ASSESSMENT OF NUTRIENT LOADING AND EUTROPHICATION IN BARNEGAT BAY-LITTLE EGG HARBOR, NEW JERSEY IN SUPPORT OF NUTRIENT MANAGEMENT PLANNING

Prepared for:

New England Interstate Water Pollution Control Commission

By:

Michael J. Kennish and Benjamin M. Fertig
Institute of Marine and Coastal Sciences
Rutgers University
New Brunswick, NJ 08901

Richard G. Lathrop
Center for Remote Sensing and Spatial Analysis
Rutgers University
New Brunswick, NJ 08901

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KEY FINDINGS

- BB-LEH is highly eutrophic and susceptible to nutrient loading. It is shallow, poorly flushed (74 days in summer), and has a highly developed watershed (34% developed, 25% urban, 10% impervious surface). The estimated range of annual total nitrogen loads from the watershed is 448,000 – 851,000 kg N yr^{-1}.

- BB-LEH experienced low DO (82 times ≤4 mg L^{-1}, 1989-2010), high TSS (max >200 mg L^{-1}) and chlorophyll a (max >40 µg L^{-1}), harmful algal blooms (200,000 cells mL^{-1}), epiphytic loading (up to 38.3% cover of seagrass), macroalgae blooms (80-100% cover 36 times, 70-80% cover 19 times, and 60-70% cover 10 times), habitat loss, >67% fewer hard clams, and food web shifts, which degraded seagrass biomass (to 2.7±8.0 g m^{-2} aboveground; 17.9±37.5 g m^{-2} belowground) and led to mortality.

- The overall seagrass response is 'Highly Degraded' throughout the estuary.

- The Index of Eutrophication is the most comprehensive and holistic assessment of BB-LEH, integrating 74,400 observations among 85 variables for ~20 indicators in 6 components: (1) Ecosystem Pressures, (2) Water Quality, (3) Light Availability, (4) Seagrass Response, (5) Harmful Algal Blooms, and (6) Benthic Invertebrate Response. Outputs are quantitative annual assessments for 3 areas on a scale of 0-100: 0-20=Highly Degraded, 20-40=Poor, 40-60=Moderate, 60-80=Good, 80-100=Excellent. Index scores assess condition and its consistency. Data availability limits its power. Though monitoring intensified over time, spatio-temporal alignment of data collection and increased sampling frequency will improve future assessments.

- Eutrophication Index values declined 34% and 36% in the central and south segments from 73 and 71 in the 1990s to 48 and 45 in 2010, respectively, indicating these segments are undergoing eutrophication and are in a state of decline. The north segment, which has experienced the highest levels of nutrient loading, has already undergone severe degradation and eutrophication. Overall, eutrophication condition is worst in the north segment but has improved modestly, in contrast to stages and trends in the south and central segments. Scores in the north segment declined sharply in 2010 (to 37, Poor), but the highest score observed in the north segment (50, Moderate) was in 2009, 3.5 times its low score (14, in 1991).

- Nutrient loading severely impacted Eutrophication Index values in BB-LEH, particularly in 2003-2010, degrading condition from Good (73) to Poor/Highly Degraded (45, 37). Initial rapid declines highlight sensitivity to loading. Beyond ~2,000 kg TN km^{-2} yr^{-1} or ~100 kg TP km^{-2} yr^{-1}, condition plateaus as Poor/Highly Degraded (50 to 2) yet variability increases, suggesting a switch in dominant factors. Perhaps this is due to community shifts, e.g., from blooms of brown tide (> 1.8 x 10^6 cells mL^{-1} in 1999-2002) to macroalgae (1998, 2004, 2005, 2008-2010).

- Total nutrient loadings were Highly Degraded in the north segment (Pressure Index = 7), but

Moderate in the central and south segments (Pressure Index = 60, 55). During 1989-1997, low DO countered favorable temperatures leading to Moderate conditions (mean Water Quality Index = 57). Favorable temperatures continued in 1998-1999, but TP increased in 2000-2003. In 1998-2003, TSS was Moderate/Good (21 to 45), epiphytic loading was Poor/Moderate (16 to 40), % surface light reaching seagrass was Highly Degraded/Poor (7 to 32), declining in 1998-2002 in the north and south segments. In 2004-2010, TP condition in BB-LEH fell from Poor to Highly Degraded (32 to 7). TSS improved steadily in the north segment, variably in the south segment, and temporarily declined in 2004-2007 in the central segment. Similar temporary Poor/Highly Degraded condition in 2004-2009 in the central segment was seen in epiphytic load (44 to 1) and % surface light reaching seagrass (41 to 0). Seagrass cover and length condition worsened over 2004-2010: ModeratePoor (34 to 14) and PoorHighly Degraded (18 to 30), respectively.

- The condition of BB-LEH has progressively worsened over time for both nitrogen and phosphorus. Periods of improvement (1989-1992, 1996-2002, and 2006-2008) have not outpaced shorter but more detrimental periods of degradation, thus leading to the overall poorer condition regarding nutrient loading.

- It is clear that throughout the entire system, nutrient loading – both total nitrogen loading and total phosphorus loading – has resulted in substantial degradation and eutrophication of BB-LEH.

- Overall, water quality condition has been declining throughout the estuary since the early 1990s.

EXECUTIVE SUMMARY

Barnegat Bay-Little Egg Harbor (BB-LEH) Estuary is a shallow, poorly flushed coastal lagoon affected by multiple anthropogenic stressors and drivers of change from an expanding human population in the adjoining coastal watershed. These factors make it particularly susceptible to nutrient enrichment and other water quality problems. Land use-land cover in the BB-LEH Watershed has changed rapidly over the past three decades, and is currently more than 25% urban. Impervious cover in the BB-LEH Watershed is currently greater than 10%, and it will exceed 12% when all available land is developed. Such changes in land use have been shown to change hydrologic dynamics by increasing the percentage of impervious surface, resulting in decreases in recharge, increases in runoff, and more extreme hydrologic peaks and low-flow events in streams. Conversion of undeveloped land to urban land use is also associated with greater concentrations and loads of nutrients (nitrogen and phosphorus nutrient species) to area creeks, streams, rivers, and the main body of the estuary.

Nutrient loading from the watershed is an important driver of biotic change in the estuary. It can cause significant shifts in primary production and plant biomass, as well as changes in the composition of autotrophs, including microalgae, macroalgae, and macrophyte assemblages which modulate higher-trophic-level dynamics. Thus, the effects of altered bottom-up controls on the biotic structure and function of the system can be far reaching. Nutrient enrichment and resulting eutrophication pose serious threats to the estuary because they are leading to long-term, ecosystem-wide decline, affecting biotic resources, essential habitats (e.g., seagrass and shellfish beds), ecosystem services, and human uses. These and other effects of urbanization will continue to increase with increasing development and alteration of the watershed, unless highly aggressive management actions and effective planning are implemented.

Regulatory protection and conservation of New Jersey's estuarine waters are based on dissolved oxygen measurements. Yet dissolved oxygen is only one indicator of ecological health, and must be monitored continuously in multiple locations for accurate assessments due to natural fluctuations over the course of a day driven by natural processes such as changes in temperature or light, as well as community photosynthesis and respiration. Therefore, it is critical that assessments of ecological health also examine biotic indicators covering a broader range of physicochemical indicators in the watershed and estuary for effective ecosystem-based assessment and management. This project establishes appropriate biotic indicators and a framework for assessment using multiple biotic indices and thus will aid New Jersey in delineating environmental impacts using a broader, more relevant range of factors.

Previous assessments of BB-LEH designated the system as moderately eutrophic in the early 1990s, but later assessments reclassified it as highly eutrophic. Examples of assessments that have been applied to BB-LEH are NOAA's National Estuarine Eutrophication Assessment (NEEA) Model and Nixon's Trophic Classification. These classifications of system

eutrophication are based on degradation of eelgrass condition and other declining ecosystem measures that have continued in concert with nitrogen loading from the BB-LEH Watershed.

Nutrient loading has been repeatedly cited as a primary cause of ecosystem eutrophication in BB-LEH. The estimated range of annual total nitrogen loads from the watershed is 448,000 – 851,000 kg N yr^{-1}, and the protracted water residence time in the estuary (74 days during the summer) facilitates nitrogen uptake by plants and nitrogen accumulation in estuarine bottom sediments. Highest nitrogen loading occurs in the north segment of the estuary due to greater development and altered land surface in northern watershed areas and the larger influent delivery systems (i.e., Toms River and Metedeconk River).

Assessments reported here document multiple severe symptoms of eutrophication in the BB-LEH estuary. These include low dissolved oxygen concentrations, harmful algal blooms, heavy epiphytic loading, loss of essential habitat (eelgrass and shellfish beds), diminishing hard clam (*Mercenaria mercenaria*) abundance, and other ecosystem component shifts. Since 2004, the condition of eelgrass (*Zostera marina* L.) has declined significantly (in 2010 the lowest eelgrass biomass values were recorded for the estuary), and macroalgal blooms have occurred frequently with increased nitrogen loading from the BB-LEH Watershed. Light reductions have been linked to lower seagrass densities, slower growth rates, stunted morphology, and higher mortalities in the estuary. The loss of seagrass beds has a secondary impact on animal populations inhabiting them. The net result is diminishing ecological integrity of the system. BB-LEH is an estuary in insidious ecological decline, as evidenced in part by the declining eutrophication condition of the central and south segments since the 1990s and an even worse eutrophication condition documented for the north segment. An array of biotic indicators reflects an impacted system.

This investigation is part of a multi-year, interdisciplinary effort by Rutgers University and the USGS that characterizes and quantifies the estuary with regard to watershed nutrient inputs, physical and water-quality properties, and biological indicators and responses. Extensive databases collected over the 1989-2011 time frame have been examined in this study. Component 1 of the study involves watershed nutrient loading quantification from existing (secondary) data. In Component 2, estuarine biotic responses to stressors and the current degree of eutrophication are quantified from new and secondary data. In Component 3, biotic indices are developed, and values of the indices are computed. The current extent and validation of eutrophication are determined in Component 4. Synthesis and management recommendations are developed in Component 5.

In this investigation, all available hydrologic, water-quality, meteorological, and land-use data were compiled and used in conjunction with a watershed loading model to determine nutrient loading on several spatial scales. Total nitrogen, total phosphorus, nitrate plus nitrite, ammonia, and organic nitrogen were quantified. PLOAD, a modeling tool for calculating concentrations, loads, and yields (area-normalized loads) of stream contaminants from water-

quality, hydrologic, and meteorological data, was used to quantify nutrient loading in runoff. PLOAD runoff load and yield were calibrated to flow values from historic hydrologic records. Baseflow nutrient concentrations, loads, and yields were calculated for growing and non-growing seasons of 1989-2011.

Turf has been mapped in the watershed with an approximately 90% overall accuracy. The mapping was deemed of sufficiently high accuracy to be used as input to the USGS watershed-based nutrient runoff modeling. Turf coverage highly correlated with urban land cover and nutrient loading,

Biotic response to nutrient loading and determination of overall eutrophic condition of BB-LEH requires the use of bioindicators and bioassessment protocols in conjunction with physicochemical water quality parameters (e.g., dissolved oxygen, nutrient concentrations, and Secchi depth). This investigation of condition and status of BB-LEH, therefore, also employs multiple plant biotic indicators. Multiple quantitative measures of benthic plant parameters must be obtained for accuracy because benthic microalgae, macroalgae, and seagrasses play major roles in primary production in BB-LEH, as in other mid-Atlantic coastal lagoons. Eutrophication of this coastal lagoon is closely coupled to plant-mediated nutrient cycling, and thus accurate assessment of eutrophy must also focus on both key pelagic and benthic autotrophic indicators.

Prior to this report, no validated, quantitative biotic index existed to assess the ecosystem health of estuarine waters of New Jersey, most notably with respect to eutrophication. Through the development and application of a comprehensive biotic index for the coastal bays of New Jersey, this project provides a measure of eutrophic impact in BB-LEH and a method to quantify the status and trends of the system. This index identifies the condition of, and relationships between, ecosystem pressures, ecosystem state, and biotic responses. The establishment of an appropriate biotic index for BB-LEH will aid New Jersey in delineating environmental impacts. A long-term goal, though beyond the scope of this project, is to extend this type of ecosystem assessment of the BB-LEH system to all estuarine waters of New Jersey in order to protect biotic communities, recreational and commercial fisheries, water quality, and habitats. Therefore, this valuable research initiative has far reaching implications for coastal environmental protection and human use in New Jersey and other coastal states.

The biotic index of eutrophic condition developed for this investigation for the BB-LEH Estuary borrows from the basic methodology of previous assessments of BB-LEH, in particular the National Estuarine Eutrophication Assessment (NEEA). The NEEA uses the ASSETS Model (Assessment of Estuarine Trophic Status). The Index of Eutrophication developed and described here employs a numeric scoring system from 0 (degraded condition) to 100 (excellent condition) rather than a qualitative scoring system used in the NEEA Model. Our approach compares observations to thresholds to assess condition. It uses ~20 (rather than 5) indicators, and determines eutrophic condition along a quantitative (rather than qualitative) scoring system

through multivariate analysis. Calculations are made for the three segments of the estuary.

Candidate indicators were selected at the outset of this project. These indicators are organized together into six components: 1) Ecosystem Pressures, 2) Water Quality, 3) Light Availability, 4) Seagrass, 5) Harmful Algal Blooms, and 6) Benthic Invertebrates. Each component is comprised of several key indicators. Data collection of these indicators often occurred at different times or in different locations, therefore annual means (or medians) for the north, central, and south estuarine segments are utilized.

The index for each of the six components is calculated by summing a Raw Score and Weighted Score, each of which contributes 50% to the component index score. Each observation of each indicator is compared to 'thresholds' to determine the 'raw score'. An indicator's thresholds can be considered to be values for that indicator that mark some type of change in other (response) variables. Thresholds are determined and defined through examination of: (a) the literature, (b) analysis of available data for BB-LEH, (c) best professional judgment, and (d) some combination of a-c. Raw scores range from 0 (degraded condition) to 50 (excellent condition) and are evenly weighted between indicators within the component index. Thus, for example, the raw score for each of the four Water Quality indicators contributes 12.5% of the score for the Water Quality Index (25% * 50% = 12.5%). Weighted Scores weight the raw scores by their variability. Principal component analysis (PCA) is conducted on the raw scores to calculate a weighting for each indicator within each component based upon their eigenvectors (variability). The weighting is calculated as the square of the eigenvector for each variable. Weighted scores are then calculated by multiplying the raw score by the weighting. Thus, for example, the weighted score for any of the four Water Quality indicators contributes 0-50% of the score for the Water Quality Index (the weighting for each variable ranges 0-100%, * 50% = 0-50%). Raw and weighted scores are summed to calculate a component index score for each of the six components. Thus, for example, each of the indicators in the Water Quality component contributes 12.5-62.5% of the Water Quality Index. Indices for each of the six components are then averaged together to calculate the overall biotic index of eutrophication. Raw, weighted, and final scores for each component and the overall Index of Eutrophication condition are calculated for each segment of the estuary for each year (1989-2010), subject to data availability.

This report documents that total nitrogen concentrations vary with location, year and season, and are largely determined by land-use patterns and precipitation. As shown in previous studies of BB-LEH and other locations, nutrient loading to the estuary has increased as watershed land has been developed, and total nitrogen concentrations in the estuary are proportional to the total nitrogen loading from the watershed. Total nitrogen concentrations are not exceptionally high (generally less than 2 mg L^{-1} as N) compared to other watersheds with large amounts of agricultural land cover and/or point sources from domestic waste-treatment plants. Thus, all data and results of nutrient loading calculations clearly show that urban land development is responsible for nutrient levels that are elevated above background levels.

BB-LEH is particularly sensitive, even to small amounts of nutrient loading, because of its small estuarine surface area and volume relative to the expanse of the watershed and because of its extreme enclosure by a barrier island complex. Hence, the effects of development and resulting nutrient loading to BB-LEH are much more significant than they would be for a deeper and more open estuary. An important observation is that loads and yields of nutrients from the BB-LEH Watershed are to a large degree controlled by precipitation totals. Although nutrient concentrations are somewhat diluted by large amounts of water during major runoff events, the variability in runoff volumes is more dynamic, and the effect is higher loading rates during wetter seasons and years. This holds true for runoff and baseflow loading, because the streams in the BB-LEH Watershed are largely groundwater fed, and the discharge levels are strongly tied to precipitation totals for these highly responsive streams.

Eutrophication condition is closely tied to indicators of light availability, and these indicators are also closely coupled to seagrass success or failure. Macroalgal blooms developed relatively frequently and impacted seagrass beds by attenuating or blocking light transmission to the beds, leaving many unvegeted bay bottom areas. From 2004 to 2010, Pre-Bloom conditions (60-70% macroalgae cover) occurred 10 times (0.45 blooms m^{-2}), Early Bloom conditions (70-80%) occurred 19 times (0.67 blooms m^{-2}), and Full Bloom conditions (80-100%) occurred 36 times (1.57 blooms m^{-2}). Blooms were more frequent during June-July (27 occurrences, 1.10 blooms m^{-2}), and August-September (22 occurrences, 0.95 blooms m^{-2}), than October-November (16 occurrences, 0.63 blooms m^{-2}). The majority of the blooms occurred during the 2008-2010 period.

Eutrophication of BB-LEH is also indicated by extensive epiphytic biomass and coverage of seagrass leaves observed in 2009, 2010, and 2011 that correlate with large-scale concurrent reduction in eelgrass biomass. Epiphytes can attenuate up to 90% of the light incident on seagrass leaves. Epiphyte biomass in 2009 peaked during June-July (mean = 121.8 mg dry wt m^{-2}). In 2010, peak epiphyte biomass occurred during August-September (mean = 67.7 mg dry wt m^{-2}). In 2011, the highest epiphyte biomass was also recorded in August-September (mean = 144.0 mg dry wt m^{-2}). Maximum biomass of epiphytes also occurred at the time of peak epiphyte areal cover on eelgrass leaves. The mean percent cover of epiphytes during all sampling periods in 2009 ranged from 19.2 to 38.3% for upper leaf surfaces and 18.4 to 38.3% for lower leaf surfaces. This is significant areal coverage. In 2010, the mean percent cover of epiphytes was generally lower than in 2009, with the values ranging from 11.3 to 25.7% for upper leaf surfaces and 10.7 to 24.4% for lower leaf surfaces. However, higher values of epiphyte percent cover were found during the October-November sampling period in 2010 than in 2009, with the mean upper leaf and lower leaf percent cover values ranging from 20 to 21% in October-November 2010 compared to mean values ranging from 18.4 to 19.2% in October-November 2009. The highest epiphyte percent cover on seagrass leaves was recorded during the August-September sampling period in 2011 when the mean upper leaf and lower leaf percent cover values were 48.1% and 48%, respectively.

Brown tide, hazardous algal blooms (HABs) caused by the pelagophyte *Aureococcus*

anophagefferens were most pronounced in BB-LEH between 1995 and 2002, but they have not been monitored since 2004. However, one brown tide bloom occurred in 2010, and others may have occurred after 2004 as well. The highest *A. anophagefferens* abundances ($>10^6$ cells mL^{-1}), Category 3 blooms (200,000 cells mL^{-1}) and Category 2 blooms (35,000 to 200,000 cells mL^{-1}), occurred in 1997 and 1999 and then again during the 2000-2002 period. Brown tides also attenuate light, and thus impact seagrass beds. In addition, hard clams cease to grow above a brown tide threshold level of 400,000 cells mL^{-1}.

A hard clam stock assessment conducted in 2001 revealed more than a 67% reduction in hard clam abundance when compared with an earlier stock assessment conducted in 1986-87. The loss of such large numbers of hard clams appears to reflect a shift or transition in the system away from one of top-down control exerted by filter feeders consuming and regulating phytoplankton populations to one of bottom-up control limited by nutrient inputs. Aside from elevated densities of brown tide, high abundances of *Nannochloris atomus* and *Synechococcus* sp. have occurred in the estuary as well. Shifts in the food web structure of the estuary due to nutrient enrichment may have significantly impacted the hard clam population.

For other light influencing factors, the mean total suspended solids (TSS) values generally ranged from 5-40 TSS units. Maximum TSS values exceeded 200 TSS units. Secchi depths generally exceeded 2 m in all estuary segments. Minimum mean Secchi depths were ~1 m. Over the 1997-2010 period, the mean chlorophyll *a* measurements generally ranged from ~1-12 mg L^{-1}. Maximum chlorophyll *a* values exceeded 40 mg L^{-1}.

Seagrass conditions documented in this report clearly show substantial degradation over time that is not isolated to one bed, but rather is estuary-wide. Such widespread response signals a broad-scale stressor, in particular eutrophication resulting from nutrient loading to the estuary and associated light attenuation due to microalgal and macroalgal blooms that directly impacted seagrass beds. Eelgrass biomass declined consistently over the 2004-2006 and 2008-2010 periods and overall from 2004-2010. Furthermore, the rate of decline of eelgrass biomass during 2008-2010 was slower than that of 2004-2006. This change in the rate of decline is related to nutrient loading and associated symptoms of eutrophication, and occurred perhaps because there was less biomass left to be lost. Though long-term monitoring of seagrass was not started early enough to observe the beginning of the initial decline prior to 2004, the pattern of biomass decline with increasing nutrient concentrations is similar to load-decline relationships described in the literature. Eelgrass areal cover also generally decreased through 2010, but the decline in plant biomass, a key water quality indicator, was most marked. A general decline in plant parameters (except blade length) was evident from 2008 to 2010 corresponding with temporal separation (yearly and seasonally of environmental parameters suggests their importance to seagrass condition). 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. A return to previous levels of eelgrass biomass therefore may be difficult to attain.

The condition of *Ruppia maritima* in the estuary also does not appear to be strong, although only one year of data (2011) has been collected on widgeon grass in the north segment since 2004. There is no way to validate the condition of widgeon grass in the north segment without additional years of sampling there. Previous years of sampling in the central and south segments, however, show conclusively that widgeon grass 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 widgeon grass was found. Somewhat higher aboveground and belowground biomass values of widgeon grass were recorded in 2011, especially in the more favorable environment of the north segment. However, no widgeon grass samples were found in the south segment during 2011. These data demonstrate that widgeon grass dominates seagrass beds only in the north segment, while eelgrass dominates the beds in all other areas. In addition, the north segment does not appear to be a major habitat for either species.

The detrimental impact of nutrient loading on the ecosystem health of BB-LEH is clearly evident in the comparison of the values of the overall index of Eutrophication vs. total nitrogen loading and total phosphorus loading. As nutrient loading increases, eutrophication condition plummets from 'Good' (a score of almost 70) to 'Poor' (a score below 40), and in some cases even 'Highly Degraded'. The initial rapid response of the decline underscores how sensitive BB-LEH is to even small increases in nutrient loading, especially at lower levels of loading. The system responds differently after reaching a threshold of nutrient loading. In excess of nutrient loads amounting to ~2,000 kg TN km^{-2} yr^{-1} or ~100 kg TP km^{-2} yr^{-1}, the Eutrophication Index values no longer decline as rapidly and level off, though with a great amount of variability, ranging between 2 and 50 (Highly Degraded to Moderate condition). Therefore, in excess of ~ 2,000 kg TN km^{-2} yr^{-1} or ~100 kg TP km^{-2} yr^{-1} another factor or set of factors may explain the variability of the eutrophication condition. However, what remains clear is that throughout the entire system, nutrient loading – both total nitrogen loading and total phosphorus loading – clearly results in substantial degradation and eutrophication of BB-LEH.

The data also indicate that different portions of BB-LEH are in different stages of degradation and eutrophication. The north segment, which experienced the highest levels of nutrient loading, has already undergone severe degradation and eutrophication, as evidenced by the lowest values of the Eutrophication Index for this segment as compared to the central or south segments. The central and south segments are similar to each other, and over 1989-2010, both have undergone significant decline in condition associated with eutrophication as nutrient loading has increased to these portions of the estuary.

There are significant and overt biotic responses to nitrogen enrichment of the estuary. The characterization of biotic response indicators in the estuary to nutrient loading entails the use of existing datasets collected between 1989 and 2010. Data collected on the indicators in

2011 are employed as a validation dataset.

In some years, the estuary has shifted to different community states. For example, from 1999-2002, BB-LEH experienced severe brown tide (> 1.8 x 10^6 cells mL^{-1}) events, but in 1998, 2004, and 2005, extensive macroalgal blooms were recorded and have persisted through ensuing years (2008-2010). Both types of bloom events are detrimental to seagrass habitat.

BB-LEH Estuary is an impaired system both in respect to aquatic life support and human use. There were 82 occurrences of dissolved oxygen (DO) levels $\leq$ 4 mg L^{-1} (the surface water quality criterion for DO is 4 mg L^{-1}) in the estuary and tributary systems determined from grab samples taken at multiple sampling sites between 1989 and 2010. Most of these low DO values occurred in the south segment (N = 63), with far fewer in the central segment (N = 13) and north segment (N = 6). These values represent only one DO measurement taken quarterly and mainly during the morning daylight hours at a sampling station and, hence, they are likely to significantly underestimate the number of DO violations that occur over time. While the estuary is designated as impaired due to low DO in the north segment, the data presented here indicate that the estuary is also likely to be impaired in the south segment due to many DO levels below 4 mg L^{-1}.

The occurrence of sea nettle blooms in the north segment has resulted in extensive areas of estuarine waters that are non-swimmable and impaired. Lower salinity waters north of Toms River have the greatest numbers of sea nettles and the most impaired bathing beaches due to sea nettle occurrence. However, other waters in the north segment have also been impacted by eruptions of sea nettle populations since 2004. Increasing eutrophic conditions and hardened shorelines have likely contributed to the problem. Currently, approximately 40-45% of the estuarine shoreline is bulkheaded. Most of the north segment of the estuary is now bulkheaded, which provides ideal overwintering habitat for sea nettles.

Based on application of the assessment model, estuarine waters in BB-LEH are worse off in terms of nitrogen than phosphorus. In addition, based on nutrient concentrations, the north segment is in much worse condition than the central or south segments. The central segment is slightly better than the south segment, but not by much. Since 1992, the condition of BB-LEH has progressively worsened over time for both nitrogen and phosphorus. Periods of improvement (1989-1992, 1996-2002, and 2006-2008) have not outpaced shorter but more detrimental periods of degradation, thus leading to the overall poorer condition regarding nutrient loading.

The bioindicators examined and the biotic index developed and applied in this study will provide the core basis for nutrient management planning, and could be included in an environmental monitoring strategy designed to provide characterization of the state's coastal resources by creating an integrated method that will effectively assess estuarine ecological conditions. This outcome has direct relevance to supporting coastal resource management

programs in New Jersey and will also be applicable to resource management programs in other coastal states nationwide.

INTRODUCTION

Human population growth and development in coastal watersheds of the U.S. have led to increasing impacts on estuarine and coastal marine environments. While great strides have been made to control point sources of pollution (e.g., sewage treatment plants) in these watersheds, nonpoint sources of nutrient enrichment associated with watershed development have contributed to the progressive eutrophication of many coastal systems and the alteration of their biotic communities (Valiela and Bowen, 2002). Land-use change resulting from urbanization of upland and shoreline habitat is a source of stressors that affect shallow lagoonal estuaries. Nutrient and organic carbon loading has been an important driver of biotic and habitat change in these lagoonal systems.

Eutrophic conditions have developed in many estuarine systems bordered by watersheds with increasing agricultural and urban land use, and the effects are most acute in shallow coastal lagoons (Kennish and Paerl, 2010). Coastal lagoons are particularly vulnerable to rapid changes in population and land use of coastal watersheds (McGlathery et al., 2007). The conversion of natural land covers to farmlands, housing developments, and industrial complexes facilitates nutrient loading to nearby estuarine waters, leading to cascading water quality and biotic impacts, debilitating impacts, and diminished ecosystem services. Natural stressors, such as hurricanes and other major storms as well as floods and droughts, can exacerbate these effects (Paerl et al., 2005, 2009). An array of mid-Atlantic estuaries, most notably coastal lagoons with restricted circulation and high water residence times, has exhibited severely stressed responses due to nutrient over-enrichment. Most lagoonal estuaries in this region are now moderately to highly eutrophic and rank among the most impacted estuarine systems in the United States (Bricker et al., 1999, 2007). Watershed management strategies to reduce nutrient loading in estuaries of this region include upgrading stormwater controls, implementing low-impact development and best management practices, advancing open space preservation, and generating total maximum daily loads (TMDLs) for nutrient limitation.

Studies of coastal lagoonal systems indicate that environmental impacts escalate as development and the amount of impervious cover in surrounding coastal watersheds increase. A watershed impact threshold is exceeded when the amount of impervious surface cover is greater than 10% (Arnold and Gibbons, 1996). Development of the BB-LEH Watershed now amounts to ~34%, and the impervious land cover exceeds 10%. Ecological impacts therefore are to be expected with increasing land alteration in the watershed (Lathrop and Conway, 2001; Kennish, 2007). The BB-LEH Estuary is in a state of insidious ecological decline. This is manifested by declining ecological conditions such as significant loss of seagrass, occurrence of nuisance and toxic algal blooms (including brown tides), markedly diminished fisheries (e.g., hard clams,

Mercenaria mercenaria), eruptions of deleterious organisms (e.g, sea nettles, *Chrysaora quinquecirrha*), decreasing biodiversity along hardened shorelines (which now cover ~40% of the estuarine shoreline), and other degrading changes. These adverse effects have become increasingly evident during the past 15 years, and they are impacting human use of the system (e.g., swimming and shellfish harvesting). Extensive studies conducted on the estuary during the past two decades have documented these problems (Kennish, 2001a; Kennish and Townsend, 2007; Kennish et al., 2007a, b; 2008, 2010; Kennish, 2009).

To accurately assess ecological change in response to diverse stressors, estuarine condition must be determined based on a suite of water quality, biotic, and habitat indicators (Paerl et al., 2005, 2007). The use of existing sampling techniques to evaluate the ecological condition of shallow estuarine systems can provide extensive and useful databases, but they are often time consuming, labor-intensive, and costly. In addition, they frequently target a single stressor. To avoid these deficiencies, there has been an effort to develop analytical techniques and environmental indicators that span the multiple levels of biological organization and are broadly applicable across geographic regions (Niemi and McDonald, 2004). This study targets a series of key water quality, biotic, and habitat indicators in the BB-LEH Estuary for assessment of ecosystem condition.

STATEMENT OF THE PROBLEM

The BB-LEH Estuary is a shallow coastal lagoon along the central New Jersey coastline (Figure 1-1). It is subject to multiple anthropogenic stressors and drivers of change from a burgeoning population in the adjoining coastal watershed. The most problematic impacts relate to nutrient loading resulting in eutrophication that threatens biotic communities and essential habitats such as submerged aquatic vegetation, shellfish beds, and finfish nursery areas. Other adverse effects on this system include nonpoint source inputs of pathogens and other pollutants, as well as the physical alteration of habitat due to bulkheading, diking and ditching, dredging, and lagoon construction. Point-source impacts of the Oyster Creek Nuclear Generating Station (i.e., biocidal releases, thermal discharges, impingement, and entrainment) significantly increase mortality of estuarine and marine organisms that inhabit the estuary. Human activities in the BB-LEH Watershed, most notably deforestation and infrastructure development, partition and disrupt habitats and also degrade water quality and alter biotic communities. Ongoing land development increases turbidity and siltation levels in tributaries, which can create benthic shading problems in the estuary.

BB-LEH has been classified as a highly eutrophic coastal lagoon based on application of NOAA's National Estuarine Eutrophication Assessment (NEEA) Model (Bricker et al., 2007) and Nixon's Trophic Classification (Kennish et al., 2007a; Kennish et al., 2010). It is highly susceptible to nutrient loading because it is shallow, poorly flushed, and bordered by highly developed and altered watershed areas that act as a conduit for nutrient transport to the estuary.

Nutrient enrichment in this water body, as well as other coastal lagoons in the mid-Atlantic region, is linked to an array of adverse impacts including depleted dissolved oxygen, harmful algal blooms (HABs), epiphytic loading, reduced biodiversity, declining fisheries, loss of essential habitat (eelgrass and shellfish beds), imbalanced food webs, and diminished ecosystem services. These impacts threaten the structure, function, and ecological integrity of the system.

Eutrophication poses the most serious threat to the long-term health of the estuary (Kennish and Townsend, 2007). The net effect of eutrophication is potentially permanent alteration of biotic communities, extensive loss of living resources and habitats, and greater ecosystem-level impacts. Nitrogen loading from the BB-LEH Watershed is a major driver of ecological change and positively correlated with total nitrogen concentrations in the estuary. Elevated total nitrogen levels have been detected in the north and south segments of the estuary (Figure 1-2). BB-LEH is highly susceptible to nutrient enrichment because it is a shallow, enclosed basin with restricted circulation and a long water residence time that result in pollution retention and recycling in the system. In addition, it is surrounded by highly developed watershed areas.

Nutrient enrichment elicits negative biotic responses in BB-LEH. For example, nitrogen loading stimulates algal growth and epiphytic infestation that cause light attenuation and shading of seagrasses. Blooms of drifting, ephemeral macroalgae (e.g., *Ulva lactuca, Enteromorpha intestinalis,* and *Gracilaria tikvahiae*) produce a thick canopy of organic matter that poses a potential danger to the seagrass beds by smothering the plants and blocking light penetration (Kennish et al., 2008; Kennish et al., 2011; Kennish and Fertig, 2012). Additionally, the accumulation of these macroalgal mats on the estuarine floor can promote an increase in sediment sulfide concentrations due to plant decomposition in anoxic, organic-rich sediment layers that is detrimental to seagrasses and benthic infaunal communities (Burkholder et al., 2007). Seagrass photosynthesis, metabolism, and growth are negatively affected by sulfide build up in bottom sediments leading to a decrease in the depth penetration of seagrasses in eutrophic waters (National Research Council, 2000; Burkholder et al., 2007).

Brown tide (*Aureococcus anophagefferans*) blooms, which repeatedly occurred in high abundances in the estuary between 1995 and 2002 (Olsen and Mahoney, 2001; Gastrich et al., 2004), are also detrimental to seagrass beds because they attenuate light in the water column over extensive areas. The highest bloom densities were recorded in Little Egg Harbor. These blooms were associated with reduced light penetration in the water column. Since seagrasses are benthic vascular plants that require high light intensity for optimal growth, brown tide and other phytoplankton blooms can significantly reduce photosynthetic activity. In order to grow, seagrass requires ~90% of the total downwelling Photosynthetically Available Radiation (PAR) (Duarte, 1991). This typically restricts seagrass habitat to shallower, less turbid benthic environments.

The minimum light requirements of seagrasses generally vary between 5 and 20% of surface irradiance (Dennison et al., 1993). Hence, light attenuation in the water column due to

suspended particulates, dissolved substances, and epiphytes on photosynthetic surfaces of the plants, can be extremely harmful to seagrass beds. These factors can also contribute significantly to depth-limitation of seagrass beds (Duarte, 1991). Nutrient over-enrichment promotes nuisance and toxic algal blooms (phytoplankton and macroalgae), as well as epiphytic growth on eelgrass blades, which reduce light availability for eelgrass function (McGlathery et al., 2007; Paerl et al., 2003, 2009). Light reductions have been linked to lower eelgrass shoot densities, slower growth rates, stunted morphology, and higher mortalities (Ochieng et al., 2010).

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 phytoplankton 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 also 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 accelerate seagrass decline (Heck and Valentine, 2007). Implications of degraded eelgrass areal cover also include elimination of habitat for bay scallops (*Argopecten irradians*), hard clams (*Mercenaria mercenaria*), and other benthic species, and can be linked to changes in ecosystem structure and function driven by bottom-up effects.

Hard-clam (*Mercenaria mercenaria*) stocks in Little Egg Harbor decreased markedly between 1986 and 2001. Celestino (2003) estimated a total of 64,803,910 hard clams in Little Egg Harbor in 2001 compared to an estimated 201,476,066 in 1986/87, representing a decrease of >67% in absolute abundance. This decrease in hard clam abundance is consistent with the decline in hard clam harvest in the estuary which was greater than 98% between 1975 and 2005 (636,364 kg in 1975 to 6,820 kg in 2005) (Figure 1-3) (Data from the National Marine Fisheries Service). The loss of such large numbers of hard clams appears to reflect a shift or transition in the system away from one of top-down control exerted by filter feeders consuming and regulating phytoplankton populations to one of bottom-up control limited by nutrient inputs.

Eruptions of the sea nettle (*Chrysaora quinquecirrha*) have worsened in the estuary during the past decade. These eruptions have impaired human use of extensive areas of the estuary, most notably in lower salinity waters in the north segment preferred as habitat by the sea nettle. Many areas in the north segment of the estuary are now effectively non-swimmable due to sea nettle occurrence.

The loss of seagrass habitat has also plagued other coastal lagoons, and even deeper estuarine systems, in the mid-Atlantic region (Orth et al., 2006; Bricker et al., 2007; Kennish, 2009; Moore, 2009; Kennish et al., 2010). As noted by Burkholder et al. (2007), an array of factors can accelerate seagrass loss, such as depressed advective water exchange from thick macroalgal growth, internal nutrient loading via enhanced nutrient fluxes from sediments to the overlying water, biogeochemical alterations including sediment anoxia with increased hydrogen

sulfide concentrations, sediment re-suspension from seagrass loss, increased system respiration and resulting oxygen stress, loss of herbivores which control algal overgrowth, and shifts favoring exotic grazers that out-compete seagrass for space. Ammonium toxicity and water-column nitrate inhibition may also contribute. The decline of seagrass beds is a serious concern in any estuary because of the multiple ecosystem services that they provide, notably major sources 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; Orth et al., 2006; Moore, 2009). These vascular plants are important indicators of overall ecosystem health of an estuary because they integrate water quality and benthic attributes (Orth et al., 2006; Burkholder et al., 2007; Kennish et al., 2008, 2010; Moore, 2009).

Since 2004, eutrophy has generally worsened in much of the BB-LEH, and the condition of the seagrass habitat has significantly degraded. For example, seagrass biomass in the estuary decreased markedly over the 2004–2006 period, and by 2010 it had dropped to a mean of 7.5 g dry wt m^{-2} (aboveground) and 26.7 g dry wt m^{-2} (belowground), which were the lowest levels ever recorded. Reduced biomass levels have persisted through 2011. Seagrass areal cover has also generally declined within beds since 2004, eliminating habitat for hard clams, bay scallops (*Argopecten irradians*), and other benthic and demersal organisms. Seagrass now covers a 5260-ha area of the BB-LEH estuarine floor (Lathrop and Haag, 2011).

SCOPE OF ECOSYSTEM CHANGE

Designated as moderately eutrophic in the early 1990s, BB-LEH was later reclassified as highly eutrophic in the late 1990s, a designation reconfirmed in 2007 (Nixon, 1995; Bricker et al., 2007; Kennish et al., 2007a). Nutrient enrichment and associated organic carbon loading in this shallow coastal lagoon have been linked to an array of cascading environmental problems such as low dissolved oxygen, loss of essential habitat (e.g., seagrass and shellfish beds), nuisance/toxic algal blooms, and impacted harvestable fisheries (e.g., hard clams) (Kennish et al., 2008, 2010).

Nutrient enrichment of the estuary has been closely coupled to development of the BB-LEH Watershed, and the history stretches across decades of time. There has been an estimated two-fold increase in nitrogen accumulation rates in the upper estuary since the 1950s (Velinsky et al., 2010). The north segment of the estuary is the most heavily impacted by nutrient loading because the northern part of the watershed is the most heavily developed and altered by human activity. In addition, the largest tributary systems (Toms River and Metedeconk River) discharge to northern Barnegat Bay.

Brown tide blooms were most severe in 1999, 2000, and 2001 when cell counts of *Aureococcus anophegefferns* exceeded 1.8 x 10^6 cells mL^{-1} each year. Hard clams cease to grow

above a threshold level of 400,000 cells mL^{-1}. A hard clam stock assessment conducted in 2001 revealed >67% reduction in clam abundance when compared with an earlier stock assessment conducted in 1986-87 (Celestino, 2003). Aside from elevated densities of brown tide, high abundances of *Nannochloris atomus* and *Synechococcus* sp. have occurred in the estuary as well. Bricelj et al. (1984) showed that hard clams poorly digest picoplankton and other diminuitive phytoplankton species, which seriously impairs their growth. Shifts in the food web structure of the estuary due to nutrient enrichment may have significantly impacted the hard clam population.

Macroalgal blooms have occurred repeatedly over the past 15 years, and the frequency of their occurrence has increased in recent years (Bologna et al., 2000, 2001; Kennish et al., 2011). These events have correlated with reduced seagrass abundance (Kennish et al., 2011). The decrease in seagrass biomass since 2004 has eliminated a significant amount of essential benthic habitat for bay scallops, hard clams (*Mercenaria mercenaria*), as well as many other benthic and demersal organisms. Hence, the eutrophic impact appears to have worsened during the past seven years.

Accelerated growth of the drifting macroalga *Ulva lactuca* has periodically produced extensive organic mats on the floor of the estuary that have altered benthic habitat (Kennish et al. 2008). For example, these mats form a mosaic of thick algal canopies covering seagrass beds that produce patches of extensive bare-bottom areas on the estuarine floor. At times, the rapid growth of other macroalgal species in the estuary, such as the rhodophytes *Agardhiella subulata*, *Ceramium* spp., and *Gracilaria tikvahiae*, also contribute to this problem. In addition, the decomposition of thick macroalgal mats can promote sulfide accumulation and the development of hypoxic/anoxic conditions in bottom sediments potentially detrimental to benthic infaunal communities. Such was the case at the Seawood Harbor area in the north segment of BB-LEH in July 2011, when serious macroalgal accumulation and decomposition events occurred, impacting extensive bottom and water column habitat.

Recurring eruptions of the sea nettle (*Chrysaora quinquecirrha*) have likewise occurred in the estuary since 2002, limiting human use most notably in the north segment and possibly causing biotic structural changes over extensive areas due to excessive zooplankton cropping. Sea nettle eruptions may be coupled to increased system eutrophy as well. These biotic and physicochemical changes can lead to further deterioration of sediment and water quality, loss of biodiversity, and disruption of ecosystem structure and function.

Shallow eutrophic estuaries and coastal lagoons often exhibit a range of ecological and biogeochemical responses that signal a shift in the balance of selective forces shaping biotic communities and habitats. The net insidious effect of these responses is the potential for major shifts in food web structure and a marked decline in ecosystem services. Shifts in plant subsystems associated with eutrophy can have serious long-term adverse effects on higher trophic levels. Changes in phytoplankton communities from diatom/dinoflagellate dominants to greater abundances of raphidophytes, picoplankton, and bloom-forming pelagophytes (e.g.,

Aureococcus anophagefferens, the causative agent of brown tides) have often led to dramatic losses of shellfish resources in other shallow estuaries (Livingston, 2000, 2003, 2006).

Nitrogen over-enrichment, when unchecked, causes significant disruption of ecosystem health (Nixon, 1995; Nixon et al., 2001; Kennish et al., 2007a). There is growing concern that escalating eutrophication will lead to severe, long-term degradation of the BB-LEH Estuary that may be intractable (Duarte et al., 2009; Kennish and de Jonge, 2011). The net insidious effects of long-term and progressive eutrophication are substantially degraded biotic and habitat components of the estuary.

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 1-1). 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 ~3.5 x 10^8 m^3 and a wet surface area of ~280 km^2 (Kennish and Lutz, 1984; Kennish, 2001a-c). 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 greatly facilitates 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 in the watershed currently amounts to ~80 million gallons per day (Robert Nicholson, US Geological Survey, personal communication, 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 50 years to more than 575,000 year-round residents 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 ~33% between 1995 and 2010 (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., 2007a).

The watershed (1,730 km^2) of the BB-LEH Estuary lies entirely in one state (New Jersey) and mainly receives nonpoint source nutrients (e.g. residential fertilizers) via both overland and groundwater (Kennish, 2001a; Kennish and Townsend, 2007). 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; Setizinger et al., 2001).

Habitats

The BB-LEH system is characterized by a wide range of habitats, including vegetated and unvegetated subtidal bay bottoms, intertidal flats and bay islands, dunes and beaches, tidal and freshwater marshes, as well as upland and wetland forests. Bottom sediments in the estuary, consisting of a mosaic of sand, silt, clay, shells, and organic matter, support an array of benthic floral and faunal communities. Urban development has resulted in the significant loss and alteration of upland and wetland forests and tidal wetlands (Lathrop and Bognar 2001; Lathrop et al. 2000). For example, 5,700 ha of forested habitat were lost to development in the BB-LEH Watershed between 1996 and 2005. About 20% (440 ha) of farmland area was also lost to development in the watershed during this time period (Richard G. Lathrop, Rutgers University, personal communication).

Water Quality

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}). The estimated range of annual total nitrogen loads from the watershed is 448,000 – 851,000 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.

Nonpoint source inputs account for almost all of the nitrogen entering the estuary. A regional wastewater treatment plant system, which has operated in the BB-LEH Watershed for more than 30 years, discharges effluent directly to the Atlantic Ocean. Only a minimal, diluted fraction of this effluent may re-enter the estuary via inlet exchange and would not be considered a point source of nitrogen. Confined animal feeding operations (52 total) cover a very small area of the watershed. With only one exception of a centrally located feeding operation in the watershed, all are located in the northern portion of the watershed. To effectively address nutrient loading problems in the estuary, it is important to determine the threshold loading of nutrients that produce observable biotic responses and impacts in the system (Kennish et al., 2008). In addition, it is critical to continuously monitor nitrogen loading to the estuary to effectively assess ecosystem health.

The highest concentrations of nitrate in surface waters in New Jersey are typically during low flows than during high flows. Low flows occur when it has not rained during the previous week, and most of the streamflow results from groundwater discharge to streams. Seitzinger et al. (2001) determined that nitrogen levels are highest in the northern part of the estuary due to the effects of heavy coastal watershed development. Elevated total nitrogen concentrations in the north segment have been corroborated by NJDEP nutrient sampling surveys conducted since 1989.

The Oyster Creek Nuclear Generating Station (OCNGS), a 635 MW power plant that has operated commercially in the BB-LEH Watershed since 1969, represents the only significant point source impact on the central bay, but biotic impact studies of the power plant have been conducted only sporadically over the past 35 years. Biocidal releases (chlorine) to Oyster Creek can affect water quality. However, the greatest impacts of the OCNGS are due to thermal discharges, impingement, and entrainment which significantly increase mortality of estuarine and marine organisms that inhabit the estuary (Kennish, 2001a).

Other adverse effects on estuarine water quality include nonpoint source inputs of pathogens and other pollutants as well as bulkheading, dredging, and lagoon construction. Human activities in the BB-LEH Watershed may not only disrupt habitats but also degrade water quality and alter biotic communities by raising turbidity and siltation levels in the estuary.

Estuarine Segmentation

Gradients in salinity, water depth, nutrient loading, total nitrogen concentrations, bottom sediments, hydrology, and basin morphology require partitioning of the estuary into segments for accurate index analysis. The estuary, therefore, has been divided into three segments (north, central, and south segments) for data assessment in this project (Figure 1- 4).

North Segment

The north segment extends from just south of the Toms River to the northern extremity at Bay Head (Figure 1-4). It is characterized by significantly lower salinities and higher total nitrogen concentrations than waters south of this segment. The type of nitrogen also differs from primarily dissolved inorganic nitrogen in the north segment to primarily dissolved organic nitrogen in the south segment. The north segment is narrower than the central segment. In addition, water depths are shallower than in the central segment (Figures 1-5). The bottom sediments in the north segment are finer grained than in the central segment largely due to diminishing tidal currents from Barnegat Inlet which transport and deposit marine sands across central Barnegat Bay (Figure 1-6). According to Psuty and Silveira (2009), sediments in the north segment exhibit a repetitive suite of morpho-sedimentary units that is related to tidal flows in the minor drainage channels emanating from the mainland. Shallow bars have formed across the mouths of micro-estuaries along the mainland such as in the Kettle Creek-Silver Bay area. A clear association of sediment type and morphology of bed structure is evident.

Central Segment

The central bay extends from an area south of Toms River to near Mill Creek (Figure 1-4). This segment is characterized by more rapid (hydrological) flushing and reduced water residence time than in the north and south segments, strong tidal currents entering at Barnegat Inlet, an extensive flood-tidal delta and its variety of forms and sediment types, deep tidal channels lined with coarse shell debris and some gravel, extensive well-sorted fine to medium sands extending north and west, finer sediments on the mainland side with a mosaic of sediment types, and seagrass beds dominating on the east side (Kennish, 2000; Psuty, 2004; Psuty and Silveira, 2009). Water circulation is greater in the central segment than the north and south segments due to the proximity of Barnegat Inlet, a wider bay area, greater fetch, and deeper waters.

South Segment

The south segment extends from the area near Mill Creek to Little Egg Inlet (Figure 1-4). Southern Barnegat Bay and Manahawkin Bay are narrow and heavily constrained by the surrounding land masses. The estuary widens again in lower Little Egg Harbor. The flow regime is thus much different here than in the central segment due to the increasing hydrologic influence of Little Egg Inlet to the south. In the Manahawkin Bay area, the water flow is restricted, and the water residence time substantially greater than that in the central segment. Kennish (2001c)

described the water circulation patterns in Little Egg Harbor. Tidal currents have greater influence than the discharge of small coastal creeks draining the mainland areas in the southern part of the estuary. Sediments in this segment consist of fine sand, silt, clay, and shell fragments (Kennish, 2001c). The greater constriction of the surrounding land and more restricted flow in the Manahawkin Bay area result in more extensive areas of finer grained sediments (silt and clay) than in the central and north segments. These finer sediments are clearly evident along the western side of Manahawkin Bay and Little Egg Harbor (Figure 1-6). Therefore, the bottom sediment patterns are substantially different in this segment than in the other two segments to the north.

East-West Segments

Each of the three segments must also be subdivided in order to separate eelgrass habitat on the east side of the estuary from the mosaic of complex morpho-sedimentary units on the west side of the estuary. Sediments differ in the three segments as shown by an estuary-wide sediment distribution map (Figure 1-6). There is a mosaic of sediment types in each segment, most notably in the western bay areas, with finer sediments clearly evident in the north and south segments. Drivers of benthic change are greater in the central bay due to strong tidal currents that account for the broad expanse of well-sorted sandy sediments to the west.

OBJECTIVES

This study had several clearly defined objectives:

- To document the influence of human altered land use on past and present nutrient export from the BB-LEH Watershed to the BB-LEH Estuary using physical and chemical watershed data and land-use patterns, and spatially explicit models.

- To determine if nutrient loading quantified by subwatershed and biotic response are stable or are temporally and spatially variable.

- To quantify baseflow, runoff, and total nutrient loads and to determine the relative importance of turf area coverage.

- To determine estuarine biotic responses to the loading of nutrients across a gradient of upland watershed development and associated estuarine nitrogen loading, and to identify key biotic responses across a variety of estuarine organisms by examining shifts in phytoplankton, benthic macroalgae, seagrass, epiphytes, benthic invertebrates, and shellfish structure and function.

- To generate a biotic index of eutrophic condition as a tool to evaluate future conditions using water quality and biotic indicators to assess eutrophication, eutrophic impacts, and overall ecosystem health of the BB-LEH Estuary and to formulate threshold levels of biotic decline and numeric loading criteria that can support an effective nutrient management plan.

- To apply a conceptual model of eutrophication and determine if ecosystem structure and function have been altered in the BB-LEH Estuary.

- To document the current biotic and seagrass habitat conditions of the BB-LEH Estuary at the end of the investigation using the most recent biotic data collected (2011) and biotic index methods developed from data collected through 2010.

APPROACH

This study has used novel methods of modeling nutrient flow to characterize the effects of rapid urbanization and altered land use in the BB-LEH Watershed. With coastal population growth increasing rapidly in the watershed, it is becoming more important to understand the effects of land-use alteration on the BB-LEH Estuary. This interdisciplinary project has integrated models of the coupled watershed-estuary system to estimate levels of nutrient loading and has employed a suite of key water quality, biotic, and habitat indicators for quantifying and characterizing estuarine responses and eutrophic conditions associated with these environmental stressors at local and estuary-wide scales.

A major fraction of primary production in BB-LEH, as in many coastal lagoons, derives from the benthic regime (i.e., benthic microalgae, macroalgae, and seagrasses). Therefore, quantitative measures of chlorophyll *a*, which are used as a proxy for phytoplankton biomass, must be supplemented with quantitative measures of benthic plant parameters to obtain an accurate assessment of ecosystem eutrophic condition. Determination of overall eutrophic condition of a coastal lagoon, such as BB-LEH, requires the use of bioindicators and bioassessment protocols in conjunction with physicochemical water quality parameters (e.g. dissolved oxygen, nutrient concentrations, total suspended solids). Eutrophication of this coastal lagoon is closely coupled to plant-mediated nutrient cycling, and therefore accurate assessment of eutrophy must focus on both key pelagic and benthic autotrophic indicators.

Quantitative loading criteria for nitrogen and phosphorus compounds, above which impairment of ecosystem structure and function occurs, have not been established for U.S. estuaries (Hameedi et al., 2007). These coastal ecosystems are highly variable in respect to the causes of, and responses to, nutrient enrichment, and therefore site-specific measures of assessment must be applied. This ecosystem-based study targeting the BB-LEH has important

implications for other coastal lagoons in the U.S. Prior to this study, the link between nutrient loading stress and biotic responses in BB-LEH was not well constrained for a number of key parameters. Such is the case for many other estuaries, most notably coastal lagoons (Kennish, 2002).

In this ecosystem-based project, we have applied multiple analyses to quantify spatial and temporal relationships between nutrient loading and biotic responses in the BB-LEH Estuary. In particular, this report describes the concurrent examination of multiple biotic responses, exploration of stressor-response relationships, and development of a comprehensive biotic index of eutrophication. Several key biotic response variables were targeted in the estuary (i.e., seagrass, phytoplankton, HABs, macroalgae, epiphytes, benthic invertebrates, and shellfish (hard clams), and were examined in the context of nutrient loading associated with human-altered land use in the adjoining BB-LEH Watershed. Important steps in the process included the determination of accurate nutrient loading values for the watershed, threshold levels of biotic decline, and numeric measures of bioindicators of ecosystem condition. To sustain and restore the health of BB-LEH, we need a better understanding of the relative importance of the predominant sources of nutrient enrichment and their relation to regional land-use patterns. This investigation has employed spatially explicit modeling of watershed nutrient sources to document the contribution of the waterborne sources of nitrogen to the estuary from subwatersheds. By coupling the nutrient loading models with *in situ* sampling of biotic responses in the estuary, we have attempted to characterize the spatial and temporal dynamics of the nutrients within the estuarine system that could be used to establish the basis for developing accurate nutrient loading criteria. Based on these findings, we have modeled how estuarine health will likely change as a result of several important policies for land use and nutrient pollution control.

METHODOLOGY ELEMENTS

This project was conducted in five components. In Component 1, loading of nutrients to BB-LEH was quantified by using all relevant data sources to meet the water quality objectives of the project. In Component 2, the biotic responses in the estuary to temporally and spatially variable nutrient loads were analyzed and reported. In Component 3, an index of eutrophic condition for the BB-LEH Estuary was computed from data collected on key water quality and biotic indicators during the 1989 to 2010 period. In Component 4, additional biotic and water quality sampling and data analysis were conducted in 2011 to further assess the current status of eutrophication of the estuary. This component also provided information to validate biotic responses in previous years. In Component 5, synthesis and management recommendations of the project were advanced. The use of study findings in nutrient management planning was also considered.

Component 1: Watershed Nutrient Loading Loading

The methodology of Component 1 is briefly described here, and in detail in Appendix 1. Available surface-water quality data for all streams in the BB-LEH watershed for 1970-2011 were compiled from the USGS's National Water Information System (NWIS) database, and from the USEPA's Storage and Retrieval (STORET) database. After thoroughly reviewing aspects of the data such as units, detection limits, and site locations, a database of quality-assured water-quality data was developed. The goal was to retain as much data as possible while maintaining a high quality standard. Hydrologic data were retrieved from the USGS's NWIS database; these data are made up of daily mean flow rates of streams from continuously-monitored gaging stations located in the watershed, and have been extensively reviewed in a multi-tiered quality assurance and evaluation program. Meteorological data in the form of daily, monthly, and annual precipitation records were retrieved from the National Climatic Data Center and from the Office of the New Jersey State Climatologist. Land-use land-cover data were retrieved from published sources and include data sets for years 1973, 1986, 1995, 2002, and 2007.

Precipitation and hydrologic data were used to conduct baseflow separation analysis for the major streams in the watershed, and to identify which water-quality data were collected during baseflow conditions and which were collected during runoff conditions. Relations between land use and water-quality were developed. Available values of streamflow and nitrogen and phosphorus concentrations were used to calculate flow-weighted mean concentrations during runoff events, referred to as event-mean concentrations (EMCs). A runoff model (PLOAD, Version 3.0 (U.S. Environmental Protection Agency, 2001)) used the EMCs, along with land-use percentages, percent impervious cover, and precipitation data to calculate concentrations, loads, and yields at the hydrologic unit code 14-digit (HUC-14) scale. Baseflow concentrations, loads, and yields were determined in an analogous way, in that baseflow-mean concentrations (BMCs) were determined for each land-use category from existing water-quality data, and were applied to the land-use fractions for each HUC-14 subbasin.

Baseflow and runoff concentrations, loads, and yields of total nitrogen and total phosphorus were estimated for each HUC-14 subbasin. Annual, growing season, and non-growing season estimates were determined for the period 1989-2011. Loads were aggregated by watershed segment (north, central, and south), to correspond with estuarine segments used in the biotic assessment.

Component 2: Estuarine Biotic Responses

The major objective of this component of the study was to characterize biotic responses in the estuary to nutrient loading and enrichment using existing datasets collected between 1989 and 2010. Data collected in 2011 was also used as a validation dataset (see Component 4). A significant outcome of this research is the determination of key biotic responses and associated

thresholds of nitrogen enrichment that lead to shifts in ecosystem structure and function signaling eutrophic degradation. In addition, a biotic index of condition is calculated to quantify the current and historical state of estuarine eutrophic effects (see Component 3). Several key bioindicators have been used in development of the biotic index.

Seagrasses

The estuary was divided into three segments (north, central, and south) to survey seagrass beds and other biotic elements. The estuarine segmentation is based on a north-to-south gradient in salinity, nutrient loading, watershed development, water depth, and other factors; there were also differences in sediment composition, hydrography, and basin morphology in the segments (Kennish, 2011). We collected seven years of comprehensive biotic response data in seagrass beds (2004-2006 and 2008-2011). During 2004-2006 and 2008-2010, biotic samples were collected at up to 120 sampling stations along 12 transects; in 2011, biotic samples were collected at 150 sampling stations along 15 transects, which included 30 sampling stations and 3 transects in the north segment (Figures 1-7, 1-8, and 1-9).

Biotic sampling was conducted at 60 stations in Little Egg Harbor during 2004 and at 60 stations in Barnegat Bay during 2005. Taxonomic surveys were conducted during 2004 and 2005 to determine the composition of macroalgae in the four seagrass beds. Biotic sampling was expanded to 80 stations in 2006, 120 stations in 2008, 2009, and 2010, and all 150 stations in 2011 (Figure 1-7). No sampling was conducted in the estuary in 2007.   An array of water quality parameters was also measured at each station during biotic sampling.

Seagrasses are responsive to regional gradients in nutrient-driven change in water transparency mediated by phytoplankton and benthic macroalgal blooms, as well as epiphytic growth on seagrass leaf surfaces that can lead to a significant decline in seagrass abundance, biomass, and other parameters. Seagrass (biomass, shoot density, blade length, and areal cover), macroalgae (areal cover), epiphytes, and shellfish (hard clams and scallops) data were collected at regular (bimonthly) intervals from June to November (see below). NJDEP water-quality data collected year-round between 1989 and 2011 were used in the data analysis of physicochemical parameters for the estuary. These data included dissolved oxygen, Secchi depth, and chlorophyll *a*, as well as total nitrogen (TN), total phosphorus (TP), total suspended solids, and temperature.

A three-pronged seagrass study was conducted over the 2004-2011 period entailing in situ quadrat and core sampling, as well as high-resolution underwater digital camera imaging and comprehensive water quality sampling as outlined by Kennish et al. (2006, 2007b, 2008). In situ sampling of seagrass beds followed the quadrat, core, and hand sampling methods of Short et al. (2002). The main objective of the seagrass study was to determine the demographic characteristics and spatial habitat change of *Zostera marina* and *Ruppia maritima* over an annual growing period, as well as the potential impacts of benthic macroalgae on the seagrass beds. Sampling stations were located with a Differential Global Positioning System

(Trimble®GeoXT™ handheld unit).

Epiphytes

Growth of epiphytes on seagrass surfaces increases with nutrient enrichment leading to a decrease in light transmission, reduced photosynthesis, and loss of seagrass biomass. Epiphyte biomass and areal cover measurements were conducted on seagrass samples collected over a three-year study period (2009-2011).

Phytoplankton

Phytoplankton communities are sensitive indicators of nutrient enrichment, which often leads to increased frequency of HABs (e.g., brown tides), cyanobacteria blooms, and nuisance blooms. Shifts in species composition to smaller phytoplankton groups, including microflagellates, picoplankton, and other smaller forms can cause serious shading and trophic impacts on benthic habitats and organisms. Measures of chlorophyll *a* are important in monitoring phytoplankton responses to nutrient enrichment, but not HABs such as brown tides.

Chlorophyll *a* measurements have been analyzed retrospectively from archived water-quality databases of the NJDEP collected in the estuary from 1989 to 2011 to assess phytoplankton biomass. From 2009 to 2011 we employed NJDEP remotely estimated chlorophyll *a* concentrations in the estuary. When high chlorophyll *a* values were detected by the NJDEP using remote sensing surveys, water samples were collected in situ within and outside of the phytoplankton bloom areas and subsequently analyzed in the laboratory for species composition and abundance. The sample analyses were completed at the Leeds Point Laboratory of the NJDEP.

Brown tide bloom events were monitored for BB-LEH by the NJDEP database over the 1995 to 2004 period. In addition, one HAB event was recorded in Little Egg Harbor in August 2010. These data were useful for retrospective analysis of brown tide activity in the estuary and incorporation into the biotic index.

Macroalgae

Drifting macroalgae are highly responsive to nutrient enrichment and thus are important indicators of eutrophic condition. The rapid increases in abundance of bloom-forming, sheet-like macroalgal forms often blanket extensive areas of seagrass habitat, blocking incident light and contributing to the loss of seagrass beds and the resident benthic and nektonic fauna.

The percent areal cover of macroalgae was also recorded, yielding data on macroalgal bloom occurrences. Diver observations were made to determine the occurrence and areal cover of macroalgae. In addition, high resolution underwater digital imaging was used to validate diver observations.

Hard clams (*Mercenaria mercenaria*)

Hard clams are typically more sensitive to local-scale conditions, and their response to persistent eutrophication and shifting phytoplankton composition is typically a decline in abundance and the loss of the resource. Hard clam abundance has been included in the biotic index development.

Hard clam (*Mercenaria mercenaria*) abundance data were obtained from field surveys conducted by the NJDEP in the estuary during 2001. More specifically, the New Jersey Bureau of Shellfisheries conducted an extensive hard clam stock assessment of Little Egg Harbor. The Bureau sampled 194 stations from 16 July to 31 August 2001 using a hydraulic dredge to determine the standing stock and relative distribution of hard clams in Little Egg Harbor.

Benthic invertebrates

Benthic invertebrate communities inhabiting eutrophic waters often experience a change in composition. Higher biomasses of benthic autotrophs generally favor greater numbers of deposit-feeding species and a progressive shift from larger, long-lived benthic fauna to smaller, rapidly growing but shorter-lived forms. These changes lead to an unbalanced benthic community.

The development of a biotic index includes a benthic invertebrate component, which is needed to measure the overall ecological condition of the estuary. Currently, no validated metric or benthic index is available to assess overall ecosystem condition for BB-LEH. Benthic invertebrates collected at ~80 sampling stations in the estuary in 2001 were used in the development of the eutrophic index for the estuary.

Component 3: Biotic Index Development

An index of eutrophication is developed for BB-LEH to quantify the status and trends of condition. The index includes a suite of ~20 metrics that are organized into six components: (1) Ecosystem Pressures; (2) Water Quality; (3) Light Availability; (4) Seagrass Response; (5) HABs; and (6) Benthic Invertebrate Response. For ecosystem pressures, the metrics include total nitrogen loading, and total phosphorus loading. For water quality, the metrics include temperature, dissolved oxygen, total nitrogen concentration, and total phosphorus concentration. For light availability, the metrics include total suspended solids, chlorophyll *a*, macroalgae areal cover, the ratio of epiphytes to seagrass, the percent of light reaching seagrass leaves, and Secchi depth. For seagrass response, the metrics include seagrass biomass (aboveground and belowground), shoot density, blade length, and areal cover. For HABs, the metrics include occurrence of brown tide blooms. For benthic invertebrate response, the metrics include benthic invertebrate species richness, Gleason's D value, EMAP index values, and hard clam abundance. A numeric impact value and a variability-weighted value are calculated for each parameter in all

three segments, and are summed to obtain an overall index of eutrophic condition for each estuary segment.

