MONITORING WETLANDS DURING WATER-LEVEL TRANSITION PERIODS

INVESTIGATING THE EFFECTS OF WATER LEVEL ON DEPTH ZONES FOR MACROPHYTE DISTRIBUTION AND ECOLOGICAL INDEX PERFORMANCE IN COASTAL MARSHES OF GEORGIAN BAY, LAKE HURON

By

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LAY ABSTRACT

The coastal wetlands in the Georgian Bay area are primarily threatened by human development and the removal of annual water-level fluctuations. From 1999-2013, the water level decreased and remained low. In 2014, the water level rose about 1 m, causing flooding of grass and trees that had grown in the meadow zone during the 14 years when the water level was low. The first goal of this thesis is to explain how and why all wetland plants are relocating during this period. The second goal is to make sure that common indicators of wetland health (water quality, plants, and fish) can still be used during a time when flooding of grasses and trees was occurring in wetlands. The findings in this thesis contribute to the ability to predict and understand how the plants will shift within a wetland during a time of flooding, as well as informing managers on appropriate sampling protocols.

GENERAL ABSTRACT

Monitoring and maintaining the health of coastal wetlands is a global concern. The greatest threat to coastal wetlands in the Great Lakes Basin are anthropogenic removal and enrichment. The coastal wetlands in Georgian Bay are relatively undisturbed by humans, but face disturbance caused by reduced annual water-level fluctuations. Since these wetlands are critical habitat for many fish, bird, amphibian, and reptile species, many efforts to accurately monitor and maintain their health have been put into place. Recently, these wetlands have been experiencing an abrupt (~1 m) transition to higher water levels, following 14 years of sustained lows, which allowed trees and shrubs to invade the meadow vegetation zone. This sustained water-level pattern has never occurred in this region before, offering the unique opportunity to study wetlands undergoing a transition, where areas of 10+ years of upland plant species growth was inundated and became part of the wetland habitat. This thesis first investigates how this change in water level affects the distribution of meadow, emergent, floating, and submerged vegetation both in physical space and area. The second chapter of this thesis presents long-term water quality, macrophyte, and fish community monitoring using ecological indices. Water quality and macrophyte indices are robust enough to monitor wetlands undergoing a transition; however, issues arise in the calculation of the wetland fish index, as the changes in macrophyte distribution described in Chapter 1 impact the ability to replicate community sampling using fyke nets. The research done throughout this thesis is highly beneficial in adding to the limited knowledge of key factors

impacting macrophyte community shifting. This work also identifies water-level scenarios where managers must adjust sampling protocols to succeed in effectively sampling wetland fish communities.

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LIST OF ALL ABBREVIATIONS AND SYMBOLS

Δ Water level	Change in water level
CHL	Chlorophyll-α
COND	Conductivity
DEM	Digital elevation model
GIS	Geographic information system
GPS	Global positioning system
IBI	Index of biotic integrity
LIDAR	Light detection and ranging
m ASL	Meters above sea level
OIH	Ontario integrated hydrology
OMNRF	Ontario Ministry of Natural Resources and Forestry
pCCA	Partial Canonical Correspondence Analysis
SAV	Submerged aquatic vegetation
SRP	Soluble reactive phosphorus
TAN	Total ammonia nitrogen
TEMP	Temperature
T_i	Tolerance to degradation value
TISS	Total inorganic suspended solids
TN	Total nitrogen
TNN	Total nitrate nitrogen

TP	Total phosphorus
TSS	Total suspended solids
TURB	Turbidity
U_i	Niche breadth value
UAV	Unmanned aerial vehicle
WFI	Wetland Fish Index
WMI	Wetland Macrophyte Index
WQI	Water Quality Index
Y_i	Abundance
YOY	Young-of-the-year

DECLARATION OF ACADEMIC ACHEIVEMENT

For the first chapter of this thesis, I participated in the collection of bathymetry data in 2016. Other elevation data was previously obtained by the lab of Dr. Pat Chow-Fraser. Post-processing of the 2016 sonar data was completed by Dan Weller. Dr. Jon Stone and I created the mathematical model for use in predicting shoreline movement. I preformed all subsequent GIS analyses. I led macrophyte surveys and stem density count sampling in 2015-2016, and historic vegetation data had been previously collected by the students of Dr. Pat Chow-Fraser. For the second thesis chapter, I undertook and oversaw all water quality, and fish and macrophyte community sampling for 2015/2016. I also completed all water quality nutrient processing in the laboratory with guidance from Julia Rutledge and Stuart Campbell. Historic data collected between 2003-2011, were compiled by previous students of Dr. Pat Chow-Fraser. Dr. Pat Chow-Fraser and I performed all statistical analyses in this thesis.

GENERAL INTRODUCTION

Georgian Bay Coastal Wetlands

Georgian Bay is a large water body extending from the north-eastern side of Lake Huron. Georgian Bay has an approximate surface area of 15 000 km^2 and a maximum depth of 165 m near the main channel where it connects to Lake Huron. Though directly associated with Lake Huron, Georgian Bay has many features that make it morphologically and functionally distinct. The basin of Georgian Bay is largely composed of granitic bedrock, with some areas of sedimentary rock at the southern and western portions of the lake. The glacial retreat 10 000 years ago carved the complex, shoreline that we see today. The shoreline of Georgian Bay consists of many islands, channels, and sheltered embayments. This intricate shoreline is also attributed to the several thousand coastal wetlands present along the shorelines of the islands and mainland (Midwood et al. 2012).

In Canada, wetlands are defined as "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation, and various kinds of biological activity adapted to wet environments" (National Wetlands Working Group 1988). Furthermore, Great Lakes coastal wetlands are defined as wetland communities located between permanent aquatic and permanent upland environments along the shores of lakes that are affected by large-lake processes such as waves, wind tides, and long-term water level fluctuations (Maynard and Wilcox 1997).

Coastal wetlands provide valuable ecosystem services such as nutrient filtration and shoreline stabilization (Maynard and Wilcox 1997), as well as being critical habitat for a number of species. The high-marsh zone of coastal wetlands is typically important for avian species (Riffell et al. 2001), and the aquatic portion of the wetland is critical spawning and nursery habitat for economically important sportfish such as the northern pike and muskellunge (Jude and Pappas 1992; Leblanc et al. 2014). It has been determined that the coastal wetlands in the Georgian Bay area are among the most pristine in the Great Lakes basin (Cvetkovic and Chow-Fraser 2011) due to the smaller amounts of anthropogenic activity in the area (Chow-Fraser 2006; Seilheimer and Chow-Fraser 2007). However, the largest current threats to these wetlands has been identified as an increase in anthropogenic development (Campbell 2017) and periods of sustained low water levels (Midwood and Chow-Fraser 2012).

Water Levels and Climate Change

Lakes Superior, Ontario, and Erie have at least one man-made flow control structure at one of the major connections to the other lakes, allowing regulation of water levels and limited annual fluctuation. Lakes Michigan and Huron are hydrologically connected through the large Straits of Mackinac and lack control structures at the St. Clair outlet of Lake Huron, resulting in the largest range in water levels of the Great Lakes. The water levels of Lake Michigan-Huron, and consequently Georgian Bay, have historically fluctuated 4 m around the mean of 176.42 m ASL (Bishop 1990).

The variability in water level occurs at different scales, and is the consequence of multiple factors that are difficult to predict and model. Storm events can influence the smaller hourly and daily fluctuations, whereas, monthly variability is usually caused by the seasonal hydrologic cycle. On a larger scale, annual and decadal changes in water level are currently attributed to changes in climate (Bishop 1990; Lenters 2001). The many factors affecting water level and the variation within each of these factors, has made it difficult to determine cycles that the water levels follow. Studies using different statistical and spectral analyses have found varying results, with the most common being cycles of ~8, ~12, and ~30 years (Liu 1970; Cohn and Robinson 1976; Hanrahan et al. 2009). In 1999, Lake Michigan-Huron entered an unprecedented period of sustained water levels lasting 14 years. In 2014, the water level rose ~ 1 m to sit above the long-term mean level. This prolonged period of low water levels, followed by an abrupt increase, does not follow the periodic patterns seen in the historic water level data and can potentially be attributed to climate change. Current models used to predict future water levels involving climate scenarios suggest a decline in the water level of Lake Michigan-Huron, with an increase in mean seasonal range (Angel and Kunkel 2010; MacKay and Seglenieks 2013). The level of uncertainty surrounding the future of climate and water-levels in this region, make it crucial to understand the influence that these potential changes have on coastal wetlands, so that management or policy update actions can be taken to protect sensitive areas.

Long-term Monitoring of Coastal Wetlands

The high economic and ecologic value of coastal wetlands have made monitoring and management of their health a priority for a number of initiatives and organizations including the Great Lakes Wetlands Conservation Action Plan (GLWCAP) that involves organizations such as Environment Canada and Climate Change, Conservation Ontario and Ducks Unlimited Canada (GLWCAP 2010). The majority of these initiatives involve the development and use of ecological indicators or indices of biotic integrity (IBIs). These indices can involve water quality parameters or any number of biotic community such as macrophytes, invertebrates, fish, or zooplankton (Lougheed and Chow-Fraser 2002; Uzarski et al. 2004, 2005; Seilheimer and Chow-Fraser 2006; Croft and Chow-Fraser 2007). In general, biotic indicators are preferred because of their cost effectiveness and ease of implementation due to the fact that generally only the identification of species is required to assess wetland health. The range in sediment composition, and general morphology of coastal wetlands throughout the Great Lakes have made it difficult to create indicators appropriate for accurately assessing health across the region (Uzarski et al. 2016). Thus, three ecological indicators using water quality, wetland macrophytes, and wetland fish, have been developed specifically for use in the Georgian Bay region (Chow-Fraser 2006; Croft and Chow-Fraser 2007; Seilheimer and Chow-Fraser 2007).

Thesis Objectives

The overarching goal of this thesis is to determine how coastal wetlands are influenced by an abrupt 1 m increase in water level following 14-years of sustained low water levels. In Chapter 1, a simple mathematical model using wetland basin slope is developed to predict how far vegetation communities within a wetland must move in order to reestablish in their preferred depth zones following a change in water level. Fine-scale digital elevation models (DEMs) created with a Geographic Information System (GIS) were used to calculate the amount of area available for macrophyte growth under both high and low water-level scenarios, and to determine the level of resilience that each vegetation type has to large water-level changes.

In Chapter 2, the water quality index (WQI), wetland macrophyte index (WMI), and wetland fish index (WFI), are used to assess the status and trends of Georgian Bay wetland health over a period of 13 years. The performance of the indicators during a period of water-level transition is examined. Ultimately, this thesis will provide insight into the scope of the physical changes in wetland macrophyte community structure during a period of water level increase, preceded by a period of sustained low levels. The findings in this thesis can be used to identify wetlands that are susceptible to sustained low water levels and inform policies and sampling protocols for monitoring wetlands in a period of transition.

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Chapter 1:

Interactions between water level and slope on vegetation distribution in coastal marshes of Georgian Bay

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Abstract

Over the last decade, Lake Michigan-Huron experienced an unprecedented period of sustained below-average water levels. The lowest experienced water level occurred in 2013 (175.93 m ASL), 0.49 m below the long-term mean (176.42 m ASL). In 2016, the water level abruptly rose 0.91 m to 176.94 m ASL, 0.52 m above the long-term mean. This scenario has provided the unique opportunity to study how four major vegetation types (meadow, emergent, floating, and submerged aquatic vegetation (SAV)) respond under two different water-level scenarios. Using fine-scale (5 m) digital elevation models (DEMs) for 25 coastal wetlands in eastern Georgian Bay, we created a simple mathematical model, and preformed GIS-based analyses to study how the slope and bathymetry of wetlands affect the distribution of vegetation faced with large, abrupt changes in water level. Slope was negatively correlated with the amount of horizontal movement travelled by vegetation communities, although the distance varied considerably within each wetland because of the highly complex, site-specific geomorphology. Using our fine-scale DEMs, we calculated that the amount of area suitable for potential emergent, floating, and submerged vegetation growth increased with higher water levels, whereas the area of the wetland suitable for meadow vegetation became compressed. We used the conserved area (i.e. the overlapping area for each vegetation type between low and high water levels) as a measure of resilience to the water-level change. The SAV was the most resilient to the water-level increase, and the meadow was the least resilient. These expected changes predicted in a GIS environment were not reflected in our field vegetation sampling during the high-water scenario, where

we found the meadow vegetation growth area expanded by an average of 1 m. We attribute this discrepancy to the lag-time required before the wetland vegetation communities can reestablish in a typical fashion.

Introduction

Freshwater coastal wetlands are found globally, wherever large freshwater lakes exist. The hydrologic connection between coastal wetlands and lakes gives them unique properties, as they are subject to many large-lake processes not present in inland wetlands. In the North American Great Lakes region, coastal wetlands are defined as those located between permanent aquatic and permanent upland environments and are affected by large-lake processes such as waves, wind tides, and water-level fluctuations (Maynard and Wilcox 1997). The coastal wetlands in the Great Lakes Basin are key habitat for many economically and ecologically important sportfish (Jude and Pappas 1992; Leblanc et al. 2014) and avian species (Riffell et al. 2001), as well as key habitat for species at risk such as the Blanding's turtle and least bittern (Ontario Ministry of Natural Resources and Forestry (OMNRF) 2017).

Not only are the Great Lakes coastal wetlands subject to decline due to anthropogenic removal (Ducks Unlimited Canada 2010) and disturbance (Cvetkovic and Chow-Fraser 2011; Campbell 2017), but they are also suffering from a lack of annual water-level fluctuations in some areas (Bishop 1990). Lakes Superior, Ontario, and Erie have at least one control structure at one of the major connections to the other lakes, allowing regulation of water levels and causing limited annual fluctuation. Lakes Michigan and Huron are hydrologically connected and lack control structures and regulations, resulting in the benefit of the largest range in water levels in this region. Due to the lack of regulation, the water level of these lakes is influenced by climate and meteorological patterns, and the dredging and continued erosion of the St. Clair River

(Lenters 2001; International Upper Great Lakes Study (IUGLS) 2012; MacKay and Seglenieks 2013). It is not uncommon for water levels in this region to fluctuate up to 4 m around the long-term mean (176.42 m ASL) (Bishop 1990; IUGLS 2012).

In general, coastal wetlands are comprised of four different vegetation types: meadow, emergent, floating, and submerged aquatic vegetation (SAV). Each type of vegetation has specific characteristics including physical structure, depth preference, and suitability as habitat (Table 1.1). The constraints on the different vegetation types to grow within specific depth ranges is what creates relatively distinct zones within the wetland where meadow, emergent, and submerged vegetation persists. The floating vegetation grows mixed within the emergent and submerged vegetation, so it does not create its own zone. Annual water-level fluctuations are extremely important to the structure of vegetation community by adding a source of disturbance so that singular plant species cannot dominate the wetland (Gathman et al. 2005; Midwood and Chow-Fraser 2012). However, absent water-level fluctuations can disrupt the wetland community by causing homogenization (Midwood and Chow-Fraser 2012) or causing extreme flooding or drying in upland environments (depending on the respective direction of the fluctuation). The amount of water-level-related disturbance a wetland is subject to, is directly related to its orientation and geomorphology (Keough et al. 1999).

The coastal wetlands in Georgian Bay (eastern extension of Lake Huron) are geomorphologically distinct from most coastal wetlands in the Great Lakes region. The granitic bedrock basin distinguishes Georgian Bay coastal wetlands from those in the more southern Great Lakes that are situated on soft sedimentary rocks (Chow-Fraser and

Albert 1999). The complex shoreline composed of many islands where several thousand small coastal marshes are located (Midwood et al. 2012), further differentiates the Georgian Bay region from the shores of Lake Superior. In addition to having unique geomorphology, as an extension of Lake Michigan-Huron, the Georgian Bay region is subject to the largest and most unpredictable water level fluctuation range compared to the other Great Lakes (Bishop 1990).

Starting in 1999, the water levels of Lake Michigan-Huron decreased below the long-term mean level (176.42 m ASL), where they remained for 14 years. This is unprecedented in this region, as it breaks most of the cycles of annual water-level fluctuations (~8, ~30 years; Liu 1970; Hanrahan et al. 2009). During the sustained, low water-level period (1999-2013), the lowest experienced water level occurred in 2013 (175.93 m ASL). In 2014, the water level abruptly rose above the long-term mean, to 176.84 m ASL (0.91 m above the minimum level in 2013), where it has remained stable through 2016. This specific scenario provided the unique opportunity to study how four major vegetation types (meadow, emergent, floating, and submerged aquatic vegetation (SAV)) respond under transition from low-water to high-water conditions.

The movement and areal change of wetland vegetation corresponding to changes in water level have been an area of interest for many researchers (Keddy and Constabel 1986; Keddy and Reznicek 1986; Gathman et al. 2005). More recently, the use of GIS (Geographic Information Systems) to quantify and model these changes have become popular (Wilcox and Xie 2007; Hebb et al. 2013). Up until this point, there have been no studies using GIS to quantify these changes in the Georgian Bay region. The primary goal of this study is to develop a simple mathematical model to predict how the location of wetland vegetation types will change under low and high water-level scenarios. This model will be compared to actual changes in location calculated using GIS. In addition, the effectiveness of wetland basin slope as a predictor of these changes will be evaluated. We expect that wetlands with a shallower basin slope will experience a greater amount of vegetation community change caused by large, abrupt water-level changes. The secondary goal is to use GIS to determine the areas that are suitable for each type of vegetation growth under low and high water levels, and their respective resilience to the observed water-level increase. An examination of how the plant communities are changing in location and density during this time were evaluated with extensive field data collected between 1981 and 2016. Being able to accurately predict changes in vegetation and subsequently habitat location and area can help inform wetland protection and management policy when determining which areas may be more at risk of reduced water-level fluctuations, as wetland protection is currently set on a basis of wetland size, location, and function (OMNRF 2013; 2016).

Methods

DEM Creation and Site Slope Calculation

First, we used a GIS (ArcGIS 10.5; ESRI, Redlands, California) to develop digital elevation models (DEMs) for 25 Georgian Bay coastal wetlands (Figure 1.1). Multiple different topographic and bathymetric data sources were used to create the best-possible DEMs, including field surveys and existing, open-access topographic and bathymetric data sets. We used the open-access Ontario Integrated Hydrology (OIH) DEM as the primary source of upland (i.e. >176.0 m ASL) elevation information. Bathymetric data for each site was collected using a commercially available sonar unit (Lowrance Elite 5-Ti or comparable) mounted on a canoe or 16 ft aluminum boat. Sonar surveys typically consisted of a pass following the 0.5m-depth contour of the wetland, and a series of parallel transects across the wetland spaced approximately 10-50 m, depending on wetland size and shape. Handheld GPS units (Garmine eTrex Legend or comparable) were used to collect spot depths (between 0 and 1 m deep) or complete shoreline traces within wetlands. Survey-grade dGPS units were used to collect spot elevations in the nearshore zone (approximately 1m above shore to 1m deep at time of survey). Depth contour data were accessed from the open-access Great Lakes bathymetry dataset from the National Oceanic and Atmospheric Administration (1996). The OIH and sonar survey data were available for all 25 wetland sites. The other elevation datasets had limited coverage and were only incorporated where available. The inclusion of additional data sources did not change slope or any subsequent calculations carried out using the DEM. We wanted to use all available data to have the most accurate product possible.

For each of the 25 wetlands in our study, we used the Topo to Raster function in ArcMap 10.5 to construct a fine-scale DEM (5 m resolution) with all elevation data available.

We used the Slope tool in ArcMap to determine the slope value at each cell of the DEM and averaged the slope values across the wetland to obtain one representative site slope. The site slope measurements were bound to elevations below 176.84 m ASL (Period 2 shoreline) and above 174.93 m ASL (1 m depth contour for Period 1). This range represents the area of the entire site that could potentially be inundated under typical wetland water depths (0 – 1m). This and all subsequent spatial analyses were completed with ArcMap 10.5.

Mathematical Model Development

The mathematical model was developed to determine the magnitude of horizontal movement that each of the vegetation types would experience for a given change in water level. This model was developed assuming a simple wetland shape with uniform slope along a completely straight stretch of shoreline. This model was created in and run using Wolfram Mathematica 7.0 (Wolfram, Champaign, Illinois).

The basic model function was defined as:

$$\text{Horizontal Movement} = \frac{\Delta\ \text{Water Level}}{\text{Slope}}$$

We used this model to predict the horizontal movement of the shoreline or any vegetation type for 25 wetlands (for which the slope was calculated in ArcMap), experiencing a 0.91 m increase in water level.

For all analyses, we focused on the change in water level from the sustained-low-water scenario (1999-2013; Period 1) to the recent rebound to above average-water levels (2014-present; Period 2). The difference in water level between these two periods was 0.91 m, which was calculated by subtracting the minimum mean monthly water level associated with Period 1 (175.93 m ASL occurring in May 2013) from the maximum mean monthly water level in Period 2 (176.84 m ASL which occurred in July 2016).

Shoreline Movement

For each wetland, we used the mathematical model to predict the horizontal movement of the shoreline given the 0.91 m water level increase between Period 1 and Period 2. The shoreline was used as a general proxy for how the distribution of the vegetation types would respond to the water-level change. We also used the site DEMs to measure the shoreline movement in a GIS. We delineated the shorelines for water levels corresponding to Period 1 (175.93 m ASL) and Period 2 (176.84 m ASL). Between the shorelines associated with Period 1 and Period 2, we drew ten evenly spaced transects perpendicular to the shore. The mean distance of the 10 transects was used as the measure of horizontal shoreline movement between Period 1 and Period 2. Minimum and maximum transect length, range, and standard error were also calculated for each site to more thoroughly describe the morphological complexity of the wetland. This measurement was repeated with the DEM for each of the 25 sites.

To determine if slope is a key factor in the horizontal distance moved by the shoreline under different water levels, a linear regression analysis was conducted on log transformed horizontal movement values calculated using a GIS, and the average wetland site slope.

In order to investigate the effectiveness of the mathematical model, using slope as the sole predictor of horizontal shoreline movement, the predicted horizontal movement values from the mathematical model and the calculated horizontal movement from the DEMs were compared. Due to non-normality in the dataset, a Wilcoxon signed rank test was used to compare horizontal shoreline movement measurements for each of the 25 wetlands.

Change in Suitable Growth Area

We used field data and ArcMap to determine changes in the location and area that is suitable for growth of each of the vegetation types between Period 1 and Period 2. High-resolution aerial imagery was not available for all 25 sites in both periods so instead we operationally defined the boundaries of each of four vegetation types based on detailed field surveys of eight coastal marshes in Georgian Bay conducted between 2004 and 2009 (during Period 1). These data had been collected as part of a synoptic survey by Cvetkovic and Chow-Fraser (2011). All wetland plants in quadrats were identified as meadow, emergent, floating or submerged aquatic vegetation (SAV), and the location and depth information were recorded. We used the 2.5% and 97.5% quantiles as the upper and lower depth thresholds, respectively. At the time that these data were collected, water levels had remained below the long-term mean for at least five years; therefore, we are confident that the wetland vegetation were found in their preferred depths.

To compare how the potential wetland area available for the growth of each of the vegetation types differed between water-level scenarios, the DEMs created for each of the wetlands were used. Each site DEM was reclassified to identify the area corresponding to the depth range identified for each vegetation type. For example, the area within the wetland suitable for meadow growth (e.g. 0 – 30 cm deep) was identified for the same wetland under low water level conditions (175.93 m ASL) and high water level conditions (176.84). The location and area of each vegetation type (e.g. emergent, SAV) was determined in this manner for both water-level scenarios in all 25 wetlands. Total wetland area, as defined by shore to the maximum depth of the SAV, was also calculated at each site and for each period. To determine any differences in area, we log transformed the area data to fit normality assumptions and preformed a paired t-test using the 25 sites comparing suitable growth area under Period 1 and Period 2 conditions.

We also sought to quantify the resilience of the different vegetation types within each wetland to the 0.91 m water level increase. The conserved area, the overlap between a vegetation type growth area between Period 1 and Period 2, was used a proxy measure for resilience. Using the floating vegetation as an example, the potential area occupied by floating vegetation during both Period 1 and 2 was defined as the "conserved area". Floating vegetation in the conserved area was considered resilient to water-level disturbance because it could grow in that location under both water-level scenarios. In contrast, any floating vegetation occurring outside of the conserved area would die back either because conditions are too deep or two shallow. For each vegetation type we calculated the conserved area between the Period 1 and Period 2.

Field Validation

We acquired vegetation species, location, and depth data for 27 (324 depth measurements) wetlands during Period 2, according to the same field protocols described previously. These data were used to compare the depth range of each vegetation type between Period 1 and Period 2. The depth distribution for each type of vegetation was plotted, and we used the 2.5% and 97.5% quantiles as the upper and lower depth thresholds, respectively, to estimate depths occupied by each vegetation type. To assess the change in the depth ranges occupied by each of the vegetation types, the range occupied in Period 2 was subtracted from the depth range occupied during Period 1.

Three wetlands in our study had been sampled in 1981 (Craig and Black 1986), when water levels had been relatively similar to those experienced in 2016 (176.81 vs 176.84 m ASL, respectively) and were subject to typical annual fluctuations, and in 2012 (Leblanc et al. 2014), during the stable, low-water conditions of Period 1. In both studies, stem counts had been collected in a standardized way along three transects per wetland that extended from shore to 1-m depth contour. We used the same protocol (Craig and Black 1986) to collect stem counts during 2016 at three sites. For each transect, ten 0.25 m^2 quadrats were sampled. In each quadrat, the number of stems of emergent and floating vegetation were recorded. In 1981, submerged vegetation cover was visually determined, and in 2012 and 2016 a rake was utilized to pull up the vegetation in the quadrat, so that submerged vegetation stem counts could be obtained.

We compared the densities of vegetation among years using Kruskal-Wallis tests due to violations of normality and equal variance assumptions. We used the Dunn Method to conduct pairwise comparisons. All statistical analyses were completed in SAS JMP 12.0 (SAS Institute, Cary, North Carolina, U.S.A.). These data were used to evaluate and quantify how vegetation types within these wetlands had actually responded to the changing water-level conditions.

Results

Following the increase in water-level of 0.91 m, the shoreline of each of the 25 wetlands moved shoreward, as derived from our GIS analysis. The extent of this movement varied inversely with the mean slope of the wetland, such that wetlands with the shallowest contours were associated with the greatest amount of horizontal movement. The quadratic function (r^2=0.76; p<0.0001) was better than the linear function (not shown; r^2=0.64; p<.0001) in terms of statistical fit through the data, and it is heuristically more appealing because the effect of slope should taper off at some point (Figure 1.2a). When we compared the quadratic curve against one derived using our mathematical model (Figure 1.2b), we found that the model significantly over-estimated the extent of movement across the entire range of slope values (Wilcoxon signed rank; p<.0001). The model predicted a mean movement of 67.71 m across all 25 sites, compared with a mean movement of 49.14 m by the GIS analysis. Due to considerable variation in wetland geomorphology between sites, the range between minimum and maximum change in shoreline movement within each wetland varied from 26.72 m for Treasure Bay with more uniform contours, and 333.35 m for Potato Island, that has much more dynamic shoreline contours (Table 1.2). In most cases, the range between minimum and maximum values for each site was between 75 – 100 m.

Total suitable growth area of wetlands when all vegetation types were considered was highest during Period 2 (Figure 1.3; paired t-test; p<.0001). Additionally, emergent, floating, and SAV suitable growth area significantly increased with water level (paired t-test; p=0.0215, p=0.0062, p<.0001, respectively). However, suitable growth area for the meadow vegetation was significantly reduced in the high water-level scenario (paired t-test; p=0.0391).

