In situ Surveys of Seagrass Habitat in the Northern Segment of the Barnegat Bay-Little Egg Harbor Estuary: Eutrophication Assessment

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ABSTRACT

A comprehensive seagrass survey has been conducted in the north segment of the Barnegat Bay-Little Egg Harbor Estuary during 2011 to assess seagrass characteristics. The primary objectives of this project were the following: (1) to determine seagrass demographics in the north segment of the estuary by investigating mixed beds of *Ruppia maritima* and *Zostera marina* over the June-November sampling period in 2011; and (2) to document seagrass characteristics across the entire estuary during 2011 as part of a separate study. Results of this investigation show conclusively that *R. maritima* dominates seagrass beds in the north segment, while *Z. marina* dominates seagrass beds in the central and south segments. *Ruppia maritima* was found at all three sampling transects (13, 14, and 15) in the north segment during all three time periods (June-July, August-September, and October-November). It was far more abundant than *Z. marina* in this segment. Mean aboveground biomass of *R. maritima* was highest at Transect 13 during Time Period 1 (19.59 g dry wt. m^{-2}) and lowest at Transect 14 during Time Period 2 (1.97 g dry wt. m^{-2}). Mean belowground biomass of *R. maritima* was highest at Transect 15 during Time Period 1 (26.39 g dry wt. m^{-2}) and lowest at Transect 15 during Time Period 2 (3.23 g dry wt. m^{-2}). By comparison, *Z. marina* was found in the north segment only at Transect 14 and Transect 15 during Time Period 1 (June-July). It was absent at these transects during later time periods. Only a small amount of *Z. marina* occurred in the north segment during the June-July sampling period in 2011 at Transect 14 (mean aboveground biomass = 0.16 g dry wt m^{-2}; mean belowground biomass = 0.72 g dry wt m^{-2}). Similarly, only a small amount of *Z. marina* occurred in the north segment during the June-July sampling period in 2011 at Transect 15 (mean aboveground biomass = 1.29 g dry wt. m^{-2}; belowground biomass = 7.18 g dry wt. m^{-2}).

Since 2004, system eutrophy has generally worsened in BB-LEH, and the condition of the seagrass habitat has markedly degraded in the central and south segments. Eelgrass biomass

declined consistently over the 2004-2006 and 2008-2010 periods and overall from 2004-2010. The 2010 eelgrass biomass values were the lowest levels recorded in the estuary. Data collected on demographic trends indicate that eelgrass beds in 2011 had yet to recover from the marked decline of plant biomass and areal cover observed in 2009 and 2010. The trend of eelgrass decline over the years has not been isolated to one bed but has been observed over extensive areas of the estuary, signaling a response to a broad-scale stressor that adversely affects plant condition across the system. Nutrient loading and eutrophication have been identified as the primary drivers of change in seagrass habitat of the estuary.

PROJECT DESCRIPTION

Introduction

Eutrophication of the Barnegat Bay-Little Egg Harbor (BB-LEH) Estuary has persisted with ongoing nitrogen loading from the Barnegat Bay watershed. Between 1998 and 2007, the total nitrogen load in surface runoff of the watershed increased from 390,000 kg N yr^{-1} to 431,000 kg N yr^{-1} (Wieben and Baker, 2009). Key biotic indicators of estuarine ecosystem condition declined significantly between 2004 and 2010, most notably eelgrass (*Zostera marina*) biomass, blade length, and areal cover (Kennish et al., 2008, 2010, 2012; Fertig et al., 2013). The mean eelgrass biomass in 2004 was 60.8 g dry wt m^{-2} aboveground and 76.5 g dry wt m^{-2} belowground, but these decreased by 2010 to 7.5 g dry wt m^{-2} and 26.7 g dry wt m^{-2}, respectively (Kennish et al., 2012). In addition, over the past two decades, recurring toxic and nuisance algal blooms (both phytoplankton and macroalgae) have occurred in the estuary, and a decrease in the shellfish (hard clams, *Mercenaria mercenaria*) abundance and harvest has been documented within the estuary (Celestino, 2003; Kennish et al., 2012). Sea nettles (*Chrysaora quinquecirrha*) increased rapidly in abundance particularly in the north segment of the estuary, impacting human use of estuarine waters (BBP, 2011). Low dissolved oxygen levels (< 4 mg/l) were recorded by the New Jersey Department of Environmental Protection (NJDEP) at several monitoring locations in the BB-LEH estuarine system between 1989 and 2010 (Kennish et al., 2012). Conversion of natural land covers to urbanized landscape has accelerated nutrient loading to estuarine tributaries, leading to cascading water quality and biotic impacts, as well as impairment of estuarine waters. Reduced freshwater inflow, restricted basin circulation, low flushing rates, and a water residence time exceeding 70 days in summer have contributed to the eutrophication problems documented in the estuary (Kennish et al., 2012; Fertig et al., 2013).

Seagrass habitat has declined significantly over the past several years in the BB-LEH in response to nutrient enrichment (Kennish et al., 2008, 2010, 2012; Fertig et al. 2013), a problem that has also plagued other coastal lagoons and some deeper estuarine systems in the mid-Atlantic region (Bricker et al., 2007; Burkholder et al., 2007; McGlathery et al., 2007; Wazniak et al., 2007; Moore, 2009). The loss of seagrass biomass is a serious concern in the estuary because of the multiple ecosystem services that the seagrass beds provide such as a major source of primary production, food for waterfowl, essential habitat and nursery areas for numerous fish and invertebrates, filters of chemical substances, agents in biogeochemical cycling, and buffers against wave and current action as well as sediment erosion (Larkum et al., 2006; Moore, 2009). These vascular plants are also important indicators of overall ecosystem health of the estuary

because they integrate water quality and benthic attributes (Orth et al., 2006; Burkholder et al., 2007; Kennish et al., 2008).

Comprehensive *in-situ* surveys of seagrass beds have been conducted annually since 2004 (excepting 2007) in the south and central segments of the estuary from southern Little Egg Harbor to Seaside Park to assess seagrass habitat and eutrophic condition. However, no comprehensive *in-situ* seagrass surveys were conducted in the north segment of the estuary prior to 2011, an area of high watershed development, elevated nutrient loads, and problematic eutrophic responses (e.g., low dissolved oxygen). This is so because previous eutrophication assessment studies of seagrass in the estuary have targeted eelgrass (*Zostera marina*) beds which are concentrated in the central and south segments of the estuary rather than beds dominated by widgeon grass (*Ruppia maritima*) in the north segment. A detailed field survey of the seagrass beds was conducted in the north segment of the estuary during the June-November period of 2011. Data collected in this project can be used as a baseline dataset for the north segment of BB-LEH, which will be useful for tracking spatial and temporal patterns of eutrophication and for determining if eutrophic conditions are improving, declining, or not changing, particularly when paired with water quality and other monitoring data.

Since 2004, system eutrophy has generally worsened in BB-LEH, and the condition of the seagrass habitat has markedly degraded in the central and south segments (Kennish et al., 2008, 2010, 2012). Eelgrass biomass declined over the 2004-2006 and 2008-2010 periods and overall from 2004-2010 (Fertig et al., 2013). The 2010 eelgrass biomass values were the lowest levels recorded in the estuary. Data collected on demographic trends indicate that eelgrass beds in 2011 had yet to recover from the marked decline of plant biomass and areal cover observed in 2009 and 2010 (Kennish et al., 2012). The trend of eelgrass decline over the years has not been isolated to one bed but has been observed over extensive areas of the estuary, signaling a response to a broad-scale stressor that adversely affects plant condition across the system. Nutrient loading and eutrophication have been identified as the primary drivers of change in seagrass habitat of the estuary (Kennish et al., 2007, 2008, 2012).

Rationale

Without a series of annual seagrass surveys in the north segment of the estuary, it is not possible to accurately document the response of seagrass to nitrogen loading in this segment of the estuary, or the effectiveness of watershed restoration efforts. This is an important segment of the estuary to characterize the effects of nutrient enrichment and the impacts of key biotic factors on ecological condition, such as blooms of nuisance phytoplankton and macroalgae, as well as eruptions sea nettles (*Chrysaora quinquecirrha*), which can alter the ecosystem structure and function.

Analysis of seagrass indicator data collected in the central and south segments of the BB-LEH over the 2004-2010 period indicates a serious declining trend in seagrass habitat condition correlated with ongoing system eutrophy (Kennish et al., 2007; 2008; 2009; 2010, 2012). There has been a significant temporal and spatial decline in eelgrass demographics in the estuary and, as a result, eelgrass beds in the central and south segment of the estuary have yet to recover from the severely depressed plant biomass values recorded in 2009 and 2010. However, a major data

gap exists for biotic responses to nutrient loading in the north segment of the estuary. This project targets this data gap so that eutrophication assessment of the seagrass community can be extended estuary-wide. The same seagrass and water quality indicators monitored in prior years of sampling in the estuary are examined in this study (Kennish et al., 2008, 2010, 2012). Results of this study are needed to holistically assess the eutrophic condition of the estuary, and to provide baseline data for ongoing ecosystem-based management of the system.

The primary objectives of this project are the following: (1) to determine seagrass demographics in the north segment of the estuary by investigating mixed beds of *Ruppia maritima* and *Zostera marina* over the June-November sampling period in 2011; and (2) to document the presence/absence, biomass, shoot density, areal cover, and eelgrass blade length of seagrass across the entire estuary. Data collected in this project are also part of a validation database for inclusion in a separate study conducted for the New England Water Pollution Control Commission (NEIWPCC).

This project is relevant to specific elements of the Barnegat Bay Partnership CCMP and will deliver an array of data under the purview of the plan, including: (1) Primary indicators: SAV distribution, abundance, plant characteristics, harmful algal blooms; and (2) Secondary indicators: temperature, pH, salinity, DO, nutrients, and turbidity. It addresses several parts of the BBNEP Monitoring Plan objectives: CCMP Chapter 5, Part 1, Monitoring Need 1: Summer survey of algal populations; CCMP Chapter 6, Part 6. Monitoring Need 2: Comprehensive habitat inventory and mapping, Part 5. Status and trend analysis of resources use: CCMP Chapter 6, Part 7. Monitoring Need 1: Population inventories of targeted species and Monitoring Need 3: Periodic species population monitoring.

Outcome

Data from this project have been integrated into an ecosystem-based study assessing biotic responses and biotic indicators of estuarine condition, particularly as they relate to the effect of watershed nutrient loading. A major outcome of this project is the compilation of quantitative measures of the distribution, biomass, density, blade length (for eelgrass), and areal cover of seagrasses in the north segment of the estuary. To this end, the data will be used in the development of a biotic index of ecosystem condition for the estuary in a related study, including an index of condition for the north segment (Kennish et al., 2012).

STUDY AREA

Physical Characteristics

BB-LEH is located along the central New Jersey coastline between 39º31'N and 40º06'N latitude and 74º02'W and 74º20'W longitude. It forms a long, narrow, and irregular tidal basin that extends north-south for nearly 70 km, being separated from the Atlantic Ocean by a narrow barrier island complex (i.e., Island Beach and Long Beach Island) that is breached by the Point Pleasant Canal in the north segment, at Barnegat Inlet in the central segment, and at Little Egg Inlet in the south segment (Kennish, 2001a-c) (Figure 1). Exchange of bay and ocean water occurs through these three inlets. The continuity of the barrier island complex restricts the exchange of water with the coastal ocean, resulting in a protracted water residence time in the

estuary amounting to 74 days in summer when eutrophication is most problematic (Guo et al., 1997, 2004).

Ranging from 2 to 6 km in width and 1 to 6 m in depth, the BB-EH Estuary has a volume of ~2.4 x 10^8 m^3 and a wet surface area of ~280 km^2 (Kennish, 2001a). Water temperature ranges from -1.5-30ºC, and salinity from ~10-32. Characterized by semidiurnal tides with a tidal range of <0.5-1.5 m, the estuary is well-mixed by wind and currents. Current velocities are typically <0.5-1.5 m s^{-1}. The shallowness of the open bay, extensive shoals and marsh islands near the inlets, and the morphology of the perimeter areas restrict current movement. The long water residence time promotes pollution retention and recycling in the estuary, thereby increasing the probability of pollution impacts and ecological damage.

