5-2022

Assessing the Impacts of a Ditch Plug on Groundwater Hydrology, Sedimentation, and Carbon Dynamics in a Salt Marsh, Phippsburg ME

Henry Stephen King
Bates College, hking2@bates.edu

Assessing the Impacts of a Ditch Plug on Groundwater Hydrology, Sedimentation, and Carbon Dynamics in a Salt Marsh, Phippsburg ME

A Senior Thesis

Presented to

The Faculty of the Department of Geology

Bates College

In partial fulfillment of the requirements for the
Degree of Bachelor of Arts

By
Henry S. King
Lewiston, Maine

[December 17th, 2021]

Acknowledgements

I would like to start by thanking all of the backers and funders of this project, as without your support none of this could have been possible. I’m immensely grateful as well to the St. John's family for allowing this research project to take place at the Bates Morse Mountain Conservation Area, and for always waving and greeting me with a smile. It was a pleasure to meet and get to know you all even in the short amount of time I was there.

I’d also like to thank all the staff working at BMMCA, Specifically Caitlin, Don and Frank. You all made my summer unforgettable with all the stories, jokes, and anecdotes you shared with me. You truly made me feel like a part of the marsh family.

A huge thank you as well to the entirety of the Bates Geology Department, as I wouldn't've been able to achieve any of this without the constant support and encouragement you provide. Thank you to all my fellow geology students who’ve helped me in the field, in the classroom, in the lab, and in every other aspect of my academic life. Thank you Phil Dostie for your unending support in this project. I cannot thank you enough as I don’t think there is a single aspect of this research that I could have done without you. And most importantly, thank you to my thesis advisor Beverly Johnson. I don’t even have the words to express how grateful I am for your support. Not only were you instrumental in the completion of my thesis, but to me making it through each and every day during this semester. And for that I cannot overstate how thankful I am that you were my advisor.

Lastly I would like to thank all my friends and family who’ve supported me throughout this project and my academic career, as without you all I wouldn’t be able to get out of bed every morning with a smile.

Abstract

The Southern Alcove (S. Alcove) of Sprague Marsh in Phippsburg Maine is a section of back barrier salt marsh that has been tidally restricted by the installation of a ditch plug into its main stream channel since the early 2000s. Previous research in the area has shown that the mean water level upstream of the ditch plug within the S. Alcove is significantly higher than the downstream area (Barry 2012). The purpose of this study is to provide more research and information on the condition of the S. Alcove (nearly ten years after it was last studied), as well as provide insight as to whether or not the ditch plug should be removed. This was done by monitoring fluctuations in the groundwater levels, assessing sedimentation rates, and analyzing sediment cores for carbon content from within and from outside of the S. alcove. Up gradient of the ditch plug and in the S. Alcove, ground water response to tidal fluctuations was muted and sedimentation rates were higher than down gradient of the ditch plug. It was also found that approximately the first 5 cm of each sediment core was very rich in organic carbon, which has been seen in other ditch plugged environments along the east coast (Vincent et al., 2013). Studies show that persistently elevated water levels in salt marshes result in decreased marsh elevation, increased soil salinity, decreased soil redox potential, decreased soil strength, and decreased carbon storage capacity (Vincent et al., 2013). Thus, I recommend removal of the ditch plug in the S. Alcove on the Sprague River Marsh. , Careful attention must be paid to the methods of removal to ensure that the marsh is restored, and monitoring practices must be employed for several years after ditch plug removal to ensure that we learn more about how this system evolves with time.

Table of Figures

Table of Tables

1.1 Maine's Coastline

The coastline of Maine is characterized by four different sections that are determined by the distinct geology and geomorphology of each region (Jacobsen et al., 1987). Due to the unique geology and geomorphology these sections each develop salt marshes in different ways. This is due to varying amounts of sediment deposition, wave protection, and overall differing landscape. These 4 sections from SW to NE include the "Arcuate Bays", "Indented Embayments", "Island-Bay Complex", and the "Eastern Cliff Shoreline" (Figure 1.1; Jacobson et al., 1987; Kelley et al., 1988).

The section furthest Southwest, the "Arcuate Bays", are characterized by large sandy beaches in between with large bedrock headlands (Jacobson et al., 1987). The salt marshes in the Arcuate Bays section of the coast form behind the beaches as back barrier marshes and make up 33% (26.4 km^2) of the existing salt marshes in Maine (Jacobson et al. 1987).

To the northeast of the "Arcuate bays" lie the "Indented Embayments". This section contains Casco Bay and many north-trending islands and peninsulas with back-barrier and fluvial salt marshes (Kelley et al. 1988). This section of the coastline contains 35% of the total salt marshes in Maine (Jacobson et al. 1987).

The next furthest northeast section is the "Island-Bay Complex" (Jacobsen et al. 1987). This section of the coast is the largest section of coastline characterized by Kelley and Jacobson. It is made up of granitic islands and exposed embayments providing little protection from wave energy (Jacobson et al 1987; Kelley et al 1988). Marshes here typically form as fringe marshes along protected areas of the coast, though other varieties are present (Jacobson et al., 1987).

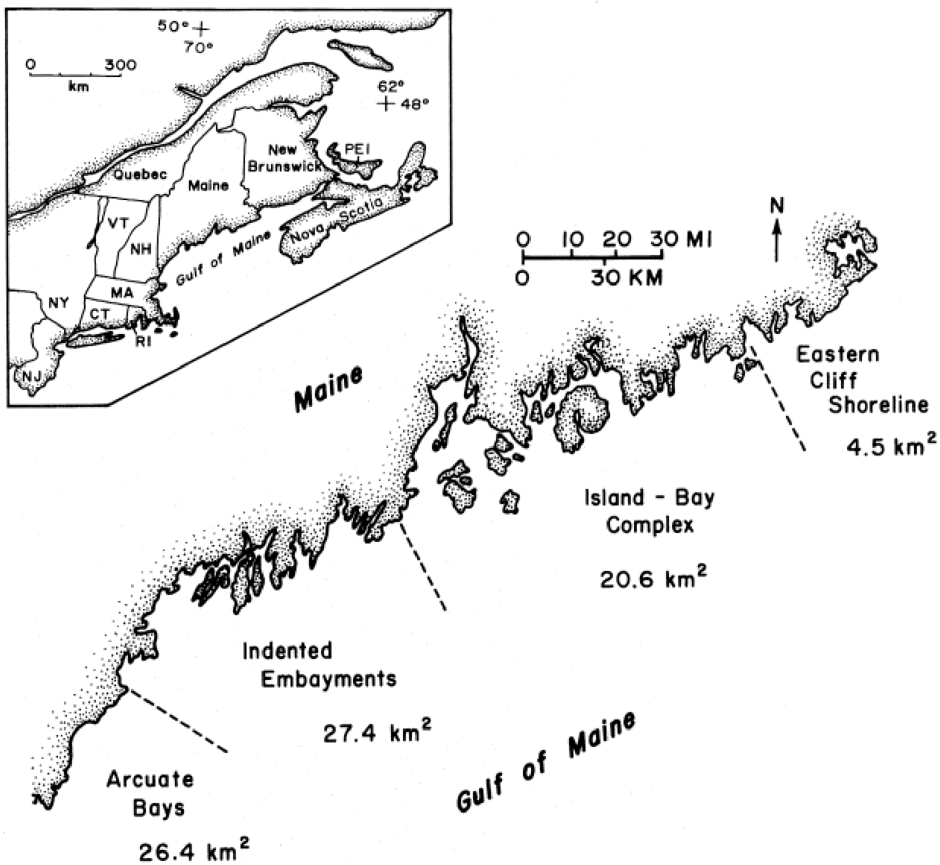

Figure 1.1: Shows a map of Maine's coastline labelled with the 4 different sections characterized by their geomorphology. Included is the total area that salt marshes cover in km^2 for each section (Jacobsen et al., 1987; Kelley et al., 1988).

Many of the salt marshes in this area are dominated by mudflats and coarse-grained sand flats and make up 25% (20.6 km^2) of all the salt marshes in Maine (Jacobson et al., 1987).

The last section along the coastline of Maine is the “Eastern-cliff shoreline” (Kelley et al. 1988). This section of the coast runs from Machias Bay all the way east to the Canadian border. The coast here mainly consists of unbroken vertical cliffs bounded by bedrock faults. It only contains 7% (4.5 km^2) of salt marshes because of such little wave energy mitigation (Kelley et al., 1988).

For Maine, the sea level in the late Quaternary has been controlled both by isostatic rebound and by eustatic sea level rise (Figure 1.2; Kelley et al., 2010; Nelleman et al., 2013). Approximately 15,000 year ago (15ka) the deglaciation of the Laurentide ice sheet left the Maine coast free of ice. However due to isostatic depression sea level then was approximately 75m higher than it is today (Kelley et al., 2010). This period can be seen as a marine environment on the present day coast where there are glaciomarine sediments at elevations lower than 220 feet (Kelley et al., 1988). When the isostatic rebound happened, sea level fell rapidly, reaching its lowest level (-60m) 12,500 years ago (Kelley et al. 2010). At this point there was an erosional unconformity and relocation of glaciomarine sediments occurred. Between 12,500 and 11,500 years ago the stabilization of the land and continued global ice melt led to a rise in the relative sea level (Kelley et al. 2010). From about 11,500-7,500 years ago there was a “slowstand” period wherein the sea level rose <5m. During this period the sea level was approximately 17-22 m below current day sea levels. It is thought that this “slowstand” period is when humans became so enticed by salt marsh environments (Kelley et al. 2010). After this “slowstand” period the sea level increased over approximately 2,500 years to the level it is at currently.

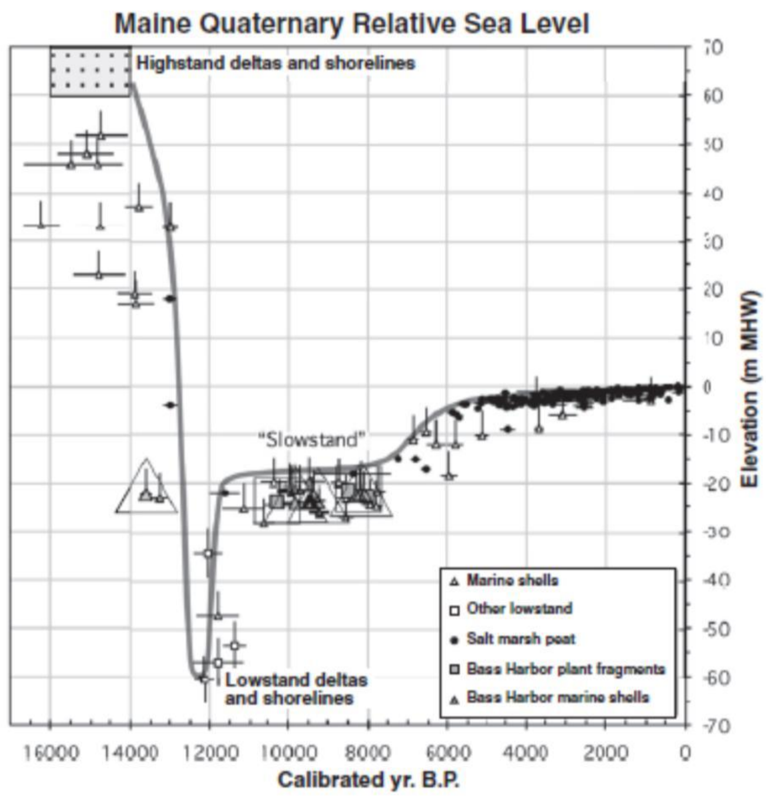

Figure 1.2: (Kelley et al., 2010) this figure depicts the relative sea level rise within Maine over the last 16,000 years.