An important goal of this project is to develop an effective and useful index of eutrophic condition for the BB-LEH Estuary. While the current determination of the impairment of New Jersey's estuarine waters is based on dissolved oxygen measurements, it is also important to examine biotic indicators and a broader range of physicochemical indicators for effective ecosystem-based assessment and management. The establishment of an appropriate biotic index for BB-LEH will aid the state of New Jersey in delineating where environmental impacts exist and in targeting resources to address these impacts. Such an index would combine ecosystem pressures (nutrient loading and water residence time), ecosystem state, and biotic responses. No validated biotic index currently exists to assess the estuarine waters of New Jersey, most notably with respect to eutrophication. A long-term goal is to extend this type of ecosystem assessment of the BB-LEH system to all estuarine waters of New Jersey in order to protect biotic communities, recreational and commercial fisheries, water quality, and habitats. Therefore, this is a valuable research initiative that has far reaching implications for coastal resource management, environmental protection, and human use in New Jersey and other coastal states.

We have applied the basic methodology used in the National Estuarine Eutrophication Assessment (NEEA) model to develop a biotic index of eutrophic condition for the BB-LEH Estuary (Bricker et al., 1999, 2007). However, we have significantly modified the approach, dividing the estuary into three segments based on environmental gradients. We have used more indicators than did Bricker et al. (1999, 2007). A numeric scoring system was used that computes an index value from key water quality and biotic indicator measurements in each of the three estuary segments for years sampled during the 1989 to 2011 period.

Component 4: Validation Dataset (2011) for Eutrophication Assessment

The collection of biotic data was continued through 2011. This additional year of data acquisition was conducted for two reasons. First, the method of determining the index of eutrophic condition developed with data collected through 2010 has been applied using 2011 data for validation. To this end, the same sampling protocols used in field surveys conducted from 2004 through 2010 were followed in 2011. Second, having data collected in 2011 enabled assessment of current conditions in the estuary. This 2011 dataset is valuable for continued tracking of spatial and temporal patterns of eutrophication and for determining if eutrophic conditions are improving, declining, or not changing.

Component 5: Synthesis and Management Recommendations

The results of the coupled nutrient loading (Component 1), estuarine biotic responses (Component 2), and biotic index development (Component 3) were analyzed to quantify spatial and temporal relationships between nutrient loading and biotic response/impact in the estuary. Water quality and sampling data were integrated into a GIS to identify hotspots of impaired water quality and eutrophication. Relationships between land use in the watershed and biotic conditions in the BB-LEH estuary were developed. From these data streams, watershed/estuary relationships and review of historic data related to the watershed and estuary, historical conditions, reference conditions (as defined by EPA-822-B-01-003, 2001), and current conditions throughout the study area were characterized. This is the data and information needed to synthesize comprehensive and representative nutrient criteria and a nutrient management plan. Recommendations for developing a management plan based on our findings will be given, and additional data and analysis needed to improve the plan will be listed.

COMPONENT 1: NUTRIENT LOADING ANALYSIS

The purpose of this component of the Barnegat Bay-Little Egg Harbor assessment project was to document the influence of human-altered land use on past and present nutrient export from the BB-LEH watershed to the BB-LEH Estuary, and quantify the spatial and temporal loading of nutrients. This component was necessary in order to link the effects of watershed nutrient loads to the environmental health of the estuary, as determined by quantitative measures of biotic, physical and chemical indicators.

Physical and chemical watershed data, land-use patterns, and a spatially explicit model were used to quantify loading of nitrogen and phosphorus species from the watershed to the estuary. In order to be consistent with the accompanying estuary research, loads and yields of nitrogen and phosphorus species were determined for years 1989-2011. Total nitrogen, total phosphorus, nitrate plus nitrite, ammonia, and organic nitrogen were quantified. Loads were calculated on an annual and seasonal (growing and non-growing seasons) basis. Baseflow loads were calculated directly from meteorological, hydrologic and water-quality data. PLOAD, Version 3.0 (U.S. Environmental Protections Agency, 2001) was used to simulate runoff loads.

Full details are compiled as an Appendix 1-1.

COMPONENT 2: ESTUARINE BIOTIC RESPONSES

This section of the report will briefly describe the data available and included in the overall study. Note, however, that the major objective of this component of the study is to characterize biotic responses in the estuary to nutrient loading and eutrophication using existing datasets collected between 1989 and 2010. Data collected in 2011 is used as a validation dataset. A significant outcome of this research is the determination of key biotic responses and associated thresholds of nitrogen enrichment that lead to shifts in ecosystem structure and function signaling eutrophic degradation. A biotic index of condition is also calculated to quantify the current and historical state of estuarine eutrophic effects. Several key bioindicators are used in development of the biotic index.

DATA COLLECTION METHODS

Physical and Chemical Measurements

Physical and chemical measurements made in the BB-LEH Estuary during 2004-2011 were compared to eelgrass characteristics. Water temperature, salinity, dissolved oxygen, pH, and depth were recorded at each sampling station on each sampling date (June/July, August/September, and October/November). These data were collected at a uniform depth (~10 cm) above the sediment-water interface using either a handheld YSI 600 XL datasonde coupled with a handheld YSI 650 MDS display unit, an automated YSI 6600 unit (equipped with a turbidity probe), or a YSI 600 XLM automated datalogger. Secchi depth was subsequently recorded. Water samples (N = 72) were also collected at all 12 transects in 2008 to determine nutrient concentrations (Kennish and Fertig, 2012). Laboratory analysis of the nutrients followed standard methods, with samples analyzed using a Lachat QuikChem FIA+ ® autoanalyzer. Additional physical and chemical measurements in BB-LEH were extracted as secondary data from long-term (1989-2011) quarterly water quality monitoring data available from the NJDEP.

Biotic Response Sampling

As noted above, comprehensive annual surveys were conducted in BB-LEH over the 2004-2011 period (excluding 2007) to obtain data on key biotic indicators used in this project (i.e., seagrass, macroalgae, epiphytes, and shellfish occurrence). Quadrat, core, and hand sampling was used to collect biotic samples along multiple transects in eelgrass beds in Barnegat Bay (~1550 ha) and Little Egg Harbor (~1700 ha) (Kennish et al., 2008, 2010) (Figure 1-8).

Sampling efforts were based on the SeagrassNet monitoring and sampling protocols of Short et al. (2002). The main modification of methods was establishing transects perpendicular to shore rather than parallel, to identify differences along a clearly defined depth gradient.

Eelgrass samples were collected during each of 3 time periods (June-July, August-September, and October-November) in all years. Widgeon grass was also collected, and sorted separately from eelgrass. The following eelgrass characteristics were recorded on all sampling dates at each sampling station: eelgrass occurrence, aboveground and belowground biomass, shoot density, blade length, and areal cover.

Quadrat Sampling

Based on the field sampling methods of Short et al. (2002), a 0.25-m^2 metal quadrat was haphazardly tossed at the sampling stations to obtain measurements of eelgrass and macroalgae areal cover. A diver estimated the percentage of the quadrat covered by eelgrass and macroalgae in increments of 5 along a scale of 0 to 100. The diver then visually inspected the eelgrass bed within the quadrat for occurrence of grazing, boat scarring, macroalgae, epiphytic loading, wasting disease, bay scallops, and hard clams. Each sampling station was also imaged using a digital camera to validate the diver observations. Subsequently, 5 replicate eelgrass blades were collected from within the quadrat, and blade lengths were measured.

Core Sampling

Coring methods also followed those of Short et al. (2002) using a 10-cm (.00785 m^2) diameter PVC coring device to collect the eelgrass samples within the quadrat, 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 eelgrass from the mesh bag, the sample was placed in a labeled bag and stored on ice in a closed container prior to transport to the Rutgers University Marine Field Station (RUMFS) in Tuckerton for laboratory analysis.

Laboratory Analysis

In the laboratory, the eelgrass samples were carefully sorted and separated into aboveground (shoots) and belowground (roots and rhizomes) components. The density of eelgrass shoots was then determined. The aboveground and belowground fractions were subsequently 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.

ECOSYSTEM PRESSURES

Water residence time and total nitrogen loading are the two key indicators of ecosystem pressure used in this project. Water residence times in the estuary range from 24 days in winter to 74 days in summer, when eutrophication is most pronounced (Guo et al., 2004). Nutrient loads from the watershed were determined annually for the time period from 1989 to 2011, including loads for total nitrogen and total phosphorus. Nutrient loads are presented in Appendix 1-1.

ECOSYSTEM STATE: WATER QUALITY

The second major category of data organization is ecosystem state which incorporates water quality variables (temperature, dissolved oxygen, total nitrogen concentration, and total phosphorus concentration) and parameters influencing light availability (chlorophyll *a*, total suspended solids, Secchi depth, macroalgae percent cover, and epiphyte percent cover). This category includes most of the project indicators. They are analyzed by segment for the estuary.

Temperature

Figure 2-1 shows the minimum, mean, and maximum temperatures recorded in the north, central, and south segments of the estuary from 1989 to 2010. Mean temperatures generally ranged from ~10-20 ºC. Minimum temperatures were less than 0 ºC, and maximum temperatures exceeded 30 ºC.

Dissolved Oxygen

The minimum, mean, and maximum concentrations of dissolved oxygen (DO) in the three estuary segments from 1989 to 2010 are illustrated in Figure 2-2. Mean DO levels generally ranged from ~4.5 to 8.5 mg L^{-1}. Minimum DO measurements were <3 mg L^{-1}, and maximum DO measurements were >12 mg L^{-1}.

Total Nitrogen

Figure 2-3 depicts the minimum, mean, and maximum concentrations of total nitrogen in the north, central, and south segments of the estuary from 1989 to 2010. Mean total nitrogen concentrations were <1000 μg L^{-1} in all estuarine segments year round. Maximum total nitrogen concentrations exceeded 1000 μg L^{-1} in the north segment of the estuary during all sampling periods from 1996 to 2010.

Total Phosphorus

The minimum, mean, and maximum concentrations of total phosphorus in the estuary are shown in Figure 2-4. Mean concentrations were <100 μg L^{-1} in all estuary segments and sampling periods from 1998 to 2010. Maximum concentrations often exceeded 100 μg L^{-1} during this period.

ECOSYSTEM STATE: LIGHT AVAILABILITY

Total Suspended Solids

Figure 2-5 illustrates the minimum, mean, and maximum total suspended solids (TSS) recorded in the north, central, and south segments of the estuary from 1989 to 2010. Mean TSS values generally ranged from 5-40 TSS units. Maximum TSS values exceeded 200 TSS units.

Secchi Depth

The minimum, mean, and maximum Secchi depths recorded in the north, central, and south segments of the estuary from 1989 to 2010 are depicted in Figure 2-6. Secchi depths generally exceeded 2 m in all estuary segments. Minimum mean Secchi depths were ~1 m.

Chlorophyll *a*

Figure 2-7 shows the minimum, mean, and maximum chlorophyll *a* recorded in the north, central, and south segments of the estuary from 1997 to 2010. Mean chlorophyll *a* measurements generally ranged from ~1-12 mg L^{-1}. Maximum chlorophyll *a* values exceeded 40 mg L^{-1}.

Macroalgae Percent Cover

Macroalgae percent cover is listed as an ecosystem state parameter because macroalgal canopy effectively shades or attenuates light to seagrass beds. As such, it must be considered as a factor influencing light availability to the benthos. More than 110 benthic macroalgal species have been identified in BB-LEH (Kennish, 2001a; Kennish et al., 2010). Both perennial forms and ephemeral, bloom-forming species occur in the estuary, with many comprising a drift community unattached to any substrate. Sheet-like masses of some species (e.g., *Ulva lactuca* and *Enteromorpha intestinalis*) are particularly problematic because they grow rapidly when light and nutrient conditions are favorable, outcompeting seagrasses and other vascular plants that constitute essential benthic habitat in the system (Coffaro and Bocci, 1997; Nelson and Lee, 2001).

In the nutrient enriched waters of this coastal lagoon, bloom-forming macroalgal species have been observed to form dense canopies more than 25-cm thick overlying seagrass beds, which block light transmission to the beds (Twilley et al., 1985). As the algal standing stocks increase, shading reduces the photosynthetic oxygen production of the seagrass plants causing diebacks (Twilley et al., 1985; Lee et al., 2007; Ralph et al., 2007). In addition, the accumulation and decomposition of decaying plant matter and ooze in bottom sediments can result in high concentrations of sulfide in the rhizosphere that decrease nutrient uptake and contribute to additional reduction in photosynthesis, growth, and leaf density, and an increase in ammonium, oxygen depletion, and seagrass mortality (Holmer and Bondgaard, 2001; Burkholder et al., 2007; McGlathery et al., 2007).

The areal percent cover of macroalgae was recorded for each sampling station. Macroalgae areal cover of 60-70% was considered 'Pre-Bloom', 70-80% was considered 'Early Bloom', and > 80% was considered 'Full Bloom' conditions (Kennish et al., 2011). The mean percent cover of macroalgae at sampling stations along each transect is illustrated in Figure 2-8. The absolute percent cover at all sampling stations ranged from 0-100%, and the mean percent cover of macroalgae ranged from 2-21% in the central and south segments of the estuary (Table 2-1, Figure 2-9).

Table 2-2 shows the frequency of occurrence of macroalgal bloom conditions in the estuary for each survey year from 2004 to 2010. There were 10 occurrences (0.45 blooms m^{-2}) of Pre-Bloom conditions (60-70% macroalgae cover), 19 occurrences (0.67 blooms m^{-2}) of Early Bloom conditions (70-80%), and 36 occurrences (1.57 blooms m^{-2}) of Full Bloom conditions (80-100%), indicating that macroalgal blooms developed relatively frequently in the estuary. Blooms were more frequent during June-July (27 occurrences, 1.10 blooms m^{-2}), and August-September (22 occurrences, 0.95 blooms m^{-2}), than October-November (16 occurrences, 0.63 blooms m^{-2}). The majority of the blooms occurred during the 2008-2010 period. There were 6 occurrences of Pre-Bloom conditions (0.20 blooms m^{-2}), 17 occurrences of Early Bloom conditions (0.57 blooms m^{-2}), and 24 occurrences of Full Bloom conditions (0.80 blooms m^{-2}) during the 2008-2010 time period. Field observations indicated that macroalgae blooms in the estuary not only developed relatively frequently, but also impacted seagrass beds. Macroalgae blooms are an important driver of change in seagrass habitat of the estuary.

Macroalgal areal cover did not exhibit significant change over 2004-2010 during the June-July and October-November sampling periods, but did exhibit a significantly declining trend (-1.5 % year-1, R^2 = 0.03, F = 19.6, p < 0.01) during the August-September time period (Table 2-3a). Although macroalgal blooms did not cover the entire area of the seagrass beds at any time during this study, the cumulative impact of the blooms across multiple locations within the beds resulted in acute loss of vegetation and the genesis of extensive bare bottom areas. *Ulva lactuca* blooms were particularly damaging in this regard.

In most years (2005, 2006, 2008, 2009), macroalgae areal percent cover significantly varied (p < 0.01) over the course of the year but did not do so consistently across years (Table 2-3b). Macroalgae areal percent cover significantly increased by time period in 2006 and 2009, decreased by time period in 2005 and 2008, and did not significantly change during 2004 and 2010 (Table 2-3b).

Benthic macroalgae are powerful drivers of change in water quality and seagrass habitat. During bloom conditions, benthic macroalgae formed a dense canopy over extensive areas of the seagrass beds. Macroalgae areal percent cover significantly correlated with multiple water quality and seagrass properties, most frequently during the June-July time period throughout 2004-2010 (Table 2-4). For example, during June-July 2004-2010, macroalgae areal percent cover negatively correlated with dissolved oxygen concentration (r = -0.11, p < 0.05, n = 550), but positively correlated with *Zostera marina* aboveground and belowground biomass (r = 0.19, p < 0.01, n = 571 and r = 0.16, p < 0.01, n = 571, respectively) and *Zostera marina* blade length (r = 0.22, p < 0.01, n = 440). These relationships did not remain significant throughout the year. Only *Zostera marina* blade length continued to be significantly correlated by August-September (r = 0.10, p < 0.05, n = 449), and none were significantly correlated during October-November (Table 2-3). Conversely, while no significant relationships between macroalgae percent cover and *Ruppia maritima* aboveground or belowground biomass were observed during June-July

2004-2010 or August-September 2004-2010, these variables were positively correlated during October-November 2004-2010 (r = 0.38, p < 0.01, n = 60 and r = 0.27, p < 0.05, n = 60) (Table 2-4).

A total of 39 macroalgal species were recorded over 2004-2005, with bloom-forming red and green algae dominating the assemblages (Kennish et al., 2010). In 2004, the sea lettuce *Ulva lactuca* was the most abundant species, occurring in 59% of the samples collected. Three red macroalgal species were also abundant, notably *Spyridia filamentosa* (55%), *Gracilaria tikvahiae* (30%), and *Champia parvula* (23%). In 2005, four red and one green macroalgal species predominated: *G. tikvahiae* (present in 70% of samples), *Bonnemaisonia hamifera* (56%), *Spyridia filamentosa* (46%), *U. lactuca* (26%), and *C. parvula* (19%).

Macroalgal blooms contributed in part to the decline of seagrass biomass in BB-LEH over the 2004-2010 period (Kennish et al., 2008, 2010, 2011). Orth et al. (2006) documented that seagrasses have high light requirements that approach 25% of the incident surface radiation (Dennison et al., 1993; Orth et al., 2006). Light extinction by macroalgae mats during bloom development threatens seagrass integrity. Macroalgae require lower light intensities than seagrass for survival (Hily et al., 2004; McGlathery et al., 2007). Hence, reduced light transmission to the estuarine floor can lead to the replacement of seagrass by rapidly growing macroalgae (e.g., *Ulva lactuca* and *Enteromorpha* spp.).

Similar bloom events in the estuary have been previously reported. For example, in 1998, Bologna et al. (2000, 2001) documented heavy benthic macroalgal blooms in the BB-LEH Estuary consisting of *Ulva*, *Gracilaria*, and *Codium*. Algal-detrital loading rates of ~400 g ash free dry weight m^{-2} derived from these blooms persisted throughout the summer and into the fall, burying extensive areas of *Z. marina* beneath a thick algal canopy. The positive correlations between *Z. marina* biomass (aboveground and belowground) and blade length in June-July reported here (Table 2-4) likely happen because larger seagrass blades trap more floating macroalgae, but once at full size later in the year, this relationship is no longer significant, and shading results in the rapid loss of aboveground and belowground biomass at several locations in the estuary (Bologna et al., 2001). Seitzinger et al. (2001) showed that benthic algal dynamics can significantly influence sediment-water nutrient fluxes in the estuary, particularly ammonium which may sustain system eutrophy.

The loss of seagrass due to the reduction in light availability from macroalgal blooms is likely accelerated by altered biogeochemical conditions in bottom sediments associated with the accumulation and decomposition of the increased algal standing stocks (Hauxwell et al., 2001, 2003; Nixon et al., 2001). The decomposition of the macroalgae causes higher nutrient efflux from the sediments to the water column enhancing eutrophication in eutrophied systems (Eyre and Ferguson, 2002). It also results in sulfide production in the rhizosphere which decreases nutrient uptake, seagrass photosynthesis, metabolism, and growth, while increasing the development of hypoxic/anoxic conditions hazardous to benthic communities (Goodman et al.,

1995; Erskine and Koch, 2000; Holmer and Bondgaard, 2001; Ralph et al., 2006). Seagrass mortality can also increase significantly in response to oxygen depletion and high pore-water ammonium concentrations (McGlathery et al., 2007).

Epiphyte Percent Cover

Seagrass leaves provide excellent substratum for epiphytic organisms which often contribute significantly to the total primary and secondary production of seagrass meadows. Epiphytic algae, or periphyton, can account for more than 50% of the total primary production in a seagrass bed, generating a rich food supply for numerous primary consumers (Borowitzka et al., 2006). They can also comprise up to 67% of the total biomass of a seagrass bed (Saunders et al., 2003). Periphyton enhances the habitat value of seagrass leaves and creates a more complex habitat within a seagrass biotope (Bologna and Heck, 1999). Despite their contribution to estuarine food webs, epiphytic assemblages reduce light availability to the seagrass blades, frequently leading to considerable loss of plant biomass and areal cover (Hily et al., 2004). When present in high abundance, epiphytes can attenuate up to 90% of light incident on seagrass blades (Brush and Nixon, 2002; McGlathery et al., 2007). Suspended particulates and dissolved substances in the water column may exacerbate these effects.

Seagrass epiphytic communities are highly variable on both temporal and spatial scales. They consist of complex and diverse interactive constitutents – bacteria, fungi, microalgae and macroalgae, herbivorous grazers, as well as organic detritus and inorganic debris typically characterized by measurement of biomass (total dry weight or ash free dry weight) (Brush and Nixon, 2002). Aside from providing habitat for epiphytic algae, seagrass leaves also serve as hosts for a wide array of epifaunal groups, both sessile and vagile forms (e.g., ascidians, barnacles, bryozoans, hydroids, polychaetes, sponges, and other taxonomic groups), which increase the habitat heterogeneity within the seagrass canopy leading to greater species richness and density of organisms (Bologna and Heck, 1999; Hily et al., 2004). The abundance and distribution of epiphytic algae, therefore, influence the abundance and distribution of faunal grazers (Fong et al., 2000; Borowitzka et al., 2006).

Grazers can control epiphytic biomass by consuming algal epiphytes plus host substrates (Peterson and Heck, 2001). Duffy et al. (2001) showed that amphipods, isopods, and copepods are important grazers of eelgrass (*Zostera marina*) periphyton. Nutrient enrichment typically enhances epiphytic biomass and productivity in a seagrass bed, while grazing suppresses both (Hasegawa et al., 2007; Jaschinski and Sommer, 2008). Escalating eutrophic conditions promote epiphytic growth on seagrass leaves, diminished light availability, and loss of seagrass (Hily et al., 2004; McGlathery et al., 2007).

The composition and abundance of epiphytic assemblages can vary greatly along an estuarine gradient in response to variable nutrient loading. Saunders et al. (2003) reported that the composition of epiphytic assemblages was reasonably consistent within a *Z. marina* bed, but exhibited significant differences at greater distances across beds at the scale of a kilometer or

more. Frankovich and Fourqurean (1997) observed pronounced compositional shifts in epiphytic assemblages across a nutrient gradient. The effect of nutrient enrichment on epiphytic loading was localized but pronounced.

Epiphytic areal cover on seagrass leaves was determined by collecting the five longest leaves from each bottom sample and visually estimating the epiphytic percent cover on both the upper and lower leaf surfaces. Using a razor, the epiphytes were subsequently scraped off of both sides of the blades and oven dried at 60°C for 48 hours to determine their biomass. The dry weight biomass of both the epiphytes and seagrass blades was then recorded to the fourth decimal place. Biomass values of both the eelgrass blades and epiphytes were recorded separately.

Table 2-5 shows the percent cover of epiphytes on seagrass leaves collected at the transect stations during the three sampling periods in 2009 and 2010. The data indicate very similar values on both upper and lower leaf surfaces of *Zostera marina* samples. The mean percent cover of epiphytes during all sampling periods in 2009 ranged from 19.2 to 38.3% for upper leaf surfaces and 18.4 to 38.3% for lower leaf surfaces. In 2010, the mean percent cover of epiphytes was generally lower than in 2009, with the values ranging from 11.3 to 25.7% for upper leaf surfaces and 10.7 to 24.4% for lower leaf surfaces. However, higher values of epiphyte percent cover were found during the October-November sampling period in 2010 than in 2009, with the mean upper leaf and lower leaf percent cover values ranging from 20 to 21% in October-November 2010 compared to values ranging from 18.4 to 19.2% in October-November 2009.

Epiphyte biomass in 2009 peaked during June-July (mean = 121.8 mg dry wt m^{-2}). In 2010, peak epiphyte biomass occurred during August-September (mean = 67.7 mg dry wt m^{-2}). The maximum biomass of epiphytes also occurred at the time of peak epiphyte areal cover on eelgrass leaves.

ECOSYSTEM BIOTIC RESPONSE

Eelgrass (*Zostera marina* L.)

Results of this project show conclusively that eelgrass condition in BB-LEH has declined substantially through time and that the rate of decline is related to nutrient loading and associated symptoms of eutrophication. In addition, the degradation rate has changed over time. 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 (except blade length) was evident from 2008 to 2010 corresponding with temporal separation (yearly and seasonally of environmental parameters suggests their importance to seagrass condition). Trends of eelgrass characteristics indicated that eelgrass biomass had yet to recover by 2010 from the

decline of plant abundance and biomass observed in 2006 (Kennish et al., 2007b, 2010). However, the rate of decline of eelgrass biomass during 2008-2010 was slower than that of 2004-2006, perhaps because less was left to be lost. Thus, biomass may be reaching a new, lower, steady state. Return to previous levels of eelgrass biomass may be difficult to attain (Duarte et al., 2009).

Though long-term monitoring was not started early enough to observe the beginning of the initial 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).

Eelgrass Biomass

Eelgrass biomass declined consistently over the 2004-2006 and 2008-2010 periods and overall from 2004-2010. The biomass in 2010 was the lowest recorded for BB-LEH (Figure 2-10). 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 (Figures 2-11 and 2-12).

Figure 2-13a-c shows relationships of chlorophyll a vs. total nitrogen (a), dissolved oxygen vs. total nitrogen (b), and dissolved oxygen vs. chlorophyll a (c) over the 2004-2010 period. Trends of eelgrass biomass showed that belowground biomass was consistently higher than aboveground biomass each year (Table 2-6). The rate of decline in eelgrass biomass was significantly sharper during 2004-2006 than in 2008-2010. Regression analysis indicated a slope of -23.8 g m^{-2} yr^{-1} (intercept = 47,765, R^2 = 0.14, p < 0.01) during 2004-2006 and -8.7 g m^{-2} yr^{-1} (intercept = 17,496, R^2 = 0.04, p < 0.01) during 2008-2010. A t-test comparing these slopes showed a significant difference (t = -6.13, p < 0.01), indicating that the decline slowed significantly in the latter three years, as can be seen in Figure 2-13d-f. In contrast, though belowground biomass also consistently declined, regression slope during 2004-2006 was -17.0 (intercept = 34,189, R^2= 0.02, p < 0.01) and during 2008-2010 was -18.4 (intercept = 37,028, R^2 = 0.04, p < 0.01), but these two slopes did not significantly differ (t = 0.25, p = 0.80).

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 2-6). 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.

Eelgrass biomass decreased during the period of increased macroalgal bloom and elevated epiphyte occurrence. The reduction of eelgrass biomass begins relatively early in the growing season each year (Table 2-6), indicating once again that the threshold value of nutrient loading leading to a substantive decline in eelgrass biomass is likely exceeded early in the growing season (June-July).

Eelgrass Shoot Density

Shoot density of eelgrass varied by sampling periods and segments (Figure 2-14), but a significant interaction term required simple effects to be reported. Highest shoot density occurred in 2010, with peak values (mean = 665 ± 460 shoots m^{-2}) recorded in June-July (Table 2-6). Lowest shoot density values were recorded in 2004 and 2006, with intermediate shoot density numbers reported in 2005, 2008, and 2009. The highest mean eelgrass shoot density measurements in 2008 were recorded during the August-September (414 ± 570 shoots m^{-2}) sampling period. Significantly lower densities of eelgrass were found in 2008 during the June-July (241 ± 435 shoots m^{-2}) and October-November (264 ± 464 shoots m^{-2}) sampling periods. Highest eelgrass shoot density also coincided with peak aboveground biomass in 2008. In 2009, the eelgrass shoot density pattern differed from that observed in 2008, with the highest mean shoot density documented during the June-July sampling period (346 ± 536 shoots m^{-2}) and progressively lower mean densities found during the August-September (265 ± 407 shoots m^{-2}) and October-November (155 ± 325 shoots m^{-2}) sampling periods. The declining eelgrass shoot density across the sampling periods in 2009 was consistent with the gradual decrease in aboveground and belowground eelgrass biomass at these times (Table 2-6). Shoot density was much lower during the summer-fall period in 2009 than in 2008.

Eelgrass Blade Length

Figure 2-15 shows the mean blade lengths of eelgrass in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010. Blade lengths were lowest in both segments during the heavily impacted year of 2006. Somewhat lower blade lengths were also recorded in 2009 and 2010. Transect explained 34% of the variation in eelgrass blade length.

The mean lengths of eelgrass blades in 2004 were 34.0 ± 10.9 cm in June-July, 32.2 ± 7.2 cm in August-September, and 31.8 ± 8.4 cm in October-November. By comparison, in 2005 the mean blade lengths of eelgrass amounted to 32.7 ± 17.6 cm in June-July, 25.9 ± 14.9 cm in August-September, and 28.5 ± 14.7 cm in October-November. Sharply lower mean blade length measurements were recorded during the heavily impacted year in 2006 as is evident by measurements in June-July (22.2 ± 24.6 cm), August-September (3.7 ± 9.8 cm), and October-November (4.6 ± 9.8 cm). The last two sampling periods in 2006 showed marked reductions in eelgrass blade lengths.

Blade lengths were more consistent during 2008, averaging 28.6 ± 12.2 cm in June-July, 22.4 ± 13.6 cm in August-September, and 31.4 ± 17.7 cm in October-November. They were somewhat reduced in 2009, when the mean lengths of eelgrass blades were 22.3 ± 13.2 cm in June-July, 24.5 ± 11.6 cm in August-September, and 21.5 ± 10.8 cm in October-November. Mean blade lengths were similar in 2010 to those in 2009, amounting to 22.2 ± 12.5 cm in June-July, 19.9 ± 10.6 cm in August-September, and 22.7 ± 13.4 cm in October-November.

Eelgrass Areal Cover

The percent cover of eelgrass was similar from 2004 to 2008 (Table 2-6). In 2004, the mean percent cover of eelgrass progressively decreased from a high of 44.8% ± 27.6% in June-July to 37.6 ± 31.3% in August-September and 21.4 ± 23.3% in October-November. A similar progressive decline was evident in 2005 when the mean percent cover of eelgrass decreased from 36.9 ± 33.1% in June-July to 23.1 ± 35.1% in August-September and 11.3 ± 11.3% in October-November. In 2006, however, the lowest mean percent cover was recorded in August-September (13.5 ± 20.6%), with higher areal cover reported in June-July (23.5 ± 35.8%) and October-November (16.4 ± 24.0%). The low eelgrass areal cover in 2006 was evident in both the central and south segments of the estuary (Figure 2-16). In 2008, the mean percent cover of eelgrass was lowest in June-July (22.2 ± 29.9%) and October-November (22.3 ± 31.1%), and highest in August-September (29.6 ± 36.3%). By comparison, the percent cover of eelgrass in 2009 decreased from 31.3 ± 35.5% in June-July to 27.2 ± 34.8% in August-September, and then decreased greatly to 14.6 ± 19.0% in October-November. Lower values were found during all sampling periods in 2010; the mean percent areal cover declined from a peak of 28.2 ± 35.7% in June-July to 21.0 ± 34.5% in August-September, and 9.2 ± 21.0% in October-November. Figure 2-17 shows the areal eelgrass cover by sampling transect during 2010.

Eelgrass Demographics

Though biomass declined from 2004-2010, the mean number of shoots generally increased from 2004 to 2010 (Table 2-6), although it decreased substantially in 2011 (see Component 4). Calculated values of r_x ranged from -0.15 yr^{-1} to +1.0. yr^{-1}; the growth rate ranged from 0.86 yr^{-1} to 1.46 yr^{-1} and negatively related to total nitrogen concentrations (Figure 2-18). Instantaneous mortality ranged from -0.80 yr^{-1} to +0.31 yr^{-1} (Table 2-7). Aside from the first year of observations, the highest proportion of the age-distribution was calculated to occur in 2010.

Widgeon Grass (*Ruppia maritima*)

Table 2-8 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 2-19 to 2-22). 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 were reported in 2005 and 2010 on the central and south segments (Table 2-8). 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%).

Other Biotic Components

Harmful Algal Blooms

Brown-tide blooms caused by the pelagophyte *Aureococcus anophagefferens* were most pronounced in BB-LEH between 1995 and 2002 (Table 2-9). While brown tides reached high densities during this span of years, they have not been monitored in the estuary since 2004, and thus no observational HAB monitoring data are available over the past eight years. Brown tides have also been reported in New York coastal bays since the mid-1980s, and in the Maryland coastal bays since 1998. Brown tides are detrimental to coastal bay ecosystems. They often discolor the water and cause negative impacts on shellfish populations (e.g., hard clams and bay scallops) and seagrasses. Gastrich and Wazniak (2002) showed that elevated levels of brown tide may significantly reduce feeding and growth of shellfish (e.g., hard clams and mussels), cause recruitment failures, and high mortality (e.g., bay scallops). Dense shading of these blooms may reduce the abundance and distribution of seagrass beds, which serve as important habitat for fish, shellfish, and other organisms. During 2000-2002, the levels of brown-tide blooms in the BB-LEH were elevated compared to levels in other estuaries that exhibited negative impacts on natural resources (Gastrich et al., 2004).

Abundances of *Aureococcus anophagefferens* in the estuary were classified using the Brown Tide Bloom Index (Gastrich and Wazniak, 2002) and mapped, along with salinity and temperature parameters, to their geo-referenced location using the ArcView GIS (Gastrich and Wazniak, 2002; Gastrich et al., 2004). The highest *A. anophagefferens* abundances (>10^6 cells mL), Category 3 blooms (200,000 cells mL^{-1}) and Category 2 blooms (35,000 to 200,000 cells mL^{-1}), occurred in 1997 and 1999 and then recurred during the 2000-2002 period, covering significant geographic areas of the estuary, especially in Little Egg Harbor (Gastrich et al., 2004). Warmer water temperatures (> 16°C) and higher salinities (> 25-26 ppt) were generally associated with Category 3 blooms, but these factors did not completely explain the timing or distribution of the blooms (Gastrich et al., 2004). Dissolved organic nitrogen concentrations were not directly linked to the blooms, which may be more closely aligned with the

concentrations of dissolved organic nitrogen in the estuary.

Extended drought conditions, low freshwater inputs, and elevated bay salinity that occurred during the 2000-2002 period appeared to promote the blooms (Gastrich et al., 2004). Abundances of *A. anophagefferens* were well above those reported to cause negative impacts on shellfish. Category 3 blooms generally occurred at water temperatures above 13-17 °C and within a salinity range between 25 and 31 ppt. An assessment of the risk of SAV habitat to brown-tide bloom categories indicates that 35% of the SAV habitat located in BB-LEH had a high frequency of Category 2 or 3 blooms for all three years of study (2000-2002). This is important considering that more than 75% of the New Jersey's eelgrass beds are located in this system (Lathrop et al., 2001), and brown tides may pose a serious risk to this habitat.

Although the presence of *Aureococcus anophagefferens* was first reported in New Jersey's coastal bays in 1988, with blooms documented in 1995, 1997 and 1999, there were insufficient data to develop trends. A monitoring program of NJDEP showed a trend in elevated abundances of brown tide from 2000-2002. However, no Category 3 blooms occurred in 2003 and 2004, indicating that high density brown tide blooms do not occur every year in the estuary. GIS analysis has shown that some seagrass habitat lies within the High-Risk Category 3 bloom 'hotspot' areas and therefore should be monitored on an annual basis.

Shellfish

Hard clam (*Mercenaria mercenaria*) harvest in BB-LEH decreased by more than 98% between 1975 and 2005 (from 636,364 kg in 1975 to 6,820 kg in 2005), with harvest statistics being unreported since 2005 (Figure 1-3). The NJDEP surveyed Barnegat Bay and Little Egg Harbor in 1985/86 and reported that hard clam population was present at densities of 1.4 and 2.5 m^{-2}, respectively. Little Egg Harbor was resurveyed in 2001, and the population density had dropped to 0.81 m^{-2} (Celestino, 2003). Based on a modeling study of the hard clam population in Islip town waters of Great South Bay, New York (Hofmann et al., 2006), a density of ~0.7 clams m^{-2} was found to be the minimum necessary to sustain the hard clam population (Kraeuter et al., 2005).

Of even greater concern was the marked decline in the hard clam stock abundance documented in Little Egg Harbor between 1986/87 and 2001. As reported by Celestino (2003), a total of 64,803,910 hard clams were estimated in LEH in 2001 compared with an estimated 201,476,066 in 1986/87, representing a decrease of over 67% in stock abundance over this period. The decline in hard clam abundance per station between the two survey years was significant *(P << 0.0002, P << 0.0002, P < 0.0001 and P < 0.0001)*. The mean size of hard clams collected in 2001 was 78.9 mm and represented a significant increase from 1986/87's mean size of 74.6 mm *(P < 0.0002)*. Recruitment indices, based on a percentage of hard clams between 30 and 37 mm collected at a specific site as compared to all sized clams collected at the same site, were significantly lower in 2001 than in 1986/87 *(P = 0.025)*. Mortality estimates were significantly greater in 2001 than in 1986/87 *(P << 0.0002)*. These statistics indicate a

shellfish population in serious decline. The loss of such large numbers of hard clams also appears to reflect a shift or transition in the system away from one of top-down control exerted by filter feeders consuming and regulating phytoplankton populations to one of bottom-up control limited by nutrient inputs.

The hard clam survey in Little Egg Harbor in 2001 occurred during a major brown tide bloom event, and subsequent to major brown tide bloom occurrences in 1999 and 2000 (Table 2-9). Eutrophication may cause significant changes in the food supply of suspension feeders. Bricelj and MacQuarrie (2007) and Bricelj (2009) have discussed the effects of brown tides on hard clams. The shift in food supply from larger diatoms and dinoflagellates to picoplanktonic pelagophytes such as *Aureococcus anophagefferens* may lead to poor growth and compromised reproductive success of hard clams, as well as poor fertilization, lower clam densities, and even altered abundances of predator populations. BB-LEH has not only exhibited a shift towards picoplanktonic pelagophytes during the past 15 years, but also has supported high abundances of other small forms such as the green alga *Synechococcus* sp. and the chlorophyte *Nannochloris atomus* (Olsen and Mahoney, 2001). Bricelj et al. (1984) has shown that these smaller phytoplankton species are poorly captured and digested by hard clams, thereby having the potential to seriously impact their growth.

Benthic Invertebrates

The USEPA collected benthic invertebrate samples at ~80 stations in the BB-LEH Estuary in 2001 as part of the Regional Environmental Monitoring and Assessment Program (REMAP) (Figure 2-23). A major goal of this project was to obtain the benthic samples in a manner consistent with EMAP's probabilistic statistical sampling design to effectively characterize the benthic invertebrate community structure contributing to the development of a benthic index of ecosystem condition. The sampling design is based on a single, annual sampling season of each station. However, the samples were not collected concurrently, but at different times in different segments of the estuary from June to August in 2001. In addition, biomass data for benthic invertebrates were not determined, which is inconsistent with benthic indices developed for other benthic invertebrate sampling programs.

National Coastal Assessment (NCA) benthic invertebrate samples collected annually in the estuary from 2000 to 2006 were not sufficiently abundant to be used in index development for this project. For example, only 4 NCA benthic invertebrate samples were collected in 2000, 2003, and 2005, while 6 samples were collected in 2002, 10 in 2004, 15 in 2001, and 16 in 2006 (Table 2-10), far too few for adequate statistical analysis for the three segments of the estuary.

An external project (i.e., benthic invertebrate indicator development project by Gary Taghon, Institute of Marine and Coastal Sciences, Rutgers University) has shown that the Virginia Province Index has incorrectly categorized many stations according to environmental conditions. In addition, ANOVAs and PCA analysis applied in this project indicate that the NCA dataset in insufficient to characterize variability in benthic habitats. Systemic errors also exist in

the NCA dataset. For example, salinity normalized total abundance significantly correlated to salinity, but it should not. Normalization should remove any correlative effect, so an inherent problem exists in the database. Significant positive correlations between salinity and most variables (exceptions of salinity-normalized-Gleason's D, I, and % Spionidae) were found, and salinity significantly differed by segment, though these other variables did not vary by segment. Most unfortunately, benthic invertebrate biomass data are unavailable in the NCA samples, but are required for existing benthic indices. These flaws in the NCA dataset cannot be overcome. Thus, these data were not included in the index of eutrophication. There is sufficient data in the REMAP database from 2001 to characterize heterogeneous habitats, and therefore this dataset was used for index development in this project, although only one year of data is represented.

Additionally, several other datasets were evaluated for suitability for inclusion in calculations of the Index of Eutrophication. Examples include NCA data (2000-2006), residence time, hydrodynamic modeling, GIS layers of seagrass coverage, counts of jellyfish, and several others. Examples of qualitative and quantitative criteria for inclusion are the number of records, location and span of dates of data collection, and ability to describe and detect heterogeneity between segments and years. Examples of statistical procedures that have been used to evaluate datasets have included (but have not been limited to) ANOVAs between segments, PCA, correlation with salinity/habitat, assessment of data availability. These evaluations indicated that the datasets mentioned above did not meet criteria for BB-LEH and cannot be included in the index. For more information about dataset evaluation for the Index of Eutrophication see Appendix 3 - 2.

COMPONENT 3: BIOTIC INDEX DEVELOPMENT

SUMMARY AND CONCLUSIONS

- The Biotic Index of Eutrophication is the most comprehensive and holistic assessment of BB-LEH conducted to date. In order to assess the ~20 indicators, the index integrates over 74,400 observations among 85 variables.

- Outputs of the index are quantitative annual assessments for 3 areas on a scale of 0-100: 0-20 =Highly Degraded, 20-40=Poor, 40-60=Moderate, 60-80=Good, 80-100=Excellent. Index scores assess condition and its consistency.

- Data availability remains a major limitation to assessment of eutrophication condition for BB-LEH. While an increasing number of indicators are being monitored, aligning data collection through space and time and increasing sampling frequency will greatly improve future assessments.

- The Index of Eutrophication is calculated for BB-LEH that includes a suite of ~20 metrics that are organized into six components: (1) Ecosystem Pressures, (2) Water Quality, (3) Light Availability, (4) Seagrass Response, (5) Harmful Algal Blooms, and (6) Benthic Invertebrate Response.

- Several key categories of data organization are analyzed in the index development process. Total nitrogen loading and water residence time are the two key indicators of Ecosystem Pressure. The second major category of data organization is Ecosystem State, which incorporates water quality variables (temperature, dissolved oxygen, total nitrogen concentration, and total phosphorus concentration) and parameters influencing Light Availability (chlorophyll *a*, total suspended solids, Secchi depth, macroalgae percent cover, and epiphyte percent cover). This category includes most of the project indicators. For ecosystem biotic response, key indicators of measurement for the project include seagrass biomass, shoot density, blade length, and areal cover; harmful algal blooms; and benthic invertebrate and shellfish abundance response. All of these indicators are analyzed by segment (north, central, and south) for the estuary.

- Observations of indicators are compared to thresholds to rescale measurements into indicator scores. Indicator scores are averaged together to calculate a Raw Score for each indicator in each component. The variability (calculated as the square of the eigenvector) for each indicator is used to weight each indicator score, which is then used to calculate a Weighted Score for each indicator in each component. The Raw Score and the Weighted Score are then summed to calculate an index for each component. The component indexes are then averaged to calculate the overall Index of Eutrophication.

- Sensitivity analyses conducted on the indicators in the water quality component tested the

impact of including or excluding indicators (which is necessary according to data availability) as well as the impact of calculating the weighting based on variability within a year, and over sets of multiple years.

- Eutrophication condition declined 34% and 36% in the central and south segments from 73 and 71 in the 1990s to 48% and 45% in 2010, respectively, indicating they are undergoing eutrophication. Overall eutrophication condition is worst in the north segment but has improved modestly, in contrast to stages and trends in the south and central segments. Scores in the north segment declined sharply in 2010 (to 37, Poor), but the highest score observed in the north segment (50, Moderate) was in 2009, 3.5 times its low score (14, in 1991).

- Total nutrient loadings were Highly Degraded in the north segment, but Moderate in central and south segments. During 1989-1997, low DO countered favorable temperatures leading to Moderate conditions. Favorable temperatures continued in 1998-1999, but TP increased in 2000-2003. In 1998-2003, TSS was Moderate/Good, epiphytic loading was Poor/Moderate, % surface light reaching seagrass was Highly Degraded/Poor, declining in 1998-2002 in the north and south segments. In 2004-2010, TP condition in BB-LEH fell from Poor to Highly Degraded. TSS improved steadily in the north segment, variably in the south segment, and temporarily declined in 2004-2007 in the central segment. Similar temporary Poor/Highly Degraded condition in 2004-2009 in the central segment was seen in epiphytic load and % surface light reaching seagrass. Seagrass cover and length condition worsened over 2004-2010: ModeratePoor and PoorHighly Degraded, respectively.

- Nutrient loading severely degraded BB-LEH, particularly in 2003-2010, degrading condition from Good to Poor/Highly Degraded. Initial rapid declines highlight sensitivity to loading. Beyond ~2,000 kg TN km^{-2} yr^{-1} or ~100 kg TP km^{-2} yr^{-1}, condition plateaus as Poor/Highly Degraded yet variability increases, suggesting a switch in dominant factors. Perhaps this is due to community shifts, e.g., from blooms of brown tide (> 1.8 x 10^6 cells mL^{-1} in 1999-2002) to macroalgae (1998, 2004, 2005, 2008-2010).

- Overall eutrophication is greatly worsened by increasing total nitrogen loading and total phosphorus loading. Initially, there are sharp declines in condition with even small increases in nutrient loading, as is the case in the central and south segments. Once loading increases beyond 2000 kg TN km^{-2} yr^{-1} or 100 kg TP km^{-2} yr^{-1}, as is the case in the north segment, eutrophication condition reaches a new, lower steady state of Poor condition.

- Total nitrogen loading and total phosphorus loading scores were lower (more degraded) during 2003-2010 than in previous years. Loading for both nutrients was higher in the north segment than the south or central segments, and thus nutrient loading in the north segment is considered 'Highly Degraded'. It is considered 'Moderate' in the central and south segments.

- Total nitrogen concentration scores were generally lowest in the north segment. Scores for total nitrogen, total phosphorus, and dissolved oxygen were either 'Highly Degraded' or 'Poor'. Overall, water quality condition has been declining throughout the estuary since the early 1990s. The poor condition of nutrients and oxygen in the estuary is directly related to

the nutrient loading from the watershed.

- Overall, light availability has been increasing in the north and central segments. Light availability greatly worsened, though temporarily, during 2005-2008 in the central segment. By 2010, overall light availability was considered 'Good' throughout the estuary. In particular, concentrations of chlorophyll *a* were low enough to be considered 'Good' throughout the estuary, while concentrations of total suspended solids were considered 'Excellent' throughout the estuary. The ratio of epiphytes to seagrass biomass was 'Moderate' in the north segment and Excellent in the central and south segments. Nevertheless, light did not penetrate deep enough into the estuary, and the percent light reaching seagrass was Poor in the north segment, Moderate in the south segment, and Good in the central segment.

- Though percent cover and shoot density indicators had slightly higher scores ('Poor'), the overall seagrass response is 'Highly Degraded' throughout the estuary.

- Five of the seven years of available data for Harmful Algal Blooms result in Highly Degraded scores for this indicator.

BACKGROUND: BUILDING ON THE NATIONAL ESTUARINE EUTROPHICATION ASSESSMENT

We applied the basic methodology used in the National Estuarine Eutrophication Assessment (NEEA) Model to develop a biotic index of eutrophic condition for the BB-LEH Estuary (Bricker et al., 1999, 2007). The NEEA uses the ASSETS model (Assessment of Estuarine Trophic Status) to examine and combine: (1) Influencing Factors, (2) Eutrophic Symptoms, and (3) Future Outlook to arrive at a qualitative assessment for each estuary in the nation.

Influencing Factors include Load (nitrogen ratio) and Susceptibility. These factors are assessed as 'Highly influenced', 'Moderately influenced', or 'Slightly influenced' and are compared in a matrix to arrive at an assessment for overall Influencing Factors.

Eutrophic Symptoms include two primary symptoms (indicators): (1) chlorophyll *a* and 2) macroalgal blooms, and three secondary symptoms (indicators): (1) dissolved oxygen, (2) submerged aquatic vegetation, and (3) nuisance/toxic blooms. Symptom expressions are determined for each symptom in each salinity zone (two salinity zones in the case of BB-LEH) resulting in a total of 15 calculations. The expression is based on a set of IF, AND, THEN, decision rules that incorporate the symptom level (e.g. concentration), spatial coverage, and frequency. The estuary-wide symptom expressions are then calculated for each symptom. First, each expression value is multiplied by the area of the salinity zone and divided by the entire area of the system to establish the weighted value. Then, the weighted expression values in the salinity zones are summed to calculate the estuary-wide symptom expression value. This process is repeated for all five eutrophic symptoms. The average of the primary symptoms is calculated

to represent the estuary-wide primary symptom value. The highest of the secondary symptom values is chosen to represent the estuary-wide secondary symptom expression value and rating. Bricker et al. (2007) chose the highest value because they felt an average might obscure the severity of a symptom if the other two have very low values. In the NEEA approach, the overall eutrophic condition is determined by using a matrix of the estuary-wide primary and secondary symptom values (determined as 'High', 'Moderate High', 'Moderate', 'Moderate Low', or 'Low') with thresholds between rating categories agreed upon by a scientific advisory committee and participants from the 1999 assessment.

Finally, the Future Outlook was determined as an attempt to identify whether conditions in an estuary will worsen, improve, or remain unchanged over the next 20 years. Expected future load (nitrogen input) and Susceptibility (flushing and dilution) are compared in a matrix. Population projections were used to determine expected future load, but these were acknowledged to be unpredictable.

We have modified the approach in three ways. First, this project divided the estuary into three segments (north, central, and south) rather than two zones, based on heterogeneity described by environmental gradients detailed in Component 1. Bricker et al. (2007) divided the estuary into two segments based solely on salinity zones. Second, this project used ~20 indicators rather than two primary and three secondary indicators (Figure 3 - 1). The indicators are organized together into six components: (1) Ecosystem Pressures, (2) Water Quality, (3) Light Availability, (4) Seagrass, (5) Harmful Algal Blooms, and (6) Benthic Invertebrates. Third, we employed a numeric scoring system from 0 (degraded condition) to 100 (excellent condition) rather than a qualitative (e.g. 'High', 'Moderate', 'Low', etc.) scoring system. Each modification is specified in detail in the approved QAPP for this project.

Despite some methodological improvements, the current project uses the core and basic methodological approach of NEEA by comparing observations to thresholds, dividing the estuary into segments, and involves a numeric scoring system. Note that in addition to the number of indicators involved, some of the differences between the NEEA methodology and the approach used in this study are due to the geographic scale and scope of analysis. The NEEA approach is intended for a national study, and thus the analysis for BB-LEH Estuary was somewhat simplified because the range of heterogeneity in one estuary is much less than that for all estuaries in the United States. Further, the availability of data across such a wide range of estuaries is quite different than that for one estuary. For a national study, commonly available data must be utilized and other types of data, though potentially important at a regional or local scale, may not be able to be analyzed at this larger scale.

GOALS AND GENERAL APPROACH FOR THE BARNEGAT BAY INDEX

OF EUTROPHICATION

An important goal of this project is to develop an index of eutrophication condition for the BB-LEH Estuary. Though the current determination of the ecological health of New Jersey's estuarine waters is based on dissolved oxygen measurements, it is also important to examine biotic indicators and a broader range of physicochemical indicators for effective ecosystem-based assessment and management. The establishment of an appropriate biotic index for BB-LEH will aid New Jersey in delineating environmental impacts. Such an index identifies the condition of and relationships between ecosystem pressures, ecosystem state, and biotic responses. Prior to this report, no validated biotic index existed to assess the estuarine waters of New Jersey, most notably with respect to eutrophication. A long-term goal, though, beyond the scope of this project, is to extend this type of ecosystem assessment of the BB-LEH system to all estuarine waters of New Jersey in order to protect biotic communities, recreational and commercial fisheries, water quality, and habitats. Therefore, this is a valuable research initiative that has far reaching implications for coastal environmental protection and human use in New Jersey and other coastal states.

The approach to developing an index of eutrophication condition involves considering ~ 20 indicators. Candidate indicators were selected at the outset of this project and are specified in the Quality Assurance Project Plan (QAPP). These indicators are organized together into six components: (1) Ecosystem Pressures, (2) Water Quality, (3) Light Availability, (4) Seagrass, (5) Harmful Algal Blooms, and (6) Benthic Invertebrates. An index is calculated for each of these six components, and the six resulting indices are integrated together to calculate the overall index of eutrophication condition.

1) Ecosystem Pressures
  Total Nitrogen Loading (kg TN yr^{-1} estuarine km^{-2})
  Total Phosphorus Loading (kg TP yr^{-1} estuarine km^{-2})
2) Water Quality
  Temperature (°C)
  Dissolved Oxygen (mg L^{-1})
  Total Nitrogen Concentration (μg L^{-1})
  Total Phosphorus Concentration (μg L^{-1})
3) Light Availability
  Total Suspended Solids (mg L^{-1})
  Chlorophyll *a* (μg L^{-1})
  Macroalgae areal cover (% cover)
  Epiphyte to seagrass ratio (g dry wt epiphytes per g dry wt seagrass)
  Secchi depth (m)
  Percent Light Reaching Seagrass Leaves (%)
4) Seagrass
  Aboveground Biomass (g m^{-2})
  Belowground Biomass (g m^{-2})

Area Cover (%)
Shoot Density (shoots m^{-2})
Blade Length (cm)
5) Harmful Algal Blooms
Aureococcus anophagefferens concentration (cells mL^{-1})
6) Benthic Invertebrates
Benthic Invertebrate Species Richness
Gleason's D value
EMAP index values
Hard Clam Abundance (clams m^{-2})

Data collection of these indicators often occurred at different times or in different locations. Therefore, to align the data for each indicator by aggregation, observations are lightly summarized as a measure of central tendency (i.e., mean or median) for each year and estuarine segment that data are available (see the section 'Available Data/Data Gaps' below).

An index for each of the six components is calculated by summing a Raw Score and Weighted Score, each of which contributes 50% to the component index score. Each observation of each indicator is compared to 'thresholds' to determine the 'raw score'. An indicator's thresholds can be considered to be values for that indicator that mark some type of change in other (response) variables. Thresholds are determined and defined through examination of: (a) the literature, (b) analysis of available data for BB-LEH, (c) Best Professional Judgment, and (d) some combination of a-c. Raw scores range from 0 (degraded condition) to 50 (excellent condition) and are evenly weighted between indicators within the component index. Thus, for example, the raw score for each of the four Water Quality indicators contributes 12.5% of the score for the Water Quality Index (25% * 50% = 12.5%).

Weighted Scores weight the raw scores by their variability. Principal component analysis is conducted on the raw scores to calculate a weighting for each indicator within each component based upon their eigenvectors (variability). The weighting is calculated as the square of the eigenvector for each variable. Weighted scores are then calculated by multiplying the raw score by the weighting. Thus, for example, the weighted score for any of the four Water Quality indicators contributes 0-50% of the score for the Water Quality Index (the weighting for each variable ranges 0-100%, * 50% = 0-50%).

Raw and weighted scores are summed to calculate a component index score for each of the six components. Thus, for example, each of the indicators in the Water Quality component contributes 12.5-62.5% of the Water Quality Index.

Indices for each of the six components are then averaged together to calculate the overall biotic index of eutrophication. Raw, weighted, and final scores for each component and the overall Index of Eutrophication condition are calculated for each segment of the estuary for each year (1989-2010), subject to data availability. Scores for the year 2011 are calculated

independently for validation.

Principal component analysis, and the comparison of the multivariate axes provide a flexible framework for objectively weighting multiple components and multiple variables within each component, especially when these variables are asynchronously available, either spatially or temporally. This technique – though tangential to the main project objectives – is an important contribution to BB-LEH, and ecosystem health assessment.

AVAILABLE DATA / DATA GAPS

Data included in the index has been assembled from a variety of sources and is available (and unavailable) asynchronously over time (years) and space (estuary segment). Data available for inclusion are shown in Figure 3 - *2*. Grid cells in black indicate data are available for all three segments (north, central, and south). Cells in teal, with 'C, S' indicate data are available for the central and south segments. Cells in red, with 'N' indicate data are available for the north segment. Cells in brown with '??' indicate data are available that year, but spatial location is unknown. Cells in white indicate no data are available that year. Note that applicability of the index to any given segment depends in part on availability of data within that segment.

Data availability is as follows. 2011 data is kept separate for validation purposes (see Component 4).

Ecosystem Pressures: Total Loading for total nitrogen and total phosphorus is available from 1989-2011 for all three segments. These data are the outputs of the USGS modeling efforts described in Component 1 of this report.

Water Quality: Data are available for all three segments when available. None of the water quality data are available during 1992. Temperature, dissolved oxygen, and total nitrogen concentrations are available from 1989-1991 and 1993-2011. Total phosphorus concentrations are available from 1999-2011. These data were obtained from the New Jersey Department of Environmental Protection, Bureau of Marine Water Monitoring, courtesy Robert Schuster and are available in summary form at (http://www.nj.gov/dep/bmw/) .

Light Availability: Chlorophyll *a*, total suspended solids, Secchi depth, the ratio of epiphyte biomass to seagrass biomass, and the percent light reaching seagrass leaves are available in all segments. Macroalgae percent cover is only available in the central and south segments, except for 2011, when it is available in all three segments. Chlorophyll *a* and total suspended solids are available for 1997-2011. Secchi depth is available from 1989-1991 and 1993-2011. Macroalgae percent cover is available from 2004-2006 and 2008-2011. The ratio of epiphyte to seagrass biomass is available as measurements from 2009-2011 and is estimated backwards to 1997. Percent light available to seagrass leaves is estimated from 1997-2011. Equations for estimating percent light available to seagrass leaves are provided in Appendix 3 -

1. Chlorophyll *a*, total suspended solids, and Secchi depth were obtained from the New Jersey Department of Environmental Protection, Bureau of Marine Water Monitoring, courtesy Robert Schuster and are available in summary form at (http://www.nj.gov/dep/bmw/). Macroalgae percent cover was obtained as part of Component 2 of this project and for previous years from Michael J. Kennish, Institute of Marine and Coastal Sciences, Rutgers University. The ratio of epiphyte to seagrass biomass and percent light reaching seagrass leaves was calculated for this report.

Seagrass response: *Zostera marina* is present in the southern two thirds of the estuary, corresponding to the central and south segments. *Ruppia maritima* is present in the northern third of the estuary, corresponding to the north segment. All seagrass variables (aboveground biomass, belowground biomass, shoot density, percent cover, and blade length) are available from 2004-2006 and 2008-2011. *Ruppia* blade lengths are not available due to its physiology. Seagrass data were obtained as part of Component 2 of this project and for previous years from Michael J. Kennish, Institute of Marine and Coastal Sciences, Rutgers University.

Harmful algal bloom concentration data are available during 1995, 1999-2002, 2005, and 2010, but its spatial extents are variously available and so assessments will only be conducted for the entire estuary. Data were obtained from reported literature values.

Benthic invertebrate data are available during 2001 from the REMAP data for all three segments (Table 3 - 1, Figure 3 - 3). These data were made available from Darvene Adams, U.S. Environmental Protection Agency, Edison, New Jersey.

Additionally, several other datasets were evaluated for suitability for inclusion in calculations of the Index of Eutrophication. Examples include NCA data (2000-2006), residence time, hydrodynamic modeling, GIS layers of seagrass coverage, counts of jellyfish, and several others. Examples of qualitative and quantitative criteria for inclusion are the number of records, location and span of dates of data collection, and ability to describe and detect heterogeneity between segments and years. Examples of statistical procedures that have been used to evaluate datasets have included (but have not been limited to) ANOVAs between segments, PCA, correlation with salinity/habitat, assessment of data availability. These qualitative and quantitative evaluations indicated that the datasets mentioned above did not meet criteria for BB-LEH and thus could not be included in index calculations. For more information regarding the analyses conducted and the conclusions drawn regarding the evaluation of datasets for potential inclusion in calculations of the Index of Eutrophication, see Appendix 3 - 2).

DATASET ASSEMBLY

All raw datasets are compiled and stored in a folder on a server housed and accessible through Rutgers CRSSA (Center for Remote Sensing and Spatial Analysis). All datasets have been validated for completeness and content. All data were collected and reported strictly

according to QAPP protocols and expressed in appropriate units and formats. In cases of data collection for this project (e.g. seagrass and associated indicators), Quality Control of the data were conducted by validation against logbooks.

The database was assembled and imported from multiple files into SAS for data analysis across dataset type. The SAS code for the database assembly is available in Appendix 3 - 3. Creating the SAS code involved ensuring that datasets were inter-operable (i.e., variable names all spelled exactly the same, same units were used, values of 0 were appropriately distinguished from those that were absent, etc.). This is important to distinguish that values of zero are included in calculating means and other statistics, while absent data are not. Absent data does not necessarily indicate an error in either fieldwork or data management because some variables may not have been measured at all stations in all years (see figure above). Assembly of multiple files into one database enables the establishment of relationships between the different dataset tables among variables of interest. Using SAS to generate a complete database makes it dynamic and versatile, enabling multiple queries and calculations of a variety of types. It is important to determine which statistical relationships can be explored between datasets spatially and/or temporally.

Data collection for the various indicators often occurred at different times or in different locations. Therefore, for the purposes of the index analysis, it is necessary to align the data to common spatial and temporal units. This was done through aggregating and summarizing data for each indicator with a measure of central tendency (i.e., mean or median) for each year and estuarine segment that data are available. The complete, lightly summarized dataset (means and medians) used for the index analysis is included in Appendix 3 - 4.

DETERMINING THRESHOLDS: RESCALING DATA

Observations of indicators are lightly summarized (by central tendency for Year and Segment) and rescaled a unitless 'raw score' for each indicator according to an equation for that indicator. The equation describes the relationship between a set of specific threshold values. Thresholds are defined values. They are not a mean and have no associated error. An indicator's threshold values can be considered to be values for that indicator that mark some type of change in other (response) variables.

The equation used to rescale observations describes the relationship between the thresholds set at defined intervals of the indicator's score. Since some equations are exponential or logarithmic, these intervals are not always equal. The equations are used to calculate raw score by inputting observations as x values, and calculated y values are the raw scores. Rescaling equations are shown in Table 3 - 2.

Rescaling observations into scores accomplishes several tasks. First, it enables integration of multiple variables by bringing them into a common, unitless dimension. Second, it homogenizes the variances, thereby not making one variable more dominant than another simply

because of the range of its scale (e.g., ~0 to 30 for temperature but 0 to 200,000 for concentration of harmful algal cells). Rescaling was completed on all variables onto the same dimension with the same variance. Raw scores all range from 0 (bad) to 50 (excellent). Weighted scores also range from 0 (bad) to 50 (excellent). The sum of the raw score and the weighted score equals the index score for each of the six components, and thus index scores range from 0 (bad) to 100 (excellent). Weighting, weighted scores, and Index scores are discussed below.

Thresholds are determined and defined through examination of: (a) the literature, (b) analysis of available data for BB-LEH, (c) Best Professional Judgment, and (d) some combination of a-c, in that order of priority. Generally, if previously established thresholds for a given indicator have not been explicitly reported in the literature for estuarine coastal lagoons, the relationships between indicators or variables were examined either in the literature or data analysis. Thresholds were set at values of indicators that indicated a change in response values – such as changes in the slope or abrupt breaks in response indicators. Best Professional Judgment is reserved only for indicators where previous thresholds are not established in the literature and data analysis yielded limited insight. In this section, we describe in detail the process of selecting thresholds for each indicator, the sources and methods considered, and the thresholds and equations used for calculating indices for each of the six components that comprise the overall Index of Eutrophication.

Ecosystem Pressures

The Ecosystem Pressures component consisted of total loading (baseflow + runoff) for total nitrogen (kg TN yr^{-1} estuarine km^{-2}) and total loading (baseflow + runoff) for total phosphorus (kg TP yr^{-1} estuarine km^{-2}). However, additional variables output from the model results of Component 1 of this study were considered, but ultimately not, included in the calculations of the Ecosystem Pressures Index, namely, Total Yield for total nitrogen (kg TN ha-1 yr^{-1}) and total phosphorus (kg TP ha-1 yr^{-1}), as well as Flow-Weighted Average Total Concentration for total nitrogen (mg L^{-1}) and total phosphorus (mg L^{-1}). Total yield strongly covaried with total loading, for both total nitrogen and total phosphorus (Figure 3 - 4), as indicated by principal component analysis. This high level of covariation is due to the fact that the calculations of total loading and total yield are proportional to each other. Thus, while they provide different pieces of information in and of themselves, inclusion of both these indicators is redundant for the purposes of an index of eutrophication. Flow-weighted average total concentration did not correlate with total loading or total yield for either total nitrogen or total phosphorus (Figure 3 - 4). However, flow-weighted average total concentration for total nitrogen did not elicit a response in light indicators (Figure 3 - 5) nor seagrass indicators (Figure 3 - 6). Similarly, flow-weighted average total concentration for total phosphorus did not elicit responses in light indicators (Figure 3 - 7) nor seagrass indicators (Figure 3 - 8). Concentrations in the watershed are irrelevant to estuarine indicators because concentrations account for volume, which is different between the watershed and the estuary. Rather, estuarine response is most strongly connected with the amount of mass of nutrients that enter the estuary from the

watershed.

Thresholds for total nitrogen and phosphorus loading were determined by examining biotic responses to nutrient loading reported in the literature, and by data analysis of the nutrient loading modeling output from PLOAD and its relationship to ecosystem state and biotic response. First, we examined relationships between nutrient loading and estuarine responses in the literature. We provide only a summary here. For more information, see for example, Burkholder et al. 2007. As nutrient loading increases, seagrass biomass and productivity decline exponentially (Tomasko et al. 1996, Figure 3 - 9), as does areal coverage (Short and Burdick 1996, Figure 3 - 10 and Valiela et al. 2000, Figure 3 - 11). Seagrass shoot density similarly declines (Deegan et al. 2002, Figure 3 - 11). Seagrass declines are mediated by linear increases in estuarine total nitrogen concentrations, as has been found in Maryland's coastal bays (Boynton et al. 1996, Figure 3 - 12) and in BB-LEH (Kennish and Fertig 2012, Figure 3 - 13). In looking for thresholds among these relationships, we have looked for values of nutrient loadings that mark a change in rate of decline of seagrass responses. However, we have also looked for values that mark the start of declines (regardless of rate), and values above which it appears that nitrogen loading is no longer a dominant factor in the change of the biotic response.

