



Assessing the Sensitivity
**OF GREAT LAKES COASTAL
WETLANDS TO CLIMATE CHANGE**



Government
of Canada

Gouvernement
du Canada

Canada

Cat. No.: CW66-778/2022E-PDF
ISBN: 978-0-660-43560-2
EC22022

Unless otherwise specified, you may not reproduce materials in this publication, in whole or in part, for the purposes of commercial redistribution without prior written permission from Environment and Climate Change Canada's copyright administrator. To obtain permission to reproduce Government of Canada materials for commercial purposes, apply for Crown Copyright Clearance by contacting:

Environment and Climate Change Canada
Public Inquiries Centre
12th Floor, Fontaine Building
200 Sacré-Coeur Boulevard
Gatineau QC K1A 0H3
Telephone: 819-938-3860
Toll Free: 1-800-668-6767 (in Canada only)
Email: enviroinfo@ec.gc.ca

Cover photo: © Environment and Climate Change Canada

© His Majesty the King in Right of Canada, represented by the Minister of Environment and Climate Change, 2022

Aussi disponible en français

Recommended Citation:

Environment and Climate Change Canada. 2022. Assessing the Sensitivity of Great Lakes Coastal Wetlands to Climate Change. Quesnelle, P., Spencer, N., Abdulhamid, N., Denomme-Brown, S., Rivers, P., Hrynyk, M., Fiorino, G, Grabas, G. 72p.

Acknowledgements

Assessing the Sensitivity of Great Lakes Coastal Wetlands to Climate Change was a priority under the 2017-2022 Great Lakes Protection Initiative to address the most pressing challenges affecting Great Lakes water quality and ecosystem health.

We acknowledge the efforts of our writing team: Pauline Quesnelle, Nora Spencer, Nash Abdulhamid, Simon Denomme-Brown, Patrick Rivers, Morgan Hrynyk, Giuseppe Fiorino, and Greg Grabas.

We also gratefully acknowledge those that supported study site access, including Wikwemikong First Nation, Walpole Island First Nation, the Mohawks of the Bay of Quinte, Essex Region Conservation Authority, Grand River Conservation Authority, Niagara Peninsula Conservation Authority, Central Lake Ontario Conservation Authority, Bruce Power, Ontario Power Generation, Ontario Parks, Parks Canada, and landowners adjacent to Anderson Creek, Georgian Bay, Lake St. Clair, the Detroit River, Fox/Dolson's Creek, Jordan Station, South Bay, and the lower Grand River.

Lastly, we thank all of the wetland experts, academic reviewers, and stakeholders that provided input and feedback that contributed to this project.

Executive summary

Coastal wetlands are dynamic and complex environments that are critical ecological and cultural components of the Laurentian Great Lakes. These systems have many important functions, including providing breeding and migratory habitat for wildlife and spawning and nursery habitat for fish. Coastal wetlands also provide many ecosystem services, including nutrient and sediment retention and protection against shoreline erosion.

Great Lakes coastal wetlands have been impacted by various human-related stressors, including nutrient and sediment loading, fragmentation, invasive species, shoreline alteration, and water-level regulation. These stressors reduce available habitat for fish and wildlife, interfere with biophysical processes, and ultimately threaten many of the services provided by Great Lakes coastal wetlands.

Future climate change represents an additional threat to Great Lakes coastal wetlands. By the end of the century, significant changes in climate are expected across the Great Lakes, including increases in over-land air temperatures, over-lake precipitation, lake water levels, and lake-level variability. These changes are expected to have environmental consequences in the Great Lakes basin. For example, changes in water levels may influence the structure and function of coastal wetlands, which may have significant implications for wildlife populations and habitat.

Environment and Climate Change Canada (ECCC) completed a study to assess the vulnerability of Great Lakes coastal wetlands to climate change, develop adaptive measures to build wetland resilience, and communicate results to wetland stakeholders and practitioners. This Climate Change Vulnerability Assessment considered how wetlands are expected to change in response to future environmental conditions driven by climate change (i.e., Sensitivity based on Exposure), plus the ability of the wetlands to continue to function despite those changes (i.e., Adaptive Capacity). This report describes the Sensitivity component of the overall Vulnerability Assessment.

Assessing Sensitivity involved estimating the adverse ecological effects of climate change on 20 Great Lakes coastal wetlands. We focused on five key ecological attributes of Great Lakes coastal wetlands: 1) total wetland area, 2) volume of the submerged and floating aquatic vegetation, 3) wetland interspersion, 4) wetland vegetation community diversity, and 5)

meadow marsh area. We used outputs from the Coastal Wetland Response Model (CWRM) developed by the Hydrodynamic and Ecohydraulic Section of ECCC's National Hydrological Services for each of the 20 coastal wetland sites. The CWRM integrated physical and ecological data to predict the response of wetland vegetation communities to climate change over time. These data included climatic exposure (e.g., temperature), modelled hydrodynamic data (e.g., water levels), surrounding land use, digital elevation models, and data collected from site-level vegetation surveys.

The CWRM used two down-scaled Global Climate Models under a mitigated greenhouse gas emission scenario of Representative Concentration Pathway 4.5. The two models (hereafter, simulations) represented different possible futures to account for uncertainty in climate projections; the lower-bound simulation and the upper-bound simulation. The lower-bound simulation had average water-level projections that were relatively similar to the projected recent past, while the upper-bound simulation projected an increase in water levels compared to the projected past. Importantly, both the lower and upper-bound simulations had increased overall variability in Great Lakes water levels compared to the projected past. For each simulation, the CWRM generated time-series outputs (one grid per year) for each of the 20 study sites. Each cell within the grid represented a 10 x 10 metre area of a particular vegetation community type. The time series was split into historic projections (hindcast years; 1981 to 2008) and future projections (forecast years; 2071 to 2098).

A response threshold was used to evaluate the negative effects of climate change on each ecological attribute. The response threshold was based on the 10th percentile in the hindcast time series for all ecological attributes (i.e., when an ecological attribute went below the lowest 10% of hindcast values); values below this threshold signified an extreme negative state. This 10th percentile threshold was applied to the forecast years to detect when extreme low values in each ecological attribute were predicted for each simulation. We then calculated Attribute Sensitivity as the proportion of extreme forecast years for each site and simulation for each ecological attribute. We also calculated overall Wetland Sensitivity for each site and simulation as the proportion of extreme forecast years across all ecological attributes. Wetland Sensitivity was classified into three categories indicating the risk that the wetland will experience negative effects of climate change (low risk, at risk, or critically at risk).

We found that area-based ecological attributes (meadow marsh area and total wetland area) were especially sensitive in the upper-bound simulation. Total wetland area in the upper-

bound simulation had the highest median Attribute Sensitivity (0.71), followed by meadow marsh area in the upper-bound simulation (0.61). Interspersion had the highest median Attribute Sensitivity in the lower-bound simulation (0.20), followed by SAV volume and meadow marsh area (both 0.16).

Wetland Sensitivity values were generally similar among water bodies in the lower-bound simulation, but were higher in the Huron-Erie Corridor, and to a lesser extent Lake Erie, in the upper-bound simulation. All three sites in the Huron-Erie Corridor were considered critically at risk in the upper-bound simulation. Four of five Lake Erie sites and six of seven Lake Huron sites were considered at risk in the upper-bound simulation, with both exceptions considered critically at risk. However, Lake Erie sites had higher Wetland Sensitivity values than Lake Huron sites, on average. Sites on Lake Ontario were the least sensitive; no site was considered critically at risk in either simulation. Additionally, there was no notable difference in Wetland Sensitivity between lacustrine and riverine wetland hydrogeomorphic types in either simulation.

Overall, we found that all coastal wetlands considered in this study were sensitive and are likely at risk to future climate change. Most notably, more frequent instances of extreme wetland loss are expected with higher lake levels where the surrounding upland is unsuitable for landward migration due to land use (e.g., transportation infrastructure, residential development) or topography (e.g., steep slopes precluding migration). This means that further land development surrounding coastal wetlands will likely exacerbate climate-driven wetland loss. Our results suggest that conservation planning for Great Lakes coastal wetlands under climate change should consider protection of the surrounding landscape. More broadly, given that coastal wetlands across the Great Lakes are continuing to be lost and degraded due to land development, the synergistic effects of climate change and land development (both resulting in habitat loss) should be considered in regional biodiversity planning and recovery strategies for wetland species at risk.

Table of Contents

Acknowledgements	i
Executive summary	ii
List of Figures.....	vii
List of Tables.....	ix
Introduction	1
Great Lakes coastal wetlands and climate change	1
Program purpose	2
Climate Change Vulnerability Assessments.....	2
Wetland Sensitivity.....	3
Ecological attribute selection	3
Rationales for selected ecological attributes	8
Total wetland area.....	8
Submerged and floating aquatic vegetation volume	10
Wetland interspersion.....	11
Wetland diversity.....	13
Meadow marsh area.....	15
Methods.....	17
Study sites.....	17
Data source.....	18
Processing of CWRM output	22
Calculating ecological attributes	22
Change-detection analysis	27
Threshold development.....	28
Hindcast verification.....	29
Attribute aggregation.....	30

Post-hoc analyses.....	31
Results.....	33
Ecological Attribute Sensitivity.....	33
Wetland Sensitivity.....	34
Post-hoc analyses.....	41
Discussion.....	49
Wetland Sensitivity to climate change: drivers	49
Uncertainty in Wetland Sensitivity estimates.....	52
Wildlife implications.....	53
Conclusions.....	54
References.....	55

List of Figures

Figure 1: Vegetation zonation in a typical coastal wetland. This arrangement of wetland plants is largely determined by moisture conditions.....	9
Figure 2: Diagram of a wetland with low and high interspersions (as viewed from above). Adapted from SOWES v3.2 (Ontario Ministry of Natural Resources 2013).....	12
Figure 3: Locations of the twenty coastal wetland sites used in the assessment of Wetland Sensitivity. Seven sites are in Lake Huron (13. Hay Bay, 14. Treasure Bay, 15. Whiskey Harbour, 16. Anderson Creek, 17. Hog Bay, 18. Baie du Doré, 20. Frances Point), three are in the Huron-Erie Corridor (3. Detroit River, 4. Lake St. Clair, 5. Johnston Bay), five are in Lake Erie (1. Rondeau Bay, 2. Fox/Dolson’s Creek, 6. Long Point, 7. Selkirk Provincial Park, 8. Grand River Mouth) four are in Lake Ontario (9. Jordan Station, 10. Lynde Creek, 11. South Bay, 12. Airport Creek), and one is in the upper St. Lawrence River (19. Hill Island East).....	17
Figure 4: Map showing meadow marsh area (yellow) and total wetland area (purple, excluding meadow marsh) at Lynde Creek, Lake Ontario.....	23
Figure 5: Map showing the depth of submerged aquatic vegetation (SAV; blue gradient), from which SAV volume was calculated, at Hill Island East, St. Lawrence River.....	24
Figure 6: Two maps showing submerged aquatic vegetation (blue) and all other vegetation communities (emergent marsh, meadow marsh, and swamp; yellow) in two different years at Lynde Creek, Lake Ontario. These vegetation groupings were used to calculate interspersions. Panel A shows a year with lower interspersions and panel B shows a year with higher interspersions.	25
Figure 7: Two maps showing four vegetation communities at Lynde Creek, Lake Ontario. These vegetation groupings were used to calculate wetland diversity. Panel A shows a year with lower wetland diversity (note there is very little submerged aquatic vegetation) and panel B shows a year with higher wetland diversity.....	26
Figure 8: Depiction of how the 10th percentile threshold was set based on the hindcast of an ecological attribute, and how this threshold was applied to the forecast. The horizontal dashed blue line is the 10th percentile threshold and red circles represent all years that fell below the threshold.....	28
Figure 9: Boxplots of Attribute Sensitivity values for all sites in upper and lower-bound simulations. The horizontal line in each box is the median, the bottom of each box is the 1st	

quartile, the top of each box is the 3rd quartile, and the whiskers represent values within 1.5 times the interquartile range..... 33

Figure 10: Wetland Sensitivity values for upper and lower-bound simulations for all coastal wetland sites (n = 20), ordered by maximum Wetland Sensitivity. The circles indicate the upper-bound simulation and the squares indicate the lower-bound simulation..... 35

Figure 11: Distribution of Wetland Sensitivity scores in the upper and lower-bound simulations. 36

Figure 12: Map showing Wetland Sensitivity scores for the 20 Great Lakes coastal wetland sites in the lower-bound simulation. Site names are in Figure 3. 37

Figure 13: Map showing Wetland Sensitivity scores for the 20 Great Lakes coastal wetland sites in the upper-bound simulation. Site names are in Figure 3..... 38

Figure 14: Partial plot of the relationship between standard deviation of site elevation (in metres) and Wetland Sensitivity for all 20 study sites in both simulations ($\beta = -0.16$, $SE = 0.07$, $t = -2.42$, $p = 0.020$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model. 41

Figure 15: Partial plot of the relationship between Z-scored three-year-mean water level and the probability of an extreme year of total wetland area for Lake Erie and Huron-Erie Corridor study sites ($\beta = 1.13$, $SE = 0.56$, $Z = 2.02$, $P = 0.044$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.. 43

Figure 16: Partial plot of the relationship between Z-scored three-year-mean water level and the probability of an extreme year of total wetland area for Lake Ontario and St. Lawrence River study sites in the lower-bound simulation ($\beta = 11.82$, $SE = 6.72$, $Z = 1.76$, $P = 0.078$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model..... 46

Figure 17: Partial plot of the relationship between Z-scored three-year-mean water level and the probability of an extreme year of total wetland area for Lake Huron study sites in the upper-bound simulation ($\beta = 49.55$, $SE = 23.51$, $Z = 2.11$, $P = 0.035$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model..... 48

List of Tables

Table 1: Summary of the ecological attributes, including a brief rationale for their inclusion and their relationship with hydrodynamic variables that are expected to be affected by climate change (primarily water levels).....	6
Table 2: Representative Concentration Pathway (RCP) and driving Global Climate Change (GCM) models used. RCP 4.5 represents a moderate emission scenario and the two GCMs represented different possible futures to account for uncertainty in climate projections (higher and lower water-level scenarios, both with increased variability in Great Lakes water levels compared to the projected past).....	19
Table 3: Community classes used in the Coastal Wetland Response Model (CWRM) to assess climate change on Great Lakes coastal wetlands. Hydrological conditions and vegetation characteristics with species examples are provided for each class. Note that open water and upland classes were not considered wetland classes, and were therefore not included in the Sensitivity assessment. Table adapted from Grabas and Rokitnicki-Wojcik 2015; Keddy and Fraser 2000; Lee et al. 1998.....	20
Table 4: Scoring system for Wetland Sensitivity for each site and simulation. Wetland Sensitivity is a proportion (0 to 1). Score is a colour classification associated with the Wetland Sensitivity (green, yellow, and red). The interpretation is based on the risk that a wetland may experience negative effects of climate change in the projected future.....	31
Table 5: Pearson correlation coefficients between Attribute Sensitivity values for all combinations of the five ecological attributes. None of the ecological attributes were considered to be correlated ($r < 0.6$).....	34
Table 6: Summary of results for each site showing wetland name, wetland hydrogeomorphic (HGM) type, Wetland Sensitivity (with colour-coded score), and most and least sensitive ecological attributes. WI = Wetland Interspersion, MMA = Meadow Marsh Area, TWA = Total Wetland Area, SAVV = Submerged Aquatic Vegetation Volume, WD = Wetland Diversity.....	39
Table 7: Results from the generalized linear mixed model for Lake Erie and Huron-Erie Corridor study sites. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value.....	42
Table 8: Results from the generalized linear mixed model for Lake Ontario and St. Lawrence River study sites. Results from the likelihood ratio test used to test the significance of the	

random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value..... 44

Table 9: Results from the generalized linear mixed model for Lake Huron study sites. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value. 44

Table 10: Results from the generalized linear mixed model for Lake Ontario and St. Lawrence River study sites for the lower-bound simulation. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value..... 45

Table 11: Results from the generalized linear mixed model for Lake Ontario and St. Lawrence River study sites for the upper-bound simulation. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value. 47

Table 12: Results from the generalized linear mixed model for Lake Huron study sites for the upper-bound simulation. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value..... 47

Introduction

Great Lakes coastal wetlands and climate change

Coastal wetlands are dynamic and complex environments that are critical components of the Laurentian Great Lakes. These systems have many important functions, including providing breeding or migratory habitat for wildlife (including birds, mammals, reptiles, and amphibians; Markle et al. 2018; Maynard and Wilcox 1997; Mortsch 1998; Sierszen et al. 2012) and spawning and nursery habitat for fish (Jude and Pappas 1992). Coastal wetlands also provide many ecosystem services such as nutrient and sediment retention (resulting in improved water quality; Sierszen et al. 2012) and protection against shoreline erosion (Johnson and Johnston 1995; Keough et al. 1999).

Human-related activities have resulted in enormous losses of wetlands worldwide (Kingsford et al. 2016). In the Laurentian Great Lakes, specifically, over 50% of coastal wetlands have been lost since European settlement, principally due to urban and agricultural development (Brazner et al. 2000). Furthermore, many remaining coastal wetlands are impacted by various anthropogenic stressors such as nutrient and sediment loading (potentially leading to eutrophication and excessive algae growth), fragmentation, invasive species, shoreline alteration, and water-level regulation (Burton 1985; Krieger et al. 1992; SOLEC 2007; Wilcox et al. 2008). These stressors reduce available habitat for fish and wildlife, interfere with biophysical processes, and ultimately threaten many of the services provided by Great Lakes coastal wetlands.

Future climate change represents an additional threat to Great Lakes coastal wetlands. By the end of the century, significant changes in climate are expected across the Great Lakes, including increased over-land air temperatures (resulting in warmer winters, less snowfall, earlier warming in the spring, more extreme heat in the summer, a longer growing season, heavier precipitation, and less ice cover), increased over-lake precipitation, increased lake water levels, and increased variability in lake water levels (Lam and Dokoska, in prep.). These changes will have numerous environmental, economic, and social consequences in the Great Lakes system. For example, changes in water levels will directly influence the structure and function of coastal wetlands (Keddy and Reznicek 1986; Smith et al. 2021), which may have significant implications for wildlife populations and habitat. However, the vulnerability of Great Lakes coastal wetlands to climate change has not yet been quantified.

Program purpose

The 2012 Canada-United States Great Lakes Water Quality Agreement (GLWQA) and the 2014 Canada-Ontario Agreement on Great Lakes Water Quality and Ecosystem Health (COA) included Annexes that recognized the effects of climate change in the Great Lakes basin. Through the Great Lakes Protection Initiative (GLPI), Environment and Climate Change Canada (ECCC) sought to provide insight into climate change impacts by assessing the vulnerability of Great Lakes coastal wetlands. This work supports Canada's commitments under the GLWQA and COA to support healthy and productive wetlands and other habitats to sustain resilient populations of native species. The objectives of this program were to assess climate change impacts on Great Lakes coastal wetlands, identify wetlands that are most vulnerable, develop adaptive measures to build wetland resilience, and communicate and share results with relevant stakeholders and practitioners.

Climate Change Vulnerability Assessments

The Intergovernmental Panel on Climate Change (IPCC) defines **Vulnerability** as “the propensity for a system or species to be adversely affected by climate change” (IPCC 2007). Climate Change Vulnerability Assessments (CCVAs) are frameworks used to evaluate the Vulnerability of a species or system to climate change (Culp et al. 2017; Ellison 2014; Upgupta et al. 2015; Wabnitz et al. 2018), and include three components: **Exposure**, **Sensitivity**, and **Adaptive Capacity** (IPCC 2007). Exposure is a measure of how much change in climate a species or system may experience (Glick et al. 2011), Sensitivity represents the degree to which a species or system will respond to climate change, and Adaptive Capacity describes the species or system's ability to accommodate or cope with the potential impacts of Exposure and Sensitivity. ECCC used a modified version of the Glick et al. (2011) framework, where the Exposure of a coastal wetland to changes in hydro-climatic variables (e.g., water levels) and the degree to which the wetland responded (i.e., Sensitivity) determined the potential impact of climate change. The impact may be moderated by a wetland's ability to adapt or cope with these changes (i.e., Adaptive Capacity).

This assessment was a collaborative effort between multiple ECCC branches: the Meteorological Service of Canada (National Hydrological Services and Hydrodynamic and Ecohydraulic Section), Science and Technology (Wildlife and Landscape Science Directorate), Strategic Policy (Regional Director General's Office - Ontario Region), and the Canadian Wildlife

Service – Ontario Region (Habitat Assessment and Data Management Unit). This report details the work led by the Canadian Wildlife Service to assess the Sensitivity of Great Lakes coastal wetlands to climate change.

Wetland Sensitivity

The goal of this Sensitivity assessment was to estimate the negative ecological effects of climate change on Great Lakes coastal wetlands and then communicate these science-based results in an easily understandable way to stakeholders, land managers, and the public. We define Sensitivity in this CCVA as the degree to which a wetland adversely responds to climate change. Note that this definition of Sensitivity differs slightly from Glick et al. (2011) because we are only estimating adverse (or negative) effects of climate change on coastal wetlands. We focused on adverse effects because the ultimate goal of this CCVA was to determine which coastal wetlands in the Great Lakes basin were most vulnerable to climate change. Therefore, to estimate Vulnerability in the context of adverse effects, Sensitivity must focus on the potential negative impacts of climate change (e.g., wetland loss).

This Sensitivity assessment used predicted wetland vegetation community maps for 20 wetland sites under different climate change scenarios. Given that vegetation communities are the basis for wildlife habitat and many ecosystem functions (Tews et al. 2004), such data were considered highly suitable to analyze ecological responses to climate change. These vegetation community maps were generated through a Coastal Wetland Response Model (CWRM) developed by the Hydrodynamic and Ecohydraulic Section of ECCC's National Hydrological Services (discussed in detail in ECCC 2022a).

Ecological attribute selection

For this Sensitivity assessment, **ecological attributes** were defined as measures of coastal wetland ecosystem composition. Our aim was to identify a simple set of attributes that would represent key ecological responses of Great Lakes coastal wetlands to climate change. Vegetation community extent would then be simulated over time based on different climate change scenarios, which we would use to calculate these ecological attributes. Ecological attributes would then be aggregated to assess the Sensitivity of different wetland sites to climate change (discussed in detail below). We considered more than 100 attributes that could be measured from grid-based maps of wetlands using spatial analysis tools (e.g.,

Hesselbarth et al. 2019; McGarigal et al. 2012). To guide the selection of attributes, we developed the following criteria.

Attributes must be:

1. Important in wetland ecology and have a strong ecological basis

The ecological attribute is based on well-established ecological theory and principles and/or has strong empirical evidence of a relationship with a wildlife species, population, habitat, or ecosystem function in wetlands (e.g., species-area relationship).

2. Responsive to changes in water levels

The ecological attribute has a strong and predictable relationship with hydrodynamic variables that are expected to be affected by climate change and that are included in the Coastal Wetland Response Model (primarily water levels).

3. Relatively independent of other attributes (have low collinearity)

The ecological attribute represents an independent or distinct ecological response of a coastal wetland to climate change. Each attribute should inform a relatively distinct part of wetland ecology; therefore, the responses among ecological attributes should not be highly correlated. Multiple, highly correlated attributes would artificially bias estimates of Wetland Sensitivity; for example, by “double-counting” the same ecological response or process.

4. Reliably extracted from CWRM grid outputs

The ecological attribute can be calculated from the CWRM grids with high accuracy at the community or ecosystem scale. Ecological attributes that required a finer resolution (less than 10 metres [which was the CWRM grid interval]) were excluded (e.g., microhabitat features).

5. Easily communicated

The scientific merit of an attribute must be easy to understand and meaningful to stakeholders, right holders, and the general public. Since the ultimate goal of the CCVA

was to inform Canadians on actions that increase coastal wetland resilience, selection of attributes requires engagement with stakeholders.

