DETERMINING TRAJECTORY USING ECOLOGICAL INDICATORS AT THE COGMAGUN RIVER SALT MARSH NINE-YEARS POST RESTORATION

by

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ABSTRACT

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The Cogmagun River salt marsh was a Ducks Unlimited impoundment that CB Wetlands and Environmental Specialists restored by reintroducing tidal flow in 2009 by removing part of the dyke embankment. Since the initial restorative actions, ecological indicators; vegetation, hydrology, soils and sediments have been monitored for five-years. In order for marsh systems to protect against sea level rise, they must be able to build vertically via trapping and deposition of sediment between shoots of salt marsh vegetation. This study monitored the ecological indicators nine-years post restoration. The aim was to discover how characteristics vary spatially, in the years following the restoration and when compared to reference conditions of a natural undisturbed salt marsh. The results showed that nine years after the re-introduction of tidal flow, the Cogmagun salt marsh displayed environmental characteristics similar to those at the reference site. The marsh exhibited a positive trajectory from the last measurements taken in 2014. The vegetation characteristics such as species richness and halophytic composition demonstrated a typical salt marsh, and was supported by the presence of zonation. Rod surface elevation tables (RSETs) and maker horizons indicate that there is compaction occurring below-ground at most sites, which was supported with the presence of anoxic layers in the soil characteristics. Net sediment accretion remains greater than compaction, contributing to a positive net change in the surface elevation, indicating vertical growth of the marsh surface.

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CHAPTER 1: Literature Review

1.1 Research problem, purpose and objectives

Salt marsh environments are dynamic ecosystems that provide a broad array of ecosystem services to the surrounding environment. These benefits include: providing a niche habitat for various organisms, providing coastal protection from storms by attenuating wave activity (Poirier et al., 2017), and playing an important role in the estuarine food web through export of macro detritus. In addition, sediment deposition on the marsh surface allows it to rise providing erosion control and counteracting some threats from sea level rise (Barbier et al., 2011). In addition, there is the potential for carbon sequestration (Connor et al., 2001). Many salt marshes in Nova Scotia have been modified to provide additional nutrient-rich agricultural land. These processes often include dyking which has several ecological implications that includes the loss of habitat, species and productivity in the dyked area (van Proosdij et al., 2010). Salt marsh restoration aims to restore a salt marsh habitat to that which existed prior to alteration and can occur via passive or active efforts. Passive salt marsh restoration involves letting a dyke breach occur due to natural conditions, such as storm surge events or sea level rise, while active restoration incorporates management techniques. These could involve removing the structure or subsection of a dyke which allows tidal flow to resume, along with gradual colonization of salt marsh flora and fauna (van Proosdij et al., 2010).

In Nova Scotia, ten salt marsh restoration projects have been conducted: Cheverie Creek salt marsh, Walton River salt marsh, Smith Gut salt marsh, St. Croix River marsh, Cogmagun salt marsh, Allain’s Creek salt marsh, Comeau Hill salt marsh, Green Creek salt marsh, Argyle salt marsh and Three Fathom Harbour salt marsh (Province of Nova Scotia, 2013; Roman and Burdick, 2012). The Cogmagun salt marsh is situated on the Cogmagun River in Hants County, Nova Scotia and consists of a 6.9-hectare tidal wetland. The Cogmagun salt marsh was a former Ducks Unlimited Canada impoundment which failed and was restored in 2010 by the creation of a 60-metre breach at the lowest elevation in the existing dyke (Bowron et al., 2010). A central channel was also excavated to establish a hydrology that is more like that which existed in the marsh before alteration, and changes have been assessed by comparing them with an undisturbed salt marsh reference site (Bowron et al., 2015).

The aim of the present project was to study environmental changes in the Cogmagun River Salt Marsh restoration site to determine if it has reached similar characteristics to those in the reference site. Monitoring protocols were used to track the changes. This project built on the work already conducted by CB Wetlands and Environmental Specialists (CBWES) who designed and carried out the initial restoration in collaboration with NS Transportation and Infrastructure Renewal (NSTIR), Saint Mary’s University (SMU), Ducks Unlimited Canada (DUC) and the property owners (Abundant Acres). A modified monitoring program was used based on the Global Programme of Action Coalition (GPAC) Regional Monitoring Protocol (Neckles et al., 2002). This was conducted one year prior to restoration and five years post restoration, and the data were compared to the reference site. The use of a reference site is included so that there is a standard for the measurements of the ecological indicators to be compared to. The reference site was located 1.5 kilometres upstream on the north side of the Cogmagun River and covered a similar area.

The restoration site was studied to address the following questions:

1. What are the current characteristics of hydrology, vegetation, soils and sediments in the reference and restoration sites?

2. How has each aspect of hydrology, vegetation, soils and sediments, changed since the last set of measurements were taken in 2014?

3. Has the Cogmagun restoration site reached a similar successional stage as the reference site, based on the ecological indicators identified in the modified GPAC monitoring protocols?

1.2 Salt marsh function

Tidal wetlands are environments in the intertidal zone, existing at the interface between land and sea. These systems experience daily flooding based on tidal activity, and where sediment deposition continually contributes to marsh accretion; the vertical growth of the marsh surface due to sediment trapping predominately amongst the vegetation. The vegetation that exists on salt marshes is adapted to these conditions and is dominated by halophytic species that can withstand varying durations of tidal inundation. Tidal wetlands are generally sheltered from significant wave activity and often have extensive mud or sand flats associated with them, especially in macrotidal environments, where the tidal range exceeds 4m (Davidson-Arnott et al., 2002).

1.2.1. Vegetation

Tidal wetlands are comprised salt tolerant and brackish-water tolerant species that are also able to survive tidal inundation. The extent and frequency of flooding ranges from that seen at low tide to that found at the highest spring tide. The vegetation can vary significantly in terms of species composition and size over a saltmarsh profile. In marsh ecosystems, high quantities of both nitrogen and phosphorus can result in rapid growth of vascular plants (Zoltai and Vitt, 1995). Although growth rates are high, decomposition rates are also high. There is an abundance vascular plants, but bryophytes have little success competing in a salt marsh environment (Zoltai and Vitt, 1995). Sodium and chloride ions are dominant in tidal marshes, making them saline, and non-alkaline ecosystems (Zoltai and Vitt, 1995). The species that dominate tidal wetlands are graminoid macrophytes such as grasses, sedges and rushes (Warner and Rubec, 1997).

Salt marsh vegetation becomes established by colonization of successional species. These pioneer species arrive by transportation of water, known as hydrochory, from adjacent tidal wetlands and through sea ice transportation. These species are angiosperms and are able to recolonize by both seed and by fragments of rhizomes (Broome et al., 1988). Tidal currents are the main method of transportation, wind speed and direction do not have a significant influence (Huiskes et al., 1995). In tidal marshes, Chang et al. (2007) indicate that tidal currents as a seed dispersal mechanism may be stochastic in nature as opposed to systematic. Chang et al. (2007) conclude that most seeds originated from local dispersal sources but storm surge events led to seeds being dispersed over longer distances. The pH, salinity, anoxia, and sediment deposition rates influence how pioneer species are able to propagate once they land on shore, but the main factor is seed availability (Erfanzadeh et al., 2009). Massive seed producers such as *Spartina* *alterniflora* are often the first colonizers (Erfanzadeh et al., 2009). Sea ice acts as a main seed dispersal mechanism in northerly climates such as Nova Scotia. When ice blocks are forced onto the marsh surface they often take pieces of the marsh surface with it. As the ice block gets transported to other locations, seeds and rhizomes are also transported (Lemieux, 2010).

Saltmarsh ecology is characterized by distinct zonation of plant species. These zonation patterns usually occur parallel to the direction of tidal flow or in concentric patterns and vary depending on the marsh surface elevation (Gray, 1992; Warner and Rubec, 1997). The surface elevation acts as a significant influence because it determines how long the species will be inundated by salt water. Varying durations of tidal inundation also influence conditions, such as the amount of light penetration and soil anoxia (Gray, 1992). Above the spring high water line, marsh grasses are typically outcompeted by terrestrial species and zonation becomes less noticeable.

Following restorative actions in a tidal wetland, the system will reach a new state of equilibrium. Once the hydrological functions are restored, vegetation colonization will proceed. This can follow a similar trend to initial establishment on a natural salt marsh. Depending on the previous use of the marsh system, the vegetation composition may exhibit significant changes from the species present before initial restoration. Salt marsh vegetation species can revegetate in the growing season immediately following restoration, with most early colonization occurring on exposed mud (Roman et al., 2002). In looking at four different case studies we can see how the vegetation succession varies depending on different marsh systems. On Sachuest Point salt marsh at Middletown, Rhode Island, Roman et al. (2002) experienced S. *patens, S. alterniflora*, and *Salicornia europaea* as initial colonizers on bare mud patches, along with an overall increase in *S. patens*, *S. alterniflora* and a decrease in *Phragmites Australis.* This combination of species was characteristic of the surrounding mature salt marsh environments. Two years post restoration; the restored marsh was still significantly different than their control site but was following a trajectory to reach natural conditions of salt marshes elsewhere in New England.

Byers and Chmura (2007) looked at two salt marshes in Atlantic Canada. These were the John Lusby marsh and the Saint's Rest marsh, both were located in the Bay of Fundy. Each site that was breached, was monitored and compared with a respective reference site, the Dipper Harbour marsh and Wood Point marsh. The authors looked at the effect of elevation on the composition of major species, comparing the plant cover and production between each restoration site and their respective reference site. Cover plots and end-of-season standing crop plots were used in each marsh in areas of varying elevation and locations near tidal channels. Byers and Chmura (2007) determined that *S. alterniflora* and *S. patens* have a vertical range that increases with tidal range. They also determined that when compared to a microtidal marsh, the species were inundated less frequently and can tolerate a greater range of frequency and depth of inundation. The *S. patens* zone may survive more than a year without inundation indicating a lesser role of hydroperiod in delimiting the species' distribution in the high marsh. Due to the elevation range of *S. alterniflora* and tolerance of high sedimentation rates, the authors conclude that restoration of Bay of Fundy salt marshes should have a promising rate of success.

CB Wetlands and Environmental Specialists (CBWES) conducted a tidal wetland restoration project in Cheverie Creek, on the Bay of Fundy, Nova Scotia and continued monitoring it for seven years. Species such as *Atriplex glabriuscula* and *Salicornia europaea* decreased following restoration, while more brackish tolerant species such as *Carex paleacea*, did not show significant changes (Bowron et al., 2013). *Spartina alterniflora* doubled in abundance while *Spartina patens* increased in both abundance and frequency (Bowron et al., 2013).

Smith and Warren (2012) looked at the importance of monitoring how vegetation responds to restored tidal flow. The authors emphasize the importance of focusing on vegetation when restoring tidal flow to an area as there may be an influx of invasive species when tidal flow is initiated explaining why *Phragmites australis* (common reed) and *Lythrum salicaria* (purple loosestrife) can be found. Restoring tidal flow to the ecosystem aims to restore the ecological function and can be measured in some-part using vegetation. Smith and Warren (2012) identify hydrology, topography, geomorphology, pore water chemistry, nutrients, seeds, soil properties, herbivory, genotype, surrounding land use, local climate and the size of restoration site as factors that influence how vegetation will recover in a restoration project. In addition, Smith and Warren (2012) identify the importance of variability in the local parameters for each site.

1.2.2. Hydrology

Tidal wetlands exist in the upper intertidal zone, at mid to high latitudes, where there is no significant wave action, allowing sediment deposition to occur (Davidson-Arnott et al., 2002). The salt-tolerant species that dominate these environments are often intersected by tidal creeks that increase in size as they approach the main tributary of tidal flooding (Davidson-Arnott, 2002). Salt marsh habitat structure and function is dependent on flooding from salt water, whereby hydrology is a key measurement. The hydroperiod is the frequency and duration that the salt marsh is inundated and is a measure of the tidal signal and elevation (Neckles and Dionne, 1999). Tidal range is the difference in height between the low tide and high tide.

As the tide rises, it floods the adjacent land causing flooding for various durations and depths. As the tide falls, the water drains from the wetland back into the tidal channel. The zonation of a marsh system is largely controlled by flooding and the levels of salinity will influence the vegetation that can grow in each zone. The upper marsh is not inundated as frequently and therefore has reduced salinity when compared to the lower marsh which experiences frequent inundation. Since tidal systems vary around the world, how the tides affect salt marshes will also vary, ranging from macrotidal in the Bay of Fundy, Canada to microtidal in the Gulf of Mexico.

The duration of tidal inundation over a marsh surface will control both vegetation and sediment characteristics and will influence nutrient infiltration (Bricker-Urso and Nixon, 1989). Following tidal inundation, how the marsh system drains is equally important. Drainage is an important component of tidal wetlands because when the water cannot drain properly, pooling can occur causing anoxic conditions in the soil. The presence of anoxic conditions can affect the soil chemistry in a way that is detrimental to the vegetation (Armstrong et al., 1985). If vegetation decay persists, this can lead to a lower salt marsh surface, increasing potential pooling and contributing to a positive feedback system resulting in a drowned marsh.

Tidal range acts as an important control on both the survival of the system and marsh characteristics, such as sediment deposition and vegetation. The stability of tidal channels and vegetation platforms increase as the tidal range increases (Kirwan and Guntenspergen, 2010). Studies have shown there is a relationship between tidal range and ability to keep pace with sea level rise, and macrotidal marshes are capable of surviving faster rates of sea level when compared to mesotidal or microtidal marshes (Stevenson et al., 1985). Increased tidal range also influences the water level from storm surge events that have less affect especially in the low and mid marsh zones (Davidson-Arnott et al., 2002).

Konisky et al. (2006) reviewed datasets from 36 salt marsh restoration activities and grouped the ecological indicators at the restoration site, the reference site and the impacted site before restoration commenced. They concluded that the impacted sites were different from the reference sites and that restoration would put them on the correct trajectory to that found at the reference site. Following restoration activities that restore tidal flow, many salt marsh environments displayed ecological similarities to their reference equivalence (Konisky et al., 2006). Tidal hydrology was similar to salinity in that the impacted site was reduced from the reference site and following restoration activities flooding increased. Although the frequency increased, the actual flood levels remained reduced, accounting for only 74% of the reference height (Konisky et al., 2006). Both salinity and hydrological components of the restoration occurred within one to two years after the initial activity.

1.2.3. Soils and sediments

In coastal environments, the sediment source varies depending on the local topography and geomorphology. Estuaries and deltas receive a large portion of their sediment from fluvial sources while embayment and barrier systems receive sediment from longshore sediment transport due to wave activity eroding the shoreline (Reed and Cahoon, 1992). In the Bay of Fundy, the marshes are minerogenic and the source of sediment is from coastal erosion of rocks and fine-grained sediments.

Allochthonous sediment originates from another location and deposition depends on elevation and salt marsh vegetation to reduce flow velocities, allowing sedimentation to occur. These marsh sediments are mostly minerogenic, comprised of silt, clay and sand. Depending on the elevation and location of the marsh, the sediment composition and grain size will range from sandy silt to clayey silt (Allen, 2009). This type of sediment deposition is known as allochthonous growth, and contributes to vertical growth of the marsh surface. The other contribution to marsh surface elevation change is the growth of organic matter through sub-surface processes (Davidson-Arnott, 2006). If the salt marsh is dominated by minerogenic sediments it is a minerogenic marsh, conversely if organic matter is dominate, it is an organogenic marsh (Allen, 2009).

Sediment accretion is important in marsh environments due to the marsh's ability to accrete with rising sea levels. Salt marshes are able to grow vertically through surface sediment deposition and expansion of below-ground organic matter. The volume of marsh soil experiences decreases through surface erosion and when organic matter decomposes it can cause compaction. The volume of marsh soil accumulates from expansion of root networks and rhizome tissues, as well as by the deposition of minerogenic sediments and refractory organic particles (Morris et al., 2016).

Chmura et al. (2001) conducted a study along the New Brunswick border on the Bay of Fundy to study sediment deposition. The authors studied seven sites and determined that the sediment accumulation decreased with elevation and distance from vegetation. A positive relationship of sediment accumulation was with tidal range, and the amounts of suspended sediment present in the water.

Once tidal flow is restored to a salt marsh, sediment deposition will occur but will vary depending on the previous conditions and restorative actions. Reed et al. (1999) looked at how tidal creeks contribute to sediment deposition and identified their importance. The authors identified relationships between sediment deposition and the tidal creek at Scolt Head Island. There was declining deposition with increasing distance from the creek, and with each measured tide. Greater suspended sediment levels were found on the flood tide versus the ebb tide.

1.3 Climate change

Climate change has a significant influence on coastal processes. Understanding how the climate is projected to change will be important for understanding the effects on coastal communities and their environments. Since the 1950s, there have been unprecedented changes (IPCC, 2013). From 1880 to 2012 the global average temperature of the land and ocean show a warming of 0.85° C, and the Northern Hemisphere was likely the warmest 30-year period from 1983 to 2012 when considering the past 1400 years (IPCC, 2013). In addition to atmospheric changes, the IPCC (2013) has stated that it is virtually certain that the upper ocean has experienced warming from 1971 to 2010. The way in which this can affect salt marshes is discussed below.

1.3.1 Flooding

Atmospheric and oceanic warming will influence sea level rise and have consequences on the erosion and flooding of coastal landscapes. In the fifth assessment, the IPCC report uses representative concentration pathways (RCPs) that assess different climatic scenarios that could occur in the future from different levels of greenhouse gas concentrations. Each scenario is a possible outcome which is dependent on the levels of greenhouse gas emitted until that point. The IPCC report (2013) determined that it is very likely that sea level rise in the 21st century will occur at a greater rate than it has from 1971-2010. Computer models indicate global mean sea level rise will range anywhere from 0.26 meters to 0.82 meters from 2081-2100 and will vary with the different RCPs. Regional sea level rise proves to be more challenging to measure due to localized variations, but the IPCC (2013) indicates it is very likely that by the end of the 21st century there will be a positive rise in sea level covering over 95% of the ocean.

Increased sea level leads to increased coastal flooding, placing both physical and social constraints on the surrounding communities. Thermal expansion of ocean waters combined with melting glaciers are the dominant driving force of sea level rise, accounting for 75% of the observed rise when excluding the Antarctic glaciers adjacent to the ice sheet (IPCC, 2013). Thermal expansion occurs as ocean water becomes warmer and the water expands, decreasing the density, causing a subsequent increase in volume (IPCC, 2013). The other main contribution to sea level rise is ice melt. This is dominated by land ice as it adds a significant input into the ocean, whereas the input provided by sea ice is only derived from density differences between fresh and salt water. The melting of glaciers also poses a significant input to global mean sea level rise.

Locally, Nova Scotia and other parts of the East Coast are experiencing disproportionate effects from relative sea level rise. This accounts for the occurrence of sea level rise combined with the relative position of the earth – largely due to subsidence caused from isostatic adjustments from glaciers that were present in the last ice age (Savard et al., 2016). On the Atlantic coast of Nova Scotia and New Brunswick, James et al. (2014) used a high emissions scenario to estimate that mean sea level elevation would be 80-100 cm higher in 2100 when compared to 1984-2005.

1.3.2 Erosion

In addition to sea level rise, climate change is also influencing soil erosion, posing a risk for sustainability of salt marsh environments. Sea level rise puts a greater stress on coastal environments and makes them more susceptible to erosion. This principle stands with all coastal environments just on differing orders of magnitude. Zhang et al. (2004) identify the main culprits for coastal erosion as sea level rise, changes in storm climate and human impacts, with sea level rise being the dominant factor. The higher sea levels rise, the greater the force of storm surge and wave activity on the adjacent land. This land protects the infrastructure both on and the marsh and behind it and without it the damage could be insurmountable.