Similarly, we examined the relationships between seagrass responses and nutrient loadings observed in BB-LEH compiled for this project. This is particularly important to calibrate the thresholds to be relevant for BB-LEH. We examined total nitrogen loading impacts on water quality indicators (Figure 3 - 14), light indicators (Figure 3 - 15), and seagrass indicators (Figure 3 - 16). Additional potential thresholds for total nitrogen loading were identified from changes in response indicators with changes in loading.

Ecosystem State: Water Quality

Temperature, dissolved oxygen, total nitrogen concentrations, and total phosphorus concentrations were all determined to be important indicators of water quality through principle component analysis. While temperature and total phosphorus were positively correlated, these indicators are different ecologically, and total phosphorus and total nitrogen were not correlated (Figure 3 - 17). Thus, they were determined to provide different pieces of information.

Water quality thresholds were also defined by examining the literature and through analysis of data assembled in this project. Specifically, we looked for optimal temperatures for seagrass growth and photosynthesis, minimum oxygen concentrations required physiologically for a variety of fish, shellfish, and invertebrate species, and nutrient concentrations that spur phytoplankton and macroalgal growth (Table 3 - 3). Kemp et al. (2004) list statistically derived concentrations of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) beyond which submerged aquatic vegetation is not present at a variety of salinity regimes (Table 3 - 3). A rough guideline has been one for Chincoteague Bay, which is a shallow, well-mixed coastal lagoon ecosystem, similar to BB-LEH. Wazniak et al. (2007) summarized pertinent thresholds regarding dissolved oxygen (Table 3 - 4), and for total nitrogen, total phosphorus, and

chlorophyll *a* (Table 3 - 5) for Maryland's coastal bays. Lee et al. (2007) reported optimal temperatures for growth and photosynthesis of seagrass (Table 3 - 6). For BB-LEH, dissolved oxygen thresholds were defined relative to the New Jersey standard of impairment, which is established at 4 mg L^{-1}.

Ecosystem State: Light Availability

Light availability is critical to maintain at high levels for shallow coastal lagoon ecosystems in order to maintain healthy dominance of benthic communities (Figure 3 - 18). Indeed, Burkholder et al. (2001) found that light reduction had a greater negative effect on seagrass shoot production than did increased nitrogen availability (Figure 3 - 19). Light availability thresholds are determined from the literature associated with physiological requirements of seagrass (Dennison 1993, Figure 3 - 20) and associated light attenuation by various factors such as plankton (chlorophyll *a*), total suspended solids, macroalgae (Kennish et al. 2011, Table 3 - 7), and epiphytic cover (Brush and Nixon, 2002; Figure 3 - 21, Figure 3 - 22), as well as measures of water clarity such as Secchi depth and the percent of surface irradiance available to seagrass leaves. Light availability (% of light available to seagrass leaves, 'PLL') is important and a potentially better measurement than Secchi depth because light often penetrates to the bottom of BB-LEH such that Secchi disks can be seen at the bottom, rendering Secchi depth readings inaccurate while also not providing a good measurement of how much light is actually available. PLL is calculated according to equations derived from empirical observations described by Kemp et al. 2004 shown in Appendix 3 - 1. Additional analysis on available data indicates that seagrass indicators responded negatively to increases in chlorophyll a (Figure 3 - 23) and total suspended solids (Figure 3 - 24).

Biotic Response: Seagrass

Thresholds for seagrass response were defined through data analysis with this project. Because few extensive data exist on seagrass in BB-LEH prior to 2004, it is difficult to establish stable reference conditions for this estuary. As discussed in Component 2 of this report, eelgrass biomass has been in general decline since monitoring commenced in 2004. Data were analyzed to identify if changes in rates of decline were evident with respect to total nitrogen loading (Figure 3 - 16), to chlorophyll *a* (Figure 3 - 23), and total suspended solids (Figure 3 - 24). However, declines had begun prior to monitoring and so assessments were adjusted given the uncertainty associated with identifying 'reference' conditions of seagrass in BB-LEH.

Biotic Response: Harmful Algal Blooms

An index of harmful algal blooms has previously been developed and is available in the literature (Gastrich and Wazniak, 2002; Figure 3 - 25). This was developed for coastal lagoon ecosystems, and thus thresholds from this index were utilized directly to derive the rescaling equation.

Biotic Response: Benthic Invertebrates

Thresholds for this component of the Index of Eutrophication are considered with respect to the REMAP assessment. They will be applied to the 2001 REMAP data. Many benthic invertebrate indices have previously been developed (see, for example, Weisberg et al., 1997, Van Dolah et al., 1999, Hale and Heltshe, 2008). Generally, they determine ideal or goal reference conditions, find locations that meet those conditions, and examine the benthic invertebrate community there with a variety of taxonomic and statistical tools. Conditions may include watershed characteristics, water quality (e.g. dissolved oxygen), contaminant concentrations, sediment composition, and bioassay survival rates. Such indices compare measurements at a new set of sites to measurements made at reference sites and test for statistically significant differences. These types of benthic invertebrate indices provide a binary response – i.e., Are unknown sites different or the same as reference conditions? Often they rely on community composition or measures of species diversity (e.g., Shannon-Weiner H or Gleason's D diversity indices) and assemble lists of species that are 'pollution indicative' or 'pollution sensitive'. Many species, however, are on both such lists.

In comparison, the Index of Eutrophication that is developed by this project compares observations at all sites directly to a spectrum of reference conditions that are termed 'thresholds'. Data are analyzed separately for each segment of the bay, because they have been determined to be heterogeneous habitats. The thresholds are biologically, physiologically, and ecologically relevant. The conditions are selected by: (a) literature review, (b) data analysis, and (c) Best Professional Judgment (in cases where a and/or b are unavailable). By comparing observations to a spectrum of reference conditions, the Index of Eutrophication provides a continuum of response, from "Healthy" to "Degraded". Validation of the methodology is conducted through comparison of multiple similar methods, and the response in 2011, as data from that year have been kept separate and out of analyses thus far.

SENSITIVITY ANALYSIS

The thresholds are defined, and the resulting equations are used to rescale observations into a unitless dimension common to all indicators within a component. These indicator scores are then equally weighted as an average to arrive at the Raw Score for the component. Additionally, a Weighted Score is calculated based on the variability (calculated as the square of the eigenvector) of the indicator, which is analyzed by principle component analysis. The Raw Score and the Weighted Score are then summed to arrive at an index for the component. Combining a direct comparison of indicators to thresholds along with the variability directly addresses the concern of identifying estuarine condition and its consistency. The utilization of principle component analysis to generate a weighting maintains the flexibility of adding additional components or indicators, provided rescaling equations could be established based on ecologically relevant threshold values. Since the weighted scores are based on the variability of

the indicators, an analysis of the sensitivity of the Weighted Score is necessary with respect to: (1) the length of time over which variability is measured, and (2) availability of individual indicators for any given year or segment.

This is particularly important because principle component analysis and other multivariate statistical tools cannot handle missing data. It is also important because, in general, indices compare a set of data to another set of old data, and the power of the index is increased with the size of the reference dataset. Data availability is therefore a critical factor for the overall index. Sufficient data are very limited for the harmful algal blooms and benthic invertebrate components. This substantially limits the ability to an index for these components for inclusion in the overall index for those years. Therefore, it is critical to understand effects on the assessment of the overall Index of Eutrophication.

Another concern was that "… a single index would be derived from an evaluation of the data collected over multiple years for multiple cause/response components. This index would then be used to evaluate the biotic health for any given year."

Scenario	Weighting	Assessment
1	Annual	Annual
2	Multiple Years	Annual
3	Multiple Years	Multiple Years

Put one way, the question is length of time over which the variability will be assessed. Put another way, it is really how frequently the indicator weightings will be updated. To address the question of the length of time to address data variability, we conducted a comparative analysis of Scenarios 1 and 2 to determine which may be more appropriate for use in BB-LEH. We anticipate that providing this sensitivity testing for the water quality component as an example addresses these concerns.

Data availability will inevitably play a role in determining weightings. When data are unavailable, variability is null, and thus weighting is considered 0%. Data availability, as discussed earlier, greatly varies. Yet, there has been significant effort on the part of federal, state, and local agencies, and academic institutions to generate increasing volumes of data. Given available data, however, Scenario 3 is not appropriate for the Index of Eutrophication because it does not meet the needs specified that the Index of Eutrophication "be used to evaluate the biotic health for any given year."

The Water Quality component was used as an example component to test sensitivity of the variability under Scenario 1 and Scenario 2. Water Quality was used because data were available for most years and for most variables (1989-2010 except 1993 for temperature, dissolved oxygen, and total nitrogen; 1999-2010 for total phosphorus). We can also therefore use the Water Quality component to examine the sensitivity of a component Index to the inclusion or

omission of a particular indicator (in this case total phosphorus), which we discuss below.

Note that conclusions from the tests comparing annual weighting to multi-year weighting can only be drawn regarding the sensitivity analysis. These sensitivity analyses were conducted using preliminary thresholds and rescaling equations and are therefore weighted scores that are not considered final results for the indicators or the Water Quality Index. No conclusions regarding an assessment of water quality can be made from the figures associated with this analysis. Analyses and conclusions regarding sensitivity analyses remain valid.

To assess sensitivity under Scenario 1, eigenvectors and weightings are calculated for each metric for each year. For Scenario 2, eigenvectors and weightings are calculated in two sets: 1989-1998 and 1999-2010. These sets of years were determined by availability of total phosphorus data. Both scenarios utilize PCA and give higher weighting for high variability, and lower weighting for low variability. Both address data gaps since unavailable data are considered to have variability = 0, and thus weighted at 0. For example, no eigenvectors or weighting can be calculated for either Scenario 1 or Scenario 2 during 1992 because no data were available that year. Effectively, the weighting for all metrics of water quality in 1992 is 0. (Following from that, the Water Quality index will have a weighting of 0 in 1992, when integrated into the overall Index of Eutrophication.)

It is important to note that under both Scenario 1 and Scenario 2, indicators receive multiple weightings over the course of the entire study period (1989-2010). For example, under Scenario 1, the weighting for total phosphorus was calculated to be 0% in 1989, 0% in 1990…, 2% in 1999, 85% in 2000… and so on (Table 3 - 8). Meanwhile under Scenario 2, total phosphorus was calculated to have two different weightings – 0% for 1989-1998 (because total phosphorus data were unavailable and thus had no variability), and 87% for data 1999-2010 (Table 3 - 9).

Weighted scores for each water quality indicator under Scenario 1 and Scenario 2 are comparable for each year and segment (Figure 3 - 26). There is no qualitative or substantial difference between scores under either scenario. This is also the case for Weighted Scores for the Water Quality Index (Figure 3 - 27). Both capture similar high and low scores for metrics and the Water Quality Index overall.

However, the multi-year scenario was determined to be more appropriate for the following reasons. In general, indices compare a set of data to another set of old data, and the power of the index is increased with the size of the reference dataset. Because data for different components were collected at different times and different locations, a common time frame and area needed across all components had to be determined. The common time frame is a year, and the common area is the segment. To maximize the power of the lightly summarized datasets, more than one year is needed to be analyzed by the principle component analysis in order to yield more than three data points (one for each segment) for any given year.

A second set of sensitivity analyses was conducted to identify the impact of inclusion or omission of an individual indicator (total phosphorus) on a component index (Water Quality). Note that these analyses were conducted using the final indicator thresholds and rescaling equations. This analysis is done for 1999-2010 and cannot be conducted for 1989-1998 because total phosphorus data are not available for this set of years. Therefore, under the multi-year scenario (1999-2010) that includes total phosphorus, the weightings are: temperature 15%, dissolved oxygen 8%, total nitrogen 13%, and total phosphorus 65%. In comparison, if total phosphorus is omitted in this same multi-year scenario (1999-2010), the weightings are: temperature 34%, dissolved oxygen 21%, and total nitrogen 45%. If total phosphorus were omitted entirely from the Water Quality component, the multi-year scenario could extend throughout the entire length of the study period (1989-2010), and in this case, the weightings would be: temperature 61%, dissolved oxygen 29%, and total nitrogen 10%. Total phosphorus was determined to be important to include as a Water Quality indicator because principle component analysis indicated that it did not co-vary with total nitrogen (Figure 3 - 17), and it affects water quality and biotic response indicators differently than temperature does, in ecological terms, even though total phosphorus tended to correlate positively with temperature.

Another example of sensitivity analysis was the determination of including the macroalgae percent cover in the Light Availability index. This was in question because this indicator had the fewest number of years of data within this component. Principle component analysis was conducted on all years of data for scenarios that excluded and included macroalgae percent cover (Figure 3 - 28). Macroalgae percent cover was determined to be an important indicator to include because, when available, it did not co-vary with any of the other Light Availability indicators. Similarly, the five seagrass indicators were examined by principle component analysis to identify potential co-variation between indicators (Figure 3 - 29).

INDICATOR SCORES

Indicator scores for Watershed Pressures were fairly consistent over time and between indicators relative to each segment (Figure 3 - 30). Nevertheless scores were somewhat lower during 2003-2010 than previously. Total Nitrogen Loading and Total Phosphorus Loading scores were always highest in the central segment and much lower in the north segment compared to either the central or south segments. There was a general decline over time in Total Nitrogen Loading and Total Phosphorus Loading scores. Total Nitrogen Loading scores ranged from 40 to 51 in the south segment, 45 to 55 in the central segment, and 5 to 14 in the north segment. Total Phosphorus Loading ranged from 70 to 87 in the south segment, 75 to 92 in the central segment, and 7 to 23 in the north segment.

Indicator scores for Water Quality indicators were highly variable (Figure 3 - 31). Scores for total nitrogen and total phosphorus were generally lower than scores for either temperature or

dissolved oxygen. No segment typically had higher or lower scores than other segments for temperature or total phosphorus. Temperature scores ranged from 27 to 46 (central), 30 to 49 (north), and 23 to 50 (south). Dissolved oxygen scores ranged from 14 to 32 (central), 20 to 33 (north), and 5 to 40 (south). Total nitrogen scores, were generally lower in the north segment (3 to 24) than the other two segments (central: 9 to 33; south: 5 to 28). Total phosphorus scores ranged from 8 to 32 (central), 11 to 26 (north), and 7 to 33 (south).

Indicator scores for Light Availability include chlorophyll *a*, total suspended solids, epiphyte to seagrass ratio, macroalgae percent cover, Secchi depth, and percent surface light available to seagrass (Figure 3 - 32). During 2004-2006, chlorophyll *a* scores were lowest in the central segment and next lowest in the north segment and highest in the south segment. In other years, chlorophyll *a* scores were comparable between segments. In 2010, chlorophyll *a* scores were 36 in the central segment, 33 in the north segment, and 37 in the south segment. Chlorophyll *a* scores ranged from 7 (in 2005) to 49 (in 2007) in the central segment, from 22 (in 2005) to 48 (in 2008) in the north segment, and from 23 (in 1998) to 47 (in 2002, 2004, and 2006) in the north segment. Total suspended solid scores ranged from 1 (in 2007) to 50 (in 2009 and 2010) in the central segment, from 35 (in 2000) to 50 (in 2008, 2009, and 2010) in the north segment, and from 21 (in 2000) to 50 (in 1997, 2008, 2009, and 2010) in the south segment. Macroalgae percent cover scores ranged from 1 (in 2009 and 2010) to 50 (in 2008) in the central segment, and from 0 (in 2009) to 39 (in 2006) in the south segment. Epiphyte to seagrass ratio scores ranged from 1 (in 2007) to 50 (in 2009) in the central segment, from 21 (in 2002) to 50 (in 2009) in the north segment, and from 16 (in 2000) to 2009 (in 2008, 2009) in the north segment. In 2010, epiphyte-to-seagrass ratio scores were 49 in the central segment, 43 in the south segment, and 37 in the north segment. Secchi depth scores ranged from 2 (in 2006) to 38 (in 2003) in the central segment, from 1 (in 2008) to 43 (in 2009) in the north segment, and from 2 (2006) to 40 (in 2005) in the south segment. Percent surface light scores ranged from 0 (in 2007) to 49 (in 2009) in the central segment, from 7 (in 1998 and 2002) to 50 (in 2009) in the north segment, and from 5 (in 2005) to 50 (in 2008) in the south segment. In 2010, percent surface light scores were 15 in the north segment, 26 in the south segment, and 36 in the central segment.

Indicator scores for Seagrass Response include those for aboveground biomass, belowground biomass, shoot density, percent cover, and blade length (Figure 3 - 33). Percent cover scores were very slightly higher in the south segment than in the central segment, but all other indicators had equivalent or higher scores in the central segment than south segment. Aboveground biomass scores ranged from 1 (in 2006, 2009, and 2010) to 4 (2005) in the central segment, and from 1 (in 2006, 2009, and 2010) to 8 in the south segment. Belowground biomass scores ranged from 2 (in 2006, 2009, and 2010) to 5 (in 2005) in the central segment and from 1 (in 2010) to 5 (in 2004) in the south segment. Shoot density scores ranged from 5 (in 2006) to 10 (in 2005) in the central segment and from 4 (in 2004, 2006) to 8 (2009) in the south segment. Percent cover scores ranged from 14 (in 2010) to 23 (in 2005) in the central segment to 18 (in 2006) to 34 (in 2004) in the south segment. Blade length scores ranged from 5 (in 2006) to 13 (in

2005) in the central segment and from 3 (in 2006) to 18 (in 2004) in the south segment.

There is only one indicator included for the Harmful Algal Bloom component (cell concentration). Indicator scores for the Harmful Algal Bloom component are equivalent to the Raw Scores, Weighted Scores, and the final Harmful Algal Bloom Index for this component. The Harmful Algae Bloom Index is shown as discrete dots due to the limited data that are available (Figure 3 - 34). Since only one variable is included (cell concentration), this indicator is weighted at 100%. Since associated spatial data are unavailable, this index cannot be broken down by segment. Harmful Algae Bloom Index values are generally low (0 in 1995, 1999, 2000, 2001, and 2002).

RAW SCORES FOR COMPONENT INDICES

Watershed Pressure Indicator scores were averaged to arrive at the Watershed Pressure Index (Figure 3 - 35). The Pressure Index ranged from 60 (in 1996) to 73 (in 2002 and 1995) in the central segment, and from 55 (in 2006, 2009, and 2010) to 69 (in 1995) in the south segment. Meanwhile, the Pressure Index was much lower in the north segment, ranging from 6 (in 1996 and 2009) to 19 (in 1995).

Raw Scores for the Water Quality component were generally consistent between segments (Figure 3 - 36). In 2010, Raw Scores for Water Quality were 19 in the north segment, 20 in the central segment, and 21 in the south segment. Raw Scores for the Water Quality component ranged from 20 (in 1996) to 35 (in 2001) in the central segment, 19 (in 2010) to 31 (in 1995) in the north segment, and 17 (in 1989) to 38 (in 2005) in the south segment.

Raw Scores for Light Availability Index were lower in the central segment during 2005-2007, but in most other years there were little differences between segments. In 2010, Raw Scores for Light Availability were 32 in the south segment, 35 in the central segment, and 36 in the north segment. Raw Scores for the Light Availability component ranged from 13 (in 2006) to 36 (in 1998) in the central segment, from 24 (in 2002) to 47 (in 2009) in the north segment, and from 22 (in 1998) to 43 (in 1997) in the south segment (Figure 3 - 37).

Raw Scores for Seagrass Response were virtually the same in the central and south segments (Figure 3 - 38). In 2010, Raw Scores for the Seagrass Response component were 6 in the central segment and 7 in the south segment. Raw Scores for the Seagrass Response component ranged from 6 (in 2006) to 11 (in 2005) in the central segment and from 6 (in 2006) to 14 (in 2004) in the south segment.

The Harmful Algae Bloom Index is shown as discrete dots due to the limited data that are available (Figure 3 - 34). These are equivalent to the Weighted scores and final Harmful Algal Bloom Index for this component. Since only one indicator is included (cell concentration), this

indicator is weighted at 100%. Since associated spatial data are unavailable this index cannot be broken down by segment. Nevertheless, Harmful Algae Bloom Index values are generally low (0 in 1995, 1999, 2000, 2001, and 2002).

WEIGHTING INDICATORS INTO COMPONENTS

As discussed above, weightings were derived for sets of multiple years according to data availability to maximize the power of the index tool. Weightings for all indicators within each component and for the components within the overall Index of Eutrophication are listed in Table 3 - 10. Weightings for Watershed Pressures were applicable to 1989-2010 and Total Nitrogen Loading and Total Phosphorus Loading were equally weighted (50% each). As discussed above, weighting for Water Quality indicators are applicable to 1989-1999 and to 2000-2010. Weightings for 1989-1999 were: temperature 66%, dissolved oxygen 33%, total nitrogen 2% and total phosphorus 0%. Weightings for 2000-2010 were: temperature 15%, dissolved oxygen 8%, total nitrogen 13%, and total phosphorus 64%. Weightings for Light Availability indicators were applicable to 1998-2010 and were: chlorophyll a 2%, total suspended solids 32%, Secchi depth 4%, epiphyte to seagrass ratio 30%, macroalgae percent cover 0%, and percent surface light reaching seagrass 31%. Weightings for Seagrass Response indicators were applicable to 2004-2010 (excepting 2007, when there were no data available) and were: aboveground biomass 8%, belowground biomass 2%, shoot density 1%, percent cover 53%, and blade length 35%. Harmful algal bloom component had only one indicator, cell concentration, which was weighted 100% when data were available.

WEIGHTED SCORES FOR COMPONENT INDICES

Weighted scores for the Watershed Pressures are equivalent to the Raw Scores for this index because Total Nitrogen Loading and Total Phosphorus Loading are evenly weighted (Figure 3 - 35).

Weighted scores for the Water Quality component were very similar between segments (Figure 3 - 36). Weighted scores for the Water Quality component ranged from 15 (in 2004) to 39 (in 1995 and 1997) in the central segment. They ranged from 15 (in 2010) to 42 (in 1997) in the north segment. They ranged from 14 (in 2003) to 40 (in 1990) in the south segment.

Weighted scores for the Light Availability component fluctuated year-to-year, the greatest in the central segment and fluctuating least in the north segment (Figure 3 - 37). During 2005-2008, weighted scores for the central segment were much lower than the other two segments. Weighted scores for the Light Availability component ranged from 3 (in 2007) to 47 (in 2009) in the central segment, from 22 (in 2002) to 49 (in 2009) in the north segment, and from 17 (in 2000) to 48 (in 2008) in the south segment.

Weighted scores for Seagrass Response were virtually the same in the central and south segments (Figure 3 - 38). Weighted scores for the Seagrass Response component ranged from 10 (in 2010) to 17 (in 2005) in the central segment and from 11 (in 2006) to 25 (in 2004) in the south segment.

Weighted scores for the Harmful Algal Bloom component are equivalent to the Raw Scores and the final Harmful Algal Bloom Index for this component. The Harmful Algae Bloom Index is shown as discrete dots due to the limited data that are available (Figure 3 - 34). Since only one variable is included (cell concentration), this indicator is weighted at 100%. Since associated spatial data are unavailable, this index cannot be broken down by segment. Harmful Algae Bloom Index values are generally low (0 in 1995, 1999, 2000, 2001, and 2002).

COMPONENT INDICES AND THE OVERALL INDEX OF EUTROPHICATION

Indices for each component provide a numeric scoring assessment based on quantitative criteria expressed as the rescaling equations and combine comparisons of the data against those criteria as well as the associated variability. The results are indices that range from 0 (Highly Degraded) to 100 (Excellent). Descriptions of the numeric scores are:

Index Value	Descriptor
80-100	Excellent
60-80	Good
40-60	Moderate
20-40	Poor
0-20	Highly Degraded

Weightings for the components into the overall Index of Eutrophication are listed in Table 3 - 10. The overall Index of Eutrophication is comprised of the Water Quality Index (100% during 1989-1997, 50% during 1998-2003, and 33% during 2004-2010), the Light Availability Index (50% during 1998-2003 and 33% during 2004-2010), and the Seagrass Response Index (33% during 2004-2010). Watershed Pressures remain separated from the other indices in terms of the overall Index of Eutrophication to avoid conflation of independent and dependent variables.

Watershed Pressure indicator scores were averaged to arrive at the Watershed Pressure Index (Figure 3 - 35). The Watershed Pressure Index was Good in the central segment, Moderate to Good in the south segment, and Highly Degraded in the north segment. In 2010, the Watershed Pressure Index was 7 in the north segment, 60 in the central segment, and 55 in the

south segment. The Watershed Pressure Index ranged from 60 (in 1996) to 73 (in 2002 and 1995) in the central segment, and from 55 (in 2006, 2009, and 2010) to 69 (in 1995) in the south segment. Meanwhile, the Pressure Index was much lower in the north segment, ranging from 6 (in 1996 and 2009) to 19 (in 1995).

The Water Quality Index indicated that water quality was generally Moderate and occasionally Good, but there were essentially no differences between segments. Water quality condition in 2010 was Poor in all three segments: 37 in the south, 36 in the central, and 33 in the north segments. The Water Quality Index ranged from 36 (in 2010) to 70 (in 1995) in the central segment, from 33 (in 2010) to 72 (in 1997) in the north segment, and from 36 (in 2003) to 74 (in 2005) in the south segment (Figure 3 - 36).

Light Availability Index values indicated that light availability was Moderate to Excellent in the south and north segments but Highly Degraded to Moderate in the central segment (Figure 3 - 37). Light availability in the central segment fluctuated widely and rapidly, with its lowest score in 2007 and its highest score only two years later. In 2010 the Light Availability Index was 70 in the south segment, 71 in the north segment, and 78 in the central segment. The Light Availability Index ranged from 19 (in 2007) to 79 (in 2009) in the central segment, from 46 (in 2002) to 96 (in 2009) in the north segment, and from 41 (in 2000) to 87 (in 1997 and 2008) in the south segment.

The Seagrass Response Index indicated that seagrass condition is Highly Degraded to Poor. There was virtually no difference between the central and southern segments of the estuary. In 2010 the Seagrass Response Index was 17 in the central segment and 19 in the south segment. The Seagrass Response Index ranged from 17 (in 2006 and 2010) to 28 (in 2005) in the central segment and from 17 (in 2006) to 39 (in 2004) in the south segment (Figure 3 - 38).

The Harmful Algae Bloom Index is shown as discrete dots due to the limited data that are available (Figure 3 - 34). These are equivalent to the Raw and Weighted scores for this component. Since only one indicator is included (cell concentration), this indicator is weighted at 100%. Since associated spatial data are unavailable, this index cannot be broken down by segment. Nevertheless, Harmful Algae Bloom Index values are generally low (0 in 1995, 1999, 2000, 2001, and 2002). Low values for this component of the index are not surprising given that sampling for harmful algae has historically been conducted when algal blooms occur in BB-LEH, and the presence of harmful algae species is anticipated.

According to the overall Index of Eutrophication, in 2010 BB-LEH was in Poor condition (37) in the north segment, Moderate condition (48) in the central segment, and Moderate condition (45) in the south segment (Figure 3 - 39). Between 1989 and 2003, the central segment had similar or slightly higher Eutrophication Index values than did the south segment, but from 2004-2010, the south segment had slightly higher Eutrophication Index values. Values of the Index of Eutrophication were always the worst in the north segment. Overall the Index of

Eutrophication ranged from 37 (in 2006) to 56 (in 2002 and 2000) in the central segment, 14 (in 1991) to 50 (in 2009) in the north segment, and from 45 (in 2010) to 71 (in 1997) in the south segment.

VALIDATION

Data from 2011 has been stored as a separate dataset and not included in the methodological analysis for the biotic index calculations. Validation results of the data for each of the datasets are provided in Component 4 of this report.

LIMITATIONS OF THE APPROACH

No assessment technique is a perfect or ideal tool, and limitations and caveats of this technique are specified here. No assessment can be more accurate than the data it draws upon. As noted in previous sections, there are many critical data gaps in previous years for most of the indicators utilized in this index. While over time more data were collected for more indicators, the paucity of data in early years limits the holistic and comprehensive assessment, particularly prior to 2004. Additionally, there are spatial misalignments or gaps among the datasets (Figure 3 - *2*), because data collection for each dataset occurred at different locations, spatial scales, and with different sampling designs. These spatial and temporal misalignments of data result from the assembly of multiple disparate, previously independent datasets with various purposes and scopes.

For this project, available data and its limitations for many indicators must be qualified to appropriately consider the confidence of the data and the assessment, which arises from its analysis. In BB-LEH, Secchi depth must be considered a type of 'censored data' – a technical statistical term defined as data that have cutoff points due to some external factor resulting in a discrete endpoint on one end of the data distribution. In this case, data 'censorship' is due to the Secchi disk hitting the bottom, which thus places an external limit (i.e., water depth) to the upper end of the observations of Secchi depth. Given the same conditions in deeper water, the recordings (and their means) for Secchi depth may have been of greater magnitude.

Frequency of data collection must also be considered a limitation to the assembled database. Dissolved oxygen data are only available from quarterly in situ observations, which are not sufficient to capture natural daily fluctuations due to processes such as photosynthesis and respiration, and further introduce bias with the confounding of temperature and sunlight irradiance. Continuous monitoring (observations recorded at 15 minute intervals) would better characterize dissolved oxygen and temperature; however, such measurements are often only able to be made in shallow water along shorelines due to capacity for sonde deployments, and so such observations would need to be reconciled with observations at depth or in open water areas of the estuary.

The expansion of the number of datasets over time provides a wealth of data for more recent years, but somewhat biases comparisons of assessments to earlier years. Epiphytic data have been calculated based on empirical observations and statistical relationships with other available observations, and though there is very good agreement between validation datasets and the calculations, additional years of measurements would strengthen the confidence in these estimates. Macroalgae and seagrass data are not available prior to 2004, creating some uncertainty regarding 'reference' or 'pristine' conditions of seagrass in BB-LEH, though these can be estimated based on empirical relationships described in the literature for other similar types of coastal lagoon estuaries.

Natural heterogeneity, either spatially or temporally, among indicators also poses a challenge to overcome. For example, due to salinity limitations, *Zostera marina* dominates seagrass beds in the central and south segments, and *Ruppia maritima* dominates the seagrass beds in the north segment. Salinity intolerance of these two species affects their data distribution in the different segments of the estuary. There is a paucity of data on harmful algal bloom concentrations, with only a few years of data available and locations of observations not available, making a spatial assessment of brown tides and other harmful algal species difficult. Furthermore, monitoring for harmful algae is only conducted when general algal blooms are occurring or if brown tide species in particular are suspected to occur. This sampling method appropriately detects presence or absence, but biases continuous assessments towards degraded conditions.

Benthic invertebrate data are only available during 2001, and biomass data are completely absent from the dataset. Benthic invertebrate biomass data are required for calculating many types of benthic invertebrate indices of environmental condition.

Threshold determination for this project has been conducted according to review of pertinent literature on similar coastal lagoons and their biotic communities, analysis of existing and collected data, best professional judgment (to as limited extent as possible), and combinations of these methods. Thresholds and rescaling equations have been calibrated for BB-LEH as a coastal lagoon. However, while there may be applicability of these thresholds to other similar coastal lagoons in New Jersey or elsewhere (such as Great South Bay, NY, Chincoteague Bay, MD/VA, Hog Island Bay, VA, etc.), the thresholds established may be of limited utility for other New Jersey waters (e.g. Raritan Bay, NY/NJ Harbor, and Delaware Bay) that do not share important characteristics. BB-LEH is in part extremely susceptible to even small amounts of nutrient loading due to its enclosed geomorphology and slow water circulation and flushing time. In contrast, coastal waters along the Atlantic Coast, Raritan Bay, and NY/NJ Harbor, and Delaware Bay have much quicker and stronger circulation patterns and therefore respond to nutrient enrichment at different time scales. Additionally, while heavy metals, inorganic, and organic toxicants may be important considerations for ecological health in some New Jersey waters, they may be of lower priority for BB-LEH. Toxicological analysis of sediments and the water column are beyond the scope of this project and have not been included in the biotic index

of eutrophication or its component indices.

DISCUSSION

Despite the limitations of the data and scope of this project, the biotic index of eutrophication remains the most comprehensive and holistic assessment of BB-LEH conducted to date. In order to assess the ~20 indicators, the index integrates over 74,400 observations among 85 variables.

Indices for each component provide a numeric scoring assessment based on quantitative criteria expressed as the rescaling equations and combine comparisons of the data against those criteria as well as the associated variability. The results are indices that range from 0 (Highly Degraded) to 100 (Excellent). Descriptions of the numeric scores can be broken down as follows:

Index Value	Descriptor
80-100	Excellent
60-80	Good
40-60	Moderate
20-40	Poor
0-20	Highly Degraded

Because index scores are comprised of raw scores and weighted scores that integrate assessments of multiple indicators and their variability, interpretations of these scores describe the overall condition and consistency of the component. Therefore, for a score of 80 to 100 indicates that most, if not all, of the indicators were consistently in excellent condition. Conversely, a score of 0 to 20 indicates that most, if not all, of the indicators were consistently in dire condition. Intermediate scores, e.g., 40 to 60, may indicate that some indicators were in good to excellent condition while others were in poor to Highly Degraded condition, or it may indicate that all indicators were in moderate condition, or it may indicate an overall inconsistency or large change in condition over time. Utilizing a Report Card analogy can help to summarize and communicate these scores to a wide variety of audiences.

The detrimental impact of nutrient loading on the ecosystem health of BB-LEH is clearly shown in a comparison of the values of the overall index of Eutrophication vs. total nitrogen loading and total phosphorus loading (Figure 3 - 40). As nutrient loading increases, Eutrophication Condition plummets from 'Good' (a score of almost 70) to 'Poor' (a score below 40), and in some cases even to 'Highly Degraded'. The initial rapid response of the decline highlights how sensitive BB-LEH is to even small increases in nutrient loading, especially at lower levels of loading. The system responds differently after reaching a threshold of nutrient

loading. In excess of nutrient loading amounting to ~2,000 kg TN km^{-2} yr^{-1} or ~100 kg TP km^{-2} yr^{-1} the Eutrophication Index values no longer decline as rapidly and level off, though with a great amount of variability, ranging between 2 and 50 (Highly Degraded to Moderate condition). Therefore, in excess of ~2,000 kg TN km^{-2} yr^{-1} or ~100 kg TP km^{-2} yr^{-1} another factor or set of factors may explain the variability of the eutrophication condition. However, what remains clear is that throughout the entire system, nutrient loading – both total nitrogen loading and total phosphorus loading – clearly result in substantial degradation and eutrophication of BB-LEH.

The data also indicate that different portions of BB-LEH 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. This is reflected in the lower values of the Eutrophication Index for the north as compared to the central or south segments. The central and south segments are similar to each other and over 1989-2010.

The Eutrophication Index scores for the central and south segments indicate that nutrient loading has resulted in severe declines in condition. Based on the entire dataset, the best Eutrophication Index score ever observed (73, described as Good) was in the central segment in 1992. Yet by 2006, the Eutrophication Index value in the central segment was at its lowest (37, Poor) and subsequently only improved to Moderate condition (48) by 2010, which still represents an overall decline in condition by 34%. Eutrophication Index scores for the south segment have declined from a high of 71 (Good) in 1997 to a low of 45 (Moderate) in 2010, representing a 36% decline.

In contrast to the south and central segments, the overall eutrophication condition of the north segment, though the lowest of the three segments, has been modestly improving. Though scores declined sharply (to 37, Poor) in 2010, the highest score observed in the north (50, Moderate) occurred in 2009, which is 3.5 times its lowest score (14, Highly Degraded), which occurred in 1991.

The indicators most important to the overall Index of Eutrophication change over time. This occurs in part due to increasingly (though never fully) holistic data availability and associated change in weighting of each of the component indices within the Index of Eutrophication over time. To examine what factors most influence the Eutrophication Index scores, we recall that a Raw Score (equal weighting of each indicator) and a Weighted Score (weighting of indicators by their variability) comprise the Eutrophication Index. Therefore, data availability and condition consistency are quite relevant. From 1989-1997, no data are available

for light availability or seagrass indicators, and thus water quality index is used. During this time period, temperature is weighted 66%, and dissolved oxygen is weighted 33% for the Weighted Score. Therefore, scores for these two indicators comprise 45% and 28%, respectively, of the overall Eutrophication Index during this time period. During this time period, dissolved oxygen condition was generally Moderate in the north and central segments but Poor to Highly Degraded in the south segment. Temperature scores generally increased from Moderate to Excellent over the same time period. The scores of these two indicators therefore largely explain the overall Moderate condition of the estuary during 1989-1997. Note that confidence in this assessment is low as measurements for dissolved oxygen in the early years of monitoring are sparsely available, with only quarterly in situ observations, as discussed above.

During 1998-2003, both the score for the Water Quality Index and the Light Availability Index equally comprise the overall Index of Eutrophication. In turn, the Water Quality Index is largely influenced by temperature scores from 1998-1999 (66% for the weighted Water Quality score) and by total phosphorus scores from 2000-2003 (64% for the weighted Water Quality score). Temperature scores were Moderate to Excellent in 1998-1999, while total phosphorus scores slid from Moderate to Highly Degraded during 2000-2003. Meanwhile, the influential indicators for the Light Availability index during 1998-2003 were total suspended solids (32%), the ratio of epiphyte to seagrass biomass (30%), and the percent of surface light reaching seagrass (31%). During this time period, total suspended solids were in Moderate to Good condition, the epiphyte to seagrass biomass ratio was Poor to Moderate, and the percent of surface light reaching seagrass was Highly Degraded to Poor, declining in the north and south segments from 1998-2002. The combination of these influential factors led to the overall Moderate to Good conditions for the overall Eutrophication Index scores that declined during 1998-2003.

Between 2004 and 2010, the Index of Eutrophication was comprised of the Water Quality Index (33%), the Light Availability Index (33%), and the Seagrass Response Index (33%). As with the previous set of years, the most influential indicator to the Water Quality Index was total phosphorus (64% for the Weighted Score), and Weighted Scores for the Light Availability Index were influenced by total suspended solids (32%), the ratio of epiphyte to seagrass biomass (30%), and the percent of surface light reaching seagrass (31%). The Seagrass Response Index was heavily influenced by the percent cover (53%) and the blade length (35%), while the aboveground and belowground biomass cumulatively contributed only 10% to the Weighted Score. Except for the anomalous year of 2005, when total phosphorus scores were 32 and 33 (Good) in the central and south segments, total phosphorus scores were generally Poor and declined to Highly Degraded (10 for all three segments) over the course of 2004-2010. Total suspended solid scores steadily improved between 2004-2010 in the north, were variable but showed general improvement in the south over that time period, and dramatically but

temporarily declined in the central segment with Highly Degraded scores during 2006-2007. The dramatic degradation between 2004-2007 and subsequent improvement (2007-2009) in the central segment was also observed in scores for the ratio of epiphyte to seagrass biomass, and the percent of surface light available to seagrass. Both seagrass percent cover and seagrass blade length indicators declined over time from 2004-2010, but the condition of percent cover was somewhat better, declining from Moderate to Poor scores, while blade length declined from Poor to Highly Degraded scores. Combined, these six indicators were the most influential on the overall Index of Eutrophication scores. The dramatic, temporary, declines of light availability indicators during 2004-2007 are observable in the decline of the Eutrophication Index scores in the central segment during that time period. Concurrently, as influential light availability indicators were improving in the north, Eutrophication Index scores in the north improved.

COMPONENT 4: VALIDATION DATASET (2011) FOR EUTROPHICATION ASSESSMENT

INTRODUCTION

In situ surveys were conducted in all three estuarine segments in 2011 to examine the characteristics of *Ruppia maritima* and *Zostera marina* during the June-November survey period (Figure 1-7). Lathrop et al. (2006) showed conclusively that widgeon grass (*R. maritima*) is the overwhelmingly dominant seagrass species in the north segment of the estuary, while eelgrass is the predominant form in the central and south segments. Biotic monitoring of the north segment of the estuary is important to holistically assess eutrophication of the entire system. Data collected in the field surveys during 2011 followed the protocols of the SeagrassNet approach that were applied in the estuary during the 2004-2010 period. These protocols were followed to maintain consistency and data integration with previous seagrass surveys to generate a validation database.

MATERIALS AND METHODS

Sampling Design

Quadrat, core, and hand sampling was conducted over the June to November period in 2011. The same sampling protocols were followed in 2011 as in previous years, but the samples were collected bimonthly at 150 stations along 15 transects in three segments (north, central, and south) of the estuary (Figure 1-7) rather than at 120 stations along 12 transects (central and south segments only) as in previous survey years (Figure 1-8). The same physicochemical and biotic data were recorded as in previous survey years (see Components 1 and 2), resulting in more than 2500 abiotic and biotic measurements for the 2011 field survey period. In addition to the field survey, water quality data collected by the NJDEP in the north segment of the estuary during the 2011 were used as secondary data. Included in this database are chlorophyll *a*, dissolved oxygen, Secchi depth, ammonia, nitrite plus nitrate, total nitrogen, phosphate, and total phosphorus.

RESULTS

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). However, it decreased markedly (mean = 16.1ºC) during the October-November sampling period (Table 4-1). 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 4-1).

Mean dissolved oxygen (DO) 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 4-1).

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 4-1).

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 4-1).

Widgeon Grass (*Ruppia maritima*)

Ruppia maritima was most abundant in the north segment of the estuary. 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 (Table 4-2).

Aboveground Biomass

Aboveground biomass of *R. 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 4-2).

The mean aboveground biomass of *R. 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 4-3). 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.

Belowground Biomass

Belowground biomass of *R. 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 4-2).

Shoot Density

The highest *R. 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 4.2).

Areal Cover

The areal cover of *R. 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 4-2).

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 4-3). 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 (*Z. marina*) varied both spatially and temporally in the estuary during 2011. 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 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 4-3).

Aboveground Biomass

Aboveground biomass of *Z. 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 4-2).

The mean aboveground biomass of *Z. 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 4-3).

Belowground Biomass

Belowground biomass of *Z. 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 4-2).

Belowground biomass of *Z. marina* in 2011 was extremely low in the north segment, where *R. 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 4-3). 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 *Z. 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 38.2 shoots m^{-2}, and it dropped to 0 during the remaining sampling periods. In the central segment, the mean shoot density was 250.4 shoots m^{-2} in June-July, 161.3 shoots m^{-2} in August-September, and 239.8 in October-November. In the south segment, the mean shoot density was 123.1 shoots m^{-2} in June-July, 212.2 shoots m^{-2} in August-September, and 208.0 in October-November (Table 4-3). These shoot densities are much lower than those reported for *Z. marina* in the estuary during 2010 (see Table 2-6).

Blade Length

The highest mean length of *Z. 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 4-3). Mean *Z. marina* blade length was also high during the October-November sampling period (31.1 cm) in the south segment 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 (Table 2-6).

Areal Cover

The mean percent cover of *Z. marina* during sampling periods in June-July, August-September, and October-November was 19.7%, 17.9%, and 16.1%, respectively (Table 4-2). 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 during both the August-September and October-November sampling periods (Table 4-3). 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 (Table 2-6).

Macroalgae

Areal Cover

The mean percent cover of macroalgae in 2011 ranged from 1 to 7.9% (Table 4-2). 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 (Table 2-1).

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 4-4). In addition, other biotic material also covered small areas of the estuarine floor ranging in mean values from 0 to 1.0% (Table 4-4).

Epiphytes

The mean percent cover of epiphytes on eelgrass leaves during all sampling periods in 2009 ranged from 19.2 to 38.3% for upper leaf surfaces and 18.4 to 38.3% for lower leaf surfaces (Table 2-5). In 2010, the mean percent cover of epiphytes on eelgrass was generally lower than in 2009, with the values ranging from 11.3 to 25.7% for upper leaf surfaces and 10.7

to 24.4% for lower leaf surfaces (Table 2-5). However, higher values of epiphyte percent cover on eelgrass leaves were found during the October-November sampling period in 2010 than in 2009, with the mean upper leaf and lower leaf percent cover values ranging from 20 to 21% in October-November 2010 compared to values ranging from 18.4 to 19.2% in October-November 2009 (Table 2-5).

Epiphyte biomass on eelgrass leaves in 2009 peaked during June-July (mean = 121.8 mg dry wt m^{-2}). In 2010, peak epiphyte biomass occurred during August-September (mean = 67.7 mg dry wt m^{-2}) (Table 2-5). The maximum biomass of epiphytes also occurred at the time of peak epiphyte areal cover on eelgrass leaves.

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 4-5).

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 4-5).

CONCLUSIONS

The degraded condition of *Z. marina* in the BB-LEH Estuary has continued through 2011, validating the progressive estuary-wide decline of this critically important seagrass species since 2004 (see Component 2). Aboveground biomass values for eelgrass in 2011 were nearly equal to the highly reduced aboveground biomass values recorded in 2009 and 2010. For example, the mean aboveground biomass measurements recorded in 2011 during the June-July, August-September, and October-November sampling periods were 7.2, 9.4, and 17.4 g dry wt m^{-2}, respectively (Table 4-2). By comparison, the mean aboveground biomass measurements of eelgrass in 2009 during these three sampling periods were 15.1, 8.0, and 3.0 g dry wt m^{-2}, respectively, and in 2010 they were 13.3, 6.6, and 2.7 g dry wt m^{-2}, respectively. All of these values are consistently low from year to year.

The condition of the belowground biomass of the eelgrass beds has worsened. For instance, the mean belowground biomass recorded for eelgrass in the estuary during the three sampling periods in 2011 (21.4, 15.7, and 15.5 g dry wt m^{-2}) is the lowest on record (Table 4-2),

including the decimated years of 2009 and 2010 (see Table 2-6). Therefore, the aboveground and belowground biomass of eelgrass in BB-LEH taken together for 2011 is highly problematic and reflective of an impacted coastal lagoon, even when considering only eelgrass in the central and south segments. This observation is also consistent with the declining trend of eelgrass in the estuary documented over the 2004-2010 period (see Component 2).

In concert with the degraded biomass condition, the shoot density of eelgrass was markedly reduced in 2011 relative to previous years of sampling from 2004 to 2010. For example, the mean shoot density values of eelgrass recorded in 2011 during the June-July, August-September, and October-November sampling periods were 157.0, 149.4, and 179.1 shoots m^{-2}, respectively (Table 4-2). Only in the severely impacted year of 2006 was a similar set of shoot density values observed, amounting to 170.3, 156.0, and 163.5 shoots m^{-2} during the June-July, August-September, and October-November sampling periods, respectively, although low values were also noted in August-September and October-November sampling periods in 2004. For all other survey years, shoot density values were much higher than those recorded during 2011, even removing the lower north segment measurements from the analysis (see Table 2-6).

The areal cover of *Z. marina* was similar to that recorded in 2010 and generally less than that recorded during the other survey years from 2004 to 2009, although somewhat higher measurements were observed when removing the shoot density values recorded in the north segment. The mean areal cover of *Z. marina* in the estuary during the June-July, August-September, and October-November sampling periods amounted to 19.7, 17.9, and 16.1%, respectively (Table 4-2). Similar to 2010, areal cover of *Z. marina* progressively decreased across the sampling periods.

The mean blade length of *Z. marina* recorded in 2011 was more consistent with that documented during previous survey years from 2004-2010. Mean blade lengths of eelgrass in 2011 amounted to 25.3, 29.1, and 31.5 cm for the June-July, August-September, and October-November sampling periods, respectively (Table 4-2).

The condition of *R. maritima* in the estuary also does not appear to be strong, although only one year of data (2011) has been collected on widgeon grass in the north segment since 2004, and hence there is no way to validate its condition in the north segment without additional years of sampling there. Previous years of sampling in the central and south segments, however, show conclusively that widgeon grass is depauparate in these areas, with mean aboveground or belowground values ≤ 1.6 g dry wt m^{-2} during all sampling periods in 2005 and 2010, when the only widgeon grass biomass values were recorded (Table 2-8). Somewhat higher aboveground and belowground biomass values of widgeon grass were recorded in 2011, especially in the more favorable environment of the north segment (Table 4-3). However, no widgeon grass samples were found in the south segment during 2011. These data demonstrate that widgeon grass dominates seagrass beds only in the north segment, while eelgrass dominates the beds in all other

areas. In addition, the north segment does not appear to be a major habitat for either species.

Since *R. maritima* propagates by runners, which may be either over or just under the sediment surface, it does not have blades in the form of *Z. marina*, but rather stem-like sections that may serve double-duty as lateral runners. The blades are technically just the tufts at the ends of these sections. While *Z. marina* canopy height can be viewed as a function of blade length, it is not accurate to measure blade length as a proxy for canopy height in *R. maritima*.

Macroalgae areal cover in 2011 was similar to that in 2010 and somewhat less than that in previous years from 2004 to 2009 (Table 2-1). The highest mean areal cover of macroalgae was reported in 2004 and 2008, when more than 20% cover was reported during at least one sampling period. The highest mean macroalgal areal cover during 2011 (7.9%) occurred during the June-July sampling period (Table 4-2).

The mean percent cover of epiphytes on eelgrass leaves during all sampling periods in 2009 ranged from 19.2 to 38.3% for upper leaf surfaces and 18.4 to 38.3% for lower leaf surfaces. In 2010, the mean percent cover of epiphytes on eelgrass was generally lower than in 2009, with the values ranging from 11.3 to 25.7% for upper leaf surfaces and 10.7 to 24.4% for lower leaf surfaces. 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 (Table 4-5). Much lower epiphyte percent cover was recorded on eelgrass leaves during the other sampling periods.

COMPONENT 5: SYNTHESIS AND MANAGEMENT RECOMMENDATIONS

INTRODUCTION

New Jersey coastal lagoons are subject to multiple anthropogenic stressors associated with increasing human population growth, land-use changes, and other alteration of coastal watershed areas. Eutrophication, left unabated, will seriously impact the structure and function as well as the overall environmental quality of these complex coastal systems and can seriously impair human uses of estuarine resources. Insidious, progressive eutrophication may even lead to the permanent alteration of estuarine biotic communities and habitats.

To better understand the ecosystem state of BB-LEH, it is instructive to review key characteristics that render the estuary susceptible to environmental impacts. First, both nonpoint and point source stressors affect the ecological integrity of the estuary. Of the various environmental problems coupled to these stressors, eutrophication (nutrient enrichment and associated cascading ecological impacts) poses the most serious threat because it creates the potential for a systemic, ecosystem-wide decline, affecting the long-term health and function of the entire system from Bay Head to Tuckerton, and impacting biotic resources, essential habitat (e.g., seagrass and shellfish beds), and human uses throughout (Figure 5-1). Some of these changes have become more evident in the estuary over the past decade.

This project examines the cause-and-effect relationships associated with lagoonal nutrient enrichment. One outcome is the need to consider nutrient loading criteria in support of nutrient management planning. A part of this effort may be directed toward the establishment of a nitrogen standard for the estuary that will have value in mitigating eutrophic impacts in the estuary.

MANAGEMENT APPLICATION

Water quality standards are established for estuarine waters by coastal states under authority of the federal Clean Water Act (CWA). To protect designated uses of a water body (primarily recreational and commercial uses, and aquatic life), water quality criteria are developed based on sound science. In the case of nutrient criteria, quantitative nutrient pollution values are set so that the biotic integrity and designated uses of a water body can be maintained.

No nutrient criteria have been established for the BB-LEH Estuary. One approach is to

establish a nutrient standard based on cause-and-effect relationships, notably making accurate measurements of variables representative of nutrient loading (causal variables) in the watershed and those based on biotic response (response variables) in the water body. While causal variables, such as nutrient loading, are vital in assessing nutrient impacts on a coastal lagoon, biotic and biogeochemical processes can significantly modify or transform them; thus, their dynamic nature can be delimiting. In addition, temporal and spatial scales play an important role in defining the relationships between causal and response variables. In the case of response variables, a suite of key variables which permit integrated assessment of biotic communities and habitats will provide more accurate data on ecosystem condition and nutrient impacts than can a single response variable. Integrated response variables may not only include biotic variables, such as phytoplankton, macroalgae, and seagrass, but also physicochemical variables, such as dissolved oxygen and total suspended solids. The complete array of causal and response variables used in this project are provided in Components 2 and 3 of this report.

DRIVERS OF CHANGE

BB-LEH, similar to other coastal lagoons, is particularly susceptible to nutrient enrichment because it is shallow with a high surface area to volume ratio. It also lies in close proximity to a highly populated and altered coastal watershed. In addition, the water residence time is protracted, promoting pollutant retention in the basin. Figure 5-2 shows total nitrogen concentrations in the estuary from 1989-2010.

The detrimental effects of eutrophication in BB-LEH are exacerbated by other factors. For example, point-source effects of the Oyster Creek Nuclear Generating Station (i.e., thermal discharges, impingement, and entrainment) increase mortality of estuarine and marine organisms inhabiting the estuary. Freshwater withdrawals in Ocean County have averaged more than 75 million gallons per day, with most of this (>70%) attributed to public use (USGS data, West Trenton, New Jersey). Centralized wastewater treatment facilities in the county discharge an average of more than 50 million gallons per day of treated wastewater to the Atlantic Ocean, and the volume of these discharges is increasing with increasing population growth (NJDEP, Trenton, New Jersey, NJPDES Municipal Flow Data). Other human factors such as bulkheading, dredging, ditching, and lagoon construction have altered hydrologic, physical, and chemical conditions in some areas of the estuary. Human activities in upland watershed areas, notably deforestation and infrastructure development, partition and disrupt habitats while also degrading water quality and altering biotic communities (Zampella, 1994; Zampella and Laidig, 1997; Dow and Zampella, 2000; Bunnell et al., 2003; Zampella et al., 2006). Soil disruption and land surface alteration increase impervious cover as well as turbidity and siltation levels in tributaries of the estuary, which can create benthic shading problems in the bays.

Human activities in the BB-LEH Watershed are the primary drivers of land use-land

cover change that require effective land-use planning and management decisions for remediation. With population growth in the watershed expected to increase from ~575,000 year-round residents (>1.2 million people during the summer tourist season) to ~850,000 people 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 increase, leading to greater input of nutrients and other pollutants to the estuary. With ongoing population growth and development, watershed habitats will continue to be partitioned and altered. The challenges posed by these changes will require more effective management measures and improved engineering controls to mitigate future impacts on the estuary.

Land alteration continues even in sensitive habitats. For example, between 1995 and 2006, riparian areas lost 625 ac of forest land cover and 373 ac of wetland land cover, with most converted to urban land cover which increased by 1,290 ac over that time period in riparian areas. By 2006, 4,205 ac of agricultural land area existed in the watershed, down by 1,097 ac in 1995. Urban land area, in turn, increased from 87,757 ac to 103,746 ac (+15,989 ac) between 1995 and 2006. Finally, 14,248 ac of forest were lost over this 11-year period (Data from the Center for Remote Sensing and Spatial Analysis, Rutgers University).

The amount of tidal marshes in the Barnegat Bay Watershed Management Area has decreased by 8% between 1995 and 2007. Based on a GIS analysis of the tidal marshes conducted by the Richard Stockton College Coastal Research Center, most of this wetland loss has occurred along the bay and tidal waterway shorelines. Additional loss of marsh habitat has taken place near areas of development in residential areas. Freshwater wetlands have also decreased in area, by ~5%, over the 12-year study period, with most of this loss ascribed to development in the watershed.

Urban land use in the BB-LEH Watershed has increased dramatically over the past four decades. 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 (Table 5-1). Urban land area in the BB-LEH Watershed now is 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.

A holistic management approach is being implemented to remediate environmental problems in BB-LEH associated with ongoing development and land use-land cover changes in the watershed. Multiple corrective strategies are being applied, such as improved stormwater control systems, implementation of best management practices in the watershed, smart

development, open space preservation, fertilizer controls, and education programs that explain to the public how and why these strategies are important and necessary for the protection of BB-LEH. Management of the watershed must also examine ways to minimize the creation of impervious surfaces, compacted soils, and sprawl, while concurrently preserving natural vegetation and landscapes. A well-coordinated and holistic management plan is critical to improving the ecological condition and resources of the estuary.

EUTROPHICATION

Eutrophication (defined as a long-term increase in nutrient and organic matter input in a water body, and associated eutrophic impacts) is responsible for insidious degradation of estuarine systems worldwide (Nixon, 1995; Boesch et al., 2001). Generally linked to nutrient loading from adjoining coastal watersheds and local airsheds, eutrophication has been deemed a priority problem of the BB-LEH Estuary (Kennish et al., 2007a; Kennish, 2009). Nutrient enrichment is problematic because it can over-stimulate the growth of phytoplankton as well as benthic microphytes and macrophytes. The result is often recurring phytoplankton blooms and the excessive proliferation of epiphytic algae and benthic macroalgae that can be detrimental to essential benthic habitats such as seagrass and shellfish beds. Dissolved oxygen levels may also be reduced.

Symptoms of serious eutrophication problems have escalated in the BB-LEH Estuary over the past decade, manifested as frequent phytoplankton and macroalgal blooms, declining shellfisheries (hard clams, *Mercenaria mercenaria*), and diminishing seagrass beds. Recurring phytoplankton blooms have been documented, including nuisance blooms (e.g., brown tides, *Aureococcus anophagefferans*) that occurred repeatedly between 1995 and 2002 (Olsen and Mahoney, 2001; Gastrich et al., 2004). Brown tide blooms were not monitored after 2004. Accelerated growth of drifting macroalgae (e.g., *Ulva lactuca*) has produced extensive organic mats that pose a potential danger to seagrass beds and other phanerogams that serve as vital benthic habitat for various recreationally and commercially important species (e.g., blue crabs, *Callinectes sapidus*; bay scallops, *A. irradians*; and tautog, *Tautoga onitus*). Rapid growth of other macroalgal species in the estuary, such as the rhodophytes *Agardhiella subulata*, *Ceramium* spp., and *Gracilaria tikvahiae*, may also have been detrimental. In addition, the decomposition of thick macroalgal mats promotes sulfide accumulation and the development of hypoxic/anoxic conditions in bottom sediments that can impact seagrasses and benthic infaunal communities.

Coastal lagoons differ from deeper estuaries in that a large fraction of the total system primary production originates in the benthic regime, notably microalgae and macroalgae, and seagrasses (Burkholder et al., 2007; McGlathery et al., 2007; Giordano et al., 2011). This is so because sunlight reaches the bottom of shallow coastal lagoons much of the time, enabling these

autotrophs to grow rapidly when nutrients and other factors are favorable. Unfortunately, benthic algae outcompetes seagrass in eutrophied estuaries often resulting in diminished production by the rooted macrophytes.

Light extinction by macroalgal mats during bloom development threatens seagrass integrity. Macroalgae require lower light intensities than seagrass for survival (Hily et al., 2004; McGlathery et al., 2007); hence, reduced light transmission to the estuarine floor can lead to the replacement of seagrass by rapidly growing macroalgae such as *Ulva lactuca* and *Enteromorpha* spp. From 2004 to 2010, 55 macroalgal bloom occurrences were recorded in the estuary (Kennish et al., 2011). These blooms not only attenuated or blocked light to the bottom of the estuary but also produced large biomasses of plant matter that may have significantly altered biogeochemical processes in bottom sediments, leading to low dissolved oxygen levels, as occurred in Barnegat Bay at Seawood Harbor (Brick) during July 2011. The Seawood Harbor macroalgal bloom in 2011 also released hydrogen sulfide gas to the atmosphere and caused respiratory problems for many people living in close proximity to the impacted bay waters. These events demonstrate the serious system and human impacts that can result from macroalgal blooms in shallow estuaries and coastal lagoons.

Frequent phytoplankton blooms can likewise cause shading of the benthos and potentially dangerous oxygen depletion. Both may result in indirect impacts on seagrass beds and other vital benthic habitat in the BB-LEH Estuary. Because excessive growth of benthic macroalgae can directly impact seagrass beds, it is also critically important to concurrently assess the effects of macroalgae on seagrasses (most notably *Zostera marina*) in the estuary.

Other significant biotic changes linked to nutrient enrichment of eutrophied estuaries have been shifts from large to small phytoplankton groups (diatoms and dinoflagellates to microflagellates and picoplankton) that can adversely affect shellfish species which consume the phytoplankton. Additional impacts include a shift from filter-feeding to deposit-feeding benthos, and a progressive change from larger, long-lived benthos to smaller, rapidly growing but shorter-lived species. The net effect therefore is the potential for a permanent alteration of biotic communities of a system (Rabalais, 2002).

Schramm (1999) and Rabalais (2002) described a predictable series of changes in autotrophic components of estuarine and marine ecosystems in response to progressive eutrophication. For those systems that are uneutrophied, the predominant benthic macrophytes inhabiting soft bottoms typically include perennial seagrasses and other phanerogams, with long-lived seaweeds occupying hard substrates. As slight to moderate eutrophic conditions arise, bloom-forming phytoplankton species and fast-growing, short-lived epiphytic macroalgae gradually replace the longer lived macrophytes; hence, perennial macroalgal communities decline. Under greater eutrophic conditions, dense phytoplankton blooms occur along with drifting macroalgal species (e.g., *Enteromorpha* and *Ulva*), ultimately eliminating the perennial and slow-growing benthic macrophytes, a situation that appears to be taking place in the BB-

LEH. With hypereutrophic conditions, benthic macrophytes become locally extinct, and phytoplankton overwhelmingly dominates the autotrophic communities.

Howarth et al. (2000a, b) and Livingston (2000) not only correlated hypereutrophication with proliferation of nuisance and toxic algal blooms but also with increased algal biomass, diminished seagrass habitat, increased biochemical oxygen demand, hypoxia/anoxia, degraded sediment quality, and loss of fisheries. Again, most of these effects appear to be occurring today in BB-LEH.

Eutrophication Conceptual Model

A general conceptual model advanced here for eutrophication in shallow coastal lagoons therefore includes a shift in plant dominance from seagrasses and perennial macroalgae to ephemeral, bloom-forming macroalgae, epiphytes, and phytoplankton. Similar conceptual models have been proposed for other shallow coastal bays in the mid-Atlantic region (see McGlathery et al., 2007; Wasniak et al., 2007). While these studies demonstrate a general shift in biotic components of these shallow coastal bays, a more complex seasonal and interannual pattern of biotic responses is evident in BB-LEH in response to watershed nutrient loading and nutrient enrichment of the estuary (Figure 5-1) (Kennish et al., 2007a, 2010, 2011).

Rather than a continuous gradient of biotic response with increasing nutrient loading as proposed by the Wazniak et al. (2007) model for the Maryland coastal bays, the BB-LEH Estuary responds somewhat differently to nutrient enrichment. When the system reaches some lower critical eutrophication threshold, the biotic responses here increase in variability and may take several different pathways. In some years, the estuary may switch to other community states. For example, during 1997, 2000-2002, BB-LEH experienced severe brown tide (*Aureococcus anophagefferens)* HAB events, but in 1998, 2004, and 2005, extensive macroalgal blooms were recorded and have persisted through ensuing years (2008-2010) (see Kennish et al., 2011). In 2006, low water clarity (likely caused by high phytoplankton-induced turbidity) resulted in estuary-wide seagrass dieoffs. Severe infestations of noxious sea nettles (*Chrysaora quinquecirrha*) were also documented; these eruptions of stinging jellyfish persisted each summer through 2011. Seagrass decline is well chronicled for the 2004-2010 period as detailed in Components 2 and 3 of this report.

Recurring blooms of drifting red and green macroalgae (e.g., *Gracilaria tikvahiae* and *Ulva lactuca*), similar to epiphyte plant overgrowth, threaten seagrass beds by attenuating or blocking light transmission to the beds. They also produce extensive organic mats that can alter biogeochemical processes in bottom sediments through the generation of sulfide in the rhizosphere which decreases nutrient uptake and contributes to additional reduction in photosynthesis, growth, and leaf density, and an increase in ammonium, oxygen depletion, and seagrass mortality (Burkholder et al., 2007; McGlathery et al., 2007). Investigations of macroalgal blooms in the BB-LEH over the six-year period from 2004-2010 (excluding 2007) revealed 55 occurrences (2.23 blooms m^{-2}) of Early Bloom (70%–80% macroalgal cover) and

Full Bloom (>80% macroalgal cover) events, which contributed to increased mortality of seagrass and the production of extensive bare bottom areas in the estuary (Kennish et al., 2011). Most of the blooms occurred from 2008-2010, a period when the loss of eelgrass biomass dropped to the lowest on record for the estuary as noted in Component 2 of this report (see also Kennish et al., 2010). The blooms were more frequent during June-July and August-September than during October-November, and these data suggest that the nitrogen loading threshold for the genesis of damaging macroalgal blooms in BB-LEH is rather low, with such events commonly initiated during late spring and early summer as nitrogen inputs increase together with the photoperiod and the level of light intensity. These factors are the key elements necessary for initiating algal bloom events.

Epiphytes can attenuate up to 90% of the light incident on seagrass leaves. The mean percent cover of epiphytes during all sampling periods in 2009 ranged from 19.2 to 38.3% for upper leaf surfaces and 18.4 to 38.3% for lower leaf surfaces. This is significant areal coverage. In 2010, the mean percent cover of epiphytes was generally lower than in 2009, with the values ranging from 11.3 to 25.7% for upper leaf surfaces and 10.7 to 24.4% for lower leaf surfaces. However, higher values of epiphyte percent cover were found during the October-November sampling period in 2010 than in 2009, with the mean upper leaf and lower leaf percent cover values ranging from 20 to 21% in October-November 2010 compared to values ranging from 18.4 to 19.2% in October-November 2009. The extensive epiphyte areal cover on seagrass leaves observed in 2009 and 2010 correlate with large-scale reduction in eelgrass biomass recorded concurrently in the estuary.

