Impacts on *Spartina alterniflora*:

Factors Affecting Salt Marsh Edge Loss.

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Abstract of the Dissertation

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Spartina marshes are found on shores along the Atlantic coast of North America. A number of natural and anthropogenic impacts are thought to affect the rate of salt marsh loss. However, few long term assessments of changes in salt marshes are available. This dissertation project characterized and ranked factors that influence the recession of the edge of *Spartina alterniflora* salt marshes, focusing on Hempstead Bay, the westernmost bay of Long Island's South Shore Estuary reserve. I used 12 sets of aerial photographs of these marshes taken from 1926 to 2007. Using a randomization process, I chose 500 points along the edge of the marsh and determined the gain or loss of marsh by the change in the location of the edge of marsh for each time period for which aerial photographs were available. For the time interval 1966-2007, I examined a number of different potential predictor variables, each associated with factors hypothesized to cause marsh loss, and assessed which variables were most correlated with salt marsh loss or gain. I then compared change in the marsh from 1966-2007 with that seen from 1926-1966 to test for the effects of different factors pre and post heavy urban development. The loss of salt marsh area from the edge was not attributable to any single influence. Edges formed artificially by dredging continued to lose marsh at a high rate long after the initial damage. The distance of the marsh to borrow pits was also a significant factor correlated with marsh loss. Urbanization and increased boat use after 1966 were also correlated with greater marsh loss. Several natural factors were also correlated with marsh loss, including having a large fetch and storm impacts and tidal flow rate. Surprisingly, increased nutrient load was not correlated with marsh loss or gain. In general, the edges of smaller channels that were most distant from both natural and anthropogenic disturbance have changed relatively little over this 81 year time frame

Dedication Page

To my parents
James D. and Catherine Browne

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Preface

My quest to complete this dissertation has been a long one, with many twists, turns and pitfalls along the path. I have worked and lived around salt marshes for several decades, and the work presented here only represents one facet of my journey toward an understanding of the systems that underlay these wetlands, but a fundamental one. I see many factors controlling the fate of salt marshes, even within the confines of the urban ecosystem studied here. My hope is that I have touched on something useful in the broader understanding of these estuarine systems and the species that depend on them.

ELEPHANT AND THE BLIND MEN

Teaching the concept Syadvada, from the Jainist tradition.

Once upon a time, there lived six blind men in a village. One day the villagers told them, "Hey, there is an elephant in the village today."

They had no idea what an elephant is. They decided, "Even though we would not be able to see it, let us go and feel it anyway." All of them went where the elephant was. Everyone of them touched the elephant.

"Hey, the elephant is a pillar," said the first man who touched his leg.

"Oh, no! it is like a rope," said the second man who touched the tail.

"Oh, no! it is like a thick branch of a tree," said the third man who touched the trunk of the elephant.

"It is like a big hand fan" said the fourth man who touched the ear of the elephant.

"It is like a huge wall," said the fifth man who touched the belly of the elephant.

"It is like a solid pipe," Said the sixth man who touched the tusk of the elephant.

They began to argue about the elephant and everyone of them insisted that he was right. It looked like they were getting agitated. A wise man was passing by and he saw this. He stopped and asked them, "What is the matter?" They said, "We cannot agree to what the elephant is like." Each one of them told what he thought the elephant was like. The wise man calmly explained to them, "All of you are right. The reason every one of you is telling it differently because each one of you touched the different part of the elephant. So, actually the elephant has all those features what you all said."

"Oh!" everyone said. There was no more fight. They felt happy that they were all right.

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Chapter 1 INTRODUCTION

Salt marshes represent a relatively simple, yet important, ecosystem in which to test ecological hypotheses. On the east coast of North America, *Spartina alterniflora* is the dominant salt marsh grass, and the ecology of this system has been studied for decades, since the classical work of Teal (1962). *Spartina alterniflora* is found on shorelines along the saline to brackish waters of rivers, estuaries, including those formed behind barrier beaches, and other sheltered bays and shores from Nova Scotia, Canada, to Florida, throughout the Gulf of Mexico, and south to Argentina (Vicarie et al. 2002). The plant community in these marshes shows a characteristic elevational zonation (Teal and Teal 1971, Niering and Warren 1980, Bertness 1991a, Bertness et al. 2002). This zonation occurs as different species tolerate or require different periods of tidal immersion. Only *Spartina alterniflora* occurs lowest on the shore, at elevations below mean high tide, where it forms a near monoculture (McKee and Patrick 1988, Waren and Niering 1993).

Spartina alterniflora is occasionally found sub-tidally in the southern part of its range where the tidal range is small, and only forms a narrow fringe along the shore in the north where tidal ranges are large (McKee and Patrick 1988). Within a given marsh, plant diversity generally increases with elevation. Above the mean high tide level the community includes *Spartina patens*, *Iva fructesens*, *Juncus jerardi*, and other species, and continues to increase in diversity until upland plants are finally encountered (Bertness and Ellison 1987, Bertness 1991a, Bertness et al. 2002). There are two commonly recognized ecomorphs of *S. alterniflora* that grade into each other. There is a short form (15-30 cm high), which covers flat expanses near mean high water and a tall form which reaches about 2 m in height and grows deeper into the water, particularly along the edges of bays, creeks, and ditches (Teal and Teal 1971, Niering and Warren 1980).

BIOLOGY AND ECOLOGY OF *SPARTINA ALTERNAFLORA*

Smooth cordgrass (*Spartina alterniflora*) is the fundamental defining plant for salt marshes on the east coast of North America (Teal and Teal 1969). Like many perennial grasses, *S. alterniflora* spreads locally though clonal growth, and can expand as a front into bare patches at approximately 12 cm y^{-1} (Hartman 1988). New patches of marsh can be initiated from clumps of marsh that are dislodged and then expand once stranded (Proffitt and Young 1999) or from seeds, which can form more than 30 tillers in one growing season (Metcalfe et al. 1986).

The flowering of *S. alterniflora* is influenced by temperature, and seeds are produced in the fall (Metcalfe et al. 1986, Callaway and Josselyn 1992). Seeds from *S. alterniflora* are generally moved by water (Metcalfe et al. 1986) or can be carried by waterfowl in the fall (Vivian-Smith and Stiles 1994). They can also travel long distances with detached *Spartina* vegetation (Sayce et al. 1997). Flower and seed predation can be severe (Bertness et al. 1987, Bertness and Shumway 1992), as well as high seedling mortality (Hopking and Parker 1984, Ungar 1987). Some studies have found high seed germination rates (Callaway and Josselyn 1992), while others have found low numbers of viable seeds in the field (Hartman 1988). It has been shown that *S. alterniflora* has a transient seed bank with no long term seed survival (Xiao et al. 2009). The number of seeds found in the fall, soon after seed production, is much higher than in the subsequent spring (Wang et al. 2009).

Inbreeding depression can occur (Davis et al. 2004), therefore dispersal by seeds is important for establishing new patches of marsh and maintaining genetic diversity (Novy et al. 2010). Natural seedlings are readily found in the spring in some marshes (Reimold et al. 1978, Metcalfe et al. 1986, Mendelsshon and Kuln 2003, Elsey-Quirk et al. 2009). Natural *S. alterniflora* seedlings, as well as significant numbers of successful experimental seedlings, were found on Louisiana marshlands by Elsey-Quirk et al. (2009). Successful seedlings were also found to add significant biomass during sediment addition experiments in Georgia and Louisiana (Reimold et al. 1978, Mendelsshon and Kuln 2003).

Low density recruitment of seedlings will form clonal patches that coalesce into continuous meadows (Davis et l. 2004). Long range dispersal by seed followed by local clonal expansion is a dispersal strategy known as stratified diffusion (Shigesada et al. 1995), and may be typical of *S. alterniflora* (Davis et al. 2004). This could account for unexpectedly high levels of genetic diversity and small clonal patches of about 100 m^2 or less that have also been found in some established marshes (Richards et al. 2004).

S. alterniflora has a number of adaptations that allow it to create marshlands in tidal estuaries. First, *S. alterniflora* is a C4 grass (Maricle et al. 2009). Plants with C4 photosynthesis tend to be adapted to high sunlight, high daytime temperatures during the growing season, and dry conditions, or limited availability of freshwater (Nobel 1983, Lanbers et al. 1998), all helpful for surviving in the sunny open salt marsh. C4 plants also have a higher photosynthetic efficiency, even when they have lower tissue levels of nitrogen (Lambers et al. 1998), as would be expected in the nitrogen poor conditions of the marsh

A second important characteristic of *S. alterniflora* is its capacity to survive in waterlogged sediment. As an emergent wetland plant, *S. alterniflora* must supply enough oxygen from the leaves and shoots to the roots to allow root metabolism in anaerobic sediment (Armstrong 1979, Colmer 2009), and reduce the negative effects of oxygen deficiency (Howes and Teal 1995, Linthurst 1979). *S. alterniflora* is capable of some anaerobic metabolism, releasing malate and limited amounts of ethanol from the roots (Mendelssohn et al. 1981). It is common among wetland plants, including *S. alterniflora*, to transport O_2 into the roots through specialized porous and gas-filled tissues, called aerenchyma (Colmer 2003), or lacunae (Howes and Teal 1994). In *S. alterniflora*, the diffusion of oxygen into the root system may be aided by hygrometric pressurization when growing in dry air (Hwang and Moris 1991, Colmer 2003). This pressurization, however, was not confirmed in greenhouse experiments by Maricle and Lee (2007). Mass transport of air through rhizomes is found in other wetland plants including the yellow waterlily, *Nuphar luteum* (Dacey 1980, Dacey 1981, Dacey and Klug 1982), the lotus, *Nelumbo* (Dacey 1987, Mevi-Schutz and Grosse 1988), and *Phragmities* (Armstrong and Armstrong 1991, Armstrong et al. 1992, Beckett et al. 2001). At present, there is no evidence that *S. alterniflora* has a flow-through air transportation system (Hwang and Morris, 1991). This finding is supported by a comparison between *S. alterniflora* and *S. anglica* (Maricle and Lee 2007). High alcohol dehydrogenase (ADH) levels were found in *S. alterniflora* roots, indicating chronic oxygen deficiency. This was not as severe in *S. anglica,* which has much smaller aerenchyma and is likely to have a flow through system (Maricle and Lee 2007).

Bioturbation, particularly from burrowing fiddler crabs, *Uca* spp., increases the available oxygen in the sediments, and increases the productivity of tall-form *Spartina* (Bertness 1985). Roots can also maintain an oxygenated rhizosphere when some of the oxygen transported into the roots is leaked into the surrounding sediment (Armstrong 1964, Teal and Kanwisher 1961), although *S. alterniflora* may not be able to maintain an oxygenated rhizosphere during warmer seasons (Howes and Teal 1994). Oxygenation of the rhizosphere may help detoxify the sediments surroundings for the roots (Mendelssohn and Postek 1982). The structure of the aerenchyma differs between the tall and short form of *S. alterniflora,* such that the specific gas transport capacity is greater in the tall form than short form increasing the available oxygen in the associated sediments (Arenovski and Howes 1992).

Oxygenation of the rhizosphere also supports a variety of micro-organisms (Bagwell et al. 1998, Lovell et al. 2000, Leaphart et al. 2003, Casciano 2007). This includes nitrogen fixing species (Teal et al. 1979, Whiting et al. 1986, Bagwell et al. 1998) and mycorrizhal fungi that are important for plant function (Burke and Hahn 2000, Casciano 2007).

A third important characteristic of *S. alterniflora* is specialized osmoregulatory physiology that allows it to live in saline waters. In addition to the drought tolerance typical of C4 plants, *S. alterniflora* has at least two additional ways of coping with seawater. *Spartina alterniflora* uses

Na^+ as an osmoregulator rather than the K^+ anion used by many other plants, thereby reducing the costly maintenance of barriers to exclude Na^+ from the roots (Vasquez et al. 2006). Specialized glands in the leaves can also excrete excess salt (Anderson 1974, McGovern et al. 1979). The ionic composition of the excreted salt is different from sea salt, having much lower amounts of Ca, Mg, and $S0_4$ (McGovern et al. 1979).

Species of *Spartina* found in salt marshes have different leaf anatomies that can reduce water loss. In salt marsh species, including *S. alterniflora*, which are adapted for surviving water stress, the stomata are only on one side of the leaf, between ridges that close when the leaf rolls in response to water stress (Maricle et al. 2009), thereby reducing water loss.

Sea salt causes stresses on *S. alterniflora* in addition to that of osmoregulation. Sulfate ions are a significant part of sea salt, comprising 7.68% of total salts by weight (Table 1.2), and just under 14% of the cation composition by weight (Thurman 1983). Sulfates are used as an electron receptor by sulfate reducing bacteria, which are capable of continuing decomposition in the anoxic zone of marine sediments (Capone and Kiene 1988). These bacteria produce hydrogen sulfide that adds to the stress of marine wetland plants beyond anoxia alone (Raven and Scrimgeour 1997).

A fourth important characteristic of *Spartina* is that it has a higher tolerance of hydrogen sulfide than other angiosperms. Hydrogen sulfide (H_2S) is toxic to most plants, including *S. alterniflora,* when it is at high concentrations (Koch et al. 1990, Howes and Teal 1994, Pezeshki et al. 1998, Reddy and DeLaune 2008). There is evidence that H_2S can inhibit the growth of *S. alterniflora* beyond the inhibition expected from anoxic conditions alone (Koch and Mendelssohn 1989, Bradley and Morris 1990). *Spartina alturniflora* has specialized enzymes that oxidize H_2S and render it nontoxic (Kraus and Doeller 1999). There is also evidence that *S. alterniflora* can actively take up sulfide (Carlson and Forest 1982), either from seawater or oxidized sulfide (Raven and Scrimgeour 1997), and that it also has a higher minimum requirement for sulfates than other plants (Stribling 1997).

Table 1. 1. The six ions that account for 99.28% of the sea salt by weight (from Thurman 1983).

Ion	Percentage
Chloride, Cl-	55.04
Sodium, Na+	30.61
Sulfate, So_4^{-2}	7.68
Magnesium, Mg^{+2}	3.69
Calcium, Ca^{+2}	1.16
Potassium, K+	1.1
Total:	99.28

A fifth important characteristic of *S. alterniflora* is that it has significant foliar absorption of NH_4^+. Low amounts of nutrient absorption through leaves has been documented in a number of plants, including red spruce (Boyce et. al. 1996), pine and fir (Tomaszewski et al. 2003), and *Spartina anglica* (Bouma et al. 2002). In these cases the foliar uptake of nitrogen compounds fulfilled < 10% of plant requirements, and was not seen as significant in *Spartina anglica* (Bouma et al. 2002). Foliar absorption is important in submerged seagrasses, where it can account for 50-100% of the plant's nutrient requirements (Stapel et al. 1996, Terrados and Williams 1997, Lee and Dunton 1999). Nitrogen availability is seen as the limiting factor for *S. alterniflora* productivity (Valiela and Teal 1979), and anoxic sediment is thought to interfere with nutrient uptake by *S. alterniflora* (King et al. 1982, Bradley and Morris 1990, Chambers et al. 1998). The ability to absorb nutrients from estuarine water was tested by Toblias et al. (2007) using ^{15}N labeled $N0_3^-$. It was found that nitrate uptake was mainly through the root system in *S. alterniflora.* More recent work by Mozdzer (2009) found significant foliar absorption by *S. alterniflora,* which may be the only member of the Poaceae with this capability. In *S. alterniflora* the uptake of NH_4^+ is particularly efficient; organic nitrogen and nitrates are also absorbed, but with lower efficiencies (Mozdzer et al. 2011). It is estimated that *S. alterniflora* can meet up to 20% of its nitrogen requirement in low nutrient waters and over 100% under eutrophic conditions by uptake from the water column (Mozdzer 2009, Mozdzer et al. 2011).

A sixth important characteristic of *S. alterniflora*, is the ability to accumulate sediment. The survival of coastal marshlands presently is dependent on the critical balance between sea-level rise and accretion rate (Warren and Niering 1993). Coastal wetlands on the eastern coast of North America accumulate 0 – 1.2 cm yr^{-1} of sediment on average, and up to 8.5 cm yr^{-1} elsewhere (Reddy and DeLaune 2008). Some authors have found sedimentation rates higher than 1.2 cm yr^{-1} for wetlands on eastern North America (e.g., Warren and Niering 1993, Kolker 2005). While salt marshes generally keep pace with sea-level rise (Kolker 2005), this is not true for all salt marshes (Warren and Niering 1993, Reddy and DeLaune 2008). Sediment trapped in marshes comes primarily from terrestrial sources through runoff and river and stream inputs, the resuspension of marsh sediment, and a small amount of marine sediments drawn into inlets (New York State Department of State 2010).

The erect leaves of *S. alterniflora* slow water movement, facilitating sediment deposition and accumulation. Sediment is collected particularly along creek edges with tall form *Spartina*, and the presence of *Spartina* prevents sediment resuspension by tidal flows (Christiansen et al. 2000). Additional sediment is removed from the water column and deposited in pseudofeces by the ribbed mussel, *Geukensia demissa*, which is associated with *S. alterniflora* (Keunzler 1961, Jordan and Valiela 1982, Smith and Frey, 1985). Sediment nitrogen levels also increase in the presence of *Geukensia*, and both above and below ground growth of *S. alterniflora* are enhanced (Bertness 1984). Nitrogen is excreted into the water by *Geukensia,* mainly as ammonia, and byssal threads and mortality transfer small amounts of nitrogen directly into the sediment (Jordan and Valiela 1982). *Geukensia* also deposits phosphate to the sediment in pseudofeces (Keunzler 1961).

ASSOCIATED SPECIES

Besides *Geukensia*, there are many other species that are closely associated with *S. alterniflora*. Necton found in the creeks and ditches in a marsh on the south shore of Long Island, NY include:

the fish *Anchoa mitchilli*, *Apeltes quadracus*, *Anguilla rostrata*, *Cyprinodon variegatus*, *Fundulus heteroclitus*, *Fundulus luciae*, *Fundulus majalus, Lucania parva*, and *Mugil curema*;

the shrimp *Crangon septemspinosa and Palaemonetes pugio*; and the crabs *Carcinus maenas* and *Callinectes sapidus*, (Corman and Roman 2011). Benthic invertebrates generally associated with *S. alterniflora* include: fiddler crabs *Uca pugnax, Uca pugilator* and *Uca minax,* the snail *Melampus bidentatus*, and some insects including *Orchelimum* spp., *Conocephalus spartinae, Triginotylus heuleri,* and Emphidridae flies (*personal observation*). The diamond-backed terrapin (*Malaclemys terrapin*) is an obligate salt marsh resident, and an attempt is being made to propose it for listing as a threatened species on the basis of habitat loss (Russel L. Burke, Hofstra, *personal communication*). Many birds are also found in *Spartina* marshes, including some that are Federally or State listed as threatened, endangered or of special concern (Table 1.2). Marsh edges are particularly important habitat for birds, as these edges provide important foraging habitat for aquatic birds (O'Connell et al. 2010).

Table 1.2. A sample of birds associated with *Spartina* marshes, including those species that are on either the Federal or New York State lists of endangered and threatened species (Federal Fish and Wildlife Service 2011, New York State Department of Environmental Conservation 2011).

Common Name	Scientific Name	Relation to *Spartina*	Federal List	New York List
Black Rail	*Laterallus jamaicensis*	Obligate	Not listed	Endangered
Sea Side Sparrow	*Ammodramus maritimus maritima*	Obligate	Not listed	Special concern
Clapper Rail	*Rallus longirostris*	Obligate	Species of concern	Not Listed
Peregrine Falcon	*Falco peregrinus*	Hunts on marsh and nests nearby or on structures	Not listed	Endangered
Roseate Tern	*Sterna dougallii dougallii*	Hunts on marsh and nests nearby	Endangered	Endangered
Short-Eared Owl	*Asio flammeus*	Can hunt and nest in Spartina marshlands	Not listed	Endangered
Pied-Billed Greeb	*Podilymbus podiceps*	Winter feeding	Not listed	Threatened
Least Bittern	*Ixobrychus exilis*	Winter feeding	Species of concern	Threatened
American Bittern	*Botaurus lentiginosus*	Winter feeding	Species of concern	Special concern
Common Tern	*Sterna hirundo*	Nesting and feeding	Species of concern	Threatened
Least Tern	*Sterna antillarum*	Preferred Summer feeding	Endangered	Threatened
Black Skimmer	*Rynchops niger*	Summer feeding	Species of concern	Special concern
Northern Harrier	*Circus cyaneus*	Preferred hunting, mostly Winter	Not listed	Endangered
Osprey	*Pandion haliaetus*	Hunts edges, nests on structures	Not listed	Special concern

SALT MARSH ECOSYSTEM FUNCTION

There is increasing concern about the loss of marshes and the ecosystem services they provide (Costanza 2008, Feagin 2010). Salt marshes are essential habitats within the estuarine ecosystems of the east coast of North America. They are known to harbor a unique assemblage of species and provide important ecosystem functions, including spawning sites for fish, and feeding and nesting sites for many birds (Christy et al. 1981, Burger et al. 1982, Erwin et al. 1994, Shriver et al. 2004, Edinger and Howard 2008, McGowan and Corwin 2008, O'Connor et al. 2010). They are essential habitat for a wide range of species (see above). Salt marshes can moderate wave action during coastal storms (Wayne 1976, Knutson 1988), thereby reducing damage to other shorelines and human construction (Möller et al. 1999, Möller et al. 2001, Costanza et al. 2008, Faegan et al. 2010).

Salt marshes are also known for their high levels of productivity. Salt marsh communities are a major carbon source for life in the estuary (Odum and De la Cruz 1967, Valiela et al. 1985, Cranford et al. 1989, Dittel et al. 2006, Quan et al. 2007, Wang et al. 2007, Montemayor et al. 2011). The average annual production of *Spartina alterniflora* alone has been estimated to be 399 - 1169 g C m^{-2} (Reidenbaugh 1983). For the Hempstead salt marshes on Long Island, NY, which are the focus of this dissertation, the estimated annual production of *S. alternaflora* ranges from 508.3 to 827.3 g C m^{-2} (Udell et al. 1969). Algae and microorganisms beneath the grass canopy add a large additional annual production that has been estimated to be high, but less than that of *S. alterniflora*. Pomeroy (1959) estimated the annual gross algal production as 200 g C m^{-2}, and Raalte et al. (1976) estimated annual production at 105.5 g C m^{-2}, or about a quarter of the production of the grass at their site, and about 20% of the total productivity of the marsh. In Delaware marshes the gross algal production was determined to be one third of the combined angiosperm production (Gallagher and Daiber 1974). Pinckney and Zigmark (2006) estimated that benthic algal annual production in North Inlet Estuary of South Carolina exceeded that of phytoplankton, but was still less than *Spartina*. Epiphitic algae on *Spartina* are another source of productivity, adding an average of 24.8 (range 15.3 -45.5) mg C (m^2 of substrate area)$^{-1}$ h^{-1} (Jones 1980). Recent stable isotope studies show that algae are the major source of carbon to the salt marsh food web, which underlines the importance of algal productivity (Sullivan and Moncreiff 1990, Cuyrrin et al. 1995, Wainright et al. 2000).

Wetlands are known for their high rates of denitrification (Seitzinger et al. 2006, Jordan et al. 2011) and for accomplishing denitrification with lower rates of NO_2 production than other habitat types. NO_2 is a powerful greenhouse gas, and healthy wetlands can remove more excess nitrogen while producing less NO_2 byproduct than other habitats (Schlesinger 2009). Salt marshes are important for denitrification (Wigand et al. 2004), but do produce NO_2 (Lindau and DeLaune 1990). The anoxic sediments in salt marshes, particularly the creek bottoms, are locations where denitrification generally occurs (Kaplan et al. 1979, Hamersley 2001). Denitrifying bacteria are both supported by organic mater from *S. alterniflora* and competing with the living cordgrass for the available NO_2 or NH_4+, producing seasonal changes in denitrification rates (Sherr and Payne 1978, Thompson et al. 1995, Hamersley and Howes 2005). Both natural increases in nitrification rates by nitrifying bacteria in <u>*Spartina's*</u> oxygenated rhyzosphere and fertilization will increase denitrification rates (Lincau and DeLaune 1991, Hamersley 2001, Hamersley and Howes 2005). *Spartina* can also remove nitrate from the water column. In an isotope tracer study, it was found that *Spartina alterniflora* was able to take up as much as 16.8% of ambient nitrate from tidal flood water each tidal cycle (Drake et al. 2009).

Thus, given their extensive importance as ecosystems, salt marshes have been the target for restoration and protection from loss.

CAUSES AND CONSEQUENCES OF MARSH LOSS

Both natural and anthropogenic influences have been identified to be important factors that can lead to loss of salt marshes. Historical records show that a large percentage of the marshlands in Eastern North America have been lost to direct physical destruction associated with construction of roads, houses, sewer treatment plants, and other forms of urban development (Dahl 1990, Kraft et al. 1992, Tiner et al. 2002, Ciappetta 2010). Filling for expansion of agricultural uses, the construction of roadways, airports (e.g., Kennedy and La Guardia airports on Long Island), golf courses, marinas, parks, and many other recreational and commercial uses have also caused the loss of large tracts of marshlands (Tiner 1987, Tiner et al. 2002, Carlisle et al. 2005, New York City Department of Environmental Protection 2007). The need for sources of landfill often resulted in the dredging of nearby salt marshes and shallow bays, and included the excavation of the large deep borrow pits (e.g., Grassy Bay in Jamaica Bay) that remain today (Swanson and Wilson 2008, K.K. Olsen, Montclair State University, NJ, *personal communication*).

In some cases, marshes were removed in association with the formation of deep water ports, including New York Harbor, Jamaica Bay, Boston Harbor, and Baltimore Harbor (Carlisle 2005, Badger 2007, K.K. Olsen, Montclair State University, NJ, *personal communication*). On Long Island, the western part of Jamaica Bay, next to the section of Gateway National Park formerly occupied by Floyd Bennett Airfield, was deepened for this purpose as far north as Canarsie, Brooklyn, NY, but never used (Anonymous 1907, Anonymous 1931, K.K. Olsen, Montclair State University, NJ, *personal communication*). Marshlands were also dredged for the creation of straightened or widened navigational channels (Herter et al. 2003, Olsen 2011). These modifications also change the hydrological properties of the surrounding water, potentially influencing neighboring salt marshes by altering sediment transport (Teal 2001, Renfro et al. 2010), and the local tidal range (Swanson and Wilson 2008).

Marshlands have also been affected by both filling and controlled flooding, which were once recommended for the control of mosquitoes (Herms and James 1961, Erwin et al. 1994). Diking for salt hay production was also common on some marshes, leading to subsidence when the dikes were abandoned (Philips 1986). In general, alteration of water flow through the marsh tends to alter the marsh sediment (Anisfeld et al. 1999) and vegetation (Roman et al. 1984).

Large scale natural and anthropogenic changes, beyond the control of local managers, can also induce marsh loss. Sea level is expected to continue to rise due to the addition of melt water from glaciers and thermal expansion as ocean water warms (Downs et al. 1994, Kemp et al. 2011). Apparent sea level rise in the New York vicinity is also caused by subsidence, which can occur from large-scale isotonic adjustments, due to the continued northern rebound after end of the last glacial period (Gornitz 1995, Gornitz et al. 2001). Subsidence can also occur from local compaction of marsh sediments or in underlying deposits (Bartholdy et al. 2010). Natural wave action is another source of marsh loss, particularly wave impact from storms (Pye 1995).

If there is an insufficient supply of sediments and production of organic mater, salt marsh accretion can be outpaced by sea level rise, resulting in increased flooding of marshlands and further loss (Orson et al. 1985, Philips 1986, Orson et al. 1987, Gammill and Hosier 1992, Downs et al. 1994, Cahoon and Reed 1995, Feagin et al. 2010). In the New York Harbor area, as recorded at the Battery Tide Gage, sea level rose at a mean rate of 0.30 cm yr^{-1} from 1905 to 2003 (Kolker 2005).

Mendelssohn and Kuln (2003) added a sediment slurry to the surface of a Louisiana, marsh to determine an optimal sediment elevation for the restoration of *S. alterniflora* marshes. The test marsh had subsided 0.94 cm y^{-1}, relative to mean sea-level, from 1944-1988. They found that in the test marshes where 40-60 cm of sediment were added, *S. alterniflora* had increased growth compared to reference plots and test plots where less sediment was added. In an earlier study done in Georgia, existing *Spartina* did not recover when buried under 61 cm or 90 cm of sediment. Regrowth succeeded when 30, 23, 15 or 8 cm of sediment was added, and new seedlings more than equaled the biomass lost when the marsh was buried. (Reimold et al 1978).

For a marsh in the Gulf of Mexico, experiments were use to determine if differences in the sediment height of the marsh affected recovery from disturbance (Stagg and Mendelssohn 2011).

The odds of the reestablishment of > 95% cover was significantly higher with low and moderate increases in marsh elevation (11-20 cm relative to ambient level) than either high levels of sediment addition (19-26 cm relative to ambient level) or no sediment addition. This work indicates that the optimal elevation relative to sea-level in many locales is likely above current marsh levels (Stagg and Mendelssohn 2011), and supports the contention that many salt marshes along the Louisiana Coast, where relative sea-level rise from 1944-1988 was 5 times the global rate, are loosing ground in the balance between sedimentation and sea-level rise (Mendelssohn and Kuln 2003).

LEGAL PROTECTION OF MARSHLAND

Although many marshlands are legally protected because of the ecosystem services they provide (Bertness et al. 2004), increasingly, there is good documentation that salt marshes are being lost (Hartig et al. 2002). Anthropogenic activities are often suspected as the major factor resulting in losses of this valuable resource (Bertness et al. 2002, Bertness et al. 2004, Kolker 2005, Bertness and Silliman 2008, Gedan et al. 2011).

Marshlands were subject to direct physical destruction in the US until the institution of tidal wetlands protection laws by local, state, and federal governments in the 1960s and 1970s. The marshes in western Long Island (the focus of this study) first received protection at the local level in 1968. Federal protection for marshlands was established by the Clean Waters Act of 1972 and was followed by New York State Tidal Wetlands Protection act in 1973. Although these protections stopped direct destruction of salt marshes and reduced loss considerably, there continues to be a total net loss of marsh area. From 1974 through 2004, Tiner et al. (2006) documented continued losses at 6 Connecticut marshes. Kolker (2005) measured the change in total area for selected islands within the South Shore Estuary Reserve (SSER) of Long Island, New York, for a ~25 year period starting in 1974. He estimated a loss of 2.0 ha $year^{-1}$ from marshes in Middle Bay and 1.7 ha $year^{-1}$ from those in East Bay. In order to conserve these wetlands, we need to understand the mechanisms underlying this background rate of loss.

Although a number of natural and anthropogenic impacts are thought to affect the rate of change in salt marsh extent through time, including changes in rates of erosion, submergence or accretion, few, if any, long term assessments are available to compare the effects of different types of impacts that lead to salt marsh loss, or determine which impacts are most important for marsh loss in a given marshland (but see Smith 2009). In addition, there is a need to assess local drivers of marsh loss (edge recession) or gain (through expansion) and to test relative importance of factors that are manageable (i.e., boat wakes) versus those that are not (i.e., wind driven waves or sea level rise) so that we can better manage and protect essential marsh habitat. Therefore, work that identifies and ranks important factors contributing to the loss or gain of marsh edges is critical for the formulation of ecosystem-based management plans. It is only after we understand these mechanisms that we can implement successful management interventions.

STUDY SYSTEM

The focus of this study was the Hempstead marshlands within the South Shore Estuary Reserve (SSER) of Long Island, New York, which is within the Town of Hempstead. The marshlands and bay bottoms are commons under the Dongan Patent of 1644, a colonial era deed that established the township of Hempstead (Van Wyke 1935). This marshland extends from the New York City line near Far Rockaway and Debs Inlet on the west to the Line Islands along the Town of Oyster Bay line on the east, and is bordered by the City of Long Beach to the south (Fig. A.1). As is evident from aerial photography, this estuary is highly urbanized (Fig. A.2). The City of Long Beach, along the southern edge, has over 33,000 citizens. The Town of Hempstead has just under 760,000 citizens, with a population density of over 1,500 citizens km^{-2}, and is within the New York City metropolitan area, which includes just under 19 million people.

The study site is a part of the SSER that is known as the Hempstead Bays (40º 37' 10” N, 73º 36' 41” W). It is usually divided into three bays, West Bay, Middle Bay and East Bay, which are unusual within the SSER due to the large tracts of *Spartina* marsh islands they contain. Historical maps (Fig. A.3) and early photographs from 1926 (Fig. A.4) indicate that large extents of fringing *Spartina* marshes still existed in 1926. The salt marsh extent prior to human modification in the 1800s is difficult to determine precisely, but it appears that it was over 6,400 hectares (Ciappetta 2010). Originally, the waters between the marshlands were not deep. Existing US Coast and Geodetic bathymetric surveys from 1880 indicate that typical depths were less than 1 meter and rarely below two or three meters below tidal datum (mean low water, MLW) (Fig. A.5).

Local tide gauges have been in place since the early 1970s at six locations in Hempstead Bay (Fig. A.6). They indicate that sea level is rising, and the rate of sea level rise varies only slightly among locations (Town of Hempstead Department of Conservation and Waterways, *unpublished data*). At Seaford, NY, located at the northeastern corner of the study area in Hempstead Bay, the gauge recorded a 15.21 cm increase in mean sea level from 1975-2001, and a 13.93 cm increase in mean highest daily tide. This averages to 0.59 cm yr^{-1} increase in mean tide and a 0.54 cm yr^{-1} increase in highest daily tide, similar to the sea level rise seen in NY Harbor (Fig. A. 7). The Freeport gauge, located at the northern edge of the study area in Hempstead Bay, was active for just under 19 years prior to replacement with a USGS gauge in 1999, and documented a 10.548 cm change in mean sea level, from January 1975 to November 1993, or an average annual sea level rise of 0.555 cm yr^{-1}. Overall, local relative sea level changes appear to be comparable to other nearby locations from 1970 to the present, including the Battery in New York harbor (Fig. A.7). The increases in mean sea level are somewhat greater than the regional estimates of 0.30 cm yr^{-1} at the Battery for 1905 to 2003 (Kolker 2005), or the 0.285 cm yr^{-1} rate for 1856 to 2011(US Coast and Geodetic Survey, NOAA, Fig. A.8), suggesting that the rate of sea level rise has increased over the past century.

Like many *Spartina* marshes in North America, this system has suffered many known anthropogenic impacts. The major historical impacts resulted from the re-engineering of this estuary by dredging and filling during the first part of the twentieth century (Herter et al. 2003), but there is some evidence of earlier construction and filling from older maps (Fig. A.3; A.11) and town documents (Fig. A.9). In 1926 there were 5,461 ha of marsh within the study site. Between 1926 and 1983, 2735 ha of marsh, slightly over 50% of the marsh, was lost to a combination of filling, channelization, and natural losses (Ciappetta 2010). After protection of this marsh in 1968, there was continued loss of 6.5 to 10 ha yr^{-1} from 1983-2004, leaving only 2496 ha (45.7% of the 1926 area) of marshland in 2004 (Ciappetta, et al. 2009). Protection of this marsh has slowed the loss, but did not stop it.

There are many challenges with quantifying changes in salt mashes through time. Prior to the widespread use of aerial photography, data about the extent of salt marsh formations was limited to a few sources, including surveys from the U. S. Coast and Geodetic Survey (USCGS), the U.S. Geological Survey (USGS), the U.S. Army Corps of Engineers (USACE) and the National Oceanographic and Atmospheric Administration (NOAA) or other nautical charts. For example, Kearney et al. (1988) looked at salt marsh changes in Nanticoke estuary, Chesapeake Bay, using a USCGS map from 1862, USGS map from 1980, and aerial photography from 1978 and 1985. Grammill and Hosier (1992) used USACE maps from 1857, 1888, 1934, and 1980 to provide reference data for using aerial photographs from 1949, 1966 and 1984. Downs et al. (1994) used aerial photography from 1938, 1952, and 1992 in combination with maps from 1849, 1901, 1942, 1973 to study changes in Bloodsworth Island, Maryland, but noted that the inferior detail in the maps impeded the measurement of internal changes and upland edges. Other studies of the changes in salt marshes through time have used just aerial photography, including Philips (1986) who compared 1940 and 1978 photographs of Delaware Bay and Tiner et al. (2006) who used photographs from 1974, 1981, 1986, 1990, 1995, 2000, and 2004 to study trends in 6 Connecticut salt marshes.

Presently, the best method to use to estimate changes in the marsh shoreline through time, rates of change, and local changes in the rates for each section of marsh edge, is aerial photography through time within a geographic information system (GIS) framework. Examples of this technique include work on Jamaica Bay, N.Y., where quantitative changes in marsh area were examined from 1924 to 1974, 1974 to 1994, and 1994 to 1999 (Hartig et al. 2002), and another study of this area from 1951 to 1974, 1974 to 1981, and 1981 to 2003 (Gateway National Recreation Area, 2007). The Western Bays of the SEER were similarly studied for changes from 1926 to 1956, 1956 to 1983, 1983 to 1994, and 1994 to 2004 (Ciappetta 2010, J. Ciappetta, Adelphi University and Conservation and Waterways, Town of Hempstead, B.A. Christensen Adelphi University *personal communication* and J.P. Browne *unpublished data*). These studies measured changes in total marsh surface area through time and related these changes to potential causes of marshland loss. Losses due to filling and construction are easily confirmed from photographic evidence. Similarly, losses due to the physical removal of marsh through dredging are straightforward to confirm. Some of these studies used too few years for a clear resolution of temporal trends. These studies are also limited to measuring changes in marsh area at the scale of several hectares resolution, and cannot detect effects acting locally or on smaller spatial scales.

To examine salt marsh loss through time, I tracked the change in location of specific points on the edge of the salt marsh over an eighty-one year time period using aerial photographs. By focusing on changes in the location of the edge of the marsh, I was able to detect changes in the marsh that could not be resolved with prior techniques, and address mechanisms that could act on spatial scales smaller than hectares of marsh area. Scale influences many aspects of ecology.

Studies covering wide spatial or long temporal extents, using large grain sizes, can average out effects that may be important and can be distinguished and studied with smaller grain sizes, and consideration of smaller spatial or shorter temporal extent (Wiens 1989). For example, studies that measure the area of entire marsh islands implicitly use a large grain size and resolve only large scale spatial trends (e.g., Kolker 2005, J. Ciappetta, Adelphi University and Conservation and Waterways, Town of Hempstead, B.A. Christensen Adelphi University *personal communication* and J.P. Browne *personal observation*).

By focusing on changes at specific points along the salt marsh edge, I was able to test the effects of potential mechanisms that act at local scales within a marsh. I was able to test several a priori hypotheses regarding factors believed to be important for salt marsh loss, including those that are considered natural, and those due to anthropogenic or secondary anthropogenic impacts on salt marsh edges. Specifically, I tested the following hypotheses: 1. Channelization leads to continued loss at edges that have been cut, 2. Boat traffic impacts salt marsh loss, 3. Nutrient loading increases marsh loss, 4. Sediment in the water may increase accretion while proximity to sediment sinks induces loss, 5. Storm impacts increase salt marsh edge loss due to erosion, and 6. Several natural conditions, including fetch (distance across open water to another shore), experienced tidal range and the velocity of water at tidal exchanges can increase or decrease edge loss.

Chapter 2 HYPOTHESES FOR MARSH LOSS

There are a number of published hypotheses about factors that contribute to salt marsh loss locally as well as regionally. This study focused on assessing the relative importance of factors that contribute to the gain or loss of salt marshes within a single estuary as detected by the change in location of the edge of the marsh (rather than loss in the middle of otherwise contiguous marsh). Here I discuss the hypotheses that are tested in this study. These hypotheses are not mutually exclusive, and each factor could potentially interact and contribute to salt marsh loss.

CHANNELIZED EDGE

Channelized edges are marsh edges that were artificially created by dredging away some section of salt marsh. A number of new marsh edges were created by cutting through marsh during engineering projects between the 1880s and 1960s, and were identified by comparing the 1880 ground survey of Hempstead Bay and aerial photographic surveys that were started in 1926 (Herter et al. 2003). For example, Sea Dog Channel was cut to provide fill for the construction of Loop Parkway between 1926 and 1950 (Fig. A.11). Another example was the widening of Hog Island and Sturm Channels for use by oil tankers in the 1950s (Fig. A.10).

Marshes with channelized edges may have different rates of loss than marshes with natural edges. Channelized edges are initially very steep, often nearly vertical, and these edges would be likely to slump soon after the channels were first cut until they achieve a stable configuration or angle of repose (Roshchupkin 1975, Mehta and Barker 1994, Fowler et al. 1995). Edges that are sharply cut could be more affected by wave-driven erosion due to the lack of stabilizing root systems. After reaching a stable configuration, however, the dominant factors affecting marsh loss was predicted to be similar to those that affect natural channels of the same width, depth, current flow and boat traffic. However, the length of time needed for stabilization is unclear.

NAVIGATIONAL CHANNELS AND BOAT WAKES

Wakes created by shipping or the use of power boats are often implicated in shoreline loss and marsh edge loss, although some studies indicate that this connection is small compared to other factors creating waves, and may be site specific (Zabawa and Ostrom 1980). Other studies show a strong connection between maritime traffic and salt marsh erosion (Darava and Moore 1997). Additional research has started to look at the effects of the passage of large vessels such as barges and fishing trawlers (Davis et al. 2009). Presently, large amounts of funding are expended marking and enforcing marsh protection zones near boating channels and areas where boat wakes could impact the marsh, therefore a clear understanding of the contribution of the effects of shipping on salt marsh loss is needed before manages can efficiently allocate money and effort. At present, the importance of boat wake impact relative to other factors is unclear.

LOCAL HYDROGRAPHIC IMPACTS.

Local hydrographic impacts are non-anthropogenic influences related to the effects of the physical structure of the estuary and typical tide and wind conditions on water motion. Local hydrographic impacts include tidal water flow and non-storm wind driven waves, which can affect specific sections of shoreline. For example, currents can remove sediment from one location, depositing it elsewhere. Additionally tidal amplitude may affect the survival of *Spartina* at the edge of marshes because it determines the period of flooding and exposure to wave action.

NUTRIENT LOADING

Nutrients originating from treated sewerage or neighborhood storm runoff can potentially influence marshes. Recent research has shown that *Spartina* can uptake ammonia directly from the water column, and this uptake can account for over 100% of the immediate nitrogen requirements of *Spartina* under eutrophic conditions (Mozdzer 2009, Mozdzer et al. 2011). Increased ambient nutrient levels also lead to measurable increased nitrogen uptake by the *S. alterniflora* (Deegan et al. 2007). There is evidence that wetlands in general are efficient at removing nitrogen from the water flowing across the marsh (Schlesinger 2009, Jordan 2011). At nitrate concentrations less than those found throughout most of the study site, up to 16.8% of available nitrate can be taken up out of each cycle of tidal flooding (Drake et al. 2006), implying that there are some mechanisms for the rapid transfer of nutrient from the water to the marshlands and marsh grasses.

Most experimental studies have found that *Spartiana alterniflora* is nitrogen limited and that the direct application of nutrients to salt marshes increases above ground growth in *S. alterniflora* (Valiela and Teal 1974, Van Raalte et al. 1976, Mendelssohn 1979, Morris 1991, Dai and Wiengert 1997, Morris and Bradley 1999, Tyler et al. 2003, Deegan et al. 2007) and sediment accumulation (Teal 2001, Morris et al. 2002, Mudd at al. 2010), even while reducing direct organic matter content of the marsh sediment (Morris and Bradley 1999). In addition, *Spartina alterniflora* requires more available nitrogen at higher salinities (Bradley and Morris 1992). Added nutrients greatly increase the leaf area and photosynthetic capacity of the short form *Spartina alterniflora* (Dal and Wiengert 1997). Nutrient loading may increase the competitive ability of *S. alterniflora*, enabling an expansion to higher shore habitats at the expense of species diversity otherwise there (Donnelly and Bertness 2001, Bertness et al. 2002, Bertness and Silliman 2004).

Kolker et al. (2005) studied sediment cores and pore water chemistry throughout marshes on Long Island and argued that marsh loss in Jamaica Bay resulted from a buildup of sulfides to toxic levels. They linked this sulfide accumulation with eutrophication due to high nutrient loading, and possibly organic material loading from sewage effluent. Alternatively, nutrient additions may augment species that then damage *Spartina alterniflora* marshes (Teal 2001), or increase the palatability of the grass and increase herbivory (Bertness et al. 2008, Alberti et al. 2011). There are also indications that nutrient loading decreases below ground growth (Agren and Ingestad 1987, Levin et al. 1989, Turner et al. 2009) and destabilizes marsh sediments (Turner 2011).

Some management programs in Jamaica Bay, NY, and public discussions are based on the assumption that nutrient loads are solely destructive for marshes, and often spend large amounts of funding to remove excess nutrients from the water (DePalma 2007, Gateway National

Recreational Area 2007). The ubiquitous influence of human activity at this time insures that an estimate of the pre-colonial levels of nutrient sequestration and recycling within pristine estuaries is no longer simple to determine. The actual interaction between nutrient concentrations and *Spartina* across a wide range of nutrient levels under field conditions is still unclear.

Other water quality factors in addition to nutrients may affect affect *Spartina* marshes. High salinity, for example, is known to stress *Spartina alterniflora* (Bradley and Morris 1992). Many chemicals, including some with potential biological activity, which could impact *Spartina* marshes, have been identified in sewerage effluent and runoff (Rodenburg 2006, PANYNJ 2011).

SEDIMENT SUPPLY BALANCE

Marshlands depend on healthy, growing *Spartina,* whose physical presence results in the accumulation of sediment carried to the marsh by the surrounding water and upland runoff. At present, continued sediment input is needed to keep pace with or exceed compaction and sea level rise (Niering and Warren 1980, Warren and Niering 1993, Teal 2001). Recent measurements on the south shore of Long Island, New York, indicate that marsh sedimentation has matched sea level rise (Kolker 2005).

A potential influence that could impede marsh gain or accelerate marsh loss is the transport and sequestration of sediment in deep water, below where it would be resuspended, and making it unavailable to marshes. Sediments that are resuspended in shallow water areas can be redistributed through the actions of waves or tidal currents and add to marshlands. There is evidence that sediment resuspended by nearby dredging can contribute to vertical accretion in *Spartina* marshes (Cademartori 2001). Sediment re-suspended by storms, including hurricanes, may also accumulate on marsh surfaces (McKee and Cherry 2009). Within the bays of the south shore of Long Island, there are few, if any, natural deep locations that would act as sediment sinks. In this area, many of the natural deep locations are scoured by water movement and are therefore represent a source of sediment to marshes. From the late nineteenth to mid twentieth century, much of the shoreline construction involved dredging nearby bay bottoms to fill in marshland, creating borrow pits. These borrow pits now reach depths of 10-20 m and provide deep locations that could trap sediments that would otherwise be available to marshes.

EFFECTS OF STORMS

The erosional energy imparted by storms is typically more important than the cumulative erosion caused by smaller waves between storm events. The energy from waves is proportional to the square of the height of the waves, and wave height is a function of wind speed, duration and fetch (the length of open water over which wind travels), and water depth (Denny 1988). Winds with speeds in the top 1-10% of those experienced annually at a given site cluster within a narrow directional range. Thus, the highest winds recorded over many storm events, will generally be from similar directions, despite the shifting winds during any given storm event. The impact from storms will be cumulative across many storms. The size and effects of storm driven waves can vary over small spatial scales due to small differences in the direction of the shoreline relative to prevailing waves, or the nearby water depth, and submarine topography, all of which can affect local patterns of erosion and deposition of sediments.

A DIFFERENCE BETWEEN EARLY 20TH CENTURY DEVELOPMENT AND LATE 20TH CENTURY URBANIZATION

Proximity to urbanization has been hypothesized to contribute to salt marsh loss. Kolker (2005) and Kolker et al. (2005) found that the general proximity to urbanized areas was correlated with increased rates of salt marsh loss for Long Island marshes. Urbanization is correlated with a number of factors, including greater nutrient loading, more storm runoff, increased sediment runoff during times of development and expansion, decreased sediment runoff once areas are developed, and possibly increased recreational boating activity. One test of this hypothesis would be to determine if there are differences in salt marsh loss during the transition from rural conditions versus after the urban expansion of the 1950's and 1960's was complete.

Chapter 3 METHODS

STUDY LOCATION

The Hempstead Bay study site (Fig. A.1) currently includes approximately 7,900 ha of wetlands and close to 2,500 ha of salt marsh, mostly covered with *Spartina alterniflora*. Patches of *Spartina patens*, *Distichlis spicata, Salicornia europaea* and other halophytes dot the marsh. *Phragmites australis*, *Iva frutescens*, *Baccharis halimifolia* and other less salt tolerant plants from a fringe along higher ground. These wetlands are typically grouped into the West Bay, Middle Bay and East Bay. Geologically, Long Island, New York, is a glacial deposit, with moraines, kames, pothole lakes, and other glacial formations. The south shore of Long Island, including the study site, lies on the outwash plain, formed of sediments carried by meltwater that overtopped the moraine system formed by the Wisconsinian ice sheet (Bennington 2003). As sea levels rose after the end of the glacial period, the wetlands within the study area formed as a shallow estuary, with a few deeper tidal channels and multiple inlets, behind barrier beaches. This estuary is mesotidal. It lacks distinct river deltas, but has meander streams, point bars, fine grained sands and silts on flats and marshes. Jones Inlet has both an ebb-tidal delta and a flood-tidal delta (Biggs 1982).

Since the late 1600s, the Hempstead Bays and their watersheds have incurred many anthropogenic modifications that were recorded on maps and aerial photography. Shallow bays have been deepened with borrow pits, some channels deepened and widened for coastal tankers, and other sections were filled. Human activity in this estuary has included fishing, hunting and boating since precolonial times. During the colonial period, woodland was cleared for farmland.

Starting in the 1800's there have been modifications to the marshlands for the construction of housing, roads, coastal shipping and recreational areas within the wetlands. These wetlands were also used as a disposal site, with two landfills for garbage that operated from the late 1960s to the early 1990s on what was once marsh. Since the 1950s, 5 sewerage outfalls have released secondarily treated effluent into the water (Fig. A.16). A separate storm runoff system sends street runoff into the estuary and its tributaries through approximately 1,000 outfalls. In this urbanized environment no natural upland margins exists. About half of the wetland edges are now hardened to protect houses, and the remaining half adjoins recreational areas and parkways that were not hardened. Recreational and commercial fishing, recreational boating, hunting, birdwatching and the general enjoyment of the vista continue in the Hempstead Bays to this day.

DATA FROM MAPS CHARTS AND AERIAL PHOTOGRAPHY SETS

There are many years of maps and aerial photographs available for Hempstead Bay since the 1600s. Maps starting in 1880, aerial photographs starting in 1926, and over 30 years of tidal and water quality measurements were used in this study. For this study, the response variable of interest was the gain or loss of salt marsh as detected by the change of marsh edges at 500 randomly generated points along the 1994 marsh edge, which was determined from aerial photography using GIS software. Independent variables were derived from water quality data, computer simulations and distances measured in GIS software.

Photographic data sets used in this study

I used series of 12 sets of aerial photographs from 1926, 1950, 1956, 1966, 1973, 1978, 1983 1989, 1994, 2000, 2004, and 2007, covering 81 years (Table 1). These photographic sets include all of the marshlands within the western bays of the SSER that collectively are known as Hempstead Bay, a section of the estuary that presently covers approximately 7,900 ha. However, before use in the study, a number of technical difficulties in the photographs were addressed.

The first difficulty is that some distortion is inherent in any attempt to project an object from one dimensionality to a lesser dimensionality. At the most basic level, projecting the curved surface of the earth onto into a map coordinate system always introduces distortion called map distortion (Snyder 1993). When an aerial photograph of a three-dimensional surface is taken from a single point above a landscape, additional distortions are introduced by this perspective projection. The distortions in a perspective projection come from two phenomena, scale variation and relief displacement (Lilles and Kiefer 2000). Scale variation is produced by the foreshortening effect at the center of the field, the photograph's principal point radially, when objects appear large relative to more distant objects at the edges of the image. Relief displacement occurs when features, such as hills, seem to lean away from the principal point in the photograph.

Orthoreferencing is one technique for removing distortions by using detailed information about the flight, including altitude, camera focal length, the attitude of the airplane, ground control points, digital elevation models and other information not available with historical photographs. A digital elevation model (DEM) is a digital data set that contains topographic information that is necessary for the removal of relief displacement from aerial photographs. Typically, the USGS topographic DEM data have a 30 m vertical increment, reducing their utility in relatively flat locations. Even the Nassau County, New York, DEM, which has 0.61 mcontours, fails to accurately capture the extremely low level of vertical variation found in marshlands. Fortunately, this lack of vertical displacement also made full ortophotographic processing unnecessary for most of the photographs used in this study. The 2000 and 2004 orthophotos were used to process the older photography.

Another difficulty is that the intended uses of a spatial data set will influence the choice of projection, which involves many trade-offs. For example, geographical latitude and longitude systems enhance navigation with a single system for the entire surface of the Earth, but introduce inaccuracies into the measurement of area and distance. The US State Plane System preserves distance and area measurement within a given zone, but at the expense of using many incompatible zones, making it unsuitable for navigation. Points within each zone are located as distances north and east of single point. Two different projections are used in the State Plane system. The Universal Transverse Mercator projection (UTM) that minimizes distortions for strips of land that are longer north to south than east to west. Lambert Conic Conformal (LCC) minimizes the distortion of area measurements for strips of land that are longer east to west (Snyder 1993). An additional choice is the particular model of the earth's surface that the map is projected from. Recent state plane projections are based on the North American Datum of 1983 (NAD83). New York State is divided into 4 incompatible zones measured from 4 different points, the Long Island and Western Zones that use LCC, and the Eastern and Central Zones that use UTM. Accurate measurements are only possible within a single zone. Recent technological advances are making it possible to use spatial data more efficiently in scientific studies. Geographic changes are now measured in standard coordinate systems from digital map data and referenced digitized aerial photography, using GIS software or other programs capable of processing spatial data.

Blue Marble Geographic Geographic Transformer® Version 6.0 (Blue Marble Geographics Incorporated) was used to reduce the error introduced by perspective distortion and project the images into NYS NAD83 LI zone. This software was chosen to produce referenced images that are portable to multiple GIS environments and, therefore, more suitable for future use by subsequent investigators using various different GIS software.

The reference data were the more recent photographic data sets of the study site, 1994 through 2007, which were orthophotos available from The New York State Department of Technology (NYS DTech) and online through the NYS GIS clearinghouse. Aerial photographs from NYS DTech are derived from flights that were flown specifically for the production of GIS compatible orthophotography and are highly accurate. The local NYS Dtech orthophotographs were referenced in US State Plane NAD 1983 New York Long Island Zone US Survey Feet projection that is based on Lambert Conic Conformal. Because the entire study site is included within the Long Island Zone, this study was able to take advantage of the more accurate measurements that are possible when using this system. The 2000 and 2004 New York State orthophotos were used as the reference data for all of the older photography in this study, and all measurements taken within this system were converted to meters before analysis.

Aerial photographs prior to 1994 were photographic prints that required digitization (scanning) and referencing before use. As an alternative to orthoreferencing, aerial photographs were referenced using points in more recent referenced photography that had known geographical locations (control points).

