J. Great Lakes Res. 25(4):799–813
Internat. Assoc. Great Lakes Res., 1999

Seasonal, Interannual, and Spatial Variability in the Concentrations of Total Suspended Solids in a Degraded Coastal Wetland of Lake Ontario

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ABSTRACT. *A 4-year (1993 through 1996) monitoring program examined the distribution of total suspended solids (TSS) in Cootes Paradise Marsh, a shallow (mean depth of 70 cm), degraded, drowned-rivermouth marsh of Lake Ontario. Monthly meteorological and hydrographical data from 1986 through 1996 revealed a hydrologically dynamic system that exhibited large seasonal and interannual variation with respect to precipitation amount, discharge volume, and water levels; the prevailing winds were shown to be oriented along the length of the marsh. Interannual variation in TSS concentrations was inversely related to mean seasonal water levels that fluctuated 45 cm over the 11 years. In a stepwise regression analysis, planktonic chlorophyll-*a *concentration only explained 2% of the variation in TSS, while inorganic and non-algal organic solids explained 70% and 18%, respectively. Mean seasonal water turbidity increased significantly with mean seasonal wind speed at 17 sampling stations during 1993 and 1994. Runoff from a summer rainstorm more than doubled water turbidities at the mouth of all three creeks over the first 36 hours. In enclosure experiments, water turbidity increased proportionately with biomass of benthivorous fish (especially common carp,* Cyprinus carpio*). When wind and carp disturbance were compared simultaneously in the field, wind speed accounted for 41% of the variation in turbidity while presence of carp explained an additional 21%. The overall temporal and spatial distribution of TSS in the marsh reflected changes in water level, wind activities, onset of rain events, and fish disturbance that acted in concert to keep Cootes Paradise Marsh extremely turbid throughout the summer.*

INDEX WORDS: *Turbidity, coastal marsh, Lake Ontario, total suspended solids.*

INTRODUCTION

Spatial and temporal variability in the distribution of suspended sediments is a prominent feature of shallow, productive lakes throughout the world (Limón *et al.* 1989, Bengtsson and Hellström 1992, Carrick *et al.* 1994, Hamilton and Mitchell 1996). The factors that contribute to this variability include water circulation patterns, wind-induced or fish-induced resuspension, and water diversion. Although the high concentration of suspended solids generally results in great water turbidity, the composition of the suspended material differ among lakes, from mostly inorganic particles in Lake Okeechobee, Florida (Havens 1995), to clay-organic matter-bacterial aggregates in Lake Chapala, Mexico (Lind and Dávalos-Lind 1991), to recently sedimented phytoplankton in seven New Zealand lakes (Hamilton and Mitchell 1997). Regardless of the source of turbidity or the distribution patterns, however, water transparency in these lakes is always poor and the growth of aquatic macrophytes is limited, even though these lakes are extremely shallow (Scheffer *et al.* 1993).

Degraded marshes of the Laurentian Great Lakes are very similar to shallow lakes in that they have become hypereutrophic and turbid because of anthropogenic pollution (Havens 1991, Chow-Fraser *et al.* 1998). As in shallow lakes, degraded wetlands have also lost most of their aquatic macrophyte communities, and extreme turbidity has similarly been blamed for their disappearance (Crowder and Painter 1991, Lougheed *et al.* 1998). Though none of these studies investigated temporal and spatial

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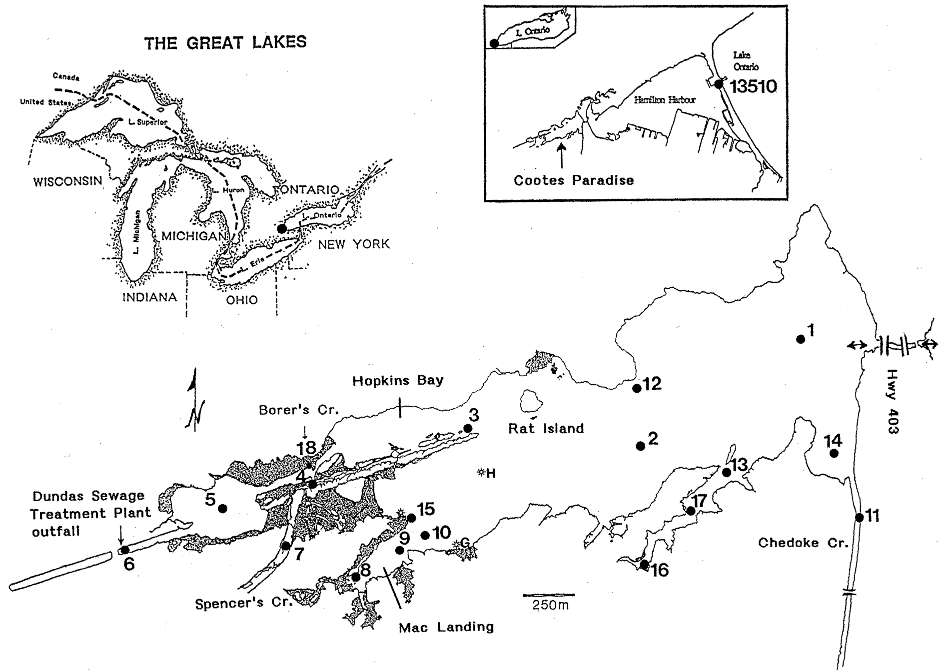

FIG. 1. Sketch of Cootes Paradise Marsh showing location of the sampling stations in the long-term monitoring program and various field studies. The inset shows the location of the marsh relative to Hamilton Harbour and the Great Lakes.

variability, such variability is expected since wetlands are subject to the same environmental factors that cause variable distribution patterns in shallow lakes. Since re-establishment of a diverse aquatic macrophyte community hinges on reduced water turbidity (Skubinna *et al.* 1995, Whillans 1996, Lougheed *et al.* 1998), the factors that contribute to patchiness in turbidity need to be understood so that effective remedial actions can be developed to restore degraded marshes.

The primary purpose of this paper is to document the seasonal and spatial variability of water turbidity in a degraded coastal Great Lakes wetland, Cootes Paradise Marsh, and to determine the factors that contribute to this variation. The factors that are considered in this study are: wind-induced resuspension, fish-induced resuspension, tributary loadings from heavy rainstorms, contribution from algal growth, and interannual changes in water level. Data from a 4-year (1993 through 1996) monitoring program are used, along with aerial photos, to document the spatial and temporal variation in total suspended solids in the marsh. The effects of each of the above factors on water turbidity are then examined through a series of field surveys and enclosure experiments conducted from 1992 to 1996.

METHODS AND MATERIALS

Description of Study Site

Cootes Paradise Marsh is located at the extreme west end of Hamilton Harbour on Lake Ontario (Fig. 1 inset). The surface area and volume of the marsh vary according to water levels; monthly mean water levels have fluctuated interannually up to 1 m over the past two decades (Chow-Fraser *et al.* 1998). According to the Stage 1 Report of the Remedial Action Plan for Hamilton Harbour (Canada-Ontario Joint Document 1992), the mean depth of the marsh is 0.7 m, the maximum surface area is 2.50 km^2, and the calculated maximum capacity is 3.57×10^6 m^3. It has a total length of 4 km and a maximum width of 1 km (Painter *et al.* 1989; Fig. 1). Three main tributaries drain into Cootes Paradise, the largest of which is Spencer Creek (station 7 in Fig. 1) which enters the marsh from the southwest, draining 79% of the 290.9 km^2 watershed. The other two smaller creeks are Borer's (station 18 in Fig. 1) and Chedoke Creek (station 11 in Fig. 1), which drain 6.9 and 9.4% respectively of the remaining watershed (T. Horvat, Hamilton Region Conservation Authority, pers. comm.).

