DOCUMENTING THE GEOMORPHIC IMPACTS OF HIGH LAKE LEVEL ON FRESHWATER COASTAL WETLANDS USING TOPOBATHYMETRIC SURVEYS: A CASE STUDY FROM SAGINAW BAY IN LAKE HURON

By

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ABSTRACT

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There has been extensive research connecting vegetative processes in coastal freshwater wetlands and water level fluctuations. However, there is little work on changes to wetland geomorphology, what those transformations may be, and how they impact the wetland. This paper aims to identify changes in wetland geomorphology and decern any correlation between water level and vegetation extents. Data for this study spans from 2012 to 2021, capturing the most recent period of rising water levels. Vegetation extent imagery and topobathymetric data were collected during field excursions in the summer of 2021 and compared to NAIP imagery acquired from USGS Earth Explorer and topobathymetric LiDAR data from NOAA Data Access Viewer. Imagery from 2021 was collected using a DJI Phantom 4 Pro quadcopter drone and 2021 topobathymetric data utilized an RTK-GPS antenna and Seafloor Systems SonarMite single-beam echosounder to conduct boating, kayaking, and wading surveys at each study site. Studies took place at Wigwam Bay and Quanicassee State Wildlife Areas located in Saginaw Bay of Lake Huron. Findings show water level had a variable impact on vegetation extent and suggest that erosion of sediment occurred during high water levels at both sites. Results show observable changes in geomorphology adjacent to the wetlands. Changes to geomorphology could potentially impact the size, health, and ecosystem services of coastal wetlands. This initial study has limitations due to the nature of available historical data but is intended as a first step towards further understanding the role of geomorphology in coastal freshwater wetland systems.

This thesis is dedicated to my mom, Rebecca Laudati.
You worked so hard to give me every opportunity to succeed and you always made sure I saw the best in myself which has carried me through this journey and many to come. I am forever grateful.

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Special thanks and appreciation to my advisor for his guidance and patience, my family who have always supported me, my friends who helped me through hard times, and Allison, my therapist, for listening to me talk about this thesis for a very long time.

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INTRODUCTION

Freshwater wetlands are vital ecosystems found throughout the world in forms such as peatlands, arctic wetlands, high-altitude wetlands, and along the coasts of oceans, rivers, and lakes (wetlands.org). These systems support diverse assemblages of terrestrial and aquatic vegetation and wildlife while benefiting the health of local, regional, and global environments (Keddy et al., 2009). Globally, healthy freshwater wetlands provide ecosystem services such as provisioning, regulation, habitat, and cultural significance to their surrounding communities and local environment (Clarkson, 2017). Clarkson (2017) describes provisioning services provided by wetlands as a broader category that includes services such as food supply, freshwater supply, and raw materials. Regulation refers to the wetland's influence on the environment such as regulating water flow and water quality, maintaining soil fertility, and preventing of erosion along coastlines. Habitat services supplied by wetlands center around supporting terrestrial and aquatic biotic components of the environment such as supporting diverse vegetation and wildlife species.

In the United States, wetlands cover 5% of the contiguous lower 48 states, of these wetlands, 95% of them are freshwater wetlands (Dahl, 2011). Found along the United States-Canadian border is the Great Lakes which are the largest source of unfrozen freshwater in the world (Herdendorf, 2004, Larson & Schaetzl, 2001). Within the United States, there are an estimated 2,892 coastal freshwater wetlands along the Great Lakes coastline; these wetlands have a total area of 252,054 ha (Sierzen et al., 2012). These Great Lakes Coastal Wetlands (GLCWs) are essential to the health of the Great Lakes basin and provide a unique opportunity for research on how to understand, protect and manage coastal wetlands. In the past, these coastal wetlands have been filled for development such as for ports or agriculture, and dredged for transportation along waterways (Albert, 2003).

Within the United States, there was a loss of 53% of wetland area across the conterminous 48 states and it has been found that higher population densities of human populations lower the density and fragmentation of wetland habitats (Gibbs, 2002). In the Great Lakes region over 59% of wetlands have been lost since European settlement, with areas reaching near complete loss of previously established wetlands in the lower lakes (Gibbs, 2000, Midwood et al., 2012, Uzarski et al., 2009). This loss underscores the importance of maintaining the remaining GLCWs and the ecosystem services they deliver.

The GLCWs are sites of high vegetative diversity and provide support to wildlife including wildfowl and other bird species through shelter and food resources (Clarkson, 2017, Midwood et al., 2012, Quinn, 2002, U.S. EPA, 2002). These wetlands offer protection from predation and wave energy allowing them to support 80% of the fish species in the Great Lakes at some portion of their life cycle; they are particularly beneficial as breeding, spawning, and nursing grounds (Chow-Fraser & Albert, 1999, Cvetkovic & Chow-Fraser, 2011, Kost et al. et al., 2007). In addition to supporting wildlife, GLCWs facilitate the uptake of excess nutrients and improve water quality through filtration (Keddy et al., 2009). This is important as there are often pollutants from roadways and run-off from agricultural land use in the Great Lakes region. The vegetation of GLCWs not only improves water quality but contributes to stabilizing coastlines by buffering wave energy and holding sediment in place with their root systems (Midwood et al., 2012, Quinn, 2002). The ecosystem services provided by the wetlands surrounding the Great Lakes are at increased risk not only due to anthropogenic disturbance, but they also contend with the impact of invasive species and climate change. Invasive vegetation jeopardizes GLCWs because many of the ecosystem services of these wetlands are dependent on the adaptations of native vegetation. Invasive species such as narrow-leaved cattail (*Typha angustifolia*) and common reed (*Phragmites australis*), which grows quickly and can reproduce asexually (Bansal et al., 2019, Wilcox, 2012), can outcompete native vegetation and threaten the diversity of GLCWs (Tulbure et al., 2007). The tightly packed stands of reed can degrade water flow regulation and energy buffering services given by coastal wetlands. For GLCWs, climate change can enhance this issue by disrupting cycles of natural water level fluctuation.

In the Great Lakes, water levels fluctuate on episodic, seasonal, and interannual scales, and GLCWs rely on these changes to maintain healthy vegetation extents (Wilcox et al., 2007, Mortsch et al., 2008). Seiches and storm waves characterize episodic changes in water levels while seasonal fluctuations follow a general pattern of higher lake levels in the spring and summer months with lower water levels at other times of the year (Wilcox et al., 2007). Interannual water cycles happen over multiple years and are characterized by long-term periods of low or high-water levels. Climate change has altered these cycles and may continue to do so. Quinn (2002) found that there has been a significant change in the timing of the seasonal cycle of high and low water levels compared to patterns of the 19th and 20th centuries; this shift was found in all the Great Lakes except for Lake Ontario. For example, in the past 100 years, the seasonal cycle of Lake Huron-Michigan has shifted with spring water levels being reached two months later than historically recorded (Quinn, 2002). Climate change can also alter the duration and magnitude of interannual water level fluctuations (Quinn, 2002). GLCWs rely on interannual water level fluctuations for stability and maintenance (Cvetkovic & Chow-Fraser, 2011, Mortsch, 2008, U.S. EPA, 2002).

The vegetation extents of healthy wetlands tend to retreat during high water levels and then recover lost area during low water levels; regular fluctuations in water levels promote greater diversity in the wetland vegetation assemblages (Mortsch, 1998, Wilcox, 2008). During low water levels, wetlands will expand if uninhibited by natural or human barriers (Freieswyk and Zedler, 2007). As the water level drops and sediment is exposed, vegetation grows from wetland seedbanks, shifting the wetland extent lakeward along the newly established soil moisture gradient (Mortsch, 1998, Wilcox & Nichols, 2008). This increases the area of available habitat and the wetland’s capacity to provide other ecosystem services. Prolonged periods of low water levels can begin to diminish wetland diversity as less hydrophytic vegetation such as wood vegetation, shrubs and grasses become dominant.

This is countered by cycles of high-water levels, during these periods’ wetland extents shift landward and the GLCWS experience a “cleaning” period, meaning any excess overgrowth that begins to limit the biodiversity of the wetland will be drowned allowing room for new growth (Mortsch, 1998). This change not only causes a retreat of the basinward wetland extent, but also shifts the soil moisture gradient landward through increased inundation (Mortsch, 1998, U.S. EPA, 2002). The inundation of the upper marsh facilitates the die-off of the woody vegetation that is established during low water levels (Mortsch, 1998, Wilcox & Nichols, 2008). During these periods emergent and submergent vegetation establish in the newly cleared areas that previously would not have supported those assemblages (Keddy & Reznicek, 1986, U.S. EPA, 2002). Emergent vegetation that was along the lakeward extent of GLCWs tends to die off due to increased water levels as oxygen availability diminishes, the space converts to open water which exposes the lakebed sediments and increases the possibility of mobilization and transport of sediment.

The processes that drive changes to wetland vegetation extents and sediment dynamics rely heavily on the dynamic nature of Great Lakes water levels. But climate change continues to disturb seasonal water cycles and appears to induce rapid and extreme fluctuations in water levels that can impact GLCWs structure and functions on long-term scales (Mortsch, 1998, Quinn, 2002, Wilcox, 2007). The Great Lakes experienced a prolonged period of low water levels that ended in 2013 when lake levels began to rise again. During the following time record, high water levels were reached at the peak of the cycle in 2020. These uncharacteristically high-water levels have largely been attributed to increased run-off in the spring and increased over-lake precipitation (Gronewold et al., 2016). Studies documenting the relationship between water level and wetland vegetation are extensive (Keddy &Reznicek 1986, Mortsch et al., 2008, Fay et al., 2016, Frieswyk & Zedler, 2007, Smith et al., 2020, Wilcox & Nichols, 2008, Chow-Fraser, 2005), but examination of sediment dynamics in these coastal systems has not been extensively explored. No studies have explored how GLCW substrate morphology changes in response to water level fluctuations and how this might impact the recovery capacity of these wetlands in the future. Any disruptions that could occur to the coastal wetland morphology during prolonged periods of high-water levels must be understood to accurately forecast future coastal wetland health and extent.

