An Aerial View of the Shoreline: Using Remotely Piloted Aircraft to Map Vegetation in Netley-Libau Marsh and Compare Plant Assemblages Across an Exposure Gradient

by

Christopher Adams

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Clayton Riddell Faculty of Environment, Earth, and Resources

University of Manitoba

Winnipeg, MB

Abstract

Netley-Libau Marsh has undergone a marked process of degradation during the past century. One of the most noteworthy changes has been a reduction in aquatic vegetation, along with a corresponding increase in open water cover. The shoreline can be understood as a critical zone, where landforms may expand, recede, or remain stable, based on interactions between vegetation and hydrogeomorphic processes. For the first component of my thesis, I developed an accessible method for using a small remotely piloted aircraft (micro-RPA) to map shoreline vegetation in prairie freshwater marshes. This method consists of two manual shoreline surveys at two different altitudes: 120 m and 10 m. The minimal regulations and low cost associated with small RPAs remove many of the barriers to entry for their use in field ecological research. Moreover, this method is likely to be robust from June to October, as it is based primarily on the shape of leaves and stems, rather than colour or spectral signature. This method is also inexpensive, as it can be done using open-source software (WebODM). The second component of my thesis used this method to compare the taxonomic composition of plant assemblages across an exposure gradient. I surveyed 23 km of shoreline between August 17 and August 29, 2020, and produced orthomosaic maps for all shoreline zones. Using data from 150 randomly placed shoreline quadrats, I used a generalized linear model to compare plant community composition along an exposure gradient. Exposure is defined as the total effect of waves on shoreline vegetation. I estimated exposure for each quadrat using an effective fetch model, integrating data on wind directional frequency from 36 compass directions with a spatial raster of the study area within ArcMap. My results indicate that exposure acts as an environmental filter, limiting the growth of *Typha* spp. in exposed shoreline areas. *Typha* spp. was less common in exposed locations, and it was also more likely to show visible damage in locations with relatively high effective fetch. The relative tolerance of emergent taxa to exposure stress should be taken into consideration when designing revegetation projects in Netley-Libau Marsh and other wetlands.

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Table of Contents

List of Figures and Tables

Chapter 1. Introduction

1.1 Background

Wetlands are among the most degraded ecosystems in southern Manitoba, having declined in cover considerably since 1870 (Hanuta 2006, Stunden Bower 2011). In coastal freshwater marshes, there has been a substantial decline in aquatic vegetation cover, and a general shift towards a turbid open-water state (Grosshans et al. 2004, Wrubleski et al. 2018, Kowal 2019, Watchorn et al. 2021). There has also been an influx of invasive species, including *Cyprinus carpio* (Badiou and Goldsborough 2011, Wrubleski et al. 2018, Kowal et al. 2022), *Lythrum salicaria* (Lindgren 2003, Lindgren and Walker 2012) and likely *Typha* X *glauca* (Grosshans et al. 2004, Tangen et al. 2022). The net result of these impacts is that many wetlands no longer provide the suite of ecological goods and services that they provided historically (Grosshans et al. 2004, Carter Johnson et al. 2010).

Aquatic plants function as ecosystem engineers, altering the course of water flow, as well as by modifying the cohesiveness of the substratum (Corenblit et al. 2009, Strayer and Findlay 2010, Gurnell 2014, Ge et al. 2019). In fluvial and tidal wetlands, the shoreline zone can be understood as a critical zone, where landforms may expand, recede, or remain stable, based on dynamic interactions between vegetation and hydrogeomorphic processes (Corenblit et al. 2009, Gurnell et al. 2014, Ge et al. 2019). The reciprocal interplay of biotic and hydrogeomorphic processes has also been applied to models of landform succession wetlands that experience fluvial (Corenblit et al. 2009, Gurnell 2014), and tidal (Sasser et al. 2018, Ge et al. 2019) forces. While some of the tidal models have been developed in marine coastal wetlands, coastal freshwater marshes can be subject to analogous physical forces of wind, waves, and wind-tides (Keddy 1982, Strayer and Findlay 2010, Mason et al. 2018, Watchorn et al. 2021).

Netley-Libau Marsh is one of the largest freshwater marshes in North America (Watchorn et al. 2012, Wrubleski et al. 2018) and it is located at the south end of Lake Winnipeg, where the Red River empties into the lake. Like many other wetlands, Netley-Libau Marsh has become degraded throughout the past century (Grosshans et al. 2004, Wrubleski et al. 2018, Watchorn et al. 2021). This degradation has been characterized by a prominent loss of emergent and submerged aquatic vegetation, as well as an increase in turbidity (Grosshans et al. 2004, Kowal 2019, Watchorn et al.

2021). Although the total cover of emergent vegetation has declined with increases in open water, the proportion of *Typha* spp. and *Phragmites australis* has increased, relative to the proportion of *Schoenoplectus* spp. (Grosshans et al. 2004, Watchorn 2015).

Successional changes in the shoreline plant communities of Netley-Libau Marsh are driven by primarily by hydrological cycles, with shoreline vegetation expanding into open water areas during dry years and dying back during wet years (Grosshans et al. 2004, Watchorn et al. 2021). Relatively dry conditions in 2018 and 2019 led to the recruitment of plants from the seed bank in areas that are normally too deep for emergent plant growth. The observed recruitment indicates that resident propagule banks, plant growth, and sediment deposition have the potential to drive the expansion of vegetated landforms, given adequate hydrological conditions.

The moderately high water in 2020 provided an opportunity to study the effects of relatively higher water on vegetated shorelines throughout Netley-Libau Marsh, following two years of relatively low water. Although existing research on Netley-Libau Marsh and other freshwater marshes has emphasized the importance of hydrological regimes in driving successional changes in vegetation (van der valk 1981, 2018; Watchorn et al. 2021), the role of wave exposure in the shoreline zone has received less attention. Nevertheless, there is no question that wetland plants have differing degrees of tolerance to the mechanical stress of wave exposure, and that these differences in tolerance can drive successional changes in plant communities (Warming 1897 [in Clements 1916], Anderson 1978, Keddy 1982). It is unclear whether the death of shoreline vegetation in wet years is driven primarily by the stress of deep water, or whether mortality tends to be greater in shoreline communities that are subjected to the combined effects of wave exposure and high water.

The research presented here will contribute to the existing body of knowledge by using inexpensive low-altitude aerial imagery to map shoreline plant communities in Netley-Libau Marsh and compare taxonomic composition along an exposure gradient. Whereas previous studies have provided a clear picture of vegetation changes in Netley-Libau Marsh on a macro-scale, these studies have limited spatial resolution, due to their use of high altitude (1920 m) aerial imagery (Grosshans et al. 2004), and Landsat imagery (Watchorn et al. 2021). Using a small remotely piloted aircraft (micro-RPA), my research will provide an understanding of shoreline communities at an intermediate scale, which is not possible using traditional sampling methods at the ground-level or using traditional aerial methods.

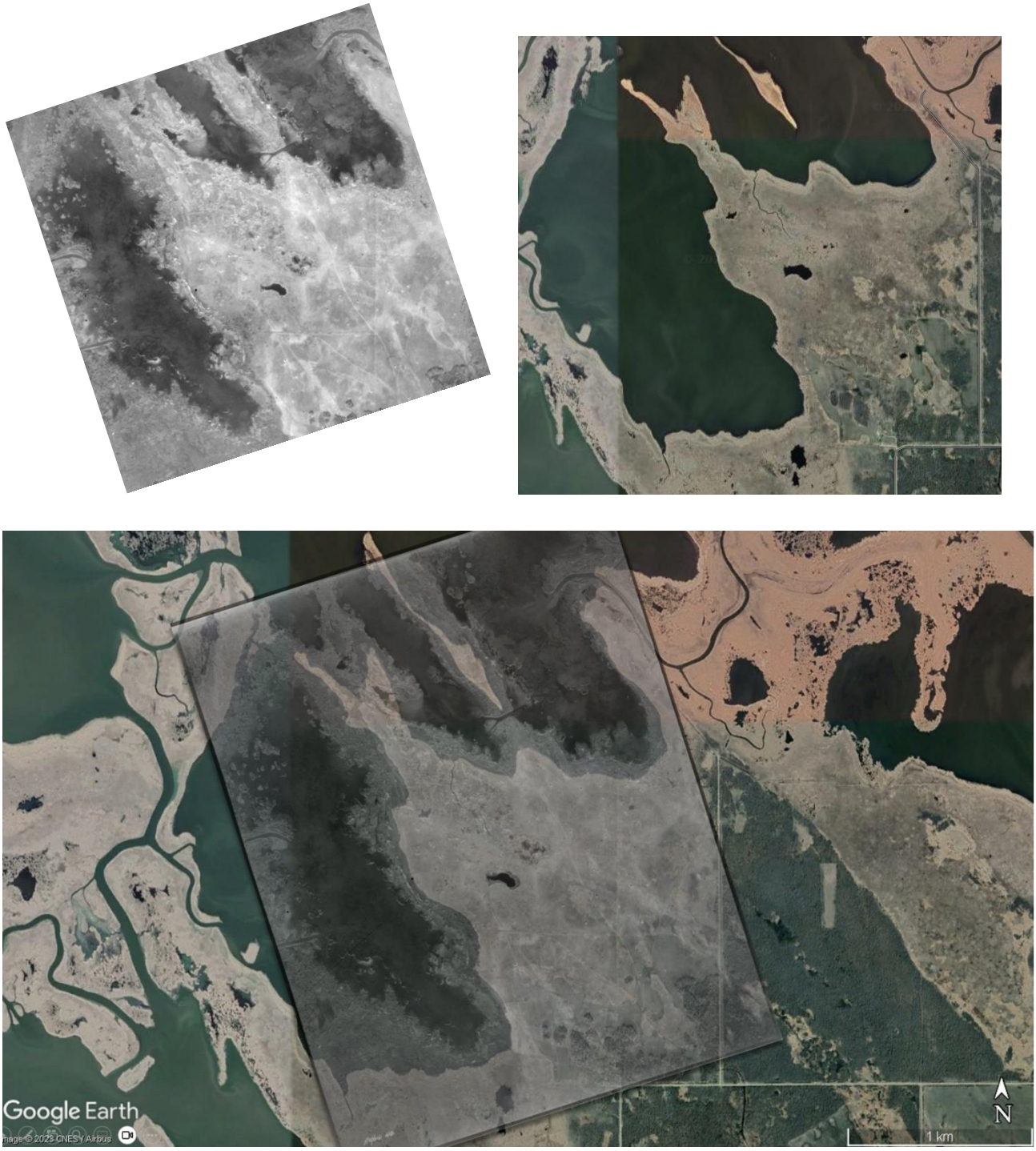

Figure 1. A portion of Netley-Libau Marsh in 1929 (top left) and 2022 (top right). Shorelines have become simplified, and the surface area of open water has increased. 1929 image obtained from the National Air Photo Library and provided by Dennis Anderson. 2022 image obtained from Google Maps.

1.2 Problem Statement and Research Objectives

Netley-Libau Marsh has become degraded over the course of the past century (Grosshans et al. 2004, Wrubleski et al. 2018, Watchorn et al. 2021). There has been a marked increase in open water area and turbidity, as well as a sharp decline in the cover of *Schoenoplectus* spp. (Grosshans et al. 2004). A decline in native emergent vegetation has been exacerbated by an influx of invasive species (Lindgren 2003, Badiou and Goldsborough 2011, Lindgren and Walker 2012), and this has led to a reduction in ecological goods and services (Grosshans et al. 2004, Watchorn et al. 2021).

The objective of the first component of my thesis was to develop an accessible method for using a micro-RPA to map shoreline vegetation in prairie freshwater marshes. This method consists of two manual shoreline surveys at two altitudes: 120 m and 10 m.

The objective of the second component of my thesis was to use the method developed in the first component of my thesis to compare the taxonomic composition of plant assemblages across an exposure gradient. I hypothesized that *Typha* spp. would be particularly vulnerable to the mechanical stress of wave exposure. Consequently, I predicted that there would be a negative relationship between wave exposure and *Typha* spp. within my dataset. I divided my prediction into two separate components, predicting that (a) *Typha* spp. would be present less frequently in exposed areas, and that (b) *Typha* spp. would tend to be present in smaller quantities in exposed areas.

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Chapter 2. Literature Review

2.1 Aerial Imagery and Vegetation Mapping

Aerial ecological surveys rose in popularity during the 1930s through the work of prominent British ecologists, in conjunction with large-scale imperial projects of resource exploration (Melvill Jones and Griffiths 1925, MNM 1925, Bourne 1928, Fischer 1928). Notably, Anker (2001) has argued that an aerial perspective contributed to Arthur Tansley's development of the modern ecosystem concept in ecology (Tansley 1935), although Tansley lifted the term itself from one of his graduate students (Anker 2001). Nearly a century later, modern authors have described the advance of inexpensive remotely piloted aircraft (RPA) technology as "revolutionary" to vegetation survey methods in remote field locations (Madden et al. 2015, Kalacska et al. 2017).

Although the method of execution has changed considerably, the four key challenges discussed in Melvill Jones and Griffiths (1925) have enduring relevance for modern RPA operators:

1. How best to ensure photography of the whole area with a minimum of flying.
2. How to arrange the photography so as to reduce to a minimum the ground work necessary to control it.
3. How to deal with the photographs when taken, and adjust them to the control.
4. Estimation of the errors to be expected in the procedure advocated.

Whereas Melvill Jones and Griffiths (1925) were willing to accept a spatial margin of error between 0.03 miles and 0.07 miles, the GPS data provided by a recreational-class RPA can provide the operator with precision within 5-10 m, and cm-level precision can easily be obtained using high precision GPS and ground control points. It is important to draw a sharp distinction between the precision of on-board GPS data as it relates to estimates of latitude and longitude, in comparison with estimates of elevation. Whereas modern technology makes it very simple to generate a three-dimensional digital elevation model using aerial imagery, elevation models will be virtually useless without the incorporation of ground control points.

In many countries, including Canada and the United States, RPAs are restricted to a maximum altitude of 120 m (*Canadian Aviation Regulations*, May 29, 2023). Because of this altitude restriction, piloted aircraft still provide opportunities for increasing the scale of surveys in a way

that is not (legally) possible for most RPA operators. However, it is possible to request and obtain a permit to fly higher than 120 m under specific conditions, if the operator can justify the importance and safety of their plans. It is conceivable that a permit could be obtained for the purpose of mapping vegetation in the context of ecological restoration projects. By flying in a piloted aircraft at an altitude of 1920 m, Grosshans et al. (2004) mapped the entirety of Netley-Libau Marsh (26,000 ha) using 106 photographs.

Despite altitude restrictions, the proliferation of RPA technology has the potential to vastly reduce the amount of time required to conduct ecological research in remote locations (Madden et al. 2015, Kalacska et al. 2017). Moreover, the ability to safely hover at low altitudes provides opportunities for intermediate-scale research that are not feasible with a piloted aircraft. The availability of high-resolution aerial imagery makes it possible to extend traditional vegetation sampling methods to a larger scale, with relatively low sampling effort. Aerial images can be stitched together to produce orthomosaic maps (Madden et al. 2015, Kalacska et al. 2017, Esposito et al. 2017, Doughty and Cavanaugh 2019, Duffy et al. 2018). High resolution cameras and open-source software make it possible to delineate vegetation classes and generate vegetation cover maps (Grosshans et al. 2004, Kalacska et al. 2017, Doughty and Cavanaugh 2019). Recent advances in RPA technology have removed financial barriers that previously limited the availability of high-quality aerial imagery (Kalacska et al. 2017, Rhee et al. 2018).

Near-infrared aerial images have been used to create orthomosaic maps describing the upland and emergent marsh vegetation of Delta Marsh (Grosshans 2001), and Netley-Libau Marsh (Grosshans et al. 2004). Orthomosaic maps generated from RPA imagery have also been used to identify submerged and emergent vegetation (Chabot et al. 2018, Meneses et al. 2018, Doughty and Cavanaugh 2019). In some cases, high resolution aerial images have facilitated reliable identification of macrophytes to the species-level (Husson et al. 2013, Chabot et al. 2018).

Multispectral imagery provides more spectral detail than natural light imagery, but multispectral cameras are expensive, and the interpretation of imagery is a specialized skill. The most accessible multispectral RPA is the Mavic 3 Multispectral Edition, which costs $4,618 USD. Conversely, it is possible to conduct aerial surveys on a modest budget, using natural light imagery, for a cost of $279-$500 USD. Natural light imagery can also be interpreted with less training than multispectral imagery. Furthermore, high resolution natural light imagery can be obtained using recreation-grade

RPAs, which fall below the 250-gram regulatory threshold that applies in Canada and the United States. This eliminates the requirement for an operator's license and RPA registration, which are non-trivial barriers to entry for researchers. It is important to keep in mind that operators must still conform to most of the same flying restrictions that licensed operators are required to follow while flying a registered RPA. The *Aeronautical Information Manual* (March 23, 2023) provides guidelines for the appropriate use of RPAs that fall below the 250-g threshold:

(a) Maintain the RPA in direct line of sight.

(b) Avoid flying your RPA above 400 ft above ground level (AGL).

(c) Keep a safe lateral distance between your RPA and other people.

(d) Stay far away from aerodromes, water aerodromes, and heliports.

(e) Avoid flying near critical infrastructure.

(f) Stay clear of aircraft at all times.

(g) Conduct a pre-flight inspection of your RPA.

(h) Keep the RPA close enough to maintain the connection with the remote controller.

(i) Follow the manufacturer's operational guidelines.

(j) Avoid special aviation or advertised events

There may be practical considerations that favour the use of a smaller or larger RPA. Smaller RPAs are easier to launch and land within a confined space, such as a canoe or small boat. Moreover, the propeller blades of small RPAs are unlikely to cut through skin in the event of an accident in the field. While all precautions should always be taken to avoid touching the blades of any RPA, larger RPAs can conceivably lead to a medical emergency in a remote field setting. The major advantages of larger RPAs are that they have greater capabilities on windy days, and they can be kept within visual line of sight over a greater distance. In circumstances where operators must launch an RPA from a confined space, safety considerations will likely outweigh these advantages of larger RPAs.

2.2 Mapping Software

Recent advances in RPA technology have been accompanied by corresponding advances in software, permitting users to generate orthomosaic maps with minimal formal training. WebODM is an accessible open-source software platform that allows users to generate high quality orthomosaic maps, using overlapping aerial imagery. For two-dimensional mosaics, RPA operators must ensure that all images being input into WebODM have a minimum of 60% spatial overlap. As long as the minimum 60% overlap is present among high-quality photographs, WebODM will stitch the images automatically.

In addition to two-dimensional maps, WebODM allows users to generate three-dimensional point cloud models, using a similar set of overlapping aerial photographs. For three-dimensional applications, a minimum overlap of 85% is recommended, and it is also recommended to fly in a grid pattern with the camera tilted at an oblique angle to capture the vertical dimension of what is being mapped. WebODM provides an accessible interface, which permits users to make basic aerial and volumetric measurements without the need for specialized GIS skills. Depending on research priorities and equipment availability, ground control points can be integrated within WebODM. Alternatively, users can simply input photographs and rely on the lower level of spatial precision present in the GPS data of the RPA.

Whereas WebODM makes it extremely simple to stitch images together, it does not assist in the image-acquisition stage. Operators who use WebODM can either acquire images manually, or they can use automated mission planning software to optimize the precision of overlap within their aerial imagery. Dronelink and Litchi are two open-source software platforms that make it possible to plan out a pre-determined flight mission, with a variety of customizable options for setting waypoints and mapping target areas. With Dronelink and Litchi, different levels of access can be purchased for a one-time cost of $25-100. However, certain advanced functions can only be accessed with a commercial subscription, which ranges from $20-30 per month.

A major advantage of automated flight software is that it allows operators to maximize the efficiency of flight paths, as well as the replicability of surveys. Once a mapping mission has been programmed, it can easily be repeated in the future. Nevertheless, it is important for operators to

be prepared to intervene manually whenever necessary. Consequently, it is preferable for RPA operators to develop some experience flying manually before relying on automated software.

Pre-planned flight missions can be simple or complex, and they can include multiple components, such as mapping and waypoints. As depicted in Figure 2, operators can adjust all aspects of the flight mission, including altitude, speed, image overlap, and waypoint positions. While the flight path depicted in Figure 2 would not be impossible to carry out manually, it would be challenging, and it would be even more challenging to replicate it manually in the future.

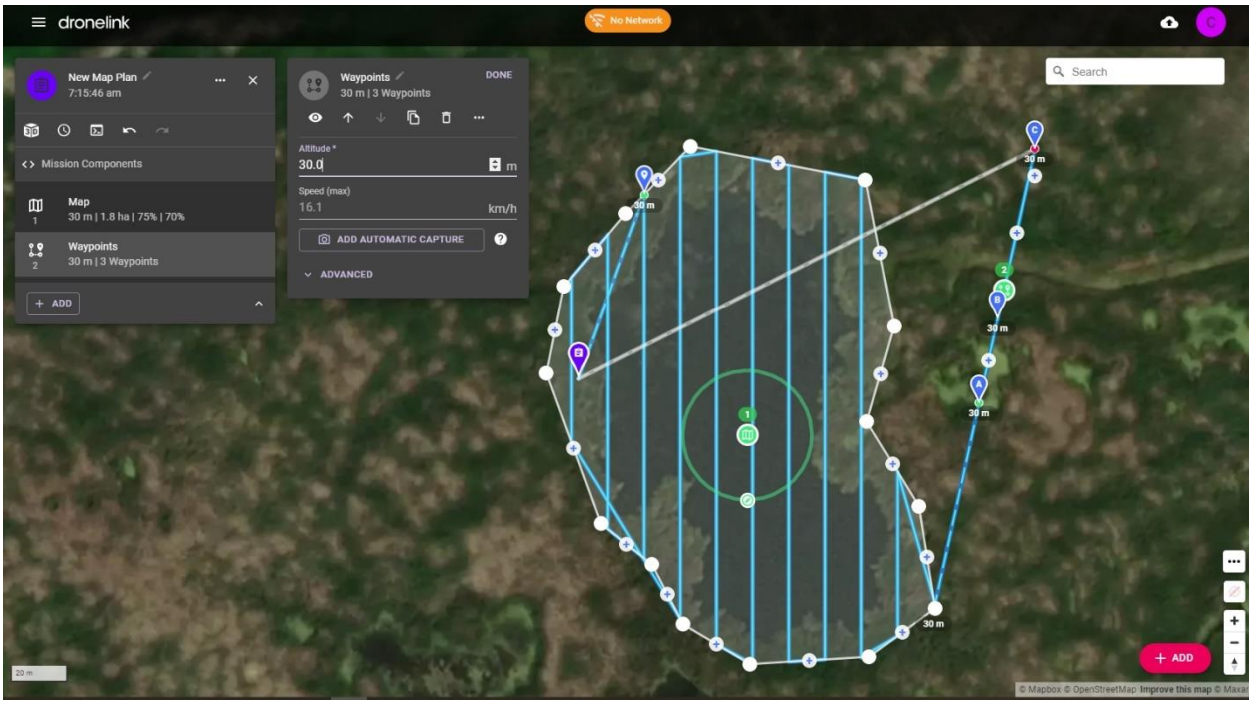

Figure 2. Example of a flight mission in Dronelink. This mission consists of two components – map and waypoints. First, the RPA will map the area with parallel straight lines. The RPA will fly at an altitude of 30 m with a maximum speed of 16.1 km/h, 75% front/back overlap, and 70% overlap in images from side-to-side. Second, the RPA will move successively to waypoints A, B, and C, taking an image of the point of interest at each waypoint.

2.3 Traditional Mapping vs Corridor Mapping

In contrast with standard automated mapping missions, in which an RPA follows a series of overlapping parallel flight paths (Figure 2), corridor mapping missions consist of one or more passes along a narrow strip, which may be either straight or curved (Figure 3). While the standard mapping flight path of overlapping straight lines is efficient across areas that have a length and width of a similar scale, this pattern is inefficient along narrow/winding strips, such as shorelines or the edges of forested areas.

Whereas most automated mapping functions can be accomplished easily and affordably using automated software, corridor mapping missions are only available using proprietary software, such as Pix4D ($150-$400 CAD per month), or a commercial subscription to an open-source platform, such as Dronelink ($20-$30 CAD per month).

Figure 3. Example of a corridor mapping mission, consisting of two passes along a curved strip. The flight path is depicted in light blue. Reproduced from https://support.dronelink.com/hc/en-us/articles/14090648504083-Linear-Corridor-Mapping.

In contrast with the limitations for corridor mapping with non-commercial mapping software, single-pass corridor mapping can be done efficiently using manual flight. As illustrated in Figure 4, standard mapping missions are restricted to a flight path of overlapping parallel transects (Figure 4A), whereas the corridor mapping mission depicted in Figure 4B follows a flight path that conforms to the shape of the shoreline. By following the shape of the shoreline, the corridor

mapping mission makes it possible to map the same area as the standard mapping mission, while flying a much shorter path. If a single pass is sufficient, this can be carried out manually. However, if two or more passes are required, this is generally not feasible using manual flight. In cases where a wide corridor map with multiple passes is necessary, this will require a commercial subscription to Dronelink or Pix4D.

Although manual corridor mapping is limited to flying corridor mapping missions in a single pass, there are some meaningful advantages to flying in a single pass when a relatively narrow strip (100 m wide at 120 m altitude) is sufficient for mapping goals. Single-pass corridor mapping requires approximately half of the image overlap that is required using standard mapping methods. This is because images only need to overlap in one direction (front-to-back), rather than two directions (front-to-back and side-to-side). All else being equal, single-pass corridor maps allow operators to map the greatest possible area, using the fewest possible images. Nevertheless, it is important to be mindful of the limitations in precision and replicability associated with manual flight. If flight time and budget are not major constraints, it will often be preferable to carry out automated flight missions.

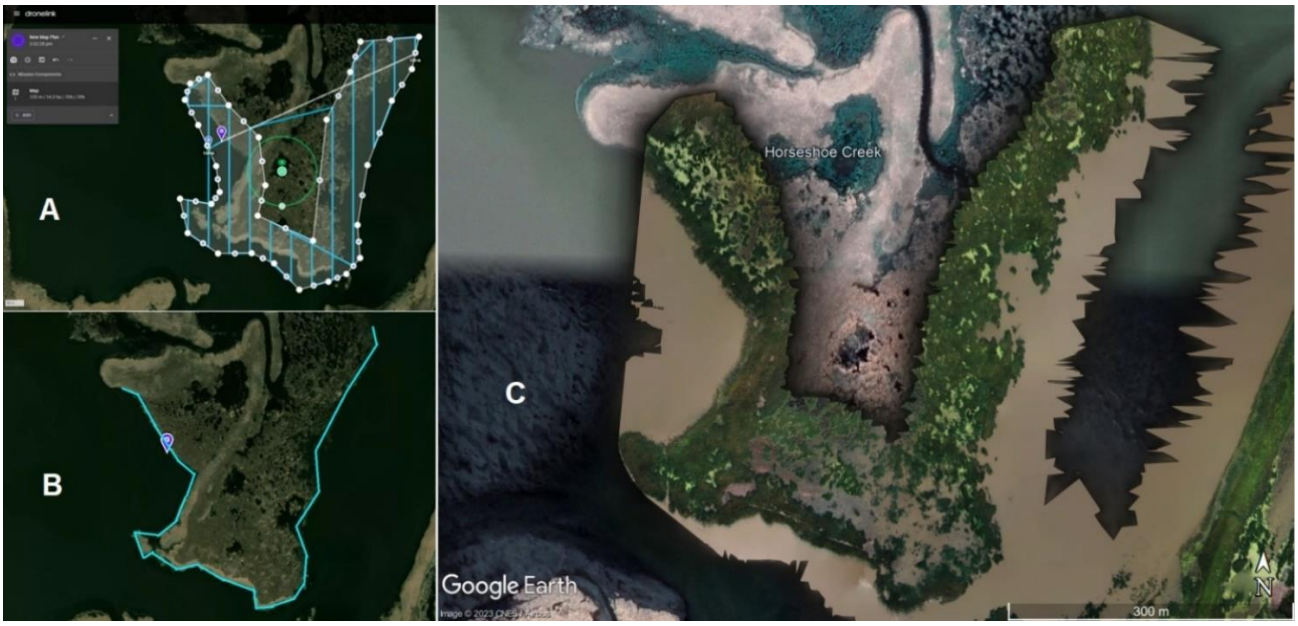

Figure 4. Comparison of flight paths (light blue) for a standard mapping mission (A) and a corridor mapping mission (B). Note that standard mapping methods require a considerably longer flight path than corridor mapping methods. Figure C is an orthomosaic that was produced in WebODM, using images captures during a manual flight path similar to Figure B.

2.4 Restoration of Prairie Freshwater Marshes

Restoration ecology is a relatively young branch of ecology that focuses on (1) describing and quantifying changes that have occurred in a natural system, (2) understanding the mechanisms driving such changes, and (3) devising solutions for how a degraded system can be improved (Hobbs and Suding 2009; Palmer et al. 2016). The success of a wetland restoration project can be influenced positively or negatively by many variables, including habitat type, hydrological regime, soil properties, topography, nutrient supplies, disturbance regimes, invasive species, seed banks and changes in biodiversity (Zedler 2000). Oftentimes, these variables are not well-understood until after a restoration attempt has either succeeded or failed (Zedler 2000). Consequently, a lack of information can cause problems when attempting to restore a site about which there is insufficient information (van der Valk et al. 1992, Zedler 2000).

