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Impact of urbanization on the water quality, fish habitat, and fish community of a Lake Ontario marsh, Frenchman's Bay

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Abstract Frenchman's Bay is a barrier beach wetland with a highly urbanized watershed located east of Toronto, along the north shore of Lake Ontario. Degradation of water quality has reduced the historically large stand of emergent vegetation to fringe emergent beds to the north and south of the Bay. Altered hydrology and runoff from the urban watershed and a nearby major highway have resulted in poor water quality, and warmer waters at the northern site. By contrast, the southern site has considerably cleaner and cooler water, as it is influenced by exchange of good-quality water with Lake Ontario. These differences in water quality were reflected in the composition of the fish assemblages that were sampled at the two sites over a 2-year period. Comparisons with past studies indicate that the dominant fish community of Frenchman's Bay has been relatively stable for the last 20 years. Scores for the Wetland Fish Index, an indicator of wetland condition, were significantly higher in the south site than in the north site, which corresponded to significant differences in Water Quality Index scores. Although the northern portion of Frenchman's Bay shows clear signs of degradation, the southern portion contains important fish habitat for western Lake Ontario.

Keywords Urbanization · Coastal wetlands · Fish habitat · Water quality · Great Lakes

Introduction

The ecology of wetlands in largely urban settings can be influenced by stressors that are unique to these systems, including recreational impacts (boating, angling), altered hydrologic regimes related to increased impervious surfaces in the watershed (Booth and

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Jackson 1997; Eyles et al. 2003), and nutrient and sediment enrichment from effluents of sewage-treatment facilities, storm sewers and culverts that drain major transportation corridors (Chow-Fraser et al. 1996; Chow-Fraser 1999; Ehrenfeld 2000). Urban impacts on coastal wetlands can be divided into two broad categories: habitat alteration or destruction, and water quality modification (i.e. increased loading of nutrient, sediment, and pollutants; Lee et al. 2006). Wetlands in western Lake Ontario have been reduced by 57%, and more developed areas have reductions in wetland area as high as 100% (Whillans 1982). Impacts of urbanization are prevalent in southern Ontario, Canada, where major cities such as Toronto and Hamilton have expanded rapidly over the past three decades, and the number of new dwellings built has kept pace with population growth (Liu et al. 2003). This unrestrained growth has severely altered the ecosystem function, and in particular drowned river-mouth marshes and protected lagoons along the Great Lake shoreline (Environment Canada 2001).

Great Lakes wetlands are important spawning and nursery habitats for fishes (Brazner and Beals 1997; Jude and Pappas 1992) and, although impacted by urbanization, coastal wetlands still provide valuable and productive habitat for biota. Urbanization may affect coastal wetland fish habitat in a more indirect way when compared to streams (e.g., altered fish life cycle from changes in hydrological regime; Freeman et al. 2001); wetland habitat will also be affected indirectly through increased loading of sediments (Trimble 1997), nutrients (Wahl et al. 1997), and pollutants associated with road runoff (Forman and Deblinger 2000). Urban impacts are further exacerbated by increased runoff due to the decreased storage capacity of urban soils (Booth and Jackson 1997). The degraded water quality and high algal biomass that result from all these urban inputs cause the loss of the submergent plant species that are vital habitat for fish and other biota (Chow-Fraser et al. 1998). Close proximity to urban development (Steedman 1988; Steward et al. 2001), even low intensity urban development, can have negative effects on the species diversity and structure of fish assemblages over time (Weaver and Garman 1994).

In this study, we will conduct a study on the long-term impact of urbanization on the ecology of an urban lagoon, Frenchman's Bay. First, we will relate long-term changes in fish habitat and fish community to changes in land use in the watershed over four decades. We will also determine if inputs from the heavily urbanized watershed have led to differences in degraded water quality at sites close to the creek mouths compared to sites farther from the creeks. Secondly, we will determine if differences in water quality were reflected in individual associations of fish species with different sites in the lagoon through the use of a newly published fish-based indicator. Finally, we will show that despite the long-term negative impacts of urbanization, Frenchman's Bay is still an important spawning and nursery habitat in western Lake Ontario, and we will identify which ecological functions have been impaired and use our results to suggest remedial actions.

Methods

Site description

Frenchman's Bay (43°49′01″ N; 79°05′34″ W) on the north shore of Lake Ontario is a semi-enclosed lagoon-type wetland located in the city of Pickering, Ontario (Fig. 1), and has a small watershed of 20 km^2 of which 80% is urbanized (Eyles et al. 2003). The population of Pickering is over 87,000 in 2001 with a density of 376.3 people/km^2 (Statistics Canada 2002). The City of Pickering, in which Frenchman's Bay is located, had

Fig. 1 Map of Frenchman's Bay, and the location of two in-marsh sampling stations (North and South) and two stream sampling stations (Amberlea Creek, downstream of Highway 401, and in Pine Creek, above Hwy 401). *Asterisks* indicate the location of stations at which water samples were taken for nutrient chemistry and suspended solids analyses

a higher population increase than the Ontario average from 1996 to 2001 (10.3 and 6.1%, respectively) and a higher building rate of new homes in Pickering than Ontario (24.1 increase and 14.3% increase, respectively) over a 10-year period from 1991 to 2001 (Statistics Canada 2002).

The semi-enclosed coastal lagoon is drained by four main tributaries: Amberlea (301 hectares, 13.5% of watershed area), Dunbarton (212 ha, 9.5%), Pine (677 ha, 30%) and Krosno (784 ha, 35%) creeks (Eyles et al. 2003; Fig. 1). Smaller tributaries on the north, west and east margins of Frenchman's Bay have been extensively engineered by pipes and culverts to form storm-water 'sewersheds' (260 ha, 12%) that empty directly into the Bay. Currently, Frenchman's Bay covers an area of 85 ha, of which approximately 47 ha is open water (Environment Canada 2001). The Bay is relatively shallow, with a maximum mid-summer depth of 3.5 m. However, its water level is closely tied to seasonal variation in Lake Ontario elevations, with maxima occurring in May and minima occurring in December. Accordingly, a drop of 30–40 cm from May to September is not unusual, and this results in a substantial shrinkage of aquatic habitat along the wetland perimeter over the growing season. Although much of the lagoon is separated from Lake Ontario by a barrier beach (900 m long × 50 m wide × 2 m high), it is kept connected to the lake by a dredged entrance that allows boats to access marinas within the Bay.

Five taxa of emergent macrophytes dominated the marsh, including three cattail species (narrow-leaved cattail *Typha angustifolia*, broad-leaved cattail *T. latifolia*, and hybrid cattail *Typha X glauca*), and two loosestrife species (swamp loosestrife *Decodon verticillatus* and purple loosestrife *Lythrum salicaria*, an invasive exotic). The latter were well established among the cattail beds along the shore. There were so few submergent macrophytes that we were unable to determine their areal extent. Of those that we encountered, we identified four native taxa (sago pondweed *Potamogeton pectinatus*, slender pondweed *Potamogeton*. sp., common waterweed *Elodea Canadensis*, and common bladderwort *Utricularia vulgaris*) and two exotic taxa (curly-leaved pondweed *P. crispus* and Eurasian milfoil *Myriophyllum spicatum*). The floating macrophytes included two native (fragrant water lily *Nymphaea odorata* and star duckweed *Lemna trisulca*) and one non-native taxa (European frog-bit *Hydrocharis morsus-ranae*).