The freshwater supply to the BB-LEH derives primarily from surface water discharges and groundwater inputs from the unconfined Kirkwood-Cohansey aquifer system. Surface and groundwater flows are generally well connected, with groundwater being the dominant (>80%) contributor to stream baseflows (i.e., as compared to surface runoff). Previous modeling efforts have predicted large decreases in the groundwater levels associated with development (Nicholson and Watt, 1997a,b). Groundwater withdrawal currently amounts to ~77 million gallons per day (BBP, 2011)). The mechanisms for loss of groundwater include higher amounts of impervious surfaces and withdrawal of groundwater for domestic uses much of which is treated at wastewater treatment plants and discharged through an ocean outfall, thus bypassing the estuary.

The human population in the watershed has increased dramatically over the past 60 years to more than 575,000 year-round residents (Table 1) and more than 1.2 million summer residents. At buildout, the population in the watershed is expected to exceed 825,000 year-round residents (Lathrop and Conway, 2001). Since 1972, the amount of developed land has risen from ~19% to ~34% of the watershed. Urban land use area increased from ~25% in 1995 to ~30% between 1995 and 2010 (BBP, 2011; Lathrop and Haag, 2011). These land-use changes have resulted in increased nonpoint source inputs of nutrients to the estuary (Kennish, 2001d; Kennish et al., 2007).

The BB-LEH watershed (1,730 km^2) lies entirely in one state (New Jersey) and mainly receives nonpoint nutrient sources from residential fertilizers, atmospheric deposition, and leaking septic systems that enter the estuary via both overland runoff and groundwater inputs (Kennish, 2001a; Kennish and Townsend, 2007). Since the wet surface area of the estuary is 280 km^2, the watershed:estuary areal ratio is ~6.5:1. A north-to-south gradient of decreasing developed watershed area and associated total nitrogen load is well documented (Hunchak-Kariouk and Nicholson, 2001; Seitzinger et al., 2001; Wieben and Baker, 2009).

Nutrient Loading

Nutrient loading to the estuary is linked to population growth and development in the watershed. In an earlier study, Hunchak-Kariouk and Nicholson (2001) calculated the total nitrogen load to the estuary of ~7.2 x 10^5 kg N yr^{-1}, with ~54% (3.9 x 10^5 kg N yr^{-1}) derived from surface water inflow, ~34% (2.4 x 10^5 kg N yr^{-1}) from atmospheric deposition, and ~12% (8.6 x 10^4 kg N yr^{-1}) from direct groundwater discharges. Wieben and Baker (2009) later estimated that the total nitrogen load to the estuary amounted to ~6.5 x 10^5 kg N yr^{-1}, with surface water

discharge contributing 66% (4.3 x 10^5 kg N yr^{-1}), atmospheric deposition 22% (1.41 x 10^5 kg N yr^{-1}), and direct groundwater discharge 12% (7.8 x 10^4 kg N yr^{-1}). According to Wieben and Baker (2009), more than 60% of the nitrogen load in surface water discharge originates from the Toms River and Metedeconk River basins. The relative contribution of nitrogen from surface water runoff, therefore, has increased substantially in recent years. Most of the nitrogen input to the estuary therefore occurs in the northern part of the BB-LEH watershed.

APPROACH

Seagrass Sampling Protocols

To accomplish the objectives of the project, an *in situ* survey was conducted on key demographic characteristics of mixed seagrass beds (*Ruppia maritima* and *Zostera marina*) in the north segment of the estuary during the June-November sampling period in 2011. A separate survey of seagrass beds in the central and south segments of the estuary was also conducted during the same sampling period, providing concurrent and complete coverage of seagrass habitat in the estuary for 2011 (Figure 1). Primary biotic data collected in the central and south segments included the presence/absence, aboveground and belowground biomass, shoot density, areal cover, and blade length (for eelgrass only) of seagrass. In addition to the percent epiphytic growth on seagrass, the presence of bay scallops and other shellfish was also recorded in the seagrass beds. The presence/absence and percent cover of macroalgae were also measured at each sampling station.

Seagrass sampling was conducted using the protocols of the SeagrassNet approach (Short et al., 2002) that were applied by Kennish et al. (2007, 2008, 2010, 2012) in prior annual seagrass surveys conducted in the estuary from 2004 to 2010 (excluding 2007). These sampling protocols were employed in this project to maintain consistency for data integration with the previous seagrass surveys.

Quadrat-and-transect sampling of seagrass beds in the north segment was conducted bimonthly using the SeagrassNet approach at 10 equally spaced sampling stations along each of 3 transects (13 southernmost transect, 14, and 15 northernmost transect) during 3 sampling periods (June-July, August-September, October-November) in 2011 (Figure 2). Thus, the target was to collect a total of 90 seagrass samples at the 30 sampling stations in this segment of the estuary during the entire study period. The same sampling protocol was followed in the north segment as in the central and south segments noted above. In addition to collecting data on the presence/absence, aboveground and belowground biomass, shoot density, areal cover, and blade length (for eelgrass only), percent epiphytic growth and the presence of bay scallops and other shellfish were recorded in the seagrass beds. The presence/absence and percent cover of macroalgae were also measured at each sampling station.

A 10-cm diameter, diver-deployed PVC corer was used to collect *in situ* seagrass samples. Diver observations were made at each sampling station to determine the occurrence and areal cover of seagrass and macroalgae, epiphytic growth, and presence of bay scallops and other shellfish species. In addition, high resolution, underwater photographs were used to validate diver observations. Sampling stations were located with a Differential Global Positioning System (Trimble®GeoXT™ handheld unit).

Temperature (°C), salinity, pH, and dissolved oxygen (mg L^{-1} and percent saturation) data were collected at each sampling station using either a handheld YSI 600 XL datasonde coupled with a handheld YSI 650 MDS display unit, an automated YSI 6600 unit, or a YSI 600 XLM automated datalogger. Secchi disk and total depth (m) measurements were likewise collected in the survey area. Water quality data (other than Secchi measurements) were collected at a uniform depth (~10 cm) above the sediment-water interface using YSI datasondes. Details of the protocols for field sampling, laboratory processing of samples, and data analysis for the proposed project can be found in Baker and Kennish (2010).

Quadrat Sampling

Following the field sampling methods of Short et al. (2002), a 0.25-m^2 metal quadrat was randomly tossed at the sampling sites to obtain measurements of seagrass and macroalgae areal cover. The percent cover of seagrass and macroalgae was estimated in the quadrat by a diver using a scale of 0 to 100 in increments of 5. Subsequently, the length of up to five seagrass blades was measured, and the mean values were recorded. The diver then visually inspected the seagrass bed within the quadrat for evidence of grazing, boat scarring, macroalgae, epiphytic loading, wasting disease, and shellfish (hard clams and bay scallops).

Core Sampling

Coring methods also followed those of Short et al. (2002) using a 10-cm (0.00785 m^2) diameter PVC coring device to collect the seagrass samples, with care taken not to cut or damage the aboveground plant tissues. The diver-deployed corer extended deep enough in the sediments to extract all belowground fractions (roots and rhizomes). Each core was placed in a 3 x 5 mm mesh bag and rinsed to separate plant material from the sediment. After removing the seagrass from the mesh bag, the sample was placed in a labeled bag and stored on ice in a closed container prior to transport back to the Rutgers University Marine Field Station (RUMFS) in Tuckerton.

Laboratory Sample Processing

In the laboratory, the samples collected in the field were carefully sorted and separated into aboveground (shoots) and belowground (roots and rhizomes) components. The density of seagrass shoots was subsequently determined. The aboveground and belowground fractions were then oven dried at 50-60ºC for a minimum of 48 hours. The dry weight biomass (g dry wt m^{-2}) of each fraction was then measured to the third decimal place more than once to ensure samples were completely dry. Epiphyte cover and biomass were also determined. These protocols were followed for all seagrass samples collected in the field.

Statistical Analysis

Water quality and seagrass data were collected in Transects 13, 14, and 15 during each of the three time periods (1 = June/July, 2 = August/September, and 3 = October/November) in 2011. Mean and standard deviations for each variable were calculated for each Transect at each Time Period (Proc MEANS, SAS Inc.). Data were tested for normal distributions with the Shapiro-Wilk statistic, histograms, and box plots (Proc UNIVARIATE, SAS Inc.). Variables that exhibited normal distributions were tested for significant differences ($\alpha = 0.05$) between transects with repeated-measures ANOVA (Proc GLM, SAS Inc.). Mauchly's criterion was used to test the assumption of sphericity, and if violated (i.e. the Chi-square approximation had an

associated p value < 0.05), then the multivariate Wilks' Lambda statistic was used. If sphericity was not violated, univariate tests were used. Cases of non-normally distributions of data were ranked (Proc RANK, SAS Inc.) before conducting two-way ANOVA (Proc GLM, SAS Inc.), which produces the equivalent results to Friedman's test (also called two-way analysis for block designs) and does not assume normal distribution. Least squares means were used to identify significant differences in cases of significant interactive effects.

RESULTS: NORTH SEGMENT

Data Summary

Mean and standard deviation values for water quality, *Zostera marina*, *Ruppia maritima*, and other parameters collected in Transects 13-15 during the three time periods (1 = June-July, 2 = August-September, 3 = October-November) in 2011 are shown in Table 2.

Normality

Dissolved oxygen concentration and percent saturation, as well as pH and Secchi depth were normally distributed during all time periods. Temperature, salinity, and all seagrass variables (both *Ruppia maritima* and *Zostera marina*) were non-normally distributed during all time periods.

Water Quality

Mean dissolved oxygen concentrations were greater than 7.0 mg L^{-1} at all transects during all time periods except at Transect 13 during Time Period 2 when the mean dissolved oxygen concentration was 6.96 mg L^{-1}. The highest mean dissolved oxygen concentration (> 10 mg L^{-1}) occurred at Transect 13 and Transect 14 during Time Period 3. Dissolved oxygen concentrations met the assumption of sphericity (chi-square = 3.39, p = 0.1838). There was a significant interaction ($p < 0.01$) between Time Period and Transect (Table 3). During Time Period 1, Transect 15 had significantly higher dissolved oxygen than either Transect 13 or Transect 14 (which did not differ). During Time Period 2, all three transects significantly differed from each other. During Time Period 3, Transect 15 had significantly lower dissolved oxygen than either Transect 13 or Transect 14 (which did not differ).

The mean dissolved oxygen percent saturation was highest at Transect 13 during Time Period 3 (115.25%) and at Transect 15 during Time Period 1 (111.74%) and Time Period 2 (113.05%). The lowest mean dissolved oxygen percent saturation was recorded at Transect 13 during Time Period 2 (87.45%). Dissolved oxygen percent saturation met the assumption of sphericity (chi-square = 2.47, p = 0.2901). There was a significant interaction ($p < 0.01$) between Time Period and Transect (Table 3). During Time Period 1, Transect 15 had significantly higher dissolved oxygen than either Transect 13 or Transect 14 (which did not differ). During Time Period 2 and Time Period 3, all three transects significantly differed from each other.

The mean pH measurements exceeded 7.0 at all transects during all time periods. The lowest mean pH value was 7.43 at Transect 13 during Time Period 2. The highest mean pH value was 8.34 at Transect 15 during Time Period 1.

Values of pH violated the assumption of sphericity (chi-square = 20.98, $p < 0.0001$). There was a significant interaction between Time Period and Transect (S = 2, M = -0.5, N = 12, DF = 4,52, F = 40.09, Wilks' Lambda < 0.01). During Time Period 1, Transect 13 had lower pH than either Transect 14 or Transect 15 (which did not differ). In Time Period 2 and in Time Period 3, pH differed between all transects.

Secchi depth measurements were unlimited at all transects and during all time periods. Secchi depth was not tested for sphericity due to insufficient error degrees of freedom, but was assumed to not violate this assumption. There was a significant interaction between Time Period and Transect ($p < 0.01$, Table 3). During Time Period 2, Secchi depth in Transect 15 was larger than either Transect 13 or Transect 14 (which did not differ).

Water temperature trends were consistent across the three time periods of sampling (Table 2). Mean temperature values were highest during Time Period 1 and lowest during Time Period 3 at all transects. The mean temperature was highest at Transect 15 during Time Period 1 (24ºC) and lowest at Transect 15 during Time Period 3 (13.9ºC).

Maximum temperatures during Time Period 1 were 23.8ºC, 23.9ºC, 24.2ºC in Transects 13, 14, and 15, respectively. During Time Period 2, maximum temperatures were 22.8ºC, 23.3ºC, and 23.3ºC in Transects 13, 14, and 15, respectively. The maximum temperatures during Time Period 3 were 15.7ºC, 14.4ºC, and 14.0ºC in Transects 13, 14, and 15, respectively.

