DISTRIBUTIONAL CHANGE IN SEAGRASS AS AN ECOLOGICAL INDICATOR OF ESTUARINE HEALTH FOR BARNEGAT BAY – LITTLE EGG HARBOR, NEW JERSEY.

By

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ABSTRACT OF THE THESIS

Distributional change in seagrass as an ecological indicator of Estuarine Health for Barnegat Bay – Little Egg Harbor, New Jersey.

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Abstract

The Barnegat Bay-Little Egg Harbor (BB-LEH) estuarine system located along the eastern shoreline of Ocean County, New Jersey contains ~ 75% of New Jersey's known seagrass habitats (Lathrop et al, 2001). Eelgrass (*Zostera marina*) is the dominant species while widgeongrass (*Ruppia maritima*) is also common in lower salinity and shallow regions of the BB-LEH. Recent remote Sensing and *in situ* surveys collected prior to 2009 have indicated that seagrass habitat has contracted from historical levels (Lathrop et al 2006; Kennish et al 2007). An estuary wide survey was conducted in the summer of 2009 to measure the current extant of seagrass habitat across the BB-LEH system. To accomplish this goal aerial imagery collected in the summer of 2009 was mapped using an object oriented image analysis techniques. A technique to classify the image objects using a Cartographic and Regression Tree (CART) was compared to the manual method used by Lathrop et al. (2006) in the 2003 survey. The visual interpretation method outperformed the CART method for mapping seagrass presence vs. absence (overall accuracy of 78% automated vs. 87% manual; kappa statistic of 45% automated vs. 73% manual). The categorical values of seagrass density, when compared

to the same validation sites, had an overall accuracy of 70% and an unweighted kappa statistic of 47% using the manual classification technique.

Results of this work indicate that seagrass habitat expanded when comparing the 2003 and 2009 remote sensing surveys from 5,184 ha in 2003 to 5,253 ha in 2009. Changes in the seasonal period of image acquisition from May 4-5th in 2003 to July 7th in 2009 provide information on the location of *R. maritima* habitat, but also make it difficult to compare between the 2003 and 2009 imagery. Overall seagrass habitat expanded slightly across the estuary system except in areas adjacent to Barnegat Inlet. Future efforts to map seagrass habitat within the BB-LEH should pay particular attention to collect imagery during peak seasonal seagrass biomass, low tide, low wind conditions, and low water turbidity to maximize the spectral difference between seagrass and other benthic habitats.

Dedication

I have enjoyed the support of a many individuals in the preparation of this thesis and my graduate studies. I would like to thank to Dr. Richard Lathrop and Dr. Michael Kennish who have provided a generous amount of time, expertise, and encouragement. Gina Petruzzelli and Gregg Sakowicz for support in the collection and processing of field data. In addition, I would like to thank the Jacques Cousteau National Estuarine Research Reserve, the Barnegat Bay Partnership (BBP), and the U.S. Environmental Protection Agency (EPA) for providing research funding for this project. Last but not least I would like to thank my family, Cayleigh, MacKenzie, Dick, Onda, Mom, Dad, Elizabeth, and particularly Colleen. Thank you all so much. I could not have done it without you.

Table of Contents

Introduction

Seagrass habitat provides important ecosystem services, including essential habitat for shellfish and finfish, and sediment stabilization. The extent of seagrass habitats worldwide has been reduced through human induced habitat changes (Short and Wyllie-Echeverria 1996; Orth et al. 2010; Waycott et al. 2009). These impacts can be broken down into two categorical stressors: (1) direct impacts due to physical alteration of benthic habitat through channel dredging, inlet modification, boat scarring, dock building; (2) indirect impacts caused by nutrient enrichment and eutrophication (Burkholder et al. 2007). Because seagrass are vascular benthic autotrophs, they require clear water and high levels of benthic Photosynthetically Active Radiation (PAR). Therefore, the success of seagrass at specific locations through time provides a potential long-term integrator of water quality (Burkholder et al. 2007). As eutrophication increases through time, the dominant primary producers in shallow marine environments tends to move across a gradient from seagrass, to macroalgae, and finally to phytoplankton (Figure 1) (Wazniak et al. 2007; Deegan et al. 2002).

The Barnegat Bay-Little Egg Harbor (BB-LEH) estuary located along the eastern shoreline of Ocean County, New Jersey contains ~ 75% of New Jersey's known seagrass habitat (Lathrop et al. 2001). Recent remote sensing and *in situ* surveys have indicated that seagrass habitat has contracted from historical levels (Lathrop et al. 2006; Kennish et al. 2008). The major focus of this research project is to assess the utility of image segmentation and object-oriented classification techniques to quantify the changes in seagrass areal distribution and density. In addition, this study examines the spatial and temporal changes in seagrass in relation to other environmental processes acting on the BB-LEH system.

Research Questions

(1) The presence and absence of seagrass habitat across the BB-LEH will be quantified for the 2009 growing season using aerial photography collected to maximize (low tide, wind, and sun angle) the ability of a trained image interpreter to discern benthic habitat features. Object-oriented image analysis techniques combined with visual interpretation (as employed by Lathrop et al. 2006) will be compared with an unsupervised (CART) technique to test whether this more automated technique improves mapping accuracy and efficiency.

(2) Aerial photography obtained in 2003, 2006, 2008 and 2009, across the BB-LEH estuary system will be used to characterize variations in the spatial distribution of seagrass across both intra (seasonal) and inter annual time frames and examine whether these patterns corroborate *in situ* field data collected during the same time periods.

(3) Aerial imagery will be used to quantify the direct anthropogenic impacts of dredging (deepwater areas), boat scarring, and dock location have on the spatial distribution of seagrass habitat across the BB-LEH and assess the significance of these impacts in relation to the broader spatial patterns of seagrass loss and gain.

(4) *In situ* data collected by the New Jersey Department of Environmental Protection Bureau of Marine Water Quality (BMWQ) in the BB-LEH will be used to

quantify total nitrogen estuary wide as well as the relationship between land cover and nutrient loading.

Background

Study Area

The Barnegat Bay-Little Egg Harbor (BB-LEH) estuary is located along the eastern edge of the New Jersey coast between 39º31’N and 40º06’N latitude and 74º02’W and 74º20’W longitude (Figure 2). The estuary forms a long, narrow, and irregular tidal basin that extends north-south for nearly 70 km, separated from the Atlantic Ocean by a narrow barrier island complex (i.e., Island Beach and Long Beach Island) that is breached at the Point Pleasant Canal in the North, Barnegat Inlet at mid bay, and Little Egg Inlet at the southern extremity (Figure 2). Ranging from 2 to 6 km in width and 1 to 6 m in depth with a mean low low tide depth of 1.5 m, this lagoonal estuary has a volume of ~3.5 x 10^8 m^3 and surface area of ~280 km^2 (Kennish 2001). Water temperature ranges from -1.5-30ºC, and salinity from ~10-32 psu (Moser 1997). Characterized by semidiurnal tides with a tidal range of <0.5-1.5 m, the estuary is well-mixed. Current velocities are typically <0.5-1.5 m s^{-1}. Circulation is restricted by the extreme shallowness of the bay and the location of the barrier island complex. Winds, tides, salinity gradients, inlet size and configuration, and the geomorphology of the estuary strongly control water movement. The shallowness of the open bay and, extensive shoals and marsh islands near the inlets restrict bay flushing times to between 60 and 120 days (Gou et al. 1997).

The Barnegat Bay watershed covers an area of 1730 km^2, with more than 500 km^2 classified as developed or urban lands (Lathrop et al. 1999). The ratio of the watershed area to the estuarine surface area is ~6.5 to 1. Small coastal plain rivers, streams, and creeks drain the watershed, and most of the freshwater discharge (>80%) is base flow derived from groundwater influx. Ocean County (which shares a similar spatial boundary to the Barnegat Bay – Little Egg Harbor watershed) has experienced exponential population growth from 30,069 people in 1900 to over 569,000 in 2008 (United States Census Bureau). Associated with this increase in population has been a rapid development of upland forest habitat to urban land cover with over 13,200 acres converted between 1995 – 2006 (Lathrop and Haag 2007). Riparian areas in some subwatersheds have been heavily altered, consisting now of over 40% urban, agriculture, or barren land (Lathrop and Haag 2007). The BB-LEH estuary faces multiple resource management issues including climate change and an associated rising sea level, increased nutrient enrichment and associated eutrophication, loss of critical habitat and species, inadequate stormwater management, and diminishing fresh water availability. To understand how these stressors are impacting the BB-LEH, it is necessary to conduct comprehensive assessments of key estuarine species and habitats.

1. Background: Seagrass

Submerged Aquatic Vegetation (SAV) is used to define all benthic primary producers including both seagrass and macroalgae species. This study focuses on seagrass distribution not on the more inclusive SAV category. Seagrasses define a group of angiosperms that can be found submerged in marine coastal regions around the world. There are a total of 12 genera and 55 species of seagrass found worldwide on every continent except Antarctica (Larkum and den, Hartog 1989). Seagrass species belong to a small number of plant families and are all contained with the superorder Alismatiflorae (Kuo and den, Hartog 2006; Waycot et al. 2009). Arber (1920) developed a set of four properties that define seagrass: (1) The plant must be able to live in a saline environment; (2) be able to grow when fully submerged; (3) be able to anchor itself; and (4) be capable of hydrodrophilous pollination. According to Kou and den, Hartog (2006) the list created by Arber (1920) does not differentiate between several taxa termed the 'eurysaline' group which can tolerate a much larger range of salinity then seagrasses species. The ability of the eurysaline group to tolerate large salinity changes leads to its inability to compete with seagrass species in marine waters (den, Hartog 1970).

Seagrass meadows, beds, and patches provide a number of important ecological services to estuarine and near-shore habitats. They provide areas of wave bafflement, allow accumulation of sediments, mitigate the re-suspension of solids, and protect benthic habitats from scouring and physical disturbances. Christiansen et al. (1981) found that declining *Z. marina* habitat accounted for 65 m of seaward shoreline expansion in Kyholm, Denmark between 1993-1954 and 1961-1960, while during periods of high *Z. marina* biomass no shoreline expansion was recorded. Through the process of photosynthesis seagrass is responsible for locally high water column dissolved oxygen levels. Seagrass export oxygen to roots and rhizomes, supplying oxygen to benthic sediments (Eyre and Ferguson 2002). Seagrass provides crucial habitat for many ecologically, and commercially important macrofauna including; Tiger prawns *Penaeus semisulcatus* and P. *esculentus*, hard clams *Mercenaria mercenaria*, blue mussels *M. edulis*, and bay scallops *Argopecten irradians* (Bologna and Heck 1999; Bologna and Heck 2000; Kenyon et al. 1996).

Seagrass Responses

Seagrass communities respond uniquely to different stressors including algal blooms, physical disturbances, and diseases. The literature review below will show that changes in seagrass habitat can implicate a stressor such as eutrophication and direct physical disturbance. This discussion will examine the effect of different environmental stressors on seagrass in general, and specifically on the primary species in the BB-LEH *Z. marina.*

Eutrophication in coastal estuary systems can lead to both direct impacts to seagrass health such as ammonium toxicity and nitrate inhibition and indirect impacts such as blooms of ephemeral macroalgae, ephipytic algae, and phytoplankton, which effectively shade seagrass habitat, causing contraction into shallower water or extirpation (Burkholder at al. 2007). This has been documented by Hauxwell et al. (2001) in Waquoit Bay, Massachusetts, who recorded losses of 80-96% of total *Z. marina* in areas where nitrogen loading was ~ 30 kg TN ha^{-1} yr^{-1} and total loss in areas where TN > 60 kg ha^{-1} yr^{-1}. Hauxwell et al. (2001) concluded that seagrass loss occurs at both the seagrass bed edge and internally within seagrass beds, and this loss can be expressed as an exponential decline in shoot density and total bed area vs nitrogen loading. Deegan et al. (2002) looked specifically at the interaction between macroalgae, *Z. marina* and fish population empirical data. They found that when nutrient loading increased, so did macroalgae biomass, while seagrass biomass, fish abundance and diversity all decreased. A temporal variability scale of impacts can be used to examine the relationship of algal blooms on seagrass species. Phytoplankton shade seagrass species for varying lengths of time depending on the local hydrodynamic cycle (flushing rate, wind, rain, tidal regimes, and bay bathymetry) and the severity of eutrophication, while benthic macroalgae species tend to be more spatial persistent but still can be impacted by the local hydrodynamic cycle. Epiphytic algae that directly attach to seagrass shoots and blades do not fluctuate on a daily or weekly cycle like macro and phytoplankton algae, and therefore can cause disturbances over longer temporal time frames. Wazniak et al. (2007) have proposed a conceptual model concerning shallow estuarine systems undergoing eutrophication that is applicable to BB-LEH. This model suggests that as nutrient loading increases in coastal estuary systems, primary production moves progressively up the water column favoring species with higher surface to volume ratios. This shift from benthic to pelagic production can occur over temporal and spatial scales within an estuarine system changing inter-year and intra-year and over a spatial eutrophication gradient.

Epiphytic algae can impact seagrass health by attenuating light and by blocking the diffusion of the limiting gas carbon over the leaf surface (Neckles et al. 1993; Beer and Koch 1996). Drake et al. (2003) showed that seagrass (*Z. marina and Thalassia testudinum*) growing in oligotrophic waters had 36% of the incident light in the peak chlorophyll absorption bands attenuated by epiphytic algae, while seagrass growing in areas of higher nutrient enrichment (Monterey Bay, CA) had ~ 60% of the total incident light absorbed. Epiphytic growth on macrophytes can be extensive, representing between 22% and 61% of the total primary productivity within *Z. marina* beds (Hemminga and Duarte 2000). Epiphytic ash-free-dry weight vs light attenuation across Photosynthetically Active Radiation (PAR 400-700 nanometers) followed a negative exponential function with higher coefficients at lower wavelengths. Epiphytes have been shown in microcosm studies to be controlled by the density of grazers (isopods, amphipods and gastropods), and independently by the amount of available nutrients, lifespan of the substrate (if biological), and water clarity. Both decreasing grazers and increasing nutrients increased the amount of epiphytes in the epiphytic matrix, and *Z. marina* only suffered a decrease in productivity with both treatments acting concurrently (Neckles et al. 1993; Johnson et al 2005). There is some evidence that episodic benthic macroalgae and pelagic algae blooms of short duration can be beneficial to seagrass habitat by shading autotrophic epiphytes and not allowing epiflora to establish on seagrass blades.

Persistent benthic macroalgae blooms in eutrophic estuaries out compete seagrass beds, becoming the dominant benthic macrophyte. Blooms are controlled primarily through light limitation (seasonal and biological), nutrient levels, temperature, grazers, and hydrodynamic cycles. Macroalgae have been shown to survive on 0.12% of incident water surface light while seagrass requires 11% of incident light (Duarte et al.1991; Sand Jensen 1992). Seagrass growth, is generally light limited while macroalgae growth is generally nutrient limited. Nitrogen has been shown to positively influence yearly macroalgae biomass in mesocosm studies, while phosphorus has been implicated as the limiting nutrient during intra-year time periods for specific macroalgae species in temperate areas. In tropical zones carbonate's ability to adsorb phosphate could create phosphorous limited areas (Valiela et al. 1997; Hauxwell et al. 2003; Lapointe et al. 1992). Nutrient uptake rate and half saturation constants in macro and microalgae are positively correlated with surface to volume ratios. Macroalgae have the ability to change form, increasing and decreasing surface area to compensate for nutrient availability (Hein et al. 1995). Macro algae blooms have been recorded with weights over 0.5 kg m^2 and with canopy heights in excess of 0.5 m. (McGlathery 2001). Macroalgae blooms can be persistent with blooms in specific estuaries recording for years or even decades (Gordon and McComb 1989; Valiela et al 1992). Macroalgae blooms impact seagrass habitat primarily through light attenuation especially in temperate regions (Dennison and Alberte, 1982; Deegan et al. 2002). Macroalgae are efficient at intercepting regenerated nutrients from benthic sediments, de-coupling benthic and pelagic biogeochemical sedimentary cycles, and contributing to the formation of hypoxia and anoxia (Valiela et al. 1992). Dense assemblages of benthic and pelagic macroalgae intermingled within seagrass habitat often serve as an indication of eutrophication since macroalgae have been show to out compete seagrass in eutrophic conditions.

Phytoplankton blooms, when present in sufficient densities, can effectively shade out benthic primary production. In shallow water regions (less then 10 m depth) total light attenuation is dominated by phytoplankton (Lorenzen 1972). Durand and Olsen. (1998) modeled the influence of various phytoplankton species on total light attenuation (scattering plus absorption). They determined that phytoplankton size (picophytoplankton < 1 µm, ultraphytoplankton 1-2 µm, and nanophytoplankton 2-20 µm), total water column biomass, as well as distribution and optical properties (accessory pigments and shape) determined total attenuation. In a mesocosm study undertaken by Short et al. (1995) *Z. marina* was exposed to different light levels (11%, 21%, 41%, 61%, and 95% of surface light) simulating the effects of increased water column light attenuation. Shoot density had a logarithmic positive response to increasing light levels. Leaf length had the opposite response with the longest average length in the lowest available light conditions. Standing biomass was positively correlated with light intensity, showing that shoot density overwhelmed shoot length for total biomass measures.

Seagrass meadows and beds can be affected by direct physical disturbances such as storms, herbivory, and anthropogenic impacts (boats scarring, dredging, and dock shading). Short and Wyllie-Echeverria (1996) found that between 1970 and 1982, 50% of the total seagrass loss worldwide could be attributed to natural direct disturbances and 50% to anthropogenic disturbances. They further found that this trend changed between 1983 and 1994 and that an increase of 75% of seagrass loss could be mostly attributed to anthropogenic-induced water clarity changes resulting in a loss of 93,000 ha of seagrass worldwide. Understanding the influence of natural impacts and the spatial patterns they create can help to differentiate the causative factors for seagrass decline. Storms can cause extensive damage to seagrass habitat by decreasing water clarity, covering seagrass beds with sediments, scouring (removing belowground root and seed stock), changing bathymetric conditions, altering current and tidal regimes, and removing organic sediments. Fonseca et al. (2002) and Fonseca and Bell (1998) found a statistical correlation between seagrass (*Z. marina* and *Halodule wrightii*) percent cover, ratio of bed perimeter to area, sediment organic content, and percent sand, silt, and clay vs. tidal current speed, exposure to waves (fetch and predominate wind direction known as REI or relative exposure index) and water depth. A tipping point for seagrass was found when current speeds of ~25cm s^{-1} and an REI of ~3x$10^{\wedge 6}$ were spatially calculated, which is near the motion current speed for local suspended sand. They hypothesize that, at these current velocities and REI levels, seagrass beds break up into discrete patches, increasing surface to volume area and the damage caused by physical disturbance. Seagrass beds may also be influenced by stochastic events (e.g. hurricanes, earthquakes, tornadoes, etc.); the relative importance of background conditions vs intense events is unknown. Dieback of *Z. marina* beds in Chesapeake Bay, has been implicated in some instances to an increase in water temperature with an associated increase in *Z. marina* respiration (Moore et al. 1996). There is some evidence that intense events radically change seagrass habitat, which is then exposed to chronic conditions. Bell (1998) observed this in Tampa Bay, which experienced large changes in seagrass percent cover after large storms (Bell unpublished in Fonseca and Bell 1998).

In summary a number of stressors can impact the spatial distribution of seagrass habitat across a variety of spatial and temporal scales. The unique impact that these different stressors leave on seagrass habitat over time can help to define the magnitude of the primary stressors on specific estuarine benthic habitats. Therefore, studying the change in seagrass habitat over time can lead to better management strategies to protect the biological integrity of these important near-shore marine environments.

Seagrass Monitoring: Overview

Assessment of seagrass habitat worldwide has become an important tool to monitor the health of near-shore marine environments. Measurements of seagrass stock, areal coverage, biomass, health, morphology, and nutrient content, involve techniques from the macro-scale such as remote sensing to the micro-scale of elemental analyzers. New information technologies such as Geographic Information Systems (GIS) and spatial and temporal models allow scientists to incorporate secondary datasets (phytoplankton bloom, water transparency, depth, wind direction, fetch, storm events, nutrient levels, hydrodynamic cycles, etc.) over a variety of temporal and spatial scales to the analysis of seagrass habitat change

Importance of Seagrass Monitoring

Seagrass originally evolved as a terrestrial biome, gradually acclimating to shallow marine environments. Seagrass species therefore have similar organs and tissues as terrestrial flowering plants (Kuo and den, Hartog 2006). Found on every continent except Antarctica seagrass species occur in estuarine and near-shore ocean systems. The study of the land-sea margin interface is key to understanding the effect of multiple resource management issues including climate change, fisheries stock management, land use / urbanization, coastal erosion, and eutrophication. Coastal zones (10 m above mean low tide) account for 2% of total land area worldwide but have nearly 10% of the worlds population and 13% of the worlds urban areas (McGranahan et al. 2007). These areas pose particular challenges for resource managers struggling with the long-term stability of coastal infrastructure due to the dynamic nature of near-shore coastal areas and the future predictions of climate change impacts.

Seagrasses are benthic-rooted vascular plants that grow in large dense communities, creating hierarchical spatial structures varying in scale from centimeters to kilometers in length. Areas of contiguous seagrass coverage of varying percent coverage over large areas are known as seagrass meadows. Seagrass beds denote contiguous areas of consistent seagrass coverage, while patch areas internal to both beds and meadows denote discrete areas of seagrass, macroalgae, or open benthic environment (Robbins and Bell 1994; Lathrop et al. 2006). Expansion of seagrass meadows, beds, and patches reflect the success of individual plants in specific geographic locations. Harwell and Orth (2002) sampled 105 *Z. marina* beds in 12 zones within the lower and middle portions of Chesapeake Bay for *Z. marina* seed viability. They found that 11 out of the 12 zones had viable *Z. marina* seeds, which indicates that *Z. marina* seedstock was well distributed across the estuary even in locations where *Z. marina* was not currently present. Because individual seagrass plants are sessile, success or failure can be generally attributed to local environmental factors, making them potential bioindicators of estuary health. Seagrass is considered a long-term integrator of estuarine condition unlike *in situ* nutrient parameters which are rapidly assimilated into plant tissue or sediment, or exchanged into coastal ocean waters (Burkholder et al. 2007). In addition, other macroflora and macrofauna species within estuarine environments are often highly mobile both temporally and spatially, necessitating a more rigorous sampling and statistical analysis to document change in density and health.

Seagrass extent and health is a worldwide indicator of estuarine and shallow marine habitat health because:

1) Seagrass is a rooted vascular plant.

2) Seagrass species have specific habitat requirements.
3) Seagrass has a worldwide distribution in a rapidly changing near shore coastal environment.
4) Seagrass is an ecologically important species altering physical-geochemical processes, creating habitat, and providing forage for other flora and fauna.

Historical information on seagrass abundance within the BB-LEH estuary system comes from early baymen. Some believed seagrass was detrimental to shellfish growth, settlement and ability to harvest (Ridgeway 2001). Seagrass was believed to be an impediment to oyster larvae settlement, and was therefore cut using an underwater submarine prior to oyster seeding (Mountford 2002). In addition, seagrass was considered a nuisance by early baymen as it confounded attempts at harvesting hard clams with tongs (Ridgway 2001). Lathrop et al. (2001) summarized the known comprehensive seagrass habitat mapping efforts for the BB-LEH. These included maps created in the following years:

1) 1968, by the US Army Corps of Engineers (survey technique unknown but most likely boat based).
2) 1979, Earth Satellite Corporation produced a 1:24,000 scale map from panchromatic aerial photography flown in June-August (Macomber & Allen 1979).
3) 1985-1987, the NJDEP Joseph et al. (1992) collected data on .4 km grids for a shellfish survey that noted presence or absence of seagrass species,

4) 1996, 1997, 1998, and 1999 Boat based surveys. (McLain and McHale 1996; Bologna et al. 2000).

In 2003 Lathrop et al. (2006) mapped seagrass habitat using 1 m spatial resolution, multi-spectral (Infrared, Red, Green, and Blue) aerial photography collected on May 4-5. Their mapping effort showed 893 ha less seagrass then the previous mapping work collected between 1996 and 1999 by McLain and McHale (1996) and Bologna et al. (2000). However because their methods were different (air photography vs *in situ* boat-based) no definitive statement could be made on seagrass habitat loss or gain. The current study therefore provides the first consistent mapping methodology with the 2003 study.

Several researchers have looked at *in situ* seagrass health and abundance within the BB-LEH. In the most extensive project Kennish et al. (2007, 2008) examined seagrasss distribution from 2004-2006, 2008 and 2009 sampling at 120 fixed stations across for major seagrass beds. Kennish found that between 2004 and 2006 in Little Egg Harbor the mean seagrass aboveground biomass declined over 87.7% percent from 59.62 to 7.31 g $m^{\wedge -2}$ dry weight. This trend, albeit with a smaller percentage loss was also recorded between 2005 and 2006 at Barnegat Inlet with a loss from 32.04 to 16.03 g m^$^{-2}$ g $m^{\wedge -2}$ dry weight. A decrease in aboveground biomass was also recorded estuary wide at all sampling locations and it represents the lowest values recorded in this estuary (Kennish et al. 2008). This episodic loss of seagrass habitat recorded between 2004 and 2006 is consistent with data collected by Bologna et al. (2001) who observed a large scale dieback of seagrass in Little Egg Harbor associated with blooms of several macroalgae species.

Gastrich et al. (2004) and Pecchioli et al. (2006) analyzed blooms of the phytoplankton species *Aureococcus anophagefferens* (brown tide) and showed that bloom densities were highest in Little Egg Harbor (Figure 2; Figure 3). The blooms were significantly different between years. High blooms were associated with warmer water temperatures, high salinity values, and low stream water flow. These results are most likely not physiological causative factors since *A. anophagefferens* blooms were also recorded in winter months. They found that *A. anophagefferens* blooms decreased Secchi disk values, indicting a reduction in solar energy for benthic fauna.

These results demonstrate that primary productivity in the BB-LEH estuary can undergo a sequence of dominant plant forms from benthic vascular plants to macroalgae (Bologna et al. 2006; Kennish et al. 2007), and finally to phytoplankton (Gastrich et al. 2004); Pecchioli et al. 2006). These changes can be seen as a gradient of eutrophication impacts (Wazniak et al. 2007). Moving along the gradient can take place very quickly through the multiplier effect and impacts of positive and negative feed back-loops (Burkholder et al. 2007).

3. Methodology

Current (2009) extant of seagrass habitat across the BB-LEH system (research question 1)

The first objective of this project was to quantify the location of seagrass across the BB-LEH estuary system for the 2009 growing season. To accomplish this an aerial photography dataset was collected, processed into image objects, and classified to create a GIS dataset showing the location of seagrass habitat across the BB-LEH. The following methods sections describe the steps used to create the output GIS dataset. In addition an accuracy assessment was undertaken to determine how well this GIS dataset maps seagrass across the BB-LEH.

a). *Aerial Photography Collection*

An aerial photography mission was undertaking during the summer of 2009. Film aerial photography was collected on June 28, July 14, and August 4, 2009 using a Navajo HS airplane equipped with a Leicca RC30 camera, lens # 13234, focal length 152.720 mm, and a variable exposure time of 260-420 milli-seconds. Two types of film were used; a grey scale AGFA 80 and color film AGFA 100. The same plane and camera was used for all three imaging missions. The plane flew at an altitude of 3,658 m and speed of 180 km hr^{-1} per hour. The plane flew three survey lines, two in the southern estuary due to bay width and one in the northern estuary for both the June 28 and July 14 aerial flyover, the August 4 date was only flown to collect imagery in the northern part of the study area. Two passes were made per day, the first to collect black and white photography and the second to collect color photography. The resultant film was then processed and scanned through a high resolution scanner resulting in a digital image with 18,278 by 18,292 pixels in a scale of 1 to 2,000. These scans were then ortho-rectified and projected into the Universal Transverse Mercator, North American Datum 1983 zone 18 north in meters. The resulting geo tiffs were mosaiced into 15 larger areas to ease the image processing procedure (Figure 2).

b). *In situ* data collection

A number of *in situ* sites were visited to collect reference information to enable the interpretation of the aerial photography (Appendix I and Figure 4). Reference sites were selected to match a subset of the *in situ* references sites selected during the 2003 Lathrop study (Lathrop et al. 2006). Reference sites were not selected in a random probabilistic manner, but rather targeted transects across the study area n = 167. In addition, 15 sample sites were selected for a late season review (October of 2009) for areas of uncertainty in the imagery. An additional 120 sample points were collected in June 2009 as part of an ongoing research project (Kennish unpublished data). These data points were also included in the study as field reference sites, although their collection used a different technique than the data points used in this study. A second *in situ* n = 124 dataset was collected to provide a validation dataset which was selected using a stratified random sampling design to focus on shallow water habitats mimicking the depth distribution of seagrass within the BB-LEH estuary (Figure 4; Figure 5). These points were distributed to match the depth distribution on the 2003 seagrass survey. To accomplish this, 2003 seagrass presence absence data from (Lathrop et al 2006) was intersected with the NOAA Nautical Charts Depth information (Charts 12324: edition 25, 1990 and 12316: edition 25, 1992 from Lathrop et al. (2001). Figure 5 shows the depth distribution of the 2003 seagrass remote sensing survey. For each 0.3048 meter depth (1 foot) category a number of field sites were randomly chosen to match the percentage of area of all seagrass habitat at that depth. This matched the random seagrass sites depth histogram to the depth histogram of the presence / absence seagrass data from 2003. In total of 150 (some points were lost since they were located in areas not able to be reached by boat or outside image acquisition area) located in Barnegat bay as a validation. These points were distributed to match the probability of finding seagrass at a specific depth. This validation dataset was not used in the image mapping and classification process but kept as an independent data set to compare with the wall-to-wall GIS map to create an error matrix, a producer's and user's accuracy assessment, and a Kappa statistic. As a secondary step after the accuracy assessment was completed the validation dataset was used to clean up the final GIS dataset.

For all of the *in situ* data collected for this project (the reference dataset n = 167 and the validation dataset n=124), field collection was accomplished as follows. The field survey was conducted from the Rutgers University Marine Field Station (RUMFS) using a 20 foot maritime skiff. Navigation to field locations was accomplished with a Garmin 530s marine GPS/Sonar system. Upon arrival at the preselected field locations, the boat weighed anchor. Next, an L shaped 4 meter x 5 meter grid made of 1.905 cm pvc (Figure 6) was lowered over the side of the boat. A diver entered the water and affixed a GPS Magellan Mobile Mapper 6 (2-5 meter horizontal accuracy) to the outside L of the survey grid (marked in Figure 6). A compass reading was taken along the left-hand axis of the sampling grid. The compass reading and the GPS position allowed precise placement of the sampling grid on the benthos to a higher level of accuracy than the boat-based GPS unit. The diver then visited grid 1 through 8 and recorded information on SAV presence / absence (yes no), percent cover of seagrass species (*R. maritima* and *Z. marina*) (0 to 100 in 10% increments), and percent coverage macroalgae (0 to 100 in 10% increments). This data was verbally relayed to the boat captain who recorded the data on write-in–the-rain paper. Upon completion of field data collection, the GPS unit was removed and the sampling grid returned to the boat. Field sheets were then signed, dated, and entered into a digital database. The precise location of each sampling grid was determined using Matlab™ and simple geometry using the GPS location in UTM Coordinates and the compass bearing. A correction for magnetic declination (difference between the North Pole and the magnetic North Pole) was calculated using NOAA website (http://www.ngdc.noaa.gov/geomagmodels/Declination.jsp for July 15th, 2009, 39.9745 N 74.1514 W magnetic declination equals 12 degrees and 47 minutes.

c.) *Creation of Image Objects*

An important step in image classification is the clumping of similar pixels into image objects for classification. To accomplish that task, each image collected in 2009 was filtered using the aggregate command available in Arc Grid™ for a 2x2 grid window selecting the median cell value. This was done to remove areas of local light scatter from wave tops, Langmuir circulation lines, and to reduce the size of the imagery for processing. The median was selected over the mean to avoid skewing from light scattering which can cause areas of high image reflectance (white capping) and shadows. Figure 7 shows the difference in image resolution between the 0.5 m and 1 m down sampled digital imagery. A noticeable reduction in the outlier points and influence of Langmuir circulation lines is evident.