Water sampling and analysis

We identified wetland habitat present in Frenchman's Bay from visual surveys on-site and from existing maps of the area (Fig. 1). Two distinctive areas of the Bay were classified as northern site near the mouths of Amberlea and Pine Creek and the southern site which was not associated with any tributaries. During the summer of 2002 (May to September), we used YSI multi-parameter probes (XL Model; YSI Inc., Yellow Springs, OH) to obtain hourly measurements of four physico-chemical characteristics (pH, temperature, dissolved oxygen (DO) and conductivity) in Amberlea and Pine creeks, respectively (Fig. 1). The station for Amberlea Creek was located downstream of a large culvert that collected runoff from Highway 401, whereas that for Pine Creek was located well above the highway and was presumably unaffected by highway runoff. All sensors in the probes were calibrated in the laboratory immediately before initial deployment, and DO sensors were maintained and calibrated monthly in the field thereafter. The record at the Amberlea Creek site was discontinued in early July due to equipment failure. In September, we re-established a YSI 6600 multi-parameter probe (pH, temperature, DO, conductivity, turbidity and chlorophyll; Yellow Springs, Inc.) in Amberlea Creek and the YSI XL in Pine Creek for the winter of 2002–2003. Two YSI 6600 probes were also deployed in Frenchman's Bay at the North and South stations. The North station (indicated by "North" in Fig. 1) was near the confluence of Amberlea, Dunbarton, and Pine Creeks; the other was located at the southwestern end of the marsh in an open-water area near a relatively intact cattail bed (indicated by "South" in Fig. 1).

On August 20, 2002, we took georeferenced measurements of pH, temperature, DO, conductivity, turbidity and chlorophyll along nine transects across Frenchman's Bay (six transects on the East to West axis and three transects on the north to south axis for a total of 179 individual measurements). This was accomplished by towing a 6600 multi-probe attached to a data-logging YSI 650 display (equipped with a Garmin GPS unit) at about 30 cm below the water surface to collect data at regular intervals. All the transect data were collected within a 6-h period. We used ESRI ArcGIS to transfer the data into a Geographic Information System and then interpolated them to raster using the inverse distance option.

From the end of June to early September 2002, we collected daily water samples from Amberlea Creek using an ISCO integrative sampler (Model 6712, Teledyne Isco, Inc; Lincoln, NE) to monitor changes in the concentrations of total suspended solids (TSS) and total phosphorus (TP) in weekly composite samples. The sampler collected four 250 ml samples of water from the Amberlea site for a daily total of 1 l of water. At approximately monthly intervals, samples were collected with a van Dorn bottle at the Amberlea and Pine

Creek stations, and at the North and South sites; the open-water site (indicated as "Open" in Fig. 2), was sampled more frequently at biweekly intervals. All water-chemistry samples were kept in the dark at 5°C during transport back to the laboratory. Laboratory processing and analysis of water samples followed standard protocols (American Public Health Association 1992) and can be found in Chow-Fraser (2006).

Habitat and fish collection

One pair of large fyke nets (13-m and 4-mm bar mesh, 1.1 × 1.4-m front opening) were set at each of the North and South stations in Frenchman's Bay. Paired fyke nets were connected from front opening to front opening with a 7-m lead connected. Each net had two 2.5-m wings that were attached at a 45° angle to the front of the net. The nets were set at the 1-m depth contour and were oriented parallel to the *Typha* beds. Nets were fished for approximately 24 h on each of 8 days between 1 August 2001 and 1 November 2002. Fish species were identified using Scott and Crossman (1998). Lengths were recorded for a representative subset of each species (20 individuals in small and large size classes for each species) and were used in published length–weight regressions (Schindler et al. 2000) to generate biomass estimates.

We used data from two sources to assess changes in the fish community over time. First, from the middle 1980s, we used a published species list from Frenchman's Bay (Stephenson 1990). Stephenson collected fish data during the ice-free periods of 1985 and 1986 derived from monthly sampling in the first year and biweekly sampling in the second. Stephenson's (1990) fish were collected with a combination of hoop nets, fyke nets, minnow traps, backpack electrofishing, and seines. For comparison with Stephenson (1990), we classified the species captured in our study as juvenile or adult based on the length distribution for each species (Scott and Crossman 1998). Second, unpublished data for the 1990s were obtained from the Toronto Regional Conservation Authority (TRCA; unpublished data). These fish were collected with a boat electrofisher, primarily in July and August from 1991 to 2000 with the exception of 1995. Although the TRCA sampled a variety of sites in the Bay, we selected only transects that corresponded to the North and South stations of our study.

Ecological indices

We calculated two ecological indices to compare temporal and spatial trends in the water quality and fish community of Frenchman's Bay. Water Quality Index (WQI; Chow-Fraser 2006), a index developed from a dataset of 110 wetlands throughout the Great Lakes, calculates a single integrated score from 12 water quality variables for comparison of the North and South sites of the marsh. WQI scores can range from −3 to +3 across six categories of wetland condition: "highly degraded" (−3 to −2), "very degraded" (−2 to −1), "moderately degraded" (−1 to 0), "good" (0 to 1), "very good" (1 to 2), and "excellent" (2 to 3). The Wetland Fish Index (WFI; Seilheimer and Chow-Fraser 2006) ranks wetlands based on the species found there, where each species has a defined tolerance to wetland degradation. WFI scores range from 1 (low quality wetland) to 5 (high quality wetland) and can be calculated based on either species presence/absence (WFI (PA)) or species abundance (WFI (AB)). Expected WFI scores reported in Lake Ontario, Lake Erie, southern Lake Huron, and southern Lake Michigan range from 1.88 to 3.81 for the WFI (PA) and 1.59 to 3.61 for the WFI (AB, Seilheimer and Chow-Fraser 2006), respectively.