Control points for the historical photographic sets prior to 1994 were selected from the NYS DTech 2000 and 2004 aerial photography. Internal features of the marsh tended to be the most stable through time and were used as control points when available. For example, locations where two ditches crossed close to perpendicular were frequently used. The registration process for all historical photographs was simplified because salt marshes were flat and do not contain relief distortion, eliminating the need for including topographical corrections. The degree of warp for removing perspective distortion and projecting was adjusted by choosing the transform algorithm used, from a non-warping affine transformation to a potentially heavily altered fourth order polynomial. After achieving a good fit with the control points, all photographs were then viewed on top of the reference image using ArcMap (ESRI Inc.) and checked for discrepancies in other parts of the image. This was accomplished by either setting the new image to be semi-transparent or by turning it on and off in rapid succession and measuring discrepancies. If discrepancies were found, then adjustments were made as needed to compensate, including adding control points, adjusting control points, increasing the order of the polynomial fit, or decreasing the order of the polynomial fit as needed for the best overall result. This process was repeated as required to achieve an acceptable result.

The 1956, 1966, 1973, 1978 and 1989 images were available as prints of contiguous sections of marsh captured during the overflight. Each of these photographic sets covered the entire extent of the study site. Images were scanned at a higher resolution than planned for the final product. Because points can only be selected to within ½ pixel, smaller pixels provided additional detail that enhanced the accuracy of the referencing. The success of the transform was then judged in ArcMap® and, if not suitable, reference points were adjusted and the effort was repeated. After referencing, the images were then saved at a resolution with pixels of 0.305 m per side. The 1956, 1966, 1973, 1979, 1983, 1989 were referenced to approximately 2 m or better as measured in ArcMap®, although tighter tolerances of 1 m or less were possible with some individual photographs.

The available photography for 1926 and 1950 presented two challenges. First, they were composite images composed of sections that were not always oriented in identical ways and they contained independent internal distortions. Composites from 1926 and 1950 were developed from physical prints and then rephotographed. While composites produced a pleasing visual result, and were the best available given the technology at the time, they could not be directly used for the measurements in this study. Unfortunately, the inherent perspective distortions of the component photographs make it impossible to simply scan and reference composite prints and with spatial accuracy. Past attempts to reference the 1926 composite Aerial Atlas images for this study site have resulted in inconsistencies of up to 100 m or more (Fred Mushacke, New York State Department of Environmental Conservation (ret.), *personal communication*).

This problem was addressed by scanning and digitally subdividing the 1926 and 1950 composite photography images into parts that contained particular original images and then referencing these separately, using only points associated with a single original image. By processing single images, in a way similar to other aerial photographic data sets both the internal distortions and the variation in orientation were addressed. The graininess of the 1926 and 1950 photographs also impaired the registration of these photographs. Using the subdivided images still allowed referencing to within 2 to 3 m using Blue Marble Geographic Geographic Transformer®. When temporary detailed corrections were added during measurement using the Spatial Analyst® extension for ESRI ArcMap®, final measurements produced were accurate within 3 m or less.

Some images contained sections of marsh that no longer exist or all possible control points were lost due to subsequent filling. More recent photographs were referenced first, and photographs of intermediate age were sometimes used when large sections of marsh had been filled and possible control points were lost. Although the use of intermediate points introduced additional error, these images were useful as historical records of original marsh extent, the creation of borrow pits, hardened edges, and some incidents of nearby channelization. Because of the marsh was lost, and later years could not be measured, points on these edges were excluded from this study.

Measuring edge change using GIS: the response variable

The response parameter for this study was the edge change through time, measured at points in order to capture local differences in the size of the marsh through time. The edge of the marsh in 1994 was used as the baseline for measuring changes in the marsh through time.

Creating the Reference Baseline

The reference baseline for the measurement of change was created using the methodology of Herder (2003). Creating this baseline required the conversion of spatial data, referred to as layers of data containing individual features of the same class, from raster data into vector data. Raster data are information about some location that is stored in rectangles, called pixels, that represent that same location, its dimensions, and its position in physical space. In the example of aerial photographs, the data represent either the color as mean spectral hues, reflectance as a gray scale, or some threshold as a binary value. Vector data are built from point positions, and can represent information about points, about lines connecting two points, about polylines of many connected points, or information about an area enclosed by a line built from connected points (i.e., a polygon). The process of extracting information from raster images is not always straightforward.

The 1994 set of infrared false color photographs were photographic tiles in the TIFF format, which were processed in photographic manipulation software. The hues representing water were selected and changed to white. Hues representing upland edges were changed to white. The

Graphic Image Manipulation Program (GIMP, The GIMP Development Team, www.gimp.org) was then used to convert these images into high contrast binary black and white images using a built-in hue detection tool. Only after detailed checking and correction of edges, cross-checking with other photography, and some visits to questionable locations to ground truth the data, the marsh areas were turned to black. The edge detection algorithm in the GRASS GIS package was then use to generate line information representing the edge of the marsh as represented in each individual tile (GRASS Development Team 2008, Neteler and Mitasova 2008). The line features that were generated for each tile, were exported as polylines that were saved in shapefile format (ESRI Inc). This resulted in a separate file for each tile that also included tile edges, stray marks from the photographs, non-marsh locations that were colored similarly to marsh, and some edge detection errors that were then corrected by hand to only represent marsh edge. The lines from each of the many photographic tiles were then merged into a single polyline data layer that was also saved in shapefile format. The final product contained only the outer edges of the salt marshes that provided the reference baseline used in this study.

Generation of the edge change measurement points

The points at which the measurements were made were chosen from an even random distribution along the entire length of the 1994 reference baseline with an automated python script written specifically for this purpose. A set of 500 points, which I called edge change measurement points in order to distinguish them from other possible point locations, were generated along the 1994 reference baseline. This set of 500 points represented fixed reference positions. The relative locations of marsh edges for all years, including 1994 as a final check and correction of the baseline, were measured relative to these reference points.

Generation of the trend measurement line

Marsh edges are typically uneven, and calculating the distance from a nearby point to an edge can change with the angular direction to the edge. The rates of marsh erosion were also found to differ even between nearby edges. Sections of peat sometimes detach from the marsh edge as a large block, or an edge can slowly erode back to a small pool or other weak area that collapses quickly. A consistent interpretation of distance to an edge change reference point was therefore difficult. The solution to this problem was to create a single trend line through each edge change reference point so that there was a single cross-section through the changes of the marsh edge through time. The distances between successive marsh edge positions along this line represented the minimum measurement of change through time. Where the trend line crossed the marsh edge in different years determined a consistent sequence of measured points along that line, somewhat analogous to measuring points along a ruler, which were used to determine the change in the location of the marsh edge through.

The script that generated the edge change measurement points simultaneously generated a polyline GIS layer containing the trend line estimates. Consistent ID numbers were assigned to edge change measurement points and all other features associated with each point so that they could be associated during future processing. An initial estimate of a trend measurement line's orientation was generated as a perpendicular to the edge. Salt marsh edges, however, are very uneven, so estimating a perpendicular to the edge did not always work well. When combined with the fact that erosion does not always occur evenly along an edge, these trend lines were only first approximations. If, when taking measurements, it was found that a line did not follow the actual erosional trend, the ArcMap rotation tool was used to rotate the trend line around the associated edge change measurement point. When a trend line was reoriented, any measurements already taken were cleared and remeasured so that a consistent set of revised points would be used in each analysis. All data were stored in the attribute table of this trend layer.

Collecting edge location data from photographic sets at the edge measurement points

In this study, the fundamental measurement was the difference between a particular edge change measurement point, derived from the 1994 data, and the point where the associated trend measurement line crossed the edge of a particular year's photographic set (Fig. A.12). That measurement was then recorded in the database as gain (positive), or loss (negative) for that interval. Estimates of edge change between other time periods were derived from these values by either adding or subtracting the fundamental intervals relative to the edge change measurement points.

Measurements of the change in the location of the marsh edge from the reference location were made using ESRI ArcMap® versions 9.0, 9.1 and 9.3 GIS software. A Visual Basic for ArcMap® script was written to automate the process.

There were several cases where, during the time between the photographic data sets, the marsh eroded through to open water so that there was no longer any marsh after 1994. Similar problems occurred for the time period before 1994, when some marsh islands had not yet formed on sand bars or dredge spoil, or the waterway was not yet dredged through an island, so the edge between the water and salt marsh did not exist yet and could not be measured. In some cases, an edge became fragmented after 1994 or had been fragmented in earlier years and was difficult to interpret. In other cases, an image of a specific location was damaged, missing or otherwise unusable in a particular year's photographic set, and was therefore not measurable for that particular year. For all of these cases, a null value was recorded in the database.

Small temporary adjustments for photographs earlier than 1994 were made using the referencing capability within ArcMap® by choosing control points close to the random shoreline points being measured. If control points were found that also occurred in either the 2000 or 2004 orthophotos, then the ArcMaps® georeferencing tool was used for this fine-tuning. Measurements most of these edge changes were accurate to within< 0.5 to 1.0 m. The 1950 and 1926 sets were less clear and an accuracy of < 1-3 m was achieved.

Trend illustrations from year class measurements

Although only the measurements from 1926, 1966 and 2007 were used in statistical analyses of factors affecting marsh edge loss, the other 9 data sets were used to assess how marsh edge loss changed over time using Loess spline smoothers and GAM models. These models were also used to look for trends in the rate and direction of change for areas of marsh that had been channelized as well as natural marsh edges to determine whether marsh loss (or gain) was linear or accelerating or decelerating over the 81-year period.

THE RESPONSE VARIABLE: TIME INTERVAL ESTIMATES

The 1966 aerial photography set was chosen as the reference for marsh gain or loss for several reasons. First, 1966 was not long after the conversion of the local watersheds from agricultural use to the highly urban community that was in place by the 1970s. Second, 1966 was not long after the sewer treatment plants started discharging into the bay during the late 1950's. Third, 1966 was not long before water sampling records were available in the early 1970's (Table 3.1).

Table 3.1, Time line of significant watershed changes to Hempstead Bay
1600
The study area was included in the Kieft Patent (1644) that founded the Town of Hempstead, and the Dongan Patent (1685) that was issued by King Charles of England soon after the land was taken from the Dutch East India Company. These deeds initiated the developement of the land and public use of the marshland.
1700
The land was extensivly developed as farmland and fishing communities were started. Grist mills and mill ponds were constructed on most stream systems. The Village of Freeport and the village of Hempstead, located further inland, both date to colonial times.
1800
By 1870's the early records of the US Fisheries Service show active fisheries within the study area based in several local communities. Drinking water reservoirs were constructed for the City of Brooklyn water supply in Bellmore Creek, East Meadow Brook, and the Mill River watersheads.

Table 3.1, Time line of significant watershed changes to Hempstead Bay
1880
An acurate map of the bay was surveyed for US Coast and Geodetic Survey that shows additional natural inlets and some ditching. Town of Hempstead Records show expecitures for small dredging and ditching projects paid for from permits to hay marshland. Parts of Bay Park, Island Park, were already developed, and Freeport was expanded.
1900
No significant specific records discovered .
1920
1926 aerial photography shows that the "Five Towns" communities, separating Hempstead Bay from Jamaica Bay to the west, were already developed as suburban areas. Housing tracts were already developed to current densities in Inwood, Lawrence, Cedarhurst, Woodmere, and parts of Hewlett, Hewlett Harbor, and Woodsburg. Sections of marshland were already filled and construction started that extended Freeport further into the bay. Island Park and Harbor Isle are already filled and built. The City of Long Beach was already established, including the filling ofsome tidal creeks and sections of marsh. Renolds Channel was already dredged for fill and navigation, including the channelization of some marsh locations. White Mills Island, not shown on prievious maps, appears just north of Renolds Channel near South Black Banks Island. Luces and Hog Island inlets, were also closed and built over in Long Beach. Zachs inlet separating Short Beach and Jones Beash is already closed. Active dredging and filling is seen in several locations including Hewlet, Freport, Baldwin, Seaford, and Point Lookout. Many streets already existed that divided the land into blocks immediately north of marshlands, however there were few houses and some blocks still showed rows of crops or nursery trees. Street drainage system installation started, but septic sewage went to cesspools.
1930
Jones Beach State Park was constructed and the Meadow Brook and Wantaugh Parkway causeways were constructed across the bay. The parkway causeway construction included channelization of Swift Creek Channel, Sea Dog Creek and some other locations that were dredged directly through marshes for use as fill in parkway construction. The New York State Boat Channel was also constructed, often through marshland.

Table 3.1, Time line of significant watershed changes to Hempstead Bay
In 1938, the United States Department of Agriculture flew aerial photograph of farmland that showed farms ocupying the north eastern portion of the Town of Hempstead, inland from the eastern portion of the salt marsh study site. House construction was seen to be expanding, but most of the land was not included in this potographic survey.
1940
The 1950 aerial photography set shows that, prior to 1950 additonal borow pits were dredged and additional filling of marshes occured in Baldwin, Freeport, Merrick, East Rockaway, Freeport, and Oceanside. The Bay Park Sewer Treatment Plant was constructed, but the main outfall and surrounding public park were not constructed.
1950
By the 1956 aerial photography set, the outfall for the Bay Park plant was installed and the surrounding park was constructed by filling marsh from a borrow pit in Hewlett Bay. Dredged material had covered the marsh islands of Pearsals Hassock, North Black Banks, South Black Banks, West Meadow, and on a small section of East Meadow. Hog Island Channel, Sturm Channel and the southern part of East Rockaway Channel were deepened and widened in order to accomidate coastal tanker traffic. Many new buildings were constructed inland from the estuary. In the Pines Brook watershed, Pines Pond was drained for use in school construction and Pines Brook channelized for drainage.
1960
By the early 1960s, most open land was already used for construction. By 1966, addional dredging, filling and building occured in Oceanside, Bellmore, Wantagh, and Merrick. In 1969, several million of cubic yards of sand were mined out of Jones Inlet and used to fill marshland for the construction of Cedar Creek Park and Cedar Creek Sewer Treatment Plant in Seaford and an outfall was constructed through the bay to the ocean. Wetland Protection efforts started in the late 1960s. Sometime in the mid 1960s, East Meadow Brook was channnelized in order to reduce flooding on Meadow Brook Parkway and speed drainage from the accociated red maple wetlands.
1970
State and Federal wetland protection regulations came into effect. The Town of Hempstead started monthly water quality testing and installed 6 tide gauges in the estuary. Data collection continues as of the time of this research.

Table 3.1, Time line of significant watershed changes to Hempstead Bay
1980
Minor dredging occured only to keep inlets open and for the removal of shoaling that endangered navigation.
1990
Minor dredging occured only to keep inlets open and for the removal of shoaling that endangered navigation.
2000
Minor dredging occured only to keep inlets open and for the removal of shoaling that endangered navigation.

For most of the analyses in this study, the period 1966 to 2007 was used. This period represents a time of relatively consistent conditions that may have been different prior to 1966. Of the original 500 points that were measured, 438 contained good measurements for both 1966 and 2007, and the change between these years was calculated.

The change in the marsh between 1926 and 1966 was also used for the estimation of change prior to the major urban development around the study estuary. This change was then compared, point-by-point, with the 1966 to 2007 changes. When measurements were not possible in all of three years, the point was excluded from the analysis set. The 12 photographic sets did not produce enough temporal points to define a time series at each spatial point, so full time series analysis was not possible. However, 12 points can indicate a trend and can also be used on shorter temporal sets to test for changes in response to change in influence for the same point. A total of 421 points contained valid measurements of all three years, 1926, 1966, and 2007.

Characteristics of response variable: Transformation method

Across the entire marsh form 1966-2007 there was an average loss of 9.7 m (SD 16.5) of marsh from the edge. The median loss over this time was 5.4 m. There were many locations where there was little change in the marsh edge, some places where marshland gained, and a few with large losses (Fig. A.10). The parameter representing marsh change between 1966 and 2007 was tested for normality with the Shapiro-Wilk test (Sahapiro and Wilk 1965) and normality was rejected ($P< 0.001$). The data were also significantly skewed (D'Agostino test for skewness, $P < 0.001$), and also significantly leptokurtotic (Pearson's measure of kurtosis = 15.6966, $P < 0.001$). Thus, the response variable was highly negatively skewed and leptokurtotic and had both negative and positive values representing eroding and accreting edges (Fig. 3.1). Because the response variable had two long tails and had both positive and negative values, typical transformations of the data were unsuccessful. A Yeo-Johnson (YJ) transform is a variation of the Box-Cox that is designed to work with variables that have both positive and negative values, however, this transformation did not normalize the data. The Quantile-Quantile-normal plots of YJ transforms showed an unusual pattern of a sharp bend in an otherwise straight line (Fig. 3.2).

A variation of the Yeo-Johnson algorithm that uses a log transform was also tried and came closer to achieving a normal distribution, except for the bend exhibited by the Quantile-Quantile-normal plot in Fig. 3.3, which was likely due to differences between accretion and erosion processes. The locations where the marsh expanded between 1966 and 2007 ($n = 35$) expanded at rates roughly half that of the areas where marsh was lost (n = 216). If the process of gain involves different mechanisms than the process of loss, an additional adjustment may be justified (F. J. Rohlf, Stony Brook University, *personal communication*). The adjustment used here was to multiply the negative values by a constant and the positive values by the inverse of that constant after transformation, which produced and a near perfect Normal QQ plot (Fig. 3.4) and a near-normal distribution (Fig. 3.5).

For this transformation, the following procedure was used:

1. For positive values (marsh growth), 1 was added to each value (to preserve the relative order) and then the values were log transformed.

2. For negative values (marsh loss), all values were made positive, 1 was added to each value, values were then log transformed, and finally made negative again.

This transform created two separate lines for positive and negative values. Therefore, I further transformed the data by multiplying each negative value by a constant, typically 0.5, and dividing the positive values by the same constant.

Based on additional tests during the hypothesis testing, some additional points were removed when they represented outliers for an independent variable, exhibited disproportionate influence, or introduced non-linearity in the fitting. Once the response variable was transformed, most independent variables did not need transformation.

Figure 3.1. The response variable, change in marsh edges between 1966 and 2007.

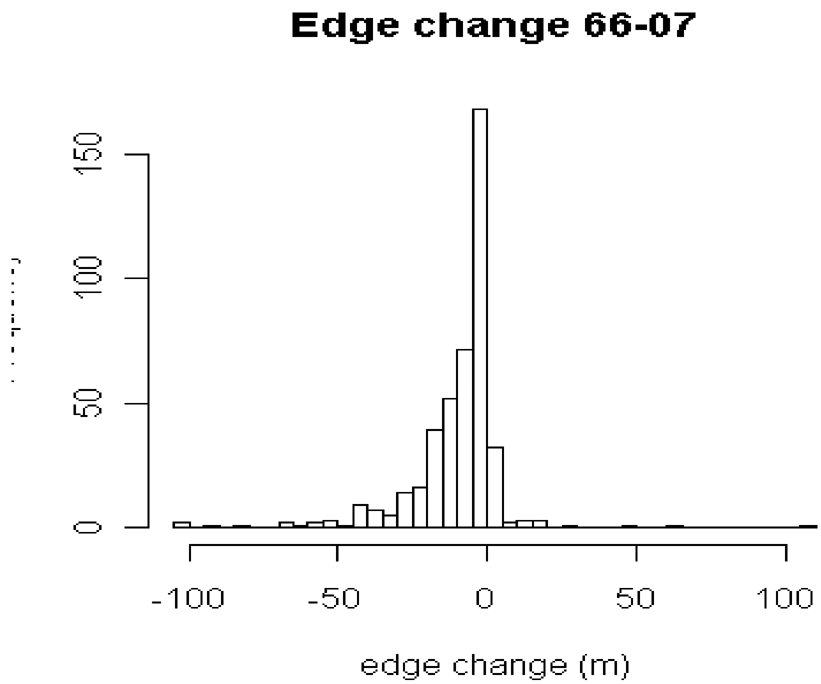


Figure 3.2. When the Yeo-Johnson transform was attempted on the linear regression of Minimum Fetch on Marsh Edge Loss for 1966-2007, the results of the transformation were poor.

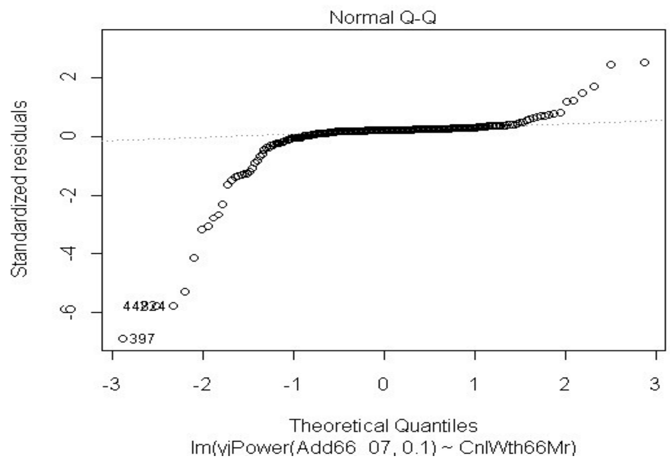


Figure 3.3. When a variation of the log transform that offsets values from 0 and inverts the negative values before and after transformation was used, a reasonable transform was achieved, but with a distinct bend in the Normal QQ plot of the residuals.

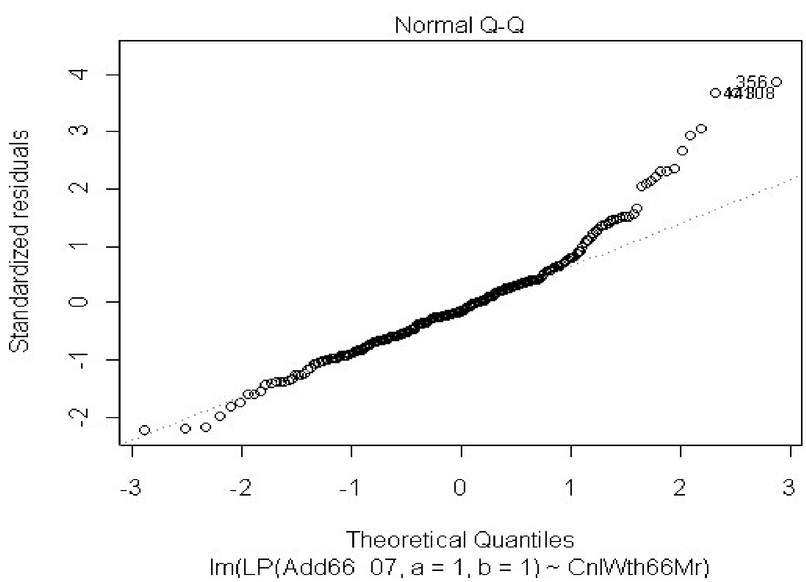


Figure 3.4. Final result with the transformation described here.

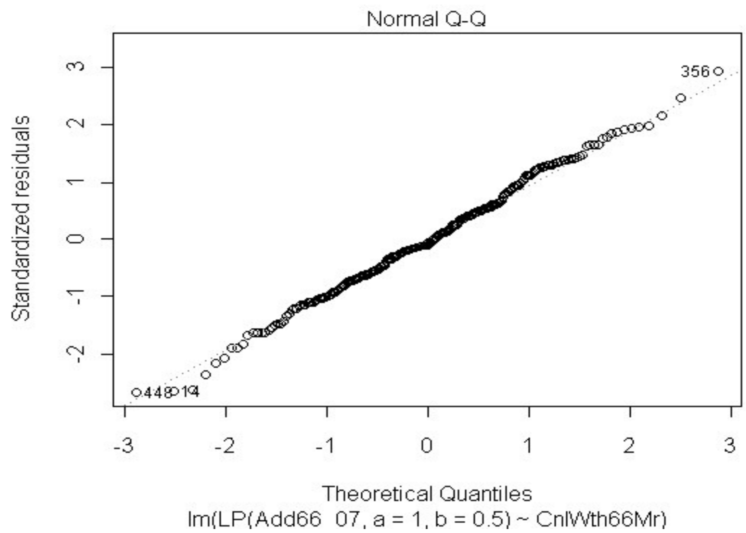
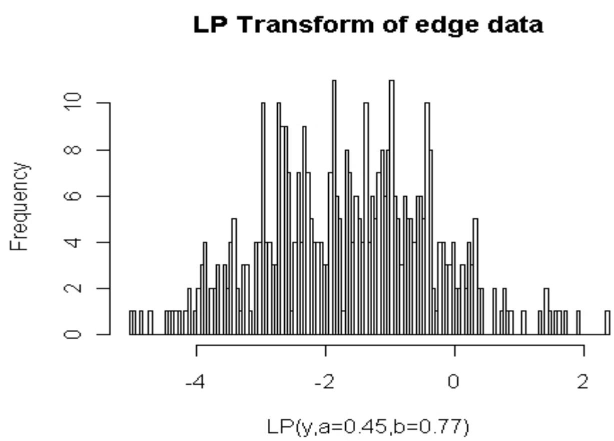


Figure 3.5. The histogram of the response variable after the LP transform.



INDEPENDENT VARIABLES

Independent variables were measured from several sources, including NOAA nautical charts, NOAA NOAS data, the photographic data sets, or acquired from water quality sampling and computer simulations.

Channelized Edges

Identification of channelized edges was primarily through historical data, derived by comparing pre and post construction spatial data (Herter et al. 2003). Channelized edges are of direct anthropogenic origin, intentionally created as part of some major construction project prior to legal protection for these wetlands. These edges were usually straight and deeply cut, marking the sharp boundary where the damage to the marsh had ended at the time. For example, the marshlands were surveyed by the US Coast and Geodetic Survey in 1879 and 1880 (Fig. A.3; Fig. A.5), providing a comparison with the photography from 1926 and later (Fig. A.2; Fig. A.4). The 1879 survey maps were made available by the Stony Brook University Library, and aided the identification of channelized edges (Fig. A.3).

Channelized marsh edges differ from natural channels in that they are sharp and expose the softer sediment underneath the marsh peat. At channelized edges, the water was deeper than natural channel edges, which could induce slumping, reduced wave attenuation, or introduce other effects that may have increased the probability of subsequent edge loss due to erosion. Therefore, channelized edges were treated as a separate class from natural marsh edges (Fig. A.13), and tested separately to determine which factors were correlated with marsh losses or gains.

Navigational Channels

For the purpose of this study, navigational channels were defined as those bodies of water that were marked and charted (NOAA chart 12352) for navigational use at some time during the period between 1968 and 2007 or had three or more bay houses that required access by boat in 2007.

There are no data sets available that directly measure the production of boat wakes within this study site, therefore two other methods were used to estimate the influence of navigational channels. A polyline shapefile was created for the marked navigational channels from NOAA chart 12352. Based on over thirty years of personal experience, I categorized them according to the typical size of boats and frequency of use (Table 3.2, Fig. A.14). The officially charted channels, mapped in ArcMap GIS, were organized into five categories of use and a sixth category representing no use, based on the size and frequency of the passing boats (Table 3.2). The size and frequently of boats and presumably their wakes, were expected to affect erosion rates.

For marsh change measurement points that faced navigation channels, the direct distance from the point to the center of the boat channel that it bordered (*channel distance*) was measured using the measurement tool in ArcMap. Due to wave attenuation, the effect of boat wakes was expected to fall off with distance and be proportional to the marsh edge change. It was hypothesized that distance from the marsh edge to the center of the channel would show more of an effect for heavily used than lightly used channels.

A third variable related to navigational channels was the euclidean distance to the nearest mapped channel (the *unlimited distance*, i.e. not limited by intervening land), which was measured for each sample point using the Point-to-Line Distance measurement tool supplied in Hawth's tools extension®for ESRI ArcMap®. These unlimited distance measurements ignored the existence of islands and other obstacles. As such, they may be less associated with direct erosional effects, but may be associated with sediment transport potential. Most channels are naturally deep, required little or no dredging, and their time of origin often predates their use for mechanically powered navigation (Fig. A.5).

Table 3.2. Navigational Channels. This variable was a categorical variable, whose rank was subjectively determined, and included waterways marked or formerly marked with navigational aids and charted on NOAA navigational chart 12352.

Category	Criterion
A	Many large boats, including commercial traffic up to 100 m at water line (LWL), use the Inland Waterway and Long Creek channels to Jones and East Rockaway inlets. Commercial traffic included coastal tankers, coastal fishing boats, party boats, ocean-going charter boats, casino boats used them. Maintenance dredging of these channels was performed as needed by federal or state agencies. Peak use could exceed 60 boats per hour.
B	Many boats, some large commercial traffic as defined for A. These were secondary channels that feed from significant concentrations of marinas and wharfs into the A channels. Occasional Town maintenance dredging of these channels was performed if needed to allow navigation. Peak use could approach 60 boats and hour.
C	These had frequent small and medium boat traffic. Peak use was about 20 boats per hour with boats up to about 8 m AWL. These were smaller tertiary channels that mostly fed into B channels from a smaller subsets of marinas.
D	Local use channels, which were mostly interconnecting side channels connecting between B and C channels, or they only fed residential docks. Small boats to about 8 m LWL used them, with peak use to about 10 per hour. Most were marked as 5 mile per hour (8 km per hour) zones.
E	Low local use channels were small side channels feeding traffic from only a small number of houses or bay leases into more heavily used channels. Navigational aids and maintenance dredging may have been discontinued due to low traffic level. Typically there were only up to about 2-3 boats per hour.
X	Not official navigational water. These waters include all of the remaining marsh edges. Widths and depths vary from broad shallow bays to narrow drainage channels of up to 3 m depth below MLW. Boat use was rare to non existent, usually less than 1-2 per day, confined to local fishermen in small shallow draft boats less than 6 m LWL.

Hardened Edges

Nautical charts and GIS maps show the locations of dredged channels and hardened shorelines such as bulkheads and riprap (Fig. A.14; Fig. A.15). Hardened shore lines, such as bulk-heads and riprap were mapped, and marsh proximity to these features measured using the direct distance measurement tool in Hawth's tools ®for ESRI ArcMap®. The variable was the Euclidean distance to the closest hardened edges without regard to obstacles that would block wave propagation. This parameter was indented for use with two hypotheses that were not included, wave reflection to opposite shorelines and reflection downward to adjoining marsh. Modeling of direct line-of-sight wave reflection was not used. Reflection downward could not be tested because no original marsh existed next to bulkheads, but only recently formed marsh on accumulating sediment. Many hardened edges were installed as part of the same construction that formed the borrow pits, and the hardened edges were therefore adjacent to borrow pits and major filling. Therefore, this variable wasnot included in analyses.

Distance to Treated Sewerage Outfalls

A point feature was created on ArcMap GIS that represented the location of treated waste outfalls (Fig. A.16). Hawth's tools® for ESRI ArcMap® utility for direct distance measurement was then used to automatically load the distances between marsh edge measurement points and outfall location into the point attribute table. The distances were not weighted for effluent load, buy water quality data (see below) did include concentrations of nutrients from both sewer and stormwater outfalls.

Distance to borrow pits

1949-1959 NOAS bathymetry from NOAA was used as the main bathymetry data set for the study site. NOAA Nautical chart 12352 and, when available, recent bathymetry data were then used to edit and update the 1949-1950 data to sufficiently locate borrow pits and estimate their size. Distances to borrow pits were calculated by outlining the 4 m depth contour into a polygon feature in ArcMap® and then calculating the centroid for each feature (Fig. A.17). The distance between the marsh edge measurement points and the nearest borrow pit centroid was determined using the setting for picking the single nearest point in Hawth's tools® for ESRI ArcMap® Distances Between Points (Between Layers) tool.

Water quality data

Water quality data included numerous parameters measured monthly at several stations from 1975 to present. Water quality parameters collected monthly by the Department of Conservation and Waterways of the Town of Hempstead, included nitrates (μmol N l^{-1}), nitrites (μmol N l^{-1}), ammonia (μmol N l^{-1}), phosphates (μmol P l^{-1}), chlorophyll (mg m^{-3}), turbidity (NPU), and Secchi depth. The Town of Hempstead Department of Conservation and Waterways (TOH C&W) has collected data from 28 water quality stations for up to 36 years. Some additional TOH C&W data were available from 1968, but at different sampling stations. The locations of these stations were mapped in GIS (Fig. A.18), giving them spatial coordinates for processing with spatial software and they were interpolated using GIS software and estimated for use with the edge measurement points.

Chlorophyll a (mg m^{-3}) was measured in the laboratory from monthly bulk water samples that were collected at a depth of 1 m (Strickland and Parsons 1984). This parameter is a widely used indicator of total phytoplankton biomass (Desortová 1981, Clesceri et al. 1998) and productivity (Boyer et al. 2009). The Chlorophyll a values represent a reasonable indicator of phytoplankton occurrence and the likely spatial distribution of phytoplankton derived sediment.

Reactive nitrates (μmol NO_3 l^{-1}), reactive nitrites (μmol NH_4 l^{-1}) and ammonia (μmol NH_4 l^{-1}) were also determined from bulk water samples (Clesceri et al. 1998). Salinity (ppt) was always measured in the field at 1 m depth. Turbidity (nephelometric turbidity units, NTU) was also available for a different set of 28 sampling stations from 2008 to present.

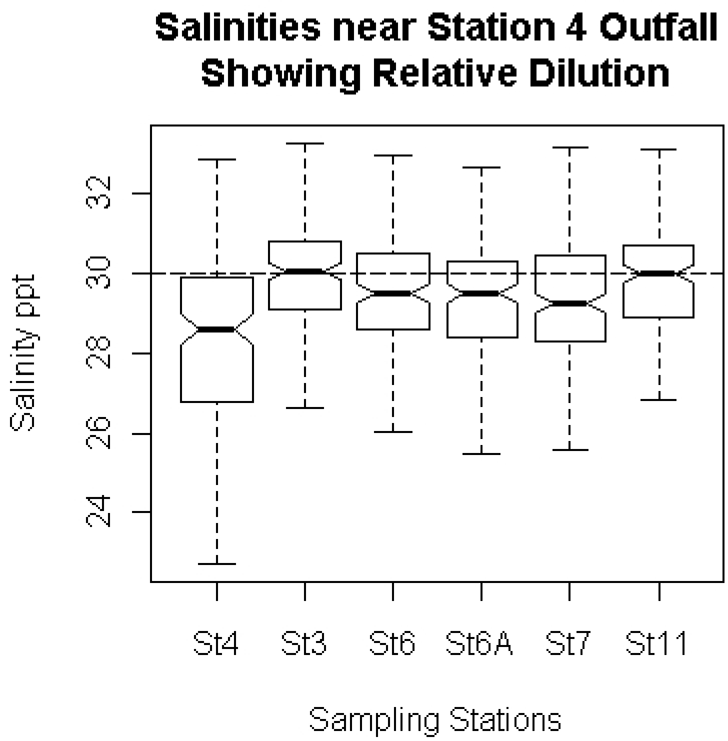
The overall mean of the monthly measurements for each parameter through time were calculated for each station. Bubble plots of 34-year means for the growing season (April-September) for nitrites illustrate the means per station (Fig. A18).

Two dimensional spline interpolations using the Spatial Analyst® extension for ESRI ArcMap® were used to estimate values of water quality for all points between sampling stations (e.g., Fig. A.19). Simple spline interpolations were seen as sufficient because particle movement in the estuary is neither a simple directional flow nor simple diffusion, but is mixed by the alternating direction of tidal flow. The values from the interpolated raster water quality layers were extracted into the attribute tables of the marsh edge measurement point feature using the Hawths Tools® (Beyer 2004) extract points tool.

Although the water quality sampling stations are distributed reasonably evenly throughout the study area, none are located at the town line, so the interpolations were done within a region bounded by the sampling stations (Fig. A.1). Some of the reference baseline and the associated edge change measurement points extended past the area enclosed within the water sampling region (Fig. A.1). Points where marsh edge was tracked that were outside of the water sampling region were given null values for water quality parameters.

At station 4, which was located at a major treated sewerage outfall just south of greater Black Banks Hassock, all parameters were significantly different than those of nearby stations (e.g. ammonia values were several times higher at station 4, but chlorophyll values were much lower). The mean salinity at this one station was 4.9% lower than the means at the five nearby stations, and 36.7% lower on one sampling day (Fig. 3.6). When interpolating, the extreme values for station 4 caused distortions that did not reflect the mixing of the water in the estuary. Spline interpolations, that were constrained to include station 4, produced negative values near other stations. Other interpolation methods that were attempted produced extremely high estimated concentrations of nutrients for the stations near station 4. Therefore, station 4 was not used, nor were two stations with short histories, leaving a total of 27 usable stations.

Figure 3.6. A boxplot of salinity comparing station 4 with nearby stations, including 6, 6A and 7, which are further inland and have several sources of water input that dilute the effluent from this station. Median values are represented as bold lines, the boxes represent the upper and lower quartiles, whiskers represent extreme values, and a Tukey notch test where the groups are significantly different if the notches do not overlap.



Tidal Flow Rates and Truncated Tidal Range

A hydrological and current modeling component for the Great South Bay Project (GSBM) was designed by R. Wilson and C. Flagg, Stony Brook University, to model the Great South Bay. This model only worked well when the entire SSER, including Hempstead Bays, was added to the system. I used output from this model to estimate peak tidal current flows and truncated tidal ranges for sample locations. Peak flow rate was the maximum modeled rate of flow through a full tide cycle (ebb and flood). Flow rate was used as an indicator of the potential for sediment transport, both erosion due to rapid flow and accretion in quiet locations.

The GSBM reports the maximum tidal excursion for high tide, and either the minimal tidal excursion for low tide or the height of the sediment, whichever was higher in elevation. This range is called a truncated tidal range, and is indication of potential wave attenuation in shallow water. The variation in tidal amplitude within the bay was modeled by the Great South Bay Project models and was observed in the tide gauge records. For locations where surrounding mud flats and sand bars are exposed at low tide, it was likely that both wind and storm driven waves and waves from boat wakes were attenuated prior to impacting the marshlands.

The output from the GSBM contained 2 spatial layers and was designed to be imported into MatLab® (MathWorks®), where the results were retrieved using scripts developed by C. Flagg. The MatLab® scripts were adapted into Octave (Eaton 2002) scripts that extracted the water velocity and water height estimates from the model output file. These estimates were then used to calculated the parameters used in this study, and the output saved to a ArcMap® compatible format. As with water quality data, the point data from this model were interpolated into a raster set for estimation of peak flow and a set for truncated tidal range as values in pixels for the waterways in the study area. The nearest neighbor interpolation method from the Spatial Analyst Extension® for ArcMap® was used for these interpolations. The information tool in ArcMap was used to retrieve near edge flow and channel center flow raster values associated with each particular edge measurement point and add them to the table associated with that point. The same was done with the truncated tidal raster to retrieve the nearby truncated tidal range value for each edge measurement point and enter them into the tidal range field of the measurement point table.

Storm Driven Significant Wave Height

Prevailing winds primarily impact the marsh through the waves that are generated. This effect is influenced by the local shoreline and near-shore topography. A simulation model that incorporates topography was used to estimate the waves that result from winds recorded at a local weather station.

The SWAN Cycle III version 40.51 (Booij et al. 1999, Padilla-Hernandez et al. 2007) was used for the estimation of significant wave height for a given wind strength, duration, and direction. The SWAN model (SWAN) computes refraction, shoaling, energy dissipation, wave-wave interaction, and hardened edge reflection for wind driven waves. By using SWAN, it was possible to estimate wave patterns for the entire complex network of water bodies that comprise the Hempstead Bays within the SSER. Input data for the model included local weather conditions and the three dimensional shape of the edges of marshland and the bottom of a water body.

The USGS installation at Point Lookout (USGS 01310740 Reynolds Channel at Point Lookout NY) was centrally located within the study site. Weather data from the Point Lookout USGS station includes wind speed and direction, recorded at 15-minute intervals when it was first installed and at 6 minute intervals after 2000. The data were provided by USGS for the 11 years period 10/01/1998 to 09/30/2009. These 11 years of records were analyzed to estimate typical local wind patterns and the dominant wind directions and strengths during storms. Wind roses (Fig. A.20) were constructed using the Wind_Rose code for MatLab® available from MatLab Central (http://www.mathworks.com/matlabcentral/fileexchange/17748-windrose). Wind roses were generated for the top 15%, 10%, 5% and 1% of wind speeds found in the provided data.

SWAN also requires a raster map representing both the surrounding shoreline and bathymetry of the water body. The bathymetry was estimated from the NOAA NOAS 1949-1950 data set edited to include changes seen in the 2008 edition of NOAA nautical chart 123452. All available recent bathymetric data, mostly from the US Army Corp of Engineers, were also included to reflect recent conditions. The resulting point data were converted into a SWAN compatible raster sets using the nearest neighbor interpolation function in the Spatial Analyst Extension® for ESRI ArcMap® and then exported as ASCII raster files. With the proper configuration settings, SWAN read the resulting bathymetry.

Calculations were made using 20 m and finer grid cells. Major impacts from wind driven waves are assumed to occur during storm conditions with above normal tides as represented by these water levels. Tide heights used in the model were at a mild storm surge height of about 0.5 m above high tide. The SWAN model was calculated using the direction and wind speed from the

5% highest wind intervals as shown in the wind rose.

The results from the SWAN model were automatically saved as *.MAT files for MatLab® and as *.TAB table files. ArcMap® could not successfully use data from either *.MAT or *.TAB files, so a small PERL script was developed that converted *.MAT files in to *.DBF files that ArcMap® was able to import correctly. Values were then interpolated in ArcMap® and finally the identify tool was used to transfer raster values into tables associated with the marsh edge measurement points.

TEST FOR COLLINEARITY AMONG INDEPENDENT VARIABLES

Patterns of intercorrelation among the independent variables in multiple regression, inflates the standardized unexplained variance and is measured as the variance inflation factor (VIF) (Sokal and Rohlf 1995). O'Brien (2007) discusses the problems encountered in dealing with colinearity and multicolinearity (terms used interchangeably) when controlling excessive VIF. Johnson and Wichern (2007), in a discussion of multivariate analysis, define collinearity as " If **Z** is not of full rank, some linear combination, such as **Z**a, must equal **0**. In this situation the columns are said to be collinear. This yields large estimated of variances … and it is difficult to detect significant regression coefficients." (p 386, Johnson and Wichern 2007). Crawley (2007), also discussing multivariate analysis, defines multicollinearity as "the near-linear relation between two of the explanatory variables, leading to unstable parameter estimates" (p448, Crawley 2007). Burt et al. (2009) state that " multicollinearity in a multiple regression equation occurs when the independent variables $X_1, X_2,...,X_{p-1}$ are intercorrelated" and point out that it becomes extremely difficult to sort out the effects of independent variables. When the multicolliniarity is perfect, the regression equations cannot even be estimated as the correlation matrix approaches singularity (Ferrar and Glauber 1967, Brurt et al. 2009). Although much of the literature seems to use the two terms interchangeably, Tu et al. (2004) define colinearity as the covariance of independently derived variables and multicolinearity as covariance stemming from having mathematically derived a variable from others. Here I use the term collinearity, in the sense defined by Tu et al. (2004), i.e., all of the variables were measured independently.

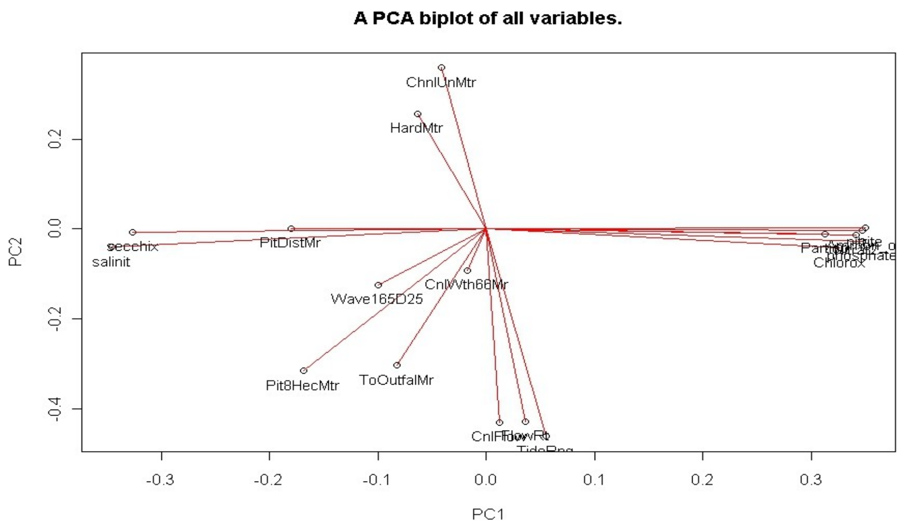
Multivariate methods were used to explore the covariance between potential driving factors in this study. An important first step was to assess the independence of the potential driving factors under study.

The spatial extent of this series of tests was defined by the boundaries that enclose the water quality data (Fig. A.1). Values representing conditions at each of the random marsh edge measurement points were stored in a single GIS attribute table. The resulting attribute table was imported into the R statistical environment for analysis (R Development Core Team, 2009). Points that were outside the extent defined by the water quality data were not used as part of the variable set reduction process. Categorical factors, categorizing channel type, were excluded from this PCA analysis. A subset of 372 points comprised of points where edges existed in both 1966 and 2007, and that were within the water quality data extent (Fig. A.1), were used.

REDUCED INDEPENDENT VARIABLE SET

Multivariate analysis was used to reduce the data set and reduce the collinearity within the data set. The reduced set of variables was primary chosen using hierarchical clustering, while visualized using PCA and biplots. PCA was performed on the set of variables using the prcomp function from the stats package in R. The PCA results were visualized using the biplot function in R. In the biplot (Fig. 3.8) illustrates the high degree of collinearity between the water quality variables, all of the nutrient variables are on the right side of the plot while Secchi depth and salinity are on the left. Fig 3.7 illustrates the first two principal components, and does not fully represent all of the covariance in the hyperspace that is being analyzed. Variables may be separated in dimensions other than those visualized in the biplot, therefore biplots are not sufficient for data reduction by themselves.

Figure 3.7. A PCA biplot of all continuous variables, particulate organics (Partico), chlorophyll (Clorox), nitrates (Nitrat2), nitrite, ammonia (Ammon_0), phosphate, turbidity (Turv2m1DW3), distance to hard edge (HardMr), Euclidean distance to navigational channels (ChnlUnMtr), Truncated Tidal Range (TideRng), edge Local Tidal Flow Rate (FlowRt), Channel Tidal Flow Rate (CnlFlow), Minimum Fetch in 1966 (CnlWth66Mr), Storm significant wave height (Wave165D25), shortest Distance to Treated Sewerage Outfall (ToOutfalMr), Secchi depth (secchix), salinity (salinit), shortest Distance to Borrow Pit (PitDistMr), and shortest Distance to Borrow Pit of 8 ha or larger (Pit8HecMr).



Hierarchical clustering then confirmed the findings from the PCA, and was used to impartially distinguish the remaining variables by degree of collinearity. Because the number of variables was small, no a priori criteria for choosing a particular clustering algorithm was obvious. In initial attempts, the dendrograms formed by simple clustering techniques were not stable during pruning. Because the intent of this process was to develop a minimally correlated set, the cluster was pruned by removing similar nodes and retaining the most dissimilar, least correlated nodes. This seemed to add to the lack of stability during pruning. Simple clustering using one algorithm was not deemed adequate. The solution was to bootstrap the calculations for each algorithm and then take a consensus tree, a process that produced stable and predictable sub trees.

Clustering was done using R package Pvclust (Suzuki and Shimodaira 2006; R Development Core Team 2005). The resulting clusters retained stable relative positions for the variables when pruned.

The initial clustering included all of the continuous variables other than direct distance to channel, as the inclusion of this variable would have restricted the analysis to only channels (Fig. 3.8). I then removed individual variables to determine the largest set of variables with minimal collinearity. The covariance matrix was also inspected to estimate the least amount of covariance among the remaining variables. The water quality variables were highly collinear and obscured interactions between other variables; therefore, the process was repeated with the water quality variables removed. The subsequent cluster, with water quality variables removed, highlighted the collinearities among the remaining variables (Fig. 3.9), which is illustrated with a biplot (Fig. 3.10). A reduced set of variables was identified so that no two members of the same collinear group were used for the same test during the analyses of edge change (Table 3.4).

Figure 3.8. A clustering of all of the continuous variables including; particulate organics (Partico), chlorophyll (Clorox), nitrates (Nitrat2), nitrite, ammonia (Ammon_0), phosphate, turbidity (Turv2m1DW3), distance to hard edge (HardMr), Euclidean distance to navigational channels (ChnlUnMtr), Truncated Tidal Range (TideRng), edge Local Flow Rate (FlowRt), Peak Channel Flow Rate (CnlFlow), Minimum Fetch in 1966 (CnlWth66Mr), Storm significant wave height (Wave165D25), shortest Distance to Treated Sewerage Outfall (ToOutfalMr), Secchi depth (secchix), salinity (salinit), shortest Distance to Borrow Pit (PitDistMr), and shortest Distance to Borrow Pit of 8 ha or larger (Pit8HecMr).

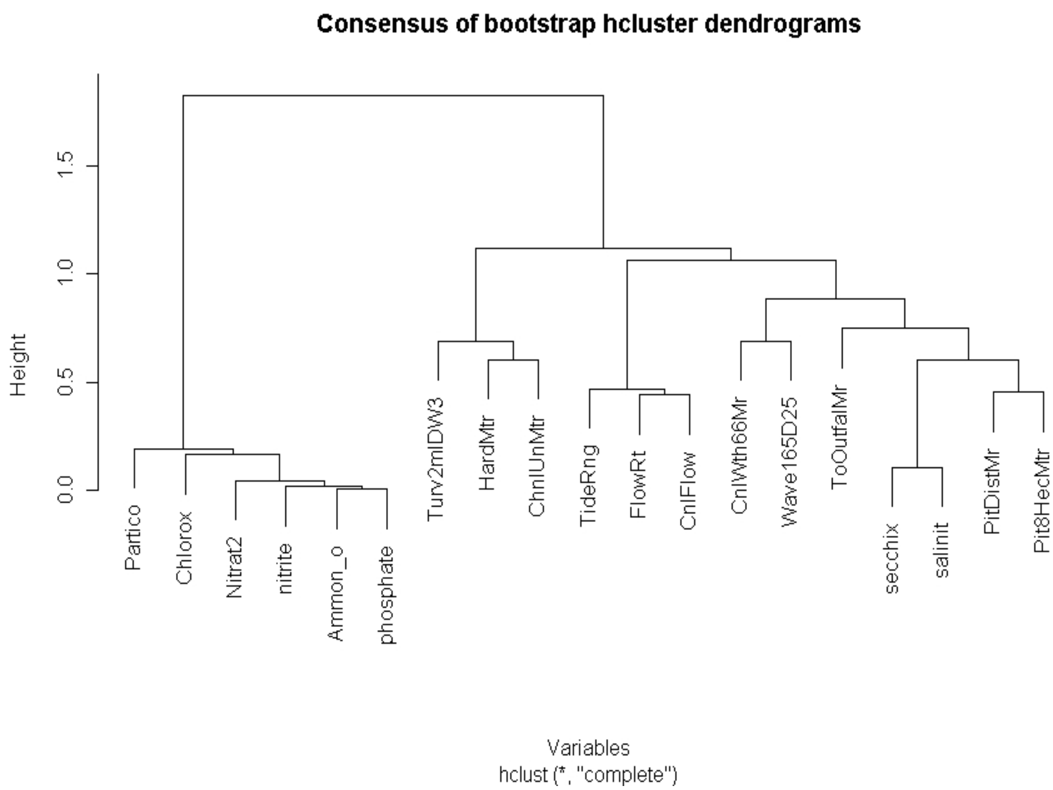


Figure 3.9. The consensus cluster of non-nutrient variables including; Turbidity (Turv2m1DW3), distance to hard edge (HardMr), Euclidean distance to navigational channels (ChnlUnMtr), Truncated Tidal Range (TideRng), edge Local Flow Rate (FlowRt), Peak Channel Flow Rate (CnlFlow), Minimum Fetch in 1966 (CnlWth66Mr), Storm significant wave height (Wave165D25), shortest Distance to Treated Sewerage Outfall (ToOutfalMr), Secchi depth (secchix), Salinity (salinit), shortest Distance to BorrowPit (PitDistMr), and shortest Distance to Borrow Pit of 8 ha or larger (Pit8HecMr).

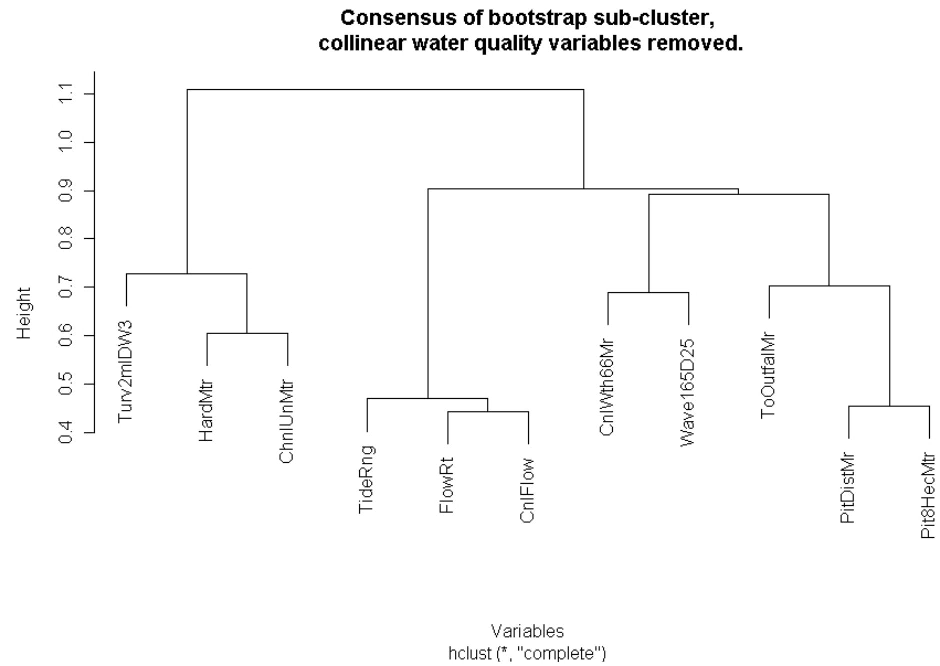
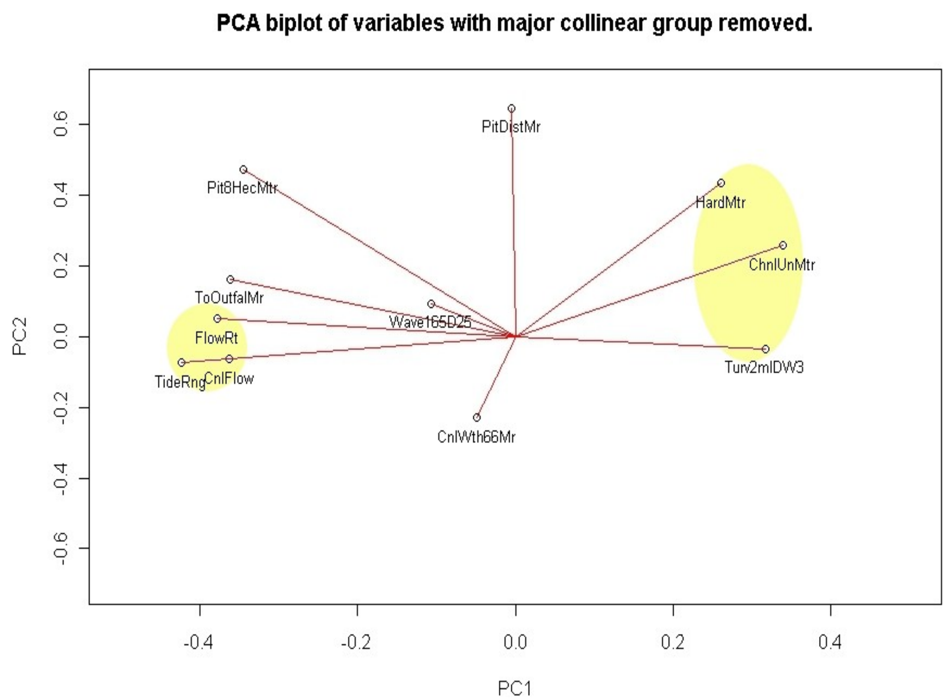


Figure 3.10. A biplot of the non-nutrient variables generated using the R function biplot and including; turbidity (Turv2m1DW3), distance to hard edge (HardMr), Euclidean distance to navigational channels (ChnlUnMtr), Truncated Tidal Range (TideRng), edge Local Flow Rate (FlowRt), Peak Channel Flow Rate (CnlFlow), Minimum Fetch in 1966 (CnlWth66Mr), Storm significant wave height (Wave165D25), shortest Distance to Treated Sewerage Outfall (ToOutfalMr), Secchi depth (secchix), Salinity (salinit), shortest Distance to Borrow Pit (PitDistMr), and shortest Distance to Borrow Pit of 8 ha or larger (Pit8HecMr). The yellow circles indicate a cluster that was collinear and reduced by using a single variable at a time.