The rectified mosaicked color photography was then imported into Ecognition™ to support image segmentation and classification. Ecognition™ is an image analysis software package that segments raster data in an unsupervised method minimizing the intra-polygon (image object) variance while maximizing inter-polygon (image object) variance. The user can control the weight of each imagery band by changing a coefficient between 0 and 1 (0 no input for that band 1 full input) by band and a unit less scale parameter which determines the average image object area. As the scale parameter increases greater spectral heterogeneity is allowed increasing the average size of the image objects. Multiple scale image objects can be created by running a multiple resolution segmentation procedure. Two-scale parameters were used for each image mosaic layer 1); a small scale parameter between 10-15; 2) a large-scale parameter 50-70. The smaller scale parameter resulted in image objects with a mean size of .073 ha, mode of .045 ha, 25 percentile of .02 ha, and the 75 percentile at .09. This scale parameter was selected to meet the target minimum mapping unit of .05 ha (500 m$^{\wedge 2}$). The minimum mapping unit defines the smallest feature delineated in the map or the amount of detail a map contains. The band coefficients used were 1 for blue, 0.7 for green, and 0.5 for red. The coefficients were selected by trial and error by the operator to maximize the difference between seagrass and other benthic habitats. Figure 8 shows the different sizes of image objects created by varying the scale parameter.

d). *Image object classification*

A manual classification where each image object was visually interpreted and assigned to one of four classes of seagrass density (high 100-80% percent cover, medium < 80%-40% cover, sparse > 40% and <= 10% cover, and no seagrass <10% - 0%). The field reference data was used to inform the interpretation. The larger scale image objects (scale parameter 50-70) were first manually classified using Ecognition™. The large image object classifications were then forced down into the smaller image objects (scale parameter of 10-15) based on the nested polygon structure. Smaller image objects on edge areas and internal to the larger image objects were then manually reclassified when necessary. This method sped up the manual classification effort allowing large contiguous areas of seagrass to be classified quickly while also allowing precise classification on seagrass edge and gap areas (Figure 9 from Lathrop et al 2006). The reference data also contained information on seagrass species and macroalgae percent cover these categories were not mapped as part of the manual classification. To create the final GIS dataset and accuracy assessment dataset the finer-scale image objects were exported to Environmental Research Institute ESRI™ shapefile format.

A secondary automated classification procedure was conducted to compare the accuracy of the manual classification versus the automated attempts. The automated classification attempt was made using a Cartographic and Regression Tree (CART) model. A CART model or decision tree predicts the dependent variable based on the values of multiple independent variables.

CART models are a form of binary recursive partitioning (Lewis 2000). They classify observations by iteratively applying a binary split at each node along a decision tree. Each binary split involves the independent variable that ‘best’ partitions the feature space between the dependant variable. Dependant variables within a CART model can be either categorical or continuous (Lawrence and Wright 2001). CART models like other categorical classification techniques will, if unchecked, over fit the predicted classification so that it matches exactly the input classification. Therefore it is necessary to prune the CART model back to a level where it can be expected to be robust (Lawrence and Wright 2001). A cross validation technique was used to prune this CART model to the appropriate node on the decision tree by randomly removing without replacement 50% of the dataset, running the CART model on the remaining data, and comparing which node on the decision tree provided the highest overall kappa coefficient vs the left out data. The initial dataset for the CART model was created by taking the *in situ* validation and training dataset mean values by spatial location and cross referencing them with the image objects created in Ecognition™. The image objects created in Ecognition™ contained information on mean values and standard deviation for each color band in the aerial photography (red, green, blue) as well as standard deviation for water depth. The CART model was run with the dependant variable seagrass absence vs presence. This model was repeated 10,000 times selecting a new cross validation dataset randomly for each run.

e). *Accuracy Assessment Manual and CART model*

To determine how well the image objects from both from the manual and the CART model described seagrass habitat across the BB-LEH an accuracy assessment was undertaken. To accomplish this the classified image objects were compared to the validation dataset within a GIS to create an accuracy assessment matrix, error of omission and commission, overall accuracy assessment, and a Kappa coefficient. This is similar to the methods employed by Lathrop et al. (2006).

Kappa Statistic

(Observed agreement - Chance agreement)/(1 - Chance agreement) (1)

The Kappa coefficient is a measure of agreement between two categorical datasets correcting for the random chance that categories will agree. These measures of accuracy were completed to determine how accurately seagrass vs. all other habitats were mapped, and to determine how well the maps reflected the density of seagrass habitat based on the *in situ* data.

For the CART model this resulted in a distribution for the output Kappa statistic and total accuracy assessment for each time the model was run. This Kappa distribution from the automated classification (CART model) seagrass presence absence was compared to the singular result obtained in the manual accuracy assessment.

Change Detection (Question 2)

To quantify how seagrass habitat has changed through time both annually and seasonally in the BB-LEH estuary system a change detection analysis was done. To complete this analysis a number of historical imagery datasets from 2003-2008 were collected and compared to the 2009 imagery and classified image objects collected as part of objective 1.

a). *Historical imagery datasets*

Geo-referenced imagery has been collected for a variety of projects across the entire study area in 2003, 2004, 2006, 2007, 2008, and 2009 (table 1). Two of the imagery datasets (2003 and 2009) were collected specifically to map benthic habitat environments. The other imagery datasets were not collected with the expressed goal of mapping benthic habitat and thereby took no special considerations to collect imagery during periods of low tide, sun angle, and or wind conditions. A visual assessment of the 2004, 2006, 2007, and 2008 imagery datasets was conducted to determine the feasibility of using these image collections to map the change in the spatial distribution of seagrass habitat through time. It was determined that most of the BB-LEH imagery datasets did not provide a consistent view of the benthic environment. Therefore, instead of using this imagery to create wall-to-wall seagrass maps, the imagery was used where image quality was appropriate to verify *in situ* observations and to characterize the seagrass reflectance across the growing season.

b). *Comparing the 2003 and 2009 image datasets*

In 2003, Lathrop et al. (2006) collected digital aerial photography across the entire LEH-BB estuarine system. This photography was collected on May 4 and 5, 2003 in both the early morning and late afternoon to minimize specular reflectance from the water surface. This digital aerial photography was processed using similar object oriented methods as the 2003 imagery (Figure 9 from Lathrop et al. 2006). The resulting classified vector data from the 2003 and 2009 classified seagrass habitat maps were manually compared using a GIS system to ascribe a reason for areas mapped as a change (presence / absence) in seagrass habitat. Each image object was given a categorical reason why the imagery classification had changed based on the assessment of a trained image interpreter. Where possible *in situ* data collected from the 2003 (Lathrop et al. 2006) and 2009 seagrass survey were used to lend support to the manual classification. The categories with definitions are listed below;

1) 'Change in Season'. A seagrass bed that existed in 2003 that had not reached peak biomass due to the spring imagery collection in 2003 but that was clearly

visible in the 2009 imagery. This classification mostly applies to *R. maritima* dominant seagrass beds in the northern portion of the BB-LEH.

2) 'Poor image quality'. When one or both of the source imagery dataset had poor image quality which did not allow the classifier to view the benthic habitat.
3) 'Misclassification'. Used when on further review of the source imagery it is believed an error of commission occurred.
4) 'Seagrass habitat gain'. Used when new seagrass habitat is found that did not exist in 2003.
5) 'Seagrass habitat loss'. Used when seagrass habitat is lost between the 2003 and 2009 imagery.

Change in seagrass habitat was further analyzed using five bay segments (Figure 10): (1) Little Egg Harbor (LEH) south of the Rt. 72 Bridge to the Tuckerton Peninsula; (2) southern Barnegat Bay (SBB) north of Little Egg Harbor to Barnegat Inlet; (3) Barnegat Inlet (BI) the area within 5 km of Barnegat Inlet; (4) central Barnegat Bay (CBB) from Barnegat Inlet to the Route 37 Bridge; (5) and northern Barnegat Bay (NBB) north of the Route 37 Bridge. For each of these segments the mapped seagrass in 2003 and 2009 was compared with the identified reason above for mismatch.

c). Deepwater Bed-edge Change Detection

An analysis was done to quantify changes in the deepwater edge of seagrass habitat by comparing the May 2003 and July 2009 imagery at a large scale (fine level of detail). This analysis was not done using the image objects but rather below the minimum mapping unit created by the 2003 and 2009 image interpretation. Where visible, the sloping deepwater edge (western edge of large contiguous seagrass beds) was manually classified into areas of bed expansion into deeper water, and contraction from deeper into shallow water. Bed expansion and contraction can occur at the scale of several meters and therefore was not always observed in the original vector change detection. Areas of change were only delineated in areas that the trained image interpreter felt confident that bed expansion or contraction had occurred, this corresponded to areas with similar features located in the field of view so that the images could be visually collocated. The eastern edge of the large contiguous seagrass beds was not analyzed because they either shallow out on intertidal mud flats (Island Beach State Park), or abruptly change to deeper depths when they are located adjacent to dredged sites.

d) . Seasonal change in seagrass occurrence

An attempt was made to quantify the change in the reflectance of seagrass habitat across the growing season by combing information from multiple imagery datasets (Table 1). Kennish et al. (2007) has shown that seagrass biomass changes throughout the growing season, with *Z. marina* having peak biomass in the June – July time period and reduced biomass from August - November. To determine if seasonal changes in seagrass coverage could be quantified using remotely sensed data, the digital numbers from several imagery missions (Table 1) were extracted from know seagrass habitat. Imagery comparisons through time to detect changes in landscape features need to be normalized to correct for differences in atmospheric absorption, scattering, bit depth, band width, sensor type, anisotropic effects, sensor calibration, processing procedures (rectification), and camera height (Yang and Lo 1998). The first step in this image normalization process was to subset a working window and a working cell size. The working window was selected adjacent to Seaside Park at the northern end of Island Beach State Park. This area was chosen because: (1) there are several imagery datasets available; (2) Kennish et al. (2007, 2008) has 30 fields sites located in this area, with data on seagrass percent cover, biomass and density from 2005, 2006, and 2008. The working cell size was the largest of the available imagery (1 m horizontal) for the U. S. Department of Agriculture 2006 National Aerial Photography Program (NAPP) aerial photography. The other imagery was resampled to match this resolution using a nearest neighbor method to preserve the original digital numbers and projected to the Universal Transverse Mercator zone 18 North America Datum of 1983 meters projection system when necessary. Images were then corrected using a radiometric correction procedure. According to Lo and Yang (1998) two different image normalization approaches are possible: (1) *in situ* ground measurements to correct for both atmospheric differences and sensor calibration; (2) Relative Radiometric Normalization where ground targets are chosen post hoc and used to compare and statistically correct two image pairs. This method results in two image pairs with similar digital number but does not remove the effects of atmospheric and sensor calibration. It simply normalizes them to one image. For this study we, used method 2 since no *in situ* data are available to normalize the imagery as required for method 1. In addition, the 2007 New Jersey Department of Environmental Protection aerial photography provides a high quality imagery dataset to normalize the other imagery, and therefore was used as the base imagery to normalize all others. To normalize the imagery a collection of Pseudo Invariant Features (PIF) was selected; in this case beach areas were selected as light targets and underwater sand flats as dark targets. The underwater sand flats were chosen to normalize for differences in water height (tidal cycle), and water clarity between image acquisition dates. The photo pairs were compared by selected pixels within the identified PIF and plotted in x y feature space. A best-fit linear regression equation was run between the two coordinate pair creating a first order-slope and an intercept using the matrix equation below. The dependent variable (y coordinate) corresponded to the non 2007 imagery, and the independent variable (x coordinate) was the 2007 imagery.

$$\mathbf{[M1\ M0] = (X^{t}X)^{-1} * X^{t}Y \quad (2)}$$

A new value for the Y coordinate was then recomputed using the difference between the linear least sum of squares line computed above and a line with a slope of 1 and a zero intercept (equation 2). The intercept and linear coefficient of the least squares line therefore represent the scaling factor between the 2007 imagery and all others. Figure 11 shows a graphics example of this rescaling process.

Equation 3: Yhat = (Y-M0) * (1/M1) (3)

Areas of mapped seagrass habitat were then used to extract the normalized digital numbers for each image and compared to determine if seasonal changes in seagrass biomass could be quantified using the image normalization procedures listed above.

Direct impacts to seagrass habitat (research question 3)

The aerial photography datasets compiled to address research question 2 were also analyzed to quantify the direct impact of boat scarring, dredging (anthropogenic created deepwater locations), and dock location on seagrass habitat. In addition a band ratio method to differentiate deepwater benthic habitats from shallow dark habitats was undertaken. This deepwater mask was used to inform the manual classification of image objects (research question 1) and to map dredged areas (research question 3).

a). *Impacts of boat scarring*

To identify areas of boat scarring the 2009 high-resolution (0.5 m horizontal) aerial photography was manually examined for linear (Figure 12; Figure 13) marks and scars. Particular attention was paid to areas adjacent to the Intracostal Waterway (ICW) and local channels by layering them in a GIS on top of the 2009 aerial photography. Areas that showed signs of boat scarring were delineated using a linear GIS file. The length in meters of identified boat scars was then totaled. It should be noted that many boat scars will be under the detectable limit for the 2009 imagery based on their small width. The 2009 aerial photography was collected at a 0.5m cell size. Because boat scars are long irregular features it was felt that they could be found below the imagery's 0.5 m cell size. This has not been shown empirically for this study. In addition, when seagrass habitat is located within areas of low sediment albedo or dense macroalgae assemblages boat scars may not appear to be different spectrally from seagrass habitat. Imagery collected in the northern part of the BB estuary had Langmuir circulation lines which closely resemble the spectral pattern of boat scars (Figure 7). These lines were different from boat scars in that they all were aligned in the same direction with equal spacing and thus could be confidently distinguished.

b). *Impact of docks*

The impact of boat docks on seagrass habitat in Barnegat Bay and Little Egg Harbor was mapped using the 2009 aerial photography. Docks on tidal creeks and within lagoonal developments were excluded because they would not naturally shade seagrass habitat. For docks that had boats present, the boat itself was included in the outline of the dock. The 2009 aerial photography had a cell size of 0.5 m and therefore few if any docks are below the minimum detectable limit. Boats located at the dock in July would likely be present all summer long and therefore would shade out any local seagrass habitat. Docks were then buffered by 2 m to represent the total area of shading as the sun angle changes throughout the day. As a secondary analysis, the area of docks within 100 m of mapped seagrass habitat was quantified. These docks are viewed as being adjacent to mapped seagrass habitat and therefore could represent a reduction in seagrass habitat through direct shading.

c). *Deepwater dredged areas*

Areas of deep water created through dredging adjacent to boat docks or part of boat channels (ICW) and local channels to and from the ICW) were mapped using the 2003, 2007, and 2009 photography where visible. The depth data created through the ratio methods (see methods section below) was also used to delineate areas of deep water vs low albedo shallow habitats. Figure 14 shows an area where the bathymetric information (band ratio) provided a clearer picture than the unprocessed aerial photography. All of these image datasets were used to create a vector file showing areas of dredged deepwater location on the eastern side of the BB-LEH estuary system (Figure 15).

d). *Bay-wide depth information.*

A method to differentiate areas of deep water and shallow dark (low albedo) benthic habitats was attempted to decrease errors of commission in the manual seagrass classification and errors of omission in the GIS layer of deepwater areas. As a first step a baywide bathymetric layer was created using the Lyzenga method of (1978). This method relies on the assumption that light attenuation is an exponential function vs. water depth. For a single band, the calculated depth using an exponential function will rely on both the water column inherent optical properties and the benthic albedo. By using two bands, a correction for albedo can be empirically derived using five empirical coefficients (Lyznga and Stumpf 2003).

$$\mathbf{Z = a0 + aiXi + ajXj \quad (4)}$$

$$\mathbf{Xi = ln[Rw(Yi) – Rinf(Yi)] \quad (5)}$$

The Lyzenga method was unsuccessful for this study area and did not result in real numbers since the albedo of shallow dark habitats is lower then some deepwater areas. This resulted in a natural log of a negative number (complex number). This is similar to the result that Stumpf et al. (2003) encountered when applying the Lyzenga (1978) method to areas of low albedo in shallow water (seagrass beds and macroalgae).

A second approach using the Stumpf et al. (2003) method of using a band ratio was applied to the data.

$$Z = m1 \ln(n\, Rw(Yi)) / \ln(n\, Rw(Yj)) – m0 \quad (6)$$

Like the Lyzenga method, this model compares the reflected values R(Yi) and R(Yj) using a log function. Instead of subtracting out areas of deep water, this method divides them and then empirically derives M1 and M0. M1 and M0 were derived by using a least squares liner regression model comparing NOAA Nautical charts to mean low low tide to train the linear regression model. M1 was the coefficient of the linear regression model, while M0 was the intercept. N was arbitrarily set to 500 to make sure the natural log solution of equation 6 was not a complex number. Stumpf et al. (2003) found that varying N from 500-1500 had no effect on the predicted depth.

$$[M1\ M0] = (X^t X)^{-1} * X^t Y \quad (7)$$

$$X = [1\ \ln(n\, Rw(Yi)) / \ln(n\, Rw(Yj))]$$

Y = Depth in Mean Low Low tide NOAA nautical Charts

The point location for each NOAA depth was buffered 15 m and converted to a raster format. These depths were then intersected with the Digital Numbers for the 2007 Aerial Photography and applied to equation 4. Theoretically, any band combination could be used (6 total combinations with 4 bands) (Figure 16). It was determined through visual interpretation that the green vs. red band ratio provided the best of all band ratios to calculate bay depth. For all 311 NJ Department of Environmental Protection aerial photographs that intersect the BB-LEH, this band ratio was run. Figure 14 shows the final result for one study area. The bay depth data was used, to locate areas of deep water (dredging), and in the manual classification of the 2009 aerial photography to delineate seagrass habitat.

Watershed development and Nitrogen loading (research question 4)

The N.J. Department of Environmental Protection (NJDEP) Bureau of Marine Water Quality (BMWQ) has an ongoing coastal ocean water quality monitoring program. This program is designed to monitor and assess water quality standards for New Jersey's coastal and estuarine waters. This dataset contains a number of attributes including total nitrogen, latitude, longitude, ammonia, nitrite + nitrate, dissolved oxygen, chlorophyll a, phosphorus, salinity, secchi, total phosphorus, temperature and total suspended solids. This dataset has been collected throughout New Jersey's coastal estuaries, rivers, and open ocean environments from 1989 to 2009 (Figure 17). In order to subset this dataset for the BB-LEH estuary, a GIS file was created showing the locations of sampling sites through New Jersey, locations within the BB-LEH system were manually selected with a GIS and coded (n = 41 site locations).

a). *Total Nitrogen map baywide*

A map (figure 18) showing total nitrogen for June-August 1989-2009 for the BB-LEH was created by selecting sampling stations that had been sampled in excess of 19 times by the NJ DEP BMWQ that fell within the BB-LEH estuary. For these points, distance based spatial interpolation (OI) was used to create a gridded spatial surface using the mean total nitrogen value by station.

b). *Nitrogen Loading vs watershed development.*

A secondary analysis for sampling stations that were located upstream (not in the bay proper) and had mean practical salinity units (PSU) of less then 5 were compared to the local subwatershed percent altered land. A low PSU value for a sample site location indicates that the majority of the water is freshwater by definition and therefore the nutrient load has not been substantially diluted with cleaner ocean water. A total of nine water quality stations fit this maximum PSU criterion. The percentage development of the subwatershed was determined by comparing the land use land cover GIS dataset created at the Center for Remote Sensing and Spatial Analysis, Rutgers University (Lathrop and Haag 2007) with the U.S. Geological Service (USGS) defined subwatershed boundaries. For each subwatershed, a percentage developed in 2006 was determined by taking the total area of all urban lands and dividing it by the total area of the subwatershed minus water (Figure 19 & Figure 20). The total nitrogen values for these select stations was regressed vs. the percent development of the upland watershed using a least squares linear regression. A 95% confidence interval was computed for the linear regression line by redistributing the residuals around the model predicated values (permutation bootsrap) with replacement and recompiling the regression line 1,000 times. The 975 and 25 ranked regression coefficient and slope were used as the 95% confidence interval.

Results and Discussion

2009 status of seagrass in the Barnegat Bay-Little estuary system (research question 1)

The 2009 seagrass Remote Sensing Survey of BB-LEH classified 5,253 ha of seagrass habitat (2,256 sparse ha, 2,527 moderate ha, and 470 thick ha) (Table 2 & Figure 21). An accuracy assessment was conducted using the 125 validation sites for both a presence absence and categorical values (sparse moderate or thick) (Table 3 & Table 4). In addition, an un-weighted Kappa statistic was used to normalize the influence of categories that cover a disproportionate area. Good results were obtained for the presence absence accuracy assessment with an overall accuracy of 87% and a Kappa value of 73%. This represents a substantial agreement between the GIS and the reference dataset. For the 4 class seagrass density map (Table 4), I obtained a total accuracy assessment of 70% and a Kappa statistic of 47% a moderate agreement. The results of the accuracy assessment show that it was difficult to differentiate between medium and thick seagrass habitat (Table 4). The procedure to select the validation sites (random vs. targeted) can drive which error (omission and commission vs. categorical) is better constrained. For example, if a larger percentage of validation sites were targeted to known seagrass habitat, a larger percentage of validation sites with seagrass would provide better estimates on errors in the categorical values (sparse, medium or thick) of seagrass habitat. If, on the other hand validation sites were selected randomly across the entire estuary a better estimate could be made on the total errors of omission and commission of seagrass habitat (presence / absence) vs. seagrass density. A better approach in future work would be to collect a larger number of samples in known seagrass habitat to provide more information on the accuracy of the categorical nature of the GIS maps.

The 2009 imagery collection was a challenge due to meteorological events (cloud cover), which caused two separate imaging attempts before good aerial photography could be obtained. Some areas of high turbidity were found in LEH and southern Barnegat Bay. On further analysis of historical Landsat satellite imagery, it was noted that these areas routinely experience higher turbidity events than other parts of the BB-LEH estuarine system. In future image missions to monitor seagrass in BB-LEH particular attention should be paid to the eastern ICW near the Route 72 Bridge (-74 15 W 39 42 N) to determine if water clarity is sufficient to discern features on the bottom of the Little Egg Harbor and Southern Barnegat Bay (SBB) estuary as this is the area that was observed to have the highest frequency of turbidity events.

Automated Classification Techniques (research question 1)

The automated classification technique of the segmented image objects created in Ecognition™ using the CART model provided a lower overall mean accuracy assessment and lower mean Kappa statistic *vis a vis* the manual classification of the same image objects (75% vs 87% overall accuracy assessment and 45% vs 73% kappa statistic respectively) (Figure 22 & Figure 23) . It was not possible to create a map showing areas of commission and omission for the CART model due to the fact that the mode had (n=10,000) unique solutions. A further refinement on this technique would be to use a Random Forest Algorithm to generate hundreds of CART models. Random Forest can estimate the dependant variable by taking the mean over the full ensemble off CART models.

The primary benefit of this type of automated classification technique is the reduction in operator time to manually classify the imagery, and a robust rule set that can be duplicated. The manual classification of the 36,000 ha study area into image objects with a minimum mapping unit of .05 ha required 3 months of trained operator time. A significant expense to the overall project budget. The CART model required 2 weeks of programming and data prep to complete and implement. Subsequent CART analysis will require much less effort if the original computer code is reused. A secondary benefit of the CART model is that all of the field data can be used for both reference and validation. Issues with using the CART model include, water clarity, image edge effects, and benthic albedo. The human eye naturally accounts for these differences by applying knowledge of spatial patterns and background information that for this application cannot be equaled by the CART model. CART models could be a good starting point for future classification of seagrass habitat. It is suggested for future seagrass remote sensing missions to start with a CART classification as a first step and then apply a manual cleaning re-interpretation to this initial classification. It is further suggested that a training dataset could be selected by a trained image interpreter by manually classifying a percentage of each bay segment. In addition by providing the CART model with contextual information such as adjacent image object color, depth and historic classification, the model might more closely mimic the results of the trained image interpreter.

Quantifying the seasonal change in seagrass habitat (question 2)

An attempt was made to discern the change in percent cover for a specific seagrass bed adjacent to the northern most part of Island Beach State Park using four imagery datasets. It is not possible to directly compare the biomass or seagrass percent cover across the entire time period because Kennish et al. (2007 & 2008) and Kennish (unpublished data 2009) only collected data from June-October and not during March or May. These images were collected on March 21st 2003, May 4-5th 2004, July 13th 2009, August 5th and 13th 2006, and October 4th 2004 (Table 1). Normalizing the different imagery (Figure 24 & Figure 25) datasets was challenging because: (1) The imagery was collected with different sensors including digital and analog aerial photography; (2) Some of the imagery consisted of multiple images mosaiced together which do not always have similar spectral responses between the different images; (3) there were clear anisotropic differences in the imagery, caused by the difference in sun angle and the reflectance of ground material; (4) water turbidity and depth were not consistent between the imagery. All of these differences between imagery can be broken down into two different types the first occur consistently or systematically across the imagery (ex water depth, overall light quality, sensor bit depth) the second are non systematic and therefore cannot be corrected using the Pseudo Invariant Feature Method (ex localized turbidity, sun angle).

The results of this study show that the differences across the growing season appear primarily in the red and green wavelength. The blue wavelength provides little information between deepwater and shallow benthic habitats (Figure 25). In addition, by combining the data into a panchromatic dataset by stacking the red, green, and blue bands, a reduction in albedo occurred with a concurrent increase in seagrass biomass and percent cover (Figure 25). Because scattering and absorption of light is wavelength dependant, the band that provides the ‘most’ information on seagrass biomass could change depending on tidal cycle and water clarity between image pairs. By stacking the bands a composite response across all wavelengths can be computed. Figure 25 shows that for the corrected May, 2003 imagery a decrease occurred in the green band compared to the spring March photography. This reduction is not seen in the peak biomass of the July imagery in the green band but rather in the red band. The July imagery was collected at lower tide level than the March and May imagery which was confirmed by the higher digital numbers for the uncorrected July imagery. The July imagery provides information on the benthic environment in the red wavelength. To compare between the May and July imagery, the bands can be combined.

This study was limited in extent due to image quality issues, particularly in the March, August, and October imagery which was not collected to support seagrass monitoring. No attention was paid to tidal cycle, wind speeds, or sun angle on data acquisition. Large areas of those images had unusable data within BB-LEH and necessitated the focus on a small study area. It is apparent by manual observation that the March and October imagery was not collected at peak seagrass biomass like the July August imagery. This can be empirically verified by expanding the Kennish et al. (2007, 2008 & 2009) *in situ* data collected in early March and May for this seagrass bed.

2003 remote sensing survey overview (research question 2)

In order to compare the 2003 remote sensing survey data to the 2009 survey data it is important to understand the accuracy of the 2003 dataset. As part of the 2003 remote sensing mission, a number of *in situ* field sites were collected (n=245). All of these field reference sites were used in the classification procedure and therefore cannot be considered a true independent validation dataset. In addition, the 245 reference sites were not selected randomly across the estuary but rather directed towards known seagrass habitat. The 2003 study had a Kappa statistic of 56.2% representing a moderate level of agreement between the classified imagery and the in situ reference sites. The overall accuracy for the entire dataset was 68% for all classes of seagrass habitat and 83% for seagrass presence / absence. Some of the reduction in accuracy is likely due to the fact that the reference dataset was not equally distributed across the estuary but rather directed towards known seagrass habitat as a training dataset. The reference sites were skewed to cover seagrass habitat. A total of 146 of the 245 reference sites contained seagrass habitat (60%), while seagrass only covered 14.5% of the entire estuarine system. A validation data set randomly distributed across the entire bay study area would have had a higher proportion of sites in non-seagrass areas (i.e. deep water areas) with a higher likelihood of correct classification.

Comparison of 2003 and 2009 seagrass survey Data (research question 2 and 4)

The following comparison discusses the results of the change detection analysis conducted between the 2003 and 2009 seagrass surveys. Where possible, ancillary information from outside research projects were used to support trends found within the comparison of the 2003 and 2009 imagery classifications.

The area of total seagrass mapped across the estuary in 2009 was 5,253 ha which represents an increase of 69 ha over the 2003 survey (Table 2 and 2a). This increase in seagrass habitat occurred in specific spatial areas; Figure 21 shows the spatial distribution of seagrass across the entire BB-LEH. This map is divided into three seagrass areas: (1) the areas mapped in 2003 only (1,490 ha); (2) the area mapped in 2009 only (1,560 ha); (3) the area and mapped in both 2003 and 2009 (3,694 ha) (Table 2c). Across the entire estuary a total of 563 ha of seagrass were mapped as lost, with an additional 785 ha of seagrass mapped as gain. Some of the difference in area represents real change in seagrass distribution and some represents an artifact of differences between imagery dates, water transparency, tidal stage and other uncontrolled factors. There is a trade off between early season image acquisition before periods of high water turbidity and peak seagrass biomass occurring in early June-July. Table 6 shows the results of the detailed analysis to differentiate the reason different areas were mapped as seagrass between the 2003 and the 2009 surveys.

Several specific seagrass beds most notably in LEH appeared to be misclassified in the 2003 seagrass survey, resulting in a reduction of 412 ha of mapped habitat and an additional 19 ha for the 2009 survey. These areas were actually areas of dark benthic habitats which closely resembled seagrass habitat in the 2003 imagery. A change in season resulted in 338 ha of seagrass to be mapped in 2009 vs. 2003, mostly the result of *R. maritima* habitat in the northern part of Barnegat Bay. For the entire BB–LEH estuarine system, seagrass habitat area seems to have expanded between the 2003 and 2009. Specific parts of the estuary, most notably the areas adjacent to Barnegat Inlet and to a lesser extent Little Egg Inlet appear to have had some seagrass dieback. This is most likely due to the dynamic nature of the inlet areas with large amounts of sediment displacement. Seagrass located adjacent to Barnegat Inlet is at risk from the impacts of direct physical alteration (dredging), an increase of dredge material on artificial islands, and the installation of a geo-textural tube to stabilize the northern part of the inlet proper. This geo-textural tube may have caused an increase in current flow during tidal transition periods and a greater scouring and or sediment deposition on adjacent seagrass habitat.

For a more detailed analysis, the estuary was subdivided into five sections. The most southern section is Little Egg Harbor which lies between 39 40 00 and 39’ 32 00 N. LEH contains shallow flats on the eastern shore extending 2.6 – 1.3 km from the barrier island complex to the deeper benthic habitats on the western side of the estuary (Figure 10).

In 2009, LEH contained 1,475 ha of mapped seagrass habitat and 1,867 ha of mapped seagrass in 2003 (Table 7 & Figure 26). This 380 ha reduction in the extent of mapped seagrass habitat between 2003 and 2009 was mostly due to a misclassification in the 2003 remote sensing project (Table 7). This misclassification occurred when a shallow dark benthic habitat was confused with seagrass habitat. Direct comparison between the 2003 and 2009 imagery provides some indication that seagrass habitat is similar between 2003 and 2009, but in specific areas it is expanding and or contracting. Figure 27 shows the loss of a seagrass bed between 2003 and 2009 in an area adjacent to the ICW. Overall, seagrass habitat appears stable between 2003 and 2009 in LEH. However, it should be noted that differences in the image acquisition period could be driving some of the increase in seagrass habitat.

LEH was the site of extensive *in situ* seagrass surveys by Kennish between 2004 and 2009 (Kennish et al. 2008; Kennish unpublished data). In addition, Gastrich et al. (2004) and Pecchioli et al. (2006) mapped extensive brown tide blooms within this estuarine system that covered known seagrass habitat in 2000, 2001, and 2002. The NJ

DEP brown tide monitoring program was stopped after the 2004; therefore no data is available after 2004. The 2002 brown tide bloom in LEH was the most severe of the five years of monitoring period with an average of 281,900 cells/ml (Table 9). This increase of brown tide was correlated with reduced Secchi disk value and higher water turbidity. Thus, increases in seagrass habitat in the northern part of LEH could be caused by the cessation of brown tide blooms during 2008 and 2009 and associated increase in light penetration. *In situ* data collected by Kennish et al. (2008) (Figures 28 and 29) for 2004, 2006, and 2008 show a decline in benthic submerged aquatic vegetation, including seagrass and macroalgae between 2004 and 2006. Unfortunately, because the brown tide monitoring program did not collect data after 2004, no information exists on brown tide bloom density or extent for 2006. The brown tide densities in 2004 compared to 2000-2002 were low with an average cell density of (15,700 cells/ml). In 2006 Kennish et al (2007) found an average secchi value of 0.97 m this is between the values found by Pecchioli et al 2006 for average secchi depths during brown tide blooms 0.8 m (2001-2002) and 1.2 m for non bloom (2003-2004). Therefore, it cannot be shown conclusively that 2006 was a brown tide bloom year, but based on the Kennish et al. (2007) *in situ* data, it is clear that macrobenthic primary producers (submerged aquatic vegetation) were severely reduced from the previous year(s). The August 2006 aerial photography collected by the U.S. Department of Agriculture (USDA) can elucidate some of the spatial trends of this apparent seagrass dieback. The August 2006 USDA NAIP photography was not optimized to record information on seagrass habitat and health; nonetheless, one specific bed (Figure 30) could clearly be seen for that year. It showed large-scale seagrass dieback in 2006, with a significant amount of internal bed loss with large areas denuded of submerged aquatic vegetation. This corresponds to what was observed *in situ* by Kennish et al. (2007) in 2006 and lends further credence to a system-wide decrease in submerged aquatic vegetation for that year. The 2006 July and August temperature data was the highest over the 2000-2008 time periods (Figure 31). In addition to possible brown tide impacts, high water temperatures in 2005 and 2006 during the mid summer peak in *Z. marina* could have caused the 2006 dieback surveyed by Kennish et al. (2007). Temperature could be either causative or correlative factor or both for *Z. marina* within the BB-LEH estuary system.

