ACCOUNTING FOR WATER LEVELS WHEN USING WATER QUALITY TO ASSESS THE HEALTH OF GREAT LAKES COASTAL MARSHES

ACCOUNTING FOR WATER LEVELS WHEN USING WATER QUALITY TO ASSESS THE HEALTH OF GREAT LAKES COASTAL MARSHES

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LAY ABSTRACT

Determining the health of coastal marshes is important in understanding the effects of human-induced disturbances such as urbanization and farming. One of the widely used ecological indicators by conservation agencies for Great Lakes coastal marshes was recently found to be sensitive to an environmental factor that was not considered during the development. The wetland water quality index (WQI; Chow-Fraser 2006) was developed to assess the health of coastal marshes using 12 water- and sediment-quality variables that detects the effects of land-uses and land-cover alterations in the connecting watershed. Due to the recent drastic increase in water levels, the WQI scores were indicating improved wetland health conditions. However, these scores are instead reflecting the dilution of nutrients via the mixing of wetland water with lake water. My thesis investigates the wetland WQI by accounting for water levels to accurately assess human-induced disturbances on the health status of coastal marshes.

GENERAL ABSTRACT

Coastal marshes are at the unique interface between land and lake water, which are home to unique wildlife. There are numerous freshwater coastal marshes along the Great Lakes coast that each have distinct geomorphologies which are home to different vegetation and fauna communities. The determination of coastal marsh health conditions is a necessity to assess influences of environmental stressors such as human-induced disturbances upstream of wetlands. Water quality indices (WQIs) are used to indicate the impacts of watershed alterations on the health of coastal marshes. It typically uses nutrient and sediment variables in the water column to assess to relate that to marsh health conditions. As coastal marshes are subjected to the constant fluctuation of water levels, it was found to be a significant factor that impacted WQI scores, likely due to the dilution effect. With increasing water levels, WQI scores also generally increased which indicated improved wetland health. Thus, it is a confounding variable against land-use and land-cover (LULC) alterations within the watershed. This study is the first to evaluate the dilution effect by relating the change of WQI scores to the change in wetland volume and wetland area. As well, it is the first to test the confounding effects of varying watershed coverage in LULC, long-term lake level ranges, and the hydrogeomorphology of coastal wetlands.

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LIST OF ABBREVIATIONS

CPM: Cootes Paradise Marsh

DEM: digital elevation model

DGPS: differential global positioning system

FFNMP: Fathom Five National Marine Park

GB: Georgian Bay

HGM: hydrogeomorphology

IGLD85: International Great Lakes Datum (IGLD) 1985

LOE: (open) lacustrine open embayment

LOS: (open) lacustrine open shoreline

LPP: (protected) lacustrine protected embayment

LULC: land-use/land-cover

MASL: meters above sea level

SAV: submerged aquatic vegetation

SOLRIS: Southern Ontario Land Resource Information System

SUBarea: subwatershed area

UNESCO: United Nations Educational, Scientific and Cultural Organization

WDepth: water depth at sampling site

WETarea: wetland area

WETvol: wetland volume

WL: water level

WOpening: width of wetland opening

WQI: Water Quality Index

% AGR: percent agricultural land

% URB: percent urban land

% FOR: percent forested land

% WET: percent wetland

Note that WQI variable abbreviations are described in Table 1 of Chapter 1

DECLARATION OF ACADEMIC ACHIEVEMENT

The following MSc thesis includes two chapters prepared as manuscripts for submission in peer-reviewed journals. The general introduction explains the events leading up to this research. Chapter 1 was submitted for publication in Ecological Processes but was rejected pending revisions. Chapter 2 is presented as a manuscript but was not submitted for publication yet. Completed references for all chapters are found below. As first author on all chapters, I analyzed the data and wrote the manuscripts under the supervision and guidance of Dr. Pat Chow-Fraser. The field data used throughout this thesis relied on the contribution of many graduate and undergraduate students in the past 2 decades.

GENERAL INTRODUCTION

Great Lakes coastal marshes are ecologically important region home to a variety of freshwater plant and wildlife community that help maintain the health of the lakes. These areas are known to filter the water of nutrients and contaminants, buffer wave action, and mitigate flooding among the numerous ecosystem services they provide (EPA 2001). Due to the unique position of being between land and lake water, they are subjected to stressors from both sides. It is well-established that human-induced disturbances such as urbanization and farming in the drainage watershed leads to the influx of nutrients in the wetland. This lowers the water quality as it makes wetlands more eutrophic. However, the mechanism that is not commonly considered when studying coastal marshes is the water levels.

The constant fluctuation of lake levels is one of the main characteristics of coastal marshes. These conditions usually allow for increased biodiversity from the periods of inundation to periods of low waters which allows for different vegetation communities whether aquatic or meadow species to survive while others are killed off. Great Lakes water level fluctuations can range from a few millimeters in an hour to nearly two meters in a few years. Generally, the difference in annual water levels between consecutive years are around 20 cm in Lake Huron. However, between 2013 to 2015, the mean annual water levels jumped by about 70 cm. This sudden tremendous increase in water levels water observed to be rather impactful.

Water levels were a recent factor of interest as it was found to be significantly related to wetland water quality. The wetland water quality index (WQI) developed by Chow-Fraser (2006) was designed to assess the impacts of human-induced disturbances in the subwatersheds such as from the alteration of natural lands to urban or agricultural land. It used 110 coastal marshes along all of the 5 Great Lakes coast, sampled between 1998 to 2002, but primarily during 2000 and 2001. During these two years, the water level fluctuations were within the typical range of around 20 cm. It was not a variable of interest, instead the final 12 variables used were water- and sediment-quality variables.

However, the sudden spike in water levels observed in Lake Huron were speculated to be the determinant of the increase WQI scores of the long-term monitored coastal marshes in Georgian Bay. Most of these marshes sampled since 2003 were regarded as pristine and reference wetlands. When the water levels increased drastically in 2015 and onward, a wetland (Black Rock) was even found to have a WQI score that exceeded the range of scores according to the index, being that it was greater than 3. Although water levels have not been drastically below the long-term mean, it would be possible that WQI scores could be below -3 during extremely low water level periods. These scores do not reflect the actual health of the wetland.

The study by Montocchio and Chow-Fraser (2021) looked at the relationship between the WQI scores and other biotic indices, to which they did not find a consensus. They compared the WQI scores during the low water level period to the high water level period and found a significant difference. This was not reflected in the biotic indices, wetland macrophyte index (WMI; Croft and Chow-Fraser 2007) and the wetland fish index (WFI; Seilheimer and Chow-Fraser 2007), as they indicated wetland health conditions that were not significantly different between the periods. Thus, the WQI was detecting an environmental parameter that resulted in an inaccurate assessment of the wetland health. It was hypothesized that the WQI scores increased due to a dilution effect where an increase in wetland volume associated with increase water levels was falsely indicating healthier wetlands.

Alongside, a study by Croft-White and others (2017) found that WQI scores seemingly suggested improved WQI scores with increased urban land. This study was conducted in two regions along Lake Ontario, the Durham Region and the Quinte Region consisting of 22 marshes. The study analyzed the effects of land-use/land-cover (LULC) alterations on the WQI scores, which they found to be as expected. Such that wetlands with watersheds higher in urban land had lower WQI scores, whereas those with higher natural, forested, and wetlands had higher WQI scores. However, the increase in WQI scores with increased urban land was unexplained in the study. This prompt the investigation of the two confounding variables being the land-use/land-cover and the water levels on the WQI scores which is important for conservation agencies to accurately assess the restoration efforts or other changing environmental conditions on the coastal marshes.

Lastly, the hydrogeomorphic site types were studied here to evaluate the mixing of wetland water with lake water. There are studies done on riverine and barrier-protected coastal marshes in the Great Lakes that found a mixing of lake water to export the nutrients from the marshes into the wetland. The size of the wetland opening was also found to be important, especially for barrier-protected marshes which typically do not mix with lake water unless it was via groundwater flow from the lake (Albert et al. 2005). Since the water levels, LULC, and hydrogeomorphology of marshes were found to influence water quality, the assessment of if and to what extent the variables impact the WQI is required.

Thesis Objectives

The overarching objectives of this thesis are to investigate the dilution effects on WQI scores and determine if the WQI is influenced by the confounding effects of water levels and land-use/land-cover alterations to accurately assess coastal marsh health conditions.

In my first chapter, I compiled 24 reference lacustrine coastal marshes from Georgian Bay to compare with the wetland volume and wetland area. To calculate the wetland volume and wetland area, I created DEMs for each wetland using bathymetric data collected in field as well as land elevation data from Weller and Chow-Fraser's (2019) 10 m resolution DEM of Georgian Bay. The change in WQI scores to the change in wetland volume and wetland area were then calculated to assess if the WQI scores were affected by the water levels. This study will help with the understanding that the WQI is sensitive to water levels due to a dilution effect in lacustrine coastal marshes.

In my second chapter, I compiled data from 29 coastal marshes with variable LULC impacts, water level ranges, and hydrogeomorphology types. Similarly, I calculated the wetland volume and area under all water level scenarios according to the sampling date for each wetland. I calculated the WQI scores from the water quality data provided by the Central Lake Ontario Conservation Authority. I measured the approximate width of the wetland opening and the elevation at the water sampling point. One of the provided variables was the water depth at the sampling location, which was considered in the analysis. As well, I calculated the % LULC for 4 classes (wetland, forest, urban, and agriculture) for 3 time periods available in the 3 versions of the Southern Ontario Land Resource Information System. All these variables were used in a multivariate analysis to determine variables that best explained the variation. This information will help ecosystem managers accurately assess wetland health conditions by accounting for water levels, % LULC, and the hydrogeomorphology of the wetland which are all factors feasible to acquire or calculate.

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ABSTRACT

Ecological indices based on changes in concentrations of nutrient and suspended sediments have been used successfully to assess the impact of human activities on the ecosystem health status of Great Lakes coastal marshes. A recent study involving wetlands of eastern Georgian Bay (GB), however, uncovered an unexpected effect of water level on index scores. Here, we test the hypothesis that increase in volume and area of wetland related to increased water level had diluted water-quality variables and is the reason for increased index scores for GB wetlands. We also assembled data from two other regions of the Great Lakes to validate the positive effect of water level on index scores. These included Cootes Paradise Marsh (CPM), a degraded urbanized wetland of western Lake Ontario that was monitored over an 8-y period before and after carp exclusion that included 11 sites, and Fathom Five National Marine Park (FFNMP) in Lake Huron, where nine marshes were monitored annually from 2007 to 2017. In all cases, we found a highly significant positive relationship between WQI scores and water levels, despite large site-to-site variation in both FFNMP and CPM. We also confirmed that a change in index score was significantly related to a corresponding change in wetland volume for GB wetlands, consistent with the hypothesis that high water levels led to a dilution of water-quality variables. Based on these results, we strongly urge wetland researchers to account for the effect of water level on scores of ecological indices that rely on nutrient and suspended sediment concentrations, and to be cautious in interpreting improvements in index scores.

INTRODUCTION

Coastal marshes of the Laurentian Great Lakes occur at the interface between land and water and are known to support high biodiversity, partly because of fluctuating water levels that regenerate buried seeds of emergent taxa and limiting growth of submersed aquatic vegetation (SAV) during low water levels while killing dominant emergent taxa and promoting growth of SAV during high water levels (Armitage 2014; Keddy and Reznicek 1986; Smith et al. 2021). In the meadow zone, grasses, reeds, and shrubs grow and die with the changing water levels that maintain high vegetation diversity (Shantz 2018; WiIcox and Nichols 2008). These taxonomically diverse aquatic terrestrial plant communities serve as important habitat for birds, amphibians, fish, invertebrate, and wildlife (Fairbairn and Dinsmore 2001; Hecnar and M'Closkey 1998; Midwood and Chow-Fraser 2011; Snell-Rood and Cristol 2003; Streever et al. 1995). Human populations also benefit from ecosystem services provided by coastal marshes such as water filtration, nutrient and carbon sequestration, erosion and wave control, and recreational opportunities (Armitage 2014).

Despite the great ecological value of coastal wetlands, McCullough (1985) estimated that 35% of the wetlands along the Canadian shorelines of Lake St. Clair, Erie, and Ontario were lost. In some areas such as along western Lake Ontario, 73 to 100% of the original coastal marshes were lost (Whillans 1982). In addition to these losses, portions of coastal wetlands throughout the Great Lakes continue to be lost due to human encroachment. The literature established a significant link between anthropogenic stressors (urbanization and agricultural development) and degradation of coastal wetlands (Danz et al. 2007; Host et al. 2011; Morrice et al. 2008). Factors such as percentage of altered land (Chow-Fraser, 2006), human population density (Danz et al. 2007; Morrice et al., 2008), and road density (Danz et al. 2007) are all established indicators of cultural degradation that tend to increase concentrations of nutrients and suspended solids in natural ecosystems.

In response to the growing concern over loss of ecosystem services in these wetlands, much research was devoted to the development of biotic indices to reflect ecosystem health. These indices are widely used by governments to identify sites that need remedial actions to monitor health of wetlands and to track the effectiveness of restoration efforts (CLOCA 2009; Randall et al. 1996; Thomasen and Chow-Fraser 2012). They are based on predictable changes in the composition of biotic groups along a degradation gradient. Indices have been developed for communities of plants and algae (Croft and Chow-Fraser 2007; McNair and Chow-Fraser 2003; Wilcox et al. 2002), invertebrates (Lougheed and Chow-Fraser 2002), and fish (Minns et al. 1994; Seilheimer and Chow-Fraser 2006; Uzarski et al. 2005).

Another class of indices were developed using water-quality variables based on well-documented relationships between land-use alterations and the concentration of nutrients and suspended solids in the wetland (Morrice et al. 2008). These include the Water Quality Index (WQI; Chow-Fraser 2006) and the

Chem-Rank index developed by Harrison et al. (2020), which were significantly related to land-use and land-cover in wetland drainage basins. It is important to distinguish between these indices that use water-quality variables to reflect landscape-level disturbances (source indices) versus the biotic indices that reflect in-marsh conditions (response indices) that may be affected by more local processes such as a revegetation program or carp exclusion (e.g. Thomasen and Chow-Fraser 2012). Such differences have been described by Wang et al. (2019) who found incongruity between an index developed for land-use changes and one used to measure in-marsh conditions in humid regions of China because of improvements in wetland conditions following restoration efforts.

Regardless of the type of variables used, investigators have developed indices to ordinate wetlands according to the degree of human disturbance because human activities have been assumed to be the most important stressor on the ecological integrity of coastal marshes (Chow-Fraser 2006; Cvetkovic and Chow-Fraser 2011; Harrison et al. 2020; Lougheed et al. 2002; Randall et al. 1996). This assumption is appropriate for most regions of the Great Lakes shoreline, but in eastern and northern Georgian Bay, however, human disturbance is minimal (Cvetkvoic and Chow-Fraser 2011; DeCatanzaro et al. 2009) and water-level fluctuations exerted a greater influence on the ecology of coastal marshes (Fracz and Chow-Fraser 2013; Leblanc et al. 2014; Midwood and Chow-Fraser 2011). Recently, Montocchio and Chow-Fraser (2021) examined the influence of water-level disturbances on the performance of two biotic indices and the WQI. Although there were no significant effects of water level on the two biotic indices, they found that WQI scores increased significantly with water level in wetlands, and they hypothesized that this was due to a dilution of nutrients and pollutants in the wetland because of increased wetland volume corresponding to higher water levels.

The primary goal of this paper is to properly test Montocchio and Chow-Fraser's (2021) hypothesis using a subset of data from 24 coastal marshes in eastern Georgian Bay (GB). We hypothesized that a change in water levels would lead to a proportional change in wetland size (volume and area) which would result in lower concentrations of pollutants and thus an increase in WQI scores. As a secondary objective, we also wanted to test the generality of this water-level influence across wetlands in other regions of the Great Lakes basin, with different geomorphology and land-use impacts. For this objective, we assembled data from 11 sites in Cootes Paradise Marsh (CPM), a degraded urbanized wetland of western Lake Ontario that was site of a marsh-wide carp exclusion (Chow-Fraser 2005; Chow-Fraser et al. 1998), and from nine coastal marshes in Fathom Five National Marine Park (Parks Canada 2010) in Lake Huron. It is not our goal to evaluate the effects of water levels on ecosystem services in the long term, but rather to explain the sensitivity of indices that rely on water-quality parameters to water levels. Results of this study should inform management agencies of the need to exercise caution when using trends in index scores based on water-quality to make inferences on change in anthropogenic stresses on wetland health.

METHODS

Water Levels

Archived water-level data were accessed from Fisheries and Oceans Canada (DFO 2019). For Georgian Bay, we used data (adjusted for the IGLD85 datum) from the Collingwood station (#11500). We used the water level corresponding to the day when water samples had been collected; if the daily value was missing, we used the average for that month. For wetlands visited in 2020, we were able to use the hourly mean water level corresponding to the time when we collected depth information to determine wetland bathymetry. Mean water levels from June to July for FFNMP were estimated from the Tobermory station (#11690). We used equation 1 in Chow-Fraser et al. (1998) and data from the Burlington station (#13150) to estimate values for CPM, as there was no water-level recorder in the marsh itself. To examine interannual changes in water levels through time, we calculated mean water levels for data collected only during the growing season from May to August inclusive. In this paper, water levels are reported as meters above sea level (MASL).

Study Site Selection

All sites in GB are part of UNESCO's Georgian Bay Biosphere Reserve, which includes 347 000 ha and extends from Port Severn to the French River (TGB 2018). One site occurs within the French River Provincial Park, two in the Key River, one in Henvey Inlet, five in the Parry Sound and Pointe au Baril region, five in the

Musquash River region and ten in the Severn Sound region (see **Figure 1.1**). Approximately half of these were accessible by road and half only accessible by boat. The FFNMP wetlands included four in Hay Bay, which occur on the mainland and are road accessible; two wetlands on Cove Island, and three on Russell Island are only accessible by boat (see **Figure 1.1**). A monitoring program had been established to study the effectiveness of a marsh-wide exclusion program of common carp in CPM prior to the spring of 1997 (Chow-Fraser et al. 1998). CPM is 250-ha and includes a large open-water area, remnant marshes in several embayments, and at the outfall of a sewage treatment plant at the western end. Long-term monitoring stations had been established at open-water sites, in two embayments, at West Pond (the sewage lagoon), at the outfall of the Sewage Treatment Plant, and in tributaries that drain into CPM (see **Figure 1** in Chow-Fraser et al. 1998). For the present study, we only included data collected in open water sites and those located in vegetated sites within the marsh (i.e. 1, 2, 8, 9, 10, 12, 13, 14, 15, 16 and 17 in Chow-Fraser et al. (1998)).

