



Contents lists available at ScienceDirect

Journal of Great Lakes Research

journal homepage: www.elsevier.com/locate/jglr

Hydrologic connections in Apostle islands coastal wetlands: Exploring dynamics and resilience to Lake Superior water level fluctuations

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ARTICLE INFO

Communicated by Anett Trebitz

Keywords:

Bathymetry
Climate change
Freshwater estuaries
Coastal lagoons
Interdunal wetlands
Vulnerability assessment

ABSTRACT

Coastal wetlands within Apostle Islands National Lakeshore (APIS) represent important reference ecosystems for Lake Superior given their relatively undegraded condition. However, this region lacks climate change-relevant nearshore bathymetric information for coastal wetlands, a gap our study helps fill. Since 2010, the Great Lakes, including Lake Superior, have experienced both extended low water levels and extreme highs. Such extremes are anticipated to increase in frequency and duration, but the exact outcomes are uncertain. Therefore, it is important to explore the resiliency of APIS wetlands against different water level scenarios. We studied seven wetlands with different geomorphic connections to Lake Superior (open, transient, or semi-permanent) and gathered water level and depth data. Using depth measurements, we created topo-bathymetric maps to investigate inundation patterns for various scenarios of Lake Superior water levels. Comparing wetland water levels to those of Lake Superior we found that water level patterns were unique per connection type, but that Lake Superior maintained baseline-water levels for all sites. Additionally, the wetlands we sampled demonstrated resiliency to substantial changes in water levels thanks to both bathymetry deeper than record low Lake Superior levels and sufficient undisturbed upslope habitats. However, the biotic communities within could shift in composition depending on future water level regimes. The increasing magnitude and frequency of Great Lakes water level fluctuations induced by climate change will influence all APIS coastal wetlands, regardless of connection type.

1. Introduction

As the largest freshwater lake in the world, Lake Superior's coastline is vast—covering over 4300 km—but only a small fraction of the coast supports wetland habitat. Where wetlands do exist, they provide habitat for an incredible diversity of plant and animal species, some of which are important sources of sustenance for the region's Indigenous people. Throughout the Laurentian Great Lakes basin, coastal wetlands are recognized as critical habitats that support fisheries (Trebitz and Hoffman, 2015), wildlife (Grand et al., 2020; Sierszen et al., 2012), and mitigate the effects of nutrient loading from upland landscapes (Carter et al., 2022; Knuth and Kelly, 2011). Coastal wetlands throughout the Great Lakes occur in shallow embayments, in river mouths, and in barrier-protected complexes (Albert et al., 2005). Geomorphic features such as barriers and sand spits often protect coastal wetlands from wind, waves, and ice scour, which supports persistence of submerged,

emergent, and wet meadow vegetation. Great Lakes coastal wetlands are influenced hydrologically by groundwater, precipitation, tributary inputs, and through connections with the adjacent lake (Mitsch and Gosselink, 2015). The relative contributions of these inputs to the water budgets of coastal wetlands influence the composition and structure of wetlands and their vegetation (Mitsch and Gosselink, 2015).

The Apostle Islands National Lakeshore (henceforth 'APIS'), located in northern Wisconsin on the southwestern coastline of Lake Superior, is home to a number of unique and important coastal wetlands. Lake Superior's coastal wetlands are situated predominantly along the southern and western shores due to underlying geology, glacial history, and topography (Brazner and Trebitz, 2018). As the last glaciers retreated approximately 10,000 years ago, the earth's crust began to rebound which continues today (Larson and Schaetzl, 2001). The rate of rebound is not consistent across the Lake Superior basin, however, and the earth's surface along the eastern and northern portions of the lake are rising

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<https://doi.org/10.1016/j.jglr.2025.102538>

Received 14 October 2024; Accepted 28 January 2025

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faster than the western and southern portions. This differential isostatic rebound causes coastal habitats along the southwestern portion of Lake Superior to subside at a rate of 27 cm/century relative to the northern and eastern shores (Bruxer and Southam, 2008). Coastal wetlands along the southwestern shoreline, therefore, are becoming increasingly inundated. The mouths of north-flowing rivers along the Wisconsin shoreline of Lake Superior, including two river mouths (Sand River and Little Sand Bay Creek) within the mainland unit of APIS and within the Red Cliff Band of Lake Superior Chippewa Reservation boundary, have become ‘drowned’ due to this differential isostatic rebound and have developed estuarine wetlands behind protective barrier beaches (Brazner and Trebitz, 2018; Epstein, 1997).

The Apostle Islands themselves are a group of 22 islands underlain by Precambrian sandstone bedrock that is exposed along some shorelines and subject to erosive forces of high energy Lake Superior waves. Other shoreline areas, especially along the southern tips of islands, experience reduced wave energy and longshore currents that allow sand deposits to form (Engstrom, 1984). Variable Lake Superior water levels over several millennia combined with differential isostatic rebound have caused these sandy shorelines to expand and recede, which facilitates development of dunes, barrier beaches, interdunal swales, and larger enclosed lagoons along the margins of islands throughout APIS (Engstrom, 1984). These coastal geomorphic features form the template in which coastal wetlands currently exist on the islands within APIS.

Designated as a National Lakeshore in 1970 by the National Park Service, APIS includes 21 of the region’s islands (Madeline Island, the largest and most populated island, is not part of the National Lakeshore) and a mainland unit (Fig. 1). Eighty percent of this land area was designated as the Gaylord Nelson Wilderness in 2004. The protected status of APIS mitigates many of the anthropogenic stressors that coastal wetlands throughout the Great Lakes experience. Anthropogenic stressors in other regions include direct loss of coastal wetland habitat via draining and filling (Mayer et al., 2004), nutrient loading from watershed land practices (Harrison et al., 2020), invasion by non-native vegetation (Lemein et al., 2017), water level stabilization (Wilcox et al.,

2005), and geomorphic alteration due to roads disconnecting coastal wetlands from adjacent lakes. Coastal wetlands in APIS are largely insulated from these impairments and represent some of the best-preserved examples of unmanipulated coastal landscapes in the Great Lakes. The Apostle Islands wetlands are not without human influence, however, as indigenous people inhabited the islands for millennia and made use of natural resources to sustain their communities by hunting, fishing, and picking berries, for example (Busch, 2008). Euro-Americans arrived by the early 17th century and periodically inhabited the islands. Fur trading, farming, fishing, quarrying, and logging all had waves of influence on the Apostle Islands ecosystems. Commercial fishing, sports fishing, and tourism continue to be influential around the Apostle Islands today (Busch, 2008). While fishing mostly occurs offshore, there are accounts of fishermen occasionally accessing the lagoon on Outer Island (J. Van Stappen, National Parks Service, 2017, personal communication).

Natural disturbance regimes for Great Lakes coastal wetlands center largely on hydrology, particularly fluctuating water levels (Keough et al., 1999; Toner and Keddy, 1997). Fluctuating water levels maintain the health and viability of coastal wetlands by flushing them of accumulated detritus, exposing and allowing for oxidation of sediments during periods of low water, and preventing dominance by any one plant species. Low water levels can expose bare soil for annual herbaceous plants to colonize or germinate from the seedbank, allowing vegetation to expand downslope. High water levels drown out later successional perennial and woody vegetation, preventing these species from encroaching and dominating (Keddy and Reznicek, 2018). Great Lakes water levels fluctuate on daily, seasonal, interannual, and longer-term intervals (Wilcox, 2004). Lake Superior coastal wetlands with a direct connection to the lake experience seiches multiple times per day (Trebitz et al., 2002). As a result, even vegetation occurring at the upland margin of these wetlands must be adapted to wetting and dewatering at frequent intervals. Seasonal water level fluctuations typically include late winter lows and late summer highs, though this varies with rates of precipitation and evaporation (Fry et al., 2020). Longer-term water level