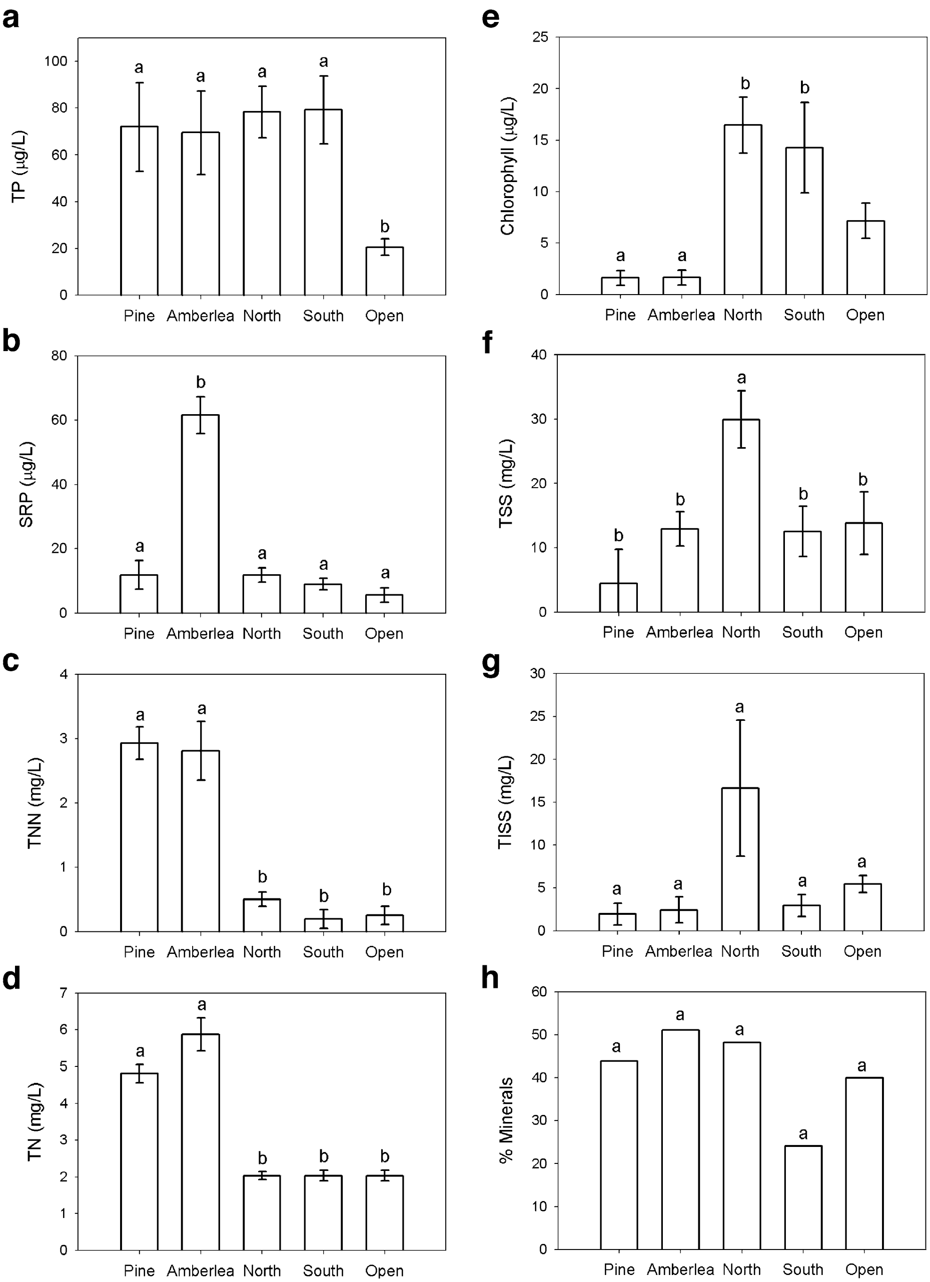


Fig. 2 Mean water quality parameters for all study sites. *Bars* are 1 S.E. of the mean. *P* values <0.05 indicate significant differences among sites as determined by ANOVA. *Bars* that share the same letter indicate that their means are statistically homogeneous as indicated by the Tukey–Kramer test. **a** *TP*=total phosphorus (P=0.017); **b** *SRP*=soluble reactive phosphorus (P=0.001); **c** *TNN*=total nitrate–Nitrogen (P< 0.001); **d** *TN*=total nitrogen (P<0.001); **e** *CHL*=chlorophyll-a (P=0.022); **f** *TSS*=total suspended solids (P= 0.021); **g** *TISS*=total inorganic suspended solids (P=0.079); **h** % minerals (P>0.1). *P* values are corrected with the Benjamini–Hochberg method and significant at an alpha of 0.05

Statistical methods

Statistical analyses were performed using SAS JMPIN, version 5.1 (SAS Institute Inc., Cary, North Carolina). When warranted, data were log10-transformed to normalize the data prior to conducting regression and correlation analyses. Analysis of variance was used to determine significant differences among means, and when appropriate, Tukey–Kramer tests were used to determine where significant differences occurred among pairs of means. The paired *t* test was used to compare water quality data from the continuous monitoring of creek and marsh sites and to compare changes in the fish community. "SE" refers to the standard error of the mean whenever it appears in this report. We used the Benjamini–Hochberg method to adjust the *P* values to account for the false discovery rate that is associated with multiple tests (Waite and Campbell 2006). All *P* values presented in this paper are adjusted with the Benjamini–Hochberg method and are significant at an alpha of 0.05.

Results

General water quality conditions

The water chemistry of Frenchman's Bay tributaries was distinct from that in the marsh. Summer mean temperatures for Amberlea and Pine creeks (17.6°C) were cooler than for the

Table 1 Physico-chemical measures for South marsh, North marsh, Pine Creek, Amberlea Creek for June to September 2002

	Month	South		North		*P* value	Pine		Amberlea		*P* value
		mean	SE	mean	SE		mean	SE	mean	SE	
Temperature (°C)	June	18.9	0.5	**20.5**	0.6	<0.001	**15.6**	0.3	15.1	0.5	0.006
	July	23.4	0.3	25.8	0.3	0.592	19.1	0.2	17.4	0.1	0.250
	Aug	23.5	0.3	**24.5**	0.4	<0.001	21.1	0.3	–	–	
	Sep	21.1	0.4	**24.4**	0.5	0.010	–	–	–	–	
Dissolved Oxygen ($mg \cdot l^{-1}$)	June	**9.8**	0.3	7.2	0.4	<0.001	2.4	0.5	**9.2**	0.1	<0.001
	July	8.4	0.2	6.0	1.0	0.393	4.8	0.5	**9.0**	0.1	0.006
	Aug	9.4	0.2	9.5	0.2	0.723	7.4	0.7	–	–	
	Sep	7.3	0.2	**8.3**	0.3	<0.001	–	–	–	–	
Conductivity ($mS \cdot cm^{-1}$)	June	0.371	0.00	**0.461**	0.01	<0.001	1.228	0.05	**1.838**	0.08	<0.001
	July	0.354	0.00	**0.424**	0.01	0.026	1.237	0.09	1.563	0.04	0.650
	Aug	0.332	0.01	**0.390**	0.01	<0.001	1.530	0.06	–	–	
	Sep	0.352	0.00	**0.446**	0.02	<0.001	–	–	–		
Turbidity (NTU)	June	8.4	0.04	**36.7**	5.45	0.004					
	July	8.6	0.03	11.9	4.01	0.153					
	Aug	8.7	0.04	4.6	1.36	0.758					
	Sep	8.1	0.04	**18.5**	3.98	0.032					
Chlorophyll a ($\mu g \cdot l^{-1}$)	June	26.4	1.55	**30.0**	1.50	<0.001					
	July	14.2	0.54	**21.4**	2.00	0.029					
	Aug	10.2	0.47	**18.3**	0.90	<0.001					
	Sep	8.8	0.65	**16.3**	0.90	0.001					

Significance was tested between South and North marsh sites and Pine and Amberlea creeks, separately. Bolded values are the higher value of the significant pairs. *P* values are corrected with the Benjamini–Hochberg method and significant at an alpha of 0.05

marsh sites (22.3°C; $P<0.001$ Table 1). Mean dissolved oxygen was also lower in the creeks (6.6 mg/l) than in the marsh sites (8.2 mg/l; $P<0.001$ Table 1), primarily due to low dissolved oxygen in Pine Creek during June and July. Conductivity showed the greatest difference in water chemistry, with creek values being nearly four times higher (1.479 mS/cm) than the marsh sites (0.391 mS/cm; $P<0.001$ Table 1).

