THE CONTRIBUTION OF SEDIMENTATION AND TIDAL INUNDATION IN
FACILITATING THE RECOVERY OF COASTAL WETLAND VEGETATION
FOLLOWING HYDROLOGIC RESTORATION

by

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ABSTRACT

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This project uses a multi-scaled approach to better understand factors driving vegetation recovery on a hydrologically restored agricultural dyke land. The objectives were to develop a simple GIS classification technique of aerial photographs to examine marsh wide restoration response; examine the relationship between tidal inundation, sediments and vegetation recovery. During the first growing season, the site was marked by a decrease in vegetation surface cover. Annual elevation surveys recorded high rates of sedimentation likely killing off non-tolerant vegetation and formed large areas of bare ground. During the second growing season an almost complete recovery in vegetation. Bare ground plots became covered with brackish or salt marsh vegetation. During the first growing season, sedimentation acted as a disturbance agent. In the second growing season, bare ground patches were colonized by brackish or salt marsh vegetation. This study highlights factors driving recovery in the early stages of restoration.

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CHAPTER 1: INTRODUCTION

1.0 Introduction

Forming some of the most fertile and accessible coastal habitats on earth, tidal marshes provide important ecosystem services, such as buffering coastal zones from storm damage, to providing habitat for ecologically sensitive species, and acting as nurseries for many commercially important fish species (McKinney et al. 2009; Craft et al. 2009; Gedan et al. 2009; Więski et al. 2009). Impacts to tidal marshes can be direct, such as the construction of dykes, weirs and levees, or indirect, where hydrological modifications by construction of water control structures, impoundments and dams limit tidal flow from entering a site (Kennish 2001; Doody 2004). Human activities on the tidal wetlands, such as the building of dykes, weirs and water control structures have severely reduced the extent of tidal wetlands (Doody 2004; Bakker et al. 2002; Warren et al. 2002; Bromberg and Bertness 2005; Konisky et al. 2006; Gedan et al. 2009). With a growing awareness of the ecosystem services provided by tidal marshes there has been a rising tide of scientific interest to better understand vegetation response to tidal inundation (Craft et al. 2009; Roman and Burdick 2012). Restoring a natural hydrology on a site, often by culvert expansion or breaching of sea walls, often occurs as legislated ecological compensation (Zedler 1996; Neckles et al. 2002; Konisky and Burdick 2004; Bowron et al. 2009; van Proosdij et al. 2010), or as a coastal management decision to protect estuaries from floods, storm surges or to address concerns regarding sea level rise (French 2006, Garbutt and Wolters 2008; Craft et al. 2009; Mossman et al. 2012). This thesis examined how vegetation responds to the return of tidal flooding on a low salinity portion of a macro-tidal river.

This project stemmed in late 2009 from the desire to better understand factors driving vegetation recovery on passively restored salt marshes within the Bay of Fundy region of Atlantic Canada. In a collaborative research effort, CB Wetlands and Environmental Specialists Inc. (CBWES) and the Intertidal Coastal Sediment Transport Research Unit (In_CoaST) of Saint Mary's University shared the purchase of a helium filled balloon and suspended camera system to offer a new research approach to examine vegetation recovery on passively restored salt marshes within the region. Vegetation response to hydrologic restoration has traditionally been studied using quadrat surveys at specified sample stations to measure species richness and abundance (Neckles et al. 2002; Konisky and Burdick 2004). Using low altitude aerial photography provides a good opportunity to support and expand the knowledge gained by traditional quadrat based surveys. Suspended camera systems are also a means of limiting the human footprint by offering a more hands-off approach to surveying wetlands or other ecologically sensitive habitats. Orthophotos and geo-referenced image mosaics collected from low altitude aerial photography have been used to: support traditional remote sensing imagery (Artigas and Pechmann 2010); document vegetation and morphological changes occurring on difficult to access environments (Ries and Marzolff 2003; Guichard et al. 2000; Boike and Yoshikawa 2003); develop habitat maps based on automated or manual processes (Mahito and Takeshi 1998; Miyamoto et al. 2004; Lesschen et al. 2008); and to examine morphological changes as well as spatial/temporal changes occurring on a variety of wetlands (Marani et al. 2006; Vericat et al. 2008). Vegetation recovery pattern on restored marshes may be linked to spatial distance from breach or a tidal network (Sanderson et al. 2000; Elsey-Quirk et al. 2009) and it was thought that low altitude aerial

photography would be able to capture these relationships on a macro-tidal wetland restoration project.

1.2 Tidal wetlands

Tidal wetlands are found in the low lying areas of coasts and are exposed to tidal flooding. Two general types of coastal wetlands exist (1) marine wetlands, which include ocean beaches, tidal flats and rocky shores, are relatively more exposed to wave action and ocean currents and other physical stressors that limit colonization of vascular plants, and (2) estuarine wetlands, which include salt and brackish marshes, are meeting zones between salt water from the ocean and fresh water from inland (Tiner 2009). Within an estuary, intertidal habitats are exposed to a gradient of marine influences, such as inundation, salinity and sulfides, and the spatial distribution of vegetation is well known to be organized in characteristic patches that parallel changes in physical stress (Silvestri et al. 2005; Crain et al. 2004; Crain et al. 2008). Odum (1988) showed that within an estuary, as tidal water moves inland from the sea it gets progressively less saline, generating marshes along a salinity continuum going from 'polyhaline' (salinity < 18 ppt); 'mesohaline' (<5 ppt); 'oligohaline' (< 0.5); to 'tidal fresh' where the limit of tidal influence is reached. Salt marshes are found in the mesohaline or polyhaline zones. Brackish marshes are described as either occurring in the mesohaline zone or the more salty regions of the oligohaline zone (Odum 1988, Brewer and Grace 1990). Marshes within the oligohaline zone lie between tidal fresh marshes (where tidal flooding exists but vegetation does not experience salt stress) and mesohaline marshes (where salinity

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plays a regular role in determining community structure) (Odum 1988; Brewer and Grace 1990, Tiner 2009).

On tidal wetlands, physical stress, such as those brought from tidal inundation, is well known to control vegetation distribution and can lead to zoned vegetation communities (Adam 1990; Bertness 1991; Crain et al. 2004; Wolters et al. 2005; Silvestri et al. 2005; Więski et al. 2009; Pétillon et al. 2010). Increase in tidal inundation decreases soil oxygen availability that negatively affects root respiration and growth, seedling development and results in a changed soil chemistry that limits plant growth (Mitsch and Gosselink 2007; Silvestri et al. 2005; Davy et al. 2011). The response of vegetation to sediment accretion by coastal vegetation can either be negative (e.g. dying and not tolerating sedimentation) (Ewing 1996; Maun 1998), or have little effect on plant growth (Maun 1998; Deng et al. 2008). Deposition of sediments carried by tidal inundation, typically consist of fine grained inorganic materials (Amos and Mosher 1985; van Proosdij et al. 2006), that result in a nutrient poor habitat for marsh vegetation (Deng et al. 2008). Nutrient limited habitats tend to favour growth of clonal patches or below ground growth (Maun 1998; Deng et al. 2008). Large deposits from single sedimentation events are also known to negatively or completely effect salt marsh plant growth (Deng et al. 2008). Zonation of vegetation on salt marshes is known to be a product of interspecific competition on high marsh borders and tolerance to physical stress on the low marsh borders (Chapman 1974; Bertness 1991; Crain et al. 2004; Ewanchuk and Bertness 2004). Experimental analysis of individual species response to stress has shown that in the absence of physical stress, such as a reduced salt stress or reduced inundation, salt marsh plants are known to become competitive subordinates in the presence of neighbours and

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are pushed towards the more frequently flooded portions by better competitors (Crain et al. 2004; Bertness 1991).

1.3 Tidal wetland restoration

Altering the tidal regime on a site (eg: the construction of dykes or inadequately sized culverts), supports habitats whose vegetation community are often composed of non-marsh vegetation, and restricted sites often act as potential habitats for invasive species (Minchinton and Bertness 2003; Silliman and Bertness 2005; Wolters et al. 2005). Hydrologic restoration, where the goal is to return natural flooding to a site (e.g.: by the excavation of dykes, expansion or removal of culverts, or ditch plugging formerly dredged creeks), aims to promote the growth of wetland vegetation to a formerly more productive state (Burdick et al. 1997; Mauchamp et al. 2002; Wolters et al. 2005; Konisky et al. 2006; Bowron et al. 2009; van Proosdij et al. 2010). As the knowledge of the ecological and economic value of tidal marshes continues to grow (Connor et al. 2001; Gedan et al. 2009) as well as a refinement of habitat protection from government organizations (Lynch –Stewart et al. 1996; Burdick et al. 1997; Neckles et al. 2002; Konisky et al. 2006; Edwards 2010), restoring tidal marshes has gained scientific interest (Warren et al. 2002; Craft et al. 2002; Wolters et al. 2005; Bowron et al. 2009; van Proosdij et al. 2010). Restoring natural hydrology to a site, frequently by culvert expansion or breaching of sea walls, often occurs as legislated ecological compensation for damage or loss to important wetlands and marshes (Zedler 1996; Nekles et al. 2002; Konisky and Burdick 2004; Bowron et al. 2009; van Proosdij et al. 2010). As restoration becomes a more viable option, ecological researchers require a better understanding of

factors influencing vegetation recovery to better understand restoration progress in stress limited environments (Byers and Chmura 2007; Bowron et al. 2009).

Restoring tidal wetlands in Canada remains a new approach. In Atlantic Canada, the Cheverie Creek salt marsh restoration project was the first large scale intended restoration project of its kind (Bowron et al. 2009). The restoration of the site, as well as other similar restoration projects that have followed, are typically initiated as legislated ecological compensation for unavoidable habitat alteration, disruption or destruction (HADD) to important fish habitat (Lynch-Stewart et al. 1996; Bowron et al. 2012). In Canada the *Fisheries Act* first implemented in 1985 and updated in 1986 aims to protect fish habitat and also guide ecological managers on courses of actions to reduce negative impacts to fish habitat (Lynch-Stewart et al. 1996; Fisheries Act 2011). Wetlands of all types are also federally regulated and protected under the 1996 *"Federal Policy on Wetland Conservation"* a strategy to achieve a '*no net loss'* of wetland function (Lynch-Stewart et al. 1996). Where there is unavoidable or harmful alteration, disruption or destruction of fish habitat (HADD), ecological managers must work to restore or improve habitats to compensate in a *'like-for-like'* manner (Lynch-Stewart et al. 1996).   In 2011 Nova Scotia implemented the '*Nova Scotia Wetland Conservation Policy'* to protect and provide direction for the management of wetlands within the province (Nova Scotia Environment 2011).

Within the Bay of Fundy region of Atlantic Canada, tidal marshes have been dyked for agricultural use since the 1630's (Ganong 1903; Bowron et al. 2012). In 1943, as a response to the increased costs of dyke maintenance as well as the recognition of the economic value of dykelands, federal regulators in partnership with the provincial

governments of Nova Scotia and New Brunswick set up the *'Maritime Dykeland Rehabilitation Committee'* to oversee maintenance costs (Edwards 2010). In 1948 the newly formed Marine Marshland Rehabilitation Administration (MMRA) gave the responsibility of dykes to federal managers whereas the dykelands themselves became provincially managed (Edwards 2010). It was during this time that mechanization techniques, such as the use of tractors, were first employed to maintain and build new dykes which resulted in many old or original dykes being destroyed or being upgraded to newer and taller structures (Milligan 1987; Edwards 2010). Between the years 1967-1970 the federal governments turned the responsibility of maintaining dykes over to provincial managers. In Nova Scotia there is the '*Agricultural Marshland Conservation Act'* to guide future action on marshes (Edwards 2010). Beginning in 1960 and continuing into the early 1970's, the provincial government of Nova Scotia began the construction of causeways across important tidal waters to promote movement between towns as well as dyke protection by way of reducing tidal oscillations (Edwards 2010; Daborn et al. 2002).

1.4 Study site

The study site for this research forms part of the St. Croix River High Salt Marsh and Tidal Floodplain Wetland Restoration Project and lies at the intersection of the tidal river and Nova Scotia Highway 101 (Bowron et al. 2008). The St. Croix River has a hydrological dam at its upper reaches that regulates fresh water flow downstream (Wells 1999). The restoration project consists of 4 separate areas, of which the St. Croix West (SCW) site is largest and is the main research site for this project (Figure 1.1). SCW has an area of 12.9 ha and lies at a mean elevation of 8.98 m (CGVD 28) (Bowron et al.

2008). The site is a dykeland that lies on what was once tidal wetland. In 1950, the St. Croix Marsh body was incorporated into the MMRA for the region (Parent 2009). Prior to incorporation the dykes along the river may have topped over on larger flood tides. However, soil samples collected on site show no indication of laminated layers associated with flooding (Bowron et al. 2010). The site likely has not flooded with saline water since it has been incorporated. Prior to restoration the site was used as agricultural land and was dominated by a mix of pasture grasses (Bowron et al. 2008). Ecological and hydrological surveys in 2007 determined that the partial, or complete, removal of agricultural dykes would allow periodic flooding of tidal water which had the potential to create a productive brackish tidal wetland habitat (Bowron et al. 2008). The site is being restored as an ecological compensation project from the unavoidable Harmful Alteration Damage or Destruction (HADD) of fish habitat due to the twinning of Hwy 101. This site is an excellent research site to study marsh recovery and temporal patterns because as a HADD project, it will require 6 years of ecological monitoring associated with the restoration projects (Fisheries Act 2011). The site is also an excellent study site for the use in this these because the timing of restoration allowed the examination of vegetation recovery as the site was breached during the fall of 2009 making the 2010 and 2011 years the first two growing seasons following hydrologic restoration.
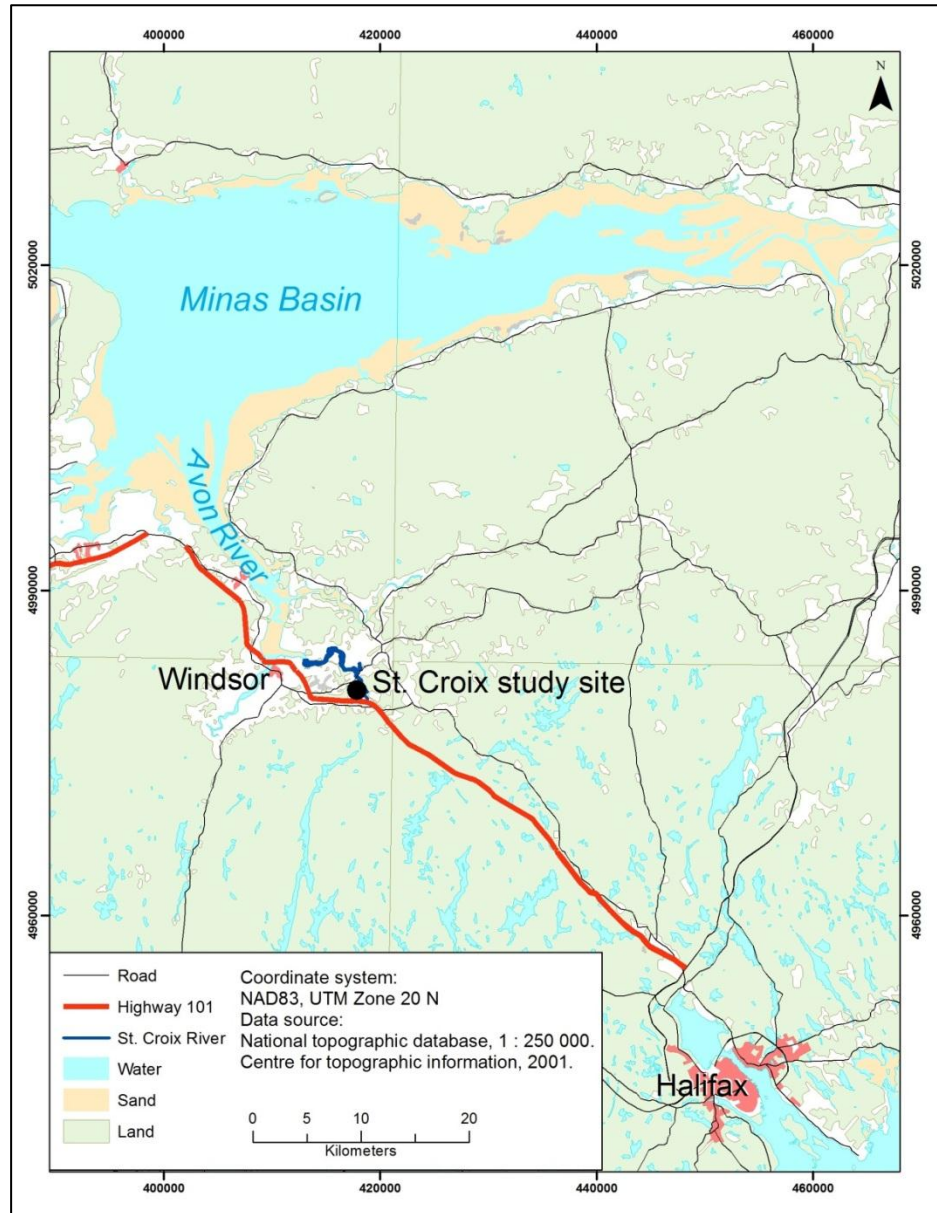
Figure 1.1 St. Croix Restoration site (black circle) located at the upper reaches of the St. Croix River. The river is a tributary of the Avon Estuary.

1.5 Research goals

Natural recovery patterns in low salinity macro-tidal environments remain a challenge to scientists and there are few examples to draw upon, even globally. It is generally not known how vegetation community structure will change and what factors drive these changes within a zone where tidal water has low salinity. This thesis examined vegetation recovery at two spatial scales; the first is a marsh wide scale where the vegetation recovery pattern on a macro-tidal brackish tidal marsh within the first two growing seasons following restoration is examined, and the second is the vegetation community species scale where the relationships between abiotic variables and individual species recovery are examined. The specific objectives addressed in this thesis were:

Chapter 2 Develop a simple classification technique of low altitude aerial photographs to examine marsh wide vegetation response to hydrologic restoration.

Chapter 3 Examine the relationship between tidal inundation, sedimentation and vegetation recovery following hydrologic restoration at the quadrat scale.

Completing this research will aid ecological researchers and restoration practitioners better understand the pattern and pace of ecosystem recovery on a macro-tidal low-saline marsh restoration project. Chapters two and three are written as independent manuscripts intended for publication. In all chapters rod sediment elevation table (RSET) data as well as pre-restoration vegetation data, original construction design and pore water soil salinity were provided by CBWES Inc. Some of the soil processing presented in chapter three was performed by the In_CoaST lab at Saint Mary's

University. Statistical analysis and interpretation of the data in both of the chapters are my own.

