GREAT LAKES COASTAL WETLAND ADAPTIVE CAPACITY

to climate change

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Executive Summary

The Laurentian Great Lakes Coastal Wetlands (GLCWs) support many ecosystem services. For example, GLCWs provide habitat for native species, improve water quality, prevent shoreline erosion, and are of great cultural and social importance. Despite these benefits, GLCWs are increasingly at risk of loss and degradation due to global climate change and its related extremes, such as changing lake water levels, temperatures, and shifts in seasonal norms such as the extent of Great Lakes ice cover. To address these risks, Environment and Climate Change Canada (ECCC) completed a five-year project titled “Assessing and Enhancing the Resilience of Great Lakes Coastal Wetlands (2017-2022)”. The project was funded by the Great Lakes Protection Initiative (GLPI) in support of Canada’s commitments pursuant to the Great Lakes Water Quality Agreement and the Canada-Ontario Agreement on Great Lakes Water Quality and Ecosystem Health. This project assessed the climate change vulnerability of 20 coastal wetlands along the Canadian shoreline of the Laurentian Great Lakes by combining the results of an analysis of Wetland Sensitivity (i.e., wetland vegetation response to climate change) with the results of an analysis of Adaptive Capacity (i.e., measuring the innate ability for a wetland to cope and adapt to climate change). Stakeholders and wetland managers can use the Vulnerability Index generated to identify, develop and prioritize adaptation strategies aimed at enhancing the resilience of GLCW ecological goods and services.

The objective of this technical report was to operationalize the concept of Adaptive Capacity by developing and interpreting the results of a composite indicator (i.e., a weighted combination of variables). Adaptive Capacity cannot be directly measured, therefore to operationalize this metric, measurable natural and anthropogenic variables with a scientifically supported effect on wetland stability and plasticity were selected. A wetland with poor stability or plasticity was proposed to have a lower ability to adapt and cope with climate-related disturbances. In this assessment, eight variables were selected to inform wetland Adaptive Capacity. They included: natural land cover surrounding the wetland (5km), vegetation biodiversity within the wetland, invasive *Phragmites australis* within and surrounding the wetland (5km), capacity for the wetland to migrate lakeward and landward, and finally, the level of protection both within and surrounding the wetland (5km). These variables were grouped into sub-indicator categories according to their influence on wetland Adaptive Capacity and aggregated using a composite indicator methodology. Variables were grouped into four sub-indicator categories; Landscape Condition, Biological Condition, Migration Potential and Protection. The end result of this analysis was a relative numerical score for each of the four sub-indicators as well as relative categorical scores of High, Moderate and Low Adaptive Capacity across the 20 GLCW sites assessed.

The results of this analysis suggest that coastal wetlands located along the shores of Lake Huron and the St. Lawrence River have a higher relative Landscape Condition compared to wetlands along the Huron-Erie Corridor, Lake Erie and Lake Ontario. Similarly, Lake Huron and St. Lawrence River wetlands also had higher relative Biological Condition compared to the Huron-Erie Corridor, Lake Erie and Lake Ontario. These results likely reflect the influence and distribution of human population and

agricultural practices within Ontario. The Huron-Erie corridor wetlands received the overall lowest relative biological condition, which is reflective not only of surrounding agriculture, but also the proportion of invasive *Phragmites australis* present at these sites.

Relative migration potential was determined by nearshore lake bathymetry, wetland hydrogeomorphology and surrounding landscape. Wetlands along Lake Erie and the Huron-Erie Corridor had the highest Migration Potential due to their shallow bathymetry and fine-grained sediments. Conversely, wetlands along Lake Huron and the St. Lawrence River received poor Migration Potential scores. Wetlands with poor migration potential were often either a protected embayment, found in bedrock, or had till derived sediments, all of which would impact their ability to migrate.

The final sub-indicator included in this study was Protection, and of the 20 sites selected, only seven were considered partially protected as defined under the Canadian Protected and Conserved Areas Database (CPCAD) which includes the most up to date data on protected areas. Of those seven wetlands, the five highest-scoring were found along Lake Erie and the St. Lawrence River, with two sites having moderate protection on Lake Huron. The lack of coastal wetland protection illustrates the need for more GLCW securement, which would increase the likelihood of wetland management and increased vigilance under a changing climate.

The final Adaptive Capacity Score represents the theoretical ability for a coastal wetland to adapt to climate change through a relative comparison of the aggregated variables. Categorical scores of High, Moderate and Low can be used to help wetland managers and stakeholders identify coastal wetlands with poor resilience to climate change. Furthermore, reviewing the underlying sub-indicators and variables that contributed to Adaptive Capacity scores can help inform the development and prioritization of adaptation measures.

Four of the six Low-scoring wetland sites were found along the Huron Erie-Corridor. Coastal wetlands in this region were characterized as having a high Migration Potential but scored poorly for Protection, Biological Condition, and Landscape Condition. The other two Low-scoring wetland sites were found on Lake Ontario. These locations scored mid-range for Migration Potential and Biological Condition but poorly for Landscape Condition and Protection.

Wetland sites receiving a Moderate score for wetland Adaptive Capacity were found across all Great Lakes included in this study. Moderately-scoring sites were not clustered and did not reflect the same underlying sub-indicator trends; however, these sites often had at least two poor to mid-range scoring sub-indicators. This result indicates that no single sub-indicator was the driving factor behind Moderate Adaptive Capacity Scores; therefore, the applied management strategies for mitigating climatic disturbance will differ across sites receiving a Moderate score.

Wetland sites receiving High Adaptive Capacity Scores were found across lakes Huron, Erie and Ontario. These sites had three or four high to mid-range scoring sub-indicators. It is important to note, that while these sites scored High relative to other wetlands, they still have sub-indicators for which management action(s) can be applied to improve their Adaptive Capacity. For example, the Adaptive Capacity of wetland sites along Lake Huron that scored High, can be further enhanced by addressing Protection, Migration Potential, and Landscape Condition.

Introduction

The Laurentian Great Lakes coastal wetlands (GLCW) support a variety of ecosystem functions and values including wildlife habitat, water quality improvement, shoreline erosion reduction, recreation and tourism, and cultural and spiritual significance. GLCWs provide necessary habitat for more than 30 species of waterfowl (Prince et al., 1992), more than 30 species of amphibians (Hecnar, 2004), and more than 80 species of fish (Jude & Pappas 1992). GLCWs can attenuate wave action from the lakes, thereby reducing erosion and protecting shorelines (Silander & Hall, 1997). GLCWs improve water quality by retaining and cycling both phosphorus (Mitsch &Reeder, 1992) and nitrogen (Tomaszek et al., 1997) which can accumulate from industrialized agricultural run-off. In addition, GLCWs support the growth of wild rice, the harvest of which is of significant cultural value for Indigenous Peoples (Vennum, 1988). Humans' use of coastal wetlands for recreational activities such as fishing, boating, and hiking are also of great social and economic value.

GLCWs wetlands are dynamic in nature and have evolved to accommodate daily, seasonal, and long-term cycles of lake water level fluctuations (Grabas & Rokitnicki-Wojcik, 2015; Mortsch, 1998). However, the variability and extremes associated with climate change have the potential to alter the structure and function of these wetlands to the extent that they can no longer provide the aforementioned ecosystem services (Gronewold et al., 2013). Climate change will affect precipitation and temperature in the Great Lakes Basin, which in turn can impact GLCW hydrology (Mortsch et al., 2000), as well as increase heat stress, flooding, pollution, soil erosion, and spread of invasive species (Erwin, 2009).

To confront the adverse effects of climate change and address coastal wetland vulnerability, Environment and Climate Change Canada (ECCC) has completed a five-year project titled "Assessing and Enhancing the Resilience of Great Lakes Coastal Wetlands". The project was funded by the Great Lakes Protection Initiative (GLPI) in support of Canada's commitments pursuant to the Great Lakes Water Quality Agreement and the Canada-Ontario Agreement on Great Lakes Water Quality and Ecosystem Health. The anticipated outcomes of the project were:

1. To identify what coastal wetlands are most vulnerable to climate change and why;
2. To develop a collection of adaptation options for enhancing coastal wetland resilience to climate change; and,
3. To improve awareness among Great Lakes rights and stakeholders, and build consensus on priorities for adaptive management action.

GLCWs are highly diverse with respect to their physiography and surrounding land use (Wilcox, 2012). Therefore, in an effort to reflect the complexity and regional diversity of GLCWs, 20 wetland sites were selected for this Vulnerability Assessment (Figure 1). Wetland sites were selected in an effort to capture variation in hydrogeomorphology and anthropogenic disturbance, as well as to reflect ecological, economic, and cultural significance. Although this study intended to cover wetland sites across all of the Great Lakes and their connecting channels, due to limitations in data availability, only wetland sites from lakes Erie, Ontario, St. Clair, Ontario and Huron and the St. Mary's, Detroit, and St. Lawrence rivers were included.

Figure 1. Map of Laurentian Great Lakes Region displaying selected Great Lakes Coastal Wetland Study Sites. 1HIE-Hill Island East, 2ACM-Airport Creek Marsh, 3SBM-South Bay Marsh, 5LCM-Lynde Creek Marsh, 6JSM-Jordan Station Marsh, 7GRM-Grand River Mouth Wetlands, 8SPP-Selkirk Provincial Park, 9LPW-Long Point Wetlands, 10RBY-Rondeau Bay, 11FCK-Fox Creek / Dolson's Creek, 12DRM-Detroit River Marshes, 12SAM-Johnston Bay, 13LSC-Lake St. Clair Marshes, 15BDD-Baie Du Doré, 16HBW-Hay Bay Wetland, 18HBW-Hog Bay Wetland, 19TBY-Treasure Bay, 22WHW-Whiskey Harbour Wetland, 23ACK-Anderson Creek, 28FPT-Francis Point

To characterize the Vulnerability of selected wetlands to climate change, three components were included within the Vulnerability Assessment Framework: Climatic Exposure, Wetland Sensitivity and Adaptive Capacity. Climatic Exposure can be defined as the extent to which a wetland is subjected to significant variations as a result of climate change. Climatic Exposure then informs Wetland Sensitivity which is a measure of how a wetland responds to climate change. Wetland Sensitivity can then be combined with Adaptive Capacity, which is defined as the ability for a wetland to cope and persist under climate change. The combination of Wetland Sensitivity and Adaptive Capacity is used to produce a measure of Wetland Vulnerability. The result is a qualitative value for each wetland that can be used to determine relative Wetland Vulnerability to climate change. Figure 2 describes the relationships between each component within the Vulnerability Assessment Framework.

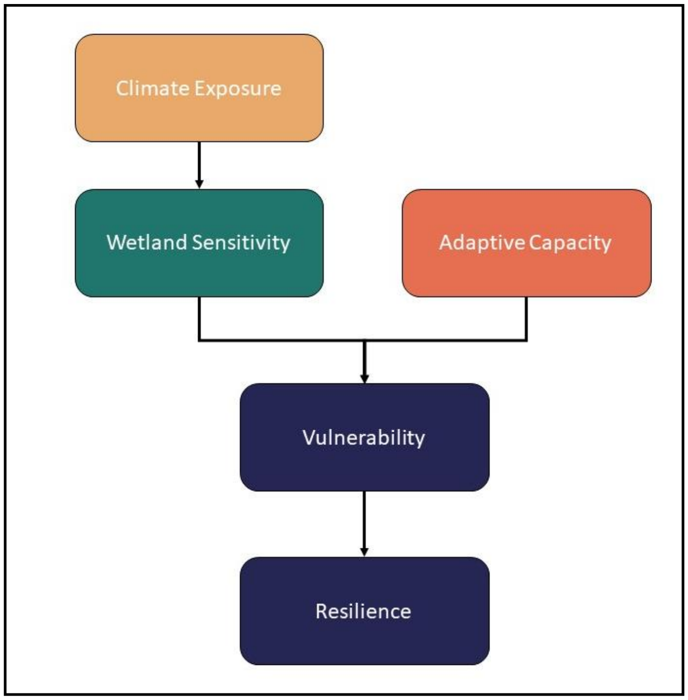


Figure 2. Schematic of Great Lakes Coastal Resilience Assessment Vulnerability Framework

Climatic Exposure was determined by simulating seasonal lake levels towards the end of the 21st century. Lake-level projections were informed by downscaled atmospheric modelling (e.g., temperature, precipitation) across the Great Lakes Region as informed by an ensemble of two Global Climate Models (GCM), one Regional Climate Model (RCM), and two Representative Concentration Pathways (RCPs) for greenhouse gas emissions (4.5 and 8.5) (ECCC, 2022a). Historic and projected water levels were then utilized in combination with wetland site elevation models, vegetation sampling, wave models, and vegetation successional models to estimate past and future change in wetland vegetation class coverage from 1980 to 2010, as well as projected change from 2070 to 2100 (ECCC, 2022b). Forecasted change in wetland class coverage was compared to changes in the hindcast. Change below the 10th percentile between the hindcast and forecast was used to inform the production of the Wetland Sensitivity Indicator (ECCC, 2022c). The remainder of this report will address the methodology, results and application of Adaptive Capacity. More information regarding the final assessment of Wetland Vulnerability (i.e., the combination of Adaptive Capacity with Wetland Sensitivity) can be found in "Assessing the vulnerability of Great Lakes coastal wetlands to climate change (ECCC, 2022d).

Adaptive Capacity

Great Lakes coastal wetlands (GLCWs) provide valuable ecosystem services, yet they continue to be degraded by both climatic and anthropogenic stressors. GLCWs are dynamic by nature and have therefore evolved to accommodate the seasonal and long-term water level fluctuations of the Great Lakes. However, the severity and duration of extreme water level events, coupled with the changes in upland hydrology, seasonal temperatures, and over-lake precipitation associated with climate change may be beyond what GLCWs can adapt to. If GLCWs are unable to adapt and cope with climate change and its related extremes, a loss of ecosystem function, a shift in ecosystem regime, or the loss of the wetland all together may occur. Understanding the degree to which a wetland is capable of adapting and coping with climate change, or wetland Adaptive Capacity, would therefore be of great value for land managers and stakeholders interested in preserving GLCWs. A qualitative measure of relative Adaptive Capacity in selected GLCWs can assist in the identification of priority management wetland sites as well as inform strategies aimed at improving the ability for wetlands to adapt to future climate disturbance.

Adaptive Capacity

The ability of a wetland in its current state to adjust and maintain its ecological regime under changing climate conditions, including variability and extremes.

The term Adaptive Capacity has been used within both the social and ecological sciences and has therefore incurred several definitions with varying degrees of specificity. In ecological studies, the Intergovernmental Panel on Climate Change's (IPCCs) definition of Adaptive Capacity is commonly referenced. It states: "*Adaptive capacity is defined as the ability of a system to adjust to climate change to moderate potential damages, to take advantage of opportunities, or to cope with the consequences*" (IPCC, 2007). This definition provides a broad concept of Adaptive Capacity that can be used to address a variety of human and natural systems at various levels of organization. However, due to this broad applicability, this study required modification to the IPCC's definition in order to increase specificity to the ecosystem of study, create clear distinction between Adaptive Capacity and other Vulnerability Framework terminology, and to provide criteria for operationalization. The definition of Adaptive Capacity used in this study is; "*The ability of a wetland in its current state to adjust and maintain its ecological regime under changing climate conditions, including variability and extremes.*"

There are two key variations between the definition used in this assessment and the original definition provided by the IPCC. The first modification is the addition of a contemporary time frame. By indicating that Adaptive Capacity should be measured using variables that represent the *'current state'* of the wetland (i.e., predisposing conditions and natural characteristics), this definition avoids the potentially misleading comparison of wetlands in an undisturbed historic state or the comparison of the estimated future change in wetlands. The second modification is the inclusion of the term '*ecological regime'* which allows for a clear distinction between 'Adaptive Capacity' and 'Resilience' (Angeler et al. 2019). Wetland resilience represents the ability of a wetland to undergo a change in regime where the wetland's structure, function, physical properties and ecological services change (Holling, 1973; Folke et al., 2004; Guttal and Jayaprakash, 2008). Alternatively, Adaptive Capacity is the ability of a wetland to cope, and adjust to disturbance events while maintaining its current regime and related ecosystem services (Angeler et al., 2019).

Adaptive Capacity Variable Criteria

Adaptive Capacity, as defined by this study, suggests both a time frame, and that an adaptive wetland will mitigate climate change-related extremes and variability. However, even with a more targeted definition, Adaptive Capacity is still a theoretical concept and cannot be directly or empirically measured. Previous studies encountering this challenge have suggested the use of surrogate variables that can both be directly evaluated and have a supported influence on ecosystem stability to be used to operationalize Adaptive Capacity (Angeler et al. 2019). In order to select appropriate variables and provide an informative qualitative Adaptive Capacity result to GLCW stakeholders and land managers, three variable criteria were implemented in this assessment. The three variable criteria are; (1) variable must reflect the current state of the wetland; (2) variables must be measured using the same method and effort across all sites; (3) variable must have a scientifically supported influence on wetland stability or plasticity.

Adaptive Capacity Variable Selection Criteria

1. Variable must reflect the current state of the wetland.
2. Variables must be measured using the same method and effort across all sites
3. Variable must have a scientifically supported influence on wetland stability or plasticity.

The first variable selection criterion corresponds directly with the definition of Wetland Adaptive Capacity used in this study. This criterion requires that any variable included within this assessment must reflect the current conditions or health of the wetland, rather than conditions that have existed in the past or may exist in the future. This criterion also helps distinguish appropriate measurable variables in the Adaptive Capacity analysis from variables to be included within the Wetland Sensitivity component of the Vulnerability Framework, which represents an estimation of future wetland change.

The second variable criterion ensures that the same sampling effort and methods are consistent across all sites selected for analysis. Twenty GLCW sites were selected for this analysis spanning the shores of Lake Huron, the Huron-Erie Corridor, Lake Erie, Lake Ontario and the St. Lawrence River. Therefore, it is important to ensure that Adaptive Capacity results reflect true variation across sites rather than variation in sampling effort, remote sensing product spatial resolution, or remote sensing classification accuracy.

The final criterion required for a variable to be included in the Adaptive Capacity analysis, is that each variable must have a scientifically supported influence on wetland stability and or plasticity. Adaptive Capacity cannot be directly measured, nor can it be verified within the limits of the Vulnerability Framework until the year 2100. However, it can be assumed that a wetland with poor stability or plasticity would have limited Adaptive Capacity. Therefore, variables with a scientifically supported influence on wetland stability or plasticity can be assumed to influence the ability for a wetland to adapt and cope with climate change.

Adaptive Capacity Variable Selection and Data Sources

Adaptive Capacity cannot be directly measured, only inferred based on contemporary surrogate variables with a scientifically supported influence on wetland stability or plasticity. Eight variables were selected to measure Adaptive Capacity, and these variables were then grouped into four sub-indicators: Landscape Condition, Biological Condition, Wetland Migration Potential, and Protection. In order to be included within this analysis each variable must meet all three variable criteria discussed in the previous section. This section will provide the year and source of variable data, as well as an explanation of how each variable selected will influence wetland stability and or wetland plasticity and which sub-indicator it contributes to. Figure 3 depicts the relationship between variables, sub-indicators, and the directionality and influence of each sub-indicator on wetland stability and ultimately Adaptive Capacity.

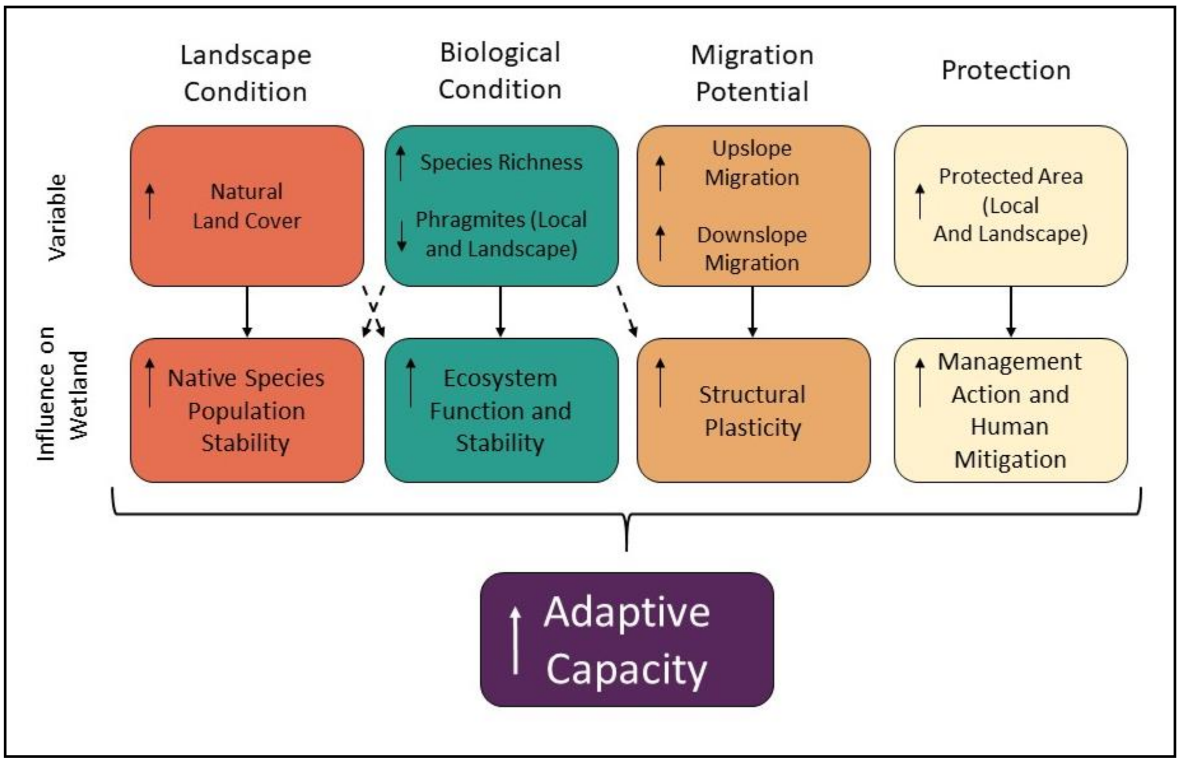


Figure 3. Schematic displaying the relationship between variables, sub-indicators, and the directionality of influence of variables on Adaptive Capacity

Given the complexity of GLCWs and their potential suite of responses to disturbance events, the list of variables included within this analysis cannot be considered exhaustive. However, the variables selected cover a broad range of influences, and encompass the characteristics of the wetland and surrounding environment that are likely to have the greatest impact on Adaptive Capacity. Several variables that were considered for this analysis but could not be included due to violation of the aforementioned variable criteria include: water quality, wetland connectivity, and sediment dynamics. The following section titled “Adaptive Capacity

Variable Rationale" provides an explanation for how each variable selected for this analysis meets all three variable selection criteria.

Landscape Condition

> **Sub-indicator:** Landscape Condition
> **Variable**: Natural Land Cover within 5km of wetland
> **Data Source(s)**: Annual Crop Inventory Data and Crop Classification Database (AAFC, 2018)
> **Influence on Wetland**: Wetlands with poor landscape condition will have less stable or adaptable wildlife populations due to the influence of landscape condition on wildlife population size, persistence and genetic diversity.