There have been efforts to map changes in coastal wetland geomorphology using bathymetric data, but this has largely been focused on marine coastal saltmarshes (e.g. Gorman et al., 1998 Collin et al., 2010 Fagherazzi et al., 2020, Deb et al. 2022). Large-scale bathymetric mapping of GLCWs only began in the early 2000s with the collection of elevation data using topobathy LIDAR (Reif, 2013). There is ongoing work to improve these methods for collection in coastal wetlands and to generate more frequent datasets but currently, records of geomorphic change in GLCWs are minimal. Wilcox (2007) mentions that morphology is a factor in influencing how water level affects shorelines. Further, Mortsch (1998) describes how a wetlands geomorphic form influences how wetland areal extent will respond to fluctuating water levels, identifying lakebed slope as a key factor. To begin addressing this gap in available data and knowledge of wetland morphodynamics, we present an analysis of vegetation extent and wetland geomorphic change at two GLCWs in response to a nearly decade-long period of high-water levels in Lake Huron.

METHODS

Study Areas

Saginaw Bay. Of the Great Lakes, Lake Huron is the second largest in surface area and the third largest of the lakes by volume (Herdendorf, 2004). Saginaw Bay is located within the southern basin of Lake Huron and is a semi-enclosed basin with the opening to Lake Huron oriented to the northeast. Saginaw Bay was chosen as an ideal area for conducting this study due to the abundance and diversity of coastal wetlands present in the bay. There is a total of 18,000 acres of coastal wetlands in Saginaw Bay making it the largest freshwater coastal wetland system left in the United States (Lynch and Waldron, 1999) The presence of coastal wetlands in the bay is due to its shallow and sheltered nature; these attributes also make changes in water level and wave intensity a concern for the low elevation coastlines. Understanding these processes in Saginaw Bay can provide valuable insight to local communities and a framework for further study of freshwater coastal wetlands. Overall, the bottom of Saginaw Bay is composed primarily of silty clay; however, there is also fine-grained sand that can be found throughout (Herdendorf, 2004). The growing season of Saginaw Bay typically ranges from May-August (Wilcox & Nichols, 2008).

The surrounding land use of Saginaw Bay has become dominantly agricultural, but wetlands can still be found in its vicinities such as shrub wetlands, fens, wet prairies, and marshes including coastal marsh systems. Much of the coastal wetlands surrounding the bay are thick stands of open marsh (Albert, 2003). In areas where there is a buildup of sediment due to the presence of sheltered bays and river deposits, these become much wider swaths of marsh wetlands. However, these marshes vary by level of sheltering and position within the bay which affects what kind of vegetation can establish and survive in the wetland. Wetlands that are open embayments and therefore are exposed directly to wave energy tend to be low in diversity and consist of vegetation that can withstand the exposure to wave action (Albert, 2003). This is reflective of our first research site, Quanicassee State Wildlife Area, which is located in the southern portion of the bay and is unsheltered by sand spits or anthropogenic structures. In addition to studying the impact of increased water levels on the open marsh wetland located in Quanicassee State Wildlife Area, we also wanted to compare the influence on sheltered coastal marsh wetland. The site selected for the sheltered wetland is Wigwam Bay State Wildlife Area. This small embayment is located on the northwestern coast of Saginaw Bay and is protected by sand spits to the north of the study site and has another small sandspit to the south of the site (Kost et al., 2007). Both sites are located within the bay and experience the same subsequent rises and falls in water levels. By choosing two wetlands with different exposure to wave action and variability in vegetation composition we can compare similarities and differences between the responses that each environment has to high-water levels (Figure 1).

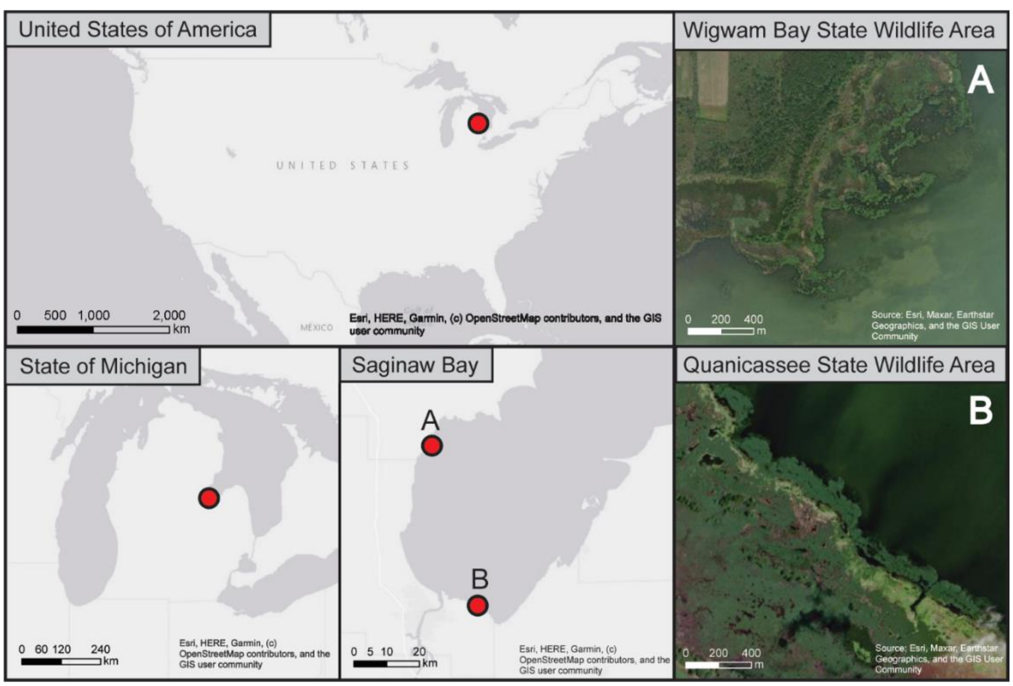

Figure 1: Locations of study sites (A) Wigwam Bay State Wildlife Area (B) Quanicassee State Wildlife Area

Quanicassee. Quanicassee State Wildlife Area is located in Essexville, Michigan (Figure 2). The wildlife area extends to both Bay County and Tuscola County. The study site is a portion of the wildlife area that spans approximately 1km of the Saginaw Bay coastline, research was focused on this section of coastal wetland and the adjacent near-shore area (Figure 2). Our study site contains a lower open marsh that is predominantly emergent vegetation and the beginning of a mudflat containing terrestrial vegetation such as grasses, shrubs, and sparse but present trees at the landward reach of the site. It also captures the adjacent submergent and open water zones at the lakeward extent of the study site. Landward of the research site are mudflats that contain sedge grasses, shrubs, and some sparse woody vegetation. At our study site, there is not a forested upland portion of the wetland though this is present in other areas of the wildlife area. The mudflats landward of the area we studied at Quanicassee are ended by agricultural fields. To the southeast of the site is the Quanicassee River; a boat ramp along this river provided access to the research site and provides sediment output that contributes to the formation of coastal wetland in this area (Kost et al., 2007). During low lake levels, the wetland at Quanicassee expanded past the historically recorded lake shoreline (DNR, 2016). The wetland has maintained some native species, notably the fringing bulrush (*Schoenoplectus tabernaemontani*) present at our research site. Unfortunately, Quanicassee has experienced extensive colonization of *P. australis*, which acts as the dominant vegetation present at our study site. Despite this, the wetlands in this area provide habitat for wildfowl and other bird populations that include species such as mallards, blue-winged teal, and American bittern (Albert, 2003, Cohen, 2020, Kost et al., 2007). They also function as breeding and nursing grounds for fish populations of Lake Huron (Albert, 2003). The wildlife area is used recreationally by local communities for fishing and hunting.

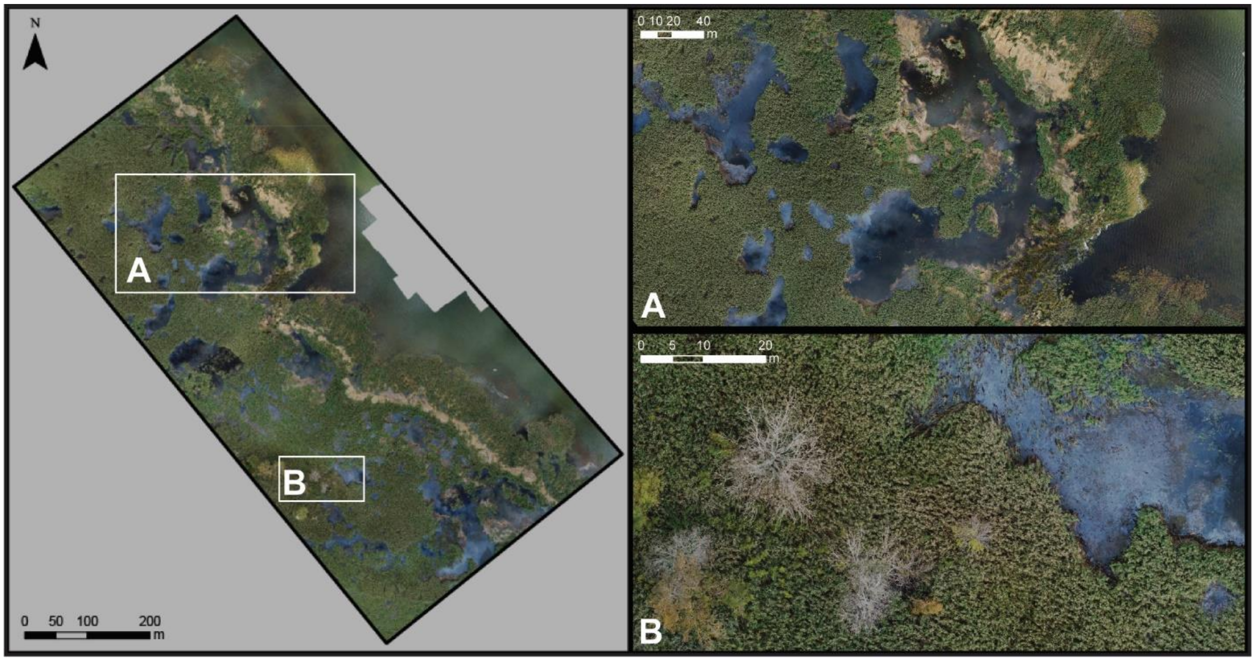

Figure 2: Quanicassee State Wildlife Area 2021 drone imagery site overview (A) West focus (B) South focus (white boxes)

Wigwam Bay. Wigwam Bay State Wildlife Area is located in Standish Michigan in Arenac County (Figure 3). The wildlife area is divided into units that were determined by the Michigan Department of Natural Resources (DNR). Our research site is within the Palmer Road unit which is the Southernmost unit of the wildlife area (DNR, 2015). Inside this unit spans about 1 km parallel to the Saginaw Bay coastline (Figure 3). Like the Quanicassee site research was focused on a section of coastal wetland and the adjacent near-shore area. Wigwam Bay has several smaller sand spit embayment's within the Wildlife Area, these provided enough shelter from wave action and sediment accumulation for the formation of coastal freshwater wetlands (Albert, 2003). To the north of our study sites is Pine River which provided access to the research site. This river also contributes sediment that aids the formation of coastal marsh in this area (Kost et al., 2007). The wetland within Wigwam Bay has poorly drained soils and supports several vegetation communities such as mixed hardwood, mixed conifer, black ash, and cedar swamps. They also support emergent marshes and shrub swamps in areas with streams and small embayments (Slaughter and Sanders 2016, Albert and Comer, 2008). Our site captures a section of emergent marsh. In addition to the emergent marsh, Wigwam Bay also has adjacent submergent vegetation lakeward of the wetland, and a small portion of shrub wetland with some woody vegetation is located at the southwest portion of the site (Figure 3B). Diversity at this site increases during lower water levels when less hydrophytic plants can establish in the shallower water or newly revealed sediment that was previously inundated. During high lake levels, diversity has lowered and from field observations, it appears that the highest volume of vegetation is from the *T. augustifolia* and *P. australis* within the study areas. However, Wigwam Bay still maintains native vegetation that is better adapted to increases in energy caused by higher water levels; prominent species are native bulrush (*Schoenoplectus tabernaemontani*), and pondweed (*Potamogeton spp*.), pickerelweed (*Pontederia cordata*) among others. The site has minimal anthropogenic disturbance beyond recreational fishing and hunting within the wildlife area.