All restored and constructed wetlands must have hydrological conditions that promote the growth of hydric plants (van der Valk et al. 1992, van der Valk 2018). Additionally, desirable hydric taxa must be present (van der Valk et al. 1992). Which taxa are considered desirable in any given restoration project will depend on the priorities of the restoration project in question. This may include native taxa in general, or a more specific subset of native taxa that are desirable for a specific purpose (e.g., erosion control). In some cases, desirable taxa will germinate naturally from a seed bank when hydrological conditions are restored to a drained site (Wienhold and van der Valk 1989, Dawson et al. 2017). The effectiveness of this method declines with the age of the drained site, as the ancestral seed bank dies off throughout time (Wienhold and van der Valk 1989). However, drained sites that receive regular flooding have been found to be more resilient against this decline than drained sites not subject to periodic flooding (Dawson et al. 2017). An emerging wetland restoration technique uses natural sediment as building material to construct restored wetland areas that are resilient against future degradation (Byrnes and Berlinhoff 2012, Audubon 2018, Louisiana Coastal Master Plan 2023). In addition to acting as construction material, sediment can contribute to plant recruitment through the seeds and propagules contained within its soil seed bank (van der Valk et al. 1992, Dawson et al. 2017).

Most constructed wetlands and many restored wetlands incorporate the addition of native seed mixes, as well as vegetative propagules and plantings (van der Valk et al. 1992, Byrnes and Berlinhoff 2012, Audubon 2018). Rather than adding seeds and propagules separately, an

alternative technique harnesses donor soils from sites that have been selected for having a desirable mix of species present in the seed bank (van der Valk et al. 1992). There are three critical factors required for such efforts to be successful (van der Valk et al. 1992). Firstly, desirable species must be present in the seed bank. Secondly, unwanted species must not be present in substantial quantities. Finally, hydrological conditions must be maintained to promote the growth of hydric species (van der Valk et al. 1992). For fluvial and tidal wetlands, hydrological conditions extend to geomorphic considerations (Gurnell 2014), and wetlands can be classified based on their hydrogeomorphology (Watchorn et al. 2012).

Vegetation species composition varies along exposure gradients (Anderson 1978, Keddy 1982), and vegetation differences can influence the complexity of the shoreline (Strayer and Findlay 2010). The simplification of shorelines can lead to an increase in wave energy that may hinder restoration attempts in wetland environments with greater wave exposure. Moreover, shoreline simplication can lead to cascading effects for animals that inhabit the shoreline zone, due to a reduction in the dissipation of physical energy (Strayer and Findlay 2010).

As shorelines become increasingly simplified, the degree to which vegetation attenuates wave energy in the shoreline zone declines, resulting in inhospitable conditions for waterbirds. For example, Laporte et al. (2014) found that wave disturbance was the leading cause of destruction among Western Grebe nests in Delta Marsh, Manitoba. The authors also indicated that a historic shift towards the dominance of *Typha* spp. has led to simplified shorelines, such that Western Grebes were forced to build their nests in exposed locations. Similarly Gonzalez-Gajardo et al. (2009) found a positive relationship between shoreline complexity and waterbird abundance.

Historically, wave energy has been quantified using relatively coarse measurements of wind fetch, based on the distance that wind may travel uninterrupted across a body of water in the direction of prevailing winds (Keddy 1982, Mason et al. 2018). Effective fetch is an adjusted fetch measurement, which applies reduced weight to long fetches in long narrow water bodies (Keddy 1982, Mason et al. 2018). Using GIS software, effective fetch can be computed for each site as a function of wind speed and 36 different direct fetch measurements, at 36 equally-spaced compass bearings radiating from each shoreline site (Keddy 1982, Rohweder 2012).

Prairie marsh succession models have focused primarily on the anoxic effects of water (van der Valk 2018). In contrast, the successional role of physical wave energy has remained largely unexamined by modern authors. Warming (1897) [translated and summarized in Clements (1916)] found that *Typha* spp. is torn out by wave action more readily than *Phragmites communis* and *Scirpus* spp. Thus, Warming (1897) concluded that the zonation of these communities is mediated by differences in wave exposure, as well as water depth. Keddy (1982) found comparable zonation along exposure gradients. However, modern authors have not explicitly incorporated exposure into models of prairie wetland succession (van der Valk 2018).

The term "hemi-marsh" was first used by Weller and Spatcher (1965) to refer to a transitional stage in wetland succession, whereby emergent vegetation and open water are highly interspersed and each occupies roughly 50% of the surface area. Weller and Spatcher (1965) and Weller and Fredrickson (1974) both found that bird abundance and diversity tended to be highest during years in which hydrological conditions gave rise to a hemi-marsh pattern of interspersion between emergent vegetation and open water. Whereas the term has been used rather loosely in the context of prairie wetland management, a hemi-marsh is not a distinct type of marsh, but rather a transitional stage between different points within the prairie wet-dry cycle (Murkin et al. 1997). Consequently, hemi-marsh conditions may occur in a wide variety of types of marsh, including prairie pothole marshes and coastal freshwater marshes.

Kaminski and Prince (1981) was particularly influential in popularizing the use of the hemi-marsh concept within wetland management, concluding that dabbling ducks had greater abundance and diversity in experimentally manipulated plots with a water:vegetation ratio of 50:50, compared with 70:30 and 30:70 (see also Murkin et al. 1997, Ma et al. 2010 and Gray et al. 2021). Masto et al. (2022) [including both Kaminski and Prince as co-authors] reanalyzed the dataset from Kaminski and Prince (1981) to resolve a problem of temporal pseudo-replication within the statistical analysis. To account for correlation between datapoints that were measured repeatedly in the same plots over the study period, Masto et al. (2022) reanalyzed the dataset from Kaminski and Prince (1981) using nested random effects in generalized linear mixed effects models. Whereas Masto et al. (2022) identified an error in the original analysis, the results of the reanalysis generalized and strengthened the original conclusions made by Kaminski and Prince (1981).

While hemi-marsh conditions are widely recognized as desirable for maximizing habitat availability for a wide variety of species, it is important to keep in mind that a hemi-marsh state is not optimal for all species in all seasons (Murkin et al. 1997, Masto et al. 2022). For example, diving ducks tend to prefer a greater proportion of open water cover than dabbling ducks (Murkin et al. 1997). Consequently, conservation programs that target large wetland complexes should manage these complexes for the greatest possible range of habitat variability to promote the greatest diversity of wildlife (Murkin et al. 1997).

Research on the hemi-marsh concept has been conducted using experimental plots ranging from 1 hectare (Kaminski and Prince 1981, Masto et al. 2022) to 7 hectares (Murkin et al. 1997). The relatively small scale of these experimental plots raises questions about the appropriate application of the hemi-marsh concept to large wetland complexes. As noted by Murkin et al. (1997), larger complexes need to be managed not only for hemi-marsh habitat, but also for the open water and upland habitat that is used preferentially by a different combination of species.

2.5 Netley-Libau Marsh

Netley-Libau Marsh has a surface area of approximately 26,000 hectares, making it one of the largest coastal freshwater marshes in North America (Grosshans et al. 2004, Wrubleski et al. 2018, Watchorn et al. 2021). Netley-Libau Marsh is located at the south end of Lake Winnipeg, and the Red River runs through the centre of the marsh before splitting into three channels and emptying into the lake (Figure 5). The west side of the marsh (Netley Marsh) is located within the Rural Municipality of St. Andrews and the east side of the marsh (Libau Marsh) is located within the Rural Municipality of St. Clements. The entire marsh complex is located within the Treaty Land Entitlement area of Peguis First Nation. Emergent vegetation in Netley-Libau Marsh consists primarily of *Typha* spp., *Schoenoplectus* spp., and *Phragmites australis* (Grosshans et al. 2004, Watchorn et al. 2021).

Netley-Libau Marsh provides habitat for many birds, mammals, fish, amphibians, and reptiles (Wrubleski et al. 2018). Netley-Libau Marsh is designated as an Important Bird Area under the Canadian Important Bird Area Program (Lindgren and the Netley-Libau Marsh Foundation Inc. 2001, Wrubleski et al. 2018). In addition to its ecological importance, Netley-Libau Marsh also has a great deal of cultural significance, as it provides habitat for animals that are taken by hunters, fishers, and trappers (Wrubleski et al. 2018). The marsh also sustains a wide range of recreational activities, including boating, bird-watching, and eco-tourism (Wrubleski et al. 2018), as well as having a history of use by Indigenous peoples (Stunden Bower 2011, Wrubleski et al. 2018).

Netley-Libau Marsh has undergone a process of marked degradation in recent decades, with emergent vegetation cover declining from 38% to 22% between 1979 and 2001 (Grosshans et al. 2004). Concurrently, the extent of open water has increased by 48% (Grosshans et al. 2004, Watchorn et al. 2021). Rather than a continuous decline, periods of stasis have been punctuated by dramatic shifts in shoreline vegetation (Watchorn et al. 2021). The cover of emergent vegetation has undergone substantial inter-annual variability, with years of extreme flooding and drought having lasting effects on vegetation cover (Watchorn et al. 2021).

Declines in vegetation cover have been exacerbated by the bioturbation impacts of invasive *Cyprinus carpio* (common carp) (Badiou and Goldsborough 2011, Watkinson 2021, Figure 6), and cultural eutrophication (Schindler et al. 2012). Netley-Libau Marsh has also been an area of high

priority for the Manitoba Purple Loosestrife Project since its inception (Lindgren 2003, Figure 7). It is widely suspected that the invasive hybrid cattail *Typha* X *glauca* is common throughout the marsh, as it is common throughout the region (Tangen et al. 2022). However, there has not been a formal survey to verify the extent of hybrid cattail cover throughout Netley-Libau Marsh. Challenges in understanding the spread of hybrid cattail have been exacerbated by difficulties inherent in distinguishing it from *T. latifolia* and *T. angustifolia*. Wasko et al. (2022) recently described a simple and reliable way to identify *T.* X *glauca* using the angle of leaf tips, but this method has not been used to study the spread of *T.* X *glauca* in Netley-Libau Marsh.

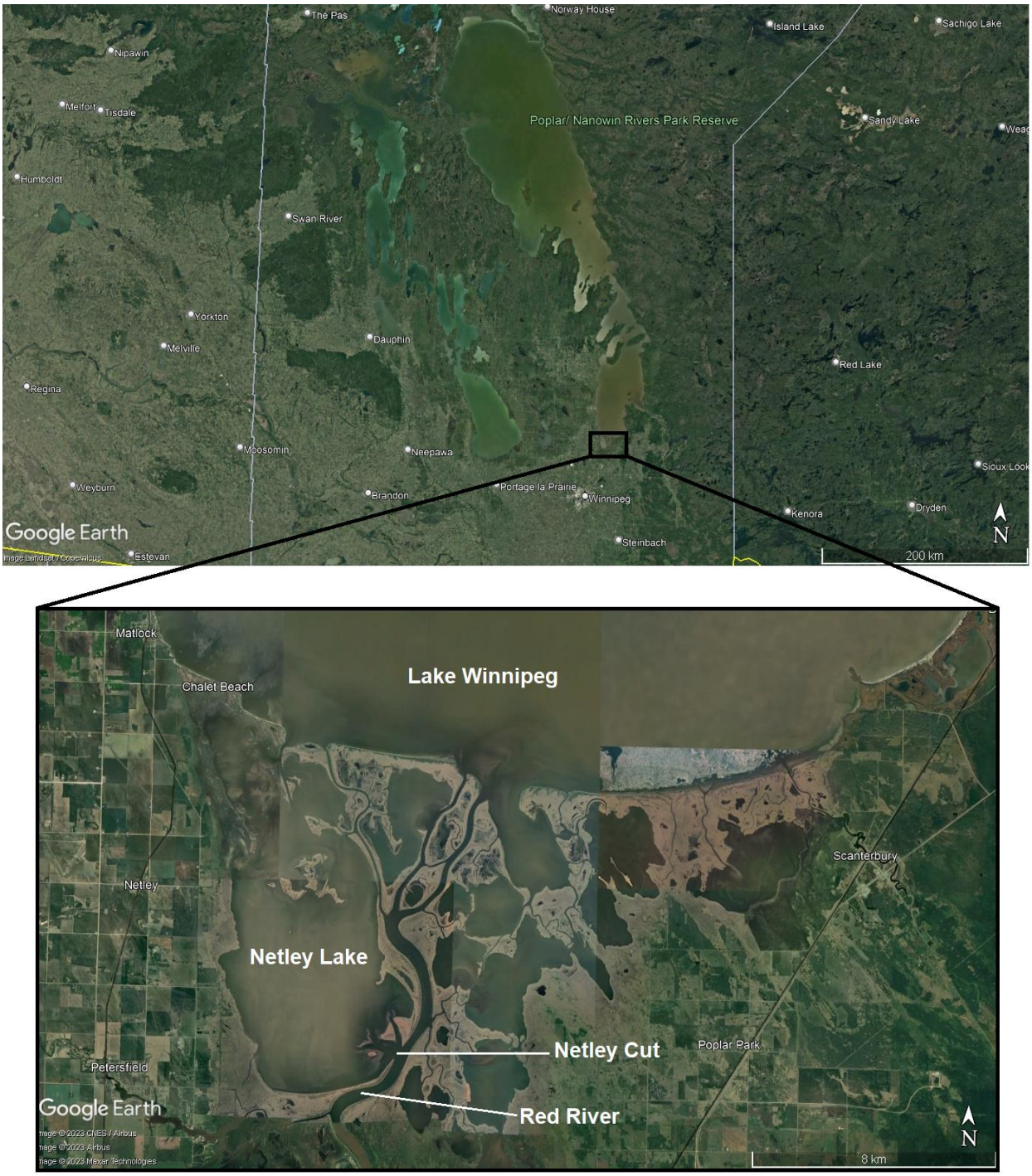

Figure 5. Site map of study area, depicting the location of Netley-Libau Marsh relative to Lake Winnipeg and the Red River.

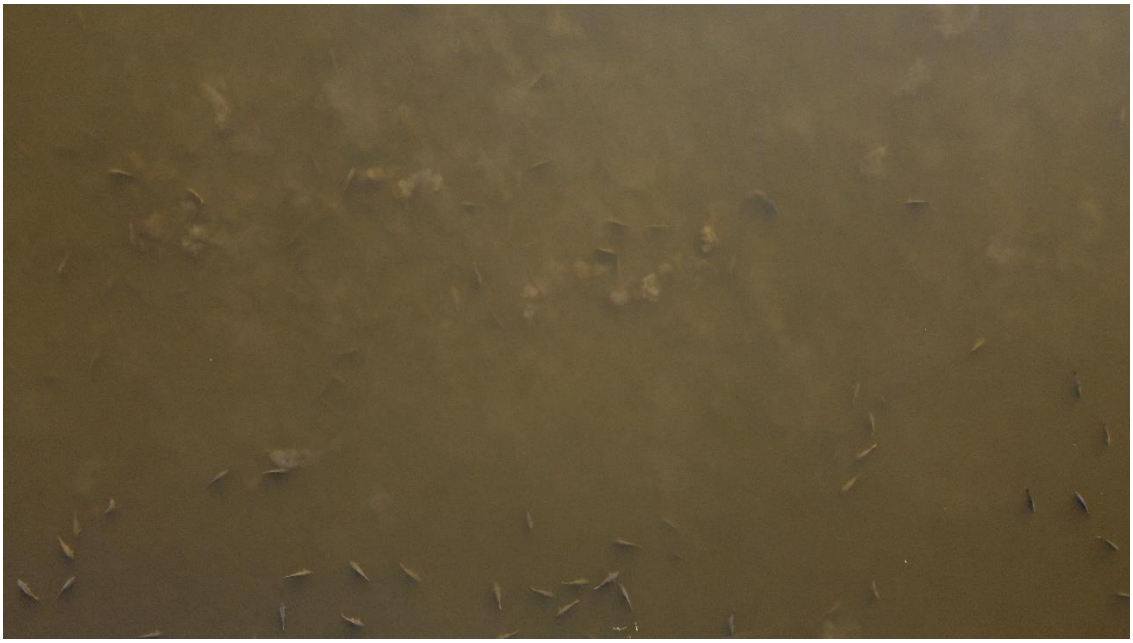

Figure 6. Bioturbation caused by common carp (*Cyprinus carpio*) in Netley-Libau Marsh. Photo taken August 24, 2020, using a DJI Mavic Mini (first generation).

Figure 7. Purple loosestrife (*Lythrum salicaria*) and a large monodominant stand of *Typha* spp. Photo taken August 27, 2020, using a DJI Mavic Mini (first generation).

In conjunction with the degradation of plant communities, there has been a decline in the ecological services provided by Netley-Libau Marsh. More than three decades ago, Verbiwski (1986) described how there had already been a marked decline in the abundance and diversity of wildlife in Netley-Libau Marsh since the 1970s. Cicek et al. (2006) and Grosshans (2014) have both described the substantial capacity of *Typha* spp. for removing nitrogen and phosphorus from the water column. The observed expansion of open water has therefore diminished the capacity of Netley-Libau Marsh to sequester nutrients, which can contribute to harmful algal blooms on Lake Winnipeg (Schindler et al. 2012). Netley Lake is the most bathymetrically dynamic portion of Netley-Libau Marsh, with some locations declining in elevation by 0.8 m between 2003 and 2010 (Watchorn et al. 2021). It is also worth noting that most of the land that has been replaced by open water in Netley Lake since 1990 was dominated by *Typha* spp. (Watchorn 2015).

Two primary mechanisms have been explored to explain the observed decline of emergent vegetation within Netley-Libau Marsh. Firstly, modifications to the historical hydrological cycle of Lake Winnipeg have been induced by Manitoba Hydro for the purpose of generating hydroelectric energy (Grosshans et al. 2004, Kowal 2019, Watchorn et al. 2021). The observed changes in vegetation are consistent with predictions made by vital attribute models of wetland succession, and similar patterns of coastal freshwater wetland decline have been observed following the hydrological stabilization of the Laurentian Great Lakes (Frieswick and Zedler 2007), Lake Manitoba (Grosshans 2001), and the St. Lawrence River (Farrell et al. 2010). With a reduction in the duration and frequency of low-water periods, there is a reduction in the opportunity for the recruitment of plants. Conversely, substantial plant recruitment during the low water of 2003 has had lasting positive effects on the cover of vegetation within Netley-Libau Marsh (Watchorn et al. 2021).

A second proposed mechanism for degradation is that there has been an increase in water flowing through the Western portion of Netley-Libau Marsh. Water from the Red River flowed through Netley Lake from 1913-1929, and then again from 1970-present (Grosshans et al. 2004, Kowal 2019, Watchorn et al. 2021, Figure 5). Estimates indicate that 37% of the water from the Red River now passes through Netley Lake (Kowal 2019, Watchorn et al. 2021). Water flow causes shear stress that can lead to the erosion of sediment and vegetation within Netley Lake (Gurnell et al.

2018, Kowal 2019). Between 1913 and 2004, the Netley Cut grew from 27.5 m to over 400 m in width (Grosshans et al. 2004).

The cumulative influence of the Netley Cut on plant growth is not fully understood, and it warrants further research. In addition to shear stress, the influx of water from the Red River results in an influx of sediment, which can have both negative and positive effects on the growth of aquatic vegetation. Over the short term, there is an increase in turbidity and a reduction in the depth of the euphotic zone – the depth that photosynthetically active radiation can reach. If photosynthetically active radiation does not reach the sediment, it will not be possible for plants to grow (Figure 8). In contrast with the negative influence of turbidity on plant growth, the influx of sediment from the Red River has the potential to have a positive influence on plant growth over the long term, through the progressive deposition of sediment, and the corresponding reduction in water depth. A miniature delta has begun to form downstream of the Netley Cut in Netley Lake (Watchorn et al. 2021), and plant growth has accompanied the local reduction in water depth (Figure 9).

Figure 8. Contrast in turbidity between open water and backwater areas in the Netley-Libau Marsh. High ambient turbidity is driven by wind mixing, bioturbation, and – in some areas – an influx of water from the Red River. Photo taken August 29, 2020, using a DJI Mavic Mini (first generation).

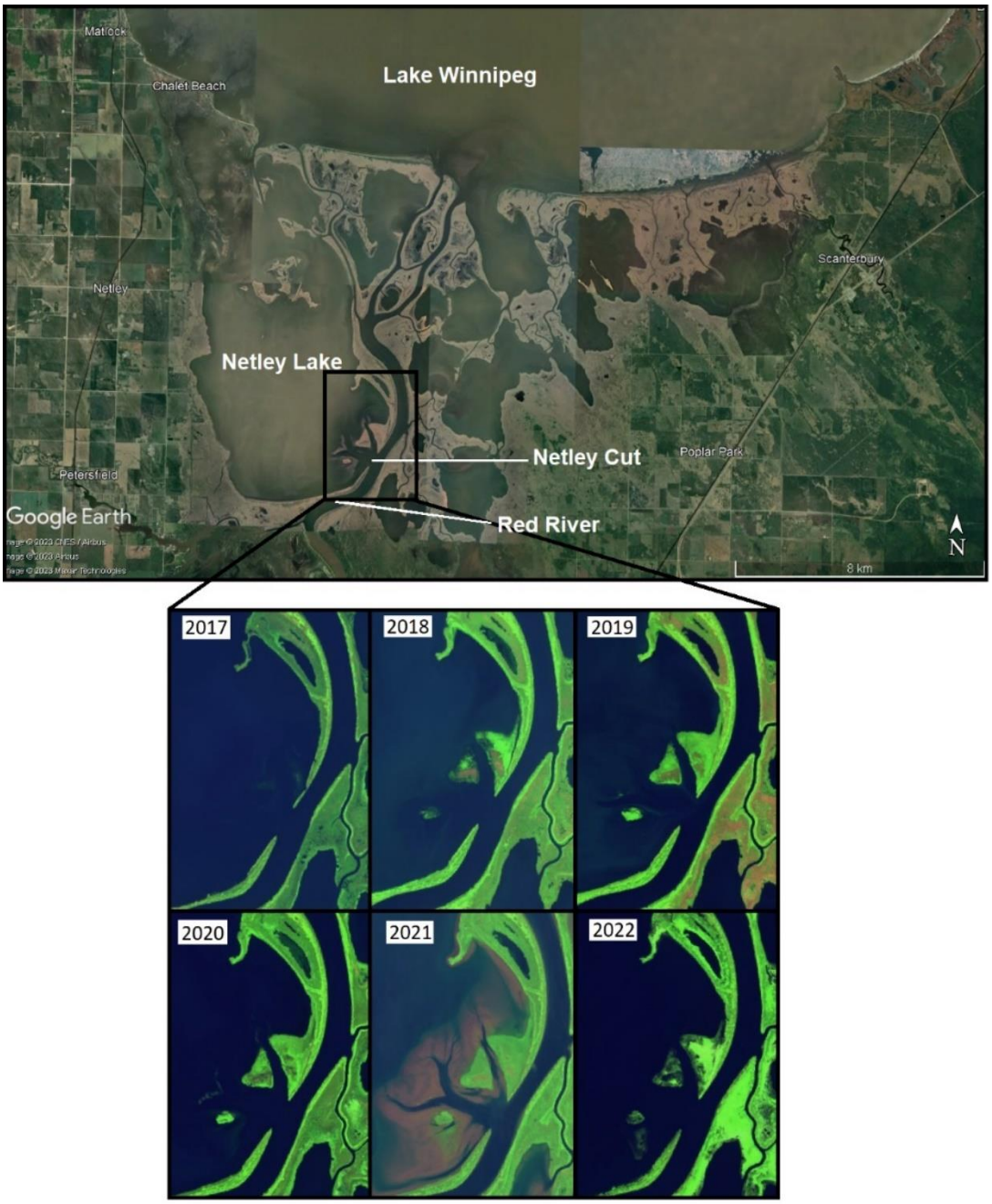

Figure 9. Site map of study area, depicting the progression of vegetation growth in south Netley Lake from 2017 to 2022. Satellite imagery obtained from SentinelHub, using the 'Agriculture' rendering setting (bands 11, 8, 2).

In addition to changes in hydrology and river connectivity, it is important to keep in mind that anthropogenic land use can prevent the marsh from expanding upland during wet years. Kowal (2019) recommended future research on the impacts of land use near coastal marshes. The presence

of anthropogenic barriers (e.g., railway lines) prevents coastal marshes from expanding during periods of increased water levels (Kowal 2019).

Despite anthropogenic changes, it is important to keep in mind that Netley-Libau Marsh is not a static system, and that Lake Winnipeg is continuously expanding southward, due to isostatic rebound (Lewis et al. 2001, Grosshans et al. 2004). While this is not the most immediate problem for Netley-Libau Marsh, the current state of land usage surrounding the marsh can be expected to inhibit the natural migration of the marsh in the coming centuries.

2.6 Community Assembly Rules

2.6.1 Historical Overview

The community assembly framework attempts to explain taxonomic composition through an understanding of (a) dispersal factors, (b) abiotic stressors, and (c) biotic interactions (Kraft et al. 2014). The modern concept of community assembly is best understood as a disjointed amalgamation of relatively recent ideas in animal ecology with much older ideas in the history of plant ecology (Booth and Larson 1999, Grime 2006). The key terms used in community assembly were first defined by Diamond (1975a) in an ornithological context, and these terms have since been adopted and extended by various plant ecologists (Keddy 1992; Weiher and Keddy 1995, 1999, 2011; Toth and van der Valk 2012; van der Valk 2018). However, Booth and Larson (1999) have argued that many of the extensions of the original formulation in Diamond (1975a) echo classic questions of plant ecology, which trace their origins at least as far back as the controversy between Clements (1916) and Gleason (1917), whose models built upon those of even earlier plant ecologists (Warming, 1896, 1909; Schimper 1898, 1903; Cowles, 1899) and geographers (Von Humboldt and Bonpland 1807, de Candolle, 1855).

The original community assembly framework described in Diamond (1975a) is a straightforward application of the theory of island biogeography (MacArthur and Wilson 1967), in combination with biotic interactions. Biotic interactions have typically been understood through the principles of competitive exclusion (Hardin 1960) and niche differentiation (Hutchinson 1957, MacArthur and Levins 1967, Diamond 1978), as described in contemporary literature. Through an understanding of dispersal pathways and competitive interactions, the original formulation of community assembly rules provided a framework for predicting incompatible combinations of species (Diamond 1975a, Gilpin and Diamond 1982), and designing natural protected areas to maximize biodiversity (Diamond 1975b).

More recent extensions of the community assembly concept have emphasized the importance of abiotic filters (Keddy 1992, Weiher and Keddy 2011, Toth and van der Valk 2012). In the context of plant ecology, the addition of abiotic filters reflects the pervasive influence of earlier plant ecologists, rather than a truly novel addition to the framework described in Diamond (1975a) (Booth and Larson 1999). This historical trajectory is best illustrated by the way that the framework

for understanding community assembly has absorbed van der Valk's (1981, 2018) 'environmental sieve' model of wetland succession (Toth and van der Valk 2012, Kraft and Ackerly 2014), which van der Valk (1981) refers as a "Gleasonian" model. Whereas van der Valk (1981) makes the influence of earlier plant ecologists explicit, other authors have been less clear about these influences (Keddy 1992, Weiher and Keddy 1995), much to the consternation of their colleagues (Wilson 1999, Booth and Larson 1999).

Whereas it is no longer controversial to include abiotic filters within the framework of community assembly, it is important to understand the extent to which the addition of these factors has broadened the scope of the community assembly concept, far beyond what was contemplated within Diamond (1975a). These differences are especially pronounced in models that place a greater emphasis on abiotic filters than on dispersal or biotic interactions (e.g. van der Valk 1981, 2018). An overview of this change is depicted in Figure 10.

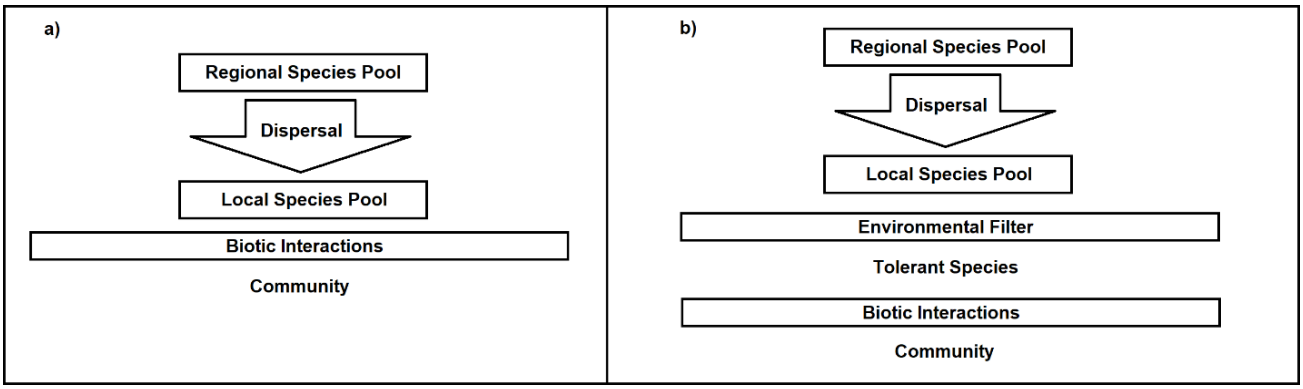

Figure 10. Simplified comparison between the community assembly framework presented in Diamond (1975a) [Figure a], and that described in the modern literature [Figure b].