Sampling

GB wetlands were sampled between late May to early September at least once in Period 1 (2003-2010) when water levels were low, and at least once in Period 2 (2015-2019) when water levels were much higher (see **Figure 1.2**). Water samples were collected in open-water at least 10 m away from emergent vegetation and where there is minimal submersed aquatic vegetation. In most cases, they were re-sampled at approximately the same vicinity; in some cases, however, it was not possible to re-sample in the same location because the higher water level had changed the shoreline configuration; in one extreme case (e.g. Key River), the site at high water level had moved 400 m inshore relative to that at low water level. We also visited some wetlands during the summer of 2020 to collect bathymetric data. The nine coastal wetlands of FFMNP were sampled by Parks Canada between the last week of June to mid-July annually from 2007 to 2017 as part of their long-term monitoring program (Parker et al. 2015). This program was designed to use Chow-Fraser's (2006) sampling protocol in open water. The CPM data came from a long-term sampling program that began in 1993 and continued annually until 2001 except in 1995. There was inconsistent sampling effort from year to year after 1998 due to resource constraints. For this study, we aggregated data by year according to habitat types within the marsh so as not to bias our results. Data from 1993, 1994, and 1996 correspond to period prior to carp exclusion, while those from 1997 to 2001 inclusive, correspond to the period following carp exclusion.

Water Quality Parameters and WQI Equations

Full details of methods used to obtain water-quality data to compute WQI scores have been documented elsewhere (Chow-Fraser et al. 1998 for CPM; Chow-Fraser 2006 for GB; Parker et al. 2015 for FFNMP). Chow-Fraser (2006) developed 9 predictive equations that could be used to compute WQI scores (see Table 5.6 in Chow-Fraser 2006). The equation we used to compute scores for GB and CPM included 12 variables (see Table 1) and which is summarized in **Eq. 1.1**. In all cases, the base of the logarithm is 10.

$$\begin{aligned} WQI = &+10.0239684 - (0.3154965 * \log TURB) \\ &(-0.3656606 * \log TSS) - (0.3554498 * \log ISS) \\ &(-0.3760789 * \log TP) - (0.1876029 * \log SRP) \\ &(-0.0732574 * \log TAN) - (0.2016657 * \log TNN) \\ &(-0.2276255 * \log TN) - (0.5711395 * \log COND) \\ &(-1.1659027 * \log TEMP) - (4.3562126 * \log pH) \\ &- (0.2287166 * \log CHL) \end{aligned} \qquad \textbf{(Eq. 1.1)}$$

Since Parks Canada only measured four variables, we used the **Eq. 2** (corresponds to Eq. 7 in Chow-Fraser 2006) to calculate WQI scores for wetlands in FFNMP:

$$\begin{aligned} WQI = &+9.2663224 -(1.367148 \times \log TURB) \\ &-(1.577380 \times \log COND) \\ &-(1.628048 \times \log TEMP) \\ &-(2.371337 * \log pH) \end{aligned} \qquad \textbf{(Eq. 1.2)}$$

Bathymetry Acquisition

We used a Lowrance depth sounder mounted on a canoe or boat to collect bathymetric information from GB wetlands. To correct for height between the transducer and the surface of the water, we added 10 cm and 20 cm when using either a canoe or boat, respectively. We uploaded the raw sonar (sound navigation and ranging) data to the BioBase website (BioBase n.d.) to produce bathymetric maps for each wetland. Mean daily (prior to 2020) or mean hourly (sampled in 2020) water levels were used to calculate elevation corresponding to boat/canoe transects. For majority of the sites, elevation data above the shoreline came from a 10-m resolution digital elevation model (DEM) of Georgian Bay (Weller and Chow-Fraser 2019), although for several wetlands, we had acquired elevation data with a differential global positioning system (DGPS). All points were inspected manually, and anomalous elevation points were removed.

Interpolation, DEM, and Volume and Area Calculation

All elevation data corresponding to points above and below the shoreline for each wetland were interpolated to produce a DEM (see **Figure 3**). We completed the interpolation using the Topo to Raster tool in ArcGIS Pro 2.8 (ESRI 2021). The options were as follows: type=point elevation, drainage enforcement=not enforced, primary data type=spot and output cell size=0.5 m. The DEM was clipped to the wetland outline, with the landward boundary defined by the historic high water-level mark and the lakeward boundary by an imaginary line linking two points at the opening of the wetland in the case of protected wetlands, or a boundary beyond the location where water samples had been collected in fringing wetlands.

We calculated wetland volume and area using the Surface Volume tool in ArcGIS Pro 2.8 and set the plane as the water level. For these calculations, we used the water level corresponding to actual dates when water samples had been collected in each wetland. Of the two measurement outputs generated, we chose the values under the column, “Area_3D". The volume calculated with this tool was verified with the traditional method (prismoidal formula) for calculating volume using contour lines. The formula is as follows where A and B are the area of adjacent contours (**Eq. 1.3**):

$$\text{Volume} = \text{stratum height}/3 * (A+B+\sqrt{(A+B)}) \qquad \textbf{(Eq. 1.3)}$$

Statistical Analysis

All analyses were carried out with SAS JMP v. 16 (SAS Institute 2021). We regressed WQI scores against water levels, volumes, and areas for GB wetlands; we also regressed the change in WQI scores against the change in volume and against the change in area. Similar linear regression analyses were performed between WQI scores and water levels for CPM and FFNMP. We first tested for a significant interaction between carp exclusion and water level on WQI scores; once we determined there was no significant interaction, we used an Analysis of Covariance to determine if there was a significant effect of carp exclusion on the WQI vs water-level relationship for the CPM data.

RESULTS

The duration when data were collected varied by region as did the range in interannual water-level fluctuations. The greatest difference between highest and lowest water levels were experienced in Georgian Bay. Over the 16-y period between 2003 and 2019, water levels in Georgian Bay varied from a low of 175.88 m in 2003 to a high of 177.11 m in 2019, corresponding to overall range of 1.23 m (**Figure 1.2a**). By comparison, over the 11 years between 2007 and 2017, water levels in Lake Huron varied from a low of 175.86 m in 2013 to a high of 176.74 m in 2017, with only a range of 0.88 m (**Figure 1.2b**). The range between lowest and highest water level in CPM was only 0.50 m over the 8 years from 1993 to 2001, with the lowest water level at 74.53 m in 1999 and the highest at 75.03 m in 1993 (**Figure 1.2c**).

We regressed WQI scores against water level for GB wetlands and found a highly significant relationship, with a slope of 1.23; WQI scores increased from 1.1 to 2.9 with a corresponding increase in elevation of 1.4 m (**Figure 1.4a**). On a site-by-site basis, all but one wetland in GB had a higher WQI score during the higher water-level period (**Table 1.2**). That wetland was Key River 3, which had a WQI score of just 0.04 lower in 2016 compared with that in 2006. When all WQI scores were grouped by water-level conditions, the mean WQI score during high water levels was 0.71 higher than that during low water levels.

The linear regression between the WQI score and the wetland volume or area was not significant (p=0.28 and 0.91, respectively; **Figure 1.4b** and **1.4c**); however, when we regressed the change in WQI score between Period 1 and Period 2 and the change in volume corresponding to the 2 water-level conditions, we found a highly significant relationship (p<0.0005; **Figure 1.4e**). We found a similarly significant relationship between the change in WQI score and the change in area between low- and high-water levels (p<0.0005; **Figure 1.4f**). We also confirmed that the change in WQI scores was also significantly related to change in water levels (**Figure 1.4d**). The maximum change in WQI scores increased by 1.3 units, with an increase in depth of 1.2 m between sampling dates. The maximum increase in wetland volume was $2 \cdot 10^5$ m^3, which occurred for big wetlands such as Oak Bay, while the maximum increase in wetland area was $6 \cdot 10^4$ m^2.

To determine the generality of the relationship, we regressed WQI scores against WLs for wetlands in the two other regions. In both cases, WQI scores increased significantly with WLs for FFNMP and CPM (p<0.05). The slope corresponding to data for CPM prior to carp exclusion was lower than that for data following carp exclusion (1.29 vs 1.86) but these were not statistically different (p>0.394). There was no significant difference in intercepts between data collected before and after exclusion (ANCOVA; p=0.111): however, holding water level constant, WQI scores were higher after carp exclusion than those prior to exclusion. For the nine coastal marshes in FFNMP, the slope was 0.81, with mean WQI scores increasing from 2.2 to 3.2 over an increase in WLs of 90 cm (**Figure 1.5b**).

DISCUSSION

There is overwhelming evidence that water levels are a major driver changing the state of coastal marshes which needs to be reflected in wetland WQIs. Our findings strongly support the hypothesis that in coastal marshes, changes in water levels will significantly influence WQI scores. This was demonstrated not only for coastal marshes of eastern Georgian Bay that are consistently found to be in excellent quality (Montocchio and Chow-Fraser 2021), but also for CPM, a large, urbanized wetland of Lake Ontario that is highly degraded, and for small coastal marshes in Lake Huron's FFNMP that are relatively undisturbed. Our results are also consistent with the hypothesis that the change in WQI scores is associated with the diluting effect of increased wetland volume that accompanies the change in water levels.

Had we interpreted the WQI scores without accounting for the positive effect of water level, we would have concluded that the impact of human disturbances in Georgian Bay had been reduced significantly between Period 1 and Period 2; and yet, there had not been any drop in human population nor decrease in cottage development (Montocchio and Chow-Fraser 2021). Similarly, human-induced stressors in FFNMP have not been reduced; to the contrary, the Municipality of Northern Bruce Peninsula has seen a 6.8% increase in population size from 2011 to 2016 (Statistics Canada 2017).

Cootes Paradise, a highly degraded marsh due to eutrophication from decades of sewage effluent discharge and the dominance of the invasive common carp, had undergone a carp-exclusion program beginning in the spring of 1997 that removed 90% of carp biomass (Chow-Fraser et al. 1998; Chow-Fraser 1998). Generally, WQI scores were higher during periods of higher water levels. Importantly, the WQI tracked the improved water quality after the carp exclusion, where WQI scores were approximately 0.2 units higher following the carp exclusion after accounting for water levels. This was likely due to improved water clarity from the drastic population decrease in common carp which are bottom feeders that contribute to sediment bioturbation. In 1999, despite the carp exclusion, the mean WQI score was the lowest recorded, primarily because it had experienced the lowest water level over the sampling period. The most parsimonious explanation for these variations is that the increase in water level in 1997 diluted pollutant concentration and when water levels dropped to low levels in 1999, the pollutant concentration greatly increased because of a reduction in volume. We obtained a significant effect of water level in CPM despite there being only an interannual difference of 0.5 m over the study period. This means that the WQI is very sensitive to water-level fluctuations and this factor must be taken into account for all long-term comparisons of dynamic ecosystems such as Great Lakes coastal marshes, especially in light of climate-induced extremes in water levels (Gronewold and Rood 2019).

Wetlands are nutrient sinks (Cheng and Basu 2017) and FFNMP is a popular tourist destination (Parks Canada 2017), and for this reason, we would expect these wetlands to have a higher influx of nutrients over time. Instead, what we saw was an overall decrease in conductivity and turbidity over time that led to higher WQI scores in these lacustrine wetlands. We suggest that these inflated scores are the result of the confounding effects of increased wetland volume associated with the higher water levels in Lake Huron in 2015 to 2017 relative to those prior to 2013.

We are aware of one long-term dataset from wetlands of Lake Ontario that can be used to investigate the effect of water level on WQI scores. Croft-White et al. (2017) assembled WQI scores (based on Eq. 6 and Eq. 7 in Chow-Fraser 2006) for 22 wetlands in two regions along the north shore of Lake Ontario that had been monitored annually from 2003 to 2014. They found that turbidity and pH decreased significantly while WQI scores increased significantly over time in both regions; they related the change in WQI scores to increased proportion of natural land in watersheds through time. They noted, however, that even though 13 of the 15 wetlands showed a decreasing temporal trend in cover of natural land and an increase in cover of urban land within watersheds, there was nevertheless a positive trend in WQI scores. Based on our results, we invoke an alternate hypothesis to explain the increased scores based on the diluting effect of increasing water level of Lake Ontario on water turbidity and pH. This hypothesis would explain why WQI scores have continued to increase despite a decline in proportion of natural lands in wetland watersheds.

There is limited availability of high-resolution bathymetric information for most of eastern GB, and consequently, we had to use boat-mounted sonars and DGPS to collect depth information. This is labour-intensive and is the primary reason for the limited number of wetlands we could include in this study. This method is also associated with unknown measurement error that could have contributed to unexplained error in our regression analyses. We do not know if wetland geomorphology may potentially influence how a particular choice of interpolation method may influence volume estimates, but this could be another source of error. The rules used to delineate wetland boundary could also affect volume calculation, and the degree of variation may also depend on wetland geomorphology.

In eastern GB, there are three broad geomorphological types as described by Albert et al. (2005) that include open lacustrine open shoreline (LOS), open lacustrine open embayment (LOE), and protected lacustrine protected embayment (LPP). These are lake-based wetlands that have little accumulation of organic sediment but that vary in terms of degree of exposure depending on the shape of the bedrock that either offers little or no physical protection (LOS, LOE) or substantial protection with narrow opening to the wetland (LPP) (see **Figure 1.6**). Since one possible explanation for the dilution may be that there is a greater exchange with water in the bay when water levels are high, the extent of such an exchange should be proportional to the size of the wetland opening. While we did not have data to test this properly, difference in WQI scores did not differ between wetlands with large openings versus those with more restricted openings. The width of wetland opening would likely influence the water exchange between the water within the marsh with the water within the lake (Albert et al. 2005).

Indices such as WQI have been used to assess the health status of wetlands across the Great Lakes basin at one time (Chow-Fraser 2006; Harrison et al. 2020; Host et al. 2011; Morrice et al. 2007; Uzarski et al. 2005). WQI scores have also been used in the Lake Huron Biodiversity Conservation Strategy to monitor wetlands in Lake Huron (TNC 2010). As well, the WQI has been used by various Conservation Authorities (Central Lake Ontario Conservation Authority and the Quinte Conservation (Croft-White et al. 2017) and Credit Valley Conservation (CVC 2010)) to manage wetlands across the north shore of Lake Ontario. These agencies have used WQI scores to track changes in wetland condition following conservation efforts, and they should account for the possible confounding effects of water level on WQI scores to avoid drawing the wrong conclusion about management actions.

CONCLUSIONS

The wetland water quality index (WQI) is sensitive to water levels even from small changes which confounds the interpretation of wetland health. Given the usefulness of the WQI and other similar indices, it is also timely for investigators to develop a method to standardize water-quality index scores to account for the significant effects of changing water level across time, and across study sites.

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Table 1.1: Explanation of acronyms, abbreviations, and units used in the wetland Water Quality Index (WQI; Chow-Fraser, 2006).

Variable label	Explanation	Units
Temp	Water Temperature	ºC
Turb	Water Turbidity	NTU
Cond	Specific conductance	µS/cm
pH	Potential Hydrogen	unitless
CHL	Chlorophyll-*a*	µg/L
TP	Total Phosphorus	mg/L
SRP	Soluble Reactive Phosphorus	µg/L
TAN	Total Ammonia Nitrogen	µg/L
TNN	Total Nitrate Nitrogen	µg/L
TN	Total Nitrogen	mg/L
TSS	Total Suspended Solids	mg/L
TISS	Total Inorganic Suspended Solids	mg/L

Table 1.2: WQI scores of Georgian Bay coastal marshes during at least one low and one high water-level date, and the difference between the mean scores. Arrows indicating the difference in WQI scores between Period 1 and Period 2.

Wetland	WQI score: Low water-level	WQI score: High water-level	Mean change in WQI score
Key River 1	0.66, 1.25	2.06, 2.93, 2.98	↑ 1.70
Black Rock	1.85, 2.22	3.22	↑ 1.19
Henvey Inlet 5	1.88	2.99	↑ 1.12
Oak Bay	1.12, 1.23, 1.57	2.01, 2.81	↑ 1.10
West Bay	0.57	1.64	↑ 1.07
North Bay 1	0.43, 1.21	1.83, 1.92	↑ 1.06
Shadow Bay	1.63	2.51	↑ 0.88
David's Bay south	1.92	2.78	↑ 0.86
Corbman Bay	1.24	2.03	↑ 0.79
Potato Island west	0.99	1.75	↑ 0.77
Charles Inlet	1.18	1.88	↑ 0.69
Miner's Creek	1.77	2.44	↑ 0.67
Green Island	0.91, 1.38	1.66, 1.77, 1.96	↑ 0.65
Venning's Bay	1.23	1.58, 1.94	↑ 0.53
Robert's Bay	1.44, 1.59, 1.80	1.87, 2.03, 2.47	↑ 0.51
Sturgeon Bay Central	1.41	1.91	↑ 0.50
Tadenac Bay	1.73	2.22	↑ 0.49
Ganyon Bay	1.43, 1.72	1.95	↑ 0.38
Ojibway Bay	1.56, 1.83	1.78, 2.11	↑ 0.25
Cormican Bay	1.86	2.02	↑ 0.16
Treasure Bay	1.51, 1.55	1.66	↑ 0.16
David's Bay north	1.61	1.76	↑ 0.15
Quarry Island	1.11, 1.51	1.45	↑ 0.14
Key River 3	1.74	1.70	↓ -0.04