1.4 Salt marsh ecosystem services

1.4.1 Counteracting sea level rise

Salt marshes are sensitive ecosystems that face threats of submergence due to current sea level rise projections. Sediment accretion is a morphodynamic process that plays a large part in reducing erosion of salt marsh environments. When sea levels are rising at normal rate, sediment accretion on the marsh surface is able to keep pace with the rising water level dependent on the frequency at which flooding is occurring and the available sediment supply (Stumpf, 1983; Crosby et al., 2016).

Accretion occurs as the soil develops through mineral sediment deposition and the accumulation of organic soils (Cahoon et al., n.d.). To determine the overall surface elevation, sediment deposition needs to be considered along with the processes occurring in the ground. These subsurface processes include compaction, root growth and decomposition and changes from shrink-swell mechanisms (Cahoon et al., n.d.). Accumulation of organic soils occurs in part due to macrophytes which exist in the intertidal zone, accumulating organic sediments while trapping inorganic sediment (Morris et al., 2002). As mentioned above, vegetation acts as a main energy dissipation method for tidal flow and wave activity. As the vegetation slows the velocity, sediment particles fall out of suspension from the water column and become deposited on the marsh surface, allowing for vertical accretion to occur.

In order to get an accurate depiction of how the marsh elevation is changing overall, the below ground processes that contribute or counteract the vertical accretion need to also be considered. These subsurface processes include decomposition and growth of the roots of plants, ground water flow, bioturbation and natural occurring consolidation and expansion (Jin, et al., 2018). How plant roots change will influence the subsurface processes and can contribute to the growing of the salt marsh or it could have the reverse effect. As the root network becomes more expansive, it can contribute to the marsh surface rising, but if anoxic conditions exist it can cause root matter to decay and cause the marsh surface to fall

Many studies indicate that there is high potential of losing salt marshes with sea level rise, but it is unknown to what extent they will be lost. Crosby et al. (2016) conducted a synthesis on the data pertaining to sea level rise and the loss of marsh ecosystems in order to look at future prospects. The study determined that there were a lot of these ecosystems that did not keep up with sea level rise within the last century, with an even higher occurrence in the past 20 years.

Schurech et al. (2018) used a global modelling program to look at how coastal wetlands will respond to sea level rise. Much of the literature looking at sea level rise at the global scale indicate that many wetland ecosystems will not be able to grow vertically fast enough to keep up with some of the sea level rise scenarios being proposed. Alternatively, literature looking at this same dilemma from a local-scale indicate the opposite. Schurech et al. (2018) aimed to analyze the gap in these two opposing views in the literature. The authors conclude that sediment accretion and accommodation space are not being adequately accounted for in the global-scale analysis. The model developed by Schurech et al. (2018) account for both limitations. The model showed that there is potential for coastal wetland gains provided that there is enough accommodations space for this growth to occur and the sediment supply remains at the same level. Schurech et al. (2018) indicate infrastructure growth and development in the coastal zone as a main limitation due to the affect it will have on the accommodation space, allowing for coastal wetland movement.

Singh et al. (2007) use the example of the Bay of Fundy to look at how restored salt marshes adjust to changes in sea level, the benefit of having a salt marsh buffer adjacent to land, the potential of marshes to store carbon all while comparing the cost-effectiveness when put against traditional methods to combat sea level rise. Singh et al. (2007) determined that salt marsh restoration is an adequate measure against sea level rise, especially when looking at the costs associated with maintaining dyke infrastructure and coastal defense, such as rocking. The authors also indicate potential for salt marsh restoration around the Bay of Fundy due to low population density, underused dykelands and tidal flow that contains high suspended sediment concentrations.

1.4.2 Carbon sequestration

In addition to the protection salt marshes provide against rising sea levels, they also act as a carbon sink, trapping atmospheric carbon and actively reducing the greenhouse effect. Connor et al. (2001) address both spatial and temporal trends of potential carbon sequestration in the tidal salt marshes located in the Bay of Fundy. Due to the high number of restored salt marshes comprising a relatively large area in this location, the authors aimed to look at how much potential there is for carbon storage. In addition, the authors looked at how the storage potential changes within different regions around the Bay of Fundy. Connor et al. (2001) implemented marker horizons along three varying elevations at seven salt marshes on the New Brunswick border. The authors used cryogenic coring to measure each marker horizon after a 12-month period had passed. In addition, soil cores were also taken at each site to measure bulk density and loss of ignition at elevations that correspond to the marker horizons. Carbon accumulation in the soils was calculated by looking at the average surface accretion combined with average carbon density from the soil core samples at each site. Connor et al. (2001) estimate that in the outer portion of the Bay, carbon accumulation in the last 30 years is 76 g C m^{-2} yr^{-1}, compared to the upper bay which is estimated to be as high as 184 g C m^{-2} yr^{-1}. Conner et al. (2001) indicate the importance for restoration due to the increased carbon storage potential with sea level rise which does not exist for the former agricultural soils.

Wollenberg et al. (2018) also looked at quantifying the amount of carbon burial in a salt marsh environment located on the Bay of Fundy. The site is in Aulac, New Brunswick and was analyzed six years after tidal flow was reintroduced. Wollenberg et al (2018) sampled soils using two methods, including the use of a 25-mm Dutch gauge corer and a Russian peat auger. Cores were taken until they met refusal, or they reached the former agricultural soils. In areas where neither corer was adequate, soil blocks were cut from the marsh surface. Loss of ignition analysis was conducted on the soil samples and a conversion factor was used to translate organic matter to organic carbon. The site showed the burial rate for carbon as 1 329 g C m^{-2} yr^{-1}, proving to be greater than a mature marsh measured nearby. Wollenberg et al. (2018) indicate that carbon storage potential is high for restoring salt marshes along the Bay of Fundy, but is likely dependent on sediment deposition following restoration, and the amount of carbon in the suspended sediment.

1.4.3 Wave activity

Vegetation has significant value for attenuating wave activity in salt marsh environments. As incoming water moves through the vegetation, frictional drag slows the water, reducing the velocity at which it can travel (Narayan, 2017). How effective the vegetation is at causing fictional drag will depend on several factors such as the plant density, height and species formation. Tempest et al. (2015) conducted an analysis on flow interactions with vegetation in salt marsh environments. The study looked at how these interactions occur so that the data could be used for flood risk assessment in salt marsh environments. Other studies have conducted similar analyses but have used modeling or field studies. The downside to using digital methods is that it involves using a lot of simplifications of how the salt marsh vegetation interacts with waves and tidal flow, which can result in over-simplifying the results. Tempest et al. (2015) aimed to provide more accurate depiction of how the vegetation attenuates coastal processes without using these assumptions. When tidal flow switches from an un-vegetated region to a vegetated region, there is a quick increase in turbulent kinetic energy and at times, velocity. Flow then becomes confined from the interaction with the vegetation, causing a turbulent wake to form at the back. Tempest et al. (2015) looked at several studies that analyzed flow characteristics of vegetation in salt marsh environments. Some studies have reported a 50% (Leonard and Croft, 2006) reduction in speed, while others have reported anywhere from 250-300% when there are unimpeded flows (Leonard and Reed, 2002). In their research, Leonard and Croft (2006) aimed to quantify velocity both horizontally and vertically, the intensity of turbulence and the total turbulent kinetic energy of *Spartina alterniflora* in North Carolina. Through this study they not only determined that about 50% of the initial mean velocity is reduced within 5 metres after reaching the vegetated foreshore, but that the reduction has an inverse relationship with the amount of biomass that is present. Leonard and Reed (2002) explain how morphology of the canopy and the actual structure of the individual plants change and influence how sediment particles move and settle, while acting as a control on fine scale hydrodynamics.

When looking at the interaction of waves with vegetation, it becomes more complicated and the relationships can vary more so than with tidal flow. The energy from waves requires a longer distance to dissipate and will often decay exponentially as the flow moves across the marsh (Jadhav et al., 2015; Tempest et al., 2015; Anderson and Smith, 2014). The importance in damage reduction often stems from reducing the height of the wave. The drag that is created as the wave moves across a vegetated surface is the main force that reduces the height of the wave, but it can also be reduced if the wave breaks because it reaches the reduced water depth. Wave height is reduced over the length of a transect, which varies from as low as 10% as the case with *Salicornia* sp. to as high as 94% from *Spartina alterniflora*.

In the research conducted by Tempest et al. (2015) they relate a reduction in flow velocity to the overall area of the plant. As vegetation density increases, there is also an increase in the capability for wave attenuation. This influence on wave attenuation also depends on the plant structure. Different structures influence the way which wave velocities become attenuated, or where greater amounts of attenuation occur over the species. Leonard and Reed (2002) looked at plant structure and how wave attenuation changes depending on the characteristics of different species. Their work showed how *Spartina alterniflora*, with a relatively simple structure, had a reduction of wave velocity that followed a curved flow profile and the reductions occur in the middle. Leonard and Reed (2002) also looked at different species and indicated that as the structure of a plant becomes increasingly complicated, the spatial variation becomes larger.

1.5 Monitoring protocols

1.5.1 GPAC Monitoring Protocols

One of the main documents outlining salt marsh restoration protocols is the Global Programme of Action Coalition for the Gulf of Maine, which established their report *Regional Standards to Identify and Evaluate Tidal Wetland Restoration in the Gulf of Maine* (2000). This report outlines a core set of monitoring variables that can be used to evaluate the success of salt marsh restoration. The purpose of this working group was to establish a set of potential restoration sites combined with a network of restored and reference condition marshes. In addition, this protocol is of importance for this report because it is the protocol that CBWES bases their monitoring protocols on.

The ecosystem indicators that were identified include: hydrology, soils and sediments, vegetation, nekton, birds and conducting baseline habitat mapping. These indicators are to be measured at the restoration site and at an undisturbed reference marsh in a similar physical setting both before and after the marsh is restored. The baseline habitat mapping includes a map of the monitored site, locator and cultural features (rivers, culverts, etc.), wetland type, invasive species, species of interest, the manipulations pre- and post-restoration, the locations where sampling will be occurring, and documentation of the base maps.

In hydrology, the core variables that need to be measured are the tidal signal and the surface elevations. Tidal signal can be measured by the deployment of automatic water level recorders. This data can be used to generate a hypsometric curve which provides estimates on what proportion of the marsh area become flooded for a given tide height. The hypsometric curve combined with the tidal signal will yield the hydroperiod. Additional variables that can be measured include tidal creek cross-sections, water table depth, surface water quality, salinity, dissolved oxygen and pH. It is important to measure cross-sectional profiles of major tidal creeks to monitor how they change throughout and following restoration and to measure surface water quality if an improvement in water quality is a main goal of the project.

The core variable to be measured with soils and sediments is pore water salinity. This is an important variable because of its influence on the distribution and abundance of plant species. This will provide insight into how the species composition changes throughout the restoration, and can be measured through a variety of means. Additional soil and sediment characteristics that can be measured include organic matter, sediment accretion, sediment elevation, redox potential and sulfide concentrations. Organic matter can be measured with the collection of soil cores to understand the influence of oxidation on marsh subsidence which can influence surface elevation. Sediment accretion is measured with marker horizons and can provide insight into what is being deposited on the marsh surface via flood waters, which is important since it is a main mechanism by which marshes build vertically and thus combatting sea level rise. Sediment elevation is the main factor for determining net balances of soil processes, indicating how surface is changing vertically, either growing or subsiding. This is typically measured through rod surface elevation tables (RSETs).

The goal of vegetation monitoring is to be able to determine how the vegetation changes through plant abundance and species composition and core variables include abundance, composition, height and stem density of species of concern. At each quadrat, plant species must be identified and the percent cover should be visually examined for both vegetation and bare ground. In addition, the height of the dominant species should be taken by using an average of the three tallest individuals. In order to measure the marsh community, the vegetation plots should be arranged systemically along transects. Species of concern should also be noted when conducting vegetation analysis and includes species such as *Phragmites* and *Lythrum*. The average height from the three tallest species should be collected in the quadrats where it occurs, but additional plots should be added if there is underrepresentation in the established plots.

The core variables in measuring nekton include identity, density, length, biomass and species richness. Throw traps are the recommended methodology for creeks and channels, whereas fyke nets are the recommendation for a vegetated marsh surface. These methods rely on trapping species which are then collected and identified where density, length and biomass can be measured. Additional measurements include fish growth, fish diet and larval mosquitos. Fish growth can be examined by grading the species by size class and fish diet involves further inspection the contents of the fish guts. Larval mosquitos should be measured if one of the main goals of the restoration project is to provide mosquitos control. These populations can be sampled in pannes and standing bodies of water with a dipper, a cup attached to a long stick. Birds are an essential component of salt marsh monitoring when the goal of the project is to increase bird use, and includes monitoring abundance, species richness and the feeding and breeding behaviour. This type of monitoring should be conducted at a variety of vantage points, where an observer will view species for 20 minute periods. Additional sampling will include small passerines and other cryptic birds of the salt marsh, birds in the buffer and waterfowl in winter.

1.5.2 The Narragansett Bay National Estuarine Research Reserve

The Narragansett Bay National Estuarine Research Reserve also released their Salt Marsh Monitoring and Assessment Program (SMMAP) which aims to promote long-term monitoring in Rhode Island. SMMAP aims to assess how salt marsh conditions, spatial extent and community composition changes in both space and time. This project not only outline monitoring protocols for restoration activities but identifies standardized protocols for data formatting and data archiving. SMMAP identify 12 monitoring parameters: geomorphic parameters, habitat composition, edaphic conditions, elevation, elevation change and accretion, hydrology and inundation, nutrients, total suspended solids, emergent vegetation, marsh crabs, nekton and birds.

The geomorphic parameters include channel and marsh edge erosion and calving rate, landward transgressive rate – to see the rate that the habitat is moving into the upland habitats due to sea level, and marsh area and ponding area. Habitat comparison will look at the community composition and plant zonation. This involves both digital analysis as well as a field component to characterize zonation and composition. Edaphic conditions were initially measured with a bearing capacity and a soil penetrometer, but have since changed to hand held shear vane which can measure the soil strength at different depths. Elevation is measured through LiDAR data or field surveys requiring RTK-GPS equipment where the remote sensing is not available. Level surveys should be conducted to collect the elevation of loggers, SETs, vegetation plots and the other monitoring infrastructure.

Elevation change and accretion is measured through the use of the surface elevation table-marker horizon method. Sediment tiles have also been used increasingly for this purpose in salt marshes. Hydrology and inundation includes the use of water level loggers allowing the inundation on the marsh surface to be calculated as well as groundwater depth. Nutrients such as nitrogen can be measured in the field, or they can be determined with a GIS land use analysis combined with nitrogen loading models. Total suspended solids are to be measured at regular intervals over a long duration of time at various marsh sites. Emergent vegetation is to be measured through a series of transects with several $1m^2$ quadrats placed along each transect. This will provide species composition, species richness, percent cover and heights and stem densities of dominant species. Additionally, aboveground and belowground plant biomass could be considered. Marsh crabs should be measured with replicated crab burrow counts on creek banks until a standardized salt marsh crab monitoring protocol is created. Nekton should be measured similar to the GPAC protocol, with the use of throw traps and ditch nets in order to gain insights into community composition, richness, density and size class. Birds act as a bio-indicator and should be measured using call-back protocols and transect and point surveys in the warm season.

CHAPTER 2: Study Area

2.1 The Bay of Fundy

The Bay of Fundy is located between Nova Scotia and New Brunswick and contains faults dated to the Paleozoic era that were activated when the Atlantic Ocean started opening due to shifting plate tectonics. This led to the sedimentary infill process that created the Fundy Basin. (Desplanque and Mossman, 2004). Throughout the last 14 000 years, the Bay has experienced considerable changes with respect to relative sea level rise. As George's Bank became submerged, the Bay of Fundy became subjected to larger tidal forces (Desplanque and Mossman, 2004). It is currently characterized as a macrotidal estuary that experiences a tidal range of 5.0 m at the mouth of the Bay, 7.3 m near the head of Chignecto Bay, with an average range of 12.0 m that can reach up to 16 m at Burntcoat Head (Desplanque and Mossman, 2004). The Minas Basin was not always macrotidal and switched from a microtidal regime sometime between 4000 to 6000 years ago (Grant, 1970). The Minas Basin has a semi-diurnal tidal regime which means that there are two high tides and two low tides that occur in a 24-hour period. The tidal range increases the closer you are to Cobequid Bay, and each tide brings an estimated 3 billion cubic meters of water in and out of the Minas Basin from each tide (Desplanque and Mossman, 2004). Due to these unique tidal conditions The Bay of Fundy is comprised of expansive tidal wetlands that are minerogenic and macrotidal in nature (Byers and Chmura, 2014).

2.2 Cogmagun River salt marsh

Cogmagun salt marsh is a tidal wetland located on the Cogmagun River, a tributary of the Avon River, located in Hants County (Figure 1). The Avon River is located near the entrance of the

Minas Basin, which is located in the Nova Scotian portion of the Bay of Fundy.The Cogmagun salt marsh consists of a 6.9-hectare tidal wetland that was a former failed Ducks Unlimited Canada impoundment (Figure 2). The site was restored in 2010 with the creation of a 60-metre breach in the existing dyke infrastructure. Previous to restoration, the site was considered a poorly drained brackish/freshwater environment that contained many areas with standing water (Bowron et al., 2015). Since the restoration, non-halophytic vegetation has decreased, and there has been an increase in indicator species with the site experiencing regular tidal flow.

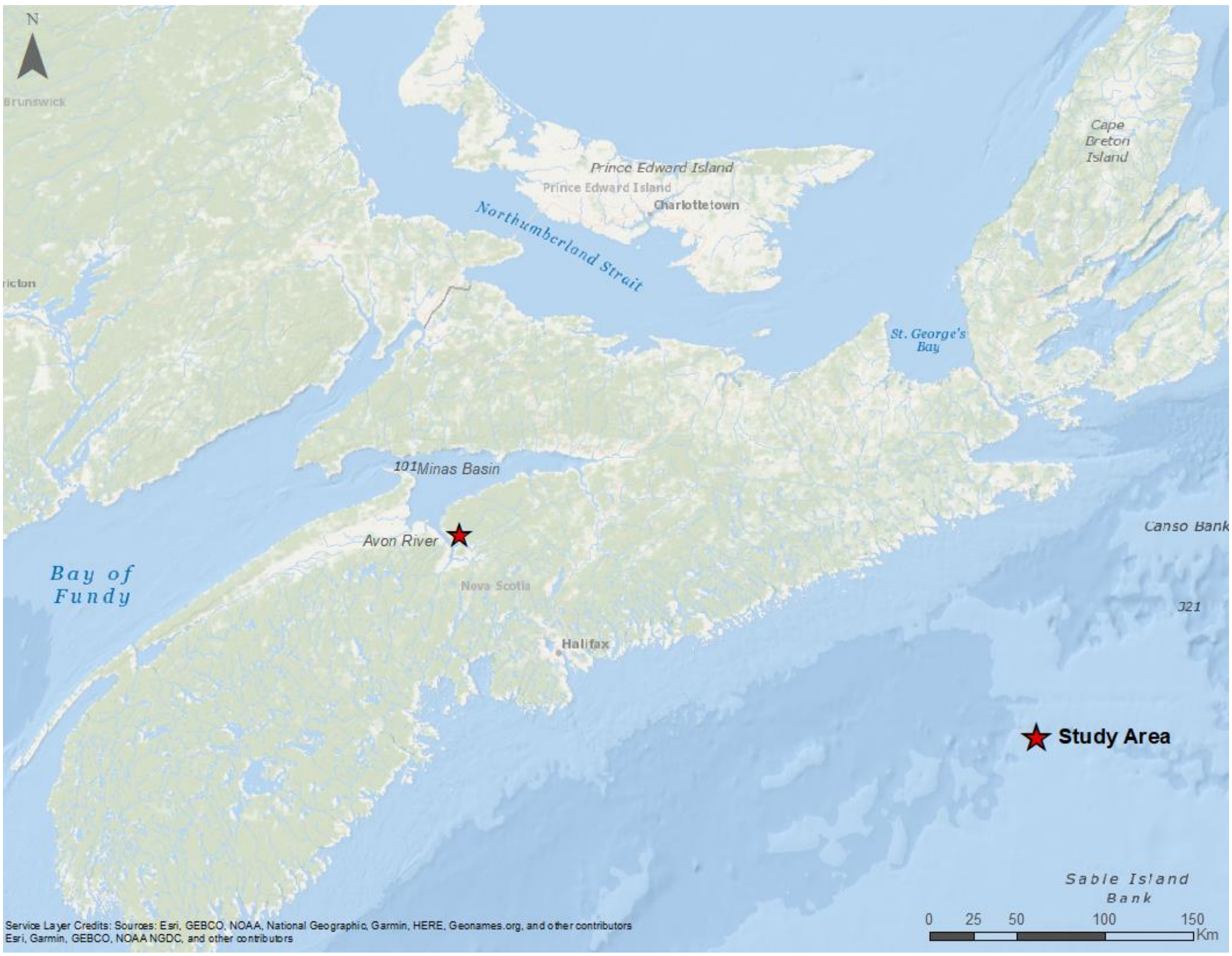

Figure 1. Map displaying where the Cogmagun study area is located.