Eelgrass abundance decreased during the period of increased macroalgal blooms and elevated epiphyte occurrence. The reduction of eelgrass biomass begins relatively early in the growing season each year (Table 2-6), indicating once again that the threshold value of nutrient loading leading to a substantive decline in eelgrass abundance and biomass is likely exceeded early in the growing season (June-July or even earlier) for this estuary. For example, aboveground eelgrass biomass peaked in June-July 2004 (mean = 109.5 g dry wt m^{-2}), and then declined markedly 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 2-6). 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. Both seasonal and interannual trends of eelgrass biomass reductions have been observed in BB-LEH in response to ongoing eutrophy of the system.

In some years, HABs were likely the primary drivers of seagrass habitat change. The highest *A. anophagefferens* abundances (>10^6 cells L^{-1}), Category 3 blooms (200,000 cells L^{-1}), occurred in 1997 and 1999; they then recurred during the 2000-2002 period (Table 2-9), covering extensive geographic areas of the estuary (Gastrich et al., 2004). These HABs were

particularly extensive in Little Egg Harbor.

A hard clam (*Mercenaria mercenaria*) stock assessment conducted in Little Egg Harbor in 2001 during a major brown tide bloom season and following several years of Category 3 blooms revealed a major decline in hard clam abundance and density from the previous hard clam stock assessment survey conducted in the mid-1980s. These reductions are consistent with coastal bays that are eutrophied. Brown tides may cause shifts in phytoplankton food supply from larger diatoms and dinoflagellates to picoplanktonic pelagophytes such as *Aureococcus anophagefferens* that can lead to poor growth and compromised reproductive success of hard clams, as well as poor fertilization, lower clam densities, and even altered abundances of predator populations. BB-LEH has not only exhibited a shift towards picoplanktonic pelagophytes during the past 15 years, but also has supported high abundances of other small forms such as the green alga *Synechococcus* sp. and the chlorophyte *Nannochloris atomus* (Olsen and Mahoney, 2001). Smaller phytoplankton species are poorly captured and digested by hard clams, thereby having the potential to seriously impact their growth (Bricelj et al., 1984).

While we presently do not understand all factors controlling the substantial intra- and interannual variability noted above, existing evidence suggests that it is keyed into weather conditions, precipitation, and the amount and source (i.e., pulses of stormwater vs. the steady influx of groundwater discharge) of freshwater inflow, which in turn alters the relative ratio of different nutrient elemental forms. The outcome is relatively clear. The biotic response in the estuary is a shift in plant dominance from seagrasses and perennial macroalgae to ephemeral, bloom-forming macroalgae, epiphytes, and phytoplankton. This is the essence of the model.

Clearly, human development and alteration of the BB-LEH Watershed have played a major role in eutrophication of the BB-LEH Estuary (Figure 5-1). In addition, recycling of nitrogen from bottom sediments due to microbial-mediated processes such as ammonification can augment continuous nitrogen influx from the watershed. Indeed, microbial mineralization of the large biomass of decaying plant matter accumulating in sediments along the estuarine floor during the summer months can provide a large secondary source of nitrogen for reentry into the water column that can hasten the eutrophication process.

Increasing nonpoint source nitrogen loading from the watershed over the spring-fall period derives from fertilizer use and other human-source activities from a burgeoning watershed population (Bowen et al., 2007). The watershed population increases dramatically in summer, more than doubling from ~575,000 people to about 1,200,000 individuals. When TN loading exceeds some critical threshold value, there is a triggering of phytoplankton and macroalgal blooms, as well as increased epiphytic growth, that can significantly reduce light transmission to seagrass beds, leading to acute die-offs of the seagrass and the resident shellfish and other benthic invertebrates inhabiting the beds. In some years, phytoplankton blooms predominate, while in other years macroalgal blooms have greater importance. Together, the blooms can severely impact the estuarine food web and modify the spatial benthic habitat structure. This

process is likely exacerbated by the decomposition of organic matter and recycling of nutrients to the water column during the warmer months of the year. Through time, this detrimental process may culminate in a "permanent" change in biotic community structure and function of the system (Figure 5-1).

A major outcome of this work is that continuous quantitative measures of seagrasses and other biotic indicators are necessary to accurately assess the overall ecological health and integrity of the estuary. In addition, threshold values of nutrient enrichment leading to declining shifts in seagrass demographics, as well as other adverse biotic responses such as nuisance and toxic algal blooms, and diminishing shellfish resources, must be assessed on a regular basis. This is the knowledge and understanding needed to synthesize comprehensive and representative nutrient criteria and to generate a highly effective, long-term nutrient management plan.

IMPAIRMENT

Dissolved Oxygen

BB-LEH Estuary is an impaired system both in respect to aquatic life support and human use as is evident in the conclusions of this study. In the case of water quality, there were 82 occurrences of dissolved oxygen (DO) levels ≤ 4 mg L^{-1} (the surface water quality criterion for DO is 4 mg L^{-1}) in the estuary and tributary systems at multiple sampling sites between 1989 and 2010 (Figure 5-3). Most of these low DO values occurred in the south segment (N = 63), with far fewer in the central segment (N = 13) and north segment (N = 6) (Figure 5-4). These values represent only one DO measurement taken quarterly and mainly during the morning daylight hours at a sampling station (and hence likely underestimate significantly the number of low DO events in the estuary); the date, time, estuary segment, and DO levels of all 82 low DO values are listed in Table 5-2. Of the 82 low DO values recorded, 18 were found in the main body of the estuary and the remainder in tributaries. The state's List of *Water Quality Limited Waters* (i.e., section 303(d) of the Clean Water Act), therefore, includes the north segment of BB-LEH, which is now designated as impaired for dissolved oxygen. This listing for the north segment is based on continuous water quality monitoring by automated datalogger instrumentation. Depressed DO levels are potentially hazardous to the maintenance of balanced indigenous populations of fish, shellfish, and other aquatic life (Breitburg et al., 2001; Breitburg, 2002).

Regulatory protection and conservation of New Jersey's estuarine waters are based on dissolved oxygen measurements. Yet dissolved oxygen is only one indicator of ecological health, and must be monitored continuously (via automated dataloggers for example) at multiple locations for accurate assessment because of natural fluctuations over the course of a day due to natural processes such as changes in temperature or light, as well as community photosynthesis

and respiration. This level of monitoring has not been done in BB-LEH. Therefore, it is critical that assessments of ecological health also examine biotic indicators covering a broader range of physicochemical indicators in the watershed and estuary for effective ecosystem-based assessment and management. This project establishes appropriate biotic indicators and a framework for assessment using multiple biotic indices that will aid New Jersey in delineating environmental impairments using a broader, more relevant range of factors. The results of this report show conclusively that much of the estuary is in a state of insidious ecological decline.

Sea Nettles

Blooms of sea nettles (*Chrysaora quinquecirrha*) have commonly occurred in BB-LEH over the past decade, most notably in the north segment of the estuary. High abundances of sea nettles have made bathing beaches and other waters in the estuary non-swimmable, creating impairment for human use. These impaired waters are predominantly found along the mainland shoreline in the north segment. This is so because sea nettles prefer warm (~25-30 ºC), low salinity (~10-17‰) waters that occur north of Cedar Creek during the summer months in an area with bulkheaded shoreline and high inflow of freshwater from larger influent systems. Bulkheading provides excellent habitat for the early life history (polyp) stage of sea nettles, which attach to the bulkhead surfaces and overwinter to repopulate the northern bay during the following spring. Sampling in 2011 had revealed much higher numbers of sea nettles at Brick (western side of Barnegat Bay) than Lavallette (eastern side of the Barnegat Bay) in the northern segment (Figure 5-5).

Adult sea nettles (medusa stage) are free-floating forms that have a well-developed, bell-shaped cap (> 10 cm in diameter) from which an array of tentacles extend downward toward the estuarine floor. The tentacles, which can be more than 1 m in length, contain numerous nematocysts that pose a threat to pelagic organisms and a hazard to unsuspecting swimmers. The unusual anatomy of sea nettles and other jellyfish species facilitates their relatively rapid transport by currents.

The occurrence of sea nettle blooms in the north segment has resulted in extensive non-swimmable waters in violation of the Clean Water Act (Figure 5-6). Lower salinity waters north of Toms River have the greatest numbers of sea nettles and the most impaired bathing areas due to sea nettle occurrence.

Repeated blooms of sea nettles have appeared in the estuary since 2004. Prior to 2000, sea nettles were not present in such high abundances in the coastal bays. The cause of recent eruptions of sea nettles has not been unequivocally established, although increasing eutrophication and hardened shorelines have likely contributed to the problem.   Currently, approximately 40-45% of the estuarine shoreline is bulkheaded. Most of the north segment of the estuary is now bulkheaded, which provides ideal overwintering habitat for sea nettles. Warmer

sea and bay temperatures have also likely led to increased abundances of sea nettles. The co-occurrence of sea nettle blooms and high nutrient inputs (>1 million kilograms per year of nitrogen to Barnegat Bay) may indicate a direct link to human activities, especially in northern coastal watershed areas, which yield the greatest nutrient load to the estuary. A similar relationship has been observed in Chesapeake Bay and its watersheds.

Research scientists Jennifer Purcell (Western Washington University) and Robert Ulanowicz (University of Maryland) have stressed the potential dangers of sea nettle blooms on estuarine food chains. Most importantly, much of the energy flow in food chains dominated by sea nettles does not pass upward to upper-trophic-level organisms, thereby reducing biotic production of the system. The result is substantially altered biotic communities.

There is no clear solution to the proliferation of sea nettles in the estuary. Remedial actions that involve physical removal of sea nettles from estuarine waters are rarely successful once they take up residence. As noted previously, attempts to net and remove jellyfish may actually increase their long-term distribution and abundance. The recommended approach is to reduce pollution inputs and eutrophic conditions in the estuarine water body, as well as hardened shorelines that provide overwintering habitat. Water quality alteration must also be minimized by improving pollution controls in the watershed source. In addition, greater enforcement of environmental regulations is necessary, as is the establishment of nutrient criteria (which currently do not exist) for estuarine waters. The long-term solution to the sea nettle problem in New Jersey coastal bays requires more effective administrative/management intervention.

Annual population surveys of sea nettles are necessary to effectively monitor their distribution and abundance in the estuary. Population eruptions of sea nettles in Barnegat Bay have occurred since 2004, and they have impaired waters for human use (i.e., swimming). This organism also poses a serious threat to the structure and function of the estuarine food web.

Shellfish Resource

Hard clam (*Mercenaria mercenaria*) harvest in BB-LEH decreased by more than 98% between 1970 and 2005 (from 636,364 kg in 1970 to 6,820 kg in 2005), with harvest statistics being unreported since 2005 (Figure 1-3). These numbers are indicative of an ongoing insidious ecological decline of the estuary. The cause of this dramatic decline has not been unequivocally established, although the diminution in hard clam landings has occurred during an escalating period of nutrient enrichment and eutrophication of the estuary. Hard clam landings are affected by several factors besides absolute abundance. For example, fishing effort, market value, and shellfish bed closures all affect hard clam harvest. Currently, BB-LEH has a very limited commercial fishery for hard clams, and it also has a limited recreational fishery. Eastern oysters (*Crassostrea virginica*) and bay scallops (*Argopecten irradians*), historically valuable shellfish resources in the estuary, are no longer of commercial or recreational importance in the system.

The NJDEP surveyed Barnegat Bay and Little Egg Harbor in 1986/87 and reported that the hard clam population was present at densities of 1.4 and 2.5 m^{-2}, respectively. Little Egg Harbor was resurveyed in 2001, and the population density had dropped to 0.81 m^{-2} (Celestino, 2003). Based on a modeling study of the hard clam population in Islip town waters of Great South Bay, New York (Hofmann et al., 2006), a density of ~0.7 clams m^{-2} was found to be the minimum necessary to sustain the hard clam population (Kraeuter et al., 2005). The decrease in population density observed in Little Egg Harbor signals a population in marked decline.

Of even greater concern was the marked decline in the hard clam stock abundance documented in Little Egg Harbor between 1986/87 and 2001. As reported by Celestino (2003), a total of 64,803,910 hard clams were estimated in LEH in 2001 compared with an estimated 201,476,066 in 1986/87, representing a decrease of more than 67% in stock abundance over this period. The hard clam population has been in a state of precipitous decline for years. The loss of such large numbers of hard clams also appears to reflect a shift or transition in the system away from one of top-down control exerted by filter feeders consuming and regulating phytoplankton populations to one of bottom-up control limited by nutrient inputs. This shift may be driven by increasing eutrophic conditions in the estuary.

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FIGURES
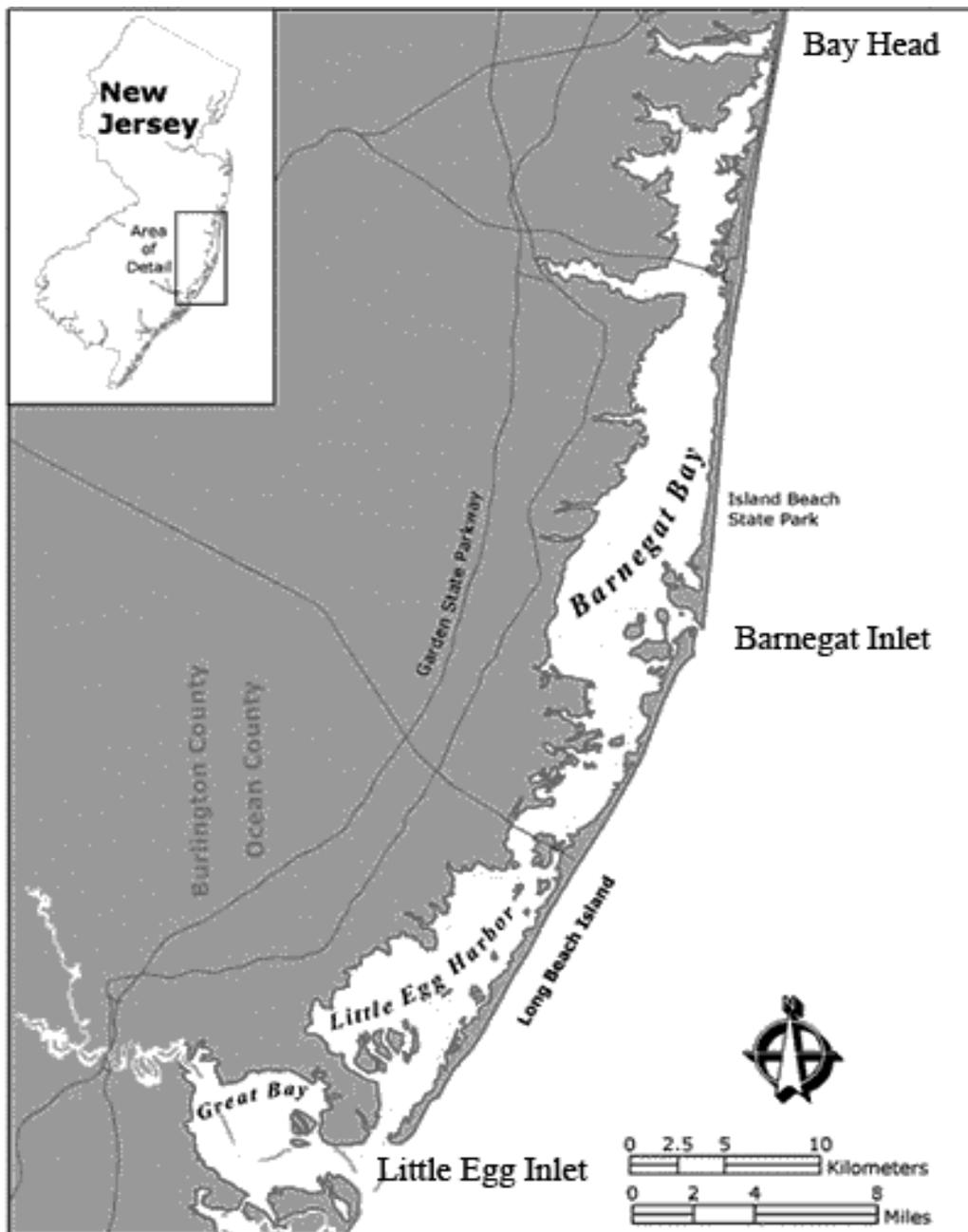
Figure 1 - 1 Map of the Barnegat Bay-Little Egg Harbor (BB-LEH) Estuary. Inset shows the location of the estuary with respect to the state of New Jersey.

Total Nitrogen Averaged 1989 - 2009

a) June - July   **b)** August - September   **c)** October - November

Total Nitrogen

μg L⁻¹		μM
> 840	🟥	> 60
701 - 840	🟧	51 - 60
561 - 700	🟨	41 - 50
421 - 560	🟩	31 - 40
< 421	🟩	< 31
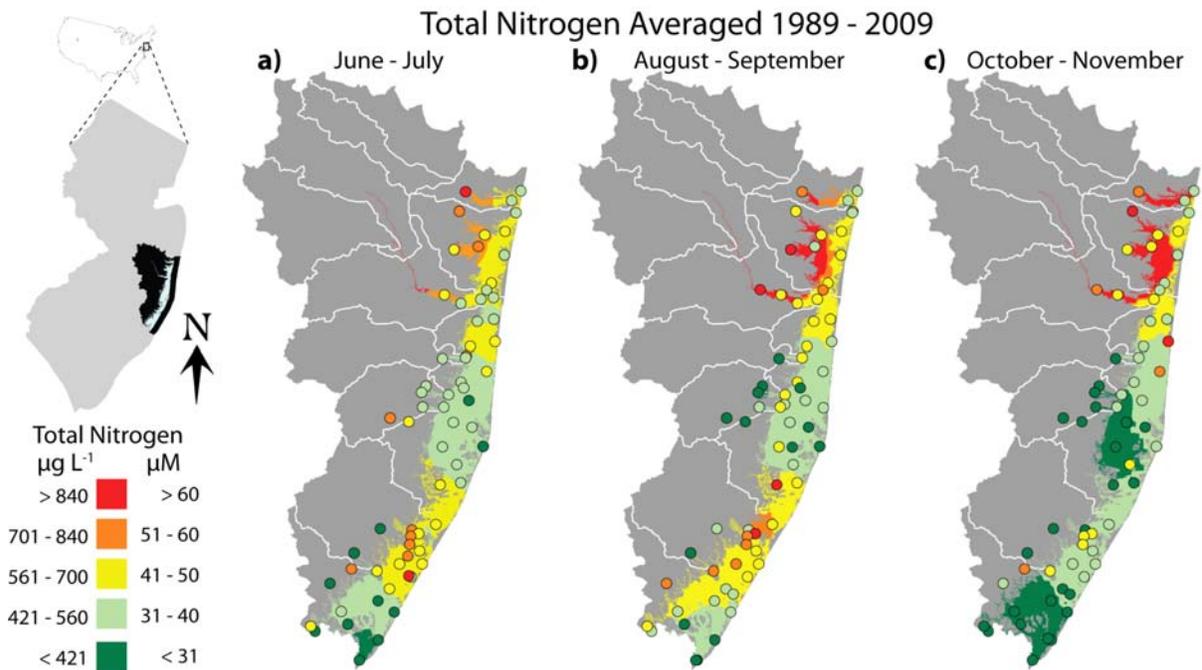
Figure 1 - 2 Mean total nitrogen concentrations in the BB-LEH Estuary from 1989-2009.
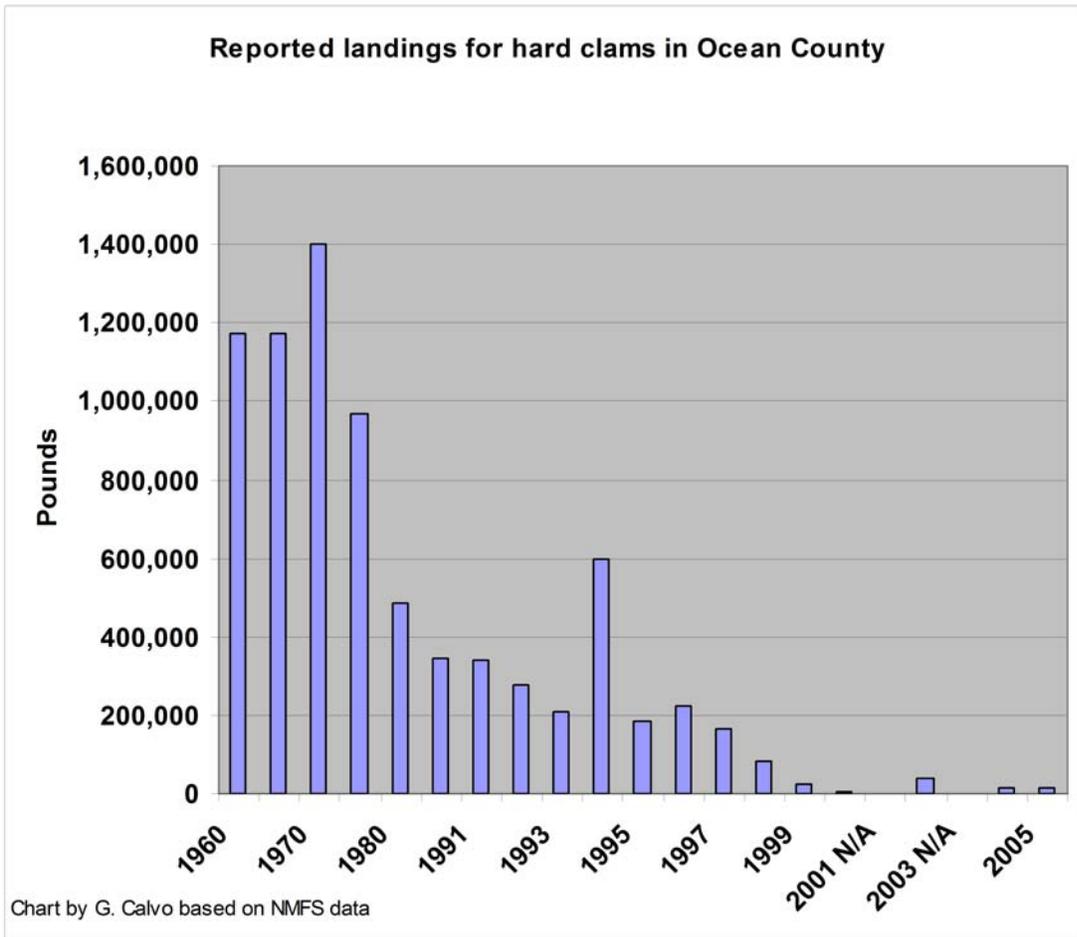
Figure 1 - 3 Hard clam landings for Ocean County showing acute decline from 1960 to 2005. Data from the National Marine Fisheries Service.
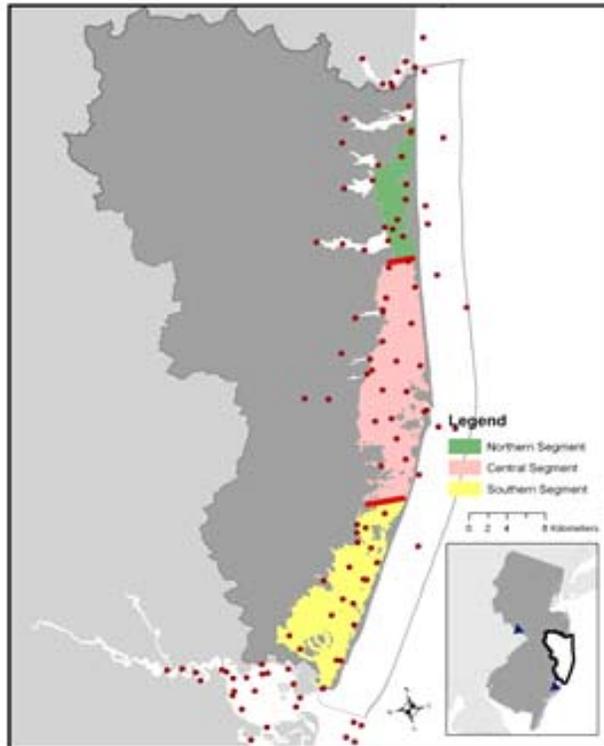
Figure 1 - 4 Map of the BB-LEH Estuary showing three segments (north, central, and south) used for index development.
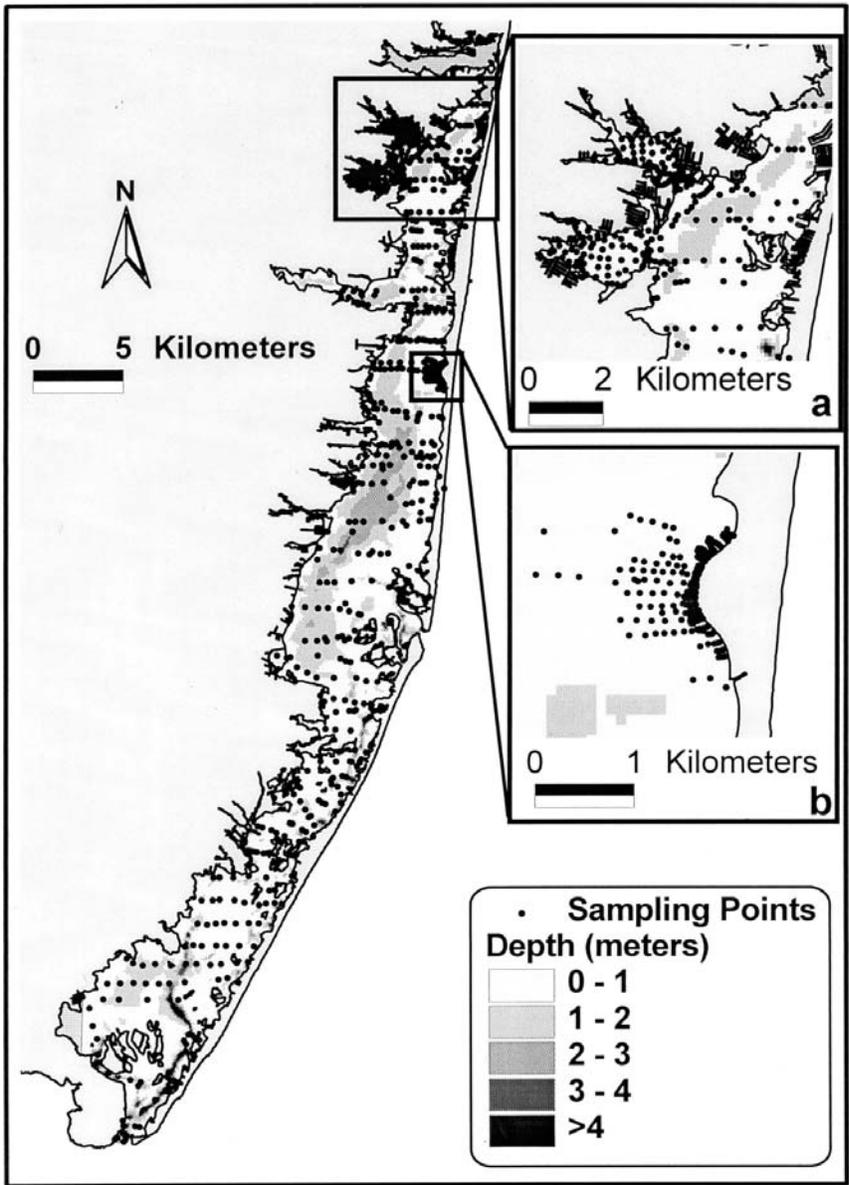
Figure 1 - 5 Map showing a grid of bottom sediment sampling stations and bathymetric measurements in the BB-LEH Estuary. (From Psuty, 2004).
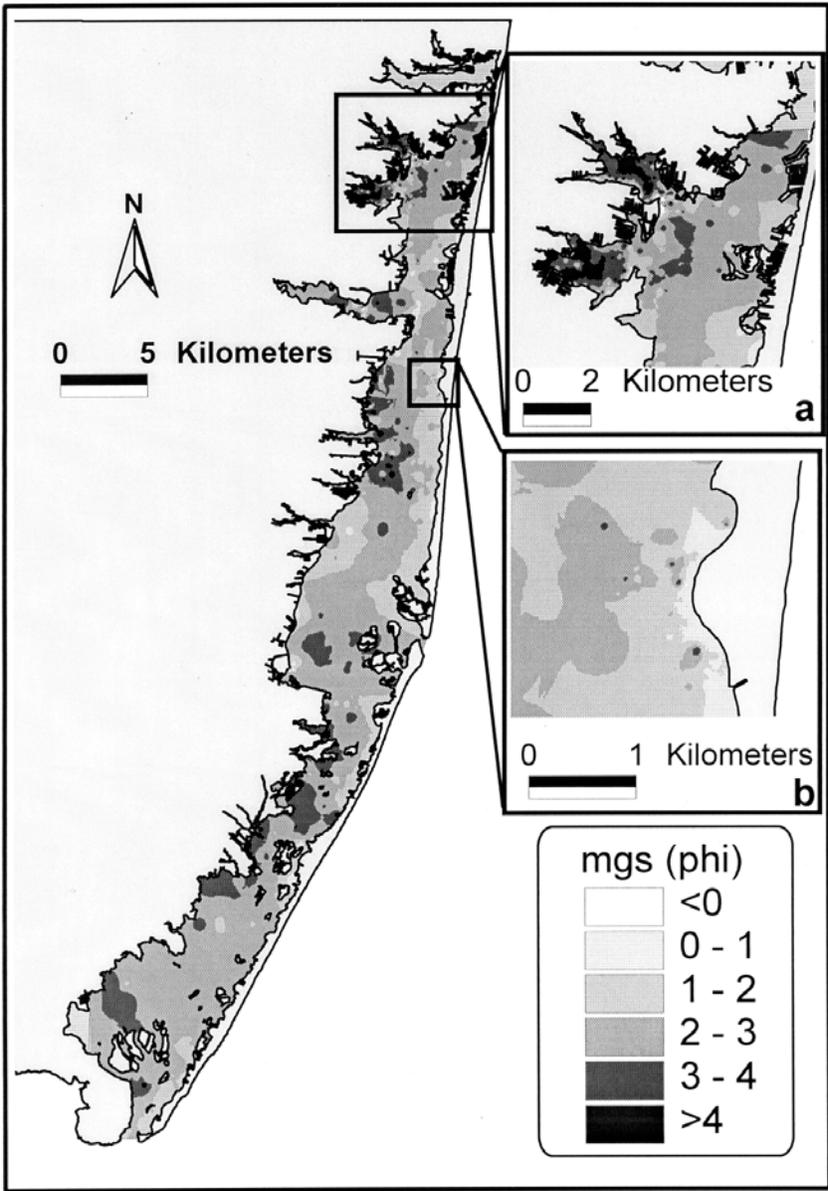
Figure 1 - 6 Bottom sediment composition and distribution (phi units) documented in the estuary. Finer grained sediments (silt, clay, and organic material) derived from upland areas, streams, and wetlands concentrate along the mainland and west side of the estuary. Well-sorted sands of marine origin and the back barrier predominate on the east side of the estuary. Sediment distribution may show a larger area of sediment type than actually exists due to the spacing of sampling locations and occurrence of mosaic patterns. (From Psuty, 2004).

Atlantic Ocean

North Branch Metedeconk River

Toms River

Cedar Creek

Barnegat Bay

Manahawkin Bay

Westecunk Creek

North Segment

Central Segment

South Segment

Little Egg Harbor

0 1 2 4 Miles
0 1 2 4 Kilometers

N
W · E
S

Washingtons Crossing

NJ NY

PA

Edwin B. Forsythe NWR

- northern_bay_seagrass
- NJDEP nutrient and physical parameters
□ Barnegat Bay Watershed boundaries
○ bay breakdownCopy
✚ Continuous stream gages
· Biological dataset: (Seagrass)
■ GIS dataset: seagrass coverage
■ GIS dataset: Urban areas 2002
■ WATER
▲ Atmospheric depostion dataset: (Insert map)

Figure 1 - 7 Map of the BB-LEH Estuary showing the location of 15 biotic sampling transects (150 sampling stations) in 2011.

Seagrass transects and sampling sites
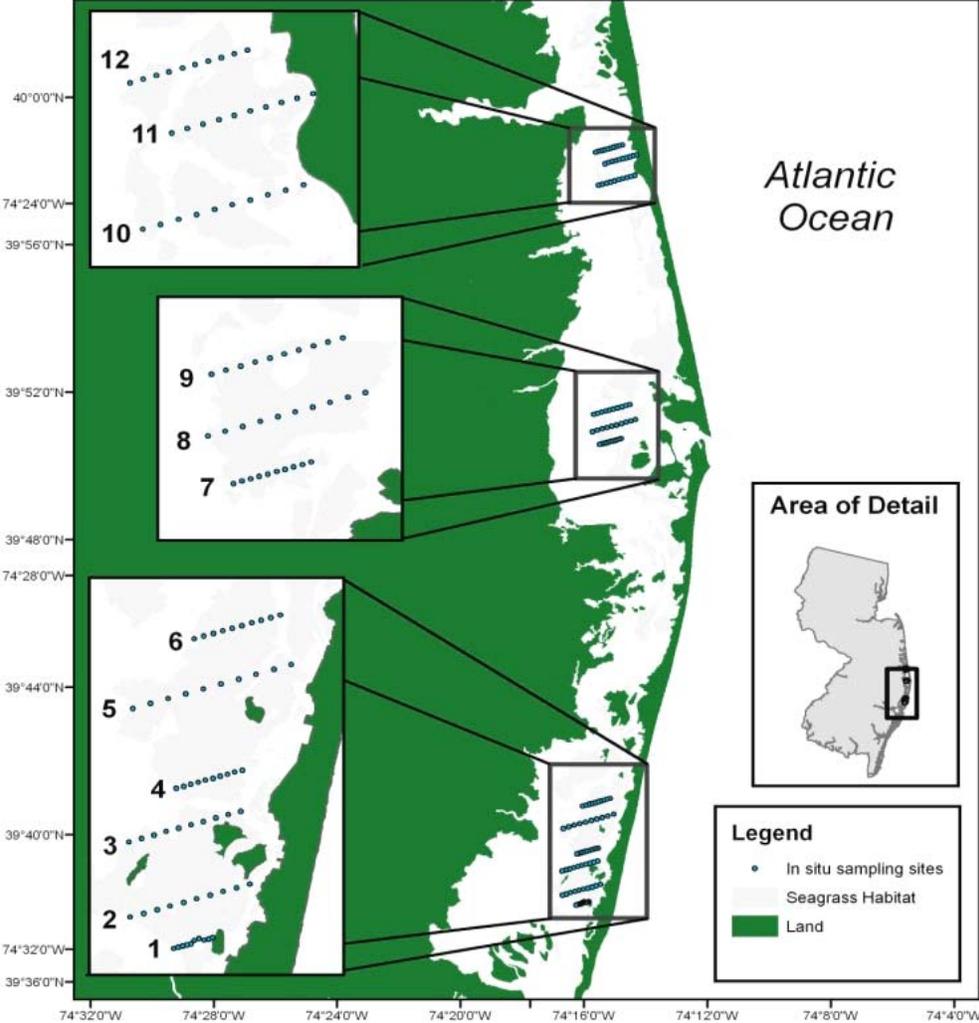
Figure 1 - 8 Study area showing 120 seagrass sampling sites along 12 transects in the BB-LEH Estuary from 2004-2010.

Seagrass Sampling Locations
in Northern Barnegat Bay
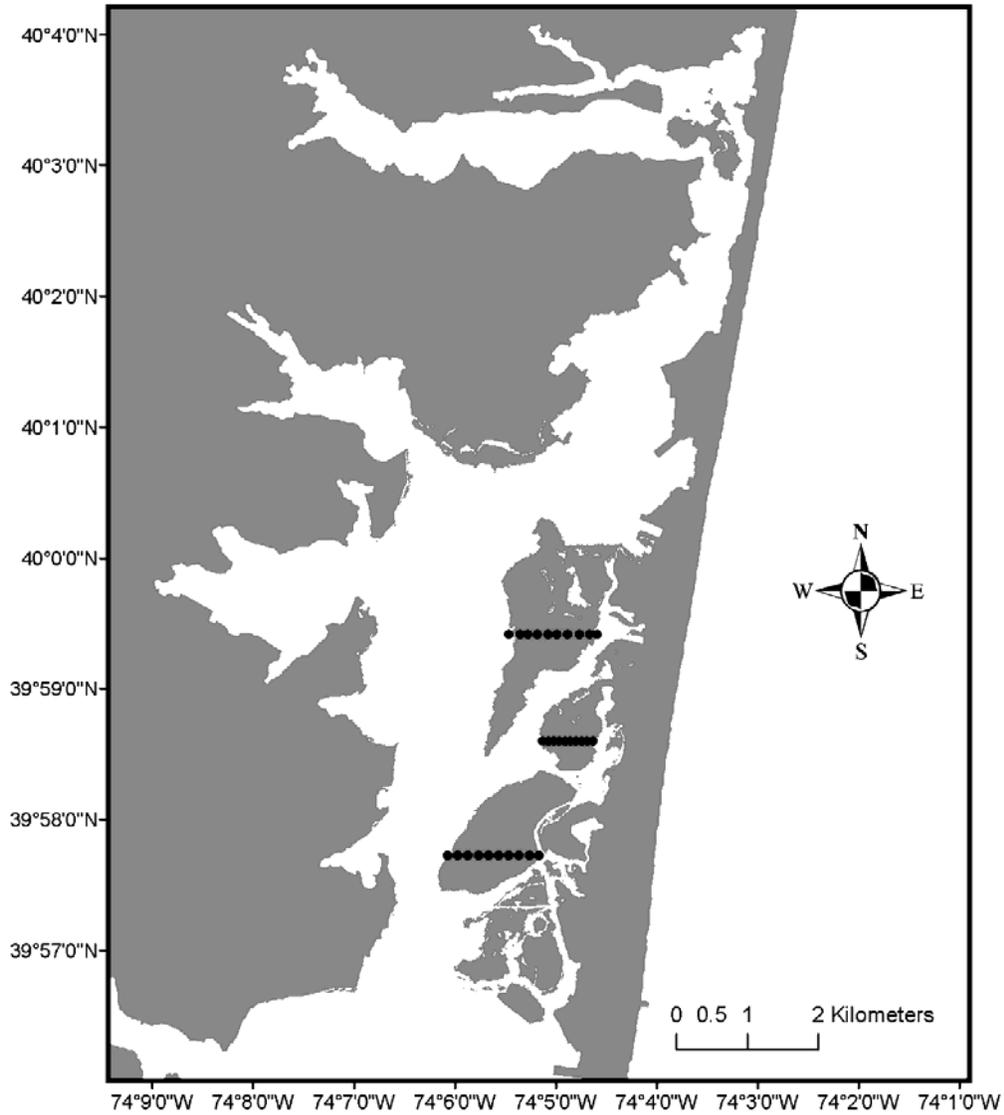
Figure 1 - 9 Seagrass transects established in the north segment of the BB-LEH Estuary for SAV sampling in 2011.
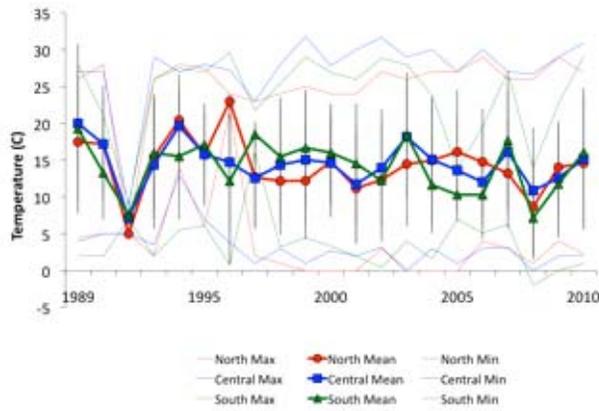
Figure 2 - 1 Minimum, mean, and maximum temperatures recorded in the BB-LEH Estuary from 1989-2010. Data from the New Jersey Department of Environmental Protection.
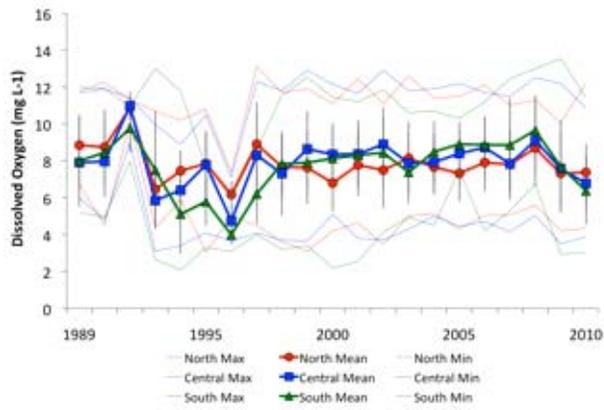
Figure 2 - 2 Minimum, mean, and maximum dissolved oxygen values recorded in the BB-LEH Estuary from 1989 to 2010. Data from the New Jersey Department of Environmental Protection.
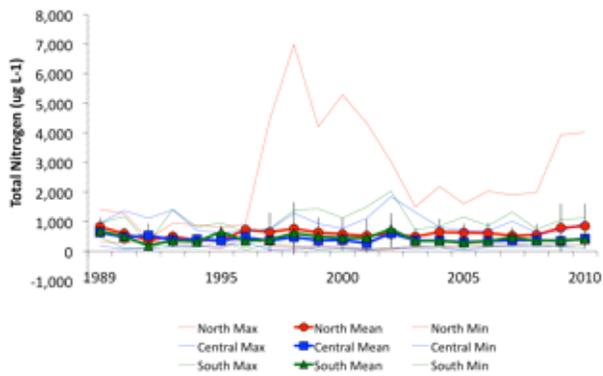
Figure 2 - 4 Minimum, mean, and maximum total phosphorus levels recorded in the BB-LEH Estuary from 1998 to 2010. Data from the New Jersey Department of Environmental Protection.

135
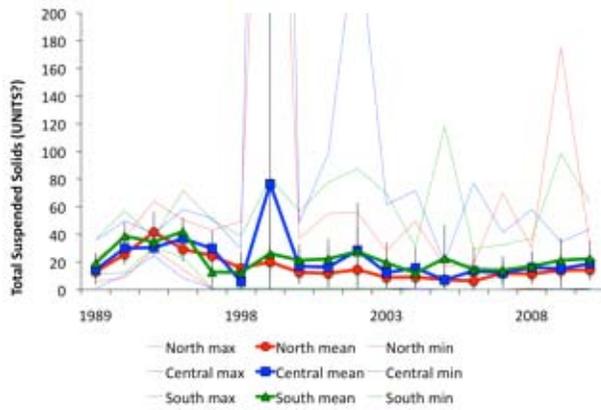
Figure 2 - 5 Minimum, mean, and maximum total suspended solids recorded in the BB-LEH Estuary from 1989 to 2010. Data from the New Jersey Department of Environmental Protection.
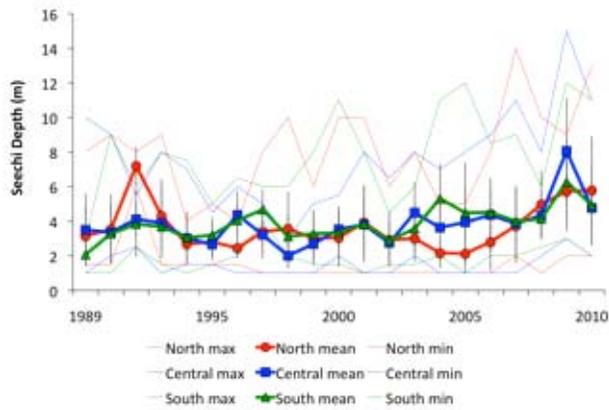
Figure 2 - 6 Minimum, mean, and maximum Secchi depth recorded in the BB-LEH Estuary from 1989-2010. Data from the New Jersey Department of Environmental Protection.

136
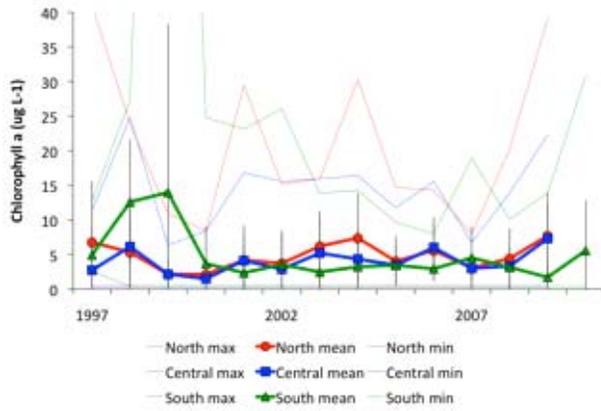
Figure 2 - 7 Minimum, mean, and maximum chlorophyll a values recorded in the BB-LEH Estuary from 1997 to 2010. Data from the New Jersey Department of Environmental Protection.
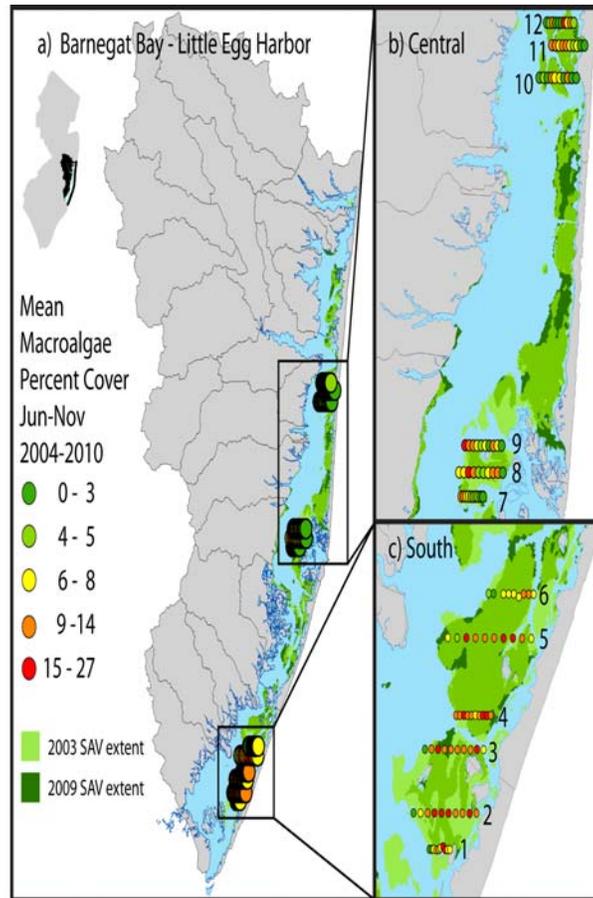
Figure 2 - 8 Mean macroalgae percent cover by sampling transect in the central and south segments of the estuary during the 2004-2010 period.
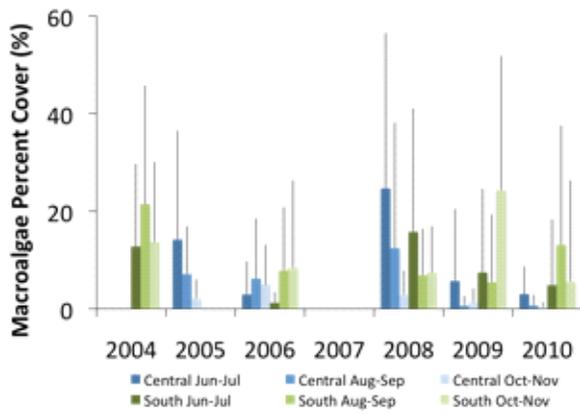
Figure 2 - 9 Mean macroalgae percent cover by sampling year (2004-2010) in the central and south segments of the estuary.
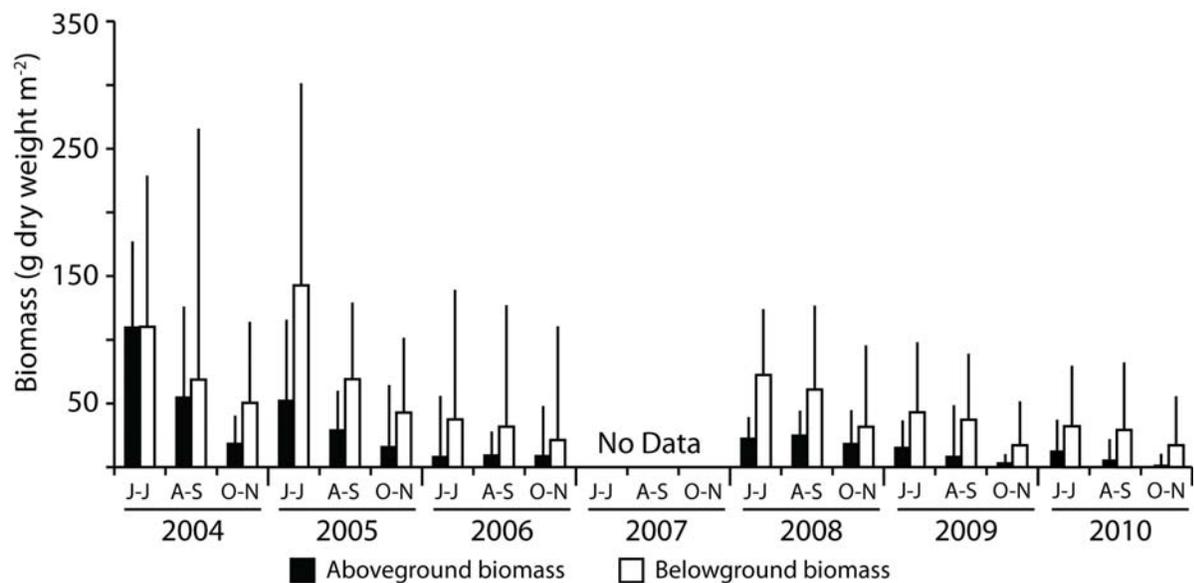
Figure 2 - 10 Mean aboveground and belowground eelgrass biomass values in the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
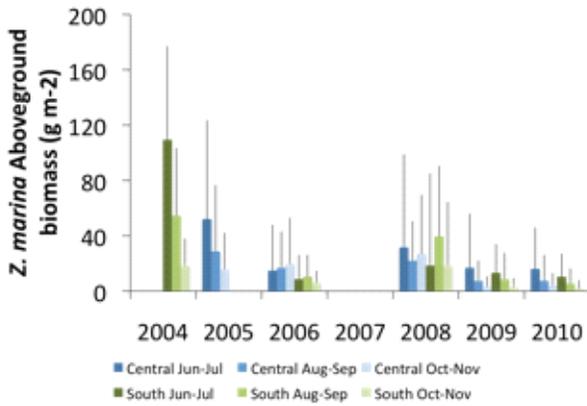
Figure 2 - 11 Mean aboveground eelgrass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
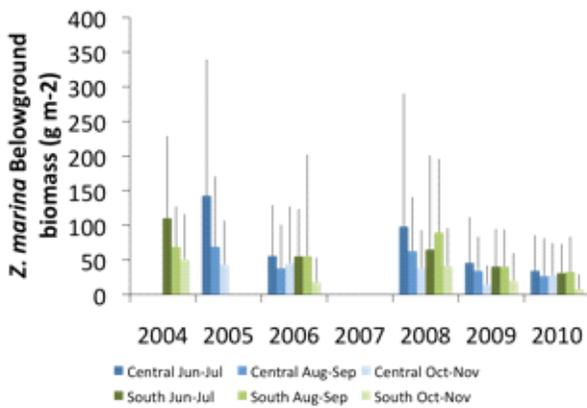
Figure 2 - 12 Mean belowground eelgrass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
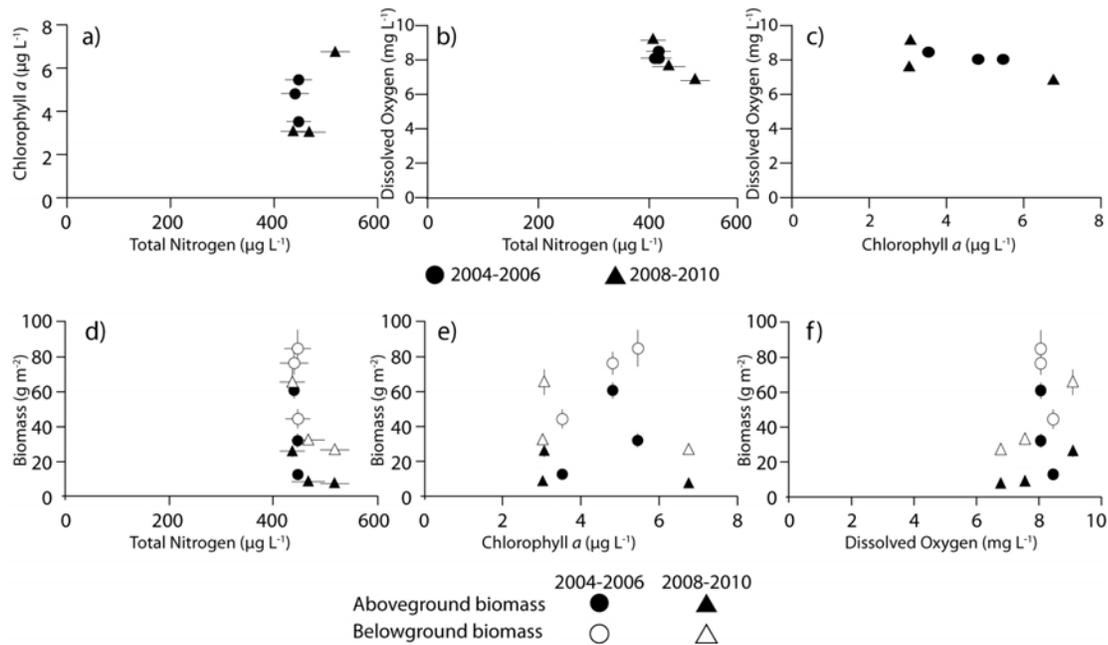
Figure 2 - 13 Variation of water quality and biological metrics for 2004-2006 (circles) and 2008-2010 (triangles). Eelgrass biomass is divided into aboveground (black) and belowground (white) components. Plots include chlorophyll a *vs. total nitrogen (a), dissolved oxygen vs. total nitrogen (b), dissolved oxygen vs. chlorophyll* a *(c), eelgrass biomass vs. total nitrogen (d), eelgrass biomass vs. chlorophyll* a *(e), and eelgrass biomass vs. dissolved oxygen (f).*
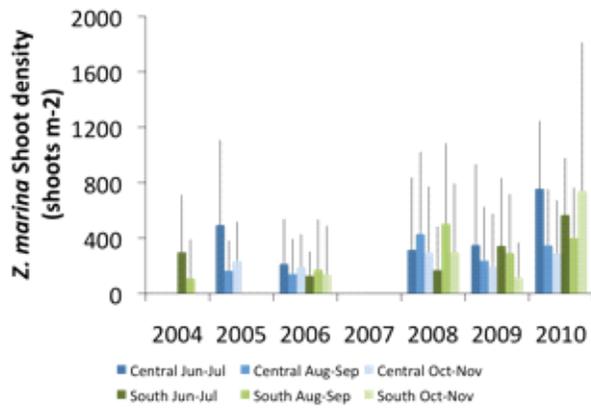
Figure 2 - 14 Mean eelgrass shoot density in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
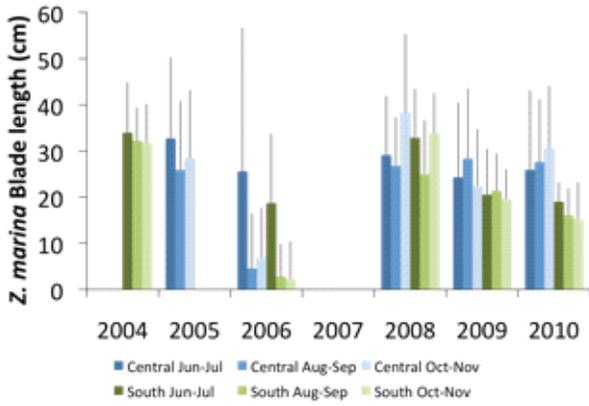
Figure 2 - 15 Mean eelgrass blade length in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
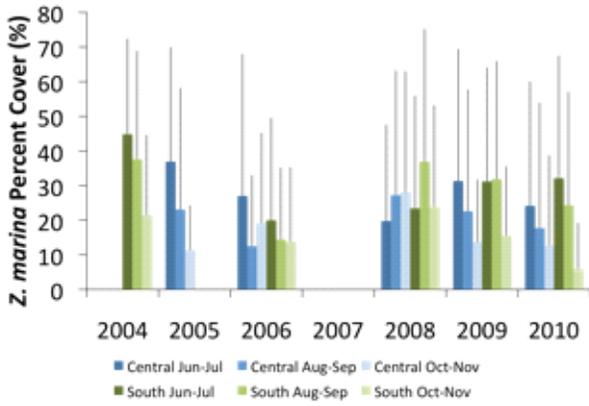
Figure 2 - 16 Mean eelgrass percent cover in the central and south segments of the BB-LEH during the spring-fall sampling periods from 2004 to 2010.
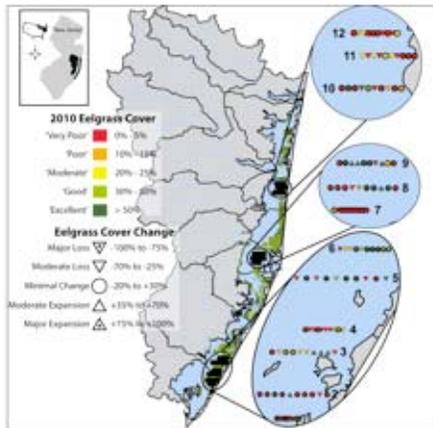
Figure 2 - 17 Eelgrass percent areal cover along 12 transects in the BB-LEH Estuary during 2010.
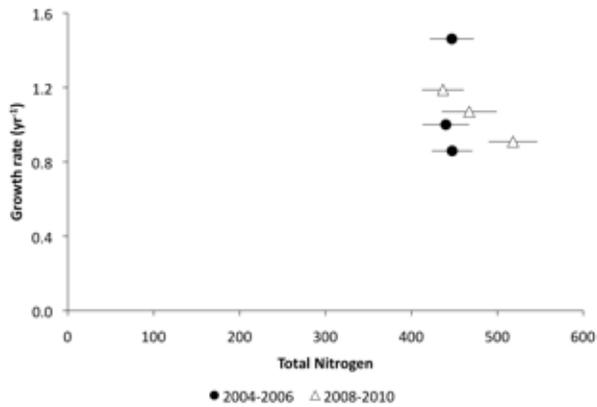
Figure 2 - 18 Annual growth rate of Z. marina in BB-LEH Estuary vs. total nitrogen concentrations, during 2004-2006 (black circles) and 2008-2010 (white triangles).
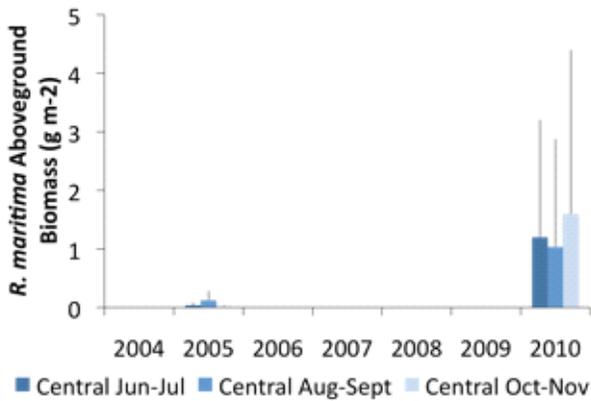
Figure 2 - 19 Mean aboveground widgeon grass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

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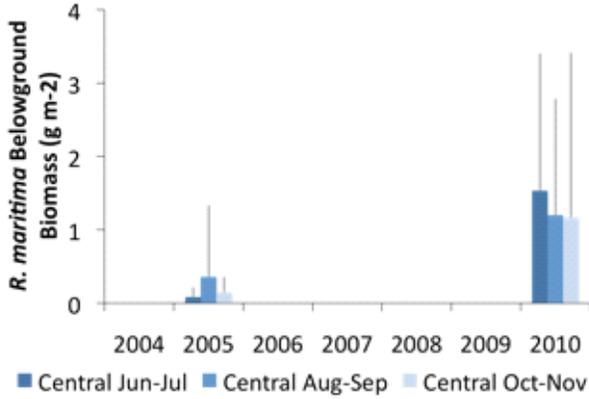
Figure 2 - 20 Mean belowground widgeon grass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
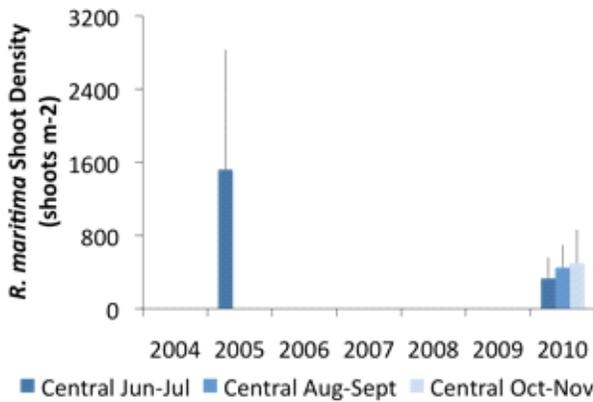
Figure 2 - 21 Mean widgeon grass shoot density values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
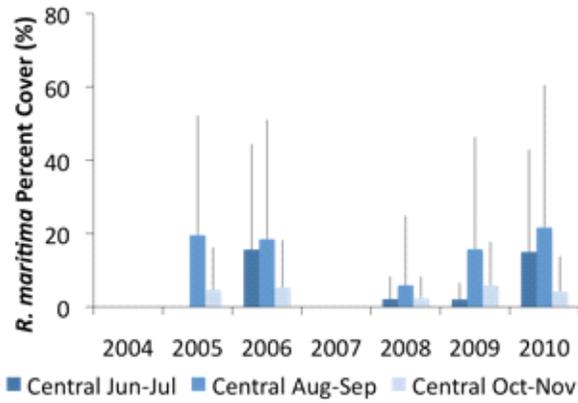
Figure 2 - 22 Mean widgeon grass percent cover values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.
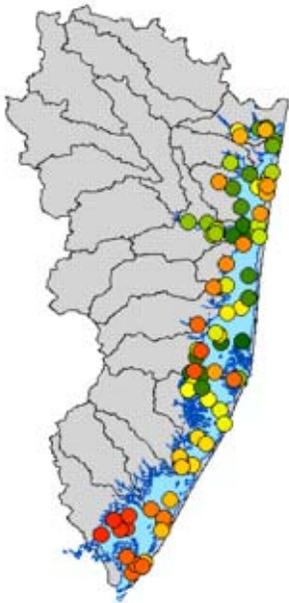
Figure 2 - 23 Benthic invertebrate sampling stations (2001) of the USEPA Regional Monitoring and Assessment Program for the BB-LEH Estuary.

Figure 3 - 1 Comparison of indicators used by Bricker et al. 2009 and those used in this Biotic Index of Eutrophication Condition.

