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Influence of water-level disturbances on the performance of ecological indices for assessing human disturbance: A case study of Georgian Bay coastal wetlands

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ABSTRACT

In this study, we compare the performance of three ecological indicators (Water Quality Index (WQI), Wetland Macrophyte Index (WMI) and Wetland Fish Index (WFI)), to assess the impact of human activities on ecosystem health of coastal marshes in eastern and northern Georgian Bay (Lake Huron) over two decades (1999–2019), when there had been a minor change in human population (increase of 7%), but a marked difference in the pattern of water-level fluctuations. Lake Huron-Michigan is known to have 8 and 12-year oscillations in water levels, but between 1999 and 2019, water levels remained 0.5 m below the long-term mean for 14 years, and then abruptly rose nearly 1 m, remaining high for the next five years. We compared index scores of wetlands surveyed during 2003–2013 (Period 1; low-water years) with those surveyed during 2014–2019 (Period 2; high-water years). In Wilcoxon signed rank pairwise comparisons, mean WQI scores increased significantly from 1.50 to 1.96 between Periods 1 and 2, respectively ($p < 0.0001$); by contrast, WMI scores remained numerically and statistically the same (3.38 vs 3.38, $p = 0.42$), while WFI scores dropped slightly, but not significantly (3.65 vs 3.59, $p = 0.15$). We hypothesize that WQI scores increased because of diluting effects from increased volume of water in wetlands due to higher water levels. Given the unpredictable influences of climate change on the pattern of Great Lakes water levels, index scores based on water-quality variables must be cautiously interpreted when they are used to compare sites across different water-level scenarios.

1. Introduction

Multiple geomorphic types of wetlands exist along the Laurentian Great Lakes coastline, but the dominant form is the coastal marsh, which the National Wetlands Working Group defines as wetlands having gradually sloping offshore zones that are hydrologically connected to deeper waters of >2 m (Albert et al., 2005; National Wetlands Working Group, 1997). These systems have fluctuating water-levels and cyclical hydrological regimes that occur naturally, but that have also been influenced by anthropogenic alterations that include: installation of regulation structures in the St. Mary's River and the St. Lawrence River for hydropower and navigation, dredging of the St. Clair and Detroit Rivers to improve navigation, diversions from Lake Michigan at Chicago for municipal use and navigation, and diversions and obstructions in the Niagara River for hydropower and for operation of piers and marinas, respectively (International Joint Commission, 1993; Quinn, 2002). The net effect of such natural and anthropogenic hydrological disturbances varies with the Great Lake, as well as with the bathymetric characteristics of the wetlands in question (Wei and Chow-Fraser, 2008; Wilcox and Nichols, 2008).

In addition to lake-level alterations, human activities through urban and agricultural development have also led to major losses in quantity and quality of coastal marsh habitat, especially in Lakes Erie and Ontario (Host et al., 2019; Uzarski et al., 2017). These effects operate at the watershed scale, with levels of nutrient and suspended solids increasing in proportion to degree of land-use alterations in the wetland's drainage basins (Chow-Fraser, 2006, 1998; Harrison et al., 2019; Morrice et al., 2008). Since increased sediment loading leads to higher water turbidity, and increased nutrients preferentially benefit the planktonic and epiphytic algae, there is reduced light availability for macrophytes (McNair and Chow-Fraser, 2003). Consequently, degraded wetlands tend to have reduced cover and diversity of submergent aquatic vegetation (SAV) (Crosbie and Chow-Fraser, 1999; Lougheed et al., 1998). Since SAV provides critical habitat for many ecologically important fish

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species, including northern pike (*Esox lucius*), muskellunge (*Esox masquinongy*), largemouth bass (*Micropterus salmoides*) and yellow perch (*Perca flavescens*), the disappearance or reduction of aquatic plants has severe implications for the long-term health of Great Lakes fish communities (Casselman and Lewis, 1996; Jude and Pappas, 1992; Leblanc et al., 2014; Weller, 2018).

Several ecological indices have been developed specifically to assess the health status of coastal marshes as impacted by human disturbance in the Great Lakes (Cooper et al., 2018; Harrison et al., 2019; Uzarski et al., 2017; Wilcox et al., 2002). One family of indices, which include the Water Quality Index (WQI; Chow-Fraser, 2006), the Wetland Macrophyte Index (WMI; Croft and Chow-Fraser, 2007), and the Wetland Fish Index (WFI; Seilheimer and Chow-Fraser, 2007, 2006), were developed based on the relationship between land-use alteration and water-quality impairment, and the cascading effects of degraded water quality on the SAV and fish communities. These indices have been applied to all five Great Lakes (Cvetkovic and Chow-Fraser, 2011), and have also been used to detect longitudinal changes in the health status of wetlands undergoing restoration (Croft and Chow-Fraser, 2007; Thomasen and Chow-Fraser, 2012). Since these indices were developed specifically to reflect changes in human activities, they have been effective in tracking the health status of wetlands that are primarily influenced by anthropogenic disturbances.

Within the Great Lakes basin, coastal marshes in the eastern arm of Lake Huron, called Georgian Bay (GB) are very unique. First, they were formed in the world's largest freshwater archipelago with over 30,000 islands. Wetlands that have formed tend to be very small (over 85% of marsh habitat are <2 ha in area; Midwood et al., 2012), have shallow substrate or exposed granite on the Canadian Precambrian Shield, and have dystrophic waters with low nutrients and high water clarity (deCatanzaro and Chow-Fraser, 2011). Compared to other coastal marshes in the Laurentian Great Lakes watershed, GB wetlands are not only geomorphologically distinct, but have been subjected to relatively low or negligible human disturbances (Croft and Chow-Fraser, 2007; Host et al., 2019; Seilheimer and Chow-Fraser, 2007). Between 1976 and 2016, human population size in eastern GB, which includes permanent residents of the Township of GB, the city of Parry Sound, Parry Sound Centre and Parry Sound NE, and the Township of Archipelago, have fluctuated between 11,382 to 13,158, with a mean of 12,455 (Fig. 1). In recent years, the population increased by 6.6% from 12,345 in 2001 to 13,158 in 2016. Tourist visits from both Canada and the U.S. to the core GB area have gradually declined between 2001 and 2006, and this region has not been as popular a tourist destination as other areas in southern Ontario (Kantar TNS Canada, 2008).

Given the low degree of human disturbance, the major disturbances to GB wetlands have been physical in nature, including ice scours, lake-effect storms, windstorms and large deviations from the historic pattern of interannual water-level fluctuations (Burnett et al., 2003; Sly and Munawar, 1988; Fig. 2). The latter has been particularly evident in the past two decades. Lake Huron-Michigan is known to have long-term quasi-periodic fluctuations of 160 and 30–33 years (Baedke and Thompson, 2000), as well as short-term 8 and 12-year oscillations in water levels (Hanrahan et al., 2009). Between 1999 and 2013 however, water levels remained 0.5 m below the long-term mean for 14 years, and then abruptly rose nearly 1 m and remained high for the next five years (Fig. 2). Such a change in hydrological regime has been linked to the complex effects of Global Climate Change on precipitation and evaporation patterns (Carter and Steinschneider, 2018; Hanrahan et al., 2010; Li et al., 2012), although no study has linked the impact of these water-level changes to the health status of GB coastal marshes.

