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# Detecting changes in ecosystem quality following long-term restoration efforts in Cootes Paradise Marsh

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ABSTRACT

Cootes Paradise Marsh is a large urban wetland of western Lake Ontario that has undergone major restoration as part of the Hamilton Harbour Remedial Action Plan. A key component of the restoration plan is exclusion of common carp (*Cyprinus carpio*) via construction of the Cootes Paradise Fishway that became operational in 1997. Here, we evaluate the response of the marsh community to carp exclusion using three approaches. First of all we analyze changes in water quality parameters and the community composition of zooplankton, macrophytes and fish. Secondly, we use ecological indices based on water quality, zooplankton, macrophytes and fish communities to track changes in quality. Lastly, we evaluate changes in the wetland quality of Cootes Paradise over the past decade in comparison with two other coastal wetlands of the Laurentian Great Lakes for which long-term data exist (Matchedash Bay of Lake Huron and Long Point Marsh of Lake Erie). Our results show that there has been variable improvement in wetland quality at Cootes Paradise, but compared to the two other wetlands, it is still the most degraded in all aspects studied. The overall trend towards moderately better water quality conditions in Cootes Paradise over the past decade is not directly reflected in the zooplankton, macrophyte and fish communities. We believe that high nutrient levels and high turbidity are preventing the progression to a clear-water macrophyte dominated system. This is one of few long-term studies that tracks the progress of restoration in a degraded marsh. It underscores the difficulty in trying to restore a 'novel ecosystem' to its original biotic and abiotic characteristics.



## 1. Introduction

Cootes Paradise (CP) Marsh is a large (250-ha) drowned river-mouth marsh, located in Hamilton Harbour at the extreme western end of Lake Ontario (Lougheed et al., 1998). At the turn of the twentieth century, over 90% of this wetland was covered with diverse vegetation, however the marsh had receded to less than 15% cover by the 1990s (Chow-Fraser et al., 1998). Coinciding with the vegetation decline, the fishery shifted from an important warmwater fishery of northern pike and largemouth bass to one dominated by planktivorous and benthivorous species such as bullheads and invasive common carp and alewife (Chow-Fraser, 1998). Despite this degradation, Cootes Paradise still provides a valuable stopover for migratory waterfowl and wetland birds (Smith and Chow-Fraser, 2010) and remains a major fish nursery for Lake Ontario (Holmes, 1988).

The degradation of CP can be attributed to a number of human activities. Agricultural and urban development of the previously forested watershed coupled with discharge of sewage effluent into the marsh for over nine decades (Chow-Fraser et al., 1998) has resulted in hypereutrophic conditions (Kelton et al., 2004). In addition, common carp (*Cyprinus carpio*), an exotic species introduced into Lake Ontario at the end of the 20th century, became established in CP, accounting for more than 90% of the fish biomass by the 1990s (Lougheed et al., 2004). Both the spawning (Lougheed et al., 1998) and feeding behaviours (Chow-Fraser, 1999) of the common carp accounted for up to 35–40% of the overall water turbidity in CP (Lougheed et al., 2004). These factors together contribute to some of the most turbid water conditions in a Great Lakes coastal wetland (Chow-Fraser, 2006), and prevent sufficient light penetration to support the growth of submersed aquatic vegetation, which is a critical component of the fish (Chow-Fraser, 1998; Randall et al., 1996) and waterfowl habitat (Prince et al., 1992). High turbidity has many other detrimental effects throughout all trophic levels, such as reducing light penetration to a level that is insufficient for periphyton growth (Newcombe and Macdonald, 1991), clogging filter-feeding structures of invertebrates (Kirk, 1991), and affecting the behaviour and survival of visually hunting predators and mating fish (Miner and Stein, 1993).

In 1985, the International Joint Commission designated Hamilton Harbour (HH) as one of 43 Areas of Concern (AOC); the harbour

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ecosystem had all fourteen “beneficial use impairments,” one of which was “loss of fish and wildlife habitat” (Hamilton Harbour Remedial Action Plan Stage 2, 1992). The largest remaining spawning and nursery habitat in the HH ecosystem is CP, and therefore a large component of the HHRAP is to restore the fish and wildlife habitat in the marsh. Thus, the HHRAP included a carp exclusion program via installation of the Cootes Paradise Fishway at the CP inlet into HH, which takes advantage of the natural migration of fish into and out of the marsh during the spring and fall (Lougheed and Chow-Fraser, 2001). The Fishway physically excludes large fish (e.g. mature breeding-age carp) from moving between the marsh and the harbour and captures them for sorting, at which point the carp are returned to HH (Wilcox and Whillans, 1999). CP is in a steady turbid-phytoplankton dominated state and it was predicted that the exclusion of common carp from the marsh would cause a shift to its former clear-macrophyte dominated state, as has been observed in many systems (Ibelings et al., 2007; Madgwick, 1999; Moss et al., 1996).

Although other restoration strategies have been implemented in addition to carp exclusion (e.g. nutrient reduction, marsh planting program), operation of the Fishway was the only one that was expected to initiate a switch to a clear-macrophyte dominated state. For this reason, we have chosen the initial date of Fishway operation as the starting point to evaluate the effectiveness of remedial actions on the overall quality of the ecosystem. In this study, ecosystem quality refers to biodiversity (species richness, diversity of functional groups in the ecosystem), presence of non-native species, and presence and abundance of known pollutant-tolerant and generalist species.

Our primary goal is to determine how ecosystem quality has responded to biomanipulation. We will use three approaches to evaluate this response. First, we will track how water quality parameters, zooplankton, macrophyte and fish species have changed from before Fishway operation to at least a decade afterwards. Secondly, we will use these measurements to calculate ecological indices developed specifically for use in Great Lakes wetlands which will provide a long-term trend to gauge the overall progress in marsh restoration. The four indices we will use are the Water Quality Index (WQI; Chow-Fraser, 2006), the modified Wetland Zooplankton Index (WZI; Lougheed and Chow-Fraser, 2002; Yantsis, 2009), the Wetland Macrophyte Index (WMI; Croft and Chow-Fraser, 2007), and the Wetland Fish Index (WFI; Seilheimer and Chow-Fraser, 2006, 2007). Lastly, to interpret the response of CP to restoration efforts in context of other coastal wetlands in the Great Lakes basin, we will compare changes with those in two other wetlands: Matchedash Bay (MB) and Long Point (LP) Marsh (location shown in Fig. 1, see Section 2.1 for site descriptions).

## 2. Materials and methods

### 2.1. Study sites

The three wetlands in this study occur along the coast of the Great Lakes (Fig. 1) and were all sampled between May and August from 1993 to 2009. CP is a windswept turbid system that is dominated by open water with a fringe of emergent and floating vegetation occurring along the shore and in two embayments. We sampled along the length of the wetland, including open-water sites located near the eastern outflow (close to the Fishway (CP1)), sites close to vegetation in the two embayments (Mac Landing (CP10) and Westdale Cut (CP16)), in a lagoon (West Pond (CP5)) near the outfall of the sewage treatment facility, and within the outflow of the main tributary, Spencer’s Creek (CP4) (see Fig. 1 inset). Many of the long-term sampling stations established in Chow-Fraser et al. (1998) have been retained in this study.

The second wetland, Matchedash Bay (MB) is one of the several bays in southern Georgian Bay, Lake Huron, in an area known as Severn Sound. Similar to CP, MB provides critical habitat for a diverse range of birds and wildlife (Sherman, 2002; Wilson and Cheskey, 2001) despite a moderately degraded state. Severn Sound was identified by the International Joint Commission as an AOC because of problems with nutrient enrichment (Sherman, 2002). A RAP was developed and implemented in 1989 to reduce nutrient inputs, and environmental conditions have improved sufficiently such that in 2003, Severn Sound was delisted as an AOC. MB is large and variable, and we have sampled at various sites within the wetland between 1998 and 2008. To ensure validity of the long-term comparison, we only use data that were collected at the same place and time during the year.