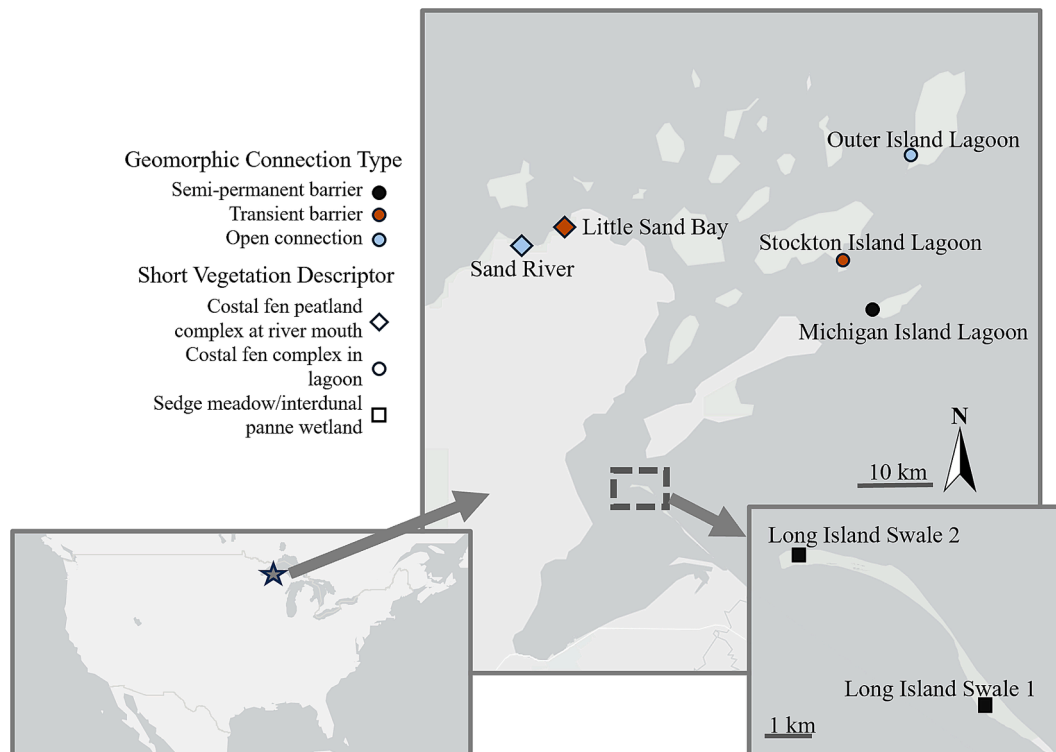


Fig. 1. Study sites within the Apostle Islands National Lakeshore in northern Wisconsin, USA. See Table 1 for more detailed vegetation descriptions per site.

dynamics result from patterns in net basin supply (precipitation and inflows vs. evaporation and outflows; VanDeWeghe et al., 2022). The period of record for Lake Superior (1918–present) reveals several relatively low-water periods and several high-water periods with a total range of 1.4 m between record high and low (Fig. 2). These water level fluctuations across the various temporal scales represent key drivers of coastal wetland ecology.

Recent modeling efforts suggest that Great Lakes water level fluctuations are likely to become more dramatic over the coming century as the climate warms (Gronewold et al., 2016; Kayastha et al., 2022; Seglenieks and Temgoua, 2022; VanDeWeghe et al., 2022). While any particular trend in mean water level for Lake Superior remains uncertain, both higher highs and lower lows, as well as more rapid transitions between extremes have all been predicted (Gronewold et al., 2016; Kayastha et al., 2022; Seglenieks and Temgoua, 2022; VanDeWeghe et al., 2022). Because fluctuations in water levels sustain diversity and vegetation structure in Great Lakes coastal wetlands (Anderson et al., 2023; Gathman et al., 2005; Wilcox and Nichols, 2008), any potential change in water level regimes could influence Lake Superior coastal wetland ecology in ways that are currently not well understood (Handler et al., 2020).

Here, we examined relationships between Lake Superior water levels and adjacent lagoonal wetlands, focusing on how these relationships vary depending on the type of geomorphological connection between the lagoons and the lake. Connection types included semi-permanent barriers, transient barriers, and open connections. To address concerns about the impact of shifting water level regimes (including more extreme high and low water levels) on the inundated area of wetlands, we developed inundation models based on topography, bathymetry (topography of land underwater), and water levels. Our investigation included wetlands on both the mainland unit (drowned river mouths) and islands (barrier beach lagoons) of APIS. Our study represents an effort to contribute to a recognized gap in nearshore bathymetry mapping and climate change-relevant models for wetlands in the Great Lakes region (Bergstrom et al., 2024).

2. Methods

Study Locations.

In summer 2017 we collected data from seven different wetlands in APIS: two were river mouth wetlands and the other five were on islands (Fig. 1, Table 1). Wetlands consisted of complexes of several habitat types, including areas of open water, emergent vegetation, and submergent aquatic vegetation. Several of the wetlands contained “poor fen” habitat with floating *Sphagnum* peat mats that graded into non-*Sphagnum* “shore fen” zones, emergent marsh, sedge meadow, shrub-scrub, or other habitats within the larger complex (Table 1). Study wetlands were chosen to represent the diversity of wetland geomorphic types found in the park (e.g., open and closed coastal lagoons and drowned river mouths) as well as a variety of wetland sizes.

Bathymetric Surveys.

Surveys of the seven coastal wetlands were conducted using a Topcon GTS 235 N Total Station (TS) theodolite (an instrument used in surveying for measuring horizontal and vertical angles) and prism mounted to a 3 m pole fitted with a 4 cm diameter round foot. The TS was affixed to a steel tripod that was set up on firm ground at the margin of each wetland and precisely leveled prior to starting surveys. Four to six control points, including the TS origin, were established in the upland margin around each wetland and marked with steel stakes driven to ground level. We established a local coordinate system for each survey by assigning an arbitrary elevation of 100 m and northing and easting coordinates of 1000 m for the first control point (TS origin). A second control point was set at the longest practical distance from the origin and used as a reference “backshot.” Thereafter, each TS sighting consisted of a horizontal angle, vertical angle, and distance to each surveyed point. From this information, the TS’s internal software calculated the local coordinates and relative elevation of each survey point. After local coordinates were established for the origin and backshot control points, we progressed to each of the remaining control points sequentially to determine their local coordinates and elevations. If the TS had to be relocated while surveying a single wetland because either the wetland was very large or because obstructed site lines precluded the survey of the entire wetland, control points from the previous survey route were

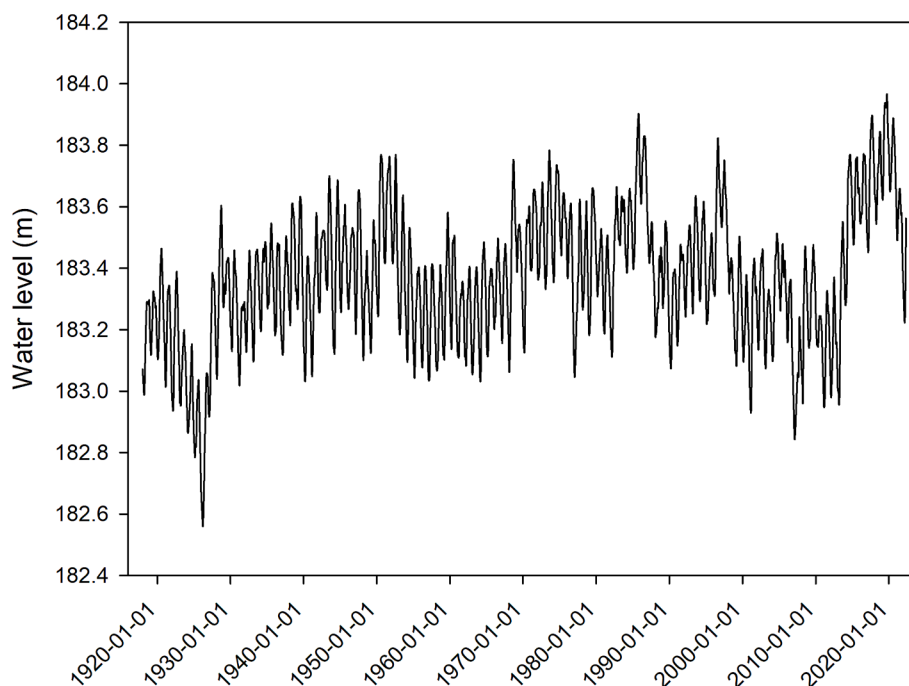


Fig. 2. Monthly mean water levels for Lake Superior in m above sea level. Data downloaded from the National Oceanic and Atmospheric Association’s Great Lakes Water Level Dashboard (https://www.glerl.noaa.gov/data/dashboard/GLD_HTML5.html).

