Multitemporal comparison of wetland communities in diked and undiked wetlands in southern Georgian Bay

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Abstract

Hydrological connection between coastal wetlands and the Laurentian Great Lakes plays an important role in maintaining aquatic biodiversity in the wetlands by preventing monocultures of emergent vegetation from forming, by facilitating frequent exchange of chemical constituents between the wetlands and lakes, and by allowing daily and seasonal migration of fish in and out of the wetlands. We hypothesize that when a wetland becomes impounded or diked, the emergent vegetation will expand at the expense of aquatic habitat (open water and submersed aquatic vegetation), the water chemistry in the marsh will become altered, and the diversity of the fish community will become reduced. Conversely, there should be no long-term impact on avian diversity where water levels are not actively manipulated to maintain bird habitat. We tested these hypotheses by comparing changes in the wetland communities of Wye Marsh, a diked wetland in Georgian Bay, with that of a nearby undiked wetland, Matchedash Bay. We used available historic air photos (from 1930 to 2008) to quantify the amount of aquatic habitat in both wetlands. Consistent with our prediction, the amount of aquatic habitat decreased significantly through time in Wye Marsh, but not in Matchedash Bay; instead, area of open-water in the undiked wetland varied directly with mean water levels of Georgian Bay. Water chemistry in both wetlands reflected surrounding agricultural land-use and exhibited differences that could be predicted on the basis of the hydrological connection with Georgian Bay. Whereas diversity of the fish community in Wye Marsh was significantly lower than that in Matchedash Bay, the avian diversity showed no significant differences. We determined that diking wetlands is not a suitable solution to limit the loss of wetland habitat due to declining water levels in Georgian Bay.

Introduction

Water level fluctuations in large lakes, such as the Laurentian Great Lakes, are an important driver for many nearshore habitats. They provide a disturbance regime at the land-water interface by fluctuating according to seasonal, annual and inter-annual cycles (Keddy and Reznicek 1986; Jude and Pappas 1992; Wilcox 2004). Coastal wetlands, which are found at that interface, experience cyclical succession following these physical disturbances (Quinlan and Mulamoottil 1987; Wilcox and Meeker 1992; Chow-Fraser et al. 1998; Wilcox 2004). Since the fluctuations are so prevalent, secondary succession of aquatic vegetation communities initiates at new water-levels, where different species hold a competitive advantage (Keddy and Reznicek 1986; Quinlan and Mulamoottil 1987). Under natural conditions succession within the coastal wetlands is continuously being interrupted as the water levels fluctuate, thus preventing the wetlands from ever reaching climax communities (Keddy and Reznicek 1986; Jude and Pappas 1992; Wilcox 2004). At high water-levels, submergent and floating vegetation dominate and at low water-levels, emergent vegetation dominates in coastal wetlands that have a connection to the lake (Chow-Fraser et al. 1998). As the water-levels rise, emergent vegetation is drowned out and space is opened up for submergent and floating species. When water levels decline, seeds of emergent vegetation germinate from the seed bank and grow in the shallow reaches of the wetlands (Keddy and Reznicek 1986; Wilcox and Nichols 2008).

Given that water-level fluctuation is a key determinant of macrophyte growth and that macrophytes provide the basic structural habitat of coastal wetlands, a loss of natural fluctuation through impoundment should lead to substantial alteration of ecosystem processes. The extent of emergent vegetation cover tends to be greater in diked wetlands with constant water levels than in undiked wetlands that experience natural fluctuations in water levels (Mitsch 1992; Sherman et al. 1996; Johnson et al. 1997; Galloway et al. 2006). In addition, Gottgens et al. (1998) found that the ratio of emergent vegetation to open water in an impounded marsh was more constant through time compared to a similar system that was hydrologically connected to the lake. Disruption of hydrological connection between a wetland and the lake can also interfere with exchange of chemical constituents such as nutrients, sediment and algae (Mitsch 1992; Fracz and Chow-Fraser 2013). In Georgian Bay, Fracz and Chow-Fraser (2013) found that the water chemistry in beaver-impounded wetlands changed significantly because of a lack of mixing with the open water of the bay. Whereas nutrients and sediments accumulated behind beaver dams and created a more turbid and nutrient-rich environment, similar wetlands that were hydrologically connected with Georgian Bay had more clear water, with water chemistry that reflected the geology of the region (deCatanzaro and Chow-Fraser 2011; Fracz and Chow-Fraser 2013). Severance of hydrological connection may also affect the movement of biota into and out of wetlands. Keast and Fox (1990) found a lower species richness of fish within a system that was naturally impounded by beavers in Ontario, Canada. This is consistent with other studies that identified hydrological connection as the major driver for the presence of top predators (Snodgrass et al. 1996; Barber et al. 2002; Bouvier et al. 2009) and may explain why invertebrate diversity has been shown to increase in impounded ecosystems (McLaughlin and Harris 1990). Within the Great Lakes context, there are many fish species that migrate into coastal wetlands to spawn and feed (Jude and Pappas 1992; Wei et al. 2004), and an impoundment or dike should similarly disrupt such migrations and lead to cascading effects on the food web.

It is equally important to determine those species that are unaffected or positively affected by diking and impoundments. Galloway et al. (2006) found that impounded wetlands had higher abundance and species richness of marsh-nesting bird species as well as overall bird abundance. Few statistically significant differences were found, however, once they controlled for differences in geographic location and wetland characteristics. Increased waterfowl use was another benefit, but this was dependent on the type of vegetation cover in the diked marsh, which varied according to site characteristics, time since impoundment and local management strategies. Monfils et al. (2014) showed similar results, with few differences between bird communities when directly comparing diked and undiked wetlands. Kadlec (1962) found that impoundments had only short-term benefits for waterfowl because the vegetation structure required continuous water-level manipulations that are expensive to maintain. Therefore, after 5 years of impoundment without active water-level manipulation to encourage cyclic succession patterns in the plant community, waterfowl habitat began to decline in Michigan wetlands (Kadlec 1962).

Within the Great Lakes basin, there have been relatively few published studies on the short-term or long-term effects of impoundment on wetlands (e.g. Johnson et al. 1997; Gottgens et al. 1998; Galloway et al. 2006). In their preliminary assessment of wetland diking as an adaptation strategy to long-term water level reductions brought on by global climate change, Galloway et al. (2006) noted that we must have a broader understanding of how dikes and dams affect ecosystem functioning before we promote their use as an adaptation strategy. In fact the impacts that we see on coastal wetlands from sustained low water levels appear to be similar to situations of diked wetlands where the water level is not fluctuated. Given that most of the studies in the past have focused on coastal marshes in Lakes Erie and Ontario, there is an obvious need to expand the geographic focus to include the other Great Lakes. A case in point is Georgian Bay, the eastern arm of Lake Huron, where sustained low water-levels over the past 14 years (Figure 1) have resulted in major losses of wetland fish habitat (Fracz and Chow-Fraser 2012). Wilcox and Nichols (2008) found that the impacts of diking were similar to the effects on the plant community that were seen with long term low water levels. No literature exists that documents the long-term or even short-term impact of diking on wetlands that occur along the eastern and southern shoreline of Georgian Bay. Hence, no data are available to guide decisions regarding the appropriate use of dikes or dams as an adaptation strategy. We should be cautious when extrapolating results from studies of wetlands in Lakes Erie and Ontario directly to those in eastern Georgian Bay since Georgian Bay wetlands are geologically unique (deCatanzaro and Chow-Fraser 2011), have vastly better water-quality scores and more diverse biotic communities compared with those in the lower Great Lakes (Cvetkovic and Chow-Fraser 2011). Similarly, Lakes Huron and Michigan experience hydrological regimes with greater interannual variation in water level compared to the other Great Lakes (see Figure 2)(Wilcox et al 2007). Water level peaks are a month later than the lower Great Lakes and two months earlier than Lake Superior.

The primary goal of this study was to document the long-term effects of diking and damming on Wye Marsh, a wetland located in southeastern Georgian Bay that has been impounded at least since the 1930s, compared to Matchedash Bay, a wetland of similar size located 20 km east of Wye Marsh that has remained hydrologically connected to Georgian Bay. Vegetation extent, water-level patterns, water quality, bird assemblages and the fish community were all compared to highlight any differences in these diked and undiked wetlands of Georgian Bay. We quantified the amount of open aquatic habitat in Wye Marsh using air photos that span eight decades (1930 to 2008) and compare them with similar measurements of Matchedash Bay (Figure 3). The two wetlands are in separate watershed but they are both impacted by surrounding agricultural land uses and have similar dominant vegetation communities. These similarities enabled us to make valid comparisons since both wetland size and basin morphometry, as well as watershed characteristics can influence how the respective ecosystems will respond to impoundments (see Galloway et al. 2006). We hypothesized that the amount of aquatic habitat in Wye Marsh would decrease through time because of succession of the emergent community whereas aquatic habitat in Matchedash Bay would be significantly and positively related to water levels in Georgian Bay in a manner consistent with coastal marshes elsewhere (see Chow-Fraser et al. 1998). Since Wye Marsh water levels are not actively managed, we did not expect the marsh bird community to show any significant benefits from impoundment. We did expect the fish species diversity in Wye Marsh to be lower than that in Matchedash Bay due to lack of access for migratory species.

Methods

Site descriptions

Wye Marsh and Matchedash bay are large, provincially significant wetlands with relatively flat bathymetry and cattails (*Typha spp.*) as the dominant emergent vegetation (Ducks Unlimited Canada 1995). Both Wye Marsh and Matchedash bay are subwatersheds within the Severn River watershed. Land use within the Severn River watershed consists of 52% natural heritage feature, 35% agriculture and 13% urban land use (Southern Georgian Bay-Lake Simcoe Source Protection Committee 2015).