LEH has a relatively small upland watershed compared to northern Barnegat Bay, and it has two inlets on either side which pump water in and out of the system. Therefore, it stands to reason that LEH and southern Barnegat Bay could have much longer water residence times then northern Barnegat Bay. Examining the NJ DEP BMWQ total nitrogen data shows two hotspots of high nitrogen enrichment within the estuary (Figure 18). The first occurs in the northern most reach of the BB-LEH, where loading is the highest. The second occurs in the northern segment of LEH and the southern segment of Barnegat Bay. This hotspot is most likely due to low bay flushing rates and associated high water residence times (Personal communication B Chant). This area also contains the highest brown tide blooms within the BB-LEH, and has an associated decrease in water clarity (Figure 3 and 18). An examination of past LandSat satellite imagery (Sept 4th, 1995, Dec 1st, 2001, and Sept 12th, 2001) show high levels of water turbidity persist in this northern section of LEH. It is therefore of particular importance to understand how seagrass beds are changing in LEH.

Seagrass habitat in LEH has been shown to be highly dynamic (Kennish et al. 2008) and that the system is capable of switching between macrobenthic primary production to phytoplankton blooms (gradient of eutrophication) and back again. This would support with the model described by Wazniak et al. (2007) which shows a degradation of benthic habitat through a categorical scale from a pristine seagrass dominated community changing from a seagrass bed infested with macroalgae; to a highly eutrophic estuary dominated by pelagic primary production (Figure 1). This can be seen through the change in seagrass extent between 2003 and 2009, the 2006 NAIP photography, and the 2004-2008 *in situ* data.. Because of the reduced flushing rates of LEH, it appears more susceptible to the impacts of nutrient loading and therefore should be the highest priority to monitor and measure the impacts of nutrient loading.

The second area of interest is the southern portion of Barnegat Bay located south of Barnegat Inlet and north of LEH 39 40’ 00’’and 39 44’ 30’’ (Figure 10 & Figure 32). This portion of the BB-LEH system has a total of 721 ha of seagrass mapped in 2009 vs. 490 ha mapped in the 2003 seagrass survey. Through closer analysis, 146 ha of the total change in area represent seagrass habitat expansion mapped in 2009, but not in 2003 (Table 6). A total of 26 ha of seagrass habitat were lost between 2003 and 2009, mostly in the northern portion adjacent to the Barnegat Inlet and the ICW. The increase of seagrass habitat between the 2003 and 2009 study periods is likely to be higher, but due to water turbidity this could not be conclusively demonstrated. A total of 169 ha of seagrass mapped in 2009 could not be verified as seagrass gain due to poor image quality in the 2003 remote sensing survey. Again, much like LEH there is an apparent gain in seagrass habitat between the 2003 and 2009 survey periods. In addition, the seagrass habitat deepwater edge (question 3), where visible in both the 2003 and 2009 aerial photography, is expanding (Figure 33) into deeper waters.

Southern Barnegat Bay (Figure 10, Figure 32 & 34) shows similar results to LEH. Brown tide blooms occur at their highest values in SBB and LEH, which suggests higher bay water residence time with an associated increase in nutrient retention. Unlike LEH, there is no extensive record of *in situ* data, and the 2006 NAIP photography provided no information on benthic habitats. Nonetheless, SBB could have had a significant dieback of seagrass in 2006 because it shares many of the same characteristic as LEH. This part of the estuary has been extensively dredged on the eastern edge adjacent to Long Beach Island for both boat access and sediment mining (Figure 15 & Figure 35). The sediment mining was likely done to provide material to Long Beach Island after the Ash Wednesday Noreaster of 1962 which caused substantial erosion to Long Beach Island LBI (Psuty Personal Communication 2010). Seagrass habitat most likely extended to the tidal flats in a similar fashion at Island Beach State Park prior to the extensive alteration to the benthic environment.

More work needs to be done to assess the seasonal and annual variation in seagrass habitat within the SBB (Figure 10 & Figure 32) segment of the BB-LEH. This portion of the BB-LEH appears susceptible to multiple stressors including harmful algal blooms, dredging, and direct impacts from boat scarring. This remains one of the least studied parts of BB-LEH in regard to seagrass habitat condition.

The Barnegat inlet region has undergone obvious changes between 2003 and 2009 (Figure 36 & 37) due mostly to shifting sand bars, ongoing dredging for channel navigation, and modifications to stabilize the inlet. A total of 384 ha of mapped seagrass was lost between 2003 and 2009 but at the same time there was a corresponding gain of 109 ha (Table 6). Some of this gain might be due to high blue mussel spat in 2008 which overwintered and was visible in the 2009 imagery. High densities of blue mussels could result in areas incorrectly mapped as seagrass beds or cause an overestimation of seagrass bed density. This could have caused some overestimation of seagrass habitat in 2009 for both presence / absence and percent cover because blue mussel spat was found within seagrass habitat during *in situ* site visits. Other confounding factors include *R. maritima* growth on shallow sand flats. This could account for seagrass habitat that was present in 2003 but not mapped due to the early image acquisition period. The Barnegat Inlet region is primarily composed of well sorted sand because high current speeds transport darker organic detritus into the deeper estuary or out the inlet and into the coastal ocean. Therefore, no habitat was classified as poor imagery (high water turbidity) since the benthic habitat could be viewed in both the 2003 and 2009 imagery collection.

The U.S. Army Corps of Engineers has undertaken extensive alteration of Barnegat Inlet to keep it open for recreational and commercial boat transit. This included placing a geo-textural tube across the northern part of the inlet in the late 1999 and early 2000 (Kennish personal communication) and dredging the ICW. This appears to have funneled current flow to the west, shifting the coarse grained sediments over the top of the two large seagrass beds. This loss of seagrass habitat could be classified as an indirect disturbance resulting from alterations to the Barnegat Inlet form and function.

Because the Barnegat Inlet region is flushed twice daily with ocean water by semi-diurnal tides it would appear less susceptible to nutrient loading and associated algal blooms (Figure 3). Seagrass habitats in Barnegat Inlet that were not physically altered by shifting sediments increased in area. This would lend support to the observation that the physical effects of Barnegat Inlet are impacting the extent of seagrass habitat in this local area. It should be noted that these impacts occurred in areas within 3-5 km of the inlet and did not extend north or south into Barnegat Bay proper.

Central Barnegat Bay (CBB) (Figure 10 & Figure 38) the section of Barnegat Bay north of Barnegat Inlet and south of the Route 37 Bridge represents a transition zone from *Z. marina* dominated habitat, to *R. maritima* dominated habitat in the north. The western edge of CBB contains little seagrass habitat vs. the extensive shoals extending 1-1.5 km into the estuary from the eastern barrier island complex (Figure 39). In total, the CBB section contained 1,662 ha of seagrass habitat in 2009 up from 1,406 ha in 2003. The 297 ha was deemed new seagrass growth, or bed expansion. A total of 50 ha was mapped as seagrass due to a change in season, and represents *R. maritima*, beds growing in the later 2009 imagery collection. A total of 61 ha of seagrass occurred mostly as bed edges representing a contraction of seagrass habitat into shallower water. The majority of the new seagrass habitat identified in the 2009 imagery is located along the shallow water sand flats adjacent to Island Beach State Park (Figure 39). Seagrass habitat therefore shifted to shallow habitat within the CBB region between 2003 and 2009. It is unclear why the sand flats adjacent to Island Beach State Park were denuded of seagrass in 2003 (Figure 40). Some of the difference could have been caused by increasing density of *R. maritima* between the 2009 and 2003 imagery datasets.

The CBB portion of the BB-LEH estuary has minimal amounts of boat dredging (Figure 15) vs. the southern three segments and the northern segment, mostly because the majority of the barrier island on the eastern edge is part of Island Beach State Park and therefore has a limited amount of boat docks (Figure 40). It is interesting to note that the protection and conservation of a terrestrial park appears to have also protected large swaths of seagrass habitat that could otherwise have been dredged for boat navigation and sediment mining. The CBB does have boat scars, one directly adjacent to Tice's Shoal that was 700 m in length (Figure 13). These boat scars, while dramatic in the photography, were not found with widths over 2.5 m and therefore do not appear to be widening through time. This suggests that they represent ephemeral impacts to seagrass habitat.

The northern-most portion of the BB-LEH system NBB (Figure 10 & Figure 41) is dominated by low salinity waters and heavy inputs of freshwater from the Toms River watershed (NJ DEP data 1989-2009). The majority of the seagrass habitat in this portion of the estuary was dominated by *R. maritima* with small pockets of *Z. marina* (Figure 41). A total of 567 ha of seagrass were mapped within NBB in 2009. This represents an increase of 188 ha of seagrass habitat vs. the 2003 data. All of this mapped increase in seagrass area most likely does not represent 'true' change but is attributed to a change in the growing season or poor image quality. It is difficult to compare imagery collected in April to imagery collected in July with respect to *R. maritima* habitat. Figure 42 shows a *R. maritima* bed with photography from May 4-5th 2003, August 2006 and July 2009. It is apparent through the *in situ* data that *R. maritima* was growing in both 2003 and 2009, but it was not visible in the May 2003 imagery. In addition, the full tidal cycle for the entire BB-LEH estuary system is ~ 2.5 hours. Low tide at Barnegat Inlet does not occur at the same time as low tide in northern Barnegat Bay. Because imagery was collected in a relatively short period of time, the entire estuary was not at a low tide during collection.

Areas with shallow water and bright sand as a backdrop did not suffer much from this issue, but northern Barnegat Bay has darker organic soils with associated lower albedos which make it more difficult to photo interpret than other regions. Because of these reasons no definitive gain or loss of seagrass habitat within northern Barnegat Bay could be determined based on the 2003 and 2009 aerial photography. Furthermore, as in the case of Southern Barnegat Bay, Northern Barnegat Bay does not have an exhaustive *in situ* dataset. This remains a high priority data gap to provide a full understanding of seagrass habitat health across the entire BB-LEH.

Direct impacts to seagrass habitat (question 3)

Boat scarring within the BB-LEH estuarine system was found throughout the entire estuary. Specific areas of the estuary, Tice's Shoal (Figure 13) for example have higher incidence of boat scarring. A total of 42.9 km of linear boat scars were mapped across the entire study areas. If each scar is assumed to be 1 meter in width that would represent a total of 4.29 ha of scarred habitat, a very small percentage of the entire estuary seagrass habitat. It is likely that a large percentage of boat scars were not found in this analysis, but even if the amount was underestimated by an order of magnitude, it would still appear that boat scarring does not play a significant role in the reduction of seagrass habitat. In addition, due to the small amount of wide seagrass scars mapped (over 2.5 meters wide) it does not appear that boat scars are expanding in size after the initial formation. Therefore, in this estuarine system, boat scars most likely represent ephemeral impacts to seagrass habitat.

A total of 1,468 docks were mapped within the BB-LEH estuarine system proper. The total area of all of these docks is 30.7 ha and 53.7 ha for docks and the 2 meter buffer zone. Out of the 1,468 mapped docks, 684 were within 100 m of mapped seagrass habitat (in either 2003 or 2009). This represents a total of 31.2 ha of buffered docks adjacent to mapped seagrass habitat. Compared to the mapped extent of seagrass habitat (5,252 ha in 2009) within the BB-LEH, this represents a small fraction of the overall seagrass habitat. Secondary effects of these boat docks include changes in the current flow, sedimentary budget, boat scarring, and dredging for boat channels. The direct impact of the boat docks on the areal extent of seagrass habitat is minor in this system. The minimal amount of seagrass impacted by boat docks can be partially explained by the NJ DEP which regulates dock construction to minimize the impact to seagrass habitat (N.J.A.C. 7:7E). It should be pointed out that the vast majority of seagrass habitat is not located adjacent to land and would therefore not be at risk to impact by the dock proper. On the other hand increasing water turbidity and dredging as a result of dock construction could have an impact on nearby seagrass habitat.

Dredged areas of the BB-LEH estuary system covered extensive areas adjacent to mapped seagrass habitat on the eastern edge of the barrier island complex (Figure 15). The mapped extent of dredged areas covered 790 ha (Table 9), excluding the western shore. In areas where no dredging has occurred (Island Beach State Park) seagrass habitat extends almost to the intertidal flats. Dredging for sediment and boat access could account for a large reduction in available habitat for seagrass within the BB-LEH estuarine system though in the longer term - outside the immediate time period of this study. The extensive non-linear dredged on the western side of the barrier island complex could be attributed to the Ash Wednesday Noreaster which occurred on March 6th-8th 1962. An analysis of older photography collected by the US Geological Survey in 1920's do not show these expansive dredged sites (Figure 35).

Conclusions

The health and spatial extent of seagrass within the BB-LEH estuary serves as a biological indicator of water quality and the impacts of eutrophication through time and space. Short (2007) found that *Zostera m.* biomass and percent cover has declined in Great Bay, NH between 1995-and 2005 while seagrass distribution has remained relatively constant. To characterize the spatial extent, health, and density of seagrass beds across the entire estuary, it is therefore necessary to combine synoptic remote sensing surveys concomitantly with comprehensive *in situ* assessment. This study produced an excellent level of agreement between manually classified image objects and an *in situ* validation dataset. This is a time intensive process which required a large time investment from a trained image interpreter (3 months for this study). The CART analysis did not provide the same level of accuracy that was obtained with the manual image interpretation techniques. This is primarily due to the inconsistent spectral response of seagrass habitat, water depth, benthic albedo, water clarity, water surface conditions, and anisotropic effects on the aerial photography that are difficult to control for. I suggest that future projects to map seagrass apply a hybrid method by first manually classifying a percentage of the image objects for each aerial photograph and then using those classified images as a training dataset within the CART model. This proposed method has several benefits including an increase in the size of the training dataset with a modest increase in cost. In addition the number of *in situ* training points would also be reduced by using this hybrid approach.

Comparing different types of imagery from scanned analog photography to digital imagery shows that the number one constraint in accurately mapping seagrass habitat is the timing of imagery acquisition to both maximize seagrass standing biomass and minimize water turbidity and water depth. It was determined that aerial photography not collected to support benthic habitat mapping could not provide system-wide or region-wide information on seagrass habitat, rather it was only useful for specific areas of high image quality. In one location high quality imagery from several different acquisition dates was processed to provide information on changes in seagrass biomass across several different years and growing seasons. This information tracks well with other *in situ* studies that show seagrass biomass and percent cover through the growing season Kennish et al. (2007). It is important to understand both intra and inter annual variability in seagrass habitat to better understand differences in imagery acquired during different periods of the growing season. Long-term trends over many years and decades should therefore take particular care to collect aerial photography during similar periods of the year to prevent seasonal bias. This can be challenging due to adverse conditions (wind, tide, cloud cover and rain) that prevent consistent imagery collection.

The comparison of the 2003 vs. 2009 remote sensing surveys suggests that the overall area of mapped seagrass cover was higher in 2009. In addition, seagrass habitat expanded into deeper habitats indicating that water clarity and/or other growth conditions were better in 2009 vs. 2003. Some of the 2009 increase may be due to an artifact of the later seasonal date of imagery collection (July 7th 2009) as compared to the 2003 aerial photography which was collected early in the spring before peak seagrass biomass (May 4-5th). Determining the causative factors for seagrass decline and expansion is difficult because seagrass habitat integrates the ecological signal over a larger period of time than the original stressor. For example, the 2003 seagrass extent was most likely impacted by the 2000-2002 brown tide blooms, which was not present during 2003 and 2004. This temporal lag is one reason that seagrass is such a good indicator of estuarine water quality, but it also shows the need for more consistent data collection to more fully understand the impact of various stressors on seagrass habitat. Collecting aerial photography later in the growing season provides more information on the extent of *R. maritima* habitat and allows *Z. marina* to reach peak biomass. Unfortunately imagery collected later in the growing season is more likely to be impacted by poor water quality, making the timing of imagery collection a tradeoff.

Direct impacts to seagrass habitat including dredging, boat docks and scarring were mapped to assess their contributions to diminishing seagrass habitat. Boat docks and boat scarring contribute a minor reduction in seagrass habitat when compared to the overall areal extent within the BB-LEH estuary. Historical dredging has significantly reduced the amount of available habitat across most of the estuarine system. A decline in seagrass habitat was observed between 2003 and 2009 in the vicinity of Barnegat Inlet. I attribute this decline to the direct physical alteration of the Inlet.

Work done in the intervening years (2004-2008) by Kennish (Kennish et al. 2007; & Kennish et al. unpublished data) showed a significant decline in submerged aquatic vegetation (seagrass and macroalgae) across the estuary. These results were corroborated by examination of the 2006 August aerial photograph which showed extensive areas of seagrass dieback as compared to the 2003 and 2009 imagery. The 2006 aerial photography as examined in this study was not useful for large areas of the BB-LEH estuary because it was not collected to maximize the ability to view benthic habitats like the 2003 and 2009 imagery. In specific areas, it does provide a snapshot view of the estuarine benthic environment (Figure 26). The 2006 imagery and the Kennish et al. (2007) *in situ* data therefore show that seagrass habitat has not been expanding in a linear fashion between 2003 and 2009.

Lathrop and Haag (2007) found that brown tide blooms were associated with 'lower freshwater inflow, higher salinity, and water temperatures'. This indicates that yearly changes in nutrient loading combined with bay flushing rates might be the key variables controlling the severity of harmful blooms across the Barnegat Bay – Little Egg Harbor Estuary System. This study has shown a statistically significant relationship between upland land cover alteration and total nitrogen loading (Figure 34). Therefore, as development increases in the Barnegat Bay-Little Egg Harbor watershed increasing amounts of nitrogen will be delivered to the estuary system. The future impact of nutrient loading will be influenced by the degree that the upland watershed is developed, that type of development practices used, the ultimate source of the nutrients, and mitigation attempts directed at reducing the loading from historic land use change. The vast size of this non source point pollution problem make mitigation challenging.

Orth et al. (2010) published a review of seagrass monitoring in Chesapeake Bay discussing the historical distribution and present range of *Z. marina*. Of particular importance is the existence of a multi–decadal aerial photography database and remote sensing survey program that has collected and processed aerial photography specifically to map seagrass habitat annually between 1984 and 2007. This longitudinal dataset has allowed researchers within the Chesapeake Bay to assess the trends in seagrass distribution through time, compare the success of various transplant and relocation efforts, and compare seagrass decline to *in situ* water quality parameters. These types of analyses are not possible in the BB-LEH because the necessary base imagery collected specifically to monitor benthic habitat do not exist on an annual basis. Imagery not collected for the purpose of mapping benthic habitats was found in this study to be inadequate for mapping the baywide distribution of seagrass habitat.

Future Considerations

Seagass habitats worldwide have been reduced through human induced habitat changes (Short and Wyllie-Echeverria 1996; Orth et al. 2010; Waycott et al. 2009). As the nutrient flux from upland habitats continue to be altered both in terms of new urban lands and mitigation attempts, it will be important to track the biological response within the BB-LEH and other similar estuary systems. In particular, seagrass habitat should be targeted as the primary ecological indicator of eutrophication in estuaries that have a history of supporting seagrass (Wazniak et al, 2007; Burholder et al. 2007). *Zostera m.* areal extent within the mid Atlantic bight is highly dynamic with some locations showing new growth and expansion of existing beds, such as the Virginia and Maryland Coastal Bays (Orth et al, 2007; Granger and Nixon, 2007; Wazniak et al, 2007) while others areas such as Great Bay NH and Chesapeake bay have shown a decline in extent (Short, 2007; Orth et al. 2010). The primary cause for seagrass decline and or expansion is related to site specific changes in water quality, temperature, direct alterations, and or disease.

(Orth et al, 2010; Burholder et al, 2007; Short, 2007). To understand what causative factors drive seagrass distribution and health and in a specific estuary it is necessary to have a dedicated monitoring program. This should be done on an annual or semiannual basis to avoid missing major changes in the extent and health of seagrass habitats as this study has shown (Orth et al, 2010). Future seagrass monitoring projects focused on change detection should incorporate a method to provide a level of the certainty of habitat change (confidence interval around the estimates of seagrass area). A useful approach would be to delineate a number of seagrass beds in the field using a GPS system, and then to apply a ratio estimator to estimates errors of omission and commission (Lathrop 2006). This would allow resource managers a better means to assess the statistical as well as real world significance of mapped changes created by remotely sensed surveys and to provide a more informed view of the impact of ecosystem level changes on seagrass habitats. Long-term attempts to increase seagrass habitat should focus on upland strategies to lower the amount of nutrient loading vs. system wide *in situ* restoration attempts that focus on very small portions of the estuarine. This study suggests that under present conditions that the seagrass beds display a high degree of resilience with strong capacity to rebound. Part of this resilience might lie in the seagrass seed bank that allows seagrass habitat to recolonize areas after a disturbance (Hauxwell and Orth, 2002). However, this does not to suggest that the seagrass beds will continue to be able to respond to extended years of high turbidity or otherwise impaired growth conditions. If a tipping point is reached such that the seagrass beds are extirpated estuary wide, then restoration might make sense if the seed bank is determined to be unviable (Hauxwell and Orth, 2002). This agrees with the conclusions of Orth et al. (2010) who state ”Restoration efforts can be important for initiating or accelerating a recovery but only if water quality is improved, and these conditions are maintained”.

To more fully understand the spatial patterns of seagrass loss and gain in relationship to the watershed nutrient inputs, a greater understanding of the spatial and temporal dynamics of bay circulation and flushing are warranted. A network of instrumented buoys would provide much needed information on bay water temperature, salinity, and algal indicators to supplement the recommended seagrass monitoring program.

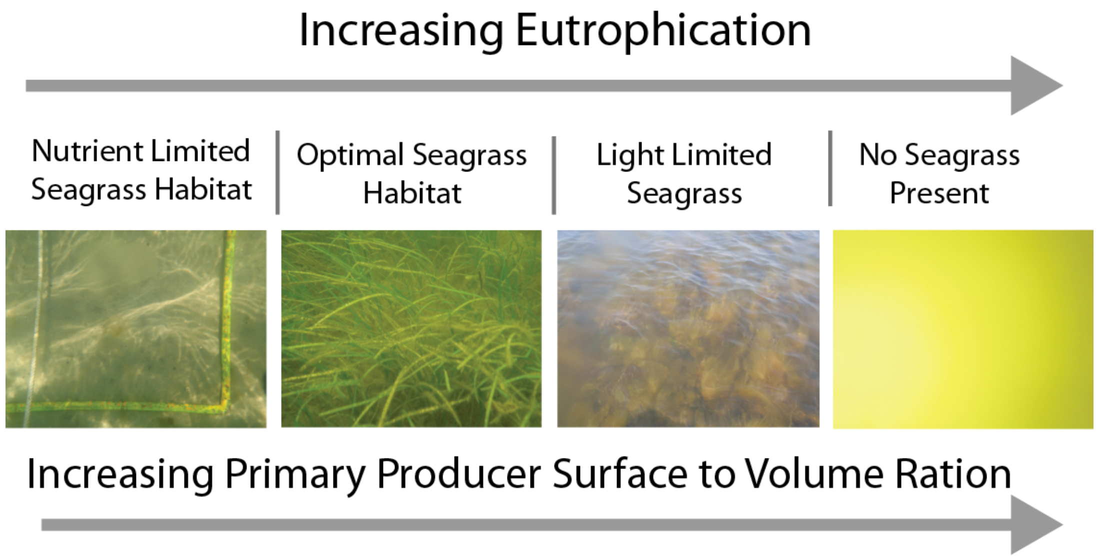

Figure 1. Model of a seagrass habitat change during a growing season in a eutrophic estuary.

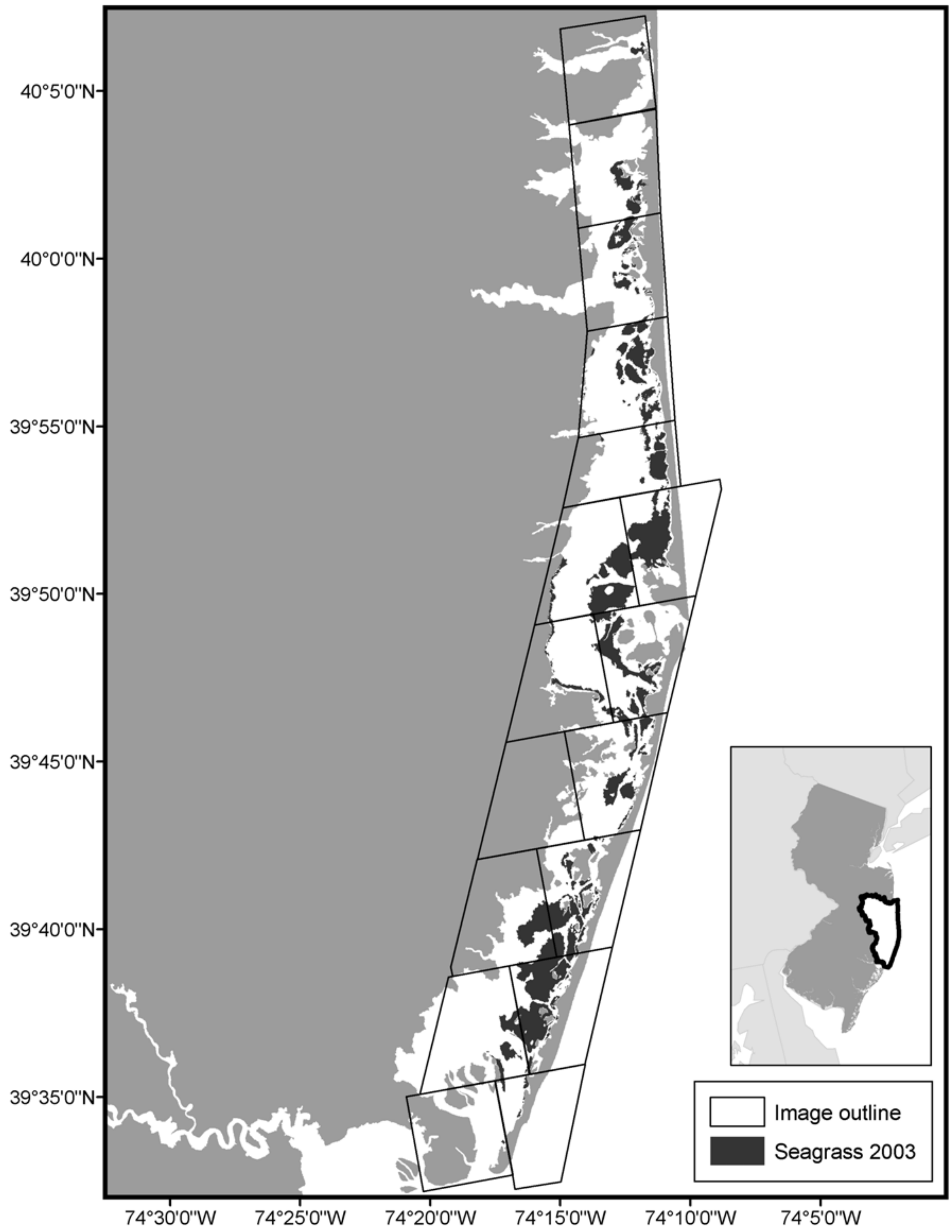

Figure 2. Outlines of the 2009 aerial photography mosaics superimposed on the 2003 mapped seagrass habitat.

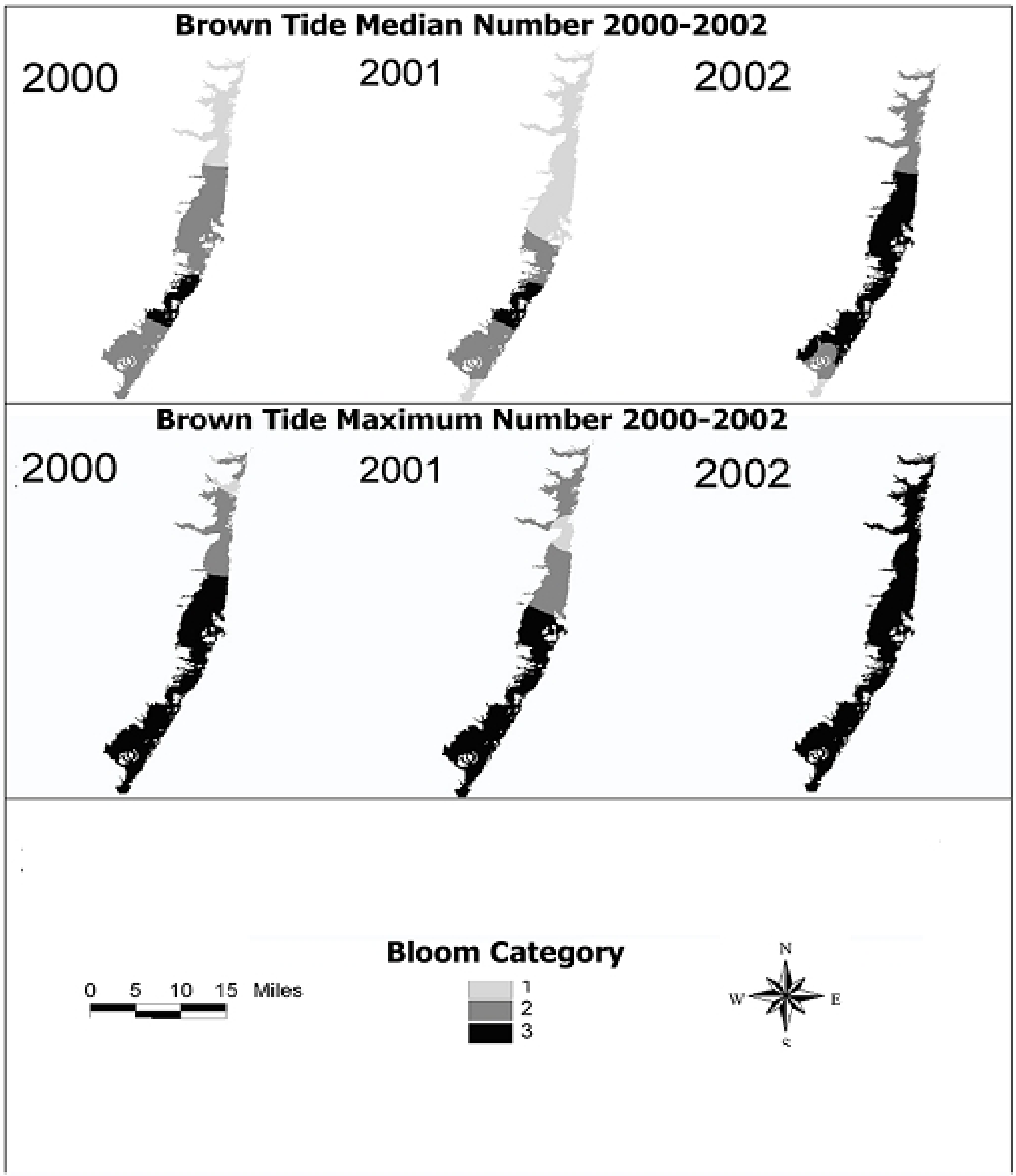

Figure 3. Map of median brown tide (A. anophagefferens) bloom category vs. seagrass beds for 2000, 2001, and 2002. (Used with permsion from Lathrop and Haag, (2007)).