LP is one of the most extensive wild areas remaining in southwestern Ontario and offers a diverse range of habitats, particularly for migratory and resident species of waterfowl (Wilcox et al., 2003), wetland birds (Smith and Chow-Fraser, 2010) and fish (Mahon and Balon, 1977). Unlike the other two marshes, Long Point (LP) is a clear-water system with diverse assemblages of emergent, submergent and floating plants, and has many habitat types that vary in depth and degree of wind exposure (Thomasen and Chow-Fraser, unpubl. data). For the comparison of long-term changes, there were only suitable historic data collected near the Provincial Park between 1998 and 2008.

### 2.2. Data collection

Details of all protocols used for water sampling and processing can be found in many of the previous publications (e.g. Chow-Fraser, 2006; Lougheed et al., 1998). It is important to note that the data included in this study span 16 years, and had not been collected intentionally for this long-term evaluation. Therefore, not all variables are available on each sampling occasion. Nevertheless, all data were collected and processed with standardized protocols and methods, and were scrutinized with respect to temporal and spatial consistency. We also ensured that water quality data (see Table 1) collected during fair-weather conditions were used because storm events can greatly alter nutrient and sediment concentrations. Zooplankton samples were collected following the protocol outlined in Lougheed and Chow-Fraser (2002). Samples were collected from June to August and a minimum of three zooplankton samples were used to calculate an average WZI score based on the recommendations by Yantsis (2009). To analyze the relative distribution of zooplankton according to their taxonomic and functional roles, we grouped zooplankton according to their size and feeding guild as suggested by Lougheed and Chow-Fraser (1998) in their analysis of the zooplankton community in CP. Herbivorous rotifers included all rotifers except for *Asplanchna*, which were classified as a predaceous rotifer. Cladocerans were grouped according to size: micro-cladoceran (<300 $\mu$m), medium cladoceran (300–600 $\mu$m) and macro-cladoceran (>600 $\mu$m). Adult copepods were grouped together according to order (cyclopoids, calanoids, or harpacticoids) and nauplii and copepods were classified as immature copepods.

The aquatic plant community was surveyed between late June and August according to the methods of Croft and Chow-Fraser (2007). In this protocol, 10–15 quadrats in representative areas of each wetland were surveyed for the presence of submergent, floating, and emergent plant taxa. Since we were only interested in fish habitat, wet meadow species were excluded.

Both electrofishing and paired fyke nets (set overnight) were used to collect fish data for the WFI calculation. Chow-Fraser et al. (2006) revealed biases in fish abundances and sizes associated with these two methods. A subsequent study showed that such differences did not lead to differences in WFI scores (Kostuk, 2006). Boat

**Table 1**
Water quality parameters for Cootes Paradise from 1993 to 2008, used in the calculation of the Water Quality Index scores. Bolded *p*-values indicate a significant relationship between parameters and time. n/a = not available.

| Parameters | 1993 | 1994 | 1996 | 1998 | 2000 | 2002 | 2008 | $r^2$ | $p$ | Overall trend |
|---|---|---|---|---|---|---|---|---|---|---|
| Turbidity (NTU) | 63.4 | 63.0 | 55.7 | 33.0 | 37.8 | 28.0 | 35.9 | 0.6 | **0.04** | ↓ |
| TSS (mg/L) | 42.9 | 63.9 | 53.1 | 54.5 | 32.9 | 31.2 | 43.2 | 0.24 | 0.26 | ↓ |
| ISS (mg/L) | 27.2 | 43.5 | 39.1 | 31.1 | 15.5 | 17.6 | 24.0 | 0.34 | 0.17 | ↓ |
| Chl *a* (μg/L) | 18.4 | 87.8 | 24.6 | 92.1 | 71.4 | 22.1 | 29.0 | 0.05 | 0.65 | ↓ |
| TP (μg/L) | 158.7 | 241.7 | 200.0 | 269.5 | 159.3 | 189.0 | 141.9 | 0.19 | 0.33 | ↓ |
| SRP (μg/L) | 13.0 | 32.8 | 29.7 | 41.3 | 6.1 | 26.3 | 15.3 | 0.07 | 0.57 | ↓ |
| TN (μg/L) | 3599 | 4443 | 4402 | 4110 | 9992 | 5096 | n/a | 0.29 | 0.27 | ↑ |
| TNN (μg/L) | 1410 | 1104 | 485 | 1064 | 452 | 548 | 87 | 0.7 | **0.02** | ↓ |
| TAN (μg/L) | 182 | 237 | 440 | 268 | 745 | 559 | 146 | 0.01 | 0.85 | ↓ |
| Conductivity (μs/cm) | 757 | 815 | 683 | 771 | 668 | 518 | 572 | 0.64 | **0.03** | ↓ |
| Temperature (°C) | 25.4 | 22.6 | 25.8 | 22.8 | 24.3 | 23.6 | 24.9 | 0.002 | 0.92 | ↑ |
| pH | 8.30 | 7.98 | 7.08 | 8.17 | 8.61 | 8.07 | 8.04 | 0.02 | 0.76 | ↑ |
| WQI score | −2.33 | −2.20 | −1.72 | −2.09 | −1.82 | −1.56 | −1.39 | 0.72 | **0.02** | ↑ |

electrofishing (see protocol in Lougheed et al., 2004) was conducted in May of 1996, 1997, and 1998, whereas paired fyke nets (see protocol in Seilheimer and Chow-Fraser, 2006) were used from May to early August in 2001–2009. We compared May 1996 to May 2008 using the Mann–Whitney Test (see Section 2.4) to control for seasonal effects. We also used Scott and Crossman (1998) to classify the fish based on species and life stages into the following functional feeding guilds: planktivorous (consuming primarily zooplankton), piscivorous, omnivorous (consuming both algae and zooplankton), herbivorous (consuming algae and plant material), carnivorous (consuming insects and other invertebrates) or benthivorous (consuming primarily benthic invertebrates and other sediment-associated organisms).

### 2.3. Description of ecological indices

The four indices we have chosen were developed specifically for coastal marshes in the Great Lakes. They are based on the premise that anthropogenic activities cause degradation in water quality and thus lead to dominance of plants, zooplankton and fish that reflect the degraded state. Operationally, the indices can be interpreted as being indicative of high-quality ecosystems when the values are high and of degraded ecosystems when values are low. Each biotic index reflects conditions of one trophic level in the ecosystem. Taken together, the abiotic index as well as biotic indices, contribute information regarding the overall condition experienced by the marsh community within the water column.