A workshop was held in March 2019 to discuss these selection criteria and four candidate ecological attributes with stakeholders, land managers, and wetland experts from across the Great Lakes Basin (LimnoTech 2019). The Canadian Wildlife Service presented the rationale for four candidate attributes: 1) total wetland area, 2) volume of submerged and floating aquatic vegetation, 3) wetland interspersion, and 4) wetland vegetation community diversity. Stakeholders and experts supported the selection criteria and ecological attributes, and identified two additional ecological attributes: meadow marsh area and non-persistent emergent vegetation area (i.e., emergent vegetation excluding cattail [*Typha spp.*] and common reed [*Phragmites australis*]). We included meadow marsh area because it met all selection criteria, but excluded non-persistent emergent vegetation area because it could not be modelled independently from emergent marsh due to lack of data on this community type (i.e., did not meet selection criteria 4). This resulted in the selection of five ecological attributes for the Sensitivity assessment (Table 1).

Table 1: Summary of the ecological attributes, including a brief rationale for their inclusion and their relationship with hydrodynamic variables that are expected to be affected by climate change (primarily water levels).

Attribute	Rationale	Relationship with Hydrodynamic Variables
Total Wetland Area	Strong evidence of a positive relationship between habitat area and species abundance, diversity, ecosystem functions, and services (e.g., Elliott et al. 2020; Smith and Chow-Fraser 2010)	Water-level fluctuations are a well-documented driver of wetland extent (reviewed in Keddy and Fraser 2000; Keddy and Reznicek 1986; Keough et al. 1999)
Submerged and Floating Aquatic Vegetation (SAV) Volume	Greater SAV volume is generally associated with increased SAV vegetation complexity and habitat structure underwater (Weller and Chow-Fraser 2019); provides critical spawning, nursery and/or foraging habitat for fish, amphibians, and reptiles (e.g., Markle et al. 2018)	Sensitive to changes in water levels (Weller and Chow-Fraser 2019)
Wetland Interspersion (Edge Density)	Interspersion is the degree of mixing between vegetation and open water (Rehm and Baldassarre 2007); measures wetland structural heterogeneity above water; proxy for hemi-marsh; represents important habitat structure for wetland wildlife (e.g., marsh birds; Hohman et al. 2021; Rehm and Baldassarre 2007); increasing interspersion is a key management action in wetland restoration (e.g., Schummer et al. 2012)	Water-level fluctuations create interspersion; high water levels cause die-back and openings in emergent vegetation, creating pockets of open water (Keddy and Reznicek 1986; Wilcox et al. 2005); low water levels allow for regrowth of emergent vegetation

Attribute	Rationale	Relationship with Hydrodynamic Variables
Wetland Diversity (Shannon-Wiener Diversity Index)	Vegetation community diversity generally provides habitat diversity (Habitat Heterogeneity Hypothesis; Stein et al. 2014); habitat diversity is generally positively related to species richness and ecosystem functioning; limiting dominance by one vegetation community (e.g., monotypic cattail stands) is a key management action in wetland restoration	Water-level fluctuations are a well-documented driver of wetland vegetation community diversity (Keddy and Fraser 2000; Keddy and Reznicek 1986; Wilcox and Nichols 2008)
Meadow Marsh Area	Highly responsive to water-level fluctuations (Wilcox et al. 2005); well-established performance indicator for Lake Ontario adaptive management used to assess ecological impacts of outflow regulation on coastal wetlands; highest species richness among wetland vegetation communities (Keddy and Reznicek 1986)	Water-level fluctuations are required for the establishment of meadow marsh; stabilization of lake levels reduces meadow marsh extent (e.g., Wilcox et al. 2005)

Rationales for selected ecological attributes

Total wetland area

The species-area relationship is a well-established concept in ecology (Rosenzweig 1999), where species richness and abundance generally increases with habitat area. Many studies support the species-area relationship (e.g., Askin and Philbrick 1987; Elliott et al. 2020), demonstrating that diverse and abundant populations in large areas can more easily tolerate a range of environmental conditions (reviewed in Elliott et al. 2020). Further, direct reductions in habitat area can negatively affect community structure by altering the strength of species interactions and trophic chain length in food webs, and can negatively affect population structure by influencing distribution, dispersal, reproductive output, foraging success, and genetic diversity (Fahrig 2003). Studies have shown many wetland birds (e.g., Swamp sparrow [*Melospiza georgiana*]) to be area-dependent, meaning that they will not breed unless there is a certain amount of good quality habitat (Elliott et al. 2020; Riffell et al. 2001; Smith and Chow-Fraser 2010). Similarly, wetland area has been shown to be positively associated with fish species richness, and is important in structuring fish community composition (Montgomery et al. 2020).

In Great Lakes coastal wetlands, total wetland area is directly affected by fluctuating water levels (Keddy and Reznicek 1986). These fluctuations are a natural process that drive the structure and diversity of wetland vegetation since many wetland plant species are adapted to a limited range of water depths (Wilcox and Nichols 2008; Figure 1). In general, as water levels increase, wetland vegetation communities can occupy higher elevations (Keddy and Reznicek 1986), whereas low water levels tend to cause wetland vegetation communities to shift lakeward, leaving higher elevations to be dominated by upland plants (Hartmann 1990; Mortsch 1998). Sustained water levels also alter vegetation communities, often causing an increase in the occurrence of homogenous communities (Keddy and Reznicek 1986). This has been described in Lake Huron, where sustained low water levels led to a reduction in fish habitat (e.g., Fracz and Chow-Fraser 2013; Midwood and Chow-Fraser 2012), and in Lake Ontario, where water-level regulation has resulted in the proliferation of cattails, primarily at the expense of meadow marsh (Wilcox and Bateman 2018; Wilcox et al. 2008). However, it is important to note that the influence of water levels on wetland area is complicated by many factors, including the frequency of water-level fluctuations at different temporal scales (e.g., Grabas and

Rokitnicki-Wojcik 2015; Grabas et al. 2019; Smith et al. 2021), local bathymetry/topography, substrate, and adjacent land cover (Keough et al. 1999; Steinman et al. 2012).

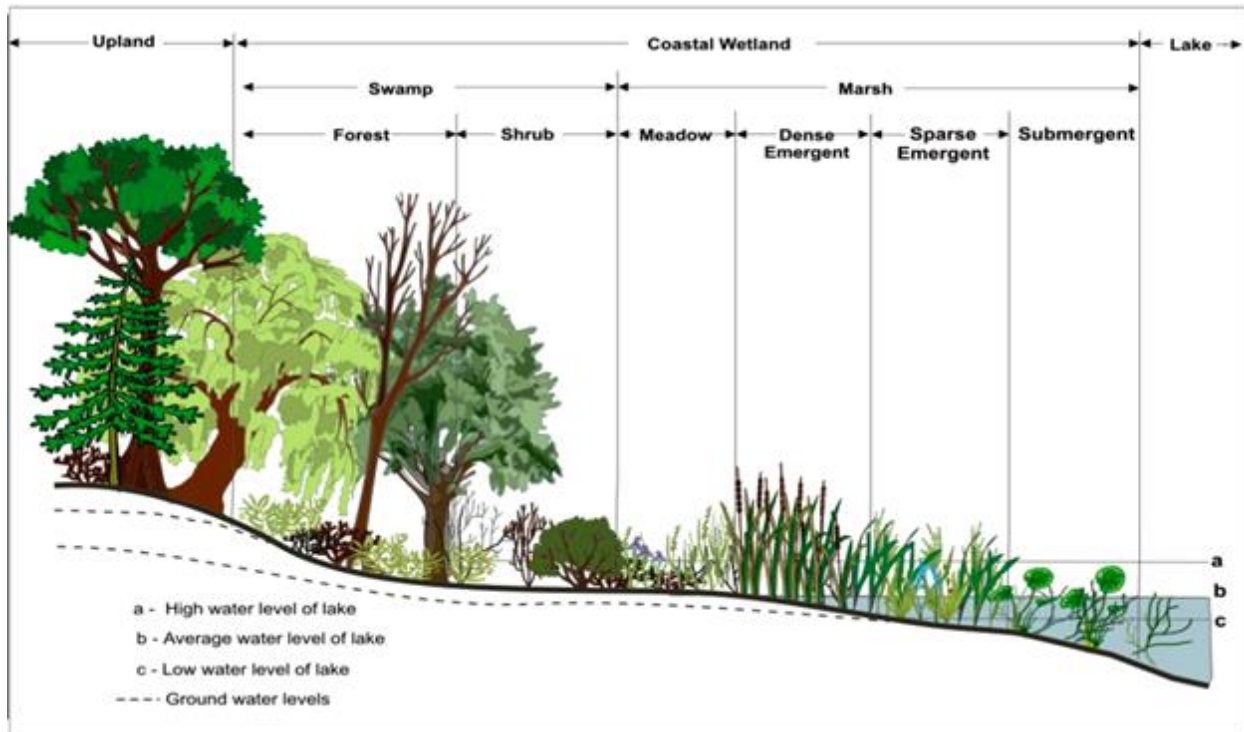


Figure 1: Vegetation zonation in a typical coastal wetland. This arrangement of wetland plants is largely determined by moisture conditions.

Long Description: This figure shows the pattern of vegetation zonation in wetland. From lake to upland, vegetation community zones occur as follows: submerged and floating aquatic vegetation, emergent, meadow marsh, shrub swamp, and forest swamp. This arrangement of wetland plants is largely determined by the ability of species within each community to tolerate different moisture conditions.

Wetland area has been dramatically reduced in Great Lakes coastal wetlands from human-related activities (e.g., wetland dredging and filling; Mortsch et al. 2006). Conversion of wetlands to urban or agricultural land-use results in an immediate loss of wildlife habitat (Montgomery et al. 2020) and significantly alters biophysical processes (e.g., nutrient and sediment cycling; Zedler and Kercher 2005). Excessive nutrient input associated with nearby urban and agricultural land use can lead to a cascading effect of altered ecosystem processes contributing to lower water quality and loss of wetland habitat (Chow-Fraser 1998; Zedler 2003).

Submerged and floating aquatic vegetation volume

Submerged and floating aquatic vegetation (hereafter referred to as SAV) is an important component of Great Lakes coastal wetlands. From a wetland function standpoint, it stores and releases nutrients (Engel 1988; Stuckey 1989), adds oxygen to the water through photosynthesis, and provides stability to sediment, reducing turbidity (McNair and Chow-Fraser 2003; Ohrel and Register 2006). SAV also affords habitat for numerous wetland fauna. For example, it provides spawning structure for adult fish and refugia for larval and young fish (Casselman and Lewis 1996; Jacobus and Ivan 2005; Jude and Pappas 1992; Trebitz et al. 2009). Lake species (e.g., yellow perch [*Perca flavescens*] and walleye [*Sander vitreus*]) migrate in spring to use SAV for spawning, seeking out the warmer temperatures and protection from the higher wave energy of the open lake (Brazner et al. 1998; Stephenson 1990; Wei et al. 2004). SAV also provides marsh-dependent birds with habitat for activities related to breeding and feeding (Bannor and Kiviat 2002; Brisbin and Mowbray 2002; Mowbray 1997; Rohwer et al. 2002), and wetlands with healthy SAV communities support more robust marsh bird communities (Grabas et al. 2012; Rokitnicki-Wojcik et al. 2014). Additionally, certain turtle species (e.g., eastern musk turtle [*Sternotherus odoratus*] and Blanding's turtle [*Emydoidea blandingii*]) prefer the warmer microclimate and foraging opportunities offered by SAV (Hartwig and Kiviat 2007; Markle et al. 2018; Millar and Blouin-Demers 2011).

Determinates of SAV community structure and establishment in wetlands are water quality, substrate, and the interaction between bathymetry and water levels (Duarte and Kalff 1986; Grabas et al. 2012; Hudon et al. 2006). When water levels decrease, stranded SAV generally dies-off and is replaced with plant species with lower water-depth tolerances (e.g., shallow emergent species; Midwood and Chow-Fraser 2012). When water levels increase, plants intolerant of flooding are replaced by SAV (Keddy and Fraser 2000; Keddy and Reznicek 1986; Smith et al. 2021). As such, rises and falls in Great Lakes water levels will influence the extent and composition of SAV communities. During periods when water levels remain around the long-term average, the SAV community generally occupies the same elevation extent over time, but becomes more dense (Hudon et al. 2006). LeBlanc et al. (2014) found that during sustained low water years in Georgian Bay, SAV community structure changed, becoming less diverse and abundant. In relation to water quality, wetlands in the lower Great Lakes with high conductivity and turbidity tend to have decreased SAV coverage (e.g., Grabas and Rokitnicki-Wojcik 2015). This sensitivity to physical and chemical conditions suggests the SAV community is a proxy for overall wetland health (Cvetkovic et al. 2010; Keough et al. 1999).

Water levels affect the characteristics of SAV communities beyond species composition, density, and areal extent, as described above. Water levels also affect the overall volume of SAV, and there is evidence that a three-dimensional measure of SAV (volume) may be a more informative parameter of habitat quality compared to area (Weller and Chow-Fraser 2019). In some cases, increases or decreases in water levels may result in more or less habitat area (Bouvier et al. 2009; Elliott et al. 2020). However, Midwood and Chow-Fraser (2012) and Weller and Chow-Fraser (2019) found that fish habitat area was not lost under sustained low water levels in eastern Georgian Bay, but both studies concluded that there was a loss of habitat volume. These results suggest that SAV volume is more sensitive to changes in water levels than area.

The vertical structure of SAV is an important component of fish habitat. Eadie and Keast (1984) found that more vertical layers of aquatic vegetation provided more fish habitat, and that fish species diversity was highly correlated with water depth. Trebitz et al. (2009) also found that deeper wetlands supported a more structurally complex SAV community and more vegetation cover, allowing more fish to thrive. Although shallower coastal wetlands provide opportunities for emergent and floating vegetation to grow, these plants provide less structural complexity to function as fish habitat (Eadie and Keast 1984; Weller and Chow-Fraser 2019).

Wetland interspersion

Wetland Interspersion is the degree of mixing between the two major cover types in a wetland: emergent vegetation and open water/SAV (Figure 2; Rehm and Baldassarre 2007). This spatial juxtaposition between emergent vegetation and water is quantified as the amount of interface (or edge) between these two cover types (i.e., edge density), where a wetland with more vegetation-water interfaces has more interspersion (Rehm and Baldassarre 2007). Maximum interspersion occurs when there is a 50:50 ratio of emergent vegetation to open water/SAV, referred to as “hemi-marsh” (Murkin et al. 1997).

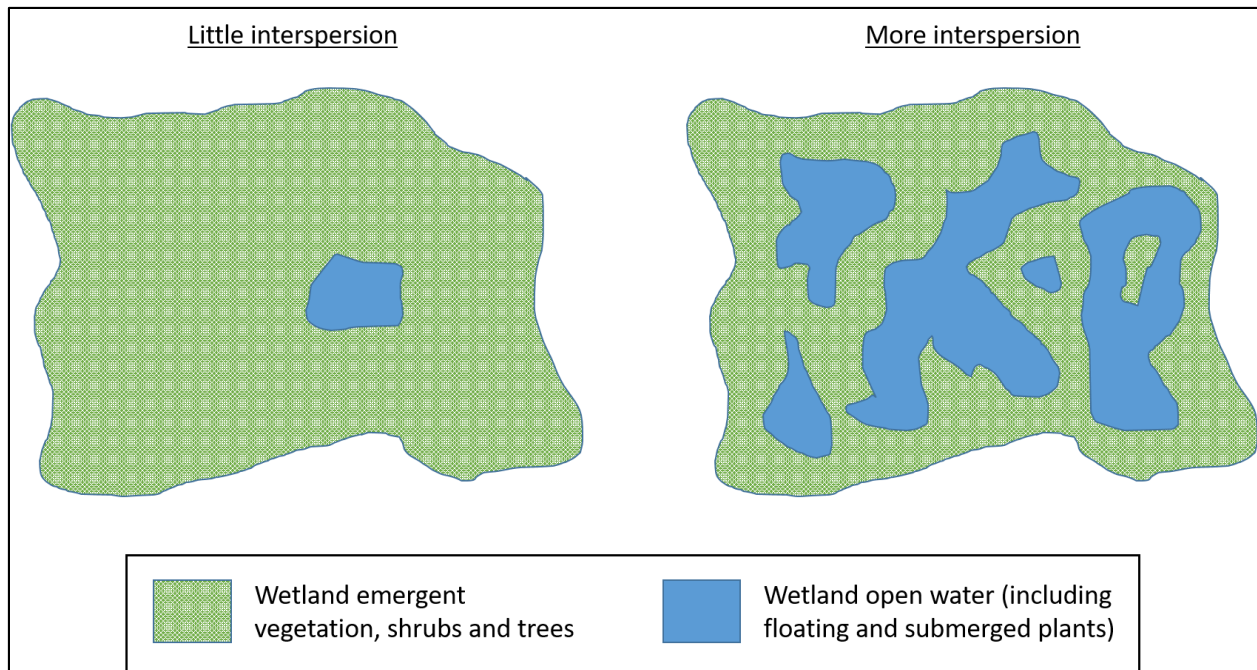


Figure 2: Diagram of a wetland with low and high interspersions (as viewed from above). Adapted from SOWES v3.2 (Ontario Ministry of Natural Resources 2013).

Long description: This figure shows different degrees of interspersions in two wetlands. The left diagram, showing little interspersions, is a wetland comprised primarily of emergent vegetation, shrubs, and trees with one small pocket of open water (including floating and submerged aquatic vegetation). The right diagram, showing more interspersions, is a wetland with multiple pools of open water (in different shapes and sizes) amongst emergent vegetation, shrubs, and trees.

Interspersions are highly influenced by hydrologic conditions, primarily water levels (Hohman et al. 2021). Periods of high lake levels cause dieback in emergent vegetation, creating openings (or “pockets”) of open water/SAV (Wilcox 2004), whereas periods of lower lake levels allow patches of emergent vegetation to grow (Wilcox 2004). It is this variation in water levels that ultimately allows for interspersed conditions. Stabilized water levels can increase the density of invasive vegetation such as *Phragmites australis* and *Typha x glauca* (Tulbure et al. 2007; Wilcox et al. 2008), which reduces interspersions by creating large dominant stands with fewer edges (Rehm and Baldassarre 2007).

Interspersions have a significant influence on wildlife. Relative abundances of multiple marsh-obligate bird species, including American bittern (*Botaurus lentiginosus*), least bittern

(*Ixobrychus exilis*), and sora (*Porzana carolina*) have been shown to increase with greater interspersed (Rehm and Baldassarre 2007). Moreover, Darrah and Kremetz (2009) found that wetlands with greater interspersed had higher occupancy of nesting king rails (*Rallus elegans*). This relationship is likely because there is more preferred feeding and nesting sites along emergent vegetation-water edges in wetlands and greater visual isolation from conspecifics (reviewed in Darrah and Kremetz 2009; Hohman et al. 2021; Rehm and Baldassarre 2007). Muskrats (*Ondatra zibethicus*) also prefer greater interspersed (Proulx and Gilbert 1983). In turn, increased herbivory by muskrat populations can create more openings and irregularly shaped channels in emergent vegetation, further increasing interspersed and suitable habitat for birds such as least bitterns (Weller and Spatcher 1965).

Interspersed in wetlands is often a focus of wetland restoration and management to increase biodiversity. Interspersed can be artificially increased through mechanical methods such as dredging and cutting. For example, in Long Point coastal wetlands on Lake Erie, the relative abundance and species richness of marsh birds were 40% and 16% higher, respectively, at dredged ponds compared to control sites (Schummer et al. 2012). Similarly, cattail marshes cut with a 50:50 ratio of vegetation to open water (i.e., maximum interspersed) had higher densities of dabbling duck pairs (such as mallards [*Anas platyrhynchos*] and blue-winged teals [*Anas discors*]) when compared to marshes with other experimentally manipulated ratios (e.g., 70:30 ratio; Murkin et al. 1982).

Wetland diversity

For this assessment, wetland diversity represents the compositional heterogeneity of vegetation communities in a wetland. We quantified wetland diversity as the variety and evenness of five wetland vegetation community types using a modified Shannon-Wiener diversity index following Fairbairn and Dinsmore (2001). The Shannon index is the most common measure of diversity used in ecological studies to assess spatial variation in habitat types (Magurran 2004).

Wetland diversity in the Great Lakes is largely driven by natural water-level fluctuations of varying frequency and amplitude (reviewed in Grabas and Rokitnicki-Wojcik 2015; Keddy and Fraser 2000; Wilcox and Nichols 2008; Wilcox et al. 2005). In general, high water levels flood out trees, shrubs, and encroaching upland vegetation at higher elevations. High water levels also cause dieback and openings in the lower extent of dominant emergent vegetation, such as

cattail species (Wilcox et al. 2005). Consequently, the area of treed and shrub thicket swamp communities decreases and the lower extent of emergent vegetation communities is temporarily replaced by more flood-tolerant communities of floating-leaved and submerged plants (e.g., Smith et al. 2021). Following these highs, lower water levels expose bare sediments, allowing germination of propagules and buried seeds. This natural drawdown promotes regeneration of numerous, less competitive emergent and meadow species that replenish the seed bank (Wilcox 2004). As a result, the extent of emergent vegetation and meadow marsh increase as the floating-leaved and submerged communities decrease due to intolerance to dewatering. Over time, dominant emergent vegetation increases in extent relative to less competitive emergent species. This natural flooding-dewatering cycle in lake levels over several time scales drives the dynamic nature of coastal wetland diversity (Keddy and Fraser 2000). In contrast, coastal wetlands on regulated lakes with dampened water-level fluctuations have lower vegetation community and structural diversity than those on unregulated lakes (Wilcox and Meeker 1992; Wilcox et al. 1993). In Lake Ontario, this decreased wetland diversity is primarily due to the loss of meadow marsh communities and expansion of cattail stands (Wilcox et al. 2008). Unsurprisingly, emulating natural water-level fluctuations through the implementation of periodic drawdowns in wetland impoundments is an important management technique to support diverse ecosystems.

For several reasons, it is important to measure and track changes in wetland diversity resulting from climate change. Firstly, the compositional heterogeneity of vegetation communities in ecosystems determines the physical structure of habitat, which greatly influences the distribution, abundance, and interactions of wildlife (reviewed in Tews et al. 2004). Ecosystems with more habitat types generally have more species (Habitat Heterogeneity Hypothesis; Cramer and Willig 2005), with plant diversity having the strongest effect on species richness across several taxa in terrestrial ecosystems (reviewed in Stein et al. 2014). Positive effects of habitat heterogeneity on biodiversity (Elliott et al. 2020; Guadagnin and Maltchik 2007; Lorenzón et al. 2016) and species abundance (Elliott et al. 2020; González-Gajardo et al. 2009) are also empirically documented in wetland ecosystems. Moreover, cattail-dominated coastal wetlands on regulated lakes have reduced marsh bird richness and abundance (Chin et al. 2014; Meyer et al. 2010), suggesting that more homogenous coastal wetlands may provide less habitat value. Ecological niche theory provides the mechanism underlying the Habitat Heterogeneity Hypothesis; increased habitat heterogeneity increases available niche space, which allows for the co-existence of an increased number of species with different ecological

requirements (i.e., niche partitioning; Ben-Hur and Kadmon 2020). It should be noted that heterogeneity-richness relationships are not always linear; unimodal relationships, where richness peaks at moderate levels of heterogeneity, have been documented in natural systems (Allouche et al. 2012; Chocron et al. 2015), including wetlands (Elliott et al. 2020). Such unimodal relationships suggest a possible trade-off between heterogeneity and the area of habitat required to support viable populations (Allouche et al. 2012).

Secondly, habitat diversity directly and indirectly drives the total number of processes or functions that an ecosystem provides (i.e., ecosystem multifunctionality), including primary productivity, carbon sequestration, and nutrient cycling (Alsterberg et al. 2017). Direct effects of habitat diversity on ecosystem multifunctionality were observed in experimental coastal marine ecosystems where more habitat types increased ecosystem functioning (Alsterberg et al. 2017). Alsterberg et al. (2017) proposed that a variety of habitats within an ecosystem promotes structural complementarity and the exchange of material and energy. Given that loss of one benthic habitat type significantly changes the metabolic status of marine coastal ecosystems (Alsterberg et al. 2017), similar complementarity is plausible among Great Lakes coastal wetland vegetation communities. Indirect effects of habitat diversity on multifunctionality occur through increased species richness. There is unequivocal evidence that reduced species richness decreases the efficiency of ecosystem functions (reviewed in Cardinale et al. 2012; Duffy et al. 2017) and that an increased number of species allows for the maintenance of multiple ecosystem functions (Zavaleta et al. 2010).