Wye Marsh is an approximately 600 ha provincially significant wetland that has been deemed an Important Bird Area and an Area of National Scientific Interest with sections designated as both Provincial Wildlife Areas and National Wildlife Areas. It is found 1.8 km upstream of Georgian Bay, Lake Huron in Tay, Ontario, Canada. A 19,600 ha agriculturally impacted watershed drains into the marsh which is underlain primarily by limestone (Maccrimmon 1980). The primary hydrological input of the marsh is the Wye River (Bufo Inc. 1978). It is unique for a wetland of its size and geographic location with respect to its management history as it has been impounded for more than 8 decades at the northern outlet where it would otherwise be connected to Georgian Bay. There is no record of the construction of the 70-m wide St. Marie Dam, although a historical 1930 aerial image of the region confirms the dam's existence to before that date. The dam did, however, break once in 1972, returning hydrological connection of the marsh to the bay until the dam was rebuilt shortly thereafter by the Ontario Ministry of Natural Resources (Ducks Unlimited Canada 1995). The Wye River empties into Georgian Bay via a vertical drop over the St. Marie Dam. The height of the vertical drop varies with the water level in Georgian Bay.

Matchedash Bay is an approximately 900 ha provincially significant wetland and, similar to Wye Marsh, is also considered an Important Bird Area (Wilson and Cheskey 2001). It is hydrologically connected to southeastern Georgian Bay, found approximately 20 km East of Wye Marsh (Figure 3). The Matchedash Bay watershed is also heavily impacted by agricultural practices and the perimeter of the marsh has been modified with some docks and cottages. The wetland is primarily underlain by limestone bedrock with some Precambrian Shield rock outlets and has two river inputs; the North River and Coldwater River (IBA Canada 2012).

Data collection and analysis

We collected and analyzed all chemical parameters according to methods described in Chow-Fraser (2006). We analyzed total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), total ammonia-nitrogen (TAN), total nitrate-nitrogen (TNN), pH, specific conductance (COND) and total suspended solids (TSS). One water sample was collected from the middle of each wetland, in open water void of submerged aquatic vegetation (Figure 4 and Figure 5) (Wye Marsh – July 23, 2012 and Matchedash Bay – July 25, 2012).

To measure daily changes in water levels during the 2013 growing season, we installed a manual stage gauge in Wye Marsh and took readings each morning. Since Matchedash Bay is connected hydrologically to Georgian Bay, we used daily water levels measured at the Midland Canadian Hydrographic Service Station (Department of Fisheries and Oceans Canada), which is located approximately 16 km linear distance away. The lowest recorded water level was subtracted from all entries at each site so that we could compare the relative change in water levels through the season. All mean annual water levels for Georgian Bay were calculated from data provided by the Canadian Hydrographic Services.

Changes in water levels at each of the wetlands were determined by calculating the relative change in water level through time. In Wye Marsh this was calculated by subtracting the lowest measured water level from all water level readings (measured using a meter stick), and in Matchedash the lowest water level in m.a.s.l was subtracted from all water level data. This gave the relative change in water levels for each site (Figure 6)

To evaluate the effect of hydrological connection on water chemistry, we assembled published data from studies that documented the conditions of wetlands with varying degrees of hydrological connection to Georgian Bay. This dataset included mean values for 17 beaver-impounded marshes, 18 hydrologically connected marshes (Fracz and Chow-Fraser 2013) and 15 open-water samples collected along eastern Georgian Bay (deCatanzaro and Chow-Fraser 2011). These data were then compared to corresponding values measured in Wye Marsh and Matchedash Bay in this study.

To assess the historical aquatic habitat availability, we obtained aerial imagery from the National Air Photo Library of Canada (1930, 1931, 1965, 1973, 1976, 1987 and 1995) and the Ontario Ministry of Natural Resources (OMNR) (2002 and 2008)(all images were 600 by 600 dpi resolution) (Table 1). All images that were not already in digital format were scanned as JPEG files (600 by 600 dpi, greyscale) and georeferenced into ArcGIS version 10 (ESRI, Redlands, California, U.S.A.) for analysis. The 2002 and 2008 images had been georeferenced by OMNR. The georeferenced 2002 image was used as a control for referencing the older images. Beginning with 1995, each image was georeferenced with the next most recent image (eg 1995 georeferenced with 2002, 1987 georefernced with 1995). A boundary was digitized around the marsh border for each site and year according to the vegetation zones. A dynamic boundary was chosen instead of using a “cookie cutter” approach, in this way only wetland habitat is captured which is particularly important as upland habitat expands. The area within each border was then digitized and tabulated as either aquatic habitat or emergent vegetation. Vegetation class boundaries were determined visually at a 1:50000 scale in the ArcGIS. This allowed for the quantification of change in habitat through time and as it related to mean annual water levels in Georgian Bay. We completed all statistical analyses in JMP version 10 (SAS Institute Inc., Cary, North Carolina, USA).

To enable valid comparisons, we wanted to use the same gear type to sample the fish communities in both wetlands. Even though fyke nets and boat electrofishing have been used successfully to survey the fish communities in wetlands of eastern Georgian Bay (see Cvetkovic et al. 2012), neither method could be used in Wye Marsh because an electrofishing boat could not enter the marsh, and all accessible areas were either too shallow or too deep for fyke nets. We therefore chose to use modified Windermere traps (Edwards et al. 1998), which could be set up on the soft, mucky sediment without use of poles. Fishing sites were selected based on accessibility and habitat type. Nets could only be set in areas that were accessible by canoe. Due to the dense nature of emergent vegetation in Wye Marsh there were a limited number of possible locations for deploying the nets. In Wye Marsh nets were set in patches of submergent vegetation that were accessible by canoe and the nets in Matchedash were set where the habitat looked visually similar to the Wye marsh locations. On two occasions in 2012 (May 8-10 and July 5-7) we surveyed the fish communities using traps that were paired 15 m apart, with one net having two conical openings and the other having only one. Net openings were placed parallel to the shoreline in areas that had low-density, patchy, submerged aquatic vegetation. The traps were deployed without wings or leads between 8:00 am and 11:00 am and left for 5 to 6 hours. The two wetlands were fished on consecutive days. When the nets were pulled, fish were identified to species, measured and released unharmed to the water where they were caught. We selected 5 sites in Wye Marsh, based on accessibility (see Figure 4), to conduct the fish surveys and 3 sites in Matchedash that had similar site characteristics (see Figure 5).

To determine how the fish communities in Wye Marsh and Matchedash Bay compared with those found in wetlands elsewhere throughout eastern Georgian Bay, we accessed data from Cvetkovic and Chow-Fraser (2011), which included fish information collected from 116 wetlands collected over a 10-year period (2001 to 2011). Since these fish had been caught with a different gear type (24-h paired fyke nets), we calculated relative abundances and only compared the most common fish taxa.

For both Wye Marsh and Matchedash Bay, point counts of marsh birds were conducted twice at each of three sites during May and June in 2012. Each point count was 25 minutes in duration, including 10 minutes of passive listening, 10 minutes of active playbacks of secretive marsh species songs, followed by a final 5 minute passive listening period. During the 10-minute playback period, 30-second songs of 10 focal secretive marsh species were broadcast from hand-held speakers, with 30 seconds of silence between playbacks. Playbacks of these secretive marsh birds included the Least bittern, American bittern, Yellow rail, Pied-billed grebe, American coot, Common moorhen, Sora, Virginia rail, King rail and Black rail. Locations of point counts at Wye Marsh were chosen to represent different habitat types, and situated at least 380 m apart (Figure 4). We surveyed close to the dam to account for potentially different habitat that may arise due to the construction of the impoundment. The middle of the marsh was chosen to account for any species that may prefer open water surrounded by emergent vegetation and/or interior marsh habitat. The third station was at the boardwalk, near the edge of the marsh and close to the visitor centre, where we observed a structurally diverse habitat. Sites in Matchedash Bay were chosen to best reflect similar attributes of the habitat surveyed in Wye Marsh (Figure 5).

Mean scores of the Shannon diversity index were used to assess the overall diversity of the bird and fish communities in both wetlands. Using all of the data collected, we also calculated scores of the Sorensen similarity index for both birds and fish (McCune et al. 2002). For the bird communities, we calculated scores of the Index of Marsh Bird Community Integrity of each site (DeLuca et al. 2004). All calculations and statistical analyses were carried out with JMP version 10 (SAS Institute Inc., Cary, North Carolina, USA).

Results

Water-chemistry variables available for Wye Marsh and Matchedash Bay have been assembled and presented in Table 2. Phosphorus concentrations in Wye Marsh were uniformly higher than those in Matchedash Bay, with almost double the TP concentrations (63.0 vs 34.2 $\mu g \cdot L^{-1}$, respectively) and 1.5 times higher SRP concentrations (19.4 vs 13.4 $\mu g \cdot L^{-1}$). By contrast, inorganic forms of nitrogen were lower in Wye Marsh than in Matchedash Bay (0.04 vs 0.07 $mg \cdot L^{-1}$ for TAN and 0.09 vs 0.27 $mg \cdot L^{-1}$ for TNN, respectively) and no differences were found for TN (both 1.04 $mg \cdot L^{-1}$). Water in Wye Marsh was more acidic (pH 6.38) than that in Matchedash Bay (pH of 8.0), and had higher specific conductance (307 vs 221 $\mu S \cdot cm^{-1}$). The concentration of total suspended solids in Wye Marsh was much lower than that in Matchedash Bay (4.40 vs 14.83 $mg \cdot L^{-1}$).

Hydrographs for both study sites were strikingly different through the 2013 growing season (Figure 6). Water levels in Wye Marsh (impounded wetland) were highest in late April and continued to decline throughout the study period, whereas those in Matchedash Bay (hydrologically connected with Georgian Bay) increased from late April to early August and then began to decline. The magnitude of change in water levels through the season in Wye Marsh (0.34 m) was about 35% lower than that for Matchedash Bay (0.53 m).

