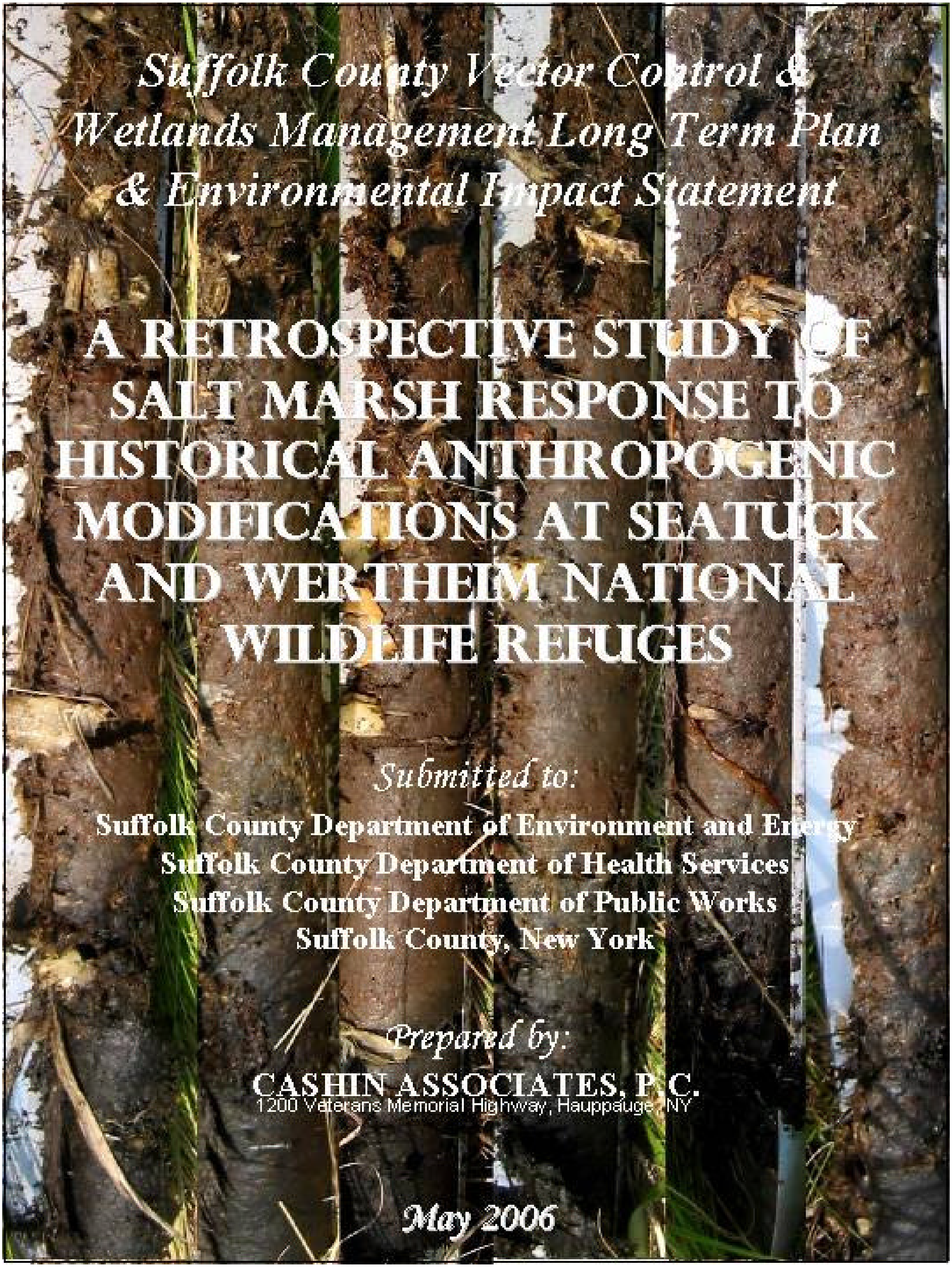

Suffolk County Vector Control & Wetlands Management Long Term Plan & Environmental Impact Statement

A RETROSPECTIVE STUDY OF SALT MARSH RESPONSE TO HISTORICAL ANTHROPOGENIC MODIFICATIONS AT SEATUCK AND WERTHEIM NATIONAL WILDLIFE REFUGES

Submitted to:

Suffolk County Department of Environment and Energy
Suffolk County Department of Health Services
Suffolk County Department of Public Works
Suffolk County, New York

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SUFFOLK COUNTY VECTOR CONTROL AND WETLANDS MANAGEMENT LONG - TERM PLAN AND ENVIRONMENTAL IMPACT STATEMENT

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Executive Summary

The fact that salt marshes accrete, up to several millimeters a year, in order to keep pace with sea-level rise, essentially allows the environmental history of each system to be preserved. This project is a comparative historical analysis of two salt marshes, located on the south shore of Long Island. The first part of this project was designed to characterize each marsh system, by laying out several transects and taking peat auger cores every three meters which were then photographed using high resolution digital photography. This preliminary assessment allows for identification of areas of interest and/or representative marsh environments to take larger cores for analysis. Using sediment cores taken from each marsh, the second part of this project examined several parameters in order to understand how the functioning of the systems changed over the past 100 years, either as a result of natural environmental conditions or anthropogenic modifications. The parameters considered include plant root and rhizome identification, dry organic content, water content, radioisotope dating using ^{210}Pb and ^{137}Cs, and grain size distribution. The hypothesis of this study is that environmental changes recorded in marsh peat deposits of Wertheim and Seatuck National Wildlife Refuges are a consequence of historical grid-ditching activities and, in Seatuck, more recent open-marsh-water-management (OMWM) alterations.

There were two significant results from this study. Primarily, the study found that there were system-wide changes in plant communities at both marshes during the past century. There is evidence of a major transition at both marshes, although the evidence is found in different forms. At both Wertheim and Seatuck there is a shift in plant community and an increase in organic content during the early 1900s (roughly corresponding to 25 cm depth). Although other anthropogenic impacts can be put forth as possible triggers for marsh change, parallel grid ditching seems the most logical explanation for Wertheim due to the fact that radioisotope dating matches plant community shifts in the early 1900s (a period of intense ditching), and studies at other sites on the East Coast has often found that ditching may inherently alters plant communities due to changes in marsh hydrodynamics. However, for Seatuck, it was found that upland development and within-marsh construction alterations had a more significant impact on marsh change than ditching did alone. Secondly, average long-term accretion rates, ranging

from 0.17 cm/yr to 0.41 cm/yr, suggest that these marshes are accreting at relatively the same rate as sea-level, presently averaged at 0.30 cm/yr.

Currently Suffolk County, New York, is implementing an OMWM project at Wertheim National Wildlife Refuge. The marsh histories of Wertheim and Seatuck suggest that the preserved history of marshes allows for a better understanding as to how physical alterations made to these systems may affect their form and functioning. Thus, by looking to the past, more knowledgeable decisions can be made when considering present and future management practices of Long Island's tidal wetlands.

1. Introduction

Salt marshes are ephemeral systems that began to form locally along the Atlantic coastline as early as 9,000 years ago as a consequence of slowing rates of sea-level rise. It was not until sea-level rise decelerated further, approximately 5,000 to 3,000 years ago, that marshes became more stable and widely established. During this latter period, the rate of sea level rise was only about one mm/yr, such that sedimentation rates in many coastal settings and shallow estuaries exceeded that of sea-level rise and thus favored marsh development (Redfield, 1972). This occurred both as an emergent process, whereby open-water habitats became sufficiently infilled to become vegetated, as well as by the slow submergence of upland that similarly favored marsh-plant colonization. Following the establishment of smooth cordgrass (*Spartina alterniflora*) in such settings, and continued sedimentation, increasing elevation created habitat for other high-marsh grasses species, such as salt-meadow cordgrass (*Spartina patens*), black grass (*Juncus gerardi*), and spike grass (*Distichlis spicata*) (Warren, 1995).

The establishment of salt marshes led them to become an increasingly tangible ecosystem to the human populations living near them. Archeological evidence suggests that Native Americans set up seasonal settlements and then eventually year-round settlements near these wetlands in recognition of their productivity and the abundant resources they supported, including grass thatch and fin and shell fish. . Following European settlement, colonists used salt marsh grasses for livestock fodder and bedding, with this practice continuing well into the beginning of the 20th century. In more recent times, salt marsh ecosystems have been subjected to a series of physical alterations which have caused both environmental degradation and marsh loss. During the past century, alterations included those from filling wetlands for railroad, highway and bridge construction, as well as modifications caused by parallel grid ditching for vector control (Rozsa, 1995).

At present, salt marshes represent a unique and important coastal ecosystem that serves as a transitional zone between the terrestrial and oceanic environments. To truly appreciate and understand the significance of salt marsh ecosystems one must consider the wide range of functions and values these systems embody. The following functions and values are reviewed in Dale and Hulsman (1990) and are also stated as the purpose for the creation of New York State's Tidal Wetlands Act (Article 25 of Environmental Conservation Law, section 661.2).

Biologically, salt marshes play a vital role in the estuarine food chain as a result of their high productivity, and in addition they provide essential habitat and/or nursery grounds for a variety of wildlife (Crooks and Turner, 1999). Physically, marshes have been credited for providing coastline protection from floods, storms and hurricanes (Mitsch and Gosselink, 2000). In addition to these biological and physical functions, the general public values salt marshes for the recreational, educational, and research opportunities they provide, as well as simply for their aesthetical appeal (Dale and Hulsman, 1990). Given these important attributes one can understand the necessity in developing and implementing salt marsh management plans that strive to preserve and protect their diversity of functions and values.

