Understanding the Effects of Managed Realignment Schemes on Salt Marsh Recovery by

Assessing the Spatiotemporal Patterns of Vegetation Colonization in the

Bay of Fundy, Canada

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Abstract

Understanding the Effects of Managed Realignment Schemes on Salt Marsh Recovery by Assessing the Spatiotemporal Patterns of Vegetation Colonization in the Bay of Fundy, Canada

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As a coastal region, Atlantic Canada is highly susceptible to the impacts imposed by climate change. As hazards such as sea level rise, storm surge, and shoreline erosion are becoming more widely recognized, there is an increased need for communities to adapt to climate change to reduce their vulnerability. Nature-based solutions (NbS) have presented themselves as a more viable long-term solution to their hard engineering counterparts. Managed dyke realignment (MR), a form of NbS, is being used to restore critical salt marsh habitat which offers several ecological, economic, and social benefits through the provision of ecosystem services. This study aimed to determine the effects of MR schemes on estuarine morphodynamics and restoration trajectories by assessing the spatial and temporal patterns of vegetation colonization of a managed realignment site in the Bay of Fundy. The evolution of habitat community structure; the spatiotemporal patterns of vegetation colonization; and the relationship between vegetation colonization and topographic features were analyzed using remote sensing and GIS techniques. Results demonstrate that the restoration trajectory is a highly successional process with initial dominance of vegetation colonization via seed and a shift toward clonal spread later in the trajectory. There was a stronger relationship between vegetation colonization and channel networks in Year 1 post-restoration than consecutive years and S. *alterniflora* and early colonizers coincided with higher accretion rates than other classes. These results provide insight regarding the trajectory of restored sites and key factors to facilitate successful MR design.

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Chapter 1: Introduction & Literature Review

1.1 Introduction and Rationale

As a coastal region, Atlantic Canada is highly susceptible to the impacts imposed by climate change. This is due to the high population of people and infrastructure that are in proximity to the coastline as well as the extent of dependence on coastal ecosystems as a resource (Lemmen and Warren, 2016). As hazards such as sea level rise, storm surge, and shoreline erosion are becoming more widely recognized, there is an increased need for communities to adapt to climate change to reduce their vulnerability.

Historically, our shorelines have been able to respond dynamically to variable conditions. However, hard infrastructure such as dyking and inappropriate development have hindered these natural processes from occurring by fixing the shoreline in place, reducing the resilience of our coastal ecosystem overall. To address these challenges, nature-based solutions (NbS) have presented themselves as a more viable solution to their hard engineering counterparts in the long term. Nature-based solutions (NbS) to coastal engineering involve techniques which harness the power of nature to address wicked problems such as climate change by increasing our resiliency in a way that doesn't reduce the ecological integrity of an ecosystem and provides social benefits (Bridges et al., 2015).

Coastal restoration as a type of nature-based solution demonstrates a paradigm shift in the way society is approaching "wicked problems" such as climate change. These approaches allow the natural environment to flourish while supporting ecosystem services for the betterment of the economy and society. Coastal restoration can be achieved through managed (dyke) realignment (MR) which involves the reintroduction of the tidal regime to areas of previously reclaimed low-

lying land, most commonly through breaching or removing the existing flood defences; or by using structures such as culverts or aboiteaux to create regulated tidal exchange (Pontee, 2014). Given appropriate physical conditions such as hydrology, sediment, and vegetation, coupled with integrated designs, coastal communities can become more resilient in the face of climate change by mitigating the impacts associated with sea level rise, storm surge, waves, shoreline erosion, and flooding through MR approaches to coastal restoration (ICF, 2018; Van Coppenolle and Temmerman, 2019). By understanding and quantifying how restored sites compare to natural (reference) sites, it is possible to continue restoring natural ecosystems with the goal of enhancing the resilience of our coastal communities.

Although realignment techniques are relatively well understood, the response of our hypertidal coastal systems within the Bay of Fundy, to MR schemes are underrepresented in the literature. Since many of these projects are undertaken primarily in Europe and the United States, there is a need to increase our collective knowledge surrounding MR schemes in a Canadian context, particularly in colder environments where ice and snow can greatly influence the feasibility and response of NbS to such techniques. For example, ice regularly damages low marsh zones through scouring and removes plant material and sediment (Ewanchuk and Bertness, 2003). However, ice may also serve as an important vector for facilitating vegetation colonization due to the contributions of ice to the sediment budget (Rabinowitz et al., 2022; van Proosdij et al., 2006). The availability of more research surrounding the restoration trajectories of MR schemes can facilitate greater adoption of these approaches on a wider scale as an alternative to hard infrastructure.

This thesis aims to demonstrate how novel, cutting-edge technologies can be used to collect and display geospatial data to accurately model site dynamics, thus, supporting our understanding of how key parameters, namely hydrology and geomorphology, influence the spatiotemporal patterns of vegetation colonization over time. This research can help to inform any adaptive management actions that may need to be undertaken to enhance restoration success.

1.2 Drivers for Restoration and Managed Realignment

1.2.1 Ecosystem Services and Climate Change Mitigation

Salt marshes, also referred to as tidal marshes, are coastal ecosystems that characterize the intertidal zone where the land meets the sea. They are often found along low-energy coastlines in temperate climates (Brunetta, et al., 2019). The unique environments of salt marshes offer several ecological, economic, and social benefits through ecosystem services. These include but are not limited to carbon sequestration, wildlife habitat (e.g., spawning grounds, nurseries, shelter, and food for fish, shellfish, birds, and other wildlife), coastal protection via wave attenuation, enhanced water quality, flood defence and erosion control, as well as social and recreational opportunities for humans (e.g., Bennett et al., 2020; Burden et al., 2013; Dale et al., 2017; Esteeves, 2014).

The complex interactions and feedbacks between hydrodynamics, vegetation, and sediment processes are critical factors that allow for salt marsh resilience (Bennett et al., 2020). In the face of climate change, these interconnected factors allow salt marshes to provide the first line of defence to mitigate climate associated hazards by buffering against storm and wind damage (Esteeves, 2014). Salt marshes are able to protect coastal communities and important infrastructure from flooding and storm surges through their natural capacity to adapt to climate induced sea level rise through sediment accretion which effectively reduces wind, waves, and

shoreline erosion (Van Coppenolle and Temmerman, 2019). Salt marshes facilitate further

coastline protection by reducing storm surge impacts through the friction imposed by the tidal

wetland vegetation and topographic features (Van Coppenolle and Temmerman, 2019).

1.2.2 Historical Context

Despite the critical importance of these coastal systems, they are globally threatened. It is

estimated that approximately 50% of salt marshes on Earth have been lost or degraded (Dale et

al., 2017), and ~30,500 ha of salt marshes just within the Bay of Fundy (Virgin et al., 2020).

There are several historical and current pressures that contribute to the complete loss or

degradation of these systems such as land reclamation, climate change, coastal squeeze, and

overexploitation (Van Coppenolle and Temmerman, 2019). These factors eliminate or reduce

the capacity of salt marsh environments to provide ecosystem services, usually due to the impact

on hydrology and sediment dynamics which are critical factors for proper ecosystem functioning

(de Vriend et al., 2015; Gerwing et al., 2020).

Land reclamation has been a major contributor to the loss of tidal marshes. According to

Burden et al. (2013), approximately 25% of the word's intertidal estuarine habitats have been

lost due to land reclamation. Many European and North American coastal wetland environments

have become degraded because they have been drained and reclaimed for agricultural, industrial,

or urban development (Dale, et al., 2017; Doody, 2013; Esteeves, 2014). Often land reclamation

has involved the construction of hard engineering structures such as dykes and sea walls, which

have been built to protect assets on the coast from erosion and flooding events. However, they

have resulted in the destruction or degradation of many natural habitats due to the restriction of

water movement, nutrients, sediment, and plant propagules, often resulting in biodiversity loss

(Esteeves, 2014; Gerwing et al., 2020).

Land reclamation that has created a static, artificial margin between land and sea result in 'coastal squeeze' (Doody, 2013). Coastal squeeze is a term used to describe the phenomenon where coastal developments hinder the capacity of coastal habitats to move landwards to higher elevations in response to sea level rise (Burden et al., 2013; Doody, 2013). A common feature contributing to coastal squeeze is historical dyke construction that was originally built for protection against flooding and to convert marshland into substrate appropriate for development or farming (Gerwing et al., 2020; Scheres and Schuttrumpf, 2019). These defence structures essentially fix the high-water mark in place, and as the low water mark rises in response to sea level rise, the opportunities for dynamic coastal processes to occur are diminished (Doody, 2013; Pontee, 2013). This results in intertidal habitat loss since a rise in sea level will gradually increase the frequency and duration of inundation until the tidal marsh is permanently submerged (Esteeves, 2014). In addition, dykes with high hydraulic loads are often protected further by grey revetments that affect the habitat value of the dykes themselves. The historical floodplains in the hinterland are also commonly turned into impermeable land for housing and industry, further diminishing the natural habitats that once existed and their associated ecosystem services (Esteeves, 2014; Scheres and Schuttrumpf, 2019).

The effects of salt marsh loss are already being experienced in the Maritimes. It is becoming more widely recognized that continual land reclamation and traditional hold-the-line practices are no longer sustainable in the long term (Doody, 2013). This is because changes in estuarine hydrodynamics can lead to undesired flooding and erosion (Bennett et al., 2020). In addition, hard engineered structures are not able to adapt to dynamic changes in sea levels. Consequently, it is becoming more important to understand the relationships and interactions among natural and built features comprising the coastal system and there is a need for more

adaptive management approaches to climate change to reduce vulnerability and enhance our resilience overall (Bridges, et al., 2015).

1.2.3 Nature-based Solutions for Coastal Resilience

There is growing evidence suggesting that restoration of degraded coastal wetlands provides an important mechanism for climate change mitigation and adaptation (Burden et al., 2013). This can be done through the adoption of nature-based solutions (NbS) which are a relatively new, internationally accepted approach to hydraulic engineering that harnesses the forces of nature to benefit the environment, economy, and society (Bridges et al., 2015; de Vriend et al, 2015; Temmerman et al., 2013). Nature-based solutions aim to be cost effective, self-sustaining and they attempt to mimic characteristics of natural features to create sustainable and integrated solutions (Bridges et al., 2015; de Vriend and van Koningsveld, 2012). This represents a paradigm shift in the way we approach climate change as there is a growing body of knowledge supporting a transition away from traditional forms of adaptation such as dykes and sea walls, towards NbS which offer more resilience while allowing our environment to prosper in tandem.

The importance and application of ecosystem-based approaches are gaining support worldwide, though they vary according to physical, geomorphological, and ecological contexts (Bridges et al., 2015). The benefits of adopting nature-based solutions are extensive. These include a reduction in cost compared to implementing and maintaining traditional approaches (Brunetta et al., 2019; Dale et al., 2017), a reduction in the frequency of maintenance and the size of defences required (Esteeves, 2014), flood risk management (e.g., attenuating wave energy and storm surges, flood water storage, erosion reduction) (Bridges et al., 2022), a buffer against hazards, and the creation of cooler ambient temperatures (ICF, 2018). NbS can be divided into

four categories, including "fully natural solutions", "managed natural solutions", "hybrid solutions that combine structural engineering with natural features" and "environmentally-friendly structural engineering" (Scheres and Schuttrumpf, 2019; *Figure 1.1*). However, depending on context, coastal structures may be deemed as more appropriate, such as in high energy settings. The best approach to these designs is achieved through collaboration between disciplines, such as ecology, economics, and social sciences to render the most holistic and appropriate solution to the problem being confronted (de Vriend et al., 2015).
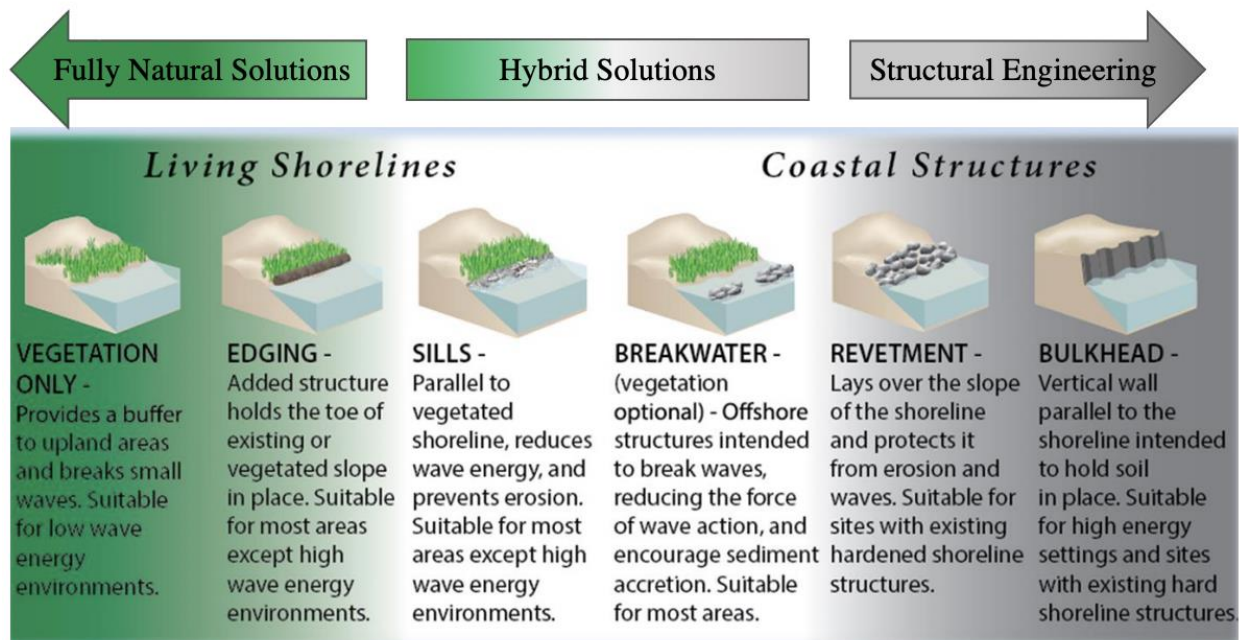
Figure 1.1. A continuum of green infrastructure to hard shoreline stabilizing techniques. Modified from NOAA (2015).

With greater adoption of this approach, there must be clear and widely accepted understanding of the foundations on which NbS are established including principles and methodological frameworks to guide their application, ensuring effective adoption of these concepts (Cohen-Shacham et al., 2016). Building with nature, or NbS, should comply with legislation, regulations, and procedures through proper permitting and governance to accommodate the needs of nature and stakeholders (de Vriend and van Koningsveld, 2012).

These approaches to enhancing coastal resilience through natural infrastructure take many forms, ranging from ecosystem restoration to hybrid approaches such as fortified dunes (ICF, 2018). However, a variety of other approaches fall within the spectrum of NbS. These include beach nourishment to counteract coastal erosion while providing opportunities for recreation, oyster and/or coral reefs to provide habitat and sediment trapping, as well as the formation of seagrass meadows, mangrove forests, and barrier islands (Bridges et al., 2015; de Vriend and Van Koningsveld, 2012; de Vriend et al., 2015). Another NbS that has become more popular over the last decade in Europe has been managed dyke realignment (Bennett et al., 2020).

1.2.4 Uncertainty Regarding the Ecosystem Services of Managed Realignment

As society continues to recognize that the traditional "hold-the-line" approach for coastal protection is no longer a viable option, managed realignment has been shown to render a more dynamic solution (Esteeves, 2014). The technique of managed realignment involves the reintroduction of the tidal regime to areas of previously reclaimed low-lying land, most commonly through breaching or removing the existing flood defences; or by using structures such as culverts or aboiteaux to create regulated tidal exchange (Pontee, 2014; *Figure 1.2*). Increasing the amount of tidal wetland habitat creates new intertidal habitat from formerly flood-defended areas of coastal land. This provides sustainable flood defences and establishes boundaries for coastal development (Bennett et al., 2020). If designed properly, MR projects can be a viable nature-based adaptation method to limit current and future risks associated with climate change.
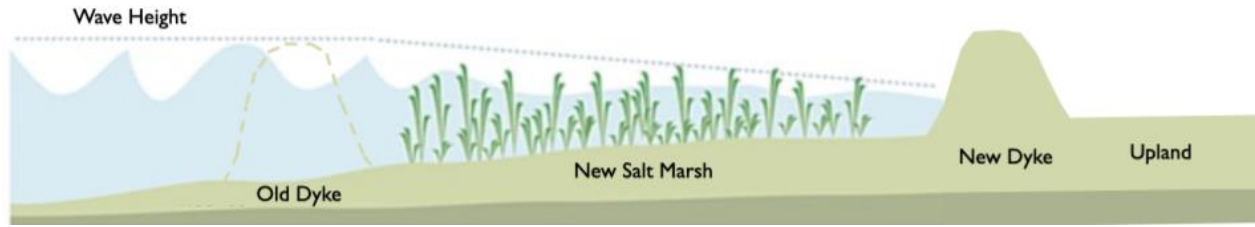
Figure 1.2. Managed realignment schematic showing reduced wave height due to established foreshore marsh. Reproduced with permission from Samantha Lewis.

Although most realignment projects have only been completed within the last 20 years (Mossman et al., 2012a), it is projected that restored salt marshes resulting from MR schemes should be able to provide similar ecosystem services to those of natural salt marshes. These include coastal protection, sediment accumulation, habitat, and carbon sequestration, though the degree by which they are able to do this is underrepresented in the literature. Despite the numerous benefits associated with MR projects, it is important to note that MR still results in artificially constrained natural processes within the estuary under consideration (French, 2006). Consequently, there is a wealth of supporting literature that suggests that the restored sites do not provide ecosystem services equivalent to a natural salt marsh and have lower biodiversity (Dale et al., 2021; Mossman et al., 2012b; Tempest et al., 2015; Weinstein et al., 1997; Williams, 2001). This is because of the nature of the original disturbance and differences in the resulting biological characteristics such as vegetation abundance and sediment characteristics, resulting in variable magnitudes of ecosystem service provision (French, 2006; Kadiri et al., 2011; Tempest et al., 2015). Though it is argued that it may take several decades before a restored marsh is able to offer equivalent functionality compared with a natural salt marsh (Kadiri et al., 2011).

1.3 Factors Affecting the Trajectory of Restoration

Restoration success is traditionally assessed by comparing indicators (biotic and abiotic) in restored saltmarshes with those of 'natural' reference sites (Zhao et al., 2016). MR often follows

a recognized trajectory once tidal flow has been reintroduced by a breach. This results in the formation of intertidal mudflats which are then colonized by saltmarsh plants to form saltmarsh habitat (Burden et al., 2013; Morris, 2012). However, the pace of this successional change from agricultural reclaimed land into functional salt marsh is highly variable and is greatly influenced by a number of factors such as topography, hydrology, sediment, and vegetation (Morris, 2012; Pontee, 2014) as they will determine the temporal and spatial trajectories of restoration success. According to Gerwing et al. (2020), it can take 60-100 years for equilibrium conditions to develop. The key physical parameters in tidal wetland restoration post breach, including hydrology, sediment, and vegetation will be assessed below.