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CHAPTER 2: USING CLASSIFIED RGB AERIAL IMAGERY TO EXAMINE THE
PACE OF VEGETATION RECOVERY ON A MACRO-TIDAL MARSH
RESTORATION PROJECT.

Abstract

High resolution large scale digital imagery collected from the air at low altitudes (typically less than a few hundred metres above the earth surface) can provide spatial and temporal information on surface cover changes. In this paper low altitude aerial photographs are classified using the image classification toolset within ArcGIS 10.0 and a maximum likelihood classification scheme to examine surface cover changes within a newly restored marsh lying within a low-salinity zone of a macro-tidal river. The use of a two class maximum classification scheme of spring and fall images from the same growing season, as well as subsequent fall images, captured remarkable changes in surface cover. In the first year, the site was marked by a period of decreased vegetation cover showing that there was a disturbance that removed surface cover biomass. The high rate of sediment elevation increase recorded by the rod sediment elevation table (RSET), ranging between 23.04 ± 0.39 cm yr^{-1} and 13.09 ± 0.35 cm yr^{-1}, killed off non-tolerant vegetation. During the second growing season there was an almost complete recovery of vegetation surface cover. The high rate of sediment accretion recorded during both years did cover the vegetation and resulted in a vegetation class having a similar spectral signal as bare ground. The limits of an automatic classification technique lie with the spectral limitations provided by red green blue (RGB) imagery. Over the first two growing seasons following restoration sedimentation likely drove surface cover changes. Sedimentation killed off vegetation in low lying areas during the first year of restoration and provided a platform for new colonization to occur in the second year. This paper shows that vegetation recovery has taken two growing seasons at the site.

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2.2 Introduction

In almost all wetlands, vegetation is typically distributed according to a gradient reflecting individual species tolerance to flooding or saturation (Odland and del Moral 2002; Mitsch and Gosselink 2007). Zonation of tidal marsh vegetation, where species diversity is low in the frequently flooded low marsh portion and generally increases in the irregularly flooded portions of marsh, is known to be a product of interspecific competition on high marsh borders, and tolerance to physical stress on the low marsh borders (Chapman 1974; Bertness 1991; Crain et al. 2004; Ewanchuk and Bertness 2004). In the absence of competition, salt marsh species are known to grow better in less saline environments than in saline environments but become subordinate species in the presence of competition (Bertness 1991; Crain et al. 2004; Ewanchuk and Bertness 2004). Where physical stress is reduced, disturbance-generated patches of bare ground are known to be important micro-habitats for competitively subordinate species (Baldwin and Mendelssohn 1998; Ewanchuk and Bertness 2004; Crain et al. 2008). Within salt marsh succession theory bare patches of ground act as platforms for primary colonization to occur in a process known as facilitation, primary colonists change the soil substrate and/or the elevation by trapping sediments allowing secondary successive species to colonize the area (Odland and del Moral 2002; Petillon et al. 2010; Silvestri et al. 2005). It has also been shown that returning tidal flow to historic land claims, by breaching of dykes or culvert expansion to allow a more natural hydrologic flow, has the potential to create high sedimentation events (Byers and Chmura 2007; Bowron et al. 2009). Sedimentation is a product of tidal range and sediment supply (Chmura et al. 2001).The return of tidal flooding from a river well known to have a high suspended sediment

concentration (van Proosdij et al. 2006) with a well-known pattern of lower mean elevation of agricultural fallow lands (Byers and Chmura 2007) explains where the high sedimentation potential exists for newly restored marshes within dykelands. The role that high sedimentation has on vegetation recovery when transforming an agricultural dyke land to former more productive state remains a point of interest.

The use of low altitude aerial photography provides a good opportunity to support and expand the knowledge gained by traditional quadrat based surveys (eg: Neckles et al. 2002; Konisky et al. 2006). High-resolution large scale digital imagery collected from the air at low altitudes (typically less than a few hundred metres above the earth surface) can provide spatial and temporal information on surface cover changes (Shuman and Ambrose 2009; Aber et al. 2010). Orthophotos and geo-referenced image mosaics collected from low altitude aerial photography have been used to: (a) support traditional remote sensing imagery (Artigas and Pechmann 2010); (b) document vegetation and morphological changes occurring on difficult to access environments (Ries and Marzolff 2003; Guichard et al. 2000; Boike and Yoshikawa 2003); (c) develop habitat maps based on automated or manual processes (Mahito and Takeshi 1998; Miyamoto et al. 2004; Lesschen et al. 2008); and (d) examine morphological changes as well as spatial/temporal changes occurring on a variety of wetlands (Marani et al. 2006; Vericat et al. 2008). Classifying images using an automated approach, where the spectral signatures of objects are used to categorize objects into surface cover classes, is a powerful tool to examine surface cover changes (Lillesand and Keifer 2000; Aber et al. 2010; Marani et al. 2006). Although the literature is rich with automatic image classification techniques of remotely sensed data with infrared capabilities (eg: Lillesand and Keifer 2000; Arnold 1997) very

few examples exist of classification techniques employed on red-green-blue colour model (RGB) imagery at the cm pixel scale. However, improvement of off-the-shelf camera equipment offers new opportunities to advance classification techniques.

This paper aims to develop a simple classification approach using RGB imagery combined with overlay analysis within ArcGIS 10.0 to examine the pattern of surface cover changes soon after restoring a natural hydrology to an agricultural dykeland within a low salinity portion of a macro-tidal river. The objectives of this research are to: (1) Determine if the Image Classification toolbar in ARCGIS 10.0 can be used to classify high resolution low altitude aerial images; (2) Determine the rate of vegetation recovery of an agricultural dykeland in the first two growing seasons following natural hydrologic restoration; (3) Examine the relationship between elevation change and aerial image features during the first growing season following hydrologic restoration.

2.3 Study area

The Bay of Fundy is a large hyper-tidal embayment located mostly within Canadian Atlantic provinces at the northeastern end of the Gulf of Maine. Semi-diurnal tides in the upper portions of the Bay of Fundy reach an excess of 16 m on larger spring tides (Desplanque and Mossman, 2004; van Proosdij et al. 2006). A large tidal prism combined with high suspended sediment concentration contributes to high sedimentation rates recorded within the region (Daborn et al. 2002; Chmura et al. 2001; van Proosdij et al. 2006). The St. Croix River High Salt Marsh and Tidal Floodplain Wetland Restoration Project is located within the upper reaches of the Avon Estuary in Nova

Scotia, Canada, Figure 1.1 (Bowron et al. 2008). This site was formerly a tidal wetland yet has been hydrologically restricted for at least 50 years. The restoration project consists of four separate areas, of which the main research site for this project was the 12.9 hectare St. Croix West (SCW) site which lies at a mean elevation of 8.98 m (CGVD-28). Elevations in this report are relative to the Canadian Geographic Vertical Datum of 1928 (CGVD28), which is referenced from the Mean Water Level (MWL) measurements made in 1928 at tide gauges across Canada. It is the current standard for vertical datum available within Canada. In 2007, ecological and hydrological surveys determined that partial, or complete, removal of the agricultural dyke would allow periodic flooding of tidal water and has the potential to create a productive brackish tidal wetland habitat (Bowron et al. 2008). In July of 2009, restoration construction started with the excavation of two ponds, positioned adjacent to areas of low surface elevation and close to remnant agricultural ditches. The dyke was breached at 5 locations between August 10 and 19, 2009 (Figure 2.1). For a full description of the construction designs refer to Bowron et al. (2011). The ecological monitoring program developed for the project was based on the 'Global Programme of Action Coalition for the Gulf of Maine' (GPAC) salt marsh restoration protocol (Neckles et al. 2002; Konisky et al. 2006). Modifications to this protocol were made based on experience gained from previous restoration successes within the area (Bowron et al. 2009; van Proosdij et al. 2010). To measure sediment elevation, a rod surface-elevation table (RSET) device was installed on site prior to restoration following a design by Cahoon et al. (2002).

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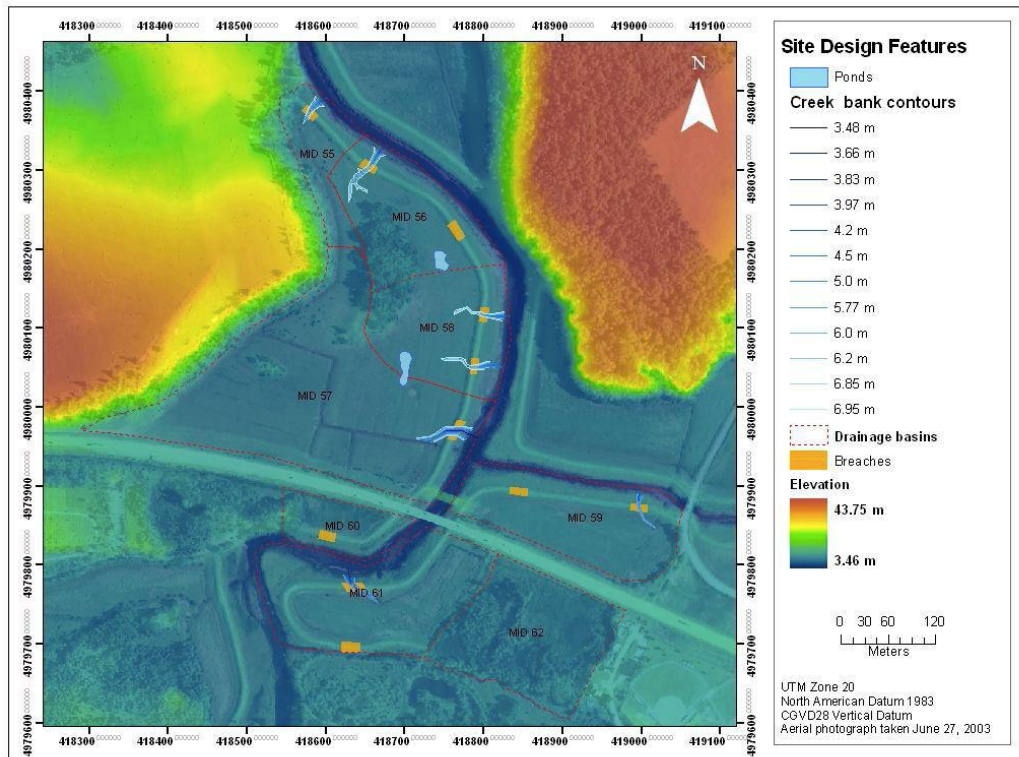
Figure 2.1 St. Croix River High Salt Marsh and Tidal Floodplain Wetland Restoration design. The locations of the breaches and the constructed ponds at the site are also shown (Sourced from Bowron et al. 2010, with

2.4 Methods

Plover I (Figure 2.2, Insert A) is a remotely operated, tethered balloon (6.3 m³ helium filled) and suspended camera system. The camera (Canon Eos Rebel XSi; www.canon.ca) and lens (Canon EFS 10-22 mm ultra wide zoom; www.canon.ca) combination has the potential to give image ground footprints of 240 m by 140 m, with 10 cm resolution when deployed at 100 m in altitude. To orthorectify the images and construct image mosaics, a total of 14 ground control points (GCP) were established on the marsh and geo-referenced using real time kinetic (RTK) from a DGPS (Coordinate system: UTM NAD83, Zone 20 N). The target signals (Figure 2.2, Insert B) were placed over the GCP and measured 40 cm X 40 cm with a 10 X 10 cm black centre. The goal was to capture temporal changes occurring over the course of the first two growing seasons following hydrologic restoration by capturing a spring and fall image in each of the growing seasons. Due to heavy rain events and windy weather combined with un-favourable tides during the spring of 2011, the capturing of the spring images was not feasible. Images reported were captured on May 2010, September 2010 and September 2011. Image distortions, known to be large from off-the-shelf camera equipment (Jungo and Jensen 2005; Laliberte et al. 2008), were corrected using DXO Optics Pro. Orthorectifaction and mosaic construction was performed using PCI Geomatica. The spring 2010 image has an accuracy on the x- axis of 0.40 root mean squared (RMS) and y-axis RMS 0.42; fall 2010 RMS = x-axis 0.26, y-axis 0.31; fall 2011 RMS = x-axis 0.07, y-axis 0.08.
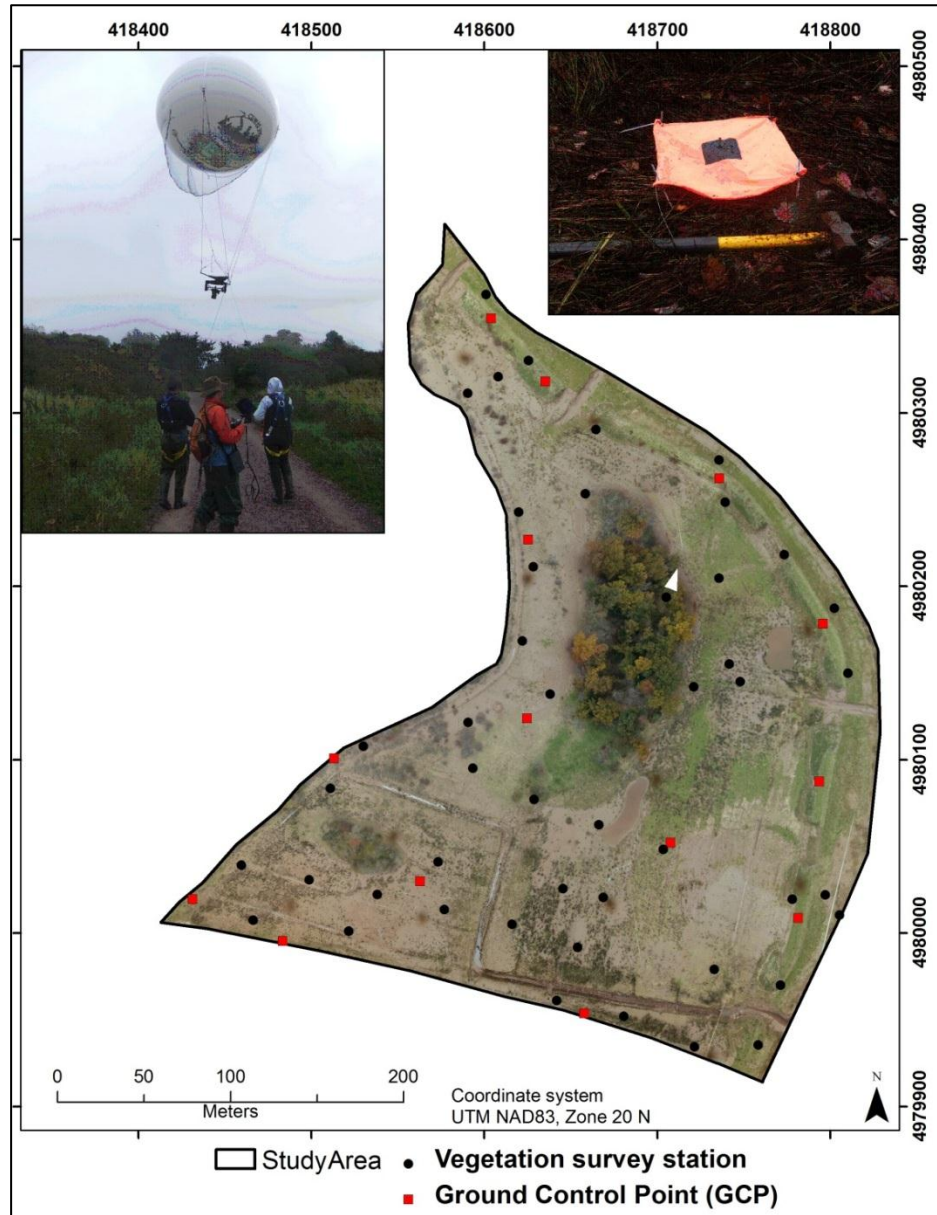
Figure 2.2 The St. Croix ecological study design showing sample stations, target signals. Insert A shows Plover I- a helium filled remotely operated balloon system. Insert B shows the target signals used in orthorectification of images

This research aimed to incorporate surface cover analysis from low altitude aerial photography into the salt marsh restoration monitoring program. To accomplish this the pixel based image classification toolbar within ArcGIS 10.0 was used. The orthorectifed image mosaics from 2010 and 2011 were classified into either vegetation or bare ground cover classes. To compare results from surface cover analysis and the actual ground conditions, a total of 47 vegetation survey stations were established (Figure 2.2). The sample stations were established on the site using a stratified random sampling technique and geo-referenced using RTK unit from a DGPS (CGVD28). At each station a point intercept quadrat method was used to determine species composition and abundance, where abundance was the frequency of contact of each species with 25 points per m^2 (Roman et al. 2001; Bowron et al. 2009).

To classify the images roughly half of the vegetation survey stations within the site and on the fringe marsh were used as training sites for the image classification, and the other half were used for accuracy assessment of the classification. The class probability tool was used to evaluate the probability of classification for each pixel, and, as needed more new cells were added to each class from examination of the images using areas known to be bare ground or areas with dense vegetation. The image mosaics were classified using a maximum likelihood classification scheme. The classified images were then smoothed as needed by using majority filtration with eight neighbours.

To determine the accuracy of the automated classification process using the suite of tools in ArcGIS 10.0, a comparison was made between the classed images from both years against the vegetation survey results from the stations not used as training sites. To make the quadrat data comparable to the classification data, results from the quadrat

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survey were classed as either dominantly bare ground if: bare ground covered more than 50% of the survey quadrat; or dominantly vegetation if: vegetation covered more than 50% of quadrat. To extract the surface cover values from the automatically classified images (Fall 2010 & Fall 2011 e.g. those images which were closest to the on the ground vegetation survey) the extract values to points tool was used at each sample station. The extracted values were then compared against the vegetation surveys to calculate percent error of the classed image. To examine changes in surface cover over time the automatically classified raster images were overlain for each instant (e.g.: Spring 2010 vs. Fall 2010; Fall 2010 vs. Fall 2011). A surface cover change matrix (Table 2.1)was created to show how temporal changes in surface cover classes was assessed.

Table 2.1 Surface cover change matrix used in overlay analysis of classified aerial photographs

	Instance 2 Bare ground	**Instance 2 Vegetation**
Instance 1 bare ground	Always bare ground	New growth
Instance 1 vegetation	Die-off	Always vegetation

2.5 Results

A comparison was made between the 2010 classified image against the 2010 quadrat data to test the accuracy of the automatic classification, as can be seen in Figure 2.3, which shows the classified image as a transparency overlay of the raw mosaic and survey stations. The classified image has an overall accuracy of 52% correct and 48% was incorrectly classified. All of the classification error is bare ground over estimation.