Figure 3 - 2. Temporal and spatial data availability for indicators used in the Index of Eutrophication.
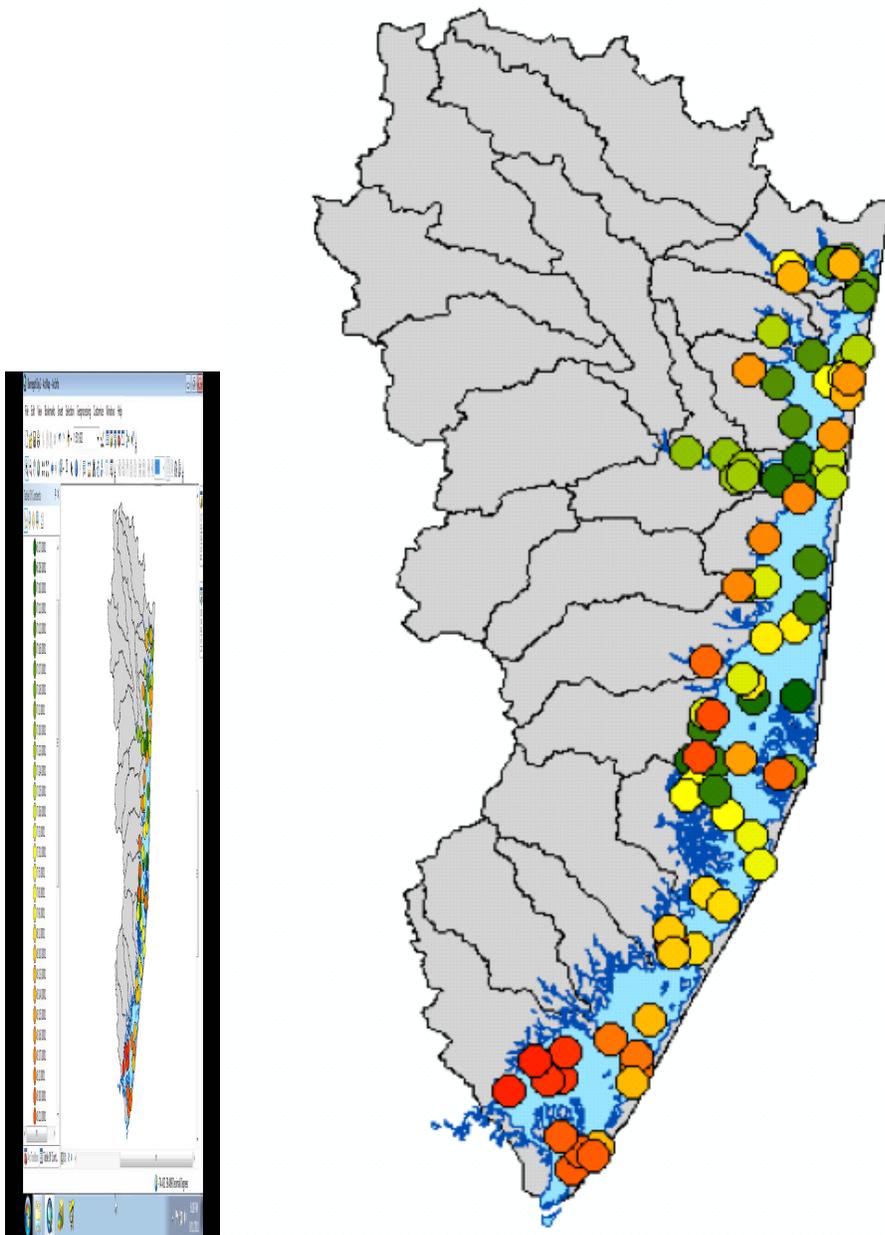
Figure 3 - 3 Map indicating REMAP sampling locations and dates.
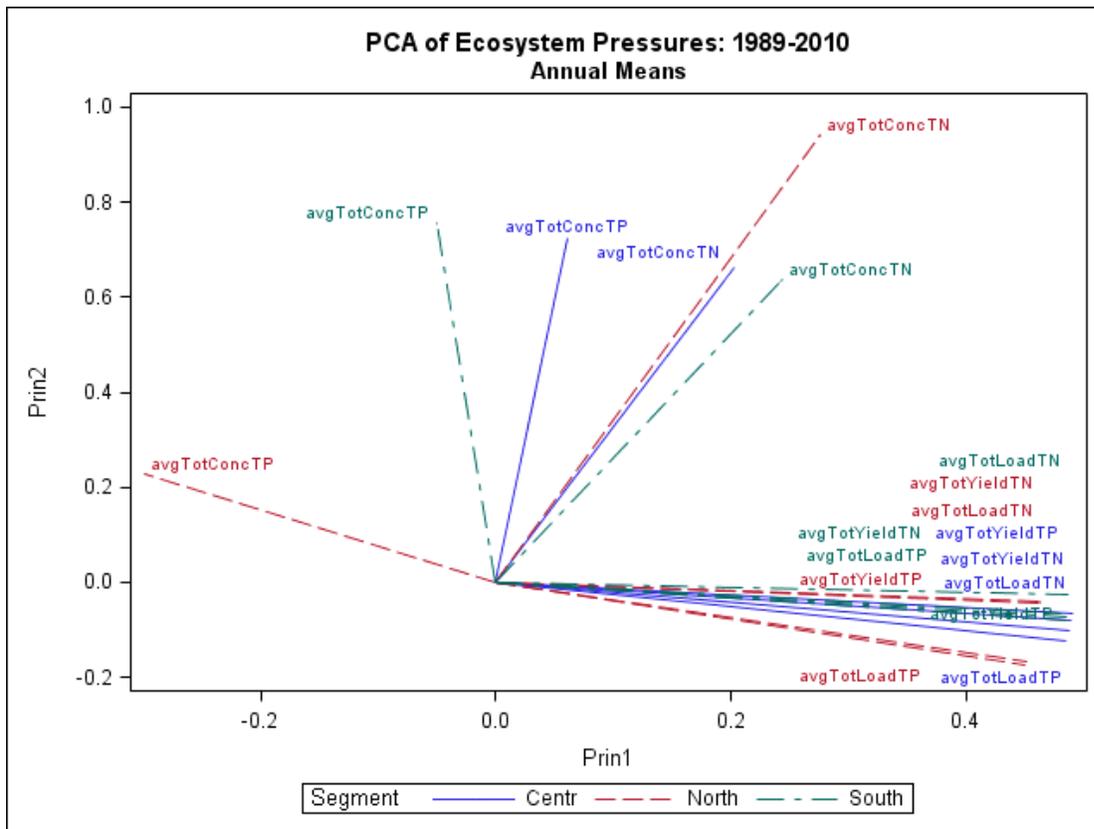
Figure 3 - 4 Principal component analysis of Total Loading, Total Yield, and Flow-weighted average total concentration for total nitrogen and total phosphorus.

Light variables vs. Total Concentration for Total Nitrogen (TN)
Annual Means
epi/sav = g DW epiphytes / g DW sav

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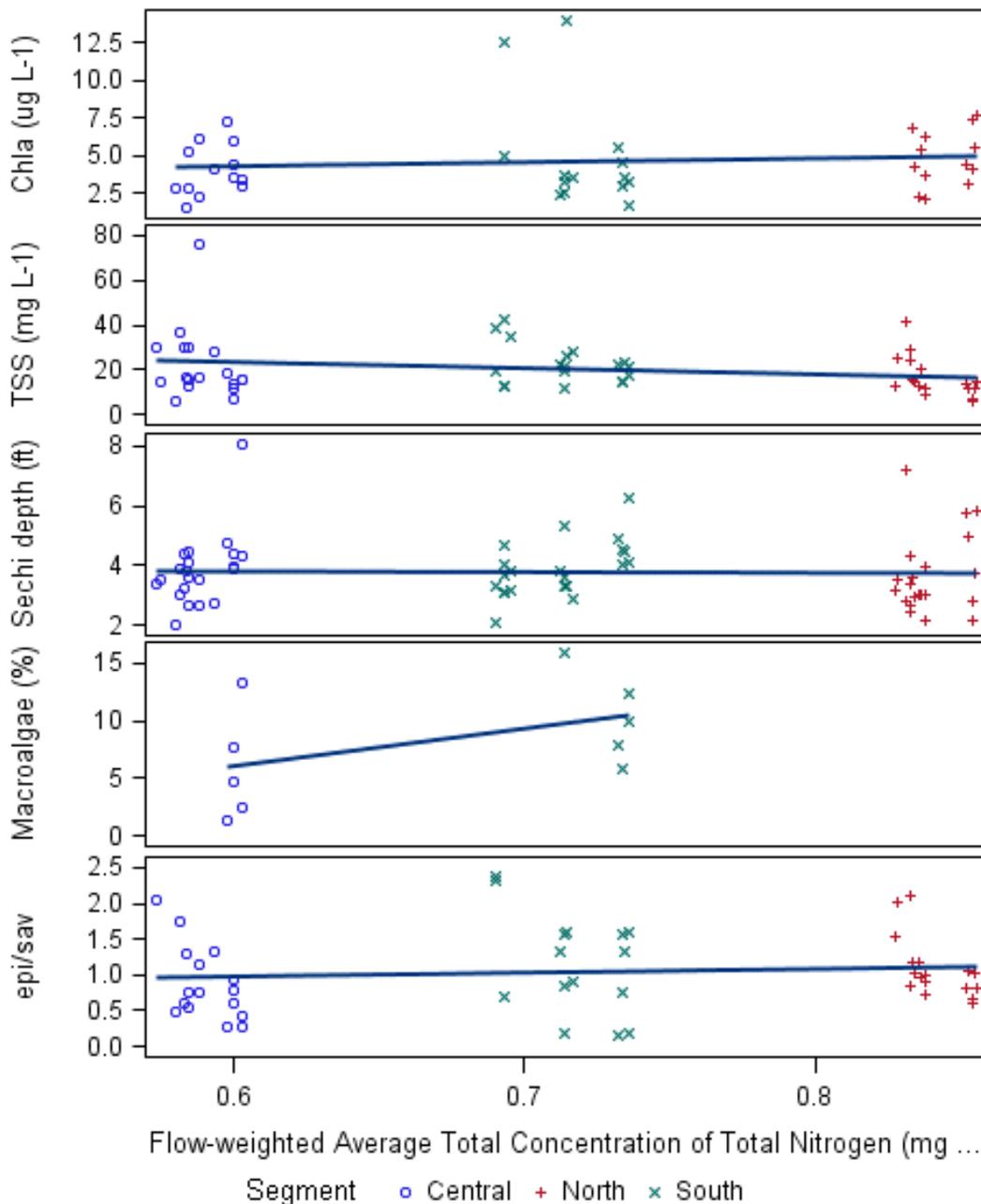
Figure 3 - 5 Light variables vs. flow-weighted average total concentration of total nitrogen (mg L⁻¹).
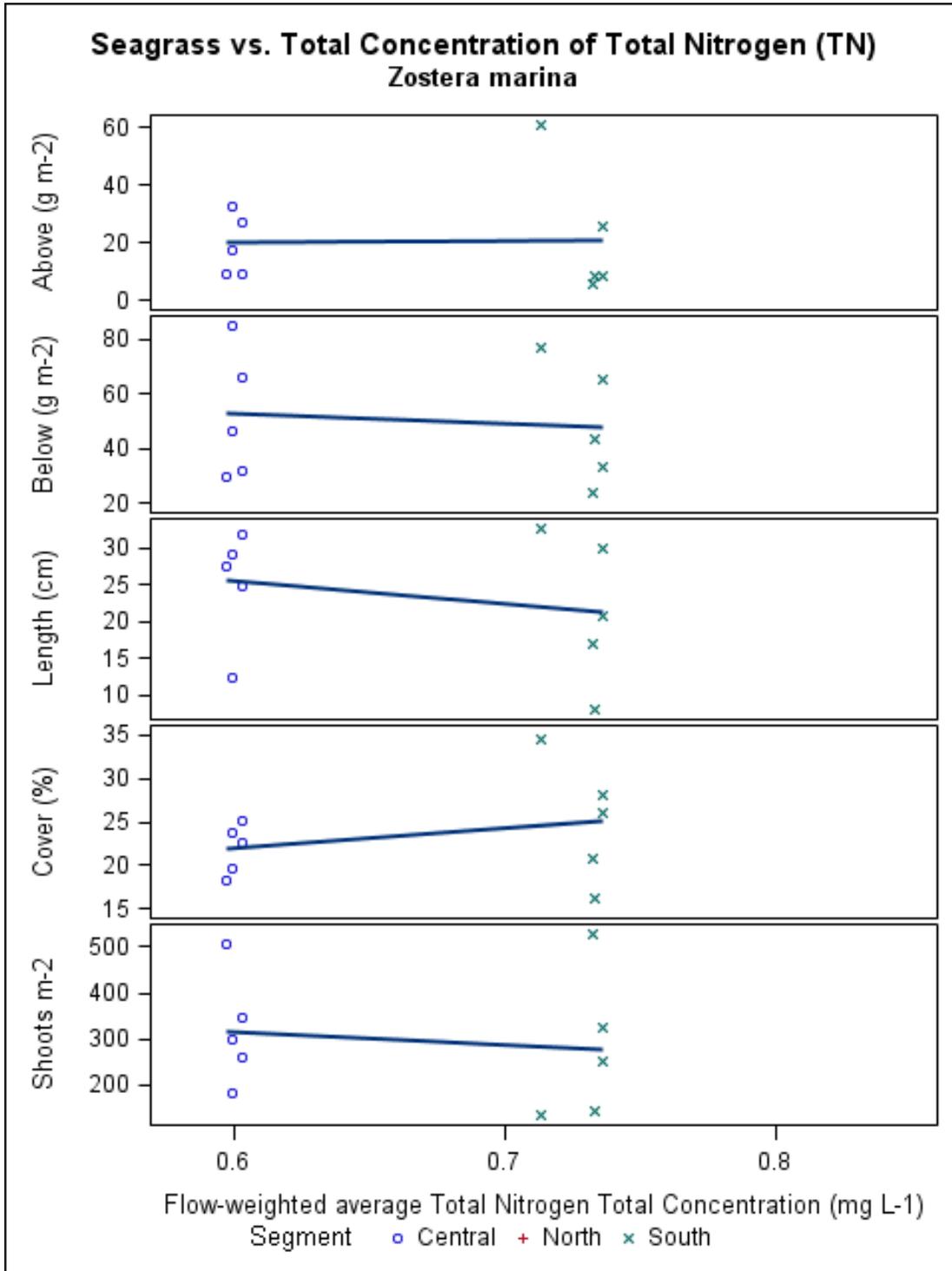
Figure 3 - 6 Zostera marina indicators vs. flow-weighted average total concentration of total nitrogen (mg L^{-1}).
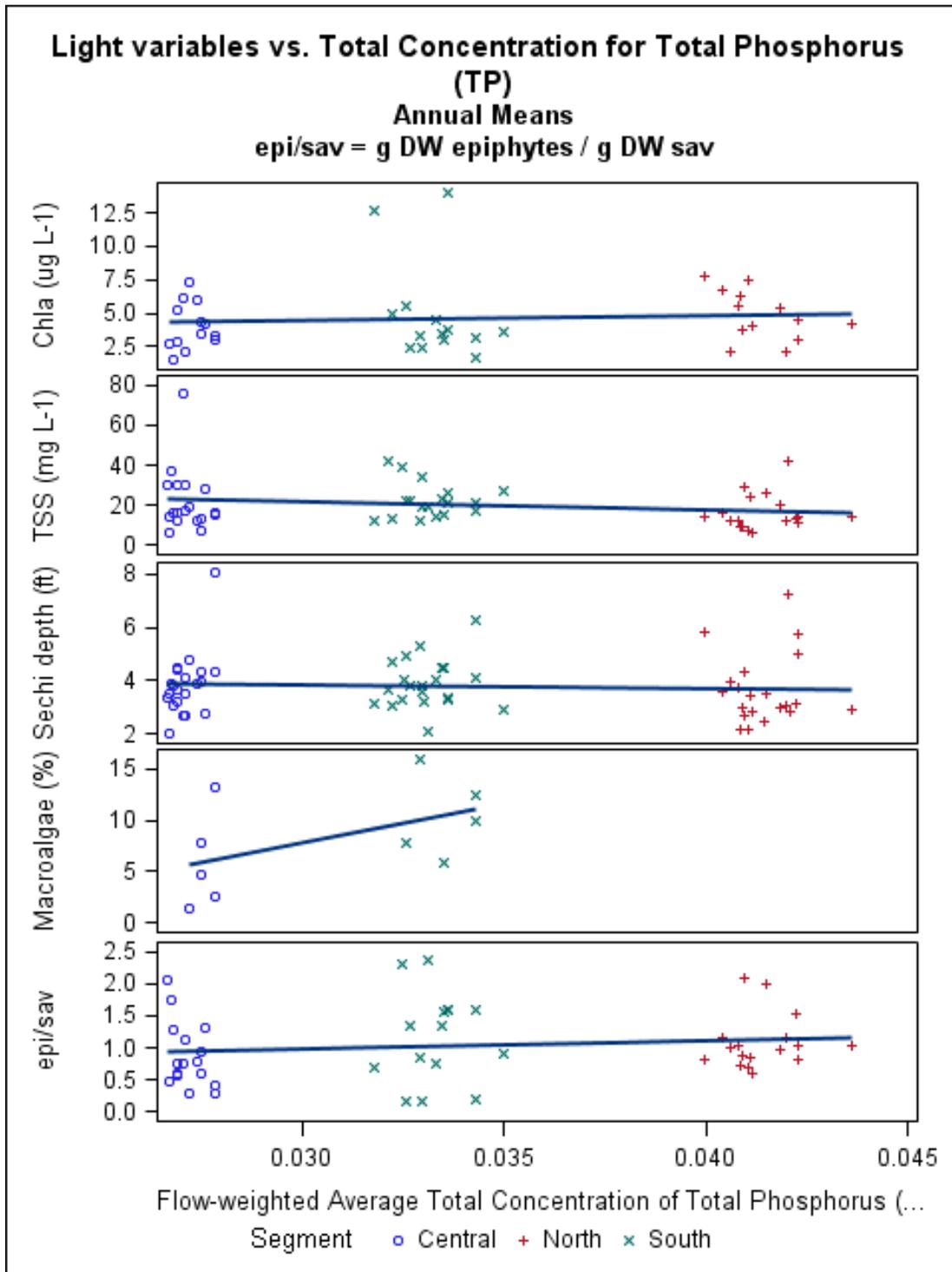
Figure 3 - 7 Light indicators vs. flow-weighted avererage total concentration of total phosphorus (mg L⁻

Seagrass vs. Total Concentration of Total Phosphorus (TP)
Zostera marina
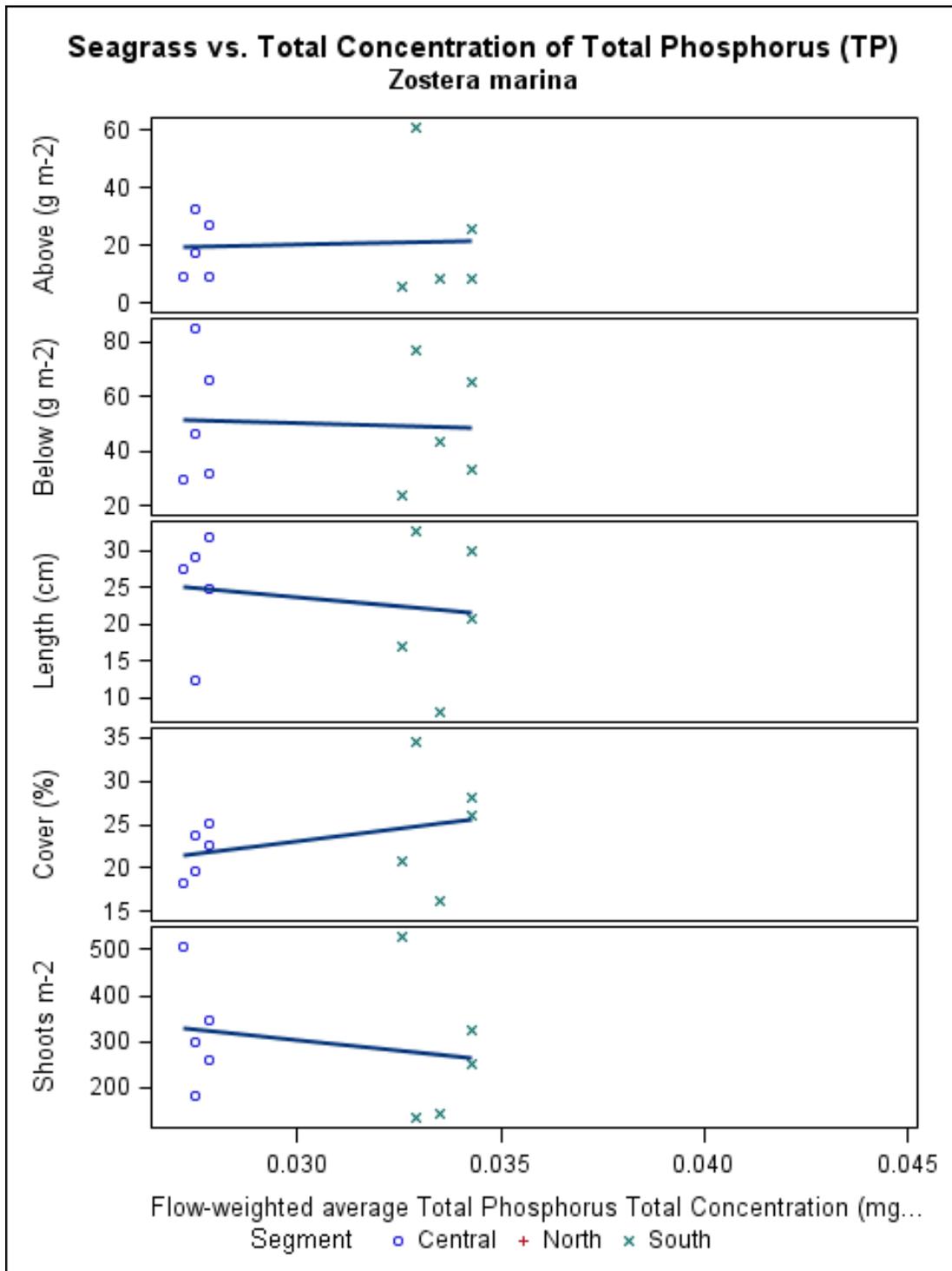
Figure 3 - 8 Zostera marina indicators vs. flow-weighted averaged total concentration of total phosphorus (mg L⁻¹).
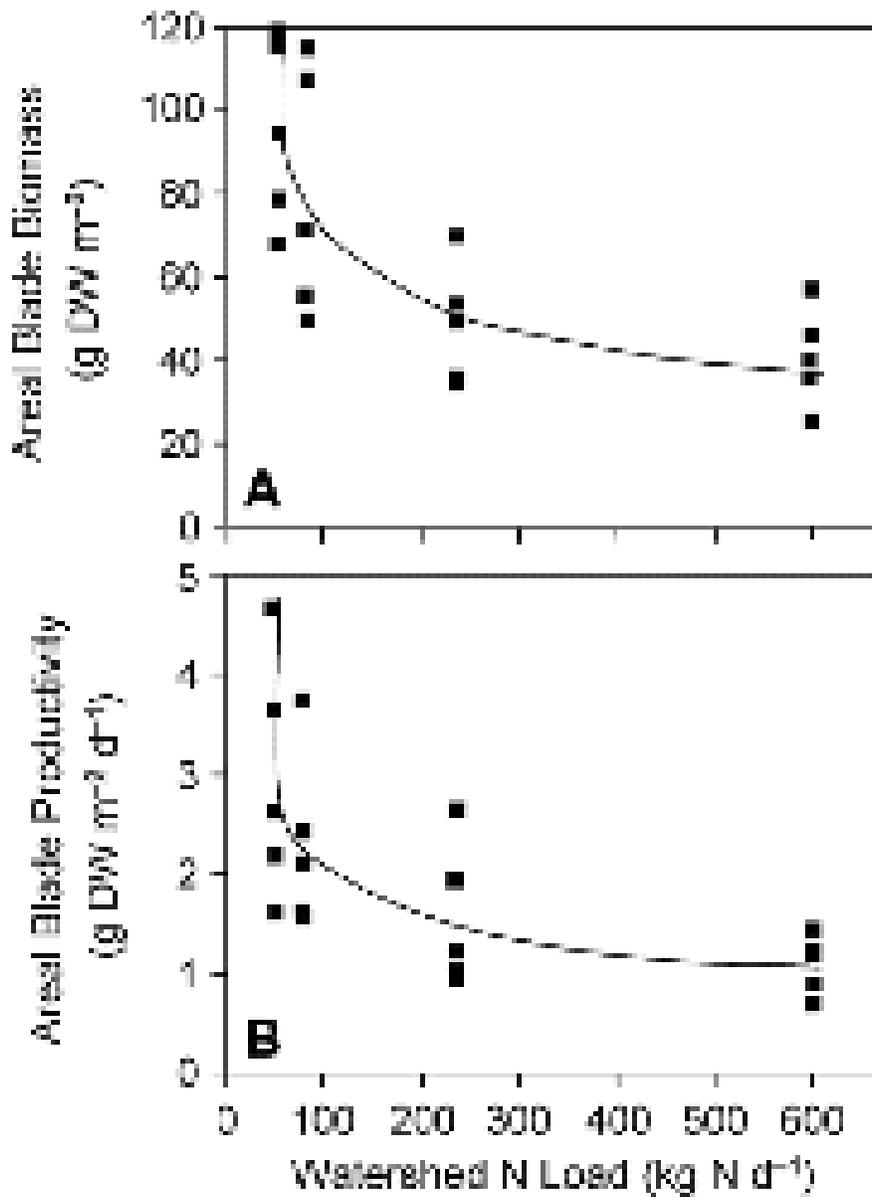
Fig. 7. (A) Areal blade biomass and (B) areal blade productivity plotted against watershed nitrogen loads for *Thalassia testudinum* from four sites in Sarasota Bay. Line is best-fit relationship. Modified from Tomasko et al. (1996).

Figure 3 - 9 Impact of nitrogen loading on seagrass biomass and productivity. (From Tomasko et al.
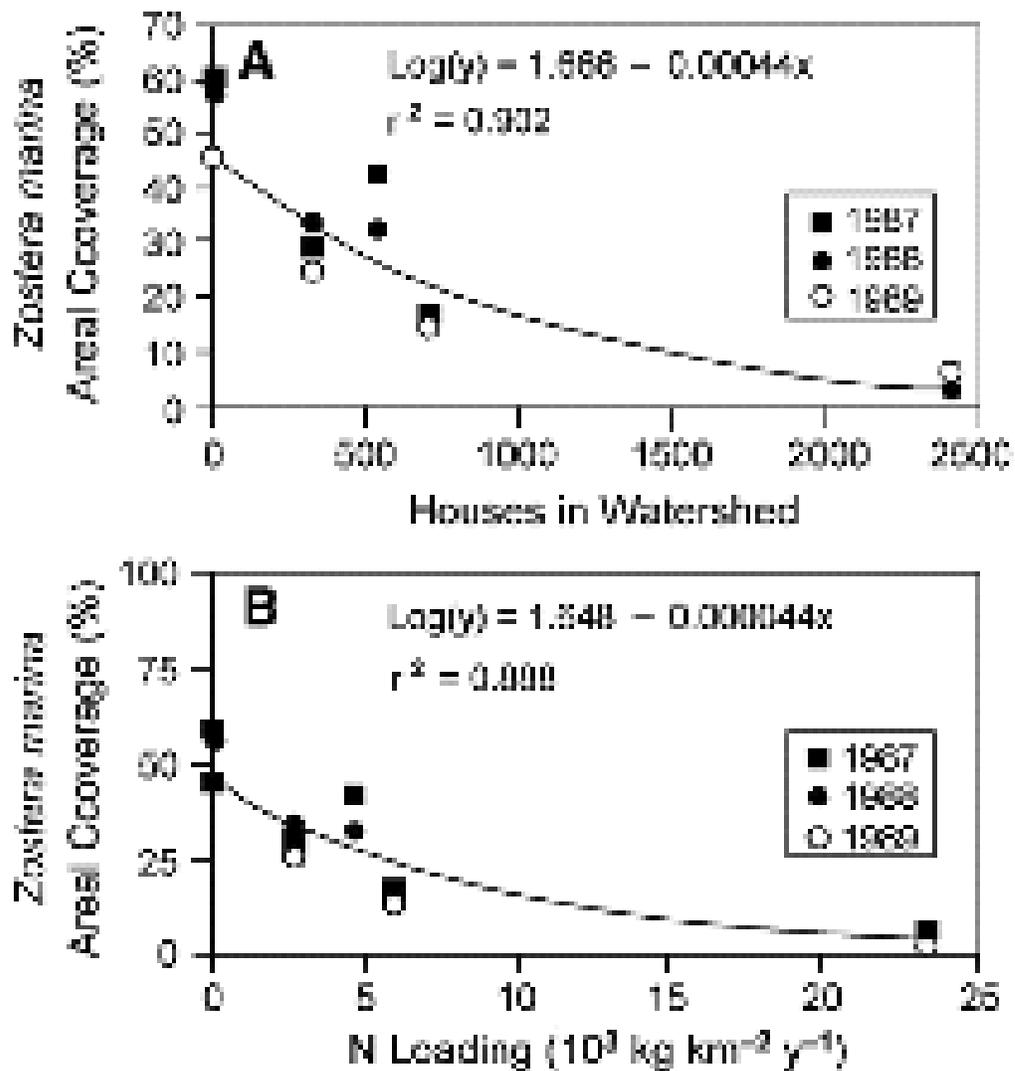
1996).

Fig. 11. Relationship between seagrass areal coverage (log of eelgrass area) in Waquoit Bay during 1987–1989 and (A) the number of houses in the sub-watersheds, and (B) nitrogen loading. From Short and Burdick (1996), with permission from the publisher; note that loading was estimated based on watershed area.

Figure 3 - 10 Losses of seagrass areal coverage with increasing nitrogen loading. (From Short and Burdick 1996).
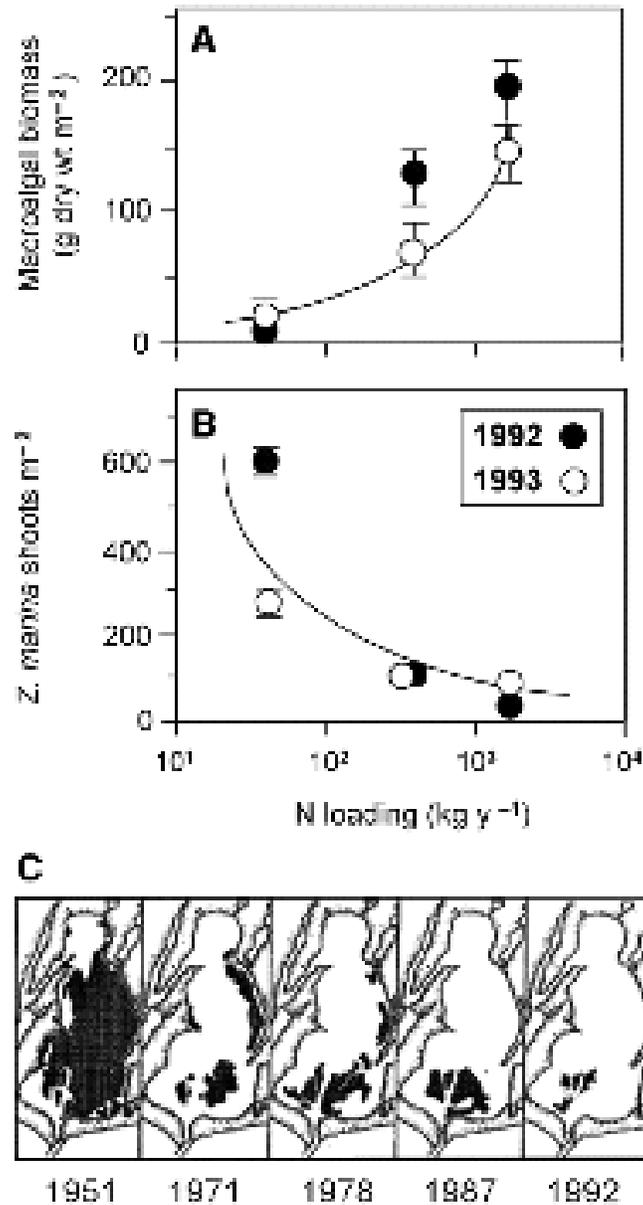
Fig. 2. Changes in plant abundance as (A) macroalgae (biomass) and (B) *Zostera marina* (shoot number) in response to N enrichment (modified from Deegan 2002; note that loading was estimated based on area of open water). (C) Change in spatial location and patch size of *Z. marina* distribution in Waquoit Bay in response to nutrient enrichment. From Valiela et al. (2000), with permission from the publisher.

Figure 3 - 11 Increase of macroalgae and decline of seagrass shoot density and areal coverage with increasing nitrogen loading. (From Burkholder et al. 2007).
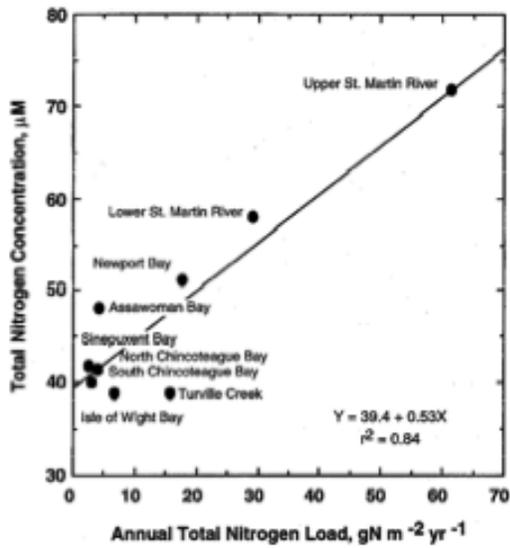
Fig. 7. A scatter plot relating annual areal total nitrogen loads to annual average total nitrogen concentrations for several regions of the Maryland coastal bays. Loading data are from Jacobs et al. (1993) and total nitrogen concentrations are from Fang et al. (1977a).
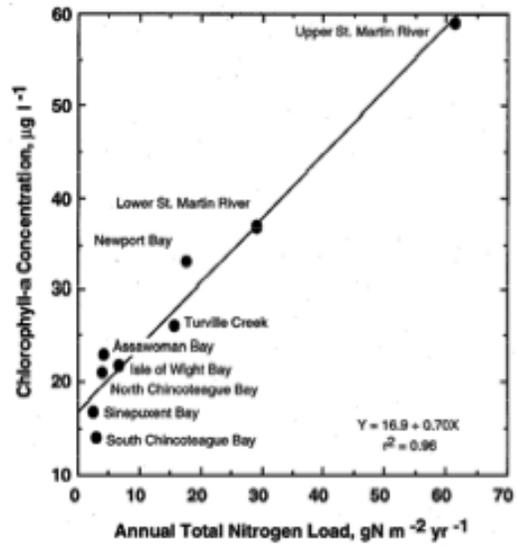
Fig. 8. A scatter plot relating annual areal total nitrogen loads to annual average chlorophyll *a* concentrations for several regions of the Maryland coastal bays. Loading data are from Jacobs et al. (1993) and total nitrogen concentrations are from Fang et al. (1977a).

Figure 3 - 12 Impact of nitrogen loading on estuarine total nitrogen and chlorophyll a concentrations in Maryland's coastal bays. (From Boynton et al. 1996).

a)

$y = 1.76x + 52.42$
$R^2 = 0.60 \quad p < 0.01$

Water Column Total Nitrogen (µg N L⁻¹)

b)

Zostera marina Nitrogen Content (%N)

c)

Nutrient Pollution Indicator (NPI)

Areal Normalized Subwatershed Total Nitrogen Loading
(kg TN km⁻² yr⁻¹)
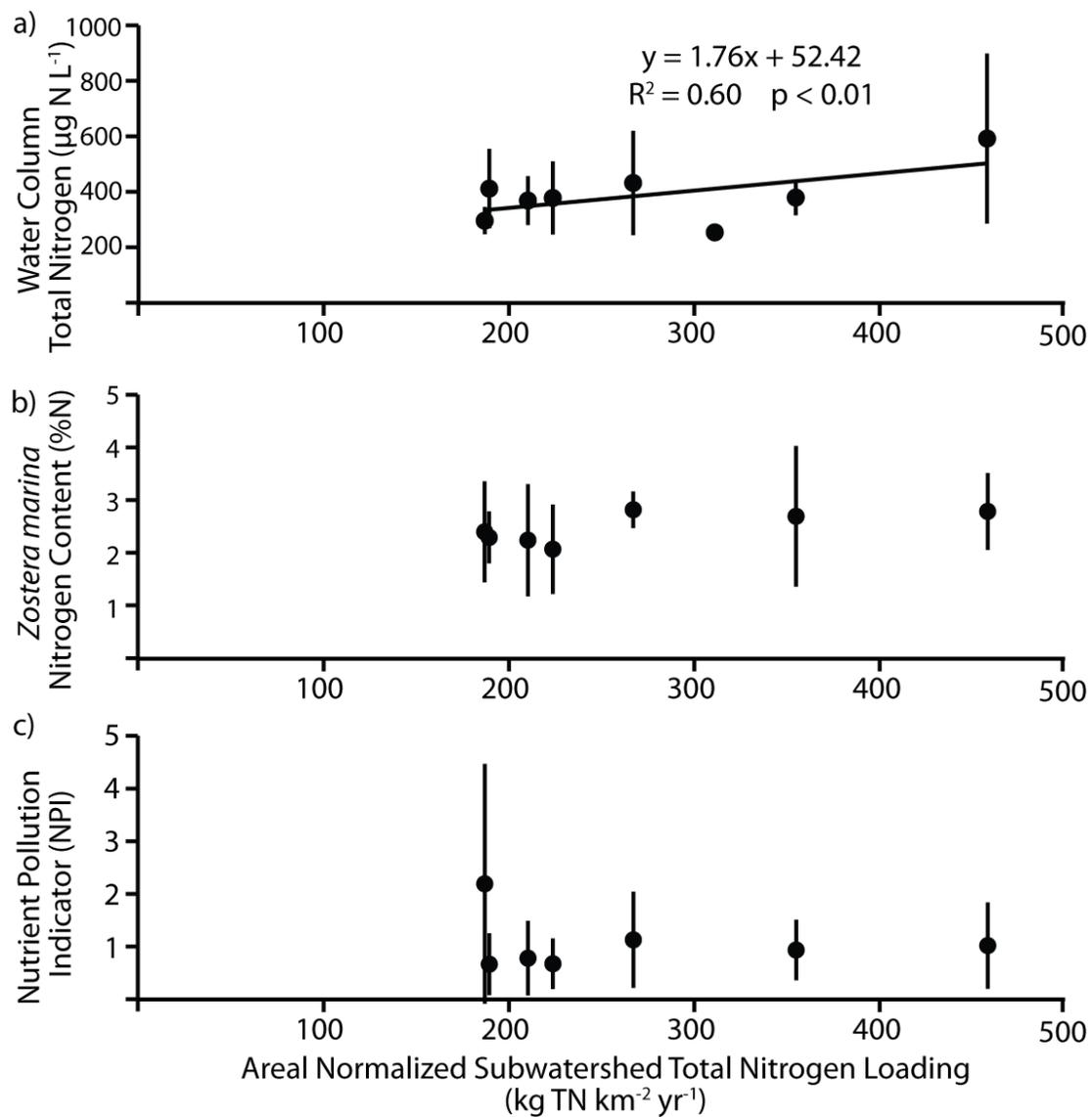
Figure 3 - 13 Estuarine nitrogen concentration, eelgrass nitrogen content, and a 'Nutrient Pollution Indicator' vs nitrogen loading in Barnegat Bay-Little Egg Harbor. (From Kennish and Fertig 2012).

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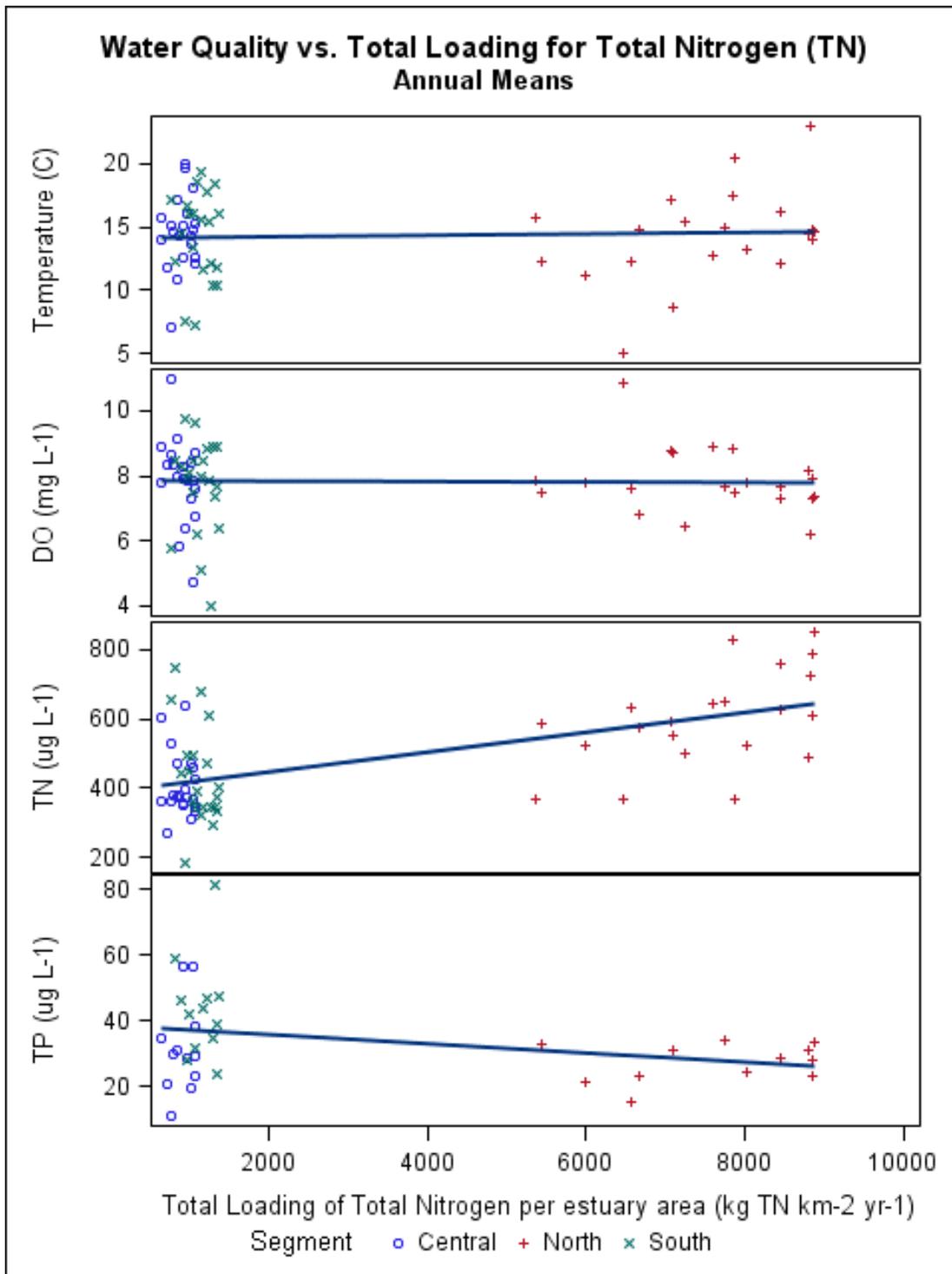
Figure 3 - 14 Water quality indicators vs. total nitrogen loading.
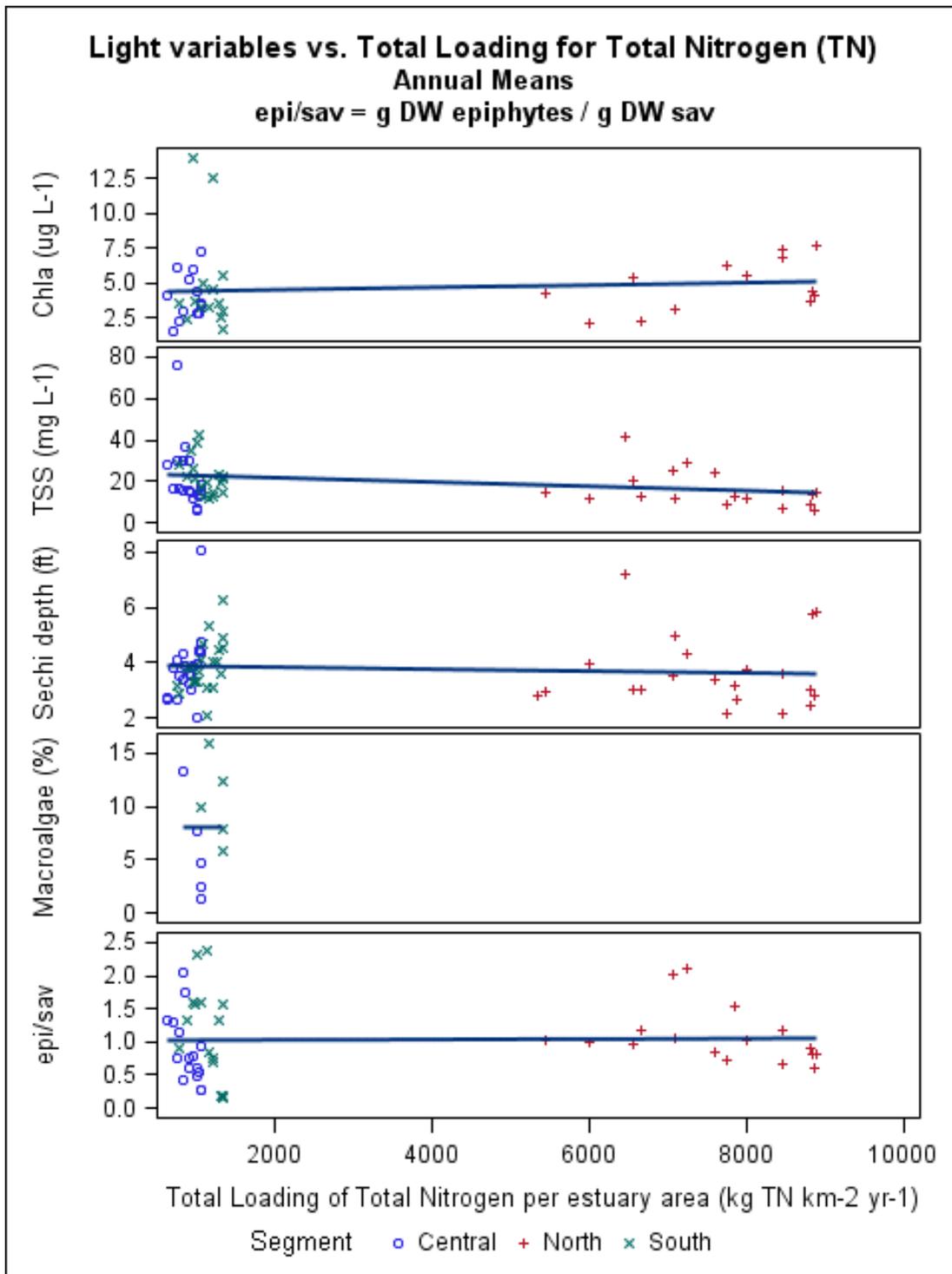
Figure 3 - 15 Light availability indicators vs. total nitrogen loading.

Seagrass vs. Total Loading of Total Nitrogen (TN)
Zostera marina
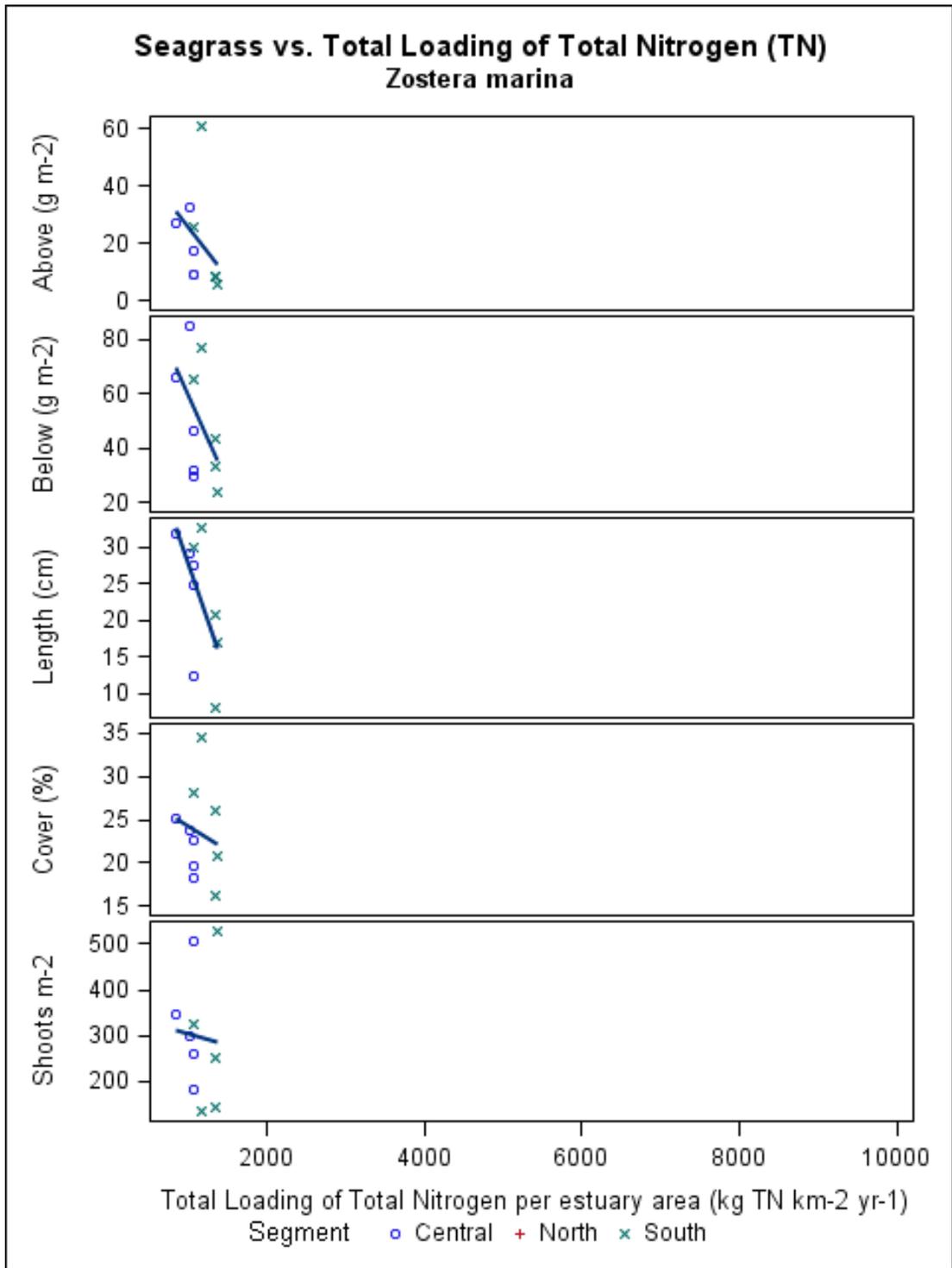
Figure 3 - 16 Seagrass indicators vs. total nitrogen loading.
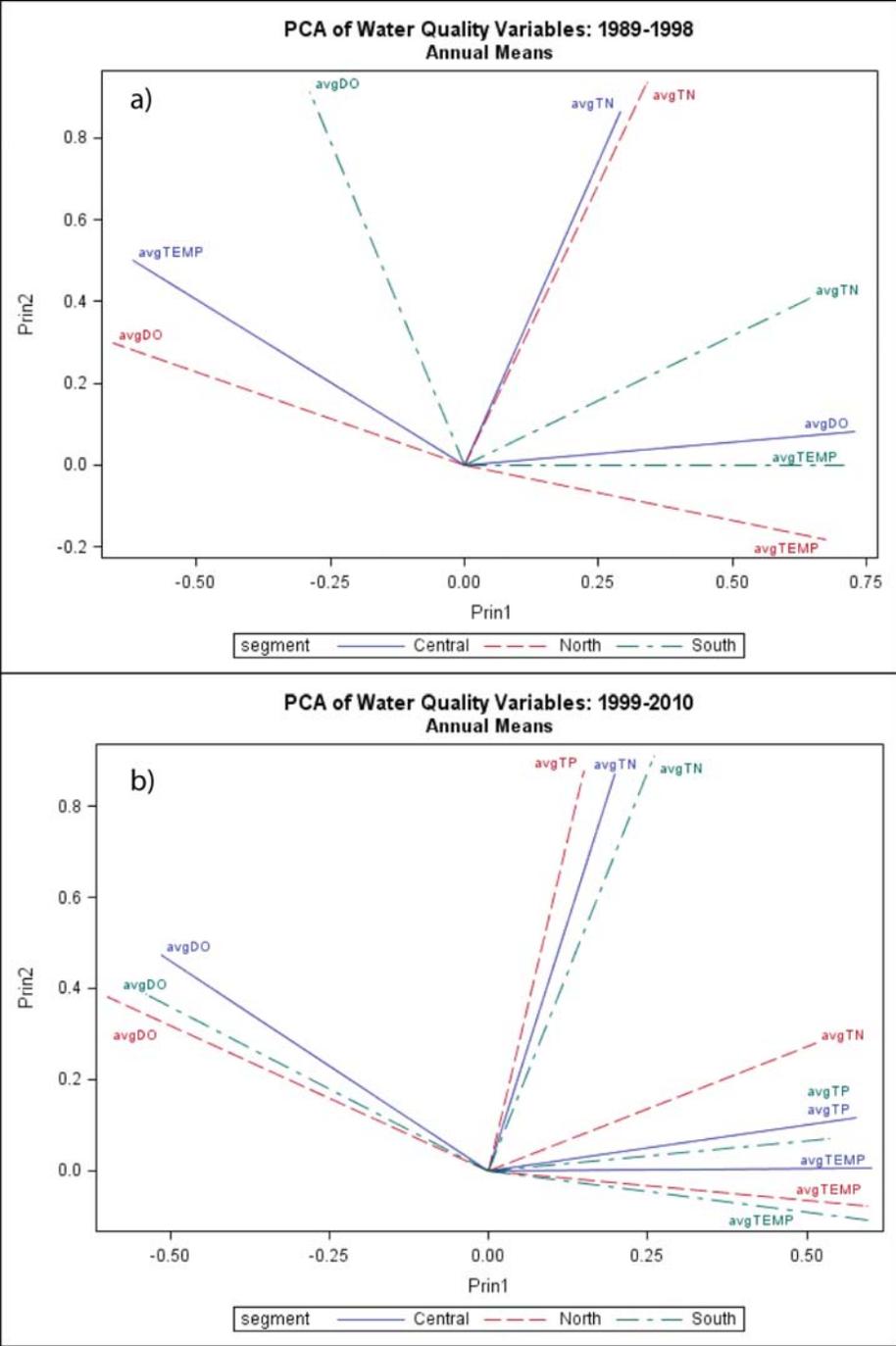
Figure 3 - 17 Principle component analysis of Water Quality indicators.
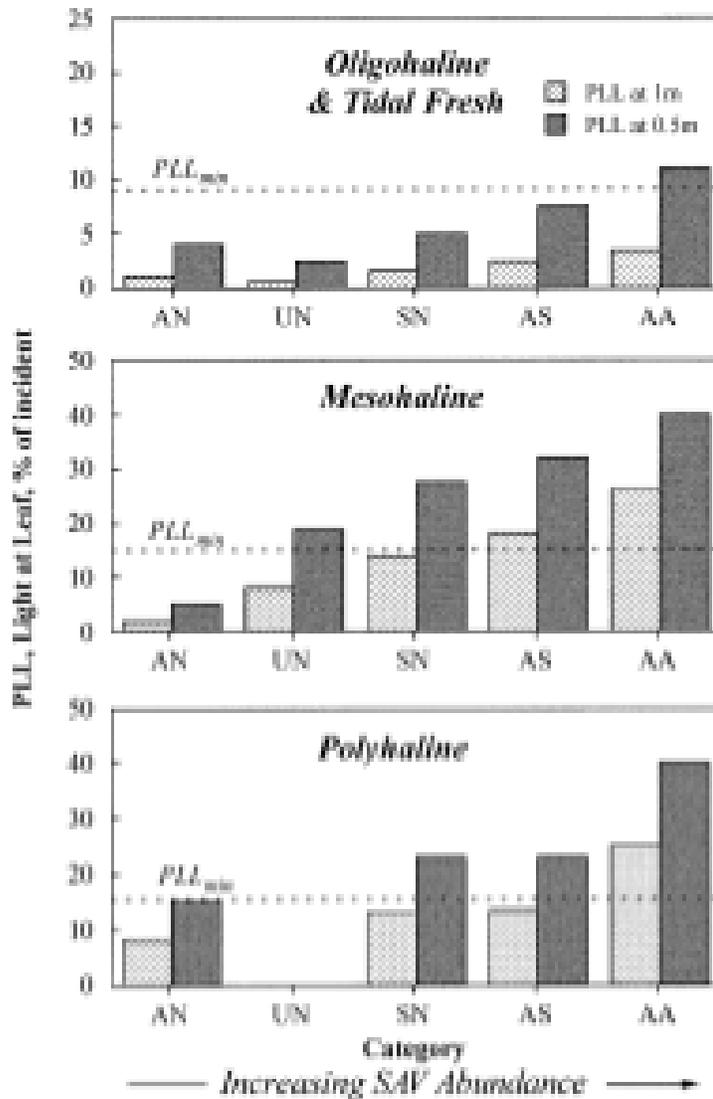
Fig. 8. Percent of surface light at SAV leaves (PLL) calculated using growing season median water quality data collected in Chesapeake Bay monitoring program at stations throughout the estuary compared to relative abundance of SAV in adjacent shallows in 1985–1996. PLL is calculated for water column depths of both 1 m (light bars) and 0.5 m (dark bars). Categories of SAV abundance (AN, always none; UN, usually none; SN, sometimes none; AS, always some; AA, always abundant) are defined in text.

Figure 3 - 18 Relationship between light availability and seagrass abundance. (From Kemp et al. 2004).
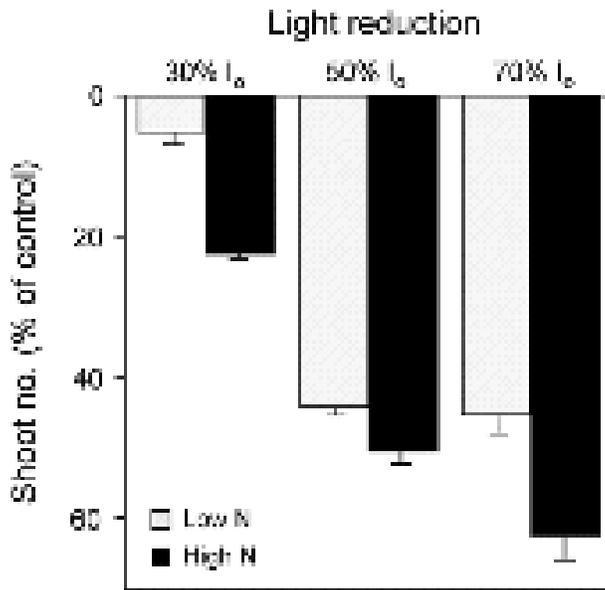
Fig. 5. The effects of water-column nitrate enrichment and light reduction on shoot production of the seagrass, *Zostera marina*. From first author's outdoor mesocosm experiments, indicated as the percent decrease from shoot production of control plants that did not receive water-column nitrate additions or light reduction (except that plants in controls and treatments all received an additional 30% light reduction for 3 h at 0900, 1200, and 1500 h on a 3-day rotation using neutral density screens to simulate conditions during high tide). Treatments were imposed for 10 weeks during the fall growing season for *Z. marina*. Controls were maintained at ambient natural light (except during simulated high tide) and nitrate (<2.15 µM or 30 µg NO_3^- NL^{-1}). Treatments included low N (at 3.57 µM or 50 µg NO_3^- L^{-1}), added daily as a pulse of enrichment and high N (at 7.14 µM or 100 µg NO_3^- NL^{-1}) at each of three imposed light levels as 30, 50, or 70% reduction at ambient surface light (I_o, accomplished using neutral density shades, with additional shading at simulated high tide as noted). *Z. marina* in all treatments with water-column nitrate enrichment declined in shoot production relative to shoot production of control plants, and the nitrate inhibition effect was exacerbated at lower irradiance (means + 1 standard error; $P < 0.05$, $n = 3$). These effects were not caused by algal overgrowth, which was maintained at low levels in controls and all treatments throughout the experiment. From Burkholder (2001), with permission from the publisher.

Figure 3 - 19 Impact of nitrate enrichment and light reduction on seagrass shoot production. (From

Burkholder 2001).

Table 1. Maximal depth limit, light attenuation coefficient (K_d), and minimal light requirements of various species of seagrass. Where Secchi depths were reported, K_d = 1.65/Secchi depth (Giesen et al. 1990). Minimal light requirements were calculated as percent light at the maximal depth limit using $100 \times I_z/I_o = e^{-K_d \cdot z}$. Range of maximal depth limit and K_d values and means $\pm$ SE of minimal light requirement given in locations with multiple data points.

Genus and species	Location	Maximal depth limit (m)	K_d light attenuation coefficient (m^{-1})	Minimal light requirement (%)
Amphibolis antarctica[*]	Waterloo Bay (Australia)	7.0	0.20	24.7
Cymodocea nodosa[*]	Ebro Delta (Spain)	4.0	0.57	10.2
C. nodosa[*]	Malta	38.5	0.07	7.3
Halodule wrightii[†]	Florida (US)	1.9	0.93	17.2
Halophila decipiens[‡]	St. Croix (US)	40.0	0.08	4.4
H. decipiens[*]	Northwest Cuba	24.3	0.10	8.8
Halophila engelmanni[*]	Northwest Cuba	14.4	0.10	23.7
Heterozostera tasmanica[*]	Victoria (Australia)	3.8–9.8	0.36–0.85	5.0 ± 0.6
H. tasmanica[*]	Chile	7.0	0.25	17.4
H. tasmanica[*]	Spencer Gulf (Australia)	39.0	0.08	4.4
H. tasmanica[*]	Waterloo Bay (Australia)	8.0	0.20	20.2
Posidonia angustifolia[*]	Waterloo Bay (Australia)	7.0	0.20	24.7
Posidonia oceanica[*]	Medas Island (Spain)	15.0	0.17	7.8
P. oceanica[*]	Malta	35.0	0.07	9.2
Posidonia ostenfeldii[*]	Waterloo Bay (Australia)	7.0	0.20	24.7
Posidonia sinuosa[*]	Waterloo Bay (Australia)	7.0	0.20	24.7
Ruppia maritima[*]	Brazil	0.7	3.57	8.2
Syringodium filiforme[*]	Northwest Cuba	16.5	0.10	19.2
S. filiforme[*]	Florida (US)	6.8	0.25	18.3
S. filiforme[†]	Florida (US)	1.9	0.93	17.2
Thalassia testudinum[*]	Northwest Cuba	14.5	0.10	23.5
T. testudinum[*]	Puerto Rico	1.0–5.0	0.35–1.50	24.4 ± 4.2
T. testudinum[*]	Florida (US)	7.5	0.25	15.3
Zostera marina[§]	Kattegat (Denmark)	3.7–10.1	0.16–0.36	20.1 ± 2.1
Z. marina[*]	Roskilde (Denmark)	2.0–5.0	0.52–0.92	19.4 ± 1.3
Z. marina[*]	Denmark	1.5–9.0	0.22–1.21	20.6 ± 13.0
Z. marina[*]	Woods Hole (US)	6.0	0.28	18.6
Z. marina[*]	Netherlands	2.5	0.49	29.4
Z. marina[*]	Japan	2.0–5.0	0.38–0.49	18.2 ± 4.5

[*]Duarte 1991.
[†]W. J. Kenworthy, personal communication, 1990.
[‡]Williams and Dennison 1990.
[§]Ostenfeld 1908.
[‖]Borum 1983.

Figure 3 - 20 Physiological light requirements for seagrass species. (From Dennison et al. 1993).

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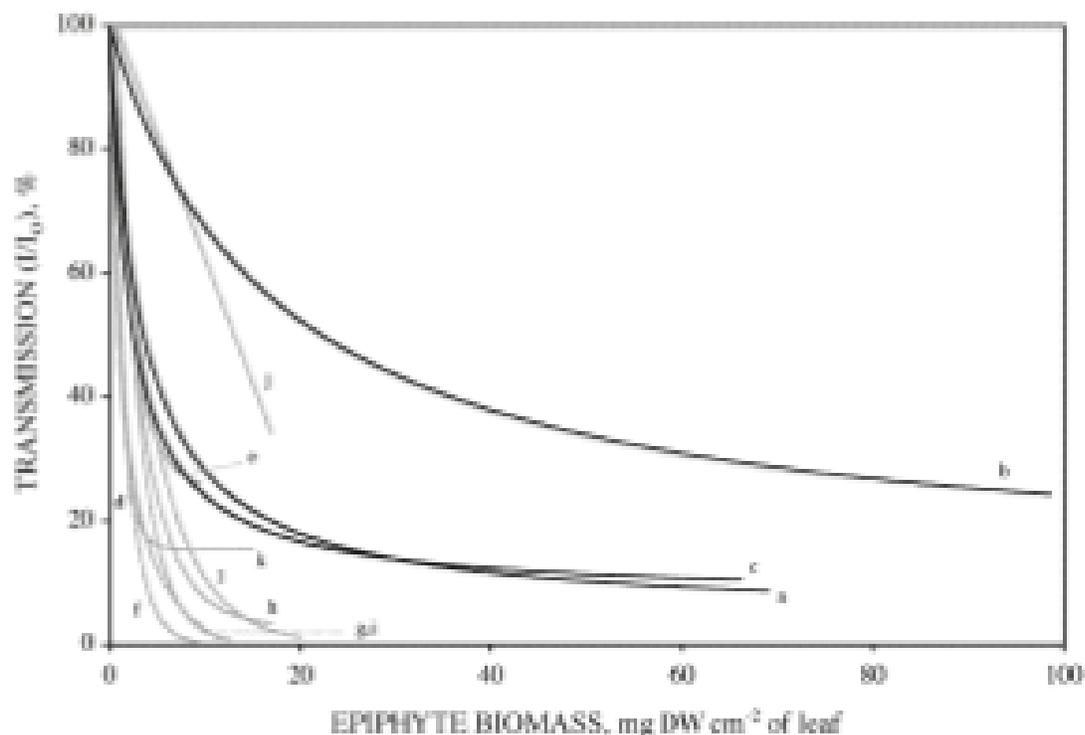
Fig. 2. Comparison of relationships between light transmission and epiphyte density in this and previous studies. Regression lines for present study are black, and denoted as follows: a, unidentified green; b, *Cladophora* sp.; c, *Polysiphornia* sp. Regression lines for past studies are gray, and are denoted as follows: d, Bulthuis & Woelkerling (1983); e, Murray (1983); f, Sand-Jensen & Borum (1983); g, Twilley et al. (1985); h, Kemp et al. (1988); i, van Dijk (1993); j, Glazer (1990); k, Stankelis et al. (1999); l, Neckles (unpubl. data). Sand-Jensen & Borum (1983) used the data of Broum & Wium-Andersen (1980). Equations for Murray (1983) and Kemp et al. (1988) were estimated from graphs of their results. Regressions were plotted only across the range of epiphytic densities used in each study (see Table 1)

Figure 3 - 21 Light attenuation due to epiphytes. (From Brush and Nixon 2002).
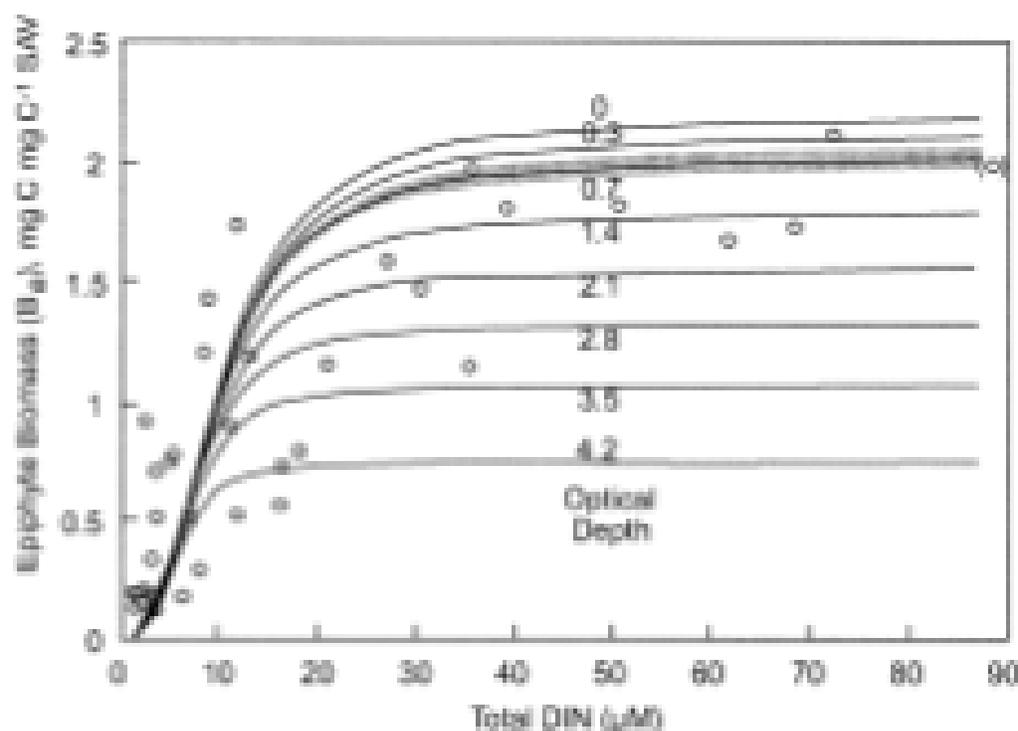
Fig. 5. Calculated responses of epiphytic algal biomass (B_e, mg C/mg C SAV) to changes in dissolved inorganic nitrogen (DIN, µM) concentration under various light (PAR, µE m^{-2} s^{-1}) conditions in estuarine waters. Each curve is described by the equation, and each represents computed response under different light regimes, characterized by the dimensionless optical depth (OD = K_dZ). These curves, which are described by (B_e = ($B\ B_e)_m$ [1 + 208 (DIN$^{-kN(OD)}$]$^{-1}$ (where ($B_e)_m$ = 2.2−[0.251 (OD$^{1.27}$)] and $K_{N(OD)}$ = 2.32(1 − 0.031OD$^{1.62}$)$^{-1}$), were generated from numerical model calculations (modified from Bartleson 1988) assuming constant biomass of host SAV plant over the growth season (May–August). The model was calibrated to data (open circles) from mesocosm studies (Murray unpublished data) for experimental light conditions (shaded area). Equations were fit to model calculations using a statistical curve-fitting routine (Kemp et al. 2000). Similar functions are predicted for B_e versus dissolved inorganic phosphorus (DIP) concentrations, with DIP = DIN/16.

