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

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New England Interstate Water Pollution Control Commission

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KEY FINDINGS

- Barnegat Bay-Little Egg Harbor (BB-LEH) is highly eutrophic and is susceptible to nutrient loading. It is shallow, poorly flushed, and affected by a 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}.

- Concentrations, loads and yields of total nitrogen and total phosphorus were quantified on annual and seasonal timescales and on 3 spatial scales: whole watershed, watershed segments corresponding to estuary segmentation, and 14-digit hydrologic unit code.

- This study confirmed that surface-water concentrations of nutrients (nitrogen and phosphorus) in the BB-LEH estuary are strongly related to land use. Total nitrogen and phosphorus are highest in areas with the highest percentages of urban and agricultural land, and with the lowest percentages of forested and undeveloped land.

- Urban development has steadily increased in the watershed since the 1970s, and this is strongly correlated with the observed increase in total nitrogen concentration in BB-LEH watershed streams. Development (and corresponding increases in total nitrogen concentrations and loads) is more intense in the north segment than elsewhere.

- Concentrations, loads and yields of phosphorus and nitrogen are generally higher during the growing season than during the nongrowing season.

- Nitrogen loads from areas covered with turf are about twice those of non-turf urban areas. Phosphorus loads from turf areas are more than eight times those from non-turf areas. Phosphorus concentrations, loads and yields are generally higher in areas with more development, and higher during runoff than in baseflow.

- Baseflow contributes more than 80% of the total nitrogen loading from streams, however, runoff contributes a higher percentage of nitrogen loading in developed areas than in undeveloped areas owing to the greater percentage of streamflow from runoff for streams in developed areas.

- From 1989 to 2010, BB-LEH experienced low dissolved oxygen (82 times $\leq$4 mg L^{-1}), high total suspended solids (max >200 mg L^{-1}) and chlorophyll *a* (max >40 µg L^{-1}), harmful algal blooms ($\geq$200,000 cells mL^{-1}), epiphytic loading (mean values up to 38.3% cover of seagrass), macroalgae blooms (80-100% cover 36 times, 70-80% cover 19 times, 60-70% cover 10 times), habitat loss, >67% fewer clams, and degraded seagrass biomass (to 2.7±8.0 g m^{-2} aboveground; 17.9±37.5 g m^{-2} belowground).

- 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 is Highly Degraded, 100 is Excellent). Index scores assess condition and its consistency. Increased availability of data would improve its resolution, though would not likely significantly change the conclusions of this report. Though monitoring intensified over time and the number of indicators monitoring are increasing, spatio-temporal alignment of data collection and increased sampling frequency will improve future assessments.

- Index of Eutrophication 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 currently undergoing eutrophication. The north segment has already undergone eutrophication. Eutrophication condition was worst in the north segment despite modest improvements, in contrast to stages and trends in the south and central segments. Scores in the north segment declined sharply in 2010 (to 37), but the highest score there (50) was in 2009, 3.5 times its low score (14, in 1991).

- Nutrient loading severely impacted Index of Eutrophication values in BB-LEH, particularly in 2003-2010, degrading condition from 73 to 45 and 37. Initial rapid declines highlight sensitivity to loading. Beyond ~2,000 kg total nitrogen km^{-2} yr^{-1} or ~100 kg total phosphorus km^{-2} yr^{-1}, condition plateaued yet variability increased (ranging 2 to 50), suggesting a switch in dominant factors.

- Total nutrient loadings in the north were very low (7), but were 60 and 55 in the central and south segments respectively. During 1989-1997, low dissolved oxygen countered favorable temperatures leading to a Water Quality Index score of 57. Favorable temperatures continued in 1998-1999, but total phosphorus increased in 2000-2003. In 1998-2003, total suspended solids scores ranged 21 to 45, epiphytic loading scores were 16 to 40, available surface light scores were 7 to 32 declining in 1998-2002 in the north and south segments. In 2004-2010, total phosphorus condition in BB-LEH fell from 32 to 7. Total suspended solids improved steadily in the north segment, variably in the south segment, and temporarily declined in 2004-2007 in the central segment. Similar temporary declines in condition during 2004-2009 in the central segment was seen in epiphytic load scores (44 to 1) and available surface light scores (41 to 0). Seagrass cover and length scores decreased over 2004-2010 from 34 to 14 and from 30 to 18, respectively.

- Increasing eutrophication of the central and south segments since the 1990s and even worse condition in the north segment was observed throughout the study period. The condition of BB-LEH progressively worsened over time for both nitrogen and phosphorus. Periods of improvement (1989-1992, 1996-2002, and 2006-2008) did not outpace shorter but detrimental periods, thus leading to overall poorer condition.

- Collectively, the direct relationship between nutrient loading from the watershed and estuarine nutrient concentrations, the degradation of an array of biotic indicators, and the relationship between nutrient loading and the Index of Eutrophication supports the conclusion that BB-LEH is an estuary that has undergone significant ecological decline.

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 30% 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.

BB-LEH is a highly eutrophic estuary. Eutrophication is defined as the process of nutrient enrichment and increase in the rate of organic matter input in a waterbody leading to an array of cascading changes in ecosystem structure and function such as decreased dissolved oxygen levels, increased microalgal and macroalgal abundance, occurrence of harmful algal blooms (HABs), loss of seagrass habitat, reduced biodiversity, declining fisheries, imbalanced food webs, altered biogeochemical cycling, and diminished ecosystem services.

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 rooted macrophyte assemblages that 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 eutrophic impacts pose serious threats to the estuary because they are leading to significant ecological decline of the estuary and affecting biotic resources, essential habitats (e.g., seagrass 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 aggressive management actions and effective planning are implemented.

Regulatory protection and conservation of New Jersey's estuarine waters are based on dissolved oxygen (DO) measurements. Yet DO is only one indicator of ecological health, and must be monitored continuously in multiple locations for accurate assessments due to natural variations over the course of a day driven by natural processes such as changes in temperature or light, as well as community photosynthesis and respiration. Routine monitoring of DO over the years in BB-LEH, a coastal lagoon, has not been conducted frequently enough or at all necessary times and sampling stations over a 24-hour period, thereby biasing sampling results. For example, DO measurements must also be made between midnight and 6 a.m. Therefore, it is important to assess the ecological health of the estuary by examining a broader range of physicochemical and

biotic indicators 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 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. The current assessment of system eutrophication is based on degradation of eelgrass condition and other declining ecosystem measures that have continued in concert with nitrogen loading from the BB-LEH Watershed, as documented by the Index of Eutrophication developed for the estuary.

Nutrient loading has been repeatedly cited as a primary cause of ecosystem eutrophication of 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; Guo et al., 1997, 2004) facilitates nitrogen uptake by plants and nitrogen accumulation in estuarine bottom sediments which can be an important secondary nitrogen source for internal cycling. 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).

The assessment reported here documents multiple 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 that has undergone significant ecological decline, as evidenced by the increasing eutrophication of the central and south segments since the 1990s ($P < 0.05$) and an even worse eutrophication condition documented for the north segment. An array of biotic indicator data collected over the past two decades 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 timeframe 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.

The term 'eutrophic condition' refers to eutrophication condition of the waterbody. The eutrophic condition of the estuary has been well documented (Seitzinger et al., 1992, 1993, 2001; Bricker et al., 1999, 2007; and Kennish et al., 2007a, 2010). 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 seagrass play major roles in primary production of 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 eutrophic condition 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 Index of Eutrophication 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 Index of Eutrophication 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 Index of Eutrophication developed for this investigation for the BB-LEH Estuary builds on previous assessments, especially the National Estuarine Eutrophication Assessment (NEEA), which the Assessment of Estuarine Trophic Status (ASSETS) Model. The methodology for this project employs a quantitative, numeric scoring system (rather than qualitative) from 0 (degraded condition) to 100 (excellent condition) for ~20 indicators (rather than 5).

Candidate indicators were selected at the outset and organized into: 1) Ecosystem Pressures, 2) Water Quality, 3) Light Availability, 4) Seagrass, 5) Harmful Algal Blooms, and 6) Benthic Invertebrates. The Water Quality, Light Availability, and Seagrass indicators comprise the 'Index of Eutrophication'. Each component includes several key indicators. Data collection often occurred at different times and / or locations, therefore annual means (or medians) for the north, central, and south estuarine segments are utilized for all calculations regarding the Index of Eutrophication. These summary data are included as an appendix to the report. Data are analyzed separately for each segment of the bay, because they have been determined to be heterogeneous habitats.

The Index of Eutrophication compares observations at all sites directly to a spectrum of reference conditions that are termed 'thresholds'. 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 and standardizes their ranges, thereby not making one variable more dominant than another. This practice is common in the literature. Validation of the methodology is conducted both through comparison of multiple similar methods, and through the response in 2011, as data from that year were kept separate and out of the analyses.

Thresholds are defined values. They are not a mean and have no associated error. 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. Thresholds are defined according to values of indicators and their relevance to biological, physiological, and ecological condition. Thresholds were defined based on thorough examination of: (a) the literature review, (b) analysis of the assembled database for calibration to BB-LEH, (c) Best Professional Judgment (in cases where a, and/or b are unavailable), and (d) some combination of a-c, in that order of priority. Best Professional Judgment was used sparingly. Best Professional Judgment was not used to determine thresholds for an indicator if the literature or data analysis provided sufficient information.

One challenge of identifying and defining thresholds is that indicators' responses were rarely starkly or drastically step-wise in function. That is, the values of thresholds are not obvious nor do indicators respond in discrete manners. Rather, ecosystems respond to various levels of stressors through continuous linear or non-linear manners with interactive effects since multiple stressors generally contribute simultaneously, in conjunction with natural processes and variability. Furthermore, many variables act as both a response and a stressor. Because ecosystems respond to stressors in complex and

interactive manners, it is unrealistic to expect obvious cusps or thresholds for any given individual stressors or response variable. Nevertheless, there is a high degree of confidence in the thresholds identified in this report based on general agreement of numerous literature studies and volume of data that were analyzed.

Raw Scores are calculated according to the mathematical relationship between an indicator's threshold values and the corresponding Raw Scores. The equations are used to calculate a Raw Score by inputting observations as x values, returning Raw Scores as y values. The rescaling equations for each indicator are provided. Raw scores range from 0 (bad) to 50 (excellent). Data are sorted and summarized by central tendency by Year and Segment. Descriptive, summery statistics of Raw Scores for each dataset are calculated for each segment during each year and stored as separate files. These files include means, medians, standard deviations, minimums, and maximums of each indicator's Raw Score. Where data were unavailable in a given segment during a given year, this was recorded as 'No Data' and was excluded from analysis. For Ecosystem Pressures, data were sorted by Year, Growing Season, and Segment before applying the rescaling equation to USGS modeled annual total nitrogen loading and annual total phosphorus loading. Rescaling equations were applied to each observation of Water Quality indicator (temperature, dissolved oxygen, total nitrogen concentration, total phosphorus concentration) during April to October (inclusive). These months were selected due to the importance of potential impacts on biological and human-use activities. Rescaling equations were applied to each observation of the six Light Availability indicators after excluding observations of each indicator where data was missing. Rescaling equations are applied to each observation of the five Seagrass indicators and the single HAB indicator.

Each indicator is weighted within its component according to a weighting that is calculated by principal component analysis (PCA). PCA was conducted using the covariance matrix of Raw Scores (not the correlation matrix) summarized by Year and Segment. Summarized data from all available years across the entire estuary (or as many segments as available) are used for PCA analysis to determine weightings. Up to three data points per year are thus plotted, and multiple years of data are required for this analysis to determine weightings for each indicator. A single weighting for each indicator is applied to data from each segment. Calculating unique weightings for each segment would be statistically inappropriate and would invalidate comparisons across segments. This method causes variables with large variances to be more strongly associated with components with large eigenvalues and causes variables with small variances to be more strongly associated with components with small eigenvalues and thus requires data with comparable units or standardized values (which is done by using the Raw Scores). Scree plots are examined to identify the cumulative explanatory power of each principal component. Generally, the first principal component explains ~50-75% of the variability, and the first two principal component axes explain ~80-90% of the variability. 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.

PCA on the covariance matrix was conducted on the median Raw Scores for temperature, dissolved oxygen, total nitrogen and total phosphorus, but this was done separately for 1989–1998 and 1999–2010 because total phosphorus data was unavailable during the first set of years. To test the effect of total phosphorus on the overall Water Quality, PCA on the covariance matrix was similarly conducted on the second set of years, but omitting the median Raw Scores for total phosphorus (see Validation below). Note that Raw Scores for Water Quality indicators are calculated on observations during April–October, inclusive. For the Light Availability indicators, PCA on the covariance matrix was conducted on median chlorophyll a Raw Scores, median TSS Raw Scores, average Secchi depth Raw Scores, average epiphyte to seagrass biomass ratio Raw Scores, and average percent light reaching seagrass leaves Raw Scores. PCA on the covariance matrix was conducted on median Raw Scores of Seagrass shoot density and mean Raw Scores for the other four Seagrass indicators. Note that PCA is not conducted for the Ecosystem Pressures because only a single number is provided for each segment in each year from the modeled nutrient loading provided by USGS. Therefore PCA cannot be conducted and total nitrogen loading and total phosphorus loading scores are averaged (each weighted 50%). PCA cannot be applied to the HAB component because there is only one indicator (weighted 100%). Each indicator's weighting is calculated as the square of the eigenvector of the first principal component for each variable.

Weighted scores are then calculated by multiplying the raw score by the weighting. Weighted scores also range from 0 (bad) to 50 (excellent). 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%). Note the important difference between the weighting and the Weighted Score. The weighting is the square of the eigenvector and represents the variability of the factor if data are available in a given segment in a given year. The Weighted Score is the Raw Score multiplied by the weighting and thus represents the consistency of the condition for that indicator. Weighted scores provide a measure of the consistency of the observations with respect to thresholds for the appropriate indicator.

The sum of the raw score and the weighted score equals the index score, and thus index scores range from 0 (bad) to 100 (excellent). 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. Thus, for example, each of the indicators in the Water Quality component contributes 12.5-62.5% of the Water Quality Index. The Water Quality, Light Availability, and Seagrass Indices for each of components with sufficient data are then averaged together for the sets of years when data are available to calculate the overall Index of Eutrophication. While ideally each index would be used as input for another PCA to calculate a weighting for each index, there was an insufficient quantity of data to do so, and equal weighting (i.e. averaging) was considered justified as an alternative. 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.

The purpose of adding the Raw Score and the Weighted Score to arrive at the Final Score for an indicator and each component index (e.g. Water Quality Index, Light Availability Index, Seagrass Response Index) is to assess both the condition and consistency of each indicator and each index. Consistency is important to include in an Index of Eutrophication because it highlights times and places when and where conditions of each indicator are changing (either positively or negatively) so that these indicators can be targeted for attention (e.g. for monitoring, management, or research). The implications for including both the condition and the consistency of eutrophication are that this tool can help prioritize decisions regarding limited resources available for various actions. For example, if an indicator is in flux, it may be worthy of more intense monitoring, research, or remediation action. If that same indicator consistently exhibited an extreme condition (e.g. 'Excellent' or 'Highly Degraded'), discussions regarding prioritization of resources may be efficiently directed towards another indicator.

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. However, 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. In addition, long water residence times promote the accumulation of nutrients within the estuarine system.

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 base-flow 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.

It is also stressed here that nitrogen and phosphorus occur in three principal media of the estuary: the water column, biotic tissue, and bottom sediments. Bottom sediments are typically the major repository of nutrients in coastal lagoons, exceeding the concentrations in the water column and biotic tissue. In fact, far greater concentrations of nitrogen are typically stored in bottom sediments of coastal lagoons (often 10-fold to 100-fold higher in bottom sediments than in the water column; Sand-Jensen and Borum, 1991; Burkholder et al., 2007). Internal nutrient loading via nutrient fluxes from bottom

sediments to the overlying water may be a significant driver of biotic change for this estuary.

The concentrations of nutrients in the water column are highly variable, particularly the dissolved inorganic components which are rapidly assimilated by autotrophs. Thus, low dissolved nitrogen concentrations in the water column may occur concurrently with algal blooms in the system due to rapid autotrophic uptake.

The amount of nutrients bound in plant tissue must also be considered when assessing eutrophication of estuarine systems; hence, the concern regarding nuisance and toxic algal blooms in these systems. In a separate study, we measured nitrogen concentrations in *Zostera marina* leaves along transects of 10 sampling stations in all three segments of the estuary. Mean leaf nitrogen concentrations ranged from 1.05 to 3.94%, reflecting a considerable amount of nitrogen assimilated from the water column and sediment pools and sequestered in plant tissue. This is a substantial amount of nitrogen when considering all seagrass leaves in the estuary. In addition, it does not consider the large amount of nitrogen concurrently bound up in the tissues of macroalgae and microalgae along the estuarine floor, which would be assimilated even faster than that taken up by seagrass.

Because of the shallow depths of BB-LEH, there is a tight benthic-pelagic coupling, as has been demonstrated in other coastal lagoons as well. In these systems, water quality monitoring of nitrogen concentrations provides only a part of the database necessary to completely assess ecosystem condition – or source of nitrogen. It also does not reflect biogeochemical processing in bottom sediments, how much nitrogen is sequestered in the sediments that may vary from year to year (and may be released to the water column), and the role of benthic microalgae in removing nitrogen released from the sediments before it reenters the pelagic domain. These processes, again, affect nitrogen levels in the water column. If nutrient measurements are not made on biotic tissue and bottom sediments, they constitute important data gaps that need to be addressed by future research and monitoring programs.

Other studies (e.g., Touchette and Burkholder, 2000) reported that phosphate in the water column of seagrass habitats typically ranges from ~0.1 to 1.7 μM compared to higher concentrations in sediment pore water ~0.3 to 20 μM. Ammonium levels in the water column were reported at 0 to 3.2 μM in the water column compared to ~1 to 180 μM in sediment pore water. Finally, nitrate + nitrite concentrations were reported at ~0.05 to 8 μM in the water column compared to ~2 to 10 μM sediment pore water.

Eutrophic condition is closely tied to indicators of light availability, and these indicators are also closely coupled to seagrass success or failure. Macroalgal blooms occurred relatively frequently and impacted seagrass beds in BB-LEH 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, signaling an increase in recent years.

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. Monitoring for *A. anophagefferens* must be conducted with the proper technique and cannot be accurately measured by chlorophyll *a* concentrations since the species does not fluoresce with this pigment (Anderson et al. 1989, 1993). 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 ($\geq$ 200,000 cells mL^{-1}) and Category 2 blooms ($\geq$ 35,000 to $\leq$ 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}. This picoplanktonic alga can cause deleterious effects on hard clam populations at levels an order of magnitude below those that cause discoloration of the water.

A hard clam (*Mercenaria mercenaria*) 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 (see Heck and Valentine, 2007). 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 (e.g. phytoplankton size structure and species composition; picoplankton blooms) due to nutrient enrichment could have impacted the hard clam population.

Only 7 hard clams were found at 120 quadrat sampling stations in the estuary in 2010 for primary biotic data. In 2011, only 9 hard clams were found at these 120 quadrat sampling stations. Only 2 bay scallops (*Argopecten irradians*) were found at these sampling stations in 2010, and none in 2011. While hard clam and bay scallop data were evaluated to determine their appropriateness for potential inclusion as an indicator for the Index of Eutrophication, there were too few data points to be able to identify threshold values and conduct assessment. Hence, these data were not included in the Index of Eutrophication.

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. From 1997-2010, 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 geographically extensive estuary. Such widespread response signals a broad-scale stressor. We attribute this response to eutrophication resulting from nutrient loading to the estuary and associated light attenuation due to microalgal and macroalgal blooms that directly impact 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 a score of almost 70 to below 40, and in some cases even lower. 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. 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 Index of Eutrophication for this segment as compared to the central or south segments. The central and south segments are similar to each other, and over the years 1989-2010, both have undergone significant decline in condition associated with increasing eutrophication.

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 as documented by low dissolved oxygen (DO) levels. There were 82 occurrences of DO levels ≤ 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. Dissolved oxygen concentrations at and below 4 mg L^{-1} are important ecologically as low oxygen stresses commercially and recreationally important species of fish, invertebrates, and other organisms. Most of the low DO values observed occurred in the south segment (N = 63), with far fewer in the central segment (N = 13) and north segment (N = 6). These values represent DO measurements taken quarterly, mainly during the morning and afternoon (daylight) hours. Hence, the number of observations of DO below 4 mg L^{-1} is quite likely to be a significant underestimate of the number of DO violations that actually occurred during this time period because nighttime measurements were not made. While the estuary is designated as impaired in the north segment due to low DO, the data presented here indicate that the estuary is also likely to be impaired in the south segment due to DO levels below 4 mg L^{-1}.

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 which are undergoing eutrophication. 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 occurrence of sea nettle blooms in the north segment has posed a hazard to human use of some waters in the estuary. Lower salinity waters north of Toms River have had the greatest numbers of sea nettles. Blooms of sea nettles have increased in the past decade. Increasing eutrophic condition and hardened shorelines may have contributed to this 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.

The bioindicators examined and the Index of Eutrophication developed and applied in this study can support nutrient management planning. The report documents the extent and limitations of available data and provides a framework for holistic ecosystem monitoring for the future that can serve as a basis for future assessments of eutrophication condition. Currently, BB-LEH is highly eutrophic and is susceptible to nutrient loading. Total nitrogen and phosphorus are highest in areas with the highest percentages of urban and agricultural land, and with the lowest percentages of forested and undeveloped land. Nitrogen loads from areas covered with turf are about twice those of non-turf urban areas. Phosphorus loads from turf areas are more than eight times those from non-turf areas. Phosphorus concentrations, loads and yields are generally higher in areas with more development, and higher during runoff than in baseflow. Index of Eutrophication values declined in the central and south segments, indicating these segments are currently undergoing eutrophication. Eutrophication condition was worst in

the north segment despite modest improvements, in contrast to stages and trends in the south and central segments.

From 1989 to 2010, BB-LEH experienced low dissolved oxygen (82 times ≤4 mg L^{-1}), high total suspended solids (max >200 mg L-1) and chlorophyll *a* (max >40 μg L^{-1}), harmful algal blooms (≥200,000 cells mL^{-1}), epiphytic loading (mean values up to 38.3% cover of seagrass), macroalgae blooms (80-100% cover 36 times, 70-80% cover 19 times, 60-70% cover 10 times), habitat loss, >67% fewer clams, and degraded seagrass biomass (to 2.7±8.0 g m^{-2} aboveground; 17.9±37.5 g m^{-2} belowground). Index of Eutrophication 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 currently undergoing eutrophication. The north segment has already undergone eutrophication and remains highly eutrophic. The Index of Eutrophication values for the northern segment decreased markedly from 2009 to 2010.

Nutrient loading severely degraded BB-LEH and initial rapid declines highlight sensitivity of the estuary to loading and that a 'tipping point' may have been crossed beyond ~2,000 kg total nitrogen km^{-2} yr^{-1} or ~100 kg total phosphorus km^{-2} yr^{-1}. Collectively, the direct relationship between nutrient loading from the watershed and estuarine nutrient concentrations, the degradation of an array of biotic indicators, and the relationship between nutrient loading and the Index of Eutrophication supports the conclusion that BB-LEH is a highly impacted estuarine system.

A holistic management approach must be accelerated to remediate environmental problems in BB-LEH associated with nutrient enrichment due to ongoing development and land use-land cover changes in the watershed. Multiple corrective strategies should be applied concurrently, such as improved stormwater control systems, implementation of best management practices in the watershed, open space preservation, fertilizer controls, soil restoration, 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 total maximum daily load (TMDL) for nitrogen and phosphorus is also a necessary element to effectively mitigate the eutrophic condition of the estuary. Application of a TMDL should be pursued concomitantly with the other management approaches noted above. It is necessary to respond aggressively at this time to nutrient loading from the watershed because of the severity of the eutrophication problems in the estuary, which may become intractable if they are not remediated in the short term. A well-coordinated and holistic management plan is critical to improving the ecological condition and resources of the estuary. This is a long-term approach to remediate the eutrophication problems in the estuary.

INTRODUCTION

Human population growth and development in coastal watersheds of the U.S. have led to increasing impacts on estuarine and coastal marine environments (Vitousek et al. 1997, Lotze et al. 2006). 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 and drivers of change that affect shallow lagoonal estuaries. Nutrient and organic carbon loading has been an important driver of biotic and habitat change in these lagoonal systems (Nixon, 1995; McGlathery et al., 2007).

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 (Nixon et al., 2001; Burkholder et al., 2007; Anderson et al., 2010; Kennish and Paerl, 2010; Giordano et al., 2011; Howarth et al., 2011). 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 an ecologically impacted system. This is manifested by declining ecological conditions such as significant loss of seagrass, occurrence of nuisance and toxic algal blooms (including brown tides), heavy epiphytic loading, 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 40-45% of the estuarine shoreline), and other degrading changes. These adverse effects have become increasingly evident during the past 15 years. Extensive studies, peer-reviewed publications (including references therein), and numerous technical reports published on the estuary during the past two decades have clearly documented these problems (Bricker et al., 1999, 2007; Bologna et al., 2000; Kennish, 2001a; Lathrop and Bognar, 2001; Seitzinger et al., 1993, 2001; Gastrich et al., 2004; Kennish and Townsend, 2007; Kennish et al., 2007a, b; 2008, 2010, 2011; Lathrop and Haag, 2007; Kennish, 2009; Moore, 2009; Barnegat Bay Partnership, 2011; Fertig et al., 2012; Kennish and Fertig, 2012).

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 (Kennish, 2001a-c). 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 (JCPL, 1978; Kennish et al., 1984; Ecological Analysts, 1986; Kennish, 2001d). 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 and these shade benthic habitats, posing problems for estuarine benthic primary producers.

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, most notably eutrophication of the waterbody.

Eutrophication is defined as the process of nutrient enrichment and increase in the rate of organic matter input in a waterbody leading to an array of cascading changes in ecosystem structure and function such as decreased dissolved oxygen levels, increased microalgal and macroalgal abundance, occurrence of harmful algal blooms (HABs), loss of seagrass habitat, reduced biodiversity, declining fisheries, imbalanced food webs, altered biogeochemical cycling, and diminished ecosystem services (de Jonge and Elliott, 2001; Kennish and de Jonge, 2011). It poses the most serious threat to the long-term health of the estuary by altering ecosystem structure and function (Kennish and Townsend, 2007; Kennish et al., 2007a). 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 (Kennish, 2001a). Blooms of drifting, ephemeral macroalgae (e.g., *Ulva lactuca, Enteromorpha intestinalis,* and *Gracilaria tikvahiae*) have produced thick canopies of organic matter that pose a potential danger to the seagrass beds by smothering the plants and blocking light penetration (Kennish et al., 2007b, 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 microbial 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; McGlathery et al., 2007).

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 (Longstaff and Dennison, 1999; Carruthers et al., 2002; Orth et al., 2006; Burkholder et al., 2007; Kennish et al., 2008, 2010; Moore, 2009).

Seagrasses are highly responsive to epiphytic growth on leaf surfaces which can cause a significant decline in seagrass abundance, biomass, and other parameters. We found considerable biomass and areal cover of epiphytes on seagrass blades in the BB-LEH over the three-year period investigated (2009-2011). Despite their contribution to estuarine food webs, epiphytic assemblages reduce light availability to the seagrass blades, frequently resulting in considerable loss of plant biomass and areal cover (Sand-Jensen 1977; Sand-Jensen et al. 1985, 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, as can macroalgal cover.

Seagrass leaves provide excellent substratum for epiphytic organisms, which can contribute significantly to the total primary and secondary production of seagrass meadows, while concurrently impacting seagrass growth, production, and biomass (Bologna and Heck, 1999). 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).

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; Hughes et al. 2004). Duffy et al. (2001), employing mesocosm experiments, 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). Reduced animal grazer communities in eutrophic estuarine systems can result in significantly increased epiphytic overgrowth on seagrass surfaces and greater light attenuation for seagrass photosynthesis (Burkholder et al., 2007; Robert W. Howarth, Cornell University, personal communication).

The composition and abundance of epiphytic assemblages typically 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 availability gradient. The effect of nutrient enrichment on epiphytic loading was localized but pronounced.

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. 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. 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, macroalgae cover, 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 seagrass blades, which reduce light availability for their function (Hauxwell et al., 2001, 2003; McGlathery et al., 2007; Paerl et al., 2003, 2009). Hauxwell et al., (2001) showed that high macroalgal canopy produced in an estuary with high nitrogen loading rate (i.e., Waquoit Bay) adversely affected shoot density, growth rate, and production of eelgrass. Ochieng et al. (2010) also linked light reductions to lower eelgrass shoot densities, slower growth rates, stunted morphology, and higher mortalities.

Diminished light transmission to the estuarine floor can cause 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.

The loss of seagrass habitat has also plagued other coastal lagoons and even deeper estuarine systems in the mid-Atlantic region primarily due to nutrient enrichment and light attenuation (Stevenson et al., 1993; 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, hydrogen sulfide toxicity, and water-column nitrate inhibition may also contribute (Goodman et al., 1995; Burkholder et al., 2007).

Since 2003, eutrophy has generally worsened in much of the BB-LEH system (see Component 3), and the condition of the seagrass habitat has significantly degraded. For example, nutrient loading severely impacted Eutrophication Index values in the estuary particularly over 2003–2010. Seagrass biomass in the estuary decreased markedly over 2004–2006, 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 recorded in this water body. Reduced biomass levels persisted through 2011. Macroalgal blooms increased over 2004–2010 as well, and epiphytic overgrowth on seagrass was substantial. Seagrass areal cover within beds has also generally declined 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 (Seitzinger and Pilling, 1992; Seitzinger et al., 1993, 2001), 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 of the estuary has been closely coupled to development of the BB-LEH Watershed, and the history stretches across decades of time. Velinsky et al. (2011) reported that sediment nitrogen accumulation rates has increased twofold in northern Barnegat Bay salt marshes starting in the mid-1950s, reflecting an increase in nutrient loading from portions of the watershed. They also concluded that their salt marsh sampling sites remain impacted by anthropogenic disturbances and have not returned to natural, reference conditions; rather, the most recent changes suggest an increase in habitat deterioration and pollution. 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 populated, developed, and altered by human activity. In addition, the largest tributary systems (Toms River and Metedeconk River) discharge to northern Barnegat Bay and deliver the highest loading of nutrients (Wieben and Baker, 2009).

Brown tide blooms were most severe in 1999, 2000, 2001, and 2002 when cell counts of *Aureococcus anophegefferns* exceeded 1.5 x 10^6 cells mL^{-1} each year. Such high densities may cause serious shading impacts on seagrass beds. Hard clams also cease to grow above a threshold level of 400,000 cells mL^{-1}. A hard clam stock assessment conducted in LEH in 2001 revealed >67% reduction in clam abundance in LEH when compared with an earlier stock assessment conducted there 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, 2012) indicated that hard clams poorly digest picoplankton and other diminuitive phytoplankton species, which seriously impairs their growth. Shifts in the food web structure (e.g., phytoplankton size structure and species composition; occurrence of picoplankton blooms) of the estuary due to nutrient enrichment could have impacted the hard clam population. Brown tide blooms also impact hard clam larvae. According to Bricelj and MacQuarrie (2007), for example, brown tides at concentrations ≥200 cells μl^{-1} are expected to cause the failure of hard clam larval populations. Larvae exposed to these concentrations of brown tides have greater susceptibility to increased secondary mortality factors.

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 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). These mats often form a mosaic of thick algal canopies covering seagrass beds that produce patches of extensive bare-bottom areas on the estuarine floor due to light shading or blocking. Epiphytic burden on seagrass plants also causes light attenuation, exacerbating the adverse effects caused by macroalgal mats. 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 (Lamote and Dunton, 2006; Burkholder et al., 2007; McGlathery et al., 2007; Anderson et al., 2010). Such was the case at the Seawood Harbor area in the north segment of BB-LEH in July

2011, when massive macroalgae accumulation and decomposition events occurred, seriously impacting extensive water column and benthic habitats.

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 may signal 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.

Recurring eruptions of the sea nettle (*Chrysaora quinquecirrha*) have likewise occurred in the estuary since 2002, posing a potential hazard to human use, most notably estuarine waters in the north segment, and possibly causing biotic structural changes due to zooplankton cropping. These biotic changes can lead to further deterioration of the system via altered food web components, loss of biodiversity, and disruption of ecosystem structure and function. Sea nettle eruptions may be coupled to increased system eutrophy as well (Kennish and Fertig, 2012).

Shallow eutrophic estuaries and coastal lagoons often exhibit a range of ecological and biogeochemical responses to nutrient enrichment that signal a shift in the balance of selective forces shaping biotic communities and habitats. The net 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).

As system eutrophy increases, the concentrations of organic matter escalate due to increased macroalgal biomass and higher phytoplankton abundance. Concomitantly, there are losses of rooted macrophytes and reduced concentrations of dissolved oxygen in these eutrophied waters. Excessive eutrophication leads to loss of ecosystem structure and function (Valiela et al., 1992, 1997; Duarte, 1995; Taylor et al., 1995; Cloern, 2001; Smith et al., 2006; McGlathery et al., 2007; Giordano et al., 2011).

Nitrogen enrichment, when unchecked, causes significant disruption of estuarine ecosystem health (Nixon, 1995; Tomasko et al., 1996; Burkholder, 2001; Cloern, 2001; Nixon et al., 2001; Deegan et al., 2002; Rabalais, 2002; Burkholder et al., 2007; Kennish et al., 2007a; Anderson et al., 2010). In coastal lagoons such as BB-LEH, the organic fraction of dissolved nitrogen comprises the vast majority of the nitrogen pool and is at least minimally biologically available and utilized by harmful algal blooms (Anderson et

al., 2002, Glibert et al., 2001, 2010). 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 effects of long-term and progressive eutrophication are substantially degraded biotic and habitat components of the estuary.

OBJECTIVES

This study had several clearly defined objectives:

1. 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.

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

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

4. 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. Each of these parameters will be examined and assessed for statistical validity and inclusion in the index development for the 1989 to 2010 period

5. To generate an Index of Eutrophication 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 develop threshold levels of biotic decline and numeric loading criteria that can support an effective nutrient management plan.

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

7. 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 index methods developed from data collected through 2010.

STUDY AREA

Physical Characteristics

BB-LEH is a coastal lagoon located 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 in many areas of the estuary facilitates pollution retention and recycling in the system, 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. Population growth in the watershed increased by more than 65% between 1980 and 2010. At build out 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; Wieben and Baker, 2009).

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 et al., 2000; Lathrop and Bognar 2001). 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, with an important component also delivered by atmospheric deposition (Gao et al., 2007). In an earlier study, Hunchak-Kariouk and Nicholson (2001) calculated the total nitrogen load to the estuary of ~7.2×10^5 kg N yr^{-1}, with ~54% (3.9×10^5 kg N yr^{-1}) derived from surface water inflow, ~34% (2.4×10^5 kg N yr^{-1}) from atmospheric deposition, and ~12% (8.6×10^4 kg N yr^{-1}) from direct groundwater discharges. Wieben and Baker (2009) later estimated that the total nitrogen load to the estuary amounted to ~6.5×10^5 kg N yr^{-1}, with surface water discharge contributing 66% (4.3×10^5 kg N yr^{-1}), atmospheric deposition 22% (1.41×10^5 kg N yr^{-1}), and direct groundwater discharge 12% (7.8×10^4 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. The wastewater treatment plant outfalls are located ~15 km north and south of Barnegat Inlet in the nearshore ocean. Because of the distances of these outfalls from Barnegat Inlet and the large dilution component, the amount of the treated discharge entering

Barnegat Bay via tidal currents through the inlet is likely to be small. There are no quantitative data available on the amount of the wastewater treatment plant effluent that enters the bay from the ocean through the inlet. However, if this were substantial, it would be reflected in nitrogen measurements in the area of the inlet. Similarly, if large amounts of nitrogen were injected into Barnegat Bay from other sources in the coastal ocean, this would also be evident as elevated nitrogen measurements in water samples taken at Barnegat Inlet compared to other bay sites through time, but this has not been observed.

We have examined the total nitrogen concentrations in water samples collected by the NJDEP over a ~10-year period at six NJDEP water quality monitoring stations, two in lower Toms River (stations 1400R11 and 1506A), one just south of Toms River (station 1636A), two in the bay just inside of Barnegat Inlet (stations 1688B and 1691E) and one in the nearshore ocean near the inlet (station A47A). These water quality sampling stations were chosen to track the transport of nitrogen and the likely source and direction of nitrogen movement, either exiting or entering the bay. Box plots showing the concentrations of total nitrogen have been produced for these six sampling stations, and they clearly illustrate the likely source (Toms River) and exit point (Barnegat Inlet) of nitrogen in the BB-LEH system. These results are consistent with the USGS findings regarding nitrogen loading which indicate that Toms River is the major source of nitrogen entering Barnegat Bay. They also reveal that the inlet is the outwelling site for the nitrogen from the bay, not the site of major nitrogen entry from the coastal ocean.

Confined animal feeding operations (52 total) cover a very small area of the watershed (Figure 1 - 4). With only one exception of a centrally located feeding operation in the watershed, all are located in the northern portion of the watershed (Fertig et al. 2012). 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 and by this study.

The Oyster Creek Nuclear Generating Station (OCNGS), a 635 MW power plant that has operated commercially in the BB-LEH Watershed since December 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 (JCPL, 1978; Kennish et al., 1984; Ecological Analysts, 1986; Kennish, 2001d).

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 (Kennish, 2011a). The estuary, therefore, has been divided into three segments (north, central, and south segments) for data assessment in this project (Figure 1 - 5).

North Segment

The north segment extends from just south of the Toms River to the northern extremity at Bay Head (Figure 1 - 5). 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 (Figure 1 - 6). 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 - 7). 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 - 5). This segment is characterized by more rapid (hydrological) flushing and reduced water residence time than in the north and south segments (Guo et al., 2000), 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 - 5). 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, Figure 1 - 7). 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 - 7). 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.

METHODS

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 condition 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 as well (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 Index of Eutrophication. Several key biotic response variables were targeted in the estuary (i.e., seagrass, phytoplankton, HABs, macroalgae, epiphytes, benthic invertebrates, and 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.

COMPONENTS

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 Eutrophication 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

The methodology of Component 1 is briefly described here, and in detail in Appendix 1-1. 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 and 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, an Index of Eutrophication 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 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 are also differences in sediment composition, hydrography, and basin morphology between 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 (Figure 1 - 8, Figure 1 - 9, and Figure 1 - 10).

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 - 8). No sampling was conducted in the estuary in 2007. An array of water quality parameters was also measured at each station during biotic sampling.

Seagrass (biomass, shoot density, blade length, and areal cover), macroalgae (areal cover), epiphytes (areal cover and biomass), 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, core and hand sampling, as well as 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, and 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

Epiphyte biomass and areal cover measurements were made on seagrass samples collected over a three-year study period (2009-2011). Bimonthly epiphytic sampling and analysis were conducted at the field sampling stations over this three-year study period. Sample collection recording was noted on a field sheet (

) Growth of epiphytes on seagrass surfaces increases with nutrient enrichment leading to a decrease in light transmission, reduced photosynthesis, and loss of seagrass biomass.

Phytoplankton

Chlorophyll *a* measurements were 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 Index of Eutrophication.

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 (Cloern, 2001). 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 (Cloern, 2001). Measures of chlorophyll *a* are important in monitoring phytoplankton responses to nutrient enrichment, but not HABs such as brown tides which do not leave a clear chlorophyll *a* signal.

Macroalgae

The occurrence and percent areal cover of macroalgae were also recorded, over the 2004-2011 period, 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.

Drifting macroalgal populations are highly responsive to nutrient enrichment and thus are important indicators of eutrophic condition (Thomsen, 2012). The rapid increases in abundance of bloom-forming, sheet-like macroalgal forms have blanketed extensive areas of seagrass habitat in estuaries, blocking incident light and contributing to the loss of seagrass beds and the resident benthic and nektonic fauna (e.g., Short and Burdick, 1996; Hauxwell et al., 2001; Cardoso et al., 2004; Huntington and Boyer, 2008; Olyarnik, 2008).

Hard clams (*Mercenaria mercenaria*)

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.

Hard clams are typically more sensitive to local-scale conditions. Their response to persistent eutrophication and shifting phytoplankton size structure and species composition can be a decline in abundance and the loss of the resource. While hard clam abundance was examined, it was not included in the development of an Index of Eutrophication.

Benthic invertebrates

The development of an Index of Eutrophication 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.

Benthic invertebrate communities inhabiting eutrophic waters commonly 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.

Component 3: Index of Eutrophication 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 Eutrophication for each estuary segment.

An important goal of this project is to develop an Index of Eutrophication using water quality and biotic indicators to assess eutrophication, impairment, and overall ecosystem health of the BB-LEH estuary. While the current determination of the impairment of New Jersey's estuarine waters is based on measurements of a single parameter (i.e., dissolved oxygen), 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 Index of Eutrophication 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, ecosystem state, and biotic responses. No validated Index of Eutrophication 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 an Index of Eutrophication 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. These segments can be compared to provide an assessment of the entire BB-LEH system. 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 available biotic indicator measurements (Table 3-2) 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 Eutrophication 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 Index of Eutrophication 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. These are 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 are given, and additional data and analysis needed to improve the plan are listed.

QUALITY ASSURANCE/QUALITY CONTROL (QA/QC)

The Quality Assurance Project Plan (QAPP) for this project (Baker and Kennish, 2010) can be found in Appendix i - 1. The QAPP specifies the field and laboratory methods used in the project and the QA/QC procedures. A sample field sheet is provided in Table i - 1. Appendix 2 of the QAPP (pages 89-91 of that document) lists the SOPs employed for data sonde measurements and biotic measurements in the project. Analytical methods employed by the NJDEP to collect water-quality data used as a source of secondary data in this report are shown in the QAPP, Appendix i - 1, Table A7-1. The Methods Detection Limits (MDL) for each type of analysis are also included in this table.

Table i - 2 gives QA/QC results for this project based on the Measurement Quality Objectives (MQOs), which are listed in Appendix i - 1, Table A7-2. QA/QC results for specific data collected on this project are listed in Table i - 3. Minor sampling changes relative to the QAPP are reported in Table i - 3. These occurred during the first year of fieldwork on the project (sampling period 3 in 2010). These minor deficiencies did not affect the data collected and were not used in calculations of the Index of Eutrophication. Changes noted in Table i - 3 were acted on, corrected, and not repeated during the remainder of the project.

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 the 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 in Appendix 1 - 1.

COMPONENT 1: SUMMARY

The following is a summary of "Concentrations, Loads, and Yields of Total Nitrogen and Phosphorus in the Barnegat Bay-Little Egg Harbor Watershed, New Jersey, 1989-2011, in Support of Investigating Spatial and Temporal Variability of Conditions in the Estuary," hereafter referred to as "the USGS watershed report." This report describes in detail the annual and seasonal loading of total nitrogen (TN) and total phosphorus (TP) to the BB-LEH Estuary for years 1989-2011. The objective of this summary is to provide the reader with a basic understanding of the temporal and spatial variability of nutrient loading to the estuary, which is necessary for understanding the nutrient-dependent water-quality and biotic cycles in the system. The reader is encouraged to read the entire USGS watershed report, which is included in its entirety as Appendix 1-1.

The purpose of the watershed loading investigation was to quantify the amount and variability of nutrient loading as a function of year (1989-2011), season (growing and nongrowing), land use, location (northern, central or southern watershed segment), hydrologic basin, and stream hydrologic condition (runoff or baseflow). The watershed investigation was accomplished with secondary (existing) data. Water quality, hydrologic, precipitation and land-use data and a spatially explicit model (PLOAD, Version 3.0, U.S. Environmental Protections Agency, 2001) were used. Contributions of lawn-care products to nutrient loading were assessed by quantifying the turf coverage and relating that to nutrient concentrations and loads. Total nitrogen and total phosphorus concentrations are reported as milligrams per liter (mg/L), loads are reported as kilograms (kg), and yields (area-normalized loads) are reported in as kilograms per

hectare (kg/ha). Once calculated, the TN and TP loads from each set of spatial and temporal conditions were available for use in relating the effects of nutrient loading to estuary health, as described in the remaining components of this report (Components 2-5).

The approach taken to estimate concentrations, loads and yields of TN and TP was to acquire and evaluate all available applicable data, obtain or develop relations for estimating values where data were not available, and calculate nutrient values from the assembled measured and estimated values. The methodology used required the following data categories: land-use data at the 14-digit hydrologic unit code (HUC-14) and watershed segment level; volumetric flow data for streams in the watershed and for each HUC-14 subbasin during runoff and baseflow conditions; daily, monthly and annual precipitation data from available meteorological stations; and water-quality data for streams in the watershed. The following four paragraphs briefly describe the data used, and additional details are presented in Appendix 1-1.

Land use. The BB-LEH watershed contains 81 subbasins at the HUC-14 scale. Distribution of land use in each of the HUC-14 subbasins was classified for five distinct years by evaluating the percentage of land in each land-use category, based on land-use land-cover digital datasets produced by the New Jersey Department of Environmental Protection for 1986, 1995, 2002, and 2007 (New Jersey Department of Environmental Protection, 1986, 2001, 2008, 2010) and from the Geographic Information Retrieval and Analysis System (GIRAS) for 1973 (Appendix 1-1, Table 1). These years are hereafter referred to as "land-use years". Land-use categories used in this study are: agriculture, barren, forest, impervious urban, residential urban, non-residential urban, water, and wetland. The percentage of each land-use type and percentage of turf coverage was determined for each HUC-14 area. Land-use percentages and turf coverage were then calculated for each watershed segment as the area-weighted percents and coverages of the HUC-14 areas of each segment.

Hydrology. Mean daily streamflow data from six continuous, real-time USGS stream gaging stations and baseflow separation procedures were used to determine annual and seasonal baseflow and runoff volumes for those six streams. Two baseflow-separation methods were used to determine the percentages of flow for each stream represented by baseflow and runoff. Flow volumes (runoff and baseflow) for the remainder of the BB-LEH watershed were estimated from relations between flow volumes at those six streams and precipitation data.

Precipitation. Precipitation data were retrieved from the Office of the New Jersey State Climatologist and from the National Climatic Data Center. Daily precipitation values were needed to determine whether baseflow or runoff conditions were in effect for all days from 1970-2011, and annual and seasonal precipitation totals were used for estimating runoff and baseflow totals at seasonal and annual increments. Average precipitation totals from stations in and near the BB-LEH watershed were used by PLOAD in runoff loading calculations, and average precipitation totals for the northern and southern halves of the watershed were used in baseflow loading calculations.

Water quality. Water-quality data were retrieved from the USGS National Water Information System (NWIS) database; the New Jersey Department of Environmental Protection (NJDEP); Brick Township Municipal Utilities Authority (BTMUA); the USEPA STORET database, which includes data collected by the USEPA, NJDEP, BTMUA and other agencies, and is composed of the Legacy Data Center (collected prior to 1998), and The Storet Warehouse (1998 and later); and other agencies. Data collected before 1970 was not used, as reliability and sensitivity (detection and reporting levels) were considered insufficient for older data. Extensive quality assurance measures were implemented to ensure that all water-quality data were in consistent and correct units and met all other quality criteria as defined by the Quality Assurance Project Plan (QAPP). Details related to data quality screening are included in Appendix 1-1.

Water Budgets

Land-surface and groundwater-based water budgets can be used to calculate runoff and baseflow values (Gray, 1970; Charles et al., 2001; Gordon, 2004; Walker et al., 2011 in Appendix 1-1). Water-budget concepts were applied to the BB-LEH watershed, based on past investigations of surface- and groundwater hydrology conducted in basins within the watershed. It was concluded that, for annual and seasonal water-budget calculations, there is no significant net change in storage in the unsaturated zone or aquifer; that withdrawals and artificial discharges to the streams are not substantial compared to the baseflow and runoff volumes; that net loss to or gain from adjacent basins is generally not substantial; and that virtually all recharged water is discharged back to the stream upstream from the gage. Therefore, the streamflow hydrology of BB-LEH, on an annual and seasonal basis, is dominated by precipitation, evapotranspiration, runoff, and baseflow. Baseflow was estimated from continuous streamflow measurements at gaging stations, baseflow separation, and relations between baseflow data from gaging stations and precipitation. Runoff was estimated by using the watershed-loading application PLOAD for each HUC-14 from precipitation and user-entered land permeability values; Additional details are given in Appendix 1-1.

Determination of Baseflow-Mean Concentrations

Mean concentrations of total nitrogen (TN) and total phosphorus (TP) during baseflow (baseflow-mean concentrations, or BMCs) were calculated for each land-use category. This involved first calculating BMCs for a set of subbasins, then determining the best-fit multiple-linear-regression relating TN or TP to the percent of each land-use category. The regression coefficient for each land-use category is the BMC for that land use. The process is described in greater detail in Appendix 1-1. To account for variability in concentrations at the sampling sites, only sites with five or more TN or TP baseflow values for a given land-use year (1973, 1986, 1995, 2002, or 2007) were selected. To be included in the calculations, the water-quality data samples must have been collected on a day when baseflow conditions were in effect. Baseflow conditions were defined as streamflow after a recession period of two to four days had transpired since the last precipitation event has occurred. The duration of the recession period depended upon the area of the subbasin, as specified in generally accepted surface-hydrology literature.

Runoff Load Calculation Using PLOAD

PLOAD

PLOAD requires the following data inputs: GIS land-use data; GIS watershed delineations; impervious factor for each land-use type; annual or seasonal precipitation; and annual or seasonal pollutant loading rates (event-mean concentrations, EMCs) for each land-use type. A comprehensive description of PLOAD input requirements and instructions is provided in the PLOAD User's Manual (U.S. Environmental Protections Agency, 2001), and is presented in Appendix 1-1.

Percent impervious values similar to those in published literature and the PLOAD manual were used in this investigation. Seasonal and annual precipitation totals were calculated from monthly totals at stations in or near the watershed, published by the meteorological agencies. EMCs, which are flow-weighted concentrations, were first determined for water-quality sites with sufficient data density to provide a representative sampling of streamflow and concentration values. Several sites in and near the BB-LEH watershed for which extensive stormwater sampling had been reported were selected. The EMCs for these sites were directly calculated from streamflow and water-quality data. To determine the EMCs for land-use categories, a multiple-regression procedure similar to that used for baseflow-mean concentrations was used, in which a best-fit equation was developed where the error between the EMCs calculated from water-quality data and from land-use percentages was minimized. More detail is available in Appendix 1-1.

Turf Analysis

A substantial portion of the watershed consists of single-family dwellings or other types of land uses with extensive areas in lawns, also referred to as turf. Remote-sensing data and geographic information systems (GIS) were used to map and quantify turf areas across the Barnegat Bay-Little Egg Harbor watershed. The New Jersey Department of Environmental Protection (NJDEP) spring 2007 color infrared aerial photography was used as the basis for the image analysis. The 2007 NJ Land Use/Land Cover data set was used to extract out urban land use areas for further analysis. The objective was to delineate what areas in urban land uses were dominated by turf/lawn land cover. More details about the turf identification and quantification are given in Appendix 1-1.

Although the original intention of this classification work was to distinguish intensively managed from less intensively managed turf, a visual assessment of the Random Forest classification results indicated that such classification was not feasible, so intensively and less intensively managed turf were grouped into one category. The accuracy assessment indicates that turf was mapped with an approximately 90% accuracy and a kappa statistic of 0.75. The turf mapping was deemed to be of sufficiently high accuracy to be used to investigate relations between turf area and nutrient loads in the watershed.

EVALUATION OF AVAILABLE WATER-QUALITY DATA

Total nitrogen data available for years 1970-2011 consisted of 1,316 values of suitable quality from 68 sites throughout the watershed. A total of 2,341 total phosphorus

values were available for 107 sites in the BB-LEH watershed. ANOVA by ranks test, followed by a Tukey multiple comparison test, determined that the order of total nitrogen and total phosphorus concentrations among the watershed segments based on available water-quality data is:

$TN_{north} > TN_{south} > TN_{central}$

$TP_{north} > TP_{south} > TP_{central}$

The order of median concentrations of both total nitrogen and total phosphorus among watershed segments is consistent with the order of percent developed land in the segments. Appendix 1-1 contains additional analysis and discussion of the variability of historical TN and TP concentrations as a function of streamflow (baseflow vs. runoff, and during the course of a storm), season, and land use.

ESTIMATES OF TOTAL NUTRIENT LOADS

Concentration, load, and yield data were determined at the HUC-14 scale for baseflow and runoff conditions; these data are provided in Table 15 of Appendix 1-1. Loads are shown at the watershed segment scale in Table 16, and for the entire watershed in Table 17 of the Appendix. The four factors that control loading at various scales (watershed, segment or HUC-14) are land use, land area, contaminant concentration, and stream flow. All four factors dictate that the north segment should have the highest loads, as was verified with hydrologic and water-quality data. During the period of study 1989-2011, total surface-water loads of TN (baseflow plus runoff loads) for the entire BB-LEH watershed ranged from about 448,000 kg as N (1995) to more than 850,000 kg as N (2011) (Table 17 of the Appendix). The north segment accounted for an average of 65.7% of the annual TN load, and the central and south segments accounted for 17.7 and 16.5%, respectively. Total phosphorus (TP) loads for the watershed ranged from 21,000 (1995) to 37,000 kg as P (2011). Similar to TN, about 65.2% of the TP load was contributed by the north segment, 18.4% by the central, and 16.3% by the south segments. The large percentage of loads discharging from the north segment is attributed to a combination of factors: the north segment is more than twice the size of the central or south segments, contains the Toms River and Metedeconk River which together make up more than 60% of the streamflow in the watershed, and contains greater proportions of agricultural, and residential and non-residential urban lands, each of which are associated with greater mean concentrations than undeveloped land. The corresponding north segment of the estuary is the smallest of the estuarine segments (Figure 2 of the Appendix). In addition to loading amounts, differences between the size of the watershed and estuarine segments in the north may be a factor contributing to higher nitrogen concentrations in the northern part of the estuary, as previously reported in Seitzinger et al. (2001) and Kennish and Fertig (2012). TN loads are slightly higher for the central segment than the south segment of the watershed, even though there is a greater proportion of urban development in the south segment, due to the larger size and greater streamflow of the central segment.

Base-Flow Loads on the Watershed Scale

Using baseflow separation to determine annual and seasonal baseflow amounts, and relationships between base-flow mean concentrations and land use, nutrient baseflow loads were estimated by year and season for each HUC-14 subbasin in the BB-LEH watershed for the years 1989-2011. Annual TN baseflow loads by HUC-14 subbasin

ranged from 318,000 kg as N (2002) to 677,000 kg as N (2011), and annual TP baseflow loads ranged from 12,500 kg as P (2002) to 25,900 kg as P (2011) (Table 17, Figure 18 of Appendix 1-1). Figure 18 of Appendix 1-1 shows that there appears to be a gradual increase in baseflow loads for 1989-2011; however, that increase is masked by a large amount of inter-year variability resulting from precipitation (and resulting streamflow) patterns. For both TN and TP, the relative contribution of baseflow loads during the growing and non-growing seasons is nearly equal, with the growing season accounting for an average of 51%, and the non-growing season accounting for an average of 49%, of the annual baseflow loads.

Base-Flow Loads on a Segment Scale

For TN, annual base-flow loads for the north segment ranged from approximately 207,000 to 437,000 kg as N, comprising an average of 65.0% of the annual TN baseflow load for the watershed (Table 16 of the Appendix). The central segment contributed 58,000-124,000 kg as N and the south segment contributed 54,000-115,000 kg as N, accounting for an average of 18.3 and 16.7% of the annual TN baseflow load, respectively (Table 16). For TP, annual baseflow loads for the north segment ranged from 7,800 to 15,900 kg as P, comprising an average of 62.2% of the baseflow TP load for the watershed. The central segment contributed 2,600-5,400 kg as P and the south segment contributed 2,200-4,600 kg as P accounting for an average of 20.5 and17.3% of the annual TP baseflow load for the watershed, respectively.

Base-Flow Loads on a HUC-14 Scale

Although there are a greater number of subbasins in the north segment that contribute the highest loads (during either dry or wet years), subbasins that contribute high baseflow loads are also found in the central and south segments, particularly along the coast. The two principal variables that determine nutrient loading on a HUC-14 scale are land use and total HUC area.

Base-Flow Yields on a HUC-14 Scale

A complete list of all yields estimated for each HUC-14 subbasin for 1989-2011 is found in Table 15 of Appendix 1-1. Yield values are load values normalized by the HUC-14 area, which explains why the HUC-14 subbasins with the highest loads do not necessarily correspond to the HUC-14s with the highest yields. Load values are more appropriate for estimating the rate of nutrient loading to the estuary, whereas yield estimates are more useful for assessing the effect of surface activities (land use) on loading from a given land area. Subbasins with the highest yields in baseflow are primarily concentrated in the northern part of the watershed, and have higher proportions of agriculture and urban land. Subbasins with the lowest yields are dominated by forests.

ESTIMATES OF RUNOFF NUTRIENT LOADS AND YIELDS

Runoff Loads on the Watershed Scale

Using PLOAD, nutrient runoff loads were estimated by year and season for each HUC-14 subbasin in the BB-LEH watershed for the years 1989-2011. Between 1989 and 2011, runoff loads for the entire watershed were approximately 98,500 to 182,600 kg TN and approximately 6,500 to 12,000 kg TP. Greater contribution of both total nitrogen and

total phosphorus during the growing season is attributed to the use of higher EMCs during the growing season and greater length (more days) of the growing season.

Runoff Loads on a Segment Scale

Annual TN runoff loads for the north segment comprised about 68.6%, and the central and south segments each contributed about 15.7%, of the annual TN runoff load. Annual TP runoff from the northern segment accounted for an average of 72.1% of the annual runoff, and the central and south segments contributed 13.8 and 14.1%, respectively. Loads contributed by the north segment are substantially higher than the other two segments in part because the land area of the north segment is considerably larger, and the greater amount of urban development in the north.

Runoff Loads on a HUC-14 Scale

As with baseflow loading, most HUC-14s with the highest runoff loads are located in the northern portion segment and along the eastern edge of the mainland part of the watershed.

Runoff Yields on a HUC-14 Scale

Subbasins with the highest TN and TP yields in runoff are located primarily in the northeastern corner of the watershed, and are dominated by urban land uses. Subbasins with the lowest yields are predominantly forested.

Relations between Turf Coverage, Land Use, and Nutrient Loads

About 67.8% of the watershed area was deemed, by satellite imagery analysis for 2007, to be about 32.2% of developed land and is most likely to include turf. About 8.0% of the watershed has been classified as developed-turf, and 24.2% as developed-non-turf. There is a strong relationship between percent turf and percent developed land as shown in Figure 25 (Appendix 1-1), such that percent turf within the watershed typically increases with percent development, and turf can be considered a reasonable predictor of the amount of development in the watershed. Discussion and exceptions to this observation are given in Appendix 1-1.

There appears to be strong relations between percent turf and annual yields of TN and TP in this watershed (Figure 26 of Appendix 1-1). When separated into baseflow and runoff, a stronger relation between yields in runoff and percent turf, than between yields in baseflow and percent turf is evident. This relation is most noticeable for total phosphorus (Figure 26 (E) of Appendix 1-1).

TN and TP loading from developed-turf areas appear to be ~twice that of developed-non-turf areas. The high N and P concentrations associated with turf are likely the result of fertilizer products being applied to lawns. The higher mean nitrogen concentration in urban non-turf areas compared to undeveloped areas shows that factors in addition to turf are contributing nitrogen loads above background levels in urban areas.

COMPONENT 2: ESTUARINE BIOTIC RESPONSES

INTRODUCTION

This section of the report briefly describes the data available and included in the overall study. Note, however, that the main objective of this component of the study is to characterize biotic responses in the estuary to nutrient enrichment using existing datasets collected between 1989 and 2010. Water quality measurements were spot checked concurrently with seagrass sample collection. 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. An Index of Eutrophication is also calculated to quantify the current and historical state of estuarine eutrophic effects. Several key bioindicators are used in development of the index.

METHODS

Water Quality Measurements

Water quality measurements were made in the BB-LEH Estuary during seagrass sampling conducted from 2004-2011. Water temperature, salinity, dissolved oxygen, pH, and depth were recorded at each sampling station on all sampling dates in 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 collected at 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 physicochemical measurements in BB-LEH were derived as secondary data from long-term (1989-2011) quarterly water quality monitoring databases of 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. This was done to identify differences along a depth gradient. Eelgrass samples were collected during each of three 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 randomly 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. Accuracy was ensured through photographic records, which were used for spot-checking and validation. 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

Total nitrogen and total phosphorus loading are the two key indicators of ecosystem pressure used in this project. 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. Qualitatively, however, we note that water residence times may also play a role in the susceptibility of the estuary to ecosystem pressures. Water residence time in the estuary ranges from 24 days in winter to 74 days in summer, when eutrophication is most pronounced (Guo et al., 2004). Long residence times are important because it leads to retention and recycling of nutrients rather than their dilution or export associated with

faster rates of oceanic exchange and flushing.

ECOSYSTEM STATE: WATER QUALITY

The second major category of data organization is ecosystem state which incorporates key water quality indicators (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 estuarine segment.

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}. These do not include any nighttime measurements of DO.

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 were generally > 2 m in all 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 (Burkholder et al., 2007; McGlathery et al., 2007; Anderson et al., 2010). As such, it must be considered as a factor influencing light availability to the benthos.

The areal percent cover of macroalgae in this study 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 and Figure 2 - 10 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. Macroalgae 'Early Blooms' (70–80%) occurred twice during 2004–2006 and 17 times during 2008–2010 (chi square $p < 0.01$). Macroalgae 'Full Blooms' (>80%) occurred 12 times during 2004–2006 and 24 times during 2008–2010 (chi square $p < 0.05$). 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 (Fertig et al. 2012).

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 extensive bare bottom areas. *Ulva lactuca* blooms were particularly damaging.

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 (Valiela et al., 1997; McGlathery, 2001). 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 - 4). Conversely, while no significant relationships between macroalgae percent cover and *R. maritima* aboveground or belowground biomass were observed during June-July 2004-2010 or August-September 2004-2010, they 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).

Epiphyte Percent Cover

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 following the methodology of Miller-Myers and Virnstein (2000). Using a razor, the epiphytes were subsequently scraped off of both sides of the blades and oven dryed at 60°C for 48 hours to determine their biomass (Frankovich and Zieman, 1995). 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 mean 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.)

Eelgrass (*Zostera marina* L.) is an important indicator of overall ecosystem health of an estuary because it integrates water quality and benthic attributes (Longstaff and Dennison, 1999; Carruthers et al., 2002; Orth et al., 2006; Burkholder et al., 2007; Kennish et al., 2008, 2010; Moore, 2009).

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, although the response of primary productivity and loss of seagrass does not appear to be linear in coastal bays like BB-LEH (Borum and Sand-Jensen, 1996; Nixon et al., 2001; Robert W. Howarth, Cornell University, personal communication) . The trend of eelgrass decline over the years has not been isolated to one bed but has been widespread in the estuary, 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; Fertig et al., 2012).

An estuary-wide survey was conducted in the summer of 2009 to measure the current extant of seagrass habitat across the BB-LEH system (Lathrop and Haag, 2011). Aerial imagery collected during the months of July and August 2009 was interpreted and mapped using an object oriented image analysis technique, similar to techniques used in the 2003 mapping survey. A boat-based *in situ* dataset was collected concurrently with the aerial photography to assist the image interpretation and for an independent accuracy assessment. We compared the remotely sensed mapping of seagrass cover change (in 2003 vs. 2009) vs. the *in situ* plot-based sampling conducted by Kennish et al. from 2004 through 2010. Appendix 2 - 1 ("Comparison of Remotely Sensed Surveys vs. *In Situ* Plot-based Assessments of Seagrass Condition in Barnegat Bay- Little Egg Harbor") provides detailed results. Comparison of the remotely sensed vs. the *in situ* plot change analysis suggests that the two methodologies had broadly similar results, with the percent area showing declines in percent cover being greater than those that exhibited increases. In conclusion, the two studies provide corroborating evidence that seagrass has declined in percent cover in the BB-LEH system during the decade of the 2000's.

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 - 11). Aboveground and belowground biomass varied considerably among sampling transects in the estuary (Figure 2 - 12 and Figure 2 - 13).

Figure 2 - 14a-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 - 14d-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 - 15), 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.

It is important to track changes in shoot density of eelgrass in BB-LEH over the past 30 years. Vaughan (1982) reported eelgrass shoot densities in the estuary in 1979-1982 ranging from ~500-1000 shoots m^{-2}. Bologna et al. (2000) documented eelgrass shoot densities in the estuary in 1999 ranging from ~650-1150 shoots m^{-2}. Over the 2004-2010 period, we found eelgrass shoot densities ranging from ~150-650 shoots m^{-2}. These values clearly reveal much lower eelgrass shoot densities over the past decade.

Eelgrass Blade Length

Figure 2 - 16 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

Diver observations and underwater videographic imaging delineated areal cover of eelgrass and widgeon grass in the estuary (Haag et al., 2008; Kennish et al., 2010). 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 - 17). 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 - 18 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). However, blade lengths over the 2004-2011 time period were much less that those reported in the late 1990s by Bologna et al. (2000). 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 was negatively related to total nitrogen concentrations (Figure 2 - 19). 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 (Figure 2 - 20, Figure 2 - 21, Figure 2 - 22, Figure 2 - 23). 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 seven years.

The most complete data sets for widgeon grass in the central and south segments were reported in 2005 and 2010 (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

Macroalgae

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, Olyarnik 2008).

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). Macroalgal bloom events increased in the estuary over the 2004-2010 period (see above). Orth et al. (2006) documented that seagrasses have high light requirements that approach 25% of the incident surface radiation (Dennison et al., 1993; Gallegos, 2001; Orth et al., 2006). Light extinction by macroalgae mats during bloom development threatens seagrass integrity (Twilley et al., 1985; Burkholder et al., 2007; Lee et al., 2007, Huntington and Boyer 2008, Olyarnik 2008). 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 from sediments which may sustain system eutrophy.

Macroalgal blooms have been shown to contribute to significant decline of seagrass beds in other nutrient-enriched coastal lagoons (McGlathery et al. (2001, 2007). Their impacts can be far reaching, altering the structure and function of these systems (Valiela et al., 1997; Lyons et al., 2012). In these systems, 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), working in Chesapeake Bay which is a much larger estuarine system, has shown that macroalgal canopies can be detrimental to seagrass beds in deeper systems. As the algal standing stocks increase, shading reduces the photosynthetic oxygen production of seagrass plants, causing diebacks (Twilley et al., 1985; Hauxwell et al., 2001, 2003; 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).

Deegan et al. (2002) demonstrated through manipulative experimentation how macroalgae alter eelgrass ecosystem support of higher trophic levels. When macroalgae were removed, eelgrass abundance increased as did the water column and benthic

boundary layer oxygen concetntrations. In addition, the lower macroalgal biomass resulted in higher fish and decapod abundance and biomass, demonstrating the importance of macroalgae in altering seagrass ecosystem support of higher trophic levels. These findings are supported by the work of Lyons et al. (2012) who recorded macroalgae bloom impacts on the structure and function of marine ecosystems.

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 load (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 of eutrophied systems (Eyre and Ferguson, 2002; McGlathery et al., 2007). 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).

Harmful Algal Blooms

Blooms of the pelagophyte *Aureococcus anophagefferens* were first identified by immunofluorescence in BB-LEH samples in 1995. However, Anderson et al. (1989) initially recorded the presence of *A. anophagefferens* (at 400 cells mL^{-1}) in an archived sample collected in the estuary in September 1986. Sieburth et al. (1988) showed that light or epifluoescence microscopy cannot be used to accurately identify *A. anophagefferens*, as was also noted by Bricelj et al. (2012).

Brown-tide blooms caused by *Aureococcus anophagefferens* were most pronounced in BB-LEH between 1995 and 2002 (Gastrich et al., 2004) (Table 2 - 9). While brown tides reached high densities during this span of years, they have not been monitored in the estuary since 2004. No observational HAB monitoring data are available since then. Nevertheless, the occurrence of brown tide blooms occurred during the period of increasing eutrophication documented in the estuary during the 1990's and after 2000 (Seitzinger et al., 2001; Bricker, 1999, 2007; Kennish et al., 2007).

Because *Aureococcus anophagefferens* does not leave a clear chlorophyll *a* signal, blooms typically go unnoticed and unchecked without a comprehensive and consistent monitoring program for HABs. Without such a program, there are likely to be underestimates of HAB events and their impacts in the estuary. Anderson et al. (1993) noted that the identification of brown tide with standard light microscopy is "uncertain," and therefore an inaccurate and unreliable way to identify and one incapable of accurately quantifying brown tides. Caron et al. (2003) showed that the application of a monoclonal-antibody technique was an effective way to detect and enumerate brown tides. Popels et al. (2003) used a quantitative polymerase chain reaction method (Popels et al., 2003) to accurately detect and enumerate brown tides.

Brown tides have also been reported in New York coastal bays since the mid-1980's, 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 (Bricelj et al., 1984; Bricelj and MacQuarrie, 2007; Bricelj, 2009). Wazniak and Glibert (2004) showed that elevated levels of brown tide significantly reduced growth of hard clams). Bricelj and MacQuarrie (2007) reported that brown tides at ≥200 cells μl^{-1} are expected to cause metamorphic failure of hard clam larval populations, leading to their increased vulnerability to secondary mortality factors. Dense shading of these blooms may reduce the abundance and distribution of seagrass beds (Cosper et al., 1987; Dennison et al., 1989), 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 other estuaries that also exhibited impacts on natural resources (Gastrich et al., 2004).

Abundances of *Aureococcus anophagefferens* in the estuary were classified using the Brown Tide Bloom Index 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 (> 16ºC) 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).

Studies have been conducted on the forms of nitrogen that stimulate and sustain brown tide blooms. Dissolved organic nitrogen may be more important than dissolved inorganic nitrogen in stimulating these blooms (Glibert et al. 2001, 2010) and in coastal lagoons similar to BB-LEH dissolved organic nitrogen has been observed as >90% of the dissolved nitrogen pool (Figure 2 - 24). Regenerated forms of nitrogen (i.e., ammonium, urea, and dissolved organic substrates) may be the primary drivers of picoplanktonic blooms in coastal lagoons with protracted water residence times. However, Mulholland et al. (2011) more recently found that *Aureococcus anophagefferens* is nutritionally versatile and capable of using a wide range of nitrogen sources, both organic and inorganic, to meet nutritional requirements. This finding suggests that both inorganic and organic forms of nitrogen may be involved in the generation of brown tide blooms.

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 ºC 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 showed that some seagrass habitat lies within the High-Risk Category 3 bloom 'hotspot' areas and therefore should be monitored on an annual basis (Gastrich et al., 2004).

Shellfish

Few live shellfish samples were collected during 2010 and 2011 during primary biotic data collection (Table 2 - 10). For example, bay scallops (*Argopecten irradians*) were observed extremely rarely during field sampling at 120 quadrat stations in 2010 and 2011. Only two live individuals were observed during 2010 and none during 2011. Similarly, hard clams (*Mercenaria mercenaria*) were also observed extremely rarely during field sampling. There were seven live individuals were observed during 2010, and nine live individuals were observed during 2011. Occasionally, there was evidence of dead shellfish (such as shell hash), but these were only observed 17 times during 2010 and four times during 2011.

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 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).

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 may 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.

Bricelj et al. (2012) stated that the increase in estimated mortality between the State surveys conducted in the 1980s and 2001 suggests that in addition to lower recruitment, an increased mortality rate is also reducing the population in Little Egg Harbor. According to Bricelj et al. (2012), this increased mortality rate may be a significant part of reduced recruitment as well. Furthermore, they contend that the trends provide evidence for historically poor and possibly declining recruitment and declining population over time in Little Egg Harbor. It is unknown, however, whether the hard clam decline has occurred estuary wide, although Bricelj et al. (2012) note that anecdotal reports indicate substantial decrease in the numbers of hard clams in Barnegat Bay as well.

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. Bricelj et al. (2012) stress that *A. anophagefferens* can cause deleterious effects on hard clam populations at levels an order of magnitude less than those that cause discoloration of the water (200,000 cells mL^{-1}).

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 - 25). 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 - 11), 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: INDEX OF EUTROPHICATION DEVELOPMENT

INTRODUCTION: BUILDING ON THE NEEA REPORT

We applied the basic methodology used in the National Estuarine Eutrophication Assessment (NEEA) Model to develop an Index of Eutrophication 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. Bricker et al. (2007) define 'susceptibility' as "a measure of a system's nutrient retention based on flushing and dilution" (p. 12) and note "susceptibility is influenced by the flow of water. The flushing capacity of a system is determined by tidal action and the amount of freshwater flowing in from its tributaries." 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. Additionally, due to heterogeneity of benthic habitats, sediment grain size, sediment total organic content, and other factors that vary along an east-to-west gradient, it is necessary to consider representativeness of any potential benthic indicator dataset along this east-west gradient (Figure 1 - 6, Figure 1 - 7). 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 groups: (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 project QAPP.

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 involving 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.

Here we provide a 'roadmap' for Component 3 of this report. This component of the report begins with a review and comparison of a previous eutrophication assessment tool (the National Estuarine Eutrophication Assessment). It continues with the goals of the Index of Eutrophication developed for this project. Substantial effort during this project (documented in project Progress Reports, responses to the Technical Advisory Committee, presentations, and other formats) went into identifying, assembling, characterizing, analyzing, and evaluating available datasets and databases. These efforts are described. Qualitatively, datasets and databases were examined for availability, completeness, and representativeness. Quantitatively, datasets and databases were examined through a variety of methods for statistical rigor, robustness, and representativeness. The goal of these efforts was to determine the suitability of including variables within these datasets and databases as indicators for inclusion in the Index of Eutrophication, as specified in the project QAPP (p. 60 of that document). The results of this effort are shown with documentation of the data availability and data gaps for Barnegat Bay-Little Egg Harbor over the study period (1989–2010; validation during 2011 is documented in Component 4). Suggestions and recommendations for filling data gaps through additional monitoring efforts are included to some extent here, but more substantially in Component 5. Details of the evaluation process for primary and secondary data are described in detail followed by the final list of indicators included in

the Index of Eutrophication. Specific details of methods for dataset assembly are then addressed. These are followed by a thorough documentation of the methods used for the Index of Eutrophication. This includes a detailed section on the determination of thresholds for each indicator, and detailed methods for calculating the Index of Eutrophication. A step-by-step example is included for illustration (Appendix 3 - 7). Following the methodology, several sections of results are documented. Results are broken down into: 1) indicator scores, 2) Raw Scores for components of the Index of Eutrophication, 3) weighting indicators into components, 4) Weighted Scores, 5) component indices and the overall Index of Eutrophication, 6) a brief section on validation, which is expanded on in Component 4. The Discussion section includes the limitation of this approach upfront before delving into the conclusions and findings of the Index of Eutrophication. This section, Component 3 of the report, ends with a summary of the main findings and conclusions.

GOALS OF THE 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 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 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.

AVAILABLE DATA / DATA GAPS

Substantial effort during this project went into identifying, assembling, characterizing, analyzing, and evaluating available datasets and databases. Qualitatively, these were examined for availability, completeness, and representativeness. Quantitatively, these were examined through a variety of methods for statistical rigor, robustness, and representativeness. The goal of these efforts was to determine the suitability of including variables within these datasets and databases as indicators for inclusion in the Index of Eutrophication, as specified in the project QAPP (page 60). Data availability is detailed below. Data from 2011 data was kept separate for validation purposes (see Component 4). Many data gaps were identified. All data (both primary data generated for this project and secondary data generated from other sources) and potential indicators were scrutinized and evaluated as described below.

Data included in the index were assembled from a variety of sources and were available (and unavailable) asynchronously over time (years) and space (estuary

segment). Data available for inclusion are shown in Figure 3 - 2 Temporal and spatial data availability for indicators used in the Index of Eutrophication. 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 were available that year. Note that applicability of the index to any given segment depends in part on availability of data within that segment.

Ecosystem Pressures: Total nitrogen loading and total phosphorus loading data are 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. No 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 was measured directly 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 primarily in the southern two thirds of the estuary, corresponding to the central and south segments. *Ruppia maritima* is present primarily 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 from 1995, 1999–2002, 2005, and 2010, but the spatial extents are variable and so assessments will only be conducted for the entire estuary. Data were obtained from reported literature values (Gastrich et al. 2004).

Benthic invertebrate data are available during 2001 from the REMAP data for all three segments (Table 3 - 1, Figure 2 - 25). These data were provided by Darvene Adams, U.S. Environmental Protection Agency, Edison, New Jersey.

Additional secondary datasets were available and considered as to suitability for inclusion for incorporation into the Index of Eutrophication. The evaluation process for these datasets is described below in the following section. Additionally, other datasets were generated by separate, ongoing projects concurrently with this project. Ideally, pertinent projects and datasets could have undergone the process of evaluation as to suitability for inclusion in the Index of Eutrophication. Timing, as a matter of practicality, however, was also necessary to consider for the successful completion of the current project without delay, hindrance, or expansion of the scope beyond that stated in the QAPP. Any potential dataset considered for inclusion into the Index of Eutrophication, or which the Index of Eutrophication is applied to must be generated completely and undergo the rigorous evaluation process to determine its suitability according to the goals and objectives of the Index of Eutrophication. Coordinating multiple separate projects was not possible for this project. The flexible framework of the Index of Eutrophication, however, does allow future datasets, such as ongoing years of water quality monitoring data, to be considered and applied by others after this project is completed.

EVALUATION PROCESS FOR INCLUSION OF SECONDARY DATA INTO THE INDEX OF EUTROPHICATION

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 approved Quality Assurance Project Plan (QAPP). Specifically, the QAPP (p. 60) specified that these include: “dissolved oxygen, Secchi depth, total nitrogen (loading), total phosphorus (loading), chlorophyll *a*; seagrass biomass, shoot density, blade length, areal cover, and epiphytic overgrowth, macroalgae abundance and areal cover, brown tide blooms, shellfish (hard clam) resource, and estuarine susceptibility (water residence time).”

Substantial effort was placed on researching each of these potential indicators, the available data sources, and developing a process for establishing thresholds for each indicator and how each component is integrated into the Index of Eutrophication. While many data gaps were identified, and some were filled over the course of this project, it was not possible to incorporate additional or new datasets while adhering to the rigorous timeline of this project. Examples of qualitative and quantitative criteria for inclusion of datasets are the number of records, location and span of dates of data collection, and ability to describe and detect heterogeneity between segments and years. This includes spatial representativeness within a segment for a given year. 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, and assessment of data availability. Within the limits of data availability, the determination to include or exclude a particular dataset was made based on the representativeness of the sampling within the spatial and temporal scope of this project. Ideally, the aggregated database for this project would be as holistic and comprehensive as possible. However, when aggregating multiple datasets collected for a variety of purposes, it is necessary to avoid bias associated with sampling design that were not designed with the current purposes of this project in mind. Inclusion of datasets that are not representative of the temporal and spatial scale of this current project would result in biased and inaccurate conclusions from this project. 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).

Residence times, available seasonally, were gathered from the literature (Guo et al. 2004). However, results from hydrodynamic modeling were unavailable for incorporation by this project. The limited availability of residence time data did not meet the criteria for number of records, location and span of dates of data collected, and the ability to describe and detect heterogeneity throughout the estuary. This rendered infeasible the determination of estuarine susceptibility with respect to water residence time. Therefore, water residence time and estuarine susceptibility could not be assessed for this project.

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 co-varied with total loading, for both total nitrogen and total phosphorus (Figure 3 - 3), as indicated by principal component analysis. This high level of co-variation 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 - 3). However, flow-weighted average total concentration for total nitrogen did not elicit a response in light indicators (Figure 3 - 4) nor seagrass indicators (Figure 3 - 5). Also, flow-weighted average total concentration for total phosphorus did not elicit responses in light indicators (Figure 3 - 6) nor seagrass indicators (Figure 3 - 7). 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.

Seagrass was scrutinized as a potential indicator for inclusion in the Index of Eutrophication. Eelgrass (*Zostera marina* L.) is an important indicator of overall ecosystem health of an estuary because it integrates water quality and benthic attributes (Longstaff and Dennison, 1999; Carruthers et al., 2002; Orth et al., 2006; Burkholder et al., 2007; Kennish et al., 2008, 2010; Moore, 2009). There is a substantial database

regarding eelgrass available from 2004–2010 (excepting 2007) in the central and south segments (Component 2). Eelgrass condition has become degraded and substantial declines in biomass (both aboveground and belowground) have been observed (Component 2, Fertig et al., 2013). Therefore, it was necessary to consider the potential future utility of this indicator in Barnegat Bay-Little Egg Harbor in the case that previous trends continue into the future. It is, unfortunately, not possible currently or within the scope of this project to predict the future of eelgrass in New Jersey waters, especially given the high variability associated with seagrass demographics (Orth et al., 2006). While the current trend of *Z. marina* in Barnegat Bay-Little Egg Harbor is grim, the characteristics of the eelgrass beds have been changing over time, with rates of decline of biomass slowing in recent years (2008–2010) as compared to previous (2004–2006) surveys (Fertig et al., 2013).

Various types of statistical analyses (e.g. shifting from parametric to non-parametric statistics) were suggested by the Technical Advisory Committee to address the question of including eelgrass as an indicator for the Index of Eutrophication. Non-parametric statistics do not assume a normal distribution (Sokal and Rohlf, 1981) and, for example, are appropriate for variables containing many values equal to zero, but are not sufficient to answer the question of including or omitting eelgrass data. Differences between transects were analyzed statistically according to a variety of methods including non-parametric analysis, as detailed in the QAPP (Sokal and Rohlf, 1981; Quinn and Keough, 2002; Underwood, 1997). These statistical analyses require distinguishing between an observed absence and missing data. 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. Further, non-normally distributed indicators (e.g. eelgrass) were transformed into normally distributed variables, i.e. Raw Scores, by rescaling eelgrass indicator data into Raw Scores according to threshold equations. Details for this methodology are provided below. Recognizing this important distinction, we ensured 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, and therefore eelgrass was concluded to be suitable as an indicator for inclusion in the Index of Eutrophication. However, there were too few instances of available GIS layers of seagrass areal coverage over the time span of the study period, failing to meet the criteria of sufficient data availability to effectively use this dataset as an indicator for assessment by the Index of Eutrophication.

The QAPP also stated (p. 60) “Benthic invertebrate data will also be examined and assessed for statistical validity and inclusion in the index development for the 1989 to 2011 period.” The Rutgers field sampling was considered as a primary source of shellfish data (bay scallops and hard clams) as potential indicators for the Index of Eutrophication. Benthic invertebrate data were also available for BB-LEH from datasets provided by U.S. EPA from the National Coastal Assessment (NCA, http://www.epa.gov/emap/nca/) and the Regional Environmental Monitoring and Assessment Program (REMAP, http://www.epa.gov/emap2/remap/index.html).

The Rutgers field sampling and primary data source for shellfish (bay scallops and hard clams) as potential indicators for the Index of Eutrophication yielded too few observations of bay scallops and hard clams (see Component 2). Only two live bay scallops (*Argopecten irradians*) were observed during 2010, and zero scallops were observed during 2011. Similarly, hard clams (*Mercenaria mercenaria*) were also extremely rare and only seven live individuals were observed during 2010, and nine live individuals were observed during 2011. These observations were thus insufficient to provide information for identifying or calibrating thresholds. Further, there was an insufficient quantity of observations to yield any quantitative assessment. Thus, neither bay scallops nor hard clams could be used as indicators, nor could they be included in the Index of Eutorphication. Qualitatively, though, it is concluded that the shellfish resources are dramatically depleted from historic populations (Bricelj et al., 2012).

While shellfish (hard clam) resource was identified as a potential indicator in the QAPP, the only such secondary data received was based on National Marine Fisheries Service hard clams landing data, which is recognized to be a measure of fishing pressure and only partially (if at all) attributable as a biotic response to the condition of eutrophication, and further, this data does not account for predation or other mortality causes. Therefore, this dataset could not be used in the Index of Eutrophication, though some general, qualitative comments are made in the Discussion section of this report. Some historical shellfish census data were available for Little Egg Harbor, but did not extend beyond this one estuarine segment, being only available for one year during the study period, and could only be partially (if at all) attributed as a biotic response to the condition of eutrophication. Therefore, this secondary dataset could not be included in the Index of Eutrophication. These qualitative and quantitative evaluations indicate that the shellfish datasets did not meet criteria for BB-LEH and thus could not be included in index calculations.

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 sampling times and sampling locations for Barnegat Bay-Little Egg Harbor Estuary for the REMAP (Figure 2 - 25, Table 3 - 1) and NCA (Table 2 - 11, Table 3 - 2) datasets are documented.

The REMAP dataset contains high spatial coverage, but only during summer of 2001. Since only one year of data was available, the REMAP dataset could not be validated, though this 2001 REMAP dataset has 80 samples that were randomized throughout the bay and sufficiently span the habitat gradients in Barnegat Bay (Figure 2 - 25). Yet sampling for the REMAP dataset occurred in a north to south direction (Figure 2 - 25). While this was efficient logistically for sampling, it introduced a serious source of spatial bias in the data (Sokal and Rohlf, 1981; Quinn and Keough, 2002; Underwood, 1997). For example, if differences in biotic response (e.g. abundance, species composition, etc.) were found between north, central, and south segments, were these due to the environmental and nitrogen loading gradients characteristic of Barnegat Bay, or were they due to the timing over the course of the summer and associated variation in temperatures, salinities, or other seasonally changing variables? What would an interaction (combination of influence) between environmental factors and timing mean,

and how much would each contribute? Seasonal bias could not be removed from the dataset, severely limiting the confidence of interpretations from this dataset. Further, based on a Pearson correlation analysis, 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 and are not necessarily reflective or responsive to eutrophication.

For future monitoring via REMAP or other benthic macroinvertebrate dataset collection, we 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.

The NCA dataset has data for several years (2000–2006), yet there are an insufficient number of sampling locations each of these years to adequately and/or representatively sample any segment during any of these years.

Due to heterogeneity of geology, morphology, bathymetry, sediments, water circulation, residence time, benthic habitats, sediment grain size, sediment total organic content, and other factors that vary along an east-to-west gradient, it is necessary to consider representativeness of any potential benthic indicator dataset along this east-west gradient in addition to the three segments along the north-south gradients (Figure 1 - 5, Figure 1 - 6, Figure 1 - 7, Figure 1 - 8, Psuty, 2004). 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 BB-LEH must sample adequately across the known gradients. Examination of three segments is supported by the QAPP, as described above, which allows for departures from the NEEA ASSETS approach that assessed two segments of BB-LEH.

Quite importantly for inclusion in the calculations for the Index of Eutrophication, the answer to the question ‘Can the NCA and REMAP datasets 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? In this case, 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 to calculating annual values of the Index of Eutrophication.

Quantitative examinations of the NCA and REMAP datasets included subjecting these and other datasets to statistical tests, as mandated by the QAPP. 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.

Several ANOVA tests were conducted to test for statistical differences in water quality and benthic habitat between the three north-south segments to gather further evidence for making a decision regarding the NCA data. A p values less than 0.05 was considered statistically significantly different (Sokal and Rohlf, 1981; Quinn and Keough, 2002; Underwood, 1997). Results of these ANOVA tests demonstrate that statistically significant differences between segments were observed for all watershed, water quality, and sediment variables but not for NCA benthic invertebrate abundance (Table 3 - 3). 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, and 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.

*Principal components analysis (PCA) were conducted on both the REMAP and the NCA datasets, individually and combined. Benthic invertebrate abundances for the three most abundant taxa (*Ampelisca vadorum, Mytilus edulis*, and* Spirobidae*) representing the majority of individuals observed were examined by PCA for the REMAP and the REMAP combined with NCA datasets. These three species datasets are plotted on principal component axes, labeled by segment (Figure 3 - 8a) and by taxa (Figure 3 - 8*

Figure 3 - 8b). 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. There was no difference whatsoever between the results of the REMAP data alone and the REMAP data combined with the NCA data since there were so few observations in the NCA dataset. Note also that combining the NCA data with the REMAP data is inappropriate 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 too few data points each year (see Table 3 - 2) to yield reliable assessment scores. From the quantitative analysis, it was concluded that the NCA dataset could not be included in the Index of Eutrophication.

INDEX OF EUTROPHICATION: FINAL LIST OF INDICATORS USED

As discussed in greater detail (See the section Available Data / Data Gaps), the final indicators selected for inclusion into the Index of Eutrophication were 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 component. Sections below describe how these indices are integrated to calculate the Index of Eutrophication.

ECOSYSTEM PRESSURES INDEX

1) Ecosystem Pressures
    - Total Nitrogen Loading (kg TN yr^{-1} $segment^{-1}$)
    - Total Phosphorus Loading (kg TP yr^{-1} $segment^{-1}$)

INDEX OF EUTROPHICATION

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)

HARMFUL ALGAL BLOOM INDEX

5) Harmful Algal Blooms
    - *Aureococcus anophagefferens* concentration (cells mL^{-1})

BENTHIC INVERTEBRATE INDEX

6) Benthic Invertebrates
    - EMAP index values

METHODS: 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 was 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.). The Quality Control at this stage refers to ensuring that the data were handled correctly by the statistical software, rather than assurances regarding the measurements themselves (i.e. non-detects). Different software packages handle data entry and data importation differently (e.g. allowing empty cells, entering a '.' or 'NA' for empty cells, etc.). It is critical to ensure that missing data are appropriately distinguished from observations of 0 in the statistical software packages. Missing data are not non-detects since no attempt at detection has been made. Zero values are not necessarily non-detects. For instance, percent cover of seagrass or macroalgae in a given quadrat may be 0% for an individual observation. In this instance, this value should not be treated as a non-detect because the visual estimation has sufficient power to correctly determine this value. The Index of Eutrophication is sensitive to the difference between zeroes and missing data/non-detects. The Index of Eutrophication 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 ensured 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. 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. 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.

METHODS: DETERMINING THRESHOLDS: RESCALING DATA

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. 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 and standardizes their ranges, 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). By comparing observations to a spectrum of conditions (i.e. 'thresholds'), the Index of Eutrophication provides a continuum of response, from "Healthy" to "Degraded". This practice is common in the literature (Bricker et al. 1999, 2007, Wazniak et al. 2007, Williams et al. 2009). Validation of the methodology is conducted both through comparison of multiple similar methods, and through the response in 2011, as data from that year were kept separate and out of the analyses.

Thresholds are defined according to values of indicators and their relevance to biological, physiological, and ecological condition. Thresholds were defined based on thorough examination of: (a) the literature review, (b) analysis of the assembled database for calibration to BB-LEH, (c) Best Professional Judgment (in cases where a, and/or b are unavailable), and (d) some combination of a-c, in that order of priority. Best Professional Judgment was used as sparingly as possible, and the reasoning and justification for the judgment is documented if it was used for an indicator. Best Professional Judgment was not used to determine thresholds for an indicator if sufficient information was available either through the literature or data analysis. Best Professional Judgment is reserved only for indicators where previous thresholds are not established in the literature and data analysis yielded limited insight. 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 are defined values. They are not a mean and have no associated error. 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. The BB-LEH database was analyzed for each segment of the bay, because these segments have been determined to be heterogeneous habitats.

Observations of indicators are 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 is the mathematical relationship between an indicator's threshold values and the corresponding Raw Scores. Since some equations are exponential or logarithmic, the intervals between thresholds are not always equal. The equations are used to calculate a Raw Score by inputting observations as x values, and calculated y values are the Raw Scores. Rescaling equations are shown in Table 3 - 4.

In this section, we describe in detail the process of selecting thresholds for each indicator, the sources and methods considered, and the thresholds that were ultimately used. The following sections describe the methods for summarizing data, applying the rescaling equations to calculate Raw Scores, the weighting of indicators and calculating Weighted Scores. This process is described more fully in later sections, with an example calculation.

One major challenge to the identification and definition of thresholds based on

data reported in the scientific literature and for data assembled or collected for this project is that the response of indicators (biological or otherwise) were rarely starkly or drastically step-wise in function. That is, the values of thresholds are not obvious nor do indicators respond in discrete manners. Rather, ecosystems respond to various levels of stressors through continuous linear or non-linear manners with interactive effects since multiple stressors generally contribute simultaneously, in conjunction with natural processes and variability. Furthermore, many variables act as both a response and a stressor. As one of only many possible examples, macroalgal cover responds to nutrient loading as macroalgal biomass, percent cover, and frequency of intensity increases as nutrient loading increases (Figure 2 - 10, Table 2 - 4) and so can be used as a biological indicator of eutrophication (Kennish et al. 2011). Yet, the presence of macroalgal cover co-located with seagrass beds serves as a stressor to seagrass at it provides shading and therefore reduces the light availability beneath it and can severely degrade seagrass condition (Figure 3 - 9). Ultimately, ecosystems respond to stressors in complex and interactive manners, and therefore it is unrealistic to expect to find an obvious cusp or threshold for any given individual stressors or response variable.

Though thresholds for indicators or response variables are not obvious or stark, there is a high degree of confidence in the thresholds that we have identified based on the numerous literature studies and volume of data that were analyzed in order to derive these thresholds. By harnessing multiple independent studies for ecosystems similar to BB-LEH as well as the long-term dataset available for this project, this project has analyzed a large volume of data, and its results are consistent with overall understanding of estuarine ecology and ecosystem health assessments.

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.

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}). These indicators were generated as output from the model results of Component 1 of this study (see Component 1 and Appendix 1-1).

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 (see, for example, Wazniak et al., 2007; Bricker et al., 1999; Bricker et al., 2007; Tomasko et al., 1996; Short and Burdick, 1996; Deegan, 2002; Valiela et al., 2000; Burkholder et al., 2007; Boynton et al., 1996; Kennish and Fertig, 2012; Stevenson et al., 1993; Duarte et al., 1995; Kiddon et al., 2003). In looking for potential thresholds among these relationships, we sought 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 or below which it appears that nitrogen loading is no longer a dominant factor in the change of the biotic response.

In examining and compiling information from the literature, loading rates for total nitrogen and total phosphorus were converted to kg N $year^{-1}$ for comparison with common units to modeled loads from BB-LEH (Component 1 of this report). Tomasko et al. (1996) and Burkholder et al. (2007) report that as nutrient loading increases, seagrass biomass and productivity decline exponentially with very sharp declines starting at ~50 kg N day^{-1}, an inflection point in the curve at ~100 kg N day^{-1} and a slower rate of decline above ~225 kg N day^{-1}. A similar type of response is seen for seagrass areal coverage in that the inflection point of the curve was below 1,000 kg N km^{-2} $year^{-1}$ and a slower rate of decline was observed above 5,000 kg N km^{-2} $year^{-1}$ (Short and Burdick, 1996; Burkholder et al., 2007). Also, seagrass areal coverage declined most dramatically at incipient levels of eutrophication, early on in the long-term analysis (Valiela et al., 2000.

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 (see, for example, Wazniak et al., 2007; Bricker et al., 1999; Bricker et al., 2007; Tomasko et al., 1996; Short and Burdick, 1996; Deegan, 2002; Valiela et al., 2000; Burkholder et al., 2007; Boynton et al., 1996; Kennish and Fertig, 2012; Stevenson et al., 1993; Duarte et al., 1995; Kiddon et al., 2003). As nutrient loading increases, seagrass biomass and productivity decline exponentially (Tomasko et al., 1996, Figure 3 - 10), as does areal coverage (Short and Burdick, 1996, Figure 3 - 11 and Valiela et al., 2000, Figure 3 - 9). Seagrass shoot density is highly variable and declines rapidly with nitrogen loading as low as 50 kg N $year^{-1}$, which slows with greater than 1,000 kg N $year^{-1}$, though at this higher loading rate the density approaches (but does not reach) 0 shoots m^{-2} (Deegan et al., 2002; Burkholder et al., 2007, Figure 3 - 9). Seagrass declines are mediated by linear increases in estuarine total nitrogen concentrations, with total nitrogen concentration in $\mu M = 39.4 + 0.53$ * the annual total nitrogen load in g N m^{-2} $year^{-1}$, as has been found in Maryland's coastal bays (Boynton et al., 1996; Burkholder et al., 2007, Figure 3 - 12) and in BB-LEH, with total nitrogen concentrations in $\mu g\ N\ L^{-1} = 52.42 + 1.76$ * the areal normalized subwatershed total nitrogen loading in kg TN km^{-2} $year^{-1}$ (Kennish and Fertig, 2012, Figure 3 - 13). In an analysis of 62 estuarine embayments, Latimer and Rego (2010) found that at $\leq$ 50 kg TN loading ha^{-1} $year^{-1}$, seagrass extent was variable and likely controlled by other ecosystem factors unrelated to nutrient loading, but above that rate eelgrass coverage declined markedly and was essentially absent at loading levels $\geq$ 100 kg TN loading ha^{-1} $year^{-1}$ (Figure 3 - 14).