1.2 Salt Marshes General

A salt marsh is an ecosystem that is situated on the boundary between terrestrial marine environments (Taylor, 2008). They are characterized by having a tidally controlled hydrologic regime that is inundated with salt water twice per day. This leads to a wide biodiversity of both halophytic flora and fauna (Silverti et al., 2003). They are predominantly vegetated by herbs, grasses, and shrubs (Kennish, 2001) and are home to many different species of birds and nektons, including many migratory and endangered species (Taylor, 2008). These habitats harbor life in the open water, drainage channels, and within the substrate and peat itself (Taylor, 2008).

Salt marshes not only provide suitable habitat for many different species, but they also acre for acre produce as much biomass as intensely farmed agricultural land (Taylor 2008). Due to this, salt marsh ecosystems have provided humans with a reliable source of food, fuel, building materials and livestock bedding for years. In addition to these provisional benefits, salt marshes filter out heavy metals and pollutants, act as a storm surge barrier, and sequester large amounts of carbon (Gedan et al., 2011). Salt marshes, along with mangroves and seagrass beds, are among the ecosystems with the highest carbon sequestration rates (Figure 1.2; Taylor et al 2008, Nelleman et al., 2009; McLeod et al., 2011).

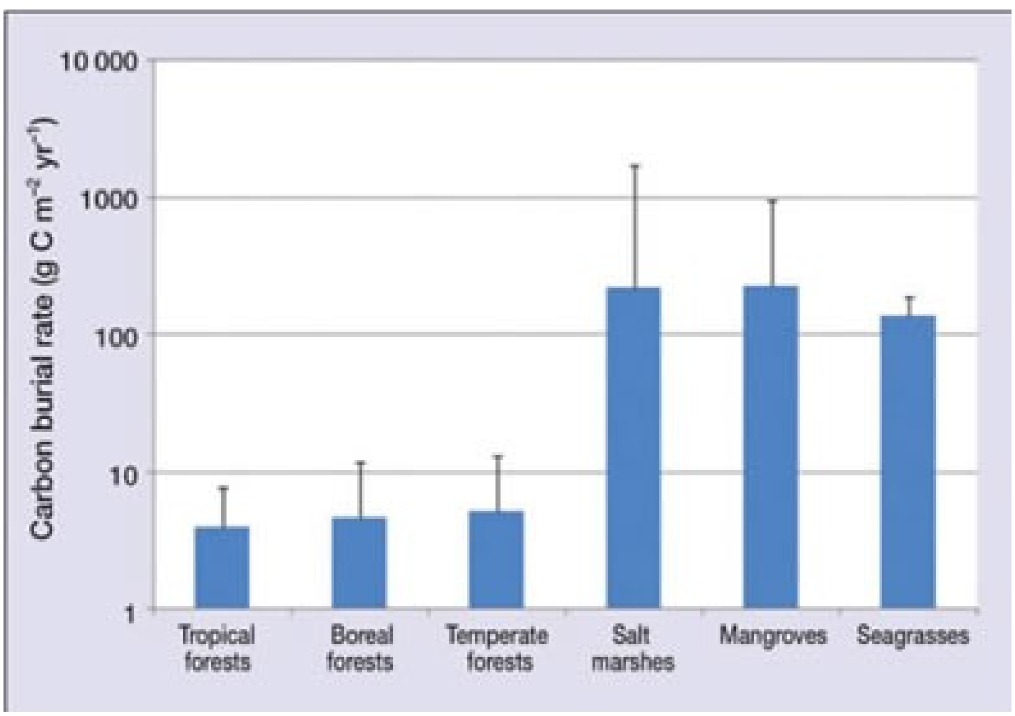

Figure 1.3: Average carbon burial rates of 6 different types of environments. The 3 environments on the left are terrestrial and have significantly lower carbon burial rates than the 3 coastal environments on the right.

1.3 Salt Marsh Formation

Salt marsh elevation is sustained by a combination of biological and sedimentological inputs (Figure 1.3). In the green boxes are all the different factors that determine the vertical formation of a salt marsh. These factors are salinity, plant growth/turnover, flooding depth/duration, sedimentation and erosion, soil elevation, biomass accumulation, decomposition, and nutrients. Each of these factors not only plays a role in the formation of the salt marsh, but also in controlling the other factors. Salt marsh plant growth, for example, is affected by salinity, hydroperiod and herbivory, and take up CO_2 to create more biomass which increases marsh elevation. This increase in marsh elevation will in turn change the hydroperiod within the marsh. In addition to the factors, in the white boxes on figure 1.3 are the different major outside inputs that can lead to a change in the processes within the marsh. Each one of these outside inputs greatly dictates its associated factors, and so when there is even a slight change in the input the effects can be drastic. If for example there was an alteration to the river flowing into a salt marsh, such as a restriction or blockage, the salinity, flooding depth/duration, and sedimentary process would all be changed. As can be seen in the model there is a lot of interconnectivity among and within the different marsh properties. While the factors and inputs of marsh formation are known, much less is known about the specific interactions between the different inputs and factors in the system.

Sediment accretion and erosion also helps to dictate the vegetation patterns on the marsh (Vincent et al., 2012) and the marshes hydroperiod (Nolte et al., 2013; Vincent et al., 2013).

Figure 1.3 shows the different impacts that sedimentation and erosion have on the environment. The figure shows that sedimentary processes are primarily driven by the hydrology in the area. Storms, sea level rise, and an altered river flow can all lead to changes in the marshes' sedimentation and erosion. This change to sedimentary processes will again have an effect on the soil elevation of the marsh, but it will also impact the amount of nutrients going into the ecosystem. This change in nutrients could in turn either help the marsh by providing plants with the requisite nutrients to thrive, or it could hinder the marsh by increasing the amount of subsurface decomposition and starting a positive feedback loop of decomposition. With the potential to fall into a positive feedback loop with an increase in hydroperiod subsequently leading to an increase in sedimentation rate (Wood et al., 1989), this is an important factor to take into account when dealing with marsh management techniques. For this reason it is necessary to know the rates of sediment accretion and erosion of a given area of salt marsh in order to be able to assess whether the area can keep up with modern levels of sea level rise (Wood et al., 1989).

As one can see all these biological and sedimentological factors are inherently linked to each other and as a whole lead to the formation of the marsh. That is why when any one factor or input is changed there can be great and lasting effects upon the processes in the area. This makes the monitoring and maintenance of salt marsh environments all the more important.

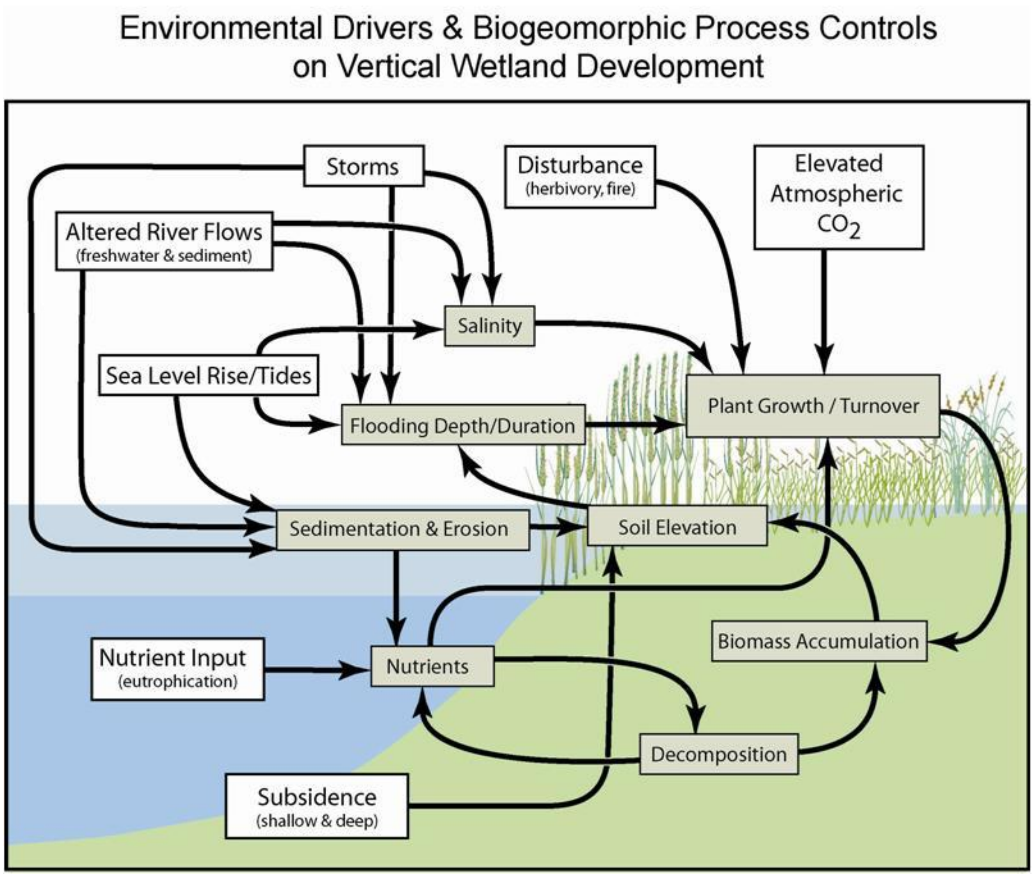

Figure 1.4: USGS Wetland Development model showing the different factors and inputs of Vertical salt marsh development.

1.4 Sprague Marsh

Sprague marsh is located within the Bates-Morse Mountain Conservation Area in Phippsburg Maine at 43°45’ N / 69°50’ W. The marsh is located just inland of Seawall beach. The proximity of this back barrier beach helps to define Sprague marsh as a hybrid back-barrier and fluvial-minor marsh (Kelley et al., 1998). The marsh is contained within a glacial valley that is bisected by the Sprague river which enters on the northern edge and drains into the Atlantic at the southeastern corner. The western side of the valley is underlined with bedrock consisting of West Marsh Granofels, West Marsh Amphibolite, Garnet Rich West Marsh Schist, Mica-rich West Marsh Schist, and West Marsh Schist (Sive et al., 2012). The eastern boundary of Sprague marsh consists of a large pegmatite intrusion that forms Morse Mountain. The majority of the sediments used in the formation of Sprague are either from the Presumpscot formation or are fine grained sands of unknown origin (Kelley et al., 1988). The approximate rate of sedimentation over the last 3,000 years was ~0.07 cm/year (Johnson et al., 2007).