1.3.1 Hydrology

Increasing the tidal prism within the MR system changes the hydrology of the site immediately as the hydrologic connectivity is re-established within the estuary (Gerwing et al., 2020; Spencer and Harvey, 2012). This can modify tidal flow both locally and in the entire estuary (Bennett et al., 2020; French, 2006). Hydrologic factors such as current velocity, tidal input, and fluvial input of a restored site plays a critical role in site trajectory as it highly influences the accretion of sediment. This in turn influences the physical functioning post-breach and morphological evolution of the site (Dale et al., 2021). This is largely due to site elevation and hence the hydroperiod, as well as hydrodynamics; these are the main factors controlling the development of salt marsh ecosystems (Kadiri et al., 2011).

Tidal flow patterns through the system are highly complex in space and time as it is controlled by topography, tidal stage, wind, waves, and vegetation (Spencer and Harvey, 2012). Tidal channels and creek networks are key features of salt marsh landscapes as they facilitate flooding and drainage of the marsh platform (Fagherazzi et al., 2012). These features are

important as they effectively distribute sediments and nutrients to the marsh ecosystem. In MR

sites, these channels may pre-exist or have been lost due to agricultural activities, which affects

the way the currents interact with the channel bottom or margins, potentially impacting sediment

delivery and stability (French, 2006). This reduction in hydrological connectivity has been

associated with modifications to the sediment structure due to dewatering and organic matter

mineralisation (Dale et al., 2021). The alteration in sediment structure typically results in an

initial dominance of sheet flow, which is defined as water flowing evenly across the surface of

the soil (Dodds and Whiles, 2010) due to the lack of developed channel networks. This is

followed by potentially rapid development of embryo creek networks which are small creeks that

result from erosion due to heterogeneity in the topography, which concentrates the surface flow

(Dale et al., 2018). The timescale at which this occurs is influenced by the tidal energy and

sediment characteristics (i.e., drainage properties, marsh gradient, presence of pre-existing

channels) (Spencer and Harvey, 2012). Given the appropriate conditions, these creek networks

can become re-established and deliver similar functionality to natural marshes and reach an

equilibrium (Spencer and Harvey, 2012).

Subsurface water flow is another important hydrological factor because it affects soil

aeration and delivery of nutrients (Tempest et al., 2015) which are important for proper

ecosystem function. Sub-surface hydrology creates spatial heterogeneity and is influenced by

grain size, degree of compaction, topography, tidal pressure, and the presence of sub-surface

vertical and horizontal features (Spencer and Harvey, 2012). Compaction and tillage of the relict

surface can reduce the frequency and size of macropores which can facilitate fast, horizontal near

surface flows while impacting the subsurface water flow rates (Tempest et al., 2015). This may

result in waterlogged surface sediment conditions and limit the vertical movement of pore waters

(Burden et al, 2013; Spencer and Harvey, 2012). This can inhibit salt marsh accretion as the ability of sediments to build up is reliant on dewatering and compaction processes (Spencer and Harvey, 2012).

Hydrology also plays a significant role in the vegetation that can become established within the MR site. The hydroperiod, defined as the length of time that the wetland is submerged, is considered to be the most important factor for determining which species may germinate and become established within the intertidal zone (Spencer and Harvey, 2012). Plant species have different ranges for the duration of inundation that they can tolerate. In addition, since hydrochory is the main mechanism for seed dispersal in salt marshes, it is important that proper hydrologic conditions are offered as the rates of tidal flow must be favourable for seed transport and retention (Spencer and Harvey, 2012).

1.3.2 Sedimentation and Marsh Surface Elevation Change

Sediment accretion is important in natural and MR sites in terms of climate mitigation as well as ecological function. The rate at which sediment can accrete within the marsh is directly related to the capacity of the coastal ecosystem to keep pace with sea level rise (Fagherazzi et al., 2012). In addition, the transport and deposition of sediment within a salt marsh ecosystem supports ecological processes such as fluxes of organic matter, nutrients, and seeds (Fagherazzi et al., 2012). This highlights the importance of considering the elevation relative to the tidal frame as it dictates the frequency and duration of tidal inundation if breaching were to occur (Dale et al., 2021; French, 2006; Oosterlee et al., 2020). These factors directly influence the spatial and temporal patterns of sedimentation rates and the type of marsh which will ultimately result post breach (French, 2006). Studies that have monitored MR sites over time have suggested that it can take up to 15 years for marsh surface elevation to stabilize, and 60-100

years for equilibrium conditions to develop (Gerwing et al., 2020; Jongepier et al., 2015). However, other studies have suggested that sediment characteristics in the restored saltmarsh do not ever become similar to those in natural saltmarshes as they vary in terms of sediment moisture and organic matter content, porosity, bulk density, and pH (Burden et al., 2013; Kadiri et al., 2011).

The sedimentation patterns in MR sites are commonly monitored using techniques such as elevation surveys, marker horizons and rod surface elevation tables (RSETs) (Burden et al., 2013). Sedimentation patterns of realignment are complex as they display varying levels and styles of accretion, resulting in high spatial heterogeneity (Dale et al., 2017; Spencer and Harvey, 2012). In general, suspended sediments are transported with the tidal currents onto the marsh platform and partially deposited in distinctive spatial patterns (Fagherazzi et al., 2012), with a pattern of accretion during the flood tide and dewatering and erosion during the ebb tide (Dale et al., 2021). Accretion can occur across the entire tidal flat but is often greater in areas adjacent to creeks (Brunetta et al., 2019), or near the breach (Dale et al., 2017), with less deposition occurring as the distance from tidal channels or the seaward marsh edge increases (Fagherazzi et al., 2012; Spencer and Harvey, 2012). These spatial sedimentation patterns that occur drive the long-term geomorphic development of the MR site.

Full tidal exchange and high sediment loads within the water column can result in very rapid sedimentation rates (Morris, 2012; Oosterlee et al., 2020). According to Dale et al. (2021), artificially lowered areas accreted at a faster rate. Rapid rates of accretion have resulted in the establishment of extensive saltmarsh habitat as elevation had increased relative to the tidal frame (Dale et al., 2017; Virgin et al., 2020; Norris et al., 2022). After this initial rapid sediment deposition, sedimentation rates decrease with increasing platform elevation because lower

portions of the marsh platform are flooded more frequently, allowing for more sediment

deposition in the low marsh (Fagherazzi et al., 2012; Spencer and Harvey, 2012).

Vertical accretion is key to ensuring a salt marsh will develop as the colonization of

vegetation is interconnected with the vertical accretion of sediment (Brunetta et al., 2019;

Fagherazzi et al., 2012). Site elevation, which is linked with the period of reclamation due to

dewatering and compaction, is important in terms of rates at which marsh colonization and

growth occurs (French, 2006). This is because low-lying sites in the tidal frame often have

modified sediment structure and properties which may inhibit the colonization of vegetation or

benthic invertebrates due to poor drainage (Oosterlee et al., 2020). It is important that there is a

balance in sedimentation rates within the site as sufficient sediment loads are required for

nutrient input and habitat creation (i.e., raising bed elevation to allow colonization of halophytes)

(Dale et al., 2017). However, excess sedimentation could prevent the desired habitat from

becoming established due to the production of anoxic conditions resulting from the burial and

decomposition of vegetation that existed prior to breaching (French, 2006). According to Pontee

(2014), high sediment supply coupled with low energy hydrologic conditions within the

realignment site can allow the mudflats to accrete vertically and evolve into saltmarsh habitats.

This is achieved by providing ideal conditions for vegetation colonization and establishment (van

Proosdij et al., 2023).

1.3.3 Vegetation

Vegetation composition and physical structure have significant influences on saltmarsh

ecosystem function, emphasizing the importance of ensuring that restored sites have comparable

vegetative structures to those of reference sites (Mossman et al., 2012b). If the proper conditions

arise post breach, the development of salt marsh vegetation can increase the rate of vertical

accretion due to the capacity of vegetation to attenuate waves, the binding ability of the roots, and the ability of plants to trap sediment on their leaves (French, 2006; Fagherazzi et al., 2012). Therefore, the presence of halophytic vegetation has major implications for climate mitigation as vegetation influences the rate at which the marsh can accrete sediment at a rate equivalent to or higher than sea level rise and reduces wave impact (Brunetta et al., 2019). According to Gerwing et al. (2020), plant communities mostly recover within 5-20 years, though other studies suggest that the diversity of plant communities that develop post breach are not equivalent to natural marshes (Dale et al., 2021; Mossman et al., 2012a).

Colonization of salt marsh species within restored sites is largely influenced by nutrient availability, salinity, relict surfaces, and seed supply, though elevation is said to be the most fundamental consideration in the design of intertidal habitats (Bridges et al., 2021). Although halophytic species colonize realignment sites quite rapidly and maintain similar species richness in comparison to references marshes after a year, the overall diversity of MR sites was different from reference sites as early successional species remained dominant (Mossman et al., 2012b). In general, vegetation development within saltmarshes is usually considered to take place in a successional manner whereby pioneer species accrete sediment by reducing flow velocities, encouraging the deposition and stabilization of sediment through root development, increasing the elevation of the site (Spencer and Harvey, 2012).

Elevation characterizes the position within the tidal range which influences the frequency, duration, and intensity of tidal inundation (Esteeves, 2014; Pontee, 2014). Upon colonization, there is a strong tendency for the species to exhibit zonation due to the varying elevations. Plants grow within a distinct elevation range, approximately between mean sea level and mean high tide (Fagherazzi et al., 2012). Vegetation zonation reflects the mixture of species

that have varying tolerances for flooding and salinity, with more flood tolerant species growing at lower elevations, and less flood tolerant species establishing at higher elevations (Bridges et al., 2022). According to Mossman et al. (2012b), sediment conditions in the low areas of realigned marshes were less oxygenated than those at similar elevations within natural sites due to waterlogged conditions, while higher elevations in MR sites remained less vegetated than natural sites likely due to hypersaline conditions. Therefore, elevations must be properly planned in MR designs to achieve optimal vegetation zonation and species diversity. This is because lower than optimum elevation results in an increase in the depth of flooding tides which can cause a decrease in plant colonization (Fagherazzi et al., 2012), resulting in reduced ability to resist climate change impacts.

It is also important that there is a suitable seed supply and the right conditions for germination and seedling establishment within the MR site. The establishment of vegetation in a site is generally not from residual seeds within the soil, but by the influx of seeds from an adjacent marsh (French, 2006). This emphasizes the importance of the MR site being in proximity to a nearby salt marsh that is a source of seeds because seeds do not travel great distances (Spencer and Harvey, 2012). Therefore, low plant abundance and diversity in MR sites may often be attributed to a lack of seed supply.

1.4 Spatial and Temporal Patterns of Vegetation Colonization

Exploring the spatial and temporal patterns of vegetation colonization is an important contributor in understanding the restoration trajectory of areas that have undergone managed realignment. There are a variety of factors that influence how vegetation may develop in a salt marsh, including the species' position within the tidal frame (Wolters et al., 2005), the mechanisms of propagule dispersal (e.g., ice, drift litter, and plant materials), the buoyancy and

flotation time of propagules (Huiskes et al., 1995), environmental conditions (Ewanchuk and Bertness, 2003), and geomorphological features (Marani et al., 2004).

Although the literature presents patterns in which vegetation may colonize, it is notable that spatiotemporal factors are highly variable and can only be described generally. This is largely due to the fact that initial site conditions, external factors such as proximity to a local seed source, and disturbances all have the capacity to enhance or prolong the rate at which a restored site expresses vegetation community structure more similarly to those of reference conditions.

1.4.1 Initial Patterns of Vegetation Colonization

Salt marshes are dynamic systems, therefore there are different physical and biological drivers that may influence the patterns that vegetation may express within a restored site over space and time. Physical factors such as winter icing can impact established vegetation across the marsh surface by damaging colonies of *Spartina alterniflora (Sporobolus alterniflorus)* in the low marsh zone as well as depositing sediment and producing scour in the middle and high marsh zones (Ewanchuk and Bertness, 2003). This phenomenon affects the microrelief of the salt marsh, leaving bare patches that may remain uncolonized for a period due to unfavourable edaphic conditions (e.g., poor drainage, anoxic soils) (Ewanchuk and Bertness, 2003; Norris et al., 2022). Despite the damage that can be inflicted by winter icing events, there is evidence that ice can serve as an important vector for the transportation of plant propagules in the Bay of Fundy. This has important implications for newly restored tidal marshes, particularly when there is no clear source of plant propagules within or outside of the site (Rabinowitz et al., 2022; van Proosdij and Townsend, 2006). According to a study conducted by Rabinowitz et al. (2022), viable rhizome material from *S. alterniflora* and *Spartina patens (Sporobolus pumilus)* was

found in ice samples within the Bay of Fundy. It is not uncommon to find evidence of sediment deposits resulting from ice deposition and subsequent melting sporadically across a marsh surface. Therefore, ice serves as a physical mode of propagule dispersal that may promote random patches of vegetation colonization early in the restoration trajectory as well as random patches of bare ground later in the restoration trajectory.

Biological factors are another important driver that influences vegetation colonization, namely dispersal ability and seed production. Depending on the primary dispersal mechanism for a given species, they may express a concentrated pattern in a particular area and become sparser away from the parent plant while other species may have greater potential for long distance dispersal (Wolters et al., 2005). This is largely due to different species' seeds having variable buoyancies and flotation periods, thus impacting the species' dispersal ability. The buoyancy of the seeds is attributed to ideal surface/weight ratios, aerial tissues, and hairs that trap air bubbles, as these factors enable them to travel longer distances (Erfanzadeh et al., 2010; Huiskes et al., 1995; Poschlod et al., 2005). The biological factors that facilitate buoyancy, coupled with the species' seed production may be responsible for the early successional phase that is often expressed in restoration sites. For example, early successional species such as *Salicornia* and *Suaeda* produce large quantities of viable seeds (300-30000 per m^{-2}), and are buoyant (Erfanzadeh et al., 2010; Wolters et al., 2008). These factors can likely be attributed to why these genera have the capacity to establish themselves early and widely within restoration sites, exhibiting variable spatial patterns. Conversely, later successional species such as *S. alterniflora* have less viable seeds (Stalter, 1973), therefore their increase in abundance mainly occurs through clonal expansion (Erfanzadeh et al., 2010) which is a characteristic of many perennial halophytes (Huiskes et al., 1995). As a result, the spatial patterns that are exhibited by *S.*

alterniflora and other perennial halophytes are characterized by dense monospecific patches that result as new stems grow along the outer edge of the initial ring, and gradually increase each year (van Proosdij and Townsend, 2006).

Although variability is expressed in spatial patterns, temporal scales are also variable. As previously described, vegetation colonization is dependent on a multitude of factors such as hydrology and sediment. The rate at which favourable conditions are achieved after salt marshes are restored is inconsistent, which is likely why successional stages are often described in terms of quantifiable or observable objectives being achieved, such as percent coverage or species diversity, rather than the time elapsed post-restoration. Within the Bay of Fundy, Virgin et al. (2020) generally describes the successional stages as: 1) high initial sediment deposition; 2) appearance and establishment of *S. alterniflora* with patchy distribution; 3) domination of *S. alterniflora* and decreased spatial variability; 4) *S. alterniflora* restricted to marsh edge and creek banks, *S. patens* density and spatial extent increasing. Since different sites within the Bay of Fundy achieve these benchmarks at different times, it is appropriate to assess when each successional stage is achieved within the study area and how long it takes to achieve the subsequent successional stage in the restoration trajectory to understand temporal scales.

1.4.2 Relationship Between Tidal Creek Formation and Vegetation Colonization

Dynamics of salt marshes are governed by complex interactions between biotic and abiotic features such as vegetation and geomorphology. Geomorphic features comprise elements that create variability within a surface through natural processes such as erosion and sediment deposition, resulting in features such as tidal creeks and channel networks. These features are described as playing an important role in facilitating seed retention (Wang et al., 2018), promoting dispersal via enhanced hydrological connectivity (Wang et al., 2021), as well as

creating topographic heterogeneity which facilitates the establishment of different species (Brooks et al., 2015). It is apparent that there are several drivers at work that may influence the dynamics of vegetation colonization within tidal marshes.

Vegetation development is intrinsically linked to the geomorphological characteristics of tidal marshes due to the coupled evolution of vegetation and morphology (D'Alpaos, 2011). One way this has been assessed is by monitoring species assemblages in relation to their distance from creeks and channels. Experimental research demonstrates that there are inconsistencies between the relationship of vegetation colonization and their distance from tidal creeks as some species have shown to grow exclusively along edges of creeks and channels, though some species do not appear to colonize in a predictable manner away from channel networks (Marani et al., 2004; Tang et al., 2022). However, other research suggests that microtopographic structures are effective tools for species re-establishment by acting as seed traps. This was demonstrated by Wang et al. (2018) as larger, deeper microtopographic structures entrapped more seeds compared to smaller structures, which in turn formed larger patches in the long term. Topographic heterogeneity can therefore be linked to enhanced seed retention as bare, homogenous surfaces do not as readily provide opportunity to trap seeds or propagules (Wang et al., 2018; Wang et al., 2021). As a result, there may be a more apparent relationship between channel/creek depth and vegetation colonization rather than distance from creek as initial channels may be small enough to facilitate seed/propagule capture without creating overly wet and anoxic conditions.

Relevant literature suggests that the more direct link between vegetation colonization in relation to channels is through the biophysical conditions that are induced through hydrologic connectivity. Tidal creeks and channels alter soil salinity and soil moisture, as soil salinity

increases with increasing distance from tidal creeks and soil moisture decreases with increasing distance from tidal creeks (Tang et al., 2022). Due to the lower elevation associated with channel network formation, these areas serve as drainage networks that are flooded more frequently and for longer periods of time. As a result, only more resistant species can survive prolonged anoxic conditions and develop monospecific conditions, while upland areas characterized by more aerated soils which support greater species diversity (Marani et al., 2004). This explains why primarily *S. alterniflora* is typically found in dense monocultures along the edges of channels and creeks in salt marshes, while middle and upper marsh boundaries are often mixed with a variety of species such as *S. patens, Spartina pectinata (Sporobolus michauxianus), Distichlis spicata, Plantago maritima, Juncus gerardii,* and other halophytic and brackish species. This heterogeneity is important for restoration trajectories as some restored sites have shown delays in the colonization of mid and upper marsh species due to inappropriate conditions such as compaction, dry/hypersaline conditions, and water retention that have resulted from a more homogenous surface (Brooks et al., 2015).

1.5 Objectives and Research Questions

The primary focus of this research is to determine the effects of MR schemes on estuarine morphodynamics and restoration trajectories by assessing the spatial and temporal patterns of vegetation colonization of a restored salt marsh within the Bay of Fundy, Nova Scotia. This was achieved by comparing vegetation community structure and surface coverage from 2019 (Year 1 post-restoration) to 2022 (Year 4 post-restoration). Additionally, the vegetation colonization was assessed in terms of the visual distribution (e.g., clonal, linear) of patches as well as their persistence and new colonization. Lastly, vegetation colonization in relation to topographic features (e.g., channels, cracks, elevation differences), building off work done by a previous

master's student (Samantha Lewis), was also assessed. Understanding the spatial and temporal variation of vegetation community development will increase our collective knowledge surrounding the restoration trajectory of salt marsh habitat and efficacy of NbS within our unique macrotidal system and can help to inform whether adaptive management should take place. The specific research questions that will be addressed within the project are as follows:

1) How did the vegetation community structure evolve over time?

