Sub-surface hydrology and vegetation drivers at macrotidal Bay of Fundy salt marshes: Implications for future restoration

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ABSTRACT

There is a growing interest to restore Bay of Fundy salt marshes diked for agriculture. Marshes recovering for several decades from storm-breached dikes can serve as analogues for restored marshes. In this study I examine factors driving sub-surface hydrology and vegetation at recovering and reference Bay of Fundy salt marshes. In Fundy marshes, groundwater at channel edges is insensitive to tidal flooding (< 10 cm change in depth) and deep draw-downs (40-100 cm) occur. Sub-surface hydrology here differs from organogenic, microtidal marshes due to low saturated hydraulic conductivity, infrequent flooding of marsh interiors, and larger hydraulic gradients imposed at channel edges. By calculating marsh elevation at dike-breach and considering *Spartina alterniflora*'s vertical range, it is apparent that salt marsh vegetation could establish when dikes breached. Multivariate analysis indicates that reference and restored/recovering sites should have similar sizes and tidal ranges. These criteria introduce problems as Fundy dikelands are more extensive than marshes not targeted for agriculture and tidal range increases exponentially up-Bay.

RÉSUMÉ

Les marais salés de la Baie de Fundy furent endigués pour l'agriculture et il existe maintenant un désir de les restaurer. En effet, les marais salés récupérant depuis plusieurs décennies suite à la destruction de digues agricoles, peuvent maintenant être considérés comme des marais subissant une restauration écologique. Dans le cadre de cette étude, j'examine les facteurs pouvant affectés l'hydrologie souterraine ainsi que la végétation des marais en restauration et des marais de références. À l'intérieur des marais de la Baie de Fundy, l'eau souterraine aux abords des canaux est insensible au changement des marées (< 10 cm de changement en profondeur) et des rabattements profonds de la nappe phréatique se produisent (40-100 cm). À cet endroit, l'hydrologie souterraine diffère des marais organogénique possédant des micro-marée car ils possèdent une conductivité hydraulique saturée très basse, une inondation du marais intérieur peu fréquente, et une plus grande gradient hydraulique près des bordures des canaux. En calculant l'élévation du marais là où la rupture de la digue s'est produite ainsi qu'en considérant la distribution verticale de *Spartina alterniflora*, il est clair que la végétation caractéristique des marais salés peut s'établir lors du bris d'une digue. De plus, les analyses multi-variées indiquent que les sites de référence et les sites restaurés doivent couvrir la même dimension ainsi que posséder une variation des marées similaires. Ces critères introduisent des problèmes car les terres drainées de la Baie de Fundy sont beaucoup plus étendues que les marais non ciblés pour l'agriculture mais aussi car la variation des marées augmente exponentiellement vers l'intérieur de la Baie.

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"For those of us who do not do science, science often seems to be the last bastion of unfuzzy logic, a place where answers are clear-cut, a moral universe where there is a right and a wrong. But we fool ourselves – it's not like that at all. Science is ruled by human passions and limitations and creativity. Science is the story we tell ourselves, or are told, to make sense of the world of atoms and cells, illness and beauty, ozone and oxygen, the world in which we – collections of atoms and cells – find ourselves."

Sue Halpern
Four Wings and a Prayer –
Caught in the Mystery of the Monarch Butterfly

It has been extremely fulfilling to turn my passion for wetlands into masters research, an important part of my science story. Like any good story it involves many people who helped shape it and who therefore deserve my heartfelt appreciation.

My fascination with the Bay of Fundy began as an undergraduate when Gail Chmura invited me to help conduct fieldwork at several salt marshes. Thanks Gail for not only introducing me to Fundy but also as my masters supervisor for providing feedback on research design and commenting on numerous drafts. As well, this research would not have been possible without the help of several people in the field - Anne Sabourin, Catherine Boleyn, and Grace Hung. A special thanks to Graham MacDonald who collected and processed much of my DGPS data as well as competently and patiently answered my countless GIS questions. Thank you all for maintaining a sense of humor while braving swarms of mosquitoes and torrential rain to gather accurate data.

Thanks to the Canadian Wildlife Service for access to John Lusby marsh and to staff of the Coastal Wetlands Institute (Mount Allison University) who graciously allowed us to use their facility to process plant samples. The research reported in this thesis was financially supported by my supervisor, Dr. Gail Chmura, a McGill Frazee Scholarship (Huntsman Marine Science Centre), and a summer NSERC awarded to my field assistant Anne Sabourin.

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My attraction to the Bay of Fundy is an extension of my deep appreciation of plants, animals, and water dynamics of aquatic environments that I gained during my childhood. Thanks mom and dad for giving me numerous opportunities to canoe Louisiana bayous, sail along the Gulf Coast, tube along riparian wetlands, and explore Florida beaches. These experiences first sparked my passion in wetlands and aquatic environments. I cannot thank you enough for assisting me in nearly every science project since middle school, including this one in which you washed and dried 72 plant samples over the course of three days.

During my freshman year of high school Dawn Kalb recognized my interest in science and invited me to enroll in an upper-level water ecology course. I will always fondly remember my first formal introduction to watery realms. Thanks also to my earth science teacher, Emily Taylor, for amazing field trips to Tunica Hills – illustrating that the natural environment has much to teach us. Thank you both for continuing to be enthusiastic and supportive mentors over the years.

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Finally, I sincerely hope that the story told by this thesis helps to make sense of the world by both enhancing our knowledge of salt marshes and ensuring that restoration efforts are directed in a meaningful way.

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Chapter 1: Introduction

Tidal salt marshes, saline wetlands regularly flooded by tidal waters, are gently sloping platforms of sediment that are colonized by halophytic (salt-tolerant) vegetation and typically dissected by a tidal channel network (Allen 2000). These marshes form in the upper inter-tidal zone, usually between mean highest high water and mean lowest low water, in areas sheltered from high wave energy. Salt marshes can be found along the coasts of most continents from the subtropical to arctic latitudes (Chapman 1974).

Human activities in coastal areas during the past several centuries have resulted in the alteration or loss of thousands of square kilometres of tidal salt marsh (e.g., Beeftink 1975; Dale and Hulsman 1990; Allen 2000; Kennish 2001). Salt marshes continue to face a variety of pressures due to increasing urbanization in coastal areas, increasing traffic through global ports, and predicted sea level rise due to greenhouse warming (Crooks and Turner 1999). Though the Bay of Fundy has a relatively low population (e.g., compared the Atlantic coast of the U.S.), only a few large urban areas, and low level of industrialization, tidal salt marshes have nonetheless been impacted by human activities. In parts of the Bay of Fundy, up to 85% of salt marshes have been converted into agricultural land via dikes and ditches since the 17^{th} century (Ganong 1903). More recently, structures such as causeways, dams, and improperly sized culverts that pass under roads have reduced or completely prevented tidal flow in Bay of Fundy salt marshes (Gulf of Maine Council Habitat Restoration Subcommittee 2004).

There is a growing interest to restore Bay of Fundy salt marshes (Gulf of Maine Council Habitat Restoration Subcommittee 2004). The economically valuable ecosystem services that salt marshes provide (Costanza et al. 1997) can help bolster restoration efforts. For example, Bay of Fundy salt marshes sequester carbon at rates equivalent to or higher than other ecosystems and in contrast to other ecosystems they do not appear to become saturated over century time scales (Conner et al. 2001). Fundy marsh sediments also serve as a sink for a variety of heavy metal contaminants (Hung and Chmura 2006), including mercury, which is of particular concern in this region (Percy 2004).

Salt marsh vegetation and hydrology

Since tidal salt marshes are regularly flooded by tidal water, there is a quite obvious need to understand surface hydrology and how it influences marsh ecology. Zonation of salt marsh vegetation is broadly due to surface elevation with respect to local tidal levels (e.g., Miller and Egler 1950; Redfield 1972; Chapman 1974). In general, a location lower in the tidal frame has a higher hydroperiod (i.e., frequency and duration of tidal flooding) and supports species capable of living in a high stress environment. In the Bay of Fundy, *Spartina alterniflora* dominates the regularly flooded low marsh zone while *Spartina patens* dominates the less flooded high marsh zone (Ganong 1903; Chmura et al. 1997). In the lower Bay, a middle marsh zone of *Plantago maritima* also occurs (Chmura et al. 1997)

The ebb and flood of tides is also an important geomorphological force in terms of both sediment delivery and erosion. Marsh vegetation traps mineral sediment and also contributes organic matter in the form of above- and below-ground production (Friedrichs and Perry 2001). Measuring sediment accretion rates in both the vertical and horizontal plane (Chmura et al. 2001; Davidson-Arnott 2002; van Proosdij et al. 2006) allows researchers to determine whether marshes can keep pace with sea level rise as well as erosive forces at the seaward edge. Research indicates that lower Bay marshes are in step with recent sea level change and are not at risk of being submerged (Chmura et al. 2001).

Allen Creek, an upper Bay marsh directly fronting the open Bay, currently has a positive sediment balance despite the fact it looses significant amounts of sediment through erosion of the marsh margin cliffs (van Proosdij et al. 2006).

Less obvious and less well studied, but of equal importance to marsh ecology, is salt marsh sub-surface hydrology. In addition to elevation relative to tidal levels, more proximate variables affect marsh vegetation. For example sediment saturation and drainage, and thus sub-surface hydrology, affects salt marsh vegetation productivity, zonation, and survival (Mendelsson and Seneca 1980; Howes et al. 1981; Weigert et al. 1983; Mendelsson and McKee 1988). Generally, the productivity of *S. alterniflora* is highest in well drained areas along creeks and is lower in the marsh interior. Prolonged waterlogging of *S. alterniflora* causes die-back. In addition to water table height, concentrations of nutrients and oxygen also influence productivity, which are controlled in part by the rate at which water moves through marsh sediments (Mendelsson and Seneca 1980; Howes et al 1981; Dacey and Howes 1984; Agosta 1985; Howes and Goehringer 1994). Movement of water through channel banks also represents an important pathway for the transfer of below-ground carbon and nutrient pools to tidal water (Howes and Goehringer 1994). Not only is the movement of water at the channel edge important, but also at the upland edge. Thibodeau et al. (1998) showed that underground flow from the upland is an important source of water for areas of salt marsh within 30 m of forested upland since it consists of freshwater that lowers sediment salinity and allows less salt-tolerant species to persist.

Marsh sub-surface hydrology is often conceptualized in terms of a water budget. Water is added to marsh sediment via tidal inundation of the marsh platform and channel banks, precipitation, underground flow from the upland, and upward movement of the regional aquifer (Nuttle 1988). Water is removed from marsh sediment via evapotranspiration, seepage into tidal channels, and downward movement of the regional aquifer (Nuttle 1988). The movement of water within marsh sediments is governed by sediment characteristics and tidal channel morphology. Grain size, % organic matter, and the presence of macropores all influence sediment infiltration (i.e, the rate at which water enters sediment) and hydraulic conductivity (i.e., the rate of water flow through sediment) (Fitts 2002). Channel morphology, including depth, shape, and presence of levees, affects the amount of drainage that occurs (Nuttle 1988; Howes and Goehinger 1994).

The ability of salt marshes to persist and continue to provide valued ecosystem services is intimately linked to sub-surface hydrology. Despite its importance, only a handful of studies have focused specifically on this process. Of the published studies, most have been conducted in microtidal, organogenic salt marshes located along the Atlantic coast of the U.S. Additional studies have been conducted in Australia, the United Kingdom, and Canada. Howes et al. (1981), Hemmond et al. (1984), and Howes and Goehringer (1994) studied the sub-surface hydrology of a Massachusetts salt marsh from a variety of perspectives including surface infiltration and export of carbon and nutrients. Agosta (1985), Jordon and Correll (1985), and Yelverton and Hackney (1986) report water table levels in a South Carolina marsh, Maryland marsh, and North Carolina marsh, respectively, for studies focused on pore water chemistry. Harvey et al. (1987) examine several geomorphological and sediment properties that influence sub-surface hydrology. Balling and Resh (1983) report the effects of mosquito control ditches on water table levels. The most comprehensive studies of marsh sub-surface hydrology include those by Nuttle (1988) in a Massachusetts marsh, Hughes et al. (1998) in an Australian marsh, and Montalto et al. (2006) in a New York marsh. Two previous studies have examined sub-surface hydrology in macrotidal salt marshes – Price and Woo (1988) examined sub-surface hydrology of an undisturbed marsh in James Bay, Canada while Blackwell et al. (2004) examined sub-surface hydrology response in a recently restored UK salt marsh.

Results from these studies provide important insight into marsh sub-surface hydrology. During periods of non-inundating neap tides there is a lowering of water table and hydraulic head (i.e, the energy available for water to flow), while inundating spring tides raise water table and hydraulic head (Nuttle 1988; Montalto et al. 2006). Even so, water is typically within 40 cm of the marsh surface. Recharge of the water table/hydraulic head near the channel edge occurs within 45 minutes of flooding (Hemmond et al. 1984; Agosta 1985; Hughes et al. 1998; Montalto et al. 2006). Precipitation events raise water table and hydraulic head by several tens of centimeters (Nuttle 1988; Hughes et al. 1998; Montalto et al. 2006). Water table/hydraulic head response to tidal fluctuations have been reported to occur at distances of 2.5 m (Nuttle 1988), 4 m (Hemond et al. 1984), and 12 to 18 m (Montalto et al. 2006) from tidal channels. The influence of tides on sub-surface hydrology is attenuated within 15 m (Nuttle 1988) to 25 m from tidal channels (Montalto et al. 2006).

Research on sub-surface hydrology in macrotidal, minerogenic Bay of Fundy salt marshes is sparse. Based on limited observations as well as results from sediment samples, Ganong (1903) concluded that water table movement is very restricted in upper Bay marshes. More recently, water-table and groundwater salinity have been measured at several Fundy salt marshes, but the data are confounded since problems exist with the water-table well sampling methodology used (CB Wetlands & Environmental Specialists 2006a and 2006b). In comparison, more is know about Fundy marsh vegetation -

numerous vegetation studies of undisturbed Fundy marshes include Hatcher and Mann (1975), Palmer (1979), Smith et al. (1980), Gordon et al. (1985), Chmura et al. (1997), and Olsen et al. (2005). There have been several vegetation studies of John Lusby, a recovering marsh on the Bay of Fundy (Van Zoost 1970; Morantz 1976; Gordon et al. 1985). From the above discussion it is clear that a complex set of interconnected variables influence zonation and productivity of marsh vegetation. Though studies of Fundy vegetation include several variables, none of the authors consider a large suite of environmental variables. Further, no implicit comparisons between vegetation of undisturbed reference marshes and those recovering from dike breaching are made.

Salt marsh restoration and recovery

As a result of dike and ditch construction, several important changes take place in former salt marshes (i.e., dikelands). Tidally derived sediments are prevented from reaching dikelands and contributing to vertical accretion. If ditches provide sufficient drainage and lower the water table, sediments compact due to dewatering and increased rates of decomposition (Roman et al. 1984; Allen 1999; Crooks and Pye 2000; Weinstein and Weishar 2002). Over time, woody species and other terrestrial vegetation establish, out compete, and replace salt marsh vegetation in drained dikelands (Ganong 1903; Portnoy et al. 1987).

Impacts due to dikes may be lessened or eliminated when dikes are breached or removed entirely. A breach, or gap, in the dike can be purposefully created in locations where humans wish to restore salt marshes (Blott and Pye 2004). Unintentional dike breaches can occur during storm events and marshes recover with little or no human influence (Crooks and Pye 2000; Crooks et al. 2002; Eertman et al. 2002; Callaway 2005; French 2006).

Ecological "trajectories" have been reported in salt marsh restoration literature (e.g., Callaway 2005) and describe the developmental pathway that begins with an unrestored ecosystem and progresses, over time and within certain bounds, towards a desired state of restoration (Society for Ecological Restoration International Science and Policy Working Group 2004). Studies of sub-surface hydrology trajectories are generally lacking. However, a short-term (4 mo) study of a UK dikeland demonstrated that water table height increased dramatically within the first five flooding events post-breach (Blackwell et al. 2004). Burdick et al. (1997) demonstrated that water table in a New Hampshire marsh was significantly closer to the marsh surface 1 to 2 yr following restoration and post-restoration water table data were similar to a reference site. In a Rhode Island salt marsh, water table depth was also significantly different before and 2 yr after restoration (Roman et al. 2002).

Vegetation trajectories have been studied in various non-constructed U.S. and European marshes. Just 1 yr after restoration of a Rhode Island marsh, there were significant changes in vegetation abundance and dissimilarity measures suggest a convergence towards typical salt marsh vegetation (Roman et al. 2002). Five yr post-recovery, Eertman et al. (2002) found that large areas of mudflat were covered with marsh vegetation at a Dutch site. Sinicrope et al. (1990) demonstrate that a Connecticut marsh contains typical salt marsh vegetation 10 yr post-restoration. Thom et al. (2002) found that during 11 yr post- dike breach low marsh vegetation has become established in a Washington marsh and similarity with a reference marsh has increased over time. Williams and Orr (2002) show that in San Francisco Bay, a vegetated marsh platform (defined as 50% cover) is achieved within less than 5 yr to more than 20 yr, depending on site conditions. In rapidly restored Connecticut salt marshes, salt marsh vegetation became established at rates of 5% of total area per yr, suggesting that restoration there takes approximately 20 yr (Warren et al. 2002). Crooks et al. (2002) found that vegetation of recovering sites in the UK over 100 yr old matched that of reference sites, concluding that recovery can be complete in less than a century.

Currently, on the Bay of Fundy, there exist two small dikelands – Musquash (15.38) and Walton River (4.95 ha) – where tidal action was returned in the past year (Ducks Unlimited Canada 2004; CB Wetlands & Environmental Specialists 2006a). Short-term data for the1-2 yr following dike breaching are currently being collected and analyzed and are yet to be published. At one of the sites there are plans to monitor up to five years post dike breach (CB Wetlands & Environmental Specialists 2006). However, investigations of large sites recovering for several decades, including marshes in this study, can 1) provide insight into large-scale, system-wide marsh response and 2) offer long-term information about vegetation, geomorphology, and recruitment of fauna. This study represents a start towards developing restoration trajectories for Bay of Fundy salt marshes and understanding salt marsh processes in order to guide future restoration plans at other sites.

Research strategy and thesis structure

In order to examine the difference in sub-surface hydrology and vegetation between reference and recovering salt marshes as well as provide insight into which factors drive these two salt marsh components a variety of techniques including mapping and sampling groundwater, sediment, and vegetation were used. Reference (undiked) and recovering (storm-breached dike) marsh pairs were chosen in both the lower and upper Bay of Fundy. Dipper Harbour (10.8 ha) is relatively undisturbed and serves as the lower Bay reference site. Saint's Rest (94.7 ha) was diked sometime between 1786 and 1864 and is the recovering site in the lower bay as tidal flooding was returned when the dike breached in the 1950's (Noel et al. 2005). In the upper Bay, Wood Point (16.9 ha), also referred to as Allen Creek, was not diked and serves as the reference site. John Lusby (600 ha) is also located in the upper Bay and was diked sometime between 1686 and 1693 (Clark 1968). It has been recovering since the dikes failed in 1947 (Graf 2004).

Chapter 2 of this thesis examines the sub-surface hydrology both within and between marshes. I believe this is the first study to compare sub-surface hydrology of recovering marshes to reference marshes several decades post dike breach. Furthermore, sub-surface hydrology has mainly been studied in organogenic, microtidal marshes. GPS/GIS mapping, salinity, and sediment results are presented in order to determine which environmental factors drive sub-surface hydrology in minerogenic, macrotidal Bay of Fundy marshes. In Chapter 3 I investigate the relationships among vegetation species and numerous environmental variables in these same marsh pairs. First, I determine the relationships among salt marsh vegetation species, surface elevation (relative to mean sea level and tide levels), and predicted frequency of tidal inundation. Then, I use multivariate statistics to determine how salt marsh vegetation percent cover and productivity respond to numerous environmental variables (i.e., distance to tidal channel, depth to sub-surface water, groundwater salinity, sediment bulk density, etc.). Results from this investigation allow me to assess the progress of vegetation recovery in two Fundy salt marshes and provide insight into the potential response of salt marshes to intentional restoration efforts elsewhere in the Bay. Both of these chapters will be submitted as separate journal articles to *Estuaries and Coasts* and are formatted accordingly, thus some repetition occurs.

Chapter 2: Studies of recovering and reference salt marshes on the Bay of Fundy, I. Sub-surface hydrology

Abstract

Soil saturation, and thus depth to groundwater, is important to salt marsh vegetation as it drives zonation, productivity, and survival. Therefore, baseline knowledge of Fundy salt marsh sub-surface hydrology is crucial in understanding changes associated with long-term projected sea level rise and abrupt return of tidal flooding resulting from dike breaches and aboiteau failure. In this study I examine drivers of sub-surface hydrology in minerogenic, macrotidal marshes and compare two pairs of reference and recovering marshes. The sub-surface hydrology of these macrotidal marshes is not greatly influenced by tidal height. Hydraulic head is displaced ~10 cm or less at inundated channel edges, a small displacement compared to microtidal marshes. Low saturated hydraulic conductivity, infrequent flooding of marsh interiors, large hydraulic gradients imposed at steep channel edges, and seasonal differences in precipitation and evapotranspiration drive sub-surface hydrology variability in Fundy marshes. Maximum groundwater depth at channel edges (60-100 cm) and the marsh interior (40-45 cm) are larger than reported at microtidal marshes. Previous research on sub-surface hydrology that has mostly been conducted in organogenic, microtidal marshes does not provide suitable models for minerogranic, macrotidal marshes. Results indicate that that sub-surface hydrology was restored at the two recovering sites. I conclude that Fundy marshes will be more ecologically resistant to rising sea level associated with greenhouse warming, as changes in tidal heights will have minimal impact on sub-surface hydrology, thus vegetation will be relatively stable. Since many Fundy marshes were drained for agricultural use, restoration of these resistant marshes could offset losses expected in more vulnerable, microtidal marshes.

Introduction

By definition tidal salt marshes are regularly flooded by tidal water. Therefore, there is a quite obvious need to understand surface hydrology and how it influences salt marsh ecology. Less obvious and less well studied, but of equal importance, is salt marsh sub-surface hydrology. Sediment saturation, and thus sub-surface hydrology, affects salt marsh vegetation productivity, zonation, and survival (Mendelsson and Seneca 1980; Howes et al. 1981; Weigert et al. 1983; Mendelsson and McKee 1988). Sub-surface hydrology influences rates of subsidence (Turner 2004); concentrations of nutrients, organic matter, and oxygen (Howes et al. 1981; Agosta 1985; Howes and Goehringer 1994); and sediment toxicity (Portnoy and Valiela 1997; Portnoy 1999). Therefore, the ability of salt marshes to persist and continue to provide valued ecosystem services - including sequestering carbon (Chmura et al. 2003) and metal contaminants (e.g., Hung and Chmura 2006) and providing habitat for various species (e.g., Weinstein and Kreeger 2000) - is intimately linked to sub-surface hydrology. As a result, baseline knowledge of salt marsh sub-surface hydrology is crucial in predicting and understanding change in salt marsh environments.

In the Bay of Fundy hydrological changes include both long-term projected sea level rise associated with greenhouse warming and sudden changes resulting from dike breaches and tide gate (locally known as aboiteau) failure. Since relative sea level in the Bay of Fundy is rising (Chmura et al. 2001), the need to understand how marsh water-tables will respond to increased tidal height is a priority (Mullally 2003). Also, there is a growing interest to restore Bay of Fundy salt marshes that were converted into agricultural land via a system of dikes (i.e., engineered earthen embankments), tide gates and ditches (Harvey 2000; Gulf of Maine Council Habitat Restoration Subcommittee 2004). Salt marshes located behind storm breached dikes represent several decades of recovery and can serve as an analogue for long-term sub-surface hydrological response of salt marshes to restoration efforts.

Marsh sub-surface hydrology is often conceptualized in terms of a marsh water budget. Groundwater is added to marsh sediment via tidal inundation of the marsh platform and channel banks, precipitation, underground flow from the upland, and upward movement of the regional aquifer (Nuttle 1988). The frequency and duration of tidal inundation are determined by the elevation of the marsh surface relative to local tide heights. Groundwater is removed from marsh sediment via evapotranspiration (ET), drainage into tidal channels, and downward movement of the regional aquifer (Nuttle 1988). During periods of non-inundating neap tides there is a lowering of water-table and hydraulic head (i.e., the energy available for water to flow determined by elevation, pressure, and velocity), while inundating spring tides raise water-table and hydraulic head (Nuttle 1988; Montalto et al. 2006). Thibodeau et al. (1998) showed that underground flow was an important source of water for areas of salt marsh within 30 m of forested upland. Importantly, this underground flow consists of freshwater that lowers the salinity of groundwater and allows less salt-tolerant species to persist (Thibodeau et al. 1998). The ratio of upland border to marsh area appears to be an important factor in governing the amount of underground flow a marsh receives; however, this issue has not yet been addressed in the literature.

Once water has been delivered to a marsh, its movement and eventual loss are governed by marsh sediment characteristics, evapotranspiration, and geomorphology. Grain size, % organic matter, and the presence of macropores all influence sediment infiltration (i.e., the rate at which water enters sediment) and hydraulic conductivity (i.e., the rate of water flow through sediment). Thus far, the majority of sub-surface hydrology studies have been conducted in organogenic marshes (Howes et al. 1981; Yelverton and Hackney 1986, Harvey et al. 1987, Nuttle 1988, Montalto et al. 2006) with relatively large pore sizes which allow for higher rates of infiltration and hydraulic conductivity (Fitts 2002). Some of these marshes also contain macropores due to root channels (Agosta 1985), crab (*Uca* spp.) burrows (Agosta 1985; Montalto et al. 2006), and muskrat

(*Ondatra zibethicus*) tunnels (Montalto et al. 2006) that allow for a rapid response of water-table and hydraulic head to tidal inundation.

Evapotranspiration is the dominant mechanism for water loss in sediments of the marsh interior, greater than 10 m from tidal channels (Nuttle 1988). However, within 10 m of the channel bank, drainage accounts for 40% of the water lost and decreases exponentially with distance from the channel (Nuttle 1988). The morphology of tidal channels affects the amount of drainage that occurs. If levees are present adjacent to tidal channels, such as in some Fundy marshes, they will cause a mound in the water-table (Nuttle 1988). This mound will cause groundwater to flow away from the channel and into the marsh interior, restricting the region from which groundwater drains into the tidal channel. Nearly vertical channel banks, such as cliffs at the bayward edge of some Fundy marshes, represent a fixed drainage boundary, while the drainage boundary of a sloping channel bank, such as V-shaped primary tidal channels, will shift as tides rise and fall (Nuttle 1988). Whether this boundary is stable or shifts should affect the amount of drainage occurring; though this issue has not been thoroughly discussed in the literature. In terms of channel depth, Howes and Goehringer (1994) found that a greater volume of groundwater drained along deep tidal channels due to the steep hydraulic gradient imposed at the channel boundary. How much of the channel is exposed and free draining during high tide (Crooks et al. 2002) should also affect the hydraulic gradient present at the channel boundary and thus drainage rates of water. For example, a creek located in John Lusby marsh on the Bay of Fundy contained water 5 to 100 cm deep during low tides (Morantz 1976). Considering that the creek is 4.76 m deep at this location (Morantz 1976), the effective depth of the channel is 3.76 to 4.71 m during low tide, maintaining a large hydraulic gradient compared to tidal channels of microtidal marshes which can remain sub-tidal during low tide (Barwis 1978).

Changes in sub-surface hydrology of macrotidal systems (tidal range > 4 m) might be expected to be dominated by the effects of tides. However, I hypothesize that due to the minerogenic nature of marsh sediment (Connor et al. 2001, Byers 2006) these marshes will have low hydraulic conductivity and infiltration rates making depth to groundwater insensitive to tidal range and tidal flooding. Yet another reason to not expect rapid response of marsh sub-surface hydrology to tidal height is the fact that high densities of large macropores are not present. Fiddler crabs (*Uca pugnax*) are present in salt marshes along the Atlantic coast of the U.S. and create burrows thereby increasing infiltration rates in areas where they are present (Bertness 1985). However, *U. pugnax* is not found in the Bay of Fundy - but clam worms (*Hediste diversicolor*), clams (*Macoma balthica*), mussles (*Mytilus edulis*), and green crabs (*Carcinus maenas*) occasionally occur in areas flooded daily (Daiber 1982).

There are only a handful of studies documenting the sub-surface hydrology response at recovering or restored salt marshes. A short-term (4 mo) study of a UK dikeland demonstrated that a ~30 cm change in depth to groundwater occurred within the first five flooding events post-breach (Blackwell et al. 2004). Burdick et al. (1997) demonstrated that water-table in a New Hampshire marsh was significantly closer to the marsh surface 1 to 2 yr following restoration and post-restoration water-table data were similar to a reference site. In a Rhode Island salt marsh, water-table depth was also significantly different before and 2 yr after restoration (Roman et al. 2002).

To my knowledge there have been no studies comparing the sub-surface hydrology of long breached (i.e., several decades or more) and un-diked reference marshes. However, there is reason to suspect that differences in sub-surface hydrology between these marsh types could exist. For example, consolidation of the original marsh sediments during cultivation combined with a deficiency in calcium carbonate led to the formation of an aquaclude (i.e., barrier to water movement) in recovering UK salt marshes (Crooks and Pye 2000). Though sub-surface hydrology was not measured, Crooks and Pye (2000) hypothesized that an aquaclude would result in higher water-tables at these recovering sites.

Thus far, surface hydrology has been studied in three Bay of Fundy marshes (Morantz 1976; Ayles and Lapointe 1996; Davidson-Arnott et al. 2002), but published data on the sub-surface hydrology of Fundy marshes are generally lacking. Based on limited observations as well as results from sediment samples, Ganong (1903) concluded that water-table movement is very restricted in upper Bay of Fundy salt marshes. More recently, water-table and groundwater salinity have been measured at several Fundy salt marshes, but the data are confounded since problems exist with the water-table well sampling methodology used (CB Wetlands & Environmental Specialists 2006a and 2006b). The present study adds much needed insight by determining 1) which factors drive sub-surface hydrology in salt marshes situated in the minerogenic, macrotidal Bay of Fundy and 2) whether differences in sub-surface exist between reference and long recovering marshes.

Methods

Study Area

The Bay of Fundy (Fig. 1), situated between New Brunswick and Nova Scotia, is renowned for its large tidal range. Tidal range during spring tides increases from approximately 5 m at the mouth to greater than 16 m in the upper reaches (Canadian Hydrographic Survey 2005). This large tidal range is one of the reasons that Fundy waters have a high suspended sediment concentration, ranging from 0.2 to 30.4 mg l^{-1} as one moves up-Bay (Miller 1966). Tides are semidiurnal with relatively small differences (~0.5 m) between the two tides (Canadian Hydrographic Survey 2005). The 18.6 yr lunar cycle also affects tidal heights in the Bay of Fundy, with the next extreme highest tides predicted to occur in 2014 (O'Reilly et al. 2005). This study was conducted during a low point in the lunar cycle.

