The biogeochemistry of a restoring macrotidal salt marsh: Cheverie Creek, Nova Scotia

By

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Abstract

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Vegetation, hydrology, sediment characteristics, and soil chemistry were studied at Cheverie Creek Salt Marsh Restoration Site, NS. Sampling was conducted during the spring and summer months of 2014 to determine how hypertidal minerogenic salt marshes influence aboveground biomass production over the growing season. Aboveground biomass, sulfide concentration, salinity, and redox potential measurements were taken at each sampling location approximately every 2 weeks throughout the growing season. Sediment cores were taken once at each location to determine bulk density, organic matter, water content and grain size. Inundation frequency and duration were determined throughout the sampling period. Hydrology measured by water level recorders was found to influence salinity and redox potential, whereas sulfide concentration increased throughout the growing season. Sediment characteristics and soil chemistry were found to influence aboveground biomass production throughout the growing season. Areas surrounding pannes were associated with low aboveground biomass, highest salinity, high sulfide and low redox potential.

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Table of Contents

List of Figures

List of Tables

Equations

$$Percent\ Water\ Content = \left(\frac{wet\ weight\ (g) - dry\ weight(g)}{wet\ weight\ (g)}\right) \times 100 \quad (1)$$

$$Dry\ Bulk\ Density = \frac{net\ dry\ weight\ (g)}{volume(ml)} \quad (2a)$$

$$Volume = \ length\ of sediment\ segment\ (cm)\ \times radius\ of\ tube\ (cm)^2 \times\ \pi \quad (2b)$$

$$Percent\ Organic\ Matter = \left(\frac{dry\ weight\ (g) - ash\ weight(g)}{dry\ weight\ (g)}\right) \times 100 \quad (3)$$

$$Sand{:}\ Silt/Clay\ ratio = \frac{Portion\ retained\ by\ sieve}{Portion\ in\ pan} \quad (4)$$

$$Sulfide\ Concentration\ (mM) = \frac{Absorbance_{sample} - \ Absorbance_{colour\ blank}}{0.542} \quad (5)$$

$$Dry\ Biomass\ (\frac{g}{cm^2}) = \frac{Dry\ Weight\ of\ Vegetation\ (g)}{625\ (cm^2)} \quad (6)$$

$$Inundation\ Frequency\ (\%) = \frac{Count\ of\ high\ tides > elevation\ of\ sample\ point}{total\ of\ high\ tides} \quad (7)$$

$$Mean\ Inundation\ Time\ (min) = \frac{Hydroperiod\ (\text{min})}{Inundation\ Frequency\ (\%)} \quad (8)$$

$$Hydroperiod\ (\text{min}) = (Count\ of\ high\ tides > elevation\ of\ sample\ point) \times 5\ min \quad (8b)$$

Chapter 1: Introduction and Literature Review

1.1 Introduction

Wetlands of all types have three features that distinguish them from other ecosystems: 1) Presence of water at the surface or within the root zone for a significant portion of the growing season; 2) Unique soil characteristics and chemistry; and 3) Vegetation that is adapted to wet conditions (Mitsch and Gosselink, 2007; Reddy and DeLaune, 2008). These three variables interact with one another while being influenced by regional factors such as climate, and landscape factors like geomorphology, hydrology and water chemistry (Figure 1.1) (Mitsch and Gosselink, 2007). Salt marshes differ from other wetland ecosystems by the salt water that inundates the site daily, residing in middle and high latitudes (Reddy and DeLaune, 2008) and the halophytic vegetation that has adapted to living in a wet saline environment and enduring freezing temperatures (Butler and Weis, 2009; Mitsch and Gosselink, 2007).

Salt marshes are highly productive ecosystems that lie at the interface between land and ocean (Allen, 2000; Colmer and Flowers, 2008; Townend et al., 2010; Butler and Weis, 2009). These ecosystems are found within low lying areas at an appropriate elevation relative to the current sea level, allowing water to enter and inundate the area (Blum and Christian, 2004). These ecosystems also provide habitats that are used by species that cannot survive within other ecosystems (Allen, 2000; Townend et al., 2010). In addition, salt marshes also provide protection from storm surges and coastal erosion, sequester carbon (Townend et al., 2010; Chmura et al., 2003; Butler and Weis, 2009; Garbutt et al., 2006) and regulate nutrient exchange between ocean and uplands (Kostka et al., 2002).

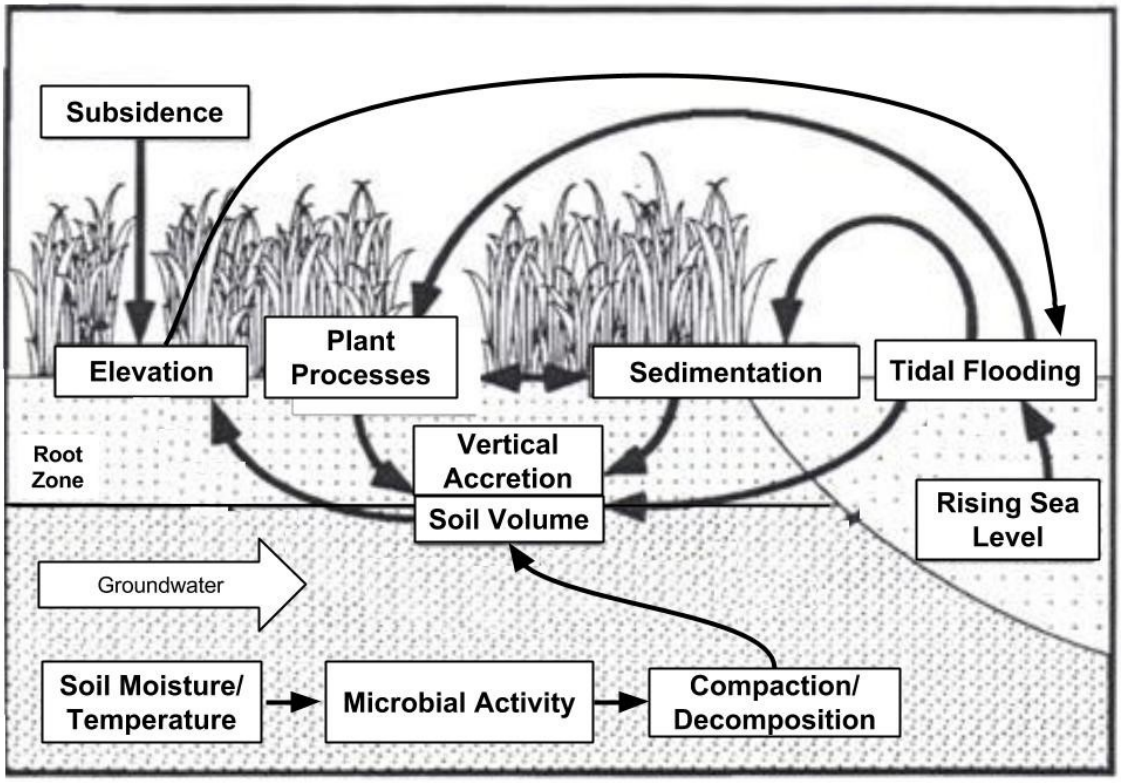

Figure 1. 1. Interactions between tidal flooding, vegetation, sediment characteristics, soil chemistry, microbial activity and groundwater in salt marsh systems (Modified from Cahoon et al., 1999).

1.2. Hydrology

Hydrology is an integral part of salt marshes as it influences the physicochemical environment, transports sediment and nutrients and influences halophytic vegetation patterns and growth (Mitsch and Gosselink, 2007). Salt marshes have a complex subsurface hydrologic system that is determined by topography, rate of sediment deposition, grain size, groundwater discharge, tidal range and wave action (Taillefert et al., 2007). Subsurface water is able to move horizontally and vertically (Taillefert et al., 2007) and is critical to consider since this water influences soil chemistry and vegetation across the marsh surface.

Tidal marshes can experience a wide range of salinity levels ranging from quite saline (30 to 50 ppt) to freshwater (0 to 0.5 ppt) depending upon position in marsh, tidal cycle, and rainfall (Butler and Weis, 2009). The low marsh that is close to the creek edge is flooded more frequently and for longer periods of time compared to high marsh (Townend et al., 2010; Pennings et al., 2005). Therefore, the salinity of the low lying area is similar to tidal waters inundating the site. However, at higher elevations, meteorological conditions play a larger role in the variability of measurements and can cause levels to increase as the sediment dries out (Townend et al., 2010; Pennings et al., 2005). Bertness et al. (1992) found that soil salinity was higher within bare patches than vegetated areas and was higher closer to the surface than subsurface. Salinity was also found to increase over the summer months as the sediment began to dry out (Bertness et al., 1992).

Salt marshes are not only influenced by tidal waters that inundate them; these ecosystems are also influenced by freshwater drainage and groundwater inputs (Reddy and DeLaune, 2008; Wilson and Morris, 2012). Groundwater influences redox potential, saturation, salinity and nutrient cycling that make it an important part of salt marsh ecosystems (Wilson and Morris, 2012). Wilson and Morris (2012) concluded that an increase in tidal amplitude increases the amount of groundwater exchange within the system when the marsh platform is inundated at high tide. However, after the platform has been inundated, any increases in mean water level will decrease flushing of groundwater (Wilson and Morris, 2012).

1.3. Sediment

The substrate of salt marshes varies around the world depending on the sediment source, the sediment supply and the vegetation that dominates the site. Therefore, salt marshes can either be classified as organogenic, where the marsh platform is dominated and sustained by below ground production of organic root matter, or minerogenic, where the marsh platform is dominated

and sustained by high inputs of inorganic sediment (Reddy and DeLaune, 2008). Minerogenic marshes tend to occur where there is a lot of suspended sediment, as in macrotidal estuaries around the world. Sediment deposition is influenced by vegetation on the marsh surface as well as by the flow of the water over this surface. Larger particles fall out of suspension closer to the source of the tidal flow while finer materials travel further into the marsh system as fines have a lower settling velocity (Allen, 2000; Stumpf, 1983). Vegetation that has thick stalks such as *Spartina alterniflora*, reduces the flow of water and causes fine sediment to fall out of suspension more quickly (Bartholdy, 2012). Organic matter remains vital to minerogenic marshes as it is the foundation of biogeochemical reactions within the sediment column. Organic material comes from belowground biomass, decaying aboveground biomass and material attached to fine grained sediments that come in with each tide.

1.3.1. Biogeochemistry

Reddy and DeLaune (2008) define biogeochemistry as "the study of the exchange or flux of materials between living and nonliving components of the biosphere". Biogeochemistry not only encompasses the reactions that occur at a microscopic scale but also the reactions that occur at a global scale. The processes that occur within wetlands at the surface or near-surface layers of sediments, govern the biogeochemical cycles, productivity of plants, microbial transformations, nutrient availability, pollutant removal, exchange between atmosphere, water and sediment, and sediment transport (Reddy and DeLaune, 2008; Pezeshki and DeLaune, 2012). Organic matter is central to biogeochemistry and encompasses nutrients such as nitrogen, phosphorous and sulfur that are important for optimal productivity (Reddy and DeLaune, 2008).

Biogeochemical cycles are regulated by the oxidation and reduction reactions that occur within sediments (DeLaune and Reddy, 2005; Reddy and DeLaune, 2008; Pezeshki and DeLaune, 2012). Oxidation and reduction reactions represent a transfer of electrons either through donating

or accepting an electrons respectively. Wetland soils are dominated by reduced forms of chemical species (Reddy and DeLaune, 2008; Pezeshki and DeLaune, 2012). The reactions are controlled by microbial communities, carbon supply (Teasdale et al., 1998; Craft, 2001; Fieldler, et al., 2007), temperature, pH, and concentration of electron acceptors (Reddy and DeLaune, 2008, Pezeshki and DeLaune, 2012, Tiner, 1991). Microbial communities use organic substances as electron donors (Sanchez et al., 1998; Craft, 2001, Reddy and DeLaune, 2008) to produce the energy that they need. Microbial communities also require an electron acceptor in order to complete the process (Reddy and DeLaune, 2008) and the preferred electron acceptor is oxygen (O_2) (Craft, 2001; Reddy and DeLaune, 2008; Fieldler, et al., 2007; Portnoy, 1999). However, oxygen diffuses 320,000 times more slowly in waterlogged than in gas-filled sediments (Colmer and Flowers, 2008; Koch and Mendelssohn, 1989, Naidoo et al., 1992). Oxygen within the soil is rapidly consumed leading to a high electron pressure or a reduced state (Colmer and Flowers, 2008; Koch and Mendelssohn, 1989; Naidoo et al., 1992). This triggers the need for a microbial community to switch to alternative electron acceptors (Koch et al., 1990, Pezeshki and DeLaune, 2012). The alternative electron acceptors include nitrate (NO_3^-), manganese (IV) oxides (MnO_2), iron (III) oxides ($Fe(OH)_3$), sulfate (SO_4^{2-}), and carbon dioxide (CO_2) in the order which provides decreased amounts of energy that can be obtained from the oxidation of organic matter (Craft, 2001; Fieldler et al., 2007; Reddy and DeLaune, 2008). The reduction of these alternative electron acceptors does not simply reduce the amount of energy that the microbial community is able to access, but also many produce phytotoxins (e.g. hydrogen sulfide) that are detrimental to vegetation growth (Koch and Mendelssohn, 1989). The incomplete decomposition of organic materials can also lead to compounds that are toxic to plants that include lactic acid ($C_3H_6O_3$), ethanol (C_2H_6O), acetaldehyde (C_2H_4O) and aliphatic acids (Anastasiou and Brooks, 2003).

Electron pressures occur as a result of the release of electrons through the oxidation of reduced compounds (Reddy and DeLaune, 2008, Pezeshki and DeLaune, 2012). This electron pressure is not completely removed within wetland systems because of the decreased availability of oxygen. The level of electron pressure within wetland sediments can be quantitatively determined by taking redox potential measurements that indicate the intensity of anaerobic conditions within the sediments (de la Cruz et al., 1989) and characterize the dominant oxidation reduction reaction occurring at the time of measurement (Reddy and DeLaune, 2008). Soil is heterogeneous which allows multiple reduction and oxidation reactions to occur simultaneously within the system (Reddy and DeLaune, 2008, Pezeshki and DeLaune, 2012). Redox potential measurements of field sediments provide an assessment of the interactions between hydrology, microbial activity, rhizome activity, sediment characteristics and the amount of available organic matter and nutrients (Catallo, 1999; Reddy and DeLaune, 2008).

Daily fluctuations in tidal inundation within salt marshes create periods of time when the sediment pore spaces are filled with water leading to anaerobic conditions. Oxidation of organic matter during these periods is believed to occur through the reduction of sulfate which is abundant in seawater, leading to sulfide production (Kostka et al., 2002; Portnoy, 1999; Morris and Whiting, 1985; Howarth, 1984). However, iron reduction under anaerobic conditions is as important in the cycling of organic matter as sulfate reduction (Palomo et al., 2013; Kostka et al., 2002).

1.3.2. Iron and Manganese

Minerogenic marshes, such as those within the Bay of Fundy, may contain high concentrations of iron (Fe^{2+}) and manganese (Mn^{2+}) (Schoepfer, et al., 2014, Reddy and DeLaune, 2008, Hung and Chmura, 2006). High iron and manganese concentrations in the soil help to buffer (poise) the redox potential and sustain redox potential around iron/manganese reduction.

These elements act as buffers because they are quickly oxidized under anaerobic and aerobic conditions by either microbial communities or spontaneously by oxygen and nitrate (Reddy and DeLaune, 2008). The oxidized forms of iron and manganese can be reused as electron acceptors by the microbes compared with oxygen reduction to water, which is irreversible. The buffered redox potential allows microbes to gain sufficient amounts of energy from the decomposition of organic matter, decrease the ability of phytotoxin formation (i.e. sulfide) and, since iron bonds with sulfide to render it inert (Schoepfer, et al., 2014), the resulted decreased sulfide levels can result in increased plant productivity.

1.4. Vegetation

The vegetation communities of salt marshes are dominated by C_4 grasses that are able to use carbon dioxide more effectively than C_3 plants (Burke et al., 2002; Mitsch and Gooselink, 2007). As previously stated, vegetation that survives within salt marshes must deal with the environmental and biological stresses which influence the pattern of vegetation across the marsh surface. Therefore, vegetation types that dominate salt marsh systems are separated into different vegetation zones that vary along elevation (Bertness et al., 1992; Bertness et al., 2002; Butler and Weis, 2009), inundation, and salinity gradients (Cooper, 1982; Pennings et al., 2005; Wang et al. 2010; Janousek and Mayo, 2013). The zonation of vegetation is also influenced by redox potential (Sanchez et al. 1998) and limiting nutrients (i.e. nitrogen: Bertness et al., 2002; Koch et al., 1990).

Salt marsh vegetation directly regulates carbon and nutrient inputs and provides oxygen to the rhizosphere (Seliskar et al., 2002; Bertness et al., 1992). Carbon and oxygen that are supplied belowground regulate the microbial communities that live there (Seliskar et al., 2002). The depth of the root zone depends on the plant community and determines the depth to which oxygen can penetrate the soil (Seliskar et al., 2002). Vegetation also stabilizes sediment and

regulates the amount of sunlight reaching the soil surface (Seliskar et al., 2002; Bertness et al., 1992).

One of the major ecological problems that hinders plant growth is soil oxygen deficiency caused by tidal flooding (Colmer and Flowers, 2008). This affects stomatal openings, photosynthesis, water and mineral uptake and hormonal balance (Tiner, 1991). In order to survive and thrive in periodically flooded soils within wetlands, plants have developed biochemical, morphological and physiological adaptations (Naidoo et al., 1992; Reddy and DeLaune, 2008; Mitsch and Gosselink, 2007). Salt marsh plants have evolved efficient systems to transport sufficient oxygen to their roots to allow respiration and chemical oxidation of phytotoxins (Naidoo et al., 1992; Reddy and DeLaune, 2008; Mitsch and Gosselink, 2007). The formation of aerenchyma tissue is one mechanism that salt marsh plants develop in order to provide oxygen to their roots so that oxygen is able to diffuse from the leaves through the stems into the root zone (Naidoo et al., 1992; Reddy and DeLaune, 2008; Koch et al., 1990).

Although *Spartina alterniflora* has an extensive system of aerenchyma tissue, it may not be enough to deal with the anoxic soil conditions that can arise within salt marsh systems (Koch et al., 1990; Mendelssohn et al., 1981). In order to combat a hypoxic root environment, *Spartina alterniflora* increases the production of an enzyme called alcohol dehydrogenase in order to allow the roots to survive (Koch et al., 1990; Mendelssohn et al., 1981). A decrease in this essential enzyme due to high sulfide levels, leads to reduced ability to take up nitrogen, decreased energy availability, and decreased growth (Koch and Mendelssohn, 1989; Koch et al., 1990; Sanchez et al., 1998).

Within coastal systems, high salinity levels require plant species to develop mechanisms to cope with salt. The mechanisms include barriers to prevent or control entry of salt, accumulation of inorganic ion and organic osmotica, salt secreting organs, and C_4 photosynthesis

(Mitsch and Gosselink, 2007). Barriers are found in root ends to restrict salt to these regions enabling plants to tolerate higher salinity (Mitsch and Gosselink, 2007). Some species such as *Spartina alterniflora*, are equipped with salt secreting organs in leaves that selectively remove certain ions from vascular tissue (Mitsch and Gosselink, 2007; Manousaki et al., 2007); ensuring an appropriate balance of sodium and potassium within the plant (Mitsch and Gosselink,, 2007).

Salt marsh plants need to be able to cope with root anaerobiosis and the production of internally produced phytotoxins and hormones as well as phytotoxins created by external microbial communities (Koch et al., 1990). These include carbon dioxide (CO_2), ethylene (C_2H_4), manganese (Mn^{2+}), iron (II) (Fe^{2+}), sulfide (S^{2-}) and carboxylic acids (McKee and McKevlin, 1993; Greenway et al., 2006). Young short shoots of *Spartina alterniflora* and *Spartina patens* are extremely vulnerable to prolonged flooding as it is difficult for these shoots to access oxygen (Blom and Voesenek, 1996; Colmer and Flowers, 2008). Unfortunately this phenomenon is compounded in *Spartina patens* which has inadequate aerenchyma development, hindering aerobic root respiration during periods of prolonged water saturation (Burdick, 1989; Pezeshki and DeLaune, 1996; Anastasiou and Brooks, 2003).

Anastasiou and Brooks (2003) studied planted *Spartina patens* in a restored wetland to determine the influence of soil pH, redox potential and elevation on the survival of these plants. They found that the survival of *Spartina patens* was not significantly influenced by soil pH but redox potentials below 50 mV greatly affected plant health (Anastasiou and Brooks, 2003). Redox potential and elevation were highly correlated but elevation alone was not a good predictor of plant health (Anastasiou and Brooks, 2003).

1.4.1. Salinity and Sulfide Effects on *Spartina alterniflora*

Salt marshes along the Atlantic Coast of North America from northern Mexico to southern Canada have patches of *Spartina alterniflora* and are usually found along tidal creeks

(Morris, 1980; Travis and Grace, 2010) that are flooded daily by tides (Bertness et al., 2002). Primary production within these salt marshes is dominated by *Spartina alterniflora* (Kostka et al., 2002) which exhibits a variation in morphology and height across the marsh surface (Morris, 1980; Teal, 1962). The variation in height is not accounted for genetically but is influenced by environmental factors including salinity, flooding, sulfide and nitrogen concentrations (Seliskar et al., 2002; Burdick et al., 1989). A short form *Spartina alterniflora* is found within waterlogged areas leading to anaerobiosis stress where ammonium (NH_4^+) levels are higher than in the more productive areas (Mendelssohn, 1979; Koch et al., 1990).

In a culture study, the varying heights of *Spartina alterniflora* was found to be due to nitrogen deficiency brought on by a change in the uptake kinetics (Morris, 1980; DeLaune et al., 1984; Buresh et al., 1980; Koch et al., 1990). Change in the uptake kinetics relates to oxygen deficiency, salinity, hydrogen sulfide, exchange capacity of the soil and diffusion of nutrients through the soil (Morris, 1980; Morris and Whiting, 1985). Overall factors that influence growth and productivity of *Spartina alterniflora* include tidal inundation, salinity, soil redox potential, ion toxicity and deficiencies of nutrients (DeLaune et al., 1984; Howes et al., 1981). DeLaune et al. (1984) were able to determine that the growth and assimilation of ammonium (NH_4^+) by *Spartina alterniflora* was not affected by redox potential but the concentration of sulfide was.

Salt marsh restoration has the ability to drastically alter the biogeochemistry within soil. Portnoy (1999) found that production of *Spartina* increased after restoration in a previously dyked-drained system as compared to a dyked-waterlogged site. These marshes were organogenic in origin (Portnoy, 1999). The biomass production of *Spartina* within the dyked-drained system was 71% higher than in the intact marshes that were dominated by the short-form of *Spartina* (Portnoy, 1999). The sites that were dyked-waterlogged experienced half the rate of biomass production than the intact marsh because of constant anaerobiosis and a decreased level of Fe

available to precipitate the sulfide (Portnoy, 1999; Portnoy and Giblin, 1997a; DeLaune et al., 1987).

Studies have been conducted to determine the effects varying salinity and water levels after disturbance (Baldwin and Mendelssohn, 1998; Bertness el al., 1992). Brown et al. (2006) investigated the effects of salinity on *Spartina alterniflora* in Louisiana marshes to determine the cause of marsh dieback and found extended drought led to high levels of salinity and significant decreases in macro- and micro-nutrient uptake. Chambers et al. (1998) found that nitrogen uptake in *Spartina alterniflora* decreased with an increase in salinity in field studies completed in Connecticut, USA (Figure 1.2).

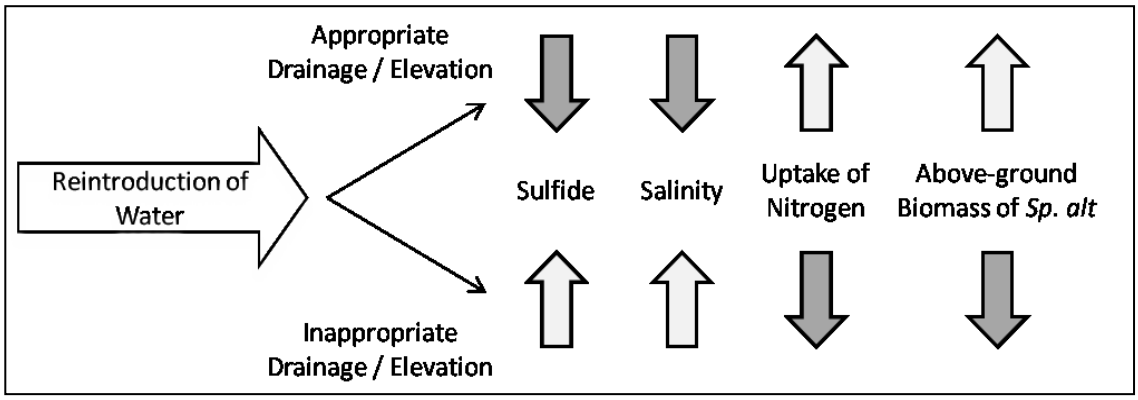

Figure 1. 2 Simplified schematic of interaction of salinity, sulfide, and drainage with the aboveground biomass of *Spartina alterniflora*.