1.1 Long Island Salt Marshes

The north and south shores of Long Island are home to an estimated 17,000 acres of tidal wetlands (Cashin Associates, 2006). Over the years, various governmental agencies and non-profit organizations have demonstrated an interest in protecting these coastal lands and their associated natural resources (Niedowski, 2000). In New York State, tidal wetlands are fundamentally protected under Article 25 of the Environmental Conservation Law which was enacted in 1973 (Part 661, Tidal Wetlands-Land Use Regulations). This article protects wetlands by providing development criteria and permitting requirements in order to preserve and protect tidal wetlands, while at the same time giving consideration to the economic and social development of the State. Most notably, this article prohibits activities destructive to wetlands such as filling and dredging. At the Federal level, tidal wetlands are protected through Section 404 of the Clean Water Act which regulates the discharge of dredged and fill materials into United States waters, including wetlands. In addition, US Code gives protection to wetlands under title 16-Conservation (Chapter 59-Wetlands Resources and Chapter 59A-Wetlands). State and Federal laws enacted for the protection and preservation of tidal wetlands provide a mechanism for the creation and implementation of appropriate management practices, however, actual practices vary widely at the local and region scales based on agency regulators and political and public climates.

1.2 Project Background

Suffolk County, New York has proposed to implement the practice of Open Marsh Water Management (OMWM) at Wertheim National Wildlife Refuge, as part of an effort to eliminate/decrease the amount of pesticides, particularly larvicides, applied for mosquito control. OMWM is a type of marsh management that physically alters the hydrodynamics of the marsh by creating a series of ponds and channels, which are ultimately connected to the adjacent bay. The overall goal of OMWM is to create suitable habitat for larvae eating fish. These fish are delivered to the marsh during high tides and prey upon mosquito larvae, or maintain permanent presence on the marsh surface via deep ponds and connecting channels. The desired outcomes of this practice are a reduction in mosquitoes, a reduction on the dependency of chemical pesticides, and improvement in the overall health of the marsh system, such as the creation of a mosaic of habitats fostering greater biodiversity. The County has hired the environmental consulting firm Cashin Associates, located in Hauppauge, New York, to design a long-term vector control and marsh management plan. The firm is overseeing numerous tasks, including the monitoring of a variety of biological and physical parameters (monitoring performed by Cashin staff, Suffolk County Department of Health Services, US Fish and Wildlife Service, and Ducks Unlimited). They will use this monitoring information as a baseline from which to determine impacts associated with the development of OMWM practices at Wertheim.

This research will contribute to this multi-faceted county project by providing the necessary component of marsh-history analysis using peat cores. These cores will be used to reconstruct approximately the past one hundred years of history at both Wertheim and nearby Seatuck National Wildlife Refuge. Results from the cores will be better interpreted by looking at the surface expression of anthropogenic and natural modifications in each coring area. Since the salt marshes are located within 32 kilometers of each other and in the same physiographic setting (Figure 1), many factors like previous weather events, tidal range, sediment supply, and salinities are largely similar between the systems. With these physical constraints, this study will test the hypothesis that early environmental changes recorded in marsh peat deposits of Wertheim and Seatuck refuges are a consequence of historical grid-ditching activities, and that more recent differences between the two sites are a result of OMWM-style ditch plugging alterations made at Seatuck in the late 1980s. By understanding the history of the two marshes, their similarities and differences, better insight into the possible long-term effects of OMWM alterations on Long Island’s South Shore salt marshes can be gained.

1.3 Open Marsh Water Management

Two important objectives for OMWM to successfully reduce mosquito abundance are to create habitat for larvivorous fish and to improve their access to the marsh surface via restoring marsh hydrodynamics. In accomplishing these goals the desired outcomes are a reduction in mosquitoes, with minimal damage to the marsh, as well as a reduction on the dependency of chemical pesticides (Cowan *et al.*, 1986). General approaches to OMWM consist of creating small ponds and deep-water reservoirs that support predatory fish (i.e., killifish *Fundulus spp.* and mummichogs *Cypridon spp.*). The areas of open water created by OMWM modifications are frequently well connected to the estuary, thus allowing the continuous movement of fish in and out of the system to predate mosquito larvae and eggs. In addition to traditional OMWM approaches, newer alternative techniques being used create open-water areas that are unconnected with the estuary, but which remain permanently flooded and thus can support fish populations (http://dep.state.ct.us/olisp/ramsar/mgmtprac.htm).

Although it has been practiced since the late 1960s, the impacts of OMWM are still not completely understood and the process is often criticized due to the fact that it requires physical alterations within the marsh. At the same time, OMWM has been credited as being a more desirable vector control than both traditional parallel grid ditching and the current practice of pesticide application (Wolfe, 1996). In recognition of the alleged ecological impacts caused by grid ditching, and of human and environmental health concerns related to the use of potentially harmful pesticides, many states have slowly shifted their perspective on vector control, and are in search of a new method that is both ecologically and human-health friendly. Atlantic coastal states, including New Jersey, Maryland, Delaware, Massachusetts, and Connecticut, have found success in the implementation of OMWM as a vector control technique (e.g., Cowan *et al.*, 1986). A number of these states have defined "success" for OMWM in terms of a reduction in pesticide use, along with a reduction in mosquito larvae and restored habitat for fish and birds (Wolfe, 1996).

On Long Island, one of earliest OMWM projects was that implemented in the late 1980s by the Seatuck Research Program at the Seatuck National Wildlife Refuge. This project consisted of cleaning filled ditches, plugging ditches, and digging shallow ditches, known as "radial spurs", from already existing ponds and deep ditches to mosquito breeding pools in order to give fish access to mosquito larvae (Lent *et al*., 1990). Although the investigators of the Seatuck OMWM project claimed success in their efforts, it was not integrated into county-wide vector control plans. Unfortunately, subsequent monitoring was limited and provides little basis for assessment. Thus this report revisits the site to reconstruct the physical environmental changes that have since occurred, as well as to compare those change occurring during the earlier part of the century at Seatuck. These results are reported in conjunction with a similar study at the OMWM demonstration site at Wertheim National Wildlife Refuge.

2. Site Descriptions

2.1 Wertheim National Wildlife Refuge

Wertheim National Wildlife Refuge was established by the US Fish and Wildlife Service in 1947. The land was donated by Cecile and Maurice Wertheim who had privately maintained the property for waterfowl hunting. An additional parcel of land was donated in 1974, and the most recent acquisition, in 1998, makes the refuge a total of 2,400 acres of land and over 700 acres of salt marsh (personal communication with NWR staff, 2004). Located along the south shore of Long Island, in the town of Shirley, Wertheim is bordered to the south by the Great South Bay (Figure 2). The Carmans River bisects the refuge, adding to both its aesthetic and recreational appeal. Large areas of the Wertheim salt marshes show a typical New England marsh zonation pattern (Teal and Teal, 1969), with the low marsh being dominated by the cordgrass, *Spartina alterniflora*, and the high marsh dominated by the cordgrass, *Spartina patens*. However, more upstream (ie., less saline) areas of high marsh were found to support a greater diversity of marsh plants including spikegrass (*Distichlis spicata*), salt marsh fleabane (*Pluchea purpurascens*), salt marsh bulrush (*Scirpus robustus*), and swamp rose mallow (*Hibiscus palustris*). In addition, the upland borders are characterized by marsh elder (*Iva frutescens*) and the invasive common reedgrass (*Phragmites australis*). The prevalent nature of *S. robustus*, in some areas, suggests that upper portions of Wertheim are brackish since this plant thrives under lower saline conditions (Niering and Warren, 1980). The tidal range for Wertheim is relatively small, with a mean range of 0.34 m and a spring range of 0.41 m (http://co-ops.nos.noaa.gov/tides05/tab2ec2a.html).

2.2 Seatuck National Wildlife Refuge

Seatuck Wildlife Refuge is located in the town of Islip, on the South Shore of Long Island, 32 km west of Wertheim (Figure 3). Originally maintained as a “gentleman’s farm” for fish and game hunting, the Seatuck land was ultimately established as a refuge for the management of migratory birds in 1968 by the US Fish and Wildlife Service. Much smaller than Wertheim, it includes about 60 acres of salt marsh (personal communication with NWR staff, 2004). The mean tidal range at Seatuck is 0.30 m and the spring tidal range is only slightly higher at 0.34 m ((http://co-ops.nos.noaa.gov/tides05/tab2ec2a.html). Champlin Creek runs parallel along the eastern border of the marsh and the Great South Bay provides the southern boundary. As with Wertheim, the vegetation pattern is similar to that of a traditional New England marsh; however, the high marsh also contains many stands of mixed *S. alterniflora* and *S. patens*. Thick stands of *I. frutescens* dominate the upland borders, as well as the edges of ditches. *P. australis* is found in dense stands along many of the ditches, with smaller offshoots seen encroaching onto the marsh. Large quantities of *P. australis* were observed at the boundaries of the marsh, a possible indicator of tidal restriction since *P. australis* is found in fresh water and brackish water (Saltonstall, 2003). Thus, growth of *P. australis* in general is stunted when flooded by seawater (Chambers *et al*., 1998). In 1992, the Seatuck tidal channel was opened, thus restoring limited tidal flow to the marsh. However, currently, the impoundment of fresh water, due in part to an obstructed drainage outlet (observation by author), apparently reduces salinities and may favor the spread of *P. australis.*

The Seatuck Research Program did a baseline study at Seatuck during 1984 to 1985, and OMWM alterations were implemented in 1986 to 1987 (Cowan *et al*., 1986; Hruby *et al*., 1990; Lent *et al*., 1990). However, no follow-up studies have been done since 1987 that give information as to how the OMWM alterations affected this marsh. Based on observations by the authors, it is clear that tidal exchange is limited in much of the marsh. For example, it was observed following persistent rains that large depressions filled with water and supported fresh to brackish ponds that persisted for many weeks (i.e., observed in late spring 2003). On one occasion a hen Gadwall (*Anas strepera*) was observed with a brood on one of these ponds, suggesting they can be suitably long-lived to encourage waterfowl nesting. However, in the absence of frequent precipitation (ie., observed late summer 2003), ponded areas dried to expose large non-vegetated pannes and were not apparently inundated (frequently) by tides, given the absence of water on the marsh surface. These occurrences apparently preclude permanent populations of fish on the marsh surface. However, it should be noted that these ponds were not dug as part of the OMWM project, but rather appear to be a consequence of limited drainage of the marsh interior due to ditch-plugging and inadvertent blockage of drainage outlets by debris.