In general, the littoral vegetation zone typically hosting SAV and floating vegetation, was the most resilient to the water level change, where on average, an overlapping zone of 4.20 ha was maintained within wetlands. Emergent and floating vegetation also exhibited overlap, maintaining an average of 2.64 and 1.85 ha of suitable area, respectively. The meadow vegetation was the least resilient to the water level change, with no growth area overlap occurring between scenarios. Consequently, all meadow vegetation would have to reestablish exclusively from seed banks and existing propagules (Figure 1.4; Appendix).

Based on the depth range data for each plant surveyed in the field during Period 1, we determined that the meadow vegetation was typically located within 0 – 30 cm of water, the emergent vegetation between 0 – 130 cm, floating vegetation between 15 – 135 cm, while the SAV typically grew between 15 – 180 cm of water. Following the increase in water levels in Period 2, however, meadow vegetation occurred from 0 – 130 cm, emergent vegetation from 30 – 160 cm, floating vegetation from 40 – 200 cm, and SAV typically from 40 – 200 cm of water. We found the same range in depth data for emergent and SAV between time periods, but for floating and meadow, the range increased by 40 and 100 cm, respectively between Periods 1 and 2 (Table 1.3).

We were able to compare the stem densities of emergent, floating and SAV in three wetlands that had been sampled during 1981, 2012 and 2016. The mean summer (June-August) water level in 1981 was similar to that in 2016 (176.81 m ASL vs 176.84

m ASL). When we compared stem densities of emergent vegetation among all years, we found a significant difference (Kruskal-Wallis; $p<0.0001$); in pairwise comparisons, stem densities did not differ significantly between 1981 and 2012, but both were significantly lower than those in 2016 (Dunn joint ranking; $p <0.005$ for all). We also found significant differences for floating vegetation among years (Kruskal-Wallis test; $p=0.0065$); in 2016, floating densities were significantly higher than those in 1981 (Dunn joint ranking; $p=0.005$), but there were no significant differences between 2016 and 2012, and 2012 and 1981 ($p>0.05$ in both cases). Despite differences in densities for emergent and floating vegetation, we found no significant differences in SAV stem densities among years (Kruskal-Wallis test; $p=0.0787$) (Figure 1.5).

In addition to changes in density, we also found changes in species richness and composition. Water-level fluctuations during the early 1980s could be described as being "normal", whereas those in 2012 had followed 13 years of sustained lows, with very little interannual fluctuations. Although there were similar numbers of emergent and floating taxa, more SAV taxa were found in 2012 than in 1981 (15 and 9, respectively). During the high water-level scenario in 2016, we found the greatest number of plant species in the three vegetation types. Whereas the dominant SAV taxa in 1981 and 2012 were *Chara* sp. and *Najas flexilis*, *Elodea canadensis* and *Bidens beckii* became the most dominant SAV species in 2016 (Table 1.4).

Discussion

Slope is a key morphologic factor that dictates how the shoreline, and consequently all vegetation types will horizontally move within a wetland under different water levels. In wetlands with shallow slopes, an increase in water level will result in greater horizontal movement required for the vegetation community to travel to re-establish. On the other hand, wetlands with steep slopes will be less affected by changes in water levels, and the relationship between the amount of horizontal movement and slope decreases in a non-linear fashion. The farther that vegetation must migrate to become re-established, the longer the lag time required for the wetland system to regain a typical state (Keddy and Constabel 1986; Keddy and Reznicek 1986). Most past vegetation modelling efforts incorporate the use of a DEM and topographic characteristics; however, slope was not directly included in the analyses (Sklar et al. 1985; Wilcox and Xie 2007; Hebb et al. 2013). Given that the effect of slope on wetland vegetation has been recognized in the literature, it is surprising how infrequently slope is reported in wetland studies compared with the specific migration patterns of the wetland plants themselves (e.g. Wilcox and Xie 2007; Hebb et al. 2013).

As shown with our modeling efforts, simple mathematical models incorporating slope and water-level change can provide fairly accurate predictions of how far different types of vegetation would migrate in order to become reestablished. Though an overestimate, the horizontal movement predicted from the model was still within the standard error range for the mean of each of the calculated horizontal movements. Since longer distances would take proportionately longer time for wetland plant species to migrate following a large increase or decrease in water level (Keddy and Constabel 1986; Keddy and Reznicek 1986), wetlands with steeper slopes will be more resilient to large water-level fluctuations. In the absence of vegetation field data, or aerial image or satellite data, wetland basin slope may be a way to assess resilience to extreme water-level changes in coastal habitats. However, it is important to recognize that once the basin slope becomes too steep, it becomes less suitable for wetland vegetation growth as the thin soils will be easily washed away and most plants will not be able to root (Duarte and Kalff 1986). As such, a trade-off exists between a gentle basin slope that is highly susceptible to water level fluctuations, and steeper basin slopes that are less suitable for macrophyte growth. Other factors that impact and limit aquatic macrophyte growth and assemblage such as fetch and nutrient availability (Schneider et al. 2015) must also be considered.

Mathematical modeling to explain different aspects of change within coastal wetlands is beneficial, but accounting for continuous variations at the site level poses a challenge (Costanza and Sklar 1985; Sklar et al. 1985; Frederico et al. 2007). Our simplified model to predict the horizontal movement of vegetation communities as a function of slope for wetlands with consistent contours generally over-predicted the amount of shoreline movement within a wetland; however, the predictions are within the standard error of the mean calculated for each of the horizontal movement values calculated using a GIS. When more accurate bathymetric data are available, we can produce more accurate estimates of horizontal movement, and include in our mathematical model, thresholds in the slope where no amount of water-level change could cause further change in vegetation zonation. Nevertheless, there may be slope constraints on vegetation growth at that point (Duarte and Kalff 1986). A more thorough investigation of wetland hydrogeomorphology and constraints to vegetation growth would be beneficial to accurately determine resilience of coastal wetland vegetation habitat on water-level disturbances.

In a situation where a GIS cannot be used to calculate wetland basin slope or horizontal movement, our simple mathematical model can provide a fairly accurate estimate. Predictions of monthly and annual water levels are publicly available through the Canadian Hydrographic Service, so the expected change in water level, can be easily obtained. Site slope can also be calculated *in situ*, without the use of a GIS. Slope can be calculated by creating a linear transect perpendicular to shore and measuring the distance from the shoreline to the 1m-depth contour. From this, we can solve for the slope of the wetland basin. This technique has been successfully applied in previous wetland habitat studies in the Georgian Bay area (Leblanc 2015). This method can be used by environmental project managers to help plan seasonal sampling by knowing in advance how much vegetation shift they might encounter. This is highly beneficial, because large water-level changes have the potential to negatively impact fish community sampling protocols and sampling efforts may need to be adapted depending on the amount of annual water-level change (Chapter 2).

Each wetland, especially in the Georgian Bay area, has unique aspects to their geomorphology (Wei et al. 2004; Fracz and Chow-Fraser 2013). This variation is most evident in this study when shoreline movement was calculated using a GIS. The large range in horizontal migration within each wetland implies that there are additional micro-habitats within each wetland that are more resilient to water-level change than in others. The unique morphology and behaviour of each wetland confirms that attempts to classify and model wetland changes at a broad regional scale is inappropriate (Keough et al. 1999; Wei et al. 2004). This again, reinforces the need for high-quality, fine-scale bathymetric information when making accurate models of any spatial habitat change in coastal wetlands.

We found that in general, the potential total wetland area was larger under the high water-level scenario. This is also true for the potential area available to emergent, floating and submerged aquatic vegetation; however, this was not the case for meadow vegetation, for which the available growth area became compressed under the high-water level scenario. Logically, this result makes sense when 'typical' wetland basin shape is considered. In general, the wetland basin becomes steeper towards the shoreline (Weller, unpub. data). We know from the slope modelling that the horizontal shift that accompanies change in water level is lessened when slopes are steeper. Since the meadow vegetation is already predominantly located in this area with the steeper slope, its movement will be restricted relative to the other vegetation types. This effect is also illustrated in the available area for the other vegetation types as the emergent vegetation that is located closest to shore has a much smaller expansion in area than the SAV, which is located in the deepest portions of the wetland. The expansion of the emergent vegetation is limited by the steeper shoreline slopes, but these slopes have little effect on the SAV expanding in the area of the wetland exhibiting gentler slopes. Wilcox and Xie

(2007) used a different approach to study the effect of increased water level in Lake Ontario on wet meadow habitat, and observed compression of meadow habitat as well, but their approach required more extensive and intensive field surveys over five consecutive years.

Although the quantity of habitat is likely to increase with water level, we cannot assume that the quality has also improved. In general, larger areas with SAV habitat would benefit littoral species such as fish (Jude and Pappas 1992). However, the compression of the meadow vegetation under high water-level scenarios could be detrimental to marsh bird species that require this specific habitat (Riffell et al. 2001). Since wetlands with different vegetation communities are used by many species, managers should consider the amount of suitable habitat for target species rather than total wetland area, and often, conditions that benefit fish species may not benefit avian species.

At this point, we have used DEMs to calculate only the area of the fundamental niche that is suitable to host the different vegetation types under high and low water-level scenarios. Using imagery collected from satellite or UAVs (unmanned aerial vehicle), actual changes in surface area of vegetation can be tracked (Chow-Fraser et al. 2016). Once more imagery becomes available, the data can be used to validate the extent of the meadow, emergent, and floating vegetation predicted in this study.

In the current case of an abrupt increase in water level following a prolonged state of low water levels, the extent to which the total area occupied by each vegetation type is maintained between water levels becomes important for vegetation distribution reestablishment. In the current high water-level transition, the meadow vegetation is the least resilient to the increase in water level, and the SAV the most resilient. Given that the growth areas for the meadow vegetation for Period 1 and Period 2 do not overlap spatially, new meadow species must grow from seed (Keddy and Resnicek 1986) and requiring a longer time for re-establishment. But even for vegetation that have overlapping distributions in different water-level regimes, they must still compete with existing vegetation in the new growth area. For example, SAV species would not be able to become established immediately in areas that had previously been dominated by emergent or meadow vegetation; after several years of inundation, however, the remnant vegetation would die off and wash away with wave action or ice forces. If water levels fluctuate annually, there should be sufficient disturbance each year to prevent dominance by any vegetation type (Odland and del Moral 2002; Gathman et al.2005).

The depth profiles of plants we surveyed in the 2015 and 2016 seasons do not reflect those sampled in 2004-2009. In theory, these depth ranges should not change between these periods since they should reflect where vegetation types prefer to grow. Instead of becoming compressed, the meadow vegetation depth range increased by 100 cm; also unexpected was the decrease in the SAV depth range by 5 cm, while the emergent depth range remained the same. However, while the range of the emergent vegetation remained the same, it was shifted 30 cm deeper (0-130 cm to 30 to 160 cm). This indicates that the emergent vegetation has so far been unable to expand in either direction to re-establish where it is ‘typically’ located. The only observation that was expected was the expansion of the floating vegetation to fill new available habitat. The increase in meadow vegetation can be attributed to the inundation of 10 years of prior vegetation growth. Much of the meadow vegetation included large shrubs (*Myrica gale*) and trees (*Pinus strobus*), which had migrated into the meadow growth area from the upland forest. It would probably take a minimum of three years of high water levels to kill off the dense meadow vegetation and allow the other vegetation types to expand into their "preferred" water depths. Midwood and Chow-Fraser (2012) suggested that wetlands in Georgian Bay would take 5-years to adjust to new water-level regimes; however, woody shrubs and pine trees are probably more difficult to "remove" compared with herbaceous species, and it may take a much longer time for aquatic vegetation to re-establish at the "new" 1m-depth zone. The actual size of the SAV range is likely larger than what we have quantified because we could not survey water deeper than 180 cm using the present protocol. We know from sonar surveys (data not shown) that we conducted in these wetlands that SAV regularly occurred below 2 m depth during Period 2.

In addition to changes plant depth, we also found changes in stem density and community composition over the time period between 1981 and 2016. The high densities of emergent vegetation in 2016 was probably a reflection of past homogenization of the high marsh during Period 1. Lack of water-level disturbance to the system usually allows emergent and meadow vegetation to proliferate (Gathman et al. 2005; Midwood and Chow-Fraser 2012). Even after being flooded for 3 years, the emergent and meadow vegetation were very dense. During this transition period from low to high water level, the depth that had previously been suitable for "true" emergent vegetation species

(*Cyperacae* sp., *Shoenoplectus* sp.) was also populated by upland shrub species (*Myrica gale*) and species adapted to living in very shallow water (*Acorus calamus*) (Newmaster et al. 1997). The few species of floating vegetation present in these wetlands do not tend to change between scenarios.

The stem densities of SAV did not change significantly between time period in part because of the high variability in our quadrats. For instance, *Schoeneplectus subterminalis* (water bulrush) has fine hair-like strands that varied greatly among quadrats. The species richness had increased in 2016, and there had been a shift in species dominance. During re-flooding in Period 2, propagules (seeds and turions) tend to be deposited along the shoreline (Odland and del Moral 2002; Gathman et al. 2005), and this may have augmented the diversity. The shift in dominance from *Chara* sp. to *Najas flexilis*, *Elodea canadensis* and *Bidens beckii* reflect the wide environmental tolerance of the latter group and their ability to colonize quickly (Newmaster et al. 1997). The overall increase in species richness in all vegetation categories following re-flooding is a positive sign that the complexity of these coastal wetland habitats can quickly rebound following a period of homogenization (Midwood and Chow-Fraser 2012).

A limitation of this study was the lack of LIDAR (Light Detection and Ranging) data. LIDAR sensors can penetrate the water's surface to map detailed underwater topographic surface, making them the tool of choice for coastal habitat mapping projects (Chust et al. 2008; Owers et al. 2016). Unfortunately, such data are not currently available for the Georgian Bay shoreline, and this lack of fine-scale bathymetric information makes accurate hydrogeomorphic modelling extremely challenging. The data sources we used to create DEMs in this study are the best currently available, but we acknowledge that some errors may have been incurred due to the resolution and accuracy of our input data sources.

The ability to accurately model and predict the fate of wetland habitats in the uncertainty of future water level and climate scenarios is highly beneficial. In the absence of vegetation and bathymetric data, slope can be estimated from field surveys and the expected horizontal shift can be calculated. This simplified model could be an asset to resource managers to crudely identify coastal wetlands with steeper slopes, that are more resilient to extreme water level changes. Understanding the lag time required for vegetation community distributions to reestablish following a prolonged period of suppressed water level fluctuation is critical for determining suitable habitat in a period of transition. More accurate bathymetric data acquisition for the Georgian Bay coastal area will be crucial for increasing the accuracy of any coastal wetland habitat modelling efforts in the future.

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Table 1.1. General descriptions of four wetland vegetation types and associated fish communities. Sources: 1) Chadde 2012 2) Scott and Crossman 1998 3) Gathman et al. 2005 4) Midwood and Chow-Fraser 2012 5) Jude and Pappas 1992 6) Newmaster et al. 1997 7) Leblanc et al. 2014 8) Kapuscinski and Farrell 2014.

Vegetation type	Description	Family of taxa present		Usefulness as fish habitat
Meadow	Contains sedges and grasses that are tolerant of some flooding but inundation for more than a few years [1]	*Calamagrostis* *Carex* *Dulichium* *Iris* *Juncus* *Agalinis*	*Cicuta* *Eupatorium* *Hypericum* *Impatiens* *Lobelia*	Negligible, except for occasional use of flooded backwater habitat by spawning pike [2]
Emergent	Vegetation that protrudes from the water's surface. Stem densities of emergent taxa are known to increase in the absence of water-level fluctuations [3,4]	*Typha* *Schoenplectus* *Eleocharis* *Eriocaulon* *Polygonum* *Pontederia*	*Sagittaria* *Sparganium* *Zizania* *Acorus* *Equisetum* *Phragmites*	Typical feeding and refugia habitat for flat-bodied *centrarchids* [2,5]
Floating	Includes any macrophytes with broad leaves that float on the water surface [6]	*Nymphaea* *Nuphar* *Brasenia*	*Potamogeton* *Sparganium* *Lemna*	Provide cover for many small *cyprinid* species and young-of-the-year (YOY) predatory fish species and protect them from avian and visual fish predators [5]
Submerged aquatic vegetation (SAV)	Consist of different growth forms such as basal growing species that cover the substrate bottom and canopy-forming species that extend throughout the water column [6]	*Bidens* *Callitriche* *Chara* *Ceterophyllum* *Elodea* *Isoetes* *Myriophyllum* *Najas*	*Nitella* *Potamogeton* *Utricularia* *Schoenoplectus* *Stuckenia* *Vallisneria* *Lobelia* *Neobeckia*	Preferred habitat for most fish species. Used particularly by *cyprinid* species, and YOY and age 1 northern pike and muskellunge [7, 8]

Table 1.2. Summary of geographic coordinates for the 25 study sites and corresponding statistics for GIS analyses to determine horizontal movement (m) of the shoreline following a 0.91 m increase in water level. Data are sorted by ascending order of mean slope calculated for each wetland.

Wetland Name	Longitude	Latitude	Mean slope	Minimum Distance Moved	Maximum Distance Moved	Range in distances moved	Mean±SE Distance Moved
Potato Island	-79.75529	44.79308	0.43	45.00	378.35	333.35	171.01±30.88
Quarry Island	-79.80881	44.83402	0.63	29.35	156.74	127.39	86.84±13.39
Oak Bay	-79.73702	44.79851	0.77	71.42	237.85	166.43	137.19±16.99
Musky Bay	-79.78038	44.81232	1.01	26.22	107.42	81.20	68.07±8.31
West Bay	-80.30484	45.42089	1.18	13.65	110.71	97.06	58.08±9.65
Corbman Bay	-80.34130	45.40855	1.19	13.54	72.00	58.46	39.39±6.79
Roberts Island	-79.83146	44.85474	1.20	16.71	155.53	138.82	69.91±15.45
Green Island	-79.74713	44.78556	1.23	19.09	178.85	159.76	90.69±14.99
Cormican Bay	-80.30908	45.40793	1.24	12.05	103.32	91.27	32.83±8.24
Davids Bay	-80.00217	45.04654	1.31	17.59	110.05	92.46	49.55±9.82
Alexander Bay	-80.00529	45.05483	1.47	14.53	58.84	44.31	29.55±4.69
Key River 1	-80.67640	45.88622	1.50	12.37	87.73	75.36	33.21±7.64
Tadenac 2	-80.69268	45.88499	1.59	16.85	60.43	43.58	30.59±4.07
Picnic Island	-79.82016	44.85952	1.63	13.17	51.76	38.59	29.50±3.62
Tadenac 1	-79.98638	45.04021	1.74	9.85	49.35	39.50	18.80±3.66
Sturgeon Bay Central	-80.41455	45.61430	1.83	16.24	86.93	70.69	46.68±6.99
Key River 3	-80.69268	45.88499	1.90	6.88	60.71	53.83	31.24±6.45
Moreau Bay	-79.94438	45.01316	1.93	11.15	135.92	124.77	39.79±11.85
Coffin Rock	-79.98738	45.04776	1.94	12.26	114.38	102.12	31.81±10.02
Charles Inlet	-80.56582	45.64726	2.00	9.38	73.72	64.34	29.20±6.36
Treasure Bay	-79.85870	44.87108	2.34	6.63	33.39	26.76	15.30±2.67
North Bay 1	-79.79414	44.89778	2.65	5.06	122.7	117.64	33.34±13.80
Miners Creek	-79.94669	45.06204	2.88	2.56	35.81	33.25	17.19±3.48
Ojibway Bay	-79.85779	44.88744	3.40	8.08	63.08	55.00	24.49±5.18
North Bay 5	-79.80274	44.88213	3.57	5.46	39.88	34.42	14.15±3.20

Table 1.3. Range in water depths (cm) where different vegetation types were found based on field surveys associated with two different water-level scenarios. Scenario 1 was a sustained low water-level period sampled between 2004 and 2011. Scenario 2 was a transition period following a 0.9 m increase in water level preceded by 5+ years of sustained low levels.

Vegetation type	**Period 1**	**Period 2**	**Difference in depth range between Periods**
Meadow	0-30	0-130	Increased by 100 cm
Emergent	0-130	30-160	No change
Floating	15-135	40-200	Increased by 40 cm
SAV	15-180	40-200	Decreased by 5 cm

Table 1.4. Dominant plant taxa present in three wetlands during three different water-level scenarios. *denotes number of families shared between the emergent and floating vegetation types, not number of species due to incomplete data availability.

Water-level scenario	Habitat type	Number of Taxa	Most Abundant Species
Fluctuating water levels (1981)	Emergent	8*	*Cyperacae* sp.
	Floating	8*	No Data
	SAV	9	*Najas flexilis* *Chara* sp. *Potamogeton gramineus*
10 y of sustained low water levels (2012)	Emergent	6	*Schoenoplectus acutus* *Typha* sp. *Schoenoplectus americanus*
	Floating	2	*Nymphaea odorata* *Potamogeton natans*
	SAV	15	*Chara* sp. *Najas flexilis* *Vallisneria americana*
3 y following abrupt increase in water level (2016)	Emergent	15	*Eriocaulon aquaticum* *Myrica gale* *Acorus calamus*
	Floating	4	*Nymphaea odorata* *Nuphar variegatum* *Potamogeton natans*
	SAV	18	*Elodea canadensis* *Bidens beckii* *Potamogeton gramineus*

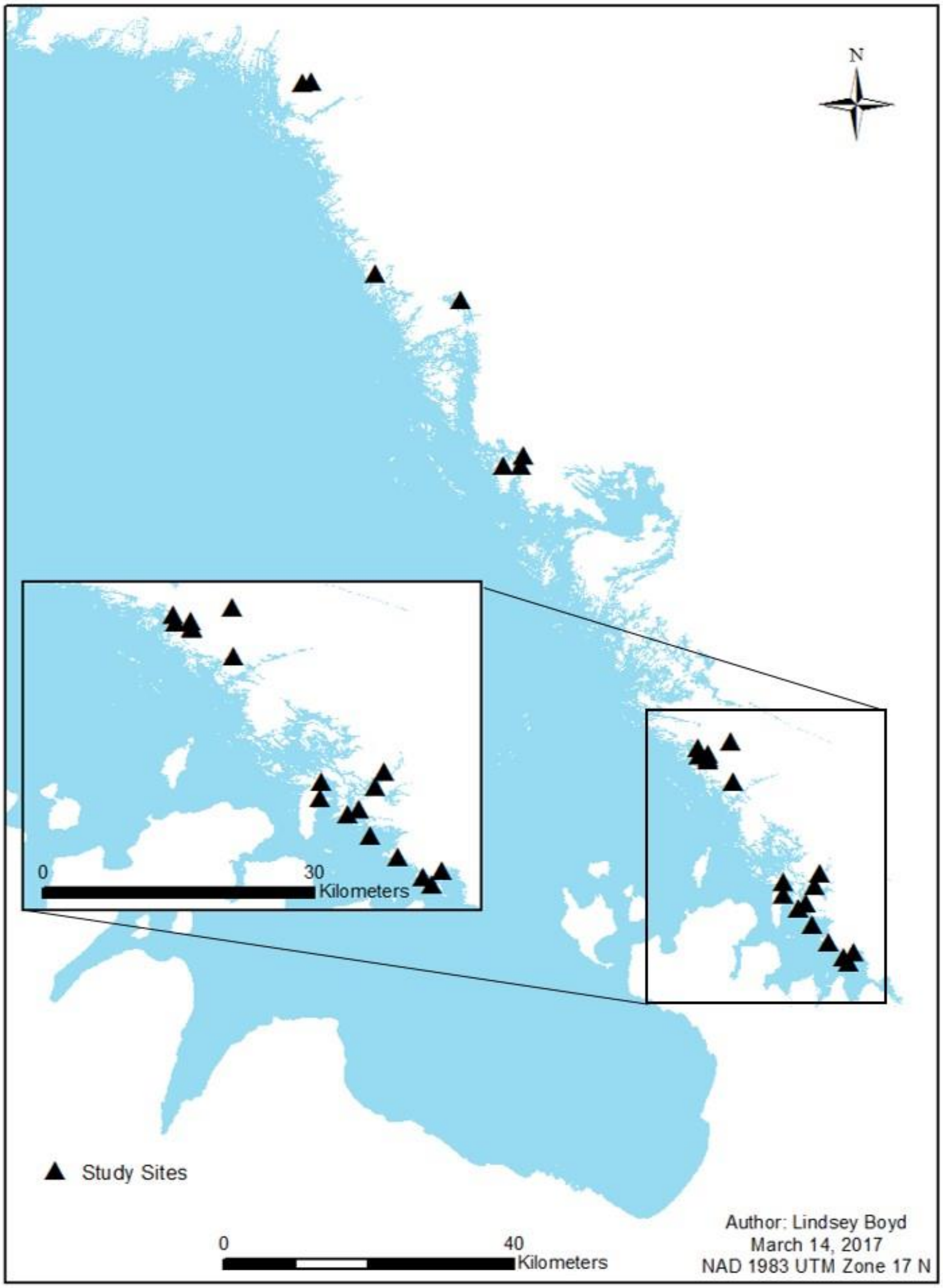

Figure 1.1. A site map indicating the locations of all 25 coastal wetlands where bathymetric sampling occurred in Georgian Bay, Lake Huron. The inset shows an expanded view of sites in the Wah Wah Taysee and Severn Sound region.

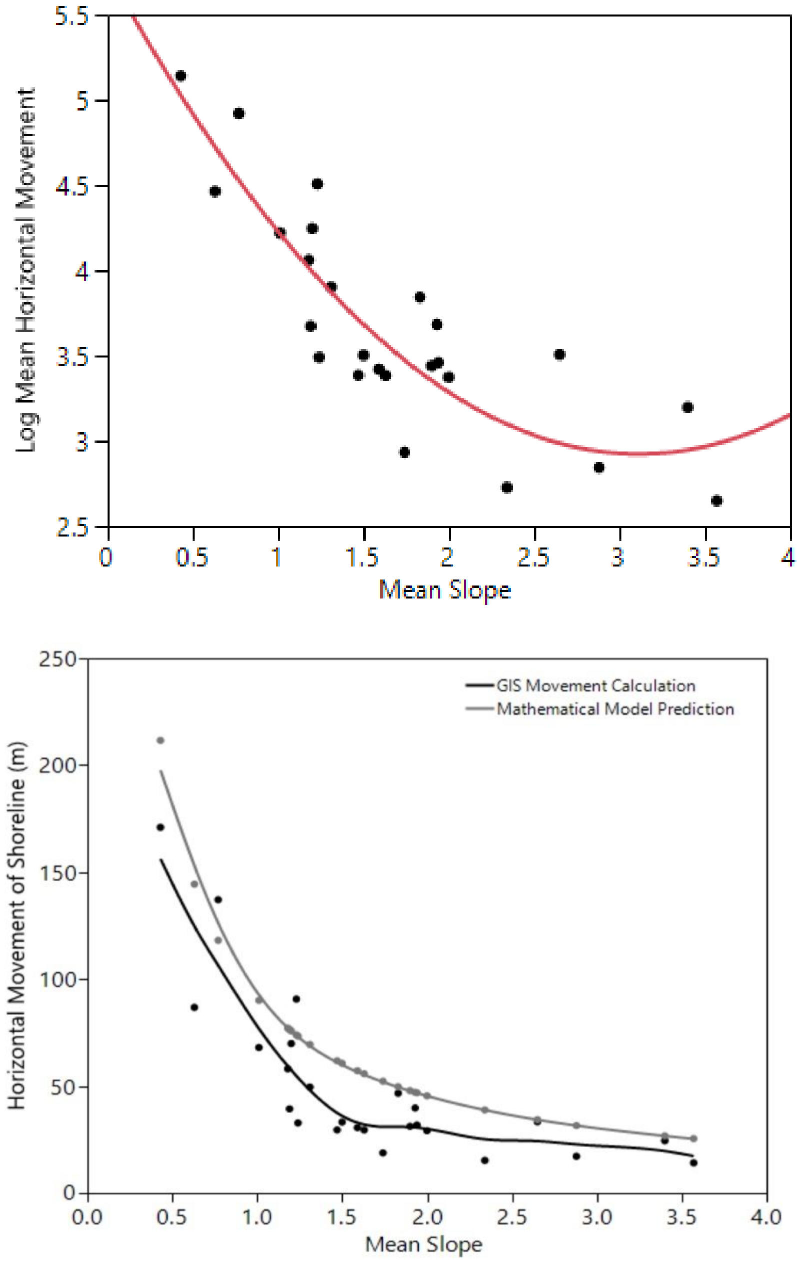

Figure 1.2 a) Quadratic relationship between log mean horizontal movement (m) of shoreline following a 0.91 m increase in water level and mean slope for all 25 wetlands.

b) Horizontal movement (m) of the shoreline as a function of the shoreline slope following a 0.91 m increase in water level. The grey line denotes values predicted by a mathematical model assuming that wetlands have uniform basin contours.