Sulfide is produced within wetlands by sulfate reducing bacteria that use sulfate as a terminal electron acceptor (Lamers, et al., 2013; Koch et al., 1990; Banat et al., 1981). A laboratory study conducted on plants taken from a Louisiana salt marsh revealed an exceedance of 1 mM of sulfide can hinder nitrogen uptake by *Spartina alterniflora* (Figure 1.2) (Koch et al., 1989; Koch et al., 1990). In a laboratory study conducted on plants taken from a salt marsh in Connecticut revealed uptake of nitrogen by *Spartina alterniflora* is unaffected by extremely high sulfide concentration and are comparable to uptake measurements under oxic conditions

(Chambers et al., 1998). Sulfide was removed from the soil as hydrogen sulfide due to bubbling of hydrogen to maintain anoxic conditions (Chambers et al., 1998). Sulfide was replaced throughout the study but only a broad range of sulfide concentration could be maintained as compared to the experiment by Koch et al. (1990) (Chambers et al., 1998). Earlier studies found that nutrient uptake and growth of *Spartina alterniflora* was hindered by, and correlated with, sulfide concentrations (Koch and Mendelssohn, 1989; Mendelssohn and Seneca, 1980; Bradley and Dunn, 1989). The total biomass of *Spartina alterniflora* was found to be reduced under elevated sulfide levels associated with anaerobic conditions (Koch and Mendelssohn, 1989). The highest total biomass production was found under anaerobic conditions that were free of sulfide when compared with aerobic soil conditions (Koch and Mendelssohn, 1989). The variation in biomass production may be accounted for by the depletion of bioavailable nitrogen (ammonium (NH_4^+)) within the aerated soil (Koch and Mendelssohn, 1989).

1.5. Salt Marsh Restoration

Salt marsh systems have been altered for hundreds of years around the world leading to significant loss of species, habitat and productivity (van Proosdij et al., 2010). Human activities have left relatively few surviving salt marshes to ensure protection of landward development from coastal flooding and storms. Coastal managers and scientists have called for restoration of these valuable ecosystems (Bromberg-Geden, et al., 2009) and restoration efforts are on the rise around the world to combat changes in sediment budgets, vegetation, water quality, and loss of habitat for fish and wildlife (Fagherazzi et al., 2004; Portnoy and Giblin, 1997a).

Salt marsh restoration activities include manipulation of hydrology, salinity, sediment, topography, microbial communities and vegetation (Howard, 2010; Zedler, 2006). Restoration of tidal flow in a tide restricted site drastically changes the biogeochemistry of soil that in turn affects the vegetation, nekton and colonization by other wildlife (Anisfeld, 2012). Studies have

been completed to determine the effects that salt marsh restoration projects have on soil chemistry which affects the productivity of salt marsh vegetation (Portnoy and Giblin, 1997a,b; Portnoy, 1999). The return of halophytic vegetation has been found to be variable after tidal restoration and related to distance from seawater entry, salinity and elevation (Smith and Warren, 2012; Smith et al., 2009). The site conditions prior to restoration will ultimately determine how biogeochemical processes will develop and the success of vegetation recolonization.

1.6. Purpose and Objectives

Changes to sediment chemistry following reintroduction of tidal exchange have implications on vegetation colonization and survival. Vegetation assists in the stabilization of the marsh platform and soil matrix. Therefore, it is crucial to understand how sediment chemistry changes in these systems after restoration and how these changes correspond to vegetation recolonization.

Several studies have been conducted on salt marshes throughout New England and the UK (Tempest et al., 2015; Portnoy, 1999; Mora and Burdick, 2013a,b) to determine the influence of soil chemistry, sediment characteristics and hydrology on aboveground biomass production but a study of this sort has not been completed in a hypertidal minerogenic system. The Bay of Fundy is a hypertidal system with a maximum tidal range of 16 m. This increases the reach of tidal water and provides a large influx of suspended sediment in the area (Desplanque and Mossman, 2004; Hinch, 2004). Minerogenic systems may affect soil chemistry differently as compared to organogenic systems (e.g. Atlantic marshes). Therefore, the Bay of Fundy provides a great laboratory to determine how hypertidal minerogenic salt marshes influence above ground biomass production over the growing season. The variables assessed in this study include: 1) Vegetation (aboveground biomass); 2) Hydrology (inundation time, inundation frequency); 3)

Sediment characteristics (bulk density, organic matter, water content, and grain size); and 4) Soil chemistry (sulfide concentration, salinity, redox potential). The objectives of this project were to:

1. Determine an appropriate soil depth for redox potential and salinity measurements;

2. Determine how variations in aboveground biomass are influenced by sulfide concentration, salinity, redox potential, inundation time, and inundation frequency over the growing season; and

3. Determine which of the above factors are significant indicators of aboveground biomass.

Chapter 2: Study Area

2.1 The Bay of Fundy

The Bay of Fundy is a macrotidal estuary located on the East Coast of Canada between New Brunswick and Nova Scotia (Figure 2.1) (Bowron et al., 2009; Davidson-Arnott et al., 2002). Macrotidal estuaries experience a tidal range greater than 4 m (Davies, 1964). The Bay of Fundy can also be classified as a hypertidal estuary because of a tidal range greater than 6 m (Archer, 2013). The Bay developed during the Appalachian orogeny which occurred approximately 286-360 million years ago (Desplanque and Mossman, 2001). The boundaries of the Bay of Fundy today are a result of the formation of a rift valley that created the Atlantic Ocean (Desplanque and Mossman, 2001). The Minas Basin is located within the upper reaches of the Bay of Fundy and is a semi-enclosed remnant of the 200 million year old rift valley (Figure 2.1) (Hinch, 2004).

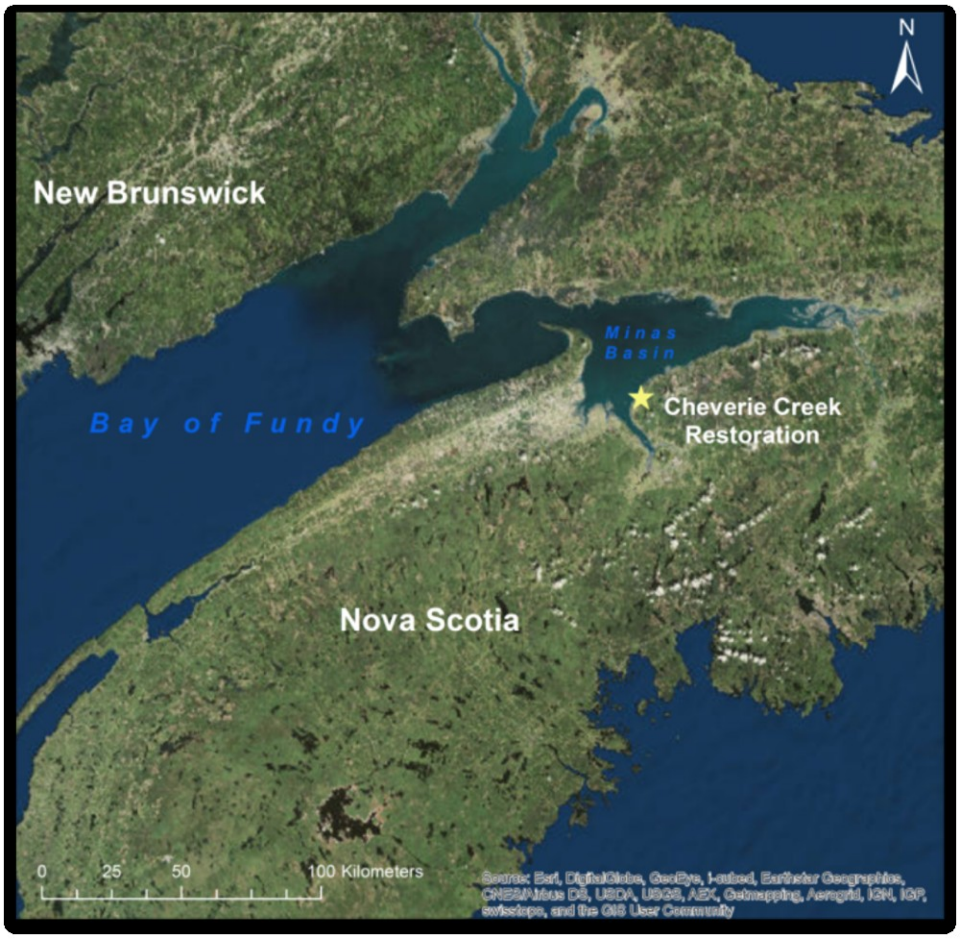

Figure 2. 1 Map showing location of Cheverie Creek Restoration Site (Bing Aerial Imagery)

The Bay of Fundy is a macrotidal estuary that experiences semidiurnal tides with an average of 16 m in the upper Bay (Desplanque and Mossman, 2004; Hinch, 2004). The estuary is dominated by large expanses of tidal mudflats and salt marshes. These systems are minerogenic in origin due to the considerable suspended sediment concentrations from basalt, sandstone and glacial mud deposits (Desplanque and Mossman, 2004) found within the Bay that range from 150 $m \cdot gl^{-1}$ on the marsh surface to 4000 $mg \cdot l^{-1}$ in the upper reaches of the Minas Basin (Bowron et al., 2009).

Approximately 80 to 85% of salt marshes within the Bay of Fundy have been lost due to anthropogenic activities that started with European settlement (Gordon, 1989). The majority of the marshes were converted to agricultural land by draining and dyking salt marsh habitat. Other

changes include construction of tidal barriers and coastal developments. The decrease in salt marsh habitat has led to a significant loss of species, habitat and productivity (Bowron et al., 2009). A loss however that can be partially mitigated by re-introducing tidal exchange through salt marsh restoration practises.

2.2 Cheverie Creek Salt Marsh

Cheverie Creek is a small tidal river located in Hants County, Nova Scotia along the southern edge of the Minas Basin (Figure 2.1). The tidal creek was historically dyked (approximately 200 years ago) and its once tidal floodplains were used for agricultural purposes (Bowron et al., 2009). Remnants of the dyke running perpendicular to the creek are still evident on the site (Figure 2.2). Highway 215 crosses Cheverie Creek and was historically a two-lane causeway with a bridge across the mouth of the main tidal river. The bridge was replaced in 1960 with a causeway and wooden box culvert with a flap gate which completely restricted tidal flow into the site (Figure 2.3) (Bowron, et al., 2009). Upland and freshwater vegetation species including trees, shrubs and grasses encroached onto the site over the last 25 years (Bowron et al., 2009). In the early 1980s, the flap on the box culvert was unintentionally removed which allowed tidal flow to flood approximately 4-5 ha of the marsh (Bowron et al., 2013).

Figure 2. 2 Northern section of Cheverie Creek Restoration Site displaying a portion of the historic dyke in relation to the culvert (C.Skinner, 2014).

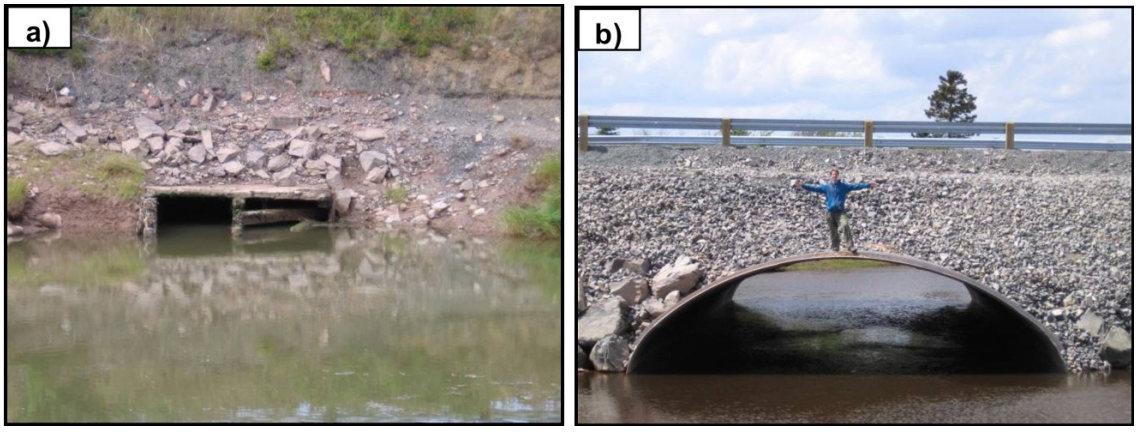

Figure 2. 3 a) Wooden box culvert in 2002 (Photograph T. Bowron, 2002; Used with permission from CBWES Inc.); b) replacement culvert installed in December 2005 (Photograph N. Neatt, 2006; Used with permission from CBWES Inc.)

2.2.1 Pre-restoration Monitoring of Cheverie Creek Salt Marsh Restoration Site

An extensive baseline study was conducted at Cheverie Creek and Bass Creek between May 2002 and October 2004 by members of the Ecology Action Center (Bowron et al., 2013) which included Nancy Neatt who completed her honours research on site (Chiasson, 2003). The baseline study was undertaken to determine conditions experienced at the site prior to restoration to determine if restoration was required and to assist in determining changes in conditions post restoration (Bowron et al., 2013). Bass Creek is an unrestricted tidal river and salt marsh located north of the Cheverie Creek restoration site. As part of the pre- and post-restoration monitoring, conditions at Cheverie Creek were compared to those at Bass Creek (located to the east) to determine the extent of ecosystem degradation and subsequent restoration at Cheverie (Chiasson, 2003). Additional baseline data were compiled by CB Wetlands and Environmental Specialists Inc. (CBWES) in the summer and fall of 2005 prior to culvert replacement in December 2005. A five year post restoration program was planned and conducted by CBWES as part of the restoration but was extended to seven years post restoration to capture longer-term changes in physical and biological components (Bowron et al., 2013).

The monitoring program was based on regional protocols developed by the Global Programme of Action Coalition for the Gulf of Maine (GPAC) for restoration of tidal wetlands within the Gulf of Maine and Bay of Fundy (Bowron, et al., 2009; Neckles et al., 2002; Neckles and Dionne, 2000). The monitoring plan included annual fish surveys, analysis of sediment characteristics (dry bulk density, organic matter, and water content), pore water salinity, sediment accretion and elevation, vegetation surveys, assessment of hydrologic network (hydroperiod and water table depth), water quality (salinity, temperature, dissolved oxygen, and pH), benthic invertebrates, and geospatial components (digital elevation model and habitat map) (Bowron et al., 2013). The monitoring program developed by CBWES was focused on the front 29.5 ha of

Cheverie because this was calculated to be the potential tidal marsh area flooded post restoration (Figure 2.4) (Bowron et al., 2013).

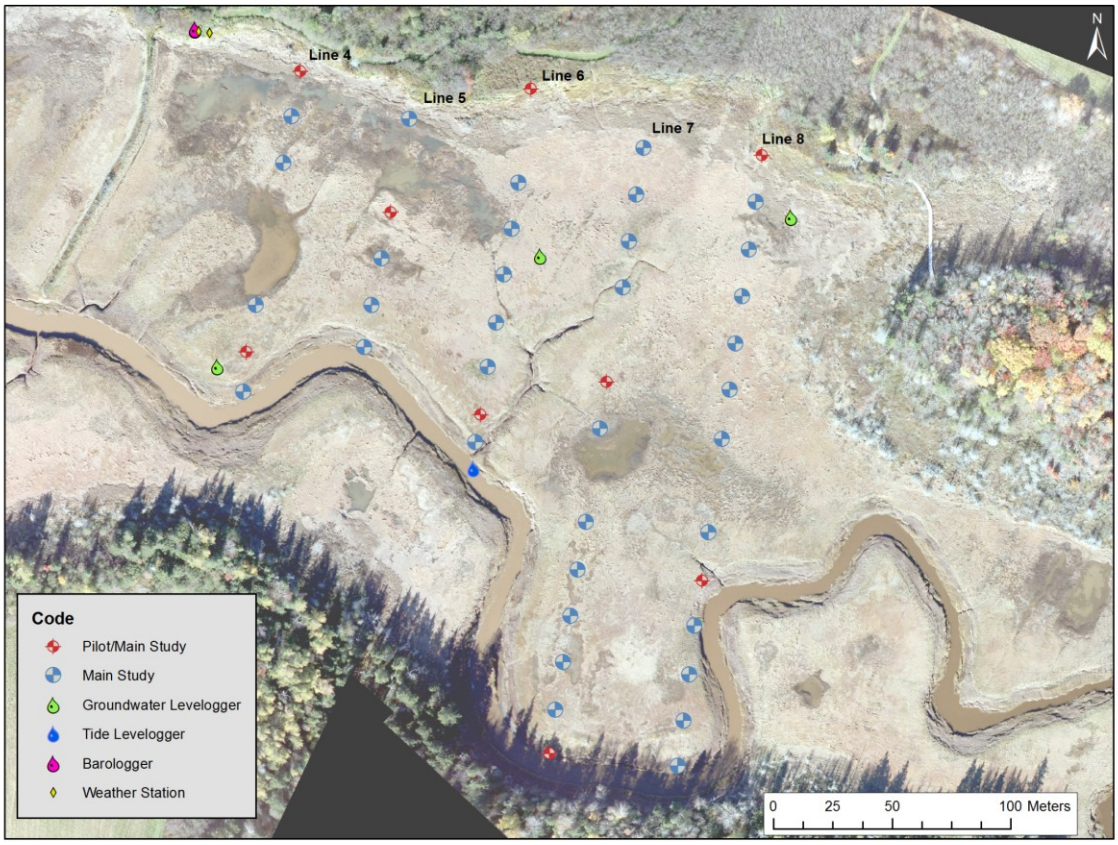

Figure 2. 4 Sampling layout for marsh pilot and marsh extent study and location of ground water wells, tidal levelogger, barologgers and weather station (Aerial Imagery property of CBWES Inc. Used with permission).

The 2004 digital elevation model derived from LiDAR, revealed the potential for restoring 29.50 hectares of salt marsh (Bowron et al., 2009; Bowron and Chiasson, 2006). Prior to restoration, approximately 75% of the marsh was not inundated by tidal water (Bowron and Chiasson, 2006). A hydrological model developed for Cheverie concluded that the existing culvert needed to be replaced with a culvert that was at least 8 m by 3 m that would allow 52% of tides to flood most of the marsh (Bowron et al., 2009; Bowron and Chiasson, 2006).

The baseline study revealed that Cheverie Creek would eventually experience similar conditions to those experienced at Bass Creek and other tidally unrestricted sites. The site was found to be dominated by freshwater and upland vegetation and differences physical attributes (hydrology, soils and sediments) as compared with Bass Creek (Bowron et al., 2009).

2.2.2 Restoration of Cheverie Creek Salt Marsh

A partnership between Nova Scotia Department of Transportation and Infrastructure Renewal (NSTIR), Fisheries and Oceans Canada - Small Craft Harbours (DFO-SCH), the Ecology Action Centre, and Ducks Unlimited Canada enabled replacement of the culvert at Cheverie Creek in the fall of 2005 (Bowron, et al., 2013). A marine habitat restoration compensation bank was also created for NSTIR and DFO-SCH to compensate for projects that interfered with Section 34 of Canada's Fisheries Act, 1985 (Hunt et al., 2011). In 2005, Section 34 of Canada's Fisheries Act, 1985 states "No person shall carry on any work or undertaking that results in the harmful alteration, disruption or destruction of fish habitat" that was commonly referred to as a "Harmful Alteration or Destruction of Fish Habitat" (HADD) (Bowron, et al., 2013). Tidal flow was restored to the site by removing the wooden box culvert (1.5 m by 1 m opening) and replacing it with an elliptical aluminum culvert (9.2m by 5.5 m opening) in 2005 (Figure 2.3) (Bowron et al., 2013, Bowron et al., 2009). This increased the culvert opening 7-fold from 4.7 m^2 to 32.6 m^2 and restored tidal flow to 43 ha of former salt marsh (Bowron et al., 2009).

Bowron, et al. (2013) established four primary goals for the restoration of Cheverie Creek salt marsh. These goals were to:

1. Significantly reduce the tidal restriction caused by the causeway-culvert highway crossing;

2. Improve hydrological conditions upstream of the causeway;

3. Increase the extent and distribution of halophytic vegetation (tidal wetland area); and

4. Improve fish passage to and within the wetland habitat upstream of the causeway.

2.2.3 Cheverie Creek: 7 years post restoration

The final year of monitoring was conducted in 2012, 7 years after restoration. The geospatial surveys completed in 2012 documented differences between Cheverie and Bass Creek (reference site). Bass Creek displayed a general increase in elevation from the creek edge to the upland, and from downstream to upstream (Bowron et al., 2013). This topography creates a habitat with an absence of panne formation. A panne is an area of pooling water within the high marsh that is characterized by high salinity (Marsh and Cohen, 2008). Cheverie Creek displayed topography typical of tide restricted sites and large marsh systems: an increase in elevation from the creek edge to the mid-high marsh, but a decrease in elevation over the remainder of the marsh platform, and then an increase into the upland (Bowron et al., 2013). This topography has resulted in establishment of an extensive panne system and enabled flood waters to reach portions of the high marsh during tides that are otherwise restricted to the main tidal creek.

Analysis of the tide signals from upstream and downstream of the culvert revealed limited tidal restriction by the replaced culvert and restoration of 43 ha of marsh (Bowron et al., 2013). There is restriction on tides larger than 7.11 m Canadian Geodetic Vertical Datum 1928 (CGVD28) but these tides only represent 4% of the recorded tides in 2012 (Bowron et al., 2013).

Pore water salinity was found to be significantly higher post-restoration compared with pre-restoration, and greatest in the high marsh and further upstream from the culvert (Bowron et al., 2013). The increase in salinity across the marsh was expected due to the increase in the frequency, extent and duration of the tidal flooding (Bowron et al., 2013). A salinity gradient, similar to other marsh systems, was evident at Cheverie Creek with a decrease in salinity with distance from the main tidal creek and culvert. An increase in salinity along Transect 5 and 9

(Figure 2.4) were paired with an increase in flooding, die-off of the vegetation and expansion of the salt pannes (Bowron et al., 2013).

Sediment characteristics over the first three years post-restoration, showed a decrease in bulk density and an increase in organic matter content with increasing distance from the main tidal creek. Lower organic matter content and higher bulk density was evident in year 5 and 7 which was also evident at Bass Creek (Bowron et al., 2013). The decrease in organic matter and increase in bulk density at Cheverie Creek is believed to be due to an increase in mineral sediment being deposited on the site created by the expansion of the tidal creek opening resulting in increased tidal exchange. Mean grain size decreased with increasing distance from main tidal creek. Coarse material was evident closer to the main tidal creek and culvert opening while finer material was evident along the upland edge and further away from the culvert opening.

Over the 7 years of post-restoration monitoring, vegetation at Cheverie Creek changed noticeably with the decrease in abundance of early succession plants and an increase in *Spartina alterniflora* that dominated the low marsh and *Spartina patens* that dominates the high marsh (Bowron et al., 2013). *Carex paleacea*, a brackish species, was found to cover a similar area over the years but other brackish species, including *Juncus gerardii* and *Solidago sempervirens* showed decline over the 7 years (Bowron et al., 2013).Vegetation communities at Cheverie Creek shifted to resemble those found at Bass Creek by year 7. Halophytic species richness at Cheverie Creek was still lower than at Bass Creek by year 7 (Bowron et al., 2013). Expansion of pannes at Cheverie Creek led to large areas of vegetation die-back compared with Bass Creek.

Species richness of nekton was similar at Cheverie Creek and Bass Creek in 2012 but with more abundance at Cheverie than at Bass (Bowron et al., 2013). The dominant species at Cheverie Creek was the mummichog whereas the Atlantic silverside had the highest frequency at Bass Creek (Bowron et al., 2013). Due to the increase in the tidal opening and panne size, more

fish species were able to enter into and reside within the Cheverie site. Higher order predators including American eel, Tomcod, and Salmonids species, have been found at both Cheverie and Bass Creek, which shows that these species now have access to Cheverie Creek (Bowron et al., 2013). Overall, by replacing the culvert, the site is trending towards conditions experienced at the reference site. The four goals of the restoration established prior to culvert replacement have been met with the removal of the tidal restriction.