Additional potential thresholds for total nitrogen loading were identified from changes in response indicators with changes in loading. This is particularly important to calibrate the thresholds to be relevant for BB-LEH. For Figures 3-15 through 3-20 the blue line indicates a regression. We examined total nitrogen loading impacts on water

quality indicators, notably temperature, dissolved oxygen, and estuarine total nitrogen and total phosphorus concentrations (Figure 3 - 15). We examined the impact of total nitrogen loading impacts on light indicators notably, chlorophyll *a*, total suspended solids, Secchi depth, macroalgae percent cover, and the ratio of epiphyte to seagrass biomass (Figure 3 - 16). We examined the impact of total nitrogen loading on seagrass indicators, specifically aboveground and belowground biomass, shoot density, percent cover, and blade length (Figure 3 - 17).

There are fewer estuarine studies that examine the relationship between total phosphorus loading and biotic responses than for the relationship between total nitrogen loading since, in general, nitrogen – not phosphorus – is the limiting nutrient for estuarine systems. Nevertheless, both phosphorus and nitrogen are important to control for estuarine watersheds, particularly those with high levels of nutrient loading, as the receiving estuaries can be phosphorus limited, nitrogen limited, or co-limited, and the nutrient that is most limiting can change both seasonally and spatially (Conley et al., 2009; Conley, 1999; Malone et al., 1996). The analyses on data assembled for this project described above were also performed for total phosphorus.

We examined total phosphorus loading impacts on water quality indicators (Figure 3 - 18), light indicators (Figure 3 - 19), and seagrass indicators (Figure 3 - 20). Again, here we looked for values of total phosphorus loadings that marked a change in the rate of decline of response indicators and for values of total phosphorus loadings that marked the start of declines (regardless of rate), and for values above which it appeared that nutrient loading was no longer a dominant factor in the change of the biotic response.

Total nitrogen concentrations increased with total nitrogen loading (Figure 3 - 15) and with total phosphorus loading in the north segment (Figure 3 - 18). Chlorophyll *a* concentrations did not appear to vary below 2,000 kg total nitrogen km^{-2} yr^{-1}, but increased linearly above ~5,000 kg total nitrogen km^{-2} yr^{-1} (Figure 3 - 16) and above ~250 kg total phosphorus km^{-2} yr^{-1} (Figure 3 - 19). All seagrass indicators declined substantially with increased total nitrogen loading (Figure 3 - 17) and total phosphorus loading (Figure 3 - 20). These declines were exponential decreases for biomass (both aboveground and belowground (Fertig et al., 2013) as well as for blade length and shoot density (Figure 3 - 17, Figure 3 - 20).

Based on the above observations and analyses, thresholds for total nitrogen loading and total phosphorus loading were defined. Defined thresholds for Ecosystem Pressures are listed in Table 3 - 11. The rescaling equations that are generated from these thresholds are listed in Table 3 - 4.

Note that since the Ecosystem Pressures only receive Raw Scores, the scores for these indicators range from 0 to 100. Ecosystem Pressures are not weighted through the PCA method because there are only two indicators, and thus PCA is not meaningful. Total nitrogen loading and total phosphorus loadings are thus not calculated but rather defined with a weighting of 50% each. Raw Scores for these indicators are averaged together to create the Ecosystem Pressure Index. The analysis conducted justifies this weighting and there is a lack of evidence justifying a different weighting for this two Ecosystem Pressures. Maximum and minimum nutrient loading values for rescaling are listed in Table 3-2. As described in more detail below, Ecosystem Pressure scores are

kept separate from the other indicators used in the Index of Eutrophication to avoid confounding assessment of causal indicators from response indicators.

Ecosystem State: Water Quality

Water quality thresholds were defined by examining the literature and through analysis of data assembled in this project. 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 - 7, Table 3 - 8), and for total nitrogen, total phosphorus, and chlorophyll *a* (Table 3 - 9) for Maryland's coastal bays.

Temperature, dissolved oxygen, total nitrogen concentrations, and total phosphorus concentrations were all determined to be important indicators of water quality through principal component analysis. While temperature and total phosphorus were positively correlated, these indicators arise from different sources, are different ecologically, and total phosphorus and total nitrogen were not correlated (Figure 3 - 21). Thus, they were determined to provide different pieces of information, and both were included as indicators of water quality. 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).

Optimal temperatures for growth and photosynthesis of seagrass (Lee et al., 2007) guided determination of temperature thresholds (Table 3 - 10). Temperature from April to October (inclusive) was considered with respect to these values for determining thresholds. In general, seagrass has peak aboveground biomass during summer months and minimal aboveground biomass during winter months (see Component 2). Lee et al. (2007) report the optimal temperature for eelgrass growth is 15.3 ± 1.6 °C, and the optimal temperature for eelgrass photosynthesis is 23.3 ± 1.8 °C (Table 3 - 10). Temperatures above 30 °C stress eelgrass though even prolonged exposure to 26 °C can also induce physiological stress (Burkholder et al., 2007; Lee et al., 2007). In addition to physiological stress of seagrass reported in the literature, analysis of the BB-LEH database revealed several relationships with temperature. There was greater variability of chlorophyll *a* concentrations above 15 °C (Figure 3 - 22). Total suspended solids and Secchi depth were inversely related to temperature (Figure 3 - 22). There was an apparent inflection point of macroalgae percent cover at ~12 °C (Figure 3 - 22). Seagrass shoot density had an apparent inflection point at ~12 °C (Figure 3 - 23).

In determining thresholds for dissolved oxygen in BB-LEH, we considered literature information, the New Jersey standard of impairment that is currently established at 4 mg L^{-1}, and analysis of the assembled database. Dissolved oxygen is a physiological requirement for fish, shellfish, and other invertebrates. Breitburg (2002) and Diaz and Solow (1999) provided literature information on physiological stress and lethal minimum oxygen concentrations. Breitburg (2002) reports seasonal patterns of dissolved oxygen in the bottom layer of a seasonally stratified temperate estuary that has undergone substantial degradation and experiences seasonal hypoxia (Figure 3 - 24). When not seasonally stressed (i.e. in winter months), dissolved oxygen concentrations can reach

~10 to 14 mg L^{-1} in the bottom layer. Due to its shallow depth and thorough mixing, BB-LEH does not stratify seasonally and is more similar to the surface layer of stratified estuaries. Therefore, dissolved concentrations in BB-LEH should exceed those of bottom layers of stratified estuaries. As dissolved oxygen concentrations reach hypoxic and anoxic conditions, lethality increases (Figure 3 - 25) and benthic communities become stressed, decreasing biomass and diversity (Figure 3 - 26, Table 3 - 7, Ritter and Montagna 1999). Chlorophyll *a* concentrations in BB-LEH were inversely related to dissolved oxygen concentrations, but total suspended solids, Secchi depth, macroalgae percent cover, and epiphyte to seagrass biomass ratio were all correlated positively with dissolved oxygen (Figure 3 - 27, Figure 3 - 28). Wazniak et al. (2007) report cutoff values for dissolved oxygen (Table 3 - 8) as < 3 mg L^{-1} 'Does not meet objectives', 3-5 mg L^{-1} 'Community threatened', 5-6 mg L^{-1} 'Borderline', > 6 mg L^{-1} 'Meets objectives', and > 7 mg L^{-1} 'Better than objectives'. Deviations from optimal temperatures were considered for threshold values. Yet limitations of dissolved oxygen monitoring noted above in previous sections create a systematic bias that misses low nighttime concentrations. These differences, in conjunction with a comparison of the primary production in BB-LEH to that of similar coastal lagoons (Fertig et al., 2009, 2013a,b, In Press; Kennish and Fertig, 2012) necessitated adjusting the dissolved oxygen thresholds upwards from the literature values in accordance with values of dissolved oxygen observed in BB-LEH and the New Jersey standard of impairment, established at 4 mg L^{-1}.

Elevated nutrient concentrations spur phytoplankton and macroalgal growth and degrade seagrass (Burkholder et al., 2007, Figure 3 - 29, Figure 3 - 30). Kemp et al. (2004) document statistically derived concentrations of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) under a variety of salinity regimes beyond which submerged aquatic vegetation is not present (<0.15 mg L^{-1} and < 0.01 mg L^{-1}, respectively, which equates to < 150 µg L^{-1} DIN and < 10 µg L^{-1} DIP) in mesohaline regions (Table 3 - 6). Kemp et al. (2004) note that these thresholds are to be applied to median values of raw data collected during the growing season (April-October, inclusive). Further, Kemp et al. show the logarithmic relationship between increasing Total DIN concentration and increasing epiphyte biomass under a variety of dimensionless optical depth regimes, where optical depth = K_d * Z = the attenuation coefficient * depth (Figure 3 - 31). Inflection points for these relationships range from 10 µM Total DIN (equivalent to 140 µg L^{-1} Total DIN) where optical depth is greatest (i.e. clearer water) to 30 µM Total DIN (equivalent to 420 µg L^{-1} Total DIN) in more opaque water (Figure 3 - 31).

Dissolved inorganic nitrogen, however, only comprises a small fraction of the total nitrogen in the water column that can be bioavailable (Figure 2 - 24), undergo uptake and recycling via the microbial loop and food webs, and thus thresholds nitrogen in estuarine waters must account for this, which can be done by utilizing total nitrogen concentrations as an indicator (Wazaniak et al. 2007).

Wazniak et al. (2007) report cutoff values for total nitrogen and total phosphorus concentrations used for Maryland's Coastal Bays (Table 3 - 9) as follows (in mg L^{-1}): Total Nitrogen < 0.55 mg L^{-1}, < 0.64 mg L^{-1}, 0.65 – 1.0 mg L^{-1}, 1.0 – 2.0 mg L^{-1}, > 2.0 mg L^{-1} (this is equivalent to < 550 µg L^{-1}, < 640 µg L^{-1}, 650 – 1000 µg L^{-1}, 1000 - 2000 µg L^{-1}, and > 2000 µg L^{-1}) and Total Phosphorus < 0.025 mg L^{-1}, < 0.037 mg L^{-1}, 0.038 –

0.043 mg L^{-1}, 0.044 – 0.100 mg L^{-1}, and > 0.100 mg L^{-1} (this is equivalent to < 25 µg L^{-1}, < 37 µg L^{-1}, 38 - 43 µg L^{-1}, 44 - 100 µg L^{-1}, and > 100 µg L^{-1}). Analysis of the assembled database revealed that in BB-LEH, seagrass biomass (both aboveground and belowground) had decreased markedly at total nitrogen concentrations greater than 400 µg L^{-1} (Figure 2 - 14, Figure 3 - 32, Fertig et al. 2013) and at total phosphorus concentrations greater than 40 µg L^{-1} (Figure 3 - 33). Defined thresholds for Water Quality indicators are listed in Table 3 - 11. The rescaling equations that are generated from these thresholds are listed in Table 3 - 4.

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 primary producer communities (Dennison et al. 1993, Table 3 - 12, Figure 3 - 34). 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. This renders 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.

Indeed, Burkholder (2001) found that light reduction had a greater negative effect on seagrass shoot production than did increased nitrogen availability (Figure 3 - 35). Light availability thresholds are determined from the literature associated with physiological requirements of seagrass (Dennison, 1993; Table 3 - 12) and associated light attenuation by various factors such as plankton (chlorophyll *a*), total suspended solids, macroalgae (Kennish et al., 2011,

Table 3 - 13), and epiphytic cover (Brush and Nixon, 2002; Figure 3 - 31, Figure 3 - 36), 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).

Dennison et al. (1993) report information from several studies that document the maximal depth limit for eelgrass (in meters) as ranging form 3.7 to 10.1 m (in Kattegat, Denmark), 2.0 to 5.0 m (in Roskilde, Denmark), 1.5 to 9.0 m (in Denmark), 6.0 m (in Woods Hole, USA), 2.5 m (in the Netherlands), and 2.0 to 5.0 m (in Japan). Importantly, Dennison et al. (1993) also report the minimal light requirements for eelgrass (*Zostera marina*) as a percent light at the maximal depth limit using $100 * I_z/I_o = e^{-Kd*Z}$, where I_z is the irradiance at depth z, I_o is the irradiance at the surface, and K_d is the light attenuation coefficient. The minimal light requirements for *Z. marina* at maximal depth are generally close to 20% of the surface irradiation and is documented at 20.1 ± 2.1 %, 19.4 ± 1.3 %, 20.6 ± 13.0 %, 18.6 %, 29.4 %, and 18.2 ± 4.5 % (Dennison et al., 1993; Table 3 - 12).

Burkholder et al. (2001, 2007) documented that light reductions decreased shoot production, and that at 30% reduction of surface light (i.e. PLL of 70%) and at 70% reduction of surface light (i.e. PLL of 30%) shoot production was inhibited even more under high nitrogen concentrations (i.e. eutrophic conditions).

Additional analysis on available data indicates that seagrass indicators responded negatively to increases in chlorophyll *a* (Figure 3 - 37) and total suspended solids (Figure 3 - 38). Seagrass biomass (both aboveground and belowground) and percent cover decreased with increasing chlorophyll *a* (Figure 3 - 37). There was perhaps a slight increase in blade length and a substantial increase in shoot density with increasing chlorophyll *a*, but chlorophyll *a* is unlikely to be a direct causal factor in this case, though shoot density may be increasing as a coping mechanism to the overall eutrophic condition (Fertig et al. 2013). All seagrass indicators except for shoot density declined substantially with increasing levels of total suspended solids (Figure 3 - 38). Shoot density declined exponentially with increasing epiphyte to seagrass biomass ratio, with changes in the rates of the decline at epiphyte biomass to seagrass biomass ratios of 0.5, 1.0, and maximum values were observed slightly above 1.5 (Figure 3 - 39). Seagrass percent cover in the central segment had maximal values at a value of ~0.5 for the epiphyte to seagrass biomass ratio (Figure 3 - 39). Blade length and seagrass biomass (both aboveground and belowground) in the central segment had maximal values at an epiphyte to seagrass biomass ratio of ~0.5 (Figure 3 - 39). Macroalgae percent cover of 7.5% was an inflection point for seagrass biomass (both aboveground and belowground), as was 5% and 12%, and similar values of macroalgae percent cover were inflection points for seagrass blade length response and seagrass percent cover and shoot density (Figure 3 - 40). Seagrass biomass (both aboveground and belowground) decreased linearly with Secchi depth to ~5 ft, but then plateau with greater depths (Figure 3 - 41).

Summertime chlorophyll *a* in Maryland's Coastal Bays has historically and recently been measured at > 40 μg L^{-1} (Boynton et al., 1996; Fertig et al., 2013), which is ~ 5 times higher concentration than the < 8 μg L^{-1} observed in BB-LEH since 2004 (Fertig et al. 2013), and areal coverage of seagrass is roughly twice as large in Chincoteague Bay (Orth et al., 2006) than it is in BB-LEH (Lathrop et al., 2001). Based on the above observations and analyses, thresholds for chlorophyll *a* concentrations, total suspended solids, macroalgae percent cover, epiphyte to seagrass ratio, Secchi depth, and percent light reaching seagrass leaves were defined. Defined thresholds for Light Availability indicators are listed in Table 3 - 14. The rescaling equations that are generated from these thresholds are listed in Table 3 - 4.

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. However, declines had begun prior to monitoring and so assessments were adjusted based on literature values of seagrass biomass within the time period of this project (Figure 3 - 42) though there remains some uncertainty associated with identifying 'reference' conditions of seagrass in BB-LEH.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 nutrient loading (Latimer and Rego, 2010; Figure 3 - 18, Figure 3 - 20), to water quality indicators (Figure 3 - 23, Figure 3 - 31, Figure 3 - 32, Figure 3 - 33) and to light availability indicators (Figure 3 - 37, Figure 3 - 38, Figure 3 - 39, Figure 3 - 40, Figure 3 - 41).

Defined thresholds for seagrass indicators are listed in Table 3 - 15. The rescaling equations that are generated from these thresholds are listed in Table 3 - 4.

Biotic Response: Harmful Algal Blooms

An index of harmful algal blooms has previously been developed for the brown tide alga *Aureococcus anophagefferens* and is available in the literature (Gastrich and Wazniak, 2002; Figure 3 - 43). This index was developed for coastal lagoon ecosystems, and thus thresholds from this index were utilized directly for the Index of Eutrophication to derive the rescaling equation. While Gastrich and Waziak use three thresholds in their index, additional intermediate thresholds along the linear function are used for this project for consistency with other indicators.

Gastrich and Wazniak (2002) defined thresholds for concentrations (cells mL^{-1}) of *A. anophagefferens* cells based on impacts of various concentrations to shellfish, including commercially or recreationally important species (see Table 3 - 16), which was in turn based on data and information available in the literature. Below 35,000 cells mL^{-1} of *A. anophagefferens*, there are no known impacts on shellfish (Bricelj et al., 2001;

Schaffner, 1999). Between 35,000 and 200,000 cells mL^{-1}, toxins from brown tide inhibit feeding rates of hard clams, reduce growth of mussels and bay scallops, and can cause high mortalities of bay scallop larvae (Bricelj et al., 2001; Bricelj, 1999; Schaffner, 1999; Bricelj and Lonsdale, 1997; Gallagher et al., 1989). Above 200,000 cells mL^{-1} of *A. anophagefferens,* water becomes discolored, bivalves may experience sub-lethal yet adverse effects, and furthermore mussels and hard clams decrease their feeding and growth rates (Gastrich and Wazniak, 2002; Bricelj and Lonsdale, 1997; Bricelj and Kuenster, 1989; Tracey, 1988; Bricelj, 1999; Bricelj et al., 2001). Furthermore, above 200,000 cells mL^{-1} of *A. anophagefferens,* bay scallops have been observed to have recruitment failures and mortalities in addition to reduced growth (Cosper et al., 1997; Bricelj and Lonsdale, 1997; Gallagher et al., 1989; Bricelj, 1987).

The thresholds for Harmful Algal Blooms are not intended to be a toxicity index (e.g. they are not based upon an identified toxin and a concentration-response) although they assume some level of toxicity to various organisms. Note that these thresholds do not predict impacts of specific concentrations of *A. anophagefferens* concentration in natural populations but do provide information on potential impacts (Table 3 - 16). It is assumed that the increased concentrations and/or increased duration of blooms may potentially cause more severe impacts.

Because of direct potential for health risks and impacts on shellfish, a precautionary approach is most appropriate for the application of these thresholds. Therefore, the maximum concentrations observed in each segment each year should be used for summarization when applying these thresholds.

According to Gastrich and Wazniak (2002), the thresholds for the Harmful Algal Blooms assume that appropriate methods are used to collect water samples and enumerate *Aureococcus anophagefferens* (Anderson et al., 1989, 1993; Caron 2001). Ideally, sampling for the brown tide algae in BB-LEH is done within each estuarine segment (north, central, south) during each year at sufficient spatial coverage. As noted above, while some data collected during the study time period are available in the literature, often the locations of sampling were not, which limits the ability to hindcast.

Note that since the Harmful Algal Blooms only receive Raw Scores, the scores for these indicators range from 0 to 100. This is because there is only one indicator and thus PCA is not meaningful and weightings are thus not calculated. Maximum and minimum harmful algal bloom concentrations for rescaling are listed in Table 3 - 4.

Defined thresholds for Harmful Algal Blooms are listed in

Table 3 - 17The rescaling equation that is generated from these thresholds is listed in Table 3 - 4.

Biotic Response: Benthic Invertebrates

Thresholds for this component of the Index of Eutrophication are considered with respect to the REMAP assessment. They were 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. For example, effects of various dissolved oxygen concentrations on benthic invertebrate communities have been studied previously and are reported by Ritter and Montagna (Table 3 - 18). 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 indices, such as the Chesapeake Bay Benthic Index of Biotic Integrity rely on community composition or measures of species diversity (e.g., Shannon-Weiner H or Gleason's D diversity indices, Table 3 - 19) and assemble lists of species that are 'pollution indicative' or 'pollution sensitive' (Weisbert et al. 1997). Many species, however, are on both such lists, limiting the ability to assess ecosystem condition.

For this project we base the assessment of benthic integrity on the EMAP sampling and index. Since only one year of data is available, there is insufficient data for validation and this dataset must suffice for the assessment of benthic invertebrates. For the EMAP index scores, a score above 0 indicated non-degraded condition and a score below 0 indicated degraded condition. To rescale this index to a similar range for comparison to the other data types used in this project, we rescale the EMAP scores based on their data distribution. There was insufficient evidence or justification for other scaling methods. Analysis of the EMAP index data variability indicated that the majority of values ranged from -2 to +2 and so equal intervals were constructed for rescaling purposes.

Defined thresholds for Benthic Invertebrates are listed in Table 3 - 20. The rescaling equations that are generated from these thresholds are listed in Table 3 - 4.

METHODS: STEP-BY-STEP CALCULATIONS

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.

Raw Scores are determined by comparing each observation of each indicator to

'thresholds' for each indicator. 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%).

Raw Scores are calculated for all datasets as follows, as documented in the SAS code used for calculating the Index of Eutrophication (see Appendix 3-5). Instances of missing data are excluded. Ecosystem Pressures data are sorted by Year, Season, and Segment and then rescaling equations are applied to USGS modeled output for the entire calendar year (rather than just the growing season, which was also provided by USGS) for both total nitrogen loading and total phosphorus loading. Rescaling equations are applied to each observation of Water Quality indicator (temperature, dissolved oxygen, total nitrogen concentration, total phosphorus concentration) during April to October (inclusive). These months are selected due to the importance of potential impacts on biological and human-use activities. Rescaling equations are applied to each observation of the six Light Availability indicators after excluding observations of each indicator where data was missing. Rescaling equations are applied to each observation of the five Seagrass indicators and the single HAB indicator.

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). Descriptive, summery statistics of Raw Scores for each dataset are calculated for each segment during each year and stored as separate files, as documented in the SAS code included in Appendix 3-5. These include means, medians, standard deviations, minimums, and maximums of each indicator's Raw Score. Where data are unavailable in a given segment during a given year, this is recorded as 'No Data'. For example, the dataset bbindex.b1wqvar_scores_mean_yr_seg contains mean, median, standard deviation, minimum and maximum temperature Raw scores for each year and segment.

Weighted Scores weight the raw scores by their variability. Principal component analysis (PCA) is conducted on the lightly summarized raw scores to calculate a weighting for each indicator within each component based upon their eigenvectors. In other words, summarized data from all available years across the entire estuary (or as many segments as available) are used for PCA analysis to determine weightings. Up to three data points per year are thus plotted, and multiple years of data are required for this analysis to determine weightings for each indicator. A single weighting for each indicator is applied to data from each segment. Calculating unique weightings for each segment would be statistically inappropriate and would invalidate comparisons across segments (Sokal and Rohlf, 1981; Quinn and Keough, 2002; Underwood, 1997). Scree plots are examined to identify the cumulative explanatory power of each principal component. Generally, the first principal component explains ~50-75% of the variability, and the first

two principal component axes explain ~80-90% of the variability. Note that PCA is not conducted for the Ecosystem Pressures because only a single number is provided for each segment in each year from the modeled nutrient loading provided by USGS (see Appendix 1). Therefore there is no variability and PCA cannot be conducted. PCA cannot be applied to the HAB component because there is only one indicator.

PCA was conducted using the covariance option on Raw Scores summarized by Year and Segment (see Appendix 3-5). The covariance option computes the principal components from the covariance matrix rather than the correlation matrix (the default setting in SAS). Using the covariance matrix causes variables with large variances to be more strongly associated with components with large eigenvalues and causes variables with small variances to be more strongly associated with components with small eigenvalues. Therefore, the covariance option should not be specified unless the units in which the variables measured are comparable or the variables are standardized in some way. As indicated above, variable (i.e., Raw Score) units are comparable as they were standardized via the rescaling equations to result in homogeneity of variance, which can be tested using the Univariate procedure in SAS.

Lightly summarized data (mean or median for each segment in each year) were used for PCA analysis (see Appendix 3-5). PCA on the covariance matrix was conducted on the median Raw Scores for temperature, dissolved oxygen, total nitrogen and total phosphorus, but this was done separately for 1989–1998 and 1999–2010 because total phosphorus data was unavailable during the first set of years. To test the effect of total phosphorus on the overall Water Quality, PCA on the covariance matrix was similarly conducted on the second set of years, but omitting the median Raw Scores for total phosphorus (see Validation below). Note that Raw Scores for Water Quality indicators are calculated on observations during April–October, inclusive. For the Light Availability indicators, PCA on the covariance matrix was conducted on median chlorophyll *a* Raw Scores, median TSS Raw Scores, average Secchi depth Raw Scores, average epiphyte to seagrass biomass ratio Raw Scores, and average percent light reaching seagrass leaves Raw Scores. PCA on the covariance matrix was conducted on median Raw Scores of Seagrass shoot density and mean Raw Scores for the other four Seagrass indicators.

The weighting is calculated as the square of the eigenvector of the first principal component 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.

The purpose of adding the Raw Score and the Weighted Score to arrive at the Final Score for an indicator and each component index (e.g. Water Quality Index, Light

Availability Index, Seagrass Response Index) is to assess both the condition and consistency of each indicator and each index.

Note the important difference between the *weighting* and the *Weighted Score*. The *weighting* is the square of the eigenvector and represents the variability of the factor if data are available in a given segment in a given year. The *Weighted Score* is the Raw Score multiplied by the weighting and thus represents the *consistency of the condition for that indicator*. Weighted scores provide a measure of the consistency of the observations with respect to thresholds for the appropriate indicator.

Consistency is important to include in an Index of Eutrophication because it highlights times and places when and where conditions of each indicator are changing (either positively or negatively) so that these indicators can be targeted for attention (e.g. for monitoring, management, or research).

The implications for including both the condition and the consistency of eutrophciation are that this tool can help prioritize decisions regarding limited resources available for various actions. For example, if an indicator is in flux, it may be worthy of more intense monitoring, research, or remediation action. If that same indicator consistently exhibited an extreme condition (e.g. 'Excellent' or 'Highly Degraded'), discussions regarding prioritization of resources may be efficiently directed towards another indicator.

Indices for each of components with sufficient data are then averaged together for the sets of years when data are available to calculate the overall Index of Eutrophication. While ideally each index would be used as input for another PCA to calculate a weighting for each index, there was an insufficient quantity of data to do so, and equal weighting (i.e. averaging) was considered justified as an alternative. 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.

VALIDATION: 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

principal 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 principal 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) the availability of individual indicators for any given year or segment.

This is particularly important because principal 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 do 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 this, 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 - 13). 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 - 14).

Weighted scores for each water quality indicator under Scenario 1 and Scenario 2 are comparable for each year and segment (Figure 3 - 44). 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 - 45). 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 timeframe and area needed across all components had to be determined. The common timeframe 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 principal 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 principal component analysis indicated that it did not co-vary with total nitrogen (Figure 3 - 21), 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. Principal component analysis was conducted on all years of data for scenarios that excluded and included macroalgae percent cover (Figure 3 - 46). 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 principal component analysis to identify potential co-variation between indicators (Figure 3 - 47).

RESULTS: INDICATOR SCORES

Indicator scores for Watershed Pressures were fairly consistent over time and between indicators relative to each segment (Figure 3 - 48). 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 - 49). 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 - 50). 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 sorth 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 - 51). 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 - 52). 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).

RESULTS: RAW SCORES FOR COMPONENT INDICES

Watershed Pressure Indicator scores were averaged to arrive at the Watershed Pressure Index (Figure 3 - 53). 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 - 54). 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 - 55).

Raw Scores for Seagrass Response were virtually the same in the central and south segments (Figure 3 - 56). 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 - 52). 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).

RESULTS: 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 - 15. 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-–999 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.

RESULTS: 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 - 48).

Weighted scores for the Water Quality component were very similar between segments (Figure 3 - 54). 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 - 55). 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 - 56). 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 - 52). 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).

RESULTS: 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 - 15. 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 - 53). 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 - 54).

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 - 55). 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 - 56).

The Harmful Algae Bloom Index is shown as discrete dots due to the limited data that are available (Figure 3 - 52). 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 - 57). 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 index calculations. Validation results of the data for each of the datasets are provided in Component 4 of this report.

DISCUSSION: 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 for many years. This frequency of data collection is not sufficient to capture natural daily fluctuations due to processes such as photosynthesis and respiration. Further, this data collection frequency introduces 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 verified 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, specifically, when chlorophyll *a* levels are elevated as measured by aerial overflights. This method however, is inappropriate for monitoring for the brown tide species *Aureococcus anophagefferens*, as is clearly demonstrated and documented in the literature (Anderson et al. 1989, 1993). Further, light microscopy methods are unable to detect this species. Monoclonal antibodies are required to positively identify the brown tide species.

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 Index of Eutrophication or its component indices.

DISCUSSION OF INDEX OF EUTROPHICATION

Despite the limitations of the data and scope of this project, the 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 - 58). 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 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 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 segment 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.

SUMMARY AND CONCLUSIONS

- The 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: Moderate→Poor and Poor→Highly 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.
- 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.
- Five of the seven years of available data for Harmful Algal Blooms result in Highly Degraded scores for this indicator.

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 1-8). 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 (Baker and Kennish, 2010; Appendix I-1). 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 1-8) rather than at 120 stations along 12 transects (central and south segments only) as in previous survey years (Figure 1-9). 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 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.

To accomplish the objectives of the project, an in situ survey was conducted in a separate study on key demographic characteristics of mixed seagrass beds (Ruppia maritima and Zostera marina) in the north segment of the estuary during the June-November sampling period in 2011 (Kennish, 2011b; Kennish et al., 2013). A survey of seagrass beds in the central and south segments of the estuary was also conducted during the same sampling period as part of the NEIWPCC project, providing concurrent and complete coverage of seagrass habitat in the three segments of the estuary for 2011. Primary biotic data collected in the central and south segments included the presence/absence, aboveground and belowground biomass, shoot density, areal cover, and blade length (for eelgrass only) of seagrass. In addition to the percent epiphytic growth on seagrass, the presence of bay scallops and other shellfish was also recorded in

the seagrass beds. The presence/absence and percent cover of macroalgae were also measured at each sampling station.

State-of-the-art seagrass sampling was conducted using the protocols of the SeagrassNet approach (Short et al., 2002) that were applied by Kennish et al. (2007, 2008, 2010, 2012) in prior annual seagrass surveys conducted in the estuary from 2004 to 2010 (excluding 2007), with the data utilized in this project report. These sampling protocols were employed in this project to maintain consistency for data integration with the previous seagrass surveys. Therefore, data comparability has been maintained throughout the project.

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

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

Physicochemical data (temperature, salinity, pH, dissolved oxygen, and depth) were also collected at each sampling station using either a handheld YSI 600 XL datasonde coupled with a handheld YSI 650 MDS display unit, an automated YSI 6600 unit, or a YSI 600 XLM automated datalogger. Secchi disk measurements were likewise collected in the survey area. Water quality data (other than Secchi measurements) were collected at a uniform depth (~10 cm) above the sediment-water interface using YSI datasondes. More than 1000 physicochemical and biotic measurements were compiled and analyzed in the project (see Kennish et al., 2013). Details of the protocols for field sampling, laboratory processing of samples, and data analysis can be found in the Quality Assurance Project Plan for both surveys (Baker and Kennish, 2010; Kennish, 2011b).

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).

VALIDATION AGAINST THE NEEA ASSESSMENT

The National Estuarine Eutrophication Assessment (NEEA) previously analyzed the condition of Barnegat Bay-Little Egg Harbor (Bricker et al. 1999, 2007). Methods for the NEEA approach are described in the section of Component 3 'Building on the National Estuarine Eutrophication Assessment'. Here, we compare our results from 2007 to findings from the NEEA report as a validation of the Index of Eutrophication that we developed in this study.

The 2007 NEEA report documents that Barnegat Bay-Little Egg Harbor had 'High Overall Eutrophic Condition' (Figure 4-1, from Bricker et al. 2007). This conclusion was reached because both primary symptoms (chlorophyll *a* and macroalgae) had high expression levels of eutrophication, and the highest secondary symptom (harmful algal blooms) also had high expression levels of eutrophication. These symptoms of eutrophication are shown visually in a conceptual diagram (Figure 4-2, from Bricker et al. 2007).

Our findings from 2007 show that the Index of Eutrophication score in the North segment was 41, in the Central segment was 43, and in the South segment was 52 (Figure

3-39). Thus, the overall eutrophication status in BB-LEH was considered 'Moderate' in 2007 for each of these three regions.

The condition of BB-LEH deteriorated over time in the Central and South segments and remained relatively constant in the North segment. This project reports a 1999 Index of Eutrophication score of 42 in the North segment, 65 in the Central segment, and 57 in the South segment (Figure 3-39). These values of the Index of Eutrophication are 'Moderate' for the North and South segments. The 1997 value of the Index of Eutrophication in the Central segment is 'Good'. The numerical difference over time is important, however.

CONCLUSIONS

The degraded condition of *Z. marina* in the BB-LEH Estuary has continued through 2011, validating the progressive system 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 $\leq$ 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 could pose a threat to human uses of estuarine resources. It 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 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 of BB-LEH. 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.

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 that inhabit the estuary (JCPL, 1978; Kennish et al., 1984; Ecological Analysts, 1986; Kennish, 2001d). 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.

Zampella et al. (2006) used biotic and environmental indicators to assess the ecological integrity of a coastal plain stream in the New Jersey Pinelands. They demonstrated that key indicators varied in relation to the percentage of altered land (developed land and upland agriculture) within the associated watersheds.

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. Impervious land cover is an important and quantifiable land use indicator of adverse impacts of pollution runoff (Arnold and Gibbons, 1996). 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, New Brunswick, New Jersey).

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 (BBP, 2011).

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.

EUTROPHICATION

Eutrophication (defined as the process of nutrient enrichment and increase in the rate of organic matter input in a waterbody leading to an array of cascading changes in ecosystem structure and function such as decreased dissolved oxygen levels, increased microalgal and macroalgal abundance, occurrence of harmful algal blooms (HABs), loss of seagrass habitat, reduced biodiversity, declining fisheries, imbalanced food webs, altered biogeochemical cycling, and diminished ecosystem services; de Jonge and Elliott, 2001; Kennish and de Jonge, 2011) is responsible for insidious degradation of estuarine systems worldwide (Nixon, 1995; Boesch et al., 2001; Burkholder et al., 2007). 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; 2011). 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 eutrophication problems have escalated in the BB-LEH Estuary over the past two decades, manifested by frequent phytoplankton and macroalgal blooms, epiphytic loading, diminishing seagrass biomass, , and other effects. Recurring phytoplankton blooms have been documented, including nuisance and toxic 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 threat to seagrass beds and other phanerogams that serve as vital benthic habitat for recreationally and commercially

important species (e.g., blue crabs, *Callinectes sapidus*; bay scallops, *A. irradians*; and tautog, *Tautoga onitus*), and many other organisms. 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 raising concerns of people living along the adjacent bayshore area, as well as government and health officials. These events demonstrate how serious macroalgal blooms can be in this coastal lagoon.

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 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 are 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, benthic microalgae, epiphytes, and phytoplankton. These changes when left unabated severely degrade habitat quality and can result in diminished production of fish and shellfish (Nixon et al., 2001; Hughes et al., 2002). 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 widespread 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 epiphytic 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; Anderson et al., 2010). 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 (Livingston, 2000, 2003, 2006). 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; Bricelj et al., 2012).

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 increases excessively, 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 as documented by low dissolved oxygen measurements. 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 as grab samples 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. 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).

In a coastal lagoon like BB-LEH, dissolved oxygen must be monitored frequently in multiple locations for accurate assessment due to large variations in this parameter over the course of a day driven by natural processes, such as changes in temperature or light, as well as community photosynthesis and respiration. Robert W. Howarth (Cornell University, personal communication) noted that "DO is often measured once a month, with no consideration of time of day for sampling; this may work for bottom waters in a highly stratified estuary, but is meaningless in a shallow lagoon where DO may oscillate from say 20% of saturation every dawn to 200% of saturation every day at dusk."

Taking one grab sample at multiple locations once a quarter, once a month, once a week, or even once a day, will not suffice – it will not provide statistically accurate or valid measurements of DO (Sokal and Rohlf 1981, Quinn and Keough 2002, Underwood 1997) in an estuarine lagoon like Barnegat Bay-Little Egg Harbor, which clearly is not stratified (Kennish et al. 2001). The variation is just too great. Previous DO samples collected by the NJDEP have been primarily obtained by collecting water samples during daylight hours in the morning and early afternoon when sampling bias will enter into the process, shifting the data results toward higher DO levels. It is not possible to correct this without also collecting and factoring in DO measurements taken during night hours, specifically from 1-5 a.m. This has not been done previously, and so the prior database on DO in this coastal lagoon is deficient. What is recommended in the future is adding moored datalogger instrumentation at more strategic locations around the estuary. If grab samples are continued, then it will be necessary to collect at least 3 grab samples at each sampling station per day (including one between 1-5 a.m.), and the sampling frequency must be increased to daily or perhaps every other day for several years' time to obtain trends. By collecting 3 grab samples per day, accurate modeling can be conducted. That would give the most accurate picture of DO levels in the estuary.

Regulatory protection and conservation of New Jersey's estuarine waters are based on DO measurements. Ideally, DO should be monitored continuously (via automated dataloggers for example) at multiple locations for accurate assessment. It is important that assessments of ecological health of BB-LEH 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.

OTHER MANAGEMENT CONCERNS

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 at times posed a hazard to human use of some areas in the north segment of the estuary. These impacted 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.

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 hardened shorelines and eutrophication have likely contributed to the problem. Currently, ~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 (>500,000 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 (Condon et al., 2001; Decker et al., 2007).

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 waterbody, as well as hardened shorelines that provide overwintering habitat. Water quality alteration must also be minimized by improving pollution controls in the watershed source. There also needs to be more administrative/management assessment of this problem.

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. This organism may also pose a threat to the structure and function of the estuarine food web because it crops substantial amounts of zooplankton which serve as a food source for many finfish and other fauna.

Shellfish Resource

Bricelj et al. (2012) have examined the status and trends of hard clam (*Mercenaria mercenaria*) populations in BB-LEH, reporting declines in both absolute abundance (documented for Little Egg Harbor), and harvest statistics (landings) over time. Hard clam 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). 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 estuary.

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 in Little Egg Harbor has been in a state of precipitous decline for years (Bricelj et al., 2012).. The loss of such large numbers of hard clams may cause 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. A shift in microalgal quality in the estuary (i.e., phytoplankton size structure and species composition; picoplankton occurrence) could be a factor in the decrease of hard clam abundance observed in the estuary (Bricelj et al., 2012). .

MANAGEMENT APPLICATIONS

A holistic management approach must be accelerated to remediate environmental problems in BB-LEH associated with nutrient enrichment due to ongoing development and land use-land cover changes in the watershed. Multiple corrective strategies should be applied concurrently, such as improved stormwater control systems (e.g., currently stormwater basin upgrades are targeting 10 of ~2700 stormwater basins), implementation of best management practices in the watershed, open space preservation, fertilizer controls, soil restoration, 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. This is a long-term approach to remediate the eutrophication problems in the estuary.

A total maximum daily load (TMDL) for nitrogen and phosphorus is also a necessary element to effectively mitigate the eutrophic condition of the estuary. Application of a TMDL should be pursued concomitantly with the other management approaches noted above. It is necessary to respond aggressively at this time to nutrient loading from the watershed because of the severity of the eutrophication problems in the estuary, which may become intractable if they are not remediated in the short term.

Results of the Index of Eutrophication applied in this study indicate that eutrophication of the estuary is greatly worsened by increasing total nitrogen loading and total phosphorus loading. 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 of the estuary, eutrophication condition reaches a new, lower steady state of poor condition. We therefore recommend a strict limit on nitrogen and phosphorus loads to 1500 kg TN km^{-2} yr^{-1} and 75 kg TP km^{-2} yr^{-1}as a starting point of control to remediate eutrophication of the estuary.

Reducing the fraction of urban area that is covered by turf will likely reduce the loads of nitrogen and phosphorus to BB-LEH. This is because concentrations of total nitrogen are substantially higher for developed turf areas than for developed non-turf areas, which in turn, are higher than those for undeveloped areas. Concentrations of total phosphorus also are higher for developed turf areas than for developed non-turf and undeveloped areas.

Better management of turf areas—for example, reducing the amounts of nitrogen- and phosphorus-containing substances applied to turf areas—will likely reduce overall loads of nitrogen and phosphorus. Reducing the volume of stormwater directly discharged to streams will also reduce the runoff component of nitrogen and phosphorus loads, and will likely reduce the total loads to BB-LEH.

Much of the land in the southern portion of the watershed is protected from intense development. Based on previous investigations in the watershed and the analysis of existing data as part of this study, future increases in development in the central and south segments will likely lead to higher concentrations and loads of nutrients in the streams located in those areas, thereby increasing nutrient inputs to the estuary.

Runoff accounts for a greater percentage of flow in the highly developed basins, and a smaller percentage of flow in the less developed basins. The total amounts of runoff and the runoff contribution of nitrogen and phosphorus loads will likely increase with additional urban development. The baseflow contributions of nitrogen loads, which are generally greater than the runoff contributions, also are strongly associated with urban land and will likely increase with increasing urban development.

A more complete understanding of nutrient cycling in the watershed could be achieved with the use of additional, targeted water-quality monitoring in conjunction with a watershed water-quality model that considers in-stream processes, shorter time steps, and that targets individual streams and reaches.

CONCLUSIONS AND RECOMMENDATIONS

BB-LEH is an estuary that has undergone significant ecological decline, as evidenced by the increasing eutrophication of the central and south segments since the 1990s ($P < 0.05$) and an even worse eutrophication condition documented for the north segment. Collectively, the direct relationship between nutrient loading from the watershed and estuarine nutrient concentrations, the degradation of an array of biotic indicators, and the relationship between nutrient loading and the Index of Eutrophication supports the conclusion that BB-LEH is a highly impacted estuarine system.

Total nitrogen loading and total phosphorus loading have caused substantial degradation and eutrophication of BB-LEH. The condition of the estuary has progressively worsened over time for both nitrogen and phosphorus (Figures 3-31, 3-36, and 3-39) resulting in an array of bottom-up impacts evident in nuisance and toxic algal blooms, declining eelgrass beds, and other parameters of change. The rate of decline of eelgrass is related to nutrient loading and associated symptoms of eutrophication. Overall, eutrophication is greatly worsened by increasing total nitrogen loading and total phosphorus loading. 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 of the estuary, eutrophication condition reaches a new, lower steady state of poor condition.

Overall, water quality condition has been declining throughout the estuary since the early 1990s. Total nitrogen loading and total phosphorus loading scores in index calculations were lower (more degraded) during 2003-2010 than in previous years, indicating a worsening condition. Loading for both nutrients is greatest in the north segment where environmental condition is most impacted. While nutrient loading has been linked to increasing eutrophication of the estuary, specific levels of total nitrogen

loading and total phosphorus loading as tipping points for ecosystem decline have not been determined.

While no nutrient criteria have been established for the BB-LEH Estuary, one remedial 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. 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.

We recommend a two-pronged management approach to address the eutrophication problems in BB-LEH. First, a TMDL for nitrogen and phosphorus should be established for the system, limiting total nitrogen and phosphorus loads to 1500 kg TN km^{-2} yr^{-1} and 75 kg TP km^{-2} yr^{-1}.

In addition, an array of other managment strategies must be aggressively applied concomitantly with a TMDL. These include measures that improve stormwater control systems, best management practices in the watershed, open space preservation, fertilizer controls, soil restoration, and support 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 long-term ecological condition and resources of the estuary.

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FIGURES

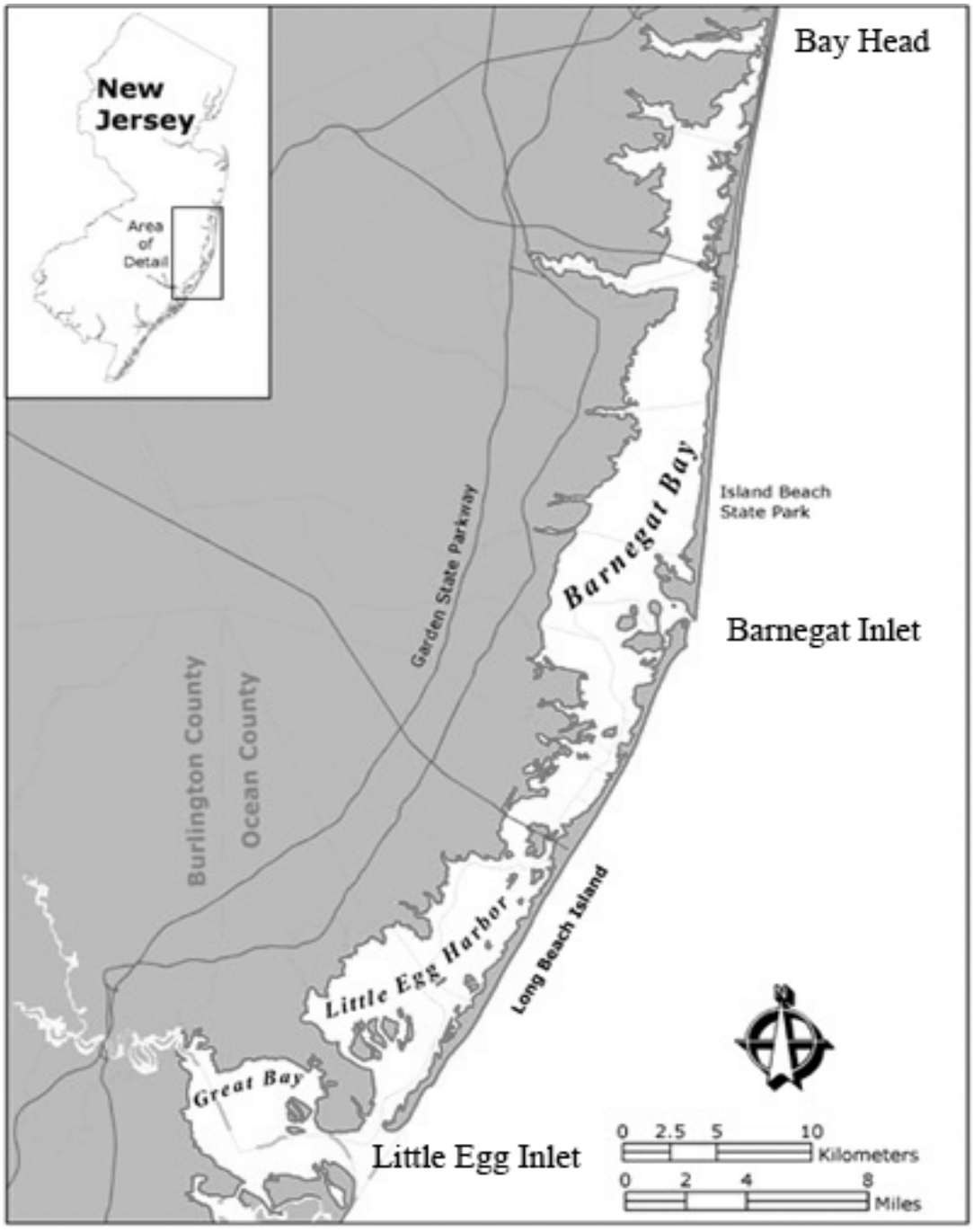


Figure 1 - 1 Map of the Barnegat Bay-Little Egg Harbor (BB-LEH) Estuary. Inset shows the location of the estuary with respect to the state of New Jersey.

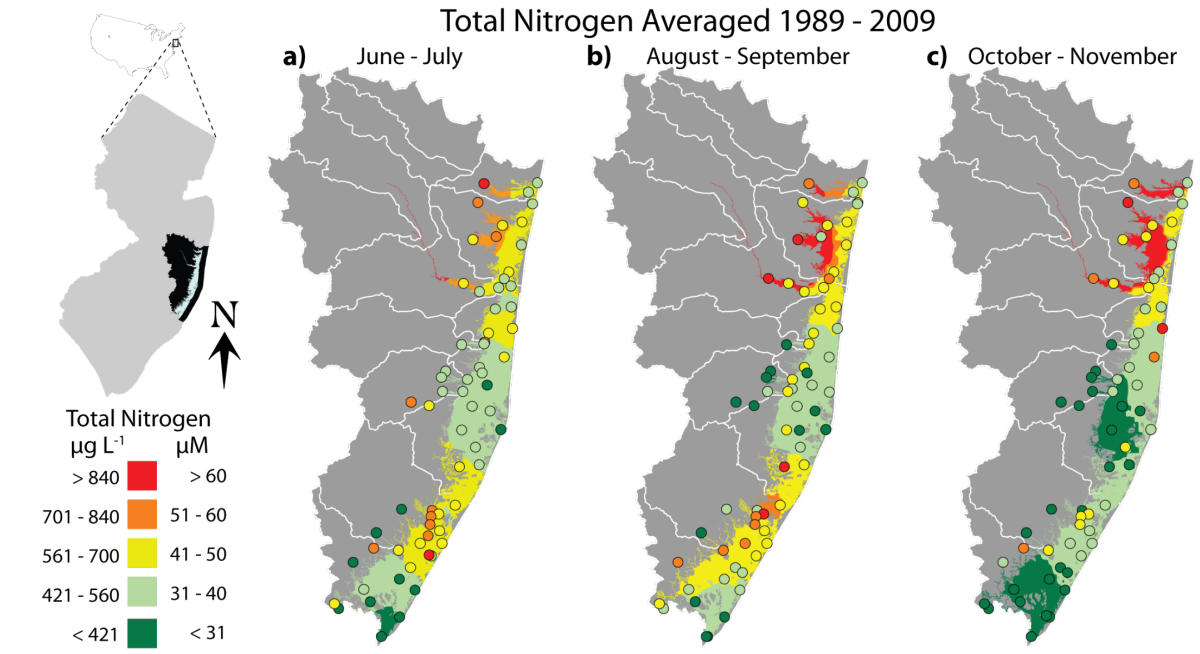


Figure 1 - 2 Mean total nitrogen concentrations in the BB-LEH Estuary from 1989-2009.

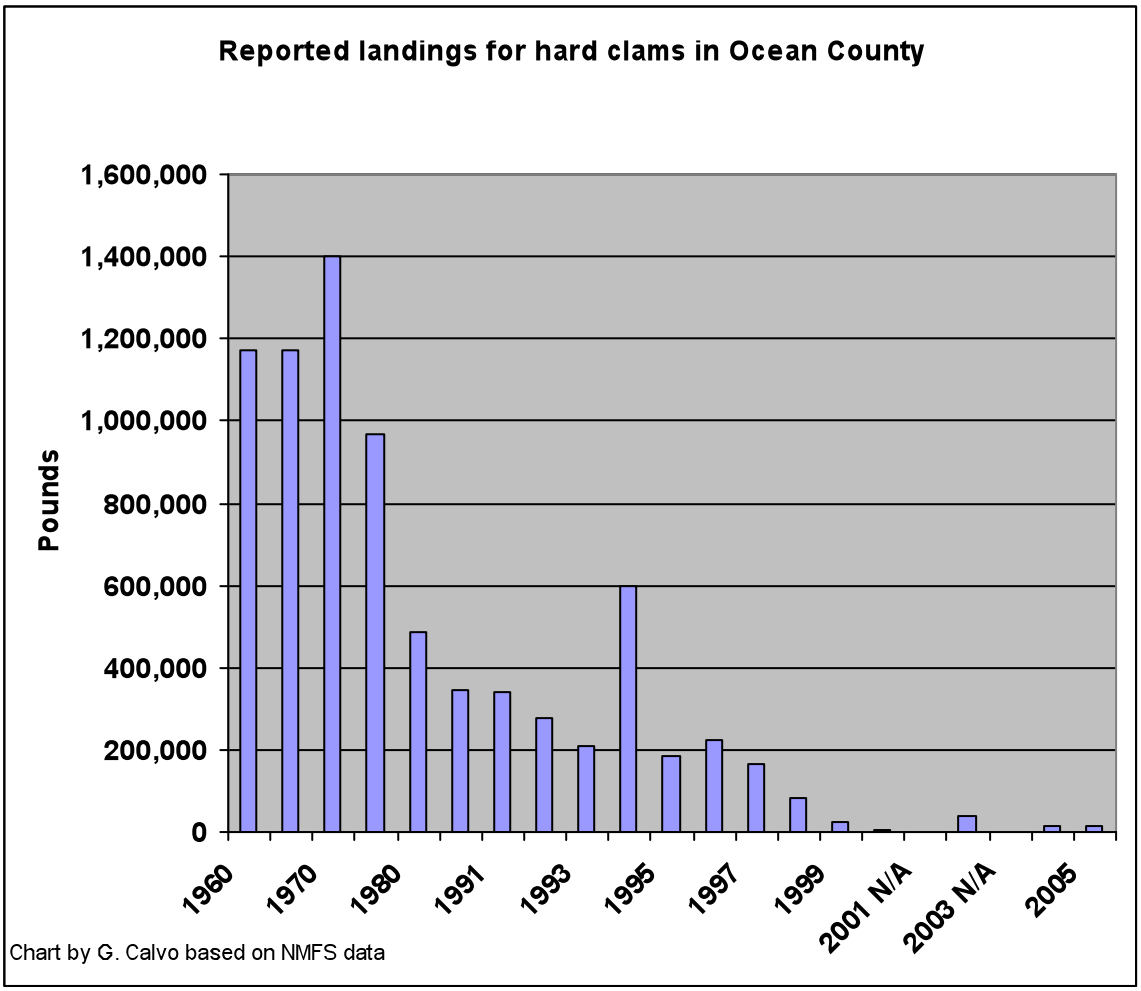


Figure 1 - 3 Hard clam landings for Ocean County showing acute decline from 1960 to 2005. Data from the National Marine Fisheries Service.

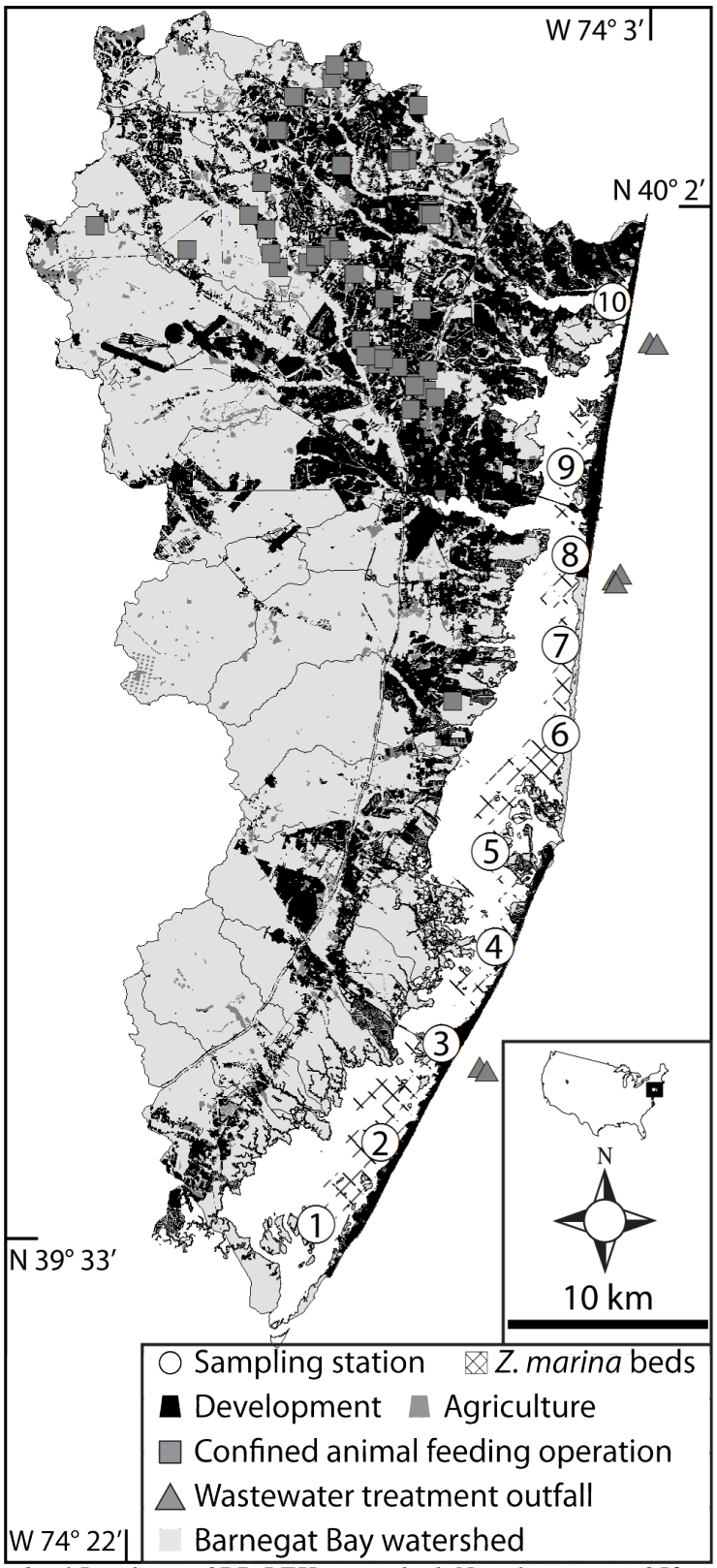


Figure 1 - 4 Land use of BB-LEH watershed. Note locations of 52 confined animal feeding operations.

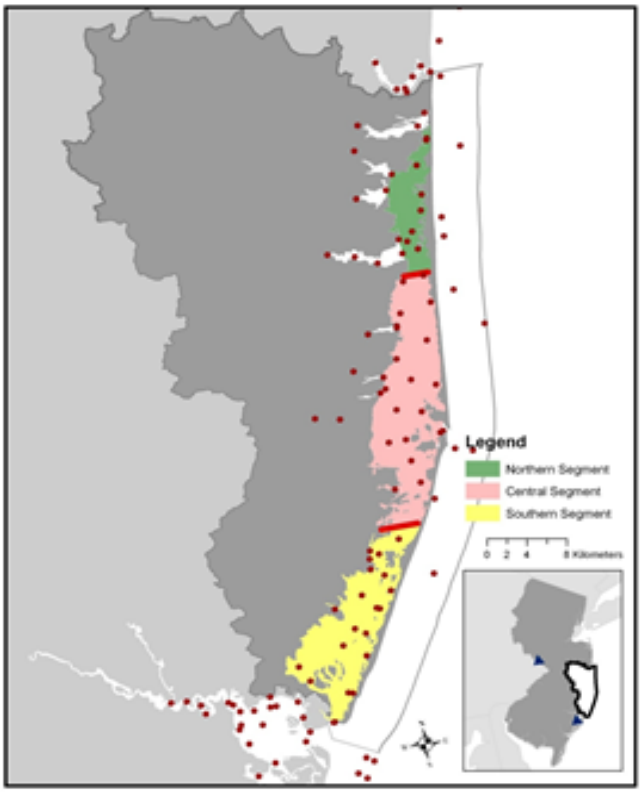


Figure 1 - 5 Map of the BB-LEH Estuary showing three segments (north, central, and south) used for index development.

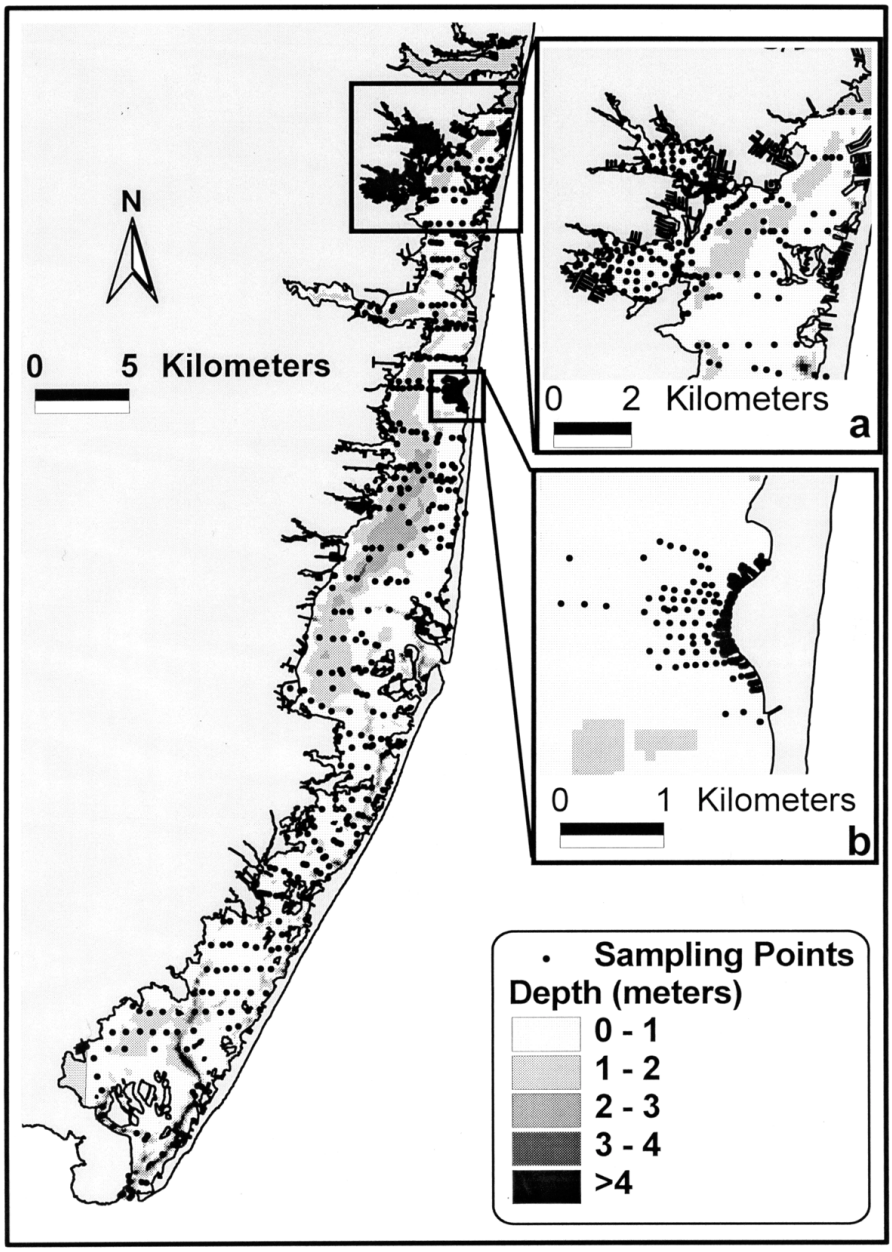


Figure 1 - 6 Map showing a grid of bottom sediment sampling stations and bathymetric measurements in the BB-LEH Estuary. (From Psuty, 2004).

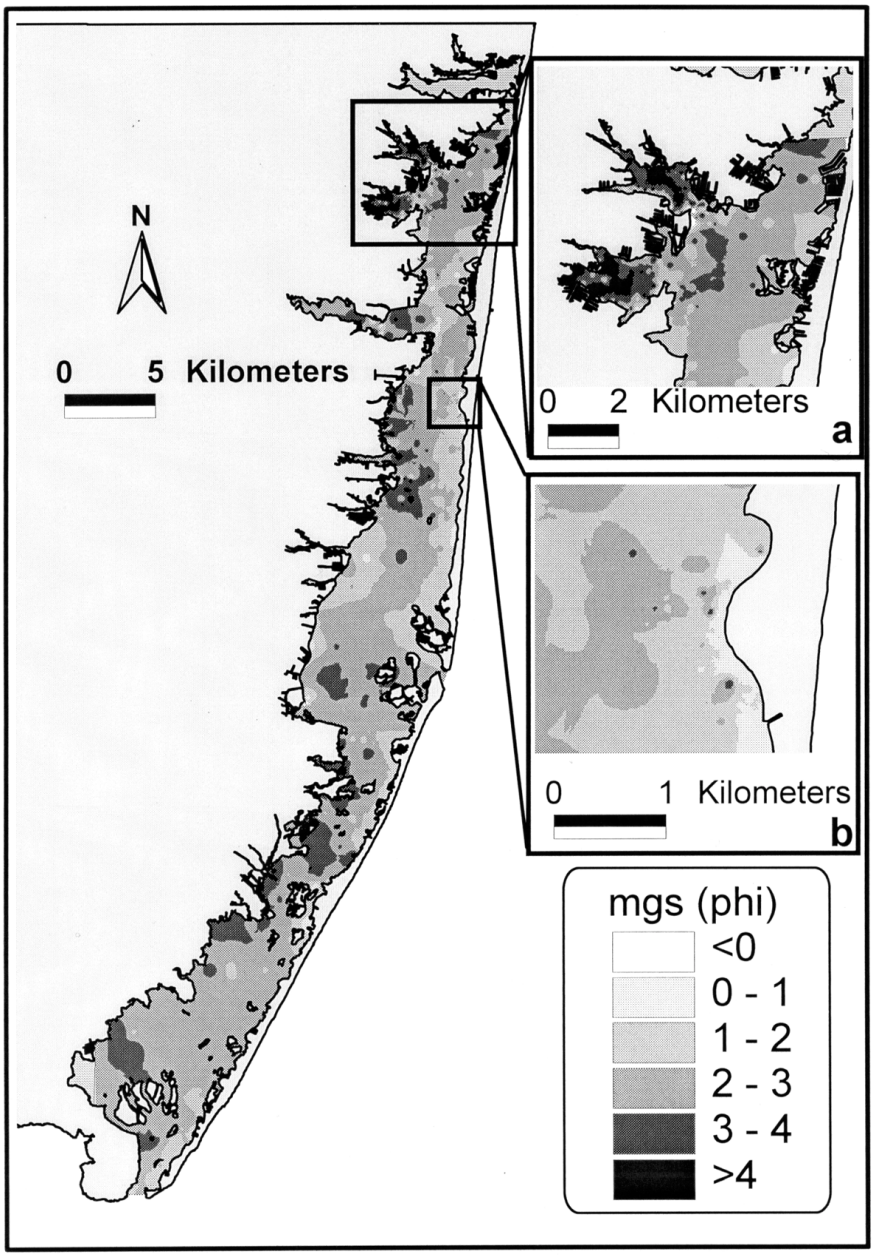


Figure 1 - 7 Bottom sediment composition and distribution (phi units) documented in the estuary. Finer grained sediments (silt, clay, and organic material) derived from upland areas, streams, and wetlands concentrate along the mainland and west side of the estuary. Well-sorted sands of marine origin 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).

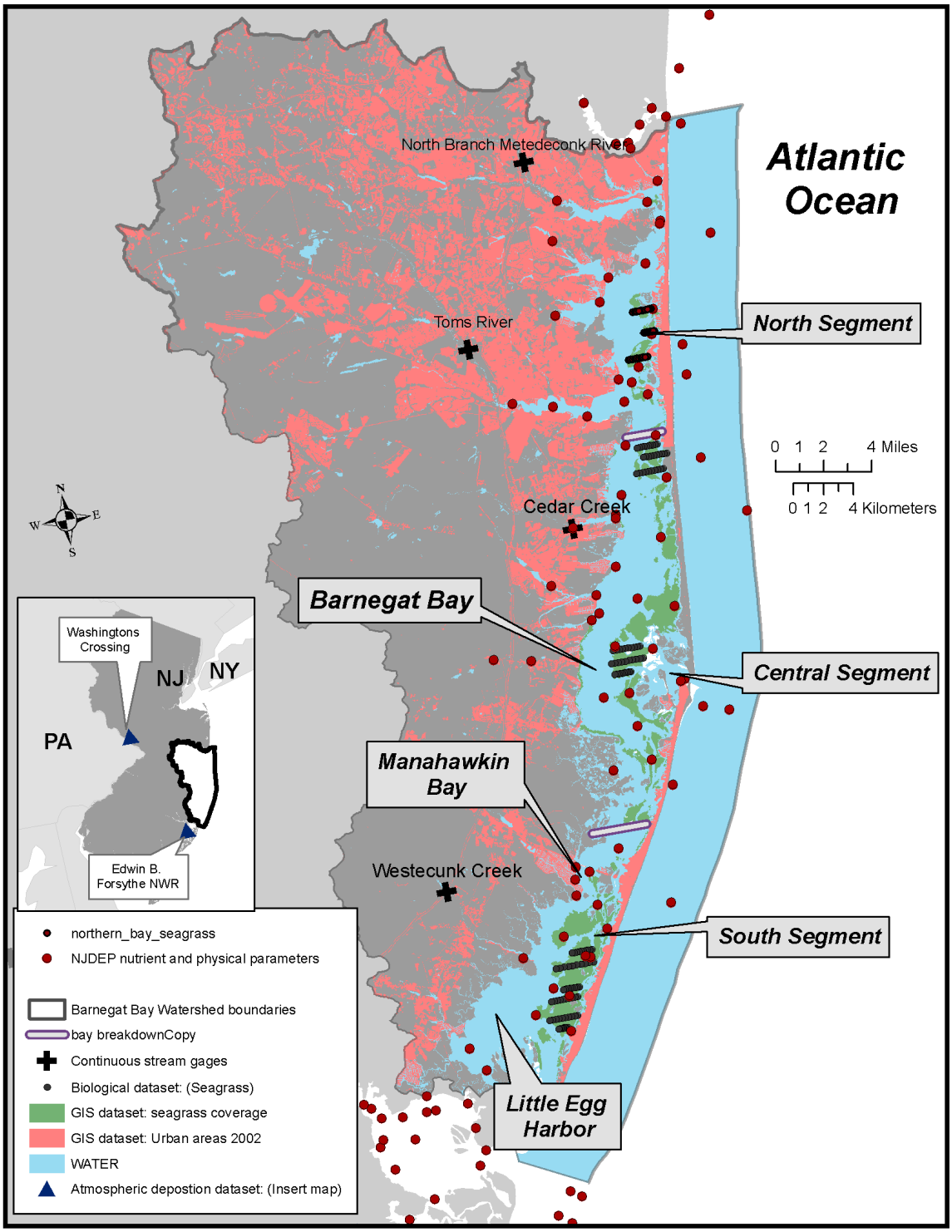


Figure 1 - 8 Map of the BB-LEH Estuary showing the location of 15 biotic sampling transects (150 sampling stations) in 2011.

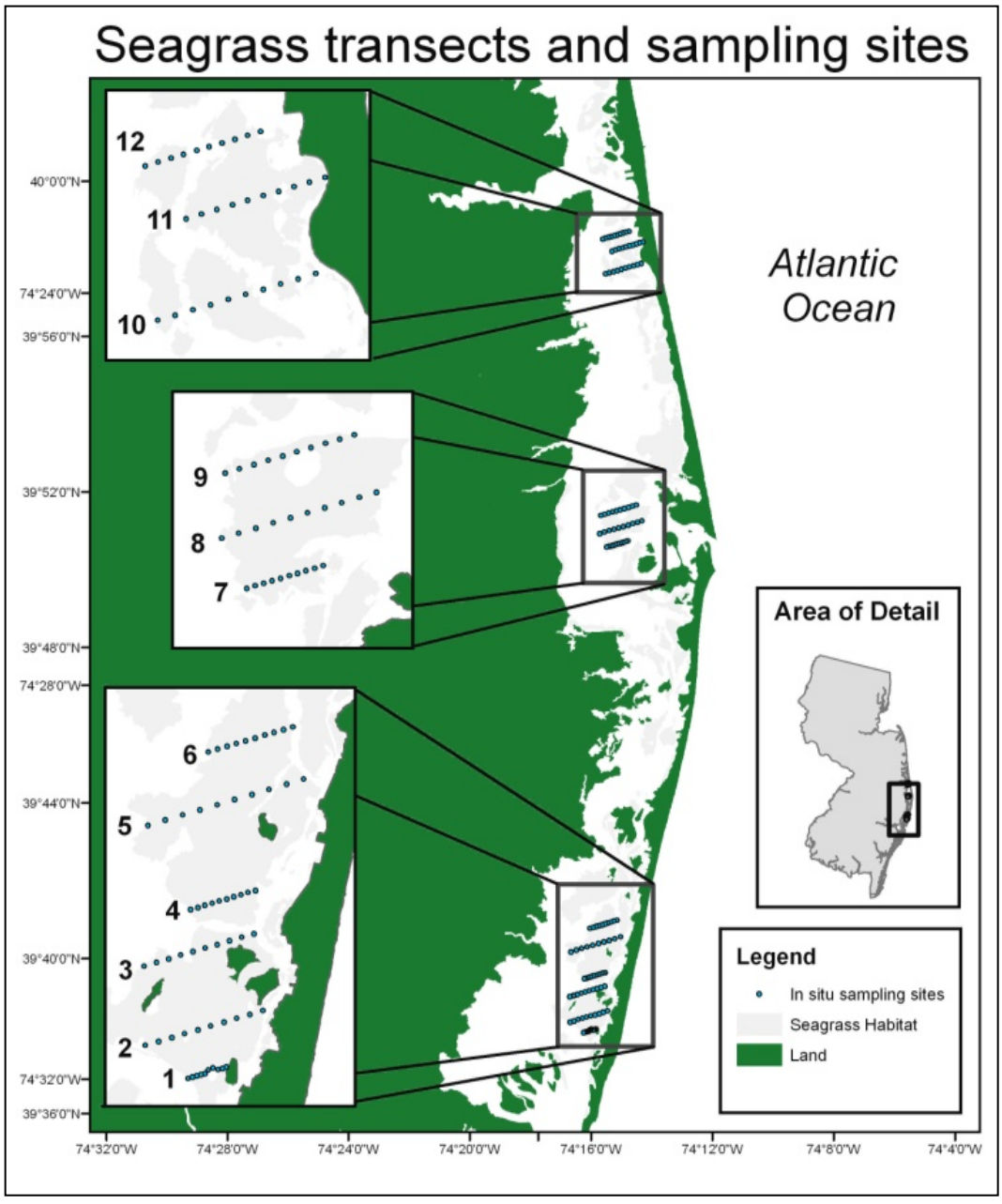


Figure 1 - 9 Study area showing 120 seagrass sampling sites along 12 transects in the BB-LEH Estuary from 2004-2010.

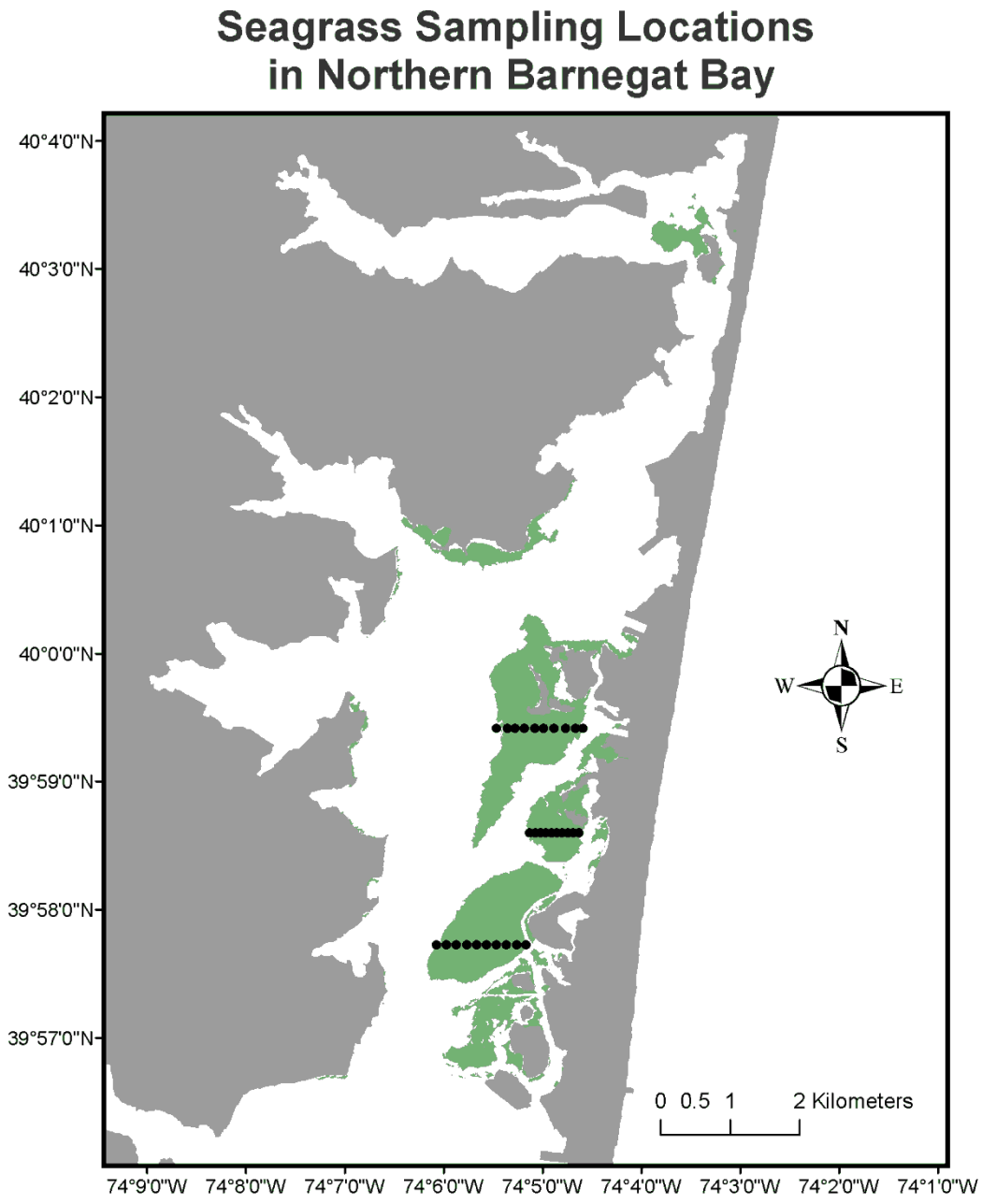


Figure 1 - 10 Seagrass transects established in the north segment of the BB-LEH Estuary for SAV sampling in 2011.

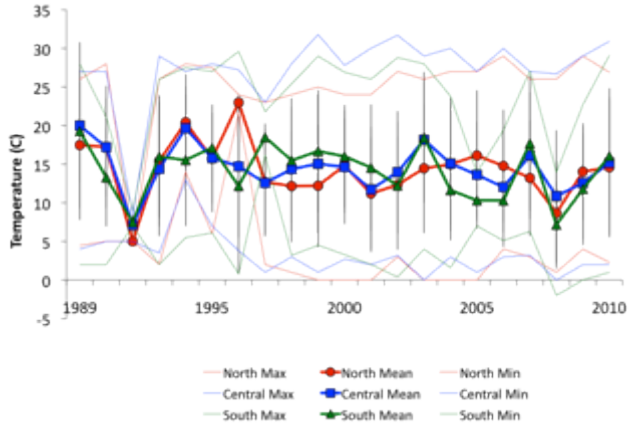


Figure 2 - 1 Minimum, mean, and maximum temperatures recorded in the BB-LEH Estuary from 1989-2010. Data from the New Jersey Department of Environmental Protection.

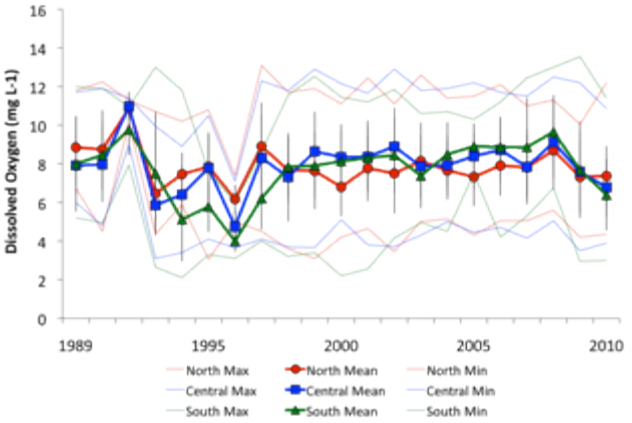


Figure 2 - 2 Minimum, mean, and maximum dissolved oxygen values recorded in the BB-LEH Estuary from 1989 to 2010. Data from the New Jersey Department of Environmental Protection.

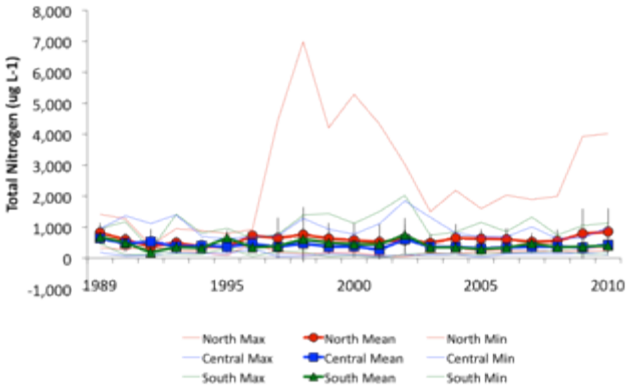


Figure 2 - 3 Minimum, mean, and maximum total nitrogen levels recorded in the BB-LEH Estuary from 1989 to 2010. Data from the New Jersey Department of Environmental Protection.

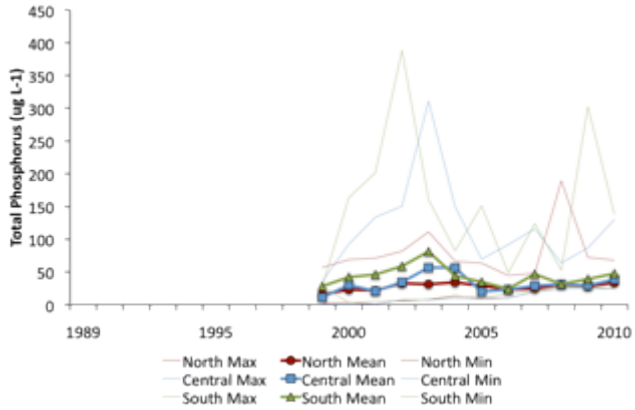


Figure 2 - 4 Minimum, mean, and maximum total phosphorus levels recorded in the BB-LEH Estuary from 1998 to 2010. Data from the New Jersey Department of Environmental Protection.

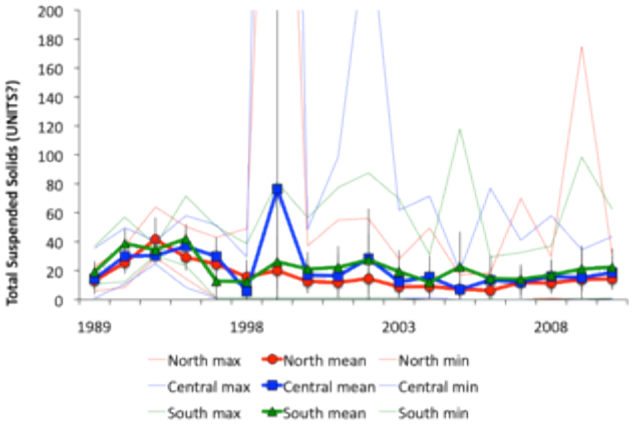


Figure 2 - 5 Minimum, mean, and maximum total suspended solids recorded in the BB-LEH Estuary from 1989 to 2010. Data from the New Jersey Department of Environmental Protection.

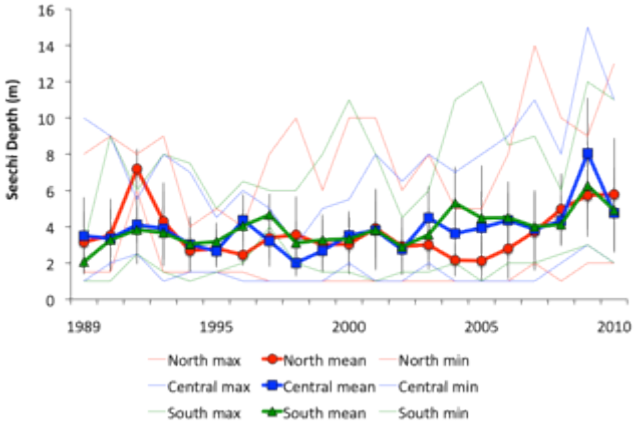


Figure 2 - 6 Minimum, mean, and maximum Secchi depth recorded in the BB-LEH Estuary from 1989-2010. Data from the New Jersey Department of Environmental Protection.

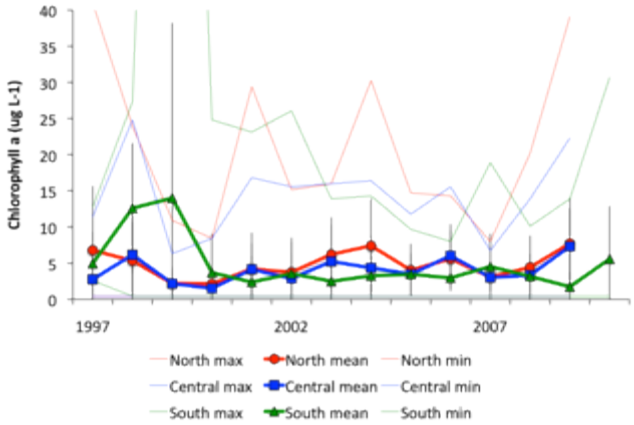


Figure 2 - 7 Minimum, mean, and maximum chlorophyll a values recorded in the BB-LEH Estuary from 1997 to 2010. Data from the New Jersey Department of Environmental Protection.

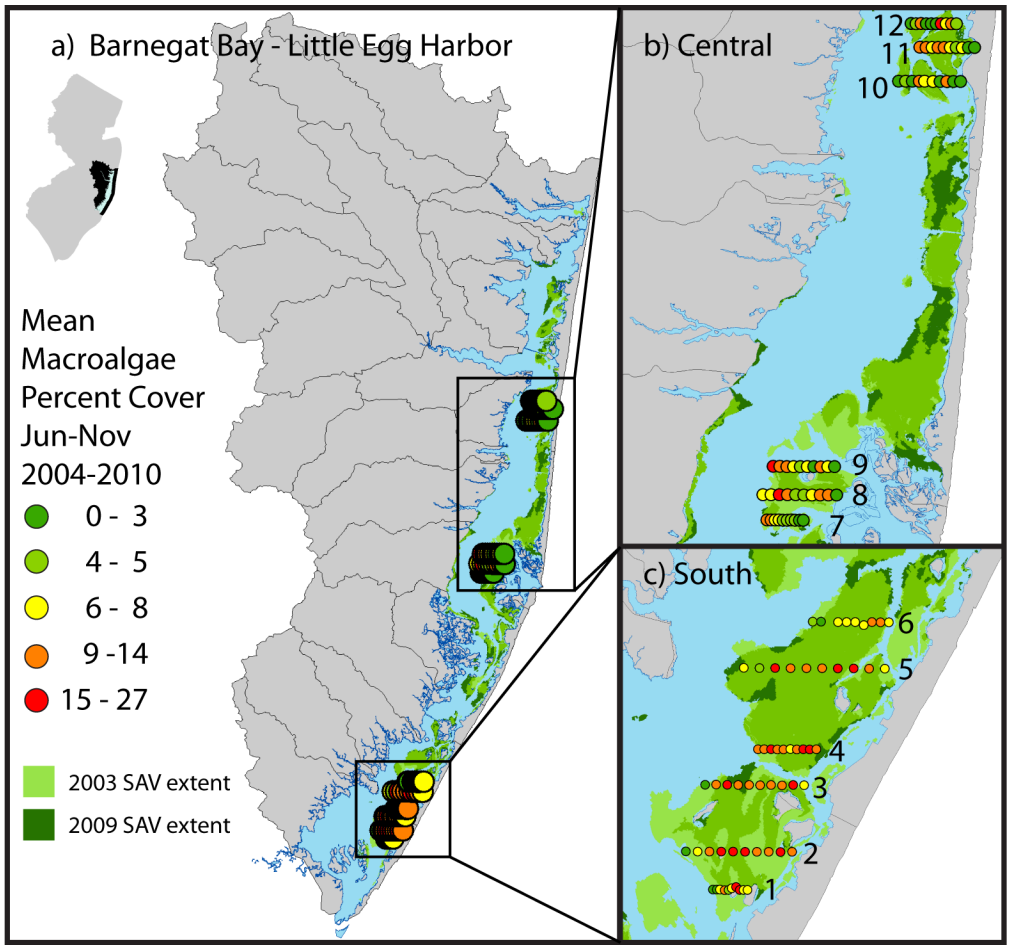


Figure 2 - 8 Mean macroalgae percent cover by sampling transect in the central and south segments of the estuary during the 2004-2010 period.

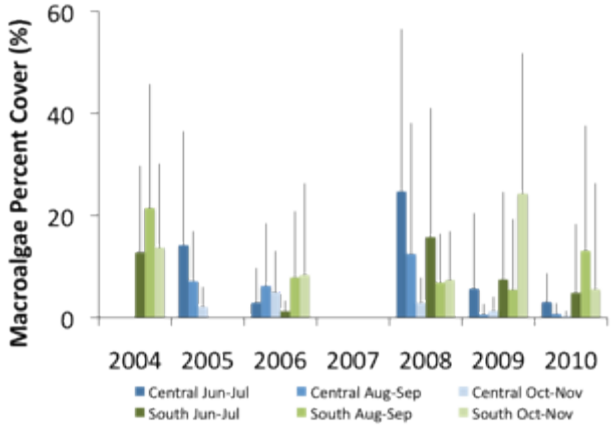


Figure 2 - 9 Mean macroalgae percent cover by sampling year (2004-2010) in the central and south segments of the estuary.

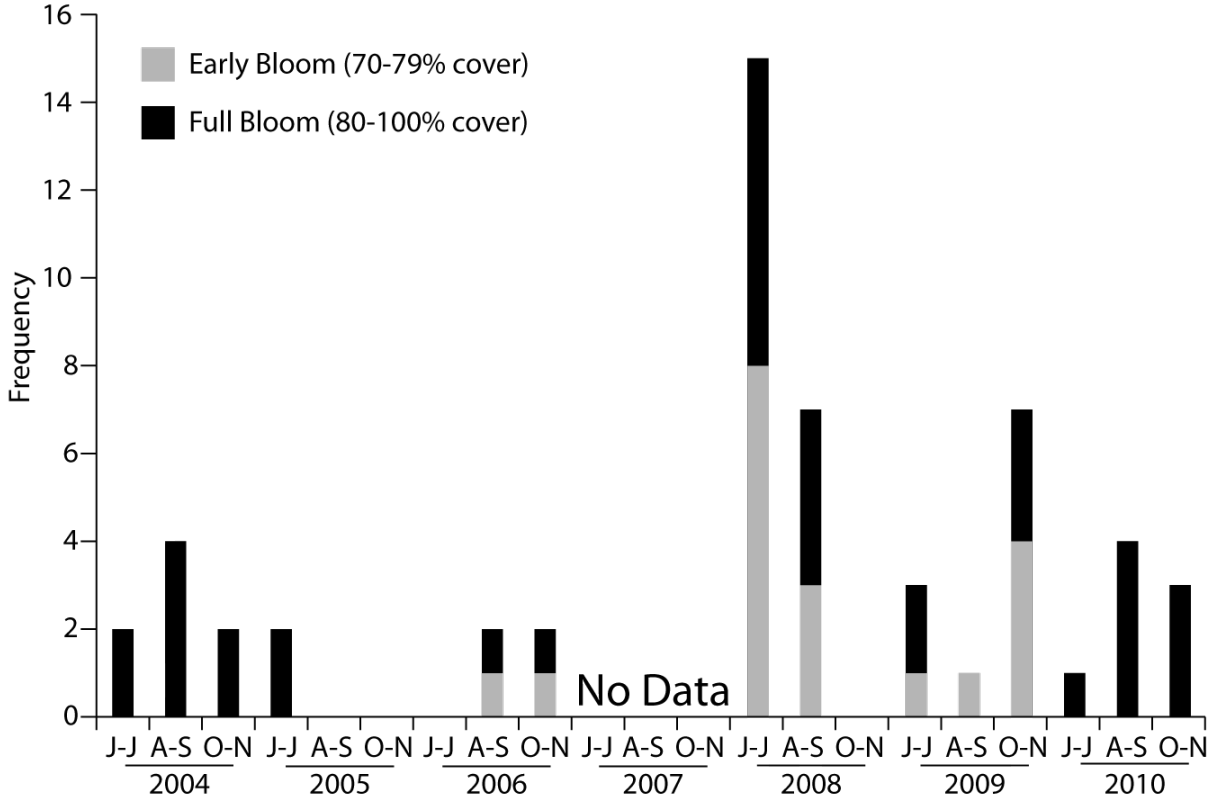


Figure 2 - 10 Frequency of macroalgae cover at 'Early Bloom' = 70-79%, and 'Full Bloom' = > 80% conditions.