Figure 2. Location of the Cogmagun Study site relative to the Cogmagun River tributary of the Avon River.

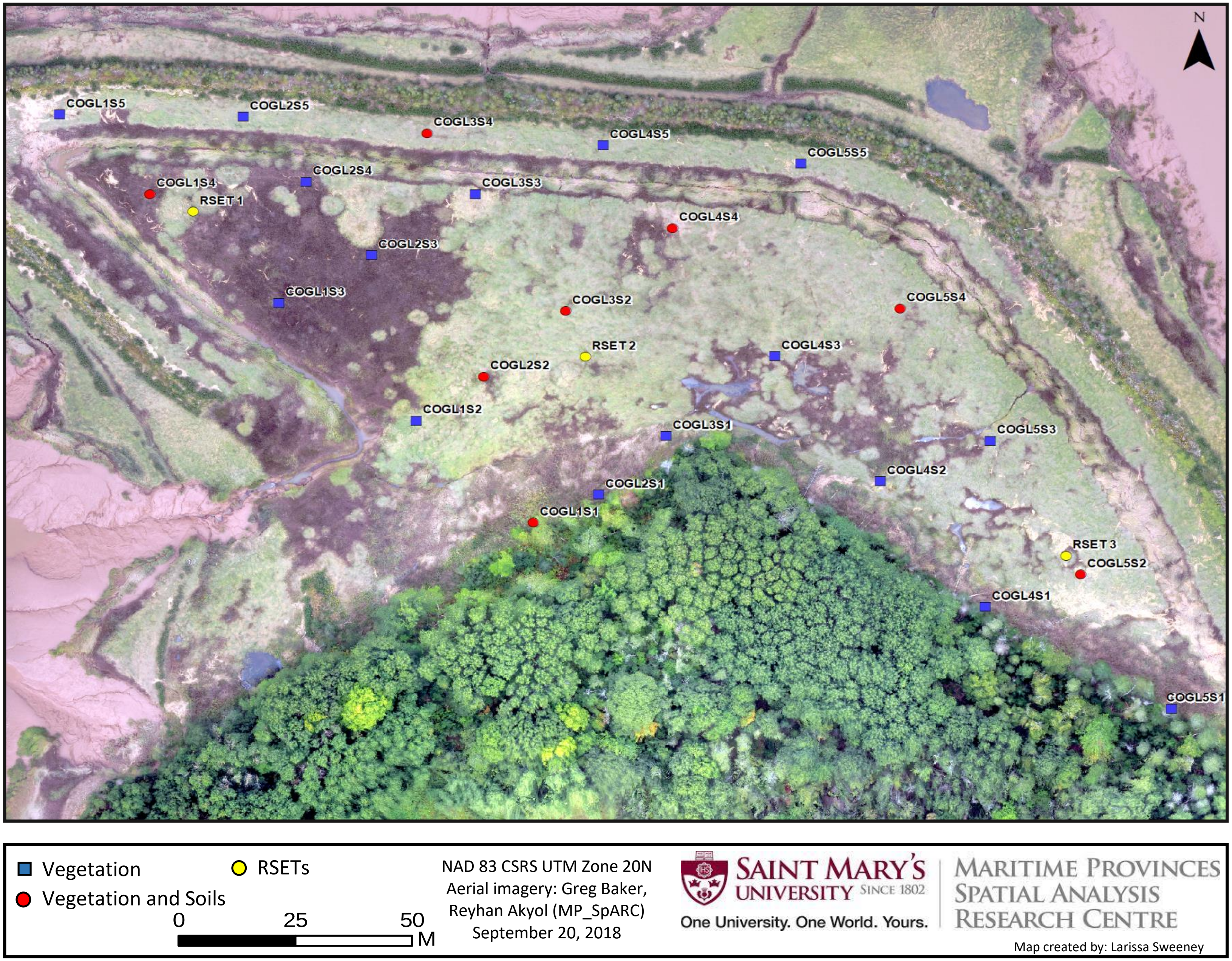

Figure 3. Locations of vegetation sampling plots at Cogmagun restoration site.

CHAPTER 3: Methodology

Coastal processes are influenced by erosion, sediment transport, sediment deposition, and how they interact with adjacent land (van Proosdij, 2016). The ecological indicators to be measured in this research include; hydrology, soils and sediment and vegetation. These measurements were compared to the last available data for the reference site to determine if they are following the similar trajectory in physical, chemical and biological characteristics. Samples were collected at the same sampling locations as the previous data, which occurs along a series of five transects at intervals that characterize different zones of marsh habitats identified by CBWES (Figure 3). All vertical data was measured in the Canadian Geodetic Vertical Datum (CGVD) 28, with the exception of the tidal data which was measured in CGVD 2013.

3.1 Vegetation

The composition, abundance and height of vegetation was measured on August 31, 2018, using 1 m^2 plots along five designated transects for a total of 24 plots at the Cogmagun restoration site (Figure 3). The grid was placed in the sampling locations that were previously flagged and identified with the RTK, where it was offset one meter from the main tidal channel It was also noted if there was a distinct high-low marsh zonation identified by signature species such as S. *alterniflora* (low-marsh) and *S. patens* (high marsh). Each vegetation plot was divided into 25 subsections that were 20 cm by 20 cm, where each smaller square was measured in the same location using the point intercept method (Roman et al., 2002). All species in the quadrat were initially identified, and a metal dowel (5 mm in diameter) was lowered into the vegetation, recording what the rod was hitting as it was lowered. The species that touched the metal dowel were recorded. Photos at each sampling station were also taken for reference (Figure 4).

Statistical analysis was used to quantify the differences between the restoration site and the reference site. Species richness and relative species abundance were used to gain insights into species diversity. Species richness is the number of different species within the sampling site with halophytic species richness looking at the number of salt tolerant species. Species abundance and halophytic species abundance measures how many individuals are within each of the different species. Plant species richness and halophytic species richness were measured by counting the number of different species within each sampling plot and within the entire site. Species abundance and halophytic species abundance was measured by counting the number of individuals that were recorded while performing the point-intercept method. These parameters were analysed using a paired T-test of the vegetation data collected in Summer 2018 and the last available data for Cogmagun reference was used as the comparison. Species abundance was analyzed through the use of a table that shows the average species abundance for each species in each year, from selected the years in 2009 to 2018.

A habitat map that differentiates vegetation was generated using ArcMap and low-altitude aerial photography taken on September 20, 2018 using a DJI Phantom Drone 3 Pro with an RGB camera. The imagery was processed using Pix4D photogrammetry software to generate an orthomosaic and a digital surface model (Pix4Dmapper User Manual, 2017). Ground control points (GCPs) were deployed in the field to increase the accuracy when generating the output.

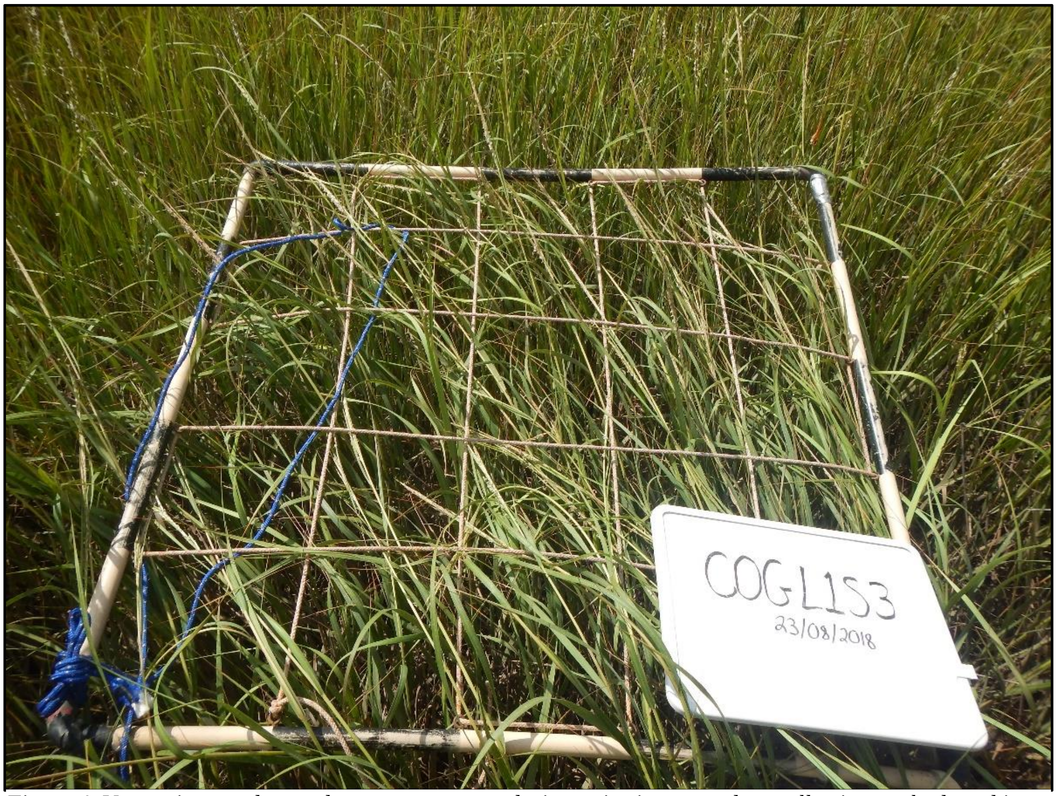

Figure 4. Vegetation quadrat to demonstrate set-up during point-intercept data collection method used in the field. Photo taken August 23, 2018.

3.2 Hydrology

The parameters for hydrology include monitoring the tidal signal and hydroperiod. Tidal signal was measured using a Solinist Model 3001 Levelogger Gold that was deployed in the excavated creek to take measurements at five-minute intervals for a total of 75 days, collecting full tidal data for both spring and neap tides. A Hobo Barologger was attached to a tree in the adjacent forested area to take measurements at the same interval in order to compensate the data for atmospheric pressure. The loggers were deployed on June 11, 2018 and retrieved on August 31, 2018. Using the recorded data from June 1 – August 25, 2018, minimum, maximum and mean water levels were calculated from the data to determine the average tidal coverage. Hydroperiod was analyzed by combining the values extracted from the tidal data with survey elevation points spanning the marsh surface. This shows how long each of the surveyed plots remain submerged, allowing you to distinguish between low marsh, high marsh and the upland edge.

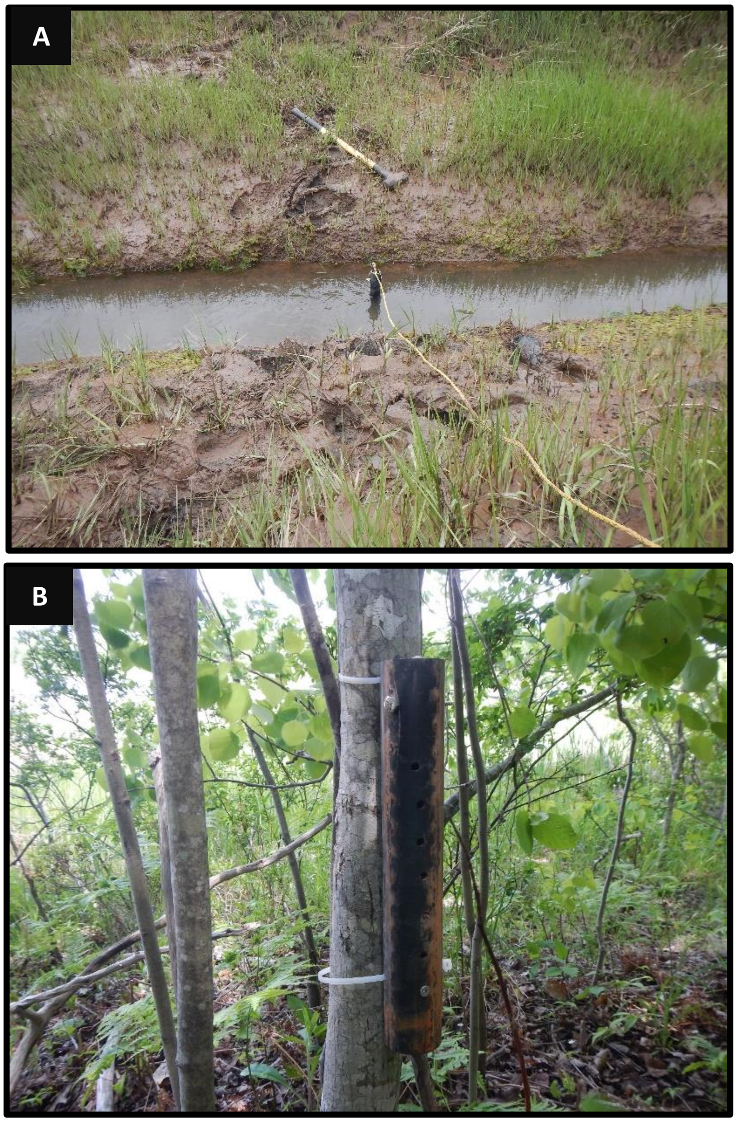

Figure 5. Location and set-up of A) Levelogger and B) Barologger in the tidal creek and adjacent forest, respectively. Photos taken June 11, 2018.

3.3 Soils and sediments

Measuring soils and sediments involve looking at factors that affect the change in surface elevation and sediment composition. Rod surface elevation tables (RSETs) (Cahoon et al., 2002) and feldspar marker horizons (Cahoon et al., 1996) were installed on August 19, 2009 and have been used to look at changes in sediment accretion and surface elevation. The RSET contains an arm with nine measuring pins that attach to the collar which is attached to the bench mark to ensure measurements are taken in the same location each time. The arm was leveled and lowered at each of the measuring pins until they reach the surface of the marsh, where they were secured (Figure 5). This provided the measurements for elevation as each pin is measured from the top of the pin to the arm. Accretion was measured using feldspar marker horizons and a cryogenic corer to take a frozen sediment core. This core was marked with a distinct feldspar layer from the time of installation (Figure 5). This method allows the amount of deposited material onto the marsh surface to be measured from a reference point.


A

Figure 6. Example of rod surface elevation table measuring technique (a) and (b) and marker horizon initial set-up. Photos taken June 4, 2018 at Belcher Street salt marsh.

One soil core and one soil syringe were taken at eight locations in the restoration site (Figure 3). Soil cores were used to measure water content and organic matter, while soil syringes measured bulk density. Soil cores were taken by pushing a metal tube that was 10 centimeters in length and 4 centimeters in diameter into the marsh surface and extracting the core while trying to minimize compaction. Each core was given a lid and sealed where they were brought back to the freezer on ice. A 60-millilitre syringe was used to take a 30-millilitre soil syringe at each location, again aiming to reduce compaction during the procedure. Each syringe was sealed and kept cold until placed in the freezer for analysis. The cores were analyzed for water content, organic content and sediment type, while the soil syringes were used to analyze bulk density. The sediment cores were thawed before laboratory analysis occurred and were characterized by noting key characteristics. These key characteristics included anoxic layers identified through the presence of black layers in the restoration site which is important, as these were not present at the reference site. Strong smells, stratification of soil layers and root and vegetated matter were also noted. Vegetative matter was noted to show evidence of established vegetation to determine if the soils can support a vegetative community.

Organic content was measured using loss on ignition (LOI) protocols (In_CoaST, 2018) where a known volume of the core was removed and dried at 105 °C for 24 hours. The difference in mass was used to determine the water content of the sample and multiplied by 100 to obtain a percent value (Equation 1).

Equation 1.

$$\text{Water content (\%)} = \left(\frac{\text{wet weight} - \text{weight } 105\ {}^{\circ}\text{C}}{\text{wet weight}}\right) * 100$$

After the water content was determined the sample was place in a muffle furnace at 550 °C for two hours. The samples were stored in a desiccator container and weighed once they were cooled, providing the LOI of organic material from the difference in mass (Equation 2).

Equation 2.

$$\text{Organic Matter content (\%)} = \left(\frac{\text{weight } 105\ {}^{\circ}\text{C} - \text{weight } 550\ {}^{\circ}\text{C}}{\text{weight } 105\ {}^{\circ}\text{C}}\right) * 100$$

Dry bulk density was determined by measuring the depth at which the core occupied the syringe, weighing the wet samples and then weighing the dry samples after placing the samples in a labelled aluminum tin at 105 °C for 24 hours. Dry bulk density was calculated by calculating the difference in wet and dry weight and dividing by the volume following Equation 3.

Equation 3.

$$\text{Dry Bulk Density } \left(\frac{\text{g}}{\text{ml}}\right) = \frac{\text{net dry weight at } 105{}^{\circ}\text{C}}{\text{volume(ml)}}$$

CHAPTER 4: Results

4.1 Vegetation

When comparing the spatial variation of vegetation within the Cogmagun restoration site in 2018, there are several trends to be noted such as zonation of vegetation. Vegetation closest to the Cogmagun River and the excavated channel in the low marsh shows that *S. alterniflora* (Figure 6) has the greatest relative abundance (16.67). In the high marsh, *S. patens* (Figure 6) has the greatest abundance (21.27) but is often mixed with *Distichlis spicata* (5.6) on the channel side of the dyke (Figure 2). The first sampling site in each transect is dominated by upland species, with *Typha angustifolia* (Figure 6) having the greatest relative abundance (4.53). The relative abundance of *Spartina alterniflora*, the dominant low marsh species, has decreased from 19.17 to 16.67, while *Spartina patens*, a dominant high marsh species, has increased from 8.21 in 2014 to 21.27 in 2018.

Figure 7. Vegetation species measured at the Cogmagun restoration site showing a) Typha angustifolia b) Spartina patens c) Spartina alterniflora and d) Suaeda maritima. Photo a by Antoine Balaz. Photo b, c, d by Larissa Sweeney. Photos taken May 8, 9, 2018.