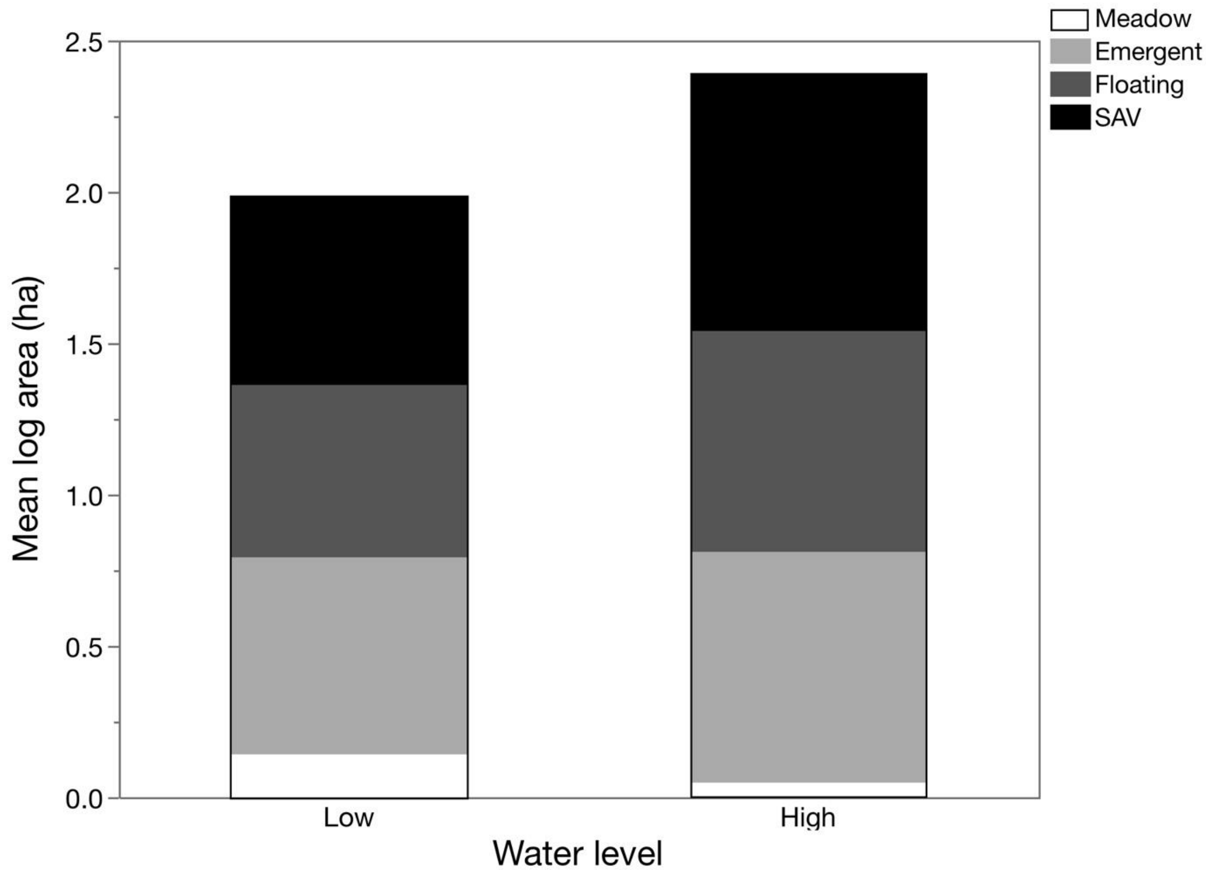

Figure 1.3. A plot of the log mean suitable area (ha) for growth of four vegetation types under two different water-level scenarios. The low-water level is 175.93 m ASL and high-water level is 176.84 m ASL.

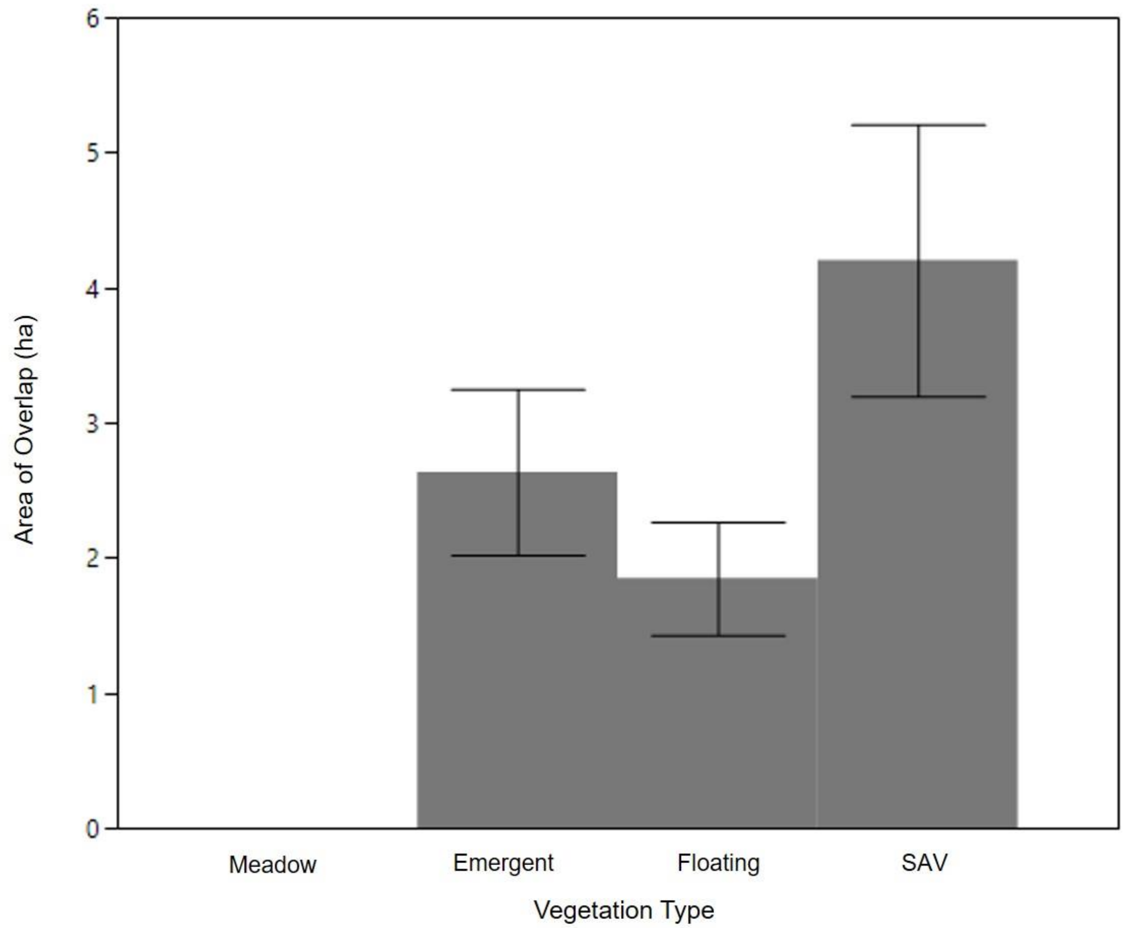

Figure 1.4. Mean±SE of the overlap (ha) between available area suitable for growth for four vegetation types during high and low water-level conditions. The low-water level is 175.84 m ASL and the high-water level is 176.93 m ASL.

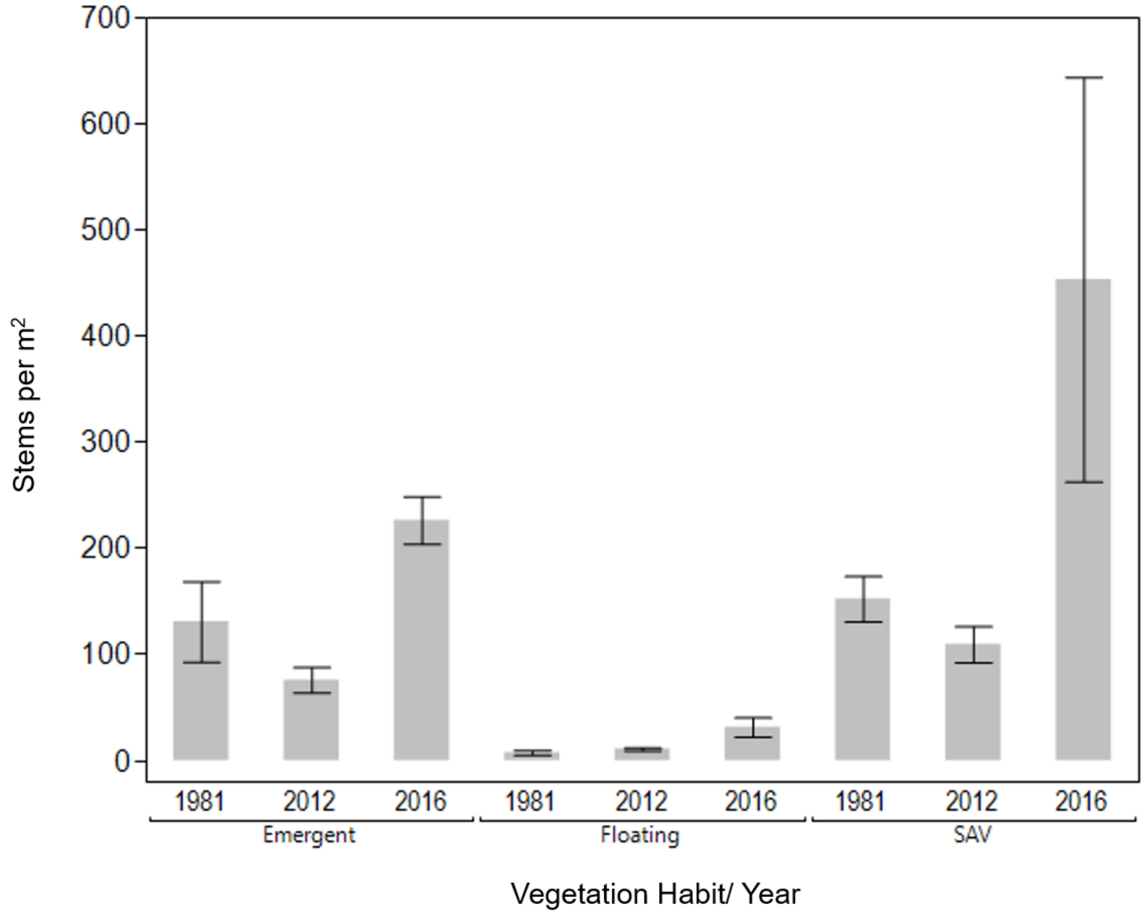

Figure 1.5. Stem densities per m^2 for emergent, floating, and SAV in three different years, each being associated with a different water-level scenario. Water levels for the 5 years preceding 1981 had fluctuated according to historic patterns, water levels for 10 years preceding 2012 had been very low, whereas those from 2014 to 2016 had risen ~0.9 m above that of 2013.

Chapter 2:

Influence of water-level induced changes on performance of ecological indices in coastal marshes of Georgian Bay: a cautionary note

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Abstract

We used three indices (Water Quality Index (WQI), Wetland Macrophyte Index (WMI), and Wetland Fish Index (WFI)) to track changes in wetland health of 40 coastal wetlands in eastern Georgian Bay over two time periods, the first between 2003 and 2008 (Period 1) and again between 2014 and 2016 (Period 2). During Period 1, water levels of Lake Michigan-Huron had fallen 0.4 m below the long-term mean (176.42 m ASL), whereas during Period 2, water levels had risen to sit 0.5 m above the long-term mean (a maximum difference of 0.9 m between periods). In each wetland, data for index calculations were collected according to standardized guidelines. WMI scores remained unchanged between periods ($\bar{x}$ WMI score of 3.59 vs 3.63 for Periods 1 and 2, respectively), indicating that wetlands were in the “excellent” category. By comparison, WQI scores significantly increased between periods ($\bar{x}$ WQI score of 1.35 vs 1.83 for Periods 1 and 2, respectively), but still indicated “very good” conditions. WFI scores, however, dropped significantly between time periods, ($\bar{x}$ WFI score of 3.66 to 3.49 from Period 1 to Period 2, respectively) even though they still indicated “very good” conditions. Given the strong association among wetland fish, macrophyte community, and water quality, the significant decrease in WFI scores was unexpected, and we attribute this to the inability of fyke nets to catch all wetland fish during Period 2 when nets had to be placed among shrubs/emergent/meadow vegetation to satisfy the prescribed depth requirement. We suggest that during periods of extreme flux in water levels, use of fyke nets alone is inappropriate and insufficient for assessing the health of wetlands that are sensitive to water-level changes.

Introduction

Freshwater coastal wetlands exist throughout the globe, and are a special class of wetlands that are hydrologically connected to large lakes. In the Laurentian Great Lakes, coastal wetlands have been defined as those located between permanent aquatic and permanent upland environments along the shores of the Great Lakes that are affected by large-lake processes such as waves, wind tides, and long-term water-level fluctuations (Maynard and Wilcox 1997). In addition to providing habitat for many bird, amphibian, and reptile species, they provide critical habitat to many fish taxa (Jude and Pappas 1992). Approximately 80% of the fish species within the Great Lakes (160 of 200) must use a wetland at some point in their life cycle (Scott and Crossman 1998; Wei et al. 2004). These fish use coastal wetlands as spawning grounds, nursery habitat, foraging arena, and refugia from predators. In particular, young-of-the-year (YOY) of top predators such as muskellunge and northern pike, seek shelter among the canopy of floating and submerged aquatic vegetation (SAV) and feed on the abundant invertebrates found there (Jude and Pappas 1992; Leblanc et al. 2014).

Over 2 million ha of wetlands were widely distributed in southern Ontario prior to European settlement (c. 1800); by 2002, only 28% of the pre-settlement wetlands remained, with up to 85% of the wetlands being lost in large urban centers (Ducks Unlimited Canada 2010). Many of the remaining wetlands in southern Ontario have been degraded (Cvetkovic and Chow-Fraser 2011) primarily because of urban and agricultural development (Chow-Fraser 2006; Seilheimer and Chow-Fraser 2007). This is not the case for coastal wetlands in northern Ontario, especially in Lake Superior and Georgian

Bay (the eastern arm of Lake Huron that lies entirely within Canada), partly because bedrock underlying these regions is granitic rock that is nutrient-poor due to their resistance to weathering, compared with the soft sedimentary rocks (limestone and shale) in the southern lakes that are nutrient-rich and suitable for farming (Chow-Fraser and Albert 1999). Eastern Georgian Bay can be further distinguished from Lake Superior by having a large number of islands and rock indentations along its shoreline that host thousands of small coastal marshes (Midwood et al. 2012). These factors contribute to Georgian Bay being considered a region with the highest concentration of pristine wetlands in the Great Lakes basin (Cvetkovic and Chow-Fraser 2011).

Besides being affected by human disturbance, Great Lakes coastal marshes can be heavily influenced by water levels of the lake to which they are hydrologically connected. Water-level disturbance such as loss of extreme low water levels following lake-level regulation in Lake Ontario and the St. Lawrence River is thought to have allowed hybrid cattail (*Typha* x *glauca)* to expand and become very dense at the expense of wet meadow habitat (Wilcox et al. 2008). Greater than five years of sustained low water levels in wetlands of eastern Georgian Bay have also led to a significant reduction in plant and fish species richness and habitat homogenization (Midwood and Chow-Fraser 2012). Therefore, loss of natural fluctuations in water levels that characterize coastal Great Lakes is the main driver affecting habitat quality rather than absolute high or low water levels *per se*.

Of all five Great Lakes, Lakes Michigan-Huron (and therefore Georgian Bay) have the largest range in fluctuations (Bishop 1990). Historically, water levels fluctuated a maximum of 4 m around the long-term mean of 176.42 m ASL (Bishop 1990). There are also hourly or daily fluctuations that can be caused by storm events, monthly variability caused by seasonal hydrologic cycle, and long-term annual or decadal fluctuations that are thought to be due to climate variability (Bishop 1990; Lenters 2001). These variations have made it extremely difficult to determine any definitive recurring pattern, although the most common statistical and spectral analyses have identified cycles of ~8, ~12, and ~30 years (Liu 1970; Cohn and Robinson 1976; Hanrahan et al. 2009). Global circulation models, however, are forecasting lower than usual water levels for the Laurentian Great Lakes over the next 50 years (Erwin 2009; Angel and Kunkel 2010) and these are expected to have a profound but unpredictable influence on the ecology of these coastal wetlands.

In 1999, water levels in Lake Michigan-Huron diverged from historical patterns and began to persist continuously at very low levels below the long-term mean for 14 years (Figure 2.1). Just as abruptly, between 2014 and 2016, water levels rebounded by approximately 0.9 m to a level just above the long-term mean. These drastic changes caused the shoreline and 1-m depth contour in all wetlands to shift towards upland vegetation, and in some wetlands by over 50 meters, flooding meadow and emergent marshes and inundating 10-y old pine trees (Table 1.2). Whereas emergent and floating vegetation dominated water depths below 1.3 m during the period of sustained low water levels, they were now submerged in water depths of up to 1.5 to 2 m during 2014 and 2016 (Table 1.3). Such a shift in vegetation zonation is expected to have an attending effect on the distribution of the fish in the wetland since many taxa are normally positively associated with submersed aquatic vegetation (Cvetkovic et al. 2010), which would have shifted to water depths beyond 1.5 to 2.0 m, depending on the basin slope of the wetland (Figure 2.2).

The abrupt rise in water levels of almost a meter in a region that is known to have minimal anthropogenic impacts offered a unique opportunity to study how water-level disturbance affects the health of coastal wetlands without confounding anthropogenic effects. Prior to the 1980s when the first biotic indicators were developed, the health of many freshwater systems was determined primarily from water-chemistry information (Karr 1981; Meador et al. 1993). Since then, ease of implementation and effectiveness at gauging wetland health have spurred the development of many indices of biotic integrity (IBI; Burton et al. 1999; Simon et al. 2001; Uzarski et al. 2004, 2005; Grabas et al. 2012) and biotic indicators (Seilheimer and Chow-Fraser 2006; Croft and Chow-Fraser 2007). Generally, biotic indices require only identification of certain taxonomic groups within the wetland surveyed once during the summer season. In almost all cases, indices have been developed for health assessment to support decisions on restoration priorities and to track effectiveness of restoration efforts. No study has yet been carried out to explicitly test how their performance would be affected simultaneously by water-level disturbance in a natural setting.

Cvetkovic and Chow-Fraser (2011) used three indices based on water-quality data (Water Quality Index (WQI); Chow-Fraser 2006), taxonomic composition of wetland macrophytes (Wetland Macrophyte Index (WMI); Croft and Chow-Fraser 2007), and the abundance or taxonomic composition of wetland fish communities (Wetland Fish Index

(WFI); Seilheimer and Chow-Fraser 2007) to rank the relative influence of human activities on the health of 200 coastal wetlands throughout the five Great Lakes. Since these easily interpretable indices are the only published ecological indices that have been developed with and validated for coastal wetlands within eastern Georgian Bay (Croft and Chow-Fraser 2007; Seilheimer and Chow-Fraser 2007; Cvetkovic and Chow-Fraser 2011), they are the most appropriate published indices to use in this area. The WMI and WFI, which are ecological response indicators, are directly related to WQI scores (Seilheimer et al. 2009), while WQI scores, indicators of anthropogenic stress, are significantly and negatively related to road density (DeCatanzaro et al. 2009), a documented indicator of human disturbance (Danz et al. 2005; Campbell 2017).

The primary goal of this study is to use the WMI, WFI, and WQI to determine how the health of coastal wetlands in eastern Georgian Bay has changed over the past decade, during a period of demonstrated water-level disturbance. Given that ecological indices were developed specifically to detect the level of human-induced degradation, and the fact that negligible impact of anthropogenic activities (as indicated by road density) has occurred in this region over the past decade (DeCatanzaro et al. 2009; Campbell 2017), we do not expect index scores to be significantly downgraded. As a secondary goal, we want to compare how the three indices perform in a period of rapid transition between water-level regimes (increase of almost a meter between 2013 and 2014), when the vegetation zone and fish community shifted spatially away from the shoreline. Continued monitoring of these pristine wetlands is imperative for early detection of areas that may be at risk of becoming degraded due to human development.

Additionally, understanding how indicators of wetland health may become confounded or biased by a change in water-level regime, will help to inform resource managers how to adapt sampling strategies during water-level transition periods.

Methods

Site Selection

All sampling took place in coastal wetlands of Georgian Bay. Wetlands were sampled during two different time periods: Period 1 (2003-2008), when mean annual water levels fluctuated at most 0.33 m, and Period 2 (2014-2016), when water levels rebounded by almost a meter to levels above the long-term mean. All sites were located along the eastern shoreline of Georgian Bay and sampled between mid-June and September (Figure 2.3). In total, 40 wetlands ranging in size from 95 to 1.5 hectares were sampled for at least one of water quality, macrophyte community, and fish community for use in indicator calculations (Table 2.2). Due to inclement weather, we were unable to collect data to calculate scores for all three indices in every wetland.

Index Sampling Protocols

All wetlands were sampled per published protocols for WQI (Chow-Fraser 2006), WMI (Croft and Chow-Fraser 2007; 2009), and WFI (Seilheimer and Chow-Fraser 2006; 2007). Sampling for water-quality parameters occurred in the approximate center of each wetland, in open water, away from floating and submerged vegetation. In total, 12 primary nutrient and physical parameters were measured for use in the WQI calculation. These included pH, temperature (TEMP), conductivity (COND), turbidity (TURB), total ammonia nitrogen (TAN), total nitrate nitrogen (TNN), total nitrogen (TN), soluble reactive phosphorus (SRP), total phosphorus (TP), chlorophyll-α (CHL), total suspended solids (TSS), and total inorganic suspended solids (TISS). Physicochemical parameters were sampled at mid-depth (between 50 and 75 cm, wetland dependent) with a YSI 6920 V2-2 Multi-Parameter Sonde equipped with a YSI Pro 1030 display (Yellow Springs, OH, USA). The YSI sonde was used to collect *in situ*, COND, TEMP, and pH. We also collected water samples and measured TURB *in vitro* using a Hach field turbidimeter. A 1-L van Dorn sampler was used to collect water from mid-depth for further processing for nutrients in the field, and analysis in the lab. In the field, we analyzed samples of TAN using a portable Hach colorimeter. Water was filtered for SRP, CHL, TSS, and TISS; water samples and filters were frozen for further analysis in the lab. Water samples for TP, TNN and TN were frozen immediately after they were collected in the field. All frozen samples were transported back to McMaster University and analyzed within 4 months of collection. Complete field and laboratory sample processing methods can be found in Chow-Fraser (2006).

The wetland macrophyte community was surveyed according to the stratified method of Croft and Chow-Fraser (2007;2009). Ten to fifteen quadrats (0.75 x 0.75 m) were placed in different vegetation zones (e.g. emergent, SAV, and floating) except in the upland meadow zone; hence, any meadow species reported had been found inundated within the aquatic portion of the wetland. Whenever possible, plants situated within the quadrat or touching the edge of the quadrat were identified to species and recorded. Sampling ceased once no new species were found in two consecutive quadrats. In shallower areas, we waded between quadrats, and in deeper areas we canoed and collected samples of submerged plants using a rake.

Fish communities were sampled with three paired fyke nets, according to published protocols (Seilheimer and Chow-Fraser 2006;2007). Nets were set parallel to shore, ideally in a combination of vegetation types (floating, SAV, and emergent); there were two sets of large fyke nets (12.7 mm bar mesh, 4.5 m length, 1.5 m X 1.25 m front opening) and one set of small nets (4.8 mm bar mesh, 2.1 m length, 0.5 m X 1.25 m front opening), and each pair of nets were connected with a 7-m lead and 2.5-m wings. The large and small fyke nets were set at the 1-m and 0.5 m depth contour, respectively. Nets were left to soak for 20-24 hours, after which all fish were collected, identified, counted, measured, and returned to the wetland. The rapid increase in water level made it difficult to sample the fish communities with fyke nets, because the 1-m depth contour (where the nets are to be set) had shifted more than a few meters in some cases. As per the standardized protocol, the nets were to be set in a mixture of wetland vegetation types, including emergent, floating and submerged vegetation (Figure 2.2).

In three wetlands, we also attempted to survey the fish community using a seine net (6.35 mm bar mesh, 18.29 m X 1.22 m net, 1.22 m X 1.22 m bag). Seining was attempted by having one person stand at the shoreline with the net, while the other end of the net was pulled straight out into the wetland, and pulled in a half-moon shape back to the shore. All fish were identified to species, counted, measured, and returned to the wetland. In two of the three cases, we were unable to complete seine net sampling due to extremely dense meadow and emergent vegetation between the 50-cm depth contour and the shoreline. Failure was due to the weighted bottom of the net rolling up and releasing the fish prematurely. The one wetland in which seining was successful had a steeper slope and therefore less dense emergent and meadow vegetation than in most of the sites sampled. To save time and resources, the seining sampling was abandoned early in the season and sampling effort was reallocated to more extensive vegetation and bathymetric surveys.

For this study, we calculated the WFI scores using presence rather than abundance data because the former eliminates biases formed by over-exaggerated abundances that are a product of different species spawning at different times throughout the summer (Seilheimer and Chow-Fraser 2006;2007). For both the WMI and WFI, values for each species corresponding to their niche breadth (U value) and tolerance to degradation (T value) were applied. U and T values were developed with a partial Canonical Correspondence Analysis (pCCA) of taxonomic and water-quality information. The pCCA allowed for species-specific treatment in the index calculation. Calculated WMI and WFI scores can range from 1-5, where high values indicate high quality and health. Most scores tend to be closer to 3, because many common, hearty species have U and T values of 3 (Seilheimer et al. 2009).