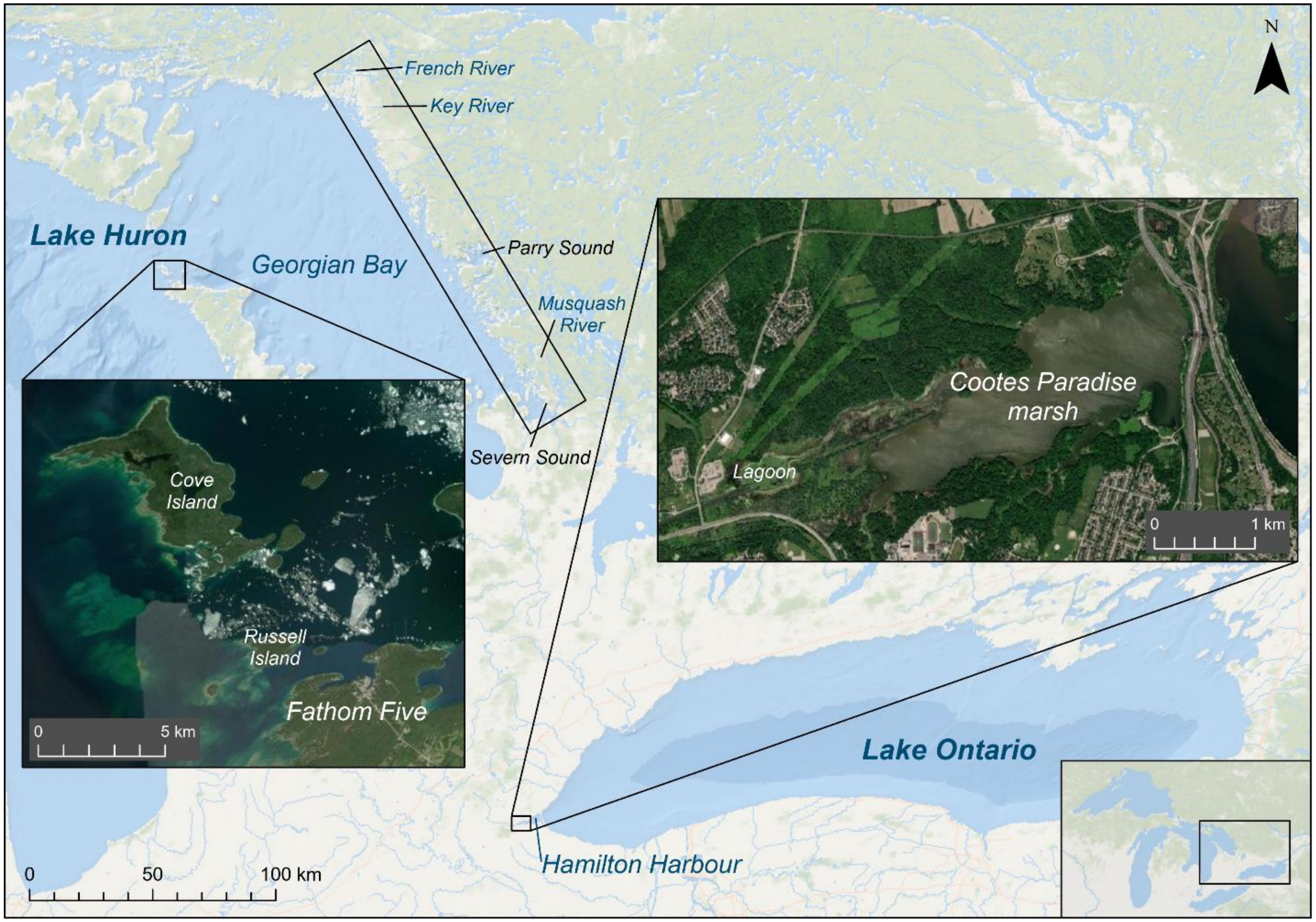


Figure 1.1: Location of this study: eastern Georgian Bay (most coastal marshes in the French River, Key River, Parry Sound, Musquash River, and Severn Sound areas), FFNMP (and the two islands), and CPM (and the lagoon) within the Great Lakes.

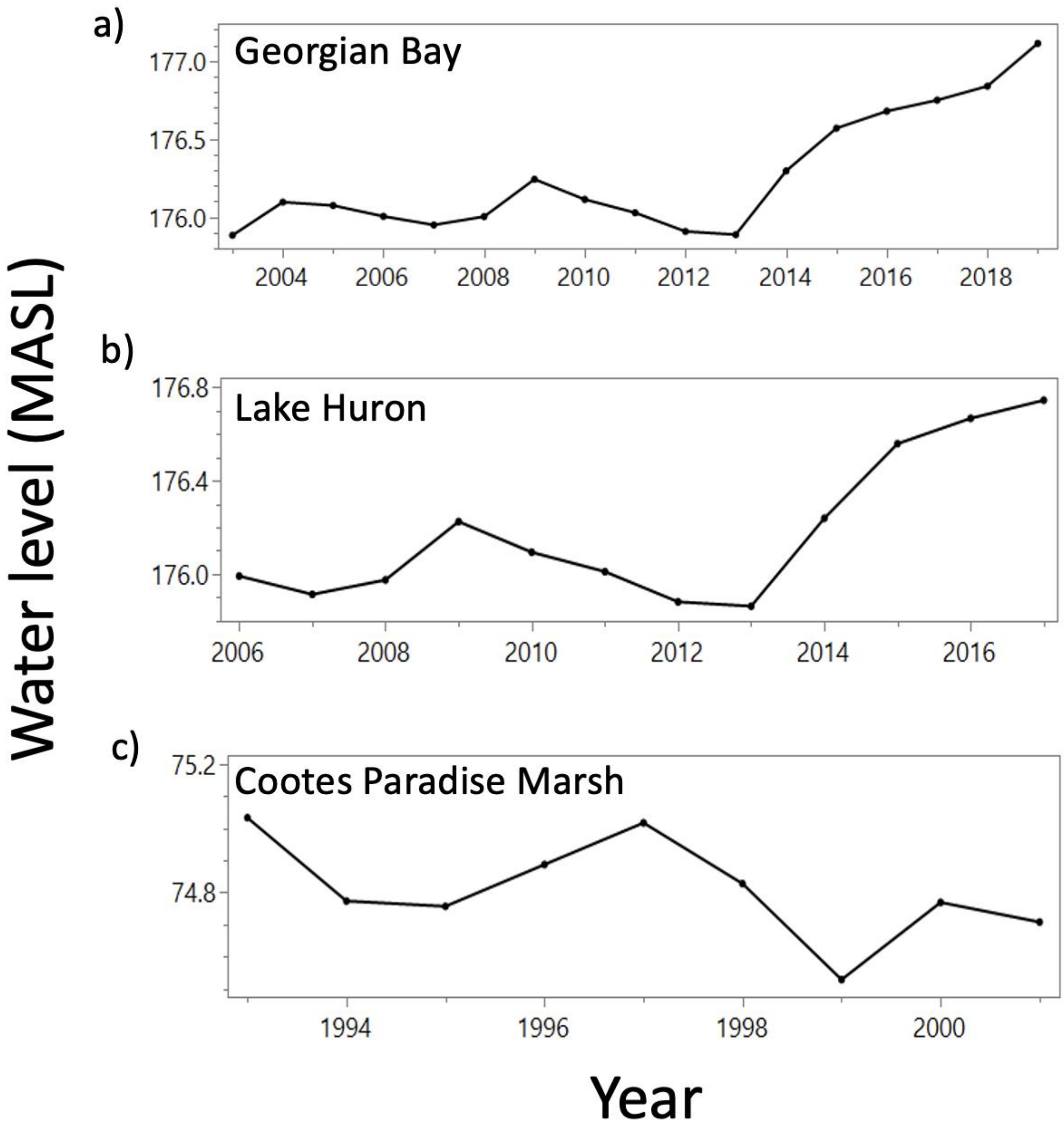


Figure 1.2: Mean annual water levels (MASL) during the study period for **a**) eastern GB using observations from the Collingwood Station #11500 **b**) FFNMP using observations from Tobermory, Lake Huron station #11690 and **c**) CPM using data from the Burlington Station #13150 and estimated with Eq.1 from Chow-Fraser et al. (1998).

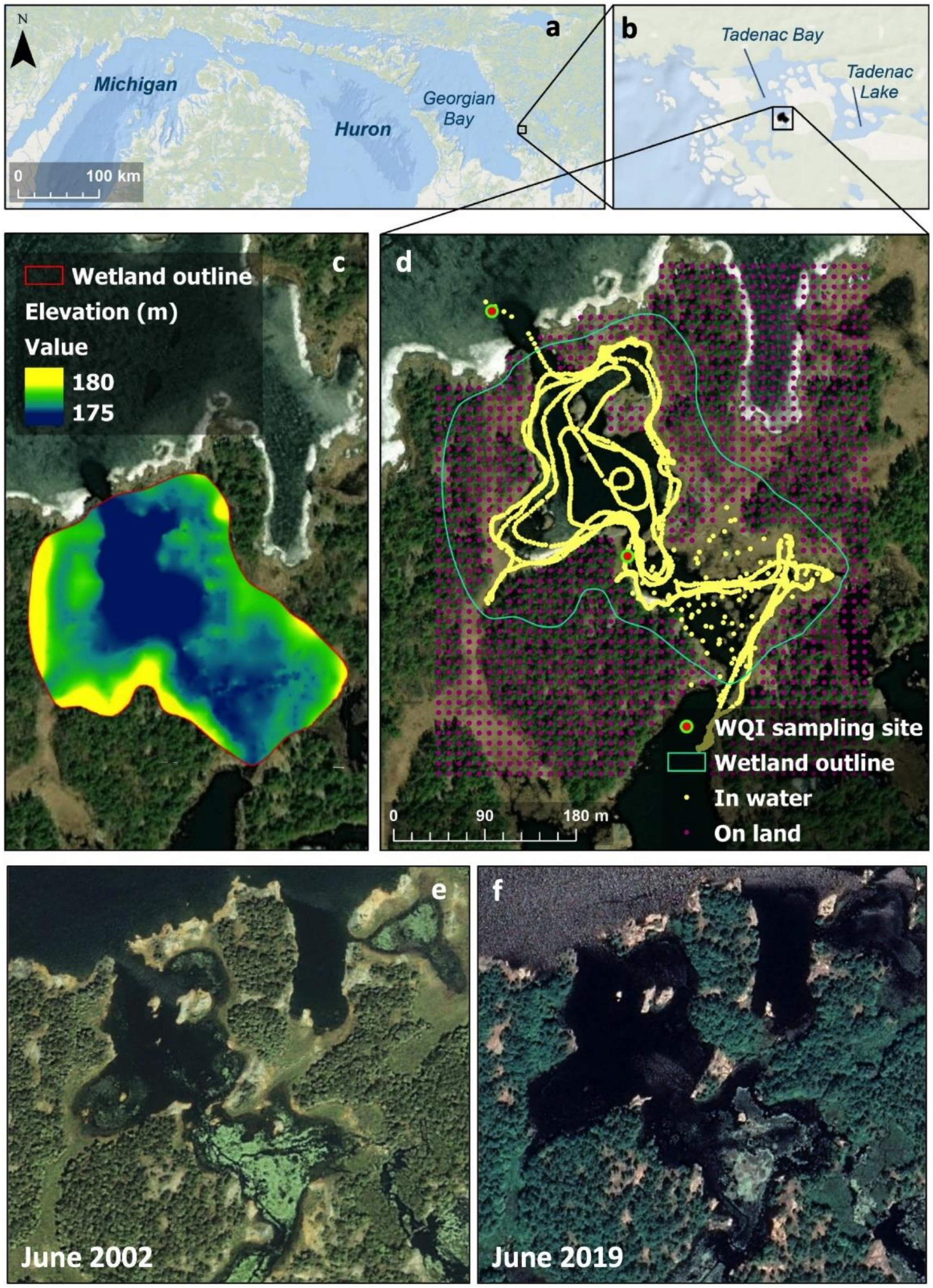


Figure 1.3: Black Rock wetland located in **a)** GB, **b)** Tadenac Bay. **c)** The DEM was interpolated with **d)** elevation data collected in water and estimated on land, and the water sampling location were within or near the wetland boundary. Google satellite imagery during **e)** year with low water level and **f)** a year with high water level.

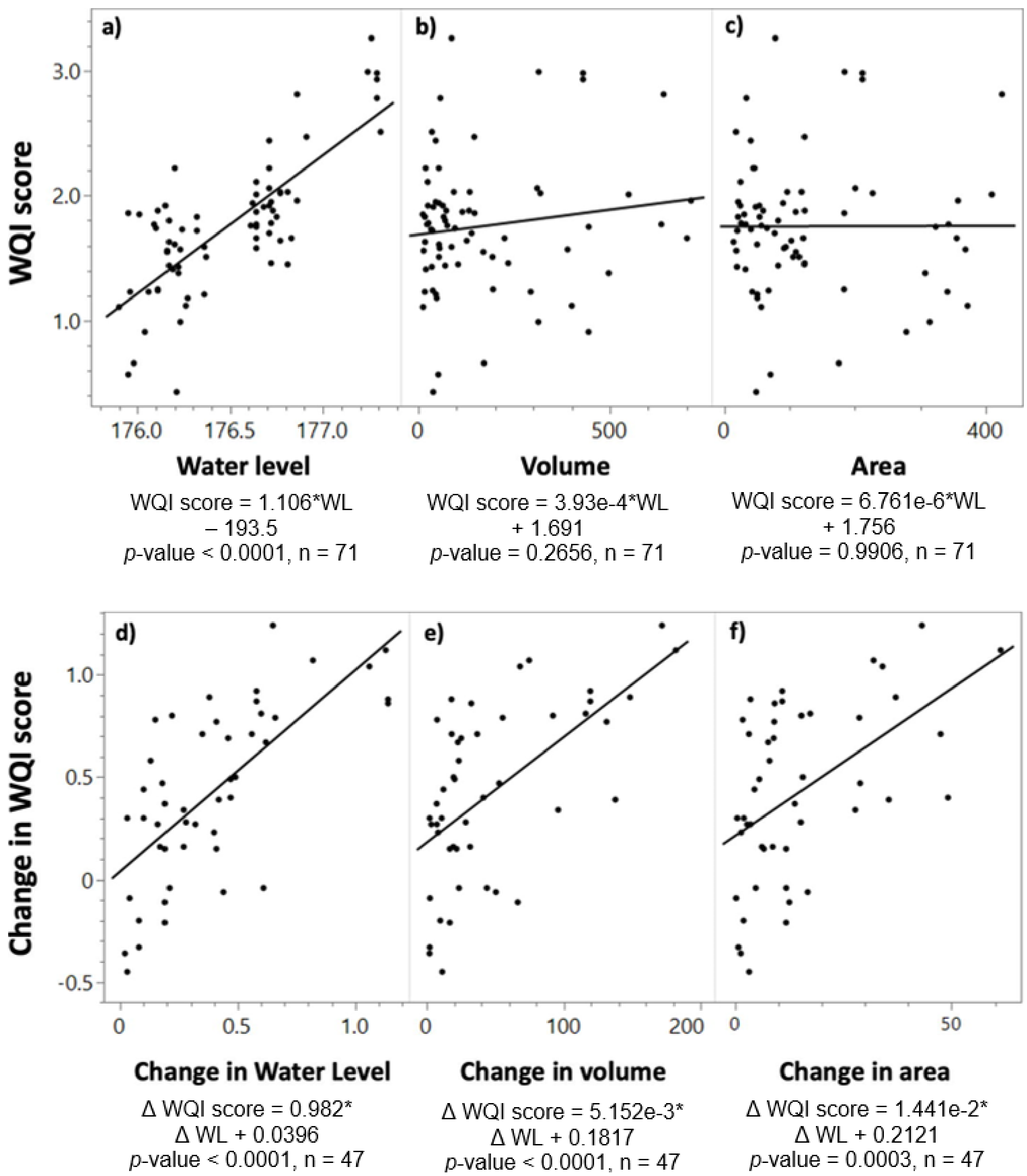
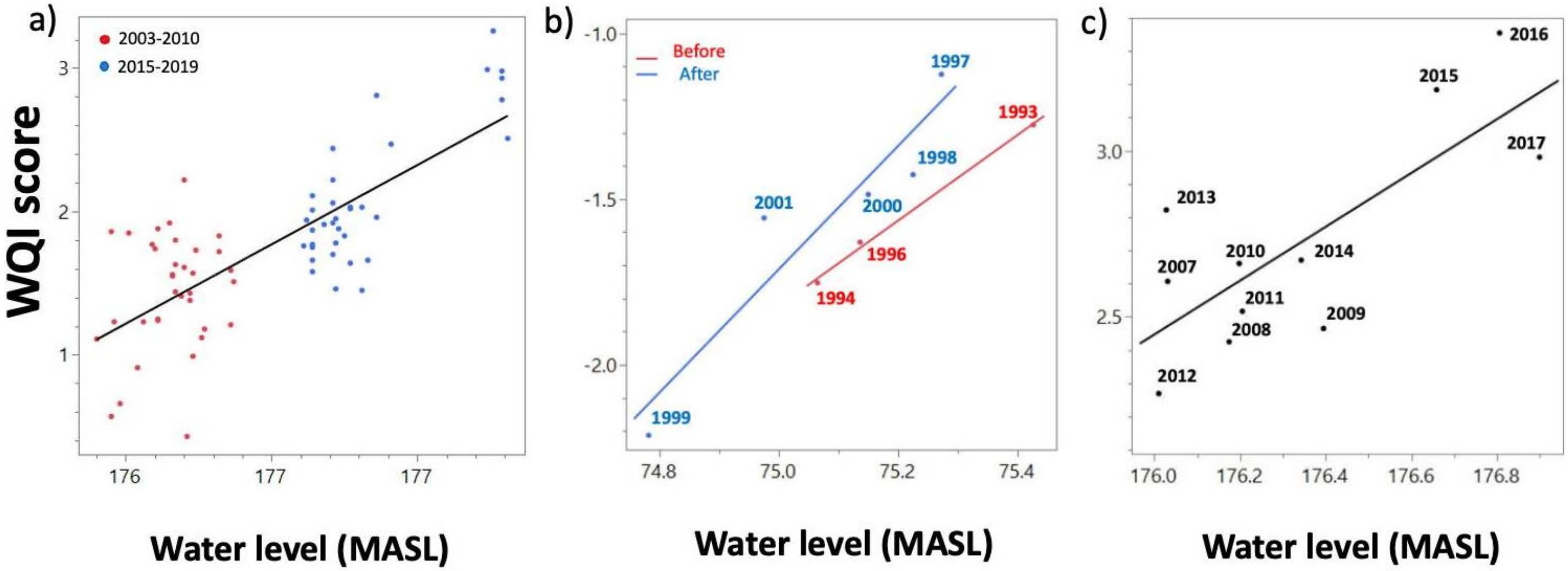


Figure 1.4: Relationship between WQI score and **a**) water level (WL; MASL) **b**) wetland volume (x10^3 m^3), and **c**) wetland area (x10^3 m^2). Relationship between change in WQI score between low and high water level periods and corresponding change in **d**) water level (MASL), **e**) wetland volume (x10^3 m^3), and **f**) wetland area (x10^3 m^2) for 24 GB wetlands.