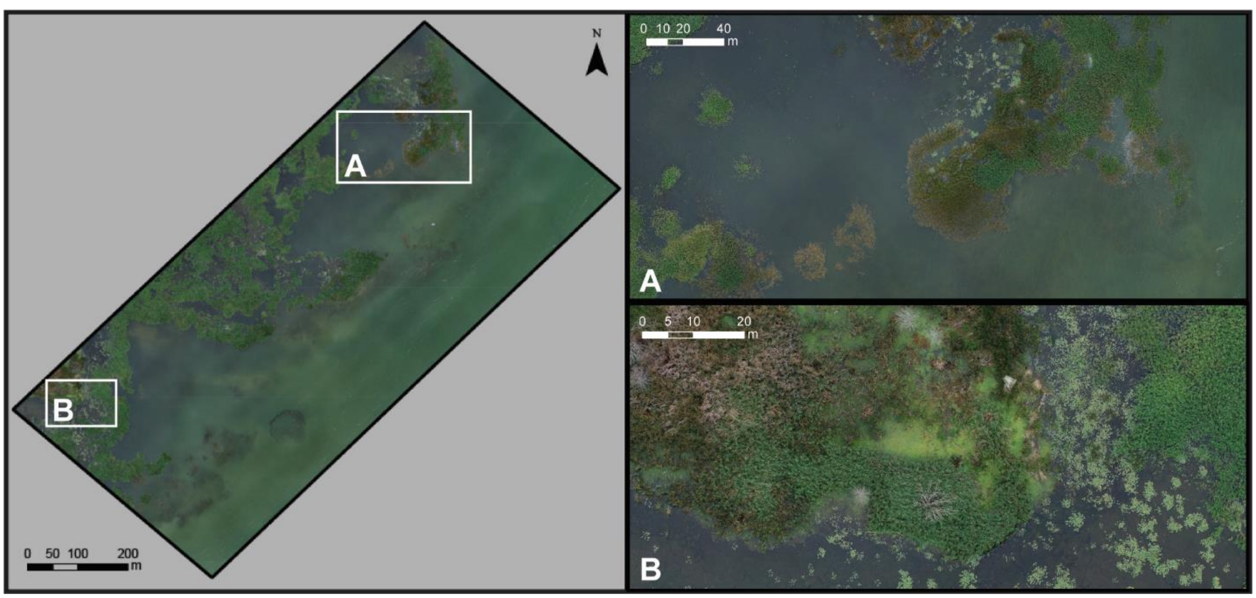

Figure 3: Wigwam Bay State Wildlife Area 2021 drone imagery site overview (A) North focus (B) Southwest focus (white boxes)

Water Level

Water level data were acquired from NOAA Tides and Currents (https://tidesandcurrents.noaa.gov/) to document the period of rising lake levels from 2013 through 2020. Specifically, hourly water level data (m; IGLD85) were downloaded for the period from January 1, 2012, to December 31, 2021, from the Essexville, MI station (ID: 9075035). This station is located about 12.5 km west of the Quanicassee site and 35.5 km southeast of the Wigwam Bay site. Site-specific hourly wind and wave estimates were also acquired for this same timeframe using model output from NOAA's Great Lakes Coastal Forecasting System. Water level, wind, and wave data were tabulated in Microsoft Excel and then plotted in Golden Software's Grapher program for visualization of hydrodynamic trends.

Water level data was plotted from 2012 to 2021 and regression of the data over time was calculated from 2013 to 2020 to capture any trends in water level change in Saginaw Bay during the rising phase of lake level. The resulting water level graph was analyzed visually, and a linear regression was used to find any notable trends on a seasonal level to observe if they were a contributing factor to sediment dynamics at either research site. The water level in 2020 was also compared to the recorded record high water level of Lake Huron which was 177.5 meters (NAVD88). Water level data is reflective of the entirety of the southern basin of Saginaw Bay and is not differentiated between the two wetland sites. Wave data were assessed for each individual site; these data were plotted for visual analysis, additionally, a linear regression was run for wave data from both sites. For the wave data, a Rosette plot was constructed for wave direction. Wave direction data were recorded in degrees; 0 degrees indicated waves are going toward the North and 90 degrees indicates waves going toward the East. Histogram bins for direction data were 45 degrees starting at 0 degrees and ending at 360 degrees. The data were as rosette diagrams to determine which direct receive the greatest amount of wave energy. This includes overall wave directions for each research site and the direction of storm waves each site experiences. In addition to graphing directional wave data, the recorded significant wave heights were graphed, and regression over time was run on the resulting plots for both Quanicassee and Wigwam Bay in order to observe any trends in significant wave height from 2012 to 2021.

To further assess high energy episodic changes in water level, storm waves were isolated from the recorded significant wave height data. Wave observations that exceed 2m are defined as storm waves. Storm wave data was split from the other significant wave height observations for each site and processed in a separate excel document to identify the duration, magnitude, directionality, and time of occurrence of these storm waves. To determine how these storm waves may impact the coastal wetlands, they were compiled into storm wave events. If the recorded storm wave observations occur within a 24-hour period from each other they are defined as a storm wave event. For example, if storm waves were recorded for 4 hours in a day and then two hours later there is another storm wave, those 5 hours would be considered a storm wave event. By doing this we can determine the number of storm events at each site and the duration of these events. A histogram was then constructed in Grapher of the storm waves, with bins being the year that the events occurred, this allows us to determine if storm wave events follow any similar trends to water level changes in Saginaw Bay. Storm wave data was analyzed by month to determine when storm waves occur most frequently throughout a given year. As stated previously, dominant directions of storm waves were graphed to find how these waves approach each of the coastal wetlands at Wigwam Bay and Quanicassee. These analyses construct a framework for how different types of water level changes may impact vegetation extent and sediment dynamics in freshwater coastal wetlands.

Aerial Imagery

Previous research has suggested that changes in water level can impact the structure and aerial extent of coastal wetlands (Mortsch, 1998). To document changes in the spatial extent of the wetlands at Wigwam Bay and Quanicassee resulting from the period of rising lake level (2013-2020), imagery from 2012, 2016, and 2021 were digitized using ESRI ArcMap. These images reflect transitioning from low to intermediate to high water levels. For the years 2012 and 2016, 4-band NAIP imagery was acquired from the United States Geological Survey (USGS) Earth Explorer for each site. The 4-band imagery allows the near-infrared (NIR) band to be utilized to distinguish between wetland vegetation and open water when digitizing. To display the NIR of Wigwam and Quanicassee the channels of the imagery were adjusted in ArcMap so that the red channel displayed band 4, green displayed band 1, and blue displayed band 2. This resulted in the imagery displaying the wetland vegetation as red and due to the shallow nature of coastal wetlands the open water was displayed as blue. The boundaries between wetland vegetation (red) and open water (blue) were then digitized as a shapefile for each of the study sites and time periods.

The 2021 aerial imagery for both sites was collected using a quadcopter drone (DJI Phantom 4 Pro) during field excursions on October 6th at Quanicassee State Wildlife area and on September 13th at Wigwam Bay State Wildlife Area. DJI Ground Station Pro was utilized to build and execute flight plans for each study site with the following standard settings: 70% front and side overlap, nadir (downward-facing) camera angle, and a ground sampling distance (GSD) of 2.7 cm/pixel. This GSD was achieved with a flight height of 100m for Quanicassee and Wigwam Bay. The resulting RGB imagery was imported into Agisoft Metashape Professional Edition for processing into orthomosaic images using structure-from-motion photogrammetry

(Turner et al., 2016). These orthomosaic images were exported from Agisoft at 5cm accuracy resolution and imported into ArcMap for digitization of the wetland extent. The 2021 images are only RGB, thus no NIR band was available for digitizing the wetland extent. However, given the high resolution and overall image clarity in these drone-acquired orthomosaic images, digitization of the wetland boundary could be achieved with a high degree of confidence.

The digitized extents were analyzed for pattern and magnitude of vegetation extent change at both wetland sites. Through visual analysis areas of high magnitudes of change were identified between 2012, 2016, and 2021 for each site. Digitizations also allowed change to be differentiated between extent retreat, which is characterized by a decrease in the distance the outer edge of the vegetation extent reached into Saginaw Bay, and a patchier loss of vegetation that is also a result of increased inundation and vegetation die-off. In addition to visual analysis, digitized wetland extents were used to calculate the amount of area lost by the square meter and the percentage of loss from year to year at each site. Area values were calculated in ArcGIS and change was calculated by subtracting the area values of 2012 from 2016, 2016 from 2021, and 2012 from 2021. The 2012 imagery was used as the "base value" that was used to calculate the percentage of wetland lost. Area values of loss between years were divided by the 2012 site extent area for each respective site.