The amount of emergent and aquatic habitat (open water and submerged aquatic vegetation) in Wye Marsh (Figure 7) and Matchedash Bay (Figure 8) has fluctuated over the past eight decades. In this study, we were most interested in how availability of aquatic habitat has changed through time since this is an essential habitat class for both fish and bird species. Between 1930 and 2008, the amount of aquatic habitat in Wye Marsh changed from 153.4 ha in 1930 to 57.38 ha in 2008, a corresponding decrease in percentage marsh habitat from 32% to 12% (Table 3). There was a general decline in this habitat category through the 78 years (Figure 1.9A), and even though there was a notable resurgence in aquatic habitat in 1976 (to 31%), we found a statistically significant negative relationship with time (F ratio = 11.32, p = 0.020, r^2 = 0.69). It is important to note that the 1976 air photo was taken shortly after the St. Marie Dam had been rebuilt and therefore reflects a short duration when the marsh had been hydrologically reconnected with Georgian Bay. To remove the confounding effect of the dam failure, we excluded data prior to 1972 and ran a second the regression analysis for Wye Marsh. This time, we found a much stronger negative relationship with time (F ratio = 16.92, p = 0.026, r^2 = 0.85) and a steeper slope that is more representative of the long-term effect of the impoundment on the plant community. By comparison, the amount of aquatic habitat in Matchedash Bay fluctuated from 409.44 ha in 1931 to 330.21 ha in 2008, with the highest value in 1973, at a time when water levels were near record highs for Lake Huron (Table 3). Even though there was a corresponding drop in percentage wetland habitat from 50% to 37% between the 77 years, we did not find any statistically significant decline with time (F ratio = 0.13, p = 0.733, r^2 = 0.03; Figure 9A).

We hypothesized that the amount of aquatic habitat in coastal wetlands should vary as a function of water levels in Lake Huron, as long as they are hydrologically connected with Georgian Bay. We tested this hypothesis by regressing the amount of aquatic habitat against mean annual water levels of Lake Huron, and found a significant positive relationship between these variables in the hydrologically connected Matchedash Bay (F ratio = 12.20, p = 0.017, r^2 = 0.71; Figure 9B). By contrast, we found no statistically significant relationship between these variables for Wye Marsh when we used all data from 1930 to 2008 (F ratio = 2.2656, p = 0.1926, r^2 = 0.31) or when we only used data from 1976 to 2008 (F ratio = 3.7211, p = 0.1493, r^2 = 0.55; Figure 9B).

The fish communities in the two wetlands differed with respect to species richness (4 vs 5 for Wye Marsh and Matchedash Bay, respectively; Table 4). Of the species caught, only the yellow perch (*Perca flavescens*) and pumpkinseed (*Lepomis gibbosus*) were common. The shorthead redhorse (*Maxostoma macrolepidotum*) and brown bullhead (*Ameiurus nebulosus*) were only caught in Wye Marsh whereas the longear sunfish (*Lepomis megalotis)*, largemouth bass (*Micropterus salmoides*) and rock bass (*Ambloplites rupestris)* were only found in Matchedash Bay. Each wetland had 2 species of migratory fish; Wye Marsh had the yellow perch and shorthead redhorse, whereas Matchedash had yellow perch and longear sunfish. Mean Shannon diversity score for Wye Marsh was significantly lower than that for Matchedash Bay (Wilcoxon rank sums Z = 2.22, p = 0.0262; Table 5). The lower score associated with Wye Marsh reflected the number of nets that were empty or had only one species present. The Sorensen similarity index was 0.44 (Table 5).

The two wetlands had similar bird community composition and diversity, especially with respect to presence of marsh-dependent species (Table 6). Unidentifiable bird species were excluded from all index score calculations (Table 6). In total, 22 bird species were identified in

Wye Marsh, including 4 marsh-dependent species (swamp sparrow; *Melospiza georgiana*, American bittern; *Botaurus lentiginosus*, marsh wren; *Cistothorus palustris,* and Virginia rail; *Rallus limicola*). By comparison, we found 17 species in Matchedash Bay, which also included 4 marsh dependent species. Three marsh-dependent species were found in both wetlands; unique species were the American bittern in Wye Marsh and the great blue heron (*Ardea herodias*) in Matchedash Bay. The mean Shannon diversity score was numerically higher for Wye Marsh (2.02 ± 0.19) compared with Matchedash Bay (1.97 ± 0.10) but the difference was not statistically significant (Wilcoxon rank sums Z = 0.44, p = 0.6625; Table 5). The Sorensen similarity index score was 0.72 (Table 5). The Index of Marsh Bird Community Integrity score for Wye Marsh (5.65 ± 0.83) was only slightly lower than that for Matchedash Bay (5.79 ± 0.40; Table 1.5), but again, we found no significant differences between wetlands (Wilcoxon rank sums Z = -0.00, p = 1.0000).

Discussion

This study has shown that diking and damming are not suitable strategies to combat the loss of wetland habitat due to declining water levels in Georgian Bay. It is the first project to document the impacts of impoundment on a Georgian Bay wetland, and will provide a basis by which managers can judge the long-term implications of loss of hydrological connection on ecosystem functions in one of the most biologically diverse ecosystems of the Great Lakes.

It is helpful to interpret differences and similarities between Wye Marsh and Matchedash Bay in light of what we know about the limnology of Georgian Bay and associated coastal marshes. Previous studies have shown that water chemistry in offshore waters of Georgian Bay is generally alkaline and has a high specific conductance (see deCatanzaro and Chow-Fraser 2011) that reflect influences of the limestone bedrock from the Bruce Peninsula and the Niagara

Escarpment to the South and West (Table 2; Weiler 1988). Open water also has high concentration of nitrates, but relatively low concentrations of total and soluble phosphorus and total suspended solids (Table 2). By contrast, water in the coastal marshes (deCatanzaro and Chow-Fraser 2011) and beaver impoundments (Fracz and Chow-Fraser 2013) has reduced alkalinity, lower ionic strength and lower concentration of nitrates. The coastal water also has higher concentrations of suspended solids and phosphorus that reflects more heavy influences from watershed runoff, particularly when a system is completely disconnected from Georgian Bay (i.e. beaver impoundments; Table 2).

The location of Wye Marsh and Matchedash Bay on primarily limestone bedrock distinguishes them from the other coastal wetlands and beaver impoundments that have been studied in eastern Georgian Bay (i.e. deCatanzaro and Chow-Fraser 2011; Fracz and Chow-Fraser 2013). The relatively high specific conductance in these marshes can be attributed to this difference in bedrock, and also to the agriculturally dominant land use in their watershed (Table 2). There are higher concentrations of soluble and total phosphorus as well as higher nitrates in the water associated with farming activities. Consistent with other studies, however, the impounded wetland had higher concentrations of phosphorus (63.0 and 19.4 $\mu g \cdot L^{-1}$ of TP and SRP, respectively) compared to the hydrologically connected marsh (34.2 and 13.4 $\mu g \cdot L^{-1}$, respectively), and this may due to differences in connectivity. These results differ from nutrients in Lake Erie, where higher TP was found in hydrologically connected wetlands but are consistent with SRP concentrations being higher in diked wetlands (Mitsch 1992). Ammonia concentrations in both wetlands were much higher than the 0.008 $mg \cdot L^{-1}$ measured in open waters of Georgian Bay, but the concentration of 0.07 $mg \cdot L^{-1}$ in Matchedash Bay is higher than that in the beaver-impounded wetlands (0.03 $mg \cdot L^{-1}$) and Wye Marsh (0.04 $mg \cdot L^{-1}$). This is inconsistent with predicted effects of hydrological disconnection.

The lower pH in Wye Marsh (6.38) relative to Matchedash Bay (8.00) may also be attributed to the effect of hydrological severance between Wye Marsh and Georgian Bay, similar to the lower pH in a beaver-impounded wetland when compared to hydrologically connected coastal wetlands in eastern Georgian Bay (5.57 vs 6.95; Table 2). The fact that Matchedash has a more alkaline pH compared to the coastal wetlands sampled by deCatanzaro and Chow-Fraser (2011) is because the former receives H^+ and phosphorus-enriched fertilizer runoff from agricultural land use and also drains limestone bedrock, whereas the coastal wetlands of eastern Georgian Bay receive primarily dystrophic runoff from the Precambrian Shield.

The open waters of Georgian Bay are naturally low in total suspended solids (0.8 mg•L^{-1} in Table 2). At the land-water interface, however, human activities, carp bioturbation and watershed runoff can all contribute to higher levels of suspended solids (Chow-Fraser et al. 1998). In eastern Georgian Bay, TSS values measured within beaver-impounded wetlands (15.5 mg•L^{-1}) were consistently higher than those in hydrologically connected coastal marshes (2.1 mg•L^{-1}), and likely reflected the lack of mixing with the dilute water of Georgian Bay. When we compare the situation between Wye Marsh and Matchedash Bay, however, differences in TSS concentrations could not be explained by the effect of impoundment. In fact, results were opposite to what we had expected with three times lower TSS values within Wye Marsh compared with Matchedash Bay (4.4 vs 14.83 mg•L^{-1}). We attribute the higher turbidity in Matchedash Bay to boat traffic (which is not allowed in Wye Marsh) and to a very large population of common carp that can keep sediment suspended by their spawning and feeding activities (Pers. obs.; Mitsch 1992; Lougheed et al. 1998; Chow-Fraser 2005).

Along with a shift in water chemistry, we also found very different trends in water-level regimes during the growing season (Figure 6). Water levels of Georgian Bay at Midland, and therefore that of Matchedash Bay, peaked in late summer (early August), which is consistent with the general pattern exhibited by Lake Huron (Figure 2). This reflected a slow recharge through the summer when snow across the large Georgian Bay watershed melts and gradually makes its way into the bay. Wye Marsh, on the other hand, experienced the peak in April after initial snow melt followed by a steady drop in water level from May to August. We suggest that Wye Marsh rapidly filled up in early May from snowmelt in the relatively small watershed, and gradually drained during the summer months. We speculated that differences in the hydrological regimes of these two wetlands were due mainly to the loss of connection between Georgian Bay and the coastal marsh, which could have led to drastic physical alterations of aquatic habitat and impact the biotic communities (Keddy and Reznicek 1986).