Human alterations on Sprague Marsh began as early as 1716 with the settling of the Pejepscot Proprietors in the region (Vincent et al., 2014). The land was regarded as a high value area as the abundance of *Spartina patens* (salt marsh hay) was conducive to livestock fodder/bedding. The marsh land was divided between the first fifty settlers in the area and property boundaries were demarcated by ditches, both to increase growth of *Spartina patens* and decrease the breeding habitats of mosquitos (Vincent et al., 2014). The next most prominent alteration on Sprague marsh took place during WWII and was the building of the main road and narrow causeway along the northern section of the marsh. This was done so that a radar tower could be built at the top of Morse mountain for monitoring ships and planes. In 1958, Junior Mellon, one of the landowners on the marsh, dredged and straightened the main tidal channel to further the drainage of the salt marsh region and allow easy boat access (Vincent et al., 2014). This straightened tidal channel is still currently the main tidal channel in Sprague marsh and the natural meandering stream channel is secondary (Vincent et al., 2014).

The history of land restoration on Sprague marsh began in January of 2000. The United States Fish and Wildlife Service (USFWS) with permission from the Natural Resource

Conservation Service (NRCS) ,The Nature Conservancy (TNC), and the Small Point Association planned to do excavation in the northern part of the marsh. The aim of this restoration was to attempt to limit the freshwater inflow as well as the growth of invasive species such as *Phragmites australis* (Common Reed) (Vincent et al., 2014). Two days into the restoration project however the marsh froze over and the project was abandoned.

The next restoration efforts would begin in 2002, this time with focus being on open marsh water management (OMWM) or ditch-plugging. The goal of this project was to increase the amount of open water on the surface of the marsh, and in consequence increase the amount of bird and nekton habitat. 3 ditch plugs were added to the north part of the marsh early in 2002 and 11 more were added south of the causeway in fall of 2002. Of the ditch plugs added south of the causeway three adjacent to the main tidal channel (DP-2 through DP-4) and eight more in a tidally restricted portion of the southern marsh (DP-1 and DP-5 through DP-11). At each site 2 sheets of plywood (8 ft x 4 ft) were pounded into the marsh channel, pools were excavated nearby, and the material removed from the pools was used to backfill the ditch behind the plywood (Vincent et al., 2014). In 2002 the Sprague river channel south below the causeway was dredged in hopes of restoring tidal flow to a portion of the marsh north of the causeway. This was done with an excavator to remove the hard stabilization underneath the bridge. Most recently in 2006 the causeway was widened in order to further increase the tidal flow back into the norther part of the marsh (Vincent et al., 2014). This final restoration effort was not followed up with additional monitoring by the USFWS, allowing room for the long-term study of marsh restoration projects.

1.5 Salt Marsh Hydrologic Conditions

The subsurface hydrologic conditions within a salt marsh are controlled by a variety of factors. The principle factor controlling them however is the source of the water. In the case of most salt marshes this water source is made up of tidal and groundwater influences, precipitation, runoff, and evapotranspiration (Knott et al., 1987). Most of these mentioned factors are forms of water input, but evapotranspiration is an output of the marsh hydrologic system; evaporation of the marshes standing surface water or water from within marsh vegetation effects the salinity and water level of the marsh. Freshwater sources such as precipitation and runoff also effect the salinity and water level of the marsh (Nuttle, 1988). Balanced hydrologic inputs are important to a properly functioning salt marsh ecosystem.

In 2011 Colin Barry assessed the hydrology of 2 sections of Sprague marsh, one of these sections being the Southern alcove. He specifically monitored the water level and conductivity within the areas so as to see the differences in hydrology between a ditched and a plugged system within the marsh. He found that in the S. Alcove upstream of the ditch plug the water table elevation was much greater than in the ditched section of the marsh. This backed up the findings of Adamowicz et al. (2002) that mean water levels upstream of a ditch plug are substantially higher than downstream. He concluded that this increased mean water level might be impacting the overall productivity of the S.Alcove and that further research, especially on the surface and in the subsurface, was needed in order to quantify the effects.

Surface and subsurface studies of marsh hydrology are not the most abundant type of publication out there, however the field has seen more growth in more recent years (Waltham et al., 2021). Early studies determined that infiltration into the salt marsh is a direct result of inundation time, hydraulic conductivity, and peat pore space (Hemond, 1984). With both sea water infiltration and precipitation being the direct controls of salinity, it can also be determined that soil salinity could serve as a proxy for salt water infiltration or precipitation. This surficial and subsurface hydrologic information would give insights into specifically how the increased mean water level caused by the ditch plug is affecting the S.Alcove.

1.6 Sediment Deposition, Accretion, and Erosion

Sedimentation rates and erosion rates within a salt marsh are two important factors for determining restoration efficacy. Marsh accretion was measured in New England by measuring sediment accumulation over a brick dust marker horizons (Wood et al., 1989). Sediment accretion rates were shown to be within 0-13 mm yr^{-1}, with back barrier marshes having the highest accumulation rates. Additionally, accretion rates using chemical markers such as ^{201}Pb (Armentano and Woodwell 1975), ^{241}Cs (Delaune et al., 1987), and ^{14}C (Belknap and Kraft, 1977) to date the sediments. The sediment accretion rates found using this dating method gave a mean sediment accretion rate of 4.5-5.5 mm yr^{-1}, which is a significantly smaller range. This difference in range is not surprising however as the dated sediment accretion rates only get a mean accretion rate over a long period of time, whereas the values obtained using the marker horizons were gathered periodically and compiled to find average accretion rates.

Sediment accretion and erosion have a variety of impacts on the marsh ecosystem (Figure 1.3). In a salt marsh ecosystem the sediment accretion and erosion is one of the main determining factors in the elevation of the salt marshes surface. For this reason it is necessary to know the rates of sediment accretion and erosion of a given area of salt marsh in order to be able to assess whether the area can keep up with modern levels of sea level rise (Wood et al., 1989).

Sediment accretion and erosion also helps to dictate the vegetation patterns on the marsh (Vincent et al., 2012) and the marshes hydroperiod (Nolte et al., 2013; Vincent et al., 2013). Figure 1.Y shows the different phases of a sediments life and the different effects that those life phases have on the environment. The figure shows that sediment deposition and sediment accretion are both linked to both the hydroperiod and biomass within the marsh. With the potential to fall into a positive feedback loop with an increase in hydroperiod subsequently leading to an increase in sedimentation rate (Wood et al., 1989), this is an important factor to take into account when dealing with marsh management techniques.

1.7 Purpose

The two main guiding questions for the research are: “What is the status of the S. Alcove?” and “Should the ditch plug be pulled out?”. The aim and objective of this study is to analyze the sedimentation rates, carbon density, and hydrologic regime in order to determine the current health of a tidally restricted salt marsh. This analysis will then be used by the management at Bates Morse Mountain Conservation Area to guide restoration efforts in the area.. This research is coming after similar thesis research that was done in the same area ~10 years ago by Colin Barry. Using groundwater wells and land coverage surveys he was able to tell that the water table within the S. Alcove was significantly higher than other sections of the marsh. He concluded that this was attributed to the ditch plug located at the mouth of the S. Alcove. The purpose of this study is to follow up with the monitoring of the S. Alcoves water table 10 years later and to determine sedimentation patterns within the S. Alcove.

2.1 Study Site

The study took place during the summer and early autumn of 2021 in Sprague Marshes Southern Alcove (Figure 2.1). The Southern Alcove of Sprague Marsh is an approximately 1 hectare section of salt marsh upstream of a ditch plug located at 43°43’52”N 69°49’15”W (shown in green on Figure 2.1). The site is primarily vegetated by *Spartina alterniflora*, *S. patens*, and *Juncus gerardii.* Upstream of the ditch plug there is a man-made main channel as well as 2 ditches dug perpendicular to the main channel. Prior to the construction of the ditch plug these channels were the primary means of drainage for the S. Alcove. Previous to this study there has been a variety of research done in the surrounding area. Among these are the studies conducted by Barry in 2011 and Vincent et al. in 2014. Barry, as previously discussed, determined that high water table elevation of the S. Alcove. Vincent et al. on the other hand looked at the effect that the ditch plug had on the vegetation within the S. Alcove. The wells at the site that were previously installed consisted of well #1 in the S. Alcove and well #4 in the healthy marsh downstream of the ditch plug. Two additional wells were installed for this study upgradient of the ditch plug (wells #2 and #3). In addition to these groundwater wells, Sediment cores were collected within the S. Alcove near well #1 and out of the well hole of well #2. Lastly sedimentation rates were gathered along 3 transects: 1 in the S. Alcove, 1 along the stream channel, and 1 going into the unplugged area downstream of the S. Alcove.

Figure 2.1: a map of the study site within and surrounding the S. Alcove. On the map the ditch plug is shown with a green marker, all the groundwater wells are shown with a red marker, and the sedimentation rate transects are shown in orange. The yellow-orange transect denotes that this transect was done parallel to the stream channel, with all replicates placed 3m from the channel.

2.2 Groundwater Monitoring

Groundwater monitoring (pressure/water depth) took place in three wells within the southern alcove (Wells #1, #2, and #3) and one well outside the alcove in a well drained part of the marsh (Well #4). The tidal stream down gradient of the ditch plug was also monitored for water depth (Stream Channel Monitor). Additionally one atmospheric pressure/temperature sensor was placed in the S. Alcove for calculations of partial and total pressure on the groundwater sensors. The sites for the three wells within the S. Alcove were arranged so that the three sites triangulated the alcove (Figure 2.1). Well location was based off a variety of factors which included whether existing wells were present, presence of standing water (at low tide), proximity to other wells, and peat structure. It was important to determine that the wells were placed far enough apart from each other to allow for a triangulation of the water table, but not so far as to be outside of the area affected by the ditch plug. Wells were installed and loaded with sensors at low tides, both to avoid working in standing water and to know that the sensors were installed in the wells at the lowest possible depth to allow for total water coverage. Peat structure was also an important factor in determining well location: if the peat did not have enough structure the well hole would collapse in on itself before a well could be installed. This happened a few times around well #2, where the core was pulled and before the well could be inserted the lip of the well hole would begin to slump and fall back into the hole. We did not install any wells in holes that did this for fear that this was happen down the length of the hole.

All well holes were made using a 1m Dutch core auger. The holes were then filled partially with sand, and the well casing was inserted. The marsh surface surrounding the well casing was then packed with sand and a thick layer of bentonite clay was deposited around the casing. Each well casing consisted of a screened section at the bottom to allow liquid to flow through but not sediments and a long closed off casing that would stick out above the highest tide (Figure 2.2). In each well two sensors were placed into the water table, one HOBOWare U20L to track pressure and temperature every 15 minutes as well as a HOBOWare U24-002-C to track the conductivity and salinity every 15 minutes (Figure 2.3). Well 2 was installed on 7/12/21 and Well 3 was installed on 7/23/21. Sensors were installed in Wells 1 and 2 on 7/12/21 and in Well 3 on 7/23/21. The atmospheric sensor was also installed on 7/12/21. On 8/28/21 the two sensors were moved from Well 1 to Well 4 to get a sense of the tidal conditions in the salt marsh downstream of the ditch plug.

On 7/23/21 the downstream channel sensor was setup. This consisted of a setup very similar to the groundwater monitoring wells, but instead of being embedded in the ground it was strapped to a light duty steel fence post that was anchored into the stream channel (Figure 2.2). A HOBOWare U20L was then installed into the well casing to record pressure and temperature data every 15 minutes. All sensors were removed on either 9/18/21 or 8/28/21.