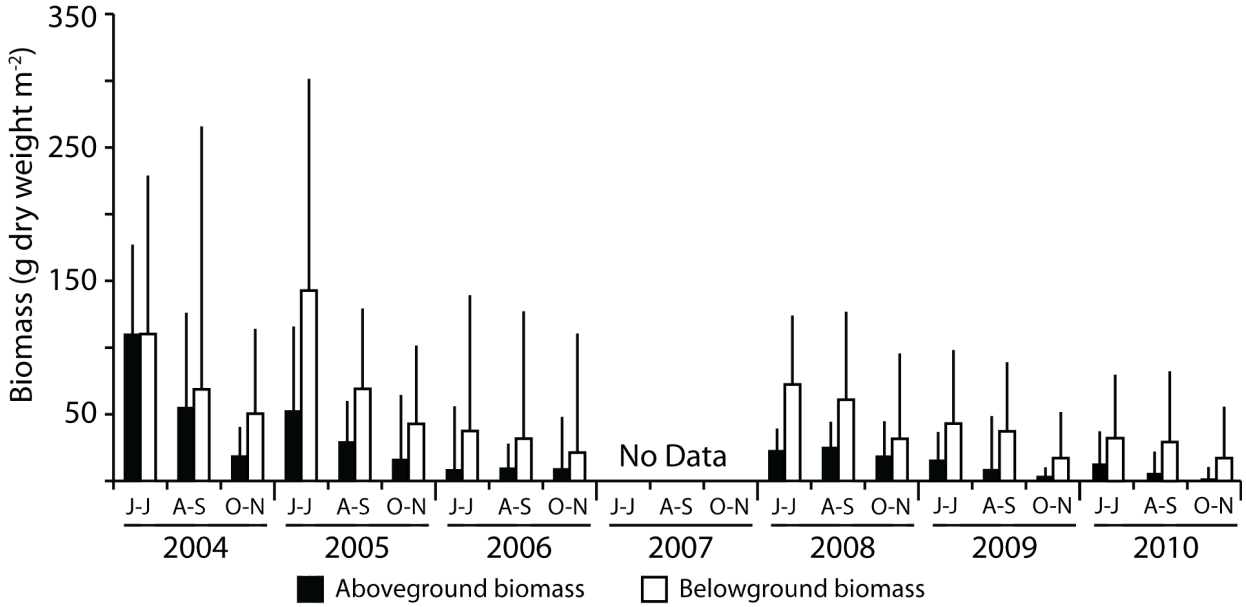


Figure 2 - 11 Mean aboveground and belowground eelgrass biomass values in the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

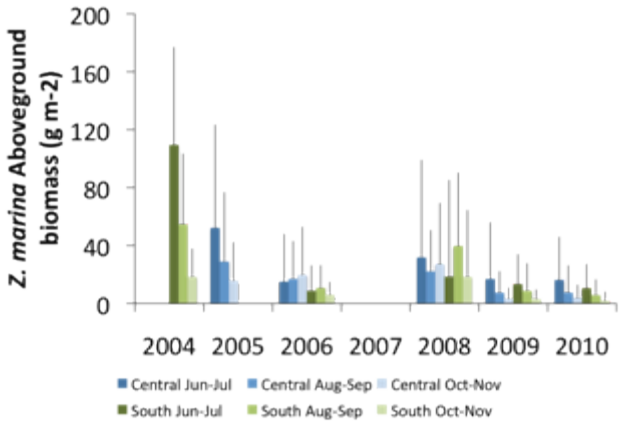


Figure 2 - 12 Mean aboveground eelgrass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

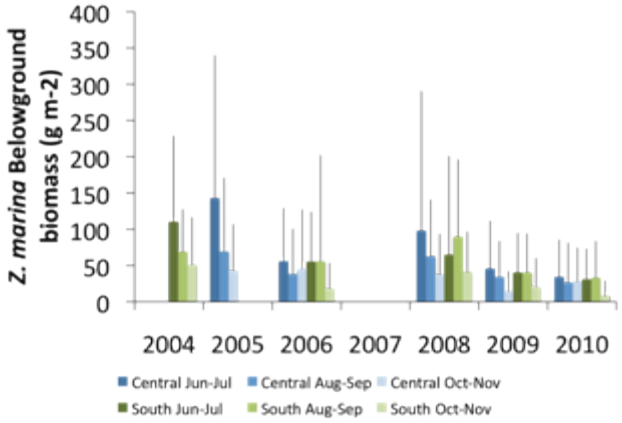


Figure 2 - 13 Mean belowground eelgrass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

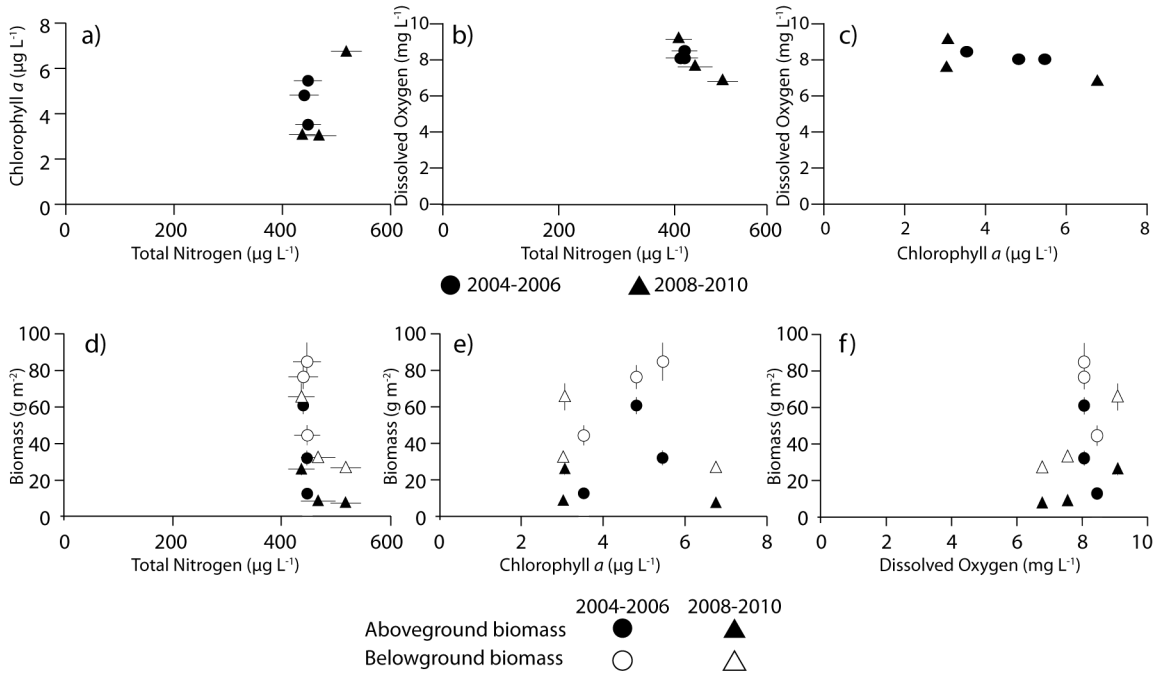


Figure 2 - 14 Variation of water quality and biological metrics for 2004-2006 (circles) and 2008-2010 (triangles). Eelgrass biomass is divided into aboveground (black) and belowground (white) components. Plots include chlorophyll a *vs. total nitrogen (a), dissolved oxygen vs. total nitrogen (b), dissolved oxygen vs. chlorophyll* a *(c), eelgrass biomass vs. total nitrogen (d), eelgrass biomass vs. chlorophyll* a *(e), and eelgrass biomass vs. dissolved oxygen (f).*

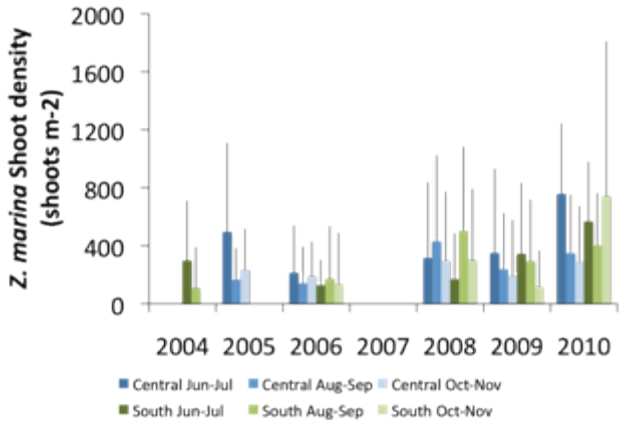


Figure 2 - 15 Mean eelgrass shoot density in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

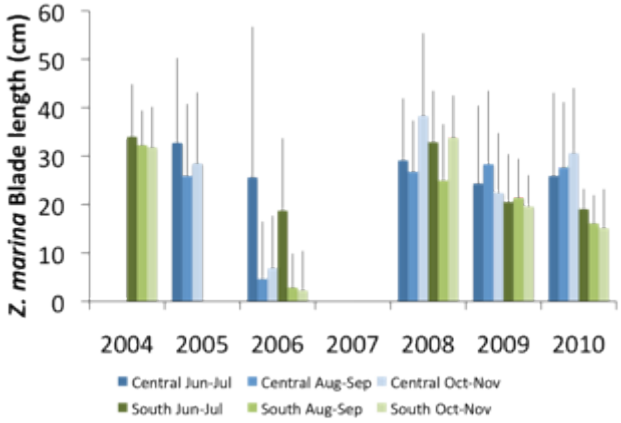


Figure 2 - 16 Mean eelgrass blade length in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

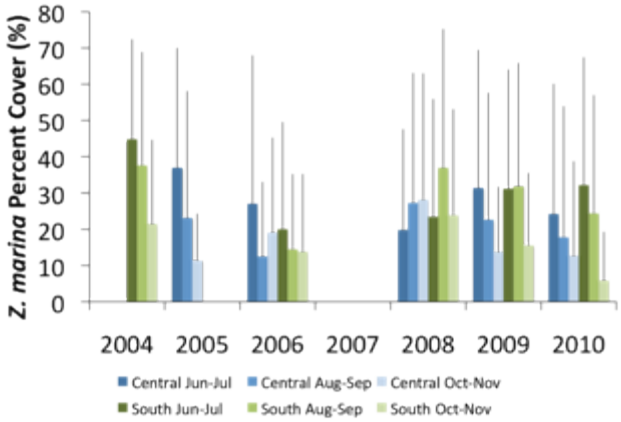


Figure 2 - 17 Mean eelgrass percent cover in the central and south segments of the BB-LEH during the spring-fall sampling periods from 2004 to 2010.

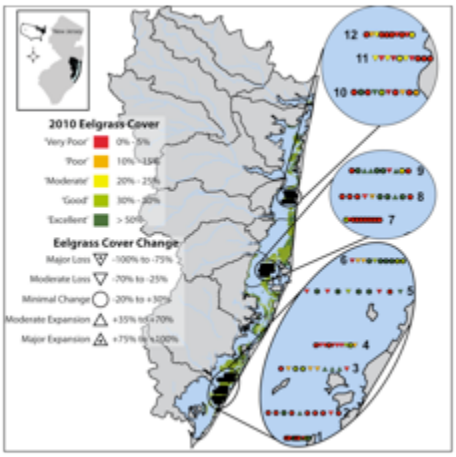


Figure 2 - 18 Eelgrass percent areal cover along 12 transects in the BB-LEH Estuary during 2010.

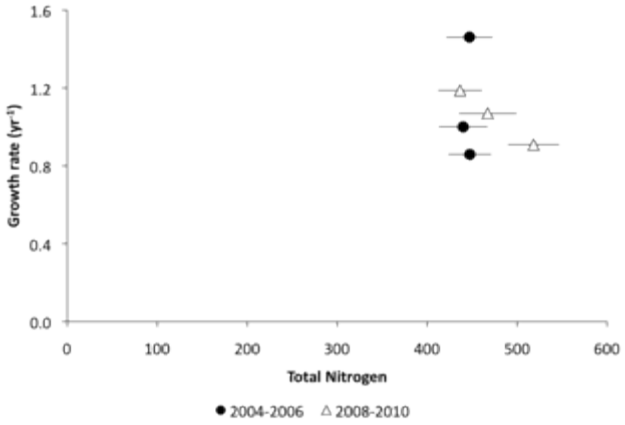


Figure 2 - 19 Annual growth rate of Z. marina in BB-LEH Estuary vs. total nitrogen concentrations, during 2004-2006 (black circles) and 2008-2010 (white triangles).

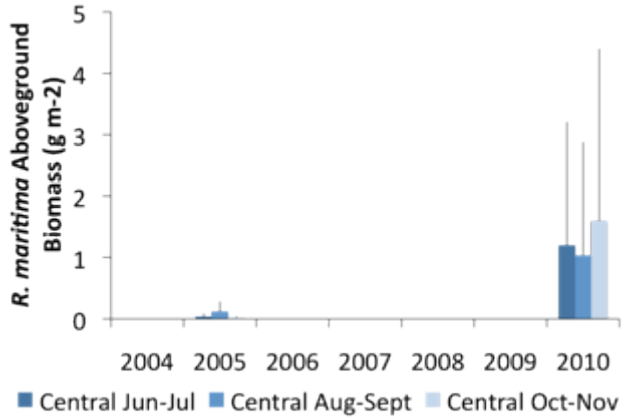


Figure 2 - 20 Mean aboveground widgeon grass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

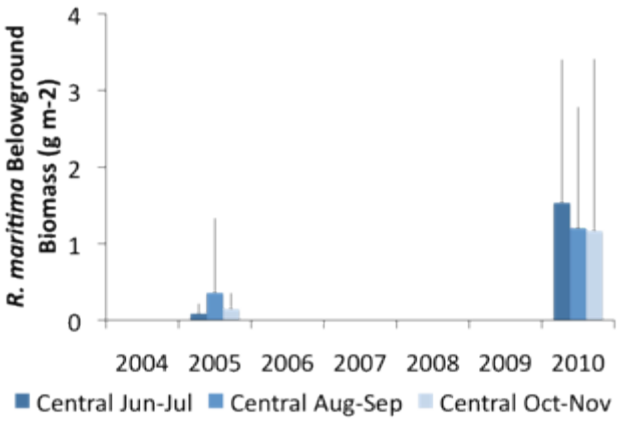


Figure 2 - 21 Mean belowground widgeon grass biomass values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

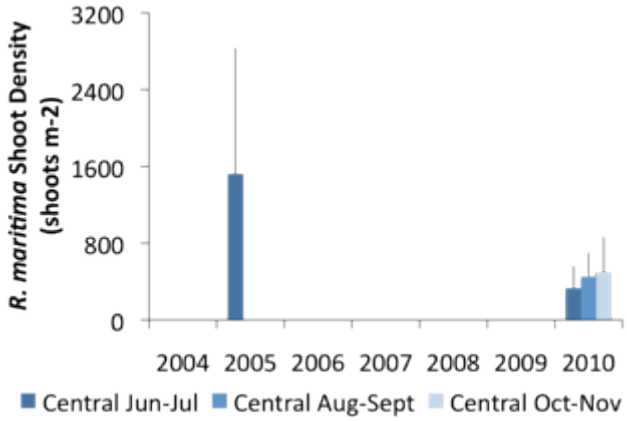


Figure 2 - 22 Mean widgeon grass shoot density values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

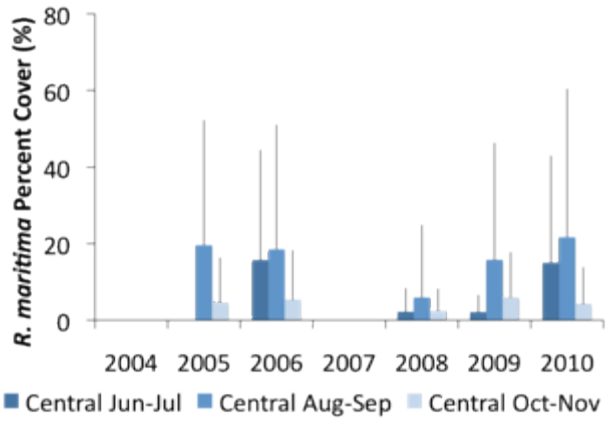


Figure 2 - 23 Mean widgeon grass percent cover values in the central and south segments of the BB-LEH Estuary during the spring-fall sampling periods from 2004 to 2010.

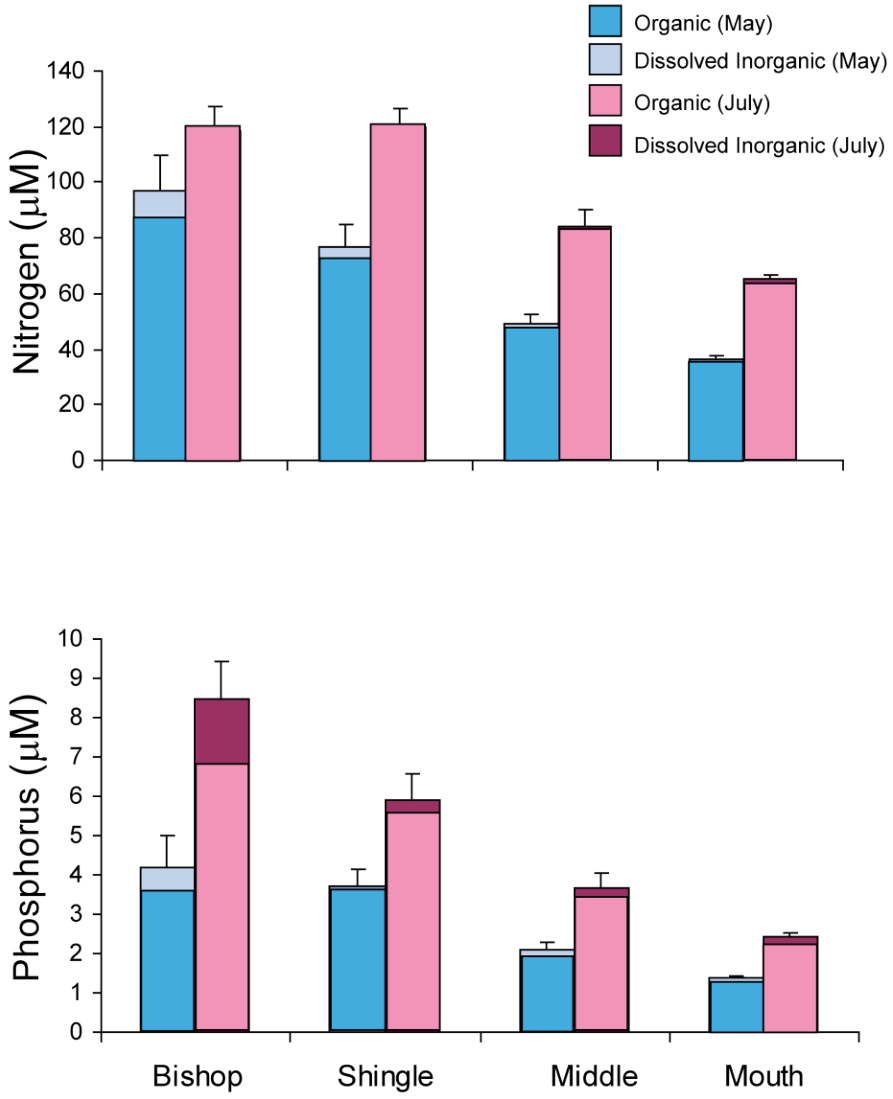


Figure 3.9: Organic and dissolved inorganic composition of nitrogen and phosphorus concentrations in the different sections of St. Martin River, May and July 2007. Error bars represent the standard error about the mean (bars). Dissolved inorganic fractions (NH_4 + NO_x and PO_4) are the upper portions of each bar graph, and organic fractions (dissolved and particulate) are the bottom portions.

Figure 2 - 24 Dissolved organic and inorganic nitrogen and phosphorus in a coastal lagoon similar to BB-LEH (from Beckert 2008)

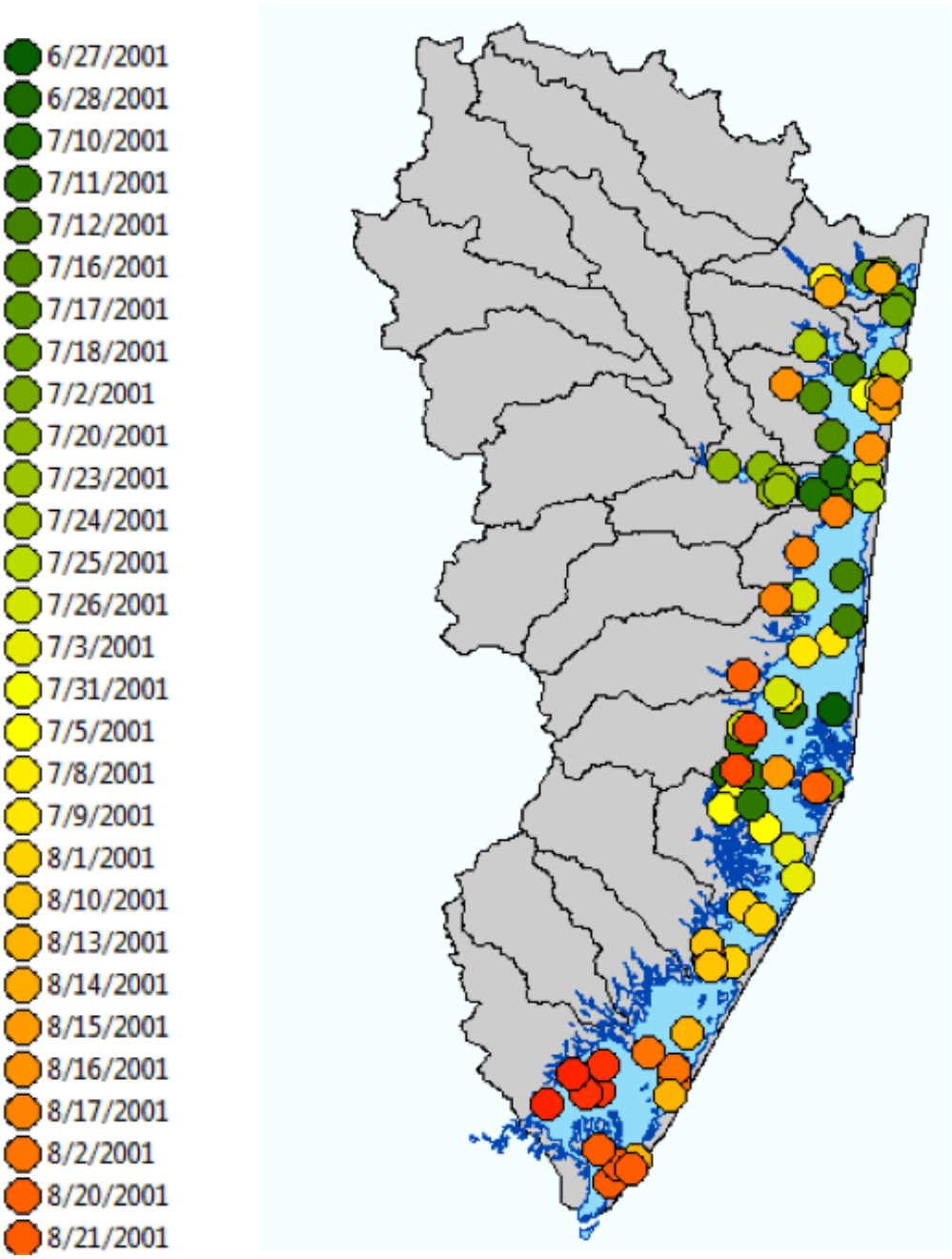
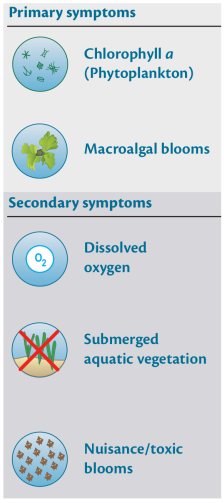


Figure 2 - 25 Benthic invertebrate sampling stations (2001) of the USEPA Regional Monitoring and Assessment Program for the BB-LEH Estuary.

This Biotic Index builds on the NEEA-ASSETS approach

- **NEEA**



RUTGERS MARINE & COASTAL SCIENCES

- **Biotic Index**
  - ~ **20** 'symptoms' or 'metrics'
  - Organization and integration necessary
  - Condition assessment in 3 segments for each year data available

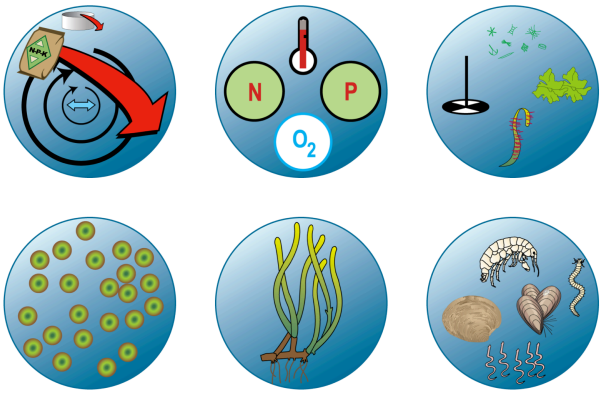


Figure 3 - 1 Comparison of indicators used by Bricker et al. 2009 and those used in this Index of Eutrophication Condition.

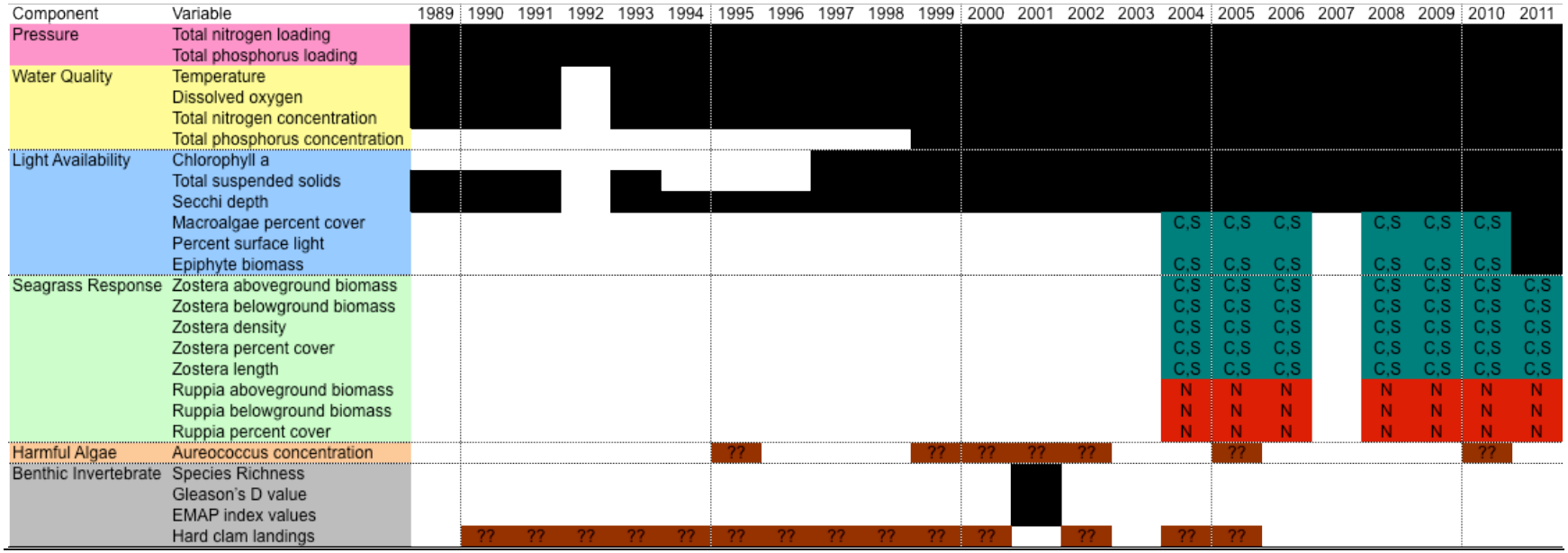


Figure 3 - 2 Temporal and spatial data availability for indicators used in the Index of Eutrophication

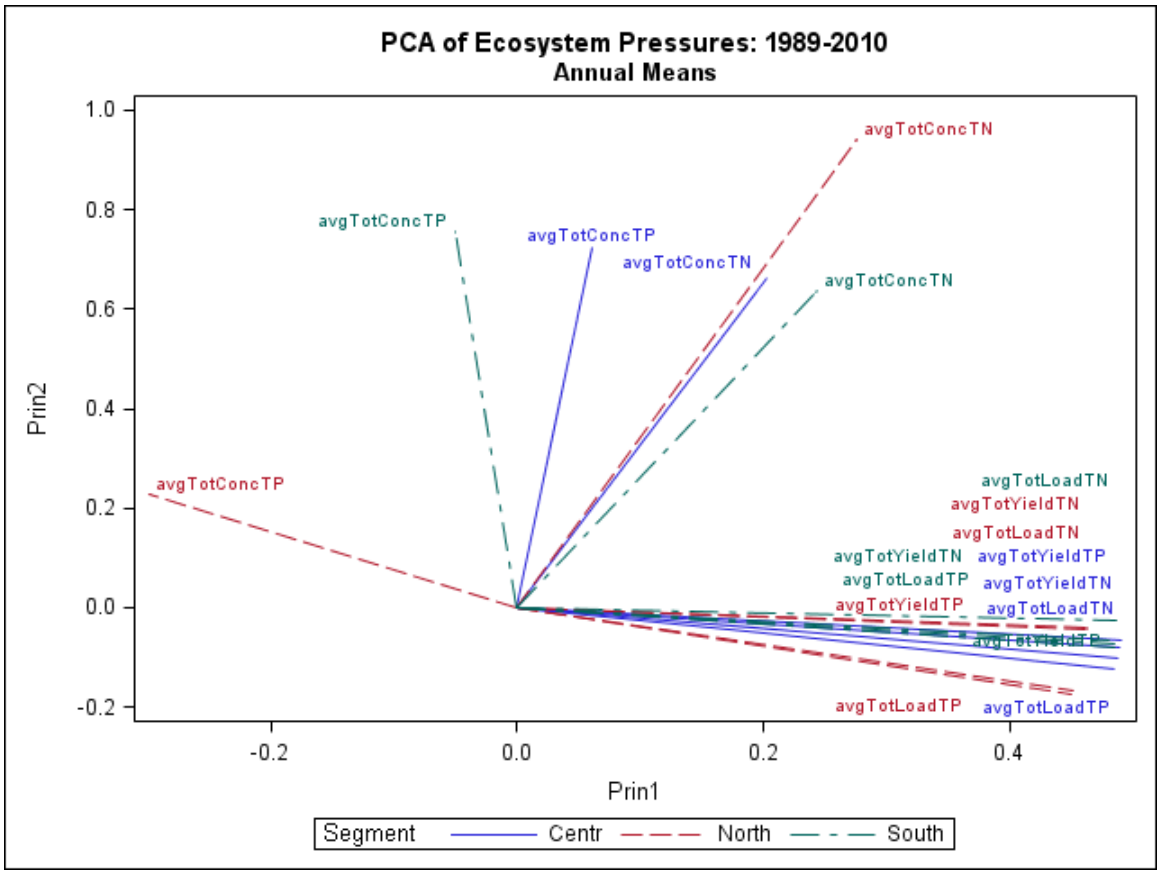


Figure 3 - 3 Principal component analysis of Total Loading, Total Yield, and Flow-weighted average total concentration for total nitrogen and total phosphorus.

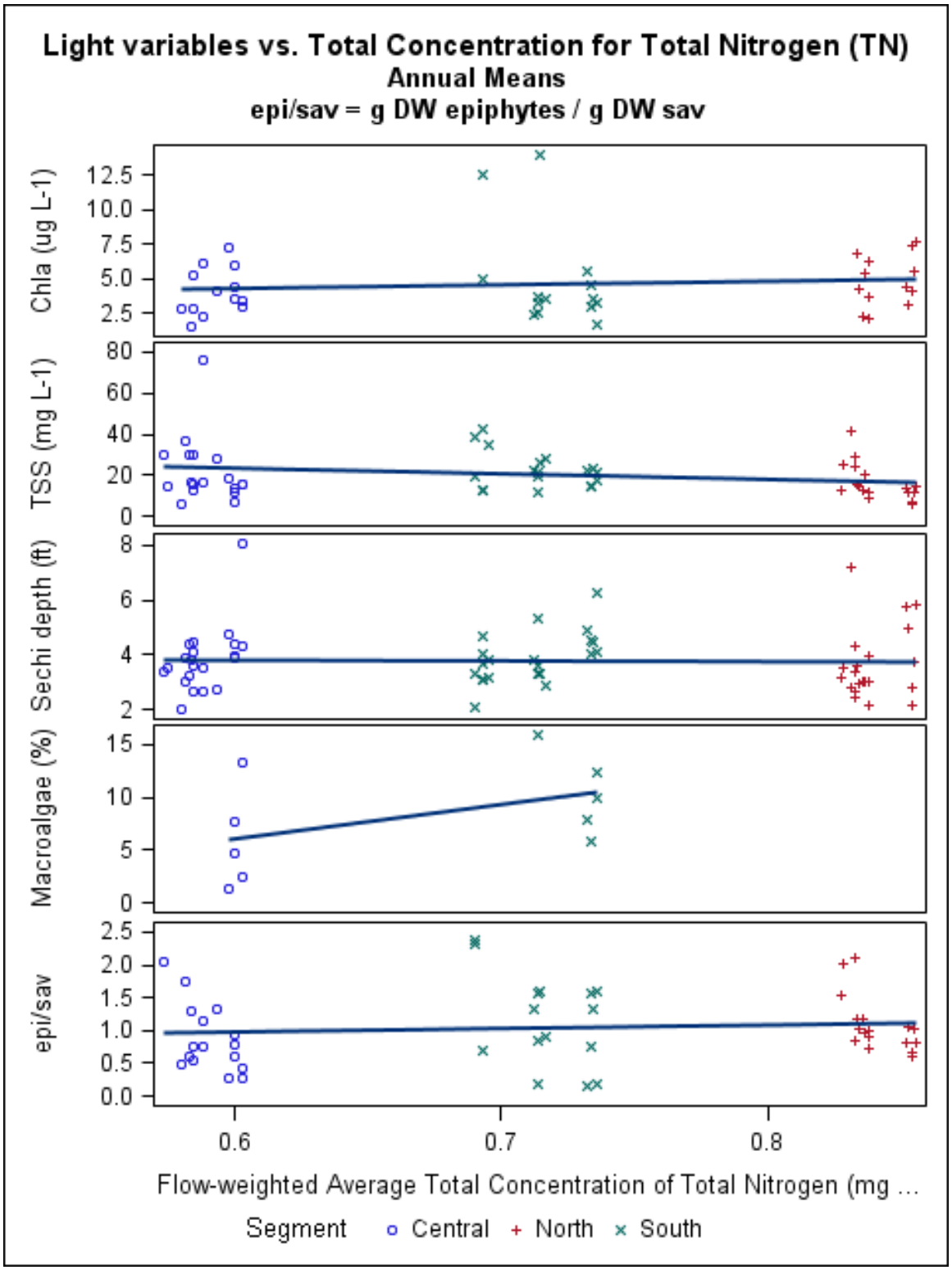


Figure 3 - 4 Light variables vs. flow-weighted average total concentration of total nitrogen (mg L^{-1}).

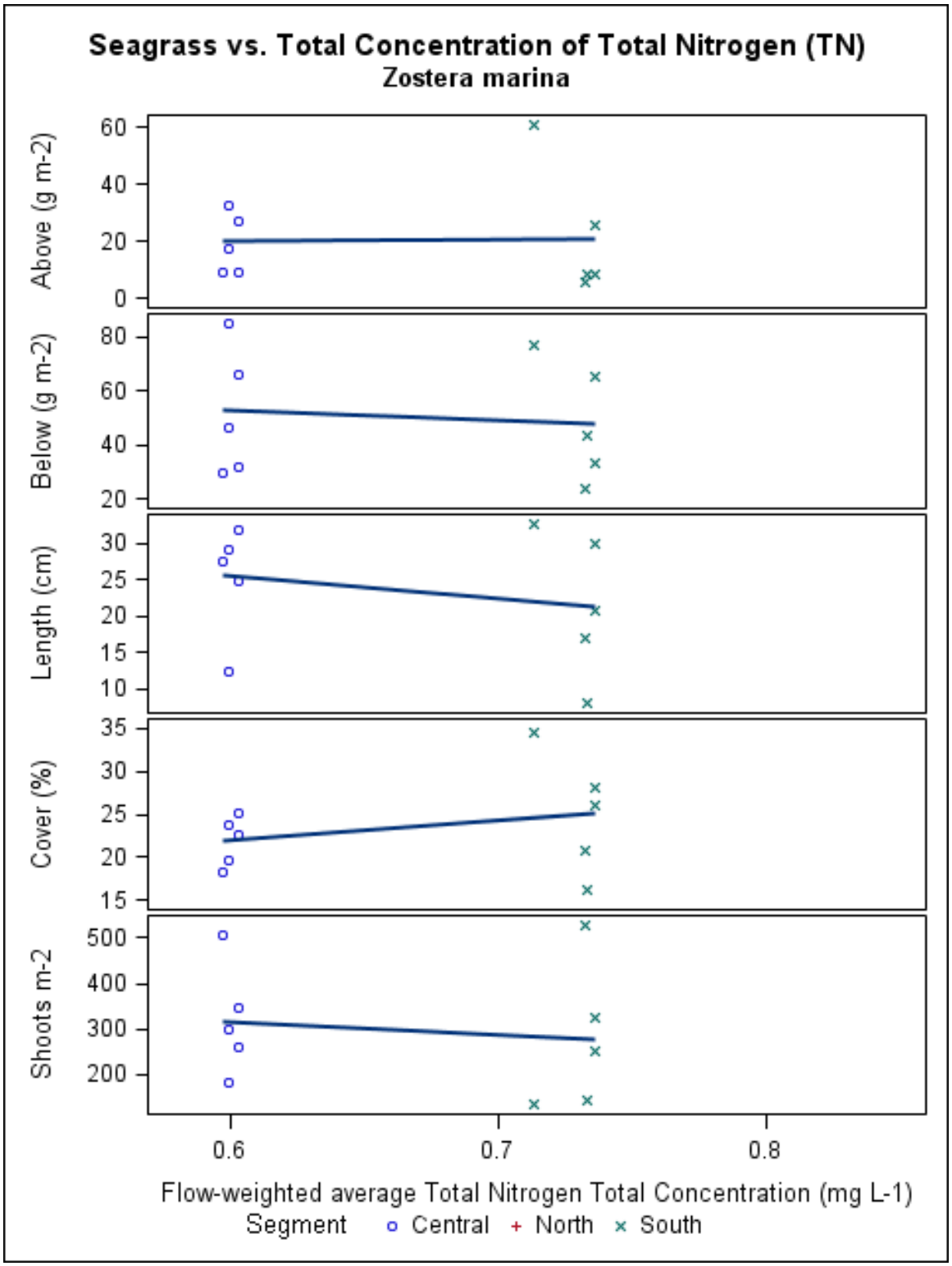


Figure 3 - 5 Zostera marina indicators vs. flow-weighted average total concentration of total nitrogen (mg L^{-1}).

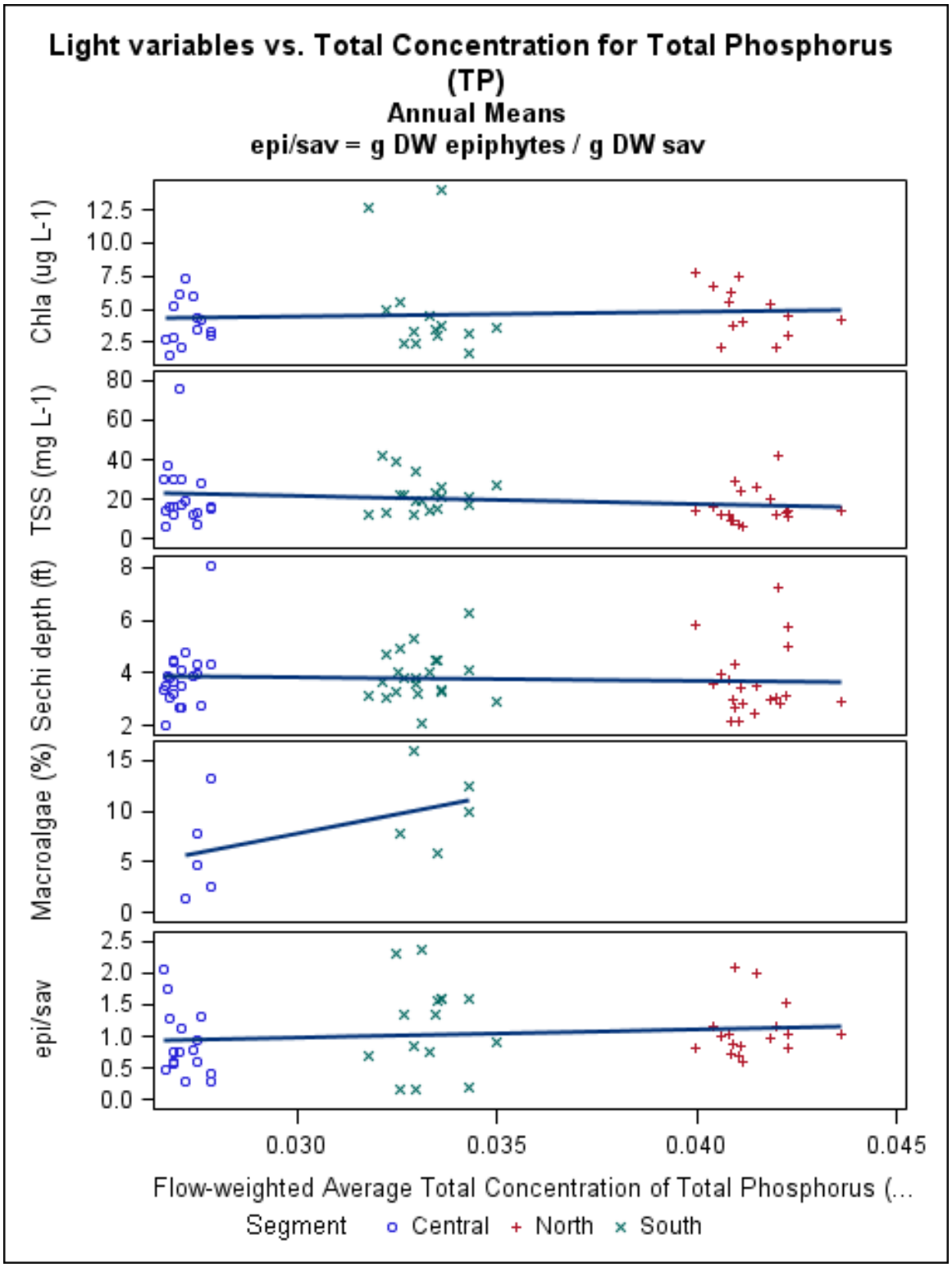


Figure 3 - 6 Light indicators vs. flow-weighted avererage total concentration of total phosphorus (mg L^{-1}).

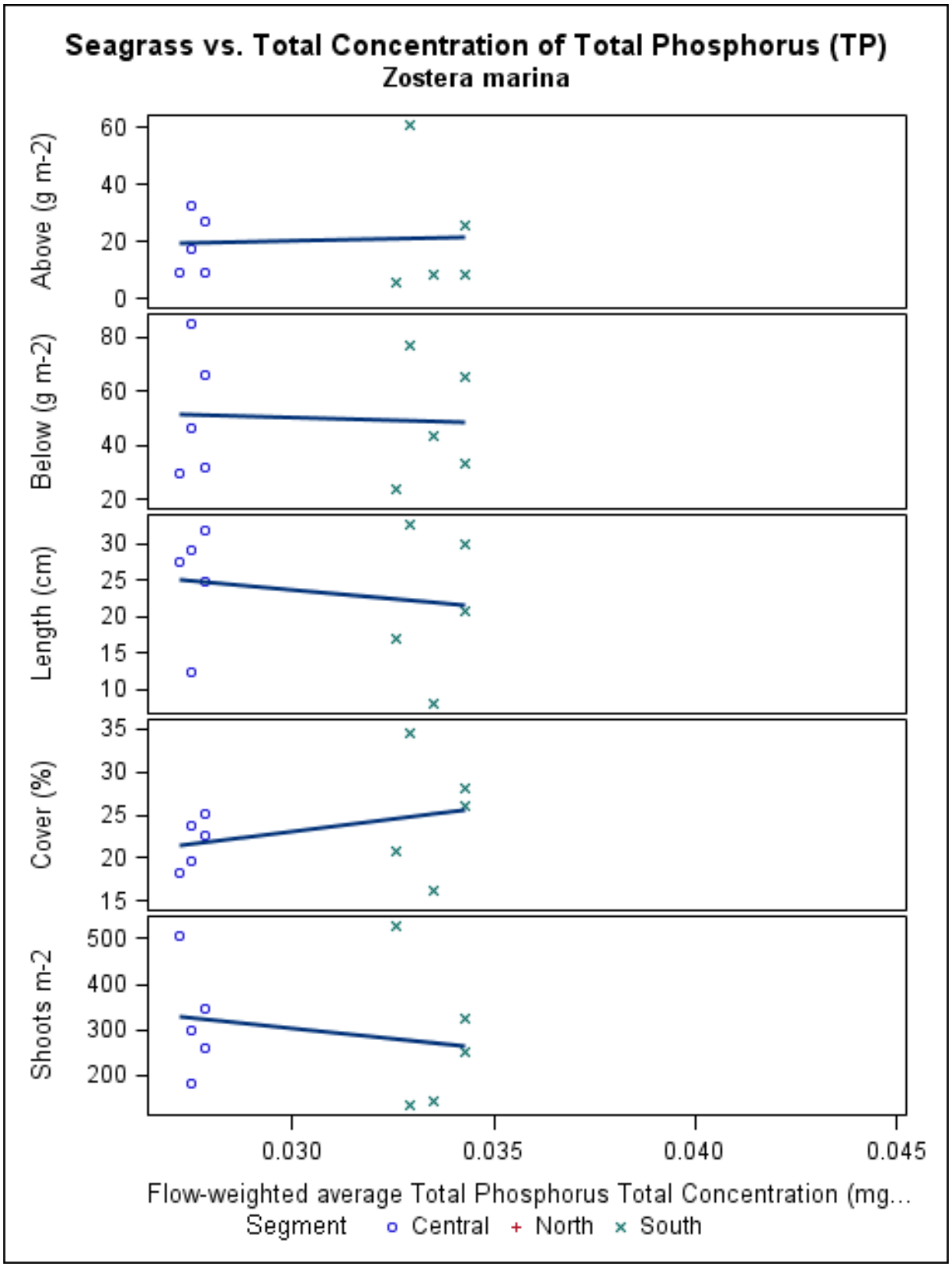
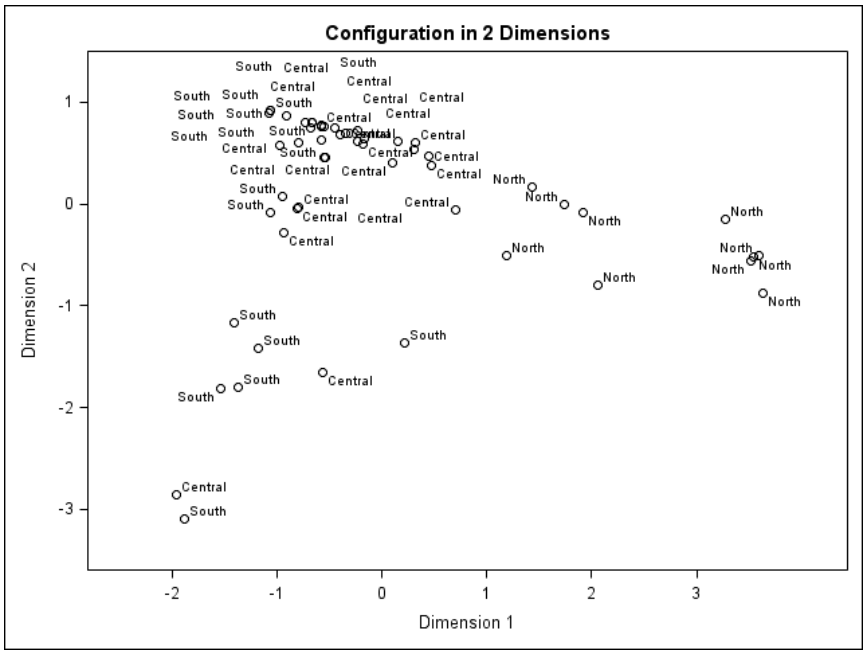
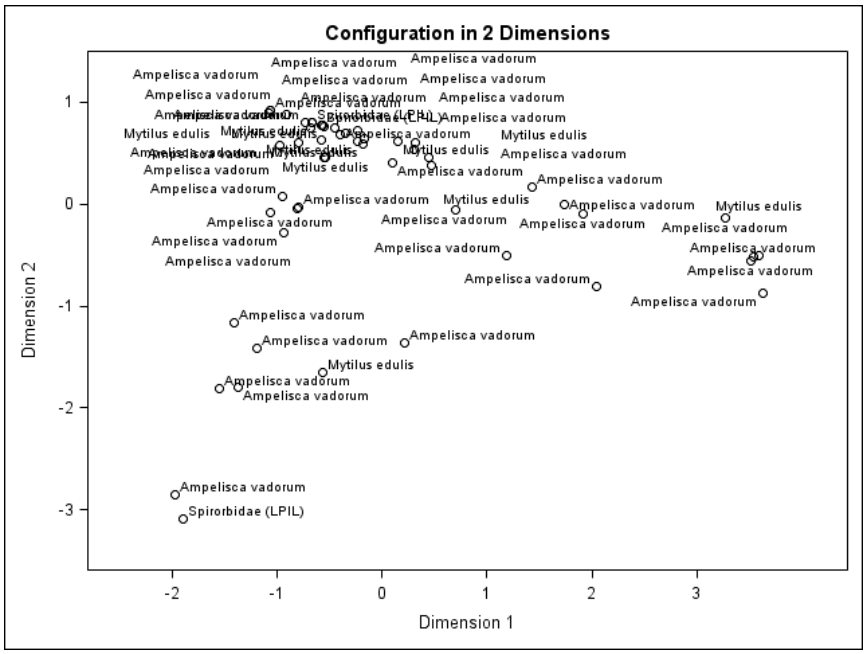


Figure 3 - 7 Zostera marina indicators vs. flow-weighted averaged total concentration of total phosphorus (mg L^{-1}).



a)



b)

Figure 3 - 8 Principal component analysis (PCA) on benthic invertebrate abundances for Ampelisca vadorum, Mytilus edulis, and Spirobidae, the three most abundant taxa in the REMAP data from US EPA labeled by segment (a) and by taxa name (b)

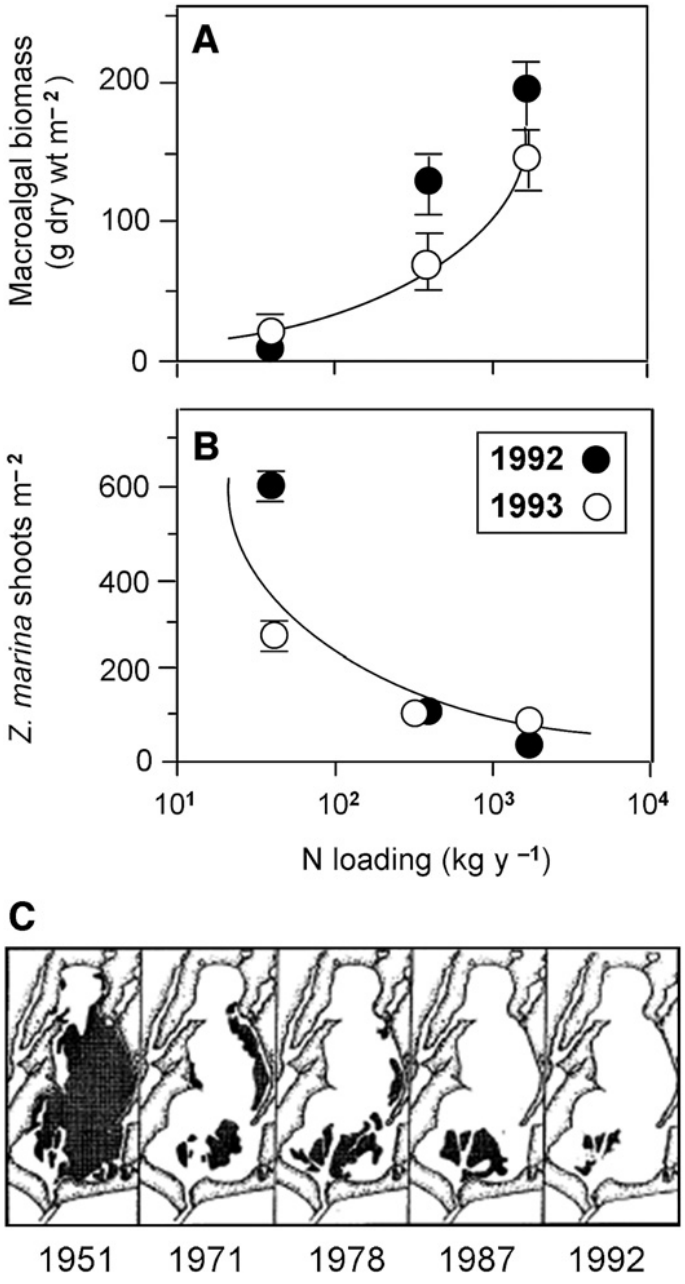


Fig. 2. Changes in plant abundance as (A) macroalgae (biomass) and (B) *Zostera marina* (shoot number) in response to N enrichment (modified from Deegan 2002; note that loading was estimated based on area of open water). (C) Change in spatial location and patch size of *Z. marina* distribution in Waquoit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

Figure 3 - 9 Increase of macroalgae and decline of seagrass shoot density and areal coverage with increasing nitrogen loading. (From Burkholder et al. 2007).

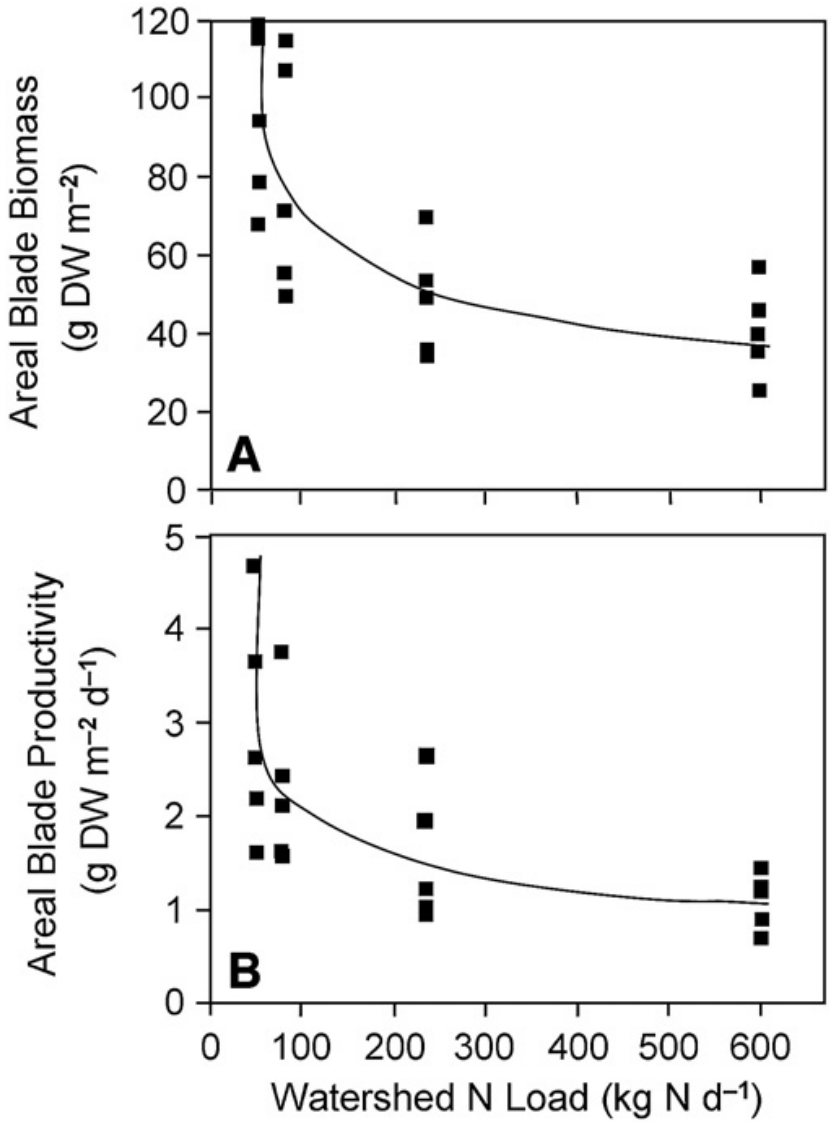


Fig. 7. (A) Areal blade biomass and (B) areal blade productivity plotted against watershed nitrogen loads for *Thalassia testudinum* from four sites in Sarasota Bay. Line is best-fit relationship. Modified from Tomasko et al. (1996).

Figure 3 - 10 Impact of nitrogen loading on seagrass biomass and productivity. (From Tomasko et al. 1996).

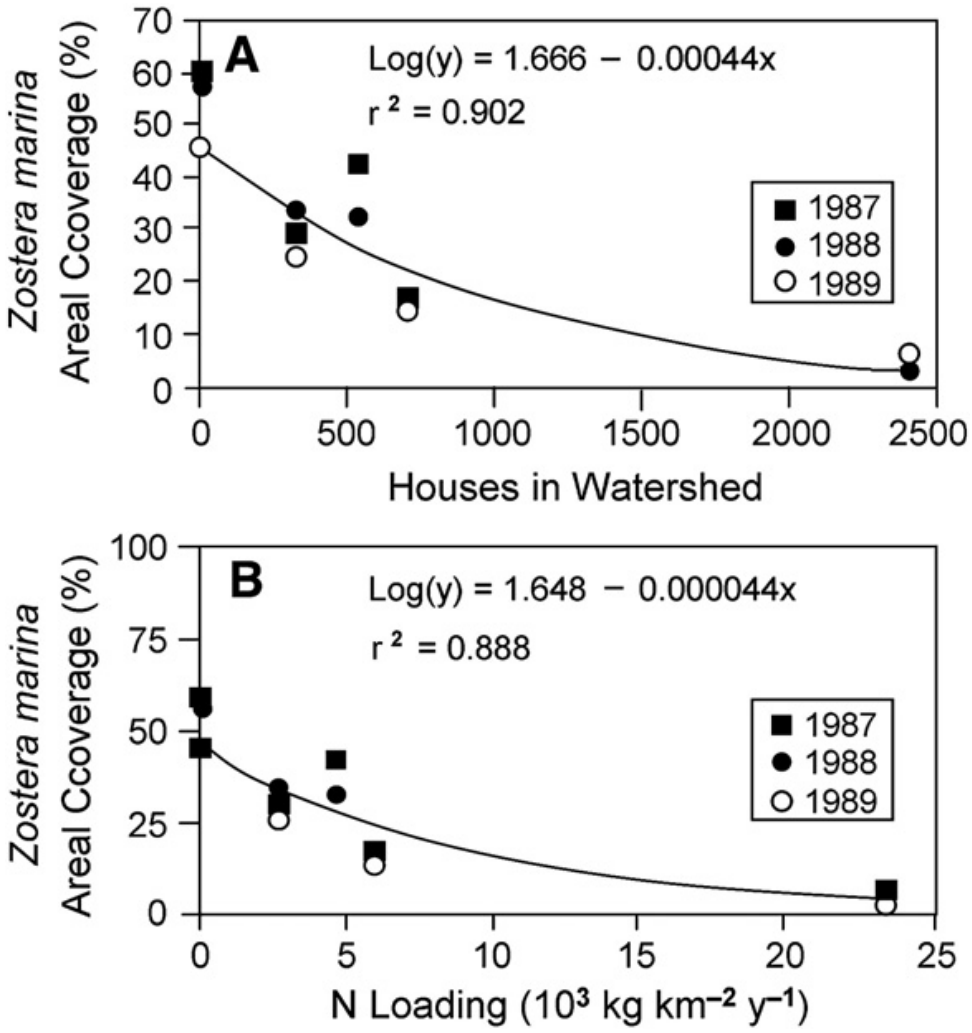


Fig. 11. Relationship between seagrass areal coverage (log of eelgrass area) in Waquoit Bay during 1987–1989 and (A) the number of houses in the sub-watersheds, and (B) nitrogen loading. From Short and Burdick (1996), with permission from the publisher; note that loading was estimated based on watershed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

Figure 3 - 11 Losses of seagrass areal coverage with increasing nitrogen loading. (From Short and Burdick 1996).

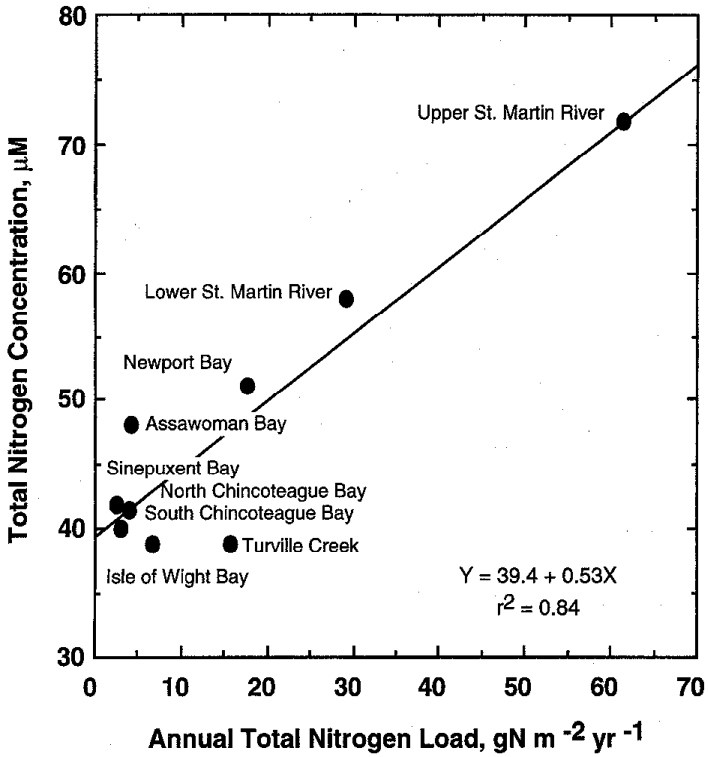


Fig. 7. A scatter plot relating annual areal total nitrogen loads to annual average total nitrogen concentrations for several regions of the Maryland coastal bays. Loading data are from Jacobs et al. (1993) and total nitrogen concentrations are from Fang et al. (1977a).

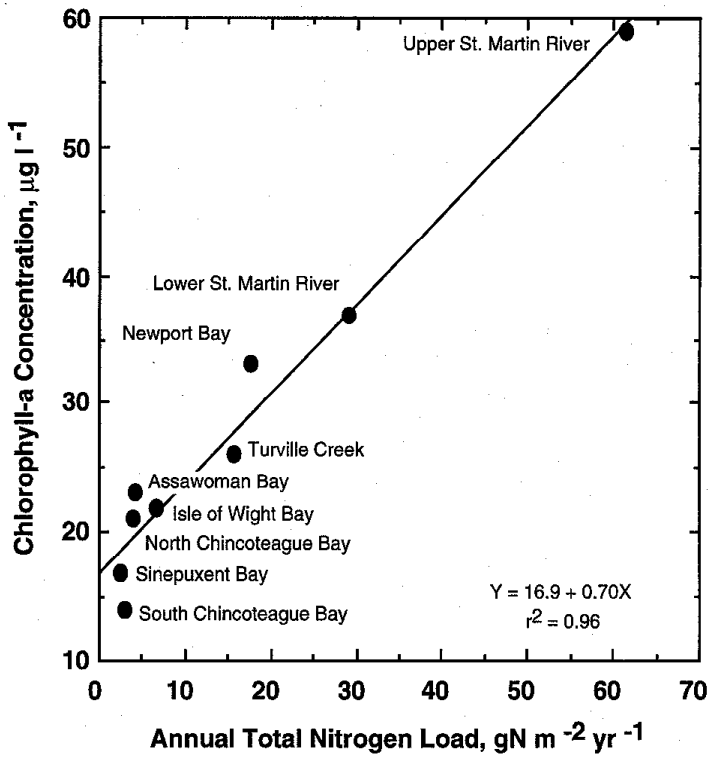


Fig. 8. A scatter plot relating annual areal total nitrogen loads to annual average chlorophyll *a* concentrations for several regions of the Maryland coastal bays. Loading data are from Jacobs et al. (1993) and total nitrogen concentrations are from Fang et al. (1977a).

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).

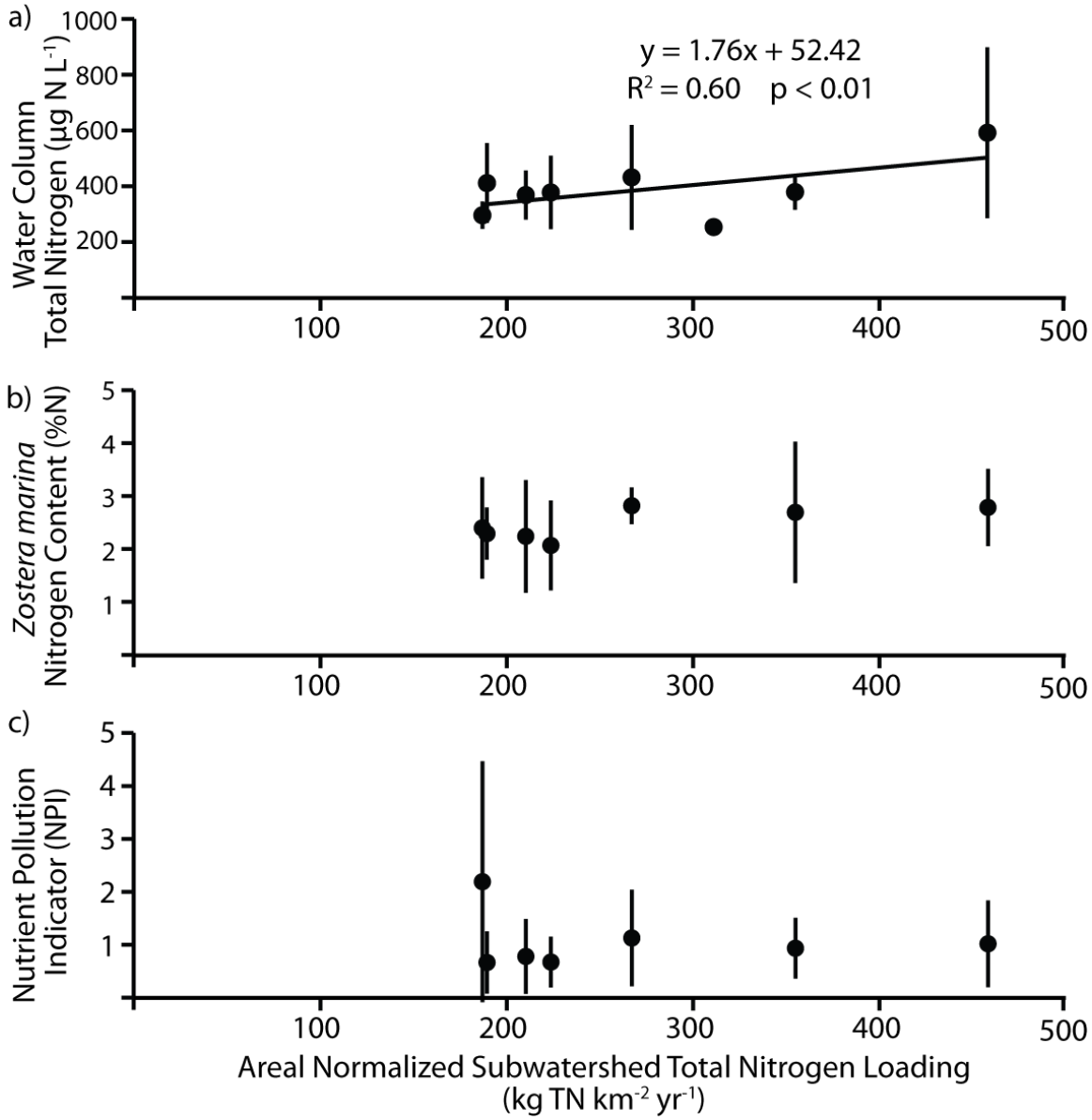


Figure 3 - 13 Estuarine nitrogen concentration, eelgrass nitrogen content, and a 'Nutrient Pollution Indicator' vs nitrogen loading in Barnegat Bay-Little Egg Harbor. (From Kennish and Fertig 2012).

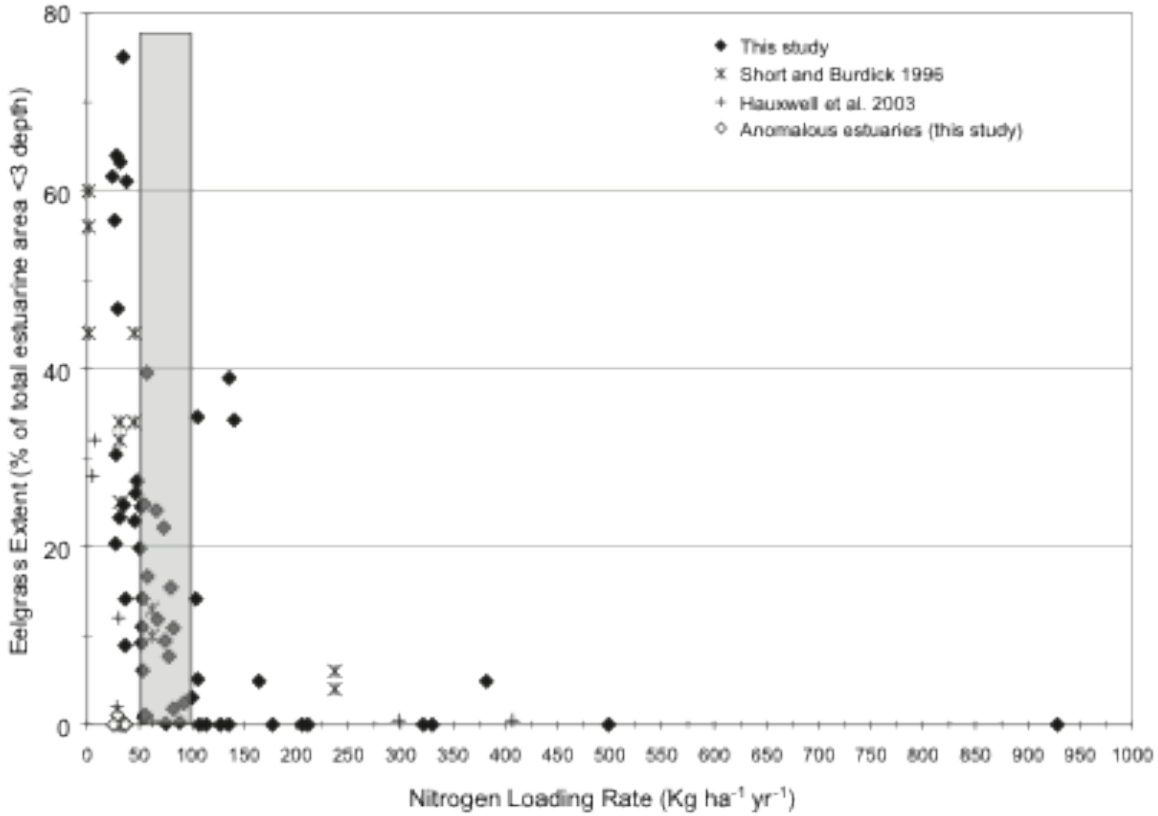


Fig. 2. Plot of eelgrass extent (percent of available habitat) vs. nitrogen loading rate (Kg N ha^{-1} yr^{-1}) (including other published values); gray bar is the nitrogen loading threshold range from the literature 50 - 100 Kg ha^{-1} yr^{-1}).

Figure 3 - 14 Eelgrass extent vs nitrogen loading in New England estuarine embayments (from Latimer and Rego 2010).

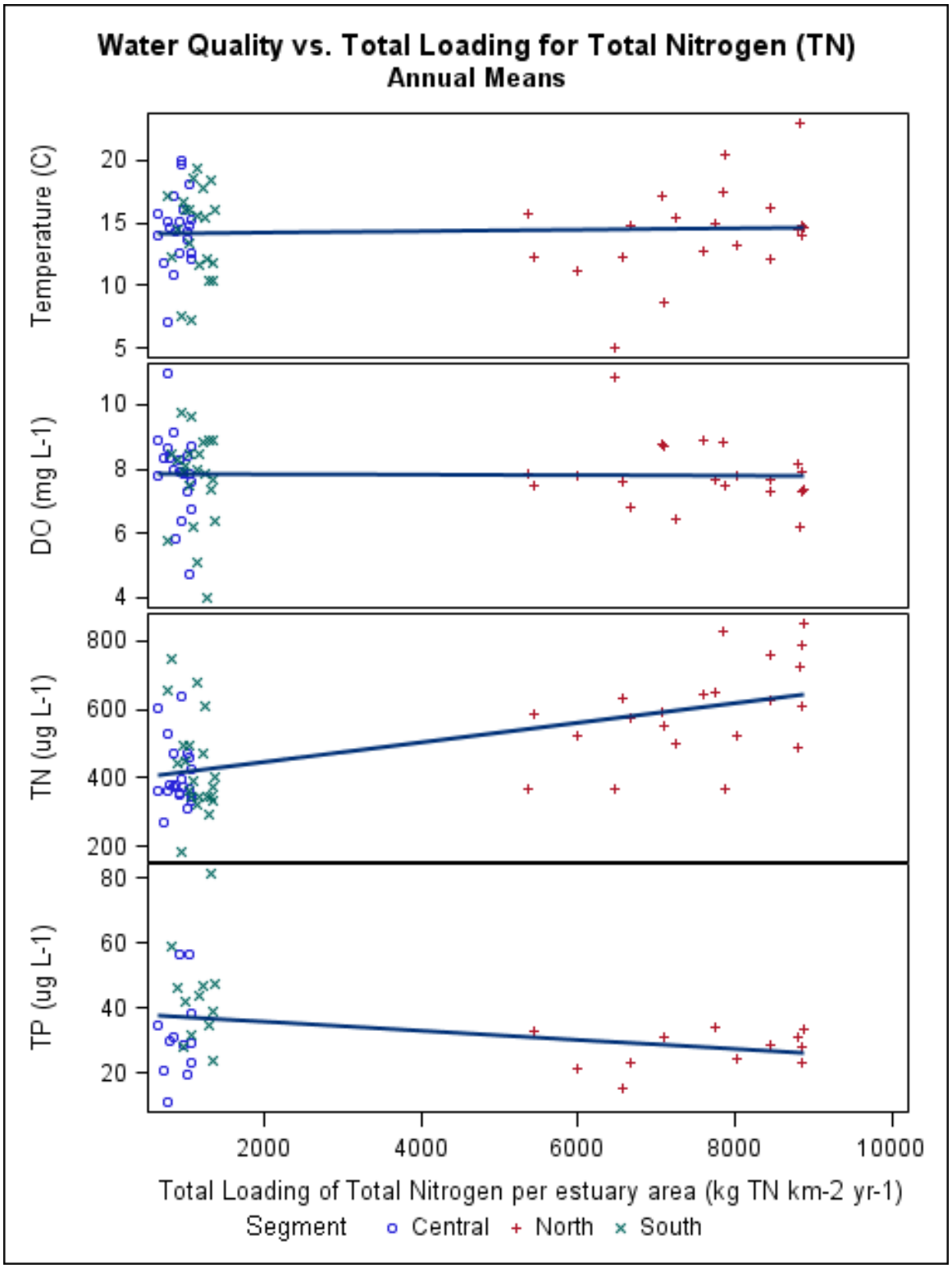


Figure 3 - 15 Water quality indicators vs. total nitrogen loading.

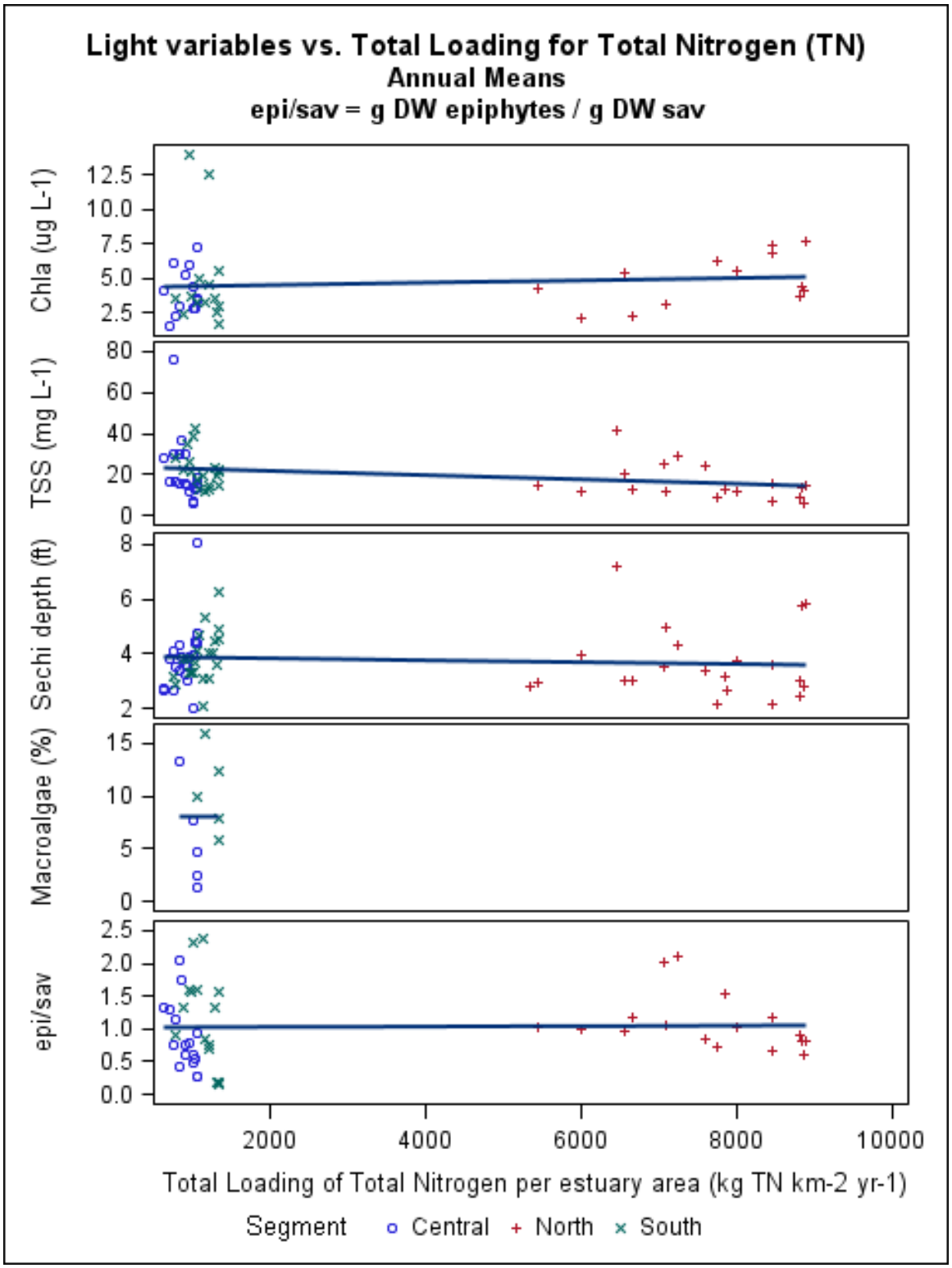


Figure 3 - 16 Light availability indicators vs. total nitrogen loading.

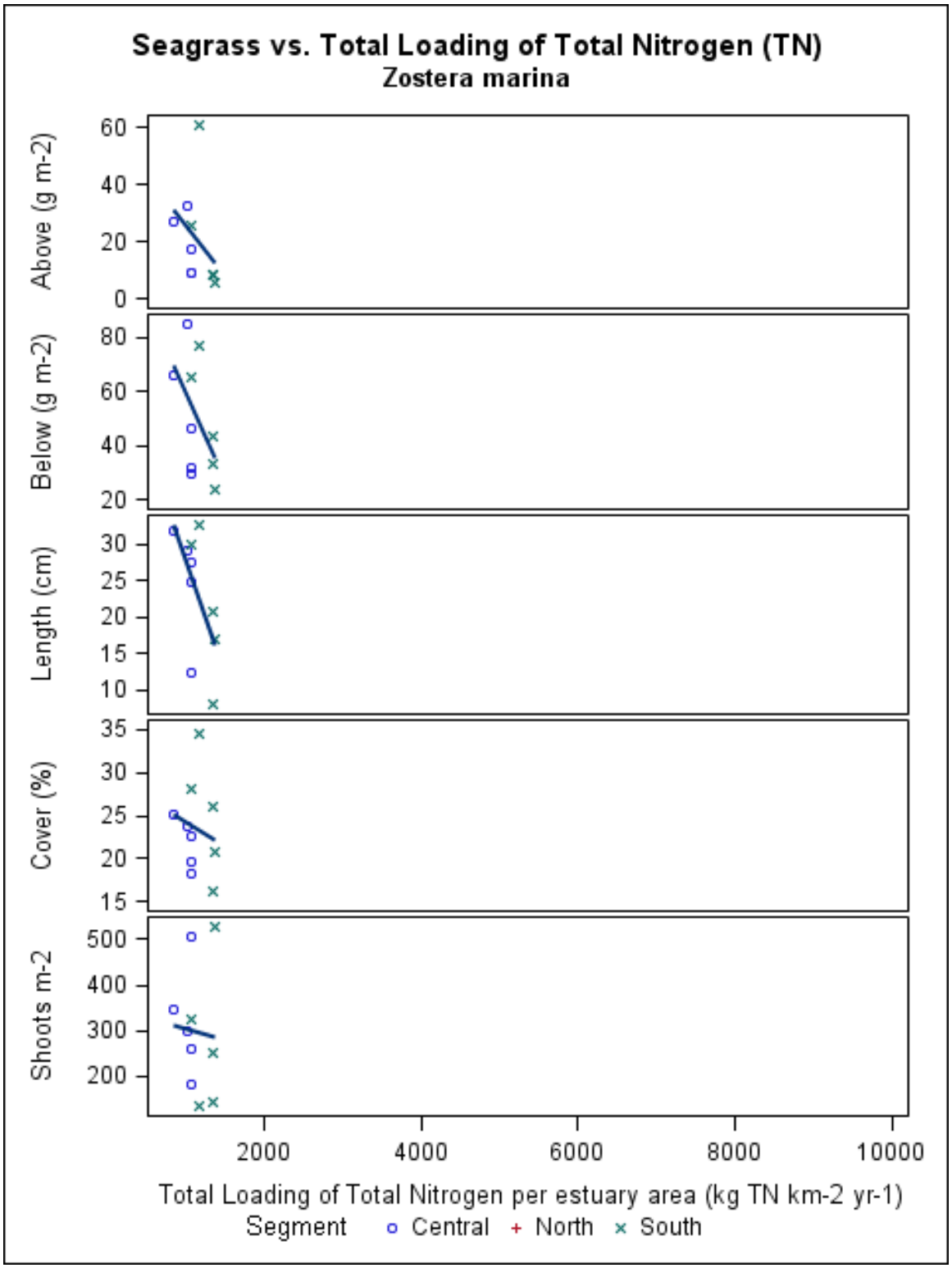


Figure 3 - 17 Seagrass indicators vs total nitrogen loading

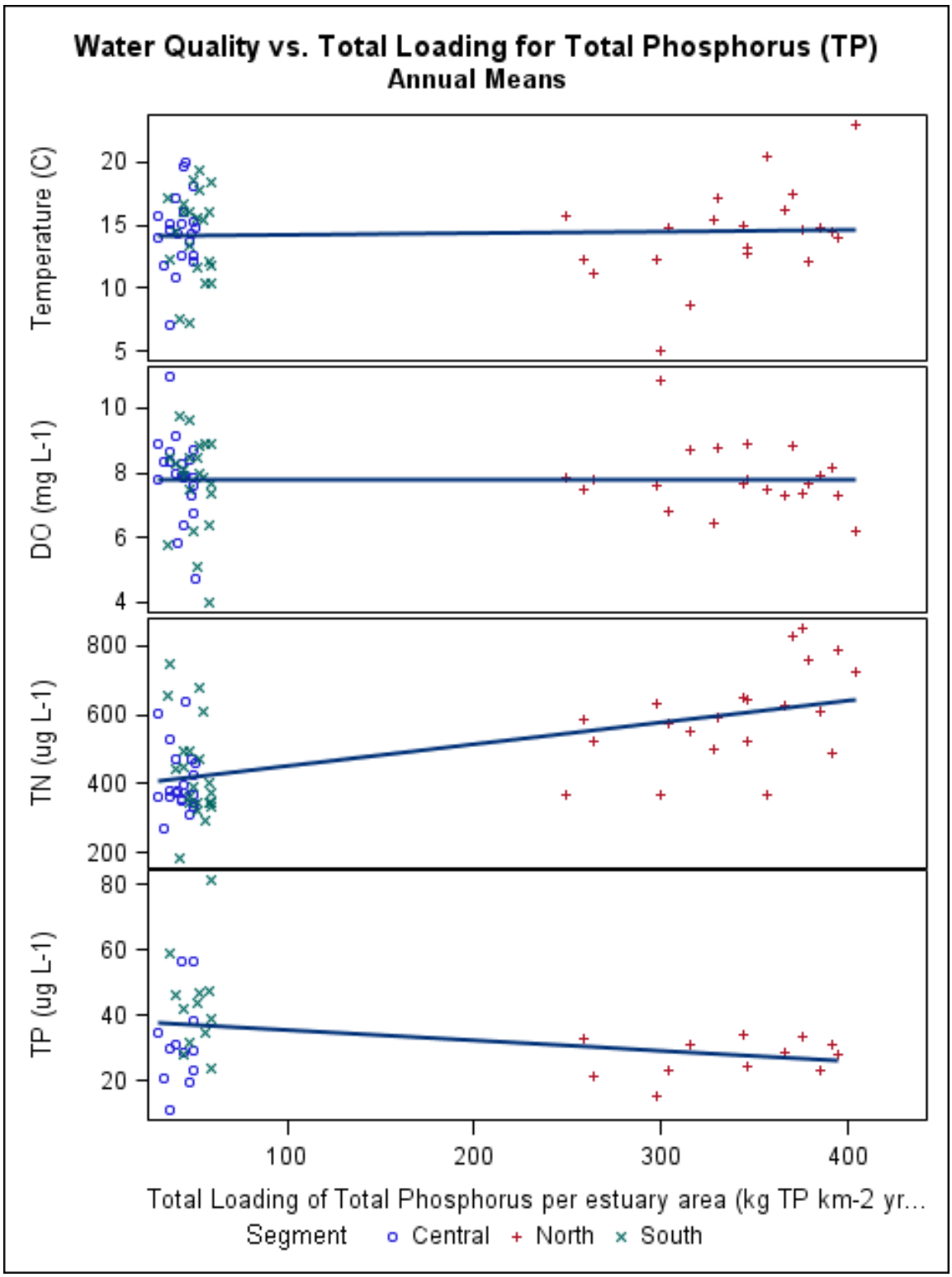


Figure 3 - 18 Water quality indicators vs. total phosphorus loading.

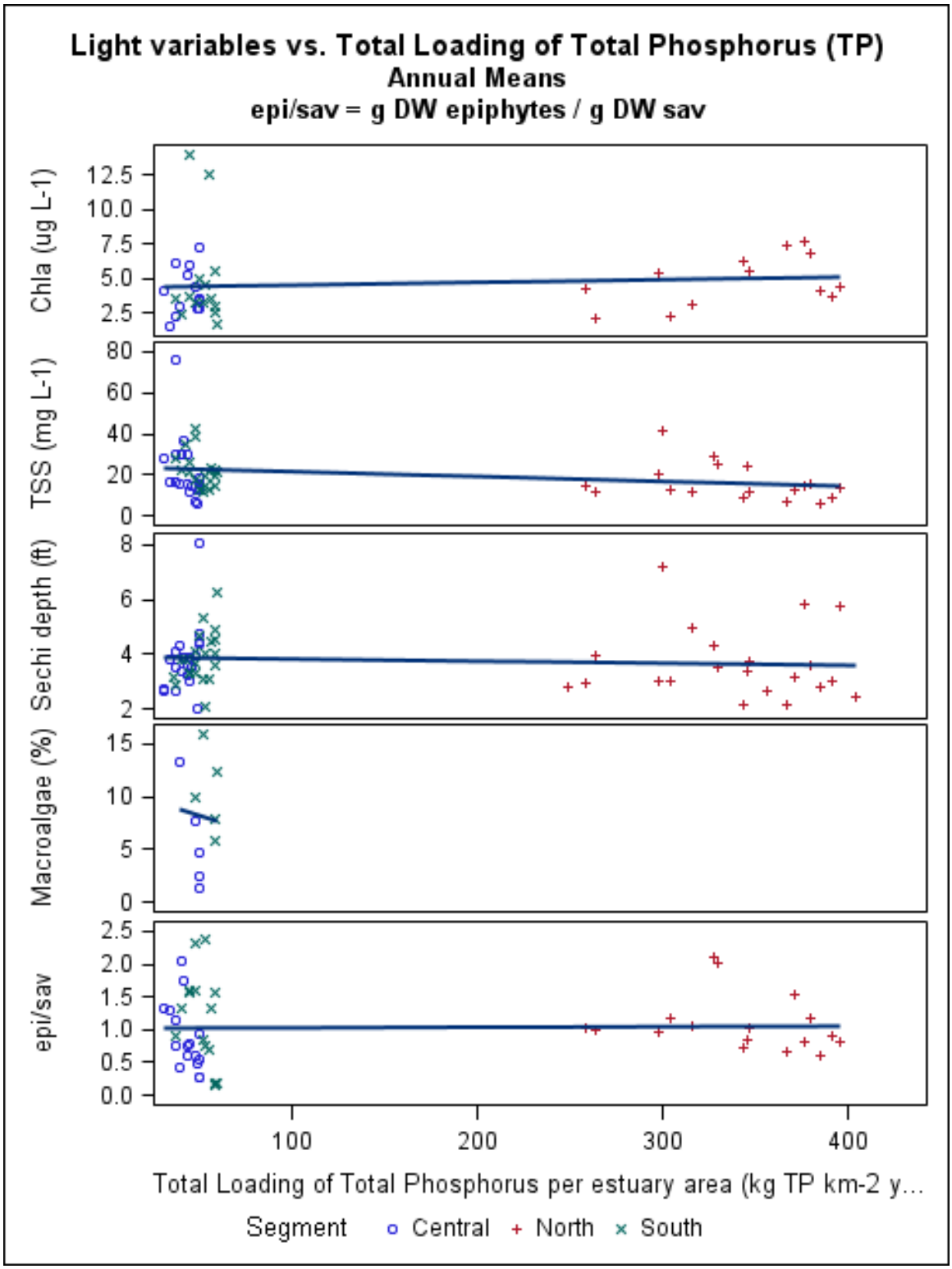


Figure 3 - 19 Light availability indicators vs. total phosphorus loading

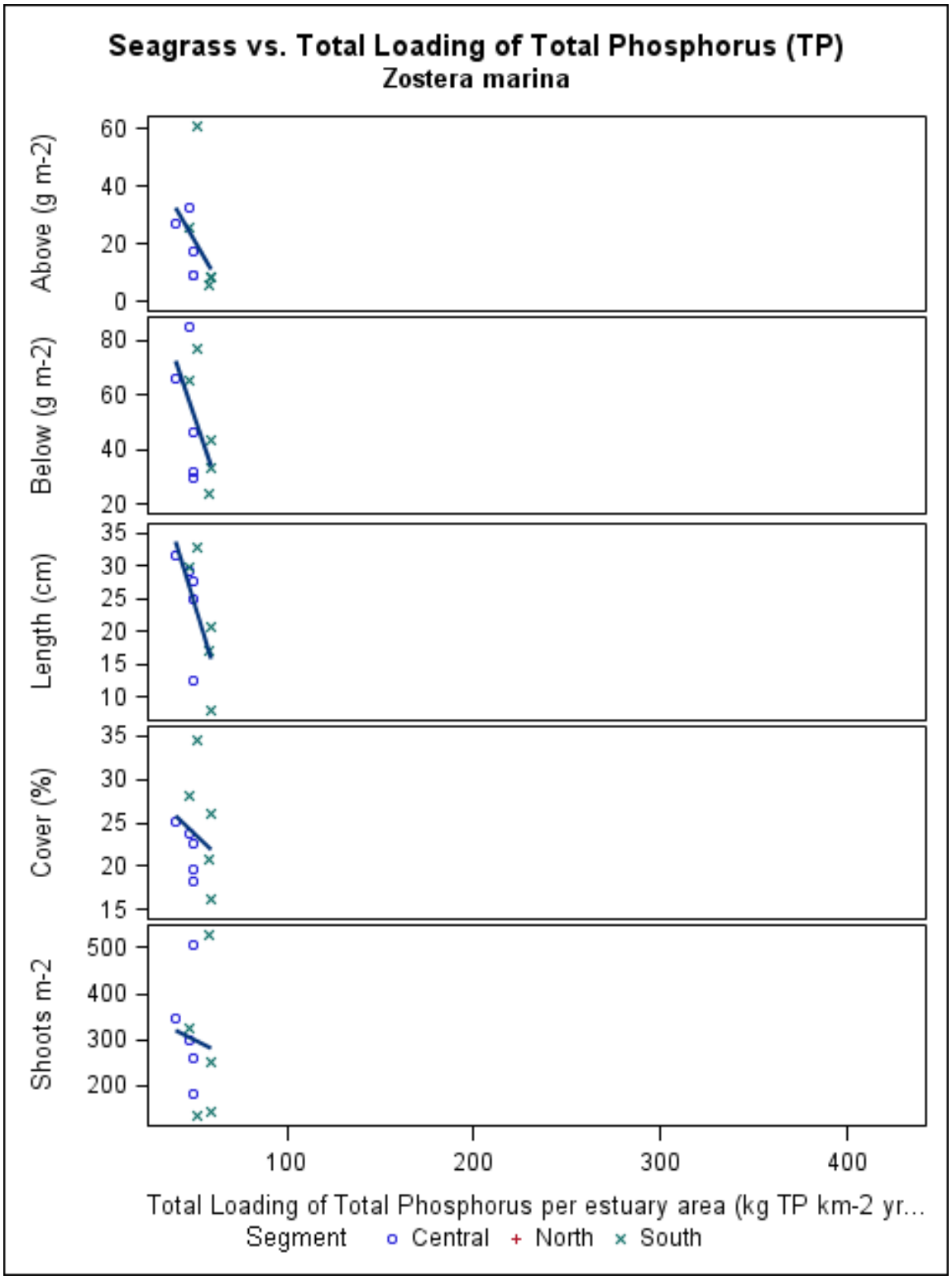


Figure 3 - 20 Seagrass indicators vs total phosphorus loading

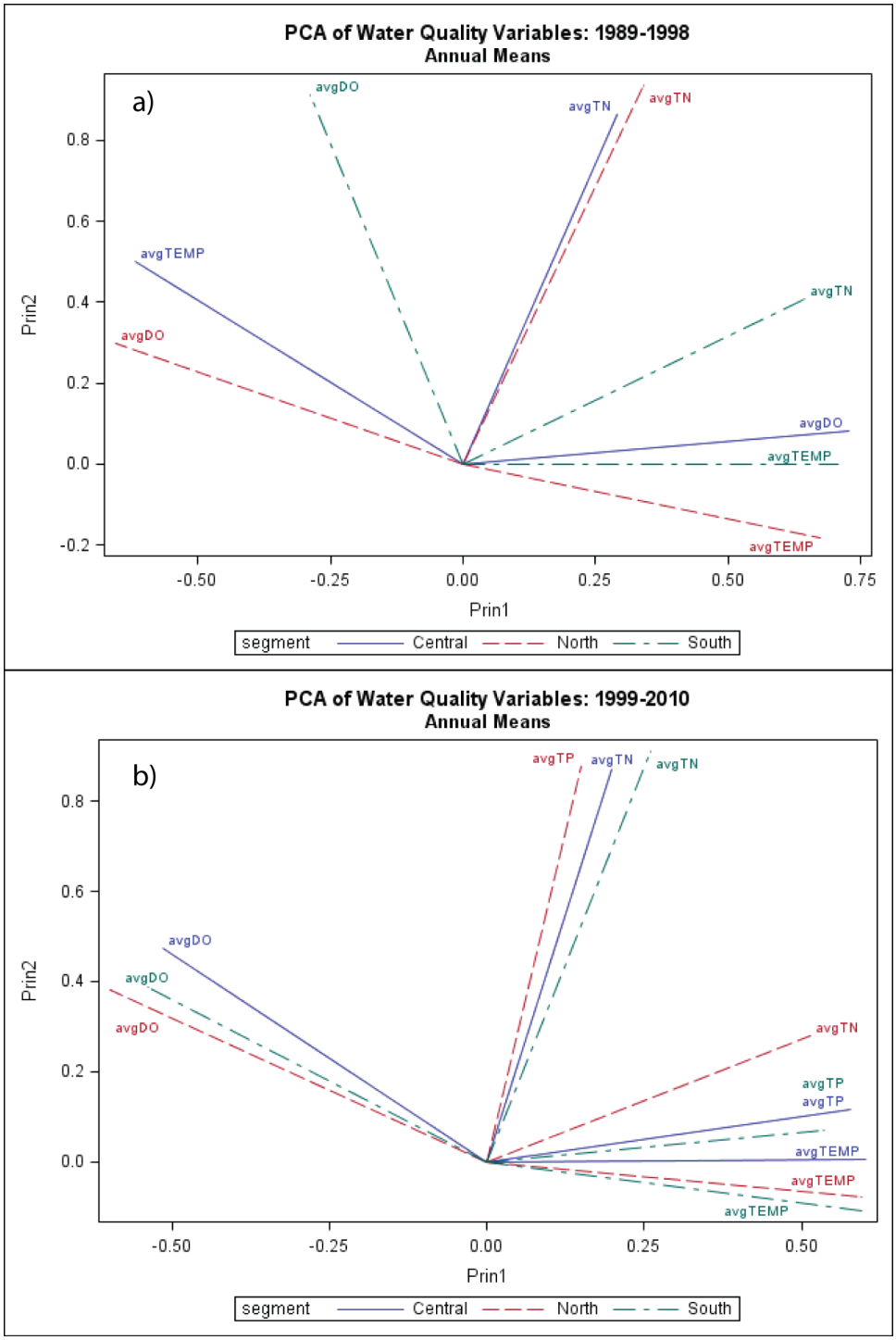


Figure 3 - 21 Principal component analysis of water quality variables (a) 1989-1998 and (b) 1999-2010

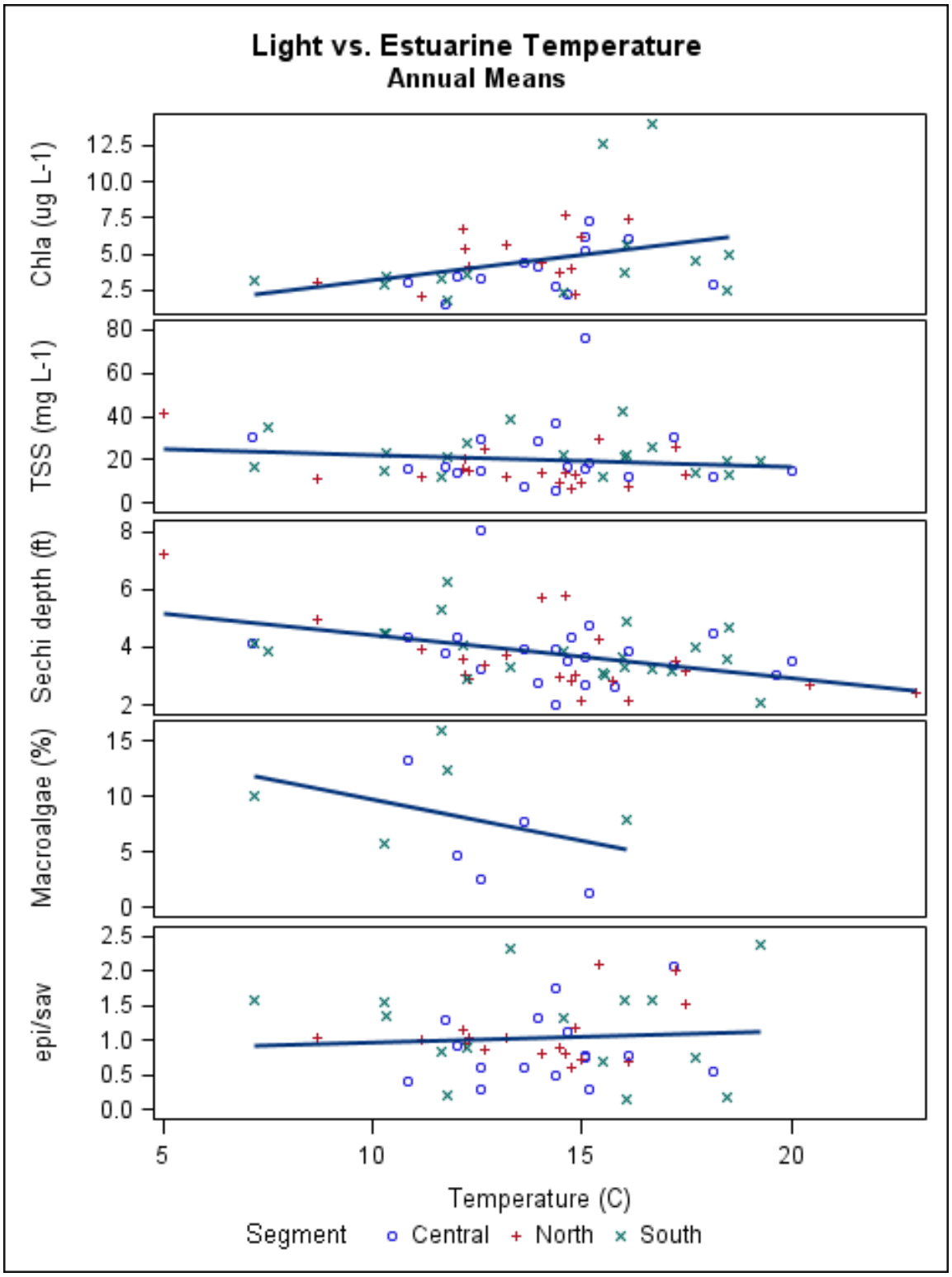


Figure 3 - 22 Light indicators vs estuarine temperature

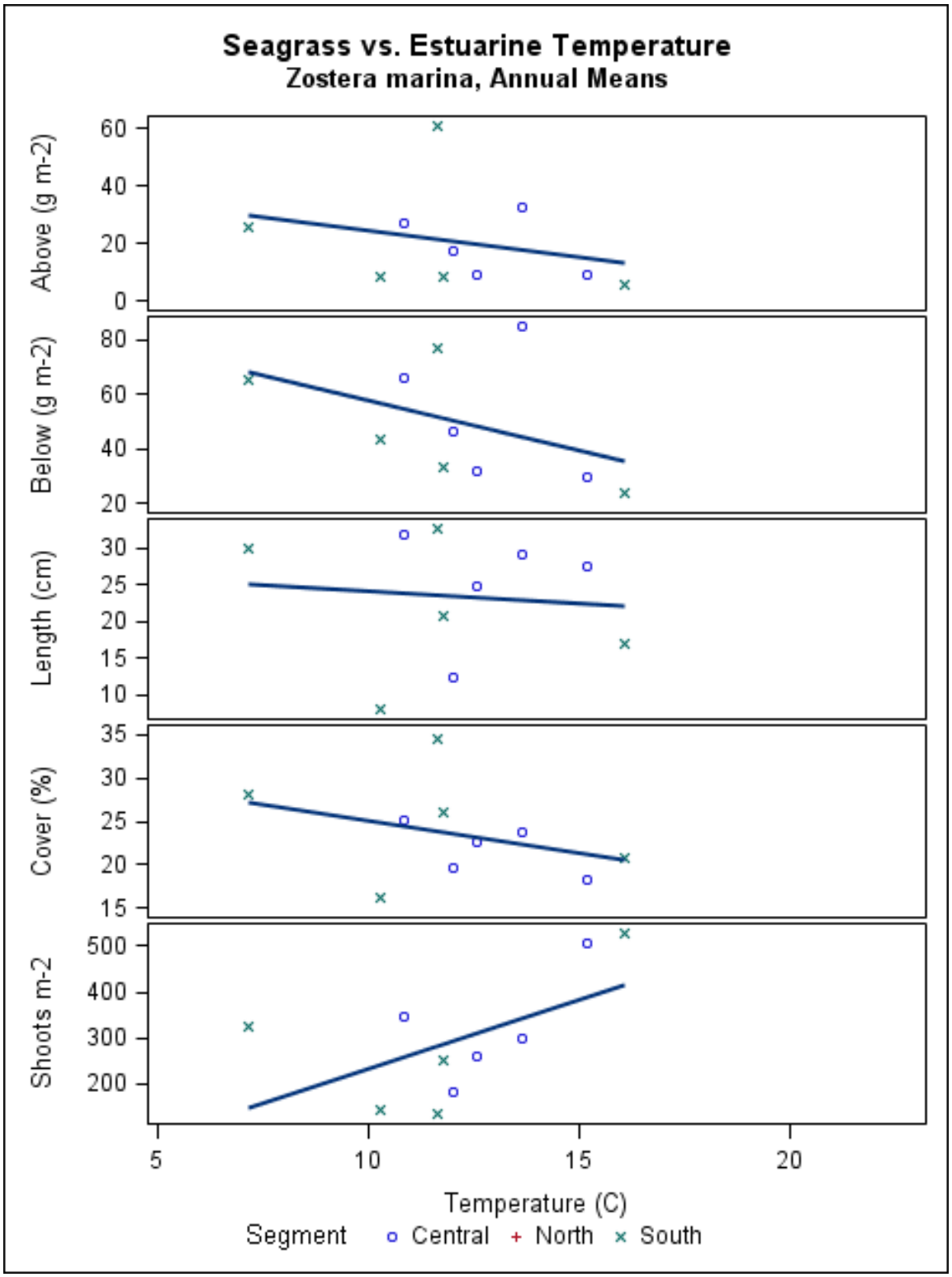


Figure 3 - 23 Seagrass indicators vs estuarine temperature

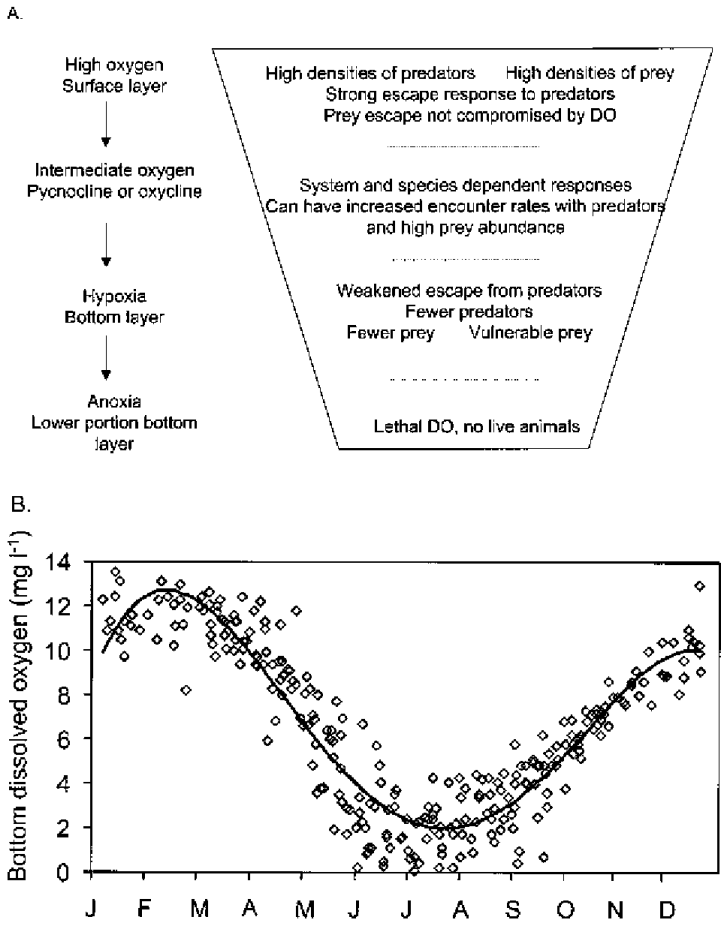


Fig. 1. Examples of spatial structure and temporal variability created by low dissolved oxygen in a temperate estuary. a) Vertical spatial structure and resultant spatial variation in direct and indirect effects of oxygen depletion. The figure shows the effect of oxygen concentration on various aspects of predator-prey interactions and abundances in different portions of a stratified water column with bottom-layer hypoxia. b) Variation in daytime bottom layer dissolved oxygen at a mesohaline site in the Patuxent River. Data show dissolved oxygen concentrations mid-channel near Broomes Island, Maryland. Variation around the fitted trend line (4th order polynomial, $r^2 = 0.87$) represent both within and among year variability in the 15-yr data set. Bottom dissolved oxygen concentrations can change rapidly as stratification is disrupted by storm-associated wind mixing. Data are from the Chesapeake Bay Program monitoring program (www.chesapeake.net).