Figure 2.4 shows the 2011 classified image as a transparency overlay of the 2011 raw mosaic and survey stations. In the 2011 classified image, 80 % of the classification is correct and 20% is incorrectly classed, where 17% vegetation is incorrectly classed as bare ground and 3% bare ground is incorrectly classed as vegetation. In both years the bare ground category is overestimated by the classification process. The bare ground overestimation is linked to vegetation that is sediment laden, and results in a vegetation class having a spectral signature similar to the bare ground class.

The spring vs. fall 2010 comparison of surface cover change, Figure 2.5, shows that by the end of the first growing season following hydrologic restoration large changes occurred on the surface of the site. Most striking was the formation of large areas of bare ground over 58% of the total surface. Area that remained bare ground throughout the first growing season (always bare ground) covered 36% of the total area and die-off of vegetation covered 22% of the site. Conversely, the 44% of vegetation was split evenly between always having had vegetation present (always vegetation) 22% and new growth or colonization with 22%. During this first growing season the bare ground and vegetation die-off classes cover a larger portion of the site and indicate that there is a general loss of biomass. Comparing the first and second growing seasons, Figure 2.6 shows that there is a significant increase in biomass, as 94 % of the site is covered by vegetation classes, with 54% being new growth and 40% having always been vegetation. There has been no change in bare ground cover over 4% of the site and 2% was attributed to species die-off.
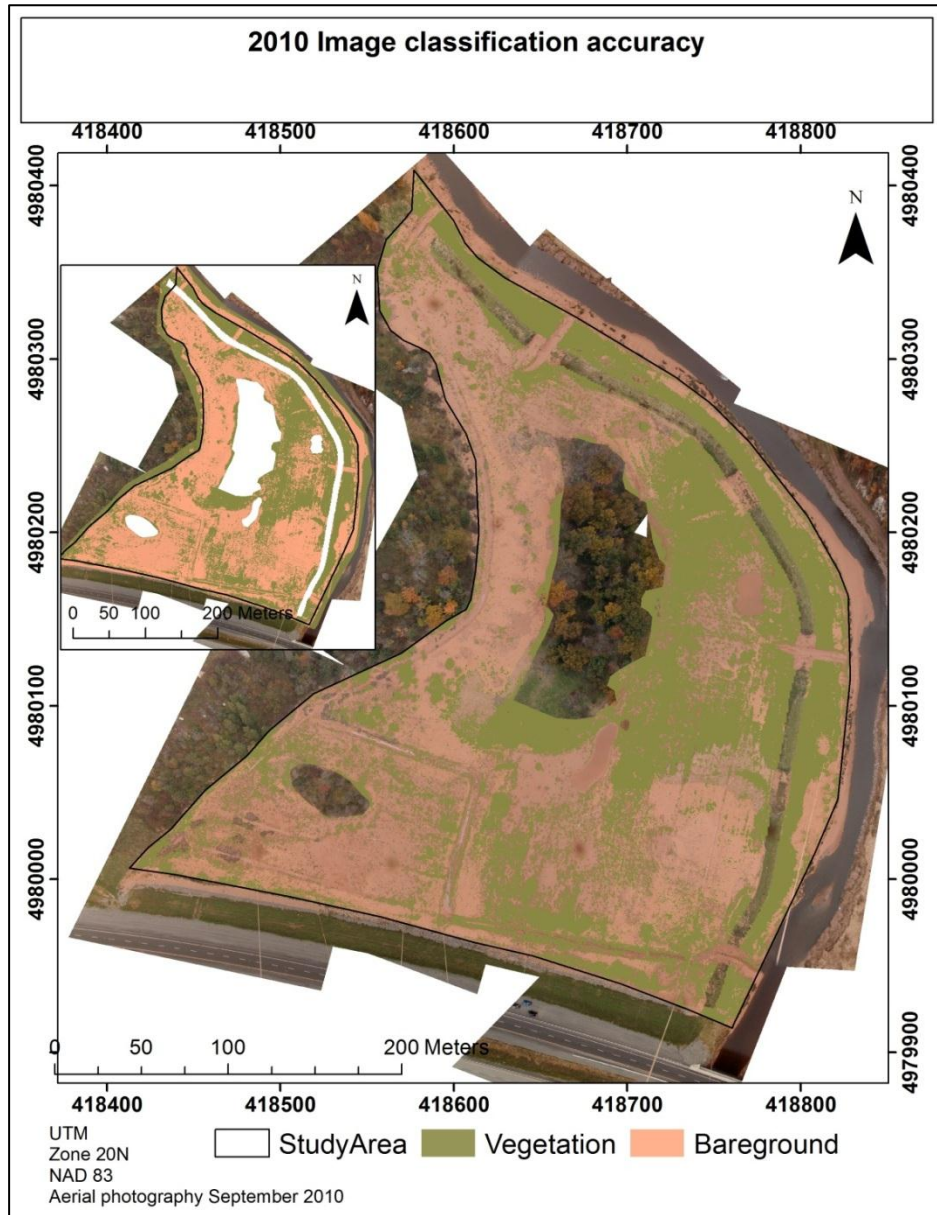
Figure 2.3 Automatic classification results for the 2010 fall image. The classified raster data is shown as a transparency overlay. The insert shows the raw image without overlay.

Figure 2.4 Automatic classification results for the 2011 fall image. The classified raster data is shown as a transparency overlay. The insert shows the raw image without overlay.
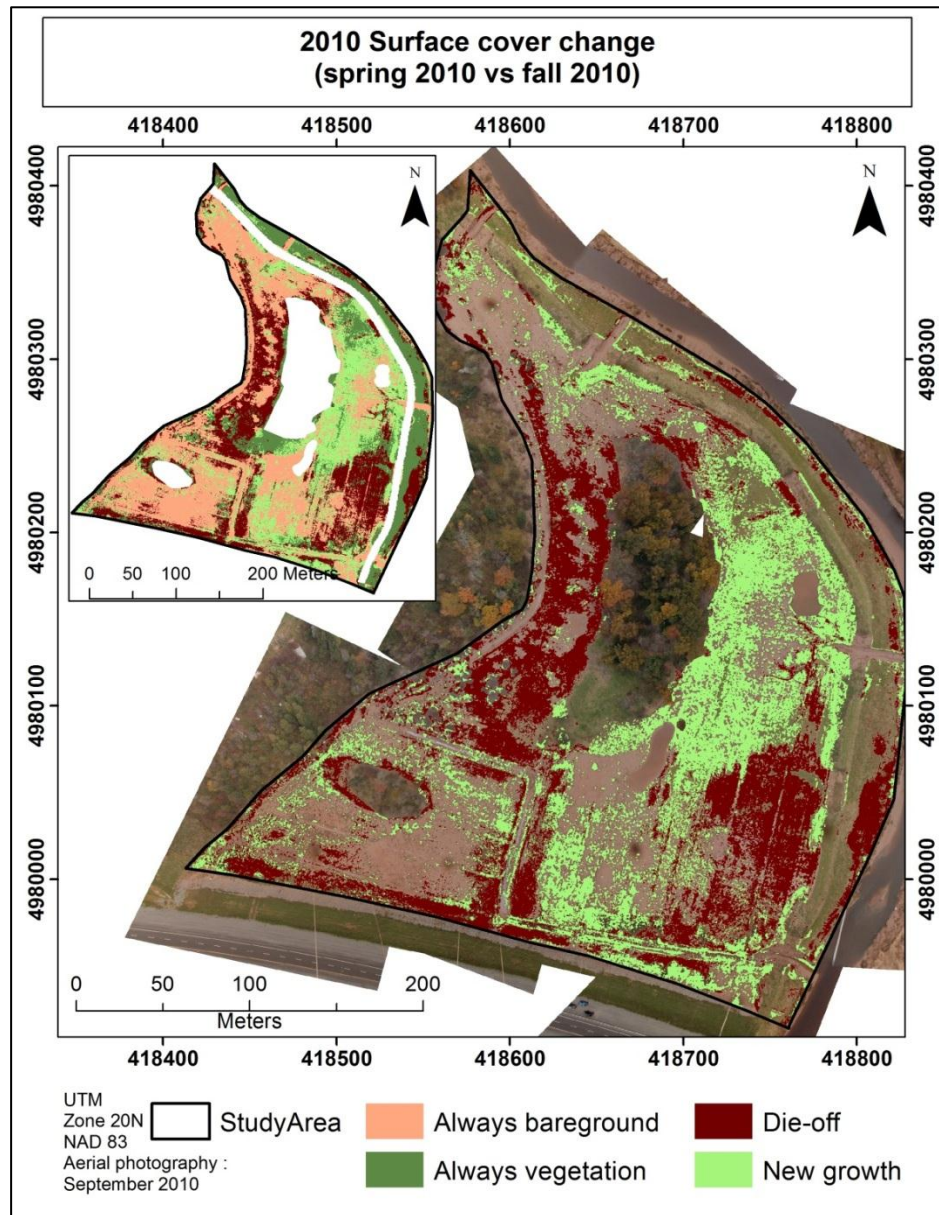
Figure 2.5 Surface cover analysis for the 2010 season. The map shows die-off and new growth overlayed over the raw image mosaic. The insert shows all the surface cover classes. The 2010 season was marked by die-off of vegetation and bare ground as well as some colonization of vegetation.
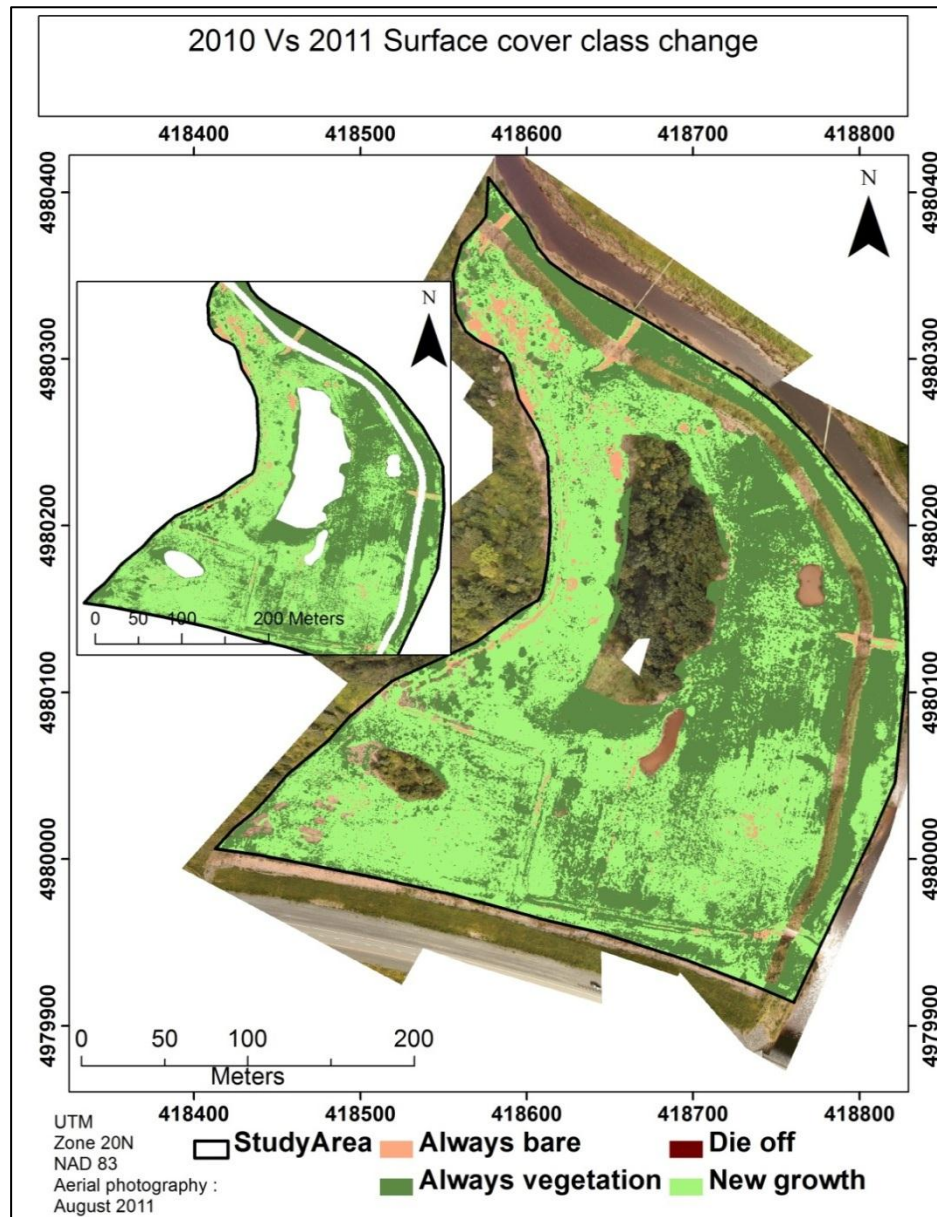
Figure 2.6 Surface cover analysis for the 2010 vs 2011 2011 season. The map shows die-off and new growth overlayed over the raw image mosaic. The insert shows all the surface cover classes. The figure shows that by the end of the second growing season most of the marsh is covered with vegetation.

To examine the relationship between elevation and surface cover changes that occurred over the first year following restoration, the cover classes (always bare, always vegetation, new growth and die-off) from the 2010 surface cover change file was used to extract Lidar DEM (2007) elevation values. The surface cover change raster was converted to centroid points file so that Lidar DEM elevation values could be extracted using the extract values to points tool and analyzed for statistical differences. Figure 2.7 is a graphic representation that confirms ANOVA analysis which shows that there is a difference in elevation among the surface cover classes (ANOVA f= 2.61, df = 3, p = < 0.001). Statistical analysis shows that the elevation of bare ground, 6.57 ± 0.37 m, is found at lower elevation where vegetation is present, 7.06 ± 0.39 m (t = 1.96, df = 36 740, p = < 0.001). The elevation of vegetation die-off, 6.74 ± 0.44 m, is occurring at a slightly higher elevation than the mean elevation of growth (6.67 ± 0.26). Low-lying areas on the marsh are known to flood more frequently and for longer periods of time than relatively higher elevations (Pétillon et al. 2010).
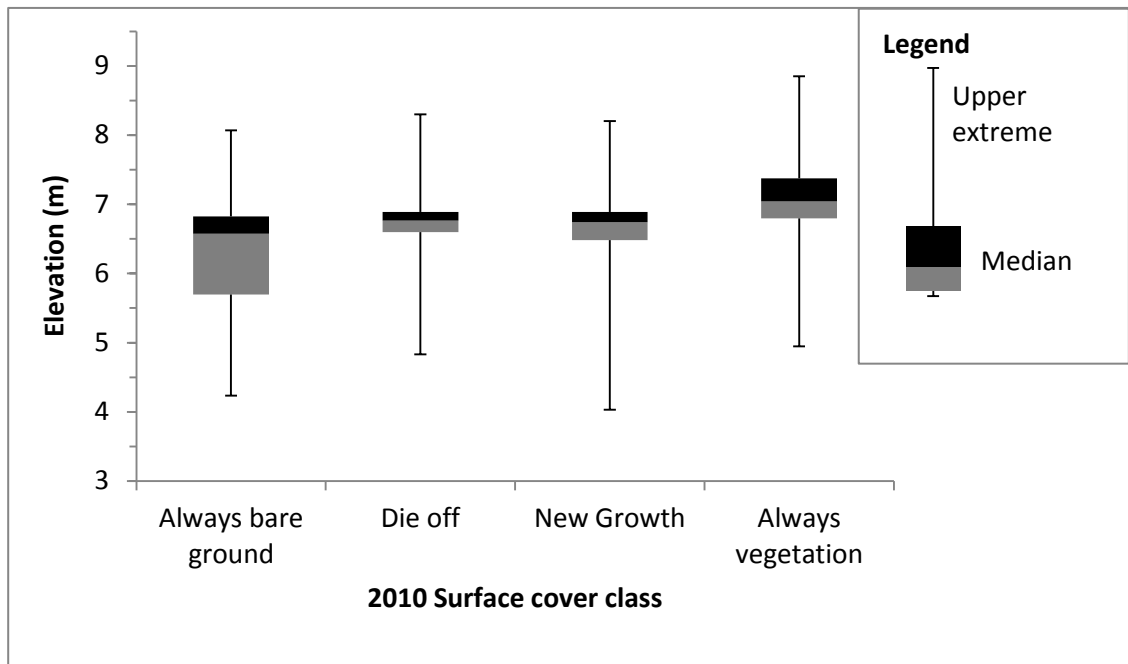
Figure 2.7 Box plot showing the elevation (m) distribution 2010 surface cover change image. For each surface cover class the median elevation is the line in between the grey and black box. The black box represents the range of the third quartile, the grey box is the range of the first quartile. The lines for each surface cover class (whiskers) represent the upper and lower extremes.