Such an increase in lake levels, can and should change the area, structure, and critically, the volume of the lake's embayments and coastal wetlands. Weller (2018) modelled the effect of 5 different lake-level scenarios on low-marsh habitat area and volume, where his findings concluded that habitat volume (as a function of wetland volume) changed significantly with changing water levels. Chow-Fraser (2005), Chow-Fraser (1999) showed that turbidity levels and total phosphorus concentrations in a coastal wetland decreased with increasing water level of Lake Ontario, presumably because the concentration of phosphorus and sediment became diluted with an increase in wetland volume. Such a reduction should affect the value of indices based on calculations of nutrient and suspended solids concentrations. To date, however, no study has examined the potential confounding effect of water-level changes on the performance of the WQI.

The recent change in water-level regime in the absence of obvious changes in human-induced disturbance offers a unique opportunity for us to assess the influence of water-level disturbance on the performance of all three ecological indicators. Given the relatively small change in population size in the region, these indices should not change significantly between 1999 and 2019, if the indices are insensitive to changes in water-level regimes. We hypothesize that the biotic (WMI and WFI) indices would not be sensitive to changes in water levels, since index scores for plant and fish taxa reflect their tolerance to degradation, rather than where they occur within vegetation zones in a wetland. For

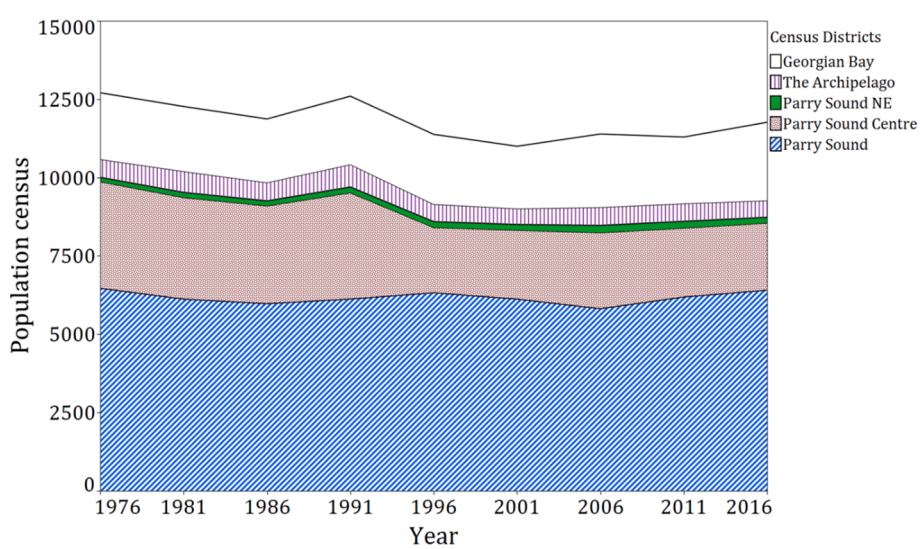

Fig. 1. Change in size of the regional human population of eastern Georgian Bay between 1976 and 2016. Census data were downloaded from the Canadian Census Analyzer (University of Toronto) and include census information for the Township of Georgian Bay, the township of the Archipelago, and the city of Parry Sound and surrounding districts.

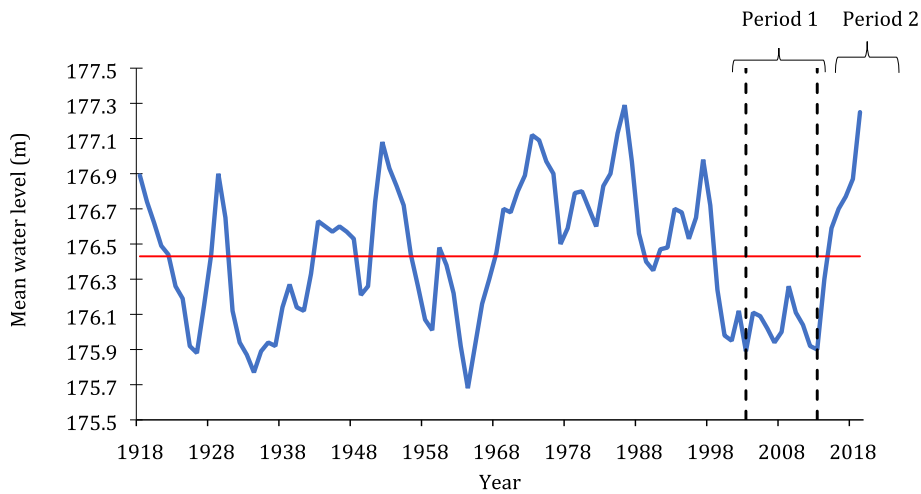

Fig. 2. Mean annual water levels of Lake Huron from 1918 to 2019, highlighting Period 1 (2003–2013; low-water years) and Period 2 (2014–2019; high-water years). Blue line = Annual mean water levels (m); Red line = Century mean for water levels (m). Water-level data were obtained from the Canadian Hydrographic Service. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

the WQI, however, we expect that an increase in water levels would result in diluted concentrations of pollutants, and thereby lead to higher WQI scores (Chow-Fraser, 2005, 1999). This assumes that the sources of pollutants have remained constant (i.e., cottage density and efficiency of septic systems have not changed). To test these hypotheses, we will use data from a long-term monitoring program, in which dozens of wetlands had been sampled with standardized protocols at least once during both periods of low and high-water levels. These periods have been defined as Period 1 (low-water years; 2003–2013), and Period 2 (high-water years; 2014–2019). Such a direct comparison of how index scores are affected by physical disturbance will facilitate accurate interpretations of long-term changes in the face of multiple stressors.

2. Methods

2.1. Wetland health indices

The WQI includes 12 parameters that are commonly used in long-term monitoring programs and was developed from data collected from 110 wetlands from all five Great Lakes (146 wetland-years, including GB). The parameters include turbidity (NTU), total suspended solids (TSS, mg/L), total inorganic suspended solids (TISS, mg/L), total phosphorus (TP, μg/L), soluble reactive phosphorus (SRP, μg/L), total ammonia-nitrogen (TAN, μg/L), total nitrate-nitrogen (TNN, μg/L), total nitrogen (TN, μg/L), specific conductivity (μS/cm), temperature (°C), and Chl-α (μg/L). Details regarding the development and calculation of this index have been published elsewhere (Chow-Fraser, 2006; Cvetkovic and Chow-Fraser, 2011). WQI scores tend to vary from −3 (indicating highly disturbed and degraded) to +3 (indicating undisturbed and excellent quality) but can be greater than this range.

The WMI and WFI were developed with data collected from 127 (154 wetland-years) and 60 Great Lakes coastal wetlands, respectively, located in all five Great Lakes, including GB. The index scores were based on presence of aquatic plant and fish assemblages and can range from a minimum of 1 to a maximum of 5. Complete details on the development of these indices and how to survey wetlands for plants and fish have been published elsewhere (for WMI: Croft and Chow-Fraser (2009), Croft and Chow-Fraser (2007); for WFI: Seilheimer and Chow-Fraser (2007)).