Chapter 3: Methodology

3.1 Sampling Design

The sampling locations used for this study were the same as those established by CB Wetlands and Environmental Specialists Inc (CBWES) for the pre- and post-restoration monitoring that was conducted on the site between 2002 and 2013. By using the established sampling locations, comparisons could be made between results found during this study and monitoring program. Sampling locations were along transects established in a non-biased, systematic, sampling design (Bowron, et al., 2013). The transects were 50 m apart measured from the upland edge running perpendicular to the main tidal creek with sampling stations 50 m apart along each transect. A total of 5 transects (Line 4 to Line 8) (Figure 3.1) encompassing 44 sampling stations were used. Stations were declared lost if they were located along the edge or within the middle of a panne preventing the collection of field data. A Leica Geosystems TCR 705 total station was used to accurately determine the location of each of the sampling points with an accuracy of 5 arc-seconds and distance and elevation accuracy to ± 5mm for site set up on April 17, 2014. The total station was placed in the middle of the remnants of the dyke to ensure full coverage of the study site.

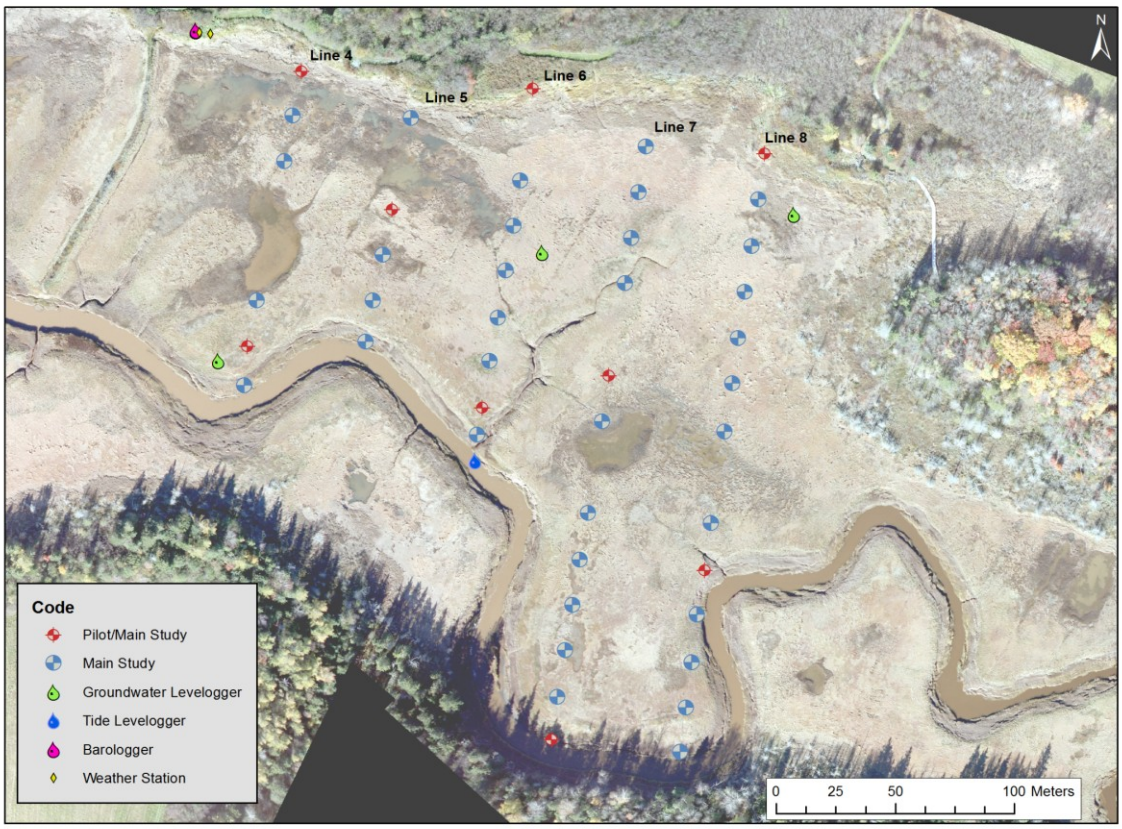

Figure 3. 1 Sampling layout for marsh pilot and marsh extent study and location of ground water wells, tidal levelogger, barologgers and weather station. (Aerial Imagery property of CBWES Inc. Used with permission).

The objectives of this research project were: 1) Determine appropriate depth for redox potential and salinity measurements; 2) Determine how variations in aboveground biomass are influenced by sulfide concentration, salinity and redox potential, inundation time, and inundation frequency over the growing season; 3) Determine which of the above factors are significant indicators of aboveground biomass.

A marsh pilot study was conducted to ensure methodologies were appropriate for Cheverie Creek, to determine variability aboveground biomass and soil chemistry over the growing season and to determine the appropriate depth for salinity and redox potential measurements to be used for the marsh extent study. A marsh extent study was conducted to examine the spatial variability of soil chemistry (sulfide concentration, salinity, redox potential),

sediment characteristics (bulk density, organic matter, and water content), aboveground biomass production, and inundation frequency/ time. The marsh extent study also provided more sampling data to determine which variables correlated best with aboveground biomass production at Cheverie Creek over the growing season in 2014.

The marsh extent study encompassed all 44 sampling stations along five transects (transects 4 to 8). Of the 44 sampling locations in the marsh extent study, 9 locations were randomly chosen for the marsh pilot study. They represented well, moderate and poorly drained sites with at least one sampling station from each transect (4 to 8) that represents the range of conditions experienced at Cheverie Creek. Data collection for the marsh pilot project began on May 21, 2014 during spring tides and therefore approximately every 2 weeks to encompass both spring and neap tides; ending on August 4, 2014. Sampling was conducted during low tide approximately one to two hours after the previous high tide. A total of 3 spring and 3 neap tides were sampled for the marsh pilot project. Sampling for the marsh extent study was conducted on August 14-16, 2014 during spring tides and August 21-22, 2014 during neap tides.

3.2 Field Methods

Meteorological conditions such as temperature and rainfall are known to affect soil chemistry and hydrology. The amount of rainfall can drastically alter the soil chemistry. For example, during drought conditions, soil would be more aerated leading to higher redox potentials and buildup of salt leading to higher salinity values (Reddy and DeLaune, 2008). On the other hand, with higher rainfall, soil can become more saturated leading to lower redox potential and lower salinity levels. In addition, temperature influences the microbial communities that live within the soil, which influences the rate of organic matter decomposition; this in turn influences the soil chemistry (Reddy and DeLaune, 2008; Charles and Dukes, 2009). Therefore, in order to

account for weather conditions, a portable weather station was deployed on site for the duration of the study (Figure 3.1).

Groundwater and tidal signal were measured throughout the duration of the study. Solinist Model 3001 Levelogger Golds were deployed in the main tidal creek along with 3 groundwater wells placed diagonally across the marsh (Level 1 is located closest to upland, Level 2 is middle of marsh, Level 3 closest to tidal creek). The groundwater wells were installed diagonally across the marsh to understand how water varied across the marsh and from creek edge to upland.

Ground water leveloggers were programmed to record data every 30 min. Each levelogger was suspended inside of a stillwell which were constructed of 2 inch diameter PVC pipe as shown in Figure 3.2. A piece of airline wire was used to suspend the leveloggers from the bolt within the stillwell to a depth of 1 m below ground surface. The leveloggers were deployed for the first month and re-checked. A build-up of sediment slurry was noticed inside of groundwater wells located in the middle and upland edge of the marsh. Therefore, each of the groundwater wells were extracted and a pair of panty hose was duct tapped to the outside of the PVC pipe. Stillwells were resurveyed after being reinstalled with the addition of the panty hose. The water was able to move freely into the well while keeping the majority of the sediment outside of the PVC pipe.

Two leveloggers were used to measure tidal signal in the main creek to ensure if one logger failed then data could still be collected for the duration of the study. Each levelogger was suspended within a 30 cm PVC (2 inch diameter) tube with holes drilled throughout to match the pattern drilled into groundwater wells. The PVC tube was mounted onto a metal bracket that was pounded into the middle of the main tidal creek until the point of resistance to ensure that the

bracket and logger could not be pulled out by the strong currents. Leveloggers were programmed to record data every 5 min to capture the rise and fall of the tides throughout the sampling period.

Barologgers were suspended within a 30 cm PVC (2 inch diameter) tube with holes drilled throughout to follow the same pattern of the other stillwells and attached to a metal bracket. These loggers were stationed next to the weather station to record atmospheric pressure and temperature to be used to compensate the data collected from the leveloggers. One barologger was programmed to record every 5 minutes to match the loggers placed in the main tidal creek and the other was programmed to record data every 30 minutes to match the loggers used to measure groundwater. Data were downloaded from the leveloggers and barologgers once a month to ensure that loggers were recording water level properly and that none of them had failed. The data collected from the leveloggers was compensated from the data measured by the barologgers to eliminate the influence of atmospheric pressure.

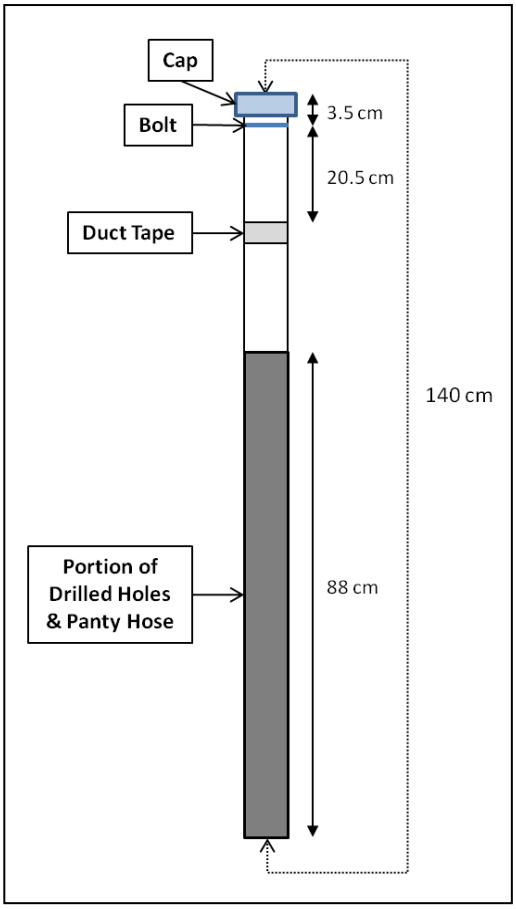

Figure 3. 2 Design of groundwater stillwells.

Organic matter, water content, grain size and bulk density influence redox potential and sulfide concentration (Reddy and DeLaune, 2008; Pezeshki and DeLaune, 2012). Therefore, sediment cores were extracted from each of the sampling locations to determine soil bulk properties as well as bulk root depth. For the purpose of this study, bulk root depth is defined as the portion of the soil matrix sampled that contains the largest amount of roots and root fragments. During the marsh pilot study, two sediment cores were extracted to a depth of 20 cm with a thin walled aluminum tube. The tube was gently hammered into the sediment with the aid of a rubber mallet. A sediment knife was used to cut any roots as the tube was pushed into the sediment. Depth measurements were taken inside and outside of the aluminum core to assess the level of compaction that occurred while the aluminum core was being inserted. A plastic cap was

placed onto the sediment core to minimize contamination during the extraction process. The soil knife was used to cut a triangular chunk of sediment away from the core and around the outside of the aluminum tube. The sediment chunk was removed and sediment core was gently but steadily extracted from the ground. A second plastic cap was placed on bottom of the core to contain the sediment. The sediment core was placed inside of a labelled Ziplock™ bag and placed in a cooler to be transported back to the lab. The cores were stored frozen at -18°C (temperature of standard freezer) until further processing occurred to stop microbial activity and decomposition.

Redox potential measurements were made using 24 constructed platinum tipped electrodes, a calomel reference electrode and milivolt meter. The platinum electrodes were constructed by Christa Skinner and Matthew Skinner based on the design by Vepraskas and Cox (2002) and recommendations by Patrick et al. (1996). Rigid heat shrink tubing was used over epoxy to seal the platinum, brass rod and copper wire to ensure that only the platinum would touch the water/sediment. A standard value of +244 mV was added to each of the field measurements to account for the difference potential of the calomel reference electrode compared with a hydrogen electrode. Redox potential probes were calibrated prior to field data collection in a mixture of pH 4.00 buffer and quinhydrone (Patrick et al., 1996). During the marsh pilot study, two probes were deployed at 5, 10, 15 and 20 cm depths at each sampling station and an average was determined for each depth. The marsh extent study consisted of deploying three probes at 5 cm that was determined to be the bulk root depth (Section 3.3).

Sulfide concentrations were determined in pore water extracted from each sampling station. A "sipper" was constructed from a meter of brake line pinched at one end. A slit was cut into the brake line 1 cm up from the pinched end. Two subsequent slits were cut approximately 0.5 cm up from the first slit. A 20 cm section of rubber tubing was attached to the opposite end of the sipper to allow a syringe to be connected to it. The metal needle was removed from the Luer-

Lok ® connector which allowed the connector to be taped to the rubber tubing. A syringe was connected to the Luer-Lok ® allowing for quick attachment and detachment. The metal tube was inserted into the ground at each sampling location to a depth of 12 cm for the marsh pilot and the marsh extent studies. The contents of the syringe were placed into a 20 ml scintillation vial. A 1 ml syringe was used to transfer 0.5 ml of pore water into another 20 ml scintillation vial that contained 12 ml of 2% Zinc Acetate that was mixed in the lab prior to field work. The vials were labeled and shaken prior to transport back to the lab for further testing. The syringes and the sipper were rinsed with tap water before being used at subsequent stations.

Salinity measurements were taken using a Field Scout soil EC probe for the marsh pilot and marsh extent studies. The probe was calibrated using 2.76 mS/cm conductivity solution at the beginning and periodically throughout the field season to ensure the meter and probe were measuring salinity accurately. For the marsh pilot study, measurements were taken at 5, 10, 15 and 20 cm depths with two replicates. There replicate measurements for the marsh extent study were taken at 5 cm depth previously determined to be the area of bulk root depth (Section 3.3).

The aboveground biomass was assessed for both studies by tossing a 25 cm x 25 cm quadrat into a larger 1 m x 1 m quadrat placed at each sampling location. All of the vegetation within the smaller quadrat was cut close to the sediment surface and retained within a plastic bag and placed in a cooler for transport back to the lab. The samples were stored in the fridge at 4°C to slow decomposition until the samples could be sorted.

3.3 Laboratory Processing

One core from each of the sampling stations for the marsh pilot study were used to determine bulk root depth. Each core was thawed in a refrigerator (4°C) for 24 hours. The core was segmented into 5 cm sections and washed through 1000 µm, 500 µm, 250 µm, and 63 µm sized sieves with a steady stream of tap water. The material that remained on the sieves was

assumed to be the plant material from the roots and was placed in a pre-weighed tin. However, sediment particles larger than 63 μm would have been trapped. After samples were dried at 60°C for 3 days and weighed, these samples were crushed and placed in a muffle furnace at 500°C for 2 hours. This eliminated mineral sediment in the calculating of root and detritus biomass in each sample.

The remaining cores from the marsh pilot study were segmented into 5 cm sections and analyzed for organic matter, water content, grain size and bulk density. The cores taken during the marsh extent study were segmented into 5 cm sections but only the top two sections (10 cm) were analyzed for bulk properties. The cores were removed from the freezer and a hair dryer was used to thaw the core slightly allowing the caps to be removed and the core to be easily extruded from the aluminum tube. A knife and a mallet were used to chop each of the cores into 5 cm segments. The segment was placed on a pre-weighed dish and covered. Wet weights were recorded after the sample was allowed to reach room temperature. Segments were placed in a drying oven at 60°C for 3 days and then cooled in a desiccator before weighing. The water content was determined using Equation 1 and dry bulk density was determined using Equation 2a and 2b. Samples were then crushed using a mortar and pestle. A portion of the crushed sample was placed on a pre-weighed crucible while the rest of the sample was placed in a plastic bag for grain size analysis. The sample placed on the crucible was heated at 550°C for 2 hours and cooled to 200°C in the muffle furnace. The crucibles were placed in a dessicator for 1 hour and then weighed. Percent organic matter was determined using Equation 3.

$$Percent\ Water\ Content = \left(\frac{wet\ weight\ (g) - dry\ weight(g)}{wet\ weight\ (g)}\right) \times 100 \quad (1)$$

$$Dry\ Bulk\ Density = \frac{net\ dry\ weight\ (g)}{volume(ml)} \quad (2a)$$

$$Volume = \ length\ of sediment\ segment\ (cm)\ \times radius\ of\ tube\ (cm)^2 \times\ \pi \quad (2b)$$

$$Percent\ Organic\ Matter = \left(\frac{dry\ weight\ (g) - ash\ weight(g)}{dry\ weight\ (g)}\right) \times 100 \quad (3)$$

The sediment was processed for grain size using the method of Mora and Burdick (2013b). Sediment samples were crushed and homogenized prior to being placed on a pre-weighed crucible. Samples were placed in a muffle furnace at 465°C for 4 hours and subsequently cooled to room-temperature in a dessicator. The remains were gently disaggregated with a mortar and pestle before being placed in a sieve stack consisting of a 63 μm sieve and pan. The samples were placed on a sieve shaker for 10 minutes. The material retained by the 63 μm sieve was classified as sand (coarse material) and the material remaining in the pan was classified as silt & clay (fine material) (Equation 4).

$$Sand{:}\ Silt/Clay\ ratio = \frac{Portion\ retained\ by\ sieve}{Portion\ in\ pan} \quad (4)$$

The Cline method (1969) was used to determine the concentration of sulfide at each sampling station. A dye solution consisting of distilled water, concentrated hydrochloric acid, p-Aminodimethytaniline, and iron (III) chloride was mixed and 10 ml was added to each vial brought back from the field (See Appendix III for further information). The vials were gently shaken, degassed and placed in the dark for 20 minutes to develop. A Varian Cary® 50 UV-Vis Spectrophotometer was used to determine the absorbance of each sample. The absorbance of the

samples were compared to the absorbance of a blank that was created with each batch of samples. The concentration was determined using Equation 5.

$$Sulfide\ Concentration\ (mM) = \frac{Absorbance_{sample} - Absorbance_{colour\ blank}}{0.542} \quad (5)$$

The vegetation samples that were collected in the field were sorted for live and dead vegetation within 5 days of collection. The live and dead portions were separated and the live portions were washed to remove sediment from the leaves. The live vegetation was dried at 60°C for three days to dry the samples to an even dry weight. The biomass per cm^2 was determined using Equation 6.

$$Dry\ Biomass\ (\frac{g}{cm^2}) = \frac{Dry\ Weight\ of\ Vegetation\ (g)}{625\ (cm^2)} \quad (6)$$

The mean inundation time and inundation frequency were determined for the sampling points used for the marsh pilot and marsh extent studies. These variables were determined using the compensated data taken from the leveloggers that were placed within the main tidal creek. Mean inundation time was determined using Equation 7 and inundation frequency was determined using Equation 8 and 8b. For equation 7 & 8b, the "count of high tides > elevation of sample point" were the total number of tides that exceeded the elevation of each sampling point.

$$Inundation\ Frequency\ (\%) = \frac{Count\ of\ high\ tides > elevation\ of\ sample\ point}{total\ of\ high\ tides} \quad (7)$$

$$Mean\ Inundation\ Time\ (min) = \frac{Hydroperiod\ (\text{min})}{Inundation\ Frequency\ (\%)} \quad (8)$$

$$Hydroperiod\ (\text{min}) = (Count\ of\ high\ tides > elevation\ of\ sample\ point) \text{x}\ 5\ min \quad (8b)$$

Metrological conditions were recorded throughout the duration of the study to determine total rainfall and variation in temperature.

3.4 Statistical Analysis

A drainage class (well, moderate, or poor) was given for each sampling location based on vegetation present and the presence of standing water when sampling was completed. The well drained areas were dominated by the tall form *Spartina alterniflora*, *Spartina pectinata* or *Carex palecea*, had an absence of sulfide and no standing water present during sampling. Moderately drained sites were dominated by *Spartina patens* and *Juncus gerardii*, an absence of sulfide smell and water present on some of the sampling periods. Poorly drained sites were dominated by short form *Spartina alterniflora,* strong smell of sulfide and presence of water at surface throughout the sampling period.

MYSTAT 12 Version 12.02.00 (A student version of SYSTAT) was used for statistical analysis. An analysis of variance (ANOVA) was performed on aboveground biomass data collected during the marsh pilot study on July 14, 2014 to determine if there was a statistical difference between mean aboveground biomass production between the three drainage classes (well, moderate and poor) at $\alpha = 0.05$. July 14, 2014 was selected for this analysis as it represented peak biomass at Cheverie Creek. Residuals were used to assess data for normality as this is a requirement of ANOVAs. The residuals were plotted in a histogram to determine how closely it resembled a bell curve or normal distribution. The aboveground biomass data was determined to be normal and did not require any transformations. Inundation frequency and time was also assessed using an ANOVA to determine if there was a statistical difference between the mean inundation frequency and time between each drainage class at $\alpha = 0.05$. No data transformation were required. The ANOVA completed on aboveground biomass data collected during the marsh extent study had to be ln transformed in order to fulfill data requirements of

ANOVAs. Organic matter at 5 to 10 cm was ln transformed. Organic matter at 0 to 5 cm, bulk density at 0 to 5 and 5 to 10 cm and water content at 0 to 5 and 5 to 10 cm were each assessed using an ANOVA and residual analyses indicated they did not require any data transformations.

A multivariate analysis of variance (MANOVA) was completed on the redox potential and salinity measurements that were gathered during the marsh pilot study. The MANOVA was used to determine the univariate and multivariate significant differences within drainage class, during spring and neap cycling and at varying depths. The MANOVA determine if there were statistical differences between sample locations that included drainage class, neap versus spring, and neap/spring and drainage at $\alpha = 0.05$. The MANOVA also determined if there were statistical differences within sample locations that included the previously mentioned sampling locations as well as depth at $\alpha = 0.05$. Redox potential at 20 cm was transformed using 3rd root. Redox potential at 5, 10, and 15 cm depths did not need to be transformed. Salinity values did not need to be transformed. For the marsh extent study, redox potential and salinity were assessed for significant differences. None of the data required transformations.

A principal component analysis (PCA) was completed to determine which variables were closely related to one another and which variables were similar. The PCA enabled the data to be simplified to allow easy assessment through a subsequent backwards stepwise regression. Data collected from the marsh extent study was used for the assessment. Sediment chemistry data collected on August 21 to 22, 2014 was entered into the PCA as this data set was compiled during neap tides. Neap tides were shown to have harsher conditions as compared to spring tides due to higher salinity values. Therefore, this data would reflect impacts on vegetation. Salinity, redox potential, water content at 5 to 10 cm, inundation frequency and inundation time did not require any data transformations before being brought into the PCA. Sulfide data were transformed into presence/absence. Bulk density 0 to 5 and 5 to 10 cm, organic matter 0 to 5 cm and 5 to 10 cm,

and water content 0 to 5 cm were all ln transformed prior to entry into PCA. The PCA was completed on correlations with minimum eigenvalue equal to 1, iterations set to 25 and convergence equal to 0.001. A correlation matrix was completed and used in the PCA. Factor scores for each sample point from the PCA analysis were recorded and used for the backwards stepwise regression. Aboveground biomass data from August 14, 2014 were used in the regression as it is closest to peak biomass.

Chapter 4: Results

4.1 Marsh Pilot Study

The marsh pilot study focused on changes over the duration of the sampling season and determining the appropriate depth for soil chemistry measurements. Each of the variables is displayed in relation to each drainage class (well, moderate and poor) and categorized by spring and neap tides to display the variability measured over drainage types and lunar periods during the growing season.

4.1.1 Aboveground Biomass

The highest aboveground biomass value for each drainage class occured on July 18, 2014 (Well: 0.08 g/cm^2; moderate: 0.05 g/cm^2; poor: 0.03 g/cm^2) (Figure 4.1). Aboveground biomass values were generally greater in the well drained sites while least in the poorly drained sites but an analysis of variance (ANOVA) revealed no significant difference was found between the three drainage classes for that date (α: 0.05; p-value: 0.196; degrees of freedom (df): 2).

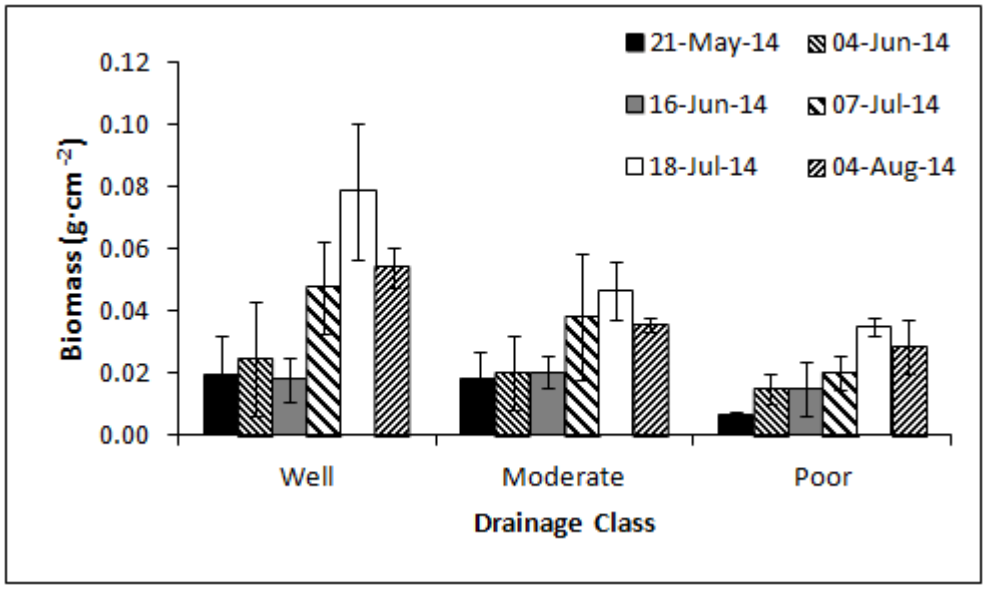

Figure 4.1 Average aboveground biomass for each drainage class for each sampling period. Solid bars represent spring tides and hashed lines represent neap tides. Error bars display standard error. (For each sampling period: Well, n=3; moderate, n=4; poor, n=2).