DISCHARGE, PRECIPITATION, WATER LEVEL AND WIND DATA

T. Horvat (Hamilton Region Conservation Authority) provided discharge data (1986 through 1996) for Spencer's Creek, which had been collected at the Water Survey of Canada Station 02HB010 at Dundas Crossing. Since there were no measured discharge volumes for the other two creeks, hydrologic inputs from these were estimated by proportionate approximation to the Spencer Creek drainage basin. The total hydrologic load calculated in this fashion is conservative because it does not consider inputs from combined sewer overflows draining into Cootes Paradise, that are distributed unevenly among the three watersheds.

Daily precipitation (volume of snowfall and rainfall; 1986 through 1996) and wind data (wind speed (km/h) and direction; 1993 and 1994) were obtained from the Canadian Atmospheric Environment Services (AES) weather station located at the Royal Botanical Gardens (RBG) in Burlington, Ontario. The weather station is located 5 km northeast of Cootes Paradise Marsh. Water level data were estimated from hourly water levels recorded at Station 13150 located at the Burlington Ship Canal connecting Hamilton Harbour to Lake Ontario (see Fig. 1 inset) as outlined in Chow-Fraser *et al.* 1998.

Water-quality Monitoring Program

On a biweekly basis from May to September in 1993 to 1996, the marsh was sampled at different locations (stations 1 to 18 inclusive, Fig. 1) as part of a long-term monitoring program. Only five generic site types were sampled consistently through the 4 years and these included: 1) vegetated areas in marsh inlets ("vegetated," stations 8, 16, and 17); 2) open-water areas in marsh inlets ("open marsh," stations 9, 10, 13, and 14); 3) open water (station 1, 2, and 12); 4) West Pond, historic sewage lagoon (station 5); and 5) Spencer's Creek (station 7). Unfortunately, water levels were too low in 1995 to permit access into the shallow marsh inlets for proper sampling, and consequently only 3 years of data will be reported for "vegetated" and "open-marsh" stations.

Sampling trips were generally carried out between 10:00 and 15:00, under sunny or overcast conditions (not during rain events). During 1993 and 1994, wind speeds (km/h) were measured with an anemometer in a canoe and corresponding wind directions were recorded at the same time. On each sampling occasion during these 2 years, water samples were collected at mid-column (total depths ranging from 0.40 to 1.2 m) and water turbidity (formazin turbidity units; FTU) was measured in triplicate with a Hach 2100 P Turbidimeter. Secchi depth (cm) transparency was recorded with a 20-cm disk. A LiCor photometer equipped with spherical submersible sensor was used to obtain underwater PAR at 10-cm intervals from the surface to within 20-cm of the sediment surface. These data were later used to generate light extinction coefficients.

Water samples that were analyzed for chlorophyll *a* (CHL), total suspended solids (TSS), and total inorganic suspended solids (TISS) for each of the 4 years were also collected at mid-column at the established sampling stations (stations 1 to 17 in Fig. 1). All analytical protocols have been documented elsewhere (Lougheed *et al.* 1998). Even though the 1993, 1994, and 1996 sampling programs were conducted by McMaster University whereas the 1995 program was performed by RBG, the data should be directly comparable because both agencies followed similar sampling and analytical protocols. This was reinforced by the results of a direct comparison in 1996 which indicated no significant differences between datasets for stations 1 to 7. Total organic suspended solids (TOSS) was calculated as the difference between TISS and TSS. To make CHL data (wet weight) comparable to the TSS data (dry weight), the values were converted to dry weight based on a 1:100 ratio between CHL and dry weight of phytoplankton (Philips *et al.* 1995). TSS data from the biweekly monitoring program (May to August inclusive) of the RBG from 1986 to 1992 (Chow-Fraser *et al.* 1998) were added to the 4-year database (1993 to 1996) to create an expanded 11-y dataset to examine changes in TSS in relation to mean summer water levels.

Enclosure Experiments with Benthivorous Fish

In May 1995 Lougheed *et al.* (1998) conducted experiments using twelve 50-m^2 enclosures that were installed in a shallow (40 to 70 cm), non-vegetated area of Cootes Paradise Marsh (near Station 10, Fig. 1). They placed three common carp (*Cyprinus carpio*) of different size into nine enclosures, with final biomass ranging between 22.8 and 2,000 kg/ha, and kept three enclosures as control (with no carp added). These were used to study the impact of spawning activities on water quality during the first 15 d of experimentation (mid-May to beginning of June). Within 30 d, other fish including brown bullhead (*Ameiurus nebulosus*), pumpkinseed (*Lepomis gibbosus*), gizzard shad (*Dorosoma cepedianum*) as well as other carp had begun to colonize the enclosures on their own. In this paper, data are presented that were collected at the end of 68 d of experimentation (mid-July), to illustrate the effects of feeding activities of carp and other benthivorous fish on water turbidity since both spawning and feeding activities can lead to sediment resuspension and only the effects of spawning have been documented by Lougheed *et al.* (1998). Immediately before all the fish were removed from the enclosures by seining, water turbidities were measured in triplicate. Total wet weight of fish in each enclosure was then measured, and the number of large (> 30 cm) carp, as well as large (approximately 30 cm) and medium-sized (approximately 10 cm) bullheads was noted. An index of benthivore activity was calculated based on the assumption that only the large carp, large bullhead, and medium-sized bullheads were benthivorous (rather than all of the fish found in the enclosures). Large carp was assigned a value of 1, large bullhead (approximately 30 cm) a value of 0.5, 15 to 25 medium-sized bulheads (aproximately 10 cm) a value of 1, and 30 medium-sized bullheads a value of 2.

Field Observations of Fish Versus Wind-induced Turbidity

From 11 June to 23 June in 1992, a field monitoring program was initiated to determine the relative impact of wind resuspension and carp spawning activity on water turbidity. Mid-column water samples were collected from four sites that encompassed a range of conditions necessary to evaluate the independent and combined effects of wind versus carp spawning activity on water turbidity. These included a site at the west end of Mac Landing (Site 8 in Fig. 1) that was protected from wind but which was prime carp spawning habitat; a site in an embayment on the south shore that was protected from wind but that was not associated with carp (site “G” in Fig 1); an exposed site that was very windy and which was associated with high carp activity (near site 15 in Fig. 1); and an exposed site that was not associated with spawning activities (site “H” in Fig. 1). All turbidity measurements were taken in triplicate in the canoe with a Hach 2100 P Turbidimeter. Presence or absence of carp was recorded at each time, and corresponding wind speed and direction were also noted. In total, 19 sets of measurements were taken during the 1992 field survey.

Tributary Inputs of Turbidity

In July, a 96-h sampling program (18 to 22 July) was conducted near the mouth of the three tributaries, Spencer, Borer’s, and Chedoke Creek (near Stns 7, 11, and 18, respectively, in Fig. 1) to monitor changes in turbidity due to a rainstorm. The storm resulted in a total precipitation of 20.8 mm of rain during the first 48 h. At approximately 2- to 4-h intervals during the first 24 h after onset of precipitation, turbidities were measured at the mouth of Spencer’s and Borer’s Creeks from a canoe. Chedoke Creek was monitored for the first time 24 h after precipitation. For remainder of the 96 h, all three creeks were monitored as frequently as possible, except when sampling could no longer be carried out safely.

Statistical Analyses

SAS Jmp 3.1.5 for the Macintosh (Statistical Analysis Systems, Inc. 1995) was used to perform all statistical analyses in this study. Where appropriate, $\log_{10}$-transformed data or reciprocals were used to normalize the variance and to render them suitable for linear regression analyses. One-way ANOVA was used to separately determine significant year-to-year and site-to-site variation in TSS, while a two-way ANOVA was used to examine the separate and interactive effects of wind and carp disturbance on water turbidity. A forward stepwise regression analysis was used to determine the degree of explained variance attributed to inorganic, algal, and non-algal organic constituents of the TSS.

