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Use of GIS Mapping and Modeling Approaches to Examine the Spatial Distribution of Seagrasses in Barnegat Bay, New Jersey

RICHARD G. LATHROP[1,*], RENEE M. STYLES[2], SYBIL P. SEITZINGER[2], and JOHN A. BOGNAR[1]

[1] *Center for Remote Sensing and Spatial Analysis, Rutgers University, New Brunswick, New Jersey 08901-8551*
[2] *Institute of Marine and Coastal Science, Rutgers University, New Brunswick, New Jersey 08901-8551*

ABSTRACT: Due to the ecological importance of seagrasses and recent indications of disease and dieback, we have synthesized existing mapped survey information concerning the spatial and temporal distribution of seagrass beds (primarily eelgrass, *Zostera marina*) in Barnegat Bay, New Jersey. Mapped surveys from the 1960s, 1970s, 1980s, and 1990s were digitized and compiled in a geographic information system to facilitate analysis. Comparison of the earlier maps with the 1990s survey shows an overall decrease of approximately 2,000 to 3,000 ha in the area of seagrass beds. While there are indications of seagrass decline, due to the great difference in mapping methods used for each of the surveys, we are cautious in directly attributing the decrease in mapped eelgrass acreage to a large-scale dieback. We examined the extent to which light could be used to predict the distribution of seagrass in Barnegat Bay. Data on Secchi depth throughout the bay were combined with a modification of an existing model (Duarte 1991) of the relationship between *Z. marina* compensation depths and light attenuation coefficients to predict the distribution of seagrasses in Barnegat Bay. When compared with mapped seagrass distribution in the bay, the model correctly predicts seagrass presence-absence over two-thirds of the time. The majority of the model error is due to errors of commission, i.e., the model predicts seagrass occurrence where it was not observed to occur. Most of this commission error is located in specific geographic areas (i.e., southern third of Little Egg Harbor and the western shoreline of the bay).

Introduction

Seagrasses play an important ecological role in coastal ecosystems due to their habitat value for many organisms (Larkum and Hartog 1989) and contribution to primary production (Hillman et al. 1989). An extensive body of literature exists on seagrasses and factors influencing their growth, depth limits, and distribution. Light, temperature, salinity, substrate, nutrient levels, epiphytes, and disease have all been found to affect the survival and distribution of seagrasses (e.g., Twilley et al. 1985; Dennison 1987; Duarte 1991; Burkholder et al. 1994; Short et al. 1995; Taylor et al. 1995; Moore et al. 1996, 1997; Short and Burdick 1996).

Many coastal ecosystems in the United States, Europe, and Australia have experienced a decline in seagrass abundance during recent decades (e.g., Orth and Moore 1983; Shepherd et al. 1989; Giesen et al. 1990; Hall et al. 1999; Sigua et al. 2000). The causes of these declines were not always clear but have been attributed to natural as well as human causes. Increased development of estuarine watersheds leading to increased anthropogenic nutrient inputs and increased water column turbidity are often considered as important factors contributing to seagrass decline (Orth and Moore 1988; Dennison et al. 1989; Koch and Beer 1996; Short and Burdick 1996; Short and Wyllie-Echevarria 1996).

A number of approaches have been used to examine factors controlling seagrass distribution including mesocosms (Twilley et al. 1985; Short et al. 1995; Taylor et al. 1995; Moore and Wetzel 2000), models, and mapping combined with measures of water quality. Field surveys and more recently remote sensing techniques including aerial photography and satellite imagery have been used to map the distribution of seagrasses throughout an estuary or coastal system (Zieman et al. 1989; Ferguson et al. 1993; Ferguson and Korfmacher 1997; Mumby et al. 1997; Robbins 1997; Ward et al. 1997). Photogrammetric interpretation of color aerial photography is generally considered the optimal method for comprehensive mapping and change detection of seagrasses and other submersed rooted vascular plants (Dobson et al. 1995). Numerous models have been developed to describe seagrass growth, survival, and/or potential habitat. These

* Corresponding author; tele: 732/932-1580; e-mail: lathrop@crssa.rutgers.edu.