Index Calculation

For each index, scores were calculated for each individual wetland for the two scenarios according to Chow-Fraser's (2006) equation:

Eq. 1:

$$
\begin{aligned}
\text{WQI score} = 10.0239684 & - (0.3154965 \times \log \text{TURB}) \\
& - (0.3656606 \times \log \text{TSS}) \\
& - (0.3554498 \times \log \text{ISS}) \\
& - (0.3760789 \times \log \text{TP}) \\
& - (0.1876029 \times \log \text{SRP}) \\
& - (0.0732574 \times \log \text{TAN}) \\
& - (0.2016657 \times \log \text{TNN}) \\
& - (0.2276255 \times \log \text{TN}) \\
& - (0.5711395 \times \log \text{COND}) \\
& - (1.1659027 \times \log \text{TEMP}) \\
& - (4.3562126 \times \log \text{pH}) \\
& - (0.2287166 \times \log \text{CHL})
\end{aligned}
$$

The formula for calculating the WMI and WFI is as follows:

Eq. 2:

$$\text{WMI or WFI} = \left(\frac{\sum_{i=1}^{n} Y_i T_i U_i}{\sum_{i=1}^{n} Y_i T_i} \right)$$

where: Y_i is the abundance – if the species is present, this value is one; if absent, the value is 0; T_i assumes a value from 1 to 3 according to the species *i* niche breadth, and U_i assumes a value from 1 to 5, depending on the tolerance of species *i* to water-quality degradation. T_i and U_i values are supplied by Croft and Chow-Fraser (2007) for WMI species and by Seilheimer and Chow-Fraser (2007) for WFI species.

Statistical Analyses

For each wetland sampled, we calculated a score for Period 1 using data from a published study (Cvetkovic and Chow-Fraser 2011), and another for Period 2 using data collected between 2014 and 2016. Differences in means between periods for all three indices were normally distributed, so paired t-tests were used to determine any differences in mean index scores. We performed all analyses in SAS JMP 12.0 (SAS Institute, Cary, North Carolina, U.S.A.).

Interpretation of Index Scores

The WQI includes information from 12 commonly measured water-quality variables and was based on information from 110 sites across the five Great Lakes, including Georgian Bay. WQI varies from -3 to +3, where positive scores denote good health and negative scores denote degraded conditions (Chow-Fraser 2006). For ease of understanding and use by others, numerical scores from all three indices correspond to an assigned category ranging from “excellent” to “highly degraded” (Table 2.1). Croft and Chow-Fraser (2007) suggested that a WQI score of zero (indicating no impairment) should correspond with a WMI score of 2.5 while that for WFI should be a score of 3.25 (Seilheimer and Chow-Fraser 2007; Cvetkovic 2008). Accordingly, the WQI, WMI, and WFI do not have the same categorical thresholds (Table 2.1).

In 2009, Seilheimer et al. (2009) compared the scores from the three indices and found that regressions of both WMI and WFI scores with WQI scores were significant. Thus, the results of the two biotic indices accurately reflected the level of degradation present in a wetland based on water quality information. The largest difference between the WMI and WFI scores was that a larger range of WMI values were present over the same range of WQI values. This is because in degraded wetlands, only a few submerged vegetation species are present, and yet a large number of moderately pollution-tolerant fish species persist.

Results

There was a significant increase (n=39 wetlands; paired t-test; $p<.0001$) in mean WQI scores between Period 1 and 2 (Table 2.3). Despite this increase, all wetlands were assessed as being in “very good” condition (scores between 1.0 and 2.0). For the 26 wetlands that we surveyed for macrophytes, we found no significant difference (paired t-test; $p=0.4441$) in mean WMI scores between time periods (Table 2.3). The status of the wetland remained “excellent” as indicated by the WMI (score >3.5). By comparison, the WFI scores in 35 wetlands decreased significantly (paired t-test; $p=0.0020$) between periods, although the status of wetlands remained in the “very good” category (scores between 3.25 and 3.75) (Table 2.3).

We grouped the macrophyte data by time period to compare the taxonomic composition of the macrophyte communities between periods for all 26 wetlands. We found 55 taxa present in both periods, 24 taxa that were unique in Period 1 and 16 that were only found during Period 2. We also noted that SAV species were more abundant during Period 1(10 vs 0), and meadow species were more abundant in Period 2 (8 vs 11) (Table 2.4).

A comparison of fish captured in the fyke nets during the two Periods showed that 7 small fusiform fish species were captured during Period 1 but not in Period 2 (Table 2.5). Besides the round goby, most of the fish captured in Period 2 that had not been captured in the earlier Period were single occurrences. In one wetland where seining was completed in addition to fyke netting, we caught five additional small, fusiform species that had not been captured in the fyke nets (Table 2.6). Using only the fish caught with fyke nets to calculate the WFI score for this wetland would have generated a score of 3.69 (very good), whereas including the fish caught with the seine in addition to those in the fyke net would have yielded a higher WFI score of 3.82 (excellent).

The vegetation zones associated with different water depths changed considerably between Period 1 and Period 2 (Figure 2.4). During Period 1, very few meadow species were encountered at the edge or below the water surface of the wetland, and they were confined to water < 40 cm deep. Under the high water-level conditions in Period 2, many more meadow species were found, and these occurred at depths ranging from the shoreline to 150 cm, most of which were located between depths of 40 and 80 cm. Emergent vegetation, which had been confined to depths below 150 cm during Period 1 were routinely found up to 180 cm in Period 2. During Period 1, most of the emergent taxa were growing in the 20 to 50 cm depth zone whereas during Period 2, they were growing in the 50 to 120 cm depth zone. We saw a similar shift in depth of floating vegetation; maximum depths of 150 cm in Period 1 extended to 200 m in Period 2. Likewise, SAV occupied depths from 20 to 200 cm in Period 1 but in Period 2, the maximum depth shifted to 300 cm. Throughout both periods, the depth zone with the most SAV occurred at 50 to 100 cm, whereas floating vegetation occurred mostly in the 80 to 100 cm depth range. Therefore, as water levels increased almost a meter between time periods, we observed a concomitant shift in the maximum depth of emergent, floating and submerged vegetation to about a meter deeper.

Discussion

Overall, we noted that the status of coastal wetlands along the Georgian Bay shoreline are still in very good health. When sorted by region, wetlands located in the more remote, northern locations had higher WQI scores (sometimes indicative of "excellent" conditions) than did wetlands located in Severn Sound, which had only recently been de-listed as an Area of Concern (Environment and Climate Change Canada 2014). We speculate that the smaller sample size of northern sites compared with the eastern and southern sites (n= 5, 21, 13, respectively) constrained analyses to elucidate regional patterns. Until more data become available, it is best to avoid separating analyses according to region and to determine temporal trends of wetland quality for Georgian Bay as a whole. Even though WQI scores improved significantly overall, there was no accompanying shift from the "very good" category to "excellent" category. It is also important to note that WQI scores increased across the entire region such that lower scores were still associated with the more anthropogenically impacted wetlands in Severn Sound (Campbell 2017) compared with those in the remote regions of eastern and northern Georgian Bay.

The higher WQI scores in Period 2 relative to Period 1 was unexpected since we have no evidence that anthropogenic disturbances had lessened between time periods. Since higher WQI scores indicate lower concentrations of nutrients and sediments in wetlands, we attribute this to a diluting effect of increased water levels between Period 1 and Period 2; previous studies have shown that TSS decreased significantly when water levels increased in a coastal wetland of Lake Ontario (Chow-Fraser 1999; Chow-Fraser

2005). The decrease in nutrients and CHL may also be related to the significant decline in phosphorus and chlorophyll α levels in the offshore zones of Lake Huron as a result of improved treatment of sewage effluent associated with the Severn Sound Remedial Action Plan in the 1990s (Croft and Chow-Fraser 2007; Dove and Chapra 2015). Water clarity, which is largely associated with reductions in TP, CHL, TSS and TURB, has also increased significantly in Lake Huron since 1970 (Binding et al. 2015). Thus, the increase in WQI scores may reflect both a reduction in suspended solids related to increased water levels, as well as lake wide reduction of nutrients in Lake Huron rather than decreased human disturbance in the wetlands.

The WMI is the only indicator for which mean scores did not significantly change between the two water-level scenarios. This can be attributed to how macrophyte sampling is normally executed, and how the index score is calculated. The protocol requires the surveyor to sample in a stratified fashion in all vegetation zones within the inundated area of the wetland, irrespective of the water level at the time of sampling (Croft and Chow-Fraser 2006). Although a total of 94 plant taxa were found in the vegetation surveys for use in the WMI calculation (Table 1.1). There were few meadow species accounted for in this sampling (only those occurring around the water's edge), since the index was developed to assess the health of only the aquatic portion of the wetland. When water levels in wetlands suddenly increased by almost a meter during Period 2, many of the meadow species were inundated and became part of the aquatic wetland area. Consequently, many more upland wildflower, grass, shrub, and tree species were found as part of the macrophyte surveys during this latter period (Table 2.4). By contrast, there were fewer taxa of SAV identified during Period 2 than in Period 1 because the plants had been found in water that was too deep to collect with a rake. The lack of sensitivity of the WMI to differences in water level makes it a better biotic index than the WFI for assessing impacts of human disturbance, and confirms the observation by Croft and Chow-Fraser (2009) that it is not dependent on completeness of plant surveys.

Normally, seasonal and annual water-level fluctuations add disturbance to the wetland habitat, allowing for a wider range of plants to establish (Keddy and Reznicek 1986; Keough et al. 1999). When wetlands experience a period of sustained low water levels, however, germination of emergent and meadow species that can tolerate drier conditions persist and create very dense stands of vegetation (Keddy and Constabel 1986; Gathman et al. 2005; Midwood and Chow-Fraser 2012). When water levels then suddenly increase by ~0.9 m (such as in 2014), the dense emergent community would be covered with 1 m of water in most areas. This transition of a meadow marsh becoming primarily aquatic habitat in a coastal wetland has been illustrated graphically based on field observations (Figure 2.5a). Based on the 2015 and 2016 field observations, densities of emergent and meadow species became drastically reduced following the recent 2-3 years of persistent inundation (Gathman et al. 2005). It is very likely that after a minimum lag time of 4-5 years of moderately high water levels, terrestrial vegetation would again dominate the wet meadow zone, while aquatic vegetation would dominate from the shoreline lakeward (Midwood and Chow-Fraser 2012; Figure 2.5b).

Since coastal wetlands in this study are primarily protected from wind, the type of vegetation found in each zone is largely a function of water depth rather than of wind and wave action that are known to structure wetland zonation of exposed wetlands (Keddy and Constabel 1986; Keddy and Reznicek 1986). The change in water depth for any wetland is primarily a function of the water level and the wetland basin slope (Chapter 1). Since coastal marshes situated in the more southern and eastern portion of Georgian Bay tend to have more shallow sloping contours than those in the northern region, they experienced a greater spatial shift in the vegetation zonations when water levels increased 0.91 m (Chapter 1). In coastal wetlands with steeper sloping bathymetry, however, the horizontal shift in macrophyte zones would be small or negligible (Figure 2.6). This is best demonstrated by our northern Georgian Bay wetlands, where we found a much more modest shift in vegetation zones (personal observations 2016). Wetland bathymetry therefore appears to be a major determinant of wetland resilience to water-level disturbance, and wetlands must be evaluated on a case-by-case basis to determine the lag time required for the macrophyte zonation to return to a 'typical' state.

The prominent change in depth zonation of coastal marshes in Period 2 resulted in a large spatial shift of dominant vegetation types (Figure 2.5a). These changes altered the available habitat for fish species within the wetland because inundated upland plants (pine trees, shrubs and meadow taxa) are not ideal fish habitat (Jude and Pappas 1992; Leblanc et al. 2014). In addition, they also influenced where fyke nets (used to survey fish for calculation of WFI scores) could be placed. A small shift in wetland plant zonation (associated with steeper slopes) would have minimal effect on placement of fyke nets whereas a large shift (associated with shallower slopes) could have a profound effect because the sampling protocol and physical net structure requires fyke nets to be set in 1 m of water. When water levels increased during Period 2, they caused most vegetation types to shift on average 0.5 m deeper towards the lake (Figure 2.4). Hence, the 1-m depth contour hosted primarily flooded emergent and meadow vegetation, with very little representation of the SAV community.

We speculate that these differences in plant communities where the fyke nets were set led to changes in fish communities in that area, resulting in different species being captured, and ultimately lowering WFI scores in Period 2. Flat bodied, *centrarchid* fish are generally found amongst more dense, emergent and floating vegetation, whereas predatory fish in their nursery stage such as northern pike, muskellunge and largemouth bass prefer less dense, canopy forming SAV (MacRae and Jackson 2001; Cvetkovic et al. 2010; Leblanc et al. 2014). The small fusiform fish such as many *cyprinid* species are also partial to canopy forming SAV (Rozas and Odum 1988; Kapuscinski and Farrell 2014). It is these small fusiform fish that are missing from the catch during Period 2 (Table 2.5). It is also these fusiform fish that have higher associated U values (niche breadth), that are used in the WFI calculation (Table 2.6). The presence of one or two of these species with high U values, would have easily increased the overall WFI score.

The overall decrease in WFI scores between Period 1 and Period 2 is problematic. Akin to the WMI, the WFI was developed as an indicator of community response to water-quality impairment that could be attributed to human activities (Seilheimer and Chow-Fraser 2006). We have no reason to believe that water quality in our wetlands had worsened between time periods; therefore, the lower WFI scores in Period 2 are likely because some of the fish associated with high U values had moved to deeper water habitats that could no longer be sampled by fyke nets. We found support for this hypothesis when we sampled the same wetland using both fyke nets and a seine net. The seine net was actively pulled through the SAV zone, approximately 1.5-2.0 m deep. Many additional fish species were caught with the seine net that had not been caught in the fyke net. Since these fish also had high U values, their absence in the fyke nets probably resulted in lower WFI scores. Combining data from both the seine and fyke nets would have yielded a higher WFI score that would have corresponded with the higher WQI score.

We do not recommend using seine nets alone to survey fish communities in coastal wetlands because many fish species that had been caught in the fyke nets were missing from the seine net (Table 2.6). As well, seine netting could not be used in most cases because the dense emergent and meadow species prevented the seine net from being pulled over the substrate without a large percentage of the fish community escaping. A possible solution to this sampling issue is to employ electrofishing instead of fyke netting. Chow-Fraser et al. (2006) found no significant differences in the number of species or functional taxa caught by fyke nets versus electrofishing (Chow-Fraser et al. 2006). It should be noted however, that electrofishing generally catches larger fish than do fyke nets, so there is a risk that the small, fusiform fish would still have been missing in our samples (Bohlin et al. 1989; Chow-Fraser et al. 2006).

Other investigators who have developed biotic indicators to assess health of coastal wetlands also acknowledge the requirement for indicators to be free of the influence of water-level fluctuations (Wilcox et al. 2002; Uzarski et al. 2004; Bhagat et al. 2007). To account for fyke net biases caused by macrophyte zone shifting due to water level fluctuations, Uzarski et al. (2005) developed two fish-based IBIs for use in Great Lakes wetlands, one for sites dominated by *Typha* species and another by *Schoenoplectus* species. These IBIs cannot be applied to eastern and northern Georgian Bay, however, because our wetlands are not principally dominated by either *Typha* or *Schoenoplectus* species due to their unique granitic base and thin soils. Additionally, the water-level increase from a period of sustained low to high was sufficiently large to cause extreme horizontal macrophyte zone shifts that made it impossible to sample within all vegetation zones in some wetlands (e.g. fyke nets would have to be set below 1.5 m to include SAV). This may explain why Uzarski et al. (2016) assessed wetlands in Georgian Bay to be in moderate to poor health, even though WFI scores of wetlands located in similar locations in this study showed them to be in "very good" health. Uzarski et al. (2016) acknowledged that more work was needed to calibrate fish-based indices for different vegetation zonations and coastal wetland habitats throughout the Great Lakes basin. This again, reinforces the susceptibility of fish-based indicators to surrounding dominant macrophyte taxa.

Despite being costly and requiring a high level of expertise to execute, the WQI was a reliable stress indicator of human disturbance even though there was a large variation in water levels between time periods. The WMI was better than the WFI as indicator of community response to anthropogenic stress, and was unaffected by large shifts in vegetation zones related to the interaction between increased water level and wetland slope. By comparison, the WFI (and by implication any fish-based indicator) may have been heavily influenced by the depth zone of the vegetation available for fyke-net placement. Since the location of different wetland fish species is dependent on macrophyte type and location (Uzarski et al. 2005; Cvetkovic et al. 2010) fyke nets should not be the only gear used to survey the fish communities when wetlands are transitioning between extremes in water levels. If we are correct, then WFI scores should continue to increase in the next few years, as long as the pattern of water-level fluctuations in Lake Huron return to that prior to 1999. Given that more recent climate models forecast modest decreases in Lake Huron water levels as well as an increase in mean seasonal range (Angel and Kunkel 2010; MacKay and Seglenieks 2013), there is an urgent need to better understand how such hydrological changes will impact both the vegetation and fish communities in these otherwise undisturbed coastal marshes.

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Table 2.1. A summary of categories associated with Water Quality, Wetland Macrophyte, and Wetland Fish Index scores.

Category	**Water Quality Index**	**Wetland Macrophyte Index**	**Wetland Fish Index**
Excellent	> 2	> 3.50	> 3.75
Very good	1 to 2	3.0 to 3.5	3.25 to 3.75
Good	0 to 1	2.5 to 3.0	2.75 to 3.25
Moderately degraded	0 to -1	2.0 to 2.5	2.25 to 2.75
Very degraded	-1 to -2	1.5 to 2.0	1.75 to 2.25
Highly degraded	< -2	< 1.5	< 1.75

Table 2.2. All locations and size of coastal wetlands sampled in Georgian Bay, Lake Huron. Sites were sampled during Period 1 under sustained low water level conditions (2003-2008) and during Period 2 under a 0.9 m increase in water level (2014-2016). Index scores for water quality, wetland macrophytes, and wetland fish were calculated where data were available. Wetland size was retrieved from coastal wetland GIS layers created by Midwood *et al.* (2012).

Site Name	Latitude	Longitude	Size (ha)	Water Quality Index Score		Wetland Macrophyte Index Score		Wetland Fish Index Score	
				Period 1	**Period 2**	**Period 1**	**Period 2**	**Period 1**	**Period 2**
Beaverstone Bay	45.98434	-81.14676	39.32	1.71	1.45	3.41	3.38	3.23	3.27
Charles Inlet	45.64780	-80.56700	73.14	1.18	1.88	3.07	3.79	3.92	3.35
Corbman Bay	45.40908	-80.34057	10.63	1.24	2.03	3.54	3.89	3.93	3.75
Cormican Bay	45.40612	-80.30580	18.34	0.78	2.02	3.67	3.68	3.76	3.39
David's Bay	45.04749	-80.00146	0.33	1.92	1.76	3.54	3.57	4.09	3.39
Deer Island	45.95986	-81.22015	22.16	2.18	1.52	3.68	3.60	3.86	3.39
Francis Point	45.41368	-80.33111	48.29	1.33	1.60	3.78	3.78	4.03	3.57
Ganyon Bay	44.91986	-79.81654	1.88	1.43	1.95	3.76	3.73	3.64	3.67
Green Island	44.78555	-79.74696	4.87	1.38	1.66	3.25	3.67	3.78	3.61
Herman's Bay	45.08630	-79.99711	2.86	1.59	1.67	3.69	3.85	3.35	3.62
Hog Bay	44.73453	-79.80366	42.44	0.72	1.65			3.70	3.61
Hole in the Wall	45.52152	-80.43932	14.16	1.78	1.64			3.72	3.43
Inukshuk Bay	45.55675	-80.38544	5.10	1.87	1.95	3.60	3.71		
Isle of Pines	45.59848	-80.51876	6.13	1.84	1.57				
Key River	45.88670	-80.67501	6.27	0.66	2.06			3.04	3.23
Key River 3	45.88474	-80.69299	5.63	1.74	1.70	3.74	3.61	3.41	3.17
Lake St. Patrick	44.97769	-79.92874	3.27	1.99	1.53	3.78	3.34	3.44	3.38
Lost Channel	45.59346	-80.51059	1.40	1.38	1.86	3.63	3.70		
Matchedash Bay	44.75950	-79.68871	873.81	-0.20	1.20			3.75	3.66
Miner's Creek	45.06153	-79.94913	5.08	1.77	2.44	3.56	3.24	3.65	3.67
Moose Bay	45.07210	-80.04958	9.84	1.85	1.61				
Moreau Bay	45.01478	-79.94396	21.97	1.17	1.79	3.69	3.69	4.03	3.26
Musky Bay	44.81257	-79.77965	17.74	1.23	2.15	2.97	3.63	3.75	3.67
Ni Bay	45.50924	-80.45599	7.32	1.05	1.68			4.13	3.45

North Bay	44.87010	-79.79499	1.90	0.43	1.83	3.50	3.62	3.68	3.58
Oak Bay	44.79784	-79.73788	50.02	1.12	2.01			3.63	3.93
Ojibway Bay	44.88783	-79.85595	1.61	1.56	1.78	3.65	3.60	3.69	3.62
Potato Island	44.78992	-79.74136	84.12	0.99	1.75			3.44	3.88
Quarry Island	44.83248	-79.80728	28.28	1.11	1.45	3.52	3.46	3.79	3.35
Robert's Bay	44.85537	-79.82870	7.43	1.44	2.03	3.65	3.46	3.76	3.66
Sand Channel	45.61295	-80.51121	6.35	1.58	2.06				
Shadow Bay	45.94717	-80.73500	11.59					3.69	3.19
Shawanaga River	45.56231	-80.36548	1.59	2.29	2.45	3.79	3.50	3.39	3.42
Sturgeon Bay Central	45.59772	-80.40207	5.68	0.52	1.91			3.22	3.72
Sturgeon Bay South	44.75624	-79.74341	95.67	0.69	1.97			3.91	3.51
Sugar John	45.93838	-81.17158	12.04	1.77	1.88	3.69	3.82	3.60	3.26
Tadenac A	45.03562	-79.99263	4.37	1.56	2.22			3.50	3.17
Treasure Bay	44.87320	-79.85739	44.62	1.55	1.66	3.58	3.51	3.80	3.59
Waterfall Bay	45.56257	-80.34536	1.22	2.14	2.48	4.00	3.85	3.44	3.27
West Bay	45.42165	-80.30710	58.09	0.57	1.64	3.50	3.63	3.43	3.46

Table 2.3. Mean WQI, WMI, and WFI scores between Period 1 (2003 – 2008) and Period 2 (2014-2016).

		Period 1			**Period 2**			
	Number of Sites	**Mean**	**Std. Error**	**Category**	**Mean**	**Std. Error**	**Category**	**p-value**
WQI	39	**1.35**	± 0.087	Very Good	**1.83**	± 0.045	Very Good	<.0001
WMI	26	**3.59**	± 0.043	Excellent	**3.63**	± 0.032	Excellent	0.4441
WFI	35	**3.66**	± 0.043	Very Good	**3.49**	± 0.034	Very Good	0.0020

Table 2.4. Wetland macrophytes identified during two sampling periods. Period 1 (2004 – 2008) was during sustained low water levels, and Period 2 (2015/2016) experienced an 0.9m increase in water level.

Sampled in both periods		**Sampled only during Period 1**	**Sampled only during Period 2**
High Marsh	**Submerged**	**High Marsh**	**High Marsh**
Calamagrostis canadensis	*Bidens beckii*	*Agalinis paupercula*	*Alnus viridis*
Carex sp.	*Callitriche sp.*	*Cicuta maculate*	*Carex aquatilis*
Dulichium arundinaceum	*Chara sp.*	*Eupatorium maculatum*	*Carex rostrate*
Iris versicolor	*Ceterophyllum demersum*	*Eupatorium perforliatum*	*Carex utriculata*
Juncus canadensis	*Elodea canadensis*	*Hypericum perforatum*	*Grass sp.*
Emergent	*Freshwater sponge sp.*	*Impatiens capensis*	*Lysimachia terrestris*
Eleocharis smallii	*Isoetes sp*	*Juncus effuses*	*Moss sp.*
Eriocaulon aquaticum	*Myriophyllum sibiricum*	*Lobelia cardinalis*	*Myrica gale*
Polygonum sp.	*Myriophyllum spicatum*	**Emergent**	*Pinus strobus*
Pontederia cordata	*Najas flexilis*	*Eleocharis acicularis*	*Rhynchospora alba*
Sagittaria graminea	*Nitella sp.*	*Sagittaria cuneate*	*Schoenoplectus cyperinus*
Sagittaria latifolia	*Potamogeton amplifolius*	*Sparganium androcladum*	**Emergent**
Schoenoplectus acutus	*Potamogeton crispus*	*Sparganium angustifolium*	*Acorus calamus*
Schoenoplectus americanus	*Potamogeton epiphydrus*	**Floating**	*Equisetum fluviatile*
Schoenoplectus validus	*Potamogeton friesii*	*Lemna trisulca*	*Phragmites australis americanus*
Sparganium eurycarpum	*Potamogeton gramineus*	*Nuphar pumila*	*Phragmites australis australis*
Typha angustifolia	*Potamogeton pusillus*	**Submerged**	**Floating**
Typha latifolia	*Potamogeton richardsonii*	*Lobelia dortmanna*	*Lemna minor*
Typha X glauca	*Potamogeton robbinsii*	*Myriophyllum alterniflorum*	**Submerged**
Zizania paulstris	*Potamogeton spirillus*	*Myriophyllum heterophyllum*	
Floating	*Potamogeton zosteriformis*	*Myrophyllum tenellum*	
Brasenia schreberi	*Schoenoplectus subterminalis*	*Neobeckia aquatica*	
Nuphar variegate	*Stuckenia pectinate*	*Potamogeton foliosus*	
Nymphaea odorata	*Utricularia gibba*	*Potamogeton illinoensis*	
Nymphoides cordata	*Utricularia intermedia*	*Potamogeton vaseyi*	
Potamogeton natans	*Utricularia minor*	*Utricularia cornuta*	
Sparganium fluctuans	*Utricularia purpurea*	*Utricularia geminiscapa*	
	Utricularia vulgaris		
	Vallisneria americana		

Table 2.5. A comparison of fish taxa captured in fyke nets in eastern Georgian Bay during two sampling periods. (Period 1: 2004 to 2008; Period 2: 2014 to 2016 inclusive). *denote a single occurrence.

Captured in both periods	Captured only in Period 1	Captured only in Period 2
Banded killifish	Fathead minnow	Central mudminnow
Black crappie	Iowa darter	Channel catfish*
Blackchin shiner	Logperch	Grass pickerel*
Blacknose shiner	Mimic shiner	Round goby
Bluegill	Muskellunge*	
Bluntnose minnow	Northern redbelly dace	
Bowfin	Rainbow darter*	
Brook silverside	Rosy-faced shiner	
Brook stickleback	Sand shiner	
Brown bullhead	Shorthead redhorse	
Common carp	Walleye*	
Common shiner		
Emerald shiner		
Golden shiner		
Johnny darter		
Largemouth bass		
Longear sunfish		
Longnose gar		
Northern pike		
Pumpkinseed		
Rockbass		
Smallmouth bass		
Spotfin shiner		
Tadpole madtom		
White crappie		
White sucker		
Yellow perch		
27 taxa	**11 taxa**	**4 taxa**

Table 2.6. Fyke net and seine net fish catch abundance from Sturgeon Channel wetland (45.59747, -80.40984) in August 2015. U and T values for use in the presence/absence (PA) calculation of the WFI are included.