Chow-Fraser (2006) developed the abiotic index called the Water Quality Index (WQI), which ranks the degree of water quality impairment based on 12 environmental parameters (i.e. major nutrients, suspended solids, chlorophyll concentrations, and physical characteristics – see Table 1 for a complete list) collected from 110 wetlands throughout the Great Lakes. Chow-Fraser (2006) used Principal Components Analysis to create an index yielding a score ranging from −3 to +3 which indicates the effect of human-induced land-use alterations on wetland quality. WQI scores were significantly related to land-use alteration and sensitive to human-induced degradation of water quality in coastal wetlands. Chow-Fraser (2006) considered all sites

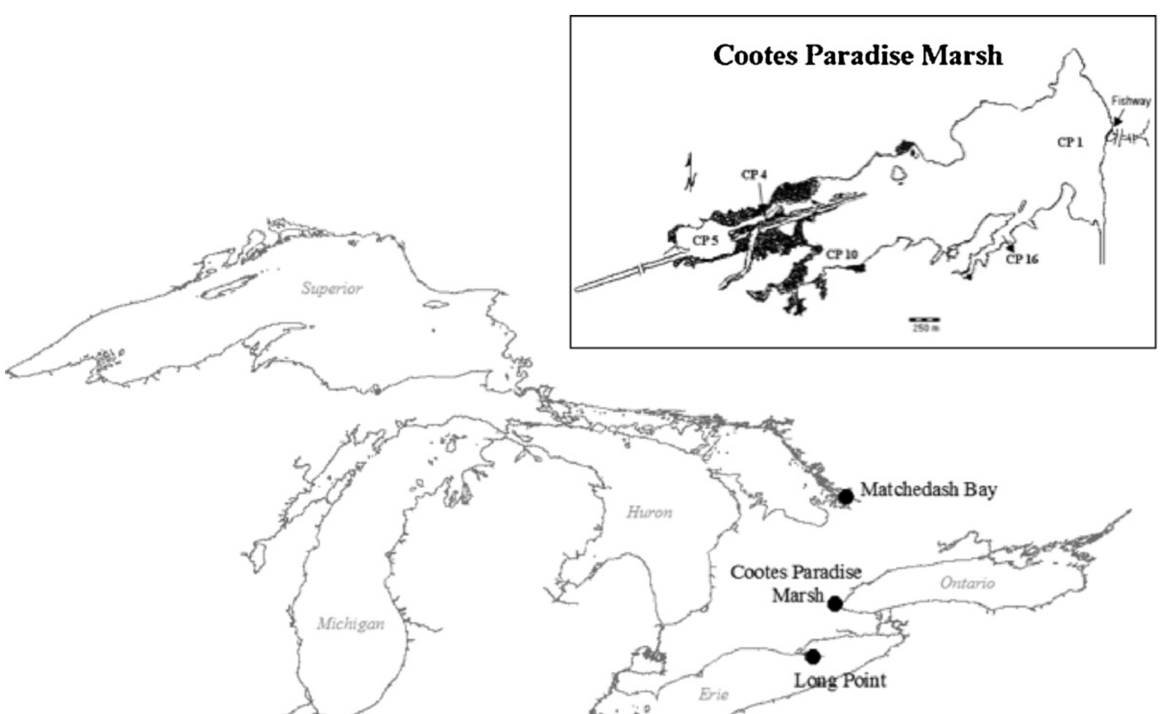


**Fig. 1.** Location of study wetlands, Cootes Paradise, Matchedash Bay and Long Point. Inset shows locations of sampling sites in Cootes Paradise (sites 1, 4, 5, 10 and 16).

associated with scores below zero to be degraded by human activities.

The three biotic indices (WZI, WMI and WFI) may be thought of as surrogates of the WQI. Lougheed and Chow-Fraser (2002) found that certain zooplankton taxa tended to be associated with clear water, macrophyte-dominated sites while others were associated with turbid, algal-dominated sites, and used this information to develop the WZI for 70 wetlands in the Great Lakes basin, excluding the pristine wetlands of eastern Georgian Bay. Yantsis (2009) modified the WZI to integrate the high-quality sites of Georgian Bay and made it more robust. The WMI was developed by Croft and Chow-Fraser (2007) who related the presence of aquatic plant species to water quality conditions in 127 wetlands from all five Great Lakes. Similar to the WMI, the basis of the WFI, developed by Seilheimer and Chow-Fraser (2006), is that the degree of water quality impairment is reflected in the taxonomic composition of the fish community. The WFI was initially developed from 40 wetlands located primarily in Lakes Ontario, Erie and Michigan and was later updated when Seilheimer and Chow-Fraser (2007) included 60 additional wetlands from Lakes Superior and Huron (including pristine sites of eastern Georgian Bay) so that the WFI could be applied to all five Great Lakes.

Canonical Correspondence Analysis was used in all three biotic indices to quantify the relationship between species occurrence and environmental variables in the form of species-specific values denoted as U and T values. First, U values (whole numbers ranging from 1 to 5) were assigned based on species' position on the synthetic degradation axis, where 1 was assigned to those that were most tolerant to degradation, and 5 to those that were least tolerant. Secondly, T values (whole numbers ranging from 1 to 3) were assigned according to the weighted standard deviation of the species scores along the axis of degradation, where 1 was assigned to species with a wide niche breadth, and 3 to those with narrow niche breadth. An index score is then calculated with respective U and T values in the following Eq. (1):

$$\text{Index score} = \frac{\sum_{i=0}^{n} Y_i U_i T_i}{\sum_{i=0}^{n} Y_i T_i} \tag{1}$$

where $Y_i$ is the presence or abundance ($\log_{10}(x+1)$) of species $i$. The index scores can range from 1 to 5, with higher values indicating better wetland quality. The U and T values for the species in this study are listed in Tables 2–4. We refer readers to the original published papers for more details on the index development and application.

### 2.4. Statistical analysis

We first used a simple linear regression to detect significant changes over time for each ecological index. We also employed the Mann–Whitney Test, also called the Wilcoxon–Mann–Whitney Test, which is a non-parametric method equivalent to the *t*-test (Zar, 2010), to compare mean scores before and after the biomanipulation. To test our hypothesis that the indicator scores based on Cootes Paradise data will be significantly lower than those for Long Point Marsh and Matchedash Bay, we used the Kruskal–Wallis single factor analysis of variance by ranks, which is the non-parametric equivalent of an analysis of variance. To determine significant group-to-group differences, we employed the post hoc non-parametric Tukey-type multiple comparison test, called the Nemenyi Test. All of the non-parametric tests we used are based on ranking the values, as opposed to the absolute values and were employed based on Zar (2010). We set the significance level to $\alpha = 0.05$ for all tests.

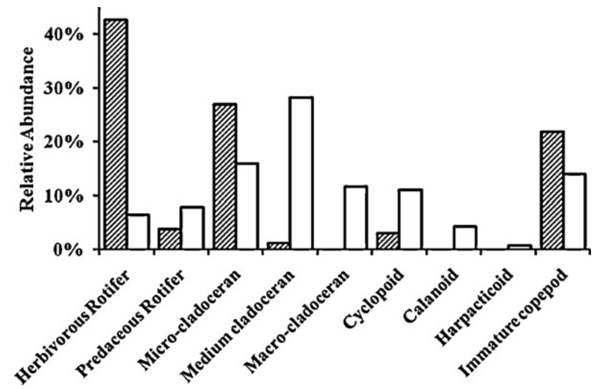


**Fig. 2.** Relative abundance of functional feeding categories for the Cootes Paradise zooplankton community in 1996 (hatched bars) and 2008 (open bars).

## 3. Results

### 3.1. Water quality and biotic communities

All water quality parameters except for TN, temperature and pH have followed a decreasing trend from 1993 to 2008 (Table 1). TN is the only nutrient that has not decreased over the study period, although we were unable to measure TN in 2008 (Table 1). Turbidity, conductivity and the nutrient TNN have significantly decreased (Table 1). Mean turbidity following exclusion (33.7 ± 4.3 NTU, 1998–2008) decreased by almost half of its original value during the pre-exclusion years (60.7 ± 4.3 NTU, 1993–1996).