Analysis of the aerial photos showed the proportion of aquatic habitat in Wye Marsh (12-32%) was always lower than that in Matchedash Bay (37-63%). Whereas aquatic habitat in Matchedash Bay varied as a function of the water level in Georgian Bay, the aquatic habitat in Wye Marsh decreased with time since impoundment. Though there are inconsistencies in the literature with respect to the extent of vegetation expansion in diked wetlands, studies conducted in the lower Great Lakes agree that mean vegetative coverage of diked wetlands that aren't actively managed for water-levels is greater than that of adjacent undiked wetlands (Mitsch 1992; Sherman et al. 1996; Johnson et al. 1997; Monfils et al. 2014). We suggest that the stabilized water level in Wye Marsh over these many decades has allowed the emergent vegetation community of mainly cattails to expand and form dense persistent floating mats irrespective of water depth, a situation that runs counter to past studies in which emergent vegetation is governed by changes in water level (Keddy and Reznicek 1986; Lyon et al. 1986; Chow-Fraser 2005; Wei and Chow-Fraser 2008; Wilcox et al. 2008; Vaccaro et al. 2009, Bufo Inc 1978). These persistent mats can become uprooted from the sediment and float, providing structure for further growth while being tolerant of water level fluctuations or deeper reaches of the marsh (Galloway et al. 2006). In 1995, a prescribed drawdown of Wye Marsh was attempted to curtail the growth of cattails. One of the reasons for the failure to successfully create a structurally diverse habitat was that these floating mats were able to withstand the drawdown (Ducks Unlimited Canada 1995, Wilcox et al 2008, Vaccaro et al. 2009). Similar result were seen in Lake Ontario where Typha spp coverage did not change because floating mats had formed that were immune to water level fluctuations (Wilcox et al. 2008). When the dam broke during the 1970s there was a pronounced reduction in the extent of emergent plants. It is unknown whether much of the sediment and cattail community was flushed out of Wye Marsh when the dam broke, or whether the cattail mats dried out when the water level dropped. As a result, larger proportion of the marsh in 1976 was identified as aquatic habitat (Figure 7). This caused the system to be reset, but when the dam was restored, the cattails began to expand as aggressively as before, so that by the late 1980s, floating mats of cattails were once again the dominant feature (Bufo Inc 1978).). By comparison, the proportion of aquatic habitat in Matchedash Bay did not vary significantly with time, but instead, was proportional with water level of Georgian Bay.

We predicted that the dam in Wye Marsh would restrict the movement of migratory fish species and lead to lower diversity index scores compared with Matchedash Bay. Although we could not discriminate our two sites on the basis of the number of migratory fish species (Table 4), we did find a lower diversity associated with Wye Marsh compared with Matchedash Bay. This is consistent with studies conducted elsewhere in the Great Lakes (Brazner 1997; Johnson et al. 1997; Bouvier et al. 2009) where reduced hydrological connectivity with the lake appeared to affect the distribution of fish species in coastal marshes and led to reduced Sorensen similarity scores between isolated sites (Table 5). Keast and Fox (1990) also found a similar reduction in species richness of fish in beaver dammed wetlands elsewhere in Ontario, Canada. As wetlands became more and more isolated from the main stream, species richness became further reduced. The two species that are considered migratory (yellow perch and shorthead redhorse; Jude and Pappas 1992; Wei et al. 2004) in Wye Marsh, could be a remnant population that survived the impoundment because they were uniformly small compared to those in Matchedash Bay and appeared to be stunted, possibly because of competition for food and the absence of a large predator (Johnson et al. 1997; Markham et al. 1997).

All fish species found were among the most commonly caught species along the eastern shores of Georgian Bay from 2001 to 2011. The two species that are common to both Wye Marsh and Matchedash Bay (pumpkinseed and yellow perch) are known to have moderate niche breadths and are tolerant of disturbance in other coastal marshes of the Great Lakes (Seilheimer and Chow-Fraser 2007). Absent from Wye Marsh were large piscivores such as largemouth bass. Bouvier et al. (2009) and Keast and Fox (1990) also found that impounded wetlands tended to lack large piscivores, and past investigators have attributed this to differences in environmental conditions or to hydrological connectivity (Snodgrass et al. 1996; Taylor 1997; Bouvier et al. 2009). For Wye Marsh, we suggest that both factors may have played a role in the long term but regardless of the mechanism, we attribute the lower fish biodiversity compared with Matchedash Bay to the presence of the St. Marie Dam.

As hypothesized, the bird communities sampled at both sites were very similar. Of the 25 species identified, 14 were common between sites including the marsh dependent Virginia rail, marsh wren and swamp sparrow. We did not find any significant differences between mean Shannon's *H* or IMBCI scores of these wetlands. Accordingly, the Sorensen similarity score between sites (0.72) was relatively high, indicating a large overlap in marsh birds. These scores also provide information with respect to the quality of the marsh habitat, which are both similarly high. Our results are consistent with those reported in Monfils et al. (2014) who showed that

67% of the bird species were common between diked and undiked wetlands of Lake Michigan. Galloway et al. (2006) also found that paired wetlands along the shores of the lower Great Lakes showed few differences in bird communities. In contrast, Nummi (1992) found greater waterfowl use after 3 years of wetland impoundment by beaver damming, but did not look into the long-term impacts. Our results are consistent with the findings of Kadlec (1962) and Harris and Marshall (1963) who showed only short-term gains from diking in situations where there was no active management of water levels.

Management implications

Impoundment structures such as dams and dikes are generally permanent, and are designed to manipulate/maintain water levels over the long term. In most instances, they have been implemented to maintain or improve plant and bird communities (Galloway et al. 2006). It is widely discussed, however, that active water-level management is required to maintain diverse plant and bird communities and that impoundment as a management strategy does not adequately consider aquatic species. In Georgian Bay, one of the most biologically diverse stretches of the Great Lakes, we observed overall negative implications of diking wetlands over the long term, consistent with past studies. We saw expansion of the emergent vegetation in the diked wetland through time, where we did not in the hydrologically connected wetland. In turn, we also saw patterns of altered water chemistry due to a loss of hydrological connection similar to what has been observed by deCatanzaro and Chow-Fraser (2011) and Fracz and Chow-Fraser (2013). Impacts to the fish community may also be attributed to changes in water chemistry, structure of the plant communities, and a barrier to migration of top predators. Our data also support past literature that bird and plant communities gain no apparent benefit from long-term impoundment without active water-level manipulation. Coastal wetlands along the eastern shoreline of Georgian Bay are relatively inaccessible and this inaccessibility would make active management costly and difficult to undertake. Even if they were accessible, these coastal wetlands are unlikely to be managed beyond their initial installation, mainly because of cost. Given these concerns, we do not believe that diking and damming are suitable strategies to combat the loss of wetland habitat due to declining water levels in Georgian Bay.

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Table 1 Air photos from 1930 to 1995 were obtained from the National Air Photo Library. The photos from 2002 and 2008 were from Ontario Ministry of Natural Resources and Forests.

	Year	Month	Spectral Range	Scale
Wye Marsh	1930	October	BW	15000
	1965	May	BW	35000
	1976	April	BW	50000
	1987	September	BW	50000
	1995	April	BW	50000
	2002	spring	Colour	resolution = 30 cm
	2008	spring	Colour	resolution = 30 cm
Matchedash Bay	1931	June	BW	15000
	1965	May	BW	35000
	1973	June	Infrared	20000
	1987	July	BW	50000
	1995	April	BW	50000
	2002	spring	Colour	resolution = 30 cm
	2008	spring	Colour	resolution = 30 cm

Table 2 Comparison of water chemistry data for Wye Marsh, Matchedash Bay, and mean values for 17 beaver-impounded wetlands (Fracz and Chow-Fraser 2013), 18 hydrologically connected coastal wetlands (Fracz and Chow-Fraser 2013) and 11 open water sites (deCatanzaro and Chow-Fraser 2011) in Georgian Bay. Values in brackets are the range. N/A = data not available. * indicates published data collected in Wye Marsh in 1998 (Lougheed and Chow-Fraser 2002) and unpublished data collected in Matchedash Bay in 2010.

Parameter	Wye Marsh	Matchedash Bay	Beaver-impounded wetland	Hydrologically connected coastal wetland	Open Georgian Bay water
Impounded	**Yes**	**No**	**Yes**	**No**	**No**
Agricultural land-use in watershed	**Yes**	**Yes**	**No**	**No**	**No**
Total P ($\mu g \cdot L^{-1}$)	**63.0**	**34.2**	**30.2** (4.5-55.6)	**15.3** (4.7-29.7)	**5.5** (4.0-7.8)
Soluble Reactive P ($\mu g \cdot L^{-1}$)	**19.4**	**13.4**	**13.3** (6.2-30.1)	**3.7** (0.6-10.8)	**0.6** (0.5-1.0)
Total Nitrogen ($mg \cdot L^{-1}$)	**1.04**	**1.04**	N/A	N/A	N/A
Total Ammonia N ($mg \cdot L^{-1}$)	**0.04**	**0.07**	**0.03** (0.00-0.23)	**0.02** (0.00-0.07)	**0.008** (0.004-0.012)
Total Nitrate N ($mg \cdot L^{-1}$)	**0.09**	**0.27**	**0.03** (0.005-0.09)	**0.04** (0.01-0.10)	**0.235** (0.19-0.26)
pH	**6.38**	**8.00**	**5.57** (4.76-7.52)	**6.95** (6.19-8.97)	**8.1** (8.0-8.2)
Specific Conductance ($\mu S \cdot cm^{-1}$)	**307**	**221**	**47** (14-131)	**134** (54-207)	**180** (159-196)
Total Suspended Solids ($mg \cdot L^{-1}$)	**4.40*** (N/A)	**14.83*** (N/A)	**15.5** (2.01-32.75)	**2.1** (0.25-7.0)	**0.8** (0.6-1.3)

Table 3 Changes, in hectares, of aquatic habitat and emergent vegetation in Wye Marsh and Matchedash Bay over 8 decades. All areas were calculated from digitized aerial images of the wetlands. Mean annual Georgian Bay water levels were obtained from the Canadian Hydrographic Service.

