The ecological effects of storm surges on Arctic bird and vegetation communities in the outer Mackenzie Delta, Northwest Territories

By

Frances Nicola Angus Shipman
B.Sc., University of British Columbia, 2016

A Thesis Submitted in Partial Fulfillment of the Requirements for the Degree of

MASTER OF SCIENCE
in the School of Environmental Studies

University of Victoria

We acknowledge and respect the lək̓ʷəŋən peoples on whose traditional territory the university stands and the Songhees, Esquimalt and W̱SÁNEĆ peoples whose historical relationships with the land continue to this day.

Supervisory Committee

The ecological effects of storm surges on Arctic bird and vegetation communities in the outer Mackenzie Delta, Northwest Territories

by

Frances Nicola Angus Shipman

B.Sc., University of British Columbia, 2016

Supervisory Committee

Dr. Trevor C. Lantz, School of Environmental Studies

Supervisor

Dr. Louise K. Blight, School of Environmental Studies

Committee Member

Abstract

Coastlines in the Western Canadian Arctic are predicted to experience increases in the frequency and intensity of storm surges as rapid climate change continues to alter weather and biophysical factors of this land-sea connection. Although storm surges have the potential to cause widespread and persistent vegetation loss, little information is available about the influence of decreasing disturbance intervals (between storms), expected timelines of recovery for vegetation, and how this dramatic vegetation change alters habitat availability and/or quality for local wildlife populations. In my MSc research, I investigated how Arctic bird diversity is affected by heterogeneous vegetation recovery post-storm and characterized ecological recovery of vegetation from repeated disturbance. In the first part of my thesis (Chapter 2), I used a combination of Landsat & Sentinel satellite imagery (1984-2019) and measured post-storm soil & vegetation attributes to characterize vegetation loss and recovery in areas of the outer Mackenzie Delta (NWT) affected by storm surges in 1999 and 2016. My observations of areas affected by the 1999 storm indicate that sites farther from the river channel lacked vegetation re-establishment and had higher soil salinity. Furthermore, our analyses suggest that sites affected by the 1999 storm that were re-inundated by the 2016 storm differed in response depending on whether sites had previously revegetated or not; more vegetation re-established at the previously unvegetated sites, whereas there were decreases in the diversity of the plant community re-establishing at the previously revegetated sites. In the second part of my thesis (Chapter 3), I employed field survey protocols from the Program for Regional and International Shorebird Monitoring (PRISM) to investigate how avian community assemblage is affected by heterogeneous re-vegetation 20 years following the 1999 storm. Comparisons of my bird survey data with vegetation and habitat factors showed that the drier, post-storm vegetation barrens were preferred by ground-nesters or species that use open habitats such as lapland longspur and semipalmated plover; whereas the wetter, mostly ponded, revegetated habitats were frequented by species of ducks, red-necked phalarope, and savannah sparrow. Taken together, my research shows that areas that have revegetated after the 1999 storm can be considered as functionally recovered in comparison to our Reference (i.e., unaffected) sites in terms of vegetation and bird communities, but that areas still exist ~20 years post-storm that do not show any characteristics of recovery.

Table of Contents

List of Tables

List of Figures

Acknowledgements

Thank you to all the institutions that supported this research and to the Inuvialuit communities of the North for allowing me to complete my field work on their lands. This work was also made possible by through logistic and financial support from the University of Victoria, the Natural Sciences and Engineering Research Council of Canada, the Northern Scientific Training Program, the Royal Canadian Geographical Society, and the Arctic Institute of North America.

I appreciate the support of my supervisor, Dr. Trevor Lantz, for guiding me through this research, and to committee member, Dr. Louise Blight, for her valuable comments along the long, long thesis journey.

Thanks also to Dr. Robert Fraser and Morgan McFarlane-Winchester from Natural Resources Canada for providing the Sentinnel1a HV image analysis for the 2016 storm.

I would like to thank everyone in Group ENVI past and present for any help and small kindness along the way. I would especially like to thank all of my Arctic Landscape Ecology lab-mates (Kiyo Campbell, Tracey Proverbs, Angel Chen, Jordan Seider, Zander Chila, Hana Travers-Smith, and Tait Overeem) for encouragement and commiseration in equal measure. Also, thank you so much to those of you who were involved in my fieldwork – you know who you are and why you have my utmost gratitude. Special thanks also to Kalina Hunter (honorary ALE lab member) for the walks, chats, and coffee breaks.

Thank you and a very sincere apology to a few of my closest friends (namely Rachel and Abby) for listening to my repetitive ranting with open ears. The MVP award, however, must be presented to my parents, Rob and Barb, for providing me with food, shelter, and frequent pokes with a sharp stick during the writing portion of my thesis.

I would like to thank the COVID-19 pandemic for providing the unique opportunity for a ‘focussed writing environment’.

Finally, I would like to dedicate this thesis to my late friend Margo Dawn Greggains (1970-2021) who provided so much wisdom, encouragement, and understanding when I was struggling the most. I wish I could have shown you the finished copy, but in all honesty, you probably wouldn’t have wanted to read it anyway!

1 Introduction

1.1 Thesis Overview & Objectives

In recent decades, climate change effects have been most pronounced at high latitudes (ACIA 2005, IPCC 2018), a trend that is predicted to continue (Collins et al., 2013). Warmer temperatures, shifts in weather patterns, and lengthening of the growing season are a few of the major changes occurring across the Arctic (Post et al., 2009). Notably, climate change is expected to affect Arctic coastlines through increased coastal erosion, sea-level rise, sea-ice decline, and changes in the timing of ice break-up (Manson & Solomon 2007, Serreze et al. 2007, Simmonds & Keay, 2009, Farquharsen et al. 2018, Marsh and Lesack, 1996). Other ecological consequences of climate change include: northward expansion of species' range margins (Killengreen 2007, Post 2009, Callaghan 2004, Mustin 2007), mismatches in phenology (Both et al. 2006, Gilg et al. 2012, Robinson et al. 2009), habitat modification (Myers-Smith et al., 2011; Tape et al., 2006), and the presence of polar bear (*Ursus maritimus*) dens along the coast versus on pack ice (Fischbach et al. 2007).

1.1.1 Coastal Climate Change

In addition to the projected sea-level rise and rapid declines in sea-ice cover, coastal areas are also predicted to experience increases in storm frequency and intensity (Scavia et al. 2002, IPCC 2007, Knutson et al. 2010, Walker 1998, Comiso et al. 2008, Sepp & Jaagus 2011). Coastal storms have the potential to produce storm surges, which are an abnormal rise in sea-level along a coastline during a storm (Harris 1963). Storms occurring during the Arctic open water season (summer and fall) that develop into surges and flood inland regions have the potential to salinize lakes and impact tundra habitat (Flint et al. 2008, Reimnitz & Maurer 1979). Some surges can result in 'salt-burned' tundra where the salinization of the terrestrial habitat is significant enough to cause wide-spread vegetation die-off (Taylor 1981, Terenzi et al. 2014, Kokelj et al. 2012). This suggests that increases in storm frequency and inundation have the potential to alter vegetation composition, ecosystem function and biodiversity of Arctic coastlands (Gornish & Miller 2010, Howes et al. 2010, Byrnes et al. 2011, Torresan et al. 2012, Tate & Battaglia 2013). However, the types of vegetation susceptible to storm surge salinization and the persistence of those effects (in terms of hindered vegetation recovery) require additional study (Person & Ruess 2003, Reimnitz & Maurer 1979, Ruz et al. 1992, Kokelj et al. 2012, Lantz et al. 2015).

1.1.2 Study Rationale

Vegetation change is especially concerning in Arctic deltas because these productive wetland complexes provide critical breeding and staging habitat for shorebirds and waterfowl. Productive tundra wetlands make up a small portion (about 20%) of the available vegetated Arctic North American land area but support a disproportionate richness and abundance of birds (Smith et al. 2020). Being wetlands, they are also prone to flooding due to their low elevation and proximity to and/or direct connection to the coastline. To my knowledge, no scientific studies have addressed the effects of surge-induced habitat change on Arctic bird populations. Under the Arctic Council Circumpolar Biodiversity Monitoring Program's (CBMP) Terrestrial Monitoring Plan, birds have been designated as Focal Ecosystem Components for monitoring ongoing climate-induced changes in the Arctic (Christensen 2013). In addition to the need for monitoring avian populations that are in decline, birds in general are particularly useful indicators of environmental change due to their relative ease of detection, high mobility, and responsiveness to habitat variation (Temple & Wiens 1989, Crick 2004).

1.1.3 My Research

My MSc thesis compares areas affected and unaffected by saline incursion to examine the persistent effects of storm surge flooding on bird habitat use and vegetation structure & composition in the outer Mackenzie Delta of the Western Canadian Arctic. My thesis includes two data chapters that explore distinct research questions and are written as stand-alone manuscripts.

In Chapter 2, I use a combination of satellite remote sensing and soil & vegetation surveys to characterize ecological consequences of saline incursion in the outer Mackenzie Delta (NWT, Canada). I used remote sensing data to distinguish sites that were affected only by a storm in 1999 from sites that were affected by storms in both 1999 and 2016 to assess whether recurrent saline inundation in the same area affects how vegetation recovery following the 1999 storm is altered by the 2016 storm. Comparing vegetation and soil data from field surveys among affected site types will provide insight into patterns of habitat recovery and help characterize ecological changes in a future where storm surges are more frequent and intense.

In Chapter 3, I conducted field surveys to characterize bird assemblages in storm-affected and unaffected areas of the outer Mackenzie Delta to test the hypothesis that storm-induced ecosystem changes affect habitat availability for Arctic bird species. Combining field data with remote sensing data, I explored whether differences in bird assemblages were associated with a suite of biophysical factors to provide insight into potential drivers of avian habitat preference in storm-affected regions of the outer delta, and to assess the utility of satellite imagery for the remote monitoring of Arctic avian habitat change.

In Chapter 4, I conclude my thesis by summarizing the findings of chapters 2 and 3, discussing limitations of my studies, and proposing future research avenues. In the remainder of this chapter, I provide background information that is relevant to contextualize my research but that is too detailed to include in chapters 2 and 3.

1.2 Critical Context

1.2.1 Storm Surge History

The interacting factors that determine whether a storm produces a surge and the severity of the surge include not only storm magnitude and duration, but also direction and timing (seasonally, tidally) (Arp et al., 2010; Lynch et al., 2004; Manson et al., 2005). Sea-ice extent at the time of the storm combines with these factors to influence the severity of flooding and erosion on coastal habitats (Lynch et al. 2004) but it can vary considerably from year to year (Arp et al. 2010). Surges can also interact with changes occurring in terrestrial permafrost (Lawrence et al. 2008) such as progressive lake expansion due to thermokarst erosion (Arp et al. 2010), thaw slumping (Lantuit & Pollard, 2008), some forms of coastal erosion (Manson & Solomon, 2007), and lowering of land levels in areas of diminishing ice-rich permafrost (Couture & Pollard 2007).

Severe windstorms that cause water levels to rise are not infrequent in the outer Mackenzie Delta. Strong onshore winds are most common in late summer and can affect water levels as far south as Inuvik (Blasco 1991). When wind speed and direction are sufficient to produce a storm surge, local Inuvialuit experts attribute the severity of its impacts in part to seasonal differences in water levels in the Mackenzie River delta (Kokelj et al. 2012). Typically, severe storm surges have occurred in the fall when the river is low and the surge-induced rise in water level can extend farther into the delta than in the summer when the freshwater river outflow is higher (Kokelj et al. 2012). With the projected warmer summer temperatures in this Arctic region, a longer open-water season will lead to more opportunities for strong storms to act upon the water's surface and produce surges (Kokelj et al. 2012). Within the past century, the Mackenzie Delta has been affected by several severe storm surges (Manson & Soloman 2007), but none caused such widespread salinization and vegetation die-off as a storm in late September of 1999 (Pisaric et al. 2011). Even 7 years after the 1999 storm, extensive unvegetated areas persisted along with high soil salinity (Kokelj et al. 2012). A study investigating the effect of the 1999 storm on microcrustaceans (Phylum Anthropoda) in a delta lake found that salinization was high enough to cause extirpation of some *Cladocera* communities (Deasley et al. 2012). Although some hypotheses exist to explain the unique effects of the 1999 storm, they remain untested due to the lack of biophysical data to compare among surges. However, studies have noted the slow vegetation recovery and potential for cumulative habitat loss with future storm surges and/or other disturbances in this area (Kokelj et al. 2012). In the context of ongoing changes in climate conditions, these effects on delta communities have significant implications for ecosystems and resource management, but little is understood about how Arctic ecosystems in this area will be affected. The 1999 storm provides an opportunity to explore factors influencing vegetation recovery after a type of disturbance that is likely to become more common in this region (IPCC 2007, Comiso et al. 2008, Goulding et al. 2009, Knutson et al. 2010). Reimnitz & Maurer (1979) estimated that recurrence intervals for large storm surges in the Canadian Arctic range from 25-50 years. Based on an estimate from the 100-year storm surge along the Alaskan coast (Reimnitz & Maurer 1979), the zone of potential saltwater flooding includes areas ≤ 2.5 meters above sea-level with direct connections to the coastline (Arp et al. 2010). Although storm events influence coastal vegetation (Taylor 1981), the severity of effects upon the terrestrial environment may vary over relatively short distances along a coastline (Arp et al. 2010, Solomon 2005).

1.3 Study Area

The Mackenzie Delta (Northwest Territories) has an area of approximately 13 000 km^2, making it Canada's largest delta (Burn & Kokelj, 2009). A low-lying alluvial plain, the delta has a maximum elevation of ~10m above sea-level and gradually slopes downward towards the Beaufort Sea (Deasley et al., 2012; Vermaire et al., 2013).

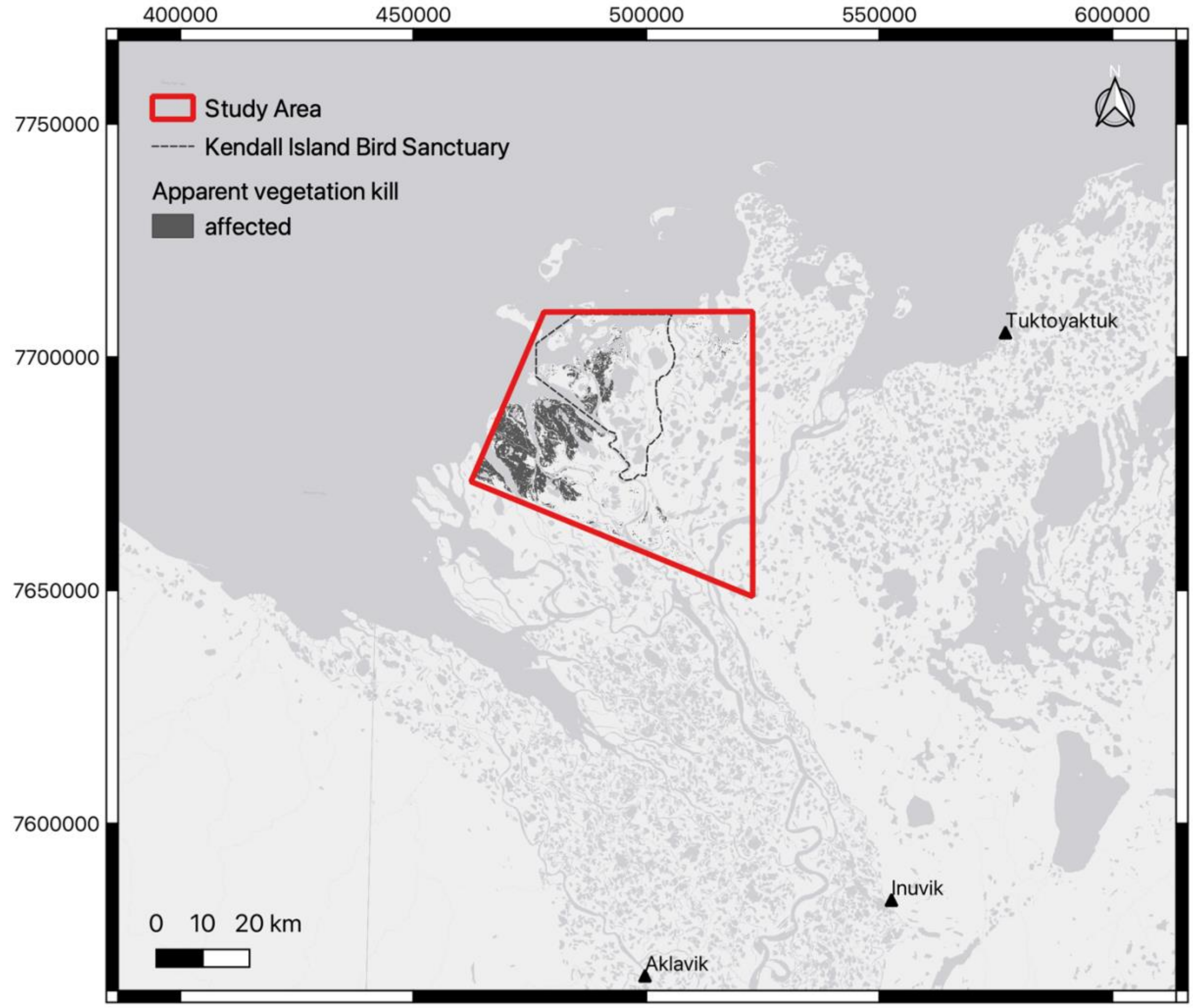


Figure 1.1. A map of the study area (outlined in red) within the Mackenzie Delta region, the 1999-affected area (in dark grey), and the boundaries of the Kendall Island Migratory Bird Sanctuary (outlined in the double-dash).

1.3.1 Inuvialuit Settlement Region

The Inuvialuit Settlement Region (ISR) is the traditional territory of the Inuvialuit and was established as part of the Inuvialuit Final Agreement (IFA) signed in 1984. The ISR spans approximately 91,000 km^2 over the entirety of the northern coasts and archipelago of Canada's Yukon and Northwest Territories. Six communities are contained within this region including: Aklavik, Inuvik, Paulatuk, Sachs Harbour, Tuktoyaktuk, and Ulukhaktok (Indian and Northern Affairs Canada [INAC] 1984). The ISR was formalized in the IFA in 1984 to recognize the land claim of the Inuit of this area (now known as Inuvialuit), maintain their rights, and protect the culture and ecosystems of their lands (INAC 1984). The Inuvialuit people retain the right to hunt and trap on these lands and many routinely engage in subsistence harvesting (INAC 1984). Through the IFA, a co-management structure was set up that recognizes five co-management bodies that work in tandem with government agencies: the Environmental Impact Screening Committee, the Environmental Impact Review Board, the Fisheries Joint Management Committee, the Wildlife Management Advisory Council: North Slope, and the Wildlife Management Advisory Council: Northwest Territories. Each of these groups is made up of equal numbers of Inuvialuit and both territorial and federal government members in order to ensure both Inuvialuit and Canadian laws and values are considered. Species regularly harvested for subsistence include (but are not limited to): caribou, moose, muskrat, geese, muskox, whitefish, seal, and beluga. A study in 2012 (Kokelj et al.) interviewing expert community members about effects of the 1999 storm surge brought the impacts of increased storm intensity on human ecosystems to light. Community experts noted the difficulty in travelling in the delta due to changes in water levels (from slumping and sedimentation post-surge) in addition to the absence of game (for subsistence hunting) in areas historically plentiful in waterfowl and/or moose (*Alces alces*) (Kokelj et al. 2012).

1.3.2 Delta Characteristics

The delta is a dynamic ecosystem underlain by deltaic and alluvial silts deposited by the Mackenzie and Peel Rivers (Rampton 1988; Canadian Wildlife Service 1992). Although it has been increasing in elevation and prograding seaward from river deposition over the past 14 000 years (Hill 1996, Manson & Soloman 2007), surveys suggest that parts of the delta front are eroding at a rate of several meters per year (Solomon 2005). The delta is also reshaped annually by ice jam flooding in the spring and occasional storm surges in the summer and fall (Hill 1996, Manson & Soloman 2007). Flooding of the delta by the spring freshet occurs on an annual basis, but salt-water flooding can also occasionally occur when sufficiently strong northwesterly winds cause storm surges (MacKay, 1963; Manson & Solomon, 2007; Vermaire et al., 2013). Along the Beaufort Sea coast, longshore currents typically come from the east (Walker, 1985), meaning westerly storms cause a rise in sea-level while easterly storms lower sea-level (Reimnitz & Maurer, 1979). This thesis focuses on the low-lying (<2m), northern, treeless part of the delta, referred to herein as the "outer delta" (Lantz et al., 2015), where gravitational tides are mild (< 0.5m) in comparison to storm surge events which can reach around 2.5 m above mean water level (Forbes & Frobel, 1985; Harper et al., 1988a; Solomon, 2005).

The frequency, duration, and timing of flooding in this region influences sedimentation, soil moisture, soil chemistry, and permafrost conditions (MacKay 1963, Marsh & Schmidt 1993, Johnstone & Kokelj 2008, Morse et al. 2009). The combination of these factors determines vegetation succession patterns (Pearce 1986, Cordes et al. 1984). Vegetation communities in the outer delta typically fall into four dominant types: graminoids, dwarf shrubs, upright shrubs, and upland tundra (Lantz et al., 2015). Terrain less than 30 cm above low tide level is generally unvegetated (Canadian Wildlife Service, 1992). Areas that flood annually are dominated by herbaceous plants whereas areas that flood 2-5 years out of 10 provide conditions for shrubs such as alder and willow (Hirst et al. 1987). For upright shrub dominated habitats, flooding typically lasts no more than a few days (Cordes et al. 1984, Pearce 1986). Dwarf shrubs grow in the most elevated surfaces of the outer delta where they are infrequently (less than every 10 years) affected by spring flooding and when flooding occurs, it is typically brief (1-2 days) (Lantz et al 2015). Alluvial vegetation (i.e. graminoid-dominated habitat) grows in areas that are flooded most frequently as this type of vegetation typically recovers rapidly from freshwater disturbance (early successional species within 1-2 years) (Pearce 1986, Johnstone & Kokelj 2008). Salt-tolerant plant species are common in Arctic coastal environments where tidal fluctuations are significant enough to influence terrestrial vegetation communities (Jefferies 1977, Handa et al. 2002); however, these species are relatively uncommon in most areas of the Mackenzie Delta that are flooded by freshwater at regular intervals (Pearce 1986, Johnstone & Kokelj 2008, Kemper & MacDonald 2009).

1.3.3 Mackenzie Delta as habitat

Although many species of mammals such as caribou (*Rangifer tarandus groenlandicus*), moose, grizzly (*Ursus arctos horribilis*) and polar bears, muskrat (*Ondatra zibethicus*), and Arctic fox (*Vulpes lagopus*) occur in the Mackenzie Delta (Berger 1978), the outer delta is of special significance to birds. The entirety of the Mackenzie Delta has been identified as “Key Migratory Bird Terrestrial Habitat” by the Canadian Wildlife Service, but the only bird-specific protected area in the outer delta is the Kendall Island Migratory Bird Sanctuary (KIMBS) (Environment Canada, 2006; Latour, 2008). This protected area, established in 1961 for the purpose of protecting the abundance of migratory birds (primarily waterbirds) in the area, encompasses approximately 606km^2 of the outer delta (Canadian Wildlife Service 1992, Bromley & Fehr 2002). Over 100 species of birds have been recorded in the sanctuary, including perching birds, shorebirds, and waterfowl (Canadian Wildlife Service, 1992; Environment and Climate Change Canada, 2019). Lesser snow geese (*Anser* *caerulescens*) use the area as a nesting and staging ground in the fall and constitute a large portion of the waterbird species (Canadian Wildlife Service, 1992). An endangered species of note, the Eskimo curlew (*Numenius borealis*), was once an abundant shorebird species that bred exclusively in the Canadian Arctic (COSEWIC, 2009). Although some recent reports indicate that a population of < 50 individuals may still survive in remote arctic regions, no records have been confirmed since 1963 (COSEWIC, 2009). There are thousands of other shorebirds nesting and staging throughout much of the delta outside of the protected area (Canadian Wildlife Service, 1992). Eleven species of Arctic-breeding shorebirds breed regularly in the outer delta and two shorebirds with limited breeding distributions in Canada (hudsonian godwit (*Limosa haemastica*) and long-billed dowitcher (*Limnodromus scolopaceus*)) nest in the outer delta (Environment Canada, 2006). This area also hosts >1% of the global breeding populations of hudsonian godwit and whimbrel (*Numenius phaeopus*) (Environment Canada, 2006). In addition to supporting significant numbers and diversity of shorebirds (Environment Canada 2006), the outer delta also provides the most important staging and breeding area for geese and swans in northern Canada (Alexander et al., 1991; Bellrose et al., 1976; Latour, 2008). Some waterfowl species of the Western Canadian Arctic are in decline (black brant goose; *Branta bernicla*), or are being harvested near their maximum limit (greater white-fronted goose; *Anser albifrons*, Canada goose; *Branta canadensis*) (Hines & Wiebe 1999; 2006). Some of the most common and numerous species to inhabit this area include savannah sparrow (*Passerculus sandwichensis*), lapland longspur (*Calcarius lapponicus*), redpoll (*Acanthis* spp.), and American tree sparrow (*Spizelloides arborea*) (Environment Canada 2006). Flood regimes in the outer delta affect the areas suitable for nesting birds as flooding patterns can change considerably from year-to-year (Gratto-Trevor 1996).

1.4 Disturbance Theory

1.4.1 Disturbance Regime

Cycles of repeated disturbance and recovery shape the dynamics of many ecosystems (White and Jensch 2001). Disturbance events produce biological legacies that interact with post-disturbance physical and environmental conditions to determine the trajectory of ecosystem recovery (Franklin et al. 2000). *Ecological memory* is a term that includes both ‘information’ legacies (species life-history traits resulting from adaptive responses to historic disturbances) and ‘material’ legacies (the biotic & abiotic structures – i.e., seeds and nutrients – left behind after a single disturbance event) (Johnstone et al. 2016). Information legacies are shaped by historical disturbances over large spatial and temporal scales, whereas material legacies are shaped by the local effects of an acute event (Johnstone et al. 2016). The characteristics of a specific disturbance can support the established ecological legacies that maintain the *'safe operating space'* of an ecosystem and result in recovery. However, when disturbance regimes and environmental conditions change overall, these legacies can be lost. This mismatch in environmental conditions and the previous disturbance legacies of an ecosystem creates a *'resilience debt'* that only becomes apparent after the system is disturbed. Identifying conditions and understanding processes that result in this misalignment between disturbance regimes and their information legacies is key in predicting when and where ecosystems will be most vulnerable to regime shifts to an alternative stable equilibrium state (Johnstone et al. 2016). An extreme disruption to information legacies can arise from a disturbance event that lies significantly outside of the historic *'safe operating space'* of the ecosystem (Johnstone et al. 2016). Nowadays, as the climate continues to change, the characteristics of disturbance regimes are also changing globally, leading to uncertainty in how disturbances will interact with their ecosystems in the future (Turner 2010). More often, changes in the severity, frequency, and/or timing of disturbances are being observed in parallel with novel disturbance events and/or disturbance interactions (Turner 2010, Trumbore et al. 2015). This rapid alteration in the nature and/or context of disturbances is leading to abrupt shifts in ecosystem states and creating novel environments in many different habitats (Frelich 2002, Hughes et al. 2013, Williams and Jackson 2007). These novel ecosystem states could be maintained by a new set of post-disturbance legacies and positive feedbacks (Scheffer et al. 2001, Bowman et al. 2015). However, anticipating whether contemporary or projected changes in the characteristics of disturbances will lead to shifts in ecosystem states is challenging because these processes unfold over decades to centuries in some ecosystems depending on the nature of the disturbance (Hughes et al. 2013, Ghazoul et al. 2015). Regardless, predicting future responses to disturbance in the face of ongoing climate change is increasingly important, especially in areas where human livelihoods and well-being are dependent on maintaining ecosystem structure and function (Seidl et al. 2016).

1.4.2 Resilience and Recovery

After a disturbance, feedback mechanisms in an ecosystem may stabilize a new system or subdue the effects of a state change (Bowman et al. 2015). Vegetation changes can feed into this process to either amplify or dampen the effects of climate change on disturbance regimes (Johnstone et al. 2016). Due to this interaction, it is especially important to understand and anticipate shifts in the dominance of plant species in areas experiencing disturbance. Ecosystem resilience is affected by the combination of individual disturbance effects, interactions among disturbances, and climate variability (Johnstone et al. 2016). As environmental change alters the context in which ecosystems recover from disturbance (Trumbore et al. 2015), ecosystems may be forced from a dynamic (but generally stationary) equilibrium state to non-equilibrium dynamics due to sustained, directional climate change effects (Johnstone et al. 2016). An extreme event may be a pivotal trigger in the degradation of an ecosystem's resilience by disrupting the relationship between species life-history traits and the disturbance characteristics to which they were adapted (Johnstone et al. 2016). When changes in an environment (i.e., changes in weather patterns, changes in dominant vegetation) contribute to changes in the frequency or intensity of regular disturbances, disturbances can 'link' to one another (Simard et al. 2011) where the material legacies of one disturbance affect the likelihood, extent, or severity of the other (Johnstone et al. 2016). 'Compound effects' from two disturbances can occur when the preceding disturbance damages the material legacies necessary for recovery following the latter disturbance event (i.e., deadwood, seeds) (Paine et al. 1998). When material legacies from a disturbance are unable to confer information about past recovery, ecosystem resilience is degraded, and the ecosystem may be forced onto a new trajectory (Johnstone et al. 2016).

1.4.3 Management considerations in a changing climate

The interaction between disturbances and ongoing climate change are likely to produce acute and potentially unprecedented responses in ecosystems (Millar and Stephenson 2015). Disturbance regimes interacting with the greater variability in environmental conditions may increasingly lead to species recruitment failures even when material legacies remain intact (Johnstone et al. 2016). Seemingly slow responses of ecosystems to environmental change can be mistaken for resilience when the ecosystem is in fact accruing *resilience debt* (a lagged response to changing drivers) (Johnstone et al. 2016). *Resilience debt* occurs when changing disturbance regimes and environmental conditions reduce the capacity of a system to recover from a disturbance, but it becomes apparent only after the system has been disturbed (Johnstone et al. 2016). Protected ecosystems are not necessarily more resilient as they tend to be better at recovering from acute stressors, but fare no better against larger-scale climate change effects (Côté & Darling 2010). Identifying the environmental legacies that support ecosystem resilience in different environments could provide key information in anticipating when disturbances may trigger abrupt shifts in ecosystem state or when ecosystems are likely to recover to their previous stable state (Johnstone et al. 2016). Understanding resilience in the context of changing climate requires detailed knowledge of species' life-history traits, disturbance characteristics, local effects of projected climate changes, and ecosystem successional dynamics (Johnstone et al. 2016). Understanding the processes that determine when, where, and how changing climate and disturbance regimes alter ecosystem dynamics is a critical question in the 21st century (Turner 2010, Trumbore et al. 2015).

1.5 References

ACIA, 2005. Arctic Climate Impact Assessment. ACIA Overview report. Cambridge University Press. 1020 pp.

Alexander, S.A., R.S. Ferguson, and K.J. McCormick. (1991). Key migratory bird terrestrial habitat sites in the Northwest Territories. Canadian Wildlife Service Occasional Paper No. 71, Ottawa, ON.

Arp, C. D., Jones, B. M., Schmutz, J. A., Urban, F. E., & Jorgenson, M. T. (2010). Two mechanisms of aquatic and terrestrial habitat change along an Alaskan Arctic coastline. Polar Biology, 33(12), 1629-1640.

Bellrose, F.C. (1980). Ducks, geese, and swans of North America. Stackpole Books, Harrisburg, PA. 540 pp.

Berger, Thomas R. 1978. "The Mackenzie Valley Pipeline Inquiry." Osgoode Hall Law Journal 16.3: 639-647. http://digitalcommons.osgoode.yorku.ca/ohlj/vol16/iss3/5

BLASCO, S.M. 1991. Regional sea-level changes and coastal retreat in the Canadian Beaufort-Mackenzie region (Abstract). In: Marsh, P., and Ommanney, C.S.L., eds. Mackenzie Delta: Environmental interactions and implications of development. NHRI Symposium No. 4. Saskatoon, Saskatchewan: National Hydrology Research Institute, Environment Canada. 173.

Both, C., S. Bouwhuis, C.M. Lessells, and M.E. Visser. 2006. Climate change and population declines in a long-distance migratory bird. Nature 441: 81–83.

Bowman DMJS, Perry GLW, and Marston JB. 2015. Feedbacks and landscape-level vegetation dynamics. Trends Ecol Evol 30: 255–60.

Bromley, B., and Fehr, A. 2002. Birds. In: Black, S., and Fehr, A., eds. Natural history of the Western Arctic. Inuvik: Western Arctic Handbook Project. 45 – 54.

Burn, C., & Kokelj, S. (2009). The environment and permafrost of the Mackenzie Delta area. Permafrost and Periglacial Processes, 20(2), 83-105. doi:10.1002/ppp.655

Byrnes, J. E., D. C. Reed, B. J. Cardinale, K. C. Cavanaugh, S. J. Holbrook, and R. J. Schmitt. 2011. Climate-driven increases in storm frequency simplify kelp forest food webs. Global Change Biology 17:2513–2524.

Callaghan TV, Bj€ orn LO, Chernov Y et al. (2004) Biodiversity, distributions and adaptations of Arctic species in the context of environmental change. Ambio, 33, 404–417.

COSEWIC. 2009. COSEWIC assessment and status report on the Eskimo Curlew Numenius borealis in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 32 pp.

Côté IM, Darling ES (2010). Rethinking Ecosystem Resilience in the Face of Climate Change. PLoS Biol 8(7): e1000438. doi:10.1371/journal.pbio.1000438

CWS (CANADIAN WILDLIFE SERVICE). (1992). Management of migratory bird sanctuaries in the Inuvialuit Settlement Region, Anderson River Delta Sanctuary, Banks Island Bird Sanctuary No.l, Banks Island Bird Sanctuary No. 2, Cape Parry Bird Sanctuary, Kendall Island Bird Sanctuary. Yellowknife, Northwest Territories: Environment Canada.

Christensen, T., J. Payne, M. Doyle, G. Ibarguchi, J. Taylor, N.M. Schmidt, M. Gill, M. Svoboda, et al. 2013. The Arctic terrestrial biodiversity monitoring plan. CAFF Monitoring Series Report Nr. 7. CAFF International Secretariat. Akureyri, Iceland. ISBN 978-9935-431-26-4.

Collins, M., R. Knutti, J. Arblaster, J.-L. Dufresne, T. Fichefet, P. Friedlingstein, X. Gao, W.J. Gutowski, et al. 2013. Long-term climate change: projections, commitments and irreversibility. In Climate change 2013: The physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change, ed. T.F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, and Y. Xia. Cambridge: Cambridge University Press.

Comiso, J., C. Parkinson, R. Gersten, and L. Stock. 2008. Accelerated decline in the Arctic sea ice cover. Geophysical Research Letters 35:1703.

Cordes, L. D., D. S. McLennan, and C. M. Pearce. 1984. Alluvial ecosystems on the Mackenzie Delta, N.W.T. BC Hydro, Burnaby, British Columbia, Canada.

Couture N, Pollard W (2007) Modeling geomorphic responses to climate change. Clim Chang 85:407–431

Crick, H. Q. P. 2004. The impact of climate change on birds. Ibis 146:48–56.

Deasley, K., Korosi, J. B., Thienpont, J. R., Kokelj, S. V., Pisaric, M. F. J., & Smol, J. P. (2012). Investigating the response of cladocera to a major saltwater intrusion event in an arctic lake from the outer mackenzie delta (NT, canada). Journal of Paleolimnology, 48(2), 287-296. doi:10.1007/s10933-012-9577-6

Department Of Public Works. 1971. Herschel Island: Feasibility of marine terminal. Ottawa: Engineering Programs Branch. 141 pp.

Environment and Climate Change Canada. (2019). *Kendall Island Migratory Bird Sanctuary* [Program descriptions; education and awareness]. Kendall Island Migratory Bird

Sanctuary. https://www.canada.ca/en/environment-climate-change/services/migratory-bird-sanctuaries/locations/kendall-island.html

Environment Canada (ENVCA). (2006). Topic 7: Wildlife and Wildlife Habitat Migratory Birds including Kendall Island Bird Sanctuary (J-EC-00136). Mackenzie Gas Project Environmental Assessment Review Written Submission. Environment Canada.

Farquharson, L. M., Mann, D. H., Swanson, D. K., Jones, B. M., Buzard, R. M., & Jordan, J. W. (2018). Temporal and spatial variability in coastline response to declining sea-ice in northwest Alaska. Marine Geology, 404, 71-83.

