LONG ISLAND TIDAL WETLANDS TRENDS ANALYSIS

Prepared for the

NEW ENGLAND INTERSTATE WATER POLLUTION CONTROL COMMISSION

Prepared by

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Executive Summary

The purpose of the Long Island Tidal Wetlands Trends Analysis project was to quantify the magnitude of landscape-level changes in wetlands loss and changes in marsh condition within the Long Island Sound, Peconic, and South Shore Estuaries including all or parts of Westchester, Bronx, Queens, Nassau, and Suffolk Counties. (See Figure 1 and Figure 2 for the geographic extents of the study area and its estuaries, respectively.) The results of this project, the observed trends in wetland area and composition change, and implications for estuary health and supply of estuarine ecosystem services are intended for use by environmental managers, conservation advocates and elected officials across a variety of regulatory agencies, environmental organizations, and governments.

Changes, including degradation, fragmentation and severe acreage losses have been observed in several Long Island, NY tidal wetland complexes during discrete and limited trends analyses. The results of this effort support other studies that have demonstrated substantial loss of tidal wetlands area over the past forty years. Typical indicators of native marsh loss (i.e., not including *Phragmites australis* marsh) that were observed in the study area include retreat of the seaward edge of the marsh, loss of marsh islands, widening of tidal creeks and ditches, panne/mudflat, pond formation, and encroachment of invasive *Phragmites australis*. In addition to native marsh loss, conversion of high marsh to low marsh is indicative of sea level rise.

The trends analysis was conducted across the three major tidal wetland classes (i.e., Intertidal, High and Fresh Marsh) and *Phragmites australis* over two time periods: 1) Year 1974 and 2) Year 2005/2008 (current year). The current year tidal wetlands were mapped using a computer-assisted image processing technique that isolates and groups pixels within an image according to the unique reflectance values, or „signatures‟, of different plant species and other marsh features. The current year tidal wetland inventory comprises the vegetated wetland areas only; unvegetated areas, such as pannes, mudflats and water bodies, were removed from the wetland map and thus are not included in the area loss/gain calculations. In a like manner, the 1974 tidal wetland inventory – which included unvegetated features within the major tidal wetland classes – was enhanced by removing unvegetated features (e.g., pannes, mudflats, and water bodies). Such enhancement allowed a one-to-one comparison between the 1974 and current year tidal wetlands. Summaries were calculated at three geographic levels, comprising the estuary, town boundary and marsh complex.

Overall, Long Island‟s estuaries have lost 13.1 % of native intertidal (IM), high marsh (HM), and coastal fresh marsh (FM) communities between 1974 and 2005/2008 (Table 3). Appendix II provides the results of the imagery analysis and trends in marsh area for each of the identified tidal wetland complexes. The Peconic Estuary and South Shore Estuaries have slightly lower percentages of marsh loss (-10.4% and -11.6%, respectively) compared to the Long Island Sound Estuary (-22.6%). Collectively, Long Island‟s three estuary complexes lost, on average, 85 acres of native marsh annually over this time. The largest loss of native marsh, i.e., 1,692 acres, occurred in the South Shore Estuary where the native marsh area declined from approximately 14,652 acres in Year 1974 to 12,959 acres in Year 2008. The native marsh in the Long Island Sound decreased by an estimated 654 acres from 2,892 acres in Year 1974 to 2,237 acres in Year 2008. Approximately 356 acres of native marsh were lost in the Peconic Estuary, declining from an estimated 3,444 to 3,078 acres from Year 1974 through 2008.

A majority of the loss of native marsh within the South Shore Estuary, i.e., 1,060 acres, occurred within the Town of Hempstead. The remaining native marsh losses within the South Shore Estuary (i.e., 599 acres) comprised approximately 120, 148, 143, and 188 acres within the Towns of Oyster Bay, Babylon, Islip, and Brookhaven, respectively (Table 16). Within the Town of Southampton, the South Shore Estuary native marsh area increased by 63 acres. Within the Long Island Sound Estuary, The largest losses in marsh area from 1974 to 2005 occurred in the Towns of Huntington (163.6 acres lost), Smithtown (132.8 acres lost), Brookhaven (67.0 acres lost), and North Hempstead (44.5 acres lost); in addition, the Bronx lost an estimated 77.8 acres of native marsh Table 31).

The Peconic Estuary spans the Towns of East Hampton, Riverhead, Shelter Island, Southampton and Southold. East Hampton sustained the largest loss of marsh habitat, losing 145.8 acres for a 13.8 percent decrease from 1974 to 2005. The Town of Southold lost nearly 10 percent of marsh habitat from 1974 through 2005, while the Town of Riverhead exhibited a slight gain in native tidal wetland area. The highest percentage loss of marsh habitat occurred in the Town of Shelter Island where marsh habitat decreased in area by 17.5 percent.

Each marsh complex was identified as “stable”, i.e. less than 10% decrease or increase in marsh area between 1974 and 2005/2008, or “at-risk”, i.e. more than 10% loss in marsh area. In the South Shore Estuary, the trends analysis identified 117 „At-Risk‟ marshes complexes – out of a total of 215 (Table 26). The number of „At-Risk‟ marsh complexes and proportion of marsh complexes identified as „At-Risk‟ declined eastward within the South Shore Estuary. The trends analysis identified 100 „At-Risk‟ marsh complexes – out of a total of 152 – in the Long Island Sound Estuary (Table 36). 86 „At-Risk marshes – out of a total of 159 – were identified within the Peconic Estuary (Table 43). „At-Risk‟ marshes are located throughout the estuary; however, clustering is apparent in the western portions of the estuary, particularly adjacent to more developed areas around Riverhead, Sag Harbor and along the north shore of Peconic Bay.

The major changes in the biological and physical structure of Long Island‟s marshes observed in this study include:

- Conversion of High Marsh to Intertidal Marsh
- Formation of Pannes and Ponds Within Marshes
- Conversion of Intertidal Marsh Islands to Mudflats
- Widening of Tidal Creeks and Man-made Ditches

- Erosion and Retreat of Seaward Edge, and
- *Phragmites australis* Encroachment

Both the conversion of high marsh to intertidal marsh and the formation of expansive panne and pond areas within marshes are indicators of marsh drowning or waterlogging. Marsh drowning may be due to the interacting effects of the failure of marsh accretional processes (such as deposition of organic sediments and accumulation of plant biomass) to keep pace with relative sea level rise and marsh subsidence related to plant mortality and subsequent decomposition of root biomass (Fagherazzi et al, 2012; Kirwan and Megonigal, 2013). This failure is due to physiological stresses such as increased flooding, sulfide accumulation, and nutrient enrichment (Turner et al, 2009; Wigand et al, 2014).

Many wetland complexes with large losses of high marsh habitats through marsh drowning and subsidence exhibit large gains in intertidal marsh as shown in Table 5, which presents the marsh area change for each of the complexes identified in Table 3, and Table 6, which presents the marsh complexes with the largest observed gains in intertidal marsh area. In contrast, other marsh complexes with large losses of high marsh habitats exhibited large reductions in total marsh area and due to the conversion of high marsh to pannes, marsh ponds, and open water, as shown at Timber Point (ID # 431, Figure 9). Wetland complexes showing significant losses in intertidal marsh, either in acreage or percent loss, are predominantly located in the western South Shore estuary (acreage) or Long Island Sound (percent loss) (Table 9 and Table 10, respectively). Considered in conjunction with the observed losses of high marsh, the observed trends in intertidal marshes indicate that substantial subsidence/drowning of tidal marshes is occurring throughout Long Island‟s three major estuary systems.

Comparison of the 1974 and 2005/2008 aerial imagery indicates widening of natural tidal creeks and man-made ditches in many Long Island marsh complexes. Creek widening in other degrading marshes in the northeastern United States has been attributed to reduced structural integrity and collapse of creek banks resulting from reduced root biomass, increased decomposition of organic matter, and increased soil water content (Deegan et al, 2012) and, in some cases, herbivory from Sesarma crabs (Smith, 2009). Creek width and cross-sectional area is related to the geomorphological and hydrological characteristics of the marsh and surrounding estuary and, as a result, may be impacted by changes in tidal prism (Vandenbruwaene et al, 2013) resulting from relative sea level rise (Stefanon et al, 2012) or anthropogenic changes in inlet or estuary bathymetry. Creek banks, particularly in large tidal channels, may also be subject to erosional forces from wind-driven waves during storms, greater water flow rates, and vessel wakes.

Many marshes in Long Island‟s estuaries exhibit pronounced retreat of the seaward marsh edge. The recession of marsh edges is influenced by similar processes to bank collapse and widening. However, due to their less sheltered position, marsh edges are subjected to greater erosional forces from vessel wakes and during storms. In addition, other anthropogenic disturbance of the seaward edge of the marsh such as the harvesting of ribbed mussels from the marsh or harvesting of shellfish from the subtidal mudflats adjacent to Spartina alterniflora may also affect the stability of the seaward edge of the marsh and Spartina recruitment.

Invasive *Phragmites australis* has colonized large areas of high marsh and coastal fresh marsh habitats on Long Island. However, many of Long Island‟s marshes were infested by *Phragmites australis* prior to 1974. Long Island‟s estuaries exhibit different trends regarding *Phragmites australis* abundance (Table 12). In the Long Island Sound, Peconic Estuary, and the coastal bays in Southampton and East Hampton, *Phragmites australis* encroachment has contributed to the drastic loss of native high marsh communities and led to the near eradication of areas classified as coastal fresh marsh. *Phragmites australis* has continued to colonize high and coastal fresh marshes within the South Shore Estuary. However, overall *Phragmites australis* coverage within the South Shore Estuary has decreased by 11.9% due to the loss of *Phragmites australis* on former dredge spoil sites west of Fire Island Inlet and in Moriches and Shinnecock Bays.

Salt marshes provide many critical benefits to human communities including fish and shellfish production, protection of shorelines from coastal storms, erosion control and sediment stabilization, water filtration through nutrient and sediment removal, carbon sequestration, and recreation and tourism (Barbier et al, 2011). The loss of nearly 3,000 acres of native wetlands implies a substantial loss of ecosystem services in Long Island‟s estuaries. The approximately 30% loss of high marsh habitats, in particular, throughout Long Island between 1974 and the mid 2000‟s and resulting loss of ecosystem services and habitat for wildlife and rare plants demands restoration efforts in complexes with greatest losses of high marsh area (Table 5) and increased management in the largest remaining high marshes (Table 6).

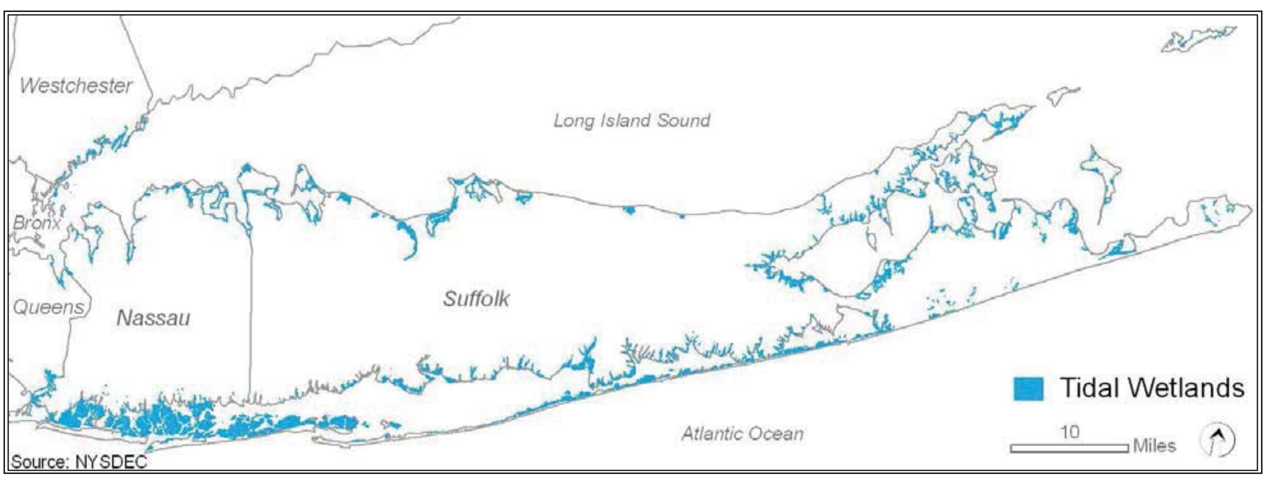
Introduction

Purpose

The purpose of the Long Island Tidal Wetlands Trends Analysis project was to quantify the magnitude of landscape-level changes in wetlands loss and changes in marsh condition within the Long Island Sound, Peconic, and South Shore Estuaries including all or parts of Westchester, Bronx, Queens, Nassau, and Suffolk Counties. The project study area is depicted in Figure 1. The results of this project, the observed trends in wetland area and composition change, and implications for estuary health and supply of estuarine ecosystem services are intended for use by environmental managers, conservation advocates and elected officials across a variety of regulatory agencies, environmental organizations, and governments. Identification of small spatial scale variation in marsh composition (i.e. less than approximately 100 square feet in area), while of ecological importance due to habitat complexity, was given lower priority under this project than providing landscape-level trends on Long Island"s tidal wetlands that can be used by regulatory agencies, environmental organizations, and governments. This work is not related to any specific environmental regulation or any specific regulatory decisions.

The project utilized Geographic Information System (GIS) and image analysis tools coupled with field reconnaissance to quantitatively and qualitatively assess Long Island"s current tidal wetlands. The delineation of current tidal wetlands is based upon the image classification of the most recently available high-resolution, color-infrared aerial photography. The current year delineation is compared with a similar assessment of tidal wetlands that was generated in Year 1974 via photo-interpretation of images taken in the same year. The comparison of the current and Year 1974 tidal wetlands areas, i.e., a quantitative trends analysis, was conducted across 527 wetland complexes, three major estuaries (i.e., Long Island Sound, Peconic and South Shore) and selected local government boundaries.

Figure 1: Project Study Area



Environmental and Ecological Context

New York‟s diverse coastal wetlands and their ecosystems are biologically, ecologically, economically, and recreationally valuable. These wetlands protect coastal water quality by acting as a sink for land derived nutrients and contaminants, constitute an important component of coastal food webs, provide valuable wildlife habitat, and protect upland and shoreline areas from flooding and erosion.

Changes, including degradation, fragmentation and severe acreage losses have been observed in several Long Island, NY tidal wetland complexes during discrete and limited trends analyses. In order to begin to develop and implement advanced protection and restoration initiatives and policies, these changes must be studied in depth. The primary goal of the Long Island Tidal Wetlands Trends Analysis was to assess the quantitative and qualitative changes, including the extent of marsh acreage lost or gained, and changes or shifts in tidal wetland vegetation since the last New York State regulatory inventory of 1974.

The study area for this project includes coastal areas of New York State within the Long Island Sound, Peconic, and South Shore estuaries including all part or parts of Westchester, Bronx, Queens, Nassau, and Suffolk Counties (see Figure 1). Trends were assessed on the individual marsh complex scale, and within and between each estuarine system.

The results of this effort support other studies that have demonstrated substantial loss of tidal wetlands area over the past forty years. Typical indicators of native marsh (i.e. not including *Phragmites australis* marsh) loss observed include retreat of the seaward edge of the marsh, loss of marsh islands, widening of tidal creeks and ditches, panne/mudflat and pond formation, and encroachment of invasive *Phragmites australis*. In addition to native marsh loss, conversion of high marsh to low marsh is indicative of sea level rise.

Funding Source and Partners

This project was funded by the New England Interstate Water Pollution Control Commission (NEIWPCC), the U.S. Environmental Protection Agency (US EPA), the New York State Department of Environmental Conservation (NYS DEC), and The Nature Conservancy (TNC). The project was administered by the NEIWPCC and representatives from the US EPA, the NYS DEC and TNC provided technical guidance under the auspices of a Technical Advisory Committee (TAC). The TAC met regularly to review project progress and findings. Representatives from the Suffolk County Departments of Health Services and Environment and Energy and the Town of East Hampton Planning Department also served on the TAC.

Trends Analysis

This trends analysis compared tidal wetland areas in the study area over two time periods: 1) Year 1974 and 2) Year 2005/2008. Appendix I provides index maps showing the location and Complex ID # for each identified tidal wetland complex. The tidal wetland boundaries for the first time period were established in 1974 via manual photo-interpretation of color-infrared imagery. Photo prints of the original pen-on-photo delineations serve as the New York State Official Tidal Wetlands Inventory. The official 1974 tidal wetland boundaries were eventually digitized – from scans of the official maps – for use with a GIS.

The tidal wetland delineation for the second time period in the trends analysis was generated from two sets of readily available color-infrared imagery for Years 2005 and 2008. The Year 2005 imagery encompassed the marsh habitat, shorelines and adjacent upland areas for the Long Island Sound and Peconic Estuaries. The Year 2008 imagery covers the entirety of the South Shore Estuary and vicinity. The tidal wetlands of the three estuaries are shown in Figure 2.

The Year 2005/2008 tidal wetland mapping – referred to as the "current year" delineation for the purposes of this study – was generated via image classification. Image classification is a computer-assisted technique which classifies pixels of a raster image based on the spectral reflectances of plant species and other natural features. This method is based on the principle that every plant species exhibits a unique spectral response to solar radiation across the electromagnetic spectrum. (Note: In practice, specific bands are established for image classification purposes wherein reflectance values are averaged; this can cause the reflectance values of species to overlap.)