Fish Species	**(U, T) values PA**	**Fyke Net Catch (Abundance)**	**Seine Net Catch (Abundance)**
Yellow perch	(3, 2)	3	1
Brown bullhead	(3, 1)	24	
Rockbass	(4, 1)	12	
Pumpkinseed	(3, 2)	34	
Longear sunfish	(4, 3)	9	
Largemouth bass	(3, 2)	30	1
Northern pike	(4, 2)	1	
Emerald shiner	(3, 2)		30
Bluntnose minnow	(3, 1)		7
Blackchin shiner	(5, 3)		11
Common shiner	(4, 3)		1
Unknown *Cyprinid* sp.	(2, 1)		3

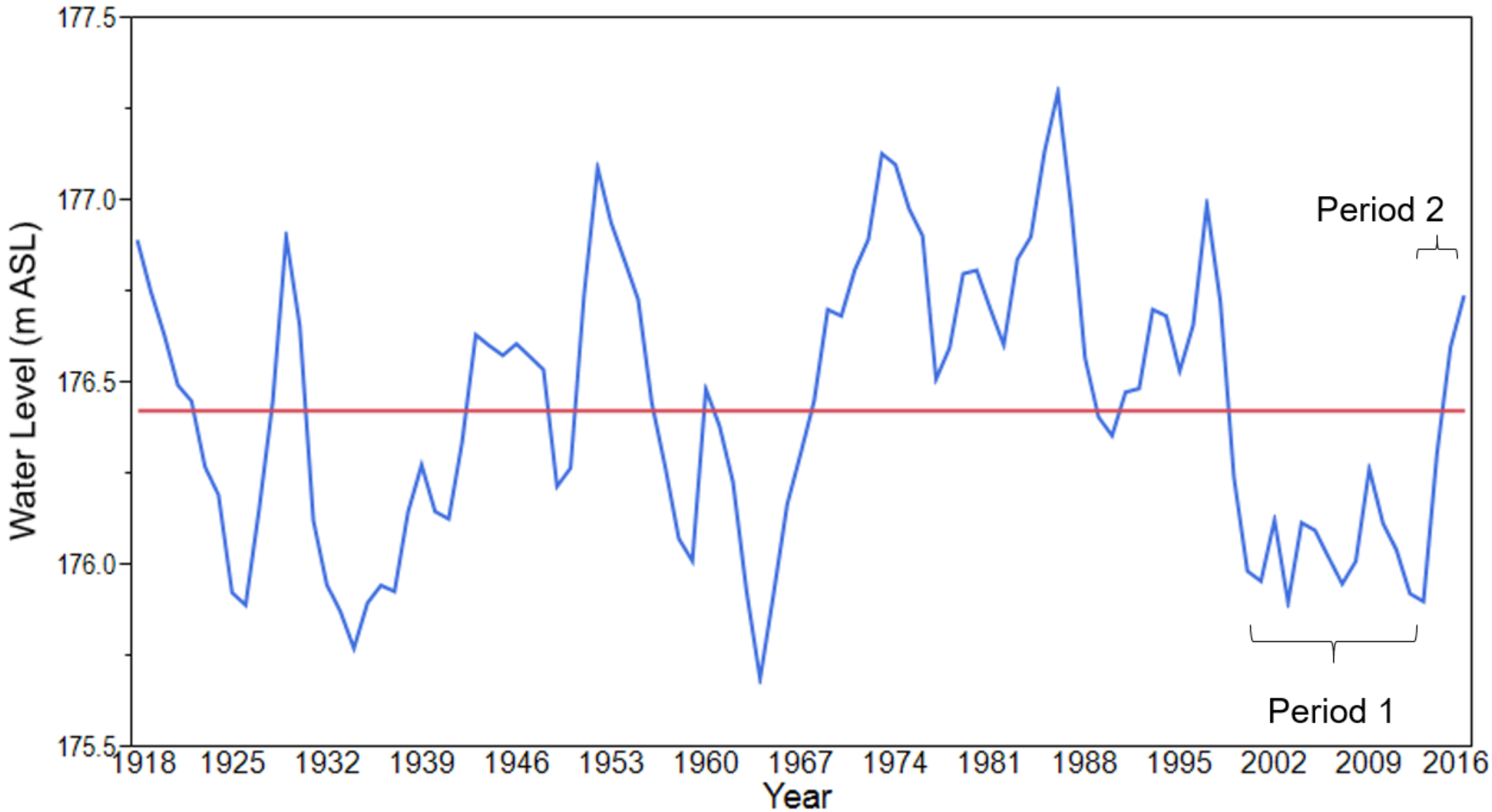

Figure 2.1. Annual average water levels in Lake Michigan-Huron from 1918 through 2016. The red line represents the long-term mean water level (176.42 m ASL). The two sampling periods are indicated; Period 1 is during a time of sustained low water levels (2003-2011) and Period 2 is when water levels were transitioning to a high scenario (2014-2016). Water level data was obtained from the Canadian Hydrographic Service.

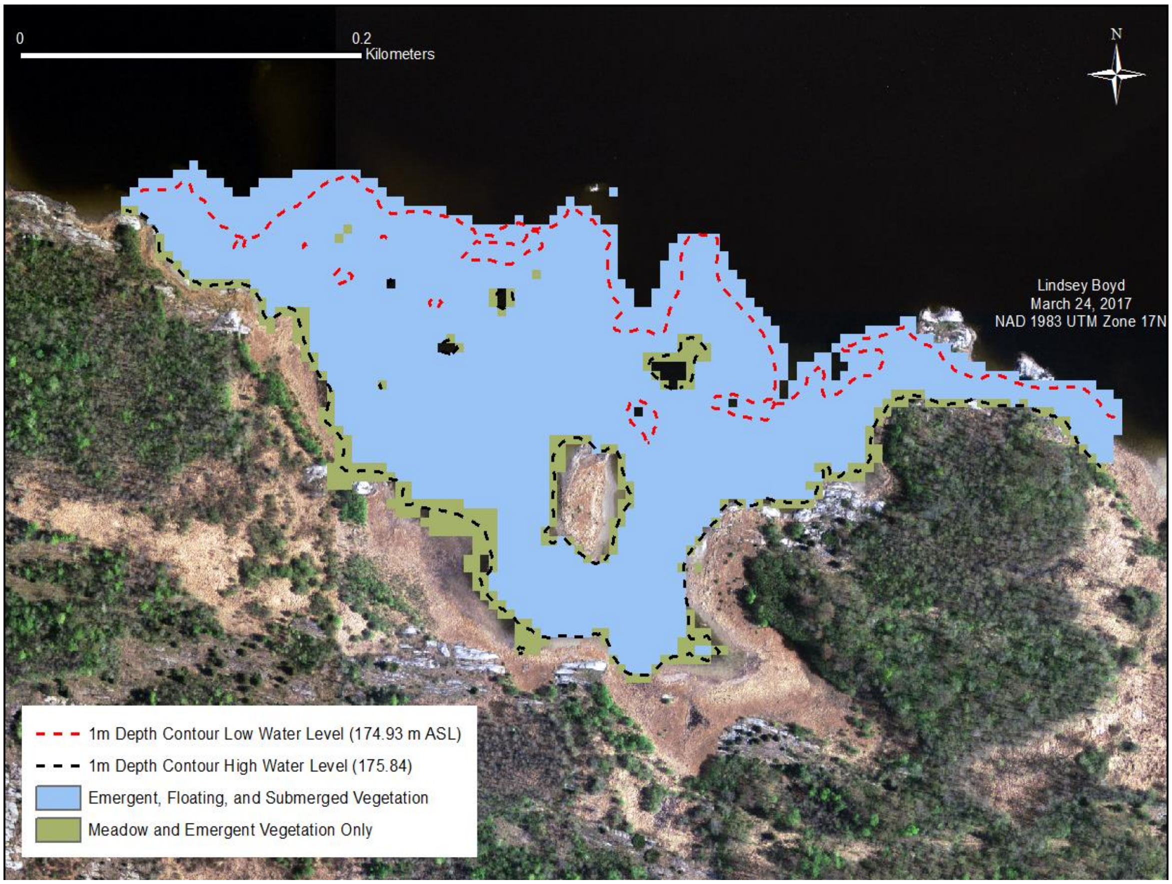

Figure 2.2. The location of the 1m depth contour in a coastal wetland during two different water levels. The location of the zone dominated by emergent, floating, and submerged vegetation where fyke nets are typically set is illustrated.

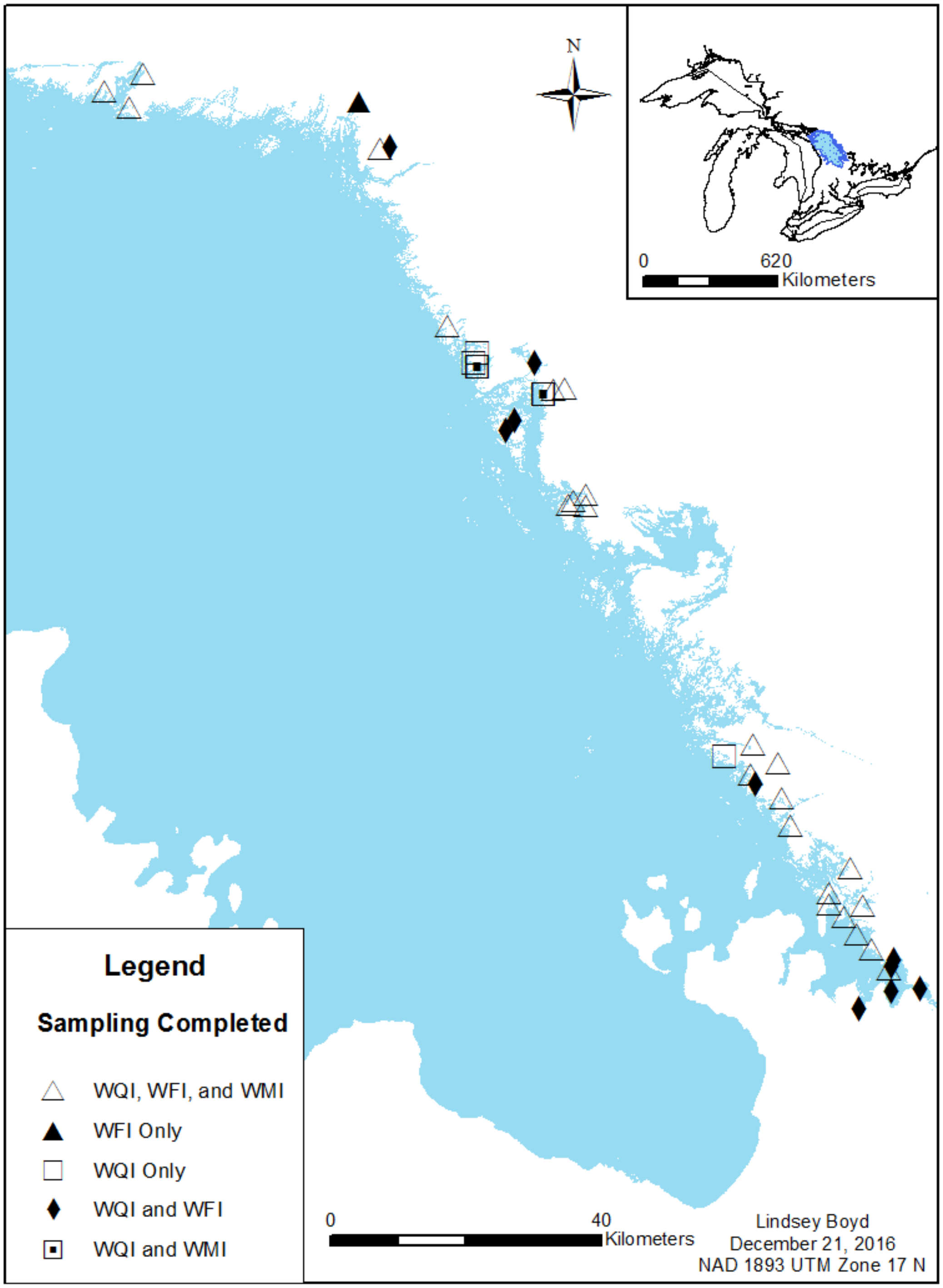

Figure 2.3. A site map indicating the locations of all 40 coastal wetlands sampled for at least one of water quality, fish, or macrophyte communities in Georgian Bay, Lake Huron.

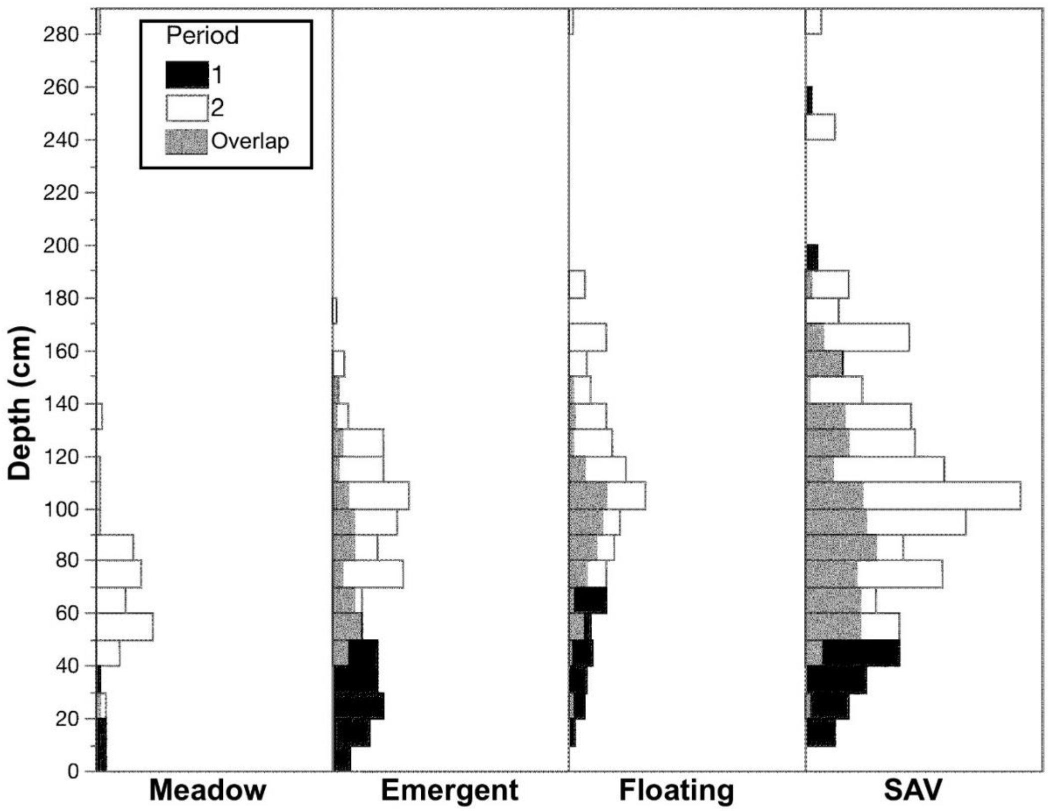

Figure 2.4. A plot showing the depths where meadow, emergent, floating, and SAV macrophytes were found during two sampling periods. Period 1 extends from 2004 – 2008, and Period 2 includes sampling completed in 2015 and 2016 following an ~0.9m increase in water level.

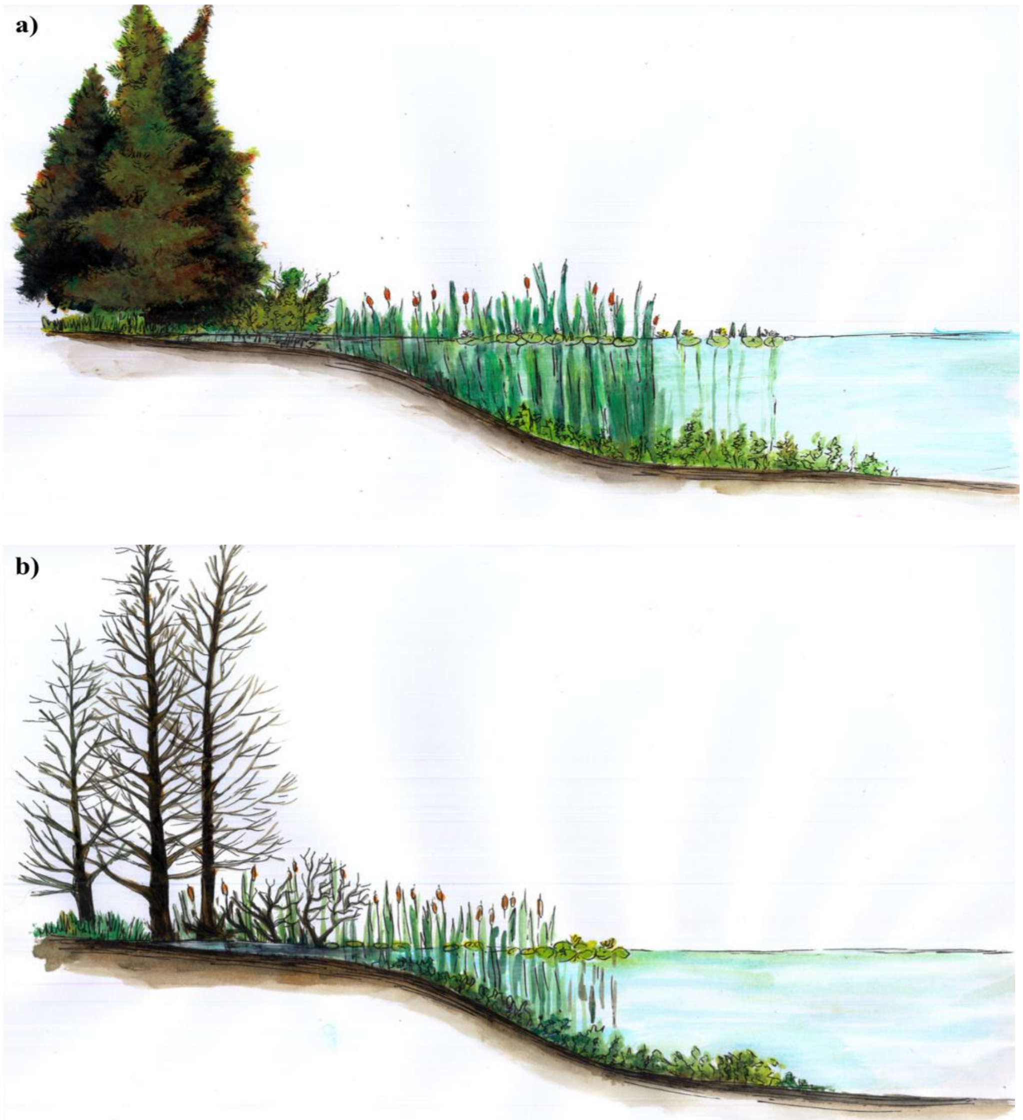

Figure 2.5a) A depiction of wetland vegetation is distributed in a wetland following a ~1 m increase in water level, preceded by 10+ years of sustained low-water levels. Vegetation includes grasses, trees, shrubs, and emergent, floating and submerged aquatic vegetation.

2.5b) Illustrates the reestablishment of vegetation types in their 'typical' depth zones following at least 5 years of inundation. Diagrams were created by Prabha Rupasinghe.

Figure 2.6. A depiction of a wetland with a steep basin slope experiencing a ~1 m increase in water level, following a 10+ year period of sustained low water levels. Diagram created by Prabha Rupasinghe.

GENERAL CONCLUSION

During a period of abrupt water-level increase, preceded by 10+ years of sustained water levels, the vegetation communities in coastal wetlands are subject to large spatial changes. The amount of change experienced is predominantly a product of the basin slope, where gentler slopes experience a larger spatial shift. In general, the amount of area available for emergent, floating, and submerged vegetation to grow is larger under high-water level conditions than low-water level conditions. Conversely, meadow vegetation growth area becomes restricted under high-water levels. In the future, more detailed field validation studies on how vegetation communities and distributions change under different water levels should be undertaken. This will allow us to identify particular wetland habitats that may be at risk under the forecasted low-water levels.

Overall, the coastal wetlands in the Georgian Bay region are still among the most pristine in the Great Lakes. While the WQI and WMI are robust enough to monitor wetlands during a period of water-level transition, special care is required when attempting to sample the fish community for any purpose during this time. The large shifts in vegetation location bias fyke net catch and impede seine netting efficiency. These effects are more prominent in the more southern region of Georgian Bay, where gently sloping wetlands exhibit more extreme vegetation shifts. In the future, sampling protocols should be revised to include multiple sampling techniques, in order to limit catch biases during a time of transition.

APPENDIX

The following figures pertain to the estimates of suitable growth area during high and low water-level conditions, as described in Chapter 1. The area of where the vegetation growth area under high-water levels overlaps with the growth area under low-water levels is also indicated. The larger, the area of overlap, the more resilient this vegetation would be to a large (~1 m) change in water level as not all new growth would be required to establish from existing buried seed banks. Area and overlap estimates were completed for meadow, emergent, floating, SAV, and whole wetland potential area for 25 wetlands in Georgian Bay using ArcMap 10.5.

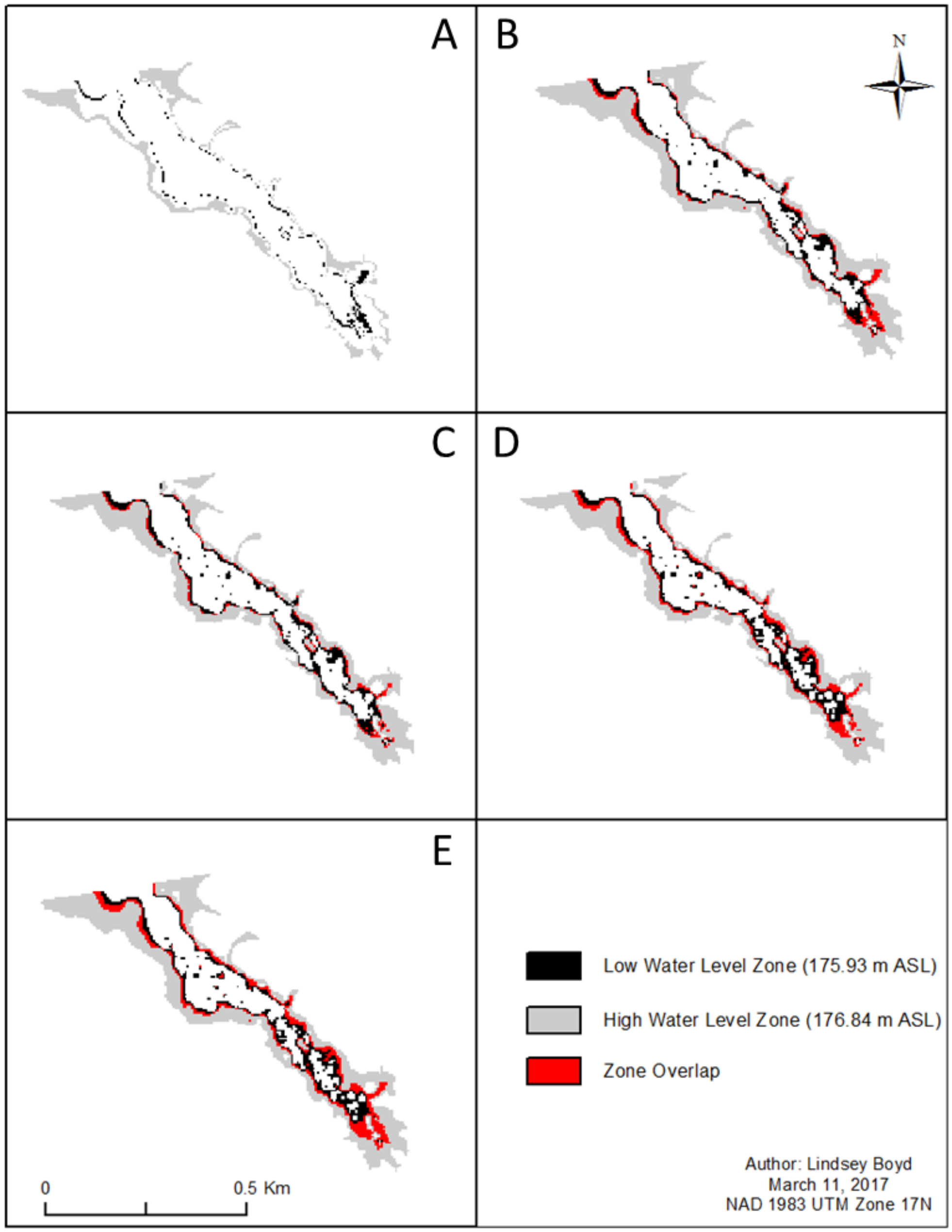

Figure 1. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Alexander Bay.

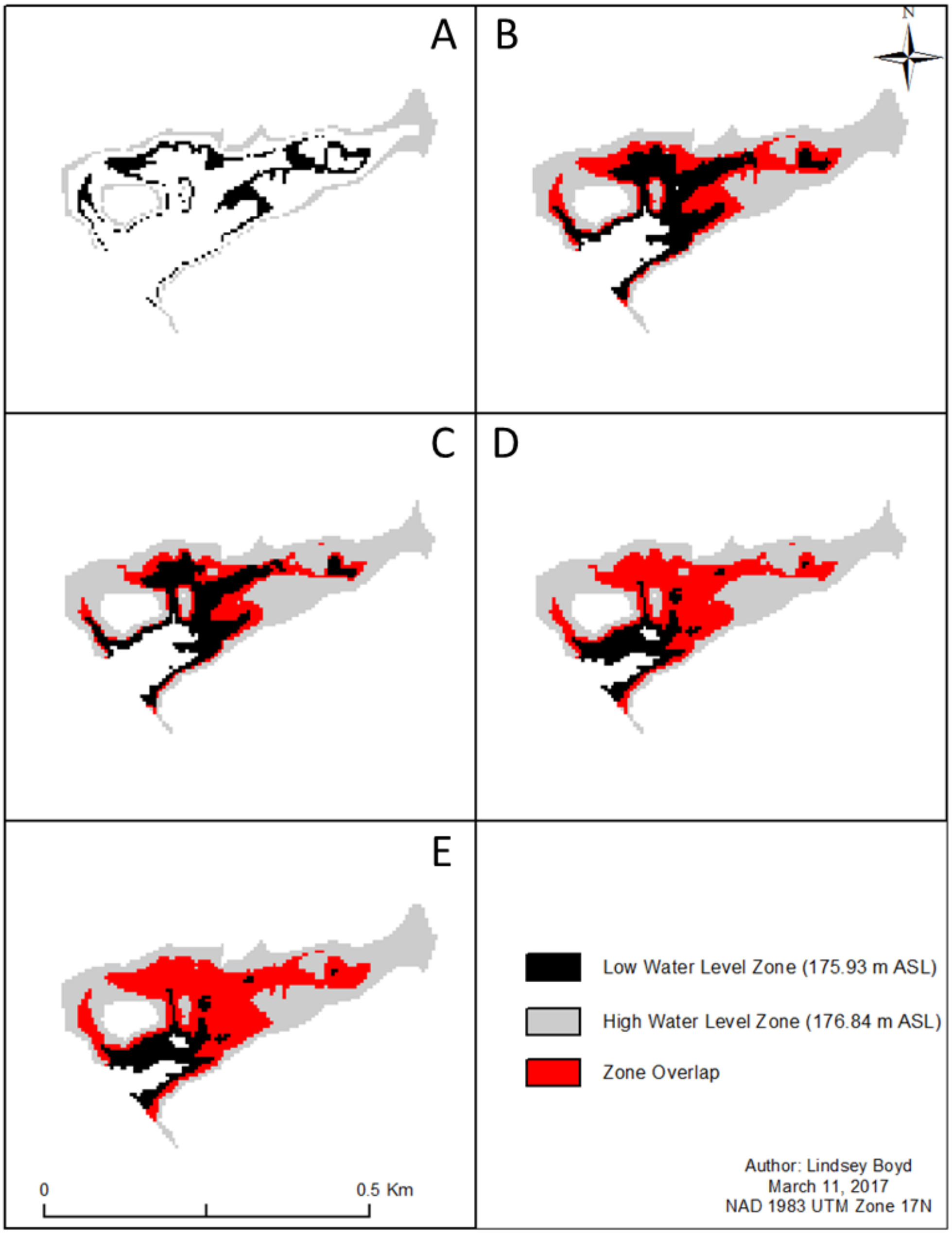

Figure 2. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Charles Inlet.