There were similar differences between stream and marsh sites in nutrient and suspended solids measured. There were no significant differences among seasonal means of total

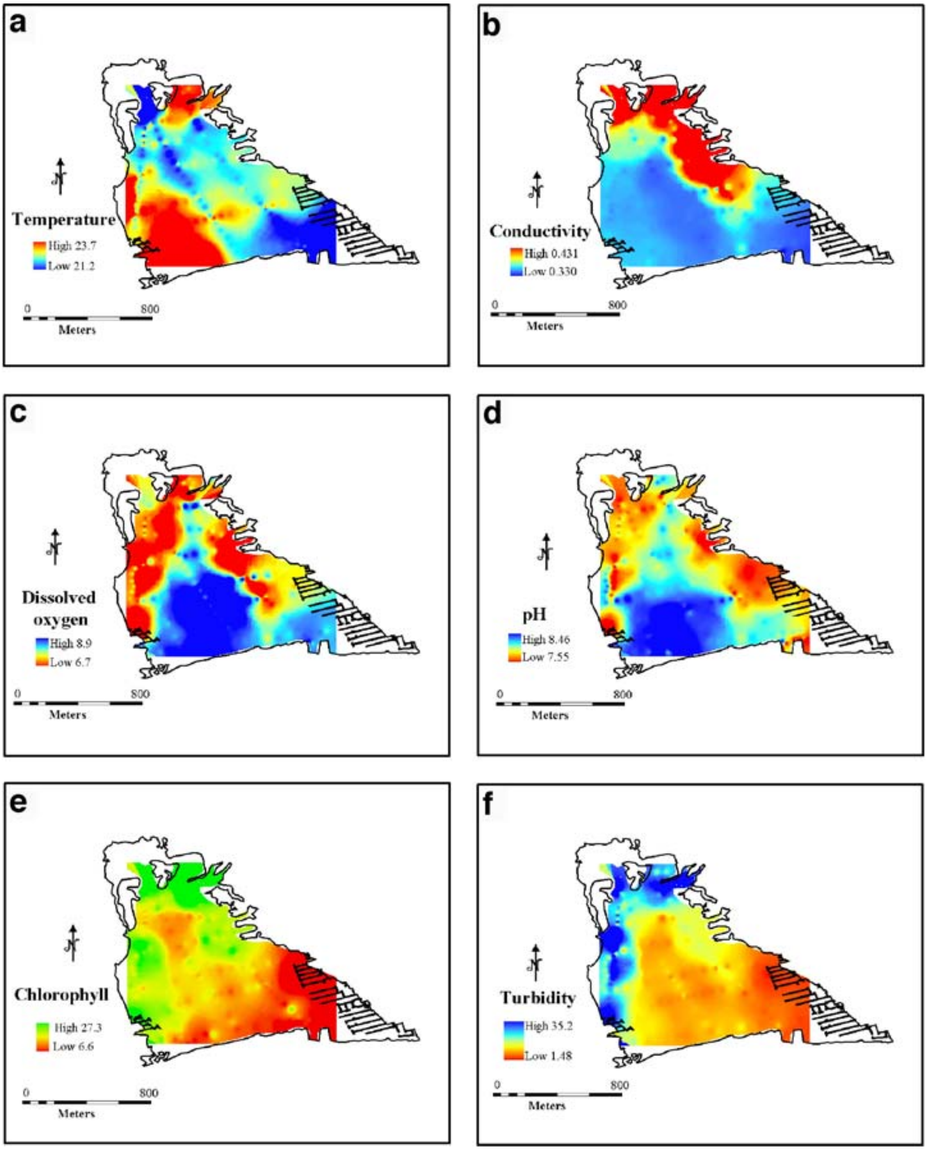
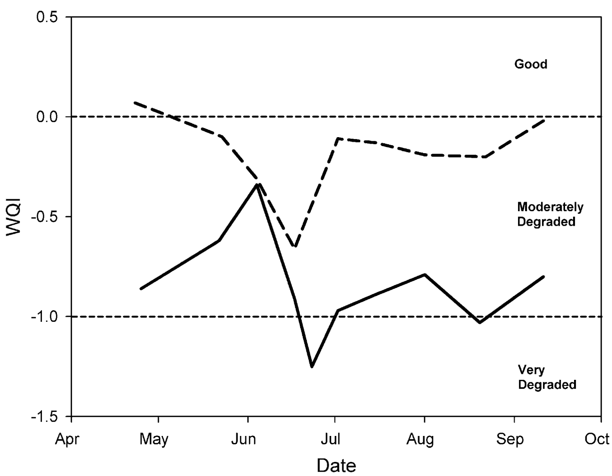


Fig. 3 GIS-rendered map showing changes in various physico-chemical characteristics of the water in Frenchman's Bay, collected August 20, 2002 with a YSI 6600 probe. **a** Temperature; **b** conductivity; **c** dissolved oxygen; **d** pH; **e** chlorophyll; **f** turbidity

phosphorus for the North, South, Amberlea and Pine creek sites, but all of these were significantly higher than that for the Open station (P=0.017, Fig. 2a). Soluble reactive phosphorus values for Amberlea were very high (about 60 μg/l; Fig. 2b), in contrast to Pine Creek and the marsh sites, which were lower and not significantly different from each other (P=0.001). Mean concentrations of total nitrate nitrogen (Fig. 2c) and total nitrogen (Fig. 2d) corresponding to both Amberlea and Pine creeks were significantly higher than those for the three marsh sites (ANOVA, Tukey–Kramer test P<0.001; Fig. 2c). Chlorophyll values were higher in the marsh sites than the creek sites, and for the shallow North and South sites than for the open water site (P=0.002, Fig. 2e). Because of the high seasonal variability in total suspended solid concentrations at both the creek and marsh sites, we found that only the North site differed significantly from the other sites (P=0.002; Fig. 2f). Total inorganic suspended solids were nearly tenfold higher at the North site, but this difference was not significant (P=0.079; Fig. 2g). It is noteworthy that except for the South site, where minerals (i.e., inorganic) comprised approximately 25% of the total suspended solids, all of the stations had percentages ranging between 40 and 50% (Fig. 2h). This means that the organic fraction (i.e. algae, protozoans and detritus) constituted a very large fraction of the suspended particles in both creek and marsh waters.

Monthly mean water temperature peaked at the South site in July and August, while the North site was warmest in July (Table 1). Water temperatures in the creek sites increased through the summer and were lower than the marsh sites in all months. Dissolved oxygen was generally high in the marsh sites and Amberlea Creek through the summer, although values as low as 0.81 mg/l were recorded at the North site and values below 2.00 mg/l occurred on 8 separate days. The minimum dissolved oxygen measurement at the South site was 3.52 mg/l, which occurred with the other four lowest measurements in the South site on a single day. Pine Creek's mean dissolved oxygen levels were lowest in June and increased until monitoring stopped in August (Table 1). Amberlea Creek had significantly higher DO values that Pine Creek in June and July (P<0.001 and P=0.006, respectively; Table 1). There was little variation in monthly mean conductivity for the marsh site but higher variability in the creek measurements. Conductivity was reduced by 0.5 to 1.0 mS/cm following rainfall (Fig. 5a), which contributed to the variability in the monthly means for creeks. Turbidity measurements in the South site were consistent across the surveyed months (8.1–8.7 NTU),

Fig. 4 Comparison of Water Quality Index (*WQI*) scores for North (*solid line*) and South (*dashed line*) sites in Frenchman's Bay during 2002



whereas the North site exhibited high values in June (36.7 NTU) followed by declines in July (11.9 NTU) and a mean that was lower than the South site in August (North 18.5 NTU and South 8.1 NTU, P=0.032; Table 1). Chlorophyll a values were highest in June, and generally declined through the summer (Table 1).

