrast, accompanying nitrate levels were extremely high in Big Creek, likely due to the sod farms in the area that use fertilizers that favour a high nitrogen ratio. Since nitrogen is generally the limiting factor for macrophyte growth (Barko et al. 1991), the combination of clear water and excess nitrate may have promoted luxuriant growth of submergent plants, despite the dominance of agricultural lands in the watershed.

Urban land use was not a significant predictor of water or sediment quality of wetlands in this study. However, urban wetlands were poorly represented in our data set, and ongoing research in our laboratory confirms that relatively few urban wetlands currently exist in southern Ontario because marshes in heavily settled areas of the province have generally been eliminated due to dredging or infilling. Thus, further attempts to increase representation of urban wetlands may not be successful.

Previous studies have shown that submergent plant cover (Robel 1962) and water clarity (Lougheed et al. 1998) can be related to the biomass of carp in wetlands. Such results have been used to justify carp exclusion/reduction as a means to restore Great Lakes coastal wetlands (e.g., Cootes Paradise Marsh and Oshawa Second Marsh). Results of this study, however, suggest that a greater emphasis on wetland conservation and restoration in the Great Lakes should be focussed on controlling non-point-source impacts because there was no consistent effect of carp resuspension on either water turbidity or the species richness of submergent vegetation. The chief determinant of water and sediment quality was the proportion of agricultural land (modified index), which accounted for about 70% of the overall variation in trophic quality. The remaining variation may be explained by such factors as carp biomass, wind and wave resuspension, and substrate type. We recommend that future studies be conducted that consider these variables simultaneously for a large number of wetlands so that the relative impacts of each can be quantified.

We have suggested that water quality impairment associated with extensive farming can be ameliorated through provision of forested buffer strips. Previous work by Peterjohn and Correll (1984) also demonstrated that riparian vegetation and forests can filter out a great deal of phosphorus, nitrogen, and suspended material in surface runoff and thus reduce loading to downstream sites. Differences in subsurface contribution of dissolved phosphorus can also be affected by the tillage practice used (Gaynor and Findlay 1995). Other best-management practices such as those oriented towards minimizing the impact of confined animal operations will also reduce the sediment and nutrient load from agricultural watersheds (Edwards et al. 1996).

The effects of buffer strips were not quantitatively evaluated in this study because it was not within the scope of this paper to fully explore their effectiveness. Nevertheless, by applying a conservative factor to our data based on benefits observed elsewhere, we hope that our results will encourage other investigators to conduct field studies to link the management value of buffer strips to the water quality of downstream wetlands. We need to quantify the beneficial effects of different types of best-management practice throughout the Great Lakes basin to provide a scientific basis for recommending the appropriate strategy to retard further degradation of wetlands in heavily farmed catchments.

Although we grouped together agriculture of all types in this study, we acknowledge that the impacts of crop farming versus cattle farming are very different (Beaulac and Reckhow 1982). Many types of farming subject the soil to long periods of exposure with correspondingly high rates of erosion. Different application rates of various types of fertilizers and herbicides will also lead to differences in pollutant loads to wetlands via runoff. Combining the power of recent geographic information systems technology with empirically derived export coefficients for the Great Lakes basin should improve the predictive power of land use models (e.g., Mattikalli and Richards 1996) and improve our ability to predict non-point-source impacts on coastal wetlands.

There is good reason why we focussed on the submergent plant community as a primary indicator of wetland quality. Submersed macrophytes spend all or most of their life cycle inside the water column and, as such, compete with algae for light. Whereas nutrient uptake by algae is through the water, macrophytes primarily utilize nutrients from the sediment (Carignan and Kalff 1980). Therefore, nutrients in the water column will disproportionately benefit the algae and allow them to shade out the macrophytes, especially in wetlands where eutrophic conditions affect water clarity. This situation is responsible for the well-documented alternate states of shallow lakes throughout the world: turbid, algae-dominated systems devoid of macrophytes or clear, macrophyte-dominated systems (e.g., Moss 1990; Chow-Fraser 1998).

Decrease in species richness with increasing water turbidity can be explained in terms of Grime's (1979) theoretical framework, in which he proposed three main types of adaptive strategies for vascular plants depending on the degree of disturbance and stress in their habitat. Wetlands with low PC1 scores are associated with seasonal rather than continuous disturbance (i.e., water depths are reduced only in late summer) and relatively low stress levels (i.e., low water turbidity and high sediment phosphorus content). According to Grime (1979), such an environment should favour many competitive species. By comparison, wetlands with high PC1 scores are associated with high continuous disturbance (e.g., uprooting activities of carp) as well as high stress levels (low light penetration because of high water turbidity). In such habitats, there is no viable strategy, and consequently, submergent growth is almost negligible (e.g., Big Creek Marsh, Cootes Paradise Marsh prior to carp exclusion, Lougheed et al. 1998). Once disturbance is reduced (e.g., Cootes Paradise Marsh after carp exclusion, P. Chow-Fraser, unpublished data), however, a few stress-tolerant species capable of growing in low light levels may become established (e.g., *Potamogeton pectinatus*, Lehmann et al. 1997; *Ceratophyllum demersum*, *Elodea canadensis*, and *Myriophyllum verticillatum*, Toivonen and Huttunen 1995).

We have shown that the trophic quality of 22 wetlands (located primarily in the catchment areas of Lakes Ontario and Erie) was significantly affected by the percent agricultural land in the respective watersheds, independent of in-marsh stresses such as carp disturbance. The decline in water quality from marshes with good water transparency and low nutrient concentrations to turbid systems enriched with phosphorus and nitrogen was accompanied by a predictable decrease in the species richness of submergent aquatic plants. Our study thus provides direct links among land use impacts, water quality impairment, and wetland plant diversity. The apparent compensatory effect of buffer strips on the water quality of downstream wetlands should be investigated more rigorously, since they may be very important in maintaining the diversity of submergent plants in Great Lakes coastal marshes.

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