Temperature significantly varied overall (DF = 8,81, F = 119.61, $p < 0.01$), and there was a significant interactive effect between Transect and Time Period, according to Friedman's test (Table 4). All Transects had significantly different temperatures ($p < 0.05$) from each other during all Time Periods, except Transect 13 during Time Period 1 and Transect 15 during Time Period 2 ($p = 0.43$).

Mean salinity values were less than 20 at all transects during all time periods. These values reflect the greater freshwater discharges of the Toms River and Metedeconk River basins than the freshwater discharges elsewhere in the system. Hence, salinity is lower in the north segment than in the central and south segments.

Salinity in Transect 13 ranged 19.4 to 19.9 in Time Period 1, 13.0 to 15.3 in Time Period 2, and 18.5 to 18.9 in Time Period 3. Salinity in Transect 14 ranged 18.9 to 19.2 in Time Period 1, from 15.6 to 16.8 in Time Period 2, and from 18.7 to 18.9 in Time Period 3. Salinity in Transect 15 ranged from 18.8 to 19.0 in Time Period 1, 15.9 to 16.5 in Time Period 2, and from 18.4 to 18.8 in Time Period 3.

Salinity significantly varied overall (df = 8,81, F = 186.43, $p < 0.01$), and there was a significant interactive effect between Transect and Time Period, according to Friedman's test (Table 4). All Transects had significantly different salinities ($p < 0.05$) from each other at all Time Periods, with two exceptions. Transects 13 and 14 did not significantly differ in salinity during Time Period 3 ($p = 0.58$). Transects 14 and 15 did not significantly differ in salinity during Time Period 2 ($p = 0.54$).

Mean specific conductivity exceeded 20 µS cm^{-1} at all transects during all time periods, and most measurements exceeded 30 µS cm^{-1}. Specific conductivity significantly varied overall (DF = 8,81, F = 206.47, $p < 0.01$), and there was a significant interactive effect between Transect and Time Period, according to Friedman's test (Table 4). All Transects had significantly different specific conductivities ($p < 0.05$) from each other at all Time Periods, with two exceptions. Transects 13 and 14 did not significantly differ in specific conductivity during Time Period 3 ($p = 0.15$). Transects 14 and 15 did not significantly differ in specific conductivity during Time Period 2 ($p = 0.65$).

Zostera marina

Zostera marina was only found in the north segment at Transect 14 and Transect 15 during Time Period 1. It was absent at those transects during later Time Periods. Mean aboveground and belowground biomass values of *Z. marina* were very low, exceeding 1 g dry wt. m^{-2} only at Transect 15 during Time Period 1 (mean aboveground biomass = 1.29 g dry wt. m^{-2}; belowground biomass = 7.18 g dry wt. m^{-2}).

Zostera marina aboveground biomass did not significantly vary in the overall ANOVA, according to the Friedman test (DF = 8,81, F = 1.62, $p = 0.13$, Table 4). *Z. marina* belowground biomass did significantly vary overall (DF = 8,81, F = 4.42, $p < 0.01$), and there was a significant interactive effect between Transect and Time Period, according to Friedman's test. Belowground biomass at Transect 15 during Time Period 1 significantly differed ($p < 0.01$) from all other Transects during all Time Periods. Note, though, that the only other time *Z. marina* was only found at Transect 14 was during Time Period 1.

Zostera shoot density did not significantly vary in the overall ANOVA, according to the Friedman test (DF = 8,81, F = 1.64, $p = 0.13$) (Table 4).

Zostera percent cover did not significantly vary in the overall ANOVA, according to the Friedman test (DF = 8,81, F = 1.00, $p = 0.44$) (Table 4).

An insufficient number of *Zostera* samples were collected to statistically test for differences between Transects and Time Periods for blade length and width.

Ruppia maritima

Ruppia maritima was found in all three Transects in the north segment at all Time Periods of 2011. It dominated the seagrass beds in the north segment, being far more abundant than *Zostera marina*. Mean aboveground biomass of *R. maritima* was highest at Transect 13 during Time Period 1 (19.59 g dry wt. m^{-2}) and lowest at Transect 14 during Time Period 2 (1.97 g dry wt. m^{-2}). Mean belowground biomass of *R. maritima* was highest at Transect 15 during Time Period 1 (26.39 g dry wt. m^{-2}) and lowest at Transect 15 during Time Period 2 (3.23 g dry wt. m^{-2}).

Ruppia maritima aboveground biomass did significantly vary overall (DF = 8,81, F = 3.98, $p < 0.01$), and there was no significant interactive effect between Transect and Time Period ($p = 0.88$), according to Friedman's test (Table 4). Both Transect and Time Period main effects were significant ($p = 0.02$ and $p < 0.01$), respectively. Aboveground biomass of *R. maritima* at Transect 13 was significantly different ($p < 0.01$) from that in Transect 14. Aboveground biomass of *R. maritima* during Time Period 1 significantly differed from that during Time Periods 2 and 3 (both $p < 0.01$).

Belowground biomass of *Ruppia maritima* did significantly vary overall (DF = 8,81, F = 4.89, $p < 0.01$), and there was no significant interactive effect between Transect and Time Period ($p = 0.40$), according to Friedman's test (Table 4). Both Transect and Time Period main effects were significant ($p < 0.05$ and $p < 0.01$), respectively. Belowground biomass of *R. maritima* at Transect 13 was significantly different ($p = 0.01$) from that at Transect 14. Belowground biomass of *R. maritima* during Time Period 1 significantly differed from that during Time Periods 2 and 3 (both $p < 0.01$).

Ruppia maritma shoot density did significantly vary overall (DF = 8,81, F = 3.60, $p < 0.01$), and there was no significant interactive effect between Transect and Time Period ($p = 0.91$), according to Friedman's test (Table 4). Both Transect and Time Period main effects were significant (both $p < 0.01$). Shoot density of *R. maritima* at Transect 13 was significantly different ($p < 0.01$) from that at Transect 14 and Transect 15. Shoot density of *R. maritima* during Time Period 1 significantly differed from that during Time Periods 2 and 3 (both $p < 0.01$).

Ruppia maritima percent cover did significantly vary overall (DF = 8,81, F = 3.34, $p < 0.01$), and there was no significant interactive effect between Transect and Time Period ($p = 0.38$), according to Friedman's test (Table 4). Both Transect and Time Period main effects were significant ($p = 0.01$ and $p < 0.01$, respectively). *R. maritima* percent cover at Transect 13 was significantly different ($p < 0.01$) from that at Transect 14. *R. maritima* percent cover during Time Period 1 significantly differed from that during Time Periods 2 and 3 ($p = 0.02$, and $p < 0.01$, respectively). Insufficient data were collected to statistically test width of *R. maritima* samples.

Macroalgae and Other Cover

Macroalgae percent cover was recorded at all three transects, but only during Time Period 1. The highest mean percent cover of macroalgae was found at Transect 15 (23%) and the lowest at Transect 14 (8%).

Macroalgae percent cover did significantly vary overall (DF = 8,81, F = 7.81, $p < 0.01$), and there was no significant interactive effect between Transect and Time Period ($p = 0.34$), according to Friedman's test (Table 4). Transect main effects were not significantly different ($p = 0.31$). Time Period main effects were significantly different ($p < 0.01$). Time Period 1 significantly differed from Time Period 2 and Time Period 3 (both $p < 0.01$).

Very low levels of other percent cover were found in the north segment. The highest mean percent other cover occurred at Transect 15 during Time Period 3 (1%), and the lowest mean percent other cover occurred at Transect 13 during Time Period 3 (0.5%). The percent other cover did not significantly vary in the overall ANOVA, according to the Friedman test (DF = 8,81, F = 1.26, p = 0.28) (Table 4).

RESULTS: ESTUARY-WIDE

Physicochemical Parameters

Water temperature during the June-July sampling period (mean = 23.5ºC) was lower than that during the August-September sampling period (mean = 25.6 ºC). It decreased markedly (mean = 16.1ºC) during the October-November sampling period (Table 5). Salinities were in the polyhaline range, with mean values of 25.4‰ and 24.9‰ registered during the June-July and August-September sampling periods, respectively. Mean salinity increased to 25.5‰ during the October-November sampling period. Salinity variation was highest during the August-September sampling period (Table 5).

Dissolved oxygen (DO) ranged from 5.8 to 11.7 mg L^{-1} during June-July, from 4.2 to 11.8 mg L^{-1} during August-September, and from 6.1 to 14.4 mg L^{-1} during October-November. Mean dissolved oxygen values amounted to 8.2 mg L^{-1} during the June-July sampling period and 7.2 mg L^{-1} during the August-September sampling period. Highest DO levels (mean = 9.3 mg L^{-1}) were recorded during the October-November period (Table 5).

The pH values were consistent across the survey area. The mean pH readings in the north segment ranged from a low of 7.7 during the August-September sampling period to a high of 8.2 during the June-July sampling period. The mean pH measurements in the central segment ranged from 7.9 to 8.1, with highest pH values recorded during the June-July sampling period. In the south segment, the mean pH values ranged from 7.9 to 8.0; higher pH values were recorded during the June-July and October-November sampling periods than during the June-July sampling period (Table 5).

Secchi measurements increased across sampling periods. In June-July, the mean Secchi reading amounted to 0.86 m. Higher Secchi values (mean = 1.05 m) were recorded during the August-September sampling period. The highest Secchi measurements (mean = 1.2 m) were found during the October-November sampling period (Table 5).

Widgeon Grass (*Ruppia maritima*)

Ruppia maritima was most abundant in the north segment of the estuary during 2011. It was essentially absent in the south segment. Density, biomass, and areal cover of widgeon grass varied considerably both in space and time during the 2011 study period (Tables 6 and 7).

Aboveground Biomass

Aboveground biomass of *Ruppia maritima* in the estuary peaked during the June-July sampling period (mean = 4.4 g dry wt m^{-2}), with lowest values (mean = 2.0 g dry wt m^{-2}) recorded during the August-September sampling period. Intermediate aboveground biomass values (mean = 3.7 g dry wt m^{-2}) were documented during the October-November sampling period (Table 6).

The mean aboveground biomass of *Ruppia maritima* was highest in the north segment; the mean values in this segment in June-July, August-September, and October-November were 13.3 g dry wt m^{-2}, 3.5 g dry wt m^{-2}, and 7.7 g dry wt m^{-2}, respectively. The aboveground biomass values of *R. maritima* were much lower in the central segment; here, the mean values in June-July, August-September, and October-November were 4.4 g dry wt m^{-2}, 3.2 g dry wt m^{-2}, and 5.4 g dry wt m^{-2}, respectively (Table 7).

Belowground Biomass

Belowground biomass of *Ruppia maritima* decreased progressively over the study period. The highest mean belowground biomass of widgeon grass was observed during the June-July sampling period (5.5 g dry wt m^{-2}), and the lowest mean belowground biomass was found during the October-November sampling period (2.6 g dry wt m^{-2}). An intermediate mean belowground biomass value occurred during the August-September sampling period (3.0 g dry wt m^{-2}) (Table 6). The mean belowground biomass of *R. maritima* was by far highest in the north segment, amounting to 19.5 g dry wt m^{-2}, 4.9 g dry wt m^{-2}, and 4.9 g dry wt m^{-2} in June-July, August-September, and October-November sampling periods (Table 7).

Shoot Density

The highest *Ruppia maritima* density (shoots m^{-2}) measurements were recorded during the October-November sampling period (mean = 1313 shoots m^{-2}). Significantly lower densities of *R. maritima* were found during the June-July (mean = 1167 shoots m^{-2}) and August-September (mean = 1002 shoots m^{-2}) sampling periods (Table 6). In the north segment, the mean *R. maritima* shoot density was highest during the June-July sampling period (4584 shoots m^{-2}) and lowest during the October-November sampling period (2979 shoots m^{-2}). An intermediate value was recorded during the August-September sampling period (2097 shoots m^{-2}) (Table 7).

Areal Cover

The areal cover of *Ruppia maritima* was relatively consistent across sampling periods. The highest mean percent areal cover was found during the August-September sampling period (9.3%), and the lowest mean percent areal cover, during the October-November sampling period (6.5%). An intermediate mean percent areal cover value was recorded during the June-July sampling period (8.3%) (Table 6).