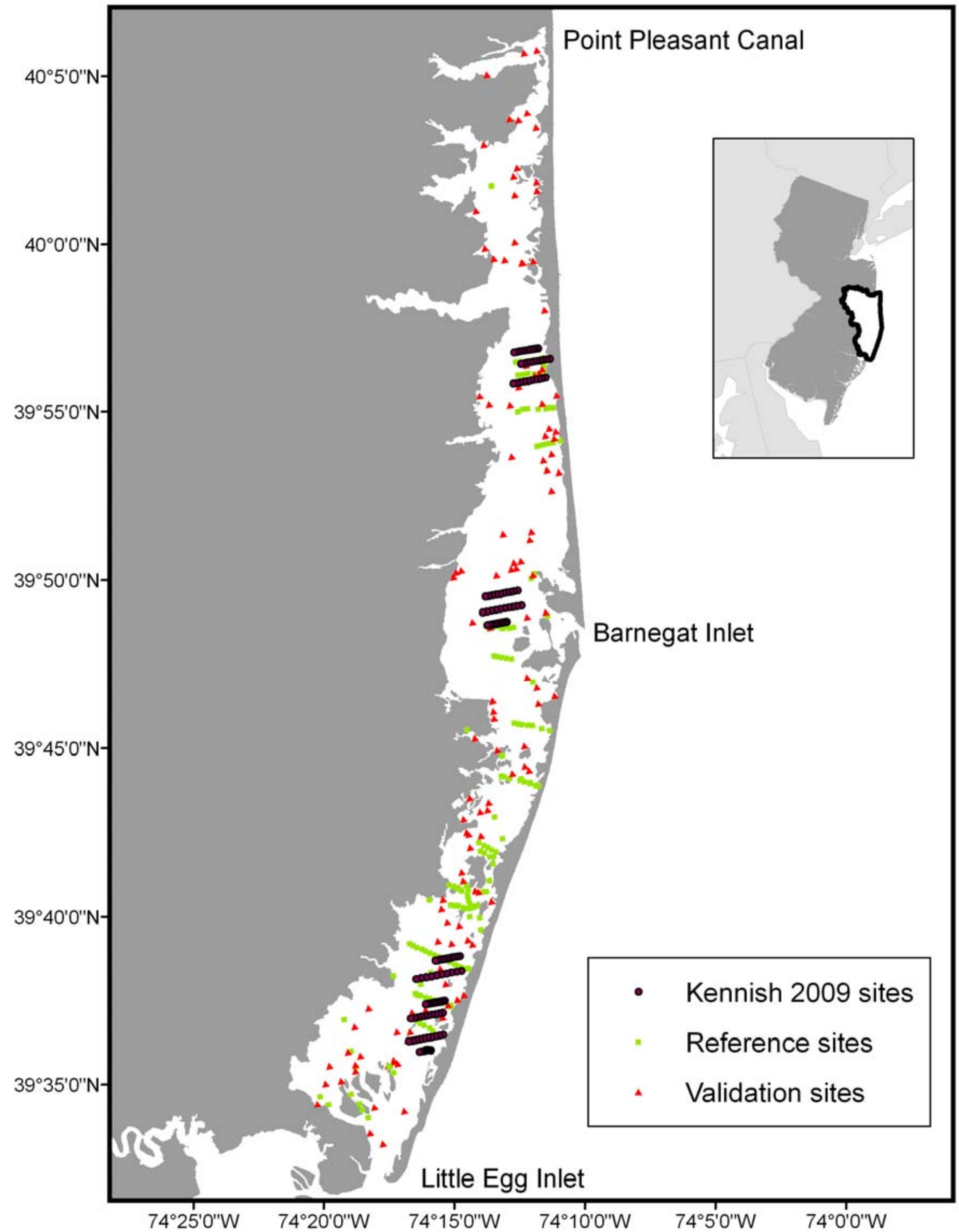

Figure 4. Location of *in situ* sites collected in 2009.

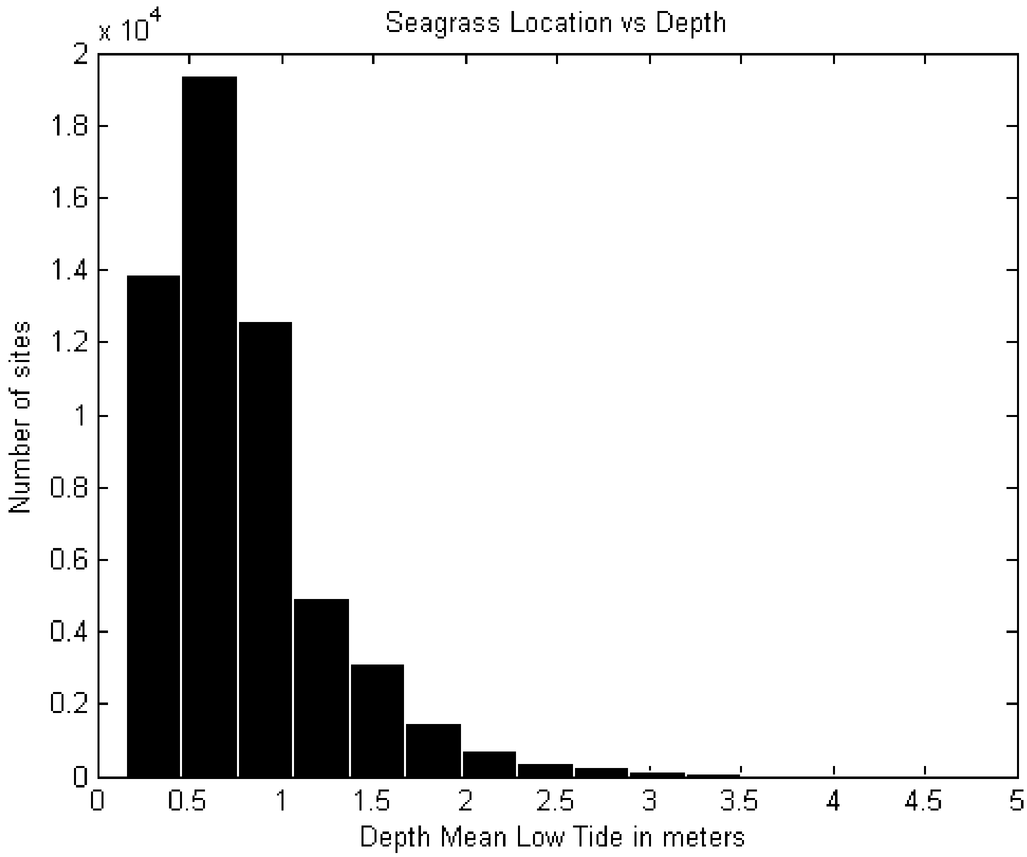

Figure 5. Mean low low tide depth in the Barnegat Bay-Little Egg Harbor estuary for the 2003 seagrass habitat.

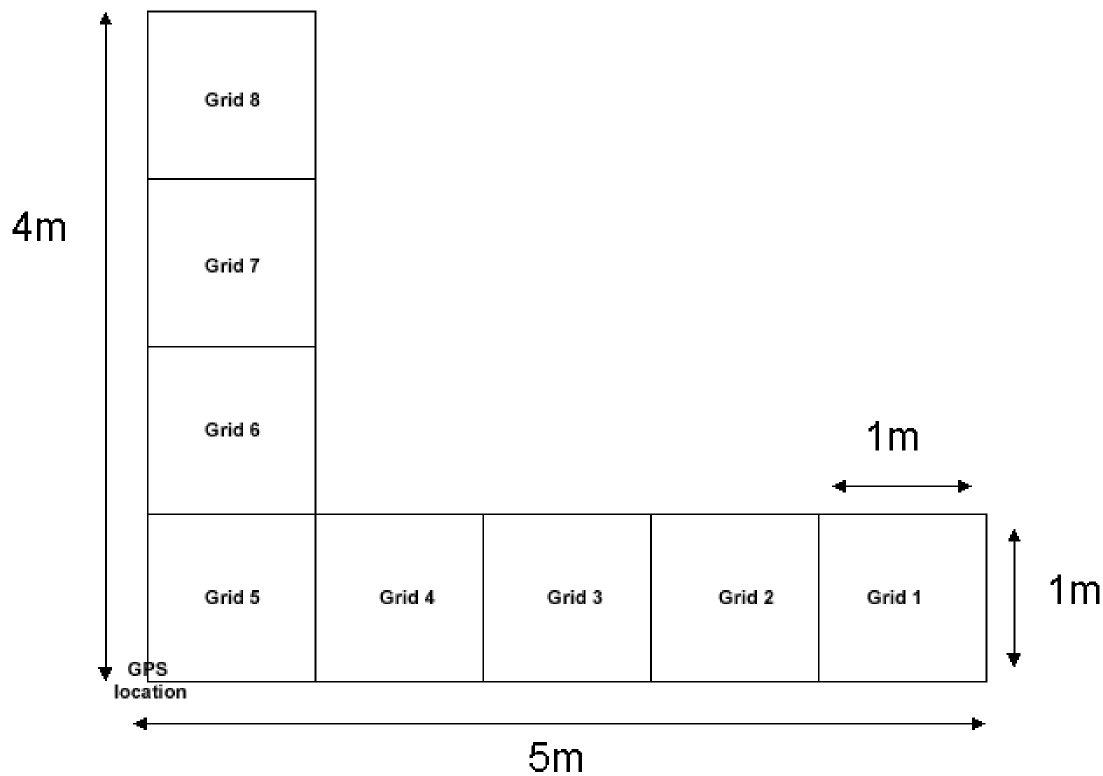

Figure 6. Sampling grid design used for the 2009 *in situ* seagrass survey.

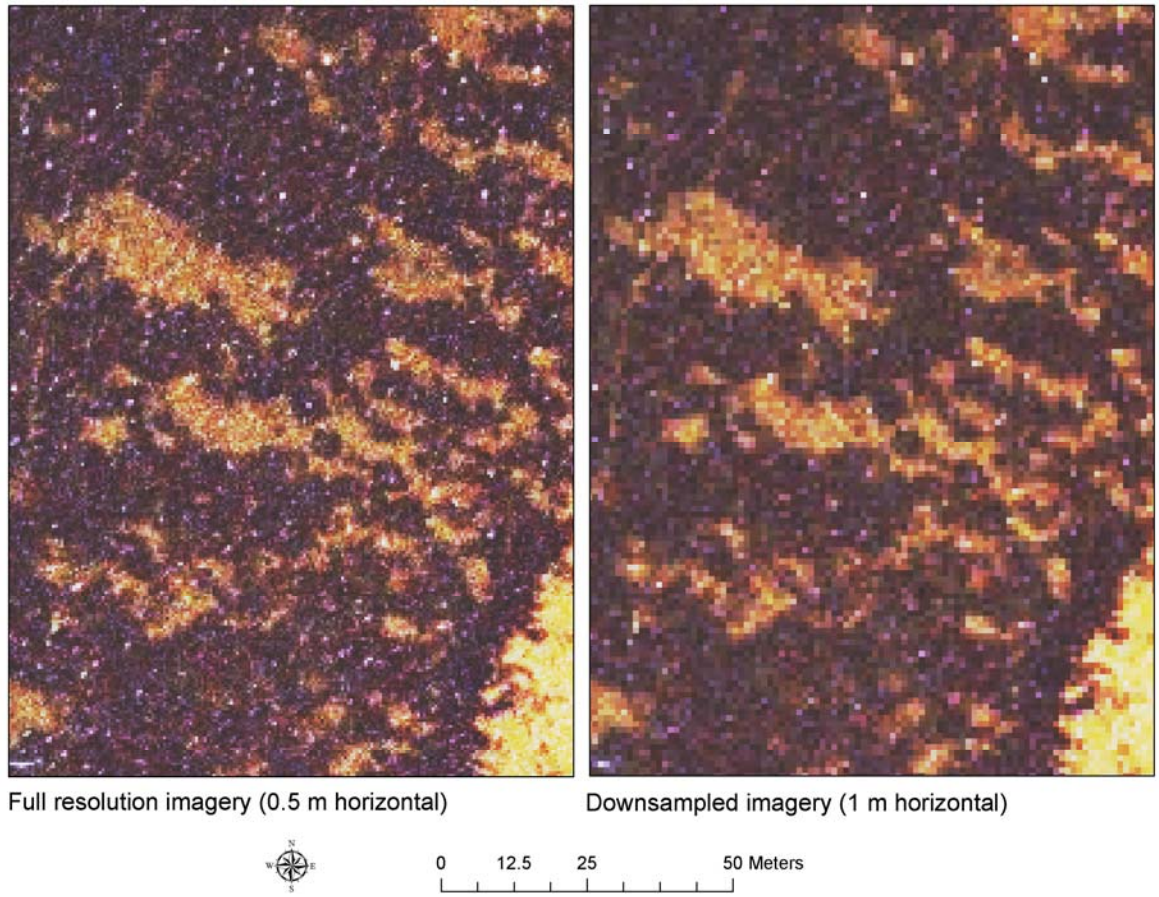

Figure 7. Full resolution of aerial photography from 2009 (left) vs. downsampled imagery (right) for processing in Ecognition™.

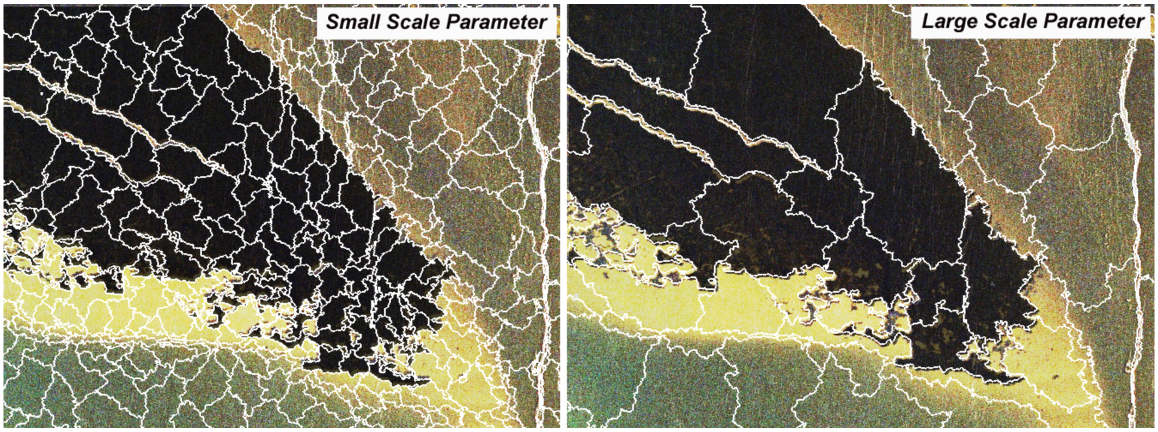

Figure 8. Graphical example of the difference in image objects size created by varying the scale parameter in the segmentation procedure. The left hand image shows the size of image objects with a scale parameter of 15. The right hand image shows the size of image objects with a 70 scale parameter. Not all the smaller image objects are nested inside of the larger (share a common boundary).

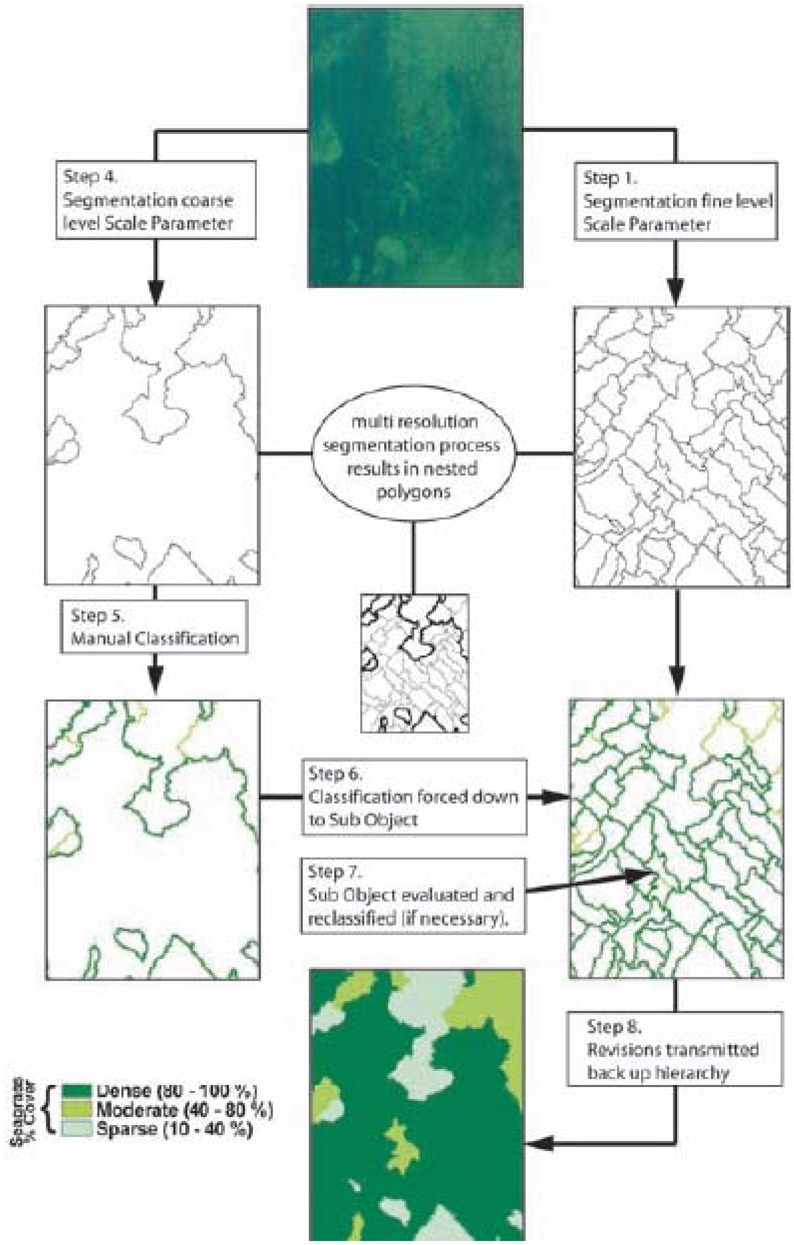

Figure 9. Process steps of the 2003 and 2009 manual seagrass classification. (Used with permission from Lathrop et al. (2006)).

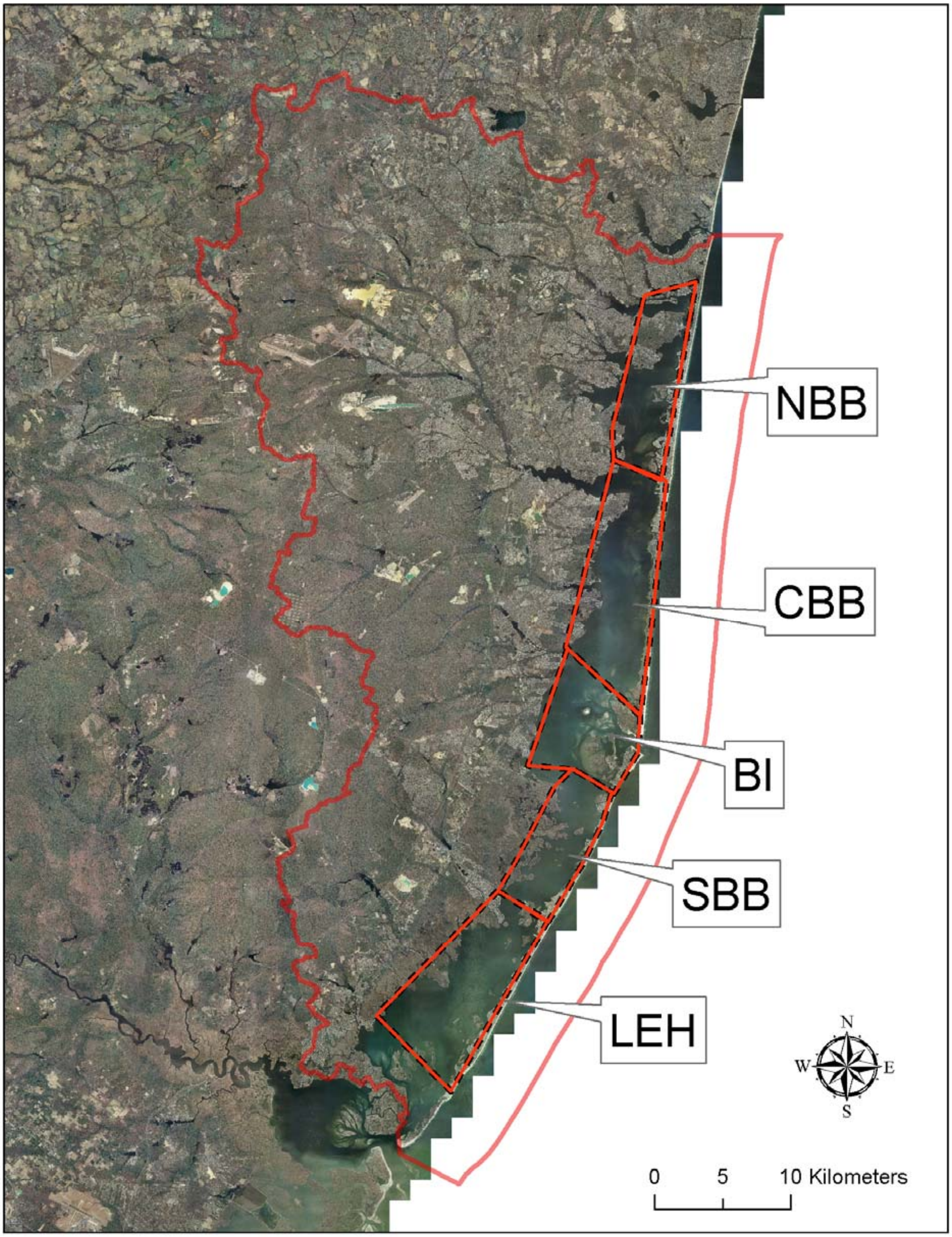

Figure 10. Location of bay segments.

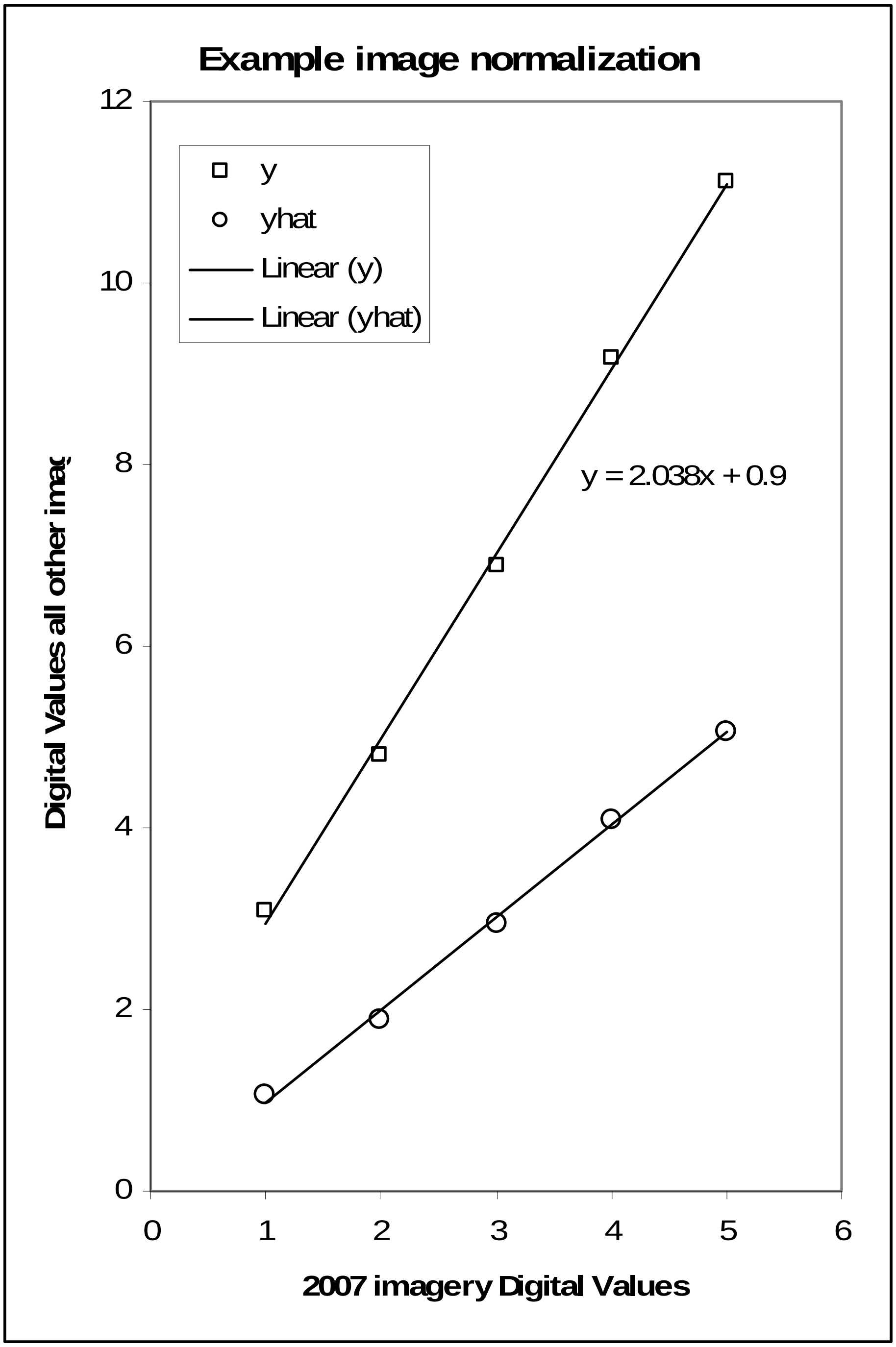

Figure 11. Graphical example of the Pseudo Invariant Feature (PIF) image normalization for the aerial photographic datasets. The top line shows a hypothetical linear relationship between the two image pairs. The bottom line shows the linear relationship between the image pairs after the y image is normalized to the x image.

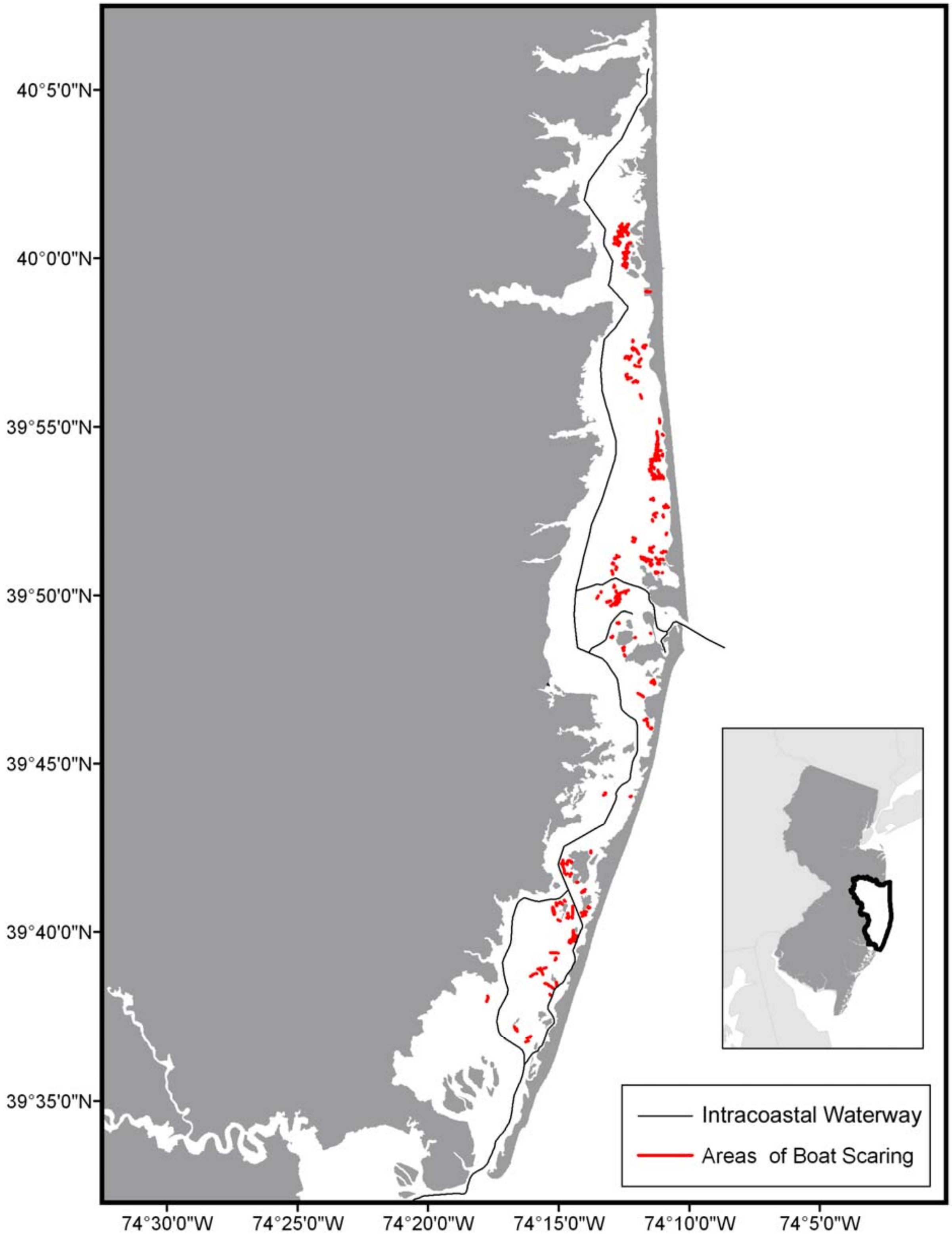

Figure 12. Areas of boat scarring manually delineated in the 2009 aerial photography.

Figure 13. Image of large boat scar adjacent to Tice’s Shoal to the west of Island Beach State Park in Central Barnegat Bay (CBB).

Figure 14. Calculation of bay depth using band ratios for the 2007 imagery.

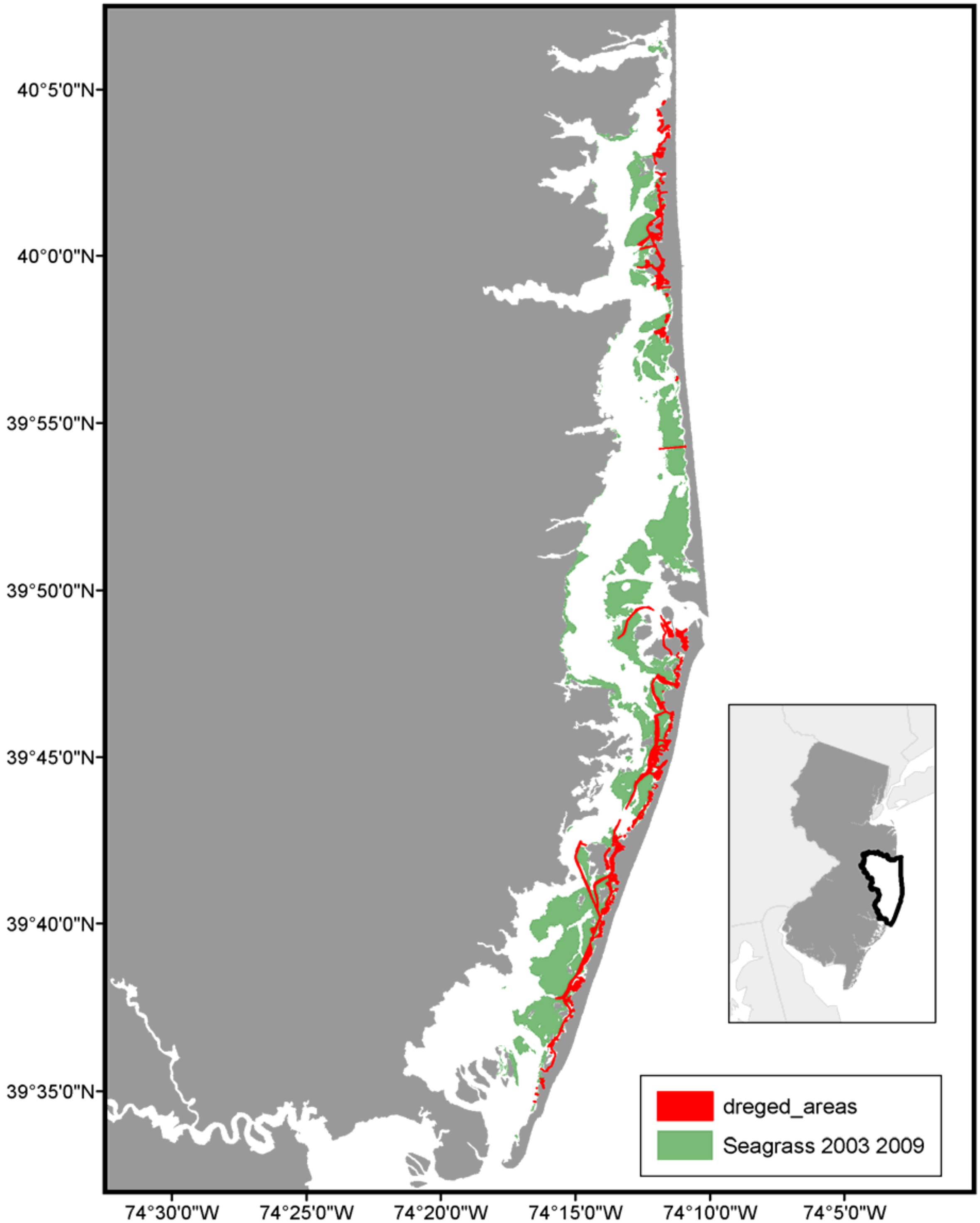

Figure 15. Location of dredging sites on the eastern side of the Barnegat Bay-Little Egg Harbor estuary. Determined through manual interpretation of 2003 and 2009 aerial photography.