2) What are the prominent/initial patterns of vegetation colonization?

3) What are the effects of topographic features (protochannels, channels, cracks, elevation differences) on the spatial and temporal patterns of vegetation colonization?

Chapter 2: Study Site

2.1 Physical Description and Characterization

This research will investigate the effects of managed realignment schemes within the Upper Bay of Fundy, Nova Scotia, Canada. Specifically, the Converse Marsh (NS044), which is part of the Tantramar Marsh System. This system is a large tidal wetland complex, located on the Chignecto Isthmus which serves as a land bridge between Nova Scotia and New Brunswick (Bowron et al., 2021) (*Figure 2.1*). The Converse Marsh lies on the eastern side of the Missaguash River, located at the mouth of the large tidal river. The Missaguash River enters the Cumberland Basin which covers 118 km^2 and approximately 2/3 of this area comprises salt marsh and mud flat habitat (van Proosdij et al, 2006).

Figure 2.1. The location of the Converse Marsh (NS044) in Nova Scotia, Canada.

Since the Converse Marsh is located within the Upper Bay of Fundy, it is subject to unique conditions when compared to MR sites in Europe and the United States. The Bay of Fundy is famous for its macro-tidal conditions, with an average tidal range of 12-14 m (Davidson-Arnott et al., 2002), though the tidal range is occasionally greater than 16 m (Tibbets and van Proosdij, 2013). This hyper-tidal environment is the result of the unique confluence of local geology, seabed morphology, geography, and oceanographic factors (Desplanque and Mossman, 2004). The site also experiences semi-diurnal tides which is defined as having two unequal high waters and two unequal low waters in one day (Desplanque and Mossman, 2004).

Suspended sediment concentrations within the Bay of Fundy are locally high but seasonally variable. The suspended sediment concentrations vary from 0.05 $g \cdot l^{-1}$ to 4.0 $g \cdot l^{-1}$ (Amos et al., 1991) and range in sediment grain size from fine to coarse silt (van Proosdij et al., 1999). These conditions create minerogenic marshes due to the extensive deposits of sediment that help to build the elevations of salt marshes and mudflats. More recently, the heavy influence of ice has been recognized as an important vector for sediment deposition in these systems. A study carried out by Rabinowitz et al. (2022) demonstrated that ice blocks contained 26.61 to 21,483.59 kg of total sediment per ice block in nearby sites, demonstrating a large contribution to the sedimentary budget of salt marshes within the Bay of Fundy.

2.2 History of Tidal Wetland Restoration in the Bay of Fundy

2.2.1 Habitat Compensation (CBWES Inc.)

Human activities such as historical dyking, coastal infrastructure, and freshwater impoundments have resulted in an approximate 80% loss of salt marsh habitat within the Bay of Fundy (MacDonald et al., 2010). This detrimental loss has reduced critical habitat for fish and birds and has presented challenges under our changing conditions due to the pressures inflicted

by climate change as we substantially impaired the defense mechanisms offered by salt marsh environments. These factors have triggered an active response over the last ~25 years to restore tidal wetlands in our province.

Since 2005, CB Wetlands and Environmental Specialists (CBWES Inc.) in partnership with Saint Mary's University has collaborated with government, academic, private sector, and NGO partners to collectively restore over 400 ha of tidal wetland habitat in Atlantic Canada. These projects have ranged from smaller scale initiatives at scales of ~10 ha (Bowron et al., 2011), to more recent, complex, and large-scale projects scaled at 90 + ha of restored floodplain habitat (Ellis et al., 2022). In all cases, the hydrology of these sites had either been improved or completely reinstated, enabling the re-establishment of salt marsh habitat. This has been conducted in several different ways including bridge or culvert replacements, breaching impoundments, and breaching dykes. However, more recently, managed dyke realignment has been used as a restoration strategy. Managed dyke realignment has been conducted by CBWES Inc. in three sites – Onslow North River, Converse, and Belcher Street *(Figure 2.2)*. The Onslow North River restoration was conducted as a large-scale habitat compensation project which involved off-setting the destruction of salt marsh habitat in Windsor by restoring habitat in Truro, Nova Scotia. The Converse and Belcher Street sites were not habitat compensation sites as they were part of the *Making Room for Wetlands project,* funded by the Department of Fisheries and Oceans Canada's Coastal Restoration Fund (CRF).

The intensive monitoring program post restoration of these sites helps to ensure all key parameters, including sediment, vegetation, and hydrology, are all following an anticipated trajectory towards successful restoration. Given that many of these projects have been some of the earliest or most complex projects to date, they have created substantial opportunities for

research and innovation. There have been numerous research projects related to these works, with each project aiming to fill the gaps in our collective knowledge regarding the response of our hyper-tidal, sediment laden system to nature-based approaches. As a result, these projects have created the foundation for coastal restoration in our region, with applications on a global scale.
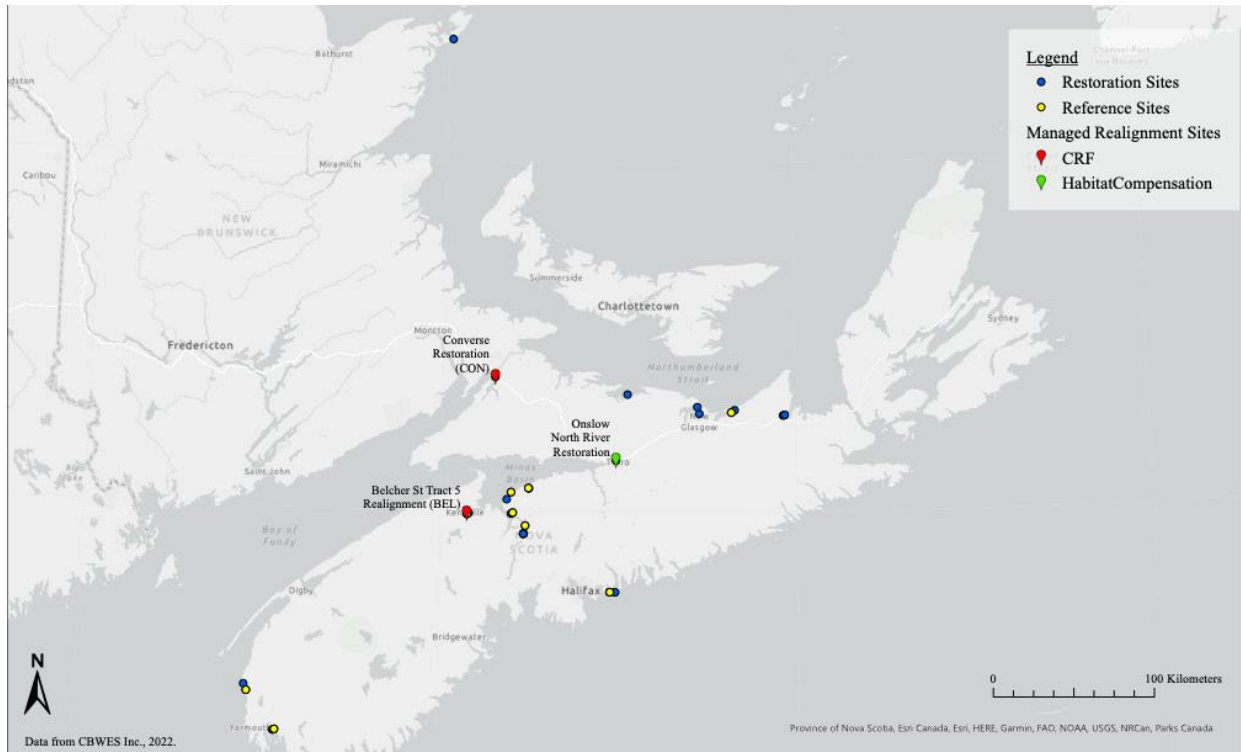
Figure 2.2. A map of all restored and reference sites conducted by CBWES Inc. in Atlantic Canada, with managed realignment sites characterized by balloon symbology. Red balloons symbolize sites that are under the CRF Making Room for Wetlands project, while the green balloon indicates a habitat compensation project.

2.2.2 Site Selection and *Making Room for Wetlands* Project

The Converse Marsh had been characterized by a long sinuous dyke system that was protecting active and fallow agricultural land. Notably, the dyke system was at risk due to its position close to the bank and the accelerated loss of foreshore marsh that had caused extensive erosion. Consequently, these events had presented high maintenance costs and the possibility of dyke failure, deeming it infeasible to maintain the dyke at its current position (Bowron et al.,

2020). As a result, this site offered unique opportunities to evaluate the efficacy of managed

dyke realignment to restore the historic floodplain, presenting itself as an ideal candidate for the

Making Room for Wetlands project. This assessment was made based on the outcome of the

AgriRisk project as it was assigned a value of high vulnerability (van Proosdij et al., 2018).

CBWES Inc. worked in collaboration with Nova Scotia Department of Agriculture

(NSDA), Land Protection Section, and TransCoastal Adaptations Center for Nature-based

Solutions at Saint Mary's University to design a realignment plan to restore tidal wetland habitat

while concurrently protecting the adjacent areas and dykeland infrastructure (Akyol, 2020). In

2018, approximately 420 m length of eroding dyke was removed, and the resultant materials

were used to backfill the drainage ditch inland of the original dyke to match the elevation of the

foreshore fringe marsh. In addition, 150 m of new dyke was constructed using materials from the

former dyke as well as the adjacent upland area, resulting in the creation of a drainage outlet and

a borrow pit directly adjacent to the new dyke (*Figure 2.3*).
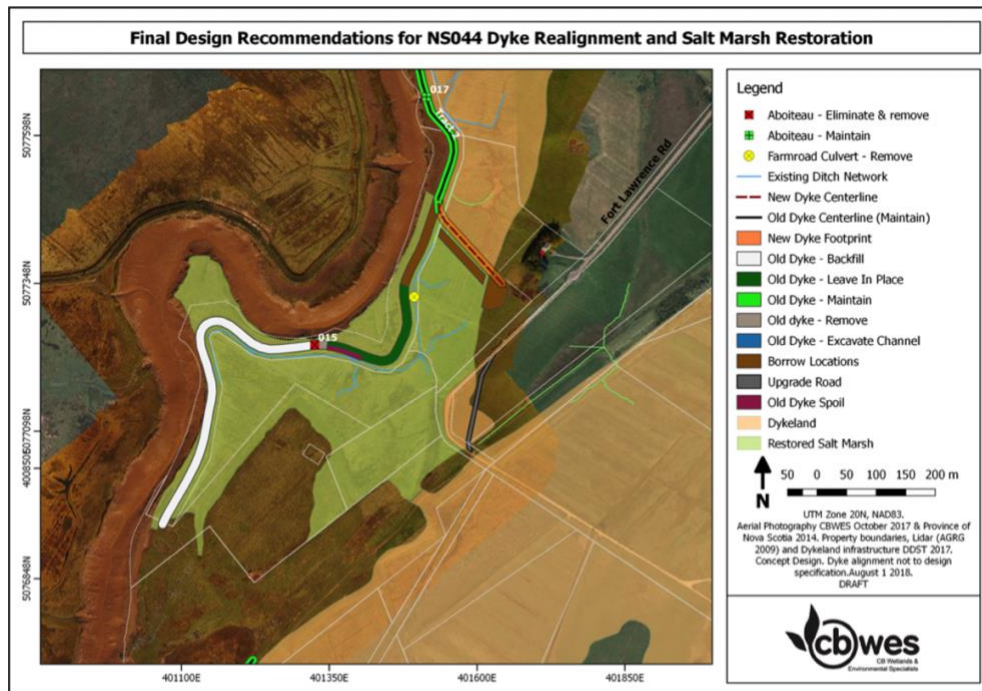
Figure 2.3. Final design of the Converse (NS004) dyke realignment and salt marsh restoration.
Reproduced with permission from Jennie Graham.

Geospatial data regarding hydrology, sediments, and vegetation have been collected each year from 2017 baseline conditions up to 2021 under a 5-year monitoring framework. The total restored area of this site was 15.4 ha (Bowron et al., 2021). The site has been slower to respond to the restored hydrology compared to other projects in the Minas Basin due to the strong influence of its antecedent landscape history such as agricultural vegetation, resulting in less favourable conditions for halophytic vegetation colonization in the first-year post-restoration. According to Lewis (2022), areas with remnant agricultural ditches typically maintained their position over time as part of the channel network and the antecedent landscape history, though there were also channels that developed within relic natural features and embryonic channels that had developed without the influence of landscape history. Areas such as the borrow pit that were stripped of their landscape history through excavation were faster to establish channel networks due to their lower elevation, though the marsh platform experienced higher rates of sediment deposition (Lewis, 2022). This appeared to have created more suitable conditions for vegetation colonization on the marsh platform as consecutive years demonstrated incremental sediment accretion and establishment of halophytic vegetation including *S. alterniflora*, *S. pectinata*, brackish species, and other early colonizers such as *Suaeda maritima (Figure 2.4).*

Figure 2.4. Development of the Converse Marsh from 2018 pre-restoration to 2022-Year 4 post restoration.

Chapter 3: Data and Methods

There were several phases of this research that built off the previous step to carry out the workflow. Initial steps involved collecting the required datasets from 2019 – 2021 and conducting field work to acquire vegetation and elevation data for 2022. The second phase involved processing the imagery, cleaning channel delineations, digitizing the vegetation data and performing the image classification of the study site. The final stage of the research was performing analyses to be able to model the spatial patterns of vegetation colonization and the relationship between topographic features. The overall workflow is outlined in *Figure 3.1*.
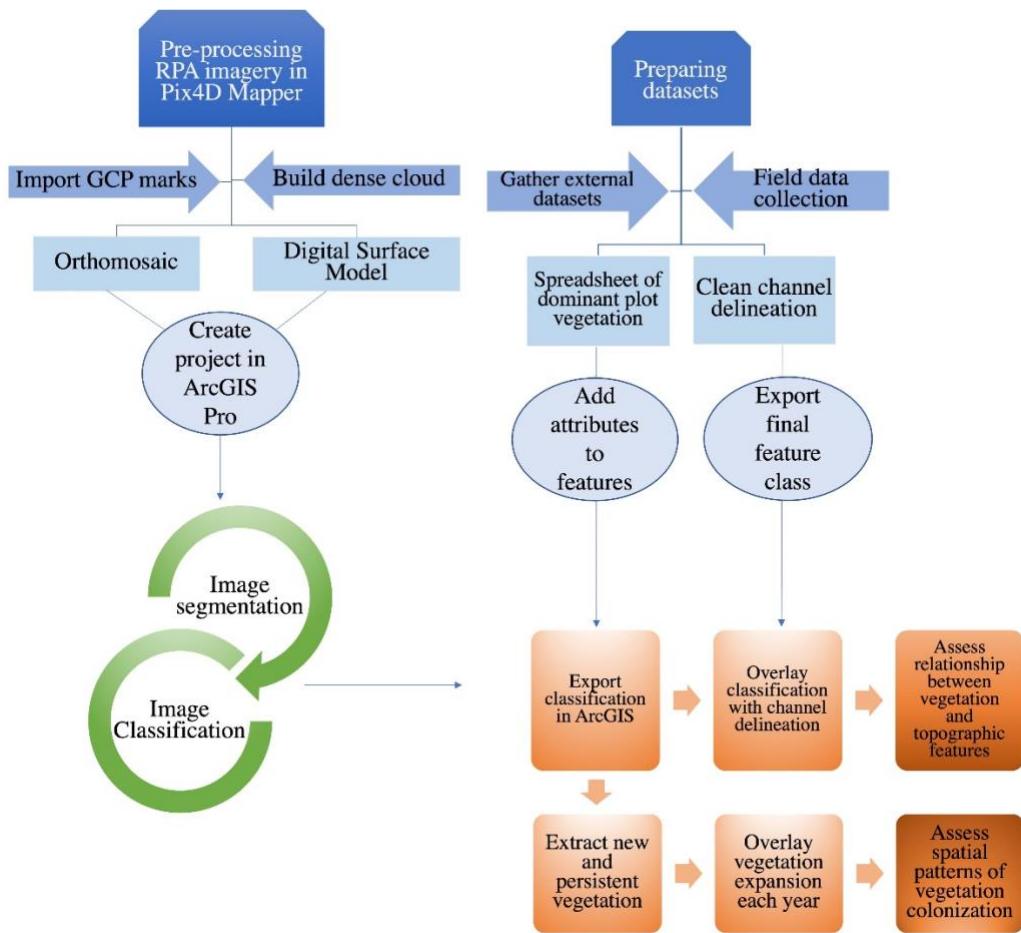
Figure 3.1. Outline of general research workflow methodology.

3.1 Preparation for Data Collection

Initial sampling stations were planned using a stratified sampling method in ArcGIS Pro. This was achieved by clipping the 2021 digital elevation model (DEM) to the extent of the study area and running the contour tool at 0.5 m intervals to create contour polygons. The result was 13 bins of elevation categories to represent the elevation gradient at the Converse marsh (*Figure 3.2*). The contour polygon was then input into the random points tool which placed plots (n=30) based on different elevation categories. The points were also compared to the 2021 habitat classification to ensure that different vegetation cover, elevation, and inundation frequencies would be captured to achieve a representative dataset of the site.

Figure 3.2. Sampling stations (n=30) placed within 13 different elevation bins at the Converse restoration site. Background imagery collected July 25, 2021 with a DJI Phantom 4 RTK RPAS.

Points that were randomly generated in inaccessible areas such as deep channels were removed, resulting in 23 new sampling stations. Other plots were added including previously studied plots from the CBWES Inc. monitoring framework (n=10) and a previous Master's student (Rabinowitz, 2020) (n=7) that were within the study area. These plots were included to create a continuous dataset by monitoring vegetation within the plots that had been surveyed in previous years, allowing change to be monitored over time. The result was 40 sampling stations across the marsh surface to establish an appropriate sample size (*Figure 3.3*). The naming conventions for the CBWES plots kept their original names, Rabinowitz (2022) plots were named TR_(unique value), and the newly placed plots were named KN_(unique value). The easting and northing values were extracted from each station, exported as a .csv and transferred onto a Leica GS14 RTK GNSS unit in preparation for field data collection.

Figure 3.3. Final sampling station locations (n=40) for field data collection. Background imagery collected July 25, 2021, by CBWES with a DJI Phantom 4 RTK RPAS.

3.2 Data Collection

3.2.1 Elevation Surveys

On August 2, 2022, the location and associated elevation of each vegetation survey station was staked out and measured with a Leica GS14 RTK GNSS unit (*Figure 3.4*). The points were marked using a survey flag and labelled for greater ease of recognizing each plot when returning to complete vegetation surveys.

Figure 3.4. Measuring the location and elevation of survey stations using a Leica GS14 RTK GNSS unit on August 2, 2022. (Credit: TransCoastal Adaptations).