While areal cover of *R. maritima* was relatively consistent across sampling periods, it was significantly different across sampling segments. For example, the mean areal cover of widgeon grass was highest in the north segment; the mean values in this segment in June-July, August-September, and October-November were 33.0%, 15.5%, and 15.5%, respectively. The mean areal cover values of *R. maritima* were generally much lower in the central segment; here,

the mean values in June-July, August-September, and October-November were 4.2%, 15.4%, and 8.8%, respectively (Table 7). This difference reflects the preference of widgeon grass for the lower salinity waters of the north segment.

Eelgrass (*Zostera marina* L.)

The biomass, shoot density, areal cover, and blade length of eelgrass (*Zostera marina*) varied both spatially and temporally in the estuary during 2011 (Tables 6 and 7). This variation in plant characteristics was most evident when comparing eelgrass in the north segment to that in the central and south segments. Only a small amount of *Z. marina* occurred in the north segment during the June-July sampling period at Transect 14 (mean aboveground biomass = 0.16 g dry wt m^{-2}; mean belowground biomass = 0.72 g dry wt m^{-2}) and Transect 15 (mean aboveground biomass = 1.29 g dry wt m^{-2}; mean belowground biomass = 7.18 g dry wt m^{-2}) and none in this segment during the other sampling periods. A marked increase in *Z. marina* was observed in the central and south segments (Table 7).

Aboveground Biomass

Aboveground biomass of *Zostera marina* in the estuary increased during each sampling period, peaking during the October-November sampling period (mean = 17.4 g dry wt m^{-2}), when the variation of biomass measurements was also greatest. Lowest values (mean = 7.2 g dry wt m^{-2}) were recorded during the June-July sampling period. Intermediate aboveground biomass values (mean = 9.4 g dry wt m^{-2}) were documented during the August-September period (Table 6).

The mean aboveground biomass of *Zostera marina* was highest in the central segment; the mean values in this segment in June-July, August-September, and October-November were 12.4 g dry wt m^{-2}, 8.5 g dry wt m^{-2}, and 26.6 g dry wt m^{-2}, respectively. Somewhat lower values were recorded in the south segment. Here, the mean aboveground biomass values of *Z. marina* in June-July, August-September, and October-November amounted to 5.3 g dry wt m^{-2}, 14.9 g dry wt m^{-2}, and 17.0 g dry wt m^{-2}, respectively (Table 7).

Belowground Biomass

Belowground biomass of *Zostera marina* was generally higher than the aboveground biomass. It decreased gradually over the study period. The highest mean belowground biomass of *Z. marina* samples was observed during the June-July sampling period (21.4 g dry wt m^{-2}), and the lowest mean belowground biomass was found during the October-November sampling period (15.5 g dry wt m^{-2}). An intermediate mean belowground biomass value was documented during the August-September sampling period (15.7 g dry wt m^{-2}) (Table 6).

Belowground biomass of *Zostera marina* in 2011 was extremely low in the north segment, where *Ruppia maritima* dominated the samples. While a mean belowground biomass value of 2.6 g dry wt m^{-2} was recorded in the north segment during the June-July sampling period, no *Z. marina* was found at the north segment stations during the August-September and October-November sampling periods. Belowground biomass values were similar in the central and south segments (Table 7). The mean belowground biomass values of *Z. marina* in the central segment in June-July, August-September, and October-November were 33.5 g dry wt m^{-2}, 11.6 g

dry wt m^{-2}, and 18.0 g dry wt m^{-2}, respectively. The mean belowground biomass values of *Z. marina* in the south segment in June-July, August-September, and October-November were 18.6 g dry wt m^{-2}, 27.7 g dry wt m^{-2}, and 20.8 g dry wt m^{-2}, respectively.

Shoot Density

Shoot density of *Zostera marina* was relatively low throughout the study period in 2011. For example, in the north segment, the mean shoot density during the June-July sampling period was only 38 shoots m^{-2}, and it dropped to 0 during the remaining sampling periods. In the central segment, the mean shoot density was 250 shoots m^{-2} in June-July, 161 shoots m^{-2} in August-September, and 240 in October-November. In the south segment, the mean shoot density was 123 shoots m^{-2} in June-July, 212 shoots m^{-2} in August-September, and 208 in October-November (Table 7). These shoot densities are much lower than those reported for *Z. marina* in the estuary during 2010.

Blade Length

The highest mean length of *Zostera marina* blades was recorded in the central segment during the October-November sampling period (31.9 cm) and the August-September sampling period (31.3 cm) (Table 7). Mean *Z. marina* blade length was also high during the October-November sampling period (31.1 cm) in the south segment. The lowest mean *Z. marina* blade length by far was found in the north segment during the June-July sampling period (15.7 cm). The north segment is a less favorable area for *Z. marina* settlement and growth. The mean blade lengths of *Z. marina* in 2011 were comparable to those recorded in 2005 and 2008, lower than those in 2004, and higher than those in 2006, 2009, and 2010.

Areal Cover

The mean percent cover of *Zostera marina* during sampling periods in June-July, August-September, and October-November was 19.7%, 17.9%, and 16.1%, respectively (Table 6). The highest percent cover of *Z. marina* in the central segment was recorded during the June-July sampling period (mean = 28.3%). In the south segment, the highest percent cover of *Z. marina* was found during the August-September sampling period (mean = 27.6%). The lowest percent cover was documented in the north segment (Table 7). Areal cover of *Z. marina* in the central and south segments during 2011 was much lower than that during 2004 and comparable to that observed from 2005 to 2010.

Macroalgae

Areal Cover

The mean percent cover of macroalgae in 2011 ranged from 1 to 7.9% (Table 6). The lowest mean percent cover of macroalgae occurred during the October-November sampling period, and the highest percent cover occurred during the June-July sampling period. Percent cover during August-September was only slightly higher (mean = 1.1%) than during October-November. These values are comparable to those recorded in the estuary during 2010, but generally less than those recorded for prior years between 2004 and 2009.

Macroalgal areal cover was highest during the June-July sampling period in the north segment (mean = 13.3%) and central segment (mean = 12.5%). Much lower macroalgal percent cover was evident during other sampling periods in all three estuarine segments (Table 8). In addition, other biotic material also covered small areas of the estuarine floor ranging in mean values from 0 to 1.0% (Table 8).

Epiphytes

In 2011, epiphyte percent cover on eelgrass leaves was highest during the August-September sampling period when the mean percent cover amounted to 48.1% on upper leaf surfaces and 48.0% on lower leaf surfaces. Much lower epiphyte percent cover was recorded on eelgrass leaves during the other sampling periods. For example, in June-July 2011, the mean percent cover of epiphytes on the upper leaf surfaces of eelgrass was only 9.1% compared to 8.6% on the lower lower leaf surfaces. These values were similar to those recorded for eelgrass leaves during the October-November sampling period when the mean percent cover of epiphytes on upper leaf surfaces was 9.7% compared to 9.0% on lower leaf surfaces (Table 9).

Epiphyte biomass on eelgrass leaves in 2011 peaked during the August-September sampling period (mean = 144.0 mg dry wt m^{-2}). Much lower epiphyte biomass on eelgrass leaves was recorded during the June-July (mean = 41.3 mg dry wt m^{-2}) and October-November (mean = 69.4 mg dry wt m^{-2}) sampling periods (Table 9).

DISCUSSION

Seagrass is an excellent indicator of estuarine water and sediment quality, as well as overall ecosystem health (Moore 2004, 2009; Duarte et al., 2006; Lamote and Dunton, 2006; Moore and Short, 2006; Kennish et al., 2008, 2009, 2010). Data collected on seagrass biomass, shoot density, areal cover, and blade length provide vital information necessary to assess key biotic responses in an estuary affected by nutrient enrichment and organic carbon loading. Rigorous and consistent sampling is needed to determine environmental trends and establish cause-and-effect relationships. The habitat requirements of seagrass can be delineated by concurrently measuring water quality and the biotic responses of the seagrass species (Moore and Short, 2006; Burkholder et al., 2007). Data collected on seagrass characteristics are also valuable for developing management strategies to restore impaired seagrass habitat.

Results of the 2011 estuary-wide seagrass survey in the BB-LEH Estuary have yielded a number of important findings. Data collected on demographic trends indicate that eelgrass beds in 2011 had yet to recover from the marked decline of plant biomass and areal cover observed in 2009 and 2010 (Kennish et al., 2009, 2010). The general decline in plant parameters observed in 2009 and 2010 continued in 2011. Eelgrass biomass and areal cover generally decreased through 2010, but the decline in plant biomass, a key water quality indicator was most marked. A general decline in plant parameters was evident from 2008 to 2010 corresponding with temporal separation (yearly and seasonally of environmental parameters suggests their importance to seagrass condition).

The north, central, and south segments of BB-LEH Estuary are in different stages of degradation and eutrophication. The north segment, which has experienced the highest levels of nutrient loading, has already undergone severe degradation and eutrophication. In this segment, *Ruppia maritima* is the dominant seagrass species. The lower aboveground biomass of *R. maritima* in the central segment is attributed to the higher salinity there and the preference of widgeon grass for lower salinity waters to the north. This segment remains in a highly eutrophic condition. The central and south segments are similar to each other, and over the 1989-2010 period, both have undergone gradual, albeit significant decline in condition associated with eutrophication (Kennish et al., 2012). In these two segments, *Zostera marina* is the dominant seagrass species.

Zostera marina is the dominant seagrass species overall in BB-LEH in terms of biomass and areal cover. While it is widely found in the central and south segments, it is very sparse in the north segment. As confirmed by sampling in the north segment during 2011, which recorded *Z. marina* during only one sampling period (June-July) in two transects (14 and 15). These findings corroborate those of Lathrop et al. (2006), who showed conclusively via aerial and *in situ* surveys that *Ruppia maritima* is the overwhelmingly dominant seagrass in the north segment.

The structure of seagrass beds in the north segment is markedly different than those in the central and south segments. While *Ruppia maritima* was found at all transects during all time periods in the north segment, the overall condition of *R. maritima* does not appear to be strong estuary-wide. Because only one year of data (2011) has been collected on this seagrass species in the north segment since 2004, there is no way to validate its condition without additional years of sampling in this segment. Previous years of sampling in the central and south segments, however, reveal conclusively that *R. maritima* is depauparate in these areas, with mean biomass values ≤ 1.6 g dry wt m^{-2} during all sampling periods in 2005 and 2010, when the only samples of *R. maritima* were found. Somewhat higher aboveground and belowground biomass values of *R. maritima* were recorded in 2011, especially in the more favorable environment of the north segment. However, no *R. maritima* samples were found in the south segment during 2011. These data demonstrate clearly that *R. maritima* dominates seagrass beds only in the north segment, while *Z. marina* dominates the beds in all other areas.

Since 2004, the condition of eelgrass beds in the estuary has generally declined estuary-wide (Kennish et al., 2010). Eelgrass biomass decreased substantially over the 2004-2006 and 2008-2010 periods and overall from 2004-2010 (Figures 3 and 4). The biomass in 2010 was the lowest recorded for BB-LEH. The rate of decline in aboveground and belowground eelgrass biomass was significantly sharper during 2004-2006 than in 2008-2010. Aboveground and belowground biomass varied considerably in the central and south segments of the estuary.

Aboveground eelgrass biomass peaked in June-July 2004 (mean = 109.5 g dry wt m^{-2}), and then declined to lowest levels in October-November 2010 (mean = 2.7 g dry wt m^{-2}). For all sampling years, aboveground biomass measurements were highest in 2004, 2005, and 2008 and lowest in 2006, 2009, and 2010 (Table 10). Belowground eelgrass biomass was a maximum in June-July 2005 (142.7 g dry wt m^{-2}) and a minimum in October-November 2009 (17.1 g dry wt m^{-2}). Similar to aboveground biomass measurements, belowground biomass measurements were

highest in 2004, 2005, and 2008 and lowest in 2006, 2009, and 2010. Overall, seagrass biomass in the estuary decreased markedly over the 2004–2006 period, and by 2010 it had dropped to a mean of 7.7 g dry wt m^{-2} (aboveground) and 27.0 g dry wt m^{-2} (belowground), which were the lowest levels ever recorded. Reduced biomass levels have persisted through 2011.

Though long-term monitoring was not started early enough to observe the beginning of the initial seagrass decline prior to 2004, the pattern of biomass decline with increasing nutrient concentrations is similar to load-decline relationships described in the literature (Nixon 1995; Cloern, 2001; Burkholder et al. 2007), and nitrogen concentrations in BB-LEH are proportional with nitrogen loading from subwatersheds. The trend of eelgrass decline over the years has not been isolated to one bed but has been estuary-wide, signaling a response to a broad-scale stressor that adversely affects plant condition across the system. Nutrient loading and eutrophication have been clearly identified as the primary drivers of change in eelgrass habitat of the estuary (Kennish et al., 2008, 2010).