2.2. Field surveys

Water samples for nutrient, chlorophyll and suspended solids measurements were collected in a 1-L van Dorn water sampler at mid water-column depth in open water of each wetland site, so that the results would not be influenced by the presence of SAV. Following collection, any necessary pre-processing was performed in the field laboratory, and samples were frozen until they were analyzed in triplicate at the University lab. All water samples were processed within 6 h of collection and all analyses were completed within 6 months of sample collection. All details for processing have been documented in Chow-Fraser (2006). The physico-chemical variables were measured *in situ* with two different multi-parameter sondes; between 2003 and 2017, we used a YSI 6920 sonde equipped with a 650 display, whereas in 2019, we used the In Situ Aqua Troll 600, equipped with a communication port that sent data to a blue-tooth enabled mobile device. During 2018, we conducted a direct comparison with both instruments at three locations along an urban stream and did not find any consistent differences between the YSI and the In Situ with respect to any relevant parameters (unpub. data). In all years, water samples used for nutrients and suspended solids were also measured in triplicate for water turbidity using a Hach Portalab turbidimeter.

Since we were interested in assessing fish habitat, we targeted vegetation that occurred primarily in the inundated portion of the wetland, which included SAV, floating and emergent taxa; we would only encounter terrestrial species normally grown in the meadow zone (i.e., shrubs, trees) if water levels were high. Therefore, during Period 1, when water levels were low, we rarely encountered terrestrial taxa, but during Period 2, when water levels were high and the meadow zone was flooded, we encountered many more terrestrial taxa than we had in Period 1. We conducted all plant surveys from mid-July to mid-August when growth of SAV was abundant and flowering. We used a stratified random sampling method (Croft and Chow-Fraser, 2009), in which 8–12 quadrats (1 m^2) were sampled in all aquatic zones (in the meadow where soil was waterlogged, along the water's edge, in the emergent zone, the floating zone and the SAV zone up to approximately 4.0 m, where tops of SAV canopy could be reached with a rake from a boat). All plant species within or touching the quadrat were identified to the species level, whenever possible, but at least to genus. This was done except for freshwater sponge, which is not strictly a macrophyte, but lives symbiotically with algae and are excellent indicators of good water quality, due to their requirement for good light penetration to support algal photosynthesis (Lauer et al., 2001). This identification process was repeated until no new species were found in two consecutive quadrats.

During Period 2, we repeated the stratified random sampling method, but because of the high-water level, part or all of the meadow zone had transitioned from terrestrial to aquatic, and many meadow species such as grasses, sedges, shrubs, and trees were covered in up to

80 cm of water. Therefore, we were unable to re-sample in the same elevations that had been sampled in Period 1. In addition, we were unable to sample for some SAV species that were located too deep in the water column for us to reach the canopy with a rake from the boat. Depending on the bathymetry of wetlands, we were unable to sample along the water's edge because the water was abutting bedrock or forests, and sometimes the navigable portion of the wetland was almost entirely flooded meadow with a large zone of dead and dying trees which had not existed during low water levels in Period 1.

During Period 1, all fish surveys were conducted with three paired fyke nets set parallel to shore at 1 m (two pairs of large nets) and 0.5 m depths (one pair of small nets). The paired fyke nets were linked via a 7-m lead. A wing was attached to either side and angled 45° from the opening to funnel fish into the nets. The nets were set up and left open for 18–24 h, and then subsequently taken down and emptied. All fish captured were counted, identified taxonomically to species if possible, and their lengths measured (for the first 15 individuals of each species). Further details of the sizes of the nets and how this sampling was conducted can be found in Seilheimer and Chow-Fraser (2007). During Period 2, we attempted to deploy the paired nets using the same protocol we used in Period 1; however, because of the dying pine trees, alders, and sedges (*Carex* sp.) dominating water depths at 0.5 m, we could not set any of the small fyke nets. As well, due to increased water depths, it was also impossible for us to set our large nets within the SAV zone, previously inhabited by piscivores (such as esocids) and suckers (*Catostomus* sp.) (Cvetkovic et al., 2012). Instead, the nets were generally set in the flooded meadow, where we found emergent vegetation, sedges, and shrubs (e.g., sweet gale *Myrica gale*).

2.3. Data analysis

The mean index score between Periods 1 and 2, defined as the low-water period (2003–2013) and high-water period (2014–2019), respectively, were statistically compared for the three indices using Wilcoxon signed-rank tests ($\alpha = 0.05$). The non-parametric equivalent to a *t*-test was used due to our small sample size and non-normality of the dataset. Three standardized linear regressions using the least sum of squares methodology were formulated. The first two regressions evaluated the relationship between the mean WMI and WFI scores against mean WQI scores. The third regression assessed and quantified the effect of mean Lake Huron water levels (m) on mean WQI scores.

3. Results

Of the 69 wetlands sampled between 2003 and 2019, only 15% had been sampled solely for WQI; therefore, almost all sites associated with the WMI and WFI dataset had also been sampled for WQI. Only 2 wetlands had been sampled for WMI and did not have either corresponding WQI or WFI scores. Over the course of this study, we surveyed 38 wetlands for all three indices at least once in both periods (Fig. 3).

Regression-analysis of the biotic indices (WMI and WFI) against WQI scores by period was conducted to assess whether these indices deviated from the WQI (Fig. 4). Neither WMI nor WFI scores varied significantly with WQI scores in either period; however, slopes associated with lines of best fit for both biotic indices were much closer to zero in Period 2 than in Period 1 (Fig. 4). In other words, changes in WQI scores during Period 2 were not closely tracked by changes in WMI and WFI scores, indicating a separation in performance between the abiotic index and biotic indices.