Finally, habitat diversity may increase the stability of ecosystems via increased biodiversity. The presence of more species stabilizes ecosystem functioning over time (Craven et al. 2018) and increases the resistance of ecosystem function to disturbances (Duffy et al. 2016; Isbell et al. 2015). This is because different species often respond differently to environmental change. In general, ecosystems with higher diversity (in terms of habitats and wildlife) should be less susceptible to the negative impacts of environment fluctuations, including climate change (ECCC 2022b).

Meadow marsh area

The area of meadow marsh (or wet meadow) has a history of being used as a biological indicator in Great Lakes coastal wetlands (Wilcox et al. 2005), in part because of its sensitivity to water-level fluctuations. Meadow marsh (which tends to be dominated by grasses and

sedges) is generally unable to tolerate persistent flooding, but the establishment of meadow marsh requires both occasional flooding and low water events (Wilcox et al. 2005). Periodic flooding staves off downslope growth by woody plant species from higher elevations, while low water events limit the expansion of emergent plant species to higher elevations (into the meadow) (Wilcox et al. 2005), and allows meadow marsh species to regenerate from buried seeds via exposure of the substrate (Keddy and Reznicek 1986). Because meadow marsh is typically situated between emergent vegetation and shrub-swamp, water-level fluctuations that influence meadow marsh will concurrently influence surrounding vegetation communities, which makes meadow marsh extent a useful indicator of overall wetland vegetation community dynamics.

The reduction of water-level fluctuations has been shown to decrease meadow marsh area substantially (Smith et al. 2021; Wilcox and Bateman 2018; Wilcox et al. 2005; Wilcox et al. 2008). The regulation of water levels in Lake Ontario since the 1960s provides a case study for how lack of hydrologic fluctuations affect meadow marsh. The regulation of Lake Ontario had a dampening effect on water-level fluctuations, thus diminishing the extremes of high and low water levels that naturally occur in the system (Wilcox et al. 2005). Without the periodic high and low water events required to curtail the spread of woody plants from higher elevations and emergent species from lower elevations into meadow marsh-dominated areas, over 50% of meadow marsh was lost since the 1960s (Wilcox et al. 2005). Much of the loss appeared to be caused by the aggressive upland expansion of cattail species over this time period (Wilcox and Bateman 2018; Wilcox et al. 2008; Wilcox et al. 2005).

The value of meadow marsh as an important ecological attribute is not, however, restricted to its degree of sensitivity to fluctuating water levels. Meadow marsh also represents the most diverse vegetation community in Great Lakes coastal wetlands (Wilcox et al. 2005), with the highest species richness among vegetative guilds (Grabas et al. 2019). Meadow marsh therefore provides diverse structural habitat for many wildlife species (Wilcox et al. 2005), including foraging areas for ungulate species (Patton and Judd 1970) and breeding and foraging habitat for many bird species (Desgranges et al. 2006; Riffell et al. 2001).

Methods

Study sites

Twenty coastal wetlands in the Great Lakes basin were assessed to determine their Sensitivity to projected climate change (Figure 3): seven sites in Lake Huron, three in the Huron-Erie Corridor (specifically, Lake St. Clair and the Detroit River), five in Lake Erie, four are in Lake Ontario, and one is in the upper St. Lawrence River. These study sites were selected to represent a range in geographical regions, surrounding land tenure and uses, and two wetland hydrogeomorphic types (riverine and lacustrine; ECCC 2022a; ECCC 2022b). Lake Superior wetlands could not be fully assessed and were not included in this report.

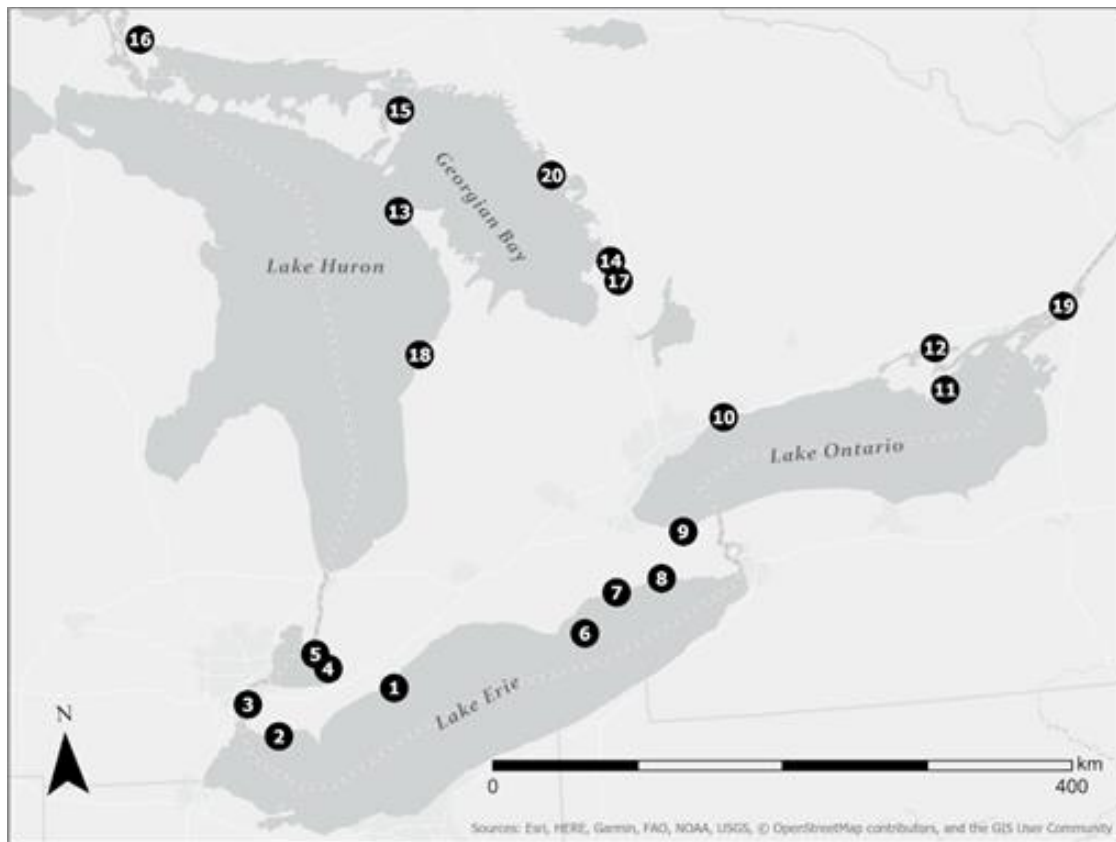


Figure 3: Locations of the twenty coastal wetland sites used in the assessment of Wetland Sensitivity. Seven sites are in Lake Huron (13. Hay Bay, 14. Treasure Bay, 15. Whiskey Harbour, 16. Anderson Creek, 17. Hog Bay, 18. Baie du Doré, 20. Frances Point), three are in the Huron-Erie Corridor (3. Detroit River, 4. Lake St. Clair, 5. Johnston Bay), five are in Lake

Erie (1. Rondeau Bay, 2. Fox/Dolson's Creek, 6. Long Point, 7. Selkirk Provincial Park, 8. Grand River Mouth) four are in Lake Ontario (9. Jordan Station, 10. Lynde Creek, 11. South Bay, 12. Airport Creek), and one is in the upper St. Lawrence River (19. Hill Island East).

Long Description: This figure is a map showing the study area and the location of the 20 wetland sites used in the assessment of Wetland Sensitivity. The study area is the Canadian side of the Great Lakes basin, excluding Lake Superior. Seven sites are in Lake Huron (Anderson Creek, Baie du Doré, Frances Point, Hay Bay, Hog Bay, Treasure Bay, Whiskey Harbour), three are in the Huron-Erie Corridor (Detroit River, Johnston Bay, Lake St. Clair), five are in Lake Erie (Fox/Dolson's Creek, Grand River Mouth, Long Point, Rondeau Bay, Selkirk Provincial Park) four are in Lake Ontario (Airport Creek, Jordan Station, Lynde Creek, South Bay), and one is in the upper St. Lawrence River (Hill Island East).

Data source

We used outputs from the Coastal Wetland Response Model (CWRM) developed by the Hydrodynamic and Ecohydraulic Section of ECCC's National Hydrological Services for each of the 20 wetland study sites. The CWRM integrated physical and ecological data to evaluate the response of wetland vegetation communities over time to climate change. Data included climatic exposure (e.g., temperature), modelled hydrodynamic data (e.g., water levels), land use, digital elevation models, and data collected from study site-level vegetation surveys (ECCC 2022a).

The CWRM used two down-scaled Global Climate Models (GCMs) under a mitigated greenhouse gas emission scenario of Representative Concentration Pathway (RCP) 4.5 (Table 2). RCPs are scenarios of emission, concentration and land-use trajectories that are used for climate modelling (Moss et al. 2008). RCP 4.5 is a moderate emission scenario in which "radiative forcing" (or the heating effect caused by atmospheric greenhouse gases) is stabilized at 4.5 watts per square metre (equivalent to approximately 650 ppm of carbon dioxide) by the end of the century due to the implementation of mitigation measures (Thomson et al. 2011). The GCMs were down-scaled to fit the Great Lakes region. The two GCM models (hereafter, simulations) represented different possible futures to account for uncertainty in climate projections; the lower-bound simulation and the upper-bound simulation. The lower-bound simulation had average water-level projections that were relatively similar to the projected recent past, while the upper-bound simulation projected an increase in water levels compared to the projected past. Importantly, both the lower and upper-bound simulations had increased

overall variability in Great Lakes water levels compared to the projected past. For each simulation, the CWRM generated 56 grid outputs (one grid per year) for each of the 20 study sites. Each cell within each grid represented 100 square metres of a vegetation community (Table 3) with the highest probability of occurrence. These grid outputs included historic projections (hindcast years; 1981 to 2008) and future projections (forecast years; 2071 to 2098).

Table 2: Representative Concentration Pathway (RCP) and driving Global Climate Change (GCM) models used. RCP 4.5 represents a moderate emission scenario and the two GCMs represented different possible futures to account for uncertainty in climate projections (higher and lower water-level scenarios, both with increased variability in Great Lakes water levels compared to the projected past).

Scenario	Driving Global Climate Change Model	Simulation ID
RCP 4.5	CanESM2 (lower water level)	Lower-bound
	GFLD-ESM2M (higher water level)	Upper-bound

Table 3: Community classes used in the Coastal Wetland Response Model (CWRM) to assess climate change on Great Lakes coastal wetlands. Hydrological conditions and vegetation characteristics with species examples are provided for each class. Note that open water and upland classes were not considered wetland classes, and were therefore not included in the Sensitivity assessment. Table adapted from Grabas and Rokitnicki-Wojcik 2015; Keddy and Fraser 2000; Lee et al. 1998.

Community Class	Hydrological Conditions	General Vegetation Description	Example Species
Open water	> 2m water depth	Little macrophyte cover; no shrub or tree cover; plankton dominated	n/a
Submerged and floating aquatic (shallow open aquatic marsh)	< 2m water depth; permanently inundated	> 25% cover of submerged, rooted, and free-floating vascular macrophytes, Characeaea, and Ceratophyllaceae; no tree or shrub cover	<i>Potamogeton pusillus</i> <i>Nymphaea odorata</i> <i>Spirodela polyriza</i>
Emergent marsh (shallow marsh)	Flooded for most of the growing season	> 25% cover of emergent plants with above-substrate growth that persists (e.g., cattail <i>spp.</i>) or does not persist (e.g., <i>Schoenoplectus spp.</i>) the following growing season; species tolerant of prolonged flooding; most species require occasional low water levels to expose substrate for seed germination	<i>Typha x glauca</i> <i>Pontederia cordata</i> <i>Schoenoplectus acutus</i>
Meadow marsh (wet meadow)	Seasonally flooded; inundated during spring, moist to dry by summer	> 25% cover of vascular plants dominated by sedges, grasses, and forbs; species less tolerant of prolonged flooding; < 25% tree and shrub cover	<i>Carex stricta</i> <i>Calamagrostis canadensis</i> <i>Stachys palustris</i>
Shrub thicket swamp	Occasionally flooded; inundated during high water levels only	> 25% shrub cover; hydrophytic woody perennials < 5 metres in height with usually more than one low-branching stem; < 25% tree cover	<i>Salix petiolaris</i> <i>Cornus sericea</i> <i>Alnus incana</i>

Community Class	Hydrological Conditions	General Vegetation Description	Example Species
Treed swamp	Occasionally flooded; inundated during high water levels only	> 25% tree cover; hydrophytic trees > 5 metres in height	<i>Fraxinus pennsylvanica</i> <i>Thuja occidentalis</i> <i>Acer saccharinum</i>
Swamp	Occasionally flooded; inundated during high water levels only	> 25% hydrophytic tree and shrub cover; used when there was insufficient data to model treed swamp and shrub swamp separately at a given wetland site	As above
Upland (terrestrial)	Water rarely above substrate	Any open, shrub, and treed communities dominated by plant species that typically do not or rarely occur in wetlands (e.g., forest, meadow, pasture, rock barren); species coefficient of wetness > 0 (Oldham et al. 1995)	<i>Acer saccharum</i> <i>Prunus virginiana</i> <i>Maianthemum canadense</i>

Processing of CWRM output

We extracted the number of grid cells in each vegetation class from 1120 CWRM grid outputs (20 sites x 56 years) to calculate the ecological attributes for each year (described below) and produce a time series of each attribute for each site and simulation (i.e., five attributes x 20 sites x two simulations = 200 time series). We only considered 100 square-metre grid cells classified as a wetland vegetation community; open water and upland classes were excluded from all analyses. These data were processed in ESRI's ArcGIS Pro and ArcPy API v2.6 (ESRI, Redlands, CA) and Python v3.6 (Van Rossum and Drake 2009). The coordinate reference system was WGS 1984 (UTM zones 17N and 18N).

Calculating ecological attributes

We calculated area for each vegetation community by summing the number of grid cells classified as the respective vegetation community and then multiplying by 100 square metres (the area of each grid cell) for each site, year, and simulation combination. For example, there were 2512 grid cells classified as meadow marsh for Lynde Creek in the 2071 projection; therefore, the total area of meadow marsh was 251200 square metres. Total wetland area was calculated by summing the total area of all wetland vegetation communities for each site, year, and simulation combination (Figure 4).

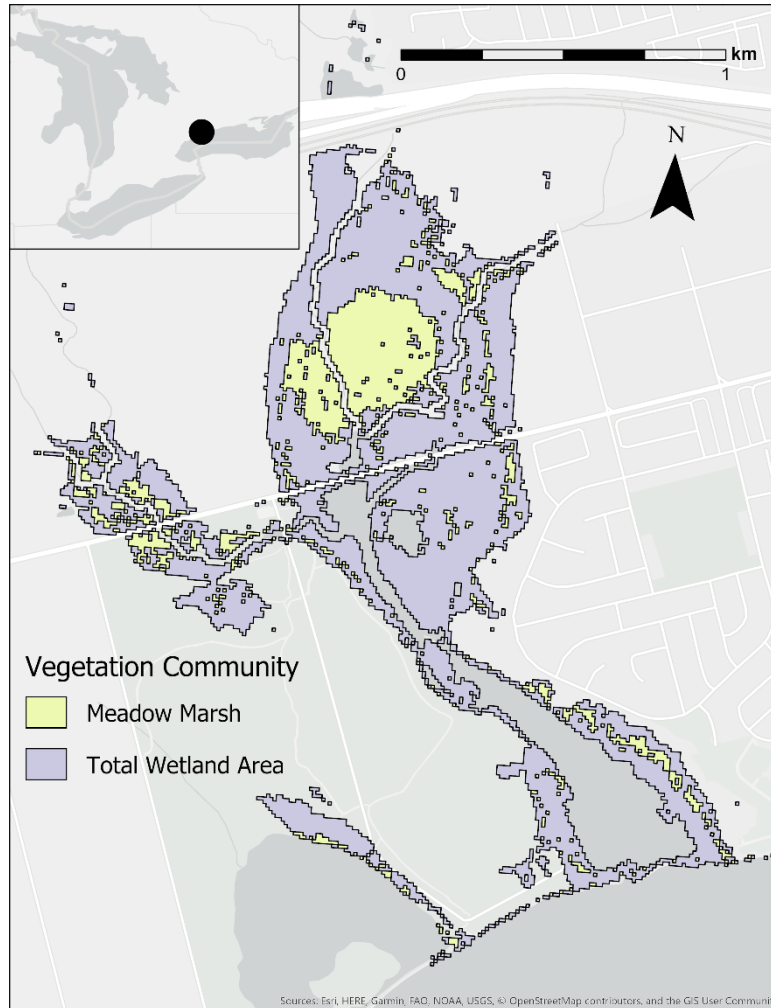


Figure 4: Map showing meadow marsh area (yellow) and total wetland area (purple, excluding meadow marsh) at Lynde Creek, Lake Ontario.

Long description: This figure is a map of the Lynde Creek study site (in Lake Ontario) showing meadow marsh area and total wetland area (excluding meadow marsh). Meadow marsh area and total wetland area were two of the five ecological attributes selected to assess Wetland Sensitivity to projected climate change.

We calculated SAV volume of each grid cell classified as SAV by multiplying the mean water depth (in metres; based on water-level projections by the Meteorological Service of Canada) by 100 square metres (the area of each grid cell; Figure 5). The resulting volume for each grid cell was then summed across a site for each year and both simulations.

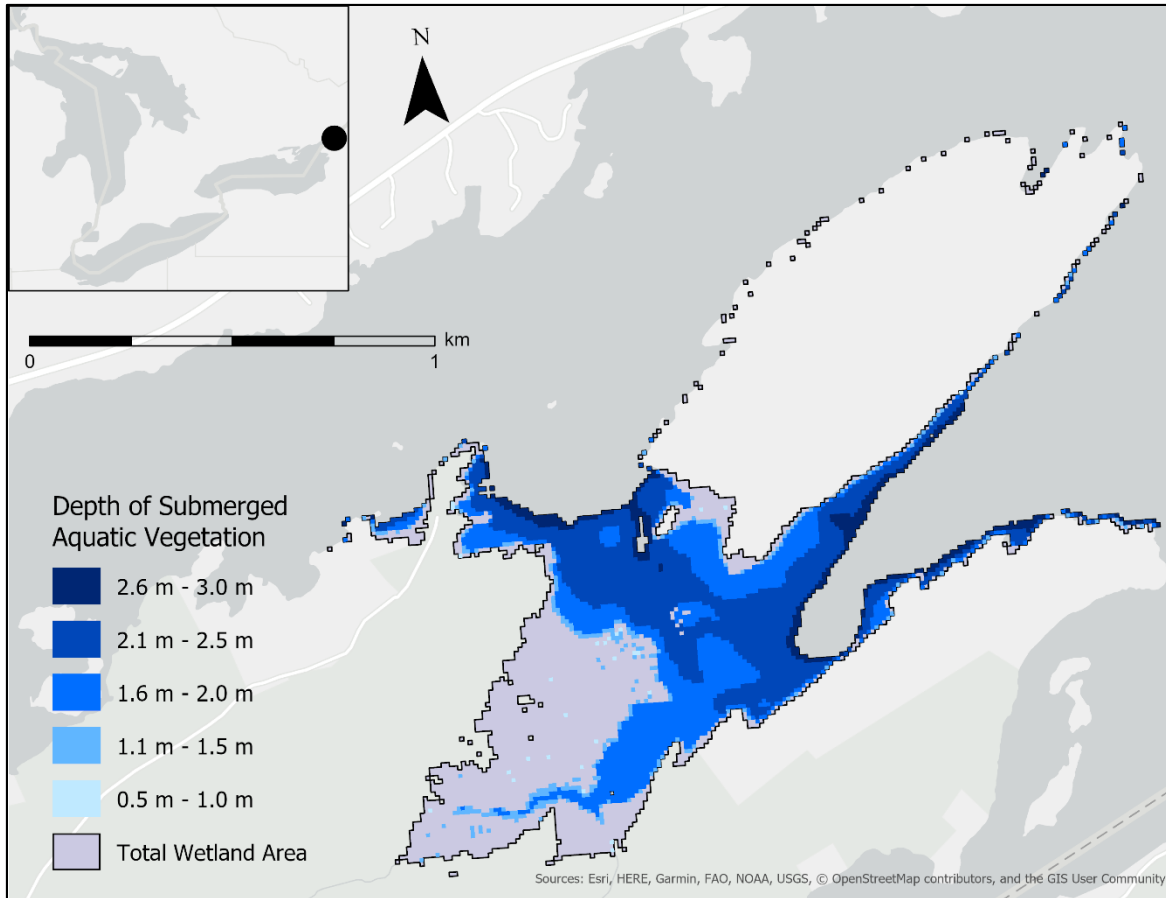


Figure 5: Map showing the depth of submerged aquatic vegetation (SAV; blue gradient), from which SAV volume was calculated, at Hill Island East, St. Lawrence River.

Long description: This figure is a map of the Hill Island East study site (in the St. Lawrence River) showing the depth of submerged aquatic vegetation (SAV). Depth was used to calculate SAV volume, which was one of the five ecological attributes selected to assess Wetland Sensitivity to projected climate change. In this example, depth varies between 0.5 metres and 3 metres. Total wetland area (excluding SAV) is also shown.

Wetland interspersion was somewhat unique among the ecological attributes selected to assess Wetland Sensitivity because it was the only measure of structural vegetation complexity; the other ecological attributes were all measures of wetland composition. To quantify interspersion, we calculated edge density, or the amount of edge between SAV and other vegetation communities for each site, year, and simulation combination. To calculate edge density, we first converted grid cells to polygons for each wetland vegetation community. We then measured the total edge length where polygons of SAV bordered other vegetation

communities using the Polygon Neighbors analysis tool in ArcGIS and divided the resulting edge length by total wetland area (Figure 6).

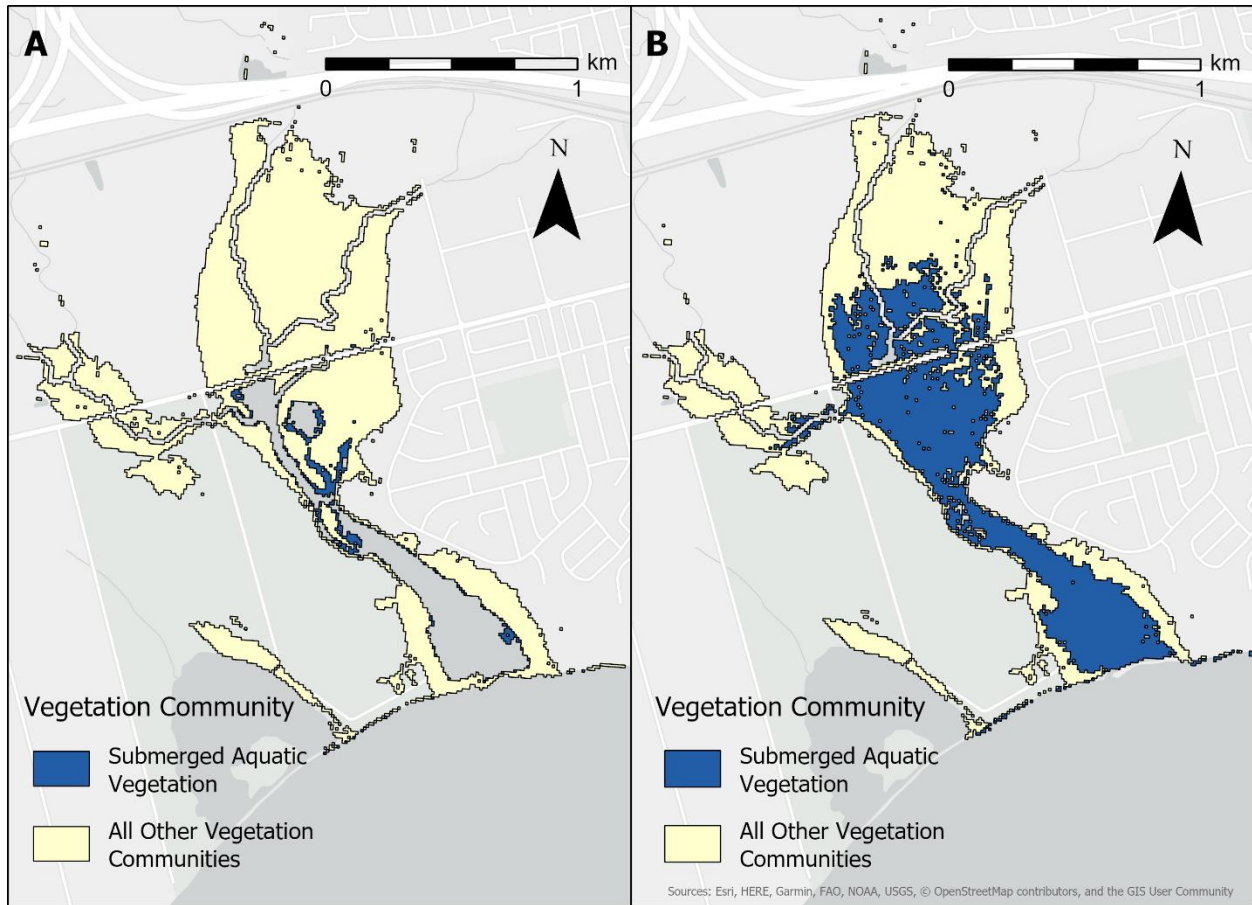


Figure 6: Two maps showing submerged aquatic vegetation (blue) and all other vegetation communities (emergent marsh, meadow marsh, and swamp; yellow) in two different years at Lynde Creek, Lake Ontario. These vegetation groupings were used to calculate interspersed. Panel A shows a year with lower interspersed and panel B shows a year with higher interspersed.