Landscape condition has a direct effect on wildlife reproduction, survival, and movement. Therefore, poor landscape condition can reduce wetland species population size, population persistence, and genetic diversity leading to a loss of wetland community structure stability. If the landscape condition surrounding a wetland cannot provide adequate suitable habitat to support local species, the wetland will be less adaptable to climate change and its related disturbances (Schloss et al., 2012).

Empirical studies show that wildlife abundances are lower in landscapes with less natural land cover, where natural land cover is a proxy for habitat area (Martin and Fahrig). This is likely because landscapes with less natural cover have fewer resources available for wildlife, which results in smaller population sizes. Wildlife that inhabit landscapes with less habitat area also have lower body condition, breeding, foraging, dispersal and survival rates (Matthysen & Currie, 1996; Yahner & Mahan, 1999; Hinam & Clair, 2008; Janin et al., 2011). These effects at the individual level contribute to population size declines and ultimately higher local species extinctions. Finally, less natural habitat in the surrounding landscape decreases the pool of potential immigrants that can reach another habitat patch by chance, thereby increasing the risk of local species extinction (Fahrig et al., in press).

Given the direct influence of landscape condition on wildlife, it is possible that landscape condition can influence a wetland ecosystem's ability to adapt to climate change through the effects of landscape structure on wildlife population size, population persistence and genetic variability. First, a higher local population size (i.e., more individuals of any wetland species) increases the probability of population persistence following a disturbance event. Moreover, greater population size increases the number of potential immigrants moving in a landscape, thereby increasing the probability of species' range expansion into suitable climatic regimes. Secondly, longer population persistence increases the amount of time available for a wetland community to adjust species assemblages, phenology, behavior, and ultimately species range expansion/shifts in response to climate change. Therefore, a larger population "buys more time" for adaptive short-term and long-term responses of wildlife to climate change. Lastly, a greater population size generally results in a greater gene pool, which increases the probability that a population contains genetic diversity allowing for adaptation to changing environmental conditions (Lande, 1988; Campbell et al., 2017).

In addition to each independent landscape condition's influence on wetland species (i.e., population size, population persistence, and genetic diversity), there are also interactive effects. Population size is critical for generating novel mutations, which create raw material allowing evolutionary adaptation to climate change (Waldvogel et al., 2020). Therefore, it is possible for a positive feedback loop to occur where population size will increase genetic diversity, which will further increase population size and support a greater range of genetic response to climate change. Given the direct influence of landscape condition on wildlife population size, population persistence and genetic diversity, as well as supporting interactive effects between population size and genetic diversity, it can be concluded that landscape condition influences the stability of wetland species populations and their adaptability. Therefore, landscape condition is an important factor to consider when assessing GLCWs Adaptive Capacity.

Natural Land Cover Variable Rationale

The Landscape Condition sub-indicator was composed of only one variable: the area of natural land cover surrounding each wetland site of interest. A landscape with more natural cover (e.g., forests and wetlands) represents a landscape with higher Landscape Condition for wetland-dependent plant and animal species. The natural area surrounding a wetland has a positive effect on the population abundance of wetland wildlife based on a global meta-analysis across several taxonomic groups (Quesnelle et al., 2015) – from moose (*Alces alces*) to western chorus frog (*Pseudacris triseriata*). This meta-analysis included GLCWs and is further supported by recent empirical evidence in the basin (Millar & Blouin-Demers, 2012; Tozer et al., 2020; Van Der Merwi et al., 2016; Patenaude et al., 2015). The positive effect of surrounding wetland area on wildlife populations is most likely due to higher food and nesting site availability in landscapes with more wetland.

Empirical evidence in the Great Lakes basin shows that forest or grassland cover has a strong positive effect on marsh birds (Tozer et al., 2020), amphibians (Hecnar & M'Closkey, 1998) and reptiles (Markle & Chow-Fraser, 2018). This is likely because forest and/or grassland cover is correlated with both water quality (e.g., Crosbie & Chow-Fraser, 1999: Lougheed et al., 2001), and terrestrial habitat types required by several wetland species. In addition, most amphibians and reptiles in the Great Lakes basin require terrestrial habitats to complete their life-cycle.

A global meta-analysis of the effects of landscape context on wetland wildlife populations found that the area of forest in the surrounding landscape had a larger effect on amphibians than the area of wetland, independent of terrestrial habitat requirements (Quesnelle et al., 2015). Quesnelle et al. hypothesized three reasons for the greater effect on amphibian population abundances of forest cover than wetland cover: 1) increased landscape complementation, or access to complementary habitat that is limiting to a species (Dunning et al., 1992); 2) decreased dispersal mortality; and, 3) increased local habitat quality (e.g. patch-scale edge effects; Chase et al., 2020).

Natural cover is typically negatively correlated with the area of developed land (agriculture and urban land use) in the Great Lakes basin (Crosbie & Chow-Fraser, 1999). A correlation analysis comparing natural land cover and agricultural cover for the 20 GLCWs used in this analysis found they were significantly negatively correlated (r = 0.8). Therefore, natural land cover can be considered the inverse measure of agricultural land use (Uzarski et al., 2004). The effect of agriculture on wetland population abundances is mixed (positive, neutral and negative), depending on agricultural intensity (Saumure et al., 2007; Collins & Fahrig, 2017), and is generally weaker in magnitude than the positive effects of wetland and forest cover (Houlahan & Findlay, 2004; Quesnelle et al., 2015).

The effects of impervious cover (roads and urbanization) are generally negative on wildlife populations (Eigenbrod et al., 2008). In studies designed to separate the independent effects of natural cover and impervious cover, natural cover in the surrounding landscape was found to have an equal or greater positive effect on wetland vegetation, benthic macroinvertebrate and amphibian communities than the negative effect of impervious cover (Eigenbrod et al., 2008; Patenaude et al., 2015). In addition, few GLCW sites selected for this analysis were surrounded by urbanized areas, which resulted in low applicability to sites surrounded by agriculture and natural land cover types. Therefore, natural land cover was selected to represent Landscape Condition due to its consistent positive effects on wildlife and applicability across GLCW study sites selected.

Natural Land Cover – Data Source

Data utilized for determining the proportion of natural land use surrounding wetlands was acquired from the Annual Crop Inventory Data and Crop Classification Database (Agriculture and Agri-Food Canada [AAFC], 2018). The Annual Crop Inventory classification utilizes a decision tree-based methodology using multi-temporal data from optical (Landsat-8, Sentinel-2, Gaofen-1) and radar (RADARSAT-2) satellite imagery. The resulting product is a 30m resolution classification with 75.88% classification accuracies for non-agricultural land cover.

Biological Condition

> **Sub-indicator:** Biological Condition
> **Variables**: *Phragmites australis* within wetland, *Phragmites australis* surrounding wetland, vegetation species richness
> **Data Source(s)**: Michigan Tech Research Institute Great Lakes Coastal Wetland and Land Use Map (2015), and Canadian Wildlife Service, Environment and Climate Change Canada (ECCC-CWS; 2018-19)
> **Influence on Wetland**: Wetlands with poor biological condition will have a reduced ecosystem stability and native species population stability.

The Biological Condition sub-indicator is comprised of three variables; *Phragmites australis* cover within the wetland, *Phragmites australis* cover within five kilometers of the wetland, and the number of vegetation species that exist within the wetland. These three variables were selected based on the Diversity – Stability hypothesis which argues that wetlands with high biodiversity have enhanced ecosystem stability (Mcnaughton 1977).

Invasive *Phragmites* was included as a measure of wetland biological condition because their monocultures are known to reduce wetland biodiversity, replace native flora, and are able to expand into new habitat after an anthropogenic or climatic disturbance event faster than native vegetation (Bourgeau-Chavez et al. 2015). Therefore, given an environmental disturbance event, a higher proportion of *Phragmites* within and surrounding a wetland may decrease native wildlife breeding, foraging and nesting opportunities resulting in a higher risk of ecosystem regime shift, and therefore a lower Adaptive Capacity. *Phragmites* coverage was estimated using the Michigan Tech Research Institute Great Lakes Coastal Wetland and Land Use Map (2015), which was classified at a 30m resolution using multi-temporal Landsat Imagery (Bourgeau-Chavez et al. 2015).

Vegetation species richness (a measure of biodiversity) was included as a variable for the Biological Condition sub-indicator because wetlands with high biodiversity are more likely to be stable following a disturbance event, and therefore more likely to continue primary producer productivity which supports wildlife. Vegetation species richness data was provided by the Canadian Wildlife Service, Environment and Climate Change Canada (ECCC-CWS); data was collected from 2018-2019.

Phragmites australis (Within and Surrounding Wetland) Variable Rationale

Phragmites australis subsp. *australis* (*Phragmites*), a perennial grass species native to Eurasia and Africa, was first introduced to the North American continent during the 1800s. Since this first introduction, *Phragmites* have expanded in range across Eastern North American wetlands where their presence has been shown to reduce native flora and fauna population and diversity (Tulbure & Johnston, 2010). Due to this observed loss of biodiversity, and the well-supported Diversity – Stability hypothesis (Mcnaughton, 1977), this study considers the potential for *Phragmites* invasion to negatively impact a wetland's ability to cope and persist under the variability and extremes associated with climate change.

As an opportunistic species, *Phragmites* can expand into new habitats faster than native flora thus allowing them to take advantage of wetland disturbances such as fluctuating water levels and the resulting exposure of unvegetated substrate (Pengra et al., 2007). In addition, human development such as land use change, modifications to hydrology, and new road construction may also act as contributing factors to *Phragmites* expansion (Johnston et al., 2008; King et al., 2007).

Once established within a wetland, *Phragmites* monocultures can displace native plant species thereby degrading habitat and reducing populations and diversity of wetland-obligate fauna such as birds, amphibians, reptiles and fish (Bourgeau-Chavez et al., 2013; Keller, 2000; Meyerson et al., 2009). Avian species richness is impacted by the presence of monotypic *Phragmites* stands, where birds, such as ducks, herons, egrets and sandpipers, which rely on cattail dominated marshes, are greatly reduced or found to be missing entirely (Benoit & Askins, 1999). This is likely because the dense homogenous stands provide low quality breeding habitat and inhibit movement and foraging behavior within the wetland. In addition, muskrats, which provide the service of improving marsh bird habitat by creating openings in dense vegetation, do not typically consume mature *Phragmites* (Benoit & Askins, 1999).

Herpetofauna have also experienced population declines in response to the loss of breeding habitat and restriction of movement caused by invasive *Phragmites* stands. At Long Point Provincial Park in Ontario, one of the 20 GLCW sites selected for this study, Fowler's Toad (*Bufo fowleri*) populations experienced an unprecedented decline in population over a 20-year time span. This decline is thought to be a direct result of the loss of shallow aquatic breeding habitat to monotypic *Phragmites* stands (Greenberg & Green, 2013). Similarly, turtles are also at risk of losing appropriate nesting sites which rely on adequate moisture and temperature ranges to ensure proper embryonic development and hatchling survival. *Phragmites* expansion into turtle nesting sites can reduce turtle reproductive success by limiting site availability, engulfing existing nests, or creating shade that reduces temperature and therefore nest viability (Bolton & Brooks, 2010). In addition, dense stands of *Phragmites* can also impede movement of herpetofauna throughout the wetland and reduce available sunny areas needed for thermoregulation (Mifsud, 2014).

Due to their ability to expand into shallow water habitats, *Phragmites* invasion also impacts fish populations. *Phragmites* stands have a drying effect on shallow water wetlands by increasing accumulation of sediment and debris which causes the surface of the shallow wetland to become more elevated and reduces water inundation. The lack of shallow water and reduced frequency of flooding results in the loss of fish spawning habitat and refuge for juvenile fish (Able & Hagan, 2003). In addition, the increased litter and sediment accumulation may also lead to the loss of first-order streams which are commonly used as fish refuge (Meyerson et al., 2009).

Loss of biodiversity due to *Phragmites* invasion reduces the probability that a species that may be able to adapt or accommodate climatic stressors will be present within the wetland. This is based on the Diversity – Stability hypothesis which argues that wetlands with high biodiversity have an enhanced ecosystem stability and functionality (Mcnaughton, 1977). This is due to the assumption that there is an increased probability that a wetland with high biodiversity will contain a species that is able to adapt to the wetland change (i.e., disturbance-tolerant) compared to a wetland with low biodiversity (Mcnaughton, 1977).Therefore, the presence of *Phragmites* surrounding a wetland may limit a wetland's ability to moderate impacts caused by climate change and its associated extremes. The potential for this limitation is proportional to the surrounding coverage of *Phragmites*, where a higher percent coverage will likely increase the probability of wetland invasion and therefore result in a reduction of wetland biodiversity.

It is possible that implementation of control methods such as rolling, burning and pesticide application can be applied to eradicate *Phragmites* and potentially increase native biodiversity. However, these control measures are costly, require repeated applications, and are not universally applied. The variability of control measures and monitoring is accounted for within the Protection sub-indicator rather than Biological Condition.

Phragmites australis Data Source

Michigan Technological Research Institute (MTRI; Bourgeau-Chavez et al., 2015) was used to inform *Phragmites australis* subsp. australis (*Phragmites*) coverage. This dataset covers the land within ten kilometres of the Great Lakes shoreline and connective waterways at a resolution of 30m. Classification was performed with a random forest classification using multi-temporal optical (Landsat-5) and synthetic aperture radar (SAR; PALSAR). Primarily, satellite imagery acquired in the spring, summer and fall of 2010 was used, however, additional imagery from 2007-2011 was used to supplement gaps in the dataset. Field validation data were collected from 2010 to 2011. Invasive *Phragmites* and wetlands were classified with 89% and 70% producer's accuracy across all lakes (Bourgeau-Chavez et al., 2015).

Vegetation Species Richness

Biodiversity is an important factor influencing wetland stability, and should therefore be considered within the Adaptive Capacity analysis. An ecosystem with high biodiversity is more likely to maintain multiple critical ecosystem functions, such as primary production, nutrient cycling and decomposition, both during and after a disturbance event (Cardinale et al., 2012; ,Duffy et al., 2017; Zavaleta et al., 2010). Biodiversity can be assessed from a genomic to ecosystem functional trait level; however, given the spatial range and diversity of GLCWs, vegetation species richness was selected as a representation of wetland biodiversity. Vegetation species richness increases the stability of an ecosystem by influencing two components: resistance and temporal variability. Resistance is the degree to which an ecosystem can persist during a disturbance event (Mccann, 2000), while temporal variability refers to the variation in an ecosystem function through time.

Biodiversity can improve ecosystem stability by providing resistance to climatic disturbance events (Isbell et al. 2015). One explanation for this is the biodiversity-ecosystem function hypothesis which argues that ecosystem function increases linearly with biodiversity. This relationship between biodiversity and ecosystem function results in species-rich ecosystems generally having a higher mean biomass or nitrogen fixation (Reich et al., 2012), which may create a short-term buffer against climate-related extremes. This phenomenon has been observed in grassland ecosystems where a higher biodiversity increased the resistance of ecosystem productivity across a wide range of climate events, including wet or dry, moderate or extreme, and brief or prolonged (Isbell et al., 2015).

Vegetation species richness can improve ecosystem temporal stability by reducing primary production variability over time through the diversity-stability hypothesis (Craven et al., 2018). A higher vegetation species richness improves the probability that primary producers with differing responses to environmental disturbances exist within the ecosystem. This differing response allows for the decrease in productivity of some species to be compensated by the increased productivity of other species, allowing for improved stability at the primary producer trophic level.

Vegetation species richness, through both short-term (resistance) and long-term (temporal stability), can contribute to ecosystem stability. These contributions can be explained by the biodiversity-ecosystem function hypothesis and diversity-stability hypothesis, respectively. It can therefore be inferred that a wetland with a higher vegetation species richness will generally have a greater biomass output that can buffer or resist climatic disturbance. In addition, this higher species richness will also have a higher probability of adapting to climate change and continuing to provide ecosystem services. Therefore, wetland ecosystems with higher vegetation species richness will have a higher Adaptive Capacity than wetlands with lower species richness.

Vegetation Species Richness - Data Collection

Vegetation species richness was informed by vegetation surveys conducted as part of Environment and Climate Change Canada's broader climate change vulnerability assessment. Vegetation surveys for each wetland occurred along 15 – 20 transects per year that were established perpendicular to elevation contours and spanned both aquatic and terrestrial environments (Figure 4). In addition to basin morphology, the orientation and length of transects were pre-determined to capture points in each vegetation community present, as determined through contemporary aerial imagery and Ecological Land Classification (Lee et al.,1998). Transects were, on average, 125m in length with a minimum length of 40m and a maximum length of 575m. Each year, 150 – 200 pre-determined quadrat sampling locations were distributed evenly across the cumulative length of all transects within a site. Between years, transects were displaced by at least 5m to minimize artifact sampling. The surveys were performed between July and September 2018 and 2019 when plant growth had peaked but before fall senescence had commenced. In response to stakeholder interest, Frances Point Marsh (27FPT) and Hill Island East (1HIE) were added to the study late in 2018 and could only be sampled once in 2019.

At each quadrat sampling location, information on the occurrence of wetland vegetation was collected using an approach similar to that of Grabas & Rokitnicki-Wojcik (2015). At each location, a 1.0m x 0.5m quadrat was placed, with the long edge perpendicular to the elevation gradient. In each quadrat vascular plants were identified to species following Newmaster et al. (1997) and Crow & Helquist (2000). Genus-level data was recorded for taxa difficult to identify to species (e.g., sterile *Cyperaceae, Characeae*). The maximum vegetation species richness sampled between the two years was used to represent richness.

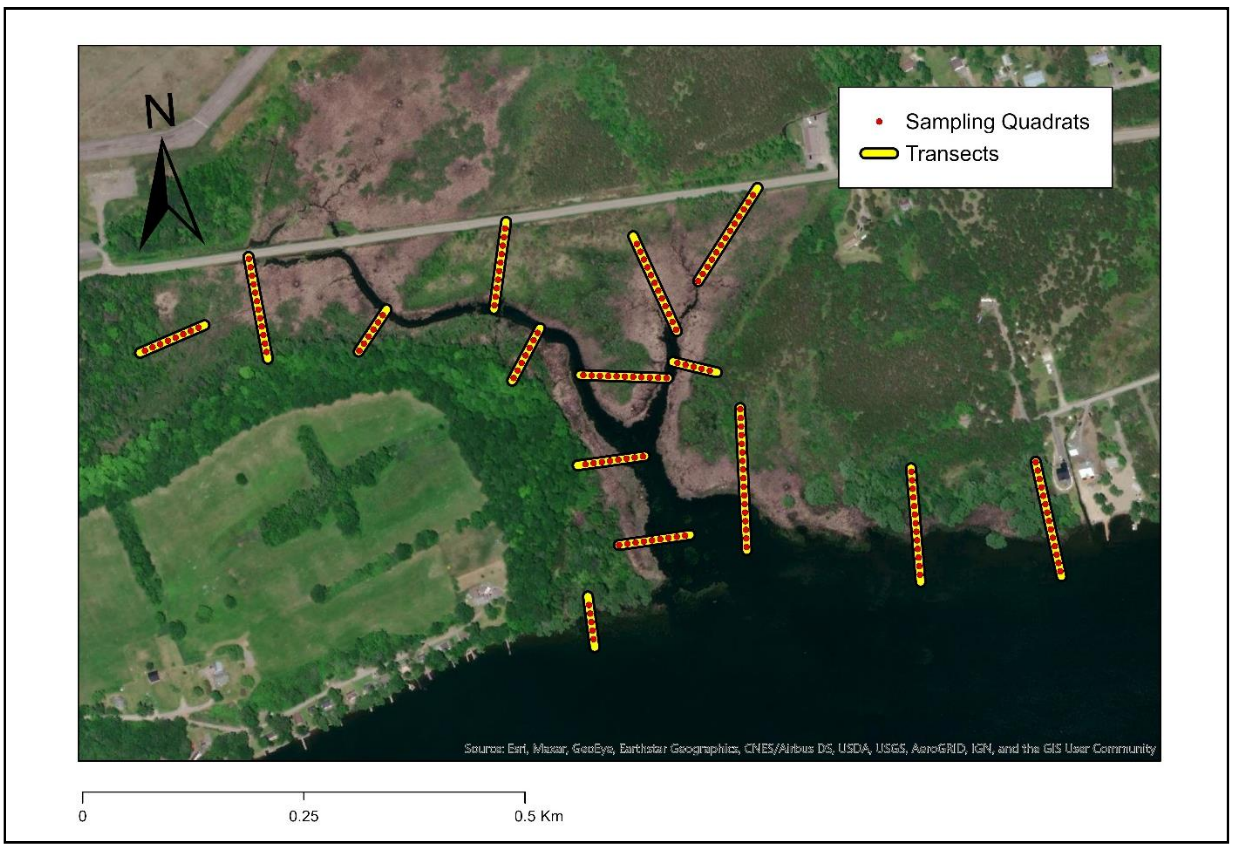


Figure 4. Airport Creek Vegetation sampling transects and quadrats

Prior to inclusion within the Adaptive Capacity analysis, maximum vegetation species richness data was evaluated to ensure that each wetland site received equal sampling effort and accurately reflected the vegetation species richness present. To determine if a relationship between sampling effort and observed species richness was present, a general linear model with total number of quadrats sampled as a predictor of observed species richness at each site was used. This model indicated no significant increase in richness among sites based on sampling effort ($p = 0.822$). To determine if vegetation species richness measured at each site accurately represented completeness (i.e., increased sampling effort would not increase species richness) sample coverage was measured (Chao & Jost, 2012). Results indicated sample coverage was high and consistent across all sites (mean = 0.95 ± 0.02). Wetland sites selected for this study exhibit a range in elevation, therefore regression analysis was used to determine if range in elevation impacted vegetation species richness. It was determined that elevation had no significant effect on observed species richness ($p = 0.93$). The results of these three statistical tests indicate that sampling effort conducted across sites was equal and sufficient to represent a complete measure of species richness which was not affected by site specific variation in elevation. Given these results, the sampling method applied and the resulting vegetation species richness data can be considered appropriate for use within the Adaptive Capacity analysis.

Wetland Migration Potential

Sub-indicator: Wetland Migration Potential
Variables: Upslope migration potential, downslope migration potential
Data Source(s): Water level projections provided by Fisheries and Oceans Canada (2020), IUGLS (2012) and ECCC (Seglenieks & Temgoua, 2021). Digital terrain models (DTM) were prepared by ECCC-NHS (Maranda et al., unpublished).
Influence on Wetland: Wetlands with poor biological condition will have a reduced ecosystem stability and native species population stability.