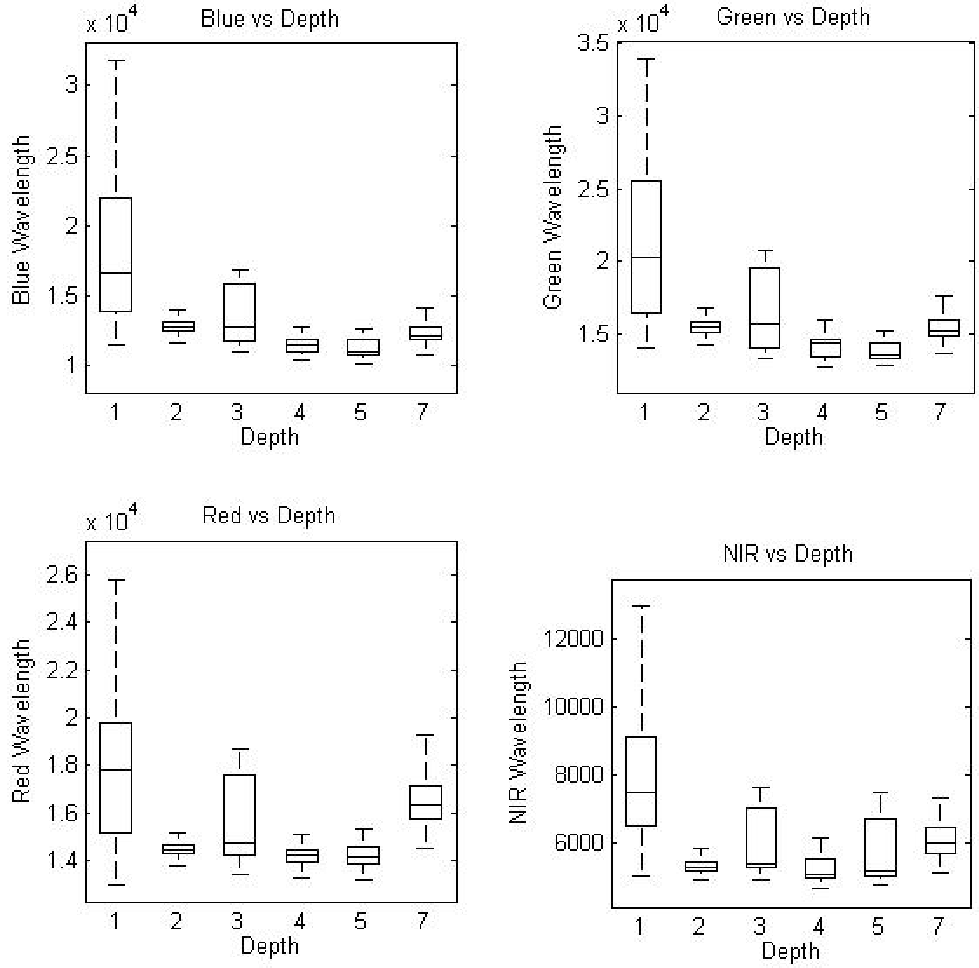

Figure 16. The responses of the March 2007 aerial photography by band vs mean low low tide water depth.

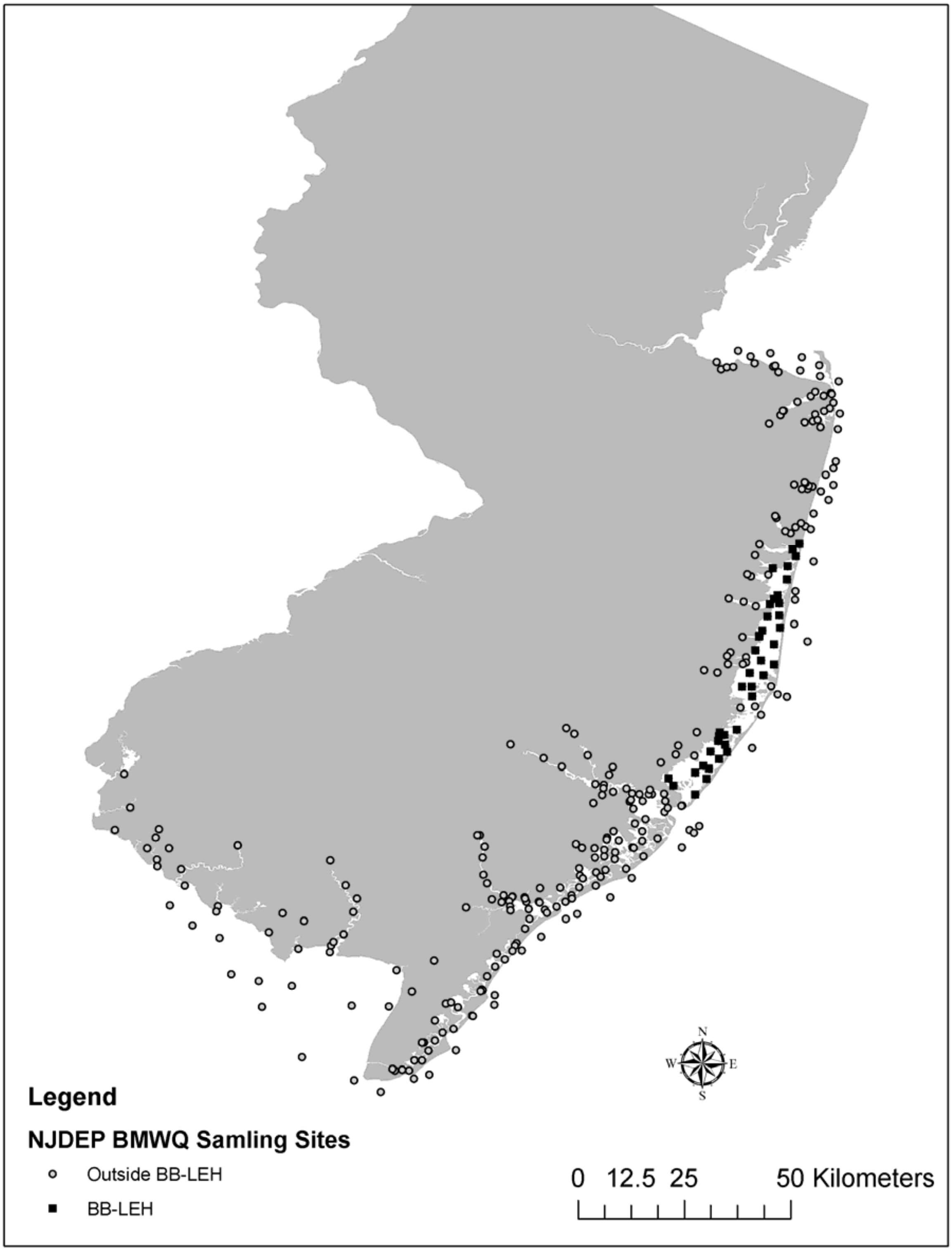

Figure 17. Location of the New Jersey Department of Environmental Protection Bureau of Marine Water Quality sampling stations in the New Jersey waters and within the Barnegat Bay-Little Egg Harbor estuary.

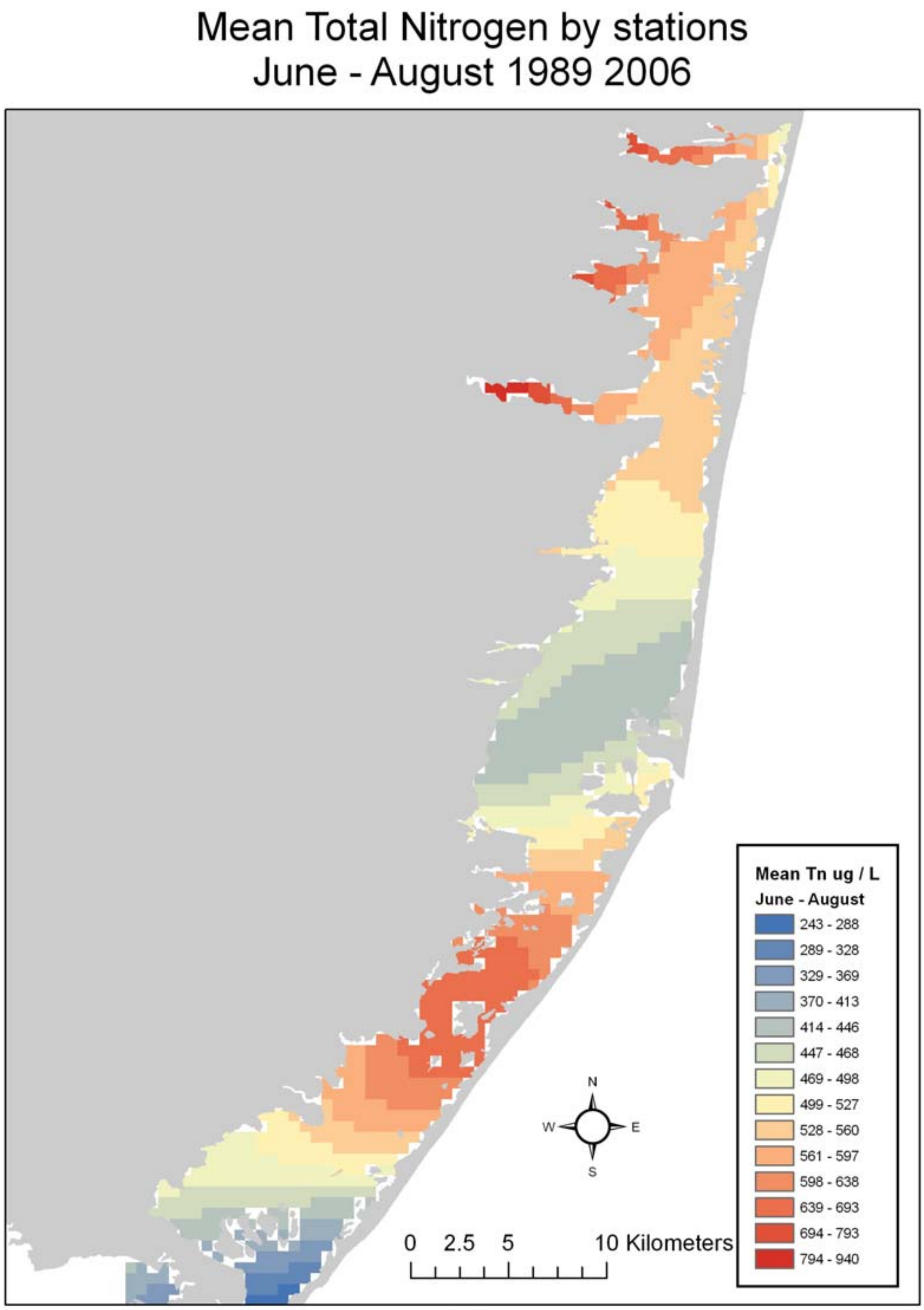

Figure 18. Gridded mean total nitrogen in the Barnegat Bay-Little Egg Harbor estuary from June-August between 1998-2006. This graphic shows two hotspots of total nitrogen within the BB-LEH estuary system.

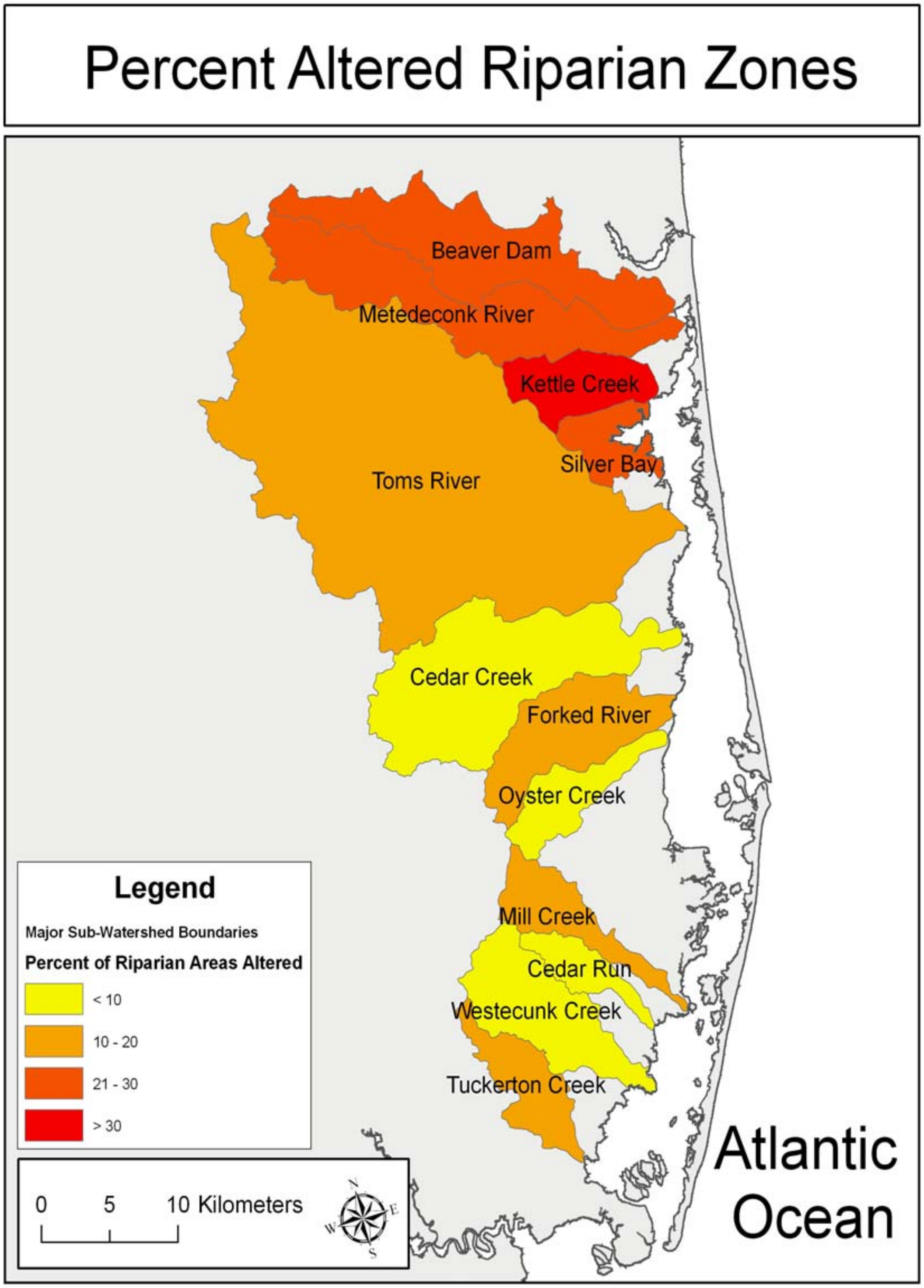

Figure 19. Location of subwatersheds within the Barnegat Bay-Little Egg Harbor watershed including percent altered riparian zones (used with permission from Lathrop and Haag, 2006).

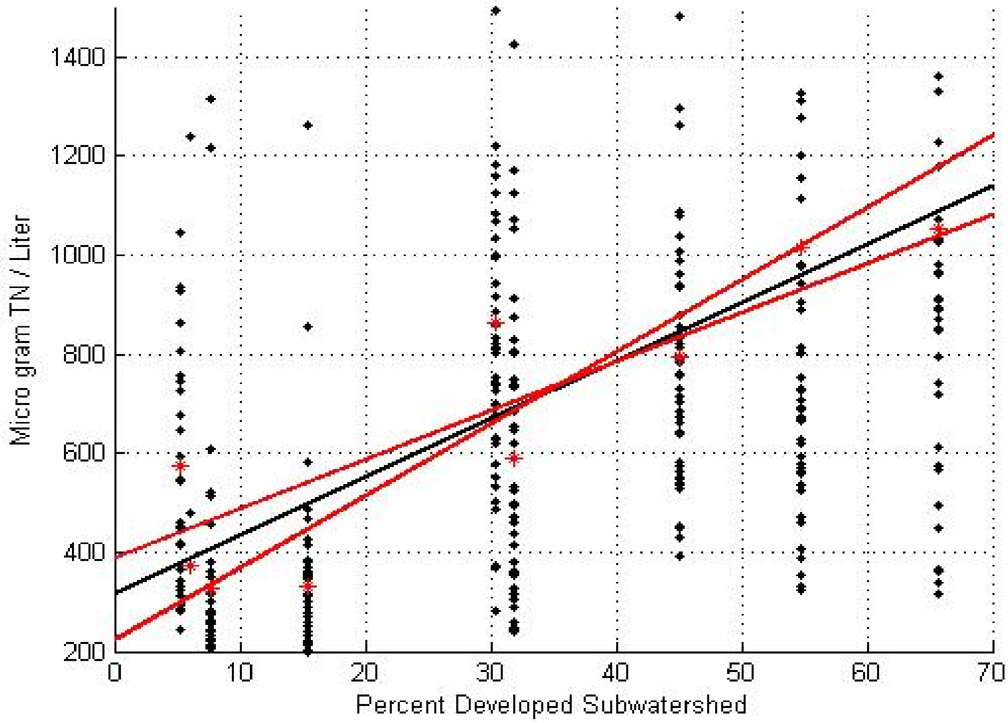

Figure 20. Percent developed subwatershed vs stream water total nitrogen values. Black line is the linear regression line. The red lines are the 95 percent confidence intervals. The red dots are the mean for each station, and the black dots are the full dataset.

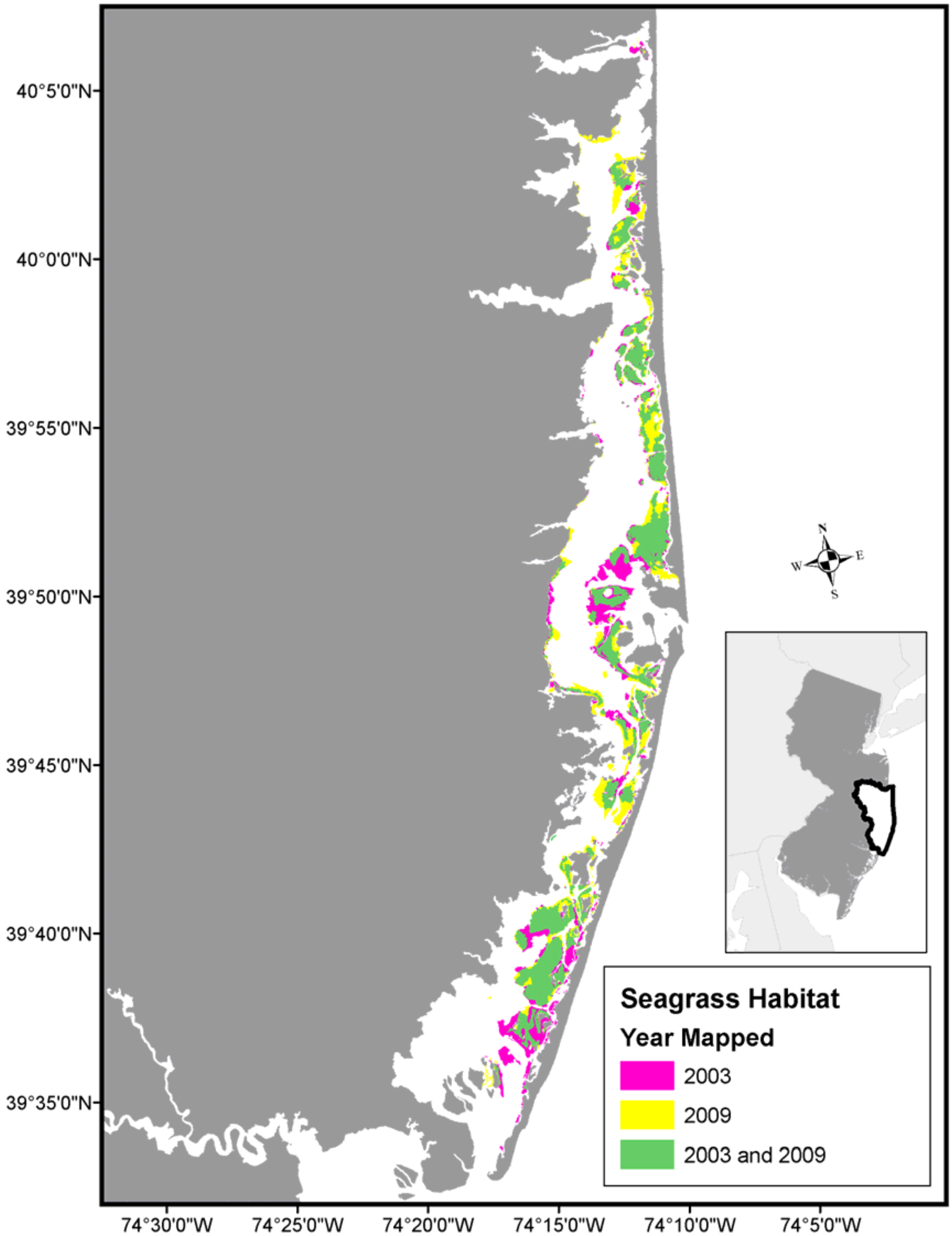

Figure 21. Distribution of seagrass mapped during the 2003 and 2009 surveys across the BB-LEH estuary system.

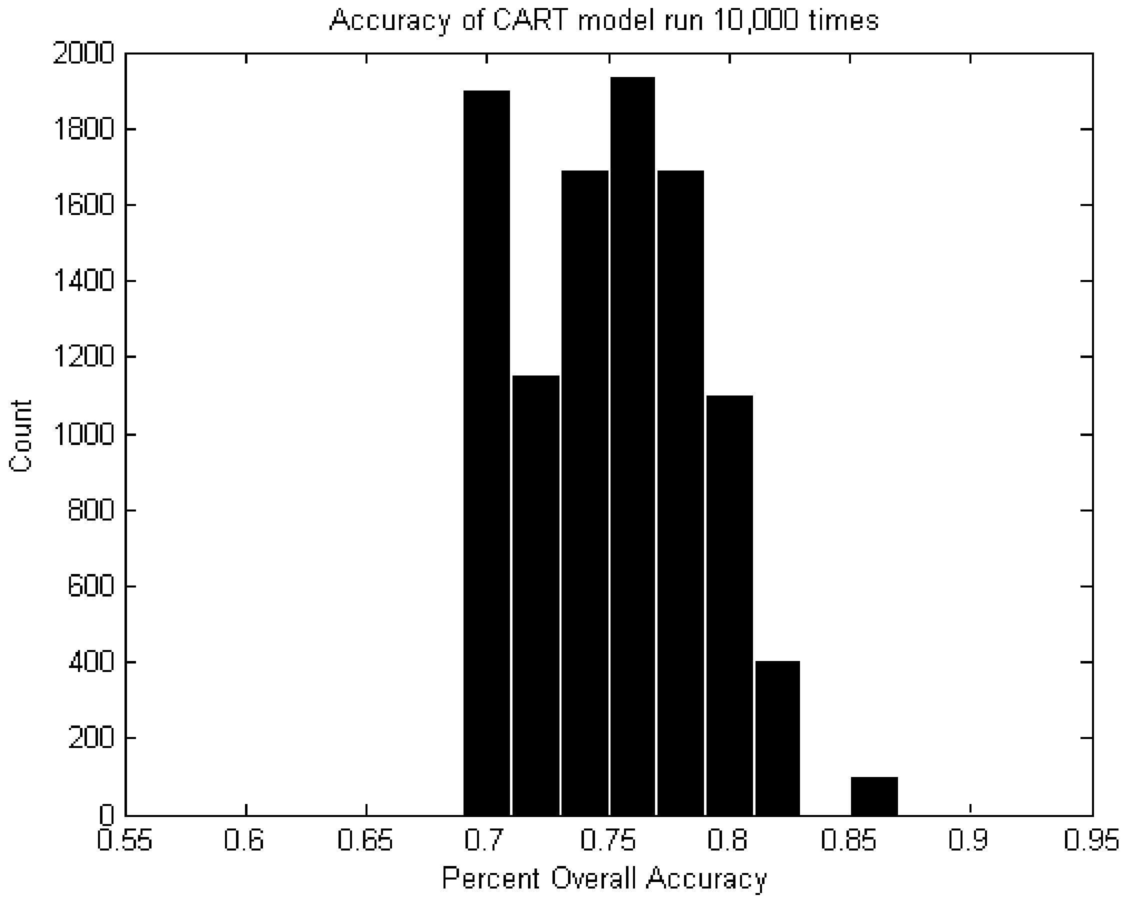

Figure 22. Overall accuracy of the CART model vs *in situ* data for the Barnegat Bay-Little Egg Harbor estuary.

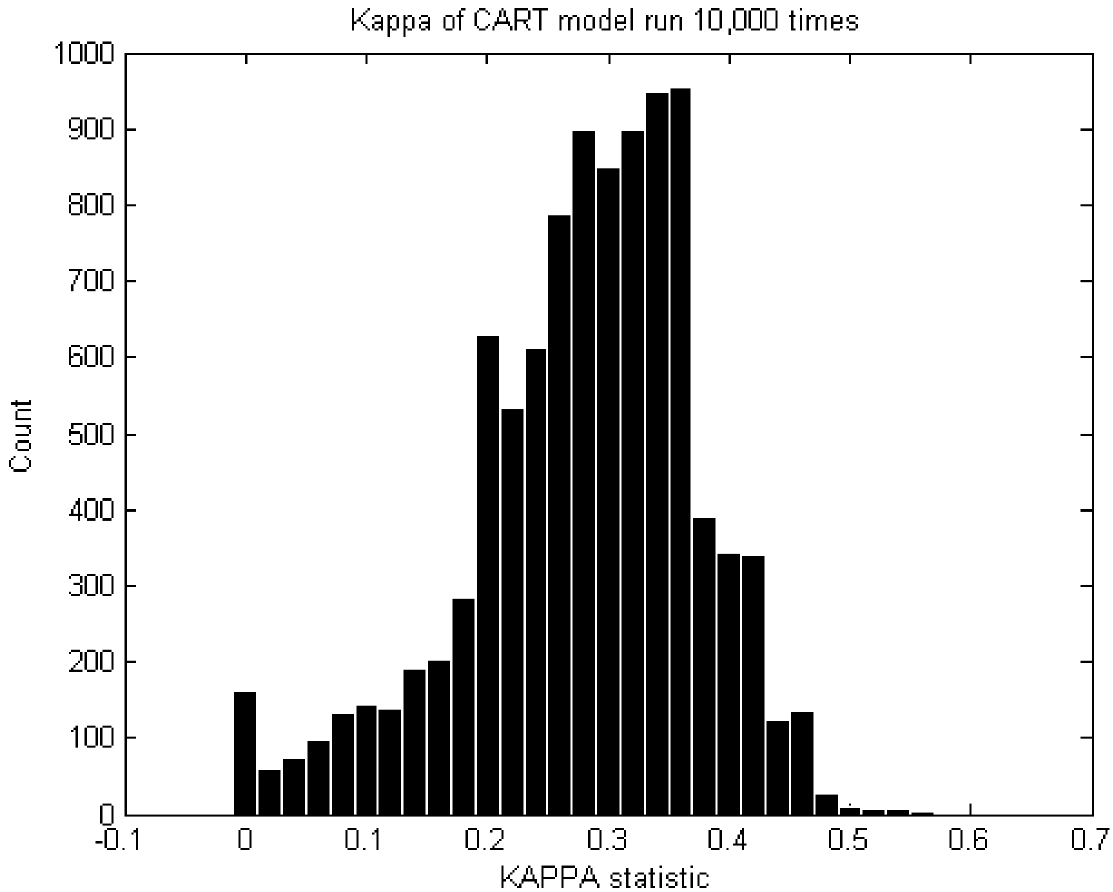

Figure 23. Kappa statistic for the CART model created using the 2009 *in situ* dataset.

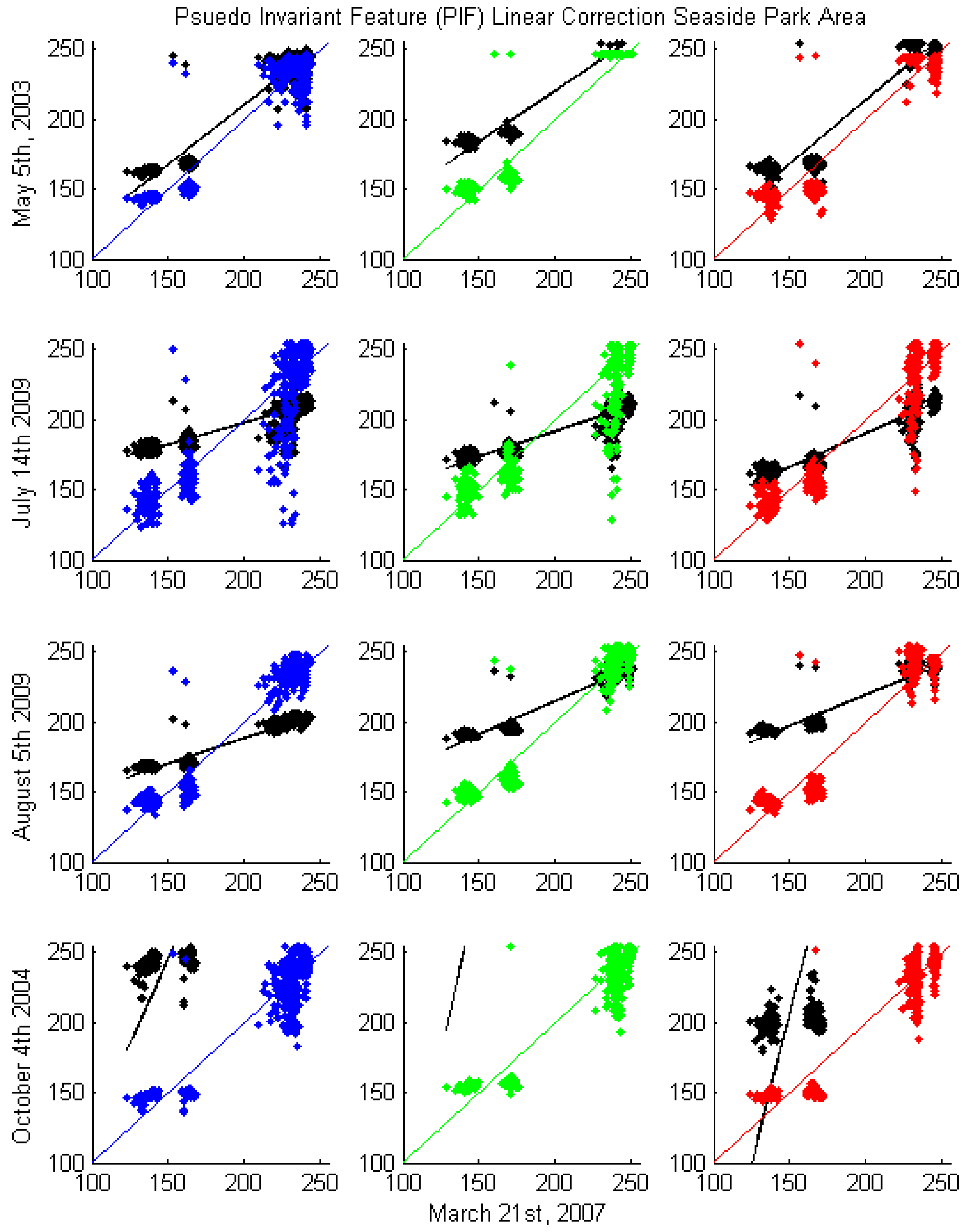

Figure 24. Imagery digital number correction for image pairs for July 14th, 2009 August 5th, 2006, and October 4th, 2004 vs March 21st, 2007 imagery. Corrected imagery shown in Blue, Green, and Red vs the original data in Black.