Fischbach A, Amstrup S, Douglas D (2007) Landward and eastward shift in Alaskan polar bear denning associated with recent sea ice changes. Polar Biol 30:1395–1405.

Flint P, Mallek E, King R, Schmutz J, Bollinger K and Derksen D 2008 Changes in abundance and spatial distribution of geese molting near Teshekpuk Lake, Alaska: interspecific competition or ecological change? Polar Biol. 31 549–56

Forbes DL & Frobel D (1985) Coastal erosion and sedimentation in the Canadian Beaufort Sea. Geol Surv Can Curr Res Pap 85(1B):69–80

Franklin JF, Lindenmayer DB, MacMahon JA, et al. 2000. Threads of continuity. Conserv Biol Pract 1: 8–16.

Frelich LE. 2002. Forest dynamics and disturbance regimes. Cambridge, UK: Cambridge University Press.

Ghazoul J, Burivalova Z, Garcia-Ulloa J, and King LA. 2015. Conceptualizing forest degradation. Trends Ecol Evol 30: 622–32.

Gilg, O., K.M. Kovacs, J. Aars, J. Fort, G. Gauthier, D. Gre´millet, R.A. Ims, and H. Meltofte, et al. 2012. Climate change and the ecology and evolution of Arctic vertebrates. Annals of the New York Academy of Sciences 1249: 166–190.

Gornish, E. S., and T. E. Miller. 2010. Effects of storm frequency on dune vegetation. Global Change Biology 16:2668–2675.

Goulding, H. L., T. D. Prowse, and S. Beltaos. 2009. Spatial and temporal patterns of break-up and ice-jam flooding in the Mackenzie Delta, NWT. Hydrological Processes 23:2654 –2670.

Gratto-Trevor, C. L. (1996). Use of Landsat TM imagery in determining important shorebird habitat in the outer Mackenzie Delta, Northwest Territories. Arctic, 11-22.

Handa, I.T., Harmsen, R., and Jefferies, R.L. 2002. Patterns of vegetation change and the recovery potential of degraded areas in a coastal marsh system of the Hudson Bay lowlands. Journal of Ecology 90:86 – 99.

Harper JR, Henry F, Stewart GG (1988) Maximum storm surge elevations in the Tuktoyaktuk region of the Canadian Beaufort Sea. Arctic 41:48–52

Harris, S. W., & Marshall, W. H. (1963). Ecology of water-level manipulations on a northern marsh. Ecology, 44(2), 331-343.

Hill, P.R. 1996. Late Quaternary sequence stratigraphy of the Mackenzie Delta. Canadian Journal of Earth Sciences 33(7):1064 – 1074.

Hines, J.E., and M.O. Wiebe. (1999). Status of waterfowl in the Inuvialuit Settlement Region. Report prepared for the Inuvialuit Wildlife Management Advisory Council. Canadian Wildlife Service, Yellowknife, NT.

Hines, J.E., and M.O. Wiebe (editors). (2006). Surveys of geese and swans in the Inuvialuit Settlement Region, Western Canadian Arctic, 1989–2001. Canadian Wildlife Service Occasional Paper No. 112. Ottawa, ON.

HIRST, S.M., MILES, M., BLACHUT, S.P., GOULET, L.A., and TAYLOR, R.E. 1987. Quantitative Synthesis of the Mackenzie Delta Ecosystem. Unpubl. report for Inland Waters Directorate, Western and Northern Region, Environment Canada, Edmonton, by Applied Ecology Ltd, North Vancouver. Available from Inland Waters Directorate, Environment Canada, 200 Twin Atria Building, 4999 - 98 Avenue, Edmonton, Alberta T6B 2X3, Canada.

Howes, N. C., D. M. FitzGerald, Z. J. Hughes, I. Y. Georgiou, M. A. Kulp, M. D. Miner, J. M. Smith, and J. A. Barras. 2010. Hurricane-induced failure of low salinity wetlands. Proceedings of the National Academy of Sciences USA 107:14014 –14019.

Hughes TP, Carpenter S, Rockstrom J, et al. 2013. Multiscale regime shifts and planetary boundaries. Trends Ecol Evol 28: 389–95.

Indian and Northern Affairs Canada (INAC). 1984. Western Arctic Claim: The Inuvialuit Final Agreement. Ottawa, ON.

IPCC. 2007. Intergovernmental Panel on Climate Change: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

IPCC. 2018. *Global Warming of 1.5°C. Special Report* Intergovernmental Panel on Climate Change

Jefferies, R.L. 1977. Vegetation of salt marshes at some coastal sites in Arctic North America. Journal of Ecology 65:661 – 672.

Johnstone, J. F., and S. V. Kokelj. 2008. Environmental conditions and vegetation recovery at abandoned drilling mud sumps in the Mackenzie Delta region, NWT, Canada. Arctic 61:199–211.

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M.C., Meentemeyer, R.K., Metz, M.R., Perry, G.L.W., Schoennagel, T. & Turner, M. G. (2016). Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment, 14(7), 369-378.

Kemper, J.T., and MacDonald, S.E. 2009. Effects of contemporary winter seismic exploration on Low Arctic plant communities and permafrost. Arctic, Antarctic, and Alpine Research 41(2):228 – 237.

Killengreen ST, Ims RA, Yoccoz NG, Bra then KA, Henden JA, Schott T (2007) Structural characteristics of a low Arctic tundra ecosystem and the retreat of the Arctic fox. Biological Conservation, 135, 459–472.

Knutson, T. R., J. L. McBride, J. Chan, K. Emanuel, G. Holland, C. Landsea, I. Held, J. P. Kossin, A. K. Srivastava, and M. Sugi. 2010. Tropical cyclones and climate change. Nature Geoscience 3:157–163.

Kokelj, S. V., Lantz, T. C., Solomon, S., Pisaric, M. F., Keith, D., Morse, P., Thienpont, J.R. & Esagok, D. (2012). Using multiple sources of knowledge to investigate northern environmental change: regional ecological impacts of a storm surge in the outer Mackenzie Delta, NWT. Arctic, 257-272.

Lantuit, H., & Pollard, W. H. (2008). Fifty years of coastal erosion and retrogressive thaw slump activity on Herschel Island, southern Beaufort Sea, Yukon Territory, Canada. *Geomorphology*, *95*(1), 84–102. https://doi.org/10.1016/j.geomorph.2006.07.040

Lantz, T. C., Kokelj, S. V., & Fraser, R. H. (2015). Ecological recovery in an Arctic delta following widespread saline incursion. Ecological applications, 25(1), 172-185.

Latour, P.B., J. Leger, J.E. Hines, M.L. Mallory, D.L. Mulders, H.G. Gilchrist, P.A. Smith, and D.L. Dickson. (2008). Key migratory bird terrestrial habitat sites in the Northwest Territories and Nunavut. 3rd Edition. Canadian Wildlife Service Occasional Paper, Yellowknife, NT. Available online at: (http://www.ngps.nt.ca/Upload/Interveners/Environment%20Canada/060411_EC_to_JRP. PDF).

Lawrence D, Slater A, Tomas R, Holland M, Deser C (2008) Accelerated Arctic land warming and permafrost degradation during rapid sea ice loss. Geophys Res Lett 35:1–6

Lynch A, Curry J, Brunner R, Maslanik J (2004) Toward an integrated assessment of the impact of extreme wind events on Barrow, Alaska. Am Meteorol Soc 85:209–221

Mackay, J. R. 1963. The Mackenzie Delta area, N.W.T. Department of Mines and Technical Surveys, Ottawa, Ontario, Canada.

Manson G, Solomon S, Forbes D, Atkinson D, Craymer M (2005) Spatial variability of factors influencing coastal change in the western Canadian Arctic. Geo-Mar Lett 25:138–145

Manson, G.K., and Solomon, S.M. (2007). Past and future forcing of Beaufort Sea coastal change. Atmosphere-Ocean 45:107 – 122, doi:10.3137/ao.450204.

Marsh, P., & Lesack, L. F. (1996). The hydrologic regime of perched lakes in the Mackenzie Delta: potential responses to climate change. Limnology and Oceanography, 41(5), 849-856.

Marsh, P., and T. Schmidt. 1993. Influence of a Beaufort Sea storm-surge on channel levels in the Mackenzie Delta. Arctic 46:35– 41.

Millar C and Stephenson NL. 2015. Temperate forest health in an era of emerging megadisturbance. Science 349: 823–26.

Morse, P. D., C. R. Burn, and S. V. Kokelj. 2009. Near-surface ground-ice distribution, Kendall Island Bird Sanctuary, western Arctic coast, Canada. Permafrost and Periglacial Processes 20:155–171.

Mustin K, Sutherland W, Gill J (2007) The complexity of predicting climate-induced ecological impacts. Climate Research, 35, 165–175.

Myers-Smith, I. H., Forbes, B. C., Wilmking, M., Hallinger, M., Lantz, T., Blok, D., Tape, K. D., Macias-Fauria, M., Sass-Klaassen, U., Lévesque, E., Boudreau, S., Ropars, P., Hermanutz, L., Trant, A., Collier, L. S., Weijers, S., Rozema, J., Rayback, S. A., Schmidt, N. M., … Hik, D. S. (2011). Shrub expansion in tundra ecosystems: Dynamics, impacts and research priorities. *Environmental Research Letters*, *6*(4), 045509. https://doi.org/10.1088/1748-9326/6/4/045509

Paine RT, Tegner MJ, and Johnson EA. 1998. Compounding perturbations yield ecological surprises. Ecosystems 1: 535–45.

Pearce, C. 1986. The distribution and ecology of the shoreline vegetation in the Mackenzie Delta, N.W.T. University of Calgary, Calgary, Alberta, Canada.

Person P, Ruess R (2003) Stability of a subarctic: plant community resistance to tidal inundation. Ecoscience 10:351–360

Pisaric, M. F., Thienpont, J. R., Kokelj, S. V., Nesbitt, H., Lantz, T. C., Solomon, S., & Smol, J. P. (2011). Impacts of a recent storm surge on an Arctic delta ecosystem examined in the context of the last millennium. Proceedings of the National Academy of Sciences, 108(22), 8960-8965.

Post E, Forchhammer MC, Bret-Harte MS et al. (2009) Ecological dynamics across the Arctic associated with recent climate change. Science, 325, 1355–1358.

Rampton VN (1988) Quaternary geology of the Tuktoyaktuk coastlands, Northwest Territories. Geological Survey of Canada, Memoir 423. pp. 8

Reimnitz E, Graves S, Barnes P (1988) Beaufort Sea coastal erosion, sediment flux, shoreline evolution, and the erosional shelf profile. US Geological Survey Map I-1182G

Reimnitz E, Maurer D (1979) Effects of storm surges on the Beaufort Sea coast, northern Alaska. Arctic 32:329–344

Robinson R, Crick H, Learmonth J et al. (2009) Travelling through a warming world: climate change and migratory species. Endangered Species Research, 7,87–99.

Ruz M, Hequette A, Hill P (1992) A model of coastal evolution in a transgressed thermokarst topography, Canadian Beaufort Sea. Mar Geol 106:251–278

Scavia, D., et al. 2002. Climate change impacts on US coastal and marine ecosystems. Estuaries 25:149–164.

Scheffer M, Carpenter S, Foley J, et al. 2001. Catastrophic shifts in ecosystems. Nature 413: 591–96.

Seidl R, Spies TA, Peterson DL, et al. 2016. Searching for resilience: addressing the impacts of changing disturbance regimes on forest ecosystem services. J Appl Ecol 53: 120–29.

Sepp, M., and J. Jaagus. 2011. Changes in the activity and tracks of Arctic cyclones. Climatic Change 105:577–595.

Serreze MC, Holland MM, Stroeve J (2007). Perspectives of the Arctic's shrinking ice cover. Science, 315:1533–1536.

Simard M, Romme WH, Griffin JM, and Turner MG. 2011. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Ecol Monogr 81: 3–24.

Simmonds, I., & Keay, K. (2009). Extraordinary September Arctic sea ice reductions and their relationships with storm behavior over 1979–2008. Geophysical Research Letters, 36(19).

Smith, P. A., McKinnon, L., Meltofte, H., Lanctot, R. B., Fox, A. D., Leafloor, J. O., ... & Smith, A. C. (2020). Status and trends of tundra birds across the circumpolar Arctic. Ambio, 49(3), 732-748.

Solomon, S. (2005), Spatial and temporal variability of shoreline change in the Beaufort-Mackenzie region, Northwest Territories, Canada, Geo-Mar. Lett., 25, 127–137.

Tape, K., Sturm, M., & Racine, C. (2006). The evidence for shrub expansion in Northern Alaska and the Pan-Arctic. *Global Change Biology*, *12*(4), 686–702. https://doi.org/10.1111/j.1365-2486.2006.01128.x

Tate, A. S., and L. L. Battaglia. 2013. Community disassembly and reassembly following experimental storm surge and wrack application. Journal of Vegetation Science 24:46–57.

Taylor R (1981) Shoreline vegetation of the arctic Alaska coast. Arctic 34:37–41

Temple, S. A., and J. A. Wiens. 1989. Bird populations and environmental changes: Can birds be bioindicators? American Birds 43:260–270.

Terenzi, J., Jorgenson, M. T., Ely, C. R., & Giguère, N. (2014). Storm-surge flooding on the Yukon-Kuskokwim delta, Alaska. Arctic, 360-374.

Torresan, S., A. Critto, J. Rizzi, and A. Marcomini. 2012. Assessment of coastal vulnerability to climate change hazards at the regional scale: the case study of the North Adriatic Sea. Natural Hazards and Earth System Sciences 12:2347–2368.

Trumbore S, Brando P, and Hartmann H. 2015. Forest health and global change. Science 349: 814–18.

Turner MG. 2010. Disturbance and landscape dynamics in a changing world. Ecology 91: 2833–49.

Vermaire, J. C., Pisaric, M. F., Thienpont, J. R., Mustaphi, C. J. C., Kokelj, S. V., & Smol, J. P. (2013). Arctic climate warming and sea ice declines lead to increased storm surge activity. Geophysical Research Letters, 40(7), 1386-1390.

Walker, H. J. 1998. Arctic deltas. Journal of Coastal Research 14:718–738.

White PS and Jentsch A. 2001. The search for generality in studies of disturbance and ecosystem dynamics. Prog Bot 62: 399–450.

Williams JW and Jackson ST. 2007. Novel climates, no-analog communities, and ecological surprises. Front Ecol Environ 5: 475–82.

2 Ecological response of vegetation communities to repeated disturbance from saline incursion in the Mackenzie Delta, Northwest Territories

F. Nicola A. Shipman[1], Trevor Lantz[1] [1]

1. School of Environmental Studies, University of Victoria

---

[1] This is the authorship planned for Ecological Applications. Study Design: FNAS and TCL, Fieldwork and Analysis: FNAS, Writing: FNAS, Revisions and Editing: FNAS and TCL.

2.1 Introduction

The rate at which the Arctic is warming has been approximately twice the global average (Graversen et al., 2008; Post et al., 2009; Serreze & Barry, 2011) and three times the rest of the northern hemisphere (Davy et al., 2018; Johannessen et al., 2016), a trend that is predicted to continue in the future (Collins et al., 2013). Accelerated warming at northern latitudes is causing rapid changes to the structure and composition of tundra vegetation and impacts a range of ecological processes (Post et al., 2009). Shifts in the dominant vegetation have already been documented in tundra ecosystems where previously lichen- and graminoid-dominated areas are being replaced by deciduous shrubs (Elmendorf et al., 2012; Lantz et al., 2013; Moffat et al., 2016; Ropars & Boudreau, 2012; Travers-Smith & Lantz, 2020). These, and other changes in vegetation, have been extensive enough to alter permafrost (Hanes et al., 2019), hydrology (Drake et al., 2019), global carbon cycling (Christiansen et al., 2018), and surface energy exchange (Blok et al., 2011).

Accelerated warming in the Arctic is also expected to impact coastlines through sea-level rise, sea-ice decline, increased storm activity, changes in the timing of ice break-up, and increased erosion from permafrost degradation, and indeed, these changes are already happening (Knutson et al., 2010; Manson & Solomon, 2007; Scavia et al., 2002; Serreze et al., 2007; Simmonds & Keay, 2009). These factors make Arctic coastlands increasingly susceptible to episodes of saline incursion from storm surges (Comiso et al., 2008; Sepp & Jaagus, 2011). Saline storm surges are likely to impact vegetation structure and composition as well as overall ecosystem function and biodiversity (Byrnes et al., 2011; Gornish & Miller, 2010; Howes et al., 2010; Tate & Battaglia, 2013; Torresan et al., 2012). The frequency, magnitude, and direction of storms, in combination with sea-level rise and the extent of open water (not covered by sea ice), dictate how the storm surges impact tundra habitats and freshwater ecosystems (Lynch et al., 2004; Manson et al., 2005; Reimnitz & Maurer, 1979). 'Salt-burned' tundra can result from storm surges where the salinization of the soils causes abrupt and severe vegetation die-off (Taylor, 1981). Previous studies have shown that vegetation communities are very sensitive to gradients in salinity (Lin et al., 2003), but that, in the context of storm surges, the resulting impacts can vary considerably over relatively short distances (Arp et al., 2010).

One Arctic coastal area that may be particularly vulnerable is the Mackenzie Delta, the world's second largest Arctic delta (~13,000 km^2) and a critical breeding habitat and crucial stop along migratory routes for many fish, mammal, and bird species (Ashenhurst & Hannon, 2008; Canadian Wildlife Service, 1992; Gratto-Trevor, 1996; Nagy, 2002; Stephenson, 2002). The outer Mackenzie Delta is already seeing more frequent and intense instances of saline flooding (Vermaire et al., 2013). In September of 1999, a storm occurred with a magnitude unprecedented in the last millennia (Pisaric et al., 2011; Vermaire et al., 2013). Previous studies assessing the ecological consequences of the 1999 storm show that the incursion of saline waters onto the land was severe enough to cause widespread and persistent vegetation loss in ~120 km^2 (~12,000 ha) of the outer delta (Kokelj et al., 2012; Lantz et al., 2015; Pisaric et al., 2011). Recent marine research, in addition to soil and vegetation field sampling, indicate that another significant storm surge event occurred in the same region from 19-21 July 2016 (Environment Canada, 2023; Lantz, unpubl. 2016; Scharffenberg, Whalen, et al., 2020). Water levels during these 3 days were ~1m higher than the rest of the month and when compared to average July water levels at this station from the previous and following year. Having occurred in the same area, the combined effect of these two large-magnitude storms on soils and vegetation is unknown. Due to the persistent nature of the ecological consequences from the 1999 storm, this second storm may have caused cumulative habitat loss in regions of the delta previously affected in 1999 (Kokelj et al., 2012).

Multiple studies have described the catastrophic effects of storm surges on vegetation, but few studies have assessed the types of vegetation that are susceptible to storm surge-induced salinization and the rate of recovery of vegetation communities versus the persistence of these effects (Arp et al., 2010; Kokelj et al., 2012; Lantz et al., 2015; Reimnitz & Maurer, 1979). Previous studies have noted that some areas in the Arctic coastal plains are more susceptible to salt-burn than others, but the drivers of this spatial heterogeneity are not well understood (Arp et al., 2010; Lantz et al., 2015). Field and remote sensing data from the outer Mackenzie Delta indicate that vegetation type (graminoid vs. dwarf shrub vs. upright shrub) can affect the rate of recovery of vegetation and soil properties post-storm (Lantz et al., 2015). Though salt-tolerant (halophytic) species are present in the outer delta, they have not colonized affected areas further inland (Lantz et al., 2015; Pisaric et al., 2011). Studies investigating the 1999 event in the Mackenzie Delta analysed its effects in comparison with the few historical records of other storms available and with field data collected following the event itself (Deasley et al., 2012; Kokelj et al., 2012; Pisaric et al., 2011; Thienpont et al., 2012, 2015), but few have followed up with long-term (>15 years) studies. Considering some ecological processes only become apparent after multiple decades (Collins, 2001), long-term datasets are crucial in evaluating ecosystem disturbance and recovery dynamics (Curtis et al., 2016). The rate at which vegetation recovers from saline incursion in varying plant communities, with different soil chemistries, and in various landscape positions would be invaluable in predicting how Arctic habitats will respond to increases in the frequency and intensity of storms projected for the near future. Investigating the long-term effects of recurrent saline incursions in Arctic deltaic ecosystems is vital to understanding the impacts of habitat change on the diversity of birds that this area supports (Latour et al., 2005; Smith et al., 2020) and to the human communities that rely on these areas for subsistence hunting and other culturally important activities (Proverbs & Lantz, 2020; Tyson et al., 2016; Wildlife Management Advisory Council (North Slope) & Aklavik Hunters and Trappers Committee, 2018).

The purpose of this study was to characterize the ecological consequences of repeated saline incursion from two major storm surge events in the outer Mackenzie Delta. We were interested in answering the following questions:

1. Are areas affected by the 1999 storm surge recovering vegetation two decades post-disturbance and do these recovering communities resemble unaffected vegetation communities?
2. How does recurrent disturbance from a second episode of storm surge-caused saline inundation influence vegetation recovery following the initial storm surge?

To assess the effects of repeated saline incursion on the recovery of vegetation communities we used a combination of satellite remote sensing and field surveys of soil & vegetation.

2.2 Methods

2.2.1 Study Area

The Mackenzie Delta is located at the northern continental edge of the Northwest Territories at the outflow of the Mackenzie and Peel rivers into the Beaufort Sea (Burn, 2017; Burn & Kokelj, 2009). A low-lying alluvial plain, the delta has a maximum elevation of ~10m above sea-level and gradually slopes downward to the Beaufort Sea (Deasley et al., 2012; Vermaire et al., 2013). This delta lies within the traditional territory of both the Inuvialuit and Gwich’in peoples, who continue to rely on the land for subsistence (Thompson & Millar, 2007; Usher, 2000). This study focused on the low-lying (< 2 masl; Fig. 1) northern part of the delta (hereafter the “outer delta”). Vegetation in the outer delta can be classified into three dominant types (graminoid wetland, dwarf shrub, and upright shrub) based on soil characteristics and frequency of flooding (Cordes et al., 1984; Lantz et al., 2015). Terrain <30 cm above low tide level is generally unvegetated (Canadian Wildlife Service, 1992; Cordes et al., 1984).

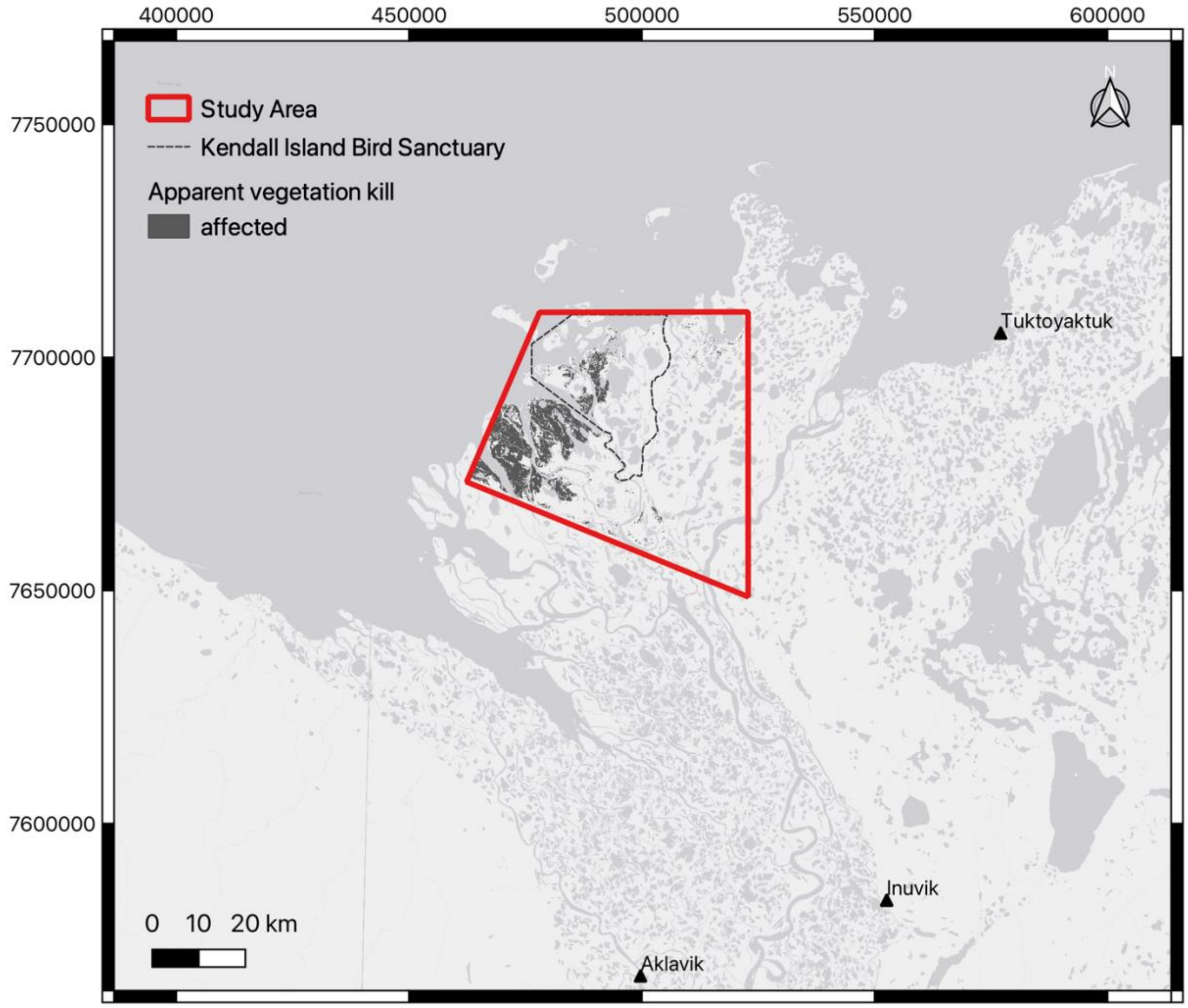


Figure 2.1. A map of our study area. It shows the extent of the Landsat imagery we assessed (outlined in red) within the Mackenzie Delta region, the 1999-affected area (in dark grey), and the boundaries of the Kendall Island Migratory Bird Sanctuary (outlined in the dash).

Within the Mackenzie Delta, the Kendall Island Migratory Bird Sanctuary encompasses approximately 606 km^2 of the outer delta (Bromley et al., 2002; Canadian Wildlife Service, 1992). Established in 1961, the purpose of this sanctuary was to protect the abundance of migratory birds (primarily waterbirds) in the area (Canadian Wildlife Service, 1992). Over 100 species of birds have been recorded in the sanctuary, including perching birds, shorebirds, and waterfowl (Canadian

Wildlife Service, 1992; Environment and Climate Change Canada, 2019). Notable species include hudsonian godwit and long-billed dowitcher, which have limited breeding grounds in Canada (Canadian Wildlife Service, 1992).

Gravitational tides in the outer delta are < 0.5m in comparison to storm surge events, which can reach around 2.5 m above mean water level (Forbes & Frobel, 1985; Harper et al., 1988b; Solomon, 2005). Storm surges are a regular occurrence in the outer delta during the open-water months (June-October) but occur more frequently in late summer and into autumn (Atkinson, 2005; Kim et al., 2021; Solomon et al., 1994). Not all storm surges result in saline incursion, as freshwater river discharge and wind direction typically keep the saltwater from reaching upstream (Carmack & Macdonald, 2002). However, as river discharge decreases over the open-water season, the potential for storm surges to cause saline flooding inland increases (Scharffenberg, Whalen, et al., 2020; Yang et al., 2015). This potential is increasing due to late summer decreases in river outflow and the extended open water season in the Arctic (Overeem et al., 2011; Vermaire et al., 2013).

2.2.2 Site Selection

To explore the effects of saline inundation on vegetation community composition and soil properties, we conducted surveys in surge-affected and surge-unaffected areas of the outer Mackenzie Delta. To assess whether repeated disturbance in the same area would affect re-vegetation, we selected sites that had been affected only in 1999 and some that had been affected both in 1999 and 2016. Since 1999-affected areas vary in their stages of vegetation recovery in the 15 years between storms, we were interested in assessing potential differences in responses of persistently dead areas and areas that displayed signs of vegetation recovery to recurrent incursion. Although vegetation community recovery from disturbance would unfold along a continuum, since we wanted to compare sites with similar types of communities that were and were not re-affected in 2016, we selected sites within areas that had remained unvegetated 1999-2015 and areas that revegetated 1999-2016 within the area affected only in 1999 and in the area affected by both storms. To select field sites for this analysis we used the Landsat record to divide the study area into five site types: 1X affected Revegetated, 1X affected Unvegetated, 2X affected Revegetated, 2X affected Unvegetated, and unaffected Reference sites. Surveys were completed at a total of 32 sites over all five site types (for divisions of replicates see Table 2.2 below). Sites were categorized using a combination of data on vegetation presence from satellite imagery, aerial photographs, and presence of surface water from satellite imagery.

Delineating affected areas

To delineate the area affected by the 1999 storm surge, we analyzed imagery from the LANDSAT 5, 7 & 8 satellites. Top of Atmosphere (TOA) imagery with less than 20% cloud cover obtained during the peak greening period (July 1 – August 15) was used to calculate the Normalized Difference Vegetation Index (NDVI) annually from 1986-2015. NDVI is calculated using the contrast between near-infrared and red reflectance (Tucker, 1979) and represents a measure of green leaf area and phytomass (Raynolds et al., 2012; Riedel et al., 2005). Using Landsat scenes from 1986-1994 we characterized inter-annual variation in NDVI before the 1999 storm and used it to establish a threshold of inter-annual variability that represented a significant departure from normal. This threshold was calculated by creating a raster stack of proportional inter-annual change from 1986-1994, where proportional inter-annual change was defined as: $[NDVI_{YEAR} - NDVI_{PREVIOUSYEAR}] / NDVI_{PREVIOUSYEAR}$. Using this stack, we calculated the average inter-annual pre-storm change on a per-pixel basis. This resulted in a single raster layer where pixels were the average inter-annual change in NDVI across all years. The mean (μ) and standard deviation (σ) of this layer were used to define the range of typical inter-annual variation as: $\mu \pm 2\sigma$, or $0.0355 \pm 2*(0.1968)$. Subsequently, the lower limit of this range was used as a threshold to delineate the extent of the area affected by the 1999 surge as $PNDVI_{2001} - PNDVI_{1998} < -0.358$, and unaffected as $PNDVI_{2001} - PNDVI_{1998} \geq -0.358$.

For delineation of the 2016 storm, we had to employ two different methods to capture affected areas. Firstly, for areas that had revegetated to some extent by 2015, we estimated the impact of the 2016 storm by applying the same thresholds from the 1999 storm to NDVI values from 2017. Specifically, we delineated areas that lost vegetation due to the 2016 storm as: $PNDVI_{2017} - PNDVI_{2015} < -0.358$ (2X affected Revegetated and four 2X affected Unvegetated). Secondly, we had to consider an additional method to delineate areas affected by the 2016 storm that had remained predominantly unvegetated since the 1999 storm (2X affected Unvegetated). Since these areas lacked vegetation in 2015, they were not identified as 2X affected by differencing 2017 and 2015 NDVI data. In order to selected for Unvegetated areas that were 're-affected' by flooding in 2016, speckle-filtered Sentinel1a HV images were used to create a polygon layer by thresholding the difference between early and late images from July 2016. These polygons are consistent with open water during this period. We then overlayed our 'affected in 1999' layer with this flooding shapefile layer to determine which Unvegetated areas had been flooded twice (2X affected Unvegetated).

Unaffected areas (Reference) were mapped as: PNDVI_{2017}- $\text{PNDVI}_{2015} \geq -0.358$ and outside of the flooding shapefile layer.

Delineating Vegetation Recovery

We selected Revegetated and Unvegetated sites within the affected portion of the outer delta using two methods that utilized satellite imagery and aerial photographs captured in 2004 (NWT Centre for Geomatics, 2007). In the first site selection method, high-resolution (effective pixel size =~1m; scale = 1:30000) air photos from 2004 (NWT Centre for Geomatics, 2007) were visually inspected. These air photos clearly show areas of bare ground and areas with green vegetation present. To verify that air photos could be used to classify sites as Revegetated or Unvegetated we extracted a subset of 2004 NDVI values from sites classified as vegetated (n=40) and unvegetated (n=44) using air photos (Table 2.1). This analysis indicates that affected sites undergoing re-vegetation had NDVI values greater than 0.25 and typically ranged from 0.303 to 0.713 (Table 2.1). NDVI values at unvegetated sites, had a maximum of 0.25 and generally ranged from 0.094 to 0.215 (Table 2.1). Based on these differences, we used 0.25 as a threshold to distinguish vegetated from unvegetated areas within the affected portion of outer delta (Table 2.1).

Table 2.1. Summary of vegetated versus unvegetated NDVI values in 2004.

Unvegetated			**Vegetated**		
Mean	Mean + 2*SD	Maximum	Mean	Minimum	Mean – 2*SD
0.1347	0.2154	0.2507	0.5080	0.2500	0.3033

Additionally, for the 1999 storm, we determined the rate of NDVI change within the affected area using a pixel-based Theil-Sen regression of a Landsat NDVI time series (2001-2012). The significance of Theil-Sen slope estimates (using a Mann-Kendall test) was assessed using a threshold of $p = 0.1$. Within affected areas, significantly trended pixels with positive slope values were classified as Revegetated and pixels with non-significant or decreasing slopes were classified as Unvegetated. This regression-based classification was compared with the air photo-based classification and field sites were selected in areas where the two classifications agreed. This method was not applicable to the 2016 storm since we only had two annual image sets post-inundation at the time (2017, 2018). Overall, we selected at least six replicates within each site type for a total of 32 sites (Table 2.2).

Table 2.2. Summary of site selection criteria using NDVI from Landsat imagery.

Site type	Classification criteria
Unaffected / Reference (n=6)	– $PNDVI_{2001}$- $PNDVI_{1998}$ > -0.358
Revegetated 2001-2012 / Affected 1X (n=6)	– $PNDVI_{2001}$- $PNDVI_{1998}$ < -0.358 – Significant increasing trend in NDVI from 2001-2012 – Green vegetation visible on 2004 air photos. – $PNDVI_{2017}$- $PNDVI_{2015}$ > -0.358
Revegetated 2001-2012 / Affected 2X (n=7)	– $PNDVI_{2001}$- $PNDVI_{1998}$ < -0.358 – Significant increasing trend in NDVI from 2001-2012 – Green vegetation visible on 2004 air photos. – $PNDVI_{2017}$- $PNDVI_{2015}$ < -0.358
Unvegetated 2001-2012 / Affected 1X (n=6)	– $PNDVI_{2001}$- $PNDVI_{1998}$ < -0.358 – No trend in NDVI from 2001-2012 – No green vegetation (brown) visible on 2004 air photos. – $PNDVI_{2017}$- $PNDVI_{2015}$ > -0.358 AND Outside polygons of 2016 flood layer.
Unvegetated 2001-2012 / Affected 2X (n=7)	– $PNDVI_{2001}$- $PNDVI_{1998}$ < -0.358 – No trend in NDVI from 2001-2012 – No green vegetation (brown) visible on 2004 air photos. – $PNDVI_{2017}$- $PNDVI_{2015}$ < -0.358 OR Inside polygons of 2016 flood layer.