	Year	Aquatic Habitat (ha)	Dense Emergent Vegetation (ha)	Total area (ha)	% Aquatic Habitat	Mean Annual Georgian Bay Water Level (m a.s.l.)
Wye Marsh	1930	153.44	332.97	486.41	31.55%	176.65
	1965	95.58	372.48	468.06	20.42%	175.92
	1976	151.11	336.56	487.66	30.99%	176.90
	1987	86.83	392.51	479.35	18.11%	176.97
	1995	66.25	425.49	491.74	13.47%	176.53
	2002	53.20	434.61	487.81	10.91%	176.12
	2008	57.38	430.44	487.83	11.76%	176.01
Matchedash Bay	1931	409.44	411.21	820.65	49.89%	176.12
	1965	356.58	532.75	889.34	40.10%	175.92
	1973	562.18	330.46	892.63	62.98%	177.12
	1987	426.81	457.06	883.87	48.29%	176.97
	1995	483.56	413.01	896.57	53.93%	176.53
	2002	375.02	534.41	909.44	41.24%	176.12
	2008	330.21	557.83	888.04	37.18%	176.01

Table 4 Comparison of fish catch per unit effort (CPUE) in Wye Marsh and Matchedash Bay during 2012. CPUE were calculated from fish surveyed at 5 sites in Wye Marsh and 3 sites in Matchedash with Windermere traps set for 6 hours. % catch in Georgian Bay wetlands indicates the average proportion of each species caught using fyke nets in each wetland from 116 wetlands in Georgian Bay between 2001 and 2011 (Cvetkovic and Chow-Fraser 2011). Species are ordered base on residence status in wetlands based on Jude and Papas (1992), with wetland dependency decreasing as you move down the list.

Common name	Scientific name	Catch per unit effort		% catch in Georgian Bay wetlands 2001-2011
		Wye Marsh	Matchedash Bay	
Bluntnose minnow	*Pimephales notatus*	0.00	0.00	7.18
Brown Bullhead	*Ameiurus nebulosus*	0.40	0.00	8.33
Largemouth Bass	*Micropterus salmoides*	0.00	0.67	5.91
Golden Shiner	*Notemigonus crysoleucas*	0.00	0.00	1.21
Pumpkinseed	*Lepomis gibbosus*	2.00	27.33	38.94
Rock Bass	*Ambloplites rupestris*	0.00	0.33	5.61
Longear Sunfish	*Lepomis megalotis*	0.00	28.33	1.29
Yellow Perch	*Perca flavescens*	0.60	25.33	5.47
Shorthead Redhorse	*Maxostoma macrolepidotum*	1.20	0.00	0.17

Table 5 Comparison of diversity index scores for the avian and fish communities of Wye Marsh and Matchedash Bay. *p*-values correspond to the probability that the index scores are significantly different between sites ($\alpha = 0.05$). Bolded *p*-values indicate significantly different means between sites. IMBCI=Index of Marsh Bird Community Integrity (Deluca et al. 2004).

Community	Diversity Index	Mean values		*p*-value
		Wye Marsh	Matchedash Bay	
Marsh birds	Shannon's H'	2.02 ± 0.19	1.97 ± 0.10	0.6625
Marsh birds	E (IMBCI)	5.65 ± 0.83	5.79 ± 0.40	1.0000
Wetland fish	Shannon's H'	0.16 ± 0.15	1.02 ± 0.11	**0.0262**
Marsh Birds	Sorensen similarity	0.72		N/A
Wetland fish	Sorensen similarity	0.44		N/A

Table 6 Comparison of marsh bird abundances in Wye Marsh and Matchedash Bay during 2012. Data are sum of all birds surveyed in three 25-minute point counts in each wetland during May and June. Species are ordered based on their IMBCI scores with scores decreasing as you move down the list (DeLuca et al. 2004).

Common name	Scientific name	Abundance	
		Wye Marsh	Matchedash Bay
Virginia Rail	*Rallus limicola*	1	1
Marsh Wren	*Cistothorus palustris*	15	27
American Bittern	*Botaurus lentiginosus*	1	0
Great-Blue Heron	*Ardea herodias*	0	1
Swamp Sparrow	*Melospiza georgiana*	16	15
Trumpeter Swan	*Cygnus buccinator*	5	3
Common Tern	*Sterna hirundo*	2	0
Common Yellowthroat	*Geothlypis trichas*	3	3
Osprey	*Pandion haliaetus*	2	0
Red-Winged Blackbird	*Agelaius phoeniceus*	24	23
Tree Swallow	*Tachycineta bicolor*	18	0
Yellow Warbler	*Setophaga petechia*	4	4
Willow Fly-Catcher	*Empidonax traillii*	2	2
Caspian Tern	*Hydroprogne caspia*	5	8
Belted King Fisher	*Megaceryle alcyon*	0	1
Barn Swallow	*Hirundo rustica*	3	5
Song Sparrow	*Melospiza melodia*	6	4
Canada Goose	*Branta canadensis*	6	1
Wood Duck	*Aix sponsa*	4	0
Mallard	*Anas platyrhynchos*	1	2
Double Crested Cormorant	*Phalacrocorax auritus*	1	0
Bufflehead	*Bucephala albeola*	1	0
Mourning Dove	*Zenaida macroura*	1	0
American Crow	*Corvus brachyrhynchos*	1	2
American Robin	*Turdus migratorius*	0	1
Duck spp.	N/A	4	0

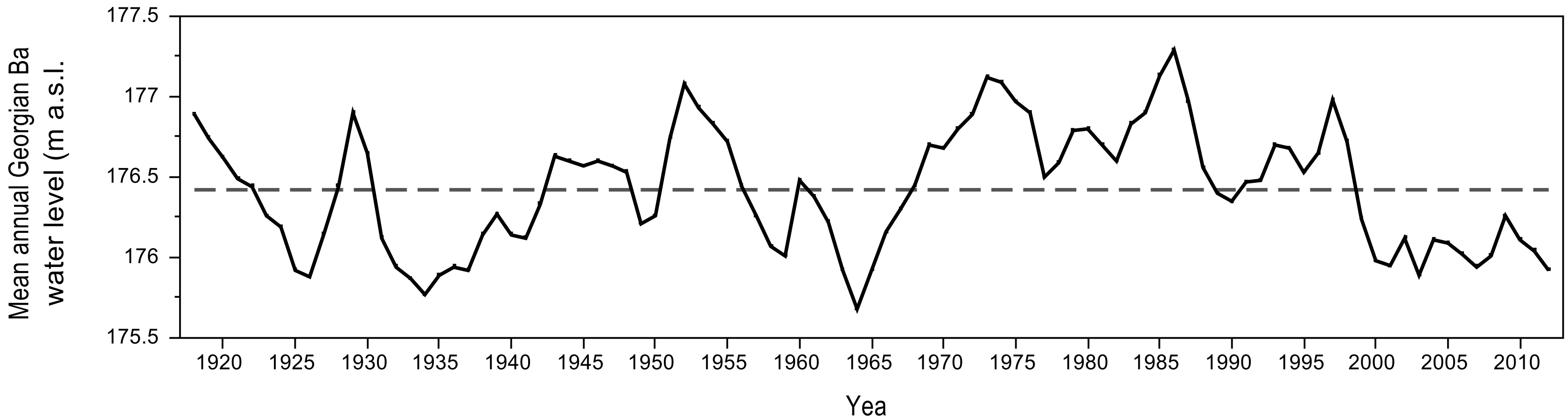

Figure 1 Mean annual Georgian Bay water levels through time (1918-2012). Dashed line indicates the long-term mean.