Spatial differences in marsh water quality

We used eight transects across Frenchman's Bay to characterize the spatial variation in August temperature, conductivity, DO, pH, chlorophyll and turbidity (Fig. 3). Water temperature at the Bay's southwestern and northeastern ends were the warmest, whereas water entering from Amberlea Creek was cooler and appeared as a plume extending through the marsh to the harbor entrance at the southeastern end (Fig. 3a). Areas with warmer water were quite shallow (less than 1 m deep) and tended to coincide with existing emergent vegetation beds near the North and South stations (see Fig. 1). Areas of higher conductivity at the northeastern portion of the Bay occurred where stream inflows from Amberlea, Dunbarton, and Pine creeks inflowed. Conductivities were reduced by 25% near the harbor entrance. Water close to the entrance also possessed higher oxygen content, as was true for water at the southwestern end and at the deep station (Fig. 3c). By comparison, water along the northern perimeter exhibited one third lower DO concentrations (Fig. 3c). Spatial patterns in pH followed those for DO and reflect the contribution of lower pH water

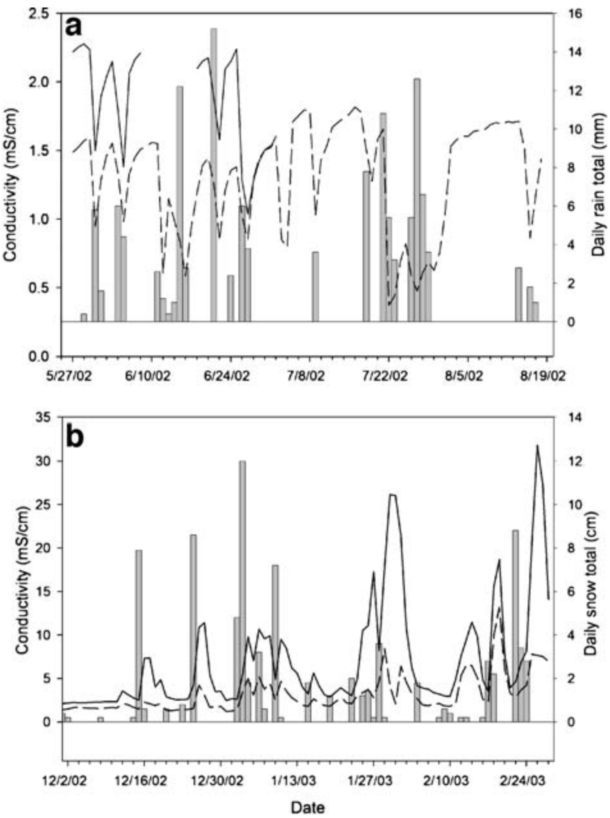


Fig. 5 Mean daily conductivity measurements for Amberlea Creek (*solid line*; downstream of Highway 401) and Pine Creek (*dashed line*; upstream of Highway 401) **a** summer May 27, 2002 to August 18, 2002 and **b** winter December 1, 2002 to February 28, 2003. Bars correspond to the daily rain totals (**a**; mm) and daily snow totals (**b**; cm) from the nearby Pickering area

from the creek at the Bay's northern end (Fig. 3d). There was an obvious northwest-to-southeast gradient in chlorophyll and turbidity values (Figs. 3e and f), which may reflect dilution from Lake Ontario water, or higher algal production in the shallow, vegetated near-shore areas in the northeast and southwest.

The patterning observed across the transects was confirmed by the North and South site continuous monitoring data (Table 1). Temperatures were significantly higher at the North site during 3–4 months (June $P<0.001$, July $P=0.592$, August $P<0.001$, September $P=0.010$; Table 1). Dissolved oxygen was significantly higher at the South site for June ($P<0.001$) and lower in September ($P<0.001$; Table 1). Conductivity was consistently higher at the North site for all months (Table 1). Turbidity was low in the South site throughout the summer (June $P=0.004$, September $P=0.032$) but the differences were not significant in July and August ($P=0.153$, $P=0.758$, respectively; Table 1). Chlorophyll measurements were significantly higher in the North site in all months (June $P<0.001$, July $P=0.029$, August $P<0.001$, September $P=0.001$). The mean WQI scores for the South site (−0.18) were significantly higher than for the North site (−0.85; paired t test, $P=0.001$; Fig. 4), although the scores for both sites fall within the "moderately degraded" category (Chow-Fraser 2006).

Table 2 Total night captured during nine collection nights and mean number fish captured per fyke net for North marsh and South marsh sites in Frenchman's Bay in 2001 and 2002

Family	Scientific name	Common name	Occurrence (8 total nights)		Mean catch (/net /night)	
			North	South	North	South
Amiidae	*Amia calva*	Bowfin	5	1	3.1	0.3
Catostomidae	*Catostomus commersonii*	White sucker	3	1	0.4	0.3
Centrarchidae	*Ambloplites rupestris*	Rockbass	–	1	–	0.1
	Lepomis gibbosus	Pumpkinseed	6	6	8.0	17.9
	Micropterus salmoides	Largemouth bass	4	4	3.8	11.4
	Pomoxis annularis	White crappie	3	2	1.4	0.3
Clupeidae	*Alosa pseudoharengus*	Alewife	5	2	9.5	6.8
	Dorosoma cepedianum	Gizzard shad	3	–	4.0	–
Cyprinidae	*Cyprinus carpio*	Common carp	6	4	2.3	0.6
	Notemigonus crysoleucas	Golden shiner	3	2	7.1	5.0
	Notropis atherinoides	Emerald shiner	4	5	4.8	1.9
	Notropis hudsonius	Spottail shiner	4	3	4.3	2.9
	Pimephales notatus	Bluntnose minnow	6	5	7.9	8.0
	Pimephales promelas	Fathead minnow	5	2	1.3	2.0
Esocidae	*Esox lucius*	Northern pike	2	3	0.4	0.5
Gasterosteidae	*Gasterosteus aculeatus*	Threespine stickleback	3	3	7.0	6.9
Ictaluridae	*Ameiurus nebulosus*	Brown bullhead	8	8	19.4	7.1
	Noturus gyrinus	Tadpole madtom	2	2	0.4	0.4
Moronidae	*Morone americana*	White perch	1	–	0.3	–
	Morone chrysops	White bass	1	–	0.1	–
Osmeridae	*Osmerus mordax*	Rainbow smelt	1	1	0.1	0.1
Percidae	*Etheostoma nigrum*	Johnny darter	2	4	0.3	1.3
	Perca flavescens	Yellow perch	5	7	2.6	29.1
	Percina caprodes	Logperch	1	0	0.1	0.0
Percopsidae	*Percopsis omiscomaycus*	Trout-perch	1	0	0.1	0.0

Highway inputs and winter conditions

During the summer, precipitation tended to dilute stream conductivity (Fig. 5a). In contrast, winter precipitation greatly elevated conductivities because of the road de-icing salts as evidenced by the coincidental timing of the onset of snowfalls and the spikes in conductivity (Fig. 5b). For both creeks, there appeared to be a base value of 0.25 mS/cm, above which conductivities increased following snowfall events. Over the course of the winter (December to February) conductivity values ranged from 1.3 to >30 mS/cm. Snowfall events of greater than 2 cm were followed by peaks in the daily mean conductivity in both Amberlea and Pine creek (Fig. 5b). Figure 5b shows that the effect on conductivity was greater in Amberlea Creek (below Highway 401; Fig. 1) than in Pine Creek (above Highway 401; Fig. 1) following each snowfall event: the maxima in Amberlea Creek approached or exceeded 20 mS/cm (Fig. 5b), whereas they seldom exceeded 15 mS/cm in Pine Creek (Fig. 5b).