In the first year following restoration the SCW RSET elevation increases ranged between 23.04 ± 0.39 cm/yr^{-1} and 13.09 ± 0.35 cm/yr^{-1} (Bowron et al. 2011). The high sedimentation within the marsh during the first growing season following restoration resulted in the formation of large mud-flats across a significant portion of the site. Visual inspection (Figure 2.8) of the areas with the most gain in sediment elevation recorded (RSET 1, RSET 2 and RSET 4) showed that they are located near or on the bare ground or the die off surface cover classes. RSET 3, the station with the lowest recorded change, is located within an area that is experiencing vegetation recovery. In the second year the

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RSET values ranged between 7.2 cm/yr^{-1} and 0.35 cm/yr^{-1}, pointing to a reduced rate of sedimentation influx and tidal inundation onto the site which could represent a less stressful environment for establishing vegetation.
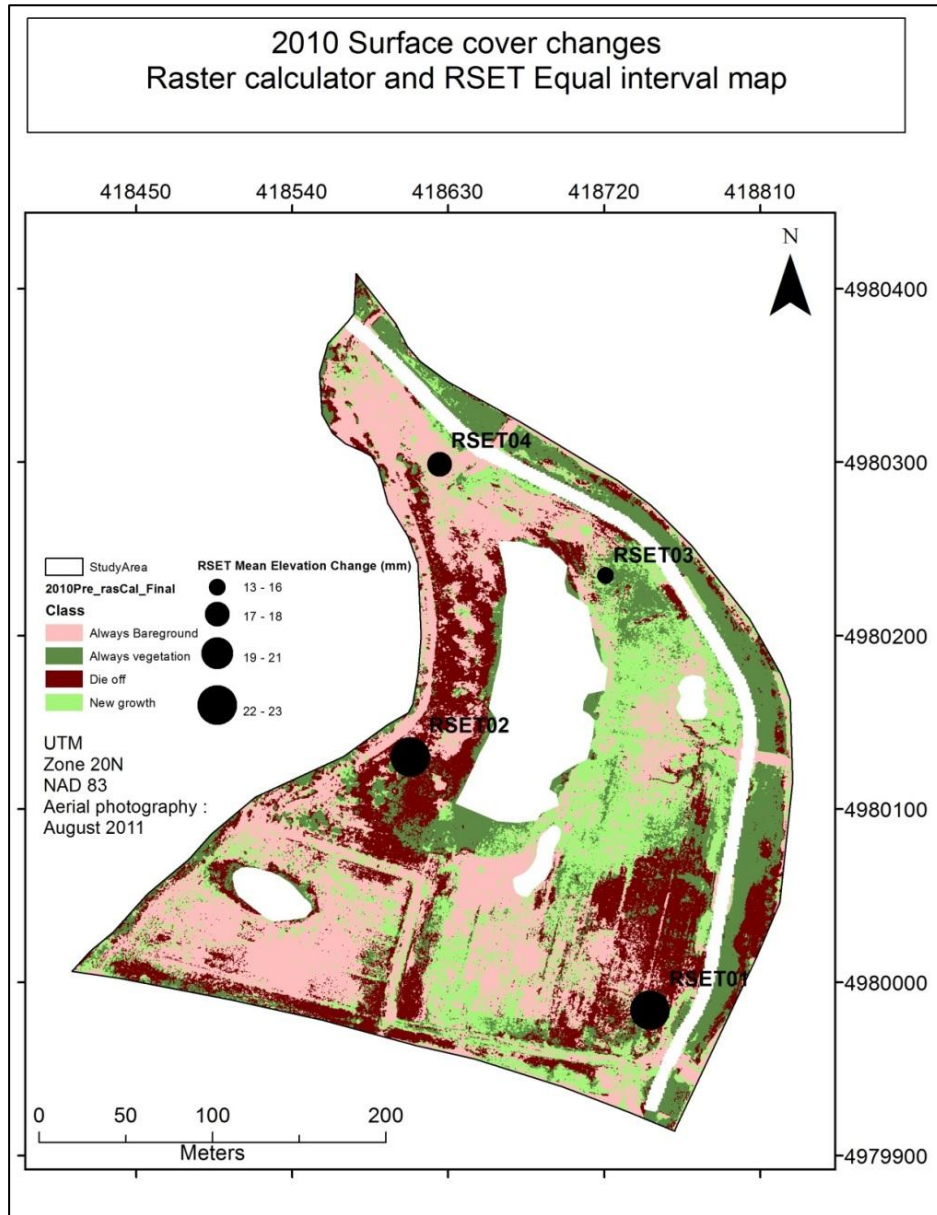
Figure 2.8 Summary of changes at the St. Croix site during the two year study period. Sediment accretion recorded at the RSET stations is shown as quartile proportional symbols. Changes in surface cover are also shown.

2.6 Discussion

Over the course of the two first growing seasons following restoration, the SCW site was marked by a period of decreased vegetation cover during the first growing season, and the second growing season experienced an almost complete vegetation recovery. Disturbance, defined as the removal of biomass by Grime (1979), occurred in the first year that was followed by a re-colonization period in the second year.

As part of the part of the monitoring program established on the site (Bowron et al. 2008) established on the restoration site (Bowron et al. 2008), pore water salinity was recorded at several stations throughout the growing season at several survey stations. Mean salinity ranged from a high of 2.6 ± 1.2 ppt and a low of 0.00 ppt (Bowron et al. 2011). The salinity readings at the site are low when compared to salinity readings in other restoration projects within the area, which record rates ranging from 18 ppt to 35 ppt (Bowron et al. 2009; van Proosdij et al. 2010). Oligohaline marshes (salinity ranging from 0.5 ppt to 5 ptt) act as intermediate marshes where vegetation experiences intermittent and reduced salt stress (Odum 1988; Brewer and Grace 1990). At the St. Croix site where there is relatively low soil salinity, it is likely that vegetation experience little salt stress. It is well-known that halophytic vegetation, such as *Spartina alterniflora, Agrostis stolonifera* and *Juncus gerardii*, tend to have higher seed germination rates and record higher growth rates in low saline environments (Rozema and Blum 1977; Shumway and Bertness 1991; Rand 2000). Given the high rate of vegetation recovery experienced at the site during the second year and the low salinity readings, seeds

collected on the mudflats or otherwise deposited at the site during that first year likely remained viable and experienced little stress that could have negatively impacted seed germination rates. The high rate of sediment elevation increase recorded by the RSET, ranging between 23.04 ± 0.39 cm/yr^{-1} and 13.09 ± 0.35 cm/yr^{-1}, killed off non-tolerant vegetation. In the second year the lower sedimentation rate, as measured by the RSET recordings ranging between 7.2 cm/yr^{-1} and 0.35 cm/yr^{-1}, and low soil salinity promoted favourable conditions for vegetation recovery.

In restoring a tidal wetland in the Bay of Fundy, where there is a high suspended sediment concentration in tidal waters (van Proosdij et al. 2006; O'Laughlin and van Proosdij 2012), it is the hydrology and high sedimentation potential that initiates changes in physical conditions by creating bare spaces that provide the platform for species colonization to occur. The low pore-water salinity found on the site is part of the success story for vegetation recovery. Zonation of the marsh at this site, where zonation is known to be a product of competition for resources away from salinity stress and tolerance to salt stress (Bertness 1991; Emery et al. 2001; Crain et al. 2004), is expected to be more competition driven and we should expect high biodiversity at this site.

2.7 Conclusion

In this research the image classification toolset within ArcGIS 10.0 was used to develop an automated classification technique on high resolution RGB imagery. The ability to capture in-season changes, as occurred between spring and fall of the first growing season, proves to be a useful low-cost approach to examine processes leading to

quick changes in surface cover within a highly dynamic state. Using a two class approach to image classification with images from spring and fall of the same growing season captured remarkable changes within the first year of growth. The high rate of sediment accretion covered the vegetation and did result in the vegetation class having a similar spectral signal as bare ground. In this research the accuracy of the image classification toolset within ArcGIS 10.0 was most limited in the first year when sediment rates were highest. In this regard the limits are with the spectral limitations of RGB imagery. To use the images to study vegetation community structure and spatial organization the use of infrared or near infrared imagery would become necessary (Aber et al. 2010). However, the use of low altitude, high resolution aerial orthorectifed aerial photographs has other values as well, such as the creation of habitat maps or the analysis of tidal channel development following hydrologic restoration (Bowron et al. 2011). The ease of operation makes the technology and techniques described in this paper an easy to use and a valuable addition to ecological assessments that are part of tidal wetland monitoring protocol already used in the region (Neckles et al. 2002; Bowron et al. 2009; van Proosdij et al. 2010).

The spatial patterns of sedimentation drove the system over the course of the two year period with the largest changes occurring during the first year. Without the high sedimentation rates, it is likely that little die-off of vegetation would have occurred, as the low pore-water salinity on the site would not have acted as a disturbance agent alone. At the St. Croix site succession is driven by sediment deposition, and, in the coming seasons

may continue to a pulsed ecosystem lying at the fringe of an ecotone between a salinity influenced ecosystem and a tidal fresh water wetland.

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CHAPTER 3: SEDIMENT DRIVEN RESPONSE: THE ROLE OF SEDIMENTATION ON VEGETATION RECOVERY FOLLOWING ON A HYDROLOGICALLY RESTORED MACRO-TIDAL AGRICULTURAL DYKE LAND

Abstract

Within an estuary, intertidal habitats are exposed to a gradient of marine influences, such as inundation, salinity and the spatial distribution of vegetation is known to be organized in characteristic patches that respond to changes in physical stress. Where physical stress is reduced, disturbance generated patches of bare ground are known to be important for competitively disadvantaged species in fresh and salt water marshes. In this paper we examine the relationship between sedimentation and species recovery following hydrological restoration. The hypothesis studied is that sedimentation and tidal flooding will drive vegetation changes. We predict that sediment created patches of bare ground would become colonized by brackish and salt marsh species. Over the course of the two year study period, the elevation of stations within the restoration site increased by 15 ± 13 cm and fringe stations increased by 9 ± 8 cm. High rates of sediment elevation increases were recorded at rod sediment elevation table (RSET) stations, ranging between 23.04 ± 0.39 cm/yr^{-1} and 13.09 ± 0.35 cm/yr^{-1}. The high rate of sedimentation created large mudflats over the course of the first year which became colonized by vegetation in the second year. Of the bare ground survey plots from the first year, 33 % became dominant salt marsh plots, 58 % became brackish dominant plots by 2011, and 1 % remained bare ground. At the St. Croix site, sedimentation was the agent driving ecosystem change by first reducing the surface biomass of non-tolerant vegetation and then aiding colonization by providing habitat for salt-tolerant vegetation within a low saline environment. This research will help guide future restoration projects within a low saline portion of a macro-tidal estuary by providing insight on driving factors of vegetation recovery.

3.0 Introduction

Within an estuary, intertidal habitats are exposed to a gradient of marine influences, such as inundation, salinity and sulfides, and the spatial distribution of vegetation is well known to be organized in characteristic patches responding to patterns of physical stress (Odum 1988; Crain et al. 2004; Silvestri et al. 2005; Crain et al. 2008). Zonation of tidal vegetation, where species diversity is low in the frequently flooded portion of the marsh and increases in the irregularly flooded portions, is known to be a product of interspecific competition on high marsh borders and tolerance to physical stress on the low marsh borders (Chapman 1974; Bertness 1991; Crain et al. 2004; Ewanchuk and Bertness 2004). In the absence of competition, salt marsh plants are known to exhibit better growth in less saline environments (reduced relative physical stress) than more saline environments (higher relative physical stress), but are competitively disadvantaged in the presence of neighbours (Bertness 1991; Crain et al. 2004; Engels et al. 2011).

Within an estuary, Odum (1988) proposed that as tidal water moves inland from the sea it gets progressively less saline generating marshes along a salinity continuum that goes from 'polyhaline' (salinity < 18 ppt); 'mesohaline' (<5 ppt); 'oligohaline' (< 0.5); to 'tidal fresh' where the limit of tidal influence is reached. Oligohaline marshes are intermediate marshes, lying between tidal fresh marshes (where tidal flooding exists but vegetation does not experience salt stress) and mesohaline marshes (where salinity plays a regular role in determining community structure) (Odum 1988; Brewer and Grace 1990). In tidal fresh and brackish wetlands, competitively disadvantaged species, such as salt

marsh species like *Spartina alterniflora* and *Salicornia sp.,* have been shown to successfully colonize disturbance-generated patches of bare ground (Baldwin and Mendelssohn 1998; Ewanchuk and Bertness 2004; Crain et al. 2008). Patches of bare ground lying high up in the tidal frame have been shown to act as a micro-habitat with a different soil chemistry than surrounding patches which act as refuge for competitively disadvantaged species (Ewanchuck and Bertness 2004; Smith et al. 2009; Davy et al. 2011). Facilitation is a term describing marsh succession, where there is a gradual replacement of a simple vegetation community by a more complex vegetation community (Odland and del Moral 2002). Primary colonizing species typically facilitate the establishment of secondary species (either by altering sediment substrates or capturing sediments thereby resulting in elevation increases) (Odland and del Moral 2002; Silvestri et al. 2005; Petillon et al. 2010). In this view, primary species are creating the conditions for future growth of secondary species by changing trapping sediments and increasing the elevation (Silvestri et al. 2005).

Where there is a tidal restriction (e.g. the construction of dykes or inadequately sized culverts) the vegetation community is often composed of pasture species or non-native species and the sites are often habitats for invasive species, such as *Phragmites australis* (Minchinton and Bertness 2003; Silliman and Bertness 2005; Wolters et al. 2008).When restoration goals are clearly outlined, restoring a natural hydrology to a site, frequently by culvert expansion or breaching of sea walls, has yielded positive results (Zedler 1996; Nekles et al. 2002; Konisky and Burdick 2004; Bowron et al. 2009; van Proosdij et al. 2010). Natural vegetation recovery on restored salt marshes has been shown to be most successful when (1) there is no restriction on the dispersal ability of

tolerant species within and nearby the restoration site (Wolters et al. 2005; Dausse et al. 2007; Morzaria-Luna and Zedler 2007; Wolters et al. 2008; Elsey-Quirk et al. 2009), (2) there is a disturbance, such as flooding or sediment deposition, to either remove non-tolerant species on the restoration sites or to allow competitively disadvantaged species to colonize the area (Crain et al. 2008; Smith et al. 2009; Davy et al. 2011), and (3) edaphic and hydrologic conditions are within tolerance ranges for tolerant species (Craft et al. 2002; Crooks et al. 2002; Konisky and Burdick 2004; Pétillon et al. 2010; Cui et al. 2011). One of the challenges remaining to restoration scientists is to understand the factors driving vegetation recovery in the low-salinity portion of the tidal frame.

Where tidal flooding has a high suspended sediment concentration, as within the Bay of Fundy region (O'Laughlin and van Proosdij 2012), the potential for sediment deposition is high (van Proosdij et al. 2006; Chmura et al. 2001; O'Laughlin and van Proosdij 2012). The response to sediment accretion by coastal vegetation can either be negative (e.g. dying and not tolerating sedimentation) (Ewing 1996; Maun 1998), or have little effect on plant growth (Maun 1998; Deng et al. 2008). There are tolerance limits associated with sedimentation for marsh species (Deng et al. 2008). Deposition of sediments carried by tidal inundation to a tidal wetland, typically consisting of fine grained materials inorganic materials (Amos and Mosher 1985; van Proosdij et al. 2006), and are known to create nutrient poor habitat known to limit vegetation growth of low marsh species, such as *Spartina alterniflora* (Deng et al. 2008). Nutrient limited habitats tend to favour growth of clonal patches or below ground growth (Maun 1998; Deng et al. 2008). Large and sudden sediment deposit events are known to more negatively influence salt marsh vegetation growth than smaller but repeated sediment deposition (Deng et al.

2008). In a macro-tidal estuary there is potential for the formation of a salt wedge (an area where dense salt water flow underlies less dense freshwater flow above it creating a wedge) that can trigger high concentrations of suspended sediments to occur in tidal waters (Burchard and Baumert 1998). Restoring tidal flow on a low saline agricultural dykeland has the potential to create high sedimentation rates that could negatively impact vegetation recovery. Tidal inundation also provides stress to plants as increases in tidal inundation results in a decrease in soil oxygen availability (Deng et al. 2008; Davy et al. 2011). Tidal inundation is determined by calculating the frequency and duration of flooding (Adam 1990, Silvestri et al. 2005; Pétillon et al. 2010). Depleted soil oxygen negatively affects root respiration and growth, seedling development and results in a changed soil chemistry that limits plant growth (Mitsch and Gosselink 2007; Silvestri et al. 2005; Davy et al. 2011). In this paper we examine the relationship between sedimentation, tidal inundation and species recovery following hydrological restoration.

3.2 Study area

The Bay of Fundy is a large hyper-tidal embayment located mostly within Canadian Atlantic provinces at the northeastern end of the Gulf of Maine. Semi-diurnal tides in the upper portions of the Bay of Fundy reach an excess of 16 m on larger spring tides (Desplanque and Mossman 2004; van Proosdij et al. 2006). A large tidal prism combined with high suspended sediment concentration contributes to high sedimentation rates recorded within the region (Daborn et al. 2002; Chmura et al. 2001; van Proosdij et al. 2006). The St. Croix River High Salt Marsh and Tidal Floodplain Wetland

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Restoration Project, Figure 1.1, is located within the upper reaches of the Avon Estuary in Nova Scotia, Canada (Bowron et al. 2008). The site is a ditched agricultural dykeland that lying on what was once tidal wetland. In 1950, the St. Croix Marsh body was incorporated into the MMRA for the region and the site has not been expected to flood with tidal waters since at least that time (Parent 2009). The vegetation community prior to restoration was dominated by pasture grass and no salt marsh species were found within the site. The restoration project consists of 4 separate areas, of which the St. Croix West (SCW) site is largest (12.9 ha) and lies at a mean elevation of 8.98 m (CGVD 28) and is the main study site used in this study (Figure 3.1). Elevations in this report are relative to the Canadian Geographic Vertical Datum of 1928 (CGVD28), which is referenced from the Mean Water Level (MWL) measurements made at tide gauges across Canada in 1928. It is the current standard for vertical datums available within Canada.

In 2007 ecological and hydrological surveys determined that the partial, or complete, removal of the agricultural dyke would allow periodic flooding of tidal water and had the potential to create a productive brackish tidal wetland habitat (Bowron et al. 2008). In 2009, restoration construction started with the excavation of two ponds, positioned adjacent to areas of low surface elevation and close to remnant agricultural ditches. The dyke was breached at 5 locations between August 10 and 19, 2009 (Figure 2.1). For a full description of the construction designs refer to Bowron et al. (2011). The ecological monitoring program developed for the restoration project was based on the global programme of action coalition for the Gulf of Maine (GPAC) regional salt marsh monitoring protocol (Neckles et al. 2002; Konisky et al. 2006) but with modifications

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based on experience gained from previous restoration successes within the area (Bowron et al. 2009; van Proosdij et al. 2010).
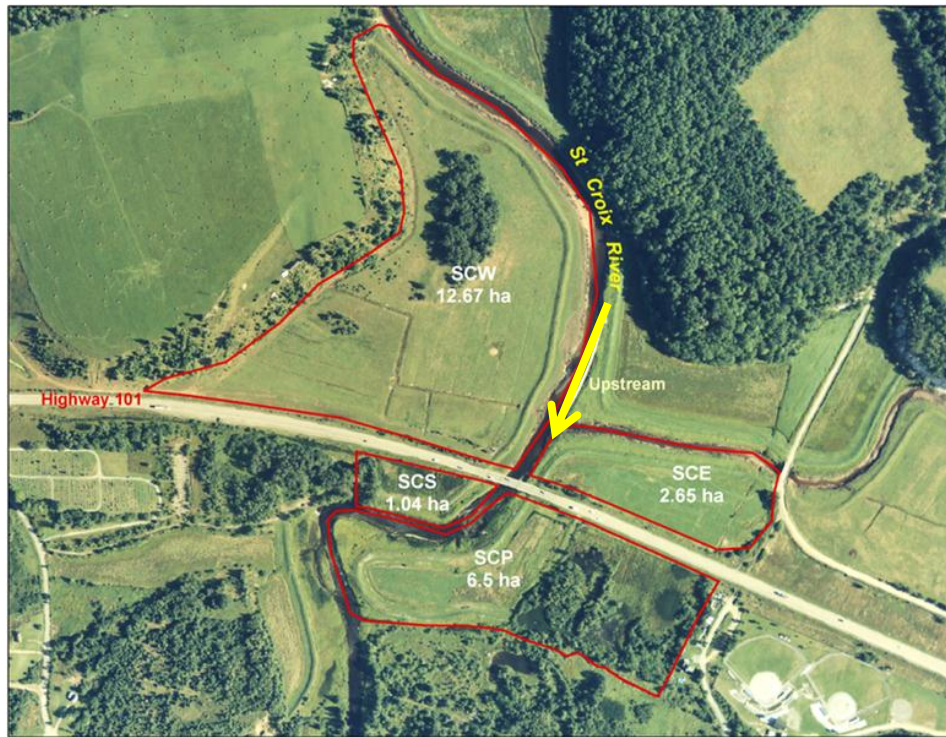
Figure 3.1 The St. Croix River High Salt Marsh and Tidal Floodplain Wetland Restoration Project is made up of 4 separate quadrats and lies at the intersection between a provincial highway and the St. Croix river. The St. Croix West site (SCW) is the main study site used in this research. Sourced with permission from Bowron et al. (2010).