WQI score = 1.106*WL – 193.5	WQI score (Before) = 1.29*WL – 98.59	WQI score = 0.8103*WL – 140.2
p-value < 0.0001, n = 71	*p*-value = 0.0401, n = 3	*p*-value = 0.0057, n = 11
	WQI score (After) = 1.861*WL – 141.3	
	p-value = 0.0182, n = 5	

Figure 1.5: Relationship between WQI score and water level (WL; MASL) for **a**) 24 marshes in eastern GB (sampling day WLs) **b)** 11 sites in CPM before and after carp exclusion (mean May through August WLs) and **c)** 9 marshes in FFNMP (mean June and July WLs)

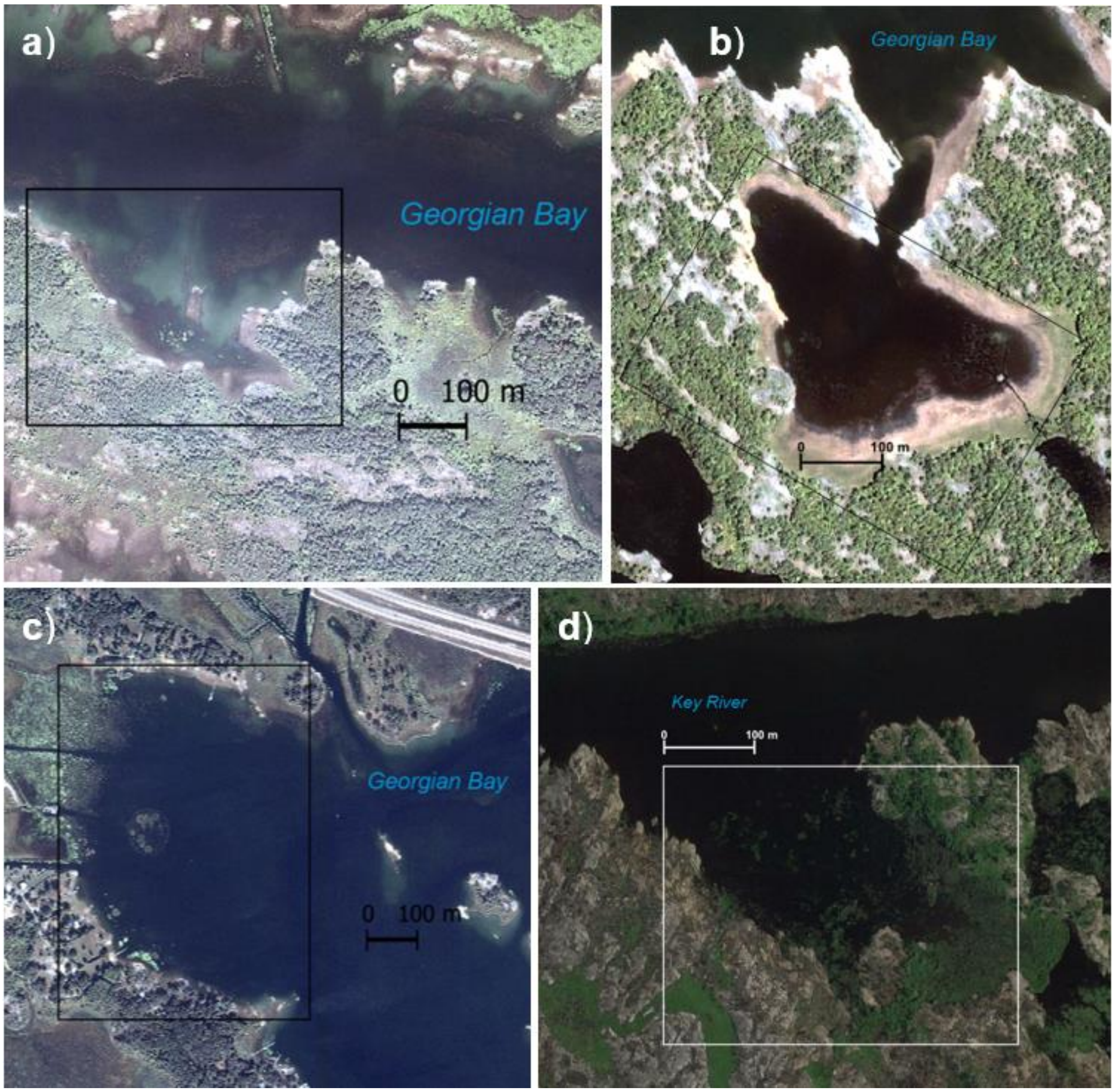


Figure 1.6: Three geomorphological types of coastal marshes (in the black or white frames) in eastern GB according to Albert et al. (2005). **a**) Open lacustrine open shoreline (LOS; Green Island, August 2008). **b**) Protected lacustrine protected embayment (LPP; Cormican Bay, March 2007). **c**) Open lacustrine open embayment (LOE; Oak Bay, May 2009). **d**) Open lacustrine open embayment (LOE; Key River 3, 2017). First 3 basemaps are IKONOS satellite and last one is Google Satellite.

Water quality indices of Lake Ontario coastal marshes reflect land use alterations, wetland geomorphology, and water levels

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ABSTRACT

Wetland Water Quality Index (WQI) scores are known to be negatively affected by the percentage (%) altered land uses (agricultural and urban land) in wetland watersheds. A recent study in an undisturbed region of eastern Georgian Bay found that WQI scores increased with Lake Huron water levels, even though there were no changes in human population size or % cover of land-use land-cover (LULC) categories in wetland subwatersheds. In this study, we investigated these apparently opposing influences on WQI scores used to assess wetland health of 29 marshes located in two developed regions of Lake Ontario (13 in Durham and 16 in Quinte). We used the Southern Ontario Land Resource Information System to estimate the % agricultural land (% AGR), % urban land (% URB), % forested land (% FOR) and % wetland (% WET) in wetland watersheds corresponding to three time periods (2002-2007, 2008-2011, and 2013-2017); during these years, water levels in Lake Ontario fluctuated by more than 1.5 m from 74.28 m to 75.81 m above sea level. Index scores were regressed against independent variables that included the four LULC categories for each time period, width of wetland opening (WOpening), water depth at sampling site (WDepth), subwatershed area (SUBarea), wetland area (WETarea) and wetland volume (Wetvol). Of the four significant multiple regression models, the best model explained 70 % of the variation in index scores, and included % URB and % AGR, WOpening, WDepth and SUBarea. WQI scores for these wetlands were more heavily influenced by % land-use alterations than by wetland hydrogeomorphology; nevertheless, managers must consider

changes in both LULC and the influence of water levels when interpreting long-term changes in WQI scores for coastal wetlands in human-disturbed regions of the Great Lakes.

INTRODUCTION

The Great Lakes basin is home to 40% of the population within Canada and the United States (EPA 2021), as well as a shoreline with freshwater coastal marshes. This intersection between humans and wetlands is known to negatively impact the quality of wetlands due to the input of nutrients. It is well-established that transformation of natural lands (e.g. wetlands and forest) in wetland watersheds to agricultural or urban lands negatively impacts the water quality of coastal wetlands (Field et al. 1996; Müller et al. 1998; Dodson and Lillie 2001; Wang et al. 2001). The water chemistry of coastal marshes is influenced by the alterations within the drainage area as well as by the geomorphology of the coastal marshes. The water chemistry of coastal marshes can be directly or indirectly influenced by the fluctuation of lake-levels, but it is also dependent on the geomorphology or shape of the wetlands. The physical characteristics are collectively known as the hydrogeomorphology (HGM) of coastal marshes which distinctly influences the sediment, vegetation, and fauna (Albert et al. 2005).

The three main hydrogeomorphic systems of Great Lakes coastal marshes are lacustrine, riverine, and barrier-protected marshes (Albert et al. 2005). Lacustrine systems occur in open or protected embayment that are directly controlled by lake water dynamics, such as the fluctuation of the water levels. Riverine systems are largely influenced by their drainage basins as it dictates the water quality and sediment input, but there is still lake influence on the marsh particularly at the mouth where lake water floods back. Barrier-protected systems have a barrier beach or alike feature that protects the marsh from wave action, where the water quality and levels are instead determined by surface drainage and groundwater which can also come from the lake.

Together, the hydrogeomorphic system type paired with the human activities occurring in the drainage basin can influence the concentration of nutrients and pollutants in the coastal marsh. Although this is generally acknowledged, few studies have been able to confirm the significant effects of both factors (Morrice et al. 2008). In an analysis of stressors and ecosystem services of the five Great Lakes, Allan et al. (2013) found the highest cumulative stress associated with Lake Ontario, especially wetlands and river mouths. The Town of Cobourg roughly splits the Canadian shoreline into a heavily urbanized western half that includes the cities of Toronto and Hamilton, and a less urbanized eastern half, with lots of wetlands and forests as well as agricultural land, and the City of Kingston (see **Figure 2.1**). Since European settlement, many of the coastal marshes along this shoreline are destroyed or degraded (Whillans 1982), but there are still dozens of coastal marshes, of different hydrogeomorphic site types, and that range in degree of degradation (Crosbie and Chow-Fraser 1999; Croft-White et al. 2017).

To assess the impacts of these human-induced disturbances on coastal wetlands, one of many ecological indicators created includes the wetland water quality index (WQI) by Chow-Fraser (2006). The WQI was created for Great Lakes coastal marshes using 12 water- and sediment-quality variables to indicate the overall wetland health conditions by considering land-use alterations. One hundred and ten coastal marshes in all five Great Lakes were sampled at least 10 m from the edge of aquatic vegetation primarily from 1998 through 2002, but the majority were during 2000 and 2001. Between these two years, the water levels varied within typical ranges. For example, Lake Ontario water levels differing by 24 cm during the regular sampling months (May through August), and 26 cm in Georgian Bay, Lake Huron. As a result, water levels within this range did not raise concerns, and so water levels were not considered as a variable in the development of the WQI.

However, a recent study found that water levels significantly affect the WQI scores of lacustrine coastal marshes in Georgian Bay which were mostly in reference conditions (Montocchio and Chow-Fraser 2021; Chen and Chow-Fraser 2022). The marshes included lacustrine open shoreline, protected embayments, and open embayments which are all directly controlled by lake water levels. The relationship between water quality index scores and water levels was highly significant ($p < 0.0001$). When compared with the wetland macrophyte index (WMI; Croft and Chow-Fraser 2007) and the wetland fish index (WFI; Seilheimer and Chow-Fraser 2007), the WQI scores illustrated a change in wetland conditions that the other 2 indices failed to match. They found a significant difference in the WQI scores during the high water level period (2003—2013) compared to the low water level period (2014—2019), but the WMI and WFI scores were not significantly different between the 2 periods, despite the biotic indices originally being highly significant with the

WQI in detecting watershed alterations. The main factor that was speculated to influence the WQI score, but not the biotic indices, was increased water levels through dilution of nutrients.

Lake level fluctuations can influence the water chemistry of coastal wetlands via the increase of volume of water as well as the export of nutrients from the marsh to the lake. The lacustrine marshes in Georgian Bay that were extensively sampled from 2003 though 2019 with a range in water levels of 1.5 m had significantly higher WQI scores during the high water level period (Montocchio and Chow-Fraser 2021). Since lacustrine marshes are directly controlled by lake water, especially open shore and embayment wetlands (Albert et al. 2005), it supports the hypothesis of the dilution effect. The export of nutrients in coastal marshes can come from the flushing of water and nutrients within the wetland into the Great Lakes. For example, Wells and Sealock (2009) observed Frenchman's Bay and found that the seiches of Lake Ontario caused hourly water level fluctuations, and in combination with temperature gradients, they contributed to the exchange of water between the bay and the lake. This allows for the mixing of the oligotrophic lake water with the more eutrophic bay water. The extent of the mixing is dependent on the width and depth of the opening such that seiche activity influence is dampened with smaller wetland mouth size and stronger tributary outflows (Trebitz et al. 2002). Similar lake and wetland water mixing via seiches were observed in a Lake Superior riverine type of coastal wetland (Morrice et al. 2004). It was also documented in a barrier beach coastal marsh on Lake Erie, where carbon from the marsh is exported to the lake (Bouchard 2007). Thus, the HGM of coastal marshes needs to be considered when studying wetland water quality and its response to watershed alterations and nutrient inputs.

One of the longest studies in the Great Lakes was done on 22 coastal marshes in the Durham and Quinte region which spanned 12 years (2003—2014) and analyzed the impacts of land-use/land-cover (LULC) on water quality (Croft-White et al. 2017). They found that WQI scores significantly increased over time in both regions, and that the relationship between WQI scores and % LULC was the most significant at the watershed-scale rather than with wetland buffers of 500, 1000, and 2000 m. At the watershed scale, the WQI scores increased significantly with an increase in the LULC classes of % natural, % forest, and % wetland, whereas it decreased with decreasing % urban. The only class where there was no significant relationship was for % agriculture. Interestingly, although wetlands in watersheds with higher % urban had lower WQI scores, 13 of the 15 wetlands that had a loss of natural land but increase in urban land unexpectedly had improved WQI scores. This could not be explained as whether it was attributed to restoration efforts or local conservation projects. However, the lake level range within this study period was 36 cm, notably larger than the range of 24 cm during the development of the WQI. Even a water level range of 50 cm in Lake Ontario water levels at Cootes Paradise was shown to have a significant effect on WQI scores (Chen and Chow-Fraser 2022).

To answer this question, our paper studies nearly the same set of marshes in the Durham region (13 marshes) and Quinte region (16 marshes) with a longer study period (2003—2017) including a high water level year by analyzing the relationship between WQI scores and % LULC combined with the effects of water levels on HGM variables such as wetland volume, wetland area, width of wetland opening, and water sampling depth. We expect WQI scores to be negatively related to % urban and % agricultural land, but positively related to % wetland and % forested land. In addition, there should also be positive effects of higher water levels especially for lacustrine marshes that are more open to mixing with lake water.

It is important to assess the influence of water levels and the HGM of marshes on WQI scores as it has confounding effects from watershed alterations which would give the wrong impression that wetland health conditions improved. Since the wetland WQI was developed, numerous conservation authorities (Central Lake Ontario Conservation Authority and the Quinte Conservation (Croft-White et al. 2017) and Credit Valley Conservation (CVC 2010)) used it to assess the health of its wetlands. Although the WQI is effective in detecting the effects of LULC on the subwatersheds, it is confounded by the interactive effects of water levels and the hydrogeomorphic site type of the marsh. In this study, we test the confounding effects of

METHODS

Wetland Monitoring Programs

Data in this study were collected as part of a regional water quality monitoring partnership among Environment and Climate Change Canada (ECCC), Central Lake Ontario Conservation Authority (CLOCA), Quinte Conservation (QC), and the Ganaraska Region Conservation Authority (GRCA). These agencies have monitored the coastal marshes located on the north shore of Lake Ontario as part of the *Durham Region Coastal Wetland Monitoring Project* (Environment Canada and Central Lake Ontario Conservation Authority (CLOCA) 2007; Grabas et al., 2012) and the *Bay of Quinte Wetland Monitoring Program* (Environment Canada, Canadian Wildlife Service, 2006; Macecek and Grabas, 2011) since the early 2000s (**Table 2.1**). These wetlands have a diverse set of LULC classes within their subwatersheds (**Figure 2.1**); those in the Durham Region to the west are heavily urbanized (**Figure 2.2**) whereas those in the Quinte Region to the east have higher proportion of natural land cover classes (forests and wetlands) (**Figure 2.3**).

Wetland and Site Selection

To simultaneously test the effects of land-use alterations and water-level fluctuations, we eliminated any wetlands that were not hydrologically connected with Lake Ontario at some point during the ice-free season. We only included barrier-beach or barred wetlands if water-level data were available from water-level dataloggers installed within the marsh and these included Cranberry, McLaughlin, Pumphouse, and Westside. Blessington Creek Marsh in the Quinte Region was split into 2 marshes, one in the west and the other in the east because the digital elevation models (DEMs) used to create drainage basins showed that these adjacent marshes had separate subwatersheds. The sampling points were also at two different locations, separated by a large stand of vegetation. We excluded any wetland that had not been sampled more than 1 or 2 years and those that had not been sampled since 2003/4. The only exception was Duffins Marsh, which had been sampled beginning in 2007; we included it because this was a relatively large watershed that was dominated by agricultural land and was therefore rare in the Durham Region. Oshawa Second Marsh and Port Newcastle were also excluded because sampling sites in these marshes were located 1000 m and 550 m, respectively, from the mouth of the wetland into Lake Ontario. We considered that too far to be influenced by mixing with water from Lake Ontario water. In total, we included 13 marshes in the Durham Region and 16 in the Quinte Region (see **Figures 2.1 to 2.3**).

Field measurements for WQI

The standardized sampling protocol used by the agencies have been described by Croft-White et al. (2017). Agencies used various multi-sondes at mid-depth to measure temperature, pH, and specific conductivity. As well, they collected water samples to measure turbidity with a turbidimeter. Water depth corresponding to the site where water had been collected was measured with a meter stick to the nearest centimeter or a weighted string if the water was greater than 1 m (Durham Regional Coastal Wetland Methodology).

They used the following equation to calculate a version of the WQI, according to Equation 7 in Table 6 of Chow-Fraser (2006):

$$\begin{aligned} WQI = +9.2663224 &- 1.367148 * \log TURB \\ &- 1.577380 * \log COND \\ &- 1.628048 * \log TEMP \\ &- 2.371337 * \log pH \end{aligned}$$

This equation explained close to 90% of the variation in WQI scores that had been calculated with Eq. 1.1 (from Table 6; Chow-Fraser 2006) that used 12 variables collected from 146 wetland-years.

Wetland Hydrogeomorphology

Albert et al (2005) classified coastal marshes in the Great Lakes according to hydrogeomorphic (HGM) characteristics that better reflect wetland processes and to improve management of these ecosystems. HGM types are classified based on geomorphic position, dominant hydrologic source and hydrologic connectivity to the Great Lake. There are three hydrologic systems (lacustrine, riverine and barrier-protected) that are further separated into geomorphic types based on degree of connectivity with the Great Lake due to wetland shape and width of wetland opening. The 29 marshes used in this study were assigned one of Albert et al.'s (2005) HGM classifications in the Great Lakes Coastal Wetland Inventory (Great Lakes Coastal Wetland Consortium 2004). We agreed with all given classifications except for one; instead of using the open lacustrine type assigned to Blessington Marsh, we assigned Blessington Marsh West as lacustrine protected embayment and Blessington March East as barrier-enclosed (see all assigned types in **Table 2.2**). The HGM classes were subsequently used to group wetlands to examine the relationship between WQI scores and water levels.