Recently, some authors have criticized the framework of community assembly rules for overestimating the importance of abiotic factors in structuring communities (Kraft et al. 2015), and for failing to integrate developments in coexistence theory into the biotic component of the framework (Chesson 2000, Cadotte and Tucker 2017). Other authors have noted that the community assembly framework needs to develop a better understanding of the physiological relationship between ramets of a single genet (Elgersma et al. 2015, Liu et al. 2016). The

translocation of resources between ramets subverts the conceptual underpinning of individual-based models by providing three distinct levels of individuality – ramets, genets, and physiologically integrated units, which may comprise a subset of ramets within a single genet (Price and Hutchings 1992, Song et al. 2013). This omission may be particularly problematic in systems like coastal freshwater marshes, which tend to be dominated by plants that have multiple ramets per genet (see van der Valk 2012).

2.6.2 Abiotic Filters and Gradients

Abiotic factors can influence community composition both directly and indirectly. There is some disagreement within the existing literature about whether terms like 'filter' and 'sieve' should be restricted to circumstances in which a particular taxon is completely excluded from an environment, or whether similar terminology can be extended to circumstances in which the outcome of biotic interactions is influenced by a differing degree of tolerance to an abiotic stressor (Kraft et al. 2015). Some authors have been criticized for operating under the assumption that abiotic factors drive observed patterns in community composition, without considering the possibility that similar patterns could be explained through reference to competitive interactions (Cadotte and Tucker 2017). Part of this criticism stems from an overall dearth of large-scale manipulative experiments, and the resulting challenge in differentiating the direct and indirect effects of abiotic stressors.

The most important abiotic filters in coastal freshwater marshes are water depth and periodicity (van der Valk 1981, 2018). The influence of water as an abiotic filter was studied in a rare example of a large-scale manipulative experiment during the Marsh Ecology Research Program (MERP), which took place in Delta Marsh, MB, from 1980 to 1989 (Murkin et al. 2000). The MERP experiments made it possible to test the predictions of the 'environmental sieve' model of succession in wetlands (van der Valk 1981, 2018). This model was developed out of disturbance-driven models of succession in forests (Noble and Slatyer 1977, Connell and Slatyer 1977), as well as the individualistic model of succession, described in Gleason (1917, 1927, 1939). Seabloom et al. (2001) extended the wetland model using cellular automaton simulations and found that the predictions of the model were generally accurate.

The impacts of wave energy can be exacerbated by tidal forces, such that shear stress and sediment burial prevent seedlings from surviving in the shoreline zone (Fagherazzi and Wiberg 2009, Ge et al. 2019). Vegetative propagules have a greater capacity than seeds to generate new ramets under stressful conditions, in both herbaceous emergent plants (Ge et al. 2019), and woody plants (Politti et al. 2018, Gurnell et al. 2018). Conversely, long-lived seeds permit plants to optimize growth and reproductive success in the face of unpredictable environmental conditions (Venable 1989, Fenner and Thompson 2004, Tellier et al. 2011, Živkovic and Tellier 2018). Persistent seed banks have been documented in arid regions (Venable and Lawlor 1980; Fenner and Thompson 2004;

Živkovic and Tellier 2018), as well as temperate regions with greater precipitation (Pederson 1983, Živkovic and Tellier 2018). In North American freshwater marshes, persistent seed banks allow a diversity of wetland plant species to thrive within the variability of wet-dry cycles (Frieswyk 2005, Frieswyk and Zedler 2007, van der Valk 2018, Wrubleski et al. 2018, Wilcox et al. 2018).

Exposure to wind and wave energy is an abiotic stressor that can influence the distribution of aquatic vegetation in lakes and wetlands (Keddy 1982, Mason et al. 2018). Among modern authors, the discussion of abiotic stress has generally fallen under the discussion of environmental gradients (Whittaker 1967, Keddy 1982, Tilman 2004). However, Warming (1897) describes the relationship between *Typha* spp. and exposure in absolute terms, which would fit the modern definition of an environmental filter in the strictest sense of the term: "[…] *Typha, Scirpus,* and *Phragmites*. […] Of the three, *Typha* is the one which is most readily torn out of place, and consequently grows in more sheltered places." (in Clements 1916). This passage asserts that *Typha* spp. is not simply subjected to a competitive disadvantage, but that it is physically torn out of place.

Gradient analysis was originally introduced by Whittaker (1951, 1967) as a method that supported the continuum concept of vegetation, in contradistinction to the community concept (Dyakov 2010, Nicolson 2016). The continuum concept of vegetation holds that species composition changes gradually through space along a variety of environmental gradients, rather than being grouped into distinct communities with a shared history of coevolution. Gradient analysis can be divided into direct gradient analysis and indirect gradient analysis (Whittaker 1967, Dyakov 2010). Direct gradient analysis, which has remained largely unchanged since its origin (Whittaker 1951, Dyakov 2010), examines gradients that have a direct physiological effect on the growth of plants, such as temperature, soil pH, or exposure stress (Keddy 1982, Austin and Smith 1989). Indirect gradient analysis examines complex gradients, such as elevation, in which plants are influenced indirectly by elevation, through the direct influence of variables like temperature and precipitation, which vary along the elevation gradient (Austin and Smith 1989, Dyakov 2010).

Whereas gradient analysis was originally described in a way that subverted the notion of distinct communities within a delineated spatial area, it should be noted that environmental gradients are abstract dimensions that need not change continuously through space (Austin 1985, Dyakov 2010). For example, Keddy (1982) used gradient analysis to describe vegetation composition along an

exposure gradient, using 25 randomly selected points along the shoreline of a lake. An important difference between elevation and exposure gradients is that exposure gradients can only be studied through a comparison of multiple sites with varied levels of exposure. This is a function of the fact that plants shield each other from wave action in a way that has no meaningful analog along an elevation gradient. Whereas it is a simple matter to record water depth at various points along an elevation gradient at a single shoreline location, it is virtually impossible to estimate the difference in wave energy between a ramet at the land/water interface, and a ramet that experiences wave energy after it has been dissipated by other ramets. Whereas elevation gradients may be gradual or steep at any given location, exposure stress may be closer to a binary distinction within a single shoreline location.

2.6.3 Biotic Interactions

Most authors who have emphasized the importance of biotic interactions within the framework of community assembly have focused on the role of competition in structuring community composition (Kraft and Ackerly 2014). In Diamond's (1975a) original formulation, the overall purpose of community assembly rules was to predict incompatible combinations of species, in which one species would necessarily be extirpated through a process of competitive exclusion (Hardin 1960), driven by the principle of limiting similarity (MacArthur and Levins 1967).

Some authors have criticized the community assembly framework for failing to incorporate recent developments in coexistence theory (Kraft et al. 2015). Coexistence theory deviates from traditional predictions about competitive exclusion by emphasizing the fact that an observed similarity in a particular phenotype may reflect similarity in fitness, rather than similarity in niche requirements. In cases where phenotypic similarity is a function of similarity in fitness, coexistence theory predicts that these similarities will not lead to competitive exclusion (Chesson 2000, Cadotte and Tucker 2017).

When studying patterns of distribution among species, it can be challenging to distinguish between the influence of biotic interactions, abiotic stressors, and constraints on dispersal (D'Amen et al. 2017). Elgersma et al. (2015) has stated that community assembly rules need to develop a better understanding of the biotic interactions between the ramets of a single genet. A weakness of most community assembly frameworks is that they define a community as an assemblage of individuals, without addressing the fact that there are three levels of individuality in clonal plants – genet, ramet, and physiologically integrated unit (Price and Hutchings 1992, Song et al. 2013).

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Chapter 3. Developing Low-Cost Manual Methods for Mapping Shoreline Vegetation in Netley-Libau Marsh Using a Micro-RPA

Abstract

Netley-Libau Marsh has undergone a marked process of degradation during the past century. One of the most noteworthy changes has been a reduction in aquatic vegetation, along with a corresponding increase in open water cover. The shoreline can be understood as a critical zone, where landforms may expand, recede, or remain stable, based on interactions between vegetation and hydrogeomorphic processes. To better understand the composition of shoreline plant communities, I developed an accessible method for using a small remotely piloted aircraft (micro-RPA) to map shoreline vegetation in prairie freshwater marshes. This method consists of two manual shoreline surveys at two different altitudes: 120 m and 10 m. Predominant taxonomic groups of vegetation were usually distinct at an altitude of 120 m, with occasional exceptions. The minimal regulations and low cost associated with small RPAs remove many of the barriers to entry for their use in field ecological research. Moreover, this method is likely to be robust from June to October, as it is based primarily on the shape of leaves and stems, rather than colour or spectral signature. This method is also inexpensive, as it can be done using open-source software (WebODM).

3.1 Introduction

Aerial imagery has been used in ecological research on a relatively coarse spatial scale for nearly a century (Melvill Jones and Griffiths 1925, MNM 1925, Bourne 1928, Fischer 1928). Notably, Anker (2001) has argued that an aerial perspective contributed to Arthur Tansley's development of the modern ecosystem concept in ecology (Tansley 1935). Nearly a century later, modern authors have described the advance of inexpensive remotely piloted aircraft (RPA) technology as "revolutionary" to vegetation survey methods in remote field locations (Madden et al. 2015). The low-altitude hovering capabilities of modern RPAs have created possibilities for novel research methods at a medium-scale, which have never been possible using airplanes or traditional survey methods at the ground-level. By increasing the availability of high-quality aerial imagery, the proliferation of RPA technology has the potential to radically alter what is possible for wetland research (Kalacska et al. 2017).

Remotely piloted aircraft provide a relatively inexpensive avenue for ecological field researchers to acquire high quality aerial imagery. Whereas RPAs that weigh more than 250 grams must be registered with Transport Canada and require that operators obtain a basic operations certificate (*Canadian Aviation Regulations*, May 29, 2023), micro-RPAs (< 250 g) can be operated without a license or registration, provided that the operator conforms to similar restrictions that are imposed upon the operation of larger RPAs (*Aeronautical Information Manual*, March 23, 2023). In combination with the proliferation of open-source mapping software, the proliferation of affordable, accessible, RPAs has vastly extended the feasibility of using aerial imagery in basic ecological research.

In many countries, including Canada and the United States, RPAs are restricted to a maximum altitude of 120 m (*Canadian Aviation Regulations*, May 29, 2023). Because of this altitude restriction, piloted aircraft still provide opportunities for increasing the scale of surveys in a way that is not (legally) possible for most RPA operators. Despite altitude restrictions, the proliferation of RPA technology has the potential to vastly reduce the amount of time required to conduct ecological research in remote locations (Madden et al. 2015, Kalacska et al. 2017). Moreover, the ability to safely hover at low altitudes provides opportunities for intermediate-scale research that are not feasible with piloted aircraft.

Recent advances in RPA technology have removed financial barriers that previously limited the availability of high-quality aerial imagery (Kalacska et al. 2017, Rhee et al. 2018). The availability of high-resolution aerial imagery makes it possible to extend traditional vegetation sampling methods to a larger scale, with relatively low sampling effort. Aerial images can be stitched together to produce orthomosaic maps and three-dimensional point cloud models (Madden et al. 2015, Esposito et al. 2017, Doughty and Cavanaugh 2019, Duffy et al. 2018). High resolution cameras and open-source software make it possible to delineate vegetation classes and generate vegetation cover maps (Grosshans et al. 2004, Kalacska et al. 2017, Doughty and Cavanaugh 2019).

Aerial surveys have been instrumental in describing the upland and emergent marsh vegetation of Delta Marsh (Shay et al. 1999, Grosshans 2001), and Netley-Libau Marsh (Verbiwski 1986, Grosshans et al. 2004). Orthomosaic maps generated from RPA imagery have also been used to identify submerged and emergent vegetation (Chabot et al. 2018, Meneses et al. 2018, Doughty

and Cavanaugh 2019). In some cases, high resolution aerial images have facilitated reliable identification of macrophytes to the species-level (Husson et al. 2013, Chabot et al. 2018).

The methods presented here use inexpensive low-altitude aerial imagery and open-source software to map shoreline plant communities in Netley-Libau Marsh. Whereas previous studies have provided a clear picture of vegetation changes in Netley-Libau Marsh on a macro-scale, these studies have limited spatial resolution, due to their use of high altitude (1920 m) aerial imagery (Grosshans et al. 2004), and satellite imagery (Watchorn et al. 2021). An additional advantage of the methods presented here is that they can be carried out using a micro-RPA, which does not require a basic operations certificate.

3.2 Problem Statement and Objective

It can be challenging to conduct vegetation surveys in remote wetland environments. Whereas aerial surveys have allowed researchers to overcome some of these challenges for nearly a century, aerial methods using piloted aircraft can be prohibitively expensive. Moreover, traditional aerial surveys provide limited spatial resolution, due to a combination of high altitude and high speed. Recent advances in remotely piloted aircraft (RPAs) have created new possibilities for vegetation surveys on an intermediate scale, through the ability to hover at low altitudes. These technological advances have also removed some of the financial barriers associated with traditional aerial methods. Despite the potential demonstrated by RPAs, new methods need to be developed to make use of these technological advances.

My objective was to develop an accessible method for using a micro-RPA (DJI Mavic Mini) to map shoreline vegetation in prairie freshwater marshes.

3.3 Description of Aircraft – DJI Mavic Mini

Aviation laws in Canada, the USA, and many other jurisdictions require a basic operations license for the operation of any RPA between 250 g and 25 kg. At the time of data collection in 2020, the first generation of the DJI Mini was the only model available in the sub-250 g class, and it was simply called the “Mavic Mini” (Figure 11). Since 2020, newer models of the DJI Mini have been released (Mini 2, Mini 3, and Mini 3 Pro). The equivalent of the first generation can still be purchased from DJI, but it has now been renamed the Mini SE. However, one important difference is that the Mini SE uses a different battery, which is interchangeable with the Mini 2, rather than the Mavic Mini.

A basic operations certificate is not required for the operation of the Mavic Mini, or for the later generations of the DJI Mini, provided that no accessories are added to increase the total weight of the RPA. Nevertheless, operators must follow all applicable laws while flying the Mavic Mini. As with larger RPAs, operators must not exceed 120 m in altitude. Visual line-of-sight must be maintained at all times, but this may be accomplished by proxy through the aid of a visual observer. Visual line-of-sight must be 'unassisted', in the sense that pilots or observers must be able to see the aircraft without the assistance of binoculars or telescopes.

The Mavic Mini is considerably less expensive than larger RPAs. Accessories are also less expensive. At the time of data collection in 2020, the base cost of the Mavic Mini (first generation) was approximately $500.00 CAD, and variations of the DJI 'Fly More Combo' could be obtained for $600-$650 CAD. The 'Fly More Combo' comes with extra cables, a propeller guard, extra propellers, a hard carrying case, and three batteries – providing a total maximum of 90 minutes fight time. The 'Fly More Combo' is adequate for many field purposes, but a fitted waterproof carrying case can be purchased for $50-$70, and additional batteries can be purchased for $60 each. Since the time of data collection in 2020, DJI has released newer versions of the Mavic Mini, which range in price from $419.00 - $759.00 USD. The original Mavic Mini has been re-released as the Mini SE, and it can be purchased for $279.00 USD as of June 2023.

Figure 11. DJI Mavic Mini (first generation) micro-RPA (top) and controller (bottom). A phone or tablet is connected to the controller during flight, displaying the perspective of the Mavic Mini during flight.

It is important to note that the propeller guards included in the Fly More Combo bring the weight of the Mavic Mini above the 250-g legal threshold. Consequently, it is illegal to use the guards without a license and registration. The same legal restriction applies to virtually all other accessories that can be purchased for the Mavic Mini, except for lens filters, which generally weigh less than 1 g. Some operators may feel more comfortable in the field with the propeller guards, or with a landing-gear attachment, but it is important to be aware of the legal implications of using

these accessories. It is also important to keep in mind that many accessories will increase the effects of wind resistance on the Mavic Mini, and this can make it more vulnerable to being blown away in strong winds.

The compact frame of the Mavic Mini presents both advantages and challenges in the field, when compared with larger RPAs. The key weakness of the Mavic Mini is that it can be carried away by wind more easily than larger RPAs. Consequently, there are fewer days in the season that are viable to fly mapping missions, compared with larger RPAs that can safely be flown in moderate wind conditions. On the other hand, larger RPAs will also produce better photographic results under calm conditions, since windy conditions cause herbaceous plants to move in the breeze. Because of this, wind limitation may not be a practical problem for many studies.

Another major advantage of the small size is that it allows the operator to take off and land within the confined space of a canoe or small boat. Flight tests with a 4-kg Bluegrass Parrot Fields RPA indicated that a much more sophisticated landing apparatus would be necessary to ensure safe operation deep within Netley-Libau Marsh, where operators cannot expect to find flat ground to set up a conventional landing pad. Furthermore, the Parrot had 10-12 minutes of flight time per battery, compared with 25-35 minutes for the Mavic Mini. Extra batteries cost approximately $300 for the Parrot, compared with $60 for the Mavic Mini (Table 1).

Table 1. Comparison between Parrot Bluegrass Fields and DJI Mavic Mini (first generation). Note that the Mavic Mini has been re-released as the Mini SE, with the same specifications as the original Mavic Mini. However, one important difference is that the Mini SE uses a different battery, which is interchangeable with the Mini 2, rather than the Mavic Mini.

	Parrott Bluegrass Fields	DJI Mavic Mini (1st generation)
Weight (g)	4,000	250
License	Required	Not Required
Safety	Dangerous blades	Blades minimally dangerous
Camera/Sensor	Multispectral, fixed angle	Natural light, 3-axis gimbal
Flight time	10-12 minutes	25-35 minutes
Maximum Visible Distance	> 1 km	< 500 m
Base Cost	$3,500 CAD	$350 CAD
Extra batteries	$300 CAD	$60 CAD (Mavic Mini)/$80 CAD (Mini SE)

3.4 Development of the Mapping Technique – Image Acquisition

I purchased the Mavic Mini in June 2020, and I spent June and July taking over 10,000 test images to assess the effectiveness of the Mavic Mini as a tool in mapping marsh vegetation, based on the vegetation classes described in Grosshans et al. (2004). At the time of data collection in 2020, there was no support for the DJI Mavic Mini with automated mission planning software, such as Pix4D or Dronelink. Consequently, all imagery was captured during manual flight using the DJI Fly application. This period was largely exploratory, and I spent a great deal of time practicing basic flight techniques, while taking automatic photographs every two seconds. During these early flight tests, I also made sure to take photographs that I could use to practice creating orthomosaic maps from aerial imagery. I used WebODM to generate many 'practice' orthomosaics during this period (more details below).

After generating several orthomosaics at an altitude of 120 m, and verifying whether the perceived vegetation classes were accurate via ground truthing, it became clear that it was much easier to identify shoreline vegetation than upland vegetation. This is because upland plant communities are much more heterogeneous in the study area, whereas the shoreline zone tends to be dominated by a small number of taxa (Grosshans et al. 2004). For similar reasons, shoreline vegetation identification was also more challenging along relatively heterogeneous shorelines. Overall, most observed shorelines were fairly homogeneous, and in many cases accurate identification of *Typha* spp. *Schoenoplectus* spp., and *Phragmites australis* was possible, based solely the 120-m orthomosaic.

After confirming that the 120-m orthomosaics were insufficient in heterogeneous areas, I decided to incorporate a second survey of each shoreline area at an altitude of 10 m. Rather than aiming the camera directly downwards at -90 degrees, the second survey aimed forwards at an oblique angle, roughly parallel to the shoreline, with the horizon just barely in view. The 10-m survey provided a clear view of the vertical profile of the shoreline vegetation, clarifying the ambiguity in most heterogeneous areas. The viewing angle of the 10-m images made them unsuitable for generating orthomosaics, but they provided a detailed qualitative supplement to the 120-m orthomosaics. The oblique angle of the 10-m survey also provided a much wider field of view than would be possible if the camera were facing directly downwards.

I based my shoreline vegetation classes loosely on the classes described in Grosshans et al. (2004). In contrast with Grosshans et al. (2004), which used near-infrared imagery to distinguish subtle spectral differences between classes, my method distinguishes classes primarily based on the physical characteristics of leaves and stems. The emphasis on vegetative characteristics makes this method robust from June to September.

The Vegetation Classes are as follows:

1. Unvegetated
2. *Typha* spp. (cattail)
3. *Phragmites australis* (giant reed)
4. *Schoenoplectus* spp. (bulrush)
5. *Bolboschoenus fluviatilis* (river bulrush)
6. *Salix* spp. (willow)
7. Forb
8. Graminoid (grasses, sedges, and rushes)
9. *Lythrum salicaria* [when flowering] (purple loosestrife)
10. *Lemna* spp. (duckweed)
11. *Sagittaria* spp. (arrowheads)

Most shoreline locations in Netley-Libau Marsh are dominated by a combination of *Typha* spp., *Schoenoplectus* spp., and *Phragmites australis*. (Verbiwski 1986, Grosshans et al. 2004) Consequently, these three taxa are the most important to be able to clearly distinguish from one another. Viewed from the side at an altitude of 10 m, these three genera have very distinct features.

Part of the structural support of *Typha* spp. leaves is a subtle twisted orientation, which makes it possible for leaves to remain upright. Inflection points within this twisted shape can clearly be seen from the side, and they are not easily confused with the features of any other common taxa. *Phragmites australis*, being the only true grass of the three genera, has distinct features that are common to grasses. In particular, *Phragmites australis* can easily be identified any time that a shoot is observed with a leaf that is clearly branching off from a stem. *Schoenoplectus* spp. lacks the distinct features described for *Typha* spp. and *Phragmites australis*, and may appear perfectly vertical from the side, or slightly flexed/bowed. *Schoenoplectus* spp. can also be distinguished by having more space between individual ramets than is present in *Typha* spp. and *Phragmites australis*.

Colour is less reliable than shape as an identifying characteristic, since colour can vary based on seasonality, physiological stress, solar conditions, and camera settings. Moreover, visible light photography is limited in the extent to which it distinguishes between shades of green. Despite these limitations, colour is valuable as a secondary identification characteristic. All else being equal, *Schoenoplectus* spp. tends to be the darkest shade of green, *Typha* spp. tends to be intermediate, and *Phragmites australis* tends to be the lightest shade of green.

Floral characteristics have limited value in this method, and they are inevitably limited to the flowering period for any given taxon, which will not necessarily converge with the flowering period of other plants within the study area. Nevertheless, the inflorescences of *Phragmites australis* are very distinct, and can therefore be used as an additional tool in classification. *Lythrum salicaria* is the only class that was distinguished primarily based on floral characteristics. This is because *L. salicaria* is the only taxon that was clearly distinguishable based on floral characteristics, as a result of its conspicuous inflorescences. *Lythrum salicaria* should be grouped with the 'Forb' class prior to mid-August, after which *L. salicaria* inflorescences will be in full bloom. Nevertheless, it is important to keep in mind that not all plants will bloom in any given year, so any analysis of blooming *L. salicaria* will be an underestimate of total cover.

In addition to vegetation classes, I also grouped shoreline zones into a series of classes, based on the presence of visible physical damage (Figures 13-26):

1. No visible damage
2. Broken shoots
3. Flattened shoots
4. Exposed roots/rhizomes
5. Sediment burial
6. Bare shelf

3.5 Summary of the Mapping Technique – Image Acquisition

The methods for image and video acquisition can be summarized as follows:

1. Manual shoreline survey at an altitude of 120 m, with the camera angled directly downwards (-90 degrees) (Figure 12). Automatic photograph every two seconds, while moving at a speed of approximately 4 m/s (Cinesmooth Mode). This method produces redundant imagery, which can be reduced during the processing stage. Image overlap will be well in excess of the minimum 60%, using every second photo. Redundant imagery can be valuable if there are any problems with individual photos.

2. Manual shoreline survey at an altitude of 10 m, with the camera angled forwards at an oblique angle so that the horizon is visible at the top of the photograph (Figure 12). Adjust the camera angle as needed to compensate for solar glare and over-exposure. Automatic photograph every two seconds, while moving at a speed of approximately 4 m/s (Cinesmooth Mode).

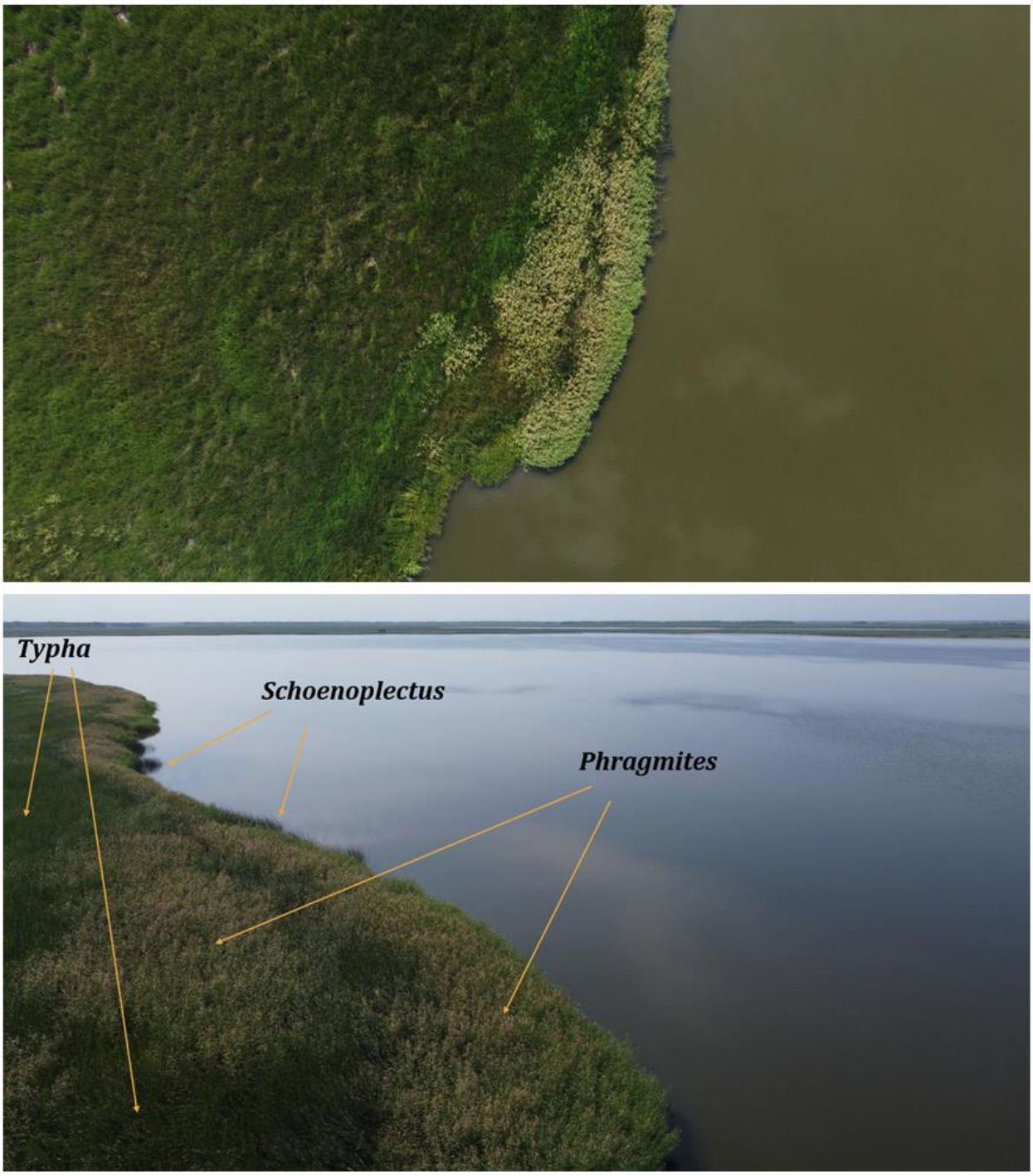

Figure 12. Example of aerial view for two surveys at 120 m (top) and 10 m (bottom). Photos taken August 29, 2020, using a DJI Mavic Mini (first generation).