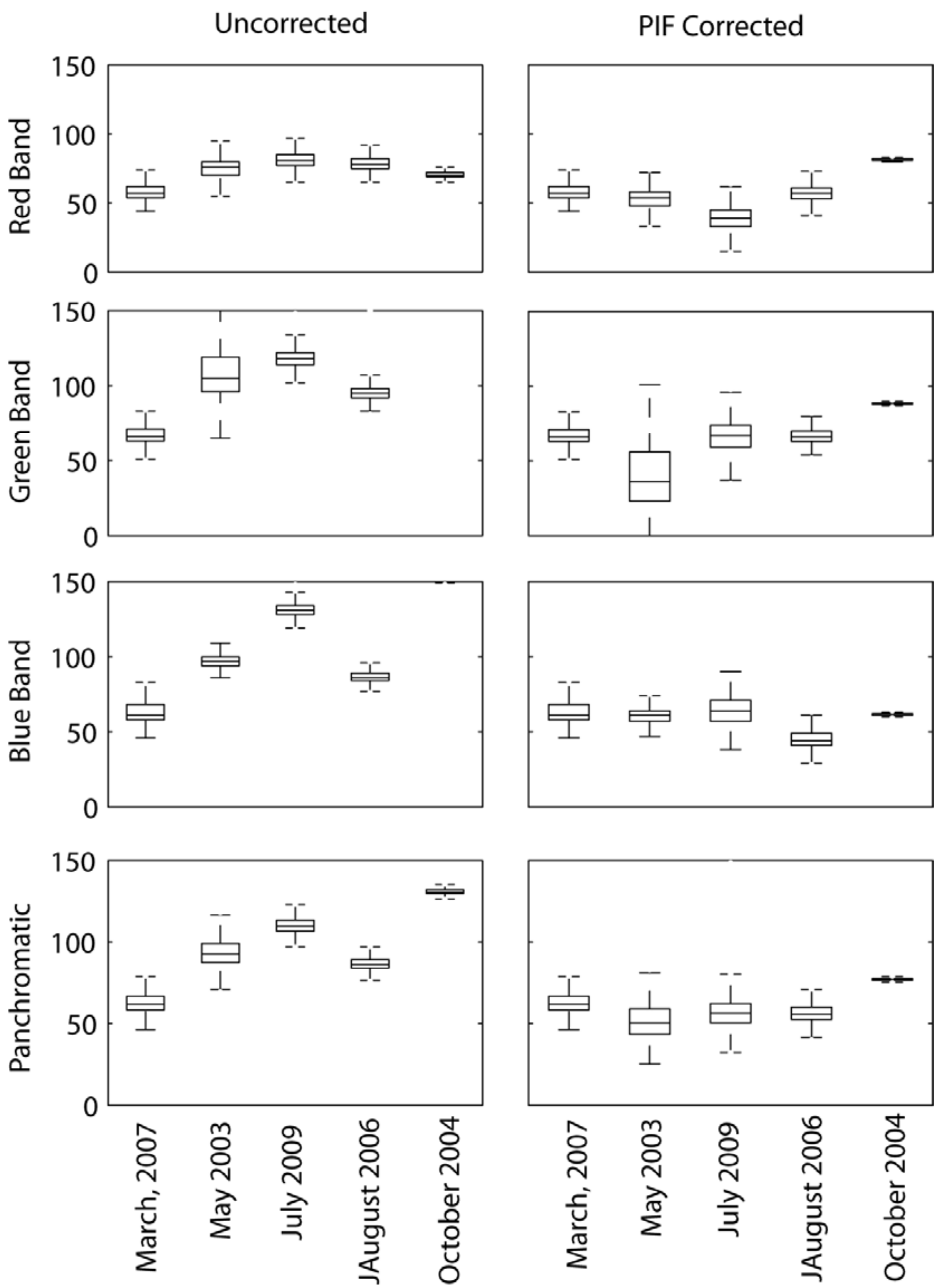

Figure 25. Boxplot showing the difference between corrected and raw imagery for Seaside Park, Barnegat Bay.

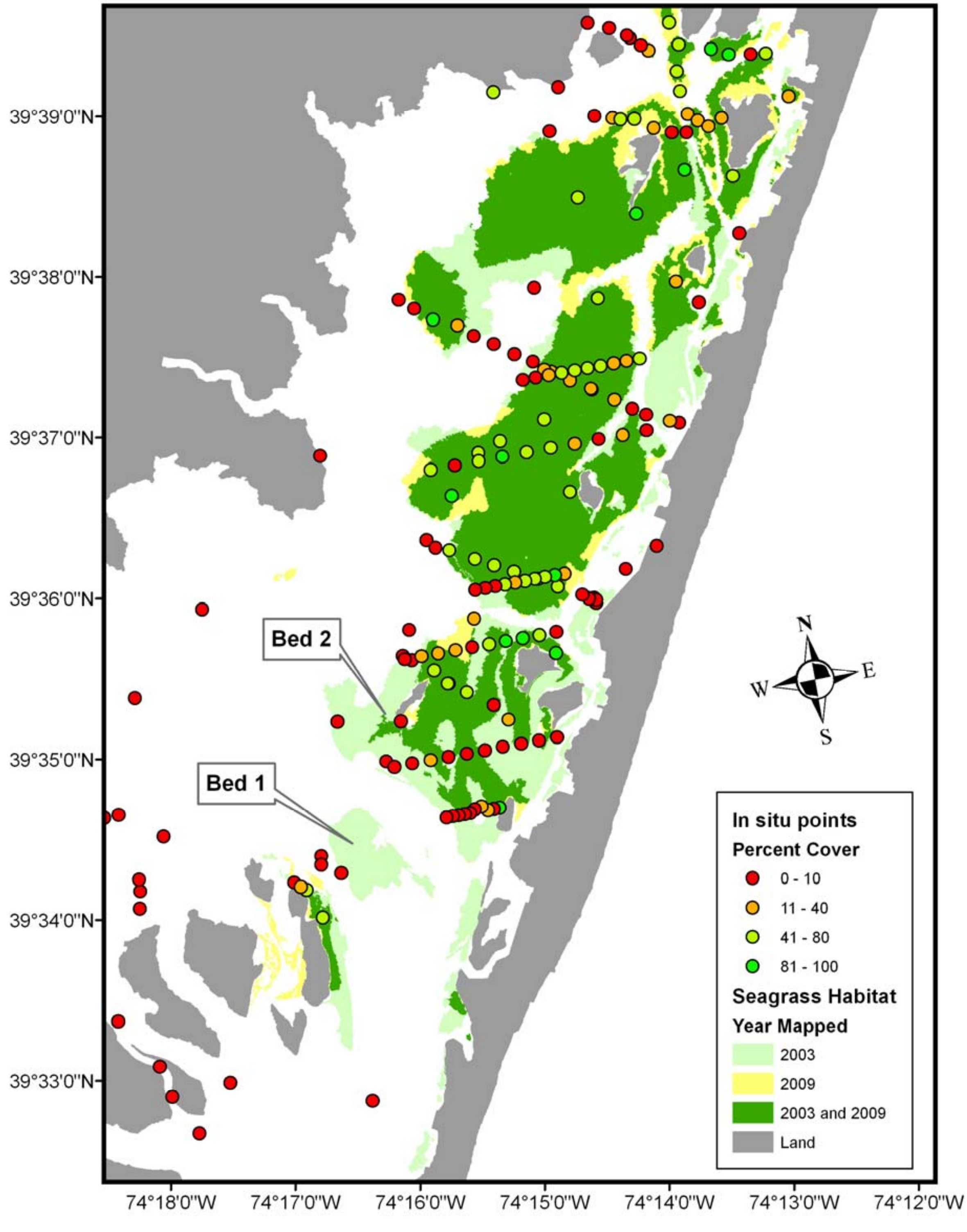

Figure 26. Seagrass habitat mapped in Little Egg Harbor during 2003 and 2009 with the 2009 *in situ* data points.

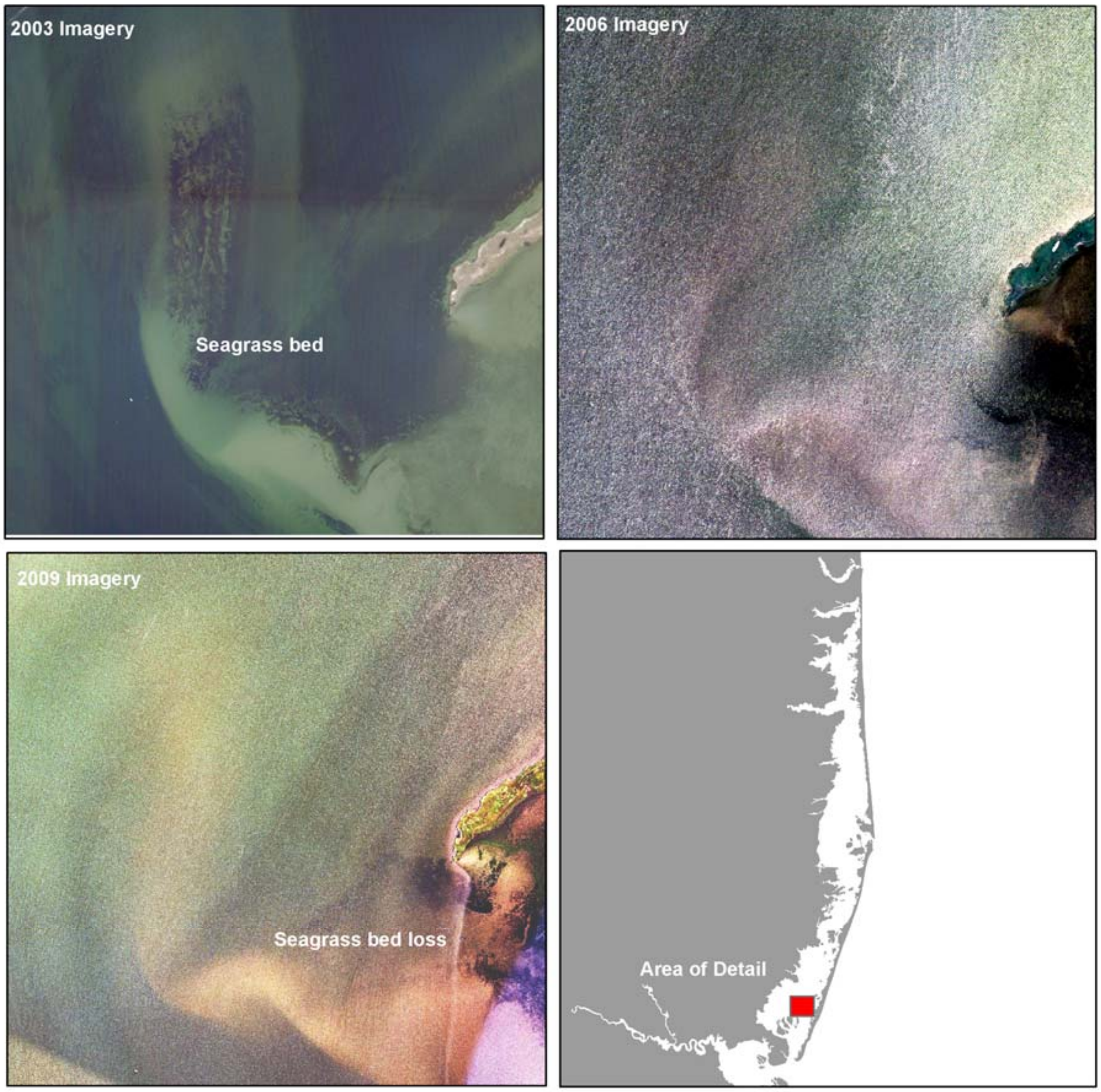

Figure 27. Aerial images showing changes in seagrass habitat in Little Egg Harbor between 2003 and 2009. The seagrass bed show in the 2003 imager (top left) is not found in the 2009 imager (bottom left). The 2006 imagery (top right) is difficult to interpret due to surface reflectance.

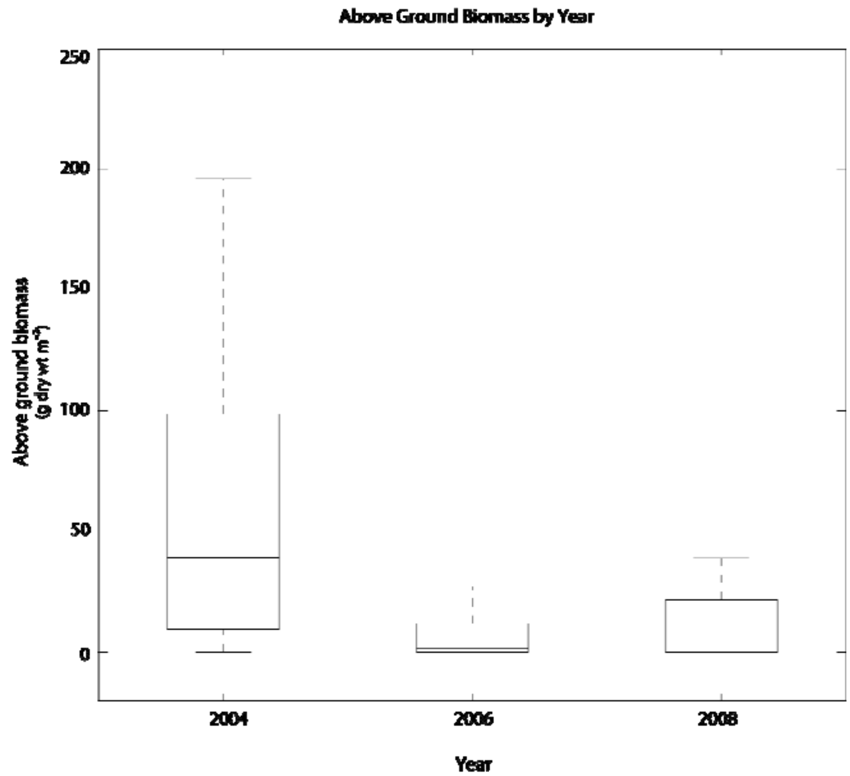

Figure 28. Boxplot showing the percent cover of *Z. marina* for 2004, 2006, 2008 in Little Egg Harbor. (Used with permission from Kennish et al, (2008))

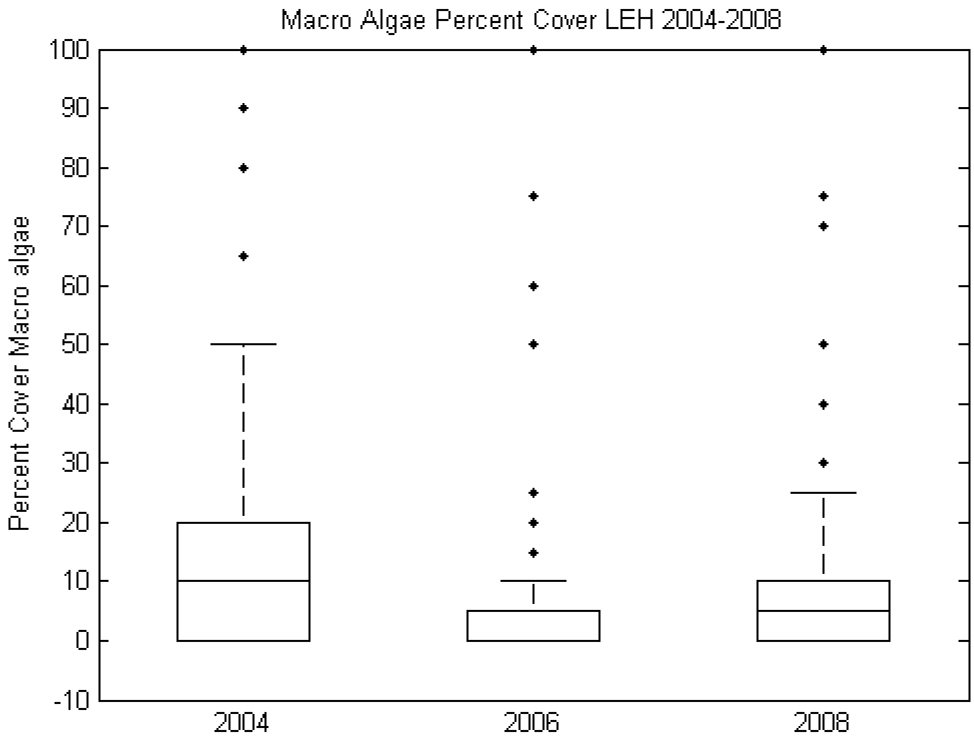

Figure 29. Boxplot showing the percent cover of Macroalgae within the Barnegat Bay-Little Egg Harbor estuary for 2004, 2006, and 2008.

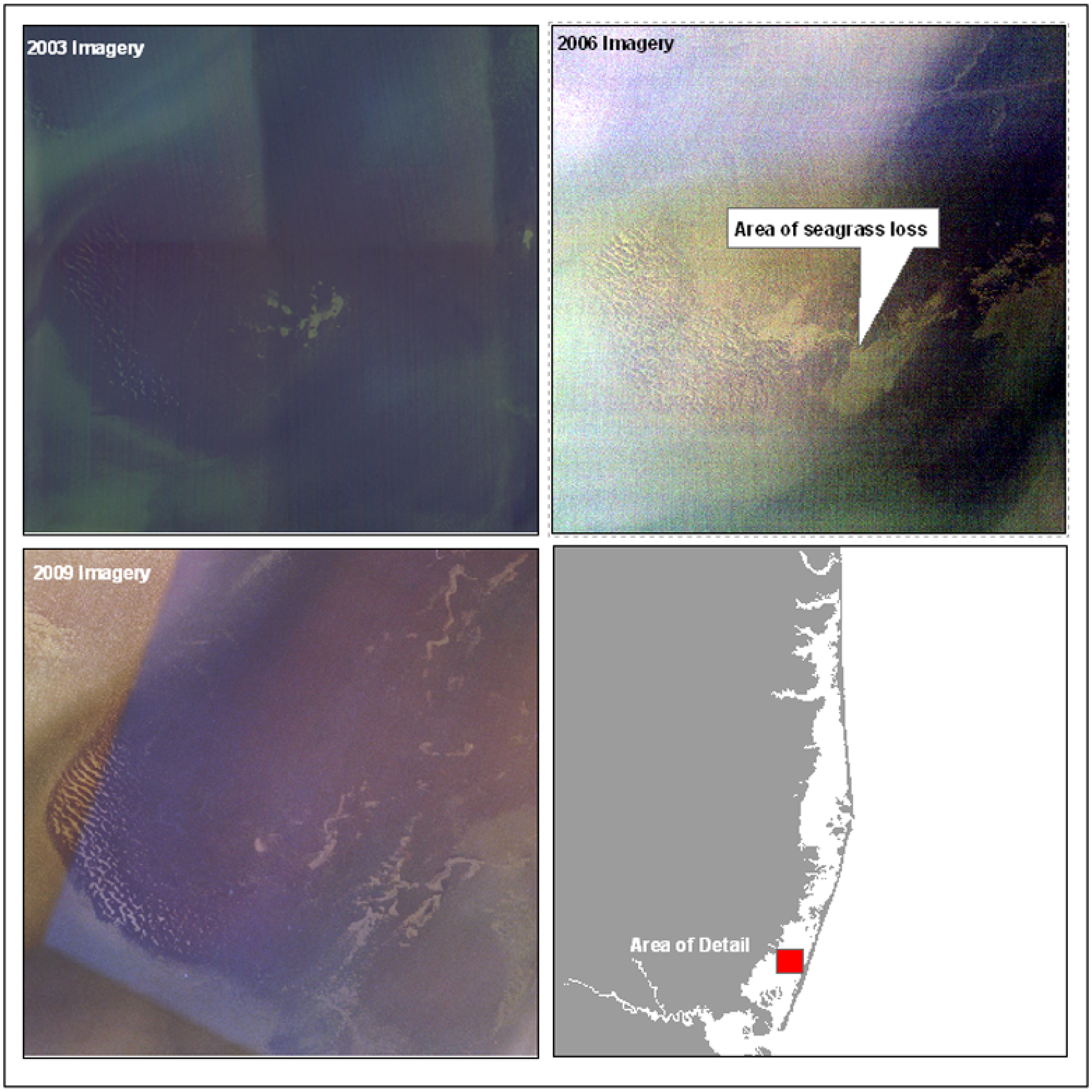

Figure 30. May 2003, July 2006, and August 2009 imagery showing a seagrass bed in 2003, dieback in 2006, and subsequent expansion in 2009. The 2003 imagery (top left) and the 2009 imagery (bottom left) show as thick seagrass bed while the 2006 imager (top right) shows several large denuded areas within the larger seagrass bed.

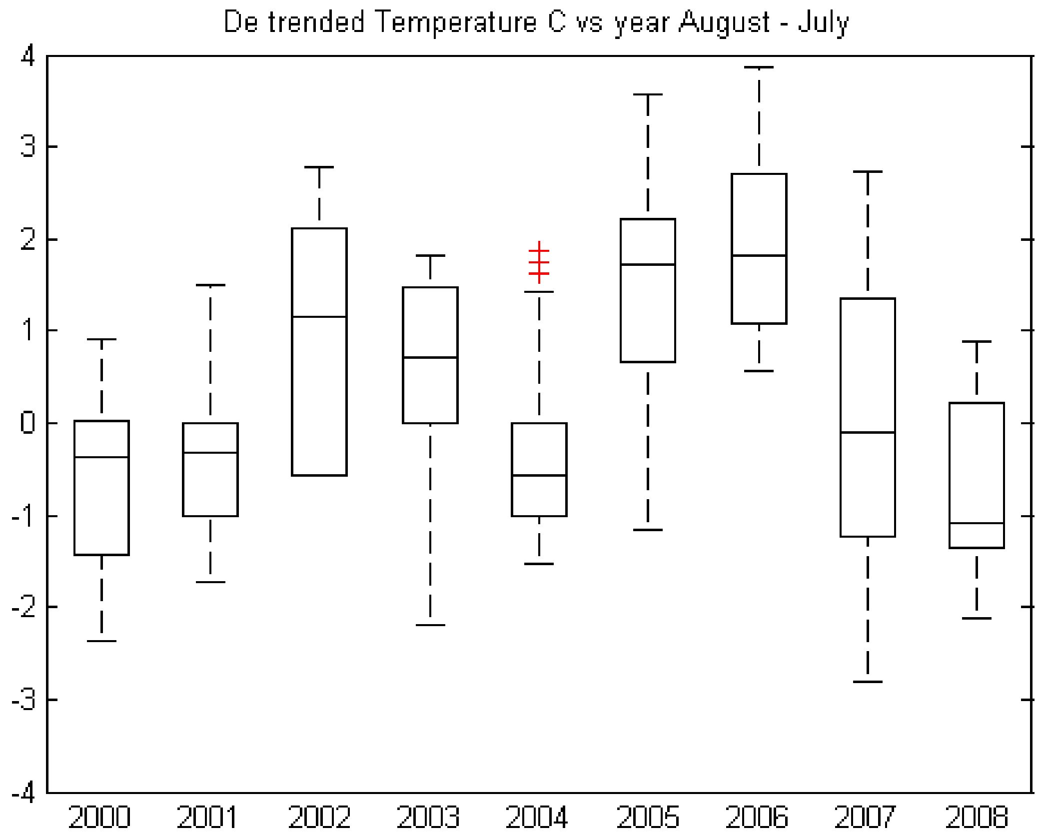

Figure 31. Detrended Temperature C for the Barnegat Bay-Little Egg Harbor estuary for July - August by year 2000-2008 (no temperature data exists in the NJ-BMWQ data for 2009).

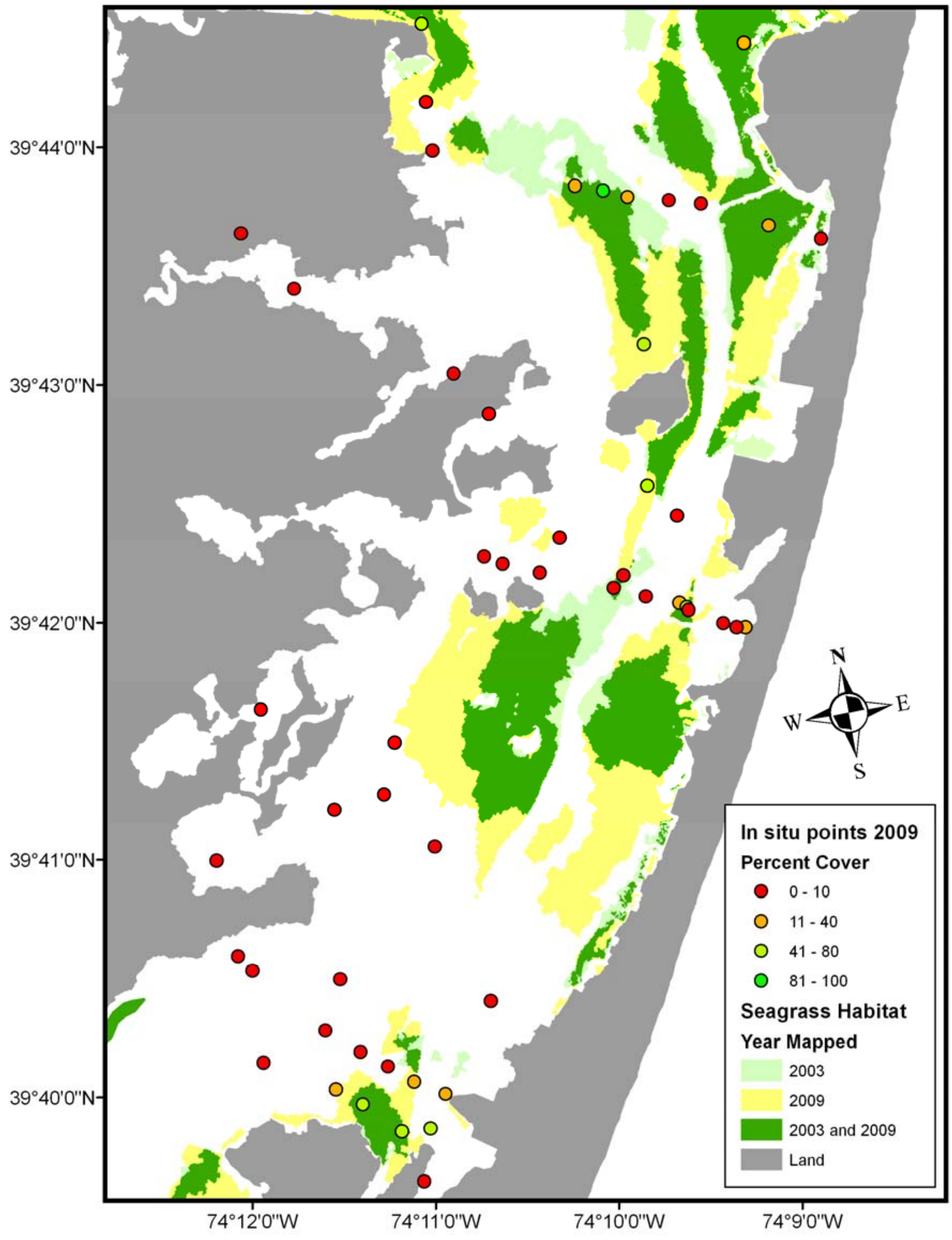

Figure 32. Seagrass habitat mapped in southern Barnegat Bay during 2003 and 2009. The 2009 *in situ* sites are show over top of the study area.

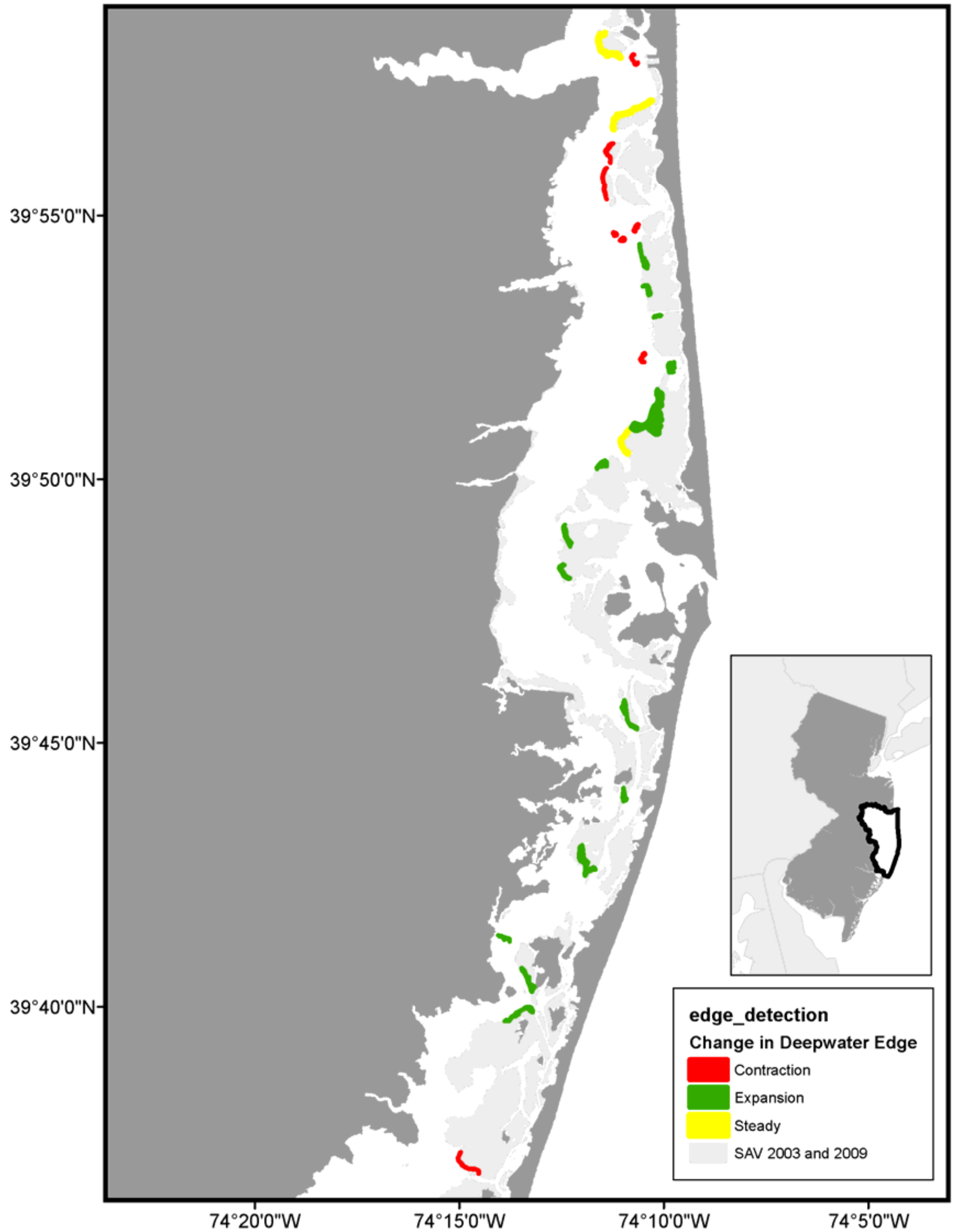

Figure 33. Change in seagrass edge in the Barnegat Bay-Little Egg Harbor estuary between the 2003 and 2009 surveys.