Figure 3 - 22 Relationship of epiphyte biomass to nitrogen concentrations under different light regimes. (From Kemp et al. 2004).

Seagrass vs. Chlorophyll a
Zostera marina, Annual Means
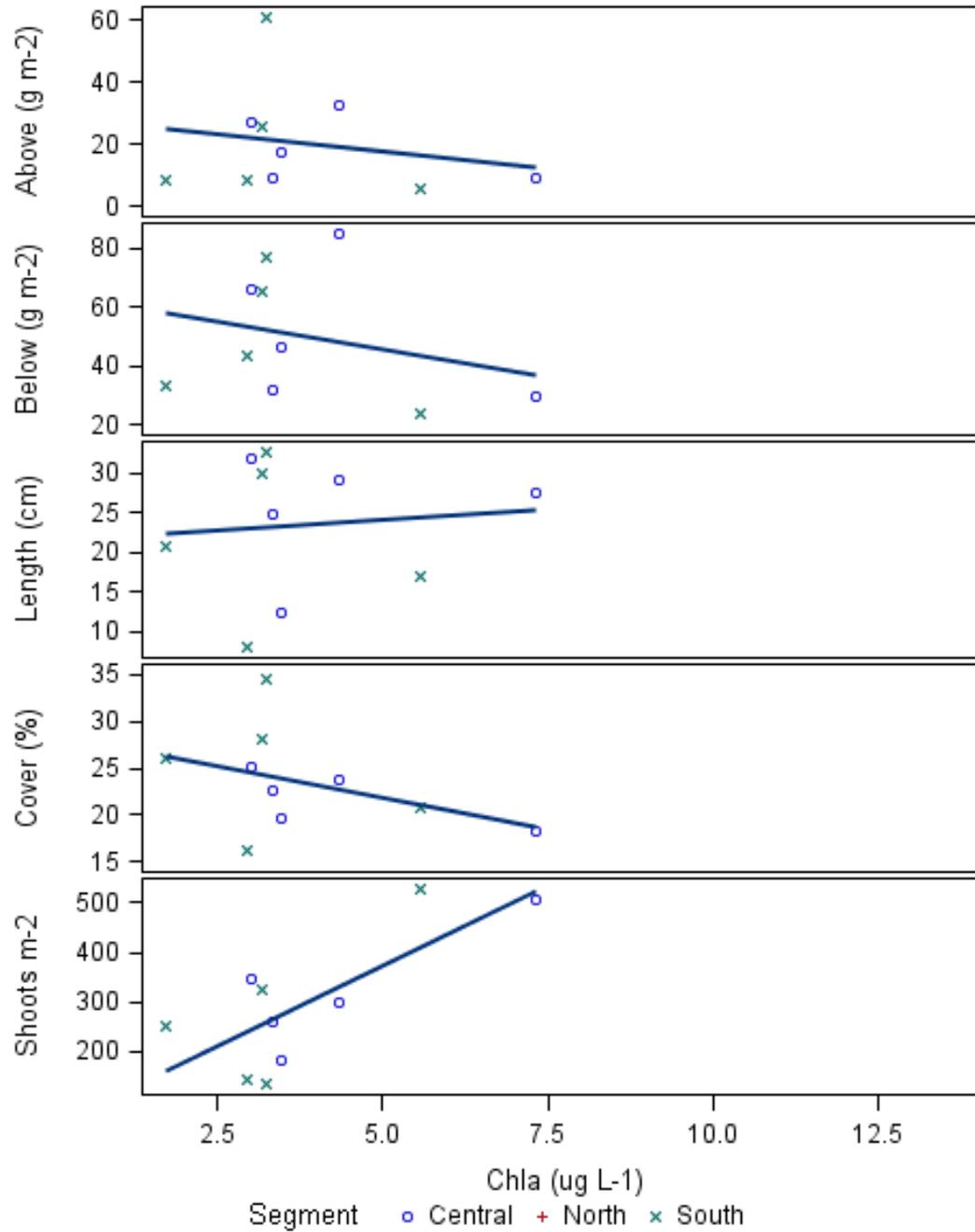
Figure 3 - 23 Seagrass indicator response to chlorophyll a.

Seagrass vs. Total Suspended Solids
Zostera marina, Annual Means
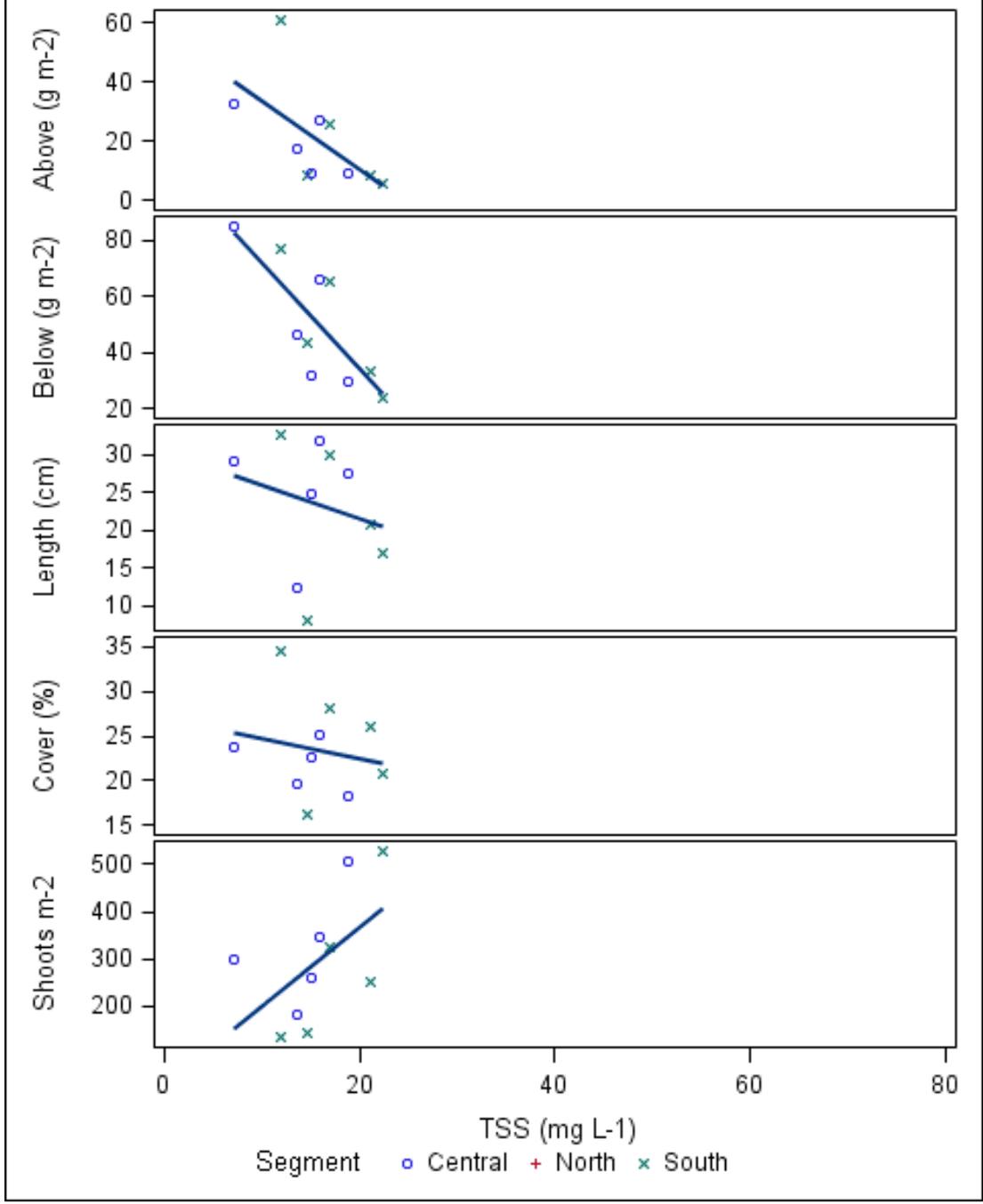
Figure 3 - 24 Seagrass indicator response to total suspended solids.
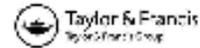
A Brown Tide Bloom Index based on the potential harmful effects of the brown tide alga, *Aureococcus anophagefferens*

Mary Downes Gastrich[1]* and Catherine E. Wazniak[2]

[1] New Jersey Department of Environmental Protection, Division of Science, Research and Technology, Trenton, New Jersey, USA

[2] Maryland Department of Natural Resources, Annapolis, Maryland, USA

*Corresponding author: New Jersey Department of Environmental Protection, Division of Science, Research and Technology, 401 E State St, Trenton, NJ 08625, USA; Tel.: (609)292-1895; Fax: (609)292-7340; E-mail: mary.downes-gastrich@dep.state.nj.us

Harmful algal blooms are an increasing phenomenon in coastal areas of the world. Recurring harmful brown tides caused by the minute alga, Aureococcus anophagefferens, are a regional problem in the northeast Atlantic states of the United States. Brown tide blooms may cause significant ecological impacts on natural resources. A Brown Tide Bloom Index was developed based on published scientific studies and agency reports that relates concentrations of the brown tide organism to potential negative impacts on natural resources including shellfish, seagrasses and protozoa. For the first time, the index provides terminology that can be used to convey accurate information about impacts to natural resources resulting from concentrations of brown tide to scientists, environmental managers and the public. The purpose of the Brown Tide Bloom Index is to provide a metric, based on available scientific studies, which can be used by environmental managers to communicate the magnitude of brown tide blooms and impacts to natural resources. The Brown Tide Bloom Index includes three categories of brown tide blooms: Category 1 blooms (algal concentrations at $<35{,}000$ cells ml^{-1}) have no reported impacts; Category 2 blooms ($\geq 35{,}000$ to $<200{,}000$ cells ml^{-1}) have potential negative impacts on feeding and growth in shellfish; Category 3 blooms ($\geq 200{,}000$ cells ml^{-1}), discolor the water a yellow-brown and may cause severe impacts and mortality on shellfish, reduction in seagrasses and planktonic organisms.

Keywords: harmful algal blooms, metric

Figure 3 - 25 An index of harmful algae blooms for coastal lagoons. (From Gastrich and Wazniak 2002).
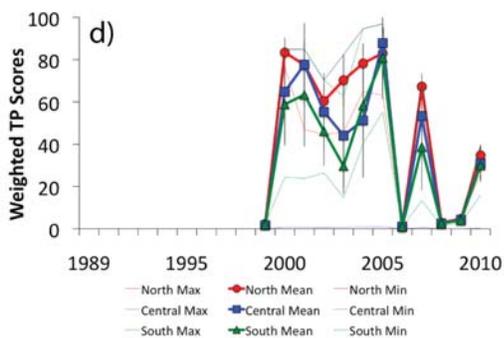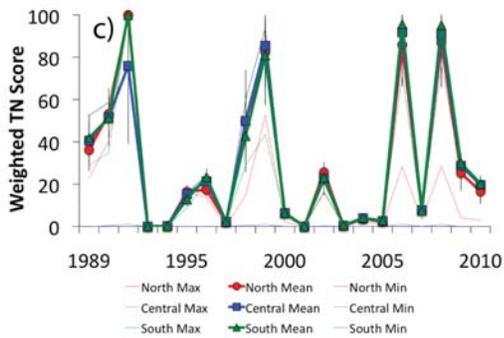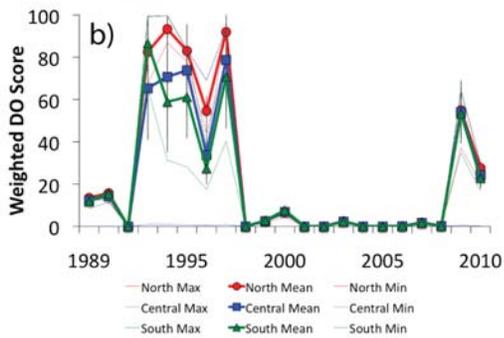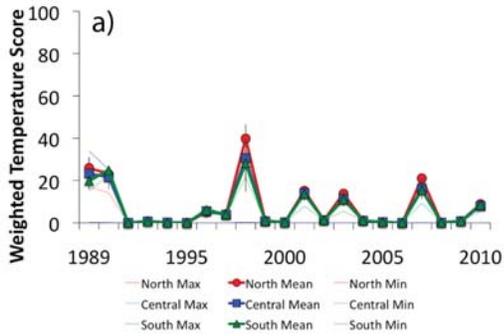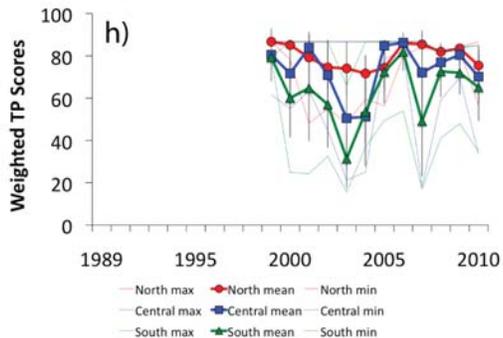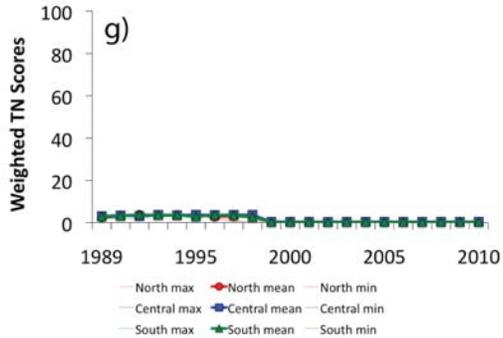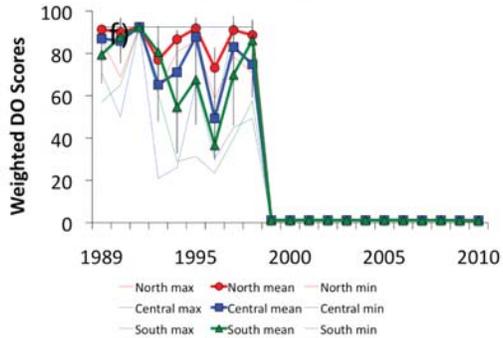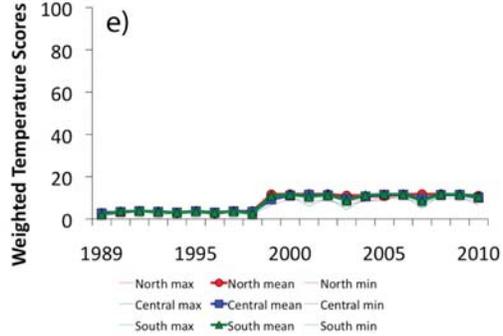
Figure 3 - 26 Weighted scores for water quality indicators under Scenario 1 and Scenario 2.

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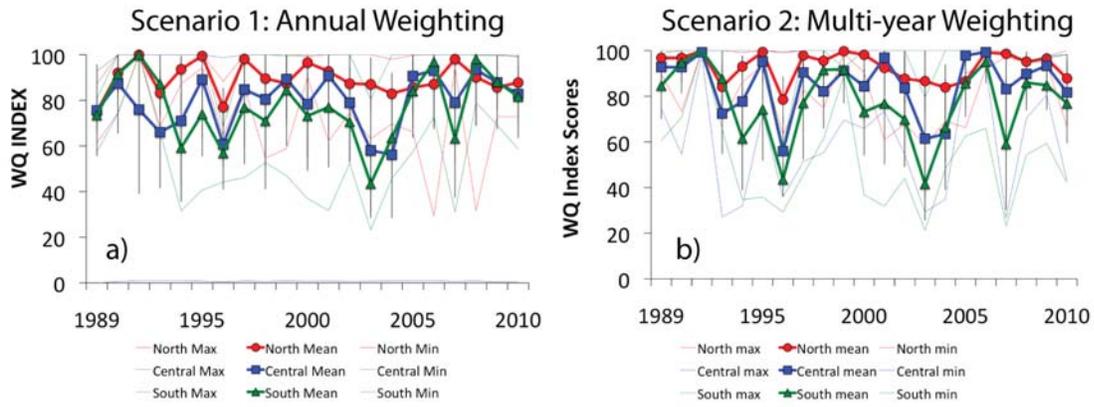
Figure 3 - 27 Weighted Scores under Scenario 1 and Scenario 2 for Water Quality Index.

PCA of Light Variables: 1998-2010
Annual Means
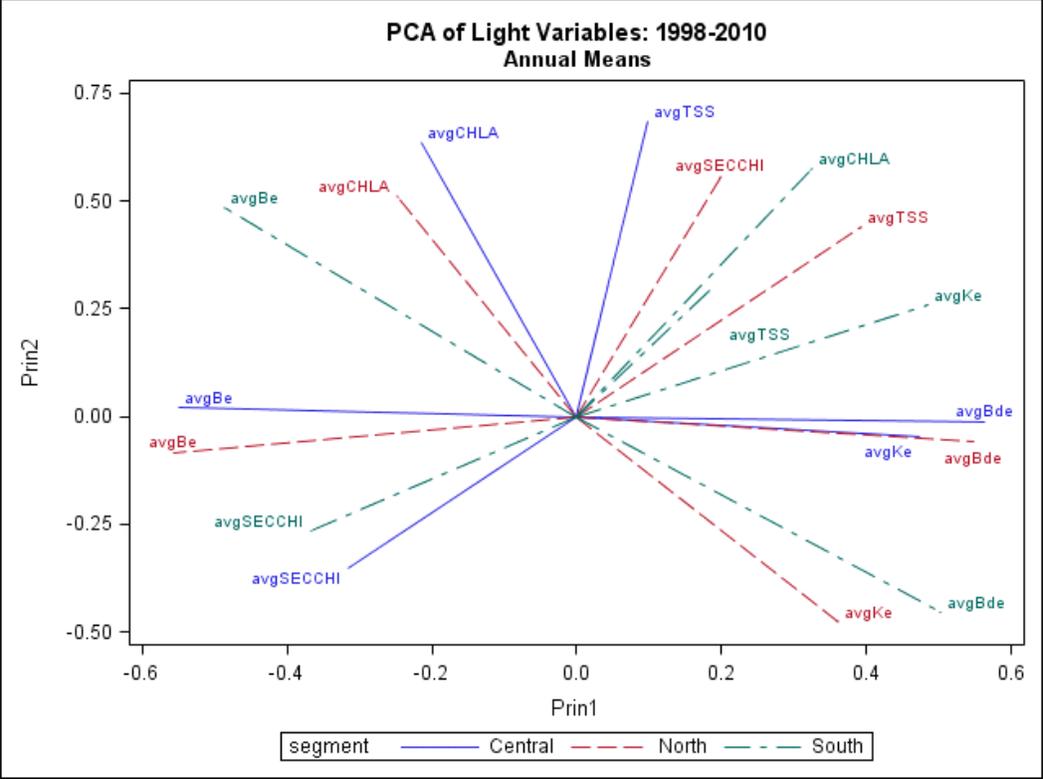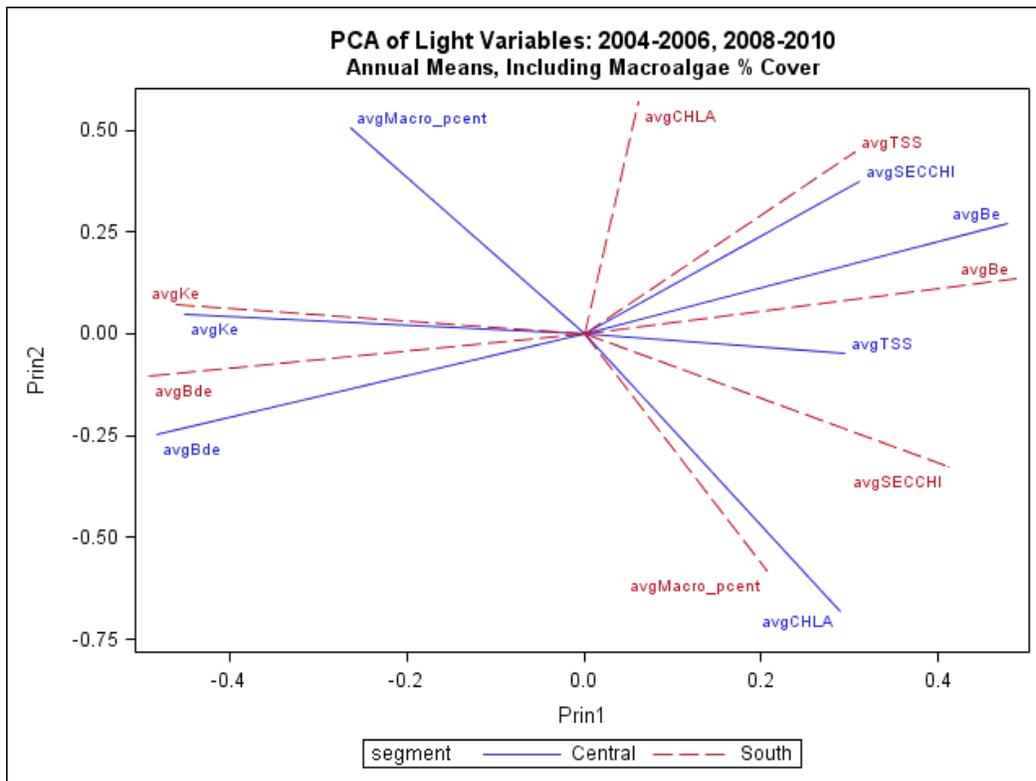
Figure 3 - 28 Principle component analysis of Light Availability indicators.
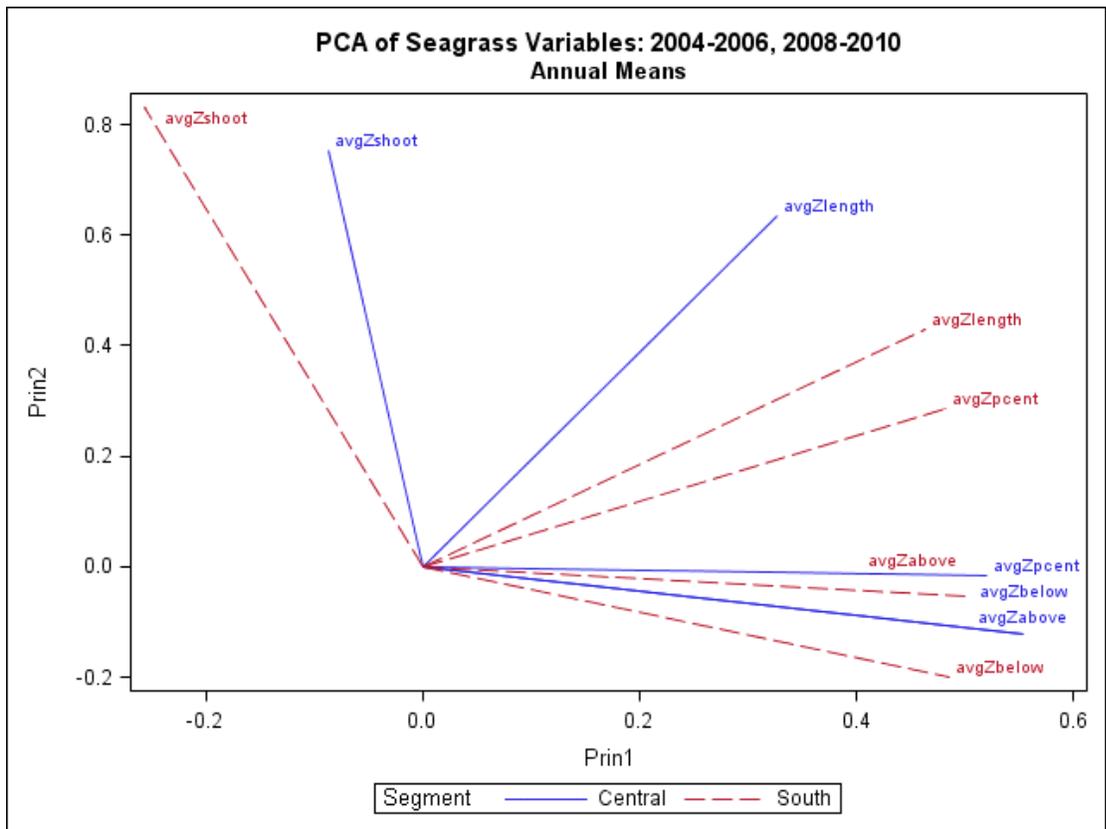
Figure 3 - 29 Principle component analysis of Seagrass indicators.
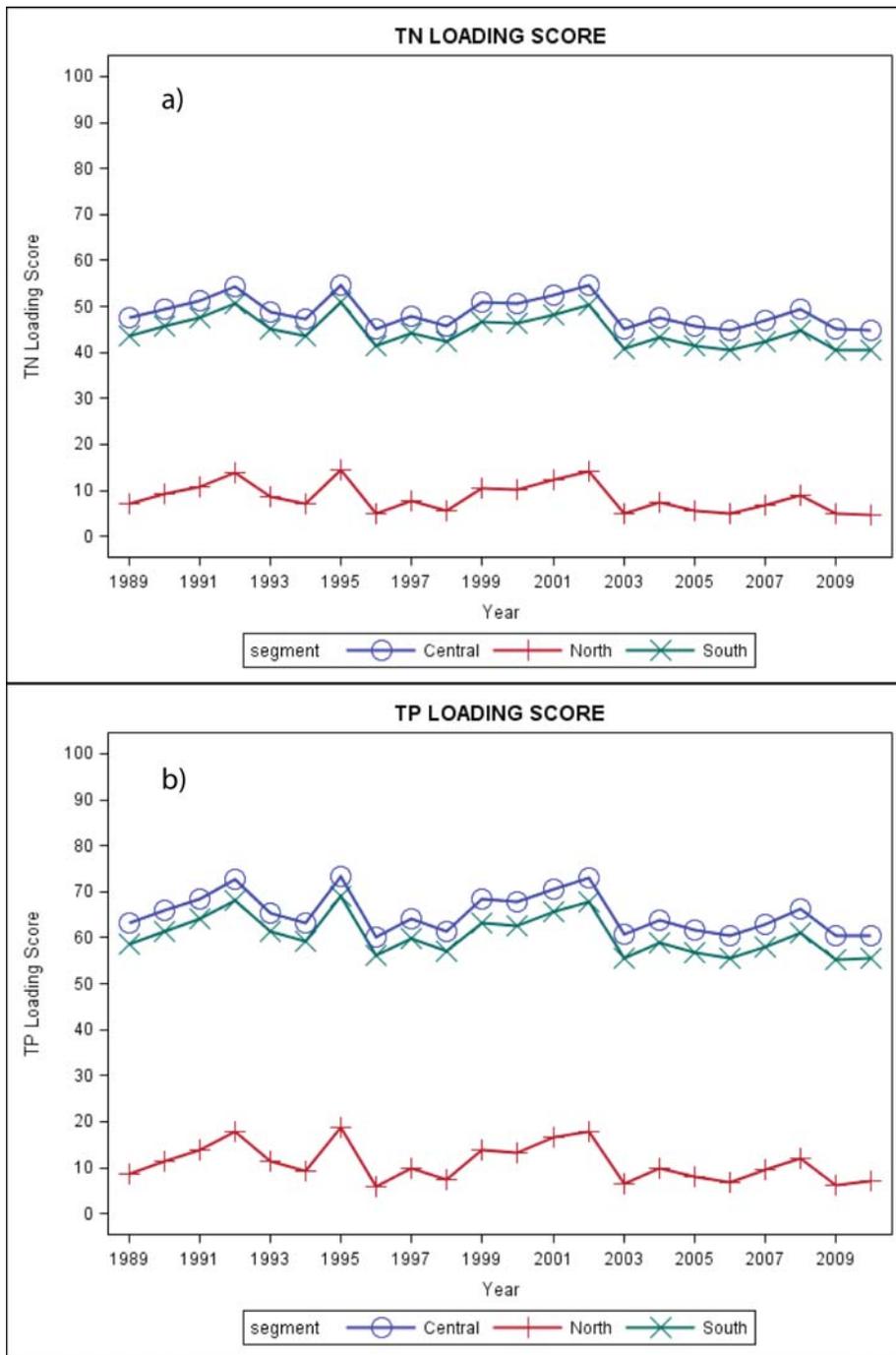
Figure 3 - 30 Raw Scores for total nitrogen loading and total phosphorus loading.
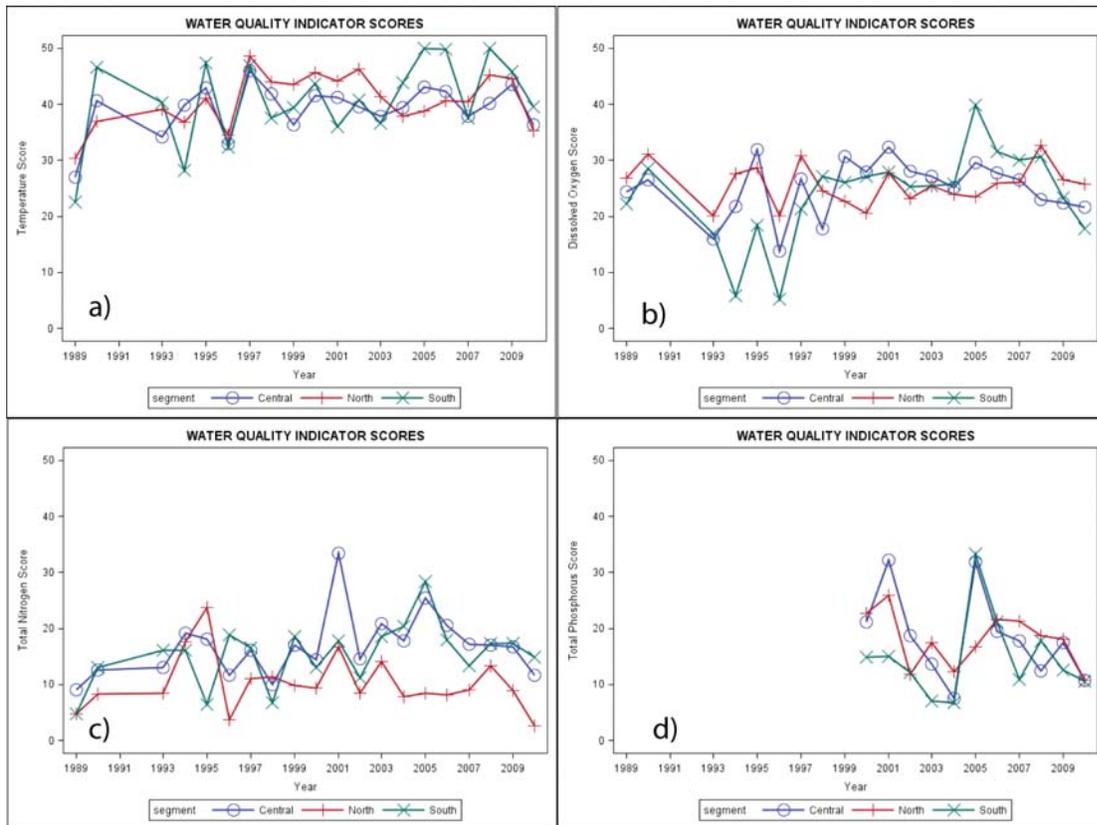
Figure 3 - 31 Scores for Water Quality indicators: (a) temperature, (b) dissolved oxygen, (c) total nitrogen, (d) total phosphorus.
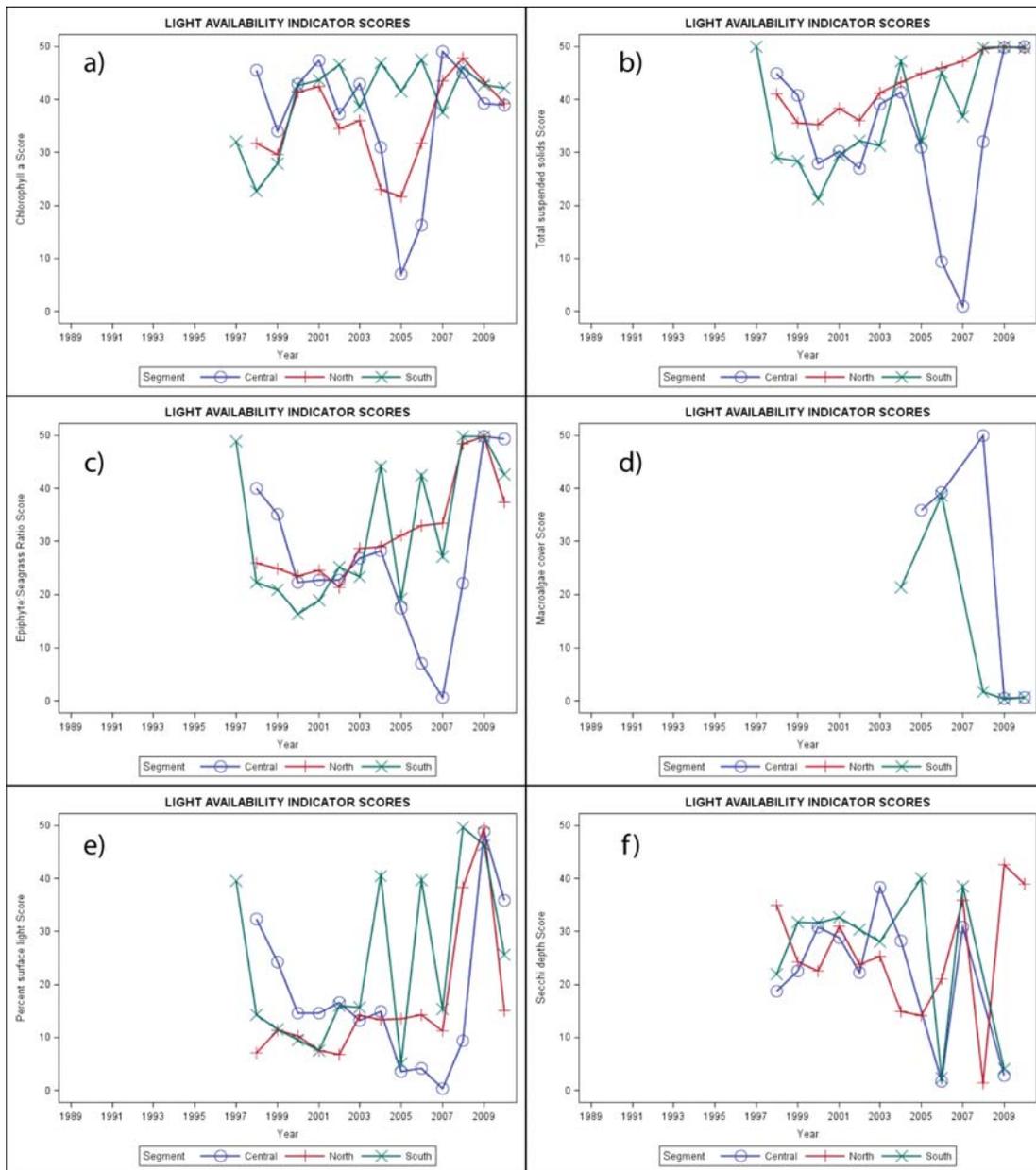
Figure 3 - 32 Scores of Light Availability indicators: (a) chlorophyll, (b) total suspended solids, (c) epiphyte:seagrass ratio, (d) macroalgae cover, (e) percent surface light, (f) Secchi depth.

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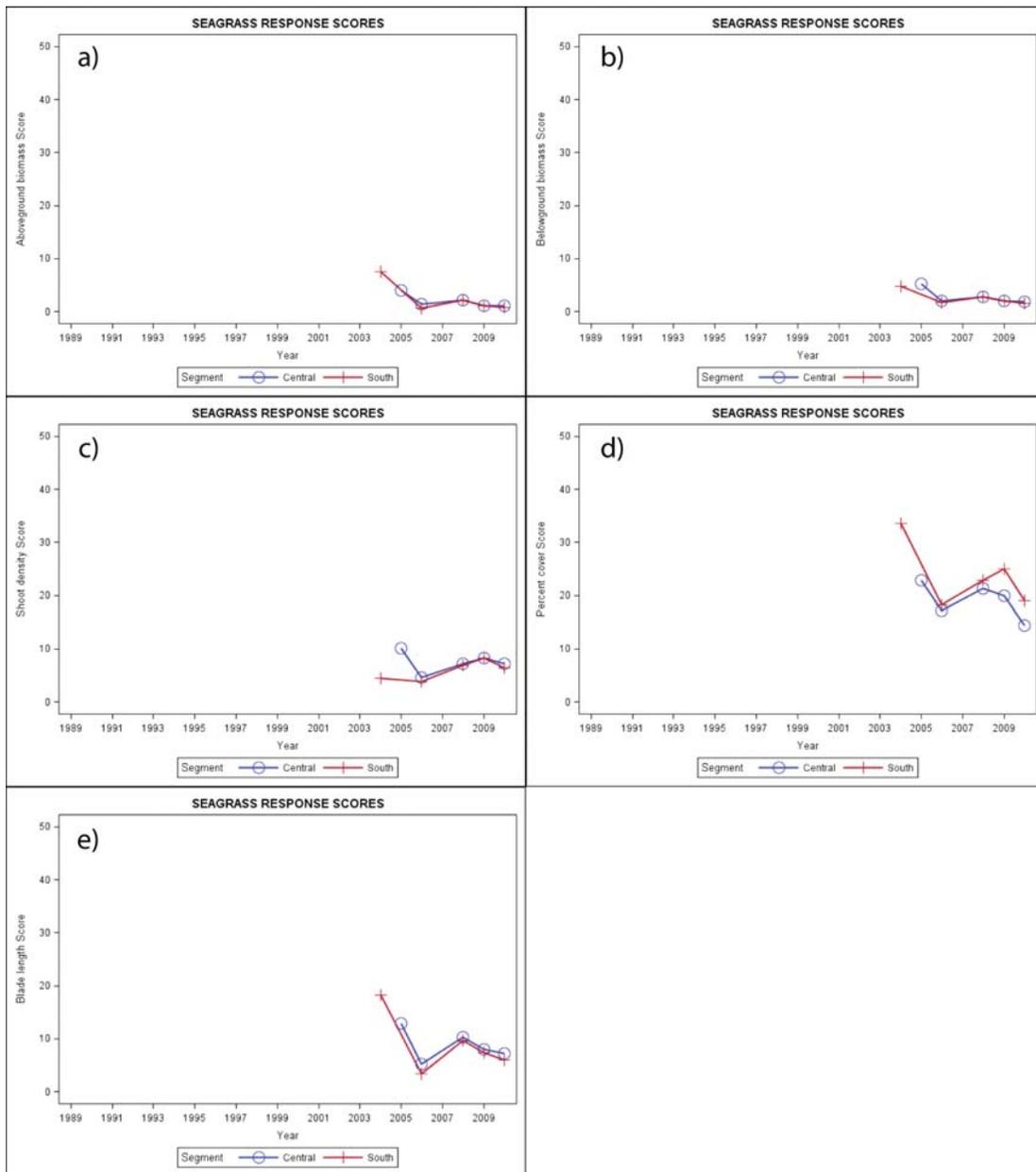
Figure 3 - 33 Scores of Seagrass Response indicators: (a) aboveground biomass, (b) belowground biomass, (c) shoot density, (d) percent cover, (e) blade length.

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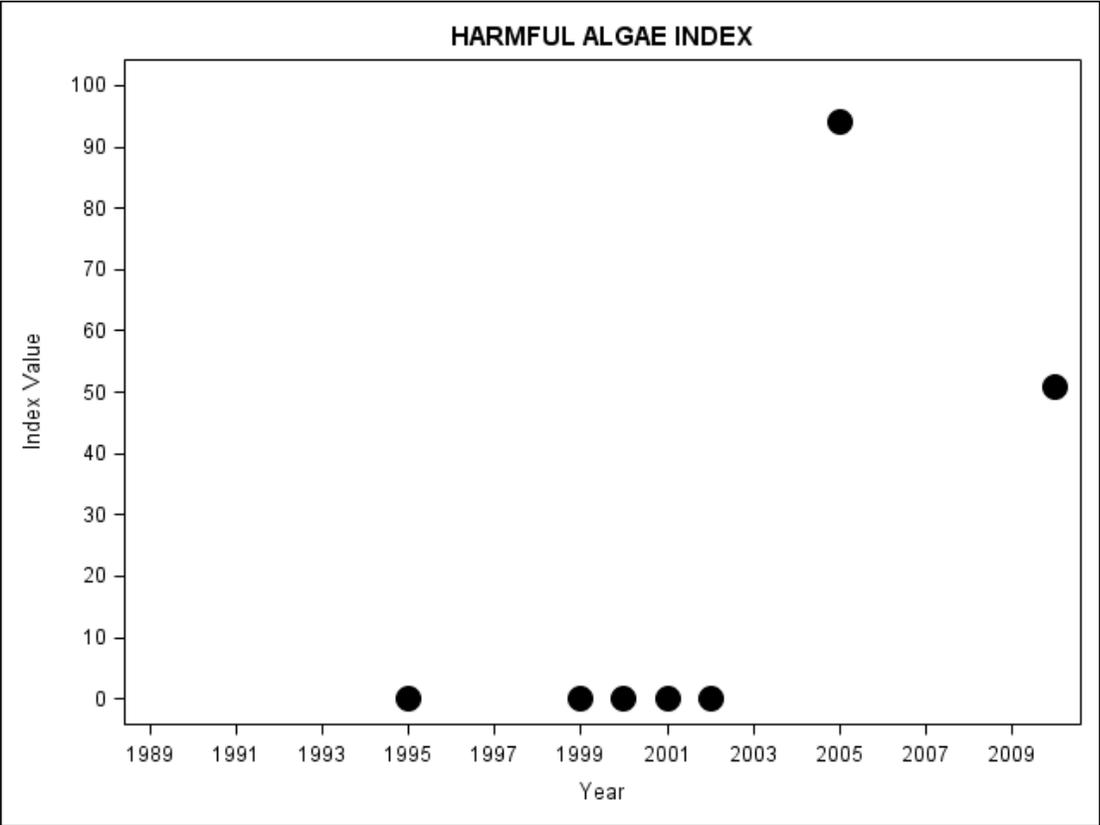
Figure 3 - 34 Harmful algal bloom Index.
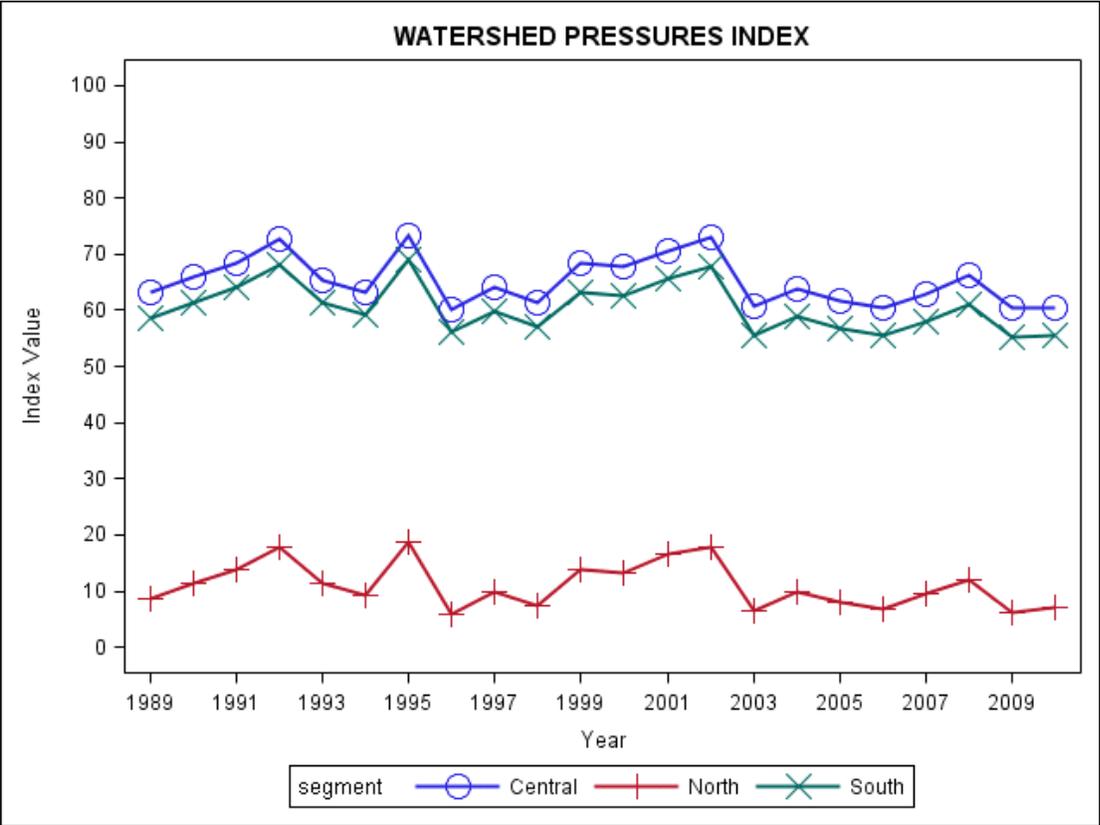
Figure 3 - 35 Watershed Pressure Index.
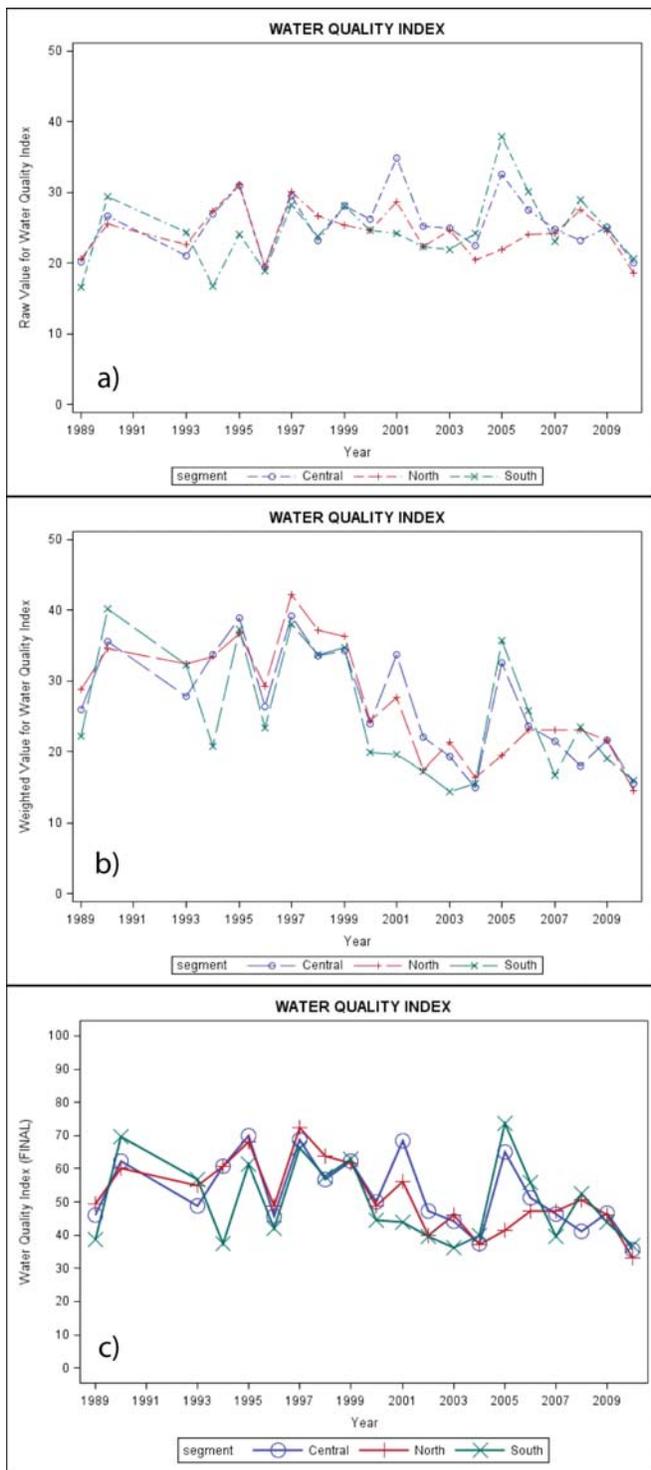
Figure 3 - 36 Water Quality Index scores: (a) Raw Scores, (b) Weighted Scores, (c) Final Scores.
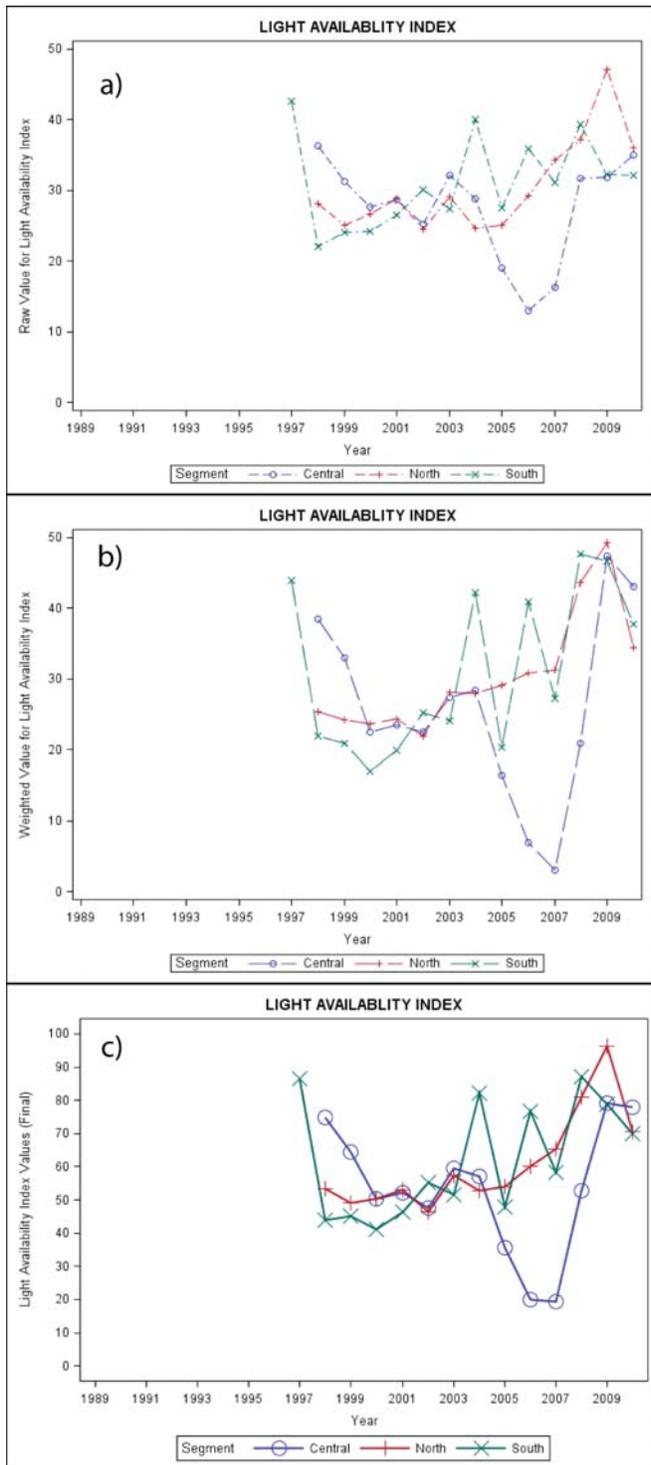
Figure 3 - 37 Light Availability Index scores: (a) Raw Scores, (b) Weighted Scores, (c) Final Scores.
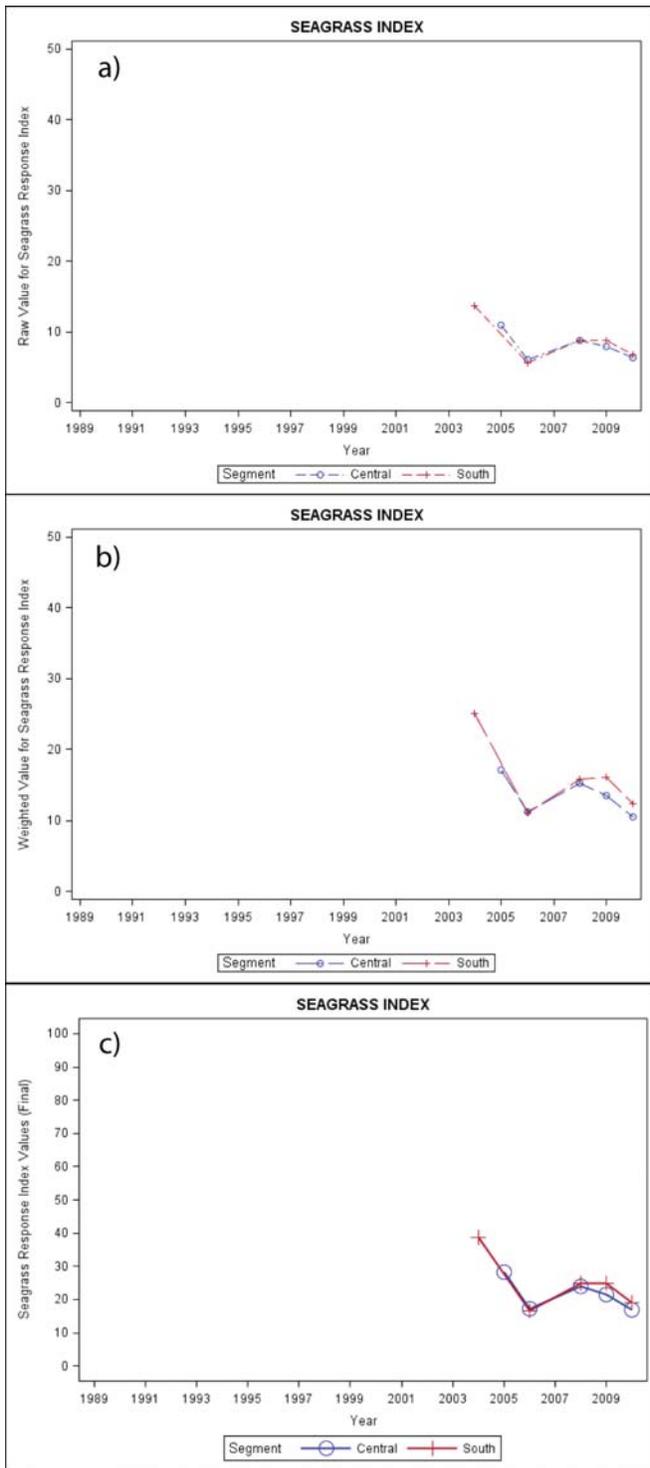
Figure 3 - 38 Seagrass Response Index scores: (a) Raw Scores, (b) Weighted Scores, (c) Final Scores.
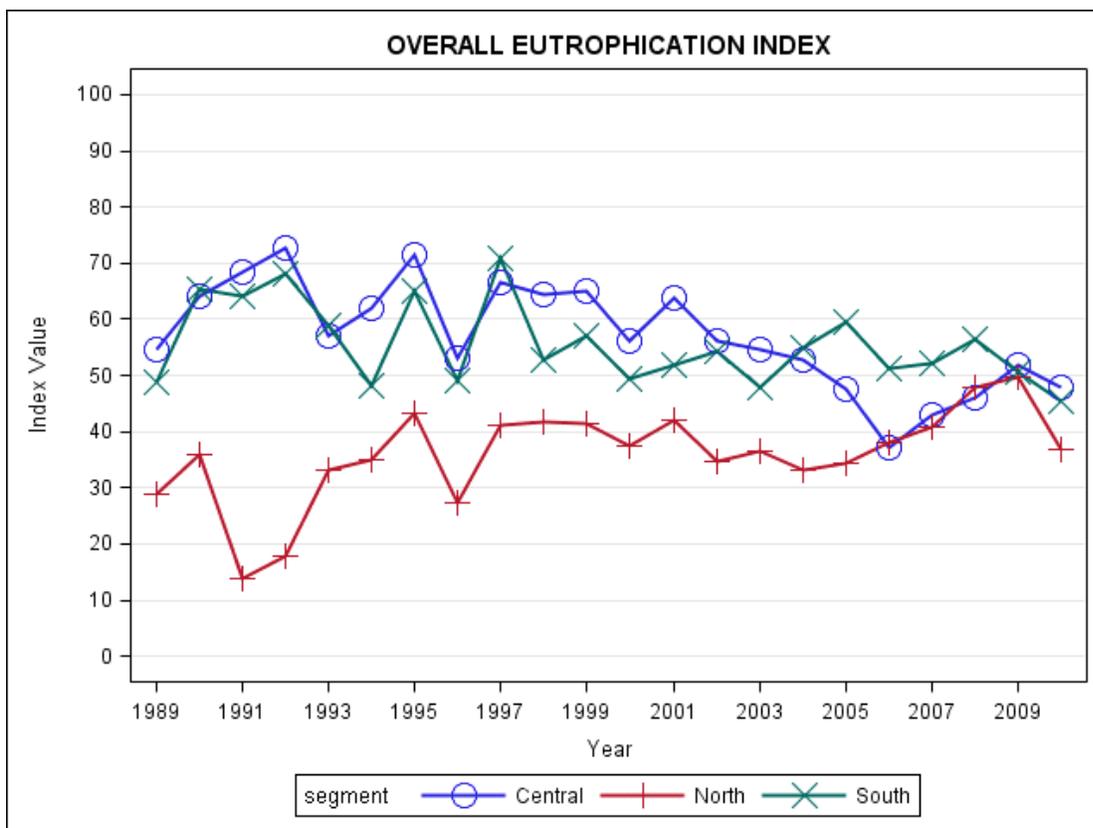
Figure 3 - 39 Overall Eutrophication Index.
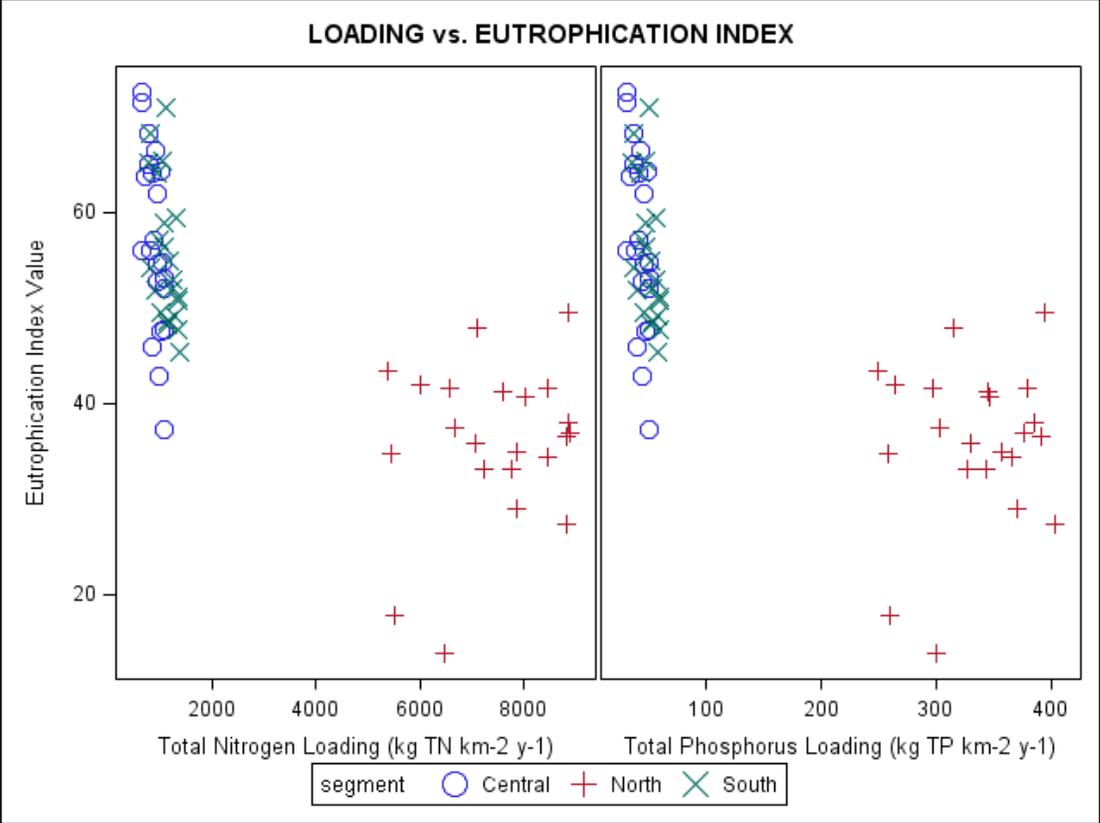
Figure 3 - 40 Eutrophication Index values vs. total nutrient loadings.
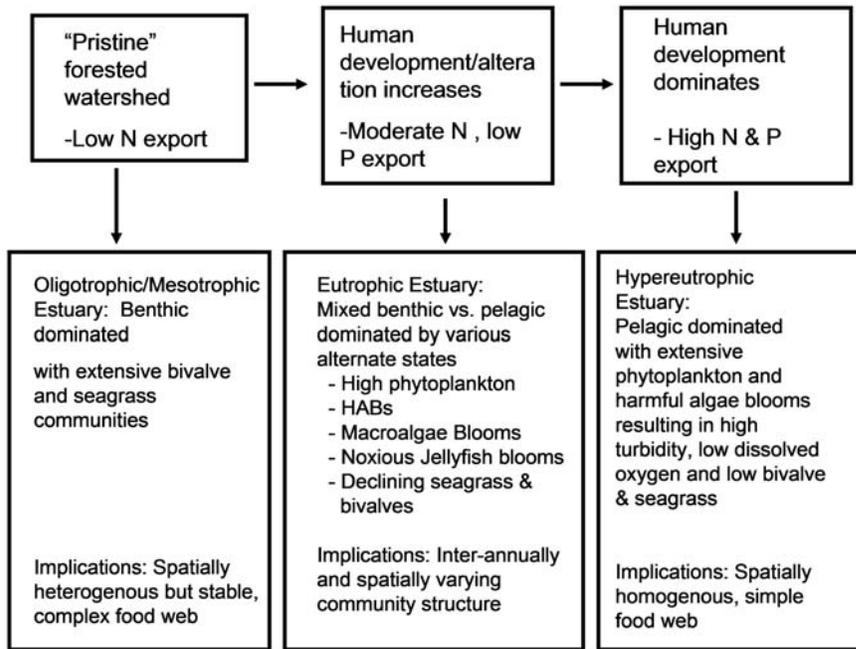
Figure 5 - 1 Conceptual model of eutrophication proposed for the BB-LEH Estuary showing variable biotic pathway responses.
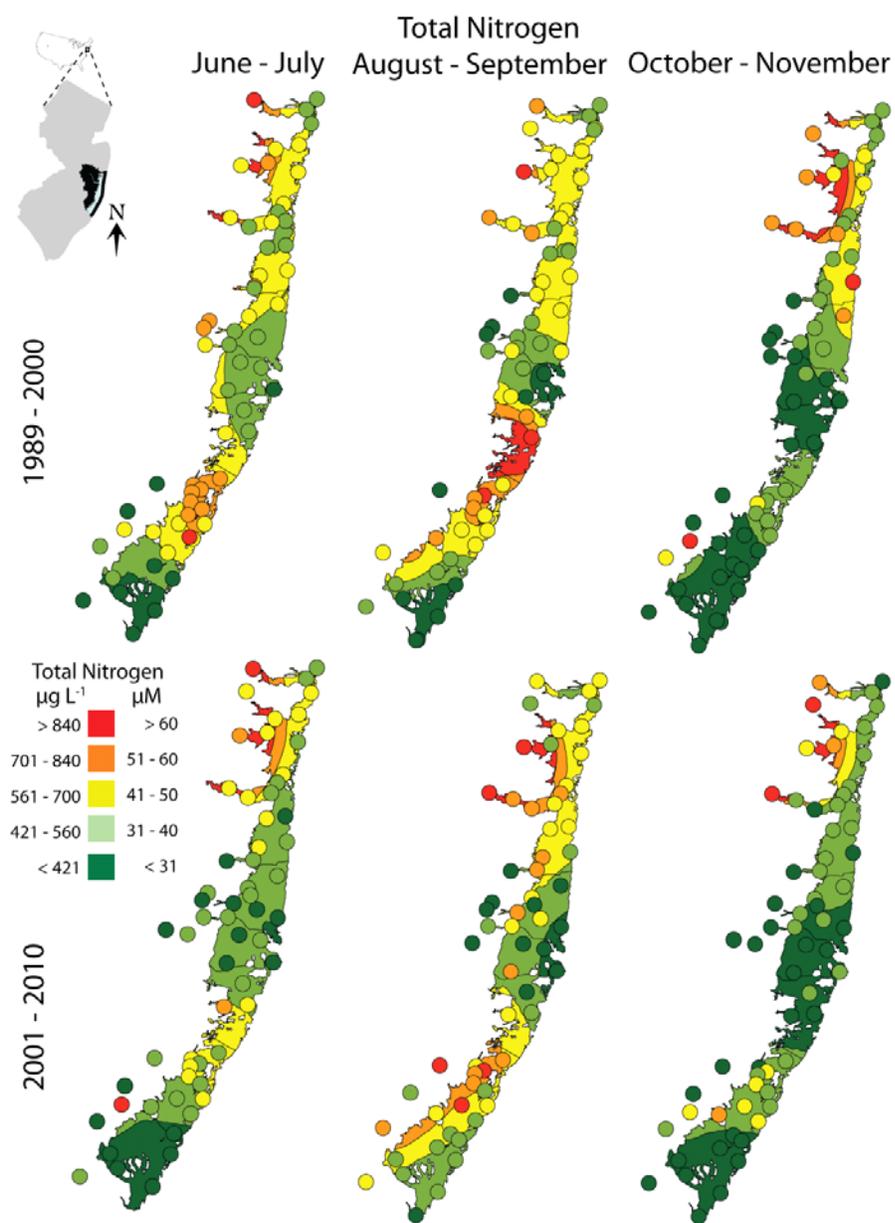
Figure 5 - 2 Total nitrogen concentrations in the BB-LEH Estuary during the June-July, August-September, and October-November sampling periods from 1989-2000 (upper graphic) and 2001-2010 (lower graphic).
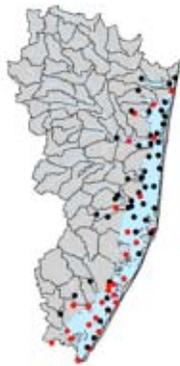
Figure 5 - 3 Water quality sampling stations of the NJDEP in BB-LEH Estuary during the 1989-2010 sampling period. Red dots show stations where dissolved oxygen values were < 4 mg L^{-1}.
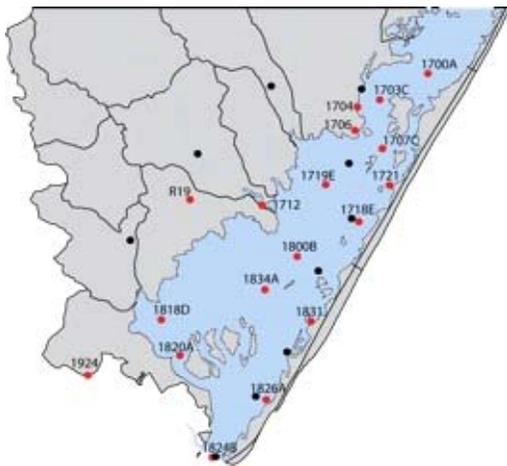
Figure 5 - 4 Sampling stations of the NJDEP for dissolved oxygen measurements in Little Egg Harbor. Note most dissolved oxygen measurements less than 4 mg L^{-1} in the BB-LEH Estuary have been recorded in the southern segment of the estuary.
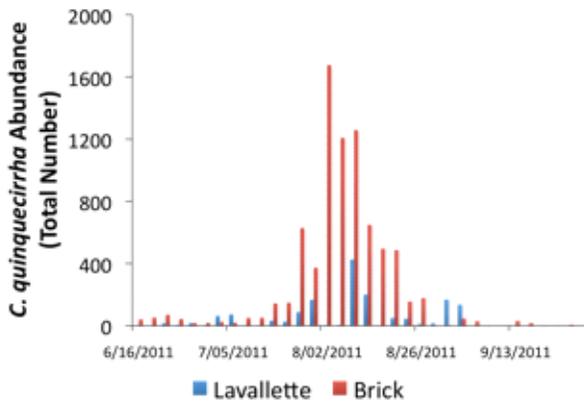
Figure 5 - 5 Abundance of sea nettles in seine sampling conducted at Brick and Lavallette sampling sites in the north segment of the BB-LEH Estuary during 2011. Note elevated abundances at Brick in lower salinity waters.