I sampled reference (un-diked) and recovering (breached dike) marsh pairs in both the lower and upper Bay (Fig. 1) in order to reflect the gradient in tidal range. At Dipper Harbour (10.8 ha), the lower Bay reference site, the primary tidal channel has nearly vertical banks, is 5 to 24 m wide, and is ~1.5 m deep. Secondary tidal channels are ~1 m wide. Dipper Harbour's low marsh is vegetated by *Spartina alterniflora* while the high marsh is dominated by *Spartina patens* (Chmura et al. 1997). Saint's Rest (94.7 ha) was diked sometime between 1786 and 1864 and is the recovering site in the lower Bay. Tidal flooding was returned when the dike breached in the 1950s (Noel et al. 2005). The primary tidal channel is 98 to 34 m wide, >4 m deep, and is bordered by a levee in some locations. Secondary tidal channels are narrower (6 to 15 m) and have nearly vertical banks. Saint's Rest is characterized by low marsh vegetation consisting mainly of *Spartina alterniflora*. A sewage treatment plant is located at the head of the primary tidal channel at Saint's Rest marsh and contributes substantial amounts of freshwater inflow.

Both sites occur in protected coastal settings. However, Dipper Harbour is situated in a narrow-valley setting while Saints Rest is a back-barrier marsh. The tidal ranges of Dipper Harbour and Saint's Rest are 6 m and 6.7 m, respectively (Canadian Hydrographic Service 2005).

Wood Point (16.9 ha) and John Lusby (600 ha) are the two upper Bay sites situated in the Cumberland Basin and have tidal ranges of 10.5 m and 11 m, respectively (see chapter 2 calculations). They are exposed to a ~5 km fetch on the Bay and their bayward edge consists of a near vertical cliff 1 to 2 m high. John Lusby was diked sometime between 1686 and 1693 (Clarke 1968) and the dikes failed in 1947 (Graf 2004). The primary tidal channels located in the John Lusby study area are 46 to 100 m wide, bordered by a levee in some locations, and are over 5 m deep. Secondary tidal channels are narrower (5 to 10 m). The marsh platform at John Lusby is vegetated by *Puccinellia americana* and *Spartina patens* (Morantz 1976). *S. alterniflora* dominates creekbank vegetation and in areas with standing water (Morantz 1976). Wood Point, also referred to as Allen Creek, was not diked and serves as the reference site. Its primary tidal channels are 15 to 20 m wide and 3 m deep. The marsh platform at Wood Point is primarily vegetated by *S. alterniflora*. The high marsh is dominated by *S. patens*.

In both pairs, the reference marsh is smaller than the recovering marsh. This discrepancy was controlled for in the following ways. For Saint's Rest, a study area of similar length and distance from the Bay was chosen for comparison with Dipper Harbour. For John Lusby, I examined an area near the bayward edge of similar width to the Wood Point marsh.

Sub-surface hydrology and piezometer transects

I used piezometers to measure hydraulic head and determine depth to sub-surface water. In the scientific literature, the term "water-table" is broadly used (e.g., Hughes et al. 1998; Blackwell et al. 2004) regardless of whether sub-surface hydrology is measured with a well, a pipe perforated along the entire length below the marsh surface, or a piezometer, a pipe perforated only at the bottom. While wells measure hydraulic head at the water-table, piezometers provide hydraulic head at the point of measurement (i.e., midpoint of the perforated section) (Schwartz and Zhang 2003). Due to vertical gradients in hydraulic head, hydraulic head measured with a piezometer may differ slightly from hydraulic head measured with a well. For clarity, I use the term "depth to groundwater" to refer to measurements made with a piezometer and reserve the word "water-table" for measurements made with wells.

Piezometers were constructed from 1.7 cm (inner) diameter PVC pipe. Approximately 30 evenly spaced holes were drilled in the lower 15 cm of each pipe and bottoms were sealed with duct tape. Three to four piezometer transects were established in each marsh during May 2005 in the lower Bay and July 2005 in the upper Bay. Transects were placed to examine the impact of various geomorphological features such as the tidal channel edge, bayward edge, upland edge, etc. Piezometers were installed as deep as possible. Within 5 m of tidal channels or the bayward edge they were installed to depths of 112 to 52 cm and the remainder to depths of 70 to 32 cm. Seventy-two additional pairs of shallow and deep piezometers were established throughout Wood Point and John Lusby in August 2005, for a total of 36 pairs per marsh. Shallow piezometers, which had holes only in the lower 10 cm, were inserted 15 cm into the marsh and $\leq$0.5 m from the deep piezometer. Piezometers were capped and since they were sometimes submerged by tides, a vent hole was not included in the design. Importantly, the caps provided watertight but not airtight seals. Seals were tested in the laboratory by submerging capped ends in water, forcing air into the pipes, and observing air bubbles. Therefore measurements of hydraulic head were made relative to ambient atmospheric pressure (i.e., no build up of pressure within the pipe).

At Dipper Harbour, transects A and B extend away from the primary channel while transect C extends away from a secondary channel (Fig. 2). Transect B terminates 1.7 m from the upland edge while transects A and C terminate at the center of peninsulas. At Saint's Rest transects D and G extend away from the primary channel and terminate 22.9 m from the upland and at the edge of a pool, respectively (Fig. 3). Saint's Rest transect E extends away from a deep ditch while transect F extends between two shallower ditches of similar depth (Fig. 3). At Wood Point, transect H and I extend away from a secondary channel and the bayward edge, respectively (Fig. 4). Transect J extends between a primary and secondary channel while transect K extends towards the upland (Fig. 4). At John Lusby, transects L and M extend from primary and secondary channels, respectively, while transect N extends between a secondary and tertiary channel (Fig. 5). Transect O extends from the bayward edge and terminates in an area of small, interconnected pools (Fig. 5). Transect P extends between two small channels ~1 m deep.

To determine marsh surface elevation at piezometer locations and location of the marsh upland border, mapping was conducted in May and August 2005 using a Carrier Phase Trimble 4700 Differential Global Positioning System (DGPS) and kinematic survey method (Trimble Corporation, Sunnyvale, CA). After collection, the GPS data were differentially corrected with base station data from New Brunswick (Dipper Harbour: #20704, Saint's Rest: #20091, Wood Point: #20379) and Nova Scotia (John Lusby: #215065) local survey benchmarks. Survey points were post-processed using Trimble GP Survey and Trimble Survey Office v. 1.52 (Trimble Corporation, Sunnyvale, California) and survey benchmark coordinates were transformed to the International

Terrestrial Reference Frame (ITRF 2000). Corrected points were imported as geographic information system (GIS) shapefiles in ArcGIS v. 9.1 (ESRI, Redlands, CA). All geographic data were transformed to the Canadian Spatial Reference System 1998 (CSRS 98), North American Datum 1983 (NAD 83). Data from Dipper Harbour and Saints Rest were projected in the Universal Transverse Mercator (UTM) zone 19N coordinate system while data from Wood Point and John Lusby were projected in UTM zone 20N. Elevation was then transformed from the NAD 83 ellipsoidal value to Canadian Geographic Vertical Datum 1928 (CGVD28) orthometric height (i.e., height above mean sea level) using the GPS.H 2.1 Geoid Height Transformation program (Natural Resources Canada 2006).

To determine the depth to groundwater a metal tube was lowered into the piezometer while simultaneously forcing air through the tubing until water bubbles were heard. The length of tubing inserted was measured with a meter stick to the nearest 0.5 cm. Hydraulic head for each piezometer was calculated as marsh surface elevation (relative to mean sea level) minus depth to groundwater. For dry piezometers, hydraulic head was graphed as piezometer depth below the marsh surface (relative to mean sea level). Therefore, all hydraulic head measurements are reported as heights above sea level, referenced to CGVD28. Hydraulic head measurements were made during neap tides for several days in May, 2005 (lower Bay) and several days in July, 2005 (upper Bay). To determine variability over a tidal cycle, 3 to 5 measurements were made 2 to 5 hr apart during neap tides in August, 2005. Hydraulic head at transect I was intensively measured at 10 min intervals. Hydraulic head of paired piezometers was also measured during August neap tides. Saturated hydraulic conductivity for each marsh was based on the height of water present in piezometers within two days after installation and calculated using the equation $K = (piR^2(\ln (d - y_1)/(d - y_2)))/(A(t_2 - t_1))$ where K is soil permeability [cm s^{-1}], R is the inside radius of the piezometer [cm], d is the depth of piezometer below water-table [cm], y_1 is the depth below water-table of water in piezometer at t_1 [cm], y_2 is the depth below water-table of water in piezometer at t_2 [cm], A is a function of the system and in this case was estimated to be 4 cm, $t_2 - t_1$ is the time required for water to rise from t_1 to t_2 [s] (Luthin and Kirkham 1949). All remaining hydraulic head measurements were made several days after piezometers were installed to allow time for their equilibration.

In the GIS, the shortest distance between each piezometer and the upland edge was also measured. Since the marshes differed in geological setting and degree of human disturbance, the upland edge usually involved a break in slope and was defined as either the start of trees, sand berms, rock fill, agricultural fields, or grass lawns. Mean high water for each of the marshes was calculated using methods described in chapter 3.

Sediment characteristics

Salinity of water collected from piezometers was measured with a hand-held refractometer. Surface infiltration rates were measured once along the longest transect at each marsh using a double ring infiltrometer (Turf-Tec International, Coral Springs, FL). Care was taken to locate the infiltrometer near piezometers while avoiding sediment compacted by foot traffic. Vegetation was clipped and the 15.24 cm diameter inner ring and 30.48 cm diameter outer ring were sharpened with a file to ease insertion. After pushing the rings 1 – 3 cm into the marsh, sediments were measured for compaction with a ruler at three locations inside and outside the 15.24 cm diameter ring. Compaction never exceeded 0.5 cm. Both rings were filled to the top with water providing an initial head of 9-7 cm. Infiltration was measured as the change in water level in the inner ring over 15 minutes. Because measurements were made on days with no precipitation and during non-inundating neap tides, infiltration rates reported represent dry conditions in the marshes.

Statistical analysis

Regressions were performed using SPSS 13.0 for Windows.

Results

Hydraulic head and salinity variation during tidal cycles

Despite a large tidal range, hydraulic head is fairly stable over the course of semidiurnal neap tidal cycles in August. Over the course of semidiurnal tidal cycles the net change in 91% of the piezometers (75 out of 82) is less than 20 cm (Fig. 6). I analyzed the maximum depth that groundwater reached during the semidiurnal cycle (Fig. 7) and found a logarithmic decrease in depth to groundwater with distance from channel at Dipper Harbour and John Lusby (Table 1). Three transects, C, H, and I, were flooded by tidal water at channel/bayward edges during August neap tide monitoring. Although piezometers 0 m along transects C, H, and I were flooded to depths of 40, 44, and 16 cm, respectively, displacement of hydraulic head was $\leq$10 cm (Fig. 8a).

Groundwater salinities were fairly stable over this time period. For the majority of piezometers, salinity either was constant or exhibited a change of one to five units (Figs. 9, 10, and 11b). In several cases, however, I detected changes of ~10 units over the course of a day; for example, along transect C (Fig. 9b), transect O (Fig. 10h), and transect B (Fig. 11b). The lowest recorded salinity was 6 (21 m along transect B) and the highest recorded salinity was 47 (15 m along transect C). The majority of measurements were in the mid 20's to mid 30's. Though salinity was sometimes highest part way between tidal channels and the upland (i.e., Figs. 9a, 9c, and 10h) or between two tidal channels (i.e., Fig. 10c) there was no relationship between salinity and distance from nearest channel or salinity and elevation. Salinity and distance from upland as well as salinity and time of day also showed no relationship.

Two marsh transects (D and J) displayed differences in depth to groundwater due to the pattern of inundating and non-inundating tides that occur as a result of the spring/neap tidal cycle. During non-inundating neap tides, depth to groundwater at transect D (Fig. 12a) was as much as 60 cm at the channel edge and 40 cm in the marsh interior. Similarly, at transect J (Fig. 13e) depth to groundwater was as much as 65 cm at the channel edge and 45 cm in the marsh interior. When spring tides flooded the marsh surface at transect D, depth to groundwater decreased to 35 cm at the channel edge and 5 cm in the marsh interior (Fig. 12b). Fifteen days after the last inundating spring tides along transect J, depth to groundwater decreased to 30-40 cm at channel edges and 10-16 cm in the marsh interior (Fig. 13f).

Seasonal variation in hydraulic head and salinity

Several piezometers at Dipper Harbour showed higher hydraulic head in May compared to August. The three most upland piezometers (within 5.6 to 1.7 m of the upland) along transect B all had water present in May (Fig. 14a). Water level ranged from at or slightly above the marsh surface to 9 –35 cm below the marsh surface. During this time, these three piezometers had markedly lower salinities (<10) (Fig. 11a). These same piezometers were dry in August, indicating that groundwater was greater than 40 cm below the marsh surface (Fig. 14b). Groundwater salinity was generally higher than in August than May (Fig. 11).

Results from the piezometers at 12 and 15 m along transect C (Fig. 14) indicate that the effects of seasonal variation can also be detected at locations far from the upland as both of these piezometers are ~84 m from the upland edge. These two piezometers both had water present in May, but these same piezometers were dry in August, indicating that groundwater was greater than 27.5 and 21 cm below the marsh surface, respectively. These two piezometers are relatively far (12 and 15 m, respectively) from the tidal channel, and have relatively high elevations (3.45 and 3.48 m, respectively).

Sediment characteristics

Depth to groundwater is a good predictor of redox potential measured at 30 cm in the sediment at Saint's Rest and John Lusby and redox potential measured at 15 cm at Dipper Harbour, Saint's Rest, and John Lusby (Fig. 15). The variability in redox potentials at Dipper Harbour (at 30 cm) and Wood Point (at both depths) cannot be explained by depth to groundwater.

Saturated hydraulic conductivities measured in Bay of Fundy sediments range from 14 x 10^{-6} cm s^{-1} to 0.059 x 10^{-6} cm s^{-1} (Table 2) and mean values for each marsh are reported in Table 3. Infiltration rates for Bay of Fundy marshes (Fig. 16) are several orders of magnitude higher than saturated hydraulic conductivities (Table 2), indicating that, on average, the rate at which water can enter the marsh sediment is greater than the rate it can move through it. Further, mean infiltration rates are higher in locations dominated by high marsh vegetation compared to locations dominated by low marsh vegetation (Fig. 16).

Multiple phreatic zones

Results from most of the paired piezometers indicate that, as expected, there is a vadose (i.e., unsaturated) zone above a phreatic (i.e., saturated) zone. However, there were three locations at John Lusby in which there is a phreatic zone near the marsh surface, followed by a vadose zone and then another, deeper phreatic zone. All three locations were within 10 m of a deep, primary channel. The results suggest that perched water-tables are present in the marsh. The first phreatic zone occurs at 7.5 to 14 cm below the marsh surface and is 2 to 7.5 cm thick. The second phreatic zone occurs at 25.5 to 50 cm below the marsh surface. The vadose zone that occurs between the two phreatic zones ranges in thickness from 9.5 to 35 cm.

Discussion

Effect of marsh geomorphology on hydraulic head

Hydraulic head varied with respect to channel depth, levees, and bedrock underlying marsh sediment. Figures 12 and 13 reveal variability with channel depth. The piezometer at 8 m along transect F is approximately equidistant from three functioning ditches of various depths. Based on August data, the hydraulic gradient between this piezometer and the deep ditch is 0.12, this piezometer and the first shallow ditch (0 m along the transect) is 0.02, and this piezometer and the second shallow ditch (12.5 m along the transect) is 0.03. Although both shallow ditches apparently influence the hydraulic head of this piezometer, the hydraulic gradients – and therefore the amount of influence – are an order of magnitude less than the gradient imposed by the deep ditch. The piezometer 0 m along transect J is located near a 3.8 m-deep channel while the piezometer 49 m along the transect is located near a 2.6 m-deep channel (Fig. 4).

Hydraulic head data during the end of neap tides in July indicate that the deeper channel results in a greater draw-down of groundwater – groundwater is 65 cm below the marsh surface at the 0 m peizometer and 40 cm below the marsh surface at the 48.5 m piezometer (Fig. 13). Levees are present at transects D (Fig. 12), J (Fig. 13), L (Fig. 17), and O (Fig. 17). At each, the hydraulic head peaks at the levee and then decreases with distance from it. Bedrock underlies and outcrops in sections of Dipper Harbour transect B. High hydraulic head at both 5 m and 15 m along this transect (Fig. 14) is likely due to impermeable and shallow bedrock. As a result, groundwater flows towards locations of lower hydraulic head, such as towards the channel, towards the piezometer located at 11 m along the transect, and towards the upland.

Hydraulic head, and therefore hydraulic gradient, near tidal channels is largely influenced by the marsh surface slope (Nuttle 1988). In Fundy marshes, the hydraulic gradient within 3 m of tidal channels is 21, 46, 41, and 20% at transects A, G, H, and J, respectively. The slope of the marsh surface within 3 m of channels in these same locations is 43, 22, 39, and 15% respectively. In all but one of the microtidal marshes studied, the slope of the marsh surface, and thus hydraulic gradient, is lower than in Fundy marshes (Table 4). Steep slopes along the channels of Fundy marshes drive sub-surface hydrology as they ensure constant drainage of groundwater through channel bank sediments.

Harvey et al. (1987) note that drainage is also influenced by marsh elevation relative to the tidal frame as marshes with higher relative elevations are exposed for longer periods of time. Therefore, losses due to drainage and ET occur over a greater percentage of the tidal cycle. Unfortunately, few researchers report marsh relative elevations; therefore, a comparison between Fundy marshes and those elsewhere is difficult. However, greater tidal range should have an effect similar to higher marsh relative elevation. As previously mentioned, flooding frequency decreases with tidal range in the Bay of Fundy (Desplanque and Mossman 2004). Infrequent flooding would allow more time for drainage to occur and deep, inter-tidal channels characteristic of macrotidal systems should result in steeper hydraulic gradients and greater drainage along channel banks (Howes and Goehringer 1994).

Hydraulic head variation during tidal cycles

Studies from microtidal marshes show that the influence of tidal height is most pronounced at and near channel edges since the most lateral and vertical infiltration and drainage occurs in this region (Montalto et al. 2005; Hughes et al. 1998; Harvey et al. 1987; Agosta 1985; Jordan and Correll 1985). If tidal height also drives the sub-surface hydrology of macrotidal marshes then one would expect large changes in hydraulic head, compared to microtidal marshes, since tidal height in macrotidal marshes changes by many meters over the course of a semidiurnal cycle. Further indication of tidal height influencing sub-surface hydrology is if hydraulic head measurements have a component with a 12.5 hr period, corresponding to the length of the semidiurnal tidal cycle (Nuttle 1988). Data from inundated channel edges of microtidal marshes indicates that groundwater is displaced anywhere from 20 to 110 cm. This displacement has a 12.5 hr period, more or less (Fig. 8b), since hydraulic head lags tidal inundation at most by 45 min (Table 4). In contrast, inundated Bay of Fundy channel edges show less hydraulic head displacement than microtidal marshes (Fig. 8a). Since there was no distinct peak or trough in hydraulic head displacement, I was unable to calculate the lag in hydraulic head relative to high tide. Thus, tidal height is not a major driver of sub-surface hydrology in Fundy marshes.

On the time scale of spring and neap tidal cycles (i.e., ~28 days), I observed hydraulic head differences of 20 to 40 cm (Figs. 12 and 13), which is comparable to other marshes (Montalto et al. 2006; Nuttle 1988). These findings suggest that inundation and length of the non-inundation period are important drivers of Fundy sub-surface hydrology. Depth to groundwater increases during the non-inundation period due to losses via ET and drainage. Inundating tides supply water to the marsh and therefore depth to groundwater decreases.

Seasonal variations in hydraulic head and salinity

Seasonal variations in hydraulic head and groundwater salinity at Dipper Harbour can be explained by differences in precipitation and ET. Precipitation was greater in May than August and 2005, in particular, represented higher than normal spring precipitation and lower than normal fall precipitation (Table 5). ET is expected to be lower in May than in August due to cooler temperatures (Table 5) and low living biomass. Greater precipitation, greater flow from the upland, and less ET in May explain higher hydraulic head and low groundwater salinities of the three most upland piezometers along Dipper Harbour transect B.

The higher elevation and greater distance from the channel of piezometers at 12 and 15 m along transect A results in less input of water from tidal inundation and less drainage of water into the tidal creek compared to other piezometers along this transect. High precipitation and low ET in May likely drove groundwater depth at these two piezometers since less time and a lesser volume of water was required to raise groundwater towards the marsh surface. Less precipitation in August along with higher ET resulted in these piezometers being dry while the other transect A piezometers had water present.

This trend of higher hydraulic head in May compared to August was not observed in any piezometers at Saint's Rest. Transects E and F occur close to a tidal channel and are located below MHW, in contrast to Dipper Harbour transects which, except for piezometers near the tidal channel, are located above MHW. Therefore, at transects E and F, any effects of precipitation on hydraulic head are likely to be masked by the effects of drainage and tidal inundation. Although transect D occurs at higher elevation and approaches the upland, distance between the nearest piezometer and the upland is 22.9 m at Saint's Rest compared to 1.7 m at Dipper Harbour. Therefore, the effect of underground flow on transect D is likely to be undetected.

Although seasonal comparisons cannot be made in the upper Bay marshes, it is interesting to note that in July and August groundwater was very close to the marsh surface along portions of transects H, I, M and O (Figs. 13 and 17) despite high temperatures and presumably high ET (Table 5). In the Bay of Fundy, Desplanque and Mossman (2004) demonstrate that as tidal range increases inundation frequency at high elevations within these marshes decreases; therefore, precipitation is likely to be a more important water delivery mechanism in irregularly flooded macrotidal marshes.

Sediment characteristics

Generally, depth to groundwater is inversely related to redox potential. However, redox potentials at Dipper Harbour (30 cm) and Wood Point contradict expected results and are likely due to factors other than depth to groundwater that have not been investigated in this study. Possible explanations include the fact that there may be a lag time between air entry and redox response (La Riviere et al. 2004) or the inability or reduced ability of plants to produce oxygen (i.e., lower productivity) and affect redox (Howes et al. 1981; La Riviere et al. 2004).

Despite the minerogenic nature of Fundy sediment (Conner et al. 2001), mean infiltration rates of the four study marshes are on the same order of magnitude as marshes studied elsewhere (Table 6). However, in Fundy marshes differences in infiltration rates between high and low marsh (Figure 16) likely play an important role in sub-surface hydrology. With greater infiltration rates in the high marsh, water that enters here can replace water lost in the low marsh. Differences in elevation between high and low marsh ensure that a hydraulic gradient is present in order for water to move from high to low marsh.

Mean saturated hydraulic conductivity in Fundy marshes is one to five orders of magnitude lower than in marshes studied elsewhere (Table 3), indicating that the movement of water in the phreatic zone is very slow. Although I did not measure hydraulic conductivity in the vadose zone, it is likely to be several orders of magnitude lower than the saturated hydraulic conductivity (Nuttle 1988) – anywhere from 10^{-7} to 10^{-10} cm s^{-1}. Hydraulic conductivity in both the vadose and phreatic zones is an important driver of sub-surface hydrology. Due to low hydraulic conductivity in the vadose zone, infiltrated water will take a long time to reach the phreatic zone. Within the phreatic zone, recharge and drainage of Fundy marshes occurs at a much lower rate than in marshes elsewhere. In organogenic marshes, water infiltrates and reaches the phreatic zone fairly quickly – changes in hydraulic head lag changes in tidal height anywhere from 0 to 90 min (Table 4). Conversely, low hydraulic conductivity in both the vadose and phreatic zones of minerogenic Fundy marshes further suggests that they are unresponsive to large changes in tidal height. When Harvey et al. (1987) varied hydraulic conductivity in their sub-surface hydrology model, they obtained similar results. In high conductivity sediments, water-table recharged and discharged quickly over time. In lower conductivity sediments, changes in hydraulic head occurred more slowly. In Fundy marshes, hydraulic head is likely to lag changes in tidal height by a much longer period of time than in organogenic marshes, potentially much longer than a semidiurnal tidal cycle.

Multiple phreatic zones

Paired shallow and deep piezometers indicate that a vadose zone lies above the phreatic zone. However, in some locations there is a shallow phreatic zone, followed by a vadose zone, followed by a deeper phreatic zone. Hydraulic conductivity in the vadose zone is likely to be several orders of magnitude lower (Nuttle 1988). In these situations, the upper phreatic zone is likely to be hydrologically disconnected from the deeper phreatic zone. In sub-surface hydrology literature this shallow phreatic zone is known as a perched water-table that forms above sediment or a geological unit of much lower hydraulic conductivity (Fitts 2002). To my knowledge, a perched water-table has not been previously documented in salt marsh sediments, thus further research is necessary to understand its occurrence.

Deep hydraulic head draw-downs

In Bay of Fundy marshes, groundwater can be quite close to the marsh surface (i.e., 0-30 cm in Figs. 12 to 14 and 17), which is comparable to marshes studied elsewhere (Table 4). However, maximum depth to groundwater recorded in the marsh interior, 40 to 45 cm, and at channel edges, 100 cm at Dipper Harbour (Fig. 14a), 65 cm at Saint's Rest (Fig. 12a and 12e), 68 cm at Wood Point (Fig. 13e) and John Lusby (Fig. 17c) can be much larger than results from other marshes (Table 4). The only other marshes with deep draw-downs of groundwater are macrotidal (i.e., James Bay and Torridge Estuary - Table 4). The deep draw-down of groundwater in macrotidal marshes can be explained by a combination of inundation and sediment characteristics. Although channel edges are frequently flooded (see chapter 3, Table 4) and infiltration rates are comparable to other marshes (Table 6), infiltrated water moves very slowly through the vadose and phreatic zones. In the relatively large distance between the marsh surface and water-table, infiltrated water can fill pore spaces or be taken up by plants (Fitts 2002), and thus never reach the phreatic zone. Water that does reach the phreatic zone is drained by steep hydraulic gradients of the channel bank. Though high marsh sediment (in marsh interiors) has higher infiltration rates than low marsh sediment (near channels) (Fig. 16), the marsh interior is flooded less frequently. Therefore, in the marsh interior water is lost to ET and drainage towards channels during the intervening non-inundation periods.

Sub-surface hydrology in reference versus recovering marshes

Shallow depth to water does not occur throughout the two recovering marshes; therefore, aquacludes documented by Crooks and Pye (2000) are not likely to be present in these marshes. At Saint's Rest, the reclamation surface is located 11-21 cm below the marsh surface (Noel et al. 2005) and at John Lusby the reclamation surface is located 100 to 130 cm below the marsh surface (Graf 2005). Based on the depths to which I was able to manually install piezometers (i.e., 31-80 cm at Saint's Rest and 34-84 cm at John Lusby), those at Saint's Rest intercepted the reclamation surface while those at John

Lusby did not. Though groundwater is close to the marsh surface in locations of Saint's Rest and John Lusby, groundwater can be several 10's of cm below the marsh surface (Figs. 12 and 13). These deep depths to groundwater indicate that a shallow, continuous aquaclude is not present throughout the two recovering marshes.

Quantitative comparison between reference and recovering marsh pairs is complicated by differences in elevation, channel depth, and marsh size. However, some transects in each marsh pair were placed in similar geomorphological settings, allowing for qualitative comparisons to be made. Results from both marsh types indicate that hydraulic head is lowest near channels, peaks at channel levees, and generally increases with distance from tidal channels. I assume that sub-surface hydrological processes are operating similarly in both recovering and reference marsh types since qualitatively similar hydrological head results were obtained.

Conclusion

In Bay of Fundy marshes, hydraulic head is lowest near channels and generally increases with distance from tidal channels, a pattern similar to marshes elsewhere. Differences in hydraulic head over the time scale of spring/neap tidal cycles as well as infiltration rates for Fundy sediments are comparable to marshes elsewhere. Despite these similarities, there are major differences between Fundy marshes and those previously studied. Sub-surface hydrology research that has mostly been conducted in organogenic, microtidal marshes does not provide suitable models for minerogranic, macrotidal marshes.

Results from recovering and reference salt marshes indicate that it is possible for sub-surface hydrology *processes* to be restored after half a century. However, different sub-surface hydrology *patterns* may exist between the two marsh types. At Saint's Rest is there a greater portion of the marsh closer to channels (i.e., $\leq$ 3 m) compared to the reference site (Chmura and MacDonald 2006). A higher channel density suggests that there is a greater portion of the marsh experiencing deep groundwater draw-downs near tidal channels. Increasing depth to groundwater has been shown to increase plant productivity (Balling and Resh 1983); this feedback could be present at Saint's Rest marsh.

Sediment, inundation patterns, geomorphology, and precipitation influence hydraulic head in Bay of Fundy salt marshes. Since tidal height is not a major driver, the sub-surface hydrology, and therefore vegetation, of these macrotidal marshes will likely be resistant to greenhouse induced sea level rise. Restoration of these resistant marshes could offset inevitable loses in less resistant, microtidal marshes of Atlantic Canada. Restoration in the Bay of Fundy can be undertaken in locations where dikes do not protect valued infrastructure, agriculture is no longer practiced, or the cost of dike maintenance outweighs the benefits.

Table 1. Regression results for maximum depth to groundwater during August neap tides versus distance from deep channels for Bay of Fundy marshes. The first column describes the types of curves fitted to the data. Only results with $p \leq 0.05$ are reported.

	Dipper Harbour		Saint's Rest		Wood Point		John Lusby	
	r^2	p	r^2	p	r^2	p	r^2	p
Linear	-	-	-	-	0.259	0.008	0.407	0.006
Logarithmic	0.303	0.018	-	-	-	-	0.330	0.016
Exponential	-	-	-	-	0.446	< 0.001	0.488	0.002

Table 2. Saturated hydraulic conductivity (K) values along various transects in Bay of Fundy marshes.