3.2.2 Vegetation Surveys

Vegetation stations were located and surveyed on August 3, 2022. Vegetation data was collected using a point intercept method which utilized 1 m^2 plots (quadrats) divided into a grid of 25 squares as per the CBWES monitoring protocol (*Figure 3.5a*). At each station, facing the river, the bottom right-hand corner was placed over the flag and the quadrat was flipped downstream (left). By offsetting the location of the vegetation survey station, it prevented the

vegetation from being trampled during the layout process. Once the quadrat was placed, each species that was observed within the quadrat was recorded, then a wooden dowel was held vertical to each sampling point until it reached the ground below. Each species that touched the rod was documented, and the process was repeated for each of the 25 intercept points at each station. A photo was taken at each station and a plant representing the dominant species within the plot was measured and recorded (*Figure 3.5*).

Figure 3.5. A) Quadrat placed over a vegetation plot; B) Documenting each species within a vegetation plot in the field notebook on August 3, 2022. (Credit: TransCoastal Adaptations)

3.2.3 RPAS Flight

A low altitude RPAS survey of Converse was conducted with a DJI Matrice 300 RTK quadcopter on September 6, 2022. This aircraft was chosen due to the survey-grade geolocation data offered by the RTK GNSS functionality. The site was flown at an altitude of 91.8 m Above Ground Level (AGL) by CBWES Inc.

A total of 9 ground control points (GCPs) were deployed throughout the site prior to the survey. Each GCP location was measured using a Leica GS14 RTK GNSS unit (*Figure 3.6*). These targets and their surveyed positions were used to georeference the data products during photogrammetric processing. Validation points (n=100) were also collected across the marsh surface using the Leica unit in areas of bare mud surfaces.

Figure 3.6. Approximate GCP deployment locations (n=9) at the Converse restoration site for the RPAS aerial survey. Background imagery collected July 25, 2021, with a DJI Phantom 4 RTK RPAS by CBWES Inc.

3.2.4 Secondary Data

Secondary data was used to create a time series from 2018 to 2022. Orthomosaic imagery was provided from 2018 to 2021 and habitat classifications were provided by CBWES Inc. from 2019 to 2021. The orthomosiac imagery that was used for this project was collected on the following dates, outlined in *Table 1*. Imagery collection dates and RPAS used to conduct the flight.

Table 1. Imagery collection dates and RPAS used to conduct the flight.

Collection Date	Drone
September 24, 2018	DJI Phantom 3 Professional
November 24, 2019	DJI Phantom 4 RTK quadcopter
August 21, 2020	DJI Phantom 4 RTK quadcopter
July 25, 2021	DJI Phantom 4 RTK quadcopter
September 6, 2022	DJI Matrice 300 RTK quadcopter

Channel delineations were provided by a previous graduate student, Samantha Lewis. The channels were delineated using a semi-automated method outlined in Lewis (2022, p.37-50). The channel delineations that were used were associated with the same dates as the previously stated orthomosaic collection dates as the channels were derived from the same orthomosaic/DSM products. Since the channel delineations were rerun at a coarser scale for the purpose of this project than what was conducted in Lewis (2022), the channel classification shapefiles required manual cleaning in ArcGIS Pro. The shapefiles overlaid the associated orthomosaic and DSM and were assessed for consistency between the imagery and the delineation in terms of location and classification of whether the channel was an embryonic/small channel or a relic ditch/large channel.

DEMs of difference (DoDs) from 2019 to 2020 and 2020 to 2021 were also provided by Samantha Lewis. These raster datasets demonstrate changes in elevation over time to quantify yearly or seasonal accretion and erosion values by subtracting the previously collected DEM from the most recent DEM. Calculating DoDs from photogrammetric elevation models are outlined in detail in Lewis (2022, p.53-61). For the purpose of this study, all secondary datasets were clipped to the extent of the study area using the Clip tool for vector datasets and Clip Raster tool for raster datasets.

3.3 Data Processing

3.3.1 Photogrammetric Processing

Photogrammetric processing of aerial imagery was conducted with *Pix4Ddiscovery* (Pix4D; Version 4.5.6). The first phase was to perform initial processing which involved converting the image coordinate system to NAD83(CSRS)/UTM Zone 20N and leaving the vertical coordinate system as the default. Image processing was then checked to enable keypoint extraction. A median of 68708 keypoints per image were detected, allowing three-dimensional information to be generated. The greater the number of keypoints, the greater the accuracy of the orthomosaic and DSM products (Pix4D, 2017).

After the initial photo alignment, the GCP locations were imported into *Pix4Ddiscovery*. The centers of the targets were manually selected to optimize the geolocation of the imagery, improving the accuracy of their geographic location up to a couple of centimeters (Akyol, 2021). This enabled a point cloud to be generated which was performed by selecting the point density as 'optimal' and the number of matches to 3.

The final step in the image processing workflow was to build the DSM and orthomosaic. The 'Geotiff' and 'merge tiles' functions were checked off for each file and the resolution for each file was rounded up to 3.5 cm for export. These outputs were exported as TIF files.

3.3.2 Image Classification

Image classification is a process that assigns classes to each pixel in a remotely sensed image. This process was performed using a supervised object-based classification in ArcGIS Pro. This method of image classification was used instead of an unsupervised classification as supervised classification methods allow the user to choose representative sample pixels and train the image processing software to identify the desired classes.

The first step of the object-based classification was to run a segmentation on the 2022 orthomosaic imagery. The image segmentation process groups neighbouring pixels together based on similar spectral properties to inform the object-based classification, which then takes into account shape characteristics and neighbourhood relationships when assigning classes to the objects (Singh et al., 2021). The input parameters were manipulated until the desired level of detail was achieved. For example, increasing the spatial detail from 15 to 20 increased the visibility of smaller and more sparse patches of vegetation which were desired features in the output classification (*Figure 3.7*). The following parameters appeared to support the optimal segmented raster image: spectral detail = 19.00, spatial detail = 18.00, and minimum segment size in pixels = 2000.

Figure 3.7. Screenshot of the result of changing the spectral detail in the image segmentation process: A) spectral detail = 15.00; B) spectral detail = 20.00.

Within ArcGIS Pro, the Classification Wizard tool offers a streamlined workflow to navigate the classification process. The 2022 orthomosaic imagery and segmentation were brought into the tool and the landcover classes were defined by the generic National Land Cover Database (NLCD) 2011 classification schema offered by ArcGIS Pro. In the Training Samples Manager within the Classification Wizard tool, the NLCD 2011 classification schema was altered to match the 2021 habitat map classes which are outlined in *Figure 3.8a*. Training samples were then generated for each class by drawing at least 20 polygons around known features of the desired class across the study area (*Figure 3.8b*).

Figure 3.8. A) Classification schema selected for September 2022 imagery, B) Training samples generated from classification schema. Orthomosaic imagery was collected on September 6, 2022, by CBWES Inc.

The classifier page offers four different classification methods: ISO cluster, maximum likelihood, random trees, and support vector machine. The method used for this research was the support vector machine (SVM) as it is less susceptible to noise and an unbalanced number/size of training samples and widely used among researchers (ESRI, 2022). Random trees (RT) was investigated as an alternative method for the image classification process due to its popularity among other researchers (e.g. Akyol, 2020), though SVM appeared to provide more accurate results and displayed less noise in the data in the initial classification process (*Figure 3.9*).
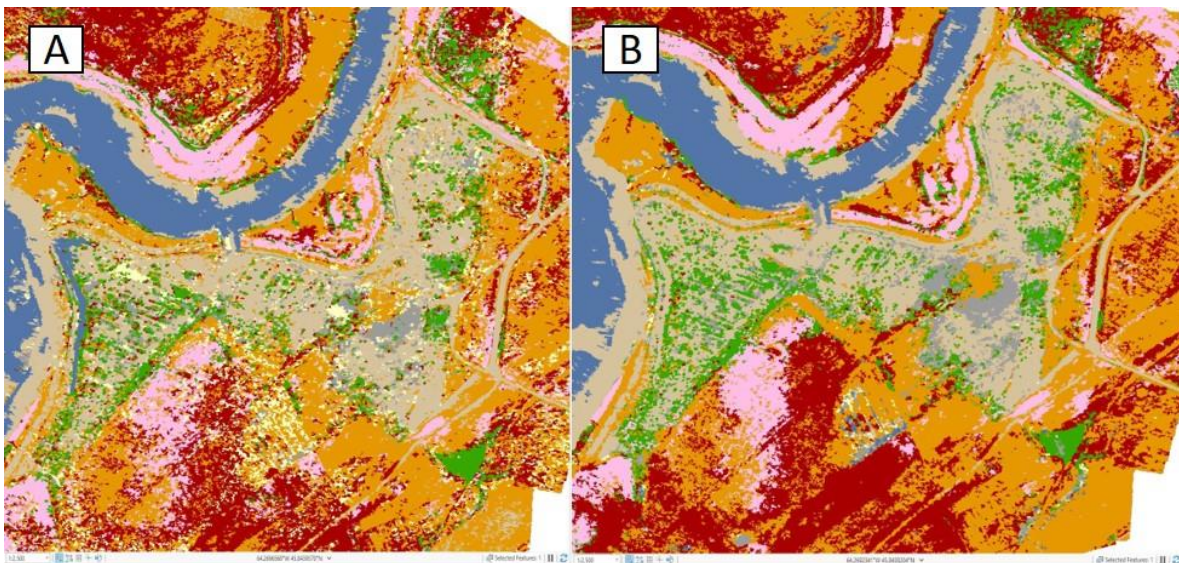
Figure 3.9. Comparison of different classifiers on the Converse imagery collected on September 6, 2022; A) Random Trees, B) Support Vector Machine.

Once the classification was completed within the classification wizard, the Raster To Polygon geoprocessing tool was run using the 'class_name' field. This allowed the classified raster to become individual vector polygons based on their associated class, allowing for greater ease of data cleaning. The Clip geoprocessing tool was then used to reduce the vector data to the extent of the 2021 habitat area (*Figure 3.10*).
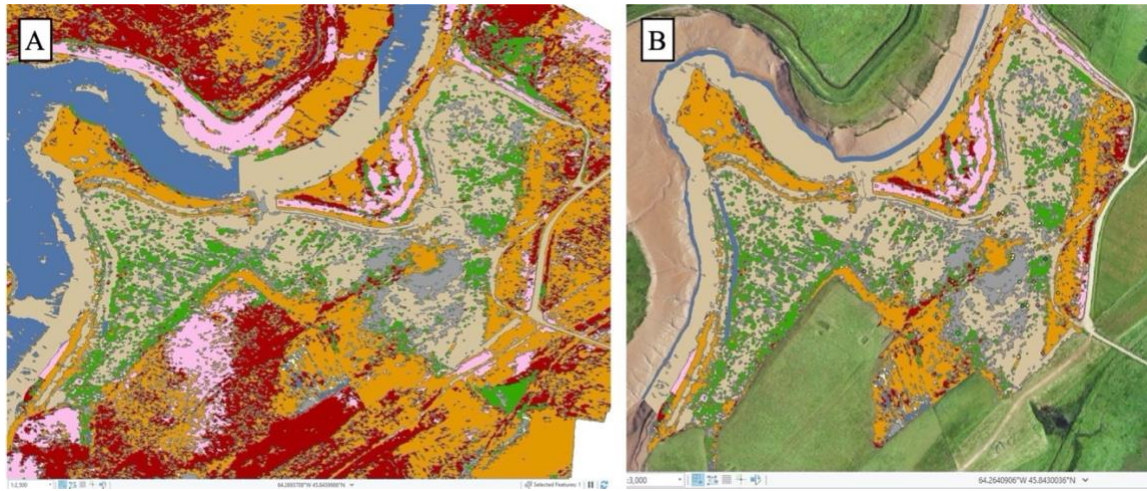
Figure 3.10. Result of clipping the raster to polygon output to the extent of the 2021 habitat data.

Vegetation data was digitized and analyzed for the dominant plant species at each survey station. The stations (n=40) and their associated dominant vegetation species were imported into the ArcGIS Pro project as a point layer to serve as ground truthing points (Appendix A). The class name of the classified polygons intersecting the vegetation plots were assessed for accuracy between the SVM assigned class and the actual vegetation data and adjacent polygons were assessed based on their spectral and spatial similarity to the assigned plots. Due to the amount of vegetation missed as well as misclassified and overclassified polygons that were found, the dataset required extensive manual cleaning (*Figure 3.11*).
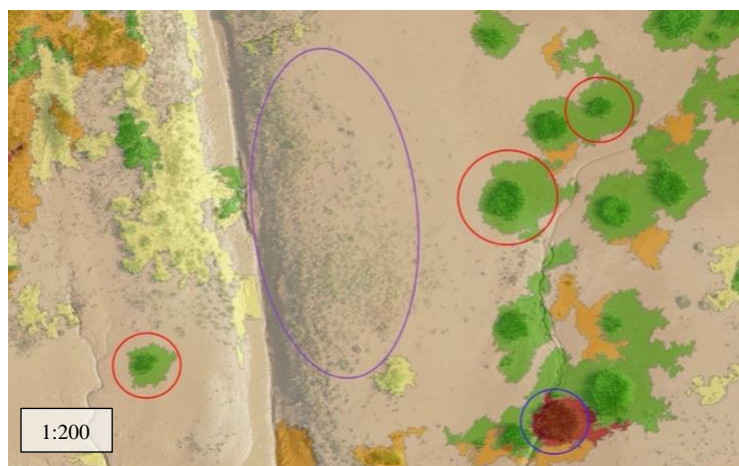
Figure 3.11. Examples of erroneous data within the classification, where red circles represent overclassified vegetation, blue circles represent misclassified vegetation, and the purple oval represents vegetation that was missed in the classification process.

Manual cleaning of the dataset was a time-consuming process that involved several tools. Where vegetation patches were misclassified, the polygons were selected which allowed their associated class name to be changed using the Calculate Field tool within the attribute table (*Figure 3.12*). This tool was also used to change all the 'dead material class' to 'early colonizers' as it was later realized that the vegetation represented by early colonizers (e.g. *Suaeda sp.*, *Salicornia sp.*) are annuals and would have been alive early in the season, but had perished by the time the flight was conducted. When polygons were too large in comparison to the patches of vegetation they were representing, the Split tool in the edit tab was used to manually outline the vegetation, and the remainder of the erroneous polygon was updated to the correct class. In areas that presented uncertainties due to the lack of nearby ground control points, species level classifications were either reduced to vegetation community (e.g. *Spartina pectinata* to brackish grasses) or the previous year's habitat map was used to inform the classes to serve as a quality assurance measure and reduce user bias.
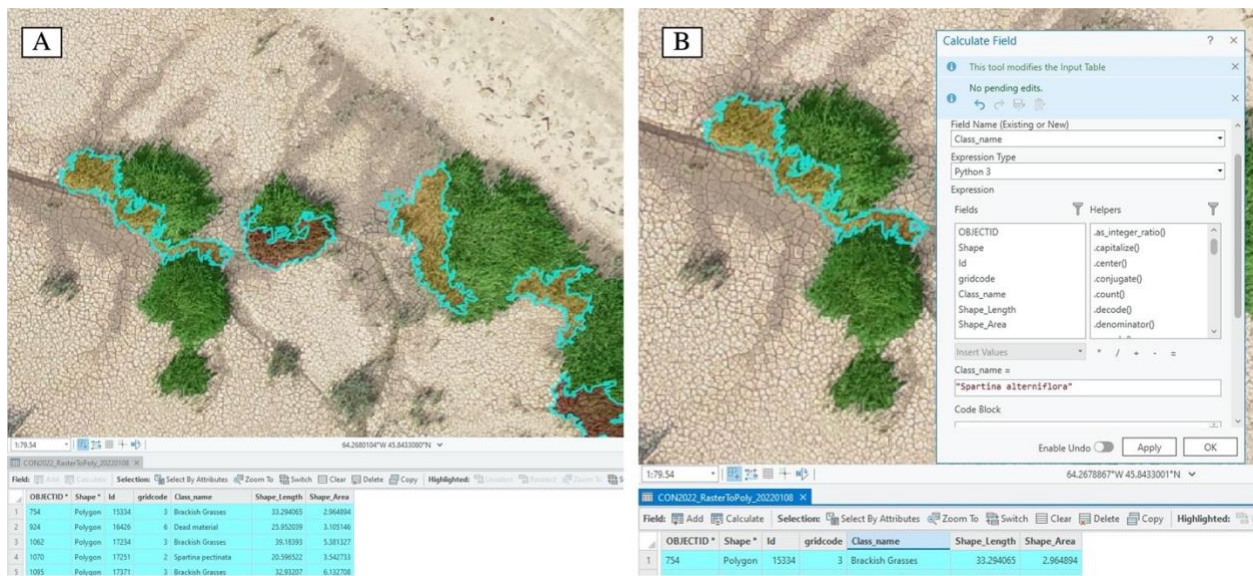
Figure 3.12. A) Misclassification of Spartina alterniflora, B) reclassifying the polygons using the calculate field tool.

The 2021 habitat classification was also used as a template for areas that were stable to reduce the amount of processing time. From site knowledge and assessing the orthomosaic imagery between 2021 and 2022, polygons were selected from the 2021 habitat shapefile to create another layer. The 2022 habitat layer was input into the Erase geoprocessing tool, using the selection from the 2021 layer as the erase feature, resulting in the removal of the existing polygons from 2022 that intersected the selected polygons from 2021 (*Figure 3.13*). The selected features were then copied to the clipboard and pasted into the 2022 habitat classification to create a continuous dataset.
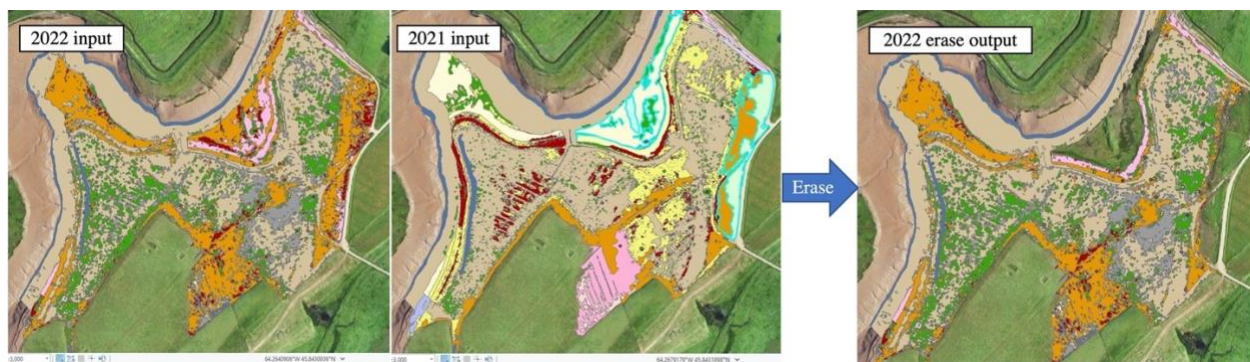
Figure 3.13. Schematic of the erase geoprocessing tool where the 2021 erase feature is used to omit the existing coincident 2022 input features.

The last step in the processing stage was to ensure that datasets were comparable. All habitat datasets were clipped to the extent of the study area and classes were merged or altered for greater ease of comparison (*Table 2*). Then, the areas of each record for each year were recalculated using the Calculate Geometry tool. This was a necessary step as the erase and clip tools do not update the reduced area of the polygons that only partially coincided with the input features.