Long Description: This figure displays two maps of the Lynde Creek study site (in Lake Ontario) showing submerged aquatic vegetation (SAV) and all other vegetation communities (emergent marsh, meadow marsh, and swamp) in two different years. These vegetation groupings were used to calculate interspersed (or edge density), which was one of the five ecological attributes selected to assess Wetland Sensitivity to projected climate change. The map on the left (panel A) has less edge where SAV borders other vegetation communities relative to total wetland area, indicating lower interspersed. The map on the right (panel B) has more edge where SAV

borders other vegetation communities relative to total wetland area, indicating higher interspersions.

Finally, we used the area of each vegetation community as described above for each site, year, and simulation to calculate the Shannon-Wiener diversity index (H):

$$H = - \sum_{i=1}^S p_i \ln p_i$$

where p is the area of vegetation community i divided by the total wetland area, and S is the total number of vegetation communities (Figure 7). Note that higher index scores indicate a greater variety and more even distribution of vegetation communities.

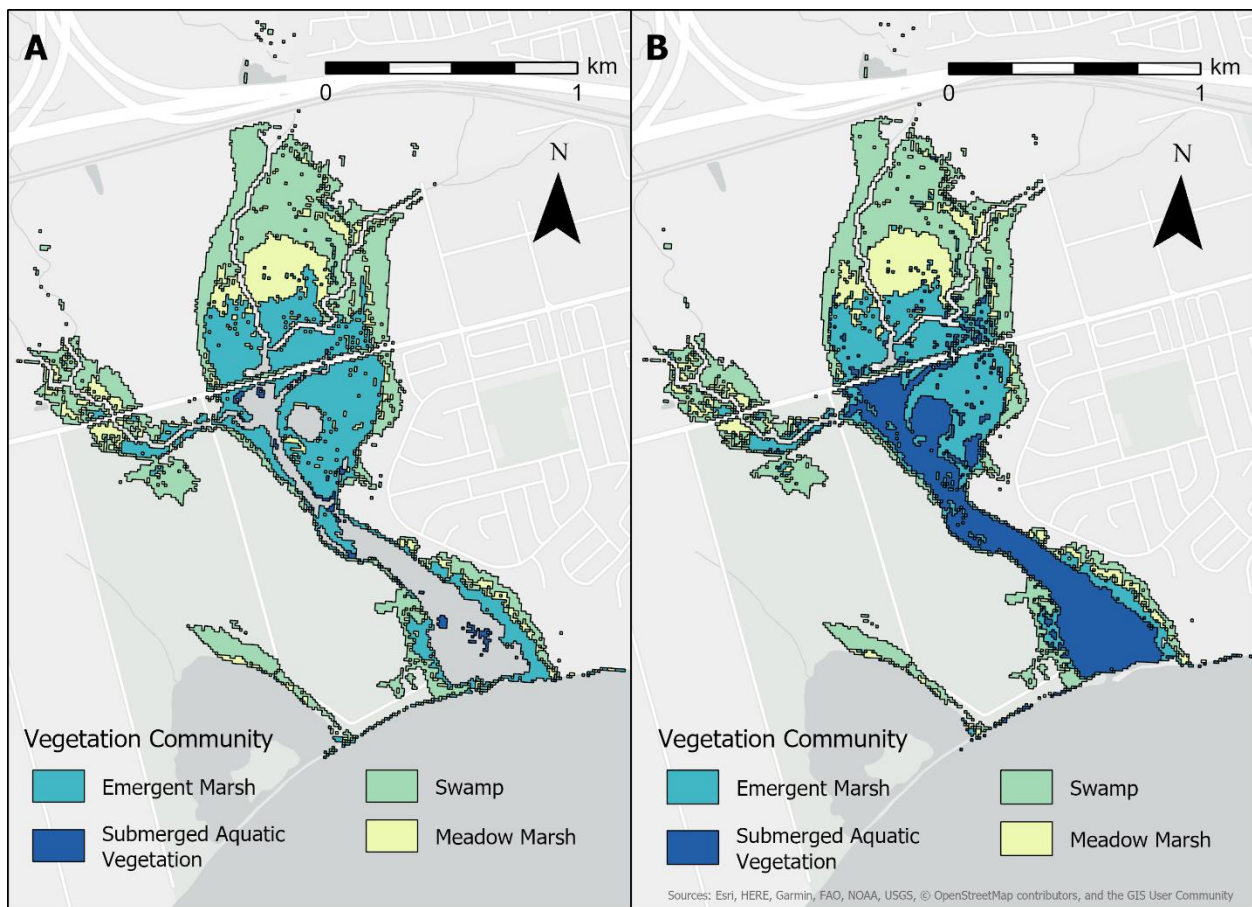


Figure 7: Two maps showing four vegetation communities at Lynde Creek, Lake Ontario. These vegetation groupings were used to calculate wetland diversity. Panel A shows a year with lower

wetland diversity (note there is very little submerged aquatic vegetation) and panel B shows a year with higher wetland diversity.

Long Description: This figure displays two maps of the Lynde Creek study site (in Lake Ontario) showing submerged aquatic vegetation (SAV), emergent marsh, meadow marsh, and swamp in two different years. These vegetation groupings were used to calculate wetland diversity (or Shannon Index), which was one of the five ecological attributes selected to assess Wetland Sensitivity to projected climate change. Both maps have the same number of vegetation communities, but the map on the left (panel A) has less evenness among vegetation communities because there is very little SAV, indicating lower wetland diversity. The map on the right (panel B) has more evenness among vegetation communities, indicating higher wetland diversity.

Change-detection analysis

For each site and simulation, projections for hindcast years (1981 to 2008) were compared to the projections for forecast years (2071 to 2098) to detect adverse effects of climate change on coastal wetlands. This change-detection approach is well established to assess effects of projected climate change (Alahuhta et al. 2011 ; Lee et al. 2015; Morales-Marin et al. 2019; Sofaer et al. 2016). The approach does not assume that a hindcast represents a reference condition or a healthy state that has not been impacted by climate change ; it is likely that all wetland study sites have been influenced by climate change to some degree. Instead, the hindcast period represents the range in wetland conditions (i.e., variability) through the recent past due to natural variation and human-related stressors. Comparing the forecast to a hindcast instead of historical observations is the best approach because it controls for any misspecification, bias, or imprecision in the GCM that could be falsely attributed to Wetland Sensitivity to climate change (Sofaer et al. 2016).

We sought a technical review of our change-detection approach by external experts (independent of ECCC). The reviewers were selected to represent a range of expertise related to climate change, national and regional wetland science, and statistics. Their input was incorporated into the following methods.

Threshold development

A response threshold was used to evaluate the negative effects of climate change on each ecological attribute. We set the response threshold based on the 10th percentile in the hindcast time series for all ecological attributes (i.e., when an ecological attribute was within the lowest 10% of hindcast values); values below this threshold signified an extreme negative state. We then applied this 10th percentile to the forecast to detect when extreme low values in each ecological attribute were predicted for each simulation (Figure 8). We also conducted our analysis using third and 20th percentile thresholds, but found that it did not meaningfully change the interpretation of the results; regardless of the threshold choice, all wetland sites were considered at risk or critically at risk in at least one simulation.

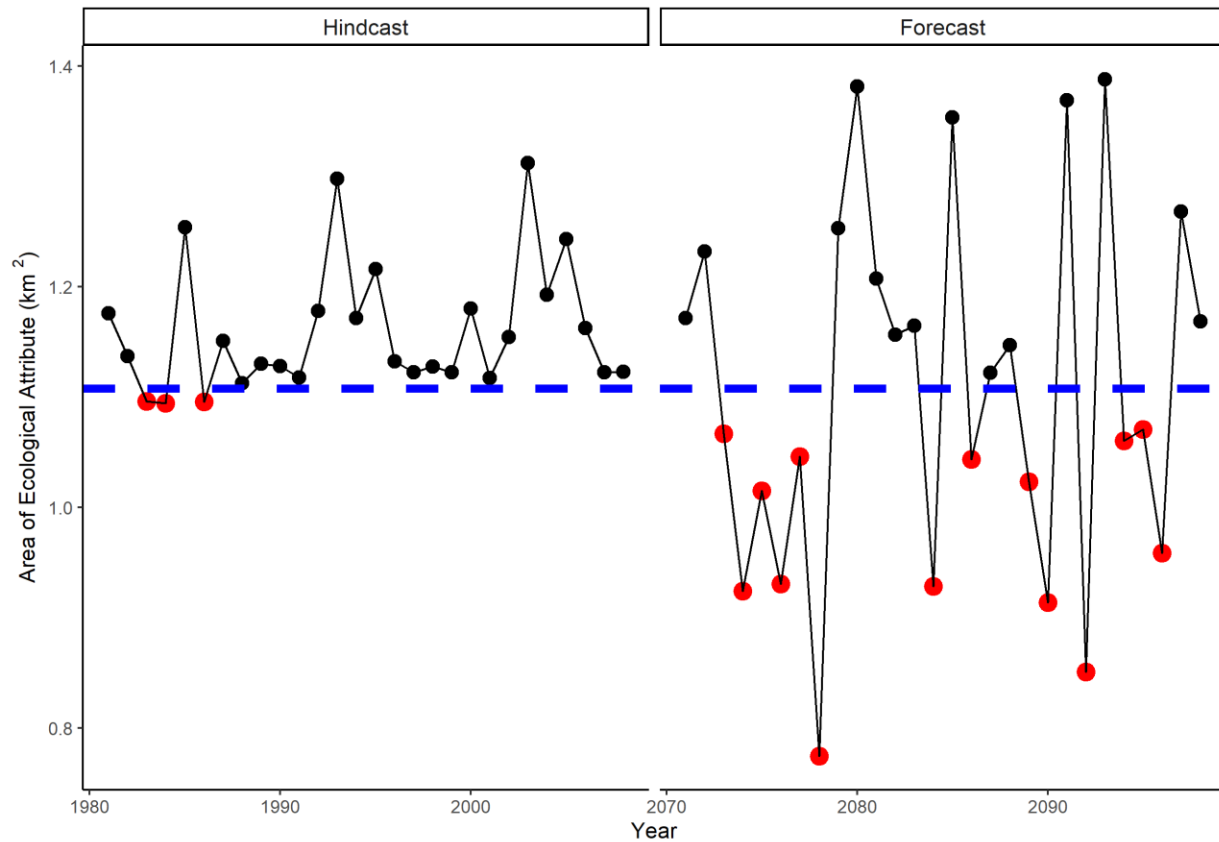


Figure 8: Depiction of how the 10th percentile threshold was set based on the hindcast of an ecological attribute, and how this threshold was applied to the forecast. The horizontal dashed blue line is the 10th percentile threshold and red circles represent all years that fell below the threshold.

Long Description: This figure provides a graphical example of how the 10th percentile threshold was set based on the hindcast data for an ecological attribute, and how that threshold was applied to the forecasted data. The y-axis of the graph is the area of an ecological attribute (in square kilometres), and the x-axis is year, split into two panels for the hindcast (1981 to 2008) and forecast (2071 to 2098) periods. In this example, the threshold based on the hindcast was set 1.11 square kilometres. There were three instances where area of the ecological attribute fell below the threshold in the hindcast, and 14 instances where it fell below the threshold in the forecast. Overall, the area of the ecological attribute was much more variable from year-to-year in the forecast than the hindcast.

Hindcast verification

Attribute Sensitivity thresholds depended on the hindcast, so we assessed whether there were any significant linear trends in the hindcasts that might lead us to set unrepresentative forecast thresholds. The threshold in a given forecast was set based on the lower 10th percentile of the associated hindcast, which, given the number of years in each hindcast, means it was effectively set based on the three years with the lowest values (or most extreme negative states) in the associated hindcast. If, for example, a strong negative trend in wetland area existed in the hindcast of a site or simulation, and the associated forecast's threshold was set based on the last three years of said hindcast, then this threshold might be considered unrepresentative since the temporal trend would suggest that more extreme negative states could continue to be observed if the time series were to continue. Setting thresholds in this way, based on the tail end of significantly linear hindcast trends, could thus lead to thresholds being set too high or low to effectively represent the frequency of extreme negative attribute states that existed in the past.

To avoid this issue, we analyzed each hindcast ($n = 200$) to detect significant linear trends using a bootstrapped student t-test ("notrend_test" function; funtimes package). Based on this test, which is designed to detect significant linear trends in time series data, nineteen hindcasts exhibited significant linear trends. We then ran simple linear models to determine the direction of the linear trend. Of the 19 significant trends, five were negative and 14 were positive. We inspected all 19 significant trends visually to determine whether they were extreme enough to have led to the establishment of unrepresentative thresholds. Upon examination, we were confident that the directional trends did not lead to unrepresentative thresholds. There

were no situations where thresholds were based on either the first three or last three years of a hindcast with a strong negative or positive trend.

Attribute aggregation

We calculated Attribute Sensitivity (i.e., the proportion of extreme forecast years) for each site and simulation by summing the number of extreme forecast years for each ecological attribute and dividing by the number of forecast years (28).

We also calculated the overall Wetland Sensitivity for each site and simulation by summing the number of extreme forecast years for all five ecological attributes and dividing by the total number of forecast years across all ecological attributes (140). This represented the proportion of years that the ecological attributes existed in an extreme negative state for each site and simulation.

We classified Wetland Sensitivity for each site and simulation into colour-coded Sensitivity scores and gave qualitative descriptors of the risk that a wetland will experience negative effects of climate change in the future (Table 4). We considered a coastal wetland to be low risk when the percentage of extreme forecast years across the five ecological attributes was less than 10%; in other words, the number of extreme forecast years did not exceed the number of extreme hindcast years. Our wetland experts and stakeholders identified that negative responses that were beyond the normal range of variation for a given wetland were of conservation priority under climate change. When there were more than 10% extreme forecast years, we categorized risk according to Table 4. While the choice of threshold defining at risk and critically at risk was subjective, splitting the remaining percentiles in this way is analogous to published work on alert thresholds for environmental variables (Van Wynsberge et al. 2013).

Table 4: Scoring system for Wetland Sensitivity for each site and simulation. Wetland Sensitivity is a proportion (0 to 1). Score is a colour classification associated with the Wetland Sensitivity (green, yellow, and red). The interpretation is based on the risk that a wetland may experience negative effects of climate change in the projected future.

Wetland Sensitivity	Score	Interpretation
< 0.1	Green	Low Risk
0.1 - 0.55	Yellow	At Risk
> 0.55	Red	Critically At Risk

One shortcoming of additive indices is that they can be biased by highly correlated variables, as these can lead to pseudoreplication or “double counting” within an index (Gibbons et al. 2006). Therefore, to ensure independence among ecological attributes (selection criteria 3), we examined the Pearson correlations among Attribute Sensitivity values. We considered ecological attributes to be correlated when $r > 0.6$.

Post-hoc analyses

To gain insight into the mechanisms driving our results, we explored the relationship between physical variables used in the CWRM (i.e., elevation and water level) and Wetland Sensitivity. Firstly, we assessed whether Wetland Sensitivity was affected by the topographic profile of a site because we suspected that sites with more variation in elevations would have a greater ability to migrate in response to water-level fluctuations, and therefore would be more resilient to climate change. Topographic elevation data above the mean water level was extracted from a digital elevation model. We used a general linear model with Wetland Sensitivity as the response variable and standard deviation in elevation and simulation as explanatory variables. An interaction term between elevation and simulation was also initially included in the model, but was dropped because it was not statistically significant. Standard deviation was used as a measure of variation in elevation at a site and was selected rather than range in elevations because the edges of a topographic profile are often represented by very little area, and therefore, range may provide a misleading representation of the distribution of elevations present at a site. Standard deviation of elevation was transformed with a log base 10 transformation to accommodate scaling issues. All model assumptions were met.

Secondly, we assessed whether water-level fluctuations were associated with the probability of an extreme year of total wetland area. We chose to assess the relationship with total wetland area because it was the most sensitive attribute to climate change in this study. We used generalized linear mixed modelling to determine if the occurrence of an extreme forecast year of total wetland area was affected by the mean water level in the preceding three years. This explanatory variable was selected because wetland vegetation reacts to water-level fluctuations with a time lag (Wilcox and Nichols 2008) and because it was shown to be highly important in the CWRM. The models were split among lakes because of differences in water levels that caused scaling issues and also to improve interpretability. Note that the three sites in the Huron-Erie Corridor were included in the Lake Erie model and the one site in the St. Lawrence River was included in the Lake Ontario model. In each of the generalized linear mixed models, we used the binary response variable of whether or not total wetland area exceeded the response threshold in a given year. Explanatory variables in each model were three-year-mean water level and simulation. An interaction term between three-year-mean water level and simulation was also initially included in each model, but was dropped from the Lake Erie model because it was not statistically significant. Year and site were initially included as random effects in all models, but site was dropped from the Lake Erie model due to issues with model fit and because it did not significantly improve the model. The significance of random effects was determined based on likelihood ratio tests. To accommodate scaling issues, three-year-mean water level was converted to a Z-score ($Z = (x - \mu) / \sigma$) in all models, and was transformed with a log base 10 transformation in the models for lakes Ontario and Huron. Mixed models were performed using the “glmer” function in the lme4 package (Version 1.1-26; Bates et al. 2015) and met model assumptions. All statistical analyses were conducted in R version 4.0.3 (R Core Team 2020).

Results

Ecological Attribute Sensitivity

Median Attribute Sensitivity for the five ecological attributes ranged from 0.07 to 0.20 in the lower-bound simulation and 0.07 to 0.71 in the upper-bound simulation, and was higher in the upper-bound simulation for three attributes (Figure 9). Area-based attributes (meadow marsh area and total wetland area) were especially sensitive in the upper-bound simulation. Total wetland area in the upper-bound simulation had the highest median Attribute Sensitivity (0.71), followed by meadow marsh area in the upper-bound simulation (0.61) (Figure 9). Interspersion had the highest median Attribute Sensitivity in the lower-bound simulation (0.20), followed by SAV volume and meadow marsh area (both 0.16). Ecological attributes were not strongly correlated ($r < 0.6$; Table 5).

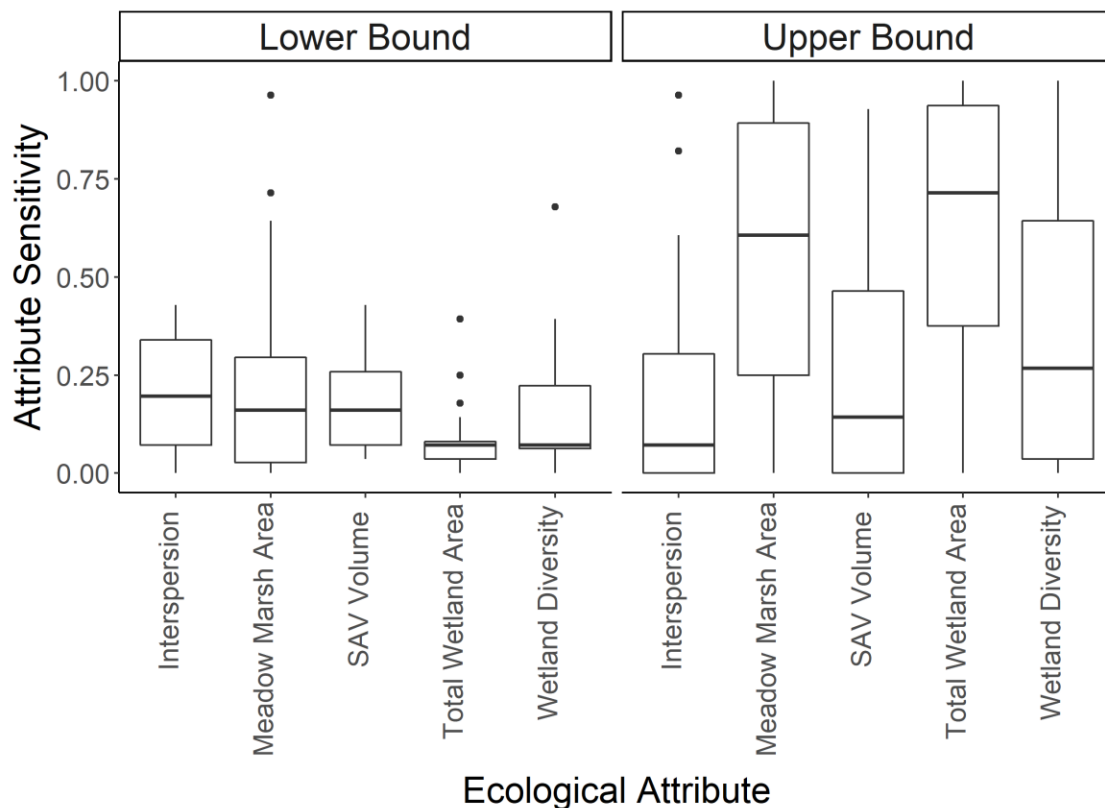


Figure 9: Boxplots of Attribute Sensitivity values for all sites in upper and lower-bound simulations. The horizontal line in each box is the median, the bottom of each box is the 1st quartile, the top of each box is the 3rd quartile, and the whiskers represent values within 1.5 times the interquartile range.

Long description: This figure displays boxplots summarizing Attribute Sensitivity values for all coastal wetland study sites for each of the ecological attributes in both the upper and lower-bound simulations. The y-axis is Attribute Sensitivity and the x-axis is the name of each ecological attribute, split into two panels for the upper and lower-bound simulations. Attribute Sensitivity was low for all ecological attributes in the lower-bound simulation, but was generally higher and more variable in the upper-bound simulation. Total wetland area in the upper-bound simulation had the highest median Attribute Sensitivity, followed by meadow marsh area in the upper-bound simulation.

Table 5: Pearson correlation coefficients between Attribute Sensitivity values for all combinations of the five ecological attributes. None of the ecological attributes were considered to be correlated ($r < 0.6$).

	Total Wetland Area	Wetland Diversity	SAV Volume	Meadow Marsh Area
Wetland Diversity	0.42	-	-	-
SAV Volume	0.46	0.33	-	-
Meadow Marsh Area	0.44	0.49	0.13	-
Wetland Interspersion	0.18	0.08	0.23	-0.22

Wetland Sensitivity

All coastal wetland study sites were considered at risk (Wetland Sensitivity ≥ 0.1) in at least one simulation, and five wetland sites (25%) were critically at risk (Wetland Sensitivity > 0.55) in at least one simulation (Figures 10 and 11). Ten sites (50%) had different Wetland Sensitivity scores between simulations, but no site was low risk in one simulation and critically at risk in the other.