The environmental agencies sampled all wetlands at 4 sites each year, although the exact location of each site varied from year to year. Since Chow-Fraser (2006) stipulated that open water should be sampled for WQI scores, we first imported all data into a GIS and visually inspected the location of all sites within each wetland. In some years, the sites were located very close to vegetation stands, such as being around 1 m distance away according to the Google satellite imagery, so they were excluded. We chose a site in each wetland that was located furthest away from the aquatic vegetation in open water; for sites with any kind of barrier, we only included the site if it were located in the main portion of the wetland within 550 m of the opening to Lake Ontario. Two exceptions were Frenchman's Bay where the closest suitable site chosen was 850 m away, and barrier-enclosed marshes were not chosen based on the distance from the opening. Since all sites for Cranberry Marsh were in open water, and the marsh is relatively small, we could not select only one site; instead, we included all data and calculated a mean for each year.

We used the 2019 Google Earth satellite image in ArcGIS to measure the width of the wetland opening (WOpening) to Lake Ontario. The difference in the width of the wetland opening between years were generally low, roughly around 1 to 15 meters difference in the riverine marshes, but the difference for lacustrine marshes were relatively low in comparison with the opening width. To keep the measurement errors consistent, we performed all measurements at a scale of 1:150 for smaller marshes to 1:4500 for larger marshes. Openings of lacustrine wetlands or open shorelines such as Robinson Cove were measured parallel to the coastline of the wetland, and approximately perpendicular to the flow of water. For wetlands that were not open embayments, we looked for signs of water exchange between the lake and the wetland in the satellite image and measured the break in the shoreline/perimeter; if there were breaches at several locations, we summed these. For barrier-enclosed wetlands such as McLaughlin Marsh, there was no obvious opening and was therefore given a value of zero. Since we only used a single Google Earth basemap image acquired in 2019, we could not account for any temporal variation in the width of the opening, presumably due to different water levels. Nevertheless, since the extent of the difference in water level was <1.5m, we have assumed the variation to be negligible. Wetland opening width measurements are presented in **Table 2.3**.

Wetland DEM, Volume, Area and Depth

We created digital elevation models (DEMs) for wetlands in the Durham Region using LiDAR point cloud data from the Greater Toronto Area (GTA; acquired in 2015) that covered wetlands from Rouge River Marsh east to Caruthers Marsh; data from CLOCA (acquired in 2018) covered wetlands from Cranberry to Pumphouse Marsh; data from OMNRF in Peterborough (acquired from 2016 to 2017) and from CLOCA were needed to complete the DEM for McLaughlin Marsh. DEMs for all wetlands in the Quinte Region were created with data from the LiDAR Eastern Acquisition Project (LEAP; acquired in 2009; **Figure 2.2**).

The LiDAR point cloud data were downloaded as LAZ (a compressed data format often used to transfer large amounts of LiDAR data). We then converted LAZ to LAS (a vector format) using the tool "Convert LAS" in ArcGIS Pro 2.8 (ESRI 2021). After the dataset was created, we used LAS Filters to select only Ground points. Without this filter, the outputted DEMs would have accounted for the height of trees and bushes within and along the marshes. Finally, we applied the tool "LAS Dataset to Raster" to the filtered dataset, with settings of "Elevation" for the field, "Binning" as the interpolation type, "Minimum" for the cell assignment and "Void Fill Method" as Natural Neighbor. To create the DEMs of 1x1 resolution, the Output Data Type was "Floating", and the Sampling Type was "Cell Size", with a Sampling Value of 1.

We used the wetland shapefile obtained from the Great Lakes Coastal Wetland Inventory, which was modified along the coast and the extent upstream to clip the DEMs of each wetland. Since water levels of Lake Ontario have not exceeded

76.0 masl in the historic data recordings, we only included areas of the DEM below this extent for further analyses. We applied the water level corresponding to the timing of sampling for each wetland to calculate the volume and area using the tool Surface Volume, in which the daily water levels of the sampling date were used as the plane height and the measurements below the plane were taken to calculate the surface area and wetland volume (**Table 2.3**). Mean depth of wetland was calculated by dividing the volume by wetland area corresponding to the same date that water samples had been collected (**Table 2.3**).

Water Levels

All wetlands are hydrologically connected with Lake Ontario for at least part of the season each year, although the degree of water exchange is expected to vary depending on the width of the wetland opening connecting to Lake Ontario and the hydrogeomorphic site type (see Albert et al. 2005). The degree of mixing with Lake Ontario (and therefore extent of dilution from Lake Ontario) would depend on Lake Ontario's water levels, which have fluctuated from a low monthly mean during the growing season (May 1 to Aug 30) of 74.65 masl occurring in August 2012 to the highest monthly mean of 75.91 masl occurring in June 2019, resulting in an increase of 1.26 m over the 7 years (see **Figure 2.4a**). The large spatial distribution and separation of the study sites prompted us to use the mean daily water level measurements from the stations collected by the Canadian Hydrographic Service at the Cobourg Station (#13590) for 6 marshes in the Durham Region and those from the Kingston Station (#13988) for all marshes in the Quinte Region (see **Figure 2.2b; Table 2.1**). For the other 7 marshes in the Durham region with water level dataloggers, we used the daily water levels corresponding to the sampling dates (**Table 2.1**).

Time Periods and SOLRIS

To examine the effects of LULC classes on WQI scores, we had to match LULC data with the appropriate timing when water-quality data had been collected. There are currently three versions of the Southern Ontario Land Resource Information System (SOLRIS). Version 1.2 was published in 2008 and used satellite images from 2000 to 2002. Since the earliest year we had data began in 2003, we had to pair information from SOLRIS 1 with WQI scores obtained between 2003 to 2007. Version 2.0 was published in 2015, and covered years 2009 through 2011, and were therefore paired with data collected during those years. Similarly, version 3.0 was published in 2019 and covered years from 2014 through 2017, and therefore we paired WQI scores obtained in 2014 to 2017 with these data. Next, we calculated mean values for each time period (i.e. 2003-2007, 2009-2011 and 2014-2017) so as not to bias the analysis in favour of time periods with a large number of data points. For each time period, we also pooled the mean daily water levels and calculated a mean value.

Creating the Subwatersheds

We created subwatersheds in ArcGIS Pro using the Provincial Digital Elevation Model of Southern Ontario (PDEM 2013), which had a pixel resolution of 30x30 m. We used tools in the following order: fill, flow direction, flow accumulation, con, stream link, and then watershed. The Ontario Hydro Network Watercourse was used to identify main tributaries so that edges of some subwatersheds that crossed streams could be modified. Small areas within subwatersheds that were delineated as not being within the subwatershed were also manually removed so that the subwatershed was dissolved as one contiguous unit. Since the elevation data within Lake Ontario were inaccurate, they caused the subwatershed outlines along the shore to be warped, and these had to be manually modified to match the shoreline from satellite imagery and perimeter of wetlands. In total, 29 subwatershed were created that were used to clip the LULC data from each version of SOLRIS. Since the DEM was published in 2013, major land-use changes that affected the elevation from 2014 onwards would not be reflected. The only wetland that had this problem was Westside Marsh which had approximately 25 ha of its wetland removed and used for extraction/mining between September 2013 and August 2014. The subwatershed created for Westside Marsh was modified to exclude the extraction area as it was a depression that did not flow into the Marsh.

Percent Land-Use/Land-Cover (LULC)

For each of the three time periods, we calculated the % cover of four major LULC classes in each wetland subwatershed to match those in Croft-White et al. (2017). These were % wetland (WET), % forested land (FOR), % urban land (URB), and % agricultural land (AGR); these were calculated by taking the sum of the LULC class and dividing it by the subwatershed area and multiplying by 100. Although Westside Marsh had a slightly decreased size in the subwatershed between periods 2 and 3 due to mining, we decided to use the same subwatershed area for all 3 periods, given the loss was only 4.4% of the subwatershed. All four of the % LULC for each marsh were calculated for the minimum and maximum amounts within the 3 time periods (**Table 2.5**).

Regional Differences in WQI scores, Subwatershed and Wetland Size, and LULC

We compared the regional differences in WQI scores, subwatershed area, wetland area, wetland volume, width of wetland opening, mean wetland depth, water depth at sampling site, % agriculture, % forest, % urban, and % wetland (**Table 2.4**). The mean of each variable by region was calculated, and the medians were put in brackets, with the parameters significantly higher in bold.

Statistical Analyses

We compared data between the two regions because Croft-White et al. (2017) had established clear differences in percentage land uses in wetland subwatersheds between Durham and Quinte Regions. To assess the effects of LULC and water levels on the WQI scores, we first looked at the data separately by region. The WQI was regressed against % LULC for each class as linear models. WQI scores used corresponded to each sampling date within the three time periods. To compare the effects of the LULC and water levels, WQI scores were regressed against water depth of the sampling site because it had the strongest positive significant relationship with WQI scores among the HGM metrics (discussed in the results). This is likely because water depth at the same sampling point accounted for the water levels at the exact sampling site over the years.

We regressed WQI scores against all dependent variables in a stepwise multivariate regression model to assess the influence of LULC and HGM metrics on WQI scores. The variables used included water depth at the sampling site (WDepth), width of the wetland opening (WOpening), wetland volume (WETvol), wetland area (WETarea), subwatershed area (SUBarea), % AGR, % FOR, % URB, and % WET. We tested variables in specific orders until the *p*-values exceeded 0.05. Although the order of some variables did not matter for the same model, sometimes the order influenced the *p*-value, so we had to redo the model using different orders. We obtained 4 significant models with comparative AICc values that contained at least 3 variables. The AICc values were used because they needed to be corrected for small sample size.

RESULTS

Mean and median WQI scores for wetlands in the two Regions were significantly different (**Table 2.4**); scores for the Quinte Region were significantly higher than those for the Durham Region. Positive WQI scores indicate wetlands are in good condition, whereas scores between -1 and -2 indicate wetlands are very degraded (Chow-Fraser 2006). We did not find any significant differences in size of subwatersheds, when either means or medians were compared. The median surface area of coastal marshes in the Durham Region was significantly greater than that in the Quinte Region, even though there were no significant differences between means. Wetlands in the Durham Region were also deeper, with a significantly larger volume (**Table 2.4**). Another important difference is that the water depths where measurements were taken in the Durham Region were shallower than those in Quinte.

We classified all wetlands according to their HGM site type (Albert et al. 2005). Those in Durham Region were either barred drowned rivermouth (RRB) or barrier-beach lagoons (BL), and this explains why they had a narrower wetland opening compared with those in the Quinte Region, which in addition to RRB and BL, also included open drowned rivermouth (RRO), protected embayments (LPP) and open embayments (LOE) (**Table 2.2**).

Next, we compared composition of LULC classes in wetland subwatersheds between the two regions (**Table 2.4**). Mean % AGR was significantly higher in

Quinte (52%) than in Durham (41%), although there were no significant differences between median values. Differences with respect to the other three LULC classes were highly significant for both mean and median values; generally, Durham subwatersheds had a mean of 40% URB vs only 7% URB for Quinte subwatersheds. By contrast, in Quinte subwatersheds, % FOR was almost double and % WET was triple that in Durham subwatersheds during the study period (**Table 2.4 and 2.5; Figures 2.2 and 2.3**).

The Lake Ontario water levels for the two regions varied annually throughout the sampling period (from May through August) with a range of 0.9 m, a minimum of 74.8 m, and a maximum of 75.7 m (**Figure 2.4b**). The water-level stations at Cobourg and Kingston recorded water levels that varied only slightly, with water levels recorded at Cobourg being marginally (< 5 cm) higher than at Kingston. Water levels varied by 50 cm from 2002 through 2016, but greatly increased by 80 cm in 2017, lowered by 60 cm in 2018, and increased by just over 60 cm again in 2019 to a record high. It is important to note that only the high water levels occurring in 2017 have been included in Period 3 in this study.

We regressed the mean WQI scores calculated for each SOLRIS time period against the four different LULC categories by region. We found that % AGR and % FOR were significant predictors of WQI scores only for wetlands in the Quinte Region (P=0.0184 and P-0.0005, respectively; **Figure 2.5 a and c**). As expected, WQI scores were negatively related to % AGR (r^2=0.12) and positively related to %

FOR (r^2=0.24). WQI scores were not significantly related to any of the LULC classes for data in the Durham Region. This figure also illustrates the generally better condition of wetlands in the Quinte Region compared with those in the Durham Region. Durham Region had a maximum of 13% WET in subwatersheds whereas the Quinte Region had a maximum of 40%. By contrast, the maximum % URB was 3 times higher in the Durham Region than in the Quinte Region.

When we combined the data for both regions to conduct the regressions, WQI scores were significantly related to all LULC categories (**Figure 2.6**) as well as to depth of wetland at time of sampling, and the width of the wetland opening. The combined class of altered land uses (% AGR and % URB) and natural land cover (% FOR and % WET) were also significant predictors of WQI scores. All these relationships are consistent with expectations except for the positive relationship between WQI and % AGR.

Of all variables related to wetland HGM (Wdepth, WOpening, WetVOL and WetAREA), bivariate regressions showed that WQI scores was most strongly related to sampling depth. We sorted the dataset according to three levels of increasing % AGR, and regressed WQI against Wdepth. Accounting for the effects of increasing agricultural land in subwatersheds resulted in significant regressions for wetlands with <38.2% AGR and between 38.2 to 56.7% AGR (**Figure 2.7a**). We used the same approach to examine the effects of % URB (**Figure 2.7b**), % FOR (**Figure 2.7c**) and % WET (**Figure 2.7d**). Significant relationships of WQI vs water depth were obtained for watersheds that had greater than 10% forested land. Data for all levels of % URB were significantly related to water depth, indicating that when the effects of urbanization are held constant, WQI scores would increase as a function of water depth in wetlands (**Figure 2.7b**)

The large number of significant bivariate regressions indicated that it would be appropriate to conduct a multiple linear regression analysis. We found four significant models with at least 3 independent variables (**Table 2.6**). Model 1 used 5 variables including % URB, % AGR, WOpening, Wdepth and SUBarea. This model accounted for 70% of the overall variation ($p < 0.0001$) with each variable being significant as a model parameter (see **Table 2.7**). The AICc was 2 units lower than that of Model 2, which included 4 variables (% URB, % AGR, WDepth and WOpening) and accounted for 68% of the overall variation. Model 3 included % WET, Wdepth and % URB and accounted for 66% of the overall variation. Model 4 included % AGR, % WET, % FOR, Wdepth and WOpening. This model had an r^2-value of 0.6672 and the highest AICc value of 176.

We chose Model 1 as the best model because it had an AICc value that was at least 2 units lower than that of the next model. The negative coefficients of % AGR, % URB and Subwatershed size reflect the negative effects of the known watershed stressors on WQI scores, while the positive coefficients of sampling depth and width of wetland opening reflect the ameliorating effects of Lake Ontario water which is dependent on water levels and HGM characteristics of the wetland (see **Eq.**

2.1). All regression coefficients and the intercept statistically significant (**Table 2.7**).

$$\begin{aligned} \text{WQI score} = 0.698493 &- 0.034591\ \%\ \text{URB.} \\ &- 0.024339\ \%\ \text{AGR} \\ &+ 0.0126921\ \text{WDepth} \\ &+ 0.0014765\ \text{WOpening} \\ &- 1.99 \cdot 10^{-9}\ \text{SUBarea} \end{aligned} \qquad \textbf{(Eq. 2.1)}$$

($F(5, 80) = 37.3936$; r^2-value of 0.70; $P < 0.0001$, RMSE of 0.6086)

DISCUSSION

Croft-White (2017) conducted a trend analysis of water-quality conditions and LULC classes over a 12-year period from 2003 through 2014 for 22 marshes that were all included in this study except Oshawa Second Marsh and Port Newcastle for the reasons explained earlier (Wetland Selection in Methods). We included additional data for 9 other wetlands. They only analyzed their data by region because of the differences in the proportion of LULC classes between regions. Another difference between their study and ours is that they related change in WQI scores against the difference in LULC categories measured in the early 2000s (SOLRIS v.1) and that measured a decade later (SOLRIS 2), whereas we regressed WQI scores against the actual LULC proportions during each of the three time periods (SOLRIS v.1, SOLRIS v. 2 and SOLRIS v.3). Although we agree that there are significant regional differences between Durham and Quinte wetlands (**Table 2.4**), we do not think that this difference should prevent us from combining the data to examine the effects of LULC classes since it was only in combining the two regions that we were able to evaluate the full range of values for all four LULC classes.

For the pooled data, all LULC variables considered had significant effects on WQI scores. The significant negative effect of increasing % URB land on WQI scores is consistent with the fact that the WQI was created using the alteration of natural lands to be developed land as an indicator of poorer wetland health. As well, many published studies that show how increased impervious land in cities lead to higher input of nutrients and pollutants (Robertson and Saad 2011), while the positive effect of increasing % FOR and % WET on water quality is also consistent with studies that show these land-cover classes can reduce surface runoff and perform ecosystem services that result in lower export of nutrients and sediments to downstream water bodies (Field et al. 1996). We do, however, believe that the positive relationship between WQI scores and % AGR is an artifact of Quinte wetlands being in far better condition than Durham wetlands but have higher proportion of agricultural land in their subwatersheds. The significant negative relationship between WQI scores and % AGR exhibited by Quinte data alone is a true reflection of the well-established damaging effects of agricultural runoff on wetland quality (Field et al. 1996).

Some problems with spurious relationships may be associated with differences in how LULC classes had been classified in all versions of SOLRIS, as well as poor classification accuracy (reportedly between 70.2% to 76.4%). As an example, the Pumphouse Marsh subwatershed was classified as having a large proportion of agricultural land. After we inspected Google Earth satellite images, it was clear that these agricultural lands were actually green space (parks and fields). In another example, there was a fairly large portion of the subwatershed of McLaughlin Marsh that had been classified as agricultural, which should have been classified as a trailer park. Over time, as trees in the trailer park matured, the land could have been classified as forested. Such inaccuracies in LULC classification may have contributed to the spurious relationship between WQI and % AGR (i.e. **Figure 2.6a**). In some cases, these problems were corrected in version 3 of SOLRIS (2014-

2017). Improved accuracy of the classification must be done manually and should be attempted in the future to see if the relationships improve. We recommend that SOLRIS-derived LULC data be verified with an independent dataset, particularly for areas that had been classified as "undifferentiated" in SOLRIS 1.

To improve predictive power of LULC classes, green spaces and trailer parks should be re-classified as a unique class since ecosystem services performed by the permeable surfaces in parks should result in better water quality than impervious surfaces that typically characterize urban land. Such misclassification is especially problematic for wetlands with small drainage basins (e.g. McLaughlin Marsh, Pumphouse Marsh and Cranberry Marsh), where a large park could account for a large proportion of the subwatershed. Another way to deal with these discrepancies may be to combine both AGR and URB as "Altered land", and the FOR and WET as "Natural land". Combining these classes increased the r^2-value from 0.10 and 0.42 for % AGR and % URB, respectively to slightly higher value of 0.44%.