Table 2. List of class names that were changed from 2019-2021 and their updated class name.

Initial Class Name	Updated Class Name
Sparse brackish grasses	Brackish grasses
Mixed colonizers	Early colonizers
Upland grasses	Remnant Agriculture
Silt covered upland veg	Dead material
Dead shrubs	Dead material

A summary statistic of each record in the attribute table was then calculated to summarize the area and frequency of each class for each year. This was achieved using the Summarize tool, with "Area_KN" as the field, "Sum" as the statistic type, and "Class_name" as the case field. The attribute table of the summary statistic and the raw data were both exported as .csv files to be used in further analyses.

3.4 Data Analysis

3.4.1 Habitat Community Structure

The tables derived from the Year 1 post-restoration to Year 4 post-restoration habitat classifications were used to evaluate and quantify how the habitat community structure changed over time. To standardize the data, each class was displayed as a percentage of the study site for each year using the following equation:

Equation 1. Calculation used to determine percent coverage of each class.

$$Percent\ coverage = \frac{Class\ area\ (m2)}{Sum\ of\ total\ area}\ x100$$

The resultant percent coverage of each class was then plotted on a stacked bar graph for ease of comparison.

3.4.2 Spatiotemporal Patterns of Vegetation Colonization

Spatial and temporal patterns of vegetation colonization were investigated by quantifying and mapping where new growth was occurring each year to be able to assess the rate of colonization and to visualize what patterns may arise. Persistence of vegetation colonization, across the marsh surface and at a species/vegetation community level, was also examined.

The first step of determining areas of new and persistent vegetation growth involved extracting the halophytic and brackish classes from the habitat shapefiles from each year post restoration. Year 1 post-restoration was omitted as there were no halophytes present in the study site. This was achieved by using the Select by Attribute tool and inputting the expression where "Class_name" is equal to the salt marsh applicable species (e.g. brackish grasses, early colonizers, *Spartina alterniflora*, *Spartina pectinata*, High marsh) (*Figure 3.14).* A new feature class was made for each year, containing only the salt marsh species/communities by using the Make Layer From Selection tool. These layers were then exported as shapefiles.
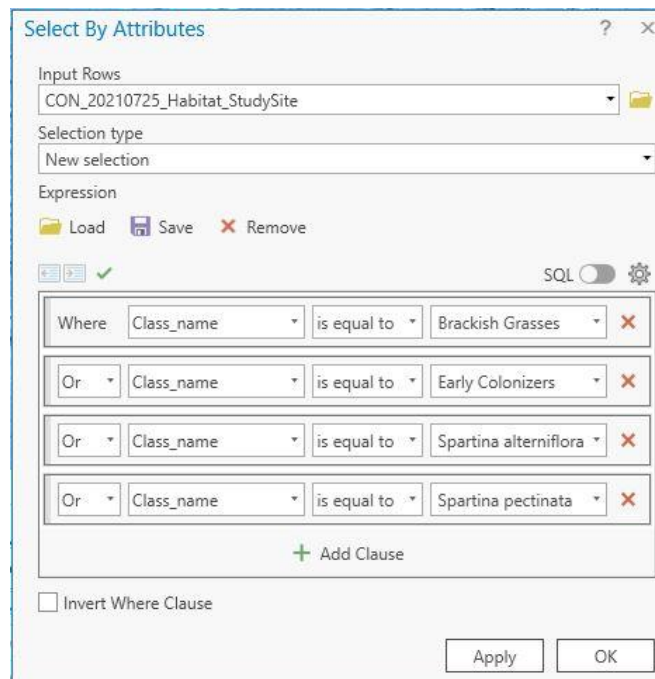
Figure 3.14. Example of the inputs used in the Select By Attributes tool to isolate the halophytic and brackish communities.

To determine where the salt marsh vegetation was persisting each year, the feature classes containing only the salt marsh vegetation from two consecutive years were inputted into the Intersect geoprocessing tool. The parameters were set to join all attributes and to maintain the input type in the output. The resultant polygons from the output feature class demonstrated where vegetation persisted spatially from 2020 into 2021, and 2021 into 2022. A new area field was added to each feature class, then the Calculate Geometry tool was used to populate the field with the updated area of each feature. Next, the Summarize tool was used on each persistent vegetation class from each year by setting the field as "Area", statistics type as "Sum" and the case fields as "Class_name_(YEAR1)" and "Class_name_(YEAR2)". The results of the outputs were tabulated, which allowed the area of the persistent vegetation to be quantified and to develop potential futures matrices by assessing how species/communities changed in consecutive years.

New growth was determined from 2020-2022 using the extracted halophytes feature classes in the first step and the persistent halophytes feature classes. All halophytes from 2020 represented new colonization, therefore the extracted salt marsh species in the first step represented new growth. New colonization/expansion was determined in 2021 by inputting the 2021 halophyte feature class as well as the 2021 persistent halophytes feature class into the Erase tool. This essentially subtracted the persistent vegetation from the total halophytes in the study area, leaving only new colonization in the output layer. An area field was then added and populated using the Calculate Geometry tool. This process was repeated using the 2022 datasets to extract the new growth/colonization that occurred in 2022. These data were then mapped to assess where the persistent and new vegetation classes were occurring spatially and to visualize the general spatial patterns of vegetation colonization (*Figure 3.15).*
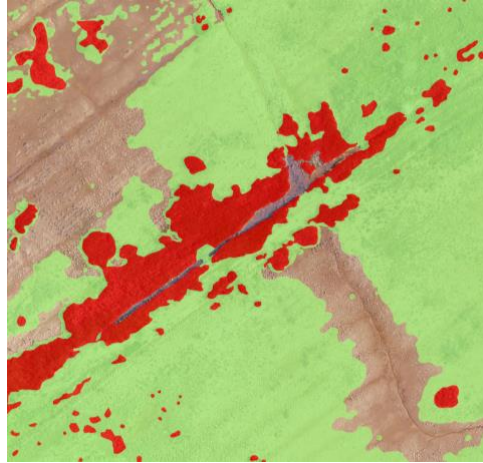
Figure 3.15. Example of the intersect output in ArcGIS Pro, with red polygons representing persistent vegetation into 2021 and green polygons representing new colonization in 2021.

3.4.3 Relationship Between Topographic Features and Vegetation Colonization

Topographic features refer to different properties on the surface of a terrain. For the purpose of this study, topographic features refer to relic ditches (> 2 cm depth), proto channels (< 2 cm depth), and the amount of sediment accretion or erosion that characterizes an area within the study site. These parameters were selected as channels characterize low-lying areas, whereas sediment accretion data allowed higher elevation areas to be investigated for a more complete assessment of spatial relationships between vegetation colonization and topographic features.

Channel delineation shapefiles were paired with the new colonization shapefiles to analyze the relationship between vegetation colonization and channel networks from 2020-2022. For this analysis, the 'Near' geoprocessing tool within ArcGIS Pro was used to calculate a Euclidian distance between the new colonization features and the closest feature in the channel network feature class. Once the data was tabulated, the average distance between the channels and each class, as well as the range of values, standard deviation, and any outliers within the dataset was able to be determined.

The relationship between vegetation colonization and accretion and erosion rates was also investigated by comparing new colonization and DoD values. Since the first steps of

calculating a DoD requires masking the vegetated areas within the DSMs to create DEMs, the

new colonization shapefiles had to be paired with the previous year's DoD to have valid bare

ground data beneath the vegetation features e.g., the 2021 new colonization layer overlaid the

2019-2020 DoD. Therefore, a comparison was only able to be observed for two growing seasons

(2021 and 2022). First, the DoD was clipped to the study area using the Clip Raster tool, then the

Zonal Statistics as Table tool was used to calculate all statistics available in the tool. This

included count, area, min, max, range, mean, standard deviation, sum, and median (Appendix D),

from the inputs outlined in *Table 3* to produce a summary of these statistics for each class.

Table 3. Inputs used in the Zonal Statistics as Table tool.

Input raster or feature zone data:	NewHalophytes_(YEAR)
Zone field:	Class_name
Input value raster:	CON_(timeframe)_DoD_BareGround_Study
Statistics type:	All
Percentile values:	90 (default)
Percentile interpolation type:	Auto-detect (default)

The data was exported and brought into excel where the accretion values were converted

from meters per year to centimeters per year. The derived mean values were used to represent the

accretion rates for each class for 2021 and 2022, and the standard error (SE) was calculated using

the following equation:

Equation 2: Standard error equation.

$$SE = \frac{STD}{\sqrt{sample\ size}}$$

Where STD is the standard deviation, derived from the output of the Zonal Statistics as Table

tool and the sample size was obtained from the summary data from the new colonization tables.

Chapter 4: Results

4.1 Habitat Community Structure

Generating habitat maps and class areas of the restoration site allows for ease of visual comparison of community structure year to year. In 2019 (Year 1 post-restoration), the Converse Marsh was dominantly characterized by bare ground (~85%), with remnants of its agricultural identity persisting (~15%) (*Figure 4.1a, Figure 4.2*). In 2020 (Year 2 post-restoration), bare ground still remained dominant in the site (~81%), though the agricultural vegetation had largely been replaced by *Spartina pectinata* and dead material (*Figure 4.1b*). Notably, this year offered the initial colonization of halophytic species and communities including *Spartina pectinata* (4%), *Spartina alterniflora* (3%), early colonizers (*Suaeda sp., Salicornia sp.*)(<1%), and brackish grasses (*Juncus gerardii, Elymus repens*) (1%), though 10% of the study site was still characterized by dead vegetation (*Figure 4.2*). Year 3 post-restoration (2021) offered an approximate 40% reduction in bare ground area. There appeared to be rapid colonization of halophyte species, notably *Spartina alterniflora* (5%) colonization in the center and northwestern section of the site and bare ground dominance shifted toward early colonizer dominance (30%) (*Figure 4.1c, Figure 4.2*). Year 4 post-restoration (2022) demonstrated a reduction of early colonizers (15%) and increase in the target species, *Spartina alterniflora* (15%) (*Figure 4.1d, Figure 4.2*). It is also notable that a more even distribution of halophytic species and communities presented themselves in Year 4 compared to Year 3, though bare ground increased slightly (~2%) between years (*Figure 4.2).*
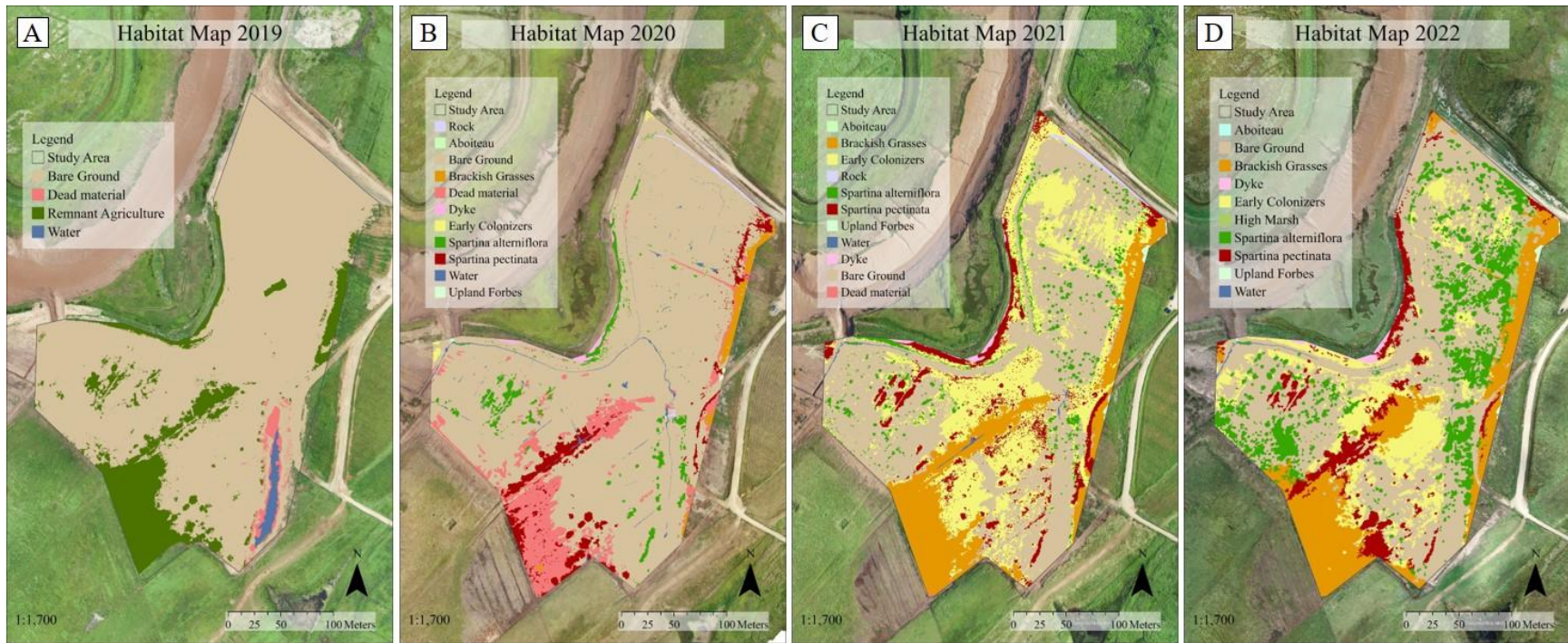
Figure 4.1. Habitat map of the study area within the Converse Marsh from 2019 (Year 1 post-restoration to Year 4 post-restoration).
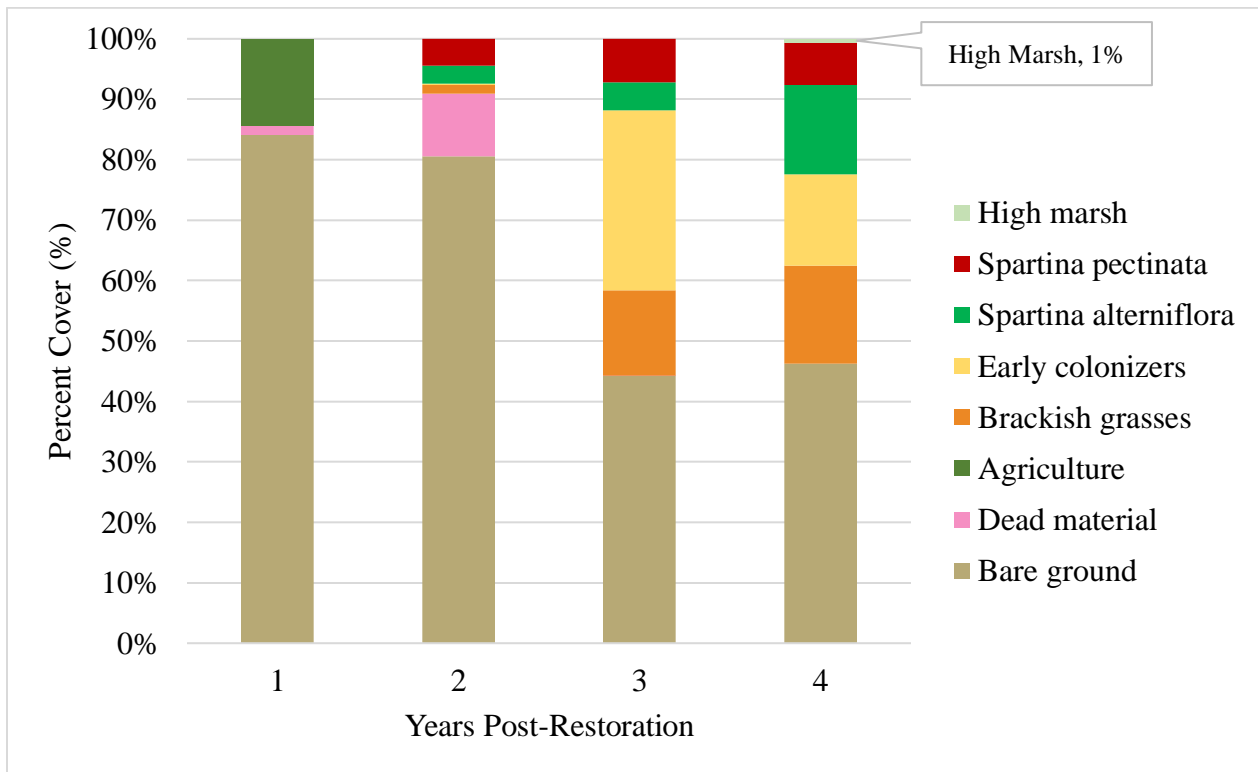
Figure 4.2. Stacked bar chart representing the percent cover of each class within the study area from year 1 post-restoration to year 4 post-restoration.

4.2 Spatiotemporal Patterns of Vegetation Colonization

Spatial and temporal patterns of vegetation colonization were assessed in two ways. First, by examining where halophytes persisted or exhibited new colonization spatially, then by assessing if the halophytes persisted from one year to the next at a species/community level.

4.2.1 Vegetation Persistence and New Colonization

Once halophytic vegetation was extracted from the shapefiles, it enabled a comparison of bare ground versus new growth versus persistent growth. Year 1 post-restoration exhibited 0% colonization of halophytic vegetation (*Figure 4.3*). The following year, 10% of the respective bare ground and halophytic vegetation area was colonized by new halophytic growth, though 90% of the area remained bare (*Figure 4.3, Table 4).* Year 3 post-restoration exhibited the

greatest abundance of new colonization of each year from Year 1 to Year 4 (*Figure 4.3*). This

was demonstrated as Year 3 had approximately ~5 times the abundance of new halophytic

colonization when compared to the previous year, resulting in 47% of the respective area being

characterized by new growth, 44% bare ground, and 9% of the Year 2 halophytes persisting

spatially into Year 3 (*Figure 4.3, Table 4*). Year 4 post-restoration was characterized by the

greatest abundance of persistent halophytes (36%) of all years (*Figure 4.3, Table 4*). Given that

the persistent or new colonization area relative to bare ground area represented 56% in Year 3

and 36% of the halophytic vegetation persisted in Year 4, this suggests that 20% of the Year 3

halophytes did not persist spatially into Year 4 *(Table 4)*. However, the cumulative area of

halophytes was greater in Year 4 when compared to Year 3, and bare ground was reduced to

41% (*Figure 4.3, Table 4),* indicative of new growth in other areas of the site. This was

supported by the 23% area representing new colonization of halophytic vegetation in Year 4

(*Table 4*).

Table 4. Summary of values representing the abundance of bare ground, persistent halophytes,
and new halophytes (new growth) from year 1 post-restoration – year 4 post-restoration.