Wetlands are highly dynamic and often adapt to changes in water level by either migrating downslope or upslope, which is an important aspect to consider regarding wetland adaptability to climate change. Wetland plants have evolved adaptations for specific hydrological conditions (Wetzel, 2001), and the range of hydrologic conditions an individual species can tolerate is referred to as its hydrologic niche (Booth & Loheide, 2012; Silvertown et al., 1999). The organization of wetland vegetation communities across a vertical gradient illustrates shared hydrologic niches amongst species, as well as differences in fitness through spatial and temporal changes in water depth (Cronk & Fennessy, 2001; Casanova & Brock, 2000; Deane et al., 2017; **Error! Reference source not found.**). GLCWs experience upslope and/or downslope shifts where vegetation communities migrate vertically in order to adapt to lake-level variability (Grabas et al., 2019; Grabas & Rokitnicki-Wojcik, 2015; Keddy & Reznicek, 1986; Keough et al., 1999; Smit & Wandel, 2006; Wilcox & Meeker, 1991; Wilcox et al., 2003; Wilcox, 2012). For example, during sustained high water-level periods, trees, shrubs and other woody vegetation at a low elevation may drown, permitting the landward expansion of aquatic vegetation. When water levels recede, so too will aquatic vegetation, allowing the regrowth of meadow and emergent marsh species from exposed seedbanks (Keddy & Reznicek, 1986). If water levels continue to recede and remain low, woody vegetation will recolonize lakeward, and the most competitive species will dominate within the meadow and emergent marsh communities (Wilcox et al., 2002).

Climate change is projected to increase both seasonal and annual water-level fluctuations within the Great Lakes, which could shift the hydrologic niches of individual plant species. Should the vertical migration of a wetland in response to higher lake-level variability be impeded (e.g., by shoreline structures, roads, urban development, etc.) local extirpations or the loss of an entire community may occur. This would constitute a regime shift, or, a persistent change in the structure, function and feedbacks of the wetland ecosystem (Angeler et al., 2019). The loss of an individual species or community could also decrease overall biodiversity, compromising wetland function and stability (see Biological Condition sub-indicator). This may diminish the ability of a wetland to cope or persist through additional climatic changes (e.g., air and surface water temperatures), as remaining populations and communities of species may lack sufficient diversity and/or redundancy in the traits required to formulate functional responses (e.g., phenotypic plasticity, seedbank potential, dispersal ability; Bengtsson et al., 2003; Folke et al., 2004; Glick et al., 2009; Kareiva, 2008).

Wetland Migration Potential - Data Sources

Determining wetland migration potential (upslope and downslope) required the combined use of multiple datasets. Historic water-level maximums and minimums, mid-century water-level projections, and mid-to-late century water-level projections for the Great Lakes were provided by Fisheries and Oceans Canada (2020), IUGLS (2012) and ECCC (Seglenieks & Temgoua, 2021), respectively. Site-specific digital terrain models (DTM) were prepared by the National Hydrological Services (NHS; ECCC). A single appropriate land classification dataset covering all wetland sites and their projected upslope migration did not exist at the time of this analysis. Therefore, three federal and provincial datasets were used in combination for this analysis. These datasets included: Ecological Land Classifications performed by the Canadian Wildlife Service – Ontario Region (CWS-ELC), The Great Lakes Shoreline Ecosystem Inventory Version 1.0 (GLSE; OMNRF, 2019) and The Southern Ontario Land Resource Information System Version 3.0 (SOLRIS; OMNRF, 2019). Table 1 summarizes the land cover datasets utilized.

Table 1. Table displaying land cover datasets used according to selected wetland sites

Unique Site ID	Wetland Name	Land Cover Dataset
1HIE	Hill Island East	SOLRIS
2ACM	Airport Creek Marsh	CWS-ELC
3SBM	South Bay Marsh	CWS-ELC

5LCM	Lynde Creek Marsh	CWS-ELC
6JSM	Jordan Station Marsh	CWS-ELC
7GRM	Grand River Mouth Wetlands	GLSE
8SPP	Selkirk Provincial Park	GLSE
9LPW	Long Point Wetlands	GLSE
10RBY	Rondeau Bay	GLSE
11FCK	Fox Creek/ Dolson's Creek	GLSE
12DRM	Detroit River Marshes	CWS-ELC
13LSC	Lake St. Clair Marshes	CWS-ELC
14SAM	Johnston Bay	CWS-ELC
15BDD	Baie Du Doré	SOLRIS
16HBW	Hay Bay Wetland	CWS-ELC
18HBW	Hog Bay Wetland	SOLRIS
19TBY	Treasure Bay	CWS-ELC
22WHW	Whiskey Harbour Wetland	CWS-ELC
23ACK	Anderson Creek Marsh	CWS-ELC
27FPT	Francis Point	CWS-ELC

Protection

In this study, GLCWs afforded full or partial protection (both within and surrounding the wetland) are expected to have a higher Adaptive Capacity than those that remain unprotected. Protected Area is defined as a "geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (International Union for Conservation of Nature [IUCN], 2013). Other Adaptive Capacity sub-indicators are derived from the natural ecosystem, but wetlands can also benefit from ecosystem management and associated laws, policies, knowledge, staff and financial resources (Barr, 2020; Kettunen & ten Brink, 2013; Stolton, et al., 2015). Consequently, relative to unmanaged areas, Canadian Protected Areas should have a greater capacity to conserve diversity, both biological and geological, and offer protection against environmental degradation and non-climatic stressors.

The conservation of wetland biological and geological diversity through Protected Area establishment and management can be considered a precautionary approach for managing the Adaptive Capacity of GLCWs. To retain ecosystem functionality, it is recognized that a minimum composition of species is required, however this threshold is seldom known (Folke et al., 2004). Therefore, conserving species compositions as they currently exist may best prepare ecosystems future climatic conditions (Angeler et al., 2019; Nyström & Folke, 2001). In addition, the conservation of geodiversity is receiving increased recognition as a strategy for maintaining biodiversity by supporting a diverse range of habitats for current and future species assemblages under various climatic conditions (Anderson & Ferree, 2010; Beier & Brost, 2010; Crofts et al., 2020).

Providing protection against environmental degradation is central to Protected Area establishment, and mitigating the impacts of non-climatic stressors is an approach frequently adopted by land managers for conserving biodiversity in the face of climate change (Heller & Zavaleta, 2009; Thomas & Gillingham, 2015). Landscapes outside of Protected Areas can be damaging to the survival and reproduction of many species due to human developmental pressures including habitat loss, habitat fragmentation, invasive species and water quality impairment. These pressures limit species migration and gene flow, and by extrapolation, the Adaptive Capacity of coastal wetlands (Heller & Zavaleta, 2009). Therefore, by mitigating non-climatic stressors, Protected Areas should leave natural communities better equipped to adapt to climate change (Beale et al., 2013).

There are a number of peer-reviewed studies that demonstrate the utility of Protected Areas in maintaining species population levels (Karanth et al., 2009; Lester et al., 2009; Sciberras et al., 2013; Sheehan et al., 2013; Taylor et al., 2011) and reducing rates of habitat loss, the chief threat to biodiversity (Edgar et al., 2014; Geldmann et al., 2018; Micheli & Niccolini, 2013). Lawson et al., (2014) demonstrated the value of Protected Areas in facilitating range shifts for the silver-spotted skipper butterfly (*Hesperia comma*) by preventing grassland habitat destruction and by promoting conservation interventions that improve habitat quality. Similarly, by evaluating temporal changes in boreal bird abundance in and outside of Protected Areas, Lehikoinen et al. (2019) demonstrated the value of Protected Areas in mitigating range retractions in species adjusting to climatic changes and in facilitating range expansions in species adjusting to new spaces. These findings are consistent with other studies on migratory species (Gillingham et al., 2015), and suggest that Protected Areas will continue to fulfil an important role in biodiversity conservation by supporting species' range shifts (Thomas & Gillingham, 2015). Not only do Protected Areas appear to provide short-term strongholds for species to persist while conservation efforts on unprotected lands yield positive effects, species reliant on protected areas may use them as "stepping stones" when first colonizing new areas (Hiley et al., 2013; Thomas et al., 2012).

Protection - Data Source

The Canadian Protected and Conserved Areas Database (CPCAD) is a spatial inventory of protected areas and Other Effective Area-based Conservation Measures (OECMs) in Canada produced and managed by the Canadian Wildlife Service of Environment and Climate Change Canada (ECCC-CWS). Protected areas and OECMs considered in the CPCAD contribute to Target One of the 2020 Biodiversity Goals for Targets for Canada, which aims to conserve at least 17 percent of terrestrial areas and inland waters by 2020 (ECCC, 2016). The most recent iteration of this dataset represents Protected Areas and OECMs as of December 2, 2019. This spatial dataset was used to inform the protection sub-indicator of Adaptive Capacity.

Additional Variables Considered

In addition to the eight variables discussed above, there were several variables considered for this analysis that could not be included as they did not meet the variable selection criteria. These variables included; water quality, wetland connectivity, and littoral sediment dynamics.

Water quality was considered as a potential measure of Adaptive Capacity as degraded water quality can negatively affect aquatic habitat and wetland biota (Gleason et al., 2003; Relyea, 2005; Sharpley and Withers, 1994). However, a standardized water quality dataset covering all 20 wetland sites was not available at the time of this analysis. In addition, with respect to GLCWs, water quality is often directly related to watershed development and surrounding land cover (Trebitz et al., 2007; Crosbie & Chow-Fraser, 1999), which was accounted for within this study.

Wetland obligate species are often non-stationary and utilize multiple wetlands in a given area. Therefore, ensuring the stability or biodiversity of one wetland may involve how connected it is to surrounding wetlands (Haig et al., 1998). For this reason, functional wetland connectivity was considered as a potential variable to be included in this analysis. To measure functional wetland connectivity, an estimate of species movement attributes across all possible land cover types is required (e.g., boundary-crossing tendency, movement speed and distance, and mortality during movement). Such estimates are difficult to develop in practice (data intensive, computationally complex) and generally species-specific. Given that this study is broad and conducted at the scale of the Great Lakes basin, the cost-to-benefit ratio of producing functional connectivity estimate was considered too great to be included at this time.

Shoreline hardening, or the practice of modifying shoreline ecosystems for human use (e.g., retaining walls), can reduce natural littoral sediment movement transport in the Great Lakes, often resulting in a local or regional sediment deficit (Wensink & Tiegs, 2016). A sediment deficit combined with the continuous erosion of existing wetland depositional features (e.g. barrier beaches), can adversely affect wetland size, bathymetric slope, and may alter vegetation community structure (Wensink & Tiegs, 2016). Therefore, the physical and biological impacts of shoreline hardening and the resulting sediment deficit may reduce GLCW stability and/or plasticity. Quantifying littoral sediment dynamics and the degree of coastal erosion could not be performed consistently across all sites within this study due to a lack of information. Long-term changes in the lakeward extent of coastal wetland habitat have not been routinely measured within Great Lakes' littoral cells, nor have thorough investigations of causation been undertaken to identify erosive drivers (Zuzek, 2021). In theory, the cumulative length of adjacent shoreline hardened could be used to infer the degree of coastal erosion; however, at the time of this study, an analysis of shoreline hardening had not been completed. Currently, Environment and Climate Canada is quantifying the degree of shoreline hardening along the Canadian shoreline of the Great Lakes as part of a Baseline Habitat Survey (BHS).

Adaptive Capacity Indicator Methodology

The methodology for assessing wetland Adaptive Capacity can be broken down into two general steps: the first step is data collection and extraction, and the second is composite indicator production from extracted data. Of the eight variables used in this analysis, five were extracted using a simple geospatial analysis. These variables include: *Phragmites* within wetland, *Phragmites* surrounding wetland (5 km), protection within wetland, protection surrounding wetland (5 km), and natural land cover surrounding wetland (5 km). Due to the complexity of the spatial analyses used for determining upslope and downslope wetland migration potential, these methods are discussed separately. The final variable, vegetation species richness, was provided directly from transect data (see Vegetation Species Richness Data Collection) and therefore did not require spatial analysis or extraction.

Following the description of the geospatial variable collection, the composite indicator methodology is described. A composite indicator is an aggregation of carefully selected sub-indicators and their variables that can be used to provide a qualitative assessment of an indirectly measurable phenomena, such as Adaptive Capacity. The resulting wetland Adaptive Capacity composite indicator score can be used to determine the relative ability of a wetland to mitigate and persist under climate change, and allows for direct comparison across selected wetland sites. In order to accomplish this, the variable data was normalized, grouped into sub-indicators, assigned weights, and aggregated according to the conceptual interactions between variables.

Geospatial Variable Collection

Geospatial analysis was used to determine the sub-indicators of Landscape Condition (natural land cover), Biological Condition (invasive *Phragmites* within and surrounding the wetland) and Protection (protected lands within and surrounding the wetland). All geospatial analyses were conducted using customized scripts produced with ESRI ArcPro (ArcPy API) 2.2.4 and Python 3.6. Statistical analysis and the production of figures was accomplished using R 3.4.4.

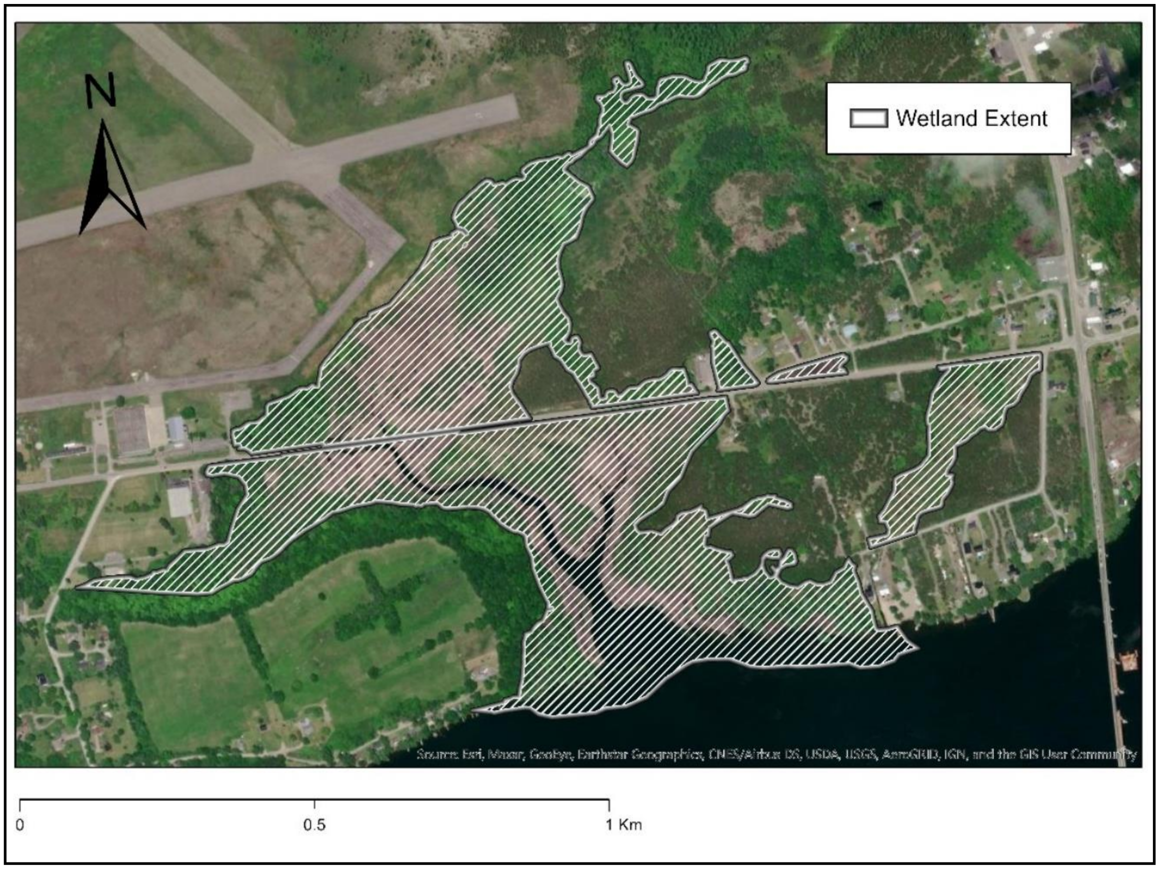


Figure 5. Hand delineated wetland extent for Airport Creek Marsh

Hand-delineated wetland areas of interest (AOIs) were digitized in ESRI ArcMap 10.6 using 0.5m resolution pan-sharpened and orthorectified World View satellite imagery (see Figure 5 for example). A five-kilometer equidistant buffer surrounding wetland AOIs was calculated using the Great Lakes Albers Equal Area (EPSG: 3174, NAD 83) projection to reduce local area distortion (Figure 6). A five-kilometer buffer was selected for landscape-level metrics in order to capture the majority of surrounding variability while considering available research and data limitations. Previous studies measuring the influence of landscape-level variables on wetland species range from 100 m (i.e., amphibians) to 20 km (birds and mammals) buffers depending on the species of interest. In order to capture the majority of species responses while staying within data limitations (i.e., MTRI dataset covered ten kilometers inland), a five-kilometer buffer was selected.

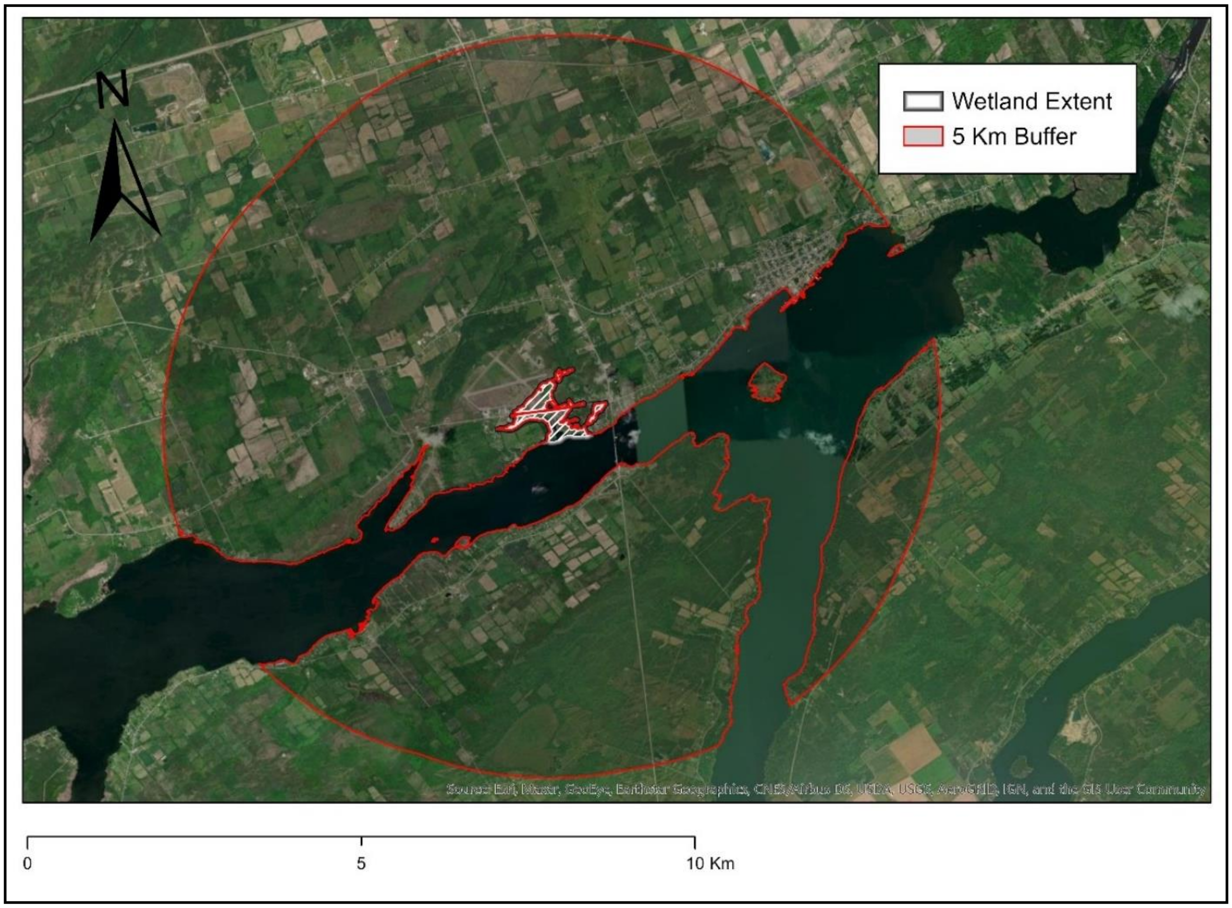


Figure 6. Five-kilometer buffer area for Airport Creek Marsh

From the resulting five-kilometer geometries, the wetland AOIs and intersecting large water bodies were removed to provide only terrestrial attributes within the AOI buffers. Because AOI lakeward boundaries were not interpreted from consistent land-use classifications or a low-water depth contour (e.g., 2m – Ontario Wetland Evaluation System; Ontario Ministry of Natural Resources [OMNR], 2013), large water bodies were removed to limit any site-specific bias when analyzing sub-indicator proportionality (see below). The resulting buffered dataset, termed the terrestrial AOI, was then intersected with the applicable geospatial dataset (i.e., Annual Crop Inventory Data and Crop Classification Database, MTRI Coastal Wetland Mapping, and the Canadian Protected and Conserved Areas Database). Each geospatial variable was then calculated as a proportion of the remaining buffer area. For a complete visual representation of the geospatial analysis, see Figure 7.

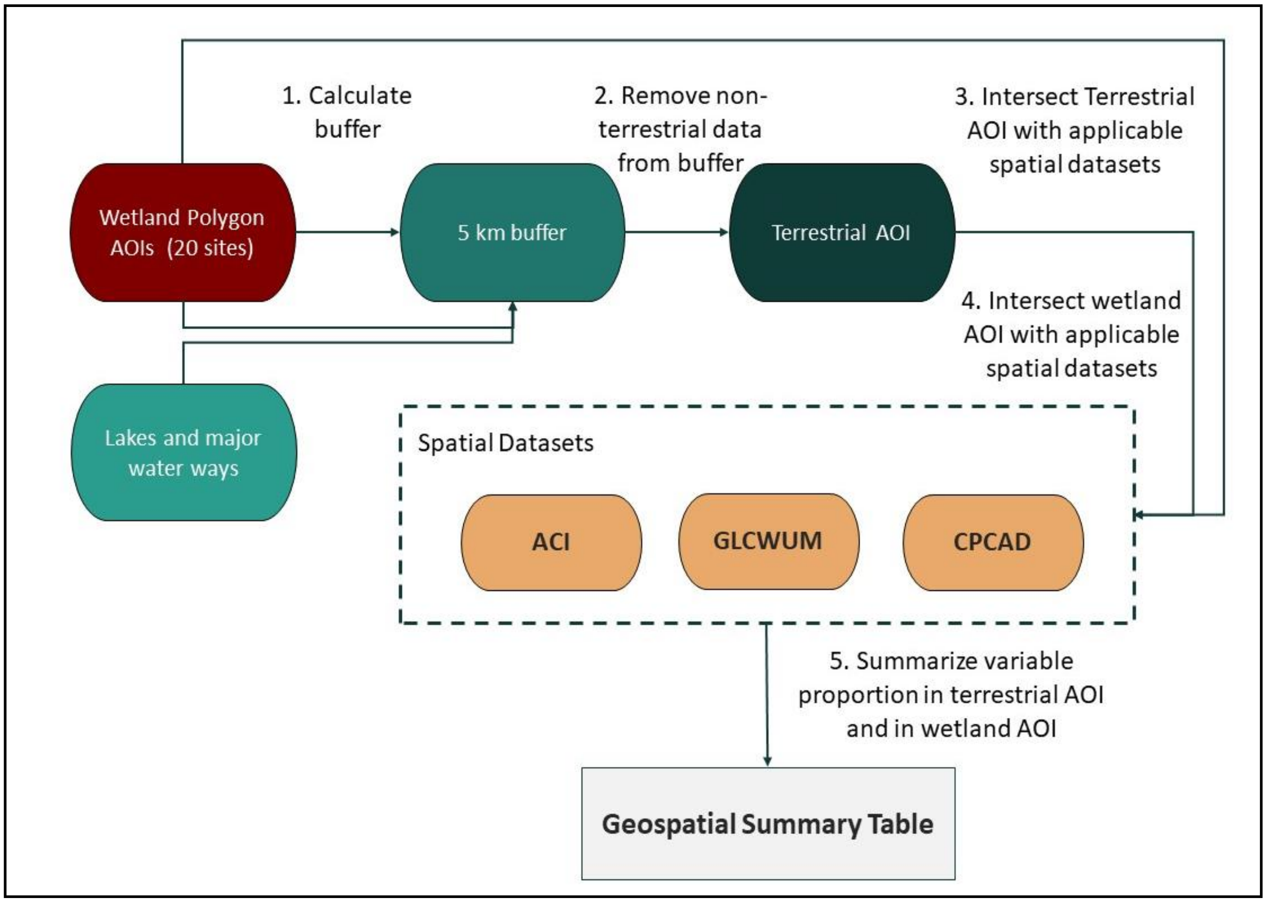


Figure 7. Schematic representation of geospatial methodology used to extract spatial variable data

Wetland Migration Potential Geospatial Analysis

The wetland migration potential sub-indicator is the combination of both upslope and downslope migration potential variables. To quantify each variable, vertical migration limits were estimated from available water-level projections for the Great Lakes. These limits were then further refined by physical variables for upslope migration, or refined by expert knowledge for downslope migration. The result of this analysis provided an estimation of the potential migration area surrounding the wetland sites of interest.