In addition to the expected effect of %LULC in wetland subwatersheds, we also documented the significant effect of the dilution effect of fluctuating water levels of Lake Ontario on WQI scores. The significant positive effect of water level on WQI scores was first documented by Montocchio and Chow-Fraser (2021) for undisturbed coastal marshes of Georgian Bay. Chen and Chow-Fraser (2022) showed that WQI scores increased with water levels because of the increased volume and area of wetlands as water levels increased. We did not see any significant effect of morphometric parameters on WQI scores in this study, and we attribute this to the difference in hydrogeomorphic site types between studies as well as the differences in % LULC. The Lake Ontario wetlands included 9 barred riverine and 7 opened riverine, 6 barrier-enclosed, 3 protected embayments, and only 4 lacustrine open shorelines, whereas those in Georgian Bay were primarily lacustrine that were either fringing shoreline or embayments.

Previous studies have emphasized the importance of geomorphic site type on marsh processes (Albert et al. 2005; Morrice et al. 2008), theorizing that water levels of the Great Lake would influence lacustrine wetlands more so than barrier beach, barred drowned river-mouth marshes or enclosed riverine site types. All the lacustrine open shoreline wetlands were in the Quinte region, which may explain in part why there were higher WQI scores in the Durham wetlands. Though this has been hypothesized (Albert et al. 2005), our study is the first to show that width of wetland opening, which reflects hydrogeomorphic site types, has a significant effect on water quality of coastal marshes. The wetlands that form barriers such as McLaughlin and Cranberry Marsh tend to have noticeably lower WQI scores compared to others with the same % LULC classes (see **Figure 2.9**), whereas the lacustrine marshes of the Quinte Region tend to have higher WQI scores than can be predicted by %LULC alone.

Though we did not see a significant relationship between WQI score and water levels, depth of the sampling station emerged as a significant predictor. Water depth is the interaction between lake water level and wetland bathymetry. Over the study period, water levels of Lake Ontario fluctuated over 1.5 m, and over that time, a large range in water depths were also recorded, depending on wetland bathymetry. Therefore, even though there was no significant effect of water level per se, the indirect effect of water level and its interaction with wetland bathymetry was evident. The strength of the regressions of WQI scores with water depth increased when data were sorted according to increasing levels of %LULC classes. When we accounted for the effects of increasing % LULC, the regressions of WQI against depth became stronger. For instance, the regression for wetlands with the two lower levels of % AGR were highly significant, showing that effects of water depth were strong when the stress of agricultural development was low (<56%; **Figure 2.7a**). Similarly, the regressions involving % URB were highly significant when data were broken into different levels of % URB (**Figure 2.7b**), with weaker relationships in the subset with highest % URB (>22%). When we accounted for natural land cover types such as forested land and wetlands, we saw highly significant regressions for wetlands with highest levels of % FOR and % WET, but not necessarily for data in the lower levels (**Figure 2.7c and 2.7d,** respectively).

The best model that emerged from the multiple regression analysis included 5 independent variables (2 LULC classes, watershed size, and 2 variables reflecting wetland hydrogeomorphology), which explained 70% of the variation in WQI scores. The Equation (Eq. 2.1) reflected the negative effects of % AGR, % URB and size of subwatershed, as well as the positive effects of % FOR and % WET, and the dilution effects of increasing water level from sampling depth and width of wetland opening. This model suggests that WQI scores are largely influenced by % URB, % AGR, more so than by effects of water level of Lake Ontario, as reflected by hydrogeomorphic variables.

Although we had water-quality data collected between 2003 and 2019, we were limited by the available LULC data that were only available for 3 time periods. It was not always possible to have an exact temporal match between WQI scores and LULC data. This was particularly problematic for Period 1, when SOLRIS data covered years 1999 to 2002 (also reported as 2000-2002 in some publications), but WQI data were not available until 2003 and in some cases 2004. We decided to combine data between 2003 to 2007 for Period 1, recognizing that the LULC data may be outdated, and data collected in the first two years were not always complete. We have not examined the extent to which this source of error affected the significance of our statistical analyses, but this is something that may be pursued to reduce residual variation in our regression models.

As a first step in our evaluation of the dataset, we examined all available data collected in each wetland. We observed unacceptably large differences among sites for the same wetland-year. Since the WQI was developed with data corresponding to sampling locations in open water, at least 10 m away from vegetated areas, we could not use most of the data made available to us, because most of them had been collected in or near vegetation stands (as seen in the ESRI basemap satellite imagery). Unfortunately, we were forced to exclude almost a dozen sites from consideration because of this problem. As mentioned in the Methods, some exclusions were necessitated because the sampling site had been located too far away from the opening of the wetland to Lake Ontario. In the future, we recommend that agencies ensure that at least one open-water (10 m away from aquatic vegetation) site is sampled in each year, which are also sampled at the same water depth.

We are certain that there were inaccuracies in how we manually measured the width of wetland opening in a GIS. To increase accuracy, geographic coordinates at both sides of the entrance should be collected each year so that they are available for future analyses. This is particularly important for the smaller marshes and in protected embayments with shallow openings. For larger marshes with very wide openings, this is not as great a problem, but more field information should be collected given the importance of metrics related to hydrogeomorphic site type in this study. We should also acknowledge that we have assumed that the only exchange between the Great Lakes and coastal marsh takes place through a visible breach in the barrier; however, if exchange occurs underground or along sub-surface breaches that cannot be seen from vertical images, then this measurement would not be accurate.

The DEMs we created for the wetlands should be considered only moderately accurate, especially for wetlands that have portions that are deeper than 2.0 m. This is because the point cloud elevation data in the inundated portion of the marsh have minimal data, given that the LiDAR could not penetrate the water. As well LiDAR are affected by sunlight reflections and water turbidity (Saputra et al. 2021). We recommend that areas with deep water (>2m) and where there is less vegetation, bathymetric data be collected with SONAR from a boat to generate better data to develop the DEMs.

We did not include wetlands that had been physically manipulated (with manual drawdown or refilling) for management purposes because their condition would not reflect impacts of LULC in their watersheds or the influences of the natural fluctuations of Lake Ontario. An example of this is Oshawa Second Marsh, which appeared to have undergone restoration efforts, with Common carp (*Cyprinus carpio*) removal during the early 2000s, and the wetland had been drained to remove *Phragmites* over the past decade (Oshawa, n.d.). We did include Cranberry Marsh, which had also been subject to a draw-down in the marsh and regulation of water level in the following years (LSCA 2005). The draining of the wetland led to a renewal of vegetation which appeared to have increased the vegetation community, and this led to improved water quality. The WQI scores were able to track the improvements as revealed in **Figure 2.9**, where CRA-1 shows the relatively poor condition of the wetland initially following restoration. 10+ years following the restoration, WQI scores for Cranberry have increased to a point that is predicted by % LULC classes.

We have demonstrated that WQI scores can be used to detect the impact of the opposing effects of % altered vs % natural land in wetland watersheds. We have also confirmed that the interaction of Lake Ontario water level with wetland bathymetry and hydrogeomorphic site type can modify the effects of % LULC on wetland water quality. It is therefore important to account for effects of water level when interpreting changes in WQI scores over the long term when water levels are variable, especially when environmental managers are attributing improved WQI scores to management actions and drawing conclusions on the health status of coastal marshes. Data to be collected at the time of sampling should include multiple water depths in the region where water samples are collected, and copious bathymetric readings from a boat-mounted sonar that can be used to modify DEMs based on LiDAR point clouds. For wetlands that are only periodically connected with Lake Ontario, we also recommend that water loggers be installed within the marsh. This would allow investigators to accurately relate water level-derived variables to WQI scores, especially for wetlands that are barred or enclosed.

This study was specific to coastal marshes in Lake Ontario that were mostly barred or enclosed types of riverine wetlands and may not be transferrable to wetlands such as in Georgian Bay, where there are no LULC impacts, and where the majority of the site types are open embayment or fringing shoreline lacustrine marshes. We have confirmed that the WQI is a useful tool for long-term monitoring of health status of disturbed watersheds on wetland water quality, despite the confounding effects of water-level fluctuations, which can be accounted for by our empirically derived multiple regression equation.

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Table 2.1 Years included and excluded ("ex.") in the study and within the 3 time periods, # of years included, site code, source of water-level data, sampling elevation. Cobourg and Kingston refer to levels measured at Cobourg station (#13590) and Kingston station (#13988). Logger refers to levels obtained from in-marsh recording devices.

Wetland name	Duration	# years included	Site Code	Data source	Elevation (masl)
Durham Region					
Bowmanville Marsh	2003-2017 ex. 2006	11	BOW	Cobourg	73.98
Carruthers Cr. Marsh	2003-2017 ex. 2006	11	CAR	Cobourg	74.31
Corbett Cr. Marsh	2004-2017 ex. 2006	10	COR	Logger	75.07
Cranberry Marsh	2004-2017 ex. 2006	9	CRA	Logger	74.90
Duffins Cr. Lakeshore Marsh	2009-2017	7	DUF	Coburg	74.09
Frenchman's Bay Marsh	2004-2017, ex. 2006	10	FRE	Coburg	74.36
Hydro Marsh	2003-2017	12	HYD	Coburg	74.38
Lynde Cr. Marsh	2004-2017 ex. 2006	11	LYN	Logger	74.36
McLaughlin Bay Marsh	2004-2017 ex. 2006	10	MCL	Logger	75.05
Pumphouse Marsh	2003-2017 ex. 2006, 2016	10	PUM	Logger	75.22
Rouge River Marsh	2004-2017 ex. 2006	10	ROU	Coburg	74.33
Westside Marsh	2004-2017	11	WES	Logger	74.85
Wilmot Cr. Wetland	2003-2017, ex. 2004 to 2007 incl.	8	WIL	Logger	74.59
Quinte Region					
Airport Cr. Marsh	2006-2017	9	AIR	Kingston	74.48
Big Island Marsh East	2005-2017	10	BIGe	Kingston	74.90
Big Island Marsh West	2005-2017 ex. 2011	9	BIGw	Kingston	74.91
Blessington Cr. Marsh East	2005-2017 ex. 2006	9	BLEe	Kingston	74.95
Blessington Cr. Marsh West	2006-2017	9	BLEw	Kingston	74.98
Carnachan Bay Marsh	2007-2017 ex. 2015	7	CBM	Kingston	75.02
Carrying Place Marsh	2006-2017 ex. 2010	8	CAP	Kingston	74.87
Dead Cr. Marsh	2006-2017	9	DEA	Kingston	74.75
Hay Bay North Marsh	2006-2017 ex. 2009, 2014	7	HAYn	Kingston	74.90
Hay Bay South Marsh	2003-2017 ex. 2004	11	HAYs	Kingston	74.91
Lower Napanee River Marsh	2006-2017 ex. 2007, 2014	7	LOWm	Kingston	75.28
Lower Sucker Creek	2006-2017	9	LOWs	Kingston	75.14
Robinson Cove Marsh	2003-2017 ex. 2004	11	ROB	Kingston	74.93
Sawguin Cr. Marsh Central	2006-2017	9	SAWc	Kingston	75.06
Sawguin Cr. Marsh Ditched	2006-2017 ex. 2010, 2014, 2015	6	SAWd	Kingston	75.26
Sawguin Cr. Marsh North	2006-2017	9	SAWn	Kingston	74.76

Table 2.2 Sampling location, wetland hydrogeomorphic class (Albert et al. 2005), and subwatershed area of wetlands. RRB=Barred Drowned River-mouth, BL=Barrier Beach Lagoon, RRO=Open Drowned River-mouth, LPP=Protected Embayment, and LOE=Open Embayment. See key to Site Code in **Table 2.1**.

Site Code	Sampling location		Hydrogeomorphic class code	Subwatershed area (ha)
	Latitude	Longitude		
BOW	43.8894699	-78.6697543	RRB	16 896
CAR	43.827635	-78.9837807	RRB	3 644
COR	43.8542363	-78.8867186	RRB	1 479
CRA	43.8436435	-78.9635719	BL	169
DUF	43.8192134	-79.0348442	RRB	28 645
FRE	43.8124486	-79.0953958	BL	1 704
HYD	43.8131073	-79.0774325	RRB	748
LYN	43.8490687	-78.9539648	RRB	13 030
MCL	43.8685293	-78.7995578	BL	265
PUM	43.8583307	-78.8418913	BL	171
ROU	43.7945286	-79.121464	RRB	33 251
WES	43.8870289	-78.6754954	BL	563
WIL	43.8962925	-78.5952424	RRB	9 849
AIR	44.1774952	-77.0984919	RRO	518
BIGe	44.1324319	-77.1953996	LPP	1 182
BIGw	44.0913364	-77.2513938	LPP	2 117
BLEe	44.1652235	-77.3165168	BL	493
BLEw	44.166399	-77.3262815	RRB	11 010
CBM	44.0757009	-77.0233748	RRO	1 727
CAP	44.0539221	-77.5723744	RRO	338
DEA	44.0673561	-77.5998361	RRO	633
HAYn	44.1794196	-76.9343773	RRO	19 694
HAYs	44.1612029	-76.8843288	LOE	1 487
LOWm	44.1994723	-76.9916703	LOE	17 471
LOWs	44.1717609	-77.1246805	RRO	13 354
ROB	44.1131322	-77.2831708	LOE	300
SAWc	44.1000818	-77.3605937	RRO	6 789
SAWd	44.0852998	-77.3417109	LPP	1 742
SAWn	44.1338957	-77.3712795	LOE	611

Table 2.3 Wetland opening width, and the minimum (Min) and maximum (Max) area, volume, and water depth of wetlands of Durham and Quinte Regions throughout the study. See key to Site Code in **Table 2.1**.

Site Code	Wetland opening (m)	Wetland area (ha)		Wetland volume (•10^5 m^3)		Water depth (cm)	
		Min	Max	Min	Max	Min	Max
BOW	45	132.04	399.28	21.85	44.35	19	62
CAR	50	49.26	193.83	10.30	20.31	52	94
COR	1	3.01	331.48	5.57	29.69	25	143
CRA	55	89.16	266.82	30.86	36.69	20	146
DUF	25	303.92	786.71	59.57	80.09	27	125
FRE	45	421.82	1 312.87	98.78	111.93	20	90
HYD	17	77.09	294.21	22.00	28.04	0	76
LYN	35	186.29	870.14	56.94	101.77	19	120
MCL	0	18.02	449.56	19.41	53.07	46	123
PUM	3	0.34	27.44	2.28	7.29	42	131
ROU	25	94.84	381.74	23.65	45.95	35	135
WES	15	20.30	223.86	14.79	28.14	0	180
WIL	2	8.99	82.98	4.58	14.51	50	150
AIR	40	3.08	89.76	1.15	30.10	45	155
BIGe	450	1.45	1 500.28	3.63	305.28	60	135
BIGw	100	1.29	1 259.04	3.33	368.56	0	65
BLEe	13	0.83	300.75	1.64	75.12	45	145
BLEw	27	1.23	158.72	2.51	34.69	35	125
CBM	430	0.38	152.67	0.88	25.12	30	220
CAP	20	0.10	49.95	0.21	12.22	20	70
DEA	130	1.19	223.18	1.60	71.96	0	125
HAYn	25	16.29	995.00	18.72	282.69	15	125
HAYs	850	0.03	240.82	0.08	48.95	45	145
LOWm	600	0.01	244.01	0.01	89.98	80	175
LOWs	425	0.06	313.76	0.16	55.97	40	155
ROB	330	1.18	154.75	2.56	21.85	90	175
SAWc	90	32.98	4 281.74	49.73	860.97	19	62
SAWd	130	0.89	1614.53	1.94	437.81	52	94
SAWn	33	0.26	199.99	0.54	39.52	25	143

Table 2.4 Comparison of median and mean (in brackets) %LULC in subwatersheds and wetland characteristics of 29 wetlands in the Durham (13) and Quinte (16) Regions. P-values correspond to the Z score for the Wilcoxon Rank Test comparison of data by region for means and medians (in bracket). The higher value is bolded if there are statistically significant differences.

Variable	Durham	Quinte	P-value
WQI score	-1.27 (-1.29)	**0.62** **(0.57)**	<0.0001 (<0.0001)
Subwatershed area (SUBarea; ha)	7 963 (1 704)	4 967 (1 607)	0.9826 (0.8970)
Wetland Area (WETarea; ha)	35.61 **(26.26)**	45.64 (16.80)	0.0615 (0.0309)
Wetland volume (WETvol; 10^3 m^3)	**209.72** **(159.98)**	106.54 (27.21)	<0.0001 (<0.0001)
Width of wetland opening (WOpening; m)	24.45 (25)	**230.81** **(115)**	<0.0001 (0.0021)
Mean wetland depth (cm)	**52.7** **(56.9)**	18.6 (13.5)	<0.0001 (<0.0001)
Water depth at sampling site (Wdepth; cm)	57.98 (53)	**80.98** **(75)**	0.0006 (0.0309)
% Agricultural land (% AGR)	40.93 (46.21)	**51.61** (50.79)	0.0174 (0.1952)
% Urban land (% URB)	**39.99** **(34.71)**	6.71 (3.24)	<0.0001 (<0.0001)
% Forested land (% FOR)	8.15 (8.08)	**14.51** **(15.37)**	<0.0001 (0.0096)
% Wetland (% WET)	8.69 (6.06)	**23.01** **(24.56)**	<0.0001 (<0.0001)

Table 2.5 Minimum (Min) and maximum (Max) % LULC classes in subwatersheds of wetlands in the Durham and Quinte Regions measured in three time periods. See key to Site Code in **Table 2.1**.