Figure 2.2: This image shows the stream channel monitor used within this study. All groundwater wells inside and outside the S.Alcove were functionally identical to this setup.

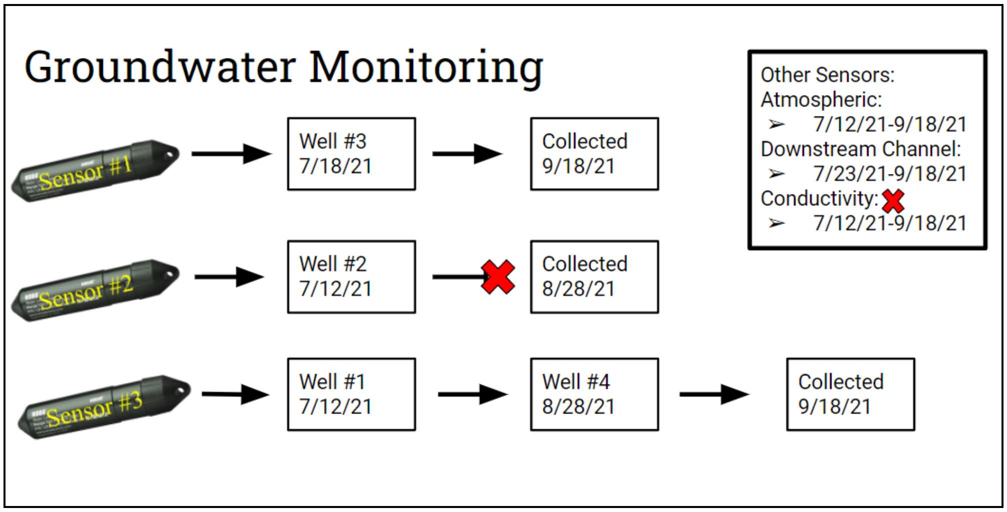

Figure 2.3: Diagram showing the dates for which the HOBOWare Dataloggers (U20L & U24-002-C) were in the field logging data. There were malfunctions that caused losses in data within Pressure/Temperature sensor #2 and within the conductivity sensors.

2.3 Sedimentation Rate Collection

To collect sedimentation rates in the S. Alcove and surrounding areas a method developed by Yellen et al. that is currently in review was used. This method involves the placement of sediment traps into the surface of a given salt marsh along a transect. At distances of 3m, 10m, and 40m four 50ml test tubes with a mesh basket over the mouth were pushed into the marsh surface so that approximately 1 cm of test tube was showing (Figure 2.4). These tubes were then left in the marsh for a certain amount of time, allowing tides to flow over them and sediment to be collected.

For this study, 3 transects were created in and around the S. Alcove. The transects were labelled “S. Alcove”, “Channel”, and “Downstream” The sediment traps were deployed on 8/3/2021 and collected on 9/18/21. Within the S. Alcove there was 1 transect placed upstream of the ditch plug. The location of the transect was chosen due to its proximity to the ditch plug, as well as for its optimal peat conditions along the transect. For the 2 transects outside of the S. Alcove. 1 was set up parallel to the stream channel with all collections sites 3m from the channel, while the other was a standard transect arrangement located 5 m downstream of the ditch plug. The sediment traps placed parallel to the stream channel were placed 3m from the stream channel and 5m away from each other. This transect was placed in hopes of seeing a shift in sedimentation patterns between the alcove and the downstream area. The downstream transect was placed 5m downstream of the channel transect and was arranged with sediment traps at 3m, 10m, and 40m. During the collection process the tubes had their mesh baskets removed, were visually inspected, and then capped. The samples were then frozen to preserve them.

Once back in the lab, the sediment traps were freeze-dried. Upon close inspection, it became apparent that the sediments were loaded with salt. Samples were de-salted by rinsing in E-Pure, centrifuging, and decanting three times. . Finally the samples were freeze dried again and then massed. This combination of methods was effective at removing the excess salt that would have interfered with total sediment mass.

Figure 2.4: Shown is an example of the sediment traps used. Galvanized steel baskets were zip tied around the mouth of the test tube in order to keep plant detritus out.

2.4 %LOI and Dry Bulk Density

For this study %LOI was run on 2 separate cores that were collected from the S. Alcove of Sprague Marsh. Core 1's location was just a few meters away from groundwater Well 1 while Core 2 was taken from the same hole as groundwater Well 2 is now in. These 2 cores are 61.0cm and 66.5cm respectively. The cores were taken using a 1m dutch peat auger and were wrapped in plastic wrap and PVC tubing for transport and storage.

Once in the lab, the cores were split lengthwise and the inside was scraped to expose a fresh and undisturbed layer. The cores were then photographed and described. The cores were subsampled every 5cm with a 2cc steel cutter; peat was extracted from the core and placed into a pre-weighed crucible. These crucibles were then massed again in order to obtain the total weight of the crucible and sample. All the crucibles were then placed into the drying oven at 40℃ for a week to allow the peat samples to dry out completely before being massed again. The last step was then to place all the samples in the muffle furnace and bring them up to 450℃ for 6 hours, and allow them to cool overnight before massing them one final time. From this point we were able to use the mass of the dry sediment and the mass of the ashes to figure out the percentage of material lost on ignition (LOI) of the sample. The %LOI value was then be used to determine the %Carbon content of the sample, as well as the bulk carbon density due using the empirically derived relationship between %LOI and %Org C (Howard et al., 2014).

3.1 Groundwater Monitoring Results

The results of the groundwater monitoring provided w useful information about the tidal signal and inundation period of both the Southern Alcove and the area downstream of it. Of the three water pressure and temperature sensors (HOBOWare U20L) only two (sensors 1+3) were able to produce data on a timescale that was conducive to this research. The last water pressure and temperature sensor (sensor 2) malfunctioned after approximately 2 weeks, only producing a fraction of the data the other sensors did. Additionally, two of the conductivity sensors that were placed also malfunctioned. These two sensors did not produce any usable data as the readings that were produced were clearly corrupted in different ways. Conductivity sensor 1 (Well 3) produced the same exact conductivity reading (33310.8 µS/cm which corresponds to a salinity of approximately 21 ppt) for 1000 straight data points before instantly dropping to 0 µS/cm for the remainder of its time in the marsh. Sensor 3 (Well 1) did a similar thing to sensor 1, though with a different reading and the date and time the sensor recorded were the same for every recording. Sensor 2 (Well 2) did however get good data showing a slight change in conductivity each subsequent reading. The average conductivity in the area was 24778 µS/cm (which corresponds to a salinity of approximately 15 ppt) with a maximum value of 26446.1 µS/cm which corresponds to a salinity of approximately 16 ppt). All other sensors functioned properly and collected the necessary data.

Beginning with water sensors at Wells 1 and 3, we were able to see a clear image of the tidal sequence within the S. Alcove between 7/18/21 and 8/28/21 (Figure 3.1). From this data we are able to see a variety of things regarding the hydrology of the area. The first thing to notice is that wells 1 and 3, while not at the same water level, seem to show a similar trend in water height, with them generally increasing and decreasing at the same time. The 2 periods with peaking water levels during the first month correspond to the high tide events causing the water levels to be much higher in the third week of July and August. The tidal signal is also pronounced in these data where high tides correspond to high water levels in both wells. The spring tides (as designated by the red box on Figure 3.1) are represented by higher amplitude shifts in pressure. Well 1 appears to be more variable sensitive to the tides than well 3 as evidenced by the higher rate of inundation.

We can also see that when the stream channel is overlaid on the data from wells 1 and 3 (figure 3.3) that these peaks correspond to the stream channel water depth as well as the S. Alcove. The water levels in the stream and the wells are driving by the tides. Wells upstream of the ditch plug experience a much lower daily fluctuation in water table height than the stream channel. The S. Alcoves water table both does not fill nor empty at a rate similar to that of the stream channel, showing a muted tidal signal throughout the entire area for the duration of the recorded period.

In Figure 3.2 we can see the groundwater data from second half of the summer when Sensor #3 was switched from Well #1 into Well #4 (downstream of the ditch plug and in the healthy part of the marsh). In Well #1 The daily tidal signal fluctuated between 0.2 and 0.3 psi (when not affected by astronomical high tides) between daily maximum water height and daily minimum water height in comparison to the tidal sequence coming into the area as shown by the water levels of the stream channel. In Well #4 we see daily fluctuations in the height of the water table that are much more in line with the stream channel's tidal signal. In the fluctuations in Well #3 we can see a similar muted tidal sequence that is consistent with the first half of the summer shown in figure 3.1. When the stream channel data is overlaid on this half of the summer we can see that the fluctuations in well #4's water pressure is very closely related to the tidal sequence the channel is showing (figure 3.4). Though the fluctuations in the stream channel are often more pronounced than well #4, with a higher maximum and a lower minimum, the peaks and rate of inundation are within the same range of values. When water pressures in Well #3 are compared to the stream channel it is seen that the tidal sequence within the S. Alcove is still greatly muted, not once being at a similar level to the channel. While Well #3 does not exhibit similarity in range of data or tidal sequence, we can see that the overall trend in water pressure is closely mimicking the minimum water pressures of the stream channel, though it never reaches the same level. There were no heavy rain events in the month of August as there were in July.

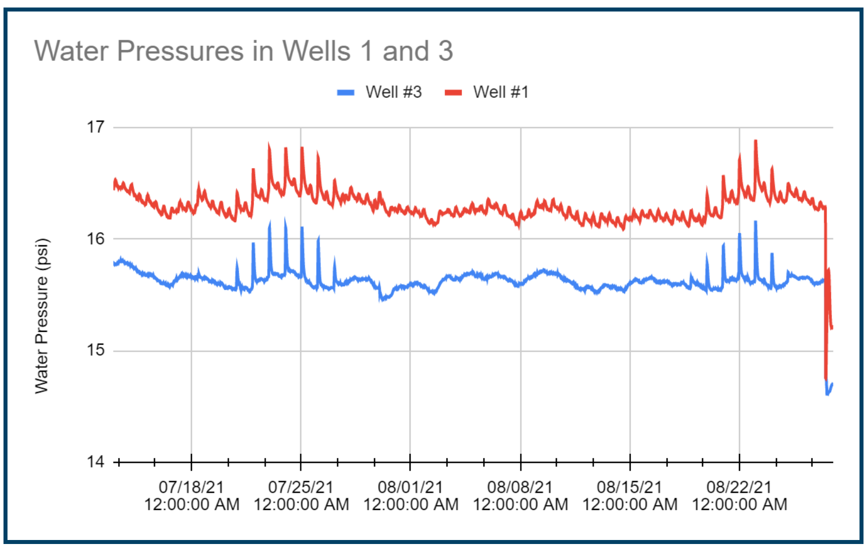

Figure 3.1: Shown is the water pressure data (psi) collected out of Wells #1 and #3 during approximately the first month of research. The two areas of peaking water pressure correspond to high tide events in the area.