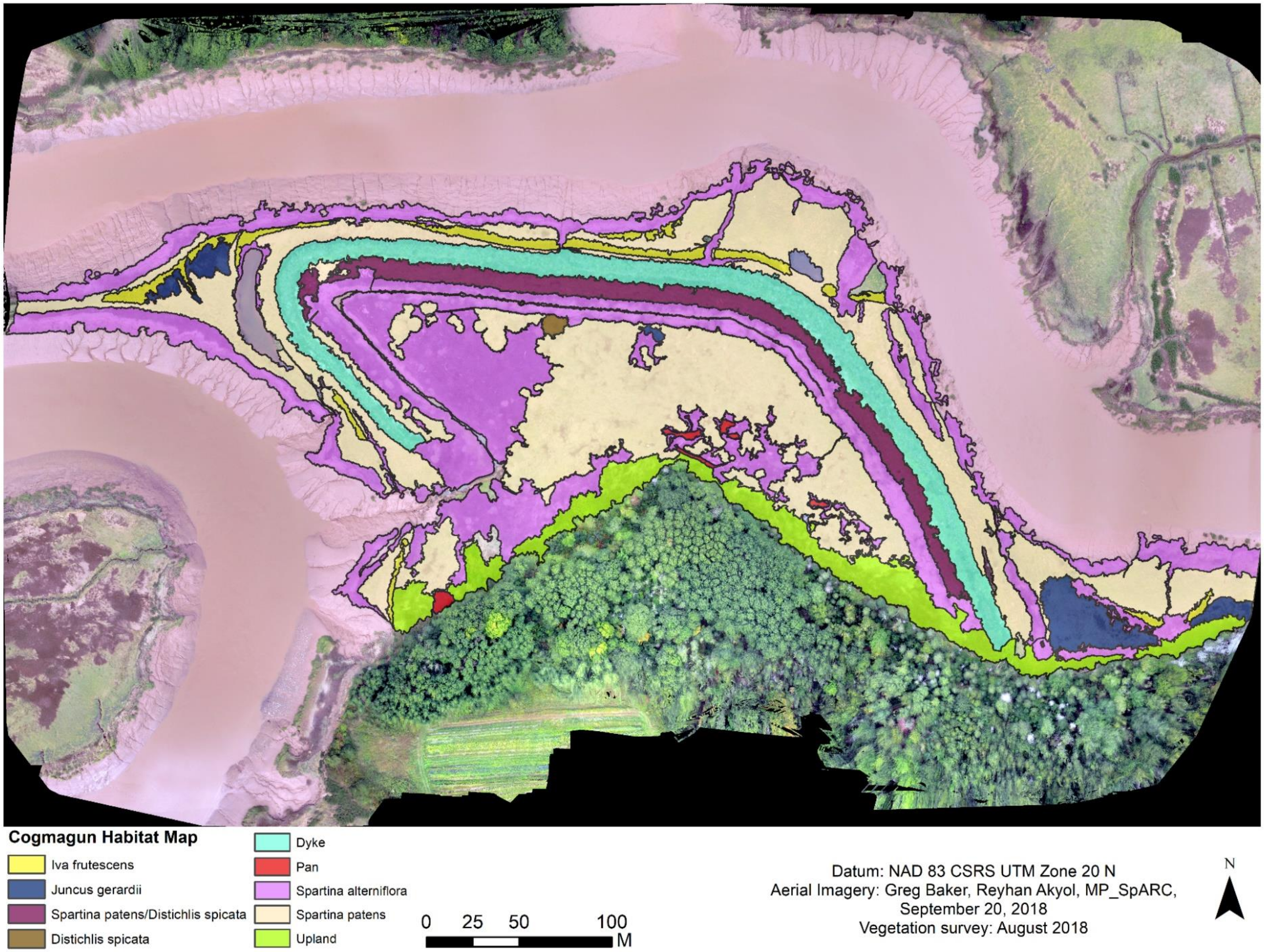

Figure 8. Cogmagun restoration habitat map for 2018 showing different species and clustering of species.

Relative vegetation abundance and frequency were also compared to past data collected for the Cogmagun restoration site to show how the site has changed over time (Figure 7). The data collected in 2018 from the Cogmagun restoration site differed from the data collected in 2014 at the same site by the absence of *Carex paleacea*, *Hierochloe odorata*, *Limonium carolinianum*, *Poa palustris*, *Scirpus maritimus*, *Solidago sempervirens*, *Symphotrichum novi-belgii*, *Triglochin maritima* and *Typha latifolia.* The only species present in 2018 that was not present in 2014 is *Aster spp*. Species richness, halophytic species richness and halophytic species abundance shows a decrease from the data collected in 2014 (Figure 8, 9, 10). One of the main differences noted in the habitat map from 2014 – 2018 is that most of the marsh surface was characterized by a transition community that comprised of low marsh, high marsh and pioneer species. In 2018, zones of vegetation from the low marsh are separating from zones in the high marsh which is noticeable in Figure 7.

Relative vegetation abundance and frequency were also compared between the Cogmagun restoration data from 2018 and the Cogmagun reference data from 2014. This was done to show how the present conditions at the restoration site compares to a mature salt marsh community. The changes were noted with the presence of *Agrostis stolonifera*, *Aster spp.*, *Phragmites australis* and *Typha angustifolia* and the absence of *Atriplex glabrisculata*, *Carex paleacea*, *Elymus repens*, *Limonium carolinianum*, *Solidago sempervirens*, *Triglochin maritima.* Species richness, halophytic species richness and halophytic species abundance all decreased from the values recorded in 2014 (Figure 8, 9, 10).

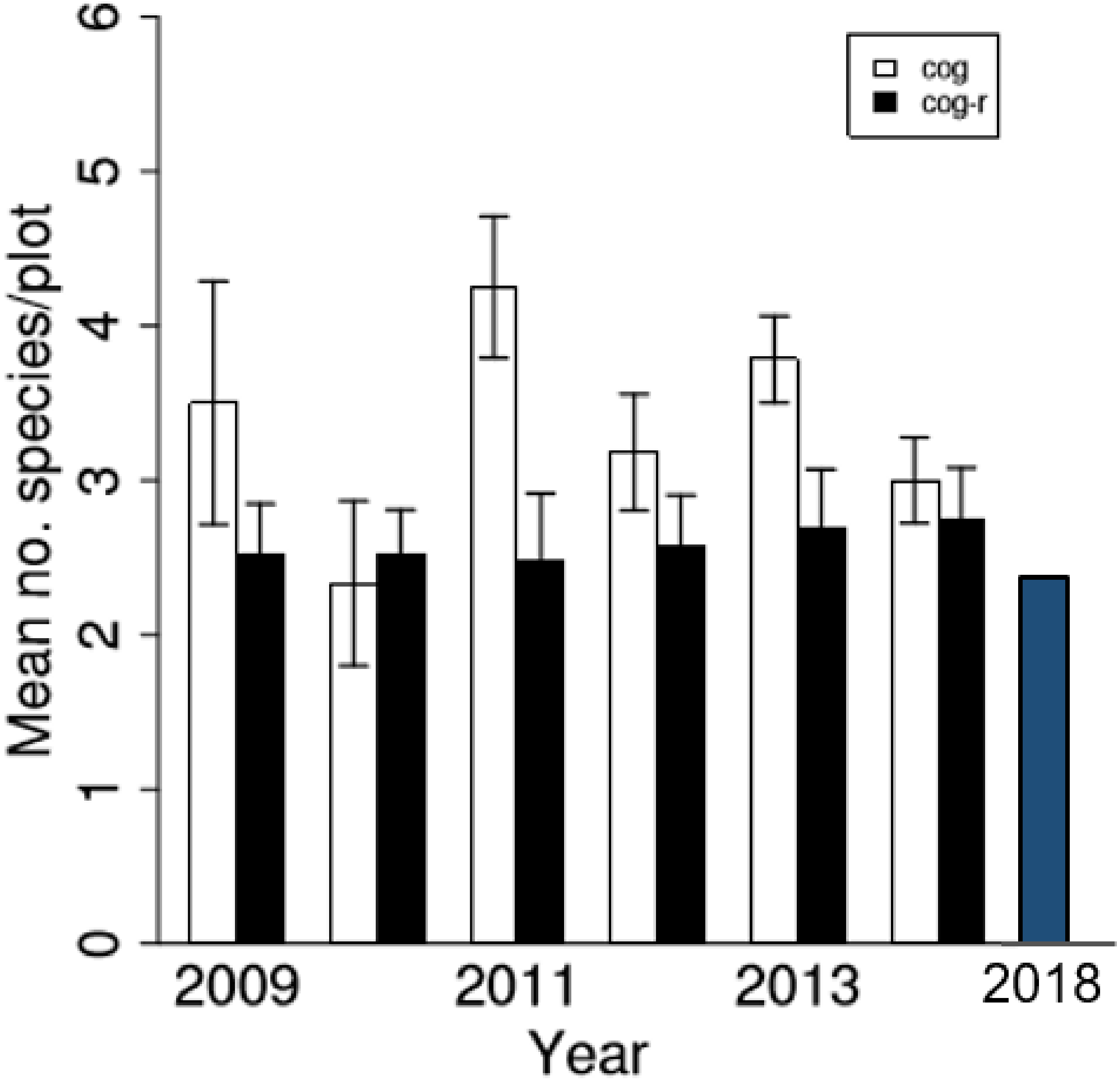

Figure 9. Species richness from 2009 - 2018 at Cogmagun restoration and Cogmagun reference (Modified from Bowron et al., 2014)

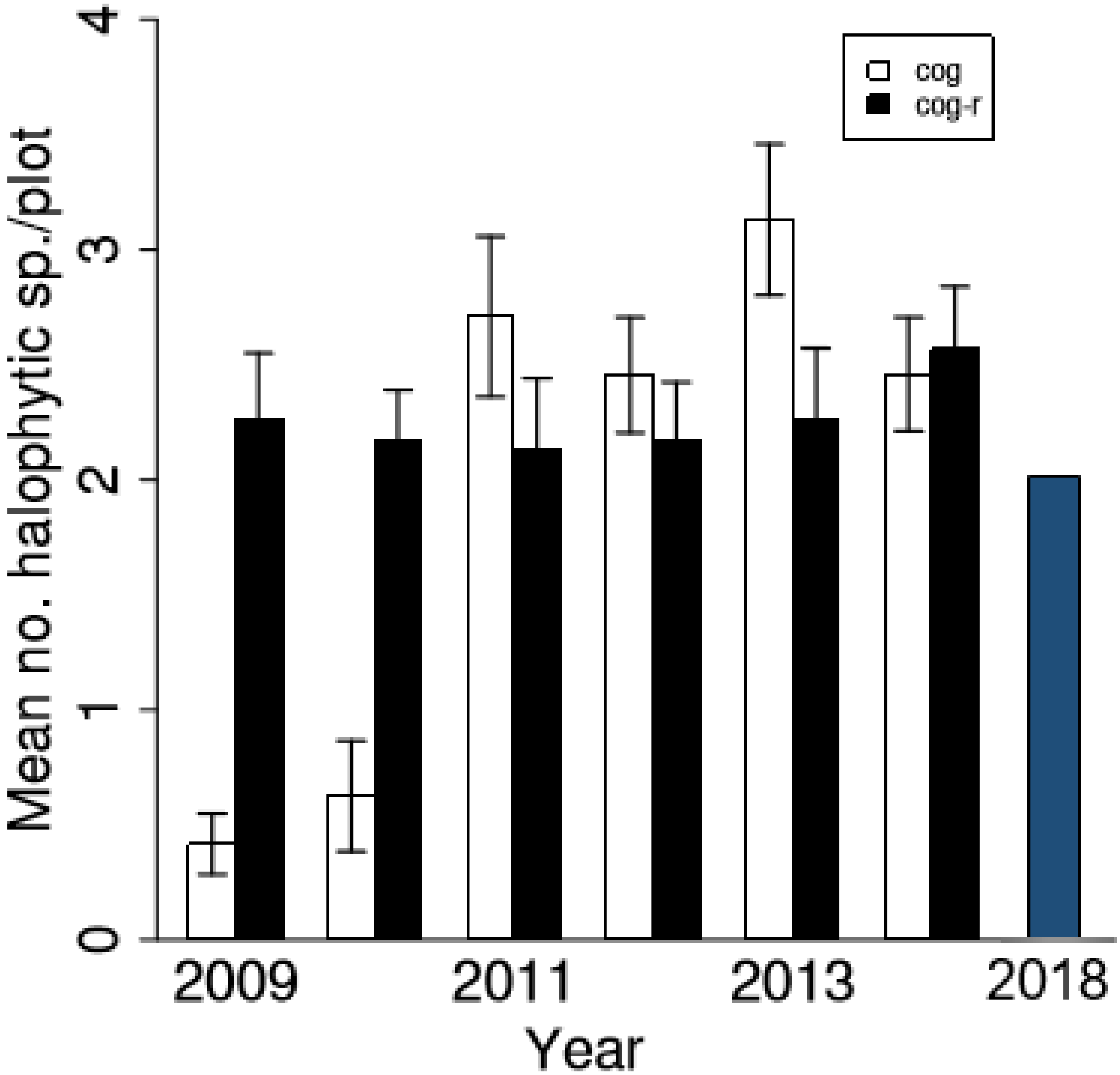

Figure 10. Halophytic species richness from 2009 - 2018 at Cogmagun restoration and Cogmagun reference (Modified from: Bowron et al., 2014)

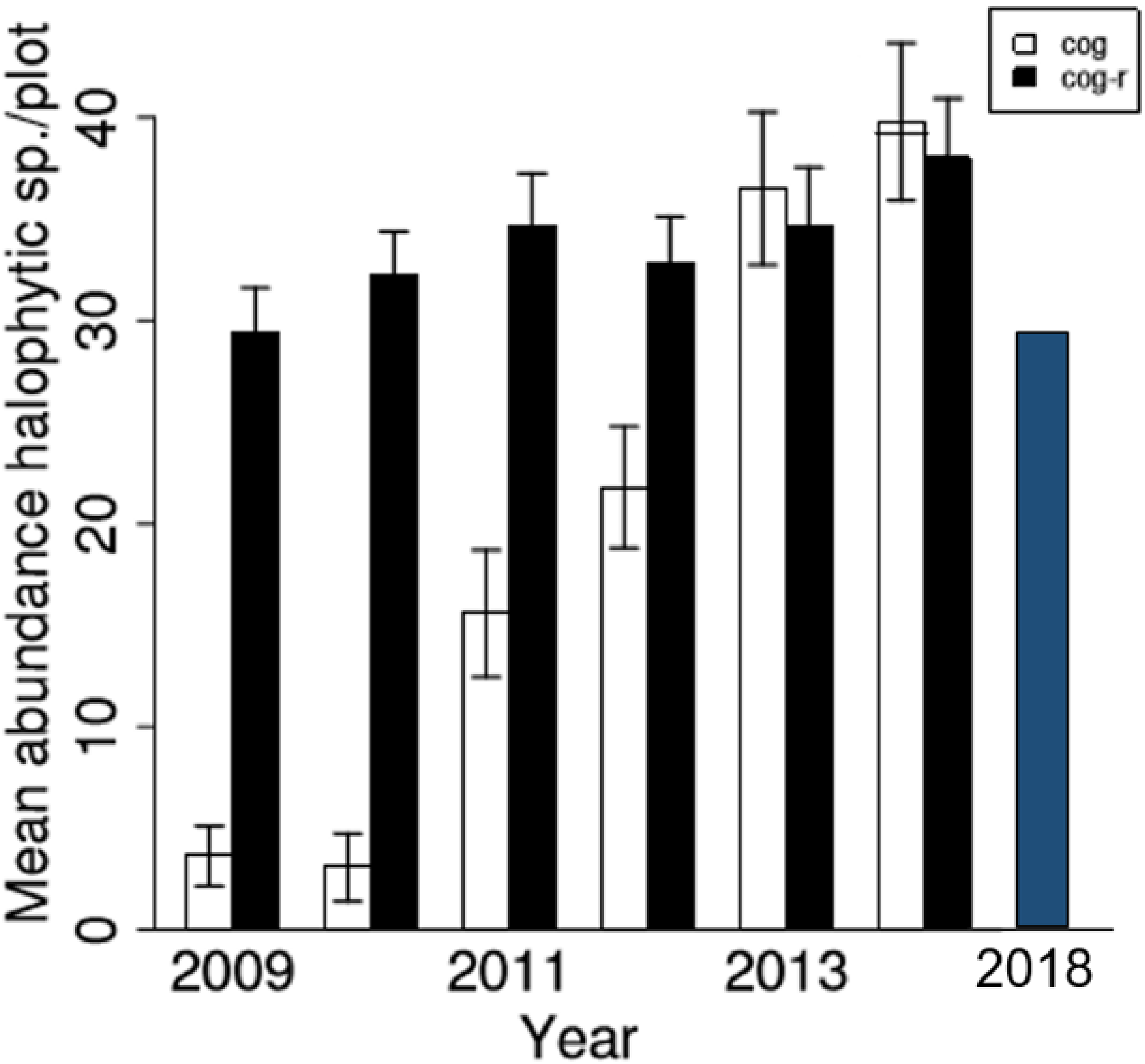

Figure 11. Relative species abundance from 2009 - 2018 at Cogmagun restoration and Cogmagun reference (Modified from Bowron et al. 2014)

These differences were also statistically analysed with vegetation data collected using the point intercept method in August, 2018. For the analysis, the Cogmagun restoration site was compared to the last available data at the Cogmagun reference site measured in 2014. Species Richness, halophytic species richness and halophytic species abundance were statistically analysed using a two-sample T-test in Minitab® 18, with the null hypothesis indicating that there was no difference between the reference and restoration site for each of the variables (H_0: $\mu_1 - \mu_2 = 0$).

Table 1. Two-sample T-test results for species richness, halophytic species richness and halophytic species abundance.

Variable	**Confidence Interval (%)**	**P-value**
Spp. Richness	95	0.355
Halophytic Spp. Richness	95	0.094
Halophytic Spp. Abundance	95	0.035

It was determined Species Richness had a p-value of 0.355, indicating that p-value is greater than the confidence interval, therefore we accept the null hypothesis (p-value > α) (Table 1). Halophytic species richness had a p-value of 0.094, which also is greater than the confidence interval (p-value > α), and we can again accept the null hypothesis that there is no significant difference (Table 1). For halophytic species abundance, the p-value was 0.035, which is less than the confidence interval (p-value < α) (Table 1). In this case we must accept the alternative hypothesis, indicating that there is a difference between the conditions at the reference site compared to the restoration site.

4.2 Hydrology

The Levelogger's that were deployed in the tidal creek measured water levels from June 11, 2018 to August 25, 2018 to account for a full range of tidal conditions that affected the Cogmagun site throughout the data collected period. Shown also on the graph is the average elevation of the low marsh (6.03 metres), high marsh (6.19 metres) and the upland edge (6.58 metes) measured in CGVD 2013 (Figure 11). These elevations were determined by averaging the surveyed vegetation plots in August, 2018 that displayed characteristic species of each marsh zone.

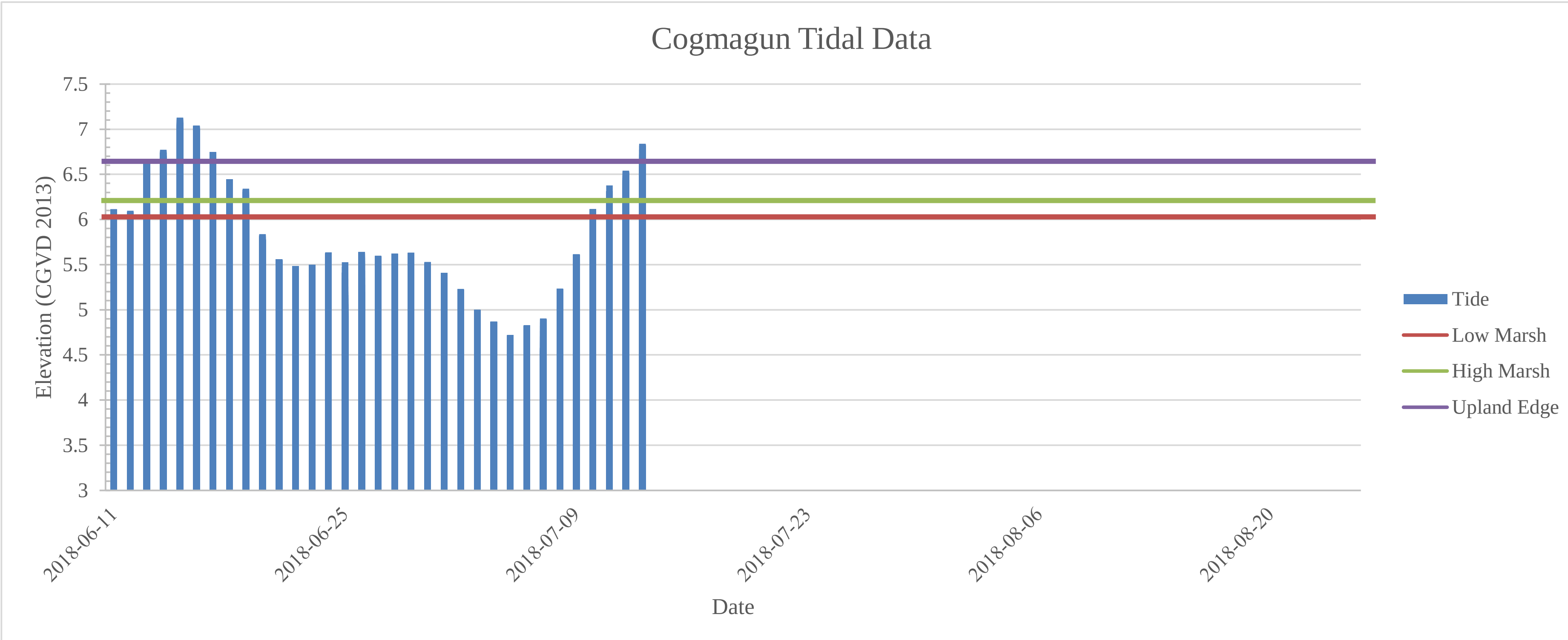

Figure 12. Tidal range data from June 11, 2018 to August 25, 2018 in CGVD 2013. Elevations calculated based on average heights of vegetation plots demonstrating each zonation characteristics.