When determining potential upslope migration area, vertical wetland migration limits (i.e., elevation) were first defined, then overlayed with physical and biological attributes which may either impede or assist in upslope wetland migration. These attributes included, but are not limited to land cover, substrate geology, light, nutrient availability, coastal processes, hydrogeology and habitat connectivity (Cronk & Fennessy, 2001; Keough et al., 1999; Uzarski et al., 2017).

A combination of vertical wetland limits (i.e., bathymetry) and expert opinion was used to determine potential area for downslope migration. Due to limited data availability, substrate and corresponding vegetation data precluded the potential for downslope migration to be further refined in the same manner as upslope migration. However, coastal processes were considered through expert consultation which provided insight to limit downslope migration beyond vertical bathymetry limits. The result of this spatial analysis is the potential area for plant communities and species to migrate to under projected lake-level variability.

Vertical Wetland Migration Limits

To establish the vertical limits for wetland migration, historic lake-level maximums and minimums (Fisheries and Oceans Canada, 2020), mid-century water level projections (IUGLS 2012) and mid-to-late century projections (National Hydrological Services, ECCC-NHS; Seglenieks &Temgoua, 2021) were considered. Using these projections and historical data, lake level maximum and minimum values were identified to inform the upper and lower vertical migration limits.

Limits for upslope migration were determined using projected lake-level maximums. Lake-level maximums were similar between IUGLS (2012) and ECCC-NHS simulations; however, in basins where the results diverged considerably (e.g., Lake Erie), professional judgment (Zuzek Inc., 2020a) was used to select the appropriate future lake-level high estimate. Two meters were then added onto the lake-level projections to establish upper topographic limits for upslope migration (

Table 2).

Limits for downslope migration were determined by using historic minimums and lake-level low estimates from IUGLS (2012). If historic and projected lake low water levels diverged, professional judgment (Zuzek Inc., 2020a) was used to select an appropriate minimum. ECCC-NHS water-level projections were not finalized as of the completion of this report and could therefore not be utilized. One meter was added in recognition that submerged aquatic and free-floating vegetation may migrate further downslope than a future lake-level low (Table 3).

Three GLCW study sites can be found on connecting channels between the Great Lakes. These include Anderson Creek Marsh (23ACK; St. Mary's River), the Detroit River Marshes (12DRM; Detroit River) and Hill Island East (1HIE; Upper St. Lawrence River). The Lake Huron migration limits were applied to Anderson Creek, the Lake Erie migration limits were applied to the Detroit River Marshes, and the Lake Ontario limits were applied to Hill Island East.

Table 2. Table comparing the upslope and bathymetric limits for wetland migration compared to historic highs and lows in meters

Lake	Huron	St. Clair	Erie	Ontario
Vertical Limit for Upslope Migration	**180.50**	**179.00**	**177.70**	**78.00**
Approx. Historic High (Fisheries and Oceans Canada, 2020)	177.50	176.10	175.10	75.90
Approx. Historic Low (Fisheries and Oceans Canada, 2020)	175.55	173.90	173.20	73.75
Bathymetric Limit for Downslope Migration	**173.55**	**171.90**	**171.20**	**72.50**

Horizontal Wetland Migration Limits (Area)

After the upper and lower vertical limits to wetland migration were defined, the area within these limits available for migration needed to be identified. Migration area was identified using the upper and lower vertical migration limits selected in the previous section in combination with site-specific digital terrain models (DTMs; Figure 8). All area existing between the minimum and maximum vertical migration elevation limits was considered potential migration area to be further limited based on land cover and expert knowledge. DTMs spanning aquatic and terrestrial environments were prepared and represent bare-earth conditions (ECCC, 2022b).

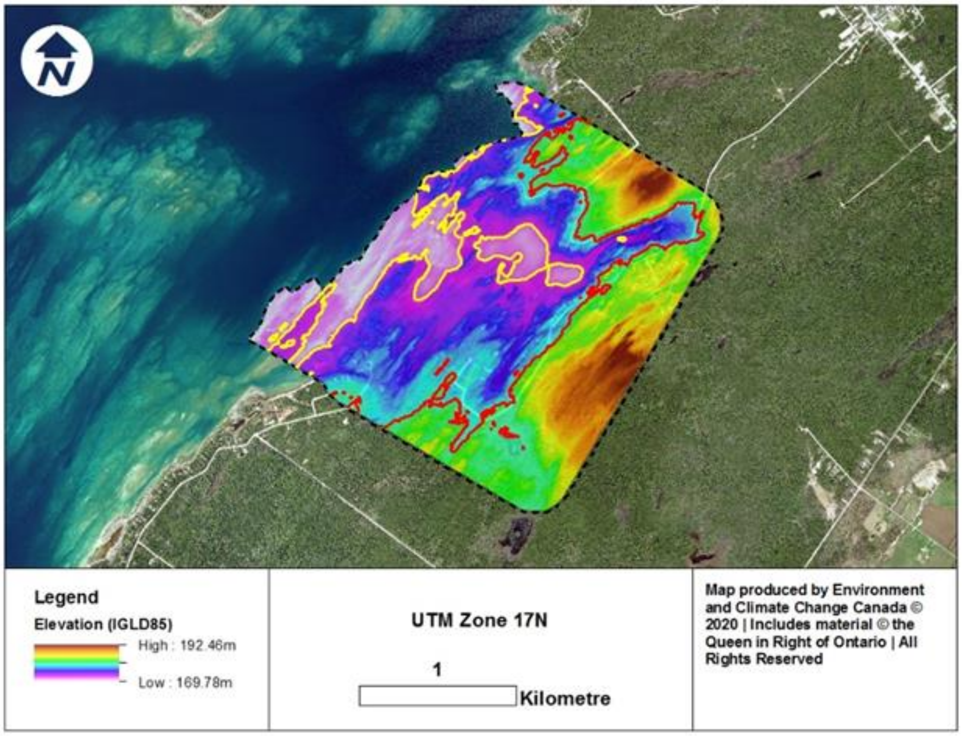


Figure 8. Digital Terrain Model (DTM) for Hay Bay Wetland. Contours illustrated are the topographic (180.50m IGLD5; red) and bathymetric limits (173.55m IGLD85; yellow) for w etland migration

Upslope Migration and Land Classification Suitability

The area available for landward migration, determined from elevation and projected water-level scenarios, was further limited using land cover data. Lands adjacent to each wetland were considered suitable for landward migration depending on their community class (Lee et al., 1998). In general, natural classes capable of transitioning into wetland habitat (e.g., forest), or capable of facilitating the vertical movement of wetland vegetation (e.g., meadow) were considered to be suitable. Conversely, anthropogenic and natural classes likely to impede or limit the vertical movement of wetland vegetation (e.g., constructed; cliff and talus, respectively) were considered unsupportive. The current footprint of each wetland, as measured through wetland community classes (e.g., marsh, swamp), was excluded from suitable upland areal estimates. Only areas with both suitable elevation and suitable land cover were selected as potential migration

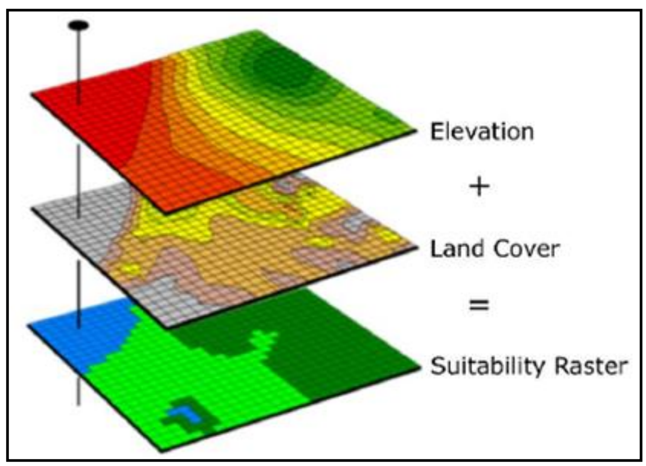


Figure 9. Schematic of combining elevation data and land cover data to determine upslope migration area (i.e., suitability raster)

areas (Figure 10). The proportion of area suitable for upslope migration out of total potential wetland migration area was used as a representative of upslope migration potential.

Downslope Migration

Unlike upslope migration potential, which utilized DTMs and multiple land cover datasets, downslope migration was assessed qualitatively using a combination of bathymetric data and expert consultation (Zuzek Inc., 2020a). Due to a lack of substrate and georeferenced aquatic vegetation data, a suitability raster layer could not be developed for downslope migration. Therefore, a decision-tree was adopted to approximate the aquatic areas suitable for downslope migration at each wetland site (Figure 10). The same land cover datasets utilized in the upslope migration analysis were used to determine the current extent of each wetland site as well as vegetation classes suitable for downslope migration under sustained lake-level lows. Emergent and shallow water marsh community classes were considered to represent the current aquatic extent of each wetland site, and open and shallow water classes were deemed suitable for downslope migration, assuming appropriate bathymetry.

Areas suitable for downslope migration were initially defined by the bathymetric limits (

Table 2). However, relying exclusively on bathymetric limits to define the potential areas for downslope migration resulted in unrealistic approximations for wetlands in regions of low vertical relief (e.g., sites on Lake St. Clair). Wave action, substrate quality, and light attenuation are expected to limit the migration of wetland vegetation several kilometers out into an exposed lakebed (Zuzek Inc., 2020a). Therefore, a 200m buffer (NAD 1983 UTM 17N) from the current wetland extent was implemented in addition to bathymetric constraints to produce a more probable approximation for lakeward migration. Whichever limit was encountered first (i.e., 200m buffer or bathymetric constraint), was chosen.

Preliminary results also informed the removal of aquatic community classes without a hydrologic connection to a lake or connecting channel prior to buffer establishment (e.g., existing ephemeral pools and adjacent wetland habitat outside the AOI), as well as shoreline reaches heavily influenced by coastal processes (e.g., sediment supply and transfer) including nearshore wave energy. Barred drowned river-mouth wetlands (such as Lynde Creek Marsh) were considered to have little to no downslope migration potential beyond the depositional features (e.g., barrier-beach) at their mouths. Similarly, the downslope migration potential of coastal wetlands without wave energy protection beyond their mouths was also considered to be limited (e.g., Fox Creek/ Dolson's Creek). Plant biomass and wave energy are negatively correlated along most vegetated shorelines (Azza et al., 2007), as wave energy can uproot seedlings, damage mature plants and erode fine sediments around roots and rhizomes (Cooper & Uzarski, 2016; Riis & Hawes, 2003). Site-specific long-term wave climates (i.e., hourly wave condition, including height, period and direction over multiple decades) were approximated from ECCC's Nearshore Assessment Framework (Zuzek Inc., 2020b) and expert consultation (Zuzek Inc., 2020a).

After areas suitable for downslope migration were adjusted for hydrologic connectivity and coastal processes, manual revisions to buffer widths and bathymetric migration limits were made based on a qualitative substrate analysis and hydrogeomorphology. The downslope migration potential of coastal wetlands situated in bedrock geology, such as Hay Bay (19HBW) on the Bruce Peninsula (limestone) was further restricted under the assumption that limited water-saturated and submersed soils would inhibit aquatic macrophyte colonization (Wetzel, 2001). The downslope migration potential of coastal wetlands situated within highly protected embayments, such as Treasure Bay (19TBY) was also extended beyond horizontal migration limits under the assumption that coastal processes (i.e., wave action) would not restrict the downslope colonization of aquatic macrophytes. Data availability prevented the quantitative assessment of other physical and chemical characteristics known to moderate the succession of wetland vegetation communities such as light and nutrient availability, and hydrogeology (i.e., groundwater resources; Cronk & Fennessy, 2001; Keough et al., 1999; Uzarski, et al., 2017; Wilcox, 2012). The areas suitable for downslope migration (ha) were then calculated and carried forward into composite indicator construction.

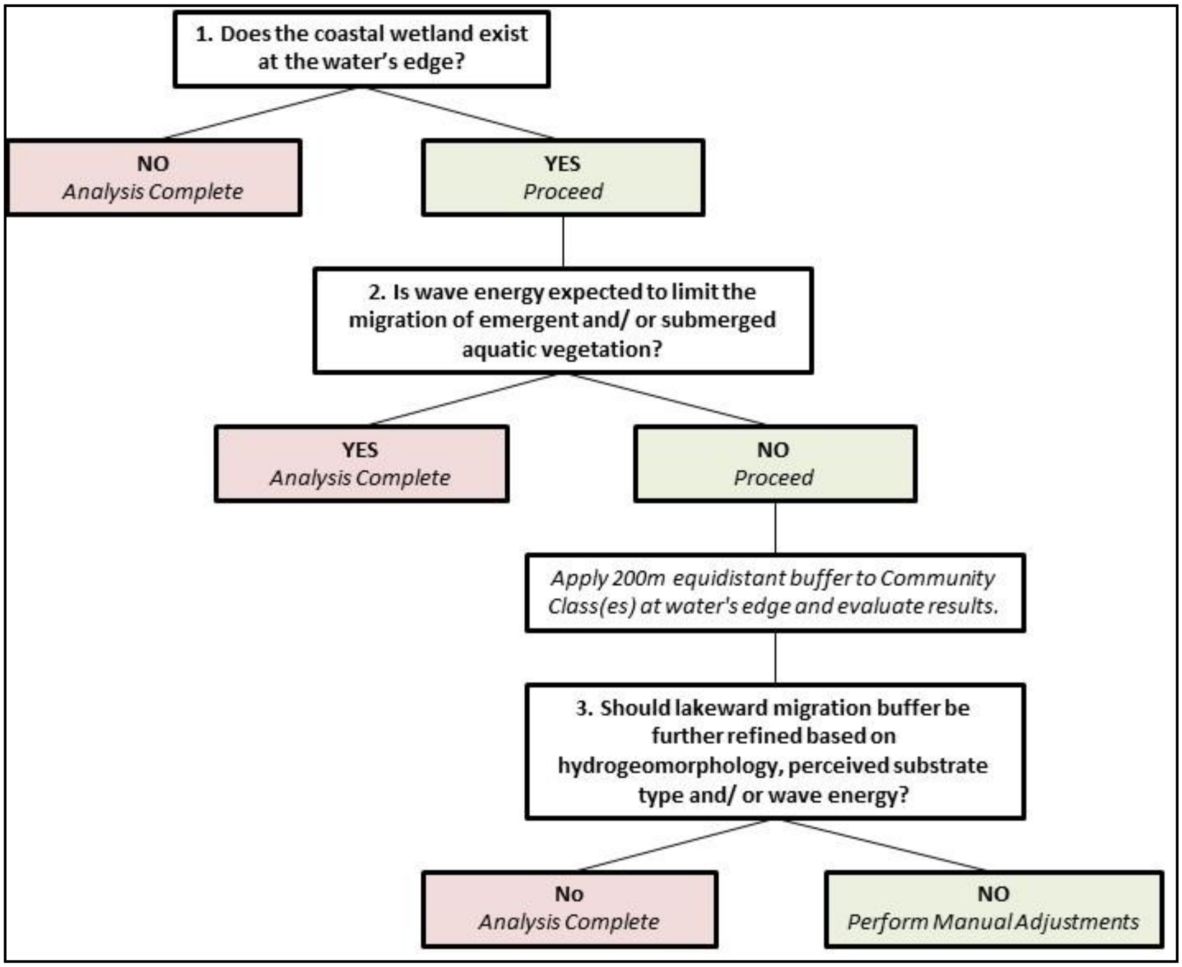


Figure 10. Decision-tree utilized to approximate the aquatic areas suitable for downslope migration at each wetland site

Composite Indicator Construction Methods

The Adaptive Capacity composite indicator is an aggregation of carefully selected sub-indicators and variables that represent key aspects of the wetland's ability to mitigate and persist under climate change. The resulting composite indicator output values quantify the theoretical concept of Adaptive Capacity to allow for direct qualitative comparison across selected wetland sites. In order to accomplish this, the data collected for each variable was normalized, assigned weights, grouped in to sub-indicators, and aggregated according to the conceptual interactions between variables (Burgass et al., 2017). Normalization was required to rescale and orient variables so that they could be combined despite having various units of measurement. Sub-indicator groupings were determined based on how the variable would contribute to the stability or plasticity of a wetland.

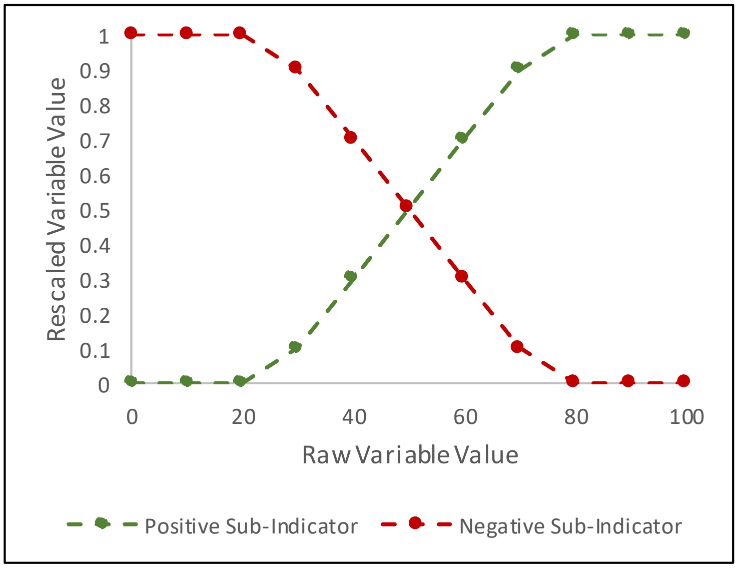
The grouping of variables into sub-indicators, as well as combining sub-indicators into the final composite indicator, required applying weights. Weighting is contentious within the scientific community as it is often interpreted as a representation of variable importance; however, in this analysis, weighting beyond equal weighting was only applied when correlation between variables existed (see Variable Grouping and Weighting for more detail). Finally, the aggregation of variables and sub-indicators required the selection of a geometric (multiplicative) or linear (additive) aggregation method. Both methods are used in this analysis due to the potential influence of variables and sub-indicators on one another (see Variable and Sub-Indicator Aggregation).

Variable Data Normalization

Variables used within this study to represent Adaptive Capacity were collected in different units of measurement and had different data distributions, therefore prior to aggregation all variables were normalized. Additionally, variables may improve or reduce the Adaptive Capacity of coastal wetlands depending on their influence. Therefore, in addition to data normalization, variables were also directionally oriented so that the influence of the variable appropriately reflected its contribution to Adaptive Capacity prior to aggregation. When selecting an appropriate normalization method, the distribution in variable data, including outliers and skewness, were considered. Normalized variables reflect the range of values measured for the sites selected, not the full range possible for coastal wetlands across the Great Lakes Region. Ultimately, two methods of normalization were used: ranking and min-max.

Ranking is a normalization technique that is robust to outliers and skewed data. The ranking method was used to normalize Migration Potential data (both upslope and downslope) as these datasets had non-normal distributions and outliers, and the rank was then divided by 20 to produce a score of 0-1. This ranking dampens the effect of skewed data on the greater composite aggregation which prevents Migration Potential from being overrepresented in the end result.

Minimum-maximum rescaling was accomplished by subtracting the minimum variable value across all study sites, and by dividing each site value by the range for a particular variable. The result is a normalized range of 0-1. This method of rescaling was applied to the following variables: vegetation species richness, invasive *Phragmites* within and surrounding the wetland, Protected Areas within and surrounding the wetland, and natural land cover. By using min-max and ranking normalization, all variables were normalized to the same unit-less scale and could be combined in subsequent steps.



Variables also had to be directionally oriented so that a value of zero would reduce the Adaptive Capacity indicator score and a value of one would improve it. With the exception of invasive *Phragmites*, all sub-indicator variables reflected this orientation. The variables associated with invasive *Phragmites* are suggested to reduce Adaptive Capacity, therefore, an observed high value (e.g., a high proportion of *Phragmites* within the surrounding area) was equated to a low sub-indicator value. To address this, variables with a reductive impact on Adaptive Capacity were not only normalized to a value of 0-1, but also inverted to allow for proper representation of the mechanism in later aggregation stages (Figure 12). This was accomplished using a fuzzy-membership function in ESRI ArcGIS Pro.

Figure 11. Diagram representing the directional orientation or reorientation of positively and negatively contributing variables

Variable Grouping and Weighting

After sub-indicator scores were normalized and prior to aggregation, variables were weighted and grouped. Three methods for providing variable weights were considered: 1) Weighting to reflect the perceived importance of variable based on expert opinion; 2) Applying equal weights to all sub-indicators; and, 3) Applying weights to reflect the variability of data within sub-indicator scores (e.g., Principal Component Analysis, PCA).

Weighting can be thought to reflect the importance of a sub-indicator's contribution to the final composite indicator, and therefore weighting based on the opinion or knowledge of sub-indicator's mechanistic action is often a default approach. Alternatively, if a decision cannot be made to provide relative weighting, equal weighting has often been selected as an alternative (Burgass et al., 2017). While this may be perceived as a neutral decision, or a decision to treat all sub-indicators as equal contributors to the composite indicator score, it is still a weighting decision and reflects the concept that the variables combined have independent influence. In this study, equal weighting and PCA weighting were utilized.

Equal weighting was used when grouping upslope and downslope variables to produce the Migration Potential sub-indicator, and when grouping protection with and surrounding wetlands to produce the Protection sub-indicator. Biological Condition was the only sub-indicator composed of more than two variables. Species richness and invasive *Phragmites* coverage both within and surrounding the wetland were correlated but have different impacts on a wetlands ability to adapt. To account for this, a PCA was conducted and the variables were weighted according to their contribution to the first principal component. Finally, the Landscape Condition sub-indicator was composed of only one variable, natural land cover, and therefore, no weighting was required.