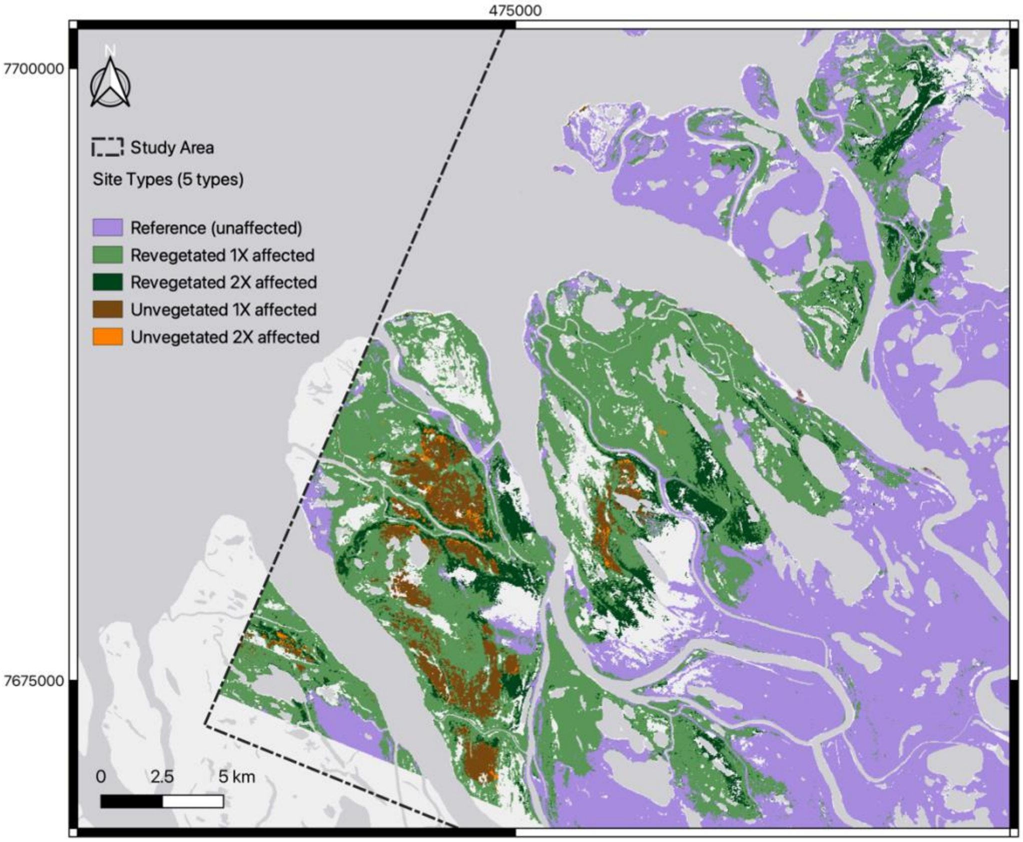


Figure 2.2. Map of five site types classified by the above classification criteria. This map was used to select sites.

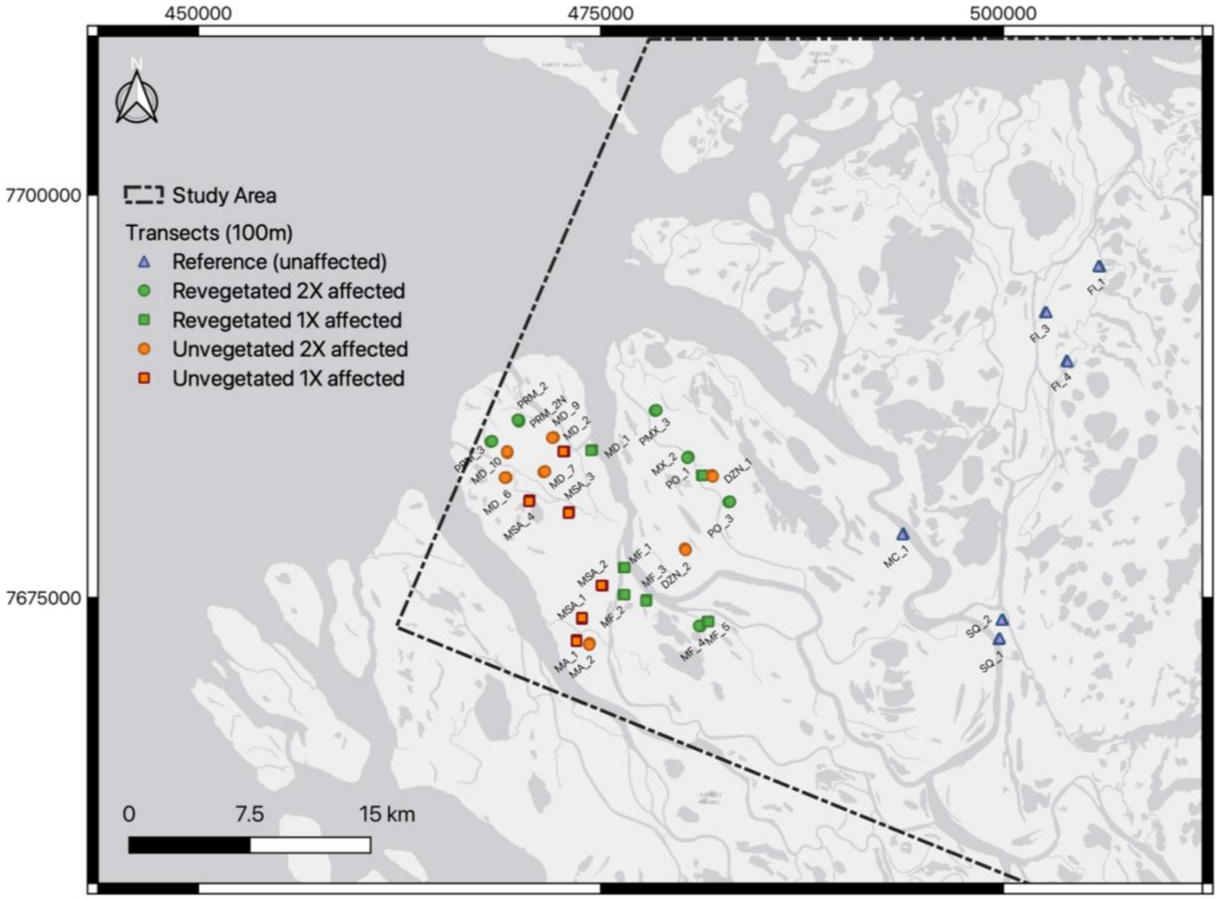


Figure 2.3. Site locations of all surveys in this study. Site types are denoted through symbols and colours; Reference sites are shown as purple triangles, Revegetated sites are shown in green, Unvegetated sites are shown in orange. 1X and 2X-affected sites are shown with square and circular symbols respectively.

2.2.3 Survey Methods

We estimated the percent cover of vegetation, thaw depth, and soil moisture in the outer delta during the summer (late June to early August) following methods previously used in the area (Lantz et al., 2015). 100m transects were set up running perpendicular to the nearest channel/water source and measurements were made every 10m along the transect. Percent cover of vegetation was assessed separately for upright shrubs and lower stature vegetation. A $4m^2$ quadrat was used for shrubs within which two $0.25m^2$ quadrats were used to estimate the cover of moss, graminoids, forbs, and lichens. If they were present, small individual shrubs were not included in the percent cover estimates for $0.25m^2$ quadrats since there cover was already estimated in the $4m^2$ quadrats. Mosses (Bryophyta) and lichens (Cryptothele) were not identified to their species levels but were recorded in grouped cover values respectively. Grasses (Poaceae spp.) and sedges (Cyperaceae spp.) were identified to the species- or genus-level when possible, but combined into their family-levels for data analysis. All other plants were identified to the species-level where possible, and to genus-level when not. The percent cover data from the two 0.25m^2 quadrats were then averaged for each 10m point. Composite active layer samples were collected at 6 points along the transect (0m, 20m, 40m, 60m, 80m, 100m) to measure salinity, pH, moisture content, and soil chemistry parameters including soluble ions and salinity following the methods outlined by McKeague (1978). Volumetric soil moisture was estimated using Delta-T Devices HH2 Moisture Meter with ML3 ThetaProbe Soil Moisture Sensor.

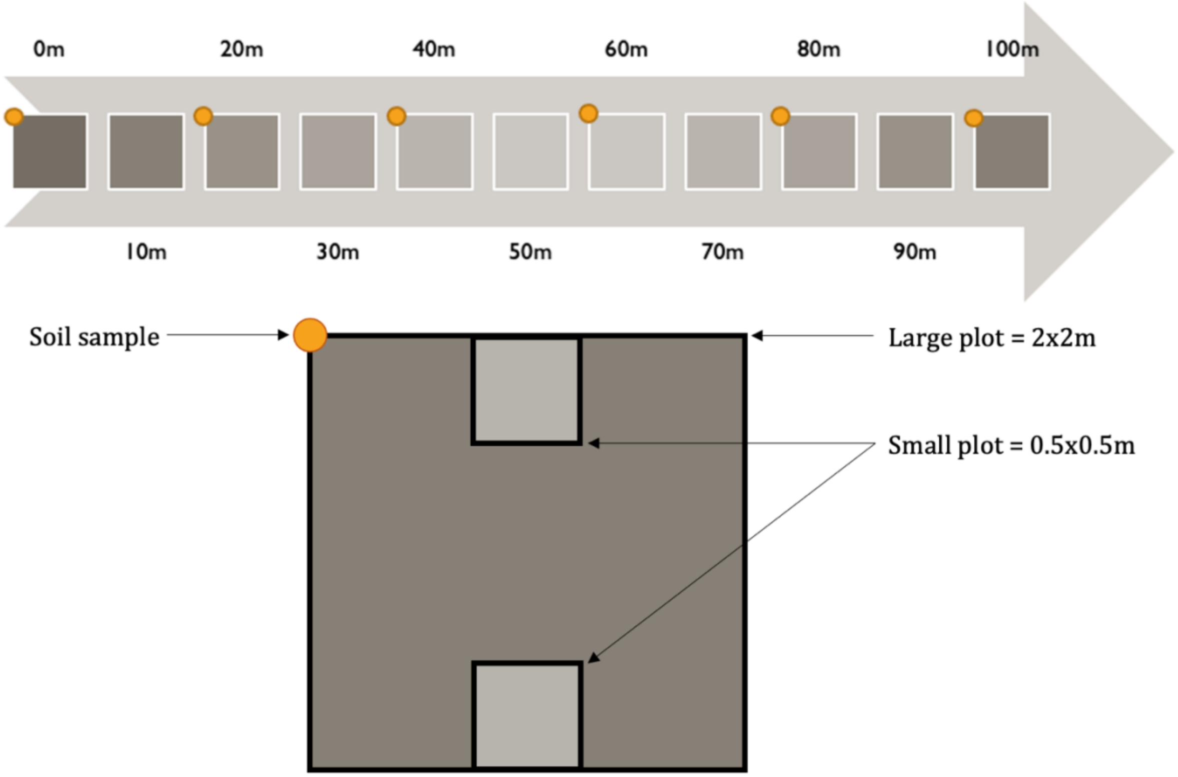


Figure 2.4. Schematic of plant and soil survey procedure along 100m transects.

Overall, 32 sites were surveyed from June 29 – August 6, 2019. This was broken down as: 6 Reference, 6 Revegetated 1X affected, 7 Revegetated 2X affected, 6 Unvegetated 1X affected, 7 Unvegetated 2X affected.

2.2.4 Remote Sensing

To assess the influence of landscape condition on the percent cover of vegetation, we used the Landsat and Sentinel archives to measure surface wetness by calculating Modified Normalized Difference Water Index (MNDWI) on Sentinel satellite imagery (from the beginning of July 2019; 10m resolution) (Du et al., 2016). Mean elevation was extracted from the ArcticDEM (2m resolution). Distance to nearest channel was also measured for each site using an MNDWI water mask derived from the Landsat 8 imagery.

2.2.5 Statistical Methods

To investigate differences in plant community composition among site types and explore relationships between plant community composition (family/genus/species) and habitat characteristics in the study area, we used Non-parametric Multidimensional Scaling (NMDS) ordination in combination with an Analysis of Similarities (ANOSIM). This analysis was completed using a Bray-Curtis dissimilarity matrix calculated from the percent cover data of the transect surveys (McGarigal et al., 2000). The relationships between habitat characteristics and plant community composition were assessed using the *envfit* function in the *vegan* package (Oksanen et al., 2015) in R (R Core Team, 2020). We also performed a Similarity Percentage (SIMPER) analysis to identify the species/genus/families making the largest contributions to community dissimilarity among site types. To characterize environmental differences among site types, a Kruskal-Wallis test (rank-based analysis of variance) was performed on a suite of variables including: 1) NDVI, 2) MNDWI, 3) distance to channel, 4) relative elevation, 5) soil pH, 6) soil salinity, 7) concentrations of major cations, 8) soil moisture, 9) thaw depth, and 10) organic soil thickness, 11) total % cover of vegetation, 12) % cover of live shrubs, 13) % cover of graminoid plants, 14) % cover of herbaceous plants, and 15) % cover of moss. Total % cover was calculated by adding the % cover from the $4m^2$ quadrat and both $0.25m^2$ quadrats together.

2.3 Results

2.3.1 Comparison of storms

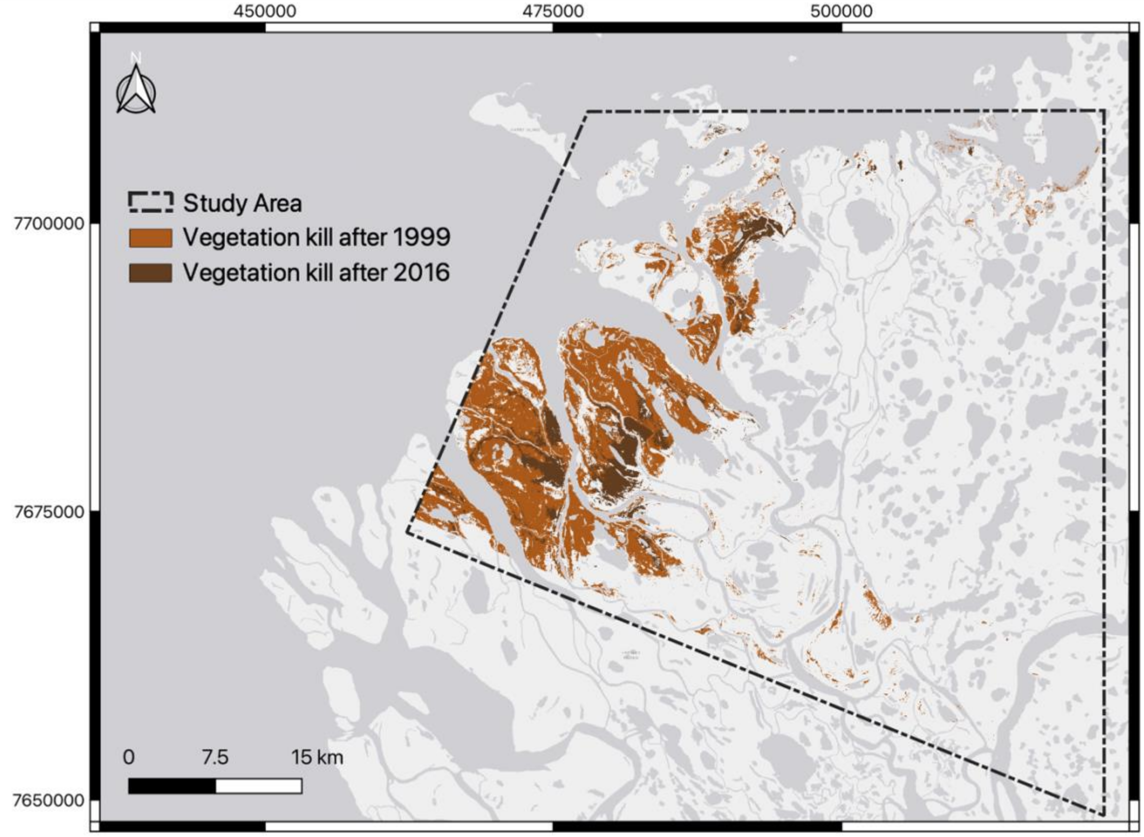


Figure 2.5. Map of the area of vegetation kill after the storms in 1999 and in 2016 estimated by differencing NDVI. Our study area is outlined by the dashed line.

The 2016 storm was 4.4 times less destructive than the 1999 storm in terms of the area of vegetation dieback (1999: 264.9843 km^2; 2016: 60.7608 km^2). By 2015, 89.8% (237.95 km^2) of the area affected in 1999 had revegetated (to 0.25 NDVI). The 2016 storm caused vegetation die back in 25.54% of this area (60.7608 km^2), and only impacted a small area outside of the area inundated in 1999. Three years following the 2016 storm (2019) 83.1% (50.4855 km^2) of the area affected by the 2016 storm had already revegetated. Comparatively, 3 years after the 1999 storm, only 35.5% (94.2039 km^2) of the affected area had revegetated. The peak height of water during the storms were 11.316m and 10.256m (gauge height above assumed datum) in 1999 and 2016 respectively (Environment Canada, 2023). High waters lasted for ~5 days in 1999 and ~3 days in 2016 (Environment Canada, 2023).

2.3.2 Plant community composition between site types

The multivariate analysis shown in Figure 2.6 indicates that vegetation community composition was significantly different among all site types; but shows that the degree of dissimilarity among site types (in comparison to within a site type) varied widely (R_{ANOSIM} = 0.137-0.692; Table 2.3).

Vegetation communities at 1X affected Unvegetated sites were the most distinct from the vegetated site types (Reference, Revegetated 1X affected, Revegetated 2X affected site types) (R_{ANOSIM} = 0.55-0.65; Table 2.3). The 1X affected Unvegetated sites had the lowest average vegetation cover and included mosses, spoonwort (*Cochlearia officinalis*), nodding campion (*Silene uralensis*) and/or grasses. Although they were occupied by the same species, the 2X affected Unvegetated sites often had more grass cover than the 1X affected Unvegetated sites. The vegetation communities at 2X affected Unvegetated sites were very dissimilar from the Reference (unaffected) sites and 2X affected Revegetated sites (R_{ANOSIM} = 0.65-0.69; Table 2.3). This was due to the relative dominance of grasses and spoonwort in combination with a lack of sedges and willow (*Salix* spp.) at the 2X affected Unvegetated sites (Table 2.5). Plant community composition at 2X affected Unvegetated sites and 1X affected Revegetated sites were moderately different from one another (R_{ANOSIM} = 0.36; Table 2.3) with higher cover of sedges at Revegetated site types and greater cover of grasses at Unvegetated site types (Table 2.5).

Reference and 1X affected Revegetated sites had species compositions that were effectively indistinguishable. The dissimilarity between these site types was also comparable to their within-group dissimilarity (R_{ANOSIM} = 0.137; Average between-group dissimilarity = 58.44%; Table 2.3 & 2.5). Reference sites were dominated by a mixture of willows and sedges, while the Revegetated 1X affected sites were dominated by a greater variety of plant types that included sedges, mosses, and willow. The species composition at the Reference sites differed considerably from that at 2X affected Revegetated site types which were characterized by greater cover of grasses, and lower cover of sedges, willow, and moss (R_{ANOSIM} = 0.435; Table 2.3 & 2.5). The 1X affected Revegetated sites also differed from the 2X affected Revegetated sites, which had increased cover of sedges and reduced cover of willow, grasses, and mosses (R_{ANOSIM} = 0.400; Table 2.3 & 2.5). Less pronounced differences between these site types (Reference vs. 2X affected Revegetated; 1X affected Revegetated vs. 2X affected Revegetated) can also be attributed to consistent cover of sedges at 2X affected Revegetated sites (Table 2.4).

The species group contributing most to differences in community composition between site types were sedges (17.82-55.38%) and willows (13.80-26.05%) (Table 2.5). The Unvegetated 1X affected site type had the most variation in community compositions (within group similarity = 24.43%) while the Revegetated 2X affected site type has the least variation in community composition between its sites (within group similarity = 57.24%) (Table 2.4).

Site types also showed differences in the presence/absence of less dominant species. Unvegetated site types (1X & 2X affected) were the only site types where seaside arrowgrass (*Triglochin maritima*), nodding campion, and Arctic buttercup (*Ranunculus hyperboreus*) were found. Arctic bearberry (*Arctous rubra*) and *Dryas* spp. were only found in unaffected Reference sites; whereas Arctic fescue (*Festuca rubra* ssp*. richardsonii*) and mare's tail (*Hippuris vulgaris*) were only found in affected site types (Revegetated (1X & 2X affected) and Unvegetated (1X & 2X affected)). Green alder (*Alnus viridis*) and common horsetail (*Equisetum arvense*) were only found in vegetated sites (Reference and 1X & 2X affected Revegetated site types; green alder was not found in 2X affected Revegetated sites). Gmelin's (yellow) buttercup (*Ranunculus gmelinii*) and white water buttercup (*Ranunculus aquatilis*) were only found in 1X & 2X affected Revegetated site types.

Figure 2.6. Nonmetric multidimensional scaling ordination of plant (species/genus/family) community composition based on Bray-Curtis similarity. The ordination shows the NMDS scores for each site type (colored symbols) and correlations between biotic and abiotic habitat variables and NMDS scores (solid arrows). The ellipses show the 95% confidence of each site type assuming a multivariate t-distribution (Wickham 2016). Plant species/genus/families are plotted in the ordination according to NMDS scores (calculated using the *scores()* function from *vegan* package). Stress = 0.136.

Table 2.3. Results of the ANOSIM analysis comparing community composition among site types. The Global R_{ANOSIM} for this test was 0.46 ($p < 0.001$).

Site types (pairwise combinations)	R_{ANOSIM} statistic
Reference / Unvegetated (2X affected)	0.649
Unvegetated (1X affected) / Unvegetated (2X affected)	0.182
Revegetated (1X affected) / Unvegetated (2X affected)	0.360
Revegetated (2X affected) / Unvegetated (2X affected)	0.692
Reference / Unvegetated (1X affected)	0.582
Reference / Revegetated (1X affected)	0.137
Reference / Revegetated (2X affected)	0.435
Revegetated (1X affected) / Unvegetated (1X affected)	0.432
Revegetated (2X affected) / Unvegetated (1X affected)	0.653
Revegetated (1X affected) / Revegetated (2X affected)	0.400

Note: All comparisons significant at $p<0.001$.

Table 2.4. Results of the SIMPER analysis characterizing similarity in vegetation community composition within the five site types. The third column shows the species/genus/families explaining the majority (>90%) of the cumulative similarity within each site type. Mean % cover (untransformed) of each species or species group at each site type is shown in the fifth column.

Site type	Average similarity (%)	Family/Species	Cumulative similarity (%)	Average % cover
Reference	51.14	Cyperaceae spp.	38.46	23.15
		Salix spp.	68.58	27.68
		Moss	91.08	21.34
Revegetated (1X affected)	42.72	Cyperaceae spp.	32.27	5.85
		Moss	51.81	12.30
		Salix spp.	68.25	8.20
		Poaceae spp.	81.31	3.39
		Equisetum variegatum	90.21	3.57
Revegetated (2X affected)	57.24	Cyperaceae spp.	96.77	26.86
Unvegetated (1X affected)	24.43	Moss	42.09	1.02
		Poaceae spp.	69.45	1.17
		Cochlearia officinalis	93.42	0.56
Unvegetated (2X affected)	37.08	Poaceae spp.	62.67	5.10
		Moss	78.11	2.49
		Cochlearia officinalis	87.08	1.34
		Triglocin maritima	92.54	0.58

Notes: The top families/species making the greatest contribution to within-group similarity (Bray-Curtis) for each site type are shown.

Table 2.5. Results of the SIMPER analysis characterizing dissimilarity in vegetation community composition between site types. The third column shows the majority (>50%) of the contribution of each species to the dissimilarity among pairwise site type combinations. Mean % cover (untransformed) of each species/genus/family at the site types being compared is shown in the fifth column.

Site type combinations	Average dissimilarity (%)	Family/Species	Contribution to dissimilarity (%)	Average % cover
Reference / Unvegetated (2X affected)	84.20	Cyperaceae spp.	23.05	23.15 / 1.24
		Salix spp.	19.53	27.68 / 0.53
		Moss	14.93	21.34 / 2.49
		Poaceae spp.	11.67	0.26 / 5.10
Unvegetated (1X affected) / Unvegetated (2X affected)	76.16	Poaceae spp.	28.88	1.17 / 5.10
		Moss	17.33	1.02 / 2.49
		Cochlearia officinalis	12.76	0.56 / 1.34
		Triglocin maritima	8.17	0.03 / 0.58
Revegetated (1X affected) / Unvegetated (2X affected)	75.75	Cyperaceae spp.	18.85	5.85 / 1.24
		Moss	15.90	12.30 / 2.49
		Salix spp.	13.80	8.20 / 0.53
		Poaceae spp.	13.64	3.39 / 5.10
Revegetated (2X affected) / Unvegetated (2X affected)	88.87	Cyperaceae spp.	38.53	26.86 / 1.24
		Poaceae spp.	18.63	0.77 / 5.10
		Moss	10.02	0.51 / 2.49
		Cochlearia officinalis	6.62	0.01 / 1.34
Reference / Unvegetated (1X affected)	92.02	Cyperaceae spp.	28.91	23.15 / 0.06
		Salix spp.	22.29	27.68 / 0.03
		Moss	16.60	21.34 / 1.02
		Equisetum variegatum	6.82	3.07 / ---
Reference / Revegetated (1X affected)	58.44	*Salix* spp.	19.99	27.68 / 8.20
		Cyperaceae spp.	18.24	23.15 / 5.85
		Moss	18.04	21.34 / 12.30
		Equisetum variegatum	10.39	3.07 / 3.57
Reference / Revegetated (2X affected)	66.26	*Salix* spp.	26.05	27.68 / 0.19
		Moss	21.36	21.34 / 0.51
		Cyperaceae spp.	17.82	23.15 / 26.86
		Equisetum variegatum	8.12	3.07 / ---
Revegetated (1X affected) / Unvegetated (1X affected)	88.01	Cyperaceae spp.	24.57	5.85 / 0.06
		Moss	16.84	12.30 / 1.02
		Salix spp.	14.23	8.20 / 0.03
		Poaceae spp.	12.49	3.39 / 1.17
Revegetated (2X affected) / Unvegetated (1X affected)	95.74	Cyperaceae spp.	55.38	26.86 / 0.06
		Poaceae spp.	10.33	0.77 / 1.17
		Moss	9.99	0.51 / 1.02
		Cochlearia officinalis	6.24	0.01 / 0.56
Revegetated (1X affected) / Revegetated (2X affected)	70.27	Cyperaceae spp.	24.57	5.85 / 26.86
		Moss	17.12	12.30 / 0.51
		Salix spp.	15.02	8.20 / 0.19
		Poaceae spp.	12.02	3.39 / 0.77

Notes: The top four species/genus/families making the greatest contribution to the between-group Bray-Curtis dissimilarity for each site combination are shown.

2.3.3 Habitat characterization of site types

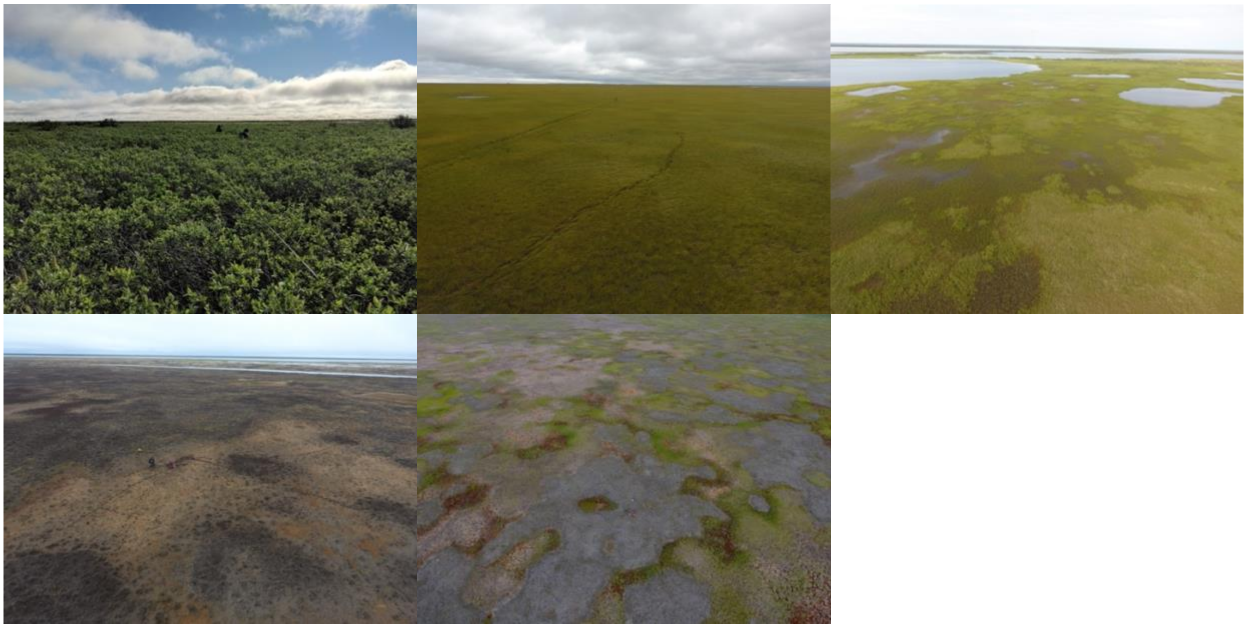


Figure 2.7. Photos from our 2019 field season showing the five site types. From left to right and top to bottom: Reference (unaffected), Revegetated (1X affected), Revegetated (2X affected), Unvegetated (1X affected), Unvegetated (2X affected).

Physical attributes tended to be similar at vegetated site types (Reference, 1X affected Revegetated, and 2X affected Revegetated), which were typically distinct from the Unvegetated (1x affected and 2X affected) site types. Both Unvegetated site types were significantly drier (at the plot scale: $4m^2$) (Figure 2.8A) and farther from channels than Reference sites or either of the Revegetated site types (Figure 2.8B). The 2X affected Revegetated sites were significantly lower in elevation than all other affected site types but were at similar elevations as the Reference sites. Only the Unvegetated site types were significantly higher than the Reference sites (Figure 2.8C).

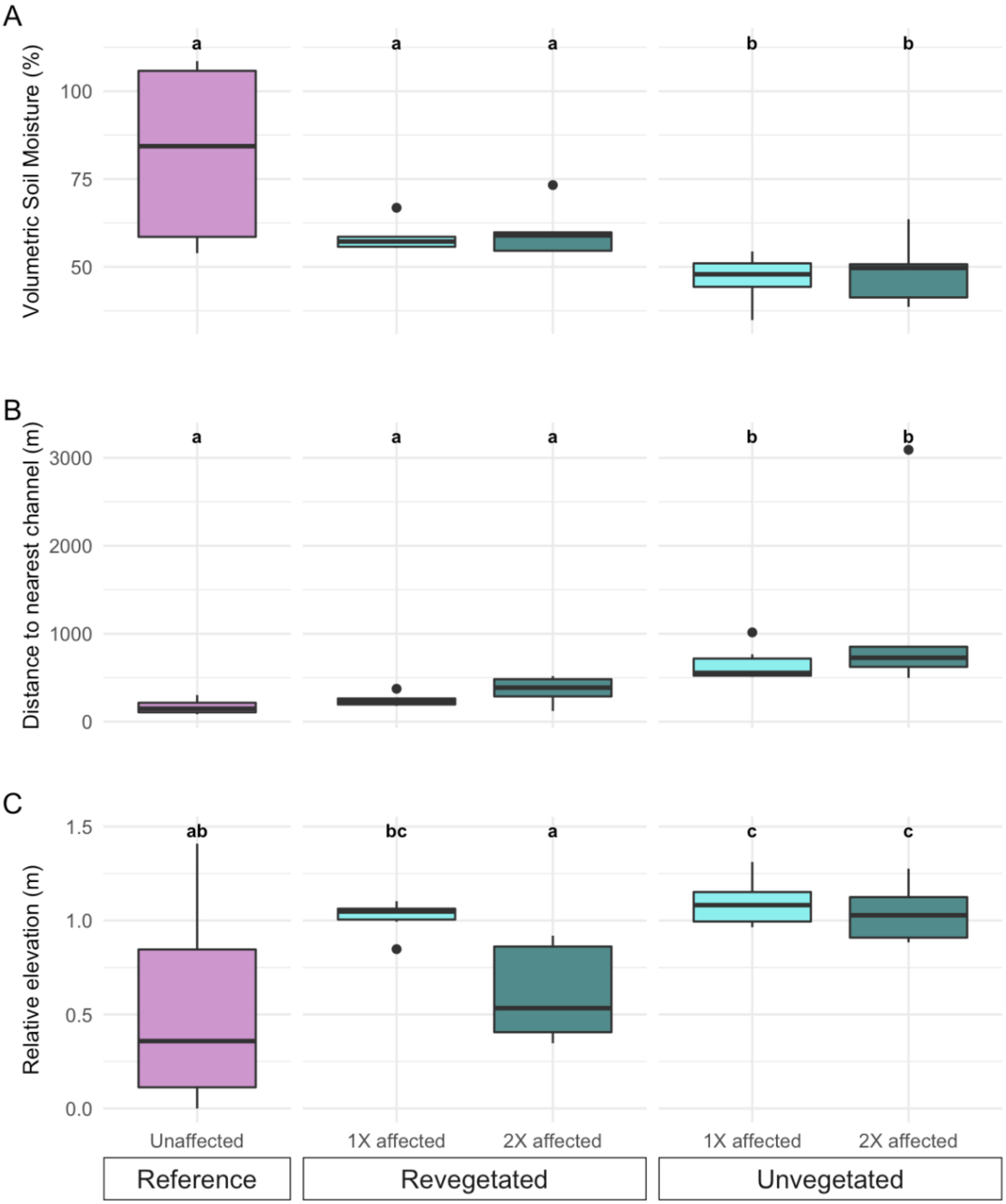


Figure 2.8. Box and whisker plots showing: A) Field soil moisture (%), B) Distance from the nearest channel (in meters), and C) Relative Elevation (in meters) at 1X affected (Revegetated and Unvegetated), 2X affected (Revegetated and Unvegetated) and unaffected (Reference) sites in the outer Mackenzie Delta. The horizontal line inside each box shows the median for each site type. The box edges show the 25th and 75th percentiles and the whiskers show [1.5*IQR] (default in R). Bars with different letters above them are significantly different (p=0.05).

Similar patterns in Unvegetated versus vegetated site types were also apparent in differences among vegetation measurements. Both Unvegetated site types had significantly lower NDVI (in 2019) at the landscape scale (900km^2) compared to the Reference and both Revegetated site types (Figure 2.9A). 1X affected Unvegetated sites also had less total vegetation cover at the plot scale (4m^2) than the Reference and Revegetated site types. However, the 2X affected Unvegetated sites were only significantly lower in vegetation than the Reference sites (not the Revegetated site types; Figure 2.9B).

The percent cover of all plant functional types was consistently higher (though not always significantly so) at Reference sites than at Unvegetated sites (Figure 2.9). The cover of most functional groups was also lower at affected Revegetated sites compared to Reference sites, but these differences were typically not significant (Figure 2.9). One notable difference is that graminoid cover was significantly higher at 2X affected Revegetated sites than most other affected site types and even higher than at unaffected Reference sites (though not significant) (Figure 2.9D). The abundance of graminoid cover also contributed heavily to the elevated total vegetation cover at these sites (Figure 2.9).

The Reference and 1X affected Revegetated site types had more live shrub cover than both the Unvegetated site types and the 2X affected Revegetated type, but this difference was not significant between 2X affected Unvegetated and 1X affected Revegetated sites (Figure 2.9C). The Reference and both Revegetated site types had more graminoid plant cover than the Unvegetated site types, although the 2X affected Unvegetated type was not significantly different from the 1X affected Revegetated and Reference site types (Figure 2.9D). Reference and 1X affected Revegetated site types had higher herbaceous plant cover than 2X affected Revegetated site type, but not significantly more than the 1X and 2X affected Unvegetated sites (Figure 2.9E). Reference and 1X affected Revegetated site types had more moss cover than both the Unvegetated site types and the 2X affected Revegetated sites, but not significantly so (Figure 2.9F).