This graph shows how the low marsh is the first zone to flood, followed by the high marsh and then the upland edge when the tide when tidal conditions exceed 6.58 metres. The maximum water level is comparable to the previous years at 7.8 m when converting the value to match the previous data in CGVD 28 (Table 2). Also derived from this data is the hydroperiod for the vegetation sampling plots as both a percent and duration in minutes and inundation frequency with the mean inundation time ordered from increasing elevation (Table 3).

Table 2. Maximum water level in study area from 2009 - 2018 in CGVD 28.

Year	**Date range**	**Maximum Water Level in Study (m CGVD 28)**
Pre 2009	July 16 – September 14	--
Post 2009	October 8 – November 23	7.6
2010	August 19 – October 4	8.2
2012	November 9 – December 5	7.6
2014	September 5 – November 24	8.0
2018	June 11 – August 25	7.8

Table 3. Hydroperiod and inundation frequency compared to dominant vegetation and zonation at the Cogmagun restoration site based on a 75-day recording from June 11, 2018 to August 25, 2018. Data are ordered from increasing elevation (CGVD 2013) that were recorded on May 18, 2018.

Zone	**Elevation (m CGVD 2013)**	**Hydroperiod (%)**	**Inundation Frequency (%)**	**Dominant Vegetation**
Low Marsh	5.95 - 6.09	3.46 - 2.74	31.13 - 27.15	*Spartina alterniflora*
High Marsh	6.12 - 6.22	2.59 - 1.91	24.5 - 21.19	*Spartina patens*
Upland Edge	6.40 - 6.88	1.52 - 0.29	18.54 - 5.30	*Typha angustifolia*

When looking at how the hydroperiod and inundation frequency compare to vegetation zonation there is a relationship between the dominant vegetation and the respective zone. This zonation allowed ranges to be identified for each hydroperiod and inundation frequency. In the low marsh zone dominated by *Spartina alterniflora*, elevation ranged from 5.95 – 6.09 m CGVD 2013, the hydroperiod ranged from 3.46 – 2.74 %, and the inundation frequency ranged from 31.13 – 27.15 % (Table 3). The high marsh was dominated by *Spartina patens* and elevation ranged from 6.12 – 6.22 m CGVD 2013, hydroperiod ranged from 2.59 – 1.91 %, and the inundation frequency ranged from 24.5 – 21.19 % (Table 3). The upland edge was dominated by *Typha Angustifolia* and the elevations ranged from 6.40 – 6.88 m CGVD 2013, the hydroperiod ranged from 1.52 – 0.29 % and the inundation frequency ranged from 18.54 – 5.30 % (Table 3).

Clustering can be noted throughout the hydroperiod and inundation frequency with small variations in the elevation, but it is most notable in the inundation frequency percent. One of the ain clustering events occurs at 6.117 m where the inundation frequency is 24.5% and remains at this value until 6.182 m indicating that all values within this range are inundated at the same frequency (Table 3).

4.3 Soils and Sediments

4.3.1 Soils

Soils samples were cut open for processing and key characteristics such as anoxic layers were noted by the presence of black layering or sections in the cores. Anoxic conditions were present at LIS4, L2S2, L3S2, L3S4, L5S2 and L5S4. In addition, root matter was present in all cores with many cores displaying grasses and other vegetative material. These samples were then analyzed for water content, organic matter and dry bulk density. Water content varied within the Cogmagun restoration site in 2018 with the highest percentage (78%) at the L1S1 sampling site and the lowest percentage (36 %) at L1S4 (Figure 12).

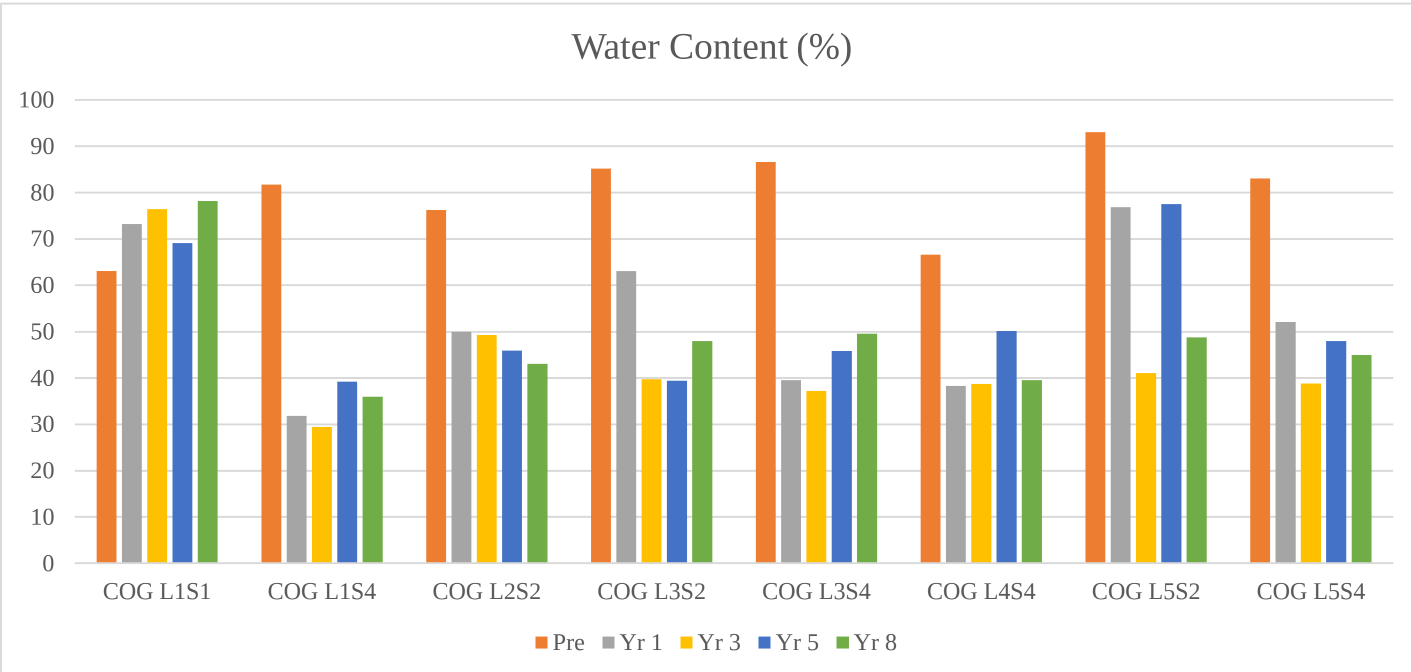

Figure 13. Water content (%) measure at Cogmagun restoration for each sampling site from pre-restoration to ninet-years post restoration.

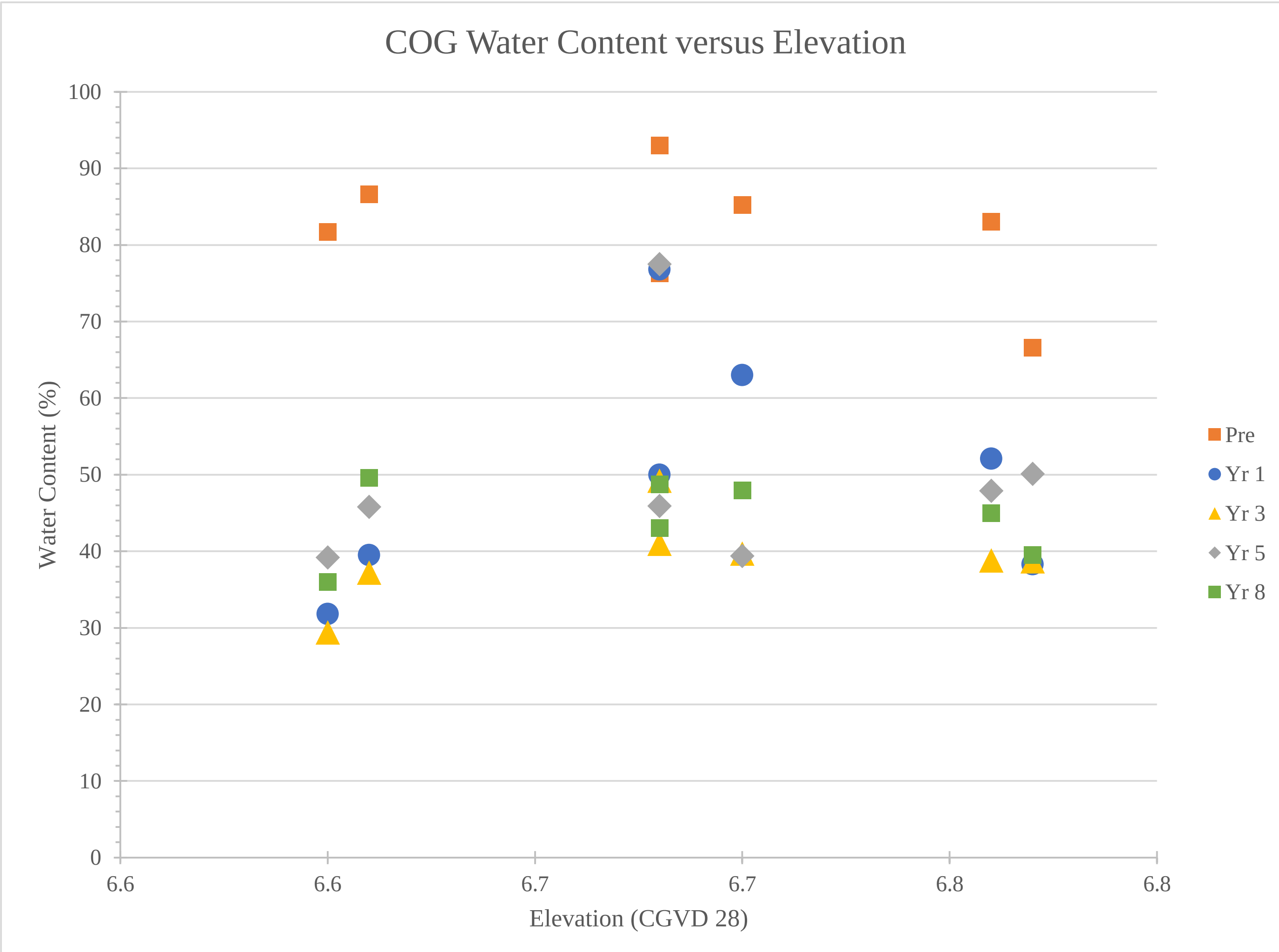

Figure 14. Water content (%) versus elevation (CGVD 28) at Cogmagun restoration. Upland site (7.5 m CGVD 28) removed to show relationship.

When comparing water content from pre-restoration to the data collected in 2018 versus elevation, there is a decreasing trend from pre-restoration to nine years post restoration. This is notable at all sites with the exception of LIS1, which experienced a general increasing trend from pre-restoration to nine years post restoration (Figure 13). When considering all data at the Cogmagun restoration site, the highest water content was recorded pre-restoration at site L5S2 reaching 93%, and the lowest water content was recorded in year three post restoration at 29.4% (Figure 12).

Comparing the 2018 values to the last available data at the reference site measured in 2014 show differences and similarities at each site. The range of water content values are similar at each site with the restoration site ranging from 36% to 78%, versus the reference site which ranges from 37% to 65%. Overall, the water content values are lower at the restoration site compared to the reference site when excluding the upland outlier at a higher elevation. The average water content values at the restoration site are 44% when excluding the upland site, compared to 53% at the reference site in 2014. When including the upland site at Cogmagun restoration, the average water content at the restoration site increases to 49%.

In 2018, the organic matter varied within the site by the highest percentage (46.2 %) at site L1S1 and the lowest percentage (5.9 %) at L1S4 (Figure 14). Organic matter plotted versus elevation

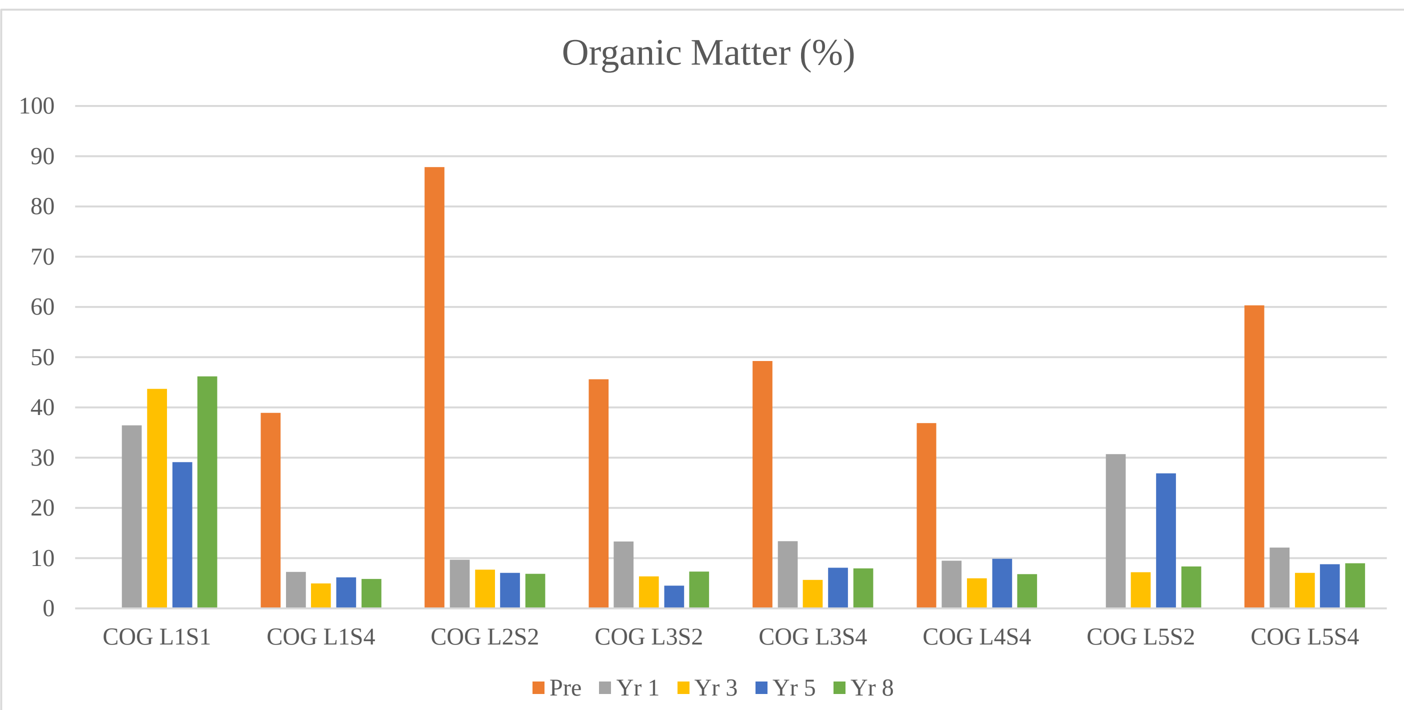

Figure 15. Organic matter (%) measured at Cogmagun restoration for each sampling site from pre-restoration to nine-years post restoration.

measured from pre-restoration to current conditions at the reference site shows a decreasing trend from pre-restoration to nine years post restoration (Figure 15). This trend is true for all sampling sites except the most upland site (L1S1), where the inverse relationship is true due to the site existing at the highest elevation (Figure 15). Both the highest and lowest recorded data, 122% and -4.2% respectively, for organic matter appear to be errors within the previous data and were excluded from Figure 14 and Figure 15. Errors aside, the highest percent of organic matter was recorded during the pre-restoration at site L5S2 which was recorded as 88% organic matter, and the lowest percent organic matter was also recorded during pre-restoration, but at site L1S1, which was measured as 5% (Figure 14).

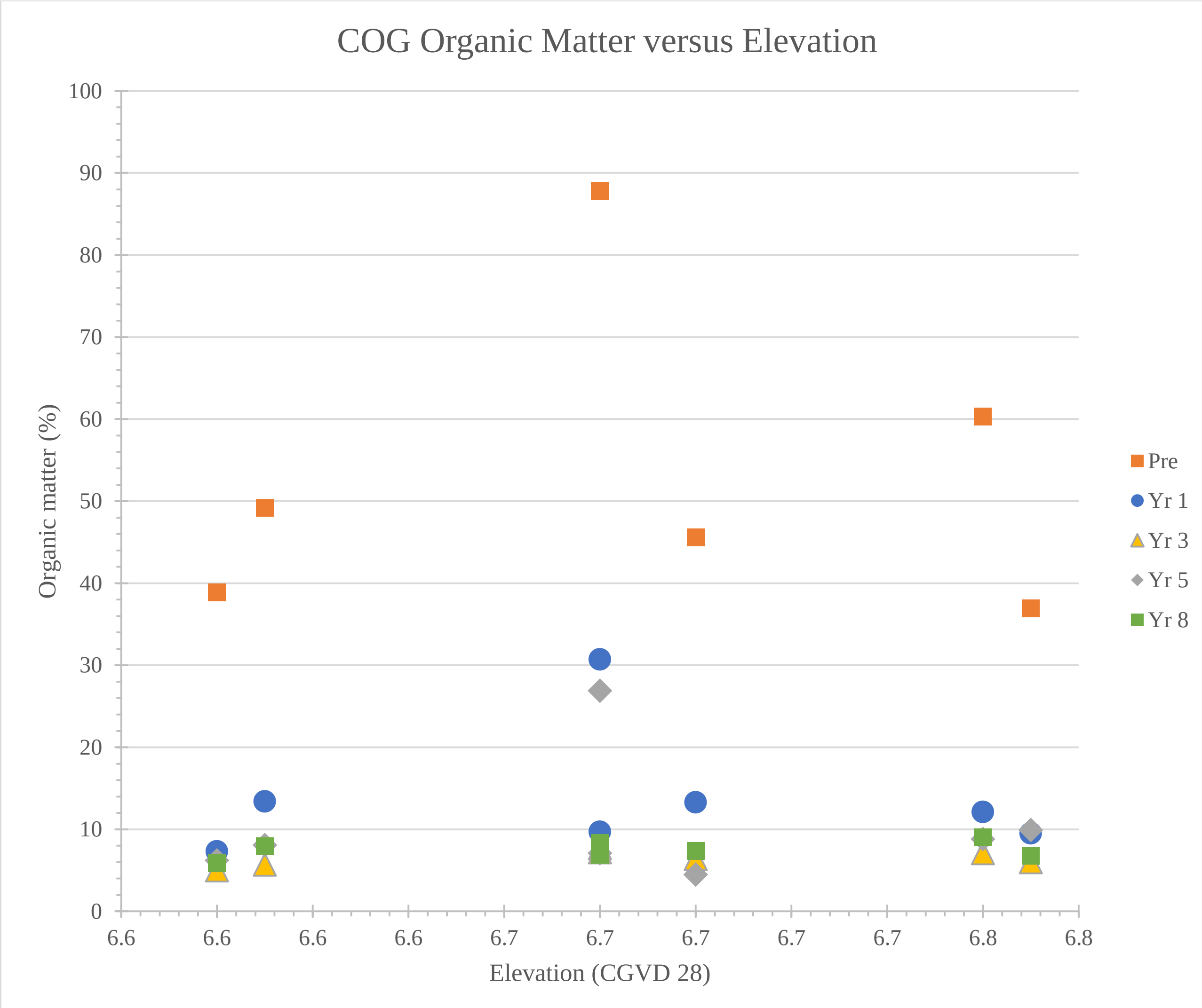

Figure 16. Organic matter (%) versus elevation (CGVD 28) at Cogmagun restoration. Upland site (7.5 m CGVD 28) removed to show relationship.