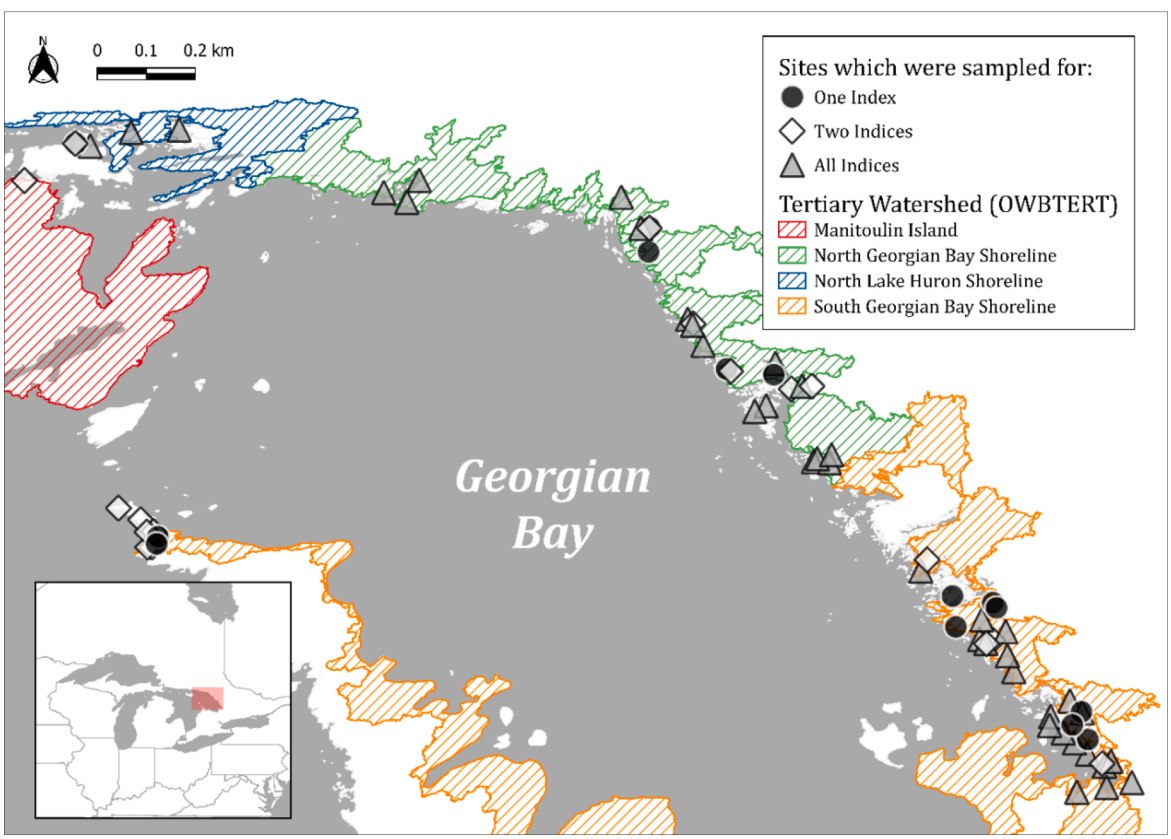

Fig. 3. The 69 total coastal wetlands sampled in Georgian Bay for both Period 1 (low-water years; 2003–2013) and Period 2 (high-water years; 2014–2019). Point symbol shape represents whether a given site was sampled for only one index, two indices, or all three. The coloured and hashed polygons represent the tertiary watershed of a given area, as delineated by the Ontario Ministry of Natural Resources and Forestry's (OMNRF) Ontario Watershed Boundary project (OWBTERT). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

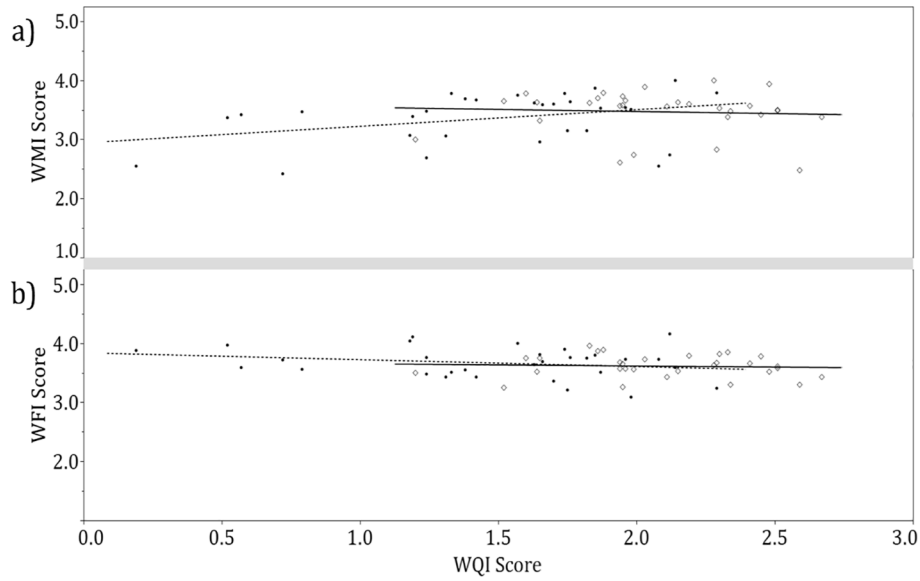

Fig. 4. Linear regression of a) WMI and b) WFI scores against WQI scores for 30 Georgian Bay coastal wetlands that had been sampled contemporaneously in both Period 1 (2003–2013; low-water years) and Period 2 (2014–2019; high-water years). Solid black dots = Period 1 scores; hollow diamonds = Period 2 scores; dashed line = Period 1 line of best fit; solid line = Period 2 line of best fit. All regressions were non-significant (F-test: p-value > 0.05).

Between Period 1 and Period 2, WQI scores increased significantly from a mean of 1.5 to 1.96 (Wilcoxon signed rank; $p < 0.0001$) (Fig. 5). Once categorized into one of the four tertiary watersheds of the GB area, all watersheds had an increased WQI score in Period 2 compared with Period 1 (Table 1). The WMI score remained constant from Period 1 to Period 2, with a mean score of 3.38 in both periods ($p = 0.42$) (Fig. 5). By comparison, the WFI scores decreased from 3.65 to 3.59 between periods, although this change was not statistically significant (Wilcoxon signed rank; $p = 0.15$) (Fig. 5).

A total of 68 macrophyte taxa were identified over the two time periods; 16 species were only encountered during Period 1, while 17 species were only found in Period 2 (Appendix 2). Of the unique species found in Period 1, 60% were submergent, consisting of two major genera, *Myriophyllum* (watermilfoil) and *Potamogeton* (pondweed). By comparison, 71% of unique species in Period 2, totalling 16% of all plant taxa identified, were those normally associated with the meadow zone (Midwood et al., 2012). Unlike the wetland plant community, we found no unique fish taxa in Period 2, but we did find 23 unique taxa in Period 1 (Appendix 3). Of these unique fish species, many were minnow and carp, which are small fusiform fish that travel in schools. Other unique species in Period 1 tended to be large benthivorous fish, like suckers, and predatory fish such as esocids, which had been observed outside of nets in deeper waters of wetlands when sampling took place during Period 2.

Table 1
Mean Water Quality Index (WQI) scores for 64 coastal wetland sites in GB that were sampled during Period 1 (low-water years; 2003–2013) and Period 2 (high-water years; 2014–2019), grouped by tertiary watershed. N = number of wetland sites sampled within a given watershed. The difference was calculated by subtracting Period 1 from Period 2.

Tertiary Watershed	N	Period 1	Period 2	Difference
Manitoulin Island	1	0.90	2.03	1.13
North Lake Huron Shoreline	4	1.61	2.16	0.55
North Georgian Bay Shoreline	24	1.52	1.86	0.34
South Georgian By Shoreline	35	1.49	2.01	0.52
Mean Total	64	1.50	1.96	0.46

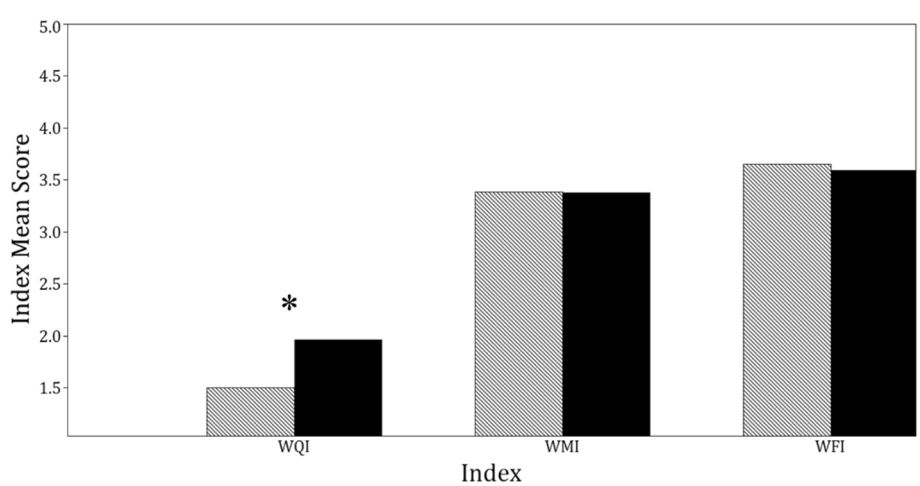

Fig. 5. Mean Index scores for Water Quality Index (WQI) (n = 64), Wetland Macrophyte Index (WMI) (n = 55), and Wetland Fish Index (WFI) (n = 43) for Period 1 (2003–2013; low-water years; bars with hashed grey lines) and Period 2 (2014–2019; high-water years; solid black bars). The asterix denotes any paired Wilcoxon signed-rank comparison between the two periods that was significant (p-value < 0.05).