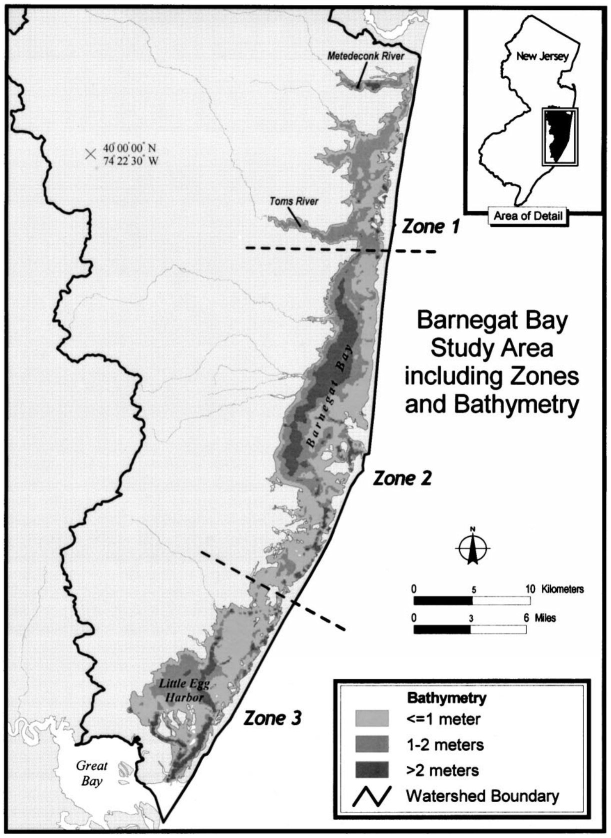

Fig. 1. Barnegat Bay study area including area zones and bathymetry.

models range from primarily optically based (Duarte 1991; Gallegos 1994; Zimmerman et al. 1994) to a combination of optical and water quality criteria (Dennison et al. 1993) to dynamic ecosystem simulation models (Madden and Kemp 1996). Each seagrass model usually has been developed or recalibrated using both water quality data and seagrass distribution information from one particular estuary or coastal system. In some cases prediction of the presence/absence of seagrasses has been compared at a few additional sites in that system. Existing seagrass models have not been applied to an independent coastal system or incorporated into a GIS approach to test the model extensively throughout the coastal system for which the model was originally developed.

Barnegat Bay is a shallow back-bay lagoonal type of estuary on New Jersey's Atlantic coast (Fig. 1) and contains approximately 75% of New Jersey's estuarine submerged aquatic vegetation habitat (Lathrop et al. 2001). Barnegat Bay shows a pattern of higher water turbidity as well as greater nutrient loading in the northern bay (Moser 1997; Hunchak-Kariouk et al. 1999) which follows the pattern of greater watershed development in the northern bay (Lathrop and Bognar 2001). The impact of increased development within the watershed coupled with the possible recurrence of wasting disease (McClain and McHale 1996; Bologna et al. 2000) has sparked concern about the status of eelgrass, *Zostera marina,* and other seagrasses in Barnegat Bay, New Jersey. Due to the ecological importance of seagrass and recent indications of disease and dieback, we synthesized existing mapped survey information concerning the spatial and temporal distribution of seagrass beds (primarily eelgrass) in Barnegat Bay. Mapped surveys from the 1960s, 1970s, 1980s, and 1990s were digitized and input to a geographic information system to facilitate visual and spatial analysis of the geographic patterns. An existing relationship between water column turbidity and seagrass depth limits (Duarte 1991) was then used to develop a GIS-based spatial model to predict the distribution of potential seagrass habitat throughout Barnegat Bay under current water quality conditions as well as under theoretical scenarios. The modeled distributions were then compared to mapped distributions.

Methods

Mapping Methods

Mapped information on the spatial distribution of seagrass beds for Barnegat Bay was derived from several sources. The first systematic survey was undertaken in 1968 (U.S. Army Corps of Engineers 1976). The methods for this study were not detailed but are presumed to be a boat-based survey. There were several mapping efforts during the 1970s. The lower portion of Little Egg Harbor was mapped based on springtime aerial photography acquired in 1977, as a pilot project to examine the feasibility of mapping submerged aquatic vegetation (SAV), including seagrass beds, with aerial photography (Good et al. 1978). Based on the success of this project, the Earth Satellite Corporation mapped the state's entire Atlantic coast and produced a 1:24,000 scale map series for the entire bay based on interpretation of black and white aerial photography and low altitude sea plane reconnaissance during the summer of 1979 (photos taken June and August, field checked July through September) (Macomber and Allen 1979). For both the 1968 and 1979 survey, they mapped four general types of SAV communities (eelgrass *Z. marina,* widgeongrass *Ruppia maritima,* mixed eelgrass and widgeongrass, and sea lettuce *Ulva lactuca* dominated) at various levels of density. We table digitized their paper maps for later GIS analysis using existing shoreline GIS data as a base map. The U.S. Fish and Wildlife Service incorporated the Earth

Satellite Corporation maps into the National Wetland Inventory for the state of New Jersey.