When comparing the 2018 organic matter data at the restoration site to the last available data from the reference site there are some notable differences. At the restoration site, organic matter ranges from 6% to 46% while the reference site ranges from 7% to 18%. Similarly to the water content analysis, the sampling site, L1S1, is an outlier at the restoration site. The average organic matter at the restoration site is lower at 7% when excluding the outlier site versus 12% at the reference site.

Within-site trends for bulk density at the Cogmagun restoration site show the highest value at L1S4 (6.6 m CGVD 28) at 1.14 g·cm³ and the lowest value at L1S1 (7.5 m CGVD 28) measuring 0.34 g·cm³ (Figure 16). This relationship is continuous amongst other elevations where bulk density decreases with increasing elevation (Figure 17). Due to this relationship, bulk density shows an inverse relationship to water content and organic matter, where there is an increasing trend from pre-restoration to nine years post restoration. Similarly to water content and organic matter, L1S1 is skewed due to it being an upland site at the higher elevation.

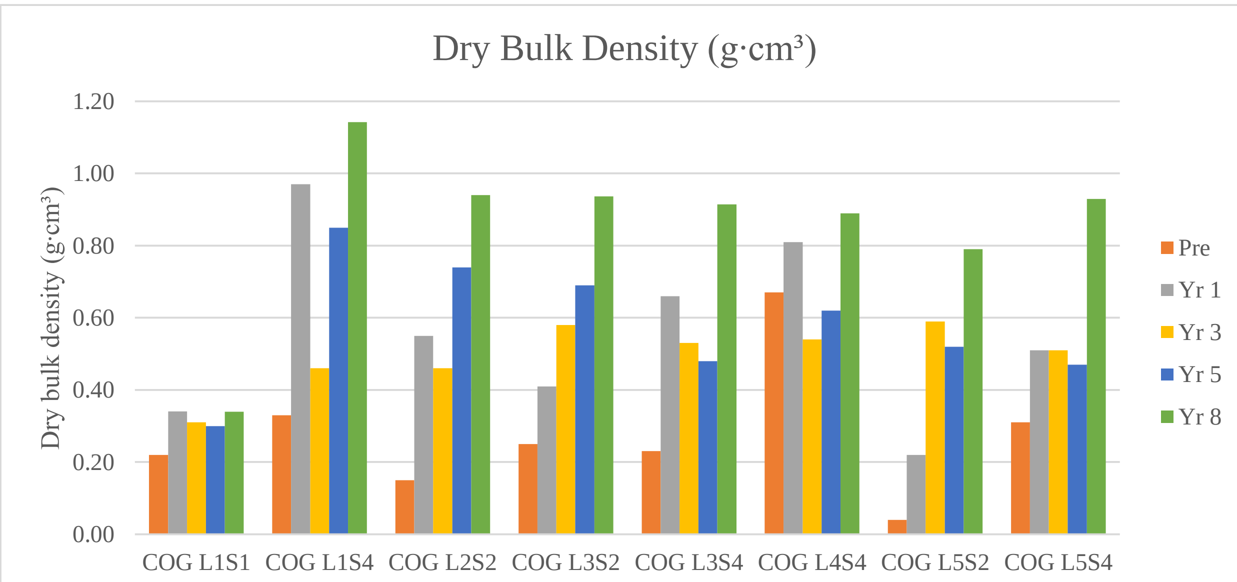

Figure 17. Bulk density (g·cm³) measured at Cogmagun restoration for each sampling site from pre-restoration to nine-years post restoration.

When looking at bulk density in the restoration site from pre-restoration to current values measured in 2018, the values measured in 2018 are higher at each site than in previous years. The lowest bulk density measurement was recorded pre-restoration (0.04 g·cm³), whereas the highest bulk density measurement was taken nine-years post restoration (1.14 g·cm³) (Figure 16). In addition, bulk density shows an increasing trend from pre-restoration to 2018 (Figure 17).

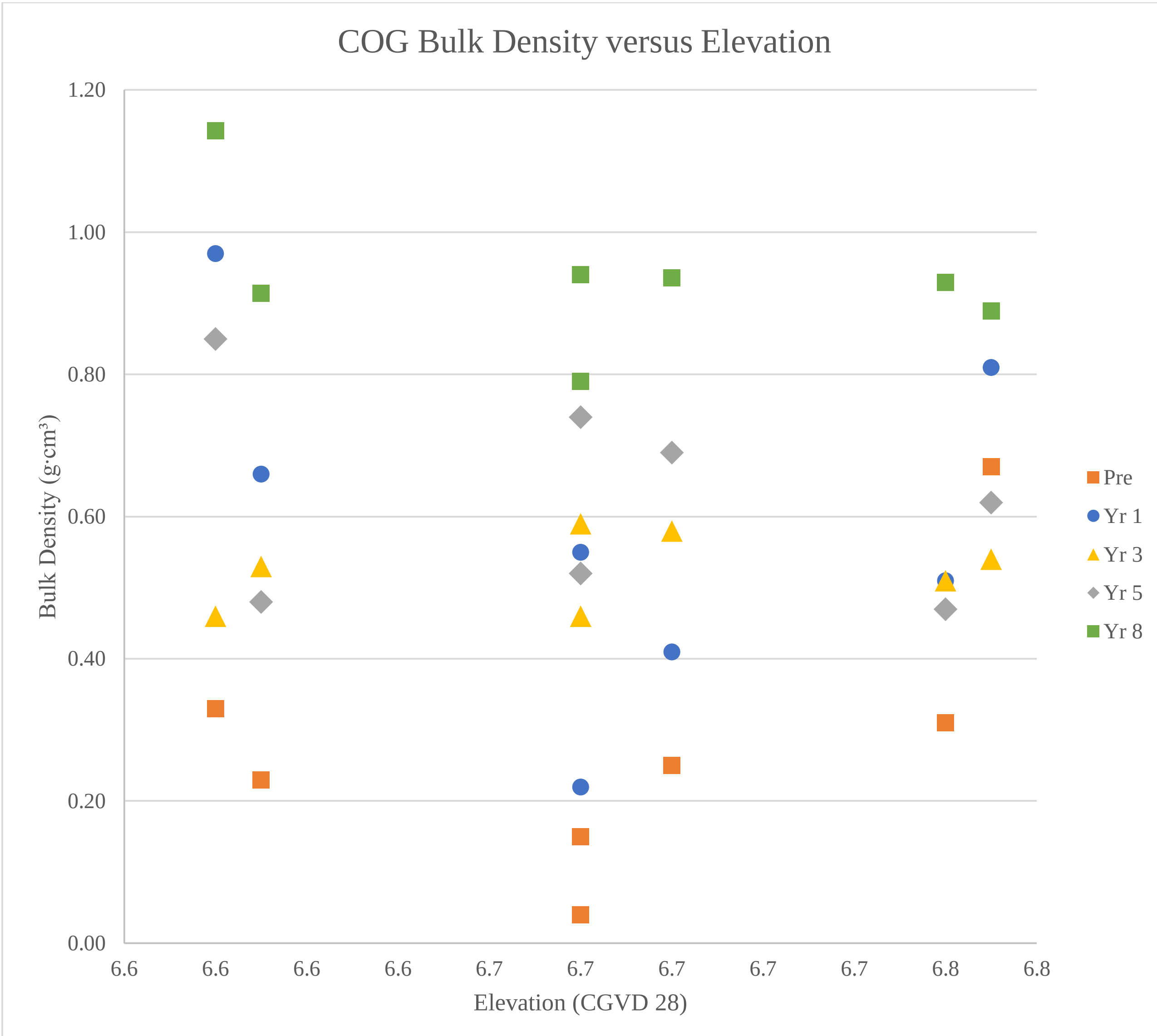

Figure 18.. Bulk density (g·cm³) versus elevation (CGVD 28) at Cogmagun restoration. Upland site (7.5 m CGVD 28) removed to show relationship.

4.3.2 Sediments

Measured RSETs show a net positive mean annual change in surface elevation when averaging the change from 2014 – 2018 over the four-year duration (Figure 18). The Cogmagun restoration site shows a mean annual change of 0.95 cm/yr, 0.47 cm/yr and 0.37/yr cm for RSET 1, 2 and 3, respectively (Figure 18). The Cogmagun reference site shows a mean annual change of 0.42 cm/yr, 0.51 cm/yr and 0.41 cm/yr for RSET 1, 2 and 3, respectively (Figure 18).

Marker horizons show net accretion that has occurred in each yearly successive period as well as the total accretion at each RSET. These measurements are taken at both the Cogmagun restoration site and Cogmagun reference site. At the Cogmagun reference site RSET 1 and RSET 2 show similar total accretion (9.15 cm and 8.36 cm, respectively) (Table 4). RSET 3 shows a lower total accretion at 5.68 cm (Table 4). At the Cogmagun reference site, RSET 1 and RSET 2 show more similar values (5.18 cm and 5.40 cm, respectively) compared to RSET 3 which has a higher total accretion at 7.55 cm (Table 5). When comparing the 2014 – 2018 data that has been averaged to represent a yearly value, the value remains lower than previous years at both the Cogmagun restoration and reference site (Table 4, 5).

Combining the total accretion with the changes in surface elevation gives measurements for the subsurface processes. All RSETs at both sites except RSET 3 at the reference site were calculated giving negative values (Table 6). When combining the subsurface processes with sediment accretion we get the net change in surface elevation. All values calculated for the net change in surface elevation are positive values (Table 6). Each of these values are between 5 – 6 cm apart from RSET 3 at the Cogmagun restoration site which only measures 2.92 cm (Table 6).

Table 1. Marker horizon data showing net accretion at the Cogmagun restoration site from 2009 - 2018. Asterisk (*) indicates that the values have been averaged over the four-year span from 2014 – 2018.

Cogmagun - MH measurements 2014-18				**net accretion from the previous year (cm/yr)**					
RSET-01 HM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**	**2013-14**	**2014-18***
core 1a	9.05	2	Good	0.00					
core 1b	8.60	1	Good	0.00					
core 1c	9.80	1	Good	0.00					
mean	**9.15**			**0.00**	**1.13**	**1.17**	**1.38**	**1.59**	**0.97**
RSET-02 - MM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**	**2013-14**	**2014-18**
core 2a		3	N/A	1.19					
core 2b	7.85	1	Fair	1.28					
core 2c	8.88	1	Fair	1.81					
mean	**8.36**			**1.43**	**1.31**	**1.14**	**1.43**	**0.50**	**0.64**
RSET -03 LM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**	**2013-14**	**2014-18**
core 3a	5.68	1	Good	0.61					
core 3b			N/A	0.86					
core 3c		3	N/A	0.89					
mean	**5.68**		0	**0.79**	**1.75**	**0.41**	**2.28**	**0.24**	**0.05**

Table 2. Marker horizon data showing net accretion at the Cogmagun reference site from 2009 - 2018. Asterisk (*) indicates that the values have been averaged over the four-year span from 2014 – 2018.

Cogmagun Reference - MH measurements 2014-18				**net accretion from previous year (cm/yr)**					
RSET-01 LM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**	**2013-14**	**2014-18***
core 1a	4.95	1	Good	0.99					
core 1b	5.50	1	Good	0.61					
core 1c	5.08	1	Good	0.63					
mean	**5.18**			**0.74**	**0.78**	**0.70**	**0.25**	**1.48**	**0.31**
RSET-02 - HM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**	**2013-14**	**2014-18**
core 2a	5.40	1	Good	0.70					
core 2b		3	N/A	1.06					
core 2c		3	N/A	1.35					
mean	**5.40**			**1.04**	**0.91**	**0.10**	**1.08**	**0.45**	**0.45**
RSET -03 MM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**	**2013-14**	**2014-18**
core 3a	7.35	1	Poor	0.99					
core 3b		2	N/A	1.25					
core 3c	7.75	1	N/A	0.93					
mean	**7.55**			**1.05**	**1.42**	**0.28**	**1.13**	**1.71**	**0.49**

Table 3. Net change in surface elevation from net sediment accretion and net changes in the subsurface processes at Cogmagun restoration and Cogmagun reference sites since measurements started in 2009 modified from Bowron et al. (2015).

Station	**Zone**	**Line**	**Elevation (m) (CGVD 28)**	**Net change in surface elevation (cm)**	**Net sediment accretion (cm)**	**Net change due to subsurface processes (cm)**
COG RSET 1	Low marsh	1	6.678	5.70	9.15	-3.45
COG RSET 2	Mid marsh	3	6.839	5.09	8.36	-3.27
COG RSET 3	High marsh	5	7.087	2.92	5.68	-2.76
COG-R RSET 1	Low marsh	2	7.181	5.05	5.18	-0.13
COG-R RSET 2	High marsh	4	7.197	5.51	5.40	0.11
COG-R RSET 3	Mid marsh	5	7.087	5.77	7.55	-1.78

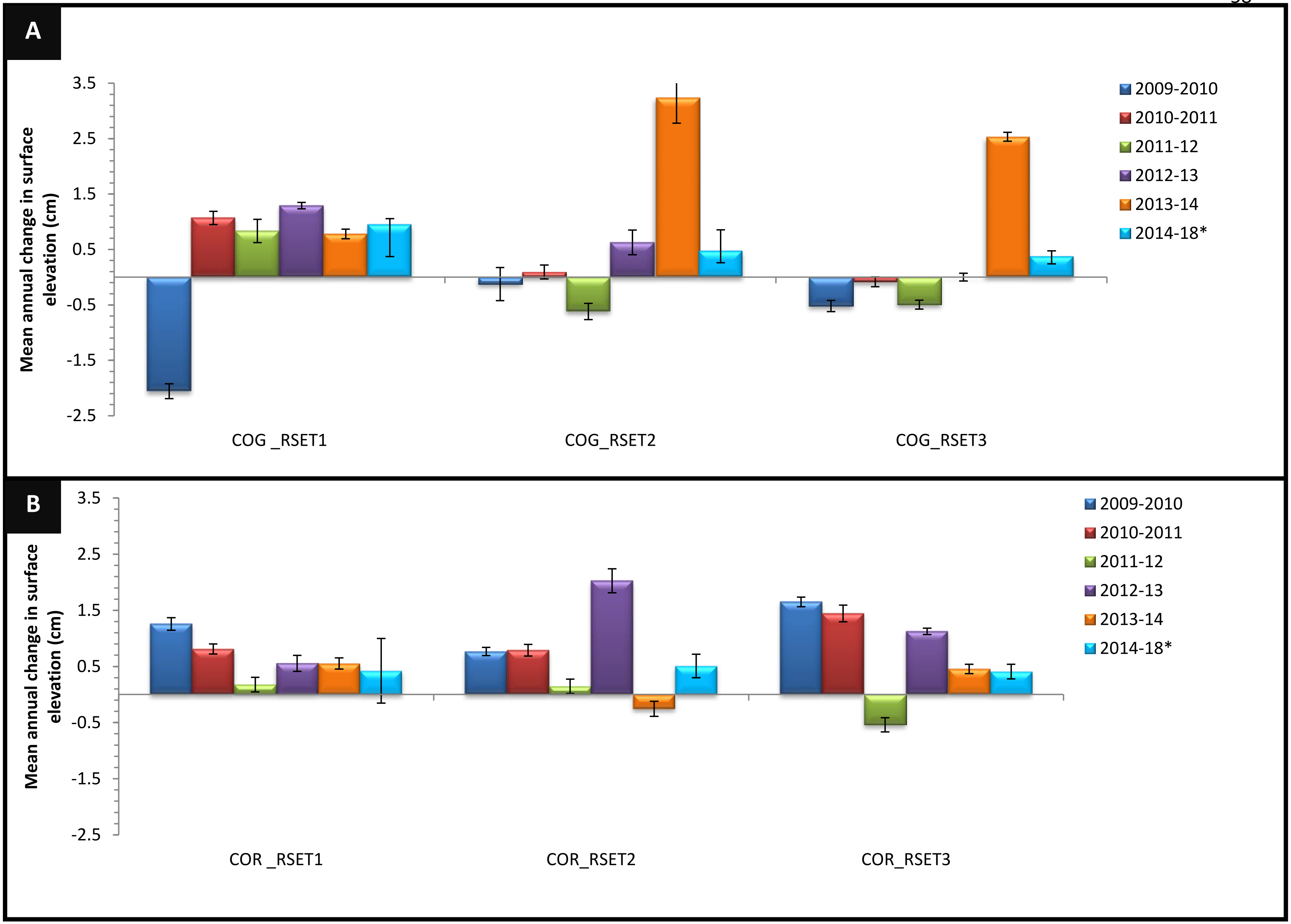

Figure 19. Mean annual change in elevation at a) Cogmagun restoration and b) Cogmagun reference site with error bars showing standard error. Asterisk (*) indicates that data is being averaged to account for yearly measurements.

CHAPTER 5: Discussion

Nine years after the re-introduction of tidal flow, the Cogmagun salt marsh is displaying environmental characteristics that are similar to the reference site and are continuing on a positive trajectory from the last measurements taken in 2014.

Pre-restoration there were many freshwater and upland species present on the site (e.g. *Carex stipata* and *Impatiens capensis*) that have not been recorded since early after the restoration. These species have been largely replaced with halophytes, however the ones inland from the first sampling site in each transect (e.g. L#S1) still contain non-halophytes and upland species. These plots at a higher elevation than the other plots and are only inundated by tidal flow during tides that are high (>6.58 m).

Particularly since 2014, Cogmagun restoration site has moved from low marsh and early successional colonizer species to a greater abundance of high marsh species. This is noted with a decrease in the relative abundance of *Salicornia europaea* and *Suaeda maritima* and increasing *Spartina patens* and *Distichlis Spicata*. The increase of cord grasses (*S. alterniflora and S. patens*) is characteristic in tidal wetland restoration as noted by Roman et al. (2002); Bowron et al. (2013) and Byers and Chmura (2007). These new species were also characteristic following the description of mature Southern New England salt marshes (Roman et al., 1984). They described the marshes they were looking at as consisting of *S. alterniflora* in the low marsh and *S. patens*, *D. spicata* and patches of *J. gerardii* in the high marsh. When looking at the habitat map, it can be noted that the restoration site is exhibiting this transition of species and zonation typical of these mature tidal wetland systems.

When comparing the current situation in the restoration site with the last available data at the reference site, the species richness and halophytic species richness are not statistically different. However, there is still a difference in abundance of halophytes between the restoration site and the reference site. A reduced abundance in the restoration site versus the reference site can be attributed to the location of the sampling plots. In the restoration site, the first sampling plots of each line are at higher elevations and dominated by upland species, whereas the first sampling plot at the reference site is located on the marsh. This would contribute to a higher abundance of upland species and a lower abundance of halophytic species at the restoration site.