4.1.2. Pore Water Sulfide Concentration

Sulfide was not found in measurable quantities at the well drained locations at any sampling time. A few of the samples within the moderately drained category had detectable sulfide concentrations on some days (Figure 4.2). Measureable concentrations of sulfide were found at the poorly drained sites on most days however, due to the small sample size (n=2) and a large difference in sulfide concentration at the two locations (Figure 4.3), statistics could not be applied to the data. At L5S3 (Figure 4.3), there was an increase in sulfide concentration over time. There does not appear to be a difference between the spring and neap tides. At L7S5 (Figure 4.3), there was also an increase in sulfide concentration over time but the increases were not consistent and concentrations were lower than at L5S3.

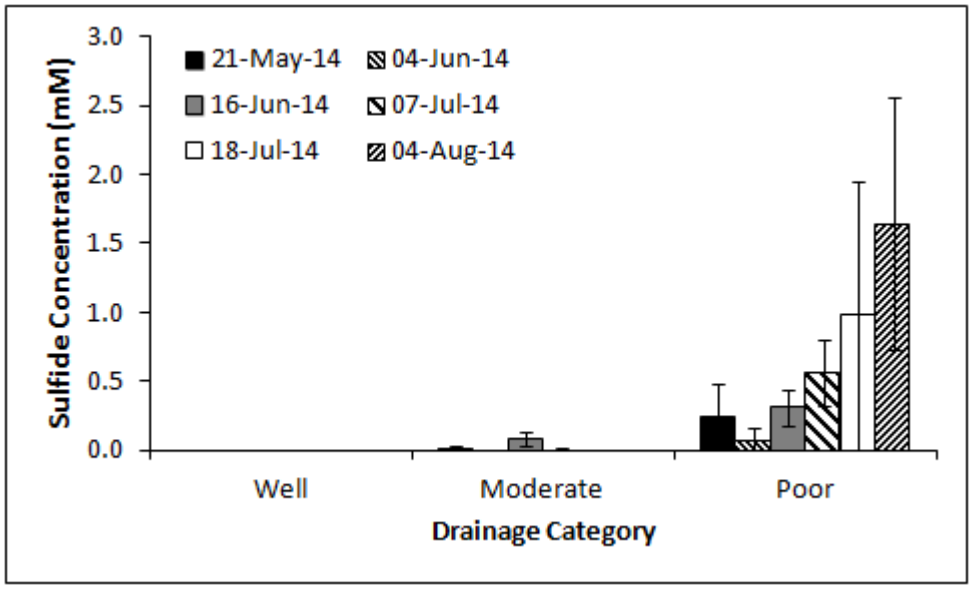

Figure 4. 2 Average sulfide concentration for each drainage class for each sampling period. Solid bars represent spring tides and hashed lines represent neap tides. Error bars display standard error and range for poorly drained sites. (Well, n=3; moderate, n=4; poor, n=2).

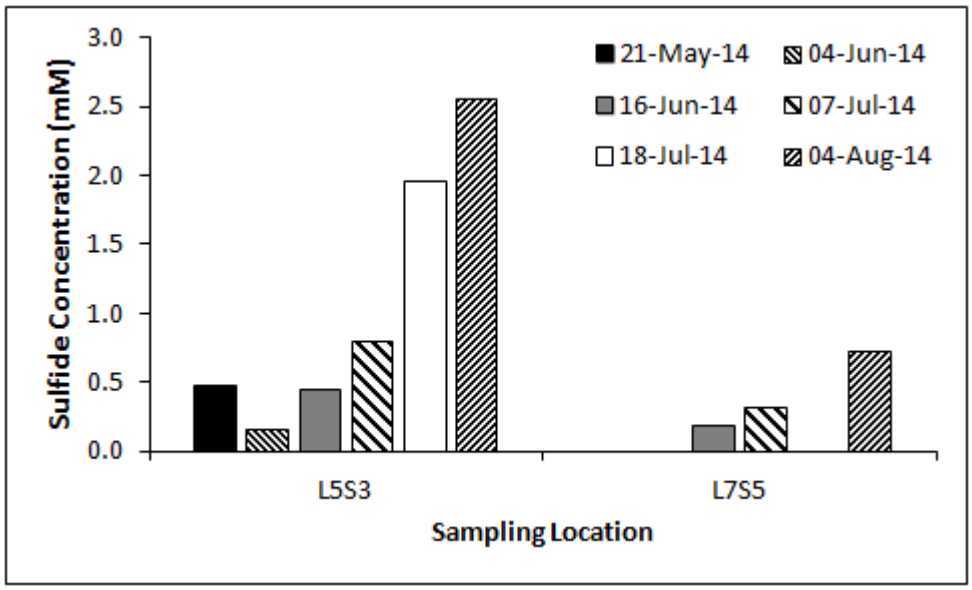

Figure 4.3 Sulfide concentration at poorly drained sampling locations for each sampling period. Solid bars represent spring tides and hashed lines represent neap tides.

4.1.3. Redox Potential

The moderately and well drained sites (Figure 4.4a,b) display a general trend of decreasing redox potential with depth. The dominant redox reactions occurring in well drained sites were oxygen and nitrate/manganese (oxygen: > +300 mV; nitrate/manganese: +100 to +300 mV) based upon the average soil redox potential. The dominant redox reaction measured in moderately drained sites was nitrate/manganese (nitrate/manganese: +100 to +300 mV) due to the average redox potential (+150 mV). The poorly drained sites displayed a great variability in redox potential with depth and over time. The lowest average redox potential measured during the sampling period fell within methane reduction (< -200 mV) that was measured on June 16 in poorly drained sites. The dominant redox reaction measured within the poorly drained sites was iron (-100 to +100 mV) and nitrate/manganese reduction.

The multivariate analysis of variance (MANOVA) and repeated measures completed on the redox potential measurements revealed a significant difference in the mean redox potential between drainage classes (well, moderate and poorly drained) (α: 0.05; p-value: 0.000; df: 2),

with varying depth (α: 0.05; p-value: 0.000; df: 3), and between neap versus spring tides (α: 0.05; p-value: 0.008; df:1). The pattern showing decline in redox from well to poorly drained soils did no change from neap to spring tides (α: 0.05; p-value: 0.266; df:2). No significant difference was also found with depth and neap versus spring (α: 0.05; p-value: 0.082; df:3). There was a significant interaction found in the redox potential measurements with depth and drainage class (α: 0.05; p-value: 0.000; df: 6). The significant interaction shows that redox potential values declined with depth for well drained, but not poorly drained marsh soils.

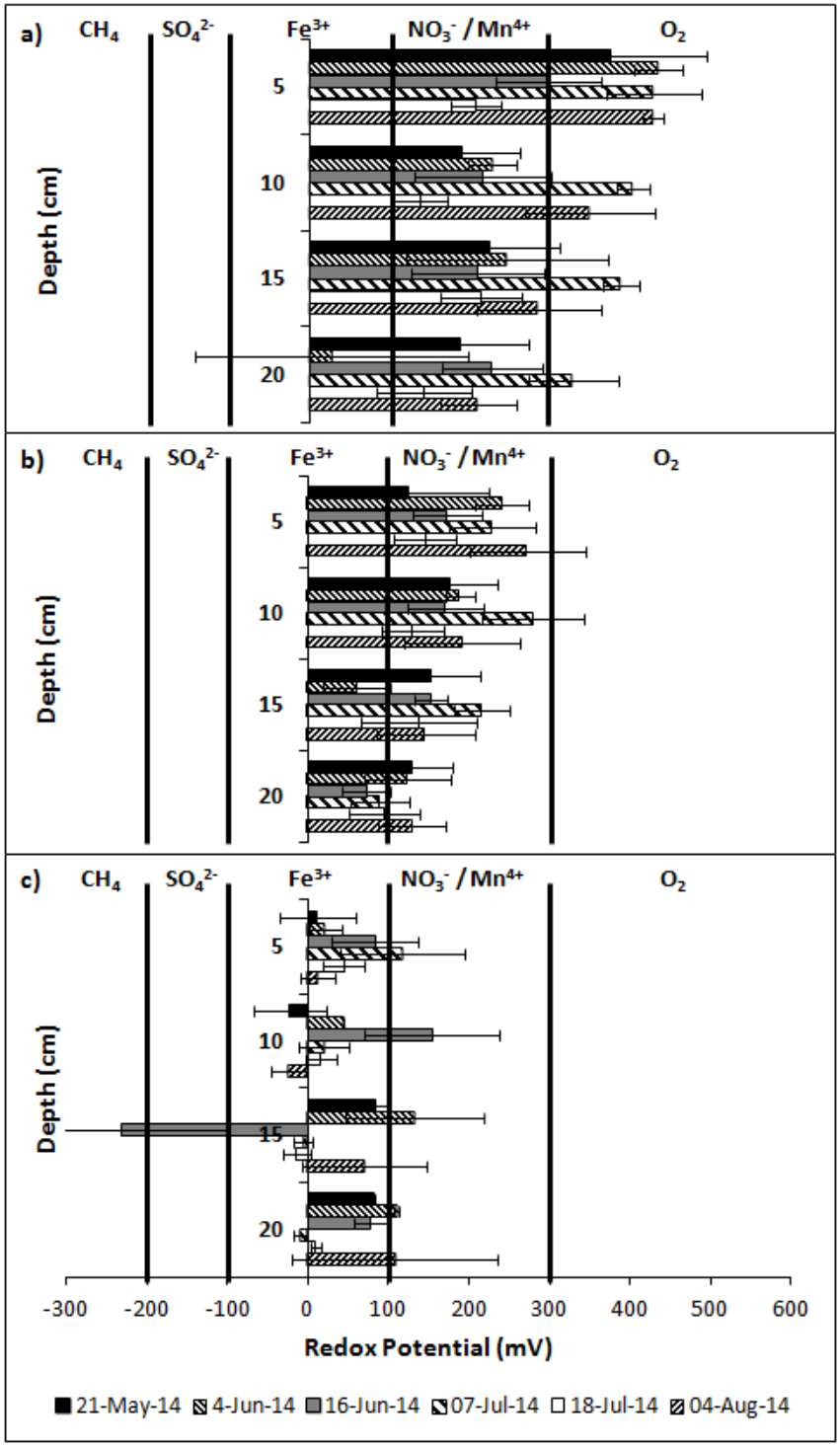

Figure 4.4 Average redox potential for each drainage class varying with depth for each sampling period. Solid bars represent spring tides and hashed lines represent neap tides. a) Well drained sites, b) Moderately drained sites, c) Poorly drained sites. Error bars display standard error. (Well, n=3; moderate, n=4; poor, n=2).

4.1.4. Salinity

The highest salinity values (Table 4.1) were measured in the poorly drained sites where there was a slight decrease in salinity with depth (Figure 4.5c). The lowest salinity values were measured at well-drained sites and intermediate levels in the moderately drained (Table 4.1; Figure 4.5). The highest salinity levels were recorded on August 4 (Figure 4.5) at all sites after a period of high temperature and low rain fall (Figure 4.21).

The MANOVA repeated measures analysis completed on salinity values to compare drainage class, neap versus spring tide periods and soil depth revealed a significant difference between the drainage classes (α: 0.05; p-value: 0.000; df: 2) and neap versus spring tides (α: 0.05; p-value: 0.015; df: 1). A significant difference was found with depth (α: 0.05; p-value: 0.000; df: 3), and the interaction between depth and drainage class (α: 0.05; p-value: 0.000; df: 6). No significant interactions were found when neap versus spring tides were compared to drainage class (α: 0.05; p-value: 0.577; df: 2) or depth (α: 0.05; p-value: 0.600; df: 3) or depth by drainage class (α: 0.05; p-value: 0.300; df: 6). Similar to redox potential, salinity varied significantly by all three main effects (drainage class, soil depth, and tide period) and the interaction of drainage class and depth. Therefore, drainage class and depth play a significant role in salinity values and are influenced by tidal period.

Table 4. 1 Range of salinity values over sampling period for each drainage class organized by depth.

Drainage Class	Depth (cm)	5	10	15	20
Well	Min	3.12 ppt (May 21)	3.11 ppt (May 21)	2.70 ppt (July 7)	2.47 ppt (May 21)
	Max	8.95 ppt (Aug 4)	9.82 ppt (Aug 4)	9.39 ppt (Aug 4)	7.39 ppt (Aug 4)
Moderate	Min	6.16 ppt (May 21)	5.22 ppt (May 21)	4.73 ppt (May 21)	5.00 ppt (May 21)
	Max	11.84 ppt (Aug 4)	10.49 ppt (Aug 4)	9.16 ppt (Aug 4)	8.60 ppt (Aug 4)
Poor	Min	12.18 ppt (June 4)	12.15 ppt (July 18)	10.63 ppt (May 21)	9.06 ppt (May 21)
	Max	21.29 ppt (Aug 4)	20.58 ppt (Aug 4)	18.4 ppt (Aug 4)	17.63 ppt (Aug 4)

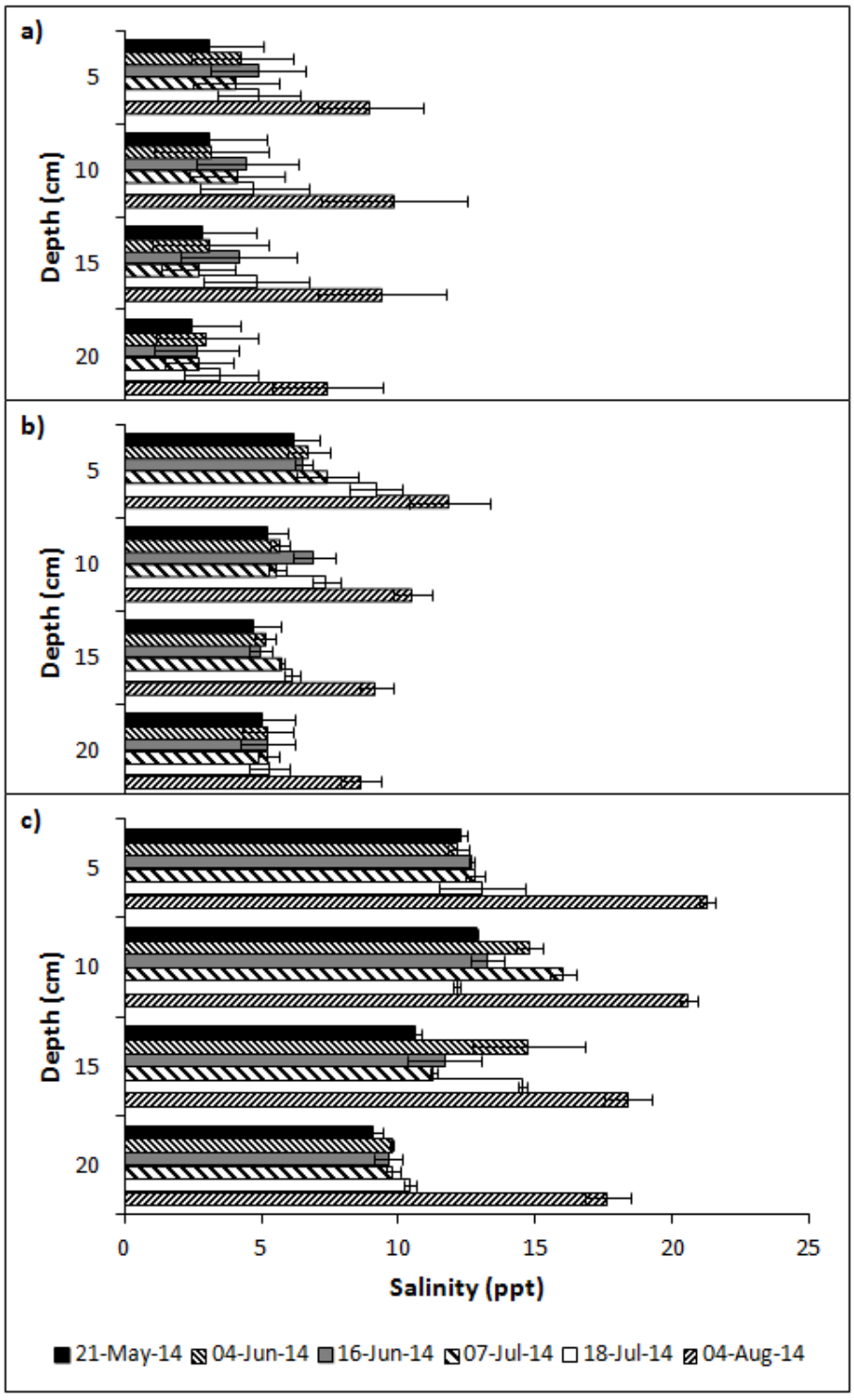

Figure 4. 5 Average salinity for each drainage class varying with depth for each sampling period. Solid bars represent spring tides and hashed lines represent neap tides. a) Well drained sites, b) Moderately drained sites, c) Poorly drained sites. Error bars display standard error. (Well, n=3; moderate, n=4; poor, n=2).

4.1.5. Hydrology

The highest mean inundation frequency and time during the sampling period (May 21 to August 4) was experienced at the poorly drained sites (Inundation frequency: 61.7% of high tides measured; Inundation time: 102.8 min per inundating tide). Moderately drained sites were in-between poor and well drained (Inundation frequency: 50.8% of high tide measured; Inundation time: 94.8 min per inundating tide). The lowest inundation frequency and time was experienced at the well-drained sites (Inundation frequency: 34.3% of high tides measured; Inundation time: 80.7 min per inundating tide) (Figure 4.6 & 4.7). Despite the large differences among means, the ANOVA revealed no significant difference between the inundation frequency for the three drainage classes (α: 0.05; p-value: 0.219; df: 2). Although there were no statistically significant differences between the drainage classes and the inundation frequency nor inundation time, there is an observable trend.

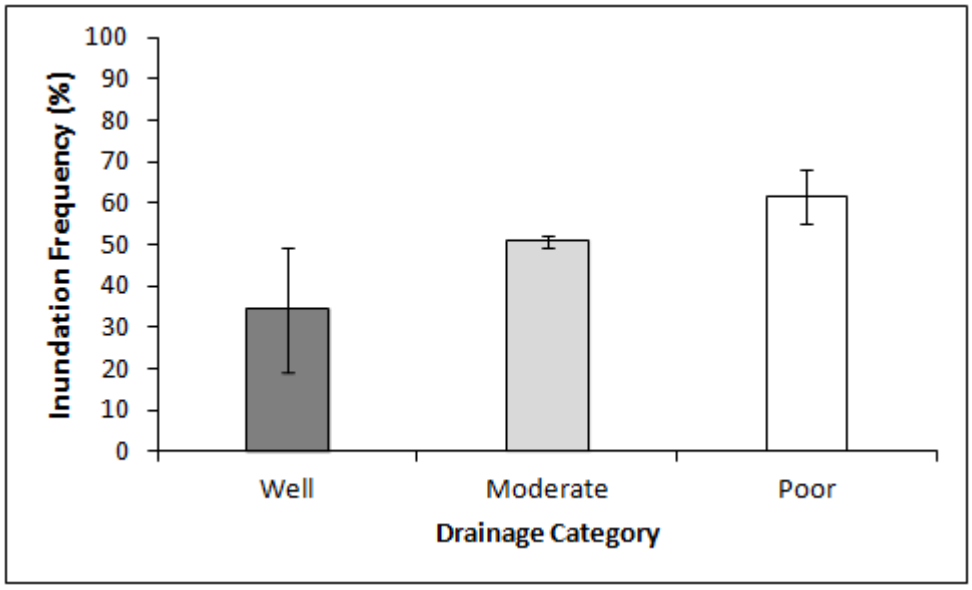

Figure 4. 6 Inundation frequency for each drainage class. Error bars display standard error. (Well: n=3; moderate: n=4; poor: n=2).

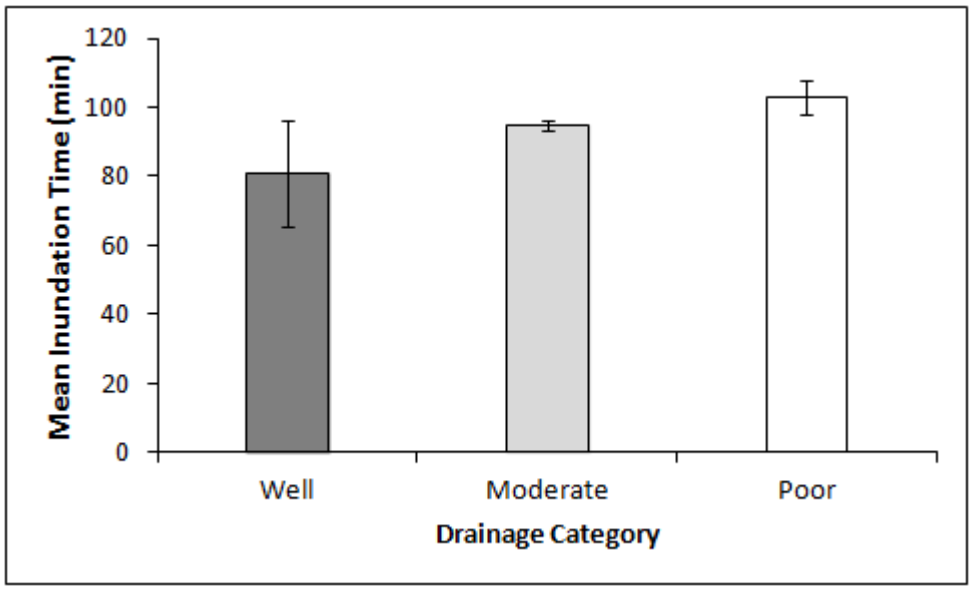

Figure 4. 7 Mean inundation time for each drainage class. Error bars are displaying standard error. (Well: n=3; moderate: n=4; poor: n=2).

4.1.6 Sediment Characteristics

The highest organic matter content in the well drained sites was found within the top 5 cm (14.6%) (Figure 4.7). The highest organic matter in the moderate and poorly drained sites was also found in 5 to 10 cm (Poor: 22.8%; Moderate: 17.4%). The poorly drained sites had more organic matter than any other site within the top 5 to 10 cm (Well: 10.1%; Moderate: 17.4%; Poor: 22.8%) and lower 10 to 15 cm (Well: 9.4%; Moderate: 8.5%; Poor: 17.9%). The MANOVA revealed no significant difference in organic matter content with depth (α: 0.05; p-value: 0.103; df: 3). A slight decrease in organic matter with depth was found in all drainage classes below 5 cm. Well drained sites showed this trend from the very top of the core. No significant interactions between organic matter content and drainage classes (α: 0.05; p-value: 0.737; df: 2) or depth with drainage classes (α: 0.05; p-value: 0.344; df: 6) were found.

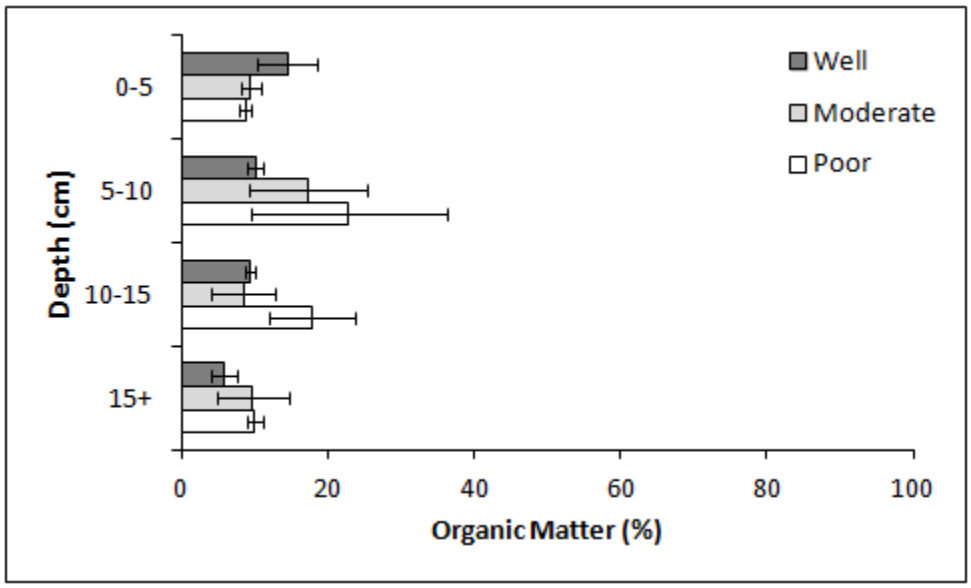

Figure 4. 8 Average organic matter content for each drainage class varying with depth. Error bars display standard error. (Well: n = 4; moderate: n=4; poor: n=2).