Composite Aerial Photographs

Vertical aerial photographs of overlapping sections of Cootes Paradise were taken from approximately 5,000 ft on 1 April, 3 June, 29 July, and 8 September in 1994. These photos were arranged to derive a composite of the entire marsh for each flight. The composites were digitized and have been computer enhanced for presentation in this paper.

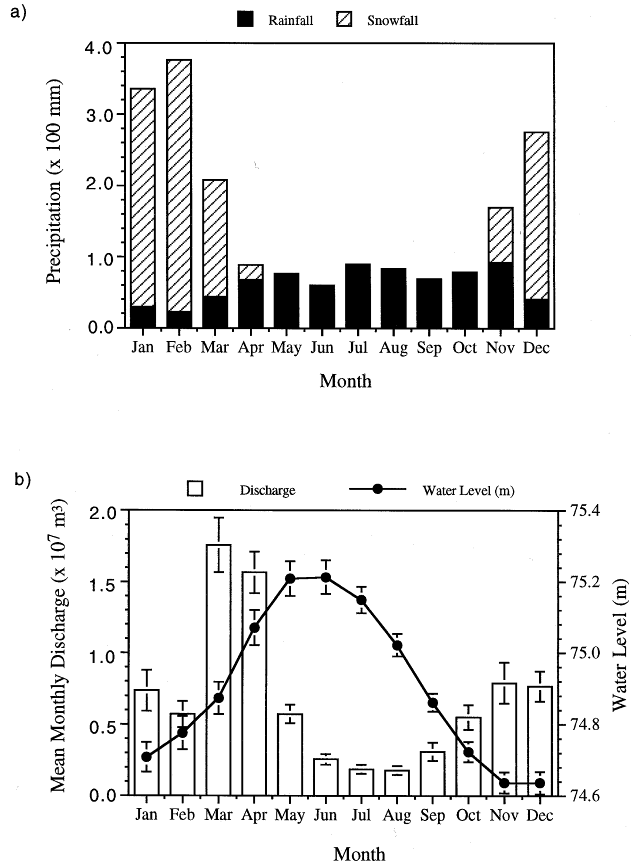

FIG. 2. a) Histogram of mean monthly precipitation data collected at the AES weather station at the Royal Botanical Gardens. b) Histogram of mean monthly discharge calculated for 1986 to 1996 data, and graph of corrresponding mean monthly water levels calculated for the same period using hourly data (see Methods). Error bars are 1 S.E. of the mean.

RESULTS

Marsh Hydrology

On average from 1986 to 1996, (Fig. 2a), the greatest amount of precipitation occurred during winter (December to February), most of it occurring as snowfall. The peak discharge occurred in March and April, when the accumulated snowfall melted in spring (Fig. 2b). A month following peak discharge, water levels were at their highest (Fig. 2b). During these 11 years, the amount of precipitation falling between October and March was a significant predictor of mean water level in May (r^2 = 0.70; P = 0.0015; n = 10). This lag between precipitation and water levels meant that marsh elevation was lowest during the winter, when precipitation was maximal (Figs. 2a &b). Based on these statistics, in an average year, one would expect the marsh to begin filling during spring (from March to May), attain maximum water levels during early summer (May and June), and drain by late fall (October and November). Using the mean annual discharge of 1.12×10^8 m^3 and a maximum capacity of 3.57×10^6 m^3, a mean residence time of 11.63 d can be calculated, which is within the range of 2 to 40 d reported in the Stage 1 Report of the Remedial Action Plan for Hamilton Harbour (Canada-Ontario Joint Document 1992).

Despite the predictable seasonal pattern, there were large interannual differences in the mean summer (May to August inclusive) water levels from 1986 to 1996 (Fig. 3a). Mean monthly water level data for Cootes Paradise from 1960 to 1993 ranged from extremely high (> 75.6 m asl) to low (< 74.4 m asl) levels four times over these 33 years (Chow-Fraser unpub. data). In this study, relatively high water levels occurred in 1986 (75.33 m asl) and in 1993 (75.30 m asl), while low water levels occurred in 1988 (74.88 m asl) and in 1995 (74.92 m asl), a duration of only 2 years between adjacaent peaks and troughs, although there were 7 years between occurrences of high water levels (Fig. 3a).

Mean seasonal high and low water levels differed by 45 cm, which is relatively large considering that Cootes Paradise only has a mean depth of 70 cm. Figure 3b illustrates the corresponding drop in water depth through the season at representative sampling sites during a year with high water level (1993), versus a year with intermediate water level (1994). Declining water levels through the summer meant that the shallow vegetated sites (stations 8, 16, and 17) were no longer inundated during August in most years, and in years with low water levels, they were not accessible at all during the summer (1994 and 1995). For deeper water areas, the drop in water level did not affect accessibility, but had an influence on wind resuspension since these were located in the exposed eastern end of the marsh (station 1). Figure 3b also illustrates obvious interannual variation in the rate at which water depths declined through the season, and these probably reflected differences in meteorological conditions (evaporation versus precipitation) during 1993 and 1994.

Wind Conditions

Since there were no continuous wind data available at or near the water surface in the marsh, mean

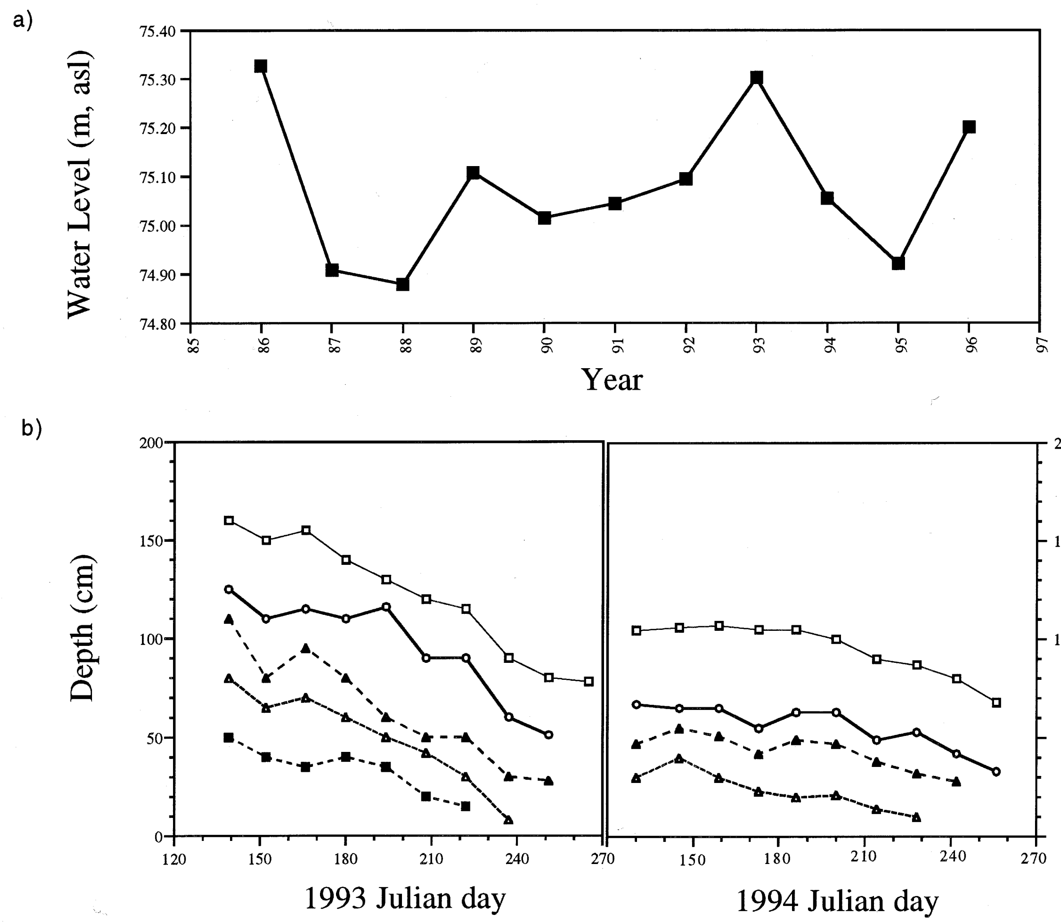

FIG. 3. a) Mean seasonal water level (May to August inclusive) for Cootes Paradise Marsh calculated from hourly data (see Methods). b) Changes in water depth through the season at different sampling stations in 1993 and 1994. (Key to symbols: open square = station 1; open circle = station 3; solid triangle = station 5; open triangle = station 8; and solid square = station 16).