K x 10^{-6} (cm s^{-1})	Transect	Distance along transect (m)
14	A	0
1.4	A	3
0.059	B	0
0.094	B	2
0.2	B	8
0.36	B	11
1.4	D	0
1.5	D	12
1.9	D	24
0.86	D	38
0.23	D	147
0.21	D	157
1.1	H	3
4.3	H	5
0.54	H	10
0.47	H	15
0.66	H	25
0.77	H	35
0.39	I	3
0.67	I	5
0.67	I	10
0.58	I	30
0.61	I	60
0.60	I	120
6.6	J	0
2.5	J	3
5.8	J	5
5.4	J	10
3.4	J	15
7.7	J	25
1.3	J	35
1.8	J	40
1.7	J	45
4.6	J	49
3.3	K	25
5.0	K	55
0.69	L	0
0.76	L	24
1.1	M	8
0.74	O	20
0.79	O	0
1.3	P	0
1.2	P	10

Table 3. Mean saturated hydraulic conductivity (K) of Bay of Fundy marshes compared to other study sites.

Marsh	K (cm s^{-1}) x 10^{-6} $\pm$ 1 S.E.	Depth (cm)	# measurements	Reference
Dipper Harbour	2.7 $\pm$ 2.3	32-105	6	this study
Saint's Rest	1.0 $\pm$ 0.3	36-67	6	"
Wood Point	2.4 $\pm$ 0.5	30-67	24	"
John Lusby	0.95 $\pm$ 0.0	46-77	7	"
Great Sippewissett	94 20000	0-40 40-110	unknown	Hemond and Fifield (1982)
Piermont	7750 $\pm$ 10	120	80	Montalto et al. (2006)
Eagle Bottom	1400	25	unknown	Harvey and Odum (1990)
Carter Creek	2000	25	unknown	Harvey and Odum (1990)
Richs Inlet	300	5-60	unknown	Yelverton and Hackney (1986)
Belle Isle	6300 100	0-60 60-100	unknown	Hemond and Chen (1990)

Table 4. Results from sub-surface hydrology studies performed at various tidal salt marshes.

Marsh and location	Tidal range (m)	Organic Matter (%)	Sediment type	Channel bank slope (%)	Depth to water channel edge (cm)	Depth to water marsh interior (cm)	Hydraulic gradient at channel edge (%)	Hydraulic head lag time (h:mm)	Reference
Petaluma, CA, USA				20	40	20-30	8		Balling and Resh 1983
Rhode River, MA, USA	0.3			2	13	1-2	4	0	Jordan and Correll 1985
Carter Creek, VA, USA	1	16	Peat over sand	12	0-10	0		0	Harvey et al. 1987
Richs Inlet, NC, USA	1	10	Sand, silt, clay	1	25	0-5	6		Yelverton and Hackney 1986
Piermont, NY, USA	1.1	12-60	Mucky peat		15-20	10		0 - 0:30	Montalto et al. 2006
Great Sippewissett, MA, USA	1.2	40-60	Peat over sand	5	30-40	0-10	3	00:20-00:38	Howes et al. 1981; Hemond et al. 1984; Howes and Goehringer; 1994
North Inlet, SC, USA	1.4		Silty	20	10-30	0	31	0	Agosta 1985

Table 4 continued. Results from sub-surface hydrology studies performed at various tidal salt marshes.

Marsh and location	Tidal range (m)	Organic Matter (%)	Sediment type	Channel bank slope (%)	Depth to water channel edge (cm)	Depth to water marsh interior (cm)	Hydraulic gradient at channel edge (%)	Hydraulic head lag time (h:mm)	Reference
Belle Isle, MA, USA	2	10	Clayey peat over clay	10	5-40	5-30	5		Nuttle 1988
Hunter River, NSW, Australia	2	8-22	Mud over silty sand	8		15		0 - 0:45	Hughes et al. 1998
James Bay, QC, Canada	3.8		Peat over silt, sand, and clay		5-10	0-90			Price and Woo 1988; Price 1991
Torridge Estuary, Devon, UK	7		Clay over peat			40-60		1:30	Blackwell et al. 2004

Table 5. Climate data for the lower Bay from Saint John, New Brunswick and upper Bay from Sackville, New Brunswick and Nappan, Nova Scotia (Environment Canada 2006) for the months during which sub-surface hydrology was studied. Nappan located 6 km south of Amherst, NS.

Location	Month	Average daily temperature (^{o}C) $\pm$ 1 s.d.		Total Monthly Precipitation (mm)	
		2005	normal	2005	normal
lower Bay	May	10.0 $\pm$ 2.8	9.4 $\pm$ 1.2	195.8	117.5
	August	17.1 $\pm$ 2.0	16.9 $\pm$ 0.8	41.0	89.6
upper Bay	July	18 $\pm$ 2.5	17.5 $\pm$ 0.9	67.3	89.8
	August	18.6 $\pm$ 2.4	17.5 $\pm$ 0.7	26.2	84.6

Table 6. Mean infiltration rate of Bay of Fundy marshes compared to other study sites.

Marsh	Infiltration Rate (cm s^{-1}) $\pm$ 1 S.E.	# measurements	Reference
Dipper Harbour	0.099 $\pm$ 0.040	6	this study
Saint's Rest	0.035 $\pm$ 0.011	8	"
Wood Point	0.066 $\pm$ 0.025	22	"
John Lusby	0.071 $\pm$ 0.020	14	"
Hunter River	0.059 to 0.0059	unknown	Hughes et al. 1998
Piermont	0.014 $\pm$ 0.009	15	Montalto et al. 2006
St. Peter's	0.02 $\pm$ 0.063 0.0063 $\pm$ 0.0021	10-30	Crooks et al. 2002

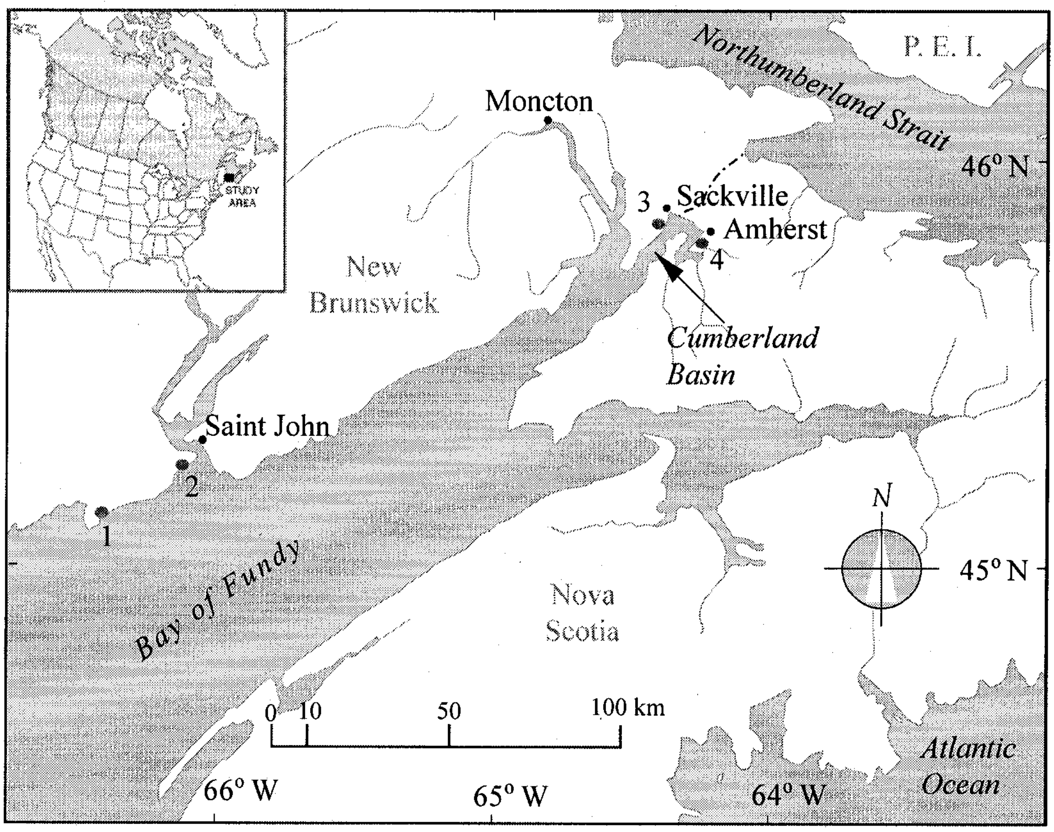

Figure 1. Map of the Bay of Fundy showing location of salt marsh study sites denoted by number: 1, Dipper Harbour marsh; 2, Saint's Rest marsh; 3, Wood Point marsh; 4, John Lusby marsh. Modified from Conner et al. (2001).

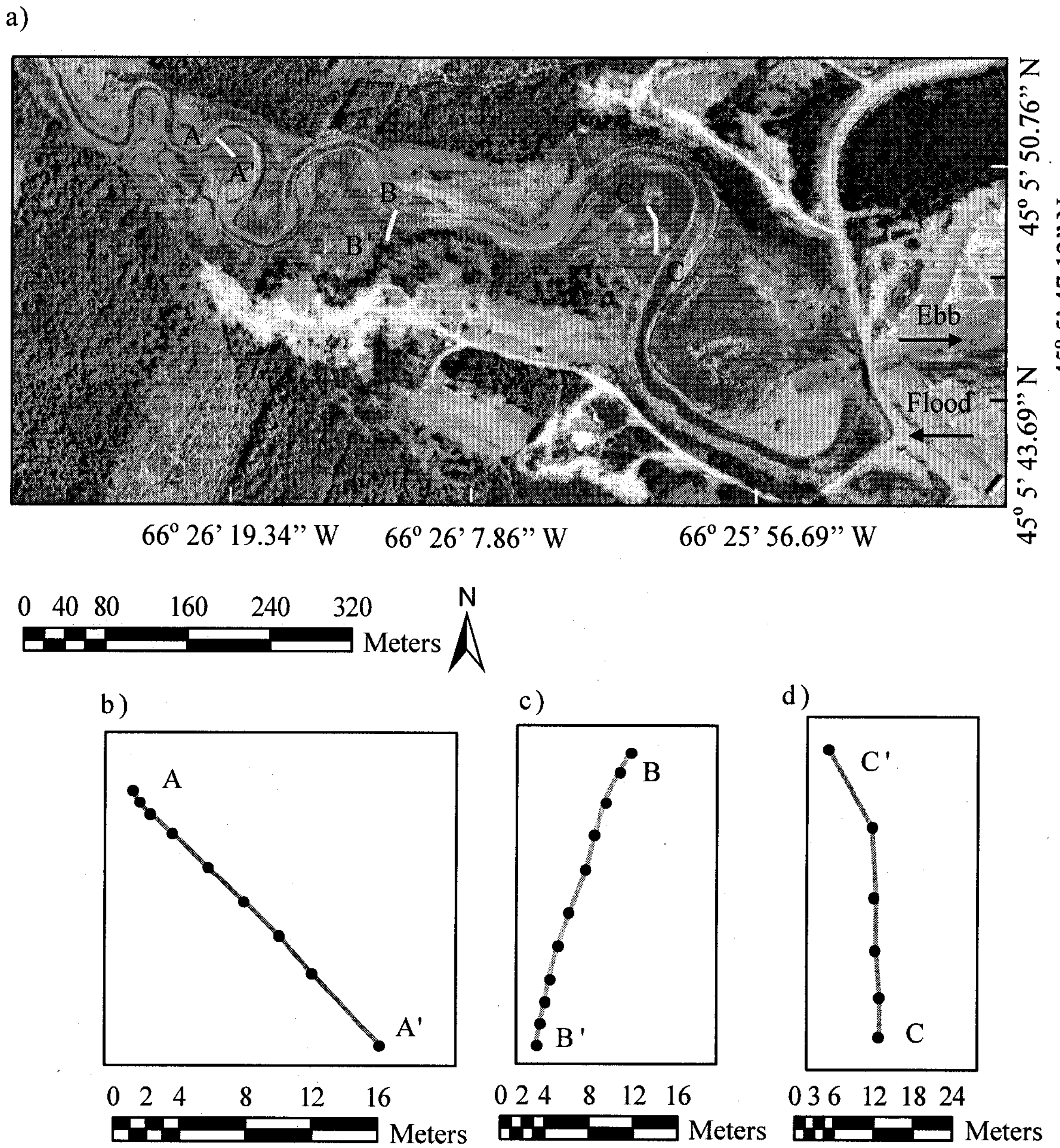

Figure 2. Piezometer transects at Dipper Harbour a) overview of marsh and transects, b) transect A, c) transect B, and d) transect C. (Photo source: Department of Natural Resources 1994).

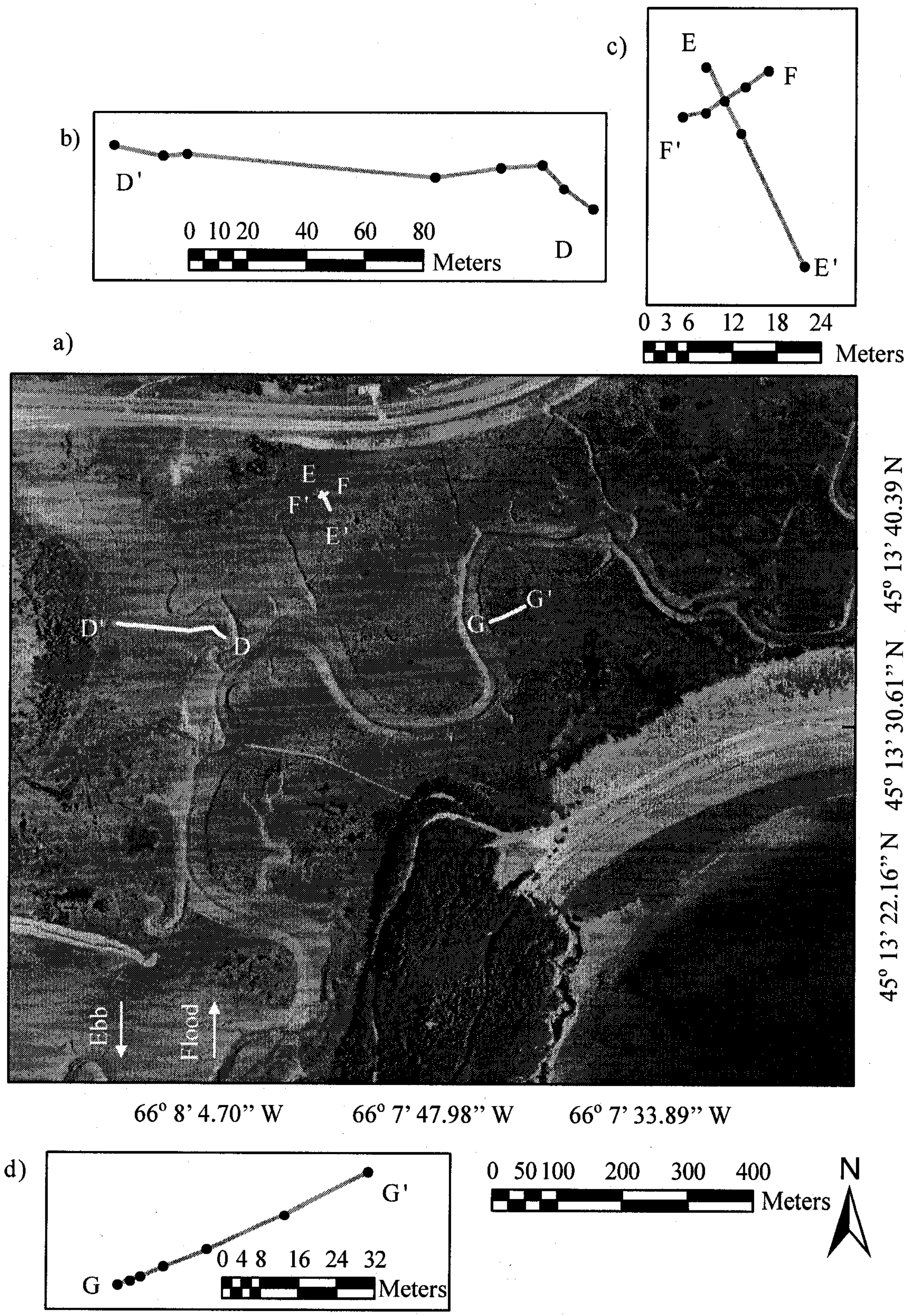

Figure 3. Piezometer transects at Saint's Rest a) overview of marsh and transects, b) transect D, b) transects E and F, c) transect G. (Photo Source: Department of Natural Resources 1994).

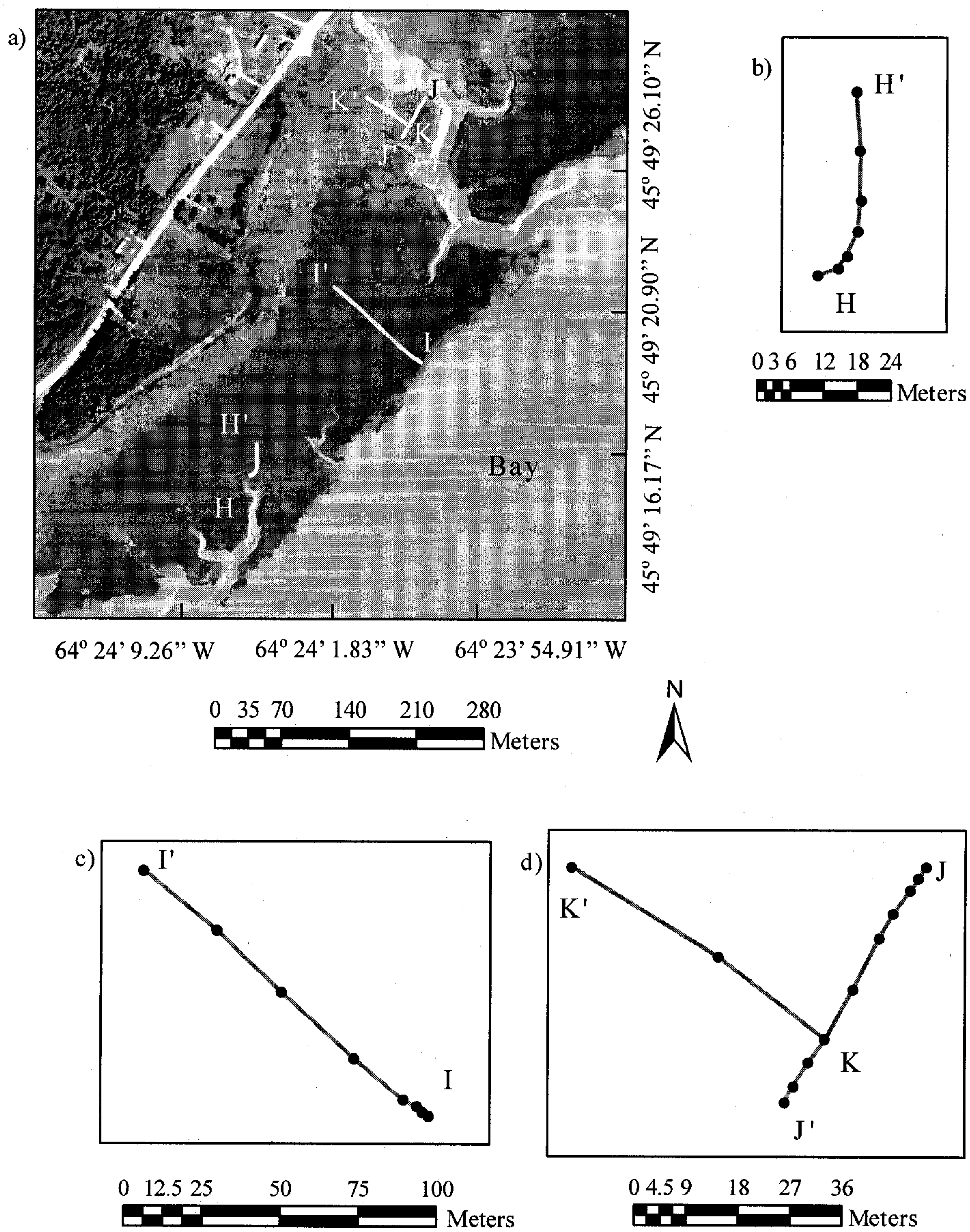

Figure 4. Piezometer transects at Wood Point a) overview of marsh and transects, b) transect H, c) transect I, and d) Transects J and K. (Photo Source: Department of Natural Resources 2001).

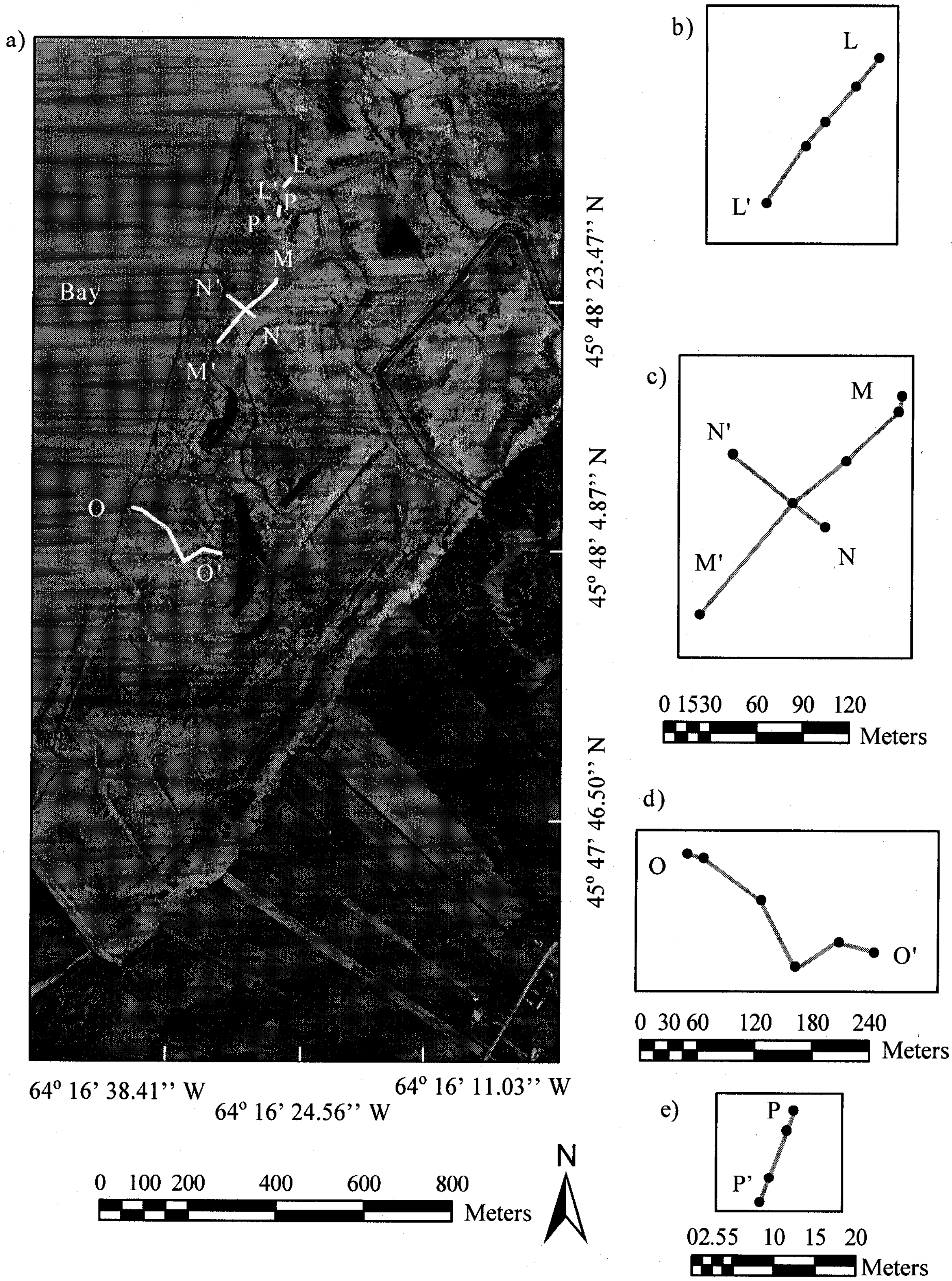

Figure 5. Piezometer transects at John Lusby a) overview of marsh and transects, b) transect L, c) transects M and N, d) transect O, and e) transect P. (Photo source: Department of Natural Resources 1995).

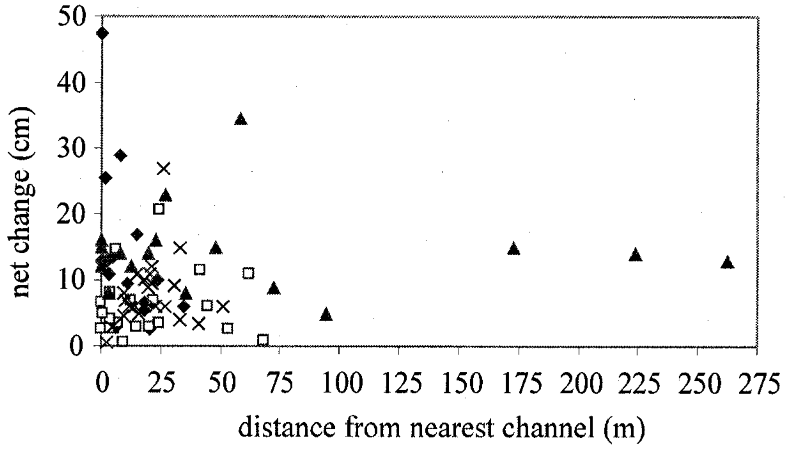

Figure 6. Net change in hydraulic head during semidiurnal neap tidal cycles. Dipper Harbour, 8/16/06 (♦); Saint's Rest, 8/18/06 (□); Wood Point, 8/5/06 (x); John Lusby, 8/4/06 (▲).

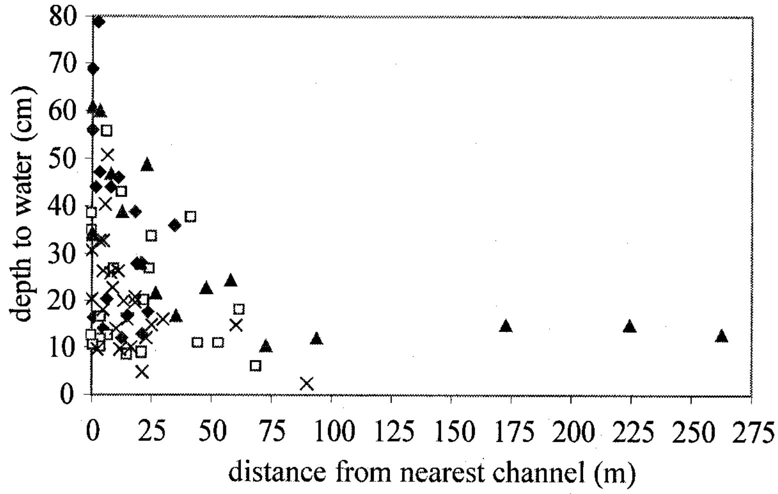

Figure 7. Maximum depth to water during semidiurnal neap tidal cycles. Dipper Harbour, 8/16/06 (♦); Saint's Rest, 8/18/06 (□); Wood Point, 8/5/06 (x); John Lusby, 8/4/06 (▲).

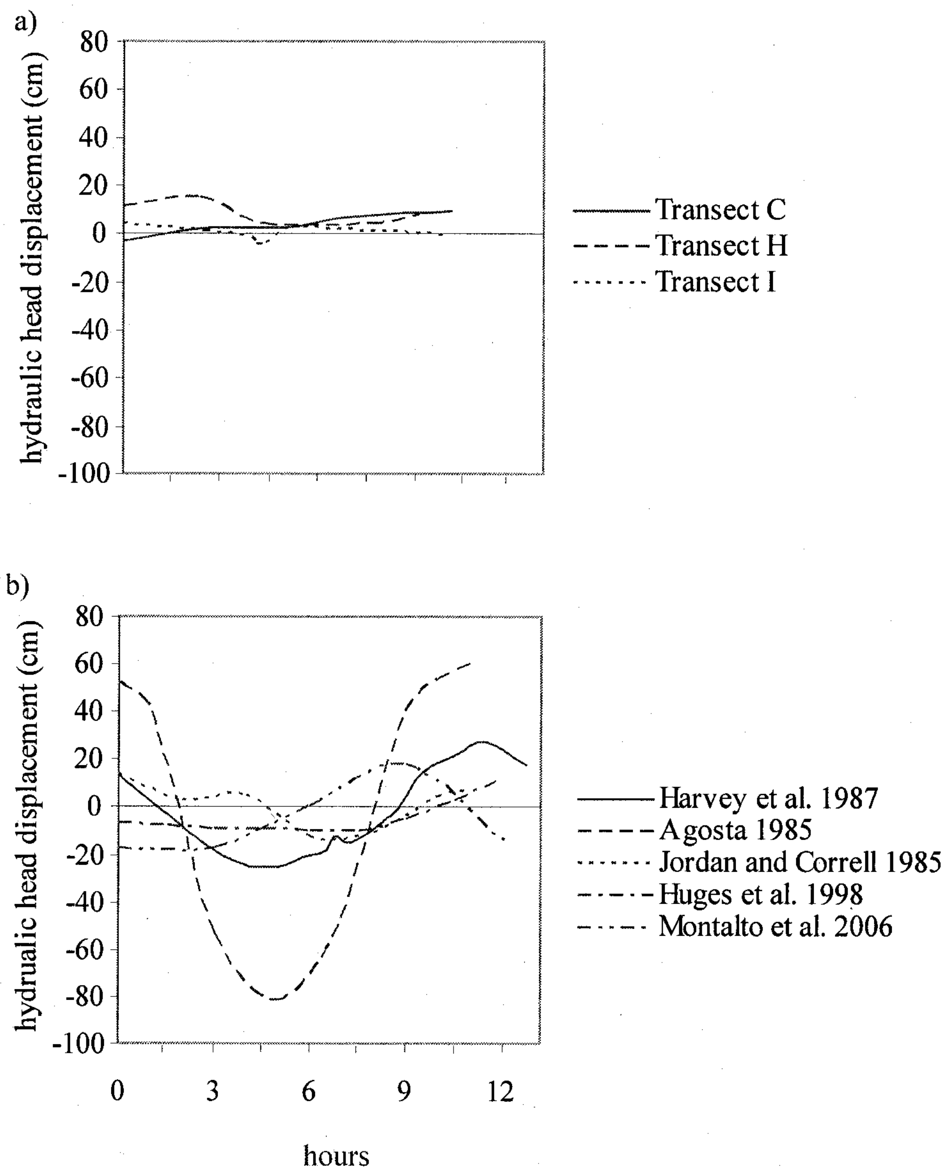

Figure 8. Magnitude of hydraulic head displacement during semi-diurnal tides near channel edges of a) Bay of Fundy marshes and b) marshes elsewhere, which are described in Table 1. Agosta (1985), Jordan and Correll (1985), and Montalto et al. (2006) measured displacement in water-table wells.