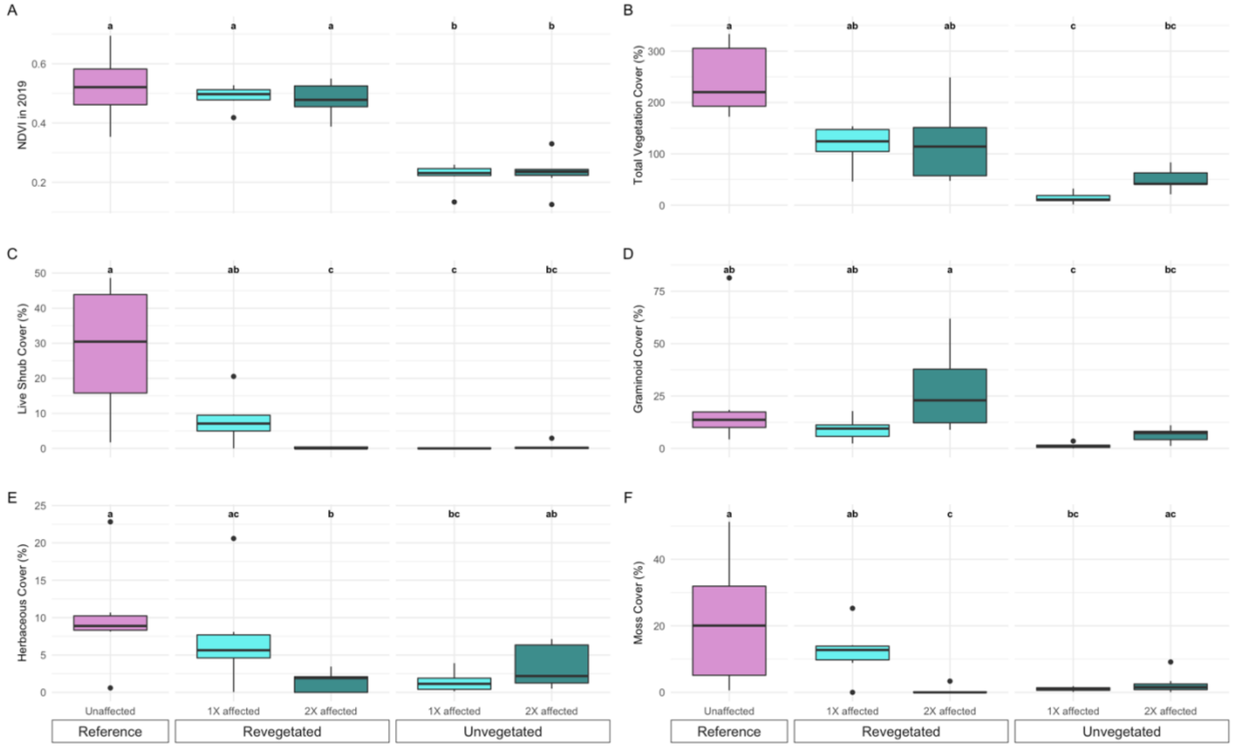


Figure 2.9. Box and whisker plots showing: A) NDVI, B) Total vegetation cover (%; cumulative % cover from all quadrats: 4m^2, (a) 0.25m^2, (b) 0.25m^2), C) % cover of live shrubs, D) % cover of graminoid plants, E) % cover of graminoid plants, F) % cover of herbaceous plants, and G) % cover of moss at 1X affected (Revegetated and Unvegetated), 2X affected (Revegetated and Unvegetated) and unaffected (Reference) sites in the outer Mackenzie Delta. The horizontal line inside box shows the median for each site type. The box edges show the 25th and 75th percentiles and the whiskers show (1.5 times the interquartile range). Bars with different letters above them are significantly different (p=0.05).

2.3.4 Soil properties of site types

For the most part, physical and chemical soil properties at Reference sites were similar to measurements made at the 1X affected Revegetated sites, which were both dissimilar from the measurements taken at both Unvegetated site types. The 2X affected Revegetated site type had overall chloride (mg/L) and sodium (mg/L) levels that were intermediate between the 1X affected Revegetated site types and the Unvegetated site types but had overall salinity (µm/cm) and magnesium (mg/L) levels that were significantly higher than both Reference and 1X affected Revegetated site types and similar to measurements from the Unvegetated site types (Figure 2.10C). Both Unvegetated site types had shallower active layer depths than the Revegetated site types, but this difference was not significant in all cases (Figure 2.10A).

1X affected Revegetated sites had significantly higher pH levels than Reference and 2X Revegetated sites, but pH levels among all other sites were not different from one another (Figure 2.10B). Both Unvegetated site types had significantly higher sodium, chloride, magnesium, and sodium than both the Reference and 1X affected Revegetated site types (Figures 2.10C, 2.10D, 2.10E, 2.10F). 2X affected Revegetated sites also had higher salinity, chloride, magnesium, and sodium compared to 1X affected Revegetated and Reference sites, but these differences were only significant for salinity and magnesium levels (Figures 2.10C, 2.10D, 2.10E, 2.10F). Reference and 1X affected Revegetated sites were lower in salinity and magnesium than all other site types and were indistinguishable from one another in all soil measurements apart from pH (Figure 2.10).

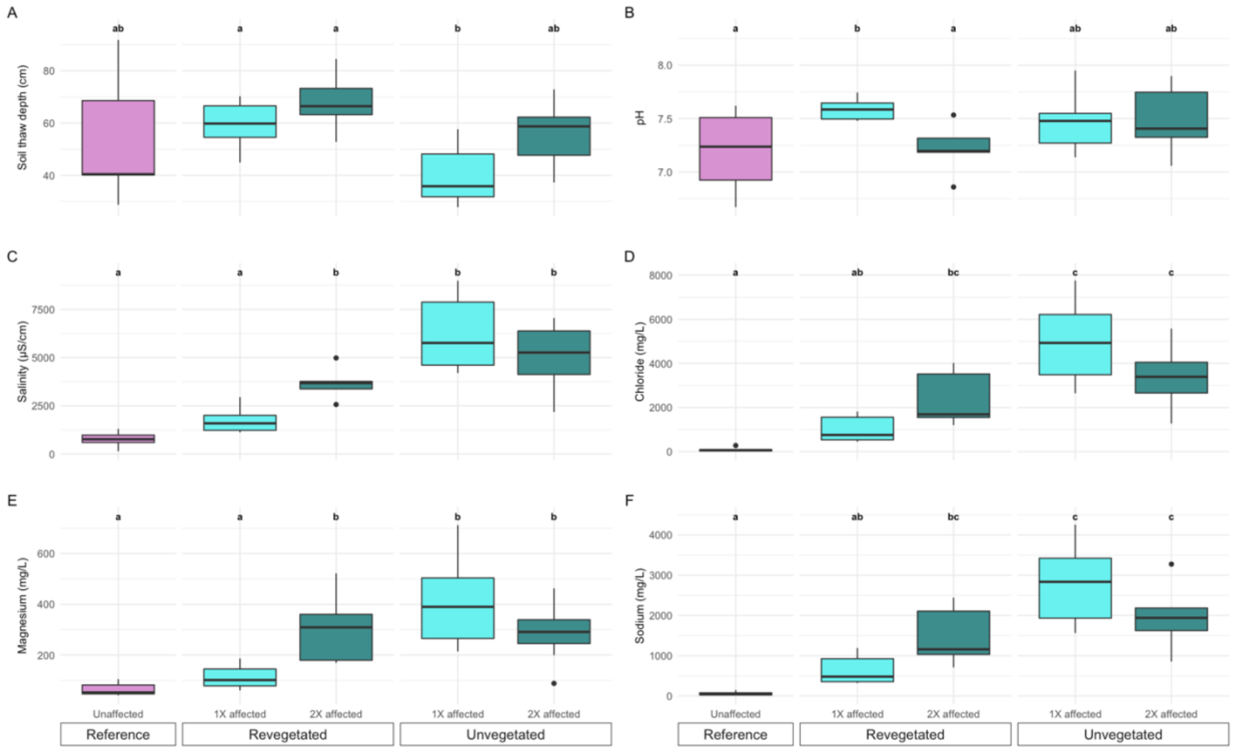


Figure 2.10. Box and whisker plots showing: A) Active layer depth (in centimeters), B) pH, C) Salinity (µS/cm), D) Chloride, E) Magnesium, and F) Sodium at 1X affected (Revegetated and Unvegetated), 2X affected (Revegetated and Unvegetated) and unaffected (Reference) sites in the outer Mackenzie Delta. The horizontal line inside box shows the median for each site type. The box edges show the 25th and 75th percentiles and the whiskers show [1.5*IQR]. Bars with different letters above them are significantly different (p=0.05).

2.4 Discussion

2.4.1 Main Findings

Our analysis shows that the effects of the 1999 storm on vegetation community composition are still apparent with reduced vegetation cover and distinct plant communities at 1X affected Unvegetated sites. This is likely due to the significantly higher salinity levels present at the 1X affected Unvegetated sites. Our study also suggests that additional flooding from a secondary saline inundation event (2016) did not hinder the revegetation of affected areas but did affect the composition of the vegetation community by reducing the cover of shrubs and moss. In our study, the storm in 2016 did not significantly affect the community composition between the 1X & 2X affected Unvegetated sites, but it did affect the vegetation recovery trajectory for the 2X affected Revegetated sites (in comparison to the 1X affected Revegetated sites). Although the remote sensing analysis showed 2X affected Revegetated sites as having similar greenness (NDVI) values as both the unaffected Reference sites and 1X affected Revegetated sites, the vegetation community composition at the 2X affected Revegetated sites was different. The 2X affected Revegetated sites were dominated by graminoids and lacked the varied cover of shrubs, herbaceous plants, and moss present at the 1X affected Revegetated sites. This is likely because less time has passed since the disturbance in 2016 than since 1999. In the Mackenzie Delta, graminoids are early successional species (re-establishment within 1-2 years) and therefore typically recover rapidly from disturbance by flooding (Johnstone & Kokelj, 2008; Lantz et al., 2015; Pearce, 1986).

Our analysis identified soil salinity levels as one of the main drivers of differences in vegetation community between the site types, which is consistent with previously published data for the 1999 storm (Lantz et al., 2015). High soil salinity is a primary factor limiting plant growth and productivity by affecting major processes such as photosynthesis, protein synthesis, and energy and lipid metabolism (Allakhverdiev et al., 2000; Parida & Das, 2005). Plant cover was highest and community composition was most varied at site types that had the lowest soil salinity levels. The only affected sites with salinity levels similar to the unaffected Reference sites were the 1X affected Revegetated sites (see below), which also had the most similar vegetation community to Reference sites. Differences in plant functional group abundance at sites also suggest that observed changes were the result of salt toxicity. Mosses, most of which are generally considered to be more susceptible than other plants to salt stress (Ćosić et al., 2020; Jorgenson, 2000; Jorgenson & Ely, 2001), were scarce in site types with higher soil salinity. Shrubs (mostly willows) were also low in abundance at sites with higher salinity because high levels of sodium and chloride ions cause premature leaf fall and/or death in willows (Iacobelli & Jefferies, 1991). Sedges, which are a highly diverse group of species, generally prefer wetter conditions, but salt-tolerance can be found in some northern clades (Hoffmann et al., 2017) and they are common in salt-marshes where water flow reduces actual soil salinity (Srivastava, 1993; Srivastava & Jefferies, 1995). This likely explains why they were abundant in Revegetated site types regardless of the salinity differences found between 1X & 2X affected Revegetated sites. These patterns are consistent with previously published data in this area that notes mosses as being highly susceptible to saline incursion and that graminoids are the first type of vegetation to re-establish in the affected area (Lantz et al., 2015).

Our analysis also showed that the average distance of sites from the nearest channel was correlated with the magnitude of vegetation recovery and average soil moisture. Flooding in the spring (freshwater) and late summer and fall (variable salinity) cause water levels to rise first in the channels which then overtake the banks to flood inland areas, so it is not surprising that low-lying areas closer to the channels would have higher soil moisture. These areas could recover more quickly from saline disturbance because they are more likely to get flooded by freshwater which could dilute or wash away some of the salts in the soil (Lantz et al., 2015). We could not assess this hypothesis within the scope of our study. Alternatively, it could be that the vegetation community present in areas that are more often affected by flooding (of both fresh and salt water) is composed of a higher quantity of fast-growing and/or salt-tolerant plants.

In our study it was challenging to separate the impacts of elevation and flood recurrence on the process of re-vegetation. Prior to this research, we thought that higher elevations would contribute to a lack of vegetation recovery because higher terrain would not be reached as often by freshwater flooding. However, we could not measure the frequency of freshwater flooding directly. Regardless, we found no significant difference in elevation between sites that remained Unvegetated and the 1X affected Revegetated sites. The two Unvegetated site types were both significantly higher than the unaffected Reference sites though. Furthermore, differences in vegetation communities at 1X affected versus 2X affected Revegetated sites could be due to elevation and frequency of freshwater flooding. However, it could also be due to the difference in time since the last saline event since 2X affected Revegetated sites were at a similar elevation as Reference sites, but 1X affected Revegetated and Reference sites are more similar in vegetation community.

It is likely that soil moisture, soil salinity levels, and frequency of disturbance/flooding at sites all played important roles in determining the presence or absence of certain species. Species that are tolerant of high salinity levels and drier soils were only found in the Unvegetated sites ; seaside arrowgrass is a species in the Juncaginaceae family that is frequently found in brackish environments and salt-marshes (Davy & Bishop, 1991; Khan & Ungar, 1999), whereas both nodding campion and spoonwort are found in a wide variety of substrates in the circumpolar arctic (dry and moist soils, as well as clay, gravel, sand, grassy, and rocky areas in a variety of elevations) but are known halophytes adapted to withstand physiological drought due to periods of high salinity (Grigore & Toma, 2021). Arctic fescue, which is a pioneering species in disturbed areas (Cody, 2000; Marteinsdóttir et al., 2013), is tolerant of short inundation but prefers drier soils (Casler & Duncan, 2003) and was found in all affected site types. Similarly, mare's tail was also found in all affected site types likely due to its ability to grow in a variety of aquatic environments including fresh water, intertidal, brackish, and saline (Aiken et al., 2007; Misson et al., 2016). Both green alder and common horsetail also prefer environments with consistent access to water (Aiken et al., 2007; Skoczowski et al., 2021). This may account for them only showing up in the vegetated sites in our study since these sites had higher soil moisture overall. Both Gmelin's (yellow) buttercup and white water buttercup are semi-aquatic species that prefer shallow ponds or margins of water (Aiken et al., 2007). White water buttercup is particularly noted as being somewhat salt tolerant and can be found at seashores or in salt marshes (Aiken et al., 2007). This tendency toward standing water and tolerance to salinity is likely why we only found them in the affected Revegetated sites.

The differences in vegetation species present at vegetated and unvegetated site types indicate that saline incursion can modify habitat for the long-term. The vegetation community present at Unvegetated sites is so distinct from that of the Reference and Revegetated sites that they seem to be on a different recovery trajectory at least ~20 years after the initial disturbance. Although very little plant cover was present at Unvegetated sites overall, the plants that had re-established did not resemble the communities of either the Revegetated or Reference sites. If these Unvegetated sites continue to support the species that are present, this could shift these areas into a different stable ecological state. Our observation that the 2X affected Unvegetated sites had more vegetation cover than the 1X affected Unvegetated sites suggests that an Unvegetated area that is subsequently inundated by a less severe storm or that has the potential to be inundated more frequently revegetates more quickly, but this was beyond the scope of our study. On the other hand, re-inundation at Revegetated sites reduced the diversity of the plant community in our study (Table 2.4).

Our analysis also shows that revegetation following the 1999 storm has progressed in some parts of the affected area such that the vegetation communities that have re-established resemble those of the Reference sites at all levels looked at in this study: NDVI greenness, total cover at the plot scale, plant functional types, and species composition. These sites were the most similar not only in vegetation community composition, but also in all other environmental factors measured. Prior to our study, we were unsure whether Revegetated areas had recovered similar vegetation communities as unaffected Reference sites. We also had no information on the likely effects of recurrent disturbance from saline incursion on previously affected areas. Observations made in this study indicate that, although there are still differences between these site types in terms of specific species' occurrences, we can conclude that the Revegetated sites that were only affected by the 1999 storm (1X affected) have recovered vegetation community structure similar to undisturbed areas (Reference).

2.4.2 Significance

Our study shows that terrestrial salinization following the 1999 storm has altered the vegetation community composition in some areas of the delta. Remote sensing estimates from 2019 suggest that ~3345 ha (~13% of the originally affected terrestrial surface area) have not recovered (to threshold 0.25 NDVI). For the 2016 storm, remote sensing analysis estimates that an additional ~1028 ha (~17% of the area that de-vegetated in 2016) have not recovered. For the most part, areas that were re-affected in 2016 supported similar vegetation communities to the same site type (Revegetated/Unvegetated) that were only affected by the 1999 storm, but some effects on the plant species present and the variety of species were also observed. To our knowledge, this is the first study assessing the effects of recurrent events of saline incursion on vegetation community composition in this area and one of the few long-term assessments of recovery from the 1999 storm. In our study, repeated flooding disturbance from a secondary saline inundation event did not hinder revegetation of the affected areas, but it did affect the composition of the vegetation community, particularly by reducing the cover of shrubs and moss. This has implications for wildlife that use these plant functional types either for shelter or for food, such as birds and caribou. Unrecovered areas are also a concern since they have remained functionally unvegetated for ~20 years and show few signs of recovering to undisturbed conditions. With the projected increases in storm surge frequency, recurrent disturbance in the affected area of the Mackenzie Delta is likely (Atkinson, 2005; Sepp & Jaagus, 2011; Simmonds & Keay, 2009). This recurrent disturbance has the potential to make recovery difficult for vegetation types, such as shrubs and mosses, that are not tolerant to prolonged inundation or to salinity. If storms become more frequent, as is predicted, there may not be enough time for salts to leach from soils before another event re-deposits more salt into the soil or for longer-lived woody shrubs to re-establish and grow before they are subjected to inundation. It is still unclear what areas of the outer delta are more protected from inundation because this likely depends heavily on factors of the storm itself (such as: salinity, height of surge, and length of inundation – among others). Although areas that are both higher in elevation, further from channels, and further inland are likely to be flooded by surges less often than areas at lower elevations, closer to channels, and nearer to the coast, if they are subjected to saline incursion, they could have reduced ecological resilience to this type of disturbance and take longer to recover than more frequently flooded areas if they recovery at all. This study highlights the importance of considering cumulative effects of storm surge events along with broader scale climate change effects and changes such as shrubification, lengthening of the growing season, coastal erosion and permafrost degradation, and oil & gas development. Our results provide insight into how future changes to the flooding disturbance regime may affect vegetation community composition and contributes to improved predictions in the rate of recovery of vegetation communities from storm surge events in the outer Mackenzie Delta and beyond. This information can also be used to inform larger scale predictions of Arctic vegetation changes in the face of climate change and help to guide management decisions for future developments in the Arctic.

2.4.3 Limitations

Firstly, although we assessed the effects of the 1999 storm in terms of the vegetation communities resulting after disturbance, we do not have any record of the vegetation community composition pre-storm. So, although we can make inferences from the satellite record as to the state of the vegetation community pre-storm via the NDVI greenness and our Reference sites, we cannot control for the effect of the previous vegetation community on the vegetation that re-establishes post-storm. We hope to have sampled a variety of sites that were affected for this to be a negligible effect on our study, but future study should endeavour to sample as much of the affected areas as possible to capture the true variety of potential vegetation communities.

Additionally, since the delta is not widely monitored for minor or moderate flooding events, it is difficult to factor the frequency of regular inundation into our assessment on storm surge effects. It is likely that areas closer to channels with lower relative elevations will be subject to flooding (by both the freshet and storm surges) more often than only during major storm surges, but we did not assess this directly in our study. Daily water level data from Reindeer Channel (Ellice Island) that has been collected from September 1982 – present shows that water levels above 10m (gauge height above assumed datum) were only reached on 21 days in the 957 days (2.2%) in July when data from this gauge station is available (Environment Canada, 2023). Water levels above 10m are slightly more common in August (4.6%) and September (3.9%) over this same time period (Environment Canada, 2023). Regardless, since the average relative elevation of our site types did show any particular pattern between the 1X and 2X affected site types, we do not think that this factor played a large role in our study.

In order to assess how areas affected by the 1999 event that remained Unvegetated in 2015 were subsequently re-affected in 2016, we had to employ a different technique of selecting the affected areas from the 2016 storm than sites that had Revegetated prior to this storm. Instead of using the only loss of vegetation greenness in NDVI from before the storm year (2015) to after the storm year (2017), we also used a flooding layer file produced from satellite imagery of 2016 in addition to the other characteristics outlined for this site type in the Methods section. We could not only use the vegetation loss method for 2016 in this case because areas that remained entirely devoid of vegetation in 2015 from the 1999 storm that were re-affected in 2016 would not be captured by this technique. Because of this, the selection of affected areas from the 1999 storm and the Revegetated areas affected again in 2016 are areas that lost vegetation as a result of the saline inundation, whereas Unvegetated areas re-affected in 2016 are simply areas where saline inundation occurred but were not necessarily where considerable vegetation loss occurred. For our study, this was the only way to capture how the Unvegetated areas from the 1999 storm would react to re-inundation by the 2016 storm. This could mean that some of the areas we selected did not undergo as dramatic vegetation loss in 2016 as they did in 1999 but seeing as the 2016 storm was already less destructive than the 1999 storm, this did not concern us for the questions we were asking.

2.4.4 Further Study

Although we found clear patterns between plant types and specific physical and/or environmental factors among our site types, more research on vegetation community composition across a greater range of abiotic conditions would allow more robust application of these findings to other areas in the Arctic and greater confidence when integrated into modelling predictions. To run a correlation analysis or model vegetation against abiotic conditions, future sampling needs to capture sufficient variation in the independent variable in order to compare the variables along a continuous scale. Because of my sampling method, I could not compare variables directly in a model but used a categorical analysis (grouped by independent variable).

The effects of annual freshwater flooding from the spring freshet were also not considered in this study. Spring flooding may contribute to vegetation loss in frequently flooded areas (Cordes et al., 1984; Pearce, 1986). In terms of recovery from saline inundation, it is not known whether "flushing" of affected areas by freshwater could assist in removing salt from soils and therefore aid in the re-establishment of salt-limited vegetation. There is also not much information available about how salinity concentrations differ among storm surge events and how deposition of salt in the soil could change depending on storm timing. Local observers of the 1999 storm have suggested that the late (September) timing of the storm was a primary reason for the severity of the salt-burn due to the persistence of salt in the soil when the ground froze for the winter (Kokelj et al., 2012). Future studies could install hourly salinity monitors and/or use the satellite archive to compare whether the vegetation communities at affected sites differ when they are flooded by freshwater at varying intervals, and whether the timing of a specific event (e.g., early vs. late in the year) similarly effects revegetation. This would aid in our understanding of vegetation recovery post-surge and would improve our predictive capabilities in determining ecological effects of future storms.

In the current literature, most storm surge events in the Mackenzie Delta have only been studied by chance (Scharffenberg, MacPhee, et al., 2020), or many years after the original event (Deasley et al., 2012; Kokelj et al., 2012; Lantz et al., 2015; Pisaric et al., 2011; Thienpont et al., 2012, 2015; Vermaire et al., 2013). While we have many descriptions of where certain vegetation types ought to be found in relation to environmental and physical factors, there are not many quantitative studies describing vegetation communities in the context of post-surge recovery and recurrent disturbance. Improved monitoring in the delta could aid in determining the factors influencing the magnitude of habitat salinization after storm surges, determine the frequency of storms that cause saline incursion, and determine the rate (and type) of recovery of vegetation post-surge.

2.5 Conclusion

This study provides evidence that vegetation changes caused by the 1999 storm persist today (~20 years later), but that some areas have recovered vegetation and soil properties similar to undisturbed areas. Repeated saline incursion (15 years between surges: 1999 & 2016) has the potential to limit the diversity of plant types in revegetated areas, at least in the shorter term (< 5 years post-storm). We identified soil salinity and distance from the nearest channel as important factors contributing to the composition of the vegetation community and presence/absence of specific plant species in our sites. How vegetation communities respond to increases in the frequency of saline flooding will depend on the interactions between many factors including – but not limited to – distance to nearest channel, elevation, soil moisture, and the pre-disturbance community. The composition of the vegetation communities in the outer delta affects many wildlife species (Terenzi et al., 2014; Chapter 3 of this thesis), which also affects local communities by impacting access to game species at traditional hunting grounds and complicates channel navigation by boat. Understanding how repeated large-scale salinization events will affect vegetation communities in previously affected areas is important for current and future monitoring of Arctic vegetation trends, especially in the context of changing disturbance regimes and ongoing climate change.

2.6 References

Aiken, S. G., Dallwitz, M.J., Consaul, L.L., McJannet, C.L., Boles, R.L., Argus, G.W., Gillett, J.M., Scott, P.J., Elven, R., LeBlanc, M.C., Gillespie, L.J., Brysting, A.K., Solstad, H., & Harris, J.G. (2007). Flora of the Canadian Arctic Archipelago: Descriptions, Illustrations, Identification, and Information Retrieval. NRC Research Press.

Allakhverdiev, S. I., Sakamoto, A., Nishiyama, Y., Inaba, M., & Murata, N. (2000). Ionic and Osmotic Effects of NaCl-Induced Inactivation of Photosystems I and II in Synechococcus sp.1. Plant Physiology, 123(3), 1047–1056. https://doi.org/10.1104/pp.123.3.1047

Arp, C. D., Jones, B. M., Schmutz, J. A., Urban, F. E., & Jorgenson, M. T. (2010). Two mechanisms of aquatic and terrestrial habitat change along an Alaskan Arctic coastline. Polar Biology, 33(12), 1629–1640. https://doi.org/10.1007/s00300-010-0800-5

Ashenhurst, A. R., & Hannon, S. J. (2008). Effects of seismic lines on the abundance of breeding Birds in the Kendall Island Bird Sanctuary, Northwest Territories, Canada. Arctic, 61(2), 190-. Gale OneFile: CPI.Q.

Atkinson, D. E. (2005). Observed storminess patterns and trends in the circum-Arctic coastal regime. Geo-Marine Letters, 25(2), 98–109. https://doi.org/10.1007/s00367-004-0191-0

Blok, D., Schaepman-Strub, G., Bartholomeus, H., Heijmans, M. M. P. D., Maximov, T. C., & Berendse, F. (2011). The response of Arctic vegetation to the summer climate: Relation between shrub cover, NDVI, surface albedo and temperature. Environmental Research Letters, 6(3), 035502. https://doi.org/10.1088/1748-9326/6/3/035502

Bromley, B., Fehr, A., & Black, S. (2002). Birds. In Natural history of the Western Arctic (pp. 45–54). Western Arctic Handbook Project.

Burn, C. R. (2017). Mackenzie Delta: Canada’s Principal Arctic Delta. In O. Slaymaker (Ed.), Landscapes and Landforms of Western Canada (pp. 321–334). Springer International Publishing. https://doi.org/10.1007/978-3-319-44595-3_23

Burn, C. R., & Kokelj, S. V. (2009). The environment and permafrost of the Mackenzie Delta area. Permafrost and Periglacial Processes, 20(2), 83–105. https://doi.org/10.1002/ppp.655

Byrnes, J. E., Reed, D. C., Cardinale, B. J., Cavanaugh, K. C., Holbrook, S. J., & Schmitt, R. J. (2011). Climate-driven increases in storm frequency simplify kelp forest food webs. Global Change Biology, 17(8), 2513–2524. https://doi.org/10.1111/j.1365-2486.2011.02409.x

Canadian Wildlife Service. (1992). Management of migratory bird sanctuaries in the Inuvialuit Settlement Region (p. 26). Environment Canada.

Carmack, E. C., & Macdonald, R. W. (2002). Oceanography of the Canadian Shelf of the Beaufort Sea: A Setting for Marine Life. Arctic, 55, 29–45.

Casler, M. D., & Duncan, R. R. (2003). Turfgrass Biology, Genetics, and Breeding. John Wiley & Sons.

Christiansen, C. T., Lafreniére, M. J., Henry, G. H. R., & Grogan, P. (2018). Long-term deepened snow promotes tundra evergreen shrub growth and summertime ecosystem net CO2 gain but reduces soil carbon and nutrient pools. Global Change Biology, 24(8), 3508–3525. https://doi.org/10.1111/gcb.14084

Cody, W. J. (2000). Flora of the Yukon Territory. NRC Research Press.

Collins, M., Knutti, R., Arblaster, J., Dufresne, J.-L., Fichefet, T., P. Friedlingstein, X. Gao, W.J. Gutowski, & et al. (2013). Long-term climate change: Projections, commitments and irreversibility (Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change). Cambridge University Press.

Collins, S. L. (2001). Long-Term Research and the Dynamics of Bird Populations and Communities. The Auk, 118(3), 583–588. https://doi.org/10.2307/4089922

Comiso, J. C., Parkinson, C. L., Gersten, R., & Stock, L. (2008). Accelerated decline in the Arctic sea ice cover. Geophysical Research Letters, 35(1). https://doi.org/10.1029/2007GL031972

Cordes, L. D., McLennan, D. S., & Pearce, C. M. (1984). Alluvial ecosystems on the Mackenzie Delta, N.W.T (Environment and Socio-Economic Services). BC Hydro. https://search.library.uvic.ca

Ćosić, M., Vujicic, M., Sabovljevic, M., & Sabovljevic, A. (2020). Effects of salt on selected bryophyte species tested under controlled conditions. Botanica Serbica, 44(1), 27–35. https://doi.org/10.2298/BOTSERB2001027C

Curtis, J. R., Robinson, W. D., & McCune, B. (2016). Time trumps habitat in the dynamics of an avian community. Ecosphere, 7(11), e01575. https://doi.org/10.1002/ecs2.1575

Davy, A. J., & Bishop, G. F. (1991). Triglochin Maritima L. Journal of Ecology, 79(2), 531–555. https://doi.org/10.2307/2260731

Davy, R., Chen, L., & Hanna, E. (2018). Arctic amplification metrics. International Journal of Climatology, 38(12), 4384–4394. https://doi.org/10.1002/joc.5675

Deasley, K., Korosi, J. B., Thienpont, J. R., Kokelj, S. V., Pisaric, M. F. J., & Smol, J. P. (2012). Investigating the response of Cladocera to a major saltwater intrusion event in an Arctic lake from the outer Mackenzie Delta (NT, Canada). Journal of Paleolimnology, 48(2), 287–296. https://doi.org/10.1007/s10933-012-9577-6

Drake, T. W., Holmes, R. M., Zhulidov, A. V., Gurtovaya, T., Raymond, P. A., McClelland, J. W., & Spencer, R. G. M. (2019). Multidecadal climate-induced changes in Arctic tundra lake geochemistry and geomorphology. Limnology and Oceanography, 64(S1), S179–S191. https://doi.org/10.1002/lno.11015

Du, Y., Zhang, Y., Ling, F., Wang, Q., Li, W., & Li, X. (2016). Water Bodies' Mapping from Sentinel-2 Imagery with Modified Normalized Difference Water Index at 10-m Spatial Resolution Produced by Sharpening the SWIR Band. Remote Sensing, 8(4), Article 4. https://doi.org/10.3390/rs8040354

Elmendorf, S. C., Henry, G. H. R., Hollister, R. D., Björk, R. G., Boulanger-Lapointe, N., Cooper, E. J., Cornelissen, J. H. C., Day, T. A., Dorrepaal, E., Elumeeva, T. G., Gill, M., Gould, W. A., Harte, J., Hik, D. S., Hofgaard, A., Johnson, D. R., Johnstone, J. F., Jónsdóttir, I. S., Jorgenson, J. C., … Wipf, S. (2012). Plot-scale evidence of tundra vegetation change and links to recent summer warming. Nature Climate Change, 2(6), Article 6. https://doi.org/10.1038/nclimate1465

Environment and Climate Change Canada. (2019, November 14). Kendall Island Migratory Bird Sanctuary [Program descriptions;education and awareness]. Kendall Island Migratory Bird Sanctuary. https://www.canada.ca/en/environment-climate-change/services/migratory-bird-sanctuaries/locations/kendall-island.html

Environment Canada. (2023). Daily Water Level Graph for MACKENZIE RIVER (REINDEER CHANNEL) AT ELLICE ISLAND (10MC011) [NT]. Water Level and Flow. https://wateroffice.ec.gc.ca/report/historical_e.html?start_year=2016&end_year=2016&y1Max=1&y1Min=1&mean1=1&median1=1&upper1=1&lower1=1&scale=normal&mode=Graph&stn=10MC011&dataType=Daily&parameterType=Level&year=2016

Forbes, D. L., & Frobel, D. (1985). Coastal erosion and sedimentation in the Canadian Beaufort Sea (Paper 85-1B; Current Research, Part B, pp. 69–80). Geological Survey of Canada. https://geoscan.nrcan.gc.ca/starweb/geoscan/servlet.starweb?path=geoscan/fulle.web&search1=R=120220

Gornish, E. S., & Miller, T. E. (2010). Effects of storm frequency on dune vegetation. Global Change Biology, 16(10), 2668–2675. https://doi.org/10.1111/j.1365-2486.2009.02144.x

Gratto-Trevor, C. L. (1996). Use of Landsat TM Imagery in Determining Important Shorebird Habitat in the Outer Mackenzie Delta, Northwest Territories. ARCTIC, 49(1), 11–22. https://doi.org/10.14430/arctic1179

Graversen, R. G., Mauritsen, T., Tjernström, M., Källén, E., & Svensson, G. (2008). Vertical structure of recent Arctic warming. Nature, 451(7174), Article 7174. https://doi.org/10.1038/nature06502

Grigore, M.-N., & Toma, C. (2021). Morphological and Anatomical Adaptations of Halophytes: A Review. In M.-N. Grigore (Ed.), Handbook of Halophytes: From Molecules to Ecosystems towards Biosaline Agriculture (pp. 1079–1221). Springer International Publishing. https://doi.org/10.1007/978-3-030-57635-6_37

Hanes, C. C., Wang, X., Jain, P., Parisien, M.-A., Little, J. M., & Flannigan, M. D. (2019). Fire-regime changes in Canada over the last half century. Canadian Journal of Forest Research, 49(3), 256–269. https://doi.org/10.1139/cjfr-2018-0293

Harper, J. R., Henry, R. F., & Stewart, G. G. (1988). Maximum Storm Surge Elevations in the Tuktoyaktuk Region of the Canadian Beaufort Sea. Arctic, 41(1), 48–52.

Hoffmann, M. H., Gebauer, S., & von Rozycki, T. (2017). Assembly of the Arctic flora: Highly parallel and recurrent patterns in sedges (Carex). American Journal of Botany, 104(9), 1334–1343. https://doi.org/10.3732/ajb.1700133

Howes, N. C., FitzGerald, D. M., Hughes, Z. J., Georgiou, I. Y., Kulp, M. A., Miner, M. D., Smith, J. M., & Barras, J. A. (2010). Hurricane-induced failure of low salinity wetlands. Proceedings of the National Academy of Sciences, 107(32), 14014–14019. https://doi.org/10.1073/pnas.0914582107

Iacobelli, A., & Jefferies, R. L. (1991). Inverse Salinity Gradients in Coastal Marshes and the Death of Stands of Salix: The Effects of Grubbing by Geese. Journal of Ecology, 79(1), 61–73. https://doi.org/10.2307/2260784

Johannessen, O. M., Kuzmina, S. I., Bobylev, L. P., & Miles, M. W. (2016). Surface air temperature variability and trends in the Arctic: New amplification assessment and regionalisation. Tellus A: Dynamic Meteorology and Oceanography, 68(1), 28234. https://doi.org/10.3402/tellusa.v68.28234

Johnstone, J. F., & Kokelj, S. V. (2008). Environmental Conditions and Vegetation Recovery at Abandoned Drilling Mud Sumps in the Mackenzie Delta Region, Northwest Territories, Canada. Arctic, 61(2), 199–211.

Jorgenson, M. T. (2000). Hierarchical Organization of Ecosystems at Multiple Spatial Scales on the Yukon-Kuskokwim Delta, Alaska, U.S.A. Arctic, Antarctic, and Alpine Research, 32(3), 221–239. https://doi.org/10.2307/1552521

Jorgenson, T., & Ely, C. (2001). Topography and flooding of coastal ecosystems on the Yukon-Kuskokwim delta, Alaska: Implications for sea-level rise. Journal of Coastal Research, 17, 124–136.

Khan, M. A., & Ungar, I. A. (1999). Effect of Salinity on Seed Germination of Triglochin Maritima Under Various Temperature Regimes. The Great Basin Naturalist, 59(2), 144–150.

Kim, J., Murphy, E., Nistor, I., Ferguson, S., & Provan, M. (2021). Numerical Analysis of Storm Surges on Canada’s Western Arctic Coastline. Journal of Marine Science and Engineering, 9(3), Article 3. https://doi.org/10.3390/jmse9030326

Knutson, T. R., McBride, J. L., Chan, J., Emanuel, K., Holland, G., Landsea, C., Held, I., Kossin, J. P., Srivastava, A. K., & Sugi, M. (2010). Tropical cyclones and climate change. Nature Geoscience, 3(3), Article 3. https://doi.org/10.1038/ngeo779

Kokelj, S. V., Lantz, T. C., Solomon, S., Pisaric, M. F. J., Keith, D., Morse, P., Thienpont, J. R., Smol, J. P., & Esagok, D. (2012). Using Multiple Sources of Knowledge to Investigate Northern Environmental Change: Regional Ecological Impacts of a Storm Surge in the Outer Mackenzie Delta, N.W.T. Arctic; Calgary, 65(3), 257–272.

Lantz, T. C. (2016). 2016 Fieldwork. Outer Mackenzie Delta.