hourly data measured at the Royal Botanical Gardens AES weather station were used. These data were compared with site-specific wind measurements obtained from a canoe during the biweekly sampling trips. This is not a direct comparison since the AES data were daily averages calculated from hourly values at one site 5 km away from the marsh, whereas the canoe data were calculated from spot measurements obtained over the course of a day throughout the marsh. Nevertheless, a statistically significant relationship was found in which AES data were consistently higher than corresponding canoe data (canoe data = 0.749 + 0.580 AES data; $r^2 = 0.68$, $P < 0.01$; $n = 17$). Hence, the AES data are suitable for describing prevailing wind conditions over the marsh, whereas canoe measurements are suitable for indicating site-to-site variation within the marsh at specific times.

Mean monthly wind data (wind speed and direction) for 1993 and 1994 are plotted in Figures 4a and b. The prevailing winds in both years blew primarily from the southwest (between 180 and 240°). Since Cootes Paradise Marsh is oriented with its long axis (4 km maximum length) in the direction of the prevailing winds, the marsh is very vulnerable to wind-resuspension in the exposed open-water areas of the marsh (stations 1, 2, 10, and 12). Wind speeds varied seasonally during 1993 and 1994. The strongest winds were encountered during winter (January and February) and the calmest during late summer (August and September, Fig. 4b). Hence, during the growing season (May to August

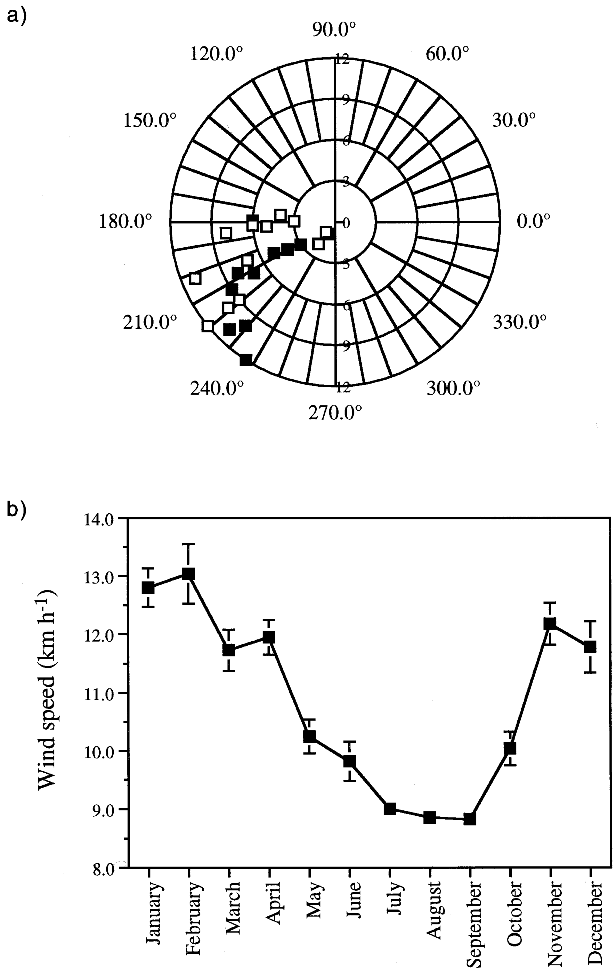

FIG. 4. a) Average monthly a) wind directions and b) wind speeds (km/h) (± SE) obtained from the AES weather station at the Royal Botanical Gardens for 1993 (closed symbols) and 1994 (open symbols). Data closest to the centre of the circle in a) correspond to January (1), whereas data furthest from the centre correspond to December (12).

inclusive), the marsh would be most vulnerable to wind-induced resuspension in early Spring (May and June).

Visual Trends from Aerial Composite

The large site-to-site variation in TSS was very evident from the series of aerial composites taken in 1994 (Plate 1). Consistent with the general pattern of water-level increases (Fig. 2b), the marsh had just begun to fill when the April 1st composite was taken; sediment-ladened water was seen entering the marsh from Hamilton Harbour through the marsh outflow below Hwy 403 (see Fig. 1). The June 3rd composite revealed a great deal of turbidity in the shallow weedy areas to the north of Station 3 (known as Hopkins Bay) and in the marsh inlet to the south (known as Mac Landing) where common carp were found in abundance. Thus, high water turbidity in these shallow regions of the marsh during early summer are likely carp-induced resuspension.

Further anecdotal evidence of the impact of carp on water clarity was provided by the installation of a temporary dike by the Royal Botanical Gardens during May 1994. This dike joined Rat Island (Fig. 1) to points on the north shore of Cootes Paradise and prevented carp from entering the entrained area. The visibly clearer water inside the diked area relative to water in Hopkins Bay may have been a result of the exclusion of carp; however, it is possible that the dike simply diverted turbid water (generated in Hopkins Bay) away from Rat Island. By July 29th, the dike had been damaged, and water behind the formerly diked area had become as turbid as other areas in Hopkins Bay. In both June and July composites, resuspended sediment from Hopkins Bay were seen to flow east and accumulate near station 2. The high turbidity measured at station 2 could not be attributed to greater fetch because wind speeds recorded at this station were much lower than that at Station 1 (7.0 versus 11.5 km/h, respectively).

The aerial composites also demonstrated temporal changes in the clarity of water entering Cootes Paradise from two of the main tributaries, Spencer's and Chedoke Creek (Fig. 1). During early June, water from both creeks appeared relatively clear (Plate 1), but by the end of July, the water had turned muddy, probably because of sediment-laden runoff resulting from a rainstorm a few days earlier. Water levels in the marsh tend to drop markedly by September (Fig. 2b). The September composite shows that water levels had dropped to such an extent that the sediment surface was visible throughout most of the marsh. One consequence of this improved clarity was prolific growth of benthic algae which could be seen covering the bottom of Mac Landing and the northeastern section of the marsh (near Station 1; Plate 1).