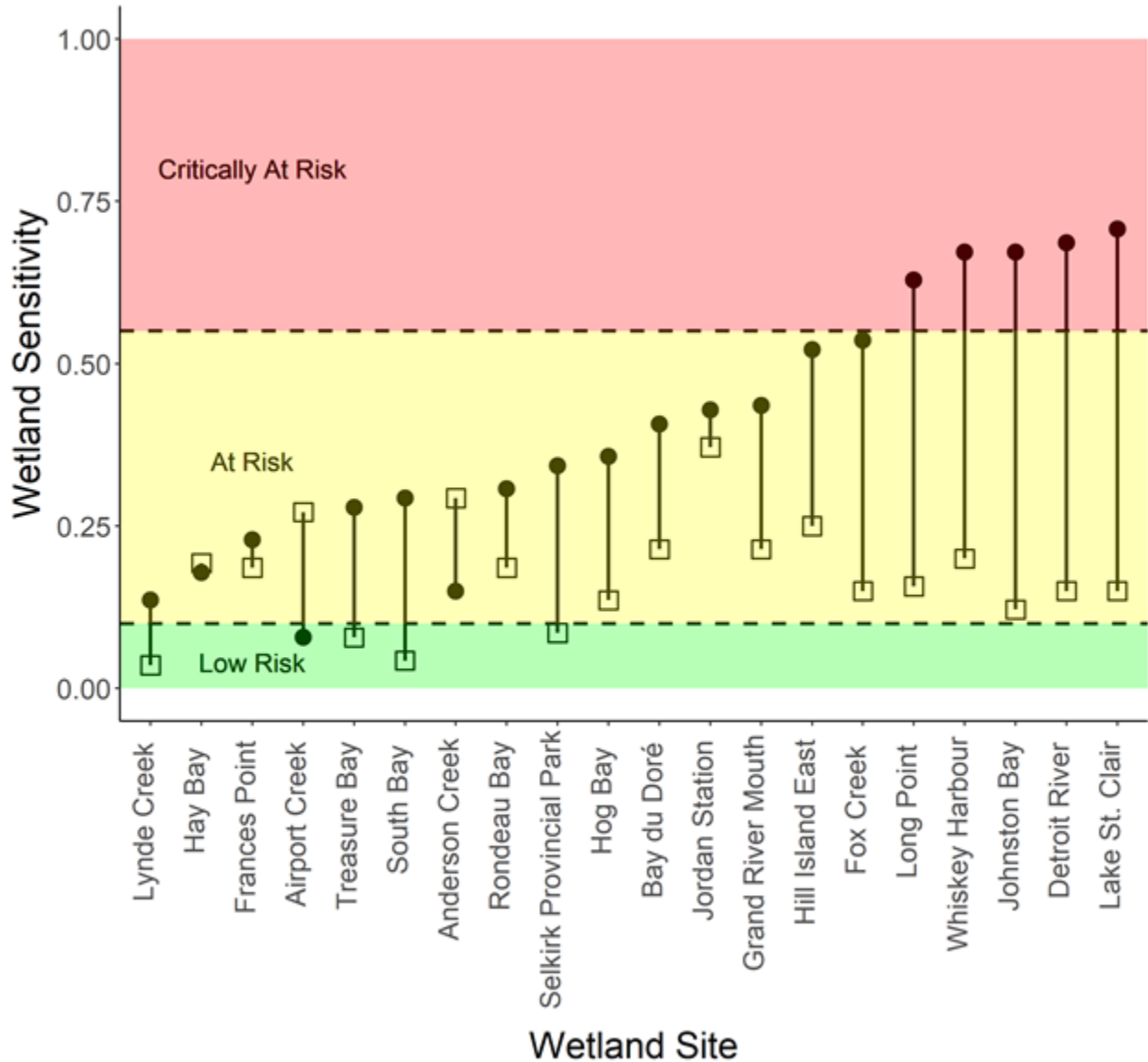


Figure 10: Wetland Sensitivity values for upper and lower-bound simulations for all coastal wetland sites (n = 20), ordered by maximum Wetland Sensitivity. The circles indicate the upper-bound simulation and the squares indicate the lower-bound simulation.

Long description: This figure displays a graph showing Wetland Sensitivity values for all 20 coastal wetland study sites in the upper and lower-bound simulations. The y-axis is Wetland Sensitivity and the x-axis is the name of each wetland site. The Wetland Sensitivity values for both simulations are provided for each site using different symbols. All study sites were considered at risk to projected climate change in at least one simulation. Five sites were considered critically at risk (all in the upper-bound simulation), and five sites were considered

low risk (four in the lower-bound simulation and one in the upper-bound simulation). For all but three sites (Airport Creek, Anderson Creek, and Hay Bay), Wetland Sensitivity was higher in the upper-bound simulation than the lower-bound simulation. Sites have been ordered from smallest maximum Wetland Sensitivity to largest. The lowest maximum Wetland Sensitivity was for Lynde Creek in the upper-bound simulation, and the highest maximum Wetland Sensitivity was for Lake St. Clair in the upper-bound simulation.

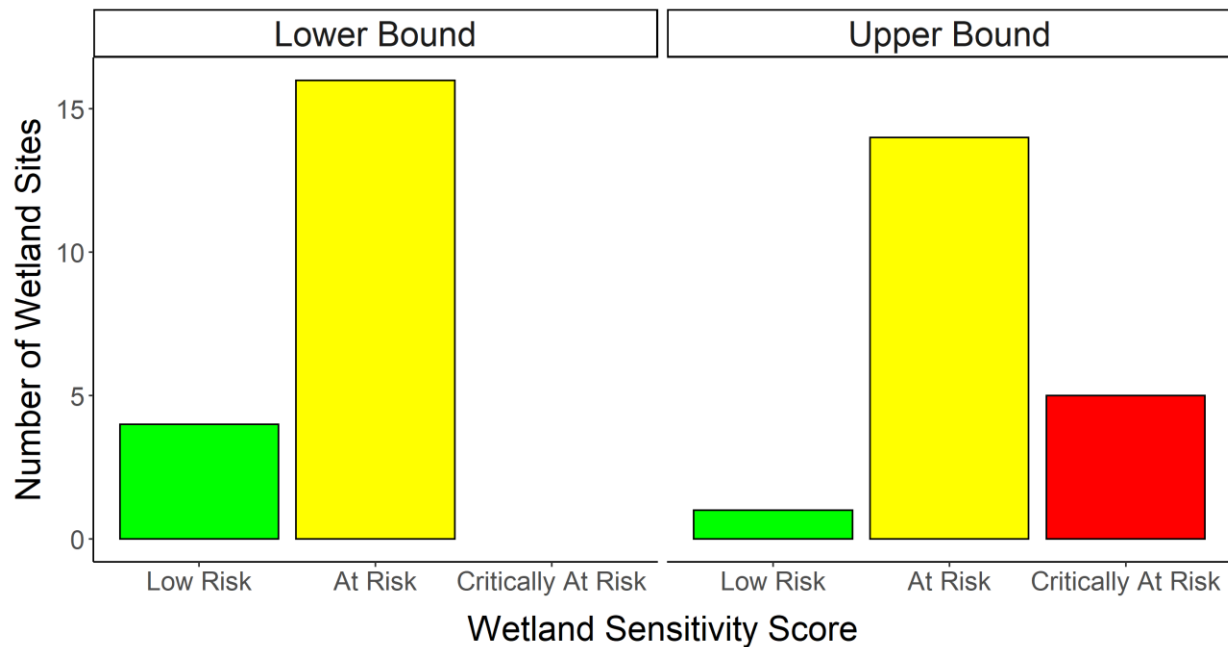


Figure 11: Distribution of Wetland Sensitivity scores in the upper and lower-bound simulations.

Long Description: This figure displays a histogram showing the number of coastal wetland study sites in the three risk categories (low risk, at risk, critically at risk) for both the upper and lower-bound simulations. The y-axis is the number of wetland sites and the x-axis are the risk categories, split into two panels for the upper and lower-bound simulations. In the lower-bound simulation, four sites were considered low risk, 16 sites were considered at risk, and no sites were considered critically at risk. In the upper-bound simulation, one site was low risk, 14 sites were at risk, and five sites were critically at risk.

Sensitivity values were generally higher in the upper-bound simulation than the lower-bound simulation, with the exception of Anderson Creek, Airport Creek, and Hay Bay. In the lower-bound simulation, 16 sites (80%) were considered at risk and no sites were critically at

risk (Figure 12). In the upper-bound simulation, 14 sites (70%) were considered at risk and five sites (25%) were critically at risk (Figure 13).

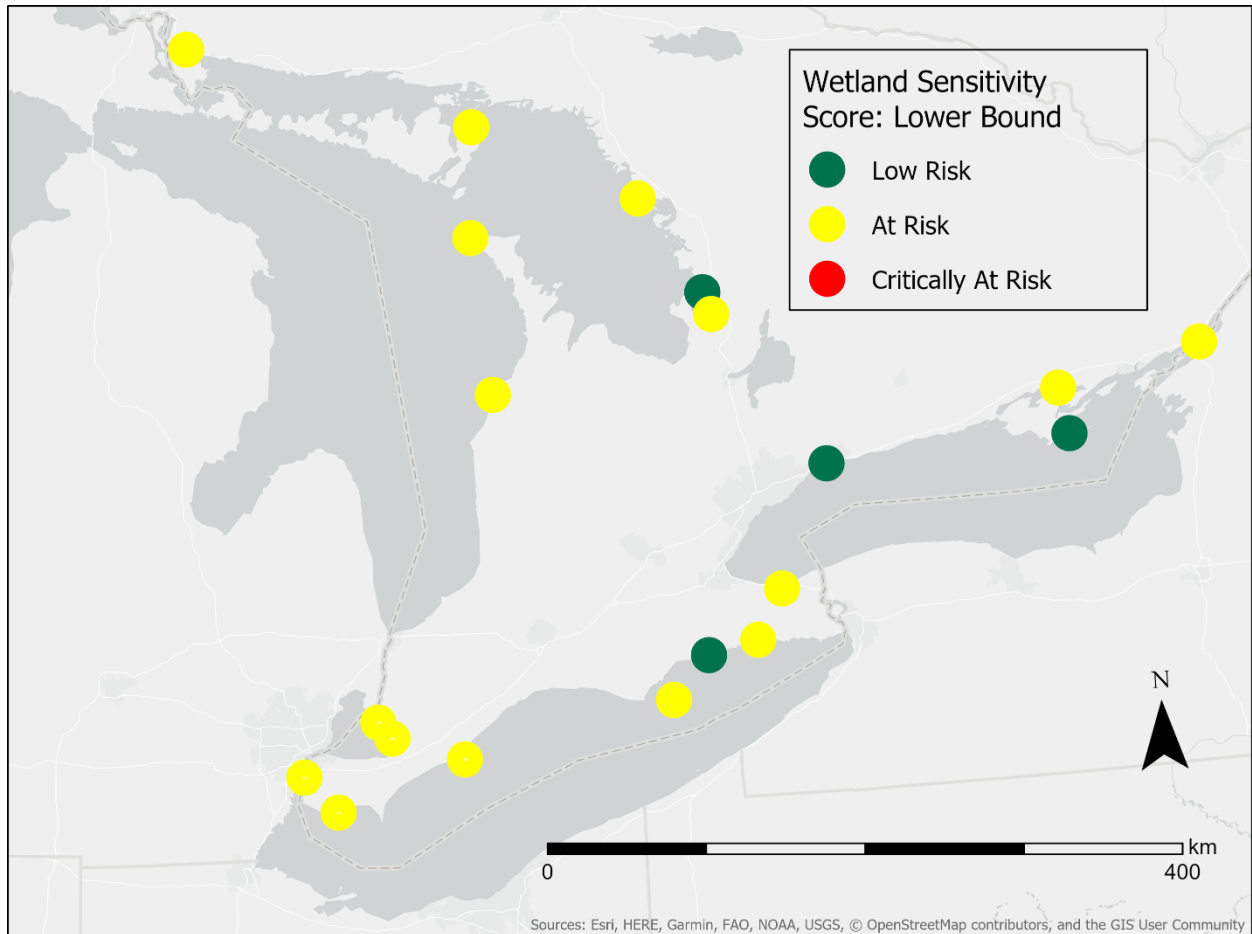


Figure 12: Map showing Wetland Sensitivity scores for the 20 Great Lakes coastal wetland sites in the lower-bound simulation. Site names are in Figure 3.

Long Description: This figure displays a map of the study area with Wetland Sensitivity scores for all 20 coastal wetland study sites in the lower-bound simulation. Four sites were considered low risk: one in Lake Huron (Treasure Bay), one in Lake Erie (Selkirk Provincial Park), and two in Lake Ontario (South Bay and Lynde Creek). All other sites were considered at risk.

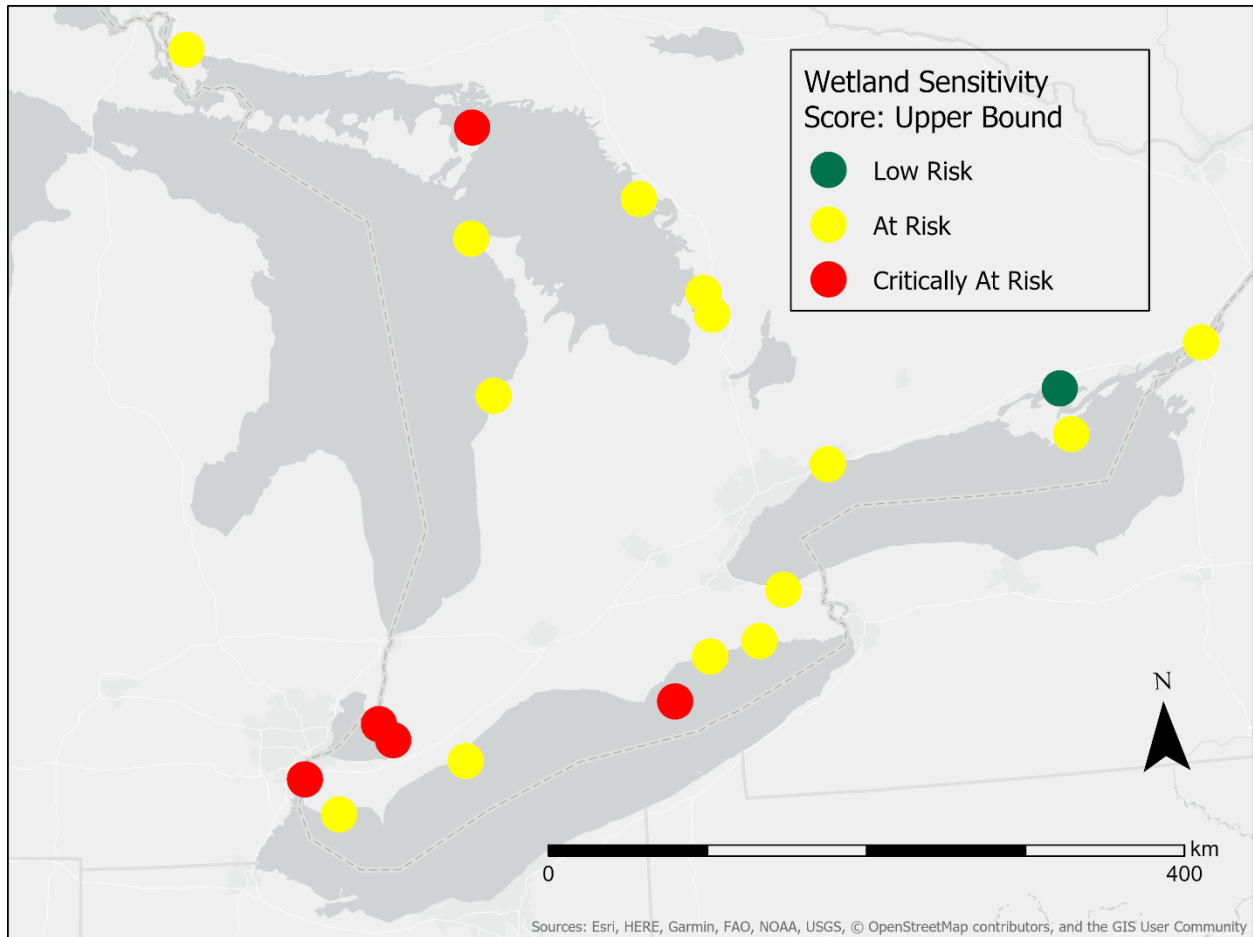


Figure 13: Map showing Wetland Sensitivity scores for the 20 Great Lakes coastal wetland sites in the upper-bound simulation. Site names are in Figure 3.

Long Description: This figure displays a map of the study area with Wetland Sensitivity scores for all 20 coastal wetland study sites in the upper-bound simulation. One site was considered low risk (Airport Creek on Lake Ontario) and five sites were considered critically at risk: one in Lake Huron (Whiskey Harbour), three in the Huron-Erie Corridor (Detroit River, Johnston Bay, Lake St. Clair), and one in Lake Erie (Long Point). All other sites were considered at risk.

Sensitivity values were generally similar among water bodies in the lower-bound simulation, but were notably higher in the Huron-Erie Corridor, and to a lesser extent Lake Erie, in the upper-bound simulation (Table 6). All three sites in the Huron-Erie Corridor were considered critically at risk in the upper-bound simulation. All Lake Erie and Lake Huron sites were considered at risk in the upper-bound simulation, with the exceptions of Long Point (Lake Erie) and Whiskey Harbour (Lake Huron), which were both considered critically at risk (but we

note that the Sensitivity of Whiskey Harbour may have been overestimated due to data limitations; ECCC 2022a). However, Lake Erie sites had higher Sensitivity values than Lake Huron sites, on average (mean = 0.45 and 0.33, respectively). Sites on Lake Ontario were the least sensitive in the upper-bound simulation; no site was considered critically at risk, and Airport Creek Marsh was the only site that was low risk across all lakes. Additionally, there was no notable difference between lacustrine and riverine wetland hydrogeomorphic types in either simulation (Table 6).

Table 6: Summary of results for each site showing wetland name, wetland hydrogeomorphic (HGM) type, Wetland Sensitivity (with colour-coded score), and most and least sensitive ecological attributes. WI = Wetland Interspersion, MMA = Meadow Marsh Area, TWA = Total Wetland Area, SAVV = Submerged Aquatic Vegetation Volume, WD = Wetland Diversity.

Site Name	HGM Type	Scenario					
		Lower-bound Simulation			Upper-bound Simulation		
		Wetland Sensitivity	Most	Least	Wetland Sensitivity	Most	Least
Rondeau Bay	Lacustrine, Sand-spit embayment	0.19	WI (0.43)	TWA & MMA (0.071)	0.31	TWA (0.96)	MMA & WD (0)
Fox Creek	Lacustrine, Barred drow ned river-mouth	0.15	SAVV (0.39)	MMA (0)	0.54	SAVV (0.93)	WD (0.18)
Detroit River	Riverine, Open shoreline	0.15	SAVV (0.32)	MMA (0)	0.69	TWA & WD (1)	MMA (0)
Lake St. Clair	Riverine, Open shoreline	0.15	WI (0.43)	TWA, SAVV & WD (0.071)	0.71	TWA (1)	WI (0.25)
Johnston Bay	Riverine, Delta marsh	0.12	WI (0.39)	MMA (0)	0.67	TWA & MMA (1)	WI (0)
Long Point	Lacustrine, Sand-spit embayment	0.16	WI (0.32)	WD (0.036)	0.63	TWA & MMA (1)	WD (0.036)
Selkirk Provincial Park	Riverine, Barred drow ned river-mouth	0.09	MMA (0.29)	TWA (0.036)	0.34	TWA (0.89)	SAVV & WI (0)

Site Name	HGM Type	Scenario					
		Lower-bound Simulation			Upper-bound Simulation		
		Wetland Sensitivity	Most	Least	Wetland Sensitivity	Most	Least
Grand River Mouth	Riverine, Barred drow ned river-mouth	0.21	WD (0.39)	WI (0.071)	0.44	TWA (0.93)	WI (0.036)
Jordan Station	Riverine, Drow ned river-mouth	0.37	MMA (0.96)	TWA (0)	0.43	MMA (0.89)	WI (0)
Lynde Creek	Riverine, Barred drow ned river-mouth	0.04	TWA & WI (0.071)	MMA & WD (0)	0.14	TWA (0.50)	SAVV & WI (0)
South Bay	Lacustrine, Open embayment	0.04	SAVV (0.071)	MMA (0)	0.29	MMA (0.79)	SAVV (0)
Airport Creek	Riverine, Open drow ned river-mouth	0.27	MMA (0.71)	WD & WI (0.071)	0.08	MMA (0.29)	SAVV (0)
Hay Bay	Lacustrine, Protected embayment	0.19	WD (0.36)	TWA & SAVV (0.071)	0.18	WI (0.61)	SAVV (0)
Treasure Bay	Lacustrine, Protected embayment	0.08	MMA (0.18)	WD (0)	0.28	TWA (0.39)	SAVV & WD (0.21)
Whiskey Harbour	Lacustrine, Protected embayment	0.20	SAVV & MMA (0.29)	TWA (0.036)	0.67	MMA (1)	WI (0.29)
Anderson Creek	Riverine, Open drow ned river-mouth	0.29	MMA (0.46)	WD (0.036)	0.15	MMA (0.68)	TWA, SAVV & WD (0)
Hog Bay	Lacustrine, Protected embayment	0.14	WD (0.25)	TWA & WI (0.036)	0.36	TWA (0.64)	WI (0.21)
Baie du Doré	Lacustrine, Open embayment	0.21	WI (0.39)	MMA (0.036)	0.41	TWA (0.82)	SAVV (0)
Hill Island East	Riverine, Protected embayment	0.25	MMA (0.64)	WI (0)	0.52	MMA (0.89)	WI (0.32)
Frances Point	Lacustrine, Protected embayment	0.19	SAVV (0.39)	MMA (0)	0.23	SAVV (0.93)	MMA (0.32)

Post-hoc analyses

Wetland Sensitivity decreased significantly with increased standard deviation in elevation at a site ($\beta = -0.16$, $SE = 0.07$, $t = -2.42$, $p = 0.020$; Figure 14). This indicates that wetlands with less topographic relief had higher Sensitivity values. In the site elevation model, Wetland Sensitivity was significantly higher in the upper-bound simulation compared to the lower-bound simulation ($\beta = 0.23$, $SE = 0.05$, $t = 4.98$, $p < 0.001$).

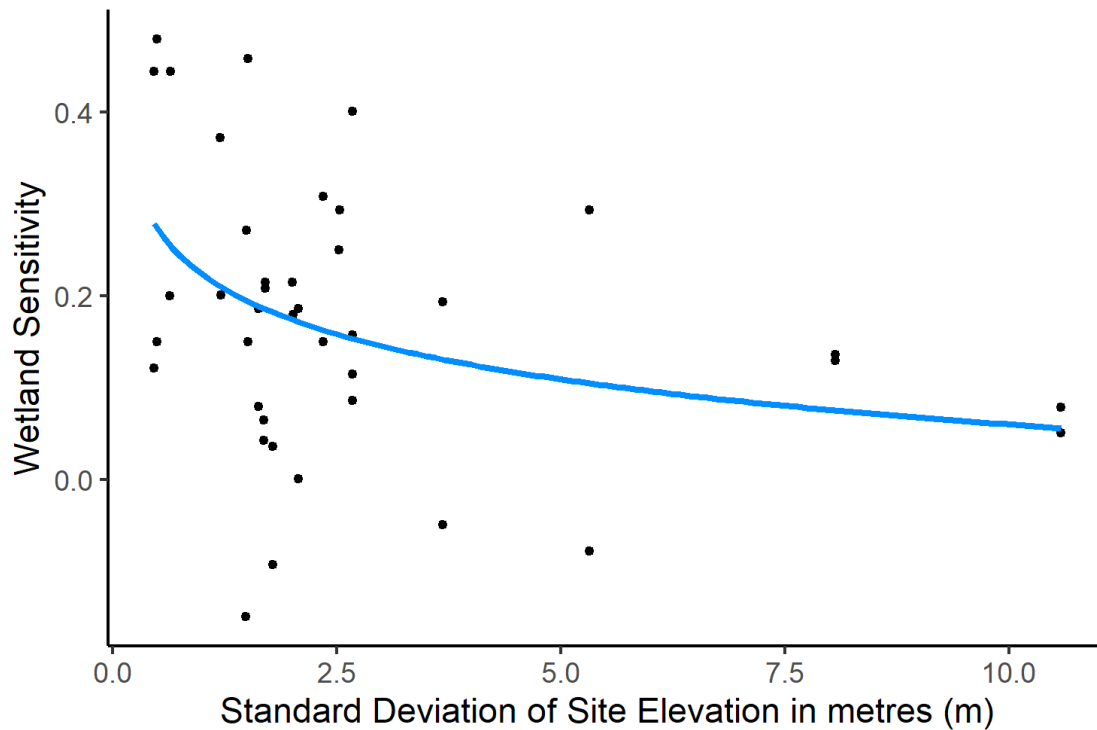


Figure 14: Partial plot of the relationship between standard deviation of site elevation (in metres) and Wetland Sensitivity for all 20 study sites in both simulations ($\beta = -0.16$, $SE = 0.07$, $t = -2.42$, $p = 0.020$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

Long Description: This figure displays a partial plot showing the relationship between the standard deviation of site elevation (in metres; x-axis) and Wetland Sensitivity (y-axis) for all 20 coastal wetland sites in both simulations. Wetland Sensitivity decreased with increasing standard deviation of site elevation ($\beta = -0.16$, $SE = 0.07$, $t = -2.42$, $p = 0.020$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

Additionally, we found that the effect of three-year-mean water level on the probability of an extreme year of total wetland area differed among lakes. In the Lake Erie model (Table 7), the probability of an extreme year significantly increased with three-year-mean water level ($\beta = 1.13$, $SE = 0.56$, $Z = 2.02$, $P = 0.044$), and was significantly higher in the upper-bound simulation compared to the lower-bound simulation ($\beta = 10.18$, $SE = 1.94$, $t = 5.24$, $p < 0.001$) (Figure 15).