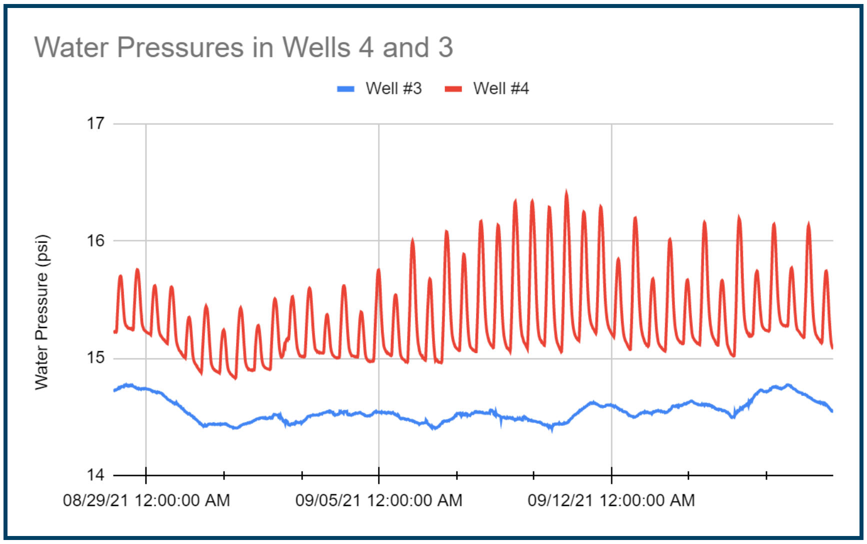

Figure 3.2: Shown is the water pressure data (psi) collected out of Wells #3 and #4 during approximately the second month of research.

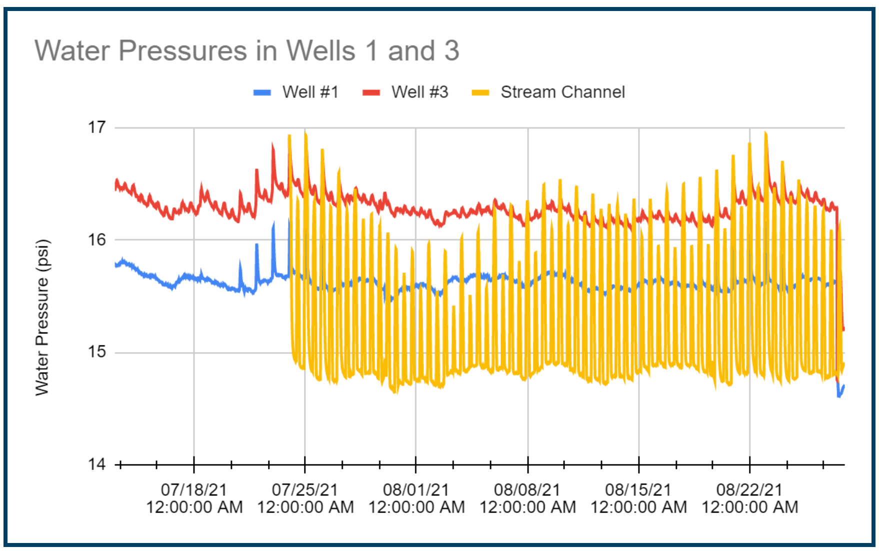

Figure 3.3:

Shown is the water pressure data (psi) collected out of Well #1, Well #3, and the stream channel during approximately the first month of research. Notice how high the fluctuations are in the stream channel as opposed to the two wells in the S.Alcove.

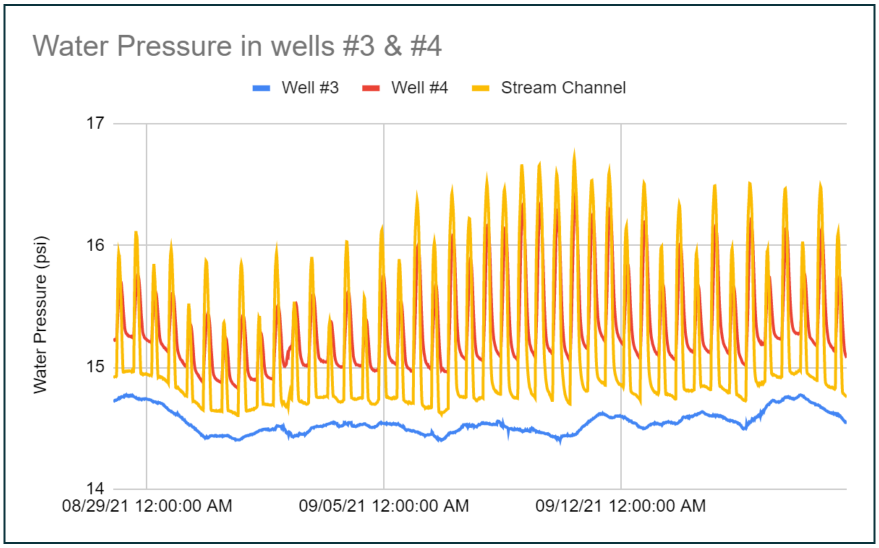

Figure 3.4: Shown is the water pressure data (psi) collected out of Well #3, Well #4, and the stream channel during approximately the first month of research. Notice how high the fluctuations are in Well #4 and the stream channel in comparison to Well #3 in the S.Alcove.

3.2 Sedimentation Rates

The results of the sedimentation data are shown below in table 3.2. Here we can see the 4 replicates taken at each sample site, as well as which sample was excluded based on the visual inspection. The non-excluded replicates were then added together and divided by the total area of the sediment traps to get a total sedimentation amount per cm^2 for each site. These values were then converted to g/m^2 and divided by the total number of days the sediment traps were placed for, giving a daily sedimentation rate in $\frac{g/m^2}{day}$. This showed a sediment distribution along each transect, allowing us to compare the distribution in the S. alcove to that of the downstream area unaffected by the ditch plug (figure 3.5). It is seen that the sedimentation rate decreases with distance along the transect, with the highest value at 3m from the stream channel and the lowest at 40m.

The sedimentation rates collected along the stream channel did not show a significant trend (figure 3.6). The values collected at 10m from the ditch plug were the lowest at 3.285 $\frac{g/m^2}{day}$, the highest rate was collected 20m from the ditch plugs with a value of 8.610 $\frac{g/m^2}{day}$, and 30m from the ditch plug a rate of 5.648 $\frac{g/m^2}{day}$ was collected. It is curious that there is no systematic change in sedimentation along the stream channel and begs for further investigation.

Location	Position	R1(g)	R2(g)	R3(g)	R4(g)	Total(g)	Total (g)(ex)	Total (g)(ex)/area (cm^2)	Total(g)(ex)/area (cm^2)/ time (days)	total(g)(ex) /area (m2)/ time (days)
S. Alcove	3m	0.2927	0.2368	0.4289	0.3184	1.2768	0.8479	0.04690	0.0010196	10.196
	10m	0.0768	0.0584	0.0936	0.091	0.3198	0.2614	0.01446	0.0003143	3.143
	40m	0.0548	0.0237	0.0592	0.0245	0.1622	0.1377	0.00762	0.0001656	1.656
Channel	1	0.1460	0.0779	0.1078	0.0875	0.4192	0.2732	0.01511	0.0003285	3.285
	2	0.1724	0.2282	0.2358	0.252	0.8884	0.716	0.03960	0.0008610	8.610
	3	0.1407	0.1637	0.1875	0.1653	0.6572	0.4697	0.02598	0.0005648	5.648
Downstream	3m	0.2722	0.1405	3.3792	0.2055	3.9974	0.6182	0.03420	0.0007434	7.434
	10m	0.0782	0.0552	0.1158	0.053	0.3022	0.1864	0.01031	0.0002241	2.241
	40m	0.039	0.0415	0.0268	0.045	0.1523	0.1255	0.00694	0.0001509	1.509

Table 3.1: In this table is all the sedimentation data collected over the duration of the study. The values highlighted in yellow were excluded based on a visual inspection of the sediment trap and its contents. Test tubes were excluded and marked "ex" if they exhibited any abnormalities when compared to the replicates that were at the same site.

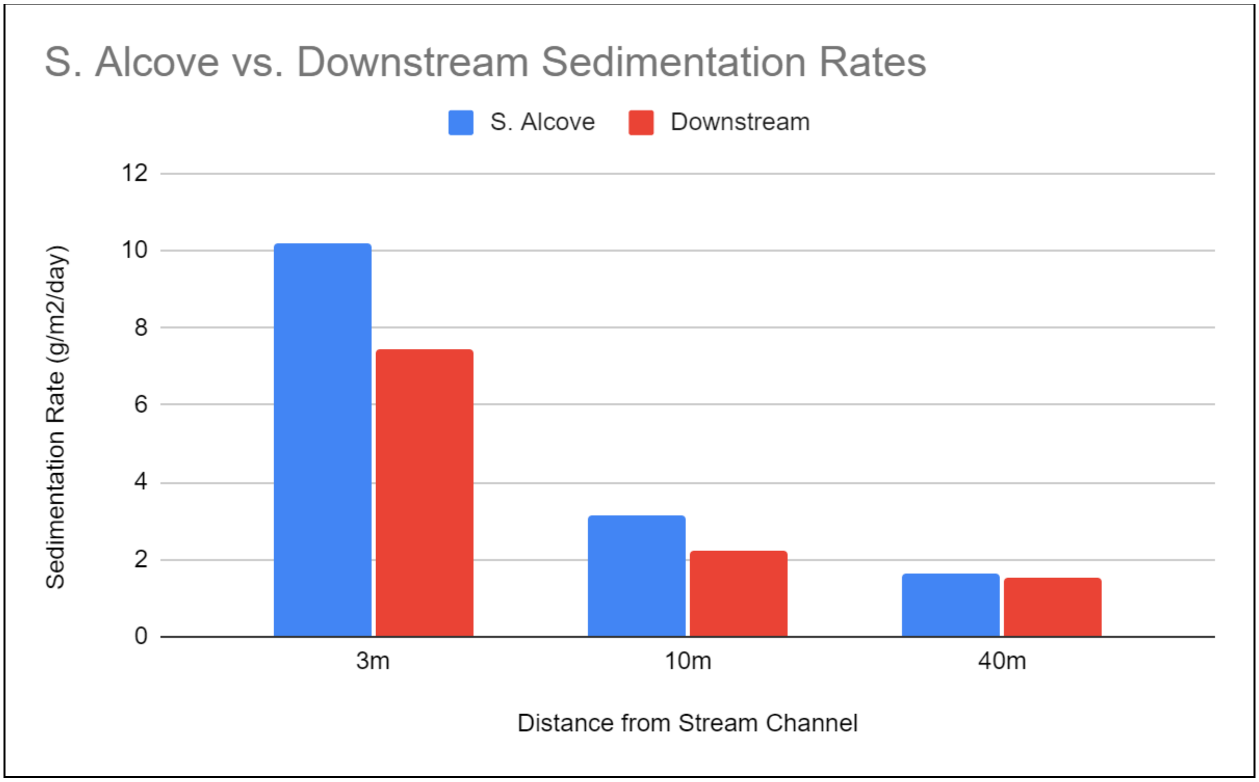

Figure 3.5: shown are the sedimentation rates collected from the transect within the S.Alcove and the one downstream of the ditch plug perpendicular to the channel.