Measured Trends in TSS Concentrations

Variation in TSS concentrations at five sites from 1993 to 1996 are detailed in Figure 5. TSS levels

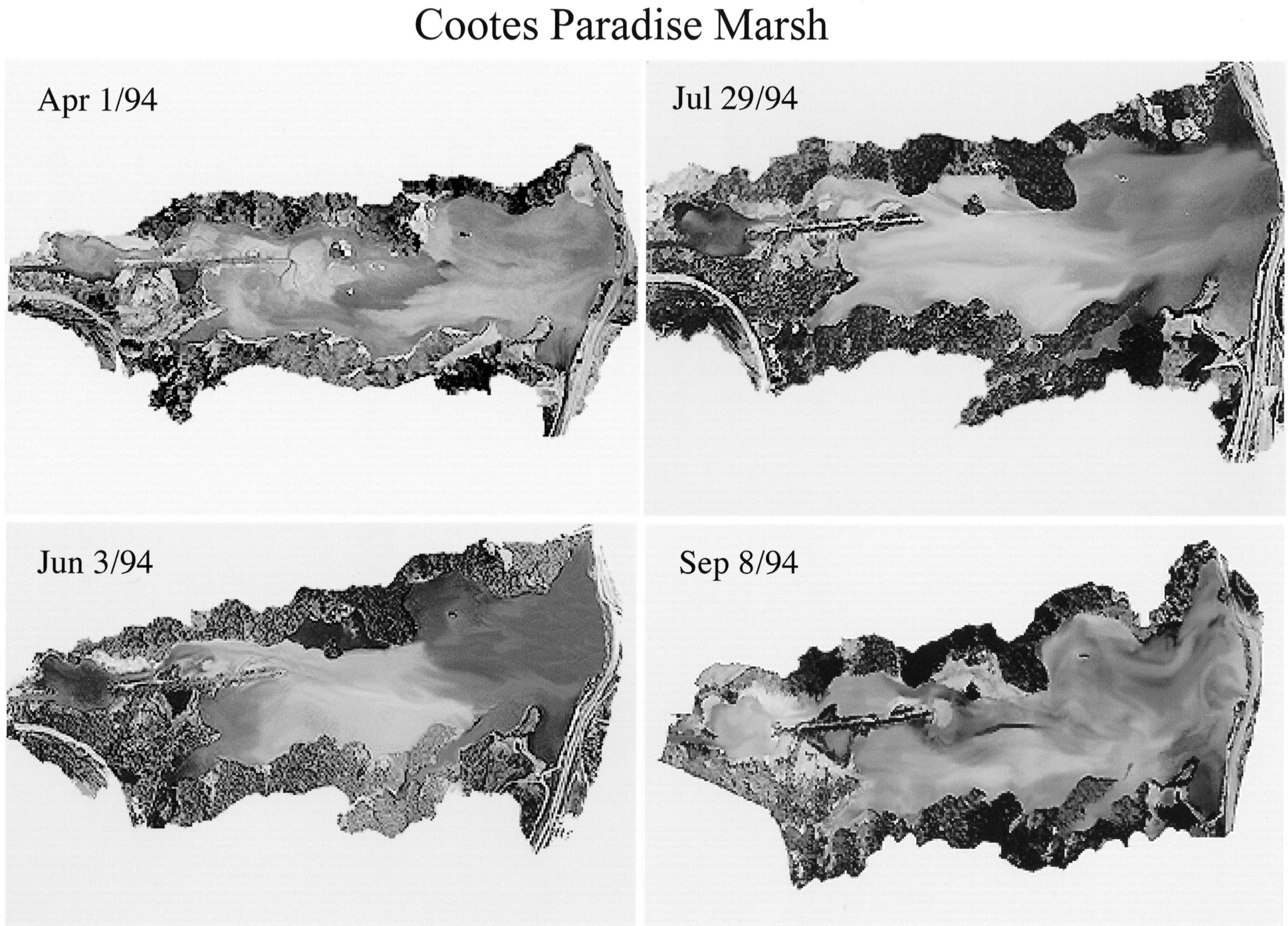

PLATE 1. ***Composite vertical aerial photograph of Cootes Paradise Marsh taken at approximately 5,000 ft on 1 April, 3 June, 29 July, and 8 September 1994.***

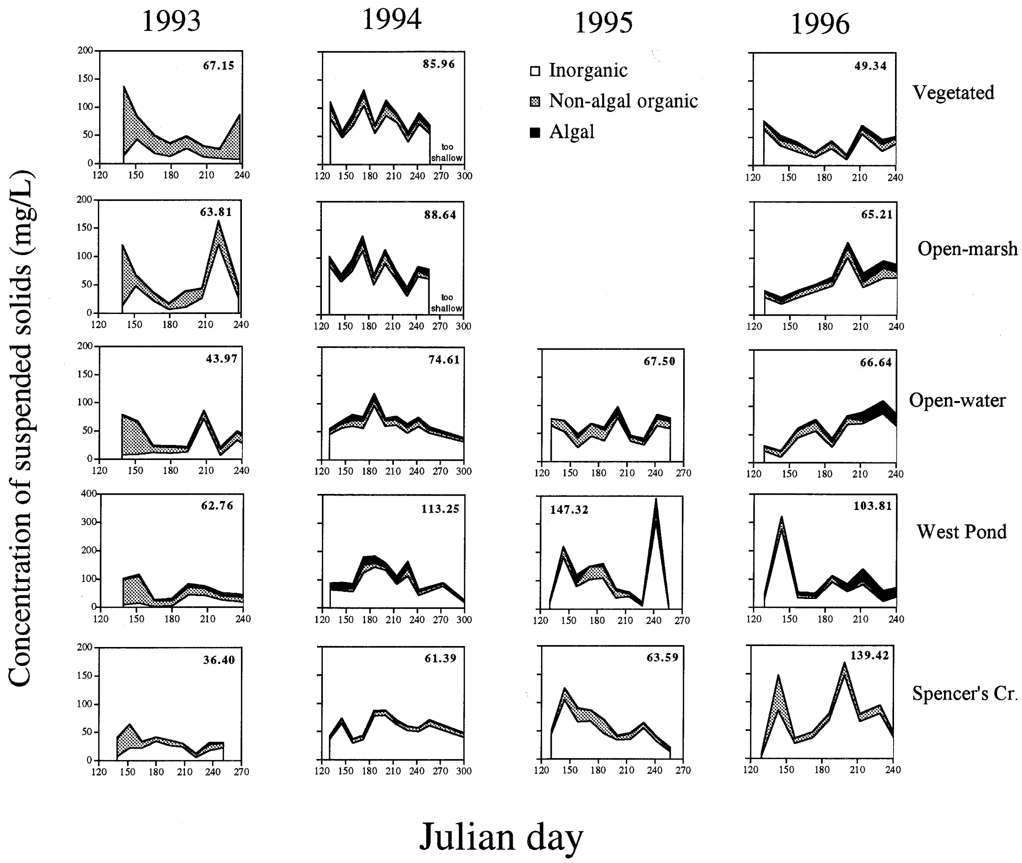

FIG. 5. Seasonal distribution of TSS concentrations (mg/L) at five site types in Cootes Paradise Marsh from 1993 to 1996. Numbers inside panels are seasonal mean values for site-year combinations. No data were collected at the vegetated and open-marsh site types in 1995 because water levels were too low to permit access to these sites.

differed significantly among years (one-way ANOVA; $P = 0.023$) when all sites were considered together, with highest overall values recorded in 1994, and lowest in 1993. Because there was considerable variation in mean summer water-levels over these 4 years (Fig. 3a), it was hypothesized that the year-to-year variation may be attributed to interannual differences in mean water levels. Since the water-level cycle in Cootes appeared to span 8+ years, this hypothesis was tested with an expanded dataset, which included TSS concentrations measured biweekly from May to August from 1986 to 1996, at stations 1 to 6 (Chow-Fraser *et al.* 1998). A significant inverse relationship was found between TSS concentrations and corresponding mean seasonal water level ($r^2 = 0.71$; $P = 0.002$) (Fig. 6), indicating that the low TSS concentrations may have been a function of higher water levels experienced in 1993 compared with 1995.

There were also significant variations in TSS levels among the five site types for the 1993 through 1996 pooled dataset (one-way ANOVA; $P = 0.011$; Fig. 5). When data were sorted on a year-by-year basis, there were no significant differences among sites in years of higher water-levels (1993 and 1996); however, in years of lower water levels (1994, $P < 0.0001$; 1995, $P = 0.023$), TSS concentrations at West Pond (station 5) were significantly higher than those at the open-water sites (stations 1, 2, and 12) and at Spencer's Creek (station 7) (Fig. 5). Each site type was examined for significant year-to-year variations in TSS concentrations. Significant differences were found among years for vegetated ($P = 0.036$), open-water ($P = 0.004$) and Spencer's Creek ($P = 0.014$) sites.