3. Objectives and Approach

This project is intended to provide a comparative historical analysis of the vegetation and surface accretion history at two Long Island salt marshes through the use of sediment cores. Cores were taken along transects at both marshes. The spatial and temporal changes in vegetation were determined and correlated with previous environmental conditions and anthropogenic modifications. Additional cores were used for radioisotope dating, using ^{210}Pb and ^{137}Cs geochronometers, to provide a timeline with which to match these vegetative changes. The complete data set also includes information obtained from plant root and rhizome identification, organic and water contents, and sediment grain-size distribution. Together, results from these analyses reveal the history of vegetation and sedimentation changes in these salt marshes over the past one hundred years. Insight from these analyses may help to understand which environmental factors and anthropogenic influences have catalyzed these changes.

Understanding the history of each marsh is essential to making informed decisions regarding future management. In particular, knowledge of how Wertheim National Wildlife Refuge and Seatuck National Wildlife Refuge have functioned in the past are important for implementing any marsh management policies. Lastly, it is hoped that the marsh histories developed from this project will address the need to bridge the gap between sound scientific knowledge and local environmental management plans.

4. Methods

4.1 Field Transects

The fieldwork for this project began in June 2004 and concluded the fall of the same year. All fieldwork was conducted nearest to low tide as possible in order to be consistent in field methods; however, given the relatively low tidal range of these two marshes, the exact time of sampling was not a major concern. The study site of the previous Seatuck project (maps in Cowan *et al.*, 1986) was revisited and surveyed for representative areas to conduct transects. Similarly, Wertheim (OMWM areas 1 and 2) was surveyed for representative areas, and transects were taken there as well. Representative areas included those along ditches, in the open "undisturbed" part of the marsh, as well as areas near rivers/estuaries. The goal was to target distinct areas of each marsh to provide an inclusive representation of most marsh sub-environments. After visually surveying the marshes, two to three transect locations for each marsh were identified (Figures 4 and 5).

Using a handheld global positioning system (GPS) with three m resolution, coordinates were taken at the beginning and end of each transect (Table 1). Transects measured 150 m in length, except for Seatuck B and C which were 72 m and 33 m respectively. A two letter code was given to each transect and subsequent large diameter cores were given a three letter code in order to simplify the identification process. The first letter of the code represents the marsh – 'S' for Seatuck and 'W' for Wertheim. The second letter represents the transect, either "A," "B," or "C." The third letter designates the core, either "A" or "B." Thus, for example, SA-A stands for Seatuck, transect A, core A and WB-B would then stand for Wertheim, transect B, core B.

Table 1. GPS Coordinates for Transects

Transect	Starting coordinates	Ending coordinates
Seatuck-A	N 40° 42.533´, W 73° 12.557´	N 40° 42.539´, W 73° 12.662´
Seatuck-B	N 40° 42.650´, W 73° 12.540´	N 40° 42.649´, W 73° 12.591´
Seatuck-C	N 40° 42.650´, W 73° 12.556´	N 40° 42.674´, W 73° 12.556´
Wertheim-A	N 40° 46.494´, W 72° 53.614´	N 40° 46.505´, W 72° 53.509´
Wertheim-B	N 40° 46.073´, W 72° 53.346´	N 40° 46.121´, W 72° 53.260´

The approach for marsh analysis conducted in this study, as described below, was designed by the authors to address the needs of the Suffolk County mosquito vector and marsh management project. Specifically, it was necessary to account for fine-scale habitat changes (e.g., over scale of meters) that are manifested differently across the much larger spatial scales of the marsh (i.e., 100s to 1000s meters). Thus, along each transect a Dutch peat auger was used to take 50 cm long cores every three m to efficiently capture fine-scale vegetation changes over large distances. Each core was photographed with a digital camera at full 3.2 megapixel resolution for maximum detail. These cores were used to determine stratigraphy and plant root and rhizome composition, and the cores were placed back into the marsh once the desired notes and photographs were taken. This assessment method was highly effective because the anaerobic, salty soil of marshes limits the rate of decomposition of plant remains. Thus, plant traces are generally preserved in the peat (Niering et al., 1977). Different types of plants characterize different physical settings within a the salt marsh (i.e. wet or dry, and level of salinity) and therefore are a tool to not only reconstruct the history of the marsh's biology, but also its physical environment (Warren, 1995).

To capture large-scale variability at the marsh site, every 15 m along the transect, the core was divided into five cm sections and placed into bags. These cores were analyzed in laboratory for dry organic content and water content. The computer program Canvas was used to piece together the photographs of each core into a single image, thus providing a linear 50 cm deep cross-section of marsh peat, allowing for an assessment of qualitative changes during approximately the past 200 years (assuming an average 2.5 mm/yr rate of accretion) (Figures 6 to 9). Information from these transects, based on detailed field descriptions and interpretation of the photographs and quantitative measurements of water and organic content, were used to tentatively identify temporal and spatial changes at each marsh. The transect information also provided context for selecting the locations of the larger cores that were used to determine the detailed history of the marshes.

4.2 Cores

Representative coring sites were chosen in each study area by selecting areas with the same stratigraphic succession (ie., sediment layering), as recognized by color and root abundance. Such layering, regardless of cause or origin, is clearly defined over large distances of the marsh (10s meters) and provides strong visual support for the general representativeness of a given site (see Figures 6 to 9). Local anomalies, such as marsh edges and ditches, were avoided except where noted. At each coring site, large-volume cores (50 to 60 cm in length and 16 cm in diameter) were taken using a PVC pipe. Care was taken to avoid compaction of the core during the extraction process. Upon removal from the ground, the cores were left inside the PVC pipe and the top was sealed with a test plug while the bottom was wrapped in plastic and duct tape in order to secure the core for transport back to the laboratory. This general coring approach is a slight modification of that used by McCaffery and Thompson in their seminal study of sedimentation processes in a Connecticut salt marsh (McCaffery and Thompson, 1980).

A total of four cores were taken at Wertheim, two along each transect, and two cores were collected at Seatuck along transect A (Table 2). Because Seatuck is a significantly smaller marsh than Wertheim and the transect information yielded a fairly uniform marsh history, only two cores were taken at Seatuck for detailed analyses.

Table 2. Locations of Marsh Cores

Core	Coordinates
SA-A	N 40° 42.532´, W 73° 12.586´
SA-B	N 40° 42.530´, W 73° 12.632´
WA-A	N 40° 46.512´, W 72° 53.581´
WA-B	N 40° 46.497´, W 72° 53.562´
WB-A	N 40° 46.099´, W 72° 53.294´
WB-B	N 40° 46.113´, W 72° 53.274´

In the laboratory, the large cores were extracted from the pipes and sectioned into one cm thick increments for the top 20 cm and two cm increments for the remainder of the core. A large, serrated knife was used to slice the cores, and both a ruler and caliper were used in order to ensure the proper width. Each section was placed in a labeled bowl and homogenized using hands and/or spoons in order to ensure uniform subsamples. The sample was then subdivided into three sealed bags for subsequent organic/water content analyses, ^{210}Pb and ^{137}Cs radioisotope measurements, and sediment grain size. Samples were stored at 4°C until needed for analysis.

4.3 Water and Organic Content

Water and dry organic content was determined by weighing approximately 10 g of wet sample in a ceramic crucible and then drying it in a 60°C oven for at least 24 hours. A temperature of 60°C was not exceeded to avoid dewatering of clay minerals, which can give an anomalously high porewater content for the sample. At the end of the drying period the samples were reweighed to calculate loss of water and its percent weight. Dried samples were then placed in a muffle furnace for six hours at 450°C in order to burn off the combustible organic component. Six hours is sufficient for complete reaction of combustable material and the 450°C was not exceeded because carbonate minerals can combust at temperatures exceeding this level. After crucibles cooled sufficiently for handling, they were reweighed. Water content and dry organic content, also referred to as loss on ignition (LOI), were calculated using the following standard equations:

$$\%H_2O = \left(1 - \left(\frac{drywt}{wetwt}\right)\right) \times 100 \qquad (1)$$

$$\%LOI = \left(1 - \left(\frac{combustedwt}{drywt}\right)\right) \times 100 \qquad (2)$$

4.4 Radioisotope Dating

Sediment dating was achieved using both alpha and gamma spectroscopy of ^{210}Pb (half-life [$T_{1/2}$] = 22.3 yr) and ^{137}Cs ($T_{1/2}$ = 30 yr) geochronometers. Analysis by alpha spectroscopy assesses ^{210}Pb activity via its direct granddaughter decay product, ^{210}Po, whereas analysis by gamma spectroscopy involves direct measure of ^{210}Pb and ^{137}Cs isotope decays (Cochran *et al.*, 1998). Given a half life of 22.3 yr, the naturally occurring radioisotope ^{210}Pb can be used to date sediments as far back as 100 years, although the best accuracy is achieved within the past 80 years (Orson *et al.*, 1998). ^{210}Pb is a product of the ^{238}U decay series, with its long-lived parent isotope being ^{226}Ra ($T_{1/2}$ = 1,602 yr). A precursor to ^{210}Pb is ^{222}Rn gas ($T_{1/2}$ = 3.8 days), which is released to the atmosphere from the ground where it decays to ^{210}Pb. Lead in the atmosphere is subsequently scavenged via precipitation and dry deposition and is supplied to the marsh where it adheres to particle surfaces, because lead is strongly particle reactive under most near-surface geochemical conditions (Armentano and Woodwell, 1975). Supported levels of ^{210}Pb are produced within the sediments from the decay of ^{226}Ra which naturally exists in sediments (Appleby and Oldfield, 1992). Sediment dating with ^{210}Pb was used to determine ages and calendar years for the sediment. In turn, sediment ages were used to calculate short-term accretion rates. These short-term accretion rates are the basis for the "constant flux" method, which assumes that the supply of ^{210}Pb to the marsh surface has been constant with time. Long-term average accretion rates were derived from ^{210}Pb by plotting a regression line of the natural log of the activity in a given core (positive values only), also known as the "constant activity" method (Appleby and Oldfield, 1992; Armentano and Woodwell, 1975; Cochran *et al*., 1998).