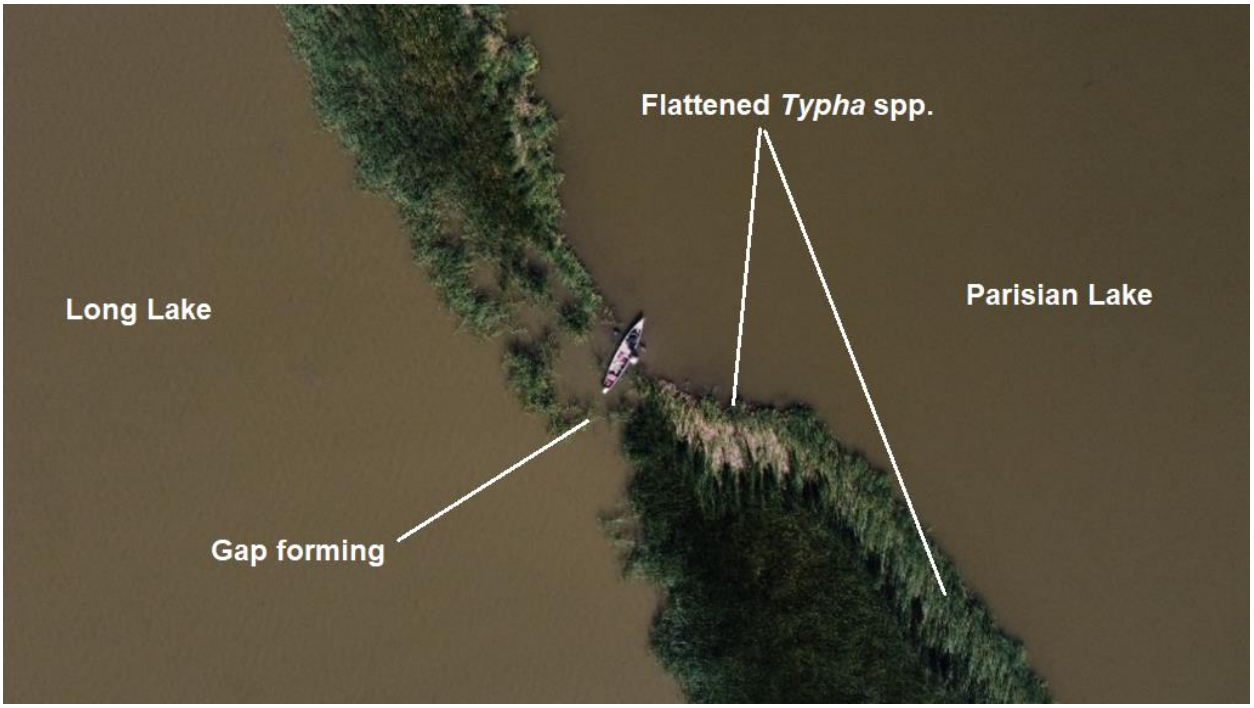

Figure 13. Flattened *Typha* spp. shoots. Gap beginning to form in island that separates Parisian Lake from Long Lake. Photo taken August 29, 2020, using a DJI Mavic Mini (first generation).

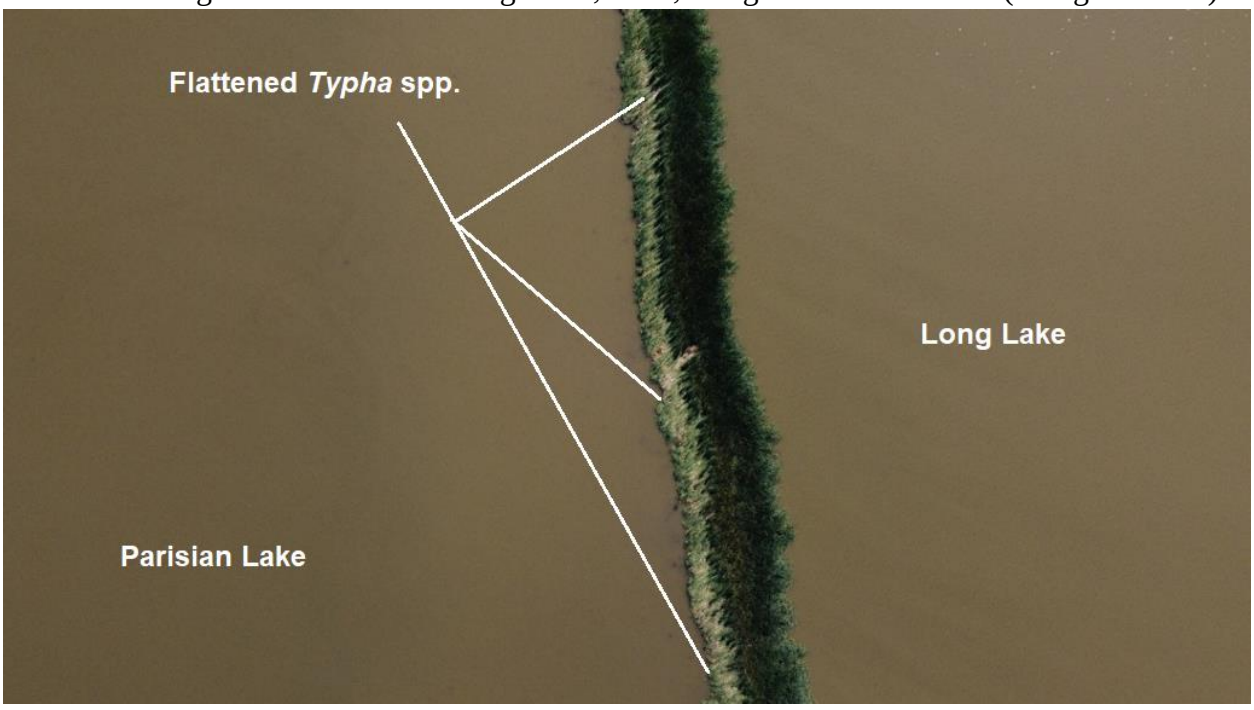

Figure 14. Flattened *Typha* spp. shoots along island that separates Parisian Lake from Long Lake. Photo taken August 29, 2020, using DJI Mavic Mini (first generation).

Figure 15. Exposed *Typha* spp. rhizomes and eroding vertical cliff along south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 16. Exposed *Typha* spp. rhizomes and bare shelf along south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 17. Flattened and broken *Typha* spp. shoots along the south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 18. Sediment suspension and deposition along the south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 19. Sediment suspension and deposition along the south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 20. Exposed *Typha* spp. rhizomes and bare shelf along south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 21. Exposed *Typha* spp. rhizomes and bare shelf along south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 22. Flattened *Typha* spp. shoots along south shore of Anderson Lake. Photo taken August 29, 2020, using Apexcam waterproof action camera.

Figure 23. Sediment suspension and deposition in north Netley Lake. Photo taken August 27, 2020, using DJI Mavic Mini (first generation).

Figure 24. Flattened and broken *Typha* spp. shoots, as well as bare shelf, along island separating Parisian Lake from Long Lake. Photo taken August 29, 2020, using DJI Mavic Mini (first generation).

Figure 25. Flattened and broken *Typha* spp. shoots, as well as bare shelf, along island separating Parisian Lake from Long Lake. Photo taken August 29, 2020, using DJI Mavic Mini (first generation).

Figure 26. Flattened and broken *Typha* spp. shoots, as well as bare shelf, along south shore of Anderson Lake. Photo taken September 2020, using DJI Mavic Mini (first generation).

3.6 Development of the Mapping Technique – Generating Practice Orthomosaics

In June 2020, I began generating practice orthomosaics using WebODM. Initially, I generated orthomosaics from a variety of different altitudes, with the intent of creating extremely detailed orthomosaics from a very low altitude. I discovered quickly that low-altitude imagery can result in substantial motion blur, and that it becomes necessary to stop the Mavic Mini for each photo, rather than taking automatic photos during a slow flight. Cinesmooth mode limits flight speed to 4 m/s, which works well for automatic photos at 120 m, but is much too fast at 30 m.

I was able to make high quality orthomosaics from 30 m, but this required a great deal of finesse with the controller to keep my speed within the range of 1-1.5 m/s, so that I could take automatic photos every 2 seconds. At 30 m, I found that it was easier to simply stop flying and take each photo manually (Figure 27). While I was satisfied with how some of these 30-m orthomosaics turned out, it was clear that these tedious methods would not be feasible on a larger scale. Because of these feasibility issues, I decided that it would be preferable to generate all orthomosaics from 120 m (Figure 28), and to supplement these orthomosaics with a second survey at 10 m. The forward viewing angle of the 10-m survey eliminated the problems of motion blur, while also providing much greater clarity, without requiring a reduction in flight speed.

I also attempted to generate many practice orthomosaics at 5 m and 15 m. These failed attempts indicated to me that extremely low-altitude orthomosaics are not advisable for herbaceous vegetation. WebODM almost always crashed when I tried to stitch photos taken from an altitude of 5 m. In reviewing the images, it seems that the photos are too dissimilar for the software to recognize the overlapping features, even when there is much more than the required 60% spatial overlap. I obtained unreliable results with images taken from 15 m, and I was not able to consistently produce high quality orthomosaics from this altitude. Most of these low-altitude attempts simply resulted in an error message from WebODM, whereas a minority of them turned out reasonably well.

My recommendation for low-altitude imagery would be to use RPAs as an extension of traditional quadrat sampling methods, such that each individual photo would represent a quadrat within the sampling design. This would allow researchers to use extremely detailed low-altitude photographs,

which would provide the greatest possible level of visual detail, while also circumventing the difficulties of stitching overlapping low-altitude imagery.

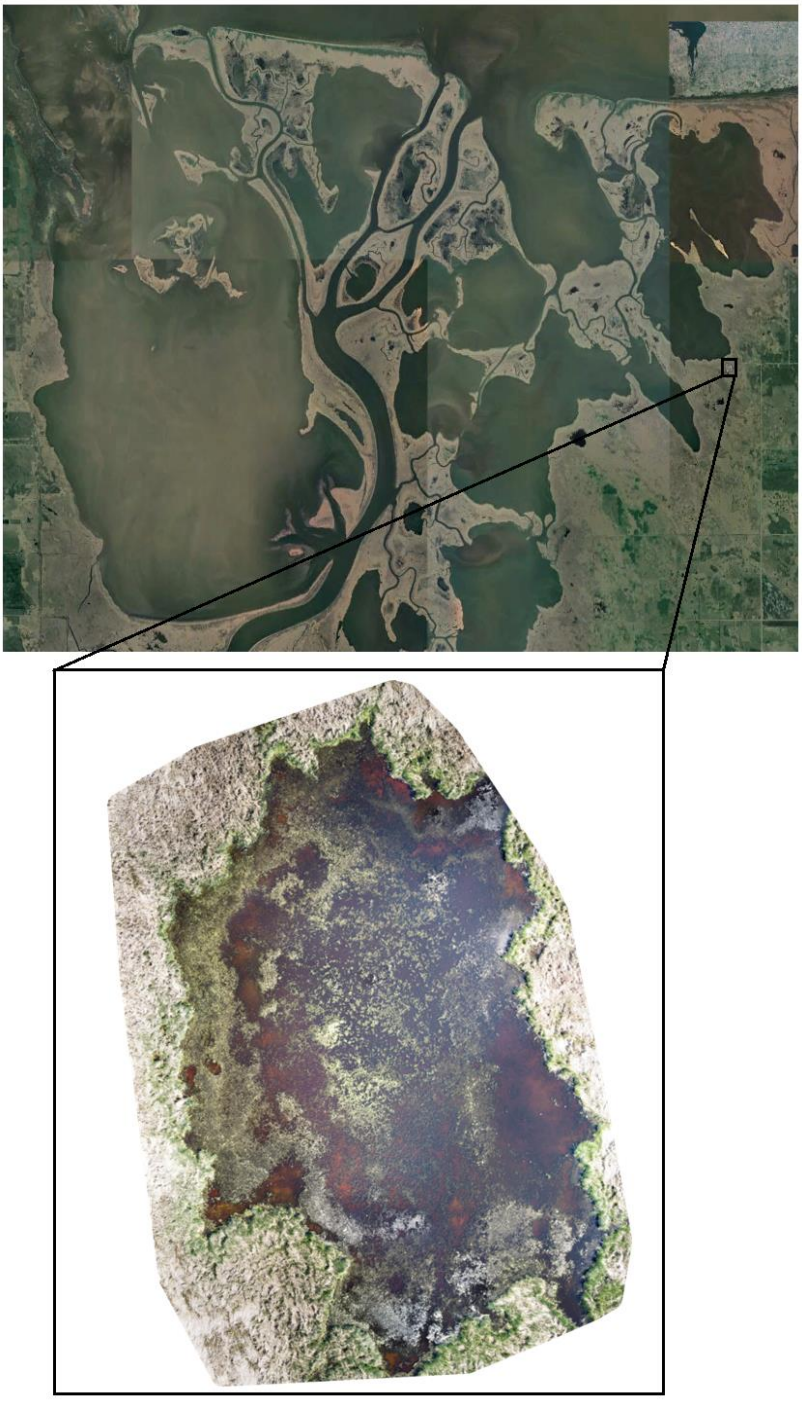

Figure 27. The first practice orthomosaic that I made, using a version of the methods described here. Photos taken June 26, 2020, using a DJI Mavic Mini (first generation). Images stitched together using WebODM. Area depicted is Stoney's Pond.

Figure 28. The first corridor mapping practice orthomosaic that I made on a scale that is comparable to the final orthomosaics that I used in my analysis (> 10 ha). Photos taken July 12, 2020, using a DJI Mavic Mini (first generation). Images stitched together using WebODM. Area depicted is Hardman Lake.

3.7 Verifying Orthomosaic Quality – WebODM Quality Reports

After producing practice orthomosaics in the fashion that I hoped to use for my thesis, I used the WebODM quality report to verify the quality of the orthomosaics that I had been producing. The quality report that WebODM automatically produces provides a variety of metrics and diagnostic graphs, which provide the user with feedback on whether they can trust the accuracy of the mosaic that they have produced (Figures 29-31).

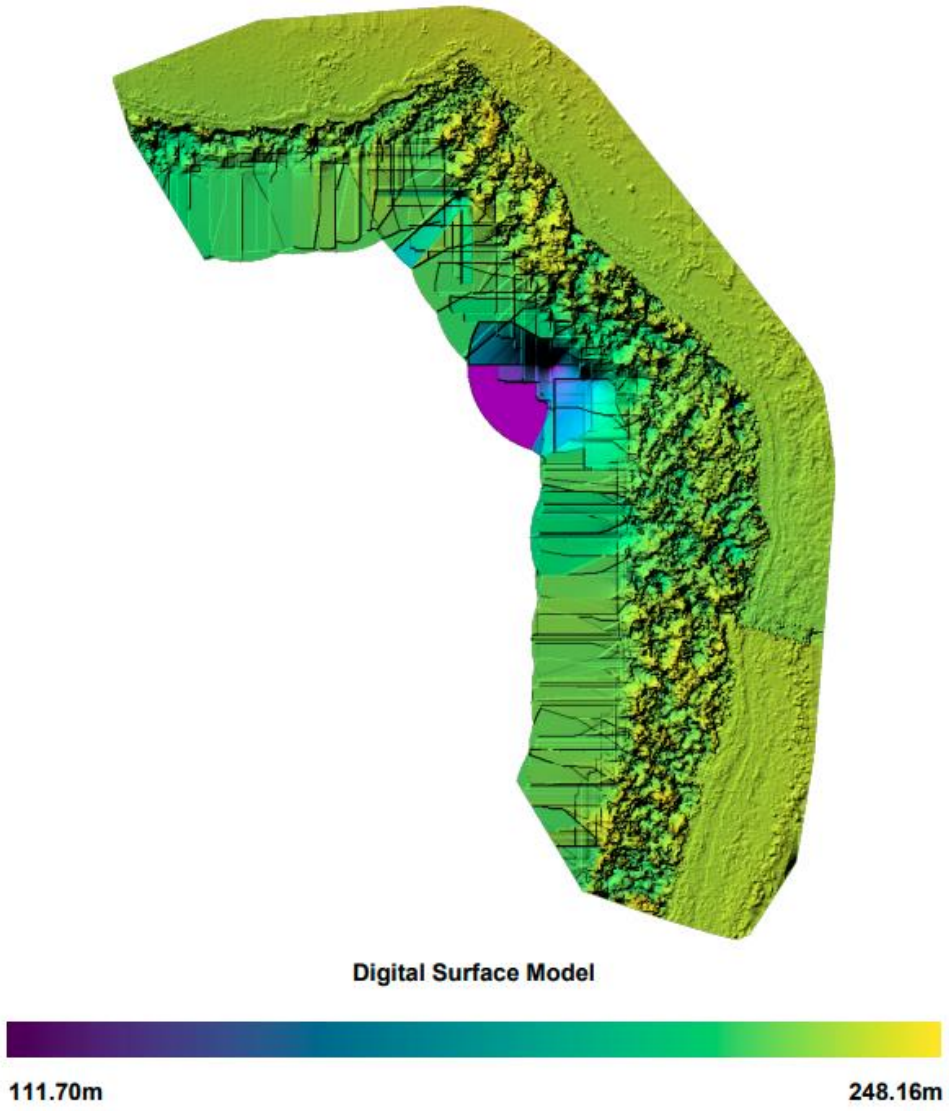

Figure 29. Three-dimensional digital elevation model (DEM), reproduced from the WebODM Quality Report. In contrast with 2D orthomosaics, the DEM output from WebODM should not be trusted in the absence of ground control points. Photos taken July 12, 2020, using a DJI Mavic Mini (first generation). Area depicted is Hardman Lake.

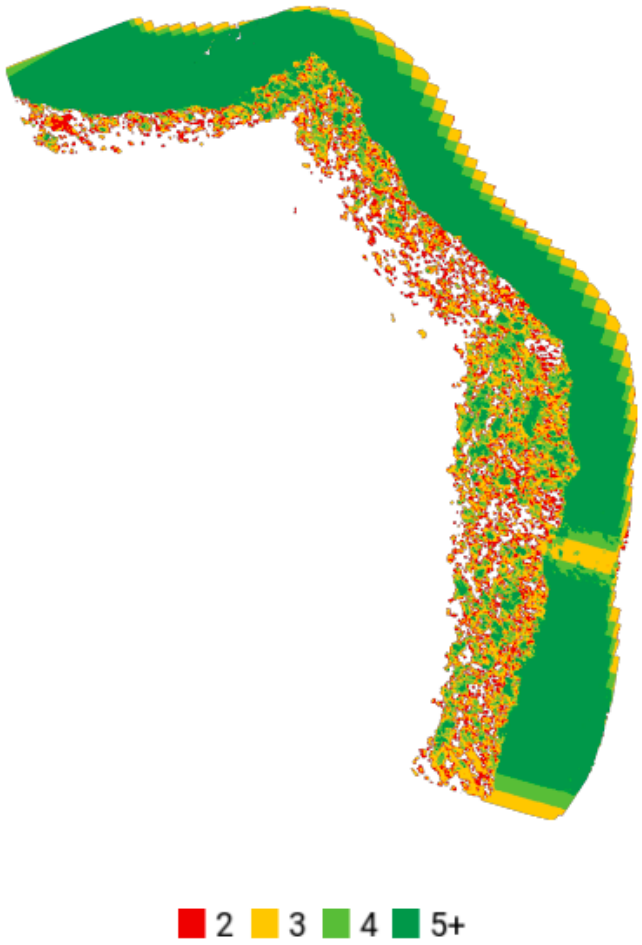

Figure 30. Diagnostic graph from the WebODM Quality Report, which provides an overview of survey quality. Survey quality is determined by the number of photos that are available for the reconstruction at any given point in space. Dark green indicates high quality (5+ images) and red indicates low quality (2 images). Gaps in survey coverage will be displayed as empty white space. Photos taken July 12, 2020, using a DJI Mavic Mini (first generation). Area depicted is Hardman Lake. A weak spot in this survey can be seen in the yellow patch. Quality is always poor towards the periphery of the survey.

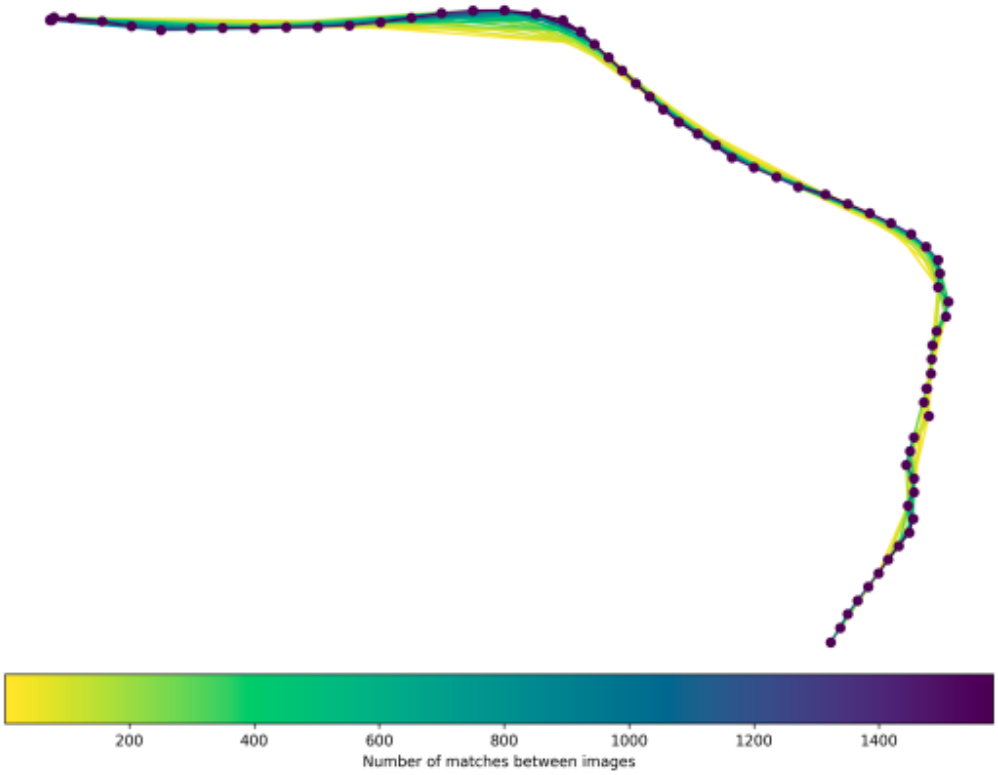

Figure 31. Diagnostic graph from the WebODM Quality Report, which provides an overview of image overlap. Purple indicates high overlap and yellow indicates low overlap. Photos taken July 12, 2020, using a DJI Mavic Mini (first generation). Area depicted is Hardman Lake.

3.8 Strengths and Limitations of Mapping Method

The manual flight methods presented here have some important strengths and limitations. The primary strength of these manual methods is that they make it possible to carry out single-pass corridor mapping without the need for a commercial subscription to mapping software. Using these methods, it is easy to follow a single circuitous flight path, such as the shoreline of a marsh or the edge of a forest. Conversely, this manual method is inefficient for standard mapping applications, which require precise overlapping transects. If the zone of interest is a relatively narrow band of 'edge' habitat, RPA operators can map this narrow strip using imagery that overlaps in a single direction only, rather than a grid of overlapping images, in which each image is overlapping with multiple images on all sides. The larger and more complex the mapping mission is, the more valuable it will be to incorporate automated flight software into a sampling design.

I tested the upper limits of the manual method developed here by attempting to map the ten experimental wetland cells that were used during the Marsh Ecology Research Program (MERP) from 1980 to 1989. The total area of this experimental complex is approximately 75 hectares, and it took 868 images to generate this orthomosaic (Figure 32). I obtained satisfactory results, but the WebODM quality report revealed some shortcomings that could have been easily overcome using automated flight software (Figures 33-36). The most obvious shortcoming was that my parallel flight paths had a very inconsistent degree of overlap, resulting in some minor gaps and some redundant photos (Figure 35).

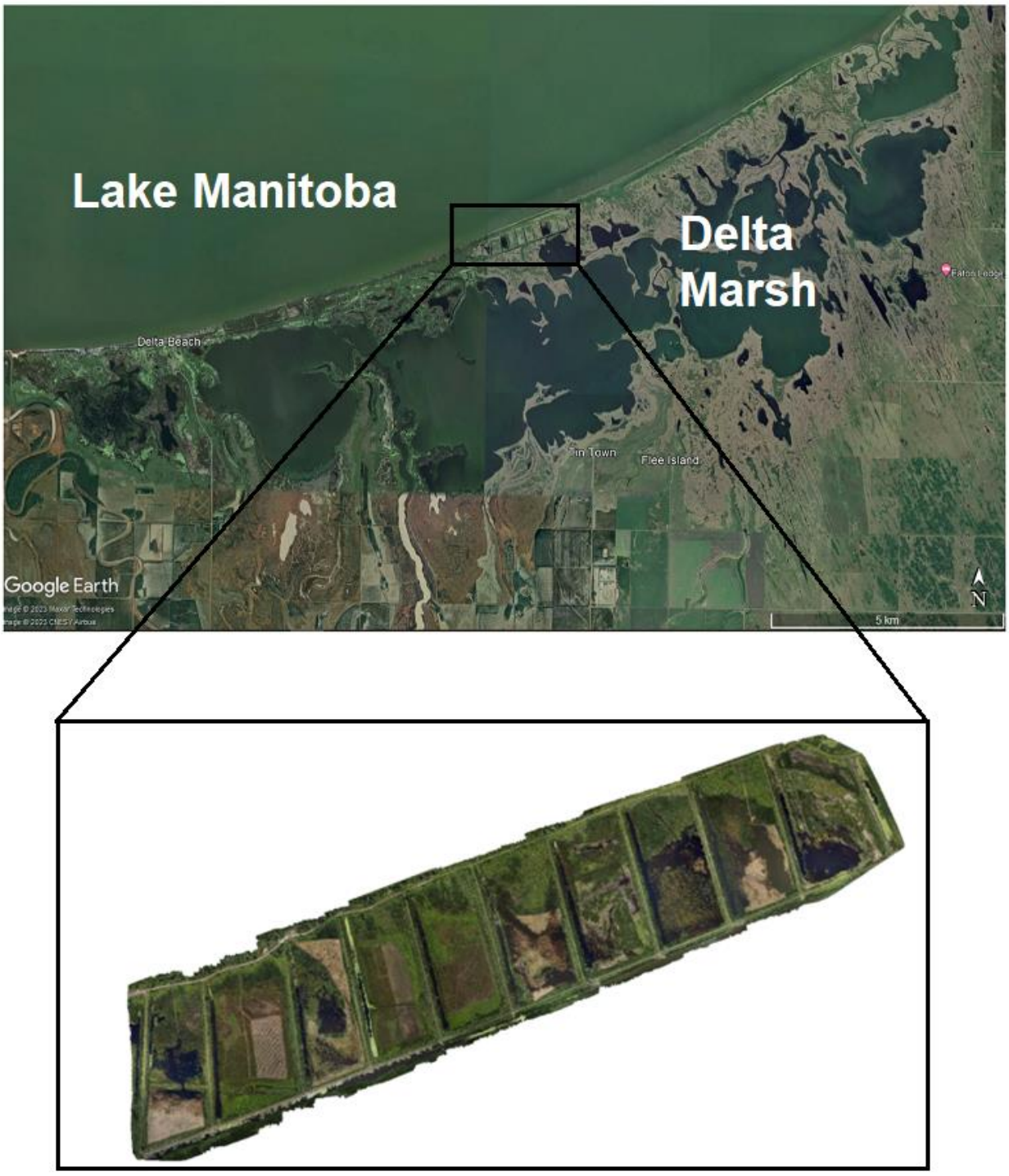

Figure 32. Two-dimensional orthomosaic, reproduced from the WebODM Quality Report. Photos taken August 20, 2023, using a DJI Mavic Mini (first generation). Area depicted is the MERP cells in Delta Marsh, Manitoba.

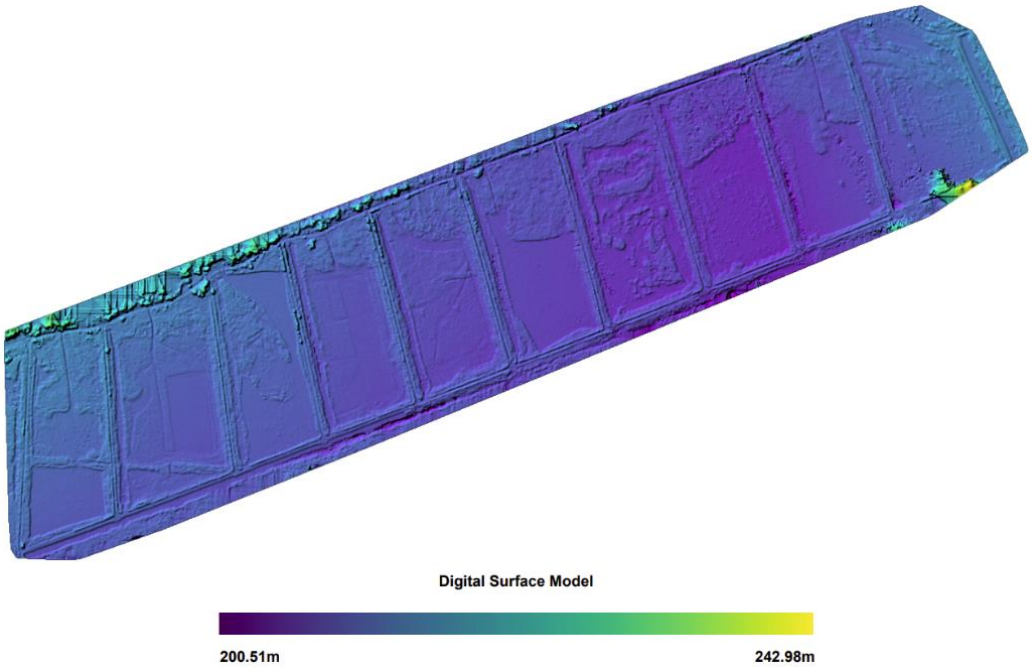

Figure 33. Three-dimensional digital elevation model (DEM), reproduced from the WebODM Quality Report. Photos taken August 20, 2023, using a DJI Mavic Mini (first generation). Area depicted is the MERP cells in Delta Marsh, Manitoba.