Figure 3 - 24 Seasonal patterns of bottom dissolved oxygen in a degraded and stratified mesohaline estuary (from Breitburg 2002)

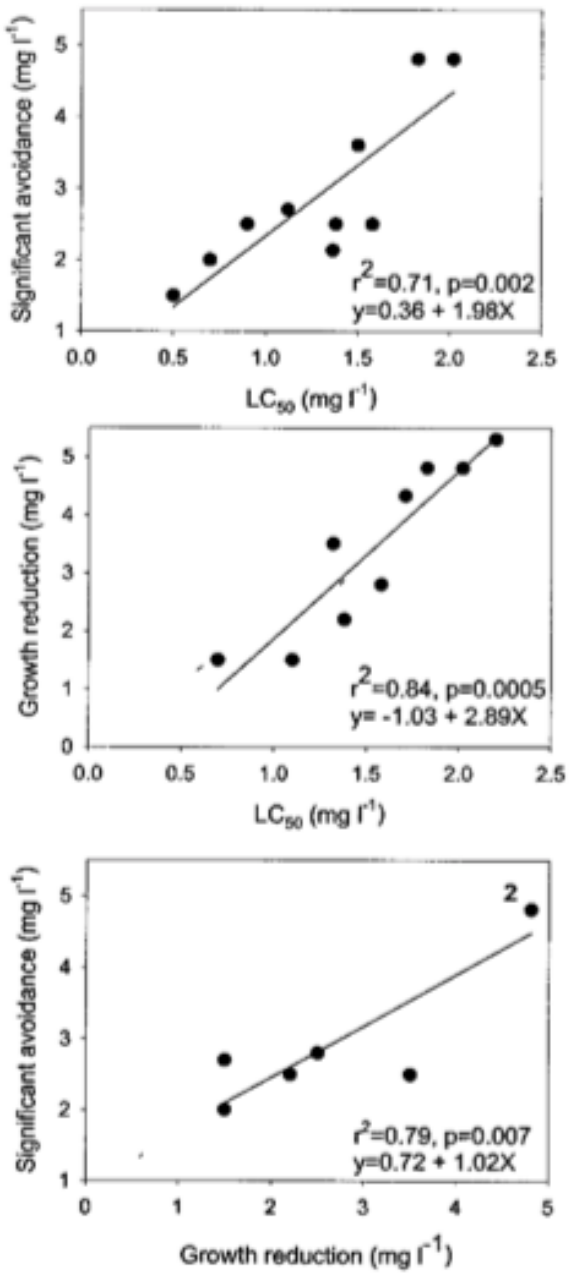


Fig. 3. Relationship between lethal dissolved oxygen concentrations and those resulting in reduced growth and behavioral avoidance of affected habitat. Top) LC_{50} versus avoidance behavior, middle) LC_{50} versus growth reduction, and bottom) growth versus avoidance behavior. Two identical points in bottom panel are indicated by the number 2 next to the data point. Data sources are as follows. Avoidance versus mortality: Barton et al. 1980; Coutant 1985; Petersen and Petersen 1990; Pihl et al. 1991a,b; Scholz and Waller 1992; Schurman and Steffensen 1992; Howell and Simpson 1994; Petersen and Pihl 1995; Wannamaker and Rice 2000; U.S. Environmental Protection Agency

Figure 3 - 25 Lethality effects of decreasing oxygen reaching hypoxic (2 mg L-1) and anoxic (0 mg L-1) conditions (from Breitburg 2002)

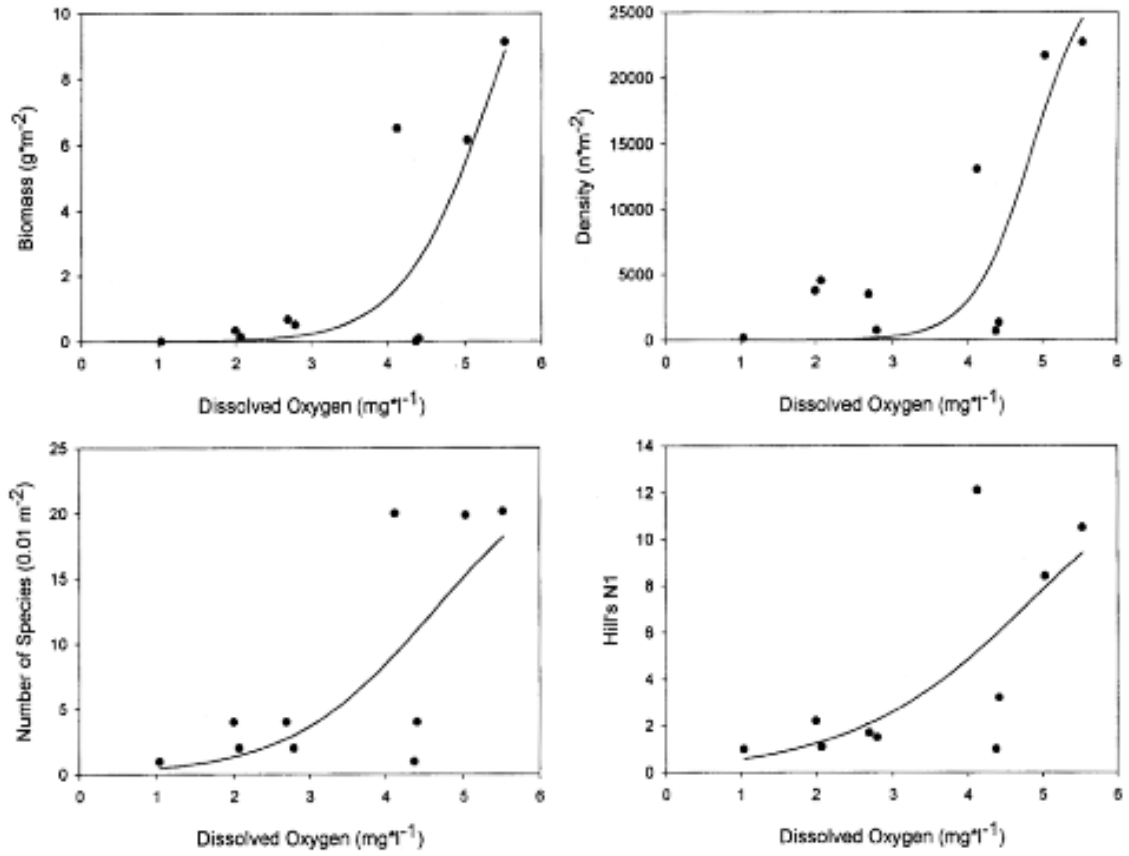


Fig. 9. Mathematical logistic models describing how dissolved oxygen concentrations influence characteristics of benthic communities: biomass, density, number of species, and Hill's Number. Model parameters as in Table 5.

Figure 3 - 26 Effects of decreasing dissolved oxygen on biomass and diversity of benthic communities (from Ritter and Montagna 1999)

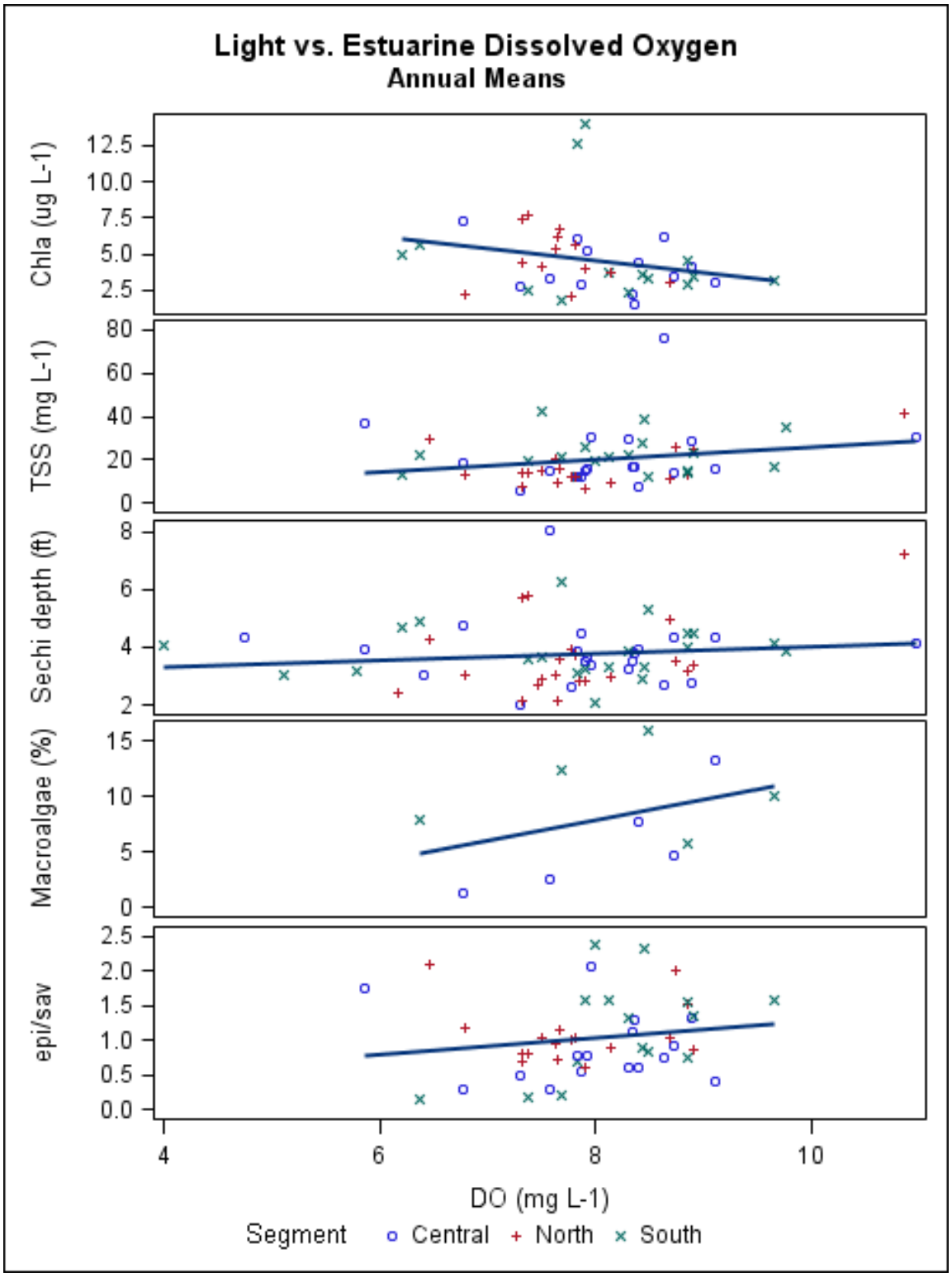


Figure 3 - 27 Light vs estuarine dissolved oxygen

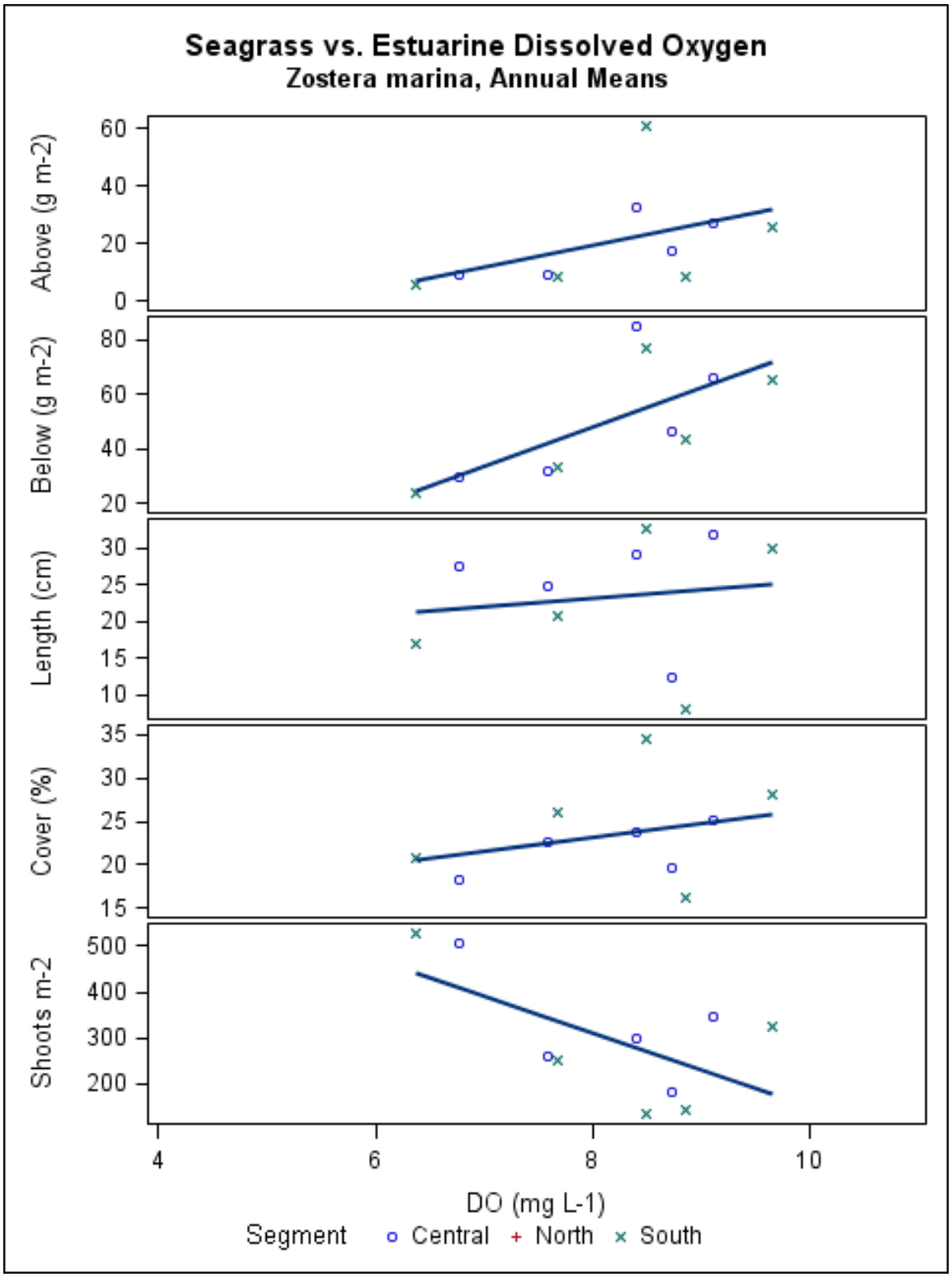


Figure 3 - 28 Seagrass indicators vs dissolved oxygen

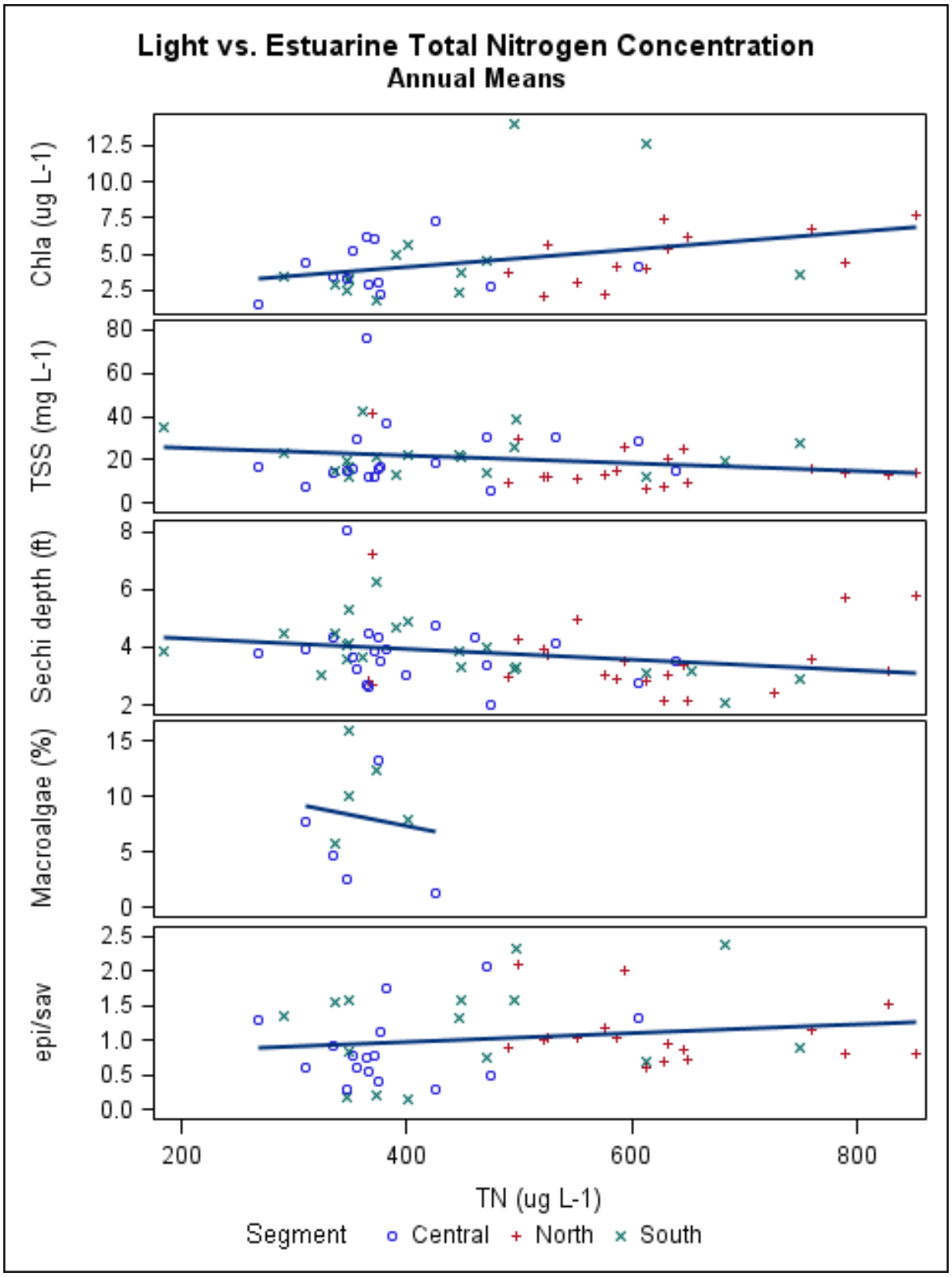


Figure 3 - 29 Light indicators vs estuarine total nitrogen concentration

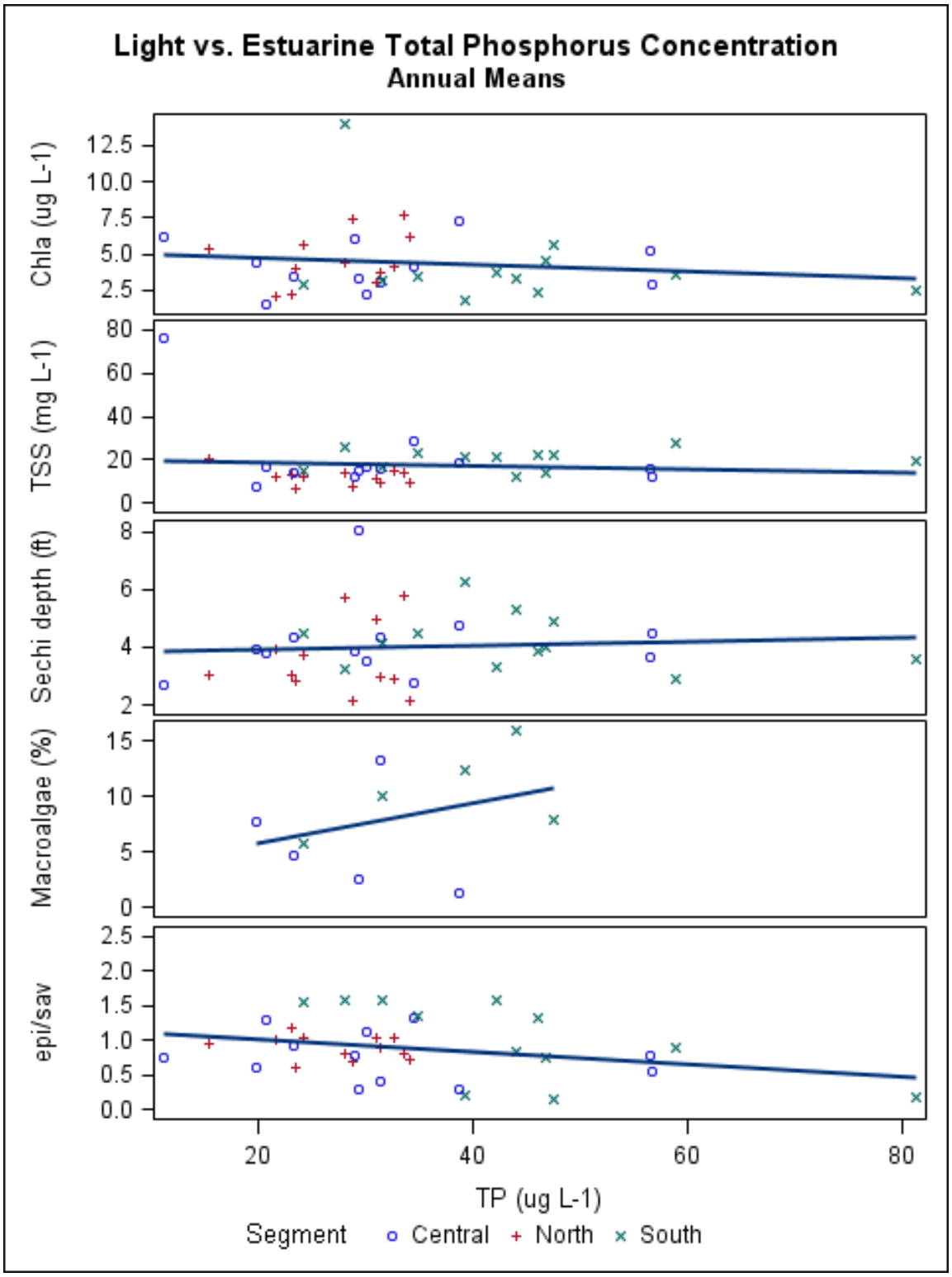


Figure 3 - 30 Light indicators vs estuarine total phosphorus concentration

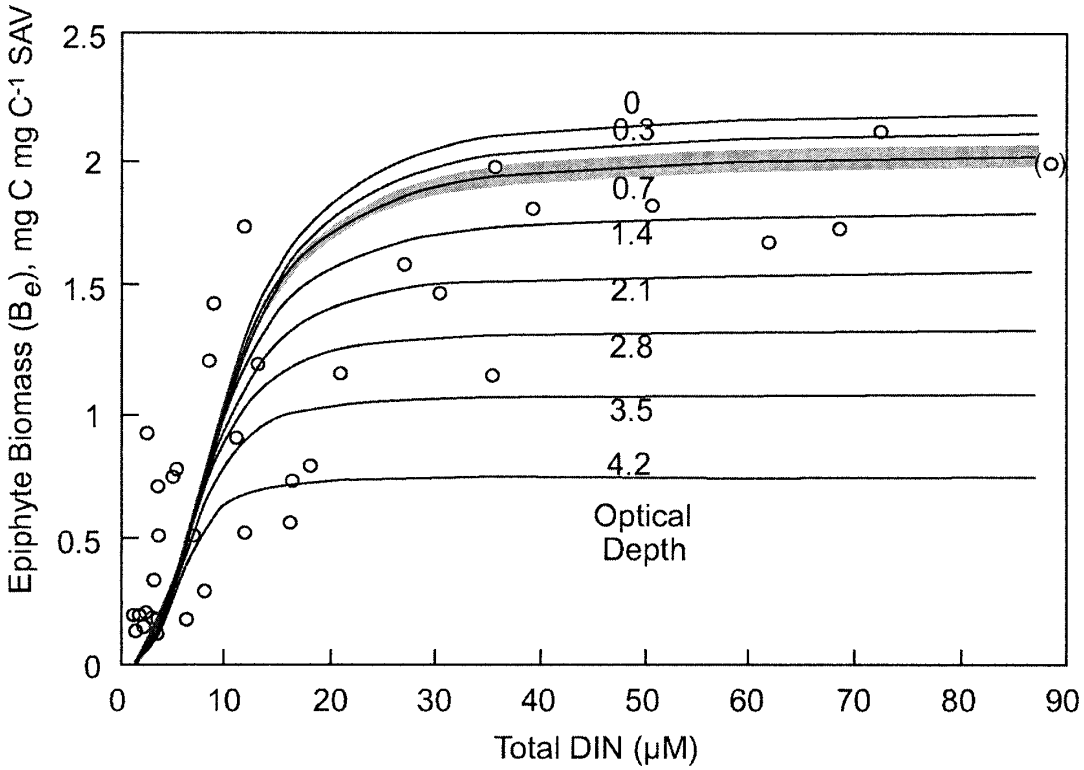


Fig. 5. Calculated responses of epiphytic algal biomass (B_e, mg C/mg C SAV) to changes in dissolved inorganic nitrogen (DIN, µM) concentration under various light (PAR, µE m^{-2} s^{-1}) conditions in estuarine waters. Each curve is described by the equation, and each represents computed response under 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 ($OD^{1.23}$)] and $K_{N(OD)}$ = 2.32(1 − 0.031$OD^{1.42}$)$^{-1}$), 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 - 31 Relationship of epiphyte biomass to nitrogen concentrations under different light regimes. (From Kemp et al. 2004).

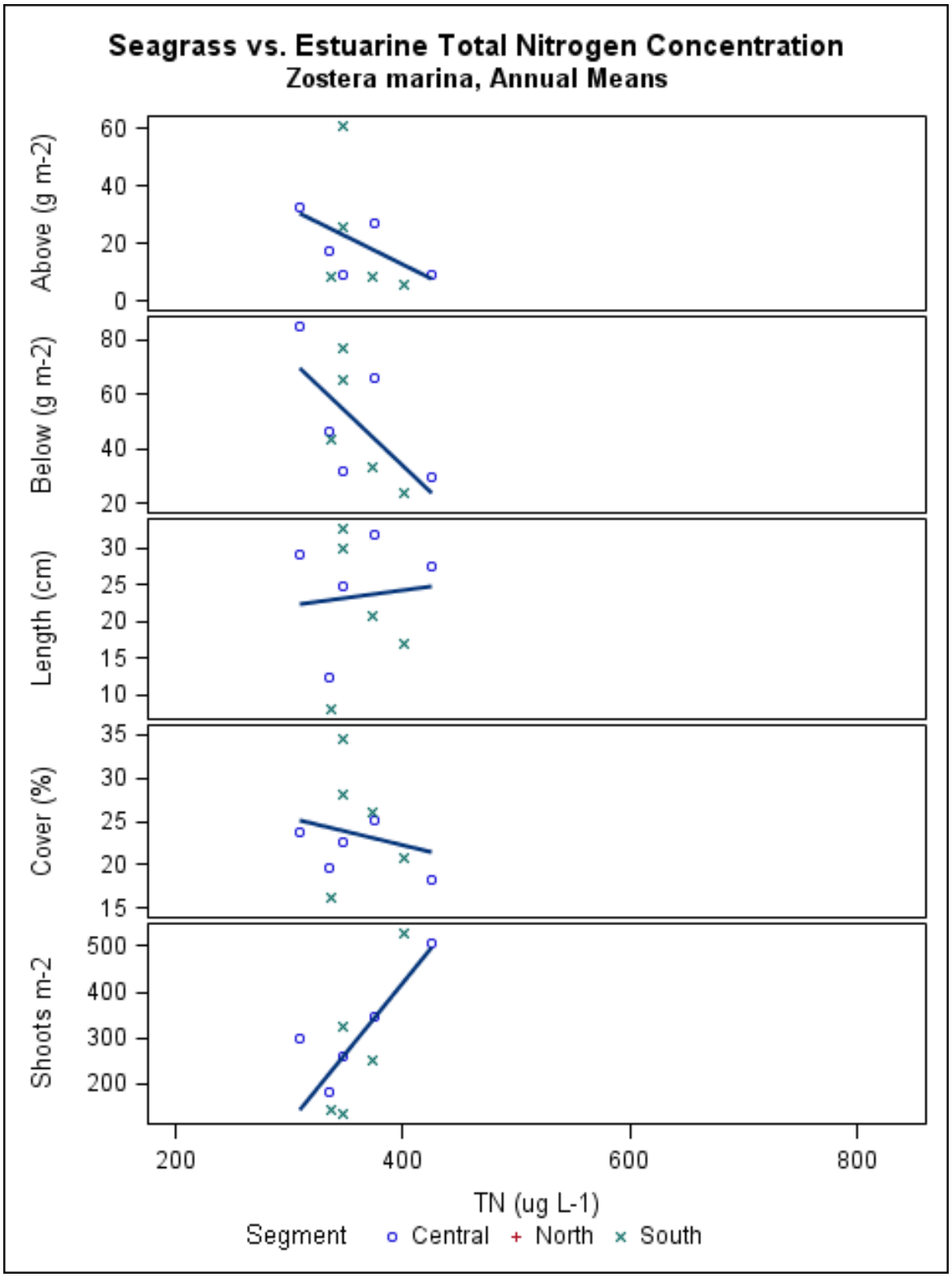


Figure 3 - 32 Seagrass indicators vs estuarine total nitrogen

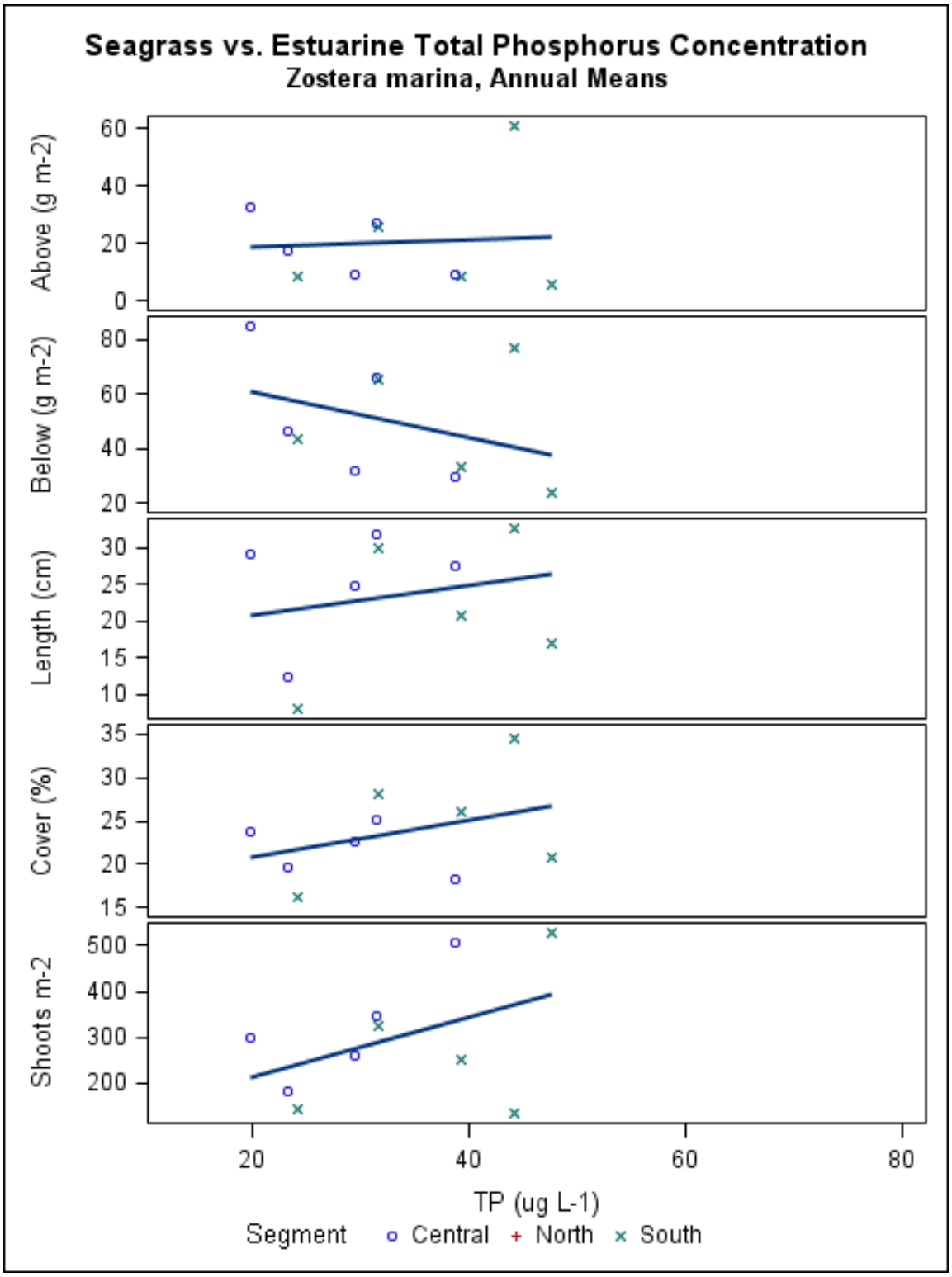


Figure 3 - 33 Seagrass indicators vs estuarine total phosphorus

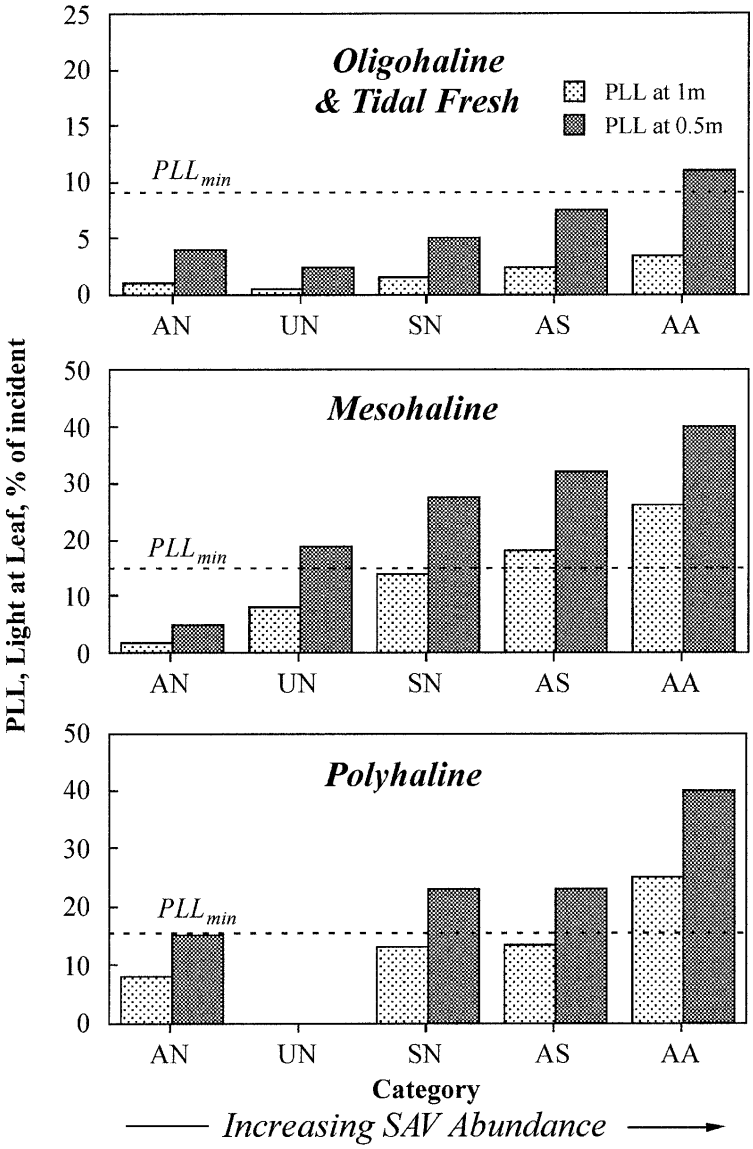


Fig. 8. Percent of surface light at SAV leaves (PLL) calculated using growing season median water quality data collected in Chesapeake Bay monitoring program at stations throughout the estuary compared to relative abundance of SAV in adjacent shallows in 1985–1996. PLL is calculated for water column 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 - 34 Relationship between light availability and seagrass abundance. (From Kemp et al. 2004).

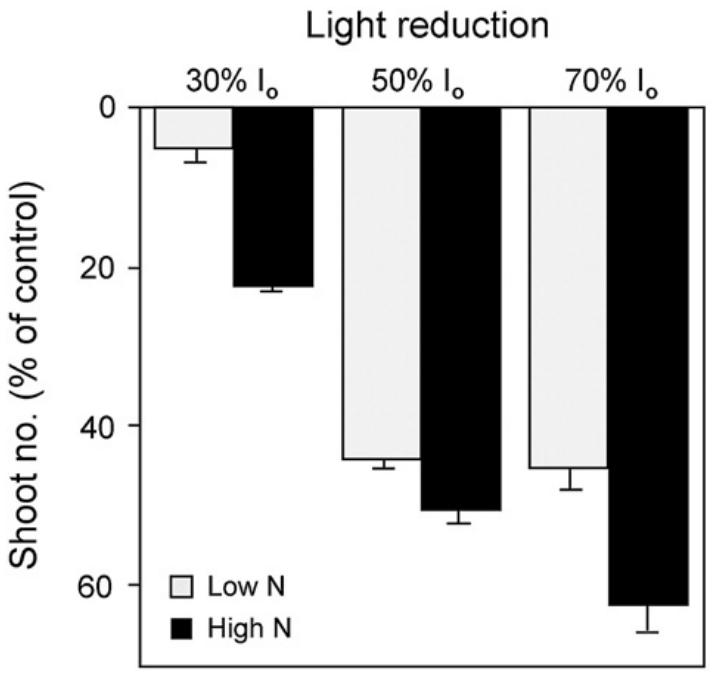


Fig. 5. The effects of water-column nitrate enrichment and light reduction on shoot production of the seagrass, *Zostera marina*. From first author's outdoor mesocosm experiments, indicated as the percent decrease from shoot production of control plants that did not receive water-column nitrate additions 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 - 35 Impact of nitrate enrichment and light reduction on seagrass shoot production. (From Burkholder 2001).

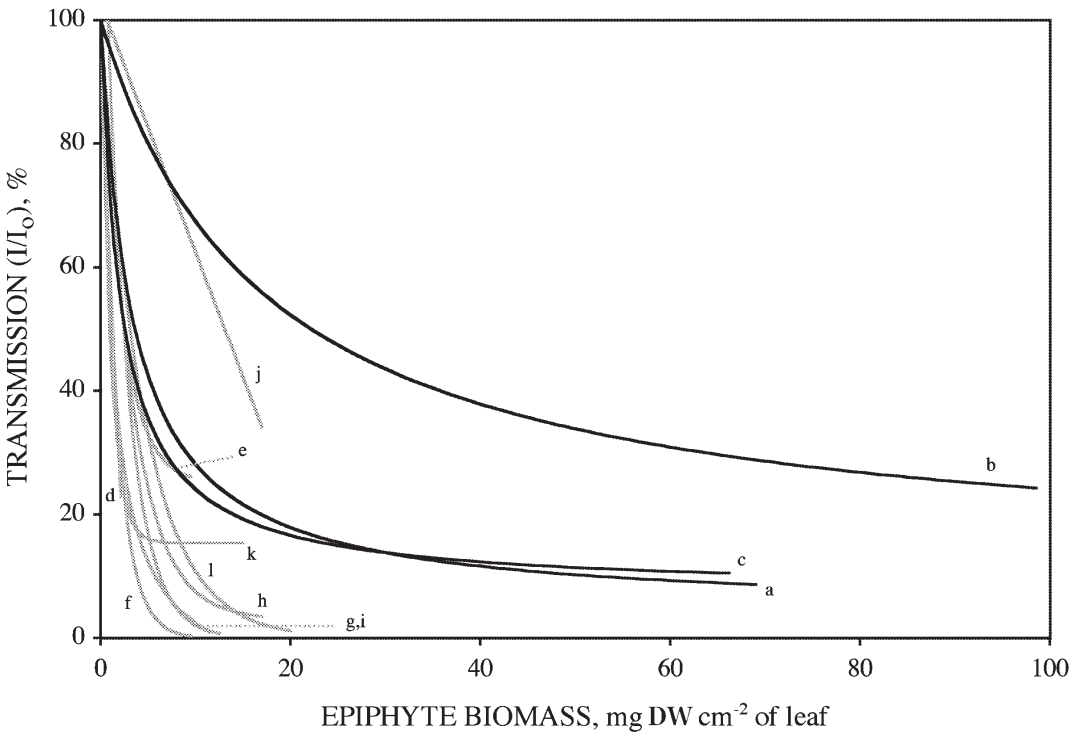


Fig. 2. Comparison of relationships between light transmssion and epiphyte density in this and previous studies. Regression lines for present study are black, and denoted as follows: a, unidentified green; b, *Cladophora* sp.; c, *Polysiphornia* sp. Regression lines for past studies are gray, and are 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 - 36 Light attenuation due to epiphytes (from Brush and Nixon 2002).

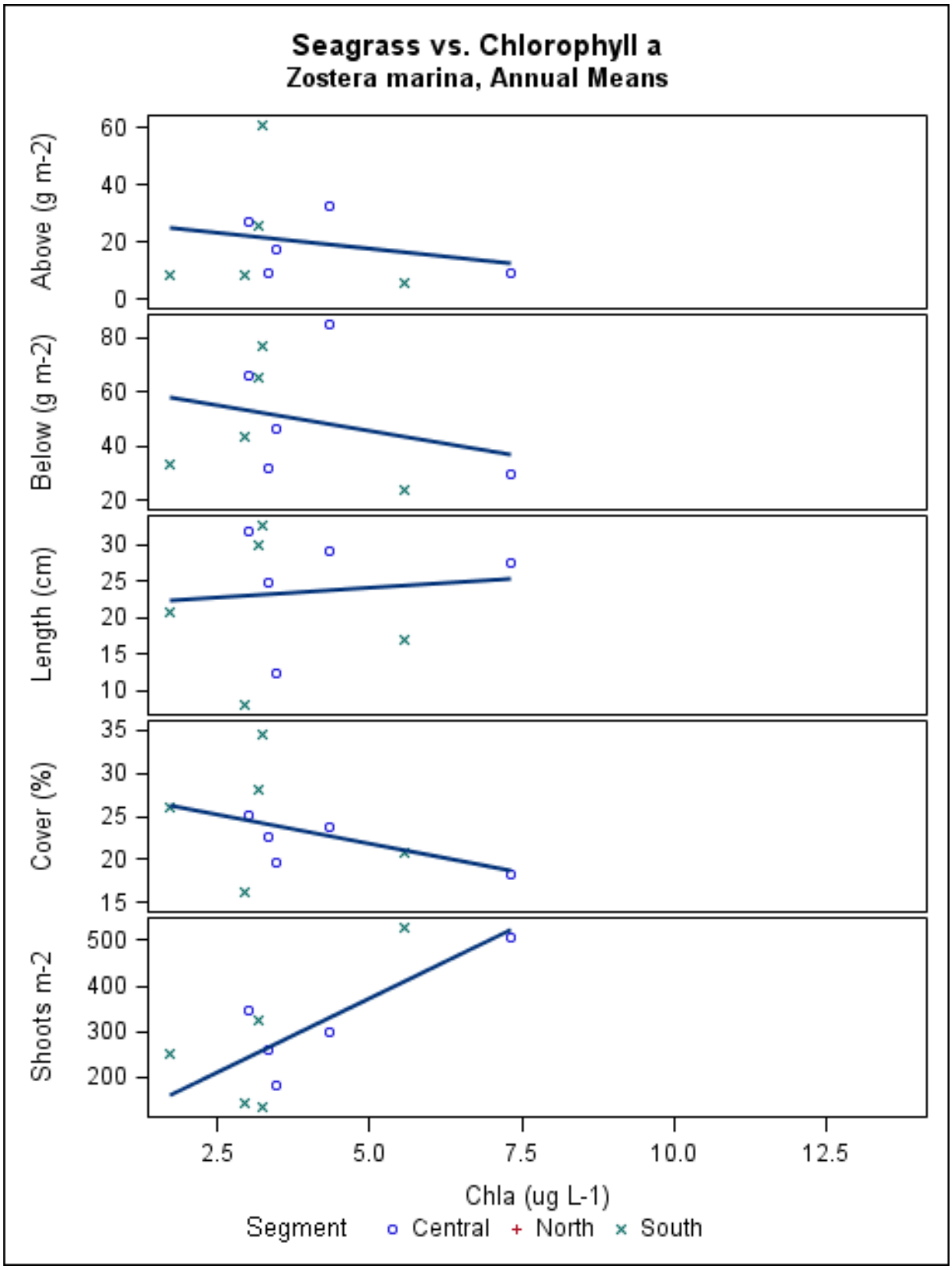


Figure 3 - 37 Seagrass indicator response to chlorophyll a.

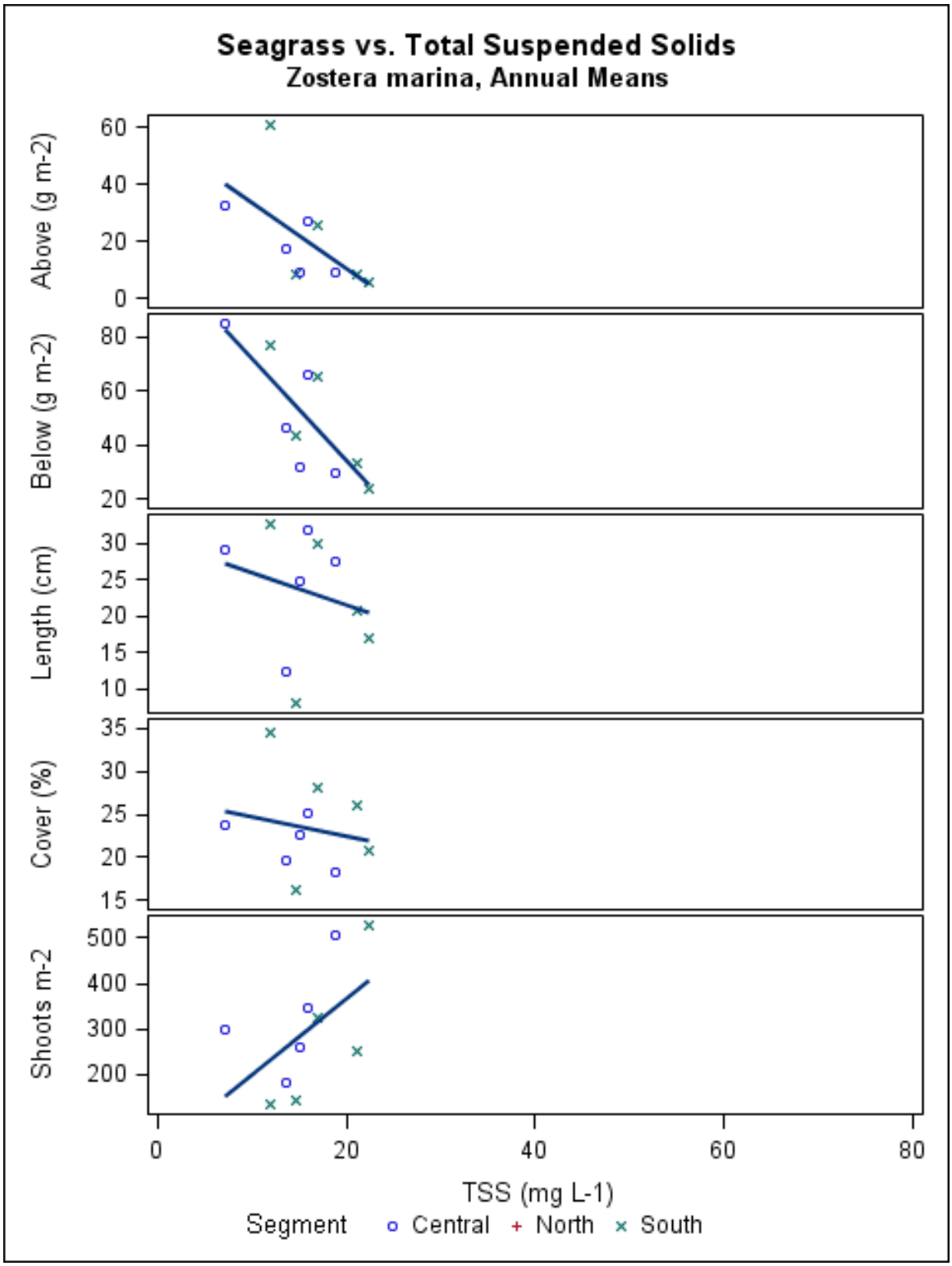


Figure 3 - 38 Seagrass indicator response to total suspended solids.

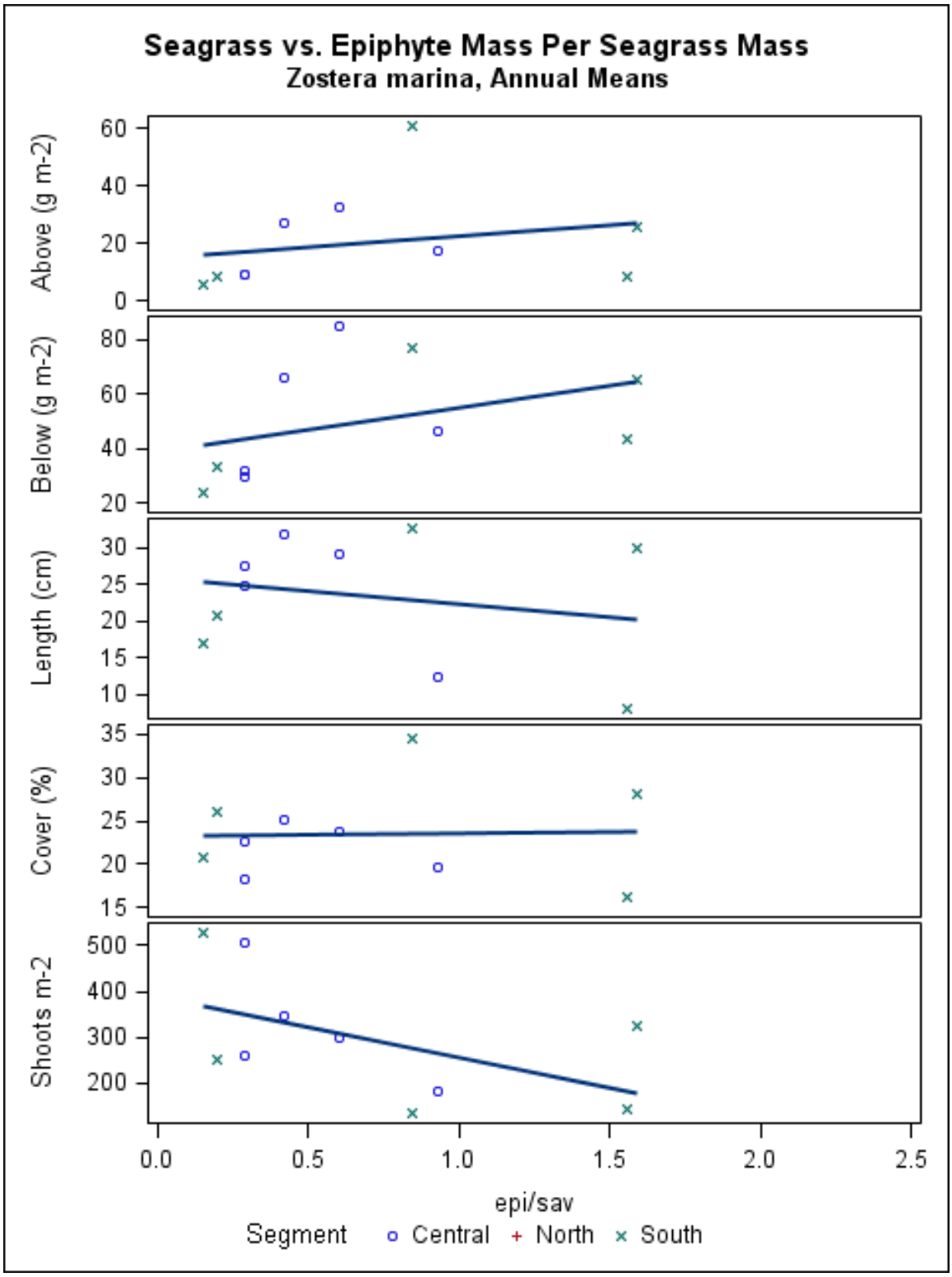


Figure 3 - 39 Seagrass indicators vs ratio of epiphyte to seagrass biomass.

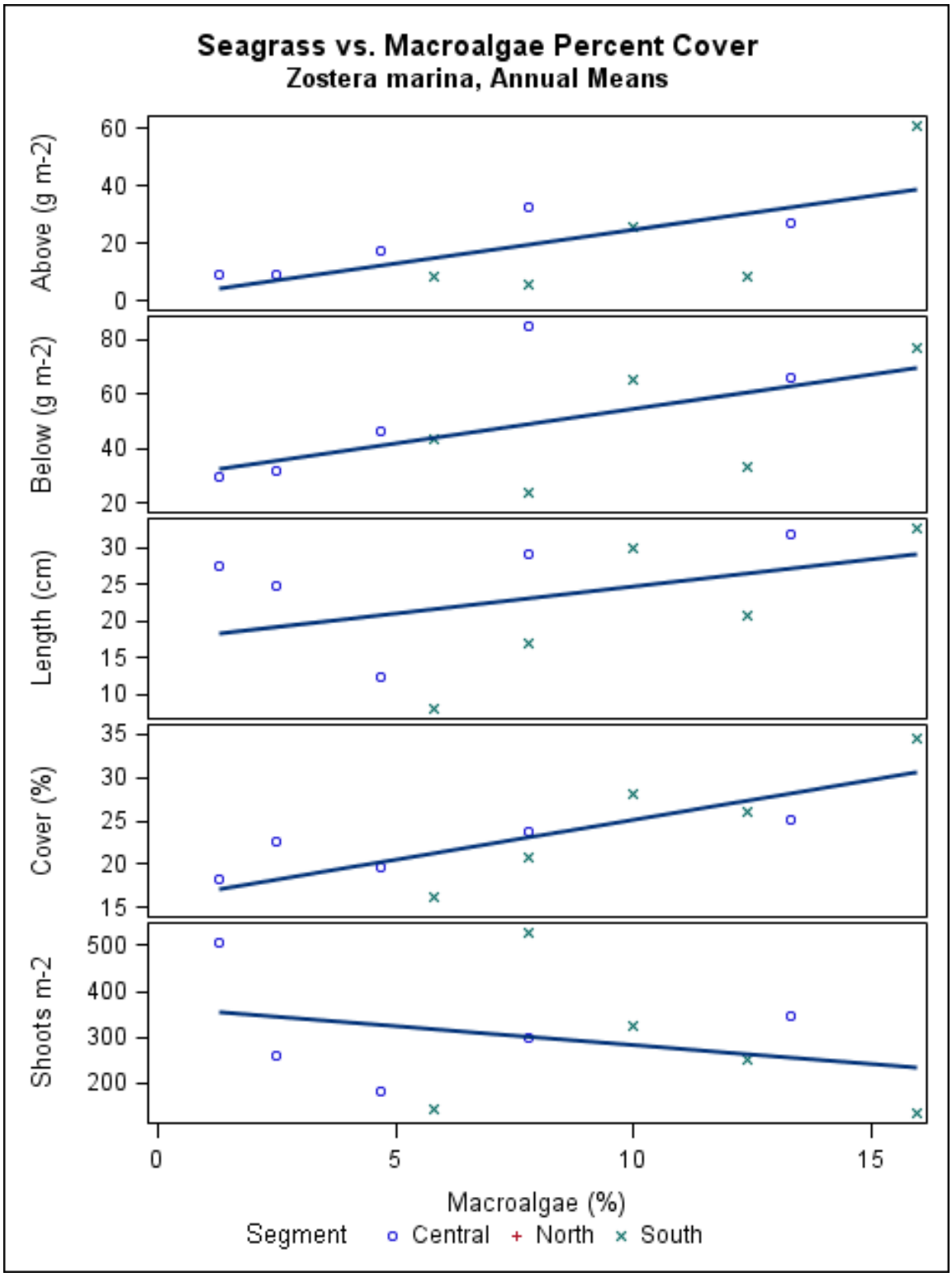


Figure 3 - 40 Seagrass indicators vs macroalgae percent cover

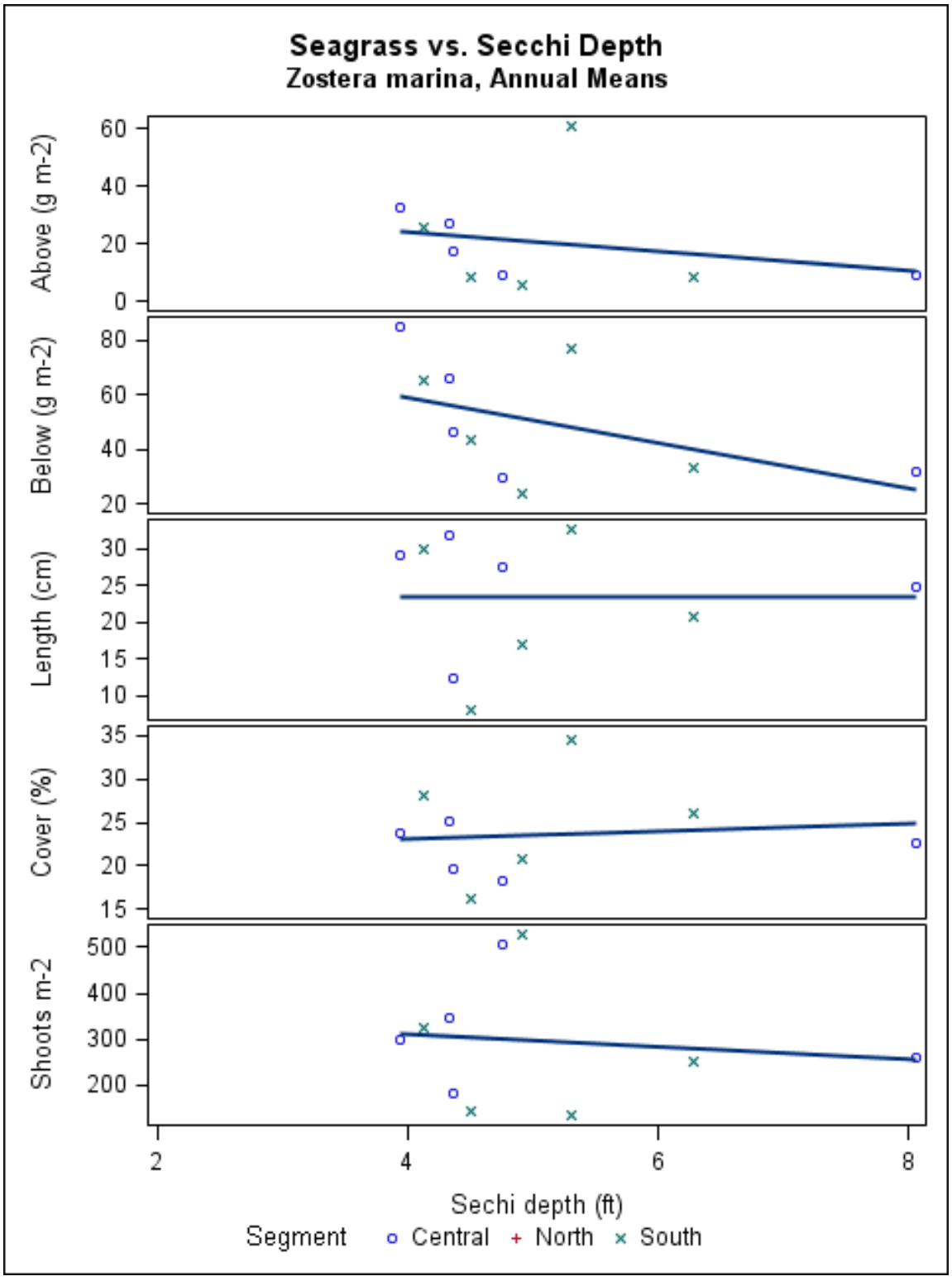


Figure 3 - 41 Seagrass indicators vs Secchi depth

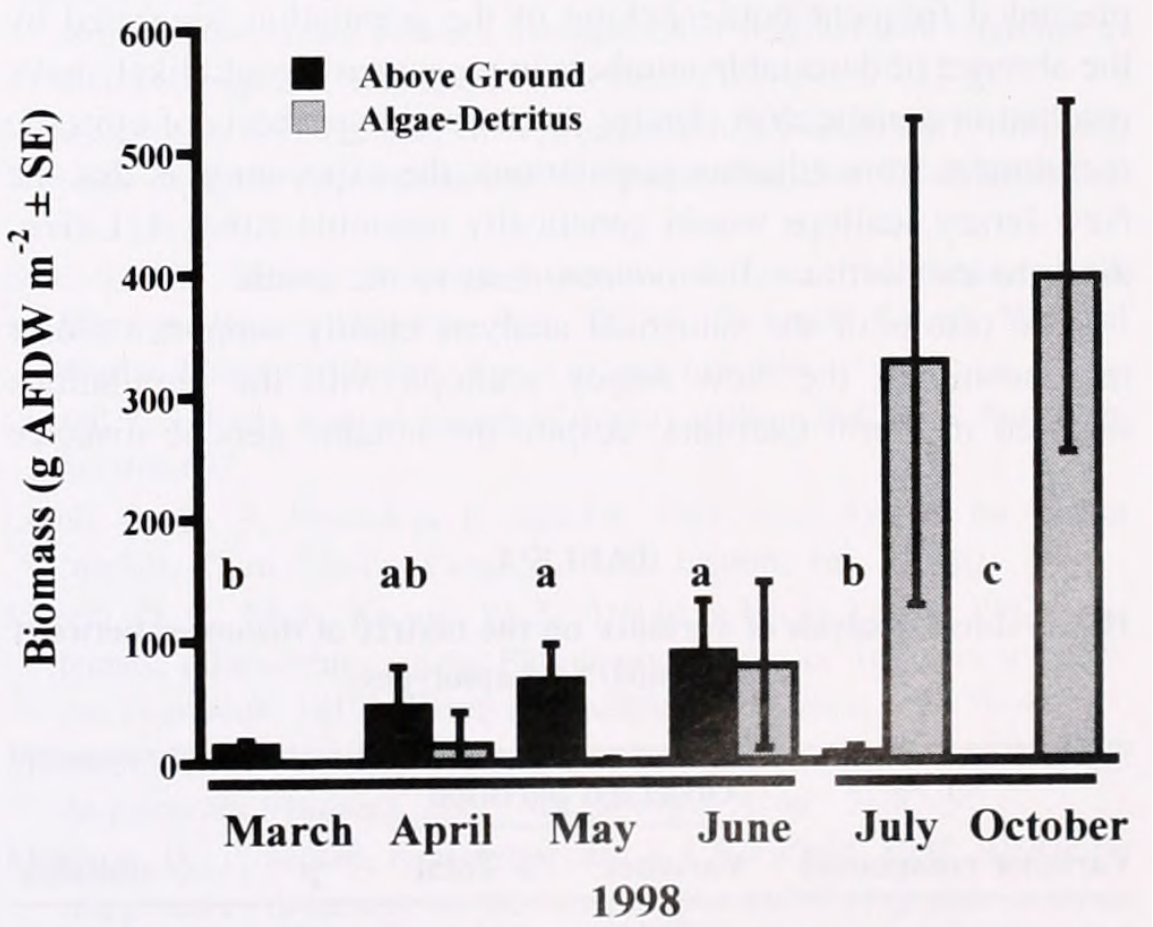


Figure 4. Seasonal distribution of *Zostera marina* above ground biomass and algal-detrital biomass from Little Egg Harbor, NJ during 1998. Values represent mean biomass expressed as gram ash free dry weight (g AFDW) per square meter ± Standard Error. Differing letters above bars represent significant differences in above ground *Z. marina* biomass among means for dates of collection ($P < 0.05$). Separated lines represent significant differences in Algae-Detritus biomass among months of collection.

Figure 3 - 42 Eelgrass biomass data during 1998 (from Bologna 2001)

Taylor & Francis

A Brown Tide Bloom Index based on the potential harmful effects of the brown tide alga, *Aureococcus anophagefferens*

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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 (≥35,000 to <200,000 cells ml^{-1}) have potential negative impacts on feeding and growth in shellfish; Category 3 blooms (≥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 - 43 An index of harmful algae blooms for coastal lagoons. (From Gastrich and Wazniak 2002).

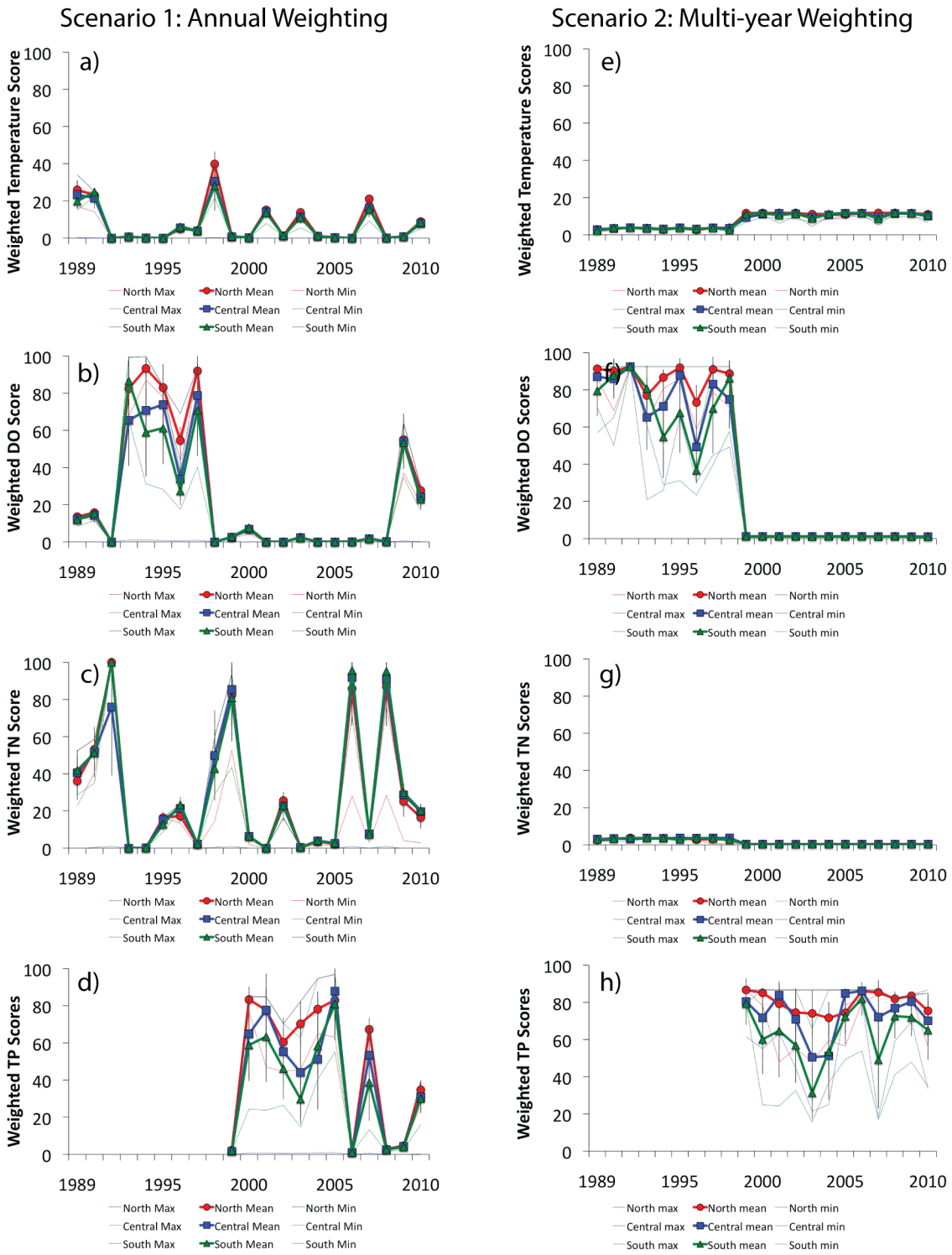

Figure 3 - 44 Weighted scores for water quality under Scenario 1 and Scenario 2.

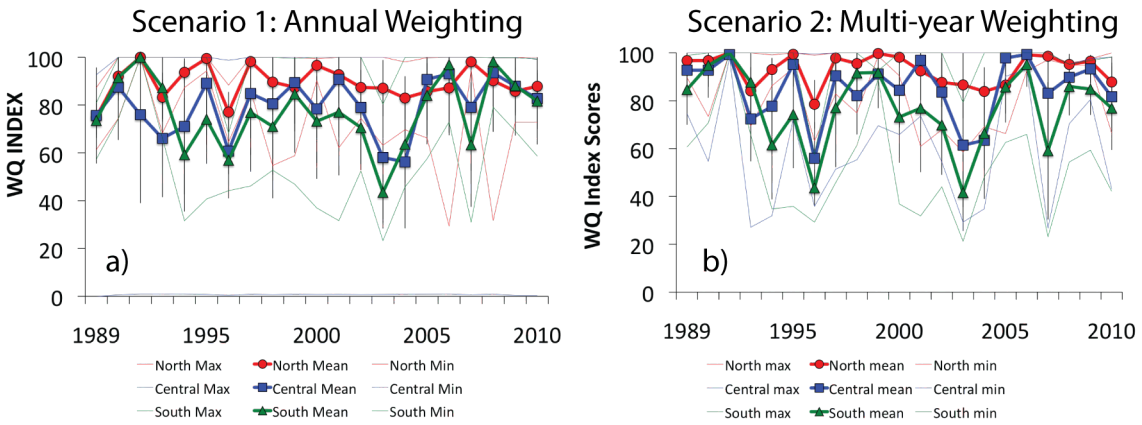

Figure 3 - 45 Weighted Scores under Scenario 1 and Scenario 2 for Water Quality Index.

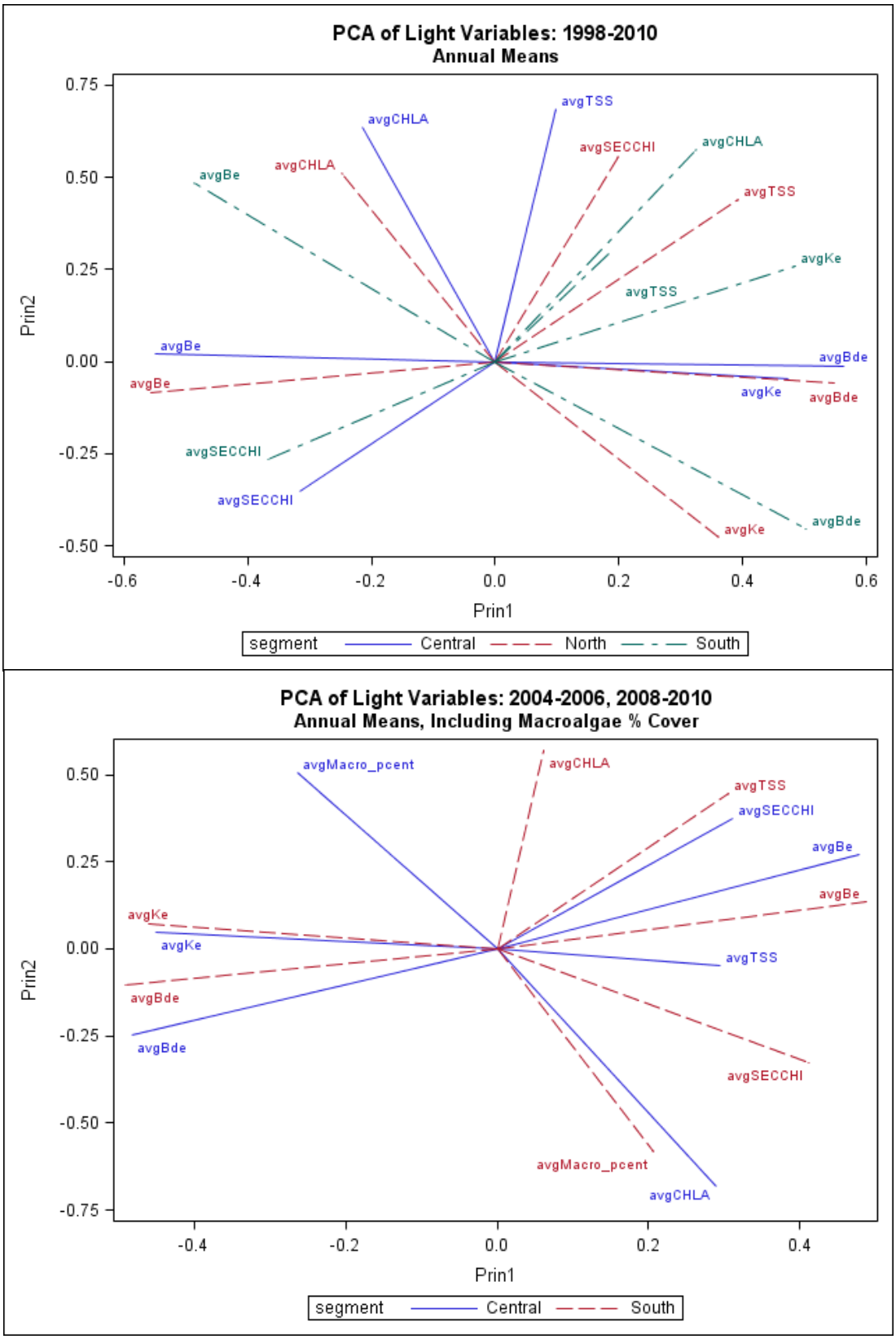

Figure 3 - 46 Principal component analysis of Light Availability indicators excluding (above) and including (below) macroalgae percent cover.

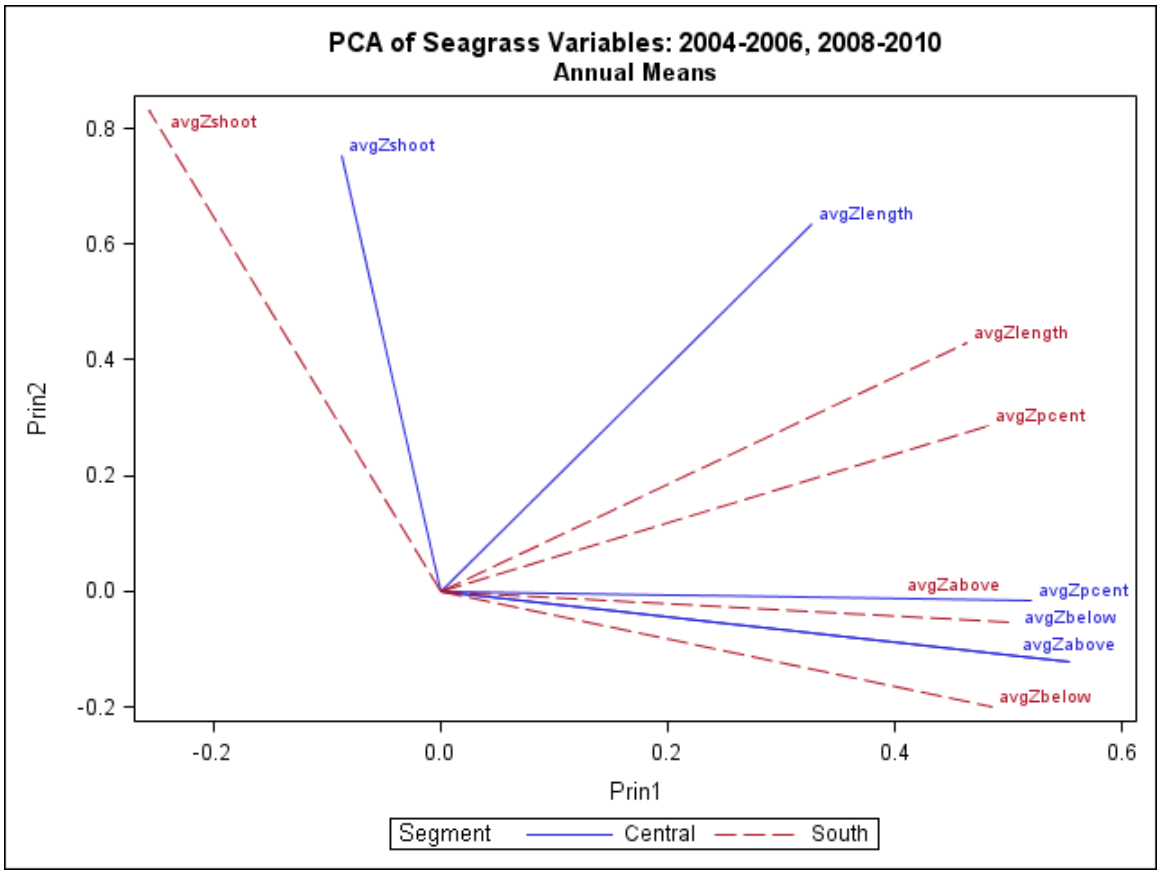

Figure 3 - 47 Principal component analysis of Seagrass indicators.

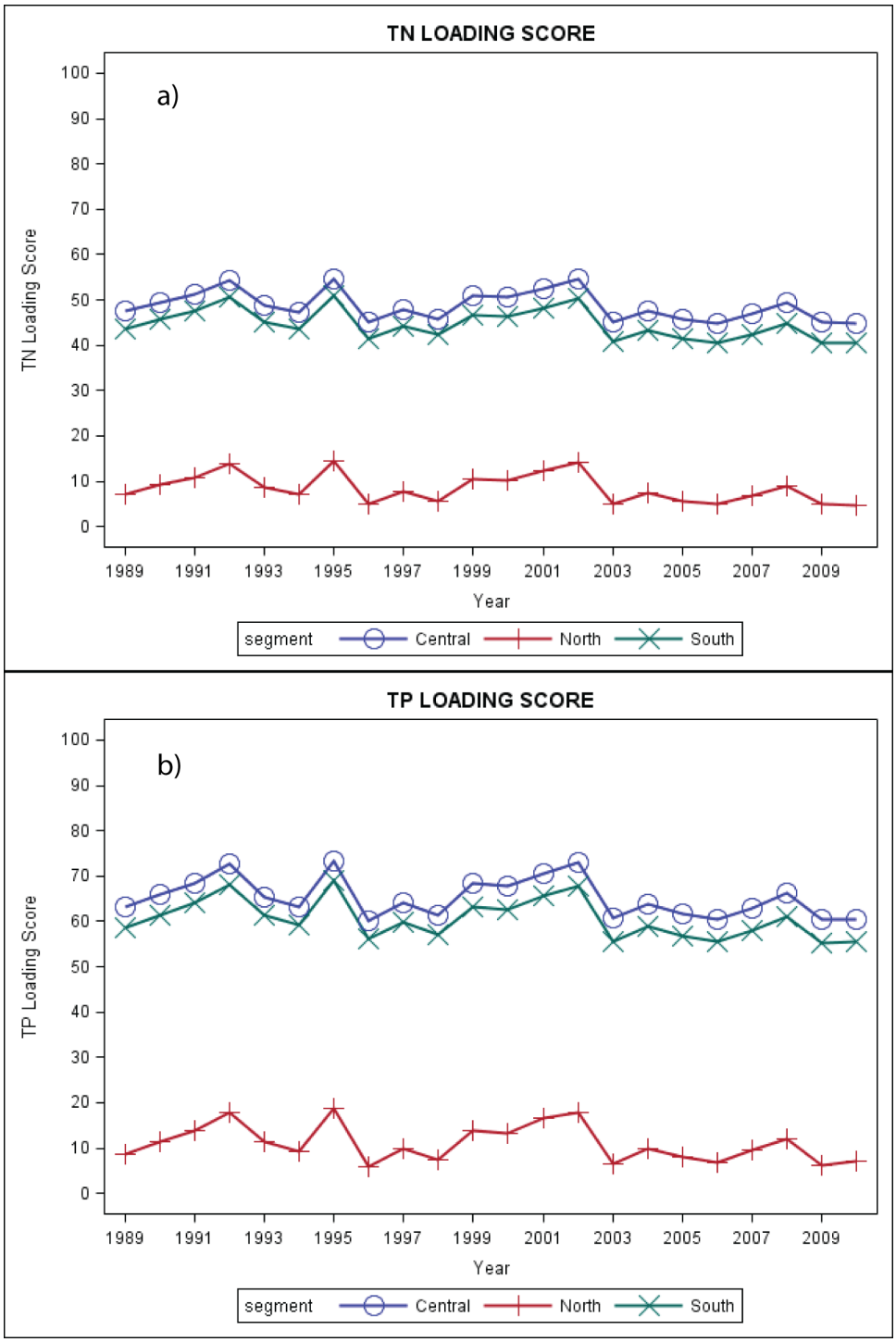

Figure 3 - 48 Raw Scores for total nitrogen loading and total phosphorus loading.

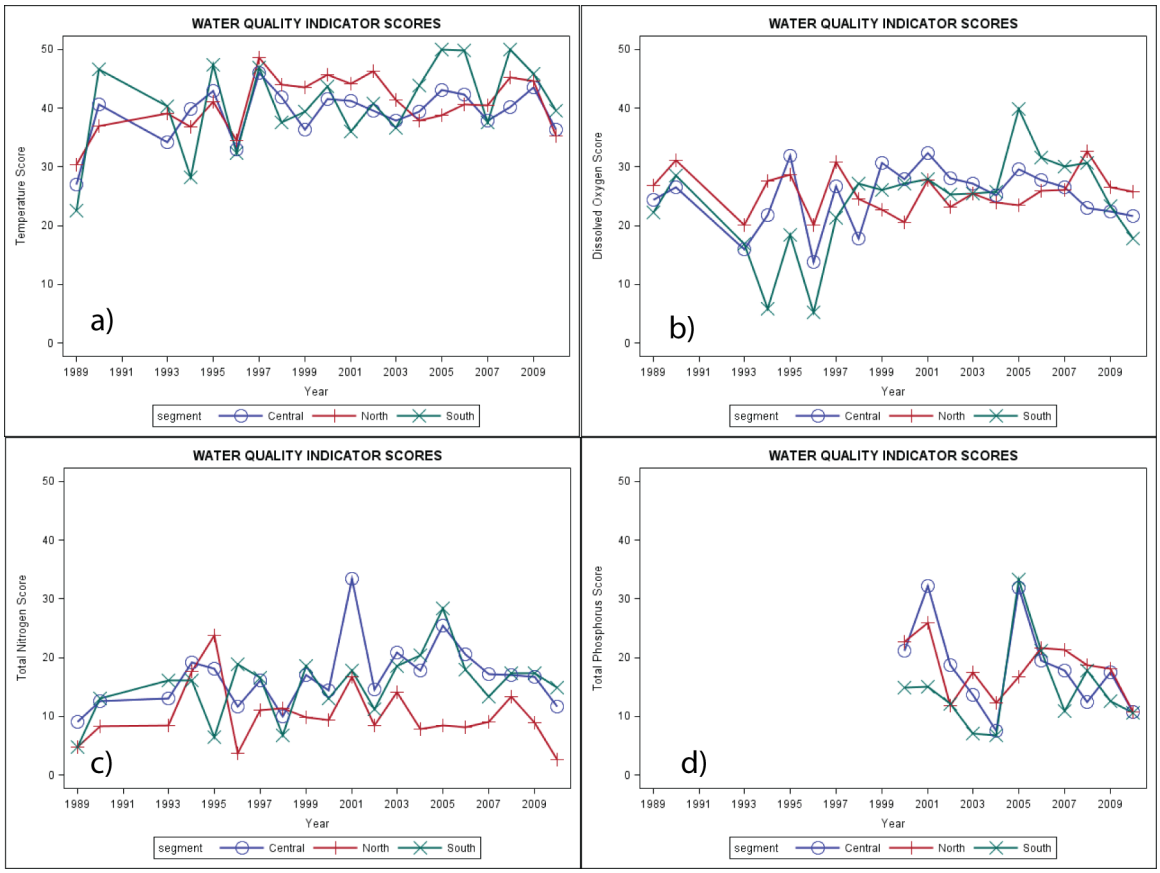

Figure 3 - 49 Scores for Water Quality indicators: (a) temperature, (b) dissolved oxygen, (c) total nitrogen, (d) total phosphorus.

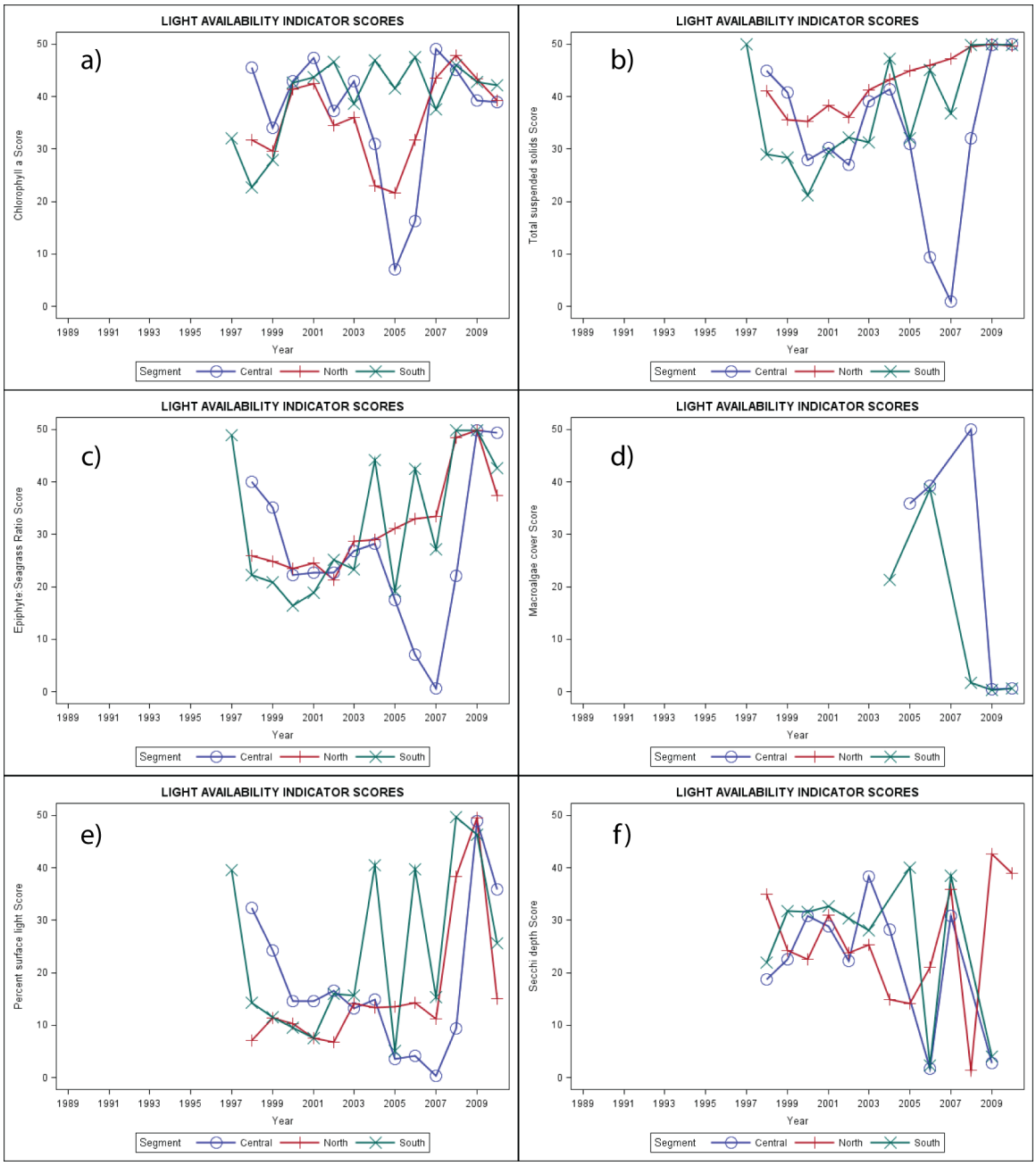

Figure 3 - 50 Scores of Light Availability indicators: (a) chlorophyll, (b) total suspended solids, (c) epiphyte:seagrass ratio, (d) macroalgae cover, (e) percent surface light, (f) Secchi depth.