The selection of variables to include was determined as follows:

1. The clustering analysis was used such that for a priori hypothesis testing, no more than one variable from each of the 6 clusters (Fig 3.8 and 3.9) was used, counting non-correlated variables as clusters with one member.

2. A primary reduced set of variables was chosen for the analyses. This set included nitrate, euclidean (unlimited) distance to the nearest channel, the modeled flow rate at the marsh edge, the euclidean distance to a borrow pit, the euclidean distance to the nearest outfall, and the channel width in 1966 (Minimum Fetch).

3. Three variables were not included in this cluster analysis as these variables are associated with navigational channels : Direct Distance to Channel (ChnlDistMr), Channel (categories of boat use and boat sizes) and Channelization (DreTh).

Some alternate variables within a cluster were used for testing some hypotheses when they represented a factor more logically and directly associated with the question of interest.

Table 3.3. The response variables.

Response Variables				
	Edge Change 1966-2007 (m) The marsh edge change between 1966 and 2007 at each measurement point.	Add66_07	Primary use variable	
	Edge Change 1926-1966 (m) The marsh edge change between 1926 and 1966 at each measurement point.	Minus26_66	Used for some comparisons	

Table 3.4. The continuous variables, with the independent variables grouped by degree of collinearity with the reduced set labeled as primary variables and showing alternative variables that were substituted in some specific hypotheses.

Independent variables			
(Collinear nutrient group)	Nitrate (μmol NO_3 l^{-1})	Nitrat2	Primary variable
(nutrient data from the Town of Hempstead)	Nitrite (μmol $N0_2$ l^{-1})	nitrite	
	Ammonia (μmol NH_4 /l^{-1})	Ammon_o	
	Phosphates (μmol PO_4 l^{-1})	phosphate	
	Chlorophyll (mg m^{-3})	Chlorox	
	Particulate Organic (mg C l^{-1})	Partico	Alternate variable
	Secchi Depth (m)	secchix	
	Salinity (ppt)	salinit	Alternate variable
(Collinear Hydrological Model group)	Peak Channel Flow (m s^{-1}) (maximum flow rate in channel over a tidal cycle)	CnlFlow	Primary variable
(From GSB model)	Truncated Tidal Range (m)	TideRng	Alternate variable
	Edge Flow (m s^{-1}) (maximum flow rate near the channel edge near the edge change measurement point over a tidal cycle)	FlowRt	Alternate variable
Collinear Mixed Group, collinear for no obvious reason	Channel Unlimited - the Euclidian distance to the channel over obstacles such as marsh islands and filled land (m)	ChnlUnMtr	Primary variable
	Hard Edge Unlimited - Euclidean distance to a hard edge over obstacles (m)	HardMtr	
	Turbidity – midlevel turbidity from 2 to 5 m depth. (NTU)	Turv2mIDW3	Alternate variable
Not co-linear	Distance to Borrow Pit – Euclidean distance to any borrow pit (m)	PitDistMr	Used for sediment
	Borrow Pit 8 Hec+ Dist Direct line distance to a borrow pit > 8 hectares in size (m)	Pit8HecMtr	
	Wave from storms - significant wave height modeled from winds 25 knots and faster. (m)	Wave165D25	Used by itself
	Distance to Treated Sewerage Outfall (m)	ToOutfalMr	Used for nutrient
	Minimum Fetch in 1966 (m) (Channel Width in 1966)	CnlWth66Mr	fetch 1966

SPATIAL AUTO-CORRELATION DUE TO MISSING VARIABLES

Spatial autocorrelation can be found when the values of a variable are correlated with the values of the same variable at nearby locations (Burt et al. 2009, Valcu and Kempenaers 2010). The Spatial auto-correlation can be either inherent when it is a property of the variable or induced if the variable responding as a function of an autocorrelated variable (Lennon 2000, Valcu and Kempenaers 2010). Spatial autocorrelation in the residuals can arise from biological process such as specialization or from modeling a non-linear system in a linear model (Dorman et al. 2007). Spatial autocorrelation also occurs if the model is lacking the proper variables, also called spatial dependency (Guisan and Thuiller 2005, Dorman 2007, Dorman et al. 2007). In some cases the failure to account for these effects can reverse the interpretation of results (Kühn 2007).

Spatial data can be subject to misinterpretation due to spatial autocorrelation which if a form of pseudoreplication (Legendre 1993, Dormann et al. 2007, Valcu and Kempeaers 2010). Simple standard designs are insufficient if spatial autocorrelation is present in the study system, including attempting to space samples far enough apart (Fortin and Dale 2009) or using randomized block design (van Es et al. 2007). Legendre et al. (2002) looked at common field sampling designs and found that, for t tests, the utility of different designs differed between spatial patterns. The response variable and the error terms from linear models should therefore be tested for spatial autocorrelation in order to determine if explicitly methods are required.

Although spatial autocorrelation is a common problem in ecological studies, there is a range of opinions regarding its importance and how or even whether to adjust for it (Diniz-Filho et al. 2007). There are a number of tests for the presence of spatial autocorrelation. In this study, three methods were used. The first was Moran's I, represented using spatial distance weights as:

$$I = \frac{(n\sum_{i=1}^{n}{}_{(C)}(X_i - \bar{X})(X_j - \bar{X}))}{(L\sum_{i=1}^{n}(X_i - \bar{X})^2)}$$

where n represents the number of points, L represents the number of joins, and $\sum_{i=1}^{n}{}_{(C)}()$ is the sum of the values of only contiguous pairs of values (Burt et al. 2009). Correlograms based on Moran's I were plotted using the pgirness packages in R (Bivand et al. 2008). The expected values of Moran's I isare < 0 and observed values greater than the expected indicate clustering while those less than the expected indicate a negative spatial autocorrelation (Burt et al. 2009). This test was used on the response variable and on some specific subsets based on the hypotheses being tested. Correlograms show Moran's I for different distance scales, Figure 3.11 shows Moran's I values indicating no spatial autocorrelation for marsh edge measurement points not along navigational channels. Figure 3.12 shows points that are along navigational channels where some spatial autocorrelation may exist.

Figure 3.11. A correlogram, based on Moran's I, which detected no spatial autocorrelation among measurement points that are not along navigational channels.

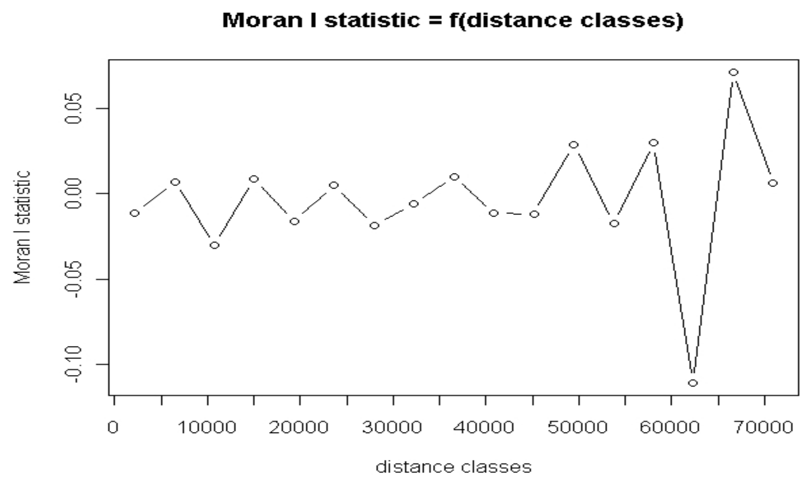
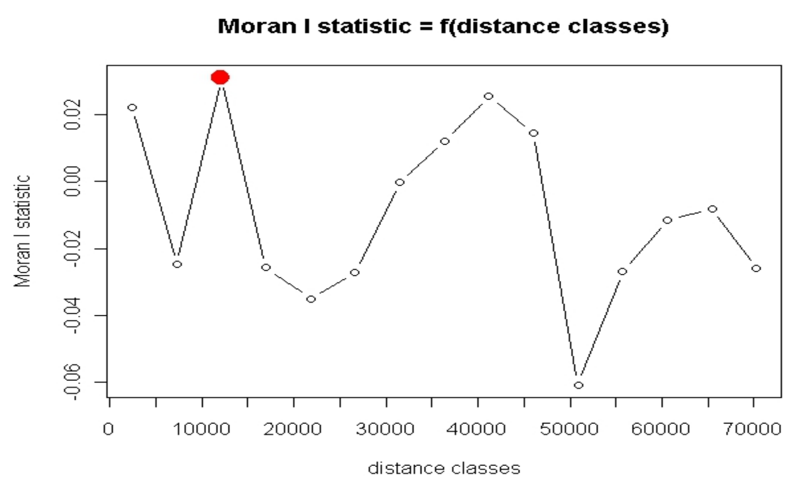


Figure 3.12. A correlogram for marsh edge measurement points along navigational channels, the red dot shows minimal significance at 10,000 m distances.



Another method for detecting autocorrelation is to use variogrames, plots of the variance in paired point measurements against distance between these points, called the lag. The term semivariogram is also used because practical algorithms allocate half the variance to each point (Bachmaier and Backes 2008). If nearby sample pairs typically show less variance than more distantly separated pairs, then autocorrelation is confirmed (Fig. 3.13). The distance at which paired variance levels off, is called the sill and the intercept with the Y-axis is called the nuget. Variograms were generated with the sp package (Bivand et al. 2008) and gstat package in R (Pebesma 2004).

Variograms were calculated as half the average squared difference between paired data values for a given distance (d)

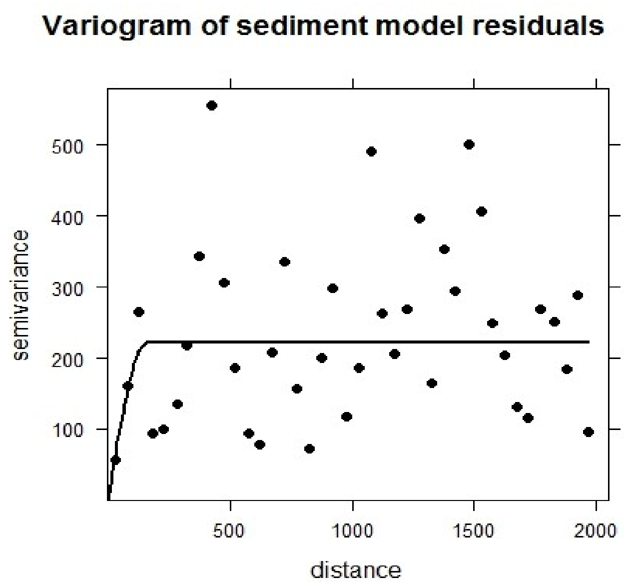
$$\gamma(d)=\frac{\sum_{(d_{ij})}(x_i-x_j)^2}{(2n(d_{ij}))}$$

where d_{ij} indicates summation across all points at distance d and x is a some variable containing observations of values that are ordered along one or more dimensions (Burt et al. 2009).

Fig. 3.13 shows a variogram plot of the residuals from a three variable model that was used to test the a priori hypothesis regarding factors that may affect marsh gain or loss through the delivery or erosion of sediment (Distance to borrow pits, turbididty and Secchi). There was no significant spatial autocorrelation.

A third method for detecting spatial autocorrelation is to plot the residuals in spatial coordinates and visually inspect the plot for a spatial pattern. Bubble plots of residuals were created using the bubble function included in the gstat package for R (Pebesma 2004). Because none of the various ways of estimating the presence of autocorrelation are perfect, spatially mapping the residuals for visual inspection is considered one of the best approaches (C. H. Graham, Stony Brook University *personal communication*)..

Figure 3.13 A variogram of sediment values (Distance to borrow pit, and turbidity) showing only a minor degree of spatial autocorrelation, if any.



TEST METHODS FOR CATEGORICAL VARIABLES

The first a priori hypotheses to be tested was that producing channelized edges resulted in different amounts of marsh loss compared natural marsh edges. Because the response variable, the amount of marsh edge loss for the time interval 1926-2007, was not normally distributed, nonparametric methods were used to compare the changes in the edges of channelized marshlands. Wilcoxon ranked sum tests were used to make comparisons of the response variable among classifications of channelized and non-channelized edges. Tests were carried out in an increasingly stringent sequence, starting with a comparison between channelized edges and all other edge types. All but one sample point for channelized edges was located on a navigational channel. Therefore, the most strict comparisons among channelized and non-channelized edges were made for points along navigational channels of the same use category and of the same range of widths as those adjoining Channelized edges.

Next, the possible effects from boat traffic (Navigational Channels) were analyzed. A series of increasingly stringent tests were made to compare non-Channelized marsh edges that were along Navigational Channels with non-Channelized edges that were not along Navigational Channels. A Kruskal-Wallis test and Wilcox ranked sum tests were used to make comparisons of the response variable among points found on channels with different boat use classification for non-channelized edges. Marsh loss was compared between edges from all navigational channels and those found on similarly sized waterways that were not used for navigation.

Finally, the hypothesis that boat use had an effect on marsh edge change was analyzed. Wilcox ranked sum tests were used to test for differences in the response variable among channel use categories. Because anthropogenic effects such as boat wake and displacement driven currents decrease with distance, the distances from the marsh edge measurement points located along channels and the navigation channels that they were located along were regressed against the changes in the response variable. The hypothesis was that the channels that were heavily used by vessels would show much reduced erosion rates with distance. Navigational channels that are rarely used should show little difference in edge loss due to the distance from navigational traffic. A T' post hock test (Sokal and Rohlf 1995) was used to compare among the different boat use classifications using the slopes produced by regressing edge change against distance separately for each boat use classification.

MULTIPLE REGRESSION AND MODEL BUILDING

Most of the remaining hypotheses tested the effects of various continuous variables (Table 3.3). .

Continuous variable methods

There were no a priori expectations for any single form of relation between independent and response variables. The possibility of non-linear relationships was explored for each hypothesis and linear methods were only used if appropriate. The GLM, GAM or GAMM from the car package (Fox and Weisberg 2011) in R were used to test the assumption of linearity for the particular model as recommended by Zuur et al. (2009). The recommendation was to use non-linear line fitting functions (smoothers), such as spline fitting, in GAMM to see if they plot as curves (Zuur, et al. 2009). If straight lines were produced, then linear models (OLM) were parsimonious and preferable, and a larger range of statistical techniques and tools based on linear regression were then valid to use (Zuur, et al. 2009).

Model mixing algorithms were used as initial steps to guide the building of linear models. AIC-based model averaging was performed using the MMIX package in R (Makowski et al. 2009, Morfin and Makowski 2009), and Bayesian model averaging was performed using the BMS package (Zeugner 2011). Model averaging is designed as an alternative method to stepwise model selection that identifies useful parameters by comparing performance across all combinations of variables. By testing all models, these methods implicitly avoid problems of path dependency and local optimization minima that cause problems when using stepwise regression. However, these methods do not produce R^2 values, fail to include interactions between variables, and do not provide output for use in software that estimates relative importance.

Forward stepwise regression was then used, with a model derived from model mixing as the starting model and a fully interacting model as its forward limit. Backward stepwise regression was then used, starting with the forward result, to check for excessive terms. This process added interaction terms, produced R^2 values, and was compatible with other software tools.

Continuous independent variables typically indicate the use of type II regression methods, but if assumptions are not met or more than correlation is needed, then OLS can be used with the understanding that power is lost and the significance values cannot be trusted while the R^2 values remain valid (A.J. Rohlf, Stony Brook University, personal communication).

THE RELATIVE IMPORTANCE OF FACTORS

For multiple regression studies the assignment of relative importance to the independent variables can be a challenge. The R^2 is a simple estimate of the explained variability, either as multiple R^2 within a given sample set or the adjusted R^2 that is corrected for sample size and inflation factors and allows for comparisons between studies. Early work on the general problem of decomposing R^2 was not encouraging (Williams 1978, Kruskal 1984, Kruskal and Majors 1989). Several methods for overcoming this problem have been proposed by (Linderman et al. 1980, Pratt 1987, Genizi 1993, Feldman 2005). The method that averages sequential sums of squares over all orderings of regressors (Linderman et al. 1980) is implemented in the relaimpo package for R (Grömping 2009) and was used here.

Chapter 4. ANALYSIS AND RESULTS: CHANNELIZATION AND NAVIGATIONAL CHANNELS

There were 4 a priori hypotheses that included categorical variables. I hypothesized that marsh loss was greater where edges were cut by dredging than along natural marsh edges. I also hypothesized that marsh edges along navigational channels that were cut by dredging eroded more than the edges of navigational channels that were not. The third hypothesis was that, for natural marsh edges, those along navigational channels erode faster than edges that were not along channels. Finally, I hypothesized that marsh loss in navigational channels would be correlated with the level of boat traffic and the proximity of the channel to the marsh.

EFFECT OF CHANNELIZATION ON SALT MARSH LOSS

Does channelization increase marsh loss?

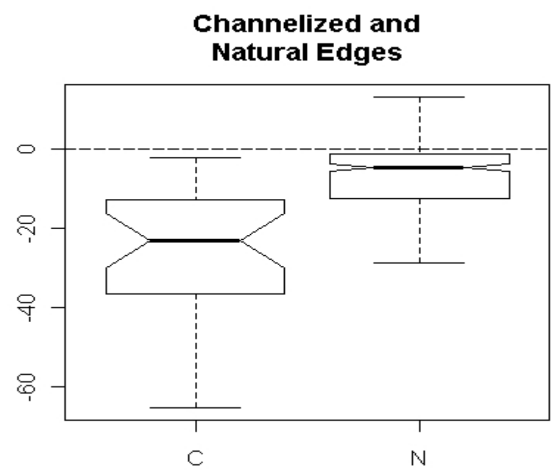
The response variable for these tests was the change in the location of the marsh edge between 1926 to 1966 and 1966 to 2007. Differences in these changes were compared between channels cut by dredging (channelized) or natural marsh edges and among channel classes described in Table 3.1.

For the edge change data from 1966 to 2007 there were 30 measurement points that fell on channelized marsh edges and 408 points that fell on natural marsh edges. The change in location of the marsh edge between 1966 and 2007 for measurement points located along channels created by dredging was compared to the change observed for those points along natural marsh edge. Because the untransformed response variable was non-normal, unpaired Wilcoxon tests were used in these comparisons. False discovery rate control (Benjamin and Hochberg 1995) was employed when multiple tests were performed for the same data set. On average, the edges of marshes that were channelized retreated 27.52 (± 3.82) m while those points on natural edges of marsh retreated on average 8.38 (± 0.76) m (Table 4.1). A simple boxplot illustrates the large difference between channelized and natural edges (Fig. 4.1).

Table 4.1. Wilcox Rank Sum test with continuity correction, for differences in salt marsh loss between 1966-2007 comparing edges of the marsh that were created by channelization and all natural marsh edges.

-	P	(n)	mean change (m)	Median (m)	SD	SE
Not Channelized	-	408	-8.38	-4.74	15.32	0.76
Channelized	<< 0.001	30	-27.52	-23.04	20.91	3.82

Figure 4.1. The change in the location of the marsh edges that were Channelized (C) between 1966 and 2007 and natural edges facing any type of water body (N). The median values are represented as bold lines, the boxes represent the upper and lower quartiles, whiskers represent extreme values, and a Tukey notch test where the groups are significantly different if the notches do not overlap.



To take into account the influence of waterway width, measurement points for both categories were restricted to marsh edges along water bodies with widths within the range typical for the channelized edges, 76 m to 400 m (Fig. 4.2). For these waterways, the channelized marsh edge retreated an average of 29.41 (± 4.19) m while those points on natural edges of marsh facing similarly sized waterways retreated on average 10.62 (± 1.18) m (Table 4.2).

Figure 4.2. The frequency distribution of channelized and natural navigational channel edges with widths of 75-400 m in 1966.

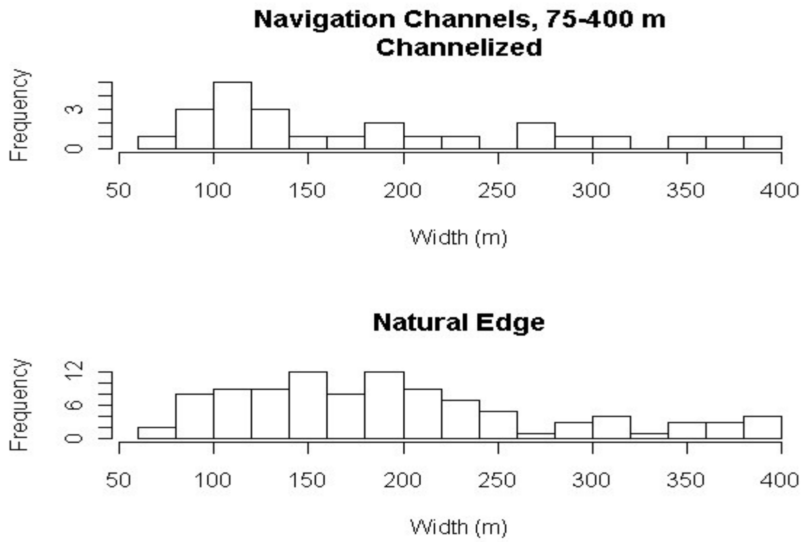


Table 4.2. Wilcox Rank Sum test with continuity correction, for 1966-2007, comparing loss of marsh between channelized areas 75 m to 400 m wide and natural marsh edges along all other waterways of the same range of widths.

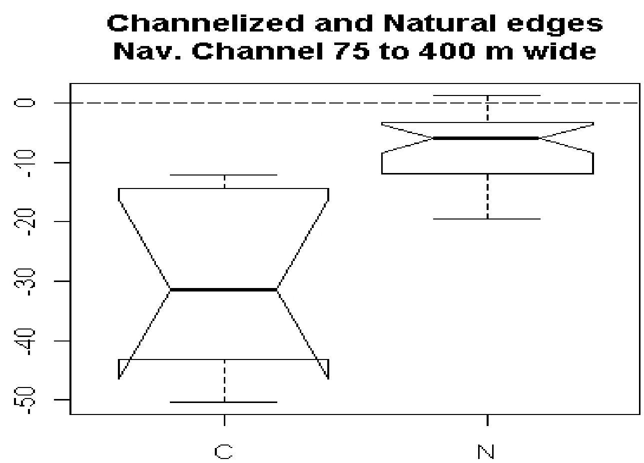
-	P	(n)	mean	median	SD	SE
Not Channelized	-	177	-10.62	-8.98	15.72	1.18
Channelized	<< 0.001	26	-29.41	-26.31	21.36	4.19

This analysis was repeated to include only measurement points on channelized marshe edges that were also along navigational channels and natural marshes that faced navigational channels 75 - 400 m wide. In this case, the channelized marsh edges retreated an average of 30.05 (± 4.21) m while those points on natural edges of marsh facing similarly sized navigational channels retreated on average 13.59 (± 1.21) m (Table 4.3). Again, there was a significant difference between these two groups.

Table 4.3. Wilcox Rank Sum test with continuity correction for 1966-2007, comparing loss of marsh between channelized areas and natural marsh edges along navigational channels 75 m to 400 m wide.

-	P	(n)	mean	median	SD	SE
Not Channelized	-	100	-13.59	-11.76	12.07	1.21
Channelized	P << 0.001	25	-30.5	-27.98	21.05	4.21

Figure 4.3. The loss of marsh from Channelized edges (C) and natural marsh edges (N) along navigational channels 75-400 m width between 1966 and 2007. These box and whisker plots show the group median values (heavy line), the upper and lower quartiles, and the whiskers represent the range. Notches show significant differences between groups if the notches do not overlap.



I then examined whether the loss of marshlands from channelized marshes was different from the loss in marshes with natural edges when both groups were classified by boat use categories for marshlands facing open water between 75-400 m across. Categories ranged from high use by larger boats (A) to occasional use by small boats (E) and edges that face waterways that were not considered navigable (X) (Table 3.1). The characteristics of these categories are summarized in Table 4.4. The data are illustrated in a violin plot Fig. 4.4, which shows the median, upper and lower quartile of data, the range as well as the distribution of data points. The results of Wilcox tests are in Table 4.5. Given the small number of measurement points for some categories, this test had relatively low power. The only significant difference was between the highest use categories (A and B) and the lowest use categories (D and E).

Table 4.4. Change in marsh edge from from 1966 – 2007 for channelized and natural marshes along navigational channels of different boat use categories. The categories are described in Table 3.1. Negative numbers indicate a loss of marsh.

Use	Channelized median (m)	mean	n	SD	SE	Natural median (m)	mean	n	SD	SE
A	-30.33	-43.68	6	33.27	13.58	-15.42	-15.89	16	13.89	3.47
B	-31.42	-27.51	11	14.6	4.4	-10.07	-13.73	19	11.72	2.69
C	-13.94	-23.02	3	17.42	10.06	-11.49	-14.02	26	15.8	3.10
D	-22.93	-22.93	2	9.63	6.81	-12.38	-13.9	23	7.12	1.48
E	-21.45	-27.62	3	20.3	11.72	-7.82	-10.00	16	9.57	2.39
X	-2.26	-2.26	1	NA	NA	-5.85	-6.77	77	18.88	2.16
All	-26.305	-29.41	26	21.36	4.19	-8.98	-10.62	177	15.72	1.18

Figure 4.4. Loss of marsh from 1966 to 2007 for channelized edges (DT), and natural edges along navigational channels categorized by levels of boat use (high A - low E, see Table 3.1), and locations away from navigational channels (X) for waterways 75 – 400 m in width. These violin plots have a box plot showing the upper and lower quartile and a median line and then surrounded by the distribution of the data.

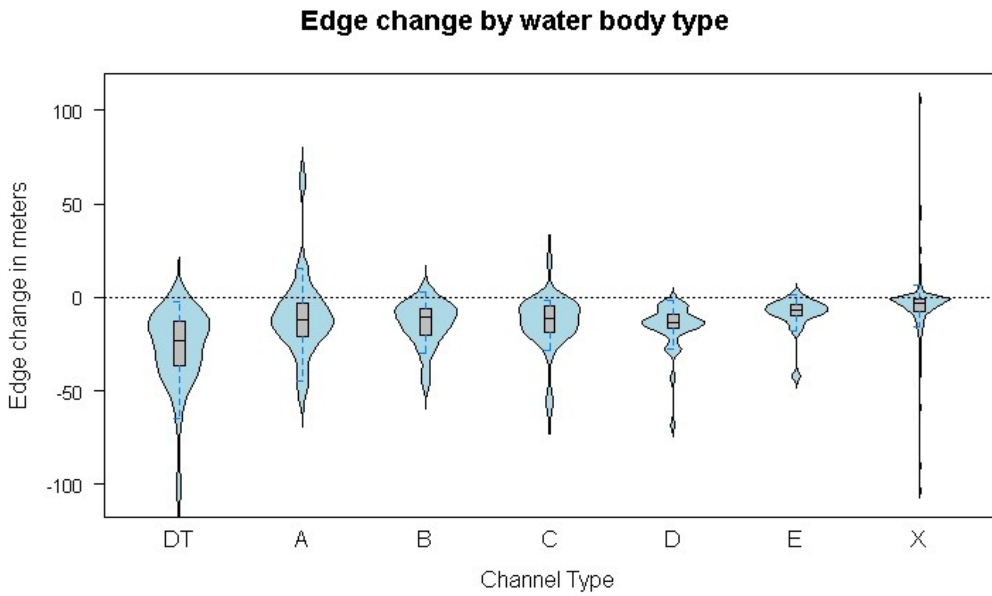


Table 4.5. The results of a Wilcox Rank Sum test for marsh loss from 1966-2007 comparing channelized versus natural shores, grouped by navigational channel classification for waters 75 - 400 m wide.

\Natural shores -------------- Channelized \	A	B	C	D	E
A	0.02	0.009*	0.009*1	0.001*	0.001*
B	0.06	0.015*	0.003*	0.005*	0.002*
C	0.79	0.36	0.39	0.35	0.05
D	0.39	0.24	0.30	0.08	0.08
E	0.56	0.16	0.20	0.28	0.03

P values that are significant when using a Benjamani-Hochbrg correction for multiple tests on the same data are indicated with a *.

Marsh edges that were channelized between 1890 and 1966 showed higher loss rates than natural marsh edges through time (Fig. 4.5). The observed losses were surprisingly linear through time

(Fig. 4.5). The loss of marsh edge (relative to the 1994 reference which provides the zero point) over the 81 year photographic record indicates that the loss of marsh from channelized edges shows little, if any, sign of slowing through time (Fig. 4.6), and is consistently greater than all natural edges of marsh along different classes of navigational channels.

Figure 4.5. A generalized additive model fit of loss of marsh from the edge through time. These plots are relatively linear, indicating that the marsh loss has been at a relatively constant rate through time. Lines represent splines and confidence limits around splines showing mean marsh change for channelized marsh (DT), and natural marsh points facing navigational changes with different degrees of boat use (A - E), and those points not on a navigational channel and not channelized (X). (See Table 3.1 for definitions of channel use categories)

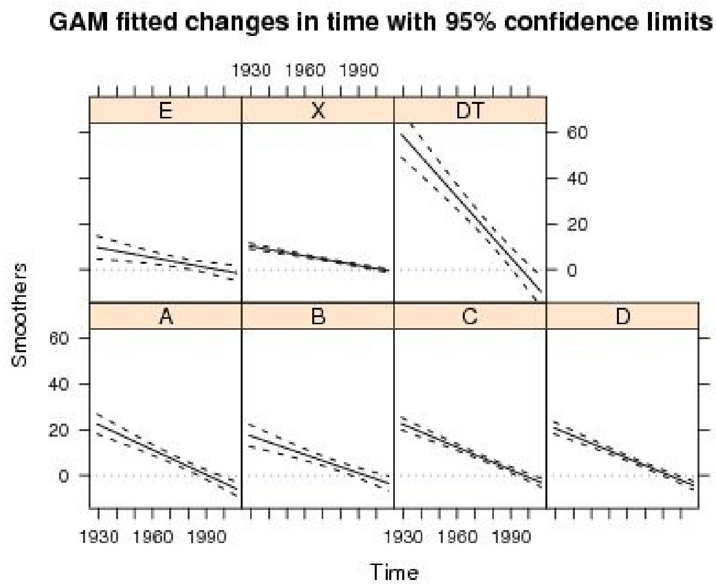
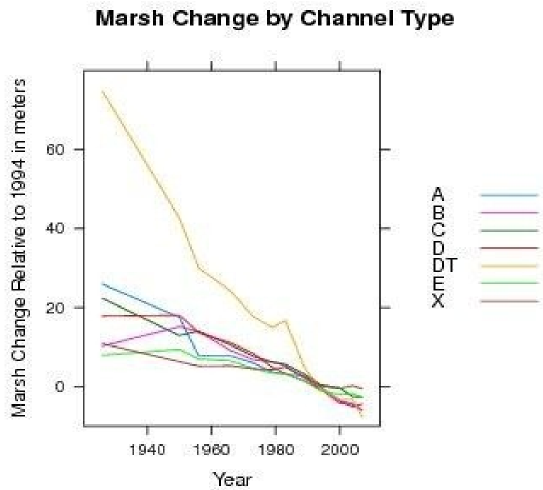


Figure 4.6. Mean values of marsh edge change relative to the 1994 marsh reference line from 1926 to 2007 for marsh edges that were channelized (DT), natural marsh edges along navigational channels with different degrees of boat use (A - E), and those points neither along a navigational channel nor channelized (X).



The overall high rate of marsh loss from areas that were channelized continued long after the initial damage to the marsh. Because the channelized marsh edges are historical artifacts, and are significantly different than the natural marsh edges of all types, channelized edges were not included in subsequent analyses.

EFFECTS OF NAVIGATIONAL CHANNELS VERSUS NON-CHANNEL WATERWAYS ON MARSH LOSS

I then tested whether there was a difference in marsh loss differed among classifications of the navigational channels for natural marsh edges from 1966 - 2007 (Table 3.1). Because of the non-normality of the response variable, non-parametric methods were used. A Kruskal-Wallis test was used to determine if significant differences existed between shores along navigational channels versus those not on such channels. Unpaired Wilcox tests were used to test for differences among categories of navigational channels with each other and with edges that did not lie along navigational channels.

There were significant differences in marsh loss rates between groups when edges away from navigational channels were included in a Kruskal-Wallis test, but no significance when only navigational channels were included in the test (Table 4.6). The general statistics show that non-channel natral marsh edge locations changed much more slowly than marshes along navigational channels, but there was no difference among marsh edges found along different types of navigational changes (categoreis) (Table 4.7). The same was true when channels a wider range of channel widths (65 - 12400 m instead of 75 - 400 m) was used; the changes in natural edges along navigational channels were still significantly different from other natural edges (Table 4.8).

Table 4.6. Kruskal-Wallis tests for change in the edge of the marsh from 1966-2007 for natural, non-channelized waters 75 - 1200 m wide showed a significant difference among groups when edges that do not lie along navigational channels were included, however no significant difference was found among navigational channel classifications.

Test	Chi-sq	df	P	n
Navigational channels versus others	24.48	5	<< 0.001	250
Among Navigational Channel classifications	4.86	4	0.30	145

Table 4.7. The mean, standard deviation, standard error, and number of points for marsh loss for all natural marsh edge navigational channel classifications

Channel use classification	mean edge change 66-07	SD	n	SE
A	-15.89	13.89	16	3.47
B	-13.73	11.72	19	2.69
C	-14.02	15.8	26	3.10
D	-13.9	7.12	23	1.48
E	-10	9.57	16	2.39
X	-6.77	18.88	77	2.15
All	-10.62	15.72	177	1.18

Table 4.8. Unpaired Wilcox Rank Sum test comparing edge change from 1966-2007 between natural edge along navigational channels and those located on other waterways for water bodies of 65 - 1200 m width.

-	P	mean	(n)	SD	SE
Not Navigational Channels.	-	-7.39	105	17.08	1.63
Navigational Channels	<< 0.001	-14.18	145	15.03	0.10

General statistics for navigational channels from 65 - 1200 m wide showed that loss rates from the highest use category (A) to the moderate use categories (D) did not decrease progressively (Table 4.9). However, loss rates were much lower for the lowest use category (E) and the edges that do not face navigation channels (X) (Table 4.9). Among the 6 waterway categories for channels from 65 - 1200 m wide, the non-channels continued to be significantly different from categories A and D but were not significantly different from B, C and E (Table 4.10). The Tukey notches in Fig. 4.7 suggest that shorelines along channels categorized as E are more like those shores categorized as X than it is to the other navigational channels.

Table 4.9. The general descriptive statistics, mean, median, and ranges, of channel and non-channel edge change, for waterways with widths in the range of 65 - 1200 m to eliminate the smaller non-channel water bodies. See table 3.1 for descriptions of the type of boat use categories used here.

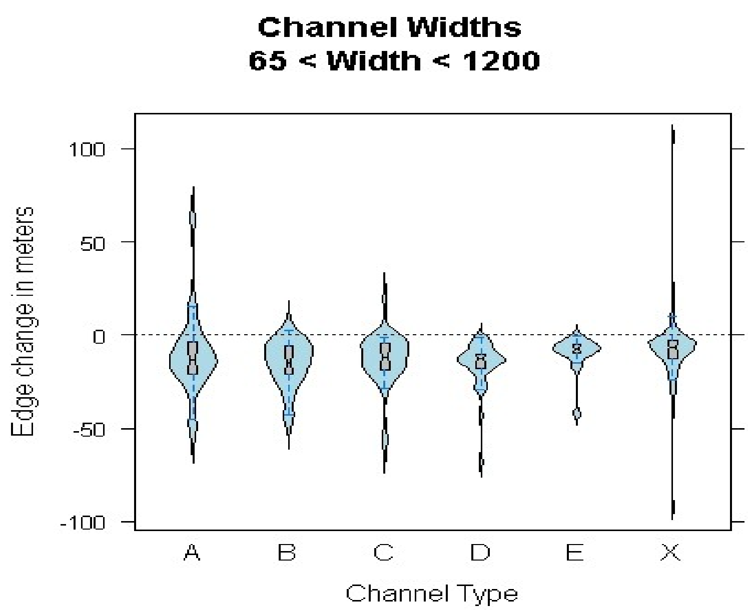
Type	Greatest loss.	1st Qu.	Median Loss	Mean	3rd Qu.	maximum gain or minimum loss.	SD	n	SE
A	-50.95	-19.90	-12.95	-10.72	-3.44	62.29	21.58	24	4.40
B	-44.84	-20.54	-14.84	-15.33	-5.69	2.80	12.49	23	2.60
C	-58.83	-18.73	-11.34	-13.17	-4.19	19.17	13.94	35	2.46
D	-68.22	-17.62	-12.50	-15.77	-10.09	-1.38	12.61	35	2.13
E	-41.98	-9.45	-7.26	-9.60	-4.54	-0.08	9.41	17	2.28
X	-91.07	-12.14	-6.13	-7.39	-2.61	105.40	17.08	105	1.67

Table 4.10. Wilcox Rank Sum test on edge change 1966-2007 for waters 65 m to 1200 mwide P-values for similarity. See table 3.1 for descriptions of the type of boat use categories used here.

-	A	B	C	D	E	(n)	SD
A	-	-	-	-	-	24	21.58
B	0.57	-	-	-	-	23	12.49
C	0.29	0.76	-	-	-	35	13.94
D	0.27	0.77	0.67	-	-	35	12.61
E	0.04*	0.57	0.62	0.08	-	17	9.41
X	0.002*	0.06	0.06	0.002*	0.23	105	17.31

P values that are significant when using a Benhamini-Hochberg correction for multiple tests on the same data are indicated with a *.

Figure 4.7. A violin plot with notched boxes that shows edge loss between 1966 and 2007 for natural edges along navigational channels and other waterways that are the same width as navigational channels. This violin plot has a box plot showing the upper and lower quartile, a median line and a Tukey notch (two groups are significantly different if the notches do not overlap) and is surrounded by the distribution of data points. Based on the Tukey notch, A through D are not significantly different from each other, but are different from E and X.



The changes in the marsh edge along navigational channels were different from other natural marsh edges, particularly when the smaller non-channel water bodies were included in the analysis. The edge change along lowest use category of navigational channels was the most similar to that of the changes along natural edges that were not navigational channels. It appeared that natural edges along navigational channels may form a separate group, distinct from other natural marsh edge types. Based on these results, I analyzed the natural marsh edges along navigational channels separately from the other natural edges in subsequent analyses that included continuous variables.

DIFFERENCES AMONG NAVIGATIONAL CHANNELS BASED ON USE LEVELS

There is often the assumption that boat use has a large impact on the salt marsh. If this is true, then it may be possible to distinguish the effects of boat use from other factors affecting marsh loss or gain. If, however, other factors mask the effects of boat use, then the impacts of boats may be less important than was previously thought. Table 4.9 shows the mean amount of marsh lost for each of the ranked categories, where A is the likely heaviest use (larger and more boats) and E the lowest use (fewer boats, no commercial traffic). There is no apparent trend in marsh loss with increased or decreased boat use. Two other approaches were used to determine if increased boat use was correlated with marsh loss. First, proximity to the center of the channel may have a greater impact on the edges of heavily used channels than lightly used ones. Second, the effect of boat use could be detected by examining the difference in marsh loss for the same channels between 1926 to 1966, when powerboat use was low, and during the latter period of 1966 to 2007, when powerboat use was much greater.

Distance to the Center of Navigational Channels

If navigational traffic has an effect, the expectation is that nearby traffic would be more damaging to the marsh and that this factor would be exaggerated with larger boats. Waves from boats attenuate with distance traveled and shorter period waves, as would be expected from small boats, attenuate more quickly (Denny 1988). Boat channel classifications with heavy use (A and B) were expected show a greater rate of erosion when the channel was close to the edge being measured than when the channel was distant. Lightly used channels (D and E) were expected to show little difference with distance, or to possible show the opposite effect if channel width was related to increased erosion, as seen below for the edges facing the remaining waterways (class X).

There were 179 measurable marsh edge points facing channels. All points in the 1966 to 2007 data set were included. This analysis also included the additional variable not included in the initial tests for collinearity, the measurement of direct distance to channel center, a measurement that is not possible for edges other than those along channels.

Over the entire 81 year period (Fig. 4.6) there was increased loss of marsh edge for shores along channels with greater boat traffic and larger boats. The distance from the marsh edge to the center of the channel also affected marsh loss, but it was not not possible to determine how these factors may interact (Fig. 4.8) as the distance to the center of a channel also differed among channel classifications (Fig. 4.9).

When all of the data were pooled across boat use categories, a single regression of LP transformed marsh edge change (see Methods: *Characteristics of response variable: Transformation*) as a function of distance to channel center was significant (pooled channel edges in Table 4.11). When shoreline points were examined by category, there was a significant effect of distance to the center of the channel for the high use category (A), where the slope showed a positive correlation between marsh edge position and the distance to the channel, indicating retention or marsh growth when distant (Table 4.11).

Table 4.11. For navigational channels as a group (pooled) and individual categories (A-E, see table 3.1) the LP transformed marsh (see methods) edges change 1926-2007 were regressed as a function of distance to channel center. A small potential trend is seen in slopes from linear regression results for edge loss within channel use classifications regressed on distance to channel center.

Use Catg.	Slope	P-value	Intercept	P-value	R^2	Adjusted R^2
Pooled	0.003	<0.001	-2.731	<<0.001	0.09	0.088
A	0.008	0.001	-3.737	<<0.001	0.38	0.354
B	0.003	0.24	-2.772	<<0.001	0.05	0.089
C	0.000	0.95	-2.238	<<0.001	0	0.000
D	0.000	0.81	-2.616	<<0.001	0	0.000
E	-0.01	0.21	-1.249	0.04	0.09	0.039

Figure 4.8. Plots of edge change between 1966-2007 along channels of use categories from high (A) to low (E). Only A had a significant correlation, which was positive, indicating that erosion was less when boats were more distant from the shore (Table 4.12).

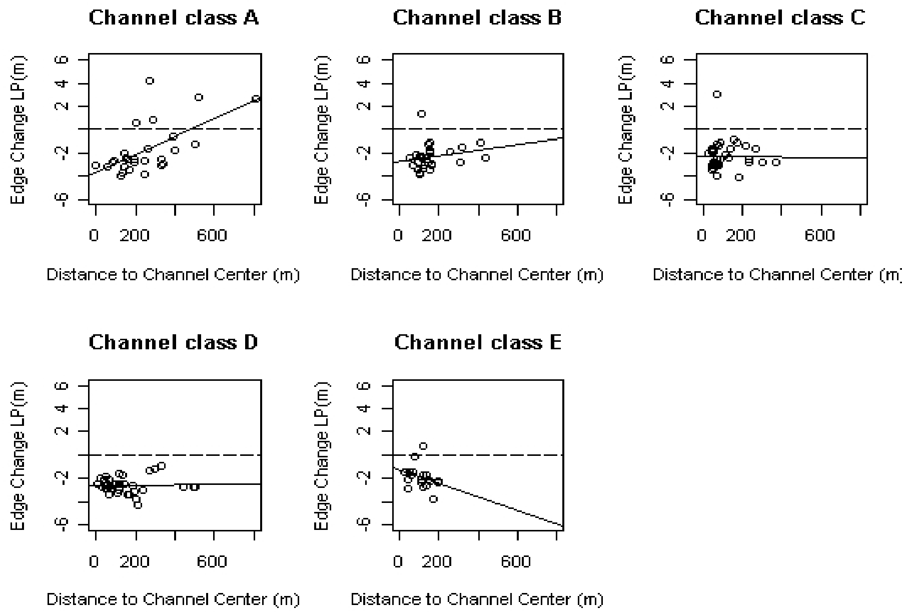
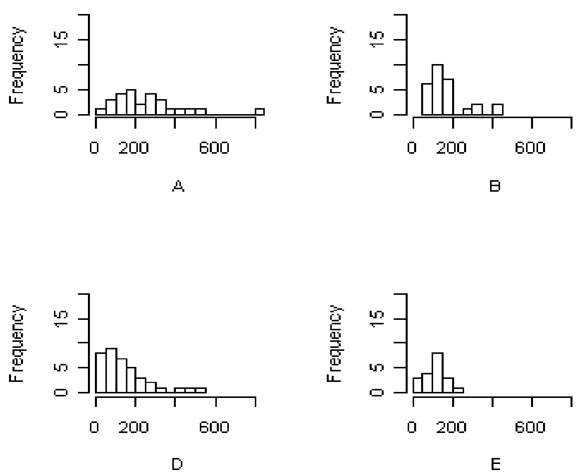


Figure 4.9. Distribution of the distance to the center of the channel for marsh edge points that were along each of the boat use categories (A-E).



I also used transformed response variable in an ANCOVA to test for differences in marsh edge change among the 5 categories of navigational channels, using distance to the center of the channel as a covariate. Other than high use category B, all use categories were significantly different from high use category A (Table 4.12), and the adjusted R^2 was 0.107.

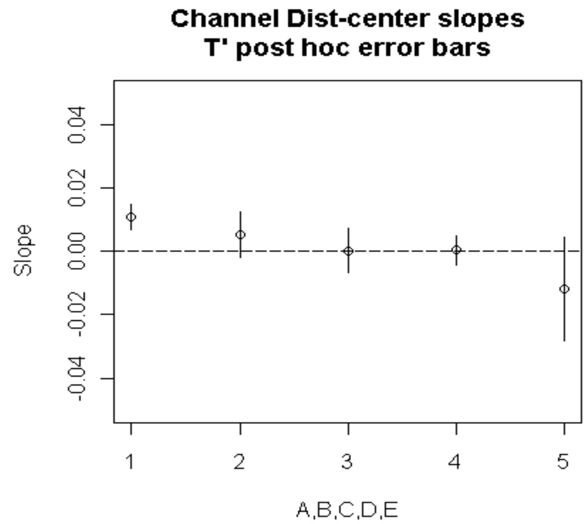
Table 4.12. By comparing the marsh edge change 1926-2007 and distance to navigational channel center between channel use categories, only the slope of A was significantly different from 0 (Table 4.14), but the other slopes differ significantly from them.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
ChannelA		-4.26	0.8	-5.32	<< 0.001
ChannelB		-2.53	1.12	1.55	0.12
ChannelC		-1.01	0.99	3.3	0.001
ChannelD		-0.63	0.96	3.79	<< 0.001
ChannelE		0.24	1.27	3.53	0.001
ChnlDistMr:ChannelA	0.65		0.15	4.29	<< 0.001
ChnlDistMr:ChannelB	0.34		0.22	-1.42	0.16
ChnlDistMr:ChannelC	0.06		0.2	-2.98	0.003
ChnlDistMr:ChannelD	-0.04		0.19	-3.74	<< 0.001
ChnlDistMr:ChannelE	-0.19		0.26	-3.19	0.002

Multiple R-squared: 0.154, Adjusted R-squared: 0.107 AIC = 268.44

The relative importance metrics from the relaimpo package (see Methods: relative importance) calculated that the model explained 11.75% of the variance in fitting edge loss by both distance to channel and channel use classification. It was also calculated that channel classification explained 7.02% of the variation, the interaction between distance to channel center and channel type explained 2.42% of the variance, and distance to channel explained the remaining 2.31%. The T' post hoc test (Sokal and Rohlf 1995) was used for a posteriori testing of significance of differences among slopes (Fig. 4.10). The overall difference in slope between the 5 categorizes was highly significant, with a value of F = 157.98 and P << 0.001. Although the differences are only significant for the endpoints, the fact that they form a trend can still be a powerful indication (F.J Rohlf, Stony Brook, *Personal communication*)

Figure 4.10. A T' post hoc test for difference in slopes found a significant difference between edge change along the high use category A channels and the two low use categories D and E, however other combinations were not significantly different, although this graph does show a trend.



A MANOVA was run with the two periods of edge change as response and the direct distance to channel center and channel classification as independent variables (Table 4.13). A difference in erosion rates between the two time periods indicates that a significant increase in edge loss occurred from 1966 - 2007 as compared to the 1926 - 1966 time interval (Table 4.14).

Table 4.13. A Hotelling-Lawley test on Channel Classification and direct channel center distance to the edge changes for the two periods, 1926 to 1966 and 1966 to 2007, for n = 142 points in common.

	df	H-L	approx F	Num DF	Den DF	Pr > F
Channel Center Distance	1	0.08	5.09	2	135	0.007
Chanel Classification	4	0.18	3.09	8	268	0.002

Table 4.14. ANOVA results for testing marsh losses from different categories of navigational channels for the two time periods, 1926-1966 and 1966-2007.

Change 1966-2007	Df	Sum Sq	Mean Sq	F val	Pr > F
Distance to Channel Center	1	33.15	33.15	10.18	0.002
Channel Classification	4	31.5	7.88	2.42	0.05
Residuals	136	442.93	3.26		
Change 1926-1966					
Distance to Channel Center	1	0.1	0.095	0.01	0.92
Channel Classification	4	131.97	32.993	3.75	0.006
Residuals	136	1196.41	8.797		

Although other effects may obscure the effects of boating on marsh edge change, the distance to the center of the channel for heavy use navigational channels had a greater effect than for light use channels. However, other factors such as channel flow in a meander stream may also show a similar relationship with distance to center of the channel. The comparison between pre and post 1966 shows a difference in the relationship between use categories that could indicate the impacts of increased powerboat use. There seems to be a relation between navigational use and edge change, but other factors may have greater effects. It is, however, important to include navigation channels in subsequent analysis of subsequent hypotheses in order to understand how edges along channels interact with other influences.

Chapter 5. ANALYSIS AND RESULTS: LINEAR MODELING

NON-CHANNELS AND NATURAL PHYSICAL FACTORS

I extracted from the entire data set those marsh change points that were not located along a navigational channel. For tests with these points, I used the variables Minimum Fetch (minimum distance to land on an opposite shore in 1966, = CnlWth66Mr), Truncated Tidal Range (TideRng), and Salinity (salinit).

General characteristics of variables: Minimum Fetch

The simple statistics for Minimum Fetch are shown in Table 5.1. An initial regression of this variable against the LP transformed showed that this was a significant factor (Table 5.2). However, the Minimum Fetch data were heteroscedastic (Fig. 5.1).

Table 5.1. The Minimum Fetch (m) for marsh sampling points in 1966. These statistics are for the untransformed data for points located along non-navigational channel edges.

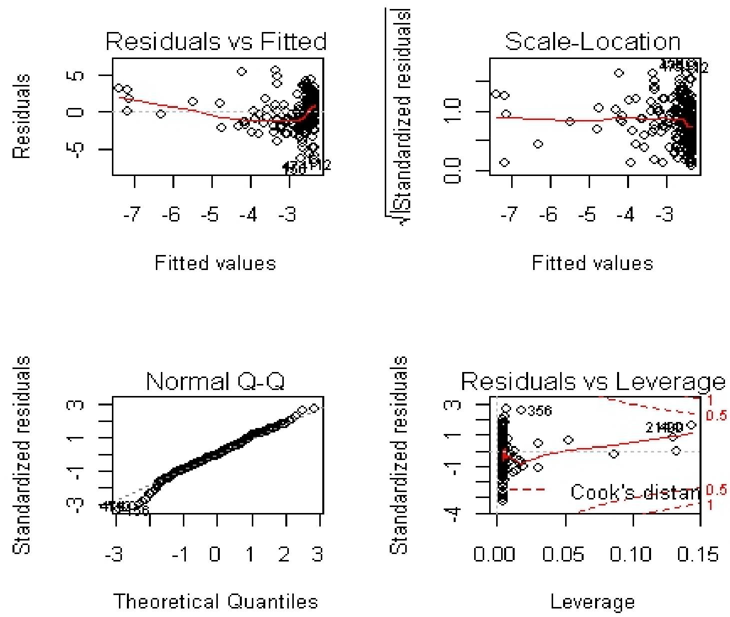
Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
0.91	15.71	50.38	150.7	133.2	1896	290.01	18.25	258

Table 5.2 The regression of Minimum Fetch in 1966 for marsh edge sampling points against the LP transformed response variable, change in marsh edge position from 1966 – 2007.

Parameter	Estimate	Std. Error	t value	Pr(>|t|)
Intercept	-2.31	0.15	15.57	<< 0.001
Slope	-0.003	0.000	-5.86	<< 0.001

Multiple R-squared: 0.014, Adjusted R-squared: 0.008

Figure 5.1. **A** diagnostics plot of the transformed response variable and the untransformed variable Minimum Fetch. The residuals did not follow the expected normal distribution.



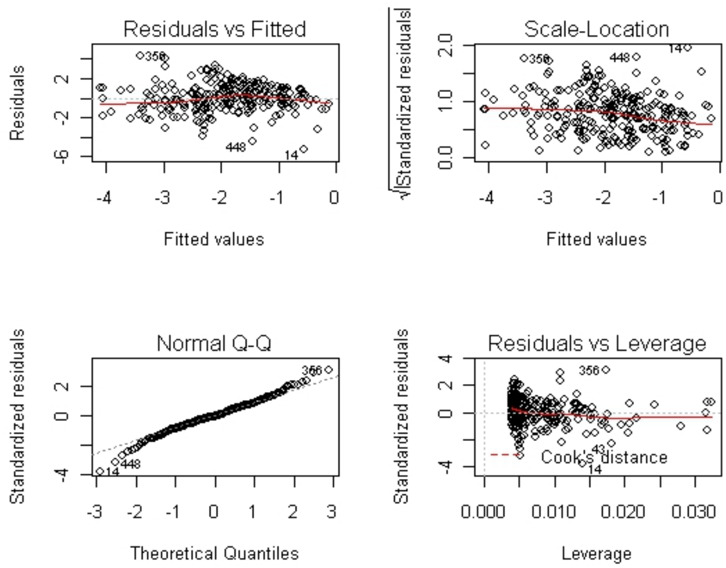
By using both the LP transform of the response variable and a Box-Cox transformation on Minimum Fetch, I was able to correct this problem (Fig. 5.2). The regression with the transformed data was still significant (Table 5.3), and the R^2 value was 0.23.

Table 5.3. The improved linear regression results for the regression of Minimum Fetch on Edge Change between 1966 and 2007 for non-channel edges.