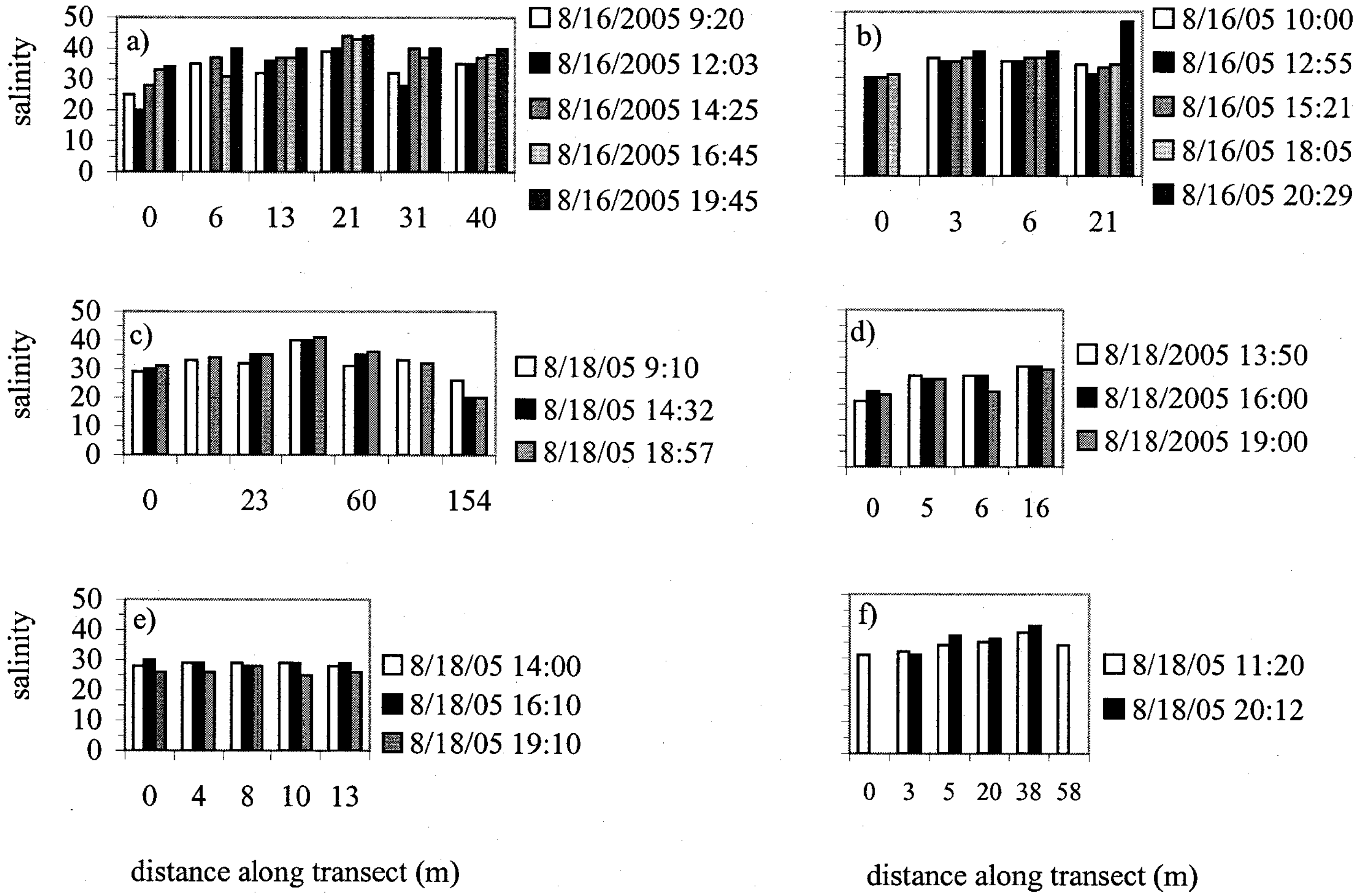

Figure 9. Change in salinity over the course of semidiurnal tidal cycles at lower Bay marshes. a) transect A, b) transect C, c) transect D, d) transect E, e) transect F, f) transect G. Start time is indicated and measurements were completed within 1 hr.

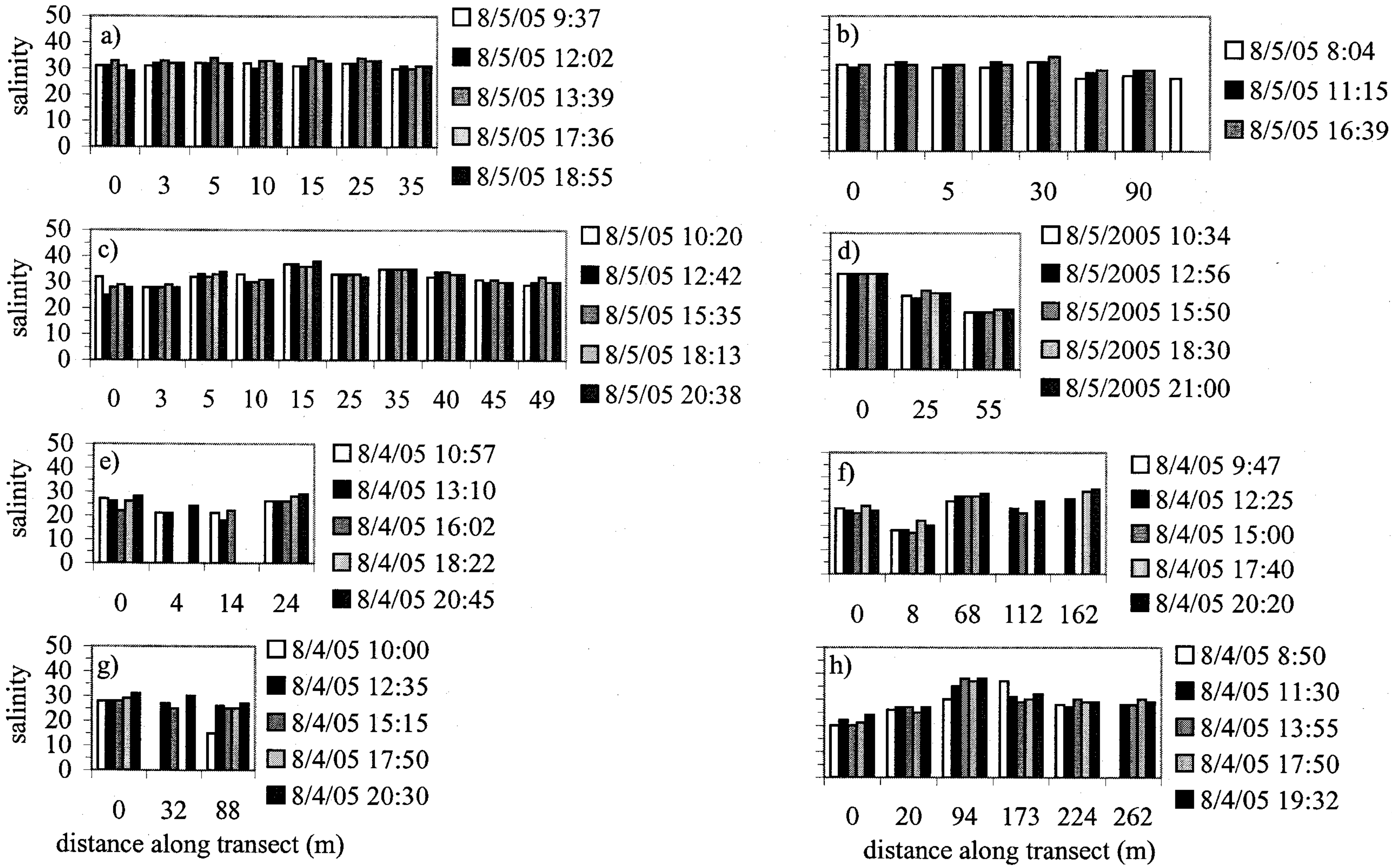

Figure 10. Change in salinity over the course of a semidiurnal tidal cycle at upper Bay marshes. a) transect H, b) transect I, c) transect J d) transect K, e) transect L, f) transect M, g) transect N, h) transect O. Start time is indicated and measurements were completed in 1 hr.

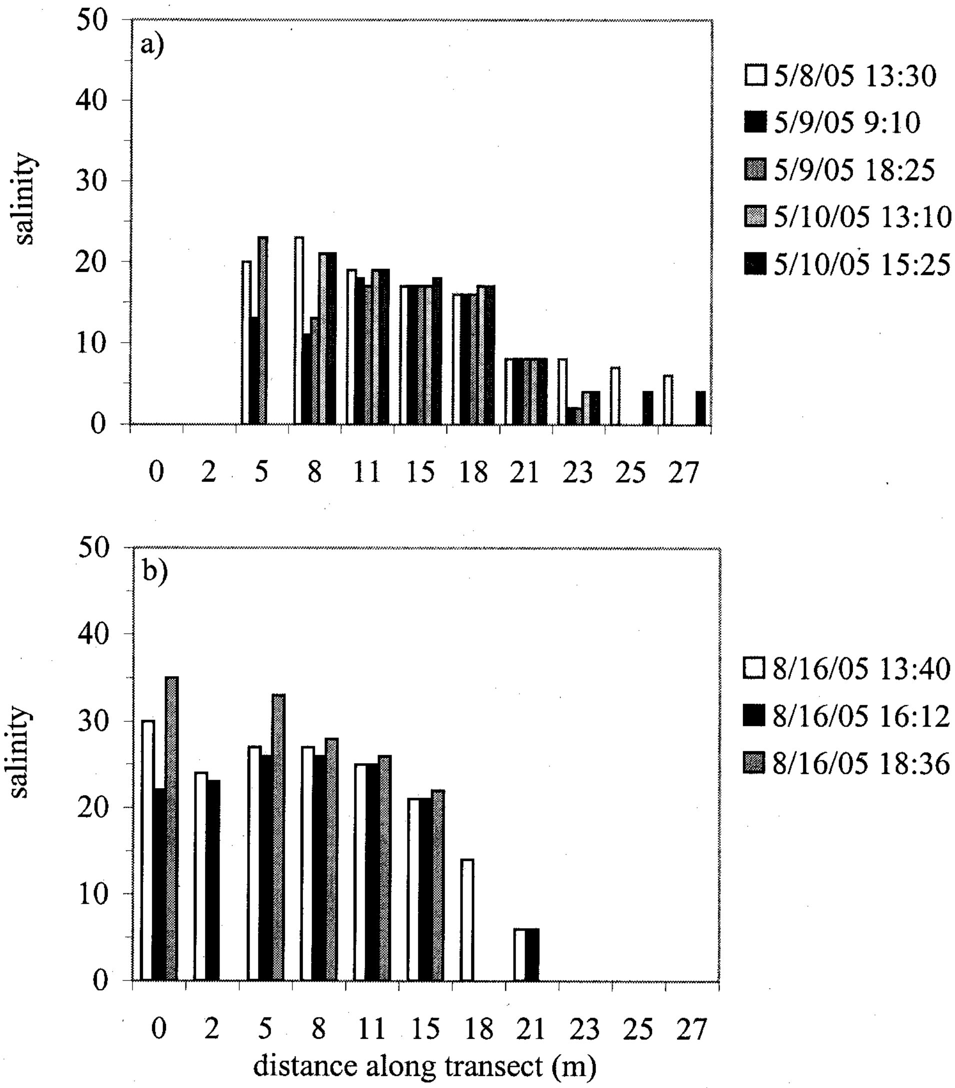

Figure 11. Seasonal comparison of salinity values along Dipper Harbour transect B. Measurements during a) May 8 – 10 and b) August 16. At distances of 0 and 2 m in May, high suspended sediment concentrations prevented accurate salinity measurements. At distances of 23-25 m in August salinity could not be measured in

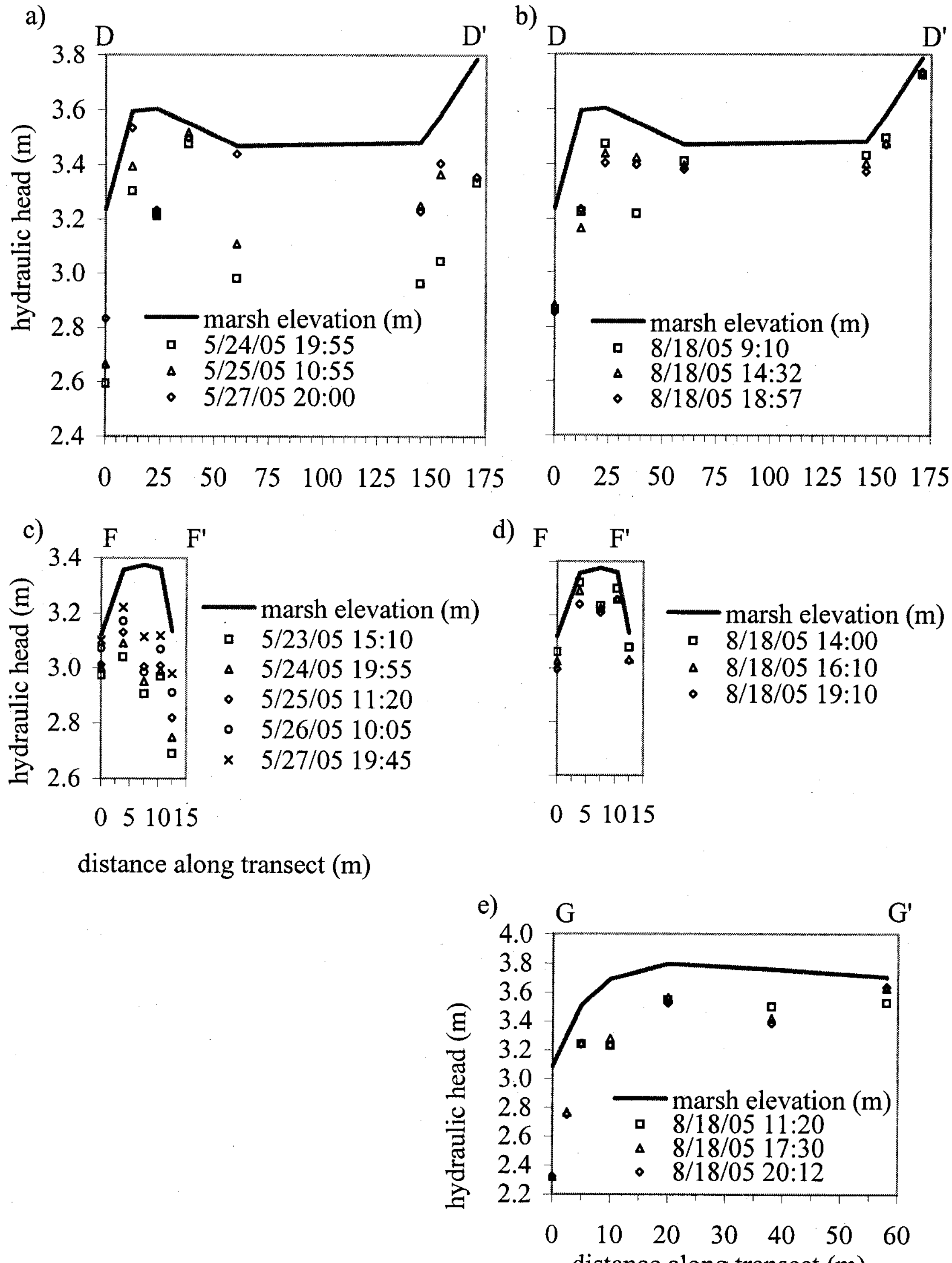

Figure 12. Saint's Rest hydraulic head measurements along a) transect D in May (2-4 days after spring tides), b) transect D August 18 (spring tides), c) transect F in May, d) transect F August 18, e) transect G August 18. Start time is indicated and measurements were completed within 1 hr. MHW at Saint's Rest is 3.3 m.

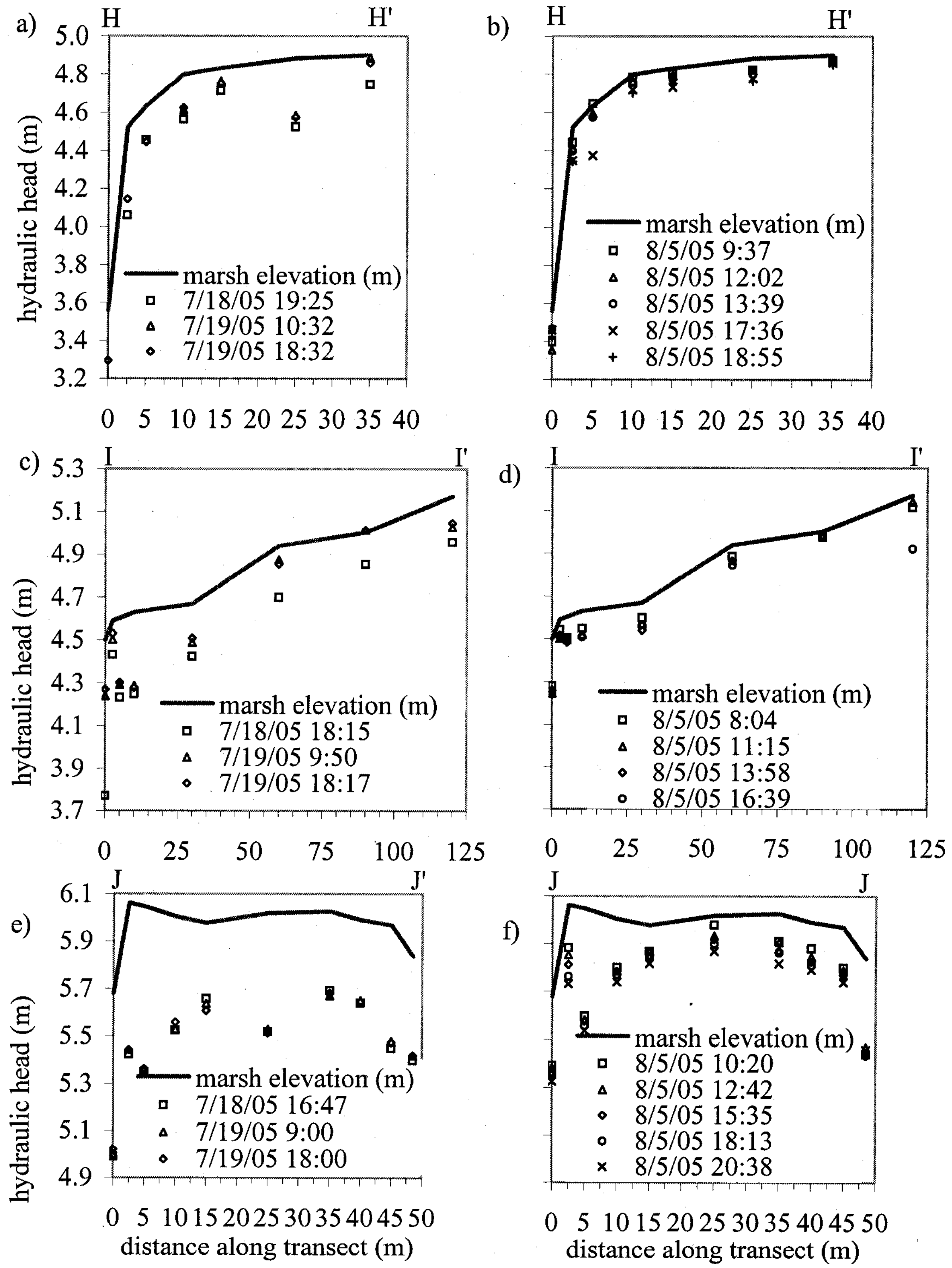

Figure 13. Wood Point hydraulic head measurements along a) transect H in July, b) transect H August 5, c) transect I in July, d) transect I August 5, e) transect J July (25-27 days after spring tides), and f) transect J August 5 (15 days after spring tides). Start time is indicated and measurements were completed within 1 hr. MHW at Wood Point is 5.2 m.

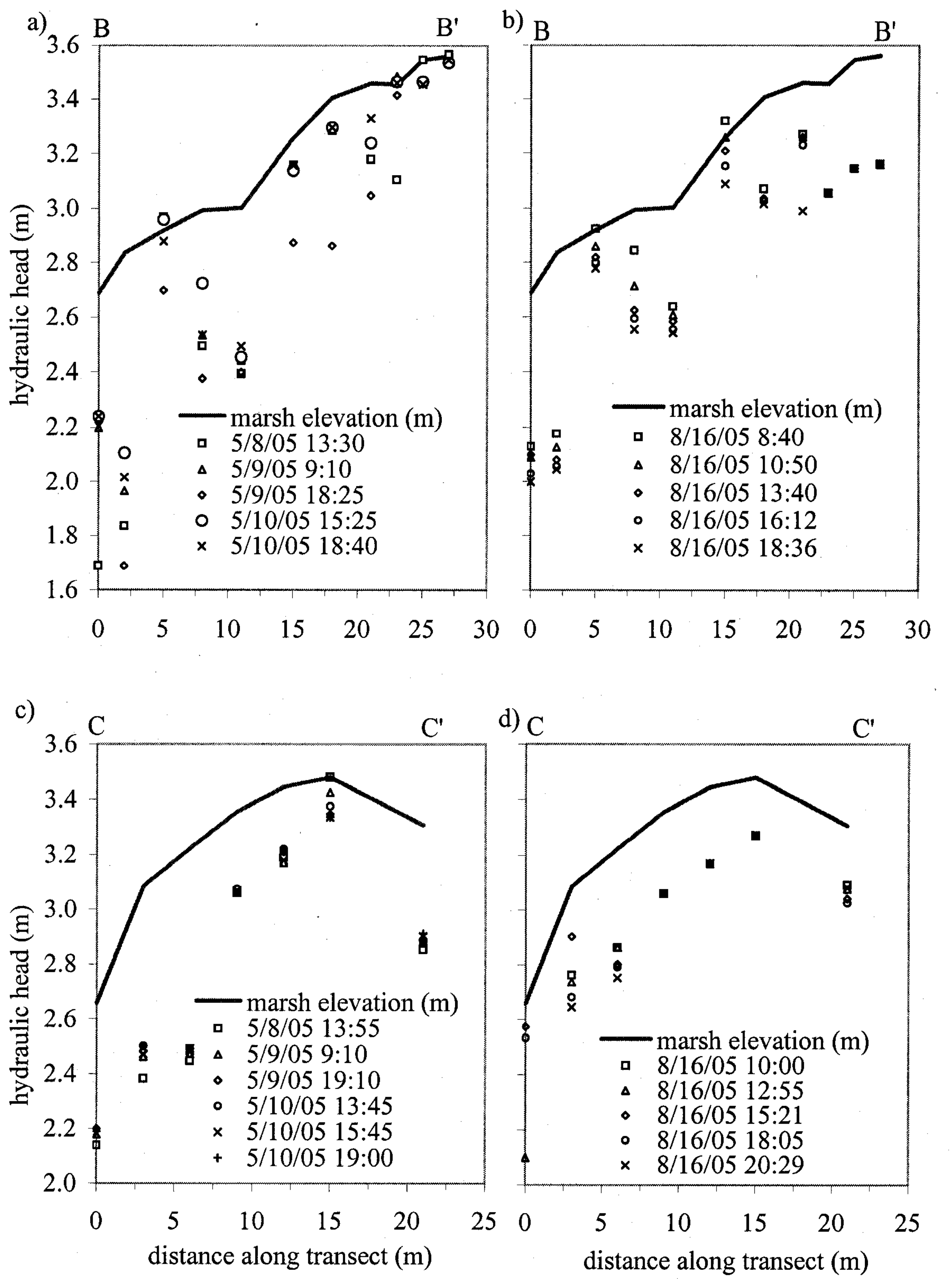

Figure 14. Dipper Harbour hydraulic head measurements along a) transect B in May, b) transect B August 16, c) transect C in May, and d) transect C August 16. Start time is indicated and measurements were completed within 1 hr. MHW at Dipper Harbour is 3.0

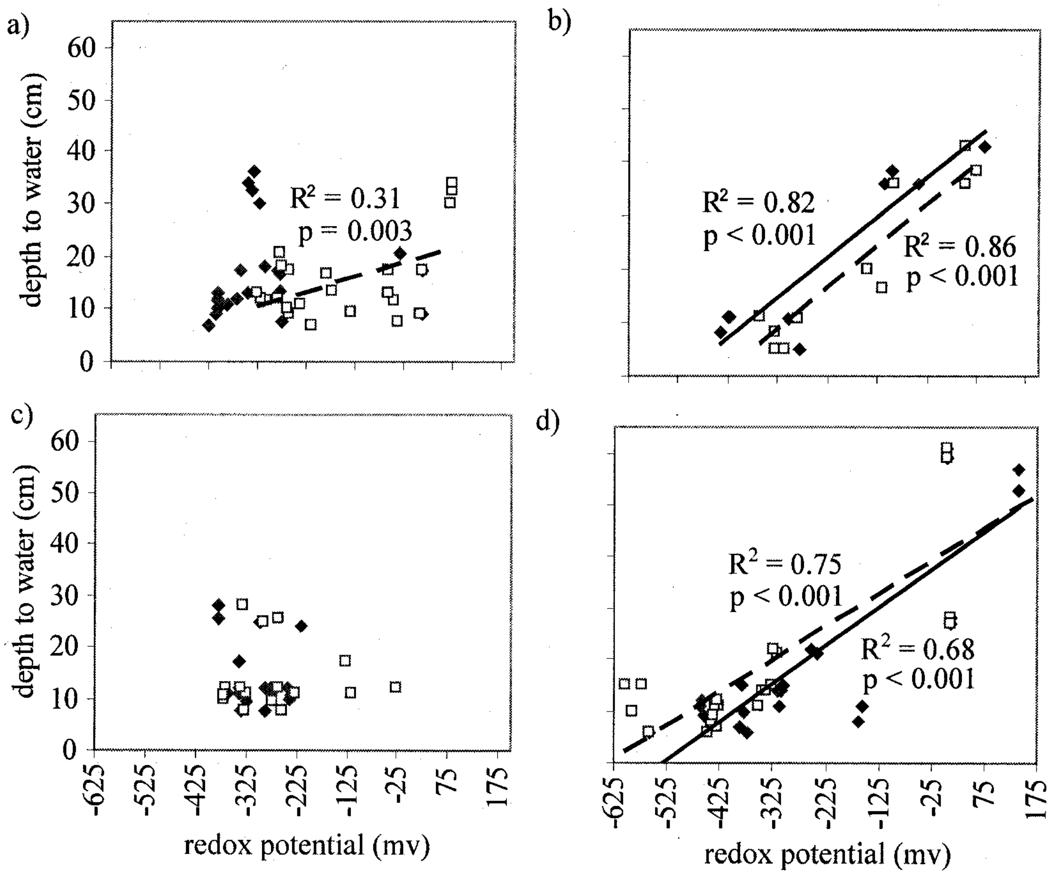

Figure 15. Redox potential at 30 cm (diamond) and 15 cm (square) versus depth to water for a) Dipper Harbour, b) Saint's Rest, c) Wood Point and d) John Lusby. Solid lines represent linear best fit for redox potential at 30 cm versus depth to water. Dashed lines represent linear best fit for redox potential at 15 cm versus depth to water.

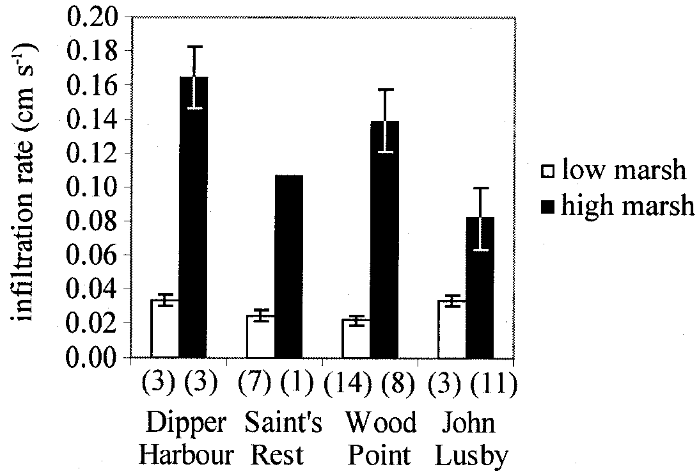

Figure 16. Infiltration rates for high and low marsh locations within each study site. Bars denote $\pm$ 1 standard error. No error bar is included for Saint's Rest high marsh because only one infiltration measurement was made. Numbers in parentheses denote number of infiltration measurments made.

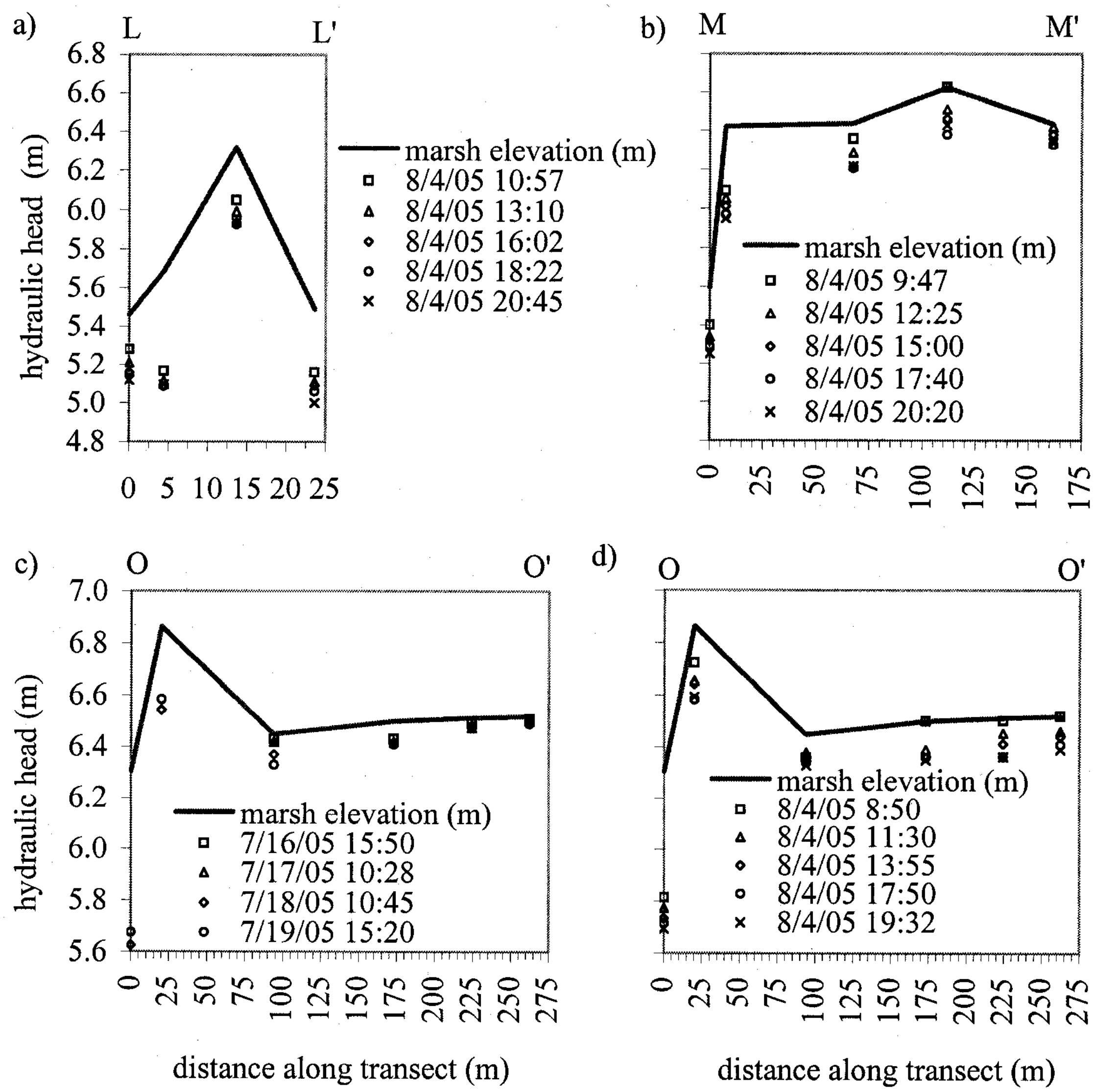

Figure 17. John Lusby hydraulic head measurements along a) transect L August 4, b) transect M August 4, c) transect O in July, d) transect O August 4. Start time is indicated and measurements were completed within 1 hr. MHW at John Lusby is 5.4 m.