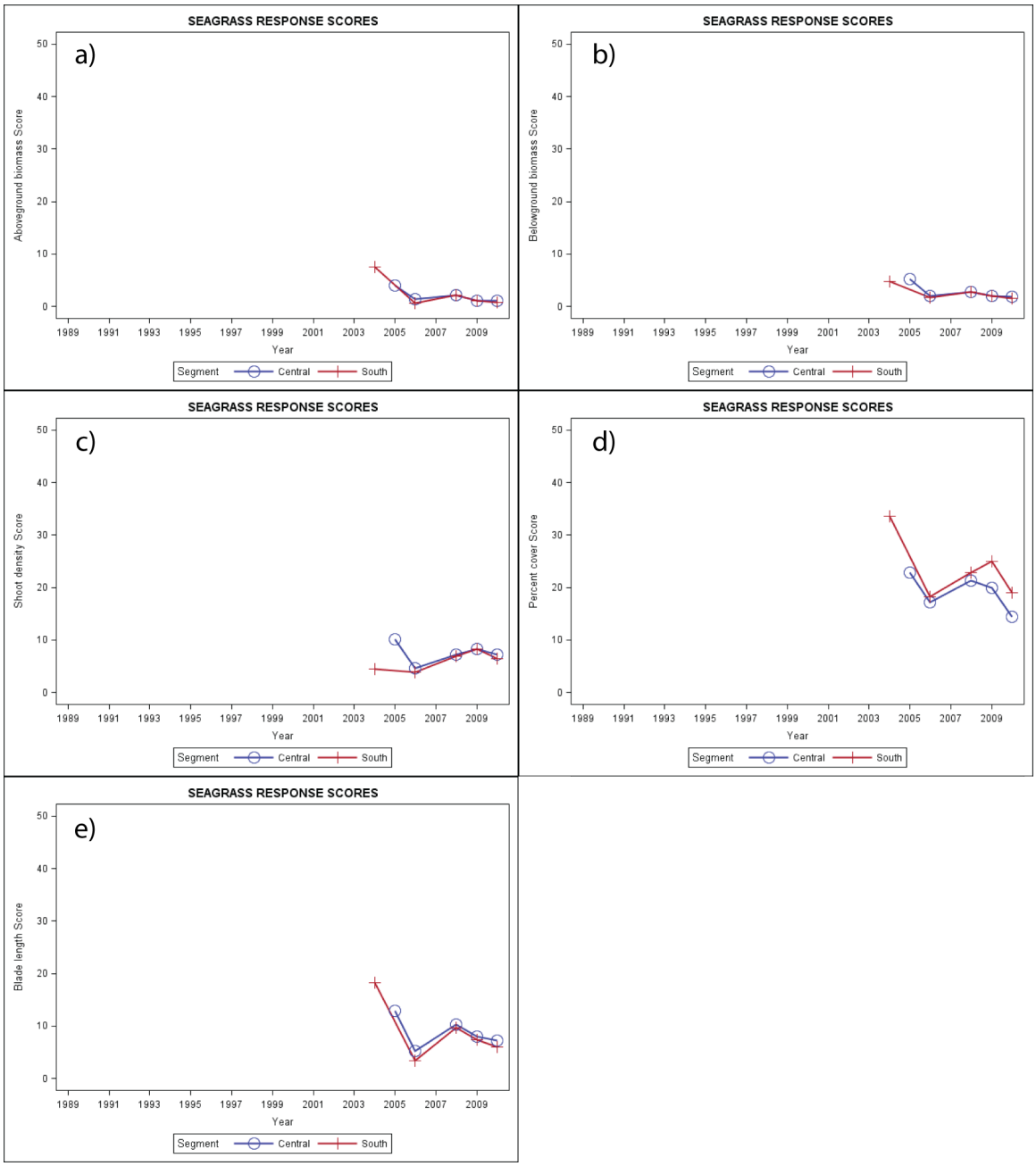

Figure 3 - 51 Scores of Seagrass Response indicators: (a) aboveground biomass, (b) belowground biomass, (c) shoot density, (d) percent cover, (e) blade length.

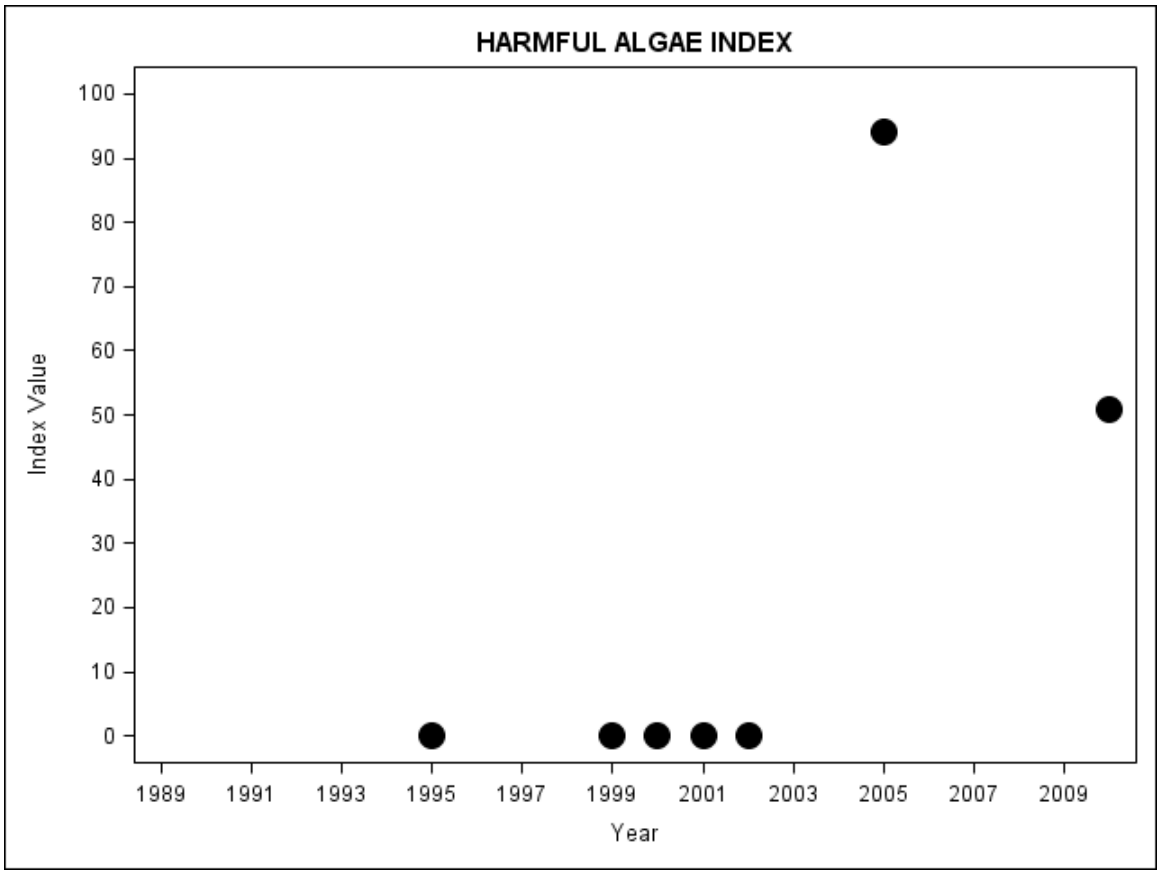

Figure 3 - 52 Harmful Algal Bloom Index.

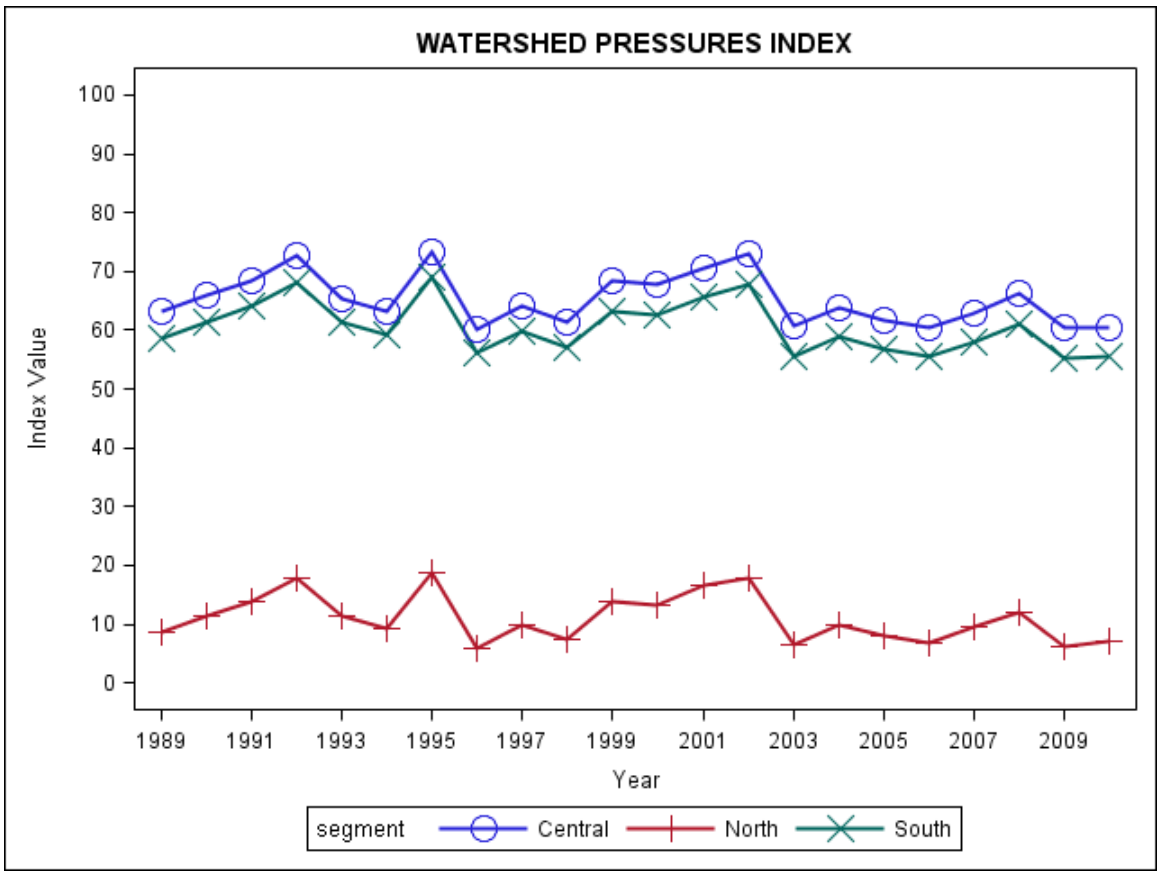

Figure 3 - 53 Watershed Pressure Index.

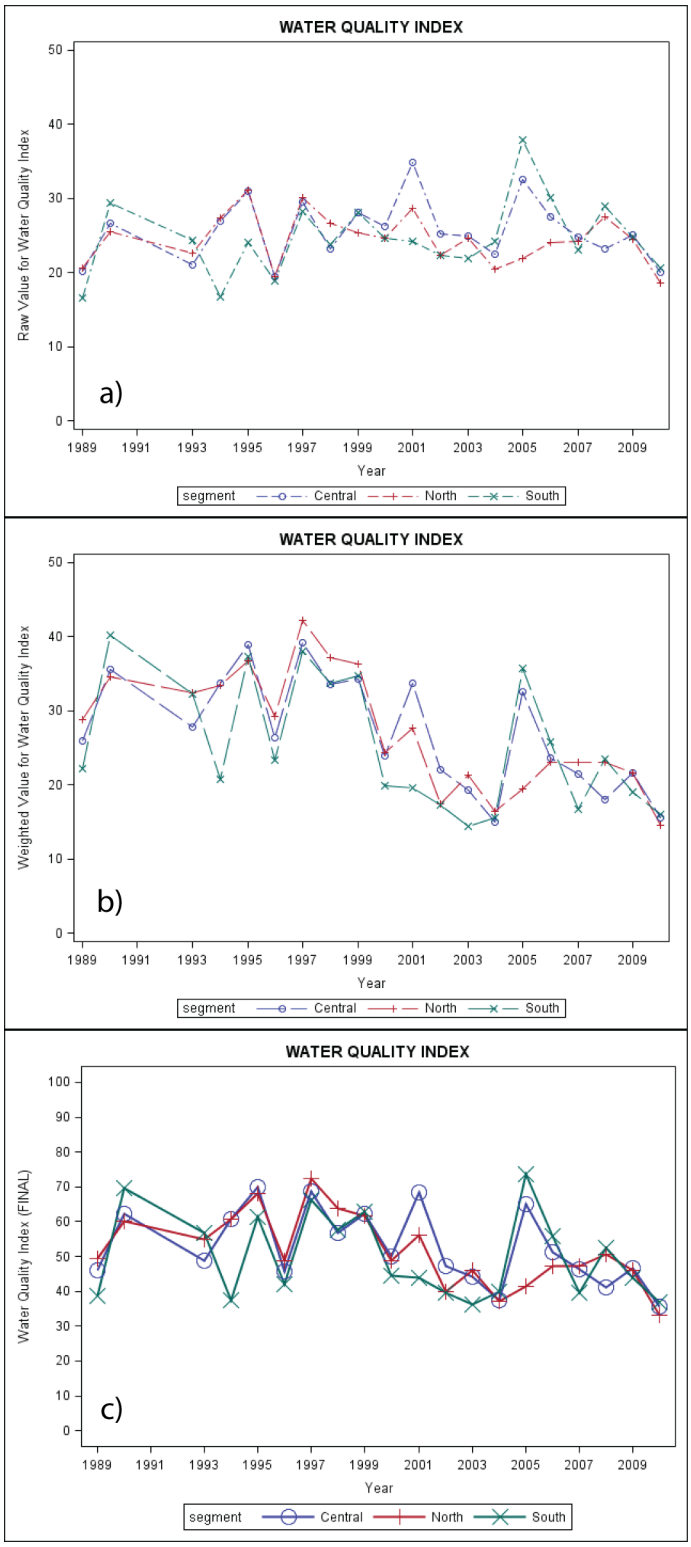

Figure 3 - 54 Water Quality Index scores: (a) Raw Scores, (b) Weighted Scores, (c) Final Scores.

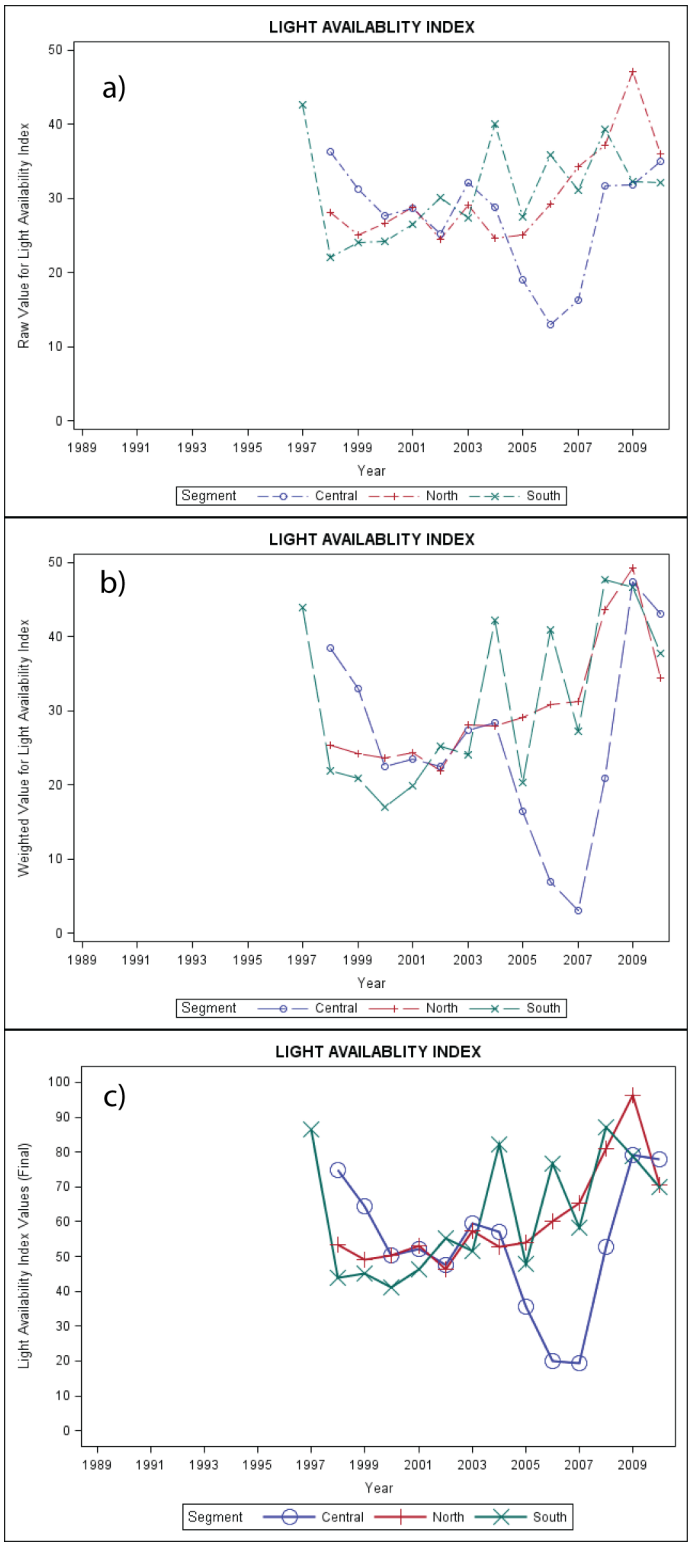

Figure 3 - 55 Light Availability Index scores: (a) Raw Scores, (b) Weighted Scores, (c) Final Scores.

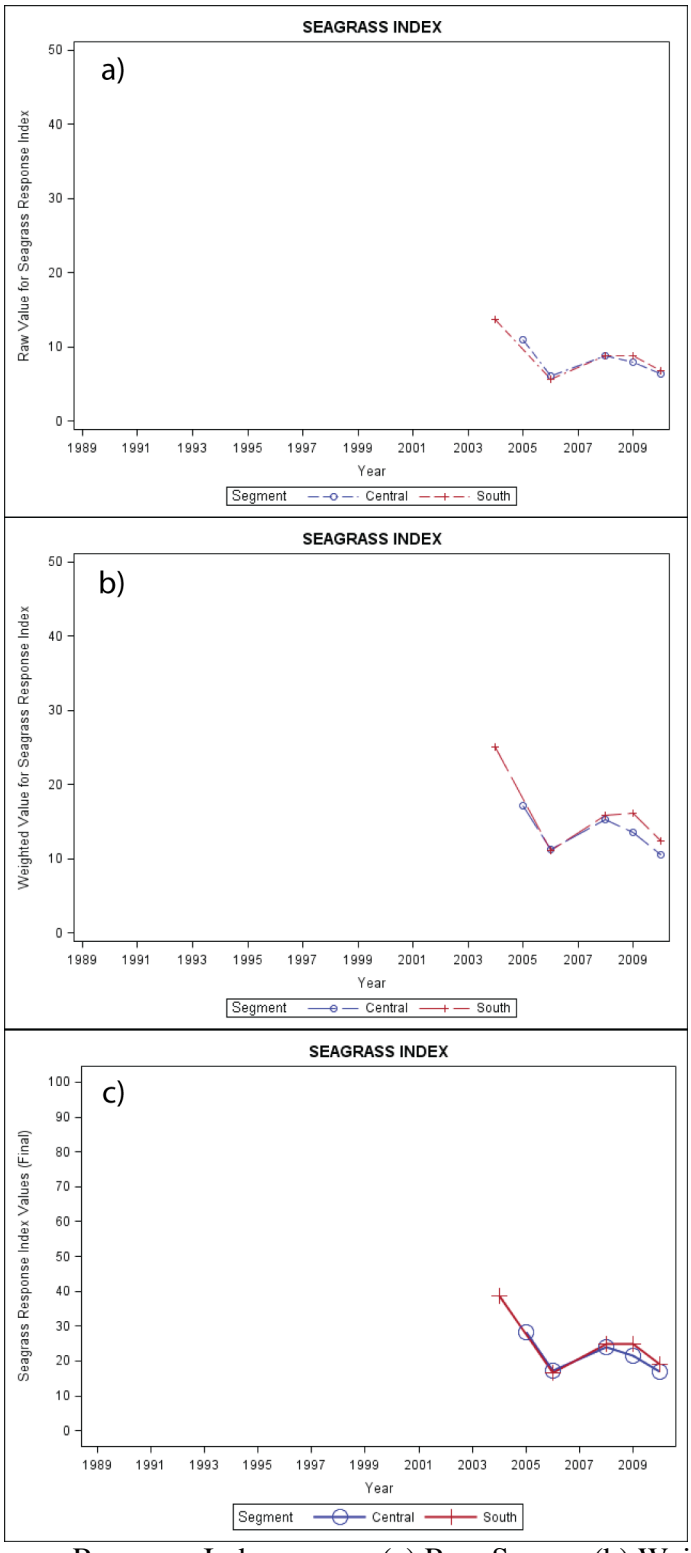

Figure 3 - 56 Seagrass Response Index scores: (a) Raw Scores, (b) Weighted Scores, (c) Final Scores.

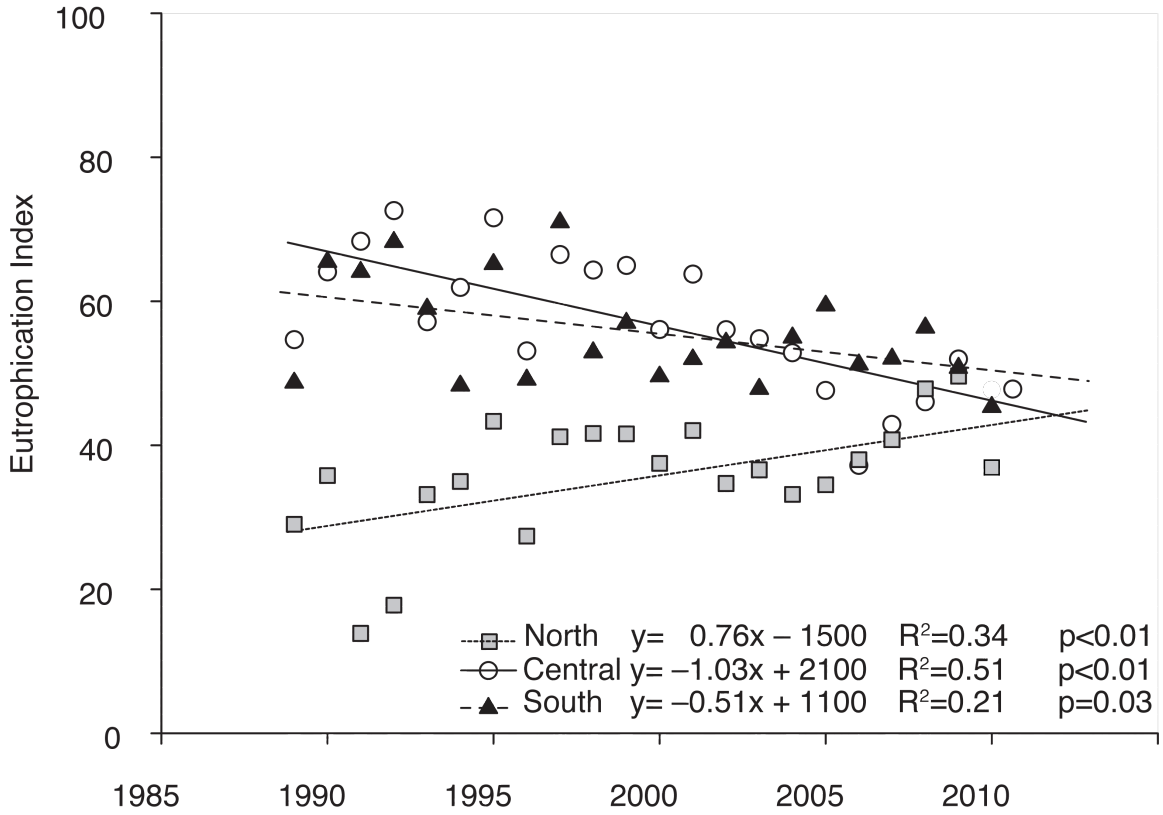

Figure 3 - 57 Overall Eutrophication Index.

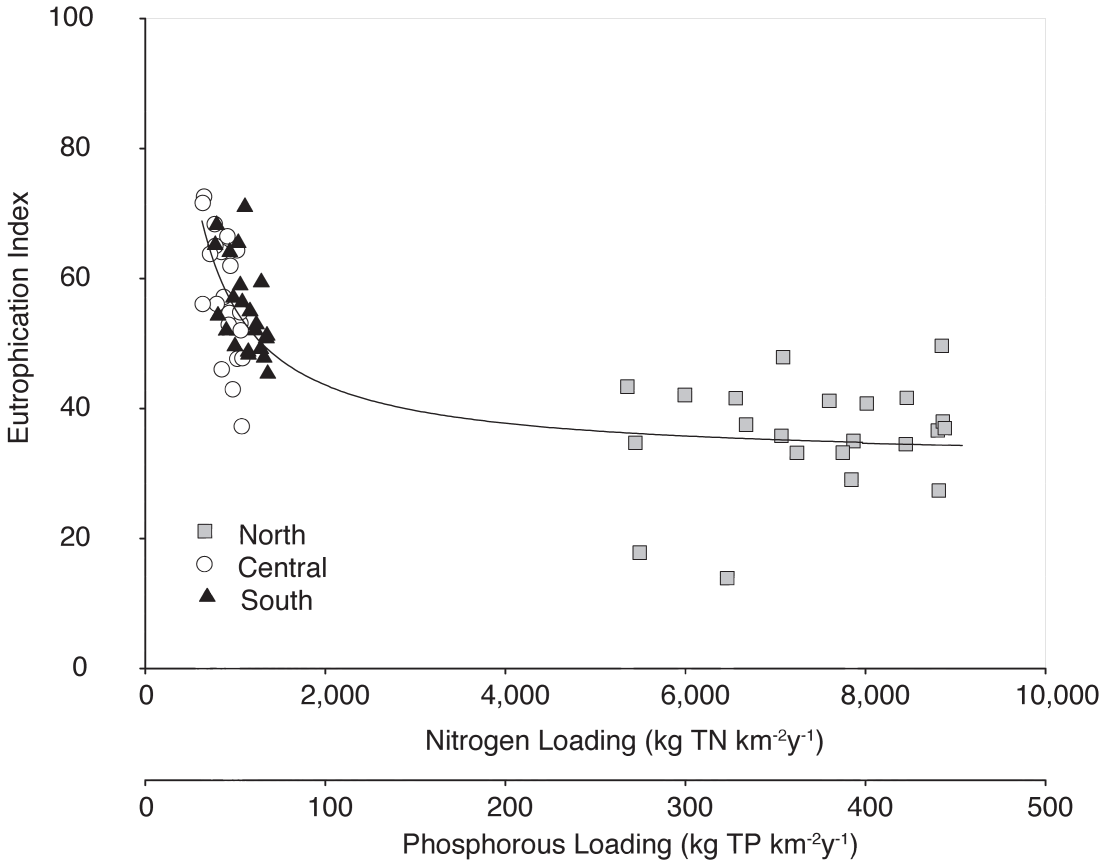

Figure 3 - 58 Eutrophication Index values vs. total nutrient loadings.

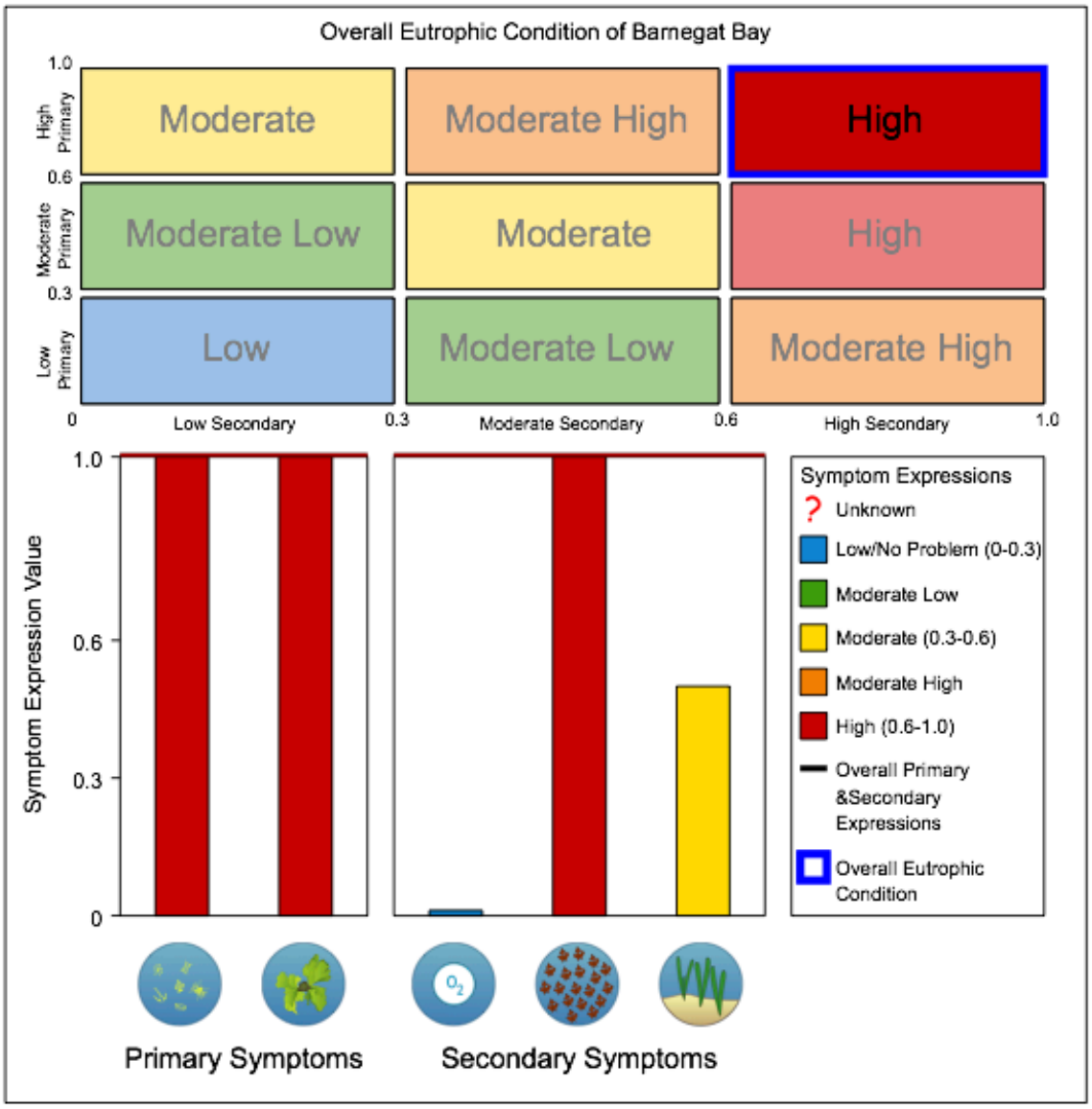

Figure 4 - 1 Results for Barnegat Bay-Little Egg Harbor from the National Estuarine Eutrophication Assessment (from Bricker et al. 2007)

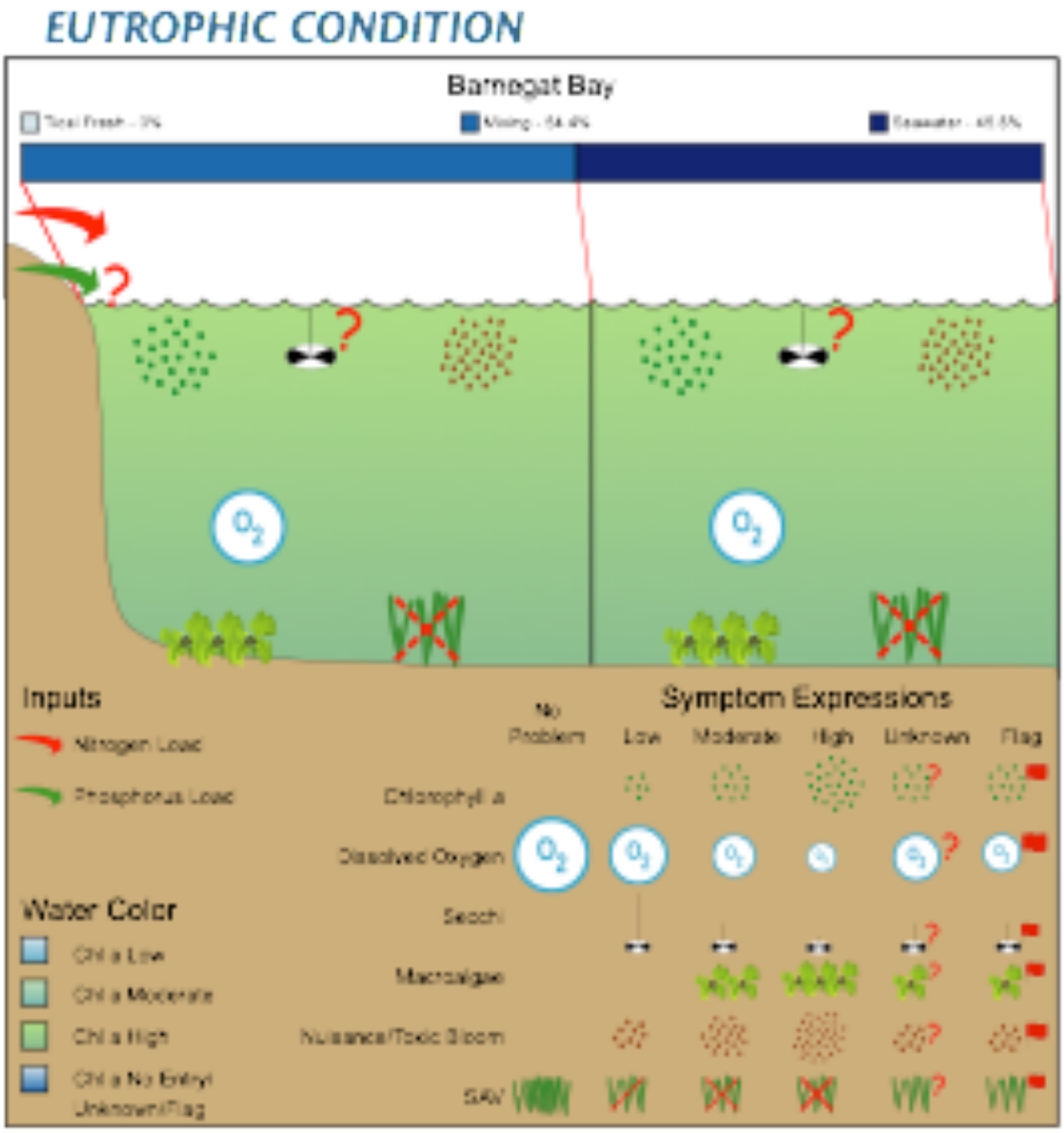

Figure 4 - 2 Conceptual diagram showing eutrophication symptoms of Barnegat Bay-Little Egg Harbor found by the National Estuarine Eutrophication Assessment (from Bricker et al. 2007)

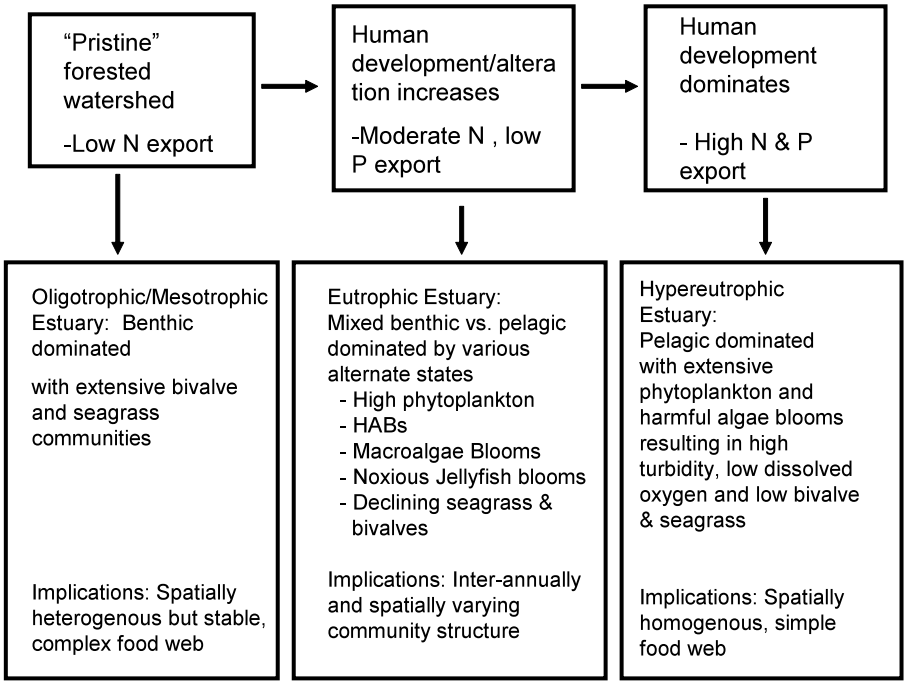

Figure 5 - 1 Conceptual model of eutrophication proposed for the BB-LEH Estuary showing variable biotic pathway responses.

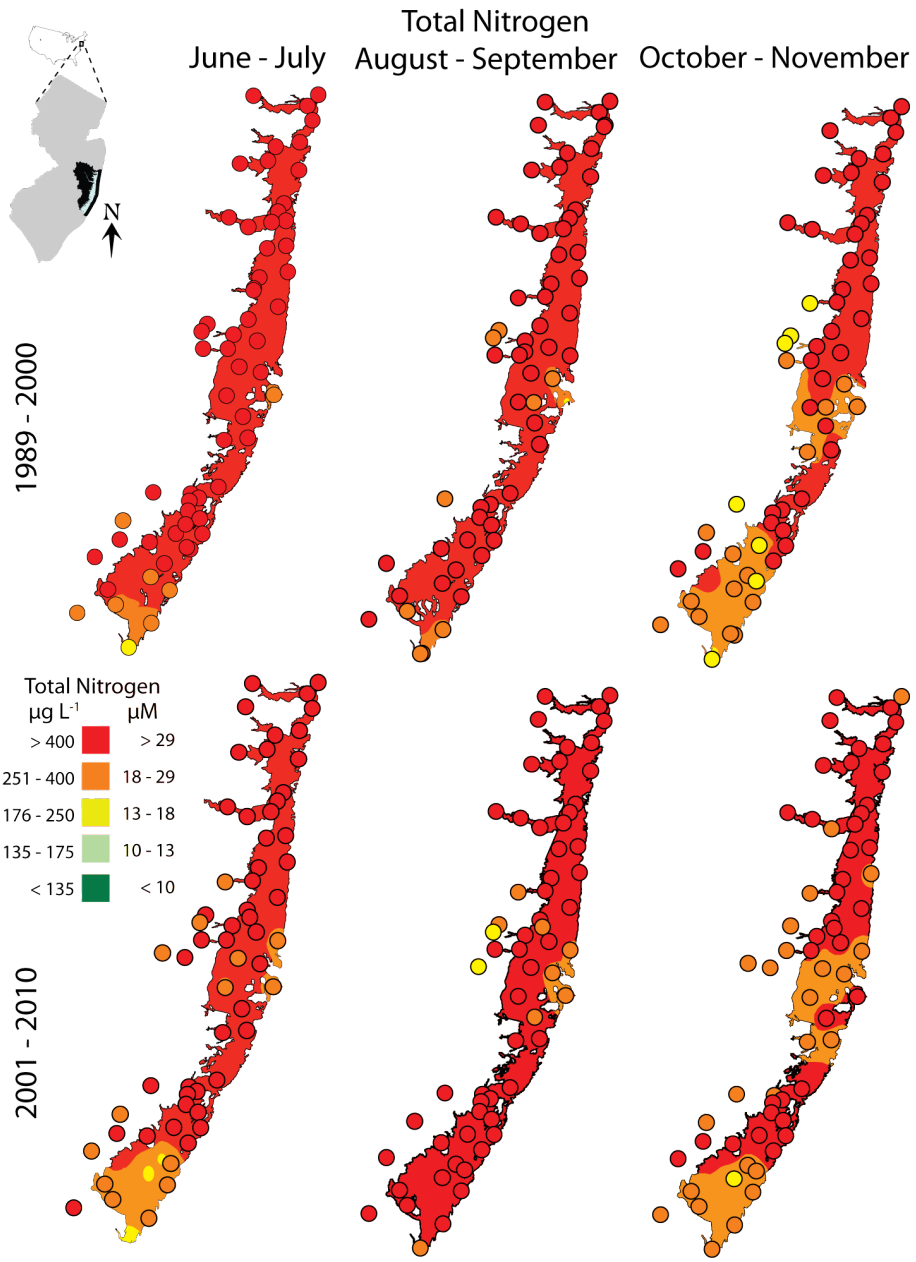

Figure 5 - 2 Total nitrogen concentrations in the BB-LEH Estuary during the June-July, August-September, and October-November sampling periods from 1989-2000 (upper graphic) and 2001-2010 (lower graphic).

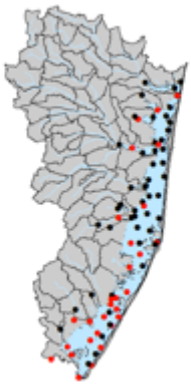

Figure 5 - 3 Water quality sampling stations of the NJDEP in BB-LEH Estuary during the 1989-2010 sampling period. Red dots show stations where dissolved oxygen values were < 4 mg L^{-1}.

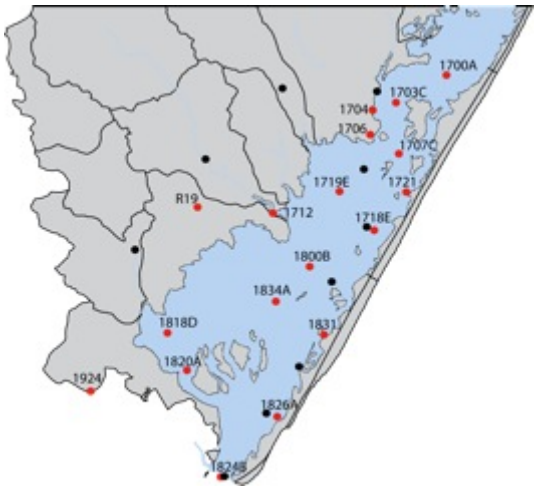

Figure 5 - 4 Sampling stations of the NJDEP for dissolved oxygen measurements in Little Egg Harbor. Note most dissolved oxygen measurements less than 4 mg L^{-1} in the BB-LEH Estuary have been recorded in the southern segment of the estuary.

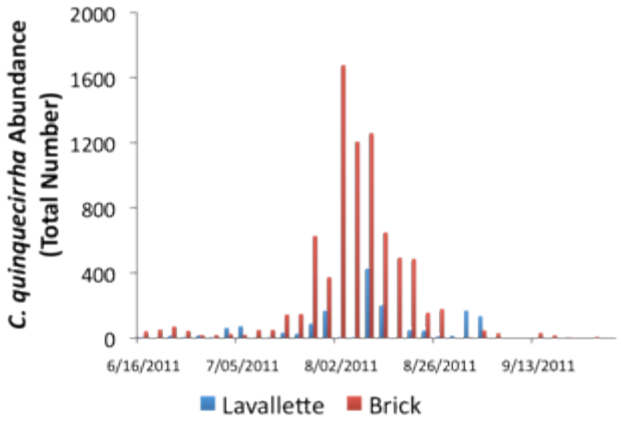

Figure 5 - 5 Abundance of sea nettles in seine sampling conducted at Brick and Lavallette sampling sites in the north segment of the BB-LEHBB-LEH Estuary during 2011. Note elevated abundances at Brick in lower salinity waters (Data Source: Barnegat Bay Partnership).

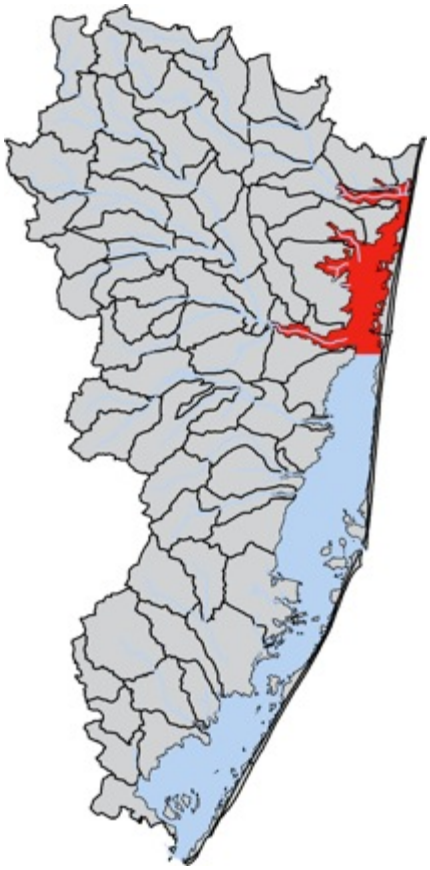

Figure 5 - 6 North segment waters of the BB-LEH Estuary (highlighted) impacted by the occurrence of sea nettles leading to extensive non-swimmable conditions and loss of human use and activity.

TABLES

Table i - 1 Project Field Sampling Sheet

Date: _____ Time (EST):_____ Transect: _____ Station: _____

Quadrat Location: _____ _____
pole antenna

boat antenna
Temp (C) _____ DO% _____

SpCond _____ DO conc _____

Salinity _____ Depth (sonde)_____m

pH _____

Depth (stick) _____cm Secchi _____cm

% Cover *Zostera* _____ % Cover *Ruppia* _____

% Cover Macroalgae (*see comments*) **_____ % Cover other** (*see comments*) **_____**

Macroalgal sample taken? Y N

Photo taken (check)? _____

Biomass sample? Y No seagrass *Zostera* **Epiphyte sample?** Y No

5 random blade lengths (mm) _____ _____ _____ _____ _____
(*Zostera* only)

Boat Scarring _____ Grazing _____

Epiphyte _____ Wasting Disease _____

Scallops_____

Clams_____

Comments:

Supervisor Initials:
_____

Table i - 2 QA/QC results for this project based on the Measurement Quality Objectives (MQOs)

Indicator / Data Type	Accuracy (Bias)	Precision	Completenes
Seagrass			
Biomass	10%	1%	100%
Density	10%	11%	100%
Areal cover	10%	5%	100%
Blade length	10%	7%	100%
Macroalgae	10%		100%
Shellfish abundance	1%	1%	100%
Bloom occurrence	1%	1%	100%
Water Column characteristics			
Dissolved oxygen	±0.5 mg L-1	8%	100%
Salinity	±1.0	1%	100%
Depth	±0.5 m	1%	100%
pH	±0.3 units	1%	100%
Temperature	±1.0 °C	3%	100%
Secchi depth	NA	8%	100%
Chlorophyll a	7%	4%	100%
Total nitrogen	9%	2%	100%

Table i - 3 QA/QC results for specific data collected on this project

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At times Gina Petruzzelli, field researcher, initialed data and tracking sheets as the proxy for the Quality Control Officer when he was not available to do so (most notable, this occurred during the inventory of equipment at the beginning of the day and end-of-day sample inventory when the Quality Control Officer did not accompany the team from/to the boat dock, but rather met them in the field).

While an incomplete set of 5 eelgrass blade measurements from sample locations with less than 5% coverage was feasible, theoretically complete sets of 5 eelgrass blade measurements should have been obtained from all sites greater than 10% coverage (assuming ample number of plants/leaves). At two stations (2-7 period 3, 11-5 period 3) we recorded percent coverage of 10% or greater but did not obtain measurement of a complete set of 5 blades. The cause of this is not clear, but was likely an error of omission. The diver should have been instructed to return to the quadrat to collect enough plants to complete the set of 5 blades. The diver was notified and the correct procedure reinforced.

One of the five epiphyte blade samples from station 12-3 during period 3 was lost between being processed and weighing. We therefore have a value for epiphyte biomass but no value for the blade biomass. A "dummy value" of -9999 was entered in place of the values (cup+sample, cup, calculated mass of the sample) for this sample.

The field data sheet was modified very slightly on one line to reflect that there was no such thing as a non-biomass station, since we sampled at all 120 stations; nor is there such a thing as a non-epiphyte station (again, since we sampled at all 120 stations).

Archived primary and secondary databases for this report are stored in dedicated files in separate locations at the Institute of Marine and Coastal Sciences and Center for Remote Sensing and Spatial Analysis at Rutgers University in New Brunswick. The Rutgers University Marine Field Station (RUMFS) was a State-certified laboratory when all primary biotic and water quality data were collected and analyzed for this project in 2010 and 2011.

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	µS 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 Shellfish samles collected at seagrass sampling sites during 2010 and 2011

2010 Bay Scallops
Period 1, Transect 4, Station 8 (Length 43mm)
Period 2, Transect 1, Station 10 (Length 53mm)

2010 Hard Clams
Period 1, Transect 9, Station 4 (Length 88mm; 63mm), both outside of quadrat
Period 2, Transect 8, Station 10 (Length 93mm)
Period 2, Transect 8, Station 7 (Length 85mm)
Period 2, Transect 1, Station 10 (Length 83mm)
Period 2, Transect 1, Station 7 (Length 76mm), collected in core
Period 1, Transect 3, Station 9
Period 1, Transect 7, Station 7 (Shell hash)
Period 1, Transect 7, Station 10
Period 1, Transect 8, Station 4
Period 1, Transect 8, Station 9
Period 2, Transect 6, Station 8 (Length 85mm)

2010 Additional Observations
Period 2, Transect 3, Station 1 (Some shell hash)
Period 1, Transect 6, Station 3 (Empty hard clam shell in quadrat)
Period 1, Transect 7, Station 8 (Shell hash and sand)
Period 1, Transect 7, Station 9 (Shell has on bottom)
Period 1, Transect 12, Station 3 (Empty oyster shell in core)
Period 2, Transect 7, Station 3 (Shell hash)
Period 2, Transect 7, Station 7 (Shell hash)
Period 2, Transect 7, Station 8 (Shell hash)
Period 2, Transect 9, Station 9 (Shell hash under sand)
Period 2, Transect 10, Station 6 (Empty mussel shells)
Period 3, Transect 1, Station 6 (Shell hash)
Period 3, Transect 1, Station 10 (Shell hash, Razor clam shells)
Period 3, Transect 2, Station 1 (Shell hash)
Period 3, Transect 2, Station 4 (Shell hash in quadrat)
Period 3, Transect 3, Station 1 (Empty hard clam shell in quadrat)
Period 3, Transect 12, Station 2 (Shell hash)

2011 Bay Scallops
NONE

2011 Hard Clams
Time Period 1, Transect 8, Station 8 (Length 95mm, 82mm)
Time Period 2, Transect 1, Station 7 (Length 91mm, shell hash)

Time Period 3, Transect 9, Station 3 (Length 80mm)
Time Period 1, Transect 8, Station 9 (Length 69 mm)
Time Period 3, Transect 1, Station 7 (Length 90mm, 92mm, 85mm, 74mm)

2011 Additional Comments
Time Period 1, Transect 7, Station 8 (Shell hash)
Time Period 2, Transect 7, Station 5 (Shell hash)
Time Period 1, Transect 7, Station 5 (Mussels)
Time Period 1, Transect 5, Station 5 (Razor clam, Length 55mm)

Table 2 - 11 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

Table 3 - 2 Temporal and spatial extent of the National Coastal Assessment (NCA) dataset (from US EPA).

Date	Station	Latitude	Longitude
8/29/00	NJ00-0049	39.9329	-74.1407
8/29/00	NJ00-0051	39.9896	-74.0740
9/6/00	NJ00-0043	39.6404	-74.2051
9/7/00	NJ00-0035	39.5054	-74.3990
7/20/01	NJ01-0040	39.5893	-74.2399
7/24/01	NJ01-0048	39.8526	-74.1021
8/8/01	NJ01-0036	39.5112	-74.2974
8/10/01	NJ01-0044	39.7187	-74.1732
10/4/01	NJ01-0104	39.9409	-74.1790
10/4/01	NJ01-0106	39.8884	-74.1108
10/5/01	NJ01-0102	39.9972	-74.1128
10/8/01	NJ01-0108	39.8023	-74.1737
10/8/01	NJ01-0110	39.7017	-74.1790
10/10/01	NJ01-0112	39.6046	-74.2610
10/11/01	NJ01-0114	39.5421	-74.3087
10/11/01	NJ01-0116	39.5351	-74.3771
10/31/01	NJ01-0042	39.6254	-74.2357
10/31/01	NJ01-0046	39.8110	-74.1676
10/31/01	NJ01-0050	39.9393	-74.0910
8/15/02	NJ02-0230	39.5365	-74.3338
8/27/02	NJ02-0240	40.0176	-74.0729
8/28/02	NJ02-0238	39.9376	-74.1101
9/4/02	NJ02-0043	39.6404	-74.2051
9/4/02	NJ02-0235	39.7643	-74.1079
9/23/02	NJ02-0227	39.4982	-74.3335
9/4/03	NJ03-0048	39.8526	-74.1021
9/4/03	NJ03-0239	39.9482	-74.1015
9/5/03	NJ03-0234	39.7488	-74.1872
9/5/03	NJ03-0236	39.8251	-74.1604
7/21/04	NJ04-0437	39.8180	-74.0990
7/26/04	NJ04-0438	39.9440	-74.1200
7/26/04	NJ04-0440	40.0400	-74.0560
7/27/04	NJ04-0231	39.6098	-74.2179
7/27/04	NJ04-0233	39.6579	-74.2185
7/27/04	NJ04-0435	39.7210	-74.1420
7/28/04	NJ04-0043	39.6400	-74.2050
8/18/04	NJ04-0228	39.5354	-74.2682
8/19/04	NJ04-0430	39.5750	-74.2780
8/27/04	NJ04-0427	39.5210	-74.3790
9/13/05	NJ05-0059	39.5050	-74.3540
9/14/05	NJ05-0052	39.8850	-74.1250
9/19/05	NJ05-0057	39.5840	-74.2650
9/19/05	NJ05-0058	39.7770	-74.1540
7/12/06	NJ06-0031	39.7870	-74.1250
7/12/06	NJ06-0039	39.7300	-74.1500
7/12/06	NJ06-0045	39.6660	-74.1930
7/12/06	NJ06-0069	39.6160	-74.2450
7/14/06	NJ06-0027	39.5000	-74.4010
7/14/06	NJ06-0035	39.4960	-74.3600
7/19/06	NJ06-0012	39.8470	-74.1100
7/19/06	NJ06-0041	39.8670	-74.0910
7/19/06	NJ06-0062	39.8520	-74.1360
7/19/06	NJ06-0063	39.9040	-74.1160
7/26/06	NJ06-0004	39.6260	-74.1990
7/26/06	NJ06-0006	39.5650	-74.3120
7/26/06	NJ06-0009	39.5680	-74.2790
7/26/06	NJ06-0050	39.5950	-74.2510
7/27/06	NJ06-0018	39.5230	-74.2980
8/8/06	NJ06-0015	39.5360	-74.3880

Table 3 - 3 ANOVA results testing for significant diferences between north-central-south segments for watershed, water quality, and sediment variables

	Variable	Units	n	df	MSE	F	p	Dataset
Watershed	TN load	kg TN year-1	9	2,6	665,190,000	14.88	**0.0047**	USGS, 2009
Watershed	areal load	kg TN year-1 km-2	9	2,6	2,900	16.43	**0.0037**	USGS, 2009
Water Quality	Salinity	ppt	3072	2, 3069	2,586	9.61	**< 0.0001**	DEP BMW, 1989-2010
Water Quality	TN in bay	ug N L-1	3050	2, 3047	145,560	91.82	**< 0.0001**	DEP BMW, 1989-2010
Water Quality	NO3	ug N L-1	3081	2, 3078	35,746	174.05	**< 0.0001**	DEP BMW, 1989-2010
Water Quality	NH3	ug N L-1	3068	2, 2065	19,947	23.93	**< 0.0001**	DEP BMW, 1989-2010
Sediments	Moisture	%	553	2, 550	318	7.99	**0.0004**	NCA, 2000-2006
Sediments	TOC	mg L-1	1510	2, 1507	1	15.11	**< 0.0001**	NCA, 2000-2006
Sediments	Sand	%	1353	2, 1350	542	11.76	**< 0.0001**	NCA, 2000-2006
Sediments	Silt-Clay	%	1353	2, 1350	542	11.73	**< 0.0001**	NCA, 2000-2006
Invertebrates	Abundance	inidviduals m-2	53	2, 50	237,920	2.35	0.1062	REMAP, 2001
Invertebrates	Abundance	inidviduals m-2	1511	2, 1508	26,316	0.89	0.4099	NCA, 2000-2006

Table 3 - 4 Equations for each indicator used to rescale observations into raw scores according to defined thresholds.

Component	Metric	Units	Rescaling equation	Maximum Score	Minimum Score	Reference
Pressures	Total nitrogen loading	kg TN	y = -19*ln(x)+177.52	x ≤ 50	x ≥ 10,000	Boynton et al. 1996, Short and Burdick 1996, Tomasko et al. 1996, Valiela et al. 2000,
Pressures	Total phosphorus loading		y = -32.81*ln(x)+204.01	x ≤ 25	x ≥ 500	Deegan et al. 2002, Burkholder et al. 2007, Kennish and Fertig 2012, This study
Water Quality	Temperature	C	y = -3.125*x+106.25	x ≤ 18	x ≥ 34	Stevenson et al. 1993, Howell and Simpson 1994, Boynton et al. 1996, Bricker et al.
Water Quality	Dissolved oxygen	mg L^{-1}	y = 4.8641*$e^{0.228*x}$	x ≥ 10	x ≤ 4	1999, Diaz and Solow 1999, Breitburg et al. 2001, Breitburg et al. 2002, Kiddon et al.
Water Quality	Total nitrogen	µg L^{-1}	y = 26721*$x^{-1.274}$	x ≤ 135	x ≥ 750	2003, Borja et al. 2004, Kemp et al. 2004, Burkholder et al. 2007, Kennish and Fertig
Water Quality	Total phosphorus	µg L^{-1}	y = 475.95*$x^{-0.977}$	x ≤ 10	x ≥ 45	2012, Lee et al. 2007, Wazniak et al. 2007, Williams et al. 2009, This study
Light Availability	% surface irradiance available	%	y = 50.084*ln(x)-122.18	x ≥ 32	x ≤ 7.818	Dennison et al. 1993, Burkholder 2001, Brush and Nixon 2002, Kemp et al. 2004,
Light Availability	Chlorophyll *a*	µg L^{-1}	y = -41.67*ln(x)+85.351	x ≤ 2.5	x ≤ 100	Burkholder et al. 2007, Lee et al. 2007, Ralph et al. 2007, Kennish et al. 2011, This
Light Availability	Total suspended solids	mg L^{-1}	y = -5*x+100	x ≤ 10	x ≥ 20	study
Light Availability	Secchi depth	cm	y = 0.125*x-12.5	x ≥ 500	x ≤ 100	
Light Availability	Macroalgae percent cover	%	y = -24.52*ln(x)+76.782	x ≤ 3	x ≥ 20	
Light Availability	Epiphyte:Seagrass biomass	g epiphyte / g seagrass	y = -20.32*ln(x)+22.744	x ≤ 2.5	x ≥ 2.0	
Seagrass response	Aboveground biomass	g m^{-2}	y = 0.125*x	x ≥ 400	x ≤ 0	Dennison et al. 1993, Duarte 1995, Valiela et al. 2000, Deegan et al. 2002, Lea et al.
Seagrass response	Belowground biomass	g m^{-2}	y = 0.0625*x	x ≥ 800	x ≤ 0	2003, Kemp et al. 2004, Burkholder et al. 2007, Lee et al. 2007, Ralph et al. 2007,
Seagrass response	Percent cover	%	y = 15.925*ln(x)-12.713	x ≥ 50	x ≤ 0	This study
Seagrass response	Shoot density	shoots m^{-2}	y = 0.0243*x+5.7143	x ≥ 1910	x ≤ 0	
Seagrass response	Blade length	cm	y = 0.625*x	x ≥ 80	x ≤ 0	
Harmful algal bloom response	Harmful algal bloom concentration	cells L^{-1}	y = -0.0004*x+113.98	x ≤ 30,000	x ≥ 260,000	Gastrich and Wazniak 2002, Gastrich et al. 2004
Benthic invertebrate response	Benthic invertebrates					Baden et al. 1990,

Table 3 - 5 Defined thresholds for Ecosystem Pressures

	PRESSURE THRESHOLDS	
	TN Total Loading	TP Total Loading
SCORE	kg TN estuary km^{-2} yr^{-1}	kg TP estuary km^{-2} yr^{-1}
100	50	25
75	250	50
50	1,000	100
25	3,000	250

Table 3 - 6 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 ($\mu g\ l^{-1}$)	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 - 7 Benthic community response to decreasing oxygen concentrations (From Ritter and Montagna 1999)

TABLE 4. Community response to different hypoxia intensity categories. Hypoxic categories are based on the average environmental and community characteristics of stations falling within each category. DO = dissolved oxygen.

Hypoxia Intensity Category	DO (mg l^{-1})	Biomass (g m^{-2})	Density (no. m^{-2})	No. Species (0.01 m^{-2})	No. Dominant Species (N1)	Dominant Species
1	>5	7.64	22,171	20	9.45	Codominance
2	4–5	3.30	7186	12	7.65	*Mediomastus ambiseta*
3	3–4	—	—	—	—	—
4	2–3	0.42	3144	3	1.63	*Streblospio benedicti*
5	1–2	0.01	189	1	1	Oligochaeta
6	0–1	—	—	—	—	—

Table 3 - 8 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 - 9 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 - 10 Optimal 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 - 11 Defined thresholds for Water Quality indicators

	WATER QUALITY THRESHOLDS			
	Temperature	Dissolved Oxygen	Total Nitrogen	Total Phosphorus
SCORE	°C	mg L^{-1}	µg L^{-1}	µg L^{-1}
50	18	10.0	135	10
38	22	9.0	175	13
25	26	7.5	250	22
13	30	4.0	400	40

Table 3 - 12 Physiological light requirements for seagrass species (from Dennison et al. 1993).

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 $100 \times I_z/I_o = e^{-K_d * Z}$. Range of maximal depth limit and K_d values and means ± 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 antaractica**	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	0.32–0.92	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.

Table 3 - 13 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 - 14 Defined thresholds for Light Availability indicators

	LIGHT AVAILABILITY THRESHOLDS					
	% Surface Irradiance Available	Secchi depth	Total suspended solids	Chlorophyll *a*	Macroalgae % cover	Epiphyte biomass per SAV biomass
	%	cm	mg L^{-1}	µg L^{-1}	%	g epiphyte / g seagrass
SCORE						
50	32	500	10.0	2.5	3	0.25
38	23	400	12.5	3.0	5	0.50
25	19	300	15.0	4.0	8	1.00
13	15	200	17.5	6.0	14	1.50

Table 3 - 15 Defined thresholds for Seagrass indicators

	SEAGRASS THRESHOLDS				
	Aboveground biomass	Belowground biomass	Shoot density	Percent Cover	Blade length
SCORE	g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm
50	400	800	1910	50	80
38	300	600	1146	25	60
25	200	400	764	10	40
13	100	200	382	5	20

Table 3 - 16 Impacts of brown tide at various concentrations used as thresholds (from Gastrich and Wazniak 2002). Table continues on next page

Table 1. Brown Tide Bloom Index: *Aureococcus anophagefferens*, cells ml^{-1}. Table is based on available scientific data; some of the available data may need to be reassessed through additional research; and there may be complex ecological interactions (e.g., trophic level interactions, presence of additional algal species, etc.) which may affect impacts which are not fully addressed in the table; Categories are relative and related to different threshold concentrations of brown tide.

Cells ml^{-1}	Index (Category)	Potential Impact	Sources
<35,000	1	• **Shellfish:** no known impact on *Mercenaria mercenaria* juveniles	Bricelj et al., 2001; Schaffner, 1999
35,000 to <200,000	2	• **Shellfish Impacts**	
		• **Hard Clams** (*Mercenaria mercenaria*)	
		The threshold concentration of toxic clones that inhibit clearance (feeding rates) on co-occurring phytoplankton species was determined to be at 35,000 to 50,000 *Aureococcus* cells ml^{-1} for juvenile (10 mm) hard clams.	Bricelj et al., 2001; Bricelj, 1999; Schaffner 1999;
		Short term feeding study (1–2 hrs) showed that an isolate of *Aureococcus* (from West Neck Bay, NY, 1995) at ≥35,000 cells ml^{-1} significantly reduced feeding (clearance rate) of juvenile hard clams (ca. 10 mm); longer term growth studies (2–3 wks) showed similar results.	Schaffner, 1999; Bricelj, 1999
		• **Mussels** (*Mytilus edulis*)	
		At 1–3 × 10^5 *Aureococcus* cells ml^{-1}, mussels in bloom areas show stronger growth reduction than quahogs relative to non-bloom sites and growth of juvenile mussels in Peconic Bay significantly reduced	Bricelj and Borrero, unpubl. Data, reported in Bricelj and Lonsdale, 1997; Bricelj and Lonsdale, 1997
		• **Bay Scallops** (*Argopecten irradians*)	
		Significant growth reduction and high mortalities of bay scallop larvae at 190,000–750,000 *Aureococcus anophagefferens* ml^{-1}	Gallagher et al., 1989
200,000 to >1,000,000	3	• **Physical Characteristics**	
		Water becomes discolored at 200,000 *Aureococcus* ml^{-1}	W. Dawydian and R. Nuzzi, Suffolk County Department of Health Services, (pers. comm)
		• **Shellfish Impacts**	
		Bivalves may experience sub-lethal, adverse effects at *Aureococcus* densities of 10^5 cells ml^{-1}	Bricelj and Lonsdale 1997; Bricelj and Kuenster, 1989
		• **Mussels and Hard Clams**	
		Grazing (clearance) rates of adult *Mytilus edulis* and *Mercenaria mercenaria* markedly inhibited during Narragansett Bay brown tide in 1985 (*Aureococcus* concentrations >10^6 cells ml^{-1}); dilution experiments in Narragansett Bay water showed *Aureococcus anophagefferens at* >2.5 × 10^5 cells ml^{-1} were required to inhibit clearance rates of adult *Mytilus edulis* on *Isochrysis galbana*	Tracey, 1988
		Effects of toxic strains of >10^6 ml^{-1} *Aureococcus* on clearance (feeding) rates of juvenile mussels	Tracey, 1988; Bricelj, 1999; Bricelj et al., 2001
		Growth of juvenile mussels significantly reduced in Peconic Bay sites at *Aureococcus* concentrations ~100,000 to 300,000 cells ml^{-1}	Bricelj and Lonsdale, 1997
		Growth of juvenile *Mercenaria mercenaria* undetectable at toxic *Aureococcus* clone concentrations ≥400,000 cells ml^{-1}	Bricelj et al., 2001

(Continued on next page)

Table 1. Brown Tide Bloom Index: *Aureococcus anophagefferens*, cells ml^{-1}. Table is based on available scientific data; some of the available data may need to be reassessed through additional research; and there may be complex ecological interactions (e.g., trophic level interactions, presence of additional algal species, etc.) which may affect impacts which are not fully addressed in the table; Categories are relative and related to different threshold concentrations of brown tide. *(Continued)*

Cells ml^{-1}	Index (Category)	Potential Impact	Sources
		• **Bay Scallops** (*Argopecten irradians*) Massive recruitment failures of the 1985 year class of the bay scallop, *Argopecten irradians*, as a result of brown tide blooms in Long Island embayments	Cosper et al., 1997
		High mortalities (up to 64–82%) of adult bay scallops in Peconic Bay after the 1995 brown tide bloom (densities reached 0.8–2.2 $\times$ 10^6 cells ml^{-1} measured from incidence of articulated "clucker" shells).	C. Smith, pers. comm. in Bricelj and Lonsdale, 1997
		Aureococcus anophagefferens causes significant growth reduction and high mortalities of *Argopecten irradians* larvae at concentrations of 190,000–750,000 cells ml^{-1}.	Gallagher et al., 1989
		Field data suggest inhibitory effects on growth of bay scallops at ca. 2 $\times$ 10^5 *Aureococcus* cells ml^{-1} in Long Island Bays	Bricelj et al., 1987
		• **Hard Clams and Mussels**	
		No significant growth (measured by change in the ash-free dry weight or organic weight of juvenile hard clams 6 mm in initial shell length) of juvenile hard clams at concentrations of the same isolate of *Aureococcus* $\geq$400,000 cells ml^{-1}; similar results with juvenile mussels	Bricelj, 1999
		Observations of a reduction in feeding and development of hard clam larvae, reported by a commercial aquaculture facility, during a 1995 brown tide bloom in Tuckerton Bay, N.J., with Aureococcus cell counts ranging from 1.1 to 1.8 $\times$ 10^6 cells ml^{-1}.	Nuzzi et al., 1996
		Reports of reductions in juvenile hard clams during 1999 and 2000 brown tide blooms in Little Egg Harbor (as reported by Biosphere, Inc., an aquaculture facility in Tuckerton, NJ), with *Aureococcus* counts reported $>10^6$ cells ml^{-1} in 1999, $>2.0 \times 10^6$ cells ml^{-1} in June 2000, and >240,000 cells ml^{-1} in June 2001	Gastrich, 2000a, b; Gastrich, 2001; Gastrich et al., 2000; NJDEP, 1999
		• **Macrobenthos Impacts**	
		Negative impacts to macrobenthos such as eelgrass, *Zostera marina* (e.g., die-off) at *Aureococcus* densities of 0.05 to 2.6 $\times$ 10^6 cells ml^{-1} with a mean of 0.66 $\times$ 10^6 cells ml^{-1} leading to an increase in light scattering and a severe reduction in light penetration) and bay scallops, *Argopecten irradians* in Long Island Bays (*Aureococcus* concentrations $>10^6$ cells ml^{-1})	Dennison et al., 1989; Cosper et al., 1987; Bricelj et al., 1987
		• **Planktonic Impacts**	
		From the onset of brown tide in West Neck Bay, N.Y. in 1995 to the peak (*Aureococcus* concentrations of 1.1 $\times$ 10^6 cells ml^{-1} microzooplankton population declined from >10,000 to <900 ind. l^{-1}.	Mehran 1996
		Copepod production in Narragansett Bay in 1985 was reduced at *Aureococcus* concentrations of 7.6 $\times$ 10^5 cells ml^{-1}.	Durbin and Durbin, 1989
		Copepod production was also reduced in West Neck Bay, NY at *Aureococcus* concentrations of 1.5 $\times$ 10^6 cells ml^{-1}.	Lonsdale et al., 1996

Table 3 - 17 Defined thresholds for Harmful Algal Blooms

	HABS THRESHOLDS
	A. anophagefferens
SCORE	cells mL^{-1}
100	30,000
75	90,000
50	150,000
25	200,000

Table 3 - 18 Literature values for dissolved oxygen effects on benthic invertebrates (from Ritter and Montagna 1999)

TABLE 4. Community response to different hypoxia intensity categories. Hypoxic categories are based on the average environmental and community characteristics of stations falling within each category. DO = dissolved oxygen.

Hypoxia Intensity Category	DO (mg l^{-1})	Biomass (g m^{-2})	Density (no. m^{-2})	No. Species (0.01 m^{-2})	No. Dominant Species (N1)	Dominant Species
1	>5	7.64	22,171	20	9.45	Codominance
2	4–5	3.30	7186	12	7.65	*Mediomastus ambiseta*
3	3–4	—	—	—	—	—
4	2–3	0.42	3144	3	1.63	*Streblospio benedicti*
5	1–2	0.01	189	1	1	Oligochaeta
6	0–1	—	—	—	—	—

Table 3 - 19 Literature values of thresholds used in the Chesapeake Bay B-IBI (from Weisberg et al. 1997)

TABLE 6. Thresholds used to score each metric of the Chesapeake Bay B-IBI.

	Scoring Criteria		
	5	3	1
Tidal Freshwater			
Shannon-Weiner	≥1.8	1.0–1.8	<1.0
Abundance (# m^{-2})	≥1,000–4,000	500–1,000 or ≥4,000–10,000	<500 or ≥10,000
Biomass (g m^{-2})	≥0.5–3	0.25–0.5 or ≥3–50	<0.25 or ≥50
Abundance of pollution-indicative taxa (%)	≤25	25–75	>75
Oligohaline			
Shannon-Weiner	≥2.5	1.9–2.5	<1.9
Abundance (# m^{-2})	≥1,500–3,000	500-1,500 or ≥3,000-8,000	<500 or ≥8,000
Biomass (g m^{-2})	≥3–25	0.5–3 or ≥25–60	<0.5 or ≥60
Abundance of pollution-indicative taxa (%)	≤25	25–75	>75
Abundance of pollution-sensitive taxa (%)	≥40	10–40	<10
Low Mesohaline			
Shannon-Weiner	≥2.5	1.7–2.5	<1.7
Abundance (# m^{-2})	≥1,500–2,500	500–1,500 or ≥2,500–6,000	<500 or ≥6,000
Biomass (g m^{-2})	≥ 5–10	1–5 or ≥10–30	<1 or ≥30
Abundance of pollution-indicative taxa (%)	≤10	10–20	>20
Biomass of pollution-sensitive taxa (%)	≥80	40–80	<40
Biomass >5 cm below sediment-water interface (%)	≥80	10–80	<10
High Mesohaline sand			
Shannon-Weiner	≥3.2	2.5–3.2	<2.5
Abundance (# m^{-2})	≥1,500–3,000	1,000–1,500 or ≥3,000–5,000	<1,000 or ≥5,000
Biomass (g m^{-2})	≥3–15	1–3 or ≥15–50	<1 or ≥50
Abundance of pollution-indicative taxa (%)	≤10	10–25	>25
Abundance of pollution-sensitive taxa (%)	≥40	10–40	<10
Abundance of carnivores and omnivores (%)	≥35	20–35	<20
High Mesohaline mud			
Shannon-Weiner	≥3.0	2.0–3.0	<2.0
Abundance (# m^{-2})	≥1,500–2,500	1,000–1,500 or ≥2,500–5,000	<1,000 or ≥5,000
Biomass	≥2–10	0.5–2 or ≥10–50	<1,000 or ≥5,000
Biomass of pollution-indicative taxa (%)	≤5	5–30	>30
Biomass of pollution-sensitive taxa (%)	≥60	30–60	<30
Abundance of carnivores and omnivores (%)	≤25	10–25	<10
Biomass >5 cm below sediment-water interface (%)	≥60	10–60	<10
Polyhaline sand			
Shannon-Weiner	≥3–5	2.7–3.5	<2.7
Abundance (# m^{-2})	≥3,000–5,000	1,500–3,000 or ≥5,000–8,000	<1,500 or ≥8,000
Biomass (g m^{-2})	≥5–20	1–5 or ≥20–50	<1 or ≥50
Biomass of pollution-indicative taxa (%)	≤5	5–15	>15
Abundance of pollution-sensitive taxa (%)	≥50	25–50	<25
Abundance of deep-deposit feeders (%)	>25	10–25	<10
Polyhaline mud			
Shannon-Weiner	≥3.3	2.4–3.3	<2.4
Abundance (# m^{-2})	≥1,500–3,000	1,000–1,500 or ≥3,000–8,000	<1,000 or ≥8,000
Biomass (g m^{-2})	≥3–10	0.5–3 or ≥10–30	<0.5 or ≥30
Biomass of pollution-indicative taxa (%)	≤5	5–20	>20
Biomass of pollution-sensitive taxa (%)	≥60	30–60	<30
Abundance of carnivores and omnivores (%)	≥40	25–40	<25
Taxa >5 cm below sediment-water interface (%)	≥40	10–40	<10

Table 3 - 20 Defined thresholds for Benthic Invertebrates

	BENTHIC INVERTEBRATES THRESHOLDS
	EMAP Index Value
SCORE	Index units
100	2
75	1
50	0
25	-1
0	-2

Table 3 - 21 Eigenvectors and annual weightings for Water Quality indicators under Scenario 1.

	WQ1 Eigenvector				Weighting for WQ1			
Year	*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 - 22 Eigenvectors and multi-year weightings for Water Quality indicators under Scenario 2.

	WQ1 Eigenvector				Weighting for WQ1			
Year	*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 - 23 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 (ºC)	Salinity (ppt)	Specific Conductivity	Dissolved Oxygen (mg L^{-1})	Dissolved Oxygen (%)	pH	Depth (cm)
North	Jun-Jul	30	23.6 (0.5)	19.2 (0.3)	30.9 (0.5)	7.9 (0.6)	103.6 (8.8)	8.2 (0.2)	-
North	Aug-Sep	30	22.9 (0.3)	15.5 (1.2)	25.4 (1.9)	7.9 (0.8)	100.0 (11.2)	7.7 (0.2)	99.8 (12.8)
North	Oct-Nov	30	14.5 (0.7)	18.7 (0.1)	30.1 (0.2)	10.0 (0.4)	110.1 (5.5)	7.9 (0.1)	119.8 (8.1)
Central	Jun-Jul	60	24.2 (1.6)	24.7 (2.9)	38.6 (4.3)	8.4 (1.3)	115.5 (17.3)	8.1 (0.1)	84.0 (31.7)
Central	Aug-Sep	60	25.6 (1.7)	24.4 (4.5)	38.4 (6.5)	7.7 (1.7)	107.5 (22.0)	8.0 (0.2)	114.1 (17.8)
Central	Oct-Nov	60	16.4 (1.7)	26.9 (5.1)	41.8 (7.1)	9.0 (1.8)	108.6 (22.3)	7.9 (0.1)	132.3 (36.5)
South	Jun-Jul	60	22.7 (1.5)	29.3 (0.1)	45.2 (0.2)	8.1 (0.7)	111.3 (9.5)	8.0 (0.1)	87.8 (25.3)
South	Aug-Sep	60	27.0 (1.2)	30.0 (0.2)	46.3 (0.3)	6.4 (1.0)	95.1 (14.9)	7.9 (0.1)	102.4 (27.5)
South	Oct-Nov	60	16.7 (0.9)	27.3 (0.6)	42.4 (0.9)	9.3 (0.5)	112.5 (6.2)	8.0 (0.1)	108.1 (14.2)

Standard deviations in parentheses

Table 4 - 2 Characteristics of submerged aquatic vegetation (SAV) by sampling period in the BB-LEH Estuary during 2011.

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

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

Standard deviations in parentheses

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

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

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

Table 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 i - 1 Quality Assurance Project Plan for this project

THIS IS A PLACEHOLDER SPOT FOR THE QAPP.

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

QUALITY ASSURANCE PROJECT PLAN (QAPP)

Prepared by Ronald J. Baker, U.S. Geological Survey, West Trenton, New Jersey

And Michael J. Kennish, Rutgers University, New Brunswick, New Jersey

QAPP REVISION 8, FEBRUARY 14, 2011

A1. PROJECT DESCRIPTION AND SIGNATURES

PROJECT DURATION: February, 2009-March, 2013

ABSTRACT

Nutrient loading to the Barnegat Bay-Little Egg Harbor (BB-LEH) Estuary will be quantified from water-quality data, atmospheric data, and loading models and related to biotic indicators of eutrophication for biotic-index development to define the estuarine ecosystem condition. Results will include thresholds of biotic and numerical loading criteria to support nutrient management planning. The ecosystem-based project will address five important elements. First, it will characterize and model how land-use decisions in the BB-LEH watershed impact nutrient fluxes to the BB-LEH estuary. All available hydrologic, water-quality, meteorological, and land-use data will be compiled and used in conjunction with watershed loading models to determine local and estuary-wide nutrient loading. Second, it will determine whether the biotic response to nutrient enrichment in the estuary represents a stable, continuous gradient or exhibits significant seasonal and inter-annual variability. Third, it will quantify to what extent variability in nutrient loading and biotic responses differs among subwatershed areas. An overall eutrophic condition index of the estuary will be calculated by integrating numeric values of key water-quality and biotic indicators across estuarine segments. This index value will serve as a standard against which future assessments of estuarine impairment can be compared. Fourth, a eutrophication conceptual model will be applied to determine if there has been significant alteration of estuarine ecological structure and function.

Fifth, threshold levels of biotic decline and numeric nutrient loading criteria will be developed for the estuary, and discussion of how these threshold levels can be integrated into a management plan will be given.

This Quality Assurance Project Plan (QAPP) was developed and organized in a manner consistent with guidance documents prepared by the USEPA (EPA/240/R-02/009 and Region 2 QAPP Guidance, Revision #1, April, 2004) and the New England Water Pollution Control Commission (NEIWPCC Guide For Development and Approval Of Quality Assurance Project Plans, March, 2006). Throughout the document, activities that use exclusively secondary data sources are presented separately from activities that involve collection of new data, because the quality-assurance requirements of these two categories of data used in this investigation are entirely different. All pre-existing (i.e., secondary) physicochemical data collected over the 1989 to 2011 period and used in this project will have been collected and analyzed in state certified laboratories of the NJDEP and will have been collected and analyzed consistently using the same methods for each parameter.

PROJECT 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 develop more sensitive modeling of loading and to determine relative contributions of nutrient loadings from lawn care practices protected riparian buffers, and stormwater management systems (SWMS).

- To determine estuarine biotic responses to the loading of nutrients across a gradient of upland watershed development and associated estuarine nitrogen loading, and 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. Each of these parameters will be examined and assessed for statistical validity and inclusion in the index development for the 1989 to 2011 period.

- To delineate 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 2011.

- To develop a biotic index of estuarine condition using water quality and biotic indicators to assess eutrophication, impairment, and overall ecosystem health of the BB-LEH estuary and formulate threshold levels of biotic decline and numeric loading criteria that can support an effective nutrient management plan.

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Benjamin M. Fertig

The U.S. Department of the Interior, U.S. Geological Survey, New Jersey Science Center, W. Trenton, NJ

Ronald J. Baker
Christine Wieben
Robert S. Nicholson

FUNDING AGENCY

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Signature _____ Date _____

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Michael J. Kennish (Rutgers)
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Benjamin M. Fertig
Gregg Sakowicz (Rutgers)
Robert S. Nicholson (USGS)
Ronald J. Baker (USGS)
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Marc Ferko (NJDEP)
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Michael Jennings (NEIWPCC)

A4. PROJECT/TASK ORGANIZATION (RESPONSIBILITIES)

ORGANIZATIONAL CHART				
FUNCTION	**NAME**	**TITLE, AFFILIATION**	**REPORTS TO:**	**RESPONSIBILITY**
Overall Project Manager	Michael J. Kennish kennish@marine.rutgers.edu	Research Professor, School of Environmental and Biological Sciences, Institute of Marine and Coastal Sciences, Rutgers University, New Brunswick, NJ	Francisco Werner, Director, Institute of Marine and Coastal Sciences, Rutgers University	Project lead; oversee estuary sampling and data analysis, bioindicator assessment, biotic index development; responsible for overall project management. Responsible for maintaining the official, approved QA Project Plan
Program Manager, USGS	Robert S. Nicholson *rnichol@usgs.gov*	Supervising Hydrologist, Environmental Studies Program Chief, USGS NJ Water Science Center	Anthony S. Navoy, Assistant Director, USGS NJ Water Science Center	U.S. Geological Survey lead; will oversee the nutrient loading component of the project
Project Manager, USGS	Ronald J. Baker *rbaker@usgs.gov*	Research Hydrologist, U.S. Geological Survey, W. Trenton, NJ	Robert S. Nicholson , Environmental Studies Program Chief, USGS NJ Water Science Center	Will determine temporal and spatial variability of nutrient loading from the watershed with the aid of mathematical models
Project Quality Assurance Manager, USGS	Jacob Gibs *jgibs@usgs.gov*	Water Quality Specialist, U.S. Geological Survey, W. Trenton, NJ	Richard H. Kropp, Director , USGS NJ Water Science Center	Responsible for QA of all water-quality data for USGS/New Jersey
Overall Project Quality Assurance Manager, Rutgers	Gregg Sakowicz sakowicz@imcs.rutgers.edu	Field researcher, Institute of Marine and Coastal Sciences, Rutgers University	Francisco Werner, Director, Institute of Marine and Coastal Sciences, Rutgers University	Will oversee and review data acquisition, analyses, and protocols. Ensures compliance with all elements of the QAPP
Project Staff, Rutgers	Richard G. Lathrop, Jr. lathrop@crssa.rutgers.edu	Professor of Environmental Monitoring, Department of Ecology, Evolution & Natural Resources Cook College, Rutgers University New Brunswick, NJ	Peter Morin, Chair Department of Ecology, Evolution & Natural Resources Cook College, Rutgers University New Brunswick, NJ	Land-use profile development, watershed data analysis, biotic index development
Project Staff, Rutgers	Benjamin M. Fertig bfertig@umces.edu	Center for Remote Sensing and Spatial Analysis, Rutgers University, New Brunswick, NJ	Michael J. Kennish and Richard G Lathrop	Technical support in all estuary field sampling, as well as watershed and estuary data analysis
Project Staff, USGS	Christine M. Wieben *cwieben@usgs.gov*	Hydrologist, U.S. Geological Survey, W. Trenton, NJ	Robert S. Nicholson , Environmental Studies Program Chief, USGS NJ Water Science Center	Will assist in nutrient loading modeling by obtaining and managing data

PROJECT ADMINISTRATION			
FUNCTION	**NAME**	**TITLE, AFFILIATION**	**EMAIL**
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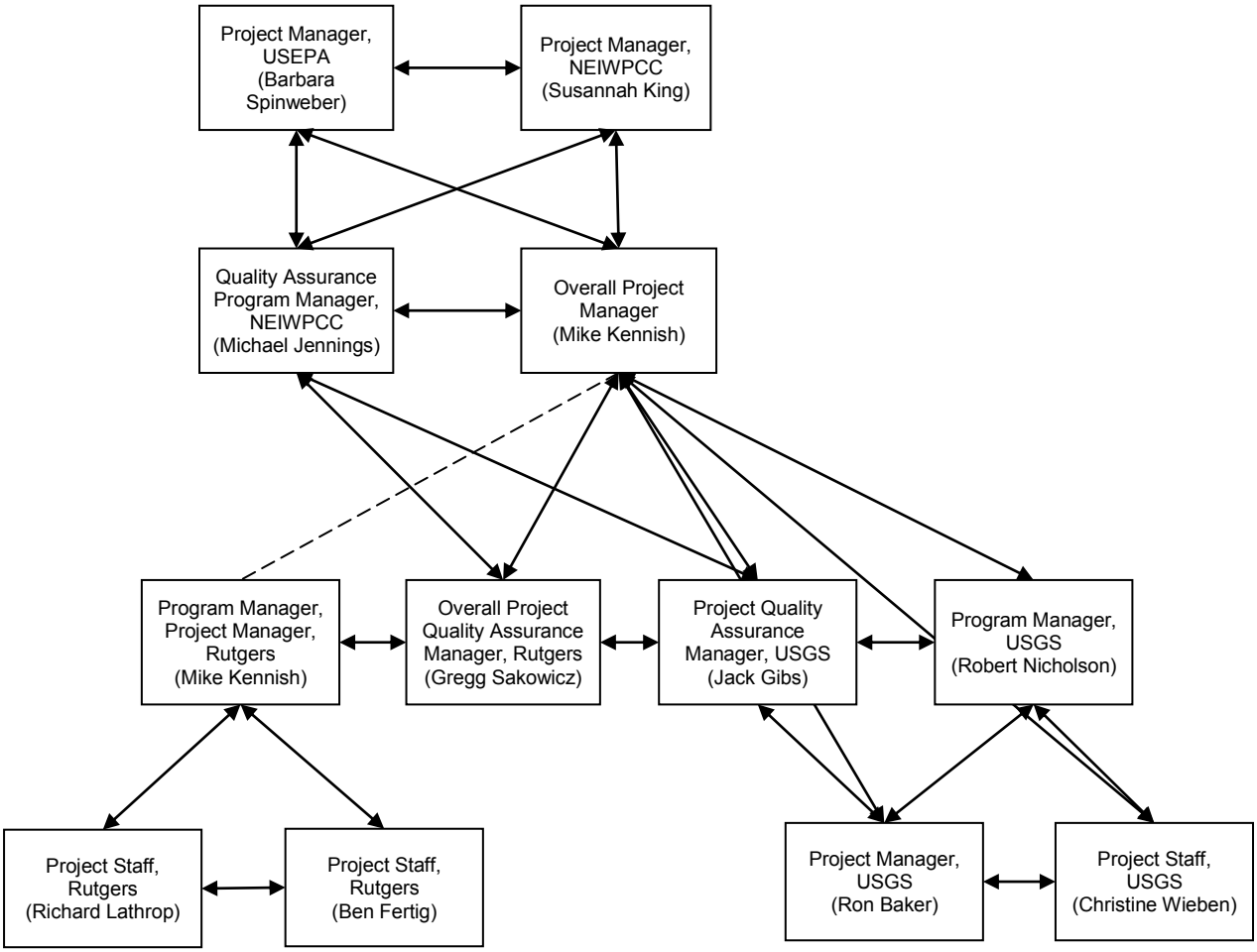
Quality Assurance Program Manager	Michael Jennings	NEIWPCC	mjennings@neiwpcc.org
Quality Assurance Officer	Donna Ringel	USEPA	ringel.donna@epa.gov

Organizational chart- lines of communication



A5. PROBLEM DEFINITION/BACKGROUND

Quantitative loading criteria for nitrogen and phosphorus compounds above which impairment of ecosystem structure and function occurs have not been established in U.S. estuaries (Hameedi et al., 2007). A regional approach for developing nutrient criteria and standards can be found in the EPA document titled, National Strategy for the Development of Regional Nutrient Criteria (1998). However, estuaries 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 investigation targets the BB-LEH estuary in New Jersey as a case study. This estuary provides an ideal setting to examine the effects of eutrophication in coastal bays because it is a shallow, poorly flushed lagoonal system impacted by nutrient enrichment (Kennish, 2007; Kennish et al., 2007a). Environmental pressures from land development in the watershed are expected to increase (Lathrop and Haag, 2007). Nutrient enrichment in the estuary is a function of land-use patterns and can result in changes in ecosystem structure and function such as the composition of micro- and macroalgae, occurrence of harmful algal blooms (HABs), loss of seagrass habitat, altered benthic invertebrate communities, diminished shellfish harvest, explosions of stinging jellyfish populations, and shifts in food webs (Kennish et al., 2007a). However, the linkage between loading stress and these effects have not been unequivocally established at this time. These priority problems also exist in many other estuaries worldwide, most notably coastal lagoons (Kennish, 2002).

This multidisciplinary analysis will quantify spatial and temporal relationships between nutrient loading and biotic responses in the BB-LEH estuary. As a component of this effort, we will assess key biotic response variables in the estuary (i.e., seagrass, phytoplankton, macroalgae, epiphytes, and shellfish resources) and nutrient loading associated with human-altered land use in the adjoining BB-LEH watershed. We suggest that, as shallow coastal bays eutrophy, there is a chaotic period when a variety of alternate ecological states is possible, depending upon weather/climate and nutrient dynamics. Key steps in the process will be to establish accurate nutrient loading values for the watershed, threshold levels of biotic decline, and numeric measures of bioindicators of ecosystem condition. In order to sustain and restore the health of our coastal aquatic ecosystems, we need a better understanding of the relative importance of the predominant sources of nutrient pollution and their relation to regional land-use patterns. Data sources will consist of secondary (pre-existing) water-quality data in the watershed and in the estuary, and newly-collected biotic data. General water-quality-parameter data such as pH, specific conductance, temperature and dissolved oxygen will be collected with automatic data sondes. New Jersey State certification will be obtained for this activity. The project will employ spatially explicit modeling of watershed sources of nutrients to determine the contribution of the waterborne sources of nitrogen to the estuary from subwatersheds. By coupling the nutrient loading models with our in situ sampling of biotic responses in the estuary, we will be able to characterize the spatial and temporal dynamics of the nutrients within the estuarine system which could be used to establish the basis for developing accurate nutrient loading criteria. Based on these findings, we will model how estuarine health will likely change as a result of several important policies for land use and nutrient pollution control.

A6. PROJECT/TASK DESCRIPTION

The study area consists of the entire watershed and water-body referred to as the Barnegat Bay-Little Egg Harbor (BB-LEH) Estuary. The Barnegat Bay–Little Egg Harbor

Estuary is a shallow, lagoonal back-barrier system located along the central New Jersey coastline between 39°31' N and 40°06' N latitude and 74°02' W and 74°20' W longitude (Figure 1).

The investigation will be conducted in five components. In **Component 1**, loading of nutrients to Barnegat Bay will be quantified by using all relevant data sources that meet the data-quality objectives of the project. Nutrient loads of fresh water reaching the estuary will be quantified annually and seasonally for all sub-watersheds at the HUC-14 resolution. It may later be necessary to aggregate HUC-14-scale results into results for larger areas in order to provide loading information that corresponds to the scale of the biotic investigation (Components 2 and 3). Processes that occur at the tidal interface, such as tidal fluxes, salinity gradients and chemical speciation will not be considered. Stormwater basin mapping will be considered in the evaluation of the effect of land use on water quality in the watershed. In **Component 2**, the biotic responses in the bay to temporally and spatially variable nutrient loads over the 1989 to 2011 period will be analyzed and reported. In **Component 3**, a biotic index of condition for the BB-LEH estuary will be computed from data collected on key water quality and biotic indicators during the 1989 to 2011 period. In **Component 4**, additional sampling and data analysis will be conducted in 2010 to assess the current status of eutrophication of the estuary. This component will also provide information to validate biotic responses in previous years. In **Component 5**, synthesis and management recommendations of the project will be advanced. The application of our findings in developing nutrient-management plans will be considered in this component.

COMPONENT 1: QUANTIFICATION OF NUTRIENT LOADING (USGS)

- The most recent and comprehensive data available will be used to determine current nitrogen and phosphorus loading from the BBLEH watershed to the estuary. Nutrient loading will be determined from:
    - Direct calculation where sufficient data are available
        - Hydrologic data (stream flow)
        - Water quality (concentrations of species of interest)
    - Model simulations where sufficient data are not available, and will rely on:
        - Nutrient loading values from sub-basins for which direct calculation are possible
        - Atmospheric-deposition data
        - Land-use-pattern data
        - Precipitation data

Two modeling tools will be used to relate explanatory variables to nutrient loading:

- PLOAD (CH2M Hill, Inc,) is a GIS-based modeling tool that calculates pollutant loading for watersheds on an annual basis by using established correlations between basin size and different land-use types, and loading of the pollutant under consideration (nitrogen and phosphorus species in this case), and
- BASINS3 (USEPA) (Better Assessment Science Integrating Point and Nonpoint Sources) functions as a geographic information system (GIS), a depot for storing and organizing data to be used in pollutant-loading modeling, and a "shell" program in which PLOAD will be run.

The combination of BASINS3 and PLOAD was selected for a number of reasons. BASINS 3 includes a comprehensive geographic information system (GIS) as its framework. This allows for analyzing landscape and land-use information and displaying relationships among variables of interest. PLOAD is included as a component of BASINS3. It is used to estimate nonpoint loads of pollutants on an annual average basis. This is appropriate for the BB-LEH watershed, because most nutrient contributions are non-point in nature. BASINS3 allows users to import their own data layers (elevation, land use, soil data, streams, point-sources) in shape file or grid file formats. Thus, either export coefficients (calculated from water-quality data) or loading-rate estimates calculated from land use, impervious surface, precipitation, and fraction of storms producing runoff can be used. Spatial variability in nutrient loading will be addressed by applying the model to subbasins of the overall study area. Temporal variability will be addressed in two ways: changes in loading variability over time will be studied by comparing model results for different years; **intra-annual** (seasonal) variability will be studied by applying the model separately to data collected during growing and nongrowing seasons. Then nutrient loading in the growing season will be compared to that in the nongrowing season for a given subbasin, and for the BBLEG estuary as a whole.

Other modeling systems could be considered. For example, SPARROW (SPAtially-Referenced Regression On Watershed attributes) Statistical methods are used in SPARROW modeling to explain in-stream measurements of water quality (constituent mass or load) in relation to upstream sources and watershed properties (soil characteristics, precipitation amounts, and land cover) that influence the transport of constituents to streams and their delivery to receiving water bodies, including estuaries. This modeling system is better suited for describing relationships between land use and water quality for large, regional areas, such as the Mississippi River watershed than for smaller areas such as BB-LEH. Additionally, it is descriptive and not predictive. NLOAD (a web-based nitrogen loading tool) could be used as an alternative to BASINS3-PLOAD. However, it is limited in its ability to include user-supplied data, and is not GIS-based. Comprehensive watershed models such as WASP require data not readily available for BB-LEH in order to achieve more accurate loading estimates than will be obtained with BASINS3-PLOAD.