The zooplankton community in 1996, the year prior to carp exclusion, was dominated by small-bodied organisms. Herbivorous rotifers, micro-cladocerans and immature copepods represented 92% of the relative abundance (Fig. 2). *Bosmina longirostris*, *Polyarthra* sp., and *Brachionus* sp. were the most common (Table 2). The zooplankton community in 2008 shifted to larger-bodied organisms and was more diverse and heterogeneous (Fig. 2). Medium cladocerans were the most common functional group (28%, Fig. 2), largely comprised of *Ceriodaphnia* sp. and *Diaphanosoma birgei* (Table 2). Overall the distribution of functional groups became more balanced, with the appearance of larger zooplankton such as macro-cladocerans and calanoid and harpacticoid copepods (Fig. 2). Although the absolute density of the cladocera zooplankton in 2008 was not nearly as high as in 1996 ($\sim$1600 L$^{-1}$ vs. $\sim$160 L$^{-1}$), species richness had increased from 20 to 23 (Table 2).

The number of macrophyte species in CP has fluctuated over the years, with the highest richness occurring in 1993 (20 species, Fig. 3) and the lowest in 2008 (10 species, Fig. 3). Relative to native species, there has been a relatively constant presence of non-native

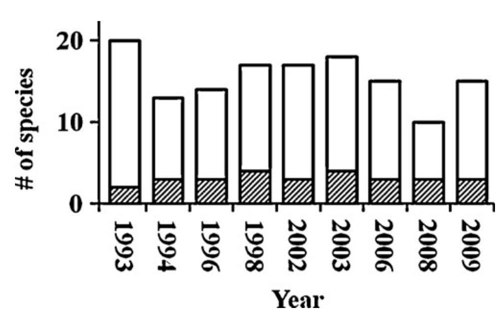


**Fig. 3.** Number of native (open bars) and non-native macrophyte species (hatched bars) at Cootes Paradise.

**Table 2**
Mean zooplankton density (# individuals/L) at Cootes Paradise from 1994 to 2008, used in the calculation of the Wetland Zooplankton Index scores. Dash (–) indicates no individuals found. (U, T) values used to calculate the WZI are listed after each species' name. *Note*: Species without (U, T) values are not included in the WZI.

| | 1994 | 1996 | 1997 | 1998 | 2008 |
|---|---|---|---|---|---|
| Copepoda | | | | | |
| Calanoid | 0.6 | 2.9 | – | 0.2 | 1.0 |
| Cyclopoid | 44.2 | 55.4 | 34.3 | 161.0 | 83.8 |
| Harpacticoid | – | – | – | – | 0.3 |
| Copepodid | 31.2 | 61.7 | 92.4 | 238.4 | 10.3 |
| Nauplii | 359.6 | 515.2 | 199.0 | 362.5 | 79.3 |
| Cladocera | | | | | |
| *Alona* sp. (4, 2) | – | – | 2.0 | – | 1.0 |
| *Bosmina* sp. (3, 2) | 414.4 | 578.9 | 15.8 | 66.6 | 45.2 |
| *Bunops serricaudata* (5, 3) | – | – | – | – | 0.2 |
| *Ceriodaphnia* sp. (4, 2) | 2.2 | 4.2 | 6.6 | 27.9 | 20.3 |
| *Chydorus* sp. (4, 2) | 12.3 | 6.4 | 40.6 | 67.7 | 25.3 |
| *Daphnia* sp. (2, 2) | 1.3 | 2.2 | – | 0.6 | 20.0 |
| *Diaphanosoma birgei* (1, 2) | 0.7 | 0.5 | 0.1 | 1.0 | 6.6 |
| *Diaphanosoma brachyurum* (3, 1) | – | – | – | – | 0.2 |
| *Eubosmina* sp. (3, 2) | – | – | – | – | 1.0 |
| *Hexarthra* sp. (1, 1) | – | – | 1.3 | – | – |
| *Kurzia* sp. (3, 3) | – | 0.2 | 4.2 | – | – |
| *Leydigia* sp. (1, 1) | 0.2 | 3.4 | 0.1 | – | – |
| *Moina* sp. (1, 1) | 51.5 | 15.0 | 0.1 | 28.7 | 2.1 |
| *Ophryoxus gracilis* (5, 3) | – | – | – | – | 0.2 |
| *Pleoroxus* sp. (4, 2) | 1.1 | 0.6 | 14.9 | 9.0 | 3.4 |
| *Polyphemus pediculus* (5, 3) | – | – | – | – | 0.2 |
| *Scapholeberis* sp. (4, 2) | 13.5 | 32.5 | 0.3 | – | 8.9 |
| *Sida crystallina* (5, 3) | – | – | – | – | 3.8 |
| *Simocephalus* sp. (5, 3) | 0.4 | – | 9.4 | 39.3 | 0.8 |
| Rotifera | | | | | |
| *Ascomorpha* sp. (1, 1) | 45.1 | 180.4 | – | – | – |
| *Asplanchna* sp. (2, 1) | 41.0 | 59.3 | 8.9 | 21.7 | 6.4 |
| *Brachionus* sp. (2, 1) | 100.6 | 186.6 | 516.5 | 404.4 | 10.7 |
| *Conochilus* sp. | – | – | 1.1 | – | – |
| *Conochiloides* sp. (4, 2) | 2.5 | 37.8 | – | 118.5 | – |
| *Euchlanis* sp. (3, 1) | 0.3 | – | 5.1 | 12.2 | 0.2 |
| *Filinia* sp. (1, 1) | 7.1 | 23.5 | 3.9 | 4.5 | 0.5 |
| *Keratella* sp. (5, 2) | 13.0 | 45.8 | 1.9 | 13.4 | 0.5 |
| *Lecane* sp. (5, 2) | 1.4 | 25.2 | – | – | – |
| *Lepadella* sp. (4, 2) | – | – | 1.5 | – | – |
| *Monostyla* sp. (5, 2) | – | – | 52.4 | 48.9 | – |
| *Mytilina* sp. (5, 3) | – | – | 16.7 | 6.1 | – |
| *Notholca* sp. (3, 1) | – | – | 1.6 | – | 0.4 |
| *Platyias* sp. (4, 2) | 0.3 | 0.2 | 5.4 | 213.9 | 1.4 |
| *Ploesoma* sp. (4, 2) | – | – | 1.4 | – | – |
| *Polyarthra* sp. (3, 1) | 294.8 | 373.9 | 119.5 | 115.7 | – |
| *Pompholyx* sp. (1, 1) | 8.7 | 34.6 | – | – | – |
| *Testudinella* sp. (4, 2) | 0.5 | – | – | – | – |
| *Trichocerca* sp. (4, 2) | – | – | 2.4 | 6.1 | – |

species (*Myriophyllum spicatum*, *Potamogetan crispus*, *Lythrum salicaria*, *Typha angustifolia*, and *T. x glauca* – Table 3, Fig. 3). With the exception of 2006, the proportion of submergent species has increased since the Fishway became operational, and this appears to have been at the expense of the emergent vegetation (Fig. 4A). The proportion of floating species has remained relatively constant post carp exclusion (Fig. 4A).

The fish community has clearly changed since 1996, when benthivores dominated (47% common carp); the community following carp exclusion had <40% benthivores (<10% common carp) (Fig. 5A). In recent years, brown bullhead and bluntnose minnows have added to the diversity of the benthivore niche (Table 4). The proportion of other types of fish has also increased. For instance, the percentage of carnivores increased two to four-fold (17–22% to 40–90%) following exclusion (Fig. 5A). Piscivores, which had been noticeably absent prior to carp exclusion reappeared beginning in 1998, however they still do not represent a large proportion of the community (Fig. 5A). Due to sampling bias related to the different gear types used, no conclusions can be made regarding the absolute abundance of catch (e.g. 4110 white perch in 2002, Table 4).