Table 1

Apostle Islands wetland study site locations and descriptions.

Site name	Location (approximate centroid)	Brief site description and geomorphic barrier type	Approximate inundated area at average lake level (ha)	Cover type description	Short cover type description
Sand River	N 46.93094 W 90.93473	Mainland drowned river mouth, open connection to L. Superior	39	Wetland complex around drowned river mouth: primarily poor fen with areas of shore fen, coastal bog, emergent marsh, shrub-scrub, tamarack swamp, deep marsh, and open water	Coastal fen peatland complex at river mouth
Little Sand Bay	N 46.948232 W 90.883276	Mainland, tributary influenced lagoon, transient barrier	3.6	Wetland complex at drowned river mouth: primarily poor fen/coastal bog, sedge meadow, shrub-scrub, deep marsh, and open water	Coastal fen peatland complex at river mouth
Stockton Island Lagoon	N 46.915382 W 90.546728	Island lagoon, transient barrier	36	Large lagoonal wetland complex dominated by poor fen, but with areas of shore fen, deep marsh, and open water; ridge-swale topography supports ericaceous shrubs on ridges and <i>Sphagnum</i> or sedges in swales	Coastal fen complex in lagoon
Michigan Island Lagoon	N 46.877515 W 90.510499	Island lagoon, semi-permanent barrier	5	Wetland complex dominated by poor fen surrounding open water of lagoon containing deep marsh; emergent marsh is present consisting of non-native and hybrid cattail (both <i>T. angustifolia</i> and <i>T. x glauca</i>)	Coastal fen complex in lagoon
Outer Island Lagoon	N 47.001478 W 90.463349	Island lagoon, open connection to L. Superior	27	Wetland complex with a mix of shore fen, poor fen, deep marsh, and open water	Coastal fen complex in lagoon
Long I. Swale 1	N 46.724094 W 90.779066	Island lagoon, semi-permanent barrier	0.3	Small interdunal swale (panne) with sedge meadow during wet periods and alder during drier periods	Sedge meadow / interdunal panne wetland
Long I. Swale 2	N 46.730319 W 90.805371	Island lagoon, semi-permanent barrier	0.5	Small interdunal swale (panne) with sedge meadow	Sedge meadow / interdunal panne wetland

included in the new route to ensure the datasets could be accurately joined.

Bathymetric surveys consisted of loops in which all control points were sighted first, then wetland survey points were sighted, then the loop was closed by returning to each control point in the reverse order that they were initially sighted. By beginning and closing each loop with the control points, errors associated with the TS tripod moving or falling out of level during a survey loop could be identified. For particularly long survey loops (>100 points), a subset of control points was sighted midway through the loop as an additional quality control measure.

Surveys were designed to capture as much topographic and bathymetric variation as possible in 1 to 4 working days at each wetland. We balanced effort against efficiency required to complete each survey in the time available at each wetland, with the goal being to collect as many survey points as feasible to produce a higher resolution of topographic and bathymetric information. Surveyors determined the spatial density of survey points as they traveled throughout each wetland. In areas with little bathymetric variation, as determined visually, fewer survey points were collected. In areas with more variation, points were located at higher density. For smaller wetlands such as the Long Island swales, survey transects radiated out from the approximate center or deepest portion of the wetland to capture bathymetric contours in approximately concentric rings. For larger wetlands such as the Stockton Island lagoon, transects running perpendicular to the long axis of the wetland were surveyed to maximize efficiency in capturing the predominant contours of the lagoon. The person operating the prism pole moved throughout each wetland by wading or by using a paddleboard and communicated with the TS operator by 2-way radio to ensure sightings were taken when the prism was correctly seated on the sediment surface and the pole was vertical. The foot of the pole was placed gently on the sediment surface taking care to not penetrate the surface where substrates were soft. Survey point data were collected throughout the inundated portion of each wetland as well as the terrestrial margins of each wetland to ensure sufficient overlap with a terrestrial LIDAR-based digital elevation model (DEM; US Geological Survey, 2019, 2016). The large area of the river mouth wetland at Sand River limited the survey to only a portion of the wetland.

Bathymetric Data Processing.

Latitude and longitude data collected in the local coordinate system

were georeferenced using the geographic coordinates of control points, which were taken with a Trimble differential GPS during the surveys. To georeference survey points, we used the *Spatial Adjustment* tool in ESRI Arc Map version 10.5.1 and matched the common points between the survey and control points using the affine method (Babiker et al., 2007). Spatial layers were projected to NAD 1983 UTM Zone 15 N for processing. We adjusted vertical positions (Z values) for survey points to true elevations by linking our survey data to a DEM derived from bare earth LIDAR that covered the upland area surrounding each wetland. DEMs were available for the entirety of Bayfield and Ashland Counties (US Geological Survey, 2019, 2016), which we joined to cover all of APIS. The LIDAR data were originally acquired in two phases (October 2015 and April 2016) at a nominal point spacing of 0.7 m and the North American Vertical Datum of 1988 (NAVD 88). Vertical accuracy of the LIDAR-based DEM was reported as 0.109 m at a 95 % confidence level based on 107 independent checkpoints (US Geological Survey, 2019, 2016). To vertical reference our survey points, we first overlaid our georeferenced terrestrial control points onto the LIDAR-based DEM. Then we applied the *Extract Values to Points* tool to obtain elevations for each control point. The mean difference between LIDAR-based elevation and our measured Z value (arbitrary datum) at each control point was used as an adjustment factor to determine the actual bathymetric surface depth in meters Above Sea Level (ASL) for each survey point within each wetland. For example, if the LIDAR-based elevation (NAVD 88) for control points in a given wetland was higher than the surveyed Z values by a mean of 100 m, the adjustment factor would be + 100 m to the Z values of the survey points. Thus, a survey point with a Z-value of 10 m would approximate to 110 m in NAVD 88.

Interpolation

Creating a continuous topographic and bathymetric surface for each wetland required interpolating our vertically adjusted and georeferenced surveyed points and integrating them with the terrestrial DEM. First, we added a string of terrestrial points around each wetland at which DEM elevations were extracted and added to the surveyed elevations. This ensured that our bathymetric maps overlapped with the DEMs and could extend into their respective surrounding terrestrial habitat. Next, we used the Geostatistical Wizard function within the Geostatistical Analyst toolbar to test various Radial Basis Function interpolation methods. We varied kernel function, number of neighbors,

and sector type to determine which combination created the lowest root-mean-square error between survey data and interpolated data (Amante and Eakins, 2016; Curtarelli et al., 2015). Output grid cell size followed Hengl's (2006) guidelines for point samples, based on the distances between each point and its closest neighbor (shortest distance). Minimum grid cell size should be the value at which, at most, 5 % of points end up in the same pixel (i.e., the 5th percentile). Maximum grid cell size, which follows the Nyquist frequency concept (Shannon, 1949), should be half the average spacing of the shortest distances (Hengl, 2006). Ultimately, there is no one-best grid cell size, so decisions should also use domain knowledge to assess for how well it reflects elevation variation and geomorphic features of the area (Borkowski and Meier, 1994; Hengl, 2006; Kienzie, 2004). The interpolated wetland bathymetric models were then joined to the terrestrial LIDAR-based DEMs using the *Mosaic to New Raster* tool to merge the two data sets, creating a continuous topographic/bathymetric surface for each wetland. Where the DEM and interpolated layers overlapped, the DEM values were used.

Water Levels.