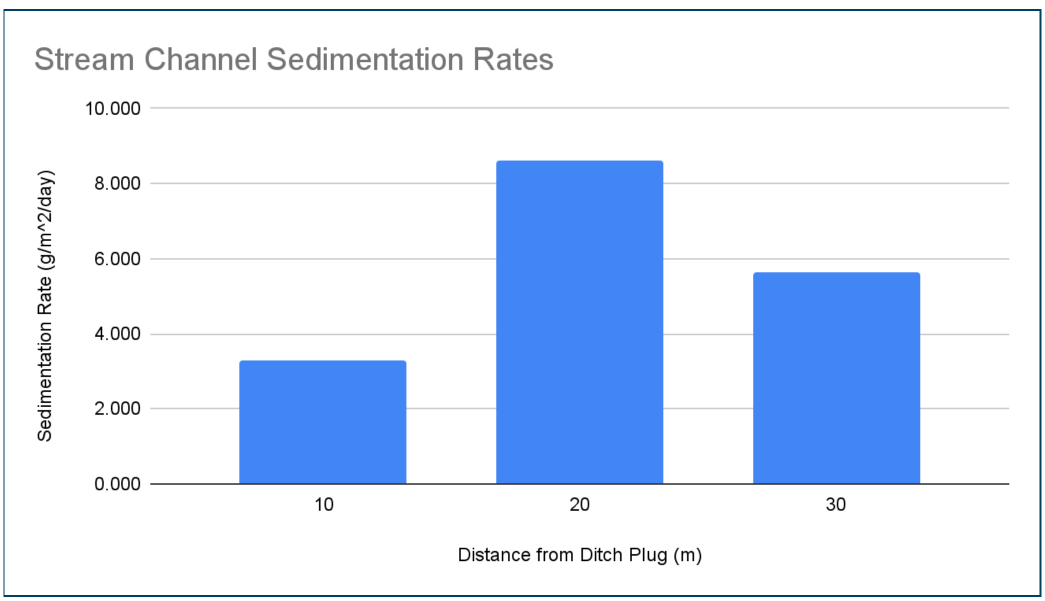

Figure 3.6: shown are the sedimentation rates collected from the transect that ran parallel to the downstream channel at a distance of 3m.

3.3 Sediment Core Data

To begin with the sediment cores the first piece of data looked at were the core descriptions (figure x). Core #1 was a total of 60cm in length and contained roots and rhizomes throughout the entirety of it. The roots within the first 0-22cm were primarily white while the roots from approximately 20-60cm were brown and yellow. The composition and color of the core also changed throughout the depth. The sediments at depth of 1-8cm were Munsell 2.5yr 3/1, with very fibrous white roots (<1mm) interwoven between and was composed of a very fine silt. From 8-12cm the color was 2.5yr 2.5/1, contained slightly thicker white roots and was also composed of a very fine silt. Depths 12-15.5cm were 5YR 4/1 and contained thick white roots and rhizomes (1-5 mm). From 15.5-31cm the color was 7.5YR 2.5/1 and the roots transitioned from the thick white roots to that of much thinner (<1mm) and densely weaved yellow and brown roots. From 31-45cm there is a sandy silty layer where the color is 7.5YR 3/2 and the roots are fibrous and yellow brown. 45-56cm the composition is very silty and has a color of 7.5YR 2.5/1 and has a very similar root coverage as the above section. The last 56-60cm were 7.5YR 4/2 in color and had similar root coverage as the above section.

For Core #2 there were fewer sections to denote as compared to core #1. The first 0-19cm were a 2.5YR 2.5/1 in color that was silty in composition with very stringy loose roots and some woody roots. The next section was at depths of 19-36cm and had a color of 2.5 YR3/2 with a silty composition. From 36-45cm the color was 5YR 3/1 and had a similar texture to that of the above section. The final section of Core #2 was at a depth of 45-65cm and was 7.5YR 4/1 in color with a very sandy composition.

Both cores along with being fully visually observed were also sampled and tested for %LOI, Bulk Density, and $\%_{org}$Carbon. Beginning with the dry bulk density of the cores we can see that the majority of core #1, aside from 10-15 cm in depth, is primarily in the range of 0.1-0.3 g/cm^2. At the 10-15cm depth there is a spike in bulk density to a value of 1.0 g/cm^2. But aside from this the majority of the core is between 0.1-0.3 g/cm^2. Core #2 follows a similar trend in that many values fall within a range of 0.1-0.3 g/cm^2 and there are some very distinct peaks. These peaks are at depths of 20cm and 45cm and have values of 1.3 and 1.1 g/cm^2 respectively. The end of the core also does something different from core #1 in that at a depth of 65cm the bulk density starts to greatly increase sharply to a final density measurement of 1.6 g/cm^2.

The bulk density data by itself is very useful, but when paired with the results from the %LOI test (figure 3.x) we are able to calculate the $\%_{Org}$Carbon ($\%_{org}$C) at each depth of the core (shown in figure 3.x). Core 1 starting at 0cm of depth has the highest recorded $\%_{org}$C of the test at 33$\%_{orc}$C. Going down the core the value drops significantly at depths of 10-15cm to a low of 1.1$\%_{orc}$C. The values then again peak back at a value of 29$\%_{orc}$C before steadily decreasing for the remainder of the depth of the core. For Core #2 the trend was slightly different in that at no point did the $\%_{orc}$C ever get to the same peak values. The core at a depth of 0cm had values of 22$\%_{orc}$C, before dipping to 11% at a depth of 5 cm, and then rising back to a high of 27$\%_{orc}$C from 10-15cm. The levels then severely dropped off again at a depth of 20cm to 3.7$\%_{orc}$C, to again increase to a level of 25$\%_{orc}$C at a depth of 25cm. Levels then drop a little over the next 15cm, before sharply decreasing one last time to a value of 3.1$\%_{orc}$C. Values then increase one last time to 17% before gradually decreasing until the end of the core.

From these $\%_{orc}C$ values we were then able to calculate the average bulk carbon density down the length of each core. Core #1 has an average bulk carbon density of 0.036±0.011 $\frac{gC}{cm^2}$ whereas Core #2 has an average bulk carbon density of 0.039±0.008 $\frac{gC}{cm^2}$. These values were found by getting the carbon density along the length of each entire core and then averaging them all together.

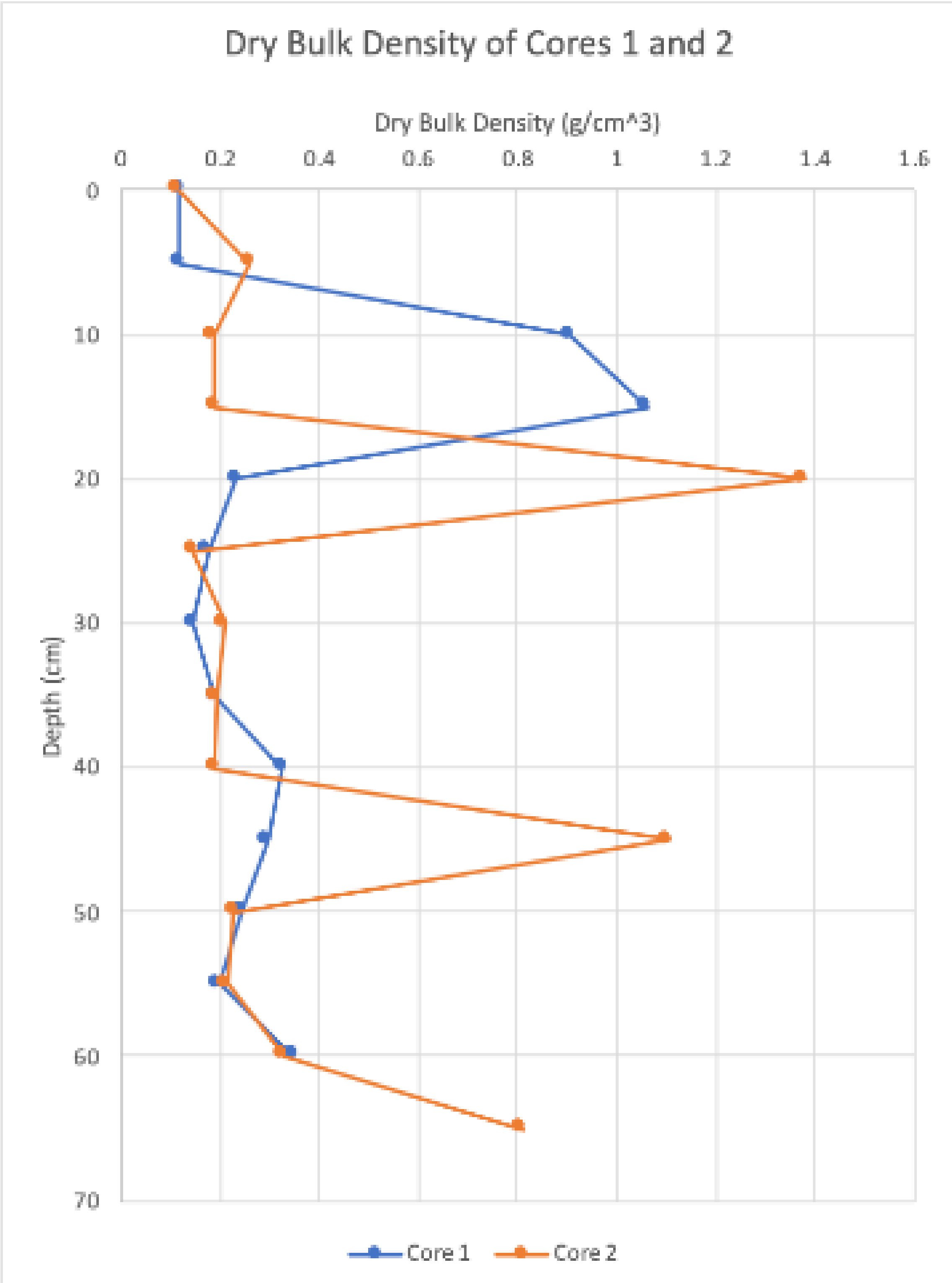

Figure 3.7: Shown are the dry bulk densities ($\frac{gC}{cm^2}$) going down in depth (cm) of Cores #1 and #2.