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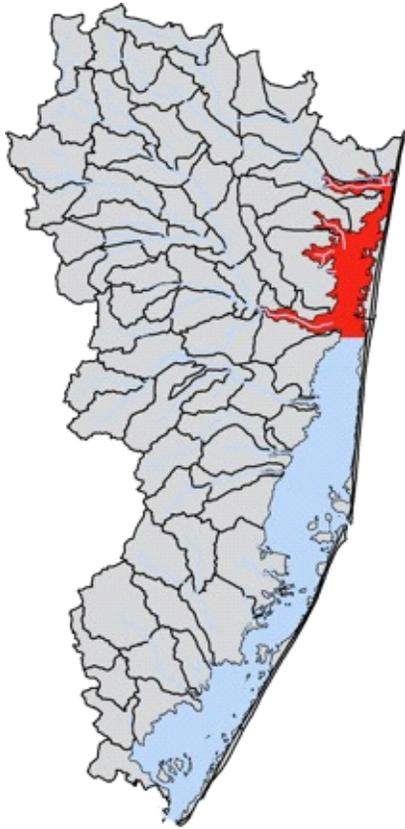
Figure 5 - 6 North segment waters of the BB-LEH Estuary (highlighted) impacted by the occurrence of sea nettles leading to extensive non-swimmable conditions and loss of human use and activity.

TABLES

Table 2 - 1 Mean (+/- standard deviation) percent cover of macroalgae on seagrass beds in the BB-LEH Estuary during 2004-2010.

Sampling Period	Percent Cover
Months	*%*
2004	
June-July	12.8 (17.0)
August-September	21.4 (24.3)
October-November	13.7 (16.5)
2005	

June-July	14.2 (22.3)
August-September	7.1 (9.8)
October-November	2.1 (3.9)

2006
June-July	2.1 (5.1)
August-September	7.0 (12.6)
October-November	6.6 (14.0)

2008
June-July	20.2 (29.0)
August-September	9.6 (19.5)
October-November	5.1 (7.9)

2009
June-July	6.5 (16.0)
August-September	3.0 (10.2)
October-November	12.8 (14.9)

2010
June-July	3.9 (10.3)
August-September	6.9 (18.4)
October-November	2.9 (14.9)

Table 2 - 2 Occurence of macroalgal blooms in the BB-LEH Estuary over the 2004-2010 study period.

Year	Pre-Bloom (60-70% cover)	Early Bloom (70-80% cover)	Full Bloom (80-100% cover)
2004	1	0	8
2005	2	0	2
2006	1	2	2

2008	0	11	11
2009	5	6	5
2010	1	0	8

Table 2 - 3 Regression analysis of macroalgae areal percent cover (a) over 2004-2010 for each of three time periods and (b) over the three time periods for each year.

a)

Time Period	n	Slope	Intercept	R^2	F	P
June-July	600	-0.66	1,338	0.00	2.91	0.09
August-September	600	-1.50	3,015	0.03	19.60	**< 0.01**
October-November	600	-0.37	760	0.00	1.40	0.24

b)

Year	n	Slope	Intercept	R^2	F	P
2004	180	0.46	15.03	0.00	0.06	0.80
2005	180	-6.04	19.86	0.11	21.61	**< 0.01**
2006	360	2.28	0.66	0.03	9.76	**< 0.01**
2008	360	-7.56	26.76	0.08	32.02	**< 0.01**
2009	360	3.11	1.22	0.02	7.74	**< 0.01**
2010	360	-0.52	5.60	0.00	0.29	0.59

Table 2 - 4 Correlation analysis between macroalgae areal percent cover and water quality, eelgrass (Zostera marina) and widgeon grass (Ruppia maritima) during three time periods over 2004-2010. Sample size (n), correlation coefficient (r), and p value (p).

	Variable	Units	June-July			August-September			October-November		
			r	p	n	r	p	n	r	p	n
Water quality	Temperature	°C	-0.05	0.19	603	0.03	0.51	678	0.06	0.19	560
	Salinity	ppt	-0.06	0.17	550	0.05	0.21	678	**0.11**	**0.01**	560
	Conductivity	$\mu S\ m^{-1}$	**-0.12**	**0.01**	460	0.07	0.10	549	**0.12**	**0.00**	560
	Dissolved Oxygen	$mg\ L^{-1}$	**-0.11**	**0.01**	550	0.02	0.59	677	0.05	0.20	559
	Dissolved Oxygen	%	**-0.10**	**0.02**	550	0.05	0.23	678	**0.12**	**0.01**	560
	pH		**-0.09**	**0.04**	550	0.02	0.57	677	0.08	0.06	556
	Secchi depth	cm	-0.03	0.66	188	0.03	0.58	256	0.00	1.00	234
Seagrass biomass	Zostera aboveground biomass	$g\ m^{-2}$	**0.19**	**0.00**	571	0.04	0.28	621	0.03	0.52	540
	Ruppia aboveground biomass	$g\ m^{-2}$	0.12	0.34	60	-0.18	0.16	60	**0.38**	**0.00**	60
	Zostera belowground biomass	$g\ m^{-2}$	**0.16**	**0.00**	571	0.04	0.28	621	0.04	0.31	540
	Ruppia belowground biomass	$g\ m^{-2}$	0.10	0.46	60	-0.20	0.13	60	**0.27**	**0.04**	60
Seagrass demographics	Zostera shoot density	$shoots\ m^{-2}$	-0.02	0.65	463	-0.04	0.38	500	-0.05	0.33	404
	Ruppia shoot density	$shoots\ m^{-2}$	-0.05	0.38	289	-0.08	0.13	336	-0.06	0.30	332
	Zostera blade length	cm	**0.22**	**0.00**	440	**0.10**	**0.04**	449	0.05	0.37	349
	Zostera areal % cover	%	-0.04	0.30	609	0.00	0.93	680	**0.10**	**0.01**	560
	Ruppia areal % cover	%	-0.04	0.31	609	-0.07	0.07	680	**-0.11**	**0.01**	560
	Other areal % cover	%	0.08	0.40	120	-0.03	0.74	120			

Table 2 - 5 Mean (+/- standard deviation) percent cover of epiphytes on upper leaf and lowr leaf surface of Zostera marina, and total epiphyte biomass on Zostera marina leaves during 2009 and 2010.

Sampling Period	Upper Leaf Percent Cover	Lower Leaf Percent Cover	Biomass
Months	*%*	*%*	*mg dry wt m^{-2}*
2009			
June-July	38.3 (26.8)	38.3 (27.1)	121.8 (495.0)
August-September	36.4 (30.4)	36.4 (30.2)	55.4 (111.7)
October-November	19.2 (24.9)	18.4 (24.6)	37.6 (100.3)
2010			
June-July	11.3 (15.4)	10.7 (15.4)	20.8 (65.9)
August-September	25.7 (23.1)	24.4 (22.9)	67.7 (113.9)
October-November	21.1 (25.8)	20.0 (25.5)	21.2 (47.0)

Table 2 - 6 Mean (+/- standard deviation) aboveground and belowground biomass, shoot density, blade length, and percent areal cover of Zostera marina recorded in the BB-LEH 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 2 - 7 Population demographics of Zostera marina in BB-LEH Estuary, 2004-2010. Time (x), number of shoots (N_x), rate of change per shoot (r_x), growth rate (λ_x), instantaneous mortality rate (m_x), survival probability (p_x), and stable-age distribution (C_x).

Year	x	N_x	r_x	λ_x	m_x	l_x	p_x	C_x
2004	0	189	1.00	1.00	0.00	1.00	1.00	1.00
2005	1	420	0.38	1.46	-0.80	0.22	0.22	0.18
2006	2	309	-0.15	0.86	0.31	0.16	0.07	0.27
2007	3	nd	nd	nd	nd	nd	nd	nd
2008	4	632	0.17	1.19	-0.72	0.33	0.20	0.24
2009	5	723	0.07	1.07	-0.13	0.38	0.11	0.35
2010	6	596	-0.10	0.91	0.19	0.32	0.08	0.62

Table 2 - 8 Mean (+/- standard deviation) aboveground and belowground biomass, shoot density, and percent areal cover of Ruppia maritima record in the BB-LEH 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)

Table 2 - 9 Abundances of Aureococcus anophagefferens in the BB-LEH Estuary.

Year	Abundance cells mL^{-1}
1988	<35,000
1995	1.0×10^6
1997	$<2.0 \times 10^5$
1999	$>1.8 \times 10^6$
2000	$>1.8 \times 10^6$
2001	$>1.8 \times 10^6$
2002	$>1.5 \times 10^6$
2003	$<2.0 \times 10^5$
2004	$<2.0 \times 10^5$
2005	$<5 \times 10^4$
2010	1.6×10^5

Table 2 - 10 Benthic invertebrate samples collected in the BB-LEH Estuary for the National Coastal Assessment Program.

Year	Number of Samples
2000	4
2001	15
2002	6
2003	4
2004	10
2005	4
2006	16
TOTAL	59

Table 3 - 1 Temporal and spatial extent of the REMAP dataset.

Date	SITE_ID	Latitude	Longitude
6/27/01	BB007	39.80475	-74.10910
6/28/01	BB004	39.76683	-74.18945
6/28/01	BB015	39.80288	-74.14280
7/2/01	BM056	39.76027	-74.11545
7/3/01	BB011	39.72190	-74.14403
7/3/01	BB046	39.70578	-74.13757
7/5/01	BB049	39.73570	-74.16197
7/5/01	BM059	39.75282	-74.18687
7/5/01	BM062	39.74630	-74.19385
7/8/01	BM054	40.05680	-74.11457
7/9/01	BB010	39.81195	-74.14552
7/9/01	BB032	39.84518	-74.11158
7/9/01	BB048	39.83877	-74.13238
7/10/01	BB006	39.92827	-74.10322
7/10/01	BB017	39.94378	-74.10728
7/10/01	BB037	39.93198	-74.12387
7/11/01	BB038	39.78635	-74.17960
7/11/01	BB039	39.76577	-74.17157
7/11/01	BB044	39.74878	-74.17295
7/12/01	BB013	39.85685	-74.09950
7/12/01	BB024	39.88258	-74.09852
7/12/01	BM068	39.86998	-74.14938
7/16/01	BB001	39.98827	-74.12333
7/16/01	BB042	40.00490	-74.09673
7/16/01	BB081	39.96615	-74.10993
7/17/01	BB002	40.04505	-74.05832
7/17/01	BB040	40.05977	-74.08108
7/17/01	BM060	40.06010	-74.06990
7/18/01	BM074	40.03847	-74.05985
7/20/01	BM053	39.94628	-74.16310
7/20/01	BM075	39.94853	-74.19247
7/23/01	BM057	39.93232	-74.15548
7/23/01	BM064	39.93868	-74.14868
7/23/01	BM078	39.93437	-74.15055
7/24/01	BB050	40.00123	-74.06680
7/24/01	BM061	40.01933	-74.12677
7/24/01	BM066	40.00698	-74.06200
7/25/01	BM063	39.93887	-74.08492
7/25/01	BM077	39.94352	-74.08338
7/25/01	BM079	39.92970	-74.08192
7/26/01	BB003	39.79495	-74.17845
7/26/01	BB021	39.81438	-74.15035
7/26/01	BM055	39.87248	-74.13413
7/31/01	BB019	39.98995	-74.08328
7/31/01	BM065	39.99237	-74.07162
8/1/01	BB016	39.68908	-74.17852
8/1/01	BB035	39.68238	-74.16638
8/1/01	BM067	39.65640	-74.18815
8/2/01	BB005	39.58587	-74.23260
8/2/01	BB027	39.59328	-74.23205
8/2/01	BB033	39.60307	-74.25117
8/3/01	BB083	39.79400	-74.17362
8/3/01	BM080	39.65878	-74.20480
8/3/01	BM084	39.76927	-74.18425
8/8/01	BB009	39.58087	-74.28887
8/8/01	BB014	39.59655	-74.28667
8/8/01	BB047	39.57935	-74.29942
8/9/01	BB025	39.57293	-74.32832
8/9/01	BB029	39.59070	-74.30907
8/10/01	BB036	39.66737	-74.20765
8/10/01	BM052	39.65422	-74.20447
8/13/01	BB018	39.61440	-74.22198
8/13/01	BM058	39.54038	-74.26237
8/13/01	BM071	39.57863	-74.23447
8/14/01	BM072	39.98190	-74.07000
8/14/01	BM082	40.05737	-74.07202
8/14/01	BM095	40.05003	-74.11215
8/15/01	BB031	39.76867	-74.15240
8/16/01	BM087	39.95817	-74.08037
8/16/01	BM097	39.99648	-74.14472
8/16/01	BM107	39.99043	-74.06933
8/17/01	BM051	39.92068	-74.10698
8/17/01	BM089	39.89665	-74.13335
8/17/01	BM093	39.86940	-74.15375
8/20/01	BM069	39.75860	-74.12295
8/20/01	BM109	39.82510	-74.17913
8/21/01	BB022	39.52743	-74.28058
8/21/01	BB028	39.53720	-74.27500
8/21/01	BB041	39.54652	-74.28970
8/21/01	BM106	39.53532	-74.26518

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Table 3 - 2 Equations for each indicator used to rescale observations into raw scores according to defined thresholds.

Component	Metric	Units	Rescaling equation	Maximum Score	Minimum Score	
Pressures	Total nitrogen loading	kg TN	$y = -19*ln(x)+177.52$	$x \le 50$	$x \ge 10{,}000$	Boynton et al. 1996, Sh
Pressures	Total phosphorus loading		$y = -32.81*ln(x)+204.01$	$x \le 25$	$x \ge 500$	Deegan et al. 2002,
Water Quality	Temperature	C	$y = -3.125*x+106.25$	$x \le 18$	$x \ge 34$	Stevenson et al. 1993,
Water Quality	Dissolved oxygen	mg L^{-1}	$y = 4.8641*e^{0.228*x}$	$x \ge 10$	$x \le 4$	1999, Diaz and Solow
Water Quality	Total nitrogen	µg L^{-1}	$y = 26721*x^{-1.274}$	$x \le 135$	$x \ge 750$	2003, Borja et al. 2004,
Water Quality	Total phosphorus	µg L^{-1}	$y = 475.95*x^{-0.977}$	$x \le 10$	$x \ge 45$	2012, Lee et al. 20
Light Availability	% surface irradiance available	%	$y = 50.084*ln(x)-122.18$	$x \ge 32$	$x \le 7.818$	Dennison et al. 1993,
Light Availability	Chlorophyll *a*	µg L^{-1}	$y = -41.67*ln(x)+85.351$	$x \le 2.5$	$x \le 100$	Burkholder et al. 2007
Light Availability	Total suspended solids	mg L^{-1}	$y = -5*x+100$	$x \le 10$	$x \ge 20$	
Light Availability	Secchi depth	cm	$y = 0.125*x-12.5$	$x \ge 500$	$x \le 100$	
Light Availability	Macroalgae percent cover	%	$y = -24.52*ln(x)+76.782$	$x \le 3$	$x \ge 20$	
Light Availability	Epiphyte:Seagrass biomass	g epiphyte / g seagrass	$y = -20.32*ln(x)+22.744$	$x \le 2.5$	$x \ge 2.0$	
Seagrass response	Aboveground biomass	g m^{-2}	$y = 0.125*x$	$x \ge 400$	$x \le 0$	Dennison et al. 1993, D
Seagrass response	Belowground biomass	g m^{-2}	$y = 0.0625*x$	$x \ge 800$	$x \le 0$	2003, Kemp et al. 200
Seagrass response	Percent cover	%	$y = 15.925*ln(x)-12.713$	$x \ge 10$	$x \le 0$	
Seagrass response	Shoot density	shoots m^{-2}	$y = 0.0243*x+5.7143$	$x \ge 1910$	$x \le 0$	
Seagrass response	Blade length	cm	$y = 0.625*x$	$x \ge 80$	$x \le 0$	
Harmful algal bloom response	Harmful algal bloom concentration	cells L^{-1}	$y = -0.0004*x+113.98$	$x \le 30{,}000$	$x \ge 260{,}000$	Gastrich and Wazniak 2
Benthic invertebrate response	Benthic invertebrates					Baden et al. 1990,

Table 3 - 3 Light and water quality thresholds relevant to seagrass. (From Kemp et al. 2004).

TABLE 2. Statistically-derived water quality thresholds beyond which submerged aquatic vegetation (SAV) are not present, and calculated minimum light requirements for SAV survival.[a]

Salinity Regime[b]	Growing Season[c]	Light Required at SAV Leaf (%), PLL_{min}	Light Required through Water (%), PLW_{min}	Water Column Light Attenuation (K_d, m^{-1})	Total Suspend Solids (mg l^{-1})	Plankton Chlorophyll *a* (µg l^{-1})	Dissolved Inorganic Nitrogen (mg l^{-1})	Dissolved Inorganic Phosphorus (mg l^{-1})
Tidal Freshwater	April–October	>9	>13	<2	<15	<15	—	<0.02
Oligohaline	April–October	>9	>13	<2	<15	<15	—	<0.02
Mesohaline	April–October	>15	>22	<1.5	<15	<15	<0.15	<0.01
Polyhaline	March–May September–November	>15	>22	<1.5	<15	<15	<0.15	<0.01

[a] Indicates that these are statistically-derived water quality threshold values, beyond which SAV were found to be absent, based on intensive field studies at selected sites in Chesapeake Bay (Batiuk et al. 1992; Dennison et al. 1993). Minimum light requirement for SAV survival given as a percent of surface light through the water column (PLW_{min}) and percent of surface light at leaves (PLL_{min}) based on Eqs. 1 and 2 (see text).

[b] Regions of the estuary defined by salinity regime, where tidal freshwater means < 0.5 psu, oligohaline means 0.5–5 psu, mesohaline means 5–18 psu, and polyhaline means >18 psu.

[c] Medians calculated over this growing season should be used to check the attainment of any of these habitat requirements, and raw data collected over this period should be used for statistical tests of attainment. For polyhaline areas, the data are combined for the two periods shown.

Table 3 - 4 Dissolved oxygen thresholds for Maryland's coastal bays. (From Wazniak et al. 2007).

TABLE 2.   Biologically relevant thresholds for dissolved oxygen in the Maryland coastal bays.

Threshold criteria category for fisheries and benthic community	DO cutoff (mg/L)
Better than objective	>7
Meets objective	>6
Borderline for community	5–6
Community threatened	3–5
Does not meet objectives	<3

Notes: The critical time period for oxygen is summer, June–August. Values are the median dissolved oxygen cutoff.

Table 3 - 5 Thresholds for total nitrogen, total phosphorus, and chlorophyll a concentrations for Maryland's coastal bays. (From Wazniak et al. 2007).

TABLE 1. Biologically relevant thresholds for nutrients and chlorophyll *a* in the Maryland coastal bays.

Biologically relevant threshold	Cutoff values		
	TN (mg/L)	TP (mg/L)	Chlorophyll *a* (μg/L)
Better than seagrass objective	<0.55	<0.025	<7.5
Meets seagrass objective	<0.64	<0.037	<15
Does not meet seagrass objective	0.65–1	0.38–0.043	15–30
Does not meet STAC objectives and/or dissolved oxygen threatened	1–2	0.44–0.1	30–50
Does not meet any objectives	>2	>0.1	>50

Notes: Critical time periods for nutrients, total nitrogen (TN) and total phosphorus (TP), are annual and the critical time period for chlorophyll is April–October (SAV growing season). STAC, Scientific and Technical Advisory Committee.

Table 3 - 6 Optmal temperatures for growth and photosynthesis of various seagrass species. (From Lee et al. 2007).

Table 7

Average values of optimal temperatures for growth and photosynthesis of temperate and tropical/subtropical seagrass species

Species	Optimal temp. (°C)	
	Growth	Photosynthesis
Temperate		
Amphibolis antarctica	26	23
Amphibolis griffithii		23
Heterozostera tasmanica		30
Phyllospadix torreyi	13	23
Posidonia australis	19	23
Posidonia oceanica	15.5±2.5	32
Posidonia sinuosa		20.5
Ruppia maritima		25.5±2.5
Zostera asiatica	12.6	
Zostera capensis	17.5	
Zostera japonica	18.5±3.5	
Zostera marina	15.3±1.6	23.3±1.8
Tropical/subtropical		
Cymodocea nodosa	24.5	31.0±0.5
Cymodocea rotundata		27
Enhalus acoroides		27
Halophila decipiens		30
Halophila johnsonii		32.5±2.5
Halophila ovalis	25	27.5
Halodule wrightii		27.7±1.5
Halodule uninervis	24.5	
Syringodium filiforme	28.7±1.8	
Thalassia hemprichii		27
Thalassia testudinum	29.1±0.3	29.0±1.2

Table 3 - 7 Area normalized occurences of macroalgae blooms in BB-LEH.

Table 1. Area normalized occurrences of macroalgal blooms (# blooms m^{-2}) in the Barnegat Bay–Little Egg Harbor Estuary over the 2004–2010 study period.

	Pre-Bloom (60–70%)			Early Bloom (70–80%)			Full Bloom (80–100%)		
	Jun–Jul	Aug–Sep	Oct–Nov	Jun–Jul	Aug–Sep	Oct–Nov	Jun–Jul	Aug–Sep	Oct–Nov
2004	0.00	0.07	0.00	0.00	0.00	0.00	0.13	0.27	0.13
2005	0.13	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00
2006	0.00	0.05	0.00	0.00	0.05	0.05	0.00	0.05	0.05
2008	0.00	0.00	0.00	0.27	0.10	0.00	0.23	0.13	0.00
2009	0.07	0.03	0.07	0.03	0.03	0.13	0.07	0.00	0.10
2010	0.00	0.03	0.00	0.00	0.00	0.00	0.03	0.13	0.10

Table 3 - 8 Eigenvectors and annual weightings for Water Quality indicators under Scenario 1.

| Year | WQ1 Eigenvector | | | | Weighting for WQ1 | | | |
	Dissolved Oxygen	Temperature	Total Nitrogen	Total Phosphorus	Dissolved Oxygen	Temperature	Total Nitrogen	Total Phosphorus
1989	0.37	0.58	-0.72		0.14	0.34	0.52	
1990	0.40	0.50	0.77		0.16	0.25	0.59	
1991	0.00	0.00	1.00		0.00	0.00	1.00	
1993	1.00	0.09	-0.01		0.99	0.01	0.00	
1994	1.00	-0.04	-0.05		1.00	0.00	0.00	
1995	0.91	0.01	0.41		0.84	0.00	0.16	
1996	0.83	-0.27	-0.49		0.69	0.07	0.24	
1997	0.97	0.20	-0.16		0.93	0.04	0.02	
1998	-0.02	0.63	0.77		0.00	0.40	0.60	
1999	-0.17	-0.10	0.97	-0.15	0.03	0.01	0.94	0.02
2000	-0.28	0.05	0.27	0.92	0.08	0.00	0.07	0.85
2001	0.00	0.39	0.01	0.92	0.00	0.15	0.00	0.85
2002	0.01	0.11	0.53	0.84	0.00	0.01	0.28	0.70
2003	0.16	0.38	-0.07	0.91	0.03	0.15	0.01	0.82
2004	0.02	0.11	-0.20	0.97	0.00	0.01	0.04	0.95
2005	0.01	0.06	-0.17	0.98	0.00	0.00	0.03	0.97
2006	-0.04	-0.01	0.99	-0.11	0.00	0.00	0.99	0.01
2007	0.14	0.46	0.29	0.83	0.02	0.21	0.09	0.68
2008	-0.05	-0.03	0.98	-0.18	0.00	0.00	0.97	0.03
2009	0.80	0.09	-0.55	0.22	0.64	0.01	0.30	0.05
2010	0.55	0.31	-0.46	0.63	0.30	0.09	0.21	0.40

Table 3 - 9 Eigenvectors and multi-year weightings for Water Quality indicators under Scenario 2.

Year	WQ1 Eigenvector				Weighting for WQ1			
------	Dissolved Oxygen	Temperature	Total Nitrogen	Total Phosphorus	Dissolved Oxygen	Temperature	Total Nitrogen	Total Phosphorus
1989	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1990	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1991	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1993	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1994	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1995	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1996	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1997	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1998	0.96	0.19	-0.19		0.93	0.04	0.04	0.00
1999	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2000	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2001	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2002	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2003	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2004	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2005	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2006	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2007	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2008	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2009	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87
2010	0.11	0.34	0.06	0.93	0.01	0.12	0.00	0.87

Table 3 - 10 Weightings used to calculate Weighted Scores for indicators in each component and for each component within the overall Index of Eutrophication Condition.

Component	Years	Variable	Weighting
Watershed Pressures	1989-2010	Total Nitrogen Loading	0.50
Watershed Pressures	1989-2010	Total Phosphorus Loading	0.50
Water Quality	1989-1999	Temperature	0.66
Water Quality	1989-1999	Dissolved Oxygen	0.33
Water Quality	1989-1999	Total Nitrogen	0.02
Water Quality	1989-1999	Total Phosphorus	0.00
Water Quality	2000-2010	Temperature	0.15
Water Quality	2000-2010	Dissolved Oxygen	0.08
Water Quality	2000-2010	Total Nitrogen	0.13
Water Quality	2000-2010	Total Phosphorus	0.65
Light Availability	1998-2010	Chlorophyll a	0.02
Light Availability	1998-2010	TSS	0.32
Light Availability	1998-2010	Secchi depth	0.04
Light Availability	1998-2010	Epiphyte : Seagrass	0.30
Light Availability	1998-2010	Macroalgae % Cover	0.00
Light Availability	1998-2010	% Light reaching seagrass	0.31
Seagrass	2004-2010	Aboveground biomass	0.08
Seagrass	2004-2010	Belowground biomass	0.02
Seagrass	2004-2010	Shoot density	0.01
Seagrass	2004-2010	Percent cover	0.53
Seagrass	2004-2010	Blade length	0.35
Harmful algae	various	Cell concentration	1.00
Eutrophication	1989-1997	Water Quality	1.00
Eutrophication	1998-2003	Water Quality	0.50
Eutrophication	1998-2003	Light Availability	0.50
Eutrophication	2004-2010	Water Quality	0.33
Eutrophication	2004-2010	Light Availability	0.33
Eutrophication	2004-2010	Seagrass	0.33

Table 4 - 1 Physicochemical measurements in the BB-LEH Estuary during submerged aquatic vegetation

(SAV) sampling in 2011.

Segment	Sampling Period	N	Temp (°C)	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 4 - 2 Characteristics of submerged aquatic vegetation (SAV) by sampling period in the BB-LEH Estuary during 2011.

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

[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 4 - 3 Characteristics of submerged aquatic vegetation (SAV) by segment of BB-LEH Estuary (2011).

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

(3303.7)   15.4 (28.6)

	Segment	Time Period			
	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 4 - 4 Areal cover of macroalgae and other biotic elements in the BB-LEH 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 4-5 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 Zostera 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 5 - 1 Barnegat Bay Watershed land use-land cover in 1986, 1995, 2002, 2007 and 2010. Data from R. Lathrop (Center for Remote Sensing and Spatial Analysis, Rutgers University).

Type	1986 acres	1995 acres	2002 acres	2007 acres	2010 acres
Urban	78,781	90,044	101,078	109,739	111,560
Agriculture/Grassland	7,693	6,314	5,532	5,227	4,965
Barren	10,518	9,206	8,549	7,594	7,410
Upland Forest	164,693	158,147	148,828	141,183	139,915
Coastal Wetland	22,402	21,715	21,493	21,472	21,469
Freshwater Wetland	66,341	63,983	63,810	63,046	62,980
Water	157,823	158,840	158,956	159,989	159,955

	Annual Net Change 86-95	Annual Net Change 95-02	Annual Net Change 02-07	Annual Net Change 07-10
Urban	+1,251	+1,576	+1,732	+607
Agriculture/Grassland	-153	-112	-61	-87
Barren	-146	-94	-191	-61
Upland Forest	-727	-1,331	-1,529	-423
Coastal Wetland	-76	-32	-4	-1
Freshwater Wetland	-262	-25	-153	-22
Water	113	17	207	-11

Table 5 - 2 Dissolved oxygen concentrations (< 4.0 mg l⁻¹) recorded in BB-LEH Estuary by the New Jersey Department of Environmental Protection from 1989-2010.

Date	Time	Segment	Station	DO (mg l^{-1})
8/06/93	10:50	South	1831	3.90
8/11/93	10:35	Central	1675	3.50
8/30/93	11:05	South	1834A	3.30
9/23/93	10:35	South	1924	3.70
9/30/93	10:35	South	1719E	3.60
9/30/93	10:50	South	1800B	2.70
10/13/93	11:00	South	1706	3.65
10/13/93	11:10	South	1704	3.55
10/13/93	11:25	South	1703C	3.70
10/13/93	11:35	South	1700A	3.60
10/13/93	11:45	South	1707C	3.60
10/13/93	11:55	South	1721	3.25
10/13/93	12:05	South	1719E	3.40
10/13/93	12:25	South	1718B	2.65
12/15/93	10:15	Central	1688B	3.10
3/23/94	10:45	South	1820A	3.85
3/30/94	8:35	South	1703C	3.75
3/30/94	9:15	South	1721	3.60
6/03/94	7:45	Central	1670D	3.40
6/09/94	9:50	South	1924	3.35
6/16/94	9:50	South	1706	3.25
6/16/94	10:40	South	1707C	3.60
6/16/94	11:05	South	1718B	3.80
6/21/94	10:35	South	1831	2.10
6/21/94	10:45	South	1818D	3.20
6/21/94	10:55	South	1820A	3.30
8/09/95	9:20	North	1506A	3.05
9/27/95	12:50	South	1719E	3.70
12/18/95	11:05	South	1824B	3.30
4/04/96	9:25	South	1924	4.00
4/04/96	10:00	South	1824B	3.90
4/04/96	10:10	South	1826A	3.80

5/23/96	9:15	South	1706	3.60
5/23/96	9:40	South	1703C	3.70
5/23/96	9:45	South	1700A	3.80
5/23/96	9:55	South	1707C	3.60
5/23/96	10:05	South	1721	3.95
5/23/96	10:10	South	1718B	3.25
5/23/96	10:45	South	1820A	3.70
5/23/96	11:05	South	1834A	3.90
5/23/96	11:10	South	1800B	3.80
5/23/96	11:15	South	1712	3.30
6/26/96	9:15	Central	1675	4.00
6/26/96	9:45	Central	1674B	3.65
9/10/96	10:45	South	1707C	3.85
9/10/96	11:05	South	1718B	4.00
9/10/96	11:15	South	1800B	3.70
9/10/96	11:25	South	1834A	3.10
9/10/96	11:40	South	1712	3.20
9/10/96	11:50	South	1719E	3.80
9/20/96	10:25	South	1824B	3.80
9/20/96	10:35	South	1826A	4.00
9/20/96	10:50	South	1834A	3.80
9/20/96	11:00	South	1818D	3.80
6/12/97	10:45	South	1924	4.00
6/23/98	9:00	Central	R14	3.70
8/18/98	8:15	Central	R14A	3.80
8/18/98	10:40	North	R10	3.80
9/30/98	12:35	North	R10	4.00
12/03/98	11:35	South	1924	3.20
01/28/99	13:00	South	1924	3.40
06/18/99	8:40	North	1605A	3.90
06/30/99	9:10	Central	R14A	3.80
08/02/99	9:25	North	1629B	3.10
08/30/99	10:25	Central	R14A	3.66
08/04/00	10:22	South	R19	2.20
08/24/01	9:00	South	R19	2.55
09/25/01	11:30	Central	R14A	3.80
08/29/02	10:05	Central	R14A	3.70
09/11/02	9:30	North	1613A	3.45
09/08/09	11:15	Central	1654C	3.50
10/14/09	11:45	South	1924	2.95
10/22/09	12:08	South	1824B	3.20
08/14/10	12:15	South	R19	3.95

08/16/10	11:00	South	1924	3.75
08/19/10	8:15	South	1718B	3.00
08/19/10	8:50	South	1831	4.00
08/23/10	10:14	South	1721	3.95
08/23/10	11:00	South	1700A	3.85
10/20/10	11:35	South	1820A	3.05
10/20/10	11:45	South	1818D	3.00
10/21/10	10:55	South	1675	3.90

APPENDICES

Appendix 3 - 1 Equations and SAS code for estimating percent surface light at seagrass leaves

```
**Variables.
PLW = percent light through the water
PLL = percent light at the leaf (considers attenuation due to epiphytes)
Z = depth of SAV growth
Kd = attenuation coefficient - diffuse attenuation of light
     Kw = attenuation due to water
     Kdoc = attenuation due to dissolved organic matter
     Kchl = attenuation due to chl a
     Ktss = attenuation due to total suspended solids
Be = biomass of epiphytes
Bde = total mass of epiphytic matieral g dry weight per g SAV
Ke = biomass-specific epiphytic light attenuation coefficient

**Equations.
PLW = 100 exp [(-Kd)(Z)]
PLL = (PLW) exp [(-Ke)(Be)]
Kd = -ln(PLW/100)/Z
Kd = K(w+doc) + Kchl + Ktss
Kd = 0.32 + 0.016[chl] + 0.094[TSS]
-ln(PLW/100) = Z(0.32 + 0.016[chl] + 0.094[TSS])
[TSS] = -(0.32 + 0.016[chl] + ln(PLW/100)/Z)/0.094
Bde = 0.107*TSS + 0.832Be
Ke = 0.07 + 0.32(Be/Bde)^-0.88
```

Appendix 3 - 2 Additional data examined for potential inclusion in the Biotic Index of Eutrophication

Qualitative examinations of the NCA and REMAP datasets included focusing on sampling design, spatial and temporal extents of data, and consideration of the datasets in light of questions to be asked of the data. The scope of the REMAP (Table 1) and NCA (Table 2) datasets are presented below. Quite importantly, the answer to the question 'Can these data reliably answer questions about X' had to pass a 'reasonability' test. That is, was the answer to that question both logically reasonable and 'Yes'? For instance, can REMAP data, all of which was collected in 2001, reasonably tell us about the benthic condition of Barnegat Bay in 2009? What about 1989? Here the answer is no, because data from 2009 (and 1989) are not available in the REMAP dataset and it is well established that benthic conditions fluctuate year to year with associated changes in habitat and water quality condition (Dauer et al. 2000). This temporal constraint is particularly important considering a major aim is to calculate annual values of the biotic index, reflecting annual biotic response.

Quantitative examinations of the NCA and REMAP datasets included subjecting these and other datasets to statistical tests to address each of the specific questions listed below that were discussed in the last conference call. Briefly, these statistical analyses address 1) segmentation and gradients within Barnegat Bay, 2) how well REMAP and NCA datasets reflect gradients in Barnegat Bay, 3) dataset correspondence, 4) dataset combination, 5) thresholds and index scores, and 6) eelgrass decline and use as a bioindicator.

- ***Is segmentation of Barnegat Bay into three (or six) segments really necessary? Do TOC and grain size really vary by spatial segment?***

Yes. Segmentation of Barnegat Bay into three north-south areas and two east-west regions is necessary. The QAPP states (page 60) that the biotic index of eutrophic condition for this project will be based on, but not exactly replicate, the NEEA approach (in which an ASSETS score is determined) and that specifically, the index will be modifying the NEEA approach by dividing the estuary into three segments (as opposed to the two in the NEEA report), in addition to the wider array of biotic indicators used. The protocols in the QAPP have previously been agreed upon.

From the QAPP: "For the period from 2004 to 2011, the NEEA model of Bricker will be applied to the water quality and biotic data collected to compare against the findings of Bricker et al. (1999, 2007) for previous years to determine if any change in eutrophic condition has occurred. However, the approach used in this project will entail dividing the estuary into three segments based on environmental gradients. A wider array of biotic indicators will also be used because more key biotic parameters have been measured in this project. "

Mike Kennish provides ample background on the reasoning for dividing Barnegat Bay into multiple segments based upon differences in geology, morphology, bathymetry, sediments, water circulation and residence time, etc. These physical characteristics create a backdrop of gradients and benthic habitats against which major differences in benthic biotic response may be expected to occur. Appropriate sampling design (a prerequisite for statistical validity and inseparable from statistical analyses) must provide sufficient and equitable opportunity to sample across expected gradients to adequately characterize variability in each of these regimes (see Sokal and Rohlf 1981, Quinn and Keough 2002, Underwood 1997). Therefore, sampling efforts designed with the purpose of characterizing benthic biotic response in Barnegat Bay must sample adequately across the known gradients.

We appreciate the thoroughness of the TAC in ensuring the validity of this segmentation of Barnegat Bay. As requested, we conducted multiple statistical analyses to verify the rationale behind these segments. We conducted several ANOVA tests to see if any observed differences in water quality and benthic

habitat were statistically different between the three north-south segments. A p values less than 0.05 was considered statistically significantly different. Results of these ANOVA tests are in Table 3, but briefly, statistically significant differences between segments **were** observed for all watershed, water quality, and sediment variable **but not** for benthic invertebrate abundance (NCA data). These variables included total nitrogen loading, areal total nitrogen loading, salinity, total nitrogen concentration in Barnegat Bay, nitrate in Barnegat Bay, ammonia in Barnegat Bay, sediment grain size and sediment total organic carbon. This suggests that indeed, the segmentation of Barnegat Bay is statistically valid, that benthic invertebrate datasets are not adequately sampled across these segments, and that future sampling designs must address these gradients to adequately characterize and assess Barnegat Bay.

- ***Were principal components analysis (PCA) or trended correspondence analysis done to conclude that Barnegat Bay is characterized by multiple gradients?***

Yes. Multivariate statistics – principal components analysis (PCA) were conducted on both the REMAP and the NCA datasets. I am not sure which analysis is requested by the term 'trended correspondence analysis', but think it might either refer to canonical correlation analysis, which is essentially a many-many correlation analysis (rather than one-one, as in a more general correlation analysis) or to another analysis that is similar in nature and output to a PCA. However, a direct Pearson correlation matrix is both sufficient and most appropriate to elucidate the correlations between individual variables from the benthic datasets (such as abundance, etc.) and other variables that exhibit gradients throughout the segments of Barnegat Bay (salinity, nitrogen loading, nitrogen concentrations (total and dissolved), and sediment characteristics (grain size and total organic carbon).

PCA analysis: For the REMAP dataset, we examined benthic shellfish abundances for the three most abundant species: 1) *Ampelisca vadorum*, 2) *Mytilus edulis*, and 3) *Spirobidae* (LPIL). Combined, these three species represent the majority of individuals observed in the REMAP dataset. Figures X and Y show the results of the PCA, labeled by segment and by species name respectively. The most important thing to note about these two plots is that the data do not cluster together by either segment or by species. For PCA analysis, the closer together data points are, the more correlated they are. Thus, the REMAP dataset does not adequately reflect the differences apparent across the north-south segments.

Pearson Correlation: The REMAP shellfish abundances for the three most numerous species was not correlated with salinity (p > 0.08) or with nitrogen loading (p > 0.17). Thus, the REMAP dataset does not reflect the gradients of these variables apparent across the north-south segments.

- ***Can the biotic index of eutrophic condition be used for other regions of New Jersey, for example in areas where SAV is known to be absent?***

Possibly. The main issue with the application of the biotic index of eutrophic condition is that it is tuned to Barnegat Bay, which is a shallow coastal lagoonal estuary that has characteristics quite different from other estuaries and water bodies within New Jersey. Most likely, the biotic index can be best applied to other similar coastal lagoonal estuaries along the New Jersey shoreline. This index has high applicability to other similar coastal lagoons in other states, such as Waquoit Bay in Massachusetts, Great South Bay in New York, Delaware's Inland Bays, Chincoteague Bay in Maryland, etc. However, it would be inappropriate to use this biotic index for regions such as the Delaware River estuary or the New York New Jersey Harbor estuary or Raritan Bay, as these are ecosystems quite different in nature – deeper, drowned river valleys with much higher exposure to oceanic circulation and mixing.

- ***Will an index of benthic macroinvertebrates be included as a component of the biotic index of eutrophication condition so that if macroinvertebrate surveys are conducted in the future, those***

conditions can be assessed?

Yes. The biotic index of eutrophic condition will include a component for benthic macroinvertebrates. This component can be used in the future to assess condition if surveys are conducted in the future. We encourage future studies to be designed to capture the variability across the gradients in Barnegat Bay as have previously been discussed.

For the current project that we are working on, conditions will be assessed (hindcast) for 2001 using the REMAP dataset. This dataset will be able to be used for the 2001 assessment.

Though this 2001 REMAP dataset has enough samples (80) that sufficiently span the gradients in Barnegat Bay, we have discovered and are concerned, however, that the timing of the sampling may introduce spatial bias. While the sampling locations were randomized throughout the bay, they were not randomly sampled – sampling occurred in a generally north to south direction over the course of the summer of 2001 (Figure 1). While this makes some amount of sense logistically for sampling, it is quite concerning statistically because it introduces a potential source of bias in the data. For example, if differences in biotic response (abundance, species composition, etc.) are found between north, central, and southern segments, are these due to the environmental and nitrogen loading gradients characteristic of Barnegat Bay or are they due to the timing over the course of the summer and associated variation in temperatures, salinities, or other seasonally changing variables? Could there be some interaction (combination of influence) between environment and timing, and if so, how much does each contribute? Potentially, we can identify and isolate this bias in a statistical manner, but this is a serious dataset flaw and requires further investigation. It is not guaranteed that such a seasonal bias can be removed from the dataset, severely limiting the interpretation of spatial information.

Going forward with REMAP or other benthic macroinvertebrate dataset collection, we highly recommend not only randomizing the locations of sampling stations within the three north-south segments and two east-west segments, but also randomizing the timing of when sampling occurs at each station. This randomization in the sampling design avoids altogether the potential for both spatial and temporal biases that may otherwise confound interpretation of the data.

- *Can NCA and REMAP datasets be combined? A statistician may be consulted to examine this possibility.*

No. To assess the past conditions of Barnegat Bay (hindcasting), data from each year will be analyzed to provide a score (assessment) for each year. REMAP data is from 2001. Data from 2001 cannot be used to generate assessments for years other than 2001. NCA data are from 2000 to 2006, however, there are only a few data points each year (see Table X). There are not enough NCA data points each year to yield reliable assessment scores. Even if the __ datapoints from the 2001 NCA dataset were to be combined hypothetically, they would have minimal effect on the results while introducing considerable detrimental effect on the reliability of the index used to generate the assessment score.

Recall that the REMAP and NCA datasets were collected for different purposes at different times, and that the objectives of these datasets were different that those of the current project. While the methods for analysis of the measurements for the REMAP and NCA datasets are the same, the data density is different between the two different data sets, and for the purpose of the biotic index of eutrophic condition the data density is paramount. The datasets should not be combined because they were taken at different times with different sampling strategies. Part of what they are trying to do is to come up with an annual index. This is quite important for hindcasting and quite important for the current status of eutrophication condition. For those reasons, the two datasets need to be looked at independently. Thus, the approach to get at the question of is each of the datasets appropriate for inclusion was to look at them independently. We welcome the opportunity to discuss this more with EPA and whatever statisticians they choose to

consult.

- ***Will additional datasets be incorporated as they become available? Specifically, NJ DEP water quality monitoring data from 2011, NCA 2011 data.***

Possibly. Doing so would be an admirable goal to achieve, and we hope to be able to do so, or that eutrophic condition and biotic response could be assessed by this index from such datasets in the future. However, we do not feel ready to commit with certainty until it is clear that doing so will not delay, hinder, or expand the project objectives stated explicitly in the QAPP. Any potential dataset for inclusion in this project will need to be examined for suitability with this project's objectives in the same manner as each of the other datasets.

- ***The TAC believes it will be informative and is eager to see what happens when the data are put into the model and to see the index scores – both with and without various components (e.g. benthic invertebrates). The TAC asked about how far along the researchers are with populating the models.***

We are eager to be moving forward and see the scores and assessments of the biotic index of eutrophic condition as well. We hope that these additional statistical analyses and rationales improve the transparency of the project and its methods. We are ready to move forward with these calculations and look forward to moving beyond the discussion of dataset incorporation.

- ***Given declines of eelgrass biomass, is a shift from parametric to non-parametric statistics necessary to separate out differences between transects?***

No. Shifting from parametric to non-parametric statistics will not provide additional statistical benefit. Transects are appropriately established according to internationally agreed upon seagrass monitoring methodologies across a wide variability of eelgrass abundances. Differences between transects are analyzed statistically according to a variety of methods, as detailed in the QAPP.

- ***Will eelgrass biomass continue to decline to the point where putting it into the model would create problems and therefore a poor indicator due to data paucity?***

It is difficult to predict the future of eelgrass in Barnegat Bay given the high variability associated with seagrass demographics as amply demonstrated in many locations nationally and internationally. While the current trend of *Zostera* biomass is grim, we do not know for certain what will be observed in the future. The model is sensitive in that it treats 'zeroes' and 'missing data' differently. A zero represents an observation of absence. Missing data represents an unknown value. A zero does not contribute to data paucity, while missing data does. Therefore, observations of absence (e.g. 0 g m^{-2} eelgrass biomass) provide important information. Recognizing this important distinction, we are taking care to ensure that values of zero for biomass or other seagrass (and other biotic response) variables are able to be included in the model of assessment of biotic response.

Appendix 3 - 3 SAS code used to assemble the datatsets into the SAS library database


```
*DATASET ASSEMBLY;
*****************;
libname BBdb "U:\My Documents\My SAS Files\9.2\BarnegatDatabase";
libname BBindex "U:\My Documents\My SAS Files\9.2\BBEutroIndex";
libname means "U:\My Documents\My SAS Files\9.2\Means";
libname BBpca "U:\My Documents\My SAS Files\9.2\BBpca";


*A. ECOSYSTEM PRESSURES (Suceptibility)
-Residence Time
data BBdb.residence;
        length estuary $ 55;
        length  residencetime_days $ 3;
        input  estuary $  residencetime_days;
        datalines;
                BarnegatBay 74
                ;
                *Data source: Guo Q., Psuty NP, Lordi GP, Glenn S, Mund MR, Gastrich MD. 2004.
Hydrographic Study of Barnegat Bay. New Jersey Department of Environmental Protection Division of
Science, Research and Technology. Research Project Summary.;


*-Nutrient Loading;
*the following imports data that were produced by USGS for Component 1 of this project.;
data bbdb.HUC14;
*length HUC14 14;
length segment $ 8;
input  segment$ HUC14;
datalines;
South   02040301130010        South   02040301130020        South   02040301130030
South   02040301130040        South   02040301130050        South   02040301130060
South   02040301130070        South   02040301130080        South   02040301140010
South   02040301140020        South   02040301140030        South   02040301140040
South   02040301140050        South   02040301140060        South   02040301920020
South   02040301920030
Central 02040301090010        Central 02040301090020        Central 02040301090030
Central 02040301090040        Central 02040301090050        Central 02040301090060
Central 02040301100010        Central 02040301100020        Central 02040301100030
Central 02040301110010        Central 02040301110020        Central 02040301110030
Central 02040301110040        Central 02040301110050        Central 02040301120010
Central 02040301120020        Central 02040301120030        Central 02040301910030
Central 02040301920010
North   02040301020010        North   02040301020020        North   02040301020030
North   02040301020040        North   02040301020050        North   02040301030010
North   02040301030020        North   02040301030030        North   02040301030040
North   02040301030050        North   02040301040010        North   02040301040020
```

North	02040301040030	North	02040301050010	North	02040301050020
North	02040301050030	North	02040301050040	North	02040301050050
North	02040301060010	North	02040301060020	North	02040301060030
North	02040301060040	North	02040301060050	North	02040301060060
North	02040301060070	North	02040301060080	North	02040301070010
North	02040301070020	North	02040301070030	North	02040301070040
North	02040301070050	North	02040301070060	North	02040301070070
North	02040301070080	North	02040301070090	North	02040301080010
North	02040301080020	North	02040301080030	North	02040301080040
North	02040301080050	North	02040301080060	North	02040301080070
North	02040301080080	North	02040301080090	North	02040301910010
North	02040301910020				

;run;

*NOTE: bbdb.baseflow contains baseflow calculation for TN from USGS.
        The original Excel filename is baseflow_load_TN_calculated.xlsx created on 4/13/2012.
        The file contains the following information:
                HUC14
                HUC14 area (ha)
                Land use year
                Year
                Season
                Measurement
                Value (metric)
                Parameter
                Precipitation (in)
                ;
proc contents data=bbdb.tnbaseflow;run;
*NOTE: bbdb.tpbaseflow contains baseflow calculation for TP from USGS.
        The original Excel filename is baseflow_load_P_calculated_20120510_for_RU.xls created on
5/10/2012.
        The file contains the following information:
                HUC14
                HUC14 area (ha)
                Land use year
                Year
                Season
                Measurement
                Value (metric)
                Units (metric)
                Parameter
                Precipitation (in)
                ;
proc contents data=bbdb.tpbaseflow;run;

*NOTE: bbdb.PLOAD contains the model output for 1989-2011 by USGS for Component 1, calibrated
on April 12, 2012.

The original Excel filename is PLOAD_TN_TP_1989_2011_calibrated_20120412_for_RU.xls created on 4/12/2012.
　　　The file contains the following information:
　　　　　HUC14
　　　　　Parameter
　　　　　Measurement
　　　　　Precipitation (in)
　　　　　Season
　　　　　Units (metric)
　　　　　Value calibrated metric
　　　　　Year
　　　　　;
proc contents data=bbdb.PLOAD;run;
proc contents data=bbdb.tnbaseflow;run;
proc contents data=bbdb.tpbaseflow;run;
proc contents data=bbdb.PLOAD;run;
proc print data=bbdb.tnbaseflow;run;
proc sort data=bbdb.tnbaseflow; by Year Season HUC14 Parameter Measurement;run;
proc sort data=bbdb.tpbaseflow; by Year Season HUC14 Parameter Measurement;run;
proc sort data=bbdb.pload; by Year Season HUC14 Parameter Measurement;run;

data bbdb.usgs;
length Season $ 13;
length Parameter $ 17;
length Measurement $ 24;
merge bbdb.tnbaseflow bbdb.tpbaseflow bbdb.pload;
by Year Season HUC14 Parameter Measurement;
Total = value_metric + value_calibrated_metric;
run;

proc sort data=bbdb.usgs; by HUC14;run;
proc sort data=bbdb.HUC14;by HUC14;run;

data bbdb.usgs2;
merge bbdb.usgs bbdb.HUC14;
by HUC14;
run;

*bbdb.huc12load is based on a summary report from 2009. This was only used for preliminary examination. It is not used for final calculations.;

data BBdb.huc12load;
　　　length huc12name $ 25;
　　　input huc12name　　　$　　　area_km2　　　tn_kgperyear;
　　　datalines;
　　　　　MetedeconkRiver　　　　　185.6　85000
　　　　　TomsRiver　　　　　332.5　170000

```
            WrangleBrook          89.3    39000
            LongSwampCreek                17.4    1700
            JakesBranch                   24.8    5200
            CedarCreek                    137.3   26000
            ForkedRiver                   62.6    14000
            OysterCreek                   33.3    7000
            MillCreek                     59.2    21000
            CedarRun                      21.4    4000
            WestecunkCreek                64.3    20000
            TuckertonCreek        41.2    11000
            ;
            *Data source: Wieben and Baker 2009;
```

*The following identifies which subwatershed the DEP sampling station are located in;

data BBdb.stationHUC12;

```
        length huc12name $ 25;
        input huc12name $ station $;
        datalines;
MetedeconkRiver         1600D
MetedeconkRiver         1601B
MetedeconkRiver         R08
TomsRiver       1502A
TomsRiver       1506A
TomsRiver       1632B
TomsRiver       R11
WrangleBrook    .
LongSwampCreek          .
JakesBranch             .
CedarCreek      1648A
CedarCreek      1648B
CedarCreek      R12
ForkedRiver     1651B
ForkedRiver     1651D
ForkedRiver     1653A
ForkedRiver     1654C
ForkedRiver     1661A
ForkedRiver     1661D
ForkedRiver     1661F
ForkedRiver     1662A
ForkedRiver     R13
ForkedRiver     R14
OysterCreek     1663A
OysterCreek     1670D
OysterCreek     1670F
OysterCreek     1688A
```