		Std. Error	t value	Pr(>|t|)
Intercept	-1.38	0.26	-5.38	<<0.001
Slope	-0.38	-0.46	-8.17	<<0.001

Multiple R-squared: 0.23, Adjusted R-squared: 0.23

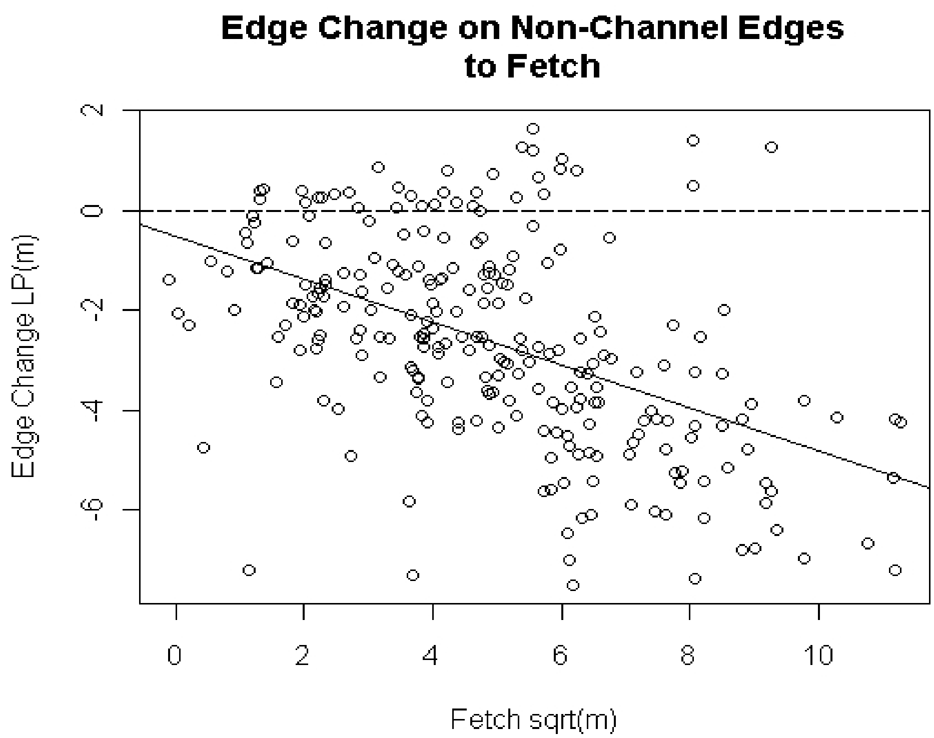
Figure 5.2. The variable diagnostics after the data transformations, including the sign adjusted Log-Power (LP) transform on the response variable (Edge Change 1966-2007) and a Box-Cox transform (lambda = 0.1) on the independent variable (Minimum Fetch).



The final regression for the Minimum Fetch regressed on marsh edge change between 1966 and 2007 for natural edges not along a navigational channel suggested two different patterns in the data. Overall, there was a significant increase in marsh edge loss with increasing fetch.

Figure 5.3. The relationship between Minimum Fetch and Edge Change from 1966-2007 using transformed data.

General characteristics of variables: Truncated Tidal Range



Truncated Tidal Range (Table 5.4) was derived from the GSBM. The regression of Truncated Tidal Range, on the change in the marsh edge from 1966-2007 (Table 5.5) was highly significant with an R^2 of 0.048. Although Truncated Tidal Range values above 0.3 m may be possible outliers, their leverage and Cooks distances remained low (Fig. 5.4). When the values of Truncated Tidal Range above 0.3 were removed, the results were similar.

Table 5.4. The Truncated Tidal Range From GSB model.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
0	0.08	0.14	0.14	0.19	0.57	0.09	0.01	258

Table 5.5. The regression table of peak Truncated Tidal Range on LP transformed marsh edge change over the period 1966-2007

		Std. Error	t value	Pr(>|t|)
Intercept	-1.97	0.25	-7.91	<< 0.001
Slope	-5.25	1.46	-3.6	< 0.001

Multiple R-squared: 0.048, Adjusted R-squared: 0.044

Figure 5.4. The diagnostics from the regression of Truncated Tidal Range on LP transformed edge change in non-channel areas between 1966-2007.

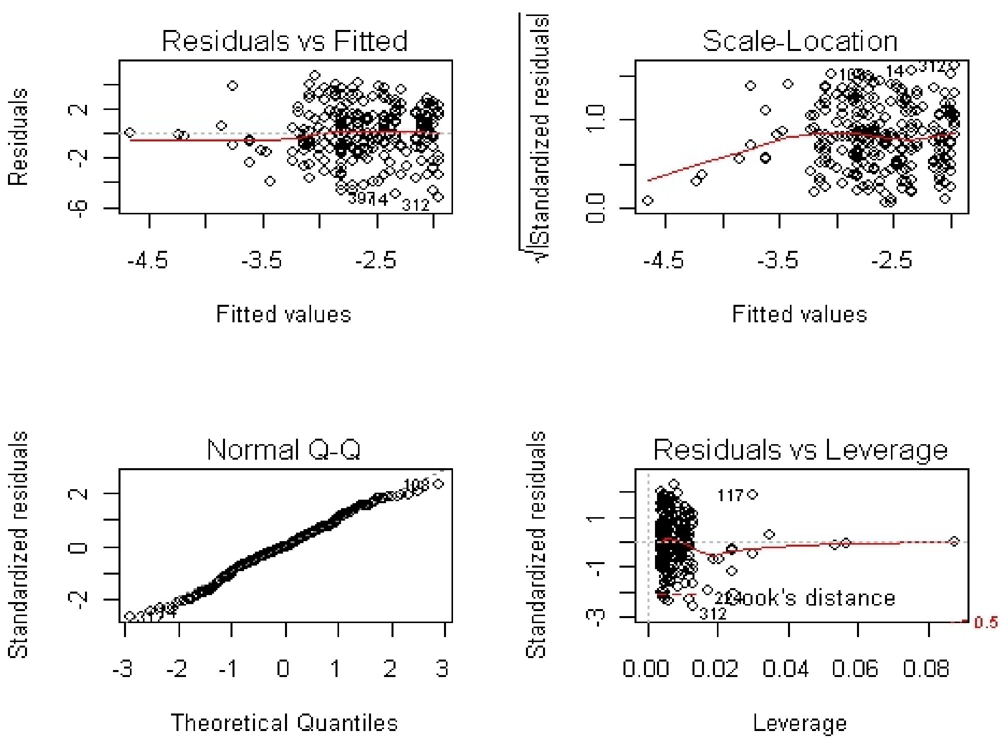
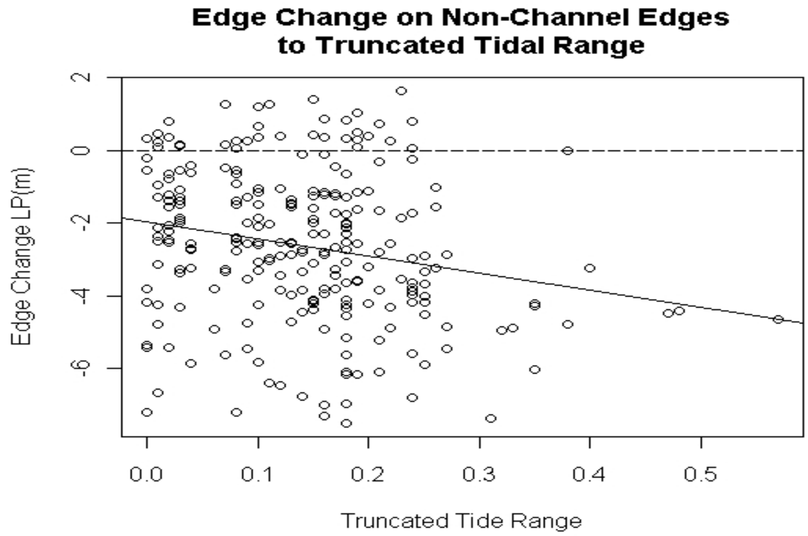


Figure 5.5. Scatter plot of the relationship between Truncated Tidal Range and marsh edge changes from 1966 to 2007.



General characteristics of variables: Salinity

The relationship between marsh loss from 1966-2007 and Salinity (Table 5.6) was also tested with a linear regression model. Several null values were found for Salinity, reflecting points outside the boundaries of the water sampling extents (Fig. A.1), which reduced the number of points used in this analysis to 215 (Table 5.6). A series of diagnostic plots did not show problems with normality or outliers (Fig. 5.6). The regression (Table 5.7) was not significant (Fig. 5.7).

Table 5.6. Salinity from TOH water quality testing data.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
28.8	30.1	30.4	30.2	30.5	30.7	0.48	0.03	215

Table 5.7. The regression table of Salinity on LP transformed marsh edge change over the period 1966-2007.

		Std. Error	t value	Pr(>|t|)
Intercept	-12.11	9.64	-1.26	0.21
Slope	0.31	0.32	0.98	0.33

Multiple R-squared: 0.004, Adjusted R-squared: <<0.001

Figure 5.6. The diagnostics plots for Salinity regressed against the transformed marsh edge change between 1966 and 2007.

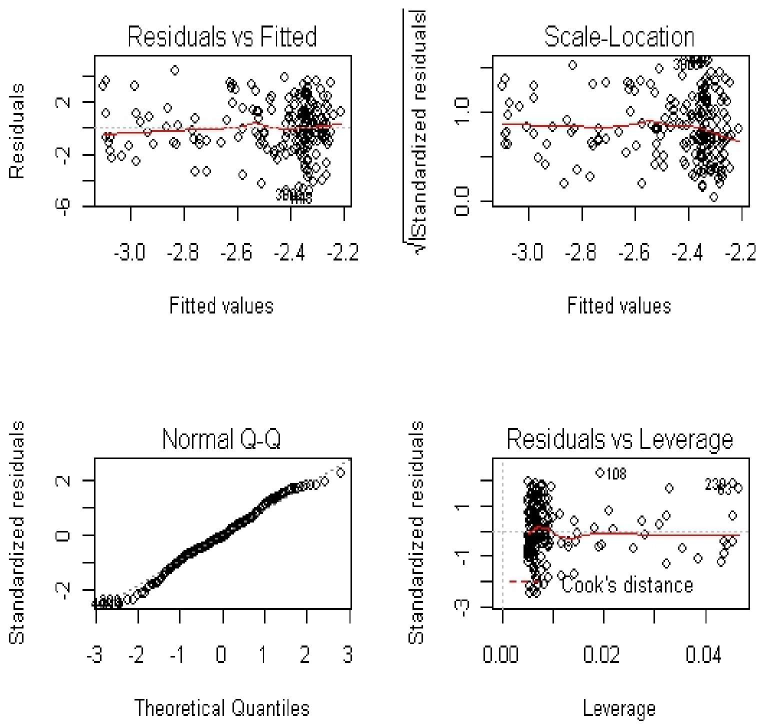
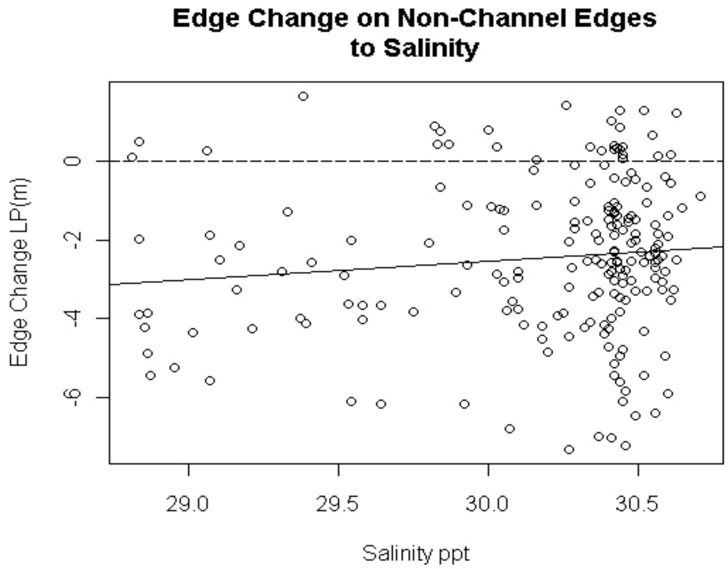


Figure 5.7. A scatter plot of Salinity regressed against the LP transformed marsh edge change between 1966 and 2007. The trend was not significant.



Building a Parsimonious Model For Natural Physical Parameters and Non-Channel Edges.

The three parameters (Minimum Fetch, Truncated Tidal Range and Salinity) were first assessed with a Bayesian model mixing algorithm in the bms package for R. An even prior distribution was used, so that no parameter was favored. The output table indicated that the Salinity term was unlikely to become part of the final model, but the Minimum Fetch term needed to be included (Table 5.11; Fig. 5.13). An AIC-based method from the MMIX package for R was also used with the same data set. The mixAic function in this package also found that Minimum Fetch and Truncated Tidal Range were the important parameters (Table 5.12 and Fig. 5.14).

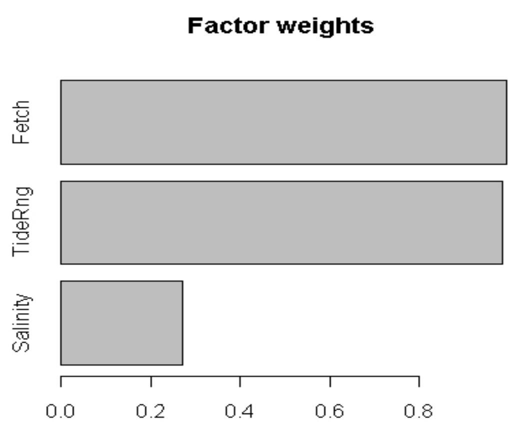
Table 5.8. The results from picking the natural physical parameters that had the greatest correlation with change in marsh edges from 1966 – 2007 for areas not on navigational channels, (PIP = posterior inclusion probability, Cond.Pos Sign = the posterior probability of a positive coefficient expected value conditional on inclusion, Idx = input order of variables).

	PIP	Post Mean	Post SD	Cond.Pos Sign	Idx
Minimum Fetch	0.97	0	0	0	1
Truncated Tidal Range	0.93	-5.2	2.18	0	2
Salinity	0.07	0	0.09	0.1	3

Table 5.9. The AIC model mixing function mixAic indicated that Minimum Fetch and Truncated Tidal Range were most important for inclusion in the best model for hypothesis regarding the impacts of natural physical factors on change in the edges of marsh not on navigational channels from 1966-2007.

	Model selected	AIC of selection
1	Edge Change 1966-2007~ Minimum Fetch+Truncated Tidal Range	937.58
2	Edge Change 1966-2007 ~ Minimum Fetch+Truncated Tidal Range+Salinity	939.58
3	Edge Change 1966-2007 ~ Minimum Fetch	946.26

Figure 5.8. Factor loadings from AIC model mixing (MMIX package) with Minimum Fetch as the most important factor and Truncated Tidal Range almost as important.



Stepwise regression.

In this simple case, both forward stepping from the best model in Table 5.9 and forward/backward stepping from a model that included Minimum Fetch, Truncated Tidal Range, and Salinity, converged on the same solution, which included interaction terms (Table 5.10). The results were checked for spatial autocorrelation with a bubble plot (Fig. 5.9) and a variogram (Fig. 5.10). Spatial autocorrelation did not appear to be present.

Table 5.10. The regression table of the effects of Minimum Fetch and Truncated Tidal Range on LP transformed change in the edge of marshes from 1966-2007.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
Intercept		.-0.43	0.72	-0.82	0.41
Minimum Fetch (BoxCox)	-0.36		0.15	-3.88	<< 0.001
Truncated Tidal Range		5.9	4.69	1.13	0.26
Minimum Fetch:Truncated Tidal Range	-2.39		0.9	0.16	0.87

Multiple R-squared: 0.179, Adjusted R-squared: 0.170

Figure 5.9. A bubble plot of residuals from a fit of Minimum Fetch and Truncated Tidal Range to non-channel marsh edge change between 1966 and 2007 indicates a grouping of negative residuals in the south-west corner of the map, but no other noticeable pattern.

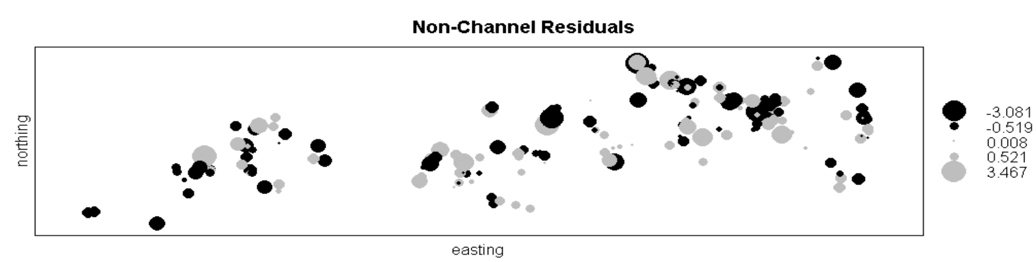
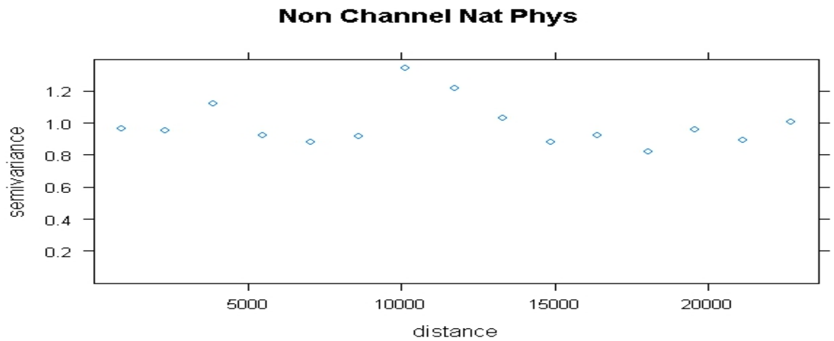
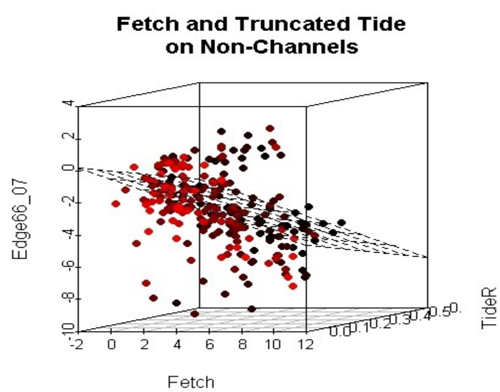


Figure 5.10. A variogram of the model fitting Minimum Fetch and Truncated Tidal Range to marsh edge loss between 1966 and 2007 for non-navigational channel areas showed no pattern of spatial autocorrelation.



The relimpo functions in R estimated that this model accounted for 17.9% of the variance in the edge change data. It estimated that 14.4% of the variance was explained by Minimum Fetch, 3.5% by Truncated Tidal Range and <<0.001% from the interaction between Minimum Fetch and Truncated Tidal Range. Figure 5.11 represents the relationship in a 3 dimensional plot.

Figure 5.11. A 3D plot of Minimum Fetch and Truncated Tidal Range with the Z value representing the marsh edge change between 1966 and 2007 for natural edges that were not located along a navigational channel.



Conclusion:

Minimum Fetch and Truncated Tidal Range both had an effect on marsh loss from 1966-2007, and interact weakly. For natural marsh edges that are not on navigational channels, the most stable marsh edges were those with the least fetch and where water covers the neighboring flats for the least amount of time (largest truncated tidal range). These results suggest that, for marshes not located on navigational channels, some aspect of general water motion is important for determining the amount of marsh lost for areas where the surrounding water is not shallow.

NAVIGATIONAL CHANNELS AND NATURAL PHYSICAL FACTORS

I then considered natural marsh edges that faced navigational channels, and tested linear models that included the correlations between natural physical variables (Minimum Fetch, Peak Channel Flow Rate, and Salinity) to determine which combination of variables best explained the changes in the edges of the marsh from 1966 – 2007.

General characteristics of variables: Peak Channel Flow Rate

The variable Peak Channel Flow Rate measured the peak water flow rate through the tidal cycle in the nearest channel to the sampling point (Table 5.14). Peak Channel Flow Rates of 0 were considered errors and those points were excluded from analysis.

Table 5.11. The Peak Channel Flow Rate (m s^{-1}) From GSB model for non-channelized navigation channels.

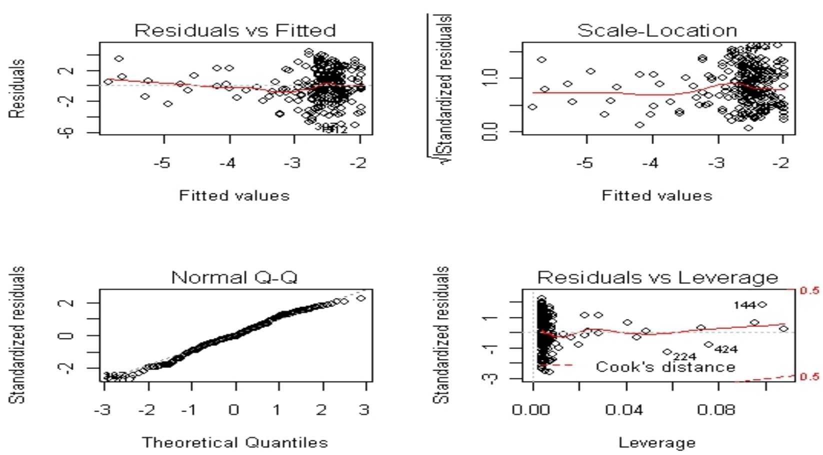
Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
0.01	0.13	0.03	0.33	0.56	1	0.24	0.02	148

Table 5.12. The relationship between the untransformed Peak Channel Flow Rate and the LP transformed response variable.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-2.07	0.21	-9.97	<< 0.001
-4.92		1.12	-4.38	<< 0.001

Multiple R-squared: 0.072, Adjusted R-squared: 0.068

Figure 5.12. The diagnostics plots for Peak Channel Flow Rate regressed against the LP transformed response variable.



The plot of residuals against fitted values showed heteroscadicity in the data. The Peak Channel Flow Rate was log transformed and the relationship with edge loss along channels was recalculated (Table 5.13). Diagnostics plots show reasonable values with a shorter Cook's distance (Fig. 5.13).

Table 5.13. The regression table of the LP transformed response variable, edge change over the period 1966-2007, as a function of log transformed Peak Channel Flow Rate.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-4.1	0.27	-15.09	<< 0.001
-1.75		0.66	-2.65	0.01

Multiple R-squared: 0.046, Adjusted R-squared: 0.039

Figure 5.13. Diagnostics plots of the transformed response variable 1966-2007 for points located on channel edges regressed against the log transformed Peak Channel Flow Rate.

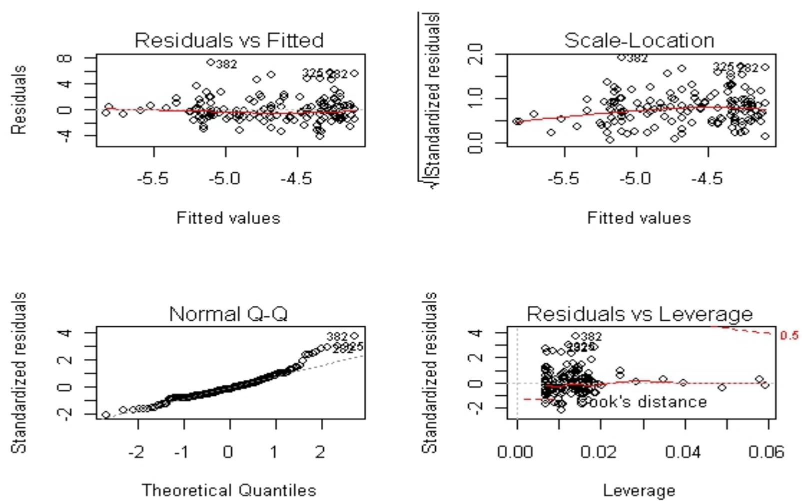
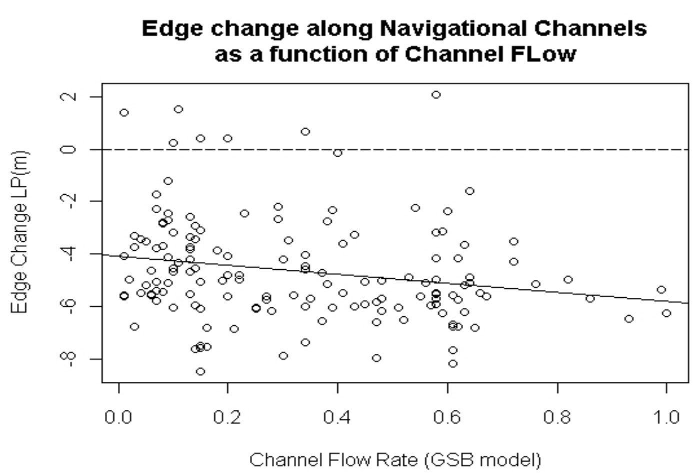


Figure 5.14. A scatter plot of the relationship between log Peak Channel Flow Rate and marsh loss along the channels edges from 1966 - 2007.



Stepwise regression.

The three parameters (Minimum Fetch, Peak Channel Flow Rate and Salinity) were first assessed with a Bayesian model mixing algorithm in the bms package for R. An even prior distribution was used, so that no parameter was favored. Because Salinity values outside of the water quality sampling area (Fig. A.1) contained null values, only 126 points were used in the model building process. The output table indicated that Salinity was unlikely to become part of the final model but Minimum Fetch needed to be included (Table 5.14). An AIC-based method from the MMIX package for R was also used for the same data set. The mixAic function in this package also found that Peak Channel Flow was important and Minimum Fetch was a less important parameter (Table 5.15 and Fig. 5.15). In both cases the Peak Channel Flow Rate was the most important term, followed by Minimum Fetch and Salinity, which made relatively little contribution as explanatory variables.

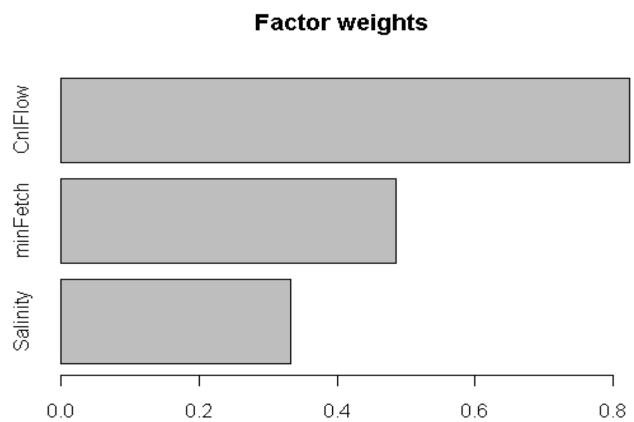
Table 5.14. Using 126 sampling points, the Bayesian model averaging package BMS in R was used with an even prior to estimate a reduction in the natural physical parameters variable set as a step in developing an optimal model for assessing factors affecting changes in navigational channel marsh edges (PIP = posterior inclusion probability, Cond.Pos Sign = the posterior probability of a positive coefficient expected value conditional on inclusion, Idx = input order of variables). Only Peak Channel Flow Rate was important.

	PIP	Post Mean	Post SD	Cond.Pos Sign	Idx
Peak Channel Flow Rate	0.54	-0.93	1.02	0	3
Minimum Fetch (B-C)	0.2	-0.03	0.08	0	1
Salinity	0.1	0.03	0.14	1	2

Table 5.15. The top three models from the AIC model mixing function mixAic are listed. Minimum Fetch and Peak Channel Flow Rate were most important for inclusion in the best mode for edge changes for 1966-2007 points along navigational channels.

	Model selected	AIC of selection
1	Edge Change 1966-2007~ Peak Channel Flow Rate	537.75
2	Edge Change 1966-2007~ Minimum Fetch + Peak Channel Flow Rate	538.07
3	Edge Change 1966-2007~ Salinity+ Peak Channel Flow Rate	539.18

Figure 5.15. Loadings from the AIC model mixing function mixAic are listed. Peak Channel Flow Rate and Minimum Fetch were most important for inclusion in the best model for marsh edge changes 1966-2007 for points along navigational channels.



The small amount of spatial autocorrelation seen in a variogram for a fit of Peak Channel Flow and Fetch (Fig. 5.15) was removed by adding the channel categories (Fig. 5.16). Adding channel classifications to the model explained the spatial autocorrelation and provided a better fit for the other variables. Subsequent visual inspection of a bubble plot of residuals did not show a spatial pattern (Fig. 5.17).

Figure 5.16. A variogram of residuals from the model of marsh edge loss from 1966 to 2007 for all navigational channel edges together. There appears to be some spatial autocorrelation due to unexplained variance at less than 500 m.

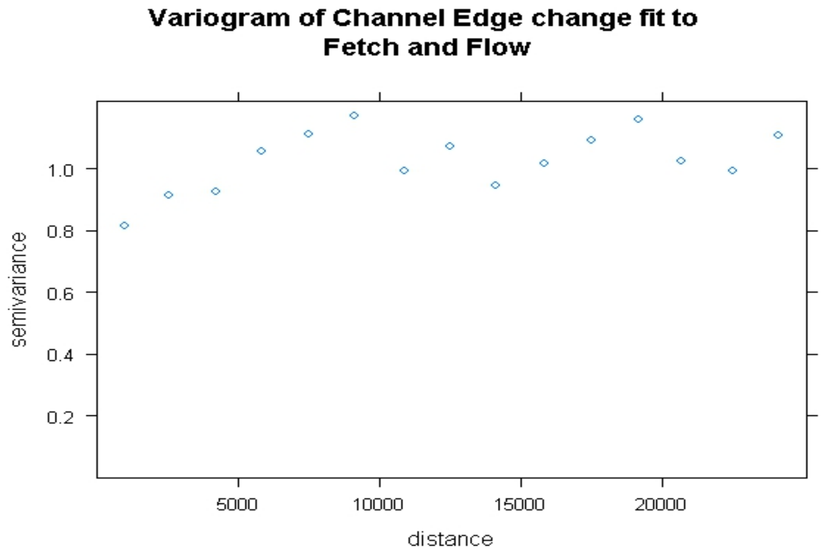


Figure 5.17. A variogram showing how the addition of channel boat use category to the model explained enough of the variance to remove the spatial autocorrelation seen in Fig. 5.16.

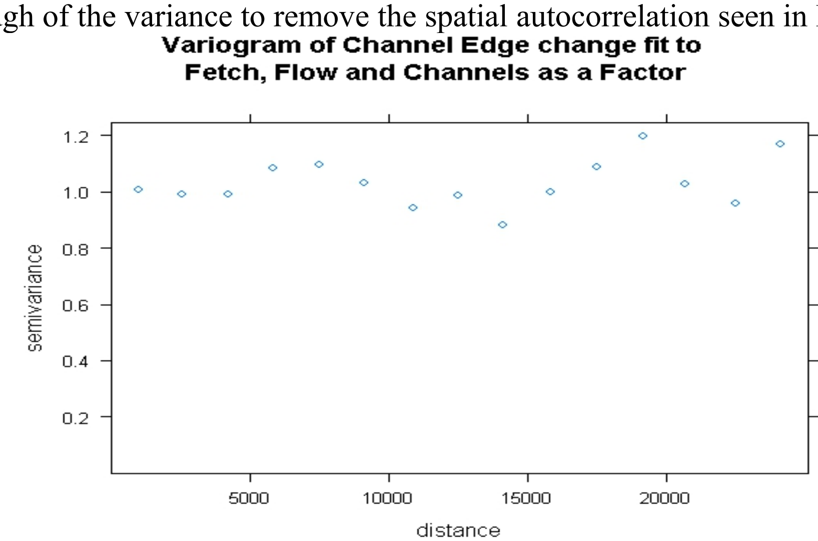
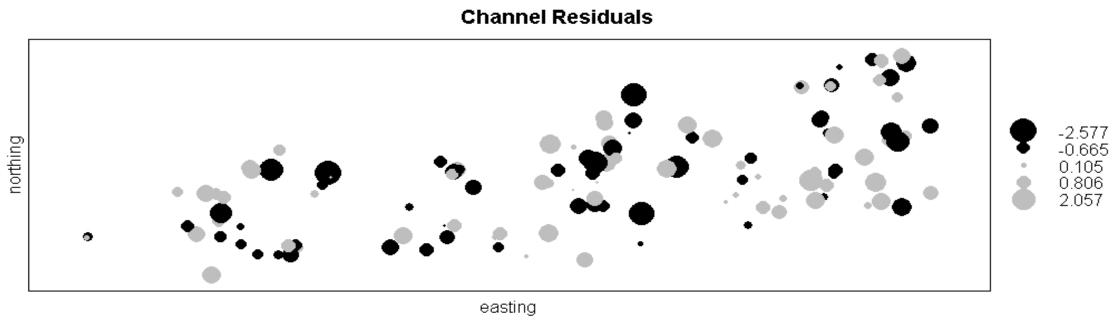


Figure 5.18. A bubble plot of residuals, which was used to check for spatial autocorrelation for fits of the effects of the variables Peak Channel Flow Rate, Minimum Fetch and Truncated Tidal Range on marsh edge loss from 1966-2007 along the navigational channels.



To determine the best-fit model, both forward (seeing the effect of adding successive variables) and backward (seeing the effect removal of variables) stepping was used. Only Peak Channel Flow Rate and Channel (the boat use categories) were retained in the final model (Table 5.16). The relative importance of the each variable in the final model was evaluated with the relaimpo package in R (Table 5.17).

Table 5.16. The regression table of the effect of the variables Peak Channel Flow Rate and Channel (the boat use classification) on the LP transformed response variable, change in the marsh edge along navigational channels from 1966-2007. This regression resulted from a forward and back stepped process that eliminated Minimum Fetch and Salinity as significant variables.

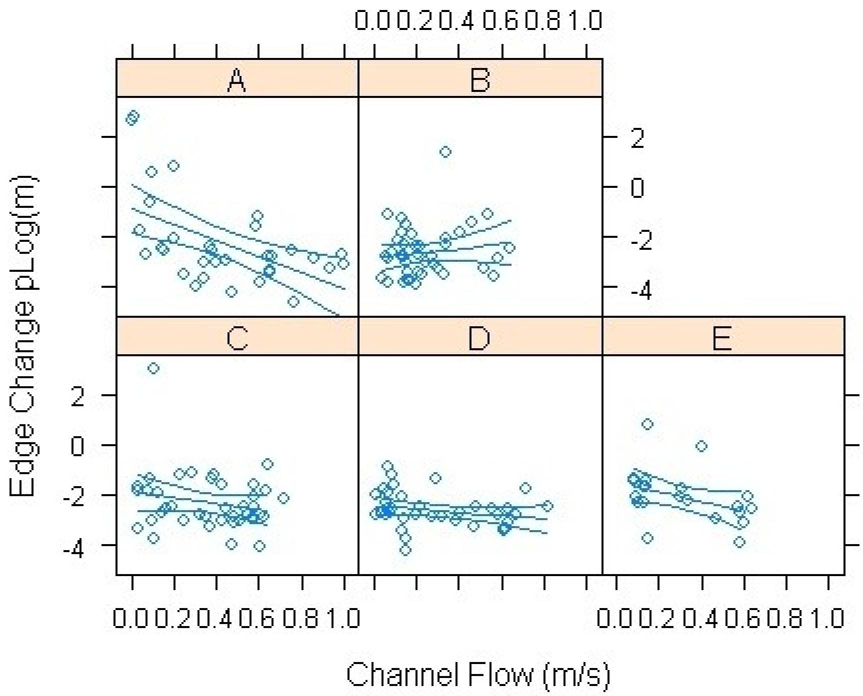
Peak Channel Flow Rate and:	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-3.36	0.488	-6.898	<< 0.001
Channel A	-2.150		0.688	-3.117	0
Channel B	-0.920		0.538	1.711	0.089
Channel C	-0.433		0.489	0.885	0.378
Channel D	-1.202		0.499	-2.407	0.02
Channel E	0.059		0.590	0.100	0.921

Multiple R-squared: 0.105, Adjusted R-squared: 0.073

Table 5.17. The relative importance of the variables in the chosen best model (Table 5.16) as determined by calc.relimp from the R package relaimpo by partitioning the variance explained by model (total variance explained 10.49%).

Variable	Percent of variance
Peak Channel Flow Rate	5.35%
Channel Class	5.13%

Figure 5.19 The relationship between Peak Channel Flow Rate and Marsh Edge Change along navigational channels by channel type. The significant relationship for boat use classifications A and D (Table 5.16) is similar to that found with Distance to Channel (Table 4.13). This suggests that the effect of water flow rate on the changes seen in the location of the edge of the marsh may confound interpretation of the effects of boat use.



Conclusion:

Of the variables tested in this section (Minimum Fetch, Peak Channel Flow Rate, and Salinity), Peak Tidal Flow Rate was the most significant in navigational channels. The water flow in the channels seemed to have the most important impact, but its importance differed among boat use categories and could confound interpretation of boat use. It is likely that that there is a relationship between the location of navigational channels with natural edges, high tidal flow rates, and high marsh erosion rates, even in the absence of powerboats. Many navigational channels are located in naturally deep water that is kept open by sediment transport driven by high tidal flow rates (Fig. A.5). There is also evidence that some of these locations are likely meander streams (Fig. A.23; Fig. A.24).

NON-CHANNELS AND NUTRIENT LOADING

The subset of marsh edge points that contained nutrient values and were not on navigational channels were used to test the effects of nutrients on marsh loss. The variables related to nutrient loading (Nitrate = Nitrat2, Distance to Treated Sewerage Outfall = ToOutfalMr) were regressed on the change in natural marsh edges from 1966 - 2007.

General characteristics of variables: Nitrate

The simple statistics for Nitrate at points along natural edges that were not on navigational channels are shown in Table 5.18. A regression model that included the relationship between Nitrate and the response variable (edge change along natural edges that were not on navigational channels from 1966-2007) showed a normal distribution of residuals (Fig. 5.20), and the regression was not significant (Table 5.19, Fig. 5.21)

Table 5.18. The general statistics for Nitrate (μg N/l) at measurement points that were natural and not on navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	n
11.56	12.61	14.02	15.16	18.25	22.21	3.04	213

Table 5.19. The regression table of the LP transformed response variable, edge change from 1966-2007, as a function of Nitrate.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-0.73	0.43	-1.68	0.095
-0.03		0.03	-0.95	0.341

Multiple R-squared: 0.004, Adjusted R-squared: -0.0

Figure 5.20. A diagnostics plot the transformed response variable 1966-2007 for points on natural edges that were not located on navigational channels regressed against Nitrate.

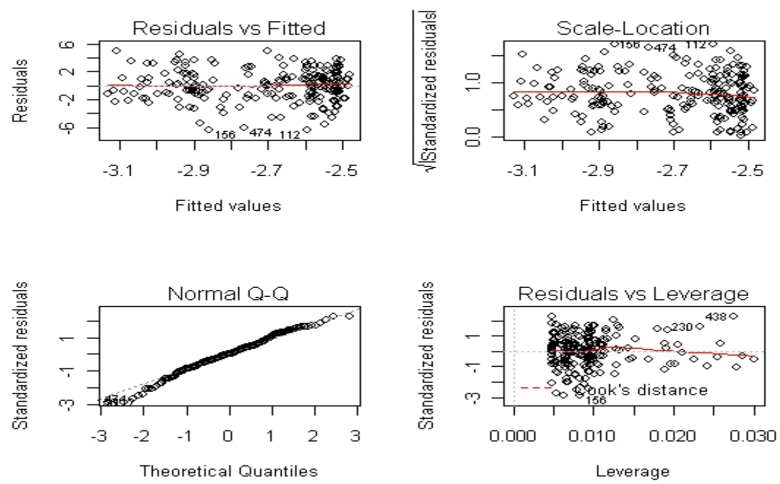
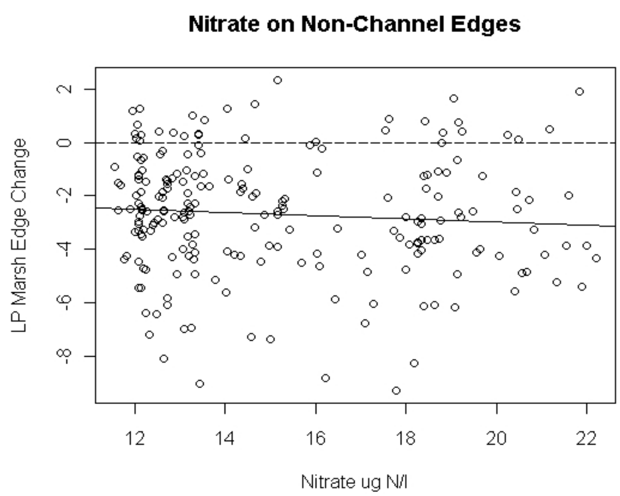


Figure 5.21. A scatter plot of marsh loss from 1966 - 2007 at points located on natural edges that were not along navigational channels edges as a function of Nitrate.



General characteristics of variables: Distance to Treated Sewerage Outfall.

The simple statistics for Distance to Treated Sewerage Outfall are shown in Table 5.20. A regression of nitrates showed the relationship of this variable to marsh edge change along natural edges that were not on navigational channels (Table 5.21; Fig. 5.23). The diagnostics showed a normal distribution of residuals (Figure 5.22), but the regression was not significant (Table 5.21)

Table 5.20. The simple statistics for the Distance to Treated Sewerage Outfall for points along natural edges that were not located along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	n
390.1	2008	3179	3459	4759	7176	1756.87	213

Table 5.21. The regression table of the LP transformed response variable, edge change from 1966-2007, as a function of Distance to Treated Sewerage Outfall.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-2.8	0.34	-8.25	<< 0.001
0		0	0.34	0.73

Multiple R-squared: 0.001, Adjusted R-squared: -0.004

Figure 5.22. Diagnostics for Distance to Treated Sewerage Outfall against marsh edge change 1966 - 2007 at points not located along navigational channels.

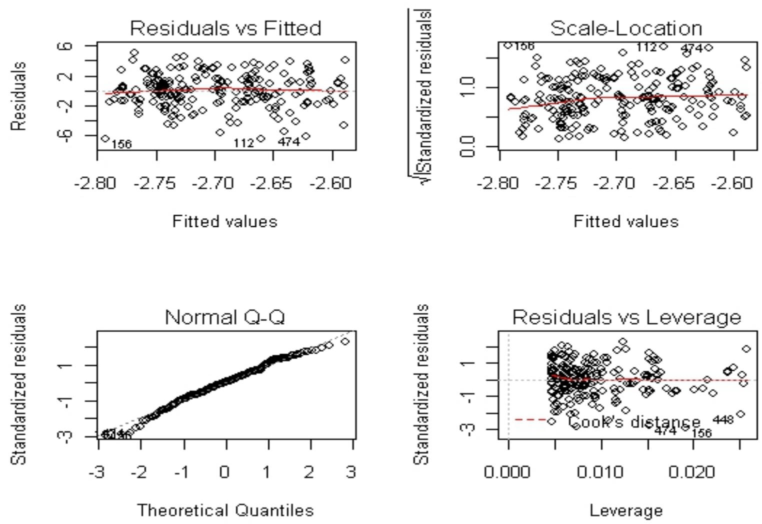
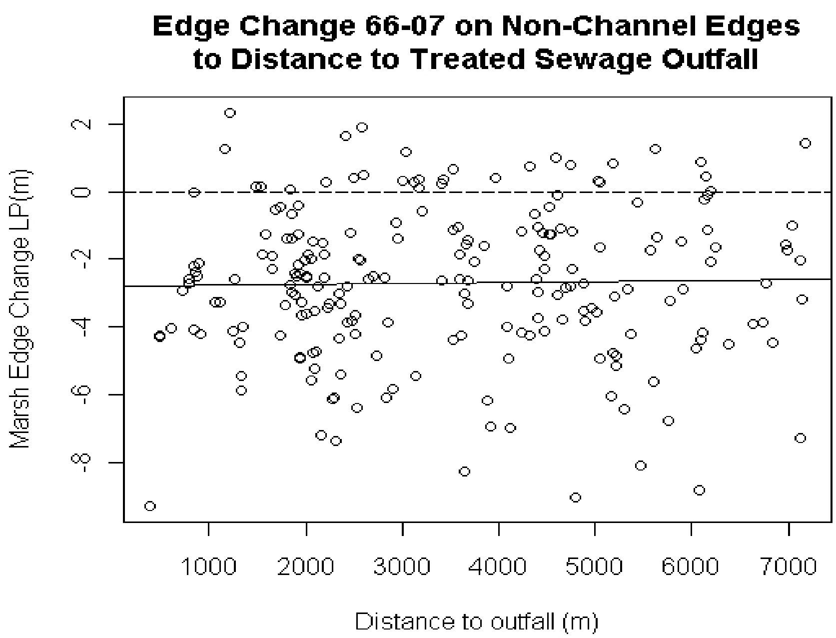


Figure 5.23. A scatter plot of marsh loss from 1966 - 2007 at points located on natural edges that were not along navigational channels edges as a function of Distance to Treated Sewerage Outfall.



Best Model For the effect of nutrient loading on marsh edges not facing navigational channels

Both variables (Nitrates and Distance to Treated Sewerage Outfall) were included in this analysis. The relationships between Nitrate and Distance to Treated Sewerage Outfall and marsh edge change along natural edges not along navigational channels from 1966-2007 were not found to be significant (Table 5.22; Fig. 5.24). In order to further confirm the small amount of edge change variation that was explained by nutrient loading, bootstrapped estimates the R^2 value were determined for both Nitrate and Distance to Treated Sewerage Outfall, and both were essentially 0 (Fig. 5.24; Fig. 5.25).

Table 5.22. The regression table showing the relationship of Nitrate and the shortest Distance to Treated Sewerage Outfall to the change in the marsh edge from 1966-2007 for natural edges that do not face navigational channels.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)	VIF
		0.2	0.34	-8.25	<<0.001	
Distance to Treated Sewerage Outfall	0		0	0.34	0.73	52.35
Nitrate		0	0.13	-1.52	0.13	6.51
Distance to Treated Sewerage Outfall : Nitrate	0		0	1.15	0.25	55.92

Multiple R-squared: 0.01, Adjusted R-squared: -0.00

Figure 5.24. A bootstrap distribution of R^2 for edge measurement points not on navigational channels, regressing Nitrate against the change in the marsh edge from 1966-2007.

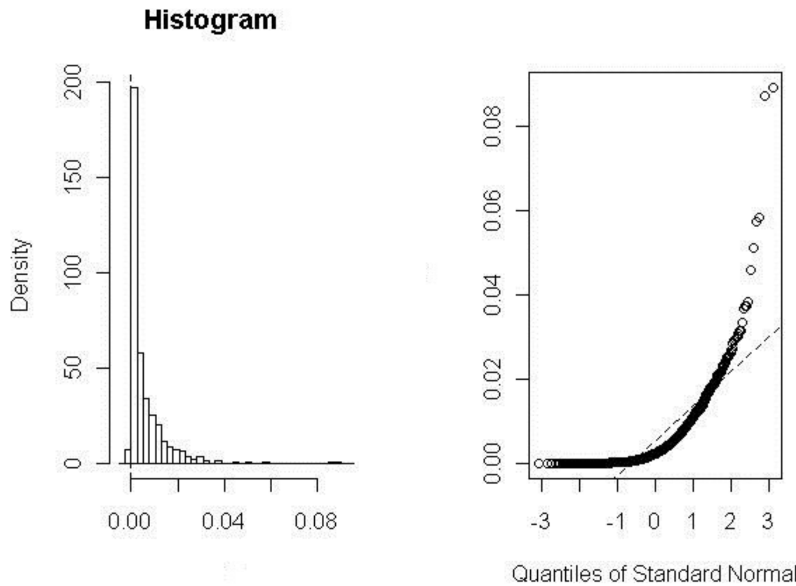
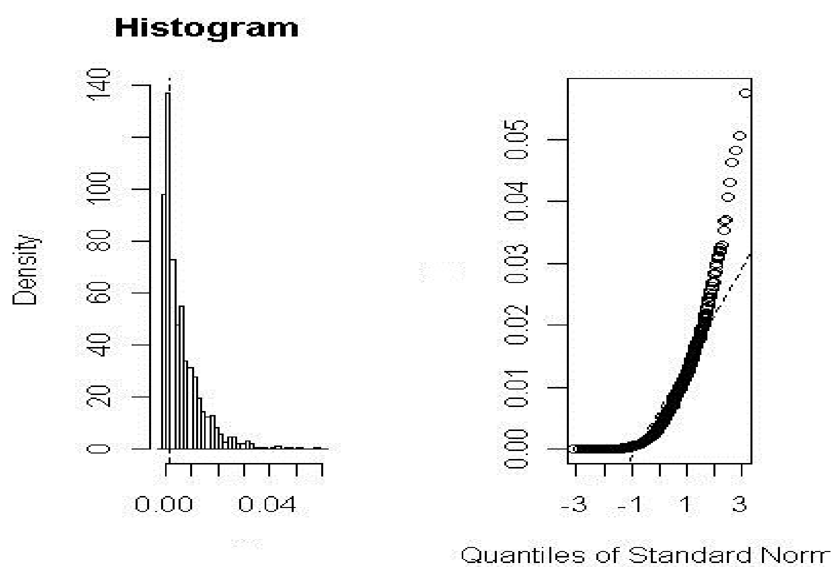


Figure 5.25. A bootstrap distribution of R^2 for edge measurement points not on navigational channels, regressing Distance to Treated Sewer Outfall against the change in the marsh edge from 1966-2007.



Conclusion:

Nutrient loading, as determined by the concentration of nitrates and proximity to treated sewerage outfalls was not correlated with the change in natural marsh edges from 1966-2007 for points along edges that were not on navigational channels.

NAVIGATIONAL CHANNELS AND NUTRIENT LOADING

I then considered only points from edges that were on navigational channels and also contained nutrient values. The variables related to nutrient loading (Nitrate = Nitrat2 and Distance to Treated Sewerage Outfall = ToOutfalMr) were regressed on the changes for natural marsh edges from 1966 - 2007 and an estimate of their relative contributions was made.

Regression for nutrient loading on edges along navigational channels

The relationship of Nitrate and Distance to Treated Sewerage Outfall are shown in Table 5.23. The regression was not significant.

Table 5.23. The regression table of Distance to Treated Sewerage Outfall on Edge Change along navigational channels 1966-2007.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-1.87	0.86	-2.18	0.03
Distance to Treated Sewerage Outfall	0.000		0.00	0.27	0.78
Nitrate	-0.006		0.05	-1.18	0.24

Conclusion:

Nitrates and Distance to Treated Sewerage Outfall did not influence on the change of location for edges along navigational channels from 1966-2007.

Edge change points that were not on navigational channels were used to test the effects of sediment variables (Particulate Organics = Partico, Distance to Borrow Pit = PitPitDistM, Turbidity = Turv2mIDW3) to determine which combination of variables best explained the changes in marsh edges from 1966 - 2007.

General characteristics of variables: Distance to Borrow Pit

Distance to Borrow Pit is a measure of the distance from each measurement point to the nearest borrow pit (Table 5.24). A diagnostic plot found the data were normally distributed, and did not require transformation (Fig. 5.26). This variable did not have a significant linear correlation with change in the edge of the marsh from 1966-2007 non-channel edges (Table 5.25; Fig. 5.27).

Table 5.24. The characteristics of the variable Distance to Borrow Pit for points along natural marsh edges that were not along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
198.7	742.5	1044	1146	1405	3227	569.13	40.86	194

Table 5.25. The relationship between the change in the marsh edge from 1966 – 2007 and Distance to Borrow Pit for points along natural marsh edges that were not along navigational channels.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-1.18	0.16	-7.4	<< 0.001
0		0	1.02	0.31

Multiple R-squared: 0.005, Adjusted R-squared: 0.00

Figure 5.26. A diagnostics plot of the transformed response variable regressed against the untransformed variable Distance to Borrow Pit for marsh edges that were not along navigational channels.

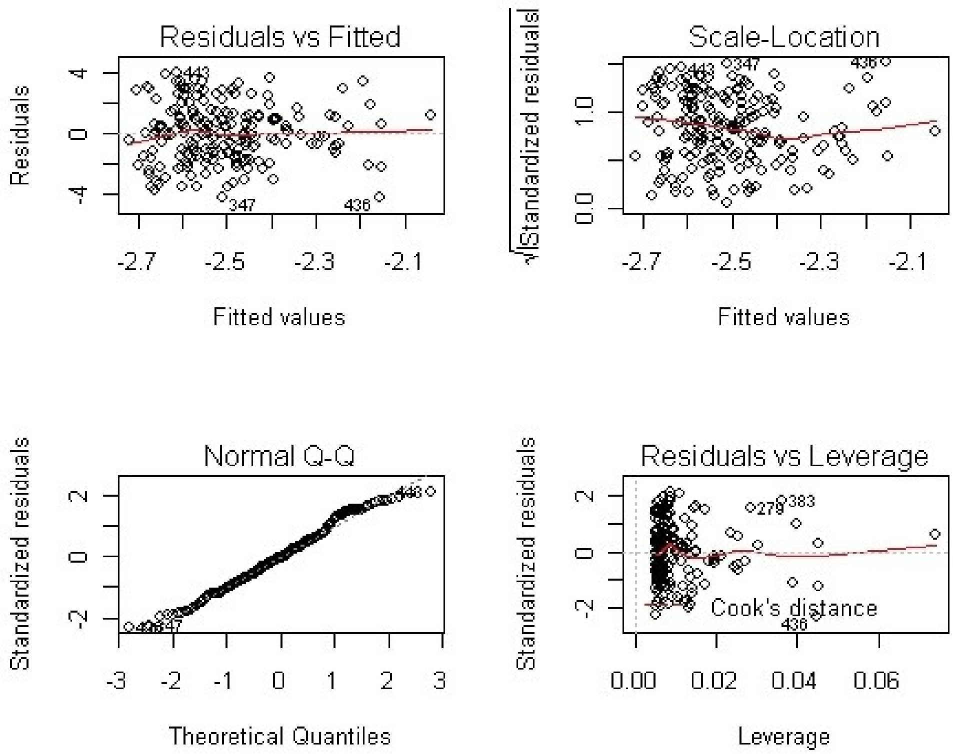
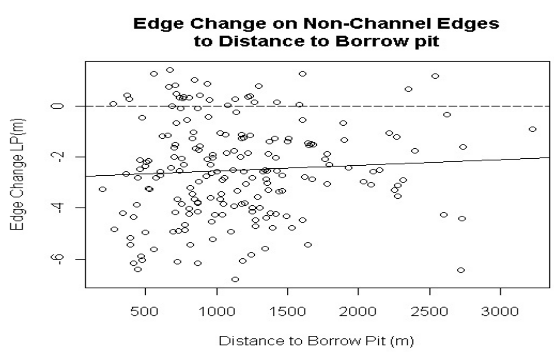


Figure 5.27. A scatter plot of the relationship between Distance to Borrow Pit and loss along the edges of marsh that were not located along channels from 1966 – 2007.



General characteristics of variables: Particulate Organics

The general characteristics of the Particulate Organics data are shown in Table 5.26. A diagnostic plot did not demonstrate problems with normality (Fig. 5.28). This variable did have a small significant linear relationship with change in the marsh edge from 1966 - 2007 for marsh not along navigational channels (Table 5.27; Fig. 5.29).

Table 5.26. The simple statistics for Particulate Organics for points along natural marsh edges that were not located along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
989.6	1036	1068	1101	1127	1459	97.53	7.00	194

Table 5.27. The relationship of Particulate Organics to change in the edge of the marsh from 1966 to 2007 at points along edges that were not located on navigational channels.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	1.101	0.790	1.393	0.17
-0.002		0.001	-2.710	0.01

Multiple R-squared: 0.037, Adjusted R-squared: 0.032

Figure 5.28. A diagnostics plot of the transformed response variable regressed against the untransformed variable Particulate Organics against the change in the position of the edge of the marsh from 1966-2007 for points along natural edges that were not along navigational-channels.