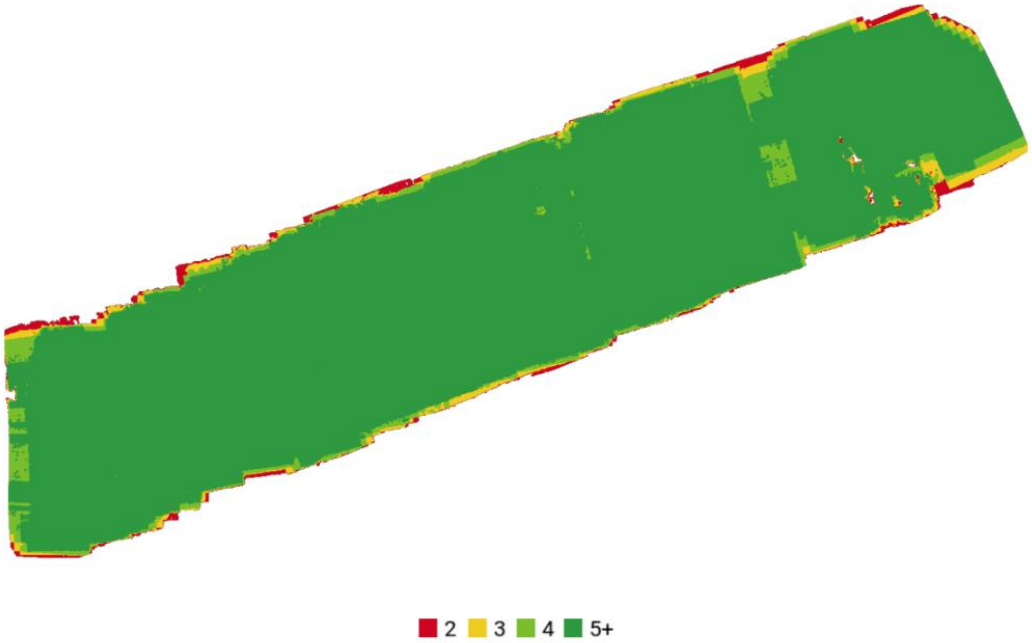

Figure 34. Diagnostic graph from the WebODM Quality Report, which provides an overview of survey quality. Dark green indicates high quality and red indicates low quality. Photos taken August 20, 2023, using a DJI Mavic Mini (first generation). Area depicted is the MERP cells in Delta Marsh, Manitoba.

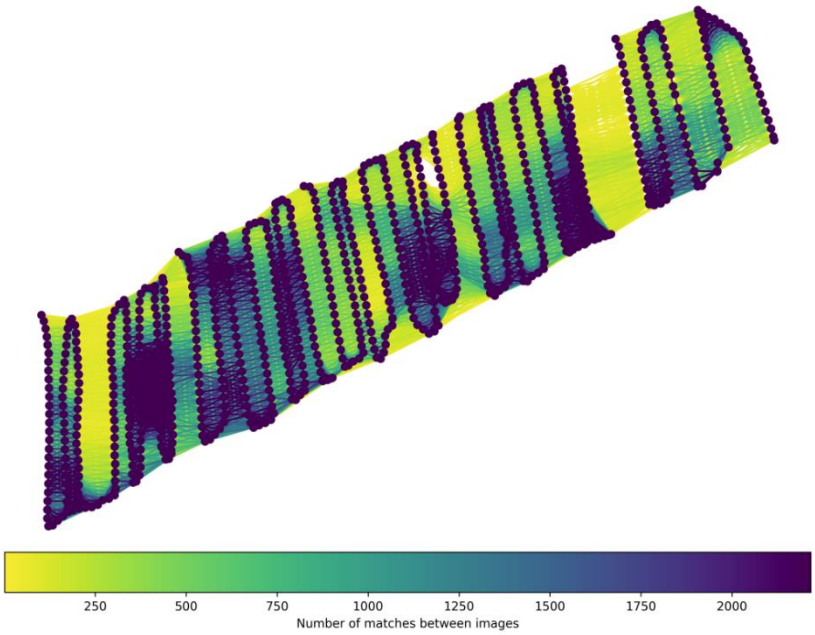

Figure 35. Diagnostic graph from the WebODM Quality Report, which provides an overview of image overlap. Purple indicates high overlap and yellow indicates low overlap. Photos taken August 20, 2023, using a DJI Mavic Mini (first generation). Area depicted is the MERP cells in Delta Marsh, Manitoba.

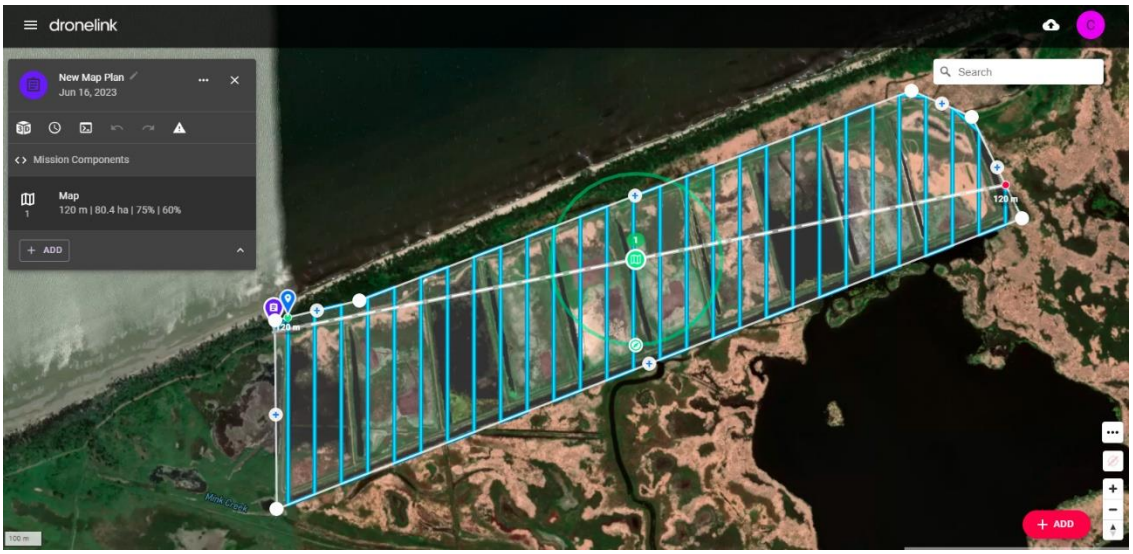

Figure 36. Mission plan for the MERP cells within Dronelink. Note that the precision of the overlapping transects is much greater than the precision of the overlapping transects from manual flight depicted in Figure 35.

3.9 Conclusions and Recommendations

The shoreline mapping methods described above are both accessible and affordable. Provided that a micro-RPA is being used, the methods presented here can even be used without the need for a basic operations certificate. The manual methods described here are particularly valuable in long/narrow sampling areas, where single-pass corridor mapping is the most efficient method of image acquisition. Whereas this study focuses on mapping a narrow strip of shoreline zone in a coastal freshwater marsh, similar methods can be applied to other edge habitats, such as the edge between a forest and a grassland. Conversely, when mapping areas that are both wide and long, mission planning software provides a degree of precision that exceeds that of manual flight, regardless of the skill level of the operator.

The recommended method for image acquisition consists of one survey at 120 m, with the camera angled directly downwards, and a second survey at 10 m, with the camera angled forwards, with the horizon in view. By combining both downwards and oblique imagery, it is possible to identify common emergent taxa to the level of genus. The 120-m imagery is stitched into an orthomosaic using open-source software (WebODM), which can be used without any specialized training. Any ambiguity in vegetation classification in the 120-m imagery can be clarified by referring to the 10-m imagery.

Whereas the methods presented here are affordable and accessible, they also have some important limitations. In many ways, physical quadrat sampling at the ground level still provides a greater degree of precision than RPA-based methods. Plants that are present in small quantities will be under-represented using the method presented here. This method also cannot distinguish between the native and invasive varieties of *Typha* spp. Moreover, there are other invasive plants (e.g., *Lythrum salicaria*), which cannot be reliably identified outside of the blooming period. The methods presented here also have some important limitations, relative to traditional aerial survey methods, due to the altitude restrictions of normal RPA flight, as stipulated in the *Canadian Aviation Regulations*.

Whereas the methods presented here include manual aerial surveys, operators should consider using automated mission planning software for many other mapping applications. The manual methods described here can easily be extended to other systems that consist of a relatively long, winding, edge habitat, such as forest edges. However, if the location of interest is a large area that

is both long and wide, it will be necessary to fly in overlapping parallel transects. In mapping missions that require parallel overlapping transects, automated flight software is much more efficient than manual flight, and it is preferable if possible. If automated flight software is not an option, the inefficiency of manual flight can be partially mitigated by taking more photos than necessary. Redundant images can be eliminated later, during the processing stage.

While there are advantages and disadvantages associated with both manual and automated flight, practical considerations of safety and legality are generally more important than considerations of efficiency or image quality. In summary, RPA operators should take the following categories into consideration before planning the details of image acquisition:

1. **Safety:** The propeller blades of many large RPAs can cause serious injuries, and the methods discussed here may be used in remote locations, where medical attention cannot be relied upon. Extreme caution must be taken if an operator wishes to fly an RPA from a confined space, such as a small boat or canoe. Micro-RPAs may be preferable in these cases, due to the minimal risk of injury from their propeller blades.
2. **Legality:** Whereas concerns of safety and legality are not mutually exclusive, it is important for operators to understand their legal obligations, which are not as simple as common-sense safety concerns. Even when an operator is using a micro-RPA, which does not require a basic operations certificate, the operator must fly within various limits, which are defined in the *Canadian Aviation Regulations* and the *Aeronautical Information Manual*.
3. **Accuracy and Precision:** The accuracy and precision of automated flight missions will generally exceed that of manual flight. However, manual flight provides sufficient precision and accuracy for many applications. For both manual and automated flight, the precision and accuracy of maps can be improved through the incorporation of ground control points. However, the onboard GPS data for many RPAs will provide sufficient accuracy and precision for many applications.
4. **Efficiency:** For standard mapping missions, which consist of overlapping parallel lines, automated mission software will provide greater efficiency than manual flight. However, in long/narrow areas, which call for corridor mapping methods, manual flight can exceed the efficiency of standard automated mapping missions. Automated

corridor mapping missions are more efficient than manual missions, but these require a commercial subscription to Dronelink or Pix4D. Manual, single-pass corridor mapping provides a simple and efficient way to map edge habitats, such as shoreline zones or the boundary between a forest and grassland.

5. **Replicability:** If the operator wishes to return to the same area in the future and replicate a previous flight mission, automated mission planning software is strongly preferred, regardless of any considerations towards efficiency.

3.10 Contribution of Authors Statement

CA wrote the paper, analyzed data, collected data, drove research direction; NK supervised graduate student, provided editorial and analytical supervision; LGG supervised graduate student, provided editorial and analytical supervision, and provided research funding.

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In the previous chapter, I described a set of accessible, affordable, methods for acquiring imagery using a micro-RPA, stitching imagery into an orthomosaic map, and identifying plants within the shoreline zone of Netley-Libau Marsh. The recommended methods for image acquisition include two manual surveys. The first survey is a single-pass corridor mapping mission, flown manually at an altitude of 120 m, with the camera aiming directly downwards. The second survey repeats approximately the same flight path at an altitude of 10 m, with the camera at an oblique angle. The 120-m imagery is stitched automatically using WebODM, and the 10-m imagery acts as a qualitative supplement to clarify any challenges in identifying vegetation from the orthomosaic.

In the following chapter, I will extend the methods described in the previous chapter to compare the composition of plant communities across an exposure gradient. I surveyed 23 km of shoreline between August 17 and August 29, 2020, and produced orthomosaic maps for all shoreline zones. Using data from 150 randomly placed shoreline quadrats, I used a generalized linear model to compare plant community composition along an exposure gradient. I estimated exposure for each quadrat using an effective fetch model that integrates data on wind directional frequency from 36 compass directions with a spatial raster of the study area within ArcMap. My results indicate that exposure acts as an environmental filter, limiting the growth of *Typha* spp. in exposed shoreline areas. *Typha* spp. was less common in exposed locations, and it was also more likely to show visible damage in locations with relatively high effective fetch. My results also indicate that damage was more common along north-facing shorelines in Netley-Libau Marsh, perhaps due to a seiche effect on Lake Winnipeg.

Chapter 4. Comparison of Taxonomic Composition across Exposure Gradient

Abstract

Netley-Libau Marsh has undergone a marked process of degradation during the past century. One of the most noteworthy changes has been a reduction in aquatic vegetation, along with a corresponding increase in open water cover and wave energy. I used aerial imagery to compare the taxonomic composition of shoreline plant assemblages across an exposure gradient in Netley-Libau Marsh. Exposure is defined as the total effect of waves on shoreline vegetation. I surveyed 23 km of shoreline between August 17 and August 29, 2020, and produced orthomosaic maps for all shoreline zones. Using data from 150 randomly placed shoreline quadrats, I used a generalized linear model to compare plant community composition along an exposure gradient. I estimated exposure for each quadrat using an effective fetch model that integrates data on wind directional frequency from 36 compass directions with a spatial raster of the study area within ArcMap. My results indicate that exposure acts as an environmental filter, limiting the growth of *Typha* spp. in exposed shoreline areas. *Typha* spp. was less common in exposed locations, and it was also more likely to show visible damage in locations with relatively high effective fetch. My results also indicate that damage was more common along north-facing shorelines in Netley-Libau Marsh, perhaps due to a seiche effect on Lake Winnipeg. The relative tolerance of emergent taxa to exposure stress should be taken into consideration when designing revegetation projects in Netley-Libau Marsh and other wetlands.

4.1 Introduction

Environmental gradients have been an area of interest for ecologists and geographers, going back at least as far as Von Humboldt and Bonpland (1807). During the 20th century, gradient analysis was formalized and extended by Whittaker (1951, 1967), and taken up by various other researchers (see Terborgh 1971, Pielou and Routledge 1976, Yeaton and Cody, 1979). Abiotic stress can influence the structure of plant communities both directly and indirectly. In some cases, abiotic stressors can be so extreme that the abiotic environment prevents the establishment or persistence of an individual, independent of any biotic interactions that might be taking place. In these cases,

the terms ‘environmental filter’ and ‘environmental sieve’ are appropriate (Kraft et al. 2015, van der Valk 2018).

Gradient analysis was originally introduced by Whittaker (1951, 1967) as a method that supported the continuum concept of vegetation, in contradistinction to the community concept (Dyakov 2010, Nicolson 2016). The continuum concept of vegetation holds that species composition changes gradually through space along a variety of environmental gradients, rather than being grouped into distinct communities with a shared history of coevolution (Whittaker 1967, Nicolson 2016). Following this view, the abundance of a particular plant will tend to increase as environmental conditions approach ecological optima for the plant in question (Whittaker 1967, Nicolson 2016).

Environmental gradients can be divided into direct gradients and indirect gradients (Whittaker 1967, Dyakov 2010). Direct gradient analysis examines gradients that have a direct physiological effect on the growth of plants, such as temperature, soil pH, or exposure stress (Keddy 1982, Austin and Smith 1989). Indirect gradient analysis examines complex gradients, such as elevation, in which plants are influenced indirectly by elevation, through the direct influence of variables like temperature and precipitation, which vary along the elevation gradient (Austin and Smith 1989, Dyakov 2010). Whereas gradient analysis was originally described in a way that subverted the notion of distinct communities within a delineated spatial area, it should be noted that environmental gradients are abstract dimensions that need not change continuously through space (Austin 1985, Dyakov 2010). For example, Keddy (1982) used gradient analysis to describe vegetation composition along an exposure gradient, using 25 randomly selected points along the shoreline of a lake.

Whereas various studies on environmental gradients and filters have often emphasized the importance of gradients in elevation (Whittaker 1956, Grosshans 2001, Jarvis et al. 2015), latitude (Dobzhansky 1950, MacArthur 1972, Fergnani and Ruggerio 2017), and water depth (Waters 1989, Waters and Shay 1990, 1992), a smaller number of researchers have extended the concept to gradients of exposure and wave energy (Keddy 1982, Schrimpf and Lynch 2021). Wave energy can be approximated using simple measurements of wind fetch distance, or it can be modeled using GIS models, which integrate data about the spatial configuration of water bodies, along with data on bathymetry, and the directional frequency of wind (Rohweder et al. 2012, Mason et al. 2018).

In tidal and seiche-influenced systems, the impacts of wave energy are influenced by the depth of water at the time of wind events (Ge et al. 2019).

Exposure stress may be particularly relevant in wetland systems like Netley-Libau Marsh, where there has been a historical process of degradation, characterized by a loss of aquatic vegetation and a corresponding expansion of turbid open water (Figure 1). Historical increases in open water have been accompanied by increased wave energy in the shoreline zone. Whereas wetland restoration projects are often designed in a way that is mindful of the relative tolerance that plants have to water depth gradients, less consideration tends to be given to the (largely unknown) differences in exposure tolerance. For example, a preliminary revegetation strategy for Netley-Libau Marsh was designed with the consideration that *Typha* spp. has high tolerance to the stress of water depth (SMM Inc. 2019). However, this revegetation plan did not take exposure tolerance into consideration, because there is minimal evidence in modern literature to support the conclusion that the common emergent taxa differ in their tolerance to exposure stress.

Whereas the modern body of scientific literature provides little guidance on differences in exposure tolerance among common emergent taxa, Warming (1897) [translated and summarized in Clements (1916)] concluded that *Typha* spp. was more vulnerable to mechanical disruption from wave action than the other common emergent taxa (*Scirpus* spp. and *Phragmites communis*). If there are in fact meaningful differences between the relative tolerance of emergent taxa to exposure stress, then these differences should taken into consideration when designing revegetation projects in Netley-Libau Marsh and other wetlands. The research presented here will attempt to answer this question by using aerial imagery to compare the composition of shoreline plant communities along an exposure gradient.

4.2 Objective and Hypotheses

My objective was to use aerial imagery to compare the composition of shoreline plant communities along an exposure gradient. Specifically, I wanted to determine whether there are any differences in tolerance to exposure stress among the three common emergent taxa in Netley-Libau Marsh: *Typha* spp., *Schoenoplectus* spp., and *Phragmites australis*.

Based on my preliminary observations of *Typha* spp. shoots that appeared to have been destroyed by wave action, I hypothesized that *Typha* spp. is particularly vulnerable to mechanical damage from wave energy. Following this hypothesis, I predicted that there would be a negative relationship between *Typha* spp. and exposure, as estimated by effective fetch. I subdivided my prediction into two distinct components:

1. Presence/absence: I predicted that *Typha* spp. would be present in fewer quadrats with high exposure, and
2. Proportional cover: Among quadrats where *Typha* spp. is present, I predicted that it would tend to be present in smaller proportions at higher exposure values.

In a similar way, I hypothesized that shoreline damage is driven by exposure, and I subdivided my prediction into two distinct components:

1. Presence/absence: I predicted that visible damage to shoots would be present in more quadrats with high exposure, and
2. Proportional cover: Among quadrats where visible damage is present, I predicted that damage would tend to be present in greater proportions at higher exposure values.

4.3 Methods

4.3.1 Implementation of Orthomosaic Mapping Technique

Between August 17 and August 29, 2020, I used a remotely piloted aircraft weighing less than 250 grams (micro-RPA) to map over 23 km of shoreline length within three regions of Netley-Libau Marsh, using the method described in Chapter 3 (Figure 37). The micro-RPA that I used for this study is the DJI Mavic Mini (first generation). Using WebODM, I stitched overlapping aerial images together to produce orthomosaic maps of my study areas (Figures 38-43).

I restricted my sampling to late-August to ensure that most plants had completed most of their growth for the season. To obtain optimal imagery, I restricted my mapping flights to days with calm weather. To minimize shading, I flew all sampling flights between 10:00 am and 2:00 pm. To ensure comparable shoreline conditions, I restricted sampling to periods where the water level was within the middle of the natural range of variability. I performed sampling flights at lake levels between 217.72 m and 217.86 m above sea level. These lake levels are relatively moderate and consistent, considering that the water level fluctuated by over 1.2 m during the growing season (217.14 m Sept 21, and 218.36 m July 19).

Figure 37. Map of Netley-Libau Marsh, reproduced from Google Earth Pro (version 7.3), depicting the locations of the three sampling regions for this study.

Figure 38. Small island within the Netley South region. Photos taken August 17, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

Figure 39. Large island within the Netley South region. Photos taken August 17, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

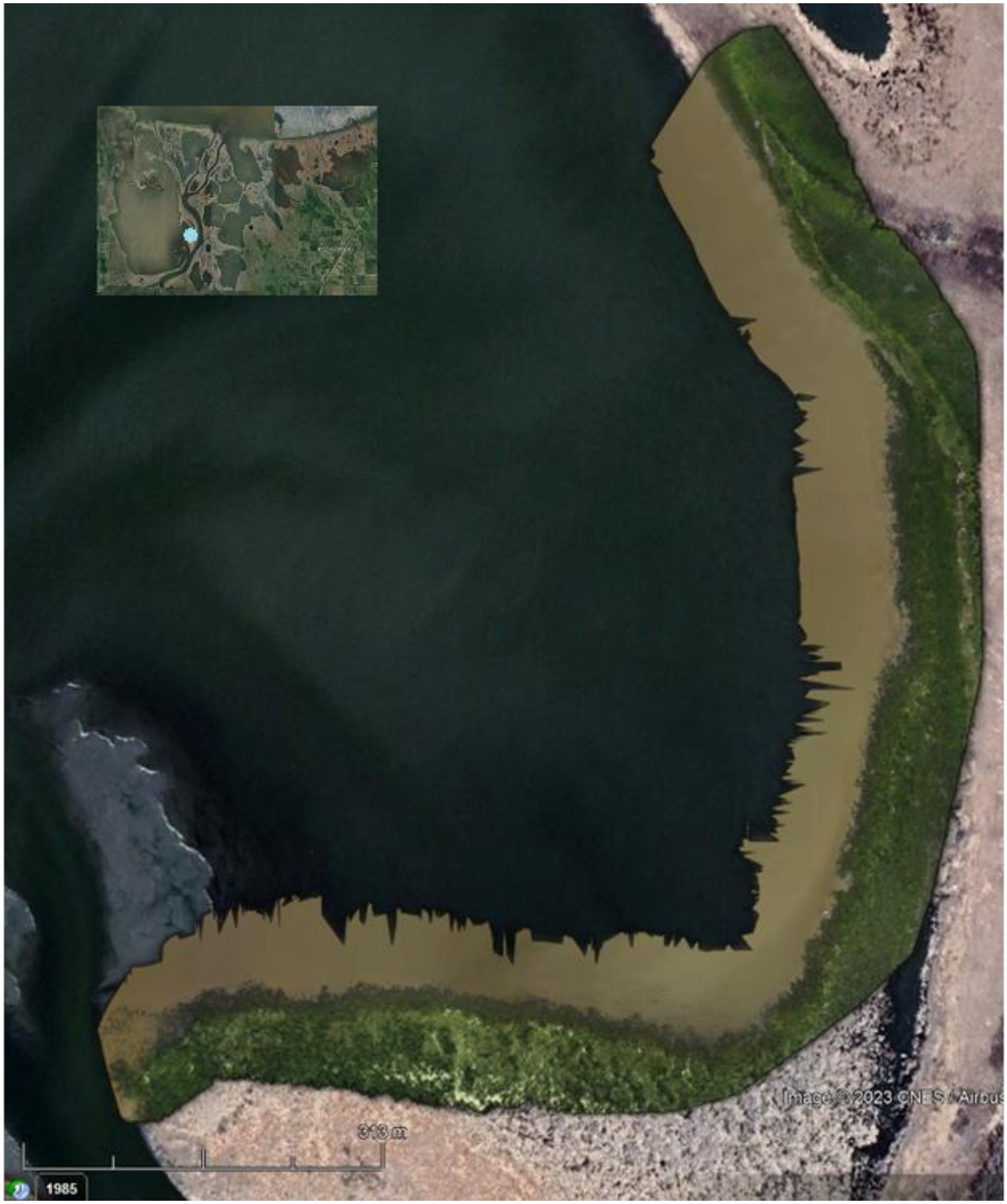

Figure 40. Shoreline and peninsula within the Netley South region. Photos taken August 17, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

Figure 41. Islands within the Netley North region. Photos taken August 27, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

Figure 42. Shoreline of Anderson Lake within the Libau region. Photos taken August 29, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

Figure 43. Shorelines of Anderson Lake, Long Lake, and Parisian Lake, within the Libau region. Photos taken August 29, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

4.3.2 Effective Fetch Model

Using the methods described in Rohweder et al. (2012), I generated a spatial model of exposure stress throughout Netley-Libau Marsh in ArcMap (version 10.8.1). I quantified exposure using effective fetch, a metric that integrates the spatial data of fetch distances from the raster, while incorporating the directional frequency of wind from 36 compass directions. The output of the effective fetch model is a spatial raster with a single numerical exposure estimate for each pixel (Figures 44-46).

In its traditional form, effective fetch can be summarized as follows (SPM 1966, Smith 1991):

$$F_{eff} = \frac{\sum X_i \cos^2 \theta_i}{\sum \cos \theta_i}$$

where

F_{eff} = effective fetch

X_i = length of the straight-line fetch

θ_i = angle from mean wind direction

The methods used here extend the traditional concept of effective fetch, by adjusting the weight of each component direction, relative to the percent frequency at which wind blew from that direction during the period of interest (Rohweder et al. 2012). I obtained data on the directional frequency of wind from the Environment Canada weather station in Gimli from May 1, 2020 to August 31, 2020. The effective fetch model allows for the integration of a simple text file that indicates each of the 36 compass directions, along with a percentage value that indicates how frequently wind blew from each direction.

I obtained a 2020 spatial raster of Netley-Libau Marsh from Agriculture Manitoba, and I made corrections to this raster through reference to satellite imagery available on SentinelHub and my own aerial imagery. Corrections were relatively minor, and they were limited to filling small gaps represented in the shoreline, which would have resulted in an over-estimate of fetch distances for some locations. The corrected raster provides a binary representation of land and water cover for Netley-Libau Marsh, where water has a value of 0 and land has a value of 1.

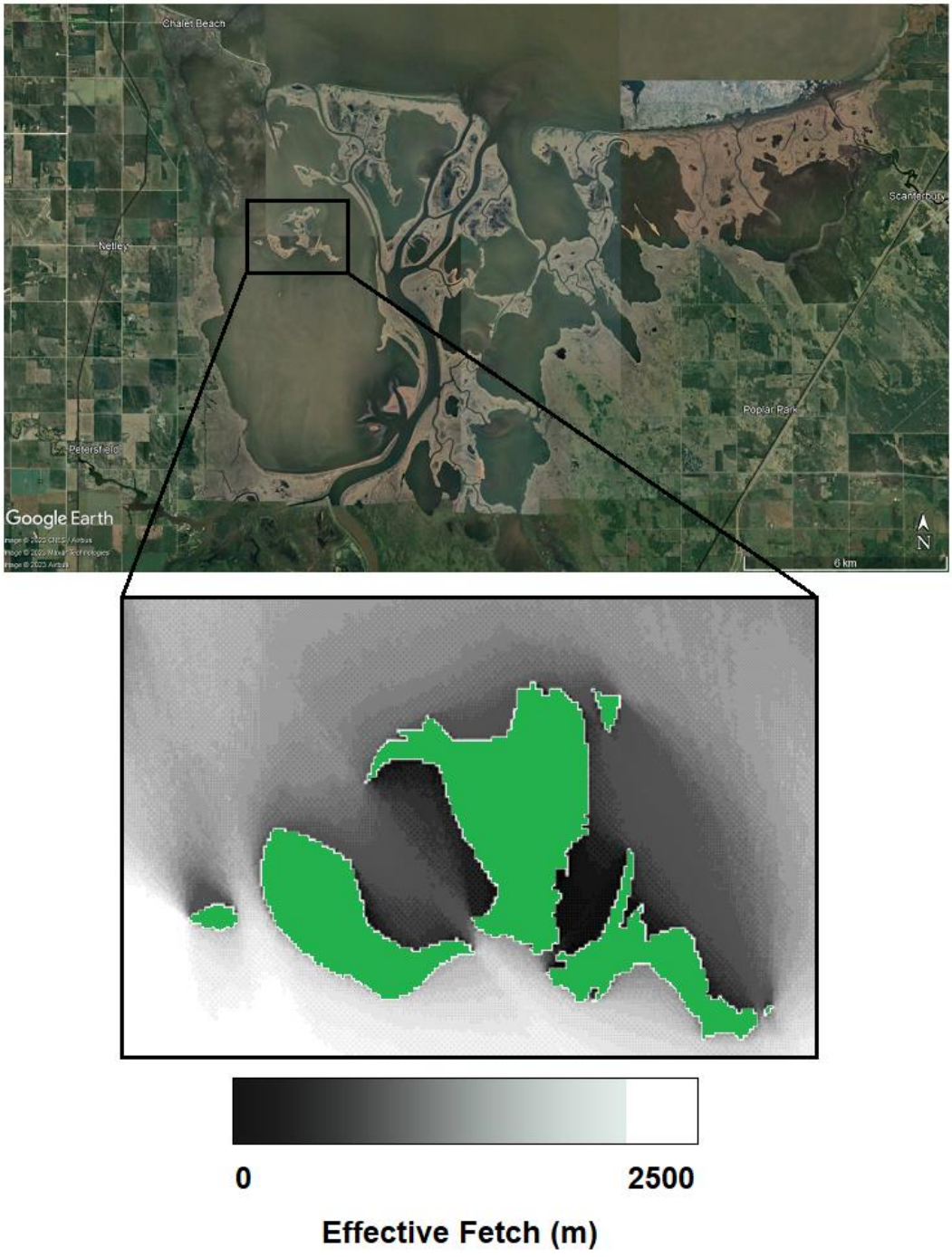

Figure 44. Effective fetch model for Netley North region, based on wind data from May-August 2020. Wind data obtained from Environment Canada weather station in Gimli. Dark gray indicates low exposure and white indicates high exposure. Green represents vegetated land.