Following the grouping of variables into their respective sub-indicators, the sub-indicator then needed to be aggregated into the final Adaptive Capacity composite indicator score. Equal weighting was applied when combing all sub-indicators due to their unique contributions to the overall composite indicator. However, sub-indicators cannot be considered completely independent of one another. Therefore, the potential for sub-indicator interaction, as well as sub-indicator compensation was addressed through liner and geometric aggregations.

Variable and Sub-Indicator Aggregation

Normalized and weighted variables and sub-indicators were combined using linear and geometric aggregation methods. Linear aggregation, or additive aggregation, is commonly applied when producing a composite indicator. In linear aggregation, a deficit or low score in one variable can be compensated by a high score in another variable. In addition, linear aggregation compensation is impacted by variable weight, where a high weighting can allow for compensation of a lower score. Conversely, geometric aggregation, or multiplicative aggregation, reduces the ability of variables to compensate for one another. Therefore, when geometric aggregation is applied, the most effective way to improve the Composite Indicator score of a site would be to bolster low sub-indicator scores as opposed to sub-indicators currently scoring high.

All variables within each sub-indicator were combined using linear aggregation (i.e., additive). This allows for variables that influence wetland stability or plasticity to compensate for one another. For example, a wetland with invasive *Phragmites* present may still be considered to have high Biological Condition if a high species richness can compensate.

Biological Condition (Equation 1), Landscape Condition (Equation 2) and Migration Potential (Equation 3) were combined together using geometric aggregation. It is expected that each of these sub-indicators independently contribute to the adaptive capacity of coastal wetlands and therefore, cannot compensate for one another. For example, the Adaptive Capacity of a wetland receiving a low Biological Condition score cannot compensate for this with a high Migration Potential sub-indicator.

Equation 1. Biological Condition sub-indicator aggregation equation, where BC represents the Biological Condition sub-indicator, PhragWW represent *Phragmites australis* within the wetland, PhragSW represent *Phragmites* surrounding the wetland, and VSR represent vegetation species richness.

$$BC = ((35.59 \times PhragWW_{sc(0-1)}) + (33.21 \times PhragSW_{sc(0-1)})) \times (31.20 \times VSR_{sc(0-1)})$$

Equation 2. Landscape Condition sub-indicator aggregation equation, where LC represent Landscape Condition, and NLC represent natural land cover

$$LC = (100 \times NLC_{min-max(0-1)})$$

Equation 3. Wetland Migration Potential sub-indicator aggregation equation, where WMP represents Wetland Migration Potential, UMP represents upslope migration potential, and DMP represents downslope migration potential

$$MP = (50 \times UMP_{rank(0-1)}) + (50 \times DMP_{rank(0-1)})$$

Equation 4. Protection sub-indicator aggregation equation, w here P represents Protection, PWW represents protection w ithin w etland, and PSW represents protection surrounsding the w etland

$$P = \left(50 \times PWW_{min-max(0-1)}\right) + \left(50 \times PSW_{min-max(0-1)}\right)$$

Protection (Equation 4) is the only sub-indicator considered to moderate Composite Indicator scores. Considering Protected Areas cannot be developed and that they have management and governance institutions implementing activities expected to benefit all other sub-indicator scores, the Protection score a site receives should be able to compensate for a low cumulative score from the other sub-indicators. To incorporate this relationship into the composite indicator aggregation, the natural logarithm of the geometrically aggregated composite score for landscape condition, migration potential and biological condition was weighted threefold when linearly aggregated with protection. The composite indicator composition is represented schematically in Figure 12.

Equation 5. Adaptive Capacity Composite Indicator aggregation equation, where AC represents Adaptive Capacity, LC represents the Landscape condition sub-indicator (Equation 2), BC represents the Biological Condition sub-indicator (Equation 1), MP represents the Wetland Migration Potential sub-indicator (Equation 3), and P represents the Protection sub-indicator (Equation 4).

$$AC = \frac{(3 \times (\log((LC \times BC \times MP)_{min-max(0-100)})) + P)}{377.53}$$

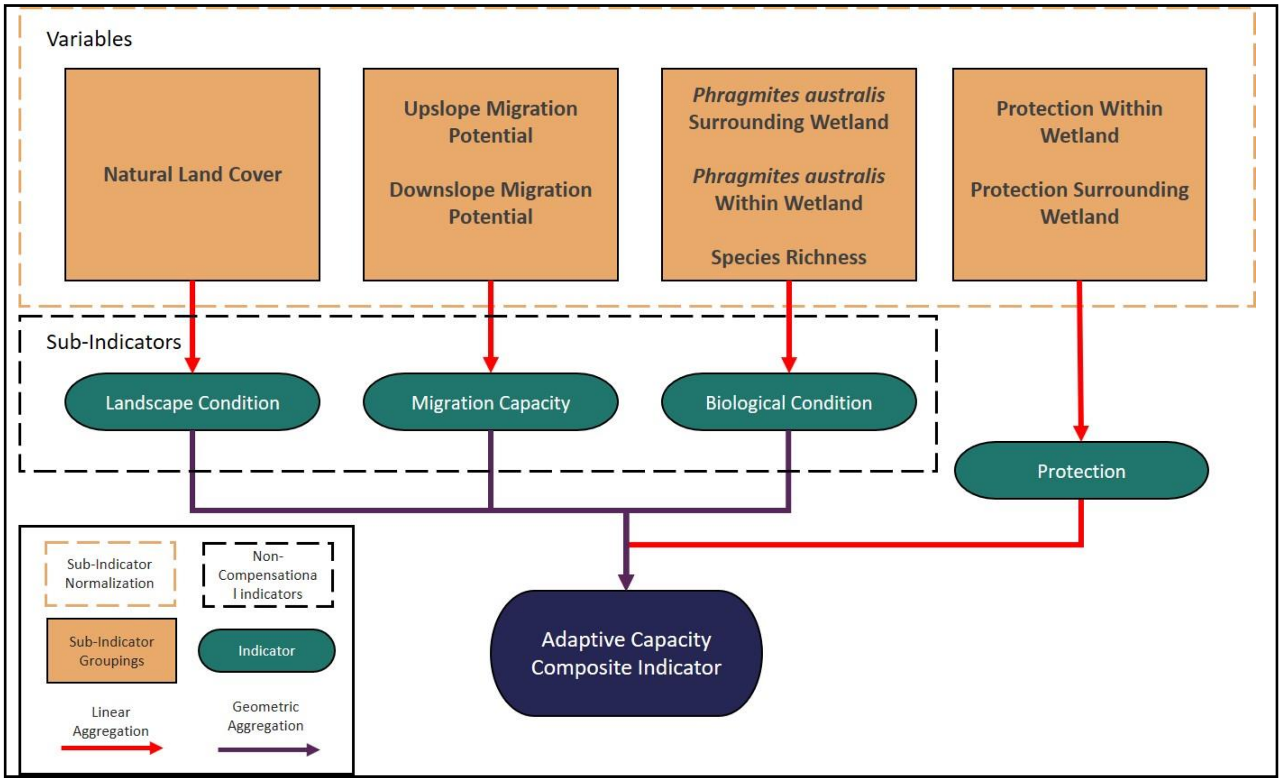


Figure 12. Schematic displaying Adaptive Capacity Composite Indicator aggregation, including the eight selected variables (natural land cover, *Phragmites australis* within wetland, *Phragmites australis* surrounding wetland, vegetation species richness, upslope migration potential, downslope migration potential, protection within wetland and protection surrounding wetland), sub-indicator groupings (Landscape Condition, Biological Condition, Migration Capacity and Protection), and aggregation method.

Results and Discussion

The results can be reviewed at a landscape or site-specific scale. At the landscape level, relative Adaptive Capacity Scores, which categorize all 20 wetland sites as having either a High, Moderate, or Low Adaptive Capacity can be reviewed to identify priority areas for adaptive management. In addition, each sub-indicator can be viewed independently, in order to determine how that sub-indicator varied across wetland sites. Sub-indicators include Biological Condition, Landscape Condition, Wetland Migration Potential, and Protection and are represented on a continuous scale of 0 to 100. Sub-indicator scores represent the relative value of the sub-indicator compared to other GLCW sites considered in this analysis. These results can be used by land managers or stakeholders in a stepwise manner. For example, if a wetland of interest received a Low Adaptive Capacity Score, the sub-indicator components can be reviewed to identify an appropriate adaptation strategy.

The results and discussion from this analysis are broken into three sections. First the sub-indicator scores and trends are reviewed, followed by Adaptive Capacity Scores and observations, and finally an independent review of each wetland site.

Sub-Indicator Scores

Landscape Condition

Regarding landscape condition, the highest-scoring wetland site was Francis Point Wetland (28FPT) in Eastern Georgian Bay, while the lowest-scoring was the Lake St. Clair Marshes (13LSC) on the eastern shoreline of Lake St. Clair (Figure 13). Similar to Biological Condition, all seven sites found on Lake Huron scored in the top 50% of sites for Landscape Condition. In addition, Hill Island East (1HIE), Airport Creek Marsh (2ACM), and South Bay Marsh (3SBM) also scored in the top 50% of sites.

Sites in the upper half of the Landscape Condition gradient that are not in Lake Huron appear to be either protected (e.g., Hill Island East), or on First Nations' land (e.g., Airport Creek Marsh). The only exception to this is South Bay Marsh in Prince Edward County. Unlike the majority of the County, which is dominated by agriculture, the lands surrounding South Bay Marsh appear to have a disproportionate amount of natural cover, particularly to the south. The limestone plateau that lies beneath Prince Edward County may be close to the surface here, making the lands adjacent to South

Bay inarable. Deep clay deposits bordering South Bay could also impede drainage, rendering the area unsuitable for crops and livestock (Chapman & Putnam, 1966).

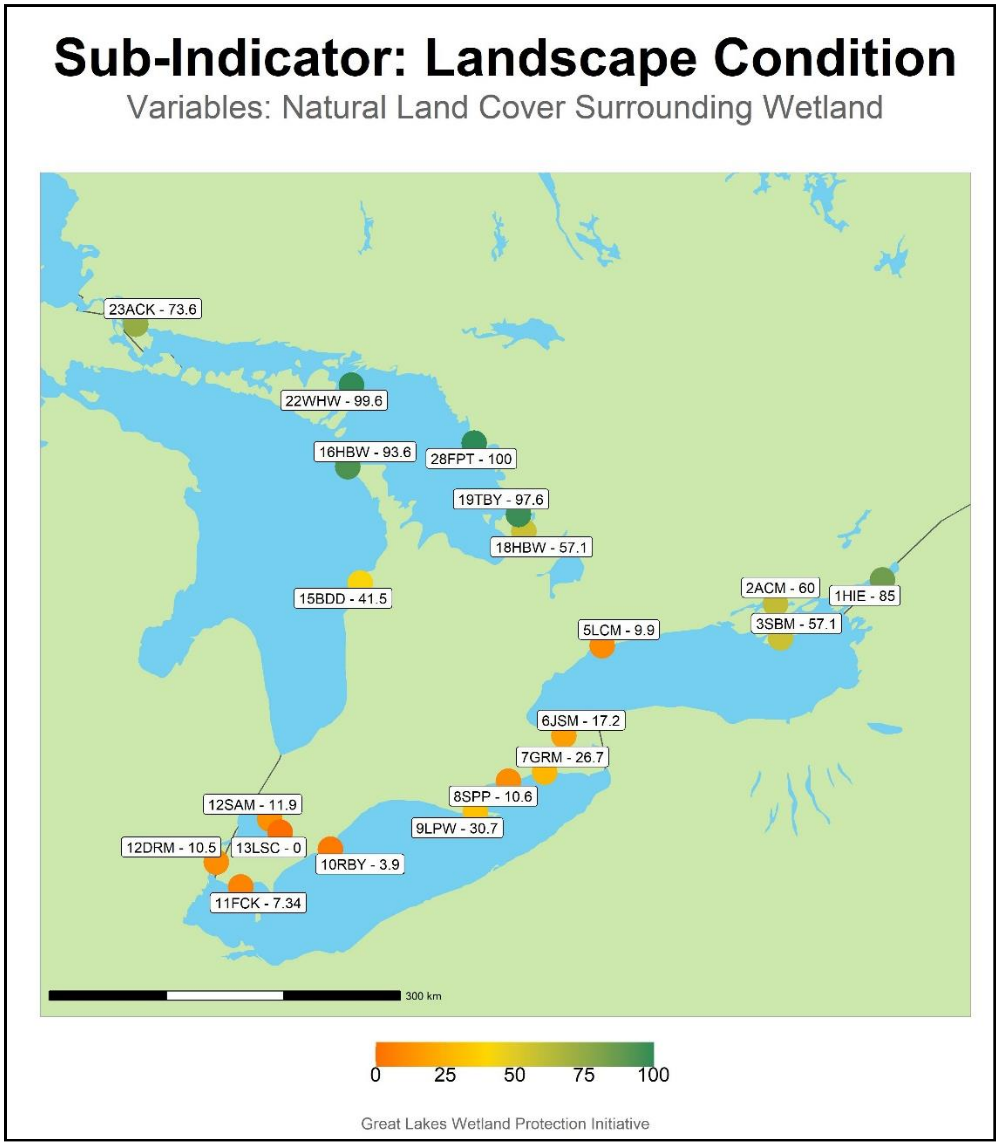


Figure 13. Map of selected Great Lakes Coastal Wetland sites displaying relative Landscape Condition sub-indicator scores (from 0 – 100). The Landscape Condition sub-indicator was composed natural land cover surrounding wetland sites. Lake St. Clair (13LSC) received the lowest Landscape condition score (0), while Francis Point (28FPT) received the highest (100).

Biological Condition

The highest-scoring wetland with respect to Biological Condition was Baie Du Doré (15BDD) on Lake Huron, while the lowest-scoring site was Johnston Bay Wetland (12SAM) at the mouth of the St. Clair River (Huron-Erie Corridor; Figure 14). All seven of the wetland sites found on Lake Huron along with two sites from Lake Ontario (2ACM and 1HIE) and one site from Lake Erie (8SPP) scored within the top 50% of sites. Three of the four lowest-scoring sites were concentrated along the Huron-Erie Corridor.

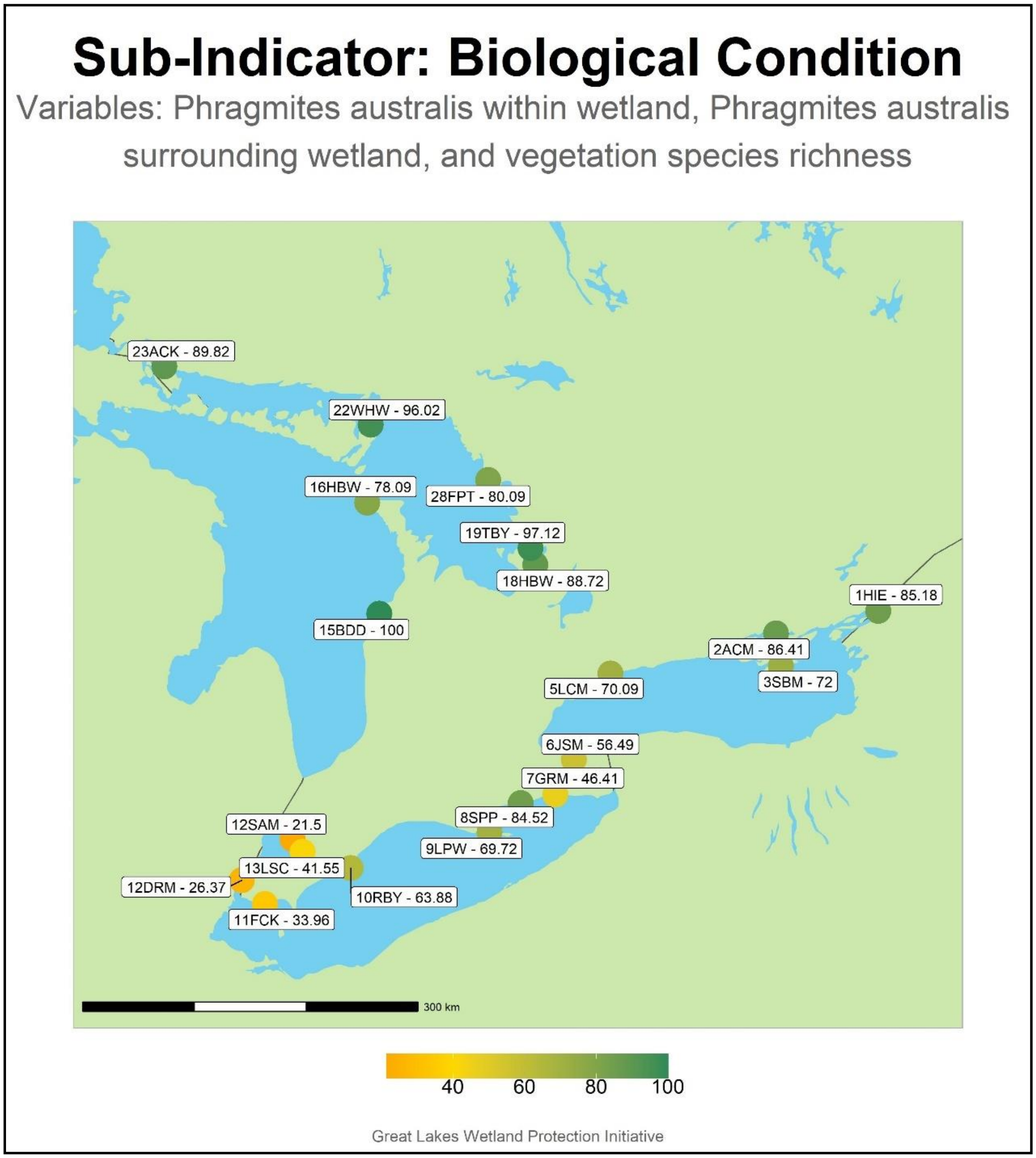


Figure 14. Map of selected Great Lakes Coastal Wetland sites displaying relative Biological Condition sub-indicator scores (from 21.5 – 100). The Biological Condition sub-indicator was composed of invasive Phragmites within and surrounding wetlands, as well as vegetation species richness. Johnston Bay (12SAM) received the lowest biological condition score (21.5) while Baie Du Doré received the highest (15BDD).

Wetland Migration Potential

The highest-scoring site with respect to Wetland Migration Potential was Johnston Bay (12SAM) which is located on the Huron-Erie Corridor. The lowest-scoring was Hill Island East (1HIE) on Lake Ontario (Figure 15). Hydrogeomorphology and wetland depth likely contribute to high Wetland Migration Potential, where shallow wetlands with fine (clay) to medium-grained sediments (sand and gravel) appear to have the highest potential to migrate. Eight of the ten highest-scoring wetlands have the following hydrogeomorphologies: delta (12SAM), open shoreline (13LSC and 12DRM), sand-spit embayments (10RBY and 9LPW) or open embayments (3SBM and 15BDD). The Grand River Mouth Wetlands (7GRM) and Airport Creek Marsh (2ACM) are the only drowned river-mouth wetlands in the upper half of the migration potential gradient, likely due to a combination of hydrogeomorphology and surrounding land use.

Although classified as a barred drowned-river mouth, the marshes at the mouth of the Grand River form somewhat of a delta downstream of the weir in Dunnville, Ontario, suggesting low topographic and/or bathymetric relief. The Grand River is also situated on the Haldimand Clay Plain, and wet sandy loams are present in the areas surrounding Dunnville (Chapman and Putnam, 1966). The relatively high migration potential for Airport Creek Marsh (2ACM) on the Bay of Quinte likely affords shelter for downslope migration, but the airport adjacent to the marsh would appear to be an impediment to upslope migration. Additionally, Airport Creek Marsh is situated on the southern edge of the Napanee Plains, which, like Prince Edward County, is underlain by a flat-to-undulating plan of limestone (Chapman and Putnam, 1966). This substrate would be difficult for wetland vegetation to colonize during a high-water event.

Six out of 11 sites classified as protected embayments scored in the lower half of the migration potential gradient. This low score is likely due to the bedrock or till-derived shorelines that form these protected bays (Albert et al., 2005) which may impede landward migration. The bedrock geology surrounding these embayments, whether it be igneous (e.g. granite), metamorphic (e.g. schist) or sedimentary (e.g. limestone) would be difficult for vegetation to colonize during a high water-event. Additionally, areas outside of each embayment are likely unsuitable for downslope colonization due to coastal or riverine processes, or steep bathymetric relief. The protected embayments considered in this study are situated on: the Canadian Shield in Lake Huron (18HBW, 19TBY, 28FPT, and 22WHW), the Niagara Escarpment on the Bruce Peninsula (16HBW), or the granitic knobs of the Frontenac Axis (1HIE; Chapman and Putnam, 1966).

The remaining four wetlands in the lower half of the Migration Potential gradient are drowned-river mouths, and their limited potential to migrate can be partially explained by hydrogeomorphology. Similar to the protected embayments in Lake Huron, Anderson Creek (23ACK) is situated on the Canadian Shield, which could impede upslope migration as discussed above. Selkirk Provincial Park (8SPP), Jordan Station Marsh (6JSM) and Lynde Creek Marsh (5LCM) are all barred drowned river-mouth wetlands, which are not conducive for downslope migration. The presence of a barrier-beach at the mouth of these locations (or the former presence in the case of Jordan Station Marsh,) suggests that significant coastal processes exist (e.g., littoral drift).

These depositional features and the forces that created them would make the colonization of downslope habitats challenging.