Winter conditions at the Open station were similar to summer values (collected on January 25, 2003). The TP was 16.22 μg/l, which is somewhat lower than the summer mean of 20.92 μg/l. Both chlorophyll and TNN values were similar to the summer means (7.40 vs

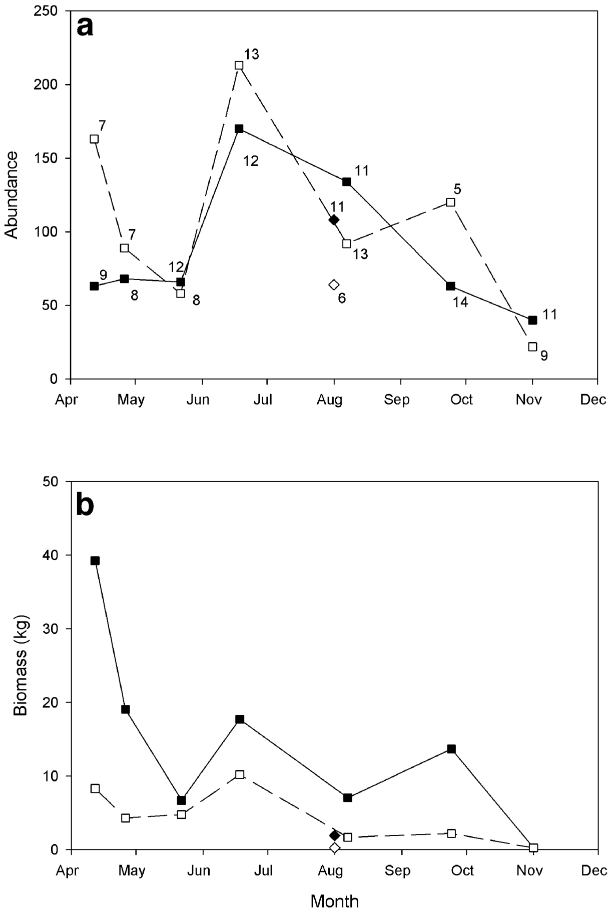


Fig. 6 Seasonal changes in **a** abundance and **b** biomass of fish caught at the North (*solid line* and *solid symbols*) and South (*dotted line* and *open symbols*) stations of Frenchman's Bay during 2001 (*diamond symbols*) and 2002 (*square symbols with solid and dashed lines*). Numbers next to symbols (a) correspond to species richness of sampling period

6.84 μg/l CHL, and 0.30 vs 0.25 mg/l TNN). Conductivity at the Open station in January was 0.320 mS/cm, which falls in the range of summer values (0.30 to 0.45 mS/cm). Overall, winter and summer levels of nutrients and chlorophyll at the Open station were very similar.

Current fish community

We surveyed the fish community in Frenchman's Bay on eight occasions between 2001 and 2002 and captured a total of 26 species from 12 families (Table 2). Although the total number of species captured varied from a low of 9 on April 26, 2002 to 17 on Aug 7, 2002, the numbers were very consistent during the May, June, August and September surveys (16 and 17; Fig. 6a). Brown bullhead, emerald shiner, and yellow perch were very common and occurred in almost every collection, including those in the early spring and late fall when species totals tended to be lower. The rare taxa (appearing in only one or two samples) included the rock bass, white perch, white bass and rainbow smelt (Table 2). Some fish tended to be caught only as adults, likely after having migrated into the marsh to spawn during the spring (white sucker and northern pike).

Since we sampled fish from both the North and South stations on every occasion, and we knew that there were significant differences between stations with respect to environmental conditions (Table 1; Fig. 2), we wanted to determine if there were any between station differences in species occurrence. We found 14 species for which abundance, biomass, or both differed significantly between stations ($P<0.05$, Paired T tests; Table 3). Six species occurred in greater numbers and/or biomass at the South station, including yellow perch, johnny darter, alewife, and largemouth bass (Table 2). Two other species were found primarily in the South station, northern pike and pumpkinseed, and were close to significant

Table 3 Summary of P values associated with paired t tests to determine significant differences between fish abundances and biomass data corresponding to North and South stations in Frenchman's Bay

Type	Species	Abundance	Biomass	TRCA data
South>North	Northern pike	0.114	0.493	Yes
	Yellow perch	**0.062**	**0.027**	Yes
	Bluntnose minnow	**0.002**	**0.004**	
	Johnny darter	**<0.001**	**0.0001**	
	Alewife	0.388	**0.030**	Yes
	Fathead minnow	0.079	**<0.001**	
	Pumpkinseed	0.094	0.1198	Yes
	Largemouth bass	0.080	**<0.001**	
	All taxa	**<0.001**	–	Yes
North>South	Brown bullhead	0.118	**0.050**	Yes
	Gizzard shad	**<0.001**	**<0.001**	
	White sucker	0.091	**0.023**	No
	Golden shiner	0.121	**0.002**	
	Bowfin	**<0.001**	**<0.001**	Yes
	Black crappie	**0.007**	**<0.001**	Yes
	Common carp	**0.045**	**0.064**	Yes
	Threespine stickleback	0.011	**<0.001**	
	All taxa	–	**<0.001**	Yes

Data were all log-transformed prior to analyses. Bolded numbers indicate that differences are significant ($P<0.05$). "Yes" indicates that data received from TRCA (Toronto Region Conservation Authority) are consistent with findings in this study, whereas "No" indicates that there was evidence to the contrary. P values are corrected with the Benjamini–Hochberg method and significant at an alpha of 0.05

(P=0.114 and P=0.094, respectively). Another eight were more abundant at the North station, including brown bullhead, gizzard shad, bowfin and common carp (Table 2). Pooling across our samples, we found that the South station catches yielded significantly greater numbers of fish (P<0.001; Fig. 6a), whereas the North station yielded a significantly higher biomass (P<0.001; Fig. 6b). This indicates that a greater number of small-sized individuals were caught at the South station, whereas fewer large-sized fish were caught at the North station. In most instances, both biomass and abundance data yielded similar seasonal trends; however, for gizzard shad, five adults contributed substantial biomass to the early spring sample, whereas the many juveniles present during summer contributed relatively little biomass, and thus gave rise to different seasonal patterns (Table 2).

The Toronto Region Conservation Authority had surveyed the fish community in Frenchman's Bay since 1991. Paired *T* tests of abundance and biomass data yielded results that were almost entirely consistent with our findings (Table 3). Fish were more abundant at the South station, but biomass was significantly higher at the North station (Table 3). Pumpkinseed, yellow perch, white sucker, alewife and northern pike were more numerous at the South station, whereas there were more brown bullhead, bowfin, black crappie, and common carp at the North station. The only discrepancy between the TRCA data and ours concerns the white sucker (Table 3), which the TRCA captured exclusively on the south side of the Bay whereas we captured them on three occasions in the North and only once at the South station (Table 2).

Differences between North and South sites were also found when comparing the WFI scores (Fig. 7). The WFI (PA) scores were higher in the South than the North throughout the summer of 2002, with the exception of June 17 (Fig. 7a). The mean WFI (PA) score for the South site

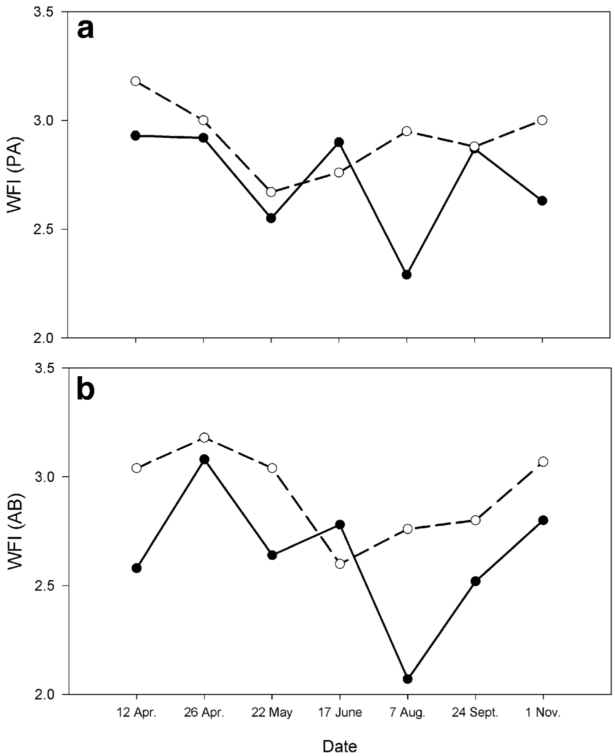


Fig. 7 Comparison of **a** Wetland Fish Index (*WFI*) (presence–absence (*PA*)) and **b** *WFI* (abundance (*AB*)) scores for North (*solid*) and South (*open*) sites in Frenchman's Bay during 2002

was higher (2.92) than that for North site (2.73; P=0.048 paired t test). The difference was even more pronounced when abundance data were used to generate WFI scores (WFI (AB)) of 2.64 vs. 2.93 for North and South, respectively (P=0.043; paired t test).