Well drained sites had an increase in bulk density with depth from top of core (Figure 4.9). Moderately drained sites displayed largest bulk density at 10 to 15 cm (1.06 g/cm^3) and the smallest at 5 to 10 cm (0.70 g/cm^3). Poorly drained sites displayed similar bulk density in the 0 to 5 cm (0.49 g/cm^3) and 15+ cm (0.54 g/cm^3). Lowest bulk density was found in 5 to 10 cm (0.34 g/cm^3) and 10 to 15 cm (0.34 g/cm^3) in poorly drained sites. The MANOVA revealed a significant difference in the bulk density values with varying depth (α: 0.05; p-value: 0.002; df: 3), and a significant interaction with depth and drainage class (α: 0.05; p-value: 0.029; df: 6). No significant difference was identified between drainage class (α: 0.05; p-value: 0.244; df: 2) alone. Any interactions between drainage classes were hidden by the strong increases in bulk density with depth effect among well and moderately drained sites.

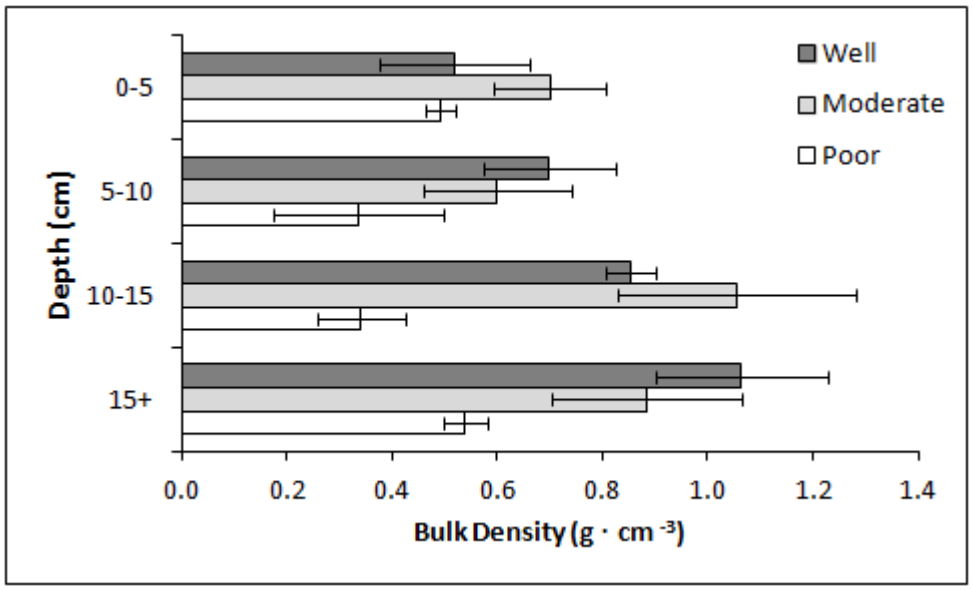

Figure 4. 9 Average bulk density for each drainage class varying with depth. Error bars display standard error. (Well: n = 4; moderate: n=4; poor: n=2).

Poorly drained sites had the highest water content values as expected (63.3%) (Figure 4.10). Well drained sites had lower water content with depth and highest water content was at 0 to 5 cm (48.4%). Moderately drained sites also had a general decrease in water content. The MANOVA revealed a significant difference in water content with depth (α: 0.05; p-value: 0.005; df: 3). No significant difference was found between drainage classes (α: 0.05; p-value: 0.192; df: 2). A small significant interaction was found between depth and drainage class (p-value: 0.052; df: 6), indicating the depth effect was absent in poorly drained soils. Any interactions between drainage classes were hidden by the strong decreases in water content with depth effect among well and moderately drained sites.

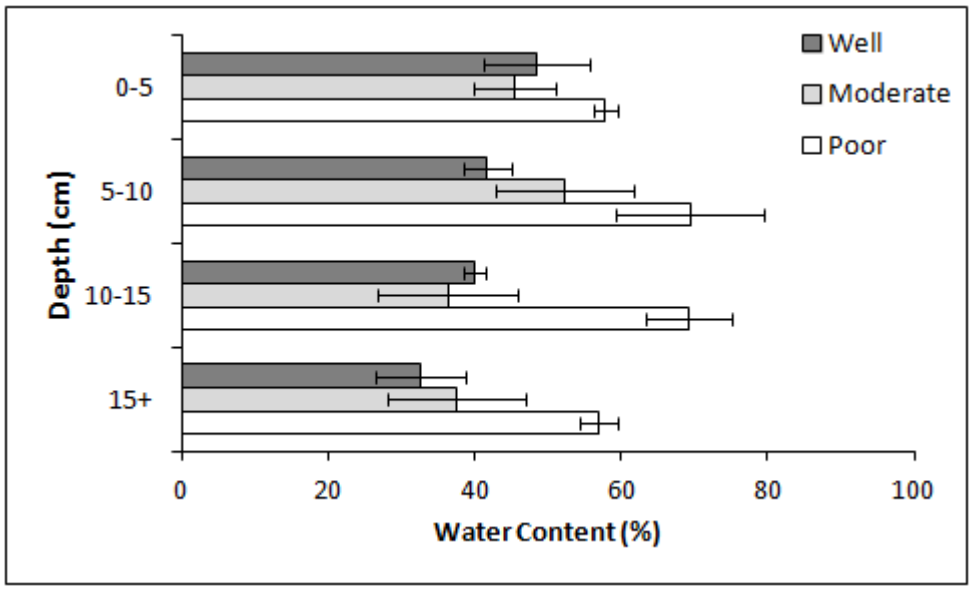

Figure 4. 10 Average water content for each drainage class varying with depth. Error bars display standard error. (Well: n = 4; moderate: n=4; poor: n=2).

4.2. Marsh Extent Study

The marsh extent study focused on understanding the relationship between aboveground biomass production and sediments (chemistry and bulk characteristics) based upon the three drainage classes. The marsh extent study also provided a larger data set to determine which variables affected aboveground biomass production during the summer of 2014 at Cheverie Creek, nine years following the restoration of tidal exchange to the marsh.

4.2.1 Aboveground Biomass

Highest aboveground biomass was measured in the well drained sites (0.08 g/cm^2) followed by moderate (0.06 g/cm^2) and poorly drained sites (0.05 g/cm^2) (Figure 4.11). An ANOVA revealed a significant difference in biomass values between drainage classes (α: 0.05; p-value: 0.000; df: 2). No significant difference was detected between neap versus spring (α: 0.05; p-value: 0.141; df: 1) nor the interaction of drainage class and neap versus spring (α: 0.05; p-value: 0.532; df: 2).

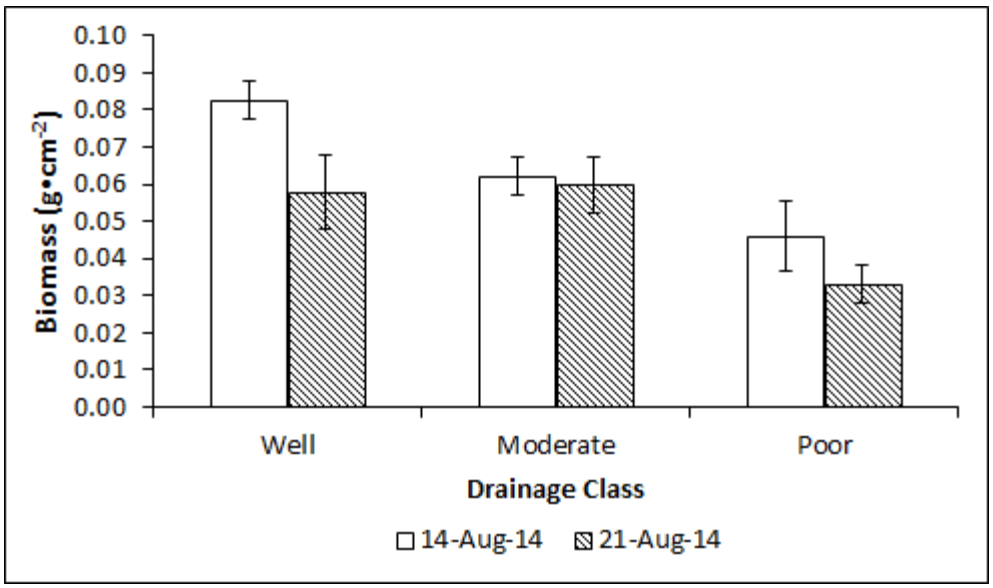

Figure 4. 11 Average aboveground biomass for each drainage class on August 14, 2014 (Spring) and August 21, 2014 (Neap). Error bars display standard error. (August 14: well: n = 9; moderate: n = 22; poor: n = 13; August 21: well: n = 8; moderate: n = 21; poor: n = 10).

4.2.2 Sulfide Concentration

Sulfide was found in detectable concentrations within moderately and poorly drained sites, but highest values were measured within poorly drained sites (Figure 4.12). A measureable concentration of sulfide was found within well drained sites only on August 14 (0.01 mM). A low sulfide concentration was measured at the moderately drained sites (Mean: 0.09 mM (August 14, 2014) and 0.1 mM (August 21, 2014)).

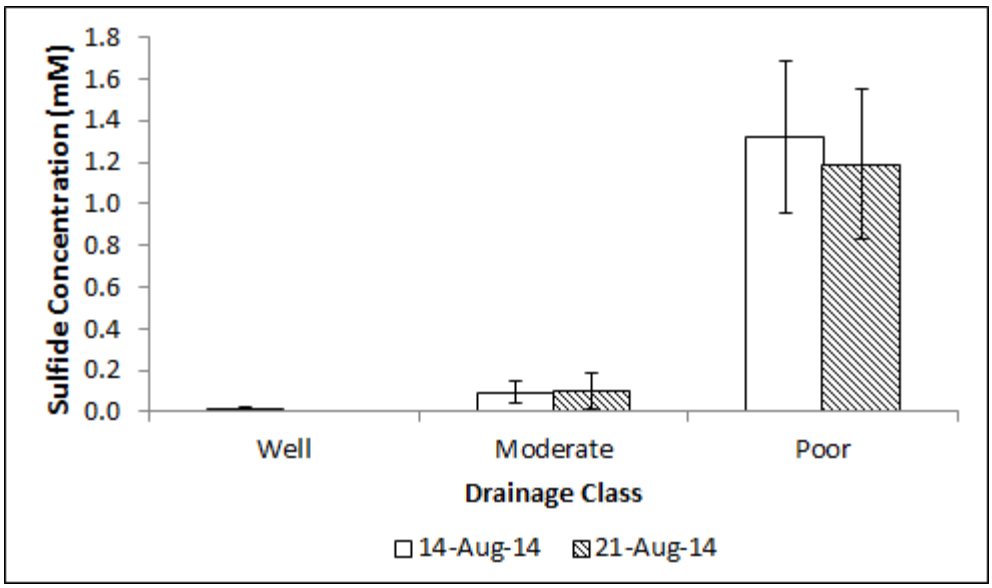

Figure 4. 12 Average sulfide concentration for each drainage class on August 14, 2014 (Spring) and August 21, 2014 (Neap). Error bars display standard error. (August 14: well: n = 9; moderate: n = 22; poor: n = 13; August 21: well: n = 9; moderate: n = 22; poor: n = 12).

4.2.3 Redox Potential

The lowest redox potentials were measured in the poorly drained sites and the highest were measured within well drained sites (Figure 4.13). The MANOVA revealed a significant difference between drainage classes (α: 0.05; p-value: 0.000; df: 2);but no significant difference was found between neap versus spring (α: 0.05; p-value: 0.324; df: 1) nor was the interaction of drainage class and neap versus spring significant (α: 0.05; p-value: 0.815; df: 2). High redox potential was measured in the well and moderately drained sites that correlates with nitrate and manganese as the dominate redox reaction at both sampling times. The dominant redox reaction measured within poorly drained sites was iron for both sampling times (Figure 4.13).

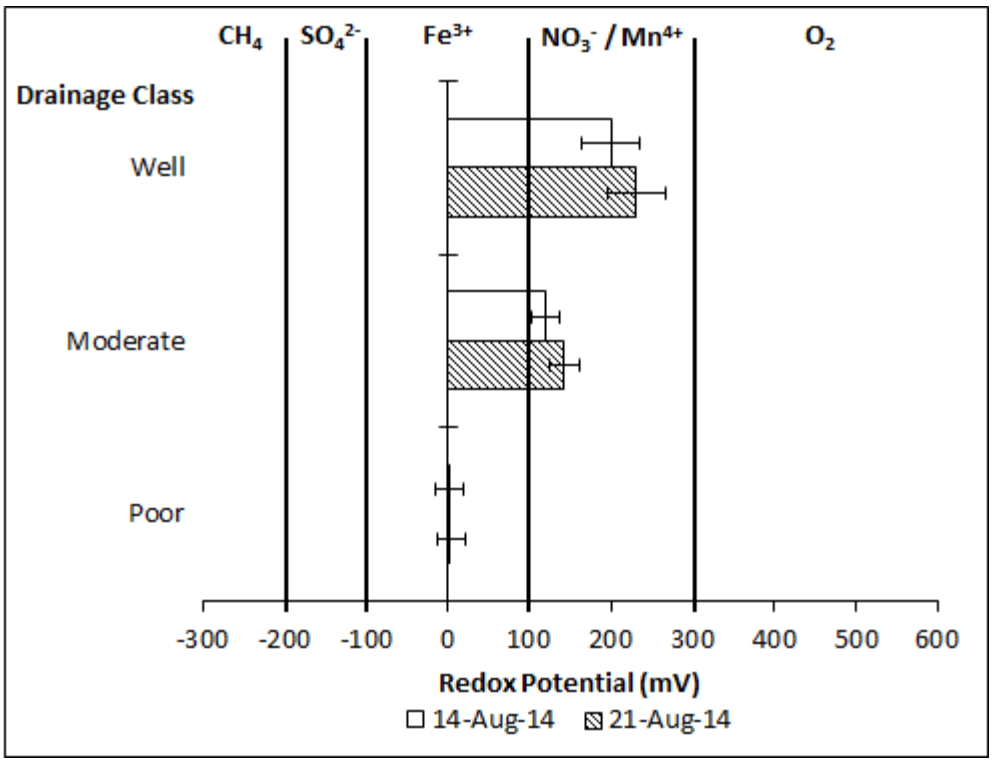

Figure 4. 13 Average redox potential for each drainage class on August 14, 2014 (Spring) and August 21, 2014 (Neap). Error bars display standard error. (August 14 & August 21: well: n = 9; moderate: n = 22; poor: n = 13).

4.2.4 Salinity

Highest salinity values were measured in the poorly drained sites and lowest salinity values were measured in well drained sites (Figure 4.14). A MANOVA completed on the data revealed a significant difference between drainage classes (α: 0.05; p-value: 0.000; df: 2) but not neap versus spring measurements (α: 0.05; p-value: 0.382; df: 1) or the interaction of drainage class with neap/spring (α: 0.05; p-value: 0.546; df: 2). Therefore, drainage class was found to have a significant influence on salinity in mid-August 2014.

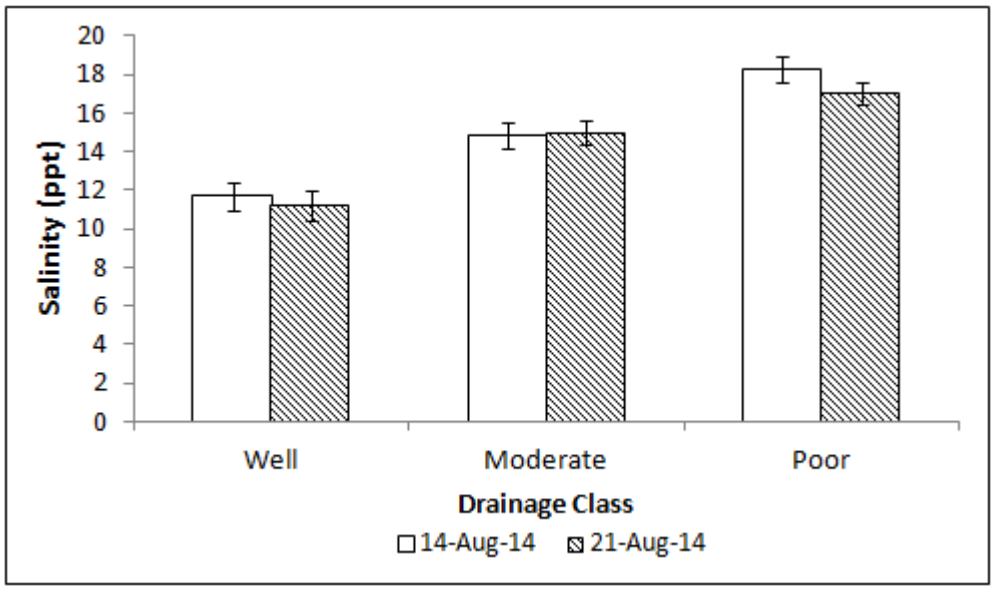

Figure 4. 14 Average salinity for each drainage class on August 14, 2014 (Spring) and August 21, 2014 (Neap). Error bars display standard error. (August 14: well: n = 9; moderate: n = 22; poor: n = 13; August 21: well: n = 9; moderate: n = 21; poor: n = 13).

4.2.5 Hydrology

Well drained sites had the highest inundation frequency (76.2 % of high tides measured) (Figure 4.15) and mean inundation time (127.6 min per tide that inundate site). This was because the well drained sites were located along the margins of the main tidal creek (Figure 3.1 & Figure 4.16). Moderately drained sites had an inundation frequency of 55.7 % of high tides measured and inundation time of 99.4 min per tide that inundated the sites while poorly drained sites had an inundation frequency of 57.4 % of high tides measured and inundation time: 100.1 min per tide that inundates the sites. Moderate and poorly drained sites had approximately the same inundation frequency and mean inundation time, but less than the well drained sites that were adjacent to tidal creeks (in contrast to the results of the pilot marsh study).

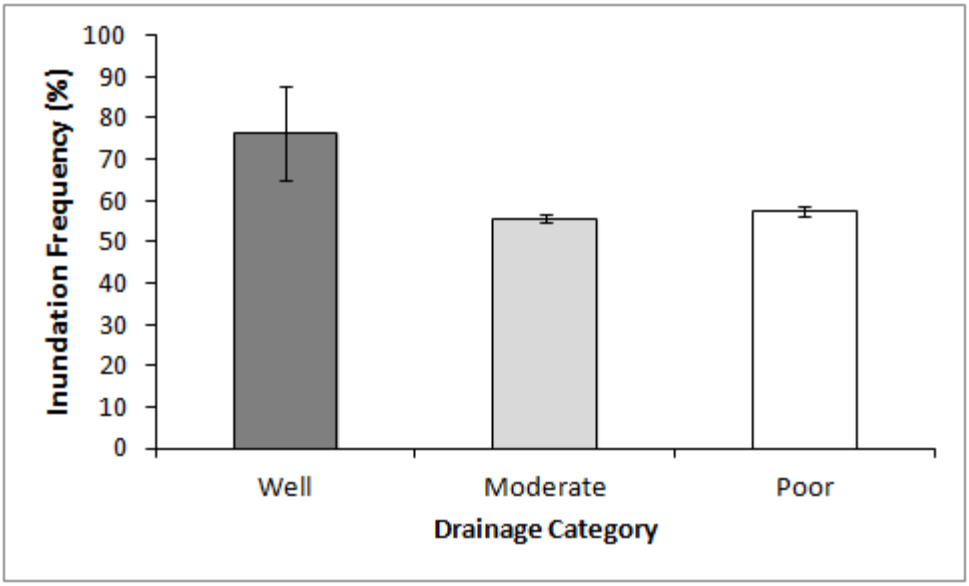

Figure 4. 15 Inundation frequency for each drainage class. Error bars display standard error. (Well: n=9; moderate: n=22; poor: n=13).

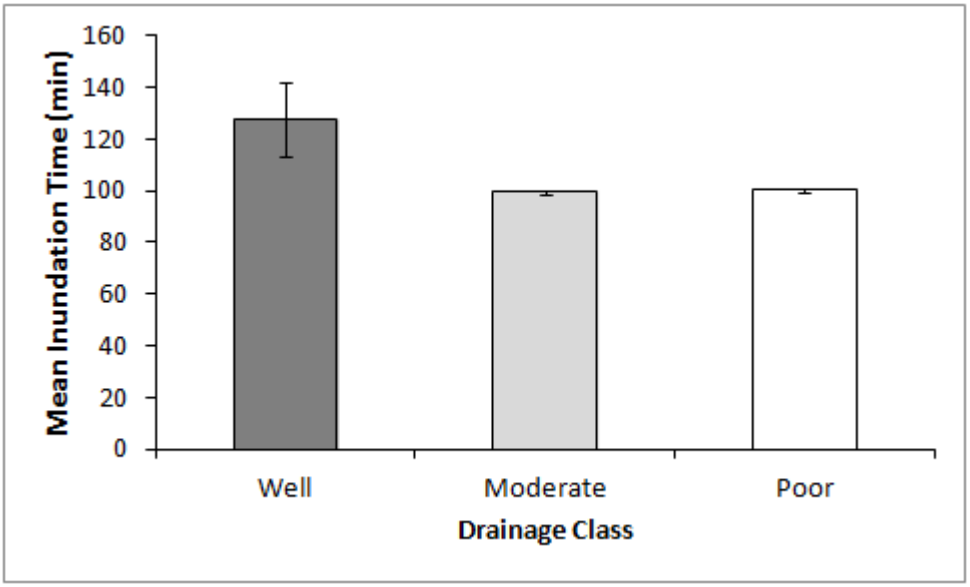

Figure 4. 16 Mean inundation time for each drainage class. Error bars display standard error. (Well: n=9; moderate: n=22; poor: n=13).

4.2.6 Sediment Characteristics

Organic matter content determined from loss on ignition was greatest within the top 5 to 10 cm for moderately and poorly drained sites (Figure 4.17). Poorly drained sites had greatest organic matter content within this zone (33.7%) than moderately drained sites (27.2%). Little

variation was found in well drained sites (0-5 cm: 8.8 %; 5-10 cm: 9.1%). An ANOVA revealed significant difference between drainage classes (α: 0.05; p-value: 0.023; df: 2) and depth (α: 0.05; p-value: 0.000; df: 2). No significant difference was found when depth was compared to drainage class (α: 0.05; p-value: 0.209; df: 2).

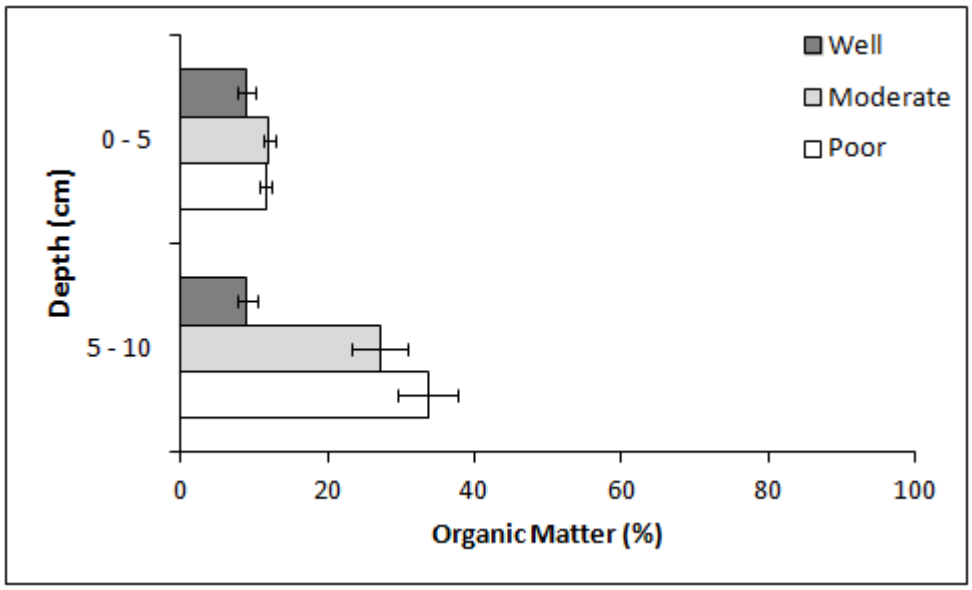

Figure 4. 17 Average organic matter for each drainage class by depth. Error bars display standard error. (Well: n=9; moderate: n=22; poor: n=13).