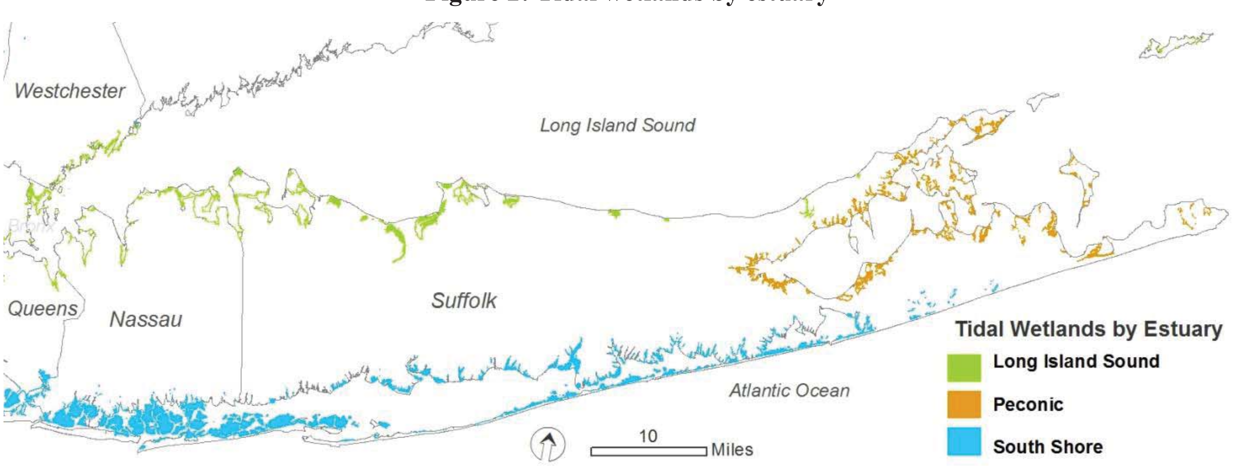
The unique spectral response of each wetland species was employed to classify each color-infrared image into species or species group for low marsh, high marsh, and *Phragmites australis*. Upland plant species and other marsh features (e.g., mudflats, salt pannes and water) were also classified based on their unique spectral signatures. New York‟s tidal wetlands contain a variety of unvegetated or sparsely vegetated substrates such as sand, gravel, or cobble beaches, salt pannes (which often contain various densities of plant such as *Salicornia sp*. and *Limonium caroliniana*), and mudflats located seaward of or in between vegetated marsh areas. Due to the absence or low density vegetation on these substrates, the image classification used in this study cannot differentiate between traditional salt pannes and marshes that have lost vegetation due to die-off or subsidence. In addition, while unvegetated sufaces, such as mudflats and pannes, can be differentiated from standing water, reliable conclusions cannot be reached by quantifying and analyzing the areas of unvegetated surfaces and standing water in the aerial imagery as the observed boundaries between unvegetated and water surfaces are arbitrary and dependent on recent rainfall, lunar phase (spring versus neap tide), and when in the tidal cycle the aerial imagery was collected. These factors may influence if a panne has some standing water in it or not or if a creek has or does not have exposed mud within its channel.

Classified species/species groups were ultimately reclassified into tidal wetland classes (i.e., intertidal marsh, high marsh, and coastal fresh marsh) based on definitions contained in the New York State Tidal Wetlands Land Use Regulations (Part 661). More detailed classifications of the ecological community types recognizing the diversity of habitat and species assemblages in New York State‟s tidal wetlands have been developed since the promulgation of Part 661, such as Reschke (1990), Edinger et al. (2002), and Sneddon and Lamont (2010). However, the tidal wetland mapping classes contained in Part 661 were utilized for this study in order to facilitate comparison with the 1974 New York State Tidal Wetlands Inventory Maps. A detailed methodology for the image classification technique is provided in the "Methodology and Data" section below.

The trends analysis was conducted across the three major tidal wetland classes (i.e., Intertidal, High and Fresh Marsh) and *Phragmites australis* and at three geographic levels, comprising – in order of decreasing geographic extent – the estuary, town boundary and marsh complex. The trend for a particular tidal wetland class is calculated by subtracting the Year 1974 area from the current year area; the change in area according to this formula can be negative or positive. If negative, the change indicates a loss of wetland for that class during the period from Year 1974 to the current year. Likewise, a positive change value indicates a gain in wetland during the same time period. The change is also calculated at the marsh complex scale to determine the total change in marsh area (i.e., comprising the three tidal wetland classes) from Year 1974 to the current year.

The estuary level is utilized as practical framework for presentation and discussion of the trends. Some results are also presented by Town to provide information to municipal land managers and regulators and to convey general information on geographic position and environmental conditions. Also, the organization of results by Town also facilitates interpretation of data tables by avoiding the presentation of large sets of results, such as the At-Risk Marshes tables, solely by alphabetical order or Complex ID #.

Figure 2: Tidal wetlands by estuary



Methodology and Data

Outline of Technical Approach

The trends analysis consisted of a comparison of four broad tidal wetland classes (i.e., Intertidal Marsh, High Marsh, Coastal Fresh Marsh and *Phragmites australis*) over two time periods. The first time period was Year 1974, when the first tidal wetland mapping was conducted. The second time period was termed the „current year‟ for the purposes of this study despite the acknowledgement that the color-infrared photos were taken in Years 2005 and 2008. The current year tidal wetlands were mapped using a computer-assisted image processing technique that isolates and groups pixels within an image according to the unique reflectance values, or „signatures‟, of different plant species and other marsh features.

Given a standard spectral library and the repeatability of a classification algorithm, the computer-assisted approach is, arguably, more consistent than the manual interpretation method employed for the 1974 tidal wetland delineation. The computer-assisted approach also easily identified and extracted numerous salt pannes, water bodies and other unvegetated areas (e.g., mudflats) within the image tiles, features that could not easily be delineated manually. Furthermore, the 1974 imagery were not orthorectified at the time of the manual delineation or prior to this project. As a result, a previous digitizing of the 1974 tidal wetland boundaries sustains sizable errors in position and shape, the latter of which affects the true area of a wetland polygon. Thus, the tidal wetland delineations from the two time periods have important differences. The methodology utilized in this project corrects for these differences, thus achieving a greater degree of equivalence, i.e., reduces difference in delineation accuracies, between the Year 1974 and Current Year delineations. Such efforts were important to the accuracy of the trends analysis.

An error analysis, that examines the relative difference between the computer-assisted and manual delineations approaches, was also conducted. This comprised the application of the computer-assisted tidal wetland image classification to randomly selected 1974 image tiles. Specifically, this analysis compares the mapped areas of the intertidal and high marshes for the manual 1974 photointerpretation and the computer-assisted wetland classification. With this „error‟ analysis, the effect of the relative difference between the two approaches on the trends analysis was estimated.

Technical Objectives

The compilation of an accurate tidal wetlands trends analysis was accomplished through the stepwise completion of several technical objectives, or tasks, which comprised the following:

1. Identification of discrete wetland complexes

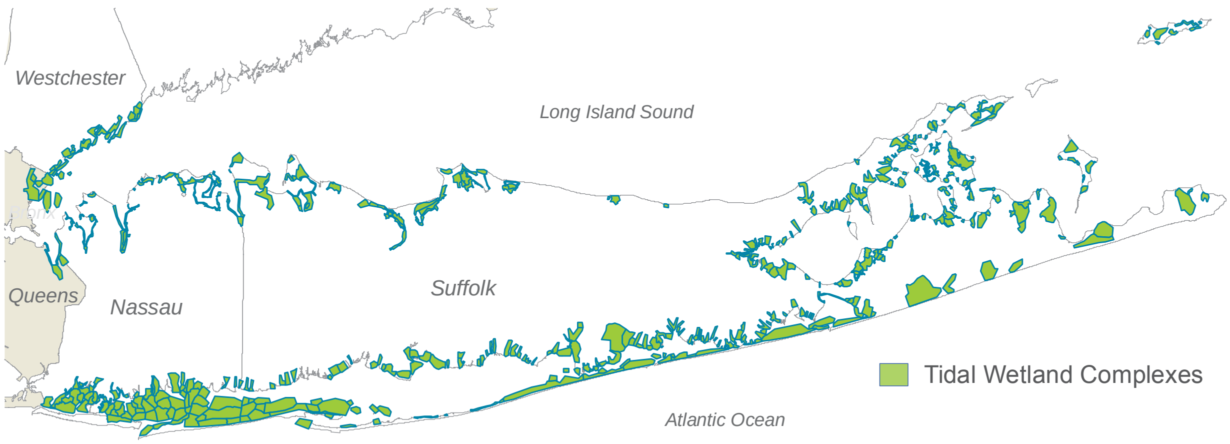
2. Scanning, orthorectification (removal of image tilt and terrain to create planimetrically correct image) and tonal balancing/mosaicking of color-infrared imagery
3. Field reconnaissance to identify and groundtruth wetland and other plant species
4. Classification of color-infrared imagery via application of a spectral library
5. Vectorization (conversion of raster classification files to vector shapefiles) and delineation of current wetland boundaries
6. Enhancement of Year 1974 tidal wetland boundaries
7. Trends Analysis
8. Error Analysis (classification error and calculation of relative difference)

It is important to note that the objective of this project was to produce tidal wetlands mapping for the current year and an enhanced version for Year 1974 that was sufficient for trends analysis. Thus, the accuracy of the current year tidal wetlands mapping is not equivalent to that achieved by field delineations. Typical mapping accuracies for automated image classifications ranges from 75 to 85 percent depending upon the classification technique employed (Thakur et al., 2012). However, the current year tidal wetlands mapping accuracy exceeds typical classification accuracies owing to the collection of numerous groundtruthed data points (i.e., for tidal wetland and other plant species) and post-classification corrections through heads-up digitizing.

Identification of discrete wetland complexes

The project team identified tidal marsh complexes using a classification system based on the Significant Coastal Fish and Wildlife Habitats (SCFWHs) identified by the New York State Coastal Atlas (NYS Department of State, 2002). Appendix I provides index maps showing the location and Complex ID # for each identified tidal wetland complex. Appendix II provides the results of the imagery and trends analysis of each of the identified tidal wetland complexes. Marsh complexes were identified and mapped in a geographically contiguous manner and delineated such that environmental impacts and indicators of marsh loss would be expected to be relatively uniform within the complex. Accordingly, many SCFWHs were subdivided into independent marsh complexes due to the large size of the SCFWHs and the disparate environmental conditions or impacts within SCFWHs. In addition, it was necessary to differentiate between marsh islands and nearby upland fringe marshes within identified marsh complexes.

Figure 3: Marsh Complexes of the Study Area



The identification of discrete wetland complexes recognized that not all tidal wetlands within the project area are located within SCFWHs as they are under federal jurisdiction (e.g., Oyster Bay National Wildlife Refuge, Fire Island National Seashore), Native American tribal ownership (e.g., Shinnecock Indian Reservation) or experienced previous adverse environmental impacts (e.g., Quantuck Bay, Meetinghouse Creek). All marsh complexes within these SCFWH-excluded areas were selected for inclusion in the trends analysis. Based on these parameters, 527 marsh complexes were identified; the boundaries of the complexes are depicted in Figure 3.

The project team also recognized that some SCFWHs contain freshwater wetland habitats (e.g., Nissequogue River, Long Pond Greenbelt) and upland habitats (e.g. Grandifolia Sandhills, Southampton Beach and Dunes). Upland or freshwater wetland portions of SCFWHs were not included within the tidal wetland complexes. Each wetland complex was named and assigned the following identifiers within the GIS project database: County, Town, SCFWH, and NYS Tidal Wetland Inventory Map Number(s).

Scanning, orthorectification, and normalization of color-infrared imagery

Imagery utilized for this trends analysis project consists of Year 1974 color-infrared photographs (employed for the first tidal wetland delineation), Year 2005 color-infrared imagery for the Long Island Sound and Peconic Estuaries, and Year 2008 color-infrared imagery for the South Shore Estuary. The Year 2008 color-infrared imagery was digitally scanned, orthorectified and mosaicked to a 2,000 meter by 2,000 meter image tile grid under a previous effort by TNC. Year 1974 and 2005 imagery, however, were not previously scanned and orthorectified and were processed in a manner comparable to the Year 2008 imagery.

The accuracy of the tidal wetland mapping conducted for this project was dependent upon the proper processing of the color-infrared imagery and its accurate orthorectification. To this end, adequate image processing parameters were established. The color-infrared imagery for Years

1974 and 2005 were scanned at a resolution of 1,000 dpi. Scanning at this resolution provided digital images with 1-foot resolution, comparable to (or exceeding) the Year 2008 aerial imagery previously compiled for the South Shore. The 1,000-dpi scan also reduced the degree of pixel mixing (averaging of pixel values that occurs for larger pixel sizes) for spectral analysis and facilitated easier and quicker differentiation between vegetation types when conducting quality control reviews.

It is noted that not all of the 1974 color-infrared images for Long Island needed to be scanned. Instead, only the 1974 images that were essential for mapping vegetated wetlands were scanned and orthorectified. 493 image tiles – from a total inventory of approximately 2,000 images – were ultimately scanned for Year 1974; the remaining image tiles contained no vegetated tidal wetlands. In addition, four of the tiles that were required for Year 1974 were missing from the image inventory.

The 1974 and 2005 images were scanned using the Wehrli RM3 or RM4 scanner. The RM-3 and RM-4 scanners are capable of scanning cut sheet or roll film media with a geometric accuracy of +/- 3 micron RMSE (root mean square error) without resampling the image data. The Wehrli scanners featured a computer controlled LED illumination system that is radiometrically calibrated across the sensor elements. The instrument utilizes a 12-Bit tri-linear sensor and an 8 micron (3,175 dpi maximum) optical system.

The orthorectified Year 2005 color-infrared imagery was also normalized and mosaicked into a 2,000-meter by 2,000-meter image grid. This grid is comparable to the original 1974 grid in terms of tile dimensions, however, the original grid utilizes a previous coordinate system based on the North American (Horizontal) Datum of 1927 (NAD27). To be current with a contemporary datum, the new grid utilizes the current North American Datum of 1983 (NAD83). (The Year 2008 imagery for the South Shore was also established in NAD83.) Thus, the original NAD27 tidal wetland grid and the new NAD83 grid used for recent imagery do not coincide.

In addition, image normalization, which includes dynamic range adjustment (adjustment of the ratio between maximum, i.e. white, and minimum, i.e. black, light intensities), tonal balancing (brightness and contrast), and color-balancing, was conducted in accordance with best practices prepared for the National Agriculture Imagery Program (NAIP), a nationwide photogrammetry program of high-resolution imagery used to map farmlands and to distinguish between crop types (USDA, 2007).

The orthorectification process employed proprietary ground control points and a high-resolution digital-elevation model. The orthorectification process attained a positional accuracy that did not exceed 3 feet of root mean square error.

Field reconnaissance to identify and groundtruth wetland and other plant species

The tidal wetland classification and delineation was supported and complemented by rigorous field reconnaissance, groundtruthing and error analysis. Field data collection was performed using a Trimble GeoXH 6000 Series receiver and post-processed to attain an accuracy ± 0.5m. Data points were collected for species having a patch size greater than 5 meters to minimize the potential for change in species from the 2005/2008 images to 2011/2012 data collection. In addition to GPS location and species identification recorded in the GIS database, the following information was recorded on a data collection form: photograph with cardinal direction, percent relative cover, patch size, plant height and growth form, and physical indicators of marsh loss.

A total of 912 data features were collected for this project throughout the study area. The field effort resulted in the collection of 805 data points for use in image classification and error analysis. (The use of groundtruthed species locations – which establishes "training points" and "test points" for image classification and error analysis, respectively – is discussed below.) Additionally, 74 area features and 29 line features – which were located at the boundary of marsh types – and 4 generic/upland data points were collected for reference purposes.

Table 1: Summary of field data points collected by Town

Town	% Points Required	Max. # Points	% Points Collected	# Points Collected
East Hampton	Up to 15%	146	4.18%	34
Southampton	Up to 21%	205	8.48%	69
Shelter Island	Up to 11%	107	3.81%	31
Southold	Up to 15%	146	4.05%	33
Riverhead	Up to 11%	107	1.11%	9
Brookhaven	4-24%	234	17.81%	145
Islip	Up to 16%	156	7.74%	63
Babylon	Up to 20%	195	6.02%	49
Smithtown	Up to 13%	127	2.70%	22
Huntington	Up to 13%	127	4.18%	34
Oyster Bay	Up to 16%	156	11.18%	91
North Hempstead	Up to 11%	107	0.00%	0
Hempstead	22-42%	410	23.71%	193
Queens	Up to 12%	117	0.25%	2
Bronx	Up to 10%	98	1.35%	11
Larchmont			1.11%	9
Mamaroneck	Up to 10%	98	0.00%	0
Rye	Up to 11%	107	1.23%	10
New Rochelle	Up to 10%	98	0.00%	0
Total # Data Points Collected				**805**

Field data collection was performed throughout the project area based on the relative Year 1974 marsh area by Town. Table 1 provides a summary of data points collected by Town, and includes the required number of points.

The field data collection effort targeted species predominately observed in each of the three vegetated marsh types as well as species commonly observed at or near the marsh boundary, as described below and summarized in Table 2:

- Intertidal Marsh – *Spartina alterniflora*
- High Marsh – *Spartina patens*, *Distichlis spicata*, *Juncus gerardii*, *Iva frutescens*
- Coastal Fresh Marsh – *Typha angustifolia*, *Schoenoplectus* spp.
- Upper High Marsh/Upland Border Species – *Phragmites australis, Baccharis halimifolia, Panicum virgatum, Morella pensylvanica, Toxicodendron radicans, Ammophila breviligulata*
- Mixed Species – Data were collected for mixed intertidal and high marsh species stands.