Table 11 shows characteristics of widgeon grass (*Ruppia maritima*) sampled in the BB-LEH Estuary during the 2004-2010 period. Since most widgeon grass is found in the north segment of the estuary, its biomass, shoot density, and areal cover values were low for the central and south segments (Figures 5 and 6). It is important to note that widgeon grass predominates over eelgrass in the north segment of the estuary, and this segment was only sampled in 2011 and not during the previous six years.

The most complete data sets for widgeon grass in the central and south segments were reported in 2005 and 2010 (Table 11). Both aboveground and belowground biomass values were low. The mean aboveground biomass ranged from 0 to 1.6 g dry wt m^{-2} during these two years of sampling; the mean belowground biomass in turn ranged from 0.1 to 1.5 g dry wt m^{-2}. Shoot densities were most consistent during 2010 when mean values gradually increased from 331 ± 231 shoots m^{-2} in June-July, 450 ± 249 shoots m^{-2} in August-September, and 499 ± 366 shoots m^{-2} in October-November. Mean areal percent cover in turn was usually less than 10%, with peak cover recorded in August-September 2005 (19.6%) and 2010 (10.8%).

Bologna et al. (2000) documented significant losses of seagrass in the estuary from the mid-1970s to 2000, which may have reduced the areal distribution of the beds by 30-60%, most significantly in Little Egg Harbor. A GIS spatial comparison analysis by Lathrop et al. (2001) suggested a contraction of the seagrass beds to shallow subtidal areas (< 2 m) during this 25-year time period due to diminished light availability in response to phytoplankton and macroalgal blooms, epiphytic overgrowth, and other factors. However, Lathrop and Haag (Center for Remote Sensing and Spatial Analysis, Rutgers University), conducting more recent aerial surveys (2003 and 2009) of seagrass distribution in the estuary, found an expansion of the bed boundaries by 69 ha during a time span when biomass and other plant demographics were dramatically declining (Lathrop and Haag, 2011). Short (2007) recently observed a similar pattern of eelgrass distribution and plant characteristics in Great Bay Estuary, New Hampshire.

Seagrasses have high light requirements, approaching 25% of incident radiation in some species (Orth et al., 2006); the minimum light requirements generally vary between 5 and 20% of the surface irradiance (Dennison et al., 1993). Hence, light attenuation in the water column due

to suspended particulates and dissolved substances, as well as resuspended detritial material, sediments, phytoplankton, and epiphytes on photosynthetic surfaces of the plants, can be detrimental to seagrass beds. Diminished light transmission to the estuarine floor can lead to the replacement of seagrass plants by opportunistic macroalgae (e.g., *Ulva* and *Enteromorpha*), filamentous epiphytic macroalgae, and benthic microalgae which require lower light intensities for survival (Hily et al., 2004; McGlathery et al., 2007). The resulting shift in the composition of bottom-up controls often resonates through upper trophic levels. The loss of seagrass habitat due to light attenuation affects trophic structure by reducing the abundance of herbivorous grazers that can control algal overgrowth (Burkholder et al., 2007). The resulting increase in algal epiphytes, therefore, may exacerbate seagrass decline via top-down effects (Heck and Valentine, 2007).

BB-LEH Estuary is susceptible to nutrient enrichment because it has a long water residence time (74 days in summer), low freshwater input (1.24×10^9 per day), and reduced flushing rate, which promote the accumulation and retention of nutrients within the system, plant uptake of the nutrients, and cascading biotic and habitat impacts. In this estuary, nutrients and other pollutants are concentrated and recycled repeatedly among system components in the water column, bottom sediments, and biota. Nutrients enter the estuary via surface-water discharge (as base flow or storm runoff), direct stormwater runoff, direct ground-water discharge, atmospheric deposition, ocean water entering inputs, and secondarily through the release of nitrogen contained in bottom sediments (Wieben and Baker, 2009). Nitrogen loading in particular stimulates a series of biotic responses. For example, the estuary has had a long history of serious nuisance and toxic algal blooms associated with nitrogen loading.

Eutrophy has progressively increased through time in the BB-LEH Estuary in concert with escalating watershed development. Urban land use in the BB-LEH watershed has increased dramatically over the past four decades, particularly in the northern part of the BB-LEH watershed. In 1972, urban land cover amounted to ~19%, but it increased to 25% of the watershed in 1995, 30% in 2006, and ~34% at present. By 2010, the watershed had 111,560 ac of urban land area compared to 78,781 ac in 1995. Agricultural land area amounted to 4,965 ac in 2010, down from 6,314 ac in 1995. Upland forest area in turn decreased from 158,147 ac in 1995 to 139,915 ac in 2010. Urban land area in the BB-LEH watershed is now more than 25 times greater than agricultural land area, and the trend is increasing (Data from the Center for Remote Sensing and Spatial Analysis, Rutgers University). Increasing urbanization of the watershed land surface leads to greater impervious cover and runoff to area streams and rivers discharging to BB-LEH, thereby promoting nutrient enrichment and other pollutant discharges to the estuary. According to Velinsky et al. (2010), there has been an estimated two-fold increase in nitrogen accumulation rates in the upper estuary since the 1950s.

Designated as a moderately eutrophic system in the early 1990s (Seitzinger et al., 2001), the estuary was later reclassified as highly eutrophic by application of NOAA's National Ecosystem Assessment Model and Nixon's Trophic Classification (Nixon, 1995; Bricker et al., 2007; Kennish et al., 2007). Nutrient enrichment and associated organic carbon loading in this shallow coastal lagoon have been linked to an array of cascading environmental effects such as excessive micro- and macroalgal growth, harmful algal blooms (HABs), reduced dissolved oxygen levels, impacted harvestable fisheries, and loss of essential habitat (e.g., seagrass and

shellfish beds) (Kennish et al., 2007; Kennish et al. 2010). Accelerated growth of drifting macroalgae (e.g., *Ulva lactuca*) has been particularly problematic, periodically producing extensive organic mats on the floor of the estuary that have posed a threat to essential benthic habitat (Kennish et al., 2008). At times, the rapid growth of other macroalgal species in the estuary, such as the rhodophytes *Agardhiella subulata*, *Ceramium* spp., *Champia parvula*, *Gracilaria tikvahiae*, and *Spyridia filamentosa*, has also created problems, as is evident in other coastal bays and lagoons as well (McGlathery et al., 2007). In addition, the decomposition of thick macroalgal mats promotes sulfide accumulation and the development of hypoxic/anoxic conditions in bottom sediments that can be detrimental to benthic infaunal communities. These biotic and physicochemical changes may lead to further deterioration of sediment and water quality, loss of biodiversity, and disruption of ecosystem structure, function, and integrity. Human uses of estuarine resources are also adversely affected. Changes in phytoplankton communities from diatom/dinoflagellate dominants to greater abundances of microflagellates, raphidophytes, and bloom forming pelagophytes (e.g., *Aureococcus anophagefferens*, the causative agent of brown tides) often lead to dramatic losses of shellfish resources (Livingston, 2000, 2003, 2006).

The net insidious effect of progressive eutrophication is the potential for the permanent alteration of biotic communities, serious ecosystem structural and functional changes, and greater ecosystem-level impacts (Burkholder et al., 2007; McGlathery et al., 2007; Kennish and de Jonge, 2012). For example, hard clam stocks in Little Egg Harbor declined by two-thirds between 1986 and 2001, and the hard clam harvest in the entire estuary declined by more than 99% between 1975 and 2005. Repeated brown tide (*Aureococcus anophagefferans*) blooms, which interfere with shellfish feeding and can significantly increase shellfish mortality, occurred nearly annually between 1995 and 2002 (Gastrich et al, 2004). Monitoring of HABs was discontinued in the estuary after 2004 (Olsen and Mahoney, 2001; Gastrich et al., 2004); however, a brown tide bloom was recorded in Little Egg Harbor in August 2010. Blooms of the sea nettle (*Chrysaora quinquecirrha*), possibly coupled to increasing eutrophic conditions, have likewise escalated in the estuary since 2004, being particularly serious during the summer of 2010 and greatly impacting human use of extensive areas in the northern segment of the estuary.

Human activities in the BB-LEH watershed linked to declining environmental conditions in the estuary are largely land use-land cover issues, and hence require effective watershed planning and management decisions for remediation. With population growth in the watershed exceeding 575,000 year-round residents (1.2 million people during the summer tourist season), adverse environmental effects on seagrass beds have continued to increase, especially as impervious cover and other land surface alteration in the watershed have facilitated the runoff of nutrients to receiving water bodies. Declining seagrass habitat in the estuary is coupled to nutrient loading from the Barnegat Bay watershed associated with ongoing development which increased from 19% in 1972 to ~34% in 2010. Land use-land cover change manifested by increasing impervious surface cover in the watershed accelerated nutrient loading to the estuary via nonpoint sources in recent years (Wieben and Baker, 2009). Currently, two-thirds of the total nitrogen load to the BB-LEH Estuary (431,000 kg yr^{-1}) derives from surface runoff of the watershed. Bowen et al. (2007) estimated that fertilizer nitrogen load alone comprised nearly 30% of the total nitrogen load from the watershed to the estuary. The total nitrogen load is expected to increase with increasing development of the watershed, unless aggressive

management actions are implemented, such as improved stormwater system controls and reduced impervious cover. Projections indicate that impervious surface cover of the Barnegat Bay watershed will increase to 12% of the land area at buildout (Lathrop and Conway, 2001).

Similar findings as reported here for the BB-LEH Estuary have been documented in other shallow coastal bays (Nixon et al., 2001). Beem and Short (2009) found significant estuary-wide decline in an array of eelgrass parameters (biomass, shoot density, canopy height, leaf area) in the Great Bay Estuary (NH) over the 2001-2007 period. They attributed this decline to the increase in watershed development and impervious surfaces contributing to greater runoff of sediments and nutrients into the estuary and an overall decrease in water quality. Short (2007) showed that despite the decrease in biomass of eelgrass beds in the estuary in 2005, eelgrass areal distribution increased slightly, a similar condition observed in this study.

Wazniak et al. (2007) observed a decrease in seagrass coverage in the Maryland coastal bays between 2001 and 2004, as did Orth et al. (2010) for Chincoteague Bay between 2001 and 2007. Both of these studies indicate that declining water quality conditions in these coastal bays are responsible for the diminishing distribution of the seagrass beds. Much of the nutrient enrichment in these systems derives from chicken farms and farmlands, while that in the BB-LEH system is attributed to urbanization of the coastal watershed. Continued monitoring -- together with conservation efforts to protect seagrass habitat in the bays and management initiatives to remediate altered watershed areas -- are necessary to effectively improve ecosystem conditions.

Nitrogen over-enrichment, when unchecked, causes significant disruption of ecosystem structure and function (Nixon, 1995; Nixon et al., 2001; Kennish et al., 2007). There is growing concern that escalating eutrophication will result in severe, long-term degradation of the BB-LEH Estuary that will be intractable (Duarte et al., 2009; Kennish et al., 2007; 2010). With population growth in the watershed expected to increase from the current population of more than 575,000 year-round residents (>1 million people during the summer tourist season) to ~850,000 people projected at buildout (~50% increase in year-round residents), aquatic environmental pressures will continue to mount, particularly as impervious cover and other land surface alteration in the watershed escalate and partition habitats, leading to the greater input of nutrients and other pollutants to the estuary.

CONCLUSIONS

Seagrass surveys conducted in the BB-LEH since 2004 have shown conclusively that *Zostera marina* (eelgrass) is the dominant seagrass species in the central and south segments of the estuary, and *Ruppia maritima* (widgeon grass) is the dominant form in the north segment. This project examined the characteristics of mixed seagrass beds (*R. maritima* and *Z. marina*) in the north segment of the BB-LEH Estuary during the June-November sampling period in 2011. State-of-the-art, targeted seagrass sampling was conducted in the north segment of the estuary using the same SeagrassNet protocols that were employed in prior seagrass surveys conducted in

the estuary since 2004. Data are also reported on seagrass surveys conducted in a separate study in the central and south segments of the estuary during the same time periods in 2011.