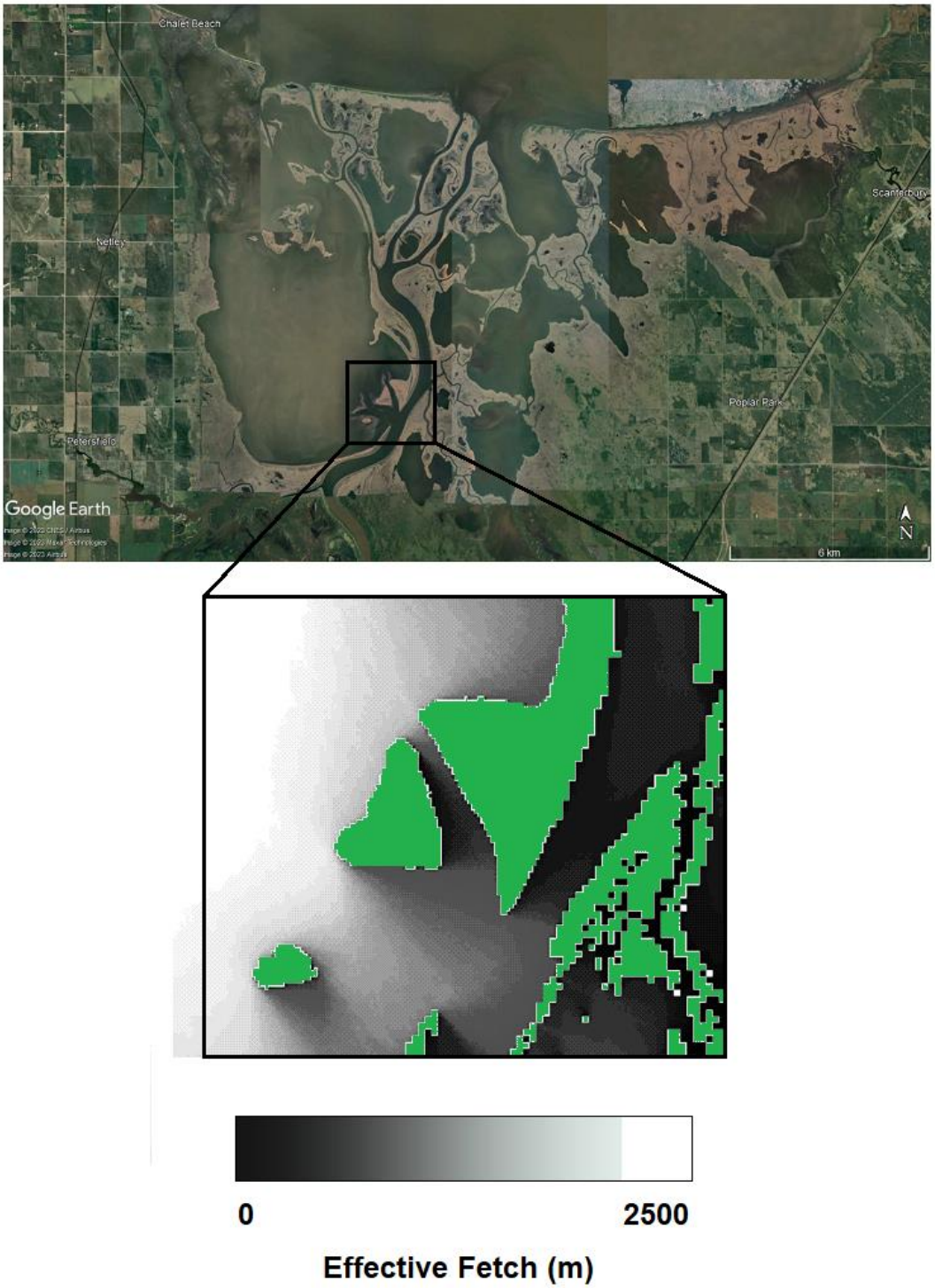

Figure 45. Effective fetch model for Netley South region, based on wind data from May-August 2020. Wind data obtained from Environment Canada weather station in Gimli. Dark gray indicates low exposure and white indicates high exposure. Green represents vegetated land.

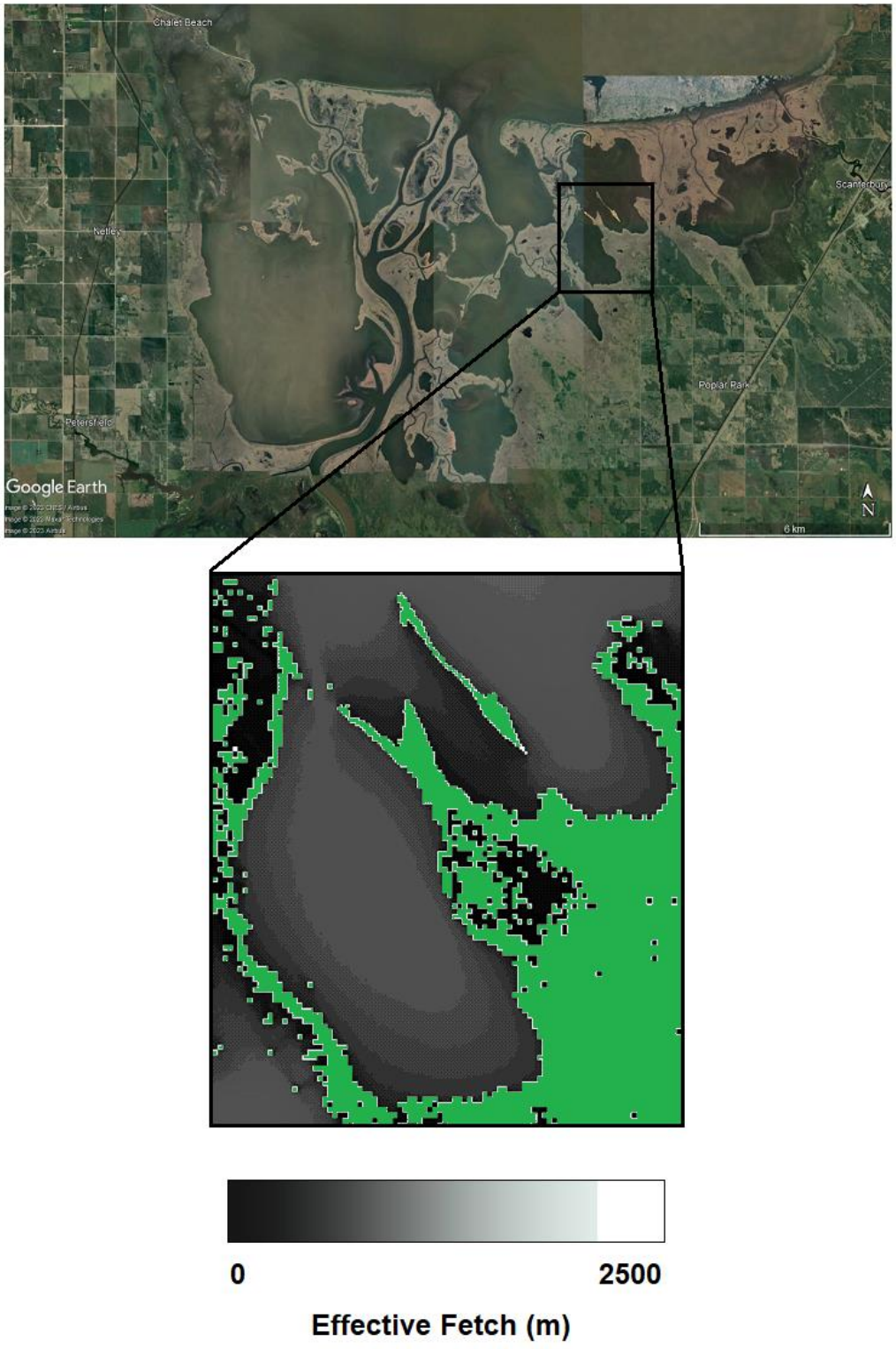

Figure 46. Effective fetch model for Libau region, based on wind data from May-August 2020. Wind data obtained from Environment Canada weather station in Gimli. Dark gray indicates low exposure and white indicates high exposure. Green represents vegetated land.

4.3.3 Shoreline Quadrats and Associated Measurements

To compare plant communities across an exposure gradient, I selected 150 random shoreline locations within my study area (Figures 47-49). Using a random number generator, I selected points along the shoreline areas that I mapped within Netley-Libau Marsh. I delineated one square 100-m^2 digital quadrat for each point in Google Earth Pro (version 7.3) (Figure 50). Within each quadrat, I manually delineated the aerial cover and shoreline length for each vegetation and damage class using the 'path' and 'polygon' tools. In cases where the 120-m imagery was unclear, I used the 10-m imagery as a qualitative supplement to clear up any ambiguity in classification. Using the effective fetch model described above, I generated a numerical estimate of exposure stress for each quadrat.

Mapped areas were clustered in three regions within the marsh, which are each at least 3.5 km from each other (Figures 47-49). The Libau region is far-removed from the Red River, and subject primarily to the influence of Lake Winnipeg. The Netley South region is a miniature delta formation where roughly 37% of the flow from the Red River enters Netley Lake (Kowal 2019). The Netley North region is located farther downstream within Netley Lake. The Netley North region is therefore subject to a considerable amount of river flow, but not in a mini-delta formation. All sites are influenced by their connection to Lake Winnipeg and fluctuate with the level of the lake (Watchorn et al. 2021).

For *Typha* spp, *Schoenoplectus* spp., and *Phragmites australis*, I estimated shoreline complexity using a simple metric of the ratio between shoreline length and surface area. I added up the total area and total shoreline length occupied by each taxon, across all 150 quadrats, and then I divided total length by total area for each taxon. The resulting ratio provides a simplified metric of how convoluted the shoreline zone is for each of the taxa.

Figure 47. Orthomosaic for the Netley North region, with markers indicating the location of all quadrats within the region. Photos taken August 27, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

Figure 48. Orthomosaic for the Netley South region, with markers indicating the location of all quadrats within the region. Photos taken August 17, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

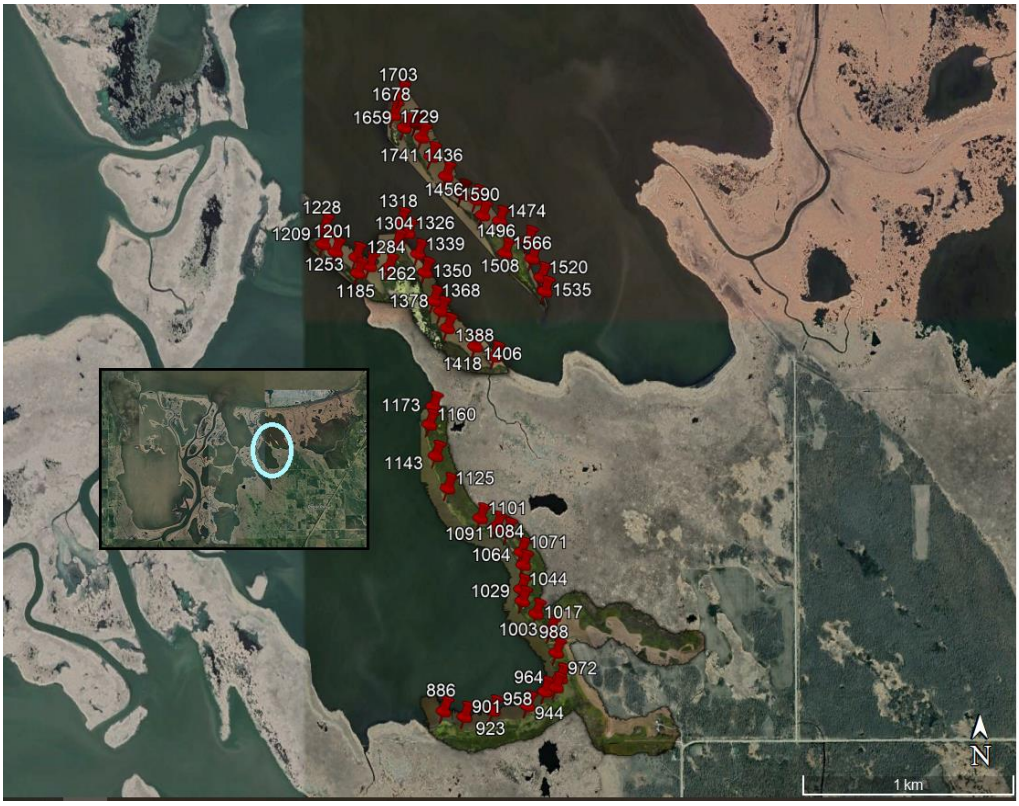

Figure 49. Orthomosaic for the Libau region, with markers indicating the location of all quadrats within the region. Photos taken August 29, 2020, using a DJI Mavic Mini (first generation). Orthomosaic stitched together using WebODM, and loaded into Google Earth Pro (version 7.3).

Figure 50. Example of one of the 150 digital quadrats used in this study. This quadrat consists solely of *Phragmites australis* and open water. Aerial and distance measurements are extracted from the quadrats using the path and polygon tools within Google Earth Pro (version 7.3).

Although I was interested in investigating the effect of direction, this effect is confounded with the influence of the seiche effect on Lake Winnipeg. Because of the north-south orientation of Lake Winnipeg and the position of Netley-Libau Marsh along the south shore, sustained north winds can result in daily fluctuations in water level of more than 1 m. The net result is that northerly winds tend to coincide with periods of high water in Netley-Libau Marsh. During my study period, northerly winds tended to converge with average water levels that were approximately 30 cm higher than those associated with southerly winds (Figure 51).

Because of the confounding influence of the seiche effect, I simplified my analysis of direction to reflect a measure of 'northness' – i.e., the extent to which a particular site is exposed to wind and waves from the north. I assigned each quadrat a simplified numerical metric for northness, based on a scale of intensity from 1 to 5, with north-facing quadrats receiving a value of 5 and south-facing quadrats receiving a value of 1 (Table 2). Because it is not possible to separate direction as a variable from the seiche effect of Lake Winnipeg, the use of a simplified scale along a north-south axis is warranted. There is no distinction between east and west, with both east and west receiving a value of 3 on this scale. This scale reflects the reality that there is no meaningful difference in average water level when wind blows from the east or the west.

Table 2. Simplified metric for northness, based on the north-south orientation of the seiche effect on Lake Winnipeg. Areas that are subjected to winds from the north will tend to experience periods of exposure stress that converge with periods of high water, due to the seiche effect.

Cardinal Direction	Degrees	Northness
N	337.5 - 22.5	5
NW	292.5 - 337.5	4
W	247.5 - 292.5	3
SW	202.5 - 247.5	2
S	157.5 - 202.5	1
SE	112.5 - 157.5	2
E	67.5 - 112.5	3
NE	22.5 - 67.5	4

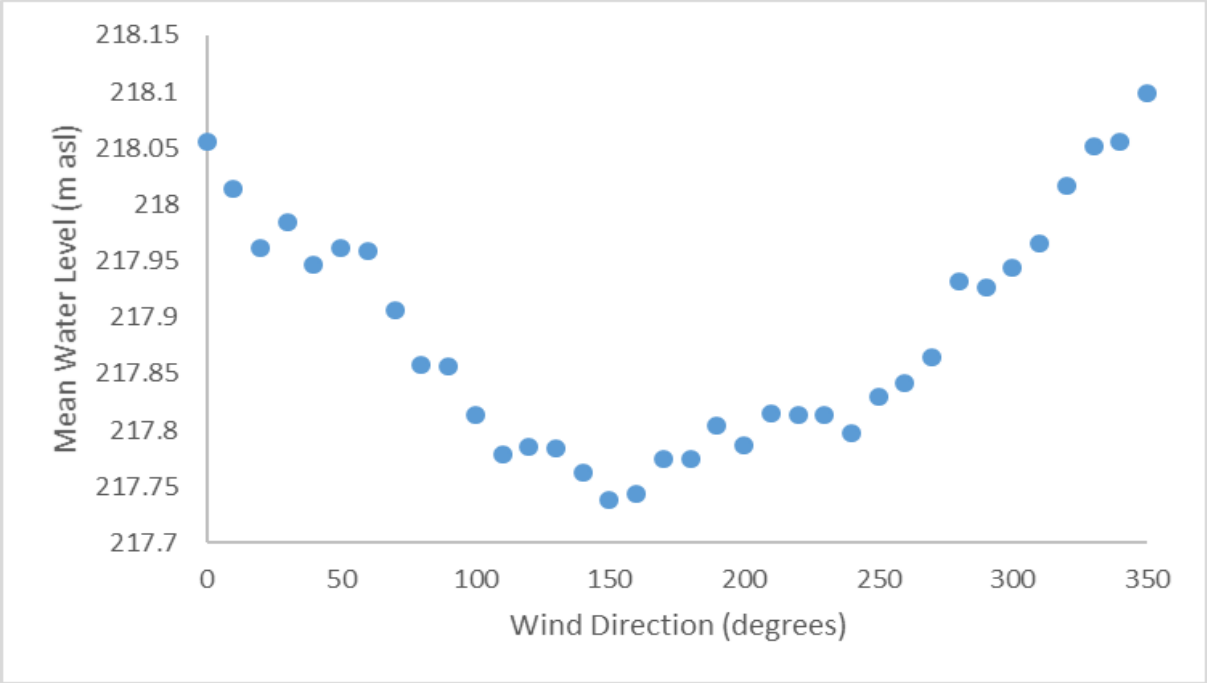

Figure 51. Mean water level from May-August 2020, for wind directions in increments of ten degrees. Data obtained from Environment Canada weather station in Gimli. On average, water was 36 cm higher when wind was blowing from 350 degrees (roughly north), compared with when it was blowing from 150 degrees (roughly south). Overall, north-facing shorelines are subjected to exposure at times that converge with high water, induced by the seiche effect on Lake Winnipeg.

4.3.4 Statistical Analysis

To compare the composition of plant communities among quadrats, I used the glm() function in RStudio version 2022.07.2+576 to run a generalized linear model. Because my dataset contains many zero-values, I divided the statistical analysis into two steps. For both damage and taxonomic composition, I analyzed 150 digital quadrats in terms of both:

1. Presence vs absence, and
2. Proportional cover when present

In the first step, my statistical analysis for each taxon asked the question: "What factors cause a particular taxon to be present in a quadrat?" In the second step, my statistical analysis asked the question: "When a particular taxon is present in a quadrat, what factors cause that taxon to be more abundant?" In addition to accommodating the large number of zero-values, this two-step process allowed me to inquire into the possibility that relationships of presence/absence and proportional cover might not be driven by the same mechanisms.

My analysis of damage followed a similar two-step process. In the first step, my statistical analysis asked the question: "What factors cause damage to be present in a quadrat?" In the second step, my statistical analysis asked the question: "When damage is present in a quadrat, what factors cause damage to be more extensive?"

For both steps in my analysis, the variables of interest were effective fetch, northness, and region. In the first step, I used a binomial distribution, and I converted all nonzero numbers to 1. In the second step, I used a normal distribution, and I excluded all zero-values from the analysis. I used the DHARMa function to verify that I was meeting the required assumptions for my models. Where assumptions were violated, I performed transformations to resolve these issues. Because all relationships were curvilinear for effective fetch, I used a natural logarithm transformation to make the relationship rectilinear and appropriate for input into a generalized linear model. To ensure that my data met the assumptions of my statistical analysis, I also performed an arcsine transformation for *Typha* spp. in the second stage of my analysis, which examines proportional cover.

4.4 Results

4.4.1 Shoreline Length

There was a marked difference in shoreline complexity among taxa, as estimated in terms of a simple ratio between shoreline length and surface area. *Schoenoplectus* spp. had relatively convoluted shorelines, with 1.31 m of shoreline length per 1 m^2 of surface area, compared with 0.36 m/m^2 for *Phragmites australis*, and 0.21 m/m^2 for *Typha* spp. (Table 3). In general, *Typha* spp. and *Phragmites australis* had relatively simple shorelines, whereas *Schoenoplectus* spp. had relatively complex shorelines.

Table 3. Summary data for area and shoreline length among shoreline plant communities in Netley-Libau Marsh, among 150 digital shoreline quadrats (10 m x 10 m).

	Mean	Standard Deviation
***Typha* Surface Area (m^2)**	50.11	36.37
***Schoenoplectus* Surface Area (m^2)**	11.27	21.11
***Phragmites* Surface Area (m^2)**	9.22	23.24
***Typha* Shoreline Length (m)**	16.45	10.52
***Schoenoplectus* Shoreline Length (m)**	41.93	24.02
***Phragmites* Shoreline Length (m)**	13.55	10.7
***Typha* Shoreline Complexity (m/m^2)**	0.21	0.21
***Schoenoplectus* Shoreline Complexity (m/m^2)**	1.31	1.09
***Phragmites* Shoreline Complexity (m/m^2)**	0.36	0.58

4.4.2 Damage

Preliminary Observations

Before any formal statistical analysis was conducted, it was already clear that there was a specific form of damage that was common among *Typha* spp. and completely absent among *Schoenoplectus* spp. and *Phragmites australis*. *Typha* spp. was frequently observed to be flattened in large sections of the shoreline zone. In some areas, these flattened areas were directly adjacent to undamaged stands of *Schoenoplectus* spp. or *Phragmites australis*. In other areas, where *Typha*

spp. was the only taxon present, these flattened areas were surprisingly long. The longest continuous area of this type was over 700 m long, along the narrow island that separates Parisian Lake from Long Lake (Figure 52). Out of 150 quadrats, damage was present in 21 quadrats. 100% of damage observed was in areas dominated by *Typha* spp.

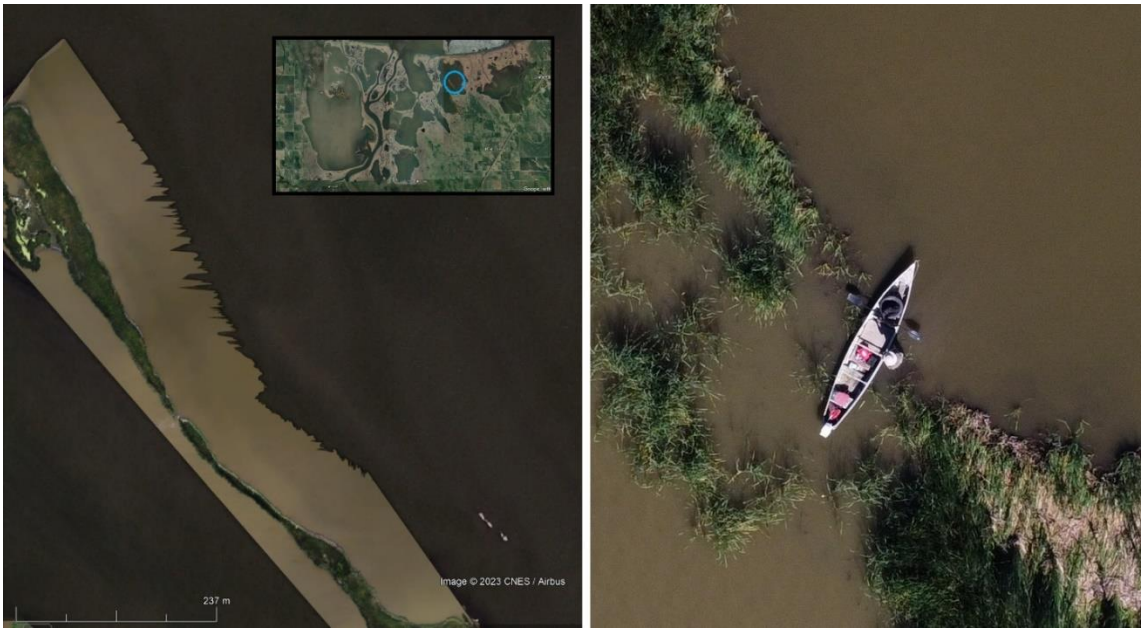

Figure 52. Orthomosaic (left) and aerial image (right), depicting the longest continuous damaged area observed within this study, which was more than 700 m in length. Photos taken August 29, 2020, using a DJI Mavic Mini (first generation).

There was a positive relationship between the presence of damage and higher effective fetch values (p = 0.000173) (Figure 53). Among quadrats with damage, the natural logarithm of effective fetch tended to be greater by 2.92 ln of m, with a standard error of 0.78 ln of m. There was also a relationship for northness, with damage being more likely to be present along north-facing shorelines (p = 0.013958). Among quadrats with damage, northness tended to be greater by 0.96, with a standard error of 0.39. There was no relationship for region, between Netley North, Netley South, and Libau (p = 0.145975 [Netley North], p = 0.989836 [Netley South]).

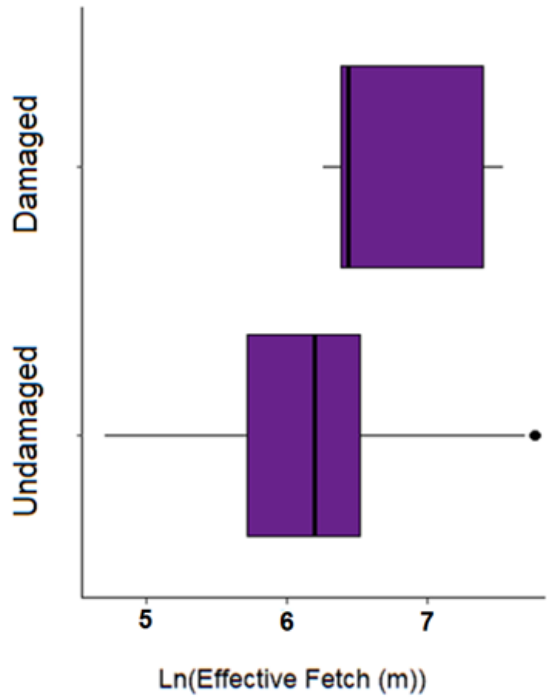

Figure 53. Boxplot depicting the presence and absence of damage in quadrats at different values of effective fetch, with a natural log transformation. Damage was more likely to be present at higher values of effective fetch.

When damage was present, its proportional cover tended to be greater along north-facing shorelines (p = 0.00797) (Figure 54). For every increase of 1 point on the intensity scale for northness, the proportional cover of damage increased by 0.1, with a standard error of 0.03. There was no relationship between effective fetch and the proportional cover of damage within the quadrat (p = 0.10395) and extent of damage did not vary by region (p = 0.29013).

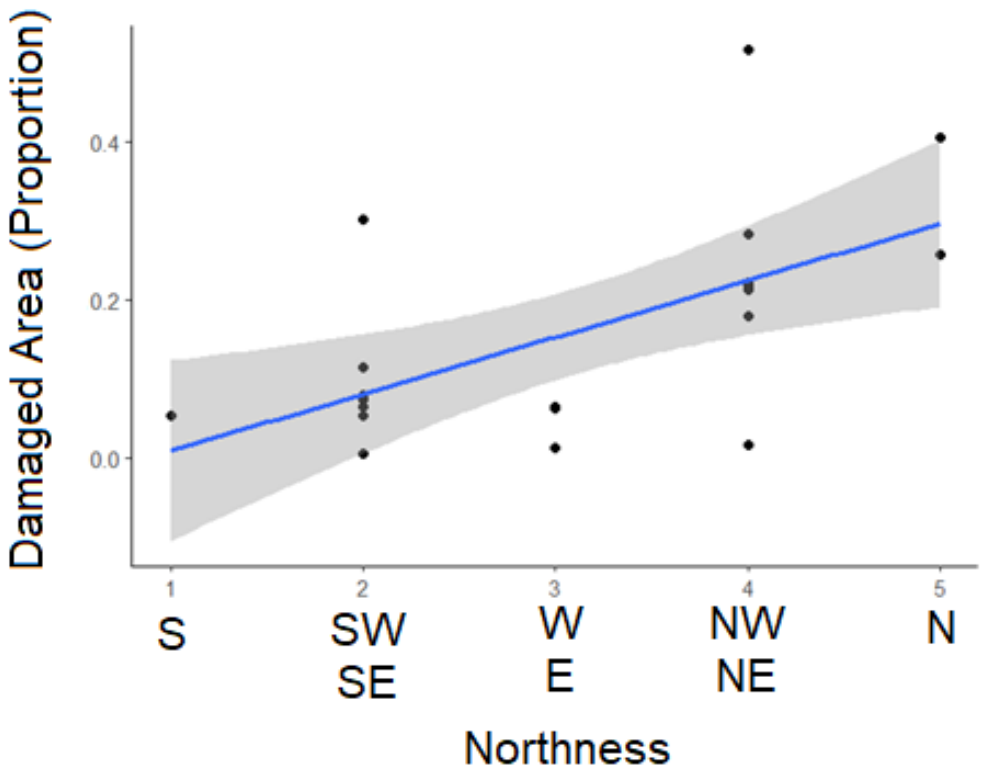

Figure 54. Scatterplot depicting the proportional cover of damage in quadrats that experience exposure from different directions. When damage was present, north-facing shorelines tended to have the highest proportional cover of damage.

4.4.3 *Typha* spp.