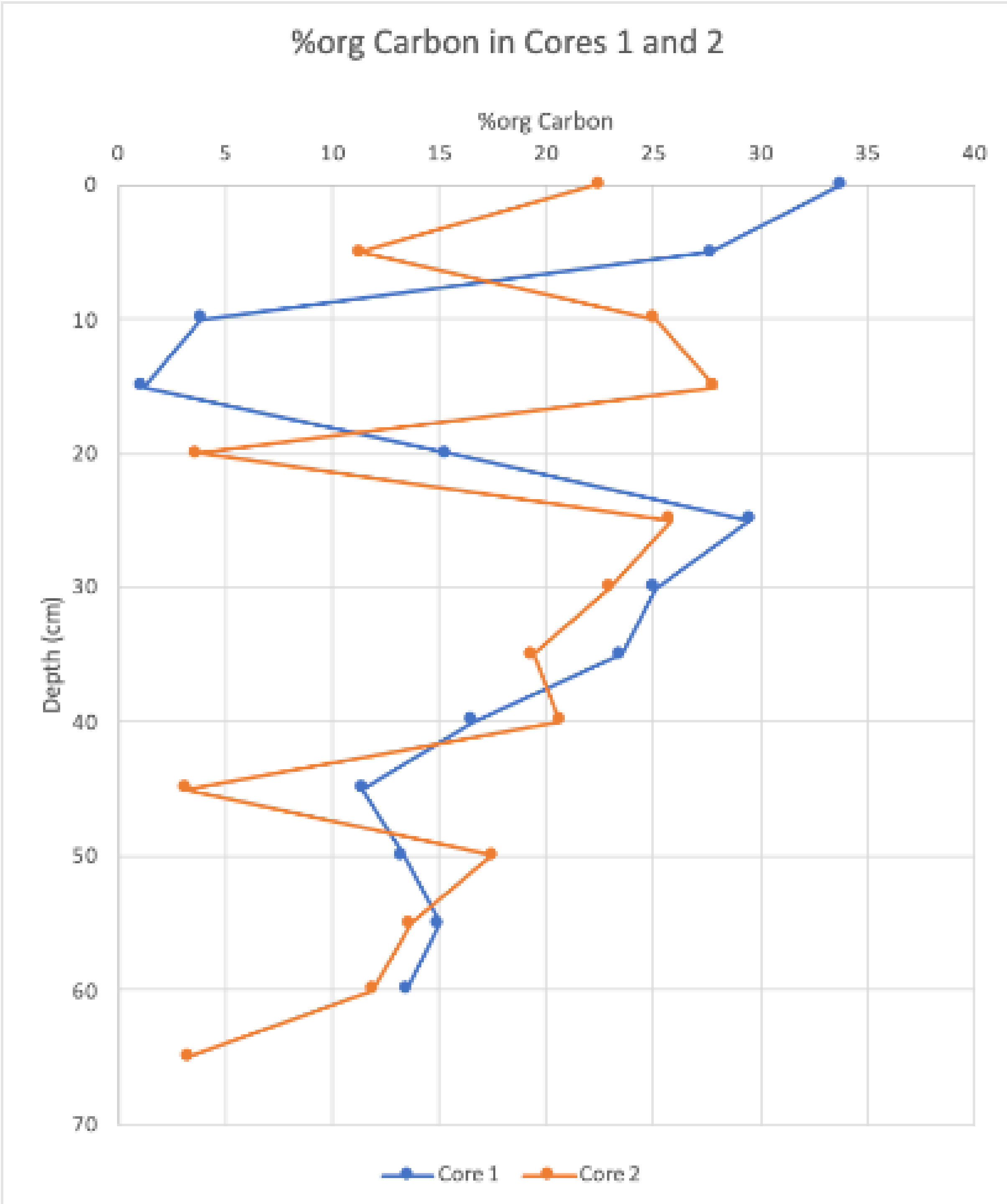

Figure 3.8: Shown is the $\%_{org}$Carbon going down in depth (cm) of Cores #1 and #2.

4.1 Tidal Sequence

Within the S. Alcove, there is a muted tidal sequence when compared to both tidal signals from the downstream area of Sprague Marsh and that of the stream channel going into the S. Alcove. This is shown in the much lower fluctuations in daily water table height as shown by the water pressure sensors. This is quite common when tidal restrictions are in place (Burdick et al., 1996).

As shown by the USGS Vertical Wetland Development model we can see that the "altered river flow" as caused by the ditch plug is indeed having a significant impact upon the flooding depth and duration of the S. Alcove. The marsh up gradient of the ditch plug is very wet, and does not drain well as observed while in the field. When looking at other examples of marshes with reduced tidal signals in past research it is not uncommon to see a strong correlation between changing vegetation and altered hydrology (Barry 2011; Vincent et al., 2014). Overall the specific change in vegetation zonation is dependent upon a variety of other factors including elevated atmospheric CO_2, salinity, and other outside disturbances. Again, while the altered hydrology is not the only thing affecting the vegetation zonation, it has been shown to be highly correlated and potentially even one of the most important factors towards zonation. These other waterway alterations include sites such as undersized culverts, roadways, and of course ditch plugs.

There were multiple shortcomings in the data collection techniques and results that contributed to a less than satisfactory look at the hydrology of the marsh. The hope was to be able to recreate the slope of the water table within the S. Alcove and therefore drainage direction of the area. This information would have been beneficial to fully understanding the hydrology of the area as it would more clearly show where the inputs were coming from and how the water was inundating the area and flowing back out. This was done in Collin Barry's thesis (2011), and at the time it suggested a slope that was dipping towards the sandy section to the southeast of the S. Alcove. It wouldn't be a far stretched extrapolation to say that this is still the case as there has been no significant visible change in the area, though it would have been helpful for long term monitoring efforts to have re-gathered the information 10 years after the Barry study.

It would have also been helpful to have gotten clear conductivity data. If clear conductivity data were to have been gathered it would have allowed one more insight into the impact of marine inundation vs freshwater from the marsh margins in the area. As shown in figure 1.4 salinity is one of the important variables to wetland development and therefore should be looked at in order to have a clearer understanding of the processes happening within the system. In the future pore water salinity should also be sampled around the site in order to map the area's salinity in some sort of resolution.

This altered tidal sequence could have a multitude of cascading effects on the marsh. These cascading effects could include a change in vegetation zonation, biomass accumulation, and the overall sediment accumulation. Each of these effects in turn could have their own cascading effects leading to a significant change in the marsh ecosystem. These could include changes in nekton usage, migrant bird population, and susceptibility to invasive species.

The altered tidal sequence in addition to changing the flooding depth and duration also changes the total amount of water flowing into the area. This change in total water flow consequently could have an effect upon the total sediment deposited. While it is uncertain whether this is a correct correlation, the sedimentation rate results do show an increase in deposition in the S. Alcove. This will be discussed in further detail in the next section, but at current it is unknown what the source of increased sediment is. My hypothesis is that there is a strong correlation between the altered tidal sequence from the ditch plug and the increased sedimentation rate.

4.2 Sedimentary Processes

The sedimentation rates shown in figure 3.5 show the daily average sedimentation rate collected over the time period that the sediment traps were out. Greater sedimentation rates closer to the stream channel in both transects suggests that marine sediments are an important component to the overall sedimentation on the marsh. Higher rates of sedimentation behind the ditch plug are surprising, given that inundation by the tidal stream is muted here. It is probable that the higher sedimentation rate behind the ditch plug represents increased input of organic matter. Microbial mats are much more abundant behind the ditch plug and are visible in the sediment traps. More data is needed to corroborate this interpretation.

This daily sedimentation value is important for determining the marshes susceptibility towards climate change (Vincent et al., 2013). If the sedimentation rate is not great enough then eventually sea level rise will catch up and overtake the wetland development processes. According to the nearest long term tide gauge the yearly sea level rise in Boston averaged about

3.4mm (NOS, CO-Ops; https://tidesandcurrents.noaa.gov), and while Boston does not share the same coastal features as Maine, it does work as a decent analog for sea level rise. If sea level rise in the area were to over take the sedimentation rate in the area the S. Alcove would likely be one of the most susceptible types of areas as it is an already stressed coastal environment. The collection methods do a good job at gathering seasonal sedimentation rates. What this means is that while we are able to get a pretty accurate approximation of the daily average sedimentation rate in the summer months that experiment was being run, these tests do no not reflect a year long daily average sedimentation rate making it hard to scale up and compare them effectively. However this doesn’t mean that the data was all for nought where the data lacks in its ability to scale up, it is fairly accurate at a daily resolution as the time period the tubes were out was so short. The data in figure 3.x shows that the sedimentation rates within the S. Alcove were across the board higher than those in the area downstream of the ditch plug which were anywhere from approximately 10% to 25% lower. This was surprising as lower rates within the S. Alcove was what I was expecting based on research saying that ditch plugged areas had lower surface elevation height. As I know that surface elevation and sedimentation rates are closely linked I had predicted that if surface elevation was lower, the sediment input must have also been lower and not able to uphold the erosional equilibrium keeping the marsh stable. This however did not seem to be the case.

Since the sedimentation rates within the S. Alcove are indeed higher than those of the area downstream; it is not far fetched to reason that the increased sedimentation rate may not have to do with the marsh elevation, but instead with the altered tidal sequence in the area caused by the ditch plug. My theory currently is that this altered tidal signal leads to an increase in surface water on the marsh during flooding, and thus an increased surface water sediment load being deposited. While the data collected in this study is not enough to discern whether this is the case or not, I do think that the ease and efficiency of this method would be effective at figuring it out. Especially with the ability for the method to scale up with more transects and distances, this test tube method has lots of potential for being able to map sediment distribution patterns during the time of the year when the marsh is not covered in snow. In scaling up the collection size and monitoring duration the data would more than likely show better trends and show a clearer picture of how sediments are actually being deposited.

In addition to being a great method to scale up for mapping purposes the test tube technique also has the added benefit of being indiscriminate in the type of sediment that it gathers. This means that if it were desired the sediments of each test tube could be tested for sediment size and composition. This would give insights into the source of the sediments, whether they're of mineral or organic, as well as the grain size. All of these would be important factors to understanding in totality the sedimentary processes in play within the S. Alcove. Unfortunately due to the scope of this thesis these tests were not able to be run upon the sediments collected. However due to the freeze drying process the sediments are preserved and are able to be tested if the need arises in the future. However I think it would be much more beneficial for the data to be recollected so as to have another set of replicates as well as a better picture of the daily sedimentation rates over different parts of the year.

The next important piece of information to look at within the sedimentary process is the data pulled out of the sediment cores from the S. Alcove. Within this data there are a couple key points of interest. The first of these is the comparison of the 2 cores gathered in this study to a core gathered by another senior thesis student in a less tidally restricted area further north in Sprague marsh near SET #3 (Meg O'Brien, pers com). All cores stayed generally in the range of 0.1-0.3 g/cm^2 in dry bulk density, and showed similar colors on the Munsell soil color charts, but that is where the similarities ended. In the 2 cores collected within the S. Alcove there were peaks 2 peaks in each core that when looked at in isolation looked as if they could be outliers, potentially from poor sampling procedure. But when looked at in tandem with the physical core description it was observed that at each peak within the bulk density and decrease in $\%_{Org}$C there was actually a change in sediment, usually either to a fine silt or a very coarse sand usually found in the back dune area. This leads me to believe that this layer is in fact a minerogenic layer that has occurred due to a change in sedimentation from the ditch plug. This would make some sense as the $\%_{Org}$C is said to increase with the addition of ditch plug (Vincent et al., 2013), meaning that a layer of more carbon rich sediment has been deposited over top of the pre ditch plug layers of more mineral rich sediments. This would likely also explain the increased amount of sedimentation that was noted within the sediment traps. While this is likely the explanation for these peaks and troughs in the bulk density and $\%_{Org}$C as well as the increased sedimentation rate, more testing would be helpful in discerning the source and composition of the sediments.

Overall the sedimentary processes are in line with the literature in having increased amounts of $\%_{org}$C, and slightly lower bulk density up gradient and behind the ditch plug. This is especially apparent when comparing the average carbon densities of our 2 with those found in previous studies (Vincent et al., 2013). Core #1 had an average bulk carbon density of 0.036±0.011 $\frac{gC}{cm^2}$ while Core #2 had a value of 0.039±0.008 $\frac{gC}{cm^2}$. These values are in line with the "Ditch plug" carbon storage values found within figure 4.x. This further backs up the notion that the sedimentary process within the S. Alcove was disrupted with the installation of the ditch plug, and that these results are similar to results obtained by other researchers.