Within each of the seven wetlands and at nearshore Lake Superior locations adjacent to Long Island and Little Sand Bay, we installed automated water-level loggers (HOBO, model U20L-04) to collect pressure and temperature data every 10 min over the summer and fall of 2017. Each logger was suspended in a perforated PVC stilling well using high strength braided fishing line. At the Little Sand Bay and Long Island wetlands, additional loggers were attached to tree trunks to collect atmospheric barometric pressure data, also at 10-minute intervals, which we used to correct the water level data for fluctuations in barometric pressure. The relative elevation of each wetland and nearshore water level logger was measured during the bathymetric surveys and vertically referenced during bathymetric data processing. As a quality control step, we made an additional comparison for wetlands with an open or intermittent connection to Lake Superior in which the average nearshore water elevation was compared to the average water elevation of Lake Superior for the same time period using data from NOAA's Ontonagon Water Level Gauging Station (Station ID: 9099044). Despite seiche amplitude and frequency differing among locations, we assumed that the average water elevation over the period of observation would be approximately the same. Because water level at the Ontonagon Gauging Station is reported in the 1985 International Great Lakes Datum, we applied a correction to NAVD88 using NOAA's online adjustment tool (<https://www.ngs.noaa.gov/cgi-bin/IGLD85/IGLD85.prl>). We applied differences between the average Ontonagon water level and our calculated mean nearshore water elevations (based on water level logger data from periods of full connection to Lake Superior that we vertically referenced using the LIDAR-based DEM) to adjust the bathymetric elevation data and water level data for Sand River (adjusted down 13 cm) and Little Sand Bay wetlands (adjusted down 29 cm). No additional adjustments were deemed necessary for the other wetlands.

Water level data were plotted for each wetland and its closest Lake Superior nearshore water level logging station. We acquired precipitation data for the same time period for the La Pointe weather station on Madeline Island (National Weather Service Station ID: 474953) and included precipitation per day in water level figures to help explain changes in water levels throughout the period.

We constructed inundation models using the topographic/bathymetric surface layers and water level data to represent the percentage of area inundated at each wetland across a gradient of water levels. These inundation models assume that water can flow freely to all cells, regardless of vegetation cover, so are not hydrologically constrained by flow paths. Therefore, the inundation scenarios should be viewed as the maximum potential inundated area for each wetland at each water level.

3. Results

Bathymetry.

Using spatial models, we generated bathymetric maps for the seven

wetlands where we collected survey data (Figs. 3, 4, S1–S5). All maps were made to a grid cell size of 1 m, which was based on calculations using distances between each point and its nearest neighbor (Hengl, 2006; Table S1). Out of the seven wetlands sampled, the lowest maximum recommended grid cell size was 1.28 m (Long Island Swale 1; Table 2) while the highest minimum recommended was 4.22 m (Stockton Island; Table S1). While too fine of grid cell sizes runs the risks of introducing local artefacts, increasing computation time, and taking up unnecessary memory (Hengl, 2006), none of these were a concern due to the small size of our areas of interest (Stockton Island being the largest at roughly 2 km²). Too coarse of grid cell sizes, on the other hand, lose accuracy from reduced detail. To stay consistent between the maps while respecting the level of detail possible with the data, we used a standard grid cell size of 1 m. Given the clustered nature of our sampling procedure, this means some areas had more extrapolation (due to low point density) than others. The resulting bathymetric maps were deemed sufficient based on our domain knowledge of the area and our purpose of providing initial efforts to model how extreme lake water levels will play out within APIS wetlands.

Both semi-permanent barrier wetlands, Stockton Island and Little Sand Bay lagoons, had similar distributions of bathymetric elevations (Fig. 6). The Stockton Island lagoon was the largest wetland sampled, both in area (Table 1) and number of survey points (Fig. 3). The lowest elevation in this lagoon, 179.8 m ASL (3.6 m below the long-term average for Lake Superior of 183.4 m ASL, hereafter 'average LS'), put the water depth there at 3.96 m during the survey period, which was the greatest water depth observed in the study. The Little Sand Bay lagoon lies primarily within a constrained channel with relatively steep-sloped sides. The deepest portion of this lagoon had an elevation of 181.2 m ASL (2.2 m below average LS), which was among the lowest elevations surveyed (Fig. S5).

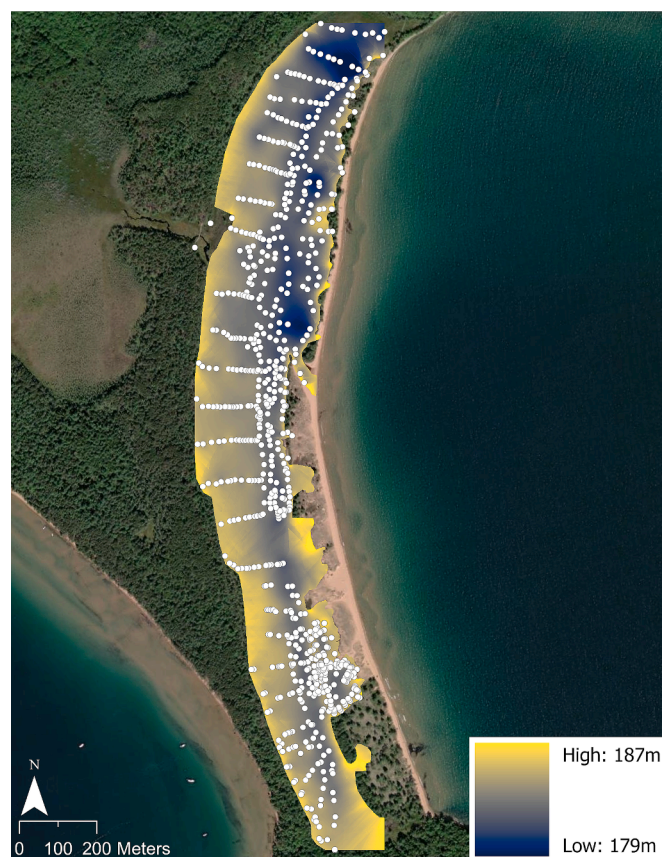


Fig. 3. Bathymetric map of the Stockton Island Lagoon. Survey points are included for reference (n = 222 points).

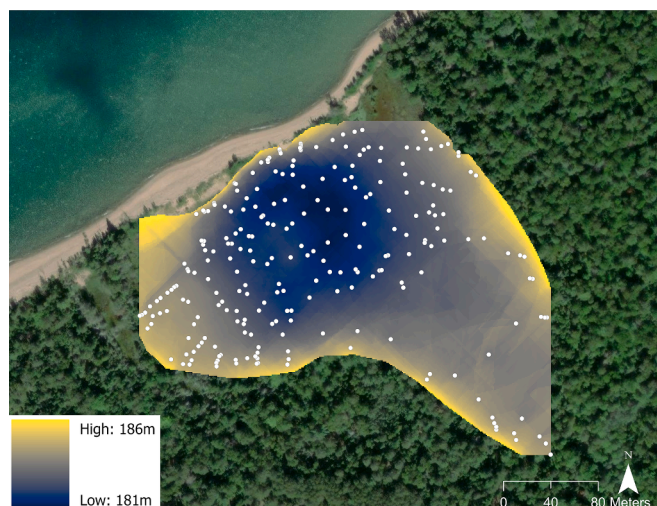


Fig. 4. Bathymetric map of the Michigan Island Lagoon. Survey points are included for reference ($n = 207$ points).

Table 2

Survey point spacing per wetland, calculated from distances between each point and its nearest neighbor, hence 'shortest distance'.

Site name	Mean shortest distance	Maximum shortest distance	Minimum shortest distance
Long Island Swale 1	2.56	1.28	0.98
Long Island Swale 2	5.24	2.62	1.22
Little Sand Bay	4.65	2.32	1.60
Michigan Island	6.88	3.44	1.25
Outer Island	8.67	4.34	1.58
Sand River	6.85	3.42	1.92
Stockton Island	12.39	6.20	4.22

Mean shortest distance is a metric used in calculating the appropriate grid cell size for interpolations.

The transient barrier wetlands, Michigan Island, and Long Island Swales 1 and 2, had different distributions of their bathymetric elevations (Fig. 6). The Michigan Island lagoon lies within a round depression and has a substantial amount of its bathymetric surface at an elevation of 181.1–181.4 m ASL (2.3–2 m below average LS; Fig. 4). The Long Island swales occupy relatively small depressions on the island (Figs. S1, S2). The depressional basin at Long Island Swale 1 was approximately 0.01 ha. with a bathymetric surface between 183.2 m and 183.5 m ASL (0.2 m below and 0.1 m above average LS) and a large surrounding plane at a slightly higher elevation (183.5–183.8 m; 0.1 m and 0.4 m above average LS).