### 3.2. Ecological indices

We calculated the annual mean WQI score from all CP sampling stations (Fig. 1 inset) to examine how water quality conditions have changed in the marsh from 1993 to 2008 and found that mean WQI scores have increased significantly over the 15 years (Fig. 6A, $r^2 = 0.72$, $p = 0.02$). We also divided the data into two periods, before and after operation of the Fishway in 1997, and found that the mean WQI score was significantly higher for the period following exclusion compared with the period before (Mann–Whitney, $p < 0.05$). Unlike the WQI, the WZI was not able to detect a significant improvement in CP through the 14 years (Fig. 6B, $r^2 = 0.04$, $p = 0.70$), and we did not find a significant difference between mean WZI scores before (1996) and after (2008) the Fishway operation (Mann–Whitney, $p > 0.05$). In addition, the WMI was unable to detect a significant trend in scores between 1993 and 2009 (Fig. 6C, $r^2 = 0.05$, $p = 0.58$); when we calculated mean WMI scores by grouping data before and after carp exclusion, we found no significant differences (Mann–Whitney, $p > 0.05$). Though not statistically significant, there has been a slight trend towards an increase in WFI scores from 1996 to 2008 (Fig. 6D, $r^2 = 0.25$, $p = 0.32$).

**Table 3**
Macrophyte species at Cootes Paradise from 1998 to 2008, used in the calculation of the Wetland Macrophyte Index scores. Presence indicated by "X". Asterisks indicate non-native species. (U, T) values used to calculate the WMI are listed after each species. *Note*: Species without (U, T) values are not included in the WMI.

| | 1993 | 1994 | 1996 | 1998 | 2002 | 2003 | 2006 | 2008 | 2009 |
|---|---|---|---|---|---|---|---|---|---|
| Floating | | | | | | | | | |
| *Lemna minor* (1, 1) | X | | | | X | X | X | | X |
| *Nuphar advena* (1, 3) | X | | | | | | | | |
| *Nuphar variegate* (2, 1) | X | X | | X | | | | | X |
| *Nymphaea odorata* (2, 1) | X | X | X | X | X | X | X | X | X |
| *Potamogetan natons* (2, 1) | | X | | X | | | | | |
| Submergent | | | | | | | | | |
| *Ceretaphyllum demersum* (1, 1) | X | | X | X | X | X | | X | X |
| *Chara* sp. (3, 2) | | | | | X | | X | | |
| *Elodea canadensis* (2, 1) | X | | X | X | X | | | X | X |
| *Myriophyllum spicatum** (1, 1) | | | X | | X | X | | X | X |
| *Najas flexilis* (3, 2) | | | | | X | X | | | |
| *Potamogeton crispus** (1, 1) | | | | X | | X | X | X | X |
| *Potamogeton foliosus* (2, 1) | | | | | | | X | | |
| *Potamogeton pusillus* (2, 1) | | | | | | | | | X |
| *Potamogeton* sp. (1, 2) | | | | | X | X | | | |
| *Stuckenia pectinata* (1, 1) | X | X | X | X | X | X | | X | |
| *Vallisneria americana* (3, 1) | X | | | | | X | | | |
| Emergent | | | | | | | | | |
| *Lythrum salicaria** (1, 1) | | X | X | X | X | | | X | |
| *Phragmites* sp. | X | X | X | X | | | | X | |
| *Polygonum amphibium* (1, 1) | X | | | | | X | X | X | X |
| *Pontederia cordata* (3, 2) | X | | | | | | | | |
| *Sagitaria latifolia* (2, 1) | X | | | | X | | X | | X |
| *Schoenoplectus* sp. (4, 1) | | X | X | X | | | | | |
| *Schoenoplectus validus* (4, 1) | X | | | | X | | X | | |
| *Sparganium androcladum* (4, 3) | | | | | | | | | X |
| *Sparganium eurycarpum* (3, 2) | X | | | | X | | X | | |
| *Typha angustifolia** (1, 1) | X | X | X | | | | | | |
| *Typha latifolia* (3, 2) | X | X | X | X | | X | X | X | X |
| *Typha x glauca** (1, 2) | X | X | X | | | X | | X | X |

**Table 4**
Absolute abundance of fish species at Cootes Paradise from 1993 to 2008, used in the calculation of the Wetland Fish Index scores. Dash (–) indicates no individuals found. Data from 1996 to 1998 correspond to boat electrofishing transects (# /100 $m^2$); data from 2001 to 2008 correspond to three sets of paired fyke nets. (U, T) values used to calculate the WFI follow each common name.

| | | 1996 | 1997 | 1998 | 2001 | 2002 | 2008 |
|---|---|---|---|---|---|---|---|
| Benthivore | | | | | | | |
| Bluntnose minnow (4, 2) | *Pimephales notatus* | – | – | 2 | 8 | 2 | 24 |
| Brown bullhead (2, 1) | *Ameiurus nebulosus* | 43 | 7 | 50 | 2 | 254 | 17 |
| Channel catfish (1, 2) | *Ictalurus punctatus* | – | – | – | 27 | – | – |
| Common carp (1, 1) | *Cyprinus carpio* | 90 | 5 | 24 | 1 | 35 | 4 |
| White sucker (3, 2) | *Catostomus commersonii* | – | 1 | – | – | – | – |
| Carnivore | | | | | | | |
| Black crappie (3, 2) | *Pomoxis nigromaculatus* | – | – | – | – | – | 1 |
| Bluegill sunfish (3, 1) | *Lepomis macrochirus* | 2 | – | 2 | 59 | 25 | – |
| Brook silverside (4, 2) | *Labidesthes sicculus* | 1 | – | – | – | – | – |
| Logperch (4, 2) | *Percina caprodes* | – | – | 6 | 1 | – | – |
| Pumpkinseed (3, 2) | *Lepomis gibbosus* | 32 | – | 94 | 56 | 33 | 34 |
| Smallmouth bass (4, 2) | *Micropterus dolomieu* | – | – | – | – | 1 | – |
| Spotfin shiner (1, 1) | *Cyprinella spiloptera* | – | – | – | 6 | – | – |
| Sunfish (3, 2) | *Lepomis* sp. | – | – | – | 106 | – | 2 |
| White bass (1, 1) | *Morone chrysops* | – | – | – | 34 | – | – |
| White perch (1, 2) | *Morone americana* | 7 | 3 | 2 | – | 4110 | 15 |
| Herbivore | | | | | | | |
| Gizzard shad (1, 2) | *Dorosoma cepedianum* | 4 | – | – | 14 | 91 | – |
| Omnivore | | | | | | | |
| Fathead minnow (2, 1) | *Pimephales promelas* | – | – | 2 | 5 | – | 12 |
| Golden shiner (3, 2) | *Notemigonus crysoleucas* | – | – | – | – | 1 | – |
| Mimic shiner (5, 3) | *Notropis volucellus* | – | – | – | – | – | 3 |
| Spottail shiner (1, 1) | *Notropis hudsonius* | 13 | 2 | 2 | 8 | 14 | – |
| Piscivore | | | | | | | |
| Largemouth bass (3, 2) | *Micropterus salmoides* | – | – | 4 | – | 1 | – |
| Northern pike (4, 2) | *Esox lucius* | – | – | – | – | – | 2 |
| White crappie (1, 1) | *Poxomis annularis* | – | – | – | – | 2 | – |
| Yellow perch (3, 2) | *Perca flavescens* | – | – | 54 | 12 | 7 | 5 |
| Planktivore | | | | | | | |
| Emerald shiner (3, 2) | *Notropis atherinoides* | – | – | 2 | 1 | – | – |
| Alewife (1, 2) | *Alosa pseudoharengus* | – | – | – | – | 1 | – |