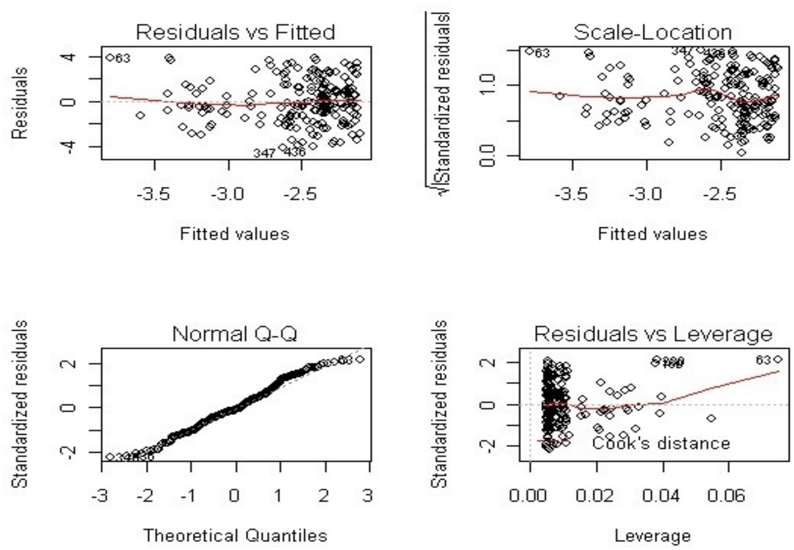
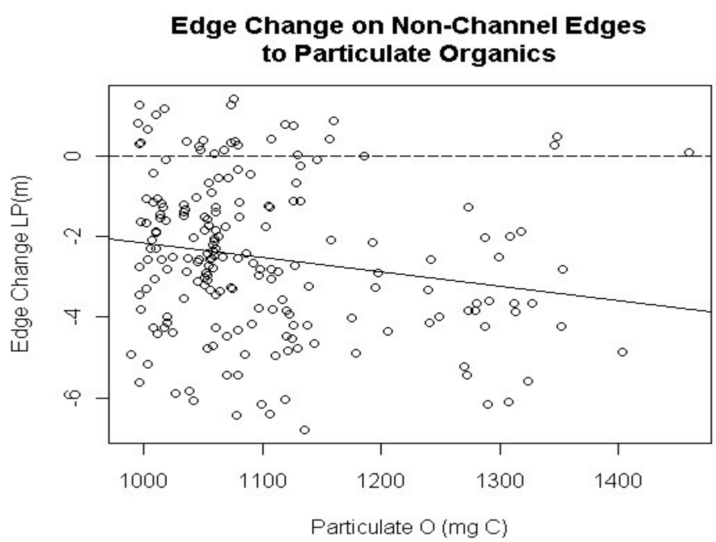


Figure 5.29. A scatter plot of the relationship between Particulate Organics and marsh loss from 1966 – 2007 for areas not located along navigational channels.



General characteristics of variables: Turbidity

The general characteristics of the Turbidity data are shown in Table 5.28. A diagnostic plot did not demonstrate problems with normality and did not need to be transformed (Fig. 5.30). This variable did not have a significant linear relationship with change in the edge of the marsh from 1966 - 2007 for areas not on navigational channels (Table 5.29; Fig. 5.31).

Table 5.28. The simple statistics for Turbidity for points along natural edges that were not located along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
11.77	13.92	14.54	14.97	16.02	23.75	1.86	0.13	194

Table 5.29. The relationship of Turbidity to change in the edge of the marsh from 1966 - 2007 at points that were not located on navigational channels.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-0.99	0.58	-1.72	0.09
-0.003		0.04	-0.07	0.94

Multiple R-squared: 0, Adjusted R-squared: 0

Fig. 5.30. A diagnostics plot the transformed change in edge position from 1966 -2007 regressed against the untransformed variable Turbidity for points that did not occur along navigational channels.

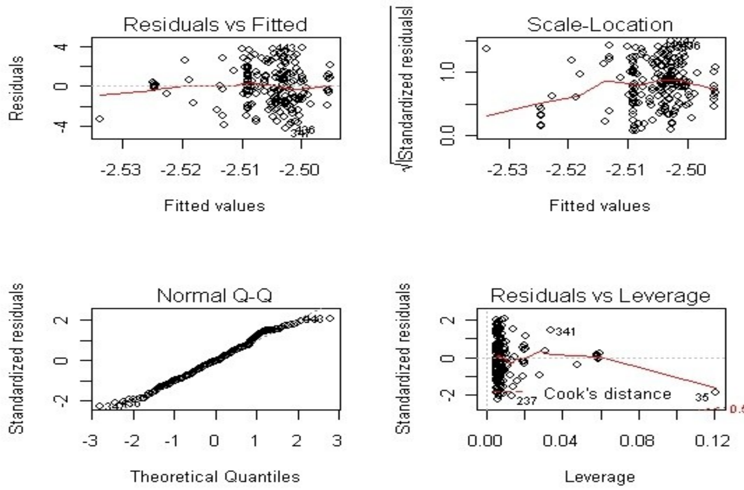
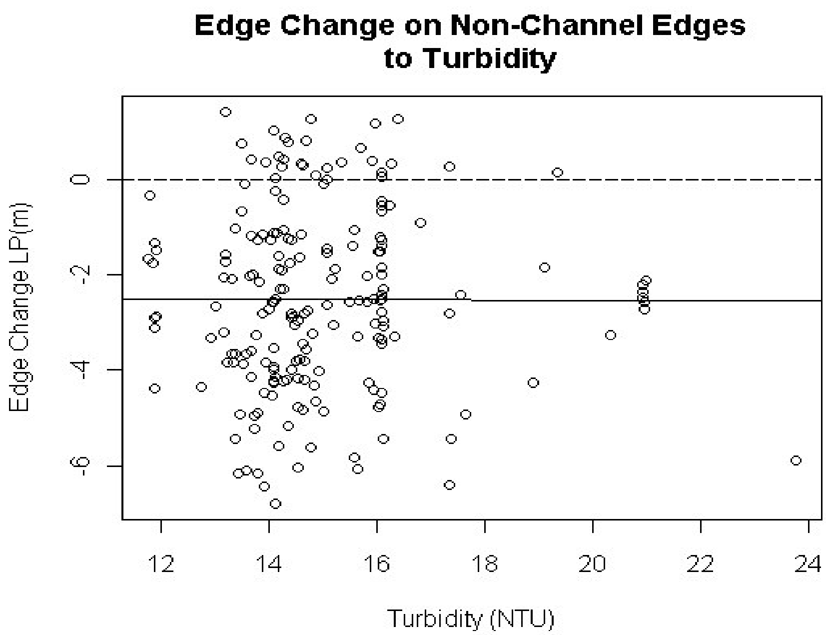


Figure 5.31. A scatter plot of the transformed change in edge position from 1966 -2007 regressed against the untransformed variable Turbidity for points that did not occur along navigational channels.



Stepwise regression.

The three parameters (Distance to Borrow Pits, Particulate Organics and Turbidity) were first assessed with a Bayesian model-mixing algorithm in the bms package for R. An even prior distribution was used, so that no parameter was favored. Because Particulate Organics values outside of the water quality sampling area contained null values, only 206 points were used in the model building process. Although none of the terms had a lot of explanatory power, t the term Distance to Borrow Pit was unlikely to become part of the final model (Table 5.30). An AIC based method from the MMIX package for R was also used for the same data set. The mixAic function in this package also found that Particulate Organics and Turbidity were the more important parameters, but both showed factor loadings of less than 0.5 and a null model including none of these variables was determined to be the best model (Table 5.31 and Fig. 5.32).

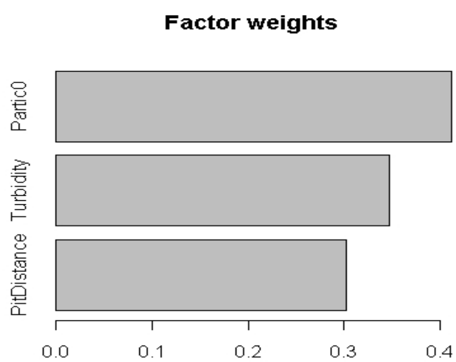
Table 5.30. Using 206 points, the Bayesian model averaging package BMS in R was used with an even prior to estimate a reduction in the sediment parameters variable set as a step in developing an optimal model for marsh edges that were not along navigational channels, however no distinct choice was obvious (PIP = posterior inclusion probability, Cond.Pos Sign = the posterior probability of a positive coefficient expected value conditional on inclusion, Idx = input order of variables).

	PIP	Post Mean	Post SD	Cond.Pos Sign	Idx
Particulate Organics	0.13	-2.44	8.59	0	1
Turbidity	0.10	7.29	3.43	1	2
Distance to Borrow Pit	0.08	1.29	8.89	1	3

Table 5.31. The top three models from the AIC model mixing function mixAic are listed. A null model was a better model than any of these variables. Particulate Organics and Turbidity were possible candidates for inclusion in the best model for points at the edges of Navigational Channels for edge changes 1966-2007.

	Model selected	AIC of selection
1	Edge Change 1966-2007 ~ 1	919.80
2	Edge Change 1966-2007 ~ Particulate Organics	920.26
3	Edge Change 1966-2007 ~ Turbidity	920.91

Figure 5.32. Loadings from the AIC model mixing function mixAic are listed. Particulate Organics and Turbidity were more important than Distance to Borrow Pit for inclusion in the best model as factors influencing edge changes 1966-2007 for points at the edges of channels not located along navigational channels.



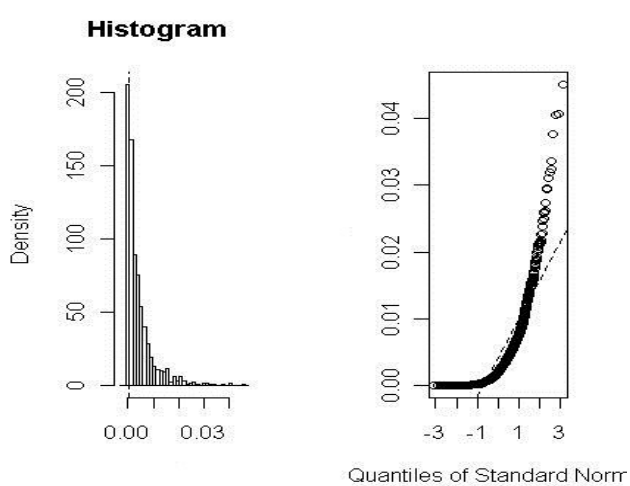
The relationship of marsh changes from 1966-2007 along edges that did not border navigational channels and Particulate Organics, Turbidity and Distance to Borrow Pit are shown in Table 5.32. The regression was not significant and the hypothesis that sediment indicators would predict non-channel edge change was not supported. In order to further confirm the small amount of edge change variation that was explained by the Distance to Borrow Pit variable for edges that were not along navigational channels, bootstrapped estimates the R^2 value were determined. The R^2 was essentially equal to 0 (Fig. 5.33)

Table 5.32. The regression table of the effects of Particulate Organics, Turbidity and Distance to Borrow Pit on LP transformed change in the edge of marshes from 1966-2007 for edges that do not occur along channels.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		19.84	34.15	0.58	0.56
Particulate Organics	-0.02		0.03	-0.61	0.54
Turbidity		18.36	2.42	-0.61	0.54
Distance to Borrow Pit		19.84	0	-1.42	0.16
Particulate Organics : Turbidity	-0.02		0	0.56	0.57
Turbidity: Distance to Borrow Pit	-0.02		0	1.5	0.13

Multiple R-squared: 0.02, Adjusted R-squared: 0.0

Figure 5.33. A bootstrapped R^2 for regressions of marsh edge change from 1966 – 2007 as a function of Distance to Borrow Pit confirms the lack of interaction during this time interval for points along edges that are not along navigational channels.



Conclusion:

The distance to a likely sediment sink (Distance to Borrow Pit) did not have an influence on marsh loss from 1966-2007 for points not on navigational channels. Turbidity also lacked significant influence on marsh loss from1966-2007 for edges that were not along navigational channels. Locations with high Particulate Organics, which would indicate a source of sediment for marsh building, seemed to lose a significant amount of marsh, rather than the hypothesized positive influence. However, the relationship between Particulate Organics and marsh edge change 1966-2007 accounted for less than 4% of the variation along edges not located along channels.

NAVIGATIONAL CHANNELS AND SEDIMENT AVAILABILITY

Natural edges that faced navigational channels were then examined, and linear models that included natural physical variables (Distance to Borrow Pit = PitDistMr, Particulate Organics = Partico, Turbidity = Turv2mIDW3 and Navigational Channel Boat Use Classification = Channel) were tested to determine which combination of variables best explained the changes in marsh edges from 1966 - 2007, and what are their relative contributions may be. I used points that were located along navigational channels that were also within the water quality measurement zone (Fig. A.1).

General characteristics of variables: Distance to Borrow Pit

The simple statistics for Distance to Borrow Pit are shown in Table 5.33. For a simple regression with the transformed response variable (the change in the marsh edge from 1966-2007 for points along navigational channels) and the variable Distance to Borrow Pit, the diagnostic plot looked reasonable (Fig. 5.34) and the regression was significant (Table 5.34; Fig. 5.35) with an R^2 of 0.055.

Table 5.33. Simple statistics for Distance to Borrow Pit for points along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
198.7	742.5	1044	1146	1405	3227	569.13	40.86	194

Table 5.34. Edge change from 1966 - 2007 for points on navigational channels as a function of Borrow Pit Distance.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-7.05	0.49	-14.49	<<0.001
0.001		0	2.75	0.01

Multiple R-squared: 0.055, Adjusted R-squared: 0.051

Figure 5.34. A diagnostics plot for edge loss along navigational channels from 1966 - 2007 regressed against Distance to Borrow Pits.

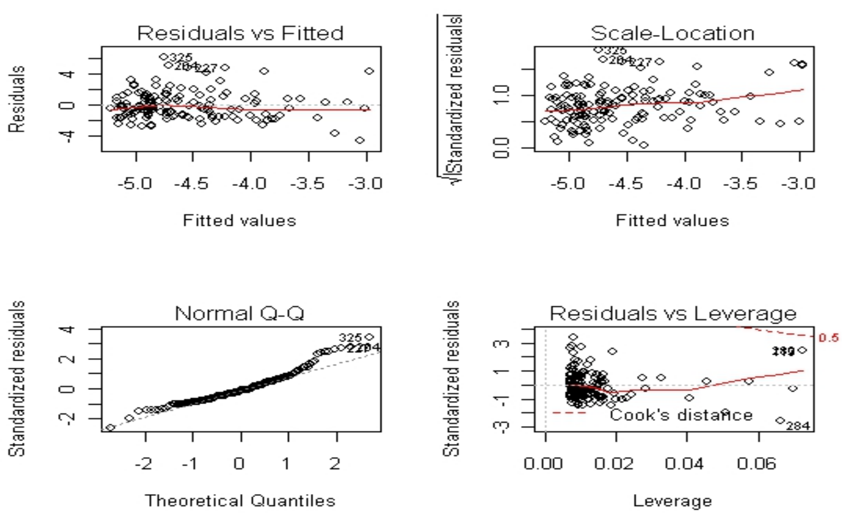
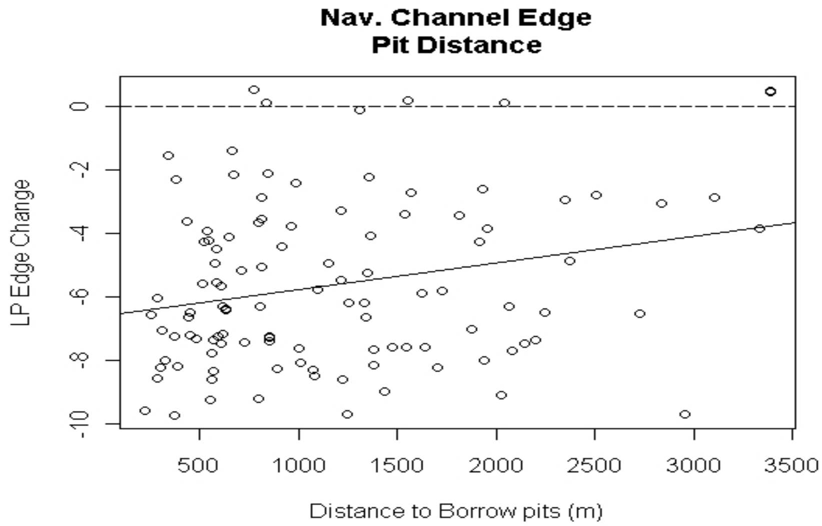


Figure 5.35. The regression of edge loss along navigational channels from 1966-2007 as a function of Distance to Borrow Pit.



General characteristics of variables: Particulate Organics

The simple statistics for Particulate Organics are shown in Table 5.35. The diagnostic plot of Particulate Organics on the change in the marsh edge from 1966-2007 looked reasonable (Fig. 5.34), and the regression was not significant (Table 5.36; Fig. 5.35) and had an R^2 of 0.002.

Table 5.35. Simple statistics for Particulate Organics for points along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
989.6	1036	1068	1101	1127	1459	97.53	7.00	194

Table 5.36. Edge change from 1966 to 2007 for points on navigational channels regressed as a function of Particulate Organics.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-6.84	2.93	-2.34	0.02
0.001		0	0.45	0.66

Multiple R-squared: 0.002, Adjusted R-squared: -0.007

Figure 5.36. A diagnostics plot for edge loss from 1966 to 2007 along navigational channels regressed against Particulate Organics.

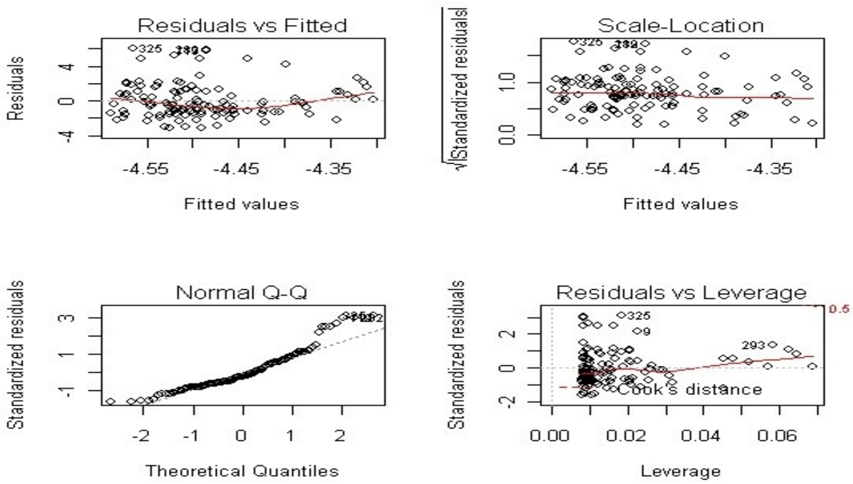
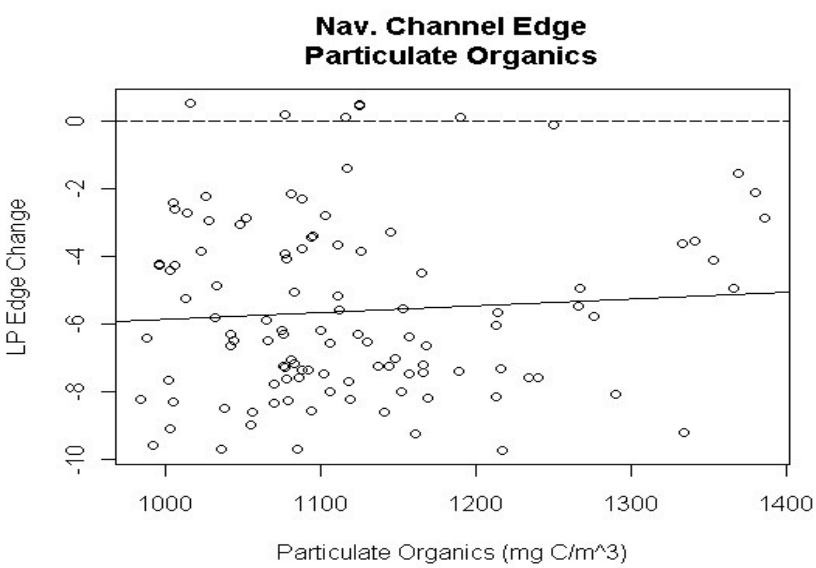


Figure 5.37. The regression of marsh edge loss from 1966 - 2007 for points along navigational channels as a function of Particulate Organics.



General characteristics of variables: Turbidity

The simple statistics for Turbidity are shown in Table 5.37. Starting with a simple regression and the transformed response variable, the diagnostic plot looked reasonable (Fig. 5.38), and the regression was not significant (Table 5.38; Fig. 5.39), with an R^2 of 0.03.

Table 5.37. Simple statistics for Turbidity for points along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
11.77	13.92	14.54	14.97	16.02	23.75	1.86	0.13	194

Table 5.38. Edge change from 1966 – 2007 for points on navigational channels regressed as a function of Turbidity.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-10.04	2.31	-4.34	<<0.001
0.31		0.16	1.92	0.06

Multiple R-squared: 0.03, Adjusted R-squared: 0.02

Figure 5.38. A diagnostics plot for edge loss along navigational channels from 1966 - 2007 regressed against Turbidity.

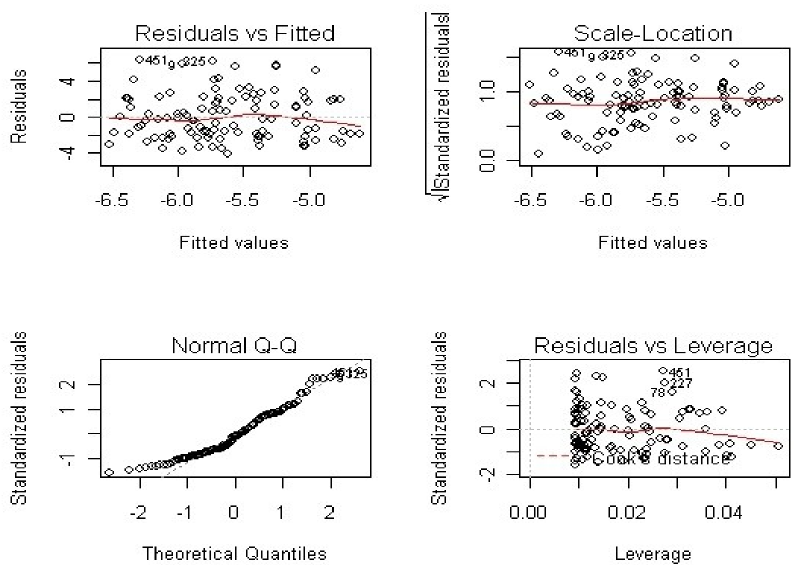
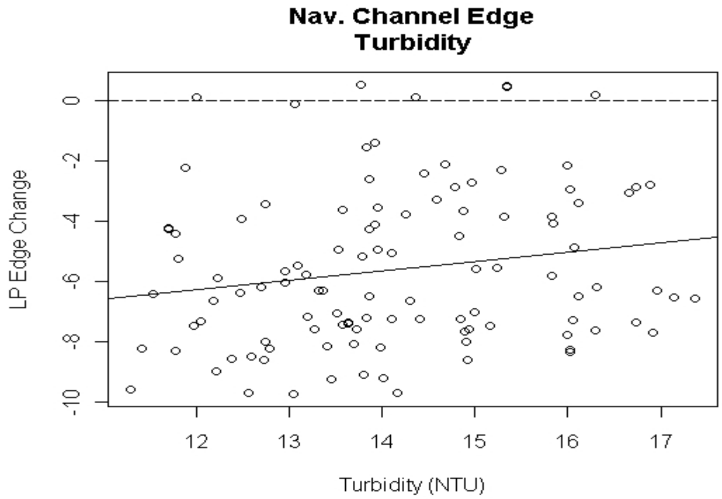


Figure 5.39. The regression of edge loss along navigational channels from 1966-2007 as a function of Turbidity.



Stepwise regression.

The three parameters (Distance to Borrow Pits, Particulate Organics and Turbidity) were first assessed with a Bayesian model-mixing algorithm in the bms package for R. An even prior distribution was used, so that no parameter was favored. Because Particulate Organics values outside of the water quality sampling area contained null values, only 206 points were used in the model building process. The output table indicated that the Distance to Borrow Pit term was the most likely to become part of the final model but the Particulate Organics and Turbidity terms were less likely to be included in the final model (Table 5.39). An AIC based method from the MMIX package for R was also used for the same data set. The mixAic function in this package also found that Distance to Borrow Pit was important, and that Particulate Organics and Turbidity were less important parameters (Table 5.40 and Fig. 5.40). In both cases the Distance to Borrow Pit was the most important term, followed by Particulate Organics and Turbidity, and only moderate interactions were found.

Table 5.39. Using 122 points, the Bayesian model averaging package BMS in R was used with an even prior to estimate a reduction in the sediment parameters variable set as a step in developing an optimal model for marsh edges that were along navigational channels. The result indicates that it is important to include Distance to Borrow Pit in a final model, but Turbidity and Particulate Organics may not be important (PIP = posterior inclusion probability, Cond.Pos Sign = the posterior probability of a positive coefficient expected value conditional on inclusion, Idx = input order of variables).

	PIP	Post Mean	Post SD	Cond.Pos Sign	Idx
Distance to Borrow Pits	0.76	0.001	0.001	1	1
Turbidity	0.19	0.036	0.097	1	3
Particulate Organics	0.14	0.000	0.002	1	2

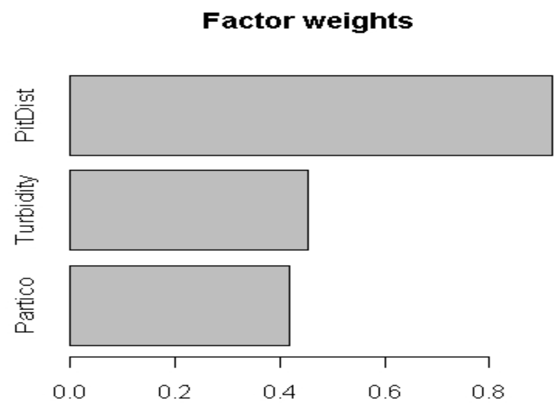
Table 5.40. The top three models from the AIC model mixing function mixAic are listed. A null model was a better model than any of these variables. Particulate Organics and Turbidity were possible candidates for inclusion in the best model for points at the edges of navigational channels for edge changes from 1966-2007.

	Model selected	AIC of selection
1	Edge Change 1966-2007 ~ Distance to Borrow Pit	616.45
2	Edge Change 1966-2007~ Distance to Borrow Pit + Turbidity	617.05
3	Edge Change 1966-2007~ Distance to Borrow Pit + Particulate Organics	617.17

Table 5.41. The relative importance of the variables in the chosen model (Table 5.42) as determined by relimpo from R package relaimpo. The proportion of variance explained by the full model was 12.39%.

Variable	Percent of variance
Distance to Borrow Pit: Channel Classification	3.91%
Channel Classification	3.85%
Distance to Borrow Pits	4.63%

Figure 5.40. Factor weights from the AIC model mixing function mixAic are listed. Distance to Borrow Pits is more important than Particulate Organics and Turbidity for inclusion in the best model of factors that influence edge changes from 1966-2007 for points at the edges of navigational channels.



The best model was found using the forward and backward stepping functions, and double-checking by hand. Only the Distance to Borrow Pit was included as an independent variable (Table 5.34; Fig. 5.35). However, some spatial autocorrelation was detected for this model (Fig. 5.41). Including Channel Classification into the model reduced the spatial autocorrelation (Fig. 5.42) and contributed to the explained variance (Table 5.41).

Figure 5.41. A variogram showing the degree of spatial autocorrelation present in the regression model that included Distance to Borrow Pit as the explanatory variable for change in the position of marsh edges from 1966-2007 for points along navigational channels.

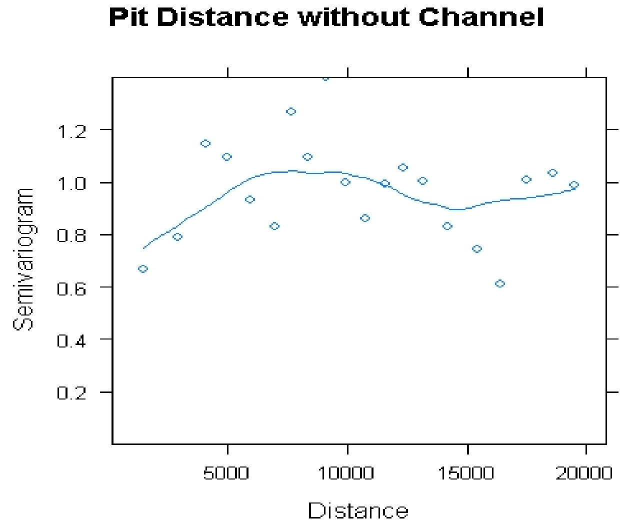
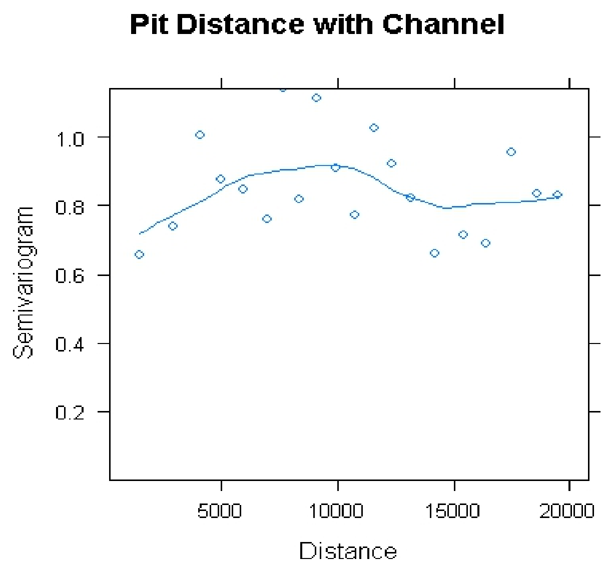


Figure 5.42. A variogram showing reduced spatial autocorrelation when Channel Classification was included.



A simple significant relationship seemed to exist when Channel Classification was not included in the model (Table 5.34; Fig. 5.35). However, marsh edges along different Channel classifications may be affected differently (Table 5.42; Fig. 5.43). Additionally, the influence of borrow pits may not be linear with distance, but may be more pronounced for locations less than 800 m from the pits (Fig. 5.43).

Table 5.42. The regression of edge loss along navigational channels from 1966-2007 as a function of Distance to Borrow Pits and Channel Classification (boat use classifications A-E, defined in Table 3.1).

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
ChannelA		-7.92	1.05	-7.52	<< 0.001
ChannelB		-6.87	0	2.44	0.01
ChannelC		-5.43	1.58	0.69	0.49
ChannelD		-8.7	1.44	1.72	0.09
ChannelE		-5.69	1.78	-0.43	0.67
Distance to Borrow Pit: ChannelA	0.002		0	2.43	0.02
Distance to Borrow Pit: ChannelB	0.001		0	-0.70	0.48
Distance to Borrow Pit: ChannelC	-0.002		0	-2.05	0.04
Distance to Borrow Pit: ChannelD	0.017		0	0.49	0.63
Distance to Borrow Pit: ChannelE	0.001		0	-0.74	0.46

Multiple R-squared: 0.12, Adjusted R-squared: 0.05

Figure 5.43. A loess fit of Marsh Edge Change along navigational channels from 1966 - 2007 and Distance to Borrow Pit with 95% confidence limits. More erosion is seen for points less than 600 m distant from borrow pits than for those further away.

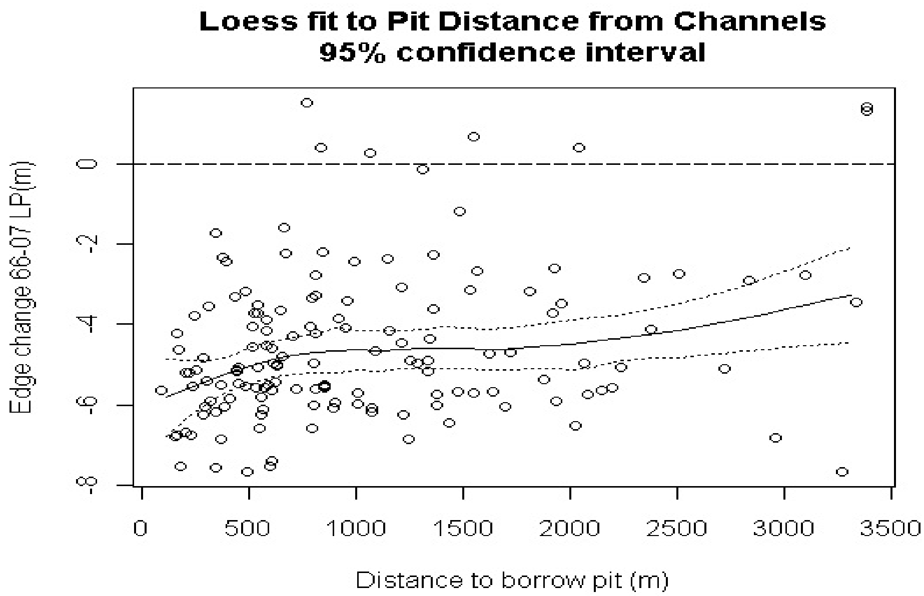
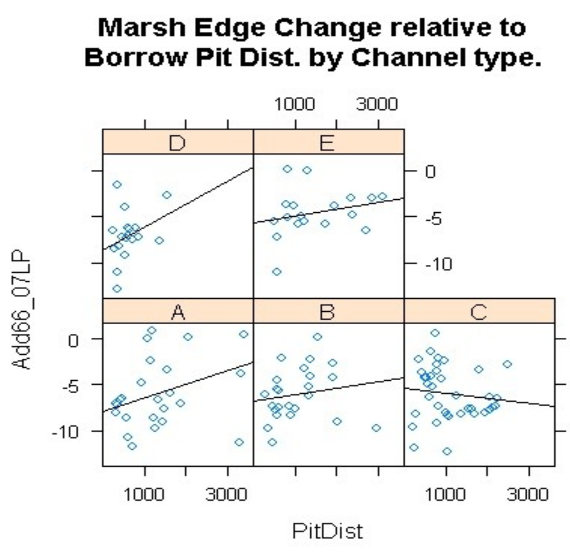


Figure 5.44. The regression of marsh edge loss from 1966-2007 for points along navigational channels and Distance to Borrow Pit for the different channel classifications.



Conclusion:

The loss of marsh from the edge from 1966-2007 for points along navigational channels was weakly influenced by the Distance to Borrow Pit. These borrow pits may act as sediment sinks for material re-suspended in the navigational channels. The influence Distance to Borrow Pit on boating channels may be more severe at distances 600 m or less.

NON-CHANNELS AND STORM IMPACTS

Edge change points that were on natural edges not on navigational channels were used to test the effects of storm driven waves (Significant Height of Storm Waves = Wave165D25) for two time frames, from 1926 - 1966 (Edge Change 1926-1966) and from 1966 - 2007 (Edge Change 1966-2007).

General characteristics of variables: Edge Change 1926-1966 for points on edges that are not located along navigational channels.

The simple statistics for Edge Change 1926-1966 for points on edges that are not located along navigational channels are shown in Table 5.43.

Table 5.43. The simple statistics for Edge Change 1926-1966 at points on natural edges not located along a navigational channel.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-144.4	-7.32	-1.78	-5.26	0.98	98.26	24.34	1.55	246

General characteristics of variables: Significant Height of Storm Waves

The simple statistics for Significant Height of Storm Waves for points on edges that were not located along navigational channels are shown in Table 5.44.

Table 5.44. The simple statistics for Significant Height of Storm Waves for points on natural edges not located along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
0	0.15	0.26	0.25	0.35	0.57	0.14	0.01	246

The effect of storm waves on marsh edges 1966 - 2007 not facing navigational channels.

For the regression of the transformed response variable, the change in the marsh edge from 1966-2007, and Significant Height of Storm Waves the diagnostic plot looked reasonable (Fig. 5.45) and the regression was significant (Table 5.45; Fig. 5.46) with an R^2 of 0.032. A loess fit to the same data revealed increased erosion along shores for wave heights above 0.2 m, likely reflecting those points more directly facing the higher waves (Fig. 5.47).

Table 5.45. A regression of Edge Change 1966 - 2007 at points on natural edges not located along navigational channels as a function of Significant Height of Storm Waves.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-2.03	0.28	-7.12	<< 0.001
-2.84		1.01	-2.83	0.01

Multiple R-squared: 0.032, Adjusted R-squared: 0.028

Figure 5.45. Diagnostics plots for edge loss for points on natural edges not located along navigational channels 1966 to 2007 regressed against Significant Height of Storm Waves.

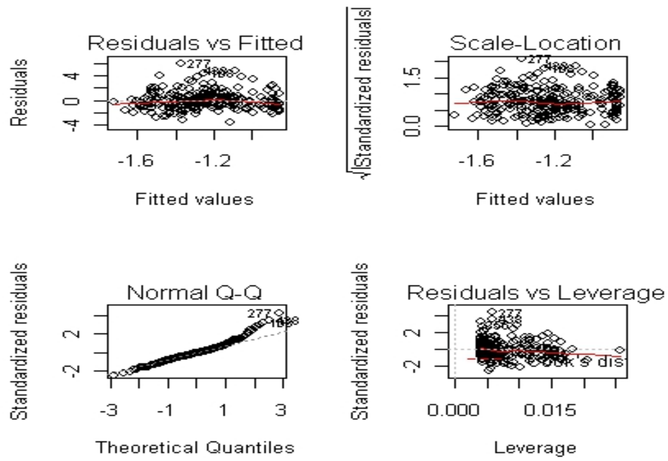


Figure 5.46. The relationship between Significant Height of Storm Waves and Edge Change 1966-2007 for natural edges not located along navigational channels.

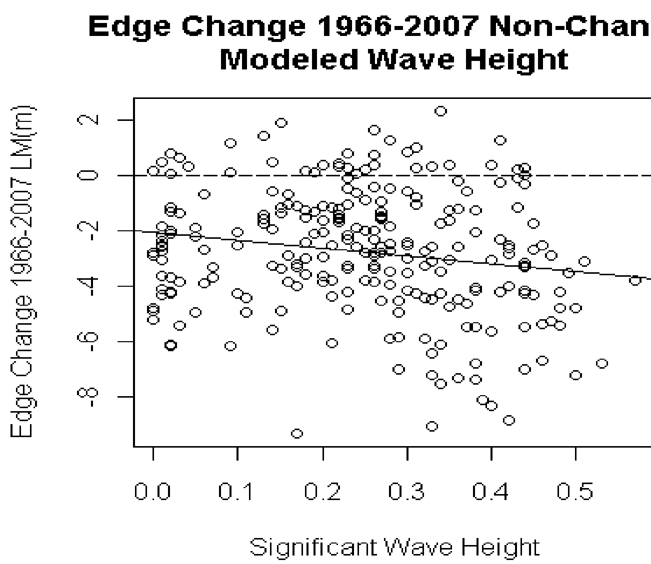
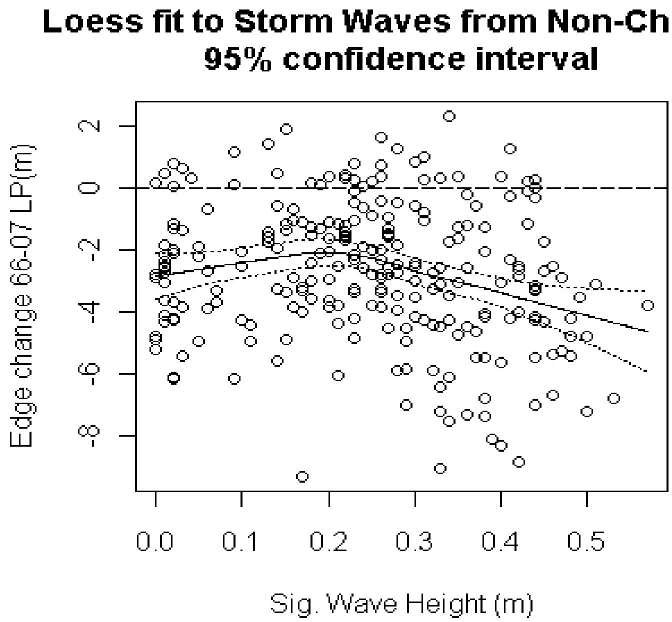


Figure 5.47. Loess fit of Significant Height of Storm Waves to Marsh Edge Change 1966-2007 for points located along natural edges that were not along navigational channels.



The effect of storm waves on marsh edges not facing navigational channels from 1926 – 1966.

For a simple regression of Significant Height of Storm Waves on the transformed response variable, the change in the marsh edge from 1926-1966, the diagnostic plot looked reasonable (Fig. 5.48) and the regression was significant (Table 5.46; Fig. 5.49) with an R^2 of 0.032. A loess fit to the same data revealed increased erosion for wave heights above 0.2 m, possibly reflecting those points more directly facing the higher waves (Fig. 5.50). This inflection was more pronounced for the marsh loss from 1926 – 1966 data than for marsh loss 1966 – 2007.

Table 5.46. A regression of Significant Height of Storm Waves and Marsh Edge Change 1926 - 1966 for natural edges not located along navigational channels.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-0.88	0.37	-2.39	0.02
-5.70		1.29	-4.64	<<0.001

Multiple R-squared: 0.083,   Adjusted R-squared: 0.079

Figure 5.48. Diagnostics plot for the regression of Marsh Edge Change for points from 1926 - 1966 on natural edges not along navigational channels and Significant Height of Storm Waves.

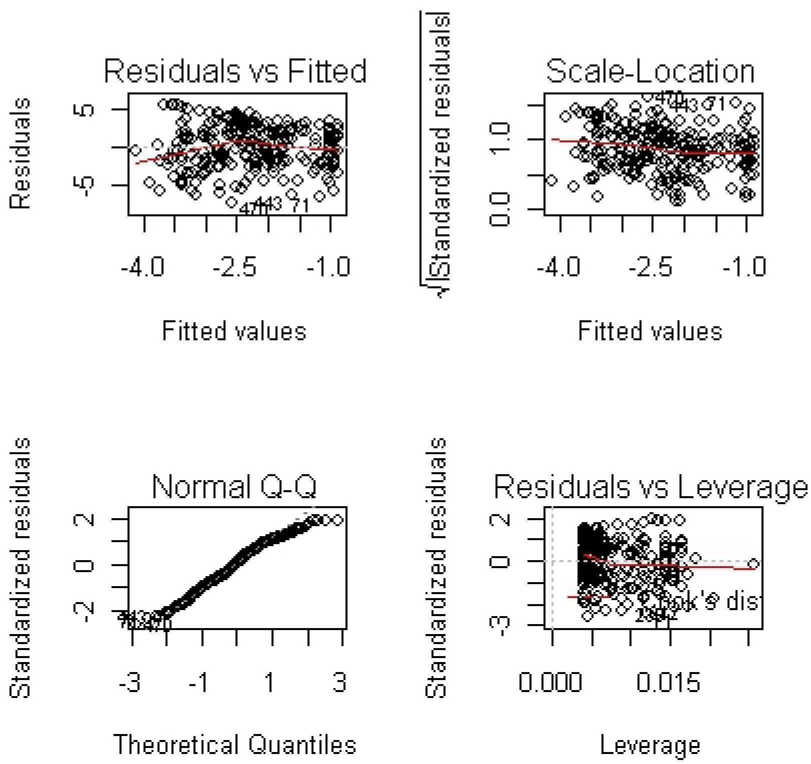


Figure 5.49. **R**egression of Significant Height of Storm Waves and Edge Change 1926-1966 for natural edges not located along navigational channels.

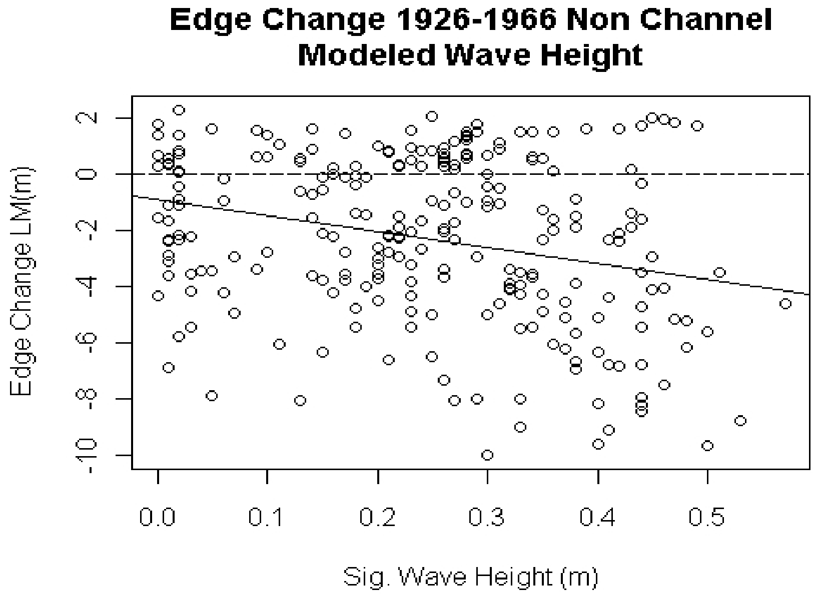
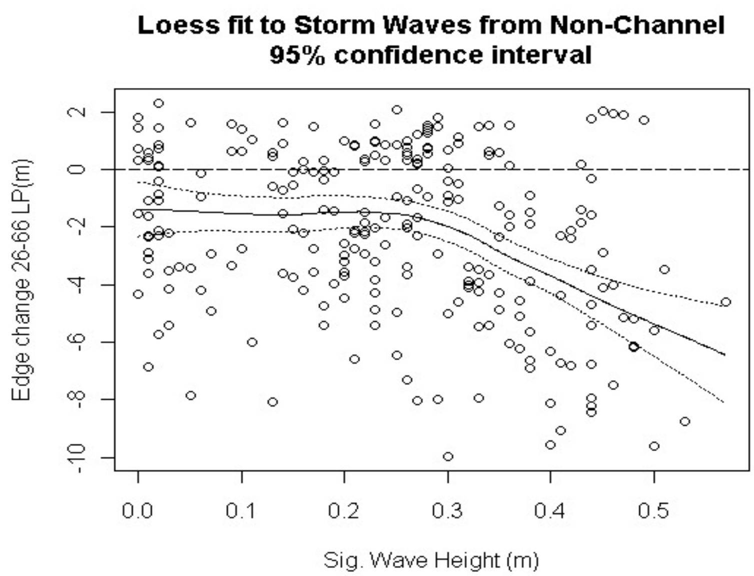


Figure 5.50. A loess fit to the relationship between Significant Height of Storm Waves and Edge Change 1926-1966 for natural edges



Conclusion:

Marshes that have natural, non-channelized edges, that are not found on navigational channels lost more marsh if the average significant height of storm waves was greater than about 0.2 - 0.3 m. This effect was seen for both time periods, 1926-1966 and 1966-2007, but was more evident for the earlier time interval.

NAVIGATIONAL CHANNELS AND STORM IMPACTS

Edge change points with natural edges that were on navigational channels were used to test the effects of storm driven waves (Significant Height of Storm Waves = Wave165D25) on the changes in marsh edges from 1926-1966 and 1966-2007

General characteristics of variables: Edge Change 1926-1966 for points on edges located along navigational channels.

The simple statistics for Edge Change 1926-1966 for points on marsh edges that were located along navigational channels are shown in Table 5.47.

Table 5.47. The simple statistics for Edge Change 1926-1966 at points located on natural edges along a navigational channel.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-93.08	-12.91	-6.07	-4.78	1.48	210.50	27.40	2.26	147

General characteristics of variables: Significant Height of Storm Waves

The simple statistics for Significant Height of Storm Waves for points on marsh edges that were located along navigational channels are shown in Table 5.48.

Table 5.48. The simple statistics for Significant Height of Storm Waves for points on natural edges located along navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
0	0.21	0.31	0.30	0.4	0.54	0.15	0.01	147

The effect of storm waves on marsh edges 1966 - 2007 for points facing navigational channels.

Starting with a simple regression of Significant Height of Storm Waves on the change in the marsh edge from 1966-2007 for points along navigational channels, the diagnostic plot looked reasonable, although not perfect (Fig. 5.51), and the regression was not significant (Table 5.49; Fig. 5.52) with an R^2 of 0.004.

Table 5.49. A regression of Edge Change 1966-2007 at points along navigational channels with natural edges as a function of Significant Height of Storm Waves.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-4.89	0.38	-12.85	<<0.001
-0.92		1.15	-0.8	0.43

Multiple R-squared: 0.004, Adjusted R-squared: 0.0

Figure 5.51. Diagnostics for Edge Change 1966- 2007 for points located on navigational channels with natural edges as a function of Significant Height of Storm Waves.

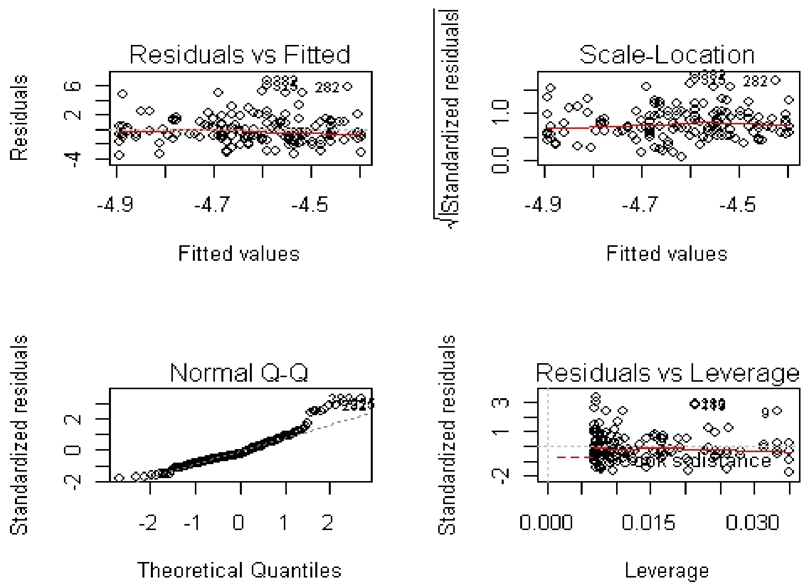
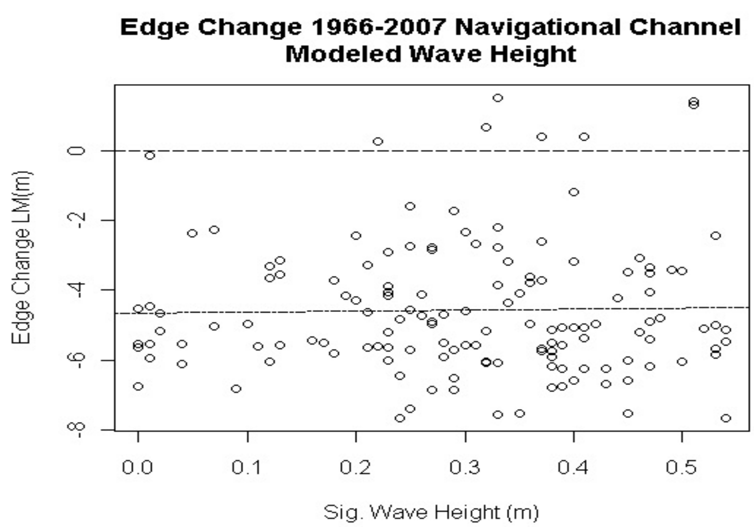


Figure. 5.52. A regression of Edge Change 1966- 2007 for points located on navigational channels with natural edges as a function of Significant Height of Storm Waves.



The effect of storm waves on marsh edges 1926 - 1966 for points facing navigational channels. Starting with a simple regression for Significant Height of Storm Waves, on the change in the marsh edge from 1926-1966 for points along navigational channels, the diagnostic plot did not look completely normal and the LP transform was used with the value 1.0, making it and unadjusted log transform (Fig. 5.53). The regression with the transformed data was not significant (Table 5.50; Fig. 5.54) with an R^2 of 0.0.

Table 5.50. A regression of edge change 1926 to 1966 at points along navigational channels as a function of Significant Height of Storm Waves.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-0.96	0.44	-2.17	0.03
-0.26		1.34	-0.2	0.85

Multiple R-squared: 0.0, Adjusted R-squared: 0.0

Figure 5.53. Diagnostics for a regression of Edge Change 1926-1966 for points along natural edges on navigational channels as a function of Significant Height of Storm Waves.

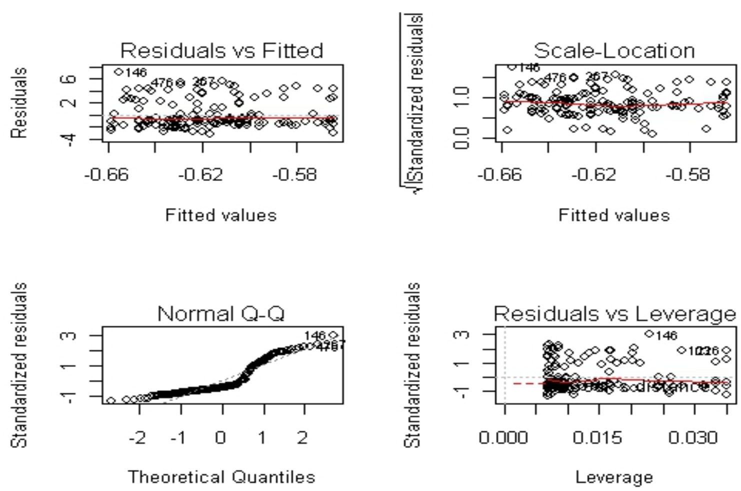
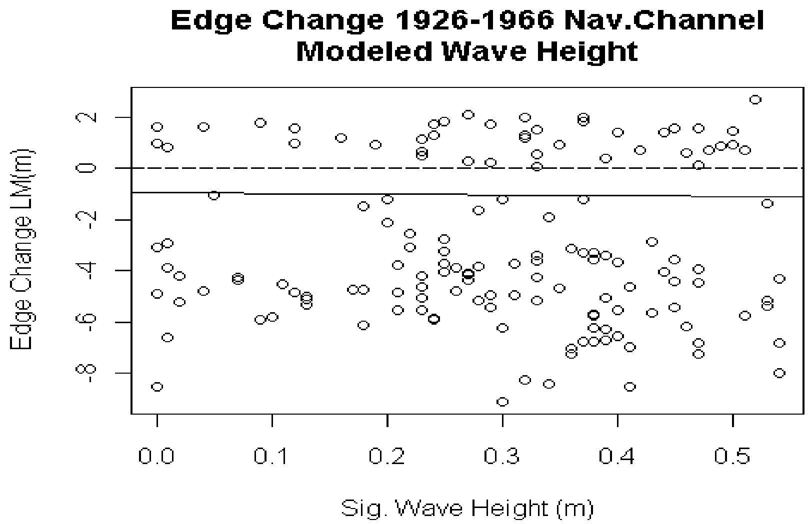


Figure 5.54. A regression of Edge Change 1926-1966 at points along natural edges on navigational channels as a function of Significant Height of Storm Waves.



Conclusion:

No storm wave effects were detectable for marsh edges located along navigational channels with natural edges for either the 1926 - 1966 or 1966 - 2007 time periods. It is likely that other effects that impact navigational channels mask any storm wave effects.

NON-CHANNELS AND ENTIRE REDUCED VARIABLE SET

Natural edges that did not face navigational channels were examined next with linear models that included natural physical variables (Fetch = CnlWth66Mr, Peak Local Tidal Flow = FlowRt, Nitrate = Nitrat2, Channel Unlimited (Euclidean) Distance = ChnlUnMtr, and Distance to Borrow Pit = PitDistMr) to determine which combination of variables best explained the changes in marsh edges from 1966 - 2007, and what are their relative contributions were.

General characteristics of variables: The Peak Local Tidal Flow Rate

The Peak Local Flow Rate ranged from 0 to 0.29 m s^{-1} and a mean of 0.1 (± 0.004) m s^{-1} (Table 5.61). A regression of the LP transformed response variable, the change in the location of the marsh edge from 1966 – 2007, for points on natural edges that were not located along navigational channels, as a function of Peak Local Flow Rate appeared normal (Fig. 5.55). This regression was not significant (Table 5.62, Fig. 5.56).

Table 5.51. The Peak Local Flow Rate (m s^{-1}) from GSB model output for points not located on navigational channels (m s^{-1}).