Table 2: Total number of data points collected by species / habitat type

Species	# Points
Spartina alterniflora	153
Spartina patens	72
Distichlis spicata	68
Juncus gerardii	20
Iva frutescens	70
Baccharis halimifolia	72
Phragmites australis	69
Typha angustifolia	11
Scheonoplectus spp.	13
Panicum virgatum	36
Morella pensylvanica	39
Toxicodendron radicans	26
Ammophila breviligulata	29
Mixed species	104
Salt panne	18
Other	5
Total # of Data Points	**805**

Classification of color-infrared imagery via application of a spectral library

The photointerpretation process for mapping the wetland boundaries comprised two major phases, i.e., supervised classification of vegetation types and other features (e.g., water bodies and salt pannes) and the grouping of vegetation types into tidal wetland classes. During the first phase of the photointerpretation process, i.e., supervised classification,, spectral analysis was used to identify and differentiate among the various wetland categories.

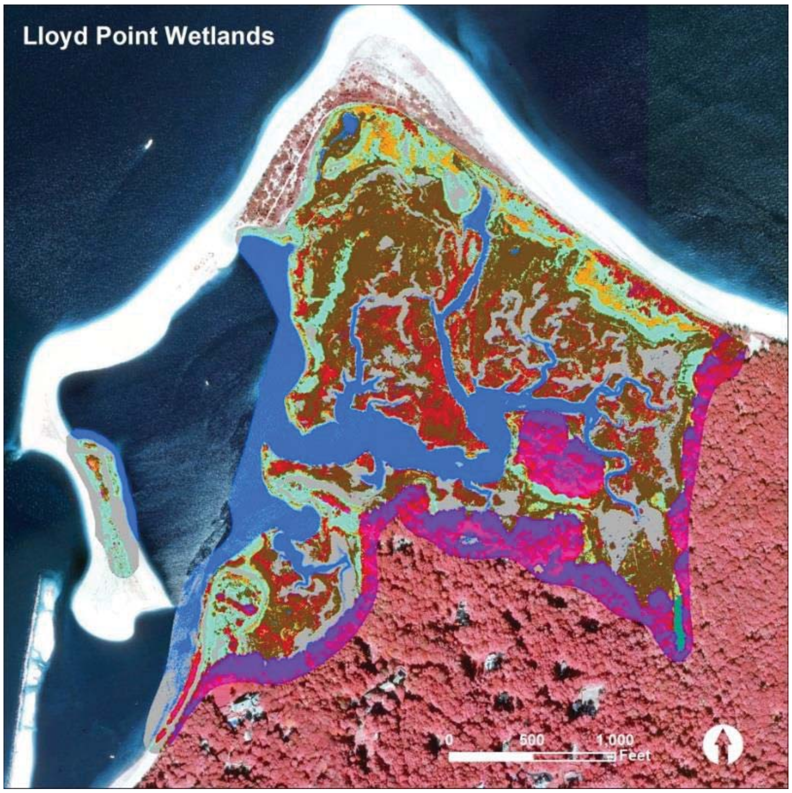
Spectral analysis is an image processing technique that is used to identify vegetation types as well as broader land cover classes by their spectral signature, i.e., by unique combinations of reflectance values within the spectral bands that comprise a false-color infrared image. Spectral analysis of the false-color infrared images that comprise the study area was performed with ENVI image analysis software. ENVI is among the most popular image analysis software packages used by remote sensing experts, scientists and researchers. ENVI provided all of the image analysis functionality required for this project including spectral analysis, image correction, „noise removal‟, and feature extraction. In addition, ENVI allowed the integration of GIS data with the raster-based color-infrared imagery, a functionality that is critical to the supervised classification process discussed below.

It is important to note that supervised classification of the color-infrared image tiles – which provides an automated method for identifying tidal wetland species – resulted in improved identification of wetlands as compared with the 1974 wetlands inventory. Thus, an overestimate of the increase in wetland area from 1974 through the current year has resulted for some wetland complexes, particularly those in Moriches and Shinnecock Bays.

Figure 4 provides an example of an image classification for the tidal wetland complex known as the Lloyd Point Wetlands (Huntington). The graph below the Lloyd Point Wetlands map in Figure 4 plots the reflectance value (i.e., a measure of the light reflected back from the particular plant species) for red light versus that for near-infrared light. Each cluster of identically colored points represents the range, or envelope, of reflectance values, according to a given species‟ response to red and near-infrared light. Note that each plant species or marsh feature occupies its own unique region on the graph (though some overlap occurs with adjacent species owing to varying plant health conditions from low to high vigor). The characteristic set of responses, or signature, of each species to two or more given wavelengths allows the image classification algorithm to identify pixels in an image as belonging to a group of reflectance values or class, e.g., *Iva frutescens, Phragmites australis, Spartina alterniflora*, etc.). In particular, this project utilized the Maximum Likelihood classification algorithm, one of a number of automated, statistical approaches for determining the appropriate class for a pixel that near the boundary zones between classes.

The Maximum Likelihood classification methodology was employed as it attained the best matches with the field-collected test points, i.e., groundtruthed data points which are used for error analysis of the image classification. In addition, although numerous other classification algorithms (e.g., Parallelepiped Classifier, Minimum Distance, Spectral Angle Mapper and Mahalanobis Distance), have been developed and perform reasonably well, the Maximum Likelihood classification algorithm is one of the most widely used techniques (Li et al., 2014). Furthermore, it is the most powerful technique where accurate groundtruthed data points – such those acquired through this project"s field reconnaissance effort – are provided (Perumal and Bhaskaran, 2010).

Figure 4: Lloyd Point Wetlands



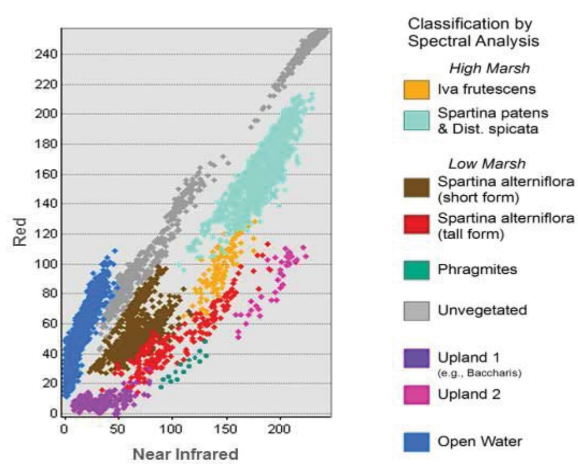
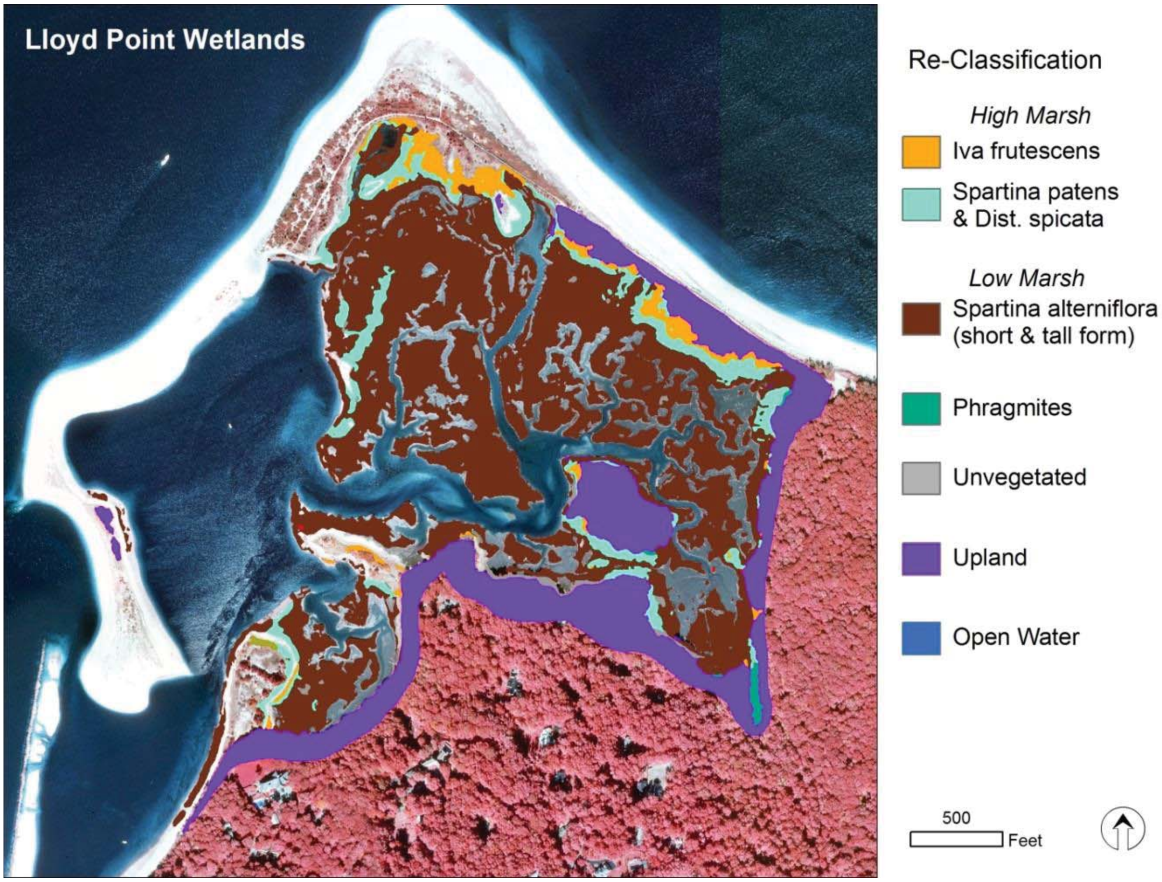


Figure 5: Lloyd Point Wetlands (Detail View)



Following the image classification as shown in Figure 4, the classified raster image was converted to a vector GIS shapefile which establishes the tidal wetlands map spatial data format. Once checked for accuracy, species and species groups were then regrouped into fewer classes as shown in Figure 5. For example, the short and tall form *Spartina alterniflora* were regrouped into one *Spartina alterniflora* class, and the two upland classes were regrouped into one upland class.

As a preparatory step to supervised classification, all digital images were tested for spectral equivalence. Images were determined to be spectrally equivalent when features in a reference, or master, image exhibited the same spectral reflectance range as those in another image, termed a servant image. (Note: the master image covered a large marsh that represented all of the species types in the spectral library and in significant quantities.) For the master and the servant image to be spectrally equivalent, the spectral ranges must match across a range of dark to bright objects, e.g., from dark water bodies, to medium brightness vegetation types to very bright (i.e., highly reflective) sandy shorelines. Thus, in the cases where the spectral reflectance ranges of features in the servant image differed substantially such that classification accuracy was reduced, i.e., more than 3 percent, from those in the master image, image normalization was conducted on the servant imager or, alternatively, the spectral library was adjusted by the difference between the master and servant images. The latter technique was utilized if, in certain instances, the differences between the master and servant image (i.e., along the gradient of light to dark spectral values) were clearly non-linear. In such instances, the spectral library was more readily adjusted than the servant image.

There were also numerous instances where the spectral values for a given species varied substantially within the image. In such instances, the image was segmented into two or more separate images for classification purposes. Each segment was then individually normalized to the master image in order to properly apply the spectral library. Such normalization and image segmentation operations posed significant analysis and computing requirements to the project but were necessary to achieve accuracy goals established in the Quality Assurance Project Plan.

This project utilized an image normalization method developed by Schott et al. (1988). Under this approach, the master and servant images were normalized by analyzing features that were statistically invariant such as concrete, asphalt pavement, rooftops, dry sand and deep water. Such features, termed "pseudo-invariant", were not expected to exhibit different spectral ranges from one image to another; digital values of the servant image were then statistically corrected to match the master image. Once the images were tested and/or corrected for spectral equivalence and the spectral signatures of the entire library features were adequately defined (i.e., through field-collected training sites), the supervised classification procedure was conducted.

Classified images were post-processed using ENVI „noise reduction" utilities including the „Sieve", „Clump" and „Majority/Minority Analysis" tools. These were used to reduce the number of stray raster cells, i.e., pixels of different value within a cluster of like values. A closer examination of the classified image in Figure 4 reveals stray cells, e.g., pixels representing small patches of *Spartina patens* or *Iva frutesens* within a larger area of *Spartina alterniflora*. Utilizing the ENVI noise reduction tools, stray cells were assigned new values that were more consistent with their surrounding values. This step also simplified the raster image so that a manageable vector version of the image was generated (i.e., one with an optimal number of vector polygons for establishing tidal wetland classes.)

The development of the spectral library of features was essential to the classification process. The locations of individual plant species – collected through "groundtruthing" and termed "training points" (or "training sites") – were used to "train" the images into classes vis-à-vis the library of spectral signatures through supervised classification.

The corrected features extracted via supervised classification were then grouped into final tidal wetland classifications according to the predominant vegetation types found within the four vegetation categories and upland borders as follows:

- **Intertidal Marsh**: Training sites were located and recorded for *Spartina alterniflora* across its variation in growth forms (i.e. tall form versus short form) and culm density.

- **Native High Marsh**: Training sites were located and recorded for stands of *Spartina patens*, *Distichlis spicata*, *Juncus gerardii*, *Iva frutescens*, and mixed-species stands. Stands dominated by short-form *Spartina alterniflora*, when found within the native high marsh, were classed as Intertidal Marsh, i.e., for patches larger than approximately 100 square feet; smaller patches of *Spartina alterniflora* were classed as High Marsh. This is consistent with the New York State Tidal Wetland Inventory Final Report mapping conventions which characterized stands predominantly comprised of short-form *Spartina alterniflora* was intertidal marsh and only included short-form *Spartina alterniflora* in high marsh when it occurred in mixed association with *Spartina patens* and/or *Distichlis spicata* (Martin et al. 1975). *Baccharis haliminfolia* was classified as upland except in instances where it was mixed and dominated by one or more of the high marsh species listed here.

- ***Phragmites australis***: Training sites were located and recorded for *Phragmites* stands including its growth forms (i.e., high vigor versus low vigor forms).

- **Native Coastal Fresh Marshes**: Training sites were located and recorded for narrow-leaved cattail (*Typha angustifolia*) and bulrushes (*Schoenoplectus* sp.). No other species were recorded for coastal fresh marshes.

- **Upland Borders**: Training sites were located and recorded for plants typically found on the landward edges of tidal wetlands including groundsel tree (*Baccharis halimifolia*), beach grass (*Ammophila breviligulata*), switchgrass (*Panicum virgatum*), poison ivy (*Toxicodendron radicans*), and northern bayberry (*Morella pensylvanica*). These plant species were selected because they are commonly observed in mixed herbaceous-shrub upland habitats which are likely to be difficult to differentiate between the adjacent mixed herbaceous habitats present in native high marshes. Note that *Baccharis halimifolia* was treated as an upland species unless mixed native high marsh species listed above.

Other features within and adjacent to marshes, such as mudflats, salt pannes, ponds, and channels/ditches, were identified. Training points were collected for salt pannes, but not for mudflats or open water features (ponds, channels, ditches). Upland habitats dominated by woody trees and shrubs were readily identified on the infrared aerial images and did not require groundtruthing.

Following spectral classification through the Maximum Likelihood classification algorithm and conversion to a vector map layer format, the predominant vegetation types were then merged into their broader tidal wetland categories, i.e., High Marsh, Intertidal Marsh, Fresh Marsh and *Phragmites australis*. It is important to note that prior to merging the vegetation types into the wetland classes, manual corrections were conducted, where necessary, on the vegetation type boundaries via photo-interpretation of the color-infrared imagery. Manual, photo-interpreted corrections were mostly relegated to the upper marsh boundary where classification accuracy was lowest and the vegetation types were often mixed. The native herbaceous marsh areas sustained a relative high classification accuracy and required limited or no manual boundary corrections.

With noted exceptions, definitions of intertidal marsh, coastal fresh marsh, and native high marsh followed New York State Tidal Wetland Land Use Regulations (Part 661) and New York Tidal Wetland Inventory Final Report (Martin et al. 1975) mapping conventions.

Intertidal Marsh – The classification shall be consistent with the NYSDEC mapping conventions and shall consider all areas dominated by tall- or short-form *Spartina alterniflora* to be intertidal marsh.

High Marsh – The New York State Tidal Wetland Land Use Regulations (Part 661) and Martin et al (1975) classify high marshes as areas subject to flooding during spring and storm tides and typically dominated by *Spartina patens* and *Distichlis spicata*. These documents also indicate that low vigor *Spartina alterniflora* and *Limonium carolinianum* may also be present and *Juncus gerardi*, *Scirpus* sp., *Iva frutescens*, and *Baccharis halimifolia* may occur at the upper edge of the high marsh. In general, plant communities dominated by these species were classified as "Native High Marsh", with exceptions noted in the following paragraphs.

Low vigor *Spartina alterniflora* could not be differentiated from short form *Spartina alterniflora* to separate out its occurrence in a high marsh versus intertidal marsh. Therefore, all occurrences of *Spartina alterniflora* were classified by species and grouped into intertidal marsh.

Baccharis halimifolia posed some challenges for the accurate differentiation of native high marsh habitats, as this species occurs at the upper limit of the high marsh. In cases where stands of *Baccharis halimifolia* occur at the landward margin of the high marsh, the tidal wetland-upland boundary commonly occurs within the stand. These stands at the landward margin of the high marsh are classified as salt shrub communities in the Ecological Communities of New York State (Edinger et al. 2002). It was not possible to consistently differentiate the tidal wetland-upland boundary on the infrared aerial imagery where this boundary was located within *Baccharis* stands. Therefore, as guidance for the mapping effort, the TAC established the landward margin of the native high marsh to be seaward margin of such *Baccharis halimifolia* stands.