	Years Post-Restoration			
Class abundance (%)	**1**	**2**	**3**	**4**
New halophytes	0%	10%	47%	23%
Persistent halophytes	0%	0%	9%	36%
Bare Ground	100%	90%	44%	41%
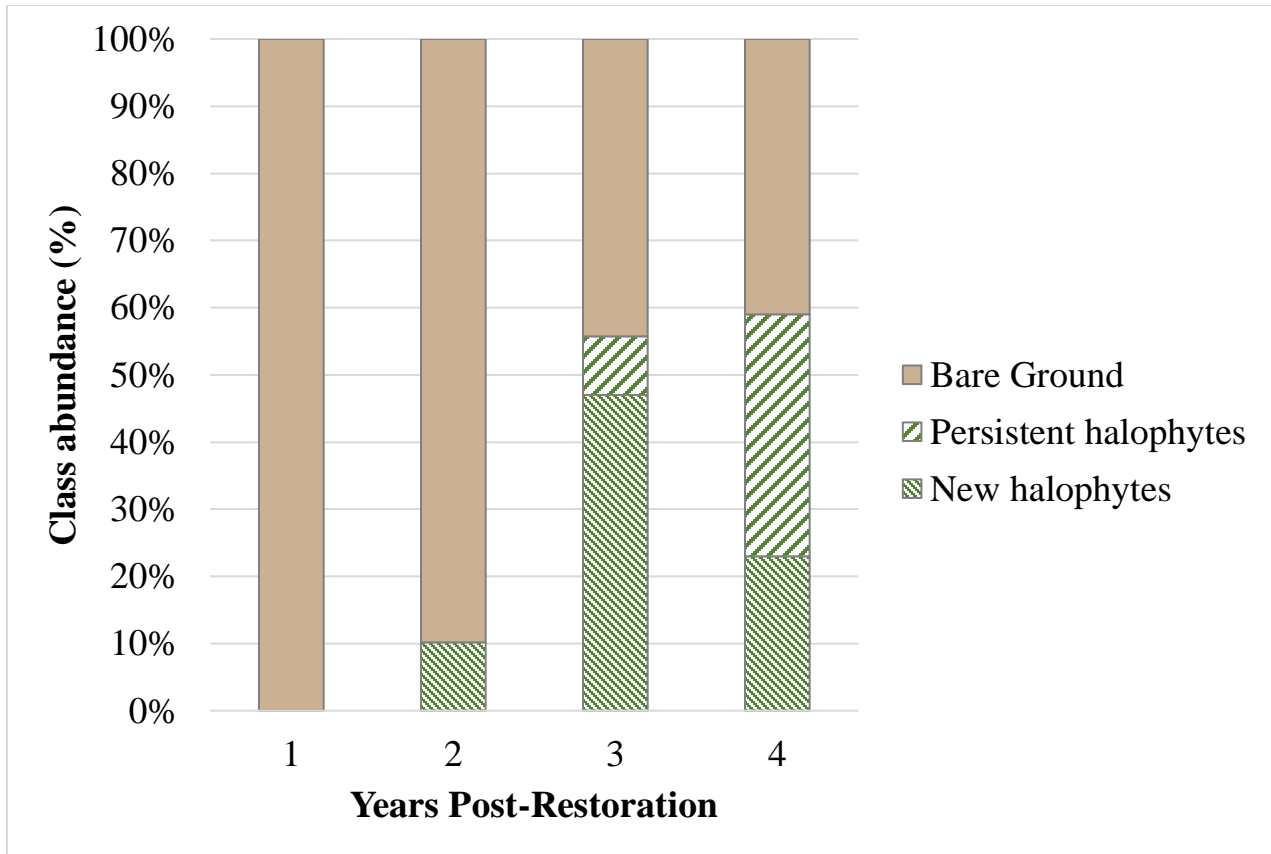
Figure 4.3. Stacked bar graph representing the abundance of bare ground, persistent halophytes, and new halophytes (new growth) from Year 1 post-restoration – Year 4 post-restoration.

Initial colonization of halophytic vegetation in 2020 appeared in higher elevation areas, particularly toward the outer bounds of the study site, depicted by the dark green polygons outlined in *Figure 4.4*. Notably, *S. pectinata* appeared to grow dominantly in a linear pattern, particularly adjacent to ditches along the west side of the site (*Figure 4.4*). New growth and colonization the following year demonstrated greater variability in patches. Expansive patches were notable in the center of the site which were dominantly characterized by early colonizers (*Figure 4.4*). There had also been evidence of clonal spread, particularly by *Spartina alterniflora* as patches in 2020 developed rings of new colonization around them in 2021. In the eastern section of the site, there appeared to be linear spread of halophytic vegetation along relic agricultural ditches (*Figure 4.4*). Year 4 post-restoration (2022) exhibited limited new

colonization. However, it was notable that small patches of *S. alterniflora* tended to merge and form larger patches of dense monocultures in the center and western areas of the study site (*Figure 4.4*).
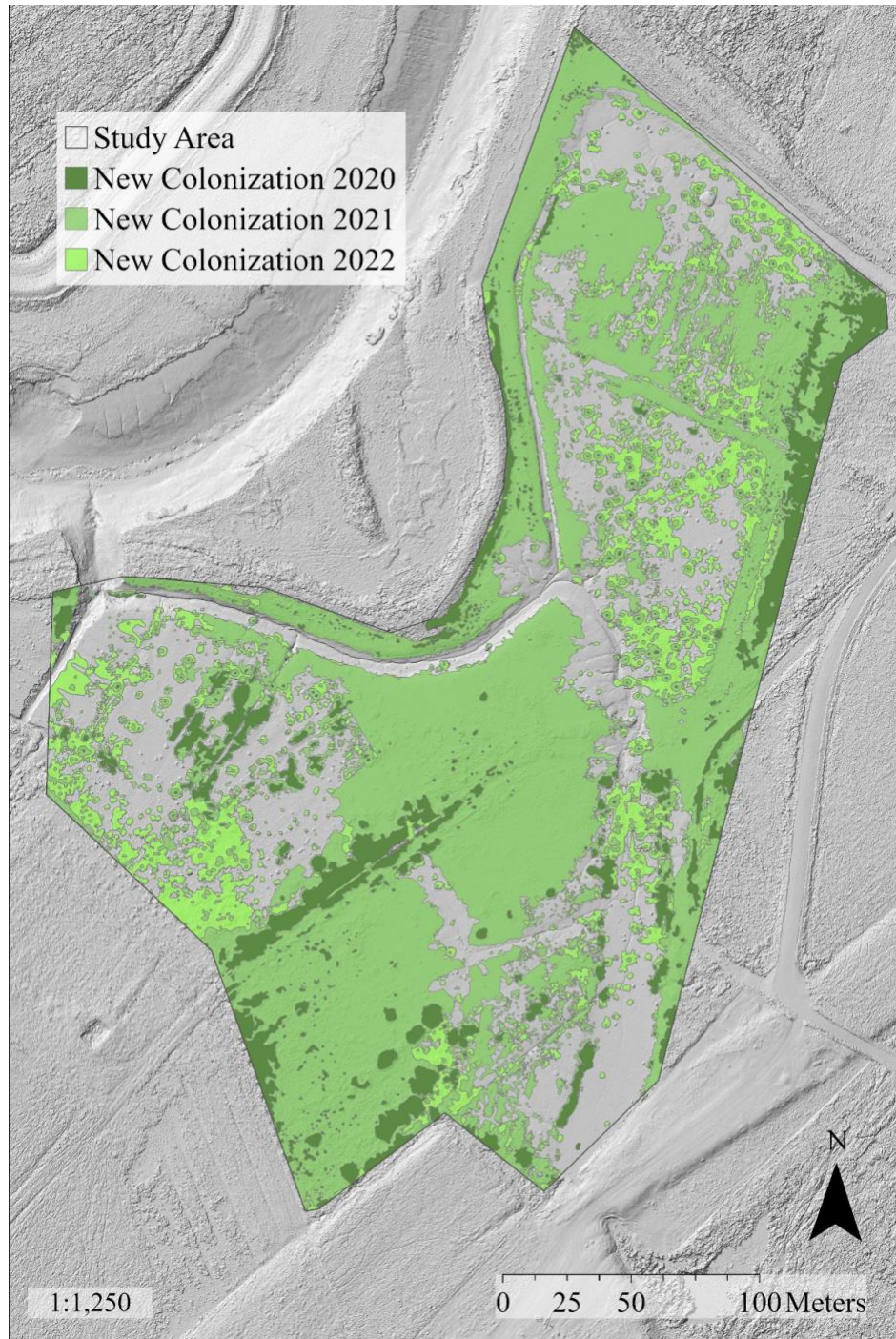
Figure 4.4. Colonization of halophytes/brackish communities from 2020-2022. Elevation data collected on September 6, 2022 using DJI Matrice 300 RTK RPAS by CBWES Inc.

4.2.3 Potential Futures

By assessing the potential futures of each halophytic class, the limitations of the previous analyses were addressed by assessing if the persistent vegetation remains the same class the following year. In *Figure 4.5,* it is evident that polygons characterized by brackish grasses remained brackish grasses, though *S. alterniflora* and *S. pectinata* were replaced by several other classes in their respective spatial area from Year 2 to Year 3 post-restoration. Between Year 3 and Year 4, the areas characterized by a certain class either remained the same or were colonized or outcompeted by a different species/community (*Figure 4.5*).
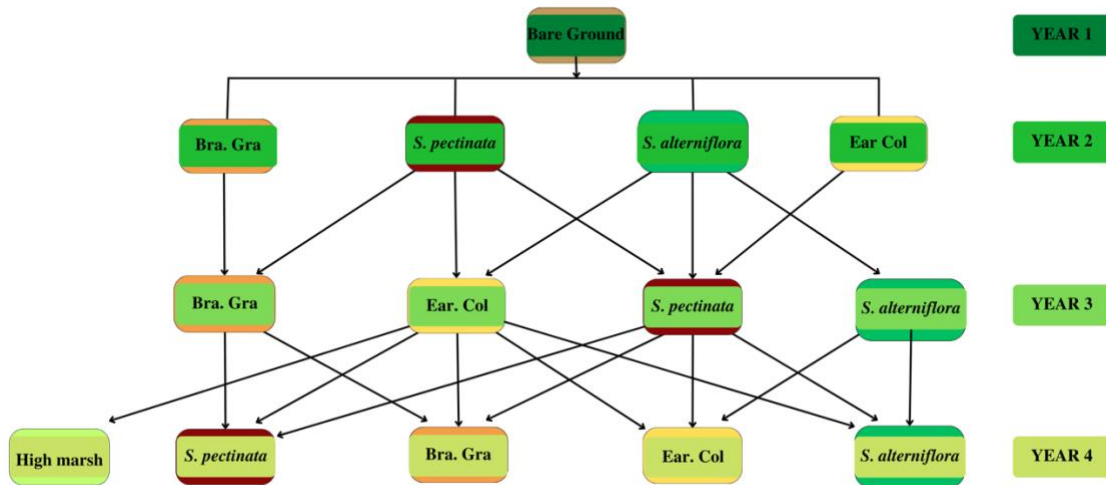
Figure 4.5. Schematic of the potential futures of each species from year 1 post-restoration to year 4 post-restoration derived from the output of the persistent vegetation analyses. Percentages of species transitioning from one to another <1%, were omitted.

The output of the persistent vegetation analysis enabled a quantitative comparison of these changes as well. Between Year 2 and Year 3 post-restoration, brackish grasses had the greatest area of each class that remained the same class, representing 15.2% of halophytic communities that persisted the following year (*Table 5a*). It is also notable that 41% of the persistent vegetation represented *S. pectinata* being replaced by the brackish grasses community and 18.8% represented *S. alterniflora* being replaced by *S. pectinata* (*Table 5a*). Other

species/communities that represented less than 10% of the persistent vegetation per class included early colonizers replaced by *S. pectinata* (1.7%), *S. alterniflora* replaced by early colonizers (6%), and *S. alterniflora* (7.8%) and *S. pectinata* (6.7%) persisting in the same spatial area the following year.

Between Year 3 and Year 4 post-restoration, the largest percentage of persistent vegetation was characterized by classes remaining the same class the following year (*Table 5a*). Brackish grasses remained the class with the greatest area that remained the same class the following year, representing 28.3% of halophytic colonization that persisted the following year (*Table 5b*). This is almost double the amount of persistent vegetation representing brackish grasses between 2020 and 2021 (*Table 5a*). It is also notable that early colonizers appeared to be replaced by other classes at a relatively equal rate across classes in Year 4, though they are being replaced by more classes between Years 3 and 4 than in Years 2 and 3 (*Table 5a, Table 5b*).

Table 5. The representative percentage of each halophytic class transitioning from one species to another or remaining the same from a) 2020 (y-axis) – 2021 (x-axis); b) 2021 (y-axis) – 2022 (x-axis). Species are abbreviated as follows: Bra. gra = brackish grasses; Ear. col = early colonizers; Spa. pec = Spartina pectinata; and Spa. alt = Spartina alterniflora.

a)		2021				
	Area (%)	**Bra. gra**	**Ear. col**	**Spa. pec**	**Spa. alt**	**High Marsh**
2020	*Bra. gra*	15.2	0.0	0.0	0.0	0.0
	Ear. col	0.0	0.2	1.7	0.0	0.0
	Spa. pec	41.5	1.1	6.8	0.8	0.0
	Spa. alt	0.1	6.0	18.8	7.8	0.0
b)		2022				
	Area (%)	**Bra. gra**	**Ear. col**	**Spa. pec**	**Spa. alt**	**High Marsh**
2021	*Bra. gra*	28.3	0.8	4.6	0.2	0.0
	Ear. col	6.1	24.7	4.2	5.8	1.0
	Spa. pec	3.2	4.5	6.4	1.0	0.0
	Spa. alt	0.3	1.4	0.3	6.8	0.4

4.3 Relationship Between Vegetation Colonization and Topographic Features

The relationship between vegetation colonization and topographic features was assessed in two ways. First, by analyzing where halophytes were colonizing in relation to channel networks each year. Next, by extracting the average sediment accretion values from DoDs associated with the coincident spatial area of the halophytic classes that colonized the following year.

4.2.1 Halophyte Colonization and Channel Networks

The near analyses demonstrated there was not a strong correlation between where halophytes were colonizing in relation to channel networks. Due to the lack of halophytic colonization in 2019 (Year 1 post-restoration), it was not used in this analysis. Year 2 post-restoration demonstrated a more prominent relationship between colonization and channel networks than in consecutive years due to the greater variability associated with the distance values of each species/community (*Figure 4.6*). However, it is notable that *S. pectinata* appeared to remain within a 2-12 m distance from channel networks while *S. alterniflora* and early colonizers were mostly within a 10 m distance from channel networks each year (*Figure 4.6*).

Due to the lack of consistent general patterns demonstrated in *Figure 4.6* and *Figure 4.7*, the correlation between vegetation colonization and channel networks is not strongly supported. Though it is evident that *S. alterniflora* is consistently proximal to channels (*Figure 4.6*) which is shown more prominently in the center and northern section of the study site, outlined by the circular patches with growth rings in *Figure 4.7C*.
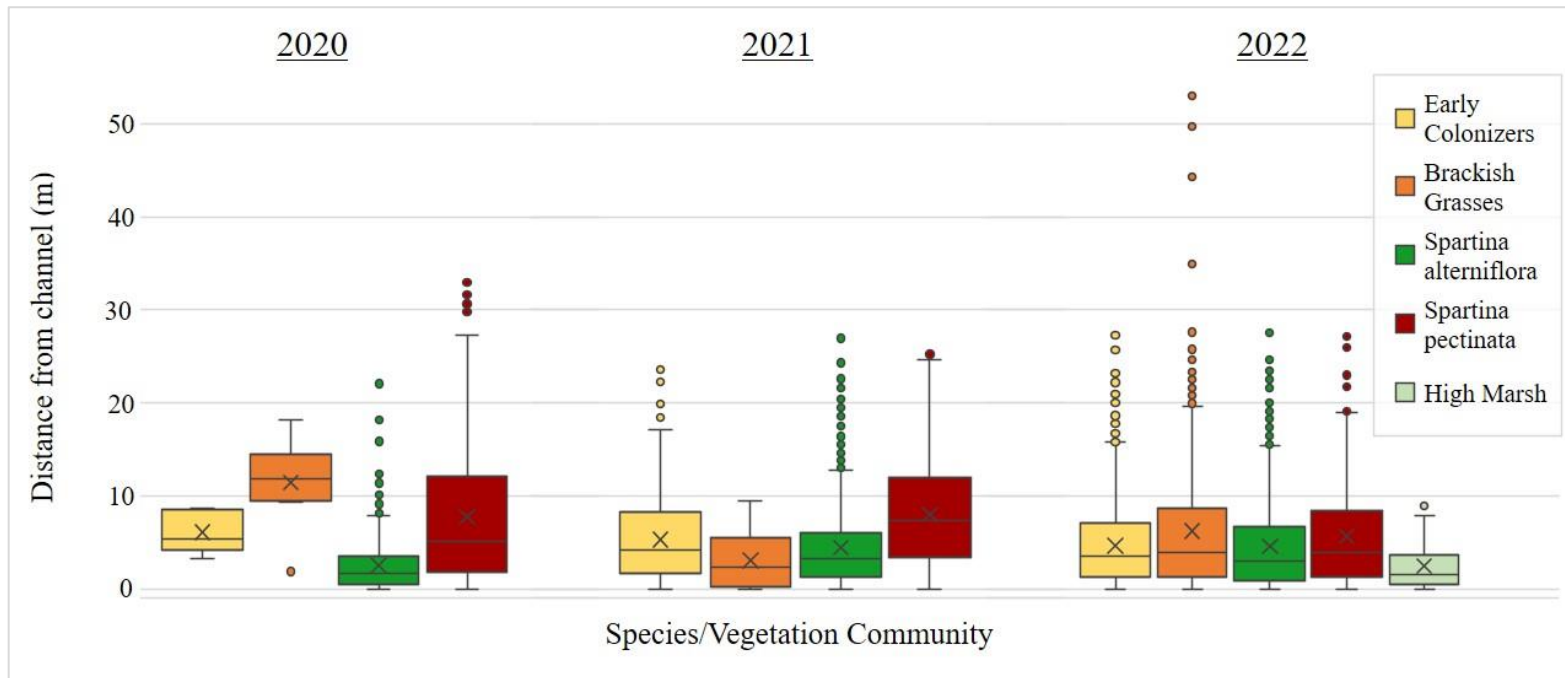
Figure 4.6. Box and whisker plot representing the distance (m) of each class relative to channel networks from 2020-2022
.

Figure 4.7. Relationship between channel networks and vegetation colonization in A) 2020; B) 2021; C) 2022. Light green represents new colonization, dark green represents persistent vegetation, solid red lines represent channels and dashed red lines represent protochannels.

4.2.2 Halophyte Colonization and DoDs

Halophyte colonization in relation to DoD values was assessed to determine if a correlation existed between sediment accretion and subsequent halophytic colonization. The results of this analysis showed that aside from the early colonizers, each species/community in 2021 colonized areas with lower average accretion values between 2019 and 2020 than the following year (*Figure 4.8*). It is also notable that *S. alterniflora* coincided with higher accretion rates, while brackish grasses coincided with lower accretion rates than other classes (*Figure 4.8*). *S. pectinata* and early colonizers appeared to colonize areas of mid-range sediment accretion between ~1.5-3 cm/year (*Figure 4.8*). Remarkably, high marsh (dominantly characterized by *S. patens*) that colonized in 2022 was associated with higher average rates of accretion (5.50 cm/year), though the standard error is quite large due to the small frequency of high marsh polygons and is therefore likely an insignificant result (*Figure 4.8; Table 6*).

Table 6. Frequency and representative area of each halophytic class in 2021 and 2022.