Connecting statement

Sub-surface hydrology and vegetation are intimately linked. The functioning of both components allows marshes to provide numerous ecosystem services; thus, restoration projects often monitor sub-surface hydrology and vegetation cover and production. Major findings of chapter 2 are that sub-surface hydrology in Fundy marshes operates differently from organogenic, microtidal marshes and that sub-surface hydrology processes can be restored on half-century time scales. Chapter 3 examines vegetation species composition and end-of-season standing crop at the same study marshes in relation to numerous environmental variables and integrates the observations on sub-surface hydrology reported in chapter 2. The aim of Chapter 3 is to determine which variables drive vegetation and assess the progress of vegetation recovery in two Fundy salt marshes.

Chapter 3: Studies of recovering and reference salt marshes on the Bay of Fundy, II. Vegetation and related environmental variables

Abstract

Since the 17th century large portions of salt marsh in the Bay of Fundy have been converted into dikelands, thus disrupting the interaction among tides, sediments, and salt marsh vegetation. However, salt marshes may recover when dikes are breached during storm events. Investigations of large sites recovering for several decades, such as marshes in this study, can 1) offer information about large-scale, system-wide marsh response and 2) provide insight into factors potentially constraining restoration at other sites. This study investigates the relationships among vegetation species and numerous environmental variables at two recovering and reference salt marsh pairs in the Bay of Fundy. In Fundy marshes, *S. alterniflora* and *S. patens* are inundated less frequently than in microtidal marshes and tolerate large variation in inundation frequency and depth. If restoration guidelines for Fundy marshes include recommendations on inundation frequency, values would need to specifically be developed for this basin. Elevation relative to mean sea level and vertical range of *S. alterniflora* and *S. patens* increase up-Bay with increasing tidal range. By calculating marsh platform elevation at time of dike failure and considering vegetation vertical ranges, it is apparent that salt marsh vegetation could establish on dikelands at the time of dike breech. Therefore, it is unlikely that after dike breech these Fundy dikelands reverted to mudflats or open water, as observed in other regions. Canonical correspondence analysis (CCA) showed that elevation (relative to mean sea level and mean high water) and distance to upland were significant predictors of vegetation cover and end of season standing crop within and between marshes. These results indicate that the most ideal reference sites will not only be located at similar tidal ranges but should also be of similar size. These criteria introduce problems with using the reference site approach in the Bay of Fundy where dikelands are more extensive than marshes not targeted for agriculture and tidal range increases exponentially up-Bay.

Introduction

Human activities in coastal areas during the past several centuries have resulted in the alteration or loss of thousands of square kilometres of tidal salt marsh (e.g., Beeftink 1975; Dale and Hulsman 1990; Allen 2000; Kennish 2001). The Bay of Fundy is no exception – in parts of the Bay up to 85% of salt marshes have been converted into agricultural dikelands via a system of earthen dikes, tide gates (locally termed aboiteau), and ditches since the 17th century (Ganong 1903).

There is growing interest to restore former Bay of Fundy salt marshes by removing dikes and other tidal barriers (Harvey 2000; Gulf of Maine Council Habitat Restoration Subcommittee 2004). Several storm-breach sites present in the Bay of Fundy can serve as analogues for future restoration projects. This study investigates the relationships among vegetation species and numerous environmental variables at two recovering and reference salt marsh pairs in the Bay of Fundy in order to inform future restoration efforts.

As a result of dike and ditch construction, dikelands cease to function as tidal salt marshes and several important changes take place. A disruption in the interaction between tides, sediments, and vegetation decreases both dikeland salinity (Ganong 1903; Daiber 1986; French 2006) and relative elevation (Roman et al. 1984; Allen 1999; Crooks and Pye 2000; Weinstein and Weishar 2002). Since freshwater from precipitation becomes the main source of water for dikelands, salts are eventually leached from the sediment. The decrease in dikeland elevation is due to three factors. First, dikeland sediment compacts due to dewatering and increased rates of decomposition due to a lowered water table. Second, tidally derived sediments are prevented from reaching dikelands and contributing to vertical accretion. Third, over the time period during which marshes have been diked, eustatic sea level has continued to rise.

These changes in salinity and water table have implications for salt marsh vegetation. Zedler et al. (1980) report increased live biomass and productivity at a tidally restricted site, likely due to lowered salinity, among other factors. If ditches provide effective drainage, salt marsh vegetation productivity may briefly increase in response to a lower water table (Balling and Resh 1983) but over time woody species and other terrestrial vegetation may establish, compete with, and replace salt marsh vegetation (Ganong 1903; Roman et al. 1984; Portnoy et al. 1987).

Impacts due to dikes may be lessened or eliminated when dikes are breached or removed entirely. A breach, or gap, in the dike can be created to restore or reactivate salt marshes (Blott and Pye 2004) or unintentional dike breaches can occur during storm events and marshes recover with little or no human influence (Crooks and Pye 2000; Crooks et al. 2002; Eertman et al. 2002; Callaway 2005; French 2006). Investigations of these recovering sites can offer long-term information about vegetation, geomorphology, and recruitment of fauna (Crooks and Pye 2000; Crooks et al. 2002; French 2006) as well as provide insight into what factors could potentially constrain restoration at other sites.

In the Bay of Fundy, there have been vegetation studies of a recovering marsh, John Lusby (Van Zoost 1970; Morantz 1976; Gordon et al. 1985). More recently, vegetation comparisons at Walton River salt marsh (4.95 ha), a recently restored (1 yr) marsh on the Bay of Fundy, and a nearby reference marsh have been made (CB Wetlands & Environmental Specialists 2006a). Musquash marsh (15.38 ha), has also been restored within the past 1 yr (Ducks Unlimited Canada 2004), but monitoring data have not yet been published. Vegetation studies of undisturbed Fundy marshes have been reported by Hatcher and Mann (1975), Palmer (1979), Smith et al. (1980), Gordon et al. (1985), Chmura et al. (1997), and Olsen et al. (2005). Though these studies investigate several environmental variables related to vegetation, none consider a large suite of environmental variables that can be used to assess controls on restoration potential.

A common means to assess progress or 'success' of restoration is by measurement of percent vegetation cover (Sinicrope et al. 1990; Williams and Orr 2002; Blott and Pye

2004) as well as similarity of vegetation cover and biomass between reference and restored sites (Crooks et al. 2002; Thom et al. 2002). In this study I compare the vegetation at two recovering and two reference marshes to assess the progress of recovery. Progress is assessed by calculation of similarity indices based on plant cover and end-of-season standing crop of each marsh pair. Use of similarity indices enables a comparison of recovery rates in Fundy marshes to those in other regions. As such comparisons are dependent upon the selection of appropriate reference sites. I compare environmental variables at each of the four Fundy marshes: surface elevation (both relative to mean sea level and tide levels) and predicted frequency of tidal inundation, as well as other environmental variables (e.g., distance to tidal channel, depth to sub-surface water, soil water salinity). I use multivariate analyses to determine which variables are most important in explaining species cover and end-of-season standing crop at one marsh pair. Using these results I assess the apparent success of marsh vegetation at two recovering Bay of Fundy salt marshes.

Methods

Vegetation and environmental variables were sampled at four Bay of Fundy salt marshes – Dipper Harbour, Saint's Rest, Wood Point, and John Lusby. See Chapter 2 for descriptions and location maps.

Vegetation

In this study, two types of vegetation plots were established in each marsh – cover plots and end-of-season standing crop plots. At John Lusby, I excluded areas behind remnant dikes that were receiving restricted tidal flow.

At cover plots, percent vegetation cover was described in August 2005 at a 1 m^2 plot surrounding each piezometer along transects established for studies of sub-surface hydrology (Chapter 2). Total cover always equalled 100%.

In order to characterize marsh platform elevation and channel locations, and thus aid in locating standing crop plots at varying elevations and distances, several data sets were employed. Transects across the four marsh platforms were recorded with a DGPS. For Wood Point, additional DGPS data collected from previous field campaigns by van Proosdij (2001) and van Proosdij et al. (2006) were used. Tidal channel shapefiles from a concurrent study of these four marshes (Chmura and MacDonald 2006) were used to determine channel location.

Standing crop plots were established in July 2005 at Wood Point and John Lusby, using the above mentioned data sets to select four sites in each marsh of at varying elevations and distances from tidal channels. These plots, 0.25 m^2, were replicated in triplicate, generating a total of 36 end-of-season standing crop plots per marsh. Due to time constraints only percent cover was determined in plots at Dipper Harbour and Saint's Rest marshes, but harvests were conducted at Wood Point and John Lusby. Locations and elevations of plots were recorded with a DGPS and are described in Table 1. Tidal channel shapefiles and aerial photographs were used to determine the distance between plots and the upland, any tidal channel, and deep tidal channels. A tidal channel was defined as 'deep' when the thalweg was > 5m from the lowest edge of channel vegetation, as deep mud deposits on Fundy channel banks present hazards.

A third sample was available at Dipper Harbour marsh - 1 m^2 control plots (i.e., unfertilized) which are part of an on-going study (Chmura unpublished data). Elevation, determined by DGPS, of these six plots was considered when determining species' elevations and tidal inundation at Dipper Harbour in order to increase the sampling size. Table 2 summarizes the data collected from the various plot types at each of the four marshes.

Elevations of three vegetated features reported by Chmura and MacDonald (2006), van Proosdij (2001), and van Proosdij et al. (2006) were analysed. The elevations of bayward edge of vegetation and vegetated edge along primary tidal channels (defined as channels first flooded by incoming flood waters) were measured at the approximate transition between vegetation and mudflat (i.e., the lowest edge of vegetation). The elevations of vegetation at Wood Point and John Lusby cliff edges were measured at the points where vegetation terminated at the top of marsh cliffs.

At Wood Point and John Lusby, vegetation was harvested from 36 plots in each marsh in late August 2005, since peak standing crop occurs at this time. Samples were washed (~6 to 8 times) until water ran clear. Remaining sediment lodged in *S. alterniflora* ligules was removed by hand. Litter was removed from each sample. A plant part was considered 'litter' if no green color was present. Samples were sorted by species, air dried for three months, then weighed.

Elevation and tidal inundation

Elevation relative to mean sea level (MSL) was measured with DGPS and is referenced to Canadian Geographic Vertical Datum 1928 (CGVD28). Mean high water (MHW) and higher high water (HHW) for each marsh were determined using methods described in Canadian Tide and Current Tables (Canadian Hydrographic Service 2005) and additional calculations. Because mean sea level is an intrinsically different measure of water level compared to MHW and HHW, it is important to consider elevations relative to both types of measurements.

MHW and HHW are referenced to chart datum (i.e., a plane below which the tide will seldom fall) and these values were converted to CGVD28 to allow their comparison with DGPS elevations. This conversion involves subtracting mean water level from both the mean tide level and large tide level (Webster et al. 2004). Values for Saint's Rest were based on the Saint John reference port, Dipper Harbour was based on the secondary port of Dipper Harbour West, and the two upper Bay marshes were based on the secondary port of Pecks Point. Wood Point and John Lusby, however, are located 9 and 22 km up-Bay from Pecks Point, respectively, requiring additional corrections to accommodate the ~0.017 m km^{-1} increase in MHW and ~0.029 m km^{-1} increase in HHW towards the head of the Cumberland Basin (Gordon et al. 1985).

I investigated the relationships among zonation of dominant species, elevation relative to mean sea level (MSL) and tide levels, frequency of tidal inundation, and depth of tidal inundation. Only mono-species plots, defined as plots in which 95% or more of the cover or end-of-season standing crop consisted of one species, were considered when determining species' elevations and tidal inundation. Mono-species plots were located in distinct marsh vegetation zones, and avoided problems with isolated patches that contained vegetation species of interest but were not representative of marsh zonation. Frequency and depth of inundation were calculated by comparing predicted elevations (relative to MSL) of high tides at Saint John for 2005, a low point in the 18.6 year tidal cycle, and 1995, a year with a large number of extreme tides, to elevations relative to MSL of various vegetated features and mono-species plots. Corrections for tidal elevation at Dipper Harbour, Wood Point, and John Lusby were made using previously described methods. As tide elevations are reported to the nearest decimeter, it was assumed that a given elevation was flooded when tide elevation equalled or exceeded this elevation.

Sub-surface hydrology and sediment characteristics

Various sub-surface hydrological and sediment parameters were measured in the vegetation plots and the types of data collected are listed in Table 2. Methods for sub-surface hydrological variables measured in cover plots associated with piezometer transects are described in chapter 2. Due to time constraints hydrological and sediment parameters were not measured in plots established at Dipper Harbour and Saint's Rest. At Wood Point and John Lusby, a shallow and deep piezometer pair was established within 0.5 m of each end-of-season standing crop plot using methods described in chapter 2, resulting in 36 piezometer pairs per marsh. Depth to water and groundwater salinity were measured two to three times during one neap tidal cycle in late August 2005 at each piezometer pair using methods described in chapter 2. At each of the 36 Wood Point and John Lusby end-of-season standing crop plots three replicate surface sediment (3 cm deep) cores were collected using a 3.6 cm diameter mini piston corer for a total of 216 cores. Wet and dry mass, after freeze drying, was determined. Bulk density was calculated as dry mass divided by volume and water content was calculated as wet mass minus dry mass divided by dry mass. Loss on ignition (Ball 1964) was used to determine percent organic content. Infiltration was also measured once in each end-of-season standing crop plot at Wood Point and John Lusby using methods described in chapter 2, providing 12 infiltration measurements per marsh.

Statistical analyses

A modified Sorensen's K index (Bray and Curtis 1957; West 1966) was used to estimate similarity in species composition, species cover, and species productivity between sites. Plots located at the marsh upland edge (i.e., those containing *Juncus gerardii* and *Carex palacea*) were excluded from this analysis since plots at Saint's Rest and John Lusby did not include this zone. An unweighted similarity index is calculated: Similarity = (2A/[2A + B +C]) x 100%, where A is the number of species in common between the two sites, B is the number of species exclusive to the first site, and C is the number of species exclusive to the second site. The term '2A' was added to the denominator so that the resulting value was between 0 and 100 (Thom et al. 2002). A weighted Sorensen index of similarity was calculated using cover or end-of-season standing crop. A is the sum of the lower values (cover or end-of-season standing crop) for each species which the two sites have in common, B is the species and summed values (cover or end-of-season standing crop) exclusive to the first site, and C is the species and summed values (cover or end-of-season standing crop) exclusive to the second site.

Vegetation and environmental data from end-of-season standing crop plots at Wood Point and John Lusby (72 plots and 19 variables, Appendix A, Tables 1 and 2) and percent cover plots at all four marshes (99 plots and 15 variables, Appendix A, Tables 3 and 4) were analysed using constrained direct ordination in the form of canonical correspondence analysis (CCA) (CANOCO for Windows v. 4.54, Biometris, The Netherlands, 2006). End-of-season standing crop plots at Dipper Harbour and Saint's Rest and un-fertilized plots (Table 2) were not analysed in CCA because not enough data were available. CCA was chosen as it allows for the simultaneous testing and modelling of multiple independent and dependent variables. Data sets for both vegetation percent cover plots and vegetation end-of-season standing crop plots were split into roughly two equal parts for the initial ordination models. One part was used for building the initial ordination model and the other part was used for testing the initial ordination model. In the final ordination models I included the total number of plots; but included only the environmental variables which had significance levels less than the Boniferroni adjustment at the alpha level and were not collinear with other variables. CANOCO tests for and reports variables that are collinear; however, collinear variables are not automatically removed; this is the responsibility of the user. I used unimodal methods as vegetation data were heterogenous – the longest gradients from detrended correspondence analyses were always greater than 4 (Leps and Smilauer 2003). Hill's scaling was focused on inter-species distances. Vegetation data were log-transformed using the following formula: y' = log(A·y+B) where y is a data value, y' is the result, and A and B are values such that after transformation the result is greater than zero. As percent cover and biomass were sometimes small, 10 was chosen for the value of A and B was given the default value of 1 (Leps and Smilauer 2003). Once the constrained ordination models were constructed, Monte Carlo permutation tests with 1000 permutations were used the test the significance of these models.

Results

Tidal range, elevation, and tidal inundation

Tidal range (Table 3) as well as MHW and HHW levels increase up-Bay (Fig. 1). Dipper Harbour has the lowest levels as it is closest to the mouth of the Bay and John Lusby has the highest levels as it is closest to the head of the Bay. MHW and HHW levels increase ~0.3 m each over the 28 km distance between Dipper Harbour and Saint's

Rest. Wood Point and John Lusby are 13 km apart and MHW and HHW levels increase by 0.2 and 0.4 m, respectively, over this distance.

The differences in elevation among vegetated features (Fig. 1) and selected plant species (Fig. 2) among the four Bay of Fundy marshes are quite striking. Increases in elevation relative to MSL up-Bay are explained by increasing tidal levels. Consider mean marsh platform elevation. Dipper Harbour and Saint's Rest have mean marsh platform elevations near MHW, 3.13 m and 3.65 m, respectively (Fig. 1). On average Dipper Harbour marsh platform is 0.1 m below MHW while Saint's Rest marsh platform is 0.6 m above MHW. Wood Point and John Lusby have mean marsh platform elevations are 5.09 and 6.08 m, respectively, which are 1.5 to 3.6 m higher than the two lower Bay marshes (Fig. 1). While the platform at Wood Point is only 0.1 m below MHW the platform at John Lusby is 1.3 m higher than MHW (Fig. 1).

Despite the fact that the platforms of Dipper Harbour and Saint's Rest are at different positions relative to tidal levels, the mean elevation of the edge of vegetation along primary channels is remarkably similar, 0.5 m below MHW for both marshes (Fig. 1). In contrast, the edge of vegetation along primary channels at Wood Point averages 1.2 m below MHW and at John Lusby is 0.2 m below MHW (Fig. 1).

Though the bayward edges of both upper Bay marshes terminate with cliffs, the Wood Point cliff is vegetated by *S. alterniflora* while the cliff at John Lusby is vegetated mainly by *S. patens*. The cliff vegetation differs between these two marshes because, on average, the cliff elevation is 1.1 m below MHW at Wood Point while at John Lusby the cliff lies, on average, at 0.3 m above MHW (Fig. 1). *S. alterniflora* is present below the cliff at John Lusby and extends towards the Bay - the bayward edge of this vegetation lies at 0.3 m below MHW, on average (Fig. 1).

In mono-species plots, *S. patens* occurred, on average, at higher elevations compared to *S. alterniflora* (Fig. 2), as expected from marsh zonation literature (e.g., Redfield 1972; Chapman 1974). Though mono-species plots of all selected species occurred below HHW, it is clear that their occurrence does not correspond to a consistent elevation relative to either HHW or MHW (Fig. 2).

It is useful to consider data from all vegetation plots (i.e., both mono-species and mixed plots) and DGPS surveys of vegetated features to determine the vertical range over which plant species occur at each marsh. Figure 3 displays the vertical range of *S. alterniflora* and *S. patens*, species for which the most complete data are available, at all four marshes relative to MHW to aid comparison. While the vertical ranges of these species overlap, *S. alterniflora* always occurs at a lower minimum elevation. The data also suggest that vertical range increases with increasing tidal range, except for *S. alterniflora* at John Lusby. The minimum elevation of occurrence for *S. alterniflora* decreases with tidal range while the maximum elevation of occurrence generally increases with tidal range, again the minimum elevation of *S. alterniflora* at John Lusby is an exception to this trend. While the maximum elevation of occurrence for *S. patens* also follows this trend, the minimum elevation of occurrence does not.

The marsh platform and mono-species plots are inundated more frequently, on average, at Dipper Harbour than Saint's Rest (Table 4). Vegetated features and mono-species plots are inundated less often and to a lesser depth at John Lusby. On average, the marsh platform and *S. patens* is flooded the least at John Lusby compared to the other three marshes. In general, frequency and depth of flooding are both greater in 1995 – a year with a large number of extreme tides (Table 4). However there are instances at

Wood Point and John Lusby where inundation frequency is less in 1995 and this is likely due to the fact that tide elevation is only reported to the nearest decimeter.

Sediment characteristics

Though the sediment characteristics of the end-of-season standing crop plots at Wood Point and John Lusby display some spatial variability, they are quite similar between the two marshes. For example, the mean bulk density ($\pm$ S.E.) of sediments at Wood Point is 0.932 ± 0.032 g cm^{-3} and at John Lusby is 0.993 ± 0.038 g cm^{-3}. Mean moisture content ($\pm$ S.E.) is $22.7 \pm 0.8\%$ at Wood Point and $21.5 \pm 1.5\%$ at John Lusby. The mean loss on ignition ($\pm$ S.E.) is $8.2 \pm 0.3\%$ at Wood Point and $8.9 \pm 0.3\%$ at John Lusby.

Vegetation characteristics

Mean plant cover and end-of-season standing crop by marsh vary between reference and recovering marshes (Tables 5 and 6). Plant cover at Dipper Harbour is roughly divided among *S. alterniflora* (29%), *Plantago maritima* (23%), and *S. patens* (24%). In contrast, Saint's Rest is dominated by *S. alterniflora* (86%). *Spartina alterniflora* (53%) dominates at Wood Point, but there is also a high percentage of *S. patens* (34%) and some *Puncinellia* spp. (4%). Though *S. alterniflora* (14%) occurs at John Lusby, this marsh is characterized by high marsh vegetation including an unidentified grass species, U1 (27%), *S. patens* (27%), and *Puccinellia spp.* (24%).

End-of-season standing crop was only measured at upper Bay sites and it paralleled the pattern in cover (Table 6). At Wood Point plots, end-of-season standing crop is roughly split between *S. alterniflora* (190 g m^{-2}) and *S. patens* (159 g m^{-2}).

Spartina patens (178 g m^{-2}) and *Hordeum vulgare* (112 g m^{-2}) dominate the end-of-season standing crop at John Lusby. Other high marsh species which have greater end-of-season standing crop at John Lusby compared to Wood Point include *Puccinellia* spp., *Hordeum jubatum*, *Limonium nashii*, and an unidentified grass (U1).

Similarity indices were used to compare reference and recovering marshes based on cover and end-of-season standing crop (Table 7). The unweighted similarity of Dipper Harbour and Saints Rest cover plots is high because both marshes have many species in common; only five species were absent from either marsh. *Spergularia canadensis*, *Pucinellia spp.*, and *Glaux maritima* are only present in plots sampled at Dipper Harbour while *Festuca rubra* and *Atriplex patula* are only present in plots sampled at Saint's Rest. The weighted similarity of Dipper Harbour and Saint's Rest is closer as the percent cover contributed by species with variable presence is quite low (194%). At Wood Point and John Lusby the unweighted similarity based on cover plots is lower because these marshes have less species in common; nine species were absent from either marsh. *Plantago maritima*, *Atriplex patula*, and *Salicornia europaea* were only present in Wood Point vegetation cover plots. *Triglochin maritima*, *Glaux maritima*, *Hordeum vulgare*, and two unidentified grasses (U1 and U2) were present only in John Lusby vegetation cover plots. The unweighted similarity of the end-of-season standing crop plots at Wood Point and John Lusby was higher than the cover plots as there were more species in common and only eight species were absent from either marsh.

The relationship of environmental variables to plant cover and end-of-season standing crop was examined through CCA (Table 8). Environmental variables (arrows/vectors in CCA biplots, Figs. 4-7) are positively correlated if they point in a similar direction to each other, are not correlated if they are at right angles to each other, and are negatively correlated if they point in opposite directions to each other (Leps and Smilauer 2003). Longer arrows represent environmental variables that have more influence on the distribution of species and samples in ordination space (Leps and Smilauer 2003). The positions of species and samples (i.e., comparing Fig. 4 and 5, comparing Fig. 6 and 7) in ordination space can be interpreted using the biplot rule where species are predicted to have the highest relative frequency in samples closest to them (Leps and Smilauer 2003). Sample and species points can also be projected perpendicular to environmental variable arrows in order to approximate the value of a particular environmental variable in relation to a particular species or sample (Leps and Smilauer 2003). For example, a projection point near zero, the coordinate system origin, corresponds to the average value of a particular environmental variable.

The CCA analysis of end-of-season standing crop at Wood Point and John Lusby reveals that elevation relative to MHW and MSL, minimum depth to water (shallow piezometer), and average ground water salinity (deep piezometer) are all significant and together these variables explain 59% of the variance (Table 8). The CCA reveals that elevation relative to MHW and MSL, distance from upland, and average soil water salinity are all significant and together these variables explain 67% of the variance in plant cover at the four marshes (Table 8). The angle between elevation relative to MSL and MHW (Figs. 6 and 7) indicates that these environmental variables are only weakly correlated. The angle between these same two variables is much smaller when considering data from only Wood Point and John Lusby marshes (Figs. 4 and 5), indicating that these variables have a stronger correlation when there is a smaller difference in tidal range between sites.

Discussion

Dike breaching (or removal) results in a rapid return of tidal water and suspended sediments. Unless the breach dimensions are limiting, a breach site has a longer hydroperiod since, relative to undiked marshes, it is low in the tidal frame (Allen 2000; Williams et al. 2002). Rapid accretion of mineral matter and the formation of tidal flats occurs at breach sites with long hydroperiods and minimal erosion (Cahoon et al. 2000; Crooks and Pye 2000; Eertman et al. 2002; Williams and Orr 2002; Blott and Pye 2004). Over time accretion rates decrease as the marsh increases its elevation in the tidal frame (Allen 1997).

Site elevation at the time of the breach strongly influences the vegetation response and its rate of recovery. A site initially lower in the tidal frame takes longer to reach an elevation suitable for establishment of salt marsh vegetation compared to a site initially higher (Williams and Orr 2002). Some sites much lower in the tidal frame have remained as open water or tidal flats for extended periods (Weinstein and Weishar 2002; Williams and Orr 2002; Blott and Pye 2004). My results from the Bay of Fundy confirm that site elevation relative to tidal levels, as well as other environmental variables, is important to vegetation response.

Elevation, tidal range, inundation, and other environmental variables

Since tidal range in the Bay of Fundy increases exponentially at the rate of 0.36% per km (Desplanque and Mossman 2004), it is extremely difficult to select a reference site with a tidal range and MHW level identical to a restored/recovering site. In this study, both marsh pairs exhibit differences in these two parameters. Differences in elevation relative to MHW explain a significant amount of variance within marshes (Table 8).

Elevation relative to MSL of salt marsh vegetation (Fig. 2) generally increased up-Bay in association with increasing high tide elevations, a result expected from the observations of Olsen et al. (2005), Gordon et al. (1981), and Palmer (1979). Results from Dipper Harbour, Saint's Rest and Wood Point (Fig. 3) are consistent with observations (Adams 1962; Redfield 1972; McKee and Patrick 1988) that the vertical range of *S. alterniflora* is increases with tidal range. However, I believe this is the first study to demonstrate that *S. patens* may also display this same trend. The smaller vertical range of *S. alterniflora* at John Lusby may represent the point where physical stress prevents it from growing at lower elevations relative to MHW.

The number of times a marsh is inundated is crucial for both sediment delivery and vegetation establishment/zonation. Harvey and Odum (1990) indicate that an *S. alterniflora* marsh is inundated more than 675 times per yr and Blum (1968) notes that *S. patens* is flooded 88 times per yr. Both of these values are based on marshes with semi-diurnal tides. While similar or higher inundation frequencies occur at low elevations of Fundy marshes, high elevations within these marshes are flooded much less frequently (Table 4). In general, as tidal range increases inundation frequency at high elevations within Fundy marshes decreases (Desplanque and Mossman 2004). For example, of the four marshes studied, John Lusby has the largest tidal range and *S. patens* mono-species plots in this marsh are inundated the least compared to other marshes. If restoration guidelines for Fundy marshes include recommendations on inundation frequency, as has been done for marshes elsewhere (Toft and Maddrell 1995 in French 2006), then values would need to specifically be developed for this basin and take into account that *S. alterniflora* and *S. patens* can tolerate large variation in inundation frequency and depth.

Morris et al. (2005) hypothesize that the distribution of salt marsh elevation relative to MHW is diagnostic of marsh stability in the face of relative sea-level rise. Specifically, highly stable marshes will have mean elevations approximately equal to MHW and well above MSL or the lower limit of vegetation growth (Morris et al. 2005). The Bay of Fundy marshes included in this study all have mean platform elevations at or above MHW (Fig. 1) and minimum platform elevations are much greater than the lower limit for *S. alterniflora* (Fig. 1 and Fig. 3). These findings suggest that in the vertical plane the marshes studied are very stable and are not imminently threatened by rising relative sea level.

In addition to relative elevation, other factors affect vegetation establishment, zonation, and productivity. French (2006) reported that in several UK breach sites, rapid sedimentation has buried previous vegetation and anoxic layers have formed, preventing the establishment of salt marsh vegetation. Other sediment characteristics related to salt marsh vegetation cover and restoration rates include soil water salinity and water table levels (Burdick et al. 1997; Roman et al. 2002; Warren et al. 2002). Channel type and density are additional factors to consider. In San Francisco Bay, sites located along interior channels far from the bay had lower suspended sediment concentrations thus inhibiting sediment accretion and vegetation establishment (Williams and Orr 2002). Eertman et al. (2002) suggested that rapid colonization of mudflats in a Dutch marsh was due to increased drainage provided by a constructed channel. Even after restoration or recovery, unrestricted tidal flow may not be present at a site due to inadequate breach or culvert dimensions. Numerous studies have documented that in this situation vegetation establishment and zonation is delayed and possibly prevented (Burdick et al. 1997; Thom et al. 2002; Williams and Orr 2002).