A similar mapping challenge was also encountered in brackish meadow communities where *Spartina patens* (and sometimes *D. spicata* and *J. gerardii*) is often found landward of the native high marsh (i.e. landward of the *Iva frutescens*-dominated upper margin of the high marsh) in sandy upland areas mixed with non-wetland vegetation. The TAC established that *S. patens*-dominated brackish meadows should – where identification is possible – comprise the high marsh community.

Phragmites australis – The original NYSDEC mapping conventions do not consider *Phragmites* except to note that that *Phragmites australis* often dominates "formerly-connected tidal wetlands", i.e., areas where tidal flow has been artificially restricted or eliminated by structures. In order to quantify the expansion of *Phragmites* on Long Island since 1974, the mapping approach estimated *Phragmites australis* coverage in both 1974 and 2005/2008 imagery. Due to its ability to grow in tidal wetland, freshwater wetland, and upland habitats, determination of an accurate wetland boundary for a monoculture stand of *Phragmites australis* was not possible. Further, one objective of this project was to quantify *Phragmites australis* expansion within and adjacent to tidal wetlands in the study area. Therefore, for the purposes of trends analysis, monoculture stands of *Phragmites australis* (typically high-vigor) were considered a separate classification. It is important to note, however, that low-vigor *Phragmites australis* also commonly mixes with high marsh species such as *Spartina patens*, *Distichlis spicata*, and *Iva frutescens*. Mixed *Phragmites australis*/high marsh stands were classified as high marsh for the tidal wetland delineation and trends analysis.

Coastal Fresh Marsh – The classification was consistent with the NYSDEC mapping conventions and Part 661 and considered areas dominated by the emergent plants narrow-leaved cattail (*Typha angustifolia*) and bulrush (*Schoenoplectus* sp.) to be coastal fresh marsh.

Formerly Connected Wetlands – No wetlands were designated as "formerly connected" as this category has no ecological meaning. Wetlands categorized as formerly connected on the 1974 maps were assessed and classified as either intertidal marsh, native high marsh, native coastal fresh marsh, or more likely, *Phragmites australis*.

Littoral Zone and Coastal Shoals, Bars and Mudflats – These habitat types were not quantified as the classification of these community types depend on the review of water depth data. Water depth cannot be discerned from aerial imagery and adequate GIS bathymetric data were not available for the entire project area; accordingly, these wetland categories were not mapped.

Other Habitats – The following habitats were readily distinguishable, either through supervised classification or manual photointerpretation: 1) Salt Pannes, 2) Ponds, 3) Creeks and 4) Mosquito Ditches. These features were mapped and grouped into a single "unvegetated" classification for the purposes of later extracting their areas from the vegetated wetlands polygons. It is recognized that New York‟s tidal wetlands contain a variety of unvegetated or sparsely vegetated substrates such as sand, gravel, or cobble beaches, salt pannes (which often contain various densities of plant such as *Salicornia sp*. and *Limonium carolinianum*), and mudflats located seaward of or in between vegetated marsh areas. Due to the absence or low density vegetation on these substrates, the image classification used in this study cannot differentiate between traditional salt pannes and marshes that have lost vegetation due to die-off or subsidence.

The classification system described above applied to the photointerpretation of the color-infrared images for Years 2005 (Long Island Sound and Peconic Estuaries) and Year 2008 (South Shore).

For the Year 1974 color-infrared images, only the *Phragmites australis* class and the salt pannes, ponds, creeks and mosquito ditches class were mapped. The newly mapped classes for Year 1974 were merged with the existing official 1974 Tidal Wetland classes per the methods described in Subsection "Enhancement of the 1974 Tidal Wetland Boundaries" below.

It is important to note that the supervised classification described above was verified using field "test points." Test points used for verification were different than those used to train the classification algorithm. By comparing the classification map with the test points, the project team conducted the first of two error analyses: 1) test of initial classification accuracy and 2) comparison of the relative error between the manual photointerpretation and the computer-assisted, tidal wetland image classification. See Error Analysis section below.

Digitize tidal wetland boundaries using "heads-up" digitizing techniques

This task was comprised of two subtasks: (1) visual assessment and correction of Year 2005/2008 classifications and (2) enhancement of Year 1974 tidal wetland boundaries. The first subtask, performed by wetland ecologists, entails the visual assessment and correction of tidal wetland polygons developed initially through spectral analysis and classification of Year 2005 and 2008 imagery. Where tidal wetland polygons boundaries and/or their particular classifications were visually determined to be incorrect – based on photo interpretation – the image analyst utilized "heads-up", i.e., on-screen, digitizing techniques to correct the tidal wetland polygons. This entailed the cutting (i.e., splitting) of formerly unique wetland polygons, the merging of adjacent polygons into one wetland category and/or renaming the wetland category within the GIS database.

Based on a review of groundtruthed data and its evaluation with respect to available Year 2005/8 color-infrared imagery, the following overlap or confusion among feature types occurred along with a resulting misclassification of tidal wetland polygons:

- Intersection of the *Phragmites australis* and *Spartina alterniflora* spectral reflectance ranges. The reflectance values for *Phragmites australis* encompass a wide range of values from low vigor to high vigor types. Consequently, there is a small portion of the *Phragmites australis* reflectance range which intersects with that of *Spartina alterniflora*. However, because these species dominate different zones – which are typically well separated spatially – they can be easily distinguished from each other in the color-infrared photography. In addition, the two plant species are texturally different which also aids in differentiating between them in the instances where spectral misclassification may occur.
- Shadows from trees in upland areas with spectral characteristics of water bodies. Tree shadows are present, in varying degrees, across all of the color-infrared images due to variations in sun angle throughout the time period during which images were taken. The darkness of the tree shadows will cause these pixels to be misclassified as other

dark features, in particular, water bodies. Depending upon the orientation of the marsh and its upland areas to the shoreline (i.e., with respect to compass points), the tree shadow can fall across upland habitat, *Phragmites australis* or the high marsh. As part of the "heads-up" digitizing and correction process, the tree-shadow areas were reclassified according to the habitat in which the shadows fall.

- Marsh / Upland transition zones due to vegetation gradients. An important transition zone is the gradient of plant distribution that exists at the boundary between a *Phragmites australis*-dominated high marsh and an upland community without trees. In these settings, dense growth of *Phragmites australis* may extend into the upland and mix with upland vines (such as poison ivy) and shrubs (such as bayberry, multiflora rose, or groundsel bush). These transition areas are more common on the south shore where less steep topography, sandy soils, and maritime influences result in a wider transition zone between high marsh and upland with less adjacent tree cover. As a result, mapping errors resulting from diffuse marsh/upland transition zones are more likely to occur in wetland complexes in the South Shore Estuary than the Long Island Sound. In this case, the presence of a „significant number‟ or ratio of one species to another will determine the boundary. The project will classify the high marsh-upland boundary, for example, based on the presence of a specified percentage of upland species. The project team will classify an area as „upland‟ if upland species covered more than 50% of the ground area.
- Mixed communities at the boundaries between community types. Consider a zone where *Spartina alterniflora* and *Spartina patens* occur in roughly equal proportions (40-60% coverage of each species).
    - Mixed communities of *Spartina alterniflora* and *Spartina patens* were not mapped separately as the image classification algorithm is capable of utilizing the spectral ranges of *Spartina alterniflora* and *Spartina patens* to define discrete boundaries between the two species. However, in addition to being found spatially adjacent within a marsh, these species are also spectrally adjacent. Groundtruth data of mixed *Spartina alterniflora* and *Spartina patens* communities were used to establish a threshold at which pixels were assigned either the value of *Spartina alterniflora* or *Spartina patens.*

Small patches of one community type can be located well within another community type. Where the small patch is less than approximately 100 square feet in area, it will be combined with the larger, surrounding community type. In contrast, the New York Tidal Wetland Inventory Final Report mapping conventions indicated that the smallest wetland category to be routinely identified and delineated within a larger wetland area was five acres, although in many instances smaller areas were mapped (Martin et al. 1975).

Small patches, less than 100 square feet approximately in area, were considered insignificant due to the landscape-scale goal of the study to quantify the magnitude of landscape-level changes in wetlands loss and changes in marsh condition and implications for estuary health and supply of estuarine ecosystem services within approximately 20,000 acres of tidal wetlands within the Long Island Sound, Peconic, and South Shore Estuaries. The results of this project, the observed trends in wetland area and composition change, are intended for use by environmental managers, conservation advocates and elected officials across a variety of regulatory agencies, environmental organizations, and governments. Identification of small spatial scale variation in marsh composition (i.e. less than 100 square feet in area), while of ecological importance due to habitat complexity, was given lower priority under this project than providing landscape-level trends on Long Island‟s tidal wetlands that can be used by regulatory agencies, environmental organizations, and governments.

In addition, the inclusion of patches smaller than 100 square feet would make the map, and its associated spatial database, unnecessarily large and unwieldy. An example can be a small patch of *Spartina alterniflora* within a high marsh stand. If the small patch were „large‟, e.g., greater than approximately 100 square feet, it would be considered a small island and would not be merged into the surrounding community type. A patch of this threshold size or greater of *Spartina alternifora* within a high marsh is likely to indicate a small depression and would be classified as intertidal marsh.

The second digitizing subtask involved the enhancement of the 1974 tidal wetland categories to include the *Phragmites australis* and unvegetated (pannes, ponds, creeks and mosquito-ditch) categories. These new polygons were extracted from the previously digitized 1974 tidal wetland polygons via a spatial "union" operation (Figure 6). Once *Phragmites australis* and unvegetated areas were removed, the acreage of Year 1974 vegetated wetland areas was recalculated to more accurately compare trends to Year 2005/2008.

It is also noted that the enhancement of the 1974 tidal wetland polygons – which were previously digitized by the NYSDEC – were also spatially corrected to match the orthorectified 1974 color-infrared images.

Figure 6: Flax Pond (Complex #103)Tidal Wetlands in 1974
[See Page D4, Appendix I for Locator Map]

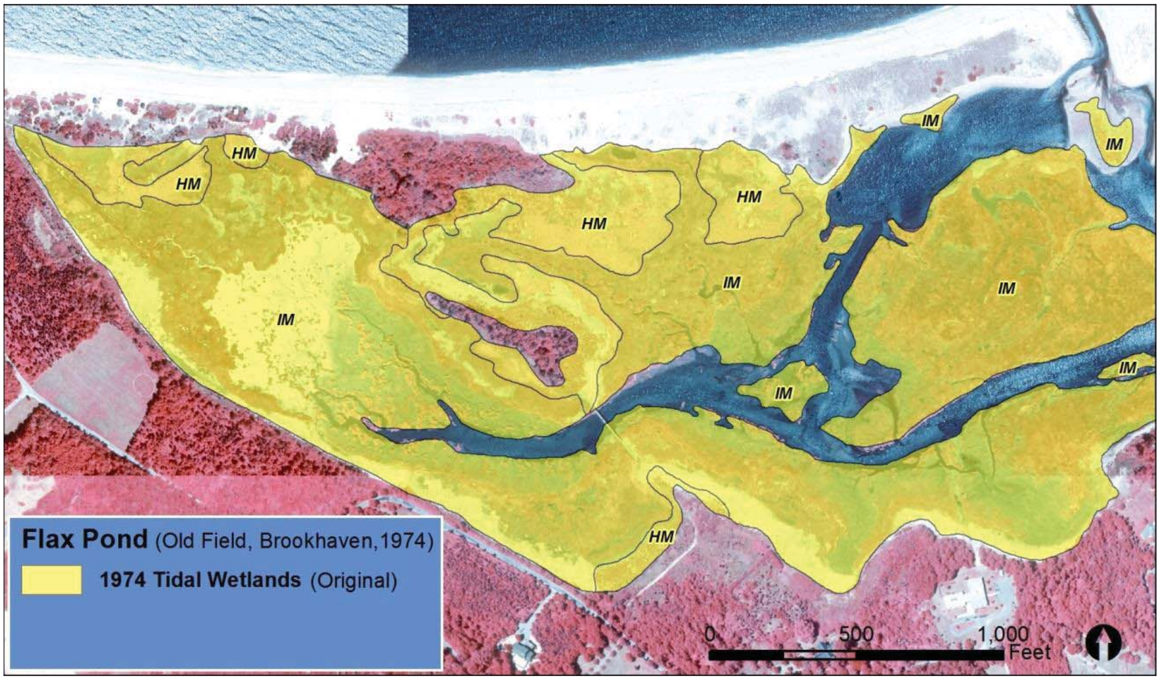
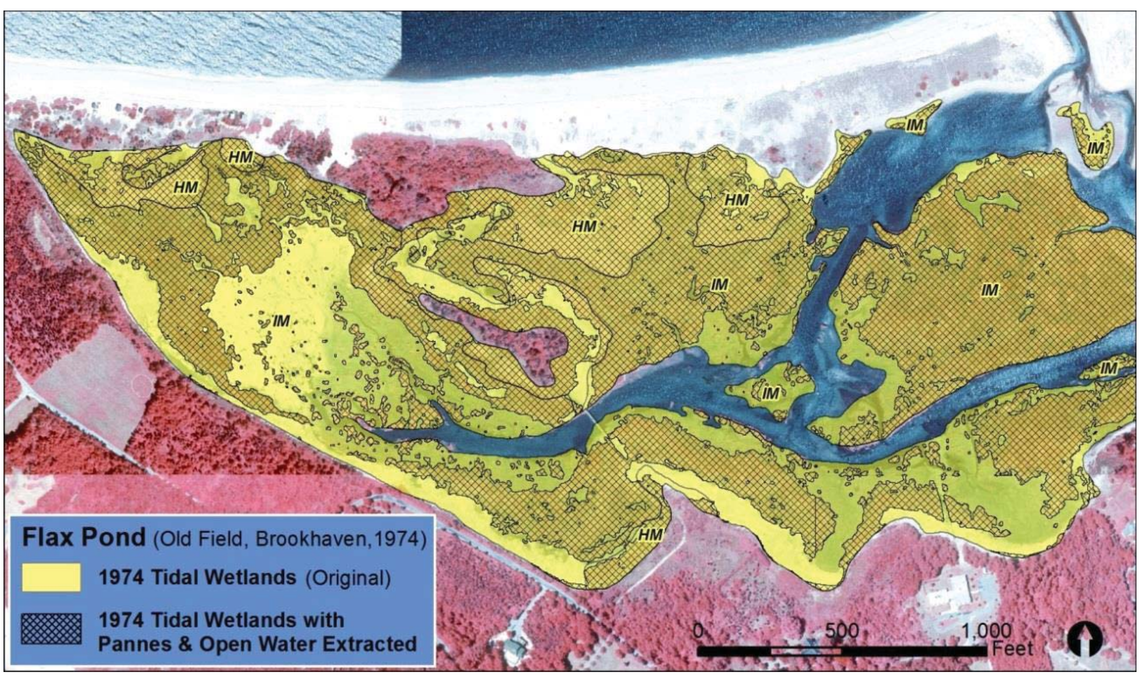


Figure 7: Flax Pond with original delineation (yellow) and enhanced delineation with unvegetated features removed (black hatch)



Trends Analysis

Total areas were calculated for intertidal marsh, high marsh, coastal fresh marsh, and *Phragmites australis* for each tidal wetland complex in 1974 and 2005/2008. The change in total vegetated tidal marsh area and total *Phragmites australis* area for each marsh complex between 1974 and 2005/2008 was calculated and tabulated as a positive or negative area and a positive or negative percentage of change.

Based on the above data, each marsh complex was identified as “stable”, i.e. less than 10% decrease or increase in marsh area between 1974 and 2005/2008, or “at-risk”, i.e. more than 10% loss in marsh area. The area and percentage change data calculated for each marsh complex described above were then summed to provide trends in tidal marsh change for larger geographic areas including:

- The entire study area;
- Major estuary systems (Long Island Sound, Peconic Estuary, South Shore Estuary); and
- Long Island Towns (North Hempstead, Oyster Bay, Hempstead, Huntington, Babylon, Smithtown, Islip, Brookhaven, Riverhead, Southold, Southampton, East Hampton, and Shelter Island);

There were a number of reasons for calculating the changes in marsh features over larger geographic areas, i.e., areas larger than the scale of the marsh complex. The NEIWPCC and the broader environmental community wish to quantify marsh loss at the regional level and for major estuary systems in order to support environmental, land use and other policy initiatives. Administrative boundaries do not necessarily follow ecosystem zones or were not purposely configured to encompass critical habitats, but local officials at the County and Town levels are interested in understanding their share of responsibility for previous marsh changes and their role in future marsh protection. Agencies of the State of New York (e.g. the Department of State and the Department of Environmental Conservation), who are charged with the proper management of wetland resources, would benefit from an understanding of marsh changes within the entire study area and the three estuary systems.

To calculate 1974 to 2005/2008 tidal wetland trends for the larger geographic areas listed above, the changes in area for each marsh feature within the specified geographic area are simply summed, or tallied, based on which marsh complexes (or portions thereof) fall within the larger geographic area. Although the larger geographic areas may or may not have a specific ecological significance, the determination of marsh loss (change) trends at different geographic scales provide important information for policy makers, elected officials and environmental managers.