Table 7: Results from the generalized linear mixed model for Lake Erie and Huron-Erie Corridor study sites. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value.

Variable – Variable Type	$\beta \pm SE$	Z	δ^2	X^2_1	P
3-year-mean water level – fixed	1.13 \pm 0.56	2.02	-	-	0.044
Simulation – fixed	10.18 \pm 1.94	5.24	-	-	< 0.001
Year – random	-	-	94.13	31.14	< 0.001

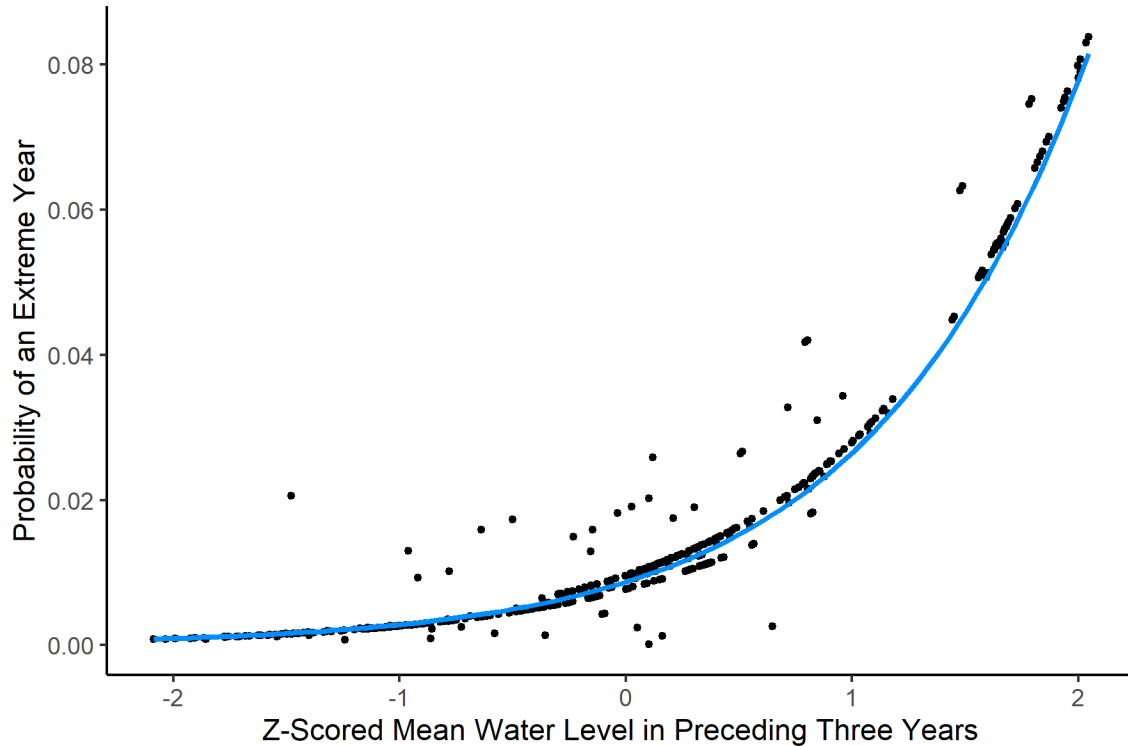


Figure 15: Partial plot of the relationship between Z-scored three-year-mean water level and the probability of an extreme year of total wetland area for Lake Erie and Huron-Erie Corridor study sites ($\beta = 1.13$, $SE = 0.56$, $Z = 2.02$, $P = 0.044$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

Long Description: This figure displays a partial plot showing the relationship between three-year-mean water level (Z-scored; x-axis) and the probability of an extreme year of total wetland area (y-axis) for Lake Erie and Huron-Erie Corridor coastal wetland sites. Probability of an extreme year significantly increased with increasing three-year-mean water level ($\beta = 1.13$, $SE = 0.56$, $Z = 2.02$, $P = 0.044$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

There was a significant interaction between simulation and three-year-mean water level in the models for lakes Ontario and Huron, indicating that the relationship between three-year-mean water level and probability of an extreme year differed significantly between simulations. In the Lake Ontario model (Table 8), there was a significant negative interaction ($\beta = -38.88$, $SE = 8.27$, $Z = -4.70$, $P < 0.001$), and in the Lake Huron model (Table 9), there was a significant positive interaction ($\beta = 36.10$, $SE = 18.33$, $Z = 1.97$, $P = 0.049$).

Table 8: Results from the generalized linear mixed model for Lake Ontario and St. Lawrence River study sites. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value.

Variable – Variable Type	$\beta \pm SE$	Z	δ^2	X^2_1	P
3-year-mean water level – fixed	19.12 \pm 4.70	4.07	-	-	< 0.001
Simulation – fixed	17.74 \pm 3.75	4.73	-	-	< 0.001
Simulation: 3-year-mean water level – fixed	-38.88 \pm 8.27	-4.70	-	-	< 0.001
Year – random	-	-	220.52	18.15	< 0.001
Site – random	-	-	226.84	11.82	< 0.001

Table 9: Results from the generalized linear mixed model for Lake Huron study sites. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value.

Variable – Variable Type	$\beta \pm SE$	Z	δ^2	X^2_1	P
3-year-mean water level – fixed	17.78 \pm 5.96	2.98	-	-	0.003
Simulation – fixed	-21.91 \pm 9.24	-2.37	-	-	0.018
Simulation: 3-year-mean water level – fixed	36.10 \pm 18.33	1.97	-	-	0.049
Year – random	-	-	315.55	17.85	< 0.001
Site – random	-	-	299.59	33.81	< 0.001

To interpret these interactions, we fit generalized linear mixed effects models for each simulation separately for both lakes. In these models, the response variable was the binary response variable of whether or not total wetland area was in an extreme negative state (i.e., exceeded the response threshold), the fixed effect was three-year-mean water level with a log base 10 transformation (to accommodate scaling issues), and the random effects were year and site. In the lower-bound simulation model for Lake Ontario (Table 10), there was a marginally significant increase in the probability of an extreme year of total wetland area with increasing three-year-mean water level ($\beta = 11.82$, SE = 6.72, Z = 1.76, P = 0.078, Figure 16).

Alternatively, in the upper-bound simulation model for Lake Ontario (Table 11), the effect of three-year-mean water level on the probability of an extreme year was not statistically significant ($\beta = -13.27$, SE = 23.77, Z = -0.56, P = 0.577). In the upper-bound simulation model for Lake

Huron (Table 12), there was a significant increase in the probability of an extreme year with increasing three-year-mean water level ($\beta = 49.55$, $SE = 23.51$, $Z = 2.11$, $P = 0.035$, Figure 17). In the lower-bound simulation model for Lake Huron, the effect of three-year-mean water level on the probability of an extreme year was positive; however, model assumptions were violated due to a significant non-linearity, meaning that the statistical inferences derived from the model were not reliable.

Table 10: Results from the generalized linear mixed model for Lake Ontario and St. Lawrence River study sites for the lower-bound simulation. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value.

Variable – Variable Type	$\beta \pm SE$	Z	δ^2	X^2_1	P
Three-year-mean water level – fixed	11.82 \pm 6.72	1.76	-	-	0.078
Year – random	-	-	75.31	3.91	0.048
Site – random	-	-	77.68	1.54	0.215

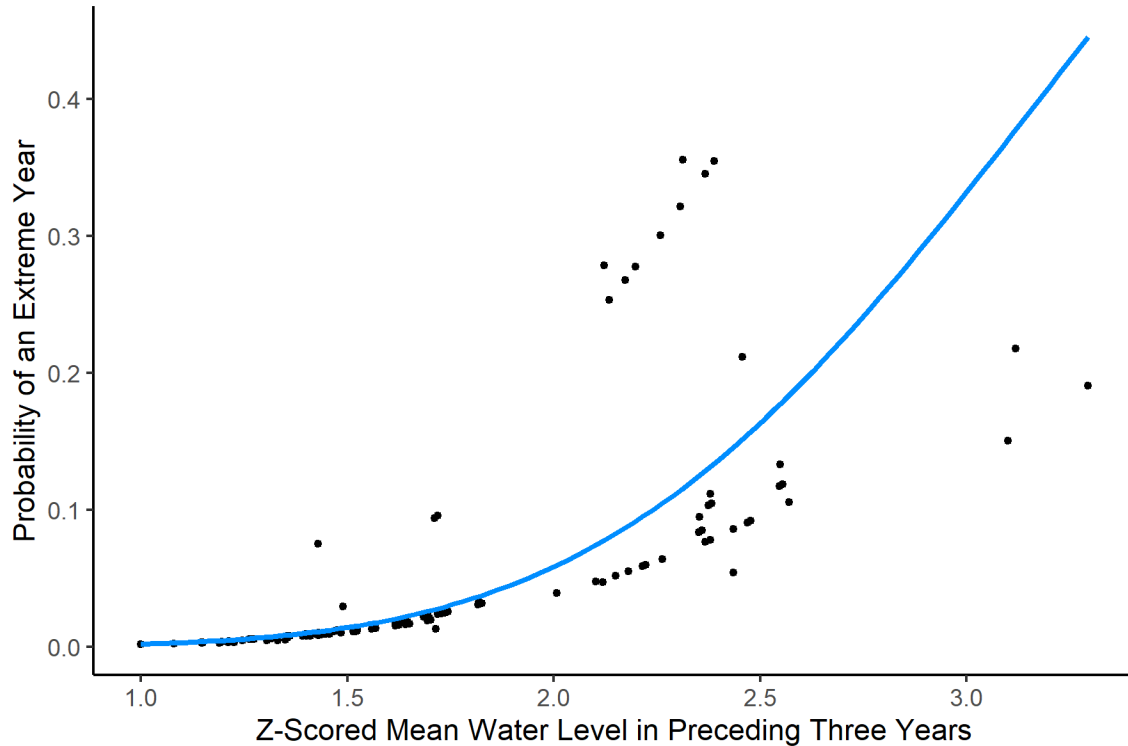


Figure 16: Partial plot of the relationship between Z-scored three-year-mean water level and the probability of an extreme year of total wetland area for Lake Ontario and St. Lawrence River study sites in the lower-bound simulation ($\beta = 11.82$, $SE = 6.72$, $Z = 1.76$, $P = 0.078$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

Long Description: This figure displays a partial plot showing the relationship between three-year-mean water level (Z-scored; x-axis) and the probability of an extreme year of total wetland area (y-axis) for Lake Ontario and St. Lawrence River study sites in the lower-bound simulation. Probability of an extreme year increased with increasing three-year-mean water level ($\beta = 11.82$, $SE = 6.72$, $Z = 1.76$, $P = 0.078$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

Table 11: Results from the generalized linear mixed model for Lake Ontario and St. Lawrence River study sites for the upper-bound simulation. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value.

Variable – Variable Type	$\beta \pm SE$	Z	δ^2	X^2_1	P
Three-year-mean water level – fixed	-13.27 \pm 23.77	-0.56	-	-	0.577
Year - random	-	-	122.17	37.28	< 0.001
Site – random	-	-	145.43	14.02	< 0.001

Table 12: Results from the generalized linear mixed model for Lake Huron study sites for the upper-bound simulation. Results from the likelihood ratio test used to test the significance of the random effect are also shown. β = estimate, SE = standard error, Z = Z-score, δ^2 = deviance, X^2_1 = Chi square value with 1 degree of freedom, P = P-value.

Variable – Variable Type	$\beta \pm SE$	Z	δ^2	X^2_1	P
Three-year-mean water level – fixed	49.55 \pm 23.51	2.11	-	-	0.035
Year - random	-	-	225.88	6.62	0.010
Site - random	-	-	182.96	49.53	< 0.001

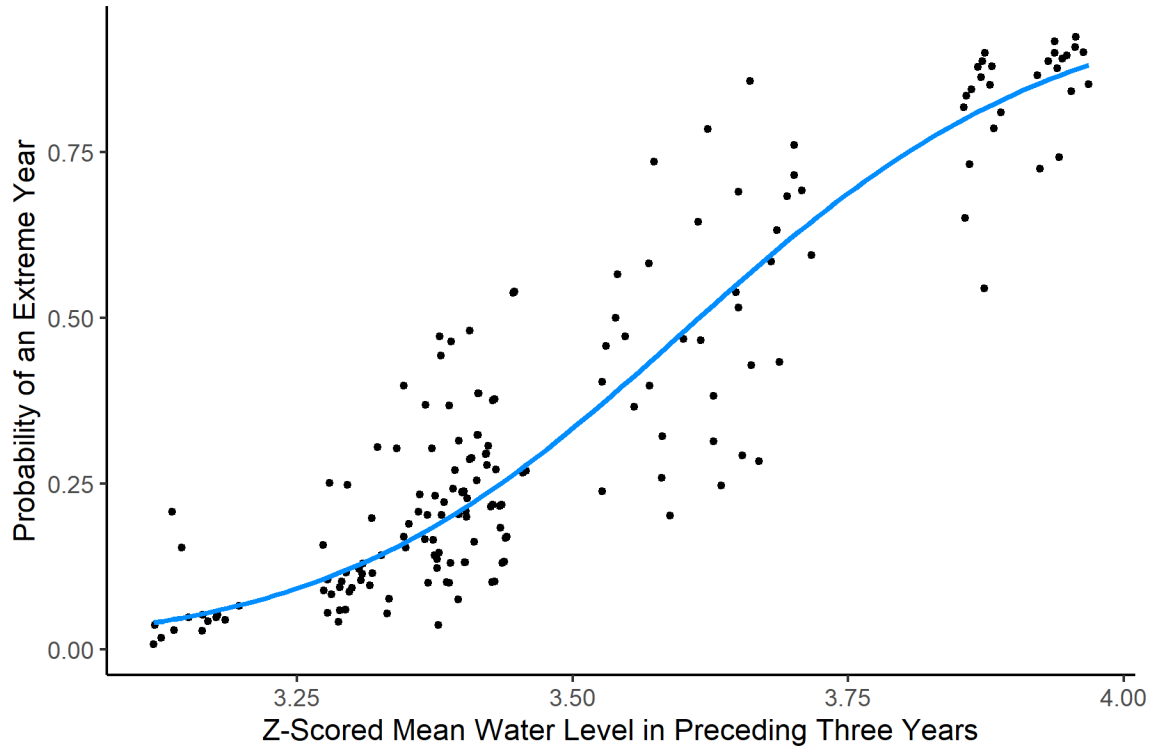


Figure 17: Partial plot of the relationship between Z-scored three-year-mean water level and the probability of an extreme year of total wetland area for Lake Huron study sites in the upper-bound simulation ($\beta = 49.55$, $SE = 23.51$, $Z = 2.11$, $P = 0.035$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

Long Description: This figure displays a partial plot showing the relationship between three-year-mean water level (Z-scored; x-axis) and the probability of an extreme year for total wetland area (y-axis) for Lake Huron study sites in the upper-bound simulation. Probability of an extreme year significantly increased with increasing three-year-mean water level ($\beta = 49.55$, $SE = 23.51$, $Z = 2.11$, $P = 0.035$). Note that a partial plot shows the relationship between two variables, after accounting for other explanatory variables in a statistical model.

Discussion

Our results suggest that all coastal wetland sites in this study are at risk under climate change. This is consistent with the IPCC assessment of wetlands as among the most vulnerable ecosystems to climate change (IPCC 2014; Lee et al. 2015). This work builds upon existing impact studies or vulnerability assessments that have found wetland flora and fauna to be vulnerable to climate change (Ellison 2015; Lamsal et al. 2017; Saintilan et al. 2019; Steen et al. 2017; Wardrop et al. 2019).

Wetland Sensitivity to climate change: drivers

As mentioned above, the two simulations used in this study represented different possible futures to account for uncertainty in climate projections; the lower-bound simulation had average water-level projections that were relatively similar to the projected recent past and the upper-bound simulation projected an increase in water levels (and both projected increased overall variability in water levels). We found that Great Lakes coastal wetlands were generally more sensitive in the upper-bound simulation, and that Sensitivity was primarily due to loss of wetland area. Although wetland area generally fluctuated over time, our results predict that there will be more years in the future with less wetland area relative to the past. We note that more years with less wetland area does not necessarily indicate wetland loss overall; however, an analysis of these data using percent mean-change from the hindcast showed that wetland area decreased by 17.6% across sites and that most sites (80%) were projected to decrease in area by 5 to 55% (ECCC 2022a). These results are consistent with other hydrodynamic studies that project wetland loss under climate change across the globe. In the Prairie Pothole Region of central North America, Sofaer et al. (2016) developed hydrologic projections forced by downscaled climate models and found that, on average, densities of wetlands are projected to decline across 10 GCMs and two emission scenarios (RCPs 4.5 and 8.5) due to drier conditions. Their results corroborate several studies predicting wetland loss as a result of climate change in that region (reviewed in Sofaer et al. 2016). In mountain wetlands in the northeastern United States, Lee et al. (2015) projected climate-induced hydrologic change under 10 GCMs and one emission scenario and found increased probability of drier conditions resulting in wetland loss. Along ocean coasts around the world, numerous climate change assessments predict that between 20% and 50% of coastal salt marshes will be lost due to climate-induced sea-level rise (reviewed in Kirwan et al. 2016a). Similarly, we found that the

probability of Great Lakes coastal wetland loss increased with increasing three-year-mean water levels across most lakes and simulations (Figure 15-17). The one exception was Lake Ontario in the upper-bound simulation (where the relationship was negative, but not statistically significant), which was likely driven by two sites (Airport Creek and Lynde Creek) where gentle slopes and surrounding land-use would allow these wetlands to migrate landward under considerably higher average lake levels. While inland wetlands are projected to be lost due to decreasing water levels and coastal wetlands are projected to be lost due to increasing water levels, the mechanism driving Wetland Sensitivity to climate change is the same – changes to hydrology. Taken together, the projected widespread loss of wetland area as a result of climate change appears to be due, in part, to climate-driven changes in water levels. This is not surprising given that climate ultimately drives water levels and variability, which have a significant influence on wetland structure (Keddy and Fraser 2000; reviewed above).

We also found that Wetland Sensitivity decreased with increasing variation in elevation. This implies that greater topographic diversity provides more locations for upward migration in response to increasing water levels, likely by providing more refugia for vegetation establishment and growth. This would include gently-rising slopes that span a large breadth in elevation. Conversely, a narrow topographic profile that is very shallow or very steep could increase Wetland Sensitivity because there would be limited opportunity for wetland migration.

Land use surrounding wetlands could also increase Wetland Sensitivity by restricting landward migration in response to increasing lake levels despite suitable topography. In the CWRM, it was assumed that specific land uses, including impervious surfaces, residential development, agriculture, and areas controlled by dikes, would be unsuitable for wetlands and restrict wetland migration into surrounding uplands (referred to as “masked areas”; ECCC 2022a). For example, Fox Creek on Lake Erie had the greatest proportion of masked area surrounding the site (0.65) and was among the most sensitive sites in the upper-bound simulation. Similarly, several sites (Detroit River, Johnston Bay, Lake St. Clair, and Selkirk Provincial Park) had high Wetland Sensitivity values in the upper-bound simulation likely because the land surrounding the wetland was classified as unsuitable for landward migration, resulting in “coastal squeeze” and wetland loss. These projections suggest that even a small area of land use adjacent to a wetland can have a large effect on Wetland Sensitivity by limiting landward migration. Therefore, it is likely that our inferences on Wetland Sensitivity are conservative because, with the exception of Fox Creek, we did not sample wetlands in the basin with more than 29% surrounding masked area.

The response of Great Lakes coastal wetlands to climate change could also depend on several other physical and biological processes that we were unable to consider in this assessment. Physical processes may include littoral sediment dynamics, riverine discharge, and nutrient and sediment loading. Of these, we expect that including littoral sediment dynamics in our assessment would have the largest effect on Wetland Sensitivity estimates. Since the delivery and deposition of sediment rebuilds substrate for vegetation growth, it is likely that natural littoral sediment transport would decrease Wetland Sensitivity by creating suitable downslope conditions for wetland migration and providing protection from storm surges and coastal erosion. However, there is strong evidence that littoral sediment transport to wetlands is significantly reduced by shoreline hardening and revetments constructed to protect coastal properties and harbours (Kirwan et al. 2016b). Most impacted in the Great Lakes are large sand-spit wetlands on Lake Erie (e.g., Point Pelee, Rondeau, and Long Point), which require littoral sediment transport from the western shoreline to maintain their physical integrity (Zuzek 2018, 2021). As shoreline alteration continues to increase across the Great Lakes, increasing sediment starvation and coastal erosion at existing wetlands could increase Wetland Sensitivity by exacerbating wetland loss.

We also were unable to assess the effect of climate change on biological processes that are temperature-controlled in wetlands. There is strong evidence from across the globe of climate change-driven increases in plant productivity (van der Wal and Stein 2014), shifting plant species composition (Liu et al. 2018), earlier phenology (e.g., spring flowering; Gonsamo et al. 2013), and poleward range shifts of plants (Harsch et al. 2009; Parmesan and Hanley 2015), all of which could influence Wetland Sensitivity. For example, Liu et al. (2018) found that warming increased grass biomass at the expense of sedges and forbs in alpine grasslands, which resulted in deeper root systems that may have allowed plant communities to acquire more water, thereby stabilizing primary production. In the Great Lakes, similar community shifts (resulting in deeper root systems) may also protect from coastal erosion by stabilizing sediment, which could decrease Wetland Sensitivity. However, it remains largely unclear how such temperature-driven changes will influence Wetland Sensitivity in the Great Lakes basin. This is a notable area for further research.

Alternatively, it is possible that invasive species could influence Wetland Sensitivity through their widespread impact on physical and biological processes in wetlands (reviewed in Bansal et al. 2019; Gallardo et al. 2016). For example, *Phragmites australis subsp. australis* grows in dense stands that displace native wetland vegetation, generally resulting in fewer

vegetation communities, decreased interspersion, and consequently lower biodiversity (Jung et al. 2017; Tulbure et al. 2007). Since ecosystem stability increases with increasing biodiversity (Craven et al. 2018), we would expect that greater areal extent of *Phragmites* would increase Wetland Sensitivity. However, dense *Phragmites* stands increase sediment retention in wetlands via greater litter accumulation and more extensive rhizome networks compared to cattail stands (Rooth et al. 2003, Bourgeau-Chavez et al. 2013). Over time, this sediment trapping results in greater vertical accretion rates in *Phragmites*-dominated wetlands (Rooth et al. 2003), which could decrease Wetland Sensitivity. Determining the net effect of *Phragmites* on Wetland Sensitivity requires additional research (ECCC 2022a).