The distribution of bathymetric elevations for the open connection wetlands at Outer Island and Sand River peaked around 182.6 m ASL (0.8 m below average LS). However, this represents a much greater proportion of the elevations within Sand River than in Outer Island, where Outer Island has a much greater range of elevations above this (Fig. 6). At Outer Island, the majority of the lagoon had a bathymetric elevation of 182.0 m–182.6 m ASL (1.4 and 0.8 m below average LS), which resulted in a depth of 1.2–1.8 m during the sampling period (Fig. S3). There also appears to be a deeper basin at the north end of the Outer Island lagoon, though bathymetry in this area was largely inferred by interpolation from the depth pattern of points surrounding the basin since no points were surveyed within it due to sampling limitations. Sand River had a narrow range in bathymetric elevation. Most of the surveyed area fell between 182.3 m and 182.9 m ASL (1.1 m and 0.5 m below average LS) with a narrow channel cutting through the surveyed

area (Fig. S4). The highest surveyed elevation at Sand River was also quite low (182.9 m ASL, 0.5 m below average LS) but this was likely due to the survey area covering only a portion of this large river mouth wetland. Bathymetric interpolation was particularly challenging in the Sand River wetland due to high density of survey points near the channel (mean shortest distance between point pairs of 4.0 m) and low density of points across the remainder of the survey area (mean shortest distance of 25.0 m).

Water Levels.

Lake Superior water level increased approximately 15 cm over the summer of 2017 based on data from our nearshore loggers (Fig. 5) and monthly mean levels at NOAA's Ontonagon Water Level Gauging Station (Station ID: 9099044). Lake Superior water level fluctuations at the hourly to daily time scales are driven primarily by surface seiches and the largest seiche we recorded occurred from June 12 through June 15. Daily minimum and maximum water levels differed by over 30 cm on June 12 at Long Island and over 60 cm at Little Sand Bay (Fig. 5). This seiche event followed a low-pressure system that swept across the lake on June 10 and 11, causing winds in excess of 64 km/h at the La Pointe weather station. Lake levels continued to fluctuate by over 50 cm at Little Sand Bay and 30 cm at Long Island through June 15. The remainder of the summer and early fall saw seiche amplitudes less than 30 cm with means ($\pm$ SE) for the entire period of 12 ± 8 cm (range: 3.9–32.9 cm) at Long Island and 18.6 ± 12.5 cm (range: 7.6–61.9 cm) at Little Sand Bay.

Water level data were collected successfully at six of the seven wetlands (the logger at Long Island Swale 2 failed shortly after deployment). Wetland water levels exhibited three distinct patterns over the study period based on the type of connection between each lagoon and Lake Superior: semi-permanent barrier (Long Island Swale 1 and Michigan Island), transient barrier (Little Sand Bay and Stockton Island), and open connection (Outer Island and Sand River). Separated from Lake Superior by their stable semi-permanent barriers, Long Island Swale 1 and the Michigan Island lagoon had the most stable water levels of the wetlands studied (Fig. 5). Their water levels reflected the daily mean Lake Superior level and had no evidence of influence from Lake Superior seiches. Daily mean water levels in Long Island Swale 1 and Michigan Island Lagoon increased throughout the summer at the same rate as Lake Superior suggesting subsurface hydrologic connectivity. This connectivity was also evident in that precipitation events had only minimal and temporary influence on the water levels of these two lagoons. For example, at Long Island Swale 1, a 3 cm rain event on June 28 raised wetland water levels temporarily but levels equilibrated within 24 h (Fig. 5). Additional minor and short-lived increases coincided with rain events at both Long Island Swale 1 and Michigan Island on July 5, July 18, and August 26 (Fig. 5).

The Little Sand Bay and Stockton Island lagoons were separated from Lake Superior by transient barriers that built up and blew out multiple times throughout the summer and fall. Consequently, these lagoons had the most dramatic fluctuations in water levels observed (Fig. 5). At Little Sand Bay, water level data indicated that a barrier formed on June 13 and water level in the lagoon quickly rose. This barrier is also apparent in Sentinel 2 L2A imagery from Jun 16 (Copernicus Sentinel Data, 2017). The lagoon level remained higher than Lake Superior and relatively stable until June 28 when a rain event caused a rapid rise of 24 cm. Water appeared to then slowly leak out of the lagoon until July 23 when a rain event led to a 35 cm increase to 184.4 m ASL, which put the lagoon level 60 cm higher than Lake Superior. At 18:00 on July 25 the barrier was breached and rapidly cut a new channel to Lake Superior, which is visible in Sentinel 2 L2A imagery for July 26 (Copernicus Sentinel Data, 2017). Within two hours, the lagoon water level fell 60 cm until equilibrating with Lake Superior at approximately 183.8 m ASL (Fig. 5). The lagoon remained connected to Lake Superior for 10 days and seiche activity is evident in the lagoon water-level record over this period. Barriers re-established and breached two additional times at this location over the study period.

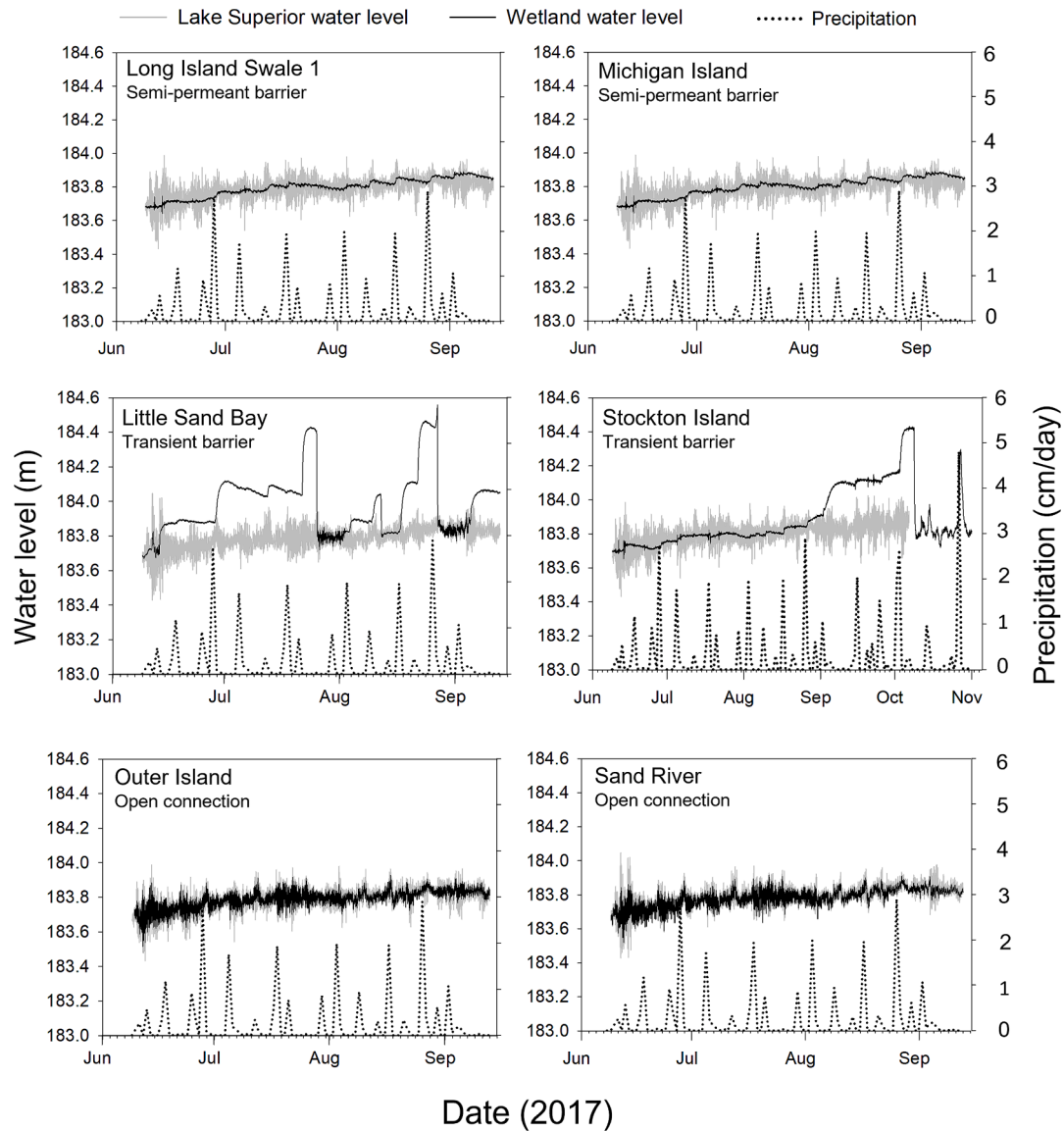


Fig. 5. Wetland water levels (black lines) and Lake Superior nearshore water levels (gray lines) for summer of 2017. Water level data were collected at 10-minute intervals. Precipitation data (dashed lines) were acquired from the La Pointe, WI weather station on Madeline Island. See Table 1 for more detailed vegetation descriptions per site.