Results and Discussion

Long Island Tidal Wetlands Trends

Long Island‟s estuaries have lost 13.1 % of native intertidal (IM), high marsh (HM), and coastal fresh marsh (FM) communities between 1974 and 2005/2008 (Table 3). Appendix II provides the results of the imagery analysis and trends in marsh area for each of the identified tidal wetland complexes. The Peconic Estuary and South Shore Estuaries have slightly lower percentages of marsh loss (-10.5% and -11.6%, respectively) compared to the Long Island Sound Estuary (-22.6%). Collectively, Long Island‟s three estuary complexes lost, on average, 85 acres of native marsh annually over this time. These results are consistent with previous studies documenting marsh loss on Long Island (Hartig et al, 2002; Mushacke, 2007; Ciapetta, 2010; Browne, 2011) and the regional loss of salt marsh observed throughout the northeastern United States. For example, substantial marsh loss in the late 20th century has similarly been reported in Connecticut (31-86% loss between 1974-2004; Tiner et al, 2006) Cape Cod (up to 50-63% between 1952/1971-2005; Smith, 2009), and Chesapeake Bay (16-29% between 1850-1990; Wray et al, 1995). These previous studies also suggest that marsh loss was likely occurring on Long Island prior to 1974, and throughout the latter half of the 20th century.

The loss of nearly 3,000 acres of native wetlands implies a substantial loss of ecosystem services in Long Island‟s estuaries. Salt marshes provide many critical benefits to human communities including fish and shellfish production, protection of shorelines from coastal storms, erosion control and sediment stabilization, water filtration through nutrient and sediment removal, carbon sequestration, and recreation and tourism (Barbier et al, 2011). Many inter-related factors contribute to marsh loss in the northeastern United States including sea level rise, eutrophication (Deegan et al, 2012), low sediment supply, altered estuary bathymetry and inlet morphology, creek and panne expansion, expansion of invasive *Phragmites australis,* erosion caused by recreational and commercial vessel wakes, altered precipitation regimes (Watson et al. 2014), and trophic cascades resulting from interactions between marsh herbivores and their predators (Silliman et al, 2005).

Long Island‟s wetland complexes exhibit tremendous variability in their stability since 1974 with observed rates of change in area varying between +210.9 and -100.0% (for marshes greater than 1 acre in 1974). This variability results from 1) the multiple mechanisms contributing to marsh loss, 2) variation in the initial size and community composition of each marsh, and 3) variation in physical, biological, and anthropogenic conditions among Long Island‟s marshes. Smaller wetland complexes are more likely to have large magnitude changes in marsh area (either gains or losses) than large complexes. For example, all marshes showing absolute changes in area (including *Phragmites australis*) greater than 50% (i.e. more than 50% gain or more than 50% loss) were observed in marshes less than 30 acres in 1974 area. Larger variability in the stability of smaller marshes is not surprising considering that these marshes are likely to change greatly in response to localized conditions that increase or decrease the survivorship of native marsh vegetation or the erosion or accumulation of sediments.

Table 3: Tidal Wetland Area Change (1974-2005/2008) in Long Island's Estuaries

Estuary	1974 IM + HM + FM Area (acres)	2005/2008 IM + HM + FM (acres)	Change in IM + HM + FM (acres)	Change in IM + HM + FM (%)
Long Island Sound	2,891.8	2,237.6	-654.2	-22.6
Peconic Estuary	3,443.9	3,077.5	-356.4	-10.4
South Shore Estuary: Total	14,651.8	12,959.4	-1,692.3	-11.6
South Shore Estuary: East Rockaway Inlet to Fire Island Inlet	10,407.2	9,027.6.0	-1,379.6	-13.3
South Shore Estuary: Fire Island Inlet to Smith Point	2,193.7	1,885.3	-308.3	-14.1
South Shore Estuary: Moriches and Shinnecock Bays	1,956.2	2,017.1	60.9	+3.1
South Fork Ponds: Mecox Bay, Sagaponack Pond, & Georgica Pond	62.7	7.3	-55.4	-88.4
TOTAL	21,050.2	18,281.8	-2,758.3	-13.1

In addition to large reductions in total marsh area, the biological composition and geophysical structure of Long Island"s tidal wetlands has also changed greatly between 1974 and 2005/2008. While this study provides analysis of trends in Long Island"s marsh area only between 1974 and 2005/2008, it is important to note that previous studies on marsh loss on Long Island and the northeastern United States (Wray et al, 1995; Hartig et al, 2002; Tiner et al, 2006; Smith, 2009; Ciapetta, 2010; Browne, 2011) indicate that marsh loss was likely occurring on Long Island prior to 1974, and throughout the latter half of the 20th century. The major changes in the biological and physical structure of Long Island"s marshes observed in this study are described in the following sections and include:

- Conversion of High Marsh to Intertidal Marsh
- Formation of Pannes and Ponds Within Marshes
- Conversion of Intertidal Marsh Islands to Mudflats

- Widening of Tidal Creeks and Man-made Ditches
- Erosion and Retreat of Seaward Edge
- *Phragmites australis* Encroachment

Conversion of High Marsh to Intertidal Marsh and Panne/Pond Formation

Both the conversion of high marsh to intertidal marsh and the formation of expansive panne and pond areas within marshes are indicators of marsh drowning or waterlogging. Marsh drowning may be due to the interacting effects of the failure of marsh accretional processes (such as deposition of organic sediments and accumulation of plant biomass) to keep pace with relative sea level rise and marsh subsidence related to plant mortality and subsequent decomposition of root biomass (Fagherazzi et al, 2012; Kirwan and Megonigal, 2013). This failure is due to physiological stresses such as increased flooding, sulfide accumulation, and nutrient enrichment (Turner et al, 2009; Wigand et al, 2014).

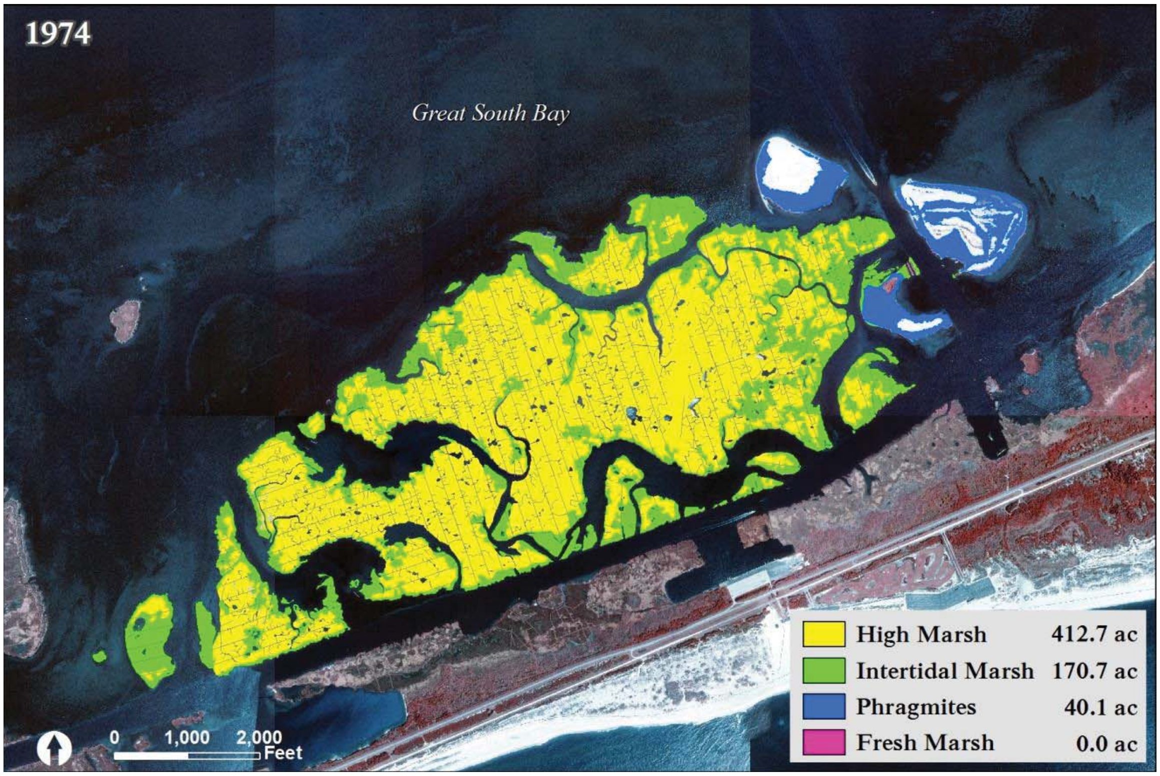
Marsh drowning has resulted in extensive conversion of native high marsh habitats to either intertidal marshes or pannes. Marshes in the Long Island Sound, Peconic, and South Shore estuaries exhibit indicators of marsh drowning with a general trend towards panne formation in the western end of the Long Island Sound and South Shore estuaries and high marsh to intertidal marsh conversion in the eastern end of the Long Island Sound and South Shore estuaries and the Peconic Estuary.

While overall rates of marsh loss over the study period range from -10.4 to -22.6%, loss of high marshes is occurring at a more rapid pace. As shown in Table 4, loss of native high marsh habitats range from -17.3 to -29.7% in the major estuary systems with a total loss of -27.1% (2,084.3 acres) of high marsh habitats. In many wetland complexes, the loss of high marsh is substantially greater than the estuary-wide totals. For example, the fifteen marsh complexes with greatest reduction in acreage of high marsh (shown in Table 5) lost -20.2 to -89.2 percent of the high marsh areas present in 1974. The cumulative area of high marsh lost in these fifteen complexes (1,467.4 acres) accounts for approximately 68% of the Long Island"s high marsh losses. High marsh can transition to intertidal marsh as observed at Cedar & Nezeras Islands (Complex ID # 401, Figure 8) or pannes and ponds at Timber Point (ID # 431, Figure 9).

Due to the reduced frequency and duration of flooding, high marsh habitats exhibit both greater plant diversity and are utilized by several avian species for nesting. Approximately thirty New York State endangered, threatened, or rare plant species are endemic to high marsh habitats (New York Natural Heritage Program, 2013). Several species of birds are high marsh-nesters, such as marsh wren, salt marsh sharp-tailed sparrow, American black duck, clapper rail, willet, and black-crowned night heron. In addition, a wide variety of wading birds, waterfowl, swallows, and terns forage in and above high marsh habitats. The disproportionate loss of high marsh habitats through conversion to intertidal marsh or panne indicates that Long Island"s marshes are becoming less suitable for these protected or declining species.

Figure 8: Cedar & Nezeras Islands (Complex ID #401)

[See Page F3, Appendix I for Locator Map]



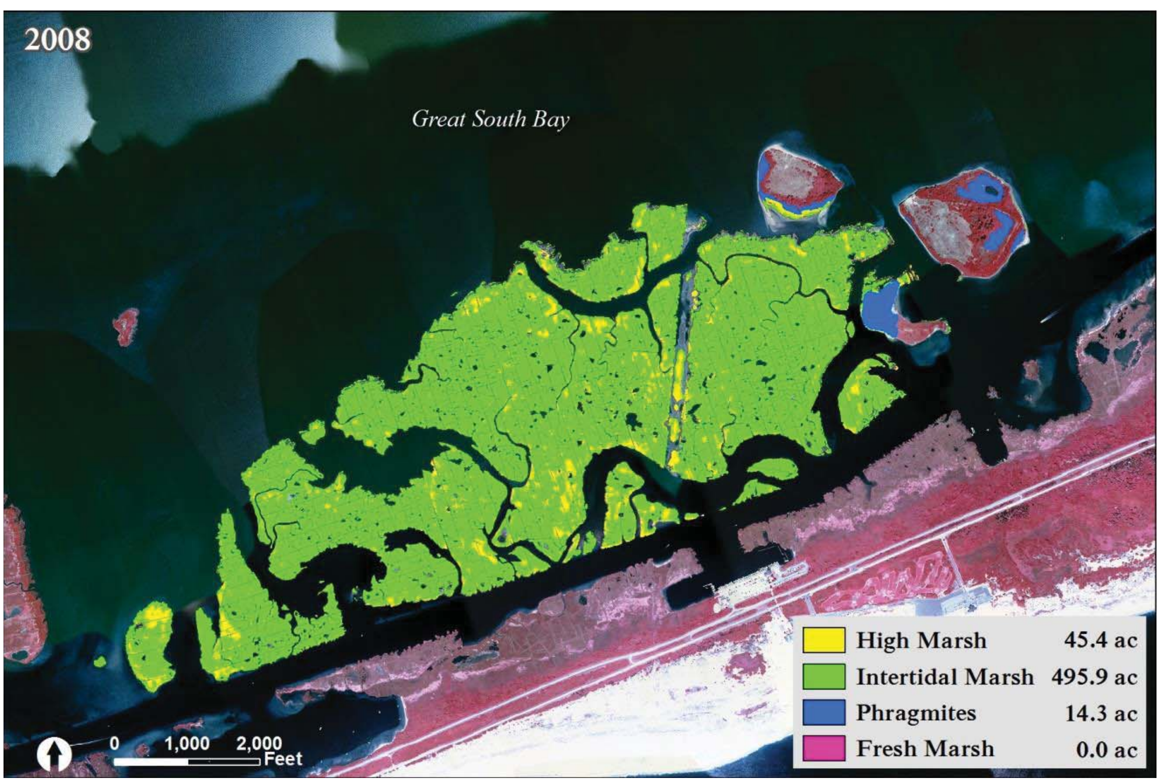
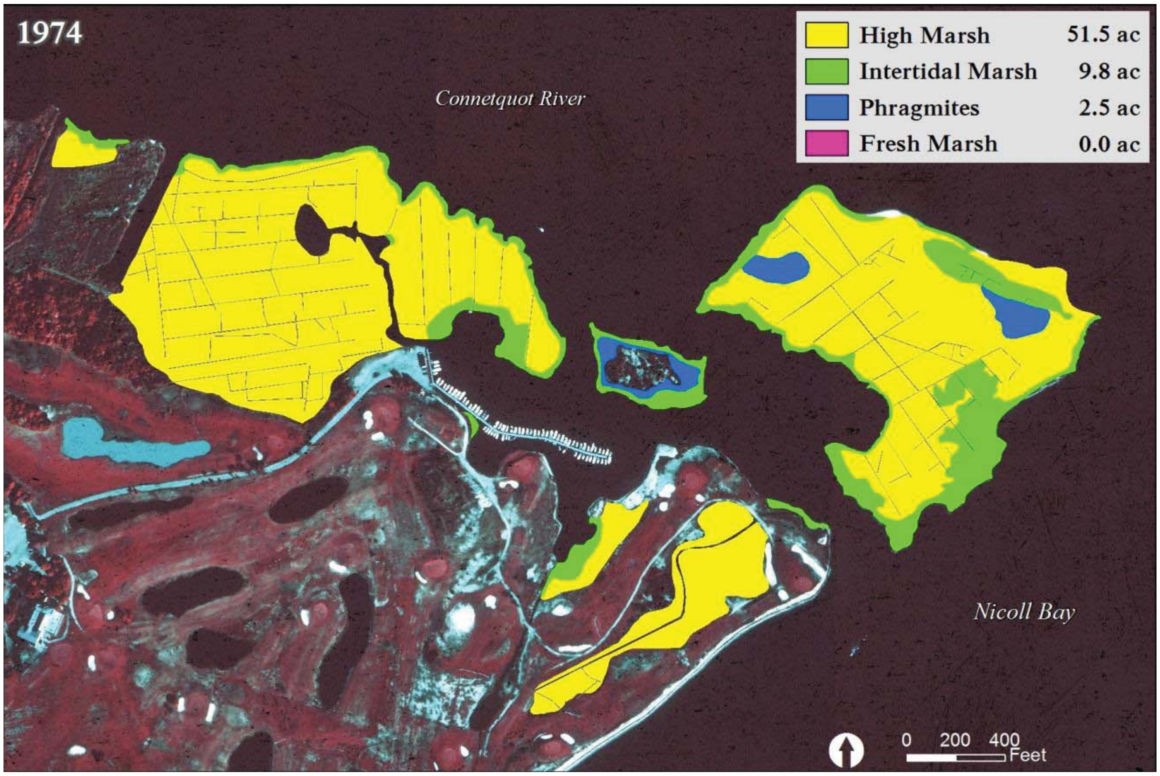
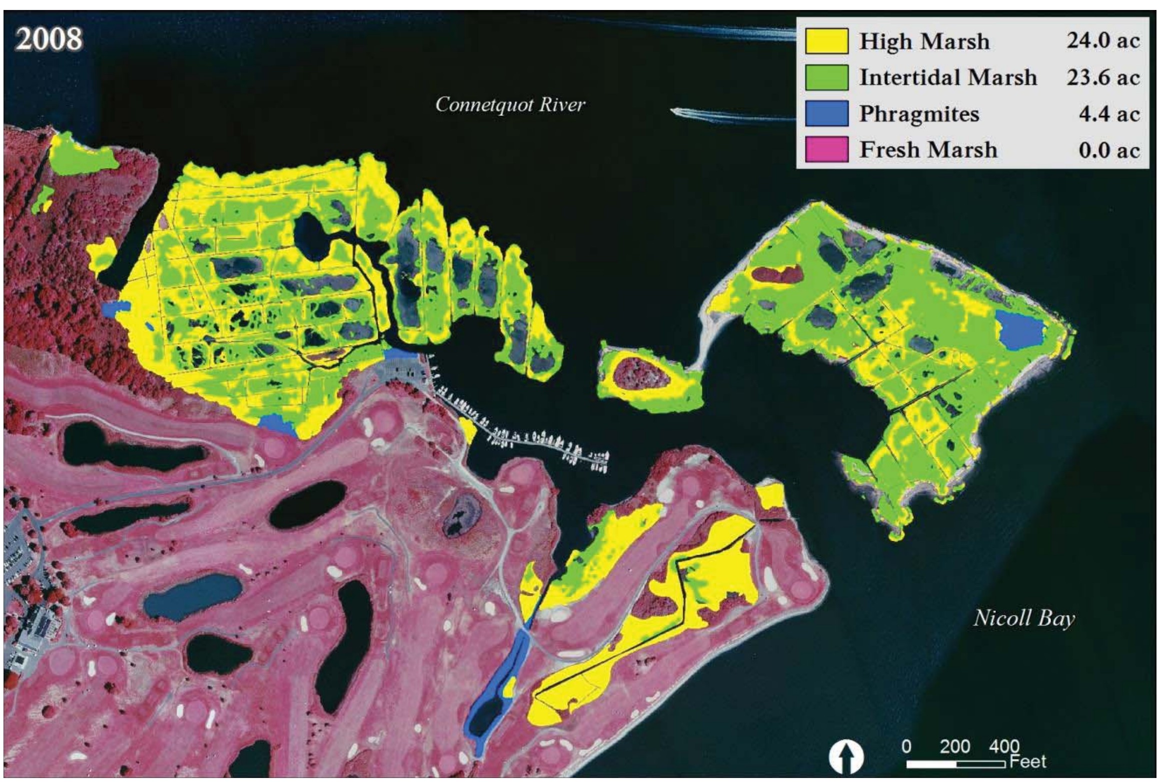


Figure 9: Timber Point (Complex ID #431)
[See Page F4, Appendix I for Locator Map]





In this study, *Iva frutescens* stands are included in the areas mapped as high marsh as are mixed stands of *Iva frutescens* and low vigor *Phragmites australis* (i.e. shoots less than approximately 6 feet in height) and low vigor *Phragmites australis* stands due to the difficulty in differentiating these cover types. Qualitative review of the mapped marshes suggests that *Iva frutescens* and *Iva frutescens/Phragmites australis* stands have increased in abundance between 1974 and 2005/2008. Increases in *Iva frutescens* and *Iva frutescens*/*Phragmites australis* stand area in high marshes (particularly landward expansion of these stands) could partially offset losses in native herbaceous high marsh dominated by *Spartina patens*, *Distichlis spicata*, and *Juncus gerardii*. Accordingly, it is likely that the native herbaceous high marshes have experienced greater losses than the high marsh loss trends presented in Table 4.