Lantz, T. C., Kokelj, S. V., & Fraser, R. H. (2015). Ecological recovery in an Arctic delta following widespread saline incursion. Ecological Applications, 25(1), 172–185. https://doi.org/10.1890/14-0239.1

Lantz, T. C., Marsh, P., & Kokelj, S. V. (2013). Recent Shrub Proliferation in the Mackenzie Delta Uplands and Microclimatic Implications. Ecosystems, 16(1), 47–59. https://doi.org/10.1007/s10021-012-9595-2

Latour, P. B., Machtans, C. S., & Beyersbergen, G. W. (2005). Shorebird and Passerine Abundance and Habitat Use at a High Arctic Breeding Site: Creswell Bay, Nunavut. ARCTIC, 58(1), 55–65. https://doi.org/10.14430/arctic389

Lin, H.-J., Shao, K.-T., Chiou, W.-L., Maa, C.-J. W., Hsieh, H.-L., Wu, W.-L., Severinghaus, L. L., & Wang, Y.-T. (2003). Biotic communities of freshwater marshes and mangroves in relation to saltwater incursions: Implications for wetland regulation. Biodiversity & Conservation, 12(4), 647–665. https://doi.org/10.1023/A:1022405213732

Lynch, A. H., Curry, J. A., Brunner, R. D., & Maslanik, J. A. (2004). Toward an Integrated Assessment of the Impacts of Extreme Wind Events on Barrow, Alaska. Bulletin of the American Meteorological Society, 85(2), 209–222. https://doi.org/10.1175/BAMS-85-2-209

Manson, G. K., & Solomon, S. M. (2007). Past and future forcing of Beaufort Sea coastal change. Atmosphere-Ocean, 45(2), 107–122. https://doi.org/10.3137/ao.450204

Manson, G. K., Solomon, S. M., Forbes, D. L., Atkinson, D. E., & Craymer, M. (2005). Spatial variability of factors influencing coastal change in the Western Canadian Arctic. Geo-Marine Letters, 25(2), 138–145. https://doi.org/10.1007/s00367-004-0195-9

Marteinsdóttir, B., Thórhallsdóttir, T. E., & Svavarsdóttir, K. (2013). An experimental test of the relationship between small scale topography and seedling establishment in primary succession. Plant Ecology, 214(8), 1007–1015. https://doi.org/10.1007/s11258-013-0226-6

McGarigal, K., Stafford, S., & Cushman, S. (2000). Multivariate Statistics for Wildlife and Ecology Research. Springer. https://doi.org/10.1007/978-1-4612-1288-1

McKeague, J. A. (1978). Manual on soil sampling and methods of analysis. Canadian Society of Soil Science.

Misson, G., Macor, A., Boscutti, F., & Casolo, V. (2016). Ecological characterisation of Hippuris vulgaris populations growing in spring water rivers. Phyton (Horn), 56(2), 209–224.

Moffat, N. D., Lantz, T. C., Fraser, R. H., & Olthof, I. (2016). Recent Vegetation Change (1980–2013) in the Tundra Ecosystems of the Tuktoyaktuk Coastlands, NWT, Canada. Arctic, Antarctic, and Alpine Research, 48(3), 581–597. https://doi.org/10.1657/AAAR0015-063

Nagy, J. (2002). Mammals. In Black, S. & Fehr, A. (Eds.), Natural history of the Western Arctic.

NWT Centre for Geomatics. (2007). Mackenzie Valley Air Photo Project 2004 [Map]. https://www.imagetest.geomatics.gov.nt.ca/arcgis/rest/services/Mosaics/MVAP_Mosaic_Combined_LCC/MapServer/info/iteminfo

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P. R., O'hara, R. B., Simpson, G. L., & Solymos, P. (2015). Vegan: Community ecology package. 2019. R Package Version, 2(10).

Overeem, I., Anderson, R. S., Wobus, C. W., Clow, G. D., Urban, F. E., & Matell, N. (2011). Sea ice loss enhances wave action at the Arctic coast. Geophysical Research Letters, 38(17). https://doi.org/10.1029/2011GL048681

Parida, A. K., & Das, A. B. (2005). Salt tolerance and salinity effects on plants: A review. Ecotoxicology and Environmental Safety, 60(3), 324–349. https://doi.org/10.1016/j.ecoenv.2004.06.010

Pearce, C. M. (1986). The Distribution and ecology of the shoreline vegetation on the Mackenzie Delta, N.W.T. [Doctoral thesis, University of Calgary]. https://doi.org/10.11575/PRISM/18346

Pisaric, M. F. J., Thienpont, J. R., Kokelj, S. V., Nesbitt, H., Lantz, T. C., Solomon, S., & Smol, J. P. (2011). Impacts of a recent storm surge on an Arctic delta ecosystem examined in the context of the last millennium. Proceedings of the National Academy of Sciences of the United States of America, 108(22), 8960–8965. https://doi.org/10.1073/pnas.1018527108

Post, E., Forchhammer, M. C., Bret-Harte, M. S., Callaghan, T. V., Christensen, T. R., Elberling, B., Fox, A. D., Glig, O., Hik, D. S., Høye, T. T., Ims, R. A., Jeppesen, E., Klein, D. R., Madsen, J., McGuire, A. D., Rysgaard, S., Schindler, D. E., Stirling, I., Tamstorf, M. P., … Aastrup, P. (2009). Ecological Dynamics across the Arctic Associated with Recent Climate Change. Science, 325(5946), 1355–1358. JSTOR.

Proverbs, T. A., & Lantz, T. C. (2020). Cumulative Environmental Impacts in the Gwich'in Cultural Landscape. Sustainability, 12(11), Article 11. https://doi.org/10.3390/su12114667

R Core Team. (2020). R: a language and environment for statistical computing. Foundation for Statistical Computing.

Raynolds, M. K., Walker, D. A., Epstein, H. E., Pinzon, J. E., & Tucker, C. J. (2012). A new estimate of tundra-biome phytomass from trans-Arctic field data and AVHRR NDVI. Remote Sensing Letters, 3(5), 403–411. https://doi.org/10.1080/01431161.2011.609188

Reimnitz, E., & Maurer, D. K. (1979). Effects of Storm Surges on the Beaufort Sea Coast, Northern Alaska. Arctic, 32(4), 329–344.

Riedel, S. M., Epstein, H. E., & Walker, D. A. (2005). Biotic controls over spectral reflectance of arctic tundra vegetation. International Journal of Remote Sensing, 26(11), 2391–2405. https://doi.org/10.1080/01431160512331337754

Ropars, P., & Boudreau, S. (2012). Shrub expansion at the forest–tundra ecotone: Spatial heterogeneity linked to local topography. Environmental Research Letters, 7(1), 015501. https://doi.org/10.1088/1748-9326/7/1/015501

Scavia, D., Field, J. C., Boesch, D. F., Buddemeier, R. W., Burkett, V., Cayan, D. R., Fogarty, M., Harwell, M. A., Howarth, R. W., Mason, C., Reed, D. J., Royer, T. C., Sallenger, A. H., & Titus, J.

G. (2002). Climate change impacts on U.S. Coastal and Marine Ecosystems. Estuaries, 25(2), 149–164. https://doi.org/10.1007/BF02691304

Scharffenberg, K. C., MacPhee, S. A., & Loseto, L. L. (2020). Upriver sightings of beluga whales (Delphinapterus leucas) follow storm surges and high water in the Mackenzie Delta, Northwest Territories, Canada. Arctic Science, 1–11. https://doi.org/10.1139/as-2020-0010

Scharffenberg, K. C., Whalen, D., MacPhee, S. A., Marcoux, M., Iacozza, J., Davoren, G., & Loseto, L. L. (2020). Oceanographic, ecological, and socio-economic impacts of an unusual summer storm in the Mackenzie Estuary. Arctic Science, 6(2), 62–76. https://doi.org/10.1139/as-2018-0029

Sepp, M., & Jaagus, J. (2011). Changes in the activity and tracks of Arctic cyclones. Climatic Change, 105(3), 577–595. https://doi.org/10.1007/s10584-010-9893-7

Serreze, M. C., & Barry, R. G. (2011). Processes and impacts of Arctic amplification: A research synthesis. Global and Planetary Change, 77(1), 85–96. https://doi.org/10.1016/j.gloplacha.2011.03.004

Serreze, M. C., Holland, M. M., & Stroeve, J. (2007). Perspectives on the Arctic's Shrinking Sea-Ice Cover. Science, 315(5818), 1533–1536. JSTOR.

Simmonds, I., & Keay, K. (2009). Extraordinary September Arctic sea ice reductions and their relationships with storm behavior over 1979–2008. Geophysical Research Letters, 36(19). https://doi.org/10.1029/2009GL039810

Skoczowski, A., Odrzywolska-Hasiec, M., Oliwa, J., Ciereszko, I., & Kornaś, A. (2021). Ecophysiological Variability of Alnus viridis (Chaix) DC. Green Alder Leaves in the Bieszczady Mountains (Poland). Plants, 10(1), Article 1. https://doi.org/10.3390/plants10010096

Smith, P. A., McKinnon, L., Meltofte, H., Lanctot, R. B., Fox, A. D., Leafloor, J. O., Soloviev, M., Franke, A., Falk, K., Golovatin, M., Sokolov, V., Sokolov, A., & Smith, A. C. (2020). Status and trends of tundra birds across the circumpolar Arctic. Ambio, 49(3), 732–748. https://doi.org/10.1007/s13280-019-01308-5

Solomon, S. (2005). Spatial and temporal variability of shoreline change in the Beaufort-Mackenzie region, northwest territories, Canada. Geo-Marine Letters, 25(2), 127–137. https://doi.org/10.1007/s00367-004-0194-x

Solomon, S., Forbes, D.L., & Kierstead, Brian. (1994). Coastal impacts of climate change: Beaufort Sea erosion study (Geological Survey of Canada). Natural Resources Canada. https://publications.gc.ca/site/eng/9.820125/publication.html

Srivastava, D. S. (1993). The role of lesser snow geese in positive, degenerative feedback processes resulting in the destruction of salt-marsh swards. https://yorkspace.library.yorku.ca/xmlui/handle/10315/18774

Srivastava, D. S., & Jefferies, R. L. (1995). The Effect of Salinity on the Leaf and Shoot Demography of Two Arctic Forage Species. Journal of Ecology, 83(3), 421–430. https://doi.org/10.2307/2261595

Stephenson, S. A. (2002). Freshwater fishes. In Black, S. & Fehr, A. (Eds.), Natural history of the Western Arctic (pp. 70–76).

Tate, A. S., & Battaglia, L. L. (2013). Community disassembly and reassembly following experimental storm surge and wrack application. Journal of Vegetation Science, 24(1), 46–57. https://doi.org/10.1111/j.1654-1103.2012.01447.x

Taylor, R. J. (1981). Shoreline Vegetation of the Arctic Alaska Coast. Arctic, 34(1), 37–42.

Terenzi, J., Jorgenson, M. T., & Ely, C. R. (2014). Storm-Surge Flooding on the Yukon-Kuskokwim Delta, Alaska. Arctic; Calgary, 67(3), 360–374.

Thienpont, J. R., C., S., Vermaire, J., Pisaric, M., Kokelj, S., & Smol, J. (2015). Synchronous changes in chironomid assemblages following a major saltwater intrusion in two coastal Arctic lake ecosystems. Journal of Paleolimnology, 53, 177–189.

Thienpont, J. R., Johnson, D., Nesbitt, H., Kokelj, S. V., Pisaric, M. F., & Smol, J. P. (2012). Arctic coastal freshwater ecosystem responses to a major saltwater intrusion: A landscape-scale palaeolimnological analysis. The Holocene, 22(12), 1451–1460. https://doi.org/10.1177/0959683612455538

Thompson, A., & Millar, N. (2007). Traditional knowledge of fish migration and spawning patterns in Tsiigehnjik (Arctic Red River) and Nagwichoonjik (Mackenzie River), Northwest Territories (No. 07–01). Gwich'in Renewable Resource Board.

Torresan, S., Critto, A., Rizzi, J., & Marcomini, A. (2012). Assessment of coastal vulnerability to climate change hazards at the regional scale: The case study of the North Adriatic Sea. Natural Hazards and Earth System Sciences, 12(7), 2347–2368. https://doi.org/10.5194/nhess-12-2347-2012

Travers-Smith, H. Z., & Lantz, T. C. (2020). Leading-edge disequilibrium in alder and spruce populations across the forest–tundra ecotone. Ecosphere, 11(7), e03118. https://doi.org/10.1002/ecs2.3118

Tucker, C. J. (1979). Red and photographic infrared linear combinations for monitoring vegetation. Remote Sensing of Environment, 8(2), 127–150. https://doi.org/10.1016/0034-4257(79)90013-0

Tyson, W., Lantz, T. C., & Ban, N. C. (2016). Cumulative effects of environmental change on culturally significant ecosystems in the inuvialuit settlement region. Arctic, 69(4), 391-. Gale OneFile: CPI.Q.

Usher, P. J. (2000). Traditional Ecological Knowledge in Environmental Assessment and Management. Arctic, 53(2), 183–193. JSTOR.

Vermaire, J. C., Pisaric, M. F. J., Thienpont, J. R., Mustaphi, C. J. C., Kokelj, S. V., & Smol, J. P. (2013). Arctic climate warming and sea ice declines lead to increased storm surge activity. Geophysical Research Letters, 40(7), 1386–1390. https://doi.org/10.1002/grl.50191

Wildlife Management Advisory Council (North Slope) & Aklavik Hunters and Trappers Committee. (2018). Yukon North Slope Inuvialuit Traditional Use Study. //www.wmacns.ca/resources/yukon-north-slope-inuvialuit-traditional-use-study/

Yang, D., Shi, X., & Marsh, P. (2015). Variability and extreme of Mackenzie River daily discharge during 1973–2011. Quaternary International, 380–381, 159–168. https://doi.org/10.1016/j.quaint.2014.09.023

3 Assessing ecological effects of storm surges on Arctic bird populations in the outer Mackenzie Delta, Northwest Territories

F. Nicola A. Shipman[1], Trevor Lantz[1], Louise K. Blight[1] [2]

1. School of Environmental Studies, University of Victoria

---

[2] This is the authorship planned for Arctic Science. Study Design: FNAS and TCL, Fieldwork and Analysis: FNAS, Writing: FNAS, Revisions and Editing: FNAS, TCL, and LKB.

3.1 Introduction

Arctic habitats are critical for many breeding bird populations, and climate-driven habitat alteration poses a significant threat to many species (Wauchope et al., 2017). Of the ~200 species of birds that are known to breed in the Arctic, 162 species are considered to have the majority of their breeding ranges there (Ganter & Gaston, A.J., 2013). For this reason, the Arctic Council's Circumpolar Biodiversity Monitoring Program (CBMP) has designated birds as "Focal Ecosystem Components" for monitoring under their Terrestrial monitoring program (Christensen et al., 2013). Of these 162 species with mainly Arctic breeding grounds, 88 are landbirds (Smith et al., 2020). Fifteen species of shorebirds (out of the 47 total Canadian species) breed exclusively in the Arctic (Donaldson et al., 2000) and several others include the Arctic within their breeding range. Though they are among the most numerous and species-rich taxa of Arctic waterbirds, many shorebird populations are declining (Clemens et al., 2016; Gratto-Trevor et al., 1998; Hitchcock & Gratto-Trevor, 1997; Jehl, 2007; Morrison et al., 2013; Piersma et al., 2016; Smith et al., 2023; Wauchope et al., 2017). Additionally, though passerines are ubiquitous and abundant in the tundra, recent trends show declines across-taxa in North America during the past decade or more (Rosenberg et al., 2019; Smith et al., 2020). Many studies list shifts in habitat characteristics and phenology due to climate change as primary factors in these declines (Baker et al., 2004; Duijns et al., 2017; Gaston et al., 2005; McKinnon et al., 2012; Robinson et al., 2005; Senner et al., 2014, 2017; Studds et al., 2017). Anthropogenic changes along long migratory routes that can stretch from the Arctic to southern South America may also be contributing (Reed et al., 2018; Studds et al., 2017).

Within the next 100 years, climate change is projected to be more rapid and uniform across the Arctic in comparison to historic warming events (Intergovernmental Panel on Climate Change (IPCC), 2013). This suggests the potential for more severe constriction of shorebird breeding distributions than in previous "bottle-necking" events of the mid-Holocene where extreme cold and arid conditions of Quaternary glaciations would have geographically restricted available breeding habitat (Buehler & Baker, 2005; Conklin et al., 2016; Wauchope et al., 2017; Wennerberg et al., 2002). A recent study assessing projected shifts in the spatial distribution of suitable habitat for Arctic-breeding shorebirds predicts considerable northward constriction at a pan-Arctic scale (Wauchope et al., 2017). These predictions specified the Canadian Arctic Archipelago as being a globally important refuge for shorebirds in the future, but also noted that these habitats currently lack government protection (Wauchope et al., 2017). Concurrent habitat shifts predicted at breeding sites include the advancement of southern predator ranges (e.g., red fox, *Vulpes vulpes*) and the expansion of potential interspecific competitors (Callaghan et al., 2004; Killengreen et al., 2007; Mustin et al., 2007; Post et al., 2009), both of which could compound the adverse effects changing shorebird habitat. In addition, some Arctic-breeding shorebird and landbird populations are already suffering from the accelerated advance of spring and increased unseasonal storms as the timing of spring arrival can greatly affect reproductive success and population dynamics (Both et al., 2010; Ganter & Gaston, A.J., 2013; Klaassen et al., 2012; Kölzsch et al., 2015; Møller et al., 2008; Saino et al., 2011; Smith et al., 2020; Tulp et al., 2009). In addition to weather-related issues caused by the advancement of spring, more complex interactions between ecosystems and their inhabitants are becoming apparent as species react to climate and weather pattern changes with varying strategies. More evidence with links to specific local effects and drivers is required to tease apart these interactions and predict future patterns in the face of ongoing climate change.

In the circumpolar Arctic, the density and diversity of birds respond to productivity gradients (Smith et al., 2020). Although heavily vegetated wetlands constitute a small portion of the available habitat, they support a disproportionate richness and abundance of bird species (Smith et al., 2020). Arctic river deltas are especially critical for migratory birds as stopover, breeding, and nesting grounds (Canadian Wildlife Service, 1992; Latour et al., 2005), but these low-lying coastal regions are particularly prone to flooding. Although freshwater flooding is a predictable annual event due to upriver snow melt in spring, saltwater flooding events can occur during severe marine weather events (MacKay, 1963; Manson & Solomon, 2007; Vermaire et al., 2013). Marine storms are expected to increase in frequency and intensity globally as a consequence of climate change (Sepp & Jaagus, 2011). Inuvialuit living in Ulukhaktok, Sachs Harbour, Tuktoyaktuk and other communities in the Beaufort Delta Region have already noted changes in weather patterns that include an increase in strong winds (Berkes & Jolly, 2001; Pearce et al., 2010; Waugh et al., 2018). Persistent strong winds in these low elevation areas can produce storm surges, which are defined as an abnormal rise in sea-level along a coastline during a storm (Harris, 1963). These surges can result in the saltwater flooding of riparian and terrestrial ecosystems in a process known as saline incursion, which can cause habitat degradation by killing vegetation that is not salt-tolerant and preventing re-establishment of plant species due to salt buildup in soils (Earle & Kershaw, 1989; Iacobelli & Jefferies, 1991). Previous studies in the Mackenzie Delta (Northwest Territories) show that saline incursion events can be severe enough to cause persistent ecological change (Pisaric et al., 2011; Thienpont et al., 2015; Vermaire et al., 2013). A decade after a storm surge in 1999, re-establishment of vegetation was underway, but some affected areas remained where little to no recolonization of plants was occurring (Lantz et al., 2015). However, due to a lack of historical monitoring of similar saline incursion events, approximate timelines for recovery of persistently barren areas remain unclear (Deasley et al., 2012; Lantz et al., 2015).

Historically, increased storm surge activity has followed trends of rising temperature and declining sea-ice (Serreze et al., 2007; Simmonds & Keay, 2009; Vermaire et al., 2013). The potential for strong winds to result in a surge is also compounded by physical and oceanographic factors such as coastal erosion and sea-level rise (Manson & Solomon, 2007; Nicholls & Cazenave, 2010; Thienpont et al., 2012). All four of these factors are projected to be adversely affected by climate change. With predictions of increasing potential for saline incursion in Arctic coastal areas, in combination with imminent contractions in predicted distributions of Arctic breeding birds, it will be critical to have a good understanding of potentially compounding emerging threats to terrestrial bird habitat in these areas. However, it is currently unknown how vegetation changes from saline incursion in the Arctic are affecting habitat use by bird species and whether affected sites can functionally recover. Since the 1999 storm, several studies have assessed the recovery of soils and vegetation, but to date, no formal investigation of the effects of saline incursion upon shorebirds or other avifauna has been conducted (Lantz et al., 2015; Pisaric et al., 2011; Thienpont et al., 2012, 2015; Vermaire et al., 2013). In this study, we used a natural experiment — namely, saltwater incursion in portions of the outer Mackenzie Delta — to assess the effects of habitat transformation caused by a severe storm surge on bird community composition and presence. We also conducted field surveys to explore the biophysical factors associated with differences in bird assemblages across the affected area.

3.2 Methods

3.2.1 Study Area

The Mackenzie Delta, North America's largest Arctic delta (~13,000 km^2), is located at the northern continental edge of the Northwest Territories at the outflow of the Mackenzie and Peel rivers into the Beaufort Sea (Burn, 2017; Burn & Kokelj, 2009). A low-lying alluvial plain, the delta has a maximum elevation of ~10m above sea-level and gradually slopes downward to the Beaufort Sea (Deasley et al., 2012; Vermaire et al., 2013). This delta lies within the traditional territory of both the Inuvialuit and Gwich'in peoples, who continue to rely on the land for subsistence (Thompson & Millar, 2007; Usher, 2000). This study focused on the low-lying (< 2 masl; Fig. 1) northern part of the delta (hereafter the "outer delta") (Lantz et al., 2015). Vegetation in the outer delta can be classified into three dominant vegetation types (graminoid wetland, dwarf shrub, and upright shrub) based on soil characteristics and frequency of flooding (Cordes et al., 1984; Lantz et al., 2015; Pearce, 1986). Terrain <30 cm above low tide level is generally unvegetated (Canadian Wildlife Service, 1992).

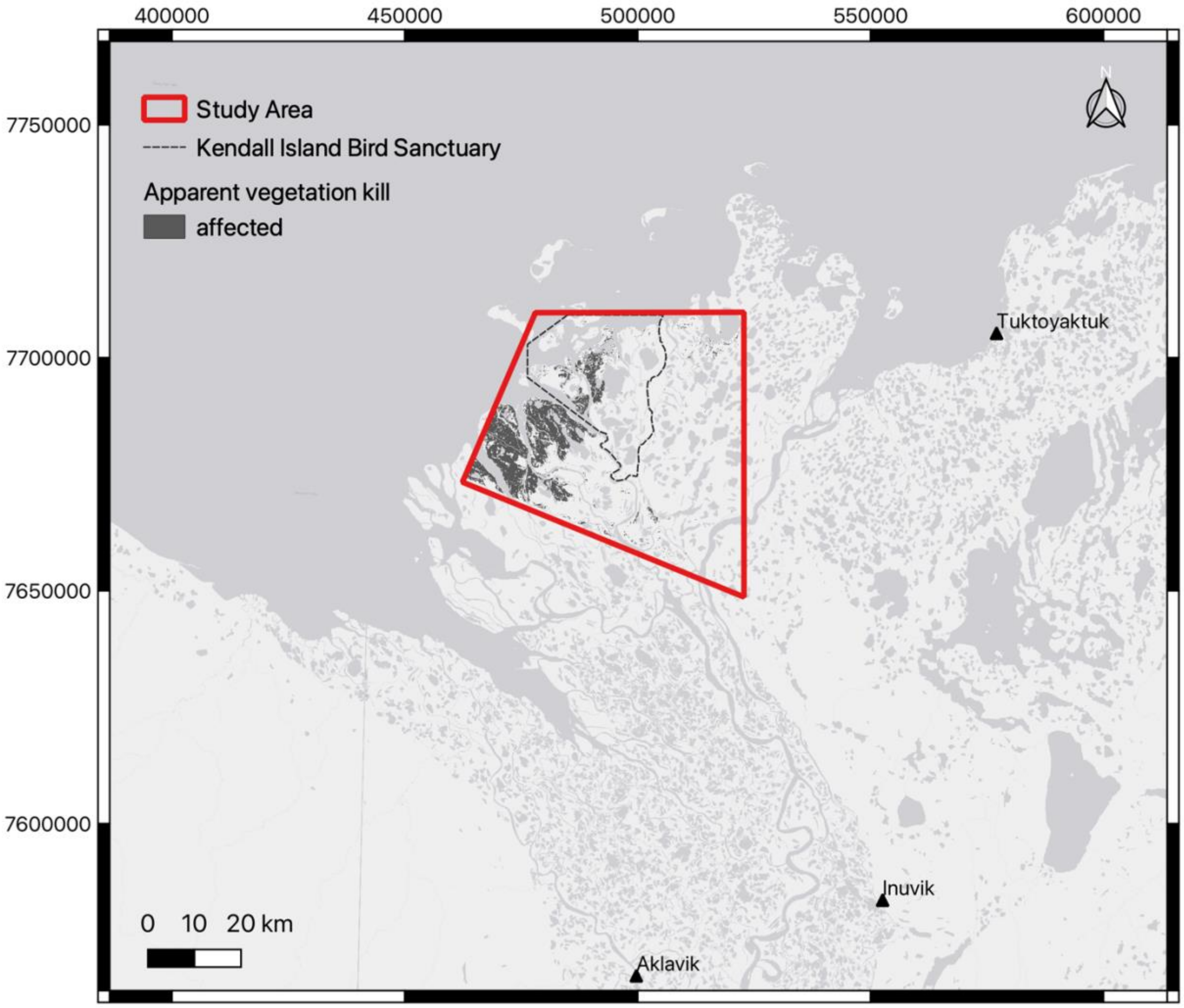


Figure 3.1. A map of our study area. It shows the extent of the Landsat imagery we assessed (outlined in red) within the Mackenzie Delta region, the 1999-affected area (in dark grey), and the boundaries of the Kendall Island Migratory Bird Sanctuary (outlined in the dash).

Within the Mackenzie Delta, the Kendall Island Migratory Bird Sanctuary encompasses approximately 606 km^2 of the outer delta (Bromley et al., 2002; Canadian Wildlife Service, 1992). Established in 1961, the purpose of this sanctuary was to protect the abundance of migratory birds (primarily waterbirds) in the area (Canadian Wildlife Service, 1992). Over 100 species of birds have been recorded in the sanctuary, including perching birds, shorebirds, and waterfowl (Canadian Wildlife Service, 1992; Environment and Climate Change Canada, 2019). Lesser snow geese (*Anser caerulescens*) use the area as a nesting and staging ground in the fall and constitute a large portion of the waterbird species (Canadian Wildlife Service, 1992). Notable species include hudsonian godwit and long-billed dowitcher, which have limited breeding grounds in Canada (Canadian Wildlife Service, 1992).

Gravitational tides in the outer delta are small (< 0.5m) in comparison to storm surge events, which can reach around 2.5 m above mean water level (Forbes & Frobel, 1985; Harper et al., 1988b; Solomon, 2005). Storm surges are a regular occurrence in the outer delta during the open-water months (June-October) but occur more frequently in late summer and into autumn (Atkinson, 2005; Kim et al., 2021; Solomon et al., 1994). Not all storm surges result in saline incursion, as freshwater river discharge and wind direction typically keep the saltwater from reaching upstream (Carmack & Macdonald, 2002). However, as river discharge decreases over the open-water season, the potential for storm surges to cause saline flooding inland increases (Scharffenberg, MacPhee, et al., 2020; Yang et al., 2015). This potential is increasing due to the extended open water season in the Arctic (Overeem et al., 2011; Vermaire et al., 2013).

A particularly anomalous surge occurred in 1999 and was retroactively documented (Kokelj et al., 2012; Lantz et al., 2015; Pisaric et al., 2011). Paleolimnological studies have shown that the magnitude of this storm surge was unprecedented in the last millennia (Pisaric et al., 2011; Vermaire et al., 2013). Previous studies assessing the ecological consequences post-1999 show that the incursion of saline water was severe enough to cause widespread and persistent (7-10 years post-storm) vegetation kill in ~120 km^2 of the outer delta (Pisaric et al., 2011; Thienpont et al., 2015; Vermaire et al., 2013). A decade after the storm surge in 1999, recovery was occurring slowly, but it was spatially variable with some areas showing no re-establishment of vegetation (Lantz et al., 2015). Though salt-tolerant species are present in the outer delta, previous surveys have not observed them colonizing affected areas, suggesting that this large-scale salinization event is unprecedented in recent history (Lantz et al., 2015; Pisaric et al., 2011).

3.2.2 Site Selection

To explore the impacts of storm-surge driven vegetation change on avian community composition and the abundance of three focal species, we conducted field studies within and outside areas affected by the 1999 surge. Specifically, we sampled areas of the outer Mackenzie Delta in three site types (Revegetated post-inundation, Unvegetated post-inundation, and unaffected Reference).

To delineate the area affected by the 1999 storm surge, we analyzed imagery from the LANDSAT 5, 7 & 8 satellites. Top of Atmosphere (TOA) imagery with less than 20% cloud cover obtained during the peak greening period (July 1 – August 15) was used to calculate the Normalized Difference Vegetation Index (NDVI) annually from 1986-2015. NDVI is calculated using the contrast between near-infrared and red reflectance (Tucker, 1979) and represents a measure of green leaf area and phytomass (Raynolds et al., 2012; Riedel et al., 2005). Using Landsat scenes from 1986-1994 we characterized inter-annual variation in NDVI before the 1999 storm and used it to establish a threshold of inter-annual variability that represented a significant departure from normal. This threshold was calculated by creating a raster stack of proportional inter-annual change from 1986-1994, where proportional inter-annual change was defined as: $[\mathrm{NDVI}_{\mathrm{YEAR}} - \mathrm{NDVI}_{\mathrm{PREVIOUSYEAR}}] / \mathrm{NDVI}_{\mathrm{PREVIOUSYEAR}}$. Using this stack, we calculated the average inter-annual pre-storm change on a per-pixel basis. This resulted in a single raster layer where pixels were the average inter-annual change in NDVI across all years. The mean (μ) and standard deviation (σ) of this layer were used to define the range of typical inter-annual variation as: $\mu \pm 2\sigma$, or $0.0355 \pm 2*(0.1968)$. Subsequently, the lower limit of this range was used as a threshold to delineate the extent of the area affected by the 1999 surge as $\mathrm{PNDVI}_{2001} - \mathrm{PNDVI}_{1998} < -0.358$, and unaffected as $\mathrm{PNDVI}_{2001} - \mathrm{PNDVI}_{1998} \geq -0.358$.

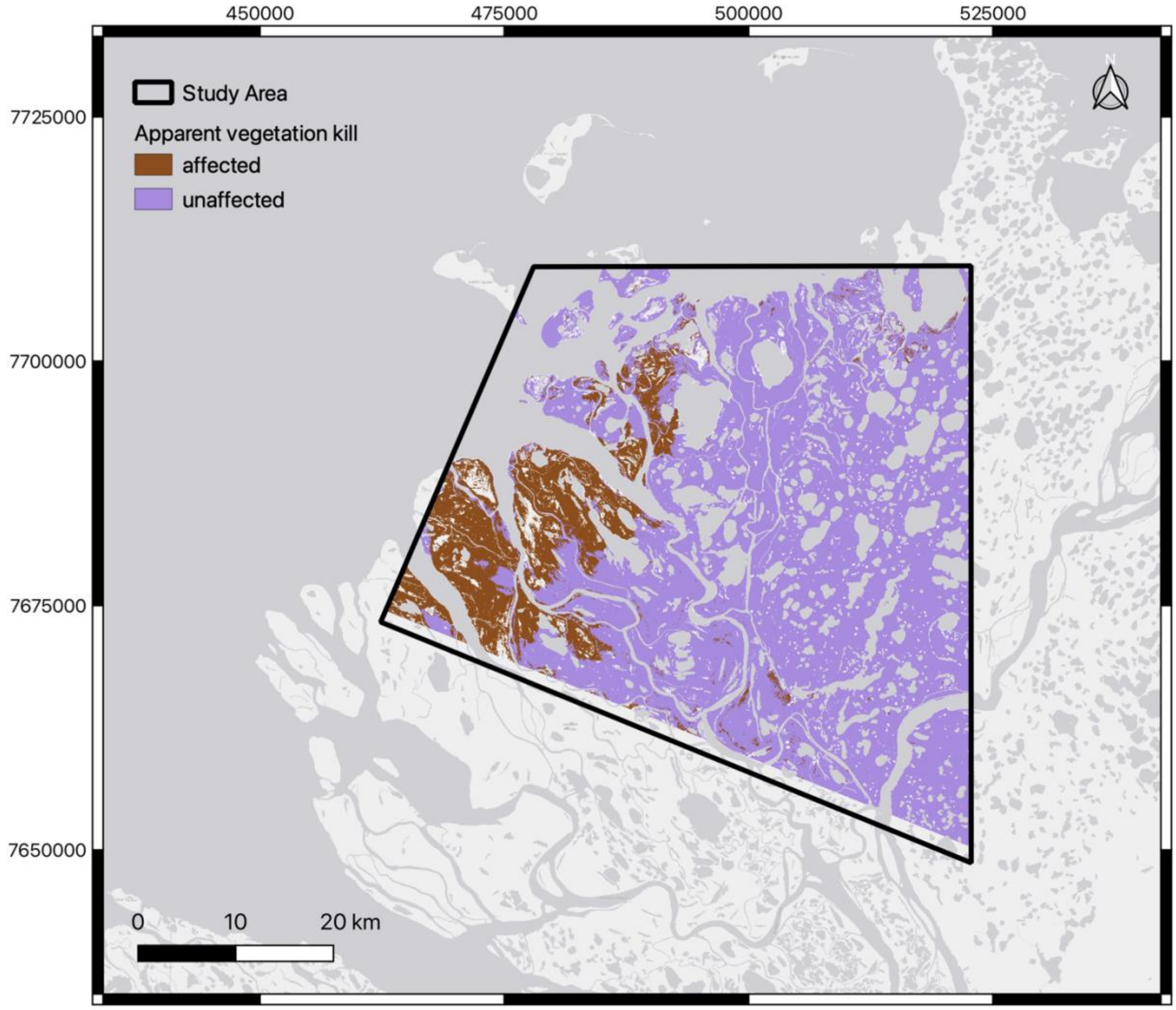


Figure 3.2. Delineation of the affected and unaffected areas post-storm as described by the proportional change in NDVI described above (affected = $PNDVI_{2001} - PNDVI_{1998} < -0.358$; unaffected = $PNDVI_{2001} - PNDVI_{1998} \geq -0.358$).

We selected re-vegetated and unvegetated sites within the affected portion of the outer delta using two methods that utilized satellite imagery and aerial photographs captured in 2004 (NWT Centre for Geomatics, 2007). In the first site selection method, high-resolution (effective pixel size =~1m; scale = 1:30000) air photos from 2004 (NWT Centre for Geomatics, 2007) were visually inspected. These air photos clearly show areas of bare ground and areas with green vegetation present. To verify that air photos could be used to classify sites as Revegetated or Unvegetated we extracted a subset of 2004 NDVI values from sites classified as vegetated (n=40) and unvegetated (n=44) using air photos (Table 3.1). This analysis indicates that affected sites undergoing revegetation had NDVI values greater than 0.25 and typically ranging from 0.303 to 0.713 (Table 3.1). NDVI values at Unvegetated sites had a maximum of 0.25 and generally ranged from 0.094 to 0.215 (Table 3.1). Based on these differences, we used 0.25 as a threshold to distinguish vegetated from Unvegetated areas within the affected portion of outer delta (Table 3.1).

Table 3.1. Summary of vegetated versus unvegetated NDVI values in 2004.

Unvegetated			**Vegetated**		
Mean	Mean + 2*SD	Maximum	Mean	Minimum	Mean – 2*SD
0.1347	0.2154	0.2507	0.5080	0.2500	0.3033

In the second site selection method, we determined the rate of NDVI change within the affected area using a pixel-based Theil-Sen regression of a Landsat NDVI time series (2001-2012). The significance of Theil-Sen slope estimates (using a Mann-Kendall test) was assessed using a threshold of p = 0.1. Within affected areas, significantly trended pixels with positive slope values were classified as Revegetated and pixels with non-significant or decreasing slopes were classified as Unvegetated. This regression-based classification was compared with the air photo-based classification and field sites were selected in areas where the two classifications agreed. Overall, we selected seven replicates within each of the affected site types and six replicates in the Reference type for a total of 20 sites (Table 3.2).