Typha spp. was less likely to be present in quadrats with higher effective fetch values (p = 0.000304) (Figure 55). Among quadrats with *Typha* spp. present, the natural logarithm of effective fetch tended to be lower by 2.13 ln of m, with a standard error of 0.59 ln of m. Northness was also significant at the presence/absence stage, with *Typha* spp. being more likely to be present along south-facing shorelines (p = 0.00641). Among quadrats with *Typha* spp. present, northness tended to be lower by 0.67, with a standard error of 0.25. Region had no effect on the presence of *Typha* spp. (p = 0.104295 [Netley North], p = 0.815897 [Netley South]).

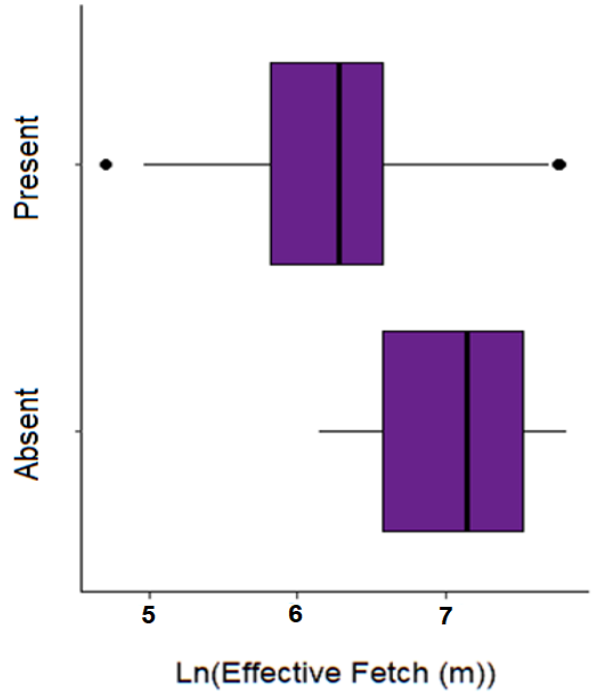

Figure 55. Boxplot depicting the presence and absence of *Typha* spp. in quadrats at different values of effective fetch, with a natural log transformation. *Typha* spp. was more likely to be present at lower values of effective fetch.

When *Typha* spp. was present, there was no relationship between effective fetch and the proportional cover of *Typha* spp. within the quadrats (p = 0.135205). Similarly, there was no relationship between northness and the proportional cover of *Typha* spp. (p = 0.094703). However, there was lower proportional cover of *Typha* spp. in Netley South than in the other two regions (p = 0.000288). Compared with the Libau region, the proportional cover of the arcsine of *Typha* spp. in the Netley South Region tended to be lower by 0.32 with a standard error of 0.09.

4.4.4 *Schoenoplectus* spp.

Schoenoplectus spp. was more likely to be present along north-facing shorelines (p = 0.00552), and in the Netley South region (p = 0.00000456). *Schoenoplectus* spp. was present in 62.2% of the quadrats within Netley South, compared with 12.2% and 12.0% in Netley North and Libau, respectively. Among quadrats with *Schoenoplectus* spp. present, northness tended to be greater by 0.58, with a standard error of 0.21. There was no relationship for the presence of *Schoenoplectus* spp. with effective fetch (p = 0.90795). *Schoenoplectus* spp. was also more abundant in north-facing (p = 0.00508), exposed (p = 0.00113), shorelines (Figures 56 and 57). For every increase of 1 ln m of the natural logarithm of effective fetch, the proportion of *Schoenoplectus* spp. increased by 0.12 with a standard error of 0.03. For every increase of 1 on the intensity scale for northness, the proportion of *Schoenoplectus* spp. increased by 0.08 with a standard error of 0.03. There was no relationship between proportional cover and region (p = 0.45759 [Netley North], p = 0.51686 [Netley South]).

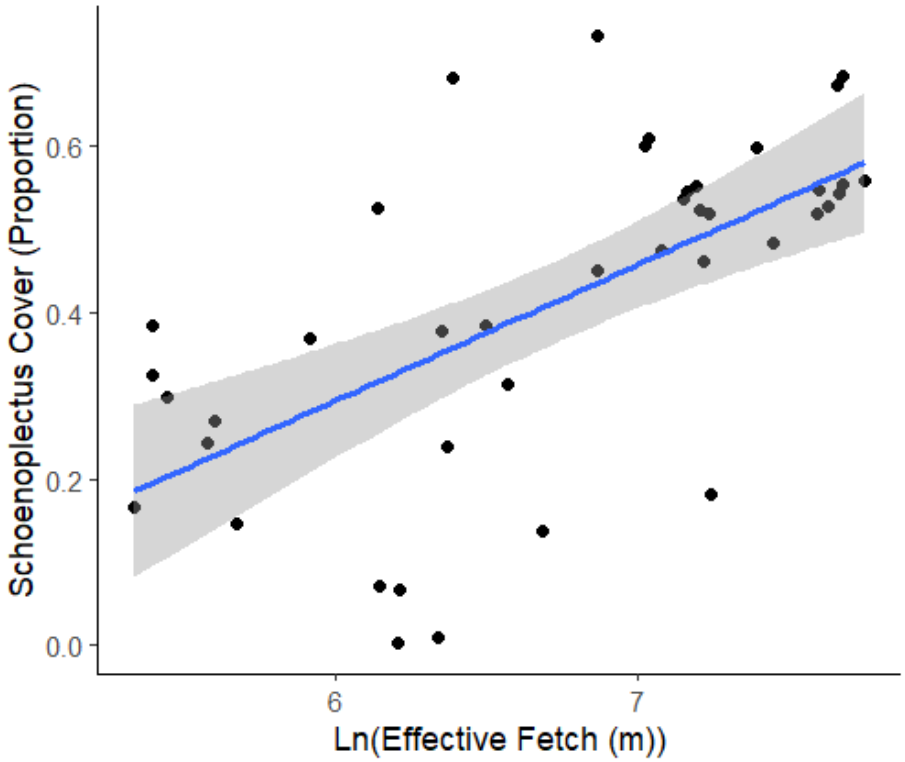

Figure 56. Scatterplot depicting the proportional cover of *Schoenoplectus* spp. in quadrats at different values of effective fetch, with a natural log transformation. The proportional cover of *Schoenoplectus* spp. tended to be greater at higher values of effective fetch.

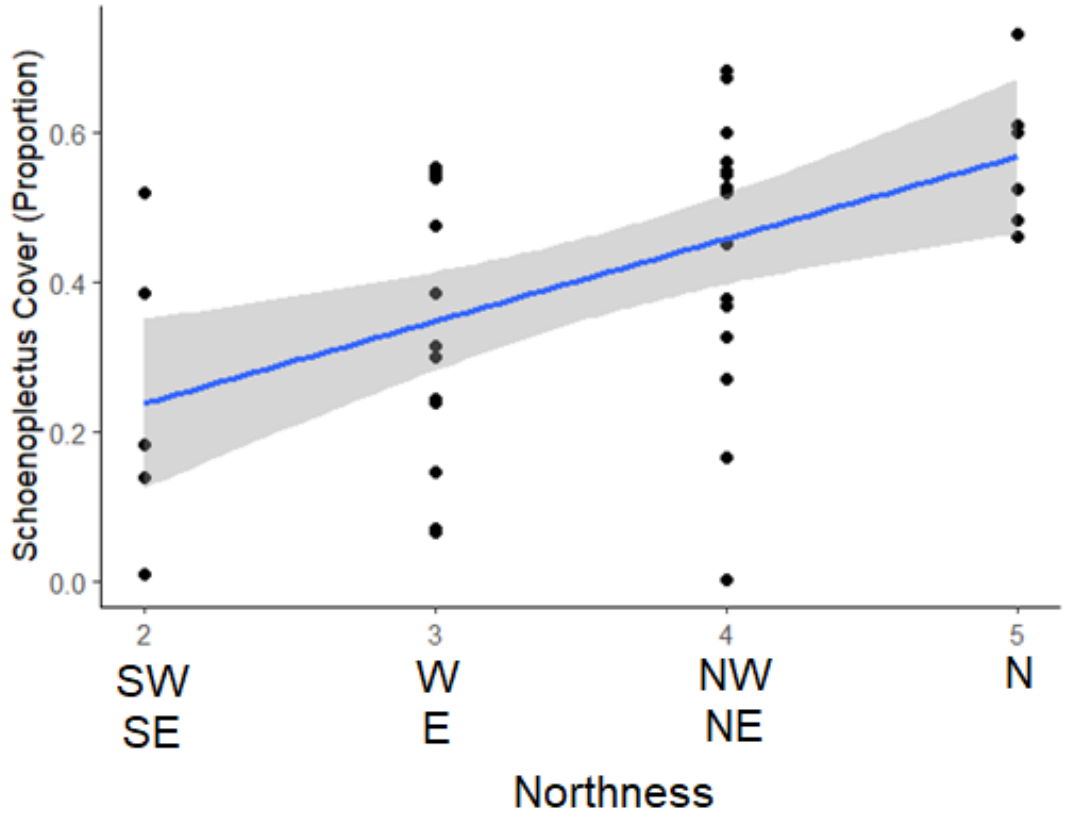

Figure 57. Scatterplot depicting the proportional cover of *Schoenoplectus* spp. in quadrats that experience exposure from different directions. The proportional cover of *Schoenoplectus* spp. tended to be greater along north-facing shorelines.

4.4.5 *Phragmites australis*

Phragmites australis was more likely to be present at higher effective fetch values (p = 0.000556) (Figure 58) and was also more likely to be present in the Libau region (p = 0.000651 [Netley North]; p = 0.000258 [Netley South]). Among quadrats with *Phragmites australis* present, the natural logarithm of effective fetch tended to be greater by 2.17 ln of m, with a standard error of 0.63 ln of m. *Phragmites australis* was present in 38.6% of the quadrats within Libau, compared with 18.2% and 10.2% in Netley South and Netley North, respectively. There was no relationship between northness and the presence of *Phragmites australis* (p = 0.255278).

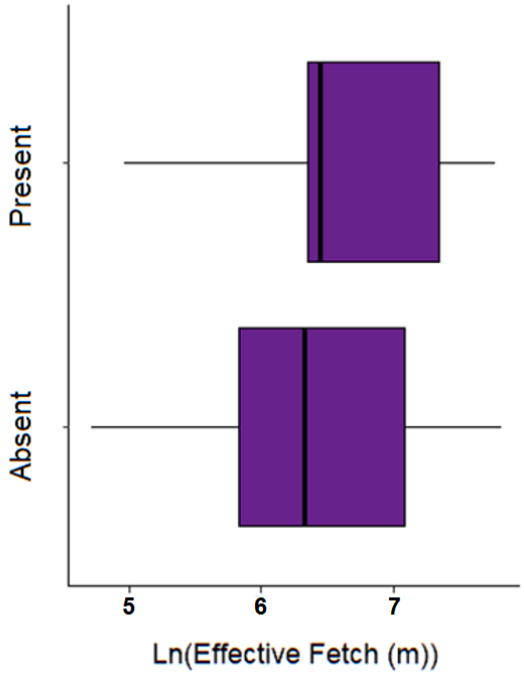

Figure 58. Boxplot depicting the presence and absence of *Phragmites australis* in quadrats at different values of effective fetch, with a natural log transformation. *Phragmites australis* was more likely to be present at higher values of effective fetch.

When *Phragmites australis* was present, there was no relationship between effective fetch and the proportional cover of *P. australis* (p = 0.22706). Similarly, there was no relationship between northness and the proportional cover of *P. australis* (p = 0.30305). Conversely, the proportional cover of *P. australis* was lower in Netley South than in the other two regions (p = 0.00234). Compared with the Libau region, the proportion of *Phragmites australis* in the Netley South region tended to be lower by 0.64 with a standard error of 0.19.

4.5 Discussion

4.5.1 Shoreline Complexity

The markedly greater shoreline complexity of *Schoenoplectus* spp. warrants some discussion, especially when coupled with the fact that there has been a decline in the relative cover of *Schoenoplectus* spp. in recent decades (Table 3, Grosshans et al. 2004). In general terms, the emergent zone in Netley-Libau Marsh appears to have undergone a historical change from a complex, *Schoenoplectus*-dominated zone, towards a simplified zone, with a greater proportion of *Typha* spp. and *Phragmites australis* (Figure 1, Grosshans et al. 2004).

Of the three main emergent taxa in Netley-Libau Marsh, only *Schoenoplectus* spp. provides the kind of interspersion and horizontal patchiness that resembles past conditions in Netley-Libau Marsh (Figure 1, Verbiwski 1986). The convoluted nature of *Schoenoplectus*-dominated shorelines also provides shelter and wave attenuation on a fine scale, which is not provided by the simplified shoreline configurations of *Typha* spp. and *Phragmites australis*. Moreover, this finer-scale shelter is not included within the scope of the effective fetch model described above, as this model is based on a coarse spatial raster, with pixels that represent 30 m x 30 m – much larger than the scale of the convolutions recorded within my 10 m x 10 m quadrats.

Whereas recent attempts at restoration in Netley-Libau Marsh have emphasized the importance of replacing open water with native emergent vegetation, less emphasis has been placed on the question of whether certain emergent taxa ought to be prioritized within revegetation projects. The convoluted nature of *Schoenoplectus*-dominated shorelines is one reason to favour *Schoenoplectus* spp. within targeted revegetation projects. In general terms, an increase in *Schoenoplectus* spp. would lead to an increase in hemi-marsh habitat. However, it should be noted that the current conditions within Netley-Libau Marsh are more polarized towards the extreme ratios of vegetation:water than those present in the primary research on hemi-marsh habitat. Whereas researchers have found a ratio of 50:50 to be preferable to 70:30 and 30:70 (Kaminski and Prince 1981, Masto et al. 2022), most areas within Netley-Libau Marsh are closer to a ratio of 100:0 or 0:100 (Figure 1). Rather than focusing on a precise target ratio of 50:50, it may be preferable to focus on increasing the amount of interspersion to ensure the widest variety of habitat types for wetland species (Murkin et al. 1997).

4.5.2 Damage

The results of the damage analysis support three broad conclusions that warrant discussion:

a) *Typha* spp. is more vulnerable to physical damage than *Schoenoplectus* spp. or *Phragmites australis*.
b) Exposure stress caused damage to be more common but not more extreme.
c) Northness caused damage to be both more common and more extreme along north-facing shorelines.

I will discuss each of these conclusions separately.

a) *Typha* spp. is more vulnerable to physical damage than *Schoenoplectus* spp. or *Phragmites australis*

Whereas the existing literature on Netley-Libau Marsh has focused on the role of water depth in driving the mortality of emergent vegetation (Grosshans et al. 2004, Watchorn et al. 2021), my own results provide ample evidence of shoreline mortality where the cause of death was clearly wave action. In 100% of these areas, all identifiable damaged plants were *Typha* spp. Before any statistical analysis was conducted, it was clear that there was a specific form of damage that was common among *Typha* spp. and completely absent among *Schoenoplectus* spp. and *Phragmites australis*. *Typha* spp. was frequently observed to be flattened in large sections of the shoreline zone.

My results run contrary to a common interpretation presented by researchers who have studied the structure and function of fibre cables running through the lacunae of leaves in *Typha* spp. (Rowlatt and Morshead 1992, Witzum and Wayne 2014, 2015, 2016). Whereas these authors have emphasized the apparent strength of *Typha* leaves, through a tensegrity structure (Witzum and Wayne 2014), my results indicate that *Typha* spp. is extremely vulnerable to mechanical stress. My results do not contradict the idea that fibre cables provide structural support to the leaves of *Typha* spp. However, my results do indicate that the tensegrity structure of *Typha* spp. leaves is less effective at resisting mechanical stress than the structure of *Schoenoplectus* spp. or *Phragmites australis*.

Conversely, my results support Eugenius Warming's (1897) assertion that *Typha* spp. is more vulnerable to being "torn out" of the shoreline zone than *Phragmites communis* or *Scirpus* spp. It is important to note that the genus *Schoenoplectus* separated from the genus *Scirpus* long after Warming's death. It is with this caveat that I conclude that my results validate Warming's claim.

b) Exposure stress caused damage to be more common but not more extreme

The greater the degree of exposure, the more likely it was for a quadrat to include some amount of visible damage. These results are consistent with my prediction. These results are also consistent with the existing literature, which has emphasized the impacts of wave exposure in the shoreline zone (e.g. Keddy 1982, Mason et al. 2018). In contrast, the second stage of my statistical analysis indicated that there was no relationship between effective fetch and the proportional cover of damage within a quadrat. These results are inconsistent with my prediction. It is challenging to interpret the meaning of these results without further research.

One possible explanation for the observed pattern is that there may be an unmeasured difference in elevation gradients across the sampled quadrats. It is conceivable that an elevation gradient could prevent wave action from reaching the ramets located upslope (Figure 59D). Unfortunately, I did not record elevation data for my quadrats. Whereas it is certainly possible that an elevation gradient could lead to the observed pattern, conclusions cannot be drawn without collecting elevation data.

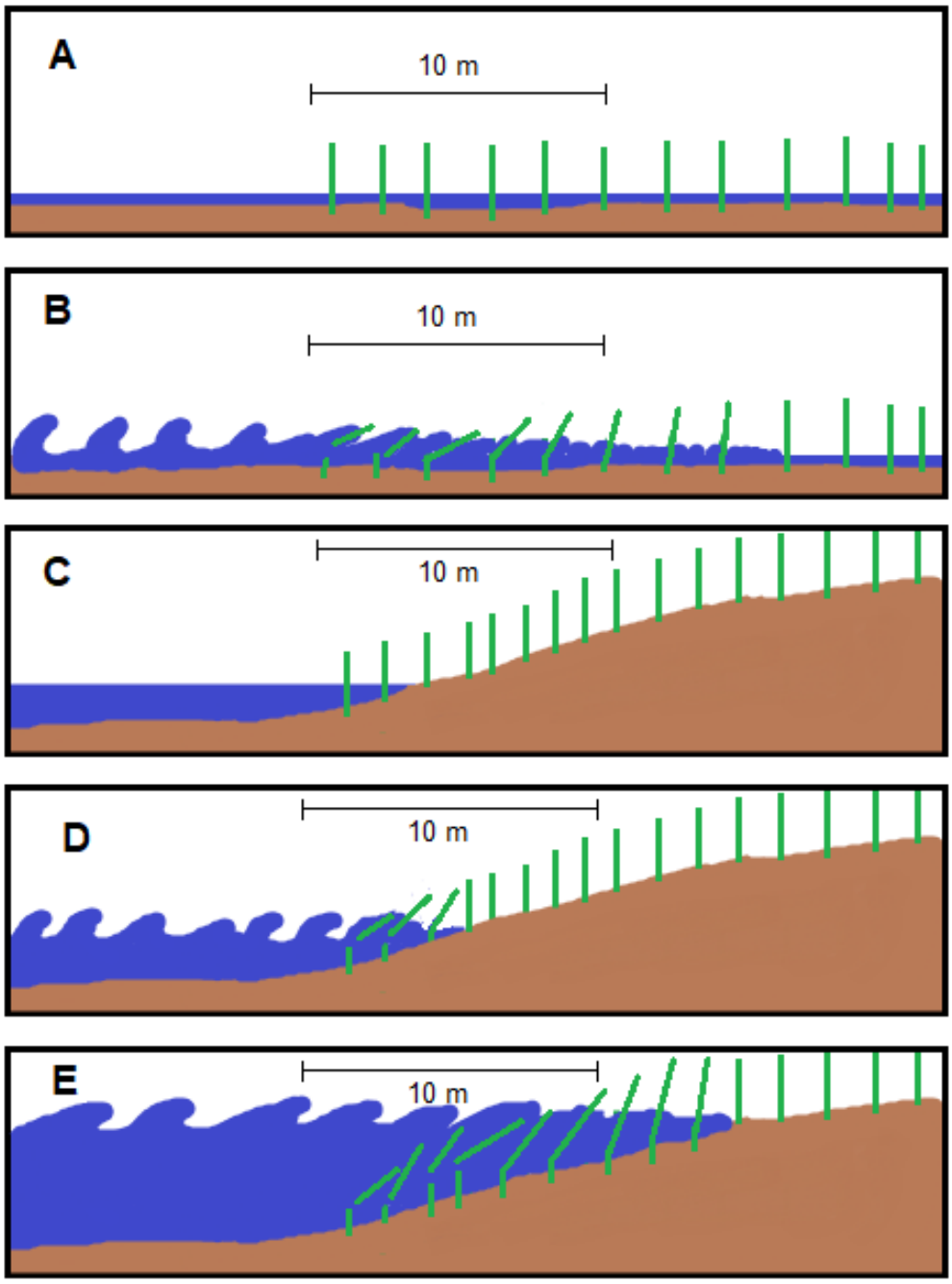

Figure 59. A hypothetical depiction of the impact of waves in the presence and absence of an elevation gradient. In scenario B, wave action is dissipated by vegetation, and this may require considerable horizontal distance. In scenario D, wave action is dissipated by vegetation across horizontal distance, but the elevation gradient places a hard limit on the horizontal distance that waves will penetrate the vegetation. In scenario E, seiche-induced high water overcomes the vertical influence of the elevation gradient and allows the waves to cause damage across a greater horizontal distance.

c) Northness caused damage to be both more common and more extreme along north-facing shorelines

In contrast with effective fetch, northness was significant at both stages of the analysis, with damage being more common and more extensive along north-facing shorelines. It is important to keep in mind that the measure of northness used in this study combines the influence of direction with the influence of water depth, as driven by the seiche effect on Lake Winnipeg. Greater damage among north-facing shorelines could therefore indicate that north winds are more severe, or that wind and wave exposure are more damaging when they coincide with periods of high water.

North-facing shorelines experienced greater damage, even though northerly winds were not more frequent. This discrepancy can be explained by the fact that waves can be expected to penetrate farther into the shoreline – both horizontally and vertically – when the water is higher (Figure 59E). As noted above, water levels tended to be approximately 30 cm higher when wind was blowing from the north than from the south. In contrast, the patterns of wind directional frequency were much less clear and provide very little evidence to support the claim that damage was driven by the direct effects of northerly winds. The compounding effect of north winds converging with periods of high water would explain why the aerial cover of damage tended to be higher along north-facing shorelines, and why the most heavily damaged quadrats were not necessarily those with the greatest exposure.

Conclusions - Damage

Typha spp. is more vulnerable to physical damage from wave exposure than either *Phragmites australis* or *Schoenoplectus* spp., and *Typha* spp. is particularly vulnerable to extensive damage along north-facing shorelines in Netley-Libau Marsh. Whereas the conclusion about exposure stress is broadly generalizable to other wetland systems, the conclusion about northness is not. This is because the effect of direction in Netley-Libau Marsh cannot be isolated from the seiche-induced effect of high water converging with north winds. The conclusions about northness may be generalizable to other coastal freshwater marshes that are situated along the north-facing shoreline of a large lake that undergoes seiche events, such as Delta Marsh. However further research would be necessary to support this claim.

4.5.3 *Typha* spp.

My results for the presence/absence of *Typha* spp. are consistent with my conclusions about the presence/absence of damage described above. It is not surprising that *Typha* spp. is less likely to be found in areas with high exposure, since *Typha* spp. is vulnerable to physical damage from wave action. As with the results for damage, the negative relationship between the presence of *Typha* spp. and exposure suggests that *Typha* spp. has lower tolerance to wind and wave exposure than *Schoenoplectus* spp. or *Phragmites australis*. Again, it is worth keeping in mind that this conclusion runs contrary to a recent current in the scientific literature, which has emphasized the importance of a tensegrity structure in permitting *Typha* spp. to withstand strong wind conditions (Rowlatt and Morshead 1992; Witzum and Wayne 2014, 2015, 2016). To be clear, my results do not contradict the conclusion that fibre cables add strength to *Typha* spp. leaves, as these authors describe. However, my results do indicate that any support conferred by these fibre cables is markedly weaker than the alternative structures that are present in *Schoenoplectus* spp. or *Phragmites australis*.

The existing body of literature on community assembly rules does not provide consistent guidance on whether the observed pattern can be accurately described as an abiotic filter, or whether it would be preferable to discuss the influence of an exposure gradient on biotic interactions. Whereas certain authors have insisted upon a very narrow interpretation of filters and sieves (Kraft et al. 2015), gradients and filters are fundamentally complementary concepts that both relate to the influence of abiotic stress. Kraft et al. (2015) also identify an important conceptual distinction that has often been overlooked by proponents of environmental filtering, criticizing proponents for jumping to conclusions about the overwhelming influence of abiotic factors. If we observe that a particular species is absent in a given location, what basis do we have to conclude that the species has been filtered out by abiotic factors? Even if we are certain that an abiotic stressor is present, would it not be equally plausible to conclude that our species was a poor competitor in a stressful environment?

For my own purposes, it is important to acknowledge that my dataset does not give me the information to distinguish between the direct effects of exposure as an abiotic filter, and the indirect ways that biotic interactions might be influenced by relative differences in tolerance to exposure. It is conceivable that the absence of *Typha* spp. in some locations is the result of

competitive exclusion, within the context of exposure stress, rather than the direct consequence of exposure exceeding the maximum physiological tolerance of *Typha* spp. Ultimately, it seems likely that exposure stress has both direct and indirect effects on *Typha* spp. Whereas the presence of wave-induced damage provides me with an indication of the direct effects of exposure as an abiotic stressor, my dataset does not provide me with the means to draw meaningful conclusions about how biotic interactions might be influenced by differences in exposure tolerance.

Whereas the influence of effective fetch on both the presence of *Typha* spp. and the presence of damage is clear, it is less clear why *Typha* spp. is not absolutely excluded from areas with high effective fetch values, as is the case with submerged vegetation (see Anderson 1978). Further research would be required to understand why it is that some stands of *Typha* spp. appear to be capable of surviving under surprisingly high values of effective fetch, in contrast with the overall trend. Several plausible explanations will be discussed here.

It is possible that there may be meaningful differences in bathymetry, which I have not incorporated into my exposure modeling. The effective fetch model that I used provides a useful estimate of exposure, but more precise exposure models can be generated by integrating bathymetric data, along with wind and spatial data. It is conceivable that a more complex exposure model might reveal important differences in bathymetry, which result in certain areas being less exposed than they appear to be using the simpler effective fetch model. Future research would be needed to verify whether this is the case. If future researchers wish to generate an exposure model that incorporates bathymetry, it would be preferable to obtain updated bathymetric data prior to running this model.

Another possible explanation is that there may be a species-specific difference between the exposure tolerance of *Typha latifolia*, *T. angustifolia*, and *T.* X *glauca*. Because my identification method limits me to the level of the genus, it is not possible for me to comment on the likelihood that such a difference in tolerance might explain the presence of some undamaged *Typha* spp. in highly exposed areas. Whereas it seems plausible that a large proportion of the *Typha* spp. in Netley-Libau Marsh is the invasive hybrid, and this is certainly the case throughout the prairie pothole region (see Tangen et al. 2022), this question has never been studied formally in Netley-Libau Marsh. Future researchers should attempt to understand the extent and implications of the spread of *Typha* X *glauca* throughout Netley-Libau Marsh.

A distinct, yet overlapping, possibility is that unknown differences in the extent of physiological integration among ramets of a single genet may explain why some *Typha* spp. ramets can tolerate a more stressful environment than other ramets. There is some evidence to suggest that *Typha* X *glauca* can integrate functions at the level of the genet (Waters 1989, Waters and Shay 1992, Elgersma et al. 2015), and that clonal integration may be an important factor in the invasion of wetlands by *Typha* X *glauca* (Travis et al. 2011) and many other invasive plants around the world (Song et al. 2013, Liu et al. 2016).

Whereas all common taxa of emergent vegetation in prairie freshwater marshes have some degree of clonal growth (van der Valk 2012), *Typha* X *glauca* has a propensity to grow in larger clonal stands, which may be physiologically integrated on a spatial scale that exceeds the scale of the quadrats used in this study. As with the discussion about species-specific differences in exposure tolerance, my dataset does not allow me to reach a conclusion on this point of ambiguity. Future research that investigates the extent of *Typha* X *glauca* throughout Netley-Libau Marsh should also attempt to understand whether clonal integration is a contributing factor in its invasiveness.

My results for the presence/absence of *Typha* spp. are consistent with my conclusions about the presence/absence of damage, described above. These results support an interpretation that *Typha* spp. tends to grow on south-facing shorelines more frequently because these areas allow it to escape the physical damage that I observed along north-facing shorelines. Because direction is inseparable from the seiche effect, my conclusions about the influence of northness have limited generalizability in other freshwater wetlands systems. Nevertheless, my results about northness may have some applicability in other coastal freshwater marshes that experience a comparable seiche effect. Competitive exclusion remains a plausible alternative explanation for the absence of *Typha* spp. in various locations along north-facing shorelines. Ultimately, it is likely that northness influences community composition in ways that are caused by both the direct and indirect effects of abiotic stress.

It is challenging to provide a clear interpretation of why *Typha* spp. tended to have less proportional cover in the Netley South region. However, The Netley South region has some unique properties that distinguish it from the other two regions. In contrast with the other two regions, Netley South is a relatively new set of landforms, forming a miniature delta where the Red River flows into Netley Lake via the Netley Cut. Sedimentation has caused the area to become shallow

enough for the growth of plants, and vegetation has been growing on the shoals from the resident seed bank from 2018 onwards. Although many details are obscure, it is uncontroversial to expect that the Netley South region is receiving an influx of propagules from the Red River that is unique among the three regions, and that these propagules have a greater opportunity to germinate and become established on the bare mudflats that are being deposited in the delta formation of the Netley South Region.