In this study, results from canonical correspondence analysis (CCA) provide insight into which variables drive vegetation differences in the Bay of Fundy, and thus, should be considered when comparing restored/recovering sites to reference sites. Elevation relative to MHW and MSL, distance to upland, soil water salinity, and minimum depth to water in shallow piezometers were all significant (Figs. 4 and 5). Because elevation relative to MHW largely determines tidal inundation, salt marsh vegetation zonation broadly reflects elevation gradients (e.g., Chapman 1974; Mitsch and Gosselink 2000). Elevation relative to MSL and distance from upland are also good predictors because each marsh in this study is located at different elevations above MSL and different distances from the upland (controlled by marsh size). Similar to Sanchez et al. (1998), this study also found that ground water salinity predicted vegetation cover. Sub-surface hydrology explained variation in species production between Wood Point and John Lusby but did not explain percent cover of all four marshes (Table 8).

Similarity measures and recovery rates

Weighted similarity measures provide a simple metric for comparing suites of recovering and reference marsh pairs. Results from the Bay of Fundy indicate that marsh pairs are as similar or more similar compared to an Elk River (Washington, US) marsh pair but less similar compared to marsh pairs in the UK (Table 7). The Elk River (Washington, US) study showed that similarity between the recovering and reference site increased rapidly after the breach and then levelled off with time with some degree of between-year variation (Thom et al. 2002).

A number of studies have examined the time period required for recovery of salt marsh vegetation in non-constructed marshes. Just one year after restoration of a Rhode

Island marsh, there were significant changes in vegetation abundance and dissimilarity measures suggest a convergence towards typical salt marsh vegetation (Roman et al. 2002). Five years post recovery, Eertman et al. (2002) found that large areas of mudflat were covered with marsh vegetation in a Dutch salt marsh. Sinicrope et al. (1990) demonstrated that a Connecticut marsh contains typical salt marsh vegetation 10 yr post-restoration. Williams and Orr (2002) show that in San Francisco Bay, a vegetated marsh platform (defined as 50% cover) is achieved within less than 5 yr to more than 20 yr, depending on site conditions. In rapidly restored Connecticut salt marshes, salt marsh vegetation became established at rates of 5% of total area per yr, suggesting that restoration there takes approximately 20 yr (Warren et al. 2002). Crooks et al. (2002) found that vegetation of recovering sites in the UK over 100 yr old matched that of reference sites, concluding that recovery can be complete in less than a century.

This study of sites recovering for a relatively long time, together with more recent restoration projects on the Bay of Fundy (Ducks Unlimited Canada 2004; CB Wetlands & Environmental Specialists 2006a), represent the first steps towards developing restoration trajectories for Fundy salt marshes. My results indicate that former Bay of Fundy salt marshes can become re-vegetated with salt marsh species with little to no human influence. Half a century after dike breach, these recovering marshes do not consist primarily of mudflats (Williams and Orr 2002) or deep open water (Weinstein and Weishar 2002; Blott and Pye 2004) but instead contain vegetated marsh platforms.

Considerations for restoration

The apparent success in Bay of Fundy marsh growth following dike breach is related to several factors – elevation at time of breach, vegetation growth range, and rapid rates of sediment deposition. Sediment cores from Saint's Rest indicate that the reclamation surface is located at depths of 11 to 21 cm in the *S. patens* dominated high marsh zone (Noel et al. 2005). This result together with the average elevation above MSL of *S. patens* (3.85 m) means that areas at Saint's Rest that are high marsh today had elevations above MSL of 3.64 to 3.74 m when the dike breached. Taking into account that sea level was 20 cm lower in 1950 (Desplanque and Mossman 2004), the marsh surface at core locations (i.e., 3.44 to 3.54 m) was well within the elevation range of *S. alterniflora* (1.53 to 3.51 m, corrected for 1950 MSL). Considering minimum elevations of *S. alterniflora* at Saint's Rest and sea level rise, the reclamation surface could have been as much as 2.3 m below current day high marsh elevations and colonized by *S. alterniflora*.

A sediment core from John Lusby indicates that the reclamation surface is located at 100 to 130 cm below the *S. patens* dominated high marsh zone (Graf 2004). This result together with the average elevation above MSL of *S. patens* (6.66 m) means that areas at John Lusby that are high marsh today had elevations above MSL of 5.36 to 5.66 m when the dike breached or 5.45 to 5.15 m when adjusted for sea level change since 1947. The marsh surface at core locations is well within the elevation range of *S. alterniflora* (4.19 to 6.44 m, corrected for 1947 MSL). Considering minimum elevations of *S. alterniflora* at John Lusby and sea level rise, the reclamation surface could have been as much as 2.1 m below current day high marsh elevations and colonized by *S. alterniflora*.

Therefore, Saint's Rest and John Lusby were high enough in the tidal frame at the time of breach to be colonized by *S. alterniflora* without first reverting to mudflat, a finding that corroborates macrofossil results from the Saint's Rest cores (Noel et al. 2005). Both Saint's Rest and John Lusby had high enough relative elevations despite being deprived of tidal sediments for 140 – 217 yr and 261 – 254 yr, respectively. Therefore, the large growth range of *S. alterniflora* (which increases with tidal range) make Bay of Fundy marshes resilient in the face of diking and ditching. At proposed restoration sites a similar analysis of reclamation surface elevation relative to the minimum elevation at which *S. alterniflora* grows for a given tidal range can be used to predict the site's response.

Addition of fill is recommended to enhance marsh restoration in some regions, but is unlikely to be required to restore Fundy marshes. Though Saint's Rest and John Lusby were lower in the tidal frame when they breached, high rates of sediment deposition – an average of 2 cm yr^{-1} at nearby Dipper Harbour low marsh and 3.5 cm yr^{-1} at nearby Wood Point low marsh (Chmura et al. 2001) – likely allowed them to quickly gain elevation. In contrast, due to the low suspended sediment concentration of the Delaware estuary in New Jersey, Weinstein and Wieshar (2002) recommend using fill prior to dike breaching to ensure that restored marshes obtain elevations high enough for vegetation establishment.

Similarity indices indicate differences in vegetation cover and end-of-season standing crop between recovering and reference marsh pairs. Elevation relative to MSL and tidal levels and distance to upland (driven in part by marsh size) partly explain these differences. These results indicate that the most ideal reference sites will not only be located at similar tidal ranges but should also be of similar size. However, it is difficult, if not impossible to locate reference sites on the Bay of Fundy which meet these criteria. First, dikelands are more extensive than marshes not targeted for agriculture. Second, as tidal range increases exponentially up-Bay, only sites directly adjacent to each other experience a similar tidal range. Some researchers have attempted to solve the adjacency issue by choosing reference marshes located seaward of dikes (e.g., Boumans et al. 2002; Crooks et al. 2002; Thom et al. 2002). Though marshes seaward of dikes appear undisturbed since they support salt marsh vegetation and experience unrestricted tidal flooding, they contain truncated, less-sinuous channels and thus altered channel habitats and surface hydrology (Hood 2004). In addition, reference marshes located seaward of dikes are further from the upland than restored sites located landward of former dikes. This study suggests that in this case vegetation differences could be a result of distance to upland rather than differences in site type (i.e., reference vs. recovering). In light of these findings, the reference site approach is of limited use in the Bay of Fundy. Instead, intensive studies of several Fundy dikelands documenting and comparing vegetation change over time, from pre-breach to many years post-breach, may yield more valid information.

Table 1. Description of end-of-season standing crop plots at Wood Point and John Lusby. A tidal channel was defined as 'deep' when the channel thalweg was > 5m from the lowest edge of vegetation.

Marsh	Plots	Distance to			Elevation relative to		Area
		deep channel	any channel	upland	MSL	MHW	
		m			m		m^2
Wood Point	1 - 3	5 - 15	5 - 15	214 - 219	5.1 - 5.2	-0.1 - 0.0	834
	4 - 6	48 - 60	25 - 34	133 - 142	5.3 – 5.5	0.1 – 0.3	1254
	7 - 9	97 - 109	26 - 38	30 - 41	5.8 – 6.1	0.6 – 0.8	2926
	10 - 12	3 - 13	6 -13	149 - 155	5.9	0.7	229
John Lusby	13 - 15	5 - 15	2 - 10	393 - 403	5.7 – 6.8	0.2 – 1.3	300
	16 - 18	4 - 11	2 - 4	442 - 451	6.5 – 6.7	1.0 – 1.3	1660
	19 - 21	81 - 115	81 - 115	596 - 625	6.6 – 6.7	1.2 – 1.3	156
	22 - 24	17 - 44	9 - 22	319 - 410	6.7	1.3	1770

Table 2. Data collected from Bay of Fundy C = cover, E = end-of-season standing crop, and U = un-fertilized plots part of on-going study (Chmura unpublished data). X denotes variables included in initial canonical correspondence analysis (CCA), excluding end-of-season standing crops at Dipper Harbour and Saint's Rest, un-fertilized plots, and infiltration measurements since limited data are available.

	Dipper Harbour			Saint's Rest		Wood Point		John Lusby	
	C	E	U	C	E	C	E	C	E
Vegetation									
percent cover	X	X	X	X	X	X		X	
end of season standing crop							X		X
Location and elevation									
distance to nearest channel	X			X		X	X	X	X
distance to deep channel	X			X		X	X	X	X
distance to upland	X			X		X	X	X	X
elevation relative to MSL	X	X	X	X	X	X	X	X	X
elevation relative to MHW	X	X	X	X	X	X	X	X	X
Sediment									
infiltration	X			X			X		X
bulk density							X		X
organic content							X		X
water content							X		X
Sub-surface hydrology - piezometer transects									
average depth to water (August)	X			X		X		X	
range in depth to water (August)	X			X		X		X	
minimum depth to water (August)	X			X		X		X	
maximum depth to water (August)	X			X		X		X	
range in depth to water (field season)	X			X		X		X	
minimum depth to water (field season)	X			X		X		X	
maximum depth to water (field season)	X			X		X		X	
average salinity (field season)	X			X		X		X	
minimum salinity (field season)	X			X		X		X	
maximum salinity (field season)	X			X		X		X	
Sub-surface hydrology - paired piezometers									
presence/absence of water (shallow)							X		X
average depth to water (shallow)							X		X
range in depth to water (shallow)							X		X
minimum depth to water (shallow)							X		X
maximum depth to water (shallow)							X		X
average depth to water (deep)							X		X
range in depth to water (deep)							X		X
minimum depth to water (deep)							X		X
maximum depth to water (deep)							X		X
average salinity (shallow)							X		X
average salinity (deep)							X		X

Table 3. Tidal range at each of the Bay of Fundy marshes. Elevation above mean sea level (MSL), referenced to CVGD28.

	Tidal range (m)	
	Mean tides	Large tides
Dipper Harbour	6.0	8.0
Saint's Rest	6.7	8.9
Wood Point	10.5	14.7
John Lusby	11.0	15.1

Table 4. Inundation frequency per year and depth for various vegetated features and mono-species plots based on predicted high tides in 2005 and 1995. Numbers denote mean (maximum, minimum). 1° channel edge and bayward edge refer to the approximate transition between vegetation and mudflat (i.e., the lowest edge of vegetation). Cliff edge refers to vegetation termination at the top of marsh cliffs. *Sa* = *Spartina alterniflora*, *Sp* = *Spartina patens*, *Hv* = *Hordeum vulgare*, and *Pm* = *Plantago maritima*. Depths of 0 m indicate that features/species were inundated less than 0.1 m since tidal elevations were reported to the nearest decimeter.

Feature/ Species	Inundation frequency per year 2005		1995		Inundation depth (m) 2005		1995	
Dipper Harbour								
1° channel edge	446	(661, 206)	534	(693, 348)	0.4	(0.7, 0.4)	0.5	(0.8, 0.3)
platform	128	(589, 0)	186	(648, 0)	0.2	(0.6, 0.0)	0.2	(0.7, 0.0)
Sa	128	(325, 70)	186	(430, 80)	0.3	(0.3, 0.2)	0.2	(0.4, 0.2)
Sp	46	(46, 70)	53	(80, 53)	0.1	(0.2, 0.1)	0.1	(0.2, 0.1)
Pm	128	(206, 70)	186	(348, 80)	0.2	(0.4, 0.2)	0.2	(0.3, 0.2)
Saint's Rest								
1° channel edge	446	(705, 18)	571	(704, 28)	0.5	(1.3, 0.2)	0.5	(1.5, 0.1)
platform	70	(258, 10)	126	(408, 1)	0.3	(0.4, 0.0)	0.2	(0.4, 0.0)
Sa	100	(258, 70)	171	(408, 80)	0.3	(0.4, 0.2)	0.2	(0.4, 0.2)
Sp	18	(33, 18)	28	(46, 28)	0.2	(0.2, 0.2)	0.0	(0.1, 0.0)
Pm	18	(33, 18)	28	(46, 28)	0.2	(0.2, 0.2)	0.0	(0.1, 0.0)
Wood Point								
Cliff edge	701	(705, 661)	704	(704, 693)	0.9	(2.0, 0.7)	1.1	(2.2, 0.8)
1° channel edge	705	(705, 589)	704	(704, 648)	1.0	(1.8, 0.6)	1.2	(2.2, 0.7)
platform	325	(705, 0)	430	(704, 0)	0.3	(1.5, 0.0)	0.4	(1.7, 0.0)
Sa	325	(631, 128)	430	(685, 186)	0.3	(0.6, 0.2)	0.4	(0.7, 0.2)
Sp	18	(70, 2)	17	(80, 0)	0.1	(0.2, 0.0)	0.1	(0.2, 0.0)
John Lusby								
bayward edge	631	(705, 401)	685	(704, 485)	0.6	(1.0, 0.4)	0.7	(1.2, 0.4)
cliff	325	(549, 128)	430	(617, 186)	0.3	(0.5, 0.2)	0.4	(0.6, 0.2)
1° channel edge	589	(705, 158)	648	(704, 241)	0.6	(1.2, 0.3)	0.7	(1.4, 0.3)
platform	10	(499, 0)	1	(580, 0)	0.0	(0.5, 0.0)	0.0	(0.5, 0.0)
Sa	206	(446, 18)	293	(534, 17)	0.3	(0.4, 0.1)	0.3	(0.5, 0.1)
Sp	10	(46, 2)	1	(53, 0)	0.0	(0.2, 0.0)	0.0	(0.1, 0.0)
Hv	10	(33, 2)	1	(35, 0)	0.0	(0.1, 0.0)	0.0	(0.1, 0.0)

Table 5. Mean percent cover (± 1 S.E.) of plots at Bay of Fundy marshes.

	Dipper Harbour (n = 24)	Saint's Rest (n = 23)	Wood Point (n = 30)	John Lusby (n = 22)
S. alterniflora	28.7 ± 7.9	86.3 ± 4.9	47.3 ± 8.2	13.6 ± 7.5
Spartina patens	23.9 ± 5.5	4.1 ± 4.1	0.7 ± 0.5	26.5 ± 8.9
Puccinellia spp.	0.2 ± 0.2		3.5 ± 1.4	23.9 ± 8.4
Plantago maritima	23.4 ± 5	1.7 ± 1.0	0.7 ± 0.5	
Bare ground	4.0 ± 2.2	2.6 ± 1.8	5.0 + 2.1	0.5 ± 0.5
Juncus gerardii	4.9 ± 2.4	< 0.05	3.3 ± 3.3	< 0.05
Carex palascea	3.5 ± 2.5	< 0.05	2.3 ± 2.3	< 0.05
Unidentified 1				27.0 ± 8.8
Triglochin maritima	1.1 ± 0.6	1.0 ± 0.6		2.5 ± 1.5
Glaux maritima	2.6 ± 6.6			1.4 ± 1.0
Sueda maritima	1.6 ± 1.1	1.6 ± 0.7		
Salicornia europea	0.7 ± 0.4	1.4 ± 0.7	0.1 ± 0.0	
Limonium nashii	1.2 ± 0.6	< 0.05		
Hordeum vulgare				2.3 ± 2.3
Festuca rubra		1.3 ± 1.3		
Rock	0.3 ± 0.3			
Atriplex patula		< 0.05	< 0.05	
Spergularia canadensis	< 0.05			

Table 6. Mean ($\pm$ 1 S.E.) end-of-season standing crop (g dry wt m^{-2}) from plots ($n = 36$) at upper Bay of Fundy study sites.

	Wood Point	John Lusby
Sparina alterniflora	190.1 $\pm$ 29.7	76.5 $\pm$ 27.7
Spartina patens	159.3 $\pm$ 39.3	178.1 $\pm$ 30.8
Hordeum vulgare		112.0 $\pm$ 43.9
Unidentified 1	4.0 $\pm$ 1.8	7.4 $\pm$ 2.1
Limonium nashii	1.2 $\pm$ 1.2	4.1 $\pm$ 2.4
Hordeum jubatum		5.7 $\pm$ 2.3
Pucinellia spp.	0.5 $\pm$ 0.4	5.3 $\pm$ 2.3
Unidentified 2		5.3 $\pm$ 3.2
Salicornia europea	0.7 $\pm$ 0.3	< 0.05
Unidentified 3		0.6 $\pm$ 0.4
Atriplex patula	0.4 $\pm$ 0.4	0.4 $\pm$ 0.2
Sueda maritima	0.1 $\pm$ 0.1	
Triglochin maritima	0.4 $\pm$ 0.4	
Unidentified 4		< 0.05

Table 7. Weighted and unweighted similarity between reference and recovering/restored sites. C = cover plots, E = end-of-season standing crop plots. Similarity for Crooks et al. (2002) calculated using data provided in their Table 2.

Marsh pairs	% similarity				Reference
	unweighted		weighted		
	C	E	C	E	
					this study
Dipper Harbour and Saint's Rest, Canada	74		90		
Wood Point and John Lusby, Canada	47	60	71	85	"
Northey Island reference and recovering, UK		82		96	Crooks et al. 2002
North Fambridge reference and recovering, UK		82		95	Crooks et al. 2002
Elk River reference and restored (year 4), USA	42		32		Thom et al. 2002
Elk River reference and restored (year 6), USA	52		78		Thom et al. 2002
Elk River reference and restored (year 11), USA	78		47		Thom et al. 2002

Table 8. Environmental variables included in final canonical correspondence analysis (CCA) models of cover (C) and end-of-season standing crop (E) plots. The significance and percent variance independently explained are reported for variables included in final models.

Variables	C p	C independent % variance	E p	E independent % variance
distance to upland	0.001	25		
elevation relative to MSL	0.004	26	0.001	31
elevation relative to MHW	0.001	27	0.001	34
average salinity (field season)	0.001	17		
minimum depth to groundwater (shallow)			0.001	14
average salinity (deep)			0.001	17
Percent variance explained by all variables included in final model		67		59

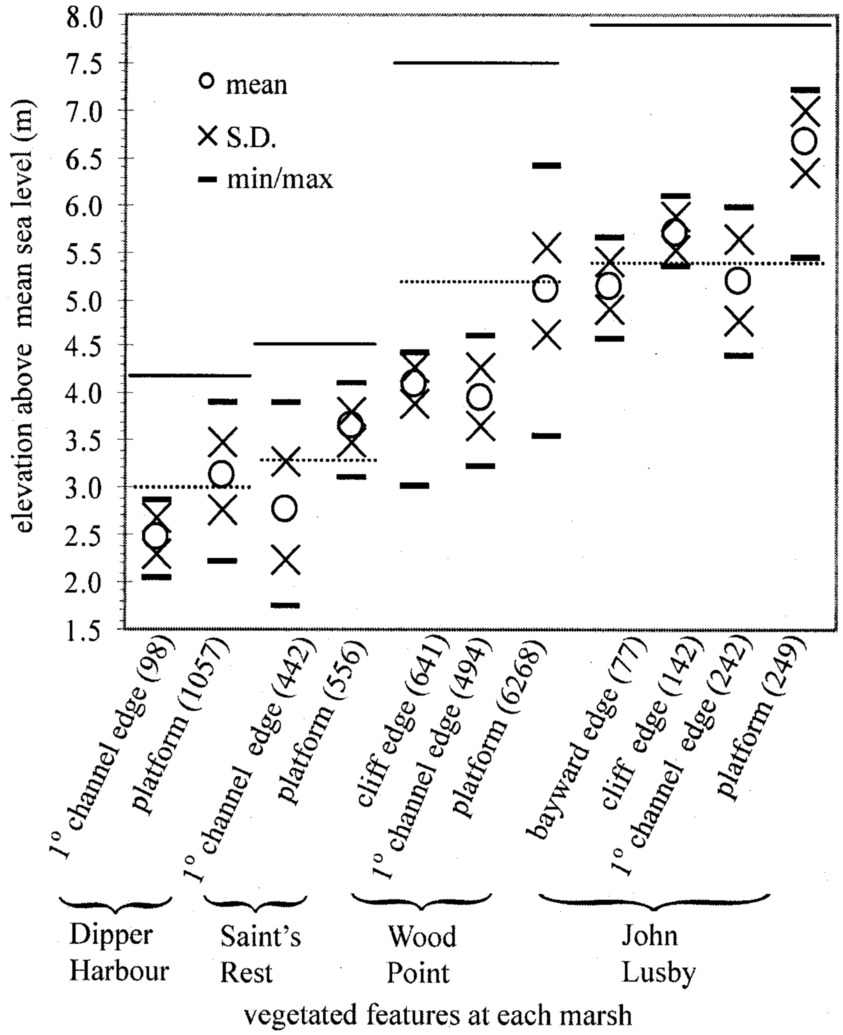

Figure 1. Elevation relative to mean sea level (referenced to CGVD28) of vegetated features at marsh study sites. Dotted lines represent mean high water (MHW) and solid lines represent higher high water (HHW). 1° channel edge and bayward edge refer to the approximate transition between vegetation and mudflat (i.e., the lowest edge of vegetation). Cliff edge refers to vegetation termination at the top of marsh cliffs. Numbers in parentheses indicate the number of measurements available.

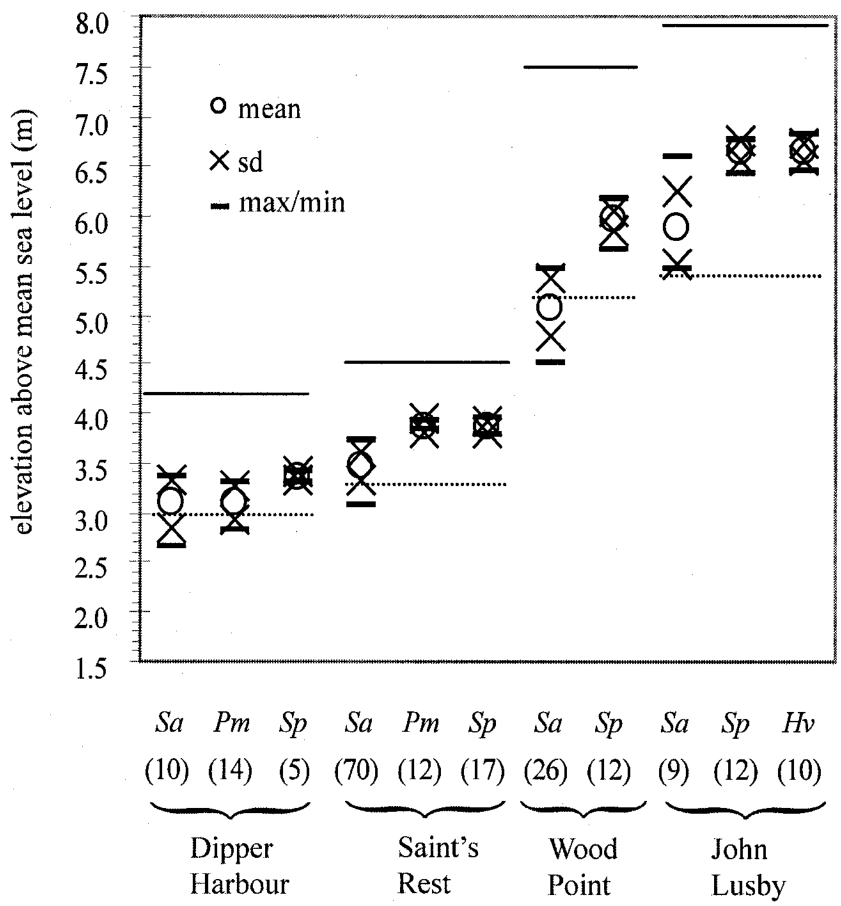

Figure 2. Elevation relative to mean sea level (referenced to CGVD28) of selected species in mono-species plots at marsh study sites. Dotted lines represent mean high water (MHW) and solid lines represent higher high water (HHW). *Sa* = *Spartina alterniflora*, *Sp* = *Spartina patens*, *Hv* = *Hordeum vulgare*, and *Pm* = *Plantago maritima*. Numbers in parentheses indicate the number of measurements available.

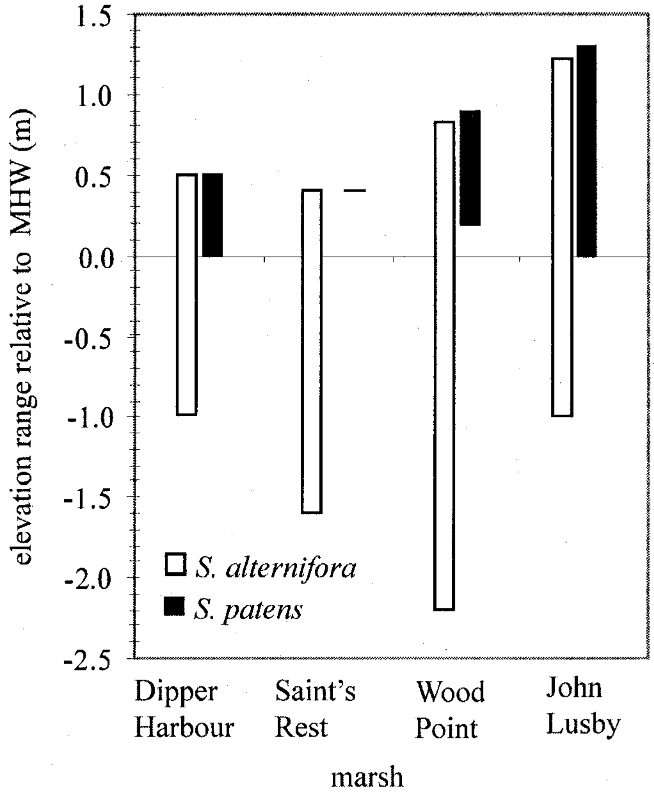

Figure 3. Elevation range of *Spartina alternifora* and *Spartina patens* relative to mean high water (MHW). These ranges are based on data from all vegetation plots (i.e., both mono-species and mixed plots) and DGPS surveys of vegetated features. Only the lower edge of *Spartina patens* was sampled at Saints Rest and it is depicted with a dash.

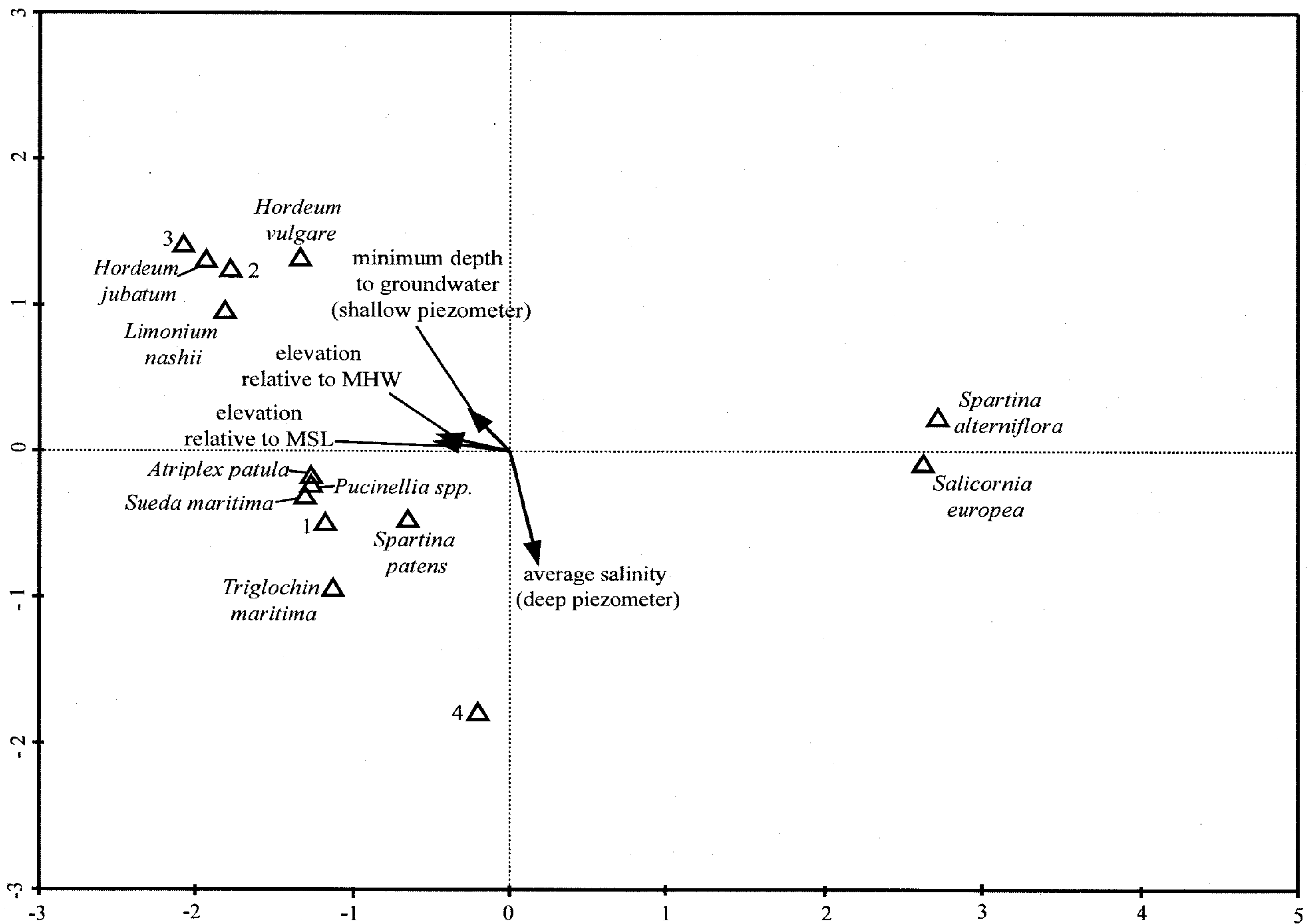

Figure 4. Vegetation end-of-season standing crop species and environmental variables biplot. Species composition is detailed in Table 1, Appendix A. Numbers denote unidentified species.