3.3 Methods

As the requirements for this project aimed to examine marsh-wide processes and vegetation recovery, it was decided to use a systematic random sampling design approach using 20 of the previously established CB Wetlands & Environmental Specialists Inc. (CBWES) stations and 27 new stations. Three stations were lost during the course of the survey as some soil data were lost during processing, or missed during vegetation and

environmental surveys. These stations were removed from analysis within ArcGIS 10.0. Sample stations were geo-referenced using a high resolution survey grade Real-time Kinematic Differential GPS system with centimeter level accuracy, a Trimble R6 radio receiver (collected in NAD83 UTM Zone 20) and geodetic elevation relative to CGVD28 (Canadian geodetic vertical datum).

Elevation surveys (with cm level accuracy) of the sample stations were performed concurrently with vegetation surveys each year (Oct 29, 2010 and August 29, 2011). The locations of the breaches and the associated created channels were surveyed in Oct 2010. Vegetation was surveyed using 1 m^2 plots in August 2010 and September 2011. At each plot a point intercept method was used to record species composition and abundance, where abundance is the frequency of contact of each species with 25 points per plot (Roman et al. 2002; Bowron et al. 2009). Statistical analysis aimed at addressing variables that would indicate progress of restoration. For each plot, individual species abundance were pooled together creating vegetation classes based on a salinity habitat class (salt marsh species, brackish marsh, tidal fresh marsh and non-flooded). These vegetation habitat classes were derived from literature and help from sources at Saint Mary's University (e.g. Magee 1981; Rodwell 2000; Tiner 2009; Dr. Jeremy Lundholm, pers. comm.). Statistical analysis was performed using a series of analysis of variance ANOVA and t-tests, where the independent variable was site location and dependent variables were tested using IBM SPSS 19.

Soil samples were collected once in the spring (May/early June) and once in the late summer/fall (August/September) of each growing season to measure organic matter (OM) and bulk density (BD). Soil samples were collected using plastic syringes (4 cm

diameter) modified by removing the needle from the barrel. Soil samples were collected to a depth of 10 cm. To gather organic matter content and water content of soil samples, each core was placed in a crucible and each sample was weighed and placed in a muffle furnace for two hours at 550 degrees C. Samples were then cooled and weighed to gain loss on ignition (LOI) of organic material. To calculate bulk density of each sample a known volume of the sample was placed in a crucible then oven-dried for 16 hours at 105 °C.

The hydroperiod for both inundation frequency and inundation time was modelled using predicted tides from Tides and Currents software and the elevations from the annual elevation survey of each sample station. Predicted tides were converted from Chart Datum (CHS) to Canadian Geodetic Vertical Datum 28 (CGVD28). Inundation frequency was calculated for 10 cm elevation increments and queried to count the total number of predicted high tides that were above each given elevation then divided by the total number of tides for the year. Inundation time was calculated using 10 cm elevation increments and queried to count all tides greater than the given elevation in 5 minute intervals and divided against the inundation frequency.

3.4 Results

To examine the relationship between marsh position, flooding and changes to the marsh elevation platform, the 47 sample stations were divided into three marsh location classes: 37 stations classed as 'within marsh', 7 stations as 'fringe marsh' (based on their locations on the river side of the dyke), and 3 stations as 'non-flood' (two stations located

on the dyke and one station within the centre island and were not expected to flood). However, one station was located on the dyke but during restoration construction become part of the marsh surface. Due to elevation changes associated with moving a point vertically from the top of the dyke to marsh surface this point was removed from elevation analysis. Annual elevation surveys of sample stations within the marsh remained subsided (6.8 ± 0.3 m CGVD28) than both the surrounding fringe marsh (7.2 ± 0.6 m) and the non-flooded sites (8.3 ± 0.4 m) as seen in Figure 3.2.

ANOVA showed that there was a difference in elevation among station location (ANOVA $f = 3.10$, df $= 2$, $p = < 0.001$). Increased flooding and sedimentation has significantly increased station elevations. Over the course of the two year study period, the elevation of stations within the restoration site increased by 15 ± 13 cm (Paired $t = 2.03$, df $= 35$, $p = < 0.001$), fringe marsh increased by 9 ± 8 cm (Paired $t = 2.45$, df $= 6$, $p = 0.03$), and there was no change to report on sites that did not flood (Paired $t = 4.30$, df $= 2$, $p = 0.41$). One station within the marsh decreased by 40 cm over the first year and this change was associated with an area of pooling prior to restoration that dewatered following restoration. Restoring a natural hydrology onto the site also had affects felt on the fringe marsh, as noted by the high increase of fringe marsh elevation.
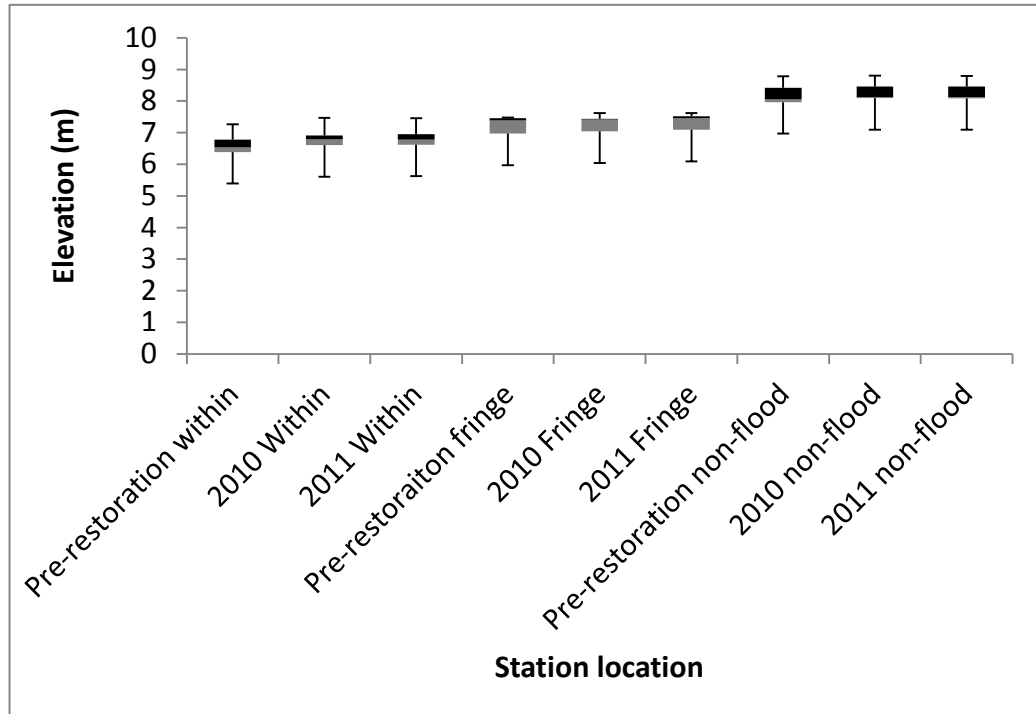
Figure 3.2 Elevation of the marsh showing subsided nature of marsh platform.

To examine the rate of elevation change over the study period, the monthly rate of change was calculated by subtracting the total change from the pre-restoration values to the two year post values and divided over the number of months passed. The final value is expressed in millimeters per month (Figure 3.3). Overall, stations within the marsh in the first year following restoration experienced the greatest increase in elevation (9 mm $\pm$ 8 mm per month) (Figure 3.3). Stations located on the fringe increased by 3 mm $\pm$ 7 mm per month during the same time period. In the second year of restoration there was a reduction in the intensity of elevation increase as stations within the marsh recorded increases of 0.1 $\pm$ 0.9 mm per month, and fringe sites recorded increases of 3 $\pm$ 5 mm per month.

RSET elevation increases within the first year of restoration also recorded high rates of sediment accretion as values ranged from a high of 23.04 $\pm$ 0.39 cm/yr^{-1} and a low of 13.09 $\pm$ 0.35 cm/yr^{-1} (Bowron et al. 2011). The high sediment elevation increases for the St. Croix site are higher than some of the other rates recorded on salt marsh restoration projects within the area: 1.7 cm/yr^{-1} on the Walton river, N.S.(van Proosdij et al. 2010) and 0.6 cm during the first year of restoration on the Cheverie river, N.S. (Bowron et al. 2011). At the St. Croix site the high sedimentation within the marsh during the first growing season following restoration resulted in the formation of large mud flats across a significant portion of the site.
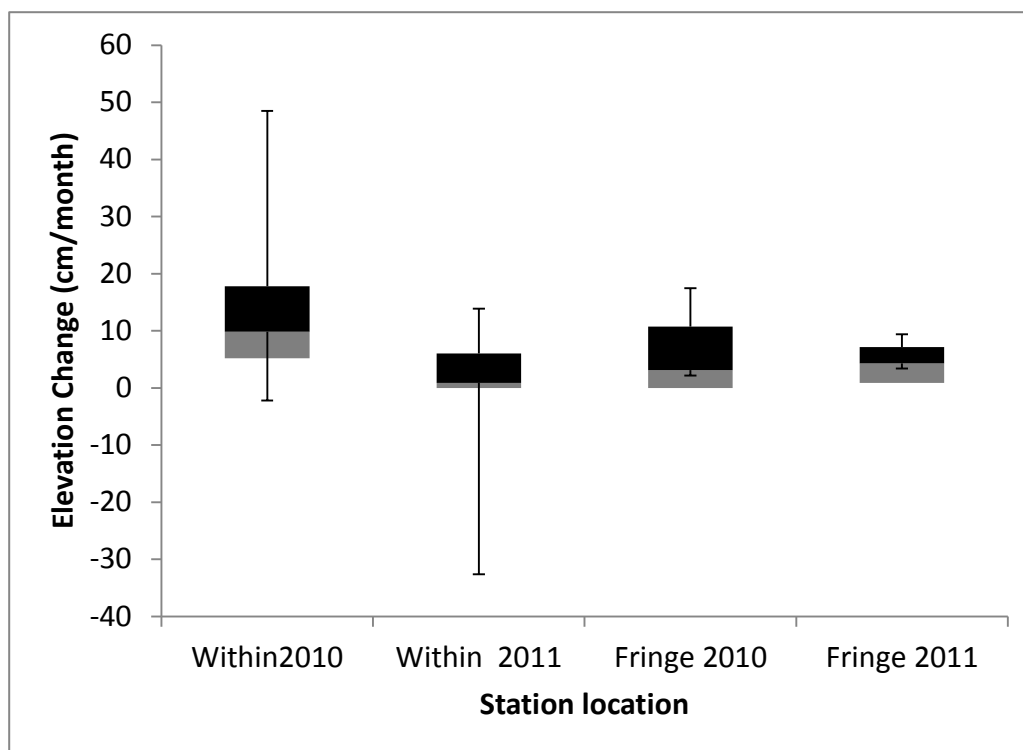
Figure 3.3 Monthly rate of elevation increase recorded at St. Croix is shown by station location and time since restoration. Large monthly rates were recorded in the first year of restoration.

To examine the spatial pattern of sediment increases recorded at the sample stations, two proportional symbol quantile display maps were created, Figure 3.4 and Figure 3.5. The first map (Figure 3.4) shows the elevation changes that occurred over the marsh in the first year of restoration when compared to the pre-restoration elevations. In 2010, there were 7 sample stations within the marsh that recorded over 20 cm increases from their pre-restoration value, 6 stations recorded more 15 cm increases, with 1 station recording a decrease in elevation of 7 cm. During the second growing season (Figure 3.5) there were only four stations that recorded increases of more than 10 cm and none were

above 15 cm, suggesting that sedimentation as an agent of change was greater during the

first year than the second.
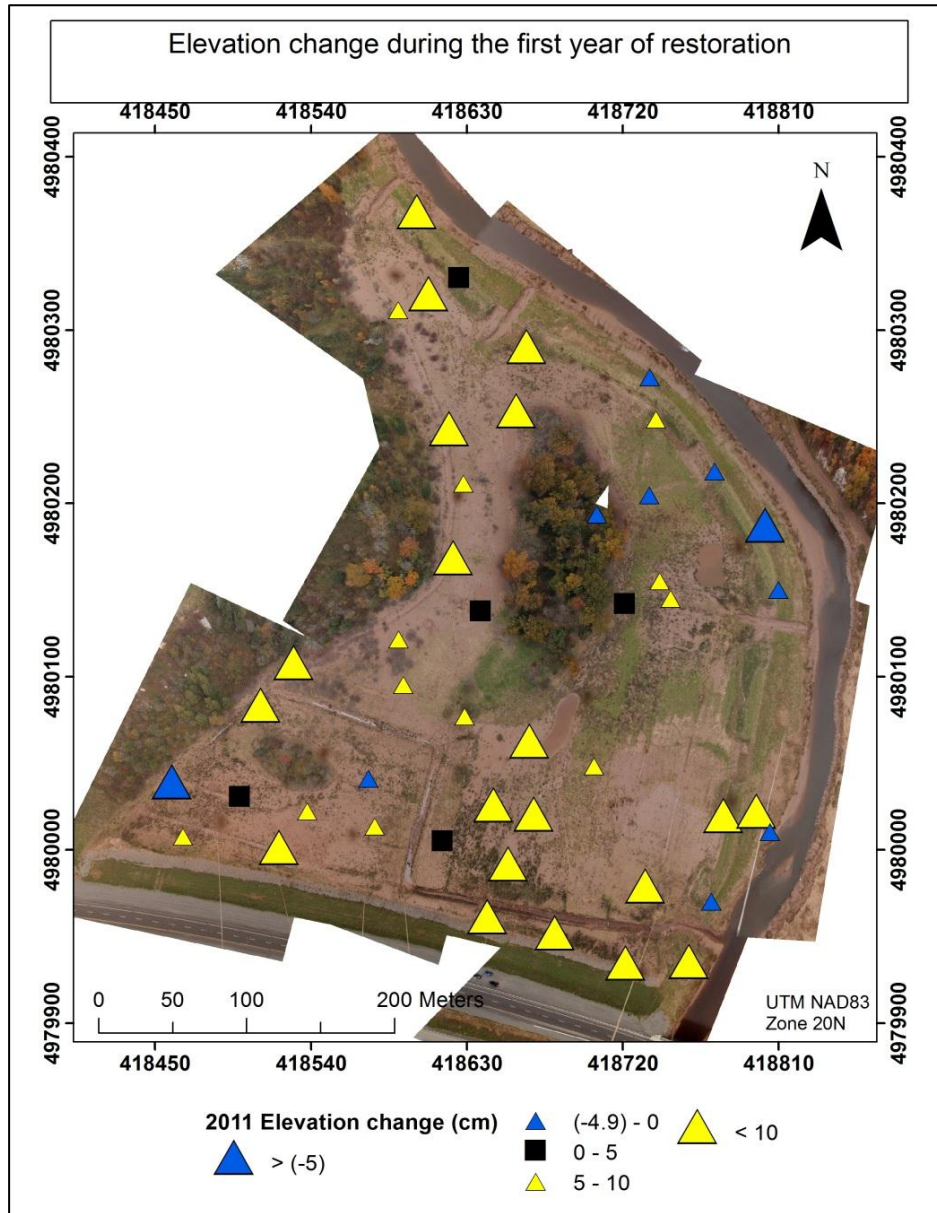
Figure 3.4 Quantile proportional symbol map showing elevation change (cm) prior to restoration and the first year of restoration recorded at sample stations.

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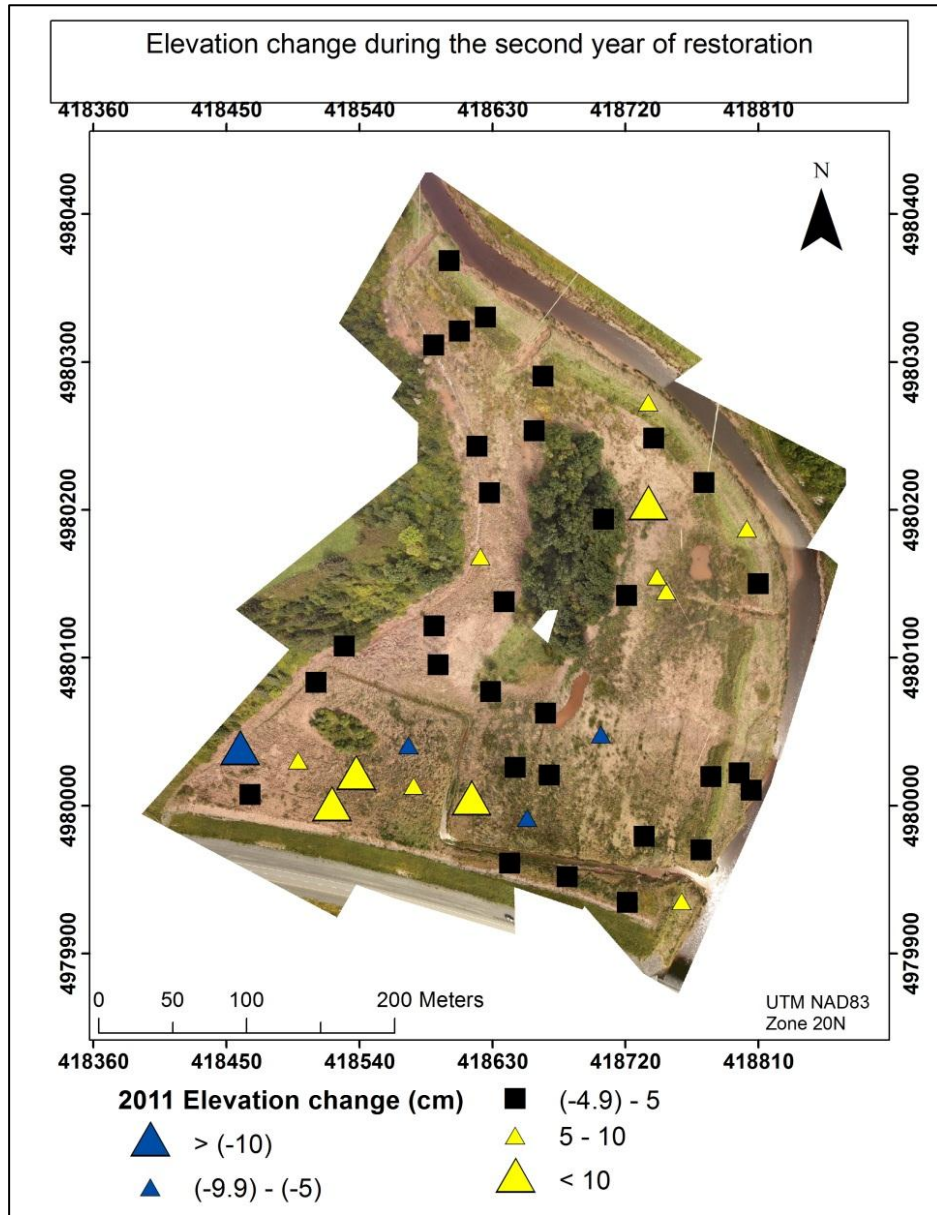
Figure 3.5 Quantile proportional symbol map showing elevation change (cm) prior to restoration and the second year of restoration recorded at sample stations.