During 1985 to 1987, information on eelgrass distribution, water depth, and bottom sediments was collected in conjunction with an estuarine shellfish inventory of Barnegat and Little Egg Harbor bays (Joseph et al. 1992). These surveys spanned the spring, summer, and autumn seasons. Benthic samples were collected at approximately 0.4 km (one-quarter mile) intervals in the deeper waters (greater than 0.9 m; 3 feet) of the bay, for a total of 489 stations. The extreme northern end of the Bay (i.e., Metedeconk River) was not surveyed. Based on this survey, the distribution of eelgrass beds was then interpolated and mapped onto a nautical chart base map and produced as figures (Joseph et al. 1992). The resulting map included two areas that were not explicitly sampled (i.e., the boat did not enter due to the shallow depth and no benthic samples were taken) but where visual reconnaissance noted the occurrence of eelgrass beds. These two areas were included as containing eelgrass (for a total of 896 ha) in our analysis. We table digitized photocopies of the eelgrass distribution maps of Joseph et al. (1992), using existing shoreline GIS data as a base map to provide ground control points.

A field survey of seagrasses was conducted during the summers of 1996, 1997, 1998 (McLain and McHale 1996), and 1999 (Bologna et al. 2000). The middle portion of the bay was mapped in 1996, the northern portion in 1997, and the southern portion (e.g., Little Egg Harbor) in 1998. The southern portion of the bay was re-mapped in 1999 using a differentially corrected (post-processing) global positioning system (GPS). During boat-based surveys, McLain and McHale (1996) and Bologna et al. (2000) identified and mapped SAV beds onto a 1:40,000 scale National Oceanic and Atmospheric Administration (NOAA) nautical chart (Charts 12324 and 12316). The dominant species (i.e., *Zostera* or *Ruppia*) were noted. We then table digitized these annotated charts and integrated them with the GPS mapped data.

In all cases, the various digitized maps were projected to a Universal Transverse Mercator (UTM) coordinate system (datum NAD83; spheroid GRS 1980). A GIS map of the bay shoreline (New Jersey Department of Environmental Protection [NJDEP] 1996) was used to provide ground control points to rubber sheet the maps where needed to a common base map. The various maps were overlaid and analyzed to examine the consistency in mapping interpretation as well as possible changes in the spatial distribution between the 1960s, 1970s, 1980s, and 1990s.

Modeling Approach

A number of modeling approaches have been used to define potential seagrass habitat. A model developed for Chesapeake Bay predicted the presence or absence of SAV at 1 m depth using a set of springtime water quality criteria (light attenuation coefficients, Secchi depth, total suspended solids, chlorophyll, and dissolved inorganic nitrogen and phosphorus) (Dennison et al. 1993). A model of potential seagrass habitat was developed for the Rhode River and Chincoteague Bay estuaries based on light attenuation coefficients (Gallegos 1994). Light attenuation was related to CDOM, chlorophyll *a,* and non-algal particulate matter in the water column. The optical component of the model was later recalibrated and applied to two sites in Indian River Lagoon, Florida (Gallegos and Kenworthy 1996). The optical model of Duarte (1991) related compensation depth of *Z. marina* to light attenuation coefficients using data from a wide range of geographic locations, including northern Europe, eastern North America, California, Mexico, and Japan. In this study, we applied (a modified form of) the Duarte (1991) model in a GIS framework to Barnegat Bay. The Duarte (1991) model was chosen because it was specific for *Z. marina,* the dominant seagrass in Barnegat Bay; it was originally developed using data from a wide range of geographic locations, and there were extensive water column turbidity data for Barnegat Bay for comparison with the mapped distribution of seagrasses in the bay.

The depth limits (compensation depth; Z_c) of *Z. marina* across a wide range of geographic locations show a significant relationship with light attenuation coefficients (K_d): $\log Z_c\ (m) = 0.27–0.84 \times \log K_d\ (m^{-1})$ ($r^2 = 0.40$, $n = 29$, $p < 0.001$; Duarte 1991). Using the data compiled in Duarte (1991), we derived an alternate form of the relationship:

$$Z_c\ (m) = 1.59/K_d\ (m^{-1}) \tag{1}$$

As noted above, *Z. marina* is the dominant seagrass in Barnegat Bay. Only a small portion of northern Barnegat Bay SAV beds, where salinity levels are lower, is dominated by *R. maritima,* although *Z. marina* is still present. Similar compensation depths have been reported for *Ruppia* and *Zostera* (Orth and Moore 1988). Equation 1 was used to map compensation depths of both seagrasses in the bay.