Comparison to historic fish community

Using a combination of existing datasets from Frenchman's Bay, we were able to make comparisons fish community (Table 4). There were 15 species that occurred across all three studies from the 1980s to the early 2000s and represented the most common species in this study (e.g., pumpkinseed, brown bullhead, yellow perch; Table 2). The remaining species

Table 4 Comparison of the species captured in this study versus studies conducted in the 1980s (Stephenson 1990) and 1990s (Toronto Regional Conservation Authority)

Common name	Scientific name	This study 2001–2002	Stephenson 1985–1986	TRCA 1991–1994	TRCA 1996–2000
Bowfin		J A	A	X	X
Brook silverside	***Labidesthes sicculus***		**A**		**X**
White sucker		J A	J A	X	X
Rockbass		A			
Pumpkinseed		J A	J A	X	X
Smallmouth bass	***Micropterus dolomieu***			**X**	**X**
Largemouth bass		J A	J	X	X
Black crappie		J A	J A	X	X
Alewife		J A	J	X	X
Gizzard shad		J A	J A	X	X
Goldfish	***Carassius auratus***			**X**	**X**
Common carp		J A	A	X	X
Emerald shiner		J A	A	X	X
Golden shiner		J A		X	X
Spottail shiner		J A	A	X	X
Bluntnose minnow		J A	A	X	X
Fathead minnow		J A	A		
Common shiner	***Luxilus cornutus***			**X**	**X**
Creek chub	***Semotilus atromaculatus***		**J**		
Northern pike		J A	J	X	X
Threespine stickleback		A			
Brown bullhead		J A	J A	X	X
Tadpole madtom		A			
White perch		A	J A	X	X
White bass		A			
Rainbow smelt		J A			
Johnny darter		A		X	X
Logperch		A			
Yellow perch		J A	J A	X	X
Trout-perch		A			
Freshwater drum	***Aplodinotus grunniens***			**X**	**X**
Central mudminnow	***Umbra limi***		**J**		
Number of species		25	19	21	22

A Adult fish captured; *J* juvenile fish captured

Species in bold were not captured during this study (2001–2002). TRCA dataset did not include life stage, so species presence is indicated with an X that we found were of low abundances or were captured only as adults, with the exception of the threespine stickleback (111 captured in this study) and golden shiner (97 captured in this study and TRCA both time periods). Stephenson (1990) found 19 species in the 1980s, including three not captured in this study. Two of these (creek chub *Semotilus atromaculatus* and central mudminnow *Umbra limi*) were captured by Stephenson only as juveniles. The brook silverside *Labidesthes sicculus* was captured only as an adult in the 1980s, but was also captured in the late 1990s (TRCA; Table 4). Seven species were captured only as adults. Both juveniles and adults were caught for seven species. Five species were captured only as juveniles. The TRCA collected 21 species in annual surveys from 1991 to 1994 and 22 species from 1996 to 2000 (Table 4), five of which were absent from our surveys but were generally present previously.

Discussion

Frenchman's Bay has a long documented history of degradation, evidenced by both sediment cores and paleolimnology. Eyles et al. (2003) used a wide range of geophysical techniques to determine the nature of bottom sediments and the distribution of contaminated sediment in the lagoon. From core analyses, they concluded that the Bay is no more than 3,000 years old, and that the first signs of human impact from the watershed occurred about 1,000 years ago. The Bay was relatively pristine for the next 800 years, as indicated by large amounts of laminated marl (fine sediment enriched in $CaCO_3$), has been typical of protected lagoons having clear water and extensive areas of submergent aquatic macrophytes such as *Chara* that secretes CaCO3 to form the chalky deposit (Eyles et al. 2003). The marl deposition was abruptly replaced circa 1840 by a layer of black-colored, foul-smelling mud rich in wood debris, partially decomposed organic matter, and the pollen of grass and weeds; this layer is referred to in other studies as the "European Settlement Layer', and indicates the onset of European settlement and the widespread forest clearances and subsequent soil erosion that followed (Weninger and McAndrews 1989). Reinhardt et al. 2005 used microfauna in sediment cores and the magnetic susceptibility of sediment layers to demonstrate the gradual occurrence of eutrophication following European settlement due to agriculture and increased nutrient loading and then the major eutrophication from urbanization and lawn runoff has occurred following the 1950s.

A primary source of contaminants to the system is runoff from Highway 401, a major transportation corridor located immediately upstream of Frenchman's Bay (Fig. 1). During the winter, large volumes of de-icing salt are applied after every snowfall, resulting in pulses of elevated conductivity. Similarly, Koryak et al. (2001) found a 20- to 30-fold increase in conductivity associated with winter thaws in Nine Mile Run, an urban stream near Pittsburgh, Pennsylvania. Summer precipitation causes declines in conductivity through dilution (Gray 2004; Koryak et al. 2001). These summer decreases and winter increases in conductivity following precipitation suggest that a relatively small increase in impervious surface area in the watershed can yield heightened stream conductivity (Walsh et al. 2005). Highway run-off can also contain elevated levels of such heavy metals as cadmium, copper, lead and zinc (Marsalek and Ng 1989; Gray 2004), which has been linked to degraded aquatic diversity and health (Maltby et al. 1995). Heavy metal contamination in the Bay is a recent phenomenon with sediment cores showing elevated levels of heavy metals only in the upper portion of the core that date back to the 1960s (Eyles et al. 2003). Currently, many parameters such as Total Keldahl Nitrogen, phosphorus, cyanide, oil and grease and total organic carbon exceed Ontario Ministry of Environment Provincial Sediment Quality Guidelines (Eyles et al. 2003). The water quality in open-water areas of Frenchman's Bay is likely helped by the permanent channel to Lake Ontario, which allows the more oligotrophic Lake Ontario water to mix with Bay water. Without the channel, the waters of the Bay would likely show increases in nutrients, conductivity, and temperature through the summer resulting in further degradation.

The composite transect maps (Fig. 3) indicate that the three marsh stations exhibit fairly distinct characteristics. The Open station was deep (about 3.0 m), with warm surface water, lower conductivity, high DO content, and low chlorophyll and turbidity. This description is consistent with the lower nutrient and suspended solids data reported for this station (Fig. 2). In contrast, the North station was shallow and warm, and seemed to be the most polluted of the three sites, with high conductivity, high chlorophyll and turbidity, and relatively low oxygen content. Like the North station, the South station was also shallow and warm, but was well-oxygenated and exhibited lower conductivity, chlorophyll and turbidity levels. Continuous hourly monitoring of the North station confirmed that the marsh can undergo periods of anoxia at night. The seasonal mean of 5.7 mg/l at the North station is an oxygen level low enough to stress certain fish species during mid-to-late summer. On the other hand, the mean of 9.22 mg/l for the South station is not likely a problem.