^{137}Cs dating is based on the known chronology of atmospheric inputs of cesium from the testing of nuclear weapons. Atmospheric testing started in the early 1950s and peaked in 1963, just before the introduction of the 1964 Test Ban Treaty (Orson et al., 1998). Therefore, ^{137}Cs is a useful isotope for dating sediment accreted between 1963 and present day. Since these cores were taken in 2004, approximately the past 40 years could be dated using ^{137}Cs. In order to calculate an average long-term accretion rate (cm/yr) since 1963, the depth in the core associated with the ^{137}Cs peak was divided by 41 years. In particular it is useful to run both ^{210}Pb and ^{137}Cs analyses on any given core so that one may be used against the other as a check. Although ^{137}Cs dating has been criticized due to potential diffusion of Cs under strongly reducing conditions (which can occur in marsh sediments), the discernible peak that results using this method suggests that it is useful, especially when coupled with ^{210}Pb dating (Cochran *et al*., 1998; Roman *et al*., 1997).

Core WA-A was analysed by alpha spectrometry, which yield data for ^{210}Pb only. The remaining five cores were dated using both ^{210}Pb and ^{137}Cs. For core WA-A ^{210}Pb analysis was achieved using a complex series of acid digestions in order to prepare the samples for alpha spectrometry, based on the original method developed by Flynn (1968). Briefly, acid digestions consisted of drying and grinding enough of each sample to produce approximately one g of sediment and adding to it 0.5 ml of ^{209}Po as a tracer. Next, digestions involving strong acids (HCl, HNO_3, and HF) were executed and heated in Teflon beakers on hotplates for specified periods of time. Lastly, a silver disk was used in the final digestion in order for the ^{210}Po to spontaneously adhere itself to the disk. The disks were then placed in a counting chamber and after approximately 24 hours the ^{209}Po and ^{210}Po integrals were recorded, determined from multi-channel-analyzer data. ^{210}Po, the granddaughter of ^{210}Pb, was used as a proxy for ^{210}Pb.

To prepare samples for gamma spectroscopy, a large amount of sample (between 50 and 100 g) was dried and twigs and other organic material were removed before grinding the sample with a mortar and pestle. Next, the samples were weighed and then sealed in aluminum cans using a manual canning device. If there was not enough sample to fill the can completely, commercial table salt (sodium chloride) was added to make up the volume difference. Samples were then counted on an intrinsic germanium detector for approximately 24 hours and the ^{210}Pb (46.5 Kev), ^{226}Ra (352 Kev) and ^{137}Cs (661.6 Kev) peak areas were recorded. Since the samples themselves can absorb gamma emissions, which in turn affects the counting efficiency of the detectors, a standard with known activities was counted on each detector. In addition, self-absorption was determined by counting for 10 seconds a hot source of ^{241}Am on each sample (Cutshall *et al.*, 1983). The ^{241}Am was also counted on a blank can five times (10 seconds each) and averaged. An efficiency versus transmission relationship was determined for each detector. The following equations were used to calculate total and excess ^{210}Pb activity for each sample based on the methods of Joshi (1987):

$$\text{Total } ^{210}\text{Pb Activity} = \frac{(\text{\# counts} - \text{background counts}) * 60 \text{ sec/min} * (\text{dpm/cpm})}{\text{time in seconds} * \text{mass of sample}} \qquad (3)$$

$$\text{Excess } ^{210}\text{Pb Activity} = \text{Total Activity} - {}^{226}\text{Ra supported Activity} \qquad (4)$$

The excess ^{210}Pb inventory was calculated using the following equation:

$$Q = \sum_i (\Delta_{xi} r_i A_i) \qquad (5)$$

Where:

Q = excess ^{210}Pb inventory
?$_x$ = thickness of interval (for gamma this was always 2 cm)
ρ = dry bulk density of the sample
A = excess ^{210}Pb activity of sample

Sediment ages were estimated using the constant flux method which assumes that the flux of ^{210}Pb to the marsh has remained constant over time, thus allowing for different accretion rates (Armentano and Woodwell, 1975). Using the constant flux method, an age for the base of each sample was calculated, using the following equation from Armentano and Woodwell (1975):

$$Q_i = Q_0 e^{-l t} \qquad (6)$$

Where:

Q_i = inventory of excess ^{210}Pb below depth interval i
Q_o = total inventory of ^{210}Pb in entire core
? = constant decay rate of ^{210}Pb
t = horizon i

Calendar years were assigned for each depth interval by subtracting the age of the sediment (t) from the year the core was taken (2004). Marsh accretion rates were determined for each depth interval using the following equation:

$$S_i = \frac{X_i - X_{i-1}}{t_i - t_{i-1}} \qquad (7)$$

Where:

S_i = sediment accretion rate at the specific interval
i = depth interval
X_i = depth of horizon i
t_i = age of horizon i

4.5 Grain Size Analysis

Grain size distribution is used as an indicator of sediment source and energy regime of the marsh. For example, sand is often an indicator of a high energy event, such as a large storm or hurricane (Donnelly *et al.*, 2001). Grain size analysis was performed at five cm intervals on wet samples from each core. Samples for grain size distribution were sieved through a 63 µm screen, using deionized water, thus allowing silt and clay particles to pass through. Approximately 15-30 g of sample were used, depending on the amount of organics in each one. Larger grains (sand) and organic material were collected on top of the sieve, and significant sand inputs were dried and saved in plastic bags.

Once each sample was sieved, it was transferred to a beaker and a small amount of household bleach was added to oxidize any remaining organics. Samples were stirred daily for 5-7 days, after which the water/bleach solution was decanted. Successive baths in deionized were administered to rinse the sediment. After 24 to 48 hours of allowing the sediment to resettle in its cleaned water, the sample was again decanted and concentrate by centrifuging at 4,000 rpm for 10 minutes. Sediments were reconstituted in 0.05 percent calgon solution (containing sodium hexametaphosphate, a dispersant) to prevent particle aggregation during analysis. Just prior to analysis, samples were sonicated for 10 to 15 minutes and introducted to a SediGraph 5100 particle size analyzer to determine the size-distribution of particles < 63 µm in size. For the interest of this project, size classes were recorded as coarse-medium silts (20 to 63 µm), fine silts (four to 20 µm) and clays (less than four µm).

5. Results

5.1 Field Transects

5.1.1 Surface Vegetation and Paleobotany

Transects were chosen in order to encompass a representative sample area of the particular marsh community. At Wertheim, transect W-A was set up in OMWM Area 1 and W-B in OMWM Area 2 (Figure 4). W-A is in an upstream area of the marsh and ran perpendicular to the bay. It encompassed an open, relatively undisturbed area that included mixed surface vegetation, including *S. patens*, *S. robustus*, *D. spicata*, *P. purpurascens*, *H. palustris*, and a scattering of *P. australias*. Multiple cores along the transect revealed dense roots and organic rich peat of primarily *S. patens* from zeo to 20 cm, with a transition to *S. robustus* at approximately 20 to 25 cm. In general, from 30 to 50 cm the cores became muddier, with less organics and varying amounts of *S. robustus* throughout. It was noted that the cores showed a thinning of peat density starting at approximately 20 cm which continued upward to the surface.

Transect W-B was about 500 m upstream of the bay and ran normal to the shoreline toward the upland. At approximately 66 m the transect traversed a one of the regularly spaced mosquito ditches that cross-cut the marsh (visible in Fig. 4). The surface vegetation was characterized by a mixture of *S. patens* and *S. alterniflora*, with *S. alterniflora* dominating along the shoreward portion of the transect. Cores revealed a generally uniform stratigraphy along the transect with the general pattern being dense organics in the upper 20 cm, transitioning to less compacted *S. patens* peat and/or *S. robustus* in the lower 30 cm of the cores. In several of the cores, *S. alterniflora* was observed between 20 and 30 cm.

At Seatuck, transect S-A was nearest to the bay and ran oblique to the shoreline, extending from the ditch along South Bay Avenue eastward to the backbeach zone of dense *Iva frutescens* (Figure 5). The surface vegetation was largely a mix of *S. patens* and *S. alterniflora*, with isolated stands of *P. australis*. The transect traversed a cross-cutting mosquito ditch at approximately 93 m. Cores collected along the transect differed in respect to their overall stratigraphy and preserved roots and rhizomes. However, the general pattern consisted of dense peat (typically *S. patens*) for the top 20 to 30 cm, transitioning to *S. alterniflora* in some cases, or remaining as *S. patens*, and ending with a fresh water peat layer beginning at approximately 45 cm. At distances of 45 m and 90 m along the the transect, *P. australis rhizomes* were found at depths of 12 to 20 cm (actual depth depended on specific core). East of the ditch located at 93 m, several of the cores contained varying sand layers beginning at approximately 20 cm.

Transects S-B and S-C were located about 200 m inland of S-A (Figure 5). S-B ran roughly parallel to the bay, from the drainage ditch to the upland shrubbery. Since this transect was only 72 m long, transect S-C was set up perpendicular to S-B, beginning at the 45 m mark on S-B. S-C was 45 m in length. Similar to transect S-A, the surface vegetation at S-B and S-C was characterized by a mix of *S. patens* and *S. alterniflora.* There were many low lying spots and small ponds that interrupted the transects. The plants and stratigraphy of the cores were fairly uniform across the two transect (Figure 5). The top 20 cm revealed a mixture of *S. patens* and *S. alterniflora* with *P. australis* interspersed in many of the cores at this interval. In transect S-C, *P. australis* was more prevalent and often continued to depths of 40 cm. Between 40 and 45 cm a black fresh water peat layer was identified. *S. patens* dominated at mid-depths between 20 and 40 cm.

5.1.2 Organic and Water Content

Organic and water content analyses reveal more significant trends at Wertheim than at Seatuck (Figures 10 to 11 vs. 12 to 14). For both transects at Wertheim, percent water and LOI data covaried, which is expected given the high water content of organic matter. That is samples with lower water content tended to also have lower organic content and vice-versa. Organics range from 9 to 82 percent for transect W-A. Transect W-B does not show quite as large a range of organics, falling between 11 and 62percent. At Wertheim, both transects A and B show a detectable and significant increased shift in organic content beginning at approximately 25 cm and increasing to the surface. Percent water is relatively uniform through both time and space, being reported at 64 to 90 percent for W-A and similarly at 59 to 83 percent for W-B.