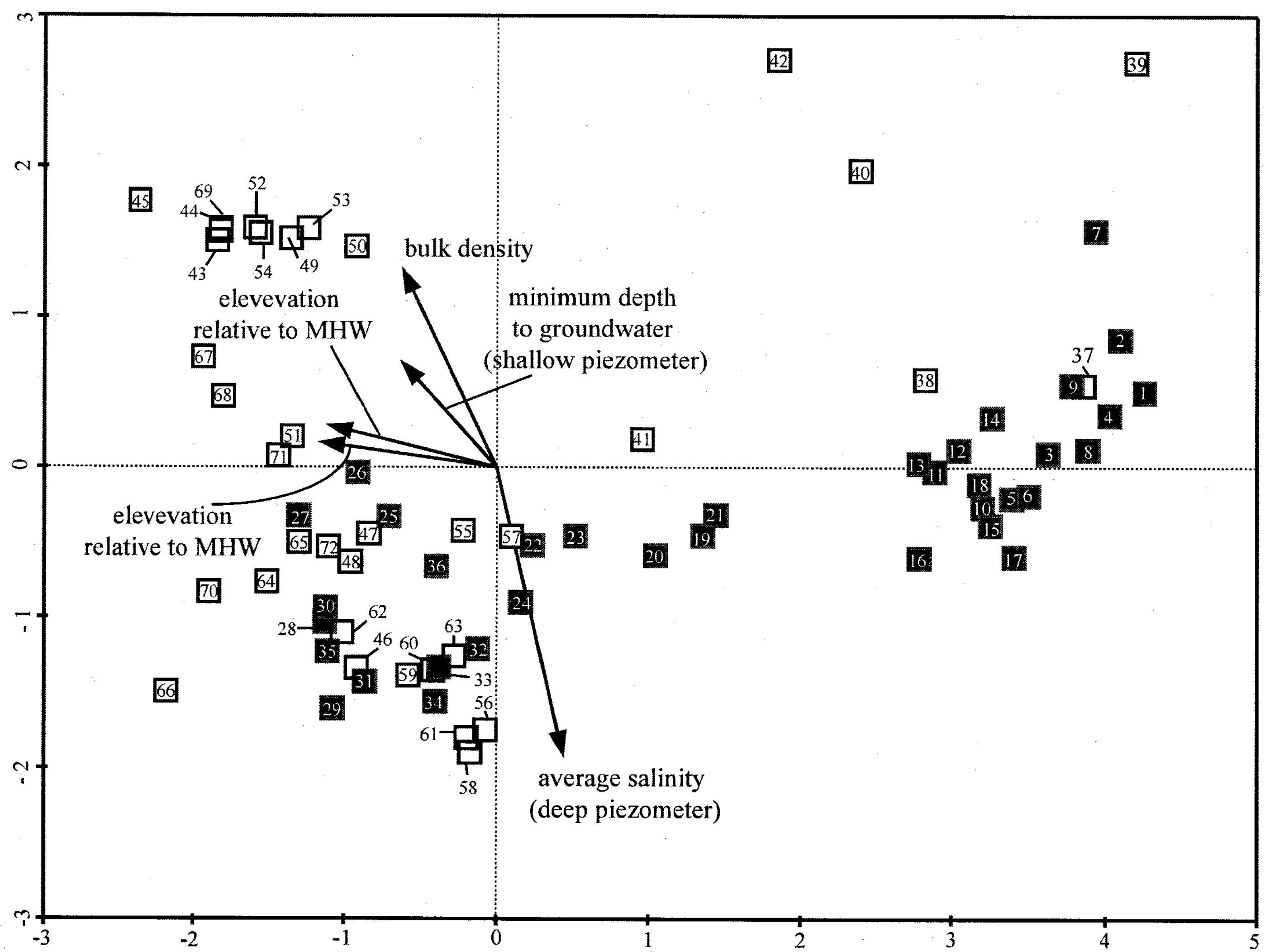

Figure 5. Vegetation end-of-season standing crop samples and environmental variables biplot. Wood Point (■) and John Lusby (□) samples. Numbers are unique identifiers of plots, listed in Tables 1 and 2, Appendix A.

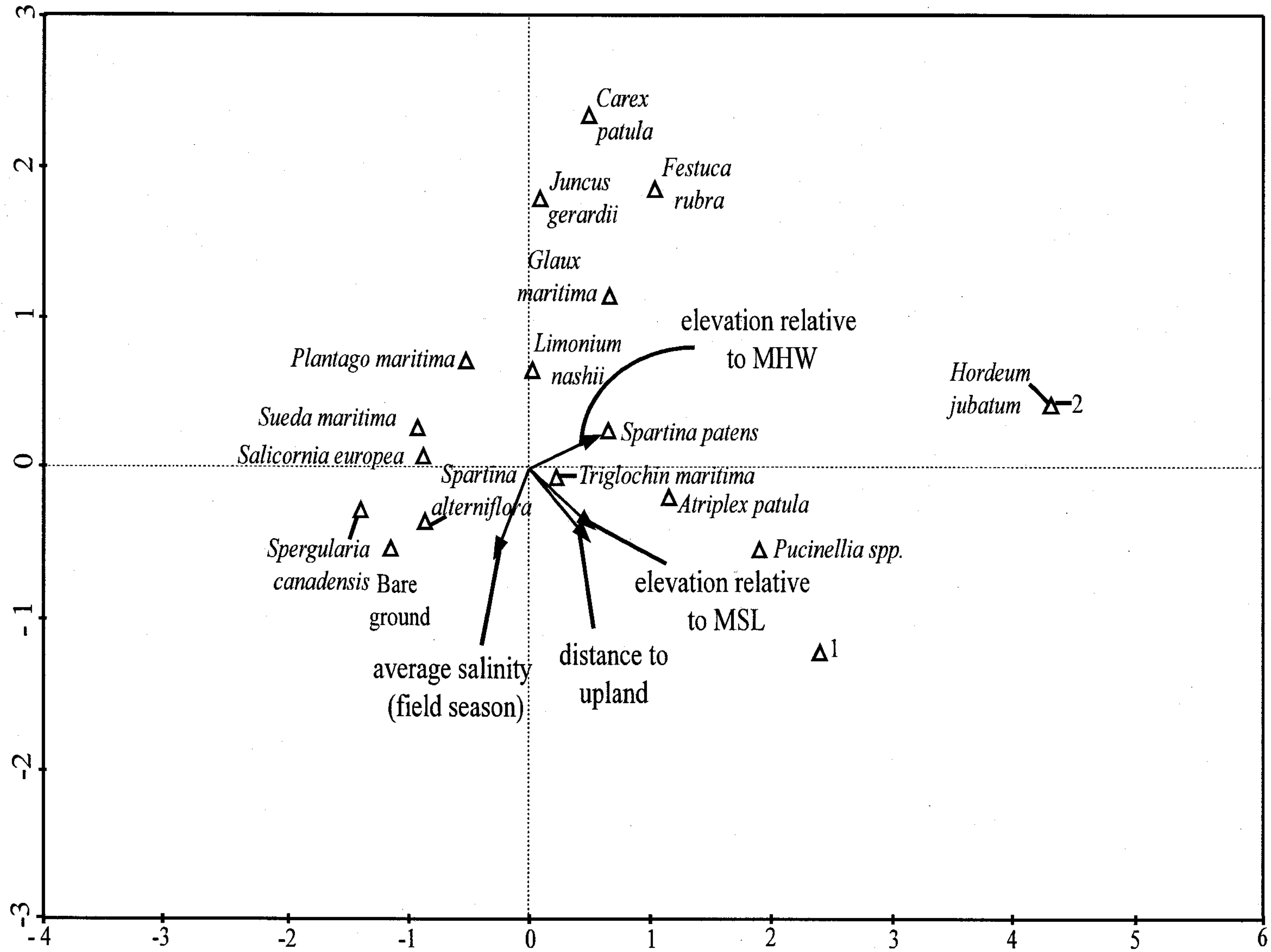

Figure 6. Vegetation percent cover species and environmental variables biplot. Species composition is detailed in Table 3, Appendix A. Numbers denote unidentified species.

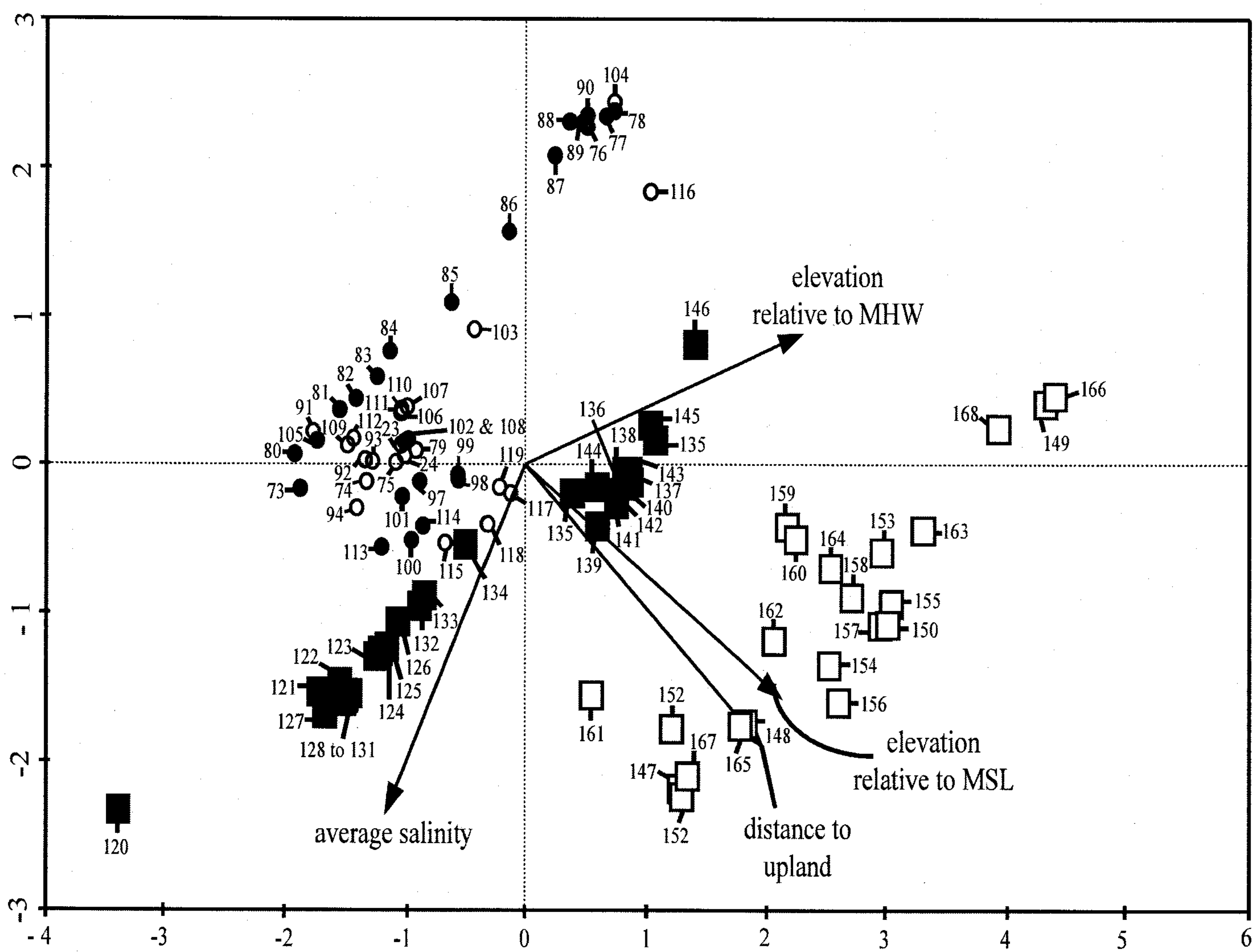

Figure 7. Vegetation percent cover samples and environmental variables biplot. Numbers are unique identifiers of plots listed in Tables 4 and 5, Appendix A. Dipper Harbour (●), Saint's Rest (O), Wood Point (■), and John Lusby (□).

Chapter 4: Conclusion and summary

Through this study I have identified factors which drive sub-surface hydrology and vegetation in Fundy marshes. Sediments, inundation patterns, geomorphology, and precipitation account for spatial and temporal variability in Fundy sub-surface hydrology. Compared to marshes elsewhere, Fundy marshes have low hydraulic conductivity, are infrequently flooded, and have steep channel banks alongside deep intertidal channels. Therefore, sub-surface hydrology research that has mostly been conducted in organogenic, microtidal marshes does not provide suitable models for minerogranic, macrotidal marshes. Though elevation relative to tide levels accounts for differences in vegetation species within and between marshes, other significant variables include elevation relative to mean sea level, distance to upland, salinity, and depth to groundwater.

After a more than a 50-yr recovery period, Saint's Rest and John Lusby are vegetated marsh platforms with sub-surface hydrological processes functioning similarly to reference sites, suggesting that in Fundy marshes these two components can recover within half century time scales. The large growth range of *S. alterniflora* combined with less compactable, low organic sediments make Bay of Fundy marshes resilient (*sensu* Grimm and Wissel 1997) in the face of diking and ditching. Fundy marshes are also resistant (*sensu* Grimm and Wissel 1997) to rising sea level associated with greenhouse warming, as changes in tidal heights and flooding will have minimal impact on depth to groundwater, thus vegetation will be relatively stable.

Macrotidal, minerogenic Bay of Fundy marshes seem ideal candidates for restoration. Though it is not desirable to restore all Fundy dikelands, restoration can be undertaken in locations where dikes do not protect valued infrastructure, agriculture is no longer practiced, or the cost of dike maintenance outweighs the benefits. Restoring these resilient and resistant marshes could offset inevitable loses in less resistant, microtidal marshes of Atlantic Canada.

This study also provides critical insight into challenges likely to be faced when monitoring restoration success at other Fundy salt marsh sites. First, it is nearly impossible to locate a reference site of similar size, geomorphology, and tidal range. Yet differences in these factors are likely to affect marsh vegetation and sub-surface hydrology. Distance to upland edge, a factor explaining vegetation cover, and ratio of marsh area to upland edge, a factor explaining shallow depth to groundwater in spring months, are affected in part by marsh size. Channel density is also related to marsh size (Pethick 1992), thus the proportion of a marsh that is close to channels and experiences greater depth to water is affected by marsh size. Other geomorphological factors that influence sub-surface hydrology, and which vary from marsh to marsh, include levees alongside channels, channel bank slope, and the amount to which channels remain free draining. Elevation relative to mean high water is critical to vegetation zonation, yet MHW increases with tidal range. In the Bay of Fundy, tidal range increases by 0.36% for every kilometer up-basin (Desplanque and Mossman 2004); therefore it is highly unlikely that any two sites will have similar tidal ranges.

Since no monitoring protocols specific to the Bay of Fundy are available, those developed for salt marshes along the Atlantic coast of the U.S. are frequently consulted (e.g., Roman et al. 2001; Neckles et al. 2002). However, difficulties may be faced when using these protocols to monitor groundwater salinity, depth to groundwater, and vegetation composition in Fundy marshes. For example, Neckles et al. (2002) suggest using wells, soil cores, or sippers to collect groundwater from 5-20 cm depths. Given the low moisture content of Fundy sediments and the fact that water takes at least 24 hr to enter piezometers, soil cores and sippers are likely to be impractical. Wells that have been allowed to equilibrate for several days seem the best alternative; however since depth to groundwater is sometimes greater than 20 cm, wells installed to this depth may remain dry. Neckles et al. (2002) do not specify a depth for wells to measure water table; however, Roman et al. (2001) recommend using 30 cm deep wells with a 4 cm inner diameter. Again, groundwater can occur at depths greater than 30 cm in Fundy marshes and pipes with a smaller diameter, such as the ones used in the study, will ensure a more rapid response to changes in hydraulic head (Fitts 2002). Roman et al. (2001) measure water table using a meter stick; however, given the sometimes large depth to groundwater this is impractical in Fundy marshes – the metal tubing used in this study seems more appropriate. These protocols suggest that wells and piezometers be permanently installed (Roman et al. 2001; Neckles et al. 2002). However, I observed sediment inside piezometers removed at the end of this field study. While small amounts of sediment are unlikely to affect readings, larger amounts could collect over long periods of time and affect piezometer/well functioning. Other researchers have used screening (Montalto et al. 2006) and nylon mesh (Agosta 1985) to prevent fine materials from entering wells. However, given the fine size of Fundy sediments (Ayles and Lapointe 1996; Van Proosdij et al. 1999), screening and mesh may become clogged. Therefore, permanent wells and piezometers in Fundy marshes may have to be removed, cleaned, and re-installed at the beginning of each field season. Finally, determining vegetation composition requires one to identify vegetation species contained within a plot. However, several grass species within upper Bay marshes could not be identified because they bloomed and senesced early in the growing season, before the period of peak standing crop, when plant harvests are usually scheduled for monitoring purposes. Thus, the period of harvest limits the ability to identify some species. This issue can be resolved with preliminary vegetation surveys earlier in the growing season.

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APPENDIX A:
Data used in canonical correspondence analysis

Table 1. End of season standing crop (g dry wt m^{-2}) from plots in upper Bay marshes. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. ates. This data used in canonical correspondence annalysis (CCA).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
	1A	1B	1C	2A	2B	2C	3A	3B	3C	4A	4B	4C	5A	5B	5C	6A
Spartina alterniflora	230.692	243.908	222.820	205.580	252.712	239.000	277.024	289.780	212.860	366.248	401.616	359.460	344.680	360.060	374.868	610.224
Spartina patens	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.632	1.360
Hordeum vulgare	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 1	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Limonium nashii	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Hordeum jubatum	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Pucinellia spp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 2	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Salicornia europea	0.000	0.000	0.000	0.120	0.000	0.372	0.496	0.000	0.552	0.472	2.436	0.340	0.144	0.428	3.660	3.044
Unidentified 3	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Atriplex patula	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sueda maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Triglochin maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 4	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Table 1 continued. End of season standing crop (g dry wt m-2) from plots in upper Bay marshes. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates.

	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32
	6B	6C	7A	7B	7C	8A	8B	8C	9A	9B	9C	10A	10B	10C	11A	11B
Spartina alterniflora	413.784	374.652	387.560	378.704	294.936	0.844	0.000	0.000	0.000	0.000	1.188	0.000	0.000	0.000	0.000	0.000
Spartina patens	0.476	0.000	2.660	0.000	21.560	580.236	631.680	693.924	634.080	578.504	246.800	291.012	548.640	281.640	66.024	171.044
Hordeum vulgare	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 1	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	42.784	5.184	6.200	45.908	7.108
Limonium nashii	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	41.816	0.000	0.000	0.000	0.000
Hordeum jubatum	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Pucinellia spp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	12.932	2.536	0.000	0.148	3.004
Unidentified 2	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Salicornia europea	2.596	2.348	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	9.012	0.000	0.000	0.000	0.000	0.000
Unidentified 3	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Atriplex patula	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	13.964	0.000	0.000	0.000	0.000	0.000
Sueda maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	2.520	0.000	0.000	0.000	0.000	0.000
Triglochin maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.220	0.000	15.744	0.000	0.000
Unidentified 4	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Table 1 continued. End of season standing crop (g dry wt m-2) from plots in upper Bay marshes. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. This data used in canonical correspondence analysis (CCA).

	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48
	11C	12A	12B	12C	13A	13B	13C	14A	14B	14C	15A	15B	15C	16A	16B	16C
Spartina alterniflora	0.000	0.000	0.000	0.000	563.188	536.156	345.524	498.808	236.024	325.384	0.000	0.000	0.000	0.588	0.000	0.000
Spartina patens	82.296	260.084	498.884	144.328	0.000	0.000	0.000	0.728	100.172	170.760	33.360	15.524	4.340	464.888	453.056	345.028
Hordeum vulgare	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 1	26.636	8.804	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	7.384	47.380	28.212	7.456	19.276	9.544
Limonium nashii	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	31.392	49.612	67.116	0.000	0.000	0.000
Hordeum jubatum	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	47.320	34.840	40.052	0.000	0.000	0.000
Pucinellia spp.	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	12.252	55.956	15.840	0.968	0.000	0.000
Unidentified 2	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	48.780	103.324	6.336	0.000	0.000	0.000
Salicornia europea	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 3	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	5.856	0.000	12.424	0.000	0.000	0.000
Atriplex patula	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	5.500	0.000	0.000	3.888	0.000	6.340
Sueda maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Triglochin maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 4	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Table 1 continued. End of season standing crop (g dry wt m-2) from plots in upper Bay marshes. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. This data used in canonical correspondence analysis (CCA).

	49	50	51	52	53	54	55	56	57	58	59	60	61	62	63	64
	17A	17B	17C	18A	18B	18C	19A	19B	19C	20A	20B	20C	21A	21B	21C	22A
Spartina alterniflora	0.000	0.000	0.000	0.000	0.000	0.000	203.004	45.376	0.000	1.244	0.000	0.000	0.000	0.000	0.000	0.000
Spartina patens	0.348	0.000	6.068	79.296	3.236	1.720	0.488	316.500	295.088	193.944	442.396	302.856	461.116	373.620	348.308	393.004
Hordeum vulgare	473.136	714.904	978.832	494.208	712.264	659.332	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 1	0.000	0.000	0.000	0.000	0.000	2.216	0.000	0.000	0.000	26.120	4.724	0.000	0.000	0.000	0.000	0.000
Limonium nashii	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Hordeum jubatum	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Pucinellia spp.	0.000	0.000	0.000	0.268	0.124	2.256	0.000	0.452	0.000	60.320	3.640	0.000	0.000	0.000	0.628	0.000
Unidentified 2	0.000	0.000	0.280	0.308	19.976	1.460	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Salicornia europea	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 3	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Atriplex patula	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sueda maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Triglochin maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 4	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.524	0.000	0.000	0.000

Table 1 continued. End of season standing crop (g dry wt m-2) from plots in upper Bay marshes. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. This data used in canonical correspondence annalysis (CCA).

	65	66	67	68	69	70	71	72
	22B	22C	23A	23B	23C	24A	24B	24C
Spartina alterniflora	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Spartina patens	465.816	574.336	67.000	65.444	57.168	87.416	149.472	140.036
Hordeum vulgare	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 1	25.112	8.440	0.000	0.000	0.796	26.704	37.332	15.464
Limonium nashii	0.000	0.000	0.896	0.000	0.000	0.000	0.000	0.000
Hordeum jubatum	0.000	0.000	9.072	37.588	38.024	0.000	0.000	0.000
Pucinellia spp.	8.244	1.788	0.000	1.048	0.000	3.472	10.200	14.508
Unidentified 2	0.000	0.000	9.532	0.000	0.000	1.648	0.000	0.000
Salicornia europea	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 3	0.000	0.000	2.584	0.000	0.000	0.000	0.000	0.000
Atriplex patula	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sueda maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Triglochin maritima	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Unidentified 4	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Table 2. Environmental data collected from upper Bay end-of-season stand crop plots. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. Piezometers which remained dry have no salinity values. 1 = presence of water in shallow piezometers; 0 = absence of water in shallow piezometers. This data used in canonical correspondence analysis (CCA).

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
	1A	1B	1C	2A	2B	2C	3A	3B	3C	4A	4B	4C	5A	5B	5C
distance to any channel (m)	5.0	5.0	5.0	10.0	10.0	10.0	15.0	15.0	15.0	33.7	32.4	29.5	28.8	29.5	28.1
distance to deep channel (m)	5.0	5.0	5.0	10.0	10.0	10.0	15.0	15.0	15.0	48.4	48.2	47.8	54.8	55.8	54.5
distance to upland (m)	216.5	215.6	213.9	217.2	216.5	215.3	219.4	218.1	217.4	142.0	141.5	140.3	136.0	135.5	134.7
elevation relative to MSL (m)	5.093	5.076	5.084	5.146	5.152	5.149	5.24	5.203	5.221	5.346	5.345	5.344	5.393	5.319	5.357
elevation relative to MHW (m)	-0.117	-0.134	-0.126	-0.064	-0.058	-0.061	0.03	-0.007	0.011	0.136	0.135	0.134	0.183	0.109	0.147
bulk density (g cm^{-3})	0.957	1.053	1.114	0.923	0.985	0.982	0.967	0.877	0.909	0.839	0.937	0.934	0.916	0.926	0.766
organic content (%)	7.250	6.503	6.700	7.404	5.000	7.662	7.304	7.497	7.795	8.288	7.792	7.600	8.050	8.296	8.242
water content (%)	19.158	20.400	21.174	20.762	21.316	21.962	21.681	21.289	20.897	23.047	22.798	21.457	21.588	23.793	17.292
presence/absence of water - shallow	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
average depth to water - shallow (cm)	0.0	2.3	2.5	3.8	11.8	8.2	0.0	0.0	4.0	1.5	4.8	0.0	4.0	0.0	5.5
range in depth to water - shallow (cm)	3.0	0.0	4.0	0.0	2.0	7.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5
minimum depth to water - shallow (cm)	0.0	2.0	0.0	2.0	11.0	7.5	0.0	0.0	3.0	0.5	4.0	0.0	3.0	0.0	5.0
maximum depth to water - shallow (cm)	0.0	2.5	4.5	5.0	12.5	9.0	0.0	0.0	6.0	3.0	5.5	1.5	5.5	0.5	6.0
average depth to water - deep (cm)	0.0	0.0	0.0	1.5	0.0	0.8	0.0	0.0	5.5	2.3	2.0	0.0	5.3	1.3	6.8
range in depth to water - deep (cm)	4.0	14.0	3.0	1.5	1.0	34.5	2.0	0.0	0.0	5.0	3.0	0.5	1.0	0.5	3.0
minimum depth to water - deep (cm)	0.0	0.0	60.0	1.0	0.0	0.0	0.0	0.0	5.5	0.5	0.0	0.0	5.0	1.0	5.5
maximum depth to water - deep (cm)	0.0	3.0	63.0	2.5	0.5	26.0	0.0	0.0	5.5	5.5	3.0	0.0	6.0	1.5	8.5
salinity (shallow)	16	30	25	33	29	30	25	29	29	30	34	28	29	28	27
salinity (deep)	28	25	35	27	28	30	15	29	23	30	27	28	26	26	27

Table 2 continued. Environmental data collected from upper Bay end-of-season stand crop plots. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. Piezometers which remained dry have no salinity values. 1 = presence of water in shallow piezometers; 0 = absence of water in shallow piezometers. This data used in canonical correspondence analysis (CCA).

	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
	6A	6B	6C	7A	7B	7C	8A	8B	8C	9A	9B	9C	10A	10B	10C
distance to any channel (m)	24.7	25.8	25.5	28.0	28.2	26.3	33.3	32.3	32.5	37.3	37.1	37.8	6.0	8.0	10.2
distance to deep channel (m)	59.5	59.3	59.1	102.2	101.1	97.2	105.7	104.6	104.8	110.0	108.9	107.8	3.2	3.1	3.1
distance to upland (m)	133.6	132.9	132.5	39.5	39.3	40.9	34.3	34.3	34.5	30.2	30.3	30.7	148.6	150.3	152.4
elevation relative to MSL (m)	5.472	5.429	5.451	5.84	5.895	5.834	5.932	5.904	5.929	6.027	5.932	6.056	5.927	5.92	5.915
elevation relative to MHW (m)	0.262	0.219	0.241	0.63	0.685	0.624	0.722	0.694	0.719	0.817	0.722	0.846	0.717	0.71	0.705
bulk density (g cm^{-3})	0.735	0.632	0.702	0.660	0.667	0.661	0.854	0.805	0.834	1.003	1.215	1.124	1.146	1.089	1.175
organic content (%)	8.546	8.504	8.700	9.345	10.405	10.795	9.405	10.539	14.514	8.787	6.847	8.188	7.900	8.208	7.496
water content (%)	18.316	17.374	20.566	18.790	19.997	16.529	19.955	20.032	19.052	20.435	17.825	17.585	18.473	18.939	19.853
presence/absence of water - shallow	1	1	1	1	1	1	1	1	1	1	1	1	0	0	0
average depth to water - shallow (cm)	3.5	0.8	3.8	6.8	5.8	6.0	6.2	8.5	8.2	10.8	12.5	14.3	16.0	14.5	15.5
range in depth to water - shallow (cm)	0.0	0.0	0.0	1.0	0.5	0.5	0.5	1.5	2.5	9.5	0.5	34.0	15.0	7.5	0.0
minimum depth to water - shallow (cm)	3.5	0.0	3.0	5.0	4.5	4.5	6.0	7.5	7.0	9.5	11.0	12.0	16.0	14.5	15.5
maximum depth to water - shallow (cm)	3.5	2.0	4.5	9.5	7.0	7.0	6.5	9.5	9.5	13.5	13.5	15.5	16.0	14.5	15.5
average depth to water - deep (cm)	1.8	1.2	1.5	6.2	0.0	6.3	5.8	3.7	4.8	13.8	13.3	12.7	23.3	21.0	18.7
range in depth to water - deep (cm)	2.0	3.0	29.5	2.0	1.0	1.5	1.0	1.5	1.0	10.5	3.5	8.0	0.5	2.5	3.0
minimum depth to water - deep (cm)	1.0	0.0	16.0	5.0	0.0	5.5	5.5	3.0	17.0	10.0	47.0	7.5	23.0	49.5	17.0
maximum depth to water - deep (cm)	3.0	3.0	45.5	7.0	0.5	7.0	6.5	4.5	18.0	20.5	50.5	15.5	23.5	52.0	20.0
salinity (shallow)	27		31		28	25	25	24	24			25			
salinity (deep)	28	29	23	19	20	18	21	19	24	18	19	18	25	31	25

Table 2 continued. Environmental data collected from upper Bay end-of-season stand crop plots. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. Piezometers which remained dry have no salinity values. 1 = presence of water in shallow piezometers; 0 = absence of water in shallow piezometers. This data used in canonical correspondence analysis (CCA).