Topobathymetry

NOAA topobathy LIDAR data, as well as RTK-GPS and echosounder surveys, were gathered to evaluate changes in wetland morphology throughout rising lake levels. Topobathy LIDAR data were downloaded from the NOAA Data Access Viewer for Wigwam Bay and Quanicassee. The data set utilized was the 2013-2015 USACE NCMP (U.S. Army Corps of Engineers National Coastal Mapping Program). This data set was collected in 2013 and was processed from 2013 to 2015. The data acquired from NOAA were Bathymetric Lidar Points in the format of ASCII X, Y, Z point files. The projection used for this download was UTM Zone 17 in the range 084W-078W. The horizontal and vertical datums used are NAD83 and NAVD88 respectively, both datums were downloaded in the units of meters and used GEOID 12B. These data were collected by the Joint Airborne Lidar Bathymetry Technical Center of Expertise using a Coastal Zone Mapping and Imaging Lidar (CZMIL) system. CZMIL uses a lidar sensor, a hyperspectral imager, and a digital camera to map bathymetry and topography (OMC Partner,s 2022). To convert this downloaded data into a usable form for this study, the files were gridded using Surfer. The gridding process was done using the natural neighbor method and a grid spacing of 30m was used to construct a digital elevation model (DEM) from the data. The acquired 2013 data had limited data points available immediately lakeward of the wetland sites due to the inability of the LiDAR to acquire accurate bathymetric data in highly turbid water. To address this shape files of the data limits were constructed at 30m gridding for Quanicassee and Wigwam Bay. These shapefiles indicate the limit of data that could be used for reliable comparison to the 2021 data sets collected at the study sites.

The 2021 topobathymetric data were collected during multiple field campaigns in the summer of 2021 using a 20-foot research vessel, a kayak, and wading surveys. Multiple survey methods were needed given the complexity of these wetland sites and the need to safely access the sites with minimal vegetation disturbance. Surveys for Wigwam Bay were conducted on June 28, 2021 (boat), July 22, 2021 (wading), and July 28, 2021 (kayak), and surveys were collected at Quanicassee on June 16, 2021 (boat) and August 3, 2021 (wading). Horizontal and vertical positioning for all survey methods were acquired using a Trimble R10-2 GNSS antenna. Real-time position corrections were acquired using network corrections from both the Michigan

Department of Transportation's Continuously Operating Reference Station (MDOT CORS) and Trimble's satellite correction service RTX; vertical errors for both data sets on average were 2.5cm and horizontal errors for Quanicassee and Wigwam Bay were 1.6cm and 1.4cm respectively. For the wading surveys, the RTK-GPS antenna was attached to a 4.22m pole. By mounting the antenna to the top of this pole it ensures that the signal of the system is not disrupted by the tall wetland vegetation. The wading survey started at the furthest lakeward point that was wade-able and proceeded landward into the wetland vegetation. An elevation data point was collected every 5 paces along the transect and transects were approximately spaced 100m apart at each site. This spacing captured variation in elevation change that occurs at different portions of the wetland in our study sites and represents variable distances that vegetation extends lakeward.

For both the boat and kayak surveys, the RTK-GPS antenna was affixed to the top of the pole and a Seafloor Systems SonarMite single-beam echosounder was attached to the bottom of the pole. For the boating and kayak surveys, the pole used was 1.83m and 0.91m tall respectively. The echosounder attached to the poles gathers depth data below the transducer and when coupled with the RTK-GPS data yields high accuracy measurements of lakebed elevation. Sonar data were collected approximately every 2s along shore normal and shore parallel transects, the system will not store any data that is gathered when there is excessive movement of the sonar (greater than 5 cm). All RTK-GPS data were collected in UTM Zone 17N and were vertically referenced to NAVD88 Geoid 12B.

Topobathymetric data were collected for each survey at Wigwam Bay and Quanicassee, the resulting data sets were XYZ data points reflecting the height of the mounted transducer. These datasets were compiled and cleaned to remove any inaccurate data points in Microsoft

Excel: this includes “blank” points in which an accurate measurement was not collected or any elevation values below 0.5m as shallow depths and movement of the watercraft could result in skewed results. After the removal of those points, the elevation data were corrected to account for antenna height and water depth below the transducer to calculate the actual elevation of the lakebed. The water depth correction only applies to the boating and kayaking surveys, as during wading surveys the transducer was not used, and measurements were taken directly on the lakebed. The actual elevation of the wetland is found by subtracting depth, which is the distance from the sonar transducer to the wetland sediment, from the elevation of the transducer. In order to correct for the antenna heights, 1.22m were subtracted from the wading survey data, 0.17m added to the boating surveys, and 1.09m added to the kayak surveys. The adjusted data was then brought into Surfer for additional cleaning and processing. Any additional spurious points were removed from the data sets. This was determined by plotting the depth data in Surfer and removing any points that had a difference of 0.1m or greater from a neighboring data point. The frequency at which the data were collected (2s) makes it unlikely that two consecutive points would have variability higher than that. These erroneous points could be generated by submerged vegetation or other underwater artifacts. For the Quanicassee and Wigwam Bay sites about 11.6% and 4.5% of the points were removed, respectively. The final compiled boat, kayak, and wading survey data sets collected for each site in 2021 were processed and analyzed for comparison to the 2013 topobathymetric data. Using Surfer, the 2021 datasets were gridded using a natural neighbor algorithm at 30m to construct the DEMs of Quanicassee and Wigwam Bay.

DOD Maps

To quantify the changes in morphology that occurred at each site the 2021 and 2013 data sets were subtracted from each other within Surfer to construct DEM of Difference (DOD) maps. These maps depict negative values (shown in red tones) as decreased sediment elevation and positive values (shown in blue tones) are areas of increased sediment elevation. To account for any instrumental error that may have occurred during the collection process, a net neutral range was determined. Any change that occurred, positive or negative, within this range was not considered reliable enough to represent a change in sediment elevation. Change that occurred within the highest vertical accuracy of the two data sets was considered neutral change. The average vertical error for the 2021 Wigwam Bay and Quanicassee data points were 0.0247m and 0.0249m respectively, however, the 2013 data had a vertical accuracy of 0.196m so the neutral range for both sites was set as +/- 0.2m. Therefore, if any changes were detected in the DOD below 0.2m we cannot be confident these are real and consider them areas with no change.

After the DOD maps were constructed for each site, they were then “blanked” in Surfer using the shapefiles that present the spatial limits of the 2013 data. Both files were loaded into Surfer into the same map. The function Assign No Data was then utilized to remove any data from the subtraction maps that did not overlap with the 2013 topobathy data, referred to as blanking. This created the final DOD maps used for analysis and discussion. This same process was applied to the original extracted DEMs of the 2021 data for each site to construct transects that show the elevation profiles at each site.

Profiles

In addition to showing overall morphology change at Wigwam Bay and Quanicassee through the DOD maps, it is important to evaluate how the elevation gradients across the wetlands changed during the period of rising lake level. Coastal wetlands follow a soil moisture gradient that determines the structure and zonation of the wetland vegetation (Frieswyk and Zedler 2007, Gathman et al. 2005). This gradient is not only influenced by water level but can be impacted by changes in wetland morphology. To document changes in the wetlands gradients that occurred during the high-water levels, transects were sliced across the DEMs at each site using Surfer.

There were 6 transects sliced at a spacing of approximately 100 meters at Wigwam Bay and Quanicassee, this spacing was then adjusted to capture areas of sufficient overlap of the 2013 and 2021 data (Figure 4). Transects were drawn from the most landward extent of the 2021 contour map to the most lakeward extent of the 2021 map. When the transects are placed over both the 2021 and 2013 contour maps, they provide elevation profiles of each transect of 2013 and 2021 for both sites. The elevation profiles of each transect for both DEM maps were plotted using Grapher and they demonstrate the changes to the profile structures that have occurred to the outer edge of each wetland over the period of high lake level. Profiles were also drawn over the un-blanked 2021 DEM maps to demonstrate the difference between available 2013 topobathymetric data and the 2021 topobathy data collected using field measurements that generate fuller coverage. To better understand the magnitude of change that occurred across each research sight, the average loss of elevation at each site was calculated. Additionally, the average maximum loss in elevation across the transects for each site was found in order to present how much change to the wetland profiles occurred from 2013 to 2021.

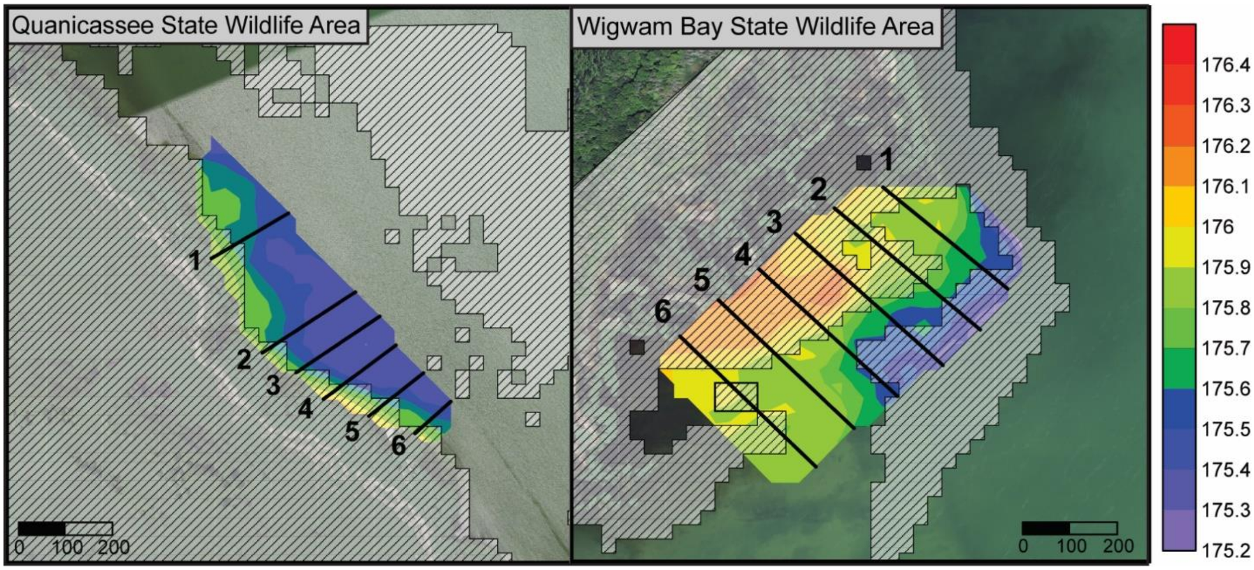

Figure 4: Transect placement and comparison of 2013 data extent to contour maps (m; NAVD88) of collected 2021 topobathymetric data for profile analysis at both the Quanicassee (Left) and Wigwam Bay (Right) study sites

RESULTS

Water Level

The water level data acquired from 2012 to 2021 document the lake level conditions over the interannual cycle of high-water levels at each research site. Lake Huron water levels began to rise in 2013 and reached a peak in 2020 (Figure 5). During this time the rate of lake level rise was 0.141 m/yr., which, as previous studies on Lake Michigan have documented, is more rapid than other periods with similar magnitudes of peak lake level (Theuerkauf & Braun, 2021). Similar seasonal patterns of annual maxima in the summer (generally July and August) and minima in the winter (generally January and February) are observed in the water level record. Record monthly water levels were set in January through August of 2020, though the record high lake level set in October of 1986 at 177.5m (NAVD88) was never exceeded.