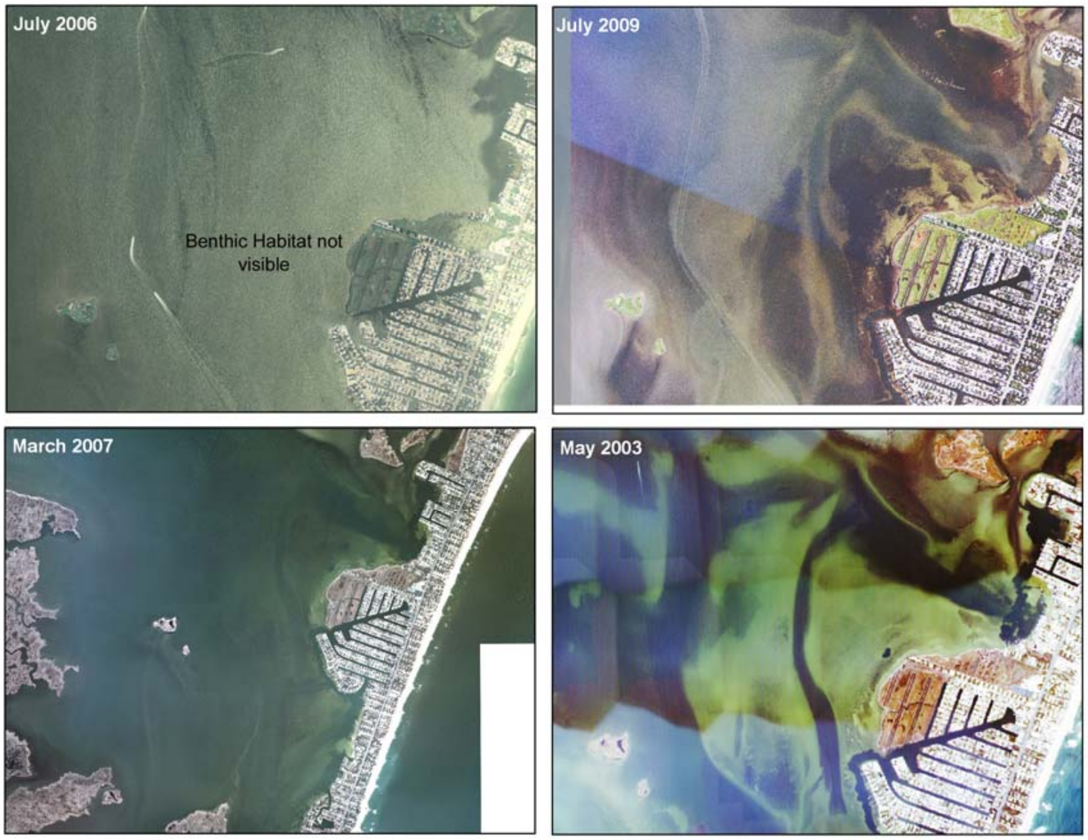

Figure 34. Comparison of the 2003, 2006, 2007, and 2009 imagery for Central Barnegat Bay (CBB). This comparison shows that the 2006 imagery (top left) does not provide information on the benthic habitat because of high surface reflectance. The 2007 imagery (bottom left) was collected in March before peak seagrass biomass.

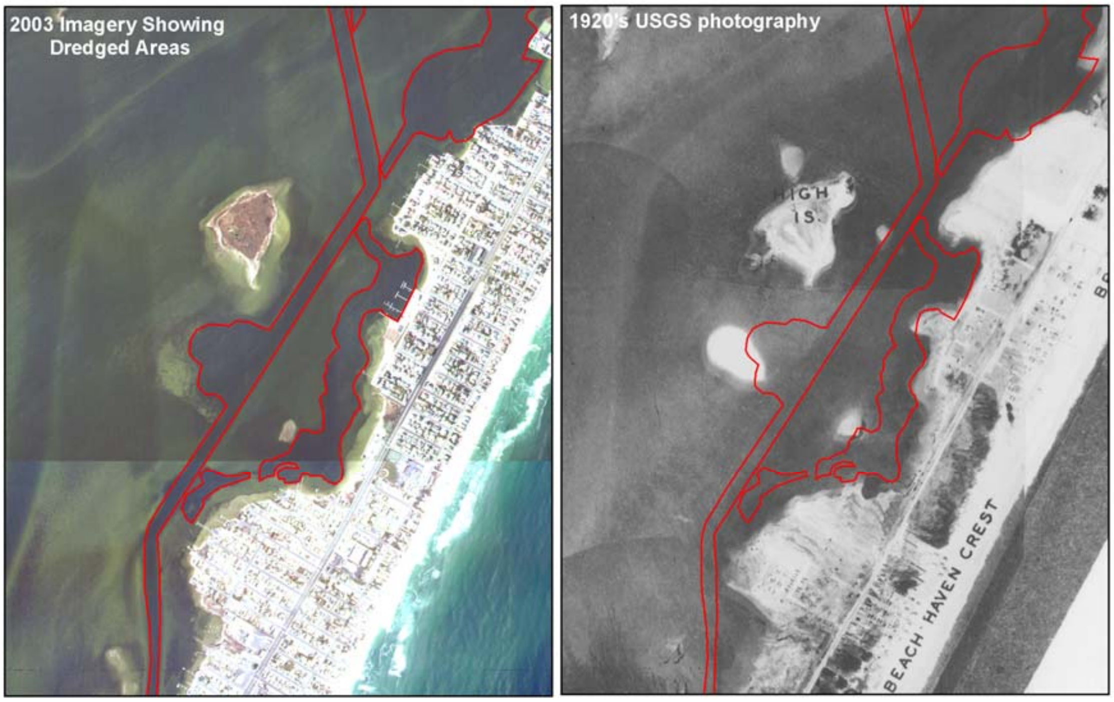

Figure 35. Dredged areas red in 2003 and the 1920's on the eastern side of Long Beach Island.

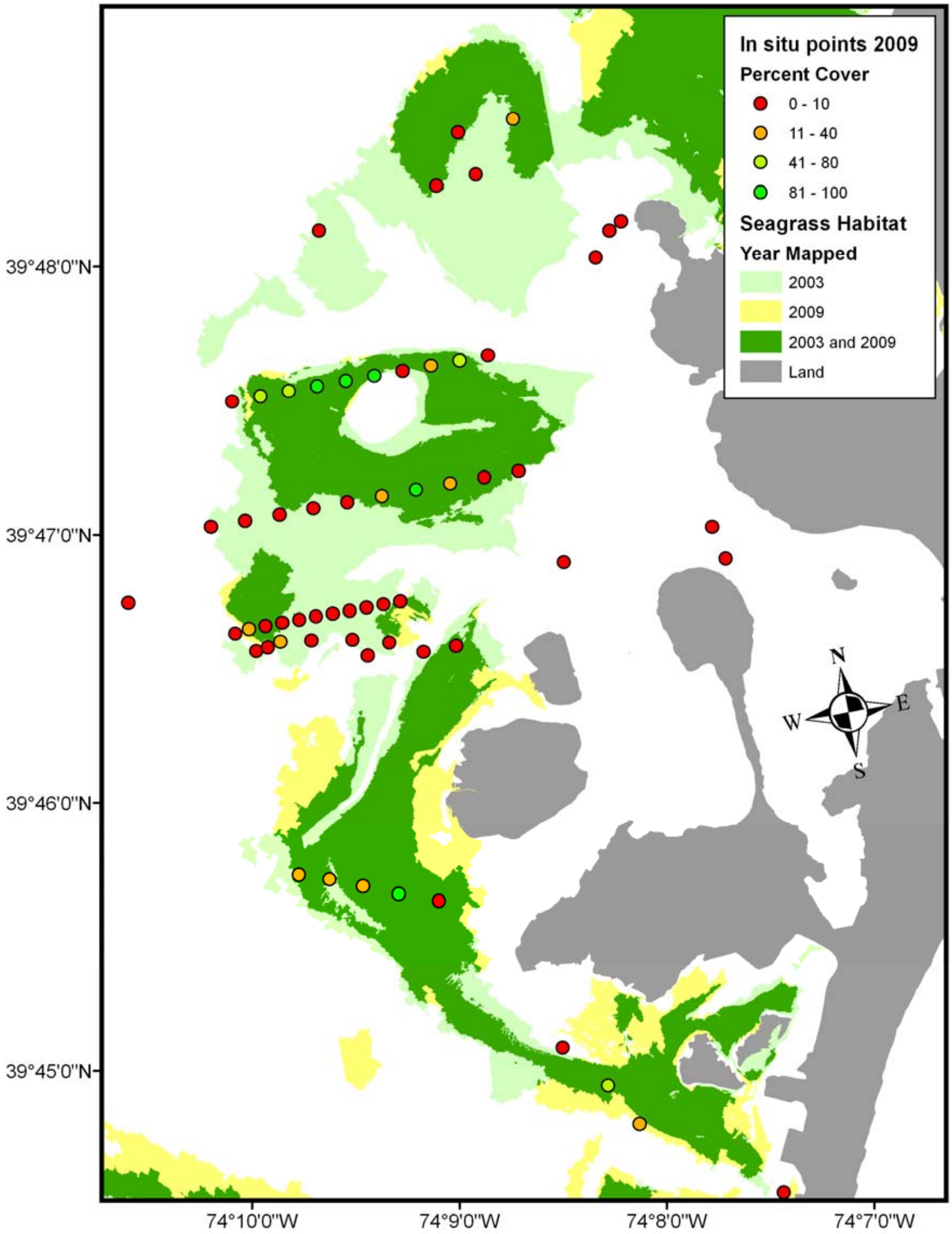

Figure 36. Seagrass habitat mapped at Barnegat Inlet during 2003 and 2009 with the 2009 *in situ* sites.

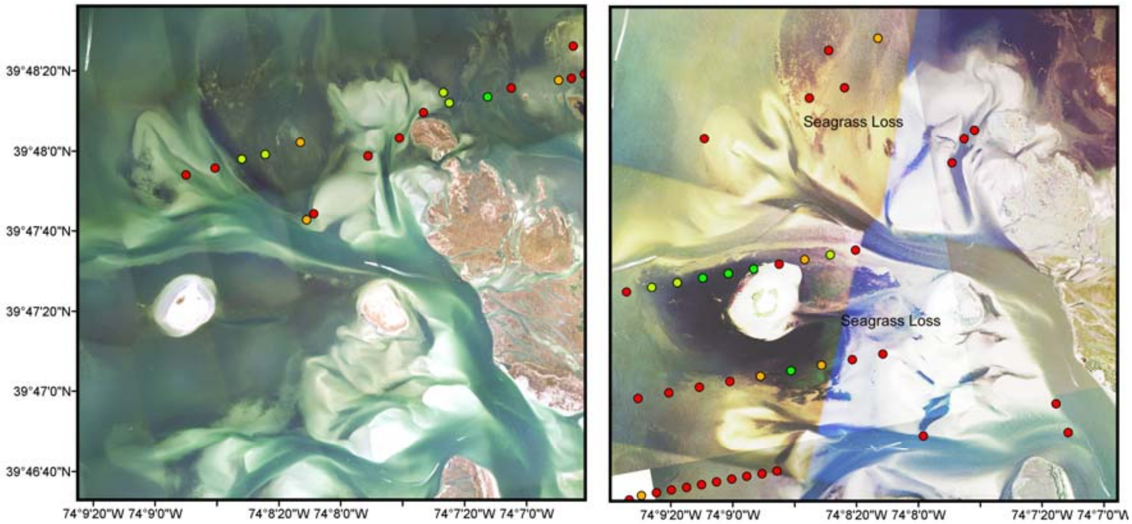

Figure 37. Imagery and *in situ* sites in 2003 (left) and 2009 (right) showing seagrass habitat decline at Barnegat Inlet.

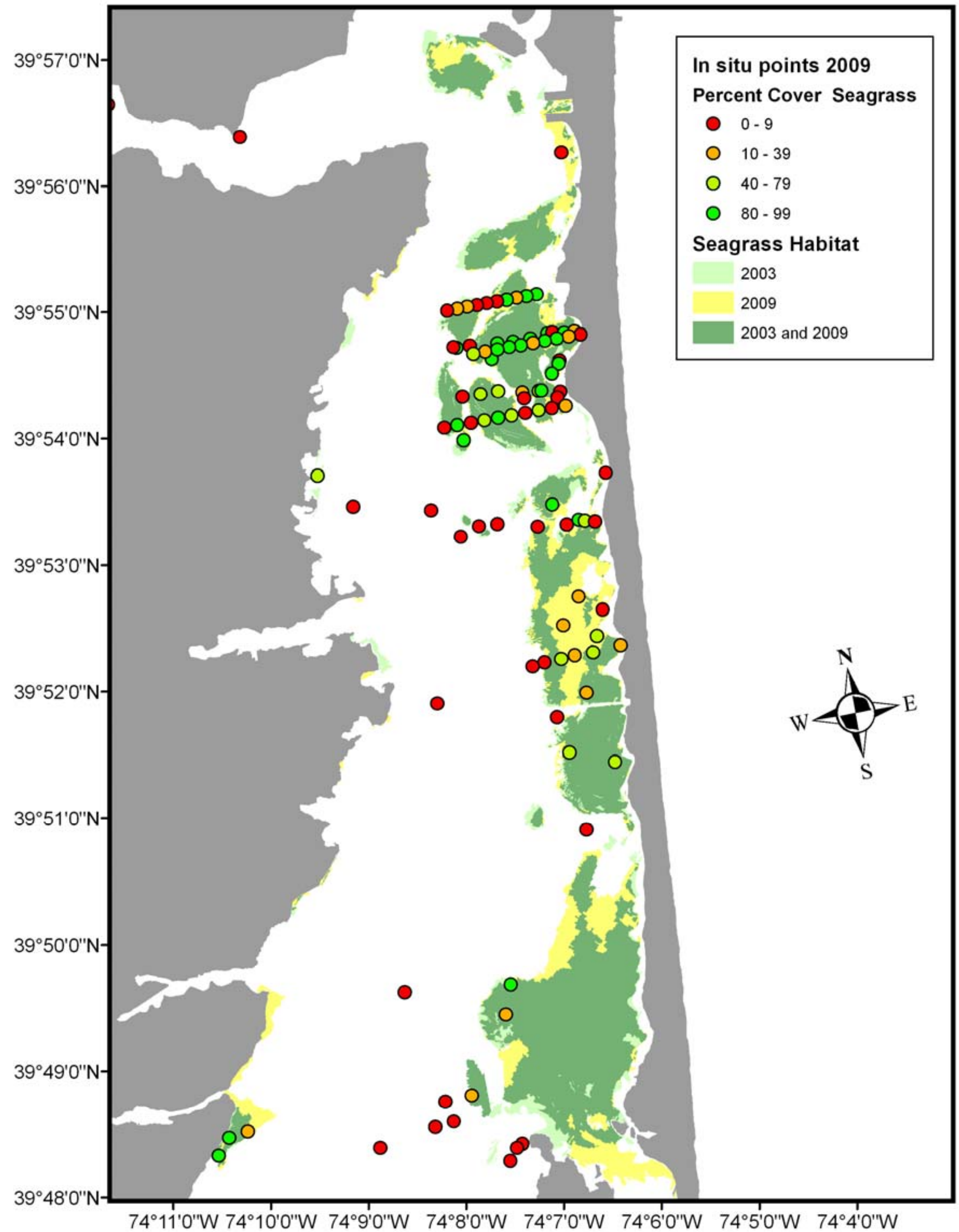

Figure 38. Seagrass habitat mapped in central Barnegat Bay during 2003 and 2009 with the 2009 *in situ* sites.

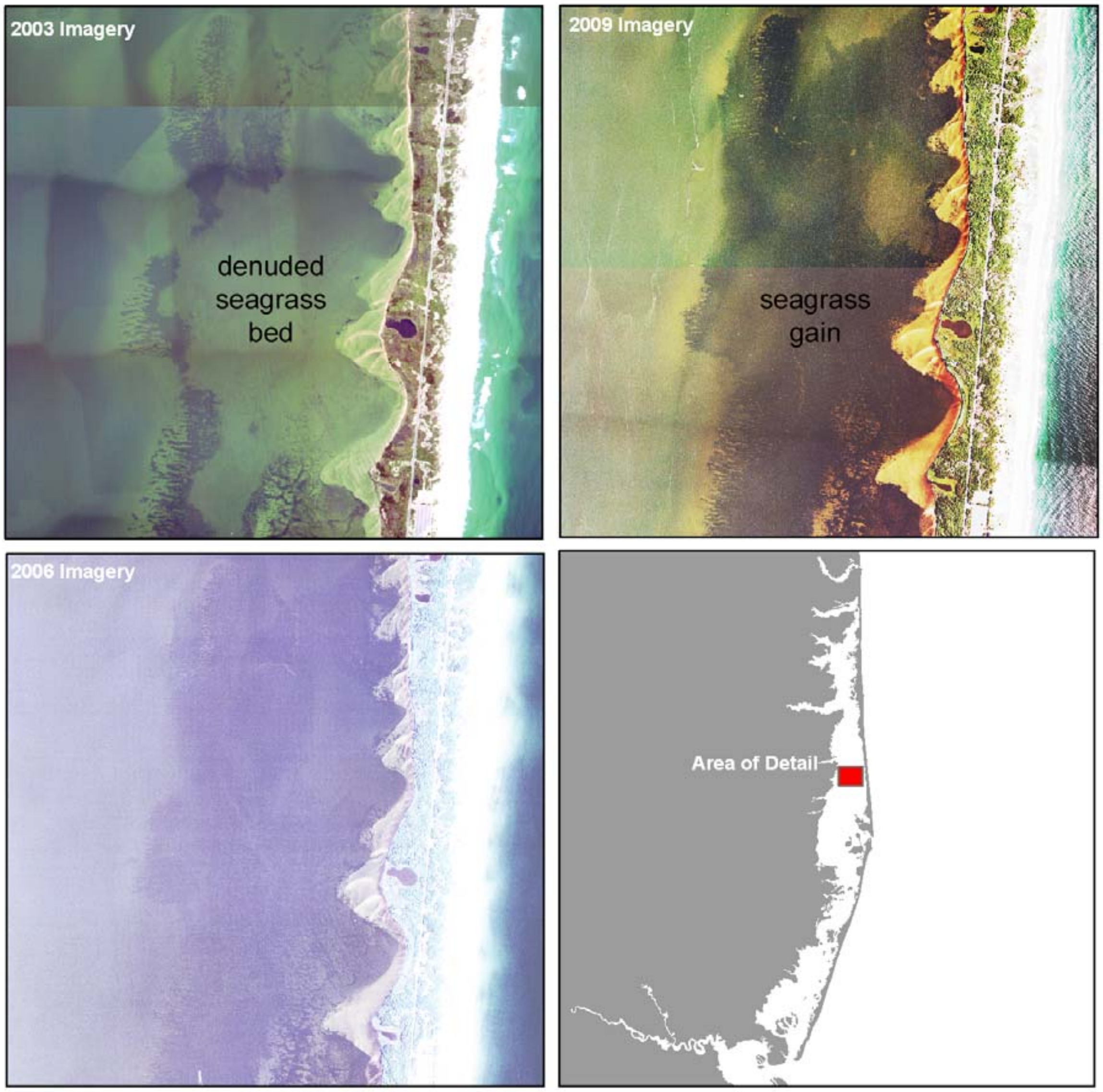

Figure 39. Imagery in 2003 (top left), 2006 (bottom left), and 2009 (top right) showing seagrass habitat decline at Barnegat Inlet.

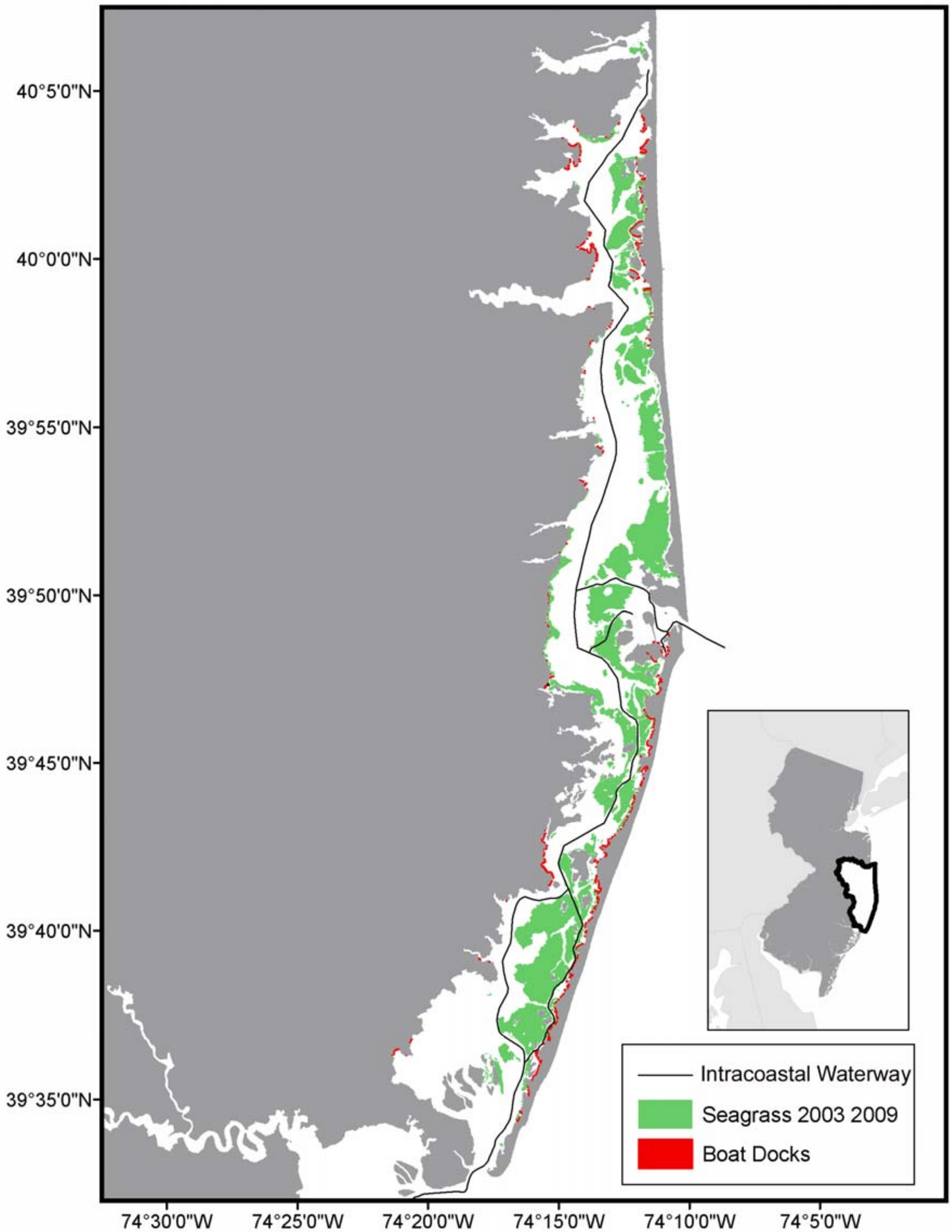

Figure 40. Location of boat docks on the Barnegat Bay-Little Egg Harbor estuary manually delineated using the 2009 aerial photography.

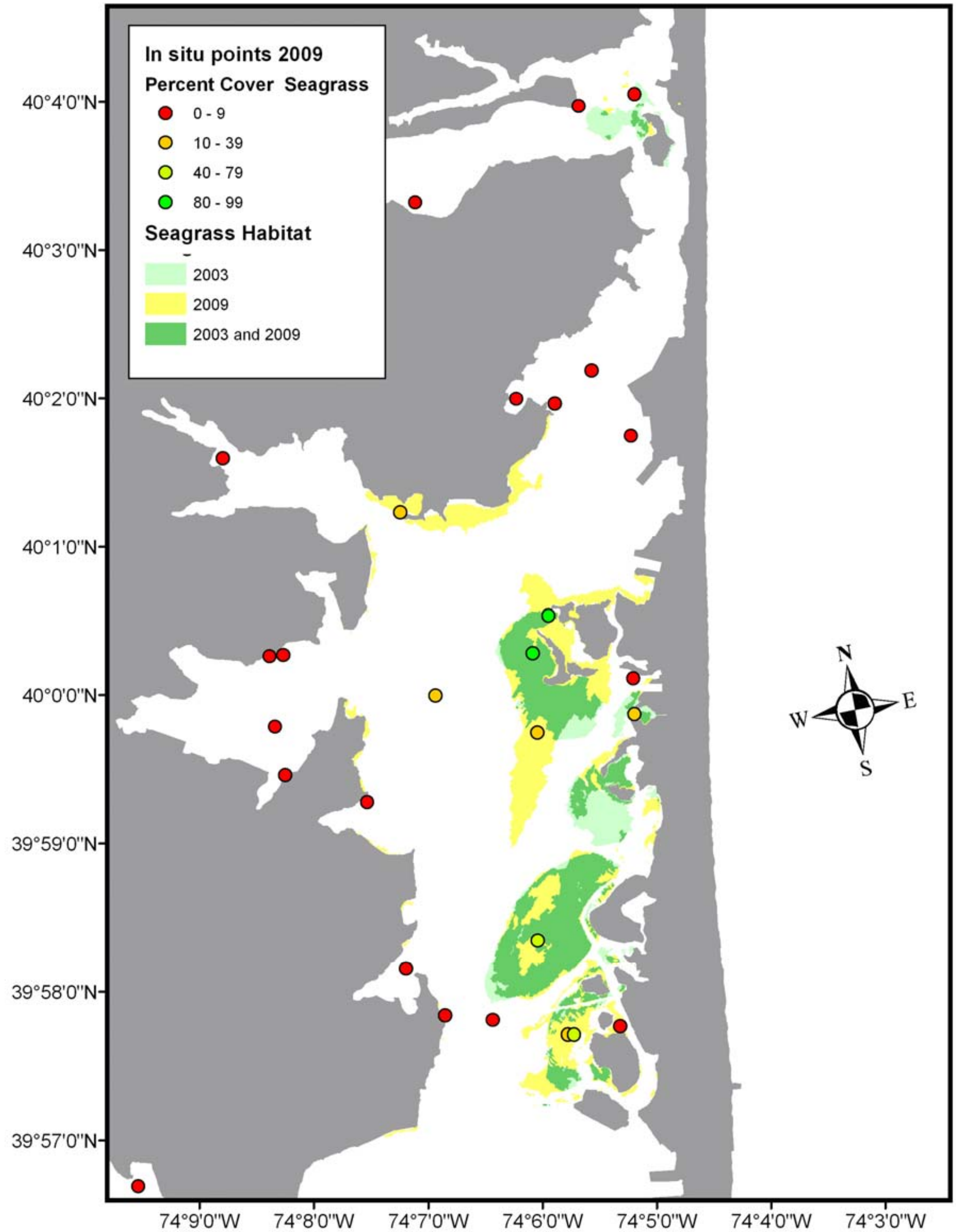

Figure 41. Seagrass habitat mapped in northern Barnegat Bay during 2003 and 2009 with the 2009 *in situ* sites.

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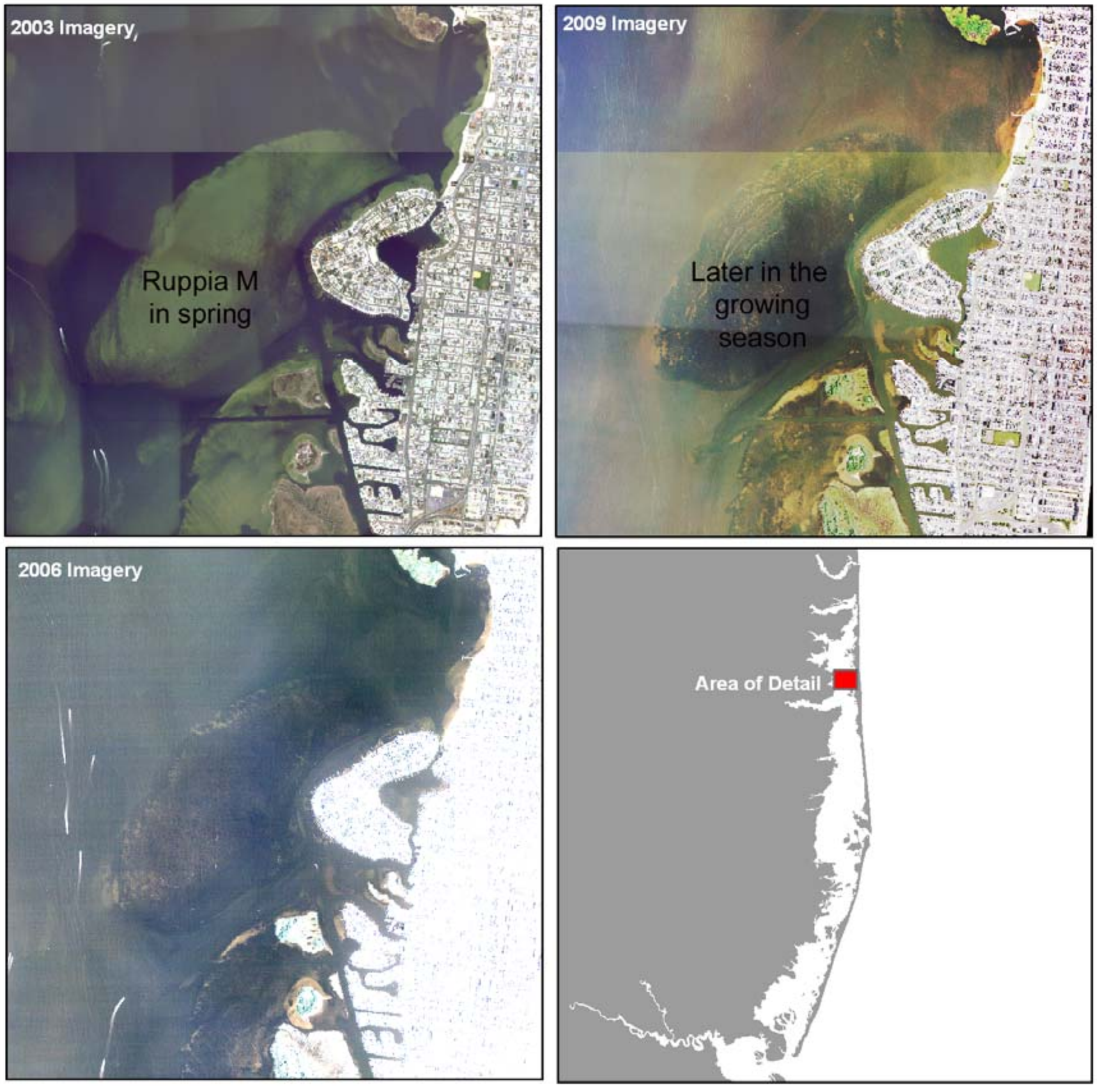

Figure 42. Imagery for March 21st 2003 (top left), August 5th, 13th 2006 (bottom left), and July 7th 2009 (top right) showing a *R. maritima* bed through the growing season.

Table 1. Ancillary photography collection data for the Barnegat Bay-Little Egg Harbor estuary.

Image Source	Collection Date	Sensor Type	Ground Resolution	Projection
CRSSA	May 4th-5th 2003	Digital	1 m	UTM
Digital Globe Quickbird	October 4th, 2004	Digital	0.7 m	UTM
USDA	July 6th, 2006	Analog	1	UTM
NJ DEP	March, 21st 2007	Analog	1 foot	NJ State Plane
USDA	August, 5th-13th 2008	Analog	1 m	UTM
CRSSA	July 14th, 2009	Analog	0.5 m	UTM

Table 2. Area of seagrass cover types mapped during the 2009 remote sensing survey.

Seagrass Type	class (ha)	Total Seagrass
Sparse (10-40%)	2,256	43%
Moderate (40-80%)	2,527	48%
Dense (80-100%)	470	9%
Total Seagrass	5,253	

Table 2a. Area of seagrass cover types mapped during the 2003 remote sensing survey.

Seagrass Type	Class (ha)	Total Seagrass
Sparse (10-40%)	1,955	38%
Moderate (40-80%)	1,093	22%
Dense (80-100%)	2,074	40%
Total Seagrass	5,184	

Table 2c. Total area of seagrass mapped during in 2003 but not in 2009 , 2009 but not in 2003, and in both the 2003 and 2009 remote sensing.

Year	Area (ha)
Solely 2003	1,490
Solely 2009	1,560
Both 2003 - 2009	3,694

Table 3. Presence / absence accuracy assessment matrix for the 2009 seagrass survey.