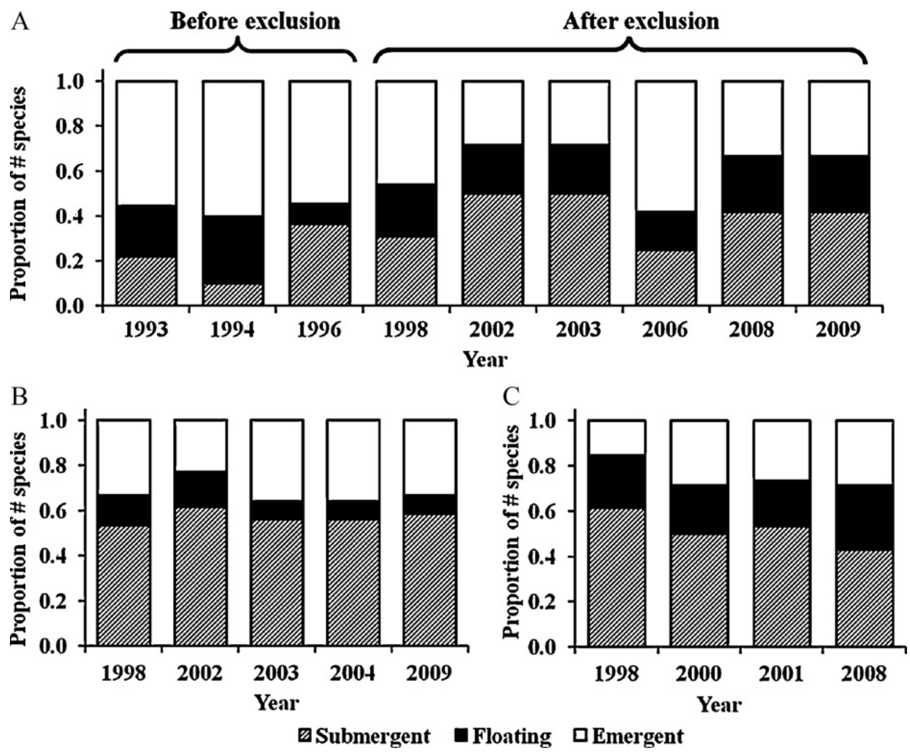


**Fig. 4.** Proportion of macrophyte species' types in (A) Cootes Paradise, (B) Matchedash Bay and (C) Long Point.

Nevertheless, when we compared scores for the period prior to carp exclusion (1996) with those obtained after carp exclusion (2008), we found the scores post-exclusion have significantly improved (Mann–Whitney, $p < 0.05$).

We also recognized some site-to-site variability in the marsh. We were able to calculate WQI and WZI for various sites within CP.

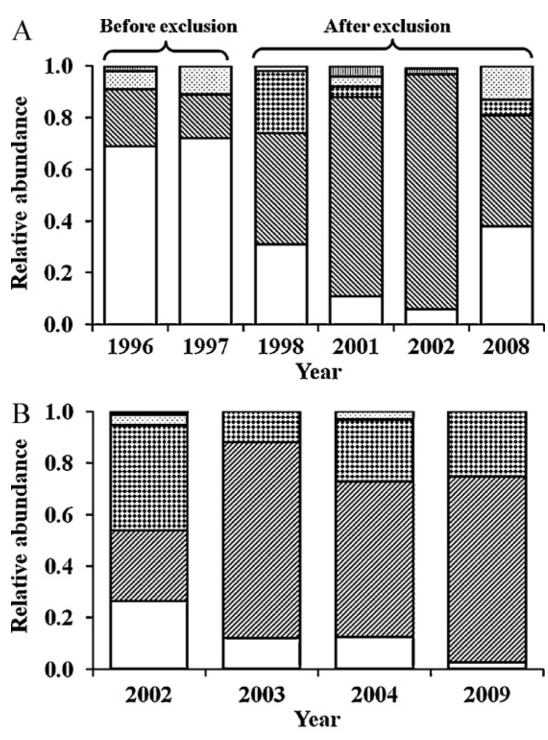


**Fig. 5.** Relative abundance of fish by functional feeding niche for each year sampled at (A) Cootes Paradise and (B) Matchedash Bay.

In 1993, all of the sites were below a WQI score of −2, except for the embayment known as Westdale Cut (CP16), which was slightly higher at −1.5 (Fig. 7). By 2008, water quality conditions at the open water site (CP1), the sewage lagoon (CP5), and at another embayment known as Mac Landing (CP10) had all improved to scores between −1.2 and −1.8, while water in Spencer's Creek outflow (CP4) had improved to −0.9, similar to that in Westdale Cut (CP16) (Fig. 7). It is noteworthy that both CP4 and CP16 had greater amount of residual vegetation compared with the other three sites. In addition, we calculated individual WZI scores for sites in CP in 1996 and 2008, and found that the sites with vegetation had higher scores (Mann–Whitney, $p < 0.05$).

### 3.3. Coastal marsh comparison

The comparison across wetlands is necessarily restricted to data collected following the Fishway implementation because no data had been collected prior to 1998 in Long Point (LP) or Matchedash Bay (MB). All of the index scores for CP were consistently lower than those for LP and MB (Fig. 8). Of all the three wetlands, MB has improved the most over 10 years, as indicated by the steepest slope relating WQI and WMI to time (Fig. 8A and C). When all three marshes were compared, we found significant differences among the three sites (Kruskal–Wallis, $p < 0.05$). WQI and WMI scores associated with LP were significantly higher than those for CP; however, scores associated with MB were not significantly different from those of either wetland (Nemenyi Test). WQI scores at MB were found to be intermediate between LP and CP (Fig. 8A), and corresponding WMI scores were more similar to those of the degraded CP in 1998 but more similar to higher quality LP in 2009 (Fig. 8C).

Compared with CP, both MB and LP have lower water turbidity and a higher proportion of submergent species (Fig. 4). Similarly, WZI scores corresponding to LP and MB were both higher than those corresponding to CP, even though they were statistically similar to each other (Kruskal–Wallis, $p < 0.05$; Nemenyi Test). Consistent with expectation, WFI scores for MB were also significantly higher