Hydrology

Inundation frequency curves (Figure 3.6) and inundation time curves (Figure 3.7) curves were created from the elevation of sample stations for each year and the predicted tide heights from the local tide signalling station. The difference in year with an increase in inundation represents an expected increase in higher tides that align with the 18.03 year Saros tide cycle (Desplanque and Mossman 2004). Repeated measures ANOVA (with station location as the independent variable and both inundation frequency and inundation time as dependent variables), shows that there is no difference in samples grouped by year but there is a difference in flooding variables (Figure 3.7). Paired *t*-test showed that there is no difference in inundation frequency between years within the marsh (Paired *t* test : t =1.74 , *df* = 36, *p* = 0.09) or on the fringe marsh (Paired *t*-test: 1.13, df = 5, *p*= 0.31). Inundation time was different within the marsh during the first year of flooding (Paired *t* test : t =2.02 , *df* = 36, *p* = < 0.001). There was no difference in inundation time between years for stations located on the fringe marsh (Paired *t* test : t = 2.57 , *df* = 5, *p* = 0.21). This shows that in the first year stations within the marsh flooded differently than any other location or year. This also supports the idea that the increase in flooding duration aided the formation of the large mudflats during the first year as higher inundation time is related to a higher potential for sediment deposition to occur (van Proosdij et al. 2006).
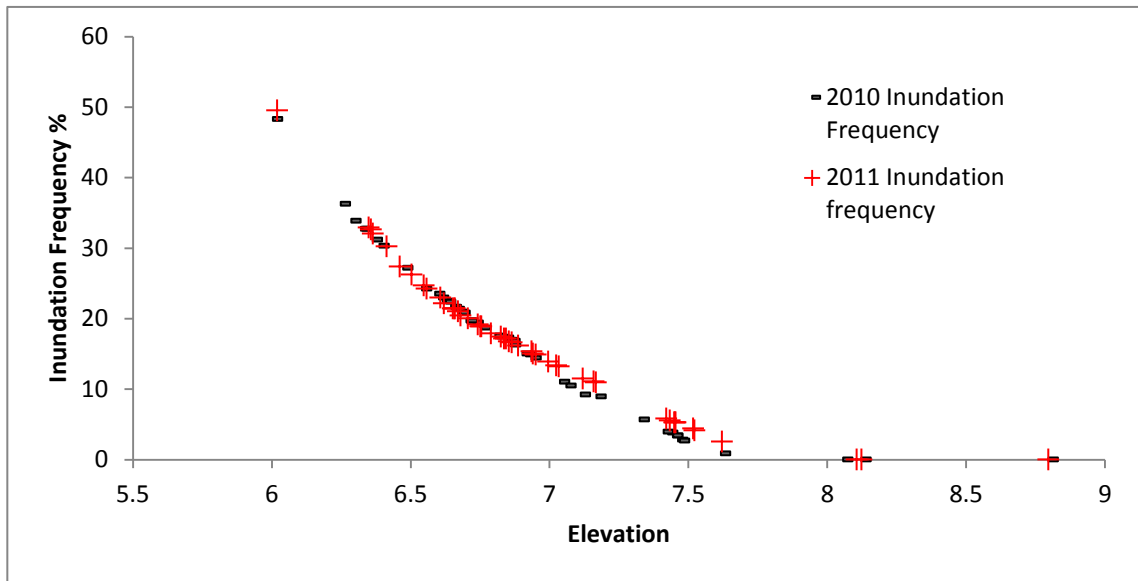
Figure 3.6 Inundation frequency curve for both 2010 and 2011.
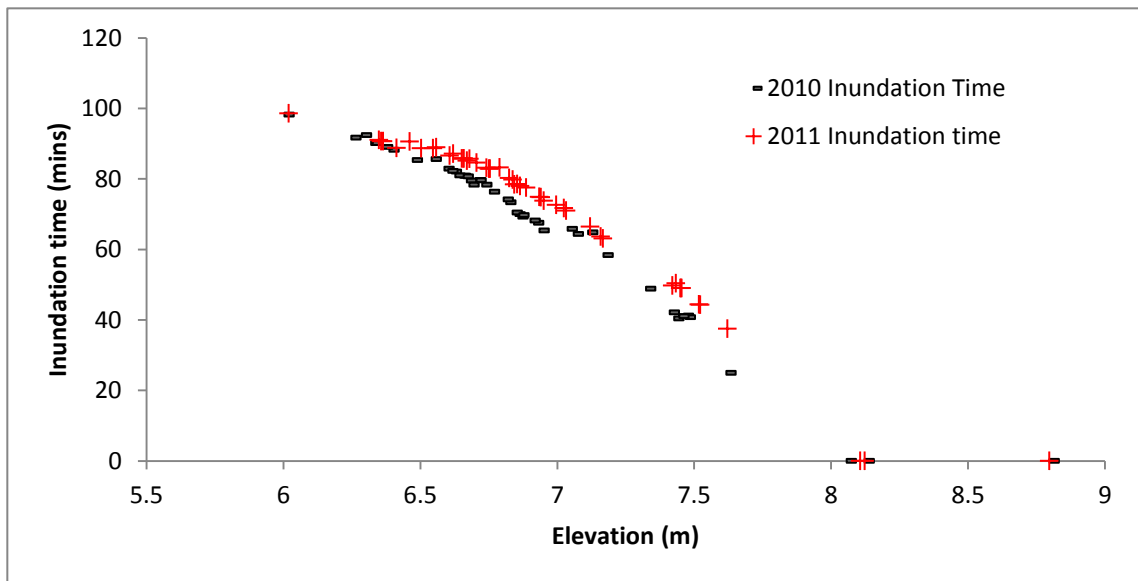
Figure 3.7 Inundation time curve for 2010 and 2011

Table 3.1 Repeated measures ANOVA for Inundation frequency % and Inundation time
(mins) for (a) fringe plots and (b) within plots.

(a)					
Factor	*df*	*SS*	MS	F	P
Station	1	799.94	799.94	2.24	0.150
Variables	1	9257.66	9257.66	25.93	< 0.001
Station x Year	20	7140.59	357.03		
(b)					
Station	1	93.33	93.33	0.86	0.354
Variables	1	122389.14	122389.14	1132.31	< 0.0001
Station x Year	144	15564.71	108.09		
df = degrees of freedom; SS = sum of squares; MS = mean of squares					

Soils and Sediments

Hydrologic restoration caused significant changes to the soil structure by bringing
in new sediments to the site from the Bay of Fundy. Examining data shared from pre-
restoration (Bowron et al. 2008) show that grain size and organic matter content from
each station experienced change. Prior to restoration, within the marsh stations have a
mean soil organic matter content of 24.3 ± 6.97 % which dropped to 5.06 ± 2.21 %, and
then 3.71 ± 1.34 %, in subsequent years (Figure 3.8). Soil organic matter content on the
fringe stations also decreased post-restoration. Soil bulk density seemed to oscillate
between years as prior to restoration mean bulk density was 1.19 ± 0.08 g/ml within the
marsh, dropped to 1.02 ± 0.16 g/ml and climbed back to 1.14 ± 0.13 g/ml (Figure 3.9).
Mean grain size prior to restoration was 119.30 ± 307.01 μm but dropped to 7.63 ± 1.30
μm, and then 8.20 ± 2.19 μm in subsequent years. Prior to restoration two soil cores
recorded high mean grain size that ranged from of 532.4 μm and 577.7 μm that fall into
the coarse sand range and had a bimodal distribution (Figure 3.11) . These two samples

were located adjacent to the large centre island which is thought to be a remnant of the glacial history of the area (Bowron et al. 2008). These two stations were excluded from the grain size graph but not from the analysis. Breaching of the dykes resulted in a soil surface trending towards a less organic a finer grained platform. Returning tidal inundation to the St. Croix site caused changes to occur within the dyke but also on the fringe marsh as well.
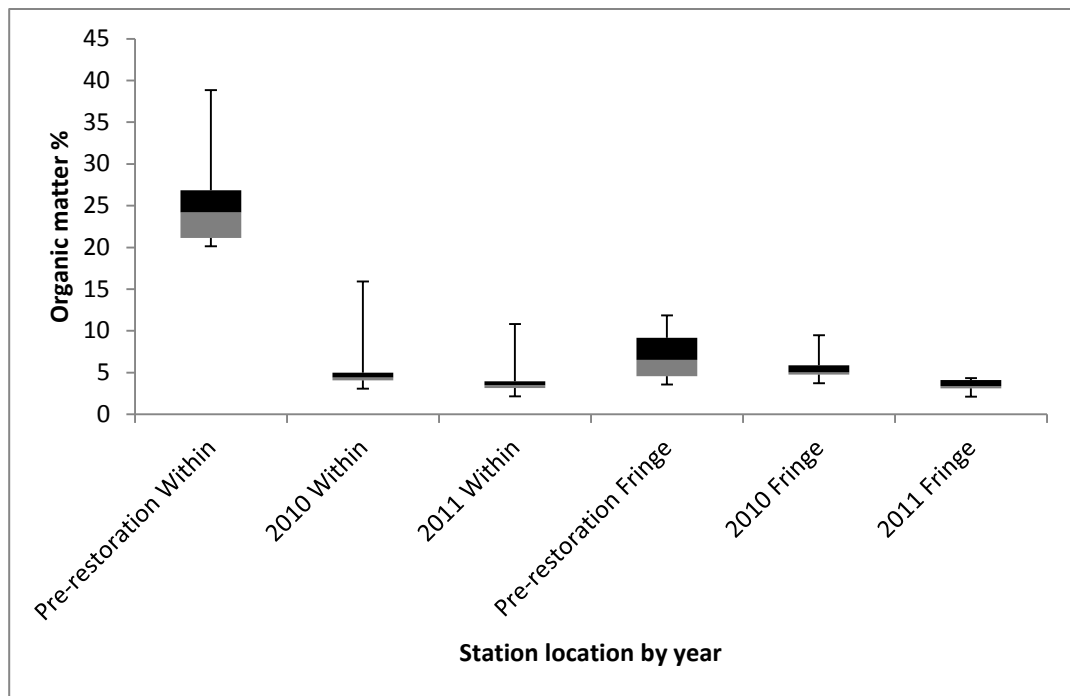
Figure 3.8 Organic matter content at the study site over the two year period shown by station location.
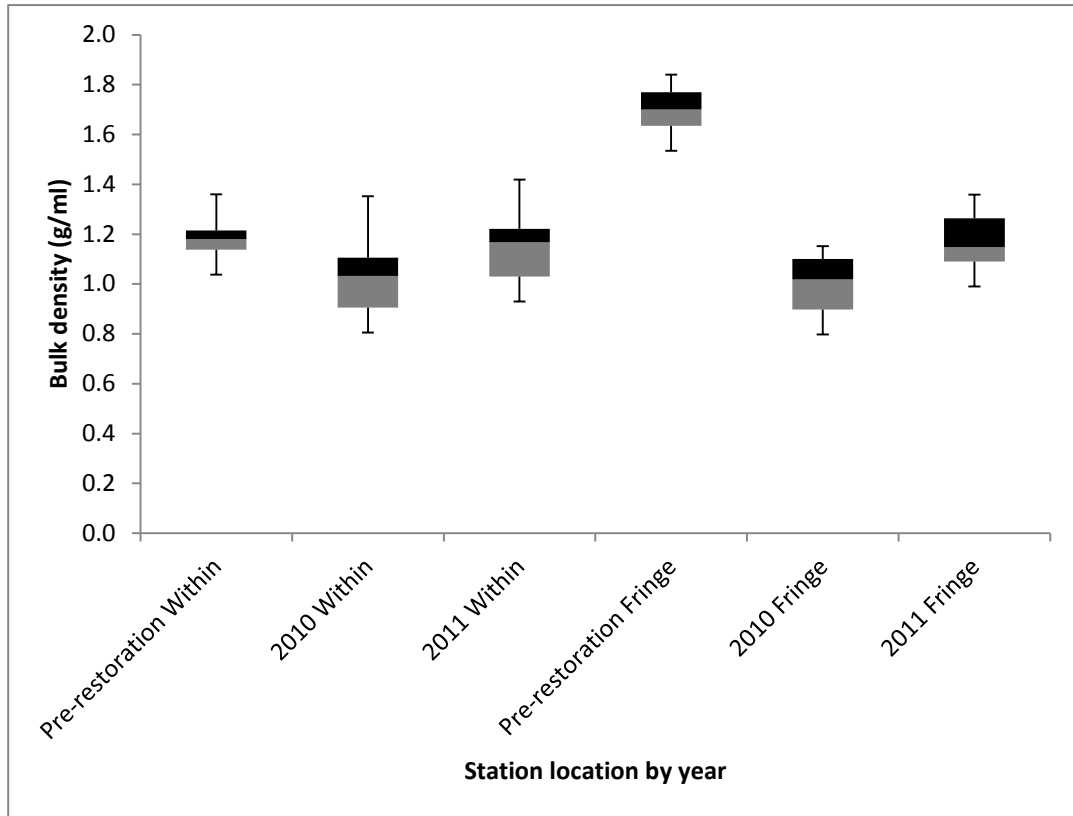
Figure 3.9 Soil bulk density (g/ml) at the study site over the two year study period.
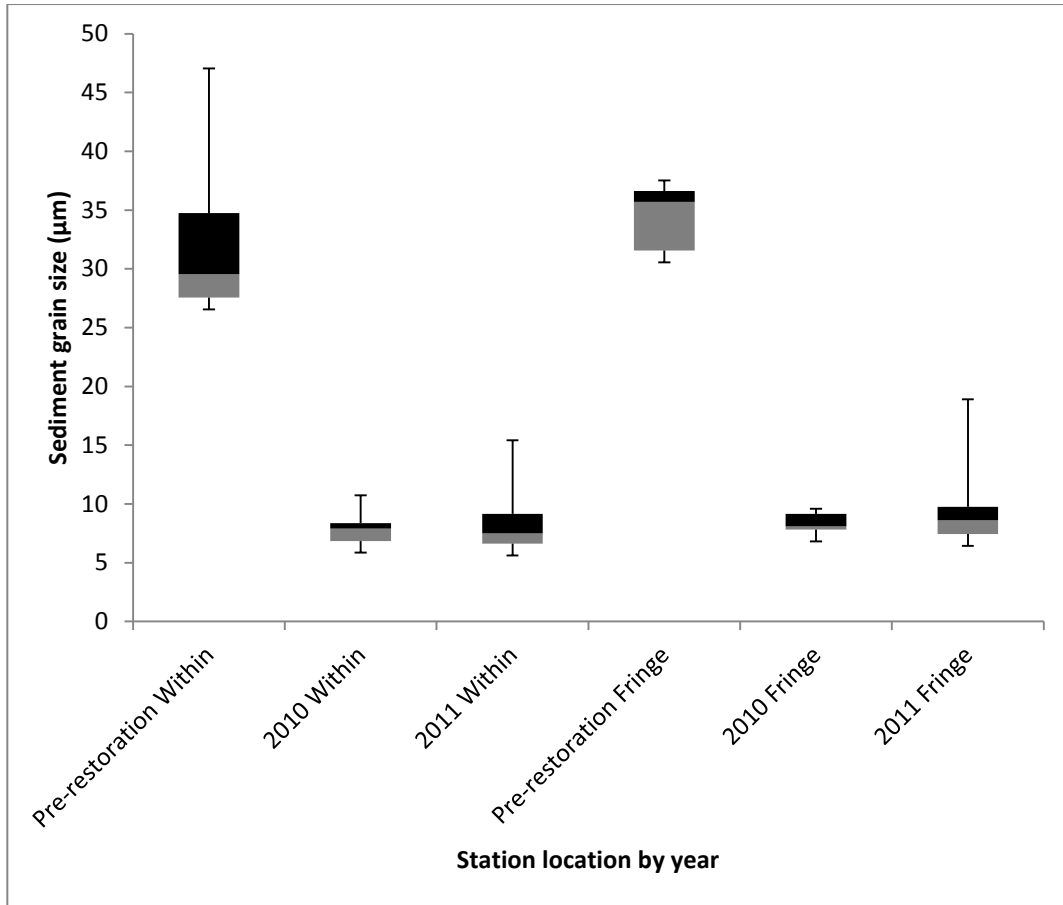
Figure 3.10 Sediment grain size at the study site over the two year study period.

Vegetation

Over the two year study period a total of 60 different species were recorded on site (Appendix A). These species were classified according to salinity habitat types using local flora guides (Magee 1981; Rodwell 2000; Tiner 2009) and a list is found in Appendix B. After two growing seasons, the return of tidal flooding to the site has increased the abundance of halophytes and brackish species but decreased the abundance of fresh water species, non-tolerant species and the bare ground cover (Figure 3.11). Repeated measures ANOVA (with station location as the independent variable and

abundance by year and vegetation class as dependent variables) show that within the marsh there is a significant effect of survey year and vegetation class; on the fringe marsh there is no effect between survey years however the vegetation classes are different between years; on non-flooded sites years are not different but the vegetation classes are different (Table 3.2).