Well drained sites were characterized by high bulk density values within the top 10 cm (Figure 4.18). Moderately and poorly drained sites were characterized by a decrease in bulk density with depth. Poorly drained sites had the lowest bulk density in the top 10 cm (0.29 g/cm^3). An ANOVA revealed a significant difference between drainage classes (α: 0.05; p-value: 0.000; df: 2), depth (α: 0.05; p-value: 0.000; df: 1) and the depth and drainage class interaction (α: 0.05; p-value: 0.000; df: 2).

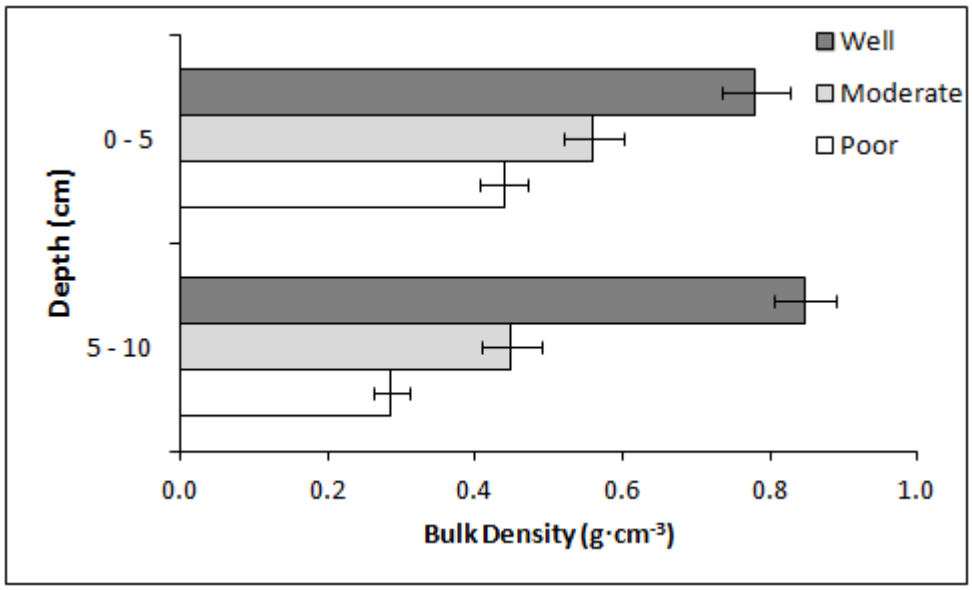

Figure 4. 18 Average bulk density for each drainage class by depth. Error bars display standard error. (Well: n=9; moderate: n=22; poor: n=13).

Poorly drained sites were characterized by greatest water content of the drainage classes and this increased slightly with depth (0-5 cm: 62.2 %; 5-10 cm: 73.01 %) (Figure 4.19). Moderately drained sites also showed an increase in water content with depth. Well drained sites were characterized by similar water content within 0 to 5 cm (42.58 %) and 5 to 10 cm (41.21%) depth. An ANOVA revealed a significant difference between drainage classes (α: 0.05; p-value: 0.000; df: 2), depth (α: 0.05; p-value: 0.000; df: 1) and depth and drainage class (α: 0.05; p-value: 0.000; df: 2).

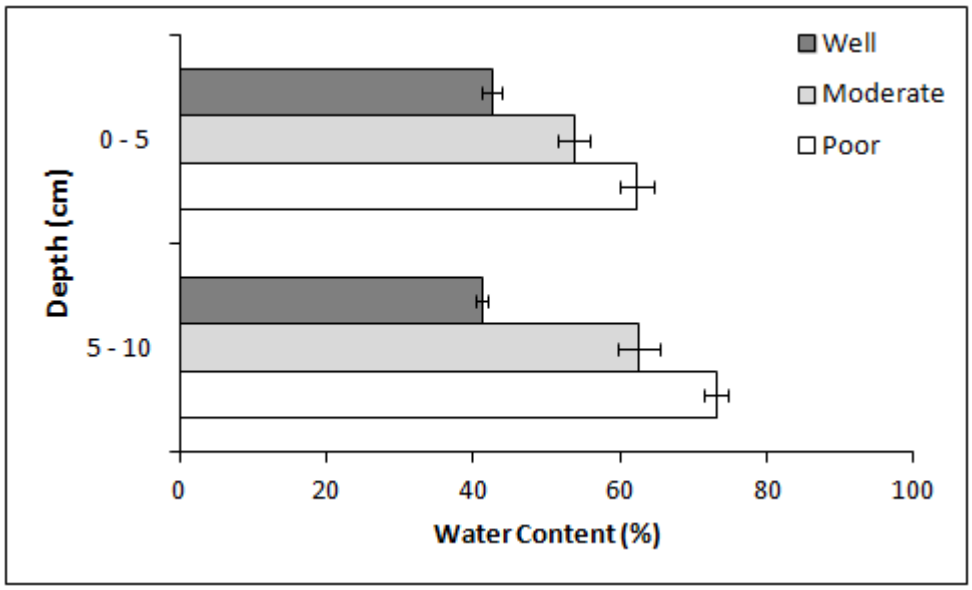

Figure 4. 19 Average water content for each drainage class by depth. Error bars display standard error. (Well: n=9; moderate: n=22; poor: n=13).

When examined by drainage class and depth, no differences were found in the ratio of sand to silt/clay, generally varying between 1.5 to 3 times more sand/silt than clay (Figure 4.20). When examined separately, well drained sites that were located on the creek edge had the highest ratio of sand to silt/clay at the 0-5 cm depth when compared to the other drainage classes (Figure 4.20).

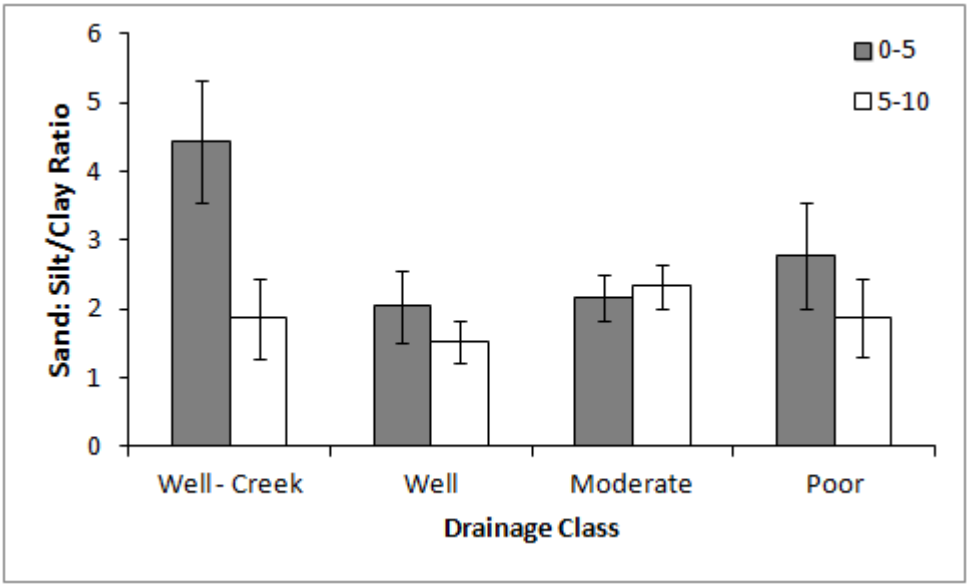

Figure 4. 20 Sand to silt/clay ratio by drainage class. Error bars are displaying standard error. (Well - creek: n = 4; well: n = 3; moderate: n = 3; poor: n = 2).

4.3. Meteorological Conditions and Groundwater Data

The meteorological data collected throughout the summer revealed a rather dry and hot summer for Cheverie. The highest total rainfall event occurred on July 26, 2014 (42.6 mm) (Figure 4.21). Hurricane Arthur passed through the region on July 4-6, 2014 but the weather station at Cheverie only received 10.8 mm of rain. Strong winds during this event may have limited rain falling into the tipping bucket rain gauge. Temperature increased from May to July with the highest temperature being recorded on July 3, 2014 at 29.8°C (Figure 4.21).

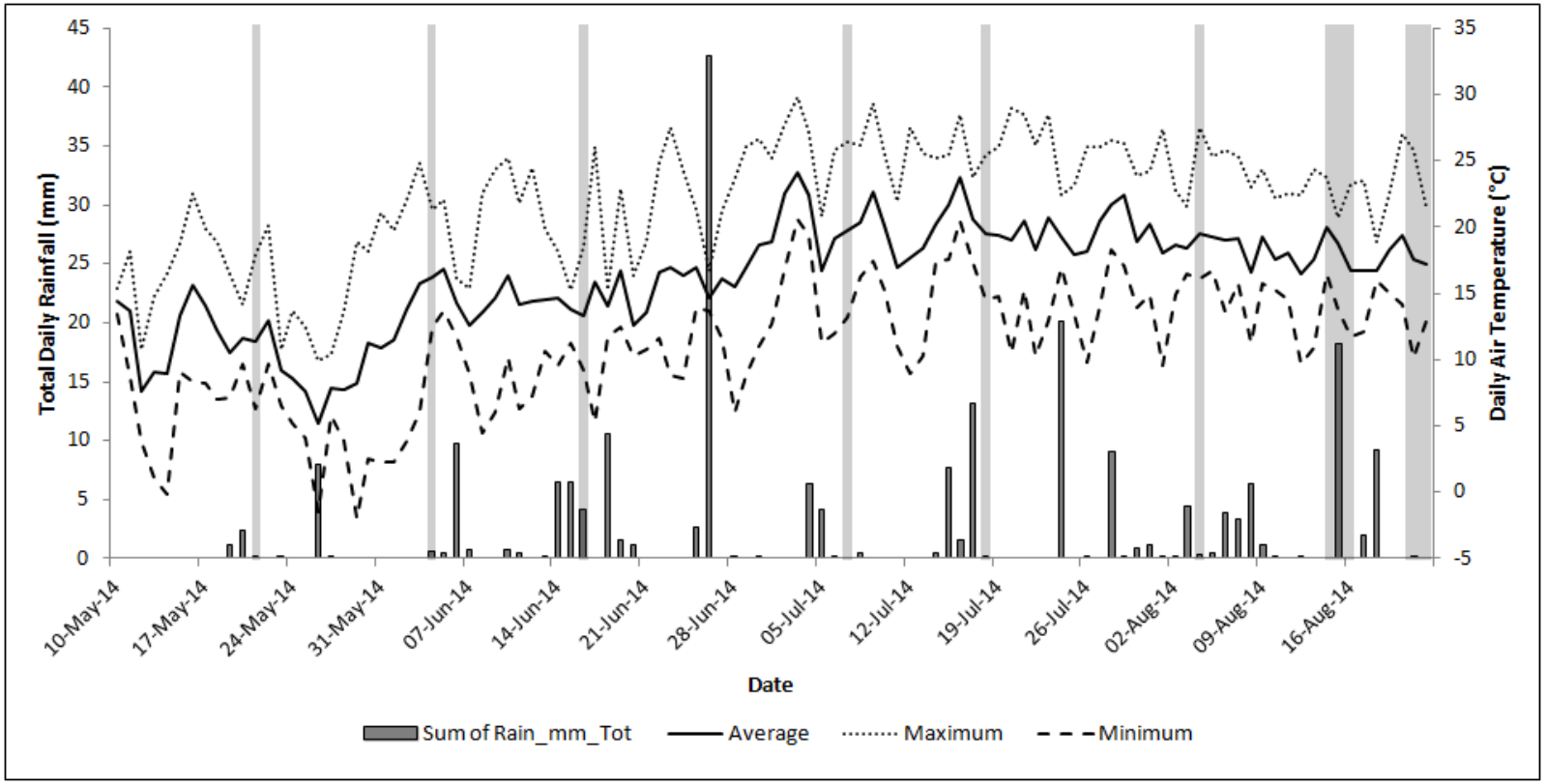

Figure 4. 21 Total daily rainfall and average, maximum, and minimum daily air temperature throughout sampling period (May 10 to August 22, 2014). Shaded areas represent sampling periods.

Groundwater measurements were collected throughout the sampling period are displayed in Figure 4.22 along with tidal data collected from the main tidal creek (grey region). All three loggers recorded the influence of tidal inundation during spring tides. At level 1, closest to upland and furthest from culvert (Figure 3.1), ground water remained approximately 10 to 30 cm below the salt marsh surface depending on length of time since last spring tide. At level 2, located in the middle of the study area (Figure 3.1), the ground water remained approximately 5 to 20 cm below the salt marsh surface. Level 3, located closest to tidal creek and culvert (Figure 3.1), experienced the greatest variability in depth to groundwater. This was due to larger neap tides and all spring tides inundating the levelogger at level 3 but not all of these tides reached level 1 and 2. Water levels steadily decreased after each set of spring tides until the next set of high spring tides inundated each location. Panty hose were installed on each stillwell decreased the influence of sediment slurry, but levels still recovered slowly. The data collected from the leveloggers should be used cautiously. The large dip in groundwater level between June and July was caused by the removal and replacement of each levelogger (Figure 4.22).

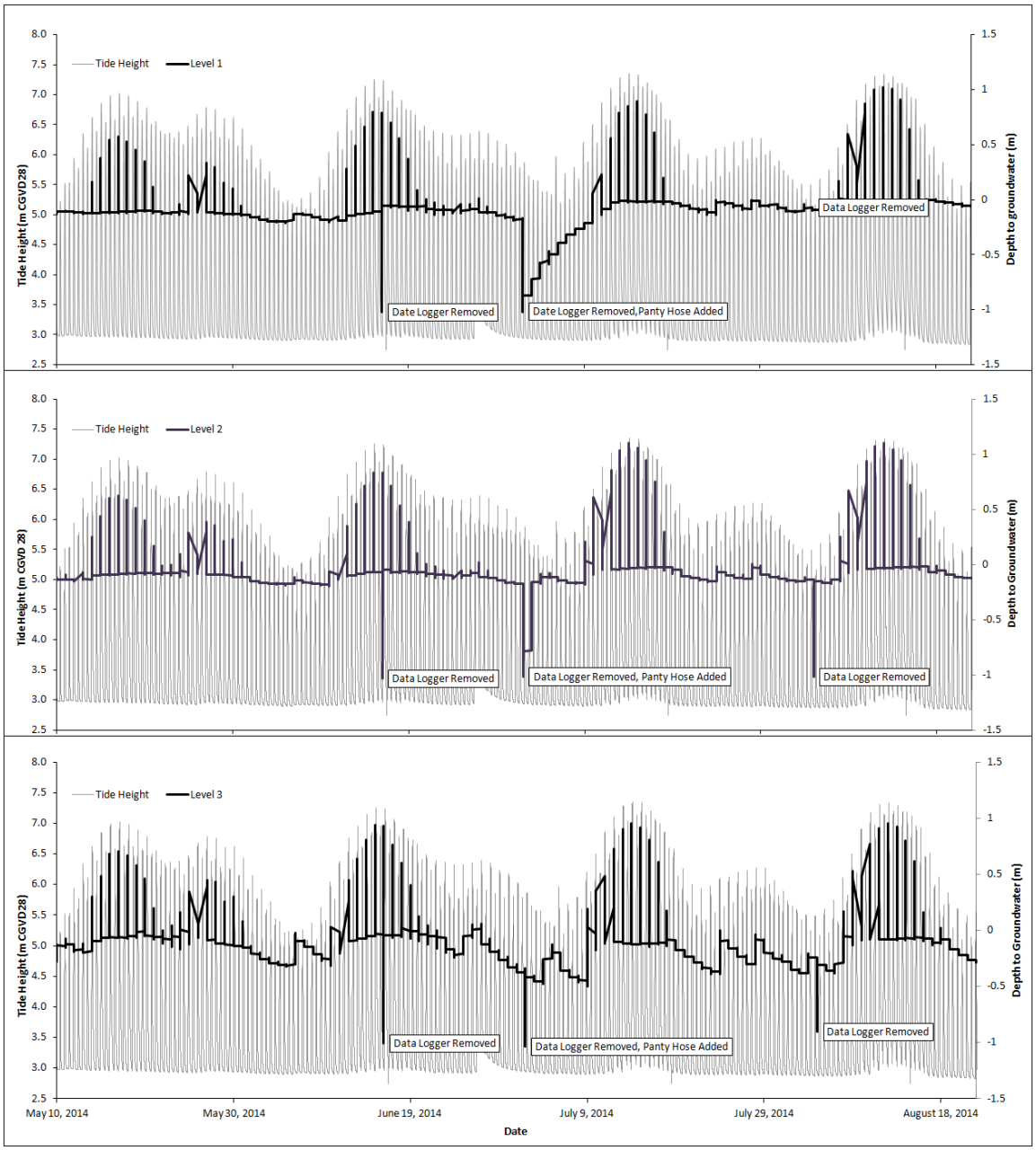

Figure 4. 22 Groundwater data collected at 3 leveloggers spread across the marsh and tide height data collected throughout the study. Zero represents soil surface for each levelogger.

4.4 Influence of Soil Chemistry, Sediment Characteristics and Inundation Frequency and Time on Above Ground Biomass Production

A principal component analysis (PCA) was completed to determine which variables were most closely related to one another and plotted to display the relationships visually (Figure 4.23). Three principal components were chosen and accounted for 86% of the variation in the sediment

properties and condition. Principal component 1 is comprised of water content and organic matter at 0 to 5 and 5 to 10 cm which cluster but are opposite to bulk density at 0 to 5 and 5 to 10 cm. This component explained 48.8% of the total variance (Table 4.2). Principal component 3 is comprised of inundation frequency and time which cluster together and explains 16.8% of the total variance. Principal component 2 is comprised of salinity and sulfide concentration which cluster and are opposite to redox potential. This component explained 20.4% of total variance.

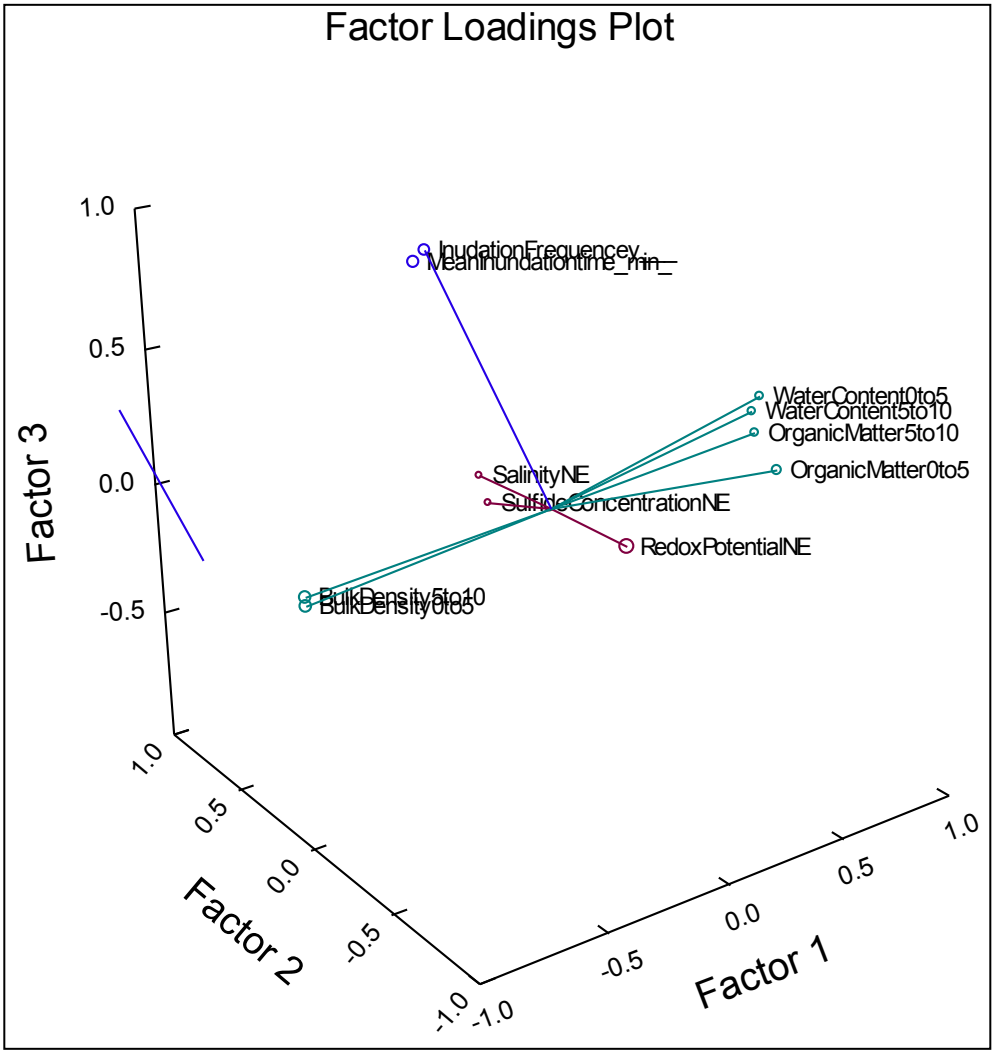

Figure 4. 23 Factor loading plot output from principal component analysis.

Table 4. 2 Principal component analysis of environmental variables and sediment characteristics

Environmental Variable	Principal components		
	1	2	3
Bulk Density 0-5cm	-0.920	0.105	-0.186
Bulk Density 5-10 cm	-0.950	0.037	-0.148
Organic Matter 0-5 cm	0.762	-0.298	-0.035
Organic Matter 5-10 cm	0.940	-0.105	0.016
Water Content 0-5 cm	0.942	-0.038	0.215
Water Content 5-10 cm	0.934	-0.004	0.167
Salinity 5 cm	0.235	0.810	-0.056
Redox Potential 5 cm	-0.308	-0.848	0.094
Sulfide (Presence/Absence)	0.199	0.824	-0.242
Inundation Frequency	-0.266	0.244	0.919
Mean Inundation Time	-0.361	0.160	0.897
Variance explained by components	5.365	2.243	1.851
Percent of total variance explained	48.769	20.394	16.829

The factor scores collected from the PCA were entered into a backward stepwise regression to determine which variables were able to best predict aboveground biomass production. The output of the regression indicated that principal components 1 and 2 could explain 17.9% of the variance in aboveground biomass (Table 4.3). Aboveground biomass was positively related with bulk density and redox potential indicating a larger bulk density and/or redox potential would correlate with a larger aboveground biomass for that site. Alternatively, organic matter, water content, presence of sulfide and salinity were all negatively correlated with aboveground biomass indicating a large value in any or all of those variables would indicate a small aboveground biomass for that site.

Table 4. 3 Backward stepwise regression of environmental and sediment characteristics principal components and above ground biomass

Biomass	Effect	Coefficient	Standard Error	Standard Coefficient	*P*
$R^2 = 0.179$	Constant	-2.949	0.102	0.000	0.000
SE = 0.664	P1	-0.199	0.104	-0.279	0.062
p-value = 0.021	P2	-0.227	0.104	-0.318	0.035

Chapter 5: Discussion and Conclusions

5.0 Introduction

The marsh pilot study was conducted to determine the appropriate depth for redox potential and salinity measurements and to determine how the measurements (redox potential, salinity, sulfide concentration, aboveground biomass) varied over time. The marsh extent study was conducted to determine variability across the marsh between the three drainage classes (well, moderate, and poor). It was also conducted to provide a dataset large enough to complete a principle component analysis to determine which environmental variables affected the production of above ground biomass.

5.1 Changes Over the Growing Season and Variability with Depth

5.1.1 Aboveground Biomass Production

Aboveground biomass production varied between the three drainage classes (well, moderate, and poorly) during the marsh pilot and marsh extent studies. Although the difference was found not to be significant during the marsh pilot study, a significant difference was found during the marsh extent study due to a larger data set for each drainage class. The variation in the aboveground biomass production between sites can be attributed to the plant species present (Figure 5.1; Appendix I) and the soil chemistry. Well drained sites were dominated by tall form *Spartina alterniflora* along the creek edge and *Spartina pectinta* and *Carex paleacea* along the upland edge. Moderately drained sites were dominated by a mixture of *Spartina patens* and *Juncus gerardii*. Poorly drained sites were dominated by short form *Spartina alterniflora*.