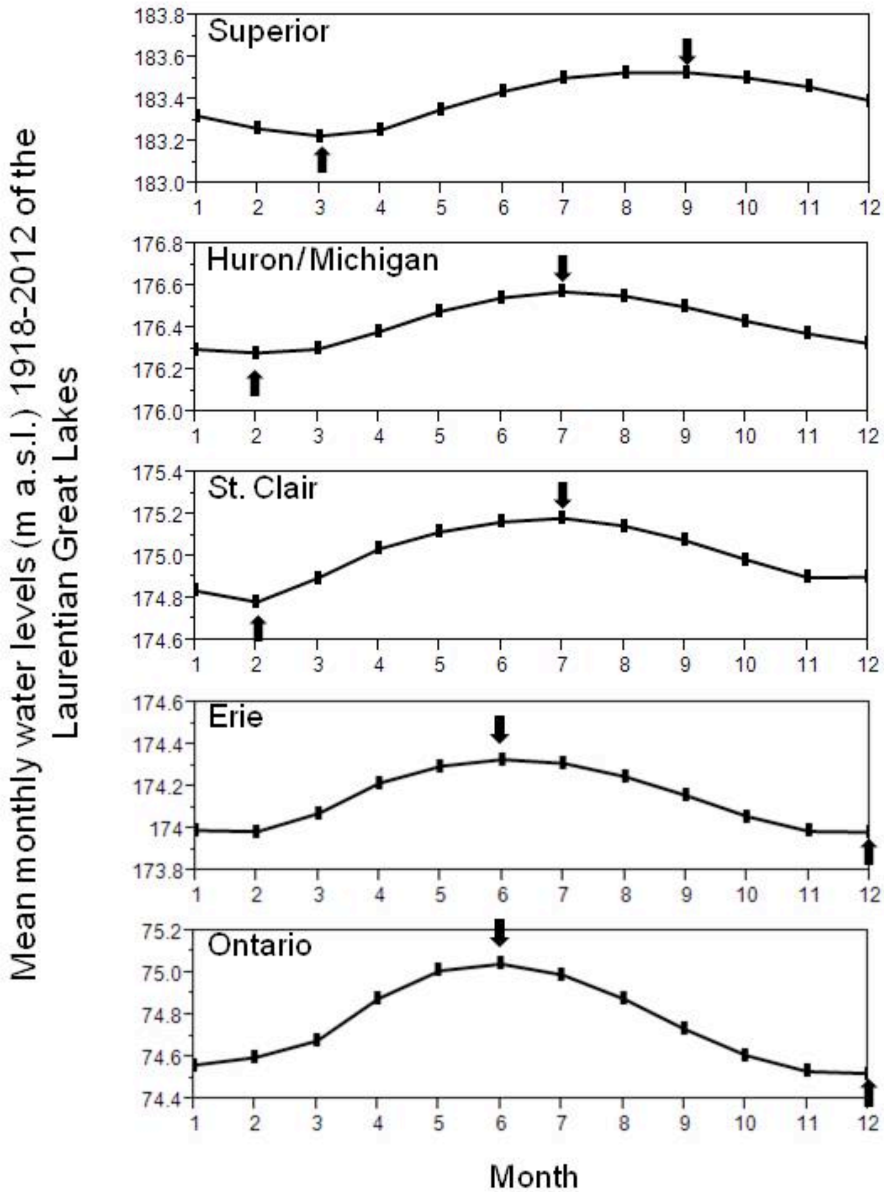

Figure 2 Hydrograph of mean monthly water levels in the Laurentian Great Lakes. Arrows indicate water level maxima and minima for each lake. Where available, data cover the period from 1918 to 2012. Obtained from the Canadian Hydrographic Service.

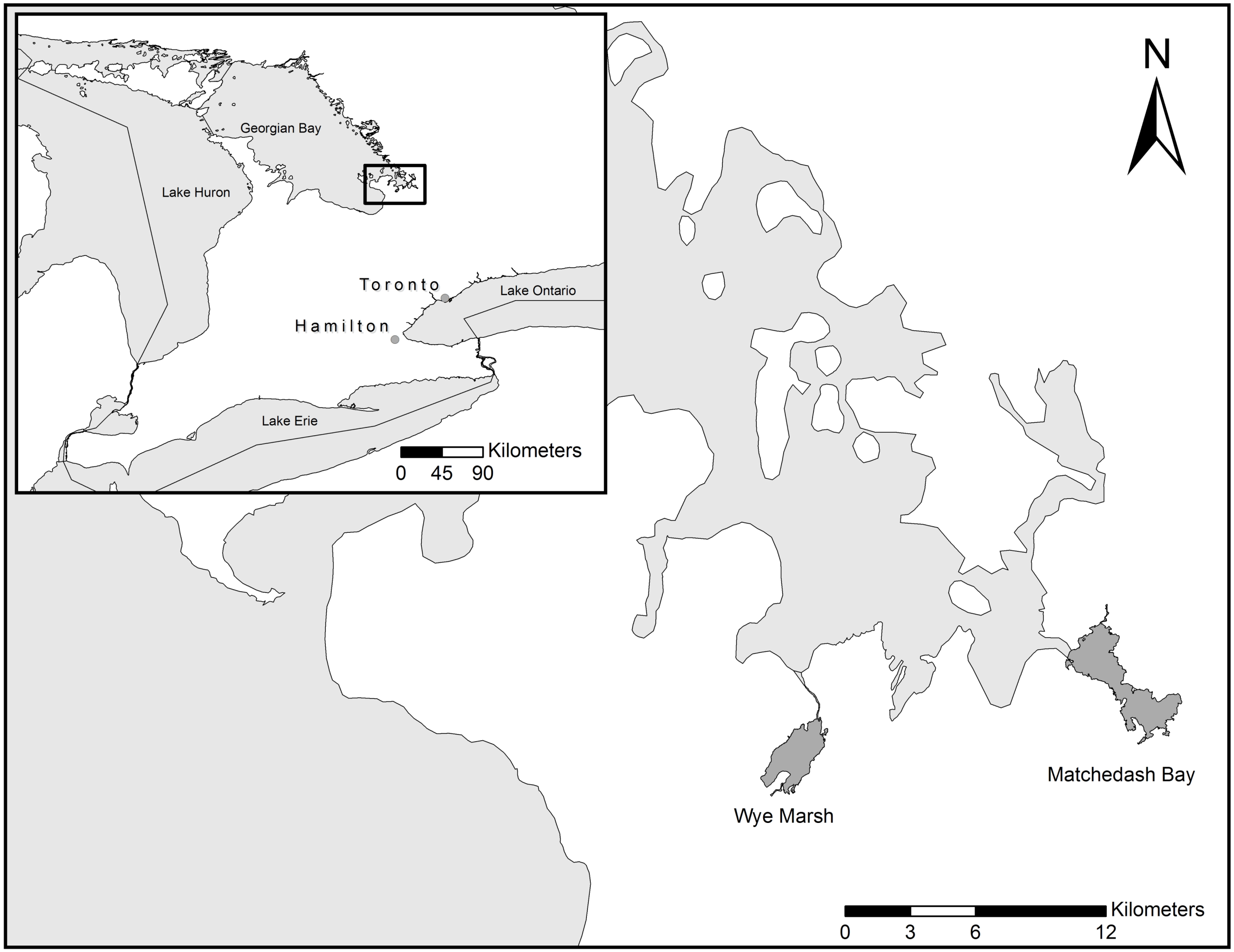

Figure 3 Location of Wye Marsh and Matchedash Bay in Southeastern Georgian Bay, Ontario.

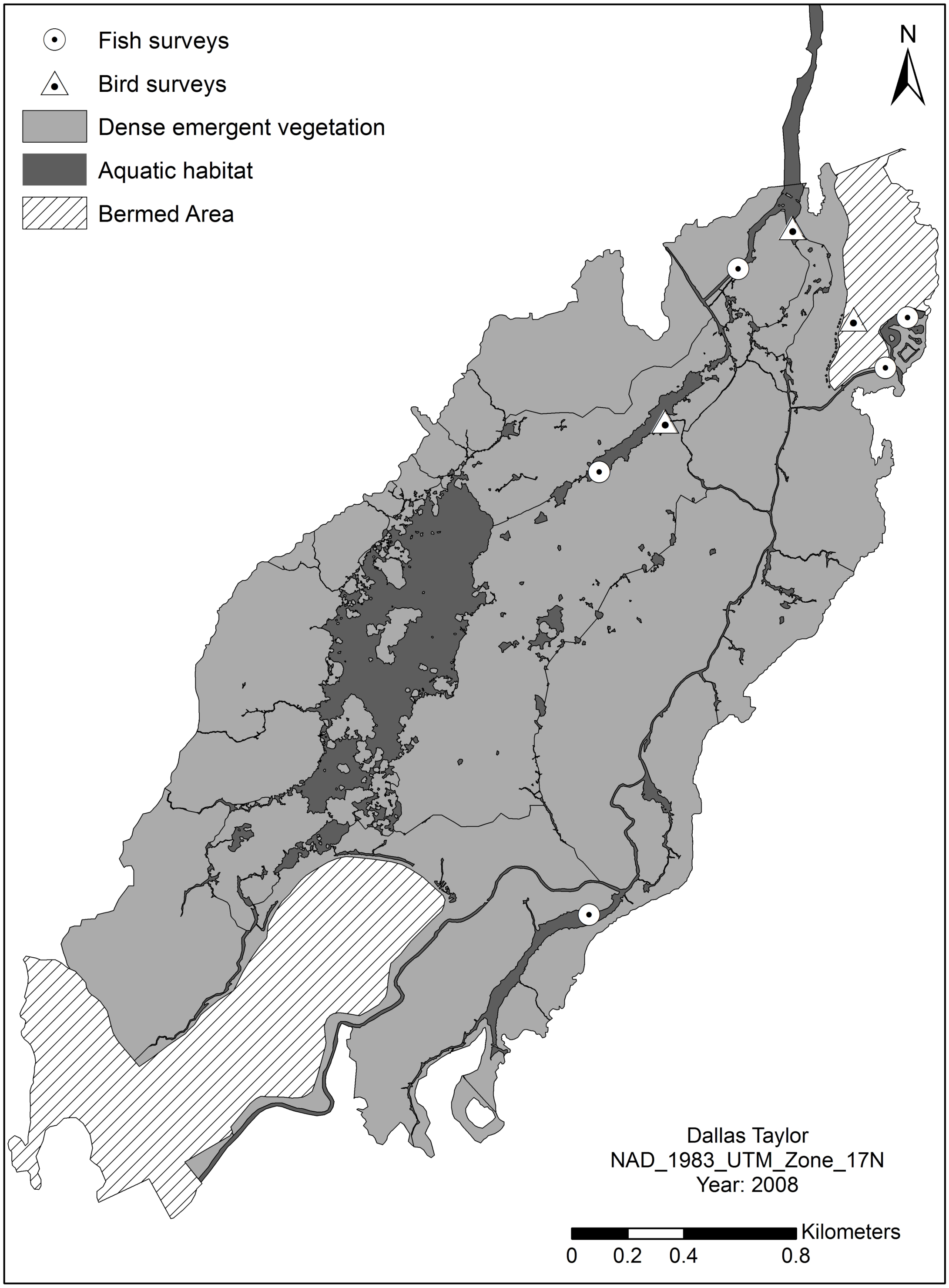

Figure 4 Map of Wye Marsh digitized from an aerial photo taken in 2008. Locations where fish and bird surveys were conducted in 2012, the bermed area (created in early 1990s), areas with dense stands of emergent vegetation and aquatic habitat are indicated.