	2021		2022	
Class Name	**Frequency**	**Area (m^2)**	**Frequency**	**Area (m^2)**
Early Colonizers	1005	20677.3	2011	1682.7
Spartina alterniflora	1311	2689.7	1625	13001.8
Spartina pectinata	1650	3439.3	370	1046.4
Brackish Grasses	30	6420.9	997	2684.2
High Marsh	NA	NA	3	0.5

	Early Colonizers	Spartina alterniflora	Spartina pectinata	Brackish Grasses	High Marsh
2021 MEAN	2.36	3.56	1.83	0.39	NA
2022 MEAN	1.74	5.56	2.83	1.85	5.50
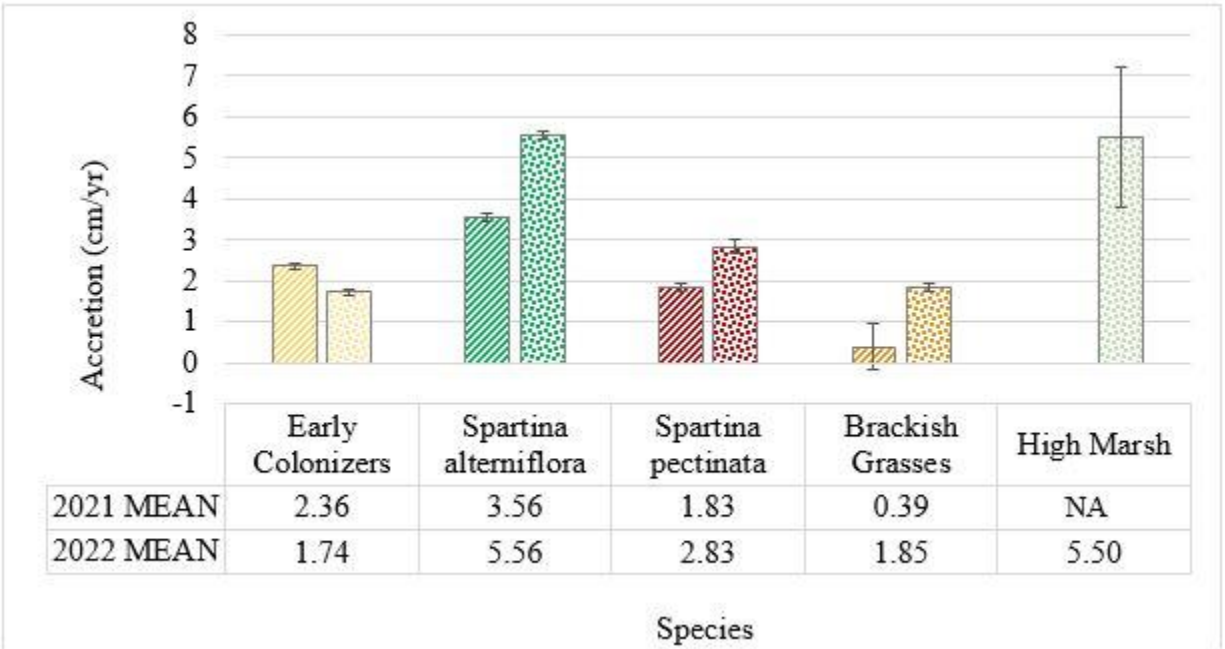
Figure 4.8. Average accretion rate extracted from the previous year's DoD for each halophyte/halophytic class representing new colonization in 2021 (hatched fill) and 2022 (dotted fill), with standard error bars.

Chapter 5: Discussion

5.1 Habitat Community Structure

The development of the habitat community structure at the Converse restoration site aligns well other research that has been conducted in the Bay of Fundy (Norris et al., 2022; van Proosdij et al., 2023; Virgin et al., 2020). In the Aulac managed realignment site in New Brunswick, four stages of vegetative community succession were observed. Namely, (i) initial rapid deposition of unconsolidated sediment and loss of terrestrial vegetation with remnants of *Spartina pectinata* persisting (1 y post-breach), (ii) colonization and spread of *S. alterniflora* and loss of *S. pectinata* (2–5 y post), (iii) high percent cover and decreased spatial variability of *S. alterniflora,* and (iv) maturation of sediments and encroachment of high marsh (Virgin et al., 2020). Recognizing the stages in which the restoration site is in over time enables a better understanding of if the site is on a positive trajectory towards reference conditions and if adaptive management is required to facilitate salt marsh development.

The first- and second-year post-restoration at Converse was characterized by limited colonization of target halophytic species with remnants of the site's agricultural vegetation persisting. The site was also dominantly characterized by bare ground resulting from initial deposition of unconsolidated sediments. Therefore, Year 1 and Year 2 post-restoration aligns with the initial successional stage outlined by Virgin et al. (2020), as the agricultural vegetation persisted in the first-year post-breach and the consecutive year experienced the initial colonization of *S. alterniflora*. However, Year 2 post-restoration may have represented an intermediate phase between phase one and phase two due to the colonization of *S. alterniflora* and *S. pectinata* in tandem.

The initial low halophytic abundance early in the restoration trajectory is well supported in the literature (van Proosdij et al., 2023; Virgin et al., 2020) and was recognized at the Converse restoration site. This may be due to the decomposing layer of dead vegetation and wet conditions that gave rise to an anoxic environment (Mossman et al., 2012). Inappropriate conditions can hinder vegetation establishment (Erfanzadeh et al., 2010) or result in a less diverse salt marsh with upper marsh species underrepresented (Brooks et al., 2015). The wet, unconsolidated sediment on the marsh surface potentially arise due to borrow pits that can create drainage issues (van Proosdij et al., 2023) which were created at the Converse restoration site before breaching the dyke. The subsequent burial of the terrestrial vegetation, which occurred in Year 1 and Year 2 post restoration, is an important mechanism for creating a clean slate for plant recolonization (van Proosdij et al., 2023). This may be due to the large resultant area of bare soil that provide low rates of interspecific competition, ideal for early colonization (Erfanzadeh et al., 2010). As the site accreted sediment and became more stable, it created more favourable conditions for colonization, which was more prominent in Year 3 post-restoration.

Year 3 post-restoration appeared to be the pivotal year in the restoration trajectory as species richness and abundance increased greatly. Early colonizers such as *Suaeda* sp. dominated the site and bare ground area was reduced by nearly half. Salt marsh annuals (*Suaeda* sp. and *Salicornia* sp.) have been recorded as the earliest colonists in other studies (Brooks et al., 2015; Erfanzadeh et al. 2010; Wolters et al., 2008), though they had colonized and were abundant by Year 2 (Mossman et al., 2012a; van Proosdij et al., 2023). This may represent a time lag in halophytic colonization at the Converse restoration site in comparison to some managed realignment sites. However, other studies such as Wolters et al. (2008), conducted in south-east

England, had similarly noted the initial colonization and spread of perennial species in Year 3 post-restoration which was demonstrated at Converse.

The expansive patches of annual halophytes represent primary salt marsh succession. Their presence promotes sediment accretion and facilitates the establishment of target perennial halophytes that characterize later successional stages (Mossman et al., 2012). This was demonstrated in Year 4 post-restoration as *S. alterniflora* represented a larger portion of the site and had begun to compete with *Suaeda sp*. in the low marsh zone *(Figure 5.1)*. As a result, Year 4 post-restoration potentially represented a variation of the third stage of vegetative community succession outlined by Virgin et al. (2020). This was shown due to the mature consolidation of sediments and *S. alterniflora* had become more well established, though remained in equal abundance to the early colonists. It is also notable that high marsh had begun to encroach in the outer bounds of the site in Year 4, characteristic of the fourth phase of the vegetative community succession in managed realignment sites (Virgin et al., 2020). This shows that the Converse restoration site achieved each successional stage within a 4-year time frame and is on a positive trajectory towards complete restoration.

Figure 5.1. Spartina alterniflora colonizing among expansive patch of early colonizers (mostly Suaeda sp.) on August 2, 2022.

5.2 Spatiotemporal Patterns of Vegetation Colonization

Salt marshes are dynamic in nature, demonstrating non-linear behaviour as vegetation dynamics change spatially and temporally. This was shown through evidence of variable dispersal mechanisms, different patch size and distributions, as well as changes in species interactions (e.g., competition or coexistence).

Year 2 post-restoration demonstrated initial colonization of halophytic species. Primarily, *S. alterniflora, S. pectinata*, and early colonizers (annuals), and their mechanism for entering the site and establishing likely differed. The composition of the reference site for Converse, positioned directly adjacent to the site on the other side of the northern dyke, is largely characterized by mature patches of *S. patens* and smaller patches of *S. alterniflora*. Given the proximity of the reference site, it is likely that *S. alterniflora* colonized the site by seed transport. This was supported by the small circular patches that were present on the mudflat in the center of the site. However, it is notable that there were also larger patches of *S. alterniflora* that had not been present the previous year, suggesting rhizome material of *S. alterniflora* was deposited via winter ice blocks (Rabinowitz et al., 2022; van Proosdij et al., 2010; van Proosdij and Townsend, 2006) or tides (Virgin et al., 2020) which is a common phenomenon in the Bay of Fundy. Both dispersal mechanisms of *S. alterniflora* can result in random initial colonization patterns.

There were similar initial spatial patterns of *S. pectinata* colonization when compared to *S. alterniflora* in Year 2 post-restoration. From the 2019 and 2020 habitat maps, *S. pectinata* appeared to colonize only in Year 2, however, there were localized patches observed in Year 1 post-restoration which had potentially survived the breach or represented new colonization (Rabinowitz, 2020). Since the dispersal of *S. pectinata* via ice blocks is not represented in the

literature, it is possible that the species was initially present in low quantities on the site which were not detected in the RGB imagery. *S. pectinata* has survived breaching in other managed realignment sites in the Bay of Fundy (Norris et al., 2022; Virgin et al., 2020), so seed from pre-existing vegetation was likely responsible for the larger patches of colonization in Year 2.

The establishment of early colonizers in Year 2 demonstrated expansive patches across the marsh surface. Given the limited availability of seed from the nearby reference site, it is likely that early colonizers were initially brought into the site via ice blocks or tide from the larger Tantramar marsh system. This is suggested as early colonizers are massive seed producers with high viability and buoyant characteristics (Erfanzadeh et al., 2010).

Consecutive years of colonization by *S. alterniflora* showed a shift from sexual reproduction to asexual reproduction dominance in later years. This result agrees with the results of Silvertown (2008) and Erfanzadeh et al. (2010) and opposes the results of Norris et al. (2022). There were several indicators of clonal spread of vegetation, particularly by *S. alterniflora.* First, there were patches that grew larger surrounding rings of vegetation in consecutive years (van Proosdij and Townsend, 2006). Next, the prominent straight lines emerging from the mature patches of *S. alterniflora* (*Figure 5.2)* would eventually merge with nearby patches to form monospecific populations (Norris et al., 2022).

The trends observed in *S. pectinata* differed from *S. alterniflora* as it appeared to grow in more linear patches on the marsh surface and eventually began to interact with the brackish grass community and early colonizers in Year 3 and Year 4. This was noted as *S. pectinata* appeared to replace brackish grasses in the central portion of the site but was replaced by brackish grasses towards the bounds of the site and early colonizers in lower elevation areas.

Figure 5.2. Mature patches of S. alterniflora beginning to merge via rhizome in August 2022.

These results further support the Converse restoration site achieving later successional stages as the species interactions increase over time and halophytic vegetation persistence increased over time. As the stability of the site increased, demonstrated by vegetation persistence and sediment consolidation, more dynamic changes and shifts from one class to another occurred (*Table 5*). This may have resulted from the initial colonizers creating more favourable conditions for other species (Mossman et al., 2012), allowing them to establish in their appropriate zonation as conditions become more favourable.

5.3 Relationship Between Vegetation Colonization and Topographic Features

Topographic heterogeneity, represented by channels, cracks, high and low elevation areas, are important in the recovery of salt marsh systems. Wang and Luo (2018) suggest three main driving factors that demonstrate the importance of microtopography in restoration design, namely, (i) provision of local seed supply; (ii) seed retention; and (iii) by modifying abiotic factors such as decreasing soil salinity and increasing soil moisture.

The relationship between vegetation colonization and topographic features described in this study are comparable with results found in other studies. Year 1 demonstrated a stronger relationship between vegetation colonization and channel networks than consecutive years which was dominantly characterized by protochannels (< 2 cm depth). Microtopography in newly restored salt marshes have shown to facilitate the re-establishment of pioneer species through seed entrapment as well as the provision of favorable water relations for seedling establishment (Wang and Luo, 2018). *S. alterniflora* and early colonizers were consistently close to channels (>10 m), which is likely highly influenced by their tolerance for high saline environments and frequent tidal inundation compared to other species. This suggests that channels could be important for these species' successful establishment, particularly to prevent salt marsh annuals from washing away due to their buoyant characteristics (Erfanzadeh et al., 2010). This is especially important as initial colonizers tend to facilitate the establishment of other perennial halophytes as they facilitate sediment trapping, raising the elevation in the tidal frame and creating more favourable for later successional species, *S. patens* (Virgin et al., 2020).

Each consecutive year appeared to demonstrate a greater variability (outliers) between channel networks and vegetation classes and closer proximity of each class to channels. It is possible that the influence of ice (van Proosdij and Townsend, 2008; van Proosdij et al., 2010) or greater seed dispersal via increased local seed source within the site boundaries gave rise to the increase in outliers. On the contrary, the closer proximity of a greater diversity of species to channels in later successional stages resonates with the findings of Morzaria-Luna et al. (2004), as they found that species richness was higher in areas close to creeks in natural marshes. Furthermore, the impact of microtopographic structures on plant recruitment and recovery was modelled by Wang and Luo (2018), which found a greater recovery effect in high marsh than in

middle marsh. This suggests a more intrinsic relationship between channel networks and the

succession of vegetation communities than individual species themselves.

Despite the greater prevalence of clonal spread from Year 1 to Year 4 post-restoration,

there was evidence of new growth of *S. alterniflora* independent of mature patches. The new

growth via seed appeared to generally present itself in lower depressions, including borrow pits

which experienced substantial surface cracking during the summer of 2022 (*Figure 5.3.*

Colonization of S. alterniflora via seed trapping in cracked sediment.. These areas have exhibited

the least amount of colonization within the study site overall. The borrow pits have demonstrated

a net gain of surface elevation each year (Lewis, 2022), as deposition typically occurs more

frequently lower in the tidal frame (Davidson-Arnott et al., 2002; Poirier, 2014). This high

sediment deposition followed by subsequent topographic heterogeneity and colonization further

suggests the importance of these features throughout the restoration process.

Figure 5.3. Colonization of S. alterniflora via seed trapping in cracked sediment.

The DoD analyses showed that *S. alterniflora* and *S. patens* tended to coincide with areas

that had experienced higher rates of sediment deposition. A similar relationship in which high

sedimentation led to rapid colonization was also reported in van Proosdij et al. (2023). *S. alterniflora* likely coincided with higher sediment accretion rates because it has a higher tolerance for more frequent tidal inundation than other species which is a characteristic of the low marsh zone. As previously mentioned, greater deposition tends to occur in low-lying areas (Lewis, 2022), driven dominantly by flocculation and sediment transport properties in the Bay of Fundy (Poirier, 2014). Other studies noted that *Spartina* showed accelerated positive accretion and as the elevation raised within the tidal frame, the vegetation communities shifted as tidal inundation became less frequent (Granse et al., 2021), which may occur at Converse over time. On the contrary, the high marsh result was unusual and is likely high due to the small representative area (<0.5 m^2) which could indicate locally high accretion rates perhaps from sediment transport from ice blocks (van Proosdij et al., 2006). Furthermore, the distinct high marsh was absent the previous year, which supports transport of *S. patens* via ice (Rabinowitz et al., 2021; *Figure 5.4*). This phenomenon was also noted by Norris et al. (2022) as the isolated *S. patens* patches were substantial in size upon initial colonization in the Aulac site in New Brunswick.
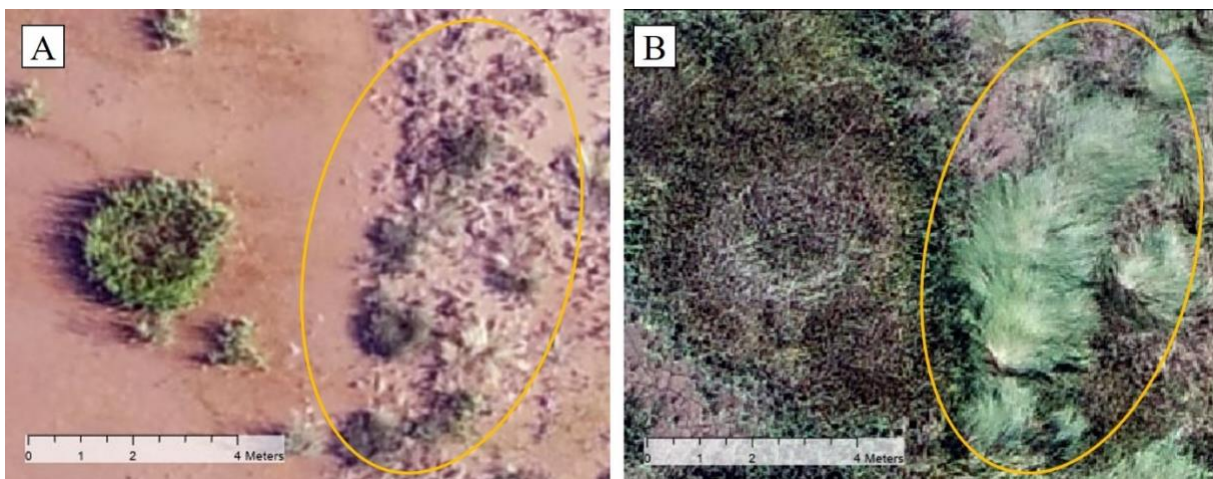
Figure 5.4. Mature patch of S. patens, greater than 4 m long that had not been there the year before: A) 2021, B) 2022.

5.4 Implications for Managed Realignment and Tidal Wetland Restoration Processes

The findings from this study resonate with other managed realignment sites in the Bay of Fundy *(Figure 5.5).* Three sites: Aulac (restored in 2010); Belcher St. (restored in 2018); and Converse (restored in 2018), ultimately converged in their succession despite differences in timing. Each site generally obliged by the stages that Virgin et al. (2022). However, the lines differentiating each phase within these sites may not be as clear cut as suggested by Virgen et al. (2022) as subtle differences in site conditions can affect vegetation recovery dynamics (Norris et al., 2022).

Figure 5.5. Locations of three comparable MR sites within the Bay of Fundy.

One key difference between the sites was the presence or absence of relic creek networks. A network of anthropogenic creeks was present at Converse and absent at Belcher St. and Aulac when the sites were breached. Vegetation colonization at Aulac was slower to develop as it was characterized by patchy revegetation in Years 2-5 post-restoration and was dominantly

characterized by *S. alterniflora* after Year 6. This slow development may have been because of drainage issues caused by borrow pits (van Proosdij et al., 2023). Although a slower development was not the case for Belcher St. as it was 95% vegetated by Year-3 (Van Proosdij et al., 2023), it was also subject to drainage issues which could have impacted the restoration trajectory. On the contrary, Converse was not subject to drainage issues despite the presence of borrow pits within the site. This was likely due to the initial presence of hydrological connectivity from relic creeks.