4. Discussion

Seilheimer et al. (2009) cross-walked this family of indices by comparing WFI and WMI scores of 32 wetlands against corresponding WQI scores. They found a significant positive relationship between the two biotic indices with WQI ($R^2 = 0.84$ for WMI; $R^2 = 0.75$ for WFI) ($p < 0.0001$). Data used in Seilheimer et al.'s study included 16 sites in GB, sampled between 2002 and 2005. By contrast, we did not find any significant relationship between the two biotic indices with WQI (all p-values >0.05; Fig. 4). Whereas WQI scores indicated significantly improved conditions in wetlands (i.e., reduction of human disturbance), both WMI and WFI scores indicated no significant change. This uncoupling between biotic index scores and the WQI was hypothesized to be based on the dilution effects of almost a meter of additional water added to wetlands that have shallow mean depths of 0.5 to 0.75 m during low-water conditions (unpub. data). The mean WQI score rose 0.46 units, which reflected a considerable change in the 12 water-quality parameters. The negative relationship of increasing water levels on turbidity and suspended sediment levels has been documented in the past for a degraded urban marsh (Chow-Fraser, 1999, 2005). We compared mean values of individual parameters between Period 1 and 2 that could be affected by volume and found a similar trend. Levels off turbidity, TSS, TISS, TP, and SRP significantly decreased between Period 1 and 2 (Wilcoxon signed rank tests; $p < 0.005$), while concentrations of TN and TNN increased significantly ($p < 0.005$), yet TAN did not change significantly ($p > 0.05$).

To correct for the effect of increased water levels on the WQI score, we selected sites that are known to experience minimal human disturbance (as indicated by cottage and road density within 0.5 km of the wetland boundary) and developed a linear relationship between WQI and water level at the time of sampling (Fig. 6). This equation may be used to adjust WQI scores associated with two different water-level conditions, so that they are directly comparable. For example, we can determine the difference between WQI scores corresponding to two water levels, and then add this value to the score for the lower water level or subtract it from the score for the higher water level.

Compared to the WQI, the WMI accurately reflected the apparent absence of change in human disturbance. The sampling protocol for WMI requires that all wetland zones of a wetland be sampled during the survey, regardless of the prevailing water level. Since all taxa in the index developed by Croft and Chow-Fraser (2007) included SAV, emergent and floating taxa, meadow species were only noted when they were encountered and did not contribute to the index score. All three types of aquatic vegetation continued to be sampled during Period 2, although there was increased representation of meadow species (including trees and shrubs) along with simultaneous decrease in canopy-forming SAV taxa that were located at depths beyond our reach. Because of redundancies in the number of plant species in each quality category, however, species substitutions and deletions between period did not result in significant changes in WMI scores; the inclusion of additional meadow species in Period 2 did not alter the WMI scores either since their presence did not contribute to the index calculation.

We emphasize that the WMI score itself does not indicate anything about biodiversity or the taxonomic composition of the wetland plant community. We found significantly fewer SAV species in Period 2 compared with Period 1, even though the scores did not differ significantly. A likely explanation is that the meadow species and trees that had established in the wet meadow zone during the sustained low-water levels in Period 1 were not allowing the SAV species to transition back to the flooded portion of the marsh during Period 2. This may mean a loss of certain edge species that require a narrow range of water depths to grow, as well as a shift of other species into even greater water depths, where our sampling equipment cannot reach. During Period 2, the rapid reflooding of the wetland that had been previously the wet meadow zone, became occupied by dying pine trees (*Pinus* sp.), alder shrubs (*Alnus* sp.), and meadow species (Fig. 7). Given that the protocol required us to sample in all inundated zones, we ended up surveying and identifying meadow species in the flooded meadow zone. According to literature (Keddy and Fraser, 2000; Wilcox and Nichols, 2008), reflooding should transition into aquatic habitat within 2–3 years, but in this case, even after 5 years of flooding, the meadow species have not relinquished their foothold.

Similar to the WMI, the WFI was unaffected by differences in water levels between Period 1 and Period 2. Cvetkovic, Wei, & Chow-Fraser (2010), found that fish communities in coastal marshes were more closely linked to the structure of the macrophyte community rather than to water quality. Although our findings generally support their conclusion, we recommend conducting further studies to determine if some of the Period 1 fish species that were missing in Period 2 was an artifact of our fishing protocol. Because of difficulties in setting the large nets within SAV as described in the Methods, we did see some large fish swimming in deeper water (>2 m), and therefore know that they are still present in the wetland. As demonstrated here, use of a standardized fishing gear and protocol in long-term monitoring programs may not be appropriate for wetlands that experience large fluctuations in water levels.

The WQI has been validated by deCatanzaro et al. (2009), who found that the development of roads and cottages on land bordering GB

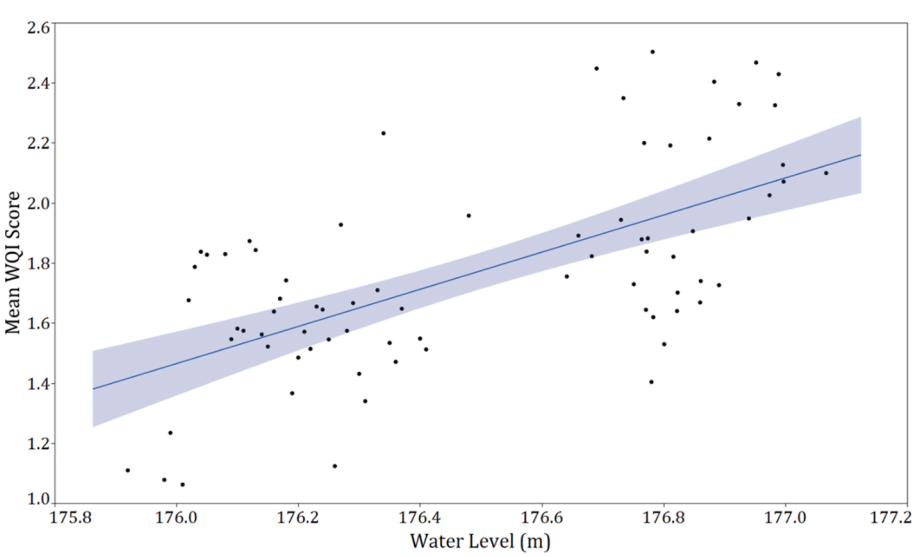

Fig. 6. Linear regression of mean WQI scores against water levels (m). Water levels are from the day the water sample was taken to calculate the respective WQI for that site. The line of best fit (blue line) is WQI = −107.4 + 0.6184 (water level x) and has an R^2 value of 0.399. The shaded blue area around the line of best fit is the 95% confidence interval. Water level data is from the Canadian Hydrographic Service, Collingwood station. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