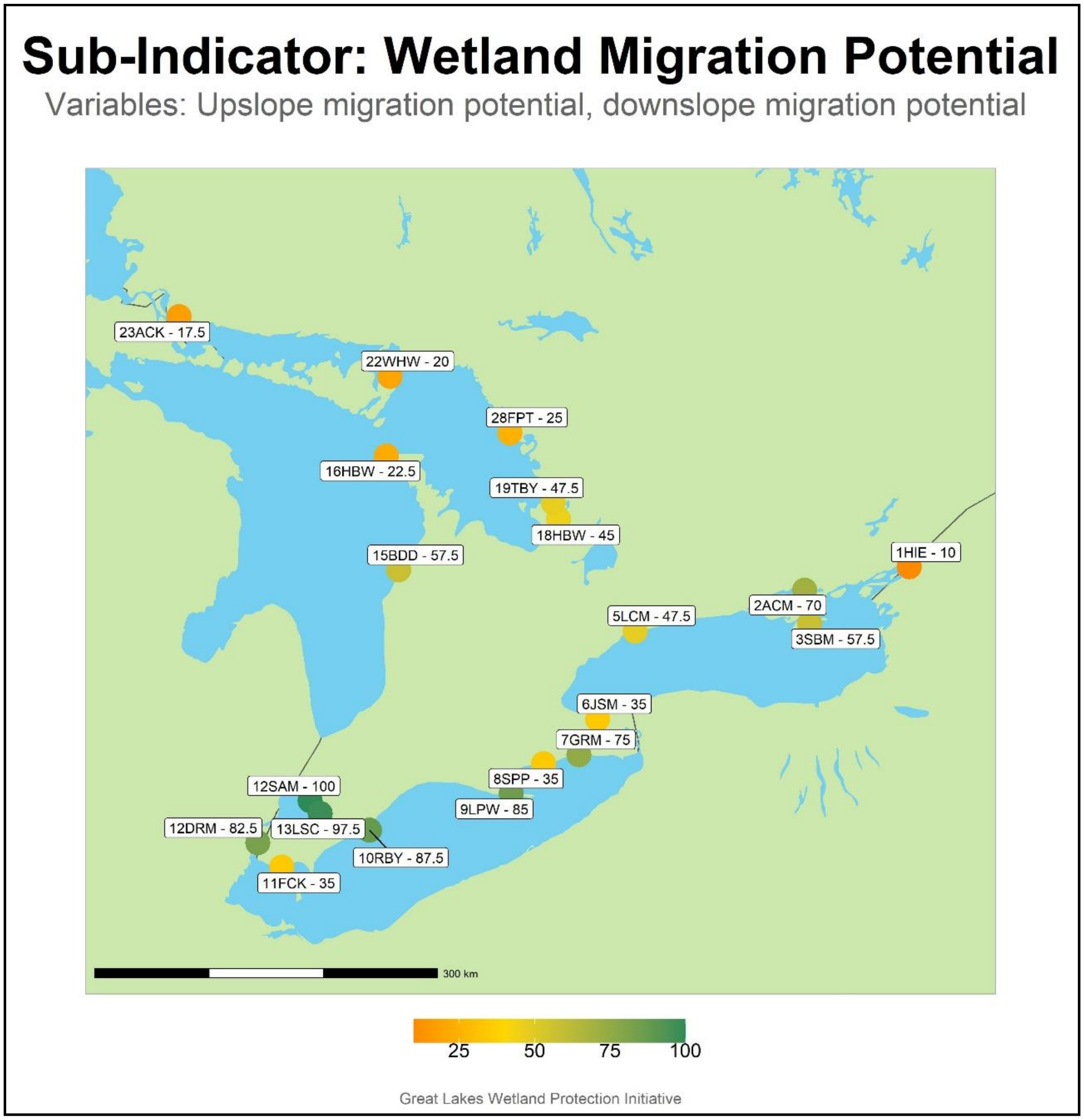


Figure 15. Map of selected Great Lakes Coastal Wetland sites displaying relative Wetland Migration Potential sub-indicator scores (from 10 - 100). The Wetland Migration Potential sub-indicator was composed upslope and downslope migration potential. Hill Island East (1HIE) received the lowest Wetland Migration Capacity Score (10), while Johnston Bay (12SAM) received the highest (100).

Protection

Seven of the 20 sites are considered to be protected and managed, as measured through the proportion of protected areas catalogued within the Canadian Protected and Conserved Areas Database (CPCAD). These seven sites are; Hay Bay Wetland (6HBW), Treasure Bay (19TBY), Long Point Wetland (9LPW), Lake St. Clair Wetland (13LSC), Selkirk Provincial Park (8SPP), Hill Island East (1HIE), and Rondeau Bay 10

RBY (Figure 16). Protected areas considered include National Parks (Georgian Bay Island, Fathom Five, Thousand Islands), Provincial Parks (Rondeau and Selkirk) and National Wildlife Areas (Long Point, Big Creek and St. Clair). If wetlands on First Nations' lands and provincial Conservation Areas met the criteria specified with the CPCAD decision-support tool, eight additional sites could be interpreted as being afforded some level of protection and management. Why these locations are not considered to be Protected Areas or Other Effective Area-based Conservation Measures (OECMs) should be determined. The remaining 13 sites not considered protected under the CPCAD received a score of zero (Figure 16).

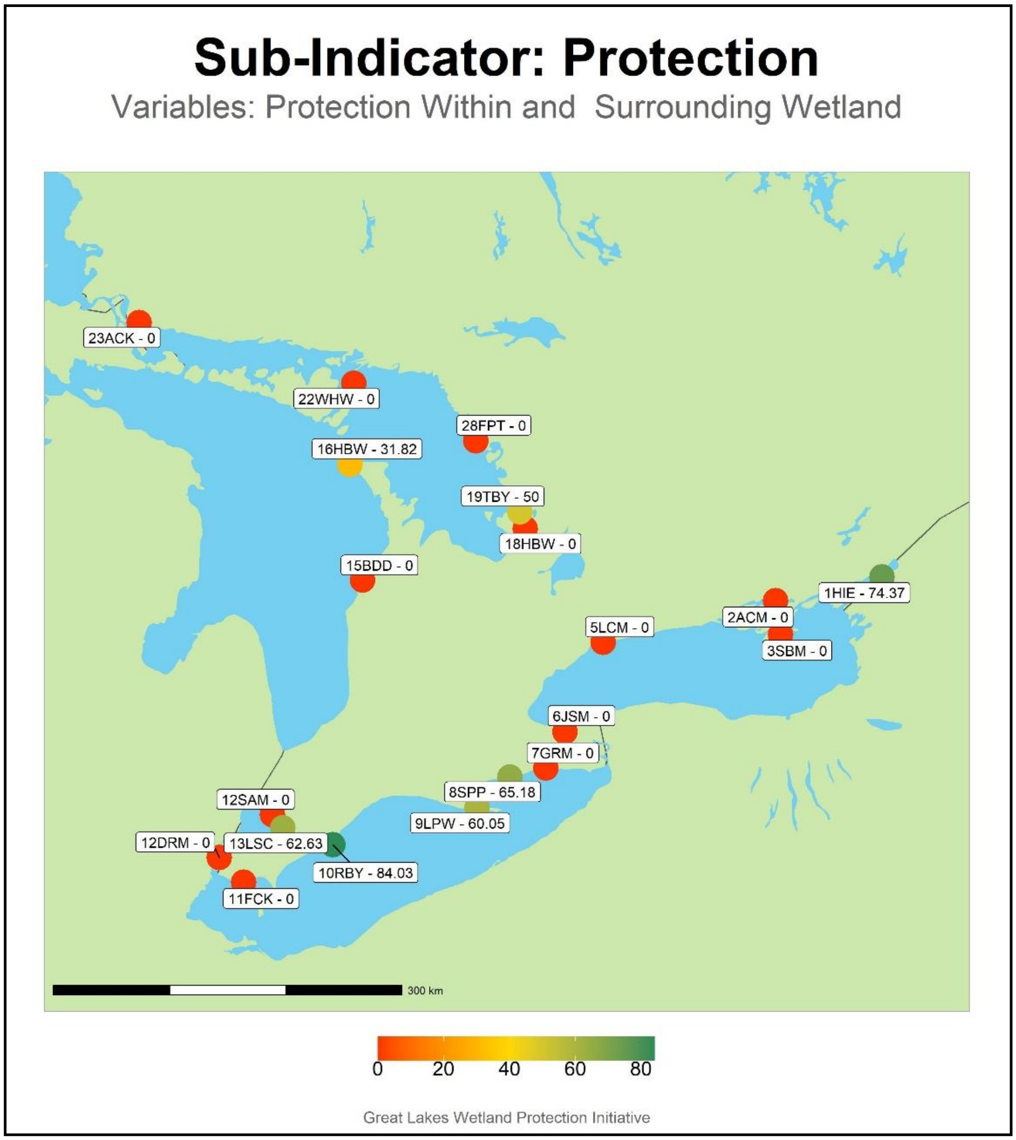


Figure 16. Map of selected Great Lakes Coastal Wetland sites displaying relative Protection sub-indicator scores (from 0 – 84.03). The protection sub-indicator was composed of Canadian Protected and Conserved Areas both within the wetland and surrounding it. Airport Creek Marsh (2ACM), South Bay Marsh (3SBM), Lynde Creek Marsh (5LCM), Jordan Station Marsh (6JSM), Grand River Mouth (7GRM), Fox Creek (11FCK), Detroit River Marshes (12DRM), Johnston Bay (12SAM), Bay du Dore (15BDD), Hay Bay Wetland (18HBY), Francis Point Wetland (28 FTP), Whiskey Harbor Wetland (22WHW), and Anderson Creek (23ACK) received scores of zero for Protection. Rondeau Bay (10 RBY) received the highest relative protection score of 84.03.

AC Level Trends and Other Observations

High Adaptive Capacity (AC Rank = 1 - 6)

The six highest ranked wetland sites are Treasure Bay (19TBY; 1), Long Point Wetlands (9LPW; 2), Airport Creek Marsh (2ACM; 3), Hay Bay Wetland (16HBW; 4), Baie Du Doré (15BDD; 5), and South Bay Marsh (3SBM; 6; Figure 17, Table 3). Three of the six highest-scoring sites are located on Lake Huron, including Treasure Bay, Hay Bay Wetland and Baie Du Doré (15BDD). All three sites received relatively high Biological Condition scores (97.1, 78.0 and 100, respectively). Treasure Bay and Hay Bay are also partially protected by their respective National Parks, but what Baie Du Doré lacks in protection it makes up for in its ability to migrate (57.5).

Though no indicator appears to be driving or biasing High Adaptive Capacity, it is worth noting that Biological Condition is the only indicator that scores high (> 50) across all sites that received a High score. For each site that received a High score, not all sub-indicators had to be high, indeed one or more of the sub-indicators scored mid-range to low, and this lower-scoring sub-indicator was varied across sites. For example, Treasure Bay and Hay Bay Wetland both scored mid-range to low with respect to migration potential (47.5 and 22.5, respectively). Long Point Wetland, scored low on Landscape Condition (30.7). Airport Creek Marsh, South Bay Marsh and Baie Du Doré scored zero on Protection.

The only site on Lake Erie to receive a High Adaptive Capacity score was Long Point (9LPW) due to its relatively high Biological Condition (69.7), potential to migrate (85.0) and level of protection (60.0). Airport Creek Marsh and South Bay Marsh were the only wetlands on Lake Ontario to receive a High Adaptive Capacity score. Both received relatively high Biological and Landscape Condition scores (86.4/60.0; and, 80.0/57.1, respectively), as well as relatively high potential to migrate (70 and 57.5, respectively). The Biological and Landscape Condition scores for these sites are in agreement with East-West water quality and land cover trends observed along the northern shoreline of Lake Ontario (Croft-White et al., 2017).

None of the wetland sites on Lake St. Clair or the connecting channels (i.e., St. Mary's River, Detroit River and St. Lawrence) had a High Adaptive Capacity Score. Although these sites did have a high potential to migrate, the wetlands in the Huron – Erie Corridor (i.e., Johnston Bay, Lake St. Clair Marshes, Detroit River Marshes) received some of the lowest Biological and Landscape Condition scores (21.5/11.9; 26.3/10.5; and, 41.5/0, respectively). Both Anderson Creek (St. Mary's River) and Hill Island East (St. Lawrence River) received high biological and landscape condition scores (89.8/73.6; and, 85.1/85.0, respectively), but being situated in bedrock geology appears to significantly hinder their ability to migrate (17.5 and 10, respectively). Similarly, no barred drowned river-mouths ranked high, mainly due to low/limited Wetland Migration Potential.

Moderate Adaptive Capacity (AC Rank = 7-13)

The seven wetland sites with Moderate Adaptive Capacity include Hill Island East (1HIE; 7), Hog Bay Wetland (18HBW; 8), Francis Point (28 FPT; 9), Whiskey Harbor Wetland (22WHW; 10), Anderson Creek (23ACK; 11), Grand River Mouth Wetlands (7GRM; 12), Selkirk Provincial Park (8SPP; 13) and Rondeau Bay (10RBY; 14; Figure 17, Table 3).

The four protected embayment wetlands in Lake Huron that received a Moderate Adaptive Capacity Score (Hog Bay Wetland, Francis Point and Whiskey Harbour Wetland and Anderson Creek) have similar Biological and Landscape Condition scores (88.7/57.1; 80.1/100; 96.0/99.6; and, 89.8/73.6, respectively). However, each received a Moderate Adaptive Capacity Score due to a lack of Protected Area (0 for all) and low ability to migrate (45, 25, 20 and 17.5, respectively). What appears to set Hill Island East apart from these wetlands with similar hydrogeomorphology is its level of protection (74.4).

The coastal wetlands in Lake Erie receiving a Moderate Adaptive Capacity Score (Grand River Mouth Wetlands, Selkirk Provincial Park and Rondeau Bay) all received low Landscape Condition scores (26.7, 10.6, and 3.9, respectively), but the other indicators offer compensation. The Wetland Migration Potential score (75) seems to raise the Grand River Mouth Wetlands above Selkirk and Rondeau Provincial Parks, but the Protection sub-indicator for each park (65.2 and 84.0, respectively) appears to be what is keeping them from receiving a low Adaptive Capacity Score. It is also worth noting that despite receiving such Low Landscape Condition scores, Selkirk and Rondeau Provincial Parks received relatively high Biological Condition scores (84.5 and 64.0, respectively), which suggests that species richness or the management of *Phragmites* may be benefitting from their protection status.

Low (AC Rank = 15 – 20)

The six sites that received the lowest Adaptive Capacity Scores were: Jordan Station Marsh (6JSM; 15), Lynde Creek Marsh (5LCM; 16), Johnston Bay (12SAM; 17), Detroit River Marshes (12DRM; 18), Lake St. Clair Marshes (13LSC; 19), and Fox Creek/Dolson's Creek Marsh (11FCK; 20; Figure 17, Table 3). Sites that were determined to have a Low Adaptive Capacity scored low for two or more sub-indicators. For example, Fox Creek/ Dolson's Creek Marsh scored poorly on Landscape Condition (7.34), Wetland Migration Potential (35) and Protection (0).

In the event that one sub-indicator was high for sites with Low Adaptive Capacity, this was not enough to compensate for multiple lower scoring sub-indicators, suggesting that geometric aggregation of migration potential, landscape condition and biological condition was sufficient to prevent overcompensation of a single sub-indicator. For example, all sites in the Huron – Erie Corridor received relatively high Wetland Migration Potential scores (Johnston Bay, Detroit River Marshes and Lake St. Clair Marshes; 100, 82.5 and 97.5, respectively), but scored poorly for both Biological and Landscape Condition.

For Lake Ontario, only sites in the Western basin received a Low Adaptive Capacity score (Jordan Station and Lynde Creek Marsh).This was due to their poor Landscape Condition scores (17.20, and 9.90; respectively) and lack of Protection (0 for both), as measured through the CPCAD. Again, these results are consistent with the East - West water quality and land cover trends observed along the Northern shoreline of Lake Ontario (Croft-White et al., 2017).

Deltaic (Johnson Bay) and open shoreline wetlands (Detroit River Marshes and Lake St. Clair Marshes) received some of the highest migration potential scores (100, 82.5, and 97.5; respectively). All three sites are located in the St. Clair Clay Plains, a region of very little topographic and bathymetric relief (Chapman & Putnam, 1966), which would support downslope and upslope wetland migration. However, this region is also highly agriculturalized because of its fertile soils and temperate climate (Chapman & Putnam, 1966). This likely contributes to relatively low Landscape and Biological Condition scores across all three sites, as well as Fox Creek/Dolson’s Creek Marsh.

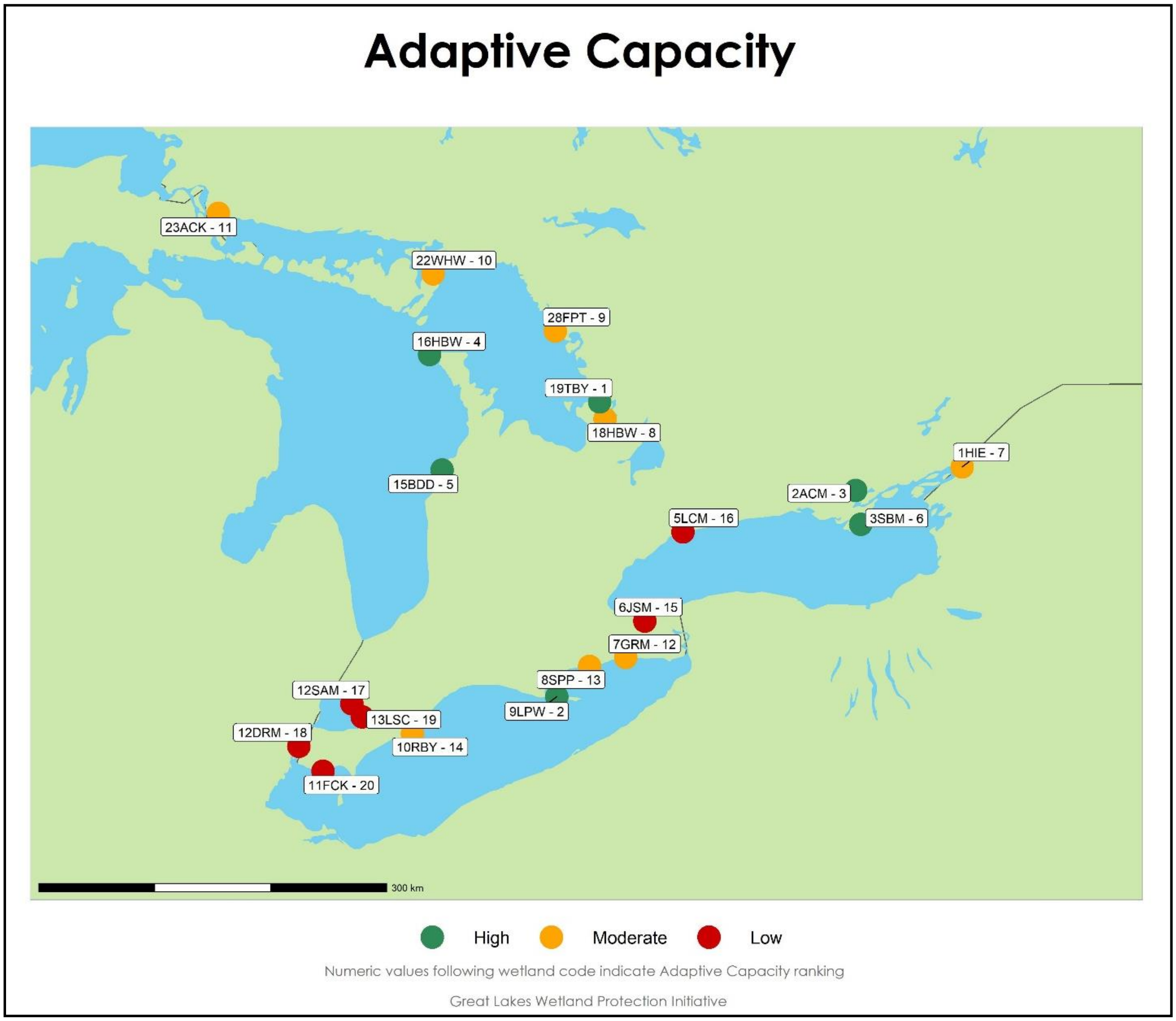


Figure 17. Map of selected Great Lakes Coastal Wetland sites displaying relative Adaptive Capacity score (High, Moderate, Low) and Rank. Treasure Bay (19 TBY) received the highest Adaptive Capacity Rank, while Fox Creek/ Dolson's Creek Marsh (11FCK) received the lowest Adaptive Capacity Rank.

Table 3. Table displaying Adaptive Capacity Level, Adaptive Capacity Rank, Adaptive Composite Indicator Score, and Sub-indicator Scores for Landscape Condition, Biological Condition, Migration Potential, and Protection according to Wetland Unique ID.

Wetland ID	Wetland Name	Adaptive Capacity Level	Adaptive Capacity Rank	Adaptive Capacity Composite Indicator Score	Landscape Condition Sub-Indicator Score	Biological Condition Sub-Indicator Score	Migration Potential Sub-Indicator Score	Protection Sub-Indicator Score
1HIE	Hill Island East	M	7	0.69	85.00	85.18	10.00	74.37
2ACM	Airport Creek Marsh	H	3	0.79	60.00	86.41	70.00	0.00
3SBM	South Bay Marsh	H	6	0.70	57.10	72.00	57.50	0.00
5LCM	Lynde Creek Marsh	L	16	0.28	9.90	70.09	47.50	0.00
6JSM	Jordan Station Marsh	L	15	0.29	17.20	56.49	35.00	0.00
7GRM	Grand River Mouth Wetlands	M	12	0.50	26.70	46.41	75.00	0.00
8SPP	Selkirk Provincial Park	M	13	0.49	10.60	84.52	35.00	65.18
9LPW	Long Point Wetlands	H	2	0.84	30.70	69.72	85.00	60.05
10RBY	Rondeau Bay	M	14	0.47	3.90	63.88	87.50	84.03
11FCK	Fox Creek / Dolson's Creek	L	20	0.00	7.34	33.96	35.00	0.00
12DRM	Detroit River Marshes	L	18	0.20	10.50	26.37	82.50	0.00
12SAM	Johnston Bay	L	17	0.23	11.90	21.50	100.00	0.00
13LSC	Lake St. Clair Marshes	L	19	0.05	0.00	41.55	97.50	62.63
15BDD	Baie Du Doré	H	5	0.70	41.50	100	57.50	0.00

16HBW	Hay Bay Wetland	H	4	0.73	93.60	78.09	22.50	31.82
18HBW	Hog Bay Wetland	M	8	0.69	57.10	88.72	45.00	0.00
19TBY	Treasure Bay	H	1	1.00	97.60	97.12	47.50	50.00
22WHW	Whiskey Harbour Wetland	M	10	0.65	99.60	96.02	20.00	0.00
23ACK	Anderson Creek	M	11	0.55	73.60	89.82	17.50	0.00
28FPT	Francis Point	M	9	0.66	100.00	80.09	25.00	0.00

Conclusion

The objective of this technical report was to operationalize the concept of wetland Adaptive Capacity and to rank the Adaptive Capacity of 20 Great Lakes Costal Wetlands. This task was accomplished using a composite indicator aggregation of eight variables. Variables considered in this analysis were selected and supported for their contribution to ecosystem stability and plasticity using available literature. However, several applicable variables such as water quality or sediment dynamics could not be included due to time and budget constraints as well as data availability. Future assessments may consider additional variables. Adaptive Capacity sub-indicator scores and composite indicator ranks are based on observed values, therefore ranking and sub-indicator scores are relative. Should this assessment be replicated with the addition or removal of wetland site, the ranking and sub-indicator scores will change.

The methodology outlined in this technical report was designed with the end goal of producing results that could be combined with Wetland Sensitivity indicator values to inform Wetland Vulnerability. However, the results of this analysis can also function as a stand-alone assessment and be used to inform adaptive management strategies. The final Adaptive Capacity Score represents the theoretical ability for a coastal wetlands to adapt to climate change through a relative comparison of the aggregated variables. Categorical scores can be used to help wetland managers and stakeholders identify coastal wetlands with poor Adaptive Capacity. Furthermore, reviewing the underlying sub-indicators and variables that contributed to these scores can help inform the development and prioritization of adaptational management of wetlands.

Although this study only considered 20 GLCWs, the resulting Adaptive Capacity scores can be used to identify priority locations for climate change adaptation across the Great Lakes. For coastal wetlands not considered in this study, it may be possible to compare proximate wetland location and hydrogeomorphology to infer if similar factors influencing Adaptive Capacity are present.