At Seatuck, organic content shows a range of values similar to those found at Wertheim; however, unlike Wertheim there is no noticeable trend in organic content. For all three transects at Seatuck, LOI ranges from zero to 77 percent. Water content has a much broader range for S-A than it does for S-B and S-C, mainly because of sand deposits found below the base of the marsh deposits. For S-A water content ranges from 16 to 87 percent. In comparison water content falls in a much narrower range for S-B and S-C at 58 to 89 percent (with just two points recorded at approximately 30 percent), which are similar to those values found at Wertheim.

5.2 Cores

5.2.1 Paleobotany

Large diameter cores taken from pre-selected areas of each marsh allowed for more detailed analyses to be performed. Review of the digital photographs, along with qualitative observations made in the field, suggested that the beginning of transect WA had "fuller" cores (the cores were cylindrical in shape and completely filled the volume of the coring device) when compared to the second half of the transect. These latter cores appeared to "thin" out (become less dense with peat/sediment and not fill the volume of the coring device) between zero and 20 cm depth. The two most likely reasons for the differences are:

- an artifact of the coring technique (with no underlying cause for the difference)
- true physical or ecological changes in the collected materials, related to environmental variability (results support the latter, see Section 6.1.1)

The potential for capturing differences in underlying core materials dictated the selection of the large diameter core locations. Core WA-A was taken at approximately 24 m along the transect and core WA-B was taken at about 72 m along the transect. Core WA-A revealed densely matted roots of *S. patens* from zero to 25 cm; from 25 to 50 cm, the core consisted of a mix of *S. patens* and *S. robustus*. Similarly core WA-B consisted of *S. patens* from zero to 20 cm, below which the core transitioned to muddy sediments with *S. robustus* throughout.

For transect WB, core WB-A was taken just landward of the ditch (approximately 75 m along the transect) and core WB-B was taken farther upland, at 114 m in an open expanse of *S. patens*. Core WB-A had dark brown to black soil, with dense roots of *S. alterniflora* from zero to 25 cm, and preserved roots and rhizomes shifted to primarily *S. patens* for the second half of the core (from 25 to 50 cm) in a muddy, light brown soil. Farther inland, core WB-B had light brown to brown colored soil and contained dense roots of *S. patens* from zero to 30 cm; from 30 to 50 cm the core became muddier and contained mainly *S. patens*, mixed amongst some *S. robustus*.

At Seatuck, core SA-A was taken at approximately 48 m along the transect and core SA-B was taken on the far side of the *I. frutescens* border and just beyond the ditch, at about 99 m. Core SA-A had several distinct transitional zones. Between zero and 20 cm there was a mix of *S. patens* and *S. alterniflora*, with *P. australis* rhizomes between 20 and 30 cm; 30 to 45 cm consisted of *S. alterniflora*, and the last five cm of the core (45 to 50 cm) was a fresh water peat layer, characterized by black peat and fine, fresh water plant roots and rhizomes. In addition, some sand was detected at depths of 15 to 16 cm and near the bottom of the core at 48 to 50 cm. Core SA-B proved somewhat of a surprise, in that the lower half of it consisted mainly of sand. Care was taken not to take cores directly next to ditches due to the possibility that dredge spoils were deposited there. Although this core was taken at what was thought to be an appropriate distance from the nearby ditch, it may not have been far enough away to avoid dredge spoils. Conversely, there may be another explanation for the sand. The top 20 cm of core SA-B consisted mainly of *S. patens*, with a small amount of sand detected in intervals 10 to 11 cm and 15 to 16 cm. Sand dominated from 20 to 38 cm, and 38 to 40 cm was a thick clay layer.

5.2.2 Organic and Water Content

As expected, organic and water content data for large diameter cores are similar to those values obtained for their respective transects. At Wertheim, all four cores show a positive correlation between LOI and percent H_2O (Figures 15a-b). For cores WA-A and WA-B organics range from 16 percent to 73 percent and for cores WB-A and WB-B, organics range from a low of 16 percent to a high of only 57 percent (Figures 16a-b). Percent water was similar between all four cores, ranging from 72 percent to 88 percent for cores WA-A and WA-B and from 62 percent to 85 percent for cores WB-A and WB-B (Figures 17a-b). Organic and water content data for all four of these cores show an increased shift starting at approximately 25 cm and continuing to the surface. More specifically, moving upward in the core from a base depth of 50 cm to a mid depth of 25 cm organic and water content in general decrease; starting at 25 cm and continuing to 10 cm organic and water content increase and from roughly 10 cm to the surface these values remain in a relatively constant state.

At Seatuck, core SA-A yielded LOI values ranging from 22 to 78 percent, similar to values reported for Wertheim (Figure 18). Core SA-B shows LOI values of two to 80 percent, with all values reported at 18 cm to the bottom of the core being two percent due to this part of the core consisting primarily of sand. Disregarding the sand intervals, all other LOI values (zero to 18 cm) fall within those values reported for SA-A. In addition, organics appear to increase from 18 cm to the surface (Figure 19). Percent water for SA-A was 70 to 85 percent, with higher water content (82 to 85 percent) being reported for 15 cm depth to surface. Water content for SA-B ranged from 20 to 88 percent, with 20 percent being reported from 18 cm to the bottom of the core, again due to the significant amount of sand found here. From 18 cm to the surface, water content increased (Figure 20).

5.2.3 Mineral Fraction versus Organic Flux

In order to determine the primary contribution to marsh accretion, the fluxes of organic and mineral fraction ($g/cm^2/y$), at specific depth intervals for each core, were calculated and an organic to mineral ratio was produced. Organic and mineral fluxes were calculated only to the supported ^{210}Pb activity levels of each core since the equation involves the accretion rate which is accurate only to supported levels of ^{210}Pb. Figures 21a-c show organic to mineral fraction ratios for each core. A ratio less than one means the marsh is accreting primarily by mineral fraction input; a ratio equal to one indicates the marsh is accreting equally by mineral fraction and organics; a ratio greater than one indicates that the marsh is accreting predominantly via organics. Cores WA-A, WA-B, SA-A, and SA-B in general show the respective marsh accreting primarily by organics in the upper parts of the cores, compared with a more mineral-rich fraction dominating the deeper core sections. Notably, cores WB-A and WB-B are distinct from the other four cores, in that they are accreting primarily via mineral deposition.

5.2.4 Radioisotope Dating

5.2.4.1 Pb-210

In general, excess ^{210}Pb activities follow exponential decay curves (Figures 22a-c). The ^{210}Pb profile for core WA-A exhibits high resolution due to intervals being only one cm thick for the top 20 cm of the core (compared to two cm intervals in all other cores). Plots of ^{210}Pb activity show that excess activity is discernable to a base depth of 22 cm for cores WA-A and WA-B and to a base depth of 26 cm for core WB-A and a base depth of 22 cm for WB-B. Excess activity is discernable to a base depth of only 20 cm for core SA-A and to 14 cm for core SA-B. Below these points where activity levels no longer decrease with depth, supported activity levels are less than one dpm/g.

Core sediment age, determined by the constant-flux model, is plotted against depth (Figures 23a-c). Data show a linear increase in age with depth for approximately the top 15 cm of each core, after which this linearity becomes less defined. A calendar year is assigned to each depth interval above the supported levels. Given that the half life of ^{210}Pb is 22.3 yr, the past 80 to 100 years can accurately be dated, thus providing a timeline by which other measured variables can be correlated. Ages are used to calculate short-term accretion rates via the constant flux model. The constant flux model shows short-term accretion rates in all cores shifting from low to comparatively higher accumulations starting at approximately 12 cm and continuing to the surface (Figures 24a-f). This depth interval roughly corresponds to the later half of the 20th century. Specifically, cores show short-term accretion rates generally being less than 0.20 cm/y for the first part of the 20th century and between 0.20 and 0.45 cm/y for the latter half of the century. This general increase in accretion rate has been noted in other studies on Long Island, and is not believed to be a function of compaction because there are no corresponding changes in percent water or bulk density of the sediments (Kolker, 2005).

Average long-term accretion rates are fairly consistent among the cores, with the exception of core WB-A which yields a slightly higher accretion rate. Long-term accretion rates via ^{210}Pb appear in Table 4. A comparison of average long-term accretion rates to Battery Station (Manhattan, New York) tide gauge records was done in order to determine if marsh accretion is keeping pace with long-term sea level rise of 0.30 cm/yr (Figure 25).

Inventories for excess ^{210}Pb activity are very consistent between the cores, with the exception of core WB-A which yields a comparatively higher inventory value. Inventory values appear in Table 3. Inventories of excess ^{210}Pb for north-eastern US salt marshes should be between 22 and 38 dpm/cm^2, assuming input is entirely from the atmosphere (Cochran *et al.*, 1998). Inventories reported for this study fall within this range, with the exception of core WB-A (low marsh core) whose inventory is reported as 47.74 ±0.4 dpm/cm^2.

Table 3. Excess Inventories for ^{210}Pb Activity

Core	*Inventory for excess ^{210}Pb activity (dpm/cm^2)*
WA-A	29.0 ±0.36
WA-B	28.8 ±0.27
WB-A	47.7 ±0.43
WB-B	34.1 ±0.32
SA-A	28.8 ±0.35
SA-B	27.8 ±0.32

5.2.4.2 Cs-137

^{137}Cs measurements were made for all cores except core WA-A, which was analyzed for ^{210}Pb only. By dividing the depth interval (cm) peak of cesium activity by 41 years (time passed since the 1963 peak of nuclear-weapons testing) an average long-term accretion rate can be determined. The five cores analyzed for cesium all show a well-defined peak, which is taken to correspond to the year 1963 (Figures 26a-c). Cesium peaked in core WA-B at 12 cm depth, yielding an accretion rate of 0.29 cm/yr; core WB-A peaked at 13 cm, yielding an accretion rate of 0.32 cm/yr; core WB-A also shows a smaller, yet just as distinctive, peak at 18 cm depth which most likely corresponds to 1954 testing inputs. Core WB-B peaked slightly shallower in the core at eight cm, giving an average rate of only 0.20 cm/yr. At Seatuck, core SA-A peaked at 10 cm, giving an accretion rate of 0.24 cm/yr and core SA-B peaked slightly shallower in the core at seven cm, yielding an average long-term accretion rate of only 0.17 cm/yr. In comparing long-term accretion rates derived from ^{137}Cs to those derived from ^{210}Pb, it is evident that both methods yield similar values for a given core (Table 4). In addition, calendar dating of the sediment puts the 1963 ^{137}Cs peak either exactly at or relatively close to similar dates derived from ^{210}Pb (Table 5).