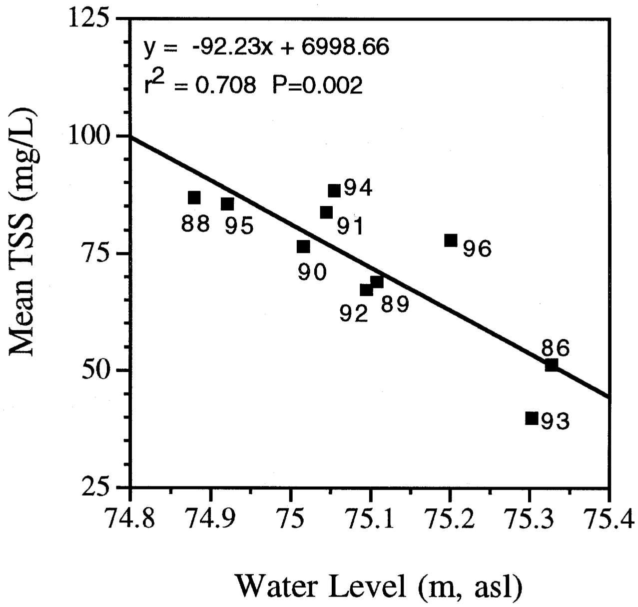

FIG. 6. Mean summer TSS concentration versus mean summer water level from 1986 to 1996 (May to August data inclusive). The year associated with each data point is indicated below the symbol.

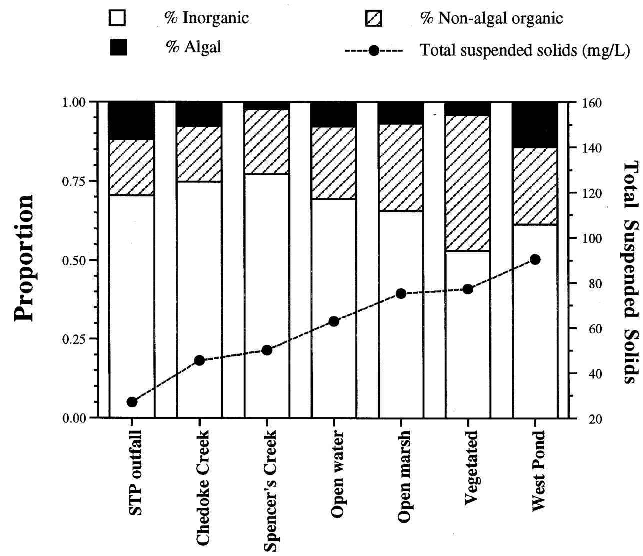

FIG. 7. Mean seasonal contribution of inorganic, non-algal organic and algal suspended solids corresponding to seven different site types in 1994. Mean seasonal TSS concentrations (closed circles) corresponding to site types are also indicated.

Contribution of Planktonic Chlorophyll

To determine how planktonic chlorophyll (CHL) influenced the distribution of TSS at different sites in the marsh, the contribution of algae in the suspended solids was determined. For this analysis, only 1994 data were used since the most comprehensive monitoring program had been carried out that year. Data from Stations 1 to 14 were sorted by site types that included the five specified in Figure 5 (Methods) as well as "STP outfall" (station 6) and "Chedoke Creek" (station 11) (Fig. 7). Even though the mean concentration was lowest at the STP outfall and highest at West Pond (station 5), they both contained a high proportion of algae in the suspended solids (12 and 15% respectively). By contrast, the vegetated sites (stations 8 and 15) that had relatively high TSS concentrations only contained a low percentage of algae (< 5%). In a stepwise regression analysis, CHL only explained 2% of the variation in TSS while the inorganic and non-algal organic solids explained 70% and 18%, respectively ($n = 378$; $P < 0.0001$). The only sites that had a significant correlation between TSS and CHL were West Pond (0.76, $P = 0.006$) and the open-water sites ($r = 0.41$, $P = 0.02$). Overall, however, the influence of algae on the temporal and spatial distribution of TSS in the marsh was low.

Comparison of TSS and Water Clarity Indicators

Besides being labor-intensive and expensive to carry out, measurement of TSS cannot be performed *in situ*. These characteristics limit its usefulness in exploratory field studies where repeated measurements are required to document the type of rapid changes resulting from wind- and fish-induced disturbance and rainstorms. Therefore a surrogate of TSS was sought that was accurate, convenient, and inexpensive to measure. For this purpose the reliability of three common indicators of water clarity were compared: water turbidity, Secchi depth transparency, and light extinction coefficient. Although they were all significant predictors of TSS, turbidity was associated with an r^2 value of 0.73, compared with 0.52 for light extinction coefficient and only 0.13 for Secchi depth transparency (Table 1). Since water turbidity is the best overall predictor of TSS, it has been used as a TSS surrogate in all subsequent analyses.

Effect of Rainstorms

A field survey was conducted in July 1993 to monitor the effects of a rainstorm on the turbidity of water in the three main tributarires because rainfall tend to be heaviest in July and August in a typi-

TABLE 1. Comparison of regression equations of the concentration of total suspended solids (TSS, mg/L) versus water turbidity (TURB; FTU), light extinction coefficient (EXT), and Secchi depth transparency (SECCHI; cm). P value for all regression coefficients were < 0.0001; numbers in bracket are the SE of the coefficients.

Independent Variable	Regression Equation	r^2	n
$\log_{10}$ TURB	$\log_{10}$ TSS = 0.665 (± 0.035) $\log_{10}$ TURB + 0.716	0.72	139
TURB	TSS = 0.995 (± 0.060) TURB + 20.054	0.67	139
$\log_{10}$ EXT	$\log_{10}$ TSS = 1.0336 (± 0.086) $\log_{10}$ EXT + 1.093	0.55	138
EXT	TSS = 13.38 (± 1.164) EXT + 2.030	0.53	118
$\log_{10}$ SECCHI	$\log_{10}$ TSS = –0.533 (± 0.068) $\log_{10}$ SECCHI + 2.527	0.31	138
1/SECCHI	TSS = 493 (± 89.465) 1/Secchi + 47.328	0.18	138

cal year (Fig. 2b). During the first 24 h following the onset of a heavy rainstorm, turbidities at the mouth of Spencer's and Borer's Creeks more than doubled (Figs. 8a and b, respectively). The amount of incoming sediment varied from creek to creek; for example, turbidities in Chedoke Creek increased 3-fold (Fig. 8c) while turbidities in the other two only doubled over the period of observation. These differences may reflect the amount of paved road surfaces in the different watersheds, since Chedoke Creek, which had the greatest urbanized areas in the watershed (Hamilton-Wentworth Pollution Control Plan, unpub. data) also had the highest increase. As a comparison, the average seasonal turbidities in Spencer's Creek during 1993 was 26.5 (± S.E. 4.73) FTU, which is similar to the level measured at the beginning of the rainstorm (Fig. 8a). Unfortunately, there are no comparable data for Chedoke and Borer's Creeks since they were not part of the 1993 biweekly monitorong program. Nevertheless, it is clear from the Spencer's Creek comparison that variable amounts of sediment-laden water can enter the marsh during a rainstorm that can raise the level of suspended solids considerably above background.