Site-Specific Adaptive Capacity Ranking

High Adaptive Capacity Sites

Treasure Bay (19TBY; AC Rank: 1)

Treasure Bay received the highest Adaptive Capacity ranking due to high Biological and Landscape Condition scores (97.1 and 97.6, respectively; Figure 18). As a protected embayment on Lake Huron, it has a relatively poor Wetland Migration Potential (47.5), but this appears to be offset by its level of Protection (50) within Georgian Bay Islands National Park.

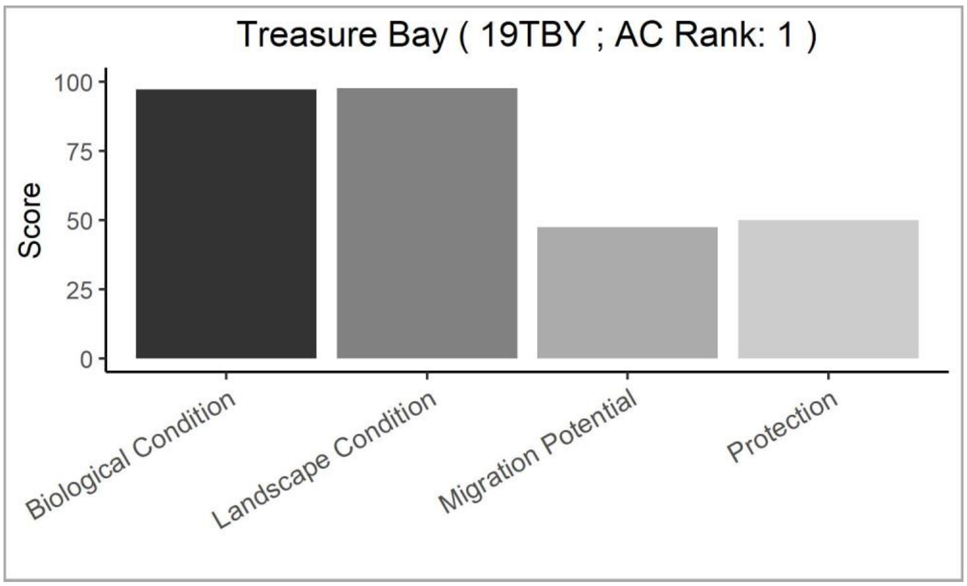


Figure 18. Bar graph displaying sub-indicator scores for Treasure Bay (10TBY). 10 TBY received an Adaptive Capacity score and rank of High, and 1, respectively.

Long Point Wetlands (9LPW; AC Rank: 2)

The Long Point Wetlands received the second highest Adaptive Capacity rank due to its high Biological Condition (69.7) and high Wetland Migration Potential (85) as a sand-spit embayment. The level of protection afforded by the two National Wildlife Areas (Big Creek and Long Point) appears to offset the poor Landscape Condition score of this site (30.7; Figure 19). If Big Creek National Wildlife Area were excluded from the wetland area of interest the Landscape Condition score of this site would likely increase, but the relative level of protection may decrease.

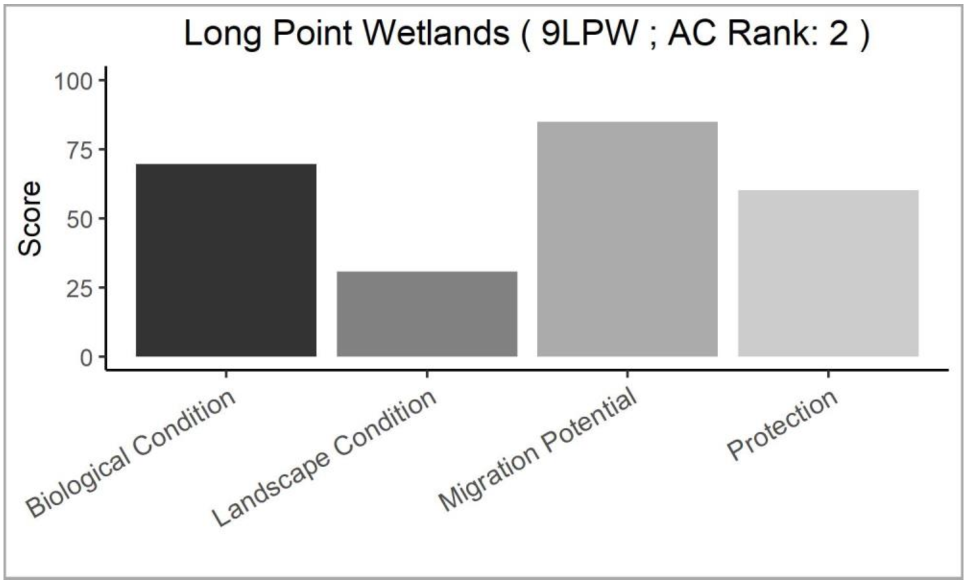


Figure 19. Bar graph displaying sub-indicator scores for Long Point Wetland (9LPW). 9LPW received an Adaptive Capacity score and rank of High, and 2, respectively

Airport Creek Marsh (2ACM; AC Rank: 3)

Although it is not considered to be protected, Airport Creek Marsh received the third highest Adaptive Capacity score due to its high Biological and Landscape Condition scores (86.4 and 60.0, respectively), as well as its relatively high Wetland Migration Potential (70; Figure 20).

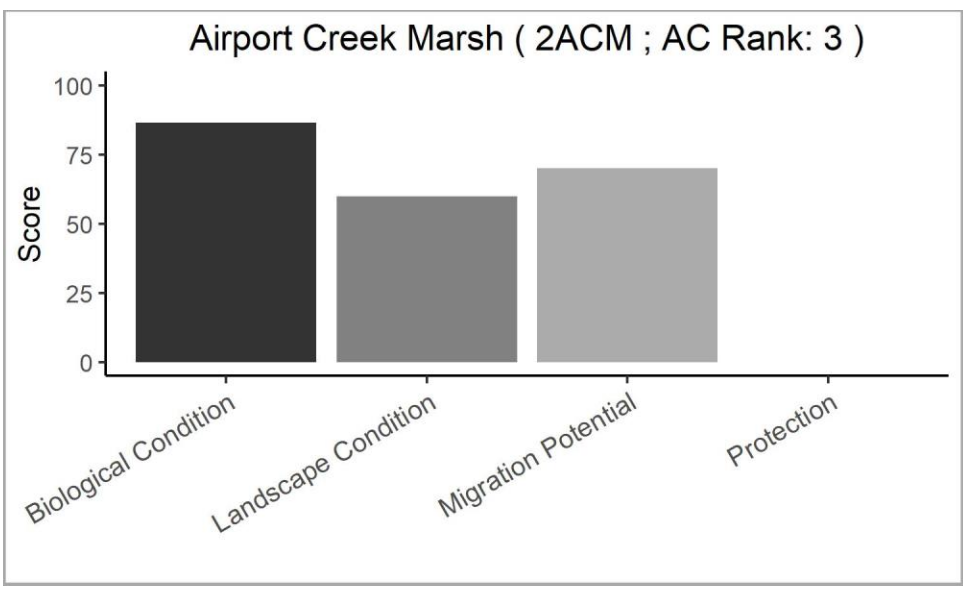


Figure 20. Bar graph displaying sub-indicator scores for Airport Creek Marsh (2ACM). 2ACM received an Adaptive Capacity score and rank of High, and 3, respectively

Hay Bay Wetland (18HBW; AC Rank: 4)

Hay Bay received the fourth highest rank due to its high Biological and Landscape condition scores (78.1 and 93.9, respectively). The level of Protection afforded by Fathom Five National Marine Park appears to compensate for the limited potential for this protected embayment to migrate (22.5; Figure 21).

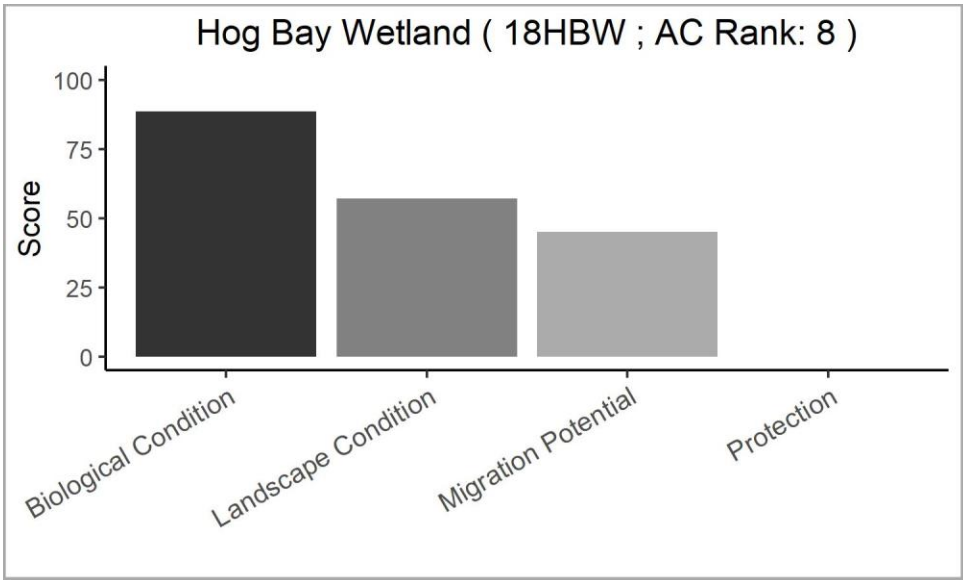


Figure 21. Bar graph displaying sub-indicator scores for Hog Bay Wetland (18HBW). 18HBW received an Adaptive Capacity score and rank of High, and 4, respectively

Baie Du Doré (15BDD; AC Rank: 5)

Baie Du Doré received the highest score for biological condition (100). This appears to compensate for modest Wetland Migration Potential (57.5) and Landscape Condition scores (41.5). Although Baie Du Doré is not considered to be protected, it might be worth investigating the *Phragmites* management techniques employed by Bruce Power to determine whether these have contributed to high species richness or low *Phragmites* coverage.

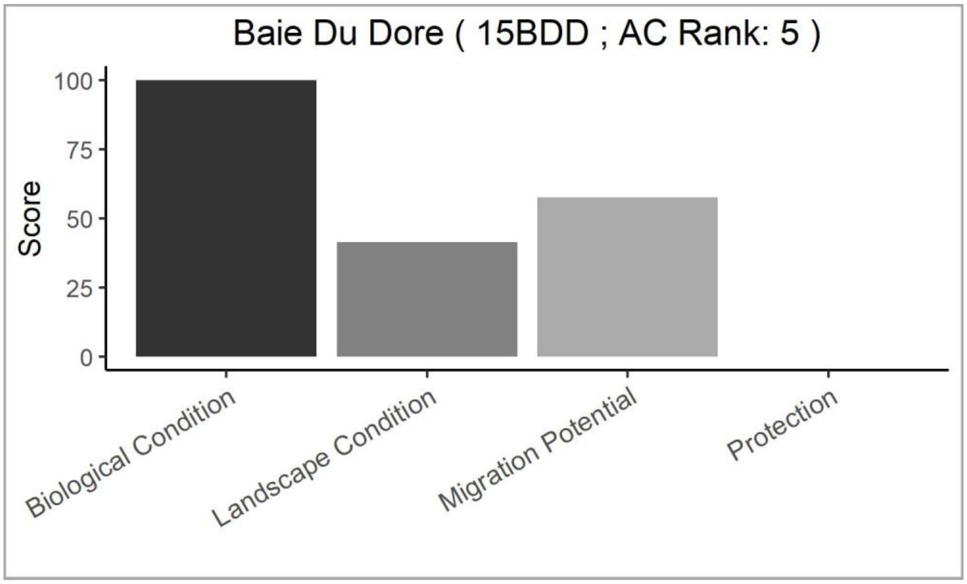


Figure 22. Bar graph displaying sub-indicator scores for Baie Du Dore (15BDD). 15BDD received an Adaptive Capacity score and rank of High, and 5, respectively

South Bay Marsh (3SBM; AC Rank: 6)

South Bay Marsh received a relatively high Biological Condition score (72.0), and modest Landscape Condition and Wetland Migration potential scores (57.1 and 57.5, respectively). South Bay Marsh is not considered to be protected, as measured through the CPCAD.

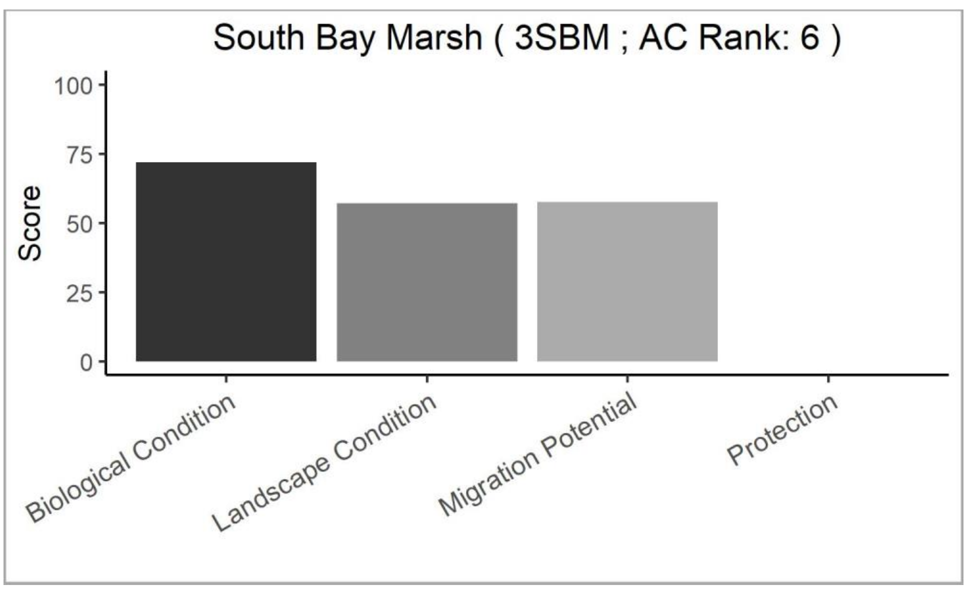


Figure 23. Bar graph displaying sub-indicator scores for South Bay Mouth (3SBM). 3SBM received an Adaptive Capacity score and rank of High, and 6, respectively

Moderate Adaptive Capacity Sites

Hill Island East (1HIE; AC Rank: 7)

If it were not for its low Migration Potential score (10), Hill Island East would likely receive a higher Adaptive Capacity rank. Hill Island East received high Biological and Landscape Condition scores (85.2 and 85.0, respectively), and received considerable Protection from Thousand Islands National Park (74.4).

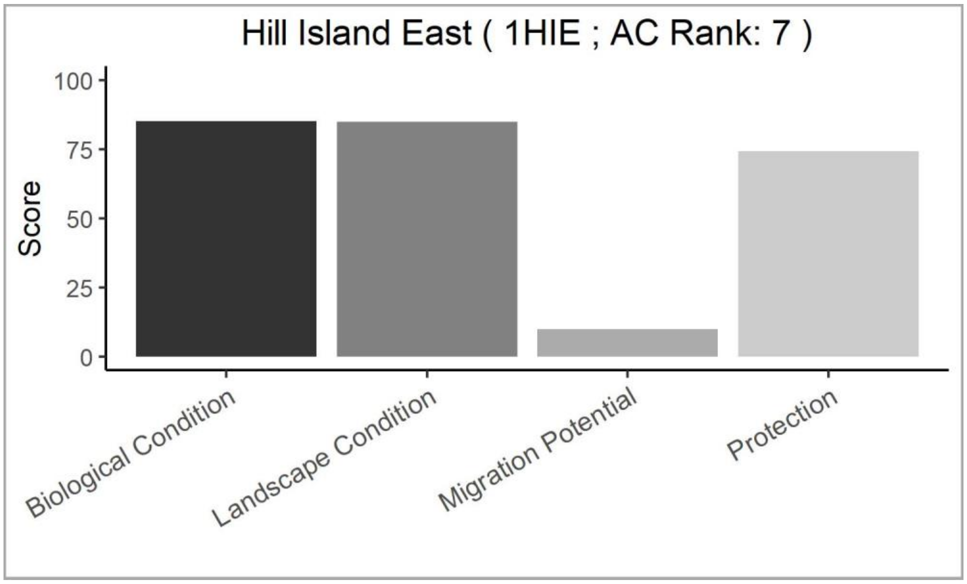


Figure 24. Bar graph displaying sub-indicator scores for Hill Island East (1HIE). 1HIE received an Adaptive Capacity score and rank of Moderate, and 7, respectively

Hog Bay Wetland (18HBY; AC Rank: 8)

Hog Bay scored eighth and received a high Biological Condition score (88.7), as well as modest Landscape Condition and Migration Potential scores (57.1 and 45, respectively). Hog Bay is not considered to be protected, as measured through the CPCAD.

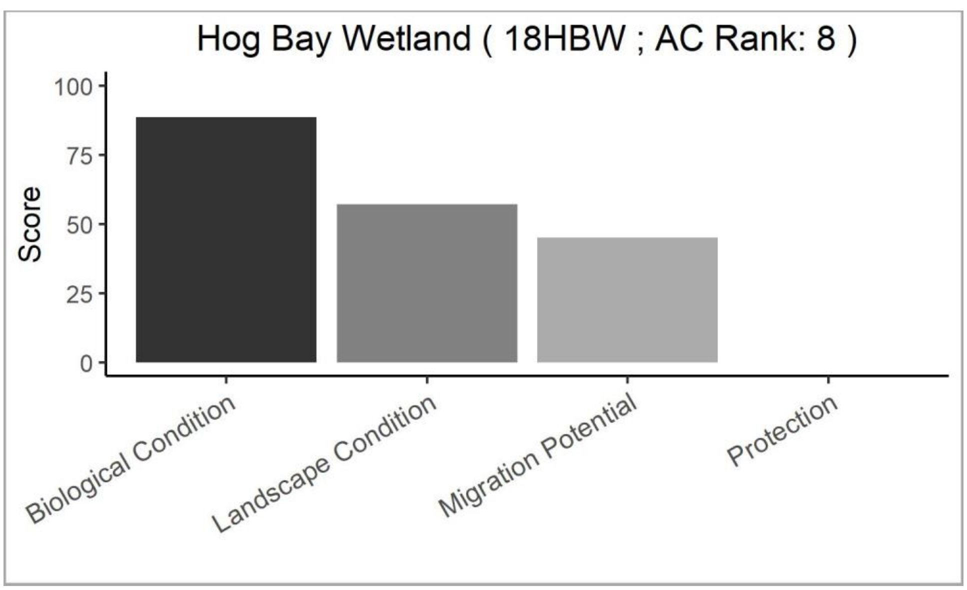


Figure 25. Bar graph displaying sub-indicator scores for Hog Bay Wetland (18HBW). 18HBW received an Adaptive Capacity score and rank of Moderate, and 8, respectively

Francis Point (28FPT; AC Rank: 9)

Situated in an undeveloped portion of the eastern Georgian Bay archipelago, Francis Point received the highest Landscape Condition score (100), and a high Biological Condition score (80.1). However, it receives a modest Adaptive Capacity Rank due to its inability to migrate (25) and its lack of Protection (0.00).

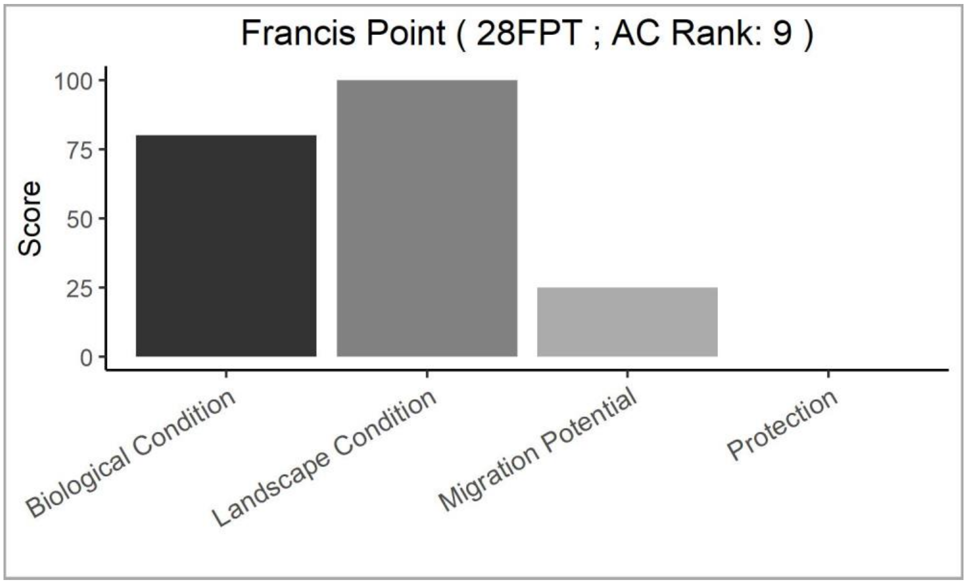


Figure 26. Bar graph displaying sub-indicator scores for Francis Point (28 FPT). 28FPT received an Adaptive Capacity score and rank of Moderate, and 9, respectively

Whiskey Harbour Wetland (22WHW; AC Rank: 10)

Whiskey Harbour received the second highest biological condition score (96.0), but it received lower Landscape Condition and Wetland Migration scores than Francis Point (99.6 and 20, respectively). Whiskey Harbour is also not considered to be protected, as measured through the CPCAD.

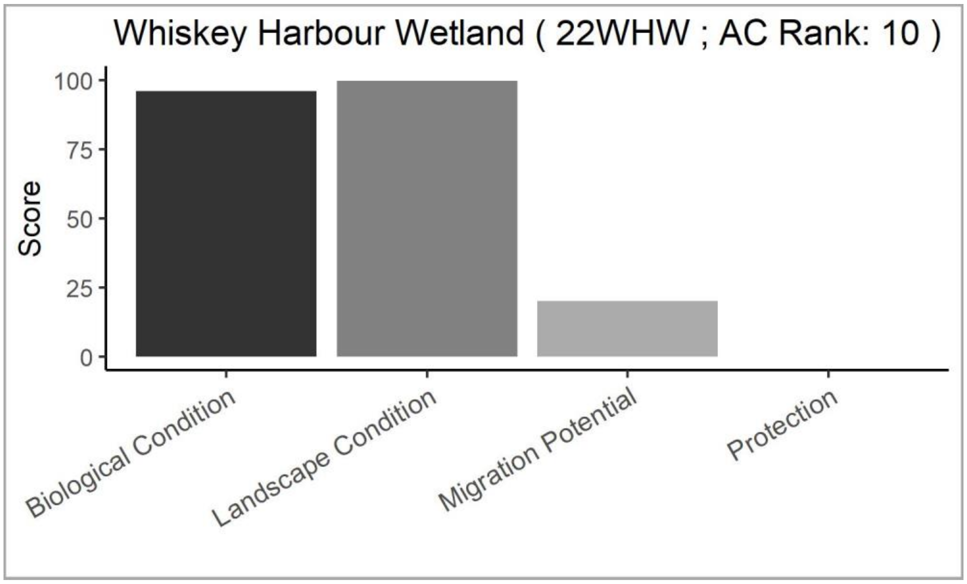


Figure 27. Bar graph displaying sub-indicator scores for Whiskey Harbour Wetland (22WHW). 22WHW received an Adaptive Capacity score and rank of Moderate, and 10, respectively

Anderson Creek (23ACK; AC Rank: 11)

Anderson Creek received a high score for Biological Condition (89.8) and a moderate score for Landscape Condition (73.6), however, its Adaptive Capacity score was hindered by a low Migration Potential (17.5) and no Protection (0), as measured through the CPCAD.