Table 4. Average Long-Term Accretion Rates Derived from Radioisotope Dating

Core	*^{210}Pb (cm/yr)*	*^{137}Cs (cm/yr)*	*Average (cm/yr)*
WA-A	0.23	n/a	n/a
WA-B	0.27	0.29	0.28
WB-A	0.41	0.32	0.37
WB-B	0.24	0.20	0.22
SA-A	0.20	0.24	0.22
SA-B	0.25	0.17	0.21

Table 5. Comparison of Calendar Dates Derived by ^{137}Cs and ^{210}Pb Radioisotopes

Core	*Depth Interval (cm) for **1963** ^{137}Cs peak*	*^{210}Pb Constant Flux Date at same interval*
WA-B	10-12	1962
WB-A	12-14	1962
WB-B	6-8	1969
SA-A	8-10	1962
SA-B	6-8	1955

5.2.5 Grain Size Analysis

Grain size analysis shows no consistent differences within or between sites, and although there is some variability of grain size within each core it is within a narrow range (Figures 27a-c). Analysis reveals that cores consist primarily of silt and clay, with the exception of core SA-B that has significant amounts of sand from 18 cm to the base of the core. Other detectable sand deposits (defined as perceptible sand grains while sieving sample) are found only in core SA-A, at intervals of 15 to 16 cm and near the bottom of the core at 48 to 50 cm. Due to technical difficulties, no data was obtained for core WA-B. With the exception of core SA-A, which shows nearly 100 percent clay (4µm) at 39 cm depth, no significant grain size pattern is discernible.

6. Discussion

6.1 Vegetation Changes

6.1.1 Wertheim

The present day surface vegetation at Wertheim indicates that the marsh transitions from a wetter *S. alterniflora* dominated marsh in the lower intertidal zone, to a higher and drier marsh characterized by *S. patens* and its associated high marsh plant communities. Area 1 of Wertheim (transect A) has a diversity of brackish high-marsh vegetation co-existing with *S. patens* whereas Area 2 (transect B) transitions from a classic intertidal low marsh (indicated by pure stands of *S. alterniflora*) to an expansive area of solely *S. patens*, exhibiting its picturesque cow-licked appearance (Redfield, 1972). The presence of *P. australis* mainly at the margins of the marsh and near ditches (as opposed to the interior of the marsh) suggests that it has invaded the marsh via edge effects such as increased elevation from dredged or sidecasted soils.

Cores from these two areas show a shift in plant community at 20 to 30 cm, corresponding to an age of early 1900s based on radioisotope dating. In general, the marsh has transitioned during the past century from a wet and brackish marsh, as indicated by abundant *S. robustus* found in the bottom half of the cores, to a drier and more saline marsh as indicated by the salt marsh plants *S. patens* and S. *alterniflora* found in the top half of the cores. Organic content data for transects and cores show a positive shift in organics corresponding to the same interval where there is a shift in plant community structure. More specifically, the plots of percent LOI vs. depth for the four large diameter cores (Figures 16a-b) show a decrease in organics from 50 cm up to approximately 25 cm (roughly the early 1800s to early 1900s), and then proceed to show an increase in organics from this mid depth to the surface. Organic matter is typically highest at the marsh surface and decreases or remains unchanged at depth due to the process of decomposition (Cochran *et al.*, 1998). Similarly, the plots of percent LOI vs depth for the transects where these cores were taken (Figures 10 and 11) show a positive shift in organic content starting at approximately 25 cm. Together, the two sets of organic content data present strong evidence that a major transition, whether anthropogenic or environmental in origin, occurred in this marsh during the early part of the 20th century (radioisotope dating discussed in greater detail in a later section).

Plots of percent water for both transects and cores (Figures 10, 11, and 17a-b) show a positive shift in water content at 25 cm, again corresponding to the early 20th century, similar to that seen in the organic data, although not quite as obvious. By plotting percent LOI with percent H_2O (Figures 15a-b) it is evident that there is a positive correlation between organics and water. This trend can be explained by the fact that organics act like a sponge and absorb water. Thus, intervals that have higher organic content will naturally have higher water content and vice versa.

6.1.2 Seatuck

The surface vegetation at Seatuck consists mainly of a mix of *S. alterniflora* and *S. patens*, with a substantial amount of *P.australis* seen invading the marsh. The abundance of *P. australis* at Seatuck is most likely attributed to a combination of suppressed tidal inundation, lack of drainage for fresh water after heavy rain storms, and major edge modifications caused by installation of deep canals and ditches. An increase in *P. australis* in many local salt marshes during the past 100 years has led to the perception that it is an introduced species. However, as Orson (1999) points out, *P. australis* has been a member of the New England plant community for thousands of years, and it has been suggested that its spread is a consequence of recent environmental changes and anthropogenic impacts. However, more recent studies have suggested that there is an introduced genotype of *Phragmites* that is a more aggressive invader than the native variety (Saltonstall, 2003). It is not clear which of these factors might play a dominant role in the spread of *P. australis*, but the interpretations are not mutually exclusive and it is likely that both may be important.

Bertness *et al.* (2002) attribute the invasion of *P. australis* to nitrogen eutrophication, and further suggest that over 90 percent of *Phragmites* expansion in Narragansett Bay, Rhode Island can be related to increasing shoreline development. Their study also notes that salt marshes without developed shorelines have less than 15 percent of their terrestrial borders dominated by *P. australis*. Given the fact that Seatuck borders a town beach and a highly trafficked road that is lined with many houses, it meets Bertness's definition for conditions favorable to the spread of *P. australis*. Seatuck has a long history of large-scale human disturbances, dating back to the 1800s. The most severe disturbance has been the fragmentation of what was once uninterrupted marsh, extending from Champlin Creek west to Orowoc Creek, which was once on the far west side of the town beach. Filling in of Orowoc Creek and subsequent development cut off tidal exchange to the west side of the marsh. Following this event, in the late 1800s three roads were built through the marsh, St. Marks Lane, South Bay Avenue, and Old Corduroy Road (see Figure 3), further fragmenting the marsh (Cowan *et al*., 1986). These major disturbances have altered the hydrodynamics of Seatuck and these changes are evidenced in the shifts in plant community structure during the past 200 years.

The disturbances at Seatuck have caused the drainage of fresh water to be restricted. The impoundment of this water provides an additional catalyst for the spread of *P. australis*. Evidence for poor drainage was noted following a heavy rainstorm when the marsh surface was inundated by approximately 20 cm of largely fresh water (as determined by taste). Further investigation revealed a blocked culvert/drainage pipe that was preventing the fresh rain water from efficiently draining from the marsh. Lowering a marsh's salinity can favor the continued establishment of *P. australis*, as it thrives in fresh to mesohaline water (zero to 18 ppt) more than in polyhaline water (18 to 35 ppt) (Saltonstall 2003).

Examination of core SA-A allows for the history of Seatuck marsh development during the past 100 years to be reconstructed. As seen in Figure 24e, the bottom of the core consists of a fresh water peat layer over which there is a layer of *S. alterniflora*, thus suggesting a transition from a fresh water marsh system to a tidal marsh within historical times. An extrapolation of dates determined from radioisotope dating places the fresh water peat layer in the beginning of the 1800s, but a more precise date is not possible below the depth of measurable radioisotope activity. This transition likely reflects landward translation of the shoreline due to natural coastal erosion, which ultimately led to increased tidal flushing of a fresh water marsh system, raising salinity and encouraging transition to a more salt-tolerant plant community. This is supported by the lack of changes in sediment grain size that might suggest a major anthropogenic disturbance or high-energy storm event. The top 20 cm of the core, which represents the majority of the 20th century, reveals a mixture of *S. patens* and *S. alterniflora*. This suggests that Seatuck has persisted as a salt marsh habitat for much or most of the 20th century. Notably, there are *P. australis* rhizomes found just below this transition at 20-25 cm depth, but these most likely postdate the early 20th century. *Phragmites* rhizomes typically grow at depths of 20 to 40 cm, and up to 100 cm below the surface (Uchytil, 1992), making them poor time markers and difficult to accurately assess time of spread based on root depth alone. However, the dominant rhizome biomass of *S. alterniflora* is found shallower than 15 cm, making it a more reliably indicator of time of change (Valiela et al., 1976). Significant ditching of Seatuck began in 1914 (Havemeyer, 1996), suggesting that the system had already converted to a salt marsh community, as grid ditching was not commonly undertaken for non-tidal wetlands.

Water and organic content data provide further evidence for these shifts in plant communities. Just prior to the 20th century, percent water decreases slightly before shifting to a higher water content for the upper portion of the core, which contains the *Spartina* plant species (Figure 20). This decrease in water content suggests that the transition to a salt marsh habitat was accompanied by an increase in relative contribution of mineral sediments (which hold less water), which might be expected due to initiation of regular tidal flushing and sediment import. This interpretation is further supported by LOI trends showing the lowest organic content from 20 to 35 cm, reflecting dilution by mineral matter (Figure 19).

The mix of surface vegetation suggests that the marsh is either not functioning as a true high or true low marsh, or that it could be in a transition state, moving from one *Spartina* species to the other. Redfield (1972) notes that the boundary between *S. patens* and *S. alterniflora* is often not defined and it is not uncommon for these two species to coexist in the high and/or low marsh. However, he also notes that a true high marsh that is well drained will be dominated by *S. patens* and its low marsh will consist of pure stands of *S. alterniflora*. Thus the present surface vegetation mixture of *S. patens* and *S. alterniflora* at Seatuck is consistent with it being poorly drained. In such conditions, local ponding of water will lead to localized and patchy distribution of *S. alterniflora* and *S. patens*.