Site Code	% Agriculture		% Urban		% Forested		% Wetland	
	Min	Max	Min	Max	Min	Max	Min	Max
BOW	62.1	62.9	10.2	11.5	17.6	17.8	7.3	7.5
CAR	46.8	58.8	24.7	37.5	7.6	8.2	6.0	6.1
COR	18.1	24.1	71.3	77.5	1.7	1.8	2.6	2.7
CRA	50.8	51.7	11.8	12.9	1.7	1.7	12.6	12.8
DUF	56.6	56.7	15.8	16.0	17.0	17.2	6.0	6.1
FRE	18.7	19.2	60.9	62.4	10.1	11.0	2.7	3.0
HYD	6.5	9.9	80.9	84.4	1.0	1.1	4.3	4.3
LYN	60.1	64.2	13.5	18.2	12.6	12.7	6.0	6.3
MCL	0.0	50.9	8.3	31.7	16.2	60.0	23.2	10.5
PUM	15.3	16.7	77.7	78.0	0.0	0.0	3.6	3.6
ROU	44.7	50.6	33.6	40.2	8.1	8.2	4.7	5.1
WES	27.6	40.2	49.5	65.9	0.6	0.9	5.2	8.5
WIL	64.2	65.4	8.4	10.2	16.1	16.7	6.8	6.8
AIR	53.3	54.1	14.2	15.0	13.1	14.2	14.9	15.5
BIGe	50.2	50.8	2.2	2.2	18.6	19.3	25.8	25.9
BIGw	38.3	38.3	4.1	4.2	21.7	21.7	34.2	34.2
BLEe	27.2	27.2	16.1	17.0	20.1	21.4	26.4	26.7
BLEw	50.7	50.8	3.2	3.2	19.6	19.7	23.6	23.7
CBM	67.2	67.9	1.5	1.5	11.8	12.4	17.5	17.7
CAP	20.5	30.6	9.9	24.3	5.1	8.0	3.5	35.1
DEA	43.2	44.4	19.9	22.9	0.0	6.0	0.0	28.6
HAYn	65.7	65.8	3.1	3.2	15.9	16.1	13.4	13.5
HAYs	80.5	80.6	1.2	1.4	4.7	4.9	10.6	10.8
LOWm	56.8	57.1	7.2	7.6	17.4	17.6	14.8	14.8
LOWs	66.2	66.2	4.2	4.2	10.6	10.7	18.0	18.0
ROB	72.0	72.2	1.8	1.8	12.1	12.4	6.5	6.6
SAWc	42.6	51.9	1.8	1.8	12.1	14.8	28.2	40.6
SAWd	38.1	45.4	3.0	3.2	16.9	19.4	27.9	37.7
SAWn	8.4	33.7	6.7	14.3	20.4	24.3	25.4	62.3

Table 2.6 Multivariate regression models using LULC and hydrogeomorphology variables with increasing r^2-value and AICc values. SUBarea=area of subwatershed; Wdepth = water depth at sampling site (cm); WOpening = Width of wetland opening (m); % AGR = % Agricultural land in subwatershed; % URB = % Urban land in subwatershed.

Model	Variables	R^2	AICc
1	% URB, % AGR, WDepth, WOpening, SUBarea	0.7003	167.85
2	% URB, % AGR, WDepth, WOpening	0.6810	170.85
3	% WET, Wdepth, % URB	0.6629	175.59
4	% AGR, % WET, % FOR, WDepth, WOpening,	0.6672	176.88

Table 2.7 Summary of parameter estimates and their statistical significance corresponding to the best multiple linear regression model. SUBarea=area of subwatershed; Wdepth = water depth at sampling site (cm); WOpening = Width of wetland opening (m); % AGR = % Agricultural land in subwatershed; % URB = % Urban land in subwatershed

Term	Estimate	Std Error	t Ratio	P-value
Intercept	0.698493	0.468215	1.49	0.1397
% URB	-0.034591	0.004969	-6.96	<0.0001
% AGR	-0.024339	0.006725	-3.62	0.0005
WOpening	0.0014765	0.000396	3.72	0.0004
Wdepth	0.0126921	0.002507	5.06	<0.0001
SUBarea (m^2)	$-1.995 \cdot 10^{-9}$	$8.79 \cdot 10^{-10}$	-2.27	0.0258

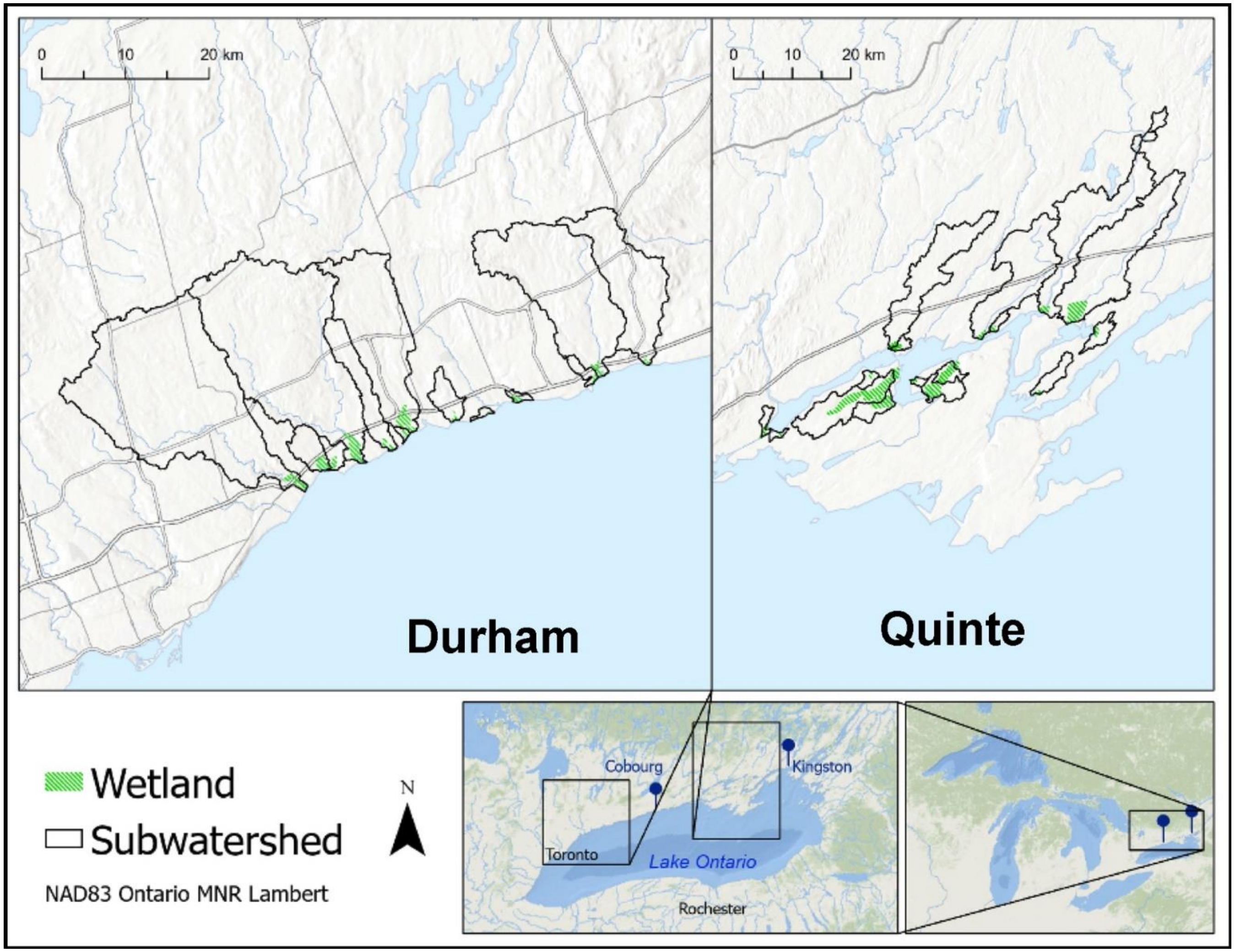


Figure 2.1 Wetlands and subwatersheds included in this study for the **a)** Durham region and **b)** Quinte Region.

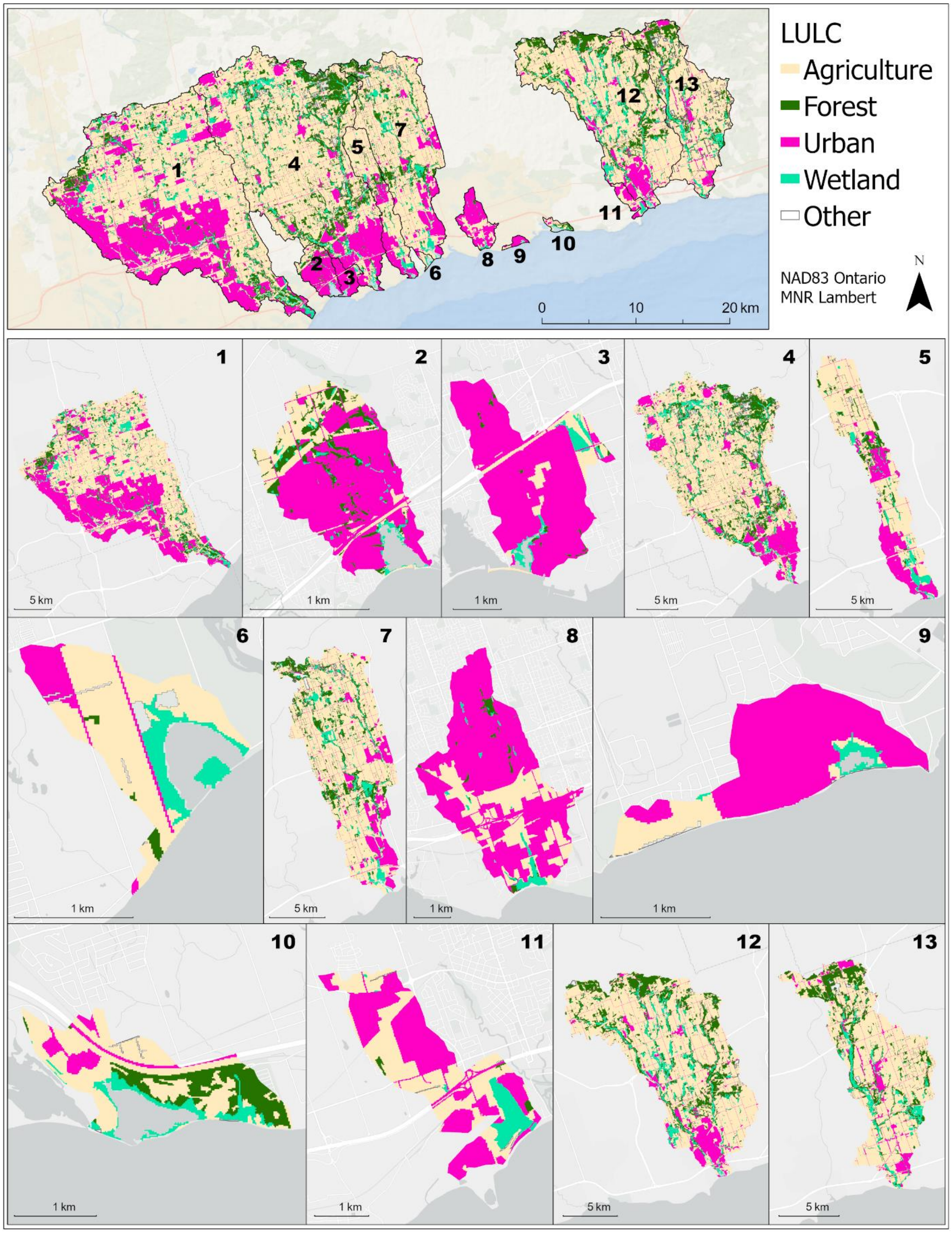


Figure 2.2 Distribution of four main land-use land-cover (LULC) classes in wetland subwatersheds in the Durham Region for SOLRIS 3. Key to marshes: **1)** Rouge, **2)** Frenchman's Bay, **3)** Hydro, **4)** Duffins, **5)** Carruthers, **6)** Cranberry, **7)** Lynde Creek, **8)** Corbett, **9)** Pumphouse, **10)** McLaughlin, **11)** Westside, **12)** Bowmanville, and **13)** Wilmot.

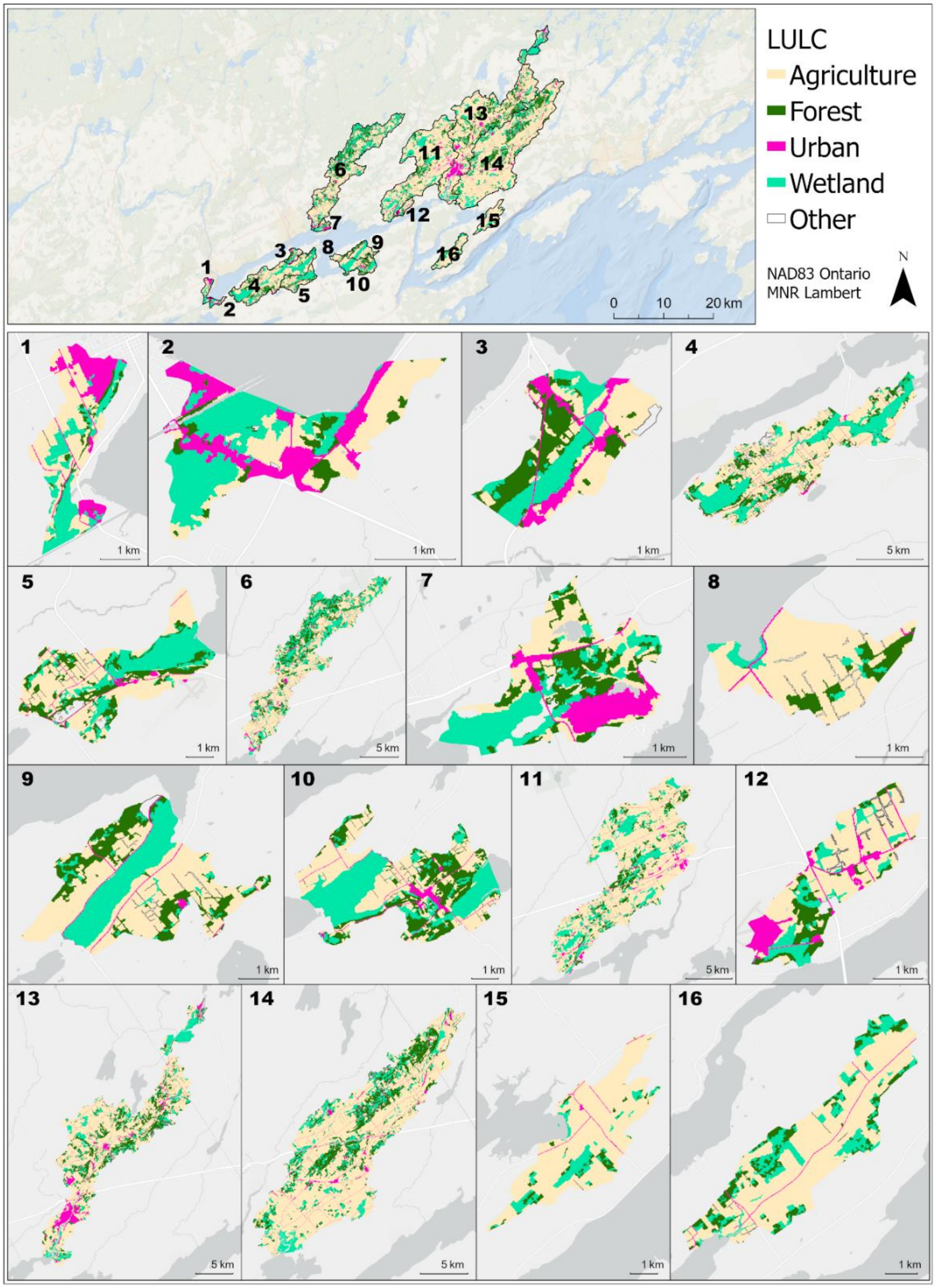
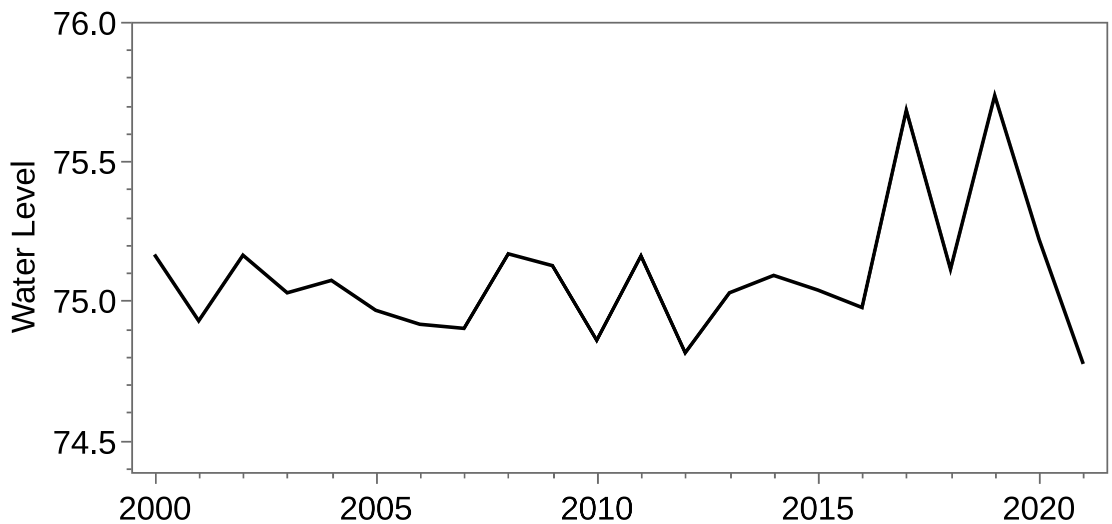


Figure 2.3 Distribution of four main land-use land-cover (LULC) classes in wetland subwatersheds in the Quinte Region for SOLRIS 3. Key for marshes: **1)** Dead Creek, **2)** Carrying Place, **3)** Sawguin North, **4)** Sawguin Central, **5)** Sawguin Ditched, **6)** Blessington W, **7)** Blessington E, **8)** Robinson Cove, **9)** Big Island E, **10)** Big Island W, **11)** Lower Sucker Creek, **12)** Airport Creek, **13)** Lower Napanee River, **14)** Hay Bay N, **15)** Hay Bay S, and **16)** Carnachan Bay.

a)



b)

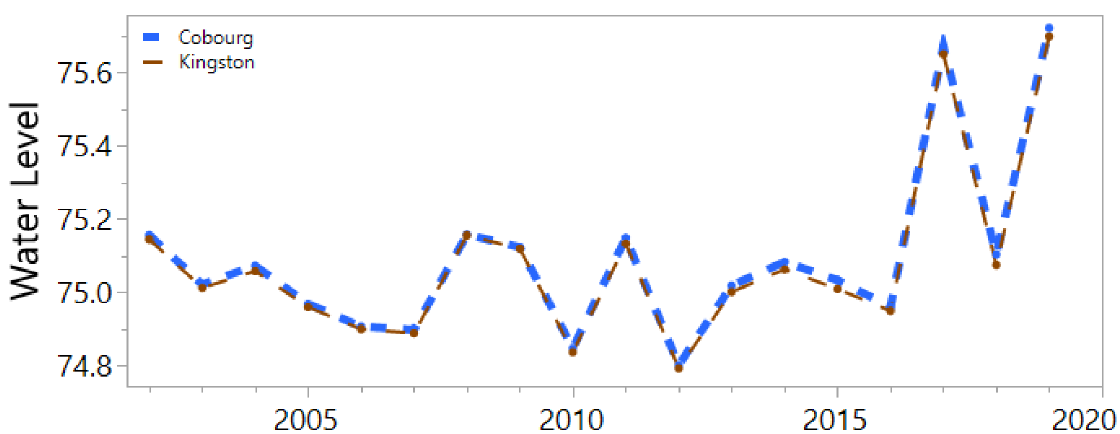


Figure 2.4 **a)** Mean annual water levels of Lake Ontario measured at Ontario Oswego and Rochester (New York), Cobourg, Port Weller, Toronto, and Kingston (Ontario) from 2000 to 2020. **b)** Mean water levels (masl) during sampling months (May through August) at the Cobourg station (#13590) for most wetlands in the Durham Region, and the Kingston station (#13988) for all wetlands in the Quinte Region.