It is also important to note that there were several species recorded in 2014 that were present but not recorded in 2018 due to being outside the measured plots or noted earlier in the season (May, 2018), and died before data collection in August, 2018. Species that were noted outside of the data collecting time include: *Carex Paleacea*, *Hierochloe odorata* and a greater abundance of *Juncus gerardii*. Species that were noted during the collection period but outside of the collection plots include: *Limonium carolinianum*, *Solidago sempervirens* and *Scirpus maritimus*. Many of these species such as *L. carolinianum* were also recorded by Roman et al. (1987) when looking at mature salt marshes.

The highest tide over the restoration site was measured at 7.13 metres (CGVD 2013), which is comparable to the highest tides measured in previous years (Bowron et al., 2014). This indicates that there are not restrictions or barriers to tidal flow over the study area. Tidal inundation acts as one main control of both soil and vegetation characteristics (Bricker-Urso and Nixon, 1989). The vegetation zonation is related to the hydroperiod and inundation frequency evident by the relationship between elevation, vegetation and hydroperiod/inundation. The low marsh zones contain the lowest elevations (5.95 - 6.09 m CGVD 2013) and have a longer hydroperiod and greater inundation frequency. This enables the growth of low marsh species that are tolerant of higher salinity levels and can remain inundated with tidal waters for longer periods. The high marsh is inundated less frequently, and spends less time inundated. This explains why there is a greater diversity of vegetation as more species can survive the longer hydroperiod and greater inundation frequency. The upland edge for each sampling plot with the exception of L3S1, contains the lowest hydroperiod and inundation frequency and has few halophytic species and many species that survive in brackish or fresh water conditions (e.g. *Typha angustifolia*).

When looking at soils characteristics from 2009 – 2018, there has been a reduction in water content along most sampling plots, with the exception of the upland outlier that was measured at an average elevation of 7.5 metres (CGVD 28). This inverse relationship can likely be explained because those points are along the upland edge, which are infrequently inundated. A similar trend was noted for the organic content, where the outliers are at an average elevation of 7.5 m (CGVD 28) and follow an inverse relationship to the other data collected. Bulk density does not demonstrate this relationship but has increased with time from 2009 – 2018. These values are still considerably different from the reference site being higher at the restoration site but with a greater range.

The combination of RSETs and marker horizons show the proportion of subsurface processes and sediment accretion that contribute to the net change in surface elevation. With all sampling locations, except RSET 3 at the reference site, experiencing negative values it shows that there is compaction occurring at most sites. The compaction is also being reflected from the soil characteristics, with the presence of anoxic layers and the increased bulk density values noted in 2018. RSET 3 at the reference site measured as a positive value indicating that roots or organic matter is contributing to growth of the marsh surface in addition to sediment deposition, as identified by Davidson-Arnott (2006). Although compaction is occurring, the mean annual change in surface elevation produced positive values across all RSETs indicating that sediment accretion is greater than compaction occurring below-ground. The surface is growing vertically and not threatened by subsidence due to the ability to keep pace with moderate sea level rise projections (Zhai et al., 2013; James et al., 2014). This difference can mostly be attributed to RSET 3, where the net accretion was only 2.92 cm/yr at the restoration site versus 5.77 cm/yr at the reference site. This trend was also noted in the study by Reed et al. (1999). They concluded that RSETs located further away from the tidal channel experienced reduced sedimentation because the tidal flow has to travel farther to reach these sites. They were inundated less frequently, leading to less sediment deposition.

When looking at the ecological indicators outlined in the modified GPAC protocols, the trajectory of the Cogmagun restoration site can be analysed. The surface elevation has continued to increase, and proper drainage has allowed the halophytic vegetation to colonize and demonstrate the zonation patterns characteristic of mature marshes. The measured variables are representative of the geomorphic evolution of the restoration as the site has progressed from a drowning impoundment to a functioning salt marsh ecosystem, capable of sustaining flora and fauna.

CONCLUSIONS

Salt marshes have shown to act as a natural barrier that provides protection to the coast from threats facing Nova Scotia as a result of sea level rise. Salt marsh restoration of degraded or former salt marsh ecosystems increases protection. In the case of the Cogmagun River salt marsh, the environmental characteristics of the restored marsh were found to be similar to the reference site and so are continuing on a positive trajectory from the last measurements taken in 2014. The vegetation and zonation patterns are now typical of mature salt marsh ecosystems. Sediment accretion is continuing to contribute to the vertical growth of the marsh platform. This vertical growth is a key factor as it enables salt marsh ecosystems to keep pace with sea level rise. The soil characteristics of the restored marsh indicate that anoxic conditions and bulk densities are such that they are unlikely to diminish vegetation growth and contribute to compaction through decay. Finally, the sediment accretion is high enough to contribute to a positive annual change in surface elevation. Long-term monitoring of salt marsh restoration projects such as that at the Cogmagun River marsh, should be continued as it is an important indicator as to whether the marsh is progressing towards natural conditions or if further adaptive management or active intervention is required.

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APPENDICES

Appendix A. Vegetation

Table A. 1. Two-Sample T-Test and CI: COR Species Richness, COG Species Richness.

μ_1: mean of COR Species Richness

μ_2: mean of COG Species Richness

Difference: $\mu_1 - \mu_2$

Equal variances are not assumed for this analysis.

Descriptive Statistics

Sample	N	Mean	St Dev	SE Mean
COR Species Richness	23	2.74	1.63	0.34
COG Species Richness	24	2.375	0.924	0.19

Estimation for Difference

Difference	95% CI for Difference
0.364	(-0.426, 1.154)

Test

Null hypothesis	H_0: $\mu_1 - \mu_2 = 0$
Alternative hypothesis	H_1: $\mu_1 - \mu_2 \neq 0$

T-Value	DF	P-Value
0.94	34	0.355

Table A. 2. Two-Sample T-Test and CI: COR Halophytic Species Richness, COG Halophytic Species Richness.

μ_1: mean of COR Halophytic Spp. Richness

μ_2: mean of COG Halophytic Spp. Richness

Difference: $\mu_1 - \mu_2$

Equal variances are not assumed for this analysis.

Descriptive Statistics

Sample	N	Mean	StDev	SE Mean
COR Halophytic Spp. Richness	23	2.52	1.24	0.26

COG Halophytic Spp. Richness	24	2.000	0.780	0.16

Estimation for Difference

Difference	95% CI for Difference
0.522	(-0.094, 1.137)

Test

Null hypothesis H_0: $\mu_1 - \mu_2 = 0$

Alternative hypothesis H_1: $\mu_1 - \mu_2 \neq 0$

T-Value	DF	P-Value
1.72	36	0.094

Table A. 3. Two-Sample T-Test and CI: COR Halophytic Species Abundance, COG Halophytic Species Abundance.

μ_1: mean of COR Halophytic Spp. Abundance

μ_2: mean of COG Halophytic Spp. Abundance

Difference: $\mu_1 - \mu_2$

Equal variances are not assumed for this analysis.

Descriptive Statistics

Sample	N	Mean	StDev	SE Mean
COR Halophytic Spp. Abundance	23	37.8	13.1	2.7
COG Halophytic Spp. Abundance	24	29.2	13.8	2.8

Estimation for Difference

Difference	95% CI for Difference
8.57	(0.65, 16.50)

Test

Null hypothesis H_0: $\mu_1 - \mu_2 = 0$

Alternative hypothesis H_1: $\mu_1 - \mu_2 \neq 0$

T-Value	DF	P-Value
2.18	44	0.035

Table A 4. Species abundance from 2009 – 2018 at the Cogmagun restoration site with comparison data from the Cogmagun reference site for 2014.

Species names	Abbreviation	Cog 2009 means	Cog 2009 freq.	Cog 2010 means	Cog 2010 freq.	Cog 2011 means	Cog 2011 freq.	Cog 2012 means	Cog 2012 freq.	Cog 2013 means	Cog 2013 freq.	Cog 2014 means	Cog 2014 freq.	Reference 2014 means	Reference 2014 freq.	Cog 2018 means	Cog 2018 freq.
Agrostis stolonifera	Agr.sto	1.62	3	1.42	3	1.5	3	2	3	0.65	2	0.22	3	0	0	0.87	1
Algae	Algae	1.08	3	0.05	2	1.76	4	0	0	0	0	0	0	0	0	0	0
Aster spp.	Ast.sp	0	0	0	0	0.14	2	0	0	0	0	0	0	0	0	0.13	1
Atriplex glabrisculata	Atr.gla	0.14	2	0.97	4	0.65	6	0.32	1	0	0	0	0	1.42	6	0	0
Betula sp.	Bet.sp	0.05	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Calamagrostis canadensis	Cal.can	0.02	2	1.21	3	0.04	1	0.5	2	0.3	1	0	0	0	0	0	0
Carex gynandra	Car.gyn	0	0	0	0	0.08	1	0	0	0	0	0	0	0	0	0	0
Carex paleacea	Car.pal	0	0	0	0	0	0	0.01	1	0.17	1	0.67	1	3.61	5	0	0
Carex pseudo-cyperus	Car.pse	0	0	0.17	1	0	0	0	0	0	0	0	0	0	0	0	0
Carex stipata	Car.sti	1.29	3	0.75	3	0	0	0	0	0	0	0	0	0	0	0	0
Taraxacum offinialis	Tar.off	0	0	0	0	0.38	1	0.32	1	0	0	0	0	0	0	0	0
Distichlis spicata	Dis.spi	0	0	0	0	0	0	0.91	1	0.57	1	4.17	7	2.17	4	5.6	7
Elymus repens	Ely.rep	0	0	0.21	1	0	0	0.09	1	0.23	2	0	0	0.61	1	0	0
Equisetum sp.	Equ.sp	0.38	2	0	0	0.01	1	0	0	0	0	0	0	0	0	0	0
Festuca rubra	Fes.rub	0.04	1	0	0	0	0	0	0	1.57	3	0	0	0	0	0	0
Gallium palustre	Gal.pal	0.93	4	0	0	0.01	1	0	0	0	0	0	0	0	0	0	0
Hierochloe odorata	Hie.odo	0.22	3	0.04	1	1.92	4	0.64	2	0.7	2	0.96	1	0	0	0	0

Hordeum jubatum	Hor.jub	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Impatiens capensis	Imp.cap	1.59	4	0	0	0.01	1	0	0	0	0	0	0	0	0	0	0
Iris versicolor	Iri.ver	0	0	0	0	0	0	0.05	1	0	0	0	0	0	0	0	0
Juncus effusus	Jun.eff	0.25	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juncus gerardii	Jun.ger	0.04	1	0.38	2	2.17	3	1.05	2	1.04	2	1.17	2	13.65	13	1.47	1
Limonium carolinianum	Lim.car	0	0	0	0	0.03	3	0.01	1	0.11	4	0.14	4	0.13	1	0	0
Lycopus americanus	Lyc.ame	0.05	2	0.01	1	0	0	0	0	0	0	0	0	0	0	0	0
Lycopus uniflorus	Lyc.uni	0.21	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lysimachia terrestris	Lys.ter	0.17	1	0.21	1	0	0	0	0	0	0	0	0	0	0	0	0
Myosotis laxa	Myo.lax	0.21	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Onoclea sensibilis	Ono.sen	0.38	2	0.04	1	0.01	1	0	0	0	0	0	0	0	0	0	0
Oxalis sp.	Oxa.sp	1.04	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polygonum sagittatum	Per.sag	0.19	3	0	0	0.04	1	0	0	0	0	0	0	0	0	0	0
Phragmites australis	Phr.aus	0	0	0	0	0	0	0	0	0	0	0.62	1	0	0	0.67	1
Poa palustris	Poa.pal	0.38	1	1.12	2	0	0	0.82	1	0	0	0.25	1	0	0	0	0
Potentilla anserina	Pot.ans	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ranunculus cymbalaria	Ran.cym	0.79	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ranunculus repens	Ran.rep	0.5	1	0.05	2	0.12	1	0	0	0	0	0	0	0	0	0	0
Rorippa palustris	Ror.pal	0	0	0	0	0.08	1	0	0	0	0	0	0	0	0	0	0
Ruppia maritima	Rup.mar	0.08	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Salicornia depressa	Sal.dep	0	0	0.04	1	1.23	15	1.56	10	7.66	17	3.54	6	0.64	4	0.07	1
Scirpus validus	Sci.val	3.17	7	0.17	2	0	0	0	0	0	0	0	0	0	0	0	0

Scirpus cyperinus	Sci.cyp	0.62	1	0.18	3	0	0	0	0	0	0	0	0	0	0	0	0
Scirpus maritimus	Sci.mar	2.21	4	0	0	2.38	8	1	1	0.61	3	0.25	1	0	0	0	0
Scutellaria galericulata	Scu.gal	0.5	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Solidago rugosa	Sol.rug	0.12	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Solidago sempervirens	Sol.sem	0	0	0	0	0	0	0	0	0	0	0.01	1	1.3	3	0	0
Spartina alterniflora	Spa.alt	0	0	0.39	3	2.94	9	9.43	17	16.36	20	19.17	19	7	10	16.67	17
Spartina patens	Spa.pat	0	0	0	0	0.8	3	2.89	9	6.43	10	8.21	9	6.04	8	21.27	14
Spartina pectinata	Spa.pec	0.38	1	1.04	1	1	2	2.09	3	0.57	1	0.3	2	3.13	3	1.6	3
Sueda maritima	Sue.mar	0	0	0.26	4	4.4	16	2.52	8	2.97	13	1.35	6	0.15	4	0.07	1
Symphotrichum lateriflorus	Sym.lat	0.08	1	0.04	1	0	0	0	0	0	0	0	0	0	0	0	0
Symphotrichum novi-belgii	Sym.nov	0	0	0.1	3	0.25	1	0	0	0.01	1	0.62	1	0	0	0	0
Thelypteris palustris	The.pal	0.01	1	0.08	1	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin maritima	Tri.mar	0	0	0	0	0	0	0	0	0	0	0.04	1	0.04	1	0	0
Typha angustifolia	Typ.Ang	7.75	11	0	0	2.38	6	1.42	3	2.22	3	3.5	4	0	0	4.53	5
Typha latifolia	Typ.lat	1.62	3	2.08	10	1.27	7	0.95	1	0.78	1	0.62	1	0	0	0	0

Appendix B. Soils

Table B. 1. Characterization of soil cores collected at Cogmagun restoration in 2018.

Cog Station	Length (cm)	Description
L1S1	10.8	• Dark brown. Core comprised on mud and fine root material. Green root near bottom of core. Fine grained sediment in core.
L1S4	9.9	• Medium brown. Black streaks throughout. Root matter throughout core, with increased amounts near bottom of core. Prominent fish smell.
L2S2	9.6	• Medium brown. Black patches throughout and heavily concentrated near bottom. A lot of root matter near bottom of core. Strong fish smell.
L3S2	10.2	• Medium brown. Black layer near bottom, with medium brown being prominent near the top of the core. Root matter interspersed throughout. Strong fish smell.
L3S4	10.1	• Darker near top and bottom of core. Medium brown throughout middle of core. Coarse black layer on top, black band from 5 – 8 cm. Dense root matter near bottom. Strong fish smell.
L4S4	9.9	• Medium brown. Root matter throughout. Grass near top and bottom of core. Mild fish smell.
L5S2	10.4	• Medium brown with two bands of streaky black material occurring from 0 – 2 cm and 6 – 9 cm. Green grass and a singular large root in the core.
L5S4	11.0	• Medium brown with a darker brown/black layer from 0 – 2 cm. Coarser material from 5 – 7 cm with high percentage of vegetative and root material.

Table B. 2. Sediment characteristics of soil cores pre, 1, 3, 5- and 8-year post restoration from COG and pre, 1, 3- and 5-years post restoration for COG-R.

Station a)	**Elev (m)**	**Water Content (%)**					**Organic Matter (%)**					**Dry bulk density (g·cm³)**				
a) COG	**Elev (m)**	**Pre**	**Yr 1**	**Yr 3**	**Yr 5**	**Yr 8**	**Pre**	**Yr 1**	**Yr 3**	**Yr 5**	**Yr 8**	**Pre**	**Yr 1**	**Yr 3**	**Yr 5**	**Yr 8**
COG L1S1	7.5	63.1	73.2	76.4	69.1	78.2	-4.2	36.4	43.7	29.1	46.2	0.22	0.34	0.31	0.30	0.34
COG L1S4	6.6	81.7	31.8	29.4	39.2	36.0	38.9	7.3	5.0	6.2	5.9	0.33	0.97	0.46	0.85	1.14
COG L2S2	6.7	76.3	50.0	49.2	45.9	43.1	87.8	9.7	7.7	7.1	6.9	0.15	0.55	0.46	0.74	0.94
COG L3S2	6.7	85.2	63.0	39.7	39.4	47.9	45.6	13.3	6.4	4.5	7.4	0.25	0.41	0.58	0.69	0.94
COG L3S4	6.6	86.6	39.5	37.2	45.8	49.6	49.2	13.4	5.7	8.1	7.9	0.23	0.66	0.53	0.48	0.91
COG L4S4	6.8	66.6	38.3	38.7	50.1	39.5	36.9	9.5	6.0	9.9	6.8	0.67	0.81	0.54	0.62	0.89
COG L5S2	6.7	93.0	76.8	41.0	77.5	48.7	122.3	30.7	7.2	26.9	8.3	0.04	0.22	0.59	0.52	0.79
COG L5S4	6.8	83.0	52.1	38.8	47.9	45.0	60.3	12.1	7.1	8.8	9.0	0.31	0.51	0.51	0.47	0.93
b) COG_R																
COG_R L1S4	7.1	43.8	32.8	24.5	36.6	NA	12.8	8.9	5.5	8.7	NA	0.78	1.02	0.77	0.76	NA
COG_R L2S1	7.1	72.0	71.0	60.4	60.0	NA	35.4	32.8	15.4	16.6	NA	0.41	0.22	0.23	0.38	NA
COG_R L2S3	7.0	55.4	54.4	52.6	58.0	NA	13.5	11.3	9.2	11.4	NA	0.61	0.71	0.42	0.39	NA
COG_R L2S5	7.1	30.9	32.6	NA	32.7	NA	9.8	2.1	NA	7.2	NA	0.91	1.14	NA	0.86	NA
COG_R L3S2	7.1	15.9	58.7	59.4	62.0	NA	NA	16.4	9.9	13.4	NA	0.33	0.49	0.31	0.32	NA
COG_R L4S1	7.1	70.0	56.8	56.5	64.9	NA	25.2	16.2	11.6	17.9	NA	0.31	0.34	0.24	0.31	NA
COG_R L4S3	7.2	45.9	46.5	NA	53.0	NA	14.1	11.5	NA	10.9	NA	0.65	0.85	NA	0.57	NA
COG_R L5S2	7.1	5.6	49.8	50.2	56.1	NA	98.9	11.3	9.2	11.2	NA	0.61	0.69	0.47	0.46	NA

Appendix C. Sediments

Table C. 1. Raw RSET data from 2009 – 2018 for Cogmagun restoration site.