The most abundant species over the course of the two first growing seasons are shown in Figure 3.12. There is a significant increase in the abundance of *Spartina alterniflora* and *Agrostis stolonifera* for salt marsh species; abundant brackish species include *Alopecurus geniculatus*, *Schoenoplectus tabermontani, Spartina pectinata, Polygonum neglectum, Typha sp.* and *Juncus articulatus.* In the second growing season, a new colonist of fresh water species, *Alisima trivale*, was recorded. However *Calamagrostis canadensis,* another fresh water tolerant species, was removed from the site. *Juncus effusus* and *Poa pratensis* both decreased in abundance. There is a general decrease in up-land species: *Lolium perenne* and *Trifolium repens*. Restoring tidal flooding to the sites has resulted in a decrease in fresh water species abundance and an increase in brackish and halophytic abundance on the marsh as time since restoration increases. A proportional symbol map for the 2010 season (Figure 3.13) shows the dominant vegetation class shown by colour and the elevation increase for the 2010 season shown by size. A proportional symbol map for the 2011 season (Figure 3.14) shows the dominant vegetation class shown by colour and the elevation increase for the 2011 season shown by size. Visual comparison between the two maps show that bare ground patches found in 2010 became colonized by brackish species or halophyte dominant plots in

2011; areas of high sediment accretion have changed location spatially; and very few

fresh water patches and bare ground patches remain in 2011.

Table 3.2 Repeated measures ANOVA for vegetation class (a) within the marsh, (b) fringe marsh, (c) non-flooded sites

(a)					
Factor	*df*	*SS*	MS	F	P
Year	1	862.77	862.77	6.97	< 0.001
Class	4	14053.81	3513.45	28.37	< 0.001
Station x Class	360	44577.95	123.83		
(b)					
Year	1	64.13	64.13	0.62	0.435
Class	4	3883.34	970.84	9.34	< 0.001
Station x Class	60	6238.29	103.97		
(c)					
Year	1	26.13	26.13	0.43	0.519
Class	4	5513.53	1378.38	22.75	<0.001
Station x Class	20	1212.00	60.60		
df = degrees of freedom; SS = sum of squares; MS = mean					
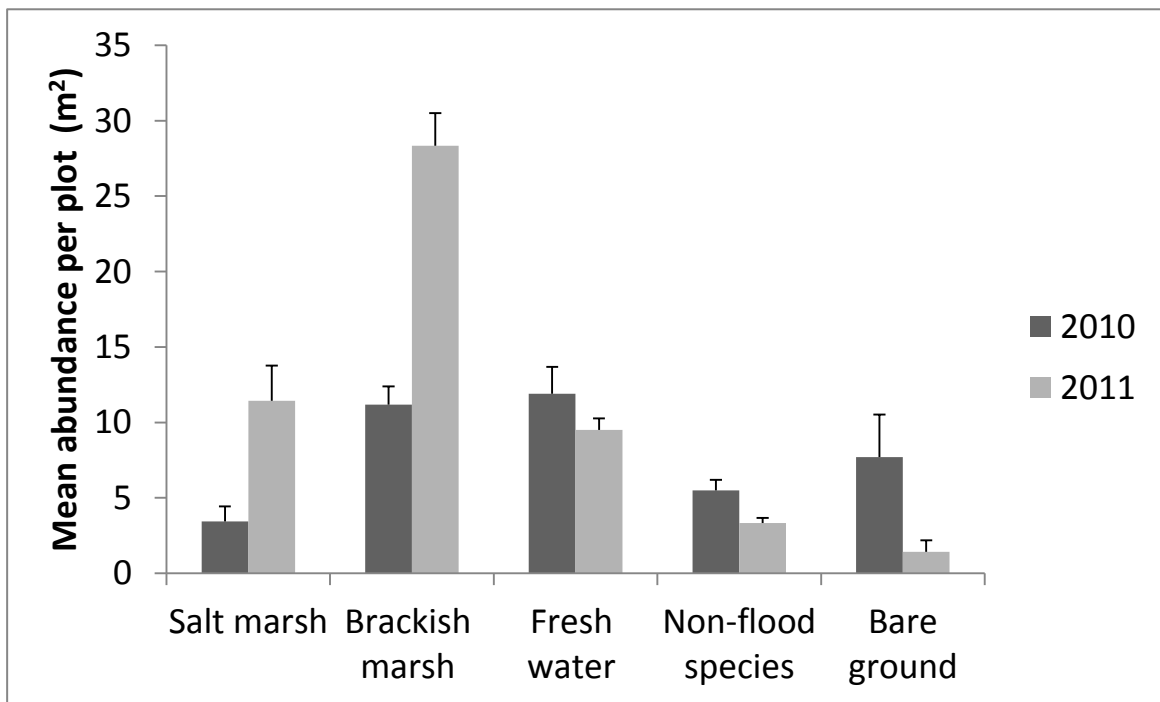
Figure 3.11 Mean abundance of vegetation by habitat class for the two year study period.
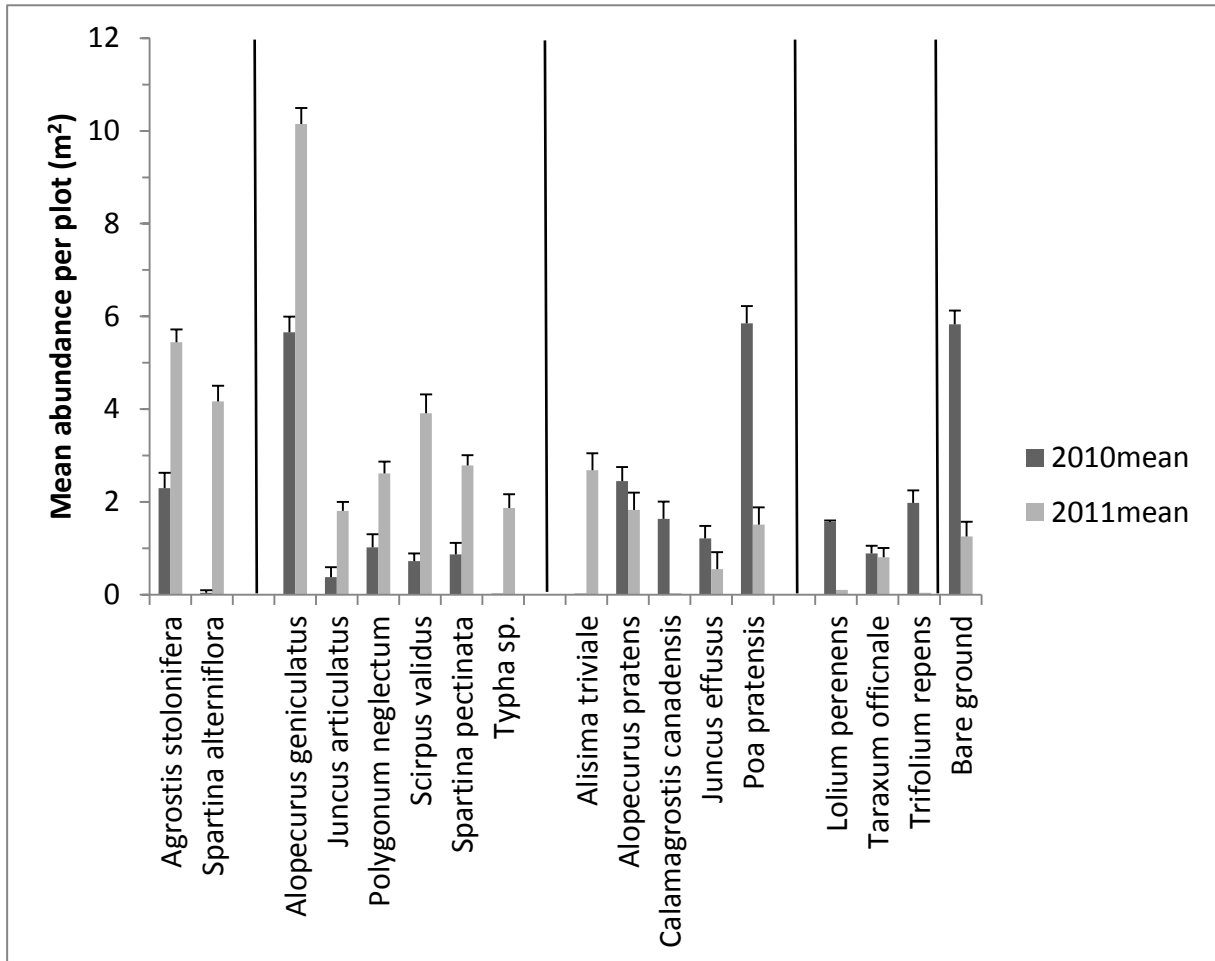
Figure 3.12 The most abundant species recorded during the two year study site. Species classed by salinity habitat with halophytes on the left, brackish species, fresh water species and non-tolerant species as well as bare ground abundance. The spaces in between columns represent habitat divides (right to left: salt marsh species, brackish marsh species, fresh water species, upland species, and bare ground).
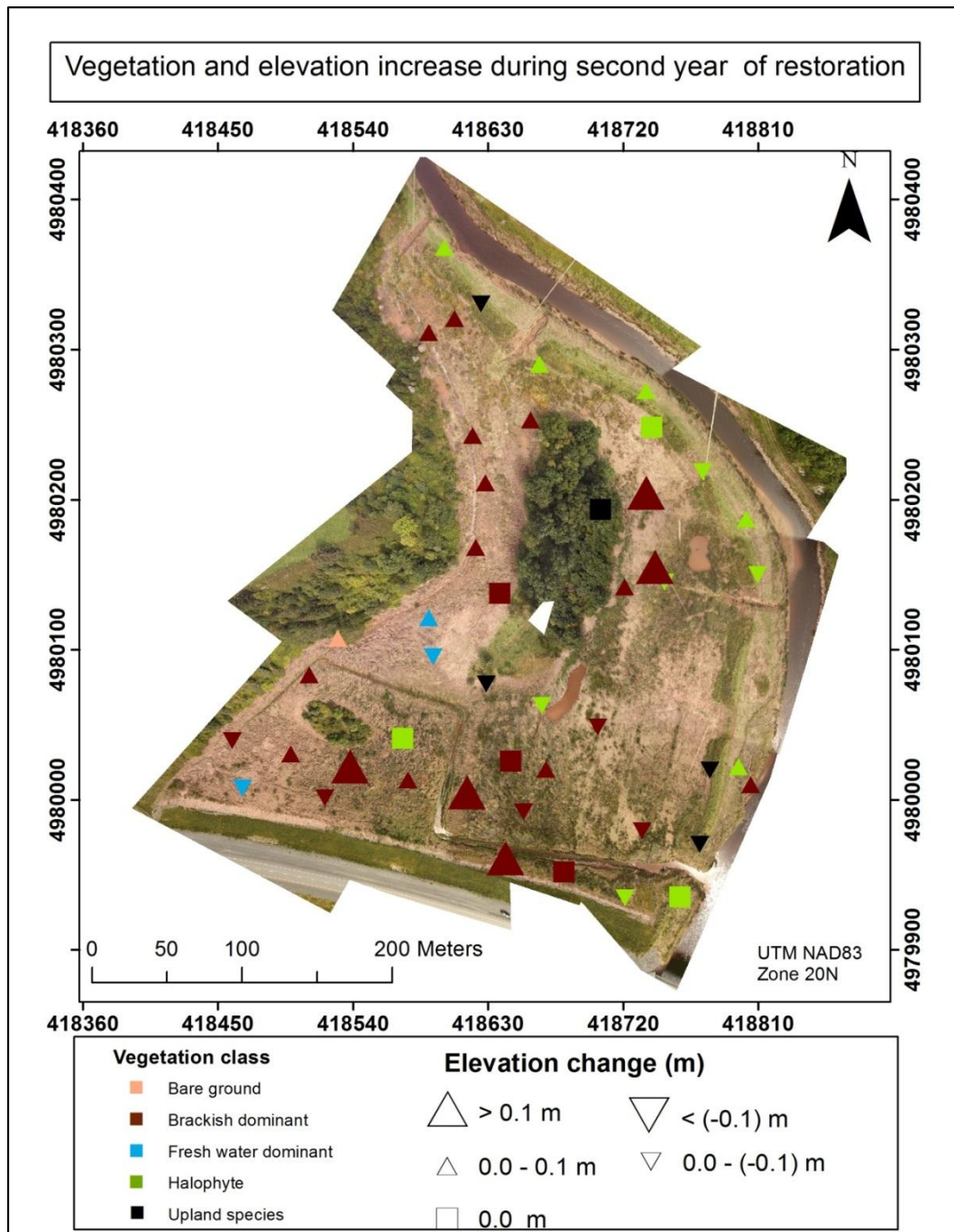
Figure 3.13 Proportional symbol map showing the dominant vegetation class for each sample station and the elevation increase during the 2010 season.

3.5 Discussion and conclusion

The return of tidal flooding to the St. Croix site triggered vegetation community change where the terrestrial agricultural vegetation present prior to restoration was removed and replaced by salt and brackish vegetation in the seasons immediately following restoration. At the St. Croix site, the return of tidal flooding triggered a high sedimentation event that aided the colonization of new species. The subsided marsh platform was shown to have higher inundation time than the fringe marsh. Higher inundation time suggests that there is a higher sediment deposition potential within the marsh than on the fringe. Bowron et al. (2010) measured suspended sediment concentration in tidal flooding near the site and was shown to have 14 % of samples fall within the fluid mud range as defined by Gaun et al. (1998). The potential for high sedimentation as the site, as given by the subsided nature of the marsh platform and the high suspended sediment concentration in the tidal waters, was realized by the breaching of the dyke. During the first year of restoration, high rates of sediment elevation increases were recorded by the RSET, which ranged between 23.04 ± 0.39 cm/yr^{-1} and 13.09 ± 0.35 cm/yr^{-1} (Bowron et al. 2011), and the elevation station surveys, which recorded an increase in elevation of 9 ± 3 cm and the fringe marsh experienced elevation increases of 4 ± 3 cm. The high elevation increase rates recorded at this site exceed other restoration sites within the region have recorded using similar methods: Walton river restoration site recorded 1.7 cm/yr-^{1} (van Proosdij et al. 2010) and the Cheverie creek restoration site recorded 0.6 cm during the first growing season (Bowron et al. 2009). The high rates of sedimentation at the site are what created the disturbance to remove vegetation prior to

restoration and created a platform for colonization to occur in the seasons following restoration. Mean pore water salinity was also recorded on the site (Bowron et al. 2010), and showed values ranging from a high of 2.6 ± 1.2 ppt and a low of 0 ppt (Bowron et al. 2011). The salinity readings at the site fit well into the oligohaine marsh category (Odum 1988) and are low when compared to salinity readings in other restoration projects within the area, that record rates ranging from 18 ppt to 35 ppt (Bowron et al. 2009; van Proosdij et al. 2010). Oligohaline marshes act as intermediate marshes where vegetation experiences intermittent and reduced salt stress relative to more saline sites (Odum 1988; Brewer and Grace 1990).

At the St. Croix site, sedimentation acted as the agent driving ecosystem change by first reducing the surface biomass of non-tolerant vegetation and then aiding colonization by providing new habitat for salt-tolerant vegetation within a low saline environment. Prior to restoration the area covered by bare ground was one hit per plot, peaked at a mean abundance of 5.83 species per plot and then dropped to 1.8 species per plot during the second growing season. During the 2010 season large mudflats were created and by the second year these became covered with vegetation. Of the survey plots that were dominant bare ground in 2010, 33 % of these became dominant salt marsh plots, 58 % became brackish plots by 2011, and 8 % remained bare ground, Table 4.1 None of the 2010 bare ground plots were located on the fringe marsh and none of these plots had upland or fresh water species return. Salt marsh vegetation is well-known to have a competitive disadvantage in the fight for resources when located beside less salt tolerant vegetation in low saline environments (Bertness 1991; Crain et al. 2004; Crain et al. 2008). Mudflats created by high accretion rates have been shown to have a different

biochemistry in their soils than patches of ground with vegetation that negatively affect less salt tolerant vegetation (Davy et al. 2011). Bare patches of ground are known to have higher salt content (Crain et al. 2008) that can also affect vegetation community structure. Within the low salinity portion of a marsh, patches of bare ground are known to act as refugia for competitively subordinate species, such as *Spartina alterniflora* and *Salicornia europaea*, which increases the overall biodiversity of a site (Bertness 1991; Emery et al. 2001; Ewanchuk and Bertness 2004; Crain et al. 2008; Davy et al. 2011).

Examination of the vegetation data also show that the colonization of *Spartina alterniflora* on the site occurred via long distance dispersal. Hydrochory, transport of seeds by water, is a well-known method of dispersal for salt tolerant vegetation (Wolters et al. 2005; Elsey-Quirk et al. 2009). The vegetation surveys from all years under the study period did not find any *S. alterniflora* located in the fringe plots or any adjacent fringe areas. *S. alterniflora* was only found within the site. T-tests comparing the inundation time and inundation frequency of plots that had *S. alterniflora* present against those that did not have the species present shows that where *S. alterniflora* occurs is flooded more frequently (t-test = 1.99, df = 85, p = 0.008) and for longer periods of time (t-test = 1.99, df = 85, p = <0.001). *S. alterniflora* is being limited to those areas where there is a stronger environmental pressure.

Overall this study demonstrates that it is possible to re-initiate tidal marsh vegetation succession in low saline environments. Sedimentation and salt pulses are the main drivers of recovery and highlights plant succession following hydrologic restoration. This research will help guide future restoration projects with a low saline portion of a macro-tidal estuary by showing that sedimentation can be an important ecosystem

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engineer within the restored site. Sedimentation created bare patches of ground that killed off non tolerant vegetation and then provided a platform for the colonization of more tolerant vegetation.

Table 4.1 Sample stations that were bare ground in the first growing season were colonized by brackish and salt marsh species in the second growing season.

Station location	2010 dominant plot	2007 most abundant species	2007 second most abundant	2011 most abundant species	2011 second most abundant	2011 dominant plot
Marsh	Bare ground	#	*Trifolium sp.*	*A. stolonifera*	*S. pectinata*	Salt marsh
Marsh	Bare ground	#	*Trifolium sp.*	*S. alterniflora*	*A. geniculatus*	Salt marsh
Marsh	Bare ground	*	*	*A. stolonifera*	*E. repens*	Salt Marsh
Marsh	Bare ground	*	*	*S. alterniflora*	*A. geniculatus*	Salt Marsh
Marsh	Bare ground	#	*Trifolium sp.*	*P. neglectum*	*C. spicata*	Brackish
Marsh	Bare ground	#	*J. effusus*	*A. geniculatus*	*A. trivale*	Brackish
Marsh	Bare ground	#	*Trifolium sp.*	*S. tebermontani*	*A. trivale*	Brackish
Marsh	Bare ground	#	*Trifolium sp.*	*J. articulatus*	Bare ground	Brackish
Marsh	Bare ground	*	*	*S. tabermontani*	*S. pectinata*	Brackish
Marsh	Bare ground	*	*	*S. tabermontani*	*S. alterniflora*	Brackish
Marsh	Bare ground	*	*	*A. geniculatus*	*S. alterniflora*	Brackish
Marsh	Bare ground	*	*	*Bare ground*	-	Bare ground

Notes:
1) # Indicates grazed pasture grass .The species could not be identified due to lack of flower and much of the leaf blades were chewed by cows.
2) * Inidcates that the station does not have data prior to restoration or bare ground dominant.