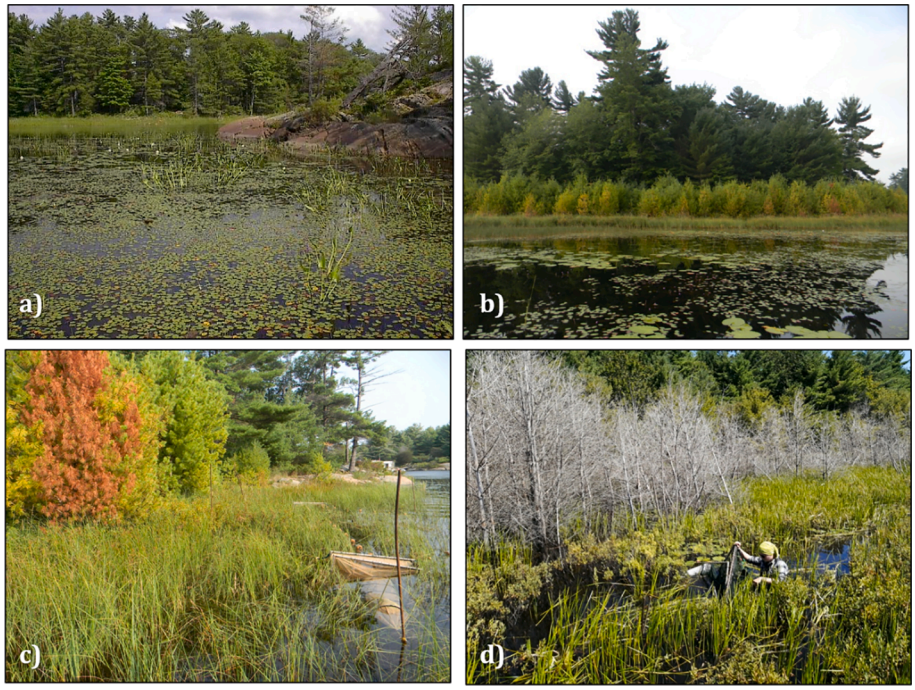

Fig. 7. Images depicting the changes in aquatic vegetation zonation for one of our sites, Moreau Bay, Georgian Bay, Lake Huron, Ontario. Images were taken in: a) 2003 (wetland dominated by pickerel weed and waterlilies), b) 2015 (inundated shrub line), c) and d) 2019 (setting nets in amongst *carex* species and dying alders).

wetlands, led to a predictable reduction in the WQI score. Using these three indices, Cvetkovic and Chow-Fraser (2011) compared the quality of coastal marshes across all five Great Lakes, and found that GB had the lowest proportion of wetlands in a degraded state, and the highest proportion in the very good and excellent categories (see index score equivalences in Table 2).

Unfortunately, the current study raises questions on utility of the WQI when wetlands are compared across different water-level scenarios. The effect of water level on water quality has not been considered in any relevant published studies of Great Lakes coastal marshes (Harrison et al., 2019; Morrice et al., 2008; Trebitz et al., 2007). Given the relative uniqueness of GB with respect to its current lack of human disturbance, the extent to which water quality in other Great lakes wetlands may respond to increased water level should be investigated separately.

Harrison et al. (2019) developed indices using physico-chemical measures and land-use metrics measured between 2011 and 2015 to infer long-term measures of human-induced pollution throughout the five Great Lakes. They collected water-quality data within ten dominant vegetation forms that may have varied in water depths. Data from their study may be used retrospectively to see if water measurements taken in wetlands with similar land-use metrics vary depending on water level. This would also require at least moderate-resolution bathymetric information that are unfortunately not always available for all Great Lakes.

Since all Great Lakes appear to be experiencing extremely high-water levels in recent years, greater effort should be made to develop an index, based on either biotic or abiotic variables, that will respond to human disturbance, while accounting for water-level impacts. To achieve this, we should further explore the ecological impacts of an altered hydrological regime on GB coastal marshes, using long-term data that capture the full range of water levels, as well as ranges of transition duration. While our regression of mean WQI scores against mean water-levels attempts to remove our hypothesized dilution effect, water level is arguably a proxy factor for the direct factor of influence, wetland volume. Therefore, until such a direct comparison can be made, we can only recommend using Fig. 6 as an interim measure to control for differential effects of water level on the WQI.

Table 2
Categories of wetland health and associated scores of the Water Quality Index (WQI), the Wetland Macrophyte Index (WMI), and the Wetland Fish Index (WFI) (Chow-Fraser, 2006; Croft and Chow-Fraser, 2007; Seilheimer and Chow-Fraser, 2006).

Category	WQI	WMI	WFI
Excellent	>2	>3.50	>3.75
Very good	1–2	3.0–3.5	3.25–3.75
Good	0–1	2.5–3.0	2.75–3.25
Moderately degraded	0 to −1	2.0–2.5	2.25–2.75
Very degraded	−1 to −2	1.5–2.0	1.75–2.25
Highly degraded	<−2	<1.5	<1.75

Investigations into the use of different techniques for fish surveys should also be pursued. For minnows, this may be the use of seine nets and for larger predatory fish in deeper waters, use of electrofishing may be a more accurate method (Cvetkovic et al., 2012). Future experimentation into the rapidly expanding field of remote operated vehicles (ROV) and camera technology for underwater survey techniques may eliminate this dependence of equipment efficacy with water levels (Sward et al., 2019).

CRediT authorship contribution statement

Danielle Montocchio: Conceptualization, Formal analysis, Investigation, Data curation, Writing - original draft, Writing - review & editing, Visualization. **Patricia Chow-Fraser:** Conceptualization, Methodology, Formal analysis, Resources, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2021.107716.

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Supplemental data

Appendix 1:

List of coastal wetlands (n = 69) sampled between 2003-2019. N_{PER1} = the maximum number of times a given site was sampled in Period 1. N_{PER2} = the maximum number of times a given site was sampled in Period 2. An X under each index represents that a given site was sampled for that specific index. Latitudes and longitudes were a mean of all sample points for a given site.