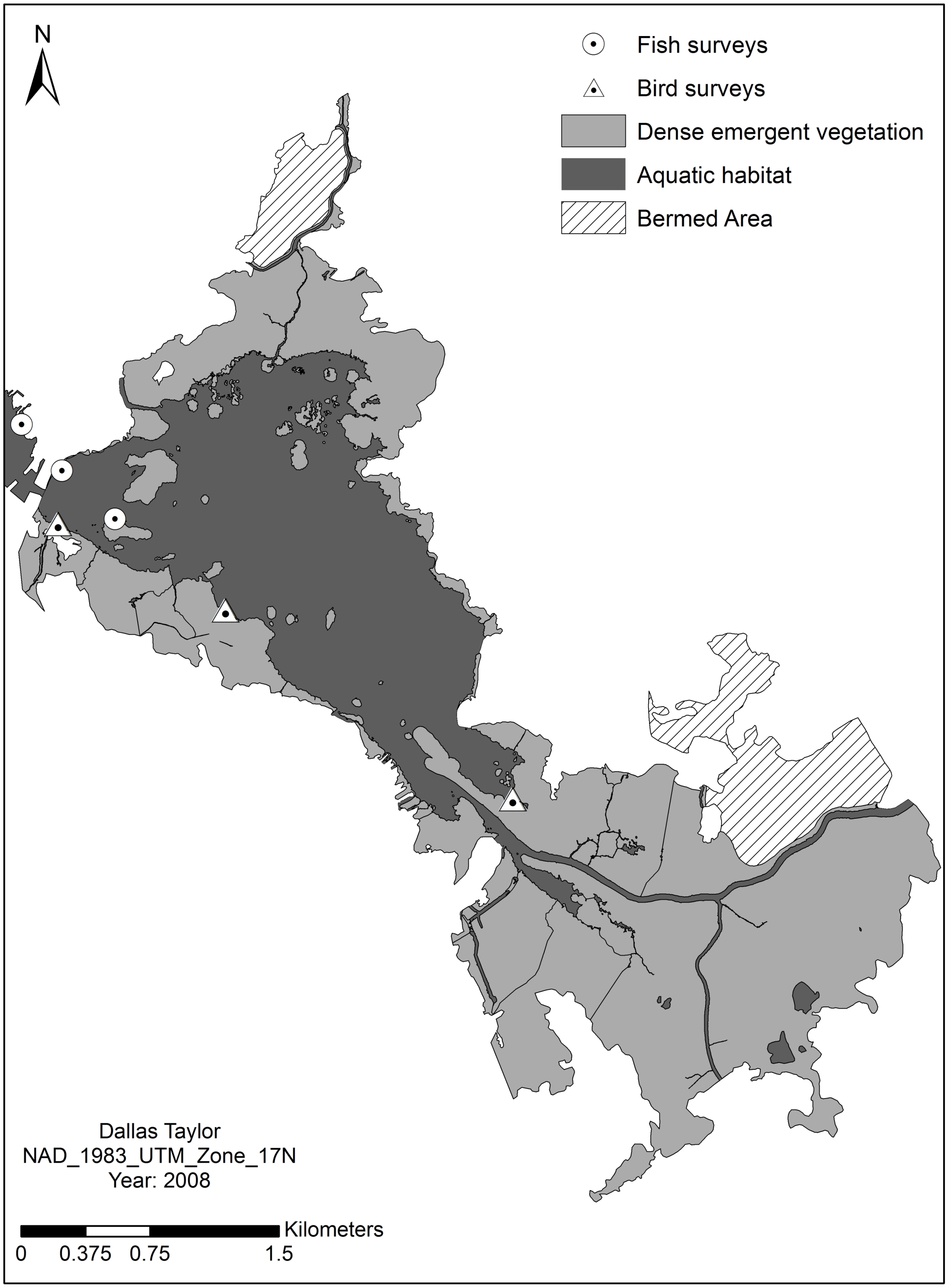

Figure 5 Map of Matchedash Bay digitized from an aerial photo taken in 2008. Locations where fish and bird surveys were conducted in 2012, the bermed area (created in early 1990s), and areas with dense stands of emergent vegetation and aquatic habitat are indicated.

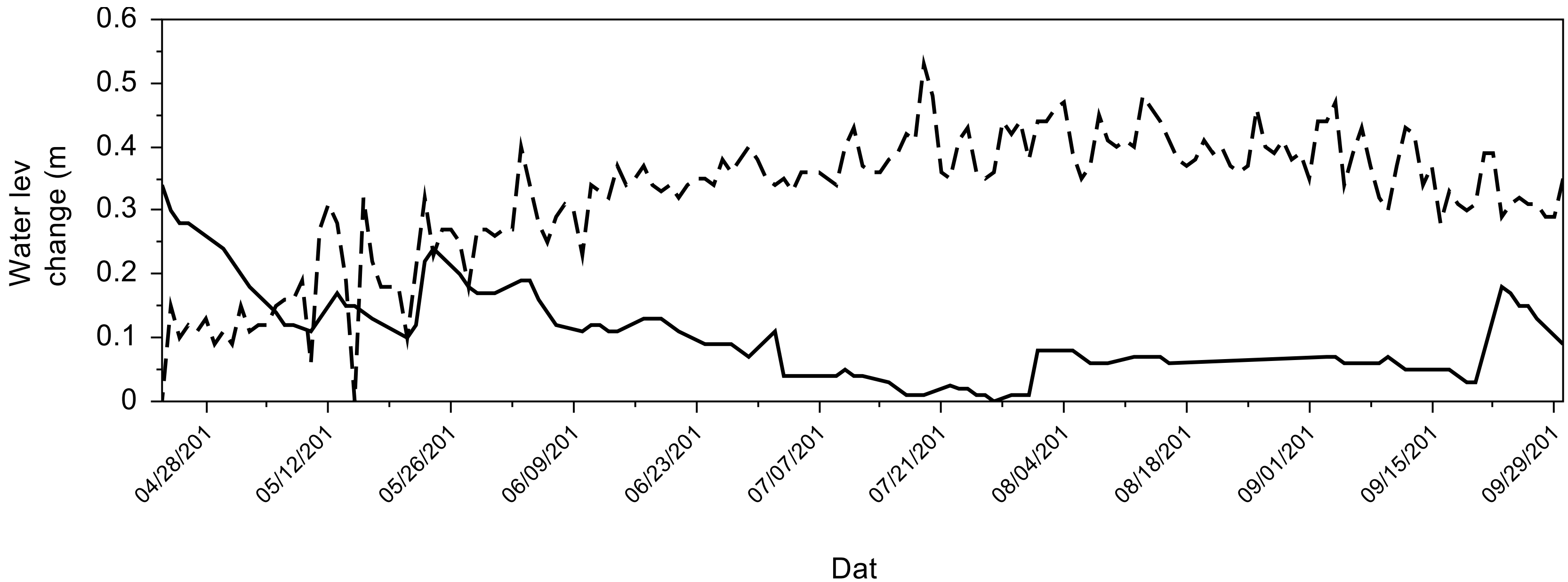

Figure 6 Relative change in water levels during the 2013 growing season between Wye Marsh (solid line) and Matchedash Bay/ Georgian Bay (broken line). Data collected in the morning at Wye Marsh and 07:00 for Matchedash Bay, from April 23, 2013 to September 30, 2013.