Cogmagun Restoration				**Net change in elevation between sampling period (cm)**								
RSET-01 high marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	28	-3.9	-2.8	-2.0	-1.5	-1.5	-2.0	-1.7	-0.6	-3.1
mean (cm)	-2.1	3	118	-1.7	-1.6	-1.8	-2.1	-1.8	-1.9	-1.0	-0.9	-0.7
stdev	0.8	5	208	-4.2	-2.0	-2.1	-2.9	-2.5	-3.2	-3.5	-2.9	-2.2
SE	0.1	7	298	-1.7	-1.9	-1.9	-1.7	-1.7	-1.8	-1.9	-1.9	-1.5
2010-2011		1	28	0.6	0.5	0.2	-0.1	-0.2	0.1	-0.3	-0.3	0.5
mean (cm)	1.1	3	118	1.0	1.1	1.0	1.4	1.3	1.4	1.3	1.4	1.3
stdev	0.7	5	208	1.0	1.1	1.2	1.1	1.0	1.1	0.9	0.3	1.0
SE	0.1	7	298	2.1	2.1	2.1	2.2	2.0	2.0	1.8	1.7	1.6
2011-2012		1	28	1.0	0.8	0.9	1.0	0.7	0.5	0.7	0.5	1.0
mean (cm)	0.8	3	118	0.6	0.6	1.1	0.6	0.8	0.8	0.7	0.8	0.9
stdev	0.2	5	208	1.0	1.0	1.0	1.2	1.0	0.8	1.0	1.2	1.1
SE	0.0	7	298	0.6	0.8	0.7	0.8	0.8	0.5	0.5	1.2	0.8
2012-2013		1	28	1.5	0.9	0.7	0.7	0.8	1.3	0.9	0.6	0.9
mean (cm)	1.3	3	118	1.4	1.1	1.0	1.3	1.7	1.9	1.6	1.4	2.0
stdev	0.4	5	208	1.2	0.8	1.4	1.2	1.2	1.5	1.4	1.1	1.4
SE	0.1	7	298	1.2	1.8	1.8	1.5	1.5	1.5	1.3	1.3	1.7
2013-14		1	28	1.0	1.1	1.2	1.1	0.4	0.4	0.3	0.8	0.4
mean (cm)	0.8	3	118	1.2	1.4	1.0	1.2	0.6	0.7	0.8	0.9	0.2
stdev	0.5	5	208	1.5	1.8	0.3	0.2	1.0	0.6	2.5	0.9	0.5
SE	0.1	7	298	0.8	-0.2	0.5	0.5	0.3	0.7	1.1	0.4	0.0
2014-18		1	20	2.6	4.0	3.7	3.3	4.2	3.7	4.2	2.4	3.6
mean (cm)	3.8	3	112	3.3	3.7	3.7	4.0	4.0	3.7	4.3	3.8	3.8
stdev	0.7	5	202	3.2	3.1	4.1	3.9	4.0	4.4	1.9	3.4	4.0
SE	0.1	7	300	3.5	4.6	4.0	5.7	4.4	3.8	4.1	3.8	4.4
RSET-02 mid marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9

2009-2010		1	230	-0.6	-0.7	-3.3	0.2	-0.1	-1.1	-0.1	0.1	-0.2
mean (cm)	-0.1	3	320	0.9	-0.6	0	0.4	0.2	-0.6	-0.9	-2.2	-1
stdev	1.8	5	50	0.1	0.9	1.1	5.5	1.9	2.8	1.3	0.6	0.3
SE	0.3	7	140	0.5	1.7	1.2	-1	-1.1	-2.7	-2.3	-5.2	-0.5
2010-2011		1	230	-0.3	0.2	0.4	-0.5	-1.5	0.0	0.5	0.9	1.1
mean (cm)	0.1	3	320	0.5	0.3	0.0	-0.1	0.3	1.0	0.2	2.5	1.8
stdev	0.8	5	50	0.3	-0.8	-0.9	-1.1	0.2	0.2	-0.4	0.3	-0.1
SE	0.1	7	140	-0.7	-0.4	-0.1	0.5	0.1	-0.1	-0.4	-0.1	-0.4
2011-2012		1	230	0.2	-0.5	-0.5	-0.5	0.2	0.0	-0.8	-0.9	-0.7
mean (cm)	-0.6	3	320	-1.5	-0.8	-0.1	-0.3	-0.4	-0.4	-0.1	-0.1	-0.1
stdev	0.9	5	50	-0.5	-0.2	0.0	-0.3	-0.5	0.0	-0.5	-0.3	-0.3
SE	0.1	7	140	-0.2	-0.7	-0.1	-1.1	-4.7	-2.3	-1.5	-1.1	-0.7
2012-2013		1	230	0.7	1.4	1.3	0.7	0.3	1.0	2.3	2.1	1.0
mean (cm)	0.6	3	320	0.3	1.4	0.5	-1.1	-1.5	-1.2	-1.8	-1.2	0.3
stdev	1.3	5	50	2.3	1.2	1.0	1.2	1.4	0.0	0.1	-0.5	-1.9
SE	0.2	7	140	0.4	0.2	-0.6	-0.5	2.7	1.8	1.0	2.5	3.8
2013-14		1	230	8.5	8.1	7.6	6.4	6.4	4.9	3.7	3.8	4.9
mean (cm)	3.2	3	320	6.1	4.8	5.6	7.4	7.4	3.7	4.8	4.1	2.0
stdev	2.8	5	50	0.5	0.6	0.4	1.3	2.5	2.0	1.3	1.1	3.4
SE	0.5	7	140	0.4	0.6	1.2	1.1	0.7	0.4	0.3	0.0	-1.4
2014-18		1	210	-1.0	-2.7	-2.9	-1.7	-1.1	0.3	0.0	0.3	-0.2
mean (cm)	1.9	3	300	3.9	3.1	2.3	-1.1	0.8	4.1	3.0	1.9	3.3
stdev	2.3	5	30	0.8	1.2	0.7	1.4	0.3	2.7	3.5	4.5	2.0
SE	0.4	7	120	3.1	3.0	3.1	3.3	5.2	4.3	5.4	5.6	5.2

Cogmagun Restoration				**Net change in elevation between sampling period (cm)**								
RSET-03 Low marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	26	-0.5	0.1	-0.6	-1.1	-1.2	-0.6	-1.3	0.2	-0.4
mean (cm)	-0.5	3	116	-0.6	0.0	0.3	-1.1	-1.0	-0.3	0.8	0.0	0.5
stdev	0.6	5	206	-0.8	-1.3	-1.0	-1.1	-1.3	-0.8	-0.9	-0.4	-1.0
SE	0.1	7	296	0.3	-1.2	-1.2	-0.7	0.1	-0.5	0.4	0.3	-0.9
2010-2011		1	26	0.3	0.5	-0.3	0.1	0.9	-0.4	-0.4	0.5	0.3
mean (cm)	-0.1	3	116	-0.1	-0.1	0.1	0.1	0.1	0.1	-0.5	-0.7	-1.1
stdev	0.5	5	206		deer footprint		0.0	0.3	-0.1	0.5	0.0	-0.7

SE	0.1	7	296	-0.5	0.3	0.5	0.1	0.0	-0.1	-0.2	-0.4	-1.9
2011-2012		1	26	-0.2	-0.9	0.4	-0.5	-0.4	-0.2	-0.4	-0.6	-0.8
mean (cm)	-0.5	3	116	-1.0	-1.1	0.3	-0.3	0.0	-1.2	-0.8	-0.6	-0.6
stdev	0.5	5	206	0.0	0.0	0.0	-0.1	-0.5	0.0	-0.2	0.3	0.1
SE	0.1	7	296	-1.2	-0.7	-0.6	-0.5	-1.0	-1.5	-1.1	-0.5	0.0
2012-2013		1	26	-0.1	-0.2	-0.6	0.1	-0.1	0.1	0.6	0.5	0.1
mean (cm)	0.0	3	116	1.1	0.2	-1.1	-0.1	-0.6	0.2	0.4	0.4	0.3
stdev	0.4	5	206	0.3	0.1	0.5	-0.3	0.4	0.1	0.2	0.1	-0.2
SE	0.1	7	296	-0.3	-0.2	-0.7	-0.4	-0.2	0.0	-0.2	-0.6	0.2
2013-14		1	26	2.4	2.5	2.7	3.1	3.4	3.1	2.2	2.2	2.5
mean (cm)	2.5	3	116	2.3	3.4	3.5	3.0	2.9	2.2	1.8	2.0	1.9
stdev	0.5	5	206	3.0	2.9	2.7	3.0	2.3	2.0	2.4	2.2	2.4
SE	0.1	7	296	2.2	1.9	3.0	3.1	2.3	2.7	2.2	1.6	2.2
2014-18		1	10	0.6	1.4	0.9	1.5	0.5	3	3.2	1.3	0.8
mean (cm)	1.49	3	100	0.7	1.5	1.4	1.2	1.2	1.1	1.3	1.5	1.2
stdev	0.62	5	200	1.2	1.4	1.4	1.1	1.7	1.8	1.9	1.3	1.8
SE	0.10	7	290	2.3	2.6	1.6	1.1	1.2	2.4	1	2.2	1.2

Table C. 2. Raw RSET data from 2009 – 2018 for Cogmagun reference site.

Cogmagun Reference RSET-01 low marsh		Position	Bearing	**Net change in elevation between sampling period (cm)** Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	57	0.8	1.0	0.1	1.5	1.5	0.5	2.6	1.1	1.0
mean (cm)	1.3	3	147	1.0	0.9	0.9	0.8	1.1	1.3	1.5	1.0	0.6
stdev	0.7	5	237	0.6	0.9	2.8	2.1	0.8	1.0	0.8	0.8	1.3
SE	0.1	7	327	1.3	0.4	1.8	2.2	1.1	1.1	3.0	1.9	2.1
2010-2011		1	57	0.7	0.5	1.2	0.4	0.2	0.5	1.2	0.9	0.2
mean (cm)	0.8	3	147	1.0	1.4	1.1	1.5	0.9	0.6	0.2	1.4	2.1
stdev	0.5	5	237	0.9	1.7	-0.4	0.7	1.4	0.8	0.4	1.0	0.0
SE	0.1	7	327	0.6	0.7	0.9	1.7	0.5	0.3	0.9	0.3	0.8

2011-2012		1	57	0.2	0.3	0.0	0.1	-0.2	0.4	-0.8	0.5	0.7
mean (cm)	0.2	3	147	0.2	0.1	-0.2	-1.0	-0.4	0.0	0.5	0.0	-1.8
stdev	0.8	5	237	-0.4	-0.2	-0.2	-1.2	-0.4	0.6	1.2	0.6	0.6
SE	0.1	7	327	1.7	0.0	0.2	0.4	2.5	1.5	0.1	0.5	0.3
2012-2013		1	57	0.5	0.5	0.9	0.6	0.8	0.7	2.1	1.3	0.7
mean (cm)	0.6	3	147	0.0	0.0	1.4	2.0	1.7	0.4	0.7	0.8	2.1
stdev	0.8	5	237	0.3	0.5	0.6	1.7	0.2	-0.3	0.1	0.9	1.2
SE	0.1	7	327	0.1	0.6	0.2	0.5	-1.5	-1.8	-0.4	0.1	-0.2
2013-14		1	57	0.5	0.4	0.2	0.0	0.2	0.6	-0.2	-0.1	0.3
mean (cm)	0.6	3	147	0.3	0.3	0.1	-0.5	0.1	1.2	0.1	0.6	0.2
stdev	0.6	5	237	0.7	1.0	1.8	0.1	0.7	1.0	0.5	0.4	0.0
SE	0.1	7	327	0.6	0.7	0.7	0.9	0.8	2.8	1.0	0.5	1.4
2014-18		1	40	2.3	1.8	1.1	1.7	1.8	0.9	2.2	1.4	1.4
mean (cm)	1.7	3	130	2.6	2.4	2.8	3.9	2.8	2.7	2.8	2.2	2.0
stdev	3.5	5	230	2.7	3.5	3.2	5.3	-17.9	2.1	2.5	1.8	2.2
SE	0.6	7	320	1.5	1.7	2.2	1.9	2.2	2.3	1.5	1.5	1.8
RSET-02 high marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	344	0.9	1.2	0.5	1.0	0.3	0.6	1.1	1.6	0.6
mean (cm)	0.8	3	74	1.1	0.5	0.7	0.8	0.2	0.9	0.9	1.0	0.5
stdev	0.4	5	164	1.3	0.6	0.5	1.0	-0.3	1.2	0.6	0.9	1.1
SE	0.1	7	254	0.9	0.4	0.2	0.0	1.2	0.0	0.6	1.3	1.7
2010-2011		1	344	0.7	0.3	1.7	0.5	0.6	1.3	0.6	-0.2	0.3
mean (cm)	0.8	3	74	0.5	1.0	0.9	0.9	1.2	0.6	-1.2	1.0	1.2
stdev	0.6	5	164	0.9	1.3	-0.1	0.5	1.4	-0.2	0.7	0.7	0.7
SE	0.1	7	254	1.3	2.1	1.7	1.8	0.5	0.9	0.5	1.1	0.7
2011-2012		1	344	0.3	-0.9	-0.4	-0.2	0.0	-0.1	-0.4	0.6	0.1
mean (cm)	0.1	3	74	0.1	0.7	0.2	0.0	0.7	0.0	1.0	0.2	-0.7
stdev	0.8	5	164	1.2	1.3	1.6	1.4	-0.2	0.8	1.0	0.6	-0.6
SE	0.1	7	254	0.2	-0.2	-0.3	-2.6	-0.2	0.2	0.1	-0.2	0.0
2012-13		1	344	3.2	3.1	2.7	3.0	2.1	1.7	1.8	0.7	1.7
mean (cm)	2.0	3	74	3.2	2.2	3.5	3.5	2.3	3.5	5.2	3.8	4.3
stdev	1.3	5	164	0.0	0.3	1.0	0.6	1.4	0.7	0.7	1.3	1.7
SE	0.2	7	254	1.0	0.6	1.5	4.1	1.1	1.0	1.2	1.8	1.4
2013-14		1	344	1.1	0.6	0.2	0.1	0.7	-0.7	-1.0	-1.6	-2.2

mean (cm)	-0.3	3	74	-0.8	-0.6	-0.8	-0.9	-0.9	-0.3	-1.3	-1.2	-1.1
stdev	0.8	5	164	0.4	0.6	0.7	0.4	-0.8	0.9	0.5	0.7	0.8
SE	0.1	7	254	0.5	0.0	0.2	-0.9	-0.6	-0.4	-0.4	-0.5	-0.5
2014-18		1	330	-0.4	0.4	0.5	0.9	1.0	1.9	3.0	3.8	4.9
mean (cm)	2.0	3	58	2.4	2.7	0.3	1.8	0.4	1.6	0.9	0.1	0.8
stdev	1.3	5	136	2.7	2.6	2.4	2.1	4.2	2.2	0.2	3.2	3.2
SE	0.2	7	240	1.8	2.7	2.6	2.6	2.9	3.1	2.9	2.0	2.8

Cogmagun Reference				**Net change in elevation between sampling period (cm)**								
RSET-03 mid marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	148	1.6	2.2	2.4	2.4	1.9	2.0	2.4	1.9	2.0
mean (cm)	1.7	3	238	1.4	1.1	0.9	2.4	0.8	1.5	2.0	2.3	1.5
stdev	0.5	5	328	2.1	1.9	2.0	1.3	0.8	1.8	1.7	1.6	1.3
SE	0.1	7	58	1.6	2.0	1.4	0.9	1.6	1.7	1.4	0.6	1.0
2010-2011		1	148	1.6	0.6	1.3	1.4	1.4	1.2	1.3	1.8	1.5
mean (cm)	1.4	3	238	1.2	3.1	2.8	0.3	2.2	0.9	0.5	0.9	0.5
stdev	0.9	5	328	0.3	0.6	0.4	1.0	2.2	1.6	3.6	3.5	2.7
SE	0.2	7	58	0.7	1.1	1.9	2.1	1.1	0.4	2.5	0.9	0.9
2011-2012		1	148	-0.3	-0.4	-1.1	-1.1	-1.2	-0.8	-0.7	-1.1	-0.5
mean (cm)	-0.5	3	238	0.1	-1.8	-0.9	0.1	-0.1	0.3	0.4	0.0	0.5
stdev	0.8	5	328	0.2	0.0	0.0	0.1	-1.1	-1.1	-2.6	-1.6	-1.2
SE	0.1	7	58	0.3	-0.8	-0.8	0.1	-0.6	0.2	-2.0	0.1	0.0
2012-2013		1	148	1.3	1.5	1.4	1.1	1.3	0.9	1.0	0.8	0.5
mean (cm)	1.1	3	238	1.2	1.4	1.0	0.6	1.0	1.4	0.9	0.7	0.9
stdev	0.3	5	328	1.3	1.5	1.3	1.5	1.2	1.3	0.9	0.6	0.9
SE	0.1	7	58	1.2	2.2	0.8	0.7	1.2	1.3	1.6	1.2	0.9
2013-14		1	148	0.1	-0.4	1.5	0.7	1.2	1.1	0.6	1.4	0.5
mean (cm)	0.5	3	238	0.1	-0.4	0.1	1.1	0.9	0.7	0.8	1.2	0.9
stdev	0.5	5	328	0.8	0.3	0.5	0.3	0.4	0.4	0.7	0.0	0.2
SE	0.1	7	58	0.0	0.0	0.4	-0.1	-0.2	-0.1	0.2	0.1	0.4
2014-18		1	130	2	2.6	1.1	2.6	0.8	0.6	1.7	0.4	1.2
mean (cm)	1.6	3	220	2.1	2.9	2.4	2.1	1.7	0.9	1.7	1	0.8
stdev	0.8	5	310	1	0.9	0.5	2.2	2.5	4.1	2.5	2.1	0.9
SE	0.1	7	60	1.5	1.8	1.5	1.5	1.2	1.6	1.3	1.4	1.7

Appendix D. Hydrology

Table D. 1. Hydroperiod and inundation frequency at the Cogmagun restoration site based on a 75-day recording from June 11, 2018 to August 25, 2018. Data are ordered from increasing elevation (CGVD 2013) that were recorded on May 18, 2018.

Point ID	N	E	Elevation (CGVD 2013)	Hydroperiod (min)	Hydroperiod (%)	Inundation Frequency (%)	Mean Inundation time (min)
COGL1S3	4992342	410941.4	5.949	3755	3.46	31.13	79.89
COGL1S4	4992377	410905.3	6.019	3380	3.12	29.80	75.11
COGL2S4	4992381	410949.1	6.074	3100	2.86	29.14	70.45
COGL2S3	4992357	410967.6	6.088	2975	2.74	27.15	72.56
COGL3S3	4992377	410996.7	6.117	2805	2.59	24.50	75.81
COGL1S5	4992402	410879.9	6.123	2785	2.57	24.50	75.27
COGL5S2	4992255	411166.7	6.118	2795	2.58	24.50	75.54
COGL1S2	4992304	410980.1	6.118	2795	2.58	24.50	75.54
COGL3S4	4992396	410983.1	6.128	2760	2.55	24.50	74.59
COGL4S2	4992285	411110.5	6.133	2755	2.54	24.50	74.46
COGL2S2	4992318	410999	6.141	2730	2.52	24.50	73.78
COGL5S3	4992298	411141.4	6.149	2695	2.49	24.50	72.84
COGL3S2	4992339	411022.1	6.152	2695	2.49	24.50	72.84
COGL4S3	4992325	411080.9	6.171	2590	2.39	24.50	70.00
COGL4S5	4992392	411032.6	6.182	2525	2.33	24.50	68.24
COGL2S5	4992401	410931.5	6.188	2495	2.30	23.84	69.31
COGL5S4	4992340	411116	6.224	2325	2.14	21.19	72.66
COGL3S1	4992299	411050.3	6.254	2205	2.03	21.19	68.91
COGL4S4	4992366	411052.1	6.268	2155	1.99	21.19	67.34
COGL5S5	4992386	411088.2	6.286	2070	1.91	21.19	64.69
COGL4S1	4992245	411139.9	6.398	1645	1.52	18.54	58.75
COGL2S1	4992280	411031.3	6.585	880	0.81	9.93	58.67
COGL5S1	4992212	411192.2	6.773	510	0.47	7.28	46.36
COGL1S1	4992271	411013	6.883	315	0.29	5.30	39.38

Appendix E. Permissions

Photo A in Figure 6 is property of Anton Balaz used with permissions