Uncertainty in Wetland Sensitivity estimates

One of our key methodological decisions was how we calculated Wetland Sensitivity (as the proportion of years that key ecological attributes existed in an extreme negative state for each site and simulation). The Meteorological Service of Canada assessed climate change impacts on the same 20 coastal wetland sites based on percent mean-change from the hindcast and found that wetland area decreased by 17.6% across sites in the upper-bound simulation, and that most (80%) sites were projected to decrease in area by 5 to 55% (ECCC 2022a). These results are consistent with our findings that coastal wetlands will be negatively impacted by climate change due to wetland loss, and suggest that our approach for assessing Wetland Sensitivity was robust.

We found that wetlands were more sensitive in the upper-bound simulation than the lower-bound simulation. As a result, our inferences on Wetland Sensitivity, and the potential negative effects of climate change on Great Lakes coastal wetlands, may be biased toward wetter future scenarios (i.e., higher lake levels represented in the upper-bound simulation). Interestingly, these wetter future scenarios are possibly more aligned with the future climate of the Great Lakes basin. Wetland responses to climate change were assessed under RCP 4.5, a moderate emission scenario in which radiative forcing is stabilized at 4.5 watts per square metre by 2100 (Thomson et al. 2011); however, recent evidence suggests that the global emissions pathway is on a trajectory towards RCP 8.5, which is characterized by a significant increase in greenhouse gas emissions over time, leading to a radiative forcing of 8.5 watts per square metre by the end of the century (IPCC 2014; Schwalm et al. 2020). Although we did not assess Wetland Sensitivity to climate change under RCP 8.5 (ECCC 2022a), ECCC (2022c) projected water levels for all the Great Lakes across 10 GCMs under RCP 8.5, and predicted higher lake-

levels that are similar to or beyond the upper-bound simulation under RCP 4.5. Given the positive relationship between three-year-mean water level and total wetland area detected in our assessment, it is likely that many Great Lakes coastal wetlands would be more sensitive to climate change due to extensive wetland loss under RCP 8.5. Taken together, we suggest that our Wetland Sensitivity values are likely underestimated, or at the very least, the upper-bound estimate is more reliable for conservation management planning for Great Lakes coastal wetlands under future climate change.

Wildlife implications

As a result of climate change, we predict there will be more years in the future with less wetland area relative to the past, which would have significant implications for wildlife. Firstly, wetland loss represents a reduction in habitat for many wetland-dependent species (i.e., wildlife requiring wetlands for a least one part of their life cycle), several of which are undergoing population declines in the Great Lakes basin (e.g., marsh birds; Tozer 2016) or are listed as federal or provincial species at risk (e.g., least bittern, Blanding's turtle). Habitat loss generally results in lower population sizes and diversity of wildlife due to lower resource availability. If there is substantial loss of coastal wetland habitat across the Great Lakes in the future as a result of climate change, there would likely be significant wildlife population declines and regional biodiversity loss. Species-specific habitat suitability modelling within our assessment framework is an important avenue of future research to support wildlife conservation. Secondly, local habitat loss at key coastal wetlands could have disproportionate effects on regional wildlife populations. For example, the coastal wetlands of Long Point are internationally recognized as important stopover habitat for several eastern populations of migratory species, including waterfowl and particularly tundra swans (*Cygnus columbianus*; Badzinski et al. 2011; Knapton and Petrie 1999; Petrie et al. 2002), monarch butterflies (Crewe and McCracken 2015), bats, and forest birds (Taylor et al. 2011). Since Long Point is projected to lose up to 55% wetland area in some years (ECCC 2022a), it is possible that the eastern populations of all of these migratory species would be negatively affected. Lastly, habitat loss is generally assumed to decrease ecosystem functioning and services provided to humans (Brisson et al. 2014; Dobson et al. 2006). The monetary value of ecosystem services provided by coastal wetlands in southern Ontario is estimated to be \$750 million per year for carbon sequestration, nutrient regulation, recreation, and aesthetic value (Troy and Bagstad 2009). Decreased ecosystem functioning would also occur indirectly from wetland loss, as both biodiversity and habitat heterogeneity are generally positively associated with habitat area.

Conclusions

The goal of this study was to provide a science-based assessment of Wetland Sensitivity to climate change to inform mitigation strategies for Great Lakes coastal wetlands. We found that all coastal wetlands considered in this study were sensitive to future climate change and are likely at risk. Most notably, significant loss of wetland area was predicted to occur more frequently as a result of higher lake levels where surrounding land is unsuitable for migration due to land use (e.g., transportation infrastructure, residential development) or topography (e.g., steep slopes precluding migration). This means that further land development surrounding coastal wetlands will likely exacerbate climate-driven wetland loss. Our results suggest that conservation planning for Great Lakes coastal wetlands under climate change should consider protection of the surrounding landscape. More broadly, given that coastal wetlands across the Great Lakes are continuing to be lost and degraded due to land development, the synergistic effects of climate change and land development (both resulting in habitat loss) should be considered in regional biodiversity planning and recovery strategies for wetland species at risk.

References

- Alahuhta, J., Heino, J., Luoto, M., 2011. Climate change and the future distributions of aquatic macrophytes across boreal catchments. *Journal of Biogeography* 38, 383–393. <https://doi.org/10.1111/j.1365-2699.2010.02412.x>
- Allouche, O., Kalyuzhny, M., Moreno-Rueda, G., Pizarro, M., Kadmon, R., 2012. Area–heterogeneity tradeoff and the diversity of ecological communities. *Proceedings of the National Academy of Sciences* 109, 17495–17500. <https://doi.org/10.1073/pnas.1208652109>
- Alsterberg, C., Roger, F., Sundbäck, K., Juhanson, J., Hulth, S., Hallin, S., Gamfeldt, L., 2017. Habitat diversity and ecosystem multifunctionality—the importance of direct and indirect effects. *Science Advances* 3, e1601475. <https://doi.org/10.1126/sciadv.1601475>
- Askins, R.A., Philbrick, M.J., 1987. Effect of changes in regional forest abundance on the decline and recovery of a forest bird community. *The Wilson Bulletin* 99, 7–21. <http://www.jstor.org/stable/4162337>.
- Badzinski, S.S., Kennedy, L., Petrie, S.A., Schummer, M.L., 2011. Variation in body composition and digestive organs of Tundra Swans during migration at Long Point, Lake Erie, Ontario. *Waterbirds* 34, 468–475. <https://doi.org/10.1675/063.034.0409>
- Bannor, B.K., Kiviat, E., 2002. Common moorhen (*Gallinula chloropus*). In *The Birds of North America*, No. 685, Poole, A.F., Gill, F.B., eds., The Birds of North America, Inc., Philadelphia, PA.
- Bansal, S., Lishawa, S.C., Newman, S., Tangen, B.A., Wilcox, D., Albert, D., Anteau, M.J., Chimney, M.J., Cressey, R.L., DeKeyser, E., Elgersma, K.J., Finkelstein, S.A., Freeland, J., Grosshans, R., Klug, P.E., Larkin, D.J., Lawrence, B.A., Linz, G., Marburger, J., Noe, G., Otto, C., Reo, N., Richards, J., Richardson, C., Rodgers, L., Schrank, A.J., Svedarsky, D., Travis, S., Tuchman, N., Windham-Myers, L., 2019. *Typha* (cattail) invasion in North American wetlands: biology, regional problems, impacts, ecosystem services, and management. *Wetlands* 39, 645–684. <https://doi.org/10.1007/s13157-019-01174-7>

- Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. *Journal of Statistical Software* 67, 1–48. <https://doi.org/10.18637/jss.v067.i01>
- Ben-Hur, E., Kadmon, R., 2020. Heterogeneity–diversity relationships in sessile organisms: a unified framework. *Ecology letters* 23, 193–207. <https://doi.org/10.1111/ele.13418>
- Bourgeau-Chavez, L., Kowalski, K., Mazur, M., Scarbrough, K., Powell, R., Brooks, C., Huberty, B., Jenkins, L., Banda, E., Galbraith, D., Laubach, Z., Riordan, K., 2013. Mapping invasive *Phragmites australis* in the coastal Great Lakes with ALOS PALSAR satellite imagery for decision support. *Journal of Great Lakes Research* 39, 65–77. <https://doi.org/10.1016/j.jglr.2012.11.001>
- Bouvier, L.D., Cottenie, K., Doka, S.E., 2009. Aquatic connectivity and fish metacommunities in wetlands of the lower Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 66, 933–948. <https://doi.org/10.1139/F09-050>
- Brazner, J., Sierszen, M., Keough, J., Tanner, D.K., 2000. Assessing the ecological importance of coastal wetlands in a large lake context. *Internationale Vereinigung für theoretische und angewandte Limnologie: Verhandlungen*, 27, 1950–1961. <https://doi.org/10.1080/03680770.1998.11901583>
- Brazner, J.C., Tanner, D.K., Jensen, D.A., Lemke, A., 1998. Relative abundance and distribution of ruffe (*Gymnocephalus cernuus*) in a Lake Superior coastal wetland fish assemblage. *Journal of Great Lakes Research* 24, 293–303. [https://doi.org/10.1016/S0380-1330\(98\)70820-2](https://doi.org/10.1016/S0380-1330(98)70820-2)
- Brisbin, I.L., Mowbray, T.B., 2002. American coot (*Fulica americana*), version 2.0. In *The Birds of North America*, Poole, A.F., Gill, F.B., eds., Cornell Lab of Ornithology, Ithaca, NY.
- Brisson, C.P., Coverdale, T.C., Bertness, M.D., 2014. Salt marsh die-off and recovery reveal disparity between the recovery of ecosystem structure and service provision. *Biological Conservation* 179, 1–5. <https://doi.org/10.1016/j.biocon.2014.08.013>
- Burton, T.M., 1985. The effects of water level fluctuations on Great Lakes coastal marshes. In *Coastal Wetlands. Proceedings of the first Great Lakes Coastal Wetlands Colloquium*, Prince, H.H., D'Itri, F.M., eds., Lewis Publishers Inc., Chelsea, MI.

- Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., Naeem, S., 2012. Biodiversity loss and its impact on humanity. *Nature* 486, 59-67. <https://doi.org/10.1038/nature11148>
- Casselman, J.M., Lewis, C.A., 1996. Habitat requirements of northern pike (*Esox lucius*). *Canadian Journal of fisheries and aquatic sciences* 53, 161-174. <https://doi.org/10.1139/f96-019>
- Chin, A.T., Tozer, D.C., Fraser, G.S., 2014. Hydrology influences generalist–specialist bird-based indices of biotic integrity in Great Lakes coastal wetlands. *Journal of Great Lakes Research* 40, 281-287. <https://doi.org/10.1016/j.jglr.2014.02.006>
- Chocron, R., Flather, C.H., Kadmon, R., 2015. Bird diversity and environmental heterogeneity in North America: a test of the area–heterogeneity trade-off. *Global Ecology and Biogeography* 24, 1225-1235. <https://doi.org/10.1111/geb.12353>
- Chow-Fraser, P., 1998. A conceptual ecological model to aid restoration of Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario, Canada. *Wetlands Ecology and Management* 6, 43–57. <https://doi.org/10.1023/A:1008495604739>
- Cramer, M.J., Willig, M.R., 2005. Habitat heterogeneity, species diversity and null models. *Oikos* 108, 209-218. <https://doi.org/10.1111/j.0030-1299.2005.12944.x>
- Craven, D., Eisenhauer, N., Pearse, W.D., Hautier, Y., Isbell, F., Roscher, C., Bahn, M., Beierkuhnlein, C., Bönisch, G., Buchmann, N., Byun, C., Catford, J.A., Cerabolini, B.E.L., Cornelissen, J.H.C., Craine, J.M., De Luca, E., Ebeling, A., Griffin, J.N., Hector, A., Hines, J., Jentsch, A., Kattge, J., Kreyling, J., Lanta, V., Lemoine, N., Meyer, S.T., Minden, V., Onipchenko, V., Polley, H.W., Reich, P.B., van Ruijven, J., Schamp, B., Smith, M.D., Soudzilovskaia, N.A., Tilman, D., Weigelt, A., Wilsey, B., Manning, P., 2018. Multiple facets of biodiversity drive the diversity–stability relationship. *Nature Ecology and Evolution* 2, 1579–1587. <https://doi.org/10.1038/s41559-018-0647-7>
- Crewe, T.L., Mccracken, J.D., 2015. Long-term trends in the number of monarch butterflies (Lepidoptera: Nymphalidae) counted on fall migration at Long Point, Ontario, Canada (1995–2014). *Annals of the Entomological Society of America* 108, 707–717. <https://doi.org/10.1093/aesa/sav041>

- Culp, L.A., Cohen, E.B., Scarpignato, A.L., Thogmartin, W.E., Marra, P.P., 2017. Full annual cycle climate change vulnerability assessment for migratory birds. *Ecosphere* 8, e01565. <https://doi.org/10.1002/ecs2.1565>
- Cvetkovic, M., Wei, A., Chow-Fraser, P., 2010. Relative importance of macrophyte community versus water quality variables for predicting fish assemblages in coastal wetlands of the Laurentian Great Lakes. *Journal of Great Lakes Research* 36, 64–73. <https://doi.org/10.1016/j.jglr.2009.10.003>
- Darrah, A.J., Kremetz, D.G., 2009. Distribution and habitat use of King Rails in the Illinois and Upper Mississippi River Valleys. *The Journal of Wildlife Management* 73, 1380-1386. <https://doi.org/10.2193/2008-561>
- Desgranges, J.L., Ingram, J., Drolet, B., Morin, J., Savage, C., Borcard, D., 2006. Modelling wetland bird response to water level changes in the Lake Ontario – St. Lawrence River hydrosystem. *Environmental Monitoring and Assessment* 113, 329–365. <https://doi.org/10.1007/s10661-005-9087-3>
- Dobson, A., Lodge, D., Alder, J., Cumming, G.S., Keymer, J., McGlade, J., Mooney, H., Rusak, J.A., Sala, O., Wolters, V., Wall, D., Winfree, R., Xenopoulos, M.A., 2006. Habitat loss, trophic collapse, and the decline of ecosystem services. *Ecology* 87, 1915–1924. [https://doi.org/10.1890/0012-9658\(2006\)87\[1915:HLTCAT\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2006)87[1915:HLTCAT]2.0.CO;2)
- Duarte, C.M., Kalff, J., 1986. Littoral slope as a predictor of the maximum biomass of submerged macrophyte communities. *Limnology and Oceanography* 31, 1072–1080. <https://doi.org/10.4319/lo.1986.31.5.1072>
- Duffy, J.E., Godwin, C.M., Cardinale, B.J., 2017. Biodiversity effects in the wild are common and as strong as key drivers of productivity. *Nature* 549, 261–264. <https://doi.org/10.1038/nature23886>
- Duffy, J.E., Lefcheck, J.S., Stuart-Smith, R.D., Navarrete, S.A., Edgar, G.J., 2016. Biodiversity enhances reef fish biomass and resistance to climate change. *Proceedings of the National Academy of Sciences* 113, 6230–6235. <https://doi.org/10.1073/pnas.1524465113>
- Eadie, J.M., Keast, A., 1984. Resource heterogeneity and fish species diversity in lakes. *Canadian Journal of Zoology* 62, 1689–1695. <https://doi.org/10.1139/z84-248>

- Elliott, L.H., Igl, L.D., Johnson, D.H., 2020. The relative importance of wetland area versus habitat heterogeneity for promoting species richness and abundance of wetland birds in the Prairie Pothole Region, USA. *The Condor* 122, duz060.
<https://doi.org/10.1093/condor/duz060>
- Ellison, J.C., 2014. Vulnerability of mangroves to climate change, in *Mangrove ecosystems of Asia*, Faridah-Hanum, I., Latiff, A., Hakeem, K.R., Ozturk, M., eds., Springer, New York, NY. https://doi.org/10.1007/978-1-4614-8582-7_10
- Ellison, J.C., 2015. Vulnerability assessment of mangroves to climate change and sea-level rise impacts. *Wetlands Ecology and Management* 23, 115–137. <https://doi.org/10.1007/s11273-014-9397-8>
- Engel, S., 1988. The role and interactions of submersed macrophytes in a shallow Wisconsin lake. *Journal of Freshwater Ecology* 4, 329–341.
<https://doi.org/10.1080/02705060.1988.9665182>
- Environment and Climate Change Canada. 2022a. Great Lakes coastal wetland response to climate change using a Coastal Wetland Response Model (CWRM). Caroline Sevigny, Dominic Thériault, Antoine Maranda, Rémi Gosselin, Mathieu Roy, Sandrine Hogue-Hugron, Nicolas Fortin, Marianne Bachand and Jean Morin. Scientific Report ST-100, Environment and Climate Change Canada, Meteorological Service of Canada, Hydrodynamic and Ecohydraulic Section, Quebec City. 523p.
- Environment and Climate Change Canada. 2022b. Assessing and Enhancing the Resilience of Great Lakes Coastal Wetlands: Adaptive Capacity to Climate Change. Hrynyk, M., Quesnelle, P., Rivers, P., Duffe, J., Grabas, G., Mayne, G. 80p.
- Environment and Climate Change Canada. 2022c. Future hydroclimate variables and lake levels for the Great Lakes using data from the Coupled Model Intercomparison Project Phase 5. Seglenieks, F. and Temgoua, A. Environment and Climate Change Canada. 58p.
- Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution and Systematics* 34, 487–515.
<https://doi.org/10.1146/annurev.ecolsys.34.011802.132419>

- Fairbairn, S.E., Dinsmore, J.J., 2001. Local and landscape-Level influences on wetland bird communities of the prairie pothole region of Iowa, USA. *Wetlands* 21, 41–47. [https://doi.org/10.1672/0277-5212\(2001\)021\[0041:LALLIO\]2.0.CO;2](https://doi.org/10.1672/0277-5212(2001)021[0041:LALLIO]2.0.CO;2)
- Fracz, A., Chow-Fraser, P., 2013. Impacts of declining water levels on the quantity of fish habitat in coastal wetlands of eastern Georgian Bay, Lake Huron. *Hydrobiologia* 702, 151–169. <https://doi.org/10.1007/s10750-012-1318-3>
- Gallardo, B., Clavero, M., Sánchez, M.I., Vilà, M., 2016. Global ecological impacts of invasive species in aquatic ecosystems. *Global Change Biology* 22, 151–163. <https://doi.org/10.1111/gcb.13004>
- Gibbons, P., Freudenberger, D., 2006. An overview of methods used to assess vegetation condition at the scale of the site. *Ecological Management and Restoration* 7, S10–S17. <https://doi.org/10.1111/j.1442-8903.2006.00286.x>
- Glick, P., Stein, B.A., Edelson, N.A., 2011. Scanning the conservation horizon: a guide to climate change vulnerability assessment. Washington, DC: National Wildlife Federation. 168 p.
- Gonsamo, A., Chen, J.M., Wu, C., 2013. Citizen Science: linking the recent rapid advances of plant flowering in Canada with climate variability. *Scientific Reports* 3, 2239. <https://doi.org/10.1038/srep02239>
- González-Gajardo, A., Sepúlveda, P.V., Schlatter, R., 2009. Waterbird assemblages and habitat characteristics in wetlands: influence of temporal variability on species-habitat relationships. *Waterbirds* 32, 225–233. <https://doi.org/10.1675/063.032.0203>
- Grabas, G.P., Blukacz-Richards, E.A., Pernanen, S., 2012. Development of a submerged aquatic vegetation community index of biotic integrity for use in Lake Ontario coastal wetlands. *Journal of Great Lakes Research* 38, 243–250. <https://doi.org/10.1016/j.jglr.2012.02.014>
- Grabas, G.P., Fiorino, G.E., Reinert, A., 2019. Vegetation species richness is associated with daily water-level fluctuations in Lake Ontario coastal wetlands. *Journal of Great Lakes Research* 45, 805–810. <https://doi.org/10.1016/j.jglr.2019.05.008>

- Grabas, P., Rokitnicki-Wojcik, D., 2015. Characterizing daily water-level fluctuation intensity and water quality relationships with plant communities in Lake Ontario coastal wetlands. *Journal of Great Lakes Research* 41, 136–144. <https://doi.org/10.1016/j.jglr.2014.12.019>
- Guadagnin, D.L., Maltchik, L., 2007. Habitat and landscape factors associated with neotropical waterbird occurrence and richness in wetland fragments. *Biodiversity and Conservation* 16, 1231–1244. <https://doi.org/10.1007/s10531-006-9127-5>
- Harsch, M.A., Hulme, P.E., McGlone, M.S., Duncan, R.P., 2009. Are treelines advancing? A global meta-analysis of treeline response to climate warming. *Ecology Letters* 12, 1040–1049. <https://doi.org/10.1111/j.1461-0248.2009.01355.x>
- Hartmann, H.C., 1990. Climate change impacts on Laurentian Great Lakes levels. *Climatic Change* 17, 49–67. <https://doi.org/10.1007/BF00149000>
- Hartwig, T.S., Kiviat, E., 2007. Microhabitat association of Blanding's turtles in natural and constructed wetlands in southeastern New York. *The Journal of Wildlife Management* 71, 576–582. <https://doi.org/10.2193/2005-619>
- Hesselbarth, M., Sciaini, M., With, K.A., Wiegand, K., Nowosad, J., 2019. landscapemetrics: an open-source R tool to calculate landscape metrics. *Ecography* 42, 1648–1657. <https://doi.org/10.1111/ecog.04617>
- Hohman, T.R., Howe, R.W., Tozer, D.C., Giese, E.E.G., Wolf, A.T., Niemi, G.J., Gehring, T.M., Grabas, G.P., Norment, C.J., 2021. Influence of lake levels on water extent, interspersion, and marsh birds in Great Lakes coastal wetlands. *Journal of Great Lakes Research* 47, 534–545. <https://doi.org/10.1016/j.jglr.2021.01.006>
- Hudon, C., Wilcox, D., Ingram, J., 2006. Modeling wetland plant community response to assess water-level regulation scenarios in the Lake Ontario – St. Lawrence River basin. *Environmental Monitoring and Assessment* 113, 303–328. <https://doi.org/10.1007/s10661-005-9086-4>
- IPCC, 2007. *Climate Change 2007: Impacts, Adaptation and Vulnerability*. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Parry M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J., Hanson, C.E., eds., Cambridge University Press, Cambridge, United Kingdom, 976 pp.