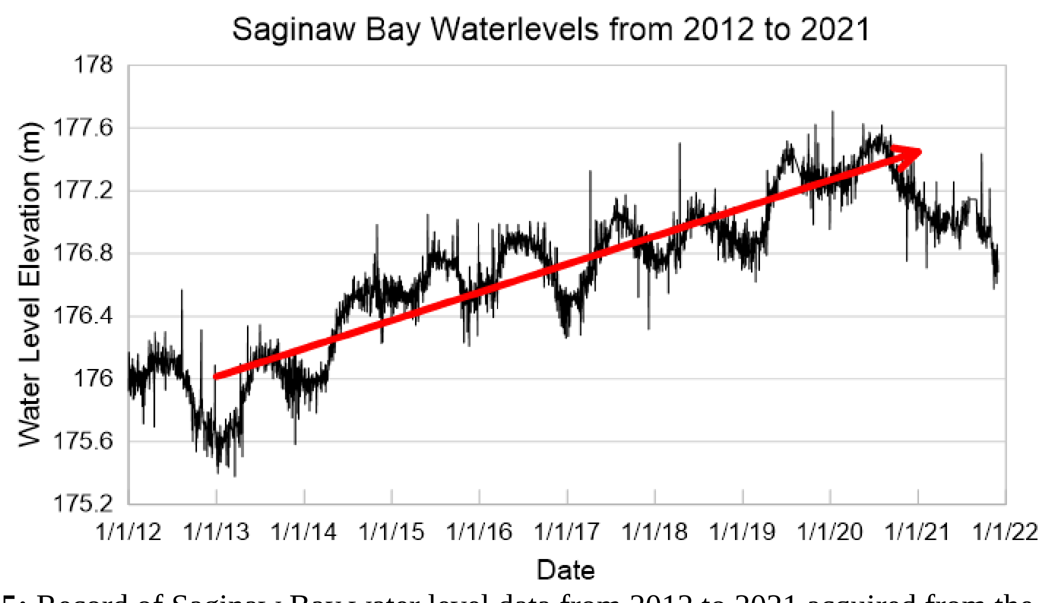

Figure 5: Record of Saginaw Bay water level data from 2012 to 2021 acquired from the NOAA station in Essexville, MI (ID: 9075035). The regression line is denoted in red

Unlike the continual rise in lake level observed, there were no consistent trends in significant wave height at either site. Although the trend in wave heights is not increasing or decreasing, there were numerous storm wave events recorded at each site. At the Quanicassee site, based on the GLCFS model output, the average significant wave height was 0.27 m, and the maximum recorded wave height was 3.7 m. The dominant wave direction at Quanicassee was from the North and accounted for 43.4% of waves that impact this site. Waves approaching Quanicassee from all other directions combined composed the remaining 56.6% (Figure 7). At the Wigwam Bay site, the average significant wave height was 0.273 m, and the maximum recorded wave height was 2.83 m. At this site, the dominant portion of waves was from the South. These waves accounted for 36.6% of the waves that impact Wigwam Bay, with the combination of waves from other directions comprising the remaining 63.4% (Figure 6).

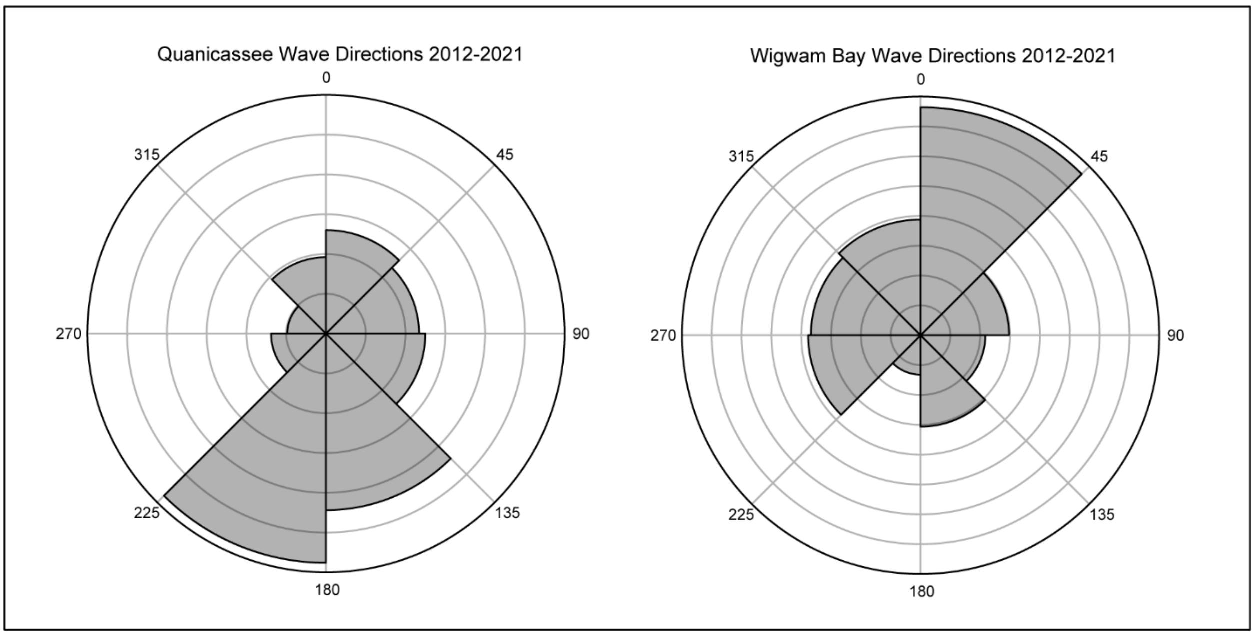

Figure 6: Rosette plots depicting which directions that waves moved toward Quanicassee State Wildlife Area (left) and Wigwam Bay State Wildlife Area (right)

Storm wave events at both sites all occur within the dominant wave directions for each respective site and have similar ranges of duration. At Quanicassee storm wave events ranged from 1 to 30 hours long and at Wigwam Bay they ranged from 1 to 31 hours long. There is variation in the number of storm wave events each site experienced from 2012 to 2021. At the Quanicassee site, there were 26 recorded storm wave events, and the site experienced a range of 1-4 storm wave events per year during the study period (Figure 7). At the Wigwam Bay site, 10 storm wave events occurred from 2012 to 2021. Unlike Quanicassee, Wigwam Bay did not experience strong wave events every year; there were no recorded storm wave events at Wigwam Bay in 2013, 2014, 2017, and 2019. Of the years when storm wave events did occur, events ranged from 1-4 in a given year. However, this site did experience the longest storm wave event which occurred in April of 2018. This uncharacteristic event lasted 31 hours and is the only storm wave event to occur outside of the range of months from August to December at Wigwam Bay during the period. Overall, the variation in storm wave events provides context for changes that occur at the two wetland sites that experience variable exposure to wave energy. Despite the contextual information provided by the storm wave events, there is no evidence in this analysis to suggest an impact on long-term lakebed morphology from storm-related episodic water level fluctuation. The only clear trend observed is in the interannual water level change which demonstrates an increasing trend in water levels in Saginaw Bay from 2012 to 2021.

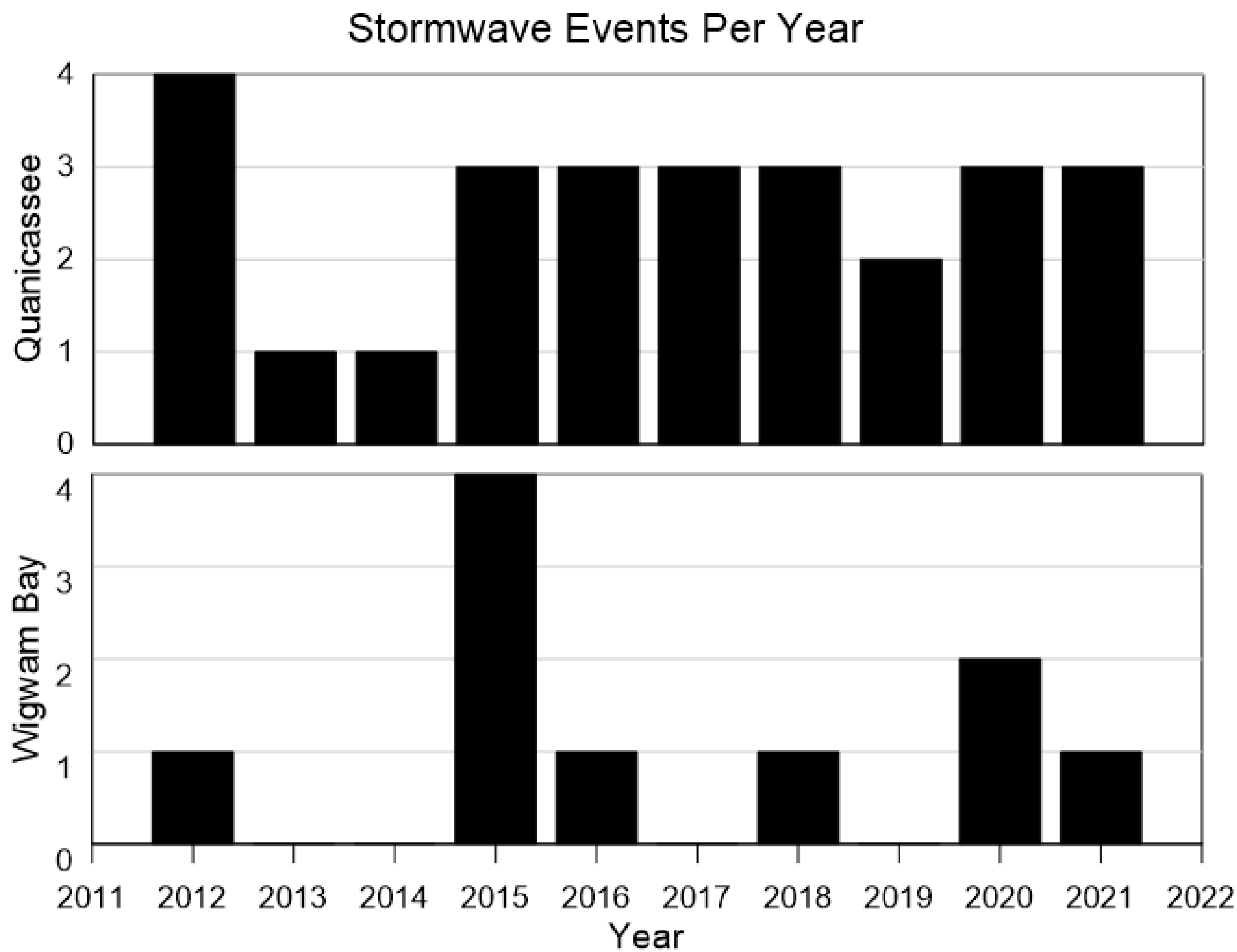

Figure 7: The number of storm wave events that occurred each year at Quanicassee (top) and Wigwam Bay (bottom) from 2012 to 2021 (NOAA GLERL)