Non-point pollution associated with urbanization and yard and garden runoff can provide high nutrient loads to streams and wetlands (Clinton and Vose 2006; Reinhardt et al. 2005), that can subsequently impact fish habitat. Our total nitrogen and TNN concentrations show that Pine and Amberlea creeks contributed nitrogen in both inorganic and organic forms to Frenchman's Bay. Most of the TP that enters Amberlea Creek and eventually discharges into Frenchman's Bay is in a form that is readily available for algal growth which, in association with high suspended sediment, results in lower light penetration and a reduced diversity and coverage of submergent vegetation (Chow-Fraser et al. 1998). Frenchman's Bay provides a valuable ecological service by quickly utilizing the nitrogen from the creeks, as shown by large difference between high creek concentration and lower marsh concentrations. Seilheimer and Chow-Fraser (2006) found a relationship between the species richness of submerged macrophytes and WFI score, and predict that a wetland with five submergent species (as we counted) should have a WFI (PA) score of 2.89. Accordingly, the mean WFI scores we observed in the marsh sites in 2002 ranged from 2.73 in the North to 2.92 in the South. Submergent vegetation provides critical spawning habitat for adults, refugia from predators, and habitat for prey of all life stages (Jude and Pappas 1992; Casselman and Lewis 1996).

An aerial photo taken of the marsh during 1999 (Environment Canada 2001) showed 47 ha of remnant emergent vegetation, down from the historical high of 68 ha found during 1960 (Williams and Lyon 1997). Part of this reduction may be attributed to an increase in water levels since the 1960s (Williams and Lyon 1997). However, the added disturbance from urban development along the shoreline together with increased sedimentation, has likely prevented their recovery now that water levels have receded. Since large intact stands of emergent vegetation are better at coping with the bioturbation associated with carp spawning and feeding (Lougheed et al. 1998; Chow-Fraser 1999) than are the small islands of vegetation that currently exist, restoring large contiguous stands like those that occupy the that characterized the northern portion of the Bay should be a priority. Degraded habitats are generally more apt to be invaded by exotic species (MacDougall and Turkington 2005). The aquatic plant community of Frenchman's Bay shows this (Wei and Chow-Fraser 2006), in that human disturbance has coincided with the invasion of purple loosestrife (*Lythrum salicaria*), which now colonizes large areas of the marsh and visually dominates the shoreline when it flowers in mid-summer. It is unlikely that native emergent species will displace the *Lythrum* without human assistance. The relationship between urbanization and wetland plants similar to Frenchman's Bay (decreased diversity and increased invaders) has been reported in other regions (New Jersey; Ehrenfeld and Schneider 1991).

The lower WFI score for the North site indicates that the fish community there is dominated by turbidity tolerant, benthivorous species. In comparison, the higher WFI score for the South site indicates that the fish community was dominated by less tolerant percid and centrachid species. The site-to-site variation is not unusual for Great Lake coastal wetlands (Lougheed and Chow-Fraser 2001; Seilheimer and Chow-Fraser 2006). Normally, more suitable fish habitats are found in areas that are protected from wind and wave-driven sediment resuspension, where aquatic vegetation flourish. Within vegetated areas, the associated water is usually cooler, less turbid, and better oxygenated throughout the day and night as compared to areas without vegetation, benefiting such species as northern pike and largemouth bass. In Frenchman's Bay, even though the water quality at the Open station is relatively good, there is insufficient suitable habitat for most of the wetland taxa because of the lack of emergent or submergent vegetation. In contrast, the marsh in the southwestern end of the lagoon is home to many more fish species during the summer and fall; heightened species richness was seen even in the remnant marsh at the Bay's northern end, although the species were more tolerant of degraded conditions. Relative to other wetlands in Lake Ontario, WFI scores for Frenchman's Bay were lower than average (South 2.92 and North 2.73). Seilheimer and Chow-Fraser 2006 report mean WFI scores for 24 wetlands of 3.09 and a range from 2.33 to 3.70. All the wetlands in Lake Ontario below 3.0 WFI score were all located in the western site of the Lake from Darlington to the Niagara Peninsula, including sites in Toronto and Hamilton. Regional impairment of fish habitat in western Lake Ontario will continue to persist with increasing development pressures.

The fish community structure of Frenchman's Bay appears to have been fairly stable over the past two decades (Stephenson 1990; TRCA, unpublished data; this study; Table 4). Habitat loss likely had large impacts on the fish community but the current regime of degraded water quality and habitat continues to impact the fish community. Spatial differences between the North and South sites as regards the dominant species have persisted since the 1990s, with higher abundances (e.g., juveniles) occurring in the South, and higher biomass (e.g., large benthivores) in the North. The historical seasonal trends in relative abundance of the dominant fish species persist (Fig. 2 in Stephenson 1990), with brown bullheads being of high abundance in early spring, followed by higher numbers of alewife in early summer, and the pumpkinseed in late summer. The only notable difference was that Stephenson found more yellow perch in the fall, whereas we found more in the spring. Two species absent from our extensive survey (brook silverside and central mudminnow) but present previously were indicators of good water quality in the WFI (Seilheimer and Chow-Fraser 2006). The loss of these two species may be due to further degradation since the 1980s. Aquatic community structure (e.g., macroinvertebrates and fishes) can be influenced by land use activities that occurred as much as 50 years earlier (Harding et al. 1998). The impacts of urbanization began 50 years ago (Reinhardt et al. 2005) have resulted in a fish community that has persisted for at least the last 20 years. Weaver and Garman (1994) found that long term exposure to urban stress reduced fish species diversity and abundance. Urbanized streams commonly have lower fish diversity, low ecological index score (e.g. index of biotic integrity), and fish with more frequent deformities (Helms et al. 2005; Wang et al. 2000). There is likely a synergistic effect acting on the fish community of Frenchman's Bay that combines the historic impacts (>50 years) with the current development in the water shed and from Highway 401 inputs.

Proper study of these ecosystems will require knowledge about ecological interactions within the marsh, as well as factors external to the marsh, such as changes in seasonal patterns of stream flow, physicochemical characteristics of source streams, and the type and amounts of pollutants that enter water courses from surface runoff. This is especially important considering that the combined effect of climate change and predicted land-use alteration in settled areas of the Great Lakes basin will likely increase surface runoff from the current 17% (calculated for 1994–2003) to 21% (calculated for 2090 to 2099, Barlage et al. 2002). We also need to include more studies that document long-term changes in wetland response to landscape alteration (e.g. Cootes Paradise Marsh in Hamilton, Chow-Fraser et al. 1998).

The importance of wetlands in the Great Lakes has only recently begun to be understood and there is little information available regarding urban impacts on Great Lake wetlands. In addition, this is one of the first studies on the impacts of urbanization on wetland fish habitat. The Frenchman's Bay fish community has been degraded by urban development. Improvement of tributary water quality could improve in fish habitat quality and enhance and the fish community. Fostering juvenile survival through improved habitat would increase the potential productivity of the fish community. A healthier wetland will also improve recreational and aesthetic opportunities for human residents in the area but tradeoffs are expected to occur between competing human and ecological functions (e.g., hiking results in reduced plant diversity) in urban wetlands (Ehrenfeld 2004). The three step approach to urban wetland management proposed by Zedler and Leach (1998) may be the most effective strategy: allow passive recreation (which encourages public support for the second and third steps), restore wetland habitat, and determine the mechanism underlying restoration activities and urban impacts influence the wetland.

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