The approximately 30% loss of high marsh habitats throughout Long Island between 1974 and the mid 2000‟s and resulting loss of ecosystem services and habitat for wildlife and rare plants demands restoration efforts in complexes with greatest losses of high marsh area (Table 5) and increased management in the largest remaining high marshes (Table 6). Appendix II provides complete marsh area data for 1974 and 2005/2008 imagery for all marsh cover types and marsh change data for each of the identified marsh complexes.

Table 4: High Marsh Area Change (1974-2005/2008) in Long Island's Estuaries

Estuary	1974 HM Area (acres)	2005/2008 HM Area (acres)	Change (acres)	Change (%)
Long Island Sound	950.2	785.9	-164.3	-17.3
Peconic Estuary	1,862.0	1,393.8	-468.2	-25.1
South Shore Estuary	4,856.8	3,414.8	-1,442.0	-29.7
East Rockaway to Fire Island Inlet	2,306.1	1,526.8	-779.3	-33.8
Fire Inlet to Smith Point	1,547.7	998.3	-549.4	-35.5
Moriches and Shinnecock Bay	1,003.0	889.7	-113.3	-11.3
South Fork Ponds	13.6	3.8	-9.8	-71.8
Long Island Total	7,682.6	5,598.3	-2,084.3	-27.1

Table 5: Wetland Complexes with Greatest High Marsh Area Change (1974-2005/2008)

Complex (ID#)	1974 High Marsh (acres)	2005 High Marsh (acres)	Δ High Marsh (acres)	Δ High Marsh (%)
Cedar & Nezeras Islands (401)	412.71	45.44	-367.28	-89.0
Captree Island & Seaganus Thatch (410)	276.85	96.61	-180.2	-65.1
Fireplace Neck & Carmans River West (461)	231.54	104.47	-127.06	-54.9
Fire Island National Seashore (445)	357.72	241.40	-116.33	-32.5
Crab Meadow (222)	157.92	60.69	-97.23	-61.60
Accabonac Harbor (156)	179.24	90.61	-88.63	-49.45
Smith Point County Park East (478)	241.16	168.90	-72.26	-30.0
Marsh Islands North of State Boat Channel (386)	73.90	18.79	-55.1	-74.6
Northwest Creek (165)	106.71	52.02	-54.69	-51.25
Carmans River East (462)	166.30	118.36	-47.94	-28.8
Napeague Meadows (154)	231.79	185.06	-46.73	-20.16
South Line Island (382)	66.32	22.80	-44.51	-65.6
Dune Road Marsh & Islands West (516)	149.62	107.37	-42.25	-28.2
Gilgo & Great Islands (394)	120.77	77.81	-42.96	-35.6
Wading River Marsh (87)	105.74	65.24	-40.49	-38.34

Table 6: Wetland Complexes with Largest High Marsh Areas in 2005/2008

Complex (ID #)	2005/2008 High Marsh (acres)
Fire Island National Seashore (445)	241.4
Napeague Meadows (154)	185.1
Smith Point County Park East (478)	168.9
Carmans River East (462)	118.4
Fireplace Neck & Carmans River West (461)	104.5
Dune Road Marsh & Islands West (516)	107.8
Tobay Sanctuary West (385)	97.6
Captree Island & Seaganus Thatch (410)	96.6
Hubbard Creek (14)	94.8

Complex (ID #)	2005/2008 High Marsh (acres)
Accabonac Harbor (156)	90.6
Gilgo & Great Islands (394)	77.8
Lawrence Marsh (322)	76.9
Quintuck Creek (429)	69.4
North & South Green Sedge Islands (325)	67.7
Wading River Marsh (87)	65.2

Many wetland complexes with large losses of high marsh habitats through marsh drowning and subsidence exhibit large gains in intertidal marsh as shown in Table 7, which presents the marsh area change for each of the complexes identified in Table 5, and Table 8, which presents the marsh complexes with the largest observed gains in intertidal marsh area. In contrast, other marsh complexes with large losses of high marsh habitats exhibited large reductions in total marsh area and due to the conversion of high marsh to pannes, marsh ponds, and open water, as shown at Timber Point (ID # 431, Figure 9). Wetland complexes showing significant losses in intertidal marsh, either in acreage or percent loss, are predominantly located in the western South Shore estuary (acreage) or Long Island Sound (percent loss) (Table 9 and Table 10, respectively). Considered in conjunction with the observed losses of high marsh, the observed trends in intertidal marshes indicate that substantial subsidence/drowning of tidal marshes is occurring throughout Long Island‟s three major estuary systems.

Table 7: Intertidal Marsh Change for Wetland Complexes with Greatest High Marsh Area Change (1974-2005/2008)

Complex	1974 Intertidal Marsh (acres)	2005 Intertidal Marsh (acres)	Δ Intertidal Marsh (acres)	Δ Intertidal Marsh (%)
Cedar & Nezeras Islands (401)	170.7	495.9	325.2	190.5
Captree Island & Seaganus Thatch (410)	315.6	441.0	123.4	38.9
Fireplace Neck & Carmans River West (461)	8.9	115.3	106.4	1200.3
Fire Island National Seashore (445)	123.9	218.0	94.1	75.9
Crab Meadow (222)	84.1	147.6	63.5	75.5
Accabonac Harbor (156)	42.2	123.7	81.4	192.8
Smith Point County Park East (478)	71.9	136.2	64.3	89.4
Marsh Islands North of State Boat Channel (386)	534.0	485.8	-48.2	-9.0
Northwest Creek (165)	41.9	85.2	43.3	103.3
Carmans River East (462)	25.9	47.0	21.2	81.8
Napeague Meadows (154)	5.2	48.0	42.8	823.8
South Line Island (382)	296.1	289.5	-6.7	-2.3
Dune Road Marsh & Islands West (516)	222.0	266.9	44.9	20.2
Gilgo & Great Islands (394)	140.2	176.9	36.7	26.2
Wading River Marsh (87)	31.8	61.1	29.3	92.2

Table 8: Wetland Complexes with Greatest Intertidal Marsh Area Gain 1974-2005/2008

Complex (ID #)	1974 IM Area (acres)	2005 IM Area (acres)	Δ IM (acres)	Δ IM (%)
Cedar & Nezeras Islands (401)	170.7	495.9	325.2	190.5
Captree Island & Seaganus Thatch (410)	317.6	441.0	123.4	38.9
Fireplace Neck & Carmans River West (461)	8.9	115.3	106.4	1200.3
Fire Island National Seashore (445)	123.9	218.0	94.1	75.9
Accabonac Harbor (156)	42.2	123.7	81.4	192.8
Smith Point County Park East (478)	71.0	136.2	64.3	89.4
Crab Meadow (222)	84.08	147.58	63.50	75.5
Tobay Sanctuary West (385)	25.7	79.8	54.1	210.8
Dune Road Marsh & Islands West (516)	222.0	266.9	44.9	19.80
Northwest Creek (165)	41.9	85.2	43.3	103.3
Napeague Meadows (154)	5.2	48.0	42.8	823.8
Gilgo & Great Islands (394)	140.2	176.9	36.7	26.2
Wading River Marsh (87)	31.8	61.1	29.3	92.2
Browns Point to Peters Neck Point (32)	11.9	40.1	28.2	237.5
Elder Island (397)	35.0	60.5	25.6	73.1

Table 9: Wetland Complexes with Greatest Intertidal Marsh Area Loss

Complex (ID #, Town)	1974 IM Area (acres)	2005 IM Area (acres)	Δ IM (acres)	Δ IM (%)
Lawrence Marsh (322, Hempstead)	540.1	461.6	-78.5	-14.5
North & South Green Sedge Islands (325, Hempstead)	279.1	205.4	-73.8	-26.4
Porpoise Channel Islands (105, Smithtown)	106.7	48.5	-58.2	-54.6
Cuba, Middle & East Islands (367, Hempstead)	222.0	170.0	-49.7	-22.6
Jones, Middle & West Cow Islands (364, Hempstead)	317.6	272.3	-45.3	-14.3
Marsh Isl. North of State Boat Channel (386, Oyster Bay)	534.0	485.8	-48.2	-9.0
Pine Marsh (356, Hempstead)	189.8	146.6	-43.2	-22.8
Cinder & North Cinder Islands (344, Hempstead)	120.1	77.3	-42.8	-35.6
East Channel Islands (341, Hempstead)	116.2	73.6	-42.6	-36.7
Garrett Marsh (339, Hempstead)	162.9	120.3	-42.6	-26.1
High Meadow Island (352, Hempstead)	158.7	117.1	-41.6	-26.2
Seadog Island (351, Hempstead)	136.2	96.6	-39.6	-29.1
Smith Meadow Island (353, Hempstead)	172.4	132.6	-39.8	-23.1
Hutchinson River (550, Bronx)	63.7	25.0	-38.7	-60.8
Alder Island/Loop Parkway (350, Hempstead)	173.3	135.2	-38.1	-22.0

Table 10: Wetland Complexes with Greatest Intertidal Marsh Area Loss (Percent)

Complex (ID #, Town)	1974 IM Area (acres)	2005 IM Area (acres)	Δ IM (acres)	Δ IM (%)
Ponquogue Islands (528, Southampton)	11.0	1.0	10.0	-90.7
Sheets Creek (275, North Hempstead)	9.8	1.0	8.8	-89.9
Northport Harbor Bird Island (225, Huntington)	16.8	2.5	14.3	-85.0
Northport Harbor (226, Huntington)	19.3	4.4	15.0	-77.4
LI Sound- Milton Point to Rye Beach (314, Rye)	9.8	2.7	7.1	-72.1
West Pond (261, Oyster Bay)	20.1	7.5	12.6	-62.6
Mitchell Creek (282, North Hempstead)	11.8	4.6	7.2	-61.1
Hutchinson River (550, Bronx)	63.7	25.0	38.7	-60.8
Huntington Harbor (233, Huntington)	18.8	7.6	11.3	-59.9
Cold Spring Harbor East (240, Huntington)	11.1	4.5	6.6	-59.7
Cold Spring Harbor Inner Harbor (241, Huntington)	12.4	5.5	6.9	-55.7
Porpoise Channel (105, Smithtown)	106.7	48.5	58.2	-54.6
West Meadow Island (334, Hempstead)	19.2	9.2	10.1	-52.3
Macy Channel & Georges Island (327, Hempstead)	32.5	16.2	16.2	-50.0

Widening of Tidal Creeks and Man-made Ditches

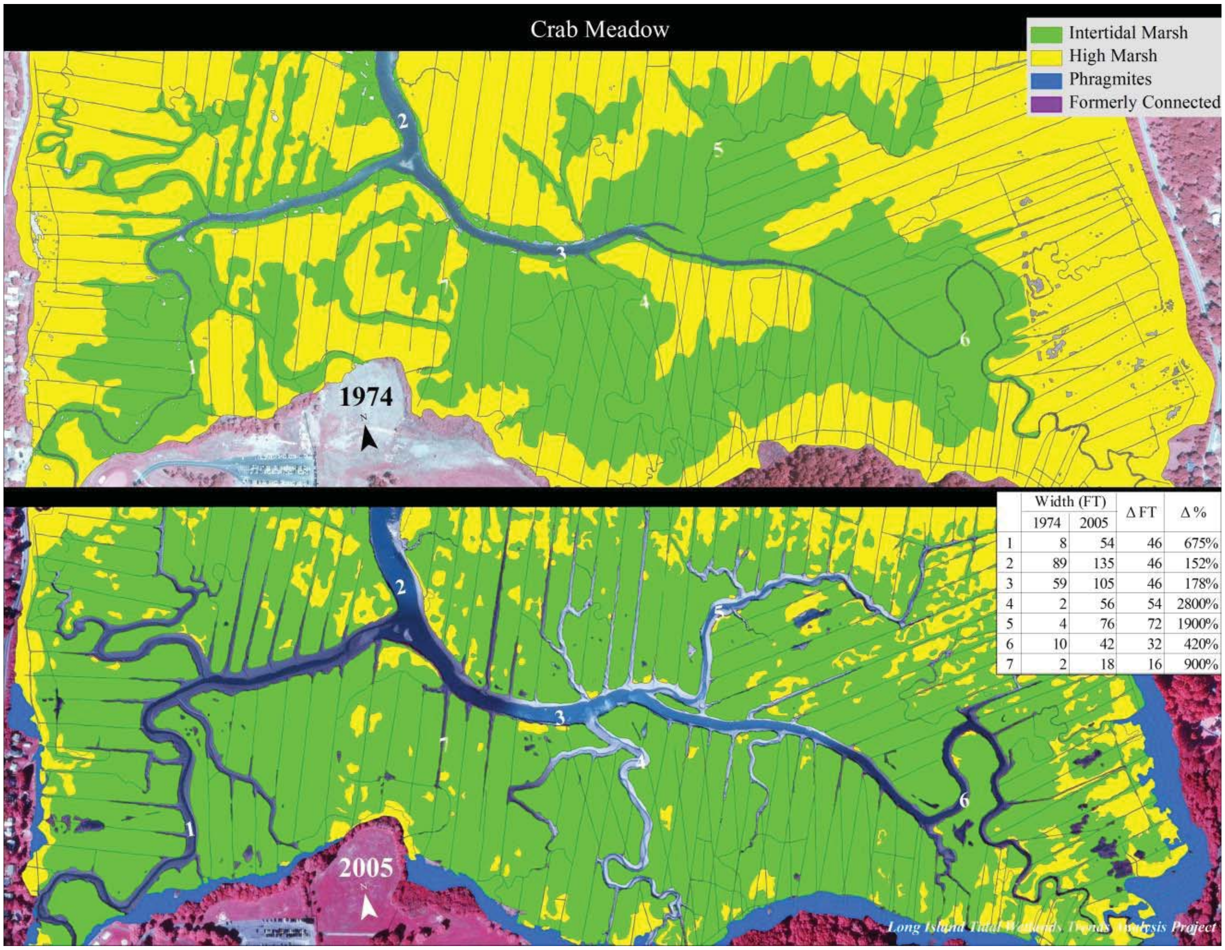
Comparison of the 1974 and 2005/2008 aerial imagery indicates widening of natural tidal creeks and man-made ditches in many Long Island marsh complexes. Creek widening in other degrading marshes in the northeastern United States has been attributed to reduced structural integrity and collapse of creek banks resulting from reduced root biomass, increased decomposition of organic matter, and increased soil water content (Deegan et al, 2012) and, in some cases, herbivory from *Sesarma* crabs (Smith, 2009). Creek width and cross-sectional area is related to the geomorphological and hydrological characteristics of the marsh and surrounding estuary and, as a result, may be impacted by changes in tidal prism (Vandenbruwaene et al, 2013) resulting from relative sea level rise (Stefanon et al, 2012) or anthropogenic changes in inlet or estuary bathymetry. Creek banks, particularly in large tidal channels, may also be subject to erosional forces from wind-driven waves during storms, greater water flow rates, and vessel wakes. At Crab Meadow Marsh (ID# 222) creek and ditch width increased by 1-72 feet (152 – 2,800%) at seven locations within the tidal creek network (Figure 10). At Pine Marsh (ID # 356), creek and ditch width increased by 12-20 feet (192 – 525%) at seven locations within the marsh (Figure 11).