Component 1 will be divided into four tasks: task 1. Selecting and characterizing the study area; task 2. Compile all data to be used in nutrient-loading determination; task 3. Calculate loading with the aid of mathematical models; and task 4. Determine the contributions of turf areas to the non-point-source loading of nutrients to the BB-LEH watershed

Task 1

The first task will be to define the study area with respect to nutrient-loading determination for the BB-LEH Estuary. In addition to the entire BB-LEH watershed, watersheds within about 100 miles North and South of the watershed boundaries will be evaluated for comparability with BB-LEH. If areas are found which can improve the accuracy of loading estimates, data from these areas will be incorporated into the loading-estimation procedures. Criteria for including such areas will be one or more similarities to BB-LEH:

- Topography/hydrology (similar to BB-LEH watershed)
- Land-use patterns (similar to BB-LEH watershed and no substantial point sources of nutrients
- Water quality (Nutrient data available that is consistent with data-quality objectives)
- Precipitation (similar to BB-LEH watershed)

- Atmospheric deposition (Reporting stations near areas of interest)

Task 2

The second task will be to compile all data that can be used for estimating nutrient loading. Only secondary (pre-existing) data will be used here, and no new sampling or analysis will be conducted. Data to be compiled, for the BB-LEH watershed and nearby watersheds that can be used to enhance loading estimates will include:

- Hydrologic (stream flow, stage, and rating curves, as available, from USGS database NWIS/ADAPS)
- Water-quality (all nutrient species, other chemical and physical measurement data, as available)
    - From the USGS database NWIS/QWDATA
    - From the New Jersey Pinelands Commission
    - From the New Jersey Department of Environmental Protection
    - Brick Township
    - From published sources
    - From other sources not yet identified
- Atmospheric-deposition data
    - From the appropriate stations of the National Atmospheric Deposition Program and USEPA CASTNET Program database
- Precipitation data
    - From the National Climatic Data Center *(http://www.ncdc.noaa.gov/oa/ncdc.html*) and Office of the New Jersey State Climatologist (*http://climate.rutgers.edu/stateclim_v1/monthlydata/index.html)*
- Land-use-pattern data from 1970s, 1986, 1995-97, 2002, 2007
    - From NJDEP (2003) and published reports (to be identified) (USGS, NJDEP, journals).
    - Turf data from CRSSA (Task 4)
- Geographic Information System (GIS) data
    - Coverages of the BB-LEH watershed currently reside on the USGS-NJ computer system. The ESRI program ARCmap will be used to organize and manipulate GIS data.
    - The new Hydrologic Unit Code 14 (HUC14) delineations developed by the NJDEP will be used.

All hydrologic, water-quality, precipitation and atmospheric-deposition data will be compiled in a Microsoft Access database. All geospatial and land-use data will be compiled in a geo-database.

Task 3

The third task will be to use all data described above in conjunction with the mathematical models BASINS3 and P-LOAD to determine loading rates of nutrients to the BB-LEH Estuary from all substantial streams, from direct-ground-water discharge, and from atmospheric deposition. Total loading to the estuary and relative contributions from the different sources will be estimated. Resolution of model simulations will be at the HUC-14 level. Elements of this task are:

- Obtain and enable up-to-date versions of the two models
- Prepare input files as required by the models
- Develop QA procedures as determined from model documentation
- Develop a database for archiving all model development and simulation activity
- Develop simulations using an iterative procedure, where models of increasing complexity are created, until simulation of all areas of interest has been completed satisfactorily.

PLOAD is not a regression or interpolation model. Rather, it is a series of mathematical expressions that directly calculates loading (of and constituent) from water quality and basin characteristics. Accuracy and precision of the model outputs will depend entirely upon the quality of input data, and applicability of the model relations to the watershed. This includes quality of the water-quality, streamflow, impervious surface, land-use and precipitation data (discussed in Section A7, Component 1).

Task 4

We will be mapping and characterizing a full suite of land uses from the 1970s to the present. The focus on turf is that this is land cover that has been inadequately mapped and quantified in the past. We will be mapping this for the first time to better characterize its spatial distribution and the intensity of management across the sub-watersheds of the Barnegat Bay-Little Egg Harbor system. We do have historical imagery and land use maps from the 1930s to characterize spatial distribution of the possible historical signal of agriculture inputs through the groundwater.

The focus on turf areas as sources of elevated nutrient loading is also due the absence of other known significant sources of nutrients such as agricultural land use, industry, and discharges from wastewater treatment plants to streams in the watershed. Large areas of the watershed are developed with single-family dwellings with lawns, and quantifying the nutrient contributions from these areas is an objective of the investigation. Quantitative nutrient loading information specifically for turf coverage is not available and existing literature usually relates loading to generalized land-use categories (such as residential-urban) or to estimated impervious surface area. The unique development characteristics of this watershed (lack of agriculture or point sources, and large tracts of housing with well-maintained lawns) will enable the study of relations between turf areas and nutrient loading. Determining these relations, whether or not a substantial portion of the nutrient loading is from this source, is an important component of this study.

Delineation of Turf areas within the BB-LEH watershed will be completed as Task 4. Erdas Imagine and eCognition will be used to view and analyze aerial and satellite imagery in this project. A geographic information systems (GIS) data layer showing turf areas for the Barnegat Bay watershed will be created using the New Jersey Department of Environmental Protection (NJDEP) spring 2007 color infrared aerial photography and a August, 2008 United Stated Department of Agriculture (USDA) panchromatic (RGB) aerial photography. These photography missions will be analyzed with the e-cognition software package to create vector objects from the geo-referenced raster datasets. Initially the resultant polygons will be classified as either not turf, or turf. A second analysis will be run to break turf areas into either managed (water or fertilizer) or unmanaged turf areas. A training dataset will be collected by randomly selected N polygons from the study area and classifying them by on screen comparison to aerial photography. This training dataset will be used to classify vector objects based on the

aggregate digital numbers of the original raster datasets. The significance of individual variables will be determined by a random forest version of the Cartographic and Regression Tree (CART) analysis. A reference manual for this statistical technique is located at *http://cran.r-project.org/web/packages/randomForest/randomForest.pdf*. Random forest uses a bootstrap version of the CART model without replacement. Bootstrapping involves randomly sub-sampling the training dataset, running a CART model and then using the remaining training dataset to compute an accuracy assessment. This technique provides an un-biased accuracy assessment while allowing the full training dataset to be used in the final model creation.

An additional accuracy assessment of the turf areas will be created by randomly selecting100 points on the landscape and the buffering them by ~ 36 meters. This will be done to create vector circles equal to 1 acre in size (radius of 35.9 meters). For each of these vector circles an operator will hand digitize turf areas showing both managed and unmanaged turf areas. These areas will be compared to the e-cognition vector polygons to show both errors of commission and errors of omission by area. In addition, the random forest model will be used to create a 95% confidence interval for the kappa statistic.

Timeline for Component 1

Item	Begin	Complete
Data collection (existing water-quality and streamflow, Atmospheric deposition, land use, meteorological)	05/29/2009	12/05/2011
Obtain, install and register models (BASINS 3, PLOAD)	06/01/2009	07/15/2009
Prepare input files for models	04/19/20109	10/28/2011
Conduct model simulations, calibrations	06/01/2010	12/12/2011
Loading estimates based on data and simulations	09/20/2010	12/23/2011

COMPONENTS 2-4: ESTUARINE BIOTIC RESPONSE; BIOTIC INDEX DEVELOPMENT; AND CURRENT (2010) EUTROPHICATION ASSESSMENT

In **Component 2**, the biotic responses in the bay to temporally and spatially variable nutrient loads over the 1989 to 2011 period will be analyzed and reported. In **Component 3**, a biotic index of condition for the BB-LEH estuary will be computed from data collected on key water quality and biotic indicators during the 1989 to 2011 period. In **Component 4**, additional sampling and data analysis will be conducted in 2011 to assess the current status of eutrophication of the estuary. This component will also provide information to validate biotic responses in previous years. Components 2-4 will use a combination of secondary (pre-existing) and new data. Only secondary nutrient, sediment and chlorophyll data will be used. New basic water-quality data will be collected with the use of automated data sondes, by personnel from a state-certified facility. Biotic data will be collected by Rutgers personnel. QA procedures are specified in the sections that follow. The sensitivity requirements for data sonde measurements in this project are listed in Appendix 1. They are appropriate for the parameters to be measured in this project. If any additional data are deemed necessary during the course of the investigation, corrective action will be taken by discussion among Project Management (Rutgers and USGS) and Project Administrative Management (USEPA, NEIWPCC and NJDEP). The QAPP will then be updated to reflect any agreed-upon changes to the project.

The work schedule for Components 2-4 is shown below.

Timeline for Components 2-4

Item	Begin	Complete
Data collection (field and secondary data) for developing biotic responses to nutrient loading	06/01/2009	11/30/2011
Biotic index development	04/01/2011	04/01/2012
Relationship between nutrient loading and biotic responses	03/01/2011	05/31/2012
Collect additional field data for model verification	06/06/2011	11/30/2011

COMPONENT 5: SYNTHESIS AND MANAGEMENT RECOMMENDATIONS (INCLUDING REPORTS AND PRESENTATION AT THE TECHNOLOGY TRANSFER SYMPOSIUM)

In **Component 5**, synthesis and management recommendations of the project will be advanced. The use of our findings in nutrient management plans will be considered in this component.

Timeline for Component 5

Item	Begin	Complete
Development synthesis and management recommendations	01/01/2012	05/31/2012
Prepare draft report, submit for review	06/01/2012	10/01/2012
Address review comments and submit final report	10/01/2012	12/31/2012
Present results at Technology Transfer Symposium	March, 2013	March, 2013

A7. QUALITY OBJECTIVES AND CRITERIA

COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

All nutrient-loading estimates for the BB-LEH estuary will be developed from secondary (pre-existing) data. All appropriate elements from the USEPA document "QAPP Guidance for Projects Using Secondary Data", EPA NE Secondary Data Guidance, Revision 2, 09/10/03 were included in the development of this QAPP. Data-quality concepts and procedures will be discussed in this section, organized by data source. Water-quality data will be gathered from many sources, and values for a given parameter may have been obtained by using multiple analytical methods. In such cases, data quality will be reviewed to ensure that all analyses meet the data-quality objectives in terms of accuracy, precision, and repeatability. Any data found to not meet these data quality objectives will not be used in this investigation.

USGS Data

All USGS data that will be used in this investigation resides in the National Water Information System (NWIS), a storage and retrieval system of water data collected through its

activities at approximately 1.5 million sites around the country. These data, which are publicly available and downloadable from *http://waterdata.usgs.gov/nwis*, were collected by USGS personnel for many purposes over years starting in 1899. NWIS is comprised of the Ground Water Site Inventory (GWSI), the Automated Data Processing System (ADAPS), the Water Quality System (QWDATA), and the Site Specific Water Use Data System (SWUDS). Additional information about NWIS is available at: *http://nwis.usgs.gov/*. NWIS data are considered "provisional, subject to revision" by the USGS until they have been published. If the data are subsequently published (usually in a USGS Annual Report produced by each water-science center, a USGS scientific publication, a journal article, or a chapter of a book or monograph) they are termed "finalized". The process of publishing a report or other product that contains NWIS data includes a requirement that the data must be reviewed and approved by an approving official (generally a Water Quality Specialist at a USGS office). When retrieving data from NWIS, however, no distinction is made between provisional and finalized data. Therefore, in an investigation such as this, data retrieved from NWIS must be reviewed by the user before they can be used with confidence. The data review will entail ensuring proper units were used, that detection limits and quantitation limits are consistent with the data quality objectives, that the data were collected after 1980, as older data may be less reliable, and that numerical data appear reasonable with respect to the constituent and the sample (for example, a pH value of 1.2 would not be considered reasonable for stream water anywhere in the BB-LEH watershed) Notes or remarks that accompany the data will be reviewed. For streamflow data, rating curves will be retrieved.

QWDATA

Analytical water-quality data obtained by analyzing samples that were collected in the field and then sent to an analytical laboratory are stored in this database. The processes of collecting samples, transporting them to the laboratory, analyzing them, reporting results, and archiving the resulting data all affect the data quality. Procedures for sample containers, collection, preservation, holding times, shipping, storage and processing are specified in the National Field Manual for the Collection of Water-Quality Data (USGS 1997-2006) (*http://water.usgs.gov/owq/FieldManual/*). These procedures are mandated for all sample collection by USGS personnel and cooperators for data that are used for USGS monitoring and research. There are minor variations in sampling methods for samples collected under the USGS National Water-Quality Assessment Program (NAWQA). NAWQA surface-water sampling protocols are described in the "Field Guide for Collecting and Processing Stream-Water Samples for the National Water-Quality Assessment Program" (USGS, 1994) (*http://www.swrcb.ca.gov/swamp/docs/qamp/appxd_usgs_nawqawatersampleprotocol.pdf*). Field work that includes sample collection is documented on USGS form (formerly designated BQA-1) "U.S. Geological Survey Surface-Water Quality Notes" and "U.S. Geological Survey Ground-Water Quality Notes".

Samples are first documented in QWDATA via a log-in procedure by the personnel submitting the sample for analysis. At this point the sample receives a "control number" which permanently and uniquely identifies the sample and documents the analyses performed.

For samples sent to USGS laboratories, a second log-in procedure is performed by laboratory personnel, and information and samples compared for accuracy and completeness. Standard, published USGS laboratory procedures are then followed as the samples are routed through the analytical laboratory for the various analyses requested. Results are submitted electronically from the analytical laboratory to the requesting party. Data are not retrievable from QWDATA if any quality assurance issues (such as non-matching dates, sampling site

identification numbers or sample control numbers) are not resolved. After all QA issues have been resolved, the data are available in QWDATA as "preliminary data, subject to revision".

QWDATA is organized such that data can be retrieved by USGS site (sampling location). In addition to nutrient data, other chemical and physical characteristics, such as major and minor anions and cat ions, pesticides, volatile organic compounds, temperature, dissolved oxygen, alkalinity, and many others. Rather than make a decision a-priori to limit the set of available parameters, all data will be retrieved from QWDATA for all USGS sites of interest. Data that do not conform to the project's data quality objectives or other project objectives will later be deleted from the Project database. Acceptance criteria will include:

- Year of sample: water-quality data collected prior to 1980 will not be used.
- Detection/reporting limit (each analytical parameter will be assigned a maximum acceptable reporting limit). Reporting limits vary for most analytical parameters in QWDATA as the result of improving instrumentation and methods over time, and varying data-quality objectives during data collection. The following maximum allowable reporting limits will be adhered to, and data for which reporting limits are greater than these values will not be used in this project
    - Total nitrogen (mg/L as N): 0.01
    - Ammonia (mg/L as N): 0.01
    - Nitrate plus nitrite (mg/L as N): 0.01
    - Nitrite (mg/L as N): 0.01
    - Total Phosphorus (mg/L as P): 0.005
    - Orthophosphate (mg/L as P): 0.005
    - Specific conductance (µS): 1

- Representativeness: samples must represent the water of interest. For example, when considering water quality of water discharging to streams, only shallow ground-water well samples will be considered; when characterizing the water quality of a stream, samples taken from minor tributaries may not be accepted if they are thought not to represent the water in the main body of the stream.
- Bias and representativeness: Where spatially clustered samples provide redundant information, only a representative subset may be used. Alternatively, the central tendency of values for a cluster of samples may be used.
- Comparability and sensitivity: Samples collected in different years, or with different data-quality objectives for the same analyses often have different reporting limits. This is an unavoidable characteristic of any analytical database developed over a long period of time. In order for all data for a given analytical parameter for all samples to be comparable, a uniform reporting limit should be used. The main issue here is "non-detect" or "left-censored" data. If a very low reporting level is adopted, then samples that were analyzed with a higher reporting limit and for which the analyte was not detected may be assigned a lower concentration than is actually present. Conversely, if a higher reporting limit is selected, then available information for samples that were analyzed under a lower reporting limit is lost. For example, many samples in QWDATA were analyzed for nitrate with a reporting level of 0.1 mg/L as N, but many more-recent samples had a reporting level of 0.01 mg/L as N. If, for comparability, the reporting level for all nitrate data was assigned as 0.1, then all values less than 0.1 (e.g. 0.03) would be "rounded" to "less than 0.1", thus losing precision information on the recent, lower-concentration samples. This issue will be addressed separately for each analyte of

interest, and the most appropriate reporting level will be selected in each case using methods described by Helsel and Hirsch (2002).
- Ranges of anticipated concentrations will vary among the analytes.

ADAPS

The User's Manual for **ADAPS** is available on-line (*http://pubs.usgs.gov/of/2003/ofr03123/adapscover.pdf*) and describes the collection of primarily surface-water days, including stream-flow and real-time-water-quality data. Field methods, record keeping, data processing, and archiving are described. The ADAPS database includes more than 100 years of stream-gaging and flow measurement data. More recently, real-time water-quality monitoring data have been archived in ADAPS. Parameters such as water pH, temperature, specific conductance, turbidity, oxidation-reduction potential and fluorescence are monitored over times ranging from instantaneous single-point readings to many years of constant monitoring. In addition to the ADAPS documentation, standard procedures for data collection, instrument maintenance and calibration, and data management are detailed in the USGS report "Guidelines and standard procedures for continuous water-quality monitors—Station operation, record computation, and data reporting" (Wagner and others, 2006) (*http://pubs.usgs.gov/tm/2006/tm1D3/pdf/TM1D3.pdf*).

Most entries in the ADAPS database are related to streamflow, measured at 1466 sites in New Jersey. Of those, 158 are "real time" sites, where time-series (recorded at fixed intervals) data are collected. Measurements are commonly recorded at 5-60 minute intervals and transmitted to the NWIS database every 1-4 hours. Many of the other sites were established and monitored for a period of time to meet objectives of specific projects of varying duration. Typically, for a given site ADAPS archives stage (stream height) and streamflow data (volume per unit time, e.g. cubic feet per second). A rating curve may also be stored. This is a mathematical relation between stage and streamflow, a log/log relation which plots as a smooth curve, sometimes approaching linearity. Methodology for developing rating curves is described by Kennedy (1984) and is available at *http://pubs.usgs.gov/twri/twri3-a10/*. As a rule, all New Jersey streamflow data in ADAPS have been reviewed, approved by the Hydrologic Data Assessment Program Chief, and published (most often in the New Jersey USGS Annual Report). As such, these data have received extensive review and are considered "finalized". Acceptance criteria will apply more to the relevance and age of the data than to their correctness. Criteria will include:

- Date of measurement: Streamflow data collected in timeframes that correspond to the collection of water-quality data is most valuable for contaminant-loading determination. Streamflow data collected during other times may be accepted and used to develop or improve rating curves
- Site of measurement: Streamflow data collected locations where water-quality data were also collected are most valuable for contaminant-loading determination. Streamflow data collected at other locations may be accepted and used to estimate streamflow at nearby streams where no streamflow data are available.

GWSI

GWSI contains data related to 19,789 ground-water measurement and sampling sites (generally wells). Information includes a unique site ID number, location (county, township, coordinates), altitude, well construction (depth, hole depth, screen information), site use, water use, construction date, aquifer, aquifer type, and well-permit number. These data have been

collected over several decades, and the quality of data varies. **Acceptance criteria** for well selection are based on completeness and accuracy of data:

- Well record: Only wells for which a well record is on file at the New Jersey Department of Environmental Protection will be selected for sampling.
- Well depth and construction: These values in GWSI will be confirmed by physically comparing them to those recorded on the well records.
- Well location: the latitude and longitude of the site will be confirmed by GPS.
- Aquifer and aquifer type: these designations will be confirmed by determining what aquifer and aquifer type are consistent with the location and depth of each well to be sampled, based on the most recent and complete ground-water-flow models available.

New Jersey Department of Environmental Protection (NJDEP) Data

Water-quality data were collected by the NJDEP during a cooperative investigation of the effects of land-use patterns on surface-water quality for tributaries to the Toms River (Baker and Hunchak-Kariouuk, 2006). All sample collection, processing and analyses were conducted according to methods described by Connell and Messler, 2004 (*http://www.state.nj.us/dep/bmw/Reports/EstMonitoring8990withData.pdf*). Analytical methods that were used are shown in Table A7-1. Subsequent to receipt by the USGS, these data were reviewed for completeness and accuracy and then published in a USGS report (Baker and Hunchak-Kariouk, 2006).

Model Simulations of Nutrient Loading

An overview of BASINS3 is shown below. The base cartographic data provided in BASINS are not detailed enough for this investigation, and additional USGS data will be used to describe the subbasins in the study area.

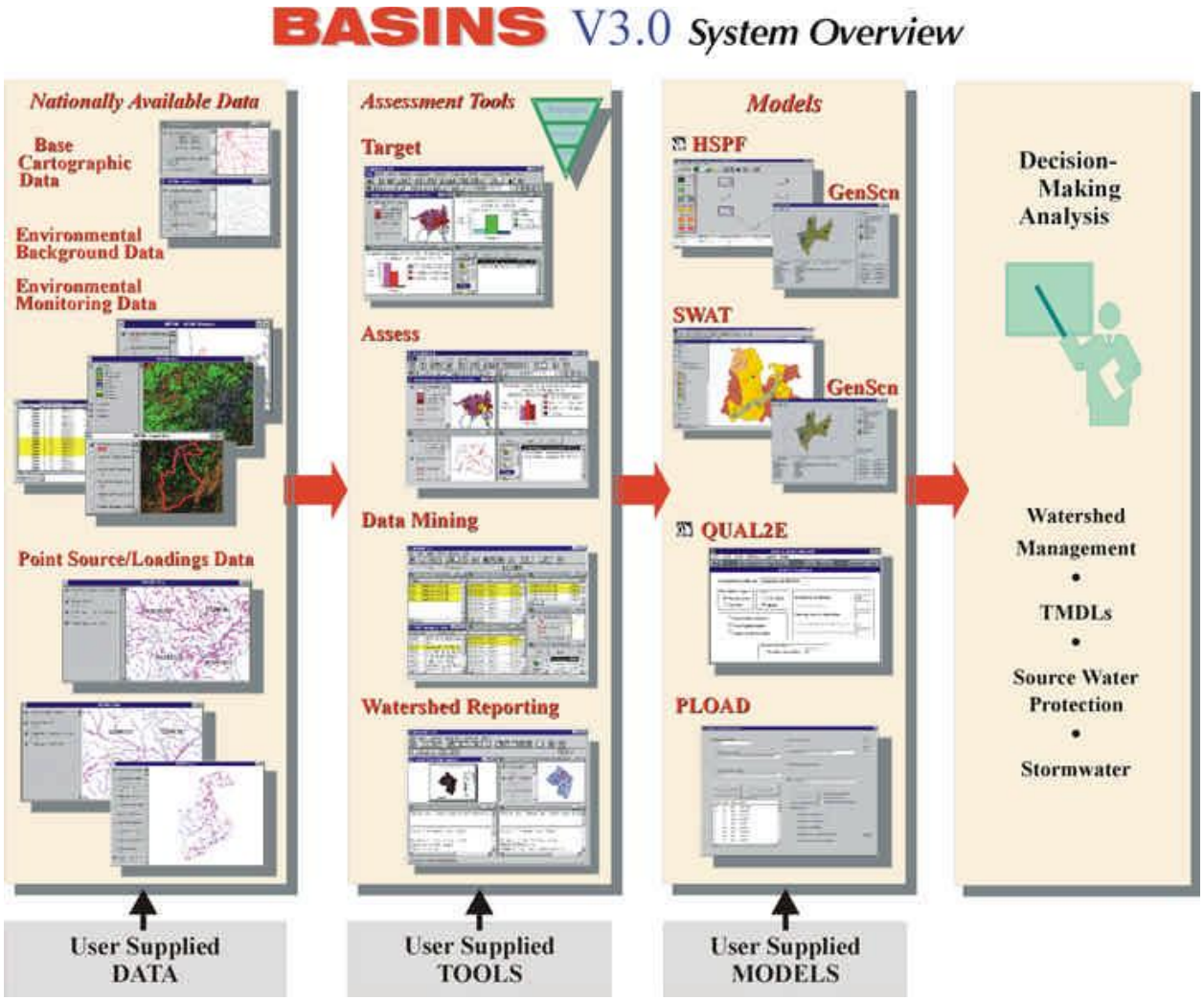


Figure A7-1. Overview of BASINS3 Watershed Modeling Tool.

Table A7-1. Parameter List and Analytical Methods that were used by the NJDEP to collect water-quality data that will be used as a source of secondary data in this investigation. (Method detection limits listed here will also be used as target method detection limits for analyses to be performed by state-certified laboratories during this investigation).

Parameter	Method	Method Detection Limit	Reporting Limit	Analytical Method	
				Method	Reference
Salinity	Conductivity (lab)	0 PPT	0 PPT	2520B	Std Meth 20th ed.
Dissolved Oxygen	Winkler Azide Mod.	0.1 mg/L	0.1 mg/L	360.2	USEPA
Total Suspended Solids	Non-filterable Residue by Drying Oven	1 mg/L	1 mg/L	160.2	USEPA
Ammonia	Automated Phenate	9.49 µg/L	18.97 µg/L	4500-NH3	Std Meth 20th ed.
Nitrate/Nitrite	Automated Cd Red. Reduction	11.25 µg/L	22.50 µg/L	353.3	USEPA
Total Nitrogen	TKN by Semi-Automated Block Digestion	18.22 µg/L	36.44 µg/L	351.2	USEPA
Orthophosphate	Orthophosphate in Estuarine & Coastal Waters	3.55 µg/L	7.09 µg/L	365.5	USEPA
Total Phosphorus		9.88 µg/L	19.76 µg/L	4500PI	

Data will be used for trend analysis only. The method of
Collection is consistent with that of the Chesapeake Bay EPA program and USGS. The data will not be used for regulatory analysis and use.

COMPONENTS 2-4: ESTUARINE BIOTIC RESPONSE; BIOTIC INDEX DEVELOPMENT; AND CURRENT (2010-2011) EUTROPHICATION ASSESSMENT

This project will determine estuarine biotic responses to the loading of nutrients across a gradient of upland watershed development and associated estuarine nitrogen loading, and identify key biotic responses across a variety of estuarine organisms by examining shifts in phytoplankton, benthic macroalgae, seagrass, epiphytes, benthic invertebrates, and shellfish structure. These major groups will be monitored across the study period to determine when numeric shifts occur in abundance, biomass, and areal cover, and other parameters which will then be correlated with nutrient loading levels (determined in subwatershed areas) to document the threshold points and levels of biotic decline. They will also be examined and assessed for statistical validity and inclusion in the index development for the 1989 to 2011 period.

The emphasis of this project is on applications using secondary data, as well as field measurements of biotic parameters, to assess eutrophic and ecological condition. A major goal is the application of environmental databases collected in the Barnegat Bay-Little Egg Harbor estuary and watershed on the development of nutrient loading and eutrophication conceptual modeling. An appropriate mechanism is to establish quality goals for the individual measurements, or measurement quality objectives (MQOs). MQOs for the various measurements (both field and laboratory) can be expressed in terms of accuracy, precision, and completeness goals (Table A7-2). These MQOs were established by obtaining estimates of the most likely data quality that is achievable based on the instrument manufacturer's specifications, scientific experience, or historical data.

The MQOs presented in Table A7-2 are used as quality control criteria both for field and laboratory measurement processes to set the bounds of acceptable measurement error. Generally speaking, MQOs are usually established for five aspects of data quality: representativeness, completeness, comparability, accuracy, and precision (Stanley and Vener, 1985). These terms are described in the context of their application to establish MQOs for each quality assurance parameter.

The relative sensitivity of an analytical method, based on the combined factors of instrument signal, sample size, and sample processing steps, must be documented in order to make a definitive statement regarding detection of an analyte at low levels - for a specific analytical method, what is the lowest concentration at which an analyte's presence can be assured above background noise? For this project, the question will be answered by calculating Method Detection Limits (MDLs) for each type of analysis. Table A7-1 lists the target MDLs for most analyses to be conducted with BB-LEH samples. Laboratories will be expected to perform in general accord with these target MDLs.

DESIRED METHOD SENSITIVITY

Method sensitivity refers to the capability of an instrument or method to distinguish a parameter of interest from background indication or “noise”. Specific methods (to be discussed later) will be selected such that the quantitative objectives of the investigation, with respect to each parameter measured, can be met. Sensitivity (in addition to precision, accuracy and bias) will be documented for each analytical method used.

REPRESENTATIVENESS

The concept of representativeness within the context of the BB-LEH project refers to the ability to accurately and precisely characterize nutrient loading and eutrophic condition in the BB-LEH estuary through the measurement of selected environmental and biotic indicators. An unbiased sampling design that includes a sufficient number of sampling sites is required to make statistically sound determinations on a system-wide basis; both spatial and temporal aspects of sampling must be considered for data collected over the 1989 to 2010 period. For this project, statistically robust tests will be applied that ensures > 90% confidence that the sampling designs are representative of estuarine systems. Temporal variation will be evaluated by repeat monitoring in 2010 and 2011, or through continued monitoring for a limited number of sites in following years if funds are available to do so.

The data quality attribute of representativeness applies not only to the overall sampling design, but also to individual measurements and samples obtained in the course of the monitoring effort. The following examples are illustrations of sample-related factors that might affect the representativeness of the study: the integrity of the sample through periods of storage must be maintained if the sample is to be regarded as representative of the conditions at the time of sampling; the use of QA/QC samples which are similar in composition to the samples being measured to provide estimates of precision and bias that are representative of the sample measurement; and that the samples are collected in an appropriate manner by gear that is specific and standardized for the study.

COMPLETENESS

Completeness is defined as "a measure of the amount of data collected from a measurement process compared to the amount that was expected to be obtained under the conditions of measurement" (Stanley and Vener, 1985). This project has established a completeness goal of 100% for the various indicators being measured (Table A7-1). The major consequence of having less than 100% complete data from all expected stations is a relatively minor loss of statistical power in the areal estimate of condition, as depicted using Cumulative Distribution Functions (CDFs). The 100% completeness goal is established in an attempt to derive the maximum statistical power from the present sampling design. Based on past years' experience, failure to achieve this goal usually results from the field crew's inability to sample at some stations because of logistical barriers, such as insufficient depth, impenetrable substrate, or adverse weather conditions. In the limited number of instances where these may be encountered, extensive efforts will be made to relocate the station or re-sample the station at a later date, always in consultation with program managers. In this way, field personnel must always strive to achieve the 100% completeness goal. In addition, established protocols for tracking samples during shipment and laboratory processing must be followed to minimize data loss following successful sample collection.

COMPARABILITY

Comparability is defined as "the confidence with which one data set can be compared to another" (Stanley and Vener, 1985). For the BB-LEH project to be effective, the data generated must be comparable to that generated from other estuarine condition monitoring projects. If the BB-LEH project is to realize its goals, the comparability of field and laboratory procedures, reporting units and calculations, detection limits, and database management processes must all be maintained to integrate with these other activities. To help ensure and document data

comparability, the BB-LEH project will utilize various data quality indicators (e.g., performance demonstrations, reference materials, and other QC samples) in conjunction with uniform, standard methods. Details of the above applications will be discussed in following sections of this plan.

Inter-laboratory calibration exercises will be conducted for certain indicators (e.g., water temperature) to help evaluate the degree of variability that exists between independent processing laboratories. For example, we will compare a Rutgers thermometer from RUMFS to the NJDEP's NIST-certified thermometer at the state-certified laboratory at Leeds Point. The Rutgers thermometer will then serve as a quality-check for our YSI datalogger thermisters at each calibration and post-calibration check.

ACCURACY, PRECISION AND BIAS

The term "accuracy" which is used synonymously with the term "bias" in this plan, is defined as the difference between a measured value and the true or expected value, and represents an estimate of systematic error or net bias (Kirchner 1983; Hunt and Wilson 1986). "Precision" is defined as the degree of mutual agreement among individual measurements, and represents an estimate of random error (Kirchner 1983; Hunt and Wilson 1986). Collectively, accuracy and precision can provide an estimate of the total error or uncertainty associated with an individual measured value. Measurement quality objectives (MQOs) for the various indicators are expressed separately as maximum allowable accuracy and precision goals (Table A7-2).

Accuracy and precision goals may not be definable for all parameters because of the nature of the measurement type. Accuracy and precision goals for biotic indicators are generally less well constrained than those for water quality indicators for this estuarine system. In order to evaluate the MQOs for precision, various QA/QC samples will be collected and analyzed for most data collection activities. We will collect samples at 120 sites during each sampling period (i.e., June, August, and October). To determine precision, most notably for the biotic indicator samples, we will collect duplicate measurements at 10% of the 120 sites or 12 randomly chosen sampling sites each sampling period for a total of 36 duplicate samples each year of sampling. Table A7-3 presents the types of samples to be used for QA/QC for each of the various data acquisition activities. The frequency of QA/QC measurements and the types of QA data resulting from these samples or processes are also presented in Table A7-3. For biomass, two cores will be taken at the 12 replicate sites. In the laboratory, two different workers will make weight measurements on the same core sample, and the difference will be calculated. For density, two different workers will count shoot density in each core sample, and the difference will be recorded. For blade length, two different workers will measure blade lengths of hand-grab samples, and the difference recorded. For areal cover, a diver will estimate the percent cover at the site, and a second worker in the laboratory will estimate the percent cover from digital imagery taken at the same site. The difference will be recorded. The same procedure will be used on macroalgae shellfish (bay scallops). A diver will count the number of bay scallops at each site, and a second worker in the laboratory will make counts from digital imagery taken at the same site. The procedure will also be applied to abundance of macroalgae. A diver will record the occurrence and abundance of macroalgae at each site, and a second worker in the laboratory will do the same using digital imagery taken in the field.

State certification will be obtained for the Rutgers facility that will collect basic water-quality data (temperature, pH, specific conductance and dissolved oxygen concentration) using automated data sondes. SOPs have been prepared describing maintenance, calibration,

measurement, and data management procedures as part of the certification process (See below). Appendix 2 lists SOPs for data sonde measurements and biotic measurements in this project.

TABLE A7-2. Measurement quality objectives for BB-LEH monitoring indicators. Accuracy (bias) goals are expressed either as absolute difference (+ value) or percent deviation from the "true" value; precision goals are expressed as relative percent difference (RPD) or relative standard deviation (RSD) between two or more replicate measurements. Completeness goal is the percentage of expected results that are obtained successfully.

Indicator/Data Type	Maximum Allowable Accuracy (Bias) Goal	Maximum Allowable Precision Goal	Completeness Goal
Seagrass:			
Biomass	10%	30%	100%
Density	10%	30%	100%
Areal cover	10%	30%	100%
Blade length	10%	30%	100%
Macroalgae	10%	30%	100%
***Shellfish*:**			
Abundance (Counts)	**10%**	**30%**	**100%**
Bloom occurrence	10%	30%	100%
Water Column Characteristics			
Dissolved oxygen	+0.5 mg/L	10%	100%
Salinity	+1.0 ‰	10%	100%
Depth	+0.5 M	10%	100%
pH	+0.3 units	10%	100%
Temperature	+1.0 °C	10%	100%
Secchi depth	NA	10%	100%
Chlorophyll *a*	10%	30%	100%
Total nitrogen	10%	30%	100%

TABLE A7-3. Quality assurance sample types, frequency of use, and types of data generated for BB-LEH monitoring.

Variable	QA Sample Type or Measurement Procedure	Frequency of Use	Data Generated for Measurement Quality Definition
Seagrass composition:			
Biomass	Core sample	Bi-monthly	Difference between two weight measurements on the same core taken by two different workers
Density	Core sample	Bi-monthly	Difference between shoot density measurements taken by two different workers
Areal cover	Quadrat areal	Bi-monthly	Diver-estimated areal cover. Second estimate by another worker using digital imagery for comparison
Blade length	Core sample	Bi-monthly	Duplicate blade length measurements taken by two different workers
Datasonde Water Quality Parameters:			
Dissolved oxygen (DO)	Air-saturated water measurement	Daily	Difference between probe value and saturation level

Salinity	QC check against standard	Daily	Difference between probe measurement and standard value
pH	QC check with standard buffers	Daily	Difference between probe and standards
Temperature	QC check against standard thermometer	Daily	Difference between probe and thermometer
Depth	QC check against depth markings on cable	Per use	Difference between probe measurement and standard marks

Variable	QA Sample Type or Measurement Procedure	Frequency of Use	Data Generated for Measurement Quality Definition
DO, salinity, pH, temperature, and depth	Performance verification at certified calibration center	Annually	Differences between instrument response and calibration standards
DO, salinity, and pH,	Calibration checks at laboratory	Monthly	Difference between instrument response and calibration standards
DO	Comparison to discrete water sample (Winklers); or side-by-side with 2nd instrument	Daily	Difference between instrument DO and reference measurement
Salinity	**Calibration of probe with YSI standards**	**Daily**	**Difference between instrument salinity and calibration standard**

A8. SPECIAL TRAINING/CERTIFICATIONS

No special training or certification is required for Component 1 (retrieving or utilizing the existing data from NWIS or to use the models BASINS3 and PLOAD). Components 2-3 require special training that pertains to the collection of chemical and biological data.

New Jersey State certification will be obtained for the Rutgers facility that will collect basic water-quality data using data sondes, as stated above. All field crews that participate in this project must first successfully demonstrate team proficiency in each component of field sampling and data collection before they will be authorized to collect actual field data and samples. Rutgers personnel will conduct structured field training sessions for those field teams that are new to estuarine bioassessment projects, as well as, for any crew that requests a refresher course. During the training, crews will be instructed on sampling protocols and methods developed for the project, then they will actively participate in hands-on exercises conducted in the field during which all components of the field sampling will be covered. After the crew has developed proficiency in the core field activities, they will be observed and evaluated by the instructors on a pass/fail basis for each component as they conduct a full the BB-LEH field sampling scenario. To be authorized to conduct BB-LEH field monitoring, the crew must pass in all areas of the certification exercise. The field reviewer will document the crew's performance on Field Crew Evaluation forms that will be turned over to the Project QA Manager and become part of the permanent record. The crews will be informed verbally by the reviewer as to whether they passed or failed the certification exercise.

TRAINING PROCEDURES

Seagrass design modified from Short et al. (2002) and Kennish et al. (2008).

In situ

An individual must be trained to identify the *Zostera marina* (eelgrass) in situ from *Ruppia maritima* and macroalgae species. In addition, the worker must be able to operate a 10-cm core device to remove the seagrass sample. Site location will be accomplished with a GPS unit so any worker must be proficient with navigating and recording GPS data.

Laboratory

A worker must be able to process and dry the aboveground and belowground portions of the seagrass sampling including drying and weighing the sample. In addition, the worker must be able to measure seagrass blade length, assess epiphyte infestation, and differentiate macroalgae from seagrass tissue.

A9. DOCUMENTATION AND RECORDS

Whenever changes or updates are made to the QA Project plan, copies of the most current copy will be electronically transmitted to all persons on the distribution list specified in Section A3. This will be the joint responsibility of the two Project Managers (Mike Kennish of

Rutgers University and Ron Baker of the USGS). Procedures that will be used for document and record keeping at the USGS and at Rutgers are described below.

COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

Only secondary (pre-existing) data will be used in Component 1. No sampling, monitoring or analysis will be conducted. All data retrieved from USGS/NWIS will be entered into a Microsoft Access database. The database also will be used to archive all input parameters, conditions, and results of model simulations for BASINS3 and P-LOAD. All metadata, documents and computer files generated during the processes of model development, calibration and validation will also be permanently archived. These documents and databases will be permanently maintained on the USGS computer system, in keeping with U.S. Government-mandated requirements for IT security, back-up protocols, and limited access. Copies of records and data obtained as printed materials (such as copies of well records) will be stored permanently at the USGS office in West Trenton, NJ.

COMPONENTS 2-4 (ESTUARINE BIOTIC RESPONSE, BIOTIC INDEX DEVELOPMENT, AND CURRENT (2010-2011) EUTROPHICATION ASSESSMENT)

The BB-LEH project will require that each data generating activity, both field measurements and laboratory analyses, be thoroughly documented in accord with the guidelines that are presented in this section. Data will be recorded in a variety of paper and digital formats. In situ physical (temperature, salinity, pH, DO, Secchi disc, etc.) and biological (percent cover and biomass of seagrass and macroalgae) measurements will be recorded on write-in-the-rain paper and backed-up with a digital picture (in case the original field sheets are lost). The biomass of the collected samples (determined in the laboratory) will be recorded on data sheets and subsequently loaded into a digital database (MS Excel and Access). The image processing software Erdas Imagine and Adobe Photoshop are also utilized to sharpen the collected imagery. These high resolution images will be analyzed for the basal area and density of seagrass cover, abundance of macroalgae, and presence of epifauna to characterize biotic conditions over well-defined temporal and spatial scales. These data will also be compared with diver observations of the sampling sites.

Field crews will initially record in-the-field data on hardcopy field sheets and, at a later date, all field data will be transcribed into an electronic format. Specific formats for both written and electronically recorded data will be prescribed to document the field monitoring and pertinent steps of laboratory analyses. Ultimately, all data will be converted into an electronic format and the data sets archived in the information management system at the Institute of Marine and Coastal Sciences at Rutgers University.

The study file includes: planning documents (QAPP), SOPS, field data sheets, laboratory notebooks or work sheets, study-related correspondence, records of peer reviews or QA assessments (reviews), and reports and publications. These records will be permanently archived by Rutgers.

Metadata (i.e., documentation of pertinent facts that define a process) will be required for each activity that generates data for this project. Metadata files will be appended to each Rutgers and USGS data set and include information such as who collected the data; how the

data were collected (e.g., equipment, instrument, and methodology); definitions of reporting units; QA/QC data; and descriptions of all aspects of data management or data analysis involved with generating the final reported value. In general, metadata should provide a future data user with a sufficient factual history of the entire process, from sample collection to final reported value, so that they can form their own assessment on the value of that data set for their particular purpose. Checklists will be prepared for use in collecting the necessary information to generate metadata files for the core indicators. Data reporting and documentation requirements, presented on a per activity basis, follow.

FIELD ACTIVITIES

Field crews will rely primarily upon hardcopy field data forms to record most field collected data; however, there are project components where self-contained dataloggers (e.g., datasondes) will be used to collect information that will be downloaded as electronic files. Standardized hardcopy forms will be used. The core field indicators/data in this project will be recorded in an approved, uniform manner. It is preferred that raw data are recorded by ballpoint pen on a real-time basis, but because of the complications with the use of pens in the field, due to wet or damp conditions, it will be acceptable to record field data with a soft-leaded pencil (although it goes against the tenets of QA). There should be a separate form for each measurement type; examples of field data sheet types to be used in this project include:

- Station Information Hydrographic Profile Instrument Calibration/Verification (hardcopy) Seagrass density, biomass, areal coverage, and blade length
- All field sheets must be identified with station ID code and dated; upon completion of the field entries, the person recording the data will sign each sheet. Field sheets are designed to lead the sampling team through a logical sequence of steps and checks that further ensures sampling protocols are followed. The field lead will verify that all field sheets are accounted for and complete prior to departing the sampling station.

All core data recorded on field data sheets will be transcribed into the field computer system within a reasonable time following collection (target period, within a week). To ensure consistency, one person will be responsible for the data entry. Data entry will be straightforward and user friendly; the fields in the electronic format will closely resemble the hardcopy raw data forms. The hardcopy data forms filled out for a given station will be compiled into a "station data package" and photo-copied to provide in-house working copies for use by NEIWPCC as well as the copies required by US EPA (study files). The original field sheets will be archived, as well as backup disks for all electronic files. These raw data will be kept on file for at least a 7-year period.

A systematic approach of sample tracking will be used to ensure accountability for the handling, storage, and transfer or shipment of the field collected samples. Chain-of-custody documentation (as per GLPs) is not required for this study; however, the system should include the following basic components:

Sample Collection:

- A master inventory of all field samples that are expected to be collected (separate list(s) for each sample type and corresponding station IDs), with check off fields providing documentation of all samples that are collected (when, and by whom)

- Sample transfer information/invoice (where, what, to whom, and when, and by whom samples are transferred or shipped)

Sample Receipt (log-in):

- Documentation (sample log-in form) of the person receiving; when and what they receive; and general condition of shipment (e.g., breakage, thawed, etc)
- Reconciliation that what was reported shipped was in fact received
- Deposition/distribution of samples (e.g., where stored and holding conditions)
- Sample release to analysts

As in the case of the field data sheets, a sample tracking system will be followed verbatim. The field team will retain copies of shipping invoices and the originals will be sent with the samples as they are transferred. The field copies should be compiled into a complete set and submitted to the Project manager to be archived for at least a period of 7 years. The recipient of the samples (processing laboratory) will inventory the physical samples against the invoice and alert the Project manager in the event of any missing samples. If a sample is missing, the laboratory should then go through appropriate channels to contact the field team as soon as possible so that they may attempt to locate the sample at their end or possibly re-sample.

LABORATORY ANALYSES

As with field collected data, the overall flow of data generated from laboratory analyses will follow the route established below:

Data Generator (raw data from laboratory)
↓
Project Manager (initial validation and formatting)
↓
NEIWPCC (additional validation/verification, and formatting)
↓
Rutgers IMCS; USGS
↓
Public Website

The specific reporting requirements for each of the major laboratory activities are described in the following sections.

State-certified analytical and processing laboratories used in this project will retain raw data files (e.g., primary standard certification, working standard preparations, instrument calibration records, results of QC check samples/measurements, instrument printouts, and final data calculations) for each indicator for a period of at least 7 years. Demonstration of laboratory certification will be required. The contractor (Rutgers/USGS) will review all data to verify that quality goals are satisfied. Upon issuing appropriate advance notification (i.e., minimum of 2 weeks), NEIWPCC and US EPA maintains the authority to access the active files and/or request copies of specific information at any time. In addition, the full set of data will be part of the study file of which NEIWPCC and US EPA will receive a copy at the completion of the project.

Sediment Characterization Analyses:

Only secondary (pre-existing) sediment data will be used in this investigation. Sediments in the seagrass study area have been characterized by recent investigations (Kennish et al., 2007a, 2008). Sediment samples were collected at the same 120 sampling sites used in this study. The samples were collected using a 10-cm diameter coring device during the 2004-2006 period, and the samples were analyzed in the laboratory for the percent composition of sand, silt (dry sieving) and clay (wet sieving through a 63-μm sieve). Laboratory records of the sediment determinations are maintained by Rutgers IMCS. Therefore, sediments have been collected and analyzed from all 120 sampling sites (100% of the sites) and will be used as secondary data for this project.

Water Quality Parameters:

Water quality measurements will include in situ analyses of water in BB-LEH during the 2010-2011 study period. Water quality parameters (temperature, salinity, dissolved oxygen, and pH) will be measured at all sampling stations using a handheld YSI 600 XL coupled with a handheld YSI 650 MDS display unit, an automated YSI 6600 unit, or a YSI 600 XLM automated datalogger as noted above. The data will be obtained prior to biotic sampling at each sampling site. Water quality data will be collected at mid-depth in the water column.

Water samples will be collected and analyzed by the marine water quality monitoring program of the NJDEP during the 2010-2011 period for nutrients determination. Sample analysis will be conducted in the NJDEP Leeds Point Laboratory, a State certified laboratory. These nutrient data will be used as secondary data, and subjected to the same review process as other secondary data used in this project. The samples will be analyzed following the methods designated in a State certified laboratory. Nitrate plus nitrite, ammonia, total dissolved nitrogen, and phosphate will be determined. Data reports for each of the nutrients will include analytical results. The participating laboratory will maintain records of sample storage conditions, standard preparations, and instrument calibrations. These records will be made available upon request to NEIWPCC and US EPA program personnel.

Water quality measurements collected by the New Jersey Department of Environmental Protection between 1989 and 2011 will be used in data analyses of physicochemical parameters for the estuary as secondary data. These pre-existing data have been collected and analyzed consistently over the 20-year period by State certified laboratories of the NJDEP. Parameters will include dissolved oxygen, Secchi depth, and chlorophyll *a*, as well as nutrient concentrations for ammonia (NH3), nitrite plus nitrate (NO2 + NO3), total nitrogen (TN), phosphate (PO4), and total phosphorus (TP). The data reports for water quality parameters will be submitted (both in hard copy and computer-readable format) to the NEIWPCC and US EPA managers.

Biotic Indicator Assessments:

Nutrient loading numbers will be developed via modeling applications discussed above. Nutrient loadings will be used as a primary indicator in assessment of estuarine biotic responses targeted in this project. Biotic shifts will be correlated with nutrient loadings developed by the aforementioned watershed models.

Among the key biotic indicators to be examined in this project are seagrasses, macroalgae, phytoplankton, epiphytes, and shellfish (see below). The major outcome will be quantitative measures of the distribution, abundance, biomass, and blade length of seagrasses, and threshold values of nutrient enrichment that lead to declining shifts in seagrass demographics, as well as other biotic responses such as nuisance and toxic algal blooms, epiphytic overgrowth, and diminishing shellfish resources. Reports (i.e., appendices) will list by station the data results by biotic group. The data report will be submitted (both in hardcopy and computer-readable formats) to the NEIWPCC and US EPA managers. Any QC data will be summarized in a hardcopy table or narrative and included with the final data package. Also, a narrative report will be included in a cover letter explaining any difficulties or irregularities encountered during the assessments (e.g., taxonomic problems, sample integrity, extraneous material in the samples).

B1. SAMPLING PROCESS DESIGN (EXPERIMENTAL DESIGN)

The estuary will be divided into three segments (north, central, and south) based on a north to south gradient in salinity, nutrient loading, and watershed development (i.e., high to low from north to south) (Figure B1-1). We currently have five years of comprehensive biotic response data (2004-2006, 2008, and 2009) collected by the lead PI and his colleagues at Rutgers University in these estuarine segments (Kennish et al., 2007a, b, 2008), and will continue to sample the same stations and parameters (excluding nutrients which will be collected by NJDEP marine water quality monitoring) using identical sampling methods for this project to ensure consistency in data acquisition with prior years of sampling. In 2010, biotic samples will be collected at 120 sampling sites (see Table B1-1 for station coordinates) using funds from NJDEP research awards. SAV (seagrasses), macroalgae, epiphytes, and shellfish samples will be collected at regular intervals (bimonthly) from June to October each year (see below). NJDEP water-quality data collected year-round between 1989 and 2011 will be used as secondary data for analysis of physicochemical parameters for the estuary; these include dissolved oxygen, Secchi depth, and chlorophyll *a*, as well as nutrient concentrations for ammonia (NH3), nitrite plus nitrate (NO2 + NO3), total nitrogen (TN), phosphate (PO4), and total phosphorus (TP).

There are three basic phases to the project: (1) field collection of environmental data and samples; (2) laboratory analyses of these samples; and (3) primary and secondary data analysis and assessment.

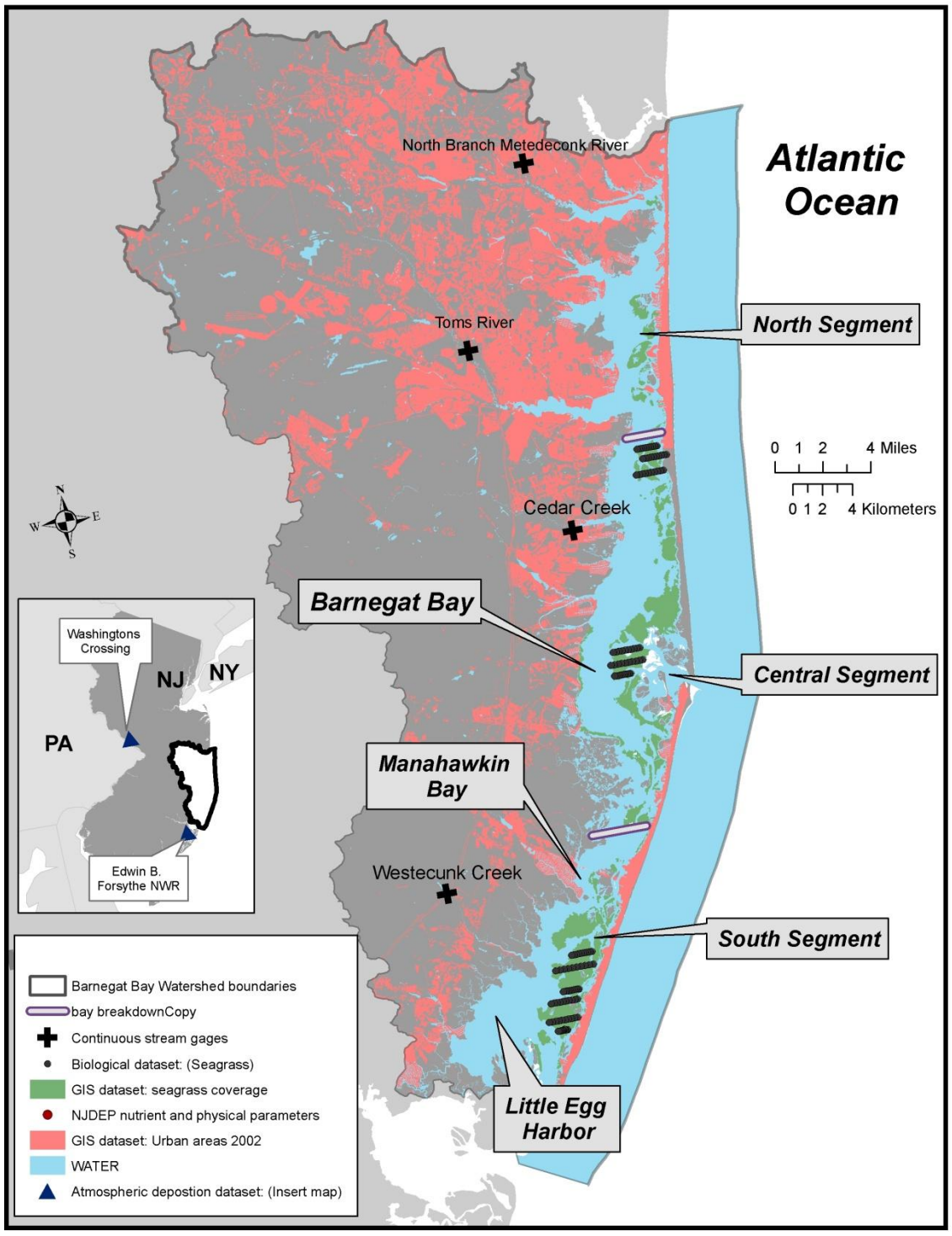


Figure B1-1. Barnegat Bay-Little Egg Harbor Estuary showing north, central, and south sampling segments.

Table B1-1. Coordinates of 120 Biotic Sampling Stations in the Estuary.

UTM X[(1)]	UTM Y[(2)]	Transect	Site
564091	4380361	1	1
564161	4380361	1	2
564231	4380361	1	3
564301	4380361	1	4
564371	4380361	1	5
564440	4380361	1	6
564510	4380361	1	7
564580	4380361	1	8
564650	4380361	1	9
564720	4380361	1	10
563594	4381056	2	1
563809	4381056	2	2
564024	4381056	2	3
564239	4381056	2	4
564454	4381056	2	5
564669	4381056	2	6
564884	4381056	2	7
565099	4381056	2	8
565314	4381056	2	9
565529	4381056	2	10
563937	4382273	3	1
564138	4382273	3	2
564338	4382273	3	3
564539	4382273	3	4
564739	4382273	3	5
564940	4382273	3	6
565140	4382273	3	7
565341	4382273	3	8
565541	4382273	3	9
565742	4382273	3	10
564894	4382925	4	1
565012	4382925	4	2
565131	4382925	4	3
565249	4382925	4	4
565368	4382925	4	5
565486	4382925	4	6
565605	4382925	4	7
565723	4382925	4	8
565842	4382925	4	9
565960	4382925	4	10
567200	4384404	5	1
566915	4384404	5	2
566629	4384404	5	3
566344	4384404	5	4
566059	4384404	5	5

565773	4384404	5	6
565488	4384404	5	7
565203	4384404	5	8
564917	4384404	5	9
564632	4384404	5	10
565879	4385250	6	1
566034	4385250	6	2
566188	4385250	6	3
566343	4385250	6	4
566498	4385250	6	5
566652	4385250	6	6
566807	4385250	6	7
566962	4385250	6	8
567116	4385250	6	9
567271	4385250	6	10
572096	4403206	7	1
572216	4403206	7	2
572336	4403206	7	3
572456	4403206	7	4
572576	4403206	7	5
572695	4403206	7	6
572815	4403206	7	7
572935	4403206	7	8
573055	4403206	7	9
573175	4403206	7	10
571964	4403959	8	1
572207	4403959	8	2
572451	4403959	8	3
572694	4403959	8	4
572937	4403959	8	5
573181	4403959	8	6
573424	4403959	8	7
573667	4403959	8	8
573911	4403959	8	9
574154	4403959	8	10
572267	4404798	9	1
572469	4404798	9	2
572672	4404798	9	3
572874	4404798	9	4
573077	4404798	9	5
573279	4404798	9	6
573482	4404798	9	7
573684	4404798	9	8
573887	4404798	9	9
574089	4404798	9	10
575935	4416264	10	1
576139	4416264	10	2
576343	4416264	10	3

576548	4416264	10	4
576752	4416264	10	5
576956	4416264	10	6
577160	4416264	10	7
577365	4416264	10	8
577569	4416264	10	9
577773	4416264	10	10
576563	4417272	11	1
576743	4417272	11	2
576923	4417272	11	3
577103	4417272	11	4
577283	4417272	11	5
577462	4417272	11	6
577642	4417272	11	7
577822	4417272	11	8
578002	4417272	11	9
578182	4417272	11	10
576296	4417973	12	1
576445	4417973	12	2
576595	4417973	12	3
576744	4417973	12	4
576894	4417973	12	5
577043	4417973	12	6
577193	4417973	12	7
577342	4417973	12	8
577492	4417973	12	9
577641	4417973	12	10

(1) **Universal Transverse Mercator coordinate, X (east) dimension**

(2) **Universal Transverse Mercator coordinate, Y (north) dimension**

FIELD COLLECTION OF ENVIRONMENTAL DATA

The field teams will collect biotic response data for each of their sampling locations. The crew will locate the sampling stations by use of Global Positioning Satellite System (GPS), preferably, differential. Agreement between the given coordinates and the actual in-the-field siting of a sampling station should be within a radius of approximately 5 m

Field activities performed at each station should require approximately 15-30 minutes per station; therefore, a team can expect to sample about 20 stations in a normal day. Of course, this is subject to such factors as weather, seas, and travel distance. At each sampling station, all sampling crews will uniformly collect a core set of data and samples following established sampling protocols and methods as outlined in Kennish et al. (2007b, 2008). Core field data samples include (these will be discussed in greater detail in following sections):

- Water column (temperature, salinity, DO, pH, depth, and Secchi depth)

- Water quality parameter (chlorophyll *a* from NJDEP databases)
- Seagrass (density, biomass, areal coverage, blade length, and epiphytes)
- Macroalgae (abundance, areal coverage)
- Shellfish (abundance of bay scallops)
- Habitat (general habitat-type; presence/absence: exotic species, and anthropogenic debris or perturbation).

Sources of variability and how this variability is addressed are shown Table B1-2 below.

Table B1-2. Sources of Data Variability and Actions Taken to Resolve or Reconcile Variability with Project Objectives

Data Category	Source of Variability	Action(s) Taken
Water column	Instrument drift	Recalibration as specified
	Instrument fouling (NA)	Cleaning and maintenance
	Instrument failure	Repair and replacement as needed
Water quality parameters	Sampling bias	Replicate samples
	Instrument drift	Recalibration as specified
Seagrass measurements	Sampling bias	Multiple composite samples
	Temporal variability	Frequent re-measurement
	Spatial variability	Spatially composited samples
Macroalgae	Sampling bias	Multiple composite samples
	Temporal variability	Frequent re-measurement
	Spatial variability	Spatially composited samples
shellfish sampling bias	Sampling bias	Multiple composite samples
	Temporal variability	Frequent re-measurement
	Spatial variability	Spatially composited samples
Habitat	Anthropogenic effects	Careful inspection of sampling areas

Samples collected from the field will be taken to the Rutgers University Marine Field Station in Tuckerton for storage and analysis. Table B1-3 is a field data sampling form to be used in this project.

TABLE B1-3. Field data sampling form to be used in this project.

Date: _____ Time (EST): _____ Transect: _____ Station: _____

Quadrat Location: _____ _____

Temp (C)	_____	DO%	_____
Sp Cond	_____	DO conc	_____
Salinity	_____	Depth (sonde)	_____
		pH	_____
Depth (stick)	_____	Secchi	_____

% Cover *Zostera* _____ % Cover *Ruppia*_____

% Cover Macroalgae (*see comments*) _____ % Cover other (*see comments*) _____

Biomass Station? Y N Y, but no seagrass

Five blade lengths (mm) (*Zostera* only _____ _____ _____

_____ _____

Boat Scarring	_____	Grazing	_____
Epiphyte	_____	Wasting Disease	_____

Scallops_____

Comments:

SUPERVISOR INITIALS: _____

LABORATORY ANALYSES OF SAMPLES

Contract Laboratories:

Biotic samples will be analyzed in Rutgers University Marine Field Station laboratories of the Institute of Marine and Coastal Sciences. Biotic measurements will include:

- seagrass shoot density, aboveground and belowground biomass, epiphyte abundance, and blade length

In-State (NJDEP) Laboratory Analyses

Secondary water quality data used in this project will be derived from the NJDEP marine water quality monitoring program. All in-state laboratory analyses have been conducted by the laboratories of NJDEP's Bureau of Marine Water Monitoring. Only parameters for which the laboratory has certification from NJDEP's Office of Quality Assurance will be used.

All of the information specified in this section (B1) is critical to the investigation, either as essential data for achieving the project objectives, or as quality-assurance parameters. None of the types of information mentioned are for information purposes only.

B2. SAMPLING METHODS: REQUIREMENTS

SAMPLING OVERVIEW

The diverse array of sampling and analytical requirements necessary in this investigation is discussed in the sections that follow. Sampling and QA procedures vary among the methods. If problems occur, such as lost, contaminated, mislabeled or improperly handled samples, these problems will be documented in the appropriate project record-keeping location (field or laboratory logs). If practical, replacement samples will be obtained. If differences between temporal or spatial characteristics of the replacement samples and the original samples have a bearing on calculations, modeling or other project activities or objectives, such differences will be noted in all subsequent documentation and products in which the replacement samples were used.

The following demographic data were obtained on all sampling dates using the methods of Kennish et al. (2007b, 2008): presence/absence of seagrass and macroalgae, aboveground and belowground biomass of seagrass, density of seagrass, percent cover of seagrass and macroalgae, and seagrass blade length. In addition, seagrass epiphyte biomass will be collected. Physicochemical data (temperature, salinity, pH, dissolved oxygen, and depth) will also be collected at each sampling site using either a handheld YSI 600 XL datasonde coupled with a handheld YSI 650 MDS display unit, an automated YSI 6600 unit, or a YSI 600 XLM automated datalogger. Secchi disk measurements will likewise be collected in the survey area. Water quality data (other than Secchi measurements) will be collected at a uniform depth (~10 cm) above the sediment-water interface using YSI datasondes.

Sampling stations along each transect will be permanently located with a Differential Global Positioning System (Trimble®GeoXT™ handheld unit). Sampling periods will commence in June, August, and October and continue until all stations are sampled.

Seagrass and Macroalgae Quadrat Sampling

Reliable quantitative biocriteria of estuarine conditions in the BB-LEH system will be developed by using seagrass and macroalgae by:

- Relating the distribution, abundance, and biomass of seagrass, and the distribution and abundance of macroalgae, to good/fair/poor estuarine conditions
- Conducting spatial/temporal (i.e. historical) trends analyses of seagrass distribution, abundance, and biomass.

State-of-the-art targeted seagrass sampling will be conducted at stations along 12 transects following the methods of Short et al. (2002). This method of sampling is acknowledged by the scientific community as the most reliable to effectively assess the condition of seagrass beds. Quadrat-and-transect sampling will be conducted bimonthly during the June-November period in 2010 and 2010, targeting disjunct seagrass beds in Little Egg Harbor (~1700 ha) and Barnegat Bay (~1550 ha). Ten equally spaced stations will be sampled along 12, east-west trending transects (transects 1-12) in four disjunct seagrass beds in Little Egg Harbor and Barnegat Bay (Figure B1-2). A total of 360 seagrass samples will be collected at the 120 transect sites during each year of sampling.

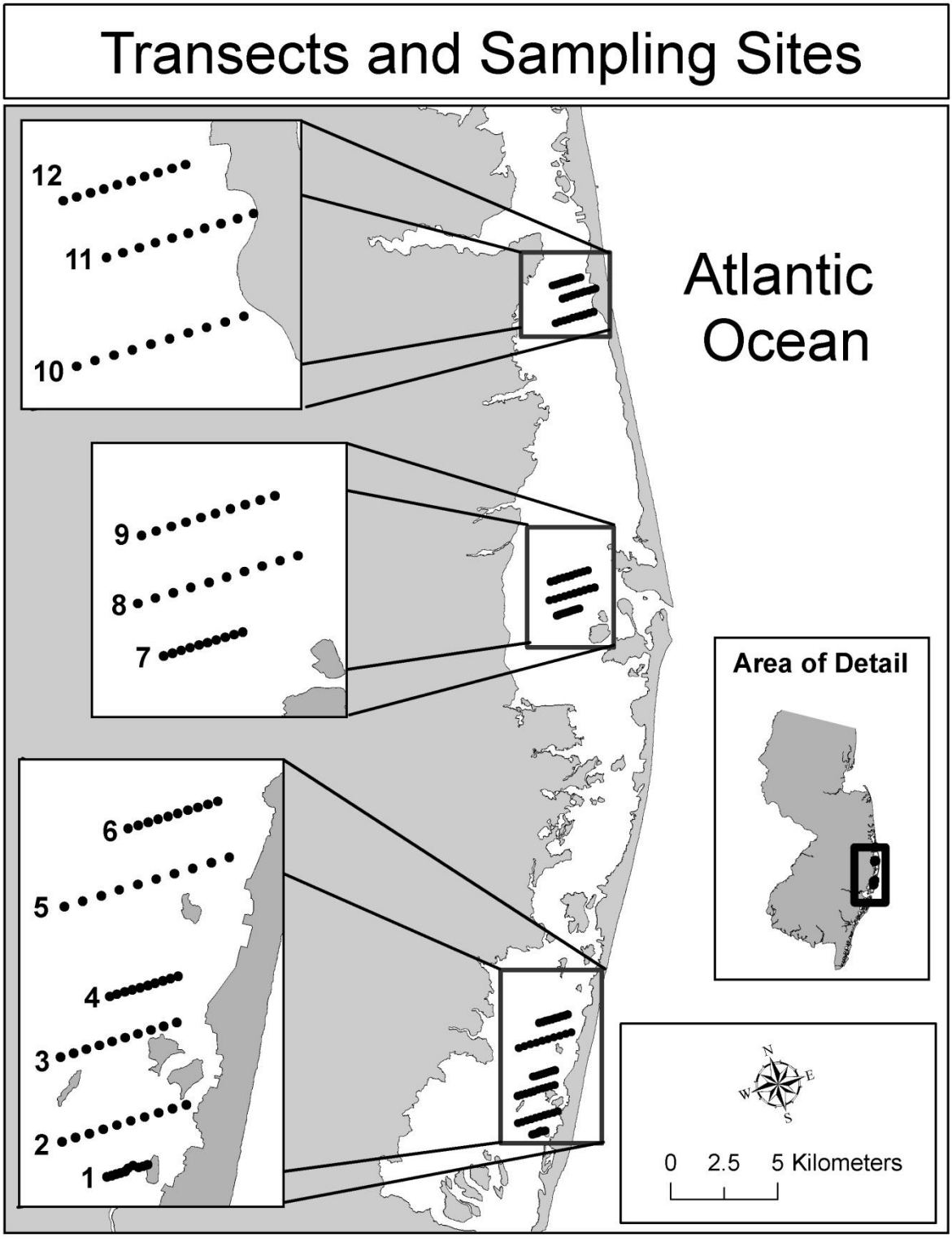


Figure B1-2. Seagrass beds showing 120 sampling stations along 12 estuary transects.

We are following state-of-the-art sampling protocols established by Dr. Fred Short at the University of New Hampshire for the nationally integrated SeagrassNet program (see Short et al., 2002). The procedure is to use an array of transects to cover the seagrass beds in an estuary. A metal quadrat measuring 0.5 m on each side with an area of 0.25 m^2 will be randomly placed at each sampling site (see Table B1-1 and Figure B1-1) to measure seagrass and macroalgae areal coverage. The percent cover of seagrass and macroalgae will then be estimated in situ by a diver using a scale of 0 to 100 in increments of 5. Subsequently, the diver will measure the length of 5 randomly chosen seagrass blades to the nearest millimeter. The diver will then visually inspect the seagrass bed within the quadrat for evidence of grazing, boat scarring, macroalgae, epiphytic loading, and wasting disease.

We are using 12 sampling transects to cover the seagrass beds in the Barnegat Bay-Little Egg Harbor system. There are 10 targeted sampling stations located along each transect. So, the seagrass sampling design provides comprehensive sampling coverage (120 sampling stations) of the estuary, and statistical validity for the study. The transects and sampling stations were originally selected by:

1) Selecting the seagrass bed of interest
2) For each bed three random points were chosen on the eastern edge of each bed.
3) Once each point was chosen the bed was divided into 9 equal segments going from the east to the west.
4) This creates a transect with 10 points going from east to west with a randomly selected north to south position.

The transects for this project were selected based on transects used by the investigators in earlier projects from 2004 to 2006 and 2008 to 2009. The original rationale for selecting transects was to break up the locations where eelgrass occurs and go from shallow to deeper water with the hypothesis that eelgrass in deeper water will respond to change faster than eelgrass in shallow water. The eelgrass beds were identified through remote sensing surveys. The methodology that was used is a common sampling strategy for eelgrass.

By pre-selecting the seagrass beds to be targeted in the study, not all seagrass beds in the estuary had an equal chance to be selected for sampling. However, we conducted comprehensive sampling of the major seagrass beds across the estuary from Tuckerton to Seaside Heights (Route 37 Bridge), yielding a detailed database consistent with the SeagrassNet approach (Short et al., 2002). Seagrass beds were not sampled in the northern segment of the estuary because there are no *Zostera* beds in that segment.

There are three sampling periods each year. Seagrass beds cover about 14% of the total area of the estuary. The 120 quadrat measurements made during each sampling period will provide a valid assessment of the system because the sampling design provides a census approach targeting *Zostera marina* beds only, and essentially covering the entire areal extent of the habitat resource. It is important to compare/contrast the following. Annual NCA sampling of the benthos in the Barnegat Bay-Little Egg Harbor Estuary between 2000 and 2005 used 20 benthic grab samples (collected in September each year) to characterize the benthic community estuary-wide (280 square kilometer estuarine area). We are using 360 quadrat samples collected in one year over a June-October period to characterize seagrass habitat over a much smaller area in the estuary. We believe we are providing a much better statistical database. The sampling station points are so well spread out across the bed and since the points were

originally randomly selected within beds we will use the points to describe the seagrass beds studied. These beds are a large majority of the entire seagrass habitat within the entire estuary system.

Demographic data collected on seagrass in the estuary will represent characteristics of the populations, both eelgrass and widgeon grass, based on the application of accepted protocols and published findings in the literature. Demographic sampling and analysis of seagrass samples will follow the methods of Short et al. (2002) and Kennish et al. (2008) which have been shown through peer-review to represent the characteristics of seagrass populations based on the application of sound statistical testing. It is expected that the data will be directly comparable to published data from other studies of mid-Atlantic coastal lagoons, although there may be greater variation and more acute changes in seagrass biomass, percent cover, and growth, as well as the rate and magnitude of nitrogen uptake during the growing season (Kennish et al., 2007a, b).

Seagrass trend analysis will be completed by a statistical comparison of data collected in 2004, 2005, 2006, 2008, and 2009. This analysis will compare differences in aboveground and belowground biomass, seagrass density and percent cover, macroalgae percent cover, and seagrass blade length using a paired statistical test (e.g. either a paired t-test if data is normally distributed or the non parametric Wilcoxon signed rank test).

Core Sampling

Coring methods will also follow those of Short et al. (2002), with a 10-cm (.00785 m^2) diameter PVC coring device used to collect the cores. Care must be taken not to cut or damage the aboveground seagrass tissues. The diver-deployed corer will be extended deep enough to extract all belowground fractions (roots and rhizomes). Each core will be placed in a 2 x 2 mm mesh bag and rinsed to separate plant material from the sediment. After removing the seagrass sample from the mesh bag, the sample will be placed in a labeled bag and stored on ice in a closed container prior to transport back to the Rutgers University Marine Field Station (RUMFS) in Tuckerton. In the laboratory, the samples will be carefully sorted and separated into aboveground (shoots) and belowground (roots and rhizomes) components. The aboveground and belowground fractions will then be oven dried at 50-60 ºC for a minimum of 48 hours (i.e. after rinsing, an aboveground and belowground portion can be delineated by differences in both color and morphology). The dry weight biomass (g dry wt m^{-2}) of each fraction will be subsequently measured to the third decimal place. Biomass samples will be collected at all 120 randomly selected sites out of the total 120 sites visited each year due to processing time sampling stations.

Macroalgae Bloom Sampling

A diver will collect macroalgae samples at transect sampling sites that exhibit bloom conditions. The samples will be removed from the seagrass bed and placed in 1-liter Nalgene bottles containing formalin adjusted to approximate ambient salinities. They will be subsequently transported to the Rutgers University Marine Field Station (RUMFS) and later examined for taxonomic identification.

Phytoplankton Bloom Sampling

Chlorophyll *a* measurements will be analyzed retrospectively from archived (secondary) water-quality databases of the NJDEP collected in the estuary since 1989 to assess phytoplankton biomass in the estuary over time. We will also analyze chlorophyll *a* data collected in 2010 and 2011 using a novel approach. In 2010 and 2011 sampling, we propose to take advantage of an ongoing survey program within the NJDEP that employs remotely estimated chlorophyll *a* concentrations to highlight potential harmful algal blooms. When high chlorophyll *a* values are detected using remote sensing surveys, water samples will be collected in situ within and outside of the bloom areas and subsequently analyzed in the NJDEP laboratory at Leed's Point for the dominant taxa. We will also attempt to identify the occurrence of harmful algal species (e.g., *Aureococcus anophagefferens*) during the bloom development. We will attempt to assess the impact of *A. anophagefferens* blooms on vital seagrass habitat by developing and applying a response-sampling protocol within the affected seagrass beds. These data will be augmented by assessing historical brown tide bloom events compiled for the BB-LEH Estuary by the NJDEP over the 1995 to 2004 period. These data will be useful for retrospective analysis of brown tide impacts in the estuary.

Determination of the dominant phytoplankton species during bloom events will be conducted at the State certified laboratory of the NJDEP at Leeds Point. The same methods employed by the Leeds Point Laboratory for phytoplankton collection and analysis in the National Shellfish Sanitation Program will be employed here. The protocols will be as follows. Water samples collected during field surveys of phytoplankton blooms will be analyzed with a light microscope. Phytoplankton cells of the dominant species will be enumerated and identified using settling chambers, or a derivation thereof. A 10-ml subsample of the sample is allowed to deposit as sediment in 10 ml counting chambers directly onto glass microscope slides. The preserved samples are analyzed for identification and enumeration to both the genus and species levels when possible, considering the most dominant taxa. Enumeration results are pooled into taxonomic categories such as diatoms, dinoflagellates and flagellate/ciliates for the purposes of examining community structure. Additional subsamples of the collections may be forwarded to other laboratories for an intercomparison analysis if there are difficulties encountered with the identification and enumeration.

If additional funding can be obtained during the project period, phytoplankton samples from each bloom event will be sent for enumeration of brown tide (Aureococcus anophagefferens) to Liping Wei at the New Jersey Institute of Technology. Protocols for processing these samples are given in Appendix 4. Historical data on brown tide blooms obtained during the 1989 to 2011 period will be included in the secondary database for hindcasting analysis.

Shellfish Sampling

Bay scallops (*Argopecten irradians*) will be enumerated in the field by a diver making in-situ observations at all sampling sites. These data will be augmented and validated by bay scallops identified in underwater camera images taken in the field along seagrass transect sites (see below). Bay scallops are confined to seagrass habitat in the estuary and are not found on unvegetated bottoms outside of the seagrass bed boundaries. Therefore, the occurrence and abundance of the species are considered to be potentially powerful indicators of seagrass habitat conditions.

Hard clam (*Mercenaria mercenaria*) resource condition will also be used in the project as secondary data for index development. Hard clam commercial landings in Ocean County will be used to assess resource status over the 1989 to 2011 study period.

Digital Imaging

Underwater camera images of seagrass and other bottom habitats will be obtained using a digital camera unit. This underwater camera offers several advantages over high-resolution, remote sensing techniques (i.e., aerial photography and satellite imaging), which yield broad spatial coverage of an estuarine system, but are not effective for imaging fine detail in spatially restricted habitat areas.

Underwater photos of bottom habitats will be obtained using a high resolution digital camera. Digital pictures will be collected at each sampling station and the photos will be cataloged and analyzed. The photos are catalogued as follows. The camera prints a date and timestamp on each photo. The photos are then copied onto the Rutgers University Marine Field Station (RUMFS) network (Dellserver") and organized into three folders (Periods 1, 2, and 3). They are then stored in individual station subfolders (120 subfolders, one for each station, per period). A laboratory worker can later call up any photo on computer and assess the imagery and specific biotic parameters. The data are then stored in databases that can be backed up on other computer systems.

Procedures for field collection of biotic (seagrass) samples and associated data for this project are based on methods developed by Short et al. (2002) and Kennish et al. (2007b, 2008). The following discussion describes the general methods and procedures for each core sampling activity. Field crews should adhere to these methods as much as possible. Additional QA/QC details for the procedures will be discussed in later sections.

Sampling locations will be provided to the field crews as coordinates of latitude/longitude in degrees-minutes, expressed to the nearest 0.01 minute (i.e., 00° 00.00'). The crews will use GPS to locate the site. The acceptable tolerance goal for siting is that the sampling station be established within (5 m) of the given coordinates. This reflects the accuracy expected from a properly functioning GPS unit of the caliber that will be used for the study. Note: the lat/long coordinates of the actual anchorage, not the "intended or given" coordinates, will be recorded on the field sheet as the sampling location. The GPS's performance should be verified on a daily basis; these details will be discussed in Section B5.

Field crews will strictly adhere to the above guidelines for siting the station, unless there are substantiated reasons that prevent sampling within that defined area. If an intended site location presents an obvious problem, two alternate locations will be considered as the sampling site. Thus, the field crew will have the discretion to use the alternate locations knowing that all three locations meet DQOs. If unusual circumstances preclude sampling at any of the three locations, the situation must be reported to the project leader, who, in turn, will discuss the specifics with appropriate NEIWPCC and US EPA personnel for resolution options. Depending on the nature of the situation, the project leader may elect to relocate the site within an acceptable range of the original location, or the site may be dropped from the sampling array. Decisions on this level (i.e., significant changes to the sampling design) are to be made only by the project leader, not by the field teams.

Field teams, however, will have a limited degree of onsite flexibility to relocate sampling sites when confronted with unexpected obstacles or impediments associated with locating sites

within the 5 m guideline. The crew chief may, for good reason (e.g., shallow conditions, currents, man-made obstructions), move the station to the nearest location from the intended site that is amenable to conduct the sampling; every effort must be made to relocate to an area that appears similar in character to that of the intended site. When it is necessary to relocate the site, the reason for the shift must be documented in the field record. Any site relocation that exceeds 5 m will be flagged and reviewed before any data collected from the station are acceptable for inclusion to the study database. At times, crews might experience difficulty in obtaining a "good core" when collecting sediment due to the nature of the bottom at their established site or some other technical deficiency. In these situations, even after they have collected the water quality samples and data, it is permissible for them to move around within the 5 m radius to locate more favorable sampling conditions without having to resample the water quality indicators.

Remote Sensing Seagrass Habitat

Remote sensing surveys to map the spatial extent of seagrass habitats within the BB-LEH estuary system have been conducted during both 2003 and 2009. The 2003 remote sensing data are complete and available via the internet at *http://www.crssa.rutgers.edu/projects/runj/sav/index.htm*. A full description of the 2003 seagrass remote sensing methods is available from the article, A Multi-scale Segmentation Approach to Mapping Seagrass Habitats Using Airborne Digital Camera Imagery, Photogrammetric Engineering & Remote Sensing, June, 2006.

Methods: 2003 Seagrass Remote Sensing Survey

To the greatest extent possible, this project followed the general guidelines established by NOAA's Coastal Services Center for remotely sensed image acquisition for benthic habitat mapping (NOAA Coastal Services Center, 2004). A digital camera with four bands was employed: blue (410 to 490 nm); green (510 to 590 nm); red (610 to 690 nm); and near-infrared (800 to 900 nm). Two GeoTiff image products were created, a truecolor imagery set and an infrared imagery set, both at a 1-m ground cell resolution and 8-bit radiometric resolution. The images were orthorectified, terrain corrected (using 7.5 min USGS DEM's), georegistered and mosaicked by flight mission with a spatial accuracy of 3 m (90% of pixels). Aerial imagery collection was scheduled for the mid- to late-spring as this time period corresponded with a sufficiently advanced growth state of the *Zostera* beds and generally low turbidity water conditions. The majority of the imagery was acquired during the early to mid-morning hours of 4 and 5 May to correspond with a low tidal stage.

To support the image interpretation and mapping, extensive field reference data were collected in the weeks before and after the image acquisition. A total of 245 field reference points were collected. Once on-site, a 1 m2 quadrat was tossed overboard and observation of the bottom was undertaken by a diver in the water using a mask for underwater viewing. For each field reference point, the following data were collected:

• Time;
GPS location (UTM);
• Date;
• Depth (meters);
• SAV species presence/dominance: *Zostera marina* or *Ruppia maritima* or macroalgae: determined by visual estimation within the 1 m2 quadrat and the 1/9 m2 core (see below);

• Percent cover (10 percent intervals): determined by visual estimation within the 1 m2 quadrat;
• Additional Comments.

Classification

The eCognition software (Standard Version 3.0) was used to segment the image into image objects at several spatial scales. The software employs a bottom-up, region-merging technique to generate homogeneous objects through a local optimization procedure (Benz *et al*., 2004). In other words, a superobject is composed of objects, which in turn can be composed of sub-objects. As sub-objects are aggregated to form an object, interior boundaries disappear, but exterior boundaries remain stable. This multi-resolution approach was adopted to segment the water portion of the image into three general levels of spatial detail. We employed a *manual* classification approach, where the image objects were visually interpreted and manually assigned a bottom type category. The field reference data were used as a general training aid in the initial stages of the visual interpretation process and were consulted during later mapping stages. Seagrass habitat was classified into three density classes sparse, moderate, and dense.

Accuracy Assessment

The resulting maps were compared with the 245 field reference points. The 1 m2 quadrat percent cover data was used to classify each point into the appropriate bottom type category. All 245 field reference points were used to support the interpretation and mapping in some fashion, and so cannot be truly considered as completely independent validation. The resulting maps were also compared with an independent set of 41 bottom sampling points collected as part of a separate seagrass-sediment study conducted by the Natural Resources Conservation Service and Ocean County Soil Conservation District during the summer of 2003 (Smith and Friedman, 2004). These additional 41 bottom sample points were collected in an area along the eastern shore of central Barnegat Bay in an area deemed of high image quality. At each sampling point, a sediment grab sample was taken and the presence/absence of seagrass visually determined for an approximately 5 m2 area. The spatial locations of the 41 sampling points were recorded using a non-differentially collected GPS receiver (Garmin Map 12) with an approximate positional error of 15 m (as compared to the 1 to 3 m for the differentially corrected 245 points). The presence/ absence data for the 245 and 41 sampling points were compared with the same location from the digital seagrass map and summarized in a contingency table and producer's/ user's accuracy and Kappa statistic (a measure of agreement corrected for chance agreement) computed.)

TABLE 2. CONTINGENCY TABLE COMPARING LEVEL 5 SEAGRASS DENSITY FROM FIELD REFERENCE DATA AND THE GIS SEAGRASS MAPS FOR 245 POINTS

TABLE 2A. FOUR CATEGORIES: SEAGRASS ABSENT, SPARSE, MODERATE VERSUS DENSE

	Reference				
GIS Map	Seagrass Absent	Seagrass Sparse	Seagrass Moderate	Seagrass Dense	User's Accuracy
Seagrass: Absent	67	20	9	3	68%
Seagrass: sparse	4	37	14	3	64%
Seagrass: moderate	0	4	40	6	80%
Seagrass: dense	6	2	7	23	61%
Producer's Accuracy	87%	59%	57%	66%	68%

TABLE 2B. TWO CATEGORIES: SEAGRASS PRESENT VERSUS ABSENT

	Reference		
GIS Map	Seagrass Absent	Seagrass Present	User's Accuracy
Seagrass Absent	67	32	68%
Seagrass Present	10	136	93%
Producer's Accuracy	87%	81%	83%

Results

The three seagrass classes accounted for 5,184 ha or approximately 14.5% of the 35,864 ha BB-LEH study area. The sparse and dense cover classes occurred in comparatively equal proportion (38% and 40%, respectively) while the moderate cover class was slightly less at 22% of the total seagrass area. The seagrass density data for the 245 field reference points were categorized into four seagrass density classes (absent, sparse, moderate, and dense), compared with the same location from the digital seagrass map and summarized in a contingency table (Table 2a above). The overall accuracy was 68.2% and Kappa statistic was 56.5%, which can be considered as a moderate degree of agreement between the two data sets. Aggregating the data into a simple presence versus absence comparison (Table 2b above) shows a higher level of agreement with an overall accuracy of 82.8% and a Kappa statistic of 63.1%. Examination of Table 2b reveals that most of the disagreement was due to a high error of omission, i.e., a number of points confirmed as seagrass in the field sampling data were not mapped as seagrass (32 out of 245 points or 13.1%). Twenty out of these 32 points (62.5%) were categorized as sparse seagrass (i.e., 10 to 39%) in the field. The final 2003 GIS remote sensing data is available for public download along with compliant Federal Geographic Data Committee (FGDC) from the CRSSA website.

2009 Seagrass remote sensing survey

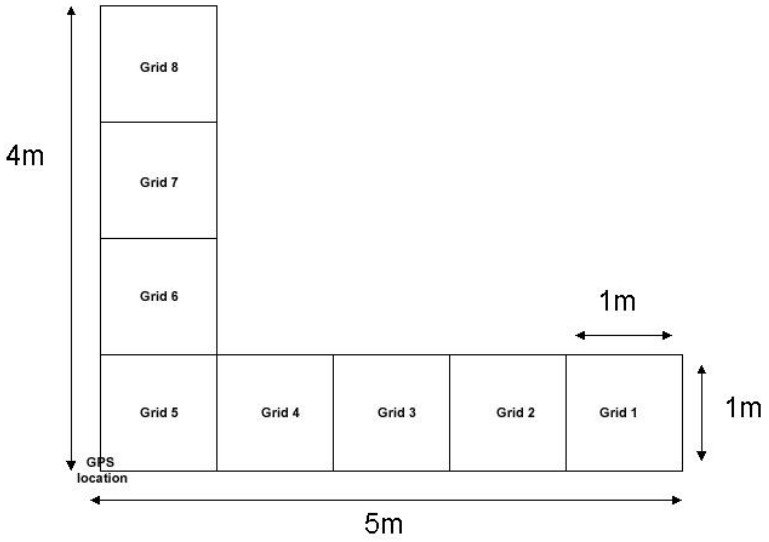
The 2009 seagrass remote sensing survey was designed to collect data in a similar manner to the 2003 remote sensing mission. Some methods have been modified including the type of imagery collected (analog vs. digital) and *in situ* field methods to mitigate the spatial variability of quadrat sampling. For a fuller description of methods and output results for the 2009 seagrass remote sensing survey, please refer to the approved QAPP for that project (**Remote Sensing survey of submersed aquatic vegetation in the Barnegat Bay Little Egg Harbor Estuary system).**

Methods

Film aerial photography was collected on June 28th, July 14th, and August 12th using a Navajo HS airplane equipped with a Leicca RC30 camera, lens # 13234, focal length 152.720 mm, variable exposure time of 260-420 (units). Two types of film were used a grey scale AGFA 80 and color film AGFA 100. The same plane and camera were used for all three imaging missions. The plane flew at an average altitude of 3,658 m with an average speed of 180 knots/hr. The plane flew three survey lines, two in the southern estuary due to bay width and one in the northern estuary for both the June 28th and July 14th fly dates. Two passes were made per day, the first to collect black and white photography, and the second to collect color photography. The resultant film was then exposed and scanned through a high resolution scanner resulting in 18,278 by 18,292 pixels with a scale of 1 to 2,000. These scans were then orthorectified and corrected and projected into the Universal Transverse Mercator, UTM North American Datum 1983 zone 18 North in meters. The resulting geo tiffs were mosaicked into 15 larger areas to ease the image processing procedure.

A number if *in situ* sites were collected to provide reference information to drive the interpretation of the aerial photography. Reference sites were selected to match *in situ* references sites selected during the 2003 (Lathrop et al.) study. Reference sites were not selected in a probabilistic or random manner, but rather along targeted transects across the study area n ~ 136. In addition 15 sample sites were selected for a late season (October 2009) review of targeted areas of uncertainty of the imagery.

A second *in situ* n ~ 120 dataset was collected to provide a validation dataset. This validation dataset was selected in a pseudo-random fashion to focus on shallow water habitats mimicking the depth distribution of seagrass within the BB-LEH estuary. This validation dataset was collected and stored until after the imagery was classified. After the imagery was collected and classified, the validation data will be used to create an error matrix, a producer's and user's accuracy assessment and a Kappa Statistic measure of agreement between categorical datasets that accounts for agreement based on chance.



For all of the *in situ* data collected for this project (the reference dataset n ~ 136 and the validation dataset n=120), field collection was accomplished as follows. Field surveys were conducted using a 20-foot maritime skiff located at the Rutgers University Marine Field Station in Tuckerton New Jersey. Navigation to field locations was accomplished with a Garmin 530s marine GPS/Sonar system. Upon arrival at the pre-selected field locations, the boat weighted anchor. Water depth (centimeters) and transparency (Secchi) depth (centimeters) data were collected with a measuring stick prior to the introduction of the sampling grid or diver. This sampling protocol avoids disturbing sediment and lowering Secchi depth values. Next a 4 m x 5 m grid made of ¾ inch pvc was lowered over the side of the boat. The diver entered the water

and affixed a GPS Magellan Mobile Mapper 6 (2-5 meter horizontal accuracy) to the outside L of the survey grid (see figure above). A compass reading was taken along the left hand axis of the sampling grid. The compass reading and the GPS position allow precise placement of the sampling grid on the benthos to a higher level of accuracy then the boat-based GPS unit. The diver then visited grid 1 through 8 and recorded information on SAV presence absence (yes/no), percent cover of seagrass species (0 to 100 in 10% increments), and percent coverage macroalgae (0 to 100 in 10% increments). These data were vocally relayed to the boat captain who recorded the data on write-in-the-rain paper. Upon completion of field data collection, the GPS unit was removed and the sampling grid returned to the boat. Field sheets were then signed, dated, and entered into Microsoft Excel. The precise location of each sampling grid was determined using matlab and simply geometry (SOHCAHTOE) using the GPS location in UTM Coordinates and the compass bearing. A correction for magnetic delineation (difference between the north pole and the magnetic north pole) was calculated using NOAA website (*http://www.ngdc.noaa.gov/geomagmodels/Declination.jsp* for July 15th, 2009 for 39.9745 N 74.1514 W magnetic delineation equals 12 degrees and 47 minutes.

The aerial photography was combined with bay depth information extracted from the NOAA nautical charts (Lathrop et al, 2003) using an Arc Macro Language (AML script). Each image was first down-sampled using the aggregate command available in Arc Grid for a 4 x 4 grid window selecting the median cell value. This was done to remove areas of local light scatter from wave tops, to reduce the size of the imagery for processing, and to remove outliers. For each color image input as a red, green, and blue photography, two more bands were added. The fourth band was an inverse distance weighting (IDW) layer created from the NOAA Nautical Charts; the fifth band is a land water layer created from the New Jersey Department of Environmental Protection (NJDEP), New Jersey Coastal Land Water polygon. These five layers were then converted to one grid stack using the make stake command and then exported from a stack to a .tif file using the grid image command. The rectified mosaicked color photography was imported into Defines eCognition to support image segmentation and classification. eCognition segments raster data in an unsupervised method minimizing the intra-polygon variance while maximizing inter polygon variance. The user can control the weight of each imagery band by changing coefficient between 0 and 1 for each band and a unit-less scale parameter which determines the average vector polygon area. As the scale parameter is increased, the average size of the polygon is increased as well. Multiple nested polygons can be created by running a multiple resolution segmentation procedure. These nested polygons will always share the boundary of the larger polygon, making them a nested polygon.

Quality Objectives and Criteria

Positional data quality objective

The Root mean square error (RMSE) horizontal positional accuracy of the mapped seagrass habitat boundaries will be less than + or - 5 m. NOAA protocols (Finkbeiner et al., 2001) suggest that the horizontal positional accuracy should be less than 13 m.

Attribute data quality objective

The *in situ* reference validation data will be compared with the classified SAV/seagrass map, and an overall accuracy and Kappa statistic (a measure of agreement corrected for chance agreement) will be computed to provide an indication of the level of agreement between

the SAV/seagrass map and the *in situ* validation dataset (i.e., thematic accuracy). NOAA protocols (Finkbeiner et al., 2001) suggest that overall thematic accuracy (presence absence) should be greater than 85% and a Kappa of > 0.5. We will use these benchmarks of accuracy as our standard.

Final GIS vector files created by this project will be documented using Federal Geographic Data Committee (FGDC) compliant metadata. These metadata will include information on:

1) source
2) scale
3) resolution
4) accuracy

Use of the 2003 and 2009 Remote Sensing Surveys

The remote sensing data will be used across the entire estuary (not just in the northern segment). It will supplement the Kennish *in situ* data and provide a full synoptic view of seagrass distribution across the entire estuary. Bay wide seagrass distribution data can only be realistically collected in a practical sense using remote sensing surveys. It provides useful information on all seagrass beds and meadows, including spatial indicators not available through *in situ* surveys (bed patchiness, surface to volume ratio, depth of the entire bed edge, etc). In addition, a GIS dataset showing seagrass distribution provides the ability to ask specific spatial questions; for example, what is the area of seagrass habitat adjacent to boat docks? What percentage of the total seagrass habitat could be affected by boat docks? These types of questions for a large estuary can only be practically addressed using a GIS/RS approach.

In addition, the 2003 and 2009 seagrass remote sensing data will be compared to each other to determine differences in seagrass distribution across the entire estuary. The categorical terms used to describe seagrass distribution in the GIS data set (sparse, moderate, and thick) correspond to percent cover categories (10-39%, 40-79%, and 80-100%, respectively).

The 2009 remote sensing mission builds upon the knowledge of previous work. We determined after the 2003 survey that, while a 1 m quadrat did a good job of quantifying a specific location, it did not always inform us of larger spatial patterns. These patterns are useful in ground truthing aerial photography. Comparison of the 2003 and 2009 imagery is similar to comparisons with other legacy datasets; care must be taken to understand the limitations between the 2003 and 2009 datasets. That being said, these surveys have much more similarities than differences. They are both aerial photography surveys, which have been processed in the same way and in the same categories. Therefore, we feel it is appropriate to compare both datasets.

As part of the 2003 remote sensing mission, a number of *in situ* field sites were collected (n=245). All of these field reference sites were used in the classification procedure and therefore cannot be considered a true independent validation dataset. In addition, the 245 reference sites were not selected randomly across the estuary but rather directed towards known seagrass habitat. The 2003 study had a Kappa statistic of 56.2%, representing a moderate level of agreement between the classified imagery and the in situ reference sites. The overall accuracy

for the entire dataset was 68% for all classes of seagrass habitat and 83% for seagrass presence/absence. When we consider thematic accuracy, we are referring to seagrass presence/absence (across the entire estuary). Some of the reduction in accuracy is likely due to the fact that the reference dataset was not distributed across the estuary but rather directed towards known seagrass habitat as a training dataset. The reference sites were skewed to cover seagrass habitat; out of 245 reference sites 146 of the field sites contained seagrass habitat (60%), while seagrass only covered 14.5% of the entire estuary system. It is likely that a randomly selected validation dataset would have provided a higher level of total accuracy, in that we would expect a higher number of non-seagrass sites that would be randomly selected. Due to the comparatively low levels of error of commission for non-seagrass sites, we expect a higher level of overall accuracy. The un-weighted Kappa statistic provides robustness for this problem, by normalizing for the number of points collected for each class. For the 2003 dataset, the Kappa statistic was over the 0.5 target and represents a moderate level of agreement. Therefore, we feel justified in the use of this dataset. In the final report that we will discuss some of the benefits and limitations of using remotely sensed datasets and, in particular, these two different studies.