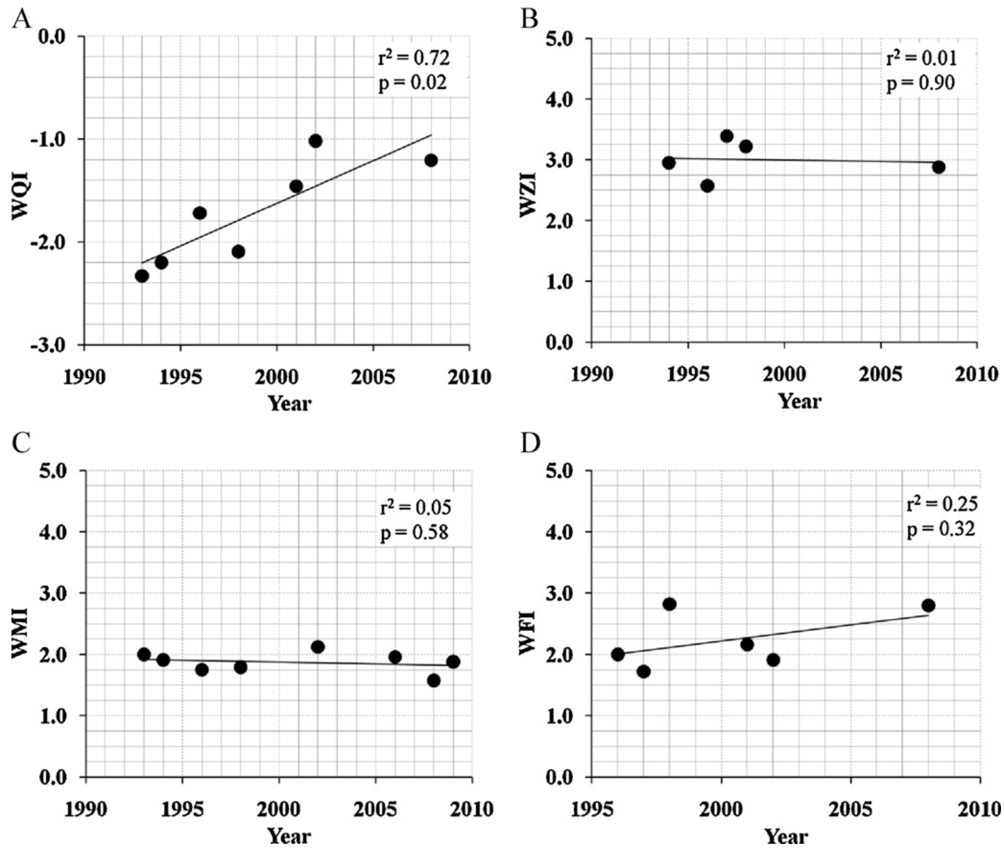


**Fig. 6.** Index scores for Cootes Paradise over the study period (A) Water Quality Index, (B) Wetland Zooplankton Index, (C) Wetland Macrophyte Index and (D) Wetland Fish Index.

than those for CP (Mann–Whitney, $p < 0.05$; Fig. 8D). Unfortunately, there were no fish data available for LP to conduct a comparison with CP. When we examined changes in the structure of the fish community, we found a higher proportion of carnivores and piscivores in MB compared with CP; by contrast, CP had a greater number of feeding guilds (Fig. 5), including a species of herbivorous fish (gizzard shad), which consumes algae. It is important to note that despite the biomanipulation, benthivores still comprise a larger proportion of the total catch in CP compared with MB (Fig. 5).

## 4. Discussion

### 4.1. Evaluation of CP quality

Our primary goal was to determine how ecosystem quality has responded to the biomanipulation in CP. We used three

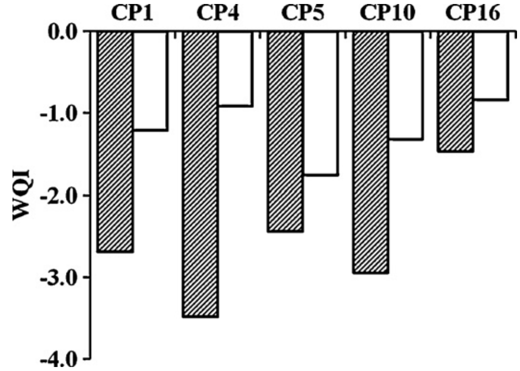


**Fig. 7.** WQI scores calculated for water samples collected at five sites in Cootes Paradise (as shown in Fig. 1 inset) during 1993 (hatched bars) and 2008 (open bars).

approaches to evaluate the response of CP to the Fishway implementation: (1) analyzing changes in water quality parameters, zooplankton, macrophyte and fish (2) using indices to quantify and assimilate information that have been proven to be indicative of ecosystem quality and (3) comparing long-term changes in CP to that of two other coastal wetlands in the Great Lakes basin. The goal of carp exclusion was to trigger the switch from the current turbid, phytoplankton-dominated state to its former clear-water, macrophyte-dominated state. Thus we will also evaluate the progress of this restoration goal, in addition to an overall assessment of the quality of CP.

Most water quality variables (Turbidity, TSS, ISS, Chl-*a*, TP, SRP, TAN, TNN) in CP have decreased over the period of study, indicating an overall improvement in wetland quality (Table 1). These observations are confirmed by an increasing trend in WQI scores, which incorporated all 12 variables into a single score (Fig. 6A). It is noteworthy that the overall WQI score is still below zero, which indicates that human-induced degradation is still occurring in the marsh. Nutrient levels in CP are still sufficiently high that conventional classification systems (e.g. Carlson's (1977) Trophic State Index) would identify it as hypereutrophic.

The zooplankton community has shifted to larger-bodied organisms (Fig. 2), which other studies have found is indicative of more oligotrophic conditions (Gannon and Stemberger, 1978; Jeppesen et al., 2000). Lougheed and Chow-Fraser (2002) observed that in general, good-quality wetlands tended to be dominated by large-bodied zooplankton that feed on epiphytic algae, whereas degraded wetlands tended to be dominated by small-bodied zooplankton that feed primarily on phytoplankton. Consistent with these observations, we found that representation by medium-sized cladocerans increased from 1% of total abundance in 1996 (prior to carp exclusion) to 28% in 2008 (Fig. 2). This group consisted of

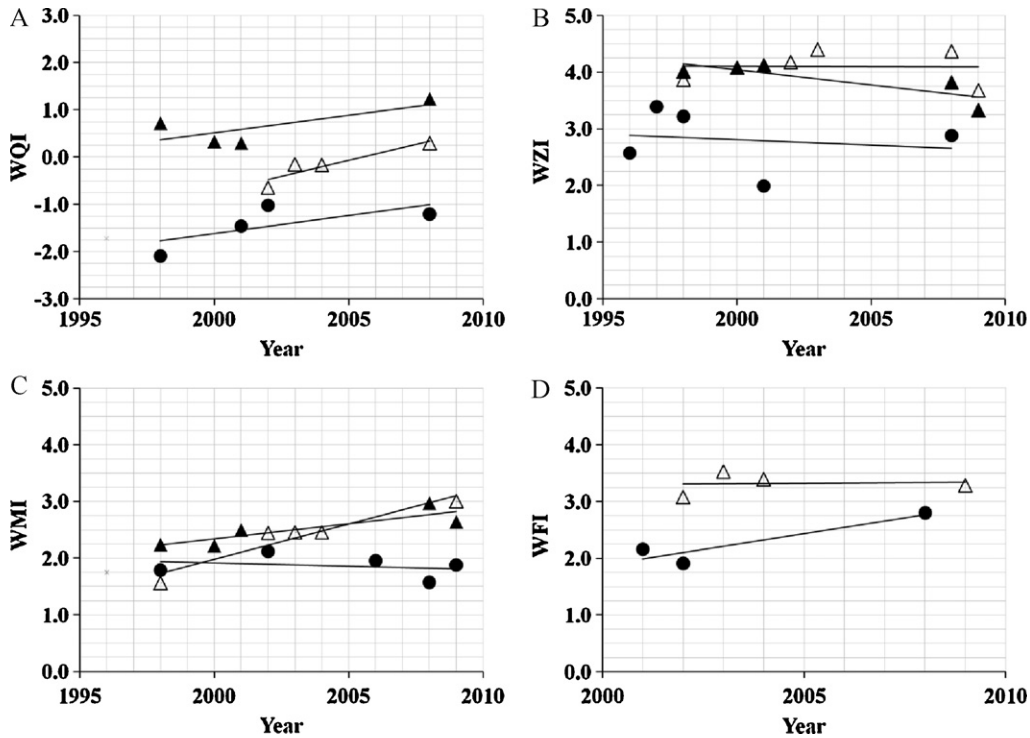


**Fig. 8.** Index scores for Cootes Paradise (closed circle), Matchedash Bay (open triangle) and Long Point (closed triangle) over the study period (A) Water Quality Index, (B) Wetland Zooplankton Index, (C) Wetland Macrophyte Index and (D) Wetland Fish Index.