Site	Latitude	Longitude	N_{PER1}	N_{PER2}	WQI	WMI	WFI
Beaverstone	45.9836	-81.1474	2	1	X	X	X
Big Tub	45.2550	-81.6814	7	4		X	
Black Rock	45.0438	-79.9742	2	1	X	X	
Boat Passage	45.2901	-81.7166	9	4	X	X	
Charles Inlet	45.6466	-80.5672	3	2	X	X	X
Corbman Bay	45.4089	-80.3414	3	2	X	X	X
Cormican Bay	45.4065	-80.3088	2	1	X	X	X
Cove Island North	45.3137	-81.7618	10	4	X	X	
David's Bay	45.0458	-80.0019	3	3	X	X	X
Deer Island	45.9586	-81.2198	3	2	X	X	X
Dogfish Bay	46.0810	-81.7360	5	2	X	X	X
Francis Point	45.4144	-80.3335	3	2	X	X	X
Ganyon Bay	44.9203	-79.8176	2	1	X	X	X
Garden Channel	45.1855	-80.1221	2	2	X	X	X
Goose Neck Bay	45.2077	-80.1091	2	1	X	X	
Green Island	44.7871	-79.7469	3	3	X	X	X
Hay Bay 1	45.2409	-81.6839	9	4		X	
Hay Bay 2	45.2341	-81.6940	8	4	X	X	
Hay Bay 3	45.2340	-81.7025	8	4	X	X	
Henvey Inlet 5	45.8370	-80.6786	1	1	X		
Hermann's Bay	45.0863	-79.9973	3	3	X	X	X

Site	Latitude	Longitude	N_{PER1}	N_{PER2}	WQI	WMI	WFI
Hog Bay	44.7339	-79.8034	1	1	X	X	X
Hole in the Wall	45.5225	-80.4382	2	2	X	X	X
Inukshuk Bay	45.5568	-80.3867	1	1	X	X	
Iroquois Bay	46.0868	-81.6380	4	2	X	X	X
Isle of Pines	45.5979	-80.5191	2	1	X		
Jumbo Bay	46.0536	-81.8198	3	1	X	X	X
Kenerick Bay	45.6999	-80.5986	3	1	X	X	X
Key River	45.8857	-80.6768	1	2		X	X
Key River 1	45.8866	-80.6765	1	2	X		
Key River 3	45.8858	-80.6942	2	2	X	X	X
La Cloche - Sturgeon	46.0588	-81.8498	2	1	X		
La Cloche (inner)	46.0582	-81.8502	2	2		X	X
Lake St. Patrick	44.9802	-79.9321	1	1	X	X	X
Lily Pond 1	44.8710	-79.8126	2	3			X
Little Current	45.9824	-81.9543	1	2	X		X
Lost Channel	45.5934	-80.5105	1	1	X	X	
Matchedash Bay	44.7533	-79.6910	7	1	X	X	X
Miner's Creek	45.0617	-79.9487	1	2	X	X	X
Moon Bay 2	45.1342	-80.0575	2	1	X		
Moon River	45.1205	-79.9751	1	1	X		
Moon River 2	45.1100	-79.9675	3	1	X		
Moose Bay	45.0711	-80.0506	3	1	X		
Moreaus Bay	45.0124	-79.9453	3	3	X	X	X
Musky Bay	44.8111	-79.7819	3	1	X	X	X
Ni Bay	45.5113	-80.4614	3	3	X	X	X
North Bay	44.8955	-79.7921	2	1	X	X	X
North Bay 1	44.8976	-79.7938	1	1	X		
Oak Bay	44.7968	-79.7335	3	2	X	X	X

Site	Latitude	Longitude	N_{PER1}	N_{PER2}	WQI	WMI	WFI
Ojibway Bay	44.8877	-79.8569	2	2	X	X	X
Potato Island	44.7918	-79.7517	1	2	X		X
Prisque Bay North	45.6903	-80.5874	2	1	X	X	
Prisque Bay South	45.6877	-80.5885	3	2	X	X	X
Quarry Island	44.8341	-79.8083	3	1	X	X	X
Roberts Bay	44.8550	-79.8310	3	3	X	X	X
Russel Island East	45.2658	-81.6902	5	4	X	X	
Russel Island West	45.2647	-81.7039	9	4	X	X	
Shadow Bay	45.9491	-80.7344	6	2	X	X	X
Shawanaga River	45.5624	-80.3650	1	1	X	X	X
Sturgeon Bay Central	45.6099	-80.4193	1	2	X	X	X
Sturgeon Bay Channel	45.5862	-80.4224	1	1	X		
Sturgeon Bay South	44.7437	-79.7426	4	3	X	X	X
Sugar John	45.9387	-81.1726	2	2	X	X	X
Tadenac Bay 1	45.0373	-79.9892	2	1	X	X	X
Tadenac Bay 2	45.0373	-79.9880	2	1	X	X	
Treasure Bay	44.8703	-79.8591	2	3	X	X	X
Vennings Bay	44.8410	-79.7806	1	1	X		
Waterfall Bay	45.5628	-80.3450	1	1	X	X	
West Bay	45.4226	-80.3044	1	3	X	X	X

Appendix 2:

Macrophyte species sampled during both periods (n = 68), or only in Period 1 (low-water years (2003-2013); n = 16) or Period 2 (high-water years (2014-2019); n = 17) sorted alphabetically and by wetland zone morphology. Meadow species = located upland from the shore, periodically flooded but prefer moist soils; Emergent species = live near and in water's edge, vascular plants with deep/dense roots inundated with water; Floating species = plants with leaves that float on the water's surface, may be rooted or free-floating; Submergent species = aquatic plants with leaves that grow entirely underwater. U = niche breadth of species and, T = pollution tolerance of species; these values were calculated by Croft and Chow-Fraser (2007) and are used to calculate the overall WMI score of a given wetland. X = species was recorded for that period.

Species	Morphology	U	T	Period 1	Period 2
Acorus calamus	Meadow	0	0		X
Alnus viridis	Meadow	0	0		X
Calamogrostis canadensis	Meadow	0	0		X
Carex rostrata	Meadow	0	0		X
Carex sp.	Meadow	0	0	X	X
Dulichium arundinaceum	Meadow	0	0		X
Filipendula ulmaria	Meadow	0	0		X
Grass sp.	Meadow	0	0		X
Iris versicolor	Meadow	0	0		X
Myrica gale	Meadow	0	0		X
Pinus strobus	Meadow	0	0		X
Potentilla sp.	Meadow	0	0		X
Rhynchospora alba	Meadow	0	0		X
Eleocharis acicularis	Emergent	4	3	X	X
Eleocharis rostellata	Emergent	4	2	X	
Eleocharis smallii	Emergent	4	2	X	X
Equisetum fluviatile	Emergent	4	2	X	X
Juncus sp.	Emergent	4	1		X
Lysimachia terrestris	Emergent	1	1		X
Lythrum salicaria	Emergent	1	1	X	X
Phragmites australis amer.	Emergent	0	0		X
Phragmites australis australis	Emergent	0	0	X	X
Polygonum amphibium	Emergent	1	1	X	X
Polygonum sp.	Emergent	1	1	X	X
Pontederia cordata	Emergent	3	2	X	X