Another notable difference between these sites was the initial deposition of sediment. Unlike Converse, Belcher St. and Aulac experienced massive deposition of sediments in Year 1 which buried pre-existing vegetation and soils, serving as a clean slate for new halophytic colonization (van Proosdij et al., 2023). However, it wasn't until Year 4 post-restoration that Belcher St. was dominated by *S. alterniflora*, which parallels with the trajectory at Converse. This could indicate that the domination of *S. alterniflora* didn't occur at Belcher St. or Aulac until the sediments within the sites had dewatered (French, 2006). These results suggest that the main driver of vegetation change at Belcher St. and Aulac was the burial of pre-existing vegetation by large quantities of sediment deposition followed by the dewatering of sediments, while the driver of vegetation change at Converse may have been attributed to initial hydrologic connectivity.

From this study and others, it is evident that a managed realignment framework is a highly successional process that can be successful when the restoration initiative is properly designed. When designing MR sites, there were several aspects that appeared to facilitate the positive trajectory of the site and accelerate the restoration process. These included topographic

heterogeneity, opportunity for ice deposition, and proximity to a nearby reference marsh as a seed source for the restoration site.

The importance of integrating topographic heterogeneity, particularly for sites that had historically been levelled for farming, has also been realized in other studies. A study conducted in Essex, UK, demonstrated that homogenous surfaces lead to hostile plant colonization environments, only suitable for pioneer species, creating very different visual appearance and ecological value when compared to a natural marsh (Brooks et al., 2015). Agricultural practices in this example that had been levelled for arable farming resulted in shallow standing water which created high saline conditions during hot and dry weather, inhibiting germination of halophytes (Brooks et al., 2015). This conveyed a much different outcome than the Converse restoration site because although Converse was historically used for agriculture, anthropogenic ditch networks were already present. The presence of topographic heterogeneity during the baseline period likely added complexity to the flow pattern on the soil surface (Van der Ploeg et al., 2012) which reduced initial sheet flow common to homogenous surfaces (Dale et al., 2018). As a result, hydrological connectivity was facilitated early in the trajectory at Converse. This likely created ideal biophysical conditions to promote the establishment of a diverse salt marsh community with target perennial species well established within a 4-year framework.

Another key factor for a positive trajectory of restoration sites in the Bay of Fundy appeared to be the opportunity for ice deposition. This could be facilitated by ensuring the breach is wide enough to allow rafted ice floes to enter the site, though the inlet tends to naturally widen to accommodate the tidal prism of the site (van Proosdij et al., 2010). The importance of providing opportunity for ice deposition is largely due to the valuable contribution of ice to the sediment budget (van Proosdij et al., 2010) as well as the deposition of rhizome material on the

marsh surface (Rabinowitz et al., 2022; van Proosdij et al., 2006). Both of which are foundational for a successful restoration initiative.

Lastly, it is well known that the proximity of a nearby reference site as a source of seeds to the restoration site is an integral component to accelerate the restoration process (Erfanzedeh et al., 2010; van Proosdij et al., 2010; Wang and Luo, 2018). Given that the reference marsh is directly adjacent to the Converse restoration site, there is ample opportunity for seed dispersal into the site and subsequent germination and colonization. This appeared to be an important contributor for initial colonization of target perennials in this study as their spatial patterns suggested they colonized from seed.

5.5 Limitations and Future Work

5.5.1 Using Classification Tools for Understanding Restoration Trajectories

Although habitat classification has shown to be an effective tool for understanding restoration trajectories, there are important limitations that should be considered. From this study, it was evident that there were errors associated with the classification that could have potentially had cascading effects when performing subsequent analyses. These results can stem from the classification algorithm itself as well as user error when selecting representative training samples, particularly when species have similar spectral signatures at certain times in the growing season. Therefore, ground truthing points to inform the classification is integral and having a high level of familiarity with site conditions is also helpful. Future work could investigate the use of multispectral imagery as an alternative or to supplement RGB imagery to assess if the classification results in a higher accuracy output. Additionally, delegating a certain amount of ground truthing points towards establishing an error matrix would likely be beneficial for quality assurance.

5.5.2 Qualitatively Assessing the Spatial Patterns of Vegetation Colonization

Assessing the spatial patterns of vegetation colonization in a qualitative way was a limitation as the spatial patterns were only able to be described generally. For example, *S. alterniflora* colonization was described as circular and subsequent years were described as larger than the previous year. To address this limitation, the PyLandStats Pythonic library could be used to quantify spatial patterns as the tool allows for a variety of landscape metrics to be assessed such as area-edge, shape, and diversity of vegetation patches. This analysis could be useful to inform probable dispersal mechanisms of individual species more accurately, derived from the quantity and size of patches from different classes.

5.5.3 Using the Near tool in ArcGIS Pro to Evaluate Patch Distance from Channels

There were limitations to using the Near tool to quantify the relationship between vegetation colonization and channel networks. The Near tool was only able to calculate the Euclidean distance from the patch edge to the nearest channel. Year 1 post-restoration may have indicated a stronger relationship between vegetation colonization and channel networks than consecutive years because as patches grow, the patch edge to the nearest channel shortens causing the high marsh species to appear to colonize close to channels in later years. In this case, investigating other tools that can calculate other distance parameters, including distance to centroid, may provide a more accurate depiction of real-world phenomenon. Additionally, other studies suggest there may be a more apparent relationship between channel/creek depth and vegetation colonization rather than distance from creek (Wang et al., 2018) which could be further evaluated.

5.5.4 Effects of Vegetation on DEM and DoD Accuracy

Another prominent limitation in this study was demonstrated by the influence of vegetation on DEM and DoD accuracy. The presence of vegetation within DEMs represents erroneous data, therefore vegetation was removed from the output DEM. As subsequent years became more vegetated, using a DoD to quantify the accretion/erosion associated with the vegetation polygons was no longer possible due to the limited bare ground in the imagery. This limitation could potentially be addressed using lidar where vegetation isn't too dense for the lidar to penetrate through to the bare ground surface. This could potentially enable DoD use later in the restoration trajectory to create a longer time series of vegetation colonization in relation to accretion/erosion values to have a more robust representative dataset to quantify this relationship.

Chapter 6: Conclusion

Coastal restoration through the adoption of nature-based solutions demonstrates a paradigm shift in the way society is approaching "wicked problems" such as climate change. These approaches allow the natural environment to flourish while supporting ecosystem services for the betterment of the economic and social realms of society. Given appropriate physical conditions such as hydrology, sediment, and vegetation, coupled with integrated designs, coastal communities can become more resilient in the face of climate change by mitigating the impacts associated with sea level rise, storm surges, waves, shoreline erosion, and flooding through MR approaches for coastal restoration (ICF, 2018; Van Coppenolle & Temmerman, 2019).

The use of GIS and remote sensing are continuously advancing our ability to monitor and quantify the recovery of salt marsh ecosystems. This research leveraged the power of these tools to assess the evolution of the habitat community structure at the Converse restoration site over time; to understand the spatial and temporal patterns of vegetation colonization; and to investigate the effects of topographic features on the spatial and temporal patterns of vegetation colonization.

Classification tools were foundational in this research to understand the successional stages at the Converse restoration site. The resultant successional stages at Converse resonated with other MR sites in the Bay of Fundy, such as the Belcher St. site (Van Proosdij et al., 2023) and the Aulac site (Norris et al., 2022). The successional stages outlined by Virgin et al. (2022) appear to generally hold true in this study and others. Although the time in which the stages occur may vary, MR sites ultimately converge in their trajectories. Therefore, future managed realignment projects within the Bay of Fundy would be expected to evolve in a similar manner.

Having insight into the successful stages of MR sites can help to inform if the restoration is on a positive trajectory or if adaptive management is needed, though advancements in the classification process could offer more accurate insight.

The spatial and temporal patterns of vegetation colonization at the Converse restoration site show a greater prevalence of colonization by seed during initial stages, including the target perennial, *S. alterniflora*. Consecutive dispersal mechanisms of *S. alterniflora* appeared to shift towards clonal spread later in the trajectory, which contrasts findings from other studies within the Bay of Fundy (Norris et al., 2022). Additionally, the site appeared to be more dynamic during the first 3 years post-restoration as species/classes more actively replaced once another each consecutive year. An increase in stability was noted between Year 3 and 4 post-restoration as species/classes tended to persist spatially as the same class, potentially indicative of the establishment of species/communities in their preferred zonation on the marsh surface.

The relationship between vegetation colonization and channel networks showed a stronger relationship in Year 1 than consecutive years. The greater prevalence of outliers each consecutive year may have been attributed to the influence of ice and/or the increase in local seed supply within the site boundaries. The relationship between vegetation colonization and DoDs showed that *S. alterniflora* and early colonizers tended to coincide with higher sediment accretion rates, brackish grasses tended to coincide with lower accretion rates, and *S. pectinata* was associated with mid-range accretion values. The spatial patterns of vegetation colonization appeared to be more strongly correlated with the biophysical conditions that arise from topographic heterogeneity, though seed trapping may also play a role.

Subtle differences between restored sites can affect vegetation recovery dynamics which highlight the importance of ensuring that managed realignment designs cater to individual

requirements and local needs. Proper design considerations and monitoring protocols are needed to ensure best management practices that will facilitate restoration success to create coastal communities that can adapt to climate change.

The following considerations in designing managed realignment sites are recommended:

- Inclusion of topographic heterogeneity

- Opportunities for ice deposition

- Proximity to a nearby reference marsh as a seed source for the restoration site

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Appendices

Appendix A: Raw Vegetation Data Used for Ground Truthing

VS Code	Date_	Dominant Veg 1	Height 1 (cm)	Dominant Veg 2	Height 2 (cm)	Dominant Veg 3	Height 3 (cm)	NOTES
CON_T2S1 22	8/3/2022	Bare Ground						
CON_T2S3 22	8/3/2022	Suaeda maritima spp maritima	23.5					
CON_T2S4 22	8/3/2022	Dead Material						dead suaeda and shrubs
CON_T2S5 22	8/3/2022	Suaeda maritima spp maritima	22					
CON_T2S6 22	8/3/2022	Distichlis spicata	37					
CON_T2S7 22	8/3/2022	Distichlis spicata	16					
CON_T2S8 22	8/3/2022	Dead Material						
CON_T3S3 22	8/3/2022	Bare Ground						
CON_T3S4 22	8/3/2022	Spartina alterniflora	86	Bare Ground				
CON_T3S5 22	8/3/2022	Elymus repens	40					
KN_01 22	8/3/2022	Spartina pectinata	101					
KN_02 22	8/3/2022	Spartina alterniflora	80					
KN_03 22	8/3/2022	Spartina pectinata	108					
KN_04 22	8/3/2022	Spartina alterniflora	122					
KN_05 22	8/3/2022	Spartina pectinata	129	Spergularia salina	18	Elymus repens	91	
KN_06 22	8/3/2022	Spartina pectinata	110					
KN_07 22	8/3/2022	Suaeda maritima spp maritima	12					
KN_08 22	8/3/2022	Spartina alterniflora	112					
KN_09 22	8/3/2022	Spartina alterniflora	115					
KN_10 22	8/3/2022	Bare Ground						
KN_11 22	8/3/2022	Juncus gerardii	65	Spartina patens	70			
KN_12 22	8/3/2022	Spartina pectinata	90					

KN_13 22	8/3/2022	Spartina alterniflora	135						
KN_14 22	8/3/2022	Spartina patens	40	Juncus gerardii	44				
KN_15 22	8/3/2022	Alopecurus pratensis	92	Dead Material					
KN_16 22	8/3/2022	Elymus repens	86						
KN_17 22	8/3/2022								
KN_18 22	8/3/2022	Suaeda maritima spp maritima	20						
KN_19 22	8/3/2022	Spergularia salina	9	Atriplex sp.	20				
KN_20 22	8/3/2022	Suaeda maritima spp maritima	20.5	Spartina pectinata	68				
KN_21 22	8/3/2022	Juncus balticus	106						
KN_22 22	8/3/2022	Dead Material							
KN_23 22	8/3/2022	Spartina patens	45						
TR_01 22	8/3/2022	Spartina pectinata	86	Plantago maritima	22				
TR_02 22	8/3/2022	Distichlis spicata	71	Puccinellia maritima	88				
TR_03 22	8/3/2022	Plantago maritima	34						
TR_04 22	8/3/2022	Suaeda maritima spp maritima	20	Puccinellia maritima	71				
TR_05 22	8/3/2022	Puccinellia maritima	48						
TR_06 22	8/3/2022	Spartina alterniflora	60	Puccinellia maritima	60				
TR_07 22	8/3/2022	Puccinellia maritima	60	Solidago sempervirens	64				

Appendix B: Raw Data of Each Habitat Class Each Year

	OID_	Class_name	Frequency	Total Area (m2)
2019	1	Bare Ground	4	59576.9
	2	Dead material	337	1075.3
	3	Remnant Agriculture	10	10194.0
	4	Water	1	782.4
2020	1	Aboiteau	1	11.0
	2	Bare Ground	27	56897.5
	3	Brackish Grasses	8	1053.3
	4	Dead material	188	7292.9
	5	Dyke	2	321.1
	6	Early Colonizers	9	116.2
	7	rock	10	287.7
	8	Spartina alterniflora	441	2108.6
	9	Spartina pectinata	365	3139.9
	10	Upland Forbes	1	33.7
	11	Water	74	573.4
2021	1	Aboiteau	1	6.2
	2	Bare Ground	97	31344.2
	3	Brackish Grasses	30	9929.5
	4	Dead material	5	25.8
	5	Dyke	1	320.6
	6	Early Colonizers	1028	21128.0
	7	Rock	1	352.3
	8	Spartina alterniflora	1338	3224.1
	9	Spartina pectinata	1667	5123.9
	10	Upland Forbes	1	137.1
	11	Water	11	189.6
2022	1	Aboiteau	1	6.2
	2	Bare Ground	4112	32948.2
	3	Brackish Grasses	3111	11475.3
	4	Dyke	75	368.7
	5	Early Colonizers	3446	10752.0
	6	High Marsh	13	433.9
	7	Spartina alterniflora	2896	10517.3
	8	Spartina pectinata	754	5021.6
	9	Upland Forbes	1	137.0
	10	Water	29	128.6

Appendix C: Raw Data of Bare Ground, and New and Persistent Vegetation

	Type	OID	Class	Frequency	Total Area (m2)
2019	**BARE**	1	Bare Ground	4	59576.9
2020	**BARE**	5	Bare Ground	27	56897.5
	NEW	1	Brackish Grasses	8	1053.3
		2	Early Colonizers	9	116.2
		3	Spartina alterniflora	441	2108.6
		4	Spartina pectinata	365	3139.9
2021	**BARE**	5	Bare Ground	97	31344.2
	NEW	1	Brackish Grasses	30	6420.9
		2	Early Colonizers	1005	20677.3
		3	Spartina alterniflora	1311	2689.7
		4	Spartina pectinata	1650	3439.3
2022	**BARE**		Bare Ground		32948.2
	NEW	0	Brackish Grasses	999	2684.3
		1	Early Colonizers	2011	1682.7
		2	High Marsh	23	21.4
		3	Spartina alterniflora	1608	12984.1
		4	Spartina pectinata	370	1046.4

	OID_	Class_1	Class_2	Frequency	Total Area (m2)
2021	1	Brackish Grasses	Brackish Grasses	6	938.6
	2	Brackish Grasses	Early Colonizers	2	0.1
	3	Brackish Grasses	Spartina pectinata	2	3.0
	4	Early Colonizers	Early Colonizers	9	10.2
	5	Early Colonizers	Spartina alterniflora	2	1.1
	6	Early Colonizers	Spartina pectinata	6	104.1
	7	Spartina alterniflora	Brackish Grasses	2	3.7
	8	Spartina alterniflora	Early Colonizers	243	372.8
	9	Spartina alterniflora	Spartina alterniflora	334	484.8
	10	Spartina alterniflora	Spartina pectinata	181	1160.4
	11	Spartina pectinata	Brackish Grasses	240	2566.4
	12	Spartina pectinata	Early Colonizers	117	67.5
	13	Spartina pectinata	Spartina alterniflora	63	48.4
	14	Spartina pectinata	Spartina pectinata	79	417.1
2022	1	Brackish Grasses	Brackish Grasses	58	8182.3
	2	Brackish Grasses	Early Colonizers	11	219.3
	3	Brackish Grasses	High Marsh	1	3.4
	4	Brackish Grasses	Spartina alterniflora	6	55.3
	5	Brackish Grasses	Spartina pectinata	10	1342.5
	6	Early Colonizers	Brackish Grasses	147	1761.5
	7	Early Colonizers	Early Colonizers	317	7143.2
	8	Early Colonizers	High Marsh	4	295.0
	9	Early Colonizers	Spartina alterniflora	354	1692.1
	10	Early Colonizers	Spartina pectinata	113	1221.7
	11	Spartina alterniflora	Brackish Grasses	161	76.0
	12	Spartina alterniflora	Early Colonizers	348	402.9
	13	Spartina alterniflora	High Marsh	55	105.5
	14	Spartina alterniflora	Spartina alterniflora	682	1978.6
	15	Spartina alterniflora	Spartina pectinata	71	74.0
	16	Spartina pectinata	Brackish Grasses	432	922.0
	17	Spartina pectinata	Early Colonizers	945	1304.6
	18	Spartina pectinata	High Marsh	3	8.7
	19	Spartina pectinata	Spartina alterniflora	148	301.4
	20	Spartina pectinata	Spartina pectinata	374	1860.2

Appendix D: Raw Data from the Zonal Statistics as Table Tool

	OID_	Class_name	ZONE_CODE	COUNT	AREA	MIN	MAX	RANGE	MEAN	STD	SUM	MEDIAN	PCT90	FREQUENCY
2021	1	Early Colonizers	1	13686032	16765.4	-0.30	0.45	0.75	0.02	0.03	20677.32	0.03	0.06	1005
	2	Spartina alterniflora	2	1292391	1583.2	-1.17	0.47	1.64	0.04	0.03	2689.70	0.04	0.07	1311
	3	Spartina pectinata	3	1266108	1551.0	-0.61	0.71	1.32	0.02	0.03	3439.32	0.02	0.06	1650
	4	Brackish Grasses	4	308567	378.0	-0.13	0.30	0.43	0.00	0.03	6420.92	0.00	0.04	30
2022	1	Brackish Grasses	1	201985	247.4	-0.49	0.19	0.68	0.02	0.03	2684.22	0.02	0.06	997
	2	Early Colonizers	2	1301006	1593.7	-0.75	0.23	0.98	0.02	0.03	1682.71	0.02	0.05	2011
	3	High Marsh	3	9099	11.1	-0.04	0.16	0.20	0.06	0.03	0.54	0.06	0.09	3
	4	Spartina alterniflora	4	4854961	5947.3	-0.52	0.35	0.86	0.06	0.03	13001.75	0.05	0.10	1625
	5	Spartina pectinata	5	232674	285.0	-0.75	0.24	0.99	0.03	0.03	1046.41	0.03	0.06	370