Effect of Fish Feeding

The number of large carp (> 30 cm) per enclosure ranged from zero to five (1,000/ha) while that of large brown bullheads (approximately 30 cm) ranged from zero to six (1,200/ha). The total biomass of fish ranged from 2.0 to 13.2 kg per enclosure (400 to 2,640 kg/ha). There were also numerous smaller bullheads (approximately 10 cm) that ranged from < 5 to 30 per enclosure. Despite the large unexplained variation ($r^2 = 0.37$), water turbidity increased significantly with the number of carp per enclosure (Fig. 9a), and this is consistent with the hypothesis that increased sediment resuspension is directly related to increased feeding activity of carp. The improved r^2-value (0.41) when turbidity was regressed against the biomass of all fish found in the enclosures suggests that other fish may also be implicated in sediment resuspension (Fig. 9b). In particular, the activity of all benthivorous fish, which included both carp and brown bullhead, yielded the highest r^2-value (r^2-0.46; Fig. 9c). This comparison illustrates that a high density of benthivorous fish (approaching 2,400 kg/ha) can significantly increase water turbidity by 20 to 30 FTU above background.

Effect of Wind Speed

The best direct evidence of the effect of wind activity on water turbidity is the highly significant ($P < 0.001$) regression between mean water turbidity and mean wind speed obtained for the 17 sampling stations in Cootes Paradise Marsh during 1993 and 1994 (Fig. 10). On average, the windiest conditions were encountered at stations 5, 10, and 1, which were all located in exposed open water areas that were easily affected by fetch. The more sheltered stations, such as station 6 (STP outfall), 7 (Spencer's Creek) and those located near weed beds (stations 8 and 16), were associated with the lowest water turbidities.

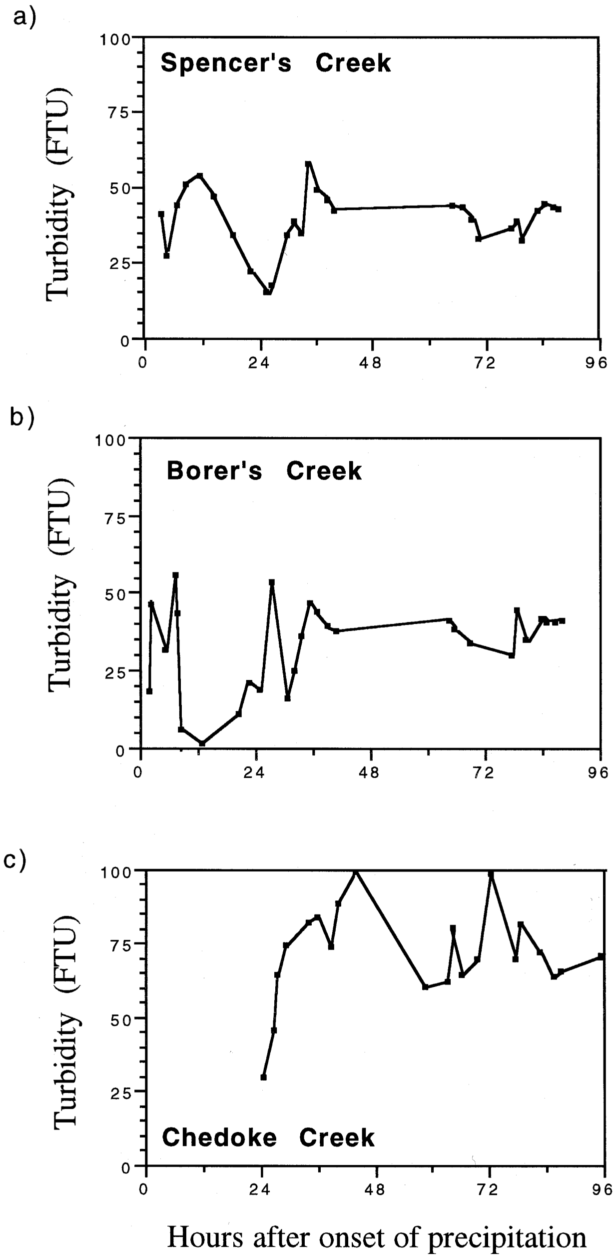

FIG. 8. Changes in turbidity after onset of a July rainstorm in 1993 at a) Spencer's Creek, b) Borer's Creek, and c) Chedoke Creek.

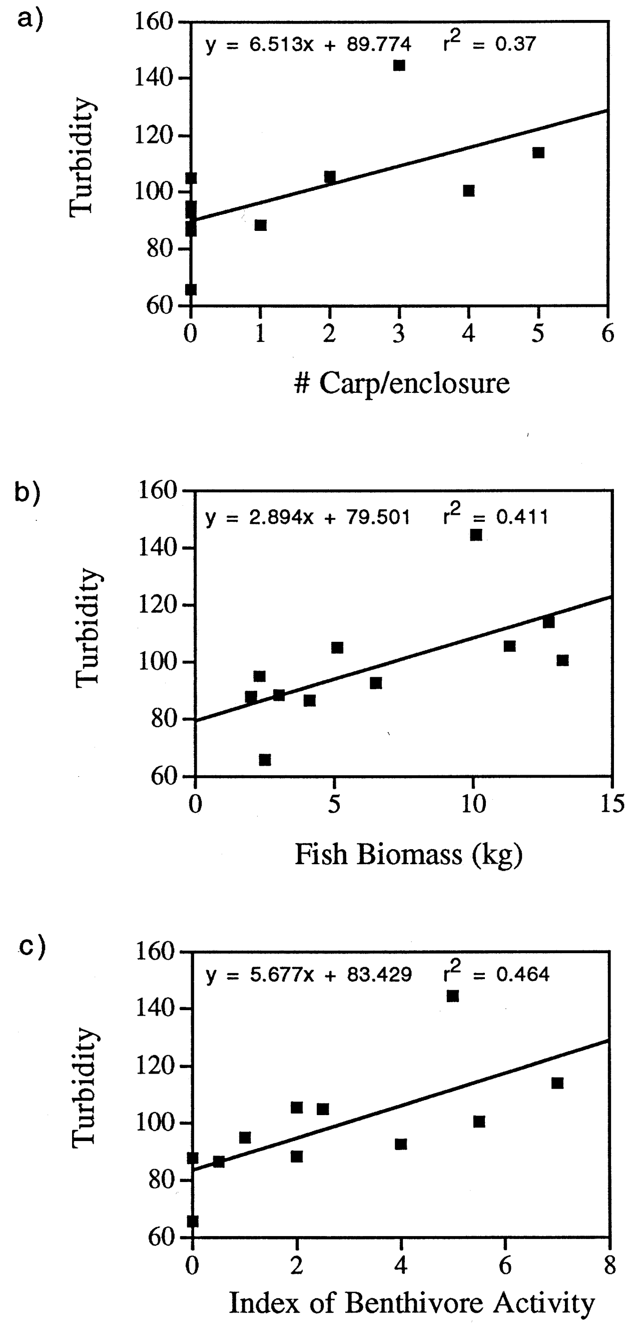

FIG. 9. Plot of turbidity (FTU) versus a) density of large carp (> 30 cm in length), b) total fish biomass (kg) in experimental enclosures and c) index of benthivore activity. See Methods for calculation of benthivore activity.