Erosion and Retreat of Seaward Edge

Many marshes in Long Island"s estuaries exhibit pronounced retreat of the seaward marsh edge. The recession of marsh edges is influenced by similar processes to bank collapse and widening.

Figure 10: Crab Meadow (Complex ID #222) - Expansion & Widening of Tidal Creeks & Man-Made Ditches

[See Page D3, Appendix I for Locator Map]

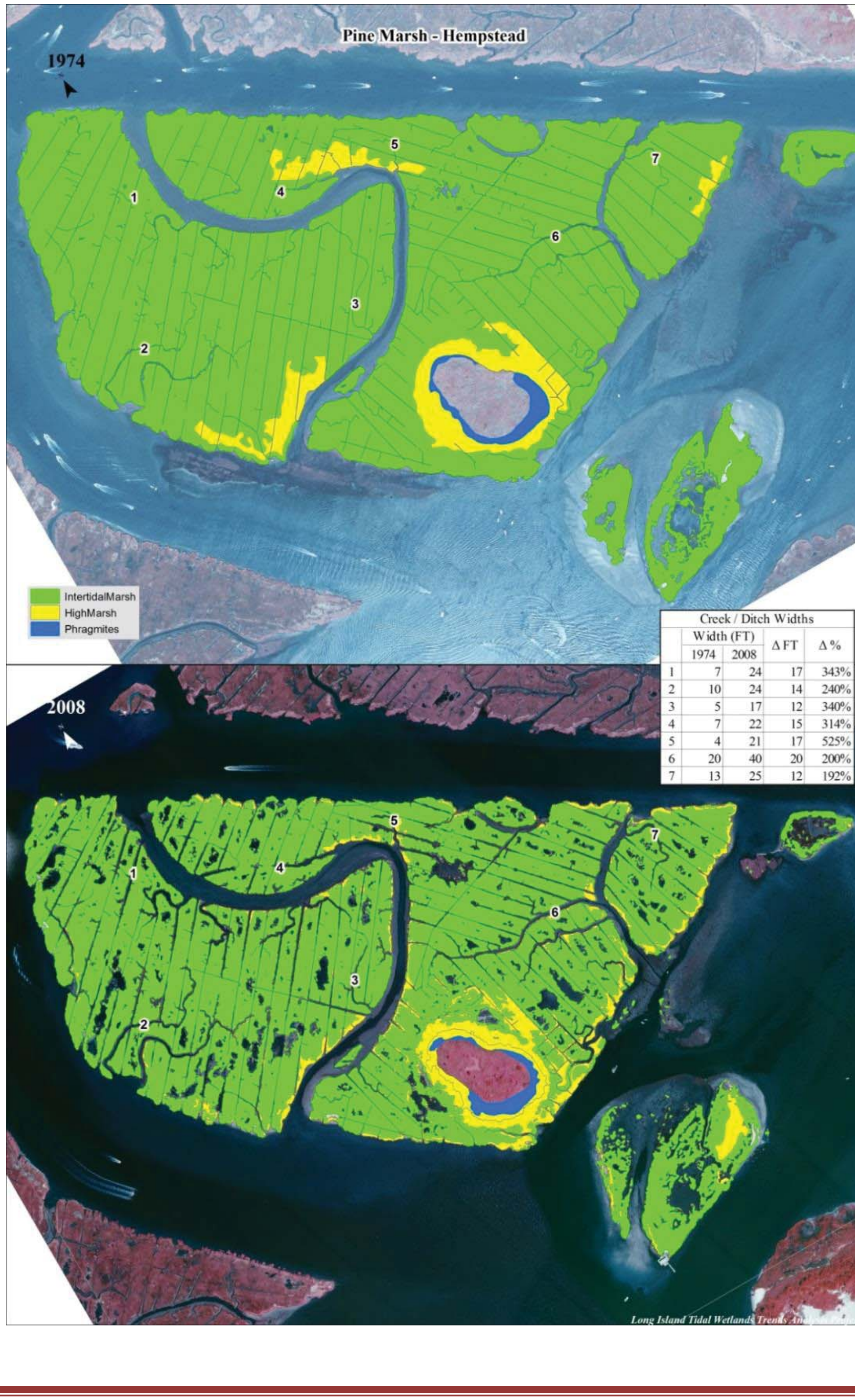


	Width (FT)		ΔFT	Δ%
	1974	2005		
1	8	54	46	675%
2	89	135	46	152%
3	59	105	46	178%
4	2	56	54	2800%
5	4	76	72	1900%
6	10	42	32	420%
7	2	18	16	900%

However, due to their less sheltered position, marsh edges are subjected to greater erosional forces from vessel wakes and during storms. Browne (2011) determined that the marsh edges in Hempstead Bay receded by 17.8 m over the past 90 years. Edge recession was 3.25 times greater in navigation channels dredged through marsh islands compared to natural marsh edges (Browne, 2011). In addition, other anthropogenic disturbance of the seaward edge of the marsh such as the harvesting of ribbed mussels from the marsh or harvesting of shellfish from the subtidal mudflats adjacent to *Spartina alterniflora* may also affect the stability of the seaward edge of the marsh and *Spartina* recruitment.

Receding marsh shorelines were frequently observed in the Long Island Sound and South Shore estuaries (Figure 12-Figure 14). Indicators of eroding marsh shorelines include scalloping of marsh shorelines, thinning of marsh peninsulas, and reticulation of the marsh shoreline (Hartig et al. 2002). The seaward marsh edge of Tobay Sanctuary (ID # 385) has receded by 102 – 200 feet based on measurements at five locations (Figure 12). The seaward marsh edge of the western shoreline of Hempstead Harbor (ID # 267) has receded by an average of 34 feet based on measurements at 32 locations (Figure 13). Thinning of marsh peninsulas and scalloped marsh edges are also visible in Cuba, Middle, and East Islands (ID # 367, Figure 14).

Figure 11: Pine Marsh (Complex ID #356) – Expansion and Widening of Tidal Creeks & Man-Made Ditches
[See Page F2, Appendix I for Locator Map]

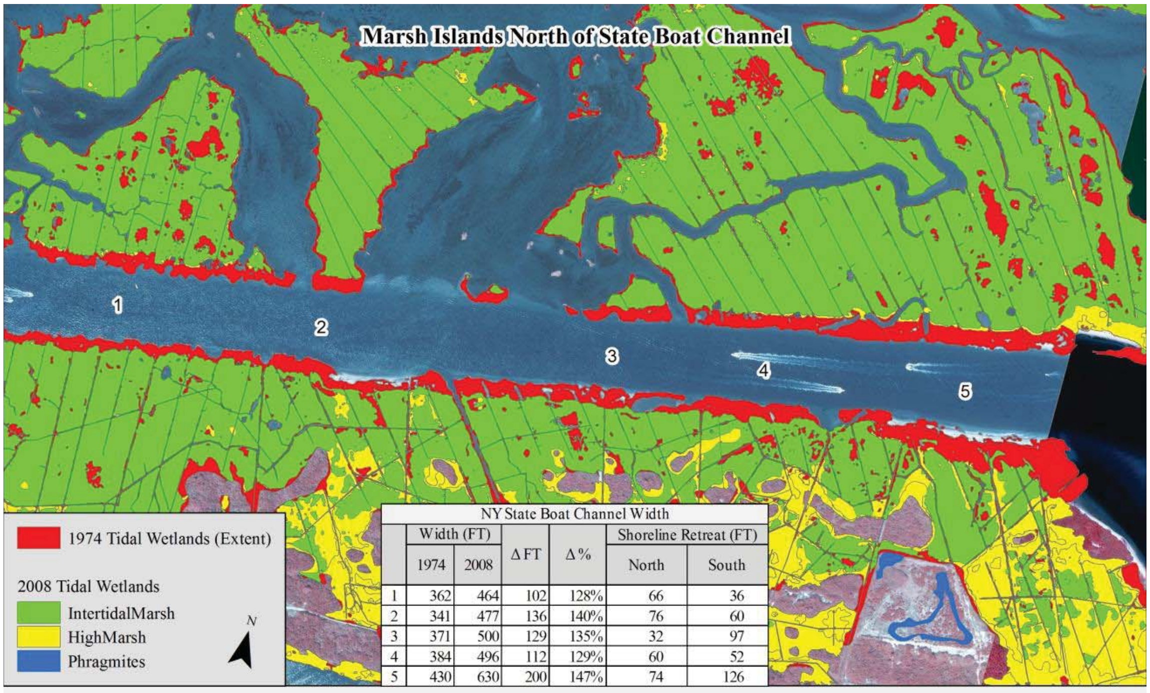


Creek / Ditch Widths

	Width (FT)		ΔFT	Δ%
	1974	2008		
1	7	24	17	343%
2	10	24	14	240%
3	5	17	12	340%
4	7	22	15	314%
5	4	21	17	525%
6	20	40	20	200%
7	13	25	12	192%

Figure 12: Tobay Sanctuary (Complex ID # 385) / Marsh Islands North & South of State Boat Channel (Complex ID # 386) – Marsh Edge Retreat

[See Page F3, Appendix I for Locator Map]



NY State Boat Channel Width						
	Width (FT)		ΔFT	Δ%	Shoreline Retreat (FT)	
	1974	2008			North	South
1	362	464	102	128%	66	36
2	341	477	136	140%	76	60
3	371	500	129	135%	32	97
4	384	496	112	129%	60	52
5	430	630	200	147%	74	126

Figure 13: Inner Hempstead Harbor West Shoreline (Complex ID # 267) – Marsh Edge Retreat

[See Page E2, Appendix I for Locator Map]

Shoreline Change Analysis

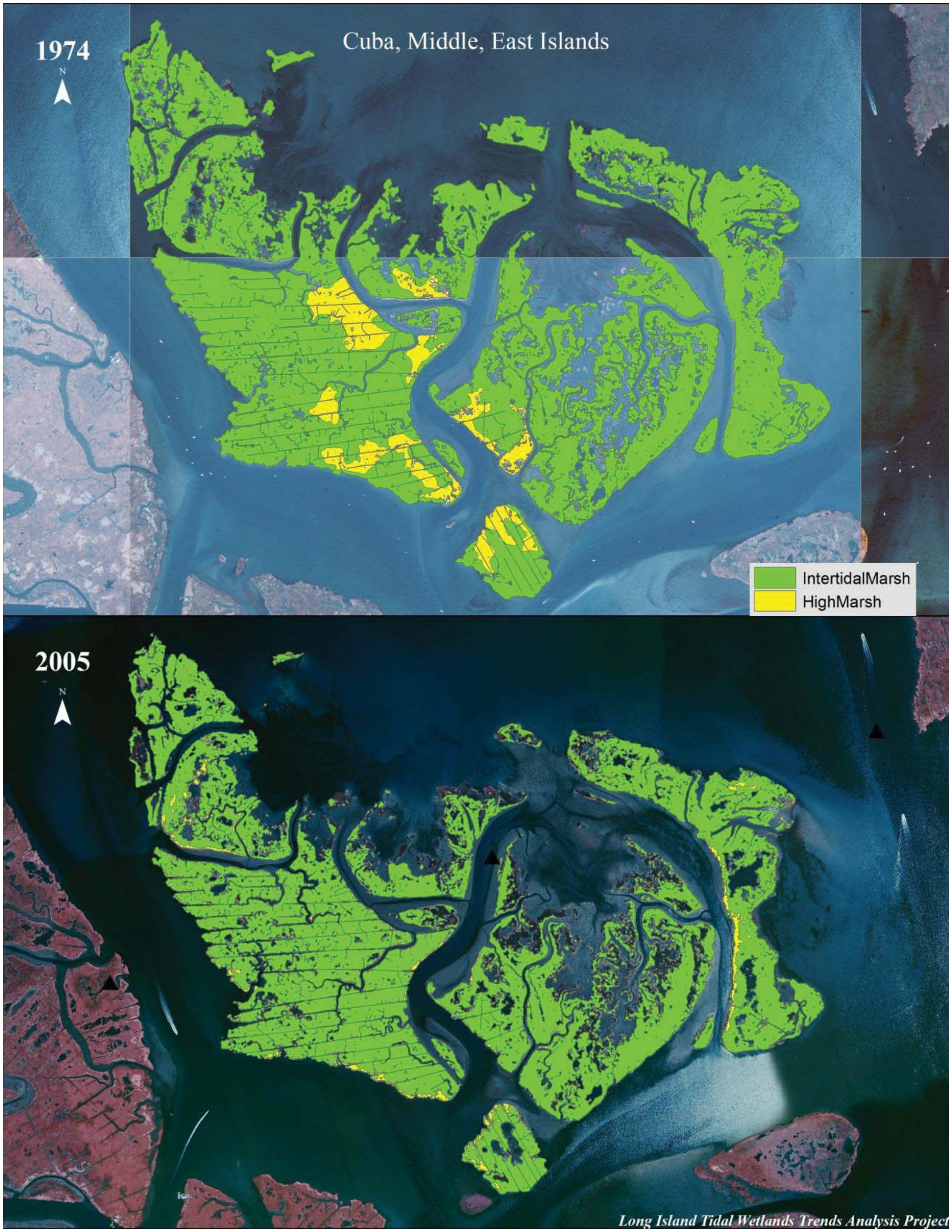
Point	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Shoreline Change (FT)	-54	-40	-23	-79	-50	-20	-3	-63	0	6	21	26	-3	-136	-104	14

Point	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32
Shoreline Change (FT)	14	-24	-15	-30	-49	-32	-45	-21	-35	-33	-72	-121	2	-9	-11	-83

# of Points	32
Average Shoreline Change (FT)	-34
Maximum Shoreline Retreat (FT)	-136
Maximum Shoreline Accretion (FT)	26

Figure 14: Cuba, Middle, and East Islands (Complex ID # 367) – Marsh Edge Retreat

[See Page F2, Appendix I for Locator Map]



Phragmites australis Expansion

Invasive *Phragmites australis* has colonized large areas of high marsh and coastal fresh marsh habitats on Long Island. However, many of Long Island‟s marshes were infested by *Phragmites australis* prior to 1974 (Table 11 and Table 12). *Phragmites asutralis* has historically been found in slightly brackish, tidal fresh marshes, and the borders of salt and brackish marshes; but has increasingly been colonizing salt and brackish marshes (Orson, 1999; Tiner, 2009). *Phragmites australis* poses significant problems for wetlands mapping due to 1) its capability to grow in tidal wetland, freshwater wetland, and upland habitats and 2) its variable spectral signature. Accordingly, *Phragmites australis* identified in Table 11 can be growing in former high marsh habitats or adjacent freshwater wetlands or upland habitats. As such, areas identified as *Phragmites australis* are not necessarily tidal wetlands and are not included in the totals for vegetated marsh area (Table 3).

Long Island‟s estuaries exhibit different trends regarding *Phragmites australis* abundance (Table 12). In the Long Island Sound, Peconic Estuary, and the coastal bays in Southampton and East Hampton, *Phragmites australis* encroachment has contributed to the drastic loss of native high marsh communities and led to the near eradication of areas classified as coastal fresh marsh. Coastal fresh marsh communities consists of various bulrushes (*Schoenoplectus* spp., *Bulboschoenus* spp., and *Scirpus* spp.), narrow-leaved cattail (*Typha angustifolia*), brackish cordgrasses (*Spartina cynosuroides* and *Spartina pectinata*), and emergent plants such as arrow arum (*Peltandra virginica*) and pickerelweed (*Pontederia* spp.) (Martin et al, 1975). Native tidal marshes most susceptible to *Phragmites australis* encroachment include marshes adjacent to disturbed upland, marshes with groundwater contributions, marshes with altered hydrology, or a combination of these impacts. Proliferation of *Phragmites australis* in the headwater creeks and shorelines of Mecox Bay (ID # 540) is likely the result of clearing, disturbance, and nutrient loading in the adjacent uplands (Figure 15). The southern portion of Accabonac Harbor (ID # 156, East Hampton) has seen the loss of approximately 39 acres of coastal fresh marsh and conversion to *Phragmites australis* (Figure 16). Baiting Hollow Marsh (ID # 85, Riverhead) has lost 19.8 acres of high and intertidal marsh to *Phragmites australis* and open water as the result of the gradual shoaling and closure of the marsh‟s inlet to the Long Island Sound (Figure 17).

Phragmites australis has continued to colonize high and coastal fresh marshes within the South Shore Estuary. However, overall *Phragmites australis* coverage within the South Shore estuary has decreased by 11.9%. Loss of *Phragmites australis* areas is most prevalent on former dredge spoil sites located west of Fire Island Inlet and in Moriches and Shinnecock Bays. In some cases, *Phragmites australis*-dominated dredge spoil has naturally transitioned to native marsh communities on dredge spoils, presumably, to sea level rise and/or restoration of tidal hydrology as shown at Pearsalis Hassock (ID # 333, Figure 18) and Blackbank Hassock (ID # 336, Figure 19). In other cases, *Phragmites australis*- dominated dredge spoils may have transitioned to upland vegetation (perhaps due to increase in elevation resulting from repeated use of the spoil site) or bare sand (perhaps due to recent use of the spoil site).