It is conceivable that the low proportional cover of *Typha* spp. in the Netley South region is a result of greater propagule pressure amongst other taxa in this region. In contrast with Netley North and especially Libau, Netley South receives a regular influx of seeds and vegetative propagules from the Red River. However, a formal seed bank study would be needed to verify whether seed density is substantially greater in the Netley South region.

It is also conceivable that differences in elevation gradients or successional stage may explain differences in the taxonomic composition of plant communities in the Netley South region. Whereas the sampled shoreline areas in the Netley North and Libau regions consist of relatively old landforms, Netley South is a dynamic miniature delta formation, which has been progressively expanding since 2018, through the deposition of sediment from the Red River. In general, the delta orientation of landforms in the Netley South region results in a steep elevation gradient along the channels between islands, along with shallow elevation gradients downstream of the delta formation. However, my dataset does not provide me with sufficient information to comment on the role of elevation or successional stages in a meaningful way.

Finally, it is conceivable that *Typha* spp. has lower proportional cover in the Netley South region, due to edaphic factors. In contrast with most areas in Netley-Libau Marsh, the delta formation in the Netley South region consists of relatively coarse-grained sediment (Greg McCullough, unpublished data). Larger particles tend to settle out of the water column before fine particles, resulting in the preferential deposition of sandy sediment in the Netley South region. Unlike other areas, the sediment is firm, rather than mucky. Further research would be needed to verify whether low proportional abundance of *Typha* spp. can be attributed to edaphic factors.

Conclusions – *Typha* spp.

My results indicate that exposure acts as an environmental filter, limiting the growth of *Typha* spp. in exposed shoreline areas. Broadly speaking, the results for *Typha* spp. are consistent with the results for damage, supporting the conclusion that (a) *Typha* spp. is more vulnerable to exposure stress than *Schoenoplectus* spp. or *Phragmites australis*, and (b) Typha spp. is especially vulnerable to exposure stress along north-facing shorelines in Netley-Libau Marsh. At the presence/absence stage, *Typha* spp. and damage were influenced by both effective fetch and northness. However, in contrast with the analysis of damage, there was no influence of northness on the proportional cover of *Typha* spp.

Whereas it is not possible to distinguish the direct influence of abiotic stress from its indirect consequences on biotic interactions, my results provide clear evidence that *Typha* spp. is more vulnerable to exposure stress than *Schoenoplectus* spp. or *Phragmites australis*. It is conceivable that the absence of *Typha* spp. in some quadrats can be attributed to competitive exclusion within the context of exposure stress. However, further research is needed to answer this question. Biotic interactions may also be influenced by unknown differences in the influx of propagules among regions with different degrees of connectivity to the Red River. Likewise, future research would be needed to provide clarity to this point.

4.5.4 *Schoenoplectus* spp.

In contrast with *Typha* spp., *Schoenoplectus* spp. has a relatively high tolerance to exposure stress. Although there was no relationship for presence/absence, *Schoenoplectus* spp. tended to have greater proportional cover in highly exposed areas. It is challenging to provide a satisfying explanation for why *Schoenoplectus* spp. did not have a significant relationship with effective fetch at the presence/absence stage. However, it is conceivable that the presence/absence of *Schoenoplectus* spp. is driven primarily by other factors, such as the availability of propagules, elevation gradients, or successional stage. Further research will be needed to clarify why *Schoenoplectus* spp. has a positive relationship with exposure for proportional cover, but not for presence/absence.

Northness was significant at both stages of analysis, with *Schoenoplectus* spp. being both more common and more abundant in north-facing quadrats. Because it is not possible to separate the influence of direction from the influence of seiche-induced water level fluctuations, the generalizability of these results are limited. Overall, these results support the conclusion that *Schoenoplectus* spp. has a relatively high tolerance to the abiotic stress caused by the seiche effect on Lake Winnipeg.

It is not possible to separate the direct and indirect effects of abiotic stress. A plausible alternative explanation for the high proportional cover of *Schoenoplectus* spp. along north-facing shorelines is that it experiences minimal competition from *Typha* spp. in these areas. This question is particularly important in areas where *Typha* X *glauca* is abundant, since this invasive hybrid is well known for its ability to outcompete native emergent vegetation (Bansal et al. 2019, Tangen et al. 2022). Further research would be needed to distinguish the direct effects of abiotic stress from their indirect consequences for the outcome of biotic interactions.

It is challenging to provide a clear interpretation of why *Schoenoplectus* spp. tended to have greater proportional cover in the Netley South region, due to the combination of factors that are unique to the Netley South region. It is conceivable that the high proportional cover of *Schoenoplectus* spp. in the Netley South region is a result of greater propagule density in this region, through regular influx of seeds and vegetative propagules from the Red River. A formal seed bank study would be needed to verify whether seed density for *Schoenoplectus* spp. is substantially greater in the Netley

South region. If there is in fact a difference in propagule availability amongst the regions in my study, then this difference could be understood within the 'dispersal' component of the community assembly framework.

It is also conceivable that differences in elevation gradients or successional stage may explain differences in the taxonomic composition of plant communities in the Netley South region. Whereas the sampled shoreline areas in the Netley North and Libau regions consist of relatively old landforms, Netley South is a dynamic miniature delta formation, which has been progressively expanding since 2018, through the deposition of sediment from the Red River and the corresponding germination of plants from the seed bank. The delta orientation of landforms in the Netley South region results in a steep elevation gradient along the channels between islands, along with shallow elevation gradients downstream of the delta formation. However, my dataset does not provide me with sufficient information to reach conclusions about the role of elevation or successional stages.

Finally, it is conceivable that *Schoenoplectus* spp. has greater proportional cover in the Netley South region because of edaphic factors. In contrast with most areas in Netley-Libau Marsh, the delta formation in the Netley South region consists of relatively coarse-grained sediment. Larger particles tend to settle out of the water column before fine particles, resulting in the preferential deposition of sandy sediment in the Netley South region. Further research would be needed to verify whether high proportional abundance of *Schoenoplectus* spp. can be attributed to edaphic factors.

Conclusions – *Schoenoplectus* spp.

In general terms, the trends for *Schoenoplectus* spp. are the opposite of those for *Typha* spp. *Schoenoplectus* spp. appears to be highly resilient against the abiotic stressors of exposure and northness. However, it is worth noting that *Schoenoplectus* spp. had greater proportional cover at higher effective fetch values, but that effective fetch had no effect on the presence of *Schoenoplectus* spp. A plausible alternative explanation for the prevalence of *Schoenoplectus* spp. along exposed north-facing shorelines is that it experiences minimal competition from *Typha* spp. in these areas. Further research would be needed to distinguish the direct effects of abiotic stress from their indirect consequences for the outcome of biotic interactions.

4.5.5 *Phragmites australis*

Both *Phragmites australis* and *Schoenoplectus* spp. have greater tolerance to exposure stress than *Typha* spp. However, *Schoenoplectus* spp. and *Phragmites australis* had positive relationships with effective fetch at different stages of the analysis. Although there was no relationship with proportional cover, *Phragmites australis* was more likely to be present in quadrats with higher effective fetch values. There is no doubt that both taxa have a higher tolerance to exposure stress than *Typha* spp., but it is not clear whether there are different mechanisms driving their ability to tolerate this stress. Further research will be needed to investigate the underlying mechanisms that make it possible for *Schoenoplectus* spp. and *Phragmites australis* to resist exposure stress.

While this falls outside the scope of my statistical analysis, a striking feature of many shoreline zones in the Libau region display zonation patterns that defy traditional predictions, with *Phragmites australis* growing at the water's edge, and *Typha* spp. growing inland. Whereas wetland diagrams invariably situate *Phragmites australis* inland/upslope of *Typha* spp. (de Swart et al. 1994, Seabloom et al. 2001), many shoreline locations within my study area display the opposite pattern. This reversal makes sense in exposed areas if we assume that *Typha* spp. is more vulnerable to exposure stress than *Phragmites australis* is to the stress of deeper water. By growing inland of *Phragmites australis*, *Typha* spp. is shielded from exposure.

Conclusions – *Phragmites australis*

In contrast with *Typha* spp., *Phragmites australis* appears to be relatively resilient against exposure and does not appear to have any meaningful relationship with northness. In contrast with *Schoenoplectus* spp., *Phragmites australis* demonstrated tolerance for exposure at the presence/absence stage instead of the proportional cover stage, while demonstrating no relationship with northness. In general, *Phragmites australis* was present in highly exposed quadrats, but it was not very abundant in these areas. Further research would be needed to distinguish the direct effects of abiotic stress from their indirect consequences for the outcome of biotic interactions.

4.6 The Role of Exposure in Netley-Libau Marsh

4.6.1 Overview

The taxonomic composition of plant communities in the shoreline zone can be understood through a combination of (a) dispersal factors, (b) abiotic filters and/or gradients, and (c) biotic interactions (Weiher and Keddy 1999, Kraft et al. 2015). In the context of Netley-Libau Marsh, it is important to keep in mind that dispersal factors are likely to be influenced by proximity to the Red River, and the resulting influx of propagules. Whereas the primary abiotic stressor in freshwater wetland systems is water depth (van der Valk 2018), my results indicate that exposure is an abiotic stressor that can be influential in structuring plant communities in marshes with large fetch values. *Typha* spp. appears to be especially vulnerable to exposure stress.

When studying patterns of distribution among species, it can be challenging to distinguish between the influence of biotic interactions, abiotic stressors, and constraints on dispersal (D'Amen et al. 2017). Whereas dispersal factors and biotic interactions play important roles in determining the taxonomic structure of plant communities (Kraft et al. 2015), my dataset provides me with the means to comment primarily on the role of abiotic stress. My results provide strong evidence to support the conclusion that *Typha* spp. is more vulnerable to exposure stress than *Schoenoplectus* spp. or *Phragmites australis*. Exposure stress causes direct physical damage to *Typha* spp., and it is also plausible that sublethal exposure stress may result in indirect effects on biotic interactions, which result in *Typha* spp. being outcompeted. Further research would be needed to distinguish the direct effects of abiotic stress from their indirect consequences for the outcome of biotic interactions.

My conclusions about exposure as an abiotic stressor connect to the community assembly framework in a similar way to van der Valk's (1981, 2018) 'environmental sieve' model of succession in wetlands. However, it is important to keep in mind that exposure is only an important factor in wetlands with large fetch distances. In contrast with Netley-Libau Marsh, many wetlands are simply too small to generate the necessary wave energy that caused damage to *Typha* spp. in my quadrats. The influence of exposure stress is therefore much less ubiquitous than the role of water depth that van der Valk (1981, 2018) has described.

My results demonstrate that the hierarchy of tolerance among common emergent taxa is different between the stressors of water depth and exposure. Whereas tolerance to water depth is usually presented as *Schoenoplectus* spp. > *Typha* spp. (rooted) > *Phragmites australis* (Seabloom et al. 2001), or *Typha* spp. (floating mat) > *Schoenoplectus* spp. > *Phragmites australis* (Wilcox et al. 2018; Watchorn et al. 2021), it appears that tolerance to exposure stress is either *Schoenoplectus* spp. > *Phragmites australis* > *Typha* spp. or *Phragmites australis* > *Schoenoplectus* spp. > *Typha* spp.

Whereas my own results provide me with limited basis to comment on biotic interactions, there is a wealth of existing literature that speaks to the competitive advantages of the invasive hybrid, *Typha* X *glauca* (Bansal et al. 2019, Tangen et al. 2022). Furthermore, despite a lack of data on the relative prevalence of different *Typha* species within Netley-Libau Marsh, it is noteworthy that *Typha* spp. has come to represent a larger proportion of total emergent vegetation in the marsh, compared with historical proportions (Grosshans et al. 2004).

Overall, the existing literature supports an expectation that *Typha* X *glauca* has a meaningful competitive advantage over *Schoenoplectus* spp. and *Phragmites australis* in the absence of exposure stress (Bansal et al. 2019), but my results indicate that this advantage declines sharply with increases in exposure stress. Beyond a certain critical threshold, *Typha* spp. begins to sustain direct physical damage from the impacts of wave energy. Further research will be needed to clarify whether outliers in my dataset can be explained by species-specific differences in tolerance, unmeasured differences in bathymetry, unmeasured differences in clonal integration, or some other unknown factor.

4.6.2 Management Implications

There are important management implications that follow from the conclusion that *Typha* spp. is more vulnerable to exposure stress than other common emergent taxa in Netley-Libau Marsh. A central management problem for Netley-Libau Marsh has been the progressive expansion of turbid, unvegetated, open water environments (Verbiwski 1986, Grosshans et al. 2004, Watchorn et al. 2021). The corresponding loss of aquatic vegetation has led to a highly exposed environment that no longer provides the same habitat value to birds, mammals, and fish. Moreover, existing shorelines now experience greater exposure stress because of larger fetch distances (Figure 1). Within the context of conservation and restoration, it is valuable to understand whether there are any taxonomic differences in tolerance to exposure stress and the further loss of shoreline vegetation at the expense of open water.

In the context of wetland conservation, my results support the conclusion that highly exposed shorelines that are dominated by *Typha* spp. are the most vulnerable to further erosion. In Netley-Libau Marsh, north-facing shorelines are especially vulnerable to damage on a large scale, due to the influence of the seiche effect on Lake Winnipeg, and the corresponding convergence of high water levels with northerly wind. Concern should be the greatest in *Typha*-dominated areas that separate distinct bays within Netley-Libau Marsh, such as the narrow island that separates Parisian Lake from Long Lake (Figure 52). These areas are vulnerable not only to loss of vegetation, but also to the amalgamation of two marsh bays into a single, larger, bay. The amalgamation of bays leads to a sudden increase in fetch distances, which can induce the erosion of *Typha* spp. along newly exposed shorelines.

In the context of wetland restoration, and particularly in the context of revegetation projects, wetland managers should be mindful about which emergent taxa are best suited to the level of exposure stress that is present in their target restoration zone. For Netley-Libau Marsh, and perhaps for other seiche-influenced wetlands, there is also a directional component that must be taken into consideration, based on the convergence of wind direction with periods of high water. The greater the exposure stress in the target restoration zone, the less likely it will be for *Typha* spp. to successfully establish in the restoration area.

4.7 Contribution of Authors Statement

CA wrote the paper, analyzed data, collected data, drove research direction; NK supervised graduate student, provided editorial and analytical supervision; LGG supervised graduate student, provided editorial and analytical supervision, and provided research funding.

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Chapter 5. Conclusions and Recommendations

5.1 Using Remotely Piloted Aircraft for Ecological Research

The proliferation of affordable and accessible RPA technology has substantially changed the scope of ecological research that can be conducted using aerial imagery (Kalacska et al. 2017). While many RPA methods are similar to those described nearly a century ago using airplanes (Melvill Jones and Griffiths 1925), advances in software have removed major skill barriers in piloting aircraft and stitching aerial imagery. Moreover, there are affordable open-source options for mission planning software (e.g. Dronelink) and for automatically stitching overlapping aerial imagery (e.g. WedODM). Whereas early aerial surveys were limited to major imperial projects of resource exploration (Bourne 1928, Anker 2001), similar methods are now accessible to smaller research programs, citizen science groups, Indigenous-led organizations, and interested individuals. In the case of micro-RPAs, these methods can be carried out legally without the need for a basic operations certificate.

There are some important legal considerations to keep in mind, regardless of whether the RPA being used for research falls below the 250-g threshold under the *Canadian Aviation Regulations*. Importantly, for both full-sized and micro-RPAs, altitude must not exceed 120-m, and the RPA must always remain within the operator's visual line of sight (or that of a designated visual observer). These restrictions must be taken into consideration when planning a research project, and it is worth noting that larger RPAs will generally be visible from a greater distance than smaller RPAs. Whereas a Mavic Mini is visible at a maximum distance of 400 m, a DJI Phantom is visible from well over 1 km. Larger RPAs also tend to have greater ability to withstand strong wind forces, although there is a great deal of variability amongst models.

For many research purposes, automated flight software will be preferable to manual flight because of the greater precision and replicability of the flight paths. Mission planning software is particularly valuable in relatively large experimental plots that require many overlapping transects, as well as in study designs that include imagery from the same location throughout time. For research purposes where the area of interest is a long, narrow, edge habitat, open-source mapping software is somewhat limited. Whereas standard mapping missions can be carried out very easily using open-source software, corridor mapping cannot be carried out without a commercial

subscription. In the absence of a commercial subscription, single-pass corridor mapping missions can be carried out manually, as described in this thesis. If corridor mapping is required using multiple passes, then a commercial subscription will be necessary.

In remote field settings, considerations of safety and practicality will often supersede considerations of efficiency and precision. In this vein, micro-RPAs have the advantage of being minimally dangerous. Whereas the propeller blades from the Mavic Mini will not cut through human skin (personal observation), the blades from many large and mid-sized RPAs will (e.g. Spry SwellPro, personal observation). Nevertheless, caution should still be taken with micro-RPAs, and operators should consider the possibility that newer micro-RPAs will have blades that are (a) more powerful than those of the Mavic Mini, (b) sharper than those of the Mavic Mini, or (c) both. For example, the DJI Mini 3 can fly in stronger winds than the Mavic Mini, and it is also noticeably quieter than the Mavic Mini (personal observation). Regardless of whether the blades of the Mini 3 can cut human skin, these design improvements suggest that the blades will likely be capable of delivering more damage than those of the Mavic Mini. Similar design improvements should be expected in future models, and operators should exercise caution.

In the present study, the compact size of the Mavic Mini was a major practical advantage, as it simply would not have been possible to launch and land a larger RPA from the confined space of a small boat or canoe. In contrast with larger RPAs, micro-RPAs can be safely landed on the operator's hand. Conversely, a larger RPA requires a proper landing setup, which is difficult to provide without having access to a patch of dry, bare, ground. It is possible to set up a landing platform on or inside a small boat, but if the boat is being moved around by waves, it will be challenging – or even impossible – to land the RPA on a platform that is secured to the boat. It is difficult to emphasize how much a small rocking motion will impede the sensors of the RPA in recognizing the platform as a safe place to land. Whereas larger RPAs generally have a higher tolerance to windy conditions while airborne, they will be more challenging and more dangerous to land in a boat under windy conditions.

While there are advantages and disadvantages associated with both manual and automated flight, practical considerations of safety and legality are generally more important than considerations of efficiency or image quality. In summary, RPA operators should take the following categories into consideration before planning the details of image acquisition:

1. **Safety:** The propeller blades of many large RPAs can cause serious injuries, and the methods discussed here may be used in remote locations, where medical attention cannot be relied upon. Extreme caution must be taken if an operator wishes to fly an RPA from a confined space, such as a small boat or canoe. Micro-RPAs may be preferable in these cases, due to the minimal risk of injury from their propeller blades.
2. **Legality:** Whereas concerns of safety and legality are not mutually exclusive, it is important for operators to understand their legal obligations, which are not as simple as common-sense safety concerns. Even when an operator is using a micro-RPA, which does not require a basic operations certificate, the operator must fly within various limits, which are defined in the *Canadian Aviation Regulations* and the *Aeronautical Information Manual*.
3. **Accuracy and Precision:** The accuracy and precision of automated flight missions will generally exceed that of manual flight. However, manual flight provides sufficient precision and accuracy for many applications. For both manual and automated flight, the precision and accuracy of maps can be improved through the incorporation of ground control points. However, the onboard GPS data for many RPAs will provide sufficient accuracy and precision for many applications.
4. **Efficiency:** For standard mapping missions, which consist of overlapping parallel lines, automated mission software will provide greater efficiency than manual flight. However, in long/narrow areas, which call for corridor mapping methods, manual flight can exceed the efficiency of standard automated mapping missions. Automated corridor mapping missions are more efficient than manual missions, but these require a commercial subscription to Dronelink or Pix4D. Manual, single-pass corridor mapping provides a simple and efficient way to map edge habitats, such as shoreline zones or the boundary between a forest and grassland.
5. **Replicability:** If the operator wishes to return to the same area in the future and replicate a previous flight mission, automated mission planning software is strongly preferred, regardless of any considerations towards efficiency.

5.2 *Typha* spp. as the Weakest Link in Exposed Shorelines

The major substantive finding of my thesis is that *Typha* spp. is considerably more vulnerable to exposure stress than either *Schoenoplectus* spp. or *Phragmites australis*. As a matter of historical curiosity, my results support an assertion that was made by Warming (1897) and summarized in English by Clements (1916). As a matter of practical importance, my findings indicate that exposure stress has the potential to be a driving force in determining the structure of plant communities in exposed wetlands.

Whereas wetland ecologists and restoration managers are well-versed in the role of water depth and hydroperiod in structuring plant communities (van der Valk 2018, Watchorn et al. 2021), virtually nothing has been written about the importance of differences in tolerance to wave exposure among the three predominant emergent taxa. Water depth and hydroperiod are important in all wetlands, and therefore it is understandable that hydrological considerations are often treated as more important than those of exposure stress. However, *Typha* spp. is often the most common emergent taxon in prairie marshes (as was the case in this study). Consequently, the vulnerability of *Typha* spp. to exposure stress is likely to be an important factor in determining the structure of plant communities in most prairie marshes with large fetch distances.

A secondary finding of my thesis is that *Typha* spp. tends to sustain damage across a greater area along north-facing shorelines in Netley-Libau Marsh, likely due to the influence of the seiche effect on Lake Winnipeg. Whereas my findings about exposure can be readily generalized to other wetland systems, this is not necessarily the case with my results about northness. While it appears plausible that other seiche-influenced coastal marshes (e.g. Delta Marsh) will experience a similar directional effect, it is not possible to extrapolate firm conclusions from my results.

5.3 The Restoration of Netley-Libau Marsh

Over the course of the past century, Netley-Libau Marsh has become substantially degraded, such that it no longer provides the same ecological goods and services that it provided historically. The present study has focused on emergent vegetation in the shoreline zone, which represents a dynamic interface between open water and dense upland vegetation. Historically, the shoreline zone consisted of substantial tracts of hemi-marsh habitat, in which emergent macrophytes and open water were highly interspersed and each occupied roughly 50% of the surface area. In recent decades, the shoreline zone of Netley-Libau Marsh has become increasingly narrow, with less than 10 m separating areas of 100% open water from areas of 100% vegetation cover. In contrast, aerial imagery from 1929 indicates that the shoreline zone was much wider, more convoluted, and more fragmented than it is currently (Figure 1).

Whereas the manual RPA methods described in this thesis are effective under the current state of the shoreline zone in Netley-Libau Marsh, it is important to emphasize that this effectiveness is partly a function of the process of the degradation that has occurred historically, and that these methods would not have worked very well for the plant communities that were present in 1929. The methods described above are effective in Netley-Libau Marsh because the shoreline zone is (a) extremely narrow, and (b) species-poor.

If the shoreline zone had been wider and more convoluted – as was the case historically – I would have needed to fly at a higher altitude or fly a multi-pass corridor mapping mission to capture the zone of transition from 100% open water to 100% vegetation. This would have required either permission from Transport Canada to exceed 120-m or a commercial subscription to Dronelink to carry out multi-pass corridor mapping. If the shoreline zone had been more taxonomically diverse – as was the case historically – it is unclear whether I would have been able to identify all the relevant taxa. Test flights in diverse plant communities indicated that identification was considerably more difficult in diverse areas.

Previous studies have recommended regular periods of draw-down within Lake Winnipeg to encourage the re-growth of emergent vegetation in Netley-Libau Marsh (Grosshans et al. 2004, Goldsborough 2015, Watchorn et al. 2021), and the growth of vegetation under dry conditions in 2003 and 2021 indicates that such a strategy would likely be effective in eliciting a growth

response. Despite these recommendations, it is not likely that Manitoba Hydro will implement such a strategy because of the possible implications for the energy security of the province.

In the absence of regular draw-down periods, the Netley-Libau Marsh Restoration Pilot Project has been exploring the possibility of inducing vegetation growth by way of a localized reduction in water depth through the deposition of sediment in a targeted restoration area (KGS 2019, RRBC 2022). Methods for reducing water depth can include the active dredging and deposition of sediment from the Red River, or the use of passive methods to induce sediment deposition from the water as it flows through Netley Lake (Watchorn 2021, RRBC 2022). In 2021, the Netley-Libau Marsh Restoration Pilot Project carried out a small experiment, dredging sediment from the Red River and depositing it into a contained zone to build shallow mudflats within Netley Lake. Monitoring is ongoing to evaluate the effectiveness of this revegetation project. In July 2023, biodegradable baffle structures will be built in Netley Lake to test whether sediment can be effectively trapped as water flows from the Red River through Netley Lake. The results of these experiments will inform future restoration efforts in Netley-Libau Marsh.

Based on the historical trends of degradation, a successful restoration strategy for Netley-Libau Marsh should include an increase in emergent vegetation, as well as an increase in the amount of interspersion between emergent macrophytes and open water within the shoreline zone. Out of the three emergent taxa that currently dominate the shoreline zone in Netley-Libau Marsh, only *Schoenoplectus* spp. tends to grow in an interspersed pattern that bears any resemblance to the convoluted, fragmented, shorelines that were present in 1929.

The convoluted nature of *Schoenoplectus*-dominated shorelines is one reason to favour *Schoenoplectus* spp. within targeted revegetation projects. In general terms, an increase in *Schoenoplectus* spp. would lead to an increase in hemi-marsh habitat. However, it should be noted that the current conditions within Netley-Libau Marsh are more polarized towards the extreme ratios of vegetation:water than those present in the primary research on hemi-marsh habitat. Whereas researchers have found a ratio of 50:50 to be preferable to 70:30 and 30:70 (Kaminski and Prince 1981, Masto et al. 2022), most areas within Netley-Libau Marsh are closer to a ratio of 100:0 or 0:100.

Rather than focusing on a precise target ratio of 50:50, it may be preferable to focus on increasing the amount of interspersion to ensure the widest variety of habitat types for wetland species

(Murkin et al. 1997). An increase of *Schoenoplectus* spp. in the shoreline zone would widen the zone of transition, resulting in an increase in areas with a ratio of 50:50, but also an increase in areas with 70:30 and 30:70. This would be preferable to the current state of the shoreline zone, in which interspersion is very limited.

In relatively exposed areas, an additional reason to favour the establishment of *Schoenoplectus* spp. is that the alternative is often *Typha* spp., and that *Typha* spp. is vulnerable to exposure stress. For example, a preliminary revegetation strategy for Netley-Libau Marsh described *Typha* spp. as being ideally suited for revegetation projects in Netley-Libau Marsh, based on its ability to "colonize disturbed sites, thrive in fluctuating water levels, as well as persist in water depths of up to 90-100 cm." (SMM Inc. 2019). Whereas both *Typha* spp. and *Schoenoplectus* spp. are capable of tolerating relatively deep water, my results have indicated that *Typha* spp. is much more vulnerable to exposure stress than *Schoenoplectus* spp., and that it is not ideally suited for exposed areas within Netley-Libau Marsh.

5.4 Areas for Future Research

Whereas this study has concluded that *Typha* spp. is more vulnerable to exposure stress than the other common emergent taxa in Netley-Libau Marsh, the mechanism driving this vulnerability is not clear. Further research will be needed to determine whether differences in exposure tolerance are driven by differences in tensile strength, flexibility, leaf geometry, or something else entirely. While previous authors have highlighted the structural role played by fibre cables that run through the lacunae of *Typha* spp., it is conceivable that these cables contribute to a scenario of catastrophic failure, in which leaves lose all their structural support as soon as the leaf is bent, and the cable is snapped.

Future research should attempt to clarify whether there are any species-specific differences in exposure tolerance within the genus *Typha*. Whereas my results clearly indicate that *Typha* spp. is more vulnerable to exposure stress than *Schoenoplectus* spp. and *Phragmites australis*, my dataset did not allow me to make distinctions below the genus level. Future research should attempt to clarify whether there are any important differences in exposure tolerance between the invasive hybrid, *Typha* X *glauca*, and the native *Typha latifolia*.

Future research should also attempt to quantify the extent of the spread of the invasive hybrid *Typha* X *glauca* throughout Netley-Libau Marsh. Considering how prevalent *Typha* X *glauca* is within the prairie pothole region (see Tangen et al. 2022), it appears probable that it has spread throughout Netley-Libau Marsh as well. However, there have not been any studies that have attempted to estimate the spread of *Typha* X *glauca* throughout Netley-Libau Marsh. Novel identification techniques described by Wasko et al. (2022) may prove to be particularly helpful in estimating the spread of *Typha* X *glauca* throughout Netley-Libau Marsh.

While my conclusions about exposure stress are generalizable to most other wetland systems, my conclusions about northness are not. Further research will be needed to determine whether other seiche-influenced marshes experience a comparable directional influence. Nevertheless, it appears plausible that a similar effect would be experienced in Delta Marsh, Manitoba's second largest coastal freshwater marsh.

Research and restoration efforts related to Netley-Libau Marsh would benefit from having a more detailed record of historical changes in the emergent vegetation zone, using historical aerial

imagery. The National Air Photo Library (NAPL) has a set of overlapping aerial photos of Netley-Libau Marsh from 1929 that is suitable for stitching into an orthomosaic map. Figure 1 includes one photo from this collection. If possible, the rest of these photos should be purchased from the NAPL and an orthomosaic map should be produced for the entire marsh complex in 1929.

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