*Ceriodaphnia* sp. and species of chydoridae that are known to be associated with aquatic plants (Fairchild, 1981; Paterson, 1993) and also have higher U and T values (see Table 2 for a list of species U and T values). Despite the obvious increased representation of species with high U and T values, WZI scores did not reflect any improvement in quality after the carp exclusion (Fig. 6B, Mann–Whitney $p > 0.05$). This may be attributed to the relatively large drop in species with low scores (e.g. *Brachionus* sp.) as compared to the smaller increase in species with high scores (e.g. *Ceriodaphnia* sp.). Total species richness of macrophytes did not increase substantially following carp exclusion, and the proportion of non-native species has not diminished; in some years (i.e. 2008) it has greatly increased (Fig. 3). Studies on the dynamics of the species present in the marsh (*Typha latifolia*, *Phragmites australis* – Wei and Chow-Fraser, 2006; *Typha* spp., *P. australis* – Wilcox et al., 2003) suggest that low water levels that we are currently experiencing may inadvertently benefit these non-native species at the expense of native species. This has been reported for some U.S. restoration sites, where a high potential for dominance by non-native species has halted the succession to more desirable native species (Zedler, 2000). We have also seen an increase in the proportion of submergent species in the marsh (Fig. 4A), although the plant community in the two other Great Lakes marshes used for comparison still have a higher representation (Fig. 4B and C). The WMI was not able to reflect the improved water quality in CP (Fig. 6C) since most of the species found in CP were pollution tolerant species that had low U and T values

As expected, the operation of the CP Fishway resulted in a dramatic reduction in the common carp population (Table 4). Not surprisingly, the feeding guilds present in the marsh following the carp exclusion have become more diverse (Fig. 5, Table 4). As mentioned previously, we have to exercise caution when comparing fish abundance and community structure information since two different sampling gears were used and there are known biases associated with each method (Chow-Fraser et al., 2006). Electrofishing was used to collect the pre-exclusion data, which in theory will target the larger piscivores, whereas the fyke nets used following the biomanipulation should target smaller fish, such as planktivores and carnivores. Therefore, the increased representation of piscivore through time cannot be attributed to a sampling bias related to different gears. However, the increased diversity of fish in more recent surveys may be attributed to the use of fyke nets since Chow-Fraser et al. (2006) showed a tendency for fyke nets to catch fish with more diverse feeding niches in degraded wetlands. Nevertheless, Kostuk (2006) showed that such differences did not lead to differences in WFI scores; therefore the higher scores in 2008 compared to 1996 are not an artefact of sampling gear (Mann–Whitney 1996–2008, $p < 0.05$). Despite the increasing trend, there were insufficient data to show a significant increase in WFI scores through time (Fig. 6D, $r^2 = 0.25$, $p = 0.32$).

In summary, both individual water quality parameters and the WQI scores show that the marsh has improved in quality following carp exclusion; however, the current status of CP is still degraded when compared with other wetlands (Chow-Fraser, 2006, Fig. 8). Although the zooplankton and fish communities have become more diverse, many species are still pollutant-tolerant generalists (e.g. *Moina* sp., brown bullhead). Species richness and the proportion of submergent species have increased, but non-native species remain prominent, and many of the native species are pollutant-tolerant generalists (e.g. *Lemna minor*, *Ceretaphyllum dermersum*). Overall, the improvement in water quality has not been accompanied by significant increases in the biotic indicators and CP remains in a turbid, algal-dominated state after more than 10 years of carp exclusion. We have observed, however, that portions of the marsh that had residual plant communities have progressed further than the open-water sites

### 4.2. Suggestions why CP is in a stable turbid state

Degraded communities, such as CP, do not always respond predictably to management efforts (Angeler et al., 2003; Zedler, 2000).

Many European shallow lakes became turbid-phytoplankton dominated systems due to external nutrient loading, and did not switch back to clear-macrophyte dominated systems with corresponding nutrient reduction (Ibelings et al., 2007; Madgwick, 1999; Moss et al., 1996). We know that CP is capable of switching to a clearer state. In 1997, abnormally low spring temperatures caused a delay of fish migration into the marsh (including planktivores), which released the zooplankton population from predation, resulting in zooplankton-mediated improvement in water clarity and an expansion of submergent vegetation in previously unvegetated shallows (Lougheed et al., 2004). However, this state was short-lived and CP switched back to its former turbid state. Many other researchers have encountered this apparent stability of the turbid-phytoplankton state and there are many explanations for the mechanisms such as herestesis (Ibelings et al., 2007), and the development of feedback between abiotic and biotic factors (Suding et al., 2004). Chow-Fraser (1998) developed a conceptual model explaining the factors that play a role in the degraded and former high-quality state of CP. It is clear that there are a number of positive and negative feedback mechanisms that are keeping CP in its current degraded state.

Chow-Fraser (1998) predicted that a reduced carp population would cause lower turbidity, but a number of other factors such as wind and wave action, high sediment loading from the watershed, and high algal biomass would continue to cause high turbidity in the marsh. Lougheed et al. (1998) predicted that exclusion of 90% of the carp would reduce water turbidity in Cootes Paradise by up to 45%, and this has largely been confirmed in the present study. Prior to biomanipulation (1993–1996), turbidity had been approximately 60 NTU, but after the Fishway became operational (1998–2008), turbidity was reduced by 44% to 33.7 NTU (Table 1). Chow-Fraser (1999) suggested that background levels (25.4 NTU) could be attributed to wind re-suspension and algal growth. Lougheed et al. (1998) also predicted that wetlands with turbidity higher than 20 NTU would likely have fewer than five species of submergent taxa, whereas wetlands below this turbidity threshold would have a more diverse aquatic plant community. According to data from the last 10 years of monitoring, CP will probably continue to have turbidity levels well above this threshold, and there is low probability that a large number of submergent species will become re-established in the near future.

In addition to high turbidity there are a number of other factors that are not facilitating the switch to a clear-water macrophyte dominated system. For example, nutrients are still very high. Summer mean TP must fall below 0.1 mg/L (Hosper and Meijer, 1993; Jeppesen et al., 2000) and TN must be below 2 mg/L (González Sagrario et al., 2005) before macrophytes can regain high areal cover. In CP, the lowest summer mean TP occurred in 2008 and was 0.14 mg/L in 2008 (Table 1). Although we did not measure TN in 2008, the lowest historic value measured in 1993 (~4 mg/L, Table 1) was twice as high as the desired level.

Expansion of the current macrophyte population is essential to the improvement in quality of CP, as macrophytes are key to stable clear-water states (Hosper and Meijer, 1993; Ibelings et al., 2007). The positive effects of the macrophyte community can be observed in CP. We observed a differential response in both WQI (Fig. 7) and WZI scores in vegetated sites, similar to that observed in other systems (Angeler et al., 2003; Moss et al., 1996; Ibelings et al., 2007). Even if CP switches to a stable clear-water system with further management actions, it would not likely return to its original undisturbed state. CP is an excellent example of a 'novel ecosystem' (Hobbs et al., 2006) that has been so transformed by human actions that it is essentially a new system, with different species assemblages, often dominated by invasive species, and should therefore be treated as such (Lindenmayer et al., 2008).

## 5. Conclusion

We have analyzed the response of the abiotic and biotic components of Cootes Paradise to restoration efforts. Although water quality is improving, it has not improved sufficiently to allow for a shift in the biotic communities towards species representative of higher quality condition. In general the communities are becoming more diverse, yet they are dominated by pollutant-tolerant species, which is reflected in the index values. This study reveals the complexity of restoration projects in highly degraded systems such as Cootes Paradise, and provides insight into the functionality of these indices as a useful management tool. The recovery of a system as complex, large and degraded as Cootes Paradise cannot be expected to be simple or inexpensive. Cootes Paradise offers a good opportunity to educate the public about unintended harmful actions caused by humans on natural systems, and serves as an important reminder that a degraded ecosystem can be difficult, if not impossible, to restore and may require management actions that are different than those appropriate for its original state.

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