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
0	0.07	0.1	0.1	0.13	0.29	0.06	0.004	216

Table 5.52. The regression table for the relationship between Peak Local Flow Rate (m s^{-1}) and Edge Change 1966-2007 for non-channel areas.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-2.31	0.32	-7.31	<<0.001
-3.55		2.66	-1.34	0.18

Multiple R-squared: 0.01, Adjusted R-squared: 0.00

Figure 5.55. A diagnostic plot of the Peak Local Flow Rate data and transformed Edge Change 1966 - 2007 for non-channel areas.

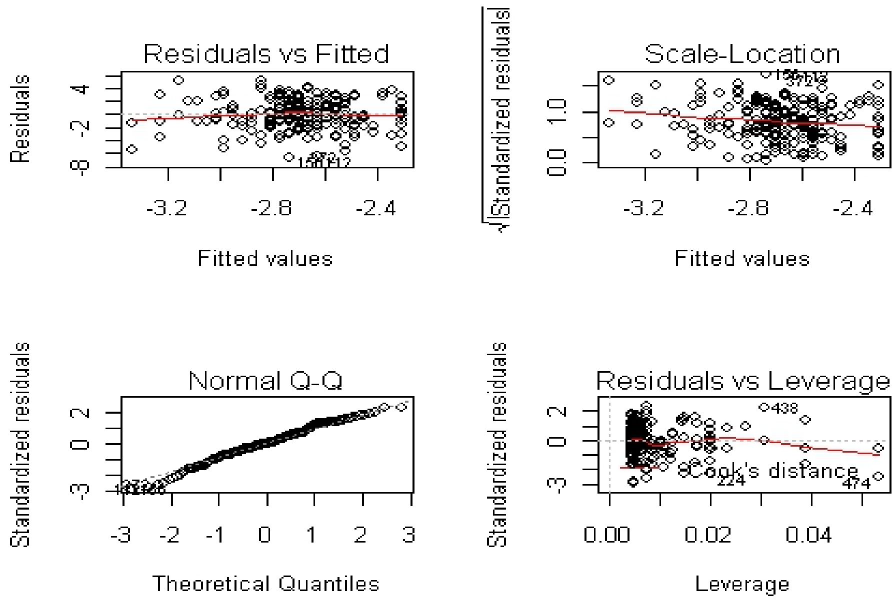
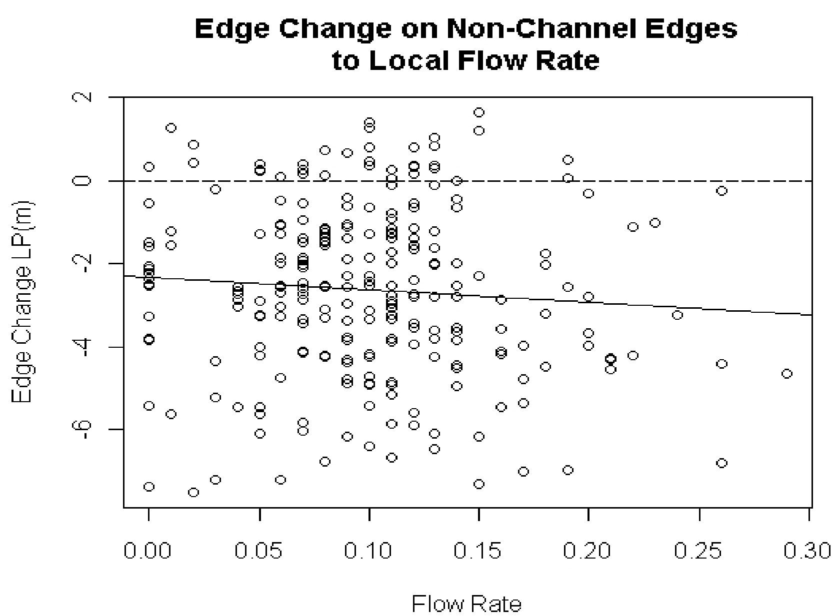


Figure 5.56. A scatter plot of the regression of Edge Change 1966-2007 as a function of Peak Local Flow Rate for points not located on navigational channels.



General characteristics of variables: Channel Unlimited Distance 1966-2007

The Channel Unlimited Distance (Channel Unlimited Distance = ChnlUnMtr) was the Euclidean distance from the measurement point to the nearest navigational channel, including over obstructions and islands. This variable is more likely to be related to sediment movement than boat wakes. The Channel Unlimited Distance ranged from 43.7 to 1009.0 m, with a mean of 396.8 (± 15.4) m (Table 5.53). A regression of the LP transformed response variable, Edge Change 1966 – 2007, for points on natural edges that were not located along navigational channels, as a function of Channel Unlimited Distance was not significant (Table 5.54, Fig. 5.58).

Table 5.53. The Channel Unlimited Distance for points not on navigational Channels and within the water quality testing zone (Fig. A.1).

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
43.72	225.5	359.4	396.8	553.9	1009	226.09	15.38	216

Table 5.54. The regression table of Channel Unlimited Distance as a function of the change in marsh edge location for points not located on navigational channel edges over the period 1966-2007.

Slope	Intercept	Std. Error	t value	Pr(>|t|)
	-2.98	0.31	-9.70	<<0.001
0.001		0.001	1.14	0.25

Multiple R-squared: 0.01, Adjusted R-squared: 0.00

Figure 5.57. A diagnostic plot of the regression of Edge Change 1966-2007 as a function of Channel Unlimited Distance for points not located on navigational channel edges.

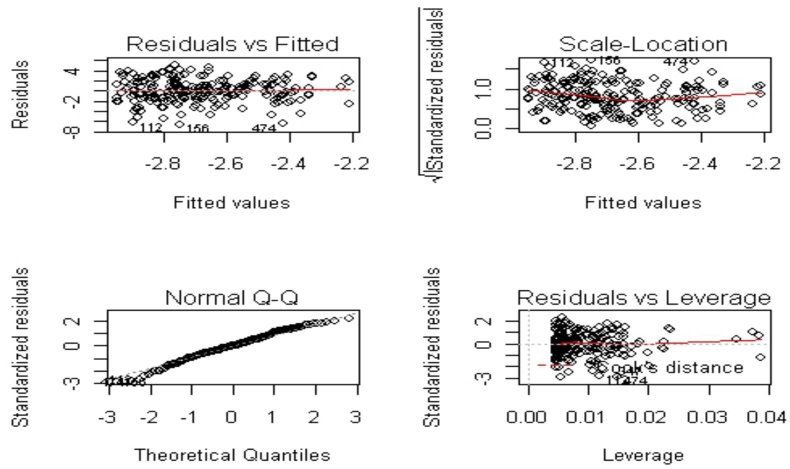
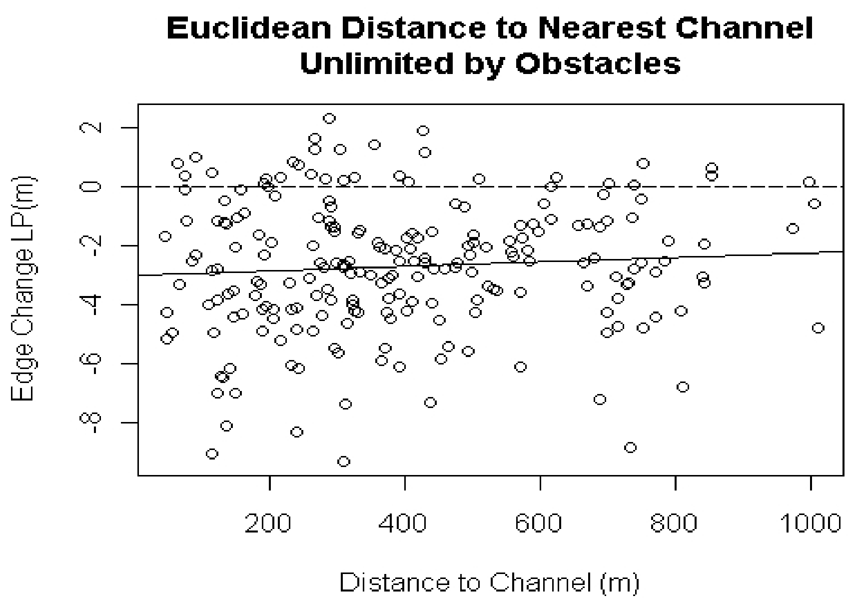


Figure 5.58. A scatterplot of the relationship between Edge Change 1966-2007 and Channel Unlimited Distance for points not located on navigational channel edges.



Stepwise regression.

The reduced set of parameters (Minimum Fetch, Peak Local Tidal Flow Rate, Nitrate, Channel Unlimited Distance, and Distance to Borrow Pit) were first assessed with a Bayesian model mixing algorithm in the bms package for R. An even prior distribution was used, so that no parameter was favored. Because Nitrate values outside of the water quality sampling area (Fig. A.1) contained null values and 3 points held erroneous Nitrate values of 0, only 213 points were used in the model building process. The output table indicated that the Minimum Fetch term was the most likely to become part of the final model and the other terms (Peak Local Tidal Flow Rate, Nitrate, Distance to Borrow Pit and Channel Unlimited Distance) were much less likely to be included in the final model (Table 5.56, Figure 5.59). An AIC based method from the MMIX package for R was also used for the same data set. The mixAic function in this package also found that Minimum Fetch was important with a weight of 1.0 but no other parameter had a weight over 0.45 (Fig. 5.60). In both cases Minimum Fetch was the most important term, and variance inflation factors were low for even the full model (Table 5.55).

Table 5.55. The variance inflation factors for the full model without interactions.

Peak Local Tidal Flow Rate	Distance to Borrow Pit	Minimum Fetch	Channel Unlimited Distance	Nitrates
1.07	1.23	1.01	1.08	0.12

Table 5.56. Using 213 points, the Bayesian model averaging package BMS in R was used with an even prior to estimate a further reduction in set of variables associated with sediment as a step in developing an optimal model for marsh edges that were not along navigational channels. Minimum Fetch was the most likely variable for inclusion in the best model with a posterior Inclusion Probability of 0.99. Fetch was the only variable with a probability of inclusion in the best model that was greater than 0.15. (PIP = posterior inclusion probability, Cond.Pos Sign = the posterior probability of a positive coefficient expected value conditional on inclusion, Idx = input order of variables).

	PIP	Post Mean	Post SD	Cond.Pos Sign	Idx
Minimum Fetch	0.99	-0.003	0.001	0	5
Peak Local Tidal Flow Rate	0.12	-0.376	1.353	0	3
Nitrate	0.12	-0.007	0.026	0	1
Channel Unlimited Distance	0.07	0.000	0.000	1	2
Distance to Borrow Pit	0.07	0.000	0	0.87	4

Figure 5.59. Bayesian Model Averaging indicated that Fetch was the only parameter needed for a model of marsh edge loss for 1966-2007 for points that were along navigational channels with natural edges.

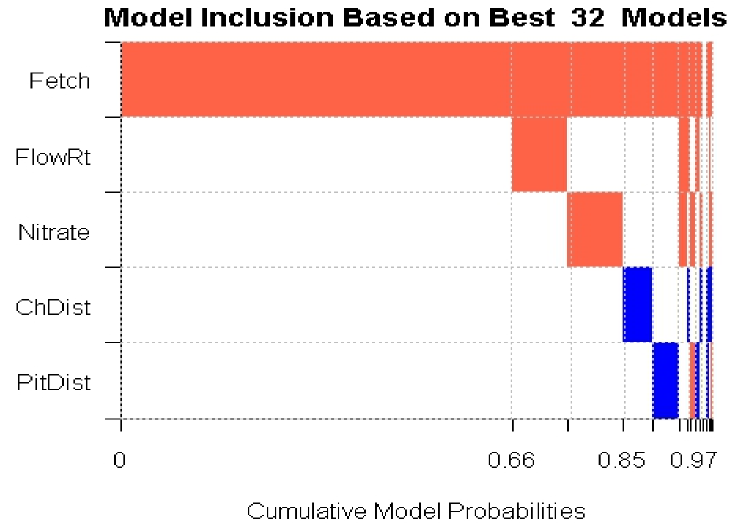
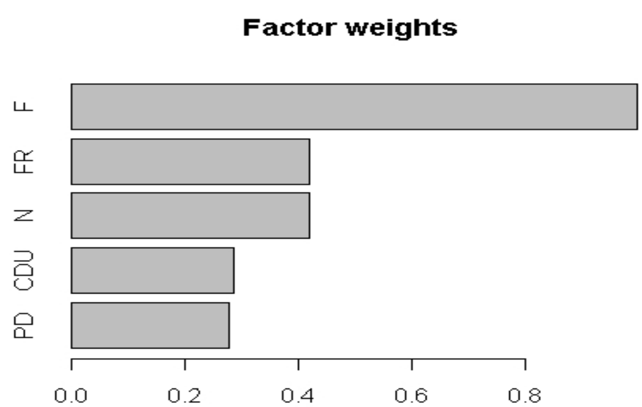


Figure 5.60. Factor weights from the AIC model mixing function mixAic are listed. Fetch (F) is more important than Peak Local Tidal Flow Rate (FR), Nitrates (N), Channel Unlimited Distance (CDU) and Distance to Borrow Pits (PD) for inclusion in the best model of factors influencing changes in th marsh edges from 1966-2007 for points at points that are not located along navigational channels.



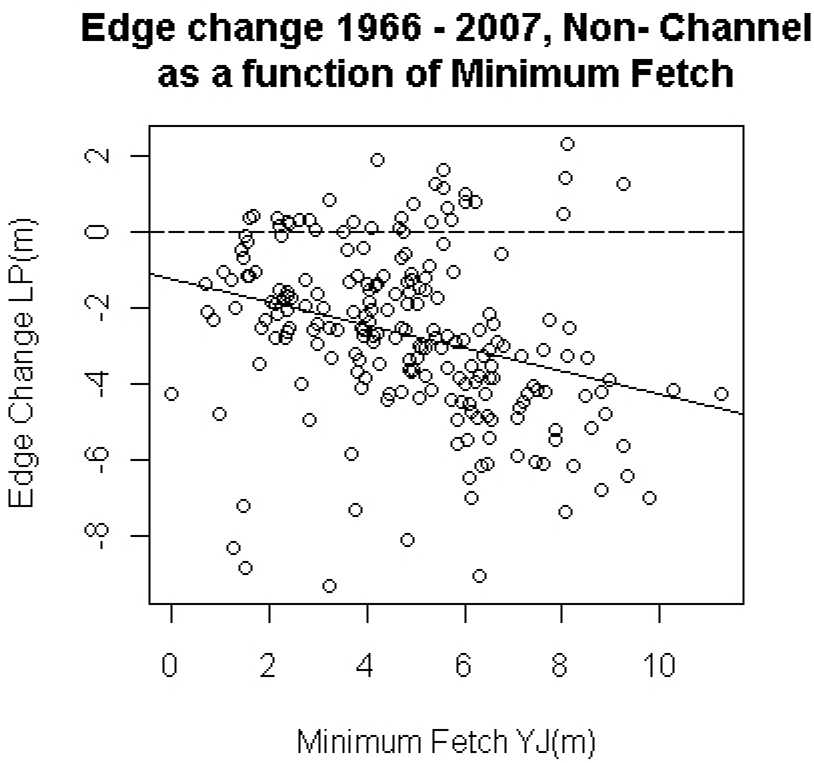
Forward and forward – backward stepwise regression were used to develop the best model for Minimum Fetch and all other variables (Nitrates, Peak Local Tidal Flow Rate, Channel Unlimited Distance, and Distance to Borrow Pits) and their possible interactions. The model with the minimum AIC included only Minimum Fetch and was highly significant (Table 5.57). The relationship between the LP transformed Edge Change 1966 – 2007 and the YJ transformed Minimum Fetch is illustrated in Fig. 5.61.

Table 5.57. The regression table of the effects of Minimum Fetch on LP transformed change in the edge of marshes from 1966-2007 for edges that were not along channels.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-2.36	0.172	-13.76	< 0.001
Minimum Fetch	-0.003		0.001	-3.9	< 0.001

Multiple R-squared: 0.07, Adjusted R-squared: 0.06

Figure 5.61. A scatterplot of the regression of LP transformed edge change 1966-2007 as a function of YJ transformed Minimum Fetch for points not located on navigational channel edges.



Conclusion:

When the entire reduced set of variables was analyzed, Minimum Fetch was the only important variable, and the model had an R^2 value of 0.07.

NAVIGATIONAL CHANNELS AND ENTIRE REDUCED VARIABLE SET

Natural marsh edges that faced navigational channels were examined next. Linear models that included all of the reduced set of variables (Distance to Borrow Pit = PitDistMr, Particulate Organics = Partico, Turbidity = Turv2mIDW3, Nitrate = Nitra2, and Navigational Channel Boat Use Classification = Channel) were tested to determine which combination of variables best explained the changes in marsh edges from 1966 - 2007. I extracted a subset of points that were located along navigational channels that were also within the water quality measurement zone (Fig. A.1).

General characteristics of variables:Channel Unlimited Distance 1966-2007

The variable Channel Unlimited Distance variable was not as accurate for the measurement to the center of an adjoining channel as the Direct Distance to Channel variable that was used in Chapter 4, but the two variables were collinear. Channel Unlimited Distance was used here so that the identical reduced set if variables was analyzed and could be compared with the previous section.

Table 5.58. Descriptive statistics for the variable Channel Unlimited Distance for navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
29.23	72.91	121.00	148.60	182.40	492.60	101.01	9.00	126

Table 5.59. Linear regression for change in the location of the marsh edge from 1966 - 2007 for navigational channels and the variable Channel Unlimited Distance.

		Std. Error	t value	Pr(>|t|)
Intercept	-5.25	0.320	-16.08	<<0.001
Slope	0.01	0.002	2.66	0.009

Stepwise regression.

For points located on the edges of navigational channels, the reduced set of parameters (Minimum Fetch , Peak Local Tidal Flow, Nitrate, Channel Unlimited Distance, and Distance to Borrow Pit) were first assessed with a Bayesian model mixing algorithm in the bms package for R. An even prior distribution was used, so that no parameter was favored. Because Nitrate values outside of the water quality sampling area contained null values and any points that held erroneous Nitrate values of 0, only 126 points were used in the model building process. The output table indicated that the indicated that Peak Local Tidal Flow Rate, Distance to Borrow Pit, and Minimum Fetch were the most likely combination to become part of the final model and the other terms (Nitrate, and Channel Unlimited Distance) were much less likely to be needed in the final model (Table 5.60, Figure 5.62).

An AIC based method from the MMIX package for R was also used for the same data set. The mixAic function in this package also found that Peak Local Tidal Flow Rate, Distance to Borrow Pit, and Minimum Fetch were important with weight factors of 1.0, 9.8 and 9.7 respectively (Table 5.61; Fig. 5.73). The Channel Unlimited Distance parameter had a weight over 0.7.3, giving it more importance than was the case with the Bayesian method, but both methods rated Nitrates as the least important variable. In both cases the Peak Local Tidal Flow Rate was the most important term.

Table 5.60. For 125 points along navigational channels 65-1,200 m width, the Bayesian model averaging package BMS in R was used with an even prior to as a step in developing an optimal model for marsh edges that were along navigational channels. Distance to Borrow Pit was the most likely variable for inclusion in the best model with a posterior Inclusion Probability of 0.92, followed closely by Peak Local Tidal Flow Rate and Minimum Fetch. (PIP = posterior inclusion probability, Cond.Pos Sign = the posterior probability of a positive coefficient expected value conditional on inclusion, Idx = input order of variables).

	PIP	Post Mean	Post SD	Cond.Pos Sign	Idx
Distance to Borrow Pits	0.94	<0.001	<0.001	1	3
Peak Local Tidal Flow Rate	0.86	-2.29	1.25	0	5
Channel Unlimited Distance	0.40	0	0.001	1	2
Minimum Fetch	0.37	0.001	<0.001	0	1
Nitrate	0.13	0.004	0.015	0.95	4

Figure 5.62. Bayesian Model Averaging and the most likely variables for inclusion in the optimal model for marsh edges along channels, indicating that it is likely that Peak Local Tidal Distance to Borrow Pits, Distance to Channel Center from the marsh edge and Flow Rate were the parameters likely needed for a model of edge loss from 1966-2007 for points that were on navigational channels of 65-1,200 m width.

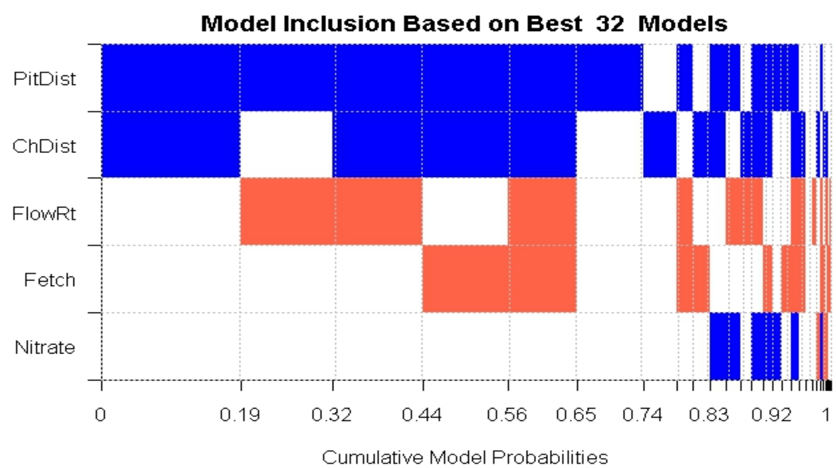


Figure 5.63. Factor weights from the AIC model mixing function mixAic are listed. Distance to Borrow Pit (PD), Peak Local Tidal Flow Rate (FR), Minimum Fetch (F) and Channel Unlimited Distance (CDU) were more important than Nitrate (N), and may all be needed for inclusion in the best model as factors influencing edge changes from 1966-2007 for points that are along navigational channels of 65-1,200 m width.

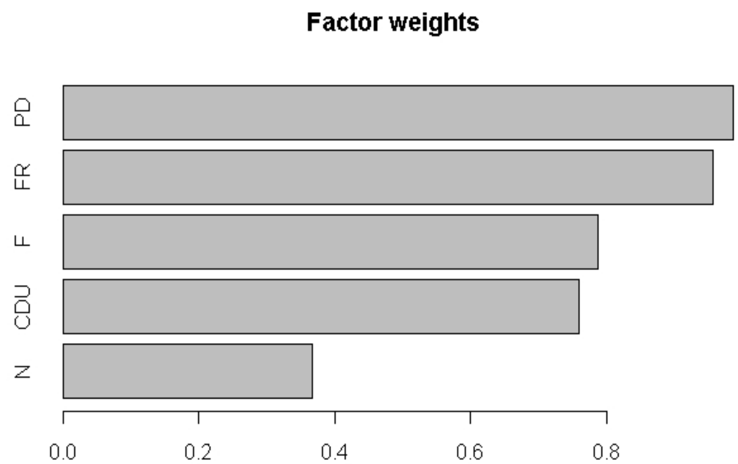


Table 5.61. The AIC model mixing function mixAic indicated that Peak Local Flow Rate (FR), Distance to Borrow Pit (PD), Channel Unlimited Distance (CDU), and Minimum Fetch (F) were the best variables for inclusion in the model for testing the reduced set of variables as factors in the change of marsh edges on navigational channels from 1966-2007. All 5 parameters together, including Nitrate (N), was the second best model

	Model selected	AIC of selection
1	Edge Change 1966-2007~ Channel Unlimited Distance + Peak Local Tidal Flow Rate + Distance to Borrow Pit + Minimum Fetch	525.78
2	Edge Change 1966-2007~ Nitrate + Channel Unlimited Distance + Peak Local Tidal Flow Rate + Distance to Borrow Pit + Minimum Fetch	526.88
3	Edge Change 1966-2007~ Peak Local Tidal Flow Rate + Distance to Borrow Pit + Minimum Fetch	527.9

The combination Peak Local Tidal Flow Rate, Distance to Borrow Pit and Minimum Fetch was used as the starting point for stepwise regression, both backstepping with a null model as a minimum and forward stepping with all 5 factors and their interactions (Table 5.62). This regression accounted for 22.55% of the variation (Table 5.63).

Table 5.62. The best model for the hypothesis free analysis of the reduced data set model for navigational channels (but not including the Channel class variable) resulted in an AIC value of 8.00 using the extractAIC function in R stats package.

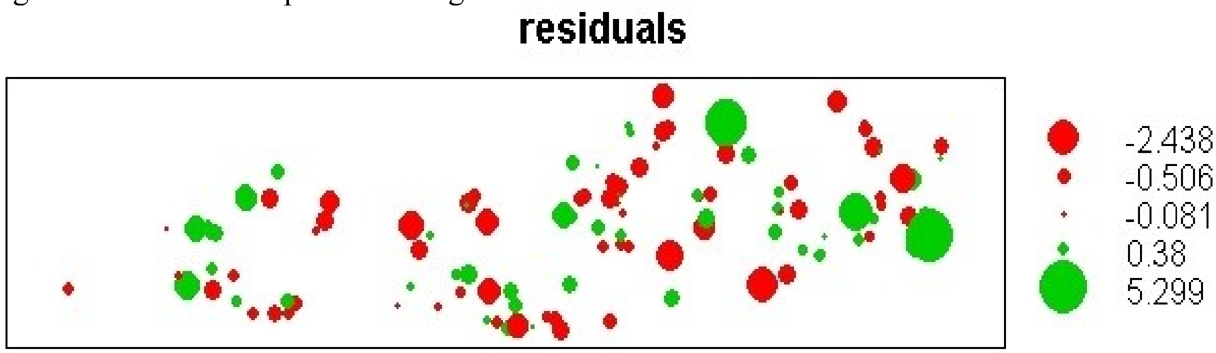
	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-1.4	0.39	-3.57	< 0.001
Channel Unlimited Distance: Peak Local Tidal Flow Rate	< 0.001		3.015	0.19	0.85
Minimum Fetch	0		0.000	-2.66	0.01
Distance to Borrow Pit	< 0.001		0.108	0.26	0.8
Peak Local Tidal Flow Rate:	-0.22		0.003	-0.14	0.1
Peak Local Tidal Flow Rate: Distance to Borrow Pit		-1.4	0.022	1.67	0.088
Peak Local Tidal Flow Rate: Channel Unlimited Distance		-1.42	0.022	-1.57	0.12
Minimum Fetch: Channel Unlimited Distance		-1.4	0.022	1.47	0.14

Multiple R-squared: 0.282, Adjusted R-squared: 0.240

Table 5.63. The relative importance of the variables in the chosen model (Table 4.8.1) as determined by relimpo from R package relaimpo reporting the proportion of variance explained by model as 28.20%.

Variable	Percent of variance
Channel Unlimited Distance	5.23%
Minimum Fetch	2.32%
Distance to Borrow Pits	6.64%
Peak Local Tidal Flow Rate	5.39%
Channel Unlimited Distance: Distance to Borrow Pits	3.14%
Channel Unlimited Distance: Peak Local Tidal Flow Rate	2.91%
Channel Unlimited Distance: Minimum Fetch	2.56%

Figure 5.64. A bubble plot of the regression residuals from Table 5.62.



After adding the variable Channel, which was for boat use categories, the stepwise regression as run again, now with 7 factors (Table 5.64). This regression accounted for 29.73% of the variation, and Peak Local Tidal Flow Rate accounted for 7.62%, Channel Classification added 5.36%, Peak Local Tidal Flow Rate interacted with Channel Classification to explain 5.56%, and the other factors continued to explain about the same amount of the variance (Table 5.65).

Table 5.64. The best model for the hypothesis free analysis of the reduced data set model for navigational channels, with the Channel variable included, resulted in an AIC value of 168.09 using the extractAIC function in R stats package

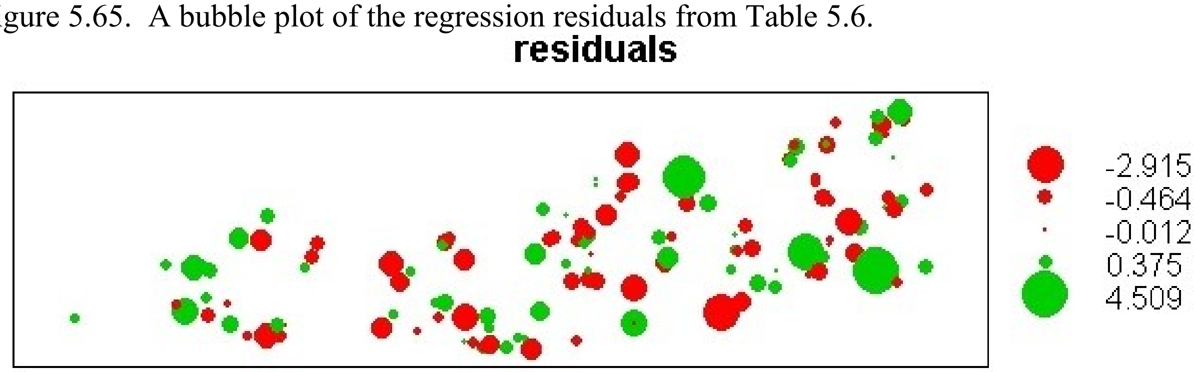
	Slope	Intercept	Std. Error	t value	Pr(>|t|)
ChannelA		-2.824	0.440	-6.42	< 0.001
Peak Local Flow Rate	0.042		1.308	0.390	0.975
Distance to Borrow Pits	0.041		< 0.001	2.410	0.017
Minimum Fetch	0.044		< 0.001	2.019	0.046
ChannelB		-2.35	0.488	0.978	0.331
ChannelC		-1.46	0.454	3.000	0.003
ChannelD		-1.9	0.427	2.166	0.032
ChannelE		-2.58	0.674	1.358	0.177
Channel Unlimited Distance	0.05		0.001	3.625	< 0.001
Minimum Fetch: ChannelB		-2.83	0.001	-2.195	0.030
Minimum Fetch: ChannelC		-2.87	0.001	-3.46	< 0.001
Minimum Fetch: ChannelD		-2.83	0.001	-3.54	< 0.001
Minimum Fetch: ChannelE		-2.83	0.003	-1.066	0.288
Channel Unlimited Distance:Peak Local Flow Rate		-2.85	0.010	-2.335	0.021

Multiple R-squared: 0.362, Adjusted R-squared: 0.299

Table 5.65. The relative importance of the variables in the chosen model (Table 4.8.1) as determined by relimpo from R package relaimpo reporting the proportion of variance explained by model as 36.19%.

Variable	Percent of variance
Minimum Fetch:Channel	9.58%
Peak Local Tidal Flow Rate	6.69%
Distance to Borrow Pits	5.71%
Channel Unlimited Distance	5.07%
Channel Unlimited Distance:Peak Local Tidal Flow Rate	3.99%
Channel	3.47%
Minimum Fetch	1.66%

Figure 5.65. A bubble plot of the regression residuals from Table 5.6.



Conclusion:

For marsh along navigational channels, the Peak Tidal Local Flow Rate was found to be the most important. When Channel Classification was included in the model, the interaction between Channel Classification and Minimum Fetch became the most important source of the variance. Distance to Borrow Pits, Minimum Fetch and Channel Unlimited Distance all had important effects as well. Only Nitrates failed to shows an influence on the change in location of edges at points located along navigational channels.

Chapter 6. ANALYSIS AND RESULTS: MARSH CHANGE IN EARLY VERSUS LATE 20TH CENTURY

The time period 1966-2007 covers most of the time over which major urbanized developed near the study area. The earlier time interval used in this study, 1926-1966, was a period of transition, with mixed suburban and rural land use. The villages of Hempstead and Freeport both date to the colonial period and other small framing and fishing communities surrounded them. The north-west portion of West Bay saw the construction of golf courses and an exclusive bedroom community as early as the 1920s and the City of Long Beach was constructed as a summer resort. However, the 1950's and 1960's were a time when many working class families were seeking to escape New York City and the farms and summer communities were replaced with year-round housing. The uses of the estuary that contains the study site shifted from fishing, shell fishing and light recreational use to heavy recreational use and remnant commercial fisheries.

In Chapter 4, I used MANOVA to analyze the difference in the edge loss rates of natural marsh edges between the time periods 1926 - 1966 and 1966 - 2007 when considering the variables Direct Distance to Channel Center and Channel boat use classification. These two variables were found to significantly affect the difference in marsh loss between these two time periods. Further analysis using ANOVA determined that the 1926 – 1966 period displayed significant differences among Channel use classifications, which were not explained by the variable Direct Distance to Channel Center (Table 4.17). For the 1966 – 2007 time interval the opposite was found There was no difference between the effect of Channel Classification and Direct Distance to Channel Center. Differences in marsh Edge Change between the two time intervals were significantly associated with Direct Distance to Channel Center (Table 4.17). This may reflect a shift from the influence of water flow rate associated with natural channels to influence from powerboat wakes.

I then tested for differences in marsh loss between the time intervals 1926 – 1966 and 1966 – 2007, and looked at how factors from the reduced set (Peak Local Flow Rate, Minimum Fetch, Particulate Organics, Channel Unlimited Distance, and Distance to Borrow Pit) effected marsh edge change for natural marsh edges.

COMPARISON OF MARSH EDGE CHANGE FOR NATURAL EDGES OF NAVIGATION CHANNELS AND NON-NAVIGATIONAL CHANNELS BETWEEN THE TIME INTERVALS 1926-1966 AND 1966-2007

Because the length of the two time periods were different, Spring 1926 to Spring 1966 is a 40 year interval while Spring 1966 to Spring 2007 is a 41-year interval, I used the average annual change for each marsh edge measurement point. Simple statistics were then computed for edges along navigational channels (Table 6.1; Table 6.2; Table 6.3; Figure 6.1) and marsh edges that were not along navigational channels (Table 6.4; Table 6.5; Table 6.6; Fig. 6.2).

Table 6.1. Descriptive statistics for the average annual marsh edge change 1926 - 1966 for navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-2.33	-0.32	-0.15	-0.12	0.04	5.26	0.69	0.06	147

Table 6.2. Descriptive statistics for the average annual marsh edge change 1966 - 2007 for navigational channels show loss during this time interval.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-1.66	-0.44	-0.28	-0.32	-0.12	1.52	0.35	0.03	147

Table 6.2. Descriptive statistics for the per point differences in average annual marsh edge change for points along navigational channels. The preliminary calculation subtracted the average annual marsh edge changes 1926 – 1966 from the average annual marsh edge changes for the 1966 - 2007 interval. Thus, for each point, a negative result indicated greater loss or smaller gain during the 1966 – 2007 time interval and a positive result indicated a greater gain or smaller loss during the 1966 – 2007 time interval.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-5.55	-0.34	-0.12	-0.20	-0.11	1.95	0.78	0.06	147

Figure 6.1. The frequency distribution of per-point differences between the marsh edges change for the time interval 1966 – 2007 and the time interval 1926 – 1966 for natural edges located along navigational channels indicates a greater loss rate for the 1966 – 2007 time interval.

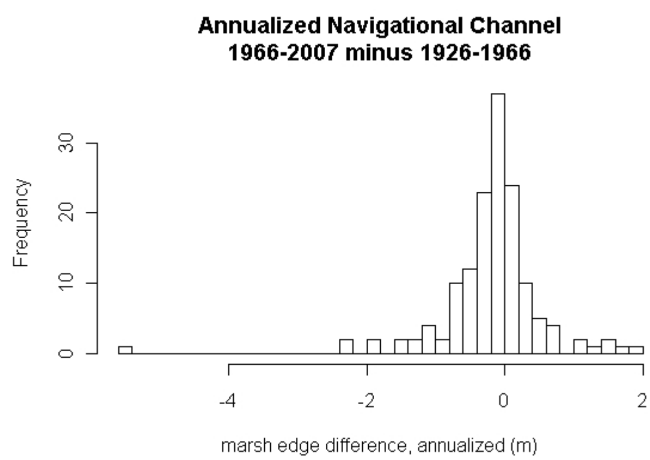


Table 6.4. Descriptive statistics for the average annual marsh edge change 1926 - 1966 for points that were along edges not located on navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-3.61	-0.19	-0.40	-0.14	0.02	2.46	0.62	0.04	247

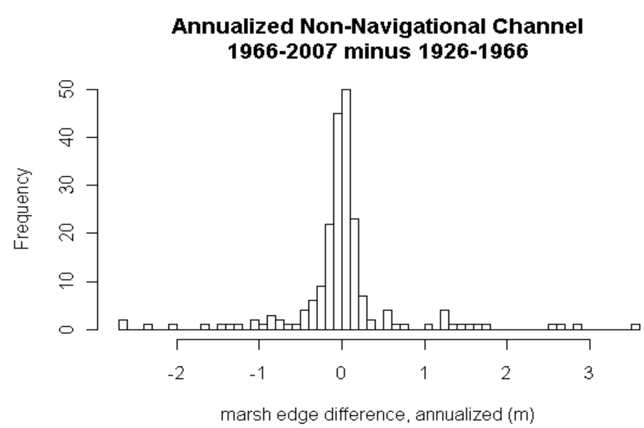
Table 6.5. Descriptive statistics for the average annual marsh edge change 1966 - 2007 for points that were along edges not located on navigational channels.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-2.52	-0.17	-0.07	-0.14	-0.02	2.57	0.38	0.02	247

Table 6.6. Descriptive statistics for the per point differences in average annual marsh edge change for points that were along edges not located on navigational channels. The preliminary calculation subtracted the average annual marsh edge changes 1926 – 1966 from the average annual marsh edge changes for the 1966 - 2007 interval. Thus, for each point, a negative result indicated greater loss or smaller gain during the 1966 – 2007 time interval and a positive result indicated a greater gain or smaller loss during the 1966 – 2007 time interval.

Min	1st Qu	Median	Mean	3rd Qu	Max.	SD	SE	n
-2.65	-0.12	0.00	0.00	0.01	3.55	0.70	0.04	247

Figure 6.2. The frequency distribution of per-point differences between the marsh edges change for the time interval 1966 – 2007 and the time interval 1926 – 1966 for natural edges located along navigational channels shows little difference in the average annual losses between the two time periods.



I extracted a subset of points from the entire set of points that contained natural edges and did not include null values for the marsh edge changes for either the 1926 – 1966 time interval or the 1966 – 2007 time interval. This resulted in an even pairing of data points that allowed a paired t-test. Paired t-tests were done for natural edges of both the navigational channels and the waterways that were not navigational channels.

The paired t-tests show a distinct difference between marsh edges changed between navigational channels and non-channel areas. There was a significant increase in marsh edge loss for navigational channels in the 1966 – 2007 time interval (Table 6.7). The difference in the average annual loss for points along marsh edges that were not located along navigational channels was not significant for these two time periods (Table 6.7). When comparing navigational channel classifications with each other, there were insufficient degrees freedom for significant tests, but higher levels of variation were found for high boat use categories (Fig. 6.3).

Figure 6.3. A box plot comparing navigational channel edge change between the 1926-1966 interval and the 1966 – 2007 interval (Channel Classifications A2-6 to E2-6 represent 1926 -1966 values for boat use classifications A-E as described in Table 3.1. Channel Classifications A6-0 to E6-0 represent the same boat use classifications for the 1966-2007 time interval). The box and whisker plots show the group median values (heavy line), the upper and lower quartiles, and the whiskers represent the range. Notches show significant differences between groups if the notches do not overlap.

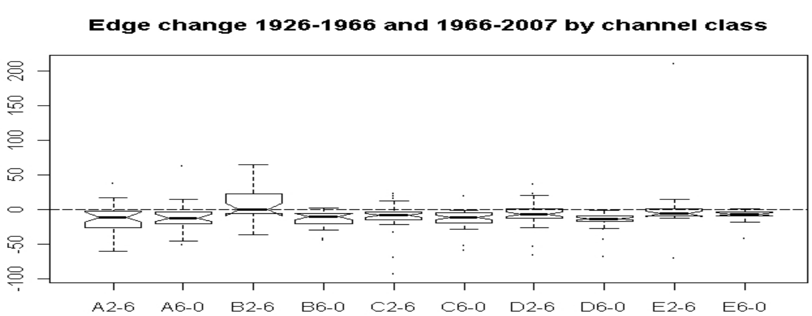


Table 6.7. Results form paired one-sided t-test between 1926-1966 and 1966-2007 paired by point. Points along navigational channels lost significantly more edge 1966 – 2007

Category	t	df	P	Difference in Annual Change
Navigational Channels	3.08	146	0.001	0.198 m
Not Navigational Channel	0.07	246	0.470	0.003

Conclusion:

Navigational channels with natural edges changed at a significantly different rate from 1966-2007 than 1926-1966. From 1966-2007 more marsh edge was lost, and was lost at a more rapid annual rate. There was no comparable difference in edge change rate for edges that were not located along navigational channels.

FACTORS AFFECTING MARSH EDGE CHANGE FOR NATURAL EDGES OF NAVIGATION CHANNELS AND NON-NAVIGATIONAL CHANNELS COMPARING THE TIME INTERVALS 1926-1966 AND 1966-2007

MANOVA was used to test for differences between marsh edge changes during the 1926- 1966 time interval and the 1966 - 2007 time interval using per point average annual changes for these two periods as the response variables and the reduced variable set (from Chapter 5) as the independent variables. MANOVA was also used to analyze the change for marsh edge measurement points on natural edges along navigational channels and again for points along natural edges that were not along navigational channels.

MANOVA of navigational channels.

I extracted from the entire data set a subset of marsh change points that only contained points from natural marsh edges (not channelized edges) that were along a navigational channel. For this series of tests, I used the variables Minimum Fetch (minimum distance to land on an opposite shore in 1966, = CnlWth66Mr), Peak Local Flow Rate (FlowRt), Distance to Borrow Pit (PitDistMr), Channel Unlimited Distance (ChnlUnMtr), Particulate Organics (Partico) and Channel Classification (Channel).

The subset only included points where the variables for Marsh Edge Change for 1926-1966 time interval, Marsh Edge Change 1966-2007 time interval, Channel Flow Rate, Distance to Borrow Pit and Minimum Fetch, did not contain null values. The edges along navigational channels showed differences in responses between the two time intervals (Table 6.9). For the time interval 1966 – 2007, the Peak Local Flow Rate, Distance to Borrow Pit, and Minimum Fetch were significant, and only Peak Local Flow was significant for the time interval 1926 – 1966 (Table 6.8; Table 6.10). The regression of Edge Change 1926 - 1966 for points along navigational channels and Peak Local Flow Rate was highly significant, with an R^2 of 0.20 (Table 6.11). A scatterplot of Edge Change 1926 - 1966 for points along navigational channels as a function of Peak Local Flow Rate shows a difference between areas of marsh accretion and areas of net erosion in relation to the Peak Local Flow Rate (Fig. 6.4). For navigational channels, the 1926 – 1966 time interval showed no effect of Distance to Borrow Pit (Table 6.12; Fig. 6.5), unlike the 1966 – 2007 time interval, where this variable was significant (Table 5.34;

Fig. 5.35).

Table 6.8. Summary of the MANOVA for the effects of Navigational Channel on marsh loss from 1926-1966 and from 1966-2007. Peak Local Flow Rate and Minimum Fetch had a greater impact from 1966-2007 and Distance to Borrow Pit had a greater impact 1926-1966.

	DF	Sum Sq	Mean Sq	F value	Pr(>F)
1926 - 1966	115				
Peak Local Flow Rate	1	47.68	47.68	5.45	0.021
Distance to Borrow Pit	1	0.99	0.99	0.11	0.738
Minimum Fetch	1	0	0	0	0.999
Particulate Organics	1	0	0	0	0.747
Channel Unlimited Distance	1	11.93	11.926	1.363	0.245
Channel Classifications	4	94.13	23.531	2.690	0.035
1966 - 2007	115				
Peak Local Flow Rate	1	44.8	44.801	12.497	0.001
Distance to Borrow Pit	1	42.30.	42.297	11.799	0.001
Minimum Fetch	1	16.63	16.627	4.638	0.033
Particulate Organics	1	0	5.903	1.647	0.202
Channel Unlimited Distance	1	0	10.549	2.943	0.089
Channel Classifications	4	18.97	4.743	1.323	0.266

Table 6.9. Summary of the MANOVA testing the comparative influence of Navigational Channels from 1926-1966 and 1966-2007. Peak Local Flow Rate and Minimum Fetch had a significantly greater impact in the later time period and the effect of Distance to Borrow Pit had a greater impact in the earlier time period.

	DF	ANOVA p>F	Pillai	Wilks	H-L
1926-1966 1966-2007	115				
Peak Local Flow Rate	1	< 0.001	< 0.001	< 0.001	< 0.001
Distance to Borrow Pit	1	0.004	0.004	0.004	0.004
Minimum Fetch	1	0.103	0.103	0.103	0.103
Particulate Organics	1	0.436	0.436	0.436	0.436
Channel Unlimited Distance	1	0.100	0.100	0.100	0.100
Channel Classification	1	0.040	0.040	0.037	0.035

Table 6.10. Summary of MANOVA results showing how marsh edges along navigational channel were affected by different variables from 1926-1966 and from 1966-2007. Peak Local Flow Rate, Minimum Fetch, and Distance to Borrow Pit had greater effects from 1966-2007. The variables that were not significant in the previous test (Table 5.8; Table 5.9), Channel Unlimited Distance, Particulate Organics and Channel Classifications, were dropped from this analysis. The DF and ANOVA are within years, the Pillai, Wilks, and Hotelling-Lawley tests are between time intervals.

	DF	ANOVA p>F	Pillai	Wilks	H-L
1926-1966	134				
Peak Local Flow Rate	1	0.221	0.005	0.005	0.005
Distance to Borrow Pit	1	0.9	0.090	0.090	0.090
Minimum Fetch	1	0.991	0.064	0.064	0.064
1966-2007	134				
Peak Local Flow Rate	1	0.002			
Distance to Borrow Pit	1	0.028			
Minimum Fetch	1	0.012			

Table 6.11. The regression table of the effects of Peak Local Flow Rate on LP transformed change for the edge of marshes along navigational channels from 1926-1966.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-4.293	0.813	-5.283	< 0.001
Peak Local Flow Rate	-0.539		0.360	-1.496	0.137

Multiple R-squared: .016, Adjusted R-squared: 0.01

Figure 6.4. A scatterplot of the effects of Peak Local Flow Rate on LP transformed change in the edge of marshes from 1926-1966 for edges along navigational channels.

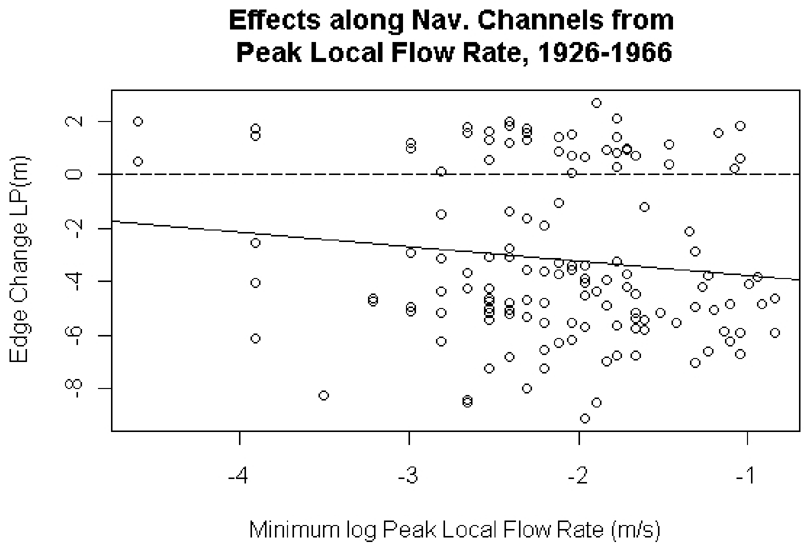
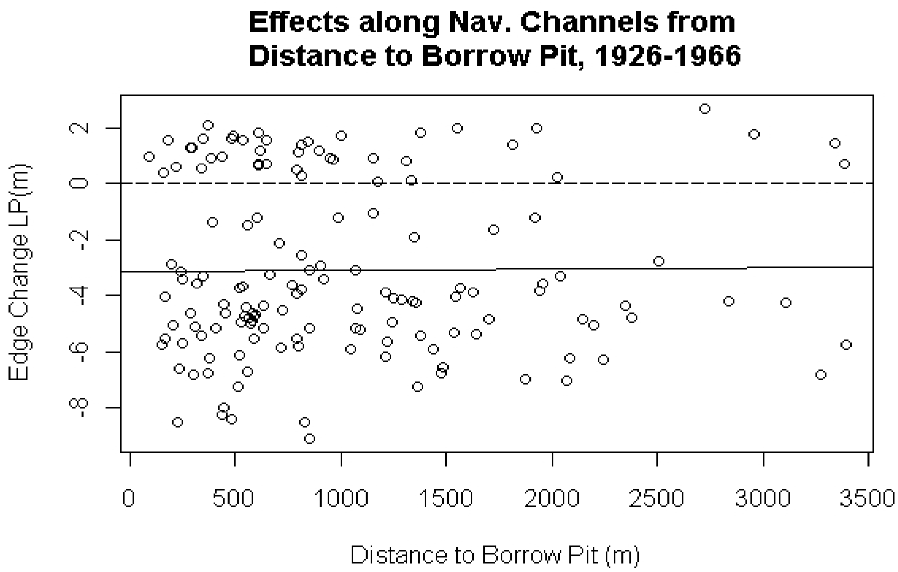


Table 6.12. The regression table of the effects of Distance to Borrow Pit on LP transformed change in the edge of marshes from 1926-1966 for edges along navigational channels.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-3.11	0.428	-7.26	< 0.001
Distance to Borrow Pit	0.000		0.000	0.1	0.92

Multiple R-squared: 0.00, Adjusted R-squared: 0.00

Figure 6.5. A scatterplot of the effects of Distance to Borrow Pit on LP transformed change in the edge of marshes from 1926-1966 for edges along navigational channels.



MANOVA of waterways that were not navigational channels.

The results of the MANOVA showed there was little difference between the time intervals 1926 – 1966 and 1966 – 2007 for Edge Change on edges that were not along navigational channels. Minimum Fetch was the only significant independent variable for both time intervals (Table 6.13; Table 6.14; Table 6.16). The regression of the correlation between Minimum Fetch and Edge Change for marsh not along navigational channels from 1926 – 1966 was significant with an R^2 of 0.2 (Table 6.15). The scatterplot shows that there was a difference between locations where marsh was accreting and where marsh was eroding (Fig. 6.6).

Table 6.13. The results of the MANOVA for the average annual change in the marsh edge for the two time periods 1926-1966, and 1966-2007 for areas that were not on navigational channels. Peak Local Flow Rate and Minimum Fetch were more important 1966-2007 and Distance to Borrow Pit was significant from 1926-1966. Channel Unlimited Distance and Particulate Organics were found to be not significant and were dropped from the analysis. The DF and ANOVA results presented are for within time intervals; the Pillai, Wilks, and Hotelling-Lawley tests are between time intervals.

	DF	ANOVA p>F	Pillai	Wilks	H-L
1926-1966	79				
Peak Local Flow Rate	1	0.214	0.462	0.462	0.462
Distance to Borrow Pit	1	0.320	0.453	0.453	0.453
Minimum Fetch	1	<< 0.001	<< 0.001	<< 0.001	<< 0.001
Particulate Organics	1	0.464	0.000	0.668	0.668
Channel Unlimited Distance	1	0.472	0.396	0.396	0.396
1966-2007	79				
Peak Local Flow Rate	1	0.958			
Distance to Borrow Pit	1	0.445			
Minimum Fetch	1	< 0.001			
Particulate Organics	1	0.595			
Channel Unlimited Distance	1	0.244			

Table 6.14. Results of the MANOVA testing the effects variables on marsh loss in non-navigational channels from 1926-1966 and 1966-2007. Only Fetch was significant and only for the early time interval.

	DF	ANOVA p>F	Pillai	Wilks	H-L
1926-1966 Minimum Fetch	204	<< 0.001	<< 0.001	<< 0.001	<< 0.001
1966-2007 Minimum Fetch	204	0.25			

Table 6.15. The regression table of the effects of Fetch on LP transformed change in the edge of marshes from 1926-1966 for edges that were not along navigational channels.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		0.49	0.432	1.13	0.26
Minimum Fetch	-0.579		0.082	-7.08	< 0.001

Multiple R-squared: 0.20, Adjusted R-squared: 0.19

Figure 6.6. Marsh Edge change for points on edges not along navigational channels 1926-1966 as a function of Minimum Fetch.

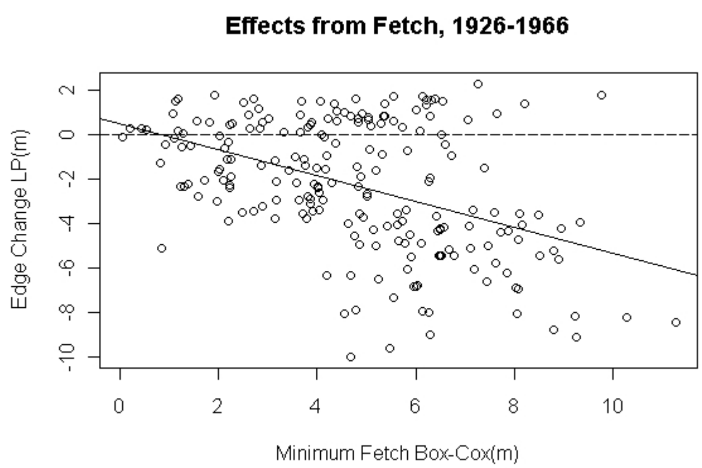


Table 6.16. MANOVA results of the effects of variables on marsh loss for points along edges that were not along navigational channels for 1926-1966 and 1966-2007. The effect of Minimum Fetch was significant for both time intervals.

	DF	Sum Sq	Mean Sq	F value	Pr(>F)
1926 - 1966	200				
Peak Local Flow Rate	1	0.02	0.02	0	0.958
Distance to Borrow Pit	1	4.1	4.1	0.59	0.445
Minimum Fetch	1	347.35	347.35	49.74	<< 0.001
Particulate Organics	1	1.98	1.98	0.28	0.595
Channel Unlimited Distance	1	9.55	9.55	1.37	0.244
1966 - 2007	200				
Peak Local Flow Rate	1	7.1	7.12	1.55	0.214
Distance to Borrow Pit	1	4.56	4.56	1	0.320
Minimum Fetch	1	92.9	92.9	20.28	<< 0.001
Particulate Organics	1	2.47	2.47	0.54	0.464
Channel Unlimited Distance	1	2.38	2.38	0.52	0.472

Marsh gains and losses greater or less than -0.15 m from 1926-1966.

The distinct V shape seen in scatterplots of Minimum Fetch (Fig. 6.6; Fig. 5.3, Fig. 5.11) raised the possibility that there was a difference in how some locations reacted to the mechanisms related to the Minimum Fetch parameter, most likely wave action. There was an apparent inflection at -0.15 m for the 1926 – 1966 time interval and at -1.15 m for the 1966 – 2007 time interval. The mechanisms that might be responsible are not known, but some factor not included in this study appears to drive the loss of marsh edge, even in locations where wave action (represented by Minimum Fetch) is contributing some sediment.

For the 1926-1966 time interval Minimum Fetch explained more of the variation in the change of marsh edges that were not located along navigational channels. There was a significant difference in factors affecting marsh gains versus losses less than 0.15 m (Table 6.17; Fig.6.7), with a R^2 of 0.63. Good results were also achieved for the time interval 1926 – 1966 when comparing the change of marsh edges that showed losses greater than 0.15 m and were not located along navigational channels. In this case, the model with Minimum Fetch, had a R^2 of 0.341 (Table 6.18; Fig. 6.8). Similar results were found for the 1966 – 2007 time interval, but in this case considering points that lost on averages less or more than 1.15 m of marsh. Locations with gains or small losses from 1966 – 2007 showed a positive slope with Minimum Fetch with a R^2 of 0.143 (Table 6.19; Fig. 6.9), and locations loosing more than 1.15 m had a negative slope and a R^2 of 0.152 (Table 6.20; Fig. 6.10). Additional research is needed to explore what the underlying mechanisms might be that could explain these patterns.