Core SA-B comprised almost exclusively *S. patens* plant remains, suggesting relatively little environmental change in this specific area of the marsh. Although Phragmites has invaded extensively in adjacent areas, this site has been dominated by high marsh S. patens since its development. The marsh peats directly overlie outwash sands, indicating the classic succession of salt marsh over a transgressed upland surface, as described by Redfield (1972). Based on extrapolation of the radioisotope dates, the development of salt marsh habitat at this site occurred in the early 1800s, about the same time that salt marsh vegetation becomes established at site SA-A nearer to the shoreline. Overall, these findings are consistent with salt marsh development at Seatuck by at least the early 1800s, followed by mixed plant transitions during the 20th century, apparently associated with successive anthropogenic modifications of the site.

Although it was hypothesized that OMWM modifications made to Seatuck during the mid 1980s potentially altered marsh functioning, there was no strong evidence for this in the peat cores. This lack of finding is due in part to only three to five cm of marsh accretion during the past 20 years, given the long-term accretion rates of 0.17-0.25 cm/yr derived from radioisotope dating. Perhaps it is too soon to tell from the marsh record how this system will ultimately adjust to these anthropogenic alterations, as suggested by the apparently transitional state of *S. patens, S. alterniflora, and P. australis* plant distributions. However, consequences of the 1980s OMWM project, and neglected follow-up/maintenance, are evidenced by the observed lack of drainage of fresh water, restricted tidal flow, and apparently favorable conditions for invasion of *P. australis*.

6.2 Radioisotope Dating

6.2.1 Sediment Ages

Using ^{210}Pb dating, the base depth of each interval was assigned an age and to this a corresponding calendar year. In general, ages appear to agree among the cores for the top few intervals and then become less defined moving down the cores, probably due to the diminishing accuracy of the ^{210}Pb given its half life of 22.3 yr. As shown in Table 5, ^{210}Pb dates match well with the depth interval of the 1963 ^{137}Cs peak, and thus by looking at the chronologies defined by ^{210}Pb in conjunction with extrapolation using the ^{137}Cs horizon, a decadal scale can be developed and matched to marsh changes.

6.2.2 Accretion Rates

Radioisotope dating via ^{210}Pb and ^{137}Cs yields similar average long-term accretion rates within each core and relatively similar values among the cores (Table 4). Overall, accretion rates for the two marshes fell between 0.17 cm/yr and 0.41 cm/yr. Variation in these rates are typical of marsh sedimentation processes, which are not steady-state but rather dominated by stochastic events and climatically forced variability (Kolker, 2005). Recent findings from Long Island also suggest that short-term variability in sedimentation is more significant in marsh systems in microtidal settings, where normal current velocities are too low for significant sediment transport and, thus, require some external forcing such as winds or waves to enhance transport capacity

(Goodbred and Hine, 1995; Kolker, 2005). Overall, marsh studies similar to this one show a comparable range of accretion rates using the ^{210}Pb dating technique. Cochran *et al.* (1998) reported rates of 0.1 to 0.4 cm/yr for their study of several Long Island salt marshes. Bricker-Urso *et al.* (1989) reported slightly higher accretion rates of 0.24 to 0.60 cm/yr for a low marsh study done at Narragansett Bay, Rhode Island, and similarly Roman *et al.* (1997) reported rates of 0.26 to 0.42 cm/yr for a study done at Nauset Marsh, in Massachusetts. Each of these values are reported with regularity from various marshes, but the accretion rates themselves do not appear to reflect health or susceptibility to marsh loss (Kolker, 2005). Thus, higher rates reported from the Rhode Island and Massachusetts studies should not be taken as a reflection of poor health or lack of function in the Long Island sites.

Core WB-A shows generally the highest rates of accretion among the three Wertheim cores, as observed via both ^{210}Pb (0.41 cm/yr) and ^{137}Cs (0.32 cm/yr). For comparison, accretion rates for the other cores ranged 0.20 to 0.29 cm/yr. However, since core WB-A was taken nearest the bay, it is likely that the comparatively high accretion rate for this core is due to the tides delivering sediments to this part of the marsh and the subsequent effectiveness of *S. alterniflora* in trapping delivered sediment. Although proximity to muddy tidal waters from the bay likely play a role in the higher sedimentation rates, the impact of storms should not be discounted (Reed, 1995). However, there are no significant variations in grain size, making it difficult to suggest the exact role that storms might play. Nevertheless, the generally greater accretion rates at this site are further reflected in the organic to mineral flux ratios (Figure 21b), which show that this area of the marsh has been accreting predominantly by mineral input for the past century.

In comparison, long-term accretion rates at Seatuck range from 0.17 to 0.25 cm/yr. Although core SA-B was analyzed only to a depth of 20 cm, in order to avoid the sand layers below this depth, the presence of sand in this core may explain why an accretion rate of only 0.17 cm/yr was achieved via ^{137}Cs. Cochran *et al.* (1998) suggest that a sandy substrate affects the retention of radioisotopes and trace metals atmospherically supplied to the marsh.

During the past 4,000 years, salt marshes have persisted in a variety of coastal settings and maintained an elevation at equilibrium with sea-level through accretion of mineral sediment delivered from tides and upland, as well as through organics produced as both above- and below-ground pplant biomass (Morris *et al.*, 2002; Reed, 1995). Together these processes allow the marsh to vertically accrete, and often one process dominates over the other. The constant flux model (Figures 24a-f) shows that in general both Wertheim and Seatuck have experienced higher accretion rates during the latter part of the 20th century, which is consistent with results from several other Long Island salt marshes (Kolker, 2005). The cause of increased accretion is not certain, but may be a consequence of rising sea level or increased climatic variability. Plots show the increase in accretion rates beginning on average in the late 1950s.

It is of interest to note that mid-century increases in accretion rates correspond roughly to a shift from mineral-dominated to organic-dominated accretion in four of the six cores. Cores WA-A, WA-B, SA-A, and SA-B all show the marsh accreting primarily via organics for approximately the past half century. In comparison cores WB-A and WB-B show the marsh accreting almost entirely via mineral fraction for the past 100 years, as a result of their close proximity to wave-resuspended sediments from Great South Bay. Thus, understanding the mode of accretion for a particular marsh, or indeed a particular area of a marsh, should be an important factor in designing specific marsh management plans, as method of accretion yields evidence as to how the marsh ultimately will maintain equilibrium with sea-level.

6.3 Changes in Marsh Form and Functioning during the 20th Century

6.3.1 Natural and Anthropogenic Impacts

Information obtained from plant community changes and radioisotope dating show evidence of widespread marsh changes in both the 19th and 20th centuries. In addition to parallel grid ditching and OMWM alterations, scenarios such as ongoing global climate change, natural closings and openings of inlets, upland development, and operation of duck farms can all be put forth as possible triggers for marsh change. Global climate change is an ongoing process that sets a background pace for vertical marsh accretion and the slow transgression of the shoreline. Such long-term changes may be responsible for the early 19th century development of salt marsh in formerly upland areas of Seatuck (transects B, C), as well as the similarly timed transition from fresh- to salt-tolerant plant communities in the upstream Wertheim site (Area 1, transect A). The history of inlet opening/closing in eastern Great South Bay (ie., “Old Inlet”) may have also played a role in this 19th century transition in parts of Wertheim to a widespread salt marsh plant community. However, no dating techniques are especially good for this time frame (between ^{210}Pb and ^{14}C capabilities), and it is not possible at present to link this as a clear causation.

When considering only the 20th century, the abrupt and widespread changes observed at Seatuck and Wertheim marshes do not appear to be correlated with any similar changes in climate or related phenomena. Thus, there is no evidence that these background processes have been a primary cause. Specifically regarding sea-level rise, results of this study show that the marshes in this study and elsewhere on Long Island (Kolker, 2005) have largely kept pace with relative sea-level rise during the 20th century. Although rates of accretion may be variable at time scales less than a decade, mean accretion rates over several decades to a century typically match or exceed the rate of sea-level rise. Therefore, it seems to be unlikely to be a primary factor in governing observed changes in the marshes.

It is worth noting that, although accretion rates remain relatively high during the 20th century, there is the prominent mid-century transition from mineral to organic-dominated accretion in Wertheim, and also somewhat at Seatuck. The cause of this change is not certain, but the change does demonstrate that marsh accretion can keep pace with sea-level rise via numerous modes. Two possible interpretions of this change to a more organic-rich peat development are:

(1) there has been a decrease in the amount of mineral sediment being delivered to the marsh surface, and/or

(2) plant productivity, particularly below-ground biomass, has increased.

Although sedimentation may play a role, the latter seems more likely to be a dominant mechanism for several reasons. First, accretion rates do not decrease after the transition, as would be expected if less sediment was being delivered to the marsh. Second, the change in accretionary styles is also associated with a shift towards increased dominance of a *Spartina patens* plant community. *S. patens* maintains most of its biomass below ground and thus typically produces the densest, most organic-rich peat of the marsh grasses (Turner et al., 2001). Thus, the question regarding marsh changes is not really an issue regarding the transition to more organic-rich accumulation, but rather the cause for spread of the *S. patens* community.

Although anthropogenic activities have apparently impacted climate change on a global scale since at least the mid 20th century (IPCC, 2001), such changes as noted above are probably too slow to have directly forced the sedimentary and floral changes observed in the Wertheim and Seatuck salt marshes. Rather, it is likely that local and regional-scale human activities, such as land use, coastal engineering, population growth, and urbanization, have had a more direct and significant impact on marshes during the 20th century. Among these, parallel grid ditching and OMWM activities were considered as possible mechanisms for observed changes at Seatuck and Wertheim. In addition, upland development and the operation of duck farms were researched for their possible link to observed marsh changes. Upland development is an important factor to consider given the fact that the population of Long Island has increased greatly during the latter half of the 20th century, including a five fold growth in Suffolk County since 1950 to nearly 1.5 million people today (www.quickfacts.census.gov).

Land use and upland development are other potentially important anthropogenic factors. In Wertheim, though, the system is much larger than Seatuck and has a correspondingly greater natural buffer from upland development. This buffer appears to largely preclude major influences on the marsh system. Specifically, changes in particulate loading to the Wertheim marshes are not likely to have been strongly affected by human activities in the drainage basin. For example, there has been discussion of potential sediment effects from road, dam, and bridge construction, as well as the small marina near the mouth. In terms of sediments, though, these are unlikely to have played a significant role because there is generally insufficient flow velocities in the river to support sediment transport. With regard to the marina, it is not especially large and unlikely to have led to any permanent or widespread change in sedimentation rates. With more upstream activities, it must be considered that extensive fresh water wetlands and lowland swamps likely intercept much of the limited sediment that the river may transport. Rather, sediment reaching the salt marsh is most likely reworked from Great South Bay by wave energy, wind-driven currents, and tides (Kolker, 2005). This notion is well supported by the bay-side core site (WB-A) having the fastest, most mineral dominated, accretion rates of any site by 50 to 100 percent. However, it is not clear what role dissolved constituents, such as nutrients derived from local groundwater flow (cesspools or fertilizers) or upstream duck farming, may play in marsh health and functioning.