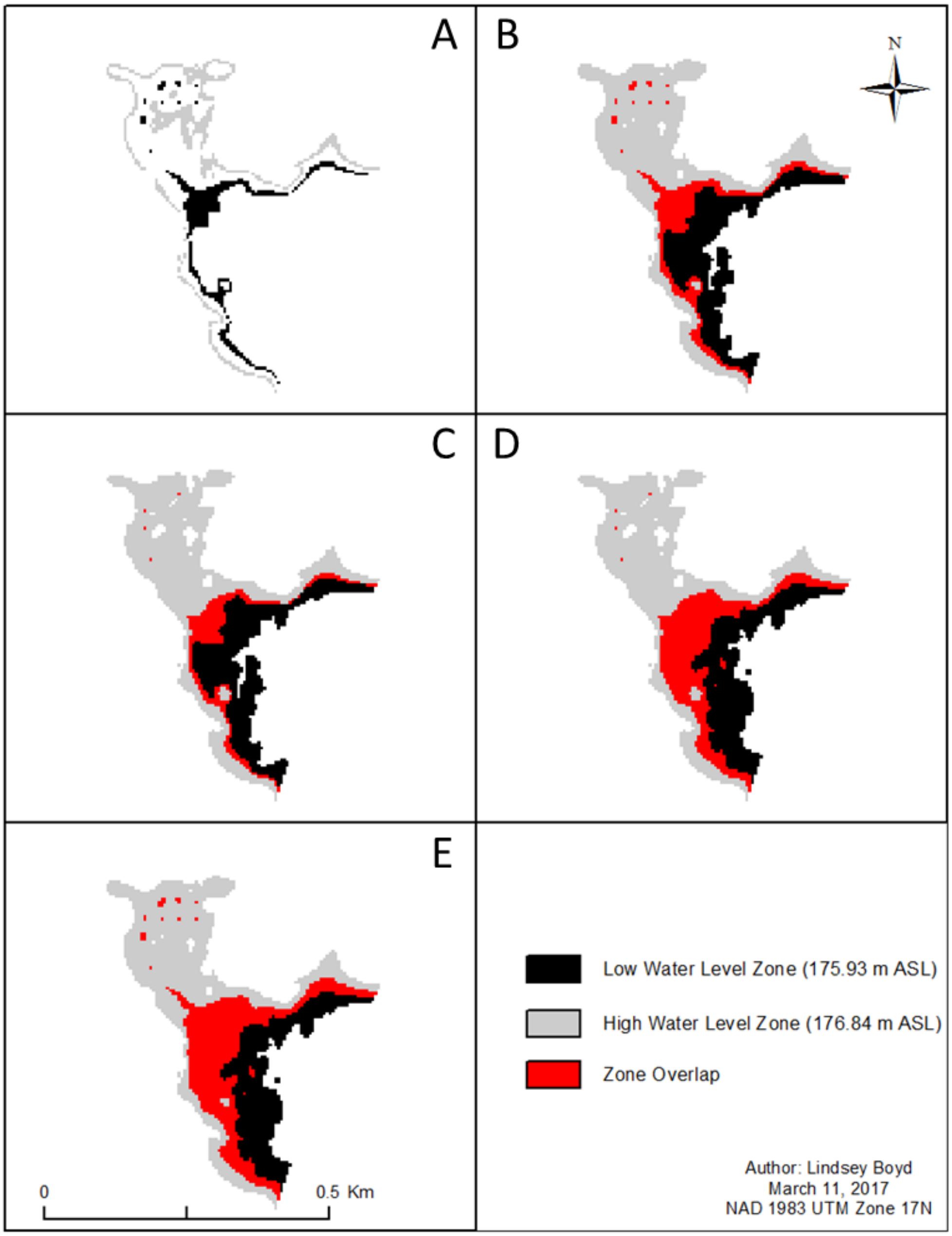

Figure 3. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Coffin Rock.

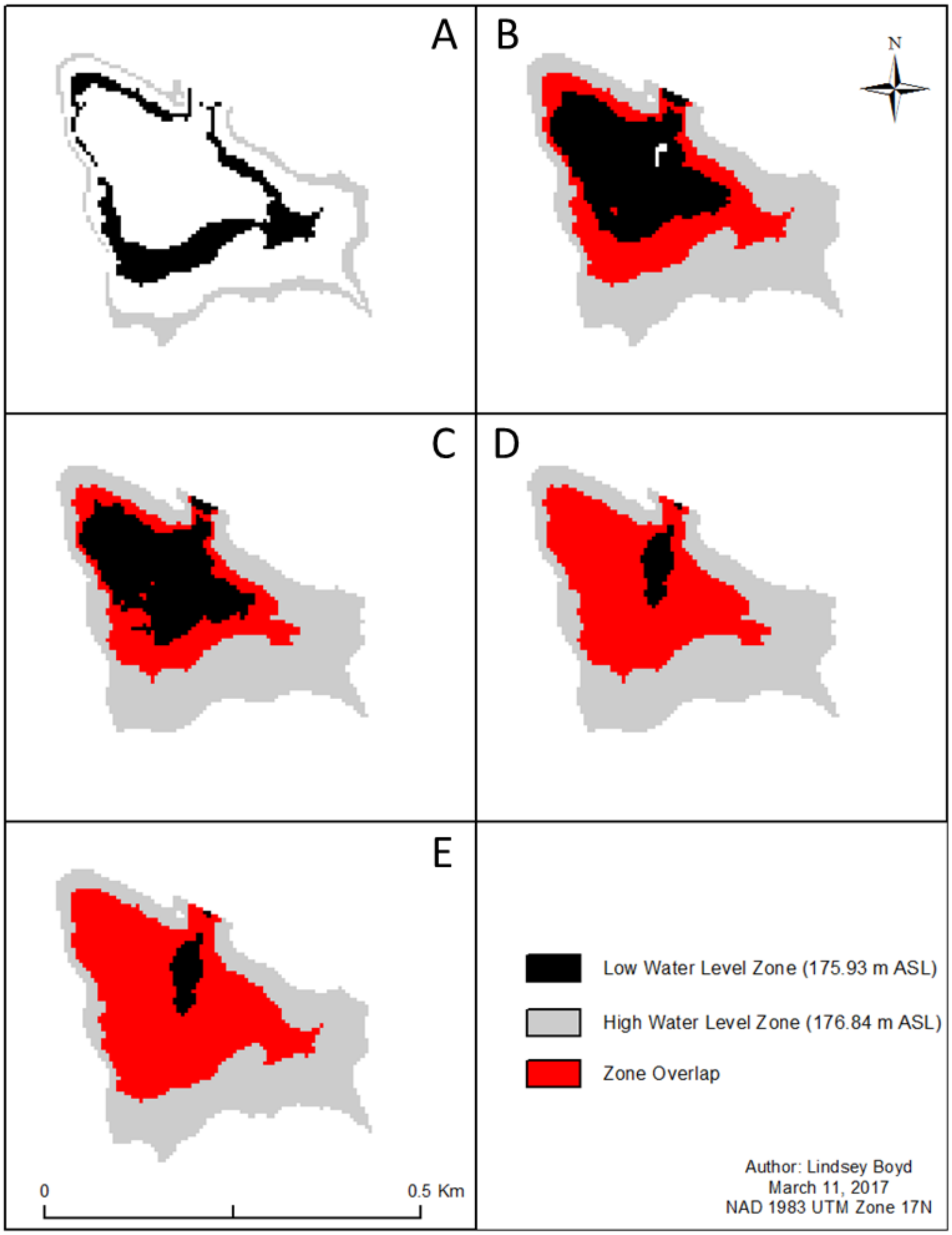

Figure 4. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Corbman Bay.

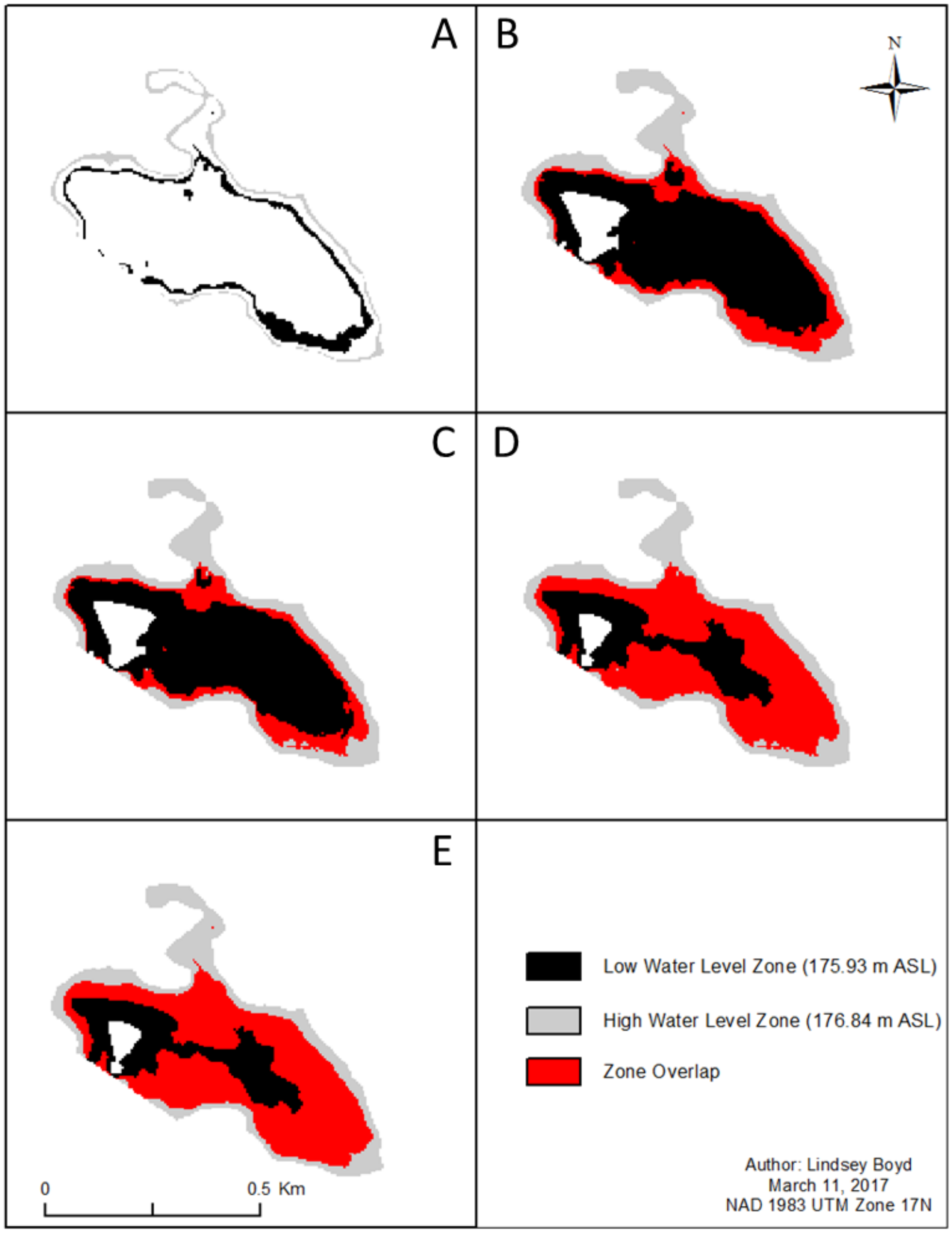

Figure 5. The area and location of the zone available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Cormican Bay.

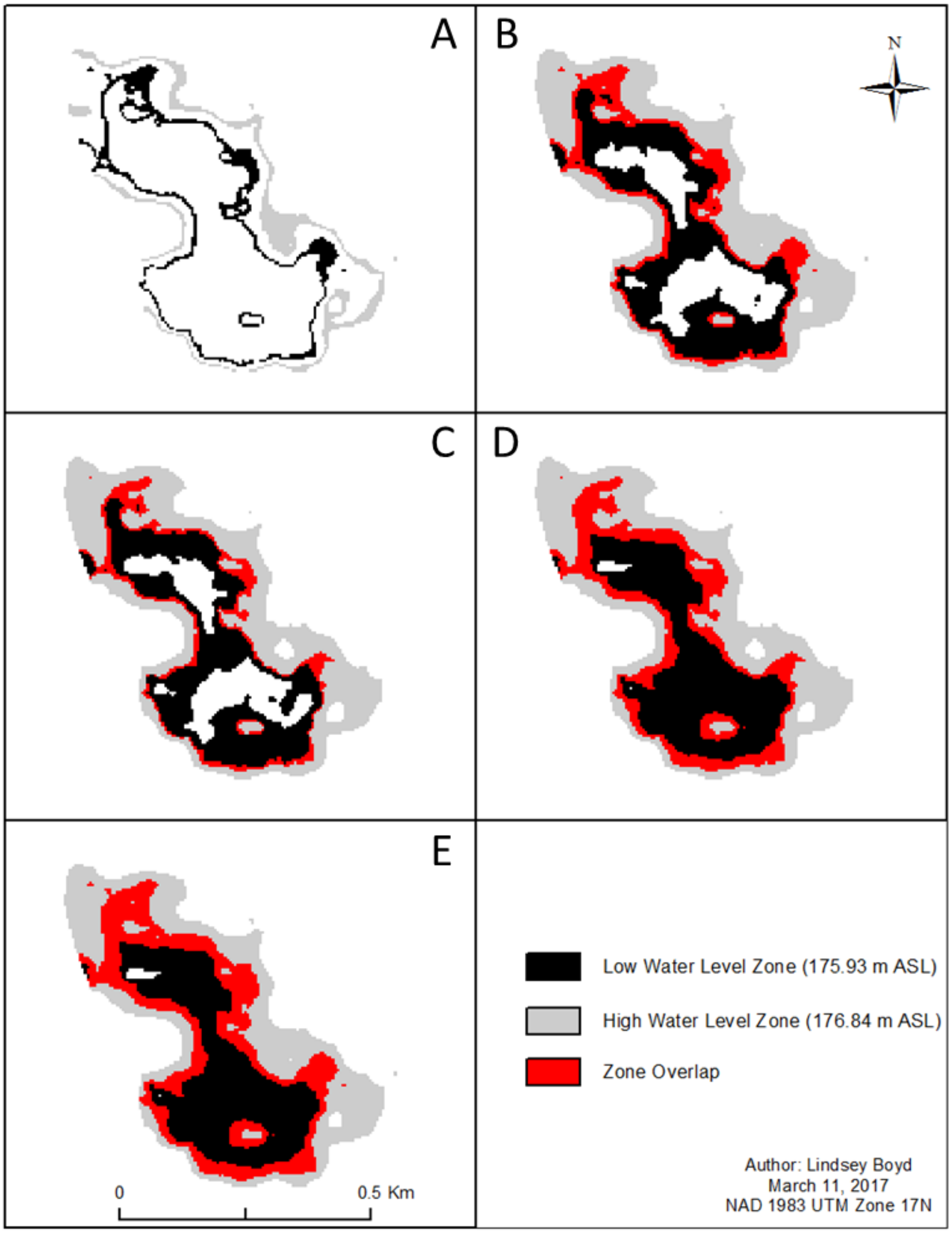

Figure 6. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: David's Bay.

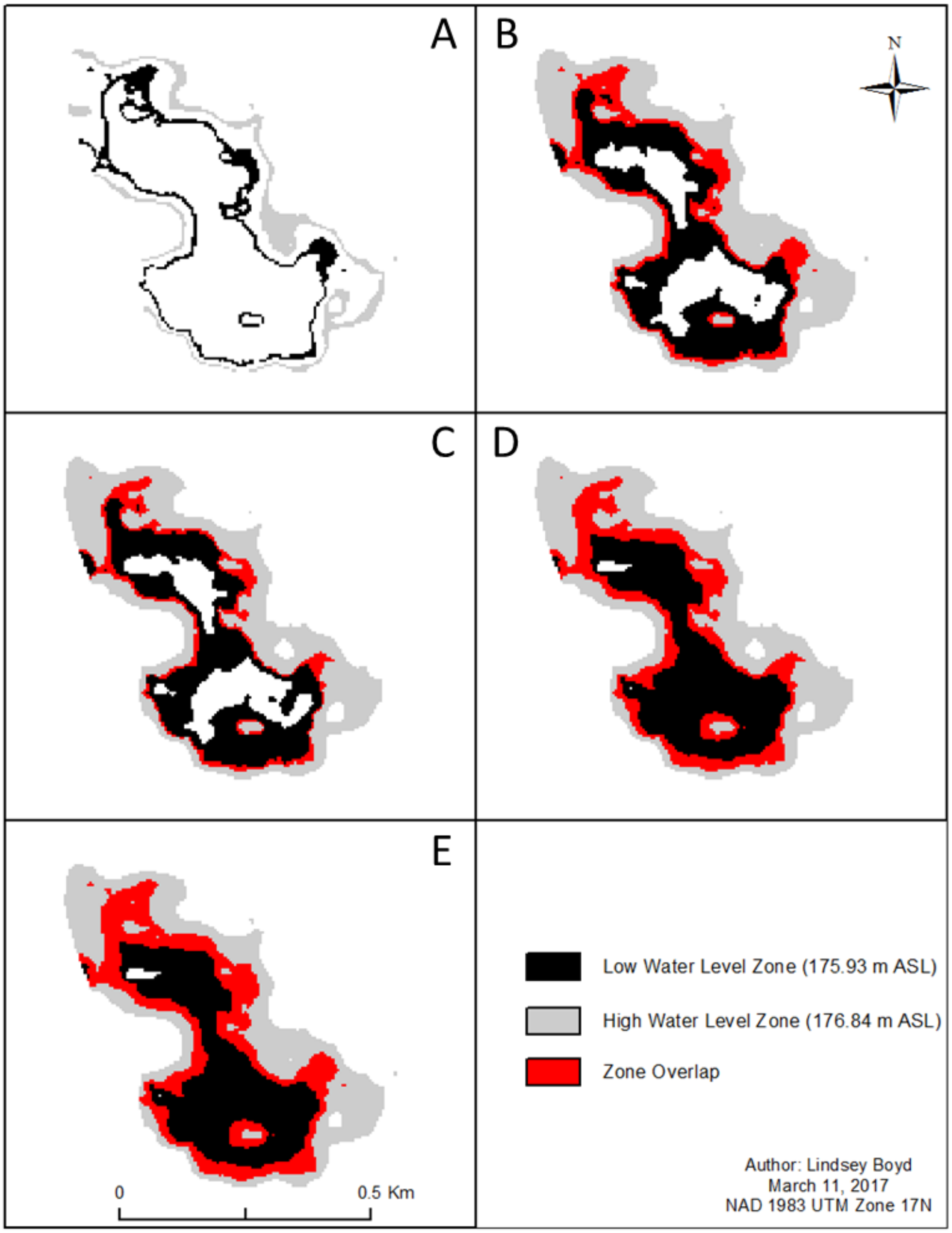

Figure 7. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Green Island.

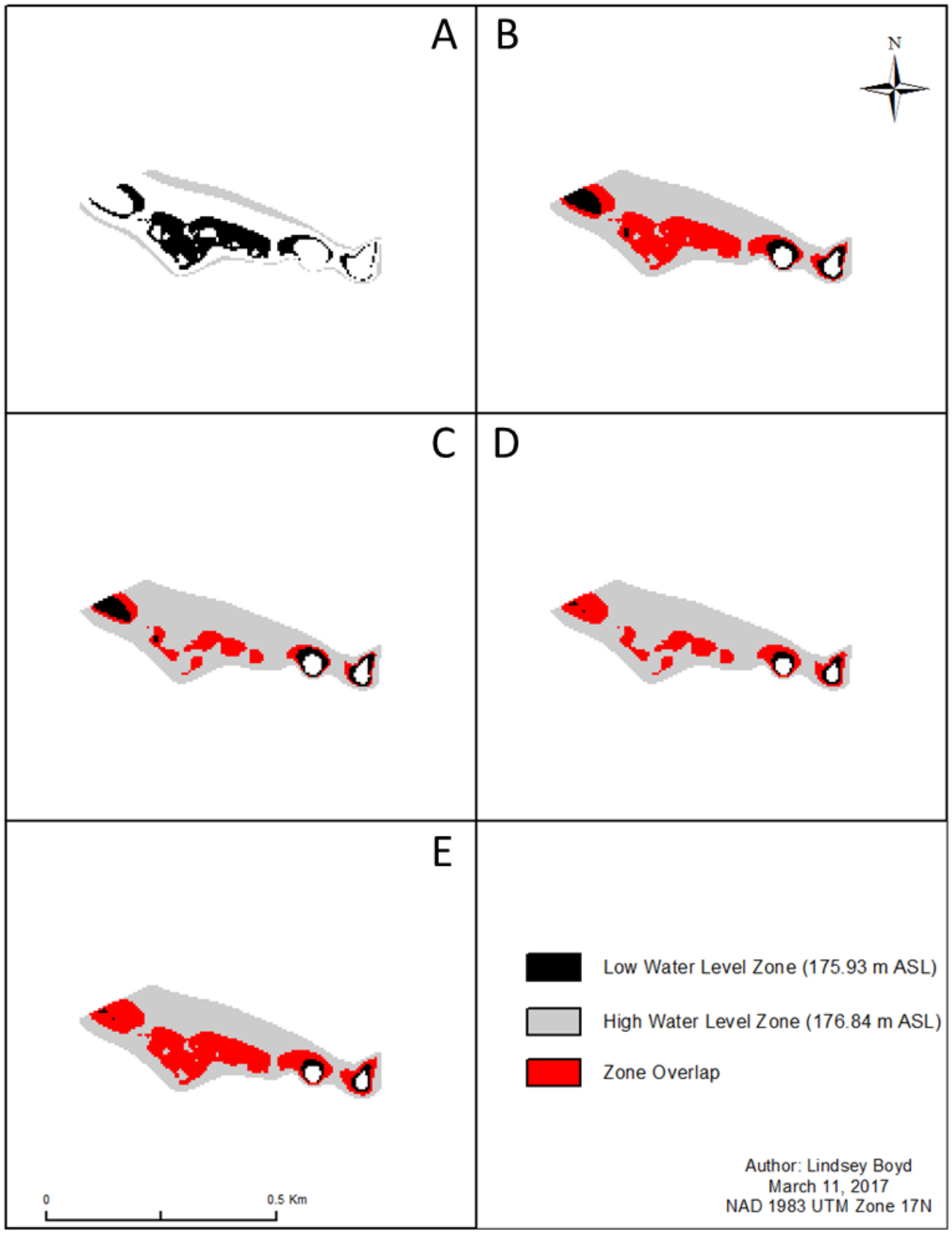

Figure 8. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Key River 1.

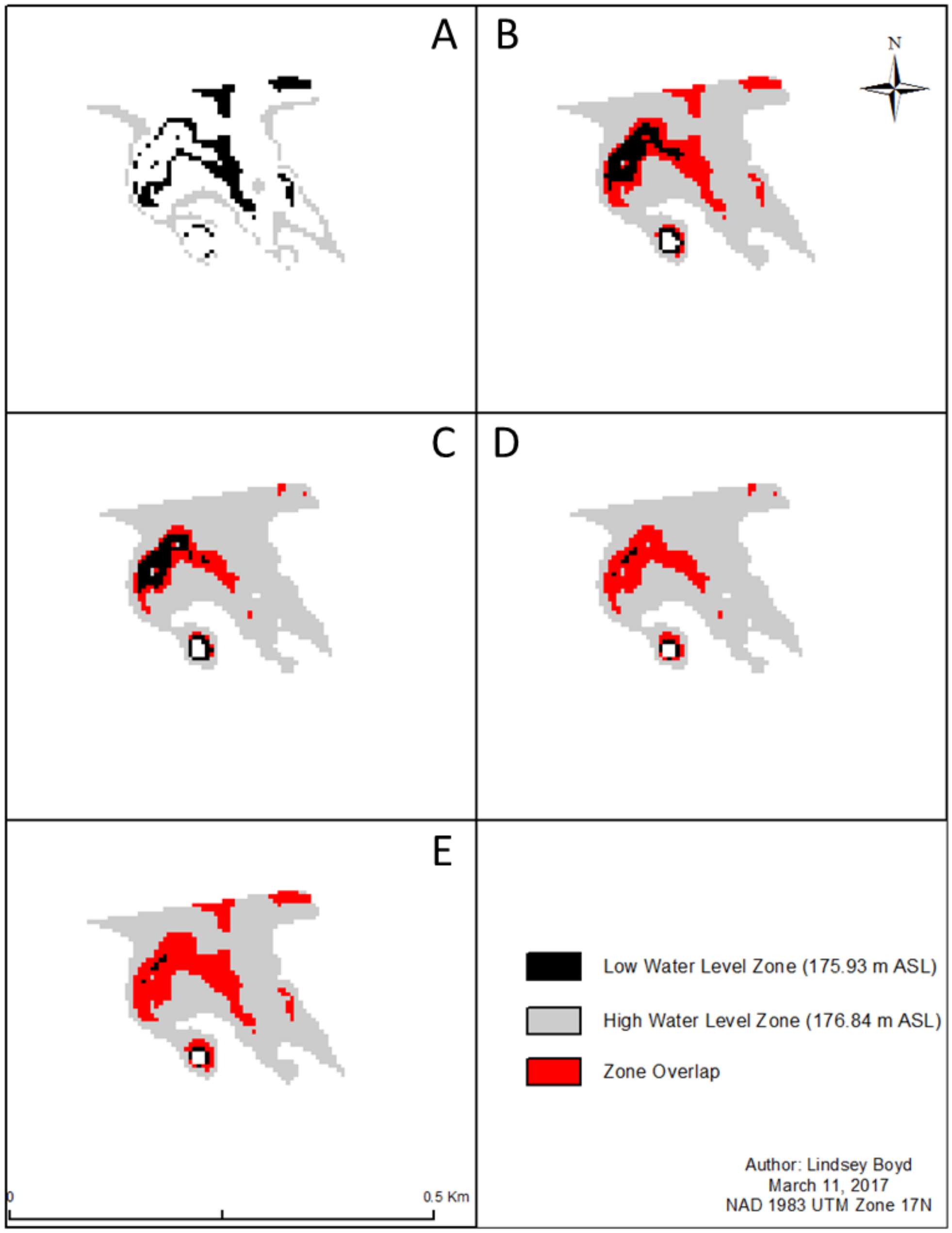

Figure 9. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Key River 3.