Water levels in the Stockton Island lagoon behaved similarly to those of Little Sand Bay, though the transient barrier appeared to be more stable at Stockton Island. Water levels in the lagoon steadily increased throughout the summer with small but rapid increases associated with rain events on July 5, July 18, and August 17 (Fig. 5). In the beginning of September, lagoon water levels rose to 184.1 m and remained stable for approximately three weeks until October 3 when a rain event caused the lagoon to rise to 184.4 m where it remained until the barrier breached on October 8. By 6:30 on October 9 the lagoon had drained 59 cm to equilibrate with Lake Superior (Fig. 5). The lagoon remained connected to Lake Superior and seiche activity is evident from October 9 until October 26. A barrier then re-formed and water levels in the lagoon surged from 183.8 m to 184.3 m in approximately 32 h. Re-formation of the barrier coincided with a 45 km/hr northeasterly wind that likely supported the sediment transport necessary for barrier formation. A 5 cm rain event occurred simultaneously, which caused water levels in the lagoon to rise behind the newly created barrier, which breached at 16:00 on October 27 causing the lagoon to drain over the subsequent 36 h to once again equilibrate with Lake Superior.

Mean water levels in Outer Island Lagoon and Sand River, which

both had open connections to Lake Superior, were identical to mean Lake Superior water level for the study period. Seiche amplitudes in these wetlands were also very similar to that of Lake Superior, though both were slightly reduced compared to the seiches measured at the Long Island nearshore station (Fig. 5). This slightly muted seiche is likely due to the offshore position of Outer Island and the narrow connection between the Sand River wetland and Lake Superior. There was no evidence that precipitation affected water levels in either Outer Island Lagoon or Sand River as their open connections appeared to allow for rapid equilibration with Lake Superior (Fig. 5).

Inundation

For five of the seven wetlands, the median elevation was between 182.9 m and 183.2 m ASL with a substantial portion of the bathymetric area falling below this elevation (Fig. 6), which suggests that even when Lake Superior is at or near its record low water levels for the period since 1918 (Fig. 2), these wetlands maintain at least some inundated area. Long Island Swale 2 has a bi-modal distribution of elevations because this wetland has both a relatively deep basin (approx. 182.6 m ASL) and a large shallower area with a mean elevation of approximately 183.5 m ASL. Interesting deviations from the typical pattern of bathymetric

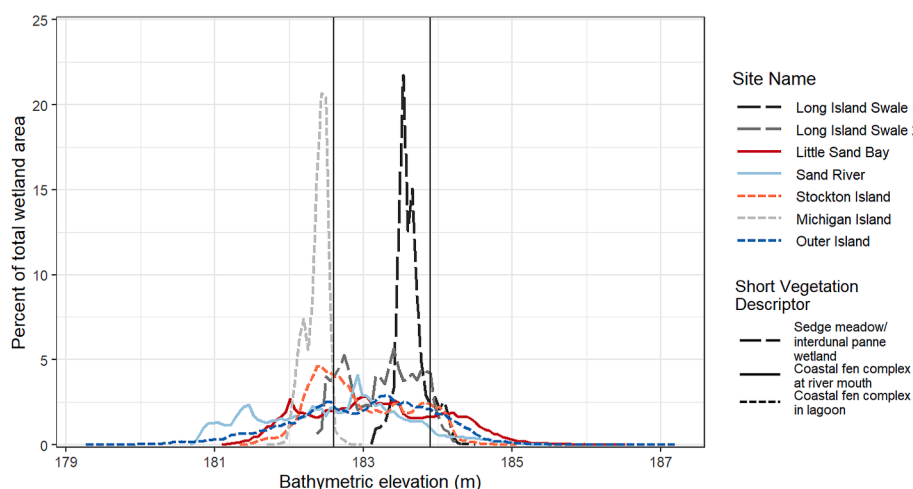


Fig. 6. Distribution of bathymetric elevations per wetland, represented as percent of the total wetland area. The complete extent for each wetland (100 %) represents the total area that was surveyed regardless of inundation at the time surveys were conducted. For Sand River, the surveyed area was only a portion of the total wetland extent. Thin vertical black lines represent record low (182.6 m) and record high (183.9 m) water levels for Lake Superior since 1918. Colors represent geomorphic connection types, where shades of black are semi-permanent barrier wetlands, shades of red are transient barrier wetlands, and shades of blue are open connection wetlands. See Table 1 for more detailed vegetation descriptions per site. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

elevations occurred at Long Island Swale 1 and Sand River. At Long Island Swale 1, the small (100 m²) and relatively deep depression was surrounded by a large flat plane, which results in the majority of the wetland not being inundated until water level exceeds 183.5 m ASL (Fig. 6). At the Sand River wetland, nearly all of the surveyed area was below 182.8 m ASL which results in the large peak in inundated area between 182.3 and 182.9 m ASL.

We also visualized inundation patterns by calculating and plotting the relative difference in inundated area compared to the area inundated when Lake Superior is at its long-term average of 183.37 m ASL (Fig. 7). Therefore, each line in this figure represents the accumulation of wetted area with increasing water level relative to the long-term average water level of Lake Superior. The unique shape of Long Island Swale 1 is again apparent with this approach; the small depression fills between 183.1 m and 183.5 m ASL and then the large surrounding plain becomes inundated when water levels rise from 183.5 to 183.8 m ASL. After removing Long Swale 1 to better visualize the remaining wetlands, the flat and relatively low bathymetric elevation of the Sand River wetland is apparent (Fig. 7). The Sand River wetland becomes 100 % inundated when water levels exceed 182.6 m ASL, though our survey data covered just a portion of the Sand River wetland and higher elevation areas existed beyond the boundary of our survey. The relatively dramatic increase in inundated area with increasing water level at Long Island Swale 2 suggests that the bathymetric slope in this system is relatively gradual and continuous so that with increasing water level, a greater percentage of inundated area is gained relative to the other wetlands.

4. Discussion

Throughout the Great Lakes, coastal wetlands have suffered from significant, numerous, and varied anthropogenic impacts, including loss of approximately 50 % area due to filling and land conversion. Among the wetlands that remain, nutrient loading (Harrison et al., 2020), non-native species invasions (Lemein et al., 2017), and hydrologic alteration (Wilcox et al., 2005) are among the most prevalent impacts throughout the Great Lakes. Coastal wetlands in protected landscapes such as APIS therefore provide valuable and increasingly rare examples of high quality and largely unmanipulated systems. Most important to our study were the relatively unmanipulated connections and barriers between APIS's coastal lagoons and Lake Superior. The hydrology of these lagoons and the wetland habitats that form within them likely function as

they have since pre-European colonization of the region. Non-degraded coastal geomorphology is rare for the Great Lakes given that the majority of coastal ecosystems throughout the basin have connections that are actively maintained for navigation or have barriers that are hardened to protect coastal infrastructure such as roads and bridges. Furthermore, in many locations nearshore sediment supply and long-shore transport have been altered by coastal revetment, which impairs barrier beach and bar formation (Gharib et al., 2023).