Species	Morphology	U	T	Period 1	Period 2
Sagittaria cuneata	Emergent	3	1	X	X
Sagittaria latifolia	Emergent	2	1	X	X
Sagittaria sp.	Emergent	2	1	X	X
Scirpus acutus	Emergent	4	2	X	X
Scirpus americanus	Emergent	5	3	X	X
Scirpus sp.	Emergent	4	1	X	X
Scirpus validus	Emergent	4	1	X	X
Sparganium androcladum	Emergent	4	3	X	
Sparganium chlorocarpum	Emergent	2	2	X	
Sparganium emersum	Emergent	1	2	X	
Sparganium eurycarpum	Emergent	3	2	X	X
Sparganium sp.	Emergent	2	2	X	X
Typha angustifolia	Emergent	1	1	X	X
Typha latifolia	Emergent	3	2	X	X
Typha sp.	Emergent	1	1	X	X
Typha x glauca	Emergent	1	2	X	X
Brasenia schreberi	Floating	4	1	X	X
Lemna minor	Floating	1	1		X
Lemna trisulca	Floating	2	2	X	
Nuphar pumila	Floating	5	2	X	
Nuphar variegata	Floating	2	1	X	X
Nymphaea odorata	Floating	2	1	X	X
Nymphoides cordata	Floating	5	3	X	X
Potamogeton natans	Floating	2	1	X	X
Sparganium fluctuans	Floating	4	2	X	X
Bidens beckii	Submergent	4	2	X	X
Callitriche sp.	Submergent	4	2	X	X
Ceratophyllum demersum	Submergent	1	1	X	X
Chara sp.	Submergent	3	2	X	X
Elodea canadensis	Submergent	2	1	X	X
Eriocaulon aquaticum	Submergent	5	3	X	X
Freshwater sponges	Submergent	5	3	X	X
Isoetes sp.	Submergent	4	3	X	X
Lobelia dortmanna	Submergent	5	2	X	
Myriophyllum alterniflorum	Submergent	5	3	X	
Myriophyllum heterophyllum	Submergent	3	2	X	
Myriophyllum sibiricum	Submergent	3	2	X	X

Species	Morphology	U	T	Period 1	Period 2
Myriophyllum sp.	Submergent	1	1	X	X
Myriophyllum spicatum	Submergent	1	1	X	X
Myriophyllum tenellum	Submergent	4	3	X	
Myriophyllum verticillatum	Submergent	4	1		X
Najas flexilis	Submergent	3	2	X	X
Neobeckia aquatica	Submergent	5	3	X	
Nitella sp.	Submergent	3	1	X	X
Potamogeton amplifolius	Submergent	4	2	X	X
Potamogeton crispus	Submergent	1	1	X	X
Potamogeton epiphydrus	Submergent	4	3	X	X
Potamogeton foliosus	Submergent	2	1	X	X
Potamogeton friesii	Submergent	2	1	X	X
Potamogeton gramineus	Submergent	4	2	X	X
Potamogeton illinoensis	Submergent	3	2	X	
Potamogeton obtusifolius	Submergent	2	1	X	
Potamogeton pusillus	Submergent	2	1	X	X
Potamogeton richardsonii	Submergent	3	2	X	X
Potamogeton robbinsii	Submergent	4	2	X	X
Potamogeton sp.	Submergent	1	2	X	X
Potamogeton spirillus	Submergent	5	2	X	X
Potamogeton vaseyi	Submergent	2	1	X	
Potamogeton zosteriformis	Submergent	3	1	X	X
Ranunculus longirostris	Submergent	2	1	X	X
Ranunculus sp.	Submergent	2	1	X	X
Sagittaria graminea	Submergent	4	3	X	X
Scirpus subterminalis	Submergent	5	2	X	X
Stuckenia pectinatus	Submergent	1	1	X	X
Utricularia cornuta	Submergent	5	3	X	X
Utricularia geminiscapa	Submergent	5	3	X	
Utricularia gibba	Submergent	5	2	X	X
Utricularia intermedia	Submergent	3	2	X	X
Utricularia minor	Submergent	3	2	X	X
Utricularia purpurea	Submergent	3	1	X	X
Utricularia rare sp.	Submergent	5	2	X	X
Utricularia sp.	Submergent	3	2	X	X
Utricularia vulgaris	Submergent	3	2	X	X
Vallisneria americana	Submergent	3	1	X	X

Species	Morphology	U	T	Period 1	Period 2
Zizania sp.	Submergent	4	2	X	X
Zosterella dubia	Submergent	2	2	X	

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Appendix 3:

Fish taxa sampled in both periods (n = 28), or only in Period 1 (low-water years (2003-2013); n = 23). There were no unique species in Period 2 (high-water years; 2014-2019). Species are sorted alphabetically and by functional feeding group. U = niche breadth of species and, T = pollution tolerance of species; these values were calculated by Seilheimer and Chow-Fraser (2006) and are used to calculate the overall WFI score of a given wetland. X = species was surveyed in that period.

Species	Functional Feeding Group	U	T	Period 1	Period 2
Ameiurus nebulosus	Benthivore	3	1	X	X
Aplodinotus grunniens	Benthivore	1	2	X	
Catostomus catostomus	Benthivore	5	3	X	
Catostomus commersonii	Benthivore	3	1	X	X
Cottus cognatus	Benthivore	4	3	X	
Etheostoma caeruleum	Benthivore	0	0	X	
Etheostoma microperca	Benthivore	4	1	X	
Etheostoma nigrum	Benthivore	3	2	X	X
Neogobius melanostomus	Benthivore	3	1	X	X
Pimephales notatus	Benthivore	3	1	X	X
Umbra limi	Benthivore	4	2	X	X
Culaea inconstans	Carnivore	3	2	X	X
Cyprinella spiloptera	Carnivore	2	1	X	X
Cyprinus carpio	Carnivore	2	1	X	X
Etheostoma exile	Carnivore	5	3	X	
Fundulus diaphanus	Carnivore	4	3	X	X
Gasterosteus aculeatus	Carnivore	2	2	X	
Labidesthes sicculus	Carnivore	4	2	X	X
Lepomis gibbosus	Carnivore	3	2	X	X
Lepomis macrochirus	Carnivore	3	1	X	X
Lepomis megalotis	Carnivore	5	3	X	X
Moxostoma macrolepidotum	Carnivore	4	3	X	
Notropis heterodon	Carnivore	5	3	X	X
Notropis heterolepis	Carnivore	4	2	X	X
Notropis rubellus	Carnivore	0	0	X	
Oncorhynchus tshawytscha	Carnivore	1	1	X	
Percina caprodes	Carnivore	3	2	X	
Pomoxis nigromaculatus	Carnivore	3	2	X	X
Pungitius pungitius	Carnivore	4	3	X	

Species	Functional Feeding Group	U	T	Period 1	Period 2
Sander vitreus	Carnivore	4	3	X	
Carassius auratus	Omnivore	1	2	X	
Chrosomus eos	Omnivore	5	3	X	
Hybognathus hankinsoni	Omnivore	1	2	X	
Luxilus cornutus	Omnivore	4	3	X	X
Notemigonus crysoleucas	Omnivore	3	2	X	X
Notropis hudsonius	Omnivore	2	1	X	
Notropis stramineus	Omnivore	4	2	X	
Notropis volucellus	Omnivore	5	3	X	X
Pimephales promelas	Omnivore	2	1	X	
Semotilus atromaculatus	Omnivore	3	1	X	
Ambloplites rupestris	Piscivore	4	1	X	X
Amia calva	Piscivore	4	2	X	X
Esox lucius	Piscivore	4	2	X	X
Lepisosteus osseus	Piscivore	5	3	X	X
Micropterus dolomieu	Piscivore	4	2	X	X
Micropterus salmoides	Piscivore	3	2	X	X
Morone americana	Piscivore	1	1	X	
Perca flavescens	Piscivore	3	2	X	X
Pomoxis annularis	Piscivore	1	1	X	X
Alosa pseudoharengus	Planktivore	2	2	X	
Esox masquinongy	Planktivore	4	3	X	
Notropis atherinoides	Planktivore	3	2	X	X
Noturus gyrinus	Planktivore	4	2	X	X