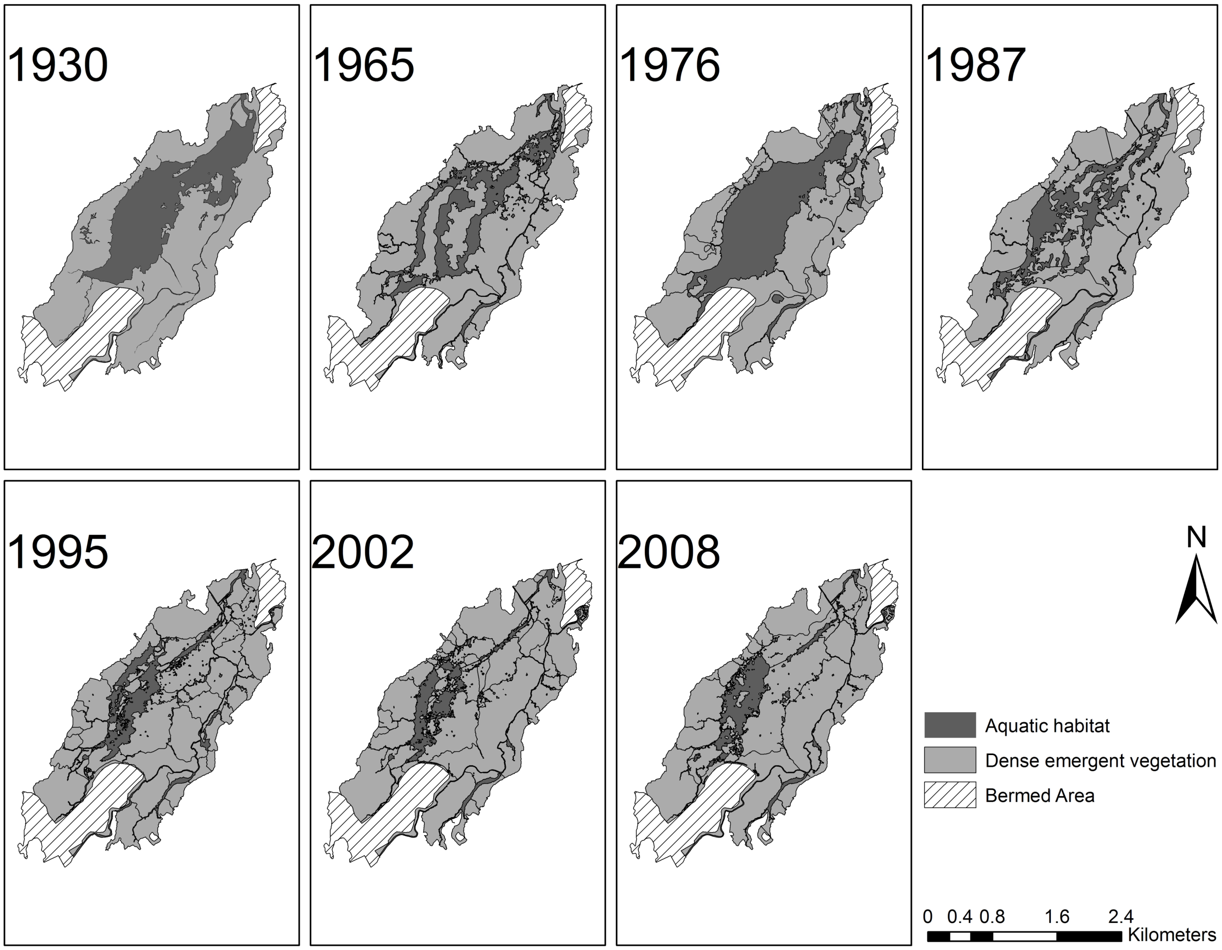

Figure 7 Changes in areal cover of dense emergent vegetation and aquatic habitat in Wye Marsh from 1930 to 2008. All maps were digitized from aerial photos acquired from April to October (National Air Photo Library, Canada and Ontario Ministry of Natural Resources).

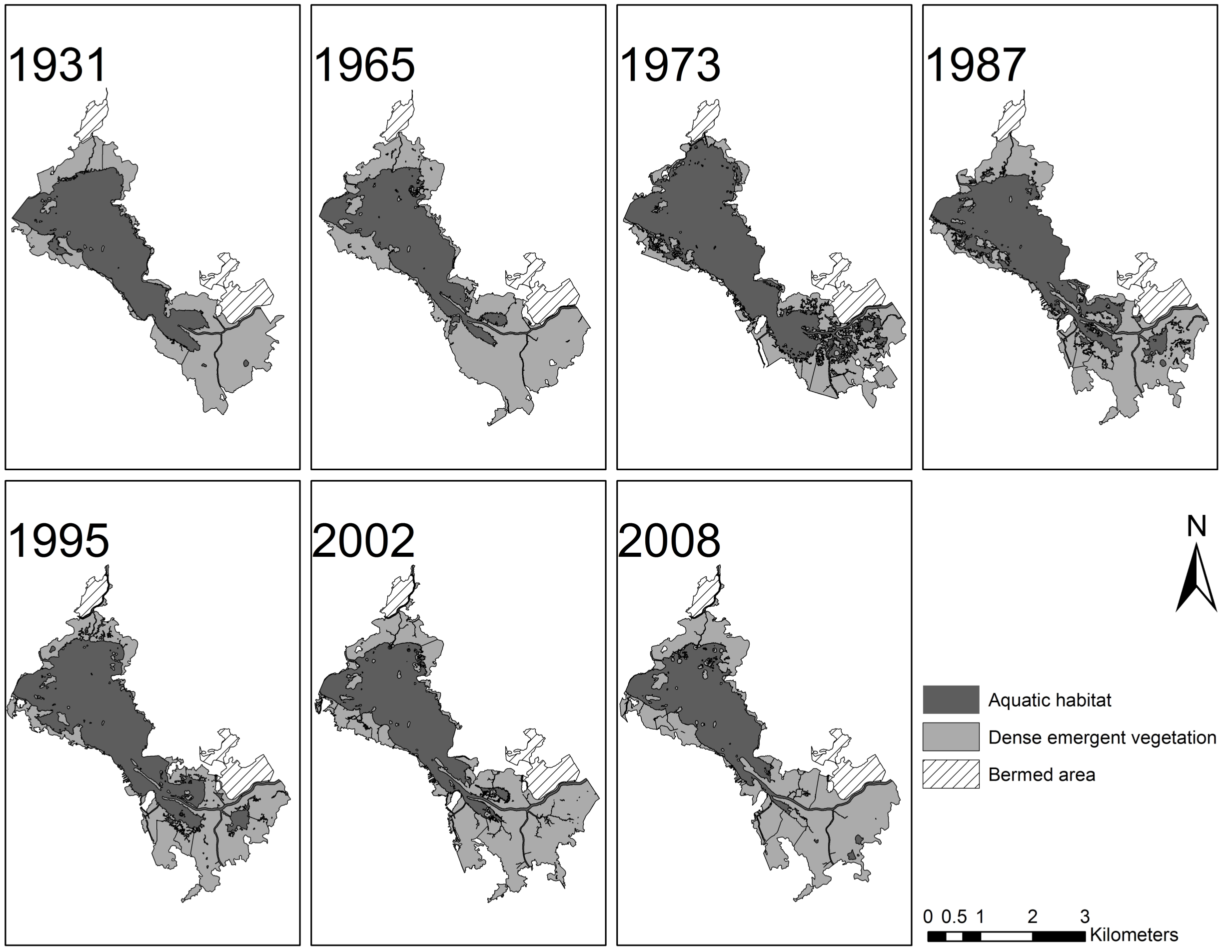

Figure 8 Changes in areal cover of dense emergent vegetation and open-water areas in Matchedash Bay from 1931 to 2008. All maps were digitized from aerial photos acquired from April to October (National Air Photo Library, Canada and Ontario Ministry of Natural Resources).

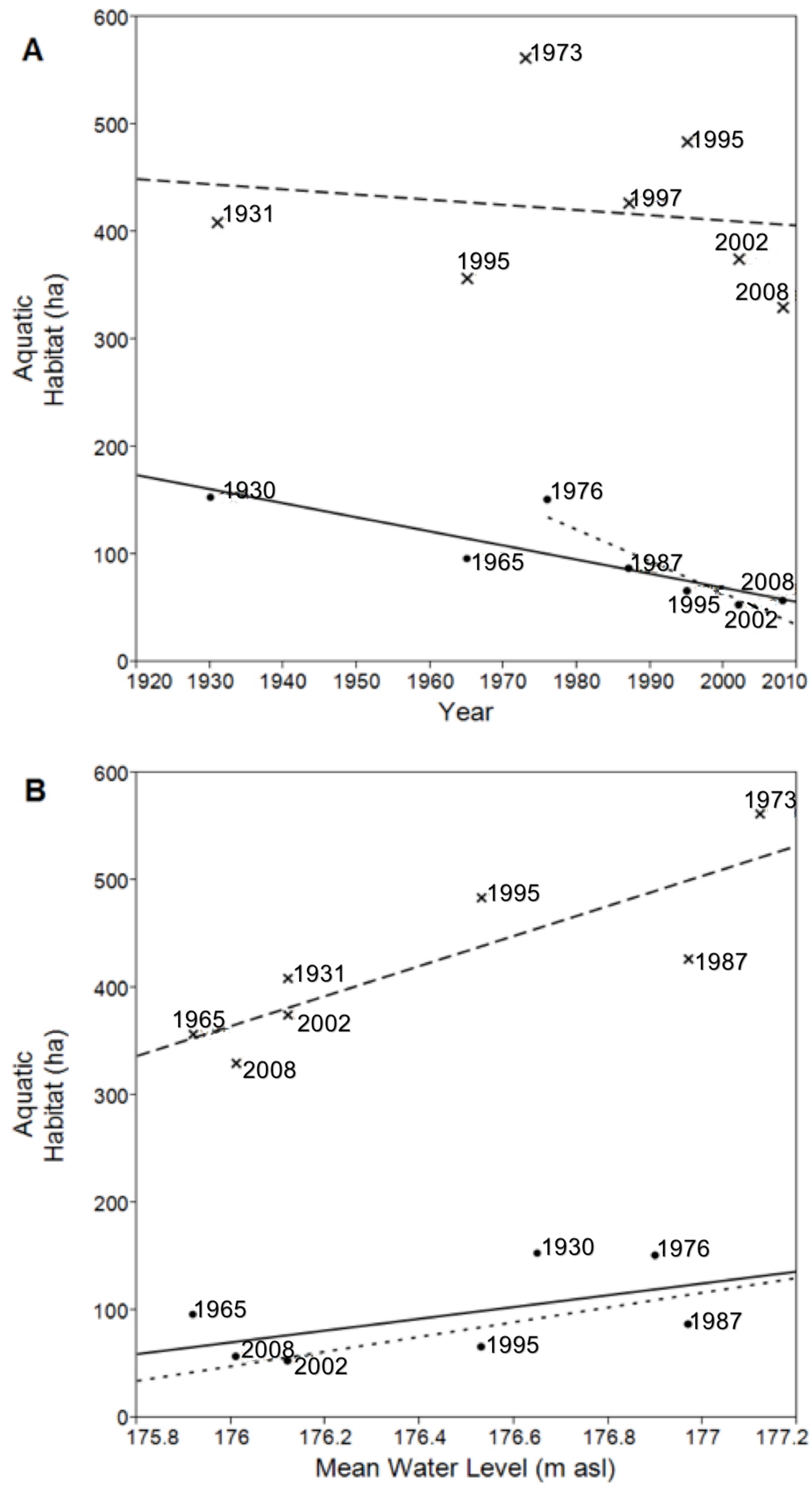

Figure 9 Plot of available aquatic habitat in Matchedash Bay (crosses) and Wye Marsh (circles) versus a) time and b) mean annual water level of Georgian Bay. Broken line is the least-squares regression line through Matchedash Bay data (1931 to 2008), while the solid line is a regression through Wye Marsh data (1930 to 2008). Also shown is a regression line through Wye Marsh data that exclude the 1930 and 1965 data points (dotted line).