Table 3.2. Summary of site selection criteria using NDVI from Landsat imagery.

Site type	Classification criteria
Unaffected / Reference (n=6)	– $PNDVI_{2001} - PNDVI_{1998} > -0.358$
Affected / Revegetated (n=7)	– $PNDVI_{2001} - PNDVI_{1998} < -0.358$ – Significant increasing trend in NDVI from 2001-2012 – Green vegetation visible on 2004 air photos. – $NDVI_{2018} > 0.25$
Affected / Unvegetated (n=7)	– $PNDVI_{2001} - PNDVI_{1998} < -0.358$ – No trend in NDVI from 2001-2012 – No green vegetation (brown) visible on 2004 air photos. – $NDVI_{2018} < 0.25$

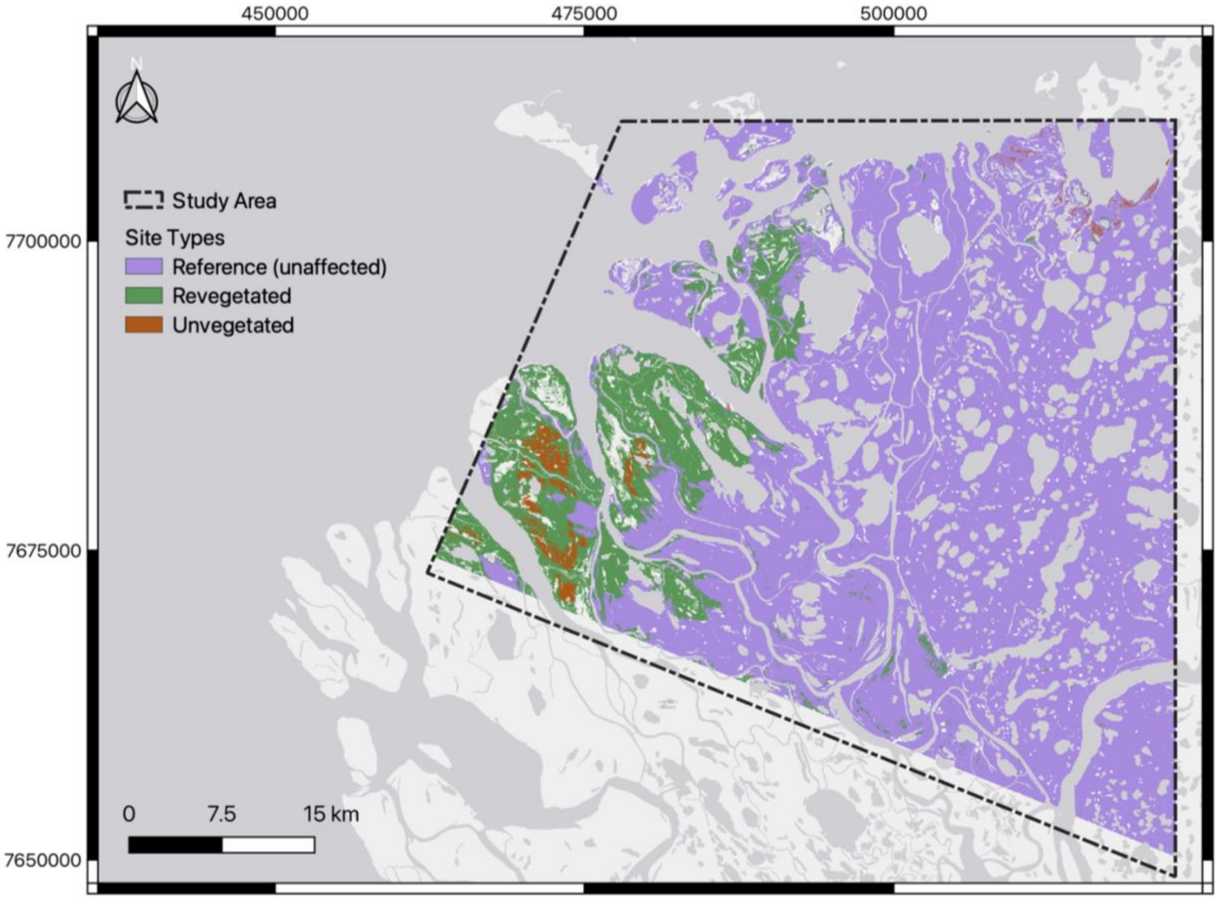


Figure 3.3. Map of three site types classified by the above classification criteria. This map was used to select sites.

3.2.3 Survey Methods

During the summer of 2019, we measured bird abundance and diversity in the outer delta during the shorebird breeding season (June 24 – July 4) using the Arctic Program for Regional and International Shorebird Monitoring's (PRISM) rapid survey protocol (Pirie et al., 2012). Although this protocol was created specifically for surveying breeding shorebirds, we were able to record all avian species present in the plots due to low general abundance. Specifically, two observers (F.N.A.S. and a field assistant) carried out walking transects spaced 25m apart in a 400 by 300m plot, repeating transects until we had covered the entire plot. Bird sightings, qualitative landcover types (i.e. tall shrubs, wet-graminoid, barren), and surface water were recorded on a grid diagram of the plot. Photographs and audio recordings were used to ensure correct species identification. If nests were found incidentally, they were recorded and photographed as well. All surveys were carried out during rainless, daylight hours to minimize weather effects on detection probability. During our time in the field, we completed 20 bird surveys (6 Reference, 7 Revegetated, 7 Unvegetated). Vegetation and soil surveys were also completed along 100m transects at each site to assess links between presence of birds and fine-scale habitat characteristics. At 11 points along these transects, nested quadrats (4m^2: tall shrubs & 0.25m^2: ground vegetation) were used to estimate the percent cover of vegetation and to measure soil characteristics (thaw depth, organic soil thickness, etc.). Plots were spaced at 10m intervals. Composite active layer samples were collected at 6 points along the transect (0m, 20m, 40m, 60m, 80m, 100m) to measure salinity, pH, and moisture content following the methods outlined by McKeague (1978). All vegetation and soil transects were placed within the bird plots perpendicular to the nearest river channel.

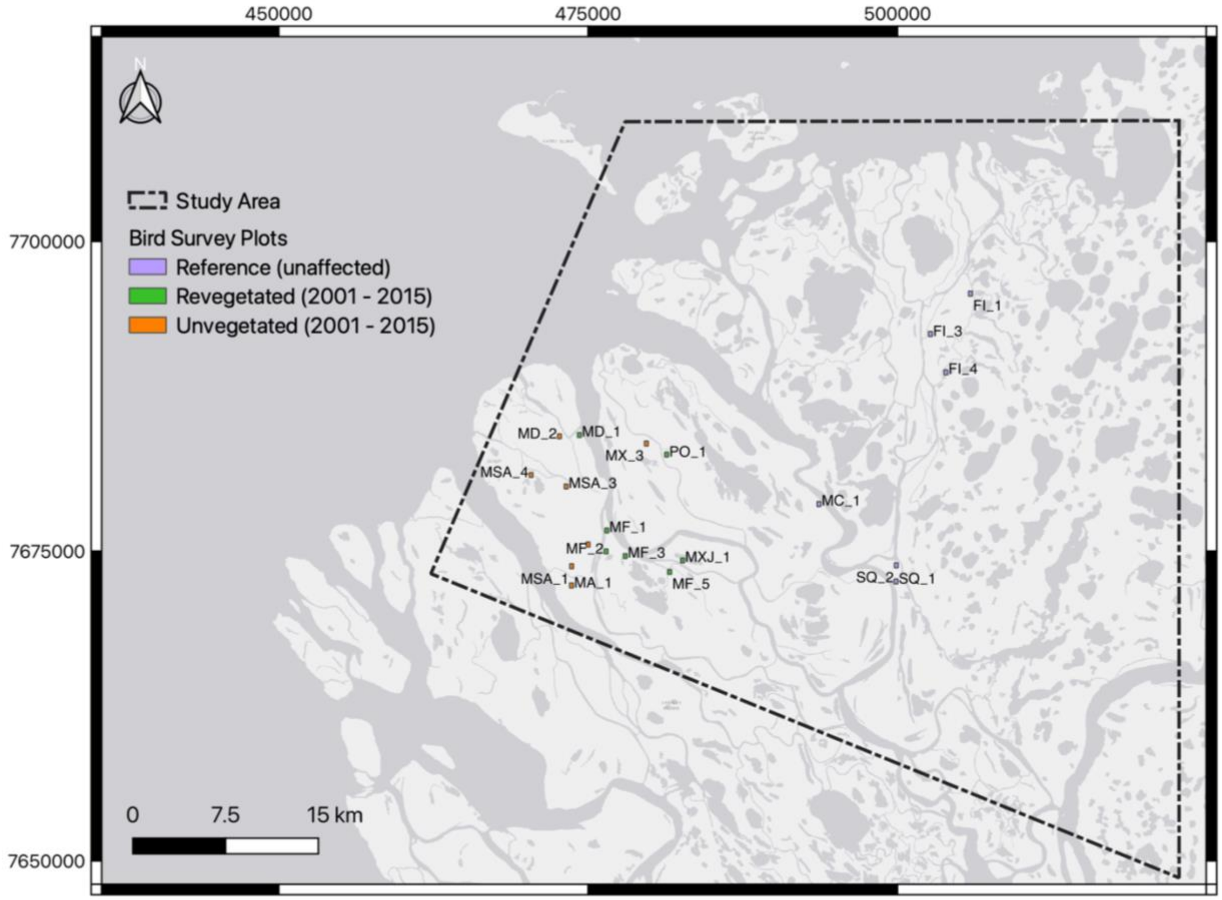


Figure 3.4. Map of all surveyed sites classified into the types created for this study. The rectangular boxes used to mark the location of the site are the actual size and orientation of the plots to scale.

3.2.4 Remote Sensing

To assess the influence of landscape variability on the abundance of focal bird species, we used the Landsat and Sentinel archives to measure surface wetness. Surface wetness was mapped by calculating Modified Normalized Difference Water Index (MNDWI) on Sentinel satellite imagery (from the beginning of July 2019; 10m resolution) (Du et al., 2016). Mean elevation was extracted from the ArcticDEM (2m resolution). Distance to nearest channel was also measured for each site using an MNDWI water mask derived from the Landsat 8 imagery.

3.2.5 Statistical Methods

To investigate differences in avian community composition among site types and explore community-habitat associations, we used a multivariate analysis: Non-parametric Multidimensional Scaling (NMDS) ordination in combination with an Analysis of Similarities (ANOSIM). This analysis was completed using a Bray-Curtis dissimilarity matrix calculated from the bird count data of the PRISM surveys (McGarigal et al., 2000). The relationships between habitat characteristics (see below) and community composition were assessed using the *envfit* function in the *vegan* package (Oksanen et al., 2015) in R (R Core Team, 2020). We also performed a Similarity Percentage (SIMPER) analysis to identify the species making the largest contributions to community dissimilarity among site types. To characterize environmental differences among site types, a Kruskal-Wallis test (rank-based analysis of variance) was performed on a suite of variables including: 1) NDVI, 2) MNDWI, 3) distance to channel, 4) elevation, 5) deciduous shrub cover, 6) vascular plant cover, 7) total vegetation cover, 8) soil moisture, 9) thaw depth, and 10) organic soil thickness. Total % cover was calculated by adding the % cover from the $4m^2$ quadrat and both $0.25m^2$ quadrats together.

Differences in the occurrence of three focal species identified in the SIMPER analysis (savannah sparrow (SAVS; *Passerculus sandwichensis*), lapland longspur (LALO; *Calcarius lapponicus*), and red-necked phalarope (RNPH; *Phalaropus lobatus*)) were assessed visually by plotting each species' abundance versus site type. Pairwise Fisher's Exact tests were then used to assess relationships between habitat variables and focal species' occurrences. We used this analysis to identify potential variables to focus on in future research or in modelling. Abundance data were transformed from counts into incidences (0/1) and habitat variables were converted into 4 categories using their quartile ranges. Each species was then individually compared to the binned habitat variables: NDVI, MNDWI, distance to channel, elevation, % cover of deciduous shrubs, % cover of vascular plants, total % cover of vegetation. Total % cover was calculated by adding the % cover from the 4m^2 quadrat and both 0.25m^2 quadrats together.

3.3 Results

3.3.1 Avian community composition between site types

The multivariate analysis (Figure 3.5) indicates that bird community composition at Unvegetated sites was significantly different from those at Revegetated and unaffected Reference sites (Table 3.3; R_{ANOSIM} = 0.84-0.85). Unvegetated sites were dominated by lapland longspur, semipalmated plover, and predatory species (ie. jaegers and hawks), while Reference and Revegetated sites were dominated by savannah sparrows and red-necked phalaropes (Table 3.5). Three "focal" species (as identified by the SIMPER analysis; savannah sparrow, lapland longspur, and red-necked phalarope) contributed to 45-61% of the dissimilarity between site types (Table 3.4; Supplementary Table 3.1). The variation in avian community composition within site type categories was the lowest in the vegetated sites (Reference and Revegetated) and highest in the Unvegetated sites (Table 3.5; Supplementary Figures 3.1, 3.2). The multivariate analysis also showed that bird community composition was highly correlated with plot-scale and remotely sensed measurements of habitat. Plots dominated by savannah sparrows & red-necked phalaropes were wetter and had higher vegetation cover than plots dominated by lapland longspur & semipalmated plover; the latter were dry and had low vegetation cover (Figure 3.5).

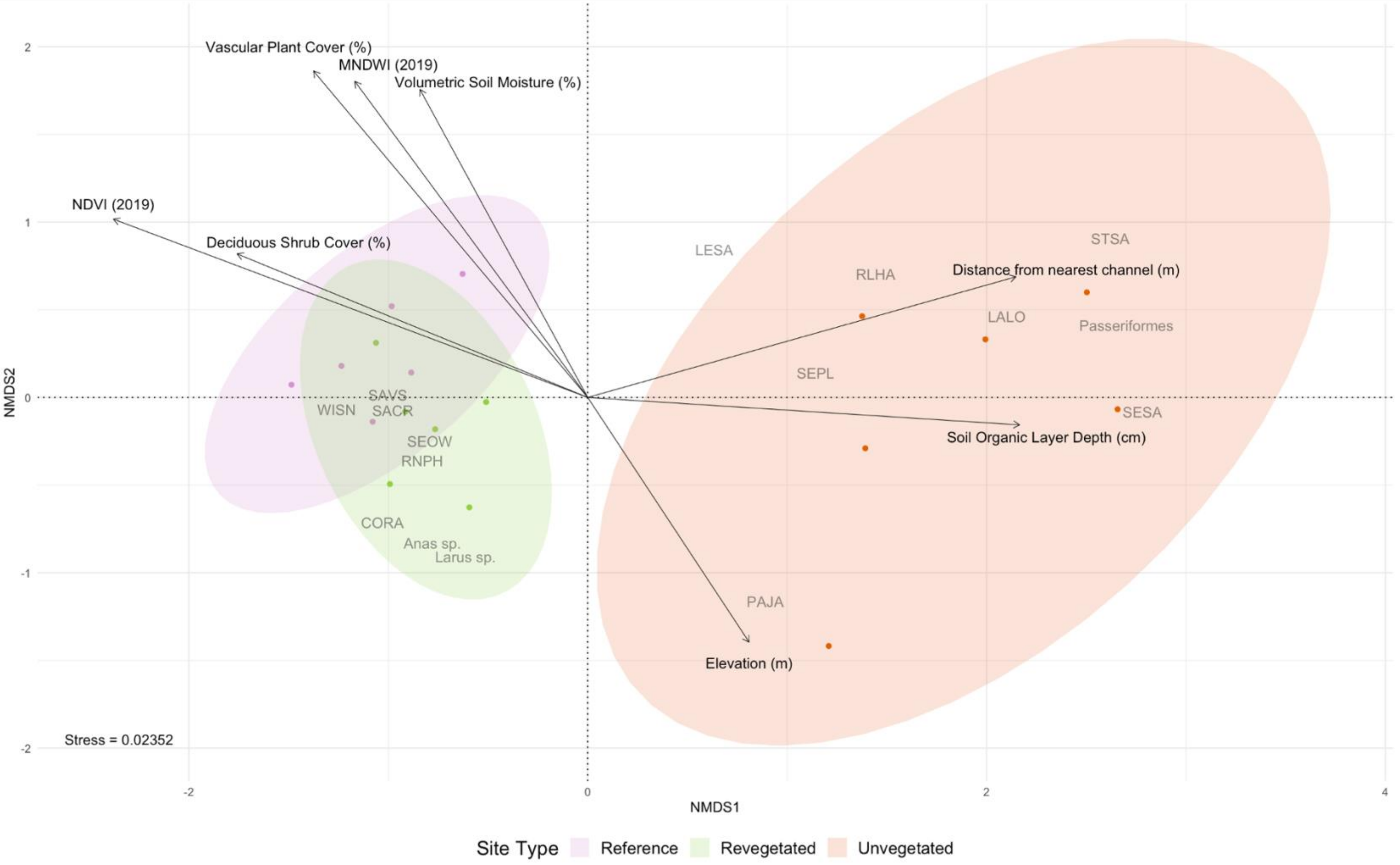


Figure 3.5. Non-parametric multidimensional scaling (NMDS) ordination of bird community composition based on Bray-Curtis similarity. The ordination shows the NMDS scores for each site type (colored points) and correlations between biotic and abiotic habitat variables and NMDS scores (solid arrows). The ellipses show the 95% confidence zone of each site type assuming a multivariate t-distribution (Wickham, 2016). Bird species are plotted in the ordination according to NMDS scores (calculated using the scores() function from the vegan package in R (Oksanen et al., 2015)). Scientific and common names corresponding to each species' abbreviation are provided in the appendix (Supplementary Table 3.2).

Table 3.3. Results of the ANOSIM analysis comparing community composition among site types. The Global R-statistic for this test was 0.589 ($p < 0.001$).

Site types (pairwise combinations)	R_{ANOSIM} statistic	Significance level (*p*)
Reference / Unvegetated	0.851	0.002
Reference / Revegetated	0.259	0.027
Unvegetated / Revegetated	0.843	0.002

Table 3.4. Results of the SIMPER analysis characterizing dissimilarity in bird community composition among three site types. The third column shows the species explaining the majority (>50%) of the cumulative dissimilarity among pairwise site type combinations. Mean abundance of birds per plot (untransformed) of each species at the site types being compared is shown in the fifth column.

Site type combinations	Average dissimilarity (%)	Species	Contribution to dissimilarity (%)	Average abundance at each site type
Reference / Unvegetated	99.00	Savannah sparrow	34.40	2.80 / 0
		Lapland longspur	20.93	0 / 1.83
		Semipalmated plover	7.35	0 / 0.76
Reference / Revegetated	44.05	Red-necked phalarope	31.21	0.35 / 1.74
		Dabbling ducks *(Anas* spp.*)*	15.40	0 / 0.94
		Savannah sparrow	13.77	2.80 / 2.70
Unvegetated/ Revegetated	95.94	Savannah sparrow	27.32	0 / 2.70
		Lapland longspur	17.28	1.83 / 0
		Red-necked phalarope	15.11	0.31 / 1.74

Notes: The top three species making the greatest contribution to the between-group Bray-Curtis dissimilarity for each site combination are shown.

3.3.2 Characterization of environmental differences among site types

As expected, Unvegetated sites had significantly lower vegetation cover (plot scale: 4m^2) and reduced NDVI in 2019 (site scale: 120 000m^2) compared to Reference and Revegetated sites (Figures 3.6A, 3.6B). Unvegetated sites were also drier, showing lower MNDWI (site scale: 120,000m^2) and volumetric moisture content (plot scale: 4m^2) (Figures 3.6C, 3.6D), and were located farther from freshwater channels than either Reference or Revegetated sites (Figure 3.6E).

Unvegetated sites had higher mean elevation than Reference sites but had similar elevation to Revegetated sites. Reference and Revegetated site also had similar percent vegetation cover, NDVI in 2019, MNDWI, soil moisture, and distance from channel (Figures 3.6A-E).

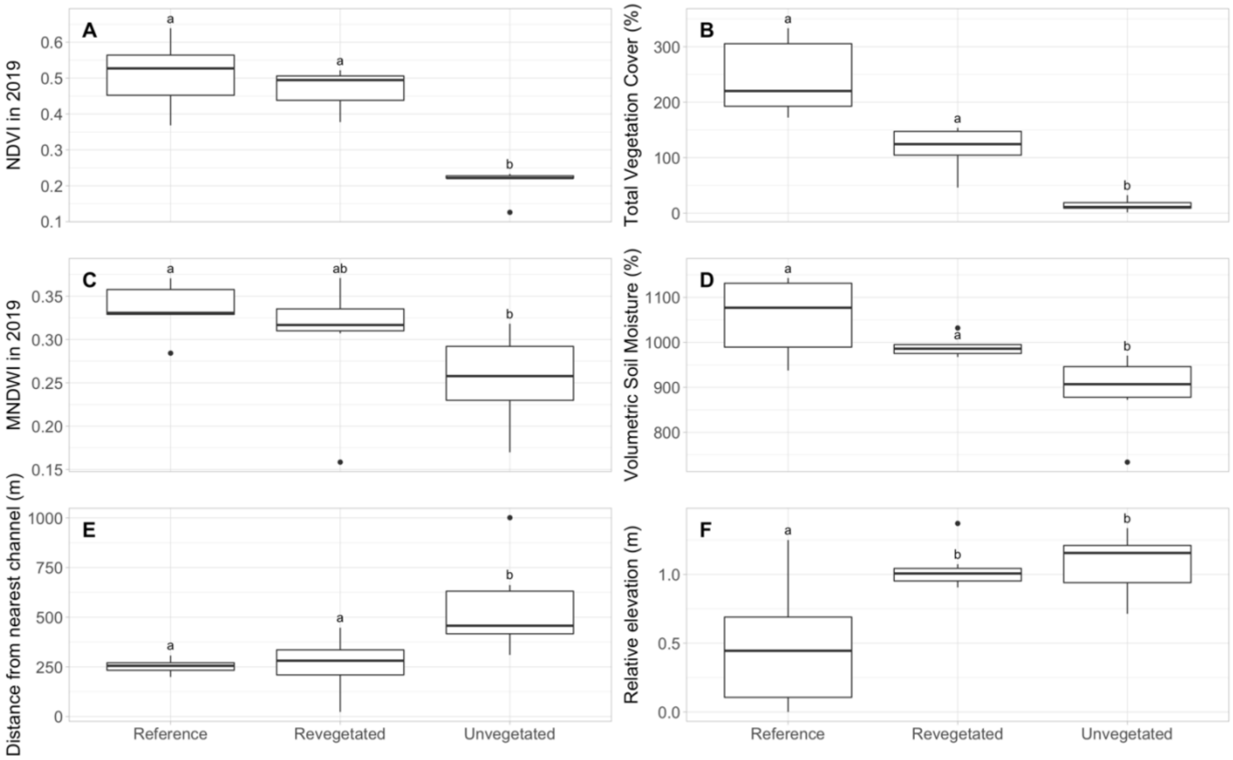


Figure 3.6. Box and whisker plots showing: (A) NDVI in 2019, (B) Total % cover of vegetation (cumulative % cover from all quadrats: $4m^2$, (a) $0.25m^2$, (b) $0.25m^2$), (C) MNDWI, (D) Volumetric soil moisture (%), (E) Distance from the nearest channel, and (F) Elevation at affected (Revegetated and Unvegetated) and unaffected (Reference) sites in the outer Mackenzie Delta. The horizontal line inside each box shows the median for each site type. The box edges show the 25th and 75th percentiles and the whiskers show [1.5*IQR]. Bars with different letters above them are significantly different ($p \leq 0.05$).

Figure 3.7. Photos from our 2019 field season of 3 site types. From left to right: Reference (unaffected), Revegetated (affected), Unvegetated (affected).

Reference sites were all dominated by sedges and variable cover of low shrubs. All Reference sites were consistently wet, and some had standing water. Revegetated sites were typically dominated by wet graminoid habitat, often interspersed with small areas of shrub/herb tundra (MF_1, MF_2 & MD_1). All these sites except one (MF_1) had some amount of standing water. Unvegetated sites were characterized by the dominance of bare ground (Figure 3.7).

3.3.3 Focal species assessment

The focal species assessment showed that savannah sparrow and lapland longspur did not co-occur at any of the sites we sampled (Figure 3.8). Savannah sparrow were only observed at Reference and Revegetated sites where NDVI was > 0.2 and lapland longspur were only observed at Unvegetated sites (Figure 3.8A). Unvegetated sites were > 300m from nearest channel and had NDVI < 0.25 (Figure 3.8A-B).

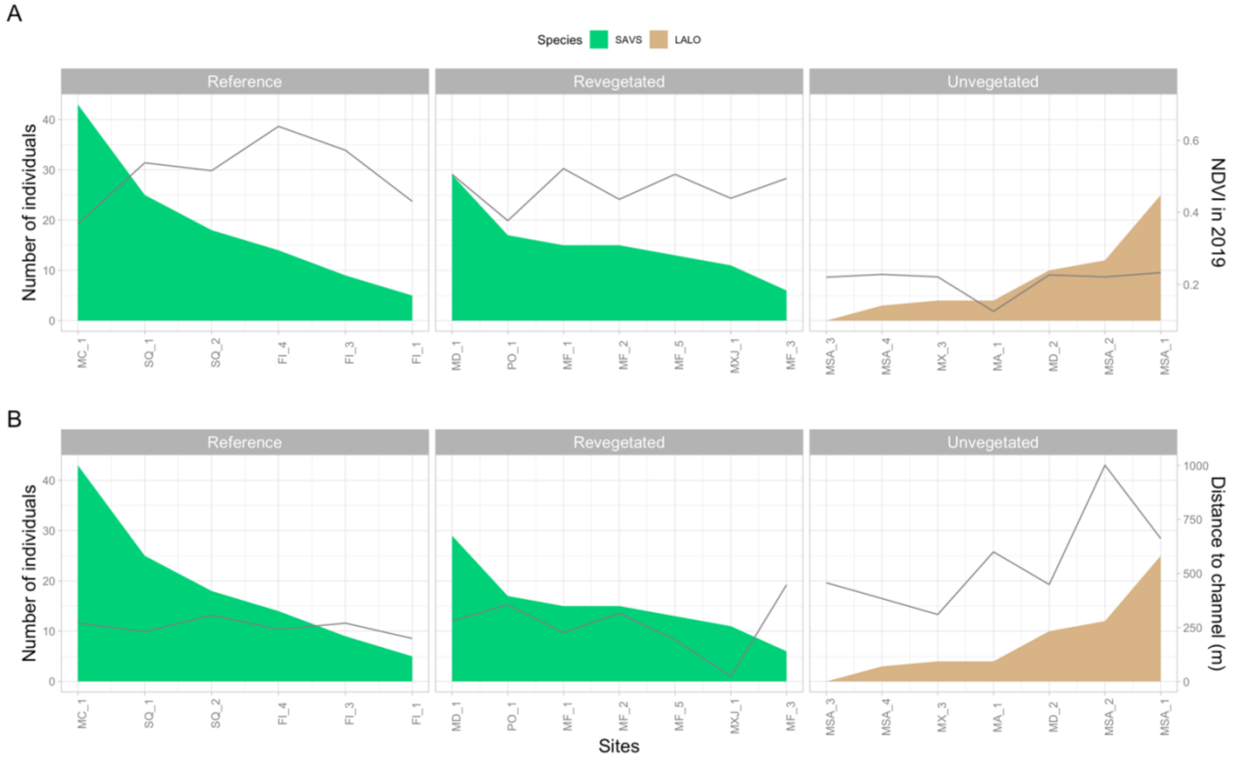


Figure 3.8. Gradient diagrams showing the number of savannah sparrows (green) and lapland longspurs (brown) at sites across the study area. Lapland longspurs were never observed at Reference and Revegetated site types and savannah sparrows were never observed at Unvegetated site types. The grey line on panel (A) shows NDVI in 2019 at each site and the grey line on panel (B) shows the distance to the nearest channel (in meters) at each site. Sites are arranged along the x-axis into their site types.

Fisher exact tests showed that focal species' occurrences were correlated with vegetation cover and habitat wetness (Table 3.5). Savannah sparrows were never present at unvegetated sites (Figure 3.8), and Fisher's exact tests showed that they preferred sites with more vegetation cover (Figure 3.9A, 3.8C). They also preferred sites that were closer to channels and had higher moisture (Figure 3.9B, 3.8D). Lapland longspur were never present at Revegetated or Reference sites (Figure 3.8) and Fisher's exact tests showed that they preferred sites with low vegetation cover and moisture further from channels (Figure 3.10A-D). Red-necked phalarope were absent from all Unvegetated sites but one and preferred sites with moderate to high cover of vegetation that were an intermediate distance from channels (Figures 3.10C, 3.10D).

Table 3.5. Global significance statistics for Fisher's Exact comparing the incidence of focal species with habitat variables. Significant associations are shown in bold text.

Species	Habitat Variable	Global p-value
Savannah sparrow	NDVI in 2019	**0.0047**
	Total cover of vegetation (%)	**0.00068**
	MNDWI in 2019	0.091
	Distance from nearest channel (m)	**0.0047**
Lapland longspur	NDVI in 2019	**0.0088**
	Total cover of vegetation (%)	**0.013**
	MNDWI in 2019	0.38
	Distance from nearest channel (m)	**0.039**
Red-necked phalarope	NDVI in 2019	**0.0053**
	Total cover of vegetation (%)	0.17
	MNDWI in 2019	1
	Distance from nearest channel (m)	0.24

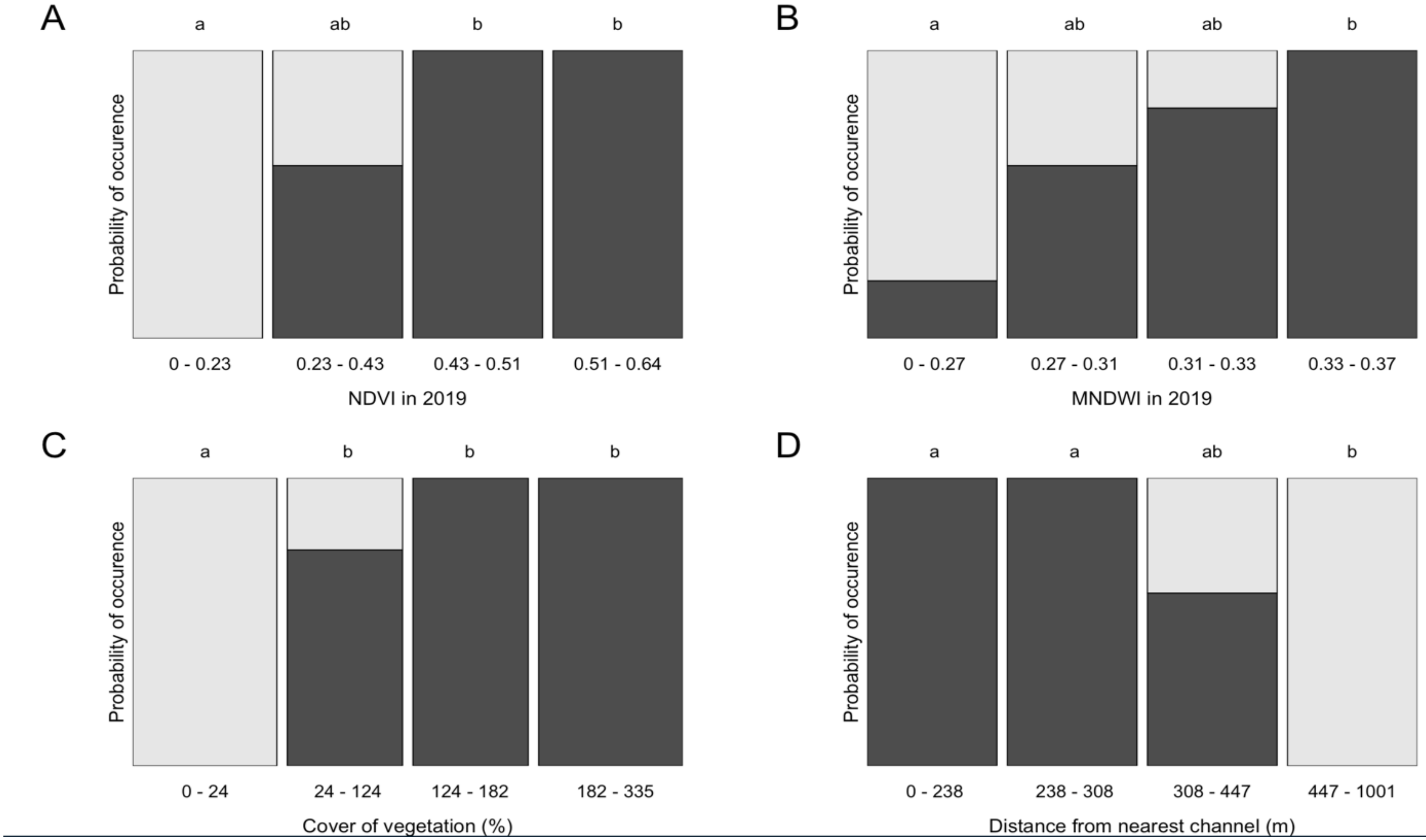


Figure 3.9. Incidence of savannah sparrow in comparison to habitat variables including: (A) NDVI in 2019, (B) MNDWI in 2019, (C) Cover of vegetation (%), and (D) Distance from the nearest channel (meters). Explanatory variables were binned based on quantiles. Bars with different letters were significantly different (Fisher's Exact test $p<0.05$).

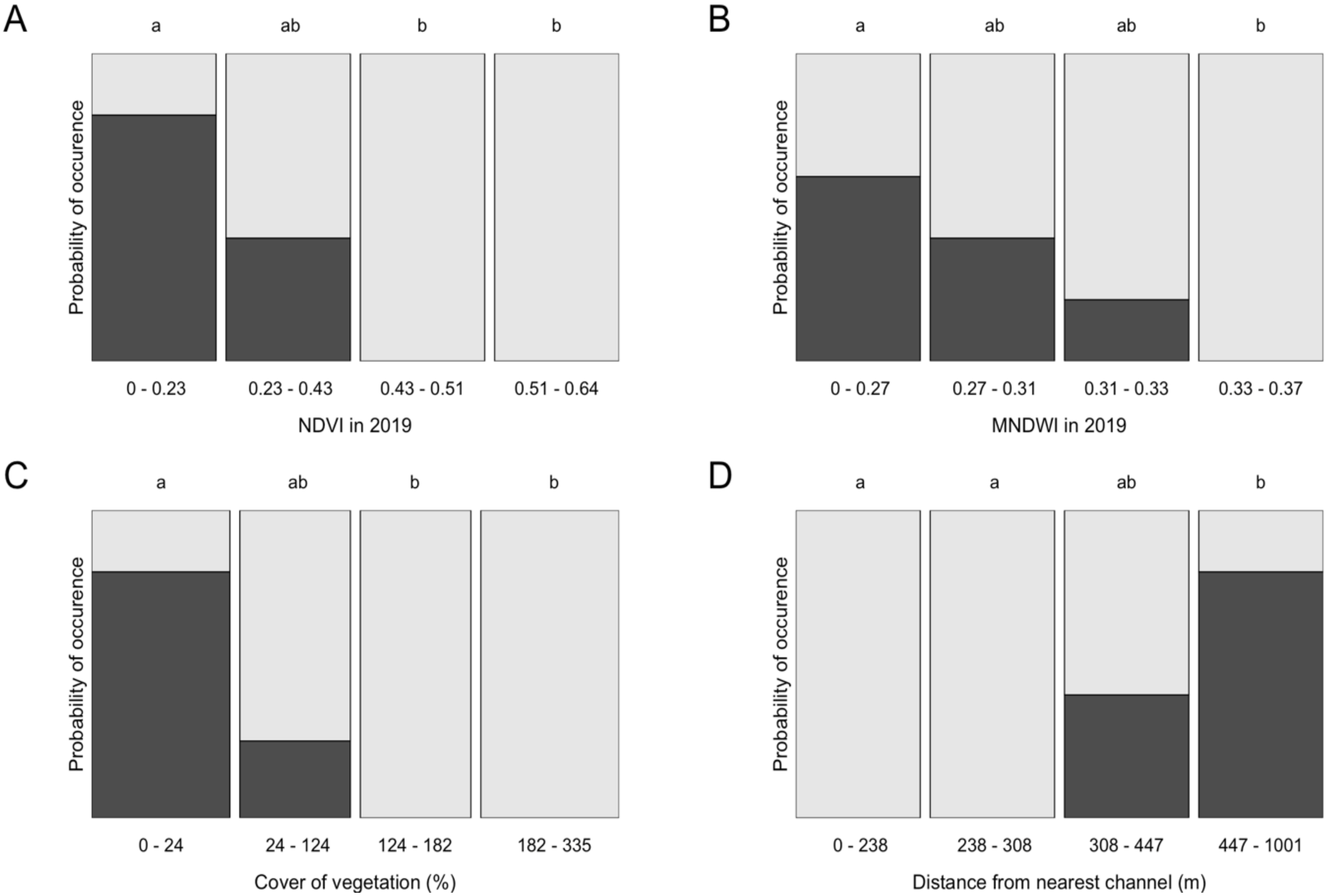


Figure 3.10. Incidence of lapland longspur in comparison to habitat variables including: A) NDVI in 2019, B) MNDWI in 2019, C) Cover of vegetation (%), and D) Distance from the nearest channel (meters). Explanatory variables were binned based on quantiles. Bars with different letters were significantly different (Fisher's Exact test $p<0.05$).