Vegetation Extents and Aerial Imagery

The response of wetland vegetation extent to rising lake levels varied between the two study sites. At the Quanicassee site from 2012 to 2021, there was little to no retreat of the outer lakeward extent of the wetland vegetation (Figure 9). From 2012 to 2016 the wetland gained area; the total area of the site increased by 7,279 m^2. Following this increase was a loss of 8,042m^2 from the vegetation extent that occurred from 2016 to 2021. At the Quanicassee site, there has been an overall loss of 2.25% of the wetland area based on the lakeward extent of the wetland vegetation. At Quanicassee there has been extensive colonization of the invasive common reed which could be contributing to the maintenance of the wetland's lakeward extent. However, this loss likely is an underestimate based on the 2021 aerial imagery collected. This imagery shows increased inundation within the wetland complex and subsequent vegetation die-off and wetland fragmentation (Figure 2). In addition, the historical imagery used in the analysis was collected at a lower resolution than that of the drone imagery collected. Therefore, digitizations in 2012 and 2016 were done conservatively. Because of this, there is likely emergent vegetation that was not included in the digitization of those extents if inundation made vegetation presence indiscernible or uncertain in the NIR imagery. In addition to increased inundation within the wetland complex and the threat of invasive vegetation, pollutants can be seen in the water within the wetland in the 2021 drone imagery (Figure 2A, 2B), this is likely due to runoff from the agricultural fields located adjacent to the wetland. Due to the resolution of the 2012 and 2016 imagery, it is unclear if pollutants were present in those extents, but both years do not display the large gaps in vegetation seen in the 2021 wetland.

At the Wigwam Bay site, there is more retreat of the lakeward extent of the wetland vegetation found at Quanicassee. There was not a gain in vegetation extent from 2012 to 2016 as was seen at Quanicassee. By 2016 the wetland area of the Wigwam Bay site had decreased by 39,825 m^2. From 2012 to 2021 there was a net loss of 58,215m^2 (20.26%) from the aerial wetland vegetation extent (Figure 9). The same potential for loss measurements to be an underestimate applies to this site as well. Vegetation loss can be seen in the Northern portion of the wetland with some areas changing in lakeward extent by up to 150m from 2012 to 2021 (Figure 9C). There is also a loss of vegetation due to inundation and drowning in the Southwest portion of the site (Figure 3B). In that part of the wetland, there is some retreat of lakeward vegetation extent and patchier stands of wetland vegetation in 2021 as compared to 2012 (Figure

3A). Wigwam Bay also shows inundation of vegetation within the wetland complex, similar to what was seen at Quanicassee. Both Wigwam Bay and Quanicassee experienced a decrease in wetland vegetation area during the most recent cycle of high-water levels; the characteristics and locations of both sites resulted in variability in this loss.

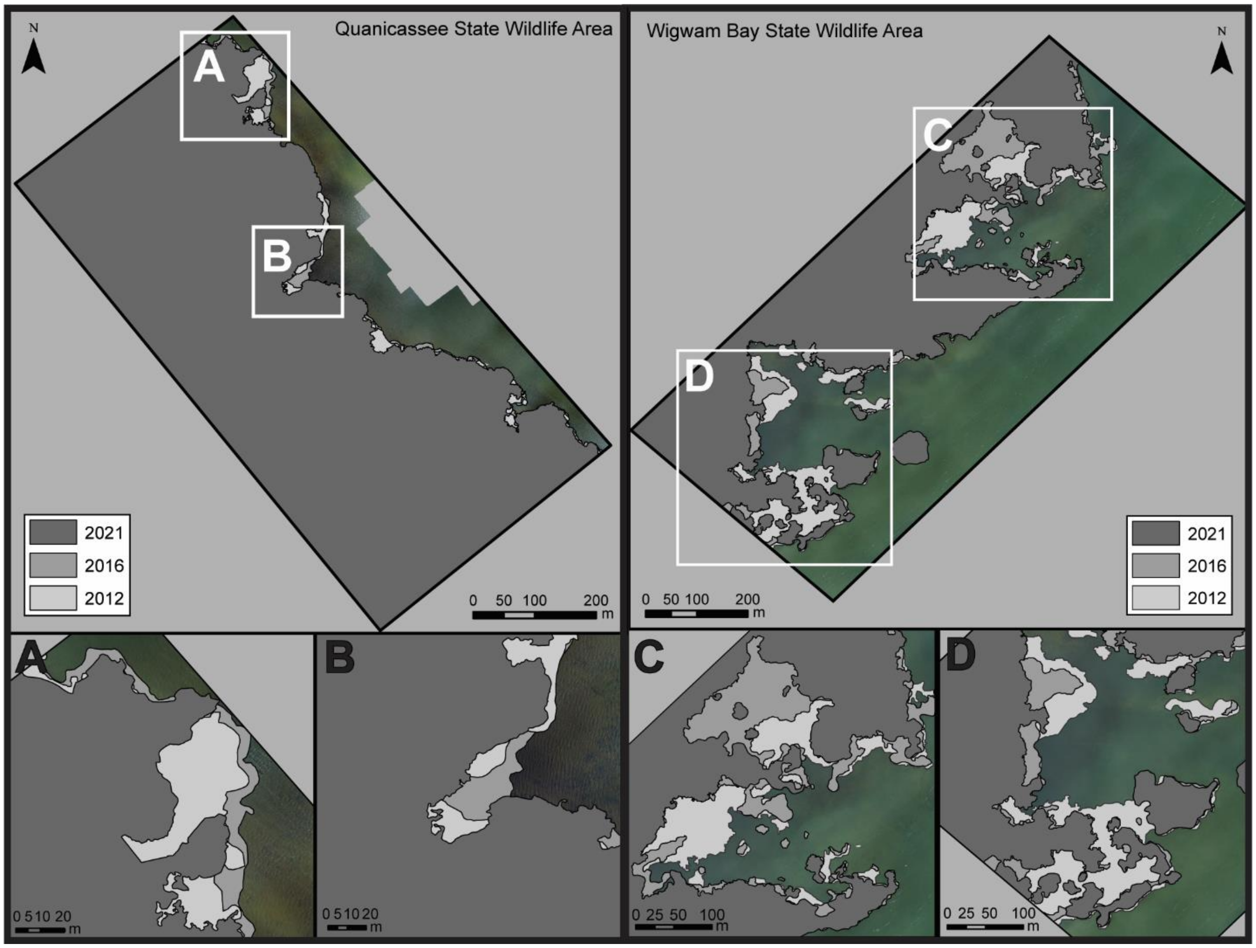

Figure 8: Digitized vegetation extents for 2012, 2016, and 2021 of Quanicassee State Wildlife Area (left) and Wigwam Bay State Wildlife Area (right) Lower panels (A-D) display closer views of the change that occurred between these years

Topobathymetry

DOD Maps

The derived DEM of Difference (DOD) maps for both Quanicassee and Wigwam Bay show a net loss of lakebed elevation indicating that erosion has occurred at both sites from 2013 to 2021 (Figure 10). This loss occurs at the most lakeward extent of the wetland vegetation and is spatially variable throughout each site. At the Quanicassee site, there is little that can be derived from the DOD map. After blanking the 2021 data to remove any points that were outside of the spatial limit of the 2013 data points, only a small portion of overlap remained of the DOD maps (Figure 10). The Quanicassee DOD map shows that there was a loss of sediment along the outer edge of the vegetation extent, however, it is unclear if additional loss or deposition occurred in the areas where there was not sufficient data overlap between the two-time steps. It is impossible with the available Quanicassee data to say anything about the fate of the material eroded from the marsh edge.

At the Wigwam Bay site, there was more overlap between the 2013 and 2021 data points resulting in a larger DOD map. The change at this site is primarily a loss of lakebed elevation along the outer vegetation extent of the wetland (Figure 9). The most lakeward portion of the DOD map shows that no net change occurred outside of the neutral range from 2013 to 2021, so the change is concentrated directly lakeward of the wetland and diminished further from the coastline. At Wigwam Bay there are also portions of the map that directly overlapped with established vegetation that did not experience any notable net change. Changes to the geomorphology caused by the loss of sediment can be seen in the profiles for each site.