4.3 Status of the Marsh

Based on the information gathered about the S. Alcoves hydrology, sediment deposition, and subsurface carbon levels, there are a couple of things that can be gleaned. The tidal signal and therefore hydrology of the S. Alcove is significantly muted in comparison to that of the marsh downstream of the ditch plug. It is known that this decreased tidal signal can have a variety of negative effects on the ecosystem of the marsh (Vincent et al., 2013). In Table 4.1 it is shown that when compared to natural pooled environments ditched salt marshes have increased water level, decreased marsh elevation, increased soil salinity, vastly decreased soil redox potential Eh, decreased soil strength, and decreased carbon storage capacity. It was observed while out on the marsh that the peat underfoot within the S.Alcove did not feel as strong or hold its structure as well as the peat in the downstream area of the marsh. While standing in one spot for any amount of time in the S.Alcove it was observed that the observer would sink anywhere from approximately 1-3 cm into the marsh. While not truly quantitative data, this would also fall in line with the decreased soil strength that was noted in the previous studies. This 1-3cm sink is due to the denser root layer at approximately 3 cm of depth (Vincent et al., 2013). In addition to

Habitat	Water level (cm)	Marsh elevation NAVD88 (cm)	Soil salinity (ppt)	Soil redox potential Eh (mV)	Soil strength (kg/cm^2)	Bulk density (mg/cm^3)	Carbon storage (mg/cm^3)	Organic content (%)
Created ditch	−2.52 c	110.98 b	26.17 b	7.85 a	4.51 b	605.01 b	34.17 a	18.97 a
Natural creek	−4.01 c	105.24 bc	26.21 b	52.18 a	4.21 bc	761.76 a	34.08 a	14.67 b
Ditch plug	5.51 a	102.57 c	30.96 a	−165.69 b	3.57 c	550.79 b	28.66 b	17.98 ab
Natural pool	−0.49 b	121.04 a	26.94 b	31.29 a	5.87 a	575.11 b	37.43 a	21.58 a

Comparisons with the same letters are not significantly different according to Tukey–Kramer post hoc tests for each variable ($p<0.05$)

Table 4.1: This table shows the different variables tracked by Vincent et al., 2014 when assessing the effects of ditch plugging on 4 different habitat replicates from three salt marsh environments.

this potential decrease in peat strength of a grayish-pink biofilm that numbed the hands with touch was also noticed upon the outside of the test tubes used to collect sediment samples. While again this is not quantitative data, it is interesting to note that this data would make sense as previous works have observed a higher organic sediment content in ditch plugged areas. This increase in organic sediments could be linked not only to the biofilm, but also could be an explanation for the increased sedimentation rate within the S. Alcove. More research would be needed to determine whether this is truly the case, though it is a curious coincidence. All of this data suggest that the S.Alcove has been negatively impacted by the tidal regime imposed by the ditch plug and that there could be a variety of unforeseen cascading effects that we have not yet noticed.

Though we were not able to measure it in this study, it has been previously found that the surface elevation of ditch plugged salt marshes was found to be up to almost 10 cm lower than natural creek salt marsh environment (Vincent et al., 2013). This would be a great area of further investigation into the status of the S. Alcove as it would give a good insight not only into the surficial processes of the marsh, but also give some insight into whether subsurface peat subsidence is occuring. It would also be worthwhile to further investigate the strength and structure of the peat within the S. Alcove. It has been stated that low soil strength that is associated with ditch plugged salt marshes can contribute greatly to increase the instability of salt marshes and the susceptibility of the area to erosion (Vincent et al., 2013). Both these effects could potentially cause issues if either left unchecked or if a restoration effort is attempted without taking them into account. If left unchecked the decreased soil stability could lead to an even further increased rate of peat subsidence as compared to a natural marsh. If a restoration effort is attempted and the ditch plug is removed without taking these factors into account, the increased susceptibility to erosion as well as the increase in tidal sequence might lead to rapid erosion of the area. For this reason I would suggest that the next studies to focus upon the S. Alcove be focused upon assessing the potential for marsh subsidence and quantifying soil strength data for the entire area.

5.1 Conclusions

In Conclusion I would like to address the 2 guiding questions that I hoped to answer at the start of my research and the start of this project. First and foremost among these is “what is the status of the S.Alcove?”. While there could be many answers to this question I think the most direct answer is that the status of the S.Alcove is poor and it could get worse. The status of the S.

Alcove is poor based on the muted tidal signal, changed sedimentary processes, decreased peat strength, and potential for rapid peat subsidence. The next question to be answered then is my 2nd guiding question of “Should the ditch plug be pulled out?”. This question does not have quite as straight forward of an answer as the last one due to the many viewpoints involved on the subject. And while I personally respect all these views, I will reserve the viewpoint in this paper to one that favors pre anthropogenic processes that can help to maintain the longevity and stability of Sprague Marsh and the S. Alcove.

With this lens for the question in mind I would recommend that the Ditch plug be removed, but only if the proper precautions and monitoring practices are put in place prior to the removal. The first of these precautions would be to assess the peat strength and elevation of the S. Alcove in relation to the area downstream of the ditch plug. This would help to inform the project management as to the true viability of the removal of the ditch plug, as a decreased soil strength increases the soil instability and susceptibility to erosion. If these were not taken into consideration when planning the ditch plug removal, the entire area could wind up getting eroded away because the tidal signal returns too abruptly for the peat structure to maintain. The next step would make sure that clear post project management and monitoring are in place. This is immensely important, not only to the health of the S. Alcove, but also to the future of climate sciences and ditch plug management research. 2 of the most important things to the future of coastal research are the continuation of long term monitoring and the site management that comes along with it (Waltham et al., 2021). If the ditch plug is pulled this data should not be overlooked as it would be a perfect example for habitat restoration monitoring since it is approximately only 1 hectare large. This long term monitoring and management would likely require quite a bit of labor, which brings me to the last precaution. Salt marsh restoration is expensive. as well as all the labor involved with the monitoring and management, you would need to hire a crew to come remove the plug. All of these costs add up to be a lot, and so I think it is important to not only take into consideration the S. Alcove, but also other parts of the marsh that could use the money. As long as these steps are taken I think that it is a good idea to remove the ditch plug.

In addition to the answer to the 2 guiding questions I also like to briefly add in my notes for future research in the area to the conclusion. I concluded that more groundwater data is needed within the S. Alcove to get a better understanding of what is happening with the hydrology. Salinity data at various points would also be an ideal piece of information to have, so a repeat of the groundwater study could be beneficial, though I would suggest new positions are chosen around the Alcove. It would also be very beneficial to repeat the sediment trap test, but with an increased amount of transects going across the marsh and increased length of the transect. If this were done properly the results would yield a sediment distribution map of the marsh, which is something that could be very helpful to determining the effects of climate change. Lastly for future work what could be done is grabbing a few more sediment cores from within the S. Alcove. These would help to discern what the composition of the peat was like throughout the entire depth of the core. In doing each of these things one would surely be adding to the body of knowledge on ditched salt marsh propers.

While much of the stuff talked about here are ideas about future work, I would like to think of my thesis as a step towards salt marsh management and research techniques going forward. Therefore I would like to get down all my thoughts on the matter so as to not only remember them myself, but also make them known to others that seek them out. These thoughts, at least in my opinion, are the most important thing for myself to have gotten out of this project. The marsh has not only endlessly fascinated me, gotten me to wake up in the morning, and provided me with many answers over the course of the last couple months, but it has more importantly brought so many different questions forth in my mind about the dynamic processes of the salt marsh and their susceptibility to climate change.

References

Adamowicz, S. C. and C. T. Roman (2002). "Initial ecosystem response of salt marshes to ditch plugging and pool creation: experiments at Rachel Carson National Wildlife Refuge (Maine)."

Barry, C. (2012). A Study of the Impact of Ditch-plug Alteration on the Hydrogeology of the Sprague River Marsh, Phippsburg, Maine. Department of Geology. Lewiston, Bates College: 113.

Burdick, D. M., et al. (1996). "Ecological responses to tidal restorations of two northern New England salt marshes." Wetlands Ecology and Management 4(2): 129-144.

Elsey-Quirk, T. and S. C. Adamowicz (2016). "Influence of Physical Manipulations on Short-Term Salt Marsh Morphodynamics: Examples from the North and Mid-Atlantic Coast, USA." Estuaries and Coasts 39(2): 423-439.

Gedan, K., et al. (2011). "Uncertain future of New England salt marshes." Marine Ecology Progress Series 434: 229-237.

Howe, A. J., et al. (2010). "Response of estuarine wetlands to reinstatement of tidal flows." Marine and Freshwater Research 61(6): 702-713.

Jacobson, H. A., et al. (1987). "Distribution and Abundance of Tidal Marshes along the Coast of Maine." Estuaries 10(2): 126.

Kelley, J., et al. (2010). "Drowned coastal deposits with associated archaeological remains from a sea-level slowstand : Northwestern Gulf of Maine, USA." Geology 38: 695-698.

Kelley, J. T., et al. (1988). "The Morphology and Origin of Salt Marshes along the Glaciated Coastline of Maine, USA." Journal of Coastal Research 4(4): 649-666.

Kennish, M. J. (2001). " Coastal Salt Marsh Systems in the U.S.: A Review of Anthropogenic Impacts." Journal of Coastal Conservation 17(3): 731-748.

Knott, J. F., et al. (1987). "Hydrologic parameters of salt marsh peat." Hydrological Processes 1(2): 211-220.

McLeod, E., et al. (2011). "A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO 2." Frontiers in Ecology and the Environment 9(10): 552-560.

Montalto, F. A. and T. S. Steenhuis (2004). "The link between hydrology and restoration of tidal marshes in the New York/New Jersey Estuary." Wetlands 24(2): 414-425.

Nolte, S., et al. (2013). "Measuring sedimentation in tidal marshes: a review on methods and their applicability in biogeomorphological studies." Journal of Coastal Conservation 17(3): 301-325.

Sive, H. R. (2012). The Structure and Stratigraphy of Small Point, Maine: D3 Inverted Transtensional Map-Scale Folds of Neoacadian Provenance. Geology, Bates College. Bachelor of Science: 84.

Taylor, P. H. (2008). Salt marshes in the Gulf of Maine: Human impacts, habitat restoration, and long-term change analysis, Gulf of Maine Council on the Marine Environment.

Vincent, R. E., et al. (2013). "Ditching and Ditch-Plugging in New England Salt Marshes: Effects on Hydrology, Elevation, and Soil Characteristics." Estuaries and Coasts 36(3): 610-625.

Vincent, R. E., et al. (2014). "Ditching and Ditch-Plugging in New England Salt Marshes: Effects on Plant Communities and Self-Maintenance." Estuaries and Coasts 37(2): 354-368.

Waltham, N. J., et al. (2021). "Tidal Marsh Restoration Optimism in a Changing Climate and Urbanizing Seascape." Estuaries and Coasts 44(6): 1681-1690.

White, D. A., et al. (1978). "Productivity and Decomposition of the Dominant Salt Marsh Plants in Louisiana." Ecology 59(4): 751-759.

Wood, M. E., et al. (1989). "Patterns of Sediment Accumulation in the Tidal Marshes of Maine." Estuaries 12(4): 237.