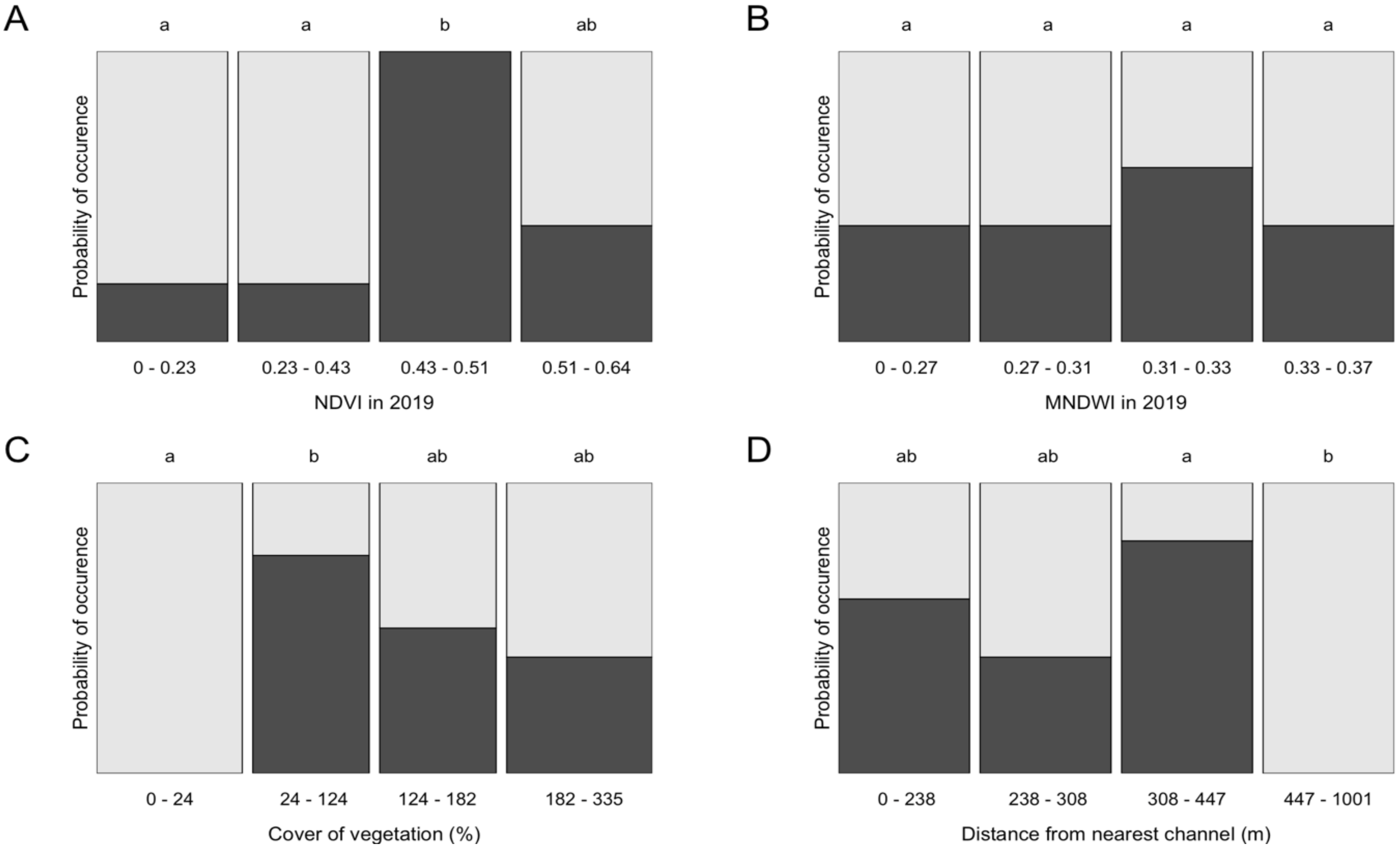


Figure 3.11. Incidence of red-necked phalarope in comparison to habitat variables including: A) NDVI in 2019, B) MNDWI in 2019, C) Cover of vegetation (%), and D) Distance from the nearest channel (meters). Explanatory variables were binned based on quantiles. Bars with different letters were significantly different (Fisher's Exact test $p<0.05$).

3.4 Discussion

3.4.1 Main findings

Our analysis shows that mesoscale differences in bird community composition and the occurrences of focal species were driven by the persistent effects of the 1999 storm surge on habitat. In our study, avian community assemblage was correlated with vegetation structure and productivity as reflected in post-storm NDVI values (Figure 3.5). This is consistent with previous studies where density and diversity of avian species responded to productivity gradients of vegetation in the Arctic (Latour et al., 2005; Smith et al., 2020). Additionally, all of our focal species showed strong associations to NDVI values from Landsat images acquired during the field season (2019) (Figures 3.8-3.11). Though this was likely due to the dramatic contrast in vegetation presence between Unvegetated and vegetated (Reference and Revegetated) site types, these associations could be a good approach for remotely identifying areas that are unsuitable for certain species.

The lack of ground cover and surface water at Unvegetated sites made them unsuitable for species that occupy aquatic habitat, such as red-necked phalaropes and ducks, or materials for camouflaging their nests and perching, such as savannah sparrows. Savannah sparrow are known to favour dense ground vegetation, especially grasses, and moist microhabitats (Wiens et al., 1986). They also prefer to nest in vegetated areas without bare ground since they require material to cover nests and to hide from predators (Davis, 2005; Dieni & Jones, 2003; Rockwell et al., 2003). In contrast, red-necked phalarope breed at or near lakes and pools with marshy margins, where their nests are rarely > 20 m from a source of water (Murphy, 1981). Their nests are sometimes concealed in clumps of sedges or grasses or under other low vegetation (Bent, 1927) so they typically prefer pools with emergent vegetation (Höhn, 1968).

The strong association between Unvegetated sites and a particular suite of species shows that vegetation die-back following saline incursion can shift the relative abundance of birds. Unvegetated sites were characterized by bare ground and the absence of surface water and were dominated by lapland longspur and semipalmated plover along with various predatory birds (such as jaegers and hawks) that prefer these conditions. In a summer study on Bylot Island (Nunavut), lapland longspur were observed foraging mainly at margins of dense vegetation or in areas with sparse clumps of plants on margins of barren spots (Drury, 1961). Although some prior studies have indicated that lapland longspur prefer low-canopy vegetated habitats (Boal & Andersen, 2005), we observed them (both audio and visual observations) only in Unvegetated sites.

Though not particularly abundant in our study, semipalmated plover also showed clear preference for the Unvegetated sites. This is likely because they require open, flat ground for running and foraging (Burger et al., 1977; Recher, 1966; Smith & Nol, 2000). In habitat selection studies on Akimsiki Island (Nunavut), semipalmated plover showed preference for open, pebbled areas for breeding even though their nests in muddier habitats were more likely to be successful (Nguyen et al., 2003, 2013). Additionally, many species observed in Unvegetated sites were predatory (parasitic jaeger, rough-legged hawk, and a short-eared owl (*Asio flammeus*)). It is possible that these species prefer hunting in these habitats where there is no cover (Liebezeit et al., 2009). We did observe territorial behaviours (alarm calling and circling) by the jaegers and hawks, so it is possible that some of these species are also using these habitats for breeding.

Our analysis suggests that revegetation 20 years following the storm surge has been adequate to restore habitat functionality for certain avian species at some sites. Avian community composition, focal species' identities, and habitat conditions were functionally similar between unaffected Reference and affected Revegetated sites. Prior to this study, we were unsure whether birds would avoid returning to affected, but revegetated areas. We also had no reference for the threshold at which enough vegetation reestablishment would overcome any potential avoidance behaviour. Observations made in this study indicate that these Revegetated sites have recovered sufficiently to host similar avian populations as the unaffected Reference sites.

From a strictly observational standpoint, the structure of the vegetation communities between the two vegetated site types appeared different on the ground. Reference sites had more shrub cover overall and/or more tall shrubs, whereas Revegetated sites tended to be mostly covered in sedges, some quite homogenously so. Despite these observed differences in plant community composition between the vegetated site types, the avian communities at these sites were similar, suggesting that functional recovery occurs within twenty years of vegetation loss in a productive Arctic delta, at least from the perspective of breeding birds. Further analysis of the vegetation cover revealed no significant differences between Revegetated and Reference site communities in terms of plant species assemblages (see Chapter 2). The avian species present at Revegetated sites were more consistent, with assemblages between sites being different due to changes in avian species richness rather than identity (Section 3.6 Appendix: Supplementary Figure 3.1, 3.2). This could be attributed to the relative homogeneity of the vegetation community and habitat structure at the Revegetated sites (Brandolin et al., 2016; Smith et al., 2020). Vegetation community composition aside, both vegetated site types were quite wet, consistently having standing water available. Although we found low differences ($R_{ANOSIM} = 0.259$, $p = 0.027$; Table 3.3) in avian community composition between Revegetated and Reference sites, we found nests and/or nesting behaviours by ducks, Wilson's snipe (*Gallinago delicata*), and red-necked phalarope only at Revegetated sites. No evidence of nesting (or "probable nesting") was found in Reference sites. This could point towards some species using these two habitat types for different purposes, but further study over a longer period is required to confirm these patterns.

3.4.2 Significance

Our study shows that the persistent impacts of terrestrial salinization following the 1999 storm have altered avian community composition in affected areas of the delta. Remote sensing estimates from 2019 suggest that ~3345 ha (~13% of the originally affected terrestrial surface area) have not recovered (to threshold 0.25 NDVI). To our knowledge, this is the first systematic study assessing the effects of terrestrial habitat change from saline incursion on Arctic birds. Our analysis suggests that habitat changes from more frequent and severe storm surges will alter bird community composition and perhaps the abundance of individual species. Vegetation loss creates habitat conditions that are avoided by certain species but selected for by others. With climate change resulting in recurrent and severe changes to future seasonal temperatures and sea-ice condition, extreme weather events in the Mackenzie Delta such as storm surges are likely to become the norm rather than the exception (Atkinson, 2005; Sepp & Jaagus, 2011; Simmonds & Keay, 2009). With this projected increase in storm surge frequency and intensity, the potential for additional areas to become persistently unvegetated also increases, something that highlights the importance of considering the cumulative effects of repeated storm surges and disturbances associated with permafrost thaw, increasing oil and gas development, and other anthropogenic disturbances (Barry & Spencer, 1976). Our results provide insight into how future vegetation changes may affect habitat availability for some avian species, contributing to improved predictions of how climate changes will affect Arctic bird populations in the Mackenzie Delta and beyond. This study can also help inform impact assessments and protected area management for Arctic breeding birds in the face of ongoing development decisions.

3.4.3 Limitations

Typically, the PRISM protocol for monitoring breeding shorebirds that we used in this study is deployed during an early spring survey window (late May & June) since some early breeders begin southward migration soon after nesting initiation in mid-May (Pirie & Johnston, 2012; Weiser et al., 2018). However, in the Mackenzie Delta, shorebird nesting initiation is limited by flooding during channel break-up (Pirie et al., 2012). As such, previous PRISM studies in the delta have delayed the surveying period until the first week of July (our survey period) with success (Pirie et al., 2012). Since we did not intend to use our data for population estimates or trends, we focussed our analyses more on incidence (presence/absence) rather than abundance to place less importance on potential missing individuals from early breeding.

Avian biodiversity studies often underestimate the natural dynamism of bird community assemblages, overemphasizing the importance of environmental variables and underestimating the importance of temporal dynamics (Collins, 2001; Curtis et al., 2016). Since our data collection was limited to a single field season and there are no recent surveys in our study area, we could not control for interannual changes in bird presence or pre-storm variation in vegetation structure (sedge wetland, upland tundra, tall shrub). However, despite the lack of historic and ongoing monitoring, this study has provided novel, valuable baseline data for the use of habitat damaged by saltwater intrusion by birds in the Mackenzie Delta. There is now a benchmark for future studies to compare against.

3.4.4 Further Study

Although we found clear, persistent effects of vegetation change on bird community structure, a more detailed understanding of habitat selection by Arctic-nesting birds would allow more precise application of fine-scale remote-sensing products. The importance of prey availability and site fidelity are some other potential drivers of the responses to environmental change in bird communities (Devictor et al., 2008; Hitch & Leberg, 2007; Knick & Rotenberry, 2000; Romañach et al., 2019). Firstly, declines in avian species with high site fidelity are already being seen in the Arctic since breeding habitat ranges of most Arctic species are shifting northward (Ganter & Gaston, 2013; Lappo et al., 2012; Smith et al., 2020). Previous studies have also looked at the association of insect emergence and shorebird hatchling success in other areas of the Canadian Arctic (Bolduc et al., 2013; McKinnon et al., 2012; van Gils et al., 2016), but prey availability in our study area has not yet been assessed in the context of saline incursion (Boelman et al., 2015; Desender & Maelfait, 1999;

Pétillon et al., 2008). Additionally, water levels change year-to-year (monthly in the summer) in the active outer delta (Gratto-Trevor, 1996). So far, most storm surge events in the Mackenzie Delta have only been studied by chance (Scharffenberg, Whalen, et al., 2020), or many years after the original event (Deasley et al., 2012; Kokelj et al., 2012; Lantz et al., 2015; Pisaric et al., 2011; Thienpont et al., 2012, 2015). Though local knowledge from communities in the delta region such as Inuvik and Tuktoyaktuk has provided invaluable insight into storms, their ecological and logistic effects, and factors that influence their intensity, the overall lack of historical data is still a hinderance when considering some research questions that rely on comparing specific data from before & after an event or identifying patterns in quantitative variables over time. Improved and consistent monitoring in the delta could aid in determining the factors influencing the magnitude of habitat salinization after storm surges and would provide better baseline data for future comparisons. Studies assessing the threshold conditions for habitat suitability and regular monitoring of storm surge activity could theoretically help to predict the severity of storm event impacts on available bird breeding habitat. This would be able to inform management decisions in a more proactive – rather than reactive – manner.

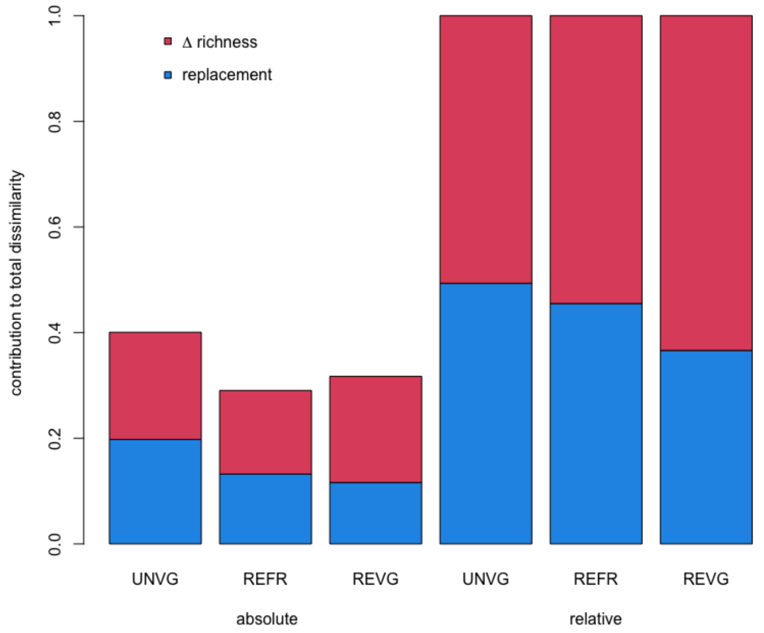
3.5 Conclusion

This natural experiment — the saltwater incursion in portions of the outer Mackenzie Delta — provides evidence that vegetation changes in storm-surge-affected habitats can have significant and lasting effects on habitat suitability for avian species. The degree to which each individual species will be affected by this habitat loss or degradation will depend on the compounding effects of several factors, including those presented in this study and pressures from outside their breeding territory. Since breeding success drives population size and diversity, the maintenance of Arctic-breeding bird populations depends on the ability of each species to utilize patchily distributed habitat in a changing Arctic landscape. Currently, there is limited information available on the breeding success of Arctic bird species in the face of breeding habitat shifts and reduction. Understanding the impacts of large-scale salinization events on habitat availability for breeding birds is important for current and future monitoring and conservation efforts, especially for avian species that breed exclusively in the Arctic.

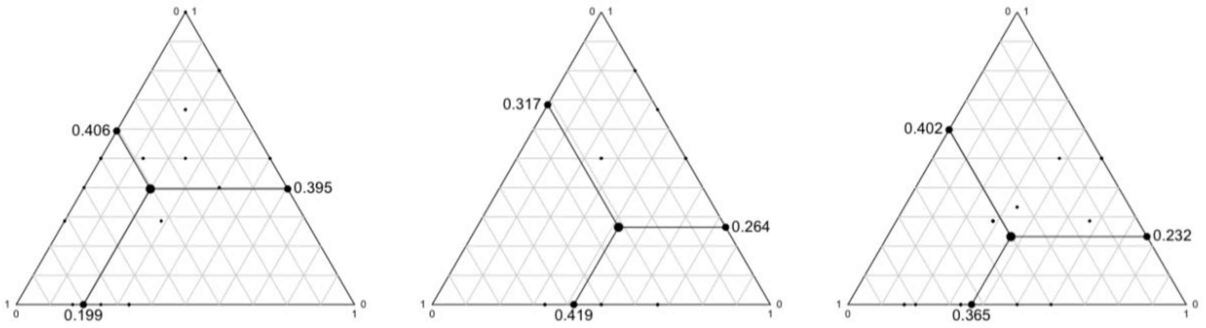
3.6 Appendix

Supplementary Table 3.1. Results of the SIMPER analysis characterizing dissimilarity in community composition between site types. The third column shows the contributions of the three focal species (SAVS, LALO, RNPH) to the cumulative dissimilarity among pairwise site type combinations. Mean abundance (untransformed) of each species at the site types being compared is shown in the fifth column.

Site type combinations	Average dissimilarity (%)	Species	Contribution to dissimilarity (%)	Average abundance
Reference / Unvegetated	99.00	Savannah sparrow	34.40	2.80 / 0
		Lapland longspur	20.93	0 / 1.83
		Red-necked phalarope	5.49	0.35 / 0.31
Reference / Revegetated	44.05	Red-necked phalarope	31.21	0.35 / 1.74
		Savannah sparrow	13.77	2.80 / 2.70
		Lapland longspur	-	- / -
Unvegetated/ Revegetated	95.94	Savannah sparrow	27.32	0 / 2.70
		Lapland longspur	17.28	1.83 / 0
		Red-necked phalarope	15.11	0.31 / 1.74



Supplementary Figure 3.1. Bar chart of both absolute and relative contributions to total dissimilarity for comparison of importance between species richness versus species replacement. (UNVG = Unvegetated site type; REFR = Reference site type; REVG = Revegetated site type). Relative measures are created by dividing by the total number of species in each site and allow direct comparison between sites with different absolute richnesses. Species replacement is more important for Unvegetated sites than for the vegetated site types. Changes in species richness are more important Revegetated sites.



Supplementary Figure 3.2. Ternary plots for the 3 different site types. From left to right: Unvegetated (affected), Reference (unaffected), Revegetated (affected). The three axes are: (bottom) similarity, (left) change in species richness, (right) species replacement.

Supplementary Table 3.2. Species codes from Figure 3.5 with their common and scientific names.

4-letter code	Species	Scientific name
ARTE	Arctic tern	*Sterna paradisaea*
CORA	Common raven	*Corvus corax*
GLGU	Glaucous gull	*Larus hyperboreus*
LALO	Lapland longspur	*Calcarius lapponicus*
LESA	Least sandpiper	*Calidris minutilla*
PAJA	Parasitic jaeger	*Stercorarius parasiticus*
PESP	Pectoral sandpiper	*Calidris melanotos*
RLHA	Rough-legged hawk	*Buteo lagopus*
RNPH	Red-necked phalarope	*Phalaropus lobatus*
SACR	Sandhill crane	*Grus canadensis*
SAVS	Savannah sparrow	*Passerculus sandwichensis*
SEOW	Short-eared owl	*Asio flammeus*
SEPL	Semipalmated plover	*Charadrius semipalmatus*
SESA	Semipalmated sandpiper	*Calidris pusilla*
STSA	Stilt sandpiper	*Calidris himantopus*
WISN	Wilson’s snipe	*Gallinago delicata*

3.7 References

Atkinson, D. E. (2005). Observed storminess patterns and trends in the circum-Arctic coastal regime. Geo-Marine Letters, 25(2), 98–109. https://doi.org/10.1007/s00367-004-0191-0

Baker, A. J., González, P. M., Piersma, T., Niles, L. J., do Nascimento, I. de L. S., Atkinson, P. W., Clark, N. A., Minton, C. D. T., Peck, M. K., & Aarts, G. (2004). Rapid population decline in red knots: Fitness consequences of decreased refuelling rates and late arrival in Delaware Bay. Proceedings of the Royal Society B: Biological Sciences, 271(1541), 875–882.

Barry, T. W., & Spencer, R. (1976). Wildlife response to oil well drilling. https://www.osti.gov/etdeweb/biblio/6268212

Bent, A. C. (1927). Life Histories of North American Shore Birds: Order Limicolae. U.S. Government Printing Office.

Berkes, F., & Jolly, D. (2001). Adapting to Climate Change: Social-Ecological Resilience in a Canadian Western Arctic Community. Conservation Ecology, 5(2). https://doi.org/10.5751/ES-00342-050218

Boal, C. W., & Andersen, D. E. (2005). Microhabitat Characteristics of Lapland Longspur, Calcarius lapponicus, Nests at Cape Churchill, Manitoba. The Canadian Field-Naturalist, 119(2), Article 2. https://doi.org/10.22621/cfn.v119i2.108

Boelman, N. T., Gough, L., Wingfield, J., Goetz, S., Asmus, A., Chmura, H. E., Krause, J. S., Perez, J. H., Sweet, S. K., & Guay, K. C. (2015). Greater shrub dominance alters breeding habitat and food resources for migratory songbirds in Alaskan arctic tundra. Global Change Biology, 21(4), 1508–1520. https://doi.org/10.1111/gcb.12761

Bolduc, E., Casajus, N., Legagneux, P., McKinnon, L., Gilchrist, H., Leung, M., Morrison, R., Reid, D., Smith, P., Buddle, C., & Bêty, J. (2013). Terrestrial arthropod abundance and phenology in the Canadian Arctic: Modelling resource availability for Arctic-nesting insectivorous birds. The Canadian Entomologist, 145, 155–170. https://doi.org/10.4039/tce.2013.4

Both, C., Van Turnhout, C. A. M., Bijlsma, R. G., Siepel, H., Van Strien, A. J., & Foppen, R. P. B. (2010). Avian population consequences of climate change are most severe for long-distance migrants in seasonal habitats. Proceedings of the Royal Society B: Biological Sciences, 277(1685), 1259–1266. https://doi.org/10.1098/rspb.2009.1525

Brandolin, P. G., Blendinger, P. G., & Cantero, J. J. (2016). From Relict Saline Wetlands to New Ecosystems: Changes in Bird Assemblages. Ardeola, 63(2), 329–348. https://doi.org/10.13157/arla.63.2.2016.ra7

Bromley, B., Fehr, A., & Black, S. (2002). Birds. In Natural history of the Western Arctic (pp. 45–54). Western Arctic Handbook Project.

Buehler, D. M., & Baker, A. J. (2005). POPULATION DIVERGENCE TIMES AND HISTORICAL DEMOGRAPHY IN RED KNOTS AND DUNLINS. The Condor, 107(3), 497–513. https://doi.org/10.1650/0010-5422(2005)107[0497:PDTAHD]2.0.CO;2

Burger, J., Howe, M. A., Hahn, D. C., & Chase, J. (1977). Effects of Tide Cycles on Habitat Selection and Habitat Partitioning by Migrating Shorebirds. The Auk, 94(4), 743–758. https://doi.org/10.2307/4085271

Burn, C. R. (2017). Mackenzie Delta: Canada's Principal Arctic Delta. In O. Slaymaker (Ed.), Landscapes and Landforms of Western Canada (pp. 321–334). Springer International Publishing. https://doi.org/10.1007/978-3-319-44595-3_23

Burn, C. R., & Kokelj, S. V. (2009). The environment and permafrost of the Mackenzie Delta area. Permafrost and Periglacial Processes, 20(2), 83–105. https://doi.org/10.1002/ppp.655

Callaghan, T. V., Björn, L. O., Chernov, Y., Chapin, T., Christensen, T. R., Huntley, B., Ims, R. A., Johansson, M., Jolly, D., Jonasson, S., Matveyeva, N., Panikov, N., Oechel, W., Shaver, G., Elster, J., Henttonen, H., Laine, K., Taulavuori, K., Taulavuori, E., & Zöckler, C. (2004). Biodiversity, Distributions and Adaptations of Arctic Species in the Context of Environmental Change. AMBIO: A Journal of the Human Environment, 33(7), 404–417. https://doi.org/10.1579/0044-7447-33.7.404

Canadian Wildlife Service. (1992). Management of migratory bird sanctuaries in the Inuvialuit Settlement Region (p. 26). Environment Canada.

Carmack, E. C., & Macdonald, R. W. (2002). Oceanography of the Canadian Shelf of the Beaufort Sea: A Setting for Marine Life. Arctic, 55, 29–45.

Christensen, T., Payne, J., Doyle, M., Ibarguchi, G., Taylor, J., Schmidt, N. M., Gill, M., Svoboda, M., Aronsson, M., Behe, C., Buddle, C., Cuyler, C., Fosaa, A. M., Fox, A. D., Heiðmarsson, S., Henning Krogh, S., Madsen, J., McLennan, D., Nymand, J., … Wedege, M. (2013). Arctic Terrestrial Biodiversity Monitoring Plan. Conservation of Arctic Flora and Fauna (CAFF). https://oaarchive.arctic-council.org/handle/11374/230

Clemens, R., Rogers, D. I., Hansen, B. D., Gosbell, K., Minton, C. D. T., Straw, P., Bamford, M., Woehler, E. J., Milton, D. A., Weston, M. A., Venables, B., Wellet, D., Hassell, C., Rutherford, B., Onton, K., Herrod, A., Studds, C. E., Choi, C.-Y., Dhanjal-Adams, K. L., … Fuller, R. A. (2016). Continental-scale decreases in shorebird populations in Australia. Emu - Austral Ornithology, 116(2), 119–135. https://doi.org/10.1071/MU15056

Collins, S. L. (2001). Long-Term Research and the Dynamics of Bird Populations and Communities. The Auk, 118(3), 583–588. https://doi.org/10.2307/4089922

Conklin, J. R., Reneerkens, J., Verkuil, Y. I., Tomkovich, P. S., Palsbøll, P. J., & Piersma, T. (2016). Low genetic differentiation between Greenlandic and Siberian Sanderling populations implies a different phylogeographic history than found in Red Knots. Journal of Ornithology, 157(1), 325–332. https://doi.org/10.1007/s10336-015-1284-4

Cordes, L. D., McLennan, D. S., & Pearce, C. M. (1984). Alluvial ecosystems on the Mackenzie Delta, N.W.T (Environment and Socio-Economic Services). BC Hydro. https://search.library.uvic.ca

Curtis, J. R., Robinson, W. D., & McCune, B. (2016). Time trumps habitat in the dynamics of an avian community. Ecosphere, 7(11), e01575. https://doi.org/10.1002/ecs2.1575

Davis, S. K. (2005). Nest-Site Selection Patterns and the Influence of Vegetation on Nest Survival of Mixed-Grass Prairie Passerines. The Condor, 107(3), 605–616. https://doi.org/10.1093/condor/107.3.605

Deasley, K., Korosi, J. B., Thienpont, J. R., Kokelj, S. V., Pisaric, M. F. J., & Smol, J. P. (2012). Investigating the response of Cladocera to a major saltwater intrusion event in an Arctic lake from the outer Mackenzie Delta (NT, Canada). Journal of Paleolimnology, 48(2), 287–296. https://doi.org/10.1007/s10933-012-9577-6

Desender, K., & Maelfait, J.-P. (1999). Diversity and conservation of terrestrial arthropods in tidal marshes along the River Schelde: A gradient analysis. Biological Conservation, 87(2), 221–229. https://doi.org/10.1016/S0006-3207(98)00058-5

Devictor, V., Julliard, R., Couvet, D., & Jiguet, F. (2008). Birds are tracking climate warming, but not fast enough. Proceedings of the Royal Society B: Biological Sciences, 275(1652), 2743–2748. https://doi.org/10.1098/rspb.2008.0878

Dieni, J. S., & Jones, S. L. (2003). Grassland Songbird Nest Site Selection Patterns in Northcentral Montana. The Wilson Bulletin, 115(4), 388–396.

Donaldson, G., Hyslop, C., Morrison, G., Dickson, L., & Davidson, I. (Eds.). (2000). Canadian Shorebird Conservation Plan. Canadian Wildlife Service.

Drury, W. H. (1961). Studies of the Breeding Biology of Horned Lark, Water Pipit, Lapland Longspur, and Snow Bunting on Bylot Island, Northwest Territories, Canada. Bird-Banding, 32(1), 1–46. https://doi.org/10.2307/4510843

Du, Y., Zhang, Y., Ling, F., Wang, Q., Li, W., & Li, X. (2016). Water Bodies' Mapping from Sentinel-2 Imagery with Modified Normalized Difference Water Index at 10-m Spatial Resolution

Produced by Sharpening the SWIR Band. Remote Sensing, 8(4), Article 4. https://doi.org/10.3390/rs8040354

Duijns, S., Niles, L. J., Dey, A., Aubry, Y., Friis, C., Koch, S., Anderson, A. M., & Smith, P. A. (2017). Body condition explains migratory performance of a long-distance migrant. Proceedings of the Royal Society B: Biological Sciences, 284(1866). https://doi.org/10.1098/rspb.2017.1374

Earle, J. C., & Kershaw, K. A. (1989). Vegetation patterns in James Bay coastal marshes. III. Salinity and elevation as factors influencing plant zonations. Canadian Journal of Botany, 67(10), 2967–2974. https://doi.org/10.1139/b89-380

Environment and Climate Change Canada. (2019, November 14). Kendall Island Migratory Bird Sanctuary [Program descriptions;education and awareness]. Kendall Island Migratory Bird Sanctuary. https://www.canada.ca/en/environment-climate-change/services/migratory-bird-sanctuaries/locations/kendall-island.html

Forbes, D. L., & Frobel, D. (1985). Coastal erosion and sedimentation in the Canadian Beaufort Sea (Paper 85-1B; Current Research, Part B, pp. 69–80). Geological Survey of Canada. https://geoscan.nrcan.gc.ca/starweb/geoscan/servlet.starweb?path=geoscan/fulle.web&search1=R=120220

Ganter, B. & Gaston, A.J. (2013). Birds. In Meltofte, H. (Ed.), Arctic Biodiversity Assessment (pp. 142–181). Convention on Arctic Flora and Fauna (CAFF). https://www.academia.edu/62645493/Ganter_B_and_Gaston_A_J_2013_Birds_In_Meltofte_H_ed_Arctic_Biodiversity_Assessment_The_Conservation_of_Arctic_Flora_and_Fauna_CAFF_Akureyri_Iceland_Pages_142_181

Gaston, A. J., Gilchrist, H. G., & Hipfner, J. M. (2005). Climate change, ice conditions and reproduction in an Arctic nesting marine bird: Brunnich's guillemot (Uria lomvia L.). Journal of Animal Ecology, 74(5), 832–841. https://doi.org/10.1111/j.1365-2656.2005.00982.x

Gratto-Trevor, C. L. (1996). Use of Landsat TM Imagery in Determining Important Shorebird Habitat in the Outer Mackenzie Delta, Northwest Territories. ARCTIC, 49(1), 11–22. https://doi.org/10.14430/arctic1179

Gratto-Trevor, C. L., Johnston, V. H., & Pepper, S. T. (1998). Changes in Shorebird and Eider Abundance in the Rasmussen Lowlands, NWT. The Wilson Bulletin, 110(3), 316–325.

Harper, J. R., Henry, R. F., & Stewart, G. G. (1988). Maximum Storm Surge Elevations in the Tuktoyaktuk Region of the Canadian Beaufort Sea. Arctic, 41(1), 48–52.

Harris, D. L. (1963). Characteristics of the hurricane storm surge (Technical Paper No. 48; p. 139). U.S. Dept. Comm., Weath. Bur. https://web.archive.org/web/20130516215010/http:/www.csc.noaa.gov/hes/images/pdf/CHARACTERISTICS_STORM_SURGE.pdf

Hitch, A. T., & Leberg, P. L. (2007). Breeding Distributions of North American Bird Species Moving North as a Result of Climate Change. Conservation Biology, 21(2), 534–539. https://doi.org/10.1111/j.1523-1739.2006.00609.x

Hitchcock, C. L., & Gratto-Trevor, C. (1997). Diagnosing a Shorebird Local Population Decline with a Stage-Structured Population Model. Ecology, 78(2), 522–534. https://doi.org/10.1890/0012-9658(1997)078[0522:DASLPD]2.0.CO;2

Höhn, E. O. (1968). Some observations on the breeding of northern phalaropes at Scammon Bay, Alaska. The Auk, 85(2), 316–317.

Iacobelli, A., & Jefferies, R. L. (1991). Inverse Salinity Gradients in Coastal Marshes and the Death of Stands of Salix: The Effects of Grubbing by Geese. Journal of Ecology, 79(1), 61–73. https://doi.org/10.2307/2260784

Intergovernmental Panel on Climate Change (IPCC). (2013). AR5 Climate Change 2013: The Physical Science Basis. https://www.ipcc.ch/report/ar5/wg1/

Jehl, J. R., JR. (2007). Disappearance of Breeding Semipalmated Sandpipers from Churchill, Manitoba: More than a Local Phenomenon. The Condor, 109(2), 351–360. https://doi.org/10.1093/condor/109.2.351

Killengreen, S. T., Ims, R. A., Yoccoz, N. G., Bråthen, K. A., Henden, J.-A., & Schott, T. (2007). Structural characteristics of a low Arctic tundra ecosystem and the retreat of the Arctic fox. Biological Conservation, 135(4), 459–472. https://doi.org/10.1016/j.biocon.2006.10.039

Kim, J., Murphy, E., Nistor, I., Ferguson, S., & Provan, M. (2021). Numerical Analysis of Storm Surges on Canada's Western Arctic Coastline. Journal of Marine Science and Engineering, 9(3), Article 3. https://doi.org/10.3390/jmse9030326

Klaassen, M., Hoye, B. J., Nolet, B. A., & Buttemer, W. A. (2012). Ecophysiology of avian migration in the face of current global hazards. Philosophical Transactions of the Royal Society B: Biological Sciences, 367(1596), 1719–1732. https://doi.org/10.1098/rstb.2012.0008

Knick, S. T., & Rotenberry, J. T. (2000). Ghosts of Habitats Past: Contribution of Landscape Change to Current Habitats Used by Shrubland Birds. Ecology, 81(1), 220–227. https://doi.org/10.1890/0012-9658(2000)081[0220:GOHPCO]2.0.CO;2

Kokelj, S. V., Lantz, T. C., Solomon, S., Pisaric, M. F. J., Keith, D., Morse, P., Thienpont, J. R., Smol, J. P., & Esagok, D. (2012). Using Multiple Sources of Knowledge to Investigate Northern Environmental Change: Regional Ecological Impacts of a Storm Surge in the Outer Mackenzie Delta, N.W.T. Arctic; Calgary, 65(3), 257–272.

Kölzsch, A., Bauer, S., de Boer, R., Griffin, L., Cabot, D., Exo, K.-M., van der Jeugd, H. P., & Nolet, B. A. (2015). Forecasting spring from afar? Timing of migration and predictability of phenology along different migration routes of an avian herbivore. Journal of Animal Ecology, 84(1), 272–283.