Table 6.17. The regression table of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1926-1966 for edges that were not along channels and either gained marsh or lost less than 0.15 m for the time interval.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		0.38	0.144	2.66	0.010
Minimum Fetch	0.114		0.031	3.65	< 0.001

Multiple R-squared: 0.063,   Adjusted R-squared: 0.052

Figure 6.7. The scatterplot of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1926-1966 for edges that were not along channels and either gained marsh or lost less than 0.15 m for the time interval.

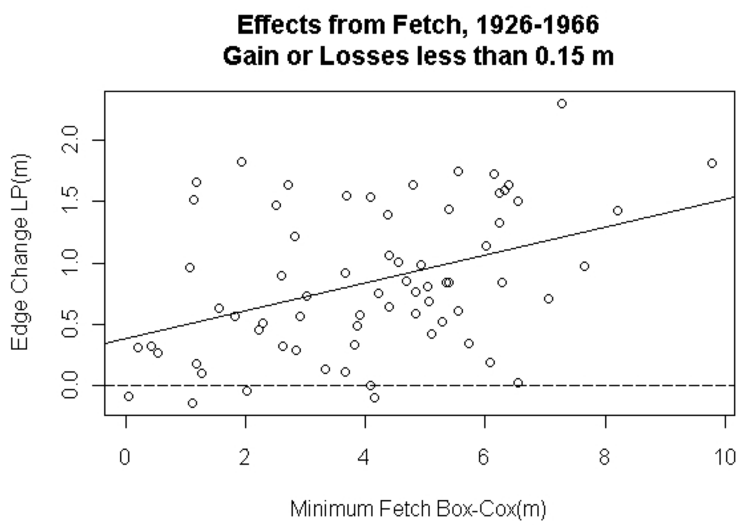


Table 6.18. The regression table of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1926-1966 for edges that were not along channels lost more than 0.15 m for the time interval.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-0.79	0.400	-1.97	0.05
Minimum Fetch	-0.598		0.072	-8.35	<< 0.001

Multiple R-squared: 0.341,   Adjusted R-squared:0.336

Figure 6.8. The scatterplot of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1926-1966 for edges that were not along channels and lost more than 0.15 m for the time interval.

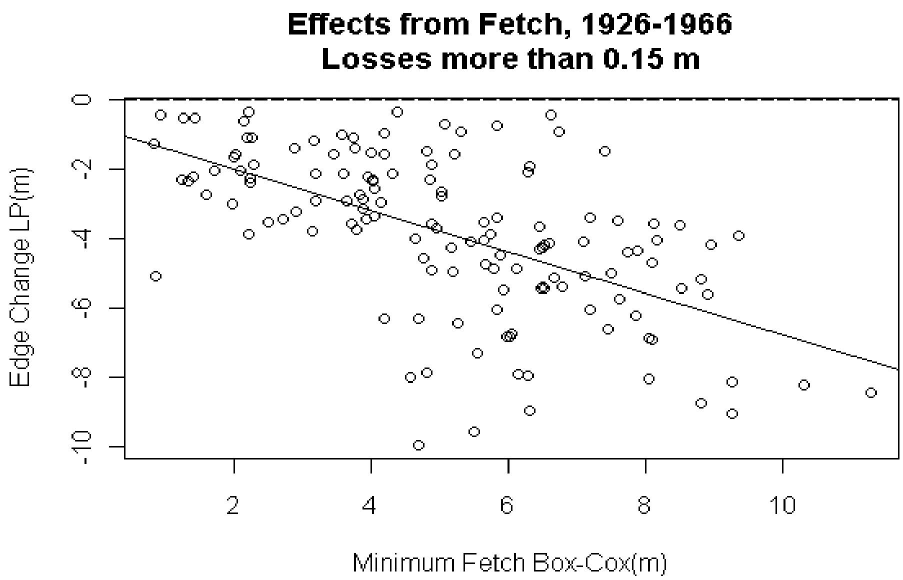


Table 6.19. The regression table of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1966 - 2007 for edges that were not along channels and either gained marsh or lost less than 1.15 m for the time interval.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-0.97	0.22	-4.51	<< 0.001
Minimum Fetch	0.19		0.05	3.68	< 0.001

Multiple R-squared: 0.143, Adjusted R-squared: 0.133

Figure 6.9. The scatterplot of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1966 - 2007 for edges that were not along channels and either gained marsh or lost less than 1.15 m for the time interval.

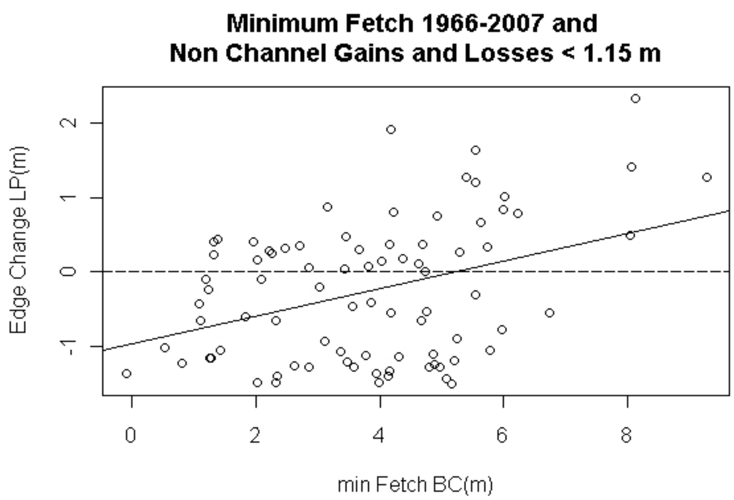
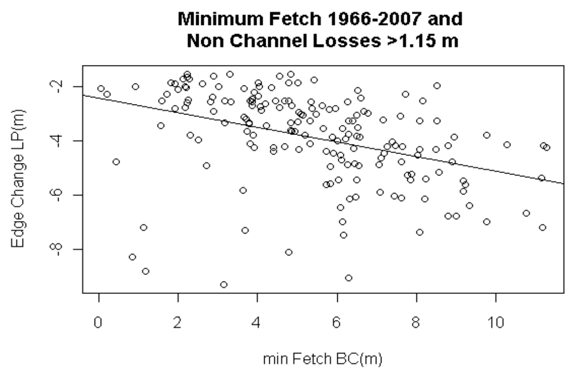


Table 6.20. The regression table of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1966 - 2007 for edges that were not along channels lost more than 1.15 m for the time interval.

	Slope	Intercept	Std. Error	t value	Pr(>|t|)
		-2.42	0.29	-8.46	<< 0.001
Minimum Fetch	-0.27		0.05	-5.58	<< 0.001

Multiple R-squared: 0.152, Adjusted R-squared:0.147

Figure 6.10. The scatterplot of the effects of Minimum Fetch on LP transformed Marsh Edge Change from 1966 - 2007 for edges that were not along channels and lost more than 1.15 m for the time interval.



Conclusion.

There are several differences in the factors affecting marsh loss between the two time intervals 1926 – 1966 and 1966 – 2007. For navigational channels that were not channelized, Peak Local Tidal Flow Rate was more important than other factors in both the 1926 – 1966 and the 1966 - 2007 time intervals. Minimum Fetch had a significant effect on the change of natural marsh edges that were not along navigational channels for both the 1926 – 1966 and the 1966 - 2007 time intervals. Distance to Borrow Pits may not have been a significant factor during the 1926 - 1966 time interval because many of the barrow pits were dug late during this time interval, thus would only have an impact for a short period of time. The re-suspension of sediment caused by the much larger number of power craft present during the 1966 - 2007 time interval is another likely factor affecting the difference in the significance of Distance to Borrow Pit as a factor between the two time intervals for areas along navigational channels.

Chapter 7 Discussion

The loss of salt marshes and similar wetlands is a global problem (Zedler and Kercher 2005). Salt marshes provide a variety of ecosystem services, from protecting coastlines, to enhancing fisheries, to critical ecosystem functions, including denitrification, and water filtration (Costanza 2008, Feagin 2010, Barber et al. 2011). Given these important services, the protection of salt marshes is considered essential for ecosystem health, conservation of shorelines and the species that inhabit marshes, and protection of human structures and cities from storm damage. A number of different mechanisms have been proposed for the accelerating loss of salt marshes (Mendelssohn and McKee 1988, McKee et al. 2004, Edwards et al., 2005, Silliman et al. 2005, Mendelssohn et al. 2006, Alber et al. 2008, Ogden and Alber 2008, Gedan et al. 2011). Given the broad array of types of marshes and habitats in which they occur, from near pristine locales to highly urbanized areas, it is unlikely that any single factor is the sole cause of marsh loss throughout the geographic range of *Spartina*. I tested the effects of a range of hypotheses that are though to be generally important, including the effects of channelization, or direct cutting of channels through the marsh, the effects of boat traffic, storms, local hydrographic conditions and increased nutrient pollution.

I found strong support for the hypothesis that channelized edges of marshes would show more erosion than those that were not channelized. Over the past 90 years, the marshes in the Hempstead Bay have retreated on average 17.8 m. Although some areas of marshlands have grown, the vast majority of areas have lost marsh and, in some cases, these losses have been extensive. Overall, channelization of the marshes, cutting large channels through otherwise intact marsh, was the single greatest factor driving salt marsh edge recession over the past 90 years, and has had a lasting effect. Unfortunately, the effects of channelization did not stop or slow over time. Instead, the continued impact from this form of disturbance has caused the steady loss of marshlands, even after 50 to 80 years past the initial damage. In addition, there has been no detectable reduction of the rate of marsh edge recession due to channelization over the time interval of this study, 1926 – 2007 (Fig 4.6). Within the study site, 16.5 ha of marsh were lost along these channelized areas between 1983 and 2004. This represented a 15.2% of the 108.6 hectares of marsh lost in the Hempstead Bay during this period, even though these channelized areas include less than 7% of the total edge of marshland. Marsh edges that were channelized retreated an average of 27.5 m over the 51-year period from 1966 and 2007, compared to a mean retreat of 8.4 m for all other areas of the marsh. The channelized edges, therefore, lost marsh from the edge at a rate 3.25 times faster than that seen for all other types together. Most of channelized areas are navigation channels. Prior studies of channelized marsh have focused on the effects of vessel traffic rather than the channelizing of the marsh per se (e.g., Price 2008, Davis et al. 2009), thus it is difficult to determine if this high rate of marsh loss due to lasting effects of deliberate cuts through the marsh is occurring elsewhere.

The practice of dredging channels directly through marshlands, or into the edges of marshland, is no longer allowed in New York State, thus no new damage of this type is expected in future years. The results of this study, however, indicate that simple legal protection from additional damage to marshes through cutting channels was insufficient for stemming marsh loss from the cuts that were made decades ago. To slow or stop the continued loss of marsh areas that have been channelized will likely require some sort of stabilization or restoration efforts. However, it is clear that any marsh management should prevent this type of damage to marshes elsewhere.

Because channelization of the marsh has such an overwhelming effect, to detect the effects of navigational boat traffic I contrasted navigational channels that were not channelized (natural navigational channels) with other natural areas of the marsh that were not channelized or exposed to boater traffic. There were significant differences between the rates of marsh edge retreat along these navigational channels and rates of retreat for edges that were not along channels. Different factors had different impacts in these two types of marsh area. Along navigational channels with natural edges, an average of 13.8 m of marsh was lost from the edge from 1966 to 2007, compared to 5.3 m for areas of marsh not on navigational channels over the same time period.

Vessel passage can cause erosion by creating short period waves and changes in currents that resuspend sediment (Dovora and Moore 1997, Hofmann et al. 2008, Davis et al. 2009). However, there are many differences between areas that are navigational channels and those that are not, especially hydrographic factors, which makes it difficult to determine if it is boat traffic that is primarily responsible for this difference. Other studies that have considered some confounding factors have concluded that boat use was a comparatively minor factor in the loss of marshlands (Zabawa and Ostrom 1980).

The strongest evidence for the relative importance of increased boat use on marsh loss is the difference in marsh loss between 1926-1966 and 1966-2007. After 1966 boat traffic increased markedly and the region around the estuary became almost totally urbanized. The rate of marsh loss along the edges of navigational channels was significantly greater in the later time period, when the size of boats and the frequency of boater traffic increased. However, marsh loss was not different between these two time periods for non-channel areas, indicating little direct influence of urbanization. Other factors such as tidal currents, storms, and wind-driven waves are likely to have been very similar over these two time periods. Consequently the increase in boat traffic is likely responsible for the greater loss seen along navigational channels.

Differences in the effects of hydrographic factors were also important for the differences seen between non-navigational channel areas and areas of marsh along navigational channels. Strong tidal currents can drive patterns of erosion and redeposition (meanders) in different parts of the marsh (Biggs 1982, Kearney et al. 1988, Klienhanz et al. 2008). High tidal flow rates can resuspend and transport sediment away from the marsh, speeding erosion (Wang 2002, Larson et al. 2009). Areas of high current flow due to tidal currents were primarily along navigational channels, and within those channels were areas of greater marsh loss for navigational channels. However, tidal currents were not an important factor for marsh loss in non-channel areas. Deep channels maintained by tidal flow were likely selected to be developed for navigational channels because they would require minimal dredging (Fig. A.5), making it difficult to separate these two factors.

Management efforts focused at reducing wakes from larger boats in channels, especially those where the boats travel close to marsh edges, will be the most effective at minimizing the impact of boat traffic. However, such management practices will be less effective at reducing marsh loss in an area where high water flow rates or wind-driven waves (areas with a large fetch in the direction of prevailing winds) are the dominant erosive factors.

For marsh not on navigational channels, both storms and factors associated with wind-driven waves affected marsh loss, however, these factors were not important for marsh along navigational channels. Away from navigational channels, the effect of storm driven waves was strong for both the 1926 – 1966 and the 1966 – 2007 time interval. The minimum fetch (the width of the channel) affects the size of normal wind driven waves and shoreline bathymetry (reflected in the truncated tidal range) both correlated with marsh edge loss. This wind driven waves along areas with shallow bathymetry, even for areas that do not face the worst storm driven waves, are important for increasing marsh loss in areas away from navigational channels. Other studies of marshlands have found similar effects of factors associated with wind-driven waves and marsh and mudflat erosion (Phillips 1986, Downs et al. 1994,, Pye 1995 Day et al. 1998, Schwimmer 2001, Fagherazzi and Wiberg 2009).

The role of sediment supply in affecting marsh loss was most important along navigational channels (Fig. 5.37). Marsh along navigational channels was lost at a great rate when in close proximity to borrow pits, but no similar correlation was found for areas of the marsh distant from navigational boat traffic. Sediment is resuspended by the turbulence and wakes from boats. This resuspended sediment will likely resettle on the marsh unless it is removed from the system by a near by sink. Borrow pits may provide such a sink. Sediment that settles into a borrow pit is unlikely to be resuspended (Renfro et al. 2011). The effect of borrow pits on marsh loss was greatest for marsh along navigational channels that were within 500 m of a borrow pit (Fig. 5.43).

It is surprising that nutrient loading did not have an affect on marsh loss in this study. Increased nitrogen favors above ground growth over below ground growth in root systems (Levin et al. 1989), including *Spartina* (Turner 2011), and may destabilize the marsh (Turner 2011). However, enhanced above ground growth will simultaneously slow water flow (Neumeir 2005) and increase sediment accumulation (Teal 2001, Morris et al. 2005), which can help stabilize the marsh grass. Turner (2009) suggested that marshes receiving high nutrient loading from the Caemarvon Diversion in the Mississippi watershed in Louisiana were weakened, resulting in sever damage due to hurricanes. The amount of nutrient loading seen in the Caemarvon Diversion was close to the nutrient concentrations in this study (Lane et al. 1999), but about 50% lower than those hypothesized to cause marsh loss in Jamaica Bay (Fitzpatrick 2001, Kolker et al. 2005). The levels of water-borne nitrate in West Bay of the SSER are close to the concentration where growth in *Spartina alterniflora* has been thought to saturate (Drake et al. 2009, Mozdzar et al. 2011), but the concentrations of ammonia are lower than the saturation level found by Mozdzar et al. (2011). The nutrient concentrations in the East Bay are lower than those in the West Bay. The switch from private septic systems to sewage treatment plants in the area around Hempstead Bay occurred in the late 1960s and early 1970s as the local population grew. However, there was no change in the rate of marsh edge recession for areas away from navigational channels from before sewage treatment plants were installed (1926-1966) and after (1966 - 2007).

The lack of a detectable affect of nutrient concentrations and marsh loss remains an open question, especially because predictions based on prior studies suggest that such a relationship should occur (Kolker 2005, Kolker et al. 2005, Turner 2011). In this case, the expected loss of marsh may be balanced by increased accretion due to the stimulated growth expected from nutrient additions (Morris et al. 2005, Mudd et al. 2010). Contaminants or non-nutrient materials in effluent may be more important and may differ between this study area and the sites of other studies, masking any effects of nutrient loading on marsh loss. Alternatively, the range of nutrient concentrations within Hempstead Bay may not cover a sufficient range to detect differences, or the negative impacts seen elsewhere may asymptote at levels of nutrient loading near or below those seen in this study. These results of this study suggest that controlling nutrient pollution alone will not have a large impact on protecting salt marshes from loss. More research is clearly needed to determine the effects of these high levels of nutrients on marshes, as well as other compounds associated with the nutrient sources and sewerage outfalls.

Unfortunately, few published studies of salt marshes include local water-borne nutrient concentrations, and most water quality studies report total nutrient loading per year or as flux into estuaries, making it difficult to compare this study with other studies. Reports focused on the effects of nutrients in an estuary typically report the amount of introduced nutrient over a time period, but not the concentrations that the organisms experience. This distinction is important; local dilution, denitrification, or absorption rates can decouple input rates or totals from observed concentrations and effects (Fisher et al. 1988, Cowen and Boynton 1996, McKee et al. 2011). In one of the few studies that incorporated both, two estuaries water-borne nutrient concentrations and biological impacts were similar in two estuaries that differed in nutrient inputs (Castro et al. 2009).

Among estuaries where water-borne nutrient concentrations have been reported, Hempstead Bay tends to have moderately high concentrations (Boynton and Kemp 2008), but not as high as those seen in Jamaica Bay (Fitzpatrick 2001), which is losing *Spartina* marsh at a high rate (Gateway National Recreational Area 2007). Nutrient concentrations in Hempstead Bay are also lower than those seen in the heavily impacted southern portions of San Francisco Bay, where *Spartina* has become an important pest species that is spreading at a rapid rate (Callaway and Josselyn 1992, Tyler et al. 2007, Sloop et al. 2011). Maximum nitrate levels in Hempstead Bay are < 70% of those in Chesapeake Bay (Fisher et al. 1988, Cowan and Boynton 1996), 50% of those found in the Hudson River (Fisher et al. 1988), and 20%-30% of those found in Delaware Bay (Fisher et al. 1988).

In Narragansett Bay, Nixon and Oviatt (1973) showed that the biomass of *S. alternilfora* increased with nutrients over a range nutrient of values 20% less than the lowest seen in Hempstead Bay. Later studies in Narragansett Bay found a similar positive relationship between increased annual total nutrient loadings and productivity for tall form *S. alternilfora*, and declines in species diversity in the high marsh and biomass of *S. patens* (Wigand et al. 2003, Wigand 2008). Unfortunately, a direct comparison with the present study is not possible because Nixon and Oviatt (1973) reported nutrient loading rates instead of nutrient concentrations in the water column.

In addition to the effects of added nutrients on the growth of *Spartina,* it has been suggested that eutrification may increase sediment pore water concentrations of H_2S, a phytotoxin that can damage the salt marsh (Kolker 2005, Kolker et al. 2005). Kolker (2005) tied sediment pore water concentrations of H_2S in Jamaica Bay to excess organic mater deposition in the marsh, but did not find a direct link between nutrients and organic matter deposition. It is unclear whether the excess organic matter was due to high algal abundances, or the accumulation of organic mater from historic raw and primary sewage outfalls or overflow from present day combined sewer outfalls, where storm drains and sewer systems are combined, resulting in raw sewage traveling through the system when rainfall is high. The high concentration of H_2S observed in

Jamaica Bay can also be related to sediment drainage and time submerged (Koch and Mendelssohn 1989, Bradley and Morris 1990, Ogburn and Alber 2006). It is not clear whether it H_2S causal, incidental, or symptomatic of marsh subsidence from other causes (Alber et al. 2008), and it is possible that the role of H_2S may vary with location and conditions.

Other factors not directly included in this study have been recently implicated in salt marsh loss, and some are seen to be particularly important in eutrophic systems, especially factors that increase grazers on *Spartina*. For example, the role of consumers (top down controls) of *Spartina* is predicted to become greater in eutrophic systems because of the higher nutritional value of marsh grass grown with higher nutrient availability (Bertness et al. 2008, Sala et al. 2008). Similarly, systems with heavy fishing pressure that removes top predators can induce a trophic cascade, resulting in the loss of marsh grass (Holdredge et al. 2008).

Several studies have implicated increases in herbivory in the loss of *Spartina* (Silliman and Bertness 2002, Silliman and Bortolus 2003, Gustafson et al. 2006, Alberti et al. 2011). In some cases predator populations can also respond to increased grazer abundances (Silliman et al. 2004, Cardoni et al. 2011), reducing the potential importance of herbivory as a main driver of marsh edge recession (Kiehn and Morris 2009).

The purple marsh crab, *Sesarma reticulatum,* is common on many shores along the Atlantic coast of North America, including Hempstead Bay. Unlike most crabs, which are predators, *Sesarma* feeds directly on *S. alterniflora.* The loss of predator control on *Sesarma* due to anthropogenic impacts has been associated with the loss of salt marsh in several areas (Holdredge et al. 2008). Outbreaks of *Sesarma* on Cape Cod were attributed to the loss of two important predators due to over fishing, blackfish (*Tautoga onitus*) and blue claw crabs (*Callinectes sapidus*), and the loss of black-crowned night-heron (*Nycticorax nycticorax*) due to habitat destruction (Holdredge et al. 2008).

While *Sesarma* is common in the SSER, damage to the marsh by *Sesarma* is typically minor, and the large swaths of grazing damage seen by Holdredge et al. (2008) and Bertness et al. (2009) are uncommon. Although located in the New York City metropolitan area, one of the most urbanized locations in the country, viable yellow-crowned and black crowned night heron rookeries are found within Hempstead Bay (McGowan and Corwin 2008). Yellow-crowned night heron are regularly seen preying on *Sesarma* during the spring at the Oceanside Marine Nature Study Area, along the northern edge of the estuary (Farina, M., Town of Hempstead, Department of Conservation and Waterways, *personal communication*) (Fig A.27). Adult blackfish, large enough to feed on *Sesarma*, typically live in deeper water on the south shore of Long Island (Conover et al. 2005) and are unlikely to be important predators on *Sesarma* in this area. With increased water temperatures, the abundance of blue claw crabs is expected to increase on Long Island. Although the harvest of crab continues, they are still abundant throughout Hempstead Bay (*personal observation*). Diamondback terrapin (*Malaclemys terrapin*) also occur within the study area and are know to feed on *Sesarma* (Tucker et al. 1995).

In this area, terrapin may be another important predator on *Sesarma*, but the food habits of the local terrapin population are not yet well understood (Burke, R.L., Hofstra University, *personal communication*).

Grazing by geese is also reported to be important for some marshes, including those in eutrophic areas (Bertness et al. 2004), particularly grazing by snow geese (Perry et al. 2001). Atlantic brant grazing has been proposed as an important factor affecting marsh die-back in Jamaica Bay (Buckley 2002). Grazing by geese is a problem for restoration sites in Jamaica Bay and other nearby locations (*personal observation*), and grubbing (feeding on underground rhizomes) by snow geese has been observed in local marshes (D. Mundey, Jamaica Bay Ecowatchers *personal communication*). However, field observations in Jamaica Bay found that the combined grazing by Canada geese, snow geese, and Atlantic brant on natural stands of *S. alterniflora* did not contribute significantly to marsh loss in Jamaica Bay (Riepe and Mundy 2002). A pilot exclusion experiment within the SSER that excluded Canada goose and Atlantic brant from foraging on natural stands of *S. alterniflora* showed no effect from grazing (*unpublished data*). Herbivory within the study area seems to be limited to small numbers of *Sesarma* in the low marsh and insects in the high marsh, including *Prokelisia* (Gustafson et al. 2006). The consumer effects observed elsewhere may have responded to the anthropogenically enhanced availability of nutrients (bottom up effects) by increased predation rates (Cardoni et al. 2011), and fishing pressure may not be sufficiently intense to release herbivores from predator control.

Sea level rise has also been implicated in the loss of marshlands (Stumpf 1983, Stevenson et al. 1985, Kearney et al. 1988, Stevenson et al. 2002). If sea-level rise outstrips marsh accretion rates, which depend on sediment deposition, marshlands will be lost (Downs et al. 1994, Wray et al. 1995, Morris et al. 2004). Similarly, other processes that either enhance or interfere with the growth or stability of Spartina and its ability to collect and stabilize the sediment, affect the rates of marsh loss. Over the past 100 years the tidal range in Jamaica Bay has increased (Swanson 2008, Swanson and Wilson 2008), and there is some indication that the tidal range of Hempsted Bay has increased over the past 40 years as well, while the tidal rage at at Sandy Hook, N.J., which is on the open coast, not in a bay, has not (R. Lawrence Swanson, Stony Brook University, *personal communication*). An increased tidal range may causee more frequent flooding of the marsh surface, stressing Spartina, and altered sedimentation patterns and rates. Several studies indicate that the accretion rates of the marsh surface are sufficient to match sea level rise (Kolker 2005, Kirwan and Temmerman 2009, Renfro et al. 2011). Many studies have found improved salt marsh survival and even growth when there is sufficient sediment availability (Shen et al. 2008, Day et al. 2011, Kirwan et al. 2011). In Hempstead Bay, locations where the marsh expanded included Hewlett Harbor (northern West Bay), East Crow Island (southern East Bay), and Meadow Island (a flood shoal inside Jones Inlet). At all of these sites, sediment accumulated in the upper intertidal zone and was then colonized by *Spartina* (*personal observation*, Fig A.25).

Factors that act locally, particularly if they are anthropogenic in origin, are potentially the most important for management efforts to protect salt marshes. These factors can be regulated, augmented, or reversed as needed by local managers for the health of the salt marsh ecosystem. Channelizing the marsh was the single most significant factor responsible for marsh loss. It is fortunate that channelized edge only represents about 7% of the total marsh edge in the SSER. It is clear that all efforts must be made to prevent any additional channelization in this marsh or other marshes. Efforts should also be made to determine how to stabilize areas that have already been channelized to reduce or prevent the continued loss of marsh at a very high rate. Other marshes with cut channels should be monitored to determine if this pattern is generalizable, or what the differences are between the SSER and areas where channelization does not result in continued marsh loss.

The other factors in this study that contributed the most to explaining marsh loss, in addition to channelizing the marsh, were all associated with sediment redistribution. For natural marsh edges not located along navigational channels, the long term trend of slow edge retreat was most affected by wind driven and storm waves. This result was found for both the 1926 – 1966 and 1966 - 2007 time intervals. The loss of protective oyster reefs along marsh margins, which reduce erosion from the edge of the marsh (Meyer et al. 1997), could contribute to marsh loss. In addition, increased sea-level, and reduced stream flow with its associated sediment, are all factors that could contribute to marsh loss due to erosion (Fig A.26). Hurricanes can cause erosion, but are also known to increase the input of sediment, measurably increasing accretion on the marsh surface (McKee and Cherry 2009). Additional efforts should be made to determine the possible roles of these factors, and potential alternative sources of sediment needed by marshes to keep pace with sea level rise.

One possible approach to offset marsh losses is the beneficial use of dredged material that would otherwise be disposed of (The Great Lakes Commission 2010). Sedimentary processes are continuing to fill navigational channels and marinas while marshlands are still in decline, and marshlands may be a significant source of the sediment filling the channels (New York State Department of State 2010). The restoration of wetlands in the Gulf of Mexico coastal region frequently uses dredged material for this purpose (Möller et al. 2001, Edwards and Mills 2005). The use of dredged material is also used as part of the managed realignment of sea defenses in England (Morris et al. 2004). Preventing marsh loss by augmenting natural sedimentation with thin layer placement of dredged material is an approach that holds promise (Ray 2007). Studies of sediment subsidy in marshes have found a number of beneficial responses, including increased inorganic content, reduced sulfide stress, reduced flooding periods, and increased sediment nutrient levels (Mendelssohn and Kuhn 2003). Projects intending to supplement marshland with sediment require careful design and planning (Broome et al. 1990, Shisler 1990). The exact level of sediment supplementation needed will vary with the local tide regime to prevent overfilling the marsh, which reduces the stability and resilience of the marsh (Stagg and Mendelssohn 2011). The mineral composition of the sediment can also influence stability (Crooks and Pye 2000). Perry et al. (2001) reviewed a number of salt marsh reconstructions, and listed several recommendations for success, including the position of the marsh relative to open water (maximum fetch), the control of grazers and nitrogen fertilization.

In light of the importance of salt marshes for wildlife, fisheries and the need to protect the urban shorelines of the Town of Hempstead and the City of Long Beach, and shorelines in general, from storm damage, it is critical that salt marshes are maintained or restored. Some have even suggested that given the importance of this habitat, new marshes should be created to mitigate the loss of services provided due to the loss of natural salt marshes (Möller et al. 1999). Within Hempstead Bay, there was no single cause of salt marsh loss. It is clear that many different factors have been at work and interact to produce the marshes we see now, and the changes in the marsh that continue to occur. The factors driving loss in this marsh may well affect all salt marshes where they occur. Although most areas lost marsh through time, there were locations that experienced a gain of marsh from 1966 – 2007 (Fig. A.21), as well as over the entire 1926 – 2007 time interval (Fig. A.22). New marsh growth is a strong indication that, given some management effort, these marshlands have the potential to spread and maintain the ecosystem services they provide.

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APPENDIX A

Figure A.1. A map of the study location within New York State. The South Shore Estuarine Reserve is indicated by the green lines. The boundary of the area with water sampling is indicated by the blue box. The dashed lines surround the area within which the random points of marsh edge were selected. and marsh edge measurement.

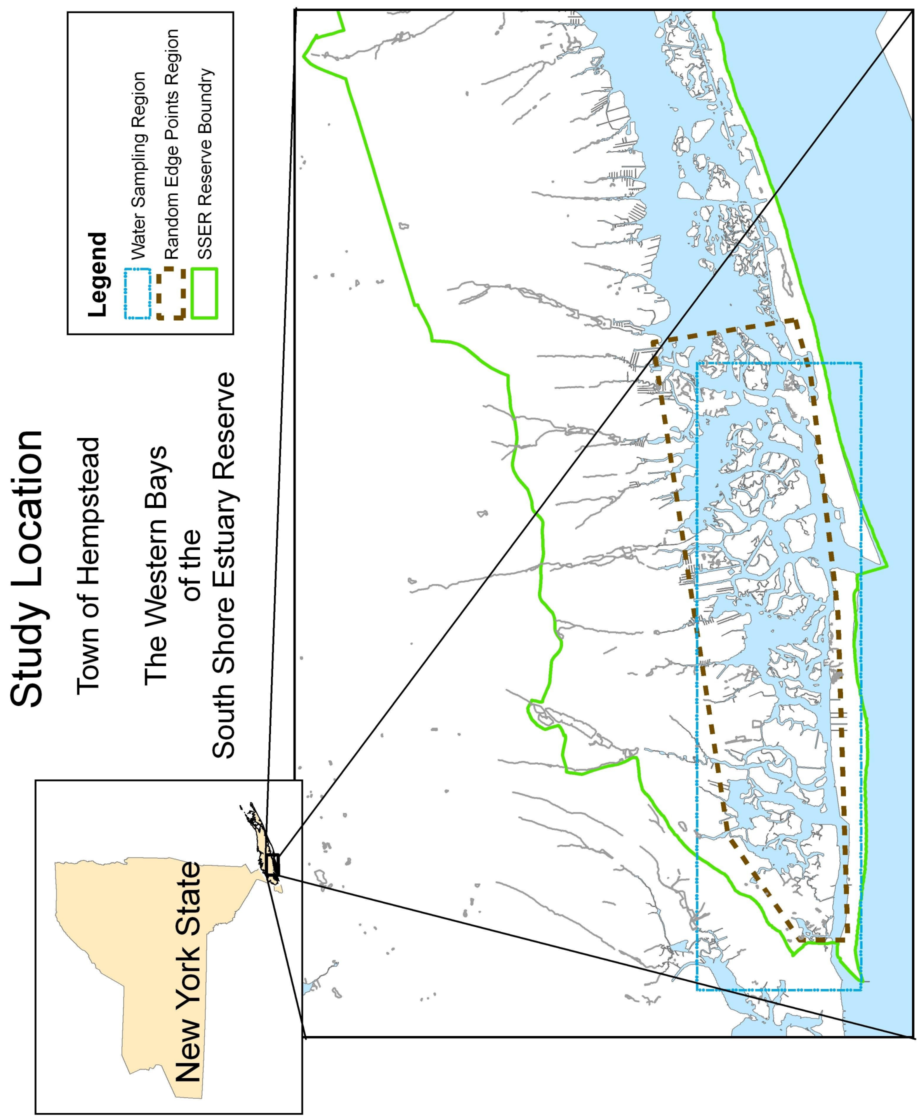

Figure A.2. A view of the study site showing urbanization in the year 2000.

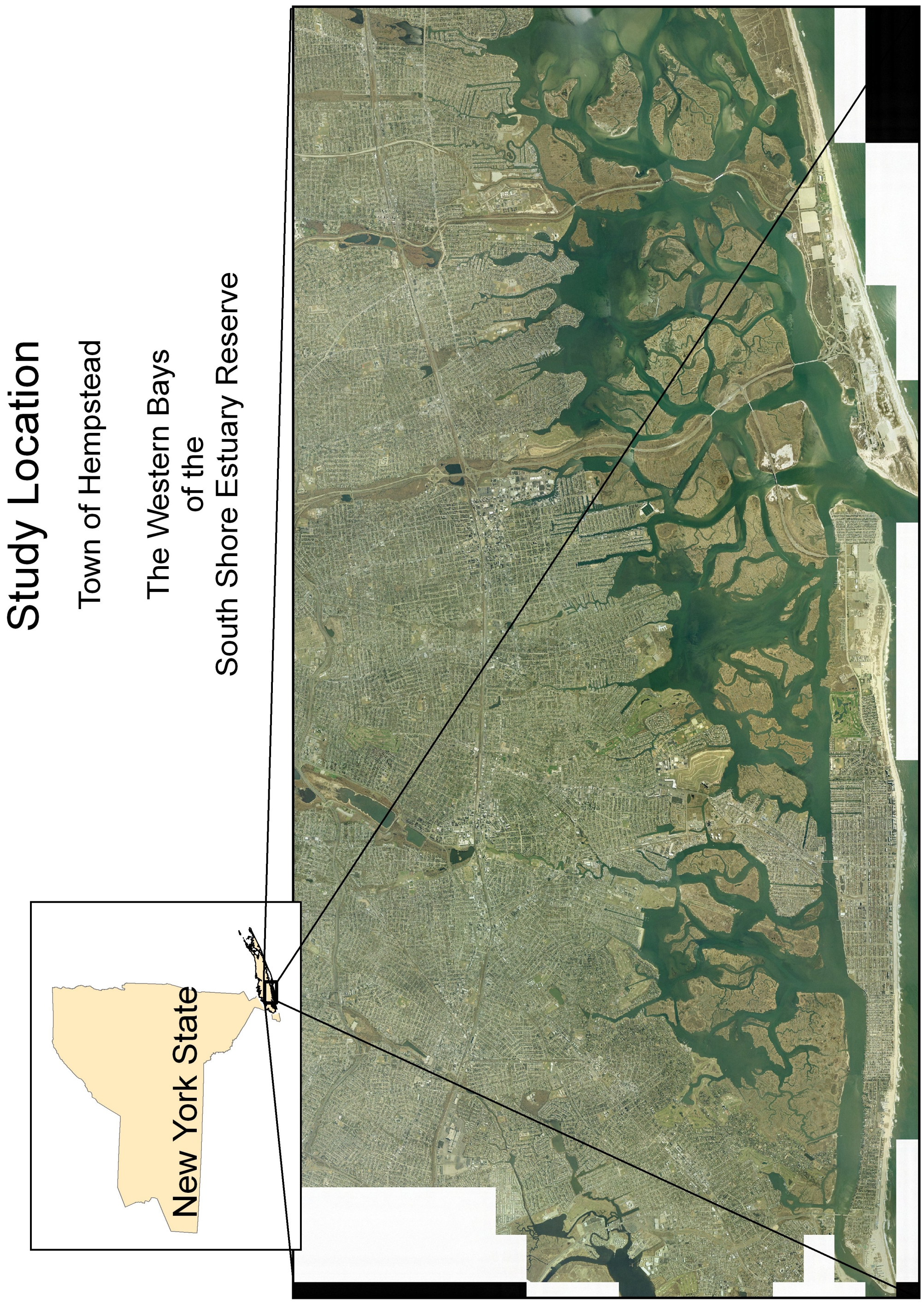

Fig A.3. Portions of the survey map including the study area that was drawn for the US Coast and Geodetic Survey in 1879.

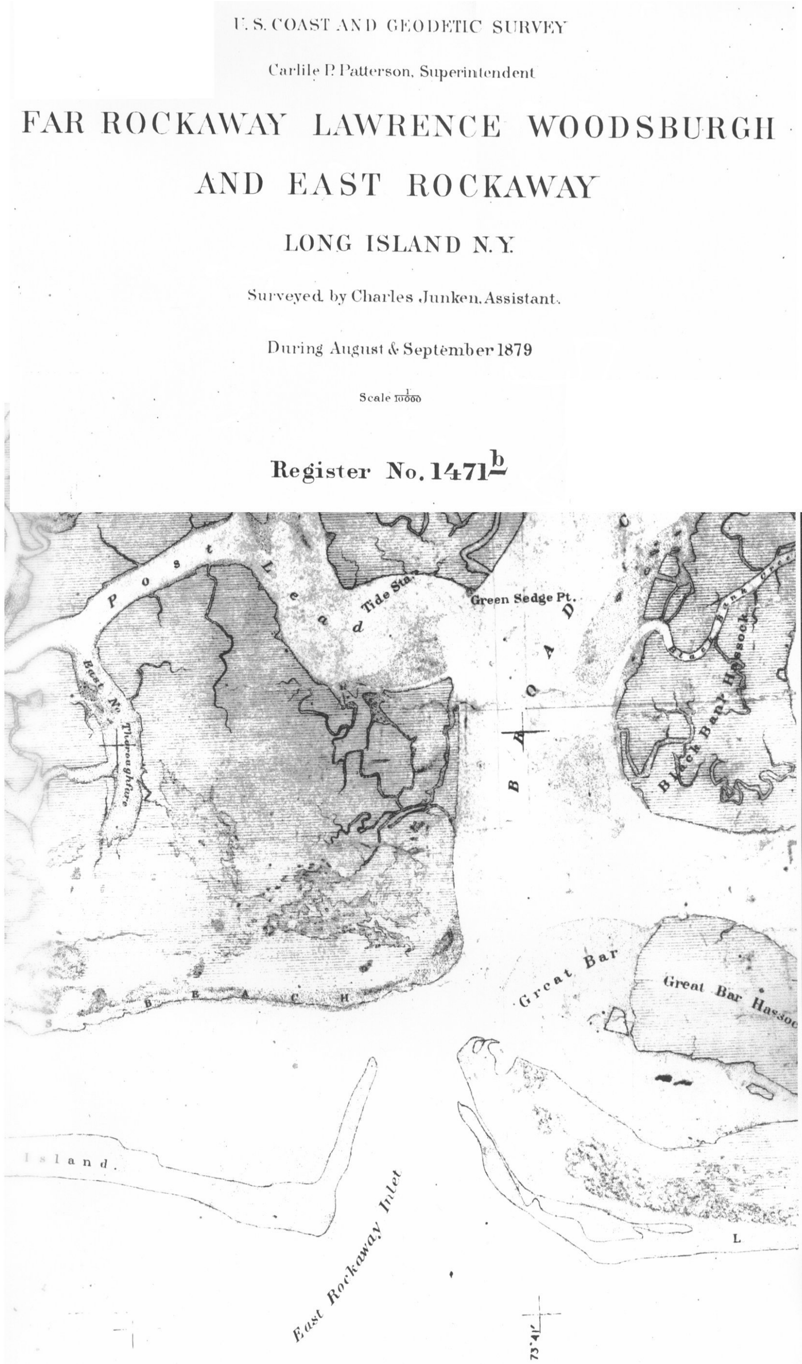

Fig A.4. Aerial photography extracted from an aerial atlas that shows the study site in 1926.

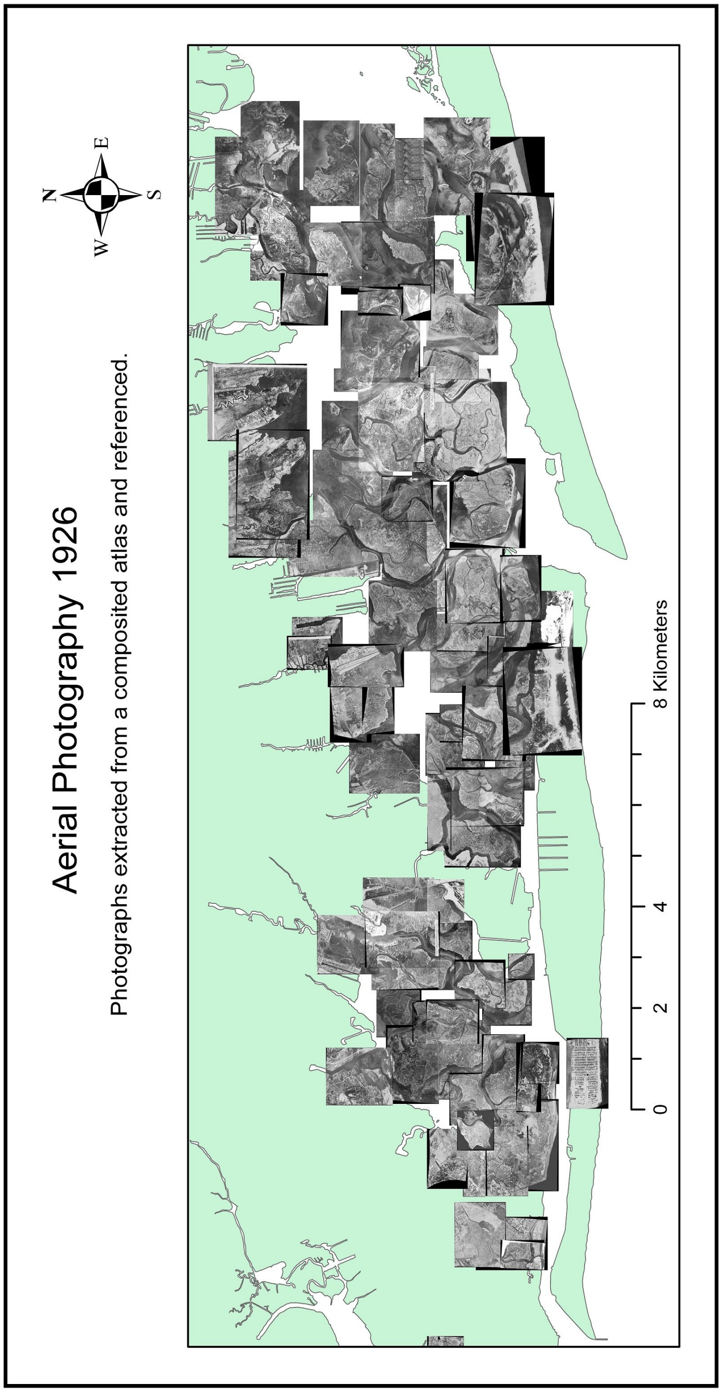

Fig A.5. A map showing the results of a bathymetric survey of the study area conducted in 1880.

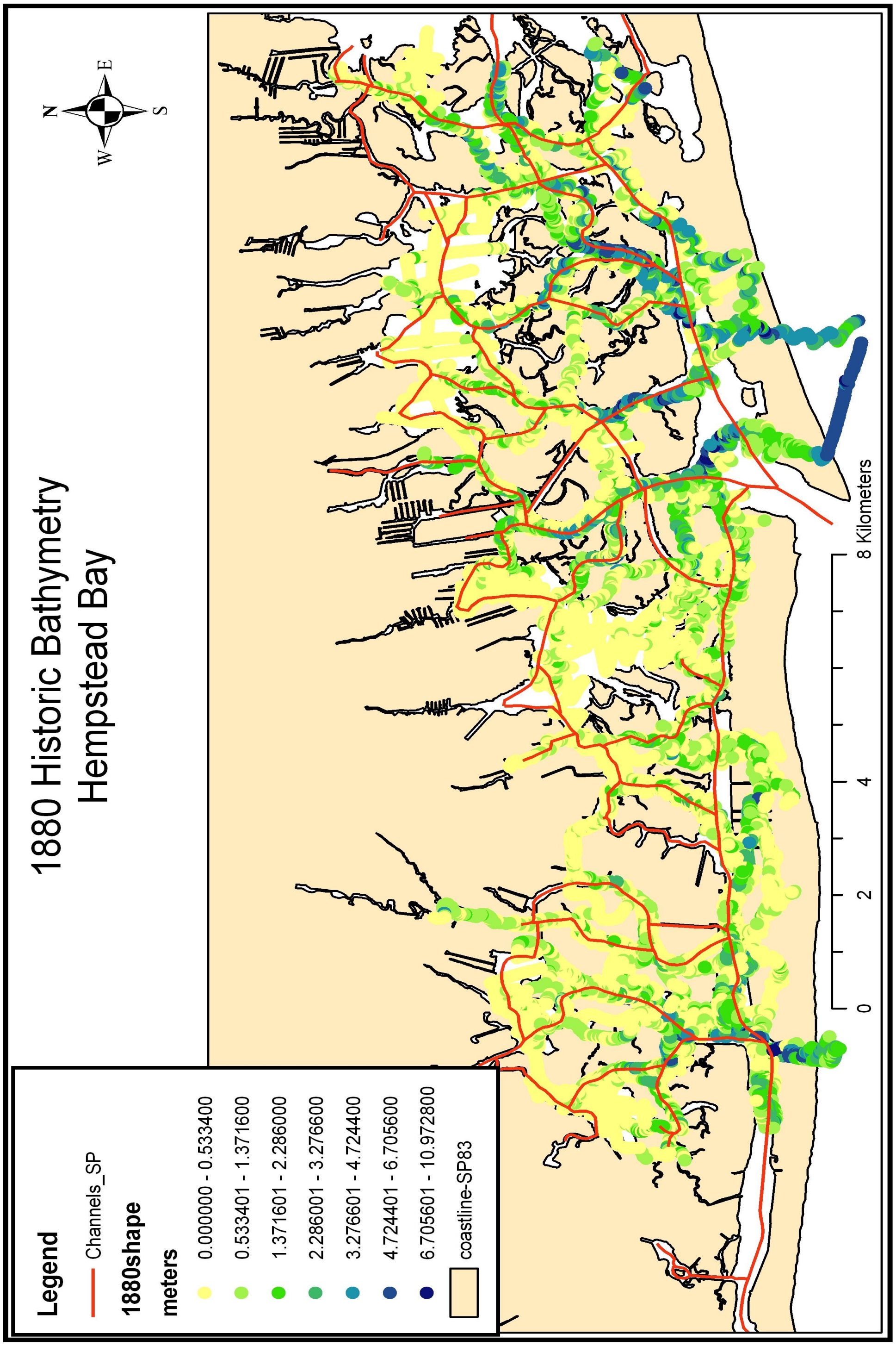

Fig A.6. Locations of Town of Hempstead tide gauges. These gauges were incrementally replaced by United States Geological Survey (USGS) and Stony Brook University School of Marine and Atmospheric Sciences (SoMAS) tide gauges during the time interval 1997 and 2011.

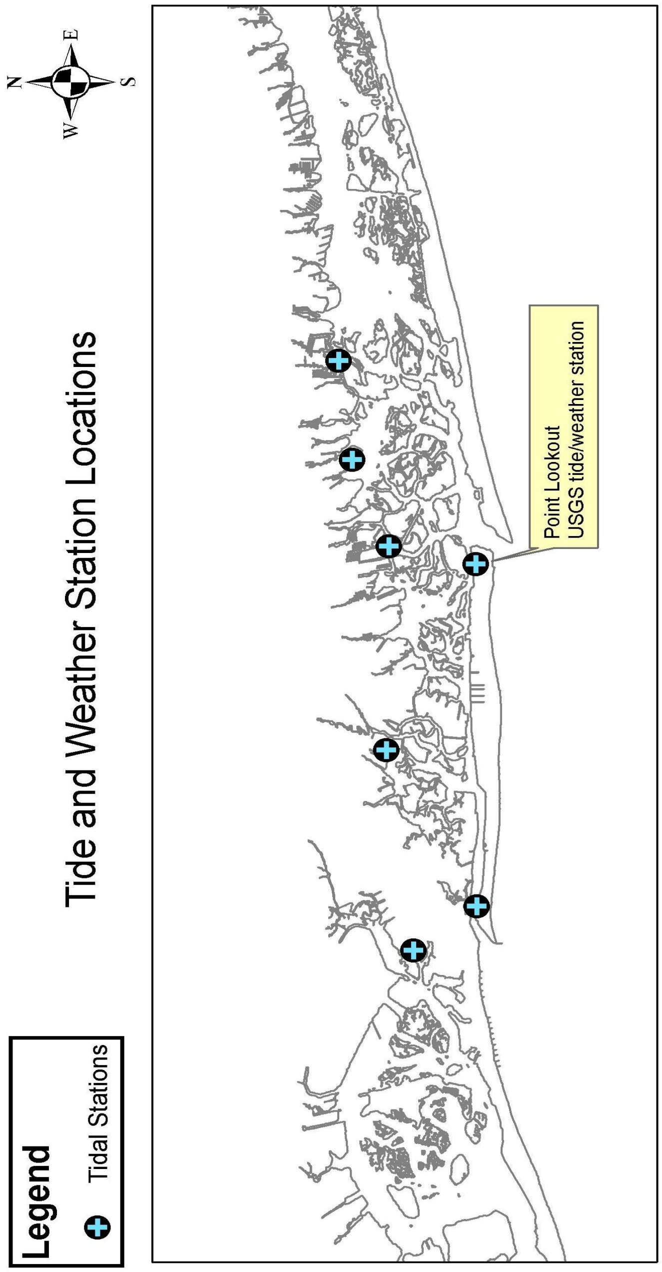

Fig A.7. Changes in maximum high tide from 1980 to present at the National Oceanic and Atmospheric Administration (NOAA) Battery Park, NYC tide gauge (red solid line is the loess of monthly means), Bay Park Town of Hempstead (TOH) (black dashed line is the loess of daily highest tide), Freeport TOH (green dashed line is the loess of daily highest tide), and Seaford TOH (orange dotted line is the loess of daily highest tide), with all tide levels adjusted to 1980 mean tidal height values as 0.

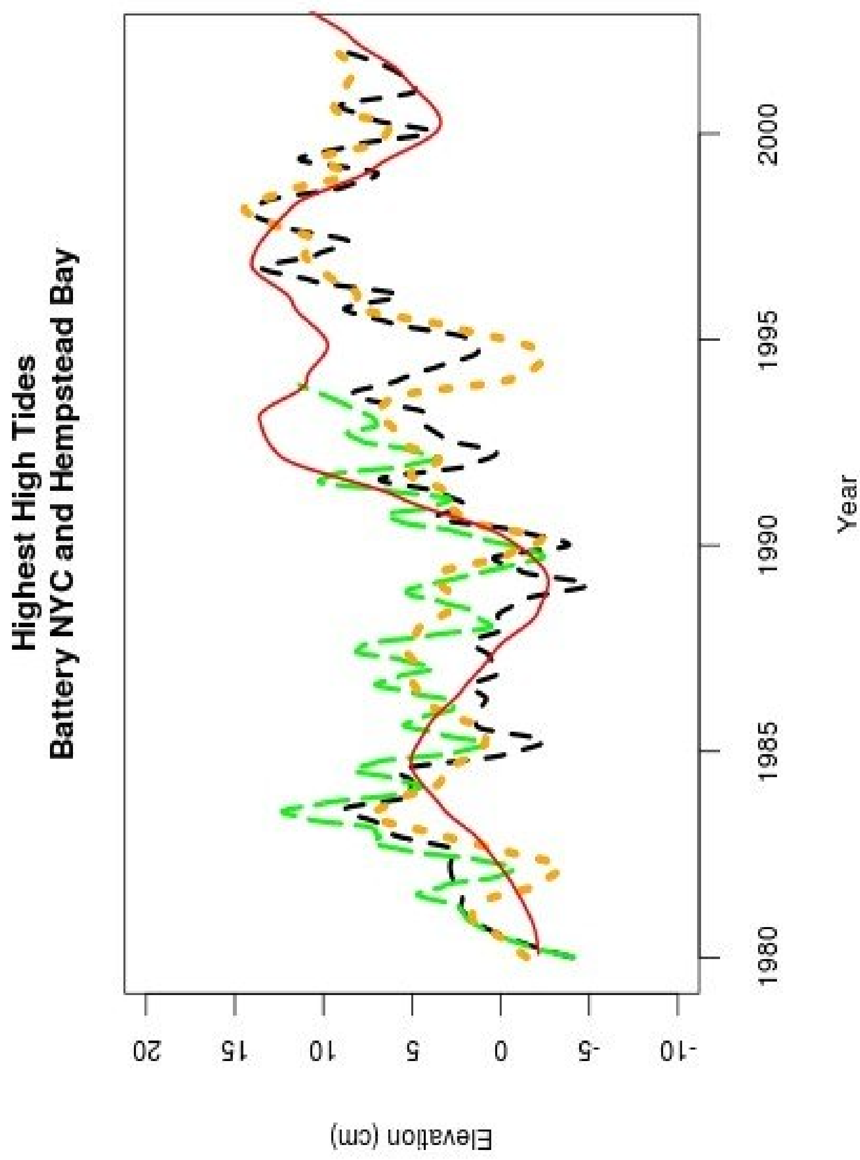

Figure A8. The monthly mean sea-level (measured in cm) at the NOAA Battery Tide Gauge, New York City, NY USA. The blue line indicates a loess fit to the data. The height is relative to the National Geodetic Vertical Datum (NGVD) 1929.