Results of this study clearly indicate that *Ruppia maritima* was far more abundant than *Z. marina* in the north segment of the estuary in 2011. Mean aboveground biomass of *R. maritima* was highest at Transect 13 during Time Period 1 (19.59 g dry wt. m^{-2}) and lowest at Transect 14 during Time Period 2 (1.97 g dry wt. m^{-2}). Mean belowground biomass of *R. maritima* was highest at Transect 15 during Time Period 1 (26.39 g dry wt. m^{-2}) and lowest at Transect 15 during Time Period 2 (3.23 g dry wt. m^{-2}).

By comparison, *Zostera marina* was found in the north segment only at Transect 14 and Transect 15 during Time Period 1 (June-July). It was absent at these transects during later time periods. Only a small amount of *Z. marina* occurred in the north segment during the June-July sampling period in 2011 at Transect 14 (mean aboveground biomass = 0.16 g dry wt m^{-2}; mean belowground biomass = 0.72 g dry wt m^{-2}). Similarly, only a small amount of *Z. marina* occurred in the north segment during the June-July sampling period in 2011 at Transect 15 (mean aboveground biomass = 1.29 g dry wt. m^{-2}; belowground biomass = 7.18 g dry wt. m^{-2}).

Previous years of sampling in the central and south segments of the estuary reveal conclusively that *Ruppia maritima* is depauparate in these areas, with mean biomass values ≤ 1.6 g dry wt m^{-2} during sampling periods in 2005 and 2010, when the only samples of *R. maritima* were recovered in these segments. Somewhat higher aboveground and belowground biomass values of *R. maritima* were recorded in 2011, especially in the more favorable environment of the north segment. However, no *R. maritima* samples were found in the south segment during 2011.

Since 2004, system eutrophy has generally worsened in BB-LEH, and the condition of the eelgrass habitat has markedly degraded, notably in the central and south segments. Eelgrass biomass declined over the 2004-2006 and 2008-2010 periods, more acutely during the 2004-2006 period. The 2010 eelgrass biomass values were the lowest levels recorded in the estuary. The loss of eelgrass biomass in the central and south segments since 2004 has eliminated habitat for hard clams, bay scallops (*Argopecten irradians*), and other benthic and demersal organisms. Only one year (2011) of comprehensive seagrass surveys has been conducted in the north segment of the estuary, and therefore additional years of sampling must be conducted to determine trends of seagrass condition in this segment.

Nutrient enrichment and eutrophication have been clearly identified as the primary drivers of change in eelgrass habitat of the estuary. Though long-term monitoring was not started early enough to observe the beginning of the initial decline of eelgrass in the estuary prior to 2004, the pattern of biomass decline with increasing nutrient concentrations is similar to load-decline relationships described in the literature, and nitrogen concentrations in BB-LEH are proportional with nitrogen loading from subwatersheds. The trend of eelgrass decline over the years has not been isolated to one bed but has been observed over much of the estuary, signaling a response to a broad-scale stressor that adversely affects plant condition across the system.

Eelgrass biomass had yet to recover by 2010 from the decline of plant abundance and biomass observed in 2006. Eelgrass biomass values for 2010 were the lowest on record for BB-LEH. Eelgrass biomass measurements in 2011 showed no improvement over those of the 2008-2010 period. Thus, biomass may be reaching a new, lower, steady state in the estuary. Return to previous levels of eelgrass biomass therefore may be difficult to attain.

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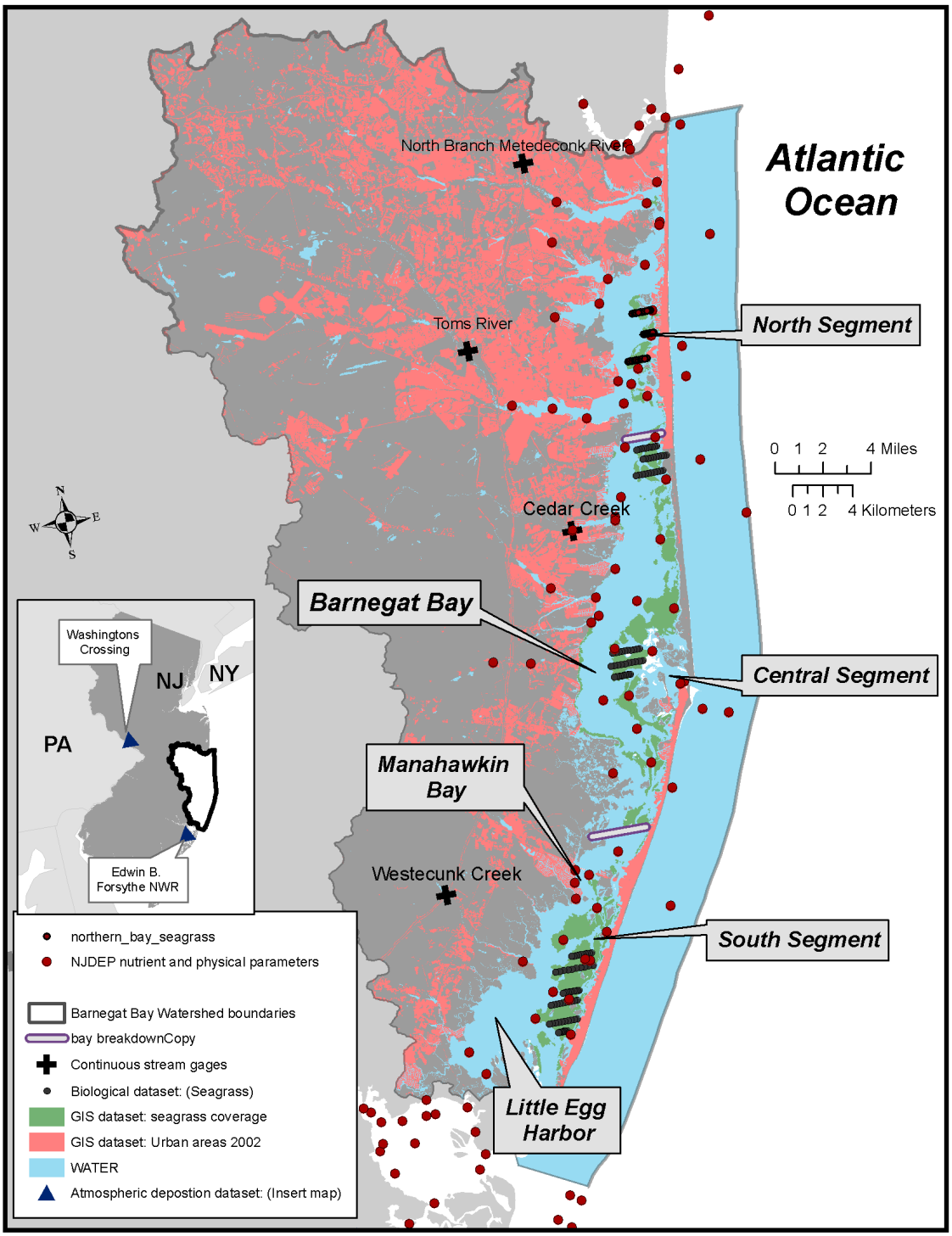

Figure 1. Graphic of the Barnegat Bay-Little Egg Harbor Estuary showing the north, central, and south segments and the seagrass sampling transects. Note north segment targeted in this study.

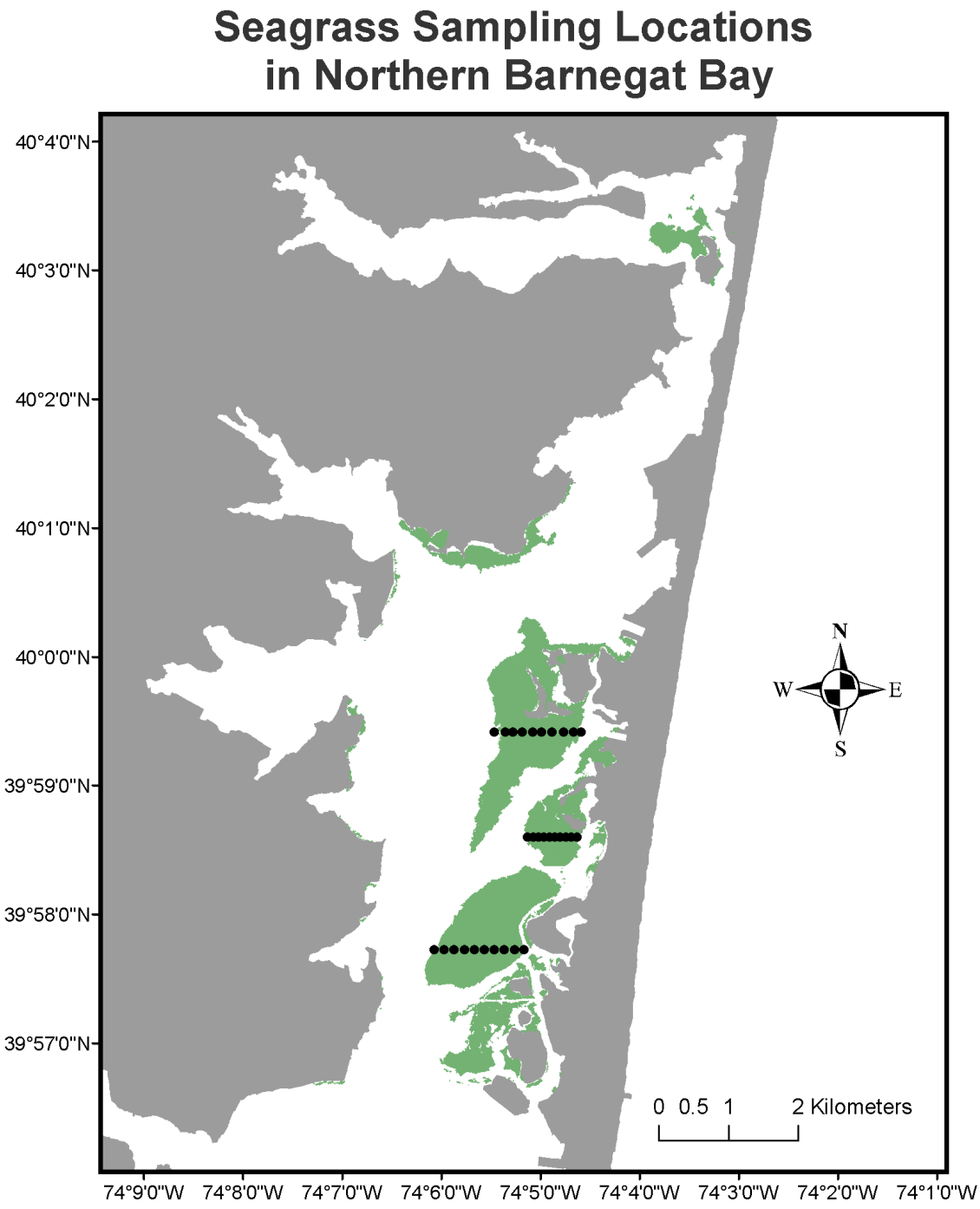

Figure 2. Sampling transects (13 southernmost, 14, and 15 northernmost) and stations in widgeon grass beds located in the north segment of the Barnegat Bay-Little Egg Harbor Estuary.

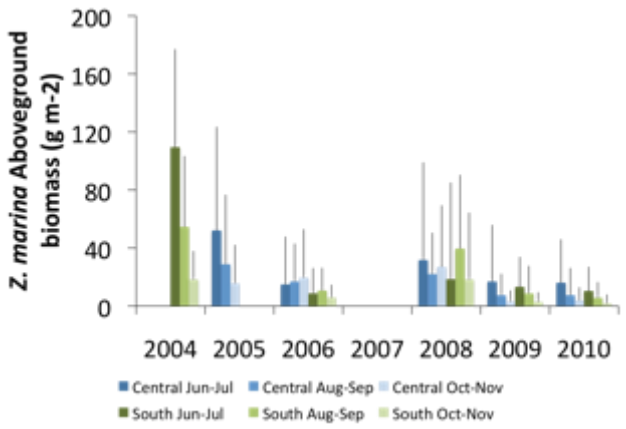

Figure 3. Mean aboveground eelgrass biomass values in the central and south segments of the Barnegat Bay-Little Egg Harbor Estuary during the spring-fall sampling periods from 2004 to 2010.