Relative Effect of Carp and Wind Activities

It is clear from the foregoing that both fish-induced (Fig. 9) and wind-induced (Fig. 10) resuspension independently contributed to increased water turbidity in Cootes Paradise Marsh. The relative impacts of each were evaluated using data from a field survey conducted during the carp spawning season at four sites in the marsh. When water turbidity was regressed against both variables in a stepwise regression, wind speed was entered first and explained 41% of the variation ($n = 19$; $P =$

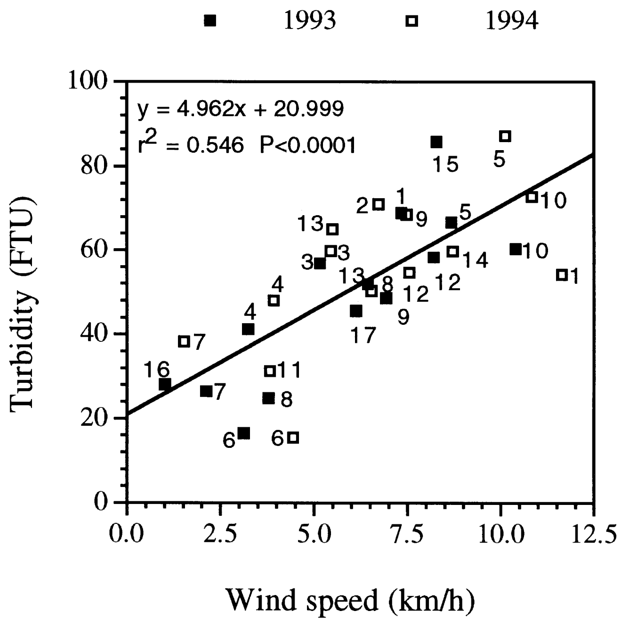

FIG. 10. Plot of mean summer water turbidity (FTU) versus mean wind speed (km/h) measured biweekly at 17 sampling stations from May to August during 1993 (solid squares) and 1994 (open squares).

0.0011). Carp absence/presence was then entered and significantly explained another 21% of the remaining variation (P = 0.0079). The multiple-regression model explained 63% of the total variation in turbidity, and had an intercept of 25.4 FTU, which may be interpreted as the turbidity that was neither accounted for by carp- or wind-resuspension, but instead reflected background levels (algal growth, residual effects of storms, etc.)

DISCUSSION

Extensive temporal variation in turbidity was observed both seasonally and interannually at the five site types in Cootes Paradise Marsh (Fig. 5). Much of the interannual differences in mean TSS could be attributed to differences in water levels. The inverse relationship between TSS levels and mean summer water levels may simply reflect dilution of suspended materials in a larger volume of water when water levels increased, or the fact that wind effects are stronger in shallower water (Limón *et al.* 1989) and thus could induce higher turbidity when water levels drop. By comparison, the large seasonal variations within a site could be attributed to a combination of factors, depending on prevailing conditions at the time of sampling. For example, turbidities at the three main tributaries were greatly affected by rainstorms, which tended to occur in an unpredictable manner during July and August. The open-water stations tended to be turbid from wind-driven resuspension (Fig. 10) and exchange of water between the harbor and the marsh while the marsh is filling (Plate 1), while the shallow, protected, weedy areas were kept turbid primarily by carp-induced resuspension (spawning and feed activities) during the summer. Turbidity in West Pond was maintained by wind-driven resuspension (Fig. 10) as well as proliferation of algal blooms which coincided with periods when wind speeds were reduced (toward end of the summer). A similar situation was observed by Stefan *et al.* (1976) and Havens (1995) who found that formation of algal blooms in shallow eutrophic lakes was inversely related to wind resuspension. In exposed shallow vegetated areas, both carp- and wind-induced resuspension could be invoked to explain variation in turbidity.

The effects of wind (or fetch) on water turbidity have been documented for large shallow lakes (Limón *et al.* 1989, Hamilton and Mitchell 1997). Bengtsson and Hellstrom (1992) showed that in smaller lakes, wind speeds greater than 12 to 15 km/h can also result in substantial resuspended materials in sediment traps. In this study, it is clear that wind-induced resuspension played an important role in maintaining high turbidity in exposed open-water areas of the marsh (station 1). During the ice-free season, wind speeds are highest in May and gradually decrease through remainder of the summer (Fig. 4b). Average wind speeds during May in 1993 and 1994 at station 1 was 10 km/h, and based on the relationship between wind speed and turbidity (Fig. 10), I estimate that wind-induced turbidities should have ranged between 60 to 70 FTU, which is close to the observed turbidity of 71.35 FTU.

According to Kelley *et al.* (1985) and Herdendorf (1992), water level of the Great Lakes fluctuates over a 7 to 10 year period; Mitsch (1992), however, suggests that the time between a high and low water level occurrence is actually between 10 and 15 years. The estimate of 8.25-y cycle for Cootes Paradise Marsh based on monthly mean water levels from 1960 to 1993 is certainly more consistent with a shorter rather than a longer cycle (Fig. 3a).

In the limnetic literature, Secchi depth transparency is used more often as indicator of water clarity compared with light extinction coefficients and water turbidity. In wetlands, however, Secchi measurements are biased when water depths drop at the same time as water begins to clear, a situation that arises at the end of the summer in the marsh inlets of Cootes Paradise Marsh (Fig. 3b). When water depths drop below 30 cm in the marsh, the photometer which is used to calculate light extinction coefficients also has limited usefulness as it cannot be operated without disturbing the sediment. By comparison, such limitations do not apply to the turbidimeter since samples can be readily collected without disturbing the sediment in water depths as shallow as 20 cm. In addition to these benefits, turbidity was found to be the best predictor of TSS values when compared with light extinction and Secchi depth transparency data (Table 1). Hence, water turbidity should be used preferentially over these other two variables in wetland monitoring programs.

Annually, when water levels in the marsh drop (Fig. 2b), the majority of the fish community (including the common carp) are forced to migrate out of the marsh into deeper areas of Hamilton Harbour and Lake Ontario to overwinter. Their return in spring is predictable, governed by temperature cues. By preventing their re-entry to the marsh during spring, RBG plans to exclude most of the large carp from Cootes Paradise Marsh. Their exclusion is expected to reduce water turbidity in at least two ways: reduced turbidity commensurate with reduced spawning activities (Lougheed *et al.* 1998), and reduced feeding activities during summer (Fig. 9a). There may also be indirect benefits through increased grazing by herbivorous zooplanton since zoplankton biomass and turbidity were inversely correlated during the spawning period (Lougheed *et al.* 1998). Unfortunately, excluding large carp may not result in reduced turbidities if brown bullheads fill the niche previously occupied by carp because in the enclosure experiments of this study, the index of benthivore activity was significantly better for predicting water turbidity than carp density alone (Fig. 9c versus 9a).

That detection of significant site-to-site differences in TSS concentrations depends on the mean seasonal water levels deserves further investigation (Fig. 5). It is possible that both carp-induced and wind-induced resuspension operate differently in shallow versus deep water. Although no explanation is proposed for the inconsistent manner in which year-to-year variation is detected among different site types, it is possible that examining differences in hydrology and food-web dynamics (algae-zooplankton-macrophyte-fish interactions) may provide some clues. The results from this study should have general relevance to other coastal Great Lakes marshes since fluctuating water levels, non-point-source pollution, and carp disturbance are common stressors in these ecosystems.

ACKNOWLEDGMENTS

I owe a great debt of thanks to the dozens of undergraduate and highschool students who contributed in various ways to my long-term monitoring program from 1992 to 1996. I salute you all! I especially want to thank Vanessa Lougheed, Barb Crosbie, Mariëtte Prent, Stu Taylor, and Paul Armstrong for planning and executing much of the field work that have been included in this paper. Mariëtte Prent also conducted the water-level frequency analyses. I acknowledge the hard work and dedication of Tricia Frayn who co-ordinated the 1994 and 1995 field programs, and whose daring coupled with the expert piloting skills of Chris Diggins resulted in the aerial photos that eventually became the aerial composites. Financial support were provided by the Tri-Council Eco-Research Program for Hamilton Harbour from 1993 to 1996; the Ontario Ministry of Natural Resources and the Ontario Ministry of Environment and Energy through their Environmental Youth Corp Program from 1993 to 1996; Human Resources Canada through their Summer Career Placement Program in 1993 to 1996; Natural Sciences and Engineering Research Council through its Undergraduate Summer Scholarship program from 1992 to 1996; and Project Paradise of the Royal Botanical Gardens in 1993 and 1994.

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Submitted: 17 December 1997
Accepted: 21 July 1999
Editorial handling: Roger Flood, David Francko