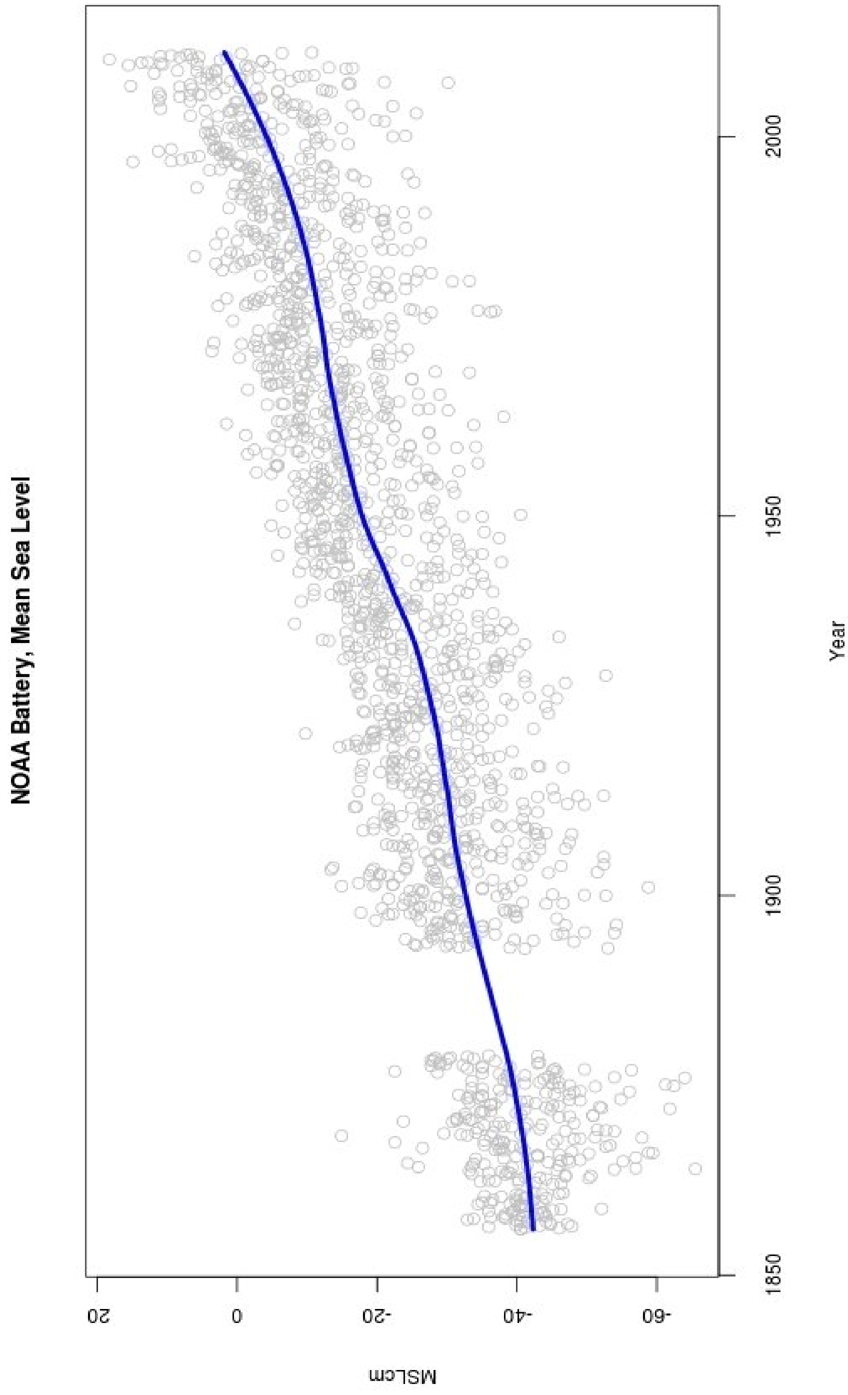

Fig A.9. Town of Hempstead records from 1893 documenting the authorization of expenditures for the ditching of marshlands and the dredging of Freeport creek.

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Hempstead March 14. 1893.

A meeting of the Town Board of the Town of Hempstead was held pursuant to law in such case made and provided in the Town Hall in said town March 14. 1893. — full Board present

On motion the following preamble and resolutions were taken up and unanimously adopted.

Whereas, the Queens County Court House is in a very unsuitable location for a majority of the people of the County and its unhealthfulness has been in years past and more particularly during the past month, clearly demonstrated. therefore be it

Resolved. by the Town Board of the Town of Hempstead that it is desirable and for the best interests of Queens County to have the Court House permanently located at Jamaica which is now the County Seat;

Resolved, that we urge the Board of Supervisors to take such steps as may be necessary for the erection of a jail at Jamaica, provided that the use of the Town Hall in said town of Jamaica can be obtained by the County on favorable terms for use as a Court House.

On motion made and duly seconded. on each of the following resolutions they were unanimously adopted

Resolved, That the time for cutting the grass on the Common Marshes of the Town of Hempstead, be the first Tuesday of September. 1893. at sunrise

Resolved. That a fine of $12.50. be imposed on any person who cuts grass. on the Common Marshes. previous to date of first Tuesday of September 1893.

Resolved, That the Supervisor be authorized to sell at public auction the grass on the Common Meadow at Freeport, at such time as he may deem best and that the net proceeds derived from the sale of the grass on the Common Meadow. be expended by the Supervisor in ditching the said meadow. and digging out Freeport creek.

Resolved, That the [illegible]

Figure A.10 Several GIS layers derived from the aerial photography from 1926, 1956, 1983, 1994, and 2004. Red indicates the extent of the marsh in 1926, and indicates a section of Long Meadow Island was channelized during the 1930s to create Sea Dog Creek and to fill marsh for the Loop Parkway. Several measurement points and measurement trend lines used to quantify marsh loss through time are also shown.

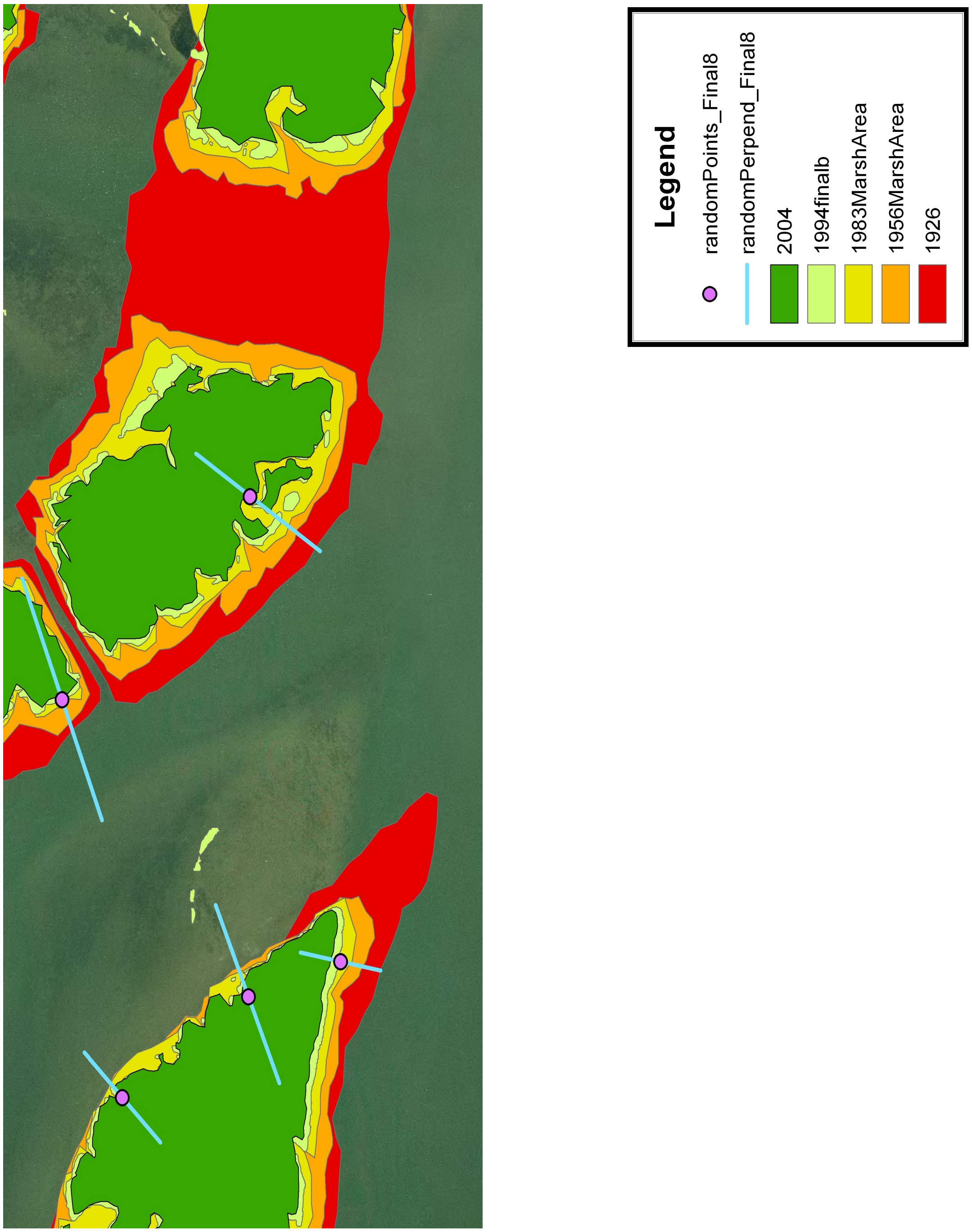

Fig A.11. The survey map of Sturm Channel and the Channelized edge of North Meadow Island drawn for the US Geological survey 1879. with recent USGS data overlaid.

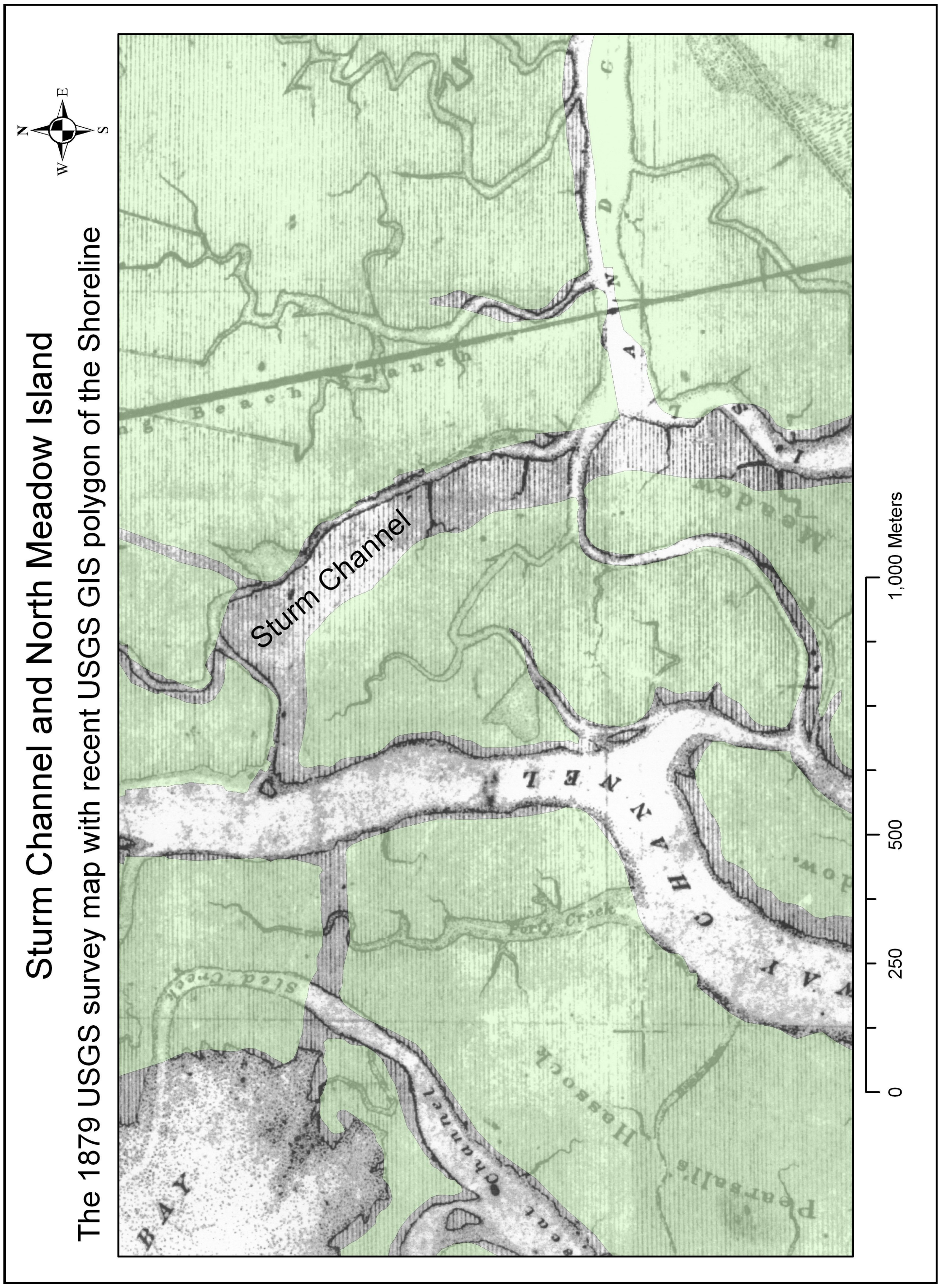

Figure A.12. A conceptual illustration of the measurement process used to quantify marsh loss or gain through time. The apparent line of change represents the measurement trend line and the point where it crossed the typical edge measurement point. The measurement of past and future edge position at that point was the difference between the edge measurement point and the points where the line of change crossed the edge at other time periods.

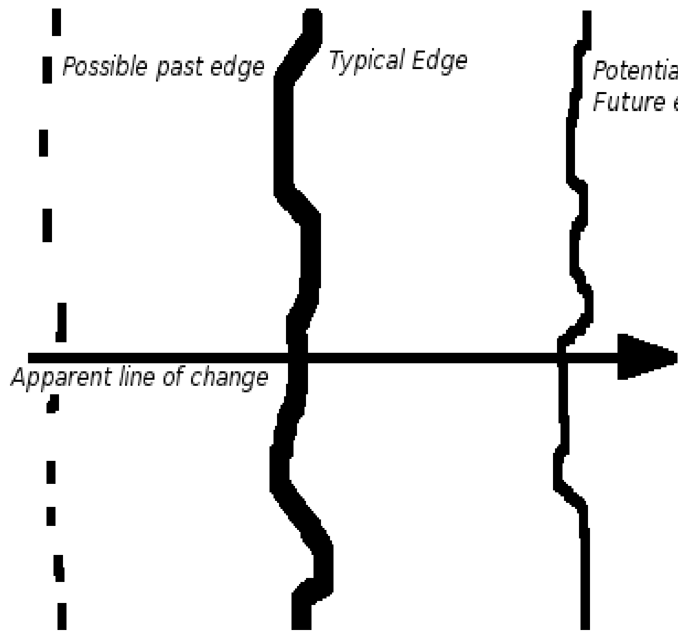

Figure A.13. The location of channelized edges within the study area.

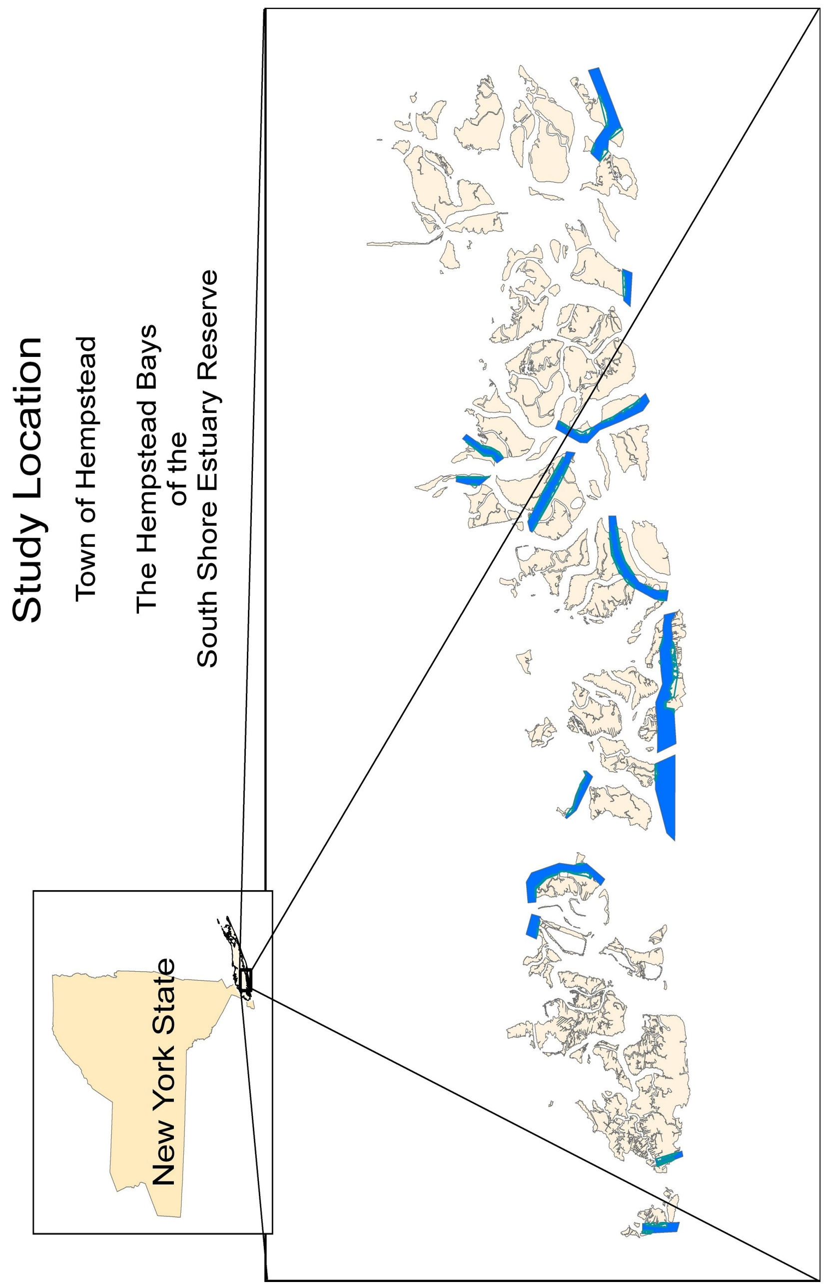

Figure A14. The Navigational Channel Classifications. Red is category A, the main channels with highest navigational use. Yellow is category B, the high use secondary channels, Tan is category C, the medium use channels, Green with a black line is category D, the low use channels, and Green is category E, the occasionally use channels. More detailed descriptions of channel categories are found in Table 3.1.

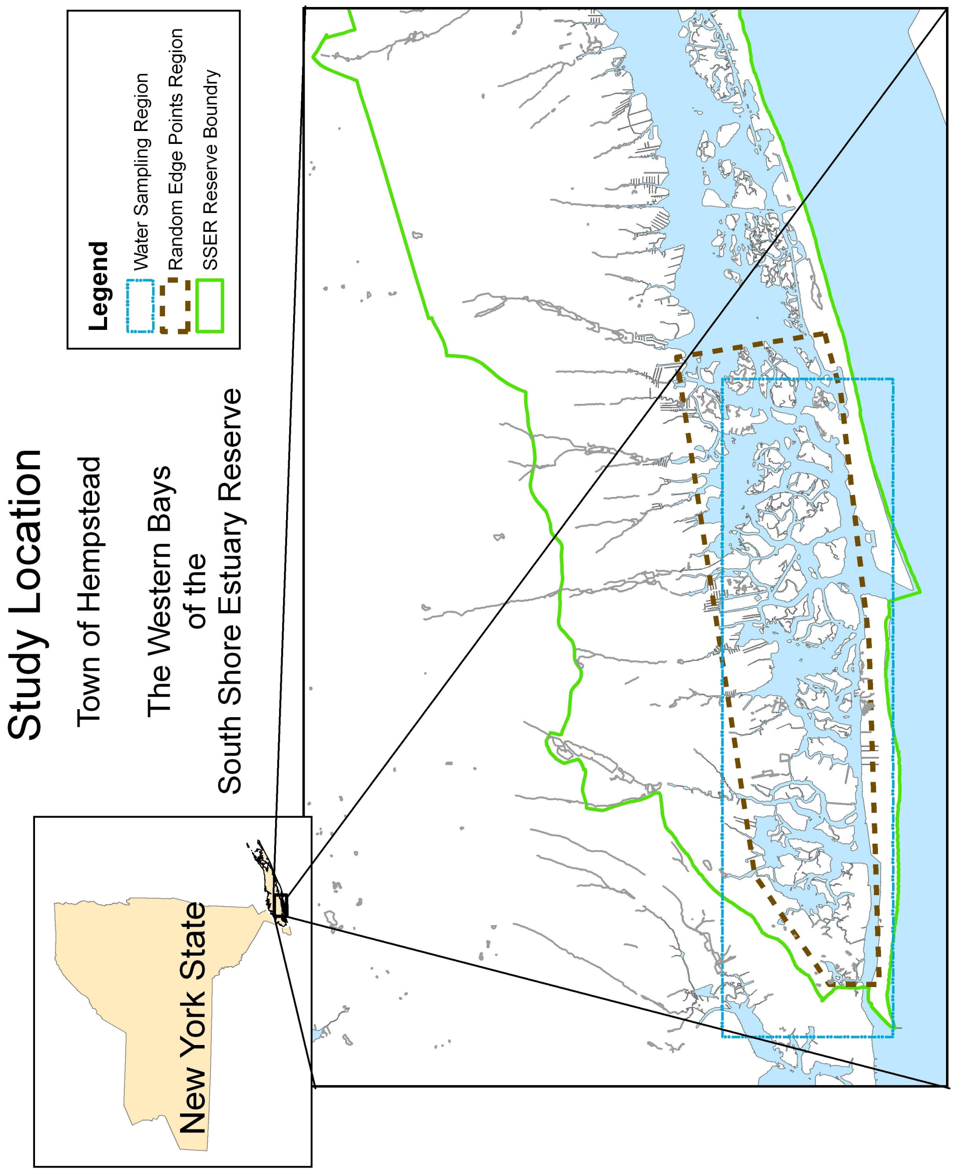

Figure A 15. Locations in the study area where the edge of the marsh has been hardened by bulkheads or stone are indicated in red.

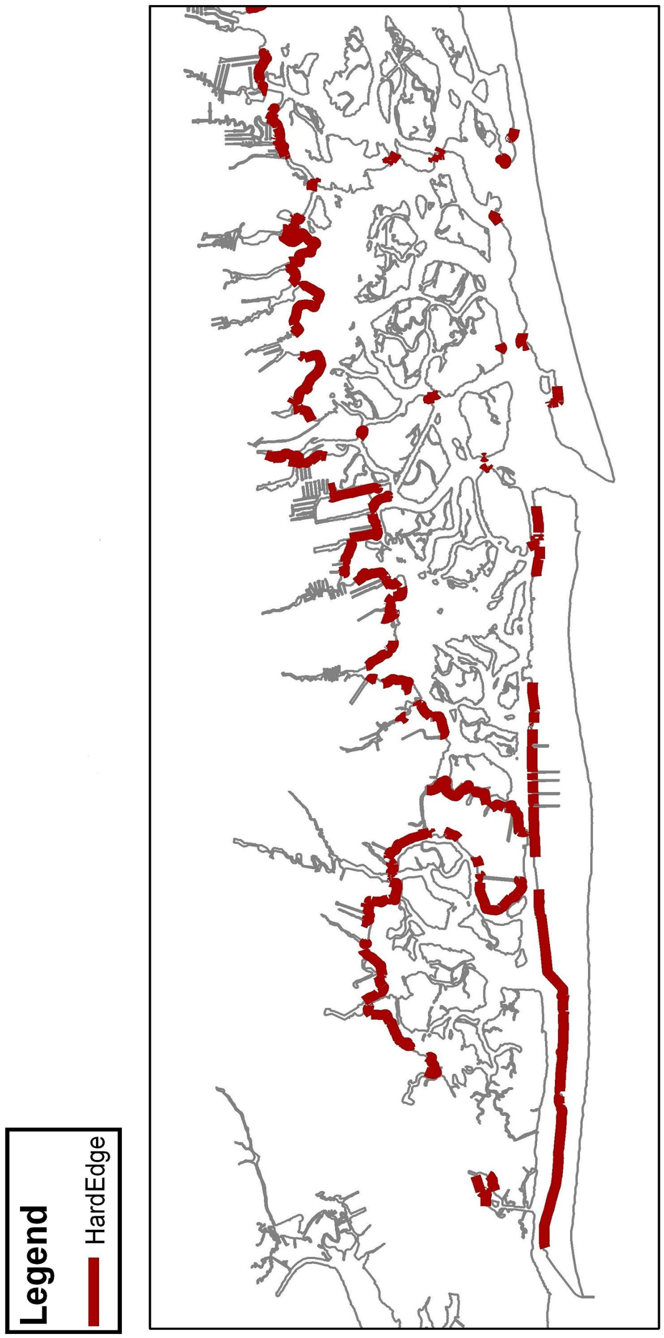

Figure A.16. Treated Sewerage Outfall Locations within the study site are marked by red crosses.

Figure A.17. Borrow Pit locations within the study site are marked in blue.

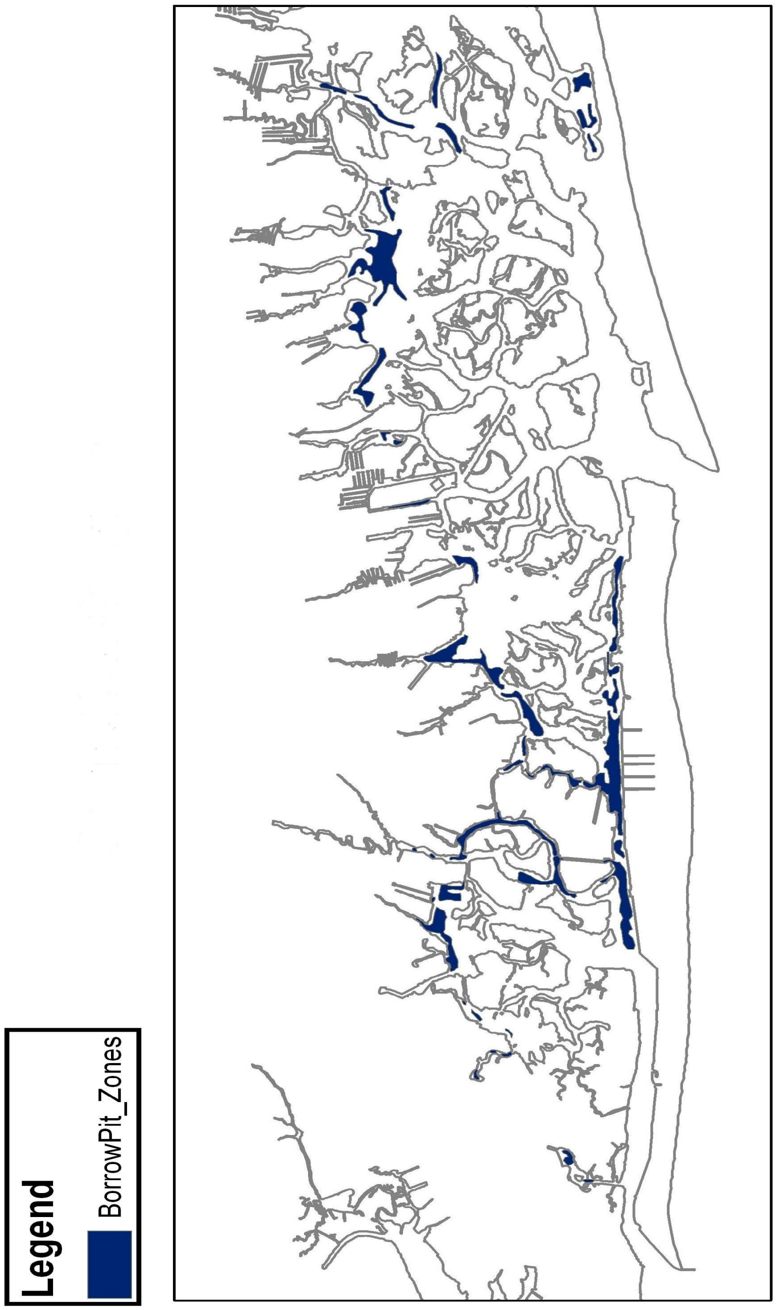

Figure A.18. A map of the Town of Hempstead, Department of Conservation and Waterways, water quality testing stations. The size of the green bubbles indicate mean nitrate levels in ug/l N.

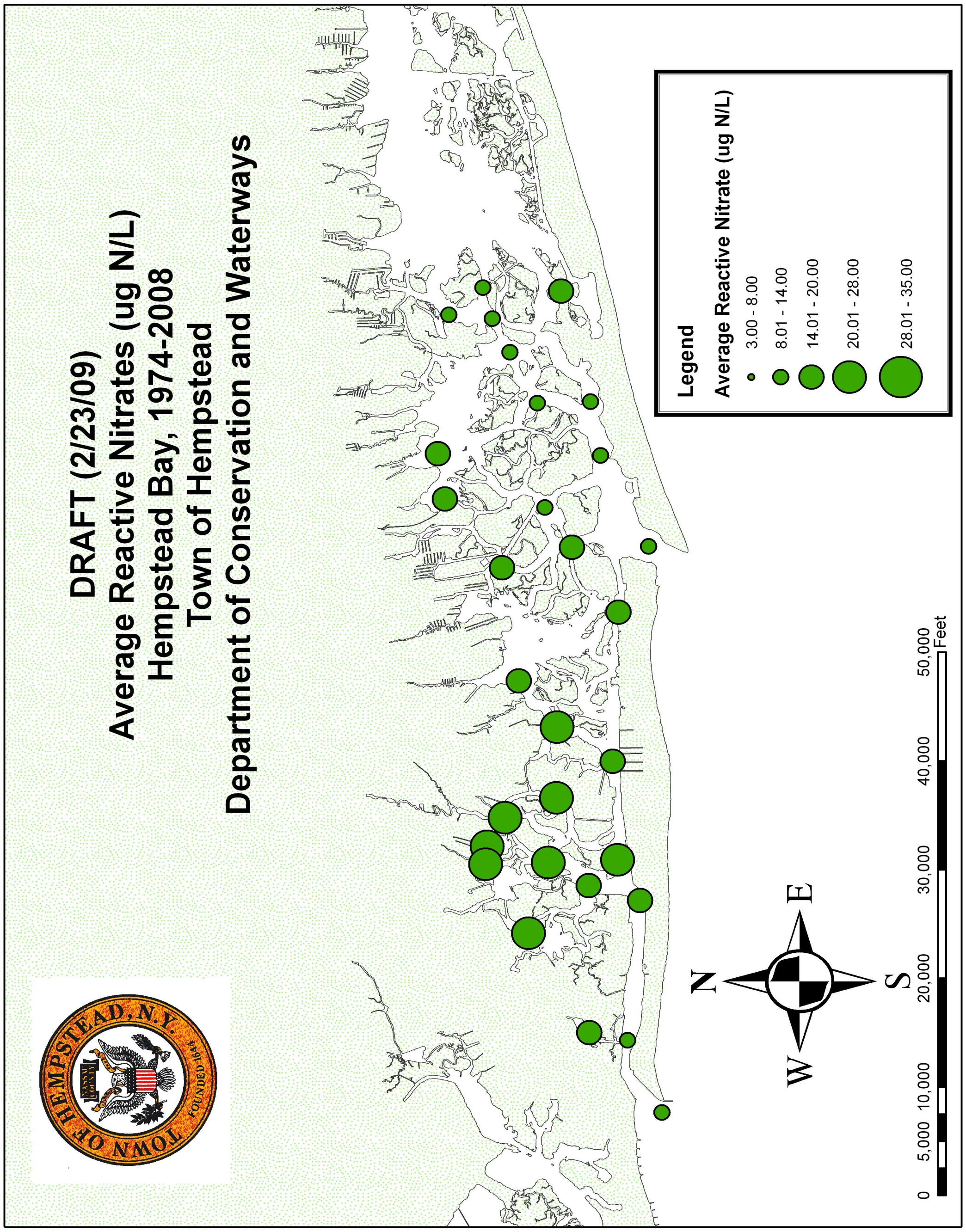

Figure A.19. Spatial interpolation was used to estimate the average nitrate concentration for each marsh edge sample point.

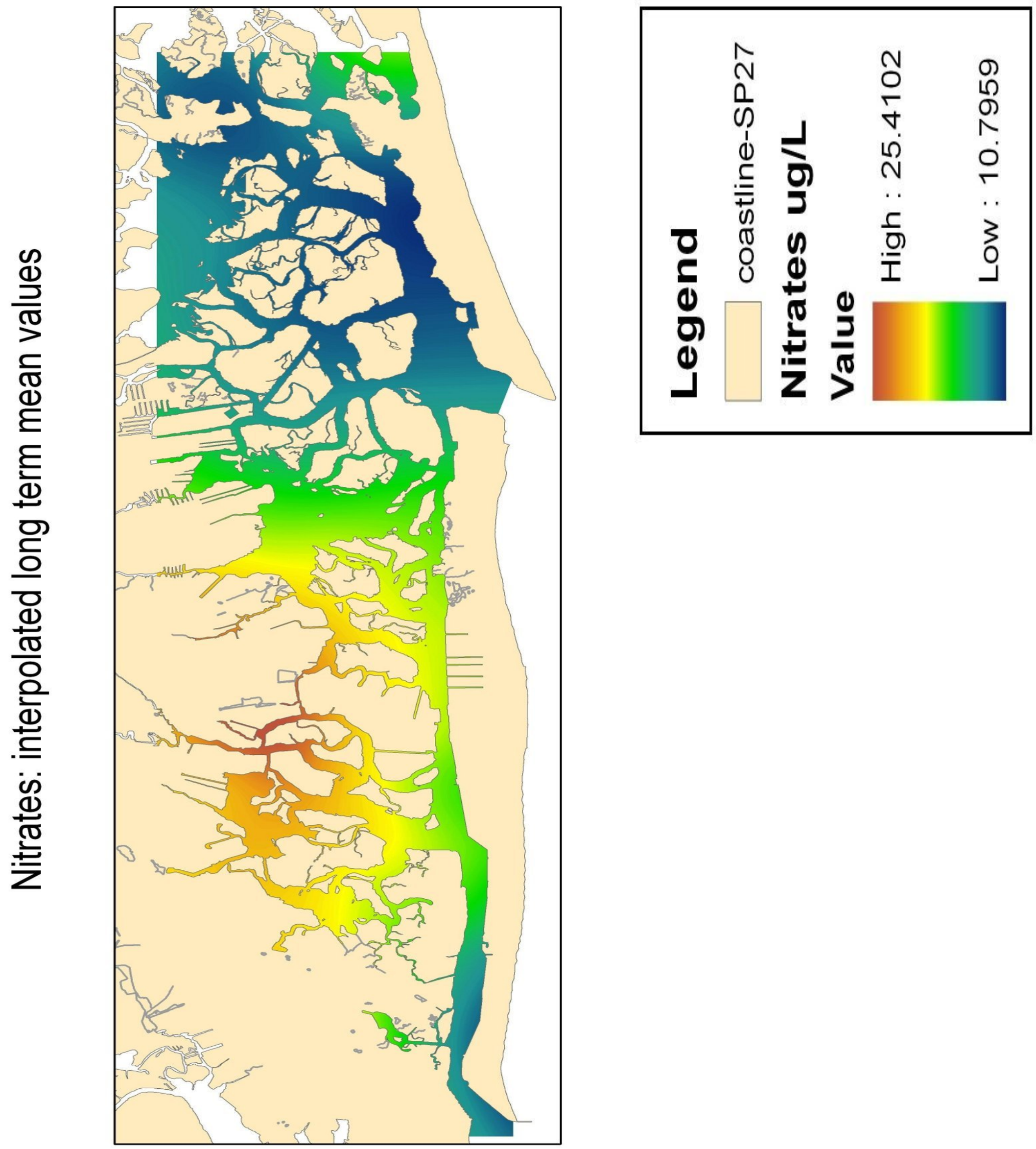

Figure A.20. A wind rose showing the direction of the top 10% of wind speeds at the Point Lookout USGS meteorological station. These data were used with the SWAN Cycle II model to generate the waves created by high winds from the indicated direction and where they impacted the shore along the salt marsh in the study area.

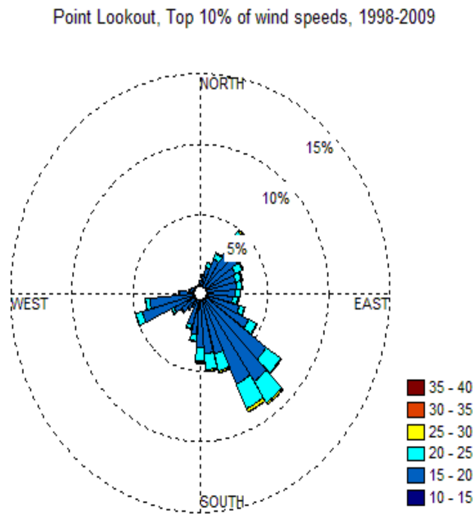

Figure A.21. The general pattern of gains and losses 1966 - 2007.

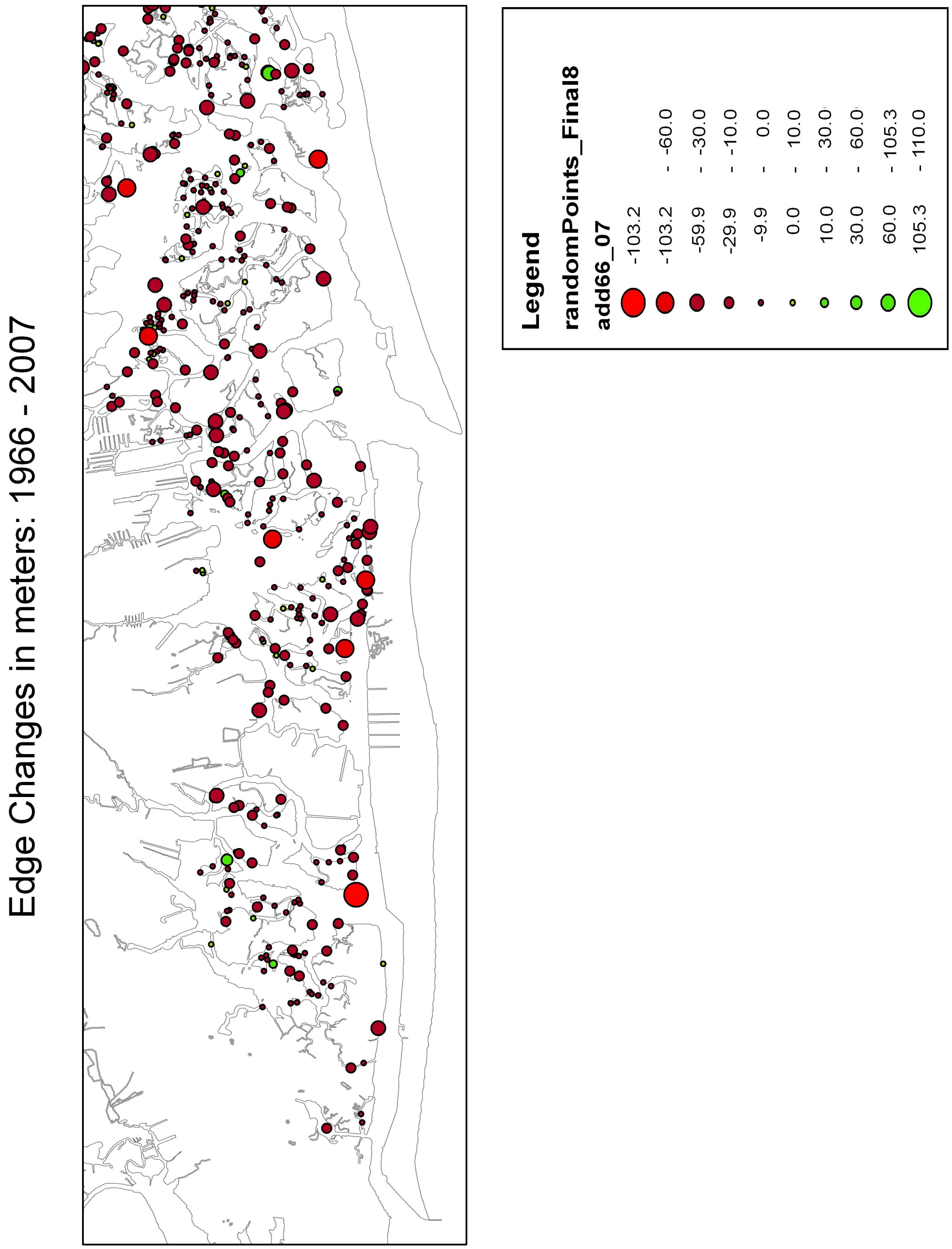

Figure A.22. The general pattern of gain and loss 1926-2007.

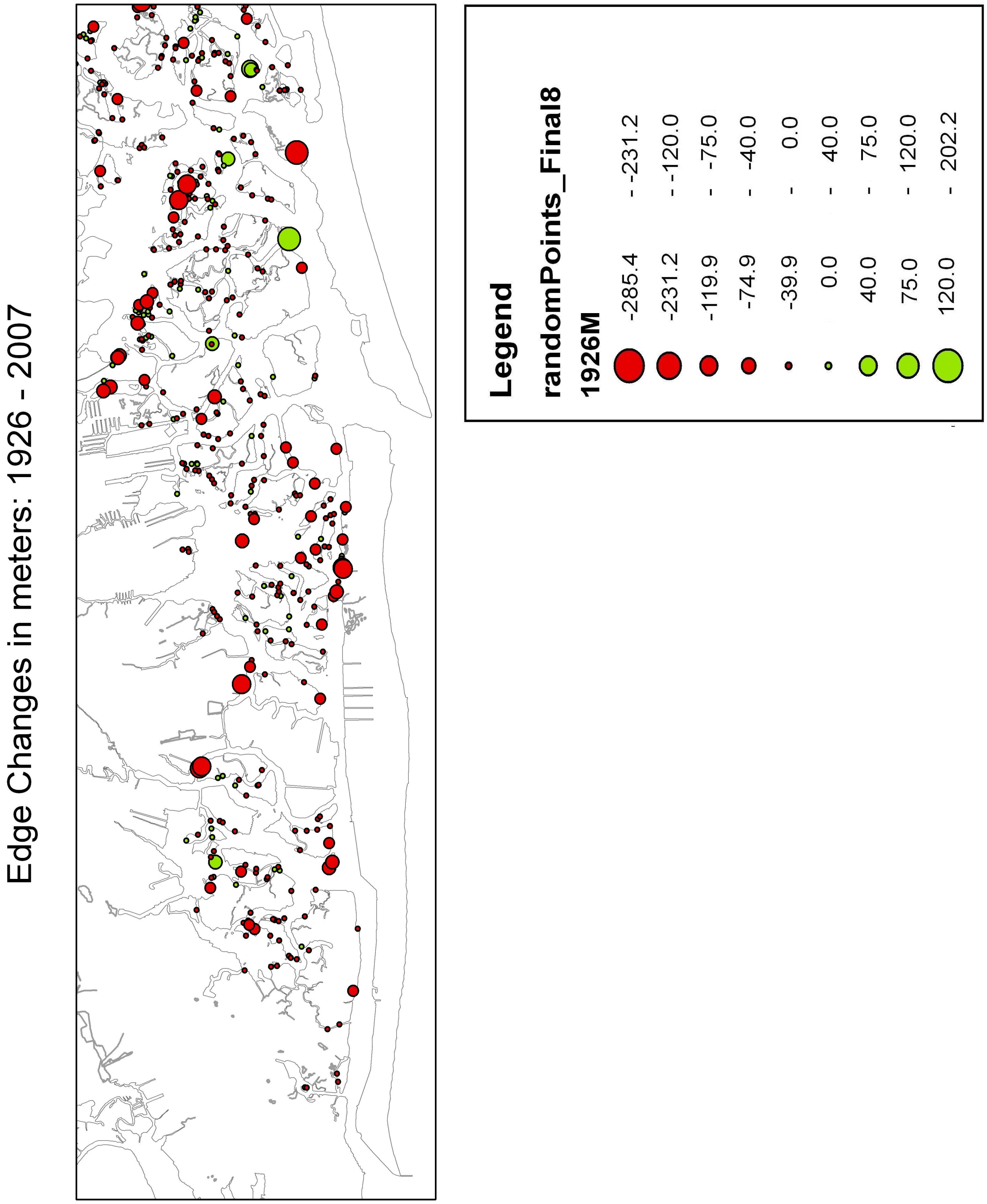

Figure A 23. A section of NOAA chart 12352 with local 2010 private aid (buoy) locations and January 2011 bathymetry data (R. Flood, SoMAS, Stony Brook University *personal communication*) overlaid on top. The navigational channel known as Broad Creek Channel had become more narrow and eroded into the southern bank of Cuba Island. The old location of the navigational channel had become a point bar, defining a classic example of a meander stream.

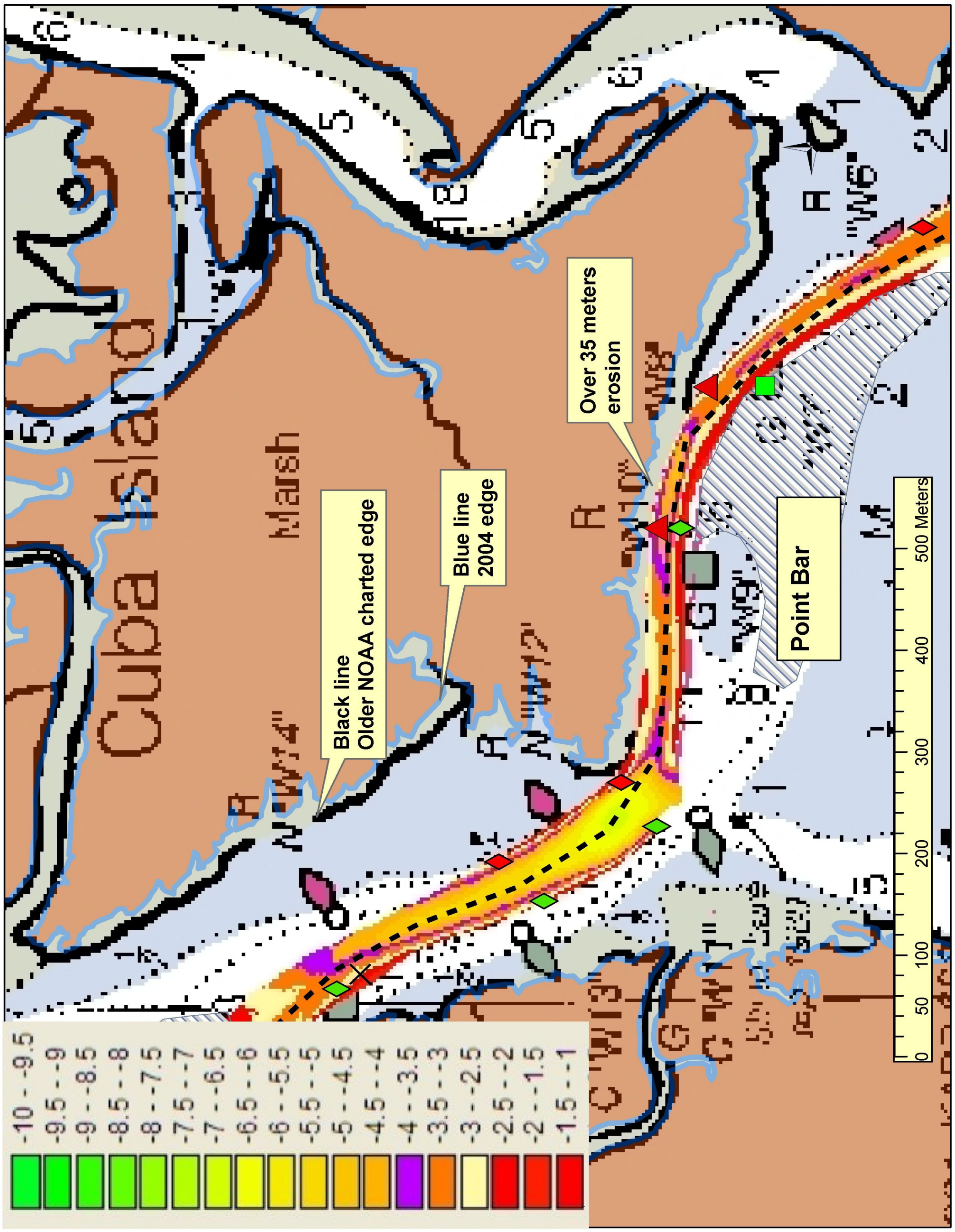

Figure A24. A section of NOAA chart 12352 with local 2010 private aid (buoy) locations and January 2011 bathymetry data (R. Flood, SoMAS, Stony Brook University *personal communication*) overlaid on top. The navigational channel known as Haunts Creek shows severe erosion of over 100 m on the eastern edge bordering Deep Creek Meadow. The western bank of Haunts Creek has new marsh expanding, but the surface ares is less than what was lost.

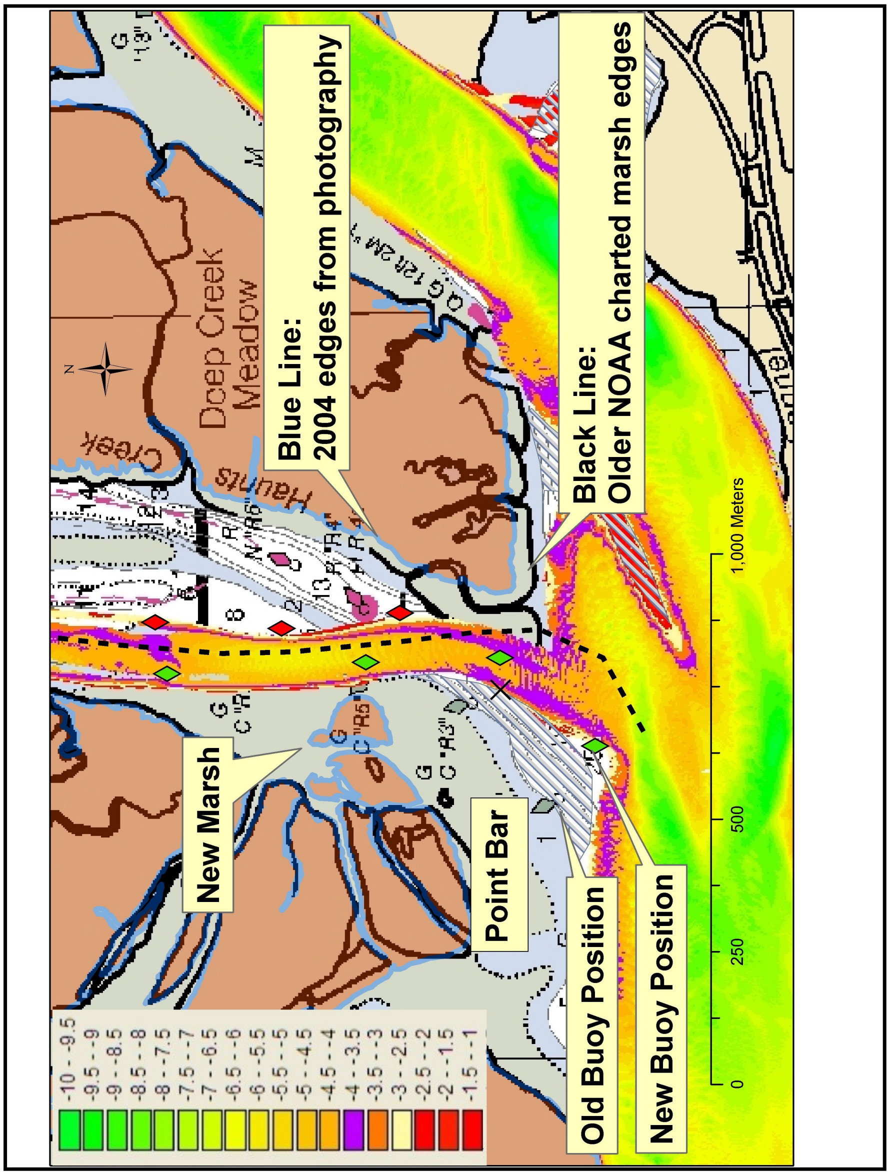

Figure A 25. An example of *Spartina alterniflora* colonizing a section of flood shoal inside Jones Inlet, 2009, and from a distance, 2011 (inset).

Figure A 26. Stream flow data from USGS gauges on Pines Brook (01311000, green solid line), East Meadow Brook (01310500, black dotted line), and Bellmore Creek (01310000, orange dashed line). There has been a reduction in stream flow into Hempstead Bay, especially for Pines Brook and East Meadow Brook through time.

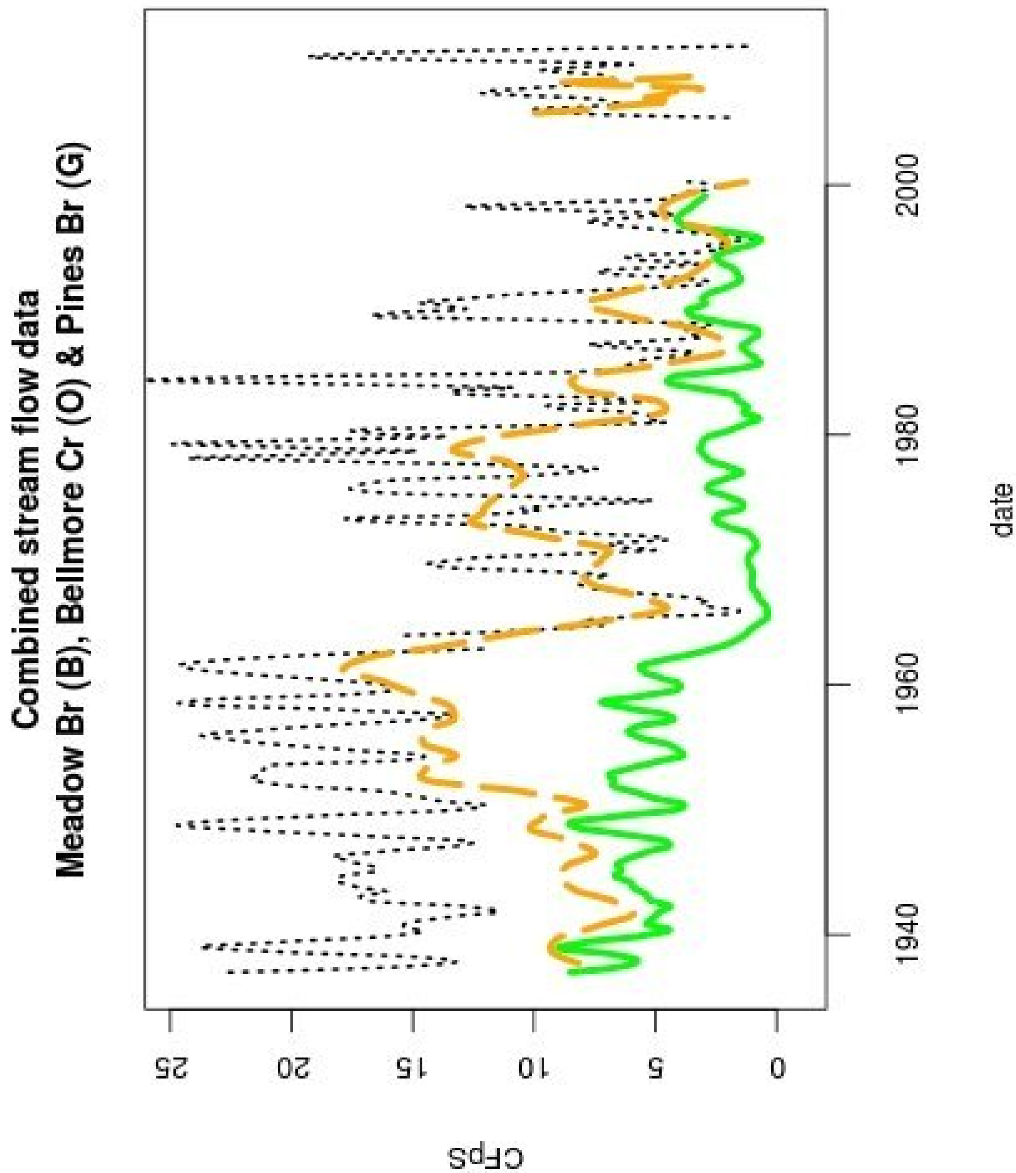

Figure A 27. A yellow-crowned night heron feeding on a purple marsh crab (*Sesarma*) at the Oceanside Marine Nature Study Area, Oceanside, NY. (photo: Michael Farina).