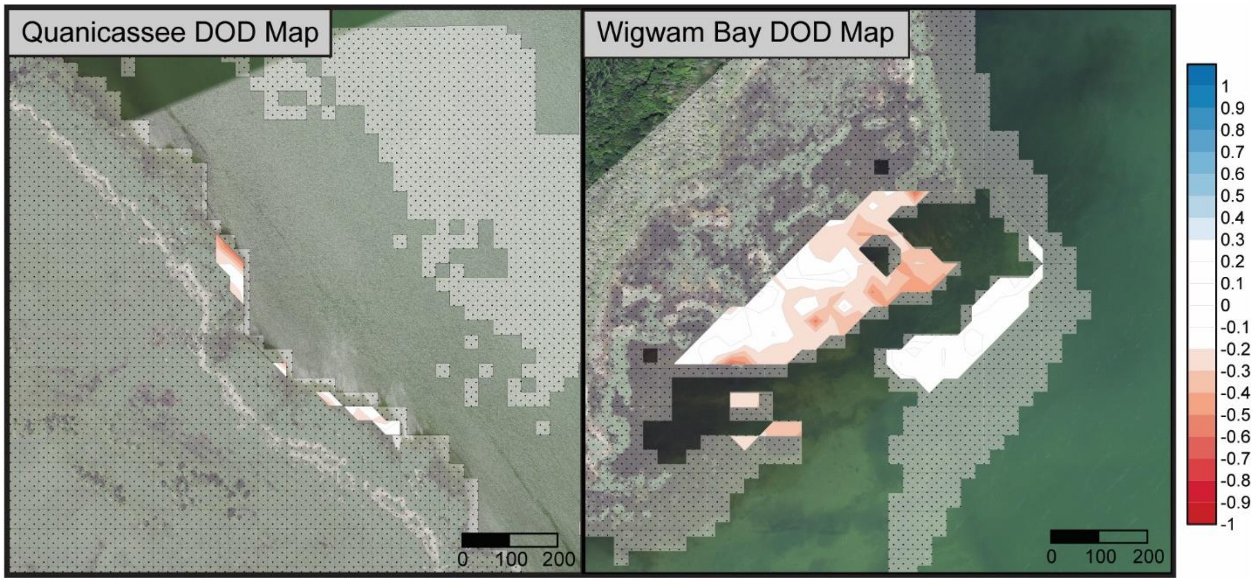

Figure 9: Derived DEM of Difference (DOD) maps (m; NAVD88) over the extent of available 2013 topobathymetric data of the Quanicassee (left) and the Wigwam Bay (right) study sites

Profiles

Changes in morphology were observed from the profiles derived from shore-normal transects at each study site (Figure 4). No sediment deposition outside of the neutral range was recorded in any of the transects. The dominant trend in change for all profiles across each site is a lowering of lakebed elevation which supports the erosion documented in the DOD maps. At the Quanicassee study site, there are various changes to the lakebed elevation and some slight structural changes across the profiles (Figure 11). The 2013 profiles across the site show a gentle decreasing slope. In profiles one and two of the Quanicassee site the elevation was rising lakeward along the 2013 profiles, however, in 2021, they demonstrate a clear decreasing slope in the same area. In profiles four, five, and six the profiles lowered but also demonstrated a change in morphology. At Quanicassee there was an average loss of 0.293m across all the profiles, the average maximum loss was 0.422m. The most loss was seen at profile 1 at the western end of the study site, loss at this profile reached 0.6m (Table 1).

Quanicassee	Profile 1	Profile 2	Profile 3	Profile 4	Profile 5	Profile 6	Average
Average Elevation Change	-0.376	-0.361	-0.352	-0.167	-0.209	-0.118	-0.264
Maximum Loss	-0.597	-0.484	-0.431	-0.295	-0.262	-0.275	-0.391
Maximum Gain	-0.191	-0.180	-0.265	-0.085	-0.126	-0.072	-0.153

Table 1: Elevation change data (m) derived from Quanicassee profiles

At the Wigwam Bay sites, the profiles were not continuous due to the blanking required for the 2013 data (Figure 12). The profiles however do represent the changes to elevation and morphology that occurred from 2013 to 2021 in the areas where overlap was present. At Wigwam Bay there appear to be ridges present in profiles 1, 3, 4, 5, and possibly 6. The ridge structures that were seen in 2013 were no longer present in profiles 1, 4, 5, and 6. In the remaining profile (profile 3) there is a ridge, but the quantity and height of the ridges are reduced from what was present in 2013. There is some deposition shown in the Wigwam Bay profiles but this deposition is not considered detectable as it is within the neutral range of change that was set for the dataset (+/- 0.2m). The average change in elevation across the profiles was a loss of 0.188m and the average maximum loss was 0.381m of elevation (Table 2).

Wigwam Bay	Profile 1	Profile 2	Profile 3	Profile 4	Profile 5	Profile 6	Average
Average Elevation Change	-0.125	-0.209	-0.147	-0.226	-0.154	-0.269	-0.188
Maximum Loss	-0.406	-0.425	-0.427	-0.337	-0.257	-0.432	-0.381
Maximum Gain	0.030	0.059	0.112	-0.079	-0.084	-0.105	-0.011

Table 2: Elevation change data (m) derived from Wigwam Bay profiles

The profiles that were derived from the transects at each site were variable in structure change and magnitude of elevation loss but demonstrate the complexity of the geomorphology in GLCWs. Our profiles show that a loss of lakebed elevation and alteration to the outer edge of the wetland sediment structure occurred from 2013 to 2021. When comparing the 2013 data to the

2021 topobathy data it is clear that the full structure of the wetland was not captured by the remote sensing methodologies deployed in 2013. Our 2021 profiles show a more complete cross-section of what the geomorphology of each site looked like in 2021 (Figure 11, Figure 12). Across the Quanicassee 2021 profiles, there is a decreasing concave-up slope to the lakebed structure and elevations level out as the profile's extent moves lakeward towards the open water (Figure 11). At Wigwam Bay the 2021 profiles demonstrate a decreasing stair-step pattern across the profiles with alternating steep and shallow decreasing slopes that vary across the study site (Figure 12). These data extend farther into the basin than collected LiDAR data and the resulting profiles can be reliably used for comparison of broader wetland geomorphic change over time.

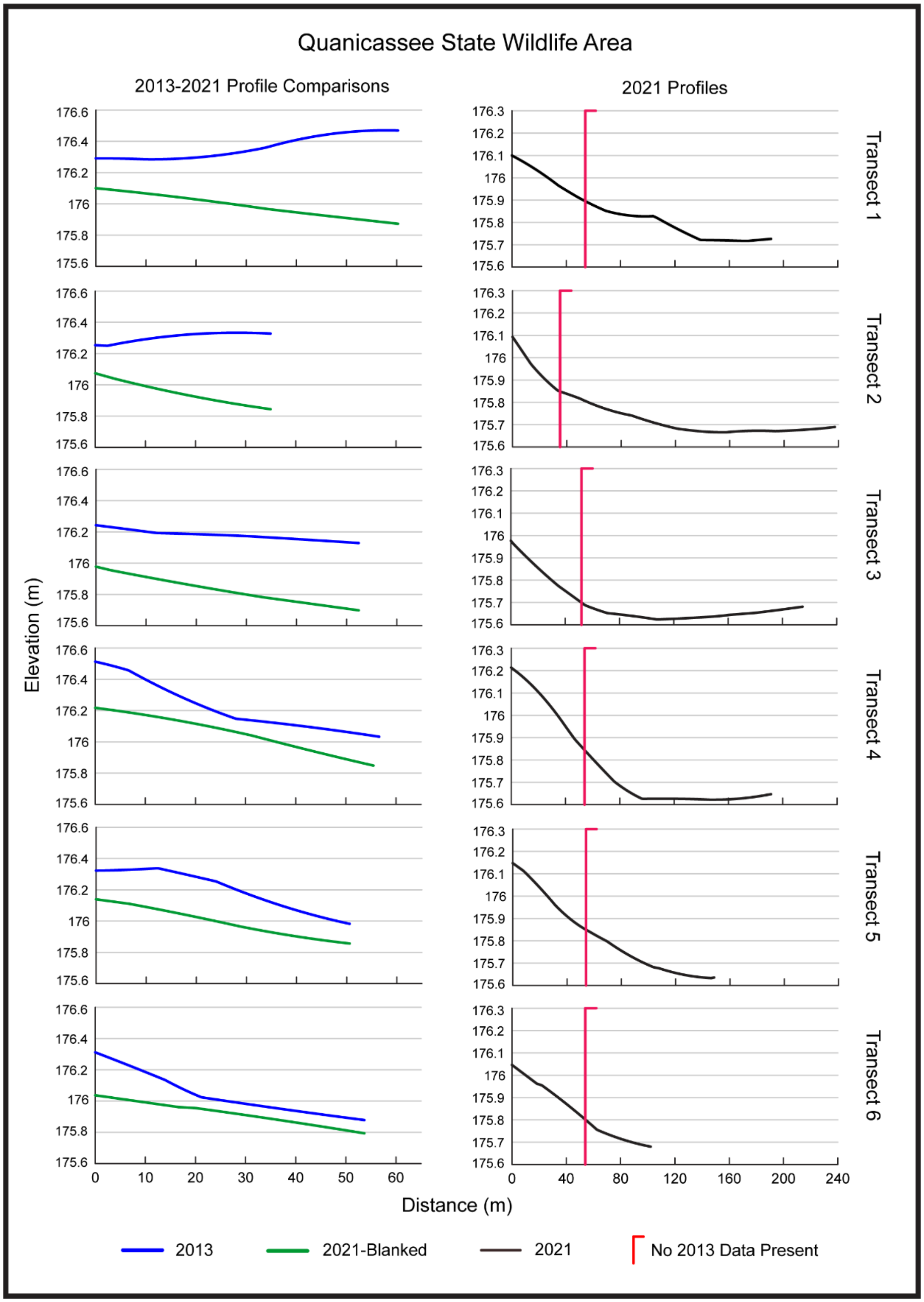

Figure 10: Derived elevation profiles for the Quanicassee Site- Comparison of 2021 and 2013 wetland profiles (left), full 2021 wetland profiles with the end of the available 2013 data marked (right)