WATER COLUMN MEASUREMENTS

The first activities that should be conducted upon arrival at a field site are those that involve water column measurements. These samples/data need to be collected before disturbing bottom sediments and habitat.

Hydrographic Data Acquisition

Water-quality data will be collected at each site including temperature, salinity, dissolved oxygen (DO), pH, and depth. Secchi depth also will be measured at each station.

Basic water-quality parameters will be measured by using hand-held YSI sondes with cable connection to a handheld 650 MDS display unit. Prior to conducting a CTD cast, the instrument will be allowed 2-3 minutes of warm-up while being maintained at near the surface, after which, the instrument will be slowly lowered to an appropriate and consistent mid-water-column depth.

Water quality parameters will be measured at mid-depth with the YSI sondes by first ascertaining on-bottom (e.g., slake line/cable), then pulling up to mid-depth. Two to three minutes will be allowed for disturbed conditions to settle before taking measurements.

Secchi depth will be determined by using a 20-cm diameter white Secchi disc. The disc will be lowered to the depth, at which it can no longer be discerned, then it is slowly retrieved until it just reappears; that depth is marked and recorded as Secchi depth (rounded to the nearest cm).

Water Quality Indicators

Total Nitrogen

TN will be analyzed from water-quality (secondary) databases of the NJDEP.

Chlorophyll *a*

Chlorophyll *a* will be analyzed from water-quality (secondary) databases of the NJDEP.

Dissolved Oxygen

Dissolved oxygen measurements collected by NJDEP marine water quality monitoring will be used as a water quality indicator. Dissolved oxygen measurements collected by datasondes in the field will also be used.

Benthic Infaunal Community

Benthic infaunal samples to be considered for index development in this project will have been collected in past surveys as part of EMAP and NCA sampling protocols (i.e., secondary data).

Habitat

Several observations will be made in the field to document certain attributes or conditions of the site that will help to characterize the overall ecological health. Observations will be made and noted for the presence of marine debris. Also, if there is obvious evidence of disruptive anthropogenic activities (e.g., dredging or landfill activity), these observations should be noted with a brief description on the appropriate field form.

A standard look-up table will be accessible for the field crews to assist them in the annotation of sampling stations. This annotation often proves to be very important in post-analysis interpretation of the data. The CMECS framework should be considered for use in this effort (Madden et al. 2005, 2009).

Index Development

An important goal of this project is to develop an index of ecological condition for the BB-LEH estuary that may be extended to other New Jersey estuaries. This index can be used to assess and define ecological impairment, and hence it will be extremely useful to NJDEP and US EPA estuarine and marine environmental assessment programs. The basic methodology used in the National Estuarine Eutrophication Assessment (NEEA) model will be applied to develop a biotic index of eutrophic condition for the estuary (Bricker et al., 1999, 2007). 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. A numeric scoring system will then be used that computes an index value from key water quality and biotic indicator measurements in each of the three estuary segments. Contingent on data availability, an index will also be developed for aggregate years sampled during the 1989 to 2011 period. The specific water quality and biotic indicators to be used in the index development include: dissolved oxygen, Secchi depth, total nitrogen (loading), total phosphorus (loading); chlorophyll *a*; seagrass biomass, shoot density, blade length, areal cover, and epiphytic overgrowth; macroalgae abundance and areal

cover; brown tide blooms; shellfish (hard clam) resource, and estuarine susceptibility (water residence time). Benthic invertebrate data will also be examined and assessed for statistical validity and inclusion in the index development for the 1989 to 2011 period. Three levels of indicator impact will be assigned: low (0.25), moderate (0.5), and high (1). A numeric impact value will be calculated for each parameter in all three estuary segments, and an area weighted value will then be summed for each segment to obtain an overall index of eutrophic condition for the estuary.

It is important to note that a biotic index will be developed for each of the three estuary segments targeted in this study (see Figure B1-1) and then an overall index will be calculated for the entire estuary. The development of the index for the northern segment will not include in situ seagrass data similar to the other two segments, but it will include remote sensing data on seagrass and data on all other indicators that are being used in this study. Application of the index to the northern segment therefore may be limited due to a lack of some data for that segment.

B3. SAMPLE HANDLING AND CUSTODY REQUIREMENTS

The following section will outline data/sample accountability guidelines for the project. Although standard formats for data/sample collection and reporting will be established for field and laboratory activities, not all aspects of sample handling will be addressed by the forms alone. Therefore, additional written documentation may be required to comply with agency reporting of field and laboratory protocols.

FIELD DATA

Field Data Forms

The project field crews will record most of their raw field data on hard copy data sheets. The field crews will also use instrumentation with self-contained data-logging capabilities (e.g., datasondes) that store values in electronic format which can be downloaded later as electronic files. The template for field data sheets will be the one designed for previous bioassessment field data acquisitions in the estuary (Kennish et al. 2007b, 2008). All pertinent field data will eventually be transcribed into an electronic format; therefore the field sheets and electronic tables should closely resemble one another.

Site/Sample Identification

Sample sites will be numbered sequentially from south to north for the 120 sampling sites. Sample ID numbers will be marked both on a label inserted into each sampling bag and marked with ink on the outside of the bag. The type of sample will also be marked on the bag. For a specific site, the crews will be provided with an abundance of preprinted site ID labels that they can use to label field sheets, sample containers, or anything related to that site.

Bags will be labeled in situ using write-in-the-rain paper and pencils. Bags are individually processed (blades scraped, aboveground and below-ground biomass separated and cleaned, and dried) in the laboratory. For each collected biomass sample, two samples are dried and both samples are labeled with write-in-the-rain paper. Results are then entered into database management software (access) by the laboratory researcher.

Sampling packets for each site will be prepared prior to the sampling date by placing a complete set of field data forms and preprinted labels into a large envelope; mark or label the outside of the envelope with the site ID number, date, area, and sampling location. These packets can then be filed numerically in a box file or cooler for transport to the field. A day or two prior to a scheduled sampling, the crew can pull the specific site packet and label a complete set of sample containers (if the labels are not waterproof, they should be covered with clear cellophane tape), then consolidate the prelabeled sample containers, data sheets, and extra labels in an appropriate size plastic bag for easy storage and transport aboard the boat, come the sampling day.

Data Transfer

Field information recorded on hardcopy must be transferred to an electronic format. The hardcopy field data will be transcribed within a week of collection to the electronic format. The electronic format will be a template similar to the hardcopy form; the same data will be entered to the electronic file that was recorded in the field. The Project Quality Assurance Manager will conduct QA/QC checks on the transfer of hardcopy data to the electronic format.

Certain field data may be collected electronically (e.g., CTD casts). If possible, these files should be downloaded and reviewed while still on site to ascertain validity (screened for incomplete files or obvious outliers). If there are any apparent problems, attempts should be made to rectify the situation and resample if necessary. Certain ancillary information related to electronically logged data still must be recorded on hardcopy forms to document data quality associated with the activity (e.g., calibration information, QC checks, etc.). These data must be indexed to the event by location, date, and time (e.g., information to document that discrete samples were collected for Site XX at YY meters).

All electronic files created during field activities must be periodically backed up on disks.

Sample Transfer

While the project protocols will not require the stringency of Good Laboratory Practices (GLPs) Chain-of-Custody protocols, the following level of accountability is expected. When the field crew returns to the dock or staging area, they will turn both the field samples and respective data forms over to their land-based support team (or designated recipient) who will again verify that all samples are accounted for by comparing actual sample containers against the field data forms. Upon inventorying samples, the crew will then temporarily store the samples under designated conditions to await shipment or delivery to the processing laboratories. In the event that a sample is missing, the person checking in samples will record the sample as missing on the inventory sheet. The boat crew responsible for the collection of that sample will be informed so that they may check the sample storage areas on the vessel. It may be that conditions in the field prevented the collection of a particular sample; in that situation, the reasons should have been recorded as a comment on the field data form. If the sample is not recovered, the crew chief will make the decision for corrective action, whether simply to re-sample while still in the area or to schedule a make-up sampling on a later date.

Samples collected in the field will be held under temporary field storage before being shipped or delivered to an appropriate processing laboratory or held at a long-term storage facility. The following protocols will be applied:

1) SAV samples: To be processed in-house by laboratory staff.
Stored on ice in coolers indoors for no longer than 72 hours (Previous sampling and testing has shown that SAV samples hold well for 3 days on-ice). If processing cannot occur within the 72-hour period, the samples will be frozen on-site at RUMFS.

2) Macroalgal samples: Stored temporarily on-ice in 1L translucent Nalgene (or Fisherbrand equivalent) bottles. Upon return to the dock, the samples (in 900mL of water from the site) will be preserved with 100mL 40% Formaldehyde and stored indoors until transported to a laboratory for analysis.

3) Brown tide samples: Raw samples will be temporarily stored in opaque brown 1L Nalgene (or Fisherbrand equivalent) bottles on-ice in a cooler during transport back to RUMFS, where they will be preserved with Gluteraldehyde (1 part of the 10% glut solution to 9 parts sample: brings it to final fixative concentration of 1%). Storage will be in glass vials wrapped in Aluminum foil (to shield from light) in a refrigerator until transported to a laboratory (TBD) for analysis.

4) Phytoplankton samples: Temporary storage is the same as brown tide samples above (same container). Approximately 1L of sample will be preserved with Lugol's and stored in the opaque brown bottles in a refrigerator until transported to a laboratory (TBD) for analysis.

A complete invoice, listing each sample ID codes, date packed, and name of person who packed the samples will accompany every batch of field samples sent from the field to a receiving facility. The field unit will retain a copy of the invoice. On the receiving end, as each sample is unpacked it will be checked-off of the invoice as received and immediately stored under prescribed holding conditions. The person receiving samples will sign, date and file the invoice. The receiving facility should immediately report any missing samples to their respective project manager, who will initiate appropriate corrective action.

Once a complete set of field collected samples are received by a processing laboratory, a master list will be compiled of all sets of samples and where they reside (e.g., freezer A, refrigerator B, or storage shed Z). The master list should be filed in the general area where the samples are held. When samples are released to (or checked out by) an analyst, the transfer will be documented on the master list by initial and date; the quantity of sample released should be recorded. If the sample or portions of it are returned to the central storage area, this should also be logged on the master list. When the laboratory uses internal tracking codes, they must be indexed to the original sample ID code (both site and sample identifiers), and all analytical results will be reported using this code.

B4. ANALYTICAL METHODS REQUIREMENTS

Procedures for the various analyses used in the collection of secondary data used in this project are based on those developed for EMAP-E and specific details for the analytical processes are documented in existing documents. Where appropriate, this QAPP will reference those documents. Disposal of samples will be done responsibly; some nonhazardous aqueous

samples can simply be poured down the drain, nonhazardous solid samples can be disposed of as normal trash. However, samples or used reagents that contain any metals or other substances that are considered toxic or hazardous, either to humans or the environment will be disposed of properly and legally. Appropriate containers will be used for the temporary storage of materials to be discarded.

B5. QUALITY CONTROL REQUIREMENTS

If changes, modifications, or additions are made to the QAPP, these items will be set in writing and sent to all of the project signatories for their review and approval prior to any and all changes being performed.

Each analysis or measurement conducted for this project will have prescribed quality control (QC) checks with quality criteria or acceptable tolerances established, where applicable. In general, the QC guidelines for this work have been adopted from those developed for the EMAP-E (US EPA) quality program. For that reason, this document will summarize the key QC elements for the field and laboratory measurements. Table A7-2 and A7-3, in this document, present summaries of the measurement quality objectives and of the QA sample types for core indicators. General discussion of the QC for individual field and laboratory activities follows.

FIELDD ACTIVITIES

QC elements associated with field monitoring activities that relate to locating the sampling site, the collection and handling of environmental samples, and direct measurements taken onsite are presented in the following sections.

Locating station

Field crews will use differential Global Position Satellite (GPS) navigation systems to locate the sampling stations. Coordinates of latitude and longitude for the previously selected random sampling stations will be issued to the field crews along with their sampling packages. The vessel operator should review navigation plans for a site at least a day prior to the scheduled sampling. Before leaving the dock, the station position will be entered into the GPS system and the operator will safely navigate to the area. As the vessel closes in on the general location, the operator will decrease speed and allow the GPS to guide the vessel onto the general location and then weigh anchor. The boat will not be anchored directly over the sample site (which would impede the work of, and pose a hazard to, the diver in the water). A weighted float will be placed on the sampling site location, assuring a highly accurate point of sampling. The site location will be recorded on the Station Information Data Sheet.

In cases where the vessel cannot navigate to within 5 m of the intended site (e.g., the site is actually landlocked or the depth too shallow), the crew will record the station as "intended-unsampleable" and thoroughly document the reason(s) on the Station Information Data Sheet. The crew will then relocate to the nearest position that permits sampling and conduct the sampling. It is not anticipated that situations like that will occur very often and less likely if suspect areas were reconnoitered prior to the monitoring window. In rare cases of very shallow sites and very low tides, a waterproof handheld GPS unit will be used to locate the station by wading through the shallow area.

Water column measurements

Water column measurements will be made using multi-parameter water quality monitoring probes (e.g., data sondes) which are connected by hardline to a handheld display unit and measurements are manually recorded as the probe is lowered or retrieved through the water column at discrete intervals of depth. These measurements will be made by personnel from a state-certified facility of Rutgers University, for which all parameters being measured have current certification. A detailed standard operating procedure (SOP), which will be developed as a part of the certification process, will be followed during instrument maintenance and calibration, and data collection.

For this purpose, YSI 6000 series datasondes will be employed to collect water quality data. They will be calibrated according to YSI specifications. Alternatively, a self-logging YSI 600 or 6600 datasonde may be used which will autonomously record data to be retrieved upon returning to port.

For each of the water quality parameters, EMAP (US EPA) has established a maximum range of allowable difference that the instrument may deviate from calibration standard (Table B5.-). It should be noted that while these limits are acceptable for the purpose of qualifying field measurements taken with the unit, when performing the daily QC check, crews should set the instrument to as near the standard as possible. The daily QC checks should not require more than slight adjustments to bring the instrument into agreement. If an instrument's performance becomes erratic or requires significant adjustments to calibrate, the unit should be thoroughly trouble-shot; problems generally can be determined as being probe-specific or related to power source (e.g., low battery voltage or faulty connections). Routine maintenance and cleaning should be performed as per the manufacturer's recommendation.

Table B5-1 Maximum acceptable differences for instrument field calibration and QC checks.

Instrument	Frequency of Check	Parameter	Checked Against	Maximum Acceptable Difference
Datasonde	Daily	Temperature	Thermometer	+ 1°C
		Salinity	Standard seawater	+ 0.2 ppt
		pH	pH buffer solution	+ 0.1 pH units
		DO	100% saturation	+3.0%
		Depth	Sea level	+0.2m

Failed QC or calibration checks should initiate a thorough inspection of the unit for obvious sign of malfunction (e.g., loose connections, damaged probes, power source, fouling on DO membrane, etc.). After any maintenance to correct problems, the unit will be re-calibrated with documentation on the appropriate field data form. In most cases, unless a probe is actually broken or damaged, the datasonde can be corrected in the field. If the unit will calibrate within the guidelines, water column measurements can be continued. If one or more parameters remain suspect, the nature of the problem should be fully documented on the field form, and the situation should be reported to the Project QA Manager for resolution. Depending on the

importance of the suspect parameter, the site may require a revisit to log an acceptable water column profile. Of course, it is always advisable to have a backup instrument available.

Secchi disk

No field calibration procedures are required for the Secchi disk. The disk must be clean, free of algae or other debris, and all surfaces white in color. All surfaces on the disk must be in good condition such that they are clearly visible. QC procedures, when using the Secchi disk to make water clarity measurements, include designating a specific crew member as the Secchi depth taker; taking all measurements from the shady side of the boat; and not wearing sunglasses when taking Secchi readings.

Pre-labeled Sample Containers

The following sections describe QC/QA procedures related to the collection of field samples developed for EMAP-E. Proper labeling of samples is a very important QA aspect and cannot be overstressed. All sample containers for a site should be pre-labeled prior to arriving on station. Pre-labeling clean, dry containers helps to ensure that labels adhere properly to the containers. A little bit of sea spray or condensation wreaks havoc on labeling. Therefore, affix all labels to sample containers in the clean comfort of the lab or motel; not at the dock, not onsite. It is best to have a "sampling packet" for each station consisting of data sheets, lat/long coordinates of station, prelabeled containers, and extra labels - all contained in a single plastic bag. The crew can then grab the packets for that day's stations, along with an extra unlabeled set, as they head out for the day.

Water Quality Samples

Field procedures for the collection of water quality samples based on EMAP-E sampling procedures basically involve the collection and filtration of water samples. This project will use secondary water quality data collected by NJDEP for application in this project and will not use water grab samples for water quality analysis.

Data Reporting Units

Both field measurements and results of laboratory analyses should be reported to the Project Quality Assurance Manager in standardized formats. Table B5-2 list the preferred data reporting format for the core indicators. Criteria included in Table B5-2 are applied to both new data collected as part of this project as well as previously collected (secondary) data. Chlorophyll *a*, TSS, TOC, and silt and clay fractions (sediments) will be applied to secondary data in this project. It is anticipated that measurements recorded by the various dataloggers will not all be displayed to the same number of places and that there will be differences due to the use of significant figures; however, effort should be made to maintain uniformity.

TABLE B5-2. Data reporting format for EMAP-E type monitoring.

Laboratory Analysis	Units	Minimum Reporting Level
Dissolved oxygen (DO)	milligrams per liter (mg/L)	0.1
Salinity	parts per thousand (ppt)	0.1
pH	pH standard units	0.1

Temperature	Degrees Centigrade	0.1
Water depth	Meters (M)	0.01
Water clarity, Secchi Depth	meters (M)	0.1
Chlorophyll *a*	micrograms per liter (µg/L)	0.1
Total suspended solids (TSS)	Milligrams per liter (mg/L)	0.1
TOC fraction of composited sediment	percent	0.1
Silt and clay fraction of composited sediment	percent	0.1

B6. INSTRUMENT AND EQUIPMENT TESTING, INSPECTION, AND MAINTENANCE

Several pieces of equipment that may be utilized to collect or analyze environmental data for EMAP-E type sampling should have periodic maintenance and calibration verification performed by manufacturer's representatives or service consultants. These procedures should be documented by date and the signature of the person performing the inspection:

Meters-biannual verification of calibration coefficient by manufacturer; analytical balances-annual verification by service representative; analytical Instrumentation (TOC Analyzer) -as per need based on general performance; service contracts recommended. All other sampling gear and laboratory instrumentation will be maintained in good repair as per manufacturer's recommendations or common sense to ensure proper function.

B7. INSTRUMENT CALIBRATION AND FREQUENCY

Appendix 3 shows a list of field equipment and instruments used in this project. Both field and laboratory equipment and instruments require routine calibration checks to verify that their performance is within acceptable quality standards. The following sections will discuss the procedures and frequency for the various instrument calibrations that are key components in the collection of accurate environmental data.

CALIBRATION PROCEDURES AND PREVENTIVE MAINTENANCE

An SOP will be developed and followed closely while maintaining, calibrating and operating data sondes and associated equipment. The QA procedures described in the SOP will be at least as detailed and rigorous as those described below.

Datasondes (dataloggers) (Yellow Spring International)

Vendor and Address:
YSI Incorporated, World Headquarters
Yellow Springs, Ohio

Phone: +1-937-767-7241
800-897-4151
Fax: +1-937-767-1058
Email: environmental@ysi.com

YSI Model used: 600XL datasonde (model #600XL, p/n 065863) paired with a 650MDS display ("handheld") (model #650MDS, p/n 650-04)

The following methods will be utilized to calibrate the probes on the 600XL; calibration procedures will be employed just prior to the commencement of each sample period (June 2010= sample period 1, August 2010= sample period 2, October 2010= sample period 3).

Calibration standards required for pH will be purchased from Y.S.I. (pH7: model #3822, p/n 003822, pH10: model #3823p/n 003823). A two-point calibration will be employed for pH, the first being pH 7 followed by pH 10. Calibration of the pH probe (model #6561, p/n 605091) will be performed via immersion in the standards and using the calibration feature of the 650MDS as per the manufacturer's instructions (i.e. allow probe to run for 60 seconds before accepting the value).

Calibration standards required for conductivity will be purchased from Y.S.I. A standard of 10 µS/cm (model #3168, p/n 060911) will be used to calibrate the conductivity probe (model #6560, p/n 006560). Calibration of the probe will be performed via immersion in the standard and using the calibration feature of the 650MDS as per the manufacturer's instructions (i.e. - allowed probe to run for 60 seconds before accepting the value).

Dissolved oxygen will be calibrated via submersion of the datasonde in a bucket of aerated tap water. The membrane (model# 5793, p/n 098094) and electrolyte (a half-saturated KCL solution; no model or p/n available, it is provided with the membranes in a "kit") is changed whenever the dissolved oxygen membrane is changed) on the oxygen probe (model #6562, p/n 006562) is changed as per the manufacturer's instructions and when anomalous data is recorded (anomalous data defined as: 1) when dissolved oxygen reads in excess of +/- approximately 5% in a 100% saturated environment (i.e.- aerated bucket of water) and cannot be rectified by re-calibration of the probe. 2) when dissolved oxygen values do not remain stable (i.e. within +/- 1%) when run continuously (sample frequency= 1 second) in a in a 100% saturated environment (i.e.- aerated bucket of water)), when bad diagnostic values (outside the following ranges: dissolved oxygen charge: range 25-75, dissolved oxygen gain: range 0.8-1.7, bad high/low test: values start high and drop to a stable point when calibrated or run continuously (sample frequency= 1 second) are observed during calibration or post-calibration (bad diagnostic values are considered "benchmark" indicators (within range= good, out of range=bad) and are not charted or recorded.), when the DO membrane is visibly punctured, folded, or otherwise damaged, or when the terminals of the DO probe are tarnished or otherwise discolored.

While not utilized to record depth in the field (a hybridized "depth stick" and Secchi disk will be utilized for this purpose), the depth sensor (no model or p/n, integral to the 600XL unit) will be calibrated along with the aforementioned probes/sensors; this will be necessary because the generation of the aforementioned values are dependent on the depth value, which is utilized by the 600XL datasonde in their calculations. Calibration of the depth sensor will be performed using the calibration feature of the 650MDS as per the manufacturer's instructions (i.e. - allow sonde to run for 60 seconds before accepting the value).

QUALITY CONTROL CHECKS

Water (Physical Parameters) Data

Post-deployment calibration checks of the Y.S.I. 6600EDS datasondes will be performed with standards of known measure to verify that the sensors were operating correctly during deployment. Erroneous measurements and/or poor diagnostic values will warrant further scrutiny of the data collected. Data will likewise be visually reviewed post-deployment, both raw numerical data and plots of the data recorded during the day's work.

Performance evaluation samples for each parameter of interest in this project shall be performed each year, and the results must be acceptable prior to performing the analysis.

FIELD CALIBRATIONS

To ensure that field measurements meet the accuracy goals established for EMAP-E type (US EPA) projects, quality controls checks are performed on a regular basis for most of the field equipment/instruments used to generate monitoring data. When QC checks indicate instrument performance outside of acceptance criteria, the instrument will be calibrated (for those instruments that allow adjustments) against an appropriate standard to re-establish acceptable level of performance; the procedure will be documented on field data forms.

Some instruments have fixed functions that cannot be adjusted under field condition. In cases where these types of measurements fail the field-QC checks, the degree of variance will be documented in field records; if possible, the situation will be rectified by changing out the faulty equipment with a backup unit until the failed unit can be repaired. If no backup is available, depending on the relative importance of that particular measurement to overall success of the monitoring operation, the crew chief must decide whether to continue operations with slightly compromised or deficient data or to suspend sampling until the situation is corrected. For example, if the GPS system is found to be totally unreliable, sampling activities should be suspended until a reliable unit is in place; to continue field operations without GPS to locate sampling sites would have dire consequences to the study design. On the other hand, if a pH probe were to break or become faulty, sampling could continue without seriously compromising the overall characterization of the environmental condition for a site. It becomes a judgment call, and if the crew has difficulty in making a decision, they should call their State QA Manager for guidance.

Differential GPS

A functional differential GPS system provides very accurate positioning data and, when in use on a regular basis, can be relied upon to operate properly from day to day. The units have a signal strength display that indicates the degree of accuracy at which the unit is currently performing. If signal strength is nominal the unit should be accurate within ~ 6 m; a weak signal may reduce accuracy to a level of ~30 m. Even though the GPS may appear to be problem-free, it should still be periodically verified by checking against a known location, such as the coordinates of latitude/ longitude for home dock or a fixed navigational marker. These verifications should be done daily in an informal mode (quick check as vessel is being readied for day) and at least once per week with documentation in the vessel logbook. If the QC check indicates the GPS to be off by more than ~60 m of the known position, it is necessary to wait for

a stronger signal or for possible interference to clear then re-check. If the unit consistently fails, a replacement should be put online.

Hydrolab Water Quality Probes (or similar unit)

Because Hydrolab Corporation's H20 Multiprobe water quality instruments have been extensively utilized in previous EMAP-E monitoring programs, this section will present calibration details specific for that instrument. The actual instruments used for EMAP-E type field monitoring may be models or brands different from the H20, but the procedures discussed here should be generic enough to address the QC issues for most other instruments of a similar design.

Hydrolab Corporation's H20 requires calibration checks on a daily basis during periods of use. The H20 is used to make instantaneous (real time) measurements that are read from a deckside display unit while the probe is lowered and raised at discrete depth intervals (e.g., at 1-m increments) through the water column. Calibration procedures are described in detail in the Hydrolab Scout 2 (display unit) and H20 (probe) Operating Manuals (and Performance Manual) (Hydrolab Corporation, 1991). The Hydrolab units will be used in applications to measure dissolved oxygen (DO), salinity, pH, temperature, and depth. Discussion of the calibration procedures and standards specific to the individual parameters follows.

DO will be calibrated by allowing the probe to equilibrate in an air-saturated-with-water environment, which represents 100% DO saturation at conditions of standard atmospheric pressure (760 mm Hg). This environment is established by positioning the polarographic DO sensor in a calibration cup that is filled with freshwater to a level just below the surface of the sensor's membrane and then placing a lid or cover over the cup to create a saturated humidity. When equilibrium is attained, the operator will activate the Hydrolab instrument to accept the condition as the calibration input for 100% DO saturation. Once calibrated, a properly functioning instrument should hold its DO calibration from day to day with only a slight drift of 2-3% from the 100% saturation standard; drift exceeding that level is indicative of the need to change the membrane and electrolyte solution. The DO meter shall be checked with the Winkler Method each week of use of the dissolved oxygen meter (at a minimum).

The pH probe requires the establishment of a two point calibration curve using two standard buffer solutions to bracket the nominal range of pH expected to be measured. For NJCBI, standard buffers of pH 7.0 and 10.0 will be used to calibrate the Hydrolab equipment. The buffer solutions must be commercially supplied with accuracy of + 0.02 pH units (or better), referenced to NIST SRMs; calibration solutions should be replaced with fresh buffer every 3-4 days.

The conductivity /salinity cell will be calibrated using a secondary, seawater standard that has had its salinity referenced against a certified standard. These procedures and results data for the preparation of the secondary standard will be logged into a QA notebook that will be maintained by State Field Managers or in-house QA personnel. Salinity of the seawater standard should be generally representative of the conditions expected in the field (e.g., for NJCBI, a mid-range salinity, 20-30 ppt). A bulk supply (5 gal) of the secondary standard can be maintained in a central location and field crews should replace their calibration allotments (300-500 ml portions) with fresh standard every 3-4 days, or at any time that it becomes suspect.

The depth sensor (a pressure transducer) is calibrated to 0.0 m of depth while the instrument is non-immersed (absence of water pressure); this in effect becomes the standard for depth calibration.

The temperature function of the Hydrolab instruments is set by the manufacturer and cannot be adjusted or calibrated in the field; historically, during 5 years of EMAP activities, there have been no malfunctions with Hydrolab's temperature sensor. However, as part of the daily calibration checks, the instrument's temperature reading will be compared to that of a hand-held laboratory thermometer (accuracy, + 1°C) as a pass/fail screen.

LABORATORY CALIBRATIONS

Analytical Instrumentations: An array of laboratory-based stoichiometric determinations will be conducted with a variety of environmental samples collected for EMAP-E type studies. These analyses require extensive utilization of certified standards for instrument calibration, plus, many incorporate the use of standard reference materials (SRMs) as a routine QC samples. The analytical standards and SRMs for all analyses will be provided by established, reputable suppliers and when available, only certified materials will be used; in cases where certified standards are not available, the analysts will obtain high purity (e.g., analytical or reagent grade) compounds to prepare in-house standards. Although the following is not a complete list, it will serve to indicate the degree of quality expected for analytical standards used to calibrate and verify analytical instrumentation:

- Analysis of total organic carbon (TOC) in sediment: NIST acetanilide standard certified reference materials such as BCSS-I (NRCC)
- Analyses of eutrophication indicators in water: Chlorophyll- Chl a extract from Analysis (Sigma Chemicals)

In general, instrument calibration for the above analyses should be verified at least twice during a batch run (i.e., continuing calibration check), when appropriate, somewhere near the middle of the run and at the end. If the analyses are run on a continual basis, the end of one run is essentially the beginning of another; if the analysis is down for a period or discontinuous, then an initial calibration check must be conducted with the first batch of the renewed series.

General Laboratory Equipment: This category includes the routine tools common to most laboratories (e.g., analytical balances, drying ovens, freezers, etc.); if not actual calibration, all of these require some documentation of performance. Each piece of equipment should have an assigned logbook in which the calibration or performance records are maintained.

Of particular interest are records for the analytical balances used for weighing out standards or analytical samples. These balances must be maintained under the manufacturer's recommended calibration schedule and the performance of the balances should be verified before each series of weighings by using a set of NIST (or previous NBS)-approved standard weights. If the performance of a particular balance is historically stable, then the verifications may only be required on an appropriate periodic basis (e.g., weekly). As much as possible, the verifications should be conducted using standard weights that reflect the magnitude of the actual weighing. The results of the verifications should be recorded in the logbook for the balance.

Certain types of samples (e.g., chlorophyll) require storage under extremely cold conditions « -50°C). These samples should be held at -70° C in an ultrafreezer that will activate an alarm if the temperature exceeds -65°C. Other equipment such as sample drying ovens should be monitored on a routine basis during periods of use to ensure their performance.

B8. INSPECTION/ACCEPTANCE REQUIREMENTS FOR SUPPLIES AND CONSUMABLES

(Element required for QA Category I documents only.)

B9. NON-DIRECT MEASUREMENTS

COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

Sources and quality-assurance criteria of secondary water-quality and stream-flow data are discussed in Section A7, Component 1. USGS/NWIS and NJDEP are the sources of secondary data for this investigation. If additional sources are identified during the course of the project, those data will be subject to same review of methods and values that the USGS and NJDEP data received. The sources of secondary data gathered during this investigation will be identified in all project deliverables. No data hierarchy will be administered; all water-quality and streamflow data that meet the data-quality objectives will receive equal consideration.

The rationale for selecting the USGS/NWIS data is that they are the most complete, well-reviewed water-quality and streamflow data available for the Barnegat Bay watershed. The rationale for selecting the NJDEP water-quality-monitoring data is that these data provide an extremely detailed analysis of nutrient loading in four tributaries to the Toms River, NJ, which is not available from any other source.

Secondary data will be used to determine the spatial and temporal variability of nutrient loading in the BB-LEH watershed. These data are essential for simulating spatial and temporal variability of loading using the mathematical models BASINS3 and PLOAD. Acceptance criteria and key data resources are described in Section A7, Component 1.

B10. DATA MANAGEMENT

This investigation requires the accumulation of many disparate types of data from many sources, as described elsewhere in this QAPP document. An additional challenge to data management is that the investigation will be conducted at two different research centers (USGS and Rutgers). The following sections describe the data management procedures that will be followed.

All data retrieved and produced will be available to all cooperating agencies. Specific features of data formatting will depend upon the types of data requested and the needs of those requesting the data. Efforts will be made to accommodate those requesting data, for example, if feasible, data requested by NJDEP would be provided in a format that can be retrieved by the Water Quality Data Exchange database.

COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

All data that will be used in Component 1 currently exists electronically, on data systems managed by Federal, State and Local government entities, and are subject to review, security, back-up and QA procedures as specified and required by the owners of the data. The USGS New Jersey Science Center data system consists of a network of individual-user computers which use the Microsoft XP-Pro operating system, connected to a network of four servers, which function as primary data storage units. The New Jersey data system is part of the national USGS data system, which is subject to rigorous operating and security requirements, as mandated by federal law. All users of the USGS data system must complete an annual IT Security training course and pass an exam in order to be qualified to use any part of the system. Backups of all files are conducted at no less than monthly intervals for all data stored in USGS/NJ servers.

This investigation will require retrieval and assembly of data from all of these disparate sources, and management of a secondary database in order to perform nutrient-loading calculations and simulations with the models BASINS3 and P-LOAD. A database format was developed by USGS personnel for use in previous investigations that included nutrient loading calculations. This will be done in the following steps by R. Baker (Project co-manager, USGS) and C. Wieben (Project Staff, USGS):

- Retrieve or download available data from appropriate primary source,
- Review data for completeness and accuracy,
- Add new data to the Microsoft Access database
- Check new entries in Access database for errors and consistency with primary-source data

Relationships among the data tables in the Access database are shown in Figure B10-1 below. Each table-value entry is unique, and all relationships are unambiguous. The tables that constitute the database are:

- Events: Meteorologic events are defined here as storms or base-flow periods during which water-quality data were collected. Events are numbered chronologically, starting from the first date for which water-quality data that meet data-quality objectives are available. Event type (storm or base-flow), start and end time, and season are listed in this table.
- Site: Each location in the study area from which data were obtained is given a unique site name and number, and some descriptive information
- Samples: each event is made up of one or more samples, each referring to a specific point in time for which data are available.
- Parameters: Water-quality and physical characteristics that are measured are described here.
- Filter: This table is used to specify whether samples were filtered before analysis
- Units: this table specifies units (e.g. µg/L, mg/L, etc.)
- Results and Results Flag: Water-quality values (parameter concentrations) and comments associated with samples are given here
- Method: Information about the analytical method(s) is given here

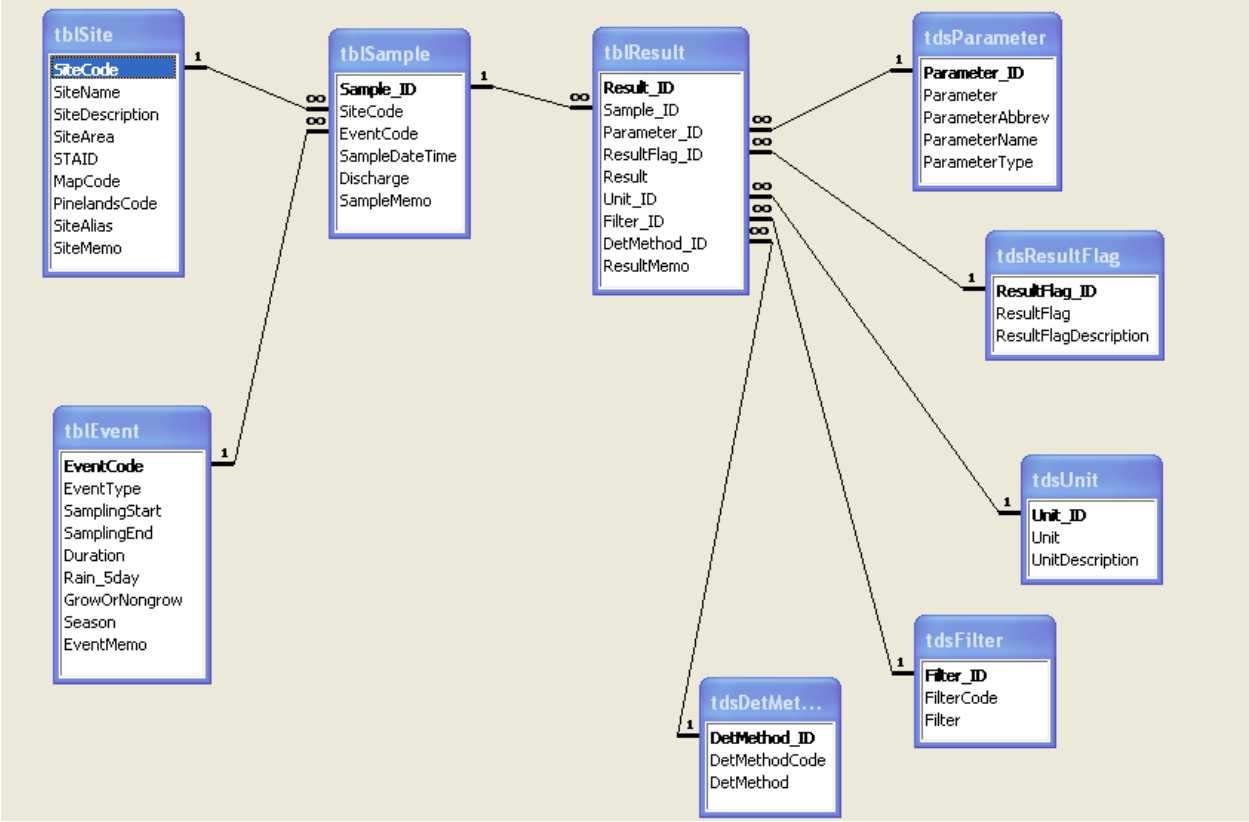


Figure B10-1. Relationships among the data tables in the Access database for secondary data

COMPONENTS 2-4 (ESTUARINE BIOTIC RESPONE, BIOTIC INDEX)

Data used in components 2-4 fall into three categories: data collected by federal and state entities with internal data management procedures; data collected by the research team for previous research projects; and data to be collected as part of the validation and assessment section of this proposal. Data management will be conducted at the Center for Remote Sensing and Spatial Analysis, Rutgers University by Benjamin Fertig, overseen by Michael Kennish (IMCS) and Richard Lathrop (CRSSA).

The Grant F. Walton Center for Remote Sensing and Spatial Analysis (CRSSA) maintains networked computers and a multi-terabyte storage devices for its research and instructional labs, as well as facility offices. The storage disks containing the research, project, and instructional databases, as well as project websites, are backed up on a bi-weekly basis.

Additionally, the CRSSA Data Bank and Image Library is a centralized data system which provides CRSSA faculty, staff, and students with geospatial data and imagery for its research, project, and instructional activities. In addition to these resources, CRSSA also builds, manages, and maintains its on-going research and project databases, as well as an archive of past-project/legacy data sets.

These components will require data retrieval and assimilation from a variety of data sources using different storage and QA/QC techniques. Digital database data (excel, access, storet) will be compiled in a Microsoft Access relational database. For geographic datasets, data will be stored in Environmental Systems Research Institute (ESRI) Geographic Information Systems data format(s). Steps listed below include:

For Electronic Digital Database data (ex NJDEP Marine Water Quality Data):

1). Download and store original data from primary sources.
2). Import primary dataset into Microsoft access.
3). Check data for both completeness (number of records compared to original dataset) and accuracy (correct level of precision).
4). Import data into Matlab Statistical Software, SAS, and R Statistical software for analysis.
5). Compare imported matlab data to the original primary source dataset to ensure import was successful, and no data were lost or altered.

For Geographic Information System Data:

1). Download and store original data and metadata when available from primary sources.
2). Check for errors with the defined projection system and datum using Arc Catalog GIS software ESRI.
3). If the data are not in ESRI-defined GIS formats, import the data using Arc tool box GIS software by ESRI.
4) For data converted into ESRI format, compare imported GIS data to the original primary source dataset to ensure import was successful, and no data were lost or altered.
5) Data analysis of GIS datasets will be undertaken in ArcMacro Language (AML), ArcToolbox, ArcMap, ArcCatalog, Matlab, R, Ecognition, and Erdas Imagine Software Packages.

Data analysis:

Data analysis will be undertaken in ArcMacro Language (AML), Arctoolbox, ArcMap, ARC Catalog, Matlab, SAS, R, Ecognition, and Erdas Imagine Software Packages. The software package used will be dependent on the specific spatial, temporal question and the input data type. Point data (one specific point in time and space) will be analyzed using the R, SAS, and Matlab statistical software packages, polygonal datasets (vector GIS), while being analyzed using the ESRI suite of GIS software, and for Raster surfaces (Aerial Photography) Erdas Imagine and Defiens Ecognition.

C1. ASSESSMENTS AND RESPONSE ACTIONS

The following sections outline the structured data reviews and assessments of data quality planned for the project. Note: Routine audits will be conducted by the Quality Assurance Officer during the course of the project, and will include review of any project environmental data collection activity.

NEIWPCC and EPA may implement, at their discretion, various audits or reviews of this project to assess conformance and compliance to the quality assurance project plan in accordance with the NEIWPCC Quality Management Plan.

FIELD MONITORING

Field Crew Certification

Prior to the start of the 2011 field monitoring, each field crew will be required to complete a 1-2-day field training to be authorized to collect actual field data and samples. Training will consist primarily of hands-on sessions during which field crew members will be instructed by the QA Manager (and associates) on the sampling methods and protocols developed for the project. If the schedule permits, training for each crew should culminate with a certification exercise in which crew members are observed and evaluated as they perform the full suite of core field activities (i.e., complete sampling for a sampling site). Although that is the preferred approach, because of time and logistical constraints, it may be necessary to certify the crews as they master each major component (e.g., sediment grabs for surficial sediment), then move on to the next, without observing in the context of a real world situation. Crews that successfully demonstrate technical competence and a thorough appreciation of field QA/QC requirements will then be authorized to initiate field activities. If a crew fails to qualify on some aspect, the members will receive further instruction in the area of their deficiencies until they perform at an acceptable level.

Field Reviews

Field teams will be responsible for the collection of environmental data and samples from the sampling sites. NEIWPCC and US EPA develop standard protocols and guidelines to help ensure that the data collected are of known quality. These guidelines allow for the use of different equipment (e.g., various hydrographic meters, work vessels, etc.) as long as the data generated meet acceptability criteria. Such performance-based QA/QC is a key factor to the project success in deriving comparable data from diverse participants. Prior to the actual collection of field data, the field crews are instructed in the approved field methods and protocols during their required initial training.

Any minor deficiencies observed during field surveillance (e.g., slight deviation from approved procedures, labeling irregularities, data reporting, etc.) should be immediately pointed out to the crew and corrective actions imposed on-the-spot. The evaluator will document with a brief note on the checklist and no further write-ups are required. If significant deficiencies (i.e., data quality is seriously compromised) are observed, the evaluator will make the appropriate on-the-spot correction, and, if the case warrants, call a halt to the field activities until the problems are resolved to the satisfaction of the Project QA Manager. All cases of this nature will be documented through a written report submitted to the Project QA Manager. A completed checklist along with a copy of the completed field data forms from the station provides the basic documentation for an evaluation of the crew's overall performance at that site.

LABORATORY ACTIVITIES

Analytical Chemistry

This project will not be collecting water grab samples for analytical chemistry. It will use secondary data collected by the NJDEP.

New Jersey State certification will be obtained for the Rutgers facility that will collect basic water-quality data using data sondes, as stated above.

C2. REPORTS TO MANAGEMENT

The contractor will prepare progress reports and submit them to NEIWPCC, as required by the contract between NEIWPCC and Rutgers. Additionally, progress reports and presentations (described in the following subsections) will be prepared for USGS Science Center management and Rutgers University management.

COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

All research projects conducted at the USGS/NJ office are assessed biannually in a formal process termed “Project reviews”. A panel consisting of the New Jersey Science Center management team, specialists in all relevant disciplines, and project management and staff convene and review the progress of each project. A Project Review document is submitted by Project Management (R. Baker, Project Co-manager) for discussion at the meeting, and includes:

- Project details: Project name, Project number, Project chief, List of planned reports.
- Brief purpose and scope.
- Project reports: List of all project publications. Approval date for annotated outline.
- Project budget: Estimate of percent funding spent, and percent of work completed. Include budgets for both the current fiscal year and for the entire project.
- Number1 Issue: Project chiefs number 1 issue/concern that may impact the project.
- Response to comments from previous project reviews
- Project Accomplishments: List of major tasks completed in past six months (include status or completion of any action items).
- Schedule/Plans: List of major tasks for next six months.
- Technical concerns: List of any technical issues/concerns you are currently working on.
- Manpower/Support: Discuss short-term and long-term needs.
- Data coordination (from cooperators): List of types of data that will be provided by cooperator and the planned dates to receive the data.
- Database Management: Types of data collected, plans to store the data, and plans to get data into district databases. List of any data collected that does not fit into NWIS databases.
- Updated Microsoft Project timeline and task list. Provide reasons for any tasks that are delayed 2 or more months from the last timeline presented.
- Recent budget information.

- Report outline: Copy of approved report outlines for a projects first review. Copies of report Table of Contents if major changes in report organization have been made since last review.

The Panel creates a list of "action items", to which Project Management must respond within agreed-upon time frames. All projects are subject to the review process until they are completed (including publication of reports and close-out of funding).

Information and records on Component 1 of the project will be prepared by the contractor and included in quarterly progress reports submitted to NEIWPCC.

COMPONENTS 2-4 (ESTUARINE BIOTIC RESPONE, BIOTIC INDEX)

The contractor will also prepare information and records on Components 2-4 of the project and submit them in quarterly progress reports to NEIWPCC, as required by the contract between NEIWPCC and Rutgers. These progress reports are due July 15, October 15, January 15, and April 15 of each year throughout the duration of the project. These reports will cover biotic as well as all other components of the project. The Program Manager will oversee this report preparation.

The status of Components 2-4 of the project will be examined in detail on a quarterly basis at meetings involving the Rutgers project management and staff team. At these meetings, results of project field surveys, laboratory analyses, data synthesis, and other activities will be covered. The Principal Investigators, Co-Principal Investigators, and staff support personnel will meet regularly to provide updates on the status of the project components. Project accomplishments and project deficiencies will be discussed in detail. Other elements will include the budget, technical issues and concerns, data management, staff and manpower considerations, and the project schedule. These meetings will form the foundation for quarterly progress reports to be submitted to NEIWPCC each quarter.

Information and records on Components 2-4 of the project that will be included in the quarterly reports to NEIWPCC are:

- Project name, project number, project manager, investigators.
- Objectives, purpose and scope.
- Project accomplishments.
- Field sampling and laboratory results.
- Status: Estuarine biotic response data.
- Status: Secondary Data.
- Status: Biotic index development.
- Status: Eutrophication assessment.
- Data provided.
- Database Management: Types of data collected, data storage and use.
- Technical concerns and problems.
- Personnel: Status and needs.
- Schedule/Plans: Tasks for next four months.

Development and Current (2010-2011) Eutrophication AssessmentT

During the implementation and execution of the project, reports are required to appropriately document QA/QC activities and to ensure that Rutgers University management is aware of pertinent items related to the general status of the project. The following reports will be expected on a routine basis, but other reports may be warranted as situations dictate.

Status Reports

Periodic status reports should be generated from both the participating investigators and possibly from within the NEIWPCC and US EPA management team. Each core activity should submit a general summary report stating their progress on the tasks with emphasis directed to any QA/QC issues.

Performance Evaluations and System Audits

All performance evaluations and system audits are the responsibility of the contractor; however, NEIWPCC and EPA may implement audits at their discretion, as described previously. Performance evaluation samples for each parameter of interest in this project shall be performed each year by the contractor, and the results must be acceptable prior to performing the analysis. The audit will consist of verification that methods and procedures specified by the QAPP and SOPs are being properly followed, assessment of project progress relative to the project objectives and timeline, and tabulation of problems that were encountered and descriptions of how they were addressed. Results of the audits will be included in the quarterly project progress reports.

The results of initial laboratory performance evaluations (PEs) will be submitted to the Project QA Manager for review. If the laboratory's results clearly meet NEIWPCC and US EPA quality criteria, the Project QA Manager will issue a letter of approval to the laboratory authorizing them to commence analyses or processing with field samples. If the laboratory's initial PE results appear deficient, the Project QA Manager will report his assessment and recommended actions to the Project Manager for concurrence or alternative corrective action. Based on that outcome, the Project QA Manager will then issue a letter to the laboratory detailing the recommended actions.

The results of all system audits (e.g., facility visits or field reviews) will be reported by the reviewer to the Project QA Manager (if other than he conducted the review). The project QA Manager will evaluate the review and formulate corrective actions where needed. As with PE evaluations (discussed above), if there are no significant deficiencies, the Project QA Manager will issue a final report of the audit results and the corrective actions, where needed, to the Project Manager and Project QA Manager, with copies sent to all key personnel involved with the project audited. If the audit results indicate serious problems or deficiencies, the Project QA Manager should be notified immediately.

Periodic Data Assessment and Quality Assurance Issues

The Project QA Manager will remain in contact with the Project Manager through personal communications during the extent of the project. As specific phases of the project are completed, the Project QA Manager will provide the Project Manager with a summary report detailing the overall data quality for that activity.

When audits of data quality are conducted onsite, the lead auditor should issue a short verbal briefing to the key personnel at the facility being audited as part of an exit interview. The briefing should address any significant observations, both positive and negative, and provide the staff with a general sense of the audit's results. If possible, a short written interim report should be prepared by the audit team and left with the appropriate staff members. A formal written report of the audit results will be issued within a month by the audit team addressed to the NEIWPCC and US EPA Project Officers and distributed to the Project QA Manager.

Anytime, when a significantly negative QA issue is encountered, it must be immediately reported to the Project QA Manager or Project Manager, who will assess the matter and, if necessary, consult with appropriate advisors to formulate corrective actions. Finding of this nature must be detailed in a report submitted to the NEIWPCC and US EPA Project Officer.

If a QA Summary Report is to be prepared, it is the responsibility of the contractor. After the completion (all analytical results reported) of the project, the NEIWPCC and US EPA management team may issue a QA Summary Report for the entire study. This report would be made available to all parties who express interest.

D1. DATA REVIEW, VERIFICATION, AND VALIDATION

REQUIREMENTS

COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

Criteria for accepting, rejecting, or qualifying project data are described in: A7. QUALITY OBJECTIVES AND CRITERIA, COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

COMPONENTS 2-4 (ESTUARINE BIOTIC RESPONE, BIOTIC INDEX); AND CURRENT (2010-2011) EUTROPHICATION ASSESSMENT, RUTGERS UNIVERSITY.

The data generated during Components 2-4 of the project will be evaluated at several junctures along their pathway from source to final incorporation into the official database.

The first and a very critical level of data review, validation and verification of data will be conducted when the raw data from the field or laboratory are reviewed while being entered into the database. If the laboratory has adhered to performance-based QA/QC requirements prescribed for their activity during the analytical phase, the submitted data should be in a reasonably sound condition. Data packages received will first be reviewed by the Project QA Manager for basic completeness and content (i.e., are these the data requested and are they expressed in appropriate units and format?). The overall data quality of each data set will then be evaluated in terms of accuracy and precision (when applicable) using the quality criteria described in this QAPP (see Section B5). These data reviews may be conducted by either the Project QA Manager or other qualified personnel (e.g., Project Manager or persons with specific expertise).

Data sets that meet the prescribed quality criteria will be accepted without further qualification for use in making environmental assessments. Data that do not meet all of the project acceptability goals because of minor deficiencies will be assigned data qualifier codes to "flag" the values in question, and they may still be included in the data set as estimates. This will enable individual data users to decide for themselves whether the data are acceptable for their specific purposes. Flagged data are the responsibility of the contractor. NEIWPCC and EPA will not participate in this task. However, flagged data may be reviewed by the NEIWPCC and US EPA management on a case-by-case basis to determine if the data are acceptable for making environmental assessments of the estuarine resource on regional or national levels. Data that consistently fail one or more quality criteria by a significant margin will be rejected and not used for NEIWPCC and US EPA assessments.

D2. VERIFICATION AND VALIDATION METHODS

COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

Processes for data verification and validation, and parties responsible for verifying and validating different components of the project data/information are described in A7. QUALITY OBJECTIVES AND CRITERIA, COMPONENT 1 (QUANTIFYING NUTRIENT LOADING, USGS)

COMPONENTS 2-4 (ESTUARINE BIOTIC RESPONE, BIOTIC INDEX AND CURRENT (2010-2011) EUTROPHICATION ASSESSMENT, RUTGERS UNIVERSITY)

The data generated during Components 2-4 of the project will be systematically reviewed with varying levels of scrutiny at several junctures along the path from time of collection to final reporting; from quick, on-the-spot screening to in-depth evaluation against established criteria or standards. For much of the field collected data, the first level of validation, a cursory screening, will occur as data are recorded; persons conducting and documenting real-time observations should be aware of the range that constitutes realistic values for a specific measure. Certainly a water temperature of 40° C should jump out as an obvious outlier and trigger an immediate response to find the source of the error. With other types of data, the initial validation may not occur in such an immediate time frame; for example, in the case of chlorophyll analysis, the analyst may first need to run several calculations to arrive at a meaningful result. Nonetheless, most data are amenable to some form of quick screening soon after being generated and the responsibility for this first- cut validation falls on the personnel performing the measurement. In addition, most laboratory analyses of the project samples will be monitored by a series of in-stream QC checks that indicate the general level of data quality for a given batch of samples. If routine screens and QC checks are adhered to and proper corrective measures enacted, there is little reason for seriously flawed data to be make it any further down the data stream. However, that assumption cannot be totally relied upon, so additional, documented verifications are required to determine if data quality remains at a level acceptable for the program. The following sections outline the format and procedures to be used for evaluating and documenting data quality for the project and discuss how issues will be resolved when they occur.

Field-Collected Data

Project field crews have the option to record field data on hardcopy data sheets or use the field computer system to directly enter the information, or a combination of both. The field computer system has a separate page for each of the primary activities conducted during the field sampling (e.g., Station Data, Water Quality Data, and Sediment Data). The pages from the computer system generically resemble hardcopy data sheets used for previous EMAP studies. The system queries the crew for specific information relevant to a sampling activity in a manner that systematically leads them through the preferred sequence of steps for collecting the field information. Regardless of the mode used to initially record data, all field data will be entered into the field computer system soon after collection (within the week is recommended).

Validation of Field Data

In the context of this document, the definition of "data validation" can be expressed as a series of questions: Are the data received actually the data expected? Are the data expressed in correct units? Are the data realistic? Are the data complete? In other words, "I was expecting one dozen oranges; did I get one dozen oranges?"

As mentioned, first-cut validation of field data occurs as the data are being collected by the field crews (e.g., are these data in the ballpark?). If the field personnel encounter situations where they question the validity of data they are collecting, they should immediately attempt to isolate and resolve the problem; if they are unable to do so, then they should describe the situation in writing on the appropriate data sheet, and, as soon as possible, consult with their respective senior Field Manager or Project Manager for corrective actions.

The next level of validation takes place as the Project Manager consolidates and formats the field data. Most of the field crew will use hardcopy data sheets to record the bulk of field data; therefore, the data must be transcribed into the field computer system. As soon as possible, upon return from the field, all raw data forms should be photocopied and the originals then placed in a secure file; the copies can then be used for entering the data. During the data entry process, the field data will be screened for missing or errant information. Any observed deficits should be notated in a bound logbook. If corrective actions are initiated (e.g., correcting a spelling error on the copied data form), the correction must be legible and the person who made the correction must document the alteration with their initial and date; a description of the correction should be noted in the bound log.

Verification of Field Data

Where "data validation" is a determination that the collected data appear appropriate and are expressed in the correct format, "data verification" is more of a process to evaluate the level of data quality (e.g., representativeness, accuracy and precision). Verification of field data involves a more critical review of QC elements or acceptance criteria such as calibration success for hydrographic equipment, acceptability of sediment grabs, siting of a station, etc. These types of evaluations can be and should be executed at each stage of the process from data collection to final review prior to data being posted to the public. However, there must be several structured check points where documented verifications are performed.

Transcription Errors

One of the first reviews field data are subjected to is an evaluation of the relative frequency of transcription errors enacted going from hardcopy into the electronic format. To determine this, a randomly selected subset of at least 10% of the station packages (the entire set of field data sheets submitted for a given station) will be pulled and the data (primarily, measurements or numerical values) manually compared against the electronic version on a field - by-field basis. Any errors will be listed in the bound logbook (see above, Field Data Validation) and a final tally derived for the station. The total number of transcription errors for a complete set of data sheets should not exceed 5. If more than 5 transcription errors are found, the entire set of field data sheets will be pulled and re-examined for review and check for errors.

Verification of Field Measurements

Measurements of water quality parameters taken directly in the field will be evaluated for accuracy by verifying the results of calibration and QC checks. These checks should be performed by the field crews on a daily basis and if the instruments are out of tolerance, they should be re-calibrated. Calibration and post-calibration of instruments will occur at the beginning/end of field sampling days. At the conclusion of the summer sampling, copies of the field records for calibration and QC checks will be provided to the Project QA Manager and Project Manager for further review. Any data that was collected, when the instruments were out of compliance, will be flagged with a qualifier code.

Other field collected data that will be evaluated on a randomly selected subset of the field data include penetration depth for benthic grabs, light-down/light-up comparisons, trawl times, and difference (distance) between intended site and actual site. These evaluations will be conducted by the Project QA Manager.

Laboratory -Generated Data

All laboratory data generated for the project will be systematically reviewed and evaluated. Laboratories that perform the analyses will conduct their internal QA/QC verifications prior to submitting the data to the Project Manager. Laboratory data will be submitted in accordance with the Standardized Data Transfer Protocols (STDP) specified in SCCWRP, 2000; the STDP stipulate that data be submitted in comma-delimited, ASCII format. The following discussion on data flow and verification is taken from the Section 1II (Roles and Responsibilities) of the above 1M plan.

Upon receipt of a data set, a temporary file will be created and a series of error checks will be performed to ensure the data: 1) are within specified ranges appropriate to each parameter measured, 2) contain all required fields, 3) have encoded valid values from constrained look-up lists where specified, and 4) are in the correct format (text in text fields and values in numeric fields, etc.).

If the data emerge from the error check routine with no errors or suspected outliers, the temporary table for that data type will be appended. If there are only a few, easily correctable errors, the changes will be made, with the consent of the submitting agency. If there are numerous errors or the corrections are difficult to implement, the data will be sent back to the submitting agency with a list of necessary corrections. The submitting agency will make the corrections and resubmit the file within one week to the Project Manager, who will subject the file to error checking again. Each of these paths will be documented as part of the submittal tracking process.

When all data for received for a particular laboratory function have been submitted, error checked, and corrected, the Project QA Manager will certify that the file is consistent with the STDP format and complete.

The Project QA Manager and Project Manager will be responsible for conducting technical reviews of the data before the data are accepted for project assessments; certain aspects of these reviews may be delegated to other staff with final approval through the above quality management personnel. Data quality of a specific data set will be assessed by a critical comparison of the submitted QA/QC results to the quality criteria or standards established by this QAPP for that analysis. If the evaluation indicates that the data, overall, meet the quality standards, with no or only minor deficiencies, then the data set will be acceptable for the project assessments without further qualification. If the data consistently fail one or more quality criteria, then the data set will be flagged with an appropriate data qualifier code. Depending on the degree of the deficiency, the data might still be used in certain project assessments (provided that data clearly carry the appropriate qualifier code), or they may be dropped entirely from the accessible database.

D3. RECONCILIATION WITH USER REQUIREMENTS

In this project the need to reconcile results to the proposed project Data Quality Objectives (DQOs) is not totally germane. The project represents an experimental application that should not be bound by success/failure criteria, but rather an iterative success/revision approach. For these reasons, the project will use Method Quality Objectives (MQOs) to evaluate success on a component level, in addition to project DQOs as criteria for the overall sampling design.

The project management team will be advised on the QC results for the individual monitoring and analytical activities as evaluated against the MQOs or quality goals established in this QAPP. Each activity for which QA/QC guidelines were described should submit a summary of those results along with their analytical results. If the data quality for a particular indicator is substandard, the project management will be charged with the decision to: 1) if consensus agreement is reached that existing criteria are overly stringent, revise the quality criteria to reflect the level of data quality attained and then use the data for environmental assessments; 2) totally reject the use of the data for environmental assessments; or, 3) flag the deficient data with qualifiers and use it conditionally for environmental assessments.

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APPENDICES

Appendix 1. Sensitivity requirements for data sonde measurements in this project.

YSI sensor specifications

Parameter: Temperature
Units: Celsius (C)
Sensor Type: Thermister
Model #: 6560
Range: -5 to 45 °C
Accuracy: +/-0.15 °C
Resolution: 0.01 °C

Parameter: Conductivity
Units: milli-Siemens per cm (mS/cm)
Sensor Type: 4-electrode cell with autoranging
Model #: 6560
Range: 0 to 100 mS/cm
Accuracy: +/-0.5% of reading + 0.001 mS/cm
Resolution: 0.001 mS/cm to 0.1 mS/cm (range dependent)

Parameter: Salinity
Units: parts per thousand (ppt)
Sensor Type: Calculated from conductivity and temperature
Range: 0 to 70 ppt
Accuracy: +/- 1.0% of reading or 0.1 ppt, whichever is greater
Resolution: 0.01 ppt

Parameter: Dissolved Oxygen % saturation
Units: percent air saturation (%)
Sensor Type: Rapid Pulse – Clark type, polarographic
Model #: 6562
Range: 0 to 500 % air saturation
Accuracy: 0-200 % air saturation, +/- 2 % of the reading or 2 % air saturation, whichever is greater; 200-500 % air saturation, +/- 6 % of the reading
Resolution: 0.1 % air saturation

Parameter: Dissolved Oxygen mg/L (Calculated from % air saturation, temperature and salinity)
Units: milligrams per Liter (mg/L)
Sensor Type: Rapid Pulse – Clark type, polarographic
Model #: 6562
Range: 0 to 50 mg/L
Accuracy: 0 to 20 mg/L, +/- 2 % of the reading or 0.2 mg/L, whichever is greater; 20 to 50 mg/L, +/- 6 % of the reading
Resolution: 0.01 mg/L

Parameter: Non-Vented Level – Shallow (Depth)
Units: feet or meters (ft or m)
Sensor Type: Stainless steel strain gauge
Range: 0 to 30 ft (9.1 m)
Accuracy: +/- 0.06 ft (0.018 m)
Resolution: 0.001 ft (0.001 m)

Parameter: pH
Units: units
Sensor Type: Glass combination electrode
Model #: 6561
Range: 0 to 14 units
Accuracy: +/- 0.2 units
Resolution: 0.01 units

Parameter: Turbidity
Units: nephelometric turbidity units (NTU)
Sensor Type: Optical, 90 ° scatter, with mechanical cleaning
Model #: 6136
Range: 0 to 1000 NTU
Accuracy: +/- 5 % reading or 2 NTU (whichever is greater)
Resolution: 0.1 NTU

Appendix 2. SOPs for data sonde measurements and biotic measurements in this project.

Before Leaving the Dock:

1) Refer to "fieldwork packing and preparation" checklist. Check off each item as it is addressed.

Locating the Station:

1) "Tighten" the resolution on console GPS unit and motor over station (into wind/current, whichever is dominant)

2) Drop the quadrat (with float) on mark.

3) Anchor nearby (as close as possible without interfering with diver in water), considering wind and current.

4) Record site location (transect and station) and time (EST) on field sheet.

Once at the Station:

1) Deploy YSI unit- allow to contact bottom, pull up approx 10cm from bottom and lash to cleat, allow to equilibrate for 2-3 minutes, record physico-chemical parameters on field sheet (refer to field sheet).

2) Obtain depth and Secchi values.

3) Diver observes quadrat, reporting percent coverage and other observations (refer to field sheet).

4) Diver takes photo(s) of area inside quadrat.

5) Diver obtains a sample of representative *Zostera* from inside quadrat and delivers to boat. Technician on board measures 5 randomly-selected seagrass blades.

6) Diver obtains core from center of quadrat and transports it back to the boat. Technician on board sieves sample, bags with appropriate label, and places in cooler.

7) A second sample is taken for epiphyte work by hand (no core). This second sample is treated like the primary biomass sample and bagged separately (labels: black text on white background= original 80 stations, white text on black background= additional 40).

8) If macroalgal presence is considered "heavy" (suggesting a bloom) a representative sample is obtained by the diver and delivered to the boat, where it will be preserved in 1 part formaldehyde to 9 parts ambient water. A label will be placed in the bottle as well as written on the outside of the bottle.

9) Diver sticks vertical PVC pipe in center of quadrat. Diver obtains actual GPS location of station with handheld unit or long antenna back to boat. If depth and current are an issue, hold position of boat close and use #s from console GPS.

10) Supervisor/delegated technician reviews field sheet and initials it if complete and approved.

11) Supervisor/delegated technician takes a photo of the field sheet.

12) Team retrieves quadrat/float and vertical PVC and moves on to next station.

Upon return:

Field Sheets:

1) Field sheets are reviewed to verify that all stations and samples are accounted.

2) Sheets are photocopied and placed in designated folder/binder and placed in an approved location at RUMFS

3) Photocopied sheets are relocated to a secondary secure location (JCNERR coastal center) and placed in the binder there.

Biomass Samples:

1) Biomass samples are cross-checked with field sheets to assure all samples are accounted for. The supervisor or senior technician should initial on a tracking sheet that all samples are accounted for. If any samples are missing, the site should be revisited at the next possible opportunity to obtain a replacement sample.

2) Biomass samples from the day are placed in a larger, single bag (small or medium wastepaper basket bag) and placed in the "coffin freezer" in the utility room in second building ("The Dark side") at RUMFS.

    - Alternatively, biomass samples can be placed in a Sterilite container and placed in the JCNERR's standing freezer in the Nutrient Laboratory. If so, the external digital thermometer should be checked daily to ensure the freezer is maintaining a temperature between -10 and -20 degrees C.

3) Epiphyte samples should be kept on ice overnight and processed the next day. If unable to process the following day, samples should be frozen along with the biomass samples.

YSI/CTD:

1) A post-calibration check of the YSI unit used to obtain physical parameters in the field should be performed to verify instrument/probe function and accuracy.

2) The YSI600 unit should be rinsed with fresh water and battery voltage assessed. If necessary, change batteries before next field day. The 650MDS display should be wiped down with a damp cloth and dried.

Other:

1) A copy of vessel float plans, detailing those on-board during each field sampling effort, should be retained for future reference.

2) The field sheets should be transcribed into a digital spreadsheet as soon as possible.

Appendix 3. List of field equipment, instruments, and supplies used in this project.

_____ Handheld YSI

_____ Spare batteries for handheld YSI

_____ Camera

_____ Spare batteries for camera

_____ Spare memory cards for camera

_____ Second (spare) camera with extra batteries

_____ Sample bags (enough+ excess for the planned day) for biomass

_____ Biomass labels for the period

_____ Sample bags (enough+ excess for the planned day) for epiphytes

_____ Epiphyte labels for the period

_____ Field sheets (enough+ excess for the planned day)

_____ Writing instruments (pencils, pens, permanent marker)

_____ Scissors (for cutting up labels)

_____ Metric ruler(s)

_____ Corer

_____ Depth stick and Secchi

_____ Sieve and/or mesh bags

_____ Translucent bottles for macroalgal samples

_____ Formaldehyde for macroalgal samples

_____ GPS and power cable

_____ GPS antenna on pole

_____ Spare GPS unit

_____ Coolers with ice

_____ Black straps for coolers

_____ Orange boat box

_____ Wetsuit

_____ Mask and snorkel

_____ VHF radio

_____ Blue and tan field box

_____ Sufficient lifejackets for all on board

Appendix 4. Protocols for field collection and laboratory analysis of brown tide samples for the Barnegat Bay-Little Egg Harbor Estuary.

These protocols describe water sample collection, filtration, pigment extraction, and quantification for brown tide enumeration in the Barnegat Bay-Little Egg Harbor Estuary. The brown tide alga *Aureococcus anophagefferens*' marker pigment 19-butanoyloxyfucoxanthin (19'-bf) is used for cell number estimation. Further using the ratio of 19'-bf to chlorophyll *a* (chl *a*), the contribution of brown tide alga to chlorophyll *a* will also be available.

Protocols

1. **Water collection.** Collect water samples at 0.5 – 1 m below the surface in clean polyethylene bottles (clean, acid free), triplicate 250 mL samples for each sampling station. Water samples can be temporarily stored in a cooler with ice while transporting.

2. **Water filtration.** Gently swirl the water sample to ensure homogeneity before pouring. The particulates are collected on 25 mm GF/F filter under low vacuum (<20 kPa) till the flow is significantly slowed (try not to allow the filtration to be more than 20 min, ideally no more than 10 min), record volume of water filtered (the water volume can be measured by weighing water sample weight loss before and after filtration, and converted to volume using water density by record the weight of 1 mL of water sample). Release vacuum in a timely manner so as not to suck the filter dry. The filtration and follow up procedure should be in dim light.

3. **Filter/particulate matter preservation.** Use a pair of tweezers, remove the filter from filtration manifold, lightly pad on a folded Kleen Wiper once to remove extra water, gently fold the filter in half (particulate matter inside the fold). The filter can then be wrapped in aluminum foil and/or suitable cryogenic storage vials, which are properly labeled (e.g. operator initial_ date_sample name_replicate number_volume), and stored cryogenically (liquid nitrogen) for later analysis.

4. **Pigment extraction**. Retrieve the filter from liquid nitrogen, submerge the filter in 1.5 mL 90% acetone (HPLC grade) in an amber microcentrifuge tube. The pigments are extracted upon sonication in ice-cold water bath (Fischer Scientific Ultrasonic FS-28) for 30 min. The extract is then vortex mixed, centrifuged, and the supernatant passed through a 0.45 m PTFE syringe filter (0.45 µm pore size, 4 mm diameter, Acrodisc) to get the particle free extract.
5. **HPLC analysis**. The HPLC system consists of Waters 2690 Separation Module, refrigerated auto sampler, column heater, Waters 486 Absorbance Detector set at 450 nm, and SRI Model 333 USB Chromatography Data System for signal conversion and integration. Pigments are separated on an Eclipse XDB C8 (150 x 4.6 mm, 3.5 µm, Agilent) using 100% methanol (J.T. Baker, HPLC grade) and methanol:28 mM aqueous TBAA (Tetra-butyl ammonium acetate, pH 6.5, made by titrating tetra-butyl ammonium hydroxide with acetic acid, Acros Organics, HPLC grade) at 70:30 v:v as mobile phases, following (Van Heukelem and Thomas 2001). The peak area of Chlorophyll a, and

diagnostic pigments representing brown tide alga (19'-butanoyloxyfucoxanthin, i.e. 19'-bf) are recorded and concentrations calculated from standard curve.

HPLC standard curve of 19'-bf and chlorophyll *a* (Chl *a*). The data are obtained by processing lab cultured exponential phase *Aureococcus anophagefferens*, followed by pigment extraction, and HPLC analysis. The standard curve establishes the correlation between 19'-bf peak area with *A. anophagefferens* cell density measured by Coulter Counter (Beckmann Coulter Multisizer 3). A standard curve of Chl *a* will be established from pure Chl a standard available in Sigma Aldrich, or less accurately, by using UV-Vis spectrophotometer (Agilent 8453) following Jeffery and Humphrey's trichromatic method (EPA method 446.0), and correlating the result from spectrophotometer to peak area in HPLC analysis.

Confirmation with standard addition. While the concentration measurement is mostly based on external standard curve as shown above, standard addition of *A. anophagefferens* pigment extraction will be used to confirm peak identification and concentration.

Data report. The results of *A. anophagefferens* cell density will be reported by taking the average and standard deviation of the triplicate field samples for each sampling station. The contribution of *A. anophagefferens* to the total chl *a* will also be available.

Reference

Van Heukelem, L. and C. S. Thomas (2001). "Computer-assisted high-performance liquid chromatography method development with applications to the isolation and analysis of phytoplankton pigments." Journal of Chromatography A 910(1): 31-49.

Appendix 1 - 1 Full Report from USGS for Component 1

THIS IS A PLACEHOLDER SPOT FOR THE USGS REPORT.

Appendix 2 - 1 Full report comparing remote sensing of seagrass to in sity monitoring.

Comparison of Remotely Sensed Surveys vs. *In Situ* Plot-based Assessments of Seagrass Condition in Barnegat Bay- Little Egg Harbor

Richard G. Lathrop Jr., Dan Merchant, Scott Haag
November 24, 2012

Abstract

The Barnegat Bay-Little Egg Harbor (BB-LEH) estuarine system located along the eastern shoreline of Ocean County, New Jersey contains ~ 75% of New Jersey's known seagrass habitat (Lathrop et al. 2001). Eelgrass (*Zostera marina*) is the dominant species while widgeongrass (*Ruppia maritima*) is also common in lower salinity and shallow regions of the BB-LEH. An estuary wide survey was conducted in the summer of 2009 to measure the current extant of seagrass habitat across the BB-LEH system (Lathrop and Haag, 2011). Aerial imagery collected during the months of July and August 2009 was interpreted and mapped using an object oriented image analysis techniques, similar to techniques used in the 2003 mapping survey. A boat-based *in situ* dataset was collected concurrently with the aerial photography to assist the image interpretation and for an independent accuracy assessment. We compared the remotely sensed mapping of seagrass cover change (in 2003 vs. 2009) vs. the *in situ* plot-based sampling conducted by Kennish et al. from 2004 through 2010. Comparison of the remotely sensed vs. the *in situ* plot change analysis suggests that the two methodologies had broadly similarly results with the percent area showing declines in percent cover was greater than those that exhibited increases. In conclusion, the two studies provide corroborating evidence that seagrass has declined in percent cover in the BB-LEH system during the decade of the 2000's.

Background on 2003 & 2009 Remotely Sensed Surveys

Results of this earlier work indicate that the overall amount of seagrass beds were similar in 2009 as compared to 2003 (5,122 ha in 2003 vs. 5,260 ha in 2009) (Lathrop and Haag, 2011). Differences in the seasonal period of image acquisition account for some of the differences in the mapped area and type of seagrass. Imagery for the 2003 survey was acquired early in the growing season (May 4-5th) while the 2009 survey was acquired on June 28th, July 7th, and August 4th. Examination of the more detailed four class seagrass cover map shows a decline in the area of dense (80-100% cover) seagrass beds in 2009 vs. 2003 (471 ha in 2009 vs. 2,074 ha in 2003; a nearly 60% decline). The loss in dense beds translated to an increase in medium (40-80% cover) density meadows (1,093 ha in 2003 vs. 2,523 ha in 2009; an increase of 130%). The extent to which this apparent thinning in the density of the seagrass meadows is real or an artifact of the poorer image quality in the 2009 imagery and the resulting lower accuracy in mapping dense seagrass meadows is less certain. Comparison of the classified seagrass presence/absence map and the *in situ* validation dataset showed an overall thematic map accuracy of 87% while the four class seagrass density map (absent, sparse, moderate, dense seagrass cover) has

an overall accuracy of 70%. For details on the methodology employed in the remotely sensed survey, refer to (Lathrop and Haag, 2011).

Methods

Several steps were taken in order to compare the results of the two different studies. The remotely sensed surveys mapped seagrass percent cover into the following four categories: Dense (80-100%), Medium (40-80%), Sparse (10-40%) and No Seagrass (<10%). The 2004-2010 *in situ* plot data were reclassified using the same classification scheme. To help visualize the changes in sea grass cover between the 2003 and 2009 remotely sensed surveys, we classed the mapped data into seven change categories (Table 1). The in situ plot data (sample size of 107 plots; dated provided by M. Kennish and B. Fertig) were similarly classed into seven change categories using the same scheme as displayed in Table 1. The *in situ* plot data were geo-located and the corresponding remotely sensed 2003-2009 change category extracted for comparison purposes.

Three sub-areas were delineated to examine more closely the correspondence between the spatial distribution of changes in seagrass cover as revealed in the remotely sensed survey vs. the *in situ* plot-based data. The three sub-areas are denoted as Northern Barnegat Bay, Barnegat Inlet and Little Egg Harbor.

Results and Discussion

Comparison of the remotely sensed vs. the *in situ* plot change analysis suggests that the two methodologies had broadly similarly results with the percent area showing declines in percent cover was greater than those that exhibited increases (Table 2; Figure 1). The remotely sensed change analysis showed approximately 37% of the seagrass area with some increase in density vs. 47% that exhibited decline. The *in situ* plot-based change analysis showed approximately 23% of the seagrass area with some increase in density vs. 51% that exhibited decline. Site-level comparison between the mapped vs. plot results shows a low degree of correspondence with only 17% of the plot results matching the mapped results (Table 3). This low level of correspondence is not unexpected given the difference in scales with the minimum mapping unit for the remotely sensed survey at 500 m^2 (0.05 ha) vs. the 1 m^2 size for the *in situ* plots. Relaxing the definition of correctness to include a more "fuzzy similarity" (i.e., exact match as well as within one class difference) showed greater degree of correspondence with approximately 60% of the plots within one class from the remotely sensed data (Table 3).

The remotely sensed 2003-2009 change map for the entire BB-LEH study area displays a rather complex pattern of loss and expansion in seagrass percent cover (Figure 2). There appears to have been an increase in the percent cover density of seagrass meadows in the north-central portions of Barnegat Bay (Figures 2 and 3), while seagrass meadows in the Barnegat Inlet area (Figures 2 and 4) and in Little Egg Harbor (Figures 2 and 5) have experienced declines in percent cover. The causal factors controlling these declines are unclear; while decreased water transparency due to eutrophication is undoubtedly

important, especially in Little Egg Harbor (Figure 4), close examination of the remotely sensed imagery acquired over the past decade suggests that many areas mapped as declining seagrass cover in the Barnegat Inlet sub-area (Figure 5) are likely due to shifting sand in this geomorphologically dynamic tidal delta area.

In conclusion, the two studies provide corroborating evidence that seagrass has declined in percent cover in the BB-LEH system during the decade of the 2000's.

References

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

Lathrop, R.G. and S. M Haag. 2011. Assessment of Seagrass Status in the Barnegat Bay – Little Egg Harbor Estuary System: 2003 and 2009. Center for Remote Sensing & Spatial Analysis, Rutgers University. 56 p. http://crssa.rutgers.edu/projects/coastal/sav/downloads/CRSSAreport2011-01_Assessment_Seagrass_in_BBAY_LEH_2003_and_2009.pdf

Table 1. Seagrass change matrix showing seven categories of percent cover change categories between 2003 and 2009. Note: that the original percent cover categories were: Dense (80-100%), Medium (40-80%), Sparse (10-40%) and No Seagrass (<10%).

2003 Classification \ 2009 Classification	None	Sparse	Medium	Dense
None	No Change	Minimal Increase	Moderate Increase	Major Increase
Sparse	Minimal Decline	No Change	Minimal Increase	Moderate Increase
Medium	Moderate Decline	Minimal Decline	No Change	Minimal Increase
Dense	Major Decline	Moderate Decline	Minimal Decline	No Change

Table 2. Comparison of remotely sensed vs. the *in situ* plot change analysis.

	Remotely Sensed Survey 2003-2009	In Situ plots 2004-2010
Major Increase	1	1
Moderate Increase	10	11
Minimal Increase	26	11
No Change	18	26
Minimal Decline	28	35
Moderate Decline	11	12
Major Decline	8	4

Table 3. Site-level comparison between the mapped vs. plot change classification results.

Map Classification	Plot Classification: Major Expansion	Moderate Expansion	Minimal Expansion	No Change	Minimal Loss	Moderate Loss	Major Loss	Total
Major Expansion		1						1
Moderate Expansion	1	1		2	1	1		6
Minimal Expansion		2		7	2	1		12
No Change		3	6	1	10	3		23
Minimal Loss		2	4	8	10	2	1	27
Moderate Loss		3	2	1	6	4	1	17
Major Loss				9	8	2	2	21
Total	1	12	12	28	37	13	4	

Figure 1. Remotely sensed (2003-2009) vs. in situ plot (2004-2010) change results.

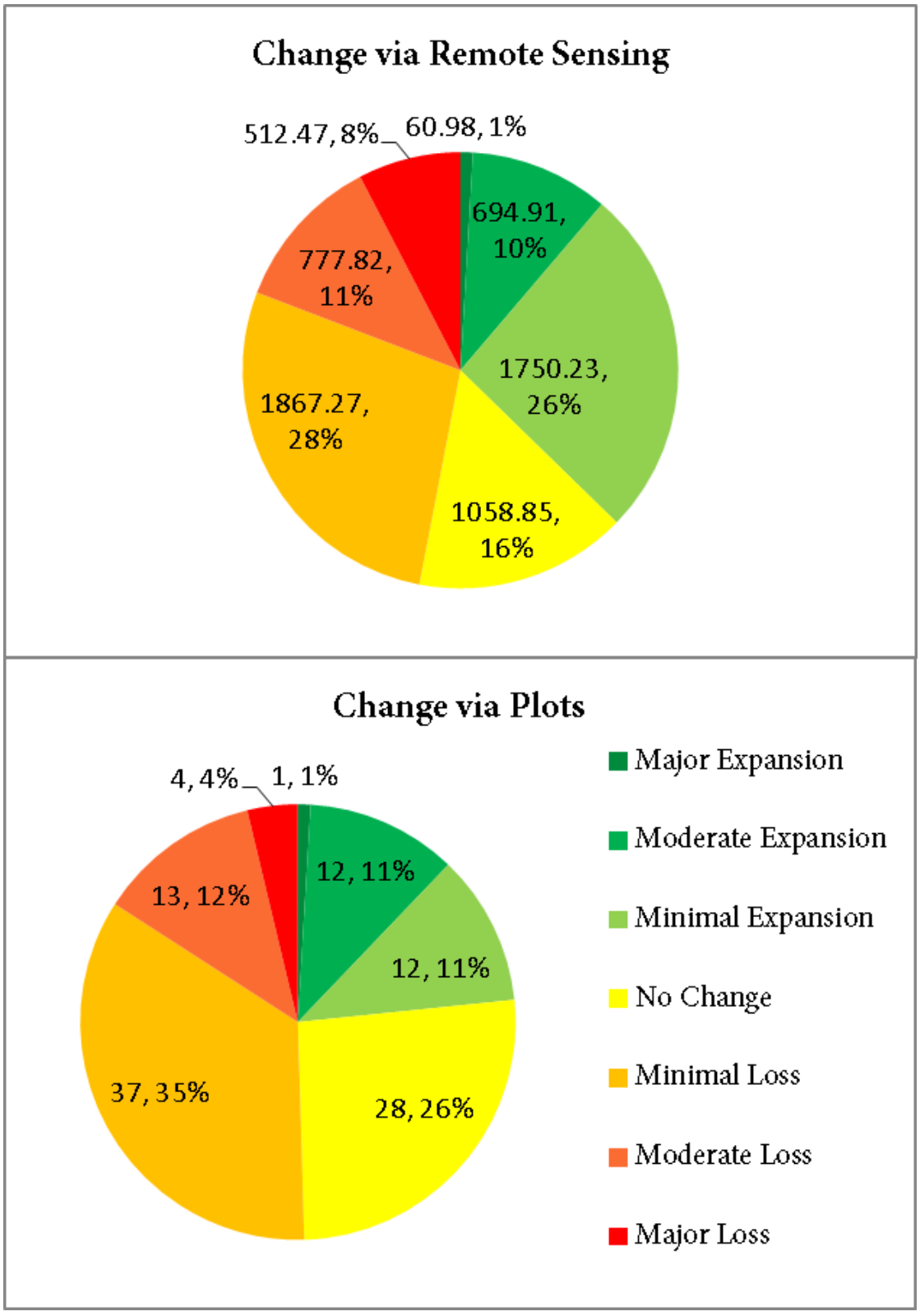


Figure 2. Seagrass 2003-2009 remotely sensed change map for entire BB-LEH study area.

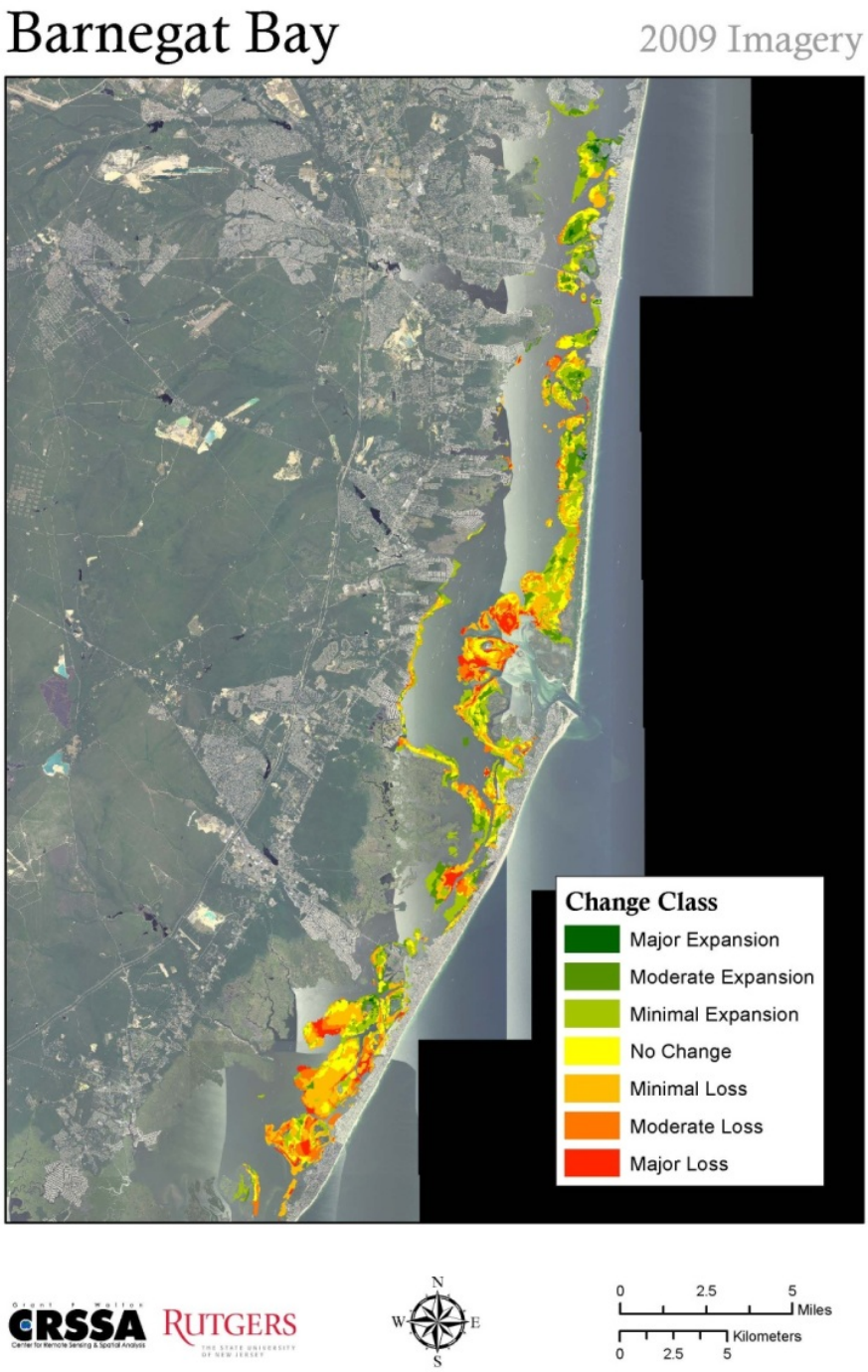


Figure 3. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Northern Barnegat Bay study sub-area.

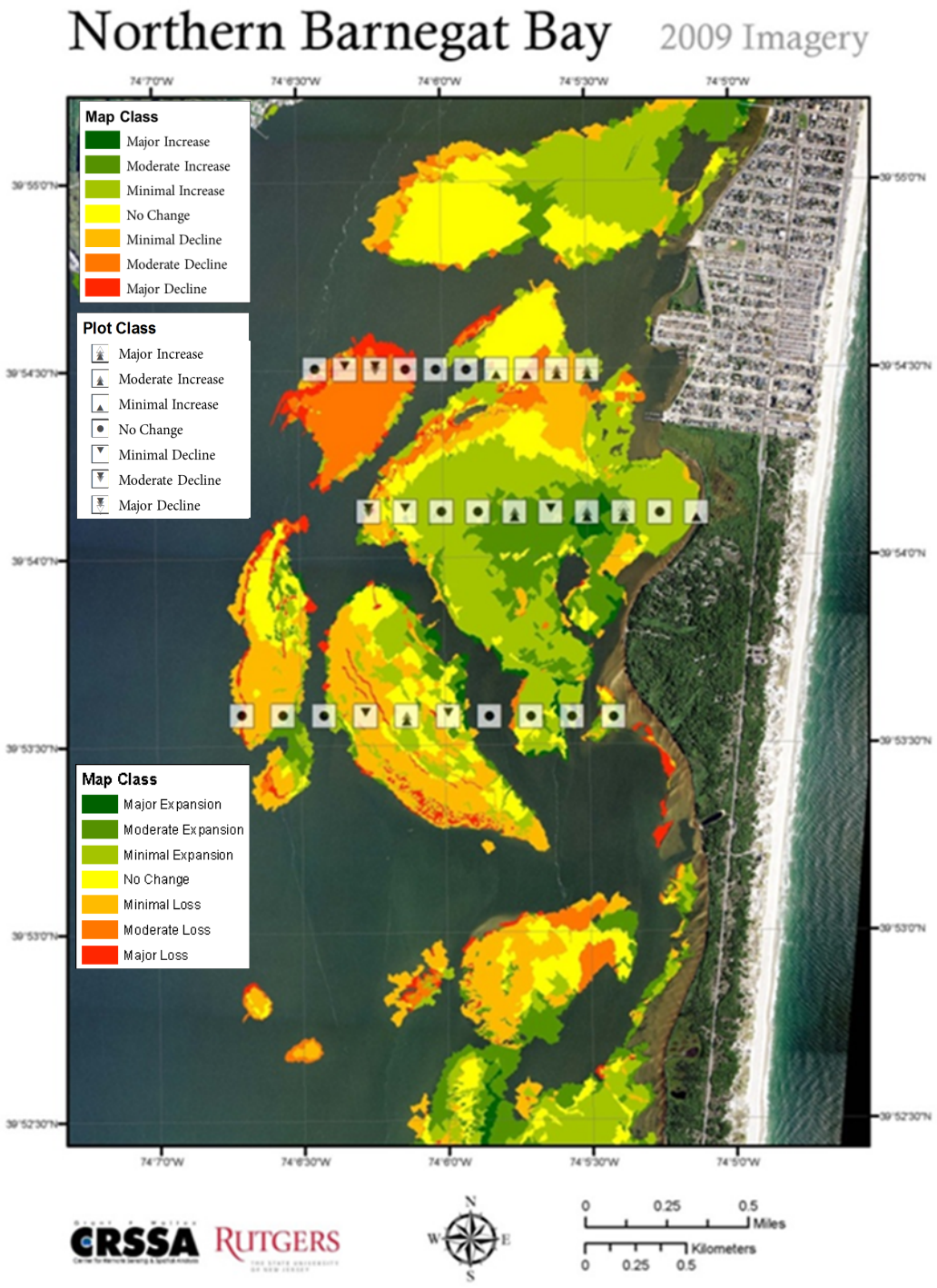

Northern Barnegat Bay
2009 Imagery
74°7'0"W
74°6'30"W
74°6'0"W
74°5'30"W
74°5'0"W
39°55'0"N
39°54'30"N
39°54'0"N
39°53'30"N
39°53'0"N
39°52'30"N
Map Class
Major Increase
Moderate Increase
Minimal Increase
No Change
Minimal Decline
Moderate Decline
Major Decline
Plot Class
Major Increase
Moderate Increase
Minimal Increase
No Change
Minimal Decline
Moderate Decline
Major Decline
Map Class
Major Expansion
Moderate Expansion
Minimal Expansion
No Change
Minimal Loss
Moderate Loss
Major Loss
CRSSA
RUTGERS
N
W
E
S
0
0.25
0.5
Miles
Kilometers
0
0.25
0.5

Figure 4. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Barnegat Inlet study sub-area.

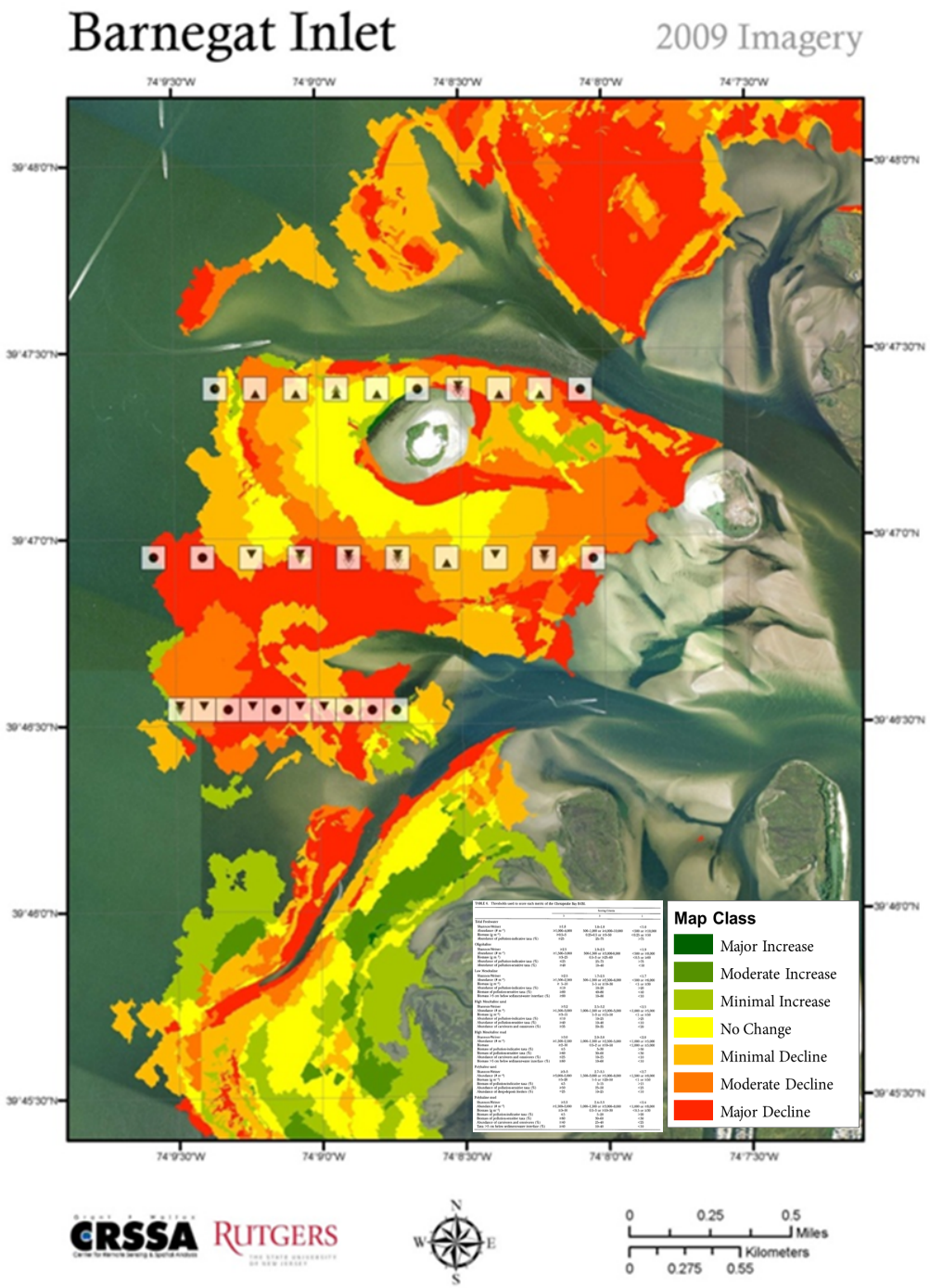
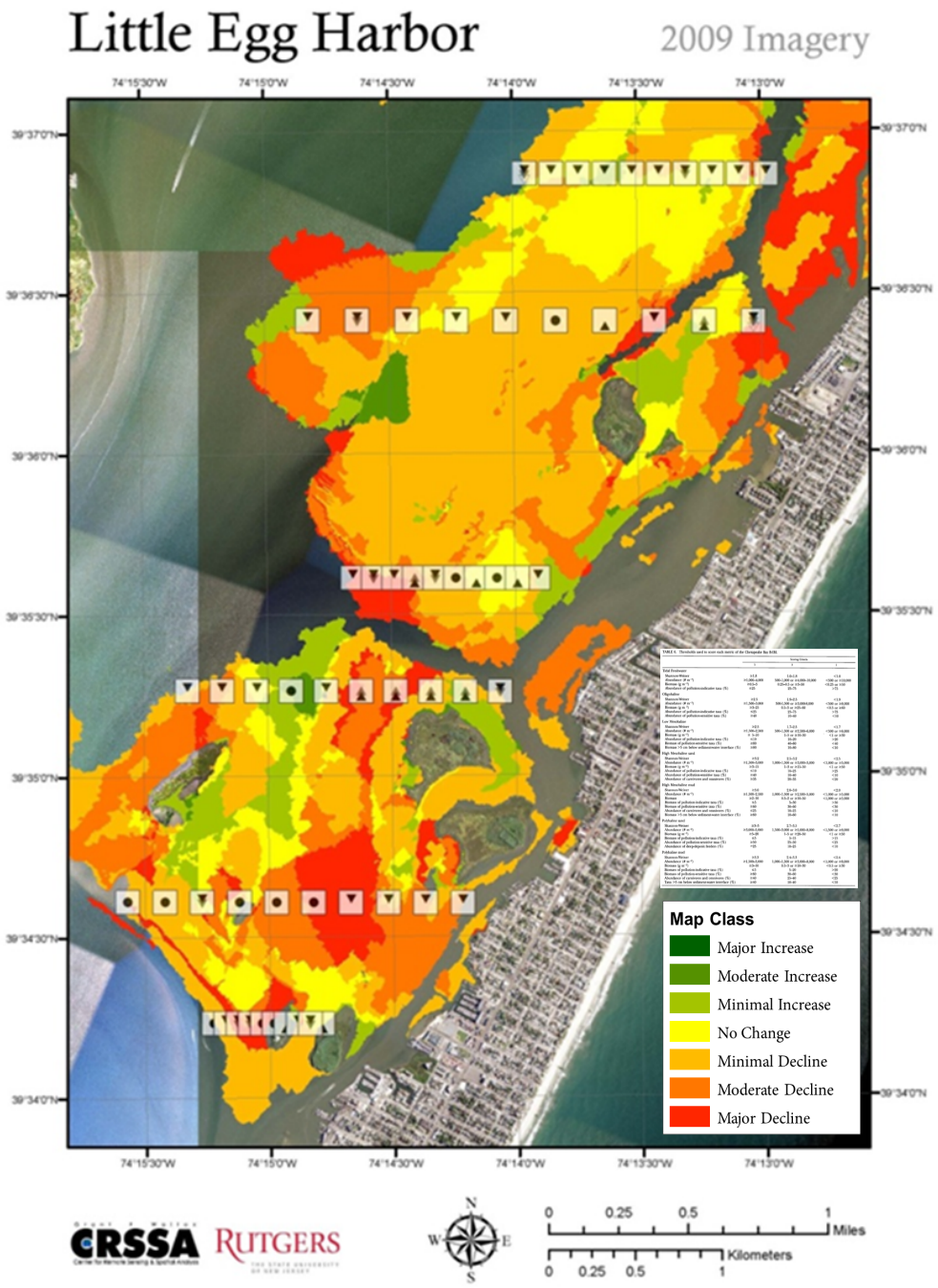


Figure 5. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Little Egg Harbor study sub-area.