While the unmanipulated connections between coastal lagoons and Lake Superior within APIS represent three unique patterns of coastal wetland hydrology, the minimum water level observed in each type was ultimately controlled by Lake Superior level. This demonstrates the overriding influence Lake Superior water levels have on coastal wetland hydrology. Thus, rising Great Lakes water levels coupled with more fluctuations between extreme highs and lows, as predicted in the coming century because of climate change, will influence all APIS coastal wetlands regardless of connection or barrier type (Kayastha et al., 2022; Seglenieks and Temgoua, 2022; VanDeWeghe et al., 2022). Despite the strong coupling of Lake Superior hydrology to lagoon water levels, our bathymetric surveys revealed that six of the seven wetlands surveyed had benthic areas lying below the record low Lake Superior elevation of 182.6 m ASL observed in April 1926. This means that even when Lake Superior reaches record low levels, wetland organisms will likely have access to inundated refuge habitats within the coastal lagoons. On the upslope margins of APIS coastal lagoons, land protection and management policies by the National Park Service have prevented encroachment by human infrastructure. This provides space for the wetlands to expand upslope during high-water periods. Both of these attributes—bathymetry deeper than the record low Lake Superior levels and sufficient natural upslope habitats—contribute to natural climate resiliency of APIS coastal wetlands.

Understanding the various drivers of wetland water level dynamics is important because water level fluctuation exerts critical influence over plant community structure in Great Lakes coastal wetlands (Wilcox, 2004). Woody vegetation and species less tolerant of prolonged flooding (e.g., *Alnus incana*, *Calamagrostis canadensis*) expand downslope during low water periods, which also facilitates replenishment of the seedbank by these species. The return of high water then pushes the woody and less flood-tolerant species back upslope while obligate emergents (e.g., *Equisetum fluviatile*) and aquatic species (e.g., *Potamogeton* spp., *Lemna minor*) fill in behind them. Long-term fluctuation in Great Lakes water

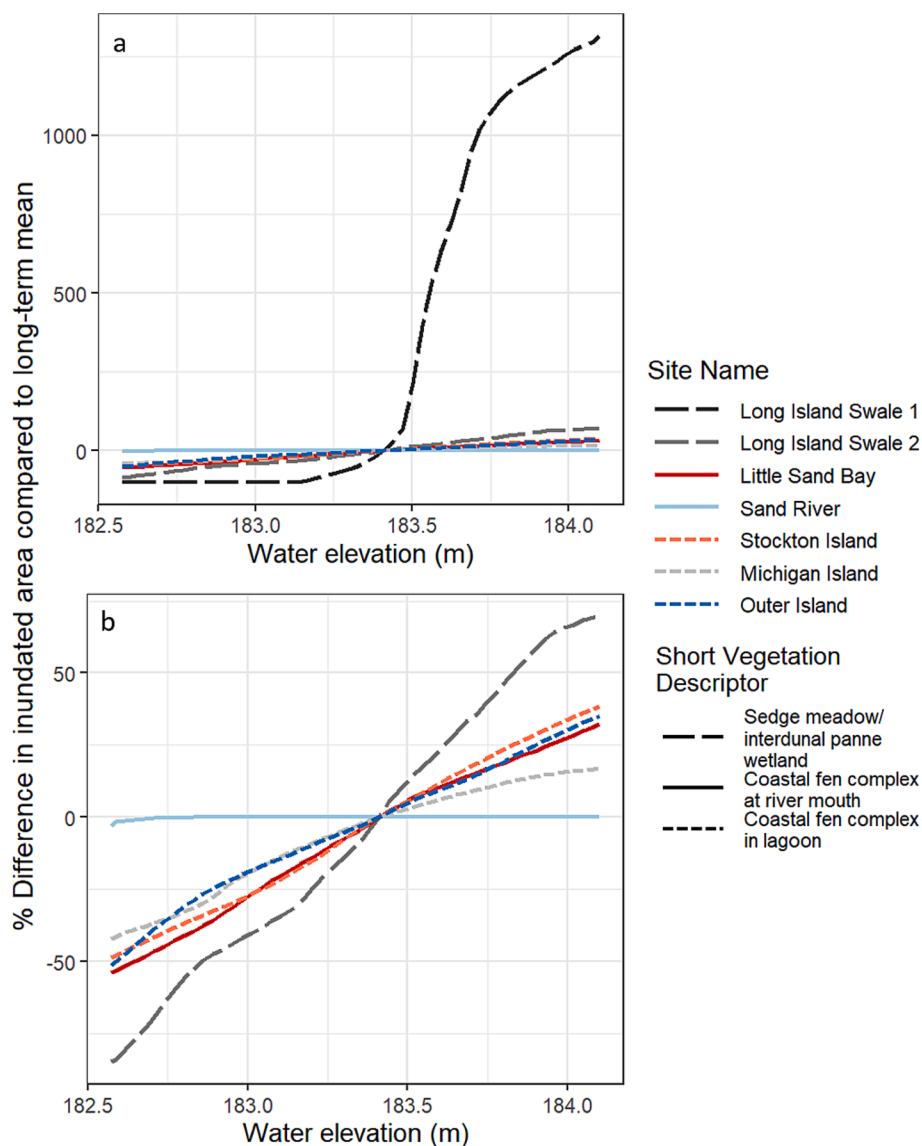


Fig. 7. Percent difference in inundated wetland area relative to inundated area at the long-term average Lake Superior water level of 183.4 m ASL. Colors represent geomorphic connection types, where shades of black are semi-permanent barrier wetlands, shades of red are transient barrier wetlands, and shades of blue are open connection wetlands. Panel a includes Long Island Swale 1, which showed a dramatic increase in inundated area when water levels increase above 183.5 m. Long Island Swale 1 is excluded in Panel b. See Table 1 for more detailed vegetation descriptions per site. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

levels drives the pattern of consistently dynamic plant communities and reduces the likelihood of any one species dominating (Anderson et al., 2023; Keddy and Reznicek, 2018). In contrast, after water levels were regulated on Lake Ontario beginning in 1958, coastal wetland plant diversity declined markedly and monodominant stands of cattail (*Typha* sp.) and later common reed (*Phragmites australis*) became common (Rupasinghe and Chow-Fraser, 2021; Wilcox et al., 2005). While the outflow of Lake Superior is partially regulated by the locks and compensating works at Sault St. Marie, the degree of flow management does not significantly impact natural water level fluctuation in Lake Superior (Lamon and Stow, 2010). Consequently, native plant communities have continued to thrive in protected coastal areas such as APIS and nearby Kakagon-Bad River Sloughs. Yet cattail invasion has emerged as a concern in parts of both wetland systems following a long-term low-water period (1998–2013) in Lake Superior (Cooper et al., 2020; Meeker et al., 2023).

In addition to the influence of Lake Superior, our study documented that each lagoon's particular connection type and watershed inputs

determined shorter-term wetland water level regimes. The lagoons experiencing the most dramatic water level fluctuations due to transient barrier formation (Little Sand Bay and Stockton Island) appeared to have vegetation adapted to these conditions. Although our study was not designed to focus on linking plant communities to inundation patterns, we did make some relevant observations during our field work. In the Stockton Island lagoon, for example, we observed large areas of floating peat mat that rose and fell as the lagoon filled and emptied. Floating peat mats were not restricted to the transient barrier lagoons, however; the Michigan Island lagoon also had significant areas of floating mat. These coastal fen mats likely reflect an adaptation to longer-term fluctuations in Lake Superior water levels where our water level analyses clearly showed that even lagoons with intact semi-permanent barriers track Lake Superior water levels (e.g., lagoon water levels equilibrated with Lake Superior within days after a large rain event).