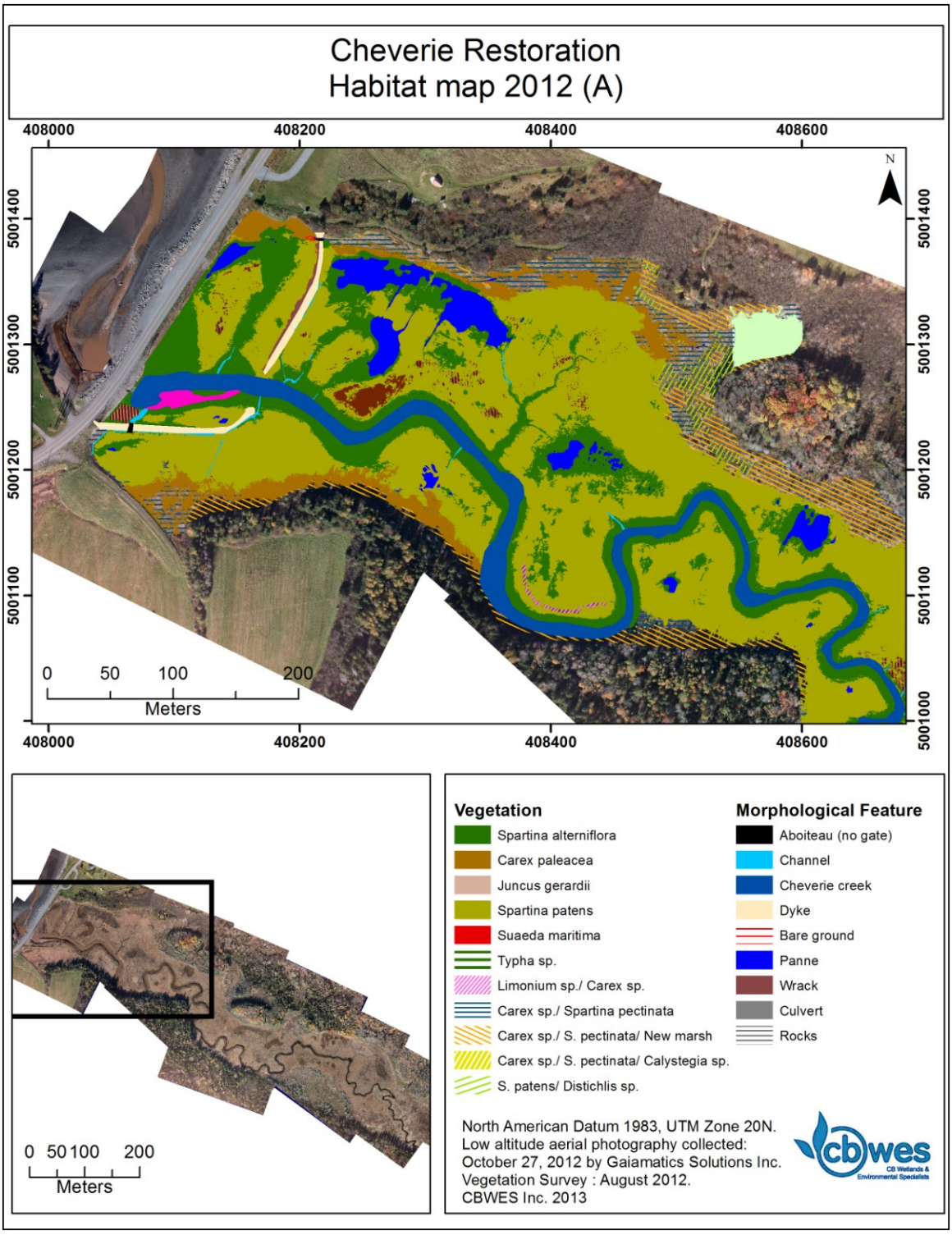

Figure 5. 1 Habitat map of Cheverie Creek from data collected in 2012. (Used with Permission from CBWES Inc.)

The height of *Spartina alterniflora* varied across the marsh surface in both the marsh pilot and marsh extent studies. The stands that were closest to the creek were noticeably taller than those located around the pannes in the high marsh. The areas closest to the creek were well drained with no water on the surface of the sediment and an absence of a sulfide smell. The areas surrounding the pannes were poorly drained with water at the surface of the sediment and a strong sulfide smell. Studies on *Spartina alterniflora* in both laboratory and field locations to determine the cause of the varying morphology (Howes, et al., 1981; Koch et al., 1989) have shown that the main limiting factors are high sulfide concentration and high salinity values (DeLaune et al., 1984; Buresh et al., 1980; Koch et al., 1990; Morris, 1980; Morris and Whiting, 1985). Some studies have linked the growth of *Spartina alterniflora* to tidal inundation, soil redox potential, ion toxicity and nutrient deficiency (DeLaune et al., 1984; Howes, et al., 1981). However, a marsh organ study conducted by Morris et al. (2002) found that biomass production peaked in areas of moderate to high inundation levels. The results of the present study would suggest that high sulfide and high salinity are the main limiting factors at Cheverie Creek, Nova Scotia.

Several studies have been conducted to determine how *Spartina patens* deals with waterlogging (Naidoo et al., 1992; Burdick et al., 1989; Burdick, 1989; Anastasiou and Brooks, 2003). *Spartina patens* was found to respond differently to salinity and flooding interactions as compared to *Spartina alterniflora* (Naidoo et al., 1992). *Spartina alterniflora* appears to be more tolerant of reduced conditions and less tolerant of high salinity compared with *Spartina patens* (Naidoo et al., 1992). In the present study, such variability in tolerance may correlate with the observed zonation of *Spartina* species across the marsh with *Spartina alterniflora* along the creek bank and panne areas, and *Spartina patens* in the mid and high marsh.

Low redox potential and high sulfide levels may interfere with the uptake of nitrogen in *Spartina patens* as shown in a study conducted in a brackish marsh along Bayou Faleau,

Lafourche Delta, USA (Burdick, et al., 1989). Salinity was not found to have significant influence on the uptake of nitrogen in *Spartina patens* but could still be an important factor (Burdick et al., 1989). Anastasiou and Brooks (2003) found that redox potential values between 0 and -60 mV caused moderate stress to *Spartina patens* plants. In the present study, *Spartina patens* was found growing in soils with redox potential values ranging from +50 to +300 mV, a range that would not have caused stress for these plants.

5.1.2 Sulfide Concentration

During the marsh pilot study, detectible amounts of sulfide were only found at the poorly drained sites and were found to increase over the sampling period. There were only two poorly drained sites sampled during the marsh pilot study (L5S3 & L7S5). The graph of sulfide concentration is shown for the poorly drained sites in Figure 4.12. Of the two sites, L5S3 was located close to the high marsh panne network and further away from any creek networks compared with L7S5. Therefore, L7S5 would be inundated by more tides than L5S3 leading to more exchange of pore water with the overlying tide water. The groundwater data showed a greater exchange in groundwater level close to the creek (low marsh) as compared with further into the marsh (high marsh). A study conducted on a marsh at Sapelo Island, Georgia also looked at natural movement of water within stands of tall and short form *Spartina alterniflora* (King et al., 1982). Significant movement vertically and laterally were found in the areas dominated by tall forms but no movement was found in the areas dominated by short form (King et al., 1982). The marsh extent study revealed a similar pattern between the drainage classes for sulfide concentration. Poorly drained sites contained high levels of sulfide as compared to moderately and well drained sites (Figure 4.13). The highest sulfide level (3.7mM) was found at L8S7 on August 14, 2014. A study conducted by Bradley and Dunn (1989) found that sulfide concentrations greater than 0.5 mM constrained biomass accumulation and plant elongation in

Spartina alterniflora. Koch et al. (1989) found that an exceedance of 1 mM could inhibit nitrogen uptake in *Spartina alterniflora*. In the present study, on August 14, 7 of the 13 poorly drained sites exceeded 1 mM sulfide and on August 21, 6 of 12 exceeded 1 mM. Therefore, a great deal of stress by high sulfide concentrations likely affected vegetation residing within the poorly drained sites of Cheverie Creek.

The sulfide concentration found at Cheverie Creek depends on where the sample was taken. The samples taken at the well drained sites along the creek edge and upland were either lacking sulfide concentration or had minimal amounts. The absence of sulfide is inconsistent with research conducted within the Gulf of Maine. Mora and Burdick (2013b) found detectible sulfide concentration ranging from 1.3 mM to 2.0 mM at all sampling sites. The difference may reflect a difference in substrate. The marshes studied by Mora and Burdick (2013a,b) were organogenic in origin as compared to Cheverie Creek which is minerogenic in origin. Minerogenic marshes have a high concentrations of iron and manganese due to the substrate (basalts) in these regions (Hung and Chmura, 2006) which can be used as an electron acceptor, obviating the need for microbes to use sulfate. In a Louisiana marsh, sites with higher levels of iron and lower level of sulfide exhibited higher biomass production (Webb et al., 1995). In the present study, the absence of sulfide concentration and higher aboveground biomass production was evident in well drained sites.

5.1.3 Redox Potential

The marsh pilot study revealed a significant difference in redox potential between drainage classes, neap versus spring, varying depth and depth and drainage class. Spring tides appeared to decrease the redox potential and neap tides increased redox potential. The spring/neap tidal signal can be explained by more water covering more of the high marsh for a longer period of time as compared with neap tides that would only come into the main tidal creek. The lack of

water on neap times would allow oxygen to penetrate into the soil matrix and raise the redox potential. A study conducted in a macrotidal site on the North Norfolk coast in the United Kingdom revealed a significant negative relationship between sediment water content and redox potential (Davy et al., 2011). This pattern was also found at Cheverie Creek as poorly drained sites had the lowest redox potential and the highest water content.

Well and moderately drained sites displayed a decrease in redox potential with depth that is consistent with results from studies in the Gulf of Maine (Howes et al., 1981) and Louisiana (Burdick et al., 1989). Highest redox potential values were measured in well drained sites which corresponded to Oxygen and Nitrate/Manganese as the dominate reduction reactions at the time of sampling throughout the study. Oxygen is unable to penetrate deep into the soil which is illustrated by the highest redox potential values being measured in the top 5 cm. Presence of water at soil surface decreases the diffusion of water into the underlying soil column. Thus, poorly drained sites which had water close to the soil surface, displayed lower redox potentials associated with stunted *Spartina alterniflora* than well and moderately drained sites. Sediments underlying stands of tall form *Spartina alterniflora* were linked to more oxidized sediment and root zone than those under the short form *Spartina alterniflora* (Howes et al., 1981). A similar pattern was found at Cheverie Creek. Creek edge sediments that were dominated by tall form *Spartina alterniflora,* had higher redox potential measurments than panne edge sediments dominated by short form *Spartina alterniflora*.

5.1.4 Salinity

The data collected during the marsh pilot study revealed that salinity was found to be significantly different between drainage class, neap versus spring, varying depth and varying depth and drainage class. Poorly drained sites were characterized by high salinity values as compared to well and moderately drained sites. The influence of spring and neap tides was

evident at each drainage class. Neap tides created higher salinities than spring tides which appeared to decrease salinity. Spring tides would cover the entirety of the marsh and allow the sediment to be flushed as compared to neap tides that would enter into the main tidal creek, overtopping the bank only on some of the higher neap tides. Sediment would dry out on neap tides as there is no water flooding the marsh platform. The meteorological conditions experienced during the relatively hot and dry summer led to increased evaporation, thus increasing salinity. Wilson and Morris (2012) determined that during spring tides, groundwater is flushed more than on neap tides and flushing is maximized on the marsh platform as this area is only inundated by spring tides. A significant difference in salinity was found with depth in the present study that was similar to the findings of Bertness et al. (1992) in a New England marsh, USA. Surface sediments also were found to have higher salinity values as compared to subsurface sediments as noted by Bertness et al. (1992).

A significant difference was identified between drainage classes in the marsh extent study but did not indicate a significant difference with neap and spring tides or when spring/neap tides and drainage class was tested. The variation in statistical significance may be due to the fact that only one spring tide and one neap tide were analyzed during the marsh extent study. A smaller variation between well and moderately drained sites may be linked to creek edge samples included in the study. These areas would have higher levels of salinity due to being flooded more frequently than locations on the marsh platform. The variability in salinity may also be linked to the elevation gradient from creek edge to upland. Bertness et al. (1992) found such a decrease in soil salinity with increasing elevation in New England, USA. Bare patches and lack of vegetation were found in poorly drained sites during the marsh scale study which correlated with high salinity values as has also been found in other studies in New England, USA (Bertness et al., 1992).

5.1.5 Sediment Characteristics

A significant increase in organic matter content with depth was found in the moderately and poorly drained sediment cores assessed for the marsh extent study. During the processing of the cores, extensive root material was found in the 5 to 10 cm range of cores collected from poor and moderately drained sites. The 5 to 10 cm range was also associated with highest water content for moderately and poorly drained sites. It appears that fibrous root material holds onto water within these zones. The increase in organic matter with depth in moderately and poorly drained cores is opposite to results found by Tempest et al. (2015) on restored salt marshes within a macrotidal system, Blackwater Estuary, England. Tempest et al. (2015) found a dramatic decrease in organic matter at the 5 cm mark where as an increase was found in the present study. No change was found in the well drained sites and values are lower than those measured in Blackwater Estuary (Tempest et al., 2015).

The marsh pilot study revealed a significant difference in water content with depth and significant interaction with depth and drainage class as compared to the marsh extent study that revealed a significant difference with drainage class, depth and significant interaction between depth and drainage class. The larger sample size of the marsh extent study provided more sample points in poorly drained sites which would allow for a significant difference varying with depth. Highest water content was found in poorly drained sites that were dominated by short form *Spartina alterniflora*, similar to results found by Howes et al. (1981).

Bulk density increased significantly with depth at all well drained sites based on data collected during both pilot marsh and marsh extent studies. Moderately and poorly drained sites experienced a decrease in bulk density from 0 to 10 cm but showed an increased with depth after 10 cm. The pattern for bulk density at well drained sites is similar data collected at Orplands Farm managed realignment site (Tempest et al., 2015). However, the pattern found at poorly and

moderately drained sites may be due to land use prior to restoration. The heavy agricultural use prior to restoration at the managed realignment site may have compacted the soil more than at Cheverie Creek leading to higher bulk density at Orplands Farm managed realignment site (Tempest et al., 2015).

Grain size analysis determined larger sand to silt/clay ratio occurred along creek areas at Cheverie Creek within 0 to 5 cm as compared to areas further in on the marsh platform. The variation in sand to silt/clay ratio is related to the inability for tidal water to carry larger particles further into the marsh (Allen, 2000; Bartholdy, 2012). Larger particles fall out of suspension closer to tidal source than finer particles that are brought further onto the marsh platform. The well drained sites analyzed for the pilot marsh study were located along the upland leading to a larger distance from tidal creek ultimately decreasing the sand to silt sand ratio experienced along the upland. Similar ratios were experienced by all drainage classes in 5 to 10 cm range. The similarity may be due to tidal restriction caused by decreased culvert size and time associated with soil development. The replacement of the culvert allows greater amount of sediment to enter the system and with more velocity, thus allowing larger particles to travel further into the system increasing sand to silt clay ratio. Similar sand to silt clay ratios were found in marshes in Gulf of Maine (Mora and Burdick, 2013b).

5.2 Relationship between soil chemistry and sediment on aboveground biomass production

The PCA combined with the backward stepwise regression determined that sediment characteristics and soil chemistry combined as two principal components providing the best indicators of aboveground biomass production at Cheverie Creek in the Summer of 2014. These relationships have been found in other salt marshes in New England (Mora and Burdick, 2013a,b; Portnoy, 1999), Atlantic Coast (Teal, 1962), Gulf Coast (Mendelssohn and Postek, 1982) and England (Tempest et al., 2015).

The data collected at Cheverie Creek revealed a positive relationship between aboveground biomass production and bulk density at 0 to 5 and 5 to 10 cm and redox potential at 5 cm. The analysis revealed a negative relationship between aboveground biomass production and organic matter and water content at 0 to 5 cm and 5 to 10 cm, salinity at 5 cm, and presence of sulfide.

A study conducted at Great Sippewissett Salt Marsh, Cape Cod, USA revealed that redox potential was positively correlated with productivity and soil water movement (Howes et al., 1981) and also with plant height in a salt marsh in Barataria Bay, Louisiana, USA (Mendelssohn et al., 1981). Bulk density values are inversely related to organic matter (Bowron et al., 2009), so aboveground biomass production should be positively correlated with bulk density, an indication of low organic matter. Low organic matter decreases microbial activity thus is associated with higher redox potential (Reddy and DeLaune, 2008).

Water content was negatively correlated with aboveground biomass production because as water content increases, redox potential decreases indicating that microbial communities are using alternative electron acceptors that give them less and less energy. The production of sulfide was negatively correlated with aboveground biomass which was expected as it inhibits among other things, the uptake of nitrogen (Bradley and Dunn, 1989; DeLaune et al., 1984; Buresh et al., 1980; Koch et al., 1990; Morris, 1980; Morris and Whiting, 1985).

Salinity also had a negative relationship with aboveground biomass production in the present study as found in other studies (Burdick et al., 1989; DeLaune et al., 1984; Buresh et al., 1980; Koch et al., 1990; Morris, 1980; Morris and Whiting, 1985).

Wiegert et al. (1983) conducted a field manipulation at a site dominated by intermediate height *Spartina alterniflora*, on Sapelo Island, Georgia. They found an increase in the productivity of *Spartina alterniflora* in areas where there was a decrease in salinity, sulfide

concentration and increased oxygen availability. Increased exchange of interstitial water and the resultant increased productivity of *Spartina alterniflora*, supports the hypothesis that exchange between tidal and interstitial water influences the productivity of the plant (Wiegert et al., 1983; Howes et al. 1981).

The measurements made at Cheverie Creek in 2014 indicate intricate interactions between sediment characteristics, soil chemistry, water content, microbial activity, and aboveground biomass over time. Cheverie Creek is characterized by several panne areas that developed due to a decrease in elevation along the marsh platform. These fill with water during spring tides (Bowron, et al., 2009). In addition, an elevation plateau developed at Cheverie as the marsh developed after reintroduction of tidal water from the installation of the larger culvert. Pools of water were observed along transects 3 to 5 prior to restoration which dried up as the summer season progressed in the first year following tidal restoration (Bowron et al., 2013). Post restoration monitoring revealed that this area and another area along transects 7 to 8 began to retain more water through the growing season (Bowron et al., 2013). Rod Sediment Elevation Tables (RSETs) and marker horizons were installed immediately post-restoration to monitor elevation change and to determine if the change was due to sediment accretion or belowground processes. Over the 7 year monitoring program, changes in sediment accretion with some change in belowground processes at RSET 3 and RSET 4 were documented (Bowron et al., 2013). At RSET 3 there was a net subsurface subsidence over the 7 year monitoring program (Bowron et al., 2013) indicating subsidence was occurring faster than accretion. In contrast, RSET 4 displayed a slight subsidence. The subsidence evident at RSET 3 and 4 can be explained by the soil chemistry and aboveground biomass data in this study (Figure 5.2). In areas of poor drainage, such as around and throughout the extensive panne system, biomass production was the lowest matched with high sulfide concentration and high salinity. In these areas, the vegetation struggles to

survive and grow. The root structures bond to sediment and help to maintain belowground stability. The microbial community uses the organic matter provided by the roots as an energy source. Depending on the amount of water saturation and reduction occurring, the microbial community will get less energy as is evident in low redox potential measurements. If the vegetation can no longer grow then the roots will begin to die and degrade. Subsidence will occur as measured by the RSETs in the area. Spring tides inundate the area and bring in more water, which becomes more saline and the waterlogging decreases redox and increases sulfide levels, which leads to a decrease in aboveground biomass production.

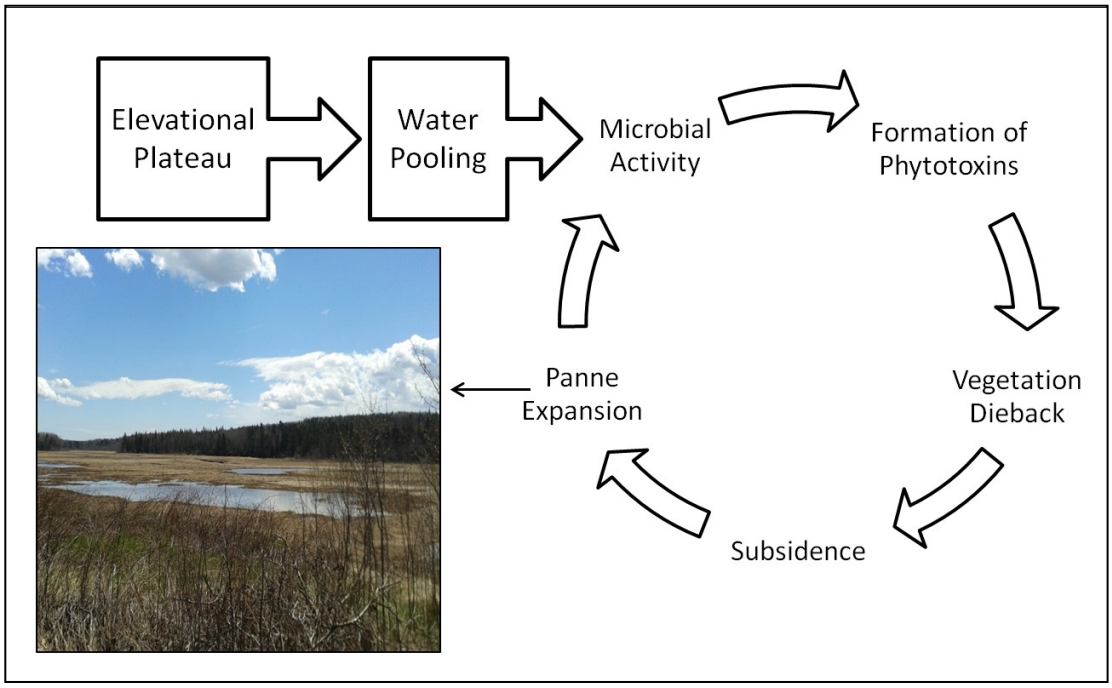

Figure 5. 2 Schematic of panne development at Cheverie Creek (Photo of large panne between transect 3 & 5 (Taken by C. Skinner, 2014))

5.3 Conclusions and Future Directions

The purpose of this study was to document the changes in soil chemistry and aboveground biomass over a single growing season at Cheverie Creek, Nova Scotia and to determine which variables (soil chemistry, sediment characteristics, or inundation frequency/time) were able to influence aboveground biomass production. Hydrology exerted the greatest influence

on salinity and redox potential measurements whereas sulfide concentration increased through the growing season. Tidal water flooding and waterlogging some areas of the marsh surface gave rise to pannes. Plants that were able to grow around the shallower portions of the pannes were stunted and these areas had the lowest aboveground biomass production, highest salinity, high sulfide and low redox potential. Sediment bulk properties also influenced aboveground biomass production and provided the basis for predicting soil chemistry at each location. Sites with high organic matter were associated with high salinity, low redox and high sulfide concentration which were associated with lower aboveground biomass production.

Overall, the data collected at Cheverie Creek is similar to data from salt marshes in other parts of North America and the UK. The data also provided an understanding of how aboveground biomass production was influenced over the growing season and may help to provide recommendation for future restoration programs in the Bay of Fundy. However, the Bay of Fundy marshes differ from the Atlantic marshes as they are minerogenic in origin (Figure 5.3). Iron and manganese found in these minerogenic marshes decreases the impact of waterlogging on aboveground biomass production. Therefore, organogenic marshes should be managed differently depending on the type of restoration required. Elevation and appropriate drainage would be more critical in organogenic marshes to ensure halophytic vegetation are able to colonize the marsh surface and thrive. With inappropriate drainage and elevation, organogenic marshes are more susceptible to sulfide production and increased sulfide toxicity (Figure 5.3). It may also suggest that under accelerated sea level rise, minerogenic marshes, due to the increase availability of iron and manganese, could be more resilient than organogenic marshes if sediment supply and ability to encroach on the upland are still available.

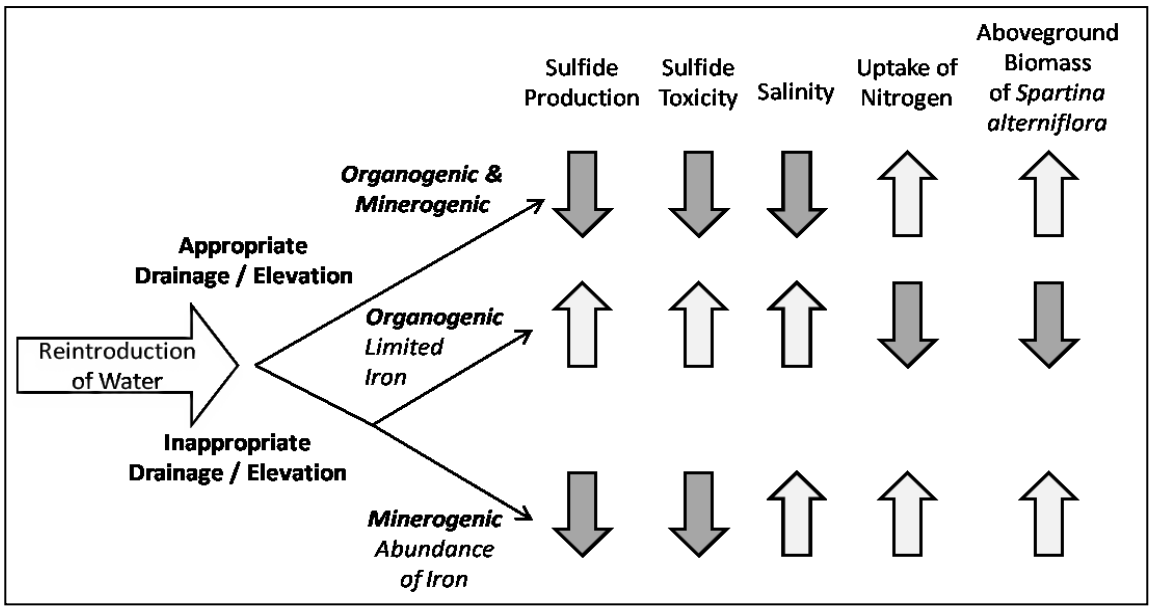

Figure 5. 3. Differences after reintroduction of tidal water between minerogenic and organogenic salt marshes.