Table 11: Phragmites Area Change (1974-2005/2008) in Long Island's Estuaries

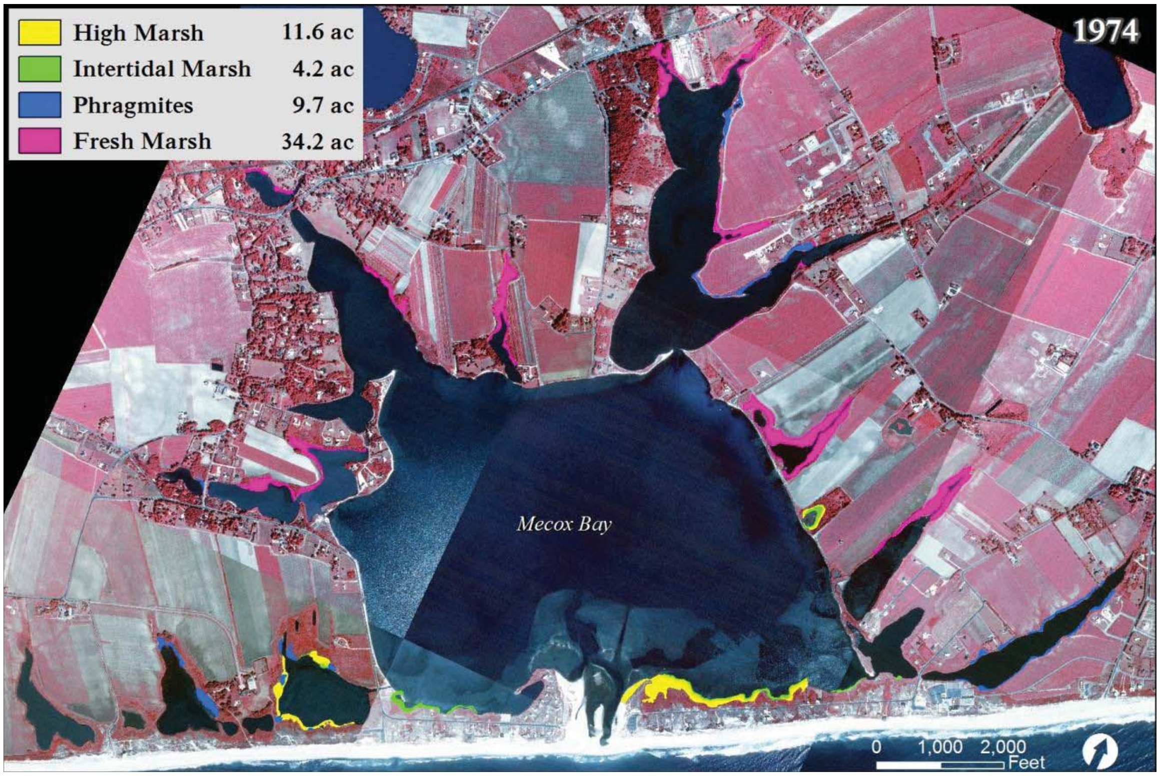
Estuary	1974 *Phragmites* Marsh (acres)	2005/2008 *Phragmites* Marsh (acres)	Change in *Phragmites* (acres)	Change in *Phragmites* (%)
Long Island Sound	317.1	423.7	106.6	+33.6
Peconic Estuary	304.2	573.6	269.3	+88.5
South Shore Estuary: Total	1,839.0	1,620.2	-218.8	-11.9
South Shore Estuary: East Rockaway Inlet to Fire Island Inlet	582.3	370.4	-211.9	-36.4
South Shore Estuary: Fire Island Inlet to Smith Point	786.1	944.0	157.9	+20.1
South Shore Estuary: Moriches and Shinnecock Bays	470.7	305.8	-164.8	-35.0
South Fork Ponds: Mecox Bay, Sagaponack Pond, Georgica Pond	21.5	106.5	85.0	+395.7
TOTAL	2,481.8	2,724.0	242.1	+9.8

Table 12: Wetland Complexes with Greatest Increase in Phragmites Area (1974-2005/2008)

Complex (ID #, Town)	1974 *Phragmites* (acres)	2005 *Phragmites* (acres)	Δ *Phragmites* (acres)
Mecox Bay & Beach (540, Southampton)	9.7	73.2	63.5
Carmans River East (462, Brookhaven)	35.8	96.4	61.6
Accabonac Harbor (156, East Hampton)	3.0	41.4	38.5
Fireplace Neck & Carmans River West (461, Brookhaven)	26.6	60.3	33.6
Jones Beach West Tip (371, Hempstead)	0.0	25.5	25.5
Northwest Creek (165, East Hampton)	32.2	55.6	23.4
Gardiner's Island Bostwick Creek (109, East Hampton)	0.0	22.9	22.9
Wading River Marsh (87, Riverhead)	7.7	25.6	17.9
Carmans River Upstream FM (463, Brookhaven)	46.1	61.8	15.7
Alewife Brook & Pond (163, East Hampton)	1.1	16.5	15.4
Little Northwest Creek (167, East Hampton)	0.0	15.0	15.0
Pepperidge State Tidal Wetlands (434, Islip)	21.8	36.2	14.4
Indian Creek (435, Islip)	21.3	35.6	14.3
Crab Meadow (222, Huntington)	6.2	19.9	13.7

Figure 15: Mecox Bay (Complex ID #540) – *Phragmites australis* Expansion

[See Page D9, Appendix I for Locator Map]



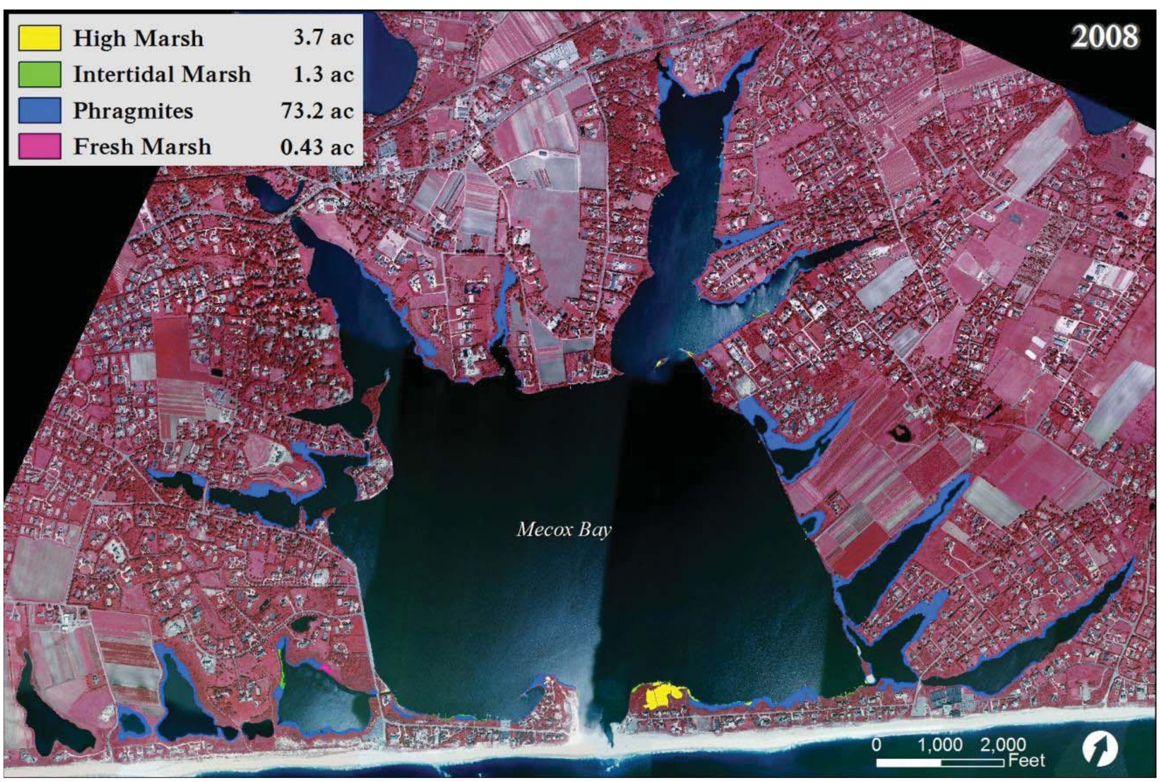
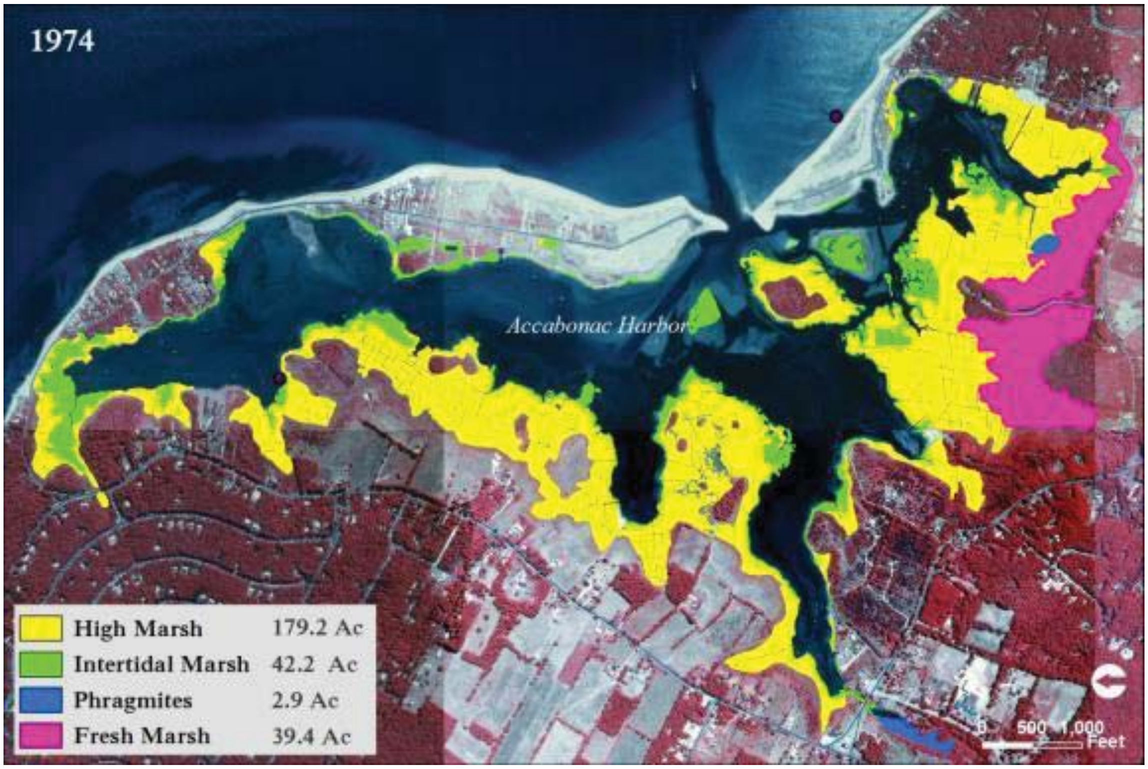


Figure 16: Accabonac Harbor (Complex ID #156) – Phragmites australis Expansion
[See Page C10, Appendix I for Locator Map]



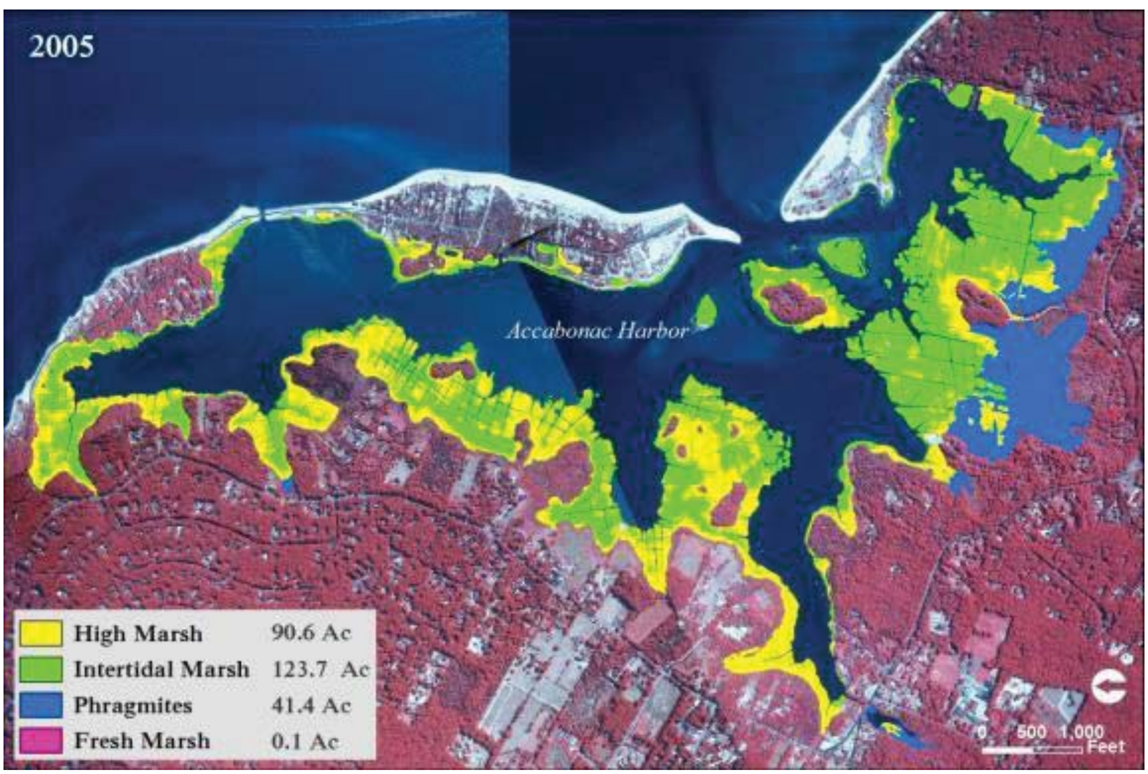


Figure 17: Baiting Hollow Marsh (Complex ID #85) – Phragmites Expansion

[See Page D6, Appendix I for Locator Map]

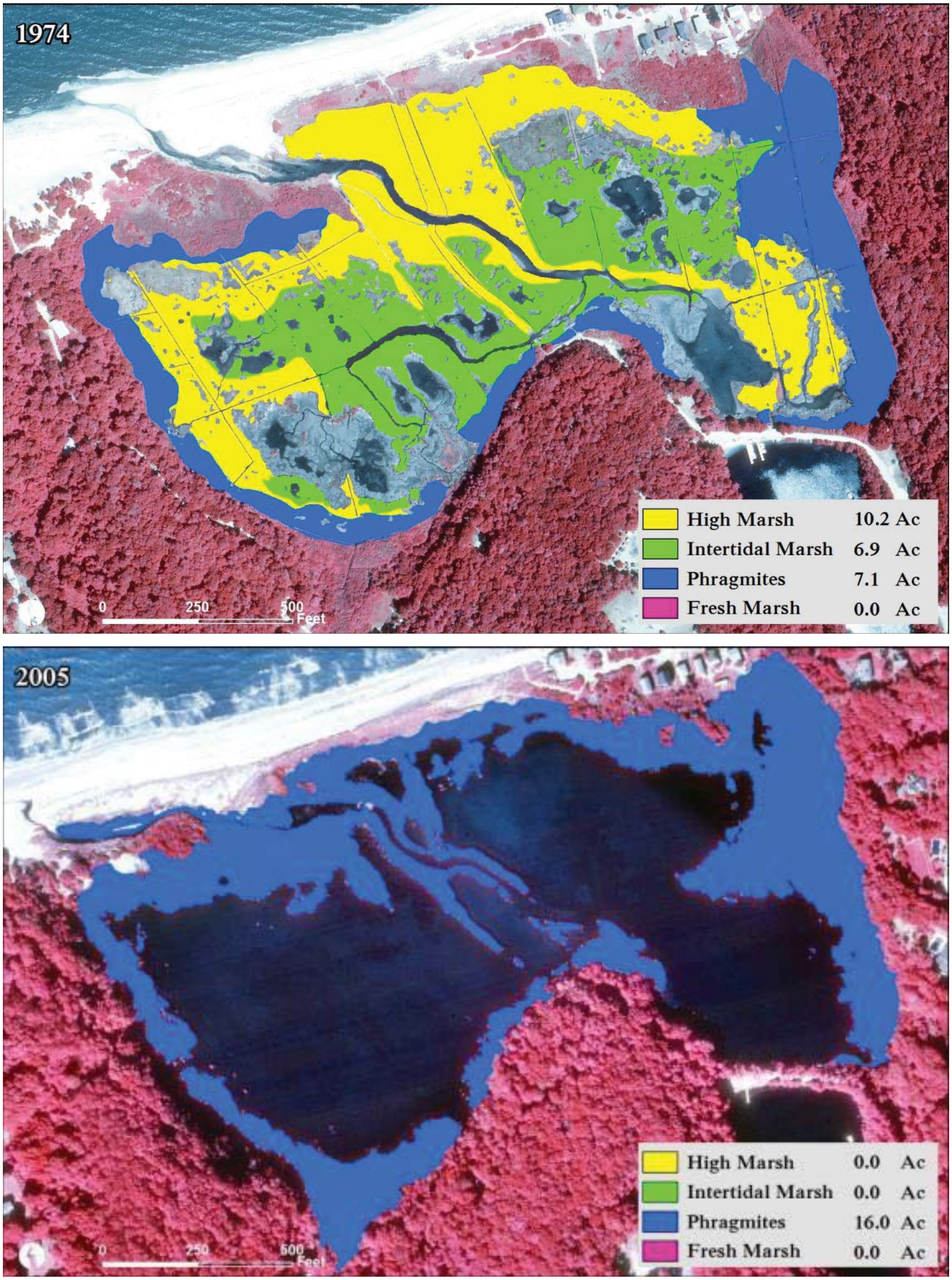


Figure 18: Pearsalis Hassock (Complex ID #333) – Conversion of former *Phragmites australis* Stands to Native Marsh and Upland
[See Page F1, Appendix I for Locator Map]