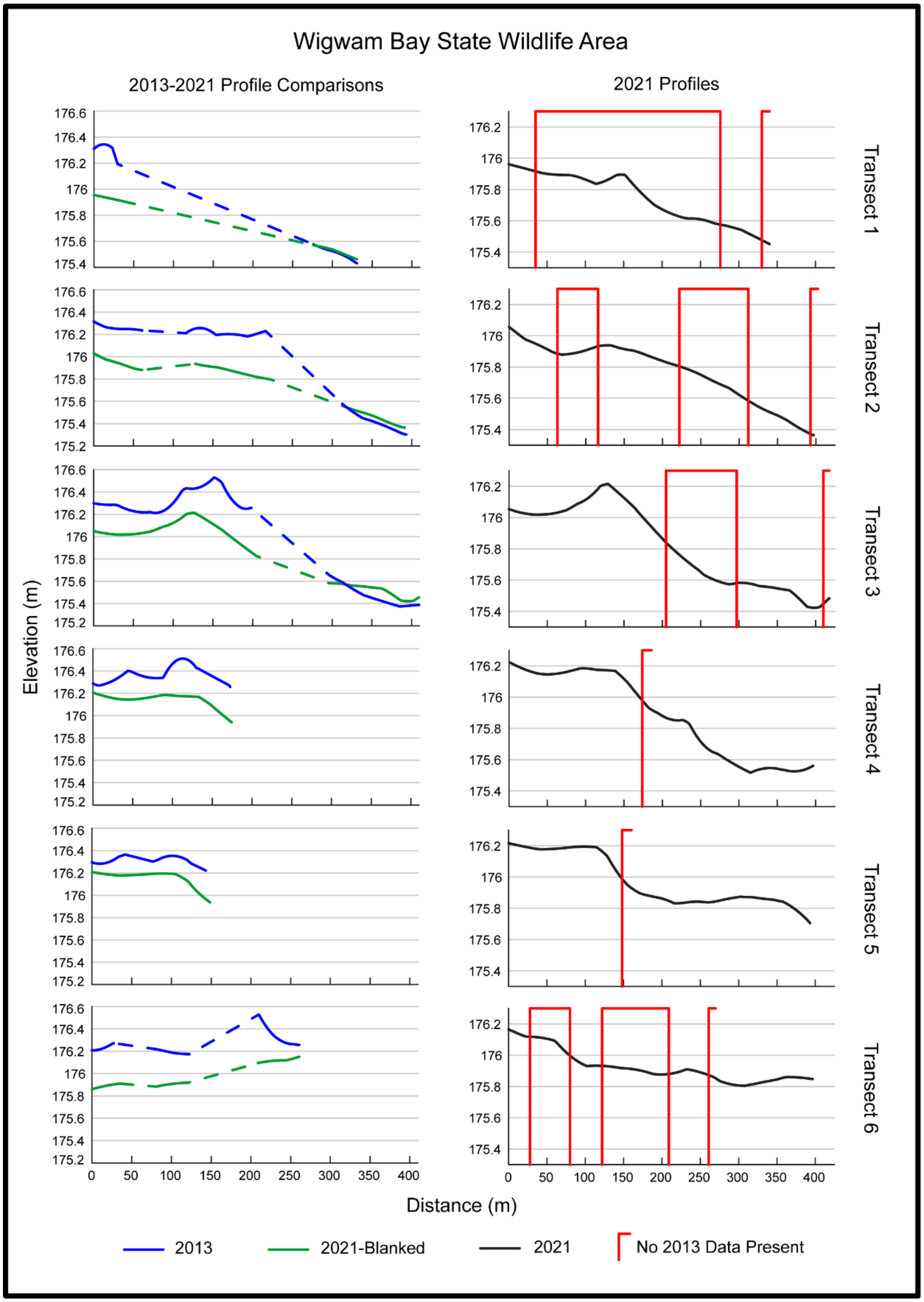

Figure 11: Derived elevation profiles for the Wigwam Bay Site- Comparison of 2021 and 2013 wetland profiles (left), full 2021 wetland profiles with the end of the available 2013 data marked (right)

DISCUSSION

Changes in the geomorphology of coastal freshwater wetlands have the potential to influence the recovery capacity of wetlands and their ecosystem services after a period of high lake level. However, changes in lakebed morphology in GLCWs have not been extensively recorded and are poorly understood. Prior to this study, there was not a time series record of topobathymetric data available for GLCWs to determine how morphology changes over time, making it difficult to predict future responses to natural water level fluctuations in the Great Lakes and disruptions that can occur to those fluctuations due to climate change.

This study is a first step in evaluating how lakebed morphology within coastal wetlands responds to prolonged periods of high-water levels. These geomorphic responses included erosion of sediment at both the Wigwam Bay and Quanicassee sites and the subsequent alteration to the lakebed structure directly adjacent to the wetlands. Implications of this loss can be derived from the geomorphology and vegetation extent changes that occurred at our study sites from 2013 to 2021. These implications include a longer-term loss of wetland areas. For example, loss of sediment observed in the GLCWs can impact oxygen and sunlight availability to wetland vegetation. Wetland vegetation relies on oxygen availability in the water column to survive which is why large wetlands can develop along wide portions of shallow water like those found along the coasts of Saginaw Bay. As the water level rises, energy in the system increases and vegetation retreats landward leaving sediment more exposed and vulnerable to transport. The DOD maps and profiles constructed at the two study sites reveal that during high water levels there was erosion and alteration to how the lakebed sediment is structured. If significant erosion occurs during high water levels, there will be fewer areas that become shallow enough during receding lake levels to support wetland recovery and/or expansion. This impacts both emergent and submergent vegetation which relies on water being shallow enough to allow sunlight to reach the submerged leaves. Overall, lowering lakebed elevation during high water levels along GLCWs can inhibit how far the wetland can recover lakeward during low water levels. This may result in the wetland not expanding as far lakeward as it had during the previous period of low lake levels depending on the extent of morphology change that takes place.

Unfortunately, the available historical topobathymetric data limits how much can be observed in regard to changes in the geomorphology. The slight changes in geomorphology and the erosion that was observed at Wigwam Bay and Quanicassee from 2013 to 2021 could potentially be attributed to being a regular component of GLCW dynamics, but this is unclear from the currently available topobathymetric data. As of right now, there is not a consistent record of geomorphic changes in Great Lakes coastlines so there is no way to establish a baseline for natural sediment fluctuation in GLCWs and therefore no way to determine when these processes become majorly disturbed. This is especially concerning because of the risks wetlands face due to climate change. Climate change is a major contributing factor to a consistent long-term increase in Great Lakes water levels over time, regardless of interannual fluctuations. As average water levels continue to climb, periods of interannual high water levels will become progressively higher and could potentially last for longer periods of time. These changes could intensify sediment loss and alterations to lakebed morphology in GLCWs during high water level cycles. This would push unobstructed wetlands to grow further landward, however much of the Great Lakes coastlines and areas adjacent to the coast are developed for anthropogenic use such as residential use or agricultural use. This will limit how far the wetlands can retreat landward, resulting in coastal squeeze of GLCWs. This shrinks the wetland's area cover and stresses native wetland vegetation as could be seen in the 2021 drone imagery of the Quanicassee and Wigwam

Bay sites. Not only does an increase in magnitude and length of high-water levels influence this but, changes to periods of low water levels can also affect wetland health. A shorter period of low lake levels can inhibit the wetland from establishing completely and prevent diverse assemblages from establishing and stabilizing in the wetlands. During uncharacteristically long periods of low water levels, woody and shrub-like vegetation can outcompete other species and there is time for invasive species to take hold in the wetlands. The results of all of this would be a reduction in the biodiversity of plant life and wildlife as competition for dwindling resources such as habitat and food increases. This threatens several ecosystem services that wetlands provide.

Reduction in wildlife presence disrupts food chain dynamics and could negatively impact native populations, especially those already considered at risk due to habitat loss or competition with invasive species. It also diminishes the recreational use of wetlands for fishing, hunting, and activities such as bird watching. Additionally, stress to and loss of wetland vegetation lessens the wetland's natural water filtration abilities through nutrient uptake. The facilitation of water quality improvement done by GLCWs is already at risk due to the colonization of invasive vegetation such as common reed that disrupts nutrient uptake. This means that increasing amounts of pollutants from runoff will make their way into Great Lakes waters. This problem will become more prevalent if there is a significant loss of wetland vegetation extents due to high water levels and the lowering of lakebed elevation. Vegetation in GLCWs also provides coastline protection through water flow regulation and buffering of high-stress events by dissipation of wave energy. Changes in wetland lakebed morphology could alter this process in multiple ways. In general, the retreat of GLCWs without the ability to expand landward will result in wave energy moving further inland than it previously would have, putting wildlife and human-made structures at risk for disturbance. There can also be less dissipation of wave energy, not only increasing the distance inland wave energy can travel but also increasing the intensity of wave energy that the wetland and coastline experience. This is because as sediment elevation lowers in front of the wetland at the lakeward side, it can reduce the presence of emergent and submergent vegetation along the edge of the wetland. This loss has the potential to contribute to a positive feedback loop in which sediment becomes more vulnerable with fewer root systems to hold it in place and increased exposure to wave energy. This could compound to increase erosion and result in scouring of sediment along the lakeward extent of GLCWs, further altering the lakebed morphology.

It is important to address that these, like many discussions on GLCWs and their morphology, are speculative findings based on limited available data and observations of GLCWs. Based on this study it is found that from 2013 to 2021 there were record high water levels in the Great Lakes and during that time two established GLCWs experienced erosion of sediment and a subsequent lowering of lakebed elevation. This lowering of lakebed elevation altered the lakebed profiles adjacent to the vegetation edge at both the study sites. It is not certain how this will impact the wetland extent during low water levels. Based on what is known of wetland vegetation response to variable water levels it is possible that the wetlands will not expand as far lakeward as they previously had. During the period of high-water levels, the lakeward extent of wetland vegetation retreated at the more sheltered Wigwam Bay site, but not considerably at the Quanicassee site, marking the variability that can be present between GLCWs. However, both sites experienced increased inundation and vegetation loss within the wetlands, which puts the previously discussed ecosystem services at risk. How much risk is a very important question that has emerged from this study? The most recent topobathymetric data publicly available for Saginaw Bay wetlands was collected in 2013 and contains considerable gaps in data due to the turbidity of coastal systems that interfered with the remote sensing methods. However, we have shown that with our methodology that it is possible to create a high-accuracy topobathymetric map of wetland geomorphology. It was also found that taking cross-section profiles across the wetland sites can provide information on the structure of wetland sediment and how it can change over time. Our collected 2021 data marks an important move forward in studying these systems and shows that it is possible to keep a record of topobathymetric data that can inform the study of wetland geomorphology.

Overall, this study provides an assessment of how these GLCWs have changed from the beginning of an interannual cycle of high-water levels to when water levels just began to decline from record peaks. There is no information to capture the changes in geomorphology that may have occurred throughout that eight-year timespan during seasonal or episodic water level fluctuations. It cannot be definitively said which processes are contributing most to the sediment movement in these systems and the changes in morphology from 2013 to 2021. This work marks the need for and importance of studying and understanding sediment movement due to the part it plays in complex coastal wetland dynamics. As technology and techniques continually improve, more knowledge can be gained as to how changes to wetland sediment morphology that occur during high water levels will truly impact vegetation extent and wetland health as water levels continue to fluctuate. Regularly collected topobathymetric data would provide insight into how sediment moves through coastal freshwater wetlands during low and high interannual water level fluctuations. It may also bring to light episodic and seasonal changes in lakebed morphology that corresponds to water level. This research is a stepping stone to a broader understanding of

GLCWs and provides a framework that can be used in the study of freshwater coastal wetlands found outside of the Great Lakes region.

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