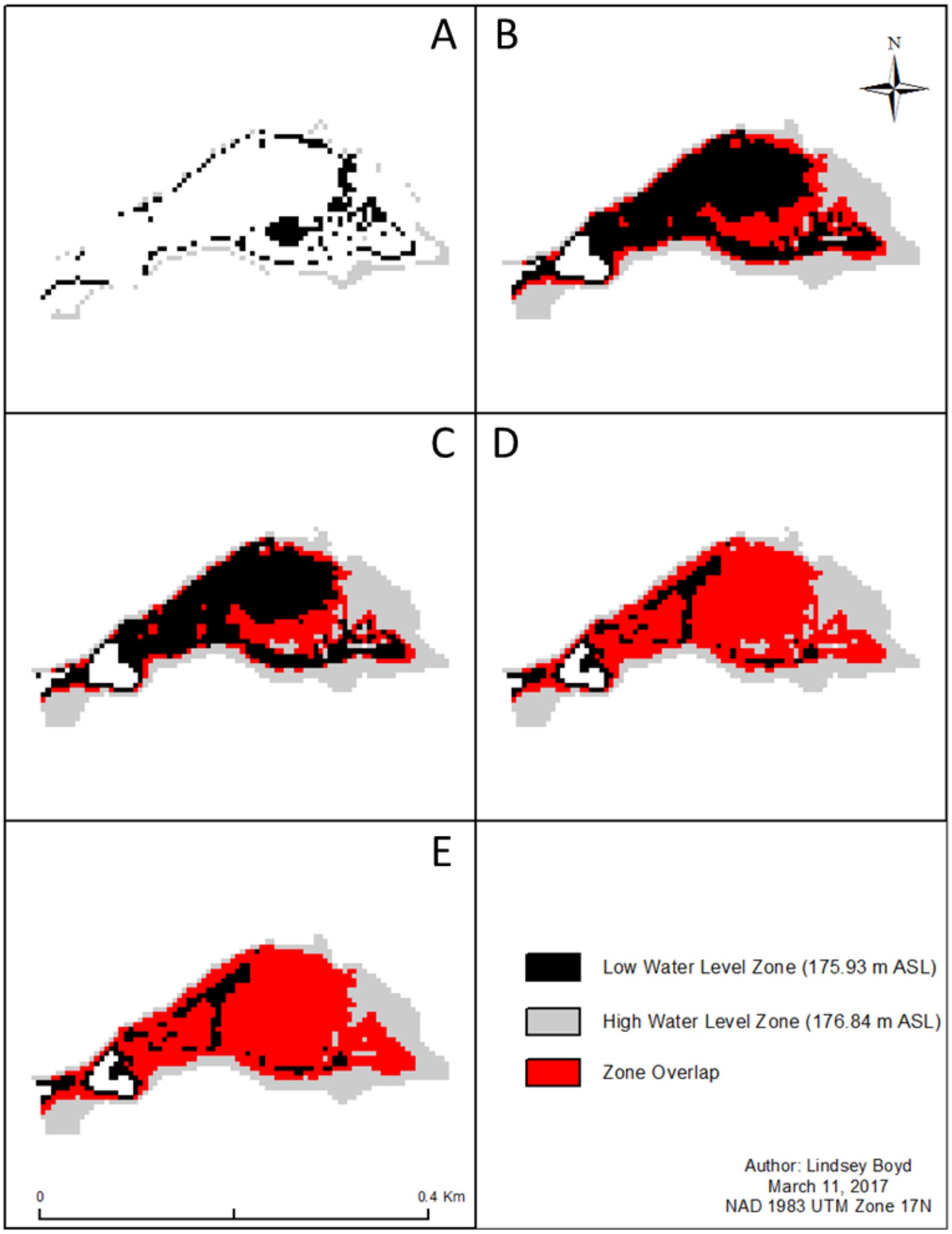

Figure 10. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Miners Creek.

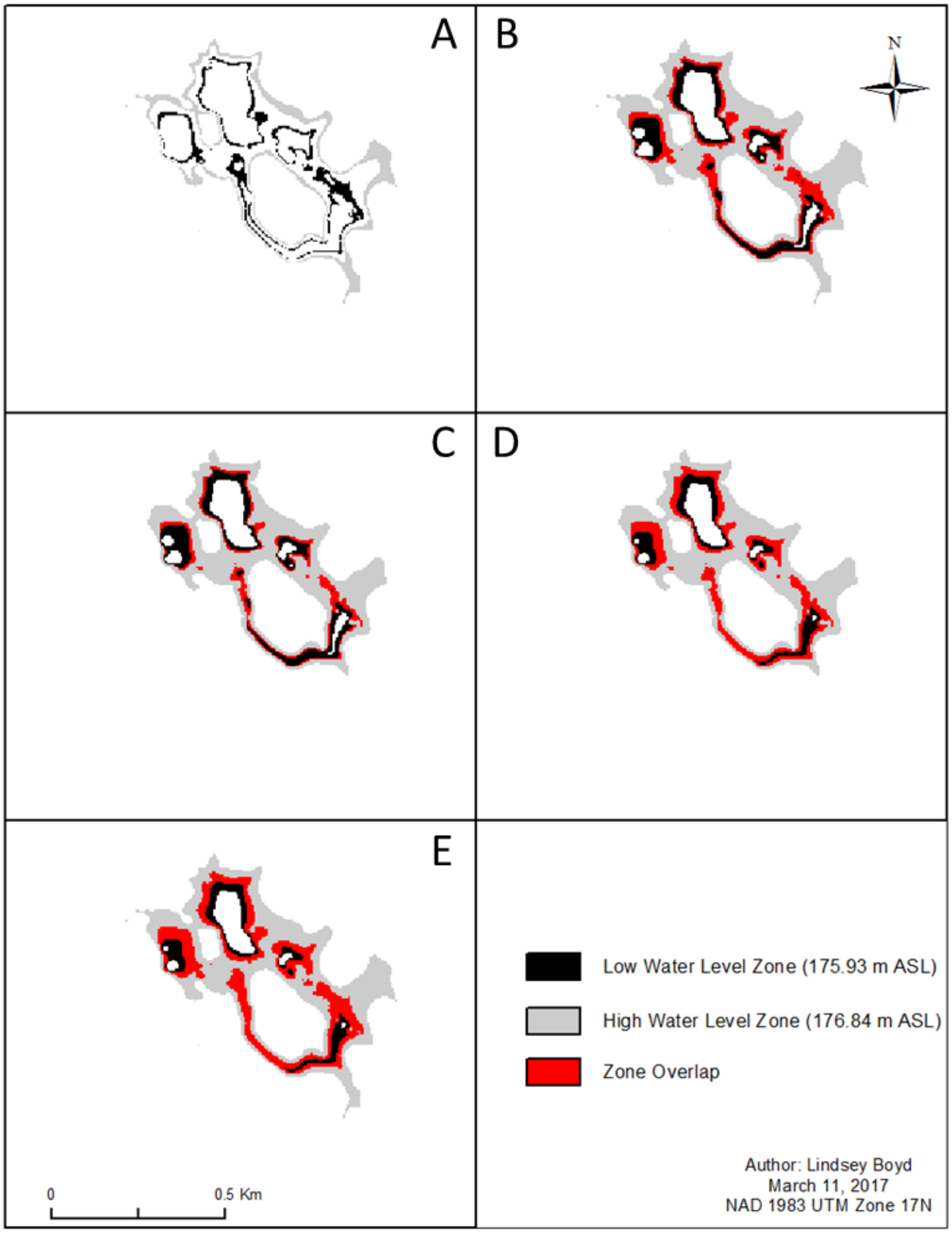

Figure 11. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Moreau Bay.

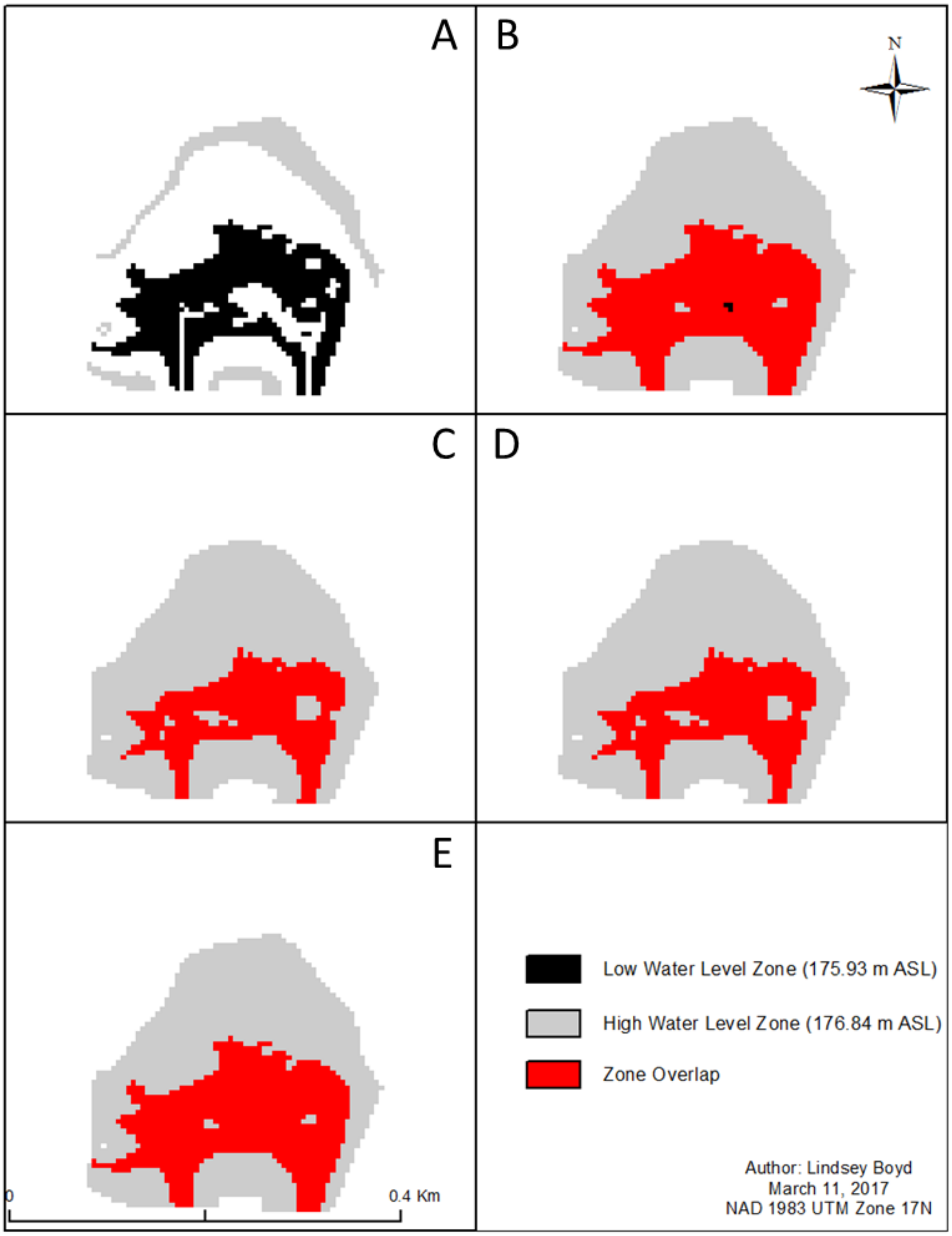

Figure 12. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Musky Bay.

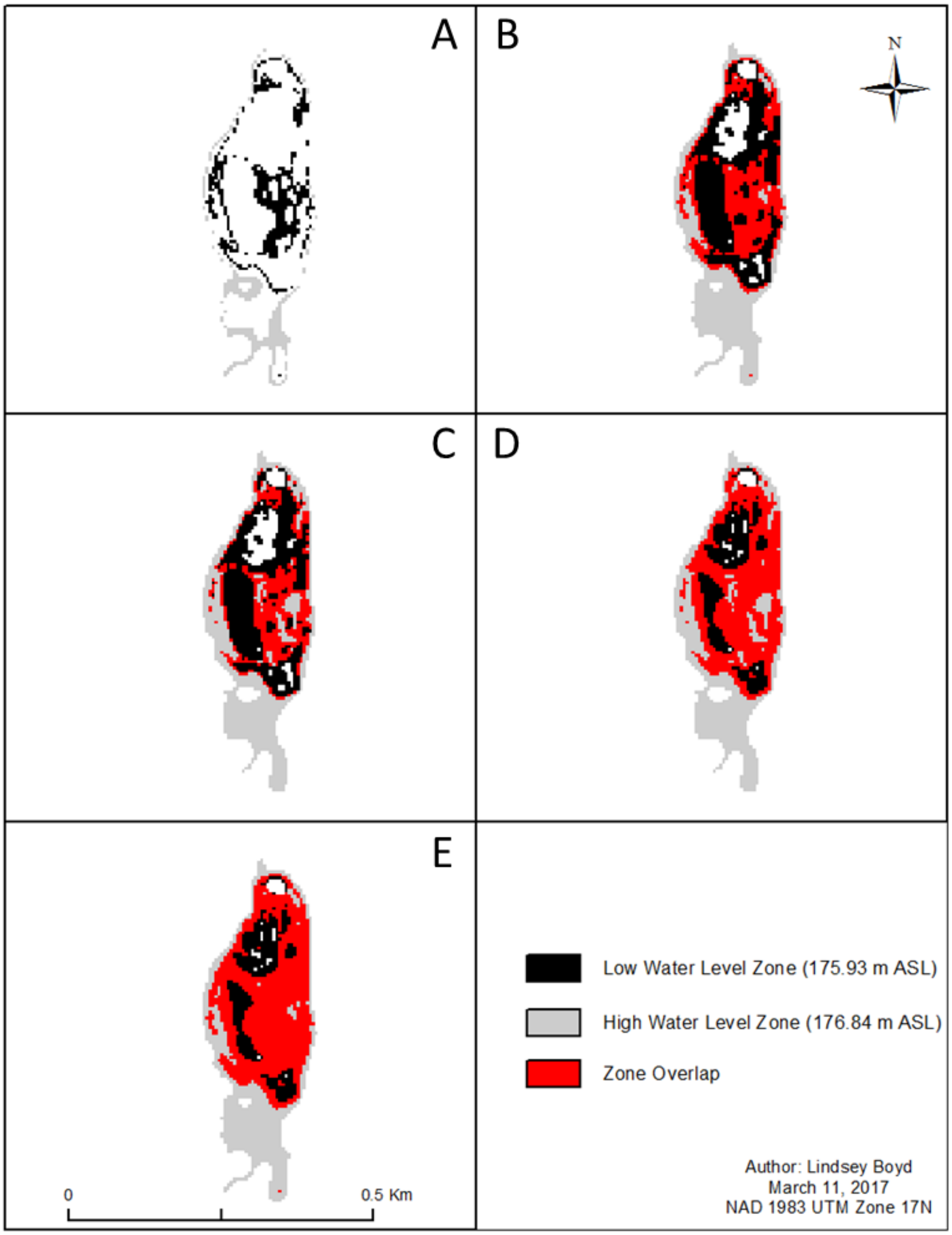

Figure 13. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: North Bay 1.

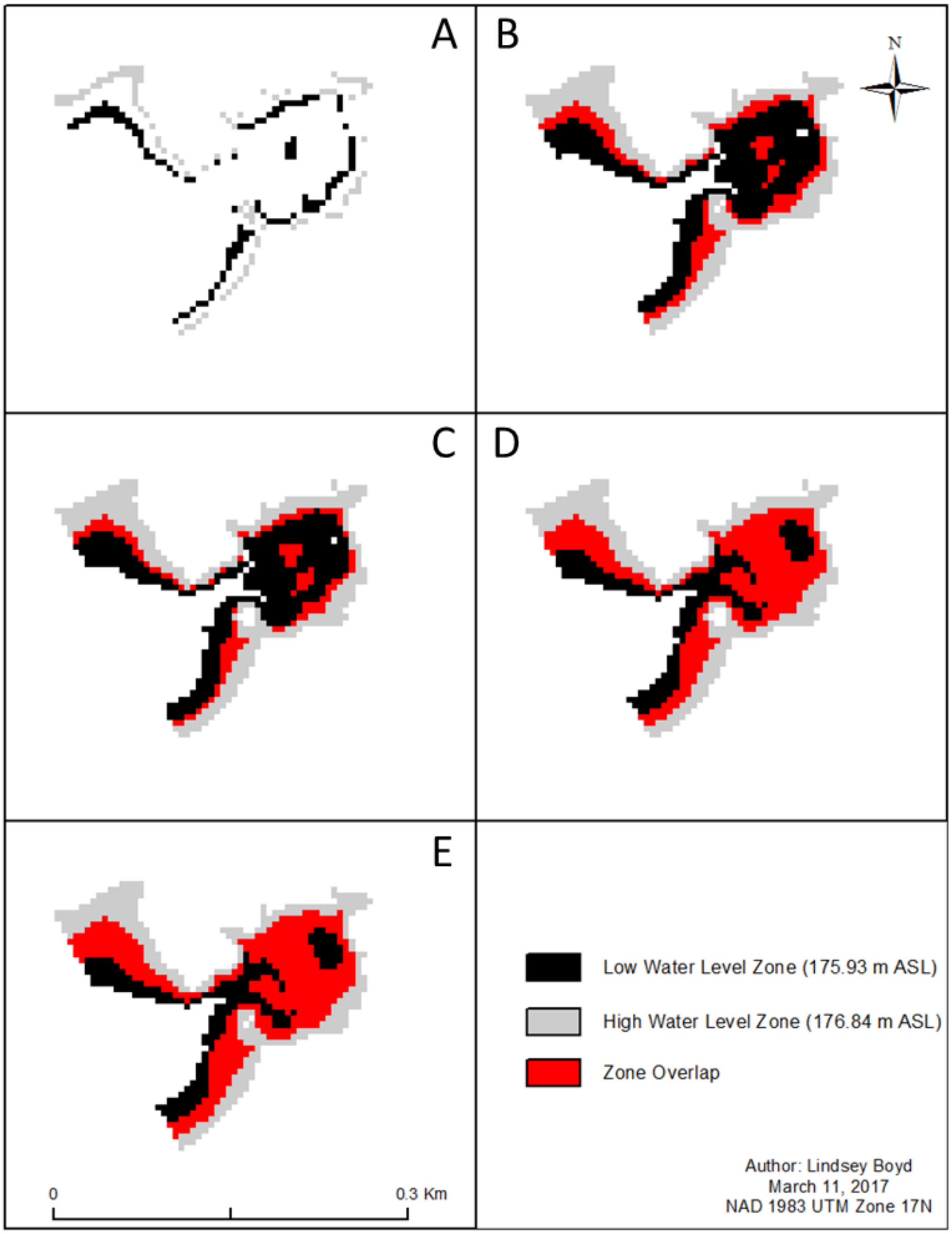

Figure 14. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: North Bay 5.

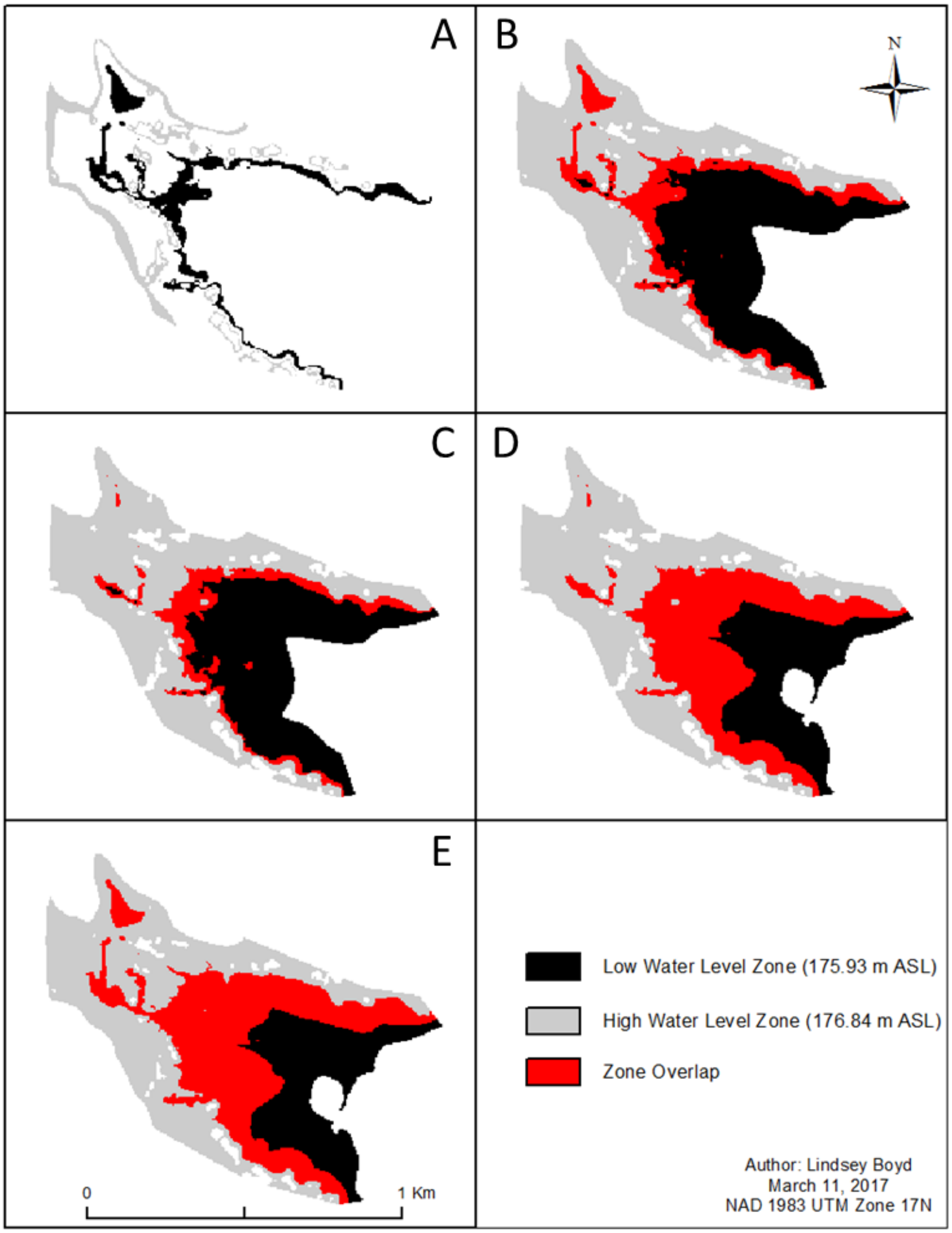

Figure 15. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Oak Bay.

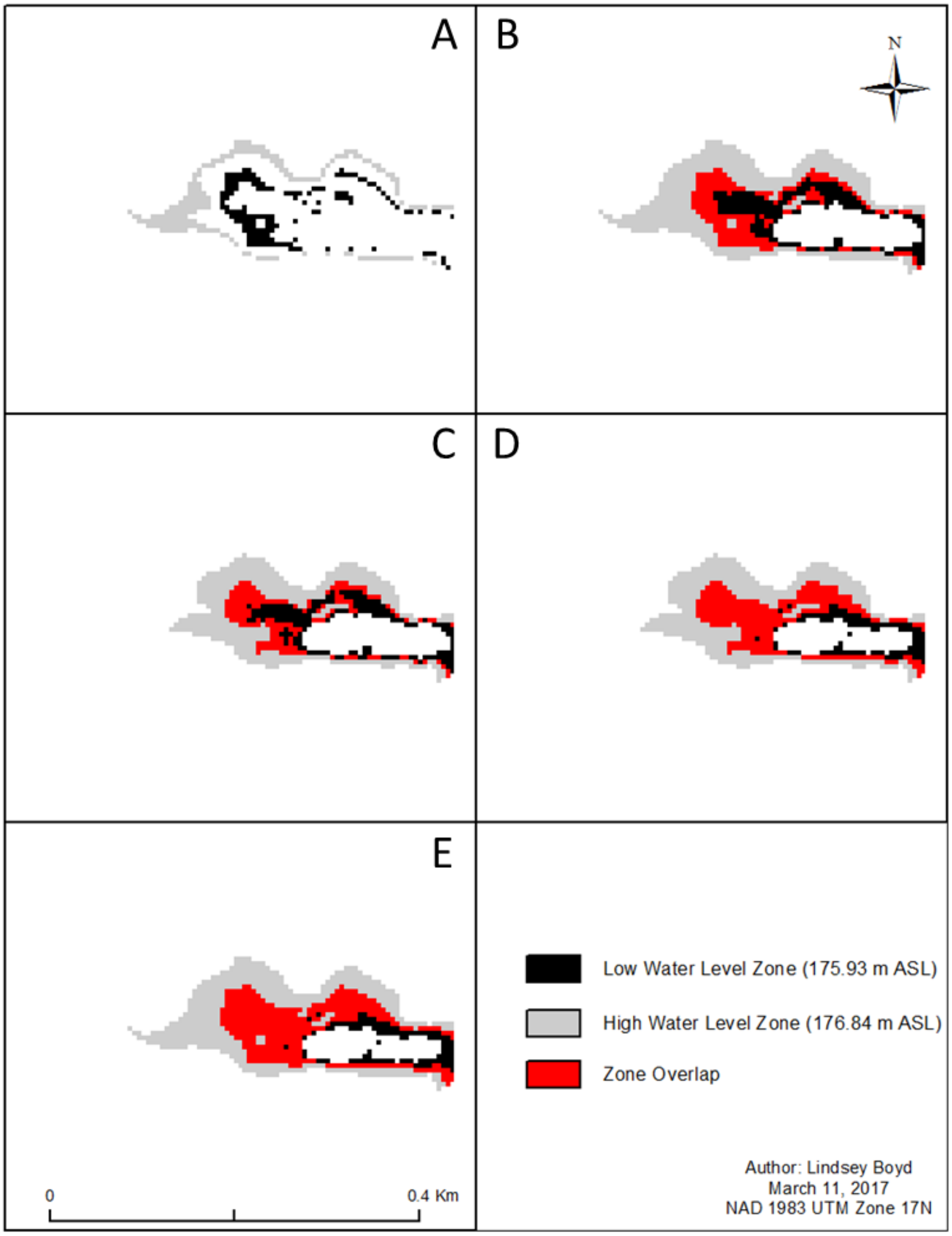

Figure 16. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Ojibway Bay.

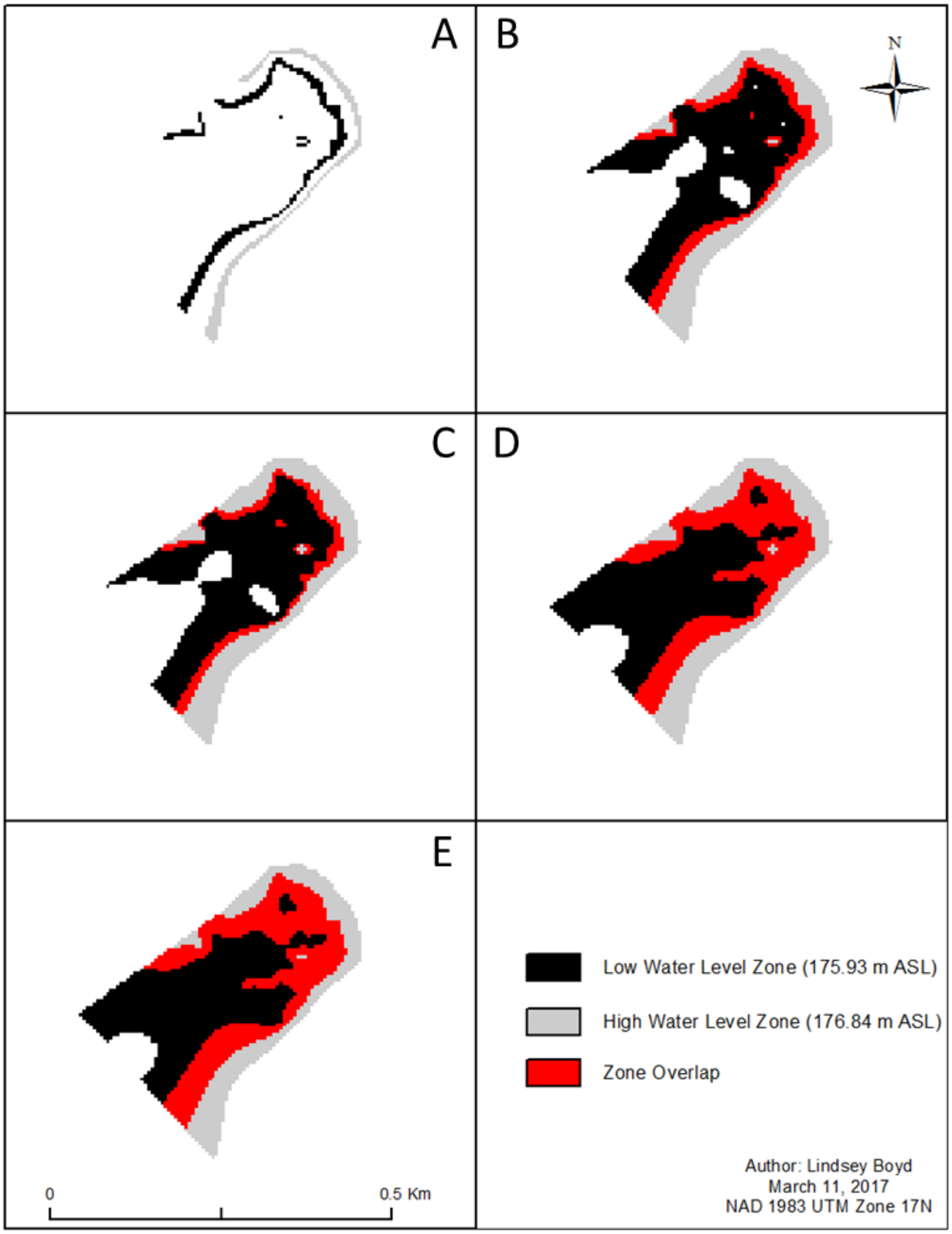

Figure 17. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Picnic Island.