It is possible that the former operation of two large duck farms in the Carmans River watershed could have contributed to the increased organic production recorded in Wertheim marsh peats during the latter half of the 20th century. The Robinson Duck Farm and Leskowicz Duck Farm were in operation between approximately 1930 and 1970 and had peak production during the 1960s (Dewitt Davies, Suffolk County Department of Planning, personal communication, 2005). In 1966, a total production of seven million ducks was reported for Suffolk County, with nearly half-a-million of these ducks originating in the Carmans River watershed (Cosulich, 1966). When one considers that two ducks can yield the suspended solids equivalent to one human, the potential for large-scale impact and pollution these farms may have had on adjoining rivers and bays can be appreciated. Since the ducks were allowed to move freely between the land and water, they deposited their wastes directly into the river and streams and also onto the riverbanks. Ultimately, wastes on the banks were likely introduced into the streams via runoff (Davids and Cosulich, 1968). Thus, the operation of duck farms may be an important factor in understanding changes to Wertheim; however, the exactly nature of such impacts and to what degree the wastes had on the marsh system are very difficult to ascertain.

Again, it is likely that the higher organic content observed in Wertheim marshes during this time does reflect an increase in organic production, not a decrease in mineral input. This interpretation is discussed above, and in addition, other anthropogenic activities (e.g., upland development, grid ditching) should only serve to increase mineral input if they have any effect at all. Thus, it is plausible that nutrient yields from the duck farming facilitated enhanced organic production in the Wertheim salt marshes. A precise interpretation is difficult because grid ditching is implemented during this same time period. Grid ditching is suggested to have favored the change in dominance from *Scirpus robustus* to *Spartina patens* in the Wertheim marsh interiors, mainly by enhancing marsh surface drainage and introduction of more saline tidal waters. Ultimately, it is reasonable to suggest that grid ditching and the duck farms each played significant roles in the observed 20th century changes in the Wertheim salt marshes.

In contrast to Wertheim, where most major construction activities were some distance from the marshes, Seatuck has a long history of more direct human disturbances, as described earlier, with both asphalt and gravel roads bisecting the marsh in several places. For the road leading to the town beach, a major canal was dredged through the marsh. Given that most of these disturbed areas are colonized almost exclusively by *Phragmites australis*, it seems apparent that these areas at least provided a suitable habitat for the invasive reed, if not actively encouraging its spread via hydrological and morphological changes. Based on the distribution of *Phragmites* at Seatuck, its presence most prominently near road/canal construction and the upland border rather than along the grid-ditching, implies that the former factors are more significant in fostering its spread. These observations of upland and within-marsh development suggest that they have been a prominent factor triggering marsh change at Seatuck in the early to middle part of the 20th century.

The orginal hypothesis of this study suggested that disturbances from parallel grid-ditching and subsequent OMWM activities triggered changes at these marshes. For Seatuck, ditching may be overshadowed by upland development and within-marsh developments, and in fact becomes just one of many anthropogenic alterations at this marsh. Additionally, it appears that any changes caused by the Seatuck OMWM project conducted in the mid 1980s are not yet evidenced in the core, although the appearance of open water habitats at the surface does suggest changes in the hydrological regime to which the system may still be adjusting. However, at Wertheim, widespread, consistent, and major shifts in plant communities are observable across the collected cores. Radioisotope dating clearly shows that observed shifts in plant communities correspond to the early 1900s, which was a period of active parallel grid ditching on the South Shore of Long Island. In addition, ditching is often said to alter the primary hydrologic functioning of the marsh (tidal inundation, salinity, etc.) which inherently will have repercussions on associated vegetation since individual plant species require a specific set of conditions in order to survive. It is also possible, but more difficult to ascertain, that nutrient loading from duck farming could have contributed to the increased organic production in the mid to late 20th century.

Based on these observations, Table 7 shows the weighting of the relative signficance of factors contributing to observed changes in the marshes over the past century. Overall, multiple factors are certain to have contributed and, thus, ultimately, marsh change is a function of complex interplay among natural and anthropogenic forcings. In general, though, it does appear that anthropogenic activities have been more significant causes of change in the 20th century, compared with that of natural processes. It is important to note that this assessment (Table 7) is based on the *magnitude* of change caused to the marsh, not necessarily changes in the health or relative value of the marsh. For example, grid ditching appears to have been one of primary causes of change in the Wertheim marshes during the 20th century, but the marsh remains relatively healthy and functioning within the range of an unaltered systems. In contrast, the primary cause of change in the Seatuck marshes, roadway construction and upland development, appears to have led to considerable degradation of the marsh, with severe *Phragmites* invasion and a highly altered hydrology. Unfortunately, it is not clear to what extent the 1985 OMWM project restored natural functioning to Seatuck, but current conditions show signs of continued change, typified by the development of episodically flooded panne habitats.

Table 7. Relative Importance of Factors Contributing to Marsh Change During the 20th Century (**1** = major influence, 2 = some influence, 3 = little influence)

Factor	Wertheim	Seatuck
Global Climate Change/SLR	3	3
Inlet Closings/Openings	2	3
Roadway/Upland Development	3	**1**
Duck Farming	2	3
Parallel Grid Ditching	**1**	2
OMWM Practices	n/a	2

6.3.2 Survival or Submergence?

The rate of global sea-level rise is increasing as a result of both environmental and anthropogenic changes (Hartig *et al*., 2002), and many questions remain as to whether salt marsh accretion will keep pace with the increasing sea-level rise, or if marshes may eventually drown in place (Reed, 1995). During the past century the average rate of sea-level rise was 0.30 cm/yr at the Battery tide gauge in New York harbor (Fig. 25). Shorter tide-gauge records from Montauk and Willets Point are similar to that of the Battery, suggesting that the rate of 0.30 cm/yr is representative for the much of the Long Island area. Although the long-term rate of rise has not changed dramatically, there has been notable interannual to sub-decadal variability, which results in rates of change that can be several fold higher than the long-term rate (Fig. 25). Kolker (2005) suggests that these short-term changes, largely driven by meteorological forcings, are primary drivers of sedimentation in the marshes bordering shallow, microtidal lagoons such as Great South Bay.

Such short-term variability in sedimentation accounts, in part, for differences observed in the average accretion rates for Wertheim and Seatuck, which range from 0.21 cm/yr to 0.37 cm/yr (Table 4). All but one site (WB-A) was determined to have mean accretion rates that are slightly less than that of long-term sea-level rise. However, all but one site (WB-B) show increased sedimentation rates during the second half of the 20th century, with values at or in excess of the

0.30 cm/yr long-term rate of sea-level rise. This acceleration of accretion rates at Seatuck and Wertheim is also consistent with results from other marshes around Long Island (Kolker, 2005). The general conclusion is that the Wertheim and Seatuck marshes are largely keeping pace with sea-level rise over the long-term. Certainly there is no suggestion that the marshes are drowning or experiencing significantly greater inundation. Rather, the considerable variability observed in short-term accretion rates (0.1 to 0.6 cm/yr) and accretionary styles (organic growth vs. mineral deposition) indicate that the marshes adapted in several ways to environmental changes of the past century. This apparent capacity for adaptation implies that the marshes are not at severe risk off loss in the future where environmental change is a given, at least that which lies within a reasonable range of stress.

Currently, predictions based on global climate models suggest that sea level may rise 30 cm by 2050 (EPA, 2005). Under such a scenario, marshes would have to accrete at twice the rate they did in the 20th century to keep pace with this accelerated rise. Based on measurements of short-term accretion rates (three to five year intervals), it is clear that these marshes can accrete at rates of 0.5 to 0.6 cm/yr. Therefore, it is very plausible that the Wertheim and Seatuck marshes could keep pace with an accelerated pace of sea-level rise. However, any further degradation of local environmental conditions (i.e., hydrology, sediment source, water quality) could easily reduce the marshes' capacity to respond favorably to stress of global climate change/sea-level rise. Thus, continued stewardship of these coastal ecosystems should may be an important component to their continued survival.

7. Management Implications

System-wide changes in plant communities were observed at both Seatuck and Wertheim National Wildlife Refuges since the 19th century and into the middle 20th century. The general overall trend, as indicated by preserved plant roots and rhizomes, has been a shift from brackish marshes to more saline and drier high marsh environments. As previously described, these changes have been a consequence of both natural and anthropogenic forcings, although changes in the last century appear to result in larger part from anthropogenic modifications. Thus, this study suggests that anthropogenic factors may be more significant in catalyzing changes to the marsh on decadal time scales. In addition, the consequences of anthropogenic modifications, as evidenced in the peat record, may not be recognized for many years, implying a time lag between environmental change and marsh response. For example, only several centimeters of marsh have accreted at Seatuck since its OMWM project in the mid 1980s, and this is not enough time to detect any significant changes, if indeed any have occurred.

One of the key outcomes of this project is evidence for both between-marsh variability, and also within-marsh variability. This finding is important to consider when designing a marsh management plan because not all marshes or even areas of the same marsh will respond to a specific alteration in the same way. For example, the data from this study shows that Area 1 of Wertheim has been accreting predominantly via organic material for the past half century; whereas Area 2 of the same marsh has been accreting primarily via mineral sediments. Certainly such factors must be taken into consideration when implementing changes to hydrology and sediment transport regimes.

Ultimately, the success of an anthropogenic modification to a marsh using OMWM lies in balancing the short-term impact to the marsh with the possible long-term success of a fully functioning marsh that uses fish to control mosquito populations, rather than pesticides. With careful planning, appropriate research, and sound management practices, today’s marshes can be restored and managed to protect their important biological and physical roles, as well as their quintessential beauty, for many centuries to come.

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