		Reference		
		Seagrass Absent	Seagrass Present	User's Accuracy
GIS MAP	Seagrass Absent	**69**	9	88%
	Seagrass Present	7	**39**	85%
	Producer's Accuracy	91%	81%	**87%**

Un-weighted Kappa statistic 73%

Table 4. Class accuracy assessment matrix for the 2009 seagrass survey**.**

		Seagrass Absent	Seagrass Sparse	Seagrass Moderate	Seagrass Dense	User's Accuracy
GIS MAP	Seagrass Absent	**69**	7	2	0	88%
	Seagrass Sparse	5	**7**	4	1	41%
	Seagrass moderate	2	4	**6**	10	27%
	Seagrass dense	0	0	2	**5**	71%
	Producer's Accuracy	91%	39%	43%	31%	**70%**

Un-weighted Kappa statistic 47%

Table 5. Distribution of CART models statistics for the automated classification of the vector polygons created in Ecognition™.

	Mean	**Mean (Manual)**	**Standard Deviation**	**5%**	**95%**	**Min**	**Max**
Kappa Coefficient	0.45	0.78	0.11	0.26	0.62	0.06	0.79
Overall Accuracy	0.75	0.92	0.05	0.66	0.83	0.59	0.91

Table5a. Presence / absence accuracy assessment matrix for the 2009 seagrass survey CART model. The matrix shows the percentage of the points found in each location because the model was run 10,000 times it will not be an exact number.

		Reference		
		Seagrass Absent	Seagrass Present	User's Accuracy
CART Model	Seagrass Absent	**47**	19	71%
	Seagrass Present	6	**28**	82%
	Producer's Accuracy	89%	60%	**75%**

Table 6. Cause of classification change in seagrass presence absence between 2003 and 2009 for the BB-LEH estuary system.

Reason	Year Mapped	Estuary wide (ha)	LEH (ha)	SBB (ha)	BI (ha)	CBB (ha)	NBB (ha)
True Loss	2003	563	86	26	384	61	0
True Gain	2009	785	135	146	109	297	0
Change in Season	2003	90	0	0	0	14	76
Change in Season	2009	338	0	0	0	51	287
Misclassification	2003	412	313	66	0	23	0
Misclassification	2009	19	0	5	0	14	0
Poor Image Quality	2003	169	132	4	0	8	24
Poor Image Quality	2009	195	20	169	0	4	3

Table7. Change in seagrass habitat between 2003 and 2009

Location	Year(s) Mapped	Area (ha)
Northern Barnegat Bay	2003	102
-	2009	290
-	2003/2009	277
Central Barnegat Bay	2003	180
-	2009	436
-	2003/2009	1,226
Barnegat Inlet	2003	423
-	2009	128
-	2003/2009	460
Southern Barnegat Bay	2003	131
-	2009	362
-	2003/2009	359
Little Egg Harbor	2003	627
-	2009	235
-	2003/2009	1,240

Table 8. Brown Tide bloom densities in the Barnegat Bay-Little Egg Harbor Estuary

Year	Overall mean (Cells/ml)	Monthy Maximum (Cells/ml)
2000	190,050	2,155,000
2001	246,500	1,883,000
2002	281,900	1,561,000
2003	8,900	54,000
2004	15,700	49,000

From Lathrop and Haag (2006).

Table 9. Total area of seagrass habitat in the Barnegat Bay-Little Egg Harbor estuary impacted by dredging, boat docks, and scars

Impact	**Result**
Boat Dock	53.7 ha
Boat Dock within 100 m of mapped seagrass habitat	30.7 ha
Boat Scarring	42.9 km
Dredging	790 ha

Appendix *in situ* reference data

percent cover *Z. marina*	percent cover *R. maritima*	percent cover macroalgae	percent cover seagrass	valididation site (1 = yes, 0 = no, 2 outside image collection area)	UTM X	UTM Y
0	0	30	0	0	564437	4383598
0	0	0	0	0	564527	4383493
65	0	5	65	0	564678	4383433
50	0	14	50	0	564957	4383278
49	0	16	49	0	565162	4383169
50	0	24	50	0	565378	4383051
50	0	12	50	0	565639	4382920
62	0	9	62	0	565850	4382787
4	0	5	4	0	566189	4382590
0	0	0	0	0	566256	4382530
9	0	11	9	0	566246	4382584
6	0	0	6	0	567593	4384424
11	0	0	11	0	567486	4384469
6	0	21	6	0	567229	4384587
5	0	6	5	0	567080	4384688
22	0	0	22	0	566895	4384827
5	0	3	5	0	566656	4384994
32	0	2	32	0	566650	4385005
38	0	1	38	0	566427	4385146
29	0	1	29	0	566225	4385288
5	0	7	5	0	566037	4385440
1	0	11	1	0	565839	4385567
2	0	6	2	0	565622	4385723
1	0	2	1	0	565412	4385856
28	0	0	28	0	565242	4386009
81	0	3	81	0	564974	4386131
5	0	4	5	0	564780	4386294
0	0	4	0	0	564618	4386428
1	0	2	1	0	563394	4384794
0	0	2	0	0	563921	4382319
10	0	2	10	0	564020	4382251
75	0	10	75	0	564253	4382087
14	0	4	14	0	564397	4381898
77	0	13	77	0	564384	4381903
69	0	10	69	0	564584	4381760
9	0	7	9	0	564867	4381560
35	0	8	35	0	565005	4381357
2	0	7	2	0	569438	4391267
4	8	8	12	0	569681	4391053
6	1	6	7	0	569876	4390901

23	2	4	25	0	570060	4390744
36	0	8	36	0	570287	4390603
1	0	1	1	0	571349	4394740
2	0	1	2	0	571484	4394657
1	14	7	15	0	571760	4394537
2	1	4	3	0	572407	4394398
0	0	4	0	0	572315	4394311
7	3	6	10	0	572554	4394198
38	1	4	39	0	572807	4394102
74	0	3	74	0	572859	4394060
7	0	3	7	0	572870	4394035
15	2	6	17	0	573287	4393821
0	0	0	0	0	573219	4393836
0	0	0	0	0	573122	4393884
4	2	7	6	0	574431	4396744
32	0	3	32	0	574040	4396924
0	0	0	0	0	573542	4397192
0	0	4	0	0	573296	4397259
27	0	0	27	0	572979	4397342
84	0	0	84	0	572796	4397426
18	0	7	18	0	572582	4397504
12	0	18	12	0	572133	4401440
16	0	19	16	0	572339	4401370
40	0	41	40	0	572559	4401283
95	0	2	95	0	572797	4401186
2	20	25	22	0	573065	4401087
1	0	10	1	0	571998	4403190
0	0	12	0	0	572119	4403044
12	0	0	12	0	572297	4403077
3	1	8	4	0	572513	4403047
0	0	0	0	0	572883	4402874
0	0	0	0	0	572797	4403001
0	0	0	0	0	573048	4402936
0	0	0	0	0	573273	4402829
6	0	12	6	0	573504	4402831
0	0	0	0	0	574956	4405339
0	0	0	0	0	575174	4405559
0	0	0	0	0	576529	4417399
82	0	0	82	0	576332	4417404
2	0	0	2	0	576287	4417424
31	2	1	33	0	576700	4427354
96	2	0	98	0	576941	4417362
91	3	0	94	0	577177	4417342
94	2	0	96	0	577435	4417340
66	17	0	83	0	577699	4417372
0	0	0	0	0	577773	4417374
96	1	0	97	0	577942	4417337
3	20	0	23	0	578102	4417332
0	0	0	0	0	577803	4416945
80	2	0	82	0	577783	4416906

0	0	0	0	0	577730	4416485
0	0	0	0	0	577676	4416409
58	13	0	71	1	577419	4416553
7	7	0	14	0	577179	4416575
56	3	0	59	0	576821	4416655
75	0	0	75	0	576561	4416660
4	0	0	4	0	576285	4416675
1	0	0	1	0	576184	4414731
0	0	1	0	0	576457	4414713
0	0	0	0	0	575891	4414621
0	0	0	0	0	577043	4414569
90	1	0	91	0	577658	4414560
4	2	0	6	0	577472	4414521
74	0	0	74	0	577752	4414533
0	0	0	0	0	577893	4414494
1	14	0	15	0	577939	4412610
52	6	4	58	0	577519	4412579
26	11	2	37	0	577244	4412586
66	0	2	66	0	577035	4412562
4	4	3	8	0	576785	4412561
0	0	1	0	0	576597	4412531
0	0	0	0	0	567261	4388138
12	0	5	12	0	567462	4388082
74	0	0	74	0	567554	4388051
51	0	11	51	0	567710	4388025
16	0	9	16	0	567917	4387879
5	0	14	5	0	568113	4387791
0	0	0	0	0	568286	4387760
14	0	7	14	0	568336	4387966
12	0	11	12	0	568436	4387876
11	0	6	11	0	568547	4387785
24	0	5	24	0	568718	4387852
48	0	1	48	0	568295	4388249
70	0	3	70	0	569362	4388503
0	0	0	0	0	569188	4388527
74	0	0	74	0	568298	4388482
69	0	1	69	0	568377	4388786
68	0	4	68	0	568314	4389061
32	0	8	32	0	568015	4388785
0	0	0	0	0	567939	4388857
0	0	0	0	0	567832	4388963
0	0	0	0	0	567800	4389001
0	0	2	0	0	567610	4389121
0	0	0	0	0	567376	4389225
12	7	0	19	0	569436	4390796
48	0	4	48	0	569625	4390641
46	0	8	46	0	569891	4390373
3	0	5	3	0	570771	4391263
69	0	2	69	0	570118	4390353
3	0	0	3	0	569996	4389953

5	0	0	5	0	569636	4389076
0	0	0	0	0	569622	4389072
59	0	6	59	0	568724	4387158
81	0	0	81	0	568186	4387328
0	0	2	0	0	560583	4377237
0	0	15	0	0	560347	4377713
0	0	5	0	0	560265	4378089
0	0	18	0	0	559881	4378697
0	0	1	0	0	558153	4378879
0	0	0	0	0	558553	4378340
0	0	2	0	2	557505	4379874
0	0	5	0	0	560409	4380150
0	0	0	0	0	560320	4381076
0	0	1	0	0	560251	4382884
96	0	0	96	0	564819	4384061
66	0	0	66	0	565493	4384587
0	0	0	0	1	566673	4382850
0	0	0	0	0	566262	4382524
2	0	7	2	0	566268	4382560
72	0	5	72	1	566193	4383856
2	0	5	2	0	566185	4382586
0	0	12	0	1	562198	4379928
72	0	4	72	0	562325	4379820
71	0	24	71	0	562455	4379470
69	0	1	69	0	574001	4399597
0	0	0	0	1	561142	4375660
0	0	0	0	1	560552	4376377
0	0	4	0	1	561047	4377757
0	0	0	0	1	559468	4379550
0	0	1	0	1	558587	4379560
0	0	4	0	1	557945	4378493
0	0	0	0	2	557023	4381009
0	0	100	0	2	557678	4380896
0	0	0	0	1	558989	4380483
0	0	5	0	1	560368	4379953
0	0	1	0	1	560421	4380290
0	0	1	0	1	560794	4380734
0	0	1	0	1	560151	4381075
0	0	0	0	1	560755	4382391
0	0	5	0	2	559654	4383387
0	0	0	0	1	561712	4383271
79	0	2	79	1	565218	4384498
54	0	2	54	1	566043	4384747
11	0	4	11	1	566148	4385321
59	0	9	59	0	566144	4388630
0	0	0	0	1	566114	4382645
94	0	0	94	1	565691	4382031
38	0	4	38	1	564824	4382599
0	0	0	0	1	566708	4388060
0	0	0	0	1	566902	4388548

80	0	1	80	1	566896	4387239
96	0	1	96	1	567533	4386926
9	0	32	9	0	568679	4386489
0	0	0	0	1	568070	4385771
24	0	4	24	1	567849	4386060
50	0	8	50	1	566918	4386027
1	0	0	1	1	566205	4386284
0	0	0	0	1	564053	4382604
0	0	0	0	1	563765	4381563
0	0	0	0	1	563033	4381695
0	0	0	0	1	563514	4381135
0	0	0	0	1	562567	4380176
0	0	0	0	1	562550	4380081
0	0	0	0	1	562758	4379942
36	0	9	36	0	562265	4379866
0	0	1	0	1	562644	4377247
0	0	74	0	1	571298	4403537
0	0	32	0	1	572206	4403055
0	0	0	0	1	574352	4403271
0	0	0	0	0	575473	4403095
0	0	0	0	1	575419	4403332
4	22	2	26	1	573735	4399918
14	0	0	14	1	574169	4399292
37	0	6	37	1	574105	4398382
1	0	10	1	1	575079	4398639
66	5	9	71	1	571617	4398985
2	0	7	2	1	571537	4398370
6	0	0	6	1	571522	4397982
0	0	0	0	1	570244	4397097
0	0	0	0	0	569906	4397609
0	0	0	0	1	571370	4396215
1	8	0	9	0	571589	4395848
55	0	7	55	1	572895	4396173
0	0	0	0	1	572913	4394788
61	0	4	61	1	572722	4395065
4	0	4	4	1	571963	4394783
0	0	0	0	1	570386	4393416
0	2	2	2	1	569389	4393864
1	1	0	2	1	570231	4393027
2	1	4	3	1	569821	4392979
0	0	35	0	1	568828	4392748
0	0	0	0	0	570553	4392547
1	0	0	1	1	569624	4391646
1	0	6	1	1	568951	4391837
0	0	4	0	1	568862	4391973
94	0	0	94	1	568739	4388669
3	0	0	3	1	568909	4391107
89	0	0	89	1	568189	4389837
27	0	5	27	1	568194	4389345
82	0	2	82	1	568932	4388567

20	0	56	20	1	569536	4387962
0	0	0	0	1	567075	4383054
0	37	0	37	1	577889	4426660
0	18	0	18	1	579143	4426673
0	0	0	0	1	579213	4427124
0	91	0	91	1	578012	4427665
10	80	0	90	1	578296	4428100
0	0	0	0	1	579733	4430156
0	0	0	0	1	579389	4431066
1	0	12	1	1	580554	4434422
0	0	29	0	1	579831	4434398
0	0	2	0	1	577571	4433563
0	0	8	0	1	578856	4430734
0	0	0	0	1	578384	4430878
0	13	0	13	1	576679	4429727
0	6	0	6	2	574897	4428210
0	0	0	0	2	574725	4428223
0	0	0	0	2	574631	4427328
0	0	0	0	2	574646	4426704
0	0	0	0	1	575612	4426183
0	74	0	74	1	577422	4424066
0	0	0	0	2	574589	4430800
0	0	0	0	1	575714	4424010
0	0	0	0	1	576099	4423334
0	0	0	0	1	576682	4423170
0	14	0	14	1	577585	4422819
0	45	0	45	1	577662	4422808
0	0	0	0	1	578257	4422802
0	0	0	0	1	578385	4419989
0	0	0	0	2	571881	4421902
0	0	0	0	2	573722	4421066
94	5	0	99	1	576188	4416031
0	0	0	0	1	578185	4415176
76	8	12	84	1	577316	4414854
7	25	50	32	1	577455	4413430
2	1	0	3	1	577776	4413181
63	11	9	74	1	577619	4412809
9	28	29	37	1	577153	4413052
21	14	0	35	1	577311	4411997
1	1	0	2	1	576818	4411722
49	8	5	57	1	576903	4411170
21	24	6	45	1	577549	4410915
0	0	2	0	1	576948	4409989
0	0	0	0	1	575102	4412235
0	0	0	0	1	575527	4415088
0	0	0	0	1	574393	4415351
61	6	0	67	1	573956	4415898
0	0	0	0	1	577187	4416483
88	4	4	92	1	577461	4416555
89	0	0	89	1	577656	4416772

78	6	6	84	1	576817	4417144
0	0	0	0	1	573854	4408093
91	1	1	92	1	575427	4407923
31	6	12	37	1	575278	4407493
11	0	10	11	1	571186	4406470
83	0	0	83	1	570901	4406421
94	0	0	94	1	570701	4406189
1	0	1	1	1	573083	4405872
5	0	23	5	1	573947	4406036
1	0	14	1	1	574233	4406066
5	0	36	5	1	574162	4406378
11	0	39	11	1	574559	4406402
0	0	0	0	1	575084	4405509

Bibliography

Alexander, R. B., R. A. Smith, G. E. Schwarz, S. D. Preston, J. W. Brakebill, R. Srinivasan, and P. A. Pacheco. (2000). Atmospheric nitrogen flux from the watersheds of major estuaries of the United States: an application of the SPARROW watershed model. Pages 119–170 in R. A. Valigura, R. B. Alexander, M. S. Castro, T. P. Myers, H. W. Paerl, P. E. Stacey, and R. E. Turner, editors. Nitrogen loading in coastal water bodies: an atmospheric perspective. American Geophysical Union, Washington, D.C., USA.

Arber A. (1920). Water Plants, A Study of Aquatic Angiosperms. Cambridge: Cambridge University Press.

Baden, S., Gullstro, M., Lunde, B., Pihl, L., & Rosenberg, R., (2003). Vanishing seagrass (*Zostera marina*) in Swedish coastal waters. Ambio 32: 374-377.

Beer, S., & Koch, E., (1996). Photosynthesis of Marine Macroalgace and seagrasses in globally changing CO2 environments. Marine Ecology Progress Series 141: 199-204.

Bell unpublished in Fonseca, M., & Bell, S. S. (1998), Influence of physical setting on seagrass landscapes near Beaufort, North Carolina, USA. Marine Ecology Progress Series, 171, 109-121.

Bologna, P. A., & Heck, L. K. (1999). Differential predation and growth rates of bay scallops within a searass habitat. Journal of experimental Marine Biology and Ecology, 239(2), 299-314.

Bologna, P., Able, K., Lathrop, R. G. & Bowers, P. (2000). Assessment of the health and distribution of submerged aquatic vegetation from Little Egg Harbor, New Jersey. Technical Report #2000-11, Institute of Marine and Coastal sciences, Rutgers University, New Brunswick, New Jersey.

Bologna, P. & Heck. K., (2000). Impacts of seagrass habitat architecture on bivalve settlement. *Estuaries* 23:449–457.

Bologna, P. , Wilbur, A., and Able, K., (2001). Reproduction population structures, and recruitment failure in a bay scallop (Argopectan irradians) population from a coastal New Jersey, USA, Journal of Shellfish Research 20:89-96.

Bologna, P., (2006). Assessing within habitat variability in plant demography, faunal density, and secondary production in an eelgrass (Zostera marina L.) bed. Journal of Experimental Marine Biology and Ecology V. 329 122–134.

Bowen, L., Ramstack, M., Mazzilli, S., & Valiela, I., (2007). NLOAD: an interactive, web-based modeling tool for nitrogen management in estuaries. Ecological Applications 17: 17–30.

Burkholder, J., Tomasko, D., & Touchette, B., (2007). Seagrass and Eutrophication. Journal and Experimental Marine Biology and Ecology 350: 46-72.

Christiansen, C., Christofferson, H., Dalsgaard, J., Nornberg, P. (1981). Coastal and near-shore changes correlated with die-back In eelgrass (Zostera marina L.). Sediment. Geology. 28: 163-173

Deegan, L., Wright, A., Ayvazian, S., Finn, T., Golden, H., Merson, R., & Harrison, J., (2002). Nitrogen loading alters seagrass ecosystem structure and support of higher trophic levels. Aquatic Conservation: Marine and Freshwater Ecosystems. 12: 193-212.

den Hartog, C. (1970) *The Seagrasses of the World*. Amsterdam, the Netherlands: North Holland.

Dennison, W., & Alberte, R., (1982). Photosynthetic responses of *Zostera marina* L. (Eelgrass) to in situ manipulations of light intensity. Oecologia. 55: 137-144.

Drake, L. A., Dobbs, F. C. & Zimmerman R.C.R. (2003). Effects of epiphyte load on optical properties and photosynthetic potential of the seagrasses Thalassia testudinum Banks ex Konig and Zostera marina. Limnology & Oceanography. 48, 456-463.

Duarte, C.M. (1991). Seagrass Depth Limits. Aquatic Botany, 40(4), 363-377.

Durand, M. D., & Olson, R. J. (1998). Contributions of phytoplankton light scattering and cell concentration changes to diel variations in beam attenuation in the equatorial Pacific from flow cytometric measurements of pico, ultra, and nanoplankton. Deep Sea Research. 43, 891-906.

Eyre, B., & Ferguson, A. (2002). Comparison of carbon production and decomposition, benthic nutrient fluxes and denitrification in seagrass, phytoplankton, benthic microalgae- and macroalgae dominated warm-temperature Australian lagoons. Marine Ecology Progress Series. 229, 43-59.

Fonseca, M., Whitfield, P. E., Kelly, N. M., & Bell, S. S. (2002), Modeling Seagrass Landscape Pattern and Associated Ecological Attributes. Ecological Applications, 12(1), 218-237.

Fonseca, M., & Bell, S. S. (1998), Influence of physical setting on seagrass landscapes near Beaufort, North Carolina, USA. Marine Ecology Progress Series, 171, 109-121.

Gastrich, M. D., Wazniak, C.E. (2002). A Brown Tide Bloom Index Based on the Potential Harmful Effects of the Brown Tide Alga, Aureococcus anophagefferens. Aquatic Ecosystems Health and Management. 33(2), 175-190.

Gastrich, M. D., Lathrop, R., Haag, S., Weinsteinn, M. P., Danko, M., Caron, D., Schaffner, R. (2004). Assessment of brown tide blooms, caused by Aureococcus anophagefferens, and contributing factors in New Jersey Coastal Bays: 2000-2002. Harmful Algae, Special Issue Brown Tides, 3, 305-320.

Gordon, D. M., & McComb, A. J.A. (1989). Growth and production of the green alga n estuary and its interpretation Cladophora montagneana in a eutrophic Australian estuary and it interpretation using a computer model. Water Resources, 23, 633-645.

Guo, G., N. Psuty, G. Lordi, and C-S. Tsali. 1997. Circulation studies in Barnegat Bay. In: Flimlin, G. E. and M. Kennish (eds.), *Proceedings of the Barnegat Bay Ecosystem Workshop*. Rutgers Cooperative Extension of Ocean County, Toms River, New Jersey, pp.17-29.Gou et al, 2004.

Harwell, M., Orth, R. (2002). Seed Bank Patterns in Chesapeake Bay Eelgrass (*Zostera marina*): A Bay-wide perspective. Estuaries. 25, 1196-1204.

Hauxwell, J., Cebrian, J., Furlong, C., & Valiela, I. (2001). Macroalgal canopies contribute to eelgrass (Zostera marina) decline in temperate estuarine ecosystems. Ecology, 82(4), 1007-1022.

Hauxwell, J., Cebrián J., and I. Valiela, 2003. Eelgrass *Zostera marina* loss in temperate estuaries: relationship to land-derived nitrogen loads and effect of light limitation imposed by algae. Marine Ecology Progress Series, 247:59-73.

Harris, L., S. Granger, and S. Nixon. 2007. Evaluation of the Health of Eelgrass (Zostera marina L.) Beds Within the Maryland Coastal Bays, Technical Report NPS/NER/NRR—2007/014. National Park Service. Philadelphia, Pennsylvania.

Hein, M. & Pedersen, M. P. & Sand-Jensen, K. (1995). Size-dependant nitrogen uptake in micro- and macroalgae. Marine Ecology Progress Series, 118, 247-253.

Hemminga, M., Duarte, C. (2000). Seagrass Ecology. Cambridge University. Press, Cambridge.

Johnson, M. P., Edwards, M., Bunker, F., & Maggs, C. Algal epiphytes of Zostera marina: Variation in assemblages structure from individual leaves to regional scale. Aquatic Botany, 82(1), 12–26.

Joseph, J., Purdy, K., & Figley, B. (1992). The influence of water depth and bottom sediment on the occurrence of eelgrass in Barnegat, Manahawkin and Little Egg Harbor bays. Marine Fisheries Administration, New Jersey Department of Environmental

Kennish, M. J., Haag, S., & Sakowicz, G. (2007). Demographic investigation of SAV in the Barnegat Bay-Little Egg Harbor Estuary with assessment of potential impacts of benthic macroalgae and brown tides. Technical Report 107-15, 47 Institutes of Marine and Coastal Sciences, Rutgers University, New Brunswick, New Jersey. 366 pp.

Kennish, M., Haag, S., & Sakowicz, G. (2008). Seagrass demographic and Spatial Habitat Characterization in Little Egg Harbor, New Jersey, Using Fixed Transects. Journal of Coastal Research, Special Issue 55, 148-170.

Kennish, M. J. (2001). Barnegat Bay-Little Egg Harbor, New Jersey: Estuary and Watershed Assessment. *Journal of Coastal Research*, Special Issue No. 32, 280 pp.

Kou, J. & den Hartog C. (2006). Seagrasses Morphology, Anatomy and Ultrastructure. In Larkum, W. D, Orth, R. J., & Duarte, C. M., Seagrasses: Biology, Ecology, and Conservation. (pp 51-87). AA Dordrecht, The Netherlands.

Kou, J. & den Hartog C. (2006). Taxonomy and Biogeography of Seagrasses. In Larkum, W. D, Orth, R. J., & Duarte, C. M., Seagrasses: Biology, Ecology, and Conservation. (pp 51-87). AA Dordrecht, The Netherlands.

Kenyon, R. A., Loneragan, N. R., Hughes, J. M., and Staples, D. J. (1997).Habitat Type Influences the Microhabitat Preference of Juvenile Tiger Prawns. Estuarine Coastal and Shelf Science, 45(30), 393-403.

Lapointe, B. E., & Clark, M. W. (2008). Nutrient inputs from the watershed and coastal eutrophication in the Florida Keys. Estuaries and Coasts, 15(4), 465-476.

Larkum, A.W.D. and den Hartog, (1989). Evolution and Biogeography of Seagrasses. In: Biology of Seagrasses (Ed. A.W.D. Larkum, A.J. McComb and S.A. Shepherd) pp 143-165, Elsevier Pub Co, Amsterdam

Lathrop, R., & Bognar, J. A., Hendrickson, A.C., & Bowers, P.D. (1999). Data Synthesis effort for the Barnegat Bay Estuary Program: Habitat Loss and Alteration in the Barnegat Bay Region, CRSSA Technical Report.

Lathrop, R.G. and Bognar, J.A. (2001). Habitat Loss and Alteration in the Barnegat Bay Region. Journal of Coastal Research, 32, 212-228.

Lathrop, R., & Conway, T,. (2001). A Build-out Analysis of the Barnegat Bay Watershed. CRSSA technical report. 1-11.

Lathrop, R., Montesano, P., & Haag, S. (2006). A muli-scale segmentation approach to mapping seagrass habitat using airborne digital camera imagery. Photogrammetric Engineering and Remote Sensing, 72(6), 665-675.

Lathrop, R., & Haag, S. (2007). Assessment of Land Use Change and Riparian Zone Status in the Barnegat Bay and Little Egg Harbor Watershed: 1995-2002-2006. Technical Report Barnegat Bay National Estuary Program (BB NEP), 1-27.

Lathrop, R.G., Styles, R., Seitzinger, S., and Bognar, J. (2001). Use of GIS Mapping and Modeling Approaches to Examine the Spatial Distribution of Seagrasses in Barnegat Bay, New Jersey. Estuaries 24, 904-916.

Lawrence, R.L., & Wright, A. (2001). Rule-based classification systems using classification and regression tree (CART) analysis, Photogrammetric Engineering & Remote Sensing, 67, 1137–1142.

Lewis, R. (2000). An introduction to the Classification and Decision Tree (CART) Analysis. Annual Meeting of the Society for Academic Emergency Medicine in San Francisco, California. Annual Meeting of the Society for Academic Emergency Medicine, San Francisco, CA, 2000.

Lorenzen, C. J. (1972). Extinction of light in the Ocean by Phytoplankton. Journal du Conseil, 34(2), 262-267.

Yang, X. Lo, C. (2000). Relative radiometric normalization performance for change detection from multi-date satellite images. Photogrammetric Engineering and Remote Sensing. 6(8), 967-980.

Lyzenga, D.R. (1978). Passive Remote-Sensing Techniques for Mapping Water Depth and Bottom Features. Applied Optics, 17, 379-383.379-383.

Macomber, R. T. & Allen D. (1979). The New Jersey submerged aquatic vegetation distribution atlas final report. Earth Satellite Corporation, Washington, D.C

McGlathery , K. (2001). Macroalgal blooms contribute to the decline of seagrass in nutrient-enriched coastal waters. Journal Of Phycology, 37, 453-456.

McGranahan, G., Balk, D., & Anderson, B. (2007). The rising tide: assessing the risks of climate change and human settlements in low elevation coastal zones. Environment and Urbanization, 19(17), 17-37.

Mclain, P. & McHale, M. (1996). Barnegat Bay eelgrass investigations 1995–1996, p. 165–172. *In* G. Flimlin and M. Kennish (eds.), Proceedings of the Barnegat Bay EcosystemWorkshop Rutgers Cooperative Extension, Toms River, New Jersey.

Moore, K., Neckles, H., and Orth, R. Zostera. (1996). Zostera marina growth and survival along a gradient of nutrients and turbidity in the lower Chesapeake Bay. Marine Ecology Progress Series, 142, 247-259.

Moser, F. C. 1997. Sources and sinks of nitrogen and trace metals, and benthic macrofauna assemblages in Barnegat Bay, New Jersey. Ph.D. Thesis, Rutgers University, New Brunswick, New Jersey.

Mountford, K., (2002). Closed Sea: From the Manasquan to the Mullica – A history of Barnegat Bay. West Creek NJ. Down the shore publishing company.

Neckles, H. A., Wetzel, R. L. & Orth, R., (1993). Relative effects of nutrient enrichment and grazing on epiphyte-macrophyte. Oecologia, 92(2), 285-295.

NOOA website for Magnetic delineations http://www.ngdc.noaa.gov/geomagmodels/Declination.jsp

Olesen, B., & Sand-Jensen, K. (2006), Seasonal acclimatization of eelgrass Zostera marine growth to light. Marine Ecology Progress Series, 94, 91-99.

Orth, J. R., Marion, S., Moore, K., and Wilcox, D., (2010). Eelgrass in the Chesapeake Region of the Mid Atlantic Coast of the USA: Challenges in Conservation and Restoration. Estuaries and Coast, 33, 139-150.

Pecchioli, J.A., Lathrop, R., & Haag, S. (2006). Brown Tide Assessment Project in NJ coastal waters: a comparison of three bloom years (2000-2002) to non bloom years (2003-2004). Research Project Summary, NJDEP.

Ridgeway, M., (2000). The Baymen: A life on Barnegat Bay. West Creek NJ. Down The Shore Publishing Company.

Robbins, B.D.,& Bell, S.S. (1994). Seagrass landscapes: A terrestrial approach to the marine subtidal environment, *Trends Ecology and Evolution*, 9, 301–304.

Seitzinger, S. and R. Styles. 1999. Water quality and primary production in the Barnegat

Short, F. T., & Wyllie-Echeveria, S. (1996). Natural and human induced disturbances of seagrass. Environmental Conservation. 23(1), 17-27.

Short, F. T., & Burdick, D. M. (1995). Mesocosm experiments quantify the effects of eutrophiciation on eelgrass, *Zoster marina*. Limnology and Oceanography, 40(4), 740-749.

Stumpf, R. P., Holdcried, K., & Sinclar, M. (2003). Determination of Water Depth with High Resolution Sattelite Imagery over Variable Bottom Types. Limnology and Oceanography, 48, 547-556.

United States Census Bureau quick facts website http://quickfacts.census.gov/qfd/states/34/34029lk.html

Valiela, I., McClelland, J., Hauxwell, J., Behr, P.J., Hersh, D., & Foreman, K. (1997). Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. Limnology and Oceanography, 42, 1105–1118.

Valiela, I., Foreman, K., LaMontagne, M., Hersh, D., Costa, J., Peckol, P., DeMea Andreson, B., D'Avanzo, C., Babiona, M., Sham, C., Brawley, J., & Lajtha, K. (1992). Couplings of watershed and coastal waters: sources and consequences of nutrient enrichment in Waquoit Bay, MA. Estuaries and Coasts. 15(4), 443-457.

Waycott, M., Duarte, C., Carruthers, T., Orth, R., Dennison, W., Olyarnik, S., Calladine, A., Fourqurean, J., Heck, K., Hughes, R., Kendrick, G., Kenworthy, J., Short., F., & Williams, S. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceeding of the National Academy of Science. 106. 12377 – 12381.

Wazniak, C.E., Hall, M. R., Carruthers, T. J., Sturgis, B., Dennison, W. C., & Orth, R. J. (2007). Linking Water Quality to Living Resources in a Mid-Atlantic Lagoon System, USA. Ecological Applications, 17(5), 64-78.