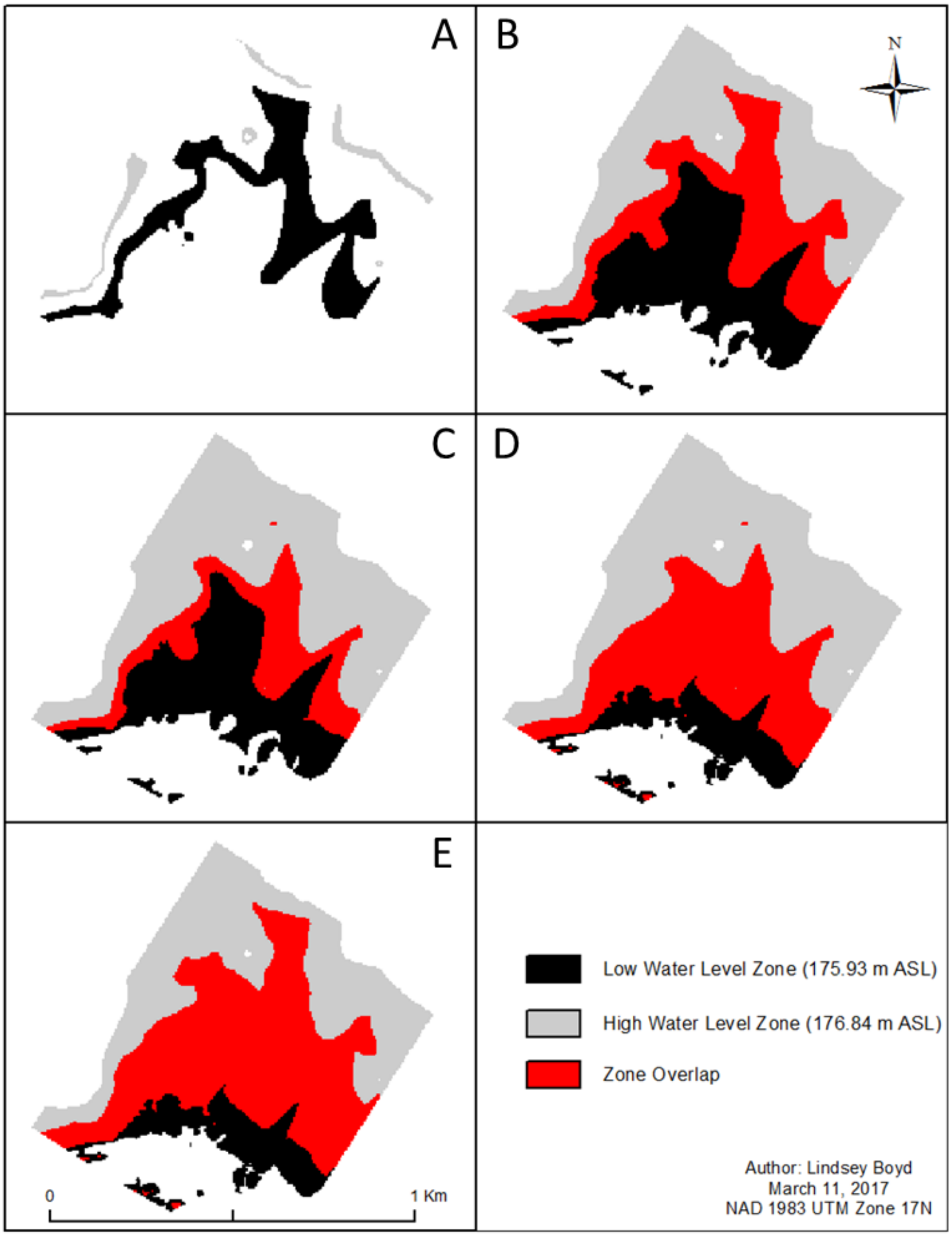

Figure 18. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Potato Island.

Figure 19. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Quarry Island.

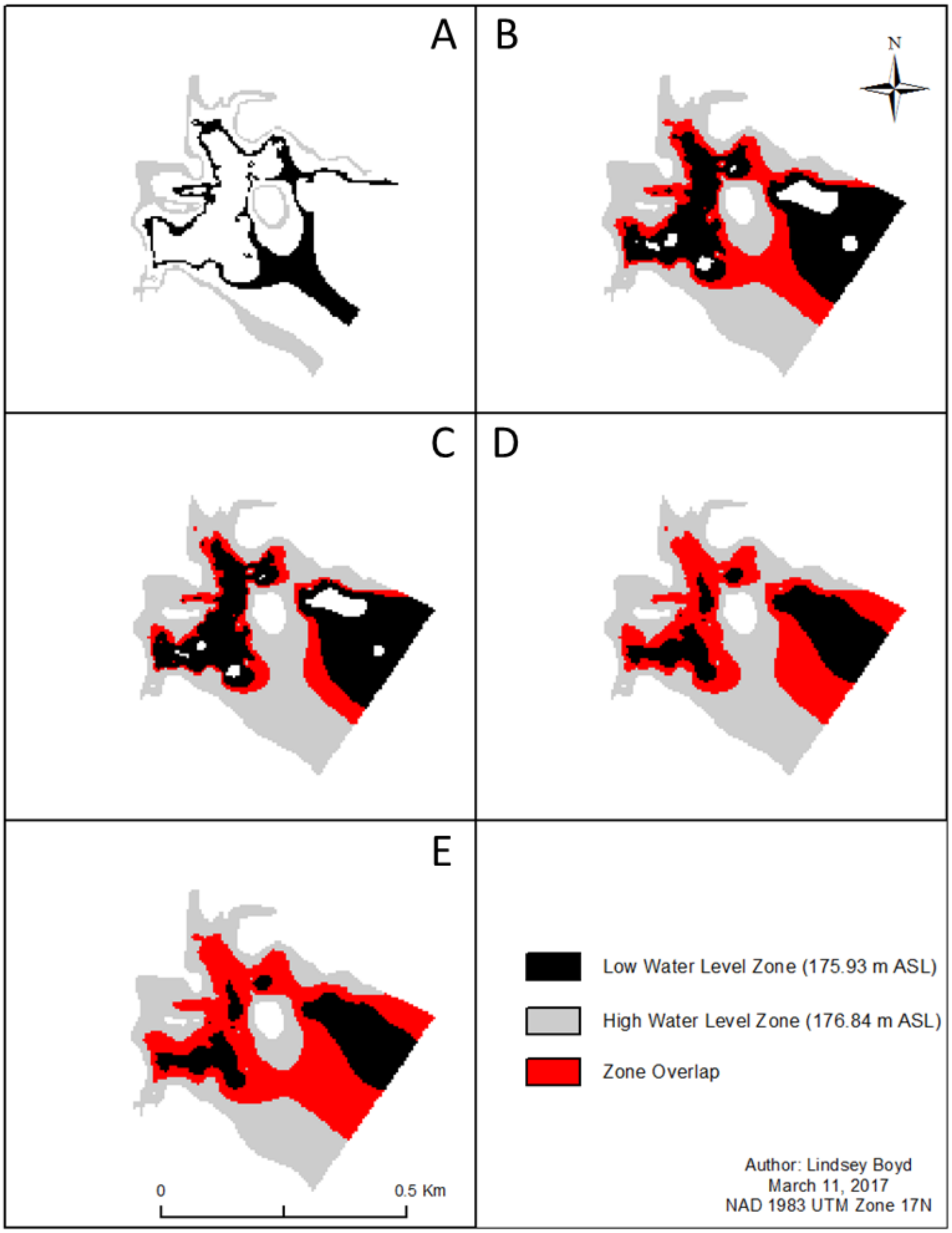

Figure 20. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Robert's Island.

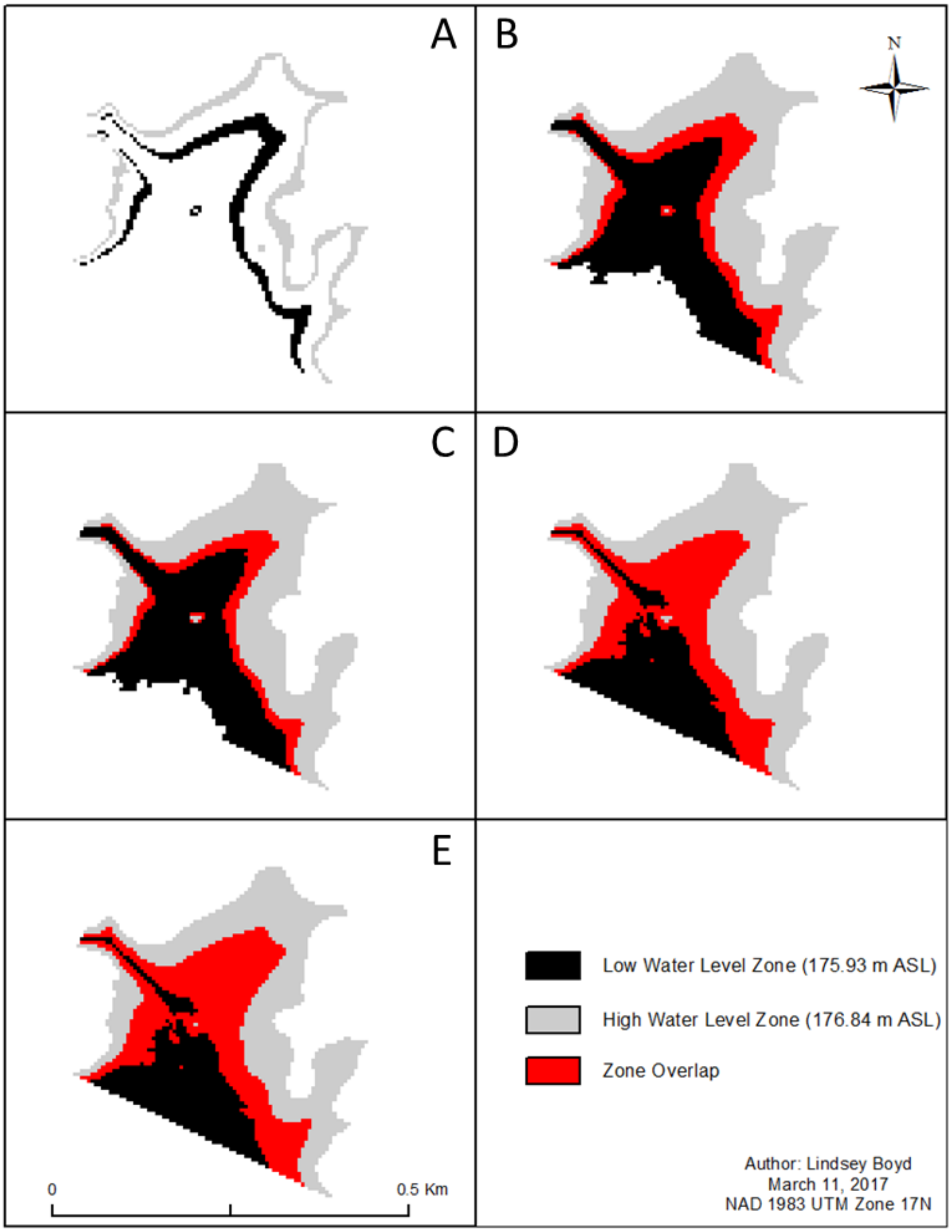

Figure 21. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Sturgeon Bay Central.

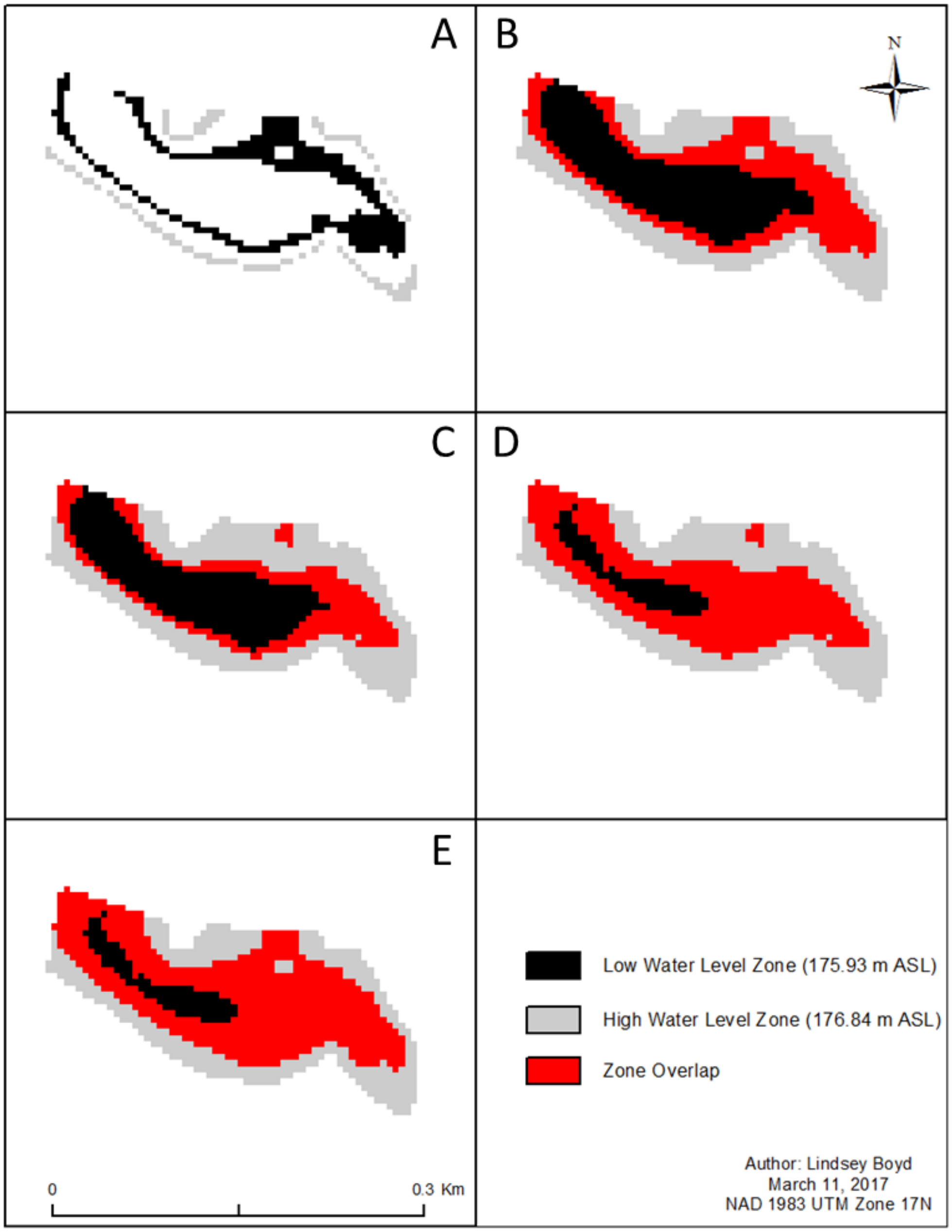

Figure 22. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Tadenac Bay 1.

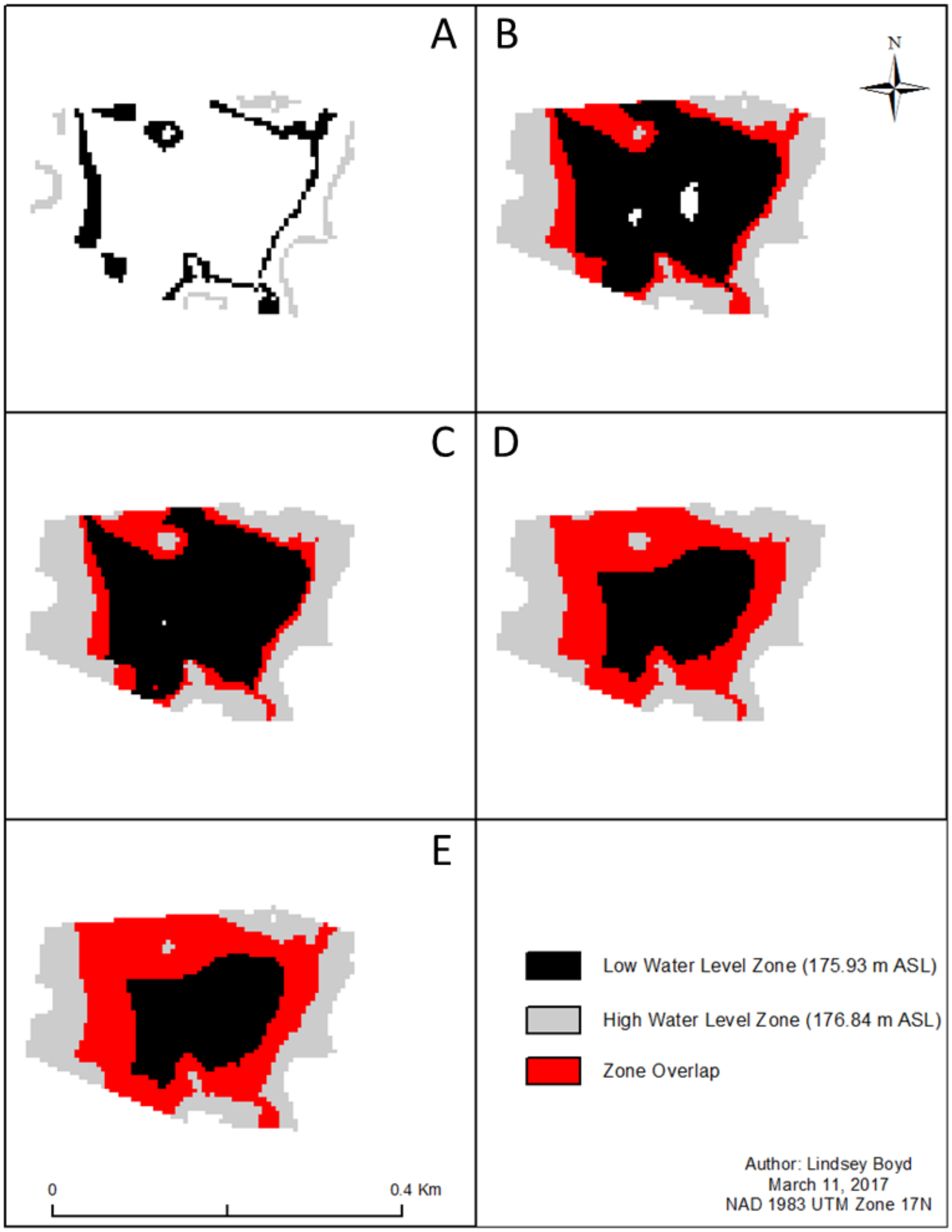

Figure 23. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Tadenac Bay 2.

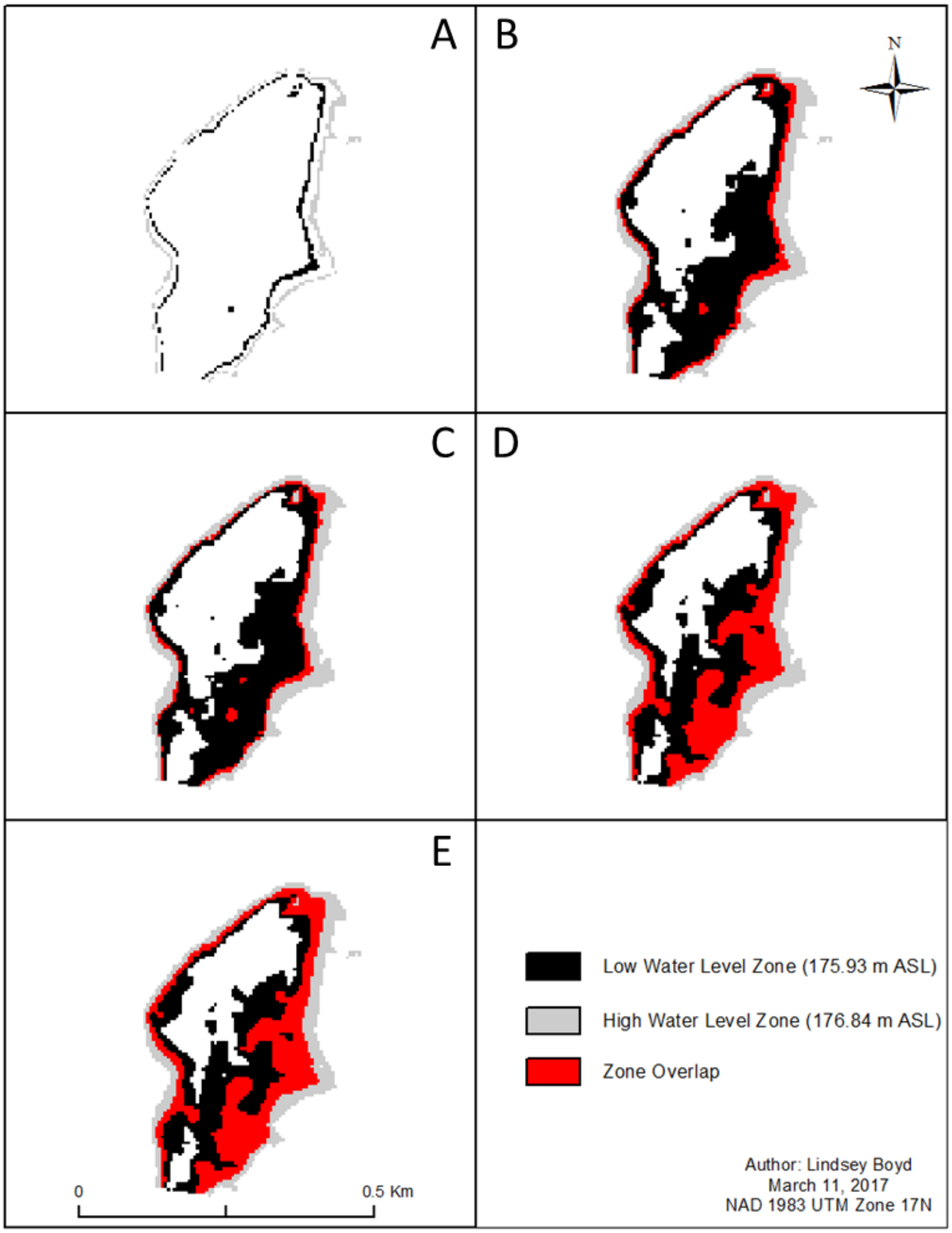

Figure 24. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: Treasure Bay.

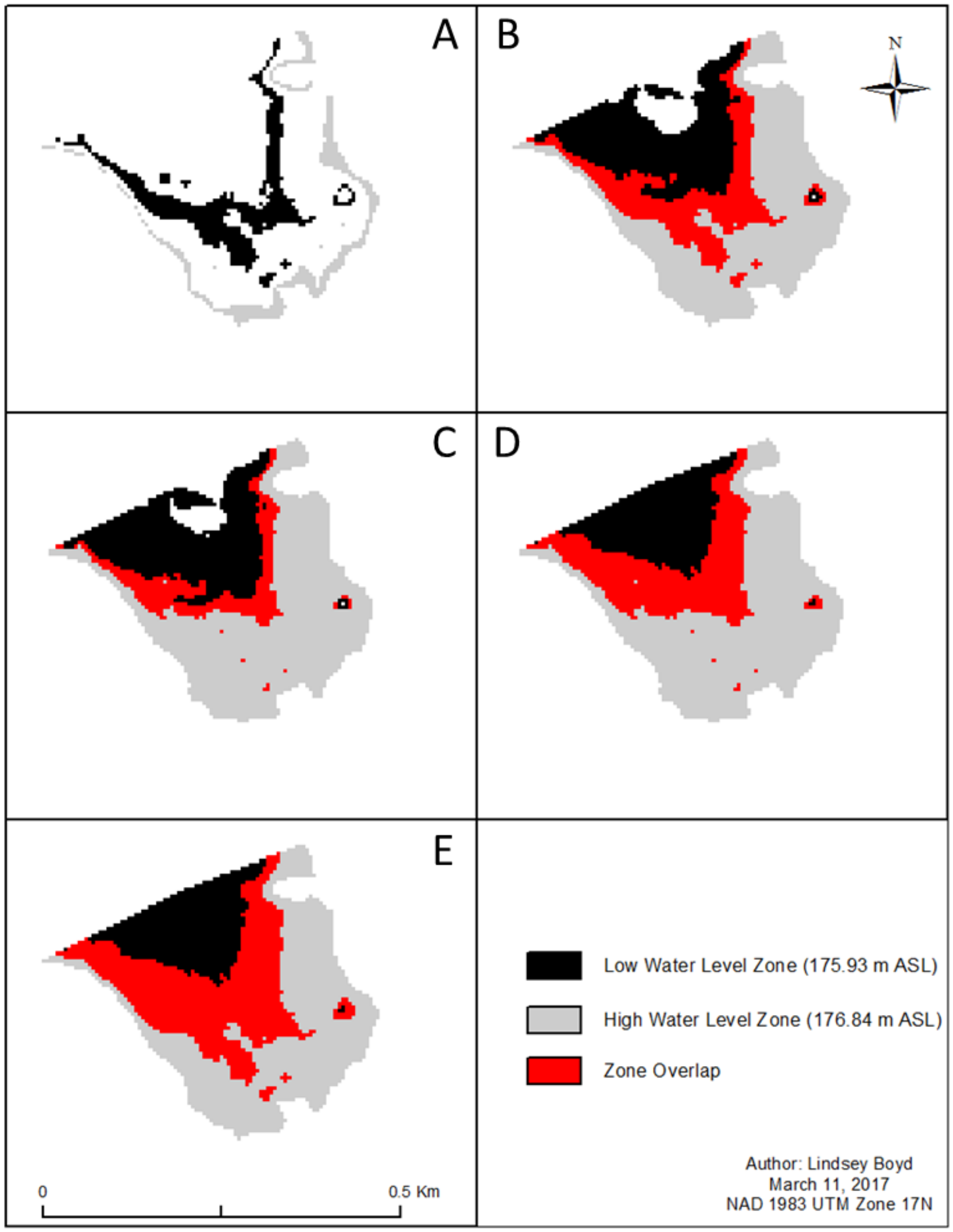

Figure 25. The area and location of the area available to be occupied by (A) Meadow vegetation (B) Emergent vegetation (C) Floating vegetation (D) Submerged Vegetation, and (E) Total wetland habitat under two water-level scenarios. Site: West Bay.