There is likely an upper limit to the amount of water level fluctuation that mat-forming native plant communities can tolerate. Peat mats in APIS coastal wetlands tend to be dominated by *Sphagnum* moss species

(Cooper et al., 2020), which have a broad range of vulnerability to flooding and drying. For example, species that occur in low-lying positions between hummocks are more susceptible to desiccation and mortality during extended droughts than hummock-forming species (Hajek and Beckett, 2007; Vile et al., 2011) and floating *Sphagnum* peat mats in coastal fens may not be able to rise high enough to avoid complete submersion during years of extreme high water. In a related study, *Sphagnum* moss cover declined in Michigan and Stockton Island lagoons in 2017 during a higher water period compared to previous surveys in 1998–1999 and 2010 (Cooper et al., 2020). Thus, while wetland water level fluctuation is a key driver of coastal wetland vegetation structure and water levels are influenced by both Lake Superior and the connections between wetlands and the lake, there are likely upper limits of water level variation beyond which plant communities would change in dramatic ways. Quantifying this threshold is beyond the scope of our study but could be done, as in Wilcox and Xie (2007), by profiling the vegetation of each site and how it responds to de-watering and flooding events and fitting these profiles into bathymetric models to simulate future water level regimes over the profiles. Not only do water levels influence vegetation, but vegetation can influence flow paths (Lightbody et al., 2008; Temmerman et al., 2005) and affect how water levels impact a given wetland (Anderson et al., 2023), which should also be accounted for to make more robust models.

In addition to the variation in water level dynamics among lagoons with different connection types, variation in bathymetric and topographic slope resulted in widely differing rates of inundation with changing water levels. Long Island Swale 1 demonstrated that the inundated area for wetlands forming within flat and low-lying landscapes can change dramatically with moderate changes in water level (Wilcox and Xie, 2007). In the face of more extreme and prolonged water level fluctuation regimes, the 12-fold increase in inundated area that would occur between average and record high Lake Superior water levels in Long Island Swale 1 would likely have a dramatic influence on vegetation communities. Relatively flat coastal habitats like Long Island Swale 1 are noted to be particularly susceptible to drastic increases in water levels, which can significantly reduce the extent of less prolonged-flood tolerant (i.e., wet meadow) species, allowing emergent and submergent vegetation zones to expand in their stead (Anderson et al., 2023; Wilcox and Xie, 2007). During prolonged low lake levels, relatively flat coastal habitats are also subject to large-scale drying. These habitats then become particularly susceptible to non-native species invasions because aquatic species die back and niche space becomes available. Some of the most aggressive invasive wetland plants in the Great Lakes region (e.g., *Typha* sp., *Phragmites australis*) are able to exploit drier conditions (Lishawa et al., 2010; Tulbure and Johnston, 2010). Invasive wetland plants can also survive extreme high-water years by exploiting the resiliency of floating peat mats, using them as refugia. So, land managers should be increasingly vigilant for non-native species invasions during years of low lake levels, especially in coastal wetlands with flatter bathymetry, and for remnants of invasives during high lake level years in coastal wetlands with floating mats.

To provide managers with the means to better assess and predict the effect of extreme de-watering and flooding events on coastal wetlands, future modeling efforts should integrate vegetation profiles, bathymetry, and wetland-lake geomorphic connection types. Few Great Lakes studies incorporate all three (e.g., Wilcox and Xie, 2007, for Lake Ontario). The present study emphasized the importance of bathymetry and geomorphic connection types when simulating inundation scenarios. Cooper et al. (2020) highlighted the significance of geomorphic connection type when assessing the influence of water level regimes on vegetation communities in APIS wetlands. Wilcox and Nichols (2008) and Smith et al. (2021) demonstrated the critical role of bathymetry and vegetation in assessing how coastal wetlands respond to altered future extreme water level fluctuations by developing conceptual models for how coastal wetland vegetation along elevation gradients responds to both low water levels in Lake Huron (Wilcox and Nichols, 2008), and

extreme highwater levels in Lake Ontario (Smith et al., 2021). Application of bathymetric models of vegetation response in coastal wetlands around Lake Superior would be especially informative to models of extreme water level impacts to Great Lakes coastal wetlands. Improved modeling of the relationship between hydrology and coastal wetland vegetation is especially important in the context of shifting wind, wave, and precipitation patterns in the region (Anderson et al., 2015; Zhang et al., 2019), which could alter the frequency and timing of transient barrier formation and persistence.

Our bathymetric models indicate that these wetlands have the capacity to persist in some form despite extreme fluctuations in Lake Superior water level. Given that many of these wetlands support wetland complexes that include unique coastal fen species, follow-up research should identify thresholds in tolerance and triggers for directional shifts in vegetation composition in this system. Regardless, to support the resiliency in biota of these and other similar coastal wetland systems, even in the face of new water level regimes, we recommend that decision makers implement management strategies to reduce and prevent major impacts to coastal wetlands. Actions could include preserving shoreline geomorphic processes, minimizing activities that cause excessive shoreline erosion, applying early detection and rapid response tactics for non-native species, limiting further anthropogenic development or promoting sustainable development, and reducing nutrient enrichment to coastal wetlands (Schmitt et al., 2022). For instance, the preservation and restoration of dune grasses along the lakeward edge of coastal wetlands is critical to minimizing shoreline erosion while still allowing for natural geomorphic processes to continue, like the intermittent direct connection between wetland and Lake Superior. Additionally, any future dock replacements or new docks should be designed to minimize changes in sediment transport patterns and long-shore current strength that could promote erosion of sand barriers. Management practices that are intended to retain and further promote the resiliency of Great Lakes coastal wetlands and their biotic communities are particularly important in a future climate likely to feature more extreme water levels (Gronewold et al., 2016; VanDeWeghe et al., 2022), coupled with more intense storms (e.g., wind-driven waves and stronger currents), especially during periods of high-water levels (Schmitt et al., 2022).

5. Conclusions

By combining on-the-ground survey techniques with terrestrial LIDAR data we developed a series of topo-bathymetric maps for coastal wetlands of Lake Superior. The approach can be used in other shallow coastal systems where near-shore LIDAR coverage is not available. With a combination of these interpolated topo-bathymetric surfaces and in-situ water level data, our study demonstrated how water levels of coastal wetlands of the Apostle Islands National Lakeshore are intrinsically linked to those of Lake Superior, regardless of connection type (open, transient, or semi-permanent). Through simulations of both low and high water levels, our results show that APIS wetlands can be quite resilient to large changes in Lake Superior water levels. Even during a scenario of the lowest recorded Lake Superior water level, most APIS wetlands we surveyed would retain some inundation, supporting their function as refugia for wetland biota. However, animals such as fish and swimming macroinvertebrates may not be able to access these wetlands from Lake Superior or vice versa at extreme low water levels. Despite the retention of inundated areas during extreme low water events, wetland plant communities may be vulnerable to substantial change in composition and structure, such as encroachment of invasive species like cattail (*Typha* spp.), which can persist even after water levels rise again, especially on the floating peat mats found within some of the wetlands. During high water periods there are few barriers to landward migration of wetlands due to expansive, relatively undeveloped coastlines around the APIS wetlands, supporting their adaptive capacity to extreme high water-levels. Future work modeling responses of vegetation and other biotic communities to future climate and water level regimes is needed

and would be improved by accounting for how bathymetry and vegetation presence influences inundation patterns, and by incorporating barrier and connection type of the coastal wetland to its respective lake.

CRedit authorship contribution statement

Olivia C. Anderson: Writing – review & editing, Writing – original draft, Formal analysis. **Sarah E. Johnson:** Writing – review & editing, Conceptualization. **Shea Schachmeyer:** Writing – review & editing, Formal analysis. **Shelly L. Ray:** Writing – review & editing, Formal analysis. **Matthew J. Cooper:** Writing – review & editing, Writing – original draft, Formal analysis, Conceptualization.

Declaration of competing interest

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

Acknowledgements

This work was supported by the National Park Service (NPS) through Great Lakes Restoration Initiative funding (collaborative agreement number P16AC01387). We are grateful for the support of NPS staff including J. Van Stappen, P. Burkman, K. Pemble and other staff who assisted with boat transports. The Mary Griggs Burke Center provided logistical support. Northland College undergraduate students contributed to this work, including M. Hein, H. Hoff, M. Mader, J. Miles, S. Tharpgeorge, and A. Hefko. Administration assistance at Northland College was provided by L. Williamson and S. Venero. J. Meeker's early wetland work in the Apostle Islands laid a strong foundation for our research. We also thank J. Elias for providing further insights into Apostle Islands coastal wetlands. Finally, the manuscript was improved with helpful feedback from the Associate Editor and two anonymous reviewers.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jglr.2025.102538>.

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