While there are few measurements of light attenuation coefficients based on detailed light profiles in Barnegat Bay, there are extensive measurements of Secchi disk depths (S_d) in the bay (NJDEP unpublished data). Therefore, we modified Eq. 1 for use with Secchi disk depths. Attenuation coefficients were estimated from the Secchi disk depths using the equation of Giesen et al. (1990):

$$K_d\ (m^{-1}) = 1.65/S_d\ (m) \tag{2}$$

Substituting Eq. 2 into Eq. 1 yields:

$$Z_c\ (m) = 0.96S_d\ (m). \tag{3}$$

The spatial distribution of water column turbidity, as measured by Secchi disk depth (S_d), was mapped throughout Barnegat Bay using the Secchi disk depth data from over 40 point stations collected at repeated intervals between 1993 and 1997 (NJDEP unpublished data). This data set was input to the ArcView GIS software for further processing and spatial analysis (UTM: datum NAD 83; spheroid WGS 1980). The original S_d data was converted from units in feet to meters. The data were extracted for the spring (April–June) and summer (July–September) seasons for a total of 41 stations for the spring season and 41 stations for the summer. Each station was sampled approximately once each season during each year. A mean S_d value was calculated for the entire time period (1993–1997) for each sampling station by season (spring and summer). The comparative utility of various techniques of interpolating the S_d point to a grid within the ArcView software, namely inverse distance weighting and kriging, were examined. Using a 100 m grid cell, the two techniques produced very similar results with S_d distributions that were not statistically significant (*t* test, $p = 0.8521$). Kriging was chosen because it uses the information from the semivariogram to determine the optimal set of weights for the estimation of the surface at unsampled grid cells (Davis 1986). The kriging parameters for both the spring and summer data sets included an exponential model and a search radius of 5,000 m.

In any interpolation procedure a major concern is how well the resulting outputs ''honor the data points'' (Davis 1986, p. 375). There is a limit on how fine the output grid cell size can be made given the spatial frequency and distribution of the input sampling points. Compared with the bottom depth data (see below), the S_d data was comparatively sparse with a mean distance between any output grid cell and the nearest input sample point of approximately 1,200 m. Various output cell sizes, ranging from 100 to 1,000 m, were examined. Comparison of the results using a 1,000 versus 100 m spacing showed very similar results with S_d distributions that were not statistically significant (*t* test, $p = 0.84$). Because the spatial distribution of S_d within Barnegat Bay is smoothly continuous without a lot of fine scale spatial heterogeneity, the distribution of sampling points appeared to adequately capture the spatial variation of S_d values within the bay (Fig. 2). A grid cell size of 100 m was chosen to provide a suitably detailed picture of the water turbidity for the seagrass modeling effort without unduly compromising the integrity of the S_d data or the mapped seagrass distribution. Equation 3 was then applied to produce an average predicted compensation depth, Z_c, for each grid cell.

A GIS map of bottom depth (m) was derived through digitization of individual depth readings (at mean lower low water) from the NOAA nautical chart (Charts 12324: edition 25, 1990 and 12316: edition 25, 1992). Due to the large number of data points (total of 2,627 sample points) well distributed throughout the bay and the sometimes irregular nature of bottom topography, a simple inverse distance weighted technique was chosen. These bottom depth readings were then interpolated to create a grid cell map of 100 m cell size in the same projection system as the S_d maps. The mean distance between any output grid cell and the nearest input sample point was approximately 180 m.

One potential problem with relying on S_d data to estimate the compensation depth, Z_c, in shallow waters is that under high transparency conditions the Secchi disk can be visible all the way to the bay bottom. Under these conditions, the S_d will be recorded as deep as the bottom depth, thereby overestimating the K_d and underestimating the Z_c for that location. Unfortunately the NJDEP data set does not record where and when this situation occurs. Most of the NJDEP sampling stations were taken in deeper water where the above situation is not normally a problem. Even if this above situation occurs, it should not materially affect the prediction of suitable seagrass habitat, in that if there is enough light at the bottom to see the Secchi disk then there should be enough light to support eelgrass.

The compensation depth, Z_c, was then subtracted from the bottom depth, Z_b, at each grid cell. Potential eelgrass habitat was then defined as those grid cells where $Z_c - Z_b > 0$, i.e., the bottom is above the Z_c and therefore sufficiently illuminated. As the water depth was based on mean lower low water, this model is a best case estimate of available light at the bottom. During high tide, some areas might not be sufficiently illuminated all the way to the bottom. If $Z_c - Z_b < 0$, then the bottom is below the Z_c and therefore not sufficiently illuminated. The resulting maps, one for spring and one for summer, of the potential or expected seagrass distribution were then compared with the observed distributions from the 1990s summer survey. The 1990s seagrass map was converted to a 100 m grid cell and the analysis conducted using the ERDAS IMAGINE Software package. Various mea-