	31	32	33	34	35	36	37	38	39	40	41	42	43	44	45
	11A	11B	11C	12A	12B	12C	13A	13B	13C	14A	14B	14C	15A	15B	15C
distance to any channel (m)	7.6	7.3	7.8	12.5	11.9	10.8	5.0	5.0	5.0	10.0	10.0	10.0	1.7	1.6	4.1
distance to deep channel (m)	7.9	7.3	7.8	12.5	12.5	12.5	5.0	5.0	5.0	10.0	10.0	10.0	15.0	15.0	15.0
distance to upland (m)	151.2	152.4	154.0	152.9	154.6	156.2	393.5	393.1	393.0	396.6	395.7	394.2	403.1	402.2	400.2
elevation relative to MSL (m)	5.904	5.909	5.913	5.925	5.928	5.932	5.653	5.768	5.754	6.071	6.129	6.115	6.747	6.72	6.757
elevation relative to MHW (m)	0.694	0.699	0.703	0.715	0.718	0.722	0.223	0.338	0.324	0.641	0.699	0.685	1.317	1.29	1.327
bulk density (g cm^{-3})	1.094	0.960	0.970	0.913	1.159	1.052	0.964	1.069	0.934	0.930	1.094	1.157	1.216	1.250	1.386
organic content (%)	8.196	8.246	7.858	8.600	7.642	5.630	7.600	7.450	7.446	7.546	6.950	7.357	6.954	7.296	6.600
water content (%)	21.501	19.211	19.579	18.414	20.368	17.974	21.498	21.354	16.682	19.662	19.677	18.374	14.971	15.168	16.366
presence/absence of water - shallow	1	1	1	1	1	1	1	1	1	0	1	1	0	0	0
average depth to water - shallow (cm)	11.0	0.3	7.8	11.3	11.7	11.2	9.0	15.0	12.5	15.0	15.0	11.8	15.0	15.5	15.0
range in depth to water - shallow (cm)	15.0	13.5	1.5	1.2	6.0	4.5	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
minimum depth to water - shallow (cm)	10.5	0.0	7.0	10.5	10.0	2.5	7.5	15.0	10.5	15.0	14.0	8.0	15.0	15.5	15.0
maximum depth to water - shallow (cm)	12.0	4.0	9.5	12.5	15.0	15.5	10.5	15.0	14.5	15.0	16.0	15.5	15.0	15.5	15.0
average depth to water - deep (cm)	9.2	8.0	13.8	11.7	11.8	12.0	48.5	6.8	69.0	50.0	51.0	47.0	46.5	21.3	41.5
range in depth to water - deep (cm)	3.0	2.0	1.5	5.0	7.0	5.5	3.0	2.5	0.0	0.0	0.0	0.0	0.0	21.5	0.0
minimum depth to water - deep (cm)	8.0	7.0	13.0	9.0	53.5	10.0	47.0	5.5	63.0	45.5	25.5	47.0	46.5	10.5	41.5
maximum depth to water - deep (cm)	11.0	9.0	14.5	14.0	60.5	15.5	50.0	8.0	69.0	50.0	51.0	47.0	46.5	32.0	41.5
salinity (shallow)	33	31	28	28	28		26					27			
salinity (deep)	32	34	31	30	31	28	26	21			21				

Table 2 continued. Environmental data collected from upper Bay end-of-season stand crop plots. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. Piezometers which remained dry have no salinity values. 1 = presence of water in shallow piezometers; 0 = absence of water in shallow piezometers. This data used in canonical correspondence analysis (CCA).

	46	47	48	49	50	51	52	53	54	55	56	57	58	59	60
	16A	16B	16C	17A	17B	17C	18A	18B	18C	19A	19B	19C	20A	20B	20C
distance to any channel (m)	2.1	4.0	2.4	2.4	4.2	2.4	2.4	4.1	3.0	115.3	115.3	115.3	107.6	107.6	107.6
distance to deep channel (m)	4.0	4.0	4.0	7.6	7.6	7.6	10.9	10.9	10.9	115.3	115.3	115.3	107.6	107.6	107.6
distance to upland (m)	445.5	442.0	441.7	445.8	446.8	447.6	450.9	448.9	446.6	595.9	601.8	599.5	611.8	611.2	611.8
elevation relative to MSL (m)	6.473	6.45	6.462	6.648	6.602	6.571	6.687	6.647	6.684	6.595	6.609	6.641	6.647	6.702	6.699
elevation relative to MHW (m)	1.043	1.02	1.032	1.218	1.172	1.141	1.257	1.217	1.254	1.165	1.179	1.211	1.217	1.272	1.269
bulk density (g cm^{-3})	1.003	1.117	1.109	1.197	1.125	1.185	1.214	1.181	1.234	0.833	0.627	0.671	0.559	0.690	0.650
organic content (%)	8.096	8.046	9.045	8.700	8.891	8.910	8.587	8.891	9.036	10.700	7.700	12.258	14.236	12.750	13.840
water content (%)	14.839	16.456	15.782	15.551	14.593	15.428	17.230	16.243	15.502	28.472	23.168	26.142	24.895	24.511	24.065
presence/absence of water - shallow	0	0	0	0	0	1	0	0	0	1	1	1	1	1	1
average depth to water - shallow (cm)	16.0	14.5	15.5	16.5	16.5	13.8	15.5	15.0	15.5	0.0	0.0	0.0	4.8	0.0	0.7
range in depth to water - shallow (cm)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
minimum depth to water - shallow (cm)	16.0	14.5	15.5	16.5	16.5	12.5	15.5	15.0	15.5	0.0	0.0	0.0	4.5	0.0	0.0
maximum depth to water - shallow (cm)	16.0	14.5	15.5	16.5	16.5	15.0	15.5	15.0	15.5	0.0	0.0	0.0	5.0	0.0	2.0
average depth to water - deep (cm)	35.3	40.0	20.5	47.3	52.0	34.0	45.5	38.0	42.5	9.8	0.0	0.0	0.0	0.0	0.0
range in depth to water - deep (cm)	1.5	2.0	4.0	0.5	0.0	5.0	2.0	1.0	0.0	2.5	3.5	2.0	1.0	2.0	0.5
minimum depth to water - deep (cm)	34.5	39.0	18.5	47.0	52.0	31.5	44.5	37.5	42.5	8.5	0.0	0.0	0.0	0.0	0.0
maximum depth to water - deep (cm)	36.0	41.0	22.5	47.5	52.0	36.5	46.5	38.5	42.5	11.0	0.0	0.0	0.0	1.0	0.0
salinity (shallow)										30	23	28	32	30	30
salinity (deep)	29	23	24			17				25	35	22	32	31	30

Table 2 continued. Environmental data collected from upper Bay end-of-season stand crop plots. The numbers in the first row uniquely identify each plot in Figure 5, chapter 3. In the second row numbers denote plots (see Table 1, chapter 3) and letters denote replicates. Piezometers which remained dry have no salinity values. 1 = presence of water in shallow piezometers; 0 = absence of water in shallow piezometers. This data used in canonical correspondence analysis

	61	62	63	64	65	66	67	68	69	70	71	72
	21A	21B	21C	22A	22B	22C	23A	23B	23C	24A	24B	24C
distance to any channel (m)	81.1	81.1	81.1	19.2	19.7	21.7	11.3	9.8	9.0	12.2	13.9	15.4
distance to deep channel (m)	81.1	81.1	81.1	41.7	43.5	44.8	16.8	18.4	19.8	20.8	22.9	20.5
distance to upland (m)	625.4	623.6	623.6	319.7	319.0	319.7	381.5	382.7	384.2	406.5	408.0	410.4
elevation relative to MSL (m)	6.731	6.726	6.712	6.766	6.758	6.761	6.778	6.766	6.775	6.727	6.69	6.673
elevation relative to MHW (m)	1.301	1.296	1.282	1.336	1.328	1.331	1.348	1.336	1.345	1.297	1.26	1.243
bulk density (g cm^{-3})	0.477	0.800	0.600	0.947	0.923	0.978	1.219	1.167	1.196	1.009	1.088	0.938
organic content (%)	11.055	9.635	14.336	8.446	9.846	8.950	7.688	7.946	7.858	8.854	9.091	8.900
water content (%)	15.411	22.408	22.439	18.131	19.239	19.602	16.933	18.415	17.891	16.875	18.990	17.550
presence/absence of water - shallow	1	1	1	1	1	0	1	1	1	1	1	1
average depth to water - shallow (cm)	6.8	2.8	4.0	4.0	0.0	14.5	4.5	3.3	13.3	15.4	7.5	8.3
range in depth to water - shallow (cm)	0.1	0.0	0.3	0.2	0.1	0.0	0.2	0.1	0.0	0.0	0.1	0.0
minimum depth to water - shallow (cm)	2.0	2.5	0.0	0.0	0.0	14.5	0.0	0.0	12.5	14.8	4.5	6.0
maximum depth to water - shallow (cm)	11.5	3.0	4.0	11.5	0.0	14.5	12.0	10.0	14.0	16.0	10.5	10.5
average depth to water - deep (cm)	5.5	3.8	0.0	27.8	0.0	34.8	45.5	44.0	40.3	48.0	29.0	12.5
range in depth to water - deep (cm)	1.0	0.5	1.5	29.5	2.5	18.5	0.0	0.0	0.5	0.0	6.0	4.0
minimum depth to water - deep (cm)	5.0	3.5	0.0	13.0	0.0	25.5	23.0	37.0	40.0	36.0	26.0	10.5
maximum depth to water - deep (cm)	6.0	4.0	0.5	42.5	0.0	44.0	45.5	44.0	40.5	48.0	32.0	14.5
salinity (shallow)	31	26	29	2				20			20	
salinity (deep)	30	28	28	28	25	27	17	19		21	20	23

Table 3. Percent cover of vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2).

	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89	90	91	92	93	94	95	96	97
	A0	A3	A6	A9	A12	A15	A21	B0	B2	B5	B8	B11	B15	B18	B21	B23	B25	B27	C0	C6	C13	C21	C31	C40	D0
Spartina alterniflora	100	10	10	0	0	0	0	90	100	90	18	78	10	0	0	10	0	0	25	20	20	97	0	10	70
Spartina patens	0	0	45	42.5	40	40	20	0	0	0	80	0	15	33.3	39	30	0	0	0	10	40	0	99	40	0
Pucinellia spp.	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Plantago maritima	0	89	45	42.5	55	40	70	9	0	0	0	0	50	0	0	54	0	0	25	0	40	1	1	40	0
Bare ground	0	0	0	0	0	0	0	0	0	10	0	10	5	0	0	0	0	0	20	50	0	0	0	0	10
Juncus gerardii	0	0	0	0	0	0	4.5	0	0	0	0	0	0	33.3	30	0	40	10	0	0	0	0	0	0	0
Carex palascea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	40	45	0	0	0	0	0	0	0
Unidentified 1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin maritima	0	0	0	10	0	0	4.5	0	0	0	0	1	0	0	1	0	1	0	0	10	0	0	0	0	0
Glaux maritima	0	0	0	0	0	10	0	0	0	0	0	0	20	33.3	30	0	19	45	0	0	0	0	0	0	0
Sueda maritima	0	1	0	0	0	0	0	0	0	0	1	10	0	0	0	0	0	0	25	0	0	1	0	0	10
Salicornia europea	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	4	10	0	0	0	0	10
Limonium nashii	0	0	0	5	0	10	1	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	10	0
Hordeum vulgare	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Festuca rubra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0
Atriplex patula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spergularia canadensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Unidentified 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 3 continued. Percent cover of vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). This data used in canonical correspondence analysis (CCA).

	98	99	100	101	102	103	104	105	106	107	108	109	110	111	112	113	114	115	116	117	118	119	120
	D12	D23.5	D38	D61	D147	D156	D173.5	E0	E5	E10	E30	F0	F4	F10.5	F12.5	G0	G2.5	G5	G10	G20	G38	G58	H0
Spartina alterniflora	98	100	100	100	100	80	0	45	100	100	90	100	100	100	100	89	100	89	70	74	79	100	100
Spartina patens	0	0	0	0	0	0	94	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pucinellia spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Plantago maritima	0	0	0	0	0	20	0	10	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0
Bare ground	0	0	0	0	0	0	0	40	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0
Juncus gerardii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Carex palascea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unidentified 1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin maritima	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	10	1	0	0
Glaux maritima	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sueda maritima	0	0	0	0	0	0	5	5	0	0	0	0	0	0	0	1	0	0	0	5	10	0	0
Salicornia europea	0	0	0	0	0	0	1	0	0	0	10	0	0	0	0	0	0	0	0	1	10	0	0
Limonium nashii	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hordeum vulgare	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Festuca rubra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	30	0	0	0	0
Rock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Atriplex patula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Spergularia canadensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unidentified 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 3 continued. Percent cover of vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). This data used in canonical correspondence analysis (CCA).

	121	122	123	124	125	126	127	128	129	130	131	132	133	134	135	136	137	138	139	140	141	142	143	144	145
	H2.5	H5	H10	H15	H25	H35	I0	I2.5	I5	I10	I30	I60	I90	I120	J0	J2.5	J5	J10	J15	J25	J35	J40	J45	J48.5	K25
Spartina alterniflora	100	100	100	100	100	99	60	70	70	80	70	100	99	100	10	0	0	0	0	20	0	20	0	20	0
Spartina patens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	70	70	100	100	100	50	80	60	85	80	100
Pucinellia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	10	0	0	0	30	20	20	15	0	0
Plantago	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	10	0	0	0	0	0	0	0	0	0
Bare ground	0	0	0	0	0	0	40	30	30	20	30	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juncus gerardii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Carex palascea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unidentified 1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Triglochin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glaux	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sueda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Salicornia	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Limonium	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0
Hordeum vulgare	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Festuca	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Atriplex	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spergularia canadensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unidentified 2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 3 continued. Percent cover of vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). This data used in canonical correspondence analysis (CCA).

	146	147	148	149	150	151	152	153	154	155	156	157	158	159	160	161	162	163	164	165	166	167	168
	K55	L0	L4	L7	L13.6	L23.6	M0	M7.5	M67.5	M111.5	M161.5	N0	N87.5	O0	O20	O94	O173	O224	O262	P0	P3	P10	P13
Spartina alterniflora	0	100	0	0	0	0	100	0	0	0	0	0	0	0	0	100	0	0	0	0	0	0	0
Spartina patens	99	0	0	0	0	0	0	0	0	0	100	0	94	90	0	0	0	100	100	50	0	50	0
Pucinellia	0	0	100	0	10	10	0	100	0	0	0	0	5	0	0	0	0	0	0	50	100	50	100
Plantago	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bare ground	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0
Juncus gerardii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Carex palascea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unidentified 1	0	0	0	0	69	60	0	0	100	100	0	100	0	0	90	0	75	0	0	0	0	0	0
Triglochin	0	0	0	0	0	20	0	0	0	0	0	0	0	0	10	0	25	0	0	0	0	0	0
Glaux	0	0	0	0	20	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sueda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Salicornia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limonium	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hordeum vulgare	0	0	0	50	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Festuca	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Atriplex	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Spergularia canadensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Unidentified 2	0	0	0	50	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 4. Environmental data collected from vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). Aug = measurements taken in August 2005 and f.s. = measurements taken throughout 2005 field season. Piezometers which remained dry throughout the field season have no average, minimum, or maximum salinity values. This data used in canonical correspondence analysis (CCA).

	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88
	A0	A3	A6	A9	A12	A15	A21	B0	B2	B5	B8	B11	B15	B18	B21	B23
distance to any channel (m)	3.3	4.3	6.1	8.8	11.9	14.8	21.4	0	2.3	4.3	7.3	10.3	13.5	17.5	20.2	23.4
distance to deep channel (m)	3.3	4.3	6.1	8.8	11.9	14.8	21.6	0	2.3	4.3	7.3	10.3	13.5	17.5	20.2	23.4
distance to upland (m)	83.9	83.9	83.8	84	83.6	84	85.4	23	21	18.6	16.2	13.6	11.3	9.4	7.3	5.6
elevation relative to MSL (m)	2.659	3.087	3.224	3.356	3.446	3.481	3.307	2.689	2.836	2.919	2.994	3.003	3.259	3.406	3.46	3.456
elevation relative to MHW (m)	-0.4	0.1	0.2	0.3	0.4	0.5	0.3	-0.3	-0.2	-0.1	0.0	0.0	0.2	0.4	0.4	0.4
average depth to water - Aug (cm)	25.7	34.1	41.0	29.5	27.5	21.0	121.5	62.0	73.9	8.5	32.8	41.8	7.7	36.9	25.8	40.0
range in depth to water - Aug (cm)	47.5	25.5	11	0	0	0	6.5	13	13	13.5	29	9.5	17	5.5	6.5	0
minimum depth to water - Aug (cm)	8.5	18.5	36.2	29.5	27.5	21	21.5	56	66	0.5	15	36.5	0	33.5	21.5	40
maximum depth to water - Aug (cm)	56	44	47.2	29.5	27.5	21	28	69	79	14	44	46	17	39	28	40
range in depth to water - f.s. (cm)	47.5	43	39	1.5	5	15.5	24	24	16	13	31	19.5	12	28	34	39.5
minimum depth to water - f.s. (cm)	8.5	18.5	36.2	28	22.5	5.5	21.5	45	66	1	15	36.5	5	11	13	0.5
maximum depth to water - f.s. (cm)	56	61.5	75.2	29.5	27.5	21	45.5	69	82	14	46	56	17	39	47	40
average salinity - f.s.	30	36	36				36	29	27	27	25	23	21	16	8	4
minimum salinity - f.s.	4	4	30				31	22	23	21	20	18	17	19	16	14
maximum salinity - f.s.	7	6	31				38	35	30	33	28	26	18	23	17	21

Table 4 continued. Environmental data collected from vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). Aug = measurements taken in August, 2005 and f.s. = measurements taken throughout 2005 field season. Piezometers which remained dry throughout the field season have no average, minimum, or maximum salinity values. This data used in canonical correspondence analysis (CCA).

	89	90	91	92	93	94	95	96	97	98	99	100	101	102	103	104
	B25	B27	C0	C6	C13	C21	C31	C40	D0	D12	D23.5	D38	D61	D147	D156	D173.5
distance to nearest channel (m)	25	27.1	0.4	6.3	12.6	20.8	23.4	34.2	0	12.2	23.3	24.5	15	20.4	25.1	35.3
distance to deep channel (m)	25	27.1	6	8.1	12.4	16.8	31.4	43.1	0	12.2	22.1	24.7	20.8	44.6	53.2	68.4
distance to upland (m)	3.8	1.7	41.2	45	49.3	54.4	62	66	185.7	175.4	167.2	152.5	129.9	47.1	39.2	22.9
elevation relative to MSL (m)	3.546	3.56	2.743	3.122	3.163	3.201	3.296	3.308	3.239	3.595	3.604	3.553	3.47	3.482	3.581	3.786
elevation relative to MHW (m)	0.5	0.5	-0.3	0.1	0.1	0.2	0.3	0.3	-0.1	0.3	0.3	0.3	0.2	0.2	0.3	0.5
average depth to water - Aug (cm)	40.0	40.0	10.4	18.2	7.6	11.6	11.8	32.5	37.3	38.7	16.5	20.7	7.5	8.0	10.0	5.3
range in depth to water - Aug (cm)	0	0	13	3	6.5	2.5	10	6	2.5	7	7	20.5	3	6	2.5	1
minimum depth to water - Aug (cm)	40	40	3.5	17.5	5.5	10.5	7.5	30	36	36	13	13	6	5	8.5	5
maximum depth to water - Aug (cm)	40	40	16.5	20.5	12	13	17.5	36	38.5	43	20	33.5	9	11	11	6
range in depth to water - f.s. (cm)	32	38.6	13	3	6.5	2.5	10	6	51	30	26	30	33	20	13	38
minimum depth to water - f.s. (cm)	8	1.4	3.5	17.5	5.5	10.5	7.5	30	13.5	6	13	3.5	3	5	8.5	5
maximum depth to water - f.s. (cm)	40	40	16.5	20.5	12	13	17.5	36	64.5	36	39	33.5	36	25	21.5	43
average salinity - f.s.	6	5	28	36	36	42	35	37	30	34	34	41	34	33	22	
minimum salinity - f.s.	6	2	20	31	32	39	28	35	29	33	32	40	31	32	20	
maximum salinity - f.s.	11	8	36	47	34	44	40	40	31	34	35	41	36	33	26	

Table 4 continued. Environmental data collected from vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). Aug = measurements taken in August, 2005 and f.s. = measurements taken throughout 2005 field season. Piezometers which remained dry throughout the field season have no average, minimum, or maximum salinity values. This data used in canonical correspondence analysis (CCA).

	105	106	107	108	109	110	111	112	113	114	115	116	117	118	119	120
	E0	E5	E10	E30	F0	F4	F10.5	F12.5	G0	G2.5	G5	G10	G20	G38	G58	H0
distance to nearest channel (m)	0	4	7	14.9	0	3.8	3.8	0.7	1.8	0	2.4	7.1	16.3	32.7	51.5	0
distance to deep channel (m)	0	4	9.1	28.8	3.8	3.8	3.8	2.5	4.1	6.3	9.1	13.8	24	41.7	61.7	0
distance to upland (m)	35.7	41.3	45.8	65.3	41.4	41.5	40.6	39	247.6	246.7	245.8	242.8	241	238.1	238.2	139.8
elevation relative to MSL (m)	2.847	3.375	3.37	3.416	3.122	3.356	3.359	3.135	3.088	3.307	3.512	3.692	3.798	3.76	3.707	3.56
elevation relative to MHW (m)	-0.5	0.1	0.1	0.1	-0.2	0.1	0.1	-0.2	-0.2	0.0	0.2	0.4	0.5	0.5	0.4	-1.7
average depth to water - Aug (cm)	30.8	15.2	10.2	7.2	9.3	7.2	8.7	8.7	76.5	54.8	26.8	44.3	25.2	32.5	11.0	14.0
range in depth to water - Aug (cm)	6.5	2.5	3.5	3	6.5	8	4	5	0	14.5	0.5	5	3.5	11.5	11	11
minimum depth to water - Aug (cm)	28.5	14	9	5.5	6	3.5	6	5.5	76.5	41	26.5	41	23.5	26	7	9.5
maximum depth to water - Aug (cm)	35	16.5	12.5	8.5	12.5	11.5	10	10.5	76.5	55.5	27	46	27	37.5	18	20.5
range in depth to water - f.s. (cm)	41	36	74	32	13	28	35.5	39	0	13	9.5	32.5	27	11.5	11	28
minimum depth to water - f.s. (cm)	14	11	2.5	2	2	3.5	3.5	5.5	76.5	41	26.5	13.5	23.5	26	7	9.5
maximum depth to water - f.s. (cm)	55	47	76.5	34	15	31.5	39	44.5	76.5	54	36	46	50.5	37.5	18	37.5
average salinity - f.s.	23	28	27	32	28	28	28	28	31	32	37		36	39	34	31
minimum salinity - f.s.	21	28	24	31	26	26	25	26	31	31	37		36	39	34	29
maximum salinity - f.s.	24	29	29	32	30	29	29	28	31	32	37		36	40	34	31

Table 4 continued. Environmental data collected from vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). Aug = measurements taken in August, 2005 and f.s. = measurements taken throughout 2005 field season. Piezometers which remained dry throughout the field season have no average, minimum, or maximum salinity values. This data used in canonical correspondence analysis (CCA).

	121	122	123	124	125	126	127	128	129	130	131	132	133	134	135	136
	H2.5	H5	H10	H15	H25	H35	I0	I2.5	I5	I10	I30	I60	I90	I120	J0	J2.5
distance to nearest channel (m)	4.92	8.01	11.44	16.15	19.26	10.13	0	2.5	5	5.05	20.81	40.97	19.12	39.44	2.89	4.31
distance to deep channel (m)	4.92	8.01	11.44	16.15	22.7	21.14	0	2.5	5	10	30	60	90	120	2.89	4.31
distance to upland (m)	141.6	141.1	138.5	131.5	123.7	114.6	188.7	187.7	184.1	179.7	161.1	132.7	109.8	87.5	100.3	99.2
elevation relative to MSL (m)	4.524	4.636	4.799	4.833	4.885	4.902	4.501	4.592	4.604	4.63	4.67	4.94	5.004	5.173	5.681	6.063
elevation relative to MHW (m)	-0.7	-0.6	-0.4	-0.4	-0.3	-0.3	-0.7	-0.6	-0.6	-0.6	-0.5	-0.3	-0.2	0.0	0.5	0.9
average depth to water - Aug (cm)	13.4	10.5	5.3	6.1	8.6	3.4	26.8	7.8	12.1	11.5	10.1	7.5	2.1	4.3	31.9	25.4
range in depth to water - Aug (cm)	10	22	8	7	6	3	9	4.5	6	6	5	11	0.5	2.5	4	15
minimum depth to water - Aug (cm)	8	4	1.5	3	6	2	22	5	20.5	8	11	4	2	3	29	18
maximum depth to water - Aug (cm)	18	26	9.5	10	12	5	31	9.5	26.5	14	16	15	2.5	5.5	33	33
range in depth to water - f.s. (cm)	38.5	22	22	9	30	13.5	19	11	27	30	17.5	18.5	13	22	40	46
minimum depth to water - f.s. (cm)	8	4	1.5	3	6	1.5	22	5	10	8	7	5.5	2	3	29	18
maximum depth to water - f.s. (cm)	46.5	26	23.5	12	36	15	41	16	37	38	24.5	24	15	25	69	64
average salinity - f.s.	32	32	32	32	33	31	32	32	32	32	34	29	29	27	28	28
minimum salinity - f.s.	31	32	30	31	32	30	31	32	32	31	33	27	28	27	25	28
maximum salinity - f.s.	32	34	33	34	34	31	32	33	32	33	35	30	30	27	32	29

Table 4 continued. Environmental data collected from vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). Aug = measurements taken in August, 2005 and f.s. = measurements taken throughout 2005 field season. Piezometers which remained dry throughoutt the field season have no average, minimum, or maximum salinity values. This data used in canonical correspondence analysis (CCA).

	137	138	139	140	141	142	143	144	145	146	147	148	149	150	151	152
	J5	J10	J15	J25	J35	J40	J45	J48.5	K25	K55	L0	L4	L7	L13.6	L23.6	M0
distance to nearest channel (m)	6.12	7.34	8.12	16	17.73	12.92	8.48	5.21	13.68	8.75	0.5	4.9	7.8	14.1	24.1	1
distance to deep channel (m)	6.12	10.72	14.99	25.2	17.73	12.92	8.48	5.21	18.31	17.78	0.5	4.9	7.8	14.1	24.1	1
distance to upland (m)	98.9	97.3	96.9	96.7	97.2	96.7	97.5	97.3	73.7	44.1	807.3	805.2	803.7	805.9	811.5	673.4
elevation relative to MSL (m)	6.048	6.005	5.978	6.019	6.027	5.991	5.969	5.838	6.053	6.16	5.46	5.687	6.623	6.318	5.49	5.596
elevation relative to MHW (m)	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.6	0.8	1.0	0.0	0.3	1.2	0.9	0.1	0.2
average depth to water - Aug (cm)	47.7	22.9	13.0	9.9	15.3	16.0	19.4	38.9	16.2	14.4	27.6	57.6	36.0	34.9	40.6	27.5
range in depth to water - Aug (cm)	6	6	5	11	9.5	9	6	3.5	9	12	16	8	0	12	16	15
minimum depth to water - Aug (cm)	45	20.5	11	4	11.5	11	17	37	10.5	9	18	52	36	27	33	19.5
maximum depth to water - Aug (cm)	51	26.5	16	15	21	20	23	40.5	19.5	21	34	60	36	39	49	34.5
range in depth to water - f.s. (cm)	39.5	30	27	46.5	27	33	49	38	30.5	19	39	10	0	12	11.5	15
minimum depth to water - f.s. (cm)	30.5	18	10	4	8.5	2	3	6	4	7	18	52	36	27	33	19.5
maximum depth to water - f.s. (cm)	70	48	37	50.5	35.5	35	52	44	34.5	26	57	62	36	39	44.5	34.5
average salinity - f.s.	33	31	37	33	35	33	30	30	27	21	26	22		20	27	26
minimum salinity - f.s.	32	30	36	32	35	32	30	29	26	20	22	21		18	26	25
maximum salinity - f.s.	34	33	38	33	35	34	31	32	29	22	28	24		22	29	28

Table 4 continued. Environmental data collected from vegetation plots along piezometer transects. The numbers in the first row uniquely identify each plot in Figure 7, chapter 3 . In the second row, letters denote transect and numbers denote distance along transect (Figs. 2-5, chapter 2). Aug = measurements taken in August, 2005 and f.s. = measurements taken throughout 2005 field season. Piezometers which remained dry throughout the field season have no average, minimum, or maximum salinity values. This data used in canonical correspondence analysis (CCA).

	153	154	155	156	157	158	159	160	161	162	163	164	165	166	167	168
	M7.5	M67.5	M111.5	M161.5	N0	N87.5	O0	O20	O94	O173	O224	O262	P0	P3	P10	P13
distance to nearest channel (m)	8.5	31	43	41.3	20.2	0	0	20	94	65.3	115.5	131.1	0	3	0	3.2
distance to deep channel (m)	8.5	48	58	72.5	35	26.8	0	20	94	173	224.3	262.3	0	3	0	3.2
distance to upland (m)	672	678.3	686.9	744.5	698.4	669.3	611.5	591.9	516.9	446.1	426.7	389.8	801.1	799.6	797.5	794.7
elevation relative to MSL (m)	6.422	6.436	6.623	6.433	6.587	6.381	6.305	6.864	6.449	5.611	6.513	6.52	5.671	6.671	5.498	6.416
elevation relative to MHW (m)	1.0	1.0	1.2	1.0	1.2	1.0	0.9	1.4	1.0	0.2	1.1	1.1	0.2	1.2	0.1	1.0
average depth to water - Aug (cm)	40.6	18.0	18.0	6.8	13.9	18.8	55.8	22.4	9.6	10.8	9.4	7.6	35.6	35.0	41.1	35.5
range in depth to water - Aug (cm)	14	15	13	9	8	7	12	14	5	15	14	13	4	0	1	0
minimum depth to water - Aug (cm)	33	8	11.5	1.5	9	14.5	49	14	7	0	1	0	41	35	41.5	33.5
maximum depth to water - Aug (cm)	47	23	24.5	10.5	17	21.5	61	28	12	15	15	13	45	35	42.5	33.5
range in depth to water - f.s. (cm)	15	15	14.5	9	11	8	19	18	10	15	14	13	16	0	22	0
minimum depth to water - f.s. (cm)	33	8	10	1.5	6	14.5	49	14	2	0	1	0	29	35	33.5	33.5
maximum depth to water - f.s. (cm)	48	23	24.5	10.5	17	22.5	68	32	12	15	15	13	45	35	55.5	33.5
average salinity - f.s.	19	32	27	33	29	24	21	26	36	32	29	29	22		25	
minimum salinity - f.s.	17	30	25	31	28	15	20	25	30	29	27	28	18		23	
maximum salinity - f.s.	22	33	30	35	31	27	24	27	38	37	30	30	26		28	