Lantz, T. C., Kokelj, S. V., & Fraser, R. H. (2015). Ecological recovery in an Arctic delta following widespread saline incursion. Ecological Applications, 25(1), 172–185. https://doi.org/10.1890/14-0239.1

Lappo, E., Tomkovich, P., Syroechkovskiy, E., & Zharikov, Y. V. (2012). Atlas of breeding waders in the Russian Arctic.

Latour, P. B., Machtans, C. S., & Beyersbergen, G. W. (2005). Shorebird and Passerine Abundance and Habitat Use at a High Arctic Breeding Site: Creswell Bay, Nunavut. ARCTIC, 58(1), 55–65. https://doi.org/10.14430/arctic389

Liebezeit, J. R., Kendall, S. J., Brown, S., Johnson, C. B., Martin, P., McDonald, T. L., Payer, D. C., Rea, C. L., Streever, B., Wildman, A. M., & Zack, S. (2009). Influence of human development and predators on nest survival of tundra birds, Arctic Coastal Plain, Alaska. Ecological Applications, 19(6), 1628–1644. https://doi.org/10.1890/08-1661.1

MacKay, J. R. (1963). The Mackenzie Delta area, N.W.T. R. Duhamel, Queen's Printer and Controller of Stationery.

Manson, G. K., & Solomon, S. M. (2007). Past and future forcing of Beaufort Sea coastal change. Atmosphere-Ocean, 45(2), 107–122. https://doi.org/10.3137/ao.450204

McGarigal, K., Stafford, S., & Cushman, S. (2000). Multivariate Statistics for Wildlife and Ecology Research. Springer. https://doi.org/10.1007/978-1-4612-1288-1

McKeague, J. A. (1978). Manual on soil sampling and methods of analysis. Canadian Society of Soil Science.

McKinnon, L., Picotin, M., Bolduc, E., Juillet, C., & Bêty, J. (2012). Timing of breeding, peak food availability, and effects of mismatch on chick growth in birds nesting in the High Arctic. Canadian Journal of Zoology, 90, 961–971. https://doi.org/10.1139/z2012-064

Møller, A. P., Rubolini, D., & Lehikoinen, E. (2008). Populations of migratory bird species that did not show a phenological response to climate change are declining. Proceedings of the

National Academy of Sciences of the United States of America, 105(42), 16195–16200. https://doi.org/10.1073/pnas.0803825105

Morrison, C. A., Robinson, R. A., Clark, J. A., Risely, K., & Gill, J. A. (2013). Recent population declines in Afro-Palaearctic migratory birds: The influence of breeding and non-breeding seasons. Diversity and Distributions, 19(8), 1051–1058. https://doi.org/10.1111/ddi.12084

Murphy, S. M. (1981). Habitat Use by Migrating and Breeding Shorebirds on the Eastern Copper River Delta, Alaska. [M.S., University of Alaska Fairbanks]. https://www.proquest.com/docview/303005378/citation/AF3D8F2045594514PQ/1

Mustin, K., Sutherland, W., & Gill, J. (2007). The complexity of predicting climate-induced ecological impacts. Climate Research, 35, 165–175. https://doi.org/10.3354/cr00723

Nguyen, L. P., Nol, E., & Abraham, K. F. (2003). NEST SUCCESS AND HABITAT SELECTION OF THE SEMIPALMATED PLOVER ON AKIMISKI ISLAND, NUNAVUT. The Wilson Bulletin, 115(3), 285–291. https://doi.org/10.1676/03-044

Nguyen, L. P., Nol, E., Abraham, K. F., & Lishman, C. (2013). Directional selection and repeatability in nest-site preferences of Semipalmated Plovers (Charadrius semipalmatus). Canadian Journal of Zoology, 91(9), 646–652. https://doi.org/10.1139/cjz-2013-0064

Nicholls, R. J., & Cazenave, A. (2010). Sea-Level Rise and Its Impact on Coastal Zones. Science, 328(5985), 1517–1520. JSTOR.

NWT Centre for Geomatics. (2007). Mackenzie Valley Air Photo Project 2004 [Map]. https://www.imagetest.geomatics.gov.nt.ca/arcgis/rest/services/Mosaics/MVAP_Mosaic_Combined_LCC/MapServer/info/iteminfo

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P. R., O'hara, R. B., Simpson, G. L., & Solymos, P. (2015). Vegan: Community ecology package. 2019. R Package Version, 2(10).

Overeem, I., Anderson, R. S., Wobus, C. W., Clow, G. D., Urban, F. E., & Matell, N. (2011). Sea ice loss enhances wave action at the Arctic coast. Geophysical Research Letters, 38(17). https://doi.org/10.1029/2011GL048681

Pearce, C. M. (1986). The Distribution and ecology of the shoreline vegetation on the Mackenzie Delta, N.W.T. [Doctoral thesis, University of Calgary]. https://doi.org/10.11575/PRISM/18346

Pearce, T., Smit, B., Duerden, F., Ford, J. D., Goose, A., & Kataoyak, F. (2010). Inuit vulnerability and adaptive capacity to climate change in Ulukhaktok, Northwest Territories, Canada. Polar Record, 46(2), 157–177. https://doi.org/10.1017/S0032247409008602

Pétillon, J., Georges, A., Canard, A., Lefeuvre, J.-C., Bakker, J. P., & Ysnel, F. (2008). Influence of abiotic factors on spider and ground beetle communities in different salt-marsh systems. Basic and Applied Ecology, 9(6), 743–751. https://doi.org/10.1016/j.baae.2007.08.007

Piersma, T., Lok, T., Chen, Y., Hassell, C. J., Yang, H.-Y., Boyle, A., Slaymaker, M., Chan, Y.-C., Melville, D. S., Zhang, Z.-W., & Ma, Z. (2016). Simultaneous declines in summer survival of three shorebird species signals a flyway at risk. Journal of Applied Ecology, 53(2), 479–490. https://doi.org/10.1111/1365-2664.12582

Pirie, L., & Johnston, V. (2012). Survey Methods for Whimbrel. In V. Johnston, J. Bart, P. A. Smith, & J. Rausch (Eds.), Arctic Shorebirds in North America (1st ed., pp. 177–184). University of California Press; JSTOR. http://www.jstor.org.ezproxy.library.uvic.ca/stable/10.1525/j.ctt1ppt1c.14

Pirie, L., Johnston, V., & Smith, P. A. (2012). Tier 2 Surveys. In V. Johnston, P. A. Smith, J. Bart, & J. Rausch (Eds.), Arctic Shorebirds in North America (1st ed., pp. 185–194). University of California Press; JSTOR. http://www.jstor.org.ezproxy.library.uvic.ca/stable/10.1525/j.ctt1ppt1c.15

Pisaric, M. F. J., Thienpont, J. R., Kokelj, S. V., Nesbitt, H., Lantz, T. C., Solomon, S., & Smol, J. P. (2011). Impacts of a recent storm surge on an Arctic delta ecosystem examined in the context of the last millennium. Proceedings of the National Academy of Sciences of the United States of America, 108(22), 8960–8965. https://doi.org/10.1073/pnas.1018527108

Post, E., Forchhammer, M. C., Bret-Harte, M. S., Callaghan, T. V., Christensen, T. R., Elberling, B., Fox, A. D., Glig, O., Hik, D. S., Høye, T. T., Ims, R. A., Jeppesen, E., Klein, D. R., Madsen, J., McGuire, A. D., Rysgaard, S., Schindler, D. E., Stirling, I., Tamstorf, M. P., … Aastrup, P. (2009). Ecological Dynamics across the Arctic Associated with Recent Climate Change. Science, 325(5946), 1355–1358. JSTOR.

R Core Team. (2020). R: a language and environment for statistical computing. Foundation for Statistical Computing.

Raynolds, M. K., Walker, D. A., Epstein, H. E., Pinzon, J. E., & Tucker, C. J. (2012). A new estimate of tundra-biome phytomass from trans-Arctic field data and AVHRR NDVI. Remote Sensing Letters, 3(5), 403–411. https://doi.org/10.1080/01431161.2011.609188

Recher, H. F. (1966). Some Aspects of the Ecology of Migrant Shorebirds. Ecology, 47(3), 393–407. https://doi.org/10.2307/1932979

Reed, E. T., Kardynal, K. J., Horrocks, J. A., & Hobson, K. A. (2018). Shorebird hunting in Barbados: Using stable isotopes to link the harvest at a migratory stopover site with sources of production. The Condor, 120(2), 357–370. https://doi.org/10.1650/CONDOR-17-127.1

Riedel, S. M., Epstein, H. E., & Walker, D. A. (2005). Biotic controls over spectral reflectance of arctic tundra vegetation. International Journal of Remote Sensing, 26(11), 2391–2405. https://doi.org/10.1080/01431160512331337754

Robinson, R. A., Clark, N. A., Lanctot, R., Nebel, S., Harrington, B., Clark, J. A., Gill, J. A., Meltofte, H., Rogers, D. I., Rogers, K. G., Ens, B. J., Reynolds, C. M., Ward, R. M., Piersma, T., & Atkinson, P. W. (2005). Long term demographic monitoring of wader populations in non-breeding areas.

Rockwell, R. F., Witte, C. R., Jefferies, R., & Weatherhead, P. J. (2003). Response of nesting savannah sparrows to 25 years of habitat change in a snow goose colony. Écoscience, 10(1), 33–37. https://doi.org/10.1080/11956860.2003.11682747

Romañach, S. S., Beerens, J. M., Patton, B. A., Chapman, J. P., & Hanson, M. R. (2019). Impacts of Saltwater Intrusion on Wetland Prey Production and Composition in a Historically Freshwater Marsh. Estuaries and Coasts, 42(6), 1600–1611. https://doi.org/10.1007/s12237-019-00572-8

Rosenberg, K. V., Dokter, A. M., Blancher, P. J., Sauer, J. R., Smith, A. C., Smith, P. A., Stanton, J. C., Panjabi, A., Helft, L., Parr, M., & Marra, P. P. (2019). Decline of the North American avifauna. Science, 366(6461), 120–124. https://doi.org/10.1126/science.aaw1313

Saino, N., Ambrosini, R., Rubolini, D., von Hardenberg, J., Provenzale, A., Hüppop, K., Hüppop, O., Lehikoinen, A., Lehikoinen, E., Rainio, K., Romano, M., & Sokolov, L. (2011). Climate warming, ecological mismatch at arrival and population decline in migratory birds. Proceedings of the Royal Society B: Biological Sciences, 278(1707), 835–842. https://doi.org/10.1098/rspb.2010.1778

Scharffenberg, K. C., MacPhee, S. A., & Loseto, L. L. (2020). Upriver sightings of beluga whales (Delphinapterus leucas) follow storm surges and high water in the Mackenzie Delta, Northwest Territories, Canada. Arctic Science, 1–11. https://doi.org/10.1139/as-2020-0010

Scharffenberg, K. C., Whalen, D., MacPhee, S. A., Marcoux, M., Iacozza, J., Davoren, G., & Loseto, L. L. (2020). Oceanographic, ecological, and socio-economic impacts of an unusual summer storm in the Mackenzie Estuary. Arctic Science, 6(2), 62–76. https://doi.org/10.1139/as-2018-0029

Senner, N. R., Hochachka, W. M., Fox, J. W., & Afanasyev, V. (2014). An Exception to the Rule: Carry-Over Effects Do Not Accumulate in a Long-Distance Migratory Bird. PLOS ONE, 9(2), e86588. https://doi.org/10.1371/journal.pone.0086588

Senner, N. R., Stager, M., & Sandercock, B. K. (2017). Ecological mismatches are moderated by local conditions for two populations of a long-distance migratory bird. Oikos, 126(1), 61–72. https://doi.org/10.1111/oik.03325

Sepp, M., & Jaagus, J. (2011). Changes in the activity and tracks of Arctic cyclones. Climatic Change, 105(3), 577–595. https://doi.org/10.1007/s10584-010-9893-7

Serreze, M. C., Holland, M. M., & Stroeve, J. (2007). Perspectives on the Arctic's Shrinking Sea-Ice Cover. Science, 315(5818), 1533–1536. JSTOR.

Simmonds, I., & Keay, K. (2009). Extraordinary September Arctic sea ice reductions and their relationships with storm behavior over 1979–2008. Geophysical Research Letters, 36(19). https://doi.org/10.1029/2009GL039810

Smith, A. C., & Nol, E. (2000). WINTER FORAGING BEHAVIOR AND PREY SELECTION OF THE SEMIPALMATED PLOVER IN COASTAL VENEZUELA. The Wilson Bulletin, 112(4), 467–472. https://doi.org/10.1676/0043-5643(2000)112[0467:WFBAPS]2.0.CO;2

Smith, P. A., McKinnon, L., Meltofte, H., Lanctot, R. B., Fox, A. D., Leafloor, J. O., Soloviev, M., Franke, A., Falk, K., Golovatin, M., Sokolov, V., Sokolov, A., & Smith, A. C. (2020). Status and trends of tundra birds across the circumpolar Arctic. Ambio, 49(3), 732–748. https://doi.org/10.1007/s13280-019-01308-5

Smith, P. A., Smith, A. C., Andres, B., Francis, C. M., Harrington, B., Friis, C., Morrison, R. I. G., Paquet, J., Winn, B., & Brown, S. (2023). Accelerating declines of North America's shorebirds signal the need for urgent conservation action. Ornithological Applications, duad003. https://doi.org/10.1093/ornithapp/duad003

Solomon, S. (2005). Spatial and temporal variability of shoreline change in the Beaufort-Mackenzie region, northwest territories, Canada. Geo-Marine Letters, 25(2), 127–137. https://doi.org/10.1007/s00367-004-0194-x

Solomon, S., Forbes, D.L., & Kierstead, Brian. (1994). Coastal impacts of climate change: Beaufort Sea erosion study (Geological Survey of Canada). Natural Resources Canada. https://publications.gc.ca/site/eng/9.820125/publication.html

Studds, C. E., Kendall, B. E., Murray, N. J., Wilson, H. B., Rogers, D. I., Clemens, R. S., Gosbell, K., Hassell, C. J., Jessop, R., Melville, D. S., Milton, D. A., Minton, C. D. T., Possingham, H. P., Riegen, A. C., Straw, P., Woehler, E. J., & Fuller, R. A. (2017). Rapid population decline in migratory shorebirds relying on Yellow Sea tidal mudflats as stopover sites. Nature Communications, 8(1), Article 1. https://doi.org/10.1038/ncomms14895

Thienpont, J. R., C., S., Vermaire, J., Pisaric, M., Kokelj, S., & Smol, J. (2015). Synchronous changes in chironomid assemblages following a major saltwater intrusion in two coastal Arctic lake ecosystems. Journal of Paleolimnology, 53, 177–189.

Thienpont, J. R., Johnson, D., Nesbitt, H., Kokelj, S. V., Pisaric, M. F., & Smol, J. P. (2012). Arctic coastal freshwater ecosystem responses to a major saltwater intrusion: A landscape-scale palaeolimnological analysis. The Holocene, 22(12), 1451–1460. https://doi.org/10.1177/0959683612455538

Thompson, A., & Millar, N. (2007). Traditional knowledge of fish migration and spawning patterns in Tsiigehnjik (Arctic Red River) and Nagwichoonjik (Mackenzie River), Northwest Territories (No. 07–01). Gwich'in Renewable Resource Board.

Tucker, C. J. (1979). Red and photographic infrared linear combinations for monitoring vegetation. Remote Sensing of Environment, 8(2), 127–150. https://doi.org/10.1016/0034-4257(79)90013-0

Tulp, I., Schekkerman, H., Klaassen, R. H. G., Ens, B. J., & Visser, G. H. (2009). Body condition of shorebirds upon arrival at their Siberian breeding grounds. Polar Biology, 32(3), 481–491. https://doi.org/10.1007/s00300-008-0543-8

Usher, P. J. (2000). Traditional Ecological Knowledge in Environmental Assessment and Management. Arctic, 53(2), 183–193. JSTOR.

van Gils, J. A., Lisovski, S., Lok, T., Meissner, W., Ożarowska, A., de Fouw, J., Rakhimberdiev, E., Soloviev, M. Y., Piersma, T., & Klaassen, M. (2016). Body shrinkage due to Arctic warming reduces red knot fitness in tropical wintering range. Science, 352(6287), 819–821. https://doi.org/10.1126/science.aad6351

Vermaire, J. C., Pisaric, M. F. J., Thienpont, J. R., Mustaphi, C. J. C., Kokelj, S. V., & Smol, J. P. (2013). Arctic climate warming and sea ice declines lead to increased storm surge activity. Geophysical Research Letters, 40(7), 1386–1390. https://doi.org/10.1002/grl.50191

Wauchope, H. S., Shaw, J. D., Varpe, Ø., Lappo, E. G., Boertmann, D., Lanctot, R. B., & Fuller, R. A. (2017). Rapid climate-driven loss of breeding habitat for Arctic migratory birds. Global Change Biology, 23(3), 1085–1094. https://doi.org/10.1111/gcb.13404

Waugh, D., Pearce, T., Ostertag, S. K., Pokiak, V., Collings, P., & Loseto, L. L. (2018). Inuvialuit traditional ecological knowledge of beluga whale (Delphinapterus leucas) under changing climatic conditions in Tuktoyaktuk, NT. Arctic Science, 1–17. https://doi.org/10.1139/as-2017-0034

Weiser, E. L., Lanctot, R. B., Brown, S. C., Gates, H. R., Bentzen, R. L., Bêty, J., Boldenow, M. L., English, W. B., Franks, S. E., Koloski, L., Kwon, E., Lamarre, J.-F., Lank, D. B., Liebezeit, J. R., McKinnon, L., Nol, E., Rausch, J., Saalfeld, S. T., Senner, N. R., … Sandercock, B. K. (2018). Environmental and ecological conditions at Arctic breeding sites have limited effects on true survival rates of adult shorebirds. The Auk, 135(1), 29–43. https://doi.org/10.1642/AUK-17-107.1

Wennerberg, L., Klaassen, M., & Lindström, Å. (2002). Geographical Variation and Population Structure in the White-Rumped Sandpiper Calidris fuscicollis as Shown by Morphology, Mitochondrial DNA and Carbon Isotope Ratios. Oecologia, 131(3), 380–390.

Wickham, H. (2016). Data Analysis. In H. Wickham (Ed.), Ggplot2: Elegant Graphics for Data Analysis (pp. 189–201). Springer International Publishing. https://doi.org/10.1007/978-3-319-24277-4_9

Wiens, J. A., Rotenberry, J. T., & Van Horne, B. (1986). A Lesson in the Limitations of Field Experiments: Shrubsteppe Birds and Habitat Alteration. Ecology, 67(2), 365–376. https://doi.org/10.2307/1938579

Yang, D., Shi, X., & Marsh, P. (2015). Variability and extreme of Mackenzie River daily discharge during 1973–2011. Quaternary International, 380–381, 159–168. https://doi.org/10.1016/j.quaint.2014.09.023

4 Conclusion

4.1 Study Synthesis

The objective of my thesis was to evaluate the long-term (> 20 years) effects of saline incursion on ecosystems in the outer Mackenzie Delta. Specifically, I conducted field surveys of plants (Chapter 2) and birds (Chapter 3) in the outer Mackenzie Delta (Northwest Territories) to investigate effects of disturbance in the context of a changing climate and to improve our understanding of recovery in this coastal Arctic ecosystem.

In Chapter 2, I used a combination of remote sensing and field-based soil & vegetation surveys to determine whether repeated disturbance from two major storm surges influences soil & vegetation communities. In this study area, I found that the 1999 storm caused more significant and persistent vegetation die-off than the 2016 storm. In my exploration of factors influencing recovery, the distance of the site to its nearest channel was the most important factor and was negatively correlated with revegetation. In comparing the 1X and 2X affected Revegetated sites to each other, I found that – although both site types were similar in the measures of NDVI in 2019 and total vegetation cover – 1X affected sites had vegetation communities with relatively even abundance of all plant types (shrubs, herbaceous, graminoids, and moss), whereas the 2X affected sites were dominated by graminoids. Salinity in Revegetated sites was higher in 2X than 1X (but not significantly so), whereas in Unvegetated sites it was lower in 2X than 1X (but not significantly so). Overall, soil salinity was higher in both types of Unvegetated sites than in both Revegetated site types. My results demonstrate that although both the 1999 and 2016 storm were large in magnitude, they differed in how significantly they affected the vegetation community of the outer delta. This adds more circumstantial support to the untested theory that widespread and persistent vegetation death following storm surges results when large storms occur late in the season when the ground is about to freeze (Kokelj et al. 2012, Lantz et al. 2015). My work also suggests that recurrent disturbance can affect the composition of the vegetation community that recovers post-disturbance, but further study is needed to verify these patterns.

In Chapter 3, I surveyed birds across affected and unaffected areas of the outer delta, and I found that bird community composition and occurrence of focal species is driven by the persistent effects of the 1999 storm. Revegetated sites supported a similar bird community to unaffected (Reference) sites but affected sites with little vegetation (Unvegetated) had a significantly different community of birds than unaffected (Reference) sites. In my study, both avian community composition as well as the occurrences of focal species were correlated with vegetation productivity gradients. My results suggest that a lack of ground cover and surface water at Unvegetated sites made them unsuitable for species that occupy aquatic habitats, such as red-necked phalaropes and ducks, or species that require materials for camouflaging their nests, such as savannah sparrows – all of which were almost exclusively found in vegetated sites (includes unaffected Reference sites and affected Revegetated sites). In contrast, Unvegetated sites were characterized by bare ground and the absence of surface water and were dominated by lapland longspur and semipalmated plover along with various predatory birds (such as jaegers and hawks). Avian community composition, focal species' identities, and habitat conditions were functionally similar between Reference (unaffected) and Revegetated (affected) sites. This indicates that revegetation 20 years following the storm surge has been adequate to restore habitat functionality for certain avian species at some sites. However, my work also suggests that areas persistently affected by the terrestrial salinization following the 1999 storm have an altered avian community composition.

Overall, my thesis demonstrates that projected increases in frequency of major storm surge salinization events like that of 1999 (Manson & Solomon 2007) have the potential to drastically alter the vegetation and avian communities in affected areas for decades post-disturbance. My findings from Chapter 2 show that persistent effects of the 1999 storm are still apparent two decades post-storm and that recurrent storm surges in the same area have the potential to limit vegetation community diversity. Chapter 3 shows that the persistent effects of the 1999 storm on vegetation is directly correlated to differences in the avian community and the species that can use these affected areas as habitat.

4.2 Limitations and Future Research Opportunities

My thesis research makes an important contribution to our understanding of how climate change-induced alterations to disturbance regimes will alter Arctic vegetation and avifauna. However, there are several limitations inherent in my research project that provide opportunities for future research. In Chapter 2, I surveyed soil properties & vegetation community composition at sites in the outer Mackenzie Delta affected by large magnitude storm surges in 1999 and 2016. I selected field sites from five site types I classified based on information derived from Landsat imagery, but I was limited by the resolution of the Landsat Imagery (1 pixel = 30m^2). Since Landsat imagery is too coarse to distinguish among vegetation cover types, I could not pre-select vegetated sites that had a variety of plant cover types. Additionally, due to the coarse resolution, only areas in larger (>30x30m) patches of unvegetated area showed up in my site selection process. Therefore, there could have been some more variation in Unvegetated sites that I couldn't identify. My data collection was also limited by the number of field sites we could feasibly get to within the short summer field season (~ one month). In total, we were able to visit 32 sites. We did take multiple measurements at each site, but when these values were averaged to control for pseudo-replication, overall statistical power was low. Because I used categorical site selection, the conclusions from my study can be used to inform patterns in how similar areas react to storm surge disturbance, but given more time and resources, randomly sampled site locations across the study area would have likely provided data with greater varieties of plant communities and environmental factors. Given the type of categorical sampling I used, there are likely habitat conditions along continuous gradients that were missed. Because we do not have data along a continuous gradient of environmental factors, we cannot directly compare our vegetation communities to these factors and extrapolate the correlations to a larger Arctic region. Instead, we can highlight patterns in environmental factors that are consistent with patterns found in other factors across the grouped site types. This method of comparison was necessary due to the amount of time and resources we had access to, but it did limit the potential for finding correlations due to our low statistical power. Longer-term monitoring is required to quantify ongoing recovery patterns of the outer delta and understand the effects of changing storm surge disturbance on the response of both floral and faunal communities in the outer Mackenzie Delta.

In Chapter 3, I surveyed all species of birds present at three site types within and outside the area affected by the 1999 storm, but the length of my survey period and the number of sites we could visit was limited due to time, financial resources, and the logistics of accessing remote sites by boat. Birds cue in on a variety of attributes when selecting breeding and staging territories, meaning that additional surveys within the study area could have provided more data on a greater variety of sites. For example, while most Unvegetated sites were drier than vegetated site types, there was one Unvegetated site (MX_3) that contained a wide but shallow pond which resulted in a much richer variety of birds being present during our sampling. That wetter Unvegetated site was the only outlier among the other sites in its group, but it is possible that if we could have sampled more sites, we would have found more similarly wet Unvegetated sites. Furthermore, because we sampled over a short period of time, we cannot know how the ephemeral nature of some of these ponds would affect our sampling and the results of our comparisons. Secondly, the method of site selection, though necessary to allow analyses with fewer data, limited the types of questions that I could ask and constrained the variety of habitat conditions (ie. soil moisture, surface water, vegetation community) to which we could compare our avian species occurrences. Finally, the very short window (~ two weeks) in which the surveys took place was not ideal because migration and staging patterns can change year to year depending on shifts in the onset of spring and timing of ice break-up (Pirie, Johnston & Smith 2012). Additionally, the PRISM protocol (which we used in this study) identifies an earlier survey window (late May & June) for monitoring breeding shorebirds since some earlier breeders begin southward migration soon after nesting (Pirie & Johnston 2012). Due to logistical reasons for reaching our field sites, we were unable to access the sites by boat prior to ice break-up on the channels. The late and short nature of my field season leaves uncertainty in the variation of bird species I observed in a single ~ two-week period, however since my research is (by my knowledge) the only study on the effects of saline incursion on birds in the western Canadian Arctic, this study still contributes a baseline for future studies.

Although the bird communities at vegetated and unvegetated sites differed, some individual species did not show significant differences along gradients of the site-characteristic habitat factors (ie. vegetation, moisture, distance from nearest channel). It is possible that we did not measure all factors relevant to those species. Avian biodiversity studies often underestimate the natural dynamism of bird community assemblages, overemphasizing the importance of environmental variables and underestimating the effect of time/season (Curtis et al. 2016; Collins 2001). More study of these factors over a longer time period is required to assess how persistent vegetation change from storm surges will affect these less abundant species. Since there is not a complete historical record informed by historical surveys in our study area, we also assumed that all three site types were similar in terms of vegetation structure before the 1999 storm. We used air photos and NDVI values to conduct site matching, but this limited our ability to ensure that all sites had uniform measures of community composition, canopy structure, and height prior to the storm.

Although we found clear, persistent effects of vegetation change on bird communities, we were unable to identify specific drivers of the patterns due to the nature of our categorical sampling method. Studies focusing on habitat around the “Revegetated” threshold (NDVI ~ 0.25) and across continuous gradients of environmental factors would provide more insight into the consistent drivers of individual species’ presences/absences in these habitat types. In particular, shorebird habitat preference often depends on subtle differences in the “understory” vegetation and water levels (Gratto-Trevor 1996). Greater variety in the sampling method (for example random point selection across the study area instead of selecting for specific NDVI values) would provide data more suited for including in individual species distribution or predictive modelling scenarios. Sampling at points along a gradient of an environmental factor of particular interest would allow direct analysis of correlations between environmental variables and bird species' presence or other habitat factors. More research on factors that have the greatest effects on habitat selection by avifauna would also allow more precise application of fine-scale remote-sensing products.

Furthermore, the importance of prey availability and site fidelity were not assessed during our study but are other potential drivers of the responses to environmental change in bird communities (Romañach et al. 2019; Knick and Rotenberry 2000, Hitch and Leberg 2007, Devitcor et al. 2008). Firstly, declines in avian species with high site fidelity are already being seen in the Arctic (van Gils, 2016). Prior to conducting our research, we had conceded the potential effect of species site fidelity on their presence/absence in lower quality habitats but could not isolate this effect with our study design. Previous studies have also looked at the association of insect emergence and shorebird hatchling success in other areas of the Canadian Arctic (Bolduc et al. 2013, McKinnon et al. 2012), but prey availability in this area has not yet been assessed in the context of saline incursion. Previous research in other areas in combination with post-storm monitoring suggests that the salinization event after the 1999 storm surge likely affected the local arthropod community, but no studies have directly investigated this to date (Desender & Maelfait 1999; Pétillon et al. 2008; Boelman et al. 2015). Moreover, since both site fidelity and mismatches in phenology of prey availability are likely species-specific, more focussed studies in the context of saline incursion are required to evaluate their importance for avian habitat suitability. Additionally, previous studies have already identified the need for more surveys of North American birds to take place outside of the breeding season (Rosenberg et al. 2019). Such surveys will provide more information about the conditions avian species encounter throughout their life cycles and potentially limiting factors outside the breeding season. Lastly, water levels change year-to-year (monthly in the summer) in the active outer delta (Gratto-Trevor 1996). Improved regular monitoring of water levels closer to coastal breeding areas could provide a clearer picture of how annual storm surge patterns will affect birds. So far, most catastrophic storm surge events in the Mackenzie Delta have only been studied (in terms of the scientific community) coincidentally; from monitors set up for another study (Scharffenberg et al., 2020), or many years after the original event (Lantz et al. 2015; Deasley et al. 2012; Kokelj et al. 2012; Pisaric et al. 2011; Thienpont et al. 2012, 2015). Though we do not have complete historical accounts, this monitoring could aid in determining the factors influencing the magnitude of habitat salinization after storm surges and provide better baseline data for future comparisons. With further study of the threshold conditions for habitat suitability and regular monitoring of storm surge activity, we could predict the severity a storm event could have on available breeding territory. This would help to inform management decision in a more proactive – rather than reactive – manner.

Long-term monitoring datasets are crucial for understanding the response of sensitive ecosystems to changing environmental conditions. Research in these areas not only contributes to our technical understanding of their biophysical processes but is extremely important for policy and decision making at a broader scale. My thesis research contributes to the literature on Arctic deltaic environments, their bird and plant communities, disturbance regimes, and soil conditions. It also provides novel baseline information that can be used as a reference point for future changes in Arctic landscapes. Continued multi-scale monitoring is still required to improve our understanding and ability to predict community recovery and long-term ecological dynamics following storm surge disturbances.

4.3 References

Boelman, N. T., Gough, L., Wingfield, J., Goetz, S., Asmus, A., Chmura, H. E., ... & Guay, K. C. (2015). Greater shrub dominance alters breeding habitat and food resources for migratory songbirds in Alaskan arctic tundra. Global Change Biology, 21(4), 1508-1520.

Bolduc, E., Casajus, N., Legagneux, P., McKinnon, L., Gilchrist, H. G., Leung, M., ... & Bêty, J. (2013). Terrestrial arthropod abundance and phenology in the Canadian Arctic: modelling resource availability for Arctic-nesting insectivorous birds. The Canadian Entomologist, 145(2), 155-170.

Collins, S. L. (2001). Long-term research and the dynamics of bird populations and communities. The Auk, 118(3), 583-588.

Curtis, J. R., Robinson, W. D., & McCune, B. (2016). Time trumps habitat in the dynamics of an avian community. Ecosphere, 7(11), e01575.

Deasley, K., Korosi, J. B., Thienpont, J. R., Kokelj, S. V., Pisaric, M. F. J., & Smol, J. P. (2012). Investigating the response of cladocera to a major saltwater intrusion event in an arctic lake from the outer mackenzie delta (NT, canada). Journal of Paleolimnology, 48(2), 287-296. doi:10.1007/s10933-012-9577-6

Desender, K., & Maelfait, J. P. (1999). Diversity and conservation of terrestrial arthropods in tidal marshes along the River Schelde: a gradient analysis. *Biological Conservation*, *87*(2), 221-229.

Devictor, V., Julliard, R., Couvet, D., & Jiguet, F. (2008). Birds are tracking climate warming, but not fast enough. Proceedings of the Royal Society B: Biological Sciences, 275(1652), 2743-2748.

van Gils JA, Lisovski S, Lok T et al. (2016) Body shrinkage due to Arctic warming reduces red knot fitness in tropical wintering range. Science, 352, 819–821.

Gratto-Trevor, C. L. (1996). Use of Landsat TM imagery in determining important shorebird habitat in the outer Mackenzie Delta, Northwest Territories. Arctic, 11-22.

Hitch, A. T., & Leberg, P. L. (2007). Breeding distributions of North American bird species moving north as a result of climate change. Conservation Biology, 21(2), 534-539.

Knick, S. T., & Rotenberry, J. T. (2000). Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. Ecology, 81(1), 220-227.

Kokelj, S. V., Lantz, T. C., Solomon, S., Pisaric, M. F., Keith, D., Morse, P., Thienpont, J.R. & Esagok, D. (2012). Using multiple sources of knowledge to investigate northern environmental change: regional ecological impacts of a storm surge in the outer Mackenzie Delta, NWT. Arctic, 257-272.

Lantz, T. C., Kokelj, S. V., & Fraser, R. H. (2015). Ecological recovery in an Arctic delta following widespread saline incursion. Ecological applications, 25(1), 172-185.

Manson, G.K., and Solomon, S.M. (2007). Past and future forcing of Beaufort Sea coastal change. Atmosphere-Ocean 45:107 – 122, doi:10.3137/ao.450204.

McKinnon, L., Picotin, M., Bolduc, E., Juillet, C., & Bêty, J. (2012). Timing of breeding, peak food availability, and effects of mismatch on chick growth in birds nesting in the High Arctic. Canadian Journal of Zoology, 90(8), 961-971.

Pétillon, J., Georges, A., Canard, A., Lefeuvre, J. C., Bakker, J. P., & Ysnel, F. (2008). Influence of abiotic factors on spider and ground beetle communities in different salt-marsh systems. Basic and Applied Ecology, 9(6), 743-751.

Pirie, L., & Johnston, V. (2012). Survey Methods for Whimbrel. In V. Johnston, J. Bart, P. A. Smith, & J. Rausch (Eds.), Arctic Shorebirds in North America: A Decade of Monitoring (1st ed., pp. 177–184). University of California Press. http://www.jstor.org/stable/10.1525/j.ctt1ppt1c.14

Pirie, L., Johnston, V., & Smith, P. A. (2012). Tier 2 Surveys. In V. Johnston, P. A. Smith, J. Bart, & J. Rausch (Eds.), Arctic Shorebirds in North America: A Decade of Monitoring (1st ed., pp. 185–194). University of California Press. http://www.jstor.org/stable/10.1525/j.ctt1ppt1c.15

Pisaric, M. F., Thienpont, J. R., Kokelj, S. V., Nesbitt, H., Lantz, T. C., Solomon, S., & Smol, J. P. (2011). Impacts of a recent storm surge on an Arctic delta ecosystem examined in the context of the last millennium. Proceedings of the National Academy of Sciences, 108(22), 8960-8965.

Romañach, S.S., Beerens, J.M., Patton, B.A. et al. Impacts of Saltwater Intrusion on Wetland Prey Production and Composition in a Historically Freshwater Marsh. Estuaries and Coasts 42, 1600–1611 (2019). https://doi-org.ezproxy.library.uvic.ca/10.1007/s12237-019-00572-8

Rosenberg, K. V., Dokter, A. M., Blancher, P. J., Sauer, J. R., Smith, A. C., Smith, P. A., ... & Marra, P. P. (2019). Decline of the North American avifauna. Science, 366(6461), 120-124.

Scharffenberg, K. C., Whalen, D., MacPhee, S. A., Marcoux, M., Iacozza, J., Davoren, G., & Loseto, L. L. (2019). Oceanographic, ecological, and socio-economic impacts of an unusual summer storm in the Mackenzie Estuary. Arctic Science, 6(2), 62-76.

Thienpont, J. R., Johnson, D., Nesbitt, H., Kokelj, S. V., Pisaric, M. F., & Smol, J. P. (2012). Arctic coastal freshwater ecosystem responses to a major saltwater intrusion: a landscape-scale palaeolimnological analysis. The Holocene, 22(12), 1451-1460.

Thienpont, J. R., Steele, C., Vermaire, J. C., Kokelj, S. V., Smol, J. P., & Pisaric, M. F. J. (2015). Synchronous changes in chironomid assemblages following a major saltwater intrusion in two coastal Arctic lake ecosystems. Journal of Paleolimnology, 53, 177-189.