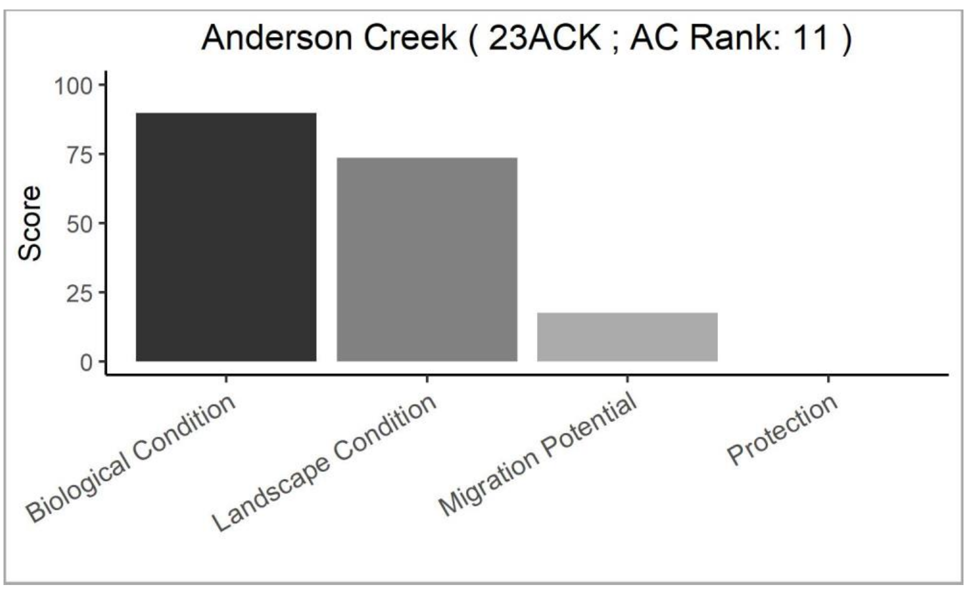


Figure 28. Bar graph displaying sub-indicator scores for Anderson Creek (23ACK). 23ACK received an Adaptive Capacity score and rank of Moderate, and 11, respectively

Grand River Mouth Wetlands (7GRM)

The Grand River Mouth Wetlands received a moderately high Wetland Migration score (75), but this does not appear to compensate for the relatively low Biological and

Landscape Condition scores (46.4 and 26.7, respectively). The Grand River Mouth Wetlands are also not considered to be protected, as measured through the CPCAD.

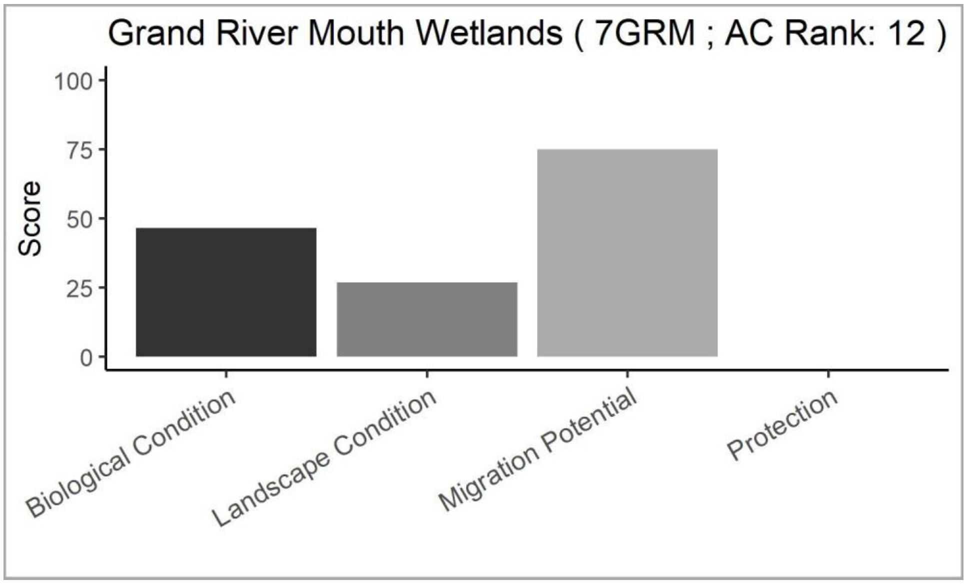


Figure 29. Bar graph displaying sub-indicator scores for Hill Island East (1HIE). 1HIE received an Adaptive Capacity score and rank of Moderate, and 12, respectively

Selkirk Provincial Park (8SPP; AC Rank: 13)

As a barred drowned river-mouth in an agricultural landscape, Selkirk Provincial Park received low scores for Landscape Condition (10.6) and Wetland Migration Potential (35). These scores appear to be somewhat compensated by a high biological condition score (84.5) and the level of protection the Provincial Park affords (65.2).

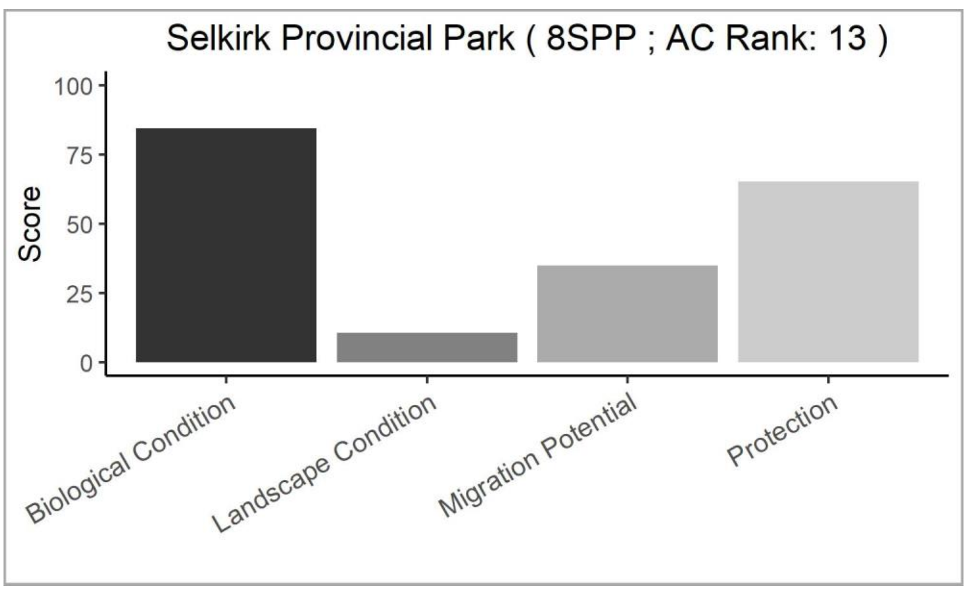


Figure 30. Bar graph displaying sub-indicator scores for Selkirk Provincial Park (8SPP). 8SPP received an Adaptive Capacity score and rank of Moderate, and 13, respectively

Rondeau Provincial Park (10RBY; AC Score: 14)

The highly agricultural landscape north of the Rondeau Peninsula appears to be compromise its Landscape Condition score (3.9). Relatively high Protection (84.0), Migration Potential (87.5) and Biological Condition scores (64.0) do not seem capable of fully compensating for this.

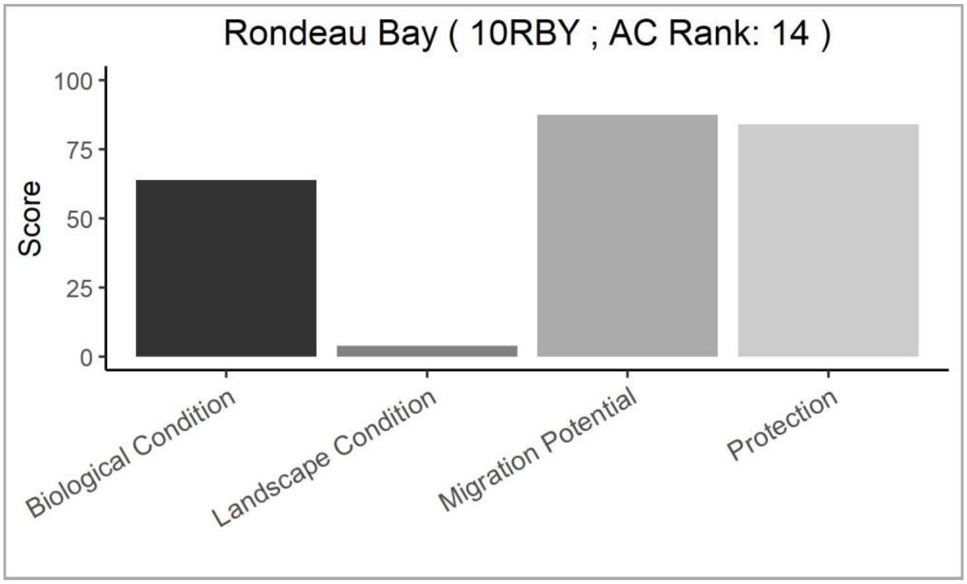


Figure 31. Bar graph displaying sub-indicator scores for Rondeau Bay (10RBY). 10RBY received an Adaptive Capacity score and rank of Moderate, and 14, respectively

Low Adaptive Capacity Sites

Jordan Station Marsh (6JSM; AC Rank: 15)

Jordan Station Marsh received a moderate Biological Condition score (56.4), but received relatively low scores for Landscape Condition (17.2) and Migration Potential (35.0). It is also not considered to be protected, as measured through the CPCAD.

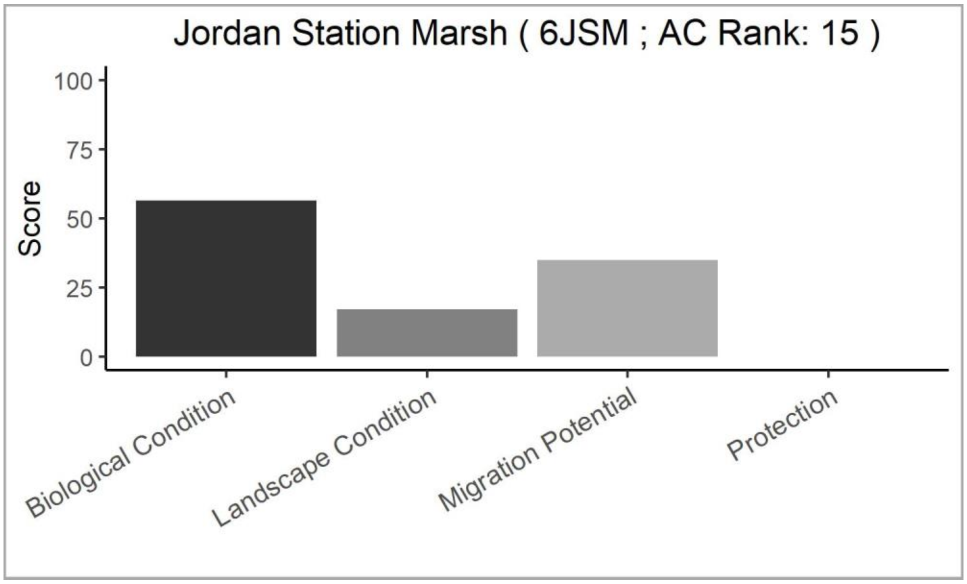


Figure 32. Bar graph displaying sub-indicator scores for Jordan Station Marsh (6JSM). 6JSM received an Adaptive Capacity score and rank of Low , and 15, respectively

Lynde Creek Marsh (5LCM; AC Rank: 14)

Lynde Creek Marsh received higher Biological Condition and Migration Potential scores than Jordan Station Marsh (70.1 and 47.5 vs. 56.5 and 25.0), but received a lower Landscape Condition score (9.9 vs 17.2). Lynde Creek Marsh is also not considered to be protected, as measured through the CPCAD.

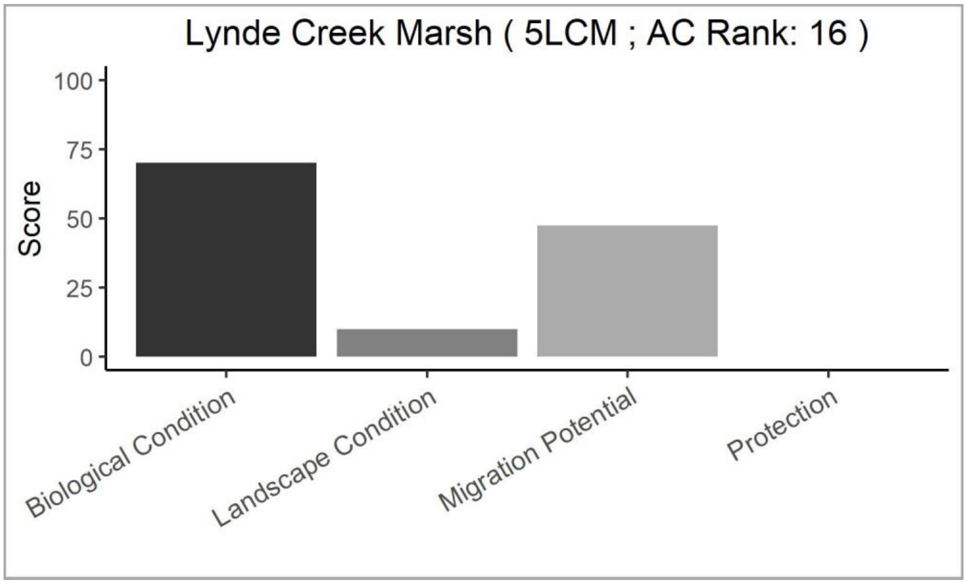


Figure 33. Bar graph displaying sub-indicator scores for Lynde Creek Marsh (5LCM). 5LCM received an Adaptive Capacity score and rank of Low, and 16, respectively

Johnston Bay (12SAM; AC Rank: 17)

Being a shallow, deltaic wetland, Johnson Bay received the highest Migration Potential score (100); however, this does not appear to compensate for its low Biological Condition (21.5) and Landscape Condition (11.9) scores. Johnston Bay is also not considered to be protected, as measured through the CPCAD.

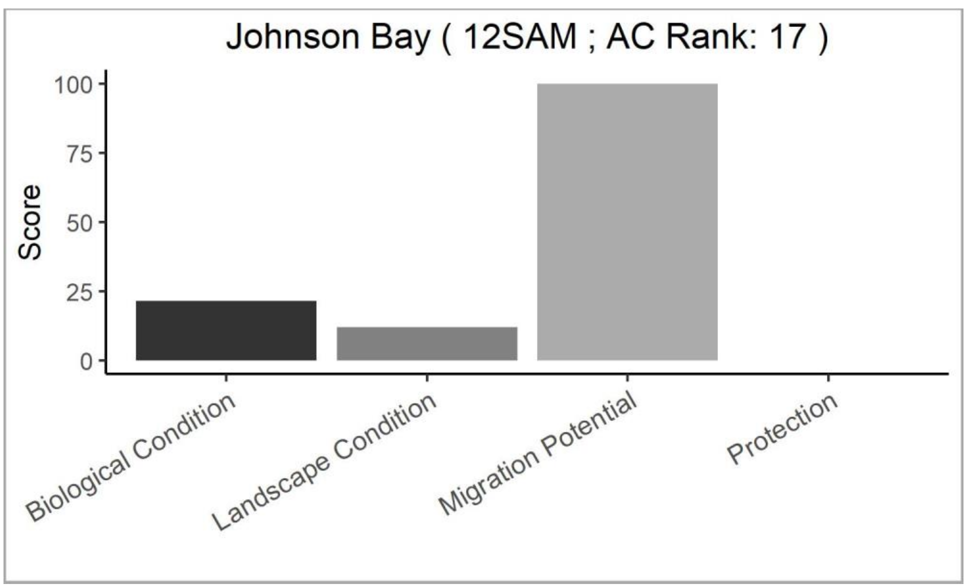


Figure 34. Bar graph displaying sub-indicator scores for Johnson Bay (12SAM). 12SAM received an Adaptive Capacity score and rank of Low, and 17, respectively

Detroit River Marshes (12DRM; AC Rank: 18)

Similar to Johnston Bay the Detroit River Marshes received a high Migration Potential score (82.5), but scored poorly for Biological Condition (26.4) and Landscape

Condition (10.5). The Detroit River Marshes are also not considered to be protected, as measured through the CPCAD.

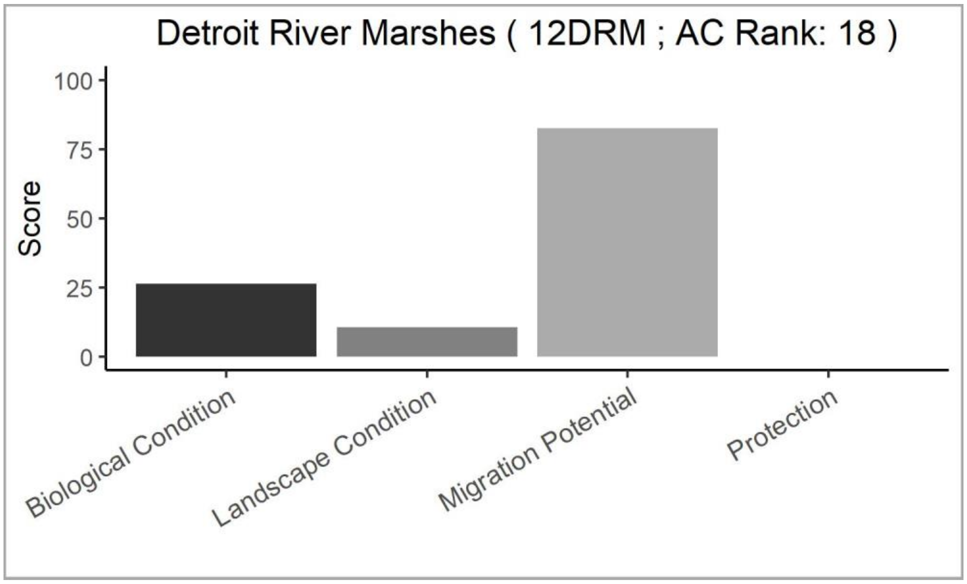


Figure 35. Bar graph displaying sub-indicator scores for Detroit River 8arsh (12DRM). 12DRM received an Adaptive Capacity score and rank of Low , and 15, respectively

Lake St. Clair Marshes (13LSC; AC Rank: 19)

The Lake St. Clair Marshes received a high Migration Potential score (97.5), but received low Landscape and Biological Condition scores (0 and 41.5, respectively). The level of Protection (62.6), does not appear to compensate for such a low Landscape Condition score.

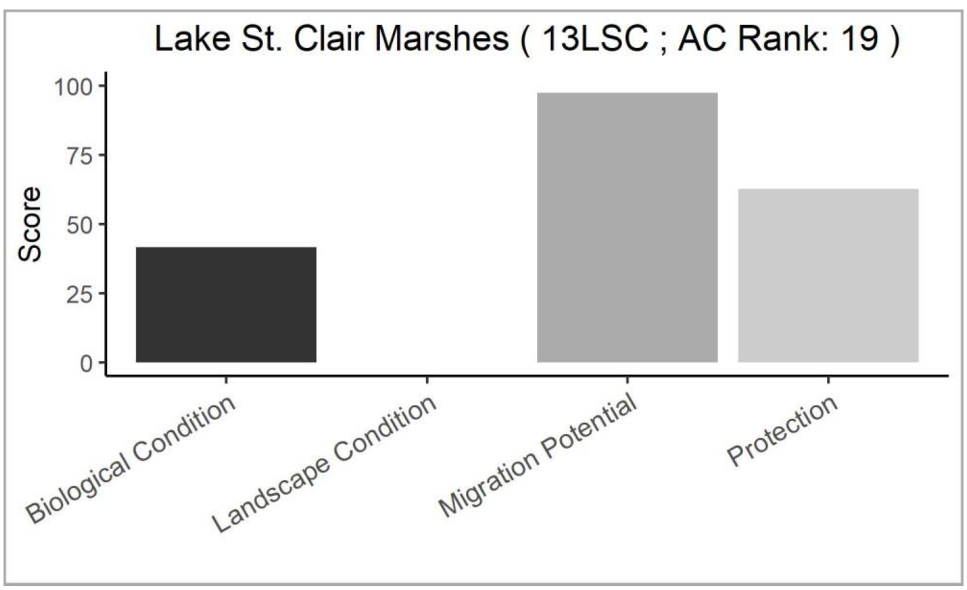


Figure 36. Bar graph displaying sub-indicator scores for Lake St. Clair Marshes (13LSC). 13LSC received an Adaptive Capacity score and rank of Low , and 19, respectively

Fox/ Dolson's Creek Marsh (11FCK; AC Rank: 20)

Fox Creek/ Dolson's Creek Marsh received the lowest Adaptive Capacity score of all sites within this analysis. It received low Biological and Landscape Condition scores (34.0, and 7.3; respectively). Fox Creek also received a low Migration Potential score (35), presumably because nearshore energy at the mouth of Fox and Dolson's Creeks would impede lakeward migration. Finally, Fox Creek/ Dolson's Creek Marsh was not considered to be protected, as measured through the CPCAD.

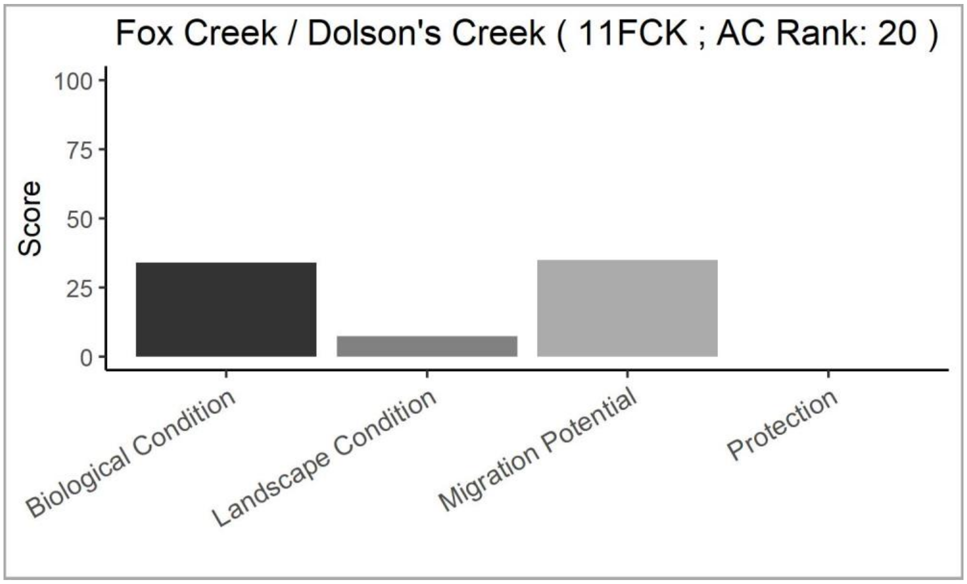


Figure 37. Bar graph displaying sub-indicator scores for Fox/ Dolson Creek Marsh (11FCK). 11FCK received an Adaptive Capacity score and rank of Low , and 20, respectively

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