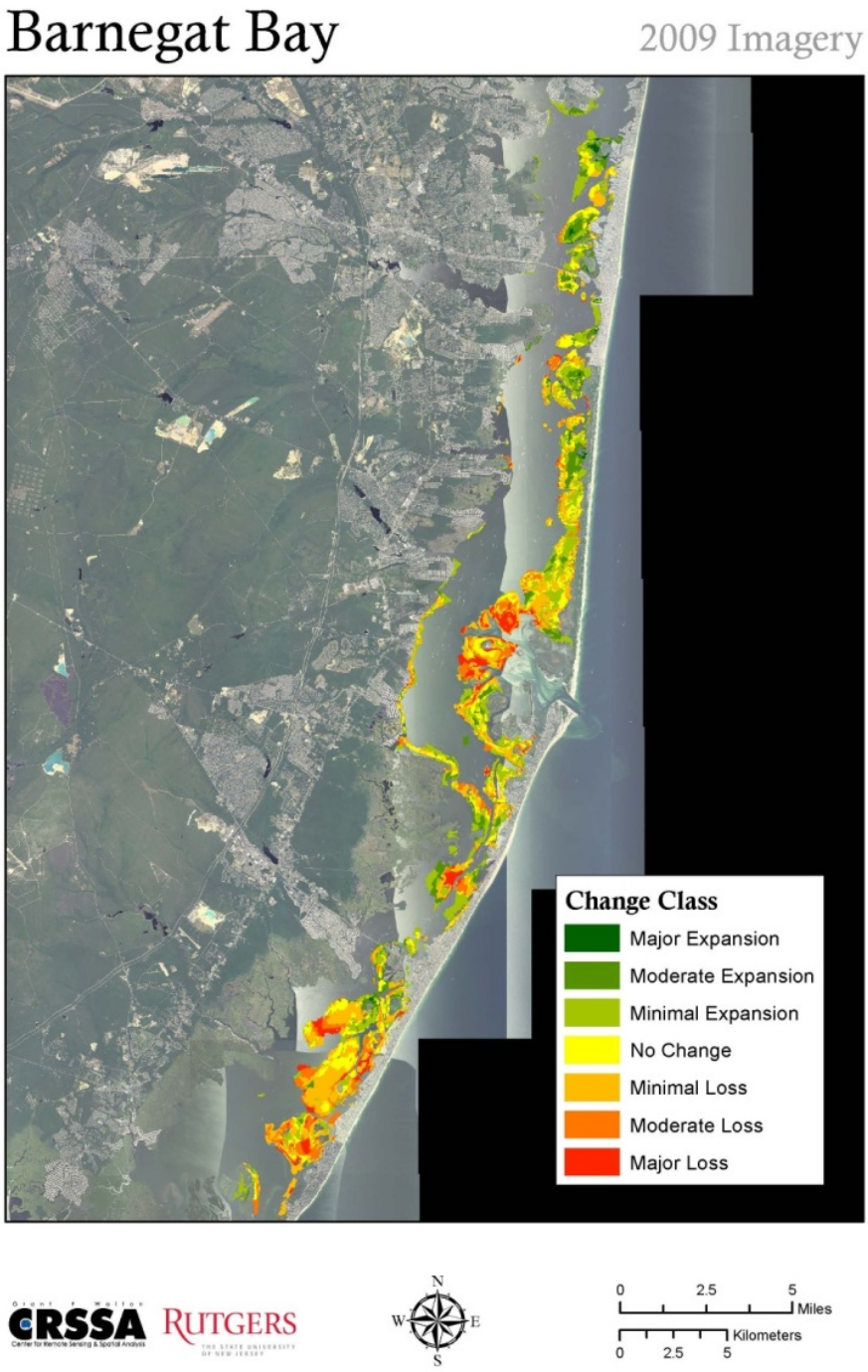


Figure 3. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Northern Barnegat Bay study sub-area.

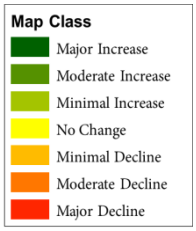


TABLE 6. Thresholds used to score each metric of the Chesapeake Bay IBI.

	Scoring Criteria		
	1	3	5
Total Freshwater			
Shannon-Wiener	0-1.0	1.01-2.0	>2.0
Abundance (# per ?)	<1,000-4,000	500-2,000 or >4,000-10,000	<500 or >10,000
Biomass (g or ?)	<0.25	0.25-1.00	>1.00
Abundance of pollution-indicator taxa (%)	>60	20-60	<20
Oligohaline			
Shannon-Wiener	<0.5	0.5-0.25	>0.25
Abundance (# per ?)	<1,500-2,000	500-1,500 or >2,000-8,000	<500 or >8,000
Biomass (g or ?)	<0.01	0.01 or 0.25-0.40	<0.5 or >0.400
Abundance of pollution-indicator taxa (%)	>60	20-60	<20
Abundance of pollution-sensitive taxa (%)	>40	10-40	<10
Low Mesohaline			
Shannon-Wiener	<0.5	1.5-0.5	>1.5
Abundance (# per ?)	<1,500-2,000	500-1,500 or >2,000-8,000	<500 or >8,000
Biomass (g or ?)	<0.25	0.25 or >2.00-3.00	<0.5 or >3.00
Abundance of pollution-indicator taxa (%)	>10	10-40	<10
Abundance of pollution-sensitive taxa (%)	>40	10-40	<10
Biomass >5 cm below sediment/water interface (%)	<80	20-80	>20
High Mesohaline sand			
Shannon-Wiener	>0.5	0.5-0.0	<0.5
Abundance (# per ?)	>1,500-3,000	1,000-1,500 or >3,000-8,000	>1,000 or >8,000
Biomass (g or ?)	>0.25	0.25 or >2.5-20	<1.1 or >100
Abundance of pollution-intolerant taxa (%)	>60	20-60	<20
Abundance of pollution-sensitive taxa (%)	>40	10-40	<10
Abundance of carnivores and omnivores (%)	>40	20-40	<20
High Mesohaline mud			
Shannon-Wiener	>1.0	2.0-1.0	<2.0
Abundance (# per ?)	>1,000-3,000	1,000-1,500 or >3,000-5,000	>1,000 or >5,000
Biomass (g or ?)	>0.25	0.25 or >2.5-20	<1.1 or >100
Abundance of pollution-indicator taxa (%)	<20	20-60	>60
Abundance of pollution-sensitive taxa (%)	>40	10-40	<10
Abundance of carnivores and omnivores (%)	>20	20-40	<20
Taxa >5 cm below sediment/water interface (%)	>80	20-80	<20
Polyhaline sand			
Shannon-Wiener	>0.5	2.5-0.5	<2.5
Abundance (# per ?)	>10,000-20,000	1,500-10,000 or >20,000-40,000	>1,500 or >40,000
Biomass (g or ?)	>0.25	0.25 or >20-40	<1.1 or >100
Abundance of pollution-indicator taxa (%)	<5	5-20	>20
Abundance of pollution-sensitive taxa (%)	>40	10-40	<10
Abundance of deep-deposit feeders (%)	>20	20-40	<20
Polyhaline mud			
Shannon-Wiener	>0.5	0.4-0.5	<0.4
Abundance (# per ?)	>1,500-2,000	1,000-1,500 or >2,000-8,000	>1,000 or >8,000
Biomass (g or ?)	>0.25	0.25 or >2.5-20	<1.1 or >100
Abundance of pollution-indicator taxa (%)	<5	5-20	>20
Abundance of pollution-sensitive taxa (%)	>40	10-40	<10
Abundance of carnivores and omnivores (%)	>40	20-40	<20
Taxa >5 cm below sediment/water interface (%)	>80	20-80	<20

Figure 4. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Barnegat Inlet study sub-area.

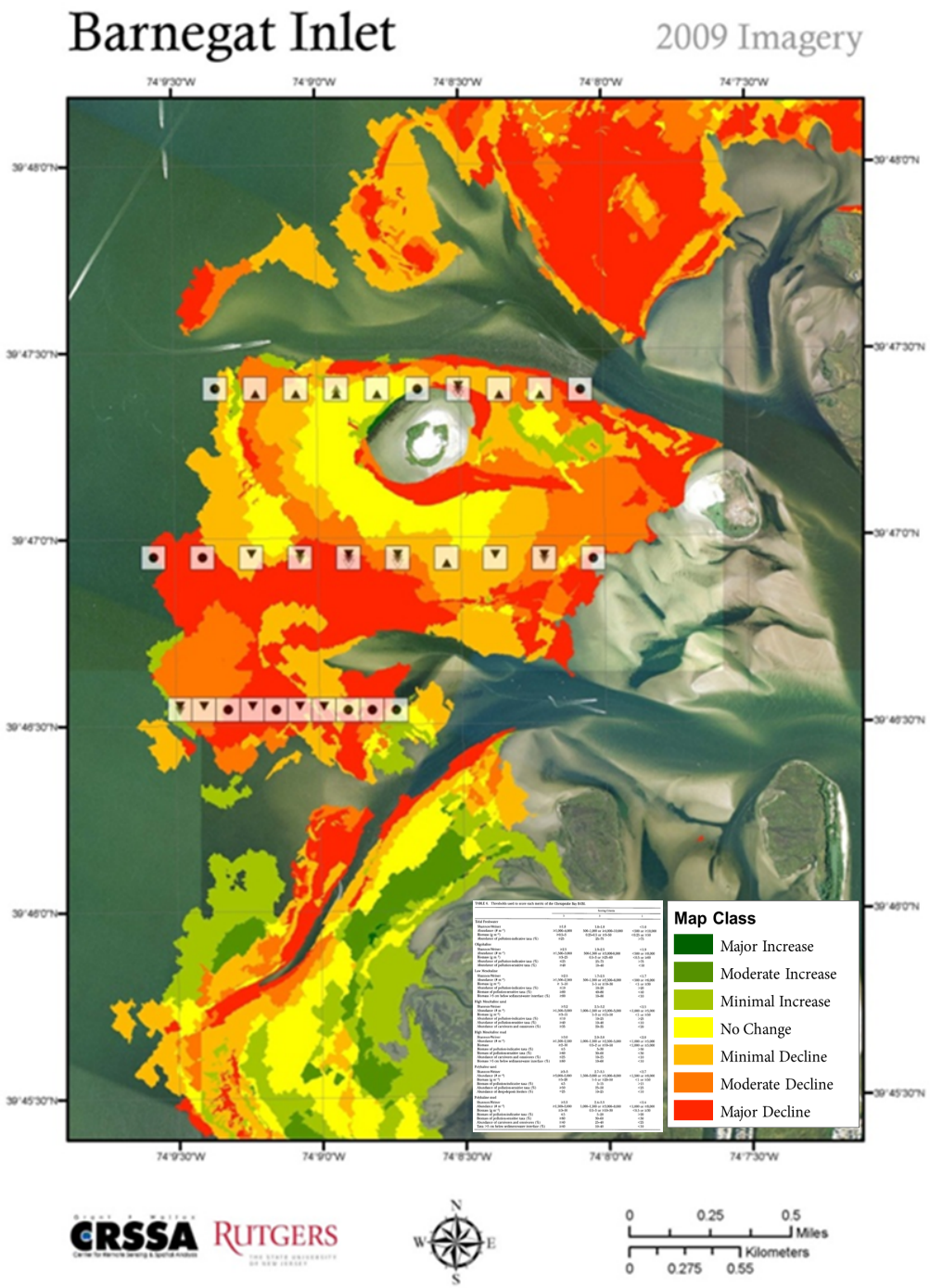
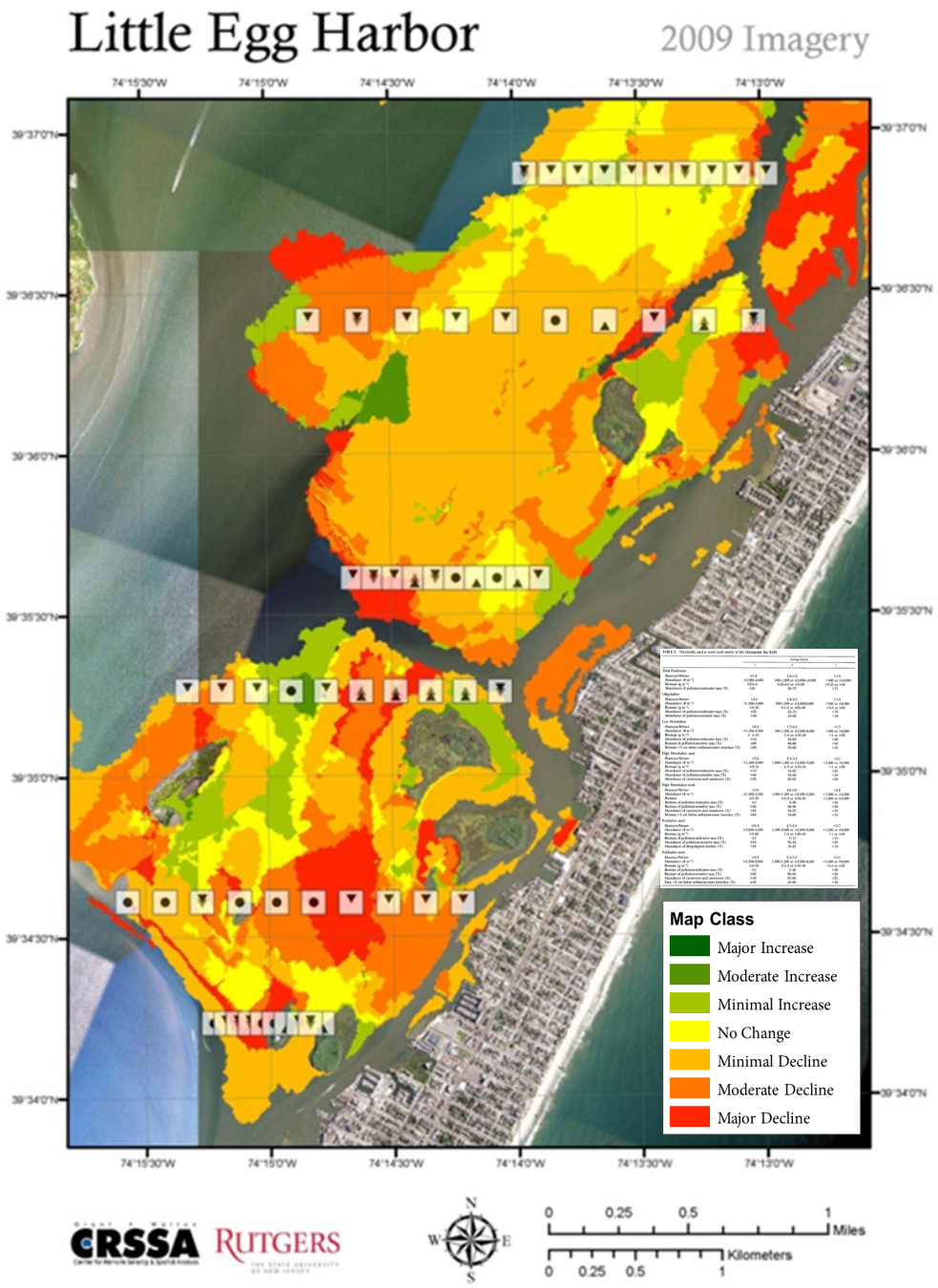


Figure 5. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Little Egg Harbor study sub-area.



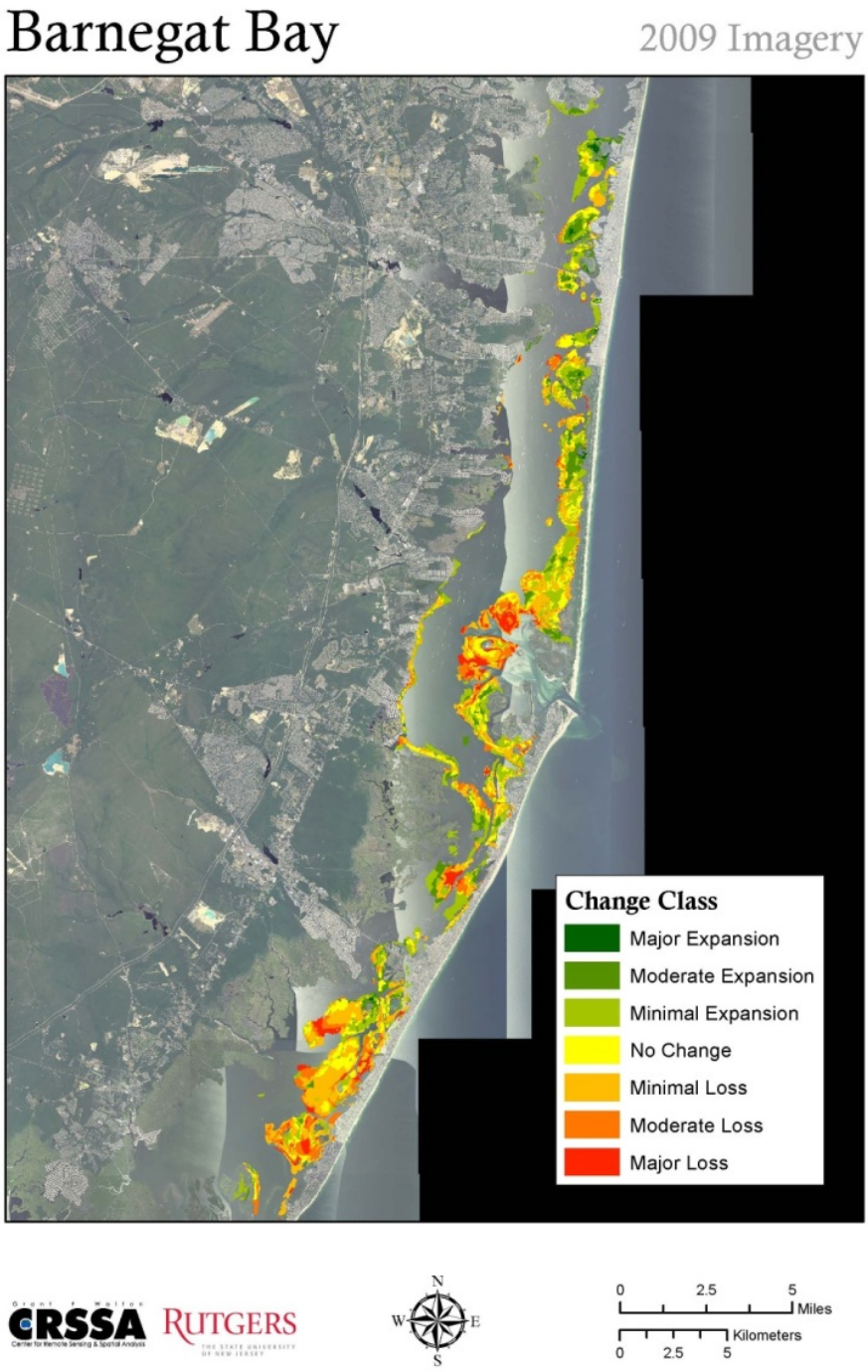
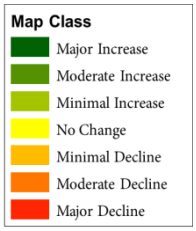


Figure 3. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Northern Barnegat Bay study sub-area.



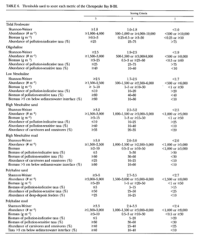


TABLE 6. Thresholds used to score each metric of the Chesapeake Bay IBI.

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Shannon-Wiener	0-1.0	1.01-2.0	>2.0	
Abundance (# per ?)	<1,000-4,000	500-2,000 or >4,000-10,000	>2000 or >10,000	
Biomass (g or ?)	<0.25	0.25-1.00	>1.00 or >10,000	
Abundance of pollution-indicative taxa (%)	>50	20-50	<20	
Oligohaline				
Shannon-Wiener	<0.5	0.5-0.25	>1.0	
Abundance (# per ?)	<1,500-2,000	500-1,500 or >2,000-8,000	>500 or >8,000	
Biomass (g or ?)	<0.01	0.01 or 0.25-.50	>0.5 or >.600	
Abundance of pollution-indicative taxa (%)	>50	20-50	<20	
Abundance of pollution-sensitive taxa (%)	<10	10-40	>40	
Low Mesohaline				
Shannon-Wiener	<0.5	1.5-0.5	>1.5	
Abundance (# per ?)	<1,500-2,000	500-1,500 or >2,000-8,000	>500 or >8,000	
Biomass (g or ?)	< 0.25	0.25 or 0.50-.50	>.50 or >.500	
Abundance of pollution-indicative taxa (%)	>10	10-40	>40	
Abundance of pollution-sensitive taxa (%)	<10	10-40	>40	
Biomass >5 cm below sediment/water interface (%)	<80	20-80	>20	
High Mesohaline mud				
Shannon-Wiener	<0.5	0.5-1.0	>1.0	
Abundance (# per ?)	<1,500-3,000	1,000-1,500 or >3,000-5,000	>1,000 or >5,000	
Biomass (g or ?)	<0.25	0.25 or >0.5-.50	>1.0 or >500	
Abundance of pollution-intolerant taxa (%)	<10	10-40	>40	
Abundance of pollution-sensitive taxa (%)	<40	10-40	>10	
Abundance of carnivores and omnivores (%)	>60	30-60	<30	
High Mesohaline sand				
Shannon-Wiener	<1.0	2.0-1.0	>2.0	
Abundance (# per ?)	<1,000-2,500	1,000-2,500 or >2,500-5,000	>1,000 or >10,000	
Biomass (g or ?)	<0.25	0.25-1.00	>1.00 or >10,000	
Abundance of pollution-indicative taxa (%)	>50	20-50	<20	
Abundance of carnivores and omnivores (%)	<10	10-40	>40	
Taxa >5 cm below sediment/water interface (%)	<60	30-60	>30	
Polyhaline mud				
Shannon-Wiener	<0.5	2.5-0.5	>2.5	
Abundance (# per ?)	<10,000-20,000	1,500-10,000 or >20,000-30,000	>1,500 or >30,000	
Biomass (g or ?)	<0.25	0.25 or >.250-.50	>.50 or >.500	
Abundance of pollution-indicative taxa (%)	>5	5-20	<20	
Abundance of pollution-sensitive taxa (%)	<10	10-40	>40	
Abundance of deep-deposit feeders (%)	<20	20-60	>60	
Polyhaline sand				
Shannon-Wiener	<0.5	0.5-1.5	>1.5	
Abundance (# per ?)	<1,500-2,000	1,000-1,500 or >2,000-8,000	>1,000 or >8,000	
Biomass (g or ?)	<0.25	0.25 or >.250-.50	>.50 or >.500	
Abundance of pollution-indicative taxa (%)	>5	5-20	<20	
Abundance of pollution-sensitive taxa (%)	<10	10-40	>40	
Abundance of carnivores and omnivores (%)	<40	20-40	>20	
Taxa >5 cm below sediment/water interface (%)	<60	30-60	>30	

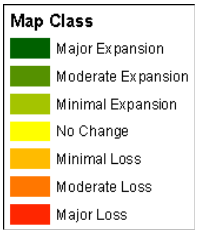


Figure 4. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Barnegat Inlet study sub-area.

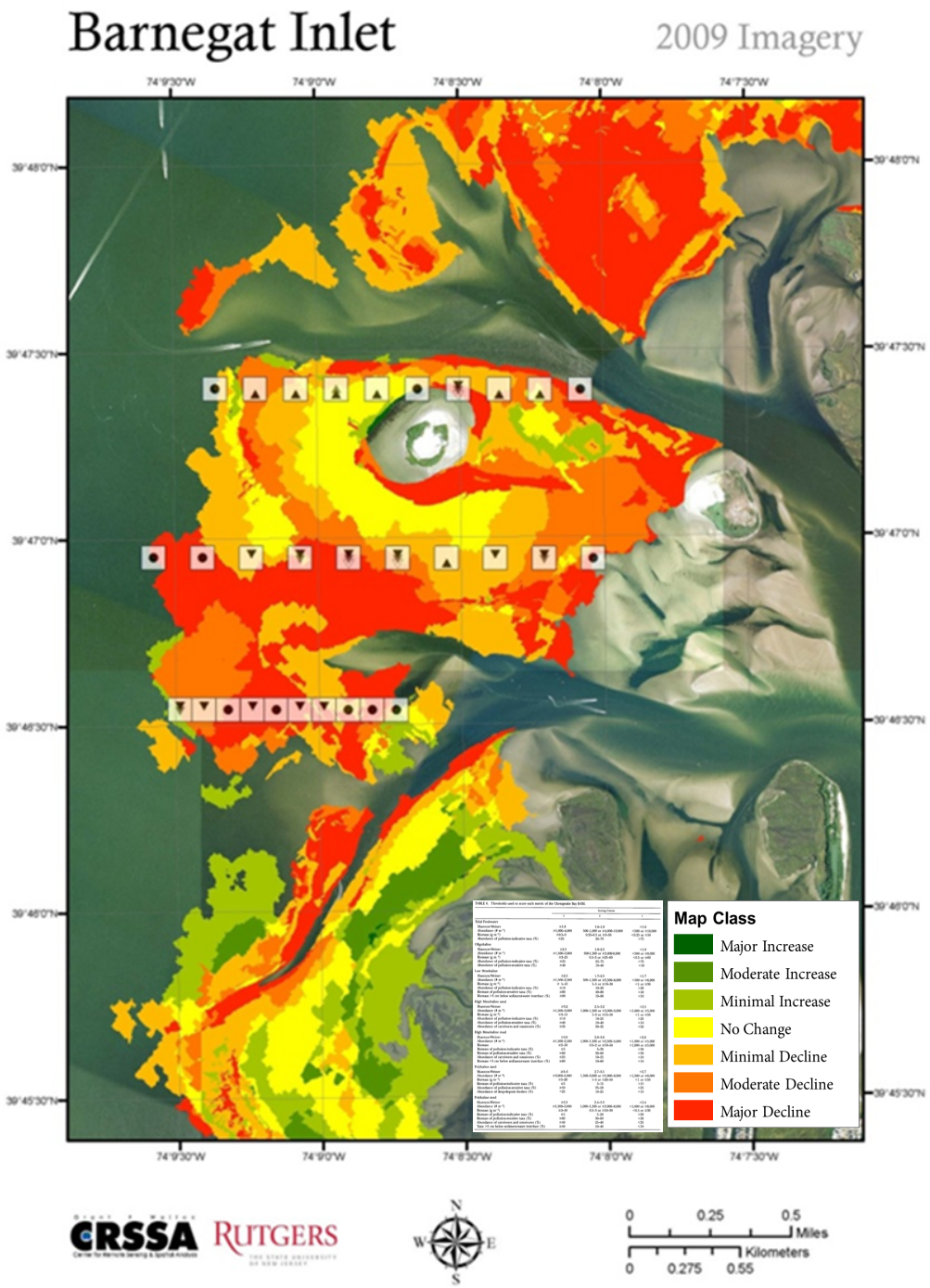
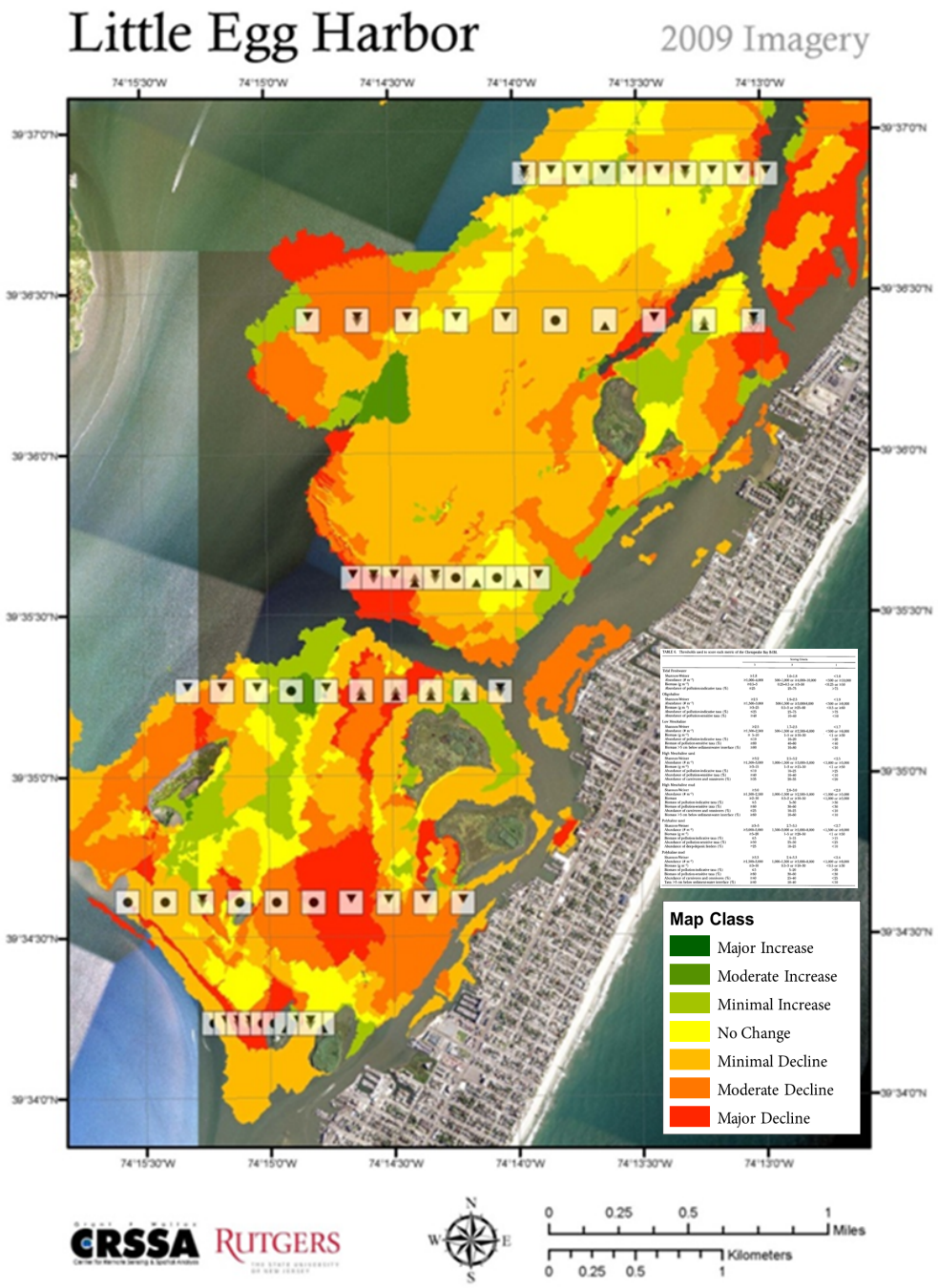


Figure 5. Seagrass 2003-2009 Change map with 2004-2010 plot-based data superimposed for the Little Egg Harbor study sub-area.



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
```

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 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 Index of Eutrophication 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 Index of Eutrophication 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 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 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 Index of Eutrophication so that if macroinvertebrate surveys are conducted in the future, those conditions can be assessed?***

Yes. The Index of Eutrophication 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 Index of Eutrophication, 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 Index of Eutrophication 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*				*Standard Error*			
Year	Segment	Total Nitrogen	Total Phosphorus	DO	Temp	TN	TP	DO	Temp	TN	TP
		kg km^{-2} yr^{-1}	kg km^{-2} yr^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}
1989	North	7844.5	370.9	8.8	17.5	828.1	NoData	0.4	1.8	83.6	NoData
1990	North	7067.2	329.8	8.7	17.2	593.1	NoData	0.3	1.2	34.0	NoData
1991	North	6466.3	299.8	10.9	5.0	368.8	NoData	0.2	0.0	14.4	NoData
1992	North	5494.3	259.2	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1993	North	7240.2	327.7	6.5	15.4	498.7	NoData	0.2	1.3	29.5	NoData
1994	North	7867.8	356.5	7.5	20.4	370.1	NoData	0.2	1.3	38.8	NoData
1995	North	5358.3	248.6	7.9	15.7	366.8	NoData	0.3	1.2	33.7	NoData
1996	North	8816.6	403.6	6.2	23.0	726.5	NoData	0.2	0.2	30.1	NoData
1997	North	7599.6	345.6	8.9	12.7	646.5	NoData	0.3	0.9	83.8	NoData
1998	North	8460.2	379.1	7.7	12.2	758.4	NoData	0.2	0.7	94.1	NoData
1999	North	6560.3	297.8	7.6	12.2	631.7	15.4	0.2	0.8	53.1	2.8
2000	North	6675.7	303.9	6.8	14.8	575.0	23.2	0.2	0.7	57.0	1.4
2001	North	5996.5	264.0	7.8	11.2	522.0	21.8	0.2	0.9	69.3	1.9
2002	North	5446.6	258.5	7.5	12.3	585.5	32.7	0.2	0.7	44.8	2.4
2003	North	8805.1	391.0	8.1	14.5	489.7	31.3	0.3	1.1	37.1	2.8
2004	North	7749.3	343.6	7.7	15.0	648.6	34.1	0.2	1.4	71.9	2.1
2005	North	8448.8	366.6	7.3	16.1	628.4	28.9	0.2	1.1	40.5	1.8
2006	North	8859.7	385.2	7.9	14.8	611.7	23.6	0.2	0.9	46.0	1.0
2007	North	8016.4	346.1	7.8	13.2	524.4	24.3	0.2	1.0	37.8	0.9
2008	North	7085.8	315.8	8.7	8.7	551.3	31.0	0.2	1.0	55.4	3.6
2009	North	8849.2	395.1	7.3	14.0	789.0	28.1	0.3	0.9	112.1	0.9
2010	North	8882.0	376.0	7.4	14.6	852.0	33.7	0.2	1.1	99.2	1.4

		Pressure		Water Quality							
		Total Loading		*Mean*				*Standard Error*			
Year	Segment	Total Nitrogen	Total Phosphorus	DO	Temp	TN	TP	DO	Temp	TN	TP
		kg km^{-2} yr^{-1}	kg km^{-2} yr^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}
1989	Central	936.3	45.2	7.9	20.0	638.3	NoData	0.3	1.4	36.6	NoData
1990	Central	846.7	40.7	8.0	17.2	470.1	NoData	0.3	0.9	33.1	NoData
1991	Central	773.0	37.0	11.0	7.1	532.1	NoData	0.3	0.4	144.6	NoData
1992	Central	653.4	31.5	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1993	Central	871.6	41.4	5.9	14.4	381.4	NoData	0.2	1.1	27.0	NoData
1994	Central	946.8	45.0	6.4	19.7	399.9	NoData	0.3	1.1	40.9	NoData
1995	Central	640.3	30.7	7.8	15.8	365.4	NoData	0.3	0.9	18.8	NoData
1996	Central	1057.9	50.5	4.7	14.7	460.2	NoData	0.2	1.7	29.9	NoData
1997	Central	913.6	43.5	8.3	12.6	355.8	NoData	0.3	1.0	16.6	NoData
1998	Central	1021.4	48.4	7.3	14.4	474.7	NoData	0.5	1.6	82.1	NoData
1999	Central	781.7	37.0	8.6	15.1	364.1	11.2	0.3	1.4	27.9	4.0
2000	Central	794.8	37.6	8.4	14.6	377.1	30.1	0.2	0.9	23.1	2.9
2001	Central	720.7	33.7	8.4	11.7	268.5	20.7	0.3	1.1	25.8	3.0
2002	Central	640.5	30.8	8.9	14.0	606.0	34.5	0.3	1.1	60.6	4.3
2003	Central	1055.8	49.5	7.9	18.2	366.6	56.7	0.2	1.0	28.7	7.7
2004	Central	929.6	43.6	7.9	15.1	351.6	56.5	0.2	1.2	20.9	6.3
2005	Central	1022.7	47.3	8.4	13.6	309.2	19.8	0.3	1.1	25.3	2.0
2006	Central	1071.8	49.6	8.7	12.0	334.9	23.2	0.2	1.1	23.3	2.4
2007	Central	971.6	44.9	7.8	16.1	371.7	29.0	0.3	1.3	29.3	2.7
2008	Central	851.2	39.8	9.1	10.9	374.4	31.4	0.4	1.3	16.8	1.3
2009	Central	1062.6	49.7	7.6	12.6	346.6	29.4	0.2	0.8	13.1	0.9
2010	Central	1082.2	49.6	6.8	15.2	425.5	38.6	0.2	1.0	17.7	2.0

		Pressure		Water Quality							
		Total Loading		*Mean*				*Standard Error*			
Year	Segment	Total Nitrogen	Total Phosphorus	DO	Temp	TN	TP	DO	Temp	TN	TP
		kg km^{-2} yr^{-1}	kg km^{-2} yr^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}	mg L^{-1}	C	µg L^{-1}	µg L^{-1}
1989	South	1144.5	53.2	8.0	19.3	681.8	NoData	0.6	2.9	37.6	NoData
1990	South	1033.5	47.7	8.5	13.3	496.2	NoData	0.2	0.8	37.3	NoData
1991	South	939.2	42.9	9.8	7.5	183.9	NoData	0.7	0.5	30.5	NoData
1992	South	795.5	36.7	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1993	South	1056.0	47.7	7.5	16.0	359.8	NoData	0.4	1.0	27.4	NoData
1994	South	1147.3	51.9	5.1	15.5	323.2	NoData	0.4	1.6	27.6	NoData
1995	South	778.1	35.6	5.8	17.1	653.4	NoData	0.3	1.0	55.3	NoData
1996	South	1283.3	58.3	4.0	12.2	345.9	NoData	0.1	1.9	22.6	NoData
1997	South	1107.5	50.2	6.2	18.5	390.0	NoData	0.5	0.5	55.8	NoData
1998	South	1236.0	55.6	7.8	15.5	612.4	NoData	0.3	1.4	56.6	NoData
1999	South	980.9	44.2	7.9	16.7	495.4	28.2	0.2	1.1	46.5	NoData
2000	South	997.7	45.1	8.1	16.0	447.4	42.2	0.1	0.6	22.4	2.7
2001	South	901.3	39.9	8.3	14.5	445.9	46.1	0.2	0.9	36.0	4.6
2002	South	808.0	37.4	8.4	12.2	748.7	58.8	0.2	0.9	59.5	6.2
2003	South	1321.6	58.9	7.4	18.5	346.2	81.3	0.3	1.5	23.9	7.1
2004	South	1163.4	51.8	8.5	11.6	347.6	44.1	0.2	1.0	23.5	2.0
2005	South	1290.7	56.2	8.9	10.3	291.5	34.9	0.1	0.4	32.8	5.5
2006	South	1353.0	59.0	8.9	10.3	336.6	24.2	0.2	0.6	22.5	1.0
2007	South	1225.5	53.2	8.9	17.7	470.2	46.8	0.4	1.7	41.6	5.2
2008	South	1078.1	47.7	9.7	7.2	347.7	31.6	0.3	1.1	20.8	1.5
2009	South	1346.1	59.6	7.7	11.8	372.9	39.2	0.3	0.8	19.1	3.8
2010	South	1361.7	58.4	6.4	16.1	400.9	47.6	0.2	0.8	20.8	3.2

Light Availability

		Mean						*Standard Error*					
Year	Segment	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light
			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	NoData	0.2	0.7
1990	North	3.5	25.6	NoData	NoData	2.0	0.9	0.3	1.3	NoData	NoData	0.1	0.3
1991	North	7.2	41.7	NoData	NoData	NoData	NoData	0.5	6.8	NoData	NoData	NoData	NoData
1992	North	NoData	NoData	NoData	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	NoData	0.1	0.2
1994	North	2.7	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	NoData	NoData	NoData
1995	North	2.8	NoData	NoData	NoData	NoData	NoData	0.2	NoData	NoData	NoData	NoData	NoData
1996	North	2.4	NoData	NoData	NoData	NoData	NoData	0.1	NoData	NoData	NoData	NoData	NoData
1997	North	3.4	24.5	NoData	NoData	0.9	15.5	0.2	2.3	NoData	NoData	0.3	5.5
1998	North	3.6	15.7	6.8	NoData	1.2	9.1	0.2	1.2	1.4	NoData	0.1	1.6
1999	North	3.0	20.0	5.3	NoData	1.0	13.6	0.1	6.1	0.6	NoData	0.1	1.9
2000	North	3.0	12.5	2.2	NoData	1.2	9.5	0.2	0.9	0.2	NoData	0.1	1.9
2001	North	3.9	11.8	2.1	NoData	1.0	10.9	0.3	1.2	0.2	NoData	0.1	1.8
2002	North	2.9	14.5	4.2	NoData	1.0	10.1	0.2	1.2	0.6	NoData	0.1	1.9
2003	North	3.0	8.8	3.7	NoData	0.9	14.0	0.2	0.8	0.6	NoData	0.1	2.2
2004	North	2.2	9.0	6.2	NoData	0.7	16.9	0.1	1.4	0.8	NoData	0.1	3.2
2005	North	2.1	7.2	7.4	NoData	0.7	19.1	0.2	0.7	0.9	NoData	0.1	2.8
2006	North	2.8	6.3	4.0	NoData	0.6	19.8	0.2	0.6	0.5	NoData	0.1	2.5
2007	North	3.7	11.7	5.6	NoData	1.0	14.4	0.3	1.4	0.5	NoData	0.1	2.4
2008	North	5.0	11.3	3.1	NoData	1.0	9.5	0.3	1.0	0.3	NoData	0.1	1.8
2009	North	5.7	13.8	4.4	NoData	0.8	8.6	0.4	3.2	0.6	NoData	0.0	0.1
2010	North	5.8	14.2	7.7	NoData	0.8	7.8	0.5	1.0	0.8	NoData	0.0	0.1

Light Availability

		Mean						*Standard Error*					
Year	Segment	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light
			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	NoData	NoData	NoData
1990	Central	3.4	30.1	NoData	NoData	2.1	1.2	0.2	1.4	NoData	NoData	0.2	0.6
1991	Central	4.1	30.3	NoData	NoData	NoData	NoData	0.3	1.9	NoData	NoData	NoData	NoData
1992	Central	NoData	NoData	NoData	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	NoData	0.4	2.1
1994	Central	3.0	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	NoData	NoData	NoData
1995	Central	2.7	NoData	NoData	NoData	NoData	NoData	0.1	NoData	NoData	NoData	NoData	NoData
1996	Central	4.4	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	NoData	NoData	NoData
1997	Central	3.2	29.8	NoData	NoData	0.6	19.8	0.1	2.6	NoData	NoData	0.3	8.2
1998	Central	2.0	5.7	2.8	NoData	0.5	26.3	0.2	1.6	1.0	NoData	0.2	3.6
1999	Central	2.7	76.4	6.1	NoData	0.7	20.3	0.2	32.7	0.8	NoData	0.2	3.5
2000	Central	3.5	16.9	2.2	NoData	1.1	14.7	0.2	1.5	0.2	NoData	0.2	3.8
2001	Central	3.8	16.3	1.5	NoData	1.3	11.5	0.3	2.1	0.2	NoData	0.1	2.6
2002	Central	2.7	28.4	4.1	NoData	1.3	13.2	0.2	4.4	0.5	NoData	0.2	3.3
2003	Central	4.5	12.2	2.9	NoData	0.6	23.2	0.2	1.5	0.5	NoData	0.2	5.0
2004	Central	3.6	15.6	5.2	NoData	0.8	16.7	0.3	2.3	0.8	NoData	0.2	6.1
2005	Central	3.9	7.0	4.3	7.8	0.6	21.8	0.4	0.9	0.7	1.1	0.1	4.8
2006	Central	4.4	13.5	3.5	4.7	0.9	16.2	0.4	2.5	0.4	0.7	0.1	3.4
2007	Central	3.9	12.0	6.0	NoData	0.8	20.4	0.4	1.4	0.7	NoData	0.2	4.8
2008	Central	4.3	15.9	3.0	13.3	0.4	21.2	0.2	1.8	0.3	1.9	0.0	0.2
2009	Central	8.1	15.2	3.3	2.5	0.3	24.1	0.4	1.0	0.3	0.7	0.0	0.1
2010	Central	4.8	18.7	7.3	1.3	0.3	23.8	0.2	1.2	0.5	0.3	0.0	0.1

		Light Availability											
		Mean						*Standard Error*					
Year	Segment	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light	Secchi	TSS	Chl a	Macroalgae	Epiphyte ratio	% light
			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	NoData	NoData	NoData
1990	South	3.3	38.7	NoData	NoData	2.3	0.4	0.2	1.3	NoData	NoData	0.1	0.1
1991	South	3.8	34.5	NoData	NoData	NoData	NoData	1.1	1.4	NoData	NoData	NoData	NoData
1992	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1993	South	3.7	42.0	NoData	NoData	NoData	NoData	0.2	1.5	NoData	NoData	NoData	NoData
1994	South	3.1	NoData	NoData	NoData	NoData	NoData	0.3	NoData	NoData	NoData	NoData	NoData
1995	South	3.2	NoData	NoData	NoData	NoData	NoData	0.2	NoData	NoData	NoData	NoData	NoData
1996	South	4.0	NoData	NoData	NoData	NoData	NoData	0.1	NoData	NoData	NoData	NoData	NoData
1997	South	4.7	12.8	4.9	NoData	NoData	NoData	0.7	5.9	2.0	NoData	NoData	NoData
1998	South	3.1	12.4	12.6	NoData	0.7	23.1	0.3	2.1	2.3	NoData	0.3	5.5
1999	South	3.3	26.2	14.0	NoData	1.6	5.1	0.2	2.3	3.1	NoData	0.2	2.2
2000	South	3.3	21.2	3.7	NoData	1.6	7.1	0.2	1.0	0.5	NoData	0.1	2.1
2001	South	3.8	22.3	2.4	NoData	1.3	6.5	0.2	1.6	0.4	NoData	0.1	1.8
2002	South	2.9	27.5	3.5	NoData	0.9	19.7	0.1	2.0	0.5	NoData	0.2	4.6
2003	South	3.6	19.4	2.5	NoData	0.2	34.1	0.3	2.6	0.6	NoData	0.1	3.2
2004	South	5.3	11.9	3.2	15.9	0.8	9.6	0.3	1.2	0.4	1.5	0.1	1.7
2005	South	4.5	22.7	3.5	NoData	1.3	4.1	0.5	4.2	0.5	NoData	0.1	1.0
2006	South	4.5	14.6	3.0	5.8	1.6	7.0	0.2	1.1	0.3	1.0	0.1	2.8
2007	South	4.0	14.1	4.5	NoData	0.8	17.9	0.4	1.7	0.7	NoData	0.2	4.1
2008	South	4.1	17.0	3.2	10.0	1.6	5.6	0.2	1.9	0.6	1.3	0.2	2.7
2009	South	6.3	21.2	1.7	12.4	0.2	32.4	0.3	1.6	0.2	1.6	0.0	0.1
2010	South	4.9	22.4	5.6	7.8	0.1	31.9	0.3	1.3	0.8	1.5	0.0	0.1

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

		Seagrass									
		Mean					*Standard Error*				
Year	Segment	Biomass above	Biomass below	Shoot density	Seagrass cover	Blade length	Biomass above	Biomass below	Shoot density	Seagrass cover	Blade length
		g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm	g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm
1989	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1990	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1991	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1992	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1993	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1994	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1995	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1996	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1997	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1998	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1999	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2000	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2001	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2002	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2003	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2004	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2005	Central	32.2	84.8	297.1	23.8	29.1	4.0	10.4	32.3	2.3	1.4
2006	Central	17.1	46.0	181.0	19.5	12.4	2.8	6.6	24.9	2.3	2.0
2007	Central	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2008	Central	26.9	66.1	346.7	25.0	31.7	4.4	11.5	48.5	2.5	1.5
2009	Central	9.1	31.4	260.1	22.5	24.8	1.9	3.8	34.5	2.4	1.5
2010	Central	9.1	29.5	506.0	18.2	27.7	1.6	3.8	55.2	2.5	1.8

		Seagrass									
		Mean					*Standard Error*				
Year	Segment	Biomass above	Biomass below	Shoot density	Seagrass cover	Blade length	Biomass above	Biomass below	Shoot density	Seagrass cover	Blade length
		g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm	g m^{-2}	g m^{-2}	shoots m^{-2}	%	cm
1989	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1990	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1991	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1992	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1993	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1994	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1995	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1996	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1997	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1998	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
1999	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2000	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2001	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2002	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2003	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2004	South	60.8	76.5	134.4	34.6	32.7	4.6	6.6	23.3	2.2	0.7
2005	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2006	South	8.4	43.0	145.4	16.1	8.0	1.3	8.8	28.0	1.8	1.2
2007	South	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData	NoData
2008	South	25.4	65.0	323.4	28.1	29.9	5.0	9.6	45.0	2.5	1.2
2009	South	8.2	33.5	250.9	26.1	20.7	1.3	3.8	30.7	2.3	0.8
2010	South	5.9	23.9	527.5	20.8	17.1	0.9	3.1	66.2	2.3	0.6

Appendix 3 - 5 Threshold information table

Water Component	Threshold Indicator	Threshold Values	Threshold value determined by:	Summary of Methods used to determine Threshold.	Threshold Limitations	Biotic Responses shown in Barnegat Bay to determine threshold values	Figures and Tables to support Threshold determination from Document	PCA Analysis Conlusions	References
Ecosystem Pressures	Total Nitrogen (kg TN yr^{-1} estuarine km^{-2})	50 kg TN estuary km^{-2} yr^{-1} = 100 250 kg TN estuary km^{-2} yr^{-1} = 75 1000 kg TN estuary km^{-2} yr^{-1} = 50 3000 kg TN estuary km^{-2} yr^{-1} = 25	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. In examining and compiling information from the literature, loading rates for total nitrogen and total phosphorus were converted to kg N year-1 for comparison with common units to modeled loads from BB-LEH (Component 1 of this report). In looking for potential thresholds among these relationships, we sought 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 or below which it appears that nitrogen loading is no longer a dominant factor in the change of the biotic response. First, we examined relationships between nutrient loading and estuarine responses in the literature (see for example, Wazniak et al., 2007; Bricker et al., 1999; Bricker et al., 2007; Tomasko et al., 1996; Short and Burdick, 1996; Deegan, 2002; Valiela et al., 2000; Burkholder et al., 2007; Boynton et al., 1996; Kennish and Fertig, 2012; Stevenson et al., 1993; Duarte et al., 1995; and Kiddon et al., 2003). Pertinent figures from these articles have	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 (see for example, Wazniak et al., 2007; Bricker et al., 1999; Bricker et al., 2007; Tomasko et al., 1996; Short and Burdick, 1996; Deegan, 2002; Valiela et al., 2000; Burkholder et al., 2007; Boynton et al., 1996; Kennish and Fertig, 2012; Stevenson et al., 1993; Duarte et al., 1995; and Kiddon et al., 2003). As nutrient loading increases, seagrass biomass and productivity decline exponentially (Tomasko et al., 1996, also Final Report Figure 3 - 9), as does areal coverage (Short and Burdick, 1996, also Final Report Figure 3 - 10 and Valiela et al., 2000, also Final Report Figure 3 - 11). Seagrass shoot density similarly declines (Deegan et al., 2002, also Final Report 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, also Final Report Figure 3 - 12) and in BB-LEH (Kennish and Fertig, 2012, also Final Report 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 including temperature, dissolved oxygen, and estuary total nitrogen and total phosphorus concentrations (Final Report Figure 3 - 14), light indicators including total suspended solids, chlorophyll a, Secchi depth, the ratio of epiphyte biomass to seagrass biomass, macroalgae percent cover, and the percent of light reaching seagrass leaves (Final Report Figure 3 - 15), and seagrass indicators including aboveground and belowground biomass, shoot density, percent cover, and blade length (Final Report Figure 3 - 16). Additional potential thresholds for total nitrogen loading were identified from changes in response indicators with changes in loading.	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 Index of Eutrophication or its component indices.	We examined total nitrogen loading impacts on water quality indicators including temperature, dissolved oxygen, and estuary total nitrogen and total phosphorus concentrations (Final Report Figure 3 - 14), light indicators including total suspended solids, chlorophyll a, Secchi depth, the ratio of epiphyte biomass to seagrass biomass, macroalgae percent cover, and the percent of light reaching seagrass leaves (Final Report Figure 3 - 15), and seagrass indicators including aboveground and belowground biomass, shoot density, percent cover, and blade length (Final Report Figure 3 - 16). Additional potential thresholds for total nitrogen loading were identified from changes in response indicators with changes in loading.	Figures: 3-9, 3-10, 3-11, 3-12, 3-13, 3-14, 3-15, 3-16, 3-17, 3-43	NA since only two indicators used for Ecosystem Pressures. Raw Scores for each weighted 50%.	Wazniak et al. 2007, Bricker et al. 1999, 2007 Tomasko et al 1996, Short and Burdick 1996, Deegan 2002, Valiela et al 2000, Burkholder et al 2007, Boynton et al 1996, Kennish and Fertig 2012, Stevenson et al 1993, Duarte et al. 1995, Kiddon et al 2003, data analysis by this study
	Total Phosphorus (kg TP yr^{-1} estuarine km^{-2})	25 kg TP estuary km-2 yr-1 = 100; 50 kg TP estuary km-2 yr-1 = 75; 100 kg TP estuary km-2 yr-1 = 50; 250 kg TP estuary km-2 yr-1 = 25	"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. In examining and compiling information from the literature, loading rates for total nitrogen and total phosphorus were converted to kg N year-1 for comparison with	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 (see for example, Wazniak et al., 2007; Bricker et al., 1999; Bricker et al., 2007; Tomasko et al., 1996; Short and Burdick, 1996; Deegan, 2002; Valiela et al., 2000; Burkholder et al., 2007; Boynton et al., 1996; Kennish and Fertig, 2012; Stevenson et al., 1993; Duarte et al., 1995; and Kiddon et al., 2003). As nutrient loading increases, seagrass biomass and productivity decline exponentially (Tomasko et al., 1996, also Final Report Figure 3 - 9), as does areal coverage (Short and Burdick, 1996, also Final Report Figure 3 - 10 and Valiela et al., 2000, also Final Report Figure 3 - 11). Seagrass shoot density similarly declines (Deegan et al., 2002, also Final Report Figure 3 - 11)	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	We examined total nitrogen loading impacts on water quality indicators including temperature, dissolved oxygen, and estuary total nitrogen and total phosphorus concentrations (Final Report Figure 3 - 14), light indicators including total suspended solids, chlorophyll a, Secchi depth, the ratio of	Figures: 3-9, 3-10, 3-11, 3-12, 3-13, 3-14, 3-15, 3-16, 3-17, 3-43	NA since only two indicators used for Ecosystem Pressures. Raw Scores for each weighted 50%.	Bricker et al. 1999, 2007 Tomasko et al 1996, Short and Burdick 1996, Deegan 2002, Valiela et al 2000, Burkholder et al 2007, Boynton et al 1996, Kennish and Fertig 2012, Stevenson et al. 1993, Kiddon et al. 2003, Wazniak et al. 2007, Duarte et al. 1995, data analysis by this study

Water Component	Threshold Indicator	Threshold Values	Threshold value determined by:	Summary of Methods used to determine Threshold.	Threshold Limitations	Biotic Responses shown in Barnegat Bay to determine threshold values	Figures and Tables to support Threshold determination from Document	PCA Analysis Conlusions	References
Water Quality	Temperature (°C)	18°C = 50; 22°C = 38; 26°C = 25; 30°C = 13	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 3-4). 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, and for total nitrogen, total phosphorus, and chlorophyll *a* (Table 3 - 9) for Maryland's coastal bays. Optimal temperatures for growth and photosynthesis of seagrass (Lee et al. 2007) guided determination of temperature thresholds (Table 3 - 10). For BB-LEH, dissolved oxygen thresholds were defined relative to the New Jersey standard of impairment, which is established at 4 mg L-1. Deviations from optimal temperatures were considered for threshold values. Temperature from April to October (inclusive) was considered with respect to these values for determining thresholds. In general,	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 3-4). 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 - 5), and for total nitrogen, total phosphorus, and chlorophyll *a* (Table 3 - 6) for Maryland's coastal bays. Optimal temperatures for growth and photosynthesis of seagrass (Lee et al. 2007) guided determination of temperature thresholds (Table 3 - 7). For BB-LEH, dissolved oxygen thresholds were defined relative to the New Jersey standard of impairment, which is established at 4 mg L-1.	Temperature data are only available from quarterly in situ observations for many years. This frequency of data collection is not sufficient to capture natural daily fluctuations. Further, this data collection frequency introduces bias with the confounding with sunlight irradiance. Continuous monitoring (observations recorded at 15 minute intervals) would better characterize 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.	effects of increased temperature on seagrass condition (including density, percent cover, biomass)	Table 3-9	weighting of 0.66 during 1989-1999; weighting of .15 during 2000-2010	Borja et al. 2004, Lee et al. 2007, Wazniak et al. 2007, Williams et al. 2009,, data analysis by this study
	Dissolved Oxygen (mg L-1)	10mg L-1 = 50; 9mg L-1 = 38; 7.5 mg L-1 = 25; 4mg L-1 = 13	Dissolved oxygen is a physiological requirement for fish, shellfish, and other invertebrates. As dissolved oxygen concentrations reach hypoxic and anoxic conditions, lethality increases (Figure 3 - 25) and benthic communities become stressed, decreasing biomass and diversity (Figure 3 - 26; Table 3 - 7, Ritter and Montagna 1999) We examined the literature and the BB-LEH database for physiological stress and lethal minimum oxygen concentrations (Breitburg 2002, Diaz and Solow 1999). Wazniak et al. (2007) report cutoff values for dissolved oxygen (Table 3-8) as < 3 mg L-1 'Does not meet objectives', 3-5 mg L-1 'Community threatened', 5-6 mg L-1 'Borderline', > 6 mg L-1 'Meets objectives', and > 7 mg L-1 'Better than objectives'. Breitburg (2002) reports seasonal patterns of dissolved oxygen in the bottom layer of a seasonally stratified temperate estuary that has undergone substantial degradation and experiences seasonal hypoxia (Figure 3 - 27). When not seasonally stressed (i.e. in winter months), dissolved oxygen concentrations can reach ~10 to 14 mg L-1 in the bottom layer. Due to its shallow depth and thorough mixing, BB-LEH does not stratify seasonally and is more similar to the surface layer of stratified estuaries and so dissolved concentrations in BB-LEH should exceed those of bottom layers of stratified estuaries.	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 3-4). 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 - 5), and for total nitrogen, total phosphorus, and chlorophyll *a* (Table 3 - 6) for Maryland's coastal bays. Optimal temperatures for growth and photosynthesis of seagrass (Lee et al. 2007) guided determination of temperature thresholds (Table 3 - 7). For BB-LEH, dissolved oxygen thresholds were defined relative to the New Jersey standard of impairment, which is established at 4 mg L-1.	Dissolved oxygen data are only available from quarterly in situ observations for many years. This frequency of data collection is not sufficient to capture natural daily fluctuations due to processes such as photosynthesis and respiration. Further, this data collection frequency introduces bias with the confounding of temperature and sunlight irradiance. Continuous monitoring (observations recorded at 15 minute intervals) would better characterize dissolved oxygen; 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.	effect of dissolved oxygen concentration on stress and survival of aquatic fauna including fish, benthic invertebrates (including shellfish)	Figure 2-14; Table 3-7	weighting of .33 during 1989-1999; weighting of .08 during 2000-2010	Bricker et al. 1999, 2007 Wazniak et al. 2007, Williams et al. 2009,Howell and Simpson 1994, Boynton et al. 1996, Diaz and Solow 1999, Breitburg 2002, Breitburg et al. 2001, Breitburg 2002, Kiddon et al. 2003, Borja et al. 2004, data analysis by this study
	Total Nitrogen Concentration (ug L-1)	135 TN ug L-1 = 50; 175 TNug L-1 = 38; 250 TNug L-1 = 25; 400 TNug L-1 = 13	Elevated nutrient concentrations spur phytoplankton and macroalgal growth and degrade seagrass (Burkholder et al. 2007). Kemp et al. (2004) document statistically derived concentrations of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) beyond which submerged aquatic vegetation is not present (<0.15 mg L-1 and < 0.01 mg L-1, respectively, which equates to < 150 µg L-1 DIN and < 10 µg L-1 DIP) in mesohaline regions (Table 3 - 6). Kemp et al. (2004) note that these thresholds are to be applied to median values of raw data collected during the growing season (April-October, inclusive). Further, Kemp et al. show the logarithmic relationship between increasing Total DIN concentration and increasing epiphyte biomass under a variety of dimensionless optical depth regimes, where optical depth = Kd * Z = the attenuation coefficient * depth (Figure 3 - 44). Inflection points for these relationships range from 10 µM Total DIN (equivalent to 140 µg L-1 total DIN) where optical depth is greatest (i.e. clearer water) to 30 µM Total DIN (equivalent to 420 µg L-1 total DIN) in more opaque water (Figure 3 - 44). Dissolved inorganic nitrogen, however, only comprises a small fraction of the total nitrogen in the water column that can be bioavailable, undergo uptake and recycling via the microbial	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 3-4). 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 - 5), and for total nitrogen, total phosphorus, and chlorophyll *a* (Table 3 - 6) for Maryland's coastal bays. Optimal temperatures for growth and photosynthesis of seagrass (Lee et al. 2007) guided determination of temperature thresholds (Table 3 - 7). For BB-LEH, dissolved oxygen thresholds were defined relative to the New Jersey standard of impairment, which is established at 4 mg L-1.	Nutrient concentrations were made quarterly in the earlier years of the project study period (1989-2010) and thus confidence of the assessment is lower during these earlier years with more limited data. Collection frequency increased over time and thus confidence increases as well for later years of the study period. 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 Index of Eutrophication or its component indices.	bioavailability and nutrient recycling resulting in increases in frequency and intensity of macroalgal blooms which is an indicator of eutrophication, as well as other primary producers including microalgae (as indicated by chlorophyll a concentration)	Figure 2-14, 3-13, 3-14, 3-23, Table 3-6, 3-8	weighting of .02 during 1989-1999; weighting of .13 during 2000-29010	Bricker et al. 1999, 2007, Burkholder et al. 2007 Kiddon et al. 2003, Kemp et al. 2004, Wazniak et al. 2007, Boynton et al. 1996, Duarte et al. 1995, Stevenson et al 1993, Kennish and Fertig 2012, , Kiddon et al. 2003,Borja et al. 2004, Williams et al. 2009,, data analysis by this study
	Total Phosphorus Concentration (ug L-1)	10 TPug L-1 = 50; 13 TP ug L-1 = 38; 22 TPug L-1 = 25; 40 TP ug L-1 = 13	Elevated nutrient concentrations spur phytoplankton and macroalgal growth and degrade seagrass (Burkholder et al. 2007). Kemp et al. (2004) document statistically derived concentrations of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) beyond which submerged aquatic vegetation is not present (<0.15 mg L-1 and < 0.01 mg L-1, respectively, which equates to < 150 µg L-1 DIN and < 10 µg L-1 DIP) in mesohaline regions (Table 3 - 6). Kemp et al. (2004) note that these thresholds are to be applied to median values of raw data collected during the growing season (April-October, inclusive). Further, Kemp et al. show the logarithmic relationship between increasing Total DIN concentration and increasing epiphyte biomass under a variety of dimensionless optical depth regimes, where optical depth = Kd * Z = the attenuation coefficient * depth (Figure 3 - 44). Inflection points for these relationships range from 10 µM Total DIN (equivalent to 140 µg L-1 total DIN) where optical depth is greatest (i.e. clearer water) to 30 µM Total DIN (equivalent to 420 µg L-1 total DIN) in more opaque water (Figure 3 - 44). Dissolved inorganic nitrogen, however, only comprises a small fraction of the total nitrogen in the water column that can be bioavailable, undergo uptake and recycling via the microbial	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 3-4). 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 - 5), and for total nitrogen, total phosphorus, and chlorophyll *a* (Table 3 - 6) for Maryland's coastal bays. Optimal temperatures for growth and photosynthesis of seagrass (Lee et al. 2007) guided determination of temperature thresholds (Table 3 - 7). For BB-LEH, dissolved oxygen thresholds were defined relative to the New Jersey standard of impairment, which is established at 4 mg L-1.	Nutrient concentrations were made quarterly in the earlier years of the project study period (1989-2010) and thus confidence of the assessment is lower during these earlier years with more limited data. Collection frequency increased over time and thus confidence increases as well for later years of the study period. 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 Index of Eutrophication or its component indices.	bioavailability and nutrient recycling resulting in increases in frequency and intensity of macroalgal blooms which is an indicator of eutrophication, as well as other primary producers including microalgae (as indicated by chlorophyll a concentration)	Table 3-6, 3-8	weighting of 0.0 during 1989-19999; weighting of .64 during 20000-2010	Burkholder et al. 2007, Bricker et al. 1999, Kiddon et al. 2003 Kemp et al. 2004, Wazniak et al. 2007, Boynton et al. 1996, Stevenson et al. 1993, Kiddon et al. 2003, Borja et al. 2004, Duarte et al. 1995, Kennish and Fertig 2012, Williams et al. 2009,, data analysis by this study

Water Component	Threshold Indicator	Threshold Values	Threshold value determined by:	Summary of Methods used to determine Threshold.	Threshold Limitations	Biotic Responses shown in Barnegat Bay to determine threshold values	Figures and Tables to support Threshold determination from Document	PCA Analysis Conlusions	References
Light Availability	Total Suspended Solids (mg L^{-1})	10 mg L-1 = 50; 12.5 mg L-1 = 38; 15mg L-1 = 25; 17.5mg L-1 = 13	Kemp et al. 2004, Bricker et al. 1999, 2007 Stevenson et al. 1993, Lee et al. 2007, Wazniak et al. 2007, Ralph et al. 2007, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study	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 (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 - 9), 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).	sampling frequency and location	degraded conditions of seagrass and other benthic primary producers, smothering of benthic fauna (especially sessile species including filter feeders)	Figure 3-25, Table 3-6,	weighting of .32 during 1998 - 2010	Kemp et al. 2004, Bricker et al. 1999, 2007 Stevenson et al. 1993, Lee et al. 2007, Wazniak et al. 2007, Ralph et al. 2007, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study
	Chlorophyll a (ug L-1)	2.5ug L-1 = 50; 3ug L-1 = 38; 4ug L-1 = 25; 6ug L-1=13	Kemp et al. 2004, Burkholder et al. 2007, Wazniak et al. 2007, Boynton et al. 1996, Bricker et al. 1999, 2007, Kiddon et al. 2003, Stevenson et al. 1993, Borja et al. 2004, Lee et al. 2007, Ralph et al. 2007, Duarte et al. 1995, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study	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 (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 - 9), 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).	sampling frequency	reduced light conditions for benthic primary producers including seagrass and benthic microalgae in sediemnts. Discoloration of water. Decreased dissolved oxygen following decomposition of microalgal detritus which	Figure 2-14, 3-13 3-24, Table 3-6, 3-8	weighting of .02 during 2000-2010	Kemp et al. 2004, Burkholder et al. 2007, Wazniak et al. 2007, Boynton et al. 1996, Bricker et al. 1999, 2007, Kiddon et al. 2003, Stevenson et al. 1993, Borja et al. 2004, Lee et al. 2007, Ralph et al. 2007, Duarte et al. 1995, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study
	Macroalgae areal cover (%cover)	3% = 50; 5% = 38; 8% = 25; 14% = 13	Kennish et al. 2011, Lee et al. 2007 Williams et al. 2009, Brush and Nixon 2002, data analysis by this study	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 (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 - 9), 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).	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.	shading and light reduction for benthic primary producers inclusing seagrass and benthci microalgae. Smothering of benthic invertebrates. Noxious odors upon decomposition resulting from macroalgal bloom population crash	Figures 2-8, 2-9, 2-10, 3-1, 3-11, 3-15, Tables 2-1, 2-2, 2-3, 2-4, 3-9.	weighting of 0 during 1998-2010 this is due to too few years of data for inclusion in the weighted score	Kennish et al. 2011, Lee et al. 2007 Williams et al. 2009, Brush and Nixon 2002, data analysis by this study
	Epiphyte to seagrass ratio (g dry wt epiphytes per g dry wt seagrass)	0.25g dry wt epiphytes per g dry wt seagrass = 50; 0.50 g dry wt epiphytes per g dry wt seagrass = 38; 1.0g dry wt epiphytes per g dry wt seagrass = 25; 1.5 g dry wt epiphytes per g dry wt seagrass = 13	Brush and Nixon 2002, Kemp et al 2004, Lee et al. 2007, data analysis by this study	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 (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 - 9), 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).	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.	shadiing and light reduction to seagrass. Noxious odors upon decomposition	Figure 3-22, 3-23	weighting off .30 during 1998-2020	Brush and Nixon 2002, Kemp et al 2004, Lee et al. 2007, data analysis by this study
	Secchi depth (m)	0.5m = 50; 0.4m = 38; 0.3m = 25; 0.2m = 13	Dennison et al. 1993, Kemp et al. 2004, Burkholder et al. 2001, Bricker et al. 1999,2007, Kiddon et al. 2003, Stevenson et al. 1993, Boynton et al. 1996, Borja et al. 2004, Lee et al. 2007, Ralph et al. 2007, Wazniak et al. 2007, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study	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 (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 - 9), 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).	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.	decreased light availabilty to benthic popiulations	Table 3-6	weighting of .04 during 1998-2010	Dennison et al. 1993, Kemp et al. 2004, Burkholder et al. 2001, Bricker et al. 1999,2007, Kiddon et al. 2003, Stevenson et al. 1993, Boynton et al. 1996, Borja et al. 2004, Lee et al. 2007, Ralph et al. 2007, Wazniak et al. 2007, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study
	Percent Light Reaching Seagrass Leaves (%)	32%=50; 23%=38; 19%=25; 15%=13	Kemp et al. 2004, Burkholder et al. 2001, Dennison et al. 1993, Stevenson et al. 1993, Boynton et al. 1996, Lee et al. 2007, Ralph et al 2007, Wazniak et al. 2007, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study	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 (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 - 9), 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).	calculated estimates based on available data	physiological light requirements	Figures 3-19, 3-20, 3-21, Table 3-6	weiighting of .31 during 1998-2010	Kemp et al. 2004, Burkholder et al. 2001, Dennison et al. 1993, Stevenson et al. 1993, Boynton et al. 1996, Lee et al. 2007, Ralph et al 2007, Wazniak et al. 2007, Williams et al. 2009, Brush and Nixon 2002, data analysis by this study

Water Component	Threshold Indicator	Threshold Values	Threshold value determined by:	Summary of Methods used to determine Threshold.	Threshold Limitations	Biotic Responses shown in Barnegat Bay to determine threshold values	Figures and Tables to support Threshold determination from Document	PCA Analysis Conlusions	References
Seagrass	Aboveground Biomass (g m^{-2})	400g m-2=50; 300g m-2=38; 200g m-2=25; 100g m-2=13	Burkholder et al. 2007, Kemp et al. 2004, Stevenson et al. 1993, Kennish et al. 2011, Kennish and Fertig 2012, Lee et al. 2007, Wazniak et al. 2007, Duarte et al. 1995, Williams et al. 2009, Dennison et al. 1993, Lea et al. 2003, data analysis by this study	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.	Eelgrass data do not start until 2004 and do not extend far back enough to capture information during steady state, and thus reference conditions for Barnegat Bay-Little Egg Harbor are difficult to estimate. Comparisons to historical coverage have been made for data collected during the study period (1989-2010). Spatially, eelgrass are not dominant in the north segment of BB-LEH due to physiological salinity requirements, although they have been observed in discrete patchy areas that are not necessarily located along established transects. The comparison to widgeongrass (Ruppia maritima) ultimately needs more time for this secondary species with lower salinity requirements to be utilized as a comparable biological indicator due to physiological differences between these two species.	population demographics, biomass declines	Figure 2-11, 2-12, 2-14, 3-10	weighting ofg 0.08 during 2004-2010	Burkholder et al. 2007, Kemp et al. 2004, Stevenson et al. 1993, Kennish et al. 2011, Kennish and Fertig 2012, Lee et al. 2007, Wazniak et al. 2007, Duarte et al. 1995, Williams et al. 2009, Dennison et al. 1993, Lea et al. 2003, data analysis by this study
	Belowground Biomass (g m^{-2})	800g m-2=50; 600g m-2=38; 400g m-2=25; 200g m-2=13	Burkholder et al. 2007, Kemp et al. 2004, Stevenson et al. 1993, Duarte et al. 1995, Kennish and Fertig 2012, Lee et al. 2007, Wazniak et al. 2007, Dennison et al. 1993, Lea et al. 2003, data analysis by this study	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.	Eelgrass data do not start until 2004 and do not extend far back enough to capture information during steady state, and thus reference conditions for Barnegat Bay-Little Egg Harbor are difficult to estimate. Comparisons to historical coverage have been made for data collected during the study period (1989-2010). Spatially, eelgrass are not dominant in the north segment of BB-LEH due to physiological salinity requirements, although they have been observed in discrete patchy areas that are not necessarily located along established transects. The comparison to widgeongrass (Ruppia maritima) ultimately needs more time for this secondary species with lower salinity requirements to be utilized as a comparable biological indicator due to physiological differences between these two species.	population demographics, biomass declines	Figure 2-11, 2-13, 2-14, 3-10	weighting of 0.02 during 2004-2010	Burkholder et al. 2007, Kemp et al. 2004, Stevenson et al. 1993, Duarte et al. 1995, Kennish and Fertig 2012, Lee et al. 2007, Wazniak et al. 2007, Dennison et al. 1993, Lea et al. 2003, data analysis by this study
	Area Cover (%)	50%=50; 25%=38; 10%=25; 5%=13	Lee et al. 2007, Burkholder et al. 2007, Kemp et al. 2004, Valiela et al 2000, Duarte et al. 1995, Deegan 2002, Stevenson et al 1993, Kennish and Fertig 2012, Wazniak et al. 2007, Williams et al. 2009, Dennison et al. 1993, Lea et al. 2003, data analysis by this study	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.	Eelgrass data do not start until 2004 and do not extend far back enough to capture information during steady state, and thus reference conditions for Barnegat Bay-Little Egg Harbor are difficult to estimate. Comparisons to historical coverage have been made for data collected during the study period (1989-2010). Spatially, eelgrass are not dominant in the north segment of BB-LEH due to physiological salinity requirements, although they have been observed in discrete patchy areas that are not necessarily located along established transects. The comparison to widgeongrass (Ruppia maritima) ultimately needs more time for this secondary species with lower salinity requirements to be utilized as a comparable biological indicator due to physiological differences between these two species.	population demographics, coverage of seagrass	Figure 2-17, 2-18, 3-11	weighting of .53 during 2004-2010	Lee et al. 2007, Burkholder et al. 2007, Kemp et al. 2004, Valiela et al 2000, Duarte et al. 1995, Deegan 2002, Stevenson et al 1993, Kennish and Fertig 2012, Wazniak et al. 2007, Williams et al. 2009, Dennison et al. 1993, Lea et al. 2003, data analysis by this study
	Shoot Density (shoots m^{-2})	1910 shoots m-2=50; 1146shoots m-2=38; 764shoots m-2=25; 382shoots m-2=13	Burkholder et al. 2007, Lee et al. 2007 Kemp et al. 2004,Kennish and Fertig 2012, Wazniak et al. 2007, Duarte et al. 1995, Lea et al. 2003, Williams et al. 2009, Dennison et al. 1993, data analysis by this study	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.	Eelgrass data do not start until 2004 and do not extend far back enough to capture information during steady state, and thus reference conditions for Barnegat Bay-Little Egg Harbor are difficult to estimate. Comparisons to historical coverage have been made for data collected during the study period (1989-2010). Spatially, eelgrass are not dominant in the north segment of BB-LEH due to physiological salinity requirements, although they have been observed in discrete patchy areas that are not necessarily located along established transects. The comparison to widgeongrass (Ruppia maritima) ultimately needs more time for this secondary species with lower salinity requirements to be utilized as a comparable biological indicator due to physiological differences between these two species.	population demographics, shoot density of seagrass	Figure 2-15, 3-12+H9	weighting of .01 during 2004-2010	Burkholder et al. 2007, Lee et al. 2007 Kemp et al. 2004,Kennish and Fertig 2012, Wazniak et al. 2007, Duarte et al. 1995, Lea et al. 2003, Williams et al. 2009, Dennison et al. 1993, data analysis by this study
	Blade Length (cm)	80cm = 50; 60cm=38; 40cm=25; 20cm=13	Burkholder et al. 2007, Kennish and Fertig 2012, Lee et al. 2007, Wazniak et al. 2007, Duarte et al. 1995, Lea et al. 2003, Dennison et al. 1993, Williams et al. 2009,, data analysis by this study	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.	Eelgrass data do not start until 2004 and do not extend far back enough to capture information during steady state, and thus reference conditions for Barnegat Bay-Little Egg Harbor are difficult to estimate. Comparisons to historical coverage have been made for data collected during the study period (1989-2010). Spatially, eelgrass are not dominant in the north segment of BB-LEH due to physiological salinity requirements, although they have been observed in discrete patchy areas that are not necessarily located along established transects. The comparison to widgeongrass (Ruppia maritima) ultimately needs more time for this secondary species with lower salinity requirements to be utilized as a comparable biological indicator due to physiological differences between these two species.	population demographics, length of seagrass	Figure 2-16	weighting of .35 during 2004-2010	Burkholder et al. 2007, Kennish and Fertig 2012, Lee et al. 2007, Wazniak et al. 2007, Duarte et al. 1995, Lea et al. 2003, Dennison et al. 1993, Williams et al. 2009,, data analysis by this study

Water Component	Threshold Indicator	Threshold Values	Threshold value determined by:	Summary of Methods used to determine Threshold.	Threshold Limitations	Biotic Responses shown in Barnegat Bay to determine threshold values	Figures and Tables to support Threshold determination from Document	PCA Analysis Conlusions	References
Harmful Algal Blooms	Aureococcus anophagefferens concentration (cells mL^{-1})	30000cells mL-1=100; 90000cells mL-1=75, 150000cells mL-1=50, 200000cells mL-1 = 25	Gastrich and Wazniak 2002, Gastrich et al 2004, data analysis by this study	An index of harmful algal blooms has previously been developed for the brown tide alga *Aureococcus anophagefferens* and is available in the literature (Gastrich and Wazniak, 2002; Figure 3 - 26). This index was developed for coastal lagoon ecosystems, and thus thresholds from this index were utilized directly to derive the rescaling equation. According to Gastrich and Wazniak (2002), the thresholds for the Harmful Algal Blooms assumes that appropriate methods are used to collect water samples and enumerate Aureococcus anophagefferens (Anderson et al. 1989, 1993 and Caron 2001). Ideally, sampling for the brown tide algae in BB-LEH is done within each estuarine segment (North, Central, South) during each year at sufficient spatial coverage.	Because of direct potential for health risks and impacts on shellfish, a precautionary approach is most appropriate for the application of these thresholds. Therefore, the maximum concentrations observed in each segment each year should be used for summarization when applying these thresholds. These thresholds are not intended to be a toxicity index (e.g. they are not based upon an identified toxin and a concentration-response) although they assume some level of toxicity to various organisms. Note that these thresholds do not predict impacts of specific concentrations of *A. anophagefferens* concentration in natural populations but do provide information on potential impacts. It is assumed that the increased concentrations and/or increased duration of blooms may potentially cause more severe impacts. As noted above, while some data collected during the study time period are available in the literature, often the locations of sampling were not, limiting the utility for hindcasting. There is a paucity of data on harmful algal bloom concentrations, with only a few years of verified 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. Specifically, when chlorophyll a levels are elevated as measured by aerial overflights. This method however, is inappropriate for monitoring for the brown tide species Aureococcus anophagefferens, as is clearly demonstrated and documented in the literature (Anderson et al. 1989, 1993). Further, light microscopy methods are further unable to detect	brown tide blooms, Gastrich et al. 2004	Figure 3-26, Figure 3-27, Figure 3-28, Table 3-14	NA. Only one variable so no PCA conducted.	Gastrich and Wazniak 2002, Gastrich et al. 2004, Anderson et al. 1989, Anderson et al. 1993, Caron 2001

Water Component	Threshold Indicator	Threshold Values	Threshold value determined by:	Summary of Methods used to determine Threshold.	Threshold Limitations	Biotic Responses Shown in Barnegat Bay to determine threshold values	Figures and Tables to support Threshold determination from	PCA Analysis Conlusions	References
Benthic Invertebrates	EMAP index values	EMAP index 2 = 100; EMAP index 1 = 75; EMAP index 0 = 50; EMAP index -1 = 25; EMAP index -2 = 0	Howell and Simpson 1994, Baden et al. 1990, Diaz and Solow 1999, Breitburg et al 2001, Breitburg 2002, Kiddon et al. 2003, data analysis by this study	There are only data from the EMAP database for 2001. In the EMAP index, 0 is used as the threshold above which is considered non-degraded and below which is considered degraded. Data distribution for 2001 was used to select the thresholds. Though there are a few stations with EMAP index values greater than 2 and less than -2, nearly all of the data falls within this range and the range was divided into equal intervals to create a linear equation which can be used to rescale the EMAP index values into another dimensionless scale with a range of 0 to 100, equal to that of the other indicators used by this project.	Only data from 2001 was available for use and no biomass data were available at all. This limits the utility of the benthic invertebrate data for an annual assessment of ecosystem condition. Validation of the benthic invertebrate data is based entirely on the original EMAP assesment. The EMAP index is known to misclassify a small percentage of samples. Some stations were located near marinas while others are not in order to test the toxicological impact of marinas. Toxicological effects may contribute to the variability and results of this data, but its impact cannot be assessed by this study. Therefore the benthic invetebrate component of this project may reflect a combination of factors in addition to eutrophication effects which cannot be separated out for this study, ultimately weakening the conclusions regarding eutrophication. Therefore, the benthic invertebrate component is considered separately from the overall Index of Eutrophication.	EMAP Index values	Figure 3-3, 3-4, Tables 2-10, 2-11, 3-1, 3-2, 3-3	NA because only one variable used and so PCA not conducted on the benthic invetebrate indicator	Howell and Simpson 1994, Baden et al. 1990, Diaz and Solow 1999, Breitburg et al 2001, Breitburg 2002, Kiddon et al. 2003, data analysis by this study

Appendix 3 - 6 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;







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*******;

*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;
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;
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;
```

Appendix 3 - 7 Example calculation of the Index of Eutrophication for 2010

Water Component	Threshold Indicator	Extreme observations if x meets conditions, use values stated. If x does not meet these conditions use the formula in column D.	Formula to determine Raw Score for this indicator	Examples performed for 2010														
				How the data should be summarized (Example: What to do with weekly samples? Daily samples? What kind of averaging should be done. Etc....) Also, how was data QA.	SEGMENT	Example: Input	Example: Raw Score	Description on How Weighting is determined for this example	Example: Weighting	Example: Weighted Score for the indicator	SEGMENT	Example: Component Index Raw Score	Example: Component Index Weighted Score	Example: Component Index Final Score	Example: Component Index Weighting	SEGMENT	Example: Overall Index of Eutrophication Condition
Ecosystem Pressures	Total Nitrogen (kg TN yr -1 estuarine km -2)	if x ≤ 50 then score =100; if x ≥ 10000 then score = 0.	y=-19*ln(x)+177.52	Annual Total TN Loading for NORTH segment each year	NORTH	8882	5	weighting is not applied to pressure scores since each are 50% of the Ecosystem Pressure Index since there are only two indicators and PCA is unnecessary to weight. Recall thatRaw Scores for Ecosystem Pressure are scale of 0 (bad) to 100 (good).	NA	NA	NORTH	NA	NA	7	NA	NORTH	7
				Annual Total TN Loading for CENTRAL segment each year	CENTRAL	1082	45			NA								
				Annual Total TN Loading for SOUTH segment each year	SOUTH	1362	40			NA	CENTRAL	NA	NA	60	NA	CENTRAL	60
	Total Phosphorus (kg TP yr-1 estuarine km-2)	if x ≤ 25 then score = 100. if x ≥ 500 then score = 0	y=-32.81*ln(x)+204.01	Annual Total TP Loading for NORTH segment each year	NORTH	376	9	weighting is not applied to pressure scores since each are 50% of the Ecosystem Pressure Index since there are only two indicators and PCA is unnecessary to weight. Recall thatRaw Scores for Ecosystem Pressure are scale of 0 (bad) to 100 (good).	NA	NA							
				Annual Total TP Loading for CENTRAL segment each year	CENTRAL	50	76			NA	SOUTH	NA	NA	56	NA	SOUTH	56
				Annual Total TP Loading for SOUTH segment each year	SOUTH	58	71			NA							
Water Quality	Temperature (°C)	if x ≤ 18 then score = 50. if x ≥ 34 then score = 0.	y=-3.125*x+106.25	average non-winter (April to October inclusive) score for each segment each year	NORTH	22.2	37	weighting by PCA on annual median Raw Scores for each segment from 2000-2010: square of the Temperature eigvalue. Multiply the weighting by the Raw Score to get the Weighted Score.	0.15	6	NORTH	20	18	37	0.33	NORTH	34
					CENTRAL	19.9	44			7							
					SOUTH	19.3	46			7							
	Dissolved Oxygen (mg L^{-1})	if x ≥ 10 then score = 50. if x ≤ 4 then score = 0	$y=4.8641*e^{0.228*x}$	average non-winter (April to October inclusive) score for each segment each year	NORTH	7.4	26	weighting by PCA on annual median Raw Scores for each segment from 2000-2010: square of the DO eigenvalue.Multiply the weighting by the Raw Score to get the Weighted Score.	0.08	2							
					CENTRAL	6.8	23			2	CENTRAL	23	19	42			
					SOUTH	6.4	21			2							
	Total Nitrogen Concentration (ug L^{-1})	if x ≤ 135 then score =50. if x ≥ 750 then score = 0	$y=26721*x^{-1.276}$	average non-winter (April to October inclusive) score for each segment each year	NORTH	852	0	weighting by PCA on annual median Raw Scores for each segment from 2000-2010; square of the TN eigenvalue.Multiply the weighting by the Raw Score to get the Weighted Score.	0.13	0							
					CENTRAL	426	12			2							
					SOUTH	401	13			2							
	Total Phosphorus Concentration (ug L^{-1})	if x ≤ 10 then score = 50. if x ≥ 45 then score = 0	$y=475.95*x^{-0.977}$	average non-winter (April to October inclusive) score for each segment each year	NORTH	33.7	15	weighting by PCA on annual median Raw Scores for each segment from 2000-2010; square of the TP eigenvalue.Multiply the weighting by the Raw Score to get the Weighted Score.	0.65	10	SOUTH	20	10	30			
					CENTRAL	38.6	13			9							
					SOUTH	47.6	0			0							
Light Availability	Total Suspended Solids (mg L^{-1})	if x ≤ 10 then score = 50. if x ≥ 20 then score = 0	y=-5*x+100	annual mean by segment	NORTH	14.2	29	weighting by PCA on annual median TSS Raw Scores for each segment from 1998-2010; square of the TSS eigenvalue. Multiply the weighting by the Raw Score to get the Weighted Score	0.32	9	NORTH	13	18	31	0.33	NORTH	34
					CENTRAL	18.7	7			2							
					SOUTH	22.4	0			0							
	Chlorophyll a (ug L-1)	if x ≤ 2.5 then score = 50. if x ≥ 7.5 then score = 0	y=-41.67*ln(x)+85.351	annual mean by segment	NORTH	7.7	0	weighting by PCA on annual median chlorophyll a Raw Scores for each segment from 1998-2010; square of the chlorophyll eigenvalue. Multiply the weighting by the Raw Score to get the Weighted Score	0.02	0							
					CENTRAL	7.3	3			0							
					SOUTH	5.6	14			0							
	Macroalgae areal cover (%cover)	if x ≤ 3 then score = 50 if x ≥ 20 then score = 0	y=-24.52*ln(x)+76.782	annual mean by segment	NORTH	No data available	NA	Weighting = 0 since data are unavailable during most years of this time period	0.00	NA							
					CENTRAL	1.3	50			0	CENTRAL	29	28	56			
					SOUTH	7.8	26			0							
	Epiphyte to seagrass ratio (g dry wt epiphytes per g dry wt seagrass)	if x ≤ 0.25 then score = 50. if x ≥ 2.0 then score = 0	y=-20.32*ln(x)+22.744	annual mean by segment	NORTH	0.8	27	weighting by PCA on annual mean epiphyte biomass to seagrass biomass ratio Raw Scores for each segment from 1998-2010; square of the ratio eigenvalue. Multiply the weighting by the Raw Score to get the Weighted Score	0.30	8							
					CENTRAL	0.3	47			14							
					SOUTH	0.1	50			15							
	Secchi depth (cm)	if x ≥ 500 then score = 50. if x ≤ 100 then score = 0	y=0.125*x-12.5	annual mean by segment	NORTH	177	10	weighting by PCA on annual mean Secchi Raw Scores for each segment from 1998-2010; square of the Secchi eigenvalue. Multiply the weighting by the Raw Score to get the Weighted Score	0.04	0							
					CENTRAL	invalid due to hitting bottom	NA			NA							
					SOUTH	invalid due to hitting bottom	NA			NA	SOUTH	28	31	59			
	Percent Light Reaching Seagrass Leaves (%)	if x ≥ 32 then score = 50. if ix ≤ 7.818 then score = 0	y=50.084*ln(x)-122.18	annual mean by segment	NORTH	7.8	0	weighting by PCA on annual mean percent light to seagrass Raw Scores for each segment from 1998-2010; square of the percent light eigenvalue. Multiply the weighting by the Raw Score to get the Weighted Score	0.31	0							
					CENTRAL	23.8	37			11							
					SOUTH	31.9	51			16							
Seagrass	Aboveground Biomass (g m^{-2})	if x ≥ 400 then score = 50. if x ≤ 0 then score = 0	y=0.125*x	June-November (inclusive) mean for each segment each year	NORTH	No data available	NA	weighting by PCA on annual mean aboveground biomass Raw Scores for each segment from 2004-2010. square of the biomass eigenvalue. Multiple the weighting by the Raw Score to get the Weighted Score	0.08	NA	NORTH	NA	NA	NA	0.33	SOUTH	42
					CENTRAL	9.1	1			0							
					SOUTH	5.9	1			0							
	Belowground Biomass (g m^{-2})	if x ≥ 800 then score = 50. if x ≤ 0 then score = 0	y=0.0625*x	June-November (inclusive) mean for each segment each year	NORTH	No data available	NA	weighting by PCA on annual mean belowground biomass Raw Scores for each segment from 2004-2010. square of the biomass eigenvalue. Multiple the weighting by the Raw Score to get the Weighted Score	0.02	NA							
					CENTRAL	29.5	2			0							
					SOUTH	23.9	1			0							
	Area Cover (%)	if x ≥ 50 then score = 50. if x ≤ 0 then score = 0	y=15.925*ln(x)-12.713	June-November (inclusive) mean for each segment each year	NORTH	No data available	NA	weighting by PCA on annual mean percent cover Raw Scores for each segment from 2004-2010. square of the percent cover eigenvalue. Multiple the weighting by the Raw Score to get the Weighted Score	0.53	NA							
					CENTRAL	18.2	33			18	CENTRAL	14	24	38			
					SOUTH	20.8	36			19							
	Shoot Density (shoots m^{-2})	if ≥ 1910 then score = 50. if x ≤ 0 then score = 0	y=0.0243*x+5.7143	June-November (inclusive) median for each segment each year	NORTH	No data available	NA	weighting by PCA on annual median shoot density Raw Scores for each segment from 2004-2010. square of the density eigenvalue. Multiple the weighting by the Raw Score to get the Weighted Score	0.01	NA							
					CENTRAL	506	18			0							
					SOUTH	528	19			0							
	Blade Length (cm)	if x ≥ 80 then score = 50. if x ≤ 0 then score = 0	y=0.625*x	June-November (inclusive) mean for each segment each year	NORTH	No data available	NA	weighting by PCA on annual mean blade length Raw Scores for each segment from 2004-2010. square of the length eigenvalue. Multiple the weighting by the Raw Score to get the Weighted Score	0.35	NA							
					CENTRAL	27.7	17			6							
					SOUTH	17.1	11			4							
Harmful Algal Blooms	Aureococcus anophagefferens concentration (cells mL^{-1})	if x ≤ 30000 then score = 100. if x ≥ 260000 then score = 0	y=-0.0004*x+113.98	maximum cells mL-1 for each segment for each year	NORTH	No data available	NA	No Data available, hence variability assumed to be null	NA	NA	NORTH	NA	NA	NA	1.00	NORTH	NA
					CENTRAL	No data available	NA	No Data available, hence variability assumed to be null	NA	NA	CENTRAL	NA	NA	NA		CENTRAL	NA
					SOUTH	No data available	NA	No Data available, hence variability assumed to be null	NA	NA	SOUTH	NA	NA	NA		SOUTH	NA
Benthic Invertebrates	EMAP index values	if x ≥ 2 then score = 100. if x ≤ -5 then score = 0	y=20.2x + 55.6	mean for East and mean for West within each segments (total 6 segments) for each year	NORTH	No data available	NA	No Data available, hence variability assumed to be null	NA	NA	NORTH	NA	NA	NA	1.00	NORTH	NA
					CENTRAL	No data available	NA	No Data available, hence variability assumed to be null	NA	NA	CENTRAL	NA	NA	NA		CENTRAL	NA
					SOUTH	No data available	NA	No Data available, hence variability assumed to be null	NA	NA	SOUTH	NA	NA	NA		SOUTH	NA