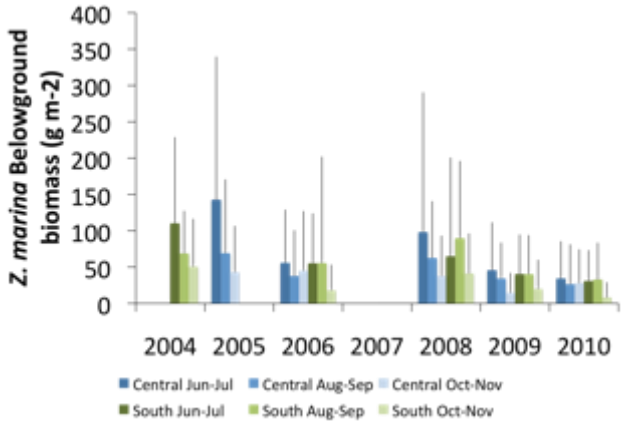

Figure 4. Mean belowground eelgrass biomass values in the central and south segments of the Barnegat Bay-Little Egg Harbor Estuary during the spring-fall sampling periods from 2004 to 2010.

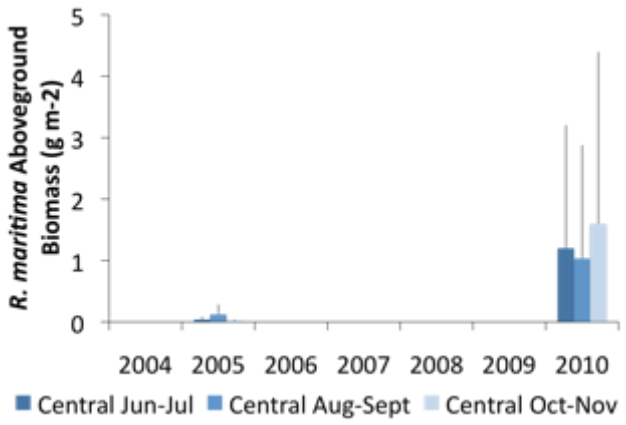

Figure 5. Mean aboveground widgeon grass biomass values in the central and south segments of the Barnegat Bay-Little Egg Harbor Estuary during the spring-fall sampling periods from 2004 to 2010.

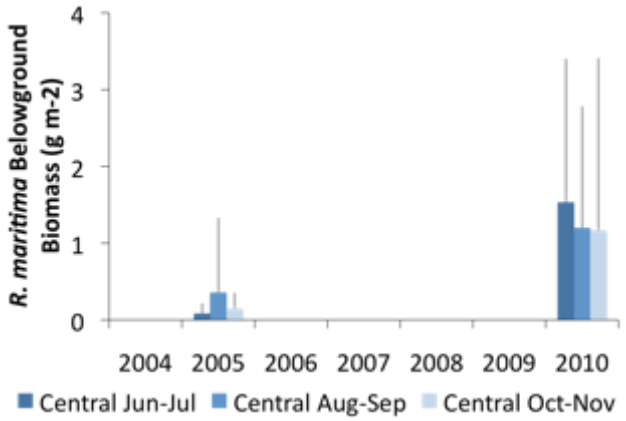

Figure 6. Mean belowground widgeon grass biomass values in the central and south segments of the Barnegat Bay-Little Egg Harbor Estuary during the spring-fall sampling periods from 2004 to 2010.

Table 1. Population in Ocean County from 1950 to 2010.

Year	Number of People
1950	56,622
1960	108,241
1970	208,470
1980	346,038
1990	433,203
2000	510,916
2010	576,567

Data Sources: Ocean County Cultural and Heritage Commission; U.S. Census

Table 2. Mean and standard deviation (SD) data for water quality (dissolved oxygen, pH, temperature, salinity, Secchi depth, specific conductivity), eelgrass (aboveground and belowground biomass, shoot density, percent cover, blade length, and blade width) and widgeon grass (aboveground and belowground biomass, shoot density, percent cover, and blade width) as well as percent cover of macroalgae and other organisms. Data are reported for the north segment of Barnegat Bay (transects 13-15), during June-July (Time Period 1), August-September (Time Period 2), and October-November (Time Period 3). Blank cells indicate no data are available.

		Transect 13						Transect 14						Transect 15					
	Time Period	1		2		3		1		2		3		1		2		3	
	N	10		10		10		10		10		10		10		10		10	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Dissolved Oxygen	mg L^{-1}	7.45	0.39	6.96	0.21	10.28	0.46	7.71	0.61	7.79	0.30	10.02	0.25	8.44	0.41	8.81	0.26	9.70	0.12
Dissolved Oxygen	%	97.39	5.24	87.45	3.05	115.25	5.51	101.70	8.58	99.56	4.00	109.91	2.83	111.74	5.44	113.05	3.39	105.28	1.41
pH		8.07	0.22	7.43	0.12	8.01	0.07	8.23	0.16	7.80	0.04	7.93	0.05	8.34	0.07	7.98	0.05	7.84	0.02
Temperature	°C	23.1	0.5	22.7	0.1	15.4	0.3	23.7	0.1	22.9	0.3	14.3	0.1	24.0	0.1	23.2	0.1	13.9	0.1
Salinity	ppt	19.6	0.2	13.9	0.8	18.7	0.1	19.1	0.1	16.3	0.4	18.8	0.1	18.9	0.0	16.2	0.2	18.6	0.1
Secchi Depth	cm			87	8	115	14			93	6	125	8	0		111	10	116	5
Specific Conductivity	µS cm^{-1}	31.47	0.26	23.02	1.27	30.12	0.19	30.72	0.16	26.58	0.55	30.21	0.08	30.45	0.05	26.50	0.29	30.02	0.14
Zostera Aboveground Biomass	g m^{-2}	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.51	0.00	0.00	0.00	0.00	1.29	3.84	0.00	0.00	0.00	0.00
Zostera Belowground Biomass	g m^{-2}	0.00	0.00	0.00	0.00	0.00	0.00	0.72	2.27	0.00	0.00	0.00	0.00	7.18	11.90	0.00	0.00	0.00	0.00
Zostera Shoot Density	shoots m^{-2}	0	0	0	0	0	0	13	40	0	0	0	0	102	223	0	0	0	0
Zostera Cover	%	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	1.6	0.0	0.0	0.0	0.0
Zostera Blade Length	cm													157.40					
Zostera Blade Width	mm							2.13						1.87	0.66				
Ruppia Aboveground Biomass	g m^{-2}	19.59	14.65	6.26	9.45	8.79	11.08	5.80	5.88	1.97	5.46	1.19	2.21	14.42	14.97	2.31	5.31	13.17	40.45
Ruppia Belowground Biomass	g m^{-2}	21.06	15.05	7.09	11.22	10.26	11.47	11.03	12.17	3.23	9.55	1.30	2.11	26.39	18.94	4.39	9.98	3.11	8.61
Ruppia Shoot Density	shoots m^{-2}	6188	4574	5080	8021	6621	8251	3501	3576	726	1731	1108	1827	4062	3190	484	1082	1210	3304
Ruppia Cover	%	36.5	26.5	18.0	19.6	35.0	38.9	22.0	26.1	9.0	13.1	4.5	9.6	40.5	23.4	19.5	18.5	5.5	8.0
Ruppia Width	mm																		
Macroalgae Cover	%	8.5	14.5	0.0	0.0	0.0	0.0	8.0	11.4	0.0	0.0	1.5	3.4	23.5	32.4	0.0	0.0	0.0	0.0
Other Cover	%	0.5	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	1.6	0.0	0.0	1.0	2.1

Table 3 Repeated measures analysis of variance (ANOVA) for dissolved oxygen ('DO', concentration and percent saturation) and Secchi depth. Transect, time period, and their interaction are tested. Degrees of freedom (DF), sum of squares (SS), mean squares (MS), F statistic (F) and p value (P) are reported.

Variable	Source	DF	SS	MS	F	P
DO mg L-1	Transect	2	8.66	4.33	23.00	< 0.01
	Time Period	2	91.77	45.88	440.45	< 0.01
	Time Period * Transect	4	15.46	3.86	37.09	< 0.01
	Error(Transect)	27	5.09	0.19		
	Error(Time Period)	54	5.63	0.10		
DO %	Transect	2	1531.97	765.99	23.72	< 0.01
	Time Period	2	1581.65	790.83	42.77	< 0.01
	Time Period * Transect	4	3330.18	832.54	45.03	< 0.01
	Error(Transect)	27	872.07	32.30		
	Error(Time Period)	54	998.37	18.49		
Secchi depth cm	Transect	2	501.88	250.94	2.58	0.11
	Time Period	1	2759.01	2759.01	97.71	< 0.01
	Time Period * Transect	2	1831.88	915.94	32.44	< 0.01
	Error(Transect)	17	1655.00	97.35		
	Error(Time Period)	17	480.00	28.24		

Table 4. Friedman's tests for water quality (temperature, salinity, specific conductivity), eelgrass (aboveground and belowground biomass, shoot density, and percent cover), widgeon grass (aboveground and belowground biomass, shoot density, and percent cover), and percent cover of macroalgae and other organisms. Transect, time period, and their interaction are tested. Degrees of freedom (DF), sum of squares (SS), mean squares (MS), F statistic (F) and p value (P) are reported.

Variable	Source	DF	SS	MS	F	P
Water Quality						
Temperature	Transect	2	1248.12	624.06	10.67	< 0.01
	Time Period	2	48093.75	24046.88	410.96	< 0.01
	Time Period * Transect	4	6651.48	1662.87	28.42	< 0.01
	Error	81	4739.65	58.51		
Salinity	Transect	2	759.65	379.83	9.83	< 0.01
	Time Period	2	53494.82	26747.40	692.56	< 0.01
	Time Period * Transect	4	3345.73	836.43	21.66	< 0.01
	Error	81	3128.30	38.62		
Specific Conductivity	Transect	2	654.62	327.31	9.34	< 0.01
	Time Period	2	5400.00	2700.00	770.27	< 0.01
	Time Period * Transect	4	3242.63	810.66	23.13	< 0.01
	Error	81	2839.25	35.05		
Eelgrass						
Zostera Aboveground Biomass	Model	8	810.00	101.25	1.62	0.13
	Error	81	5064.50	62.52		
Zostera Belowground Biomass	Transect	2	594.07	297.03	3.61	0.03
	Time Period	2	1125.00	562.50	6.84	< 0.01
	Time Period * Transect	4	1188.13	297.03	3.61	< 0.01
	Error	81	6665.30	82.29		
Zostera Shoot Density	Model	8	819.20	102.40	1.64	0.13
	Error	81	5055.30	62.41		
Zostera Percent Cover	Model	8	180.00	22.50	1.00	0.44
	Error	81	1822.50	22.50		
Widgeon grass						
Ruppia Aboveground Biomass	Transect	2	3935.52	1967.76	4.26	0.02
	Time Period	2	10211.72	5105.86	11.06	< 0.01
	Time Period * Transect	4	546.72	136.68	0.30	0.88
	Error	81	37400.05	461.73		
Ruppia Belowground Biomass	Transect	2	2765.00	1382.50	3.19	0.05
	Time Period	2	12429.87	6214.93	14.33	< 0.01
	Time Period * Transect	4	1764.13	441.03	1.02	0.40
	Error	81	35135.50	433.77		
Ruppia Shoot Density	Transect	2	4778.45	2389.23	5.04	< 0.01
	Time Period	2	8430.62	4215.31	8.89	< 0.01
	Time Period * Transect	4	459.48	114.87	0.24	0.91
	Error	81	38421.45	474.34		
Ruppia Percent Cover	Transect	2	5010.22	2505.11	4.65	0.01
	Time Period	2	7125.35	3562.68	6.61	< 0.01
	Time Period * Transect	4	2281.93	570.48	1.06	0.38
	Error	81	43659.00	539.00		
Other cover						
Macroalgae Percent Cover	Transect	2	513.80	256.90	1.19	0.31
	Time Period	2	11950.32	5975.16	27.76	< 0.01
	Time Period * Transect	4	988.63	247.16	1.15	0.34
	Error	81	17436.25	215.26		
Other Percent Cover	Model	8	855.00	106.88	1.26	0.28
	Error	81	6885.00	85.00		

Table 5. Physicochemical measurements in the Barnegat Bay-Little Egg Harbor Estuary during submerged aquatic vegetation (SAV) sampling in 2011.