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CHAPTER 4: SYNTHESIS

The goal of this research was to investigate factors driving the response of an agricultural dyke land located within a macro-tidal estuary to hydrologic restoration. This was achieved by using two spatial scales over the course of the two first growing seasons of restoration. Low altitude aerial photography was used to examine marsh wide response, and on the ground quadrat surveys examined changes to ecosystem properties and vegetation recovery. By using the two research scales a broader understanding of vegetation response to tidal inundation was achieved. The findings of this research showed that restoring tidal inundation to a subsided agricultural dykeland can result in rapid and dynamic changes over a short (less than two years) period of time.

The use of low altitude aerial photography provided insight on the marsh-wide response to tidal inundation. Classification of subsequent aerial images, by employing the toolsets in ArcGIS 10.0, showed marsh-wide changes that occurred following restoration. The first growing season was marked by high sedimentation rates that resulted in the formation of large mudflats over large areas of the marsh. Classified images from the first year showed that 58 % of the marsh surface is covered with bare ground, with 36 % being attributed to having been bare ground throughout the first year and 22 % as species die-off. By the end of the second growing season, the mudflats were replaced by an almost complete cover (93 %) of vegetation of the surface of the marsh. The use of aerial photography to understand ecosystem response to restoration is limited without an investigation of the on the ground conditions. Analysis of quadrat surveys showed that vegetation recovery followed the edaphic response to tidal flooding of an agricultural dykeland. Mean grain size of sediments on the marsh prior to restoration were 119.30 ± 307.01 μm (categorized as coarse sand) but dropped to 7.63 ± 1.30 μm, and then 8.20 ±

2.19 μm (categorized as silt and clay) in subsequent years. The finer sediment grain size class of soil samples following restoration, point to a tidal origin of sediments. Sediments are flocculated have smaller grain sizes. Organic matter content decreased in the seasons following restoration, which were 24.3 ± 6.97 % prior to restoration but dropped to 5.06 ± 2.21 %, and 3.71 ± 1.34 % in subsequent years, pointing to a mineorganic origin of soils typically associated with the Bay of Fundy (O'Laughlin and van Proosdij et al. 2012). Re-introducing tidal flooding to the site had a positive effect on halophytic and brackish species abundance and a negative influence on fresh water and upland species. In the subsequent growing seasons following restoration, tidal flooding has increased the presence of halophyte species to 11.4 ± 2.3 species/ m^2 , brackish species abundance increased to 28.3 ± 2.2 species/ m^2,while fresh water species abundance decreased to 9.5 ± 0.8 species/ m^2, and non-tolerant vegetation decreased to 3.3 ± 0.3 species/ m^2.

Overall, the return of tidal flooding to a subsided agricultural dyke land located near the tidal limit of a macro tidal estuary was driven by a two phase process. To summarize the series of events that occurred over the first two growing seasons following hydrologic restoration a figure created in Appendix C. The return of tidal flooding on a subsided marsh platform, where the area within the marsh lies at a lower elevation than the surrounding fringe marsh, resulted in higher inundation time and an increased sedimentation potential within the marsh than the surrounding fringe marsh. Increased tidal flooding on the site triggered an ecological engineering phase where high sedimentation rates are recorded and resultant decrease in vegetation cover. During the first year of restoration bare ground covered most of the low lying areas of the marsh. The vegetation recovery phase that occurred during the second year of growth showed that

bare ground plots were colonized by brackish and salt marsh vegetation. The high sedimentation rates and subsequent mudflat formation recorded in the first year provided a platform for colonization of brackish and salt marsh vegetation.

Given the overall position of the restoration site within the estuary and the dam restricting tidal flow up river, there is a high potential for saline and fresh water to converge. In 2009, Bowron et al. (2010) showed that more than 14% of the suspended sediment concentration measured from tidal water near the site was within the range of fluid mud, as described by Gaun et al. (1998). The up-estuary propagation of relatively more dense saline tidal flow and a lack of mixing with less dense fresh water can result in a stratified water column positioned in a wedge shape at the extreme turbidity maximum (ETM) (Burchard and Baumert 1998; Uncles 2002). High suspended sediment concentrations are recorded at the tip of the salt wedge (Burchard and Baumert 1998; Uncles 2002). The mixing of fresh water and salt water also creates the salt induced flocculation phenomena, where suspended materials have their ionic strengths modified by the physiochemical properties of water, resulting in increased sedimentation (Burchard and Baumert 1998; Thill et al. 2001). The travel up and down an estuary of the ETM is known to be linked to a spring/neap tide cycle in river flows (Schoelhammer et al. 2000; Doxaran et al. 2009). The potential for the ETM to travel up and down the estuary suggests that the vegetation at the St. Croix site has the potential to experience intermittent affects from the changes of suspended sediment concentrations within the tidal frame.

A biological disturbance, defined by Grime (1979) as the destruction of biomass, opens up space and resources to be utilized by new individuals on a tidal marsh is

typically from biological sources such as herbivory (Rand 2000;) or from physical sources such as salt pulses (Flynn et al. 1995; Howard and Mendelssohn 1999). However, few studies have suggested that sedimentation can be act as disturbance agents within newly restored tidal wetlands. Greenhouse examples have shown that even salt marsh vegetation exhibits a tolerance threshold to sediment accretion, where once a tolerance level is reached species die-back will occur (Maun 1998; Deng et al. 2008). The pulsed nature of tidal flooding at the St. Croix site created a period of high sedimentation that eliminated the vegetation lying on the site prior to restoration. However, due to a lack of a consistent salinity and sedimentation inputs on the site, the vegetation of St. Croix should not be expected to exhibit zoned community pattern (Odum 1988; Crain et al. 2004; Suchrow and Jensen 2010). It would appear that the St. Croix site lies at an ecotone, and due to its location within the zone of ETM, is an intersection zone between the tidal fresh water and salt water marshes which has the potential to create a site with high species biodiversity.

Species co-existence between salt marsh, brackish marsh and fresh water communities will be related to the ability of species recovery and the distribution of bare ground patches within the site at periodic disturbances. One of the models used to explain pulsed disturbances and high biodiversity is the intermediate disturbance hypothesis (IDH). Connell (1978) suggested that within coral reefs and tropical rain forests, the intermediate disturbance had the potential to create high biodiversity. The mechanism suggested is that within a frequently disturbed environment, species with an ability to quickly re-generate themselves (e.g. better dispersers) will be favoured over those better competitors (e.g. poor dispersers). Conversely, where there are infrequent disturbances

low species diversity would be a product of the better competitors excluding those better dispersers. An intermediate disturbance would therefore create a habitat that has patches of ground with different ages of disturbance and carry species in various stages of recovery and colonization where competitively inferior species would occupy the recently disturbed sites (Wilson 1990; Roxburgh et al. 2004). Within oligohaline marshes, disturbance in the form of periods of high salinity salt pulses, can affect vegetation community structure as individual species recovery ability (e.g. resiliency) become important determinants of community structure (Howard and Mendelssohn 1999). In this thesis, however, it was shown that sedimentation likely acted as the main disturbance agent during the first year. In the second year a reduction in sediment elevation increases were recorded and the reduction in intensity coincides with the increase in vegetation cover.

Overall this study demonstrates that it is possible to re-initiate tidal marsh vegetation succession in low saline environments. During the study period there was a positive influence of sedimentation on driving vegetation recovery. This study provided insight into the factors controlling vegetation recovery in the first two growing seasons following restoration. The results from this study and the low-altitude aerial photography techniques presented here can aid ecological management to better understand factors contributing or limiting vegetation recovery within a low saline environment. Although this study examined factors driving recovery there are still questions that warrant further investigation.

1. Examining the relationship between tidal transport of sediments, nutrients within the sediments and the role it has on different vegetation communities would help us understand micro-habitat patches on the site.

2. This project focussed on using RGB imagery to document vegetation changes soon after hydrologic restoration. As the site continues to be covered by vegetation the RGB imagery will continue to be limited. Future studies wanting to examine vegetation dynamics should try to use infrared or near infrared imagery on the site. Vegetation community structure is expected to change with the a continued influence the tidal flooding and having technology to examine marsh wide vegetation response will better inform restoration progress.

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Appendix A: Species abundance over the study period

Species	2010 mean	2010 st dev	2011 mean	2011 st dev
Achillea millefolium	0.32	0.45	0.16	0.17
Agrostis perennans	*	*	0.68	+
Agrostis stolonifera	0.29	0.33	0.85	0.27
Alisima trivale	*	*	0.36	0.37
Alepecurus geniculatus	0.44	0.34	0.66	0.35
Algae	0.24	0.16	*	*
Alopecurus pratens	0.35	0.31	0.49	0.37
Atriplex hastata	*	*	0.30	0.45
Atriplex sp.	0.05	0.06	*	*
Bareground	0.37	0.30	0.34	0.32
Bidens cernua	*	*	0.28	0.34
Bromus inermis	*	*	0.44	+
Calamagrostis canadensis	0.51	0.37	*	*
Carex gynandra	*	*	0.00	+
Carex lenticularis	*	*	0.48	+
Carex lurida	0.04	+	*	*
Carex paleacaea	0.40	0.47	0.61	0.37
Carex scoparia	0.12	0.14	0.20	
Carex striata	0.07	0.05	*	*
Cerastium vulgare	0.20	+	0.24	
Daucus carota	0.04	+	*	*
Dead material	0.55	0.25	0.11	0.05
Echinochloa crus-galli	*	*	0.04	+
Elymus repens	0.21	0.17	0.32	0.39
Elymus trachychalus	*	*	0.24	+
Galium palustre	0.04	0.06	0.06	0.08
Glechoma hederacea	0.24	+	*	*
Glyceria grandis	*	*	0.56	0.40

Species	2010 mean	2010 st dev	2011 mean	2011 st dev
Impatens capensis	0.08	+	*	*
Juncus effusus	0.23	0.27	0.26	0.36
Juncus gerardii	0.15	0.11	0.24	0.28
Jucus breviculatus	0.10	0.08	0.04	0.06
Juncus articulatus	0.24	0.21	0.28	0.19
Juncus bufonius	*	*	0.20	0.28
Lolium perenens	0.99	0.02	0.20	+
Plantago major	0.10	0.08	*	*
Poa pratens	0.52	0.37	0.57	0.37
Phloeum pratens	*	*	0.36	0.00
Polygonum remosium	*	*	0.12	
Rubus sp.	1.00	+	*	*
Rumex crispus	0.04	+	0.04	+
Polygonum hydropiper	*	*	0.17	0.21
Polygonum lapathifolium	0.30	0.31	0.12	0.10
Polygonum neglectum	0.27	0.28	0.27	0.25
Polygonum persicaria	*	*	0.12	0.00
Polygonum sagittatum	*	*	0.16	+
Scirpus americanus	0.08	+	0.16	+
Scirpus atrovirens	0.51	0.31	0.30	0.32
Scirpus validus	0.17	0.17	*	*
Spartina altineflora	0.04	0.06	0.49	0.33
Spartina pectinata	0.16	0.24	0.29	0.22
Stellaria sp.	0.08	+	0.04	+
Symphotrichum novi-belgii	0.10	0.03	*	*
Taraxum officnale	0.15	0.16	0.17	0.20
Taraxum sp.	0.14	0.12	*	*
Trifolium repens	0.06	0.08	0.17	0.17
Typha sp.	*	*	0.35	0.30

Appendix B: Species classification by habitat preference

Salt marsh species	Brackish species	Fresh water species	Upland species
Agrostis stolonifera	Achillea millefolium	Agrostis perenans	Anthemis cotula
Carex paleaceae	Alopecurus geniculatus	Alisima triviale	Bromus inermus
Juncus gerardii	Aster sp.	Alopecurus pratens	Daucus carota
Spartina alterniflora	Atriplex hastata	Bidens cernua	Echinocloa crus-galii
	Atriplex sp.	Calamagrostis canadensis	Geleopsis tetrahit
	Carex scoparia	Carex lenticularis	Gnaphalium uliginosum
	Cerastium vulgare	Carex lurida	Lolium perenens
	Elymus repens	Carex remosa	Rubus allegheniensis
	Elymus trachycaulus	Carex spicata	Taraxum officnale
	Galium palustre	Carex stricta	Taraxum sp.
	Glyceria grandis	Glechoma hederacea	Trifolium repens
	Juncus articulatus	Impatens capensis	
	Juncus bufonius	Jucus breviculatus	
	Plantago major	Juncus effusus	
	Polygonum lapathifolium	Phleum pratense	
	Polygonum neglectum	Poa pratensis	
	Polygonum persicaria	Polygonum hydropiper	
	Polygonum remosissimum	Polygonum sagittatum	
	Rumex crispus		
	Scirpus americanus		
	Scirpus atrovirens		
	Scirpus maritimum		
	Scirpus validus		
	Spartina pectinata		
	Stelleria Sp.		
	Symphotrichum novi-belgii		
	Typha sp.		

Appendix C : Vegetation recovery succession at the St. Croix site

Return of tidal flooding

- **Subsided nature of marsh platform**
  - Higher inundation time within the marsh than on the fringe
  - Incrased sedimentation potential within the marsh than on the fringe marsh

Vegetation die-off and mudflat formation

- **1st year of restoration : Ecological engineering phase**
  - High sedimentation rates within the marsh results in vegetation die-off and subsequent mudflat formation

Mudflat colonization

- **2nd year of restoration : Vegetation response phase**
- Low pore water salinity on the site represents reduced salt stress for vegetation
- Colonization of bare ground by brackish and salt marsh species

Appendix D: Sample stations and their coordinates

(UTM NAD83, Zone 20N)

Station location	Station	Easting	Northing	Pre-elevation	2010 Elevation	2011 Elevation
Marsh	L1S1	418590.33	4980311.48	6.64	6.70	6.75
Marsh	L1S2	418607.78	4980320.87	6.89	7.04	7.05
Dyke	L1S3	418625.35	4980330.28	8.79	8.80	8.80
Marsh	L2S1	418619.64	4980242.92	6.74	6.91	6.94
Marsh	L2S2	418658.30	4980253.38	7.31	7.45	7.45
Fringe	L2S4	418735.53	4980272.79	7.47	7.43	7.52
Marsh	L3S1	418621.85	4980168.51	6.56	6.79	6.84
Forest	L3S3	418705.10	4980193.63	7.87	8.12	8.12
Marsh	L3S4	418735.43	4980204.70	6.88	6.83	6.95
Marsh	L3S5	418773.14	4980218.36	0.00	0.00	0.00
Marsh	L4S2	418593.28	4980095.18	6.86	6.91	6.89
Marsh	L4S3	418628.71	4980077.03	7.05	7.10	7.03
Marsh	L4S4	418666.04	4980062.58	6.38	6.55	6.55
Marsh	L4S5	418703.35	4980048.34	6.71	6.81	6.74
Marsh	L4S7	418777.91	4980019.64	7.23	7.47	7.43
Fringe	L4S8	418805.12	4980010.38	6.00	6.00	6.02
Marsh	L5S1	418459.56	4980039.22	6.71	6.65	6.30
Marsh	L5S2	418498.65	4980030.71	6.22	6.25	6.34
Marsh	L5S3	418537.93	4980022.22	6.26	6.32	6.44
Marsh	L5S4	418576.81	4980013.55	6.53	6.59	6.66
Marsh	L5S5	418615.81	4980005.11	6.65	6.65	6.78
Marsh	L5S6	418653.71	4979991.70	6.36	6.61	6.54
Marsh	L5S8	418732.79	4979979.11	6.83	7.05	7.02
Dyke	L5S9	418771.14	4979970.11	8.06	8.06	8.05
Marsh	N01	418590.51	4980121.50	6.61	6.71	6.71
Marsh	N02	418664.29	4980290.43	6.92	7.11	7.12
Marsh	N03	418747.68	4980144.99	6.81	6.90	7.00
Marsh	N04	418739.18	4980248.46	7.10	7.17	7.17
Fringe	N05	418796.98	4980022.18	6.65	6.76	6.76
Fringe	Y06	418758.11	4979935.42	7.30	7.45	7.52
Marsh	Y07	418721.41	4979934.52	6.18	6.67	6.67
Marsh	Y08	418680.52	4979951.91	6.62	6.86	6.85
Marsh	Y09	418641.64	4979961.15	6.48	6.68	6.68
Marsh	Y10	418521.47	4980001.14	6.42	6.54	6.65
Marsh	Y11	418466.17	4980007.54	6.39	6.47	6.46
Marsh	Y13	418573.20	4980041.03	6.62	6.60	6.55
Marsh	Y14_Y02	418510.91	4980083.39	6.24	6.36	6.36
Marsh	Y15	418529.92	4980107.88	6.22	6.33	6.35
Marsh	Y16	418637.91	4980137.75	6.79	6.81	6.86
Marsh	Y17	418628.15	4980211.40	6.79	6.84	6.84
Fringe	Y18	418600.88	4980368.34	7.47	7.62	7.62
Fringe	Y19	418802.16	4980187.41	7.38	7.33	7.42
Fringe	Y20	418809.94	4980150.01	7.44	7.41	7.45
Marsh	Y21	418645.31	4980025.74	6.42	6.63	6.62
Marsh	Y22	418668.57	4980020.61	6.49	6.66	6.66
Marsh	Y23	418720.93	4980142.11	6.77	6.81	6.83
Marsh	Y24	418741.40	4980155.15	6.77	6.86	6.94

Appendix E: Letter of Permission

CBWES Inc.

www.cbwes.com
November 1, 2012

To: Ben Lemieux
         MSc Candidate Saint Mary's University
         Halifax, Nova Scotia

Re: Letter of Permission

Dear Mr. Lemieux,

With this letter CBWES Inc. grants you permission to use material (written, figures, or tables) from the *St. Croix River High Salt Marsh and Floodplain Wetland Restoration Project* Reports (2008 – 2011), prepared for the Nova Scotia Department of Transportation and Infrastructure Renewal to complete your thesis for the Degree of Master of Science in Applied Science.

Take care.

Sincerely,

Tony M. Bowron
Director, Coastal Wetland Ecologist
CBWES Inc.