- IPCC, 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Core Writing Team, Pachauri, R.K., Meyer, L.A., eds., IPCC, Geneva, Switzerland, 151 pp.
- Isbell, F., Craven, D., Connolly, J., Loreau, M., Schmid, B., Beierkuhnlein, C., Bezemer, T.M., Bonin, C., Bruelheide, H., De Luca, E., Ebeling, A., Griffin, J.N., Guo, Q., Hautier, Y., Hector, A., Jentsch, A., Kreyling, J., Lanta, V., Manning, P., Meyer, S.T., Mori, A.S., Naeem, S., Niklaus, P.A., Polley, H.W., Reich, P.B., Roscher, C., Seabloom, E.W., Smith, M.D., Thakur, M.P., Tilman, D., Tracy, B.F., van der Putten, W.H., van Ruijven, J., Weigelt, A., Weisser, W.W., Wilsey, B., Eisenhauer, N., 2015. Biodiversity increases the resistance of ecosystem productivity to climate extremes. *Nature* 526, 574–577.
<https://doi.org/10.1038/nature15374>
- Jacobus, J., Ivan, L.N., 2005. Evaluating the effects of habitat patchiness on small fish assemblages in a Great Lakes coastal marsh. *Journal of Great Lakes Research* 31, 466–481. [https://doi.org/10.1016/S0380-1330\(05\)70277-X](https://doi.org/10.1016/S0380-1330(05)70277-X)
- Johnson, B., Johnston, C.A., 1995. Relationship of lithology and geomorphology to erosion of the western Lake Superior coast. *Journal of Great Lakes Research* 21, 3–16.
[https://doi.org/10.1016/S0380-1330\(95\)71016-4](https://doi.org/10.1016/S0380-1330(95)71016-4)
- Jude, D.J., Pappas, J., 1992. Fish utilization of Great Lakes coastal wetlands. *Journal of Great Lakes Research* 18, 651–672. [https://doi.org/10.1016/S0380-1330\(92\)71328-8](https://doi.org/10.1016/S0380-1330(92)71328-8)
- Jung, J.A., Rokitnicki-Wojcik, D., Midwood, J.D., 2017. Characterizing past and modelling future spread of *Phragmites australis ssp. australis* at Long Point Peninsula, Ontario, Canada. *Wetlands* 37, 961–973. <https://doi.org/10.1007/s13157-017-0931-3>
- Keddy, P., Fraser, L.H., 2000. Four general principles for the management and conservation of wetlands in large lakes: The role of water levels, nutrients, competitive hierarchies and centrifugal organization. *Lakes Reservoirs: Science, Policy and Management for Sustainable* 5, 177–185. <https://doi.org/10.1046/j.1440-1770.2000.00111.x>
- Keddy, P.A., Reznicek, A.A., 1986. Great Lakes vegetation dynamics: The role of fluctuating water levels and buried seeds. *Journal of Great Lakes Research* 12, 25–36.
[https://doi.org/10.1016/S0380-1330\(86\)71697-3](https://doi.org/10.1016/S0380-1330(86)71697-3)

- Keough, J.R., Thompson, T.A., Guntenspergen, G.R., Wilcox, D.A., 1999. Hydrogeomorphic factors and ecosystem responses in coastal wetlands of the Great Lakes. *Wetlands* 19, 821–834. <https://doi.org/10.1007/BF03161786>
- Kingsford, R.T., Basset, A., Jackson, L., 2016. Wetlands: conservation's poor cousins. *Aquatic Conservation: Marine and Freshwater Ecosystems* 26, 892–916. <https://doi.org/10.1002/aqc.2709>
- Kirwan, M.L., Temmerman, S., Skeeahan, E.E., Guntenspergen, G.R., Fagherazzi, S., 2016a. Overestimation of marsh vulnerability to sea level rise. *Nature Climate Change* 6, 253–260. <https://doi.org/10.1038/nclimate2909>
- Kirwan, M.L., Walters, D.C., Reay, W.G., Carr, J.A., 2016b. Sea level driven marsh expansion in a coupled model of marsh erosion and migration. *Geophysical Research Letters* 43, 4366–4373. <https://doi.org/10.1002/2016GL068507>
- Knapton, R.W., Petrie, S.A., 1999. Changes in distribution and abundance of submerged macrophytes in the Inner Bay at Long Point, Lake Erie: Implications for foraging waterfowl. *Journal of Great Lakes Research* 25, 783–798. [https://doi.org/10.1016/S0380-1330\(99\)70777-X](https://doi.org/10.1016/S0380-1330(99)70777-X)
- Krieger, K.A., Klarer, D.M., Heath, R.T., Herdendorf, C.E., 1992. Coastal wetlands of the Laurentian Great Lakes: current knowledge and research needs. Preface: a call for research on Great Lakes coastal wetlands. *Journal of Great Lakes Research* 18, 525–528.
- Lam, S., Dokoska, K., in preparation. Climate change in the Great Lakes basin: summary of trends and impacts. Ontario Climate Consortium, Toronto, ON.
- Lamsal, P., Kumar, L., Atreya, K., Pant, K.P., 2017. Vulnerability and impacts of climate change on forest and freshwater wetland ecosystems in Nepal: A review. *Ambio* 46, 915–930. <https://doi.org/10.1007/s13280-017-0923-9>
- Leblanc, J.P., Weller, J.D., Chow-Fraser, P., 2014. Thirty-year update: Changes in biological characteristics of degraded muskellunge nursery habitat in southern Georgian Bay, Lake Huron, Canada. *Journal of Great Lakes Research* 40, 870–878. <https://doi.org/10.1016/j.jglr.2014.08.006>

- Lee, H., Bakowsky, W., Riley, J., Bowles, J., Puddister, M., Uhlig, P., McMurray, S., 1998. Ecological land classification for southern Ontario: first approximation and its application. Ontario Ministry of Natural Resources, Southcentral Science Section, Science Development and Transfer Branch. SCSS Field Guide FG-02. 225 pp.
- Lee, S.-Y., Ryan, M.E., Hamlet, A.F., Palen, W.J., Lawler, J.J., Halabisky, M., 2015. Projecting the hydrologic impacts of climate change on montane wetlands. PLOS ONE 10, e0136385. <https://doi.org/10.1371/journal.pone.0136385>
- LimnoTech, 2019. Summary report for: Assessing and enhancing the resilience of Great Lakes coastal wetlands – experts meeting. Report to Environment and Climate Change Canada. 35 pp.
- Liu, H., Mi, Z., Lin, L., Wang, Y., Zhang, Z., Zhang, F., Wang, H., Liu, L., Zhu, B., Cao, G., Zhao, X., Sanders, N.J., Classen, A.T., Reich, P.B., He, J.-S., 2018. Shifting plant species composition in response to climate change stabilizes grassland primary production. Proceedings of the National Academy of Sciences 115, 4051–4056. <https://doi.org/10.1073/pnas.1700299114>
- Lorenzón, R.E., Beltzer, A.H., Olguin, P.F., Ronchi-Virgolini, A.L., 2016. Habitat heterogeneity drives bird species richness, nestedness and habitat selection by individual species in fluvial wetlands of the Paraná River, Argentina. Austral Ecology 41, 829–841. <https://doi.org/10.1111/aec.12375>
- Magurran, A. E., 2004. Measuring biological diversity, 2nd ed. Blackwell Science Ltd, Oxford, U.K.
- Markle, C.E., Rutledge, J.M., Chow-Fraser, P., 2018. Factors affecting coastal wetland occupancy for eastern musk turtles (*Sternotherus odoratus*) in Georgian Bay, Lake Huron. Herpetologica 74, 236–244. <https://doi.org/10.1655/Herpetologica-D-18-00002>
- Maynard, L., Wilcox, D.A., 1997. Coastal Wetlands. Background paper for State of the Lake Ecosystem Conference 1996. Environment Canada, Guelph, ON, and United States Great Lakes Science Center National Biological Service, Ann Arbor, MI.
- McGarigal, K., Cushman, S.A., Ene, E., 2012. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps. Computer software program produced by

the authors at the University of Massachusetts, Amherst.
<http://www.umass.edu/landeco/research/fragstats/fragstats.html>

- McNair, S.A., Chow-Fraser, P., 2003. Change in biomass of benthic and planktonic algae along a disturbance gradient for 24 Great Lakes coastal wetlands. *Canadian Journal of Fisheries and Aquatic Sciences* 60, 676–689. <https://doi.org/10.1139/f03-054>
- Meyer, S.W., Badzinski, S.S., Petrie, S.A., Ankney, C.D., 2010. Seasonal abundance and species richness of birds in common reed habitats in Lake Erie. *The Journal of Wildlife Management* 74, 1559–1566. <https://doi.org/10.1111/j.1937-2817.2010.tb01284.x>
- Midwood, J.D., Chow-Fraser, P., 2012. Changes in aquatic vegetation and fish communities following 5 years of sustained low water levels in coastal marshes of eastern Georgian Bay, Lake Huron. *Global Change Biology* 18, 93–105. <https://doi.org/10.1111/j.1365-2486.2011.02558.x>
- Millar, C.S., Blouin-Demers, G., 2011. Spatial ecology and seasonal activity of Blanding's turtles (*Emydoidea blandingii*) in Ontario, Canada. *Journal of Herpetology* 45, 370–378. <https://doi.org/10.1670/10-172.1>
- Montgomery, F., Reid, S.M., Mandrak, N.E., 2020. Extinction debt of fishes in Great Lakes coastal wetlands. *Biological Conservation* 241, 108386. <https://doi.org/10.1016/j.biocon.2019.108386>
- Morales-Marin, L.A., Rokaya, P., Sanyal, P.R., Sereda, J., Lindenschmidt, K.E., 2019. Changes in streamflow and water temperature affect fish habitat in the Athabasca River basin in the context of climate change. *Ecological Modelling* 407, 108718. <https://doi.org/10.1016/j.ecolmodel.2019.108718>
- Mortsch, L., 1998. Assessing the impact of climate change on the Great Lakes shoreline wetlands. *Climatic Change* 40, 391–416. <https://doi.org/10.1023/A:1005445709728>
- Mortsch, L., Ingram, J., Hebb, A., Doka, S., 2006. Great Lakes coastal wetland communities: vulnerabilities to climate change and response to adaptation strategies. Final report submitted to the Climate Change Impacts and Adaptation Program, Natural Resources Canada. Environment Canada and the Department of Fisheries and Oceans, Toronto, ON.

- Moss, R.H., Babiker, M., Brinkman, S., Calvo, E., Carter, T., Edmonds, J.A., Elgizouli, I., Emori, S., Erda, L., Hibbard, K., Jones, R., Kainuma, M., Kelleher, J., Lamarque, J.F., Manning, M., Matthews, B., Meehl, J., Meyer, L., Mitchell, J., Nakicenovic, N., O'Neill, B., Pichs, R., Riahi, K., Rose, S., Runci, P.J., Stouffer, R., VanVuuren, D., Weyant, J., Wilbanks, T., van Ypersele, J.P., Zurek, M., 2008. Towards new scenarios for analysis of emissions, climate change, impacts, and response strategies. IPCC Expert Meeting Report on New Scenarios, Intergovernmental Panel on Climate Change. Pacific Northwest National Lab, Richland, WA.
- Mowbray, T.B., 1997. Swamp sparrow (*Melospiza georgiana*). In *The Birds of North America*, No. 279, Poole, A., Gill, F., eds., The Academy of Natural Sciences, Washington, DC.
- Murkin, H.R., Kaminski, R.M., Titman, R.D., 1982. Responses by dabbling ducks and aquatic invertebrates to an experimentally manipulated cattail marsh. *Canadian Journal of Zoology* 60, 2324–2332. <https://doi.org/10.1139/z82-299>
- Ohrel, H., Register, K., 2006. Chapter 18: Submerged Aquatic Vegetation, in *Volunteer Estuary Monitoring Manual, a Methods Manual*, Second Edition. The Ocean Conservatory, Washington, DC, and United States Environmental Protection Agency, Washington, DC.
- Oldham, M.J., Bakowsky, W.D., Sutherland, D.A., 1995. Floristic quality assessment for southern Ontario. OMNR, Natural Heritage Information Centre, Peterborough. 68 pp.
- Ontario Ministry of Natural Resources, 2013. Ontario wetland evaluation system: southern manual, 3rd edition, Version 3.2. Ontario Ministry of Natural Resources, Toronto, ON.
- Parmesan, C., Hanley, M.E., 2015. Plants and climate change: complexities and surprises. *Annals of Botany* 116, 849–864. <https://doi.org/10.1093/aob/mcv169>
- Patton, D.R., Judd, B.I., 1970. The role of wet meadows as wildlife habitat in the Southwest. *Journal of Range Management* 23, 272–275. <https://doi.org/10.2307/3896220>
- Petrie, S.A., Badzinski, S.S., Wilcox, K.L., 2002. Population trends and habitat use of Tundra Swans staging at Long Point, Lake Erie. *Waterbirds* 25, 143–149.

- Proulx, G., Gilbert, F. F., 1983. The ecology of the muskrat, *Ondatra zibethicus*, at Luther Marsh, Ontario. *Canadian Field-Naturalist* 97, 377–390.
- R Core Team, 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>.
- Rehm, E.M., Baldassarre, G.A., 2007. The influence of interspersion on marsh bird abundance in New York. *The Wilson Journal of Ornithology* 119, 648–654. <https://doi.org/10.1676/06-060.1>
- Riffell, S.K., Keas, B.E., Burton, T.M., 2001. Area and habitat relationships of birds in Great Lakes coastal wet meadows. *Wetlands* 21, 492–507.
- Rohwer, F. C., Johnson, W.P., Loos, E. R., 2002. Blue-winged teal (*Anas discors*). In *The Birds of North America*, No. 625, Poole, A.F., Gill, F.B., eds., *The Birds of North America*, Inc., Philadelphia, PA.
- Rokitnicki-Wojcik, D., Grabas, G., Brett, J., 2014. An association between marsh-nesting obligate bird species and submergent vegetation in lower Great Lakes coastal wetlands. *Ontario Birds* 32, 27–40.
- Rooth, J., Stevenson, J., Cornwell, J., 2003. Increased sediment accretion rates following invasion by *Phragmites australis*: The role of litter. *Estuaries* 26, 475–483. <https://doi.org/10.1007/BF02823724>
- Rosenzweig, M.L., 1999. Heeding the Warning in biodiversity's basic law. *Science* 284, 276–277. <https://doi.org/10.1126/science.284.5412.276>
- Saintilan, N., Rogers, K., Kelleway, J.J., Ens, E., Sloane, D.R., 2019. Climate change impacts on the coastal wetlands of Australia. *Wetlands* 39, 1145–1154. <https://doi.org/10.1007/s13157-018-1016-7>
- Schummer, M.L., Palframan, J., McNaughton, E., Barney, T., Petrie, S.A., 2012. Comparisons of bird, aquatic macroinvertebrate, and plant communities among dredged ponds and natural wetland habitats at Long Point, Lake Erie, Ontario. *Wetlands* 32, 945–953. <https://doi.org/10.1007/s13157-012-0328-2>

- Schwalm, C.R., Glendon, S., Duffy, P.B., 2020. RCP8.5 tracks cumulative CO2 emissions. *Proceedings of the National Academy of Sciences* 117, 19656–19657.
<https://doi.org/10.1073/pnas.2007117117>
- Sierszen, M.E., Morrice, J.A., Trebitz, A.S., Hoffman, J.C., 2012. A review of selected ecosystem services provided by coastal wetlands of the Laurentian Great Lakes. *Aquatic Ecosystem Health and Management* 15, 92–106.
<https://doi.org/10.1080/14634988.2011.624970>
- Smith, I.M., Fiorino, G.E., Grabas, G.P., Wilcox, D.A., 2021. Wetland vegetation response to record-high Lake Ontario water levels. *Journal of Great Lakes Research* 47, 160–167.
<https://doi.org/10.1016/j.jglr.2020.10.013>
- Smith, L.A., Chow-Fraser, P., 2010. Implications of the species-area relationship on sampling effort for marsh birds in southern Ontario. *Wetlands* 30, 553–563.
<https://doi.org/10.1007/s13157-010-0048-4>
- Sofaer, H.R., Skagen, S.K., Barsugli, J.J., Rashford, B.S., Reese, G.C., Hoeting, J.A., Wood, A.W., Noon, B.R., 2016. Projected wetland densities under climate change: habitat loss but little geographic shift in conservation strategy. *Ecological Applications* 26, 1677–1692.
<https://doi.org/10.1890/15-0750.1>
- SOLEC, 2007. State of the Lake Ecosystem Conference final report. Environment Canada and United States Environmental Protection Agency. ISBN 978-0-662-47328-2.
- Steen, V., Sofaer, H.R., Skagen, S.K., Ray, A.J., Noon, B.R., 2017. Projecting species' vulnerability to climate change: Which uncertainty sources matter most and extrapolate best? *Ecology and Evolution* 7, 8841–8851. <https://doi.org/10.1002/ece3.3403>
- Stein, A., Gerstner, K., Kreft, H., 2014. Environmental heterogeneity as a universal driver of species richness across taxa, biomes and spatial scales. *Ecology letters* 17, 866–880.
<https://doi.org/10.1111/ele.12277>
- Steinman, A.D., Ogdahl, M.E., Weinert, M., Thompson, K., Cooper, M.J., Uzarski, D.G., 2012. Water level fluctuation and sediment–water nutrient exchange in Great Lakes coastal wetlands. *Journal of Great Lakes Research* 38, 766–775.
<https://doi.org/10.1016/j.jglr.2012.09.020>

- Stephenson, T.D., 1990. Fish reproductive utilization of coastal marshes of Lake Ontario near Toronto. *Journal of Great Lakes Research* 16, 71–81. [https://doi.org/10.1016/S0380-1330\(90\)71399-8](https://doi.org/10.1016/S0380-1330(90)71399-8)
- Stuckey, R.L., 1989. Western Lake Erie aquatic and wetland vascular-plant flora: its origin and change. In *Lake Erie estuarine systems: Issues, resources, status, and management*, Krieger, K.A., ed., U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Estuarine Programs Office, Washington, DC, pp. 205–256.
- Taylor, P.D., Mackenzie, S.A., Thurber, B.G., Calvert, A.M., Mills, A.M., McGuire, L.P., Guglielmo, C.G., 2011. Landscape movements of migratory birds and bats reveal an expanded scale of stopover. *PLOS ONE* 6, e27054. <https://doi.org/10.1371/journal.pone.0027054>
- Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M.C., Schwager, M., Jeltsch, F., 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. *Journal of Biogeography* 31, 79–92. <https://doi.org/10.1046/j.0305-0270.2003.00994.x>
- Thomson, A.M., Calvin, K.V., Smith, S.J., Kyle, G.P., Volke, A., Patel, P., Delgado-Arias, S., Bond-Lamberty, B., Wise, M.A., Clarke, L.E., Edmonds, J.A., 2011. RCP4.5: a pathway for stabilization of radiative forcing by 2100. *Climatic Change*, 109, 77–94. <https://doi.org/10.1007/s10584-011-0151-4>
- Tozer, D.C., 2016. Marsh bird occupancy dynamics, trends, and conservation in the southern Great Lakes basin: 1996 to 2013. *Journal of Great Lakes Research* 42, 136–145. <https://doi.org/10.1016/j.jglr.2015.10.015>
- Trebitz, A.S., Brazner, J.C., Danz, N.P., Pearson, M.S., Peterson, G.S., Tanner, D.K., Taylor, D.L., West, C.W., Hollenhorst, T.P., 2009. Geographic, anthropogenic, and habitat influences on Great Lakes coastal wetland fish assemblages. *Canadian Journal of Fisheries and Aquatic Sciences* 66, 1328–1342. <https://doi.org/10.1139/F09-089>
- Troy, A., Bagstad, K., 2009. Estimating ecosystem services in southern Ontario. Prepared for the Ontario Ministry of Natural Resources. Spatial Informatics Group, LLC, Pleasanton, CA.

- Tulbure, M.G., Johnston, C.A., Auger, D.L., 2007. Rapid invasion of a Great Lakes coastal wetland by non-native *Phragmites australis* and *Typha*. *Journal of Great Lakes Research* 33, 269–279. [https://doi.org/10.3394/0380-1330\(2007\)33\[269:RIOAGL\]2.0.CO;2](https://doi.org/10.3394/0380-1330(2007)33[269:RIOAGL]2.0.CO;2)
- Ugupta, S., Sharma, J., Jayaraman, M., Kumar, V., Ravindranath, N.H., 2015. Climate change impact and vulnerability assessment of forests in the Indian Western Himalayan region: A case study of Himachal Pradesh, India. *Climate Risk Management* 10, 63–76. <https://doi.org/10.1016/j.crm.2015.08.002>
- van Der Wal, R., Stien, A., 2014. High-arctic plants like it hot: a long-term investigation of between-year variability in plant biomass. *Ecology* 95, 3414–3427. <https://doi.org/10.1890/14-0533.1>
- Van Rossum, G., Drake, F.L., 2009. *Python 3 Reference Manual*. CreateSpace, Scotts Valley, CA.
- Van Wynsberge, S., Gilbert, A., Guillemot, N., Payri, C., Andréfouët, S., 2013. Alert thresholds for monitoring environmental variables: a new approach applied to seagrass beds diversity in New Caledonia. *Marine Pollution Bulletin* 77, 300–307. <https://doi.org/10.1016/j.marpolbul.2013.09.035>
- Wabnitz, C.C.C., Lam, V.W.Y., Reygondeau, G., Teh, L.C.L., Al-Abdulrazzak, D., Khalfallah, M., Pauly, D., Palomares, M.L.D., Zeller, D., Cheung, W.W.L., 2018. Climate change impacts on marine biodiversity, fisheries and society in the Arabian Gulf. *PLOS ONE* 13, e0194537. <https://doi.org/10.1371/journal.pone.0194537>
- Wardrop, D.H., Hamilton, A.T., Nassry, M.Q., West, J.M., Britson, A.J., 2019. Assessing the relative vulnerabilities of Mid-Atlantic freshwater wetlands to projected hydrologic changes. *Ecosphere* 10, e02561. <https://doi.org/10.1002/ecs2.2561>
- Wei, A., Chow-Fraser, P., Albert, D., 2004. Influence of shoreline features on fish distribution in the Laurentian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 61, 1113–1123. <https://doi.org/10.1139/f04-061>
- Weller, J.D., Chow-Fraser, P., 2019. Simulated changes in extent of Georgian Bay low-marsh habitat under multiple lake levels. *Wetlands Ecology and Management* 27, 483–495. <https://doi.org/10.1007/s11273-019-09673-4>

- Weller, M.W., Spatcher, C.S., 1965. Role of habitat in the distribution and abundance of marsh birds. No. 43. Agricultural and Home Economic Experiment Station, Iowa State University of Science and Technology, Ames, IA. <http://publications.iowa.gov/id/eprint/21788>
- Wilcox, D.A., 2004. Implications of hydrologic variability on the succession of plants in Great Lakes wetlands. *Aquatic Ecosystem Health and Management* 7, 223–231. <https://doi.org/10.1080/14634980490461579>
- Wilcox, D.A., Bateman, J.A., 2018. Photointerpretation analysis of plant communities in Lake Ontario wetlands following 65 years of lake-level regulation. *Journal of Great Lakes Research* 44, 1306– 1313. <https://doi.org/10.1016/j.jglr.2018.08.007>
- Wilcox, D.A., Ingram, J.W., Kowalski, K.P., Meeker, J.E., Carlson, M.L., Xie, Y., Grabas, G.P., Holmes, K.L., Patterson, N.J., 2005. Evaluation of water level regulation influences on Lake Ontario and upper St. Lawrence River coastal wetland plant communities. Final Project Report.
- Wilcox, D.A., Kowalski, K.P., Hoare, H.L., Carlson, M.L., Morgan, H.N., 2008. Cattail invasion of sedge/grass meadows in Lake Ontario: photointerpretation analysis of sixteen wetlands over five decades. *Journal of Great Lakes Research* 34, 301–323. [https://doi.org/10.3394/0380-1330\(2008\)34\[301:CIOGMI\]2.0.CO;2](https://doi.org/10.3394/0380-1330(2008)34[301:CIOGMI]2.0.CO;2)
- Wilcox, D.A., Meeker, J.E., 1992. Implications for faunal habitat related to a altered macrophyte structure in regulated lakes in northern Minnesota. *Wetlands* 12, 192–203. <https://doi.org/10.1007/BF03160609>
- Wilcox, D.A., Meeker, J.E., Elias, J., 1993. Impacts of water-level regulation on wetlands of the Great Lakes. Phase 2 Report to Working Committee 2, International Joint Commission, Great Lakes Water Levels Reference Study.
- Wilcox, D.A., Nichols, S.J., 2008. The effects of water-level fluctuations on vegetation in a Lake Huron wetland. *Wetlands* 28, 487–501. <https://doi.org/10.1672/07-129.1>
- Zavaleta, E.S., Pasari, J.R., Hulvey, K.B., Tilman, G.D., 2010. Sustaining multiple ecosystem functions in grassland communities requires higher biodiversity. *Proceedings of the National Academy of Sciences* 107, 1443–1446. <https://doi.org/10.1073/pnas.0906829107>

Zedler, J.B., 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. *Frontiers in Ecology and the Environment* 1, 65–72. [https://doi.org/10.1890/1540-9295\(2003\)001\[0065:WAYSRI\]2.0.CO;2](https://doi.org/10.1890/1540-9295(2003)001[0065:WAYSRI]2.0.CO;2)

Zedler, J.B., Kercher, S., 2005. Wetland resources: status, trends, ecosystem services, and restorability. *Annual Review of Environment and Resources* 30, 39–74. <https://doi.org/10.1146/annurev.energy.30.050504.144248>

Zuzek, P.J., 2018. Chatham-Kent Lake Erie shoreline study. Prepared for The Municipality of Chatham-Kent. Zuzek Inc., Waterdown, ON.

Zuzek, P.J., 2021. Recommendations for Long-term Conservation of Barrier Protected Coastal Wetlands. Zuzek Inc., Waterdown, ON.