Segment	Sampling Period	N	Temp (ºC)	Salinity (ppt)	Specific Conductivity	Dissolved Oxygen (mg L^{-1})	Dissolved Oxygen (%)	pH	Depth (cm)
North	Jun-Jul	30	23.6 (0.5)	19.2 (0.3)	30.9 (0.5)	7.9 (0.6)	103.6 (8.8)	8.2 (0.2)	-
North	Aug-Sep	30	22.9 (0.3)	15.5 (1.2)	25.4 (1.9)	7.9 (0.8)	100.0 (11.2)	7.7 (0.2)	99.8 (12.8)
North	Oct-Nov	30	14.5 (0.7)	18.7 (0.1)	30.1 (0.2)	10.0 (0.4)	110.1 (5.5)	7.9 (0.1)	119.8 (8.1)
Central	Jun-Jul	60	24.2 (1.6)	24.7 (2.9)	38.6 (4.3)	8.4 (1.3)	115.5 (17.3)	8.1 (0.1)	84.0 (31.7)
Central	Aug-Sep	60	25.6 (1.7)	24.4 (4.5)	38.4 (6.5)	7.7 (1.7)	107.5 (22.0)	8.0 (0.2)	114.1 (17.8)
Central	Oct-Nov	60	16.4 (1.7)	26.9 (5.1)	41.8 (7.1)	9.0 (1.8)	108.6 (22.3)	7.9 (0.1)	132.3 (36.5)
South	Jun-Jul	60	22.7 (1.5)	29.3 (0.1)	45.2 (0.2)	8.1 (0.7)	111.3 (9.5)	8.0 (0.1)	87.8 (25.3)
South	Aug-Sep	60	27.0 (1.2)	30.0 (0.2)	46.3 (0.3)	6.4 (1.0)	95.1 (14.9)	7.9 (0.1)	102.4 (27.5)
South	Oct-Nov	60	16.7 (0.9)	27.3 (0.6)	42.4 (0.9)	9.3 (0.5)	112.5 (6.2)	8.0 (0.1)	108.1 (14.2)

Standard deviations in parentheses

Table 6. Characteristics of submerged aquatic vegetation (SAV) by sampling period in the Barnegat Bay-Little Egg Harbor Estuary during 2011.

SAV	Sampling[1] Period	Aboveground Biomass (g dry wt m^{-2})	Belowground Biomass (g dry wt m^{-2})	Shoot Density (Shoots m^{-2})	Areal Cover (%)	Blade Length (cm)
Zostera	Jun-Jul	7.2 (19.9)	21.4 (43.3)	157.0 (304.3)	19.7 (30.0)	25.3 (15.7)
	Aug-Sep	9.4 (37.6)	15.7 (37.8)	149.4 (443.2)	17.9 (32.9)	29.1 (12.3)
	Oct-Nov	17.4 (51.0)	15.5 (33.4)	179.1 (395.8)	16.1 (30.3)	31.5 (13.3)
Ruppia	Jun-Jul	4.4 (9.1)	5.5 (11.2)	1167.1 (2548.2)	8.3 (17.8)	
	Aug-Sep	2.0 (5.8)	3.0 (9.5)	1001.6 (3175.9)	9.3 (21.0)	
	Oct-Nov	3.7 (13.1)	2.6 (6.8)	1313.1 (3731.4)	6.5 (16.5)	
Macroalgae	Jun-Jul				7.9 (18.2)	
	Aug-Sep				1.1 (5.0)	
	Oct-Nov				1.0 (3.0)	
Other	Jun-Jul				0.2 (1.1)	
	Aug-Sep				0.1 (0.9)	
	Oct-Nov				0.5 (1.8)	

[1]Sample size is 150 for all parameters except blade length
Sample size for blade length (Jun-Jul) is 76
Sample size for blade length (Aug-Sep) is 57
Sample size for blade length (Oct-Nov) is 73

Standard deviations in parentheses

Table 7. Characteristics of submerged aquatic vegetation (SAV) by segment of the Barnegat Bay-Little Egg Harbor Estuary during 2011.

SAV	Segment	Sampling Period	Aboveground Biomass (g dry wt m^{-2})	Belowground Biomass (g dry wt m^{-2})	Shoot Density (Shoots m^{-2})	Areal Cover (%)	Blade Length (cm)
Zostera							
	North	Jun-Jul	0.5 (2.5)	2.6 (7.5)	38.2 (134.4)	0.2 (0.9)	15.7
	North	Aug-Sep	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	-
	North	Oct-Nov	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	-
	Central	Jun-Jul	12.4 (29.0)	33.5 (57.5)	250.4 (378.7)	28.3 (32.6)	29.9 (203.1)
	Central	Aug-Sep	8.5 (29.8)	11.6 (32.9)	161.3 (585.1)	17.2 (34.1)	31.3 (154.9)
	Central	Oct-Nov	26.6 (58.5)	18.0 (34.9)	239.8 (426.6)	24.8 (35.5)	31.9 (154.4)
	South	Jun-Jul	5.3 (10.3)	18.6 (32.8)	123.1 (253.5)	23.9 (31.1)	21.0 (73.1)
	South	Aug-Sep	14.9 (51.0)	27.7 (47.3)	212.2 (371.9)	27.6 (36.3)	27.8 (98.4)
	South	Oct-Nov	17.0 (53.8)	20.8 (37.9)	208.0 (439.2)	15.4 (29.3)	31.1 (106.7)
Ruppia							
	North	Jun-Jul	13.3 (13.4)	19.5 (16.4)	4583.7 (3873.9)	33.0 (25.8)	
	North	Aug-Sep	3.5 (7.0)	4.9 (10.0)	2096.6 (5086.7)	15.5 (17.3)	
	North	Oct-Nov	7.7 (23.9)	4.9 (9.0)	2979.4 (5693.3)	15.5 (26.9)	
	Central	Jun-Jul	4.4 (7.9)	3.9 (7.0)	626.0 (1185.0)	4.2 (8.9)	
	Central	Aug-Sep	3.2 (7.3)	5.2 (12.7)	1455.7 (3303.7)	15.4 (28.6)	
	Central	Oct-Nov	5.4 (11.3)	4.0 (8.1)	1793.1 (3978.9)	8.8 (15.6)	
	South	Jun-Jul	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	
	South	Aug-Sep	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	
	South	Oct-Nov	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	

Sample size is 30 for all time periods of sampling in the North segment
Sample size is 60 for all time periods of sampling in the Central and South segments
Standard deviations is parentheses

Table 8. Areal cover of macroalgae and other biotic elements in the Barnegat Bay-Little Egg Harbor Estuary during 2011.

Biota	Segment	Time Period	Sample N	Areal Cover (%)
Macroalgae				
	North	Jun-Jul	30	13.3 (22.0)
	North	Aug-Sep	30	0.0 (0.0)
	North	Oct-Nov	30	0.5 (2.0)
	Central	Jun-Jul	60	12.5 (22.4)
	Central	Aug-Sep	60	1.7 (6.8)
	Central	Oct-Nov	60	2.1 (4.3)
	South	Jun-Jul	60	0.7 (2.2)
	South	Aug-Sep	60	1.2 (3.9)
	South	Oct-Nov	60	0.1 (0.6)
Other				
	North	Jun-Jul	30	0.3 (1.3)
	North	Aug-Sep	30	0.0 (0.0)
	North	Oct-Nov	30	0.3 (1.3)
	Central	Jun-Jul	60	0.3 (1.6)
	Central	Aug-Sep	60	0.0 (0.0)
	Central	Oct-Nov	60	1.0 (2.6)
	South	Jun-Jul	60	0.0 (0.0)
	South	Aug-Sep	60	0.3 (1.4)
	South	Oct-Nov	60	0.0 (0.0)

Table 9. Mean (+/-) standard deviation percent cover of epiphytes on upper leaf and lower leaf surfaces of *Zostera marina*, and total epiphyte biomass (mg dry wt m^{-2}) on *Z. marina* leaves during 2011.

Sampling Period	Upper Leaf Percent Cover	Lower Leaf Percent Cover	Biomass
Months	*%*	*%*	*mg dry wt* m^{-2}
2011			
June-July	9.1 (12.8)	8.6 (12.9)	41.3 (270.6)
August-September	48.1 (27.7)	48.0 (27.8)	144.0 (164.0)
October-November	9.7 (14.4)	9.0 (14.4)	69.4 (182.5)

Table 10. Mean (± standard deviation) aboveground and belowground biomass, shoot density, blade length, and percent areal cover of *Zostera marina* recorded in the Barnegat Bay-Little Egg Harbor Estuary during 2004-2010.

Sampling Period	Aboveground Biomass[1]	Belowground Biomass[1]	Shoot Density[2]	Blade Length	Areal Cover
Months	*g dry wt m^{-2}*	*g dry wt m^{-2}*	*shoots m^{-2}*	*cm*	*%*
2004					
June-July	109.5 (67.6)	110.2 (118.8)	297.8 (414.7)	34.0 (10.9)	44.8 (27.6)
August-September	54.6 (48.8)	68.7 (58.8)	108.2 (282.1)	32.3 (7.2)	37.6 (31.3)
October-November	18.2 (19.8)	50.5 (66.0)	0.0 (0.0)	31.8 (8.4)	21.4 (23.3)
2005					
June-July	52.1 (71.4)	142.7 (197.1)	494.4 (614.5)	32.7 (17.6)	36.9 (33.1)
August-September	28.8 (48.0)	69.0 (101.8)	163.4 (220.0)	25.9 (14.9)	23.1 (35.1)
October-November	15.7 (26.6)	42.8 (64.0)	233.4 (284.4)	28.5 (14.7)	11.3 (12.9)
2006					
June-July	11.8 (26.4)	55.5 (70.7)	170.3 (263.3)	22.2 (24.6)	23.5 (35.8)
August-September	13.7 (21.7)	46.5 (112.6)	156.0 (311.2)	3.7 (9.8)	13.5 (20.6)
October-November	12.8 (25.4)	31.6 (64.7)	163.5 (299.4)	4.6 (9.8)	16.4 (24.0)
2008					
June-July	22.3 (63.6)	72.4 (158.6)	241.7 (435.3)	28.6 (12.2)	22.2 (29.9)
August-September	24.7 (39.4)	60.9 (89.3)	414.2 (570.4)	22.4 (13.6)	29.6 (36.3)
October-November	18.1 (40.6)	31.6 (51.8)	264.4 (464.6)	31.4 (17.7)	22.3 (31.1)
2009					
June-July	15.1 (31.2)	43.0 (60.3)	346.7 (536.3)	22.3 (13.2)	31.3 (35.5)
August-September	8.0 (17.1)	37.2 (51.7)	265.0 (406.9)	24.5 (11.6)	27.2 (34.8)
October-November	3.0 (7.2)	17.1 (34.5)	154.8 (325.0)	21.5 (10.8)	14.6 (19.0)
2010					
June-July	13.3 (24.3)	32.6 (47.0)	664.5 (459.6)	22.2 (12.5)	28.2 (35.7)
August-September	6.6 (15.3)	29.6 (52.8)	376.9 (379.8)	19.9 (10.6)	21.0 (34.5)
October-November	2.7 (8.0)	17.9 (37.5)	439.8 (708.3)	22.7 (13.4)	9.2 (21.0)

Table 11. Mean (± standard deviation) aboveground and belowground biomass, shoot density, and percent areal cover of *Ruppia maritima* recorded in the Barnegat Bay-Little Egg Harbor Estuary during 2004-2010.

Sampling Period	Aboveground Biomass	Belowground Biomass	Shoot Density	Areal Cover
Months	*g dry wt m^{-2}*	*g dry wt m^{-2}*	*shoots m^{-2}*	*%*
2004				
June-July				0.3 (1.6)
August-September				0.2 (1.3)
October-November				0.0 (0.0)
2005				
June-July	0.0 (0.1)	0.1 (0.1)	1521.2 (1310.5)	0.0 (0.0)
August-September	0.1 (0.2)	0.4 (1.0)	0.0 (0.0)	19.6 (32.7)
October-November	0.0 (0.0)	0.1 (0.2)	0.0 (0.0)	4.7 (11.7)
2006				
June-July			0.0 (0.0)	7.9 (21.7)
August-September			0.0 (0.0)	9.3 (24.7)
October-November			0.0 (0.0)	2.8 (9.5)
2008				
June-July				1.1 (4.5)
August-September				3.0 (13.4)
October-November				1.2 (4.3)
2009				
June-July			0.0 (0.0)	1.0 (3.4)
August-September			0.0 (0.0)	7.9 (22.9)
October-November			0.0 (0.0)	3.0 (8.9)
2010				
June-July	1.2 (2.0)	1.5 (1.9)	331.0 (231.5)	7.5 (21.1)
August-September	1.0 (1.8)	1.2 (1.6)	449.9 (249.4)	10.8 (29.4)
October-November	1.6 (2.8)	1.2 (2.2)	498.7 (366.0)	2.1 (7.1)