OysterCreek 1688B
OysterCreek　　1688C
OysterCreek　　1691A
OysterCreek　　1691E
OysterCreek　　R14A
OysterCreek　　R15
OysterCreek　　R16
MillCreek　　　1700A
MillCreek　　　1703
MillCreek　　　1703C
MillCreek　　　1704
MillCreek　　　1706
CedarRun　　　1707C
CedarRun　　　1718B
CedarRun　　　1718C
CedarRun　　　1719E
CedarRun　　　1721
CedarRun　　　1721C
CedarRun　　　1800B
CedarRun　　　1800D
CedarRun　　　R17
WestecunkCreek　　1712
WestecunkCreek　　R18
TuckertonCreek1818D
TuckertonCreek1820A
TuckertonCreek1824A
TuckertonCreek1824B
TuckertonCreek1826A
TuckertonCreek1826B
TuckertonCreek 1828A
TuckertonCreek1831
TuckertonCreek1834A
TuckertonCreek 1924
TuckertonCreekR19
TuckertonCreekR20
KettleCreek　　1613A
KettleCreek　　R09
SilverBay　　　1604A
SilverBay　　　1605A
SilverBay　　　1609B
SilverBay　　　1615A
SilverBay　　　1617E
SilverBay　　　1618A
SilverBay　　　1622E
SilverBay　　　1627
SilverBay　　　1629
SilverBay　　　1629B

```
SilverBay       1631E
SilverBay       1635E
SilverBay       1636A
SilverBay       1645C
SilverBay       1645G
SilverBay       R10
SilverBay       R10A
GunningRiver    1674B
GunningRiver    1675
GunningRiver    1683C
;
*The following associates loading and subwatershed data;
proc sort data=huc12load; by huc12name;run;
proc sort data=stationHUC12; by huc12name;run;
data BBdb.huc12; merge  huc12load stationHUC12; by   huc12name; run;


*B. ECOSYSTEM STATE
-B.1. Water quality
        -Temperature
        -Dissolved oxygen
        -Total Nitrogen
        -Total Phosphorus
        ;
PROC IMPORT OUT= BBdb.BMW_Nutrients
        DATATABLE= "BMW_Nutrients"
        DBMS=ACCESS REPLACE;
    DATABASE="C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2\nj dep
bmwm\bmw_bb_data.mdb";
    SCANMEMO=YES;
    USEDATE=NO;
    SCANTIME=YES;
RUN;
        *Source: NJ DEP BWM water quality monitoring program
*The following associates subwatershed and loading data with DEP data.
        -select year(s) of data
        -parse out the year, month, sampling period
        -remove stations outside Barnegat Bay - Little Egg Harbor
        -associate station within each segment of BB-LEH;
proc sort data=BBdb.huc12;by station;run;
proc sort data=BBdb.BMW_Nutrients; by station;run;
data BBdb.BMW_NutrientsALL;
set bbdb.BMW_Nutrients;
merge bbdb.BMW_Nutrients bbdb.huc12; by station;
length segment $ 8;
if Characteristic_Row = 'CHLA' then CHLA=Results;
if Characteristic_Row = 'DO' then DO=Results;
```

```
if Characteristic_Row = 'ENT' then ENT=Results;
if Characteristic_Row = 'FC' then FC=Results;
if Characteristic_Row = 'NH3' then NH3=Results;
if Characteristic_Row = 'NO3' then NO3=Results;
if Characteristic_Row = 'PO4' then PO4=Results;
if Characteristic_Row = 'SAL' then SAL=Results;
if Characteristic_Row = 'SECCHI' then SECCHI=Results;
if Characteristic_Row = 'TEMP' then TEMP=Results;
if Characteristic_Row = 'TN' then TN=Results;
if Characteristic_Row = 'TP' then TP=Results;
if Characteristic_Row = 'TSS' then TSS=Results;
year = year(datepart(ActivityStartDate));
month = month(datepart(ActivityStartDate));
if month = 6 then Time_Period=1;
if month = 7 then Time_Period=1;
if month = 8 then Time_Period=2;
if month = 9 then Time_Period=2;
if month = 10 then Time_Period=3;
if month = 11 then Time_Period=3;
if Station eq '1506'  then delete;
if Station eq '1703A' then delete;
if Station eq '1823B' then delete;
if Station eq '1900B' then delete;
if Station eq '1903'  then delete;
if Station eq '1903E' then delete;
if Station eq '1903L' then delete;
if Station eq '1906D' then delete;
if Station eq '1908C' then delete;
if Station eq '1911A' then delete;
if Station eq '1911C' then delete;
if Station eq '1917A' then delete;
if Station eq '1921B' then delete;
if Station eq '1923B' then delete;
if (Station ge '1502A') and (Station le '1632B') then segment = 'North';
if (Station ge 'R08') and (Station le 'R11')  then segment = 'North';
if (Station ge '1635E') and (Station le '1691E') then segment = 'Central';
if (Station ge 'R12') and (Station le 'R16') then segment = 'Central';
if (Station ge '1700A') and (Station le '1924')  then segment = 'South';
if (Station ge 'R17') and (Station le 'R20') then segment = 'South';
if segment ne "";
run;


*B.2. Light availability ;

*-Secchi          -- imported into BMW_Nutrients ;
*-Macroalgae % cover    -- imported into the SAV files;
```

```
*-Chlorophyll;
filename inf 'C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\bmw_aircraftremotesensing.dbf';
proc dbf db4=inf out=chl_depaircraft;run;
data BBdb.chl_depaircraft;
set  chl_depaircraft;
Latitude = Lat * 10;
Longitude = Long * 10;
drop Lat Long;
run;
        *Source: NJ DEP BWM chlorophyll remote sensing by aircraft;


*-Epiphytes  -- from field sampling along with SAV - data from Kennish et al. ;
PROC IMPORT OUT= BBDB.epiphytes2009
        DATAFILE= "C:\Documents and Settings\Administrator\My Docume
nts\My SAS Files\9.2\epiphytes 2009.xls"
        DBMS=EXCEL REPLACE;
    RANGE="Sheet1$";
    GETNAMES=YES;
    MIXED=YES;
    SCANTEXT=YES;
    USEDATE=YES;
    SCANTIME=YES;
RUN;
PROC IMPORT OUT= BBDB.epiphytes2010
        DATAFILE= "C:\Documents and Settings\Administrator\My Docume
nts\My SAS Files\9.2\epiphytes 2010.xls"
        DBMS=EXCEL REPLACE;
    RANGE="Sheet1$";
    GETNAMES=YES;
    MIXED=YES;
    SCANTEXT=YES;
    USEDATE=YES;
    SCANTIME=YES;
RUN;
data epiphytes2009_1;
set bbdb.epiphytes2009;
Year = 2009;
Station_char=put(Station,8.0);
run;
data epiphytes2010_1;
set bbdb.epiphytes2010;
Year = 2010;
Station_char=put(Station,8.0);
run;
```

```
proc contents data=epiphytes2009_1;run;
proc sort data=epiphytes2009_1; by Year Time_Period Transect Station;run;
proc sort data=epiphytes2010_1; by Year Time_Period Transect Station;run;
data epiphytes;
length Segment $ 8;
merge epiphytes2009_1 epiphytes2010_1;
by Year Time_Period Transect Station;
if Transect ge 1 and Transect le 6 then Segment = 'South';
if Transect ge 7 and Transect le 12 then Segment = 'Central';
if Transect ge 13 and Transect le 15 then Segment = 'North';
Epiphyte_biomass_mg = Epiphyte_biomass*1000;
run;
data epiphytes2;
set epiphytes;
drop Station;
run;
data epiphytes3;
set epiphytes2;
rename Station_char=Station;
run;
data bbdb.epiphytes;
set epiphytes3;
if Transect ne .;
run;
        *Source: Mike Kennish (Rutgers), Gregg Sakowicz (JCNERR);


*C. ECOSYSTEM BIOTIC RESPONSE
        C.1. -SAV
                -aboveground biomass
                -belowground biomass
                -shoot density
                -blade height
                -Zostera % cover
                ;
PROC IMPORT OUT= WORK.SAV2004import
        DATATABLE= "SAV2004"
        DBMS=ACCESS REPLACE;
    DATABASE="C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\savfieldwork2004-2009_sent1.mdb";
    SCANMEMO=YES;
    USEDATE=NO;
    SCANTIME=YES;
RUN;
data BBdb.sav2004;
length Station $ 8;
length Boat_Scarring Grazing Wasting_disease Epiphytes $ 2;
```

```
format Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
informat Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
set sav2004import;
Depth_cm=Depth*100;
DO_percent = DO_per;
DO_mgL = Do_mg_l;
Zostera_aboveground_biomass_gm2= above_ground_grams_m2;
Zostera_belowground_biomass_gm2= below_ground_grams_m2;
Zostera_percentcover = Z_marina;
Zostera_bladelength = mean(Sav_Height1, Sav_Height2, Sav_Height3, Sav_Height4, Sav_Height5);
Zostera_shootdensity = Stems_in_Core / 0.007853982;
Ruppia_percentcover = R_Maritima;
Macroalgae_percentcover = Per_cover_Macro_Algae;
Boat_Scarring = Boat_Scaring;
PH=pH;
Epiphytes = epiphytes;
Wasting_disease=wasting_disease;
Stationchar = substr(station_id,5,6);
Station=input(Stationchar,best2.);
drop station_id;
run;
PROC IMPORT OUT= WORK.SAV2005import
        DATATABLE= "SAV2005"
        DBMS=ACCESS REPLACE;
    DATABASE="C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\savfieldwork2004-2009_sent1.mdb";
    SCANMEMO=YES;
    USEDATE=NO;
    SCANTIME=YES;
RUN;
data BBdb.sav2005;
length Station $ 8;
length Boat_Scarring Grazing Wasting_disease Epiphytes $ 2;
format Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
informat Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
set sav2005import;
DO_percent = DO_per;
DO_mgL = Do_mg_l;
PH=pH;
Zostera_aboveground_biomass_gm2= above_ground_grams_m2z;
Zostera_belowground_biomass_gm2= below_ground_grams_m2z;
Zostera_percentcover = Z_marina;
Zostera_bladelength = mean(Sav_Height1, Sav_Height2, Sav_Height3, Sav_Height4, Sav_Height5);
Zostera_shootdensity = Stems_in_Core_z / 0.007853982;
Ruppia_aboveground_biomass_gm2 = ruppia_above;
Ruppia_belowground_biomass_gm2 = ruppia_below;
Ruppia_percentcover = R_Maritima;
```

```
Ruppia_shootdensity = Stems_in_Core_r / 0.007853982;
Macroalgae_percentcover = Per_cover_Macro_Algae;
Epiphytes = epiphytes;
Wasting_disease=wasting_disease;
Boat_Scarring = Boat_Scaring;
Stationchar = substr(station_id,5,6);
Station=input(Stationchar,best2.);
drop station_id;
run;
PROC IMPORT OUT= WORK.SAV2006_CSV
        DATAFILE= "\\ad-rsc\data\users\bfertig\My Documents\My SAS F
iles\9.2\SAV2006.csv"
        DBMS=CSV REPLACE;
   GETNAMES=YES;
    DATAROW=2;
RUN;
data sav2006_csv2;
informat        Station $8.;
informat        Date___Time datetime19.;
format          Date___Time datetime19.;
set sav2006_csv;
Station = Station_num;
Date___Time=DHMS(Date_Excel,0,0,Time_Excel);
run;
data bbdb.SAV2006;
length Station $ 8;
length Boat_Scarring Grazing Wasting_disease Epiphytes $ 2;
format Boat_Scarring Grazing Wasting_disease Epiphytes $2. ;
informat Boat_Scarring Grazing Wasting_disease Epiphytes $2. ;
set sav2006_csv2;
Boat_Scarring = Boat_Scaring;
DO_percent = DO_per;
DO_mgL = Do_mg_l;
Secchi_cm = secchi;
PH=pH;
Zostera_aboveground_biomass_gm2= above_ground_grams_m2;
Zostera_belowground_biomass_gm2= below_ground_grams_m2;
Zostera_percentcover = Z_marina;
Zostera_bladelength = mean(Sav_Height1, Sav_Height2, Sav_Height3, Sav_Height4, Sav_Height5);
Zostera_shootdensity = Stems_in_Core_z / 0.007853982;
Ruppia_percentcover = R_Maritima;
Ruppia_shootdensity = Stems_in_Core_r / 0.007853982;
Macroalgae_percentcover = Per_cover_Macro_Algae;
Epiphytes = epiphytes;
Wasting_disease=wasting_disease;
Time_Period = sample_period;
drop date Sav_Height1_lab Sav_Height2_lab Sav_Height3_lab Sav_Height4_lab Sav_Height5_lab utmx
```

```
utmy levels line Turbidity;
run;
PROC IMPORT OUT= WORK.SAV2008import
       DATATABLE= "SAV2008"
       DBMS=ACCESS REPLACE;
   DATABASE="C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\savfieldwork2004-2009_sent1.mdb";
   SCANMEMO=YES;
   USEDATE=NO;
   SCANTIME=YES;
RUN;
data BBdb.sav2008;
length Station $ 8;
length Boat_Scarring Grazing Wasting_disease Epiphytes $ 2;
format Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
informat Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
set sav2008import;
where Transect ne 0;
Boat_Scarring = Boat_Scaring;
DO_percent = DO_per;
DO_mgL = Do_mg_l;
Secchi = secchi;
PH=pH;
Zostera_aboveground_biomass_gm2= above_ground_grams_m2;
Zostera_belowground_biomass_gm2= below_ground_grams_m2;
Zostera_percentcover = Z_marina;
Zostera_bladelength = mean(Sav_Height1, Sav_Height2, Sav_Height3, Sav_Height4, Sav_Height5);
Zostera_shootdensity = stems_in_core_z___m_2;
Ruppia_percentcover = R_Maritima;
Ruppia_shootdensity = Stems_in_Core_r / 0.007853982;
Macroalgae_percentcover = Per_cover_Macro_Algae;
Epiphytes = epiphytes;
Wasting_disease=wasting_disease;
Month = month(datepart(Date___Time));
if Month = 6 then Time_Period=1;
if Month = 7 then Time_Period=1;
if Month = 8 then Time_Period=2;
if Month = 9 then Time_Period=3;
if Month = 10 then Time_Period=3;
if Month = 11 then Time_Period=3;
Stationchar = substr(station_id,5,6);
Station=input(Stationchar,best2.);
drop station_id;
drop date;
run;
PROC IMPORT OUT= WORK.SAV2009import
       DATATABLE= "SAV2009"
```

```
        DBMS=ACCESS REPLACE;
    DATABASE="C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\savfieldwork2004-2009_sent1.mdb";
    SCANMEMO=YES;
    USEDATE=NO;
    SCANTIME=YES;
RUN;
data BBdb.sav2009;
length Station $ 8;
length Boat_Scarring Grazing Wasting_disease Epiphytes $ 2;
format Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
informat Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
set sav2009import;
Boat_Scarring = Boat_Scaring;
DO_percent = DO_per;
DO_mgL = Do_mg_l;
Secchi_cm = secchi;
PH=pH;
Zostera_aboveground_biomass_gm2= above_ground_grams_m2;
Zostera_belowground_biomass_gm2= below_ground_grams_m2;
Zostera_percentcover = Z_marina;
Zostera_bladelength = mean(Sav_Height1, Sav_Height2, Sav_Height3, Sav_Height4, Sav_Height5)/10;
Zostera_shootdensity = stems_in_core_z___m_2;
Ruppia_percentcover = R_Maritima;
Ruppia_shootdensity = Stems_in_Core_r / 0.007853982;
Epiphytes = epiphytes;
Wasting_disease=wasting_disease;
Macroalgae_percentcover = Per_cover_Macro_Algae;
Stationchar = substr(station_id,5,6);
Station=input(Stationchar,best2.);
drop station_id;
run;

PROC IMPORT OUT= WORK.SAV2010_CSV
        DATAFILE= "\\ad-rsc\data\users\bfertig\My Documents\My SAS F
iles\9.2\SAV2010.csv"
        DBMS=CSV REPLACE;
    GETNAMES=YES;
    DATAROW=2;
RUN;

data sav2010convert;
informat Station $8.;
set SAV2010_CSV;
RAbove=input(Above_ground_grams_m2r,best.);
RBelow=input(Below_ground_grams_m2r,best.);
Station = Station_num;
```

```
Secchi_num=input(Secchi_cm,best.);
drop Secchi_cm;
run;
data bbdb.sav2010;
length Boat_Scarring Grazing Wasting_disease Epiphytes $ 2;
format Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
informat Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
informat Date___Time datetime19.;
format Date___Time datetime19.;
set sav2010convert;
Date___Time=DHMS(Date,0,0,Time);
format Date___Time datetime19.;
DO_mgL = DOmg;
Conductivity=SpCond;
Temperature=Temp;
PH=pH;
Zostera_aboveground_biomass_gm2= Above_ground_grams_m2z;
Zostera_belowground_biomass_gm2= Below_ground_grams_m2z;
Zostera_percentcover = Percent_Cover_Zostera;
Zostera_bladelength = (mean(Length1, Length2, Length3, Length4, Length5))/10;
Zostera_shootdensity = Z_number_of_stems / 0.007853982;
Ruppia_aboveground_biomass_gm2 = RAbove;
Ruppia_belowground_biomass_gm2 = RBelow;
Ruppia_percentcover = Percent_Cover_Ruppia;
Ruppia_shootdensity = R___stems / 0.007853982;
Macroalgae_percentcover = Percent_Cover_Macroalgae;
Other_percentcover = Percent_Cover_Other;
Epiphytes = Epiphyte;
run;
PROC IMPORT OUT= WORK.SAV2011_CSV
        DATAFILE= "X:\projects\barnegat_bay\sav\databases\2011\SAV2011 for sas.csv"
        DBMS=CSV REPLACE;
    GETNAMES=YES;
    DATAROW=2;
RUN;
```

*Source: Mike Kennish (Rutgers), Gregg Sakowicz (JCNERR);

```
data bbdb.savALL;
length Station Segment $ 8;
format Date DATE9.;
retain
Year
Time_Period
date
Date___Time
Transect
```

Station
Segment
Zostera_aboveground_biomass_gm2
Zostera_belowground_biomass_gm2
Zostera_percentcover
Zostera_bladelength
Zostera_shootdensity
Ruppia_aboveground_biomass_gm2
Ruppia_belowground_biomass_gm2
Ruppia_percentcover
Ruppia_shootdensity
Macroalgae_percentcover
Other_percentcover
Epiphytes
Grazing
Wasting_disease
Temperature
Salinity
Conductivity
PH
DO_mgL
DO_percent
Secchi_cm
Boat_Scarring
;
length Boat_Scarring Grazing Wasting_disease Epiphytes $ 2;
format Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
informat Boat_Scarring Grazing Wasting_disease Epiphytes $2.;
set bbdb.sav2004 bbdb.sav2005 bbdb.sav2006 bbdb.sav2008 bbdb.sav2009 bbdb.sav2010;
Date=datepart(Date___Time);
Year = year(datepart(Date___Time));
Month = month(datepart(Date___Time));
if Month = 6 then Time_Period=1;
if Month = 7 then Time_Period=1;
if Month = 8 then Time_Period=2;
if Year ne 2008 and Month = 9 then Time_Period=2;
if Year eq 2008 and Month = 9 then Time_Period=3;
if Month = 10 then Time_Period=3;
if Month = 11 then Time_Period=3;
Transect=transect;
if Transect ge 1 and Transect le 6 then Segment = 'South';
if Transect ge 7 and Transect le 12 then Segment = 'Central';
if Transect ge 13 and Transect le 15 then Segment = 'North';
keep Year Time_Period Date Date___Time Transect Station Segment
Zostera_aboveground_biomass_gm2 Zostera_belowground_biomass_gm2 Zostera_percentcover
 Zostera_bladelength Zostera_shootdensity Ruppia_aboveground_biomass_gm2
Ruppia_belowground_biomass_gm2

```
 Ruppia_percentcover Ruppia_shootdensity Macroalgae_percentcover Other_percentcover Epiphytes
Grazing Wasting_disease Temperature Salinity Conductivity PH DO_mgL DO_percent Secchi_cm
Boat_Scarring;
drop imagery biomasscollected;run;

proc contents data=bbdb.savall;run;
proc contents data=bbdb.sav2004;run;
proc contents data=bbdb.sav2005;run;
proc contents data=bbdb.sav2006;run;
proc contents data=bbdb.sav2008;run;
proc contents data=bbdb.sav2009;run;
proc contents data=bbdb.sav2010;run;
proc contents data=bbdb.savall;run;

*C.2. -HABS (concentration);
data BBdb.browntide;
input year        minmax? $        cells_per_ml ;
        datalines;
        1988    max             35000
        1995    approx  1000000
        1997    .               .
        1999    min             1800000
        2000    min             1800000
        2001    min             1800000
        2002    min             1500000
        2005    max     50000
        2010    approx  158000
        ;
        *Source:

*C.3. -Benthic Invertebrates
        -hard clam landings - pounds, value;

PROC IMPORT OUT= BBdb.clams
        DATAFILE= "C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\Ocean COunty hard clam landings.xls"  DBMS=EXCEL REPLACE;
    RANGE="data$"; GETNAMES=YES;MIXED=YES;SCANTEXT=YES;USEDATE=YES;
SCANTIME=YES;
RUN;
        *source: 1960-1980: Watershed Management Plan 1993 from NMFS data.
                        1990-2005: Calvo inquire to Gaipo - NMFS data;

*C.4. -NCA & REMAP;
*----REMAP---;
PROC IMPORT OUT=REMAP_physical
        DATAFILE= "C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\REMAPall.xls" DBMS=EXCEL REPLACE; RANGE="physical$";
```

```
GETNAMES=YES;MIXED=YES;SCANTEXT=YES;USEDATE=YES;SCANTIME=YES;
RUN;

PROC IMPORT OUT=REMAP_sedtox
        DATAFILE= "C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\REMAPall.xls"
                        DBMS=EXCEL REPLACE;
                RANGE="Sed_tox$";
GETNAMES=YES;MIXED=YES;SCANTEXT=YES;USEDATE=YES;SCANTIME=YES;
RUN;

PROC IMPORT OUT=REMAP_sedchem
        DATAFILE= "C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\REMAPall.xls"
                        DBMS=EXCEL REPLACE;
                RANGE="sedchem$";
GETNAMES=YES;MIXED=YES;SCANTEXT=YES;USEDATE=YES;SCANTIME=YES;
RUN;

PROC IMPORT OUT=REMAP_taxonomy1
        DATAFILE= "C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\REMAPall.xls"
                        DBMS=EXCEL REPLACE;
                RANGE="taxonomy1$";
GETNAMES=YES;MIXED=YES;SCANTEXT=YES;USEDATE=YES;SCANTIME=YES;
RUN;

PROC IMPORT OUT=REMAP_taxonomy2
        DATAFILE= "C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\REMAPall.xls"
                        DBMS=EXCEL REPLACE;
                RANGE="taxonomy2$";
GETNAMES=YES;MIXED=YES;SCANTEXT=YES;USEDATE=YES;SCANTIME=YES;
RUN;

PROC IMPORT OUT=REMAP_abundance
        DATAFILE= "C:\Documents and Settings\Administrator\My Documents\My SAS Files\9.2
\REMAPall.xls"
                        DBMS=EXCEL REPLACE;
                RANGE="abundance$";
GETNAMES=YES;MIXED=YES;SCANTEXT=YES;USEDATE=YES;SCANTIME=YES;
RUN;
        *Source: Darvene Adams (EPA) and Bob Schuster (NJ DEP);

proc sort data=REMAP_physical; by SITE_ID; run;
proc sort data=REMAP_sedtox; by SITE_ID; run;
proc sort data=REMAP_sedchem; by SITE_ID; run;
```

```
proc sort data=REMAP_taxonomy1; by SITE_ID; run;
proc sort data=REMAP_abundance; by SITE_ID; run;

data BBDB.REMAP;
merge REMAP_physical REMAP_sedtox REMAP_sedchem REMAP_taxonomy1 REMAP_abundance;
by SITE_ID
run;
```

Appendix 3 - 4 Complete dataset summarized by Year and Segment

		Pressure					Water Quality	
		Total Loading				Mean		
		Total	Total					
Year	Segment	Nitrogen	Phosphorus	DO	Temp	TN	TP	
		kg km^{-2} yr^{-1}	kg km^{-2} yr^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}	n
1989	North	7844.5	370.9	8.8	17.5	828.1	NoData	
1990	North	7067.2	329.8	8.7	17.2	593.1	NoData	
1991	North	6466.3	299.8	10.9	5.0	368.8	NoData	
1992	North	5494.3	259.2	NoData	NoData	NoData	NoData	N
1993	North	7240.2	327.7	6.5	15.4	498.7	NoData	
1994	North	7867.8	356.5	7.5	20.4	370.1	NoData	
1995	North	5358.3	248.6	7.9	15.7	366.8	NoData	
1996	North	8816.6	403.6	6.2	23.0	726.5	NoData	
1997	North	7599.6	345.6	8.9	12.7	646.5	NoData	
1998	North	8460.2	379.1	7.7	12.2	758.4	NoData	
1999	North	6560.3	297.8	7.6	12.2	631.7	15.4	
2000	North	6675.7	303.9	6.8	14.8	575.0	23.2	
2001	North	5996.5	264.0	7.8	11.2	522.0	21.8	
2002	North	5446.6	258.5	7.5	12.3	585.5	32.7	
2003	North	8805.1	391.0	8.1	14.5	489.7	31.3	
2004	North	7749.3	343.6	7.7	15.0	648.6	34.1	
2005	North	8448.8	366.6	7.3	16.1	628.4	28.9	
2006	North	8859.7	385.2	7.9	14.8	611.7	23.6	
2007	North	8016.4	346.1	7.8	13.2	524.4	24.3	
2008	North	7085.8	315.8	8.7	8.7	551.3	31.0	
2009	North	8849.2	395.1	7.3	14.0	789.0	28.1	
2010	North	8882.0	376.0	7.4	14.6	852.0	33.7	

		Pressure				Water Quality		
		Total Loading			Mean			
		Total	Total					
Year	Segment	Nitrogen	Phosphorus	DO	Temp	TN	TP	
		$kg\ km^{-2}\ yr^{-1}$	$kg\ km^{-2}\ yr^{-1}$	$mg\ L^{-1}$	C	$\mu g\ L^{-1}$	$\mu g\ L^{-1}$	n
1989	Central	936.3	45.2	7.9	20.0	638.3	NoData	
1990	Central	846.7	40.7	8.0	17.2	470.1	NoData	
1991	Central	773.0	37.0	11.0	7.1	532.1	NoData	
1992	Central	653.4	31.5	NoData	NoData	NoData	NoData	N
1993	Central	871.6	41.4	5.9	14.4	381.4	NoData	
1994	Central	946.8	45.0	6.4	19.7	399.9	NoData	
1995	Central	640.3	30.7	7.8	15.8	365.4	NoData	
1996	Central	1057.9	50.5	4.7	14.7	460.2	NoData	
1997	Central	913.6	43.5	8.3	12.6	355.8	NoData	
1998	Central	1021.4	48.4	7.3	14.4	474.7	NoData	
1999	Central	781.7	37.0	8.6	15.1	364.1	11.2	
2000	Central	794.8	37.6	8.4	14.6	377.1	30.1	
2001	Central	720.7	33.7	8.4	11.7	268.5	20.7	
2002	Central	640.5	30.8	8.9	14.0	606.0	34.5	
2003	Central	1055.8	49.5	7.9	18.2	366.6	56.7	
2004	Central	929.6	43.6	7.9	15.1	351.6	56.5	
2005	Central	1022.7	47.3	8.4	13.6	309.2	19.8	
2006	Central	1071.8	49.6	8.7	12.0	334.9	23.2	
2007	Central	971.6	44.9	7.8	16.1	371.7	29.0	
2008	Central	851.2	39.8	9.1	10.9	374.4	31.4	
2009	Central	1062.6	49.7	7.6	12.6	346.6	29.4	
2010	Central	1082.2	49.6	6.8	15.2	425.5	38.6	

		Pressure				Water Quality		
		Total Loading			Mean			
		Total	Total					
Year	Segment	Nitrogen	Phosphorus	DO	Temp	TN	TP	
		kg km^{-2} yr^{-1}	kg km^{-2} yr^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}	n
1989	South	1144.5	53.2	8.0	19.3	681.8	NoData	
1990	South	1033.5	47.7	8.5	13.3	496.2	NoData	
1991	South	939.2	42.9	9.8	7.5	183.9	NoData	
1992	South	795.5	36.7	NoData	NoData	NoData	NoData	N
1993	South	1056.0	47.7	7.5	16.0	359.8	NoData	
1994	South	1147.3	51.9	5.1	15.5	323.2	NoData	
1995	South	778.1	35.6	5.8	17.1	653.4	NoData	
1996	South	1283.3	58.3	4.0	12.2	345.9	NoData	
1997	South	1107.5	50.2	6.2	18.5	390.0	NoData	
1998	South	1236.0	55.6	7.8	15.5	612.4	NoData	
1999	South	980.9	44.2	7.9	16.7	495.4	28.2	
2000	South	997.7	45.1	8.1	16.0	447.4	42.2	
2001	South	901.3	39.9	8.3	14.5	445.9	46.1	
2002	South	808.0	37.4	8.4	12.2	748.7	58.8	
2003	South	1321.6	58.9	7.4	18.5	346.2	81.3	
2004	South	1163.4	51.8	8.5	11.6	347.6	44.1	
2005	South	1290.7	56.2	8.9	10.3	291.5	34.9	
2006	South	1353.0	59.0	8.9	10.3	336.6	24.2	
2007	South	1225.5	53.2	8.9	17.7	470.2	46.8	
2008	South	1078.1	47.7	9.7	7.2	347.7	31.6	
2009	South	1346.1	59.6	7.7	11.8	372.9	39.2	
2010	South	1361.7	58.4	6.4	16.1	400.9	47.6	

			Mean					Light Availability	Sta		
Year	Segment	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light	Secchi	TSS	Chl a	N
			mg L^{-1}	µg L^{-1}	%		%		mg L^{-1}	µg L^{-1}	
1989	North	3.1	12.9	NoData	NoData	1.5	2.6	0.4	1.2	NoData	
1990	North	3.5	25.6	NoData	NoData	2.0	0.9	0.3	1.3	NoData	
1991	North	7.2	41.7	NoData	NoData	NoData	NoData	0.5	6.8	NoData	
1992	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1993	North	4.3	29.2	NoData	NoData	2.1	0.7	0.3	1.6	NoData	
1994	North	2.7	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	
1995	North	2.8	NoData	NoData	NoData	NoData	NoData	0.2	NoData	NoData	
1996	North	2.4	NoData	NoData	NoData	NoData	NoData	0.1	NoData	NoData	
1997	North	3.4	24.5	NoData	NoData	0.9	15.5	0.2	2.3	NoData	
1998	North	3.6	15.7	6.8	NoData	1.2	9.1	0.2	1.2	1.4	
1999	North	3.0	20.0	5.3	NoData	1.0	13.6	0.1	6.1	0.6	
2000	North	3.0	12.5	2.2	NoData	1.2	9.5	0.2	0.9	0.2	
2001	North	3.9	11.8	2.1	NoData	1.0	10.9	0.3	1.2	0.2	
2002	North	2.9	14.5	4.2	NoData	1.0	10.1	0.2	1.2	0.6	
2003	North	3.0	8.8	3.7	NoData	0.9	14.0	0.2	0.8	0.6	
2004	North	2.2	9.0	6.2	NoData	0.7	16.9	0.1	1.4	0.8	
2005	North	2.1	7.2	7.4	NoData	0.7	19.1	0.2	0.7	0.9	
2006	North	2.8	6.3	4.0	NoData	0.6	19.8	0.2	0.6	0.5	
2007	North	3.7	11.7	5.6	NoData	1.0	14.4	0.3	1.4	0.5	
2008	North	5.0	11.3	3.1	NoData	1.0	9.5	0.3	1.0	0.3	
2009	North	5.7	13.8	4.4	NoData	0.8	8.6	0.4	3.2	0.6	
2010	North	5.8	14.2	7.7	NoData	0.8	7.8	0.5	1.0	0.8	

							Light Availability				
				Mean							Sta
Year	Segment	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light	Secchi	TSS	Chl a	M
			mg L^{-1}	µg L^{-1}	%		%		mg L^{-1}	µg L^{-1}	
1989	Central	3.5	14.4	NoData	NoData	NoData	NoData	0.4	2.5	NoData	
1990	Central	3.4	30.1	NoData	NoData	2.1	1.2	0.2	1.4	NoData	
1991	Central	4.1	30.3	NoData	NoData	NoData	NoData	0.3	1.9	NoData	
1992	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1993	Central	3.9	36.9	NoData	NoData	1.7	2.6	0.3	1.7	NoData	
1994	Central	3.0	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	
1995	Central	2.7	NoData	NoData	NoData	NoData	NoData	0.1	NoData	NoData	
1996	Central	4.4	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	
1997	Central	3.2	29.8	NoData	NoData	0.6	19.8	0.1	2.6	NoData	
1998	Central	2.0	5.7	2.8	NoData	0.5	26.3	0.2	1.6	1.0	
1999	Central	2.7	76.4	6.1	NoData	0.7	20.3	0.2	32.7	0.8	
2000	Central	3.5	16.9	2.2	NoData	1.1	14.7	0.2	1.5	0.2	
2001	Central	3.8	16.3	1.5	NoData	1.3	11.5	0.3	2.1	0.2	
2002	Central	2.7	28.4	4.1	NoData	1.3	13.2	0.2	4.4	0.5	
2003	Central	4.5	12.2	2.9	NoData	0.6	23.2	0.2	1.5	0.5	
2004	Central	3.6	15.6	5.2	NoData	0.8	16.7	0.3	2.3	0.8	
2005	Central	3.9	7.0	4.3	7.8	0.6	21.8	0.4	0.9	0.7	
2006	Central	4.4	13.5	3.5	4.7	0.9	16.2	0.4	2.5	0.4	
2007	Central	3.9	12.0	6.0	NoData	0.8	20.4	0.4	1.4	0.7	
2008	Central	4.3	15.9	3.0	13.3	0.4	21.2	0.2	1.8	0.3	
2009	Central	8.1	15.2	3.3	2.5	0.3	24.1	0.4	1.0	0.3	
2010	Central	4.8	18.7	7.3	1.3	0.3	23.8	0.2	1.2	0.5	

					Light Availability				Sta
				Mean					
Year	Segment	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light	Secchi	TSS	Chl a	N
			mg L^{-1}	µg L^{-1}	%		%		mg L^{-1}	µg L^{-1}	
1989	South	2.1	19.1	NoData	NoData	2.4	0.3	0.1	1.9	NoData	
1990	South	3.3	38.7	NoData	NoData	2.3	0.4	0.2	1.3	NoData	
1991	South	3.8	34.5	NoData	NoData	NoData	NoData	1.1	1.4	NoData	
1992	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1993	South	3.7	42.0	NoData	NoData	NoData	NoData	0.2	1.5	NoData	
1994	South	3.1	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	
1995	South	3.2	NoData	NoData	NoData	NoData	NoData	0.2	NoData	NoData	
1996	South	4.0	NoData	NoData	NoData	NoData	NoData	0.1	NoData	NoData	
1997	South	4.7	12.8	4.9	NoData	NoData	NoData	0.7	5.9	2.0	
1998	South	3.1	12.4	12.6	NoData	0.7	23.1	0.3	2.1	2.3	
1999	South	3.3	26.2	14.0	NoData	1.6	5.1	0.2	2.3	3.1	
2000	South	3.3	21.2	3.7	NoData	1.6	7.1	0.2	1.0	0.5	
2001	South	3.8	22.3	2.4	NoData	1.3	6.5	0.2	1.6	0.4	
2002	South	2.9	27.5	3.5	NoData	0.9	19.7	0.1	2.0	0.5	
2003	South	3.6	19.4	2.5	NoData	0.2	34.1	0.3	2.6	0.6	
2004	South	5.3	11.9	3.2	15.9	0.8	9.6	0.3	1.2	0.4	
2005	South	4.5	22.7	3.5	NoData	1.3	4.1	0.5	4.2	0.5	
2006	South	4.5	14.6	3.0	5.8	1.6	7.0	0.2	1.1	0.3	
2007	South	4.0	14.1	4.5	NoData	0.8	17.9	0.4	1.7	0.7	
2008	South	4.1	17.0	3.2	10.0	1.6	5.6	0.2	1.9	0.6	
2009	South	6.3	21.2	1.7	12.4	0.2	32.4	0.3	1.6	0.2	
2010	South	4.9	22.4	5.6	7.8	0.1	31.9	0.3	1.3	0.8	

						Seagrass			Sta
		Biomass above	Biomass below	*Mean* Shoot density	Seagrass cover	Blade length	Biomass above	Biomass below	
Year	Segment								
		g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm	g m^{-2}	g m^{-2}	sl
1989	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1990	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1991	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1992	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1993	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1994	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1995	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1996	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1997	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1998	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1999	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2000	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2001	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2002	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2003	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2004	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2005	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2006	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2007	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2008	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2009	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2010	North	NoData	NoData	NoData	NoData	NoData	NoData	NoData	

		Seagrass						Star
Year	Segment	Biomass above	Biomass below	Mean Shoot density	Seagrass cover	Blade length	Biomass above	Biomass below
		g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm	g m^{-2}	g m^{-2}	sl
1989	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1990	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1991	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1992	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1993	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1994	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1995	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1996	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1997	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1998	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
1999	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2000	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2001	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2002	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2003	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2004	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2005	Central	32.2	84.8	297.1	23.8	29.1	4.0	10.4	
2006	Central	17.1	46.0	181.0	19.5	12.4	2.8	6.6	
2007	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	
2008	Central	26.9	66.1	346.7	25.0	31.7	4.4	11.5	
2009	Central	9.1	31.4	260.1	22.5	24.8	1.9	3.8	
2010	Central	9.1	29.5	506.0	18.2	27.7	1.6	3.8	

					Seagrass			Stai
				Mean				
		Biomass	Biomass	Shoot	Seagrass	Blade	Biomass	Biomass
Year	Segment	above	below	density	cover	length	above	below
		g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm	g m^{-2}	g m^{-2}	sl
1989	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1990	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1991	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1992	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1993	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1994	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1995	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1996	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1997	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1998	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1999	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2000	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2001	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2002	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2003	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2004	South	60.8	76.5	134.4	34.6	32.7	4.6	6.6
2005	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2006	South	8.4	43.0	145.4	16.1	8.0	1.3	8.8
2007	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2008	South	25.4	65.0	323.4	28.1	29.9	5.0	9.6
2009	South	8.2	33.5	250.9	26.1	20.7	1.3	3.8
2010	South	5.9	23.9	527.5	20.8	17.1	0.9	3.1

Appendix 3 - 5 SAS Code used for index calculations

```
*******************;
*INDEX CALCULATION*;
*******************;
libname BBdb "U:\My Documents\My SAS Files\9.2\BarnegatDatabase";
libname BBindex "U:\My Documents\My SAS Files\9.2\BBEutroIndex";
libname means "U:\My Documents\My SAS Files\9.2\Means";
libname BBpca "U:\My Documents\My SAS Files\9.2\BBpca";

**STEP 1: COMPARE DATA TO THRESHOLD EQUATIONS TO CALCULATE
```

INDICATOR SCORES;

```
*A. Ecosystem Pressures;
title1 Pressures;
proc sort data=bbdb.TotalLoadKgKm2;
by Year Season Segment;
run;
data bbindex.A1pressurescore;
set bbdb.TotalLoadKgKm2;
if Season='All' then do;
        if TN_TotLoadKgKm2 le 50 then tnloadscore=100; else
        if TN_TotLoadKgKm2 ge 10000 then tnloadscore=0; else
        tnloadscore = -19*log(TN_TotLoadKgKm2) + 177.52;

        if TP_TotLoadKgKm2 le 25  then tploadscore=100; else
        if TP_TotLoadKgKm2 ge 500        then tploadscore=0;  else
        tploadscore = -32.81*log(TP_TotLoadKgKm2) + 204.01;
        end;
run;
proc print data=bbindex.A1pressurescore;
where Season='All';
var Year Segment TN_TotLoadKgKm2 TP_TotLoadKgKm2 tnloadscore tploadscore;
run;

*----------------------------------------------------------------------------;


*B. Ecosystem State;
title1 Water Quality;
data BBindex.B1wqvar_scores_raw;
set BBdb.BMW_NutrientsALL;
where month ge 4 and month le 10  ;

if Characteristic_Row = 'TEMP' and Results le 18.0 then tempscore=50; else
if Characteristic_Row = 'TEMP' and Results > 34.0  then tempscore=0; else
if Characteristic_Row = 'TEMP' then tempscore = -3.125*Results + 106.25;

if Characteristic_Row = 'DO' then doexp = 0.228*Results;
if Characteristic_Row = 'DO' and Results ge 10.0 then doscore=50; else
if Characteristic_Row = 'DO' and Results <  4.0 then doscore=0; else
if Characteristic_Row = 'DO' then doscore = 4.8641*exp(doexp);
```

```
if Characteristic_Row = 'TN' and Results < 135 then tnscore=50; else
if Characteristic_Row = 'TN' and Results > 750 then tnscore=0; else
if Characteristic_Row = 'TN' then tnscore = 26721*(Results**(-1.274));

if Characteristic_Row = 'TP' and Results < 10      then tpscore=50; else
if Characteristic_Row = 'TP' and Results > 45 then tpscore=0; else
if Characteristic_Row = 'TP' then tpscore = 475.95*(Results**-0.977);

keep
year month segment station
Characteristic_Row Results
TEMP DO TN TP
tempscore doscore tnscore tpscore;
run;
proc sort data=BBindex.B1wqvar_scores_raw; by Year Segment;run;
proc means data=BBindex.B1wqvar_scores_raw n mean median stddev min max;
by Year Segment;
where month ge 4 and month le 10;
var tempscore doscore tnscore tpscore;
output out=bbindex.b1wqvar_scores_mean_yr_seg
        mean(tempscore doscore tnscore tpscore)= avgTEMP_score avgDO_score avgTN_score
avgTP_score
        median(tempscore doscore tnscore tpscore) = medTEMP_score medDO_score
medTN_score medTP_score
        stddev(tempscore doscore tnscore tpscore)= sdTEMP_score sdDO_score sdTN_score
sdTP_score
        min(tempscore doscore tnscore tpscore)= minTEMP_score minDO_score minTN_score
minTP_score
        max(tempscore doscore tnscore tpscore)= maxTEMP_score maxDO_score maxTN_score
maxTP_score
;
run;
proc print data=bbindex.b1wqvar_scores_mean_yr_seg;run;

*---------------------;


title1 Light Availability;
proc contents data=bbdb.pllestimate2;run;
proc sort data=bbdb.pllestimate2; by Year Segment; run;
data BBindex.B2lightvar_scores_raw;
set BBdb.PLLestimate2;
```

if PLLest ne . and PLLest > 32 then pllscore = 50;else
if PLLest ne . and PLLest < 7.818 then pllscore = 0;else
if PLLest ne . then pllscore = 50.084*log(PLLest)-122.18;

if Secchi_cm ne . and Secchi_cm > 500 then secchiscore = 50;else
if Secchi_cm ne . and Secchi_cm < 100 then secchiscore = 0;else
if Secchi_cm ne . then secchiscore = 0.1250*Secchi_cm - 12.5;

if CHLA ne . and CHLA < 2.5 then chlascore = 50; else
if CHLA ne . and CHLA > 7.75 then chlascore = 0; else
if CHLA ne . then chlascore = -41.67*log(CHLA)+85.351;

if TSS ne . and TSS le 10 then tssscore = 50; else
if TSS ne . and TSS ge 20 then tssscore= 0; else
if TSS ne . then tssscore = -5*TSS+100;

if Macroalgae_percentcover ne . and Macroalgae_percentcover le 3 then macroscore = 50; else
if Macroalgae_percentcover ne . and Macroalgae_percentcover ge 20 then macroscore = 0; else
if Macroalgae_percentcover ne . then macroscore = -24.52*log(Macroalgae_percentcover)+
76.782;

if Bde ne . and Bde le 0.25 then epiphytescore = 50; else
if Bde ne . and Bde ge 2.0 then epiphytescore = 0; else
if Bde ne . then epiphytescore = -20.32*log(Bde)+22.744;

keep
Year Segment
CHLA TSS Secchi Macroalgae_percentcover Bde PLLest
chlascore tssscore secchiscore macroscore epiphytescore pllscore;
run;
proc sort data=BBindex.B2lightvar_scores_raw; by Year Segment; run;
proc means data=BBindex.B2lightvar_scores_raw n mean stddev min max;
by Year Segment;
var chlascore tssscore secchiscore macroscore epiphytescore pllscore;
output out=bbindex.b2lightvar_scores_mean_yr_seg
        mean(chlascore tssscore secchiscore macroscore epiphytescore pllscore)=
avgCHLA_score avgTSS_score avgSECCHI_score avgMACRO_score avgEPI_score
avgPLL_score
        stddev(chlascore tssscore secchiscore macroscore epiphytescore pllscore)=
sdCHLA_score sdTSS_score sdSECCHI_score sdMACRO_score sdEPI_score sdPLL_score
        median(chlascore tssscore secchiscore macroscore epiphytescore pllscore)=

medCHLA_score medTSS_score medSECCHI_score medMACRO_score medEPI_score medPLL_score
       min(chlascore tssscore secchiscore macroscore epiphytescore pllscore)=
minCHLA_score minTSS_score minSECCHI_score minMACRO_score minEPI_score minPLL_score
       max(chlascore tssscore secchiscore macroscore epiphytescore pllscore)=
maxCHLA_score maxTSS_score maxSECCHI_score maxMACRO_score maxEPI_score maxPLL_score
;
run;
proc print data=bbindex.b2lightvar_scores_mean_yr_seg;run;

proc print data=bbindex.b2lightvar_scores_mean_yr_seg;
var Year Segment avgSECCHI_score avgMACRO_score;
run;

*--------------------;

title1 Seagrass (Biotic Response);
proc contents data=bbdb.savall;run;
proc sort data=bbdb.savall; by Year Segment; run;
data BBindex.C1savvar_scores_raw;
set bbdb.savall;

if Zostera_aboveground_biomass_gm2 le 0 then abovescore = 0; else
if Zostera_aboveground_biomass_gm2 ge 400 then abovescore = 50; else
abovescore = 0.125 * Zostera_aboveground_biomass_gm2;

if Zostera_belowground_biomass_gm2 le 0 then belowscore = 0; else
if Zostera_belowground_biomass_gm2 ge 800 then belowscore = 50; else
belowscore = 0.0625 * Zostera_belowground_biomass_gm2;

if Zostera_shootdensity le 0   then densityscore = 0; else
if Zostera_shootdensity ge 1910 then densityscore = 50; else
densityscore = 0.0243 * Zostera_shootdensity + 5.7143;

if Zostera_percentcover le 0 then percentscore = 0; else
if Zostera_percentcover ge 50 then percentscore = 50; else
percentscore = 15.925 * log(Zostera_percentcover) - 12.713;

if Zostera_bladelength le 0 then lengthscore = 0; else

```
if Zostera_bladelength ge 80 then lengthscore = 50; else
lengthscore = 0.625 * Zostera_bladelength;

keep
Year
Segment
Zostera_aboveground_biomass_gm2
Zostera_belowground_biomass_gm2
Zostera_shootdensity
Zostera_percentcover
Zostera_bladelength
abovescore
belowscore
densityscore
percentscore
lengthscore;
run;
proc sort data=BBindex.C1savvar_scores_raw; by Year Segment; run;
proc means data=bbindex.C1savvar_scores_raw;
by Year Segment;
output out=bbindex.C1savvar_scores_mean_yr_seg
        mean(abovescore belowscore densityscore percentscore lengthscore)= avgABOVE_score
avgBELOW_score avgDENSITY_score avgPCENT_score avgLENGTH_score
        median(abovescore belowscore densityscore percentscore lengthscore)=
medABOVE_score medBELOW_score medDENSITY_score medPCENT_score
medLENGTH_score
        stddev(abovescore belowscore densityscore percentscore lengthscore)= sdABOVE_score
sdBELOW_score sdDENSITY_score sdPCENT_score sdLENGTH_score
        min(abovescore belowscore densityscore percentscore lengthscore)= minABOVE_score
minBELOW_score minDENSITY_score minPCENT_score minLENGTH_score
        max(abovescore belowscore densityscore percentscore lengthscore)= maxABOVE_score
maxBELOW_score maxDENSITY_score maxPCENT_score maxLENGTH_score
;
run;
proc print data=bbindex.c1savvar_scores_mean_yr_seg;run;

*--------------------;


title1 HABS (Biotic Response);
data bbindex.C2HABvar_scores_raw;
set bbdb.browntide;
```

```
if cells_per_ml ne . and cells_per_ml ge 260000 then habscore = 0;else
if cells_per_ml ne . and cells_per_ml lt 30000  then habscore = 100;else
habscore = -0.0004*cells_per_ml + 113.98;
run;
proc sort data=bbindex.C2HABvar_scores_raw; by Year; run;
proc print data=bbindex.C2HABvar_scores_raw; by Year; run;
proc means data=bbindex.C2HABvar_scores_raw; by Year;
var habscore;
output out=bbindex.C2HABvar_scores_mean_yr
                mean=avgHAB_score
                median=medHAB_score
                stddev=sdHAB_score
                min=minHAB_score
                max=maxHAB_score
;
        run;
proc print data=bbindex.c2habvar_scores_mean_yr;run;

*------------;


title1 BENTHIC (Biotic Response);

data C3REMAPvar_scores_raw;
set bbdb.remap;
run;




**************************************************************************
*;

*STEP 2: CONDUCT PCA ON VARIABLE SCORES AND USE EIGENVECTORS
TO CACLULATE WEIGHTING OF EACH VARIABLE WITHIN EACH COMPONENT;

        *PRESSURES - NO VARIABILITY, SO NO WEIGHTING.

        *WATER QUALITY;
```

```
proc sort data=bbindex.b1wqvar_scores_mean_yr_seg; by Year;run;
proc print data=bbindex.b1wqvar_scores_mean_yr_seg;
where Year ge 1989 and Year le 1999;
var  Year Segment medtemp_score meddo_score medtn_score;
run;

*1989-1999;
proc princomp data=bbindex.b1wqvar_scores_mean_yr_seg
covariance
out=bbpca.B1WQscores89_99
outstat=bbpca.B1WQscores89_99stat;
where Year ge 1989 and Year le 1999;
var medtemp_score meddo_score medtn_score;
run;
*1999-2010;
proc princomp data=bbindex.b1wqvar_scores_mean_yr_seg
covariance
out=bbpca.B1WQscores00_10
outstat=bbpca.B1WQscores00_10stat;
where Year ge 2000 and Year le 2010;
var medtemp_score meddo_score medtn_score medtp_score ;
run;
proc sort data=bbpca.B1WQscores89_99stat; by _TYPE_ ;run;
proc sort data=bbpca.B1WQscores00_10stat; by _TYPE_ ;run;
*2nd decade with TP;
data bbindex.B1WQweight;
set bbpca.B1WQscores89_99stat bbpca.B1WQscores00_10stat;
by _TYPE_ ;
where _NAME_ = 'Prin1';
if _TYPE_ = 'SCORE' then WeightTemp = medtemp_score*medtemp_score;
if _TYPE_ = 'SCORE' then WeightDO = meddo_score*meddo_score;
if _TYPE_ = 'SCORE' then WeightTN = medtn_score*medtn_score;
if _TYPE_ = 'SCORE' then WeightTP = medtp_score*medtp_score;
run;
proc print data=bbindex.B1WQweight;run;



        *LIGHT AVAILABILITY;
proc sort data=bbindex.b2lightvar_scores_mean_yr_seg; by Year;run;
*1998-2010;
proc princomp data=bbindex.b2lightvar_scores_mean_yr_seg
```

```
covariance
out=bbpca.B2LAscores98_10
outstat=bbpca.B2LAscores98_10stat;
var  medCHLA_score medTSS_score avgSECCHI_score avgEPI_score avgPLL_score;
run;
proc sort data=bbpca.B2LAscores98_10stat; by _TYPE_ ;run;
data bbindex.B2LAweight;
set bbpca.B2LAscores98_10stat;
by _TYPE_ ;
where _NAME_ = 'Prin1';
if _TYPE_ = 'SCORE' then WeightChla = medCHLA_score*medCHLA_score;
if _TYPE_ = 'SCORE' then WeightTSS = medTSS_score*medTSS_score;
if _TYPE_ = 'SCORE' then WeightSecchi = avgSECCHI_score*avgSECCHI_score;
if _TYPE_ = 'SCORE' then WeightEpiphyte = avgEPI_score*avgEPI_score;
if _TYPE_ = 'SCORE' then WeightPLL = avgPLL_score*avgPLL_score;
run;
proc print data=bbindex.B2LAweight;run;


        *SEAGRASS;
proc sort data=bbindex.C1savvar_scores_mean_yr_seg; by Year;run;
*2004-2006, 2008-2010;
proc princomp data=bbindex.C1savvar_scores_mean_yr_seg
covariance
out=bbpca.C1SAVscores04_06_08_10
outstat=bbpca.C1SAVscores04_06_08_10stat;
var avgABOVE_score avgBELOW_score medDENSITY_score avgPCENT_score
avgLENGTH_score;
run;
proc sort data=bbpca.C1SAVscores04_06_08_10stat; by _TYPE_ ; run;
data bbindex.C1SAVweight;
set bbpca.C1SAVscores04_06_08_10stat;
by _TYPE_;
where _NAME_ = 'Prin1';
if _TYPE_ = 'SCORE' then WeightAbove = avgABOVE_score*avgABOVE_score;
if _TYPE_ = 'SCORE' then WeightBelow = avgBELOW_score*avgBELOW_score;
if _TYPE_ = 'SCORE' then WeightDensity = medDENSITY_score*medDENSITY_score;
if _TYPE_ = 'SCORE' then WeightPercent = avgPCENT_score*avgPCENT_score;
if _TYPE_ = 'SCORE' then WeightLength = avgLENGTH_score*avgLENGTH_score;
run;
proc print data=bbindex.C1SAVweight;run;
```

*HARMFUL ALGAE - NO VARIABILITY, SO NO WEIGHTING;

*BENTHIC INVERTEBRATES - NO VARIABILITY, SO NO WEIGHTING;


```
**************************************************************************
*********;

*STEP 3: CALCULATE UNWEIGHTED, WEIGHTED, and INDEX SCORES FOR EACH
COMPONENT;

        *PRESSURES - NO WEIGHTING;
                *Total Loading for TN and Total Loading for TP are evenly weighted.;
                *Weighted scores are not calculated for Pressures since there is no variability
within Year-Segment.
                Therefore Unweighted scores equal the PRESSURE INDEX;
title1 Pressure Index;
proc sort data=bbindex.A1pressurescore;
by Year Segment;
data bbindex.A1pressureindex;
set bbindex.A1pressurescore;
where Season='All';
by Year Segment;
PressureIndex = mean(tnloadscore, tploadscore);
run;
proc print data=bbindex.A1pressureindex;
var Year Segment tnloadscore tploadscore PressureIndex;
run;
proc sort data=bbindex.A1pressureindex;by  Segment;run;
proc sgplot data=bbindex.A1pressureindex;
title 'TN LOADING SCORE';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
YAXIS LABEL = 'TN Loading Score' VALUES = (0 TO 100 BY 10);
series x = Year y = tnloadscore / name="TN Loading Score" markers MARKERATTRS=(size=
15) LINEATTRS=(pattern=solid thickness=2)        group=Segment;
```

```
run;
proc sgplot data=bbindex.A1pressureindex;
title 'TP LOADING SCORE';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
YAXIS LABEL = 'TP Loading Score' VALUES = (0 TO 100 BY 10);
series x = Year y = PressureIndex / name="TP Loading Score" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)        group=Segment;
run;
proc sgplot data=bbindex.A1pressureindex;
title 'WATERSHED PRESSURES INDEX';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
YAXIS LABEL = 'Index Value' VALUES = (0 TO 100 BY 10);
series x = Year y = PressureIndex / name="Pressures Index" markers MARKERATTRS=(size=
15) LINEATTRS=(pattern=solid thickness=2)        group=Segment;
run;


        *WATER QUALITY;
title1 WATER QUALITY INDEX;
proc print data=bbindex.b1wqvar_scores_mean_yr_seg;run;
proc print data=bbindex.B1WQweight;run;
proc sort data=bbindex.b1wqvar_scores_mean_yr_seg; by Year;run;
data bbindex.B1WQ_Index;
set bbindex.b1wqvar_scores_mean_yr_seg;

UnweightedWQ_Index = mean(avgTEMP_score, avgDO_score, avgTN_score, avgTP_score);

if Year ge 1989 and Year le 1999 then WeightTemp = 0.6571;
if Year ge 2000 and Year le 2010 then WeightTemp = 0.1502;

if Year ge 1989 and Year le 1999 then WeightDO = 0.3275;
if Year ge 2000 and Year le 2010 then WeightDO = 0.0760;

if Year ge 1989 and Year le 1999 then WeightTN = 0.0154;
if Year ge 2000 and Year le 2010 then WeightTN = 0.1285;

if Year ge 1989 and Year le 1999 then WeightTP = 0.0000;
if Year ge 2000 and Year le 2010 then WeightTP = 0.6454;

WtdTEMP_score = WeightTemp*avgTEMP_score;
```

```
WtdDO_score = WeightDO*avgDO_score;
WtdTN_score = WeightTN*avgTN_score;
WtdTP_score = WeightTP*avgTP_score;

WeightedWQ_Index = sum(WtdTEMP_score, WtdDO_score, WtdTN_score, WtdTP_score);

WQ_Index = UnweightedWQ_Index + WeightedWQ_Index;

run;
proc print data=bbindex.B1WQ_Index;run;
proc sort data=bbindex.B1WQ_Index;by Segment;run;
proc sgplot data=bbindex.B1WQ_Index;
title 'WATER QUALITY INDICATOR SCORES';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
*YAXIS LABEL = 'Temperature Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Dissolved Oxygen Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Total Nitrogen Score' VALUES = (0 TO 50 BY 10);
YAXIS LABEL = 'Total Phosphorus Score' VALUES = (0 TO 50 BY 10);
*series x = Year y= avgTEMP_score / name="Temperature Score" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)       group=Segment;
*series x = Year y= avgDO_score / name="Dissolved Oxygen Score" markers
MARKERATTRS=(size=15) LINEATTRS=(pattern=solid thickness=2)   group=Segment;
*series x = Year y= avgTN_score / name="Total Nitrogen Score" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)       group=Segment;
series x = Year y= avgTP_score / name="Total Phosphorus Score" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)       group=Segment;
legend;
run;
proc sgplot data=bbindex.B1WQ_Index;
title 'WATER QUALITY INDEX';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
*YAXIS LABEL = 'Water Quality Index (FINAL)' VALUES = (0 TO 100 BY 10);
*YAXIS LABEL = 'Raw Value for Water Quality Index' VALUES = (0 TO 50 BY 10);
YAXIS LABEL = 'Weighted Value for Water Quality Index' VALUES = (0 TO 50 BY 10);
*series x = Year y = WQ_Index / name="WQ Index" markers MARKERATTRS=(size=15)
LINEATTRS=(pattern=solid thickness=2)   group=Segment;
*series x = Year y = UnweightedWQ_Index / name="Unweighted" markers LINEATTRS=
(pattern=dashdashdot) group=Segment;
series x = Year y = WeightedWQ_Index / name="Weighted" markers LINEATTRS=
(pattern=longdash)     group=Segment;
```

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```
legend  ;
run;


        *LIGHT;
title1 LIGHT AVAILABILITY INDEX;
proc print data=bbindex.b2lightvar_scores_mean_yr_seg;run;
proc print data=bbindex.B2LAweight;run;
proc sort data=bbindex.b2lightvar_scores_mean_yr_seg; by Year;run;
data bbindex.B2LA_Index;
set bbindex.b2lightvar_scores_mean_yr_seg;

UnweightedLA_Index = mean(avgCHLA_score, avgTSS_score, avgSECCHI_score,
avgMACRO_score, avgEPI_score, avgPLL_score);

WeightCHL = 0.0244;
WeightTSS = 0.3209;
WeightSECCHI = 0.0413;
WeightMACRO = 0.0000;
WeightEPI = 0.3004;
WeightPLL = 0.3130;

WtdCHL_score = WeightCHL*avgCHLA_score;
WtdTSS_score = WeightTSS*avgTSS_score;
WtdSECCHI_score = WeightSECCHI*avgSECCHI_score;
WtdMACRO_score = WeightMACRO*avgMACRO_score;
WtdEPI_score = WeightEPI*avgEPI_score;
WtdPLL_score = WeightPLL*avgPLL_score;

WeightedLA_Index = sum(WtdCHL_score, WtdTSS_score, WtdSECCHI_score,
WtdMACRO_score, WtdEPI_score, WtdPLL_score);

LA_Index = UnweightedLA_Index + WeightedLA_Index;

run;
proc print data=bbindex.B2LA_Index;run;
proc sort data=bbindex.B2LA_Index;by Segment;run;
proc sgplot data=bbindex.B2LA_Index;
title 'LIGHT AVAILABILITY INDICATOR SCORES';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
*YAXIS LABEL = 'Chlorophyll a Score' VALUES = (0 TO 50 BY 10);
```

```
*YAXIS LABEL = 'Total suspended solids Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Secchi depth Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Macroalgae cover Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Epiphyte:Seagrass Ratio Score' VALUES = (0 TO 50 BY 10);
YAXIS LABEL = 'Percent surface light Score' VALUES = (0 TO 50 BY 10);
*series x = Year y = avgCHLA_score / name="Chlorophyll a" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)        group=Segment;
*series x = Year y = avgTSS_score / name="Total Suspended Solids" markers
MARKERATTRS=(size=15) LINEATTRS=(pattern=solid thickness=2)    group=Segment;
*series x = Year y = avgSECCHI_score / name="Secchi depth" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)        group=Segment;
*series x = Year y = avgMACRO_score / name="Macroalgae cover" markers
MARKERATTRS=(size=15) LINEATTRS=(pattern=solid thickness=2)    group=Segment;
*series x = Year y = avgEPI_score / name="Epiphyte:Seagrass Ratio " markers
MARKERATTRS=(size=15) LINEATTRS=(pattern=solid thickness=2)    group=Segment;
series x = Year y = avgPLL_score / name="Percent surface light" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)        group=Segment;
legend;
run;
proc sgplot data=bbindex.B2LA_Index;
title 'LIGHT AVAILABLITY INDEX';
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
*YAXIS LABEL = 'Light Availability Index Values (Final)' VALUES = (0 TO 100 BY 10);
*YAXIS LABEL = 'Raw Value for Light Availability Index' VALUES = (0 TO 50 BY 10);
YAXIS LABEL = 'Weighted Value for Light Availability Index' VALUES = (0 TO 50 BY 10);
*series x = Year y = LA_Index / name="LA Index" markers MARKERATTRS=(size=15)
LINEATTRS=(pattern=solid thickness=2)    group=Segment;
*series x = Year y = UnweightedLA_Index / name="Unweighted" markers LINEATTRS=
(pattern=dashdashdot) group=Segment;
series x = Year y = WeightedLA_Index / name="Weighted" markers LINEATTRS=
(pattern=longdash)     group=Segment;
legend  ;
run;




        *SEAGRASS;
proc print data=bbindex.C1savvar_scores_mean_yr_seg;run;
proc print data=bbindex.C1SAVweight;run;
proc sort data=bbindex.C1savvar_scores_mean_yr_seg; by Year;run;

data bbindex.C1SAV_Index;
```

```sas
set bbindex.C1savvar_scores_mean_yr_seg;

UnweightedSAV_Index = mean(avgABOVE_score, avgBELOW_score, avgDENSITY_score,
avgPCENT_score, avgLENGTH_score);

WeightABOVE = 0.0841;
WeightBELOW = 0.0244;
WeightDENSITY = 0.0111;
WeightPCENT = 0.5336;
WeightLENGTH = 0.3458;


WtdABOVE_score = WeightABOVE*avgABOVE_score;
WtdBELOW_score = WeightBELOW*avgBELOW_score;
WtdDENSITY_score = WeightDENSITY*avgDENSITY_score;
WtdPCENT_score = WeightPCENT*avgPCENT_score;
WtdLENGTH_score = WeightLENGTH*avgLENGTH_score;

WeightedSAV_Index = sum(WtdABOVE_score, WtdBELOW_score, WtdDENSITY_score,
WtdPCENT_score, WtdLENGTH_score);

SAV_Index = UnweightedSAV_Index + WeightedSAV_Index;

run;
proc print data=bbindex.C1SAV_Index;run;
proc sort data=bbindex.C1SAV_Index;by Segment;run;
proc sgplot data=bbindex.C1SAV_Index;
title 'SEAGRASS RESPONSE SCORES';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
*YAXIS LABEL = 'Aboveground biomass Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Belowground biomass Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Shoot density Score' VALUES = (0 TO 50 BY 10);
*YAXIS LABEL = 'Percent cover Score' VALUES = (0 TO 50 BY 10);
YAXIS LABEL = 'Blade length Score' VALUES = (0 TO 50 BY 10);
*series x = Year y = avgABOVE_score / name="Aboveground biomass" markers
MARKERATTRS=(size=15) LINEATTRS=(pattern=solid thickness=2)    group=Segment;
*series x = Year y = avgBELOW_score / name="Belowground biomass" markers
MARKERATTRS=(size=15) LINEATTRS=(pattern=solid thickness=2)    group=Segment;
*series x = Year y = avgDENSITY_score / name="Shoot density" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)           group=Segment;
*series x = Year y = avgPCENT_score / name="Percent cover" markers MARKERATTRS=
```

```
(size=15) LINEATTRS=(pattern=solid thickness=2)          group=Segment;
series x = Year y = avgLENGTH_score / name="Blade length" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)          group=Segment;
legend;
run;
proc sgplot data=bbindex.C1SAV_Index;
title 'SEAGRASS INDEX';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
*YAXIS LABEL = 'Seagrass Response Index Values (Final)' VALUES = (0 TO 100 BY 10);
*YAXIS LABEL = 'Raw Value for Seagrass Response Index' VALUES = (0 TO 50 BY 10);
YAXIS LABEL = 'Weighted Value for Seagrass Response Index' VALUES = (0 TO 50 BY 10);
*series x = Year y = SAV_Index / name="SAV Index" markers MARKERATTRS=(size=15)
LINEATTRS=(pattern=solid thickness=2 )   group=Segment;
*series x = Year y = UnweightedSAV_Index / name="Unweighted" markers LINEATTRS=
(pattern=dashdashdot) group=Segment;
series x = Year y = WeightedSAV_Index / name="Weighted" markers LINEATTRS=
(pattern=longdash)     group=Segment;
legend  ;
run;




        *HAB ;
                *Only one variable is used to determine the HAB index, and there is no
variability
                so there is no weighting, so the concentrations are directly rescaled into the
                HAB index.;
proc print data=bbindex.c2habvar_scores_mean_yr;run;
proc sort data=bbindex.c2habvar_scores_mean_yr; by Year;run;
data bbindex.C2HAB_Index;
set bbindex.C2habvar_scores_mean_yr;
HAB_Index = avgHAB_score;
run;
proc print data=bbindex.C2HAB_Index;run;
proc sgplot data=bbindex.C2HAB_Index;
title 'HARMFUL ALGAE INDEX';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
YAXIS LABEL = 'Index Value' VALUES = (0 TO 100 BY 10);
scatter x = Year y = HAB_Index / name="HAB Index"  MARKERATTRS= (symbol=circlefilled
color=black size=15) ;
```

legend ;
run;

;

      *BENTHIC;

****
*********;

*STEP 4: CONDUCT PCA ON COMPONENT SCORES AND USE EIGENVECTORS
TO CALCULATE WEIGHTING OF EACH COMPONENT INDEX WITHIN OVERALL
INDEX;

proc sort data= bbindex.A1pressureindex; by Year Segment; run;
proc sort data= bbindex.B1WQ_Index; by Year Segment; run;
proc sort data= bbindex.B2LA_Index; by Year Segment; run;
proc sort data= bbindex.C1SAV_Index; by Year Segment; run;
data bbindex.EUTRO;
merge
bbindex.A1pressureindex
bbindex.B1WQ_Index
bbindex.B2LA_Index
bbindex.C1SAV_Index;
by Year Segment;
run;
proc print data=bbindex.EUTRO;
var Year SEgment PressureIndex WQ_Index LA_Index SAV_Index;run;
*1989-1997;
proc princomp data=bbindex.EUTRO
covariance
out=bbpca.EUTROscores89_97
outstat=bbpca.EUTROscores89_97stat;
where Year ge 1989 and Year le 1997;

```
var WQ_Index;
run;
*1998-2003;
proc princomp data=bbindex.EUTRO
covariance
out=bbpca.EUTROscores98_03
outstat=bbpca.EUTROscores98_03stat;
where Year ge 1998 and Year le 2003;
var WQ_Index LA_Index;
run;
*2004-2010;
proc princomp data=bbindex.EUTRO
covariance
out=bbpca.EUTROscores04_10
outstat=bbpca.EUTROscores04_10stat;
where Year ge 2004 and Year le 2010;
var WQ_Index LA_Index SAV_Index;
run;
proc sort data=bbpca.EUTROscores89_97stat; by _TYPE_ ;run;
proc sort data=bbpca.EUTROscores98_03stat; by _TYPE_ ;run;
proc sort data=bbpca.EUTROscores04_10stat; by _TYPE_ ;run;
data bbindex.EUTROweight89_97;
set bbpca.EUTROscores89_97stat;
by _TYPE_;
where _NAME_ = 'Prin1';
if _TYPE_ = 'SCORE' then WeightWQ = WQ_Index*WQ_Index;
run;
data bbindex.EUTROweight98_03;
set bbpca.EUTROscores98_03stat;
by _TYPE_;
where _NAME_ = 'Prin1';
if _TYPE_ = 'SCORE' then WeightWQ = WQ_Index*WQ_Index;
if _TYPE_ = 'SCORE' then WeightLA = LA_Index*LA_Index;
run;
data bbindex.EUTROweight04_10;
set bbpca.EUTROscores04_10stat;
by _TYPE_;
where _NAME_ = 'Prin1';
if _TYPE_ = 'SCORE' then WeightWQ = WQ_Index*WQ_Index;
if _TYPE_ = 'SCORE' then WeightLA = LA_Index*LA_Index;
if _TYPE_ = 'SCORE' then WeightSAV = SAV_Index*SAV_Index;
run;
```

```
data bbindex.EUTROweight;
set bbindex.EUTROweight89_97  bbindex.EUTROweight98_03  bbindex.EUTROweight04_10;
run;
proc print data=bbindex.EUTROweight;
var WeightWQ WeightLA WeightSAV;run;
```

```
**************************************************************************
*********;

**STEP 5: CALCULATE OVERALL UNWEIGHTED, WEIGHTED, and FINAL
EUTROPHICATION INDEXC SCORES;

data bbindex.EUTRO_Index;
set bbindex.EUTRO;
EUTRO_Index = mean(PressureIndex, WQ_Index, LA_Index, SAV_INDEX);
run;
proc print data=bbindex.EUTRO_Index;
*var Year Segment PressureIndex WQ_Index LA_Index SAV_Index EUTRO_Index;
run;
proc sgplot data=bbindex.EUTRO_Index;
title 'OVERALL EUTROPHICATION INDEX';
where Year ge 1989 and Year le 2010;
XAXIS LABEL = 'Year'  values = (1989 to 2010 by 1) ;
YAXIS LABEL = 'Index Value' grid VALUES = (0 TO 100 BY 10);
series x = Year y = EUTRO_Index / name="Eutrophication Index" markers MARKERATTRS=
(size=15) LINEATTRS=(pattern=solid thickness=2)          group=Segment;
legend;
run;

**********
COMPARING INDEXES;
**********
;

proc contents data=bbindex.EUTRO_Index;run;
proc sgplot data=bbindex.EUTRO_Index;
scatter x = PressureIndex y = SAV_Index / group=Segment;
run;
```

```
proc sort data=bbindex.EUTRO_Index; by Segment;run;
proc sgscatter data=bbindex.EUTRO_Index;
title 'LOADING vs. EUTROPHICATION INDEX';
where Year ge 1989 and Year le 2010;
compare y= EUTRO_Index x =(TN_TotLoadKgKm2 TP_TotLoadKgKm2) /
MARKERATTRS=(size=15) group = Segment ;

label EUTRO_Index = 'Eutrophication Index Value'
            TN_TotLoadKgKm2='Total Nitrogen Loading (kg TN km-2 y-1)'
            TP_TotLoadKgKm2='Total Phosphorus Loading (kg TP km-2 y-1)';
run;
```