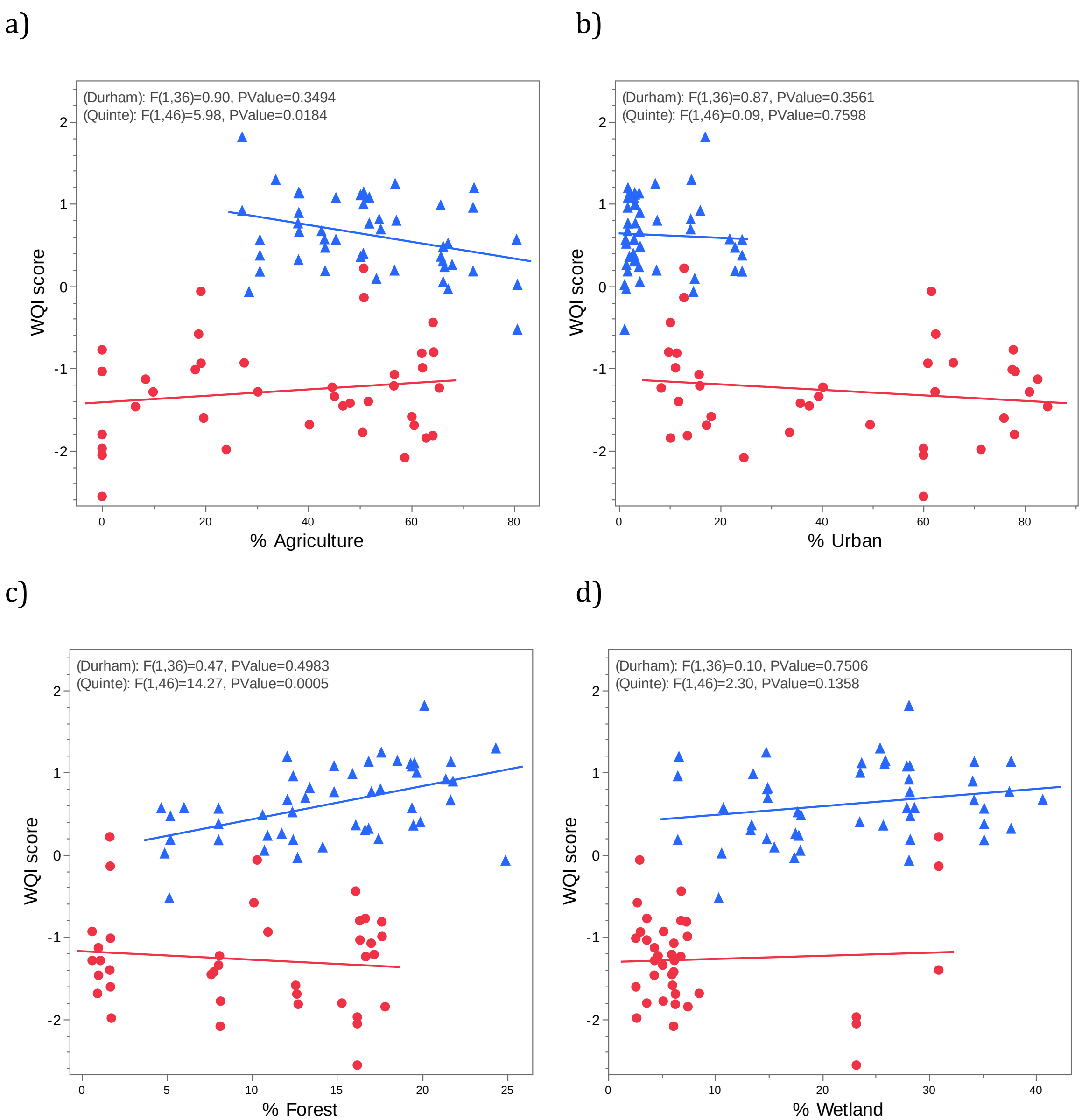


Figure 2.5 Mean WQI scores calculated by time periods versus percent **a)** agricultural land, **b)** urban land **c)** forested land, and **d)** wetland, plotted separately for the Durham (red circles) and the Quinte (blue triangles) Regions.

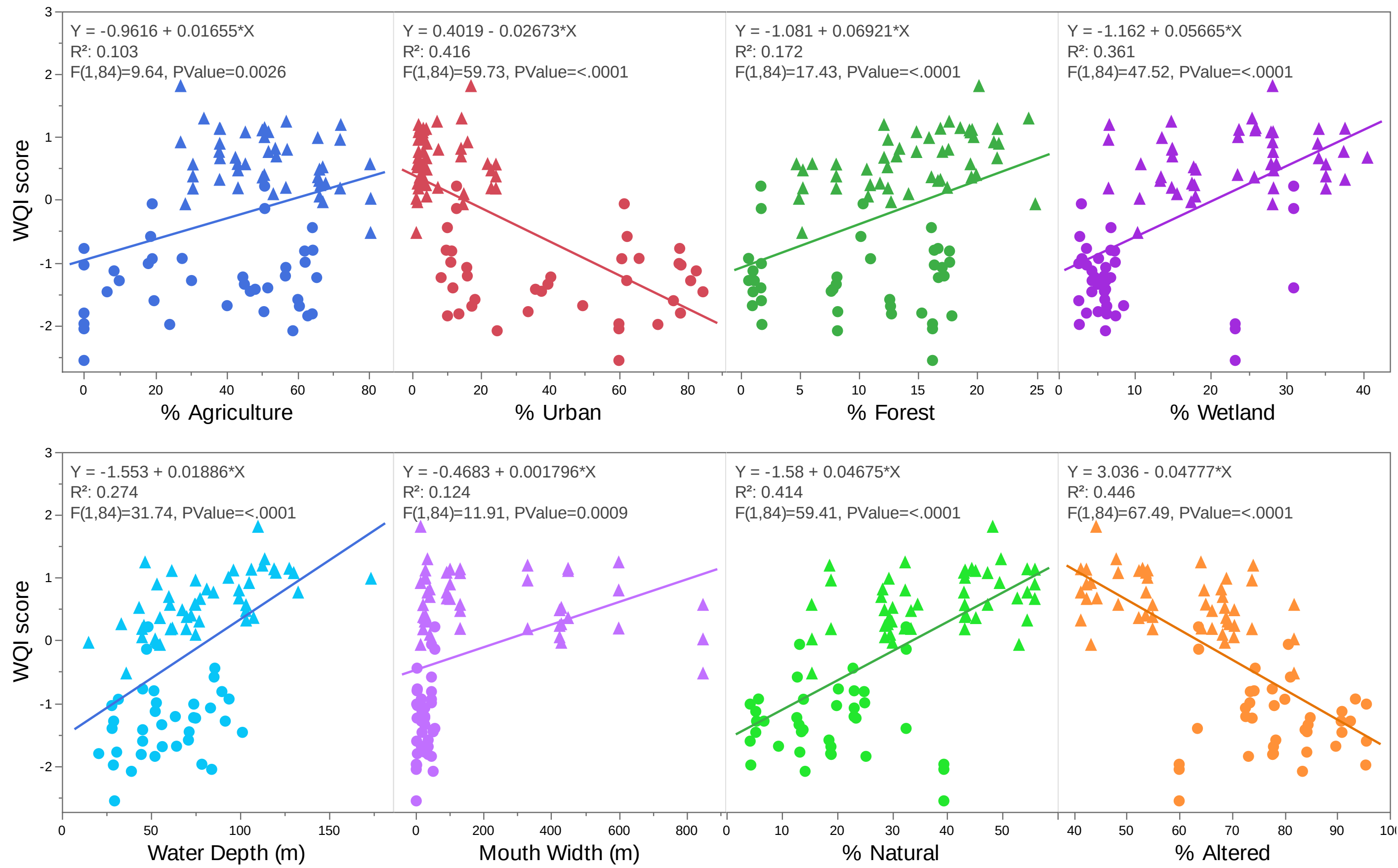


Figure 2.6 WQI scores regressed against LULC classes as well as water depth at the time of sampling and width of the wetland opening to Lake Ontario. % Natural is the sum of % FOR and % WET; % Altered is the sum of % AGR and % URB. Circles correspond to wetlands in the Durham Region whereas triangles correspond to wetlands in the Quinte Region.

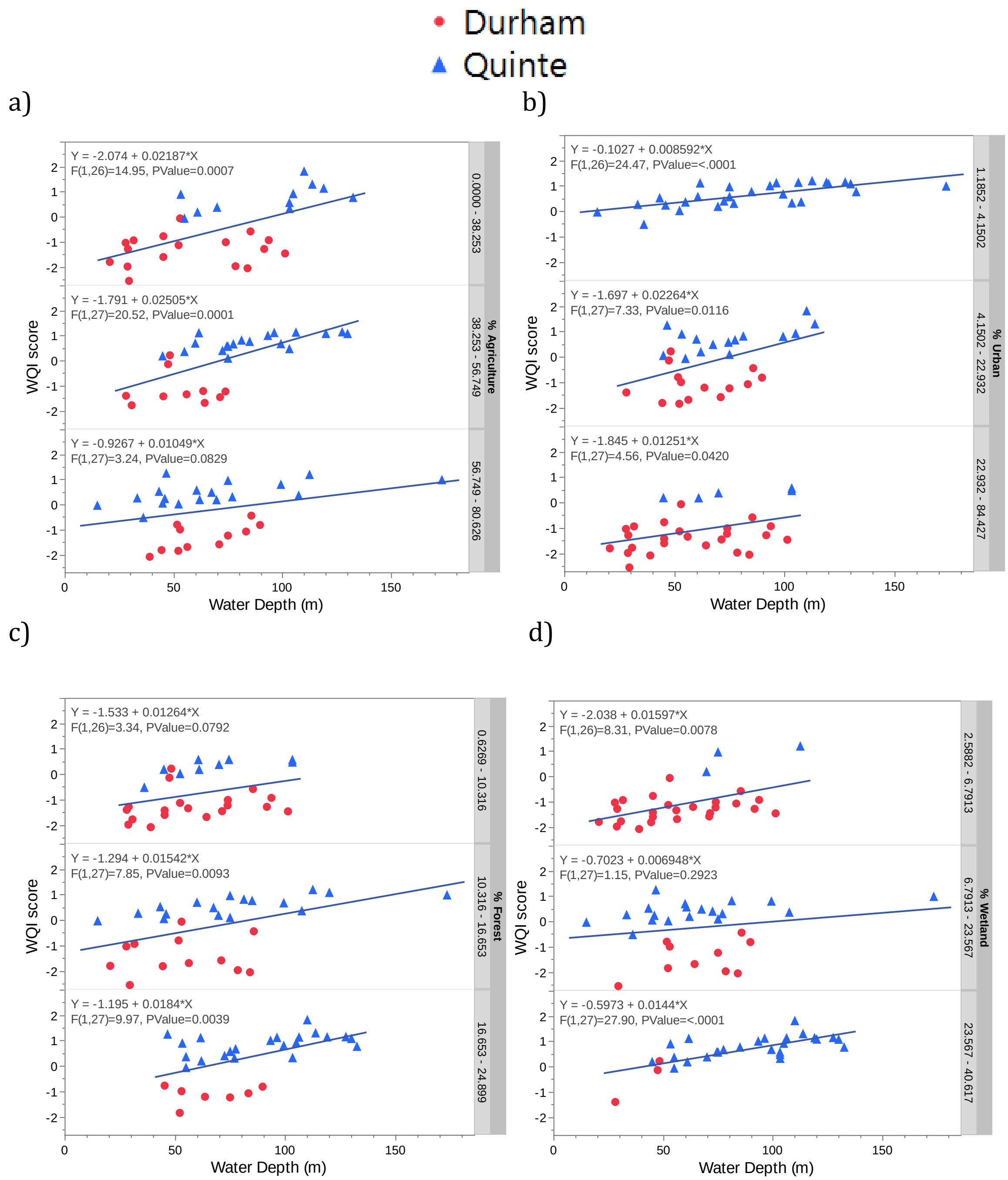


Figure 2.7 WQI score vs water depth (cm) plotted according to three increasing intervals of **a)** % AGR, **b)** % URB **c)** % FOR and **d)** %WET (n=29,29,28).

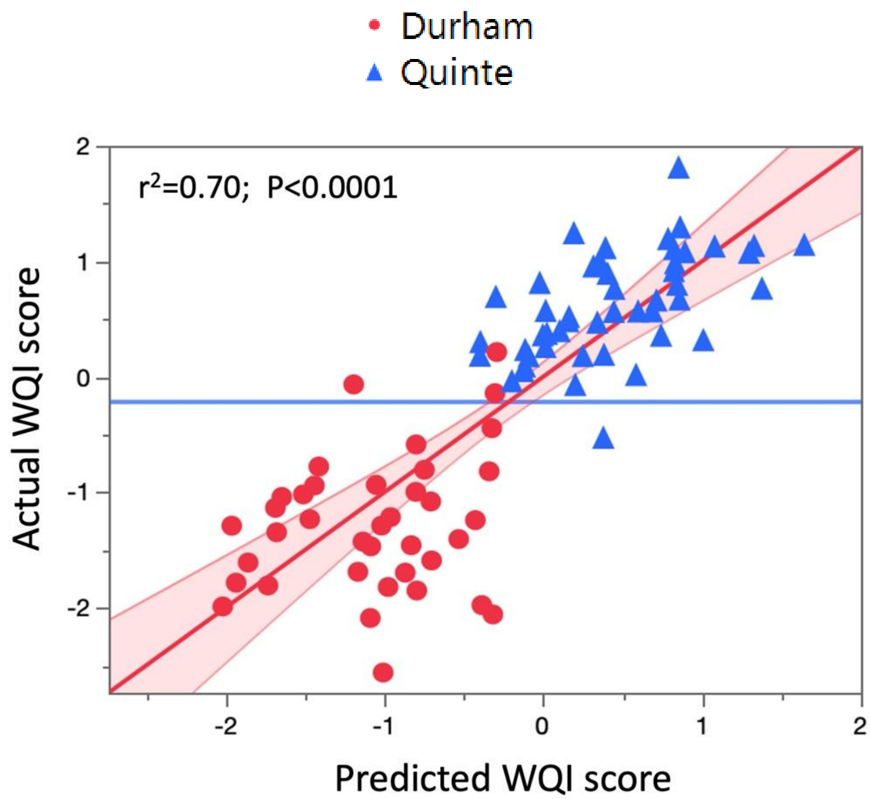


Figure 2.8 Actual WQI scores vs Predicted WQI score calculated from Eq. 2.1.

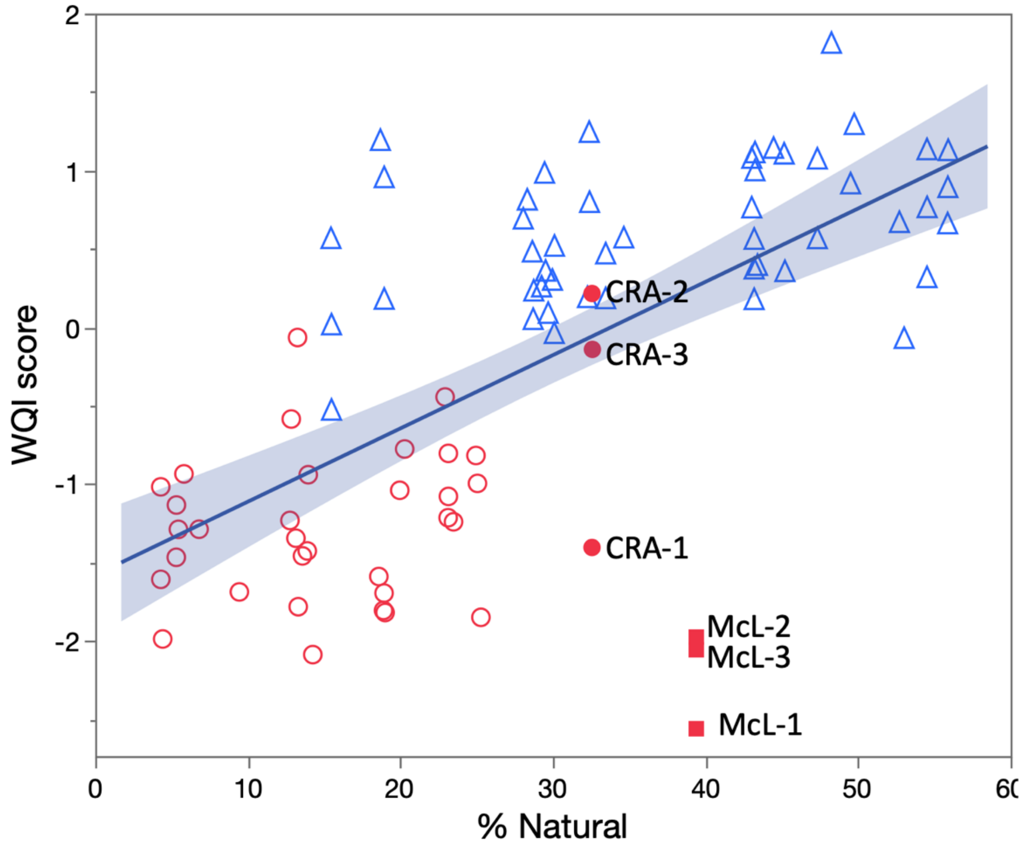


Figure 2.9 WQI scores vs % Natural Land in wetland subwatersheds. Durham wetlands are circles, while Quinte wetlands are triangles. CRA = Cranberry Marsh and McL = McLaughlin Marsh. The numbers appending the site codes indicate the time period 1 (2002-2007), 2 (2008-2011) and 3(2013-2017).