Further studies to quantify iron and manganese levels in the sediment should be undertaken in marshes throughout the Bay of Fundy and possibly the Atlantic coast. This would enable a better understanding of how vegetation is affected by sediment chemistry and sediment characteristics. A further understanding of the importance of iron and manganese in these systems would allow restoration professionals a better understanding of how the marsh will develop without human intervention. Especially in difficult restoration projects that require manipulation of the marsh platform or planting of vegetation

Future studies should also incorporate more devices (i.e. leveloggers) in groundwater wells distributed throughout the marsh along with monitors that measure salinity. This would enable more effective modeling of groundwater movement and exchange. Salt marshes are influenced by tidal flows but are also strongly influenced by freshwater input from upland sources. Therefore, some of the variability in salinity values across the marsh are almost certainly linked to freshwater sources. Additional monitoring of these aspects and others at Cheverie Creek

over more seasons would result in more information could better predict spatial variation in biomass production and lead to a better overall understanding of salt marshes throughout the Maritimes.

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Appendix I. Aboveground Biomass Composition and Values

Table A-1. Aboveground biomass composition and value for marsh pilot study (Values are in $g \cdot cm^{-2}$)

Sample ID	21-May-14	04-Jun-14	16-Jun-14	07-Jul-14	18-Jul-14	04-Aug-14	Drainage Class	Dominant Species
L4S1	0.045	0.056	0.036	0.097	0.069	0.031	Moderately	*Carex paleacea*
L4S7	0.008	0.012	0.011	0.020	0.044	0.033	Moderately	*Spartina patens* & *Distichlis spicata*
L5S3	0.008	0.020	0.024	0.026	0.032	0.037	Poor	*Spartina alterniflora* & *Salicornia europaea*
L6S1	0.045	0.061	0.031	0.077	0.117	0.053	Well	*Carex paleacea* & *Spartina pectinata*
L6S8	0.009	0.005	0.014	0.028	0.025	0.036	Moderately	*Spartina patens* & *Distichlis spicata*
L7S5	0.005	0.010	0.006	0.014	0.037	0.020	Poor	*Spartina alterniflora*
L7S12	0.010	0.008	0.020	0.007	0.047	0.042	Moderately	*Spartina patens* & *Distichlis spicata*
L8S1	0.009	0.011	0.016	0.038	0.076	0.065	Well	*Spartina pectinata*
L8S10	0.004	0.001	0.006	0.028	0.042	0.043	Well	*Spartina alterniflora* & *Spartina patens*

Table A-2. Aboveground biomass composition and value for marsh extent study (Values are in g•cm -2)

Sample ID	14-Aug-14	21-Aug-14	Dominant Species
L4S1	0.077	0.032	*Carex paleacea*
L4S2	0.033	0.018	*Spartina alterniflora*
L4S3	0.035	0.025	*Spartina alterniflora & Spartina patens*
L4S6	0.090	0.172	*Spartina patens*
L4S7	0.056	0.024	*Spartina patens & Distichlis spicata*
L4S8	0.082	0.098	*Spartina alterniflora*
L5S1	0.118	0.064	*Spartina alterniflora*
L5S3	0.030	0.044	*Spartina alterniflora & Salicornia europaea*
L5S4	0.056	No Data	*Spartina alterniflora*
L5S5	0.048	0.037	*Spartina patens*
L5S6	0.094	0.097	*Spartina alterniflora*
L6S1	0.100	0.028	*Carex paleacea & Spartina pectinata*
L6S3	0.024	0.033	*Spartina patens*
L6S4	0.023	0.021	*Spartina alterniflora & Spartina patens*
L6S5	0.047	0.050	*Spartina patens*
L6S6	0.034	0.031	*Spartina patens*
L6S7	0.075	0.061	*Spartina patens & Distichlis spicata*
L6S8	0.033	0.039	*Spartina patens & Distichlis spicata*
L6S9	0.086	No Data	*Spartina alterniflora*
L7S1	0.048	0.064	*Spartina patens*
L7S2	0.071	0.109	*Spartina patens*
L7S3	0.088	0.057	*Spartina patens*
L7S4	0.049	0.050	*Spartina alterniflora & Spartina patens*
L7S5	0.007	0.012	*Spartina alterniflora*
L7S6	0.066	0.032	*Spartina alterniflora*
L7S9	0.008	No Data	*Spartina patens*
L7S10	0.095	0.050	*Spartina patens*
L7S11	0.103	No Data	*Spartina patens*
L7S12	0.056	0.051	*Spartina patens & Distichlis spicata*
L7S13	0.032	0.043	*Spartina patens & Distichlis spicata*
L7S14	0.064	0.037	*Spartina alterniflora*
L8S1	0.054	0.042	*Spartina pectinata*
L8S2	0.102	0.085	*Spartina patens*
L8S3	0.088	0.038	*Spartina patens*
L8S4	0.086	0.102	*Spartina patens*
L8S5	0.061	0.076	*Spartina patens*

Table A-2 Cont'd. Aboveground biomass composition and value for marsh extent study (Values are in g•cm -2)

Sample ID	14-Aug-14	21-Aug-14	Dominant Species
L8S6	0.032	0.058	*Spartina patens*
L8S7	0.006	0.015	*Distichlis spicata (Mostly bare patch)*
L8S9	0.079	No Data	*Spartina alterniflora & Spartina patens*
L8S10	0.078	0.032	*Spartina alterniflora & Spartina patens*
L8S11	0.098	0.068	*Spartina alterniflora*
L8S12	0.054	0.036	*Spartina alterniflora & Spartina patens*
L8S13	0.051	0.056	*Spartina patens*
L8S14	0.087	0.060	*Spartina alterniflora*

Appendix II. Soil Chemistry

Table A-3. Soil chemistry data from marsh pilot study.

Date	Sample ID	Drainage Class	Salinity (ppt)				Redox Potential (mv)				Sulfide Concentration (mM)
			5 cm	10 cm	15 cm	20 cm	5 cm	10 cm	15 cm	20 cm	
21-May-14	L4S1	Moderate	4.07	3.80	2.20	1.60	169.3	203.0	148.2	139.6	0.00
	L4S7	Moderate	5.15	5.23	5.75	6.55	325.3	278.5	310.1	159.5	0.00
	L5S3	Poor	12.60	12.79	10.98	9.56	77.8	40.4	62.5	80.7	0.48
	L6S1	Well	0.07	0.07	0.06	0.07	590.6	336.0	395.4	356.9	0.00
	L6S8	Moderate	7.18	4.71	4.27	4.92	160.7	220.4	140.0	230.8	0.05
	L7S5	Poor	11.92	12.93	10.29	8.57	-53.4	-85.5	106.3	84.1	0.00
	L7S12	Moderate	8.25	7.14	6.71	6.92	-151.9	6.2	16.2	-12.0	0.00
	L8S1	Well	2.63	2.26	1.91	1.50	173.4	129.6	102.1	93.5	0.00
	L8S11	Well	6.68	7.02	6.59	5.84	362.1	105.2	178.8	114.9	0.00
04-Jun-14	L4S1	Moderate	4.89	4.77	4.26	2.67	144.5	212.9	93.6	130.4	0.00
	L4S7	Moderate	8.12	6.20	6.11	6.31	275.5	193.5	80.0	142.6	0.00
	L5S3	Poor	11.64	15.49	11.85	9.85	-7.8	45.7	252.9	105.9	0.16
	L6S1	Well	1.58	0.09	0.07	0.79	493.7	209.0	494.2	242.8	0.00
	L6S8	Moderate	6.05	5.45	4.97	4.96	294.3	214.6	131.5	242.9	0.00
	L7S5	Poor	12.73	14.13	17.66	9.68	51.1	44.0	10.1	113.8	0.00
	L7S12	Moderate	7.84	6.21	5.22	6.87	253.1	137.3	-60.6	-19.2	0.00
	L8S1	Well	3.38	2.32	2.11	1.48	414.4	187.0	155.1	150.1	0.00
	L8S11	Well	7.82	7.09	7.20	6.68	398.0	289.0	90.6	-309.7	0.00

Table A-3. Cont'd. Soil chemistry data from marsh pilot study.

Date	Sample ID	Drainage Class	Salinity (ppt)				Redox Potential (mv)				Sulfide Concentration (mM)
			5 cm	10 cm	15 cm	20 cm	5 cm	10 cm	15 cm	20 cm	
16-Jun-14	L4S1	Moderate	5.61	5.28	3.94	2.67	179.2	177.9	211.7	53.0	0.00
	L4S7	Moderate	7.04	7.61	5.17	7.16	257.8	277.5	134.1	135.6	0.25
	L5S3	Poor	12.81	14.12	9.80	8.86	8.0	36.3	-42.6	107.2	0.44
	L6S1	Well	1.49	1.14	0.67	0.31	412.8	344.7	374.0	349.6	0.00
	L6S8	Moderate	6.54	6.12	4.73	4.75	53.3	181.9	143.6	105.8	0.05
	L7S5	Poor	12.55	12.38	13.56	10.39	159.4	273.1	-420.4	48.1	0.18
	L7S12	Moderate	6.89	8.62	5.94	6.37	202.4	46.3	125.0	0.2	0.03
	L8S1	Well	5.96	4.73	3.82	2.04	187.3	54.4	122.8	146.4	0.00
	L8S11	Well	7.20	7.56	8.00	5.57	295.4	247.4	133.1	187.0	0.00
07-Jul-14	L4S1	Moderate	4.45	5.58	6.02	4.39	177.8	110.2	132.9	-1.7	0.00
	L4S7	Moderate	9.95	5.68	5.67	5.25	389.8	388.7	297.8	168.3	0.00
	L5S3	Poor	13.27	16.73	11.50	10.20	8.8	-23.5	-22.2	-20.1	0.81
	L6S1	Well	1.25	1.43	1.14	0.78	529.2	443.3	398.4	434.4	0.00
	L6S8	Moderate	7.17	6.27	5.49	5.01	188.4	262.2	238.1	67.8	0.04
	L7S5	Poor	12.32	15.33	11.08	9.44	227.6	64.7	11.5	2.7	0.32
	L7S12	Moderate	7.95	4.73	5.78	6.33	163.7	359.5	198.7	123.5	0.00
	L8S1	Well	4.17	3.56	1.61	2.34	433.7	392.6	344.4	240.8	0.00
	L8S11	Well	6.74	7.35	5.35	5.04	326.0	375.4	422.5	311.8	0.00

Table A-3. Cont'd. Soil chemistry data from marsh pilot study.

Date	Sample ID	Drainage Class	Salinity (ppt)				Redox Potential (mv)				Sulfide Concentration (mM)
			5 cm	10 cm	15 cm	20 cm	5 cm	10 cm	15 cm	20 cm	
18-Jul-14	L4S1	Moderate	7.14	7.39	6.02	3.26	127.9	39.2	123.3	150.4	0.00
	L4S7	Moderate	9.13	6.89	6.90	5.58	196.4	201.1	329.0	177.9	0.00
	L5S3	Poor	15.26	12.33	14.32	10.09	8.6	-10.9	-38.2	0.9	1.96
	L6S1	Well	2.55	2.07	2.05	1.31	256.0	202.2	313.7	258.8	0.00
	L6S8	Moderate	8.66	6.36	5.54	5.66	217.5	185.8	119.9	-18.8	0.00
	L7S5	Poor	10.80	11.98	14.80	10.74	81.2	43.0	10.0	18.1	0.00
	L7S12	Moderate	11.72	8.78	5.93	6.67	42.6	95.9	-19.9	69.9	0.00
	L8S1	Well	4.45	3.50	3.89	3.27	220.0	128.1	165.5	66.4	0.00
	L8S11	Well	7.65	8.59	8.52	5.94	147.6	84.0	163.4	102.1	0.00
04-Aug-14	L4S1	Moderate	7.46	8.56	7.53	6.56	156.8	11.3	14.0	80.5	0.00
	L4S7	Moderate	13.93	11.83	10.46	9.00	481.0	355.6	295.8	243.3	0.00
	L5S3	Poor	20.89	20.12	17.20	18.82	-18.0	8.2	-39.3	-72.4	2.56
	L6S1	Well	6.46	6.26	5.71	4.89	436.0	475.8	416.9	256.3	0.00
	L6S8	Moderate	13.01	10.86	9.39	8.69	249.0	171.7	189.1	59.1	0.00
	L7S5	Poor	21.70	21.05	19.61	16.45	43.0	-55.3	180.4	289.0	0.73
	L7S12	Moderate	12.97	10.73	9.28	10.14	205.9	231.5	86.8	135.4	0.00
	L8S1	Well	7.63	8.17	8.67	5.85	446.2	373.7	294.7	258.4	0.00
	L8S11	Well	12.77	15.03	13.79	11.44	404.1	199.3	147.5	117.4	0.00

Table A-4. Soil chemistry data from marsh extent study.

Date		August 14, 2014 (Spring)			August 21, 2014 (Neap)		
PtID	Drainage Class	Salinity (ppt)	Redox Potential (mV)	Sulfide Concentration (mM)	Salinity (ppt)	Redox Potential (mV)	Sulfide Concentration (mM)
L4S1	Moderate	9.21	30.73	1.04	8.72	27.97	0.00
L4S2	Poor	16.16	47.77	0.00	16.89	23.13	0.03
L4S3	Poor	15.55	-36.30	0.14	18.71	59.30	0.42
L4S6	Moderate	11.62	154.13	0.00	11.96	122.23	0.00
L4S7	Moderate	12.81	253.07	0.00	12.22	247.73	0.00
L4S8	Well	10.27	260.17	0.00	11.90	286.57	0.00
L5S1	Poor	16.77	-36.33	2.12	16.76	3.43	2.42
L5S3	Poor	19.74	-29.37	1.29	18.80	-26.57	No Data
L5S4	Poor	21.17	-52.90	2.96	20.40	-99.87	3.32
L5S5	Moderate	13.34	190.17	0.00	18.07	176.57	0.00
L5S6	Well	13.72	286.53	0.00	13.24	191.27	0.00
L6S1	Well	7.96	380.50	0.00	6.75	395.60	0.00
L6S3	Poor	18.20	112.50	0.00	13.25	134.43	0.00
L6S4	Moderate	22.73	-25.83	0.22	18.73	-7.37	1.91
L6S5	Poor	19.85	102.90	0.11	14.42	50.50	0.00
L6S6	Moderate	17.09	-32.60	0.00	17.97	17.83	0.00
L6S7	Moderate	14.15	50.17	0.00	15.28	110.87	0.00
L6S8	Moderate	11.85	104.33	0.00	11.35	282.07	0.00
L6S9	Well	13.90	105.47	0.11	12.37	45.70	0.00
L7S1	Moderate	11.28	108.70	0.00	18.54	174.50	0.00
L7S2	Moderate	16.62	52.83	0.00	16.47	161.57	0.00

Table A-4 Cont'd. Soil chemistry data from marsh extent study.

Date		August 14, 2014 (Spring)			August 21, 2014 (Neap)		
PtID	Drainage Class	Salinity (ppt)	Redox Potential (mV)	Sulfide Concentration (mM)	Salinity (ppt)	Redox Potential (mV)	Sulfide Concentration (mM)
L7S3	Moderate	16.81	158.23	0.00	17.43	246.20	0.00
L7S4	Moderate	16.66	233.00	0.00	No Data	166.77	0.00
L7S5	Poor	21.25	49.20	0.00	17.28	75.57	0.18
L7S6	Poor	21.10	-76.40	3.05	17.73	-51.97	3.36
L7S9	Poor	16.82	-64.50	1.12	16.93	-32.50	0.64
L7S10	Moderate	15.21	26.30	0.00	11.84	160.53	0.00
L7S11	Moderate	17.12	276.00	0.00	13.50	147.63	0.00
L7S12	Moderate	11.00	135.00	0.00	13.67	107.60	0.00
L7S13	Moderate	13.77	210.63	0.00	13.49	230.47	0.00
L7S14	Well	12.50	288.03	0.00	12.45	167.87	0.00
L8S1	Well	9.62	159.33	0.00	7.98	325.73	0.00
L8S2	Moderate	12.45	70.93	0.22	15.93	23.37	0.13
L8S3	Poor	14.87	16.63	1.70	14.71	-46.80	1.14
L8S4	Moderate	16.84	138.37	0.00	17.12	132.03	0.00
L8S5	Moderate	17.88	154.47	0.00	16.02	94.07	0.00
L8S6	Moderate	18.85	9.00	0.52	17.36	77.70	0.14
L8S7	Poor	19.41	-35.63	3.70	18.38	-26.33	1.73
L8S9	Poor	16.46	27.10	0.98	16.89	-15.10	1.07
L8S10	Well	10.79	167.73	0.00	11.45	181.43	0.00
L8S11	Well	12.61	77.63	0.00	11.65	289.30	0.00
L8S12	Moderate	14.78	154.00	0.00	15.13	268.97	0.00

Table A-4 Cont'd. Soil chemistry data from marsh extent study.

Date		August 14, 2014 (Spring)			August 21, 2014 (Neap)		
PtID	Drainage Class	Salinity (ppt)	Redox Potential (mV)	Sulfide Concentration (mM)	Salinity (ppt)	Redox Potential (mV)	Sulfide Concentration (mM)
L8S13	Moderate	13.50	160.67	0.00	13.63	159.10	0.00
L8S14	Well	14.01	66.83	0.00	13.16	195.83	0.00

Appendix III. Methodology used for sulfide concentration

Methodology from David Burdick (May 25, 1994)

Sulfides - fix at once.

1. Measure out 12 mL 2% ZnAc into the appropriate number of scintillation vials. Use 12.4 mL for high sulfide water, e.g. *Spartina* and *Scirpus* cores. Don't forget an extra for a reagent blank, and one extra per sample to be used as a colour blank.

2. Collect 0.5 mL in syringe from microcosm port and transfer to one of the scintillation vials without air.

> Note: If high sulfide is expected (>2.5 mM), use 0.1 mL porewater in 12.4 mL ZnAc. Correct for dilution later (in this case 0.5/0.1 = 5X dilution).*

3. Add 10 mL DPDH dye solution and quickly cap to prevent loss of H_2S.

4. Shake the vial and store in dark 20 min, no longer than 2 hr to let colour develop.

5. Read absorbance at 670 nm using 1 cm light path. Set zero absorbance with reagent blank = .5 mL DI water + 12 mL ZnAc + 10 mL DPDH.

6. Read absorbance of colour blank, i.e. degassed sample (from anion aliquot) treated sample.

7. Calculate conc.:

> Conc. (mM) =
>
> $(Abs_{sample} - Abs_{colour\ blank} / 0.542)$ X dilution factor*

Dye recipe:

> To 1 liter vol. flask
>
> > Add about 300 mL H_20
> >
> > Plus 496 mL conc. HCl
> >
> > Then add 3.728 g DPDH (p-Aminodimethytaniline, Sigma Chemical); TOXIC! Use dust mask!
> >
> > Plus 6.0 g [$FeCl_3 \cdot 6H_20$]
> >
> > Fill to mark with distilled water

Appendix IV. Permissions

Dear Tony and Nancy:

I am completing a Master's thesis at Saint Mary's University, Halifax, Nova Scotia entitled "Biogeochemistry of a recently restored macrotidal salt marsh: Cheverie Creek Restoration Site, Nova Scotia, CA". I am requesting the use of the mosaic and habitat map that were created in 2012 and presented within the 2013 monitoring report[1] for Cheverie Creek.

The requested permission extends to any future revisions of my thesis, to the public circulation of my thesis by the Saint Mary's University Library, and to the prospective reproduction of the thesis by the National Library of Canada or its agents.

If these arrangements meet with your approval, please sign this letter where indicated below and return a scanned signed copy to me. Thank you.

Sincerely,

Christa Skinner

PERMISSION GRANTED FOR THE USE REQUESTED ABOVE:

CB Wetlands and Environmental Specialists

Date: June 1, 2015

[1] Bowron, T., Neatt, N., Graham, J., van Proosdij, D., Lundholm, J., and Lemieux, B. 2013. Post-restoration monitoring (year 7) of the Cheverie Creek Salt Marsh Restoration Project. A report for Nova Scotia Department of Transportation and Infrastructure Renewal.

CB Wetlands and Environmental Specialists
331 Newbury Road
Hammonds Plains, Nova Scotia
B4B 0C6

July 8, 2015

Dear Tony and Nancy:

I am completing a Master's thesis at Saint Mary's University, Halifax, Nova Scotia entitled "Biogeochemistry of a recently restored macrotidal salt marsh: Cheverie Creek Restoration Site, Nova Scotia, CA". I am requesting the use of the comparison photos of the wooden box culvert and new larger culvert (Figure 3; pg 5) that was included within the 2013 monitoring report[1] for Cheverie Creek.

The requested permission extends to any future revisions of my thesis, to the public circulation of my thesis by the Saint Mary's University Library, and to the prospective reproduction of the thesis by the National Library of Canada or its agents.

If these arrangements meet with your approval, please sign this letter where indicated below and return a scanned signed copy to me. Thank you.

Sincerely,

Christa Skinner

PERMISSION GRANTED FOR THE USE REQUESTED ABOVE:

Nancy C. Neatt

CB Wetlands and Environmental Specialists

Date: July 9, 2015

[1] Bowron, T., Neatt, N., Graham, J., van Proosdij, D., Lundholm, J., and Lemieux, B. 2013. Post-restoration monitoring (year 7) of the Cheverie Creek Salt Marsh Restoration Project. A report for Nova Scotia Department of Transportation and Infrastructure Renewal.

CB Wetlands and Environmental Specialists

April 22, 2016

331 Newbury Road

Hammonds Plains, Nova Scotia

B4B 0C6

Dear Tony Bowron and Nancy Neatt,

I am completing a Master of Science thesis at Saint Mary's University, Halifax, Nova Scotia entitled "The biogeochemistry of a restoring macrotidal salt marsh, Cheverie Creek Restoration Site, Nova Scotia". I am requesting the use of the habitat map constructed in 2012 for Cheverie (Figure 23a; pg 37) that was included within the 2013 monitoring report[1] for Cheverie Creek.

The requested permission extends to any future revisions of my thesis, to the public circulation of my thesis by the Saint Mary's University Library, and the prospective reproduction of the thesis by the National Library of Canada or its agents.

If these arrangements meet with your approval, please sign this letter where indicated below and return a scanned copy to me. Thank you.

Sincerely,

Christa Skinner

PERMISSION GRANTED FOR THE USE REQUESTED ABOVE:

CB Wetlands and Environmental Specialists

Date: 25/4/16

[1] Bowron, T., Neatt, N., Graham, J., van Proosdij, D., Lundholm, J., and Lemieux, B. 2013. Post-restoration monitoring (year 7) of Cheverie Creek Salt Marsh Restoration Project. A report for Nova Scotia Department of Transportation and Infrastructure Renewal.

CB Wetlands and Environmental Specialists

April 22, 2016

331 Newbury Road

Hammonds Plains, Nova Scotia

B4B 0C6

Dear Tony Bowron and Nancy Neatt,

I am completing a Master of Science thesis at Saint Mary's University, Halifax, Nova Scotia entitled "The biogeochemistry of a restoring macrotidal salt marsh, Cheverie Creek Restoration Site, Nova Scotia". I am requesting the use of the habitat map constructed in 2012 for Cheverie (Figure 23a; pg 37) that was included within the 2013 monitoring report[1] for Cheverie Creek.

The requested permission extends to any future revisions of my thesis, to the public circulation of my thesis by the Saint Mary's University Library, and the prospective reproduction of the thesis by the National Library of Canada or its agents.

If these arrangements meet with your approval, please sign this letter where indicated below and return a scanned copy to me. Thank you.

Sincerely,

Christa Skinner

PERMISSION GRANTED FOR THE USE REQUESTED ABOVE:

CB Wetlands and Environmental Specialists

Date: 25/4/16

[1] Bowron, T., Neatt, N., Graham, J., van Proosdij, D., Lundholm, J., and Lemieux, B. 2013. Post-restoration monitoring (year 7) of Cheverie Creek Salt Marsh Restoration Project. A report for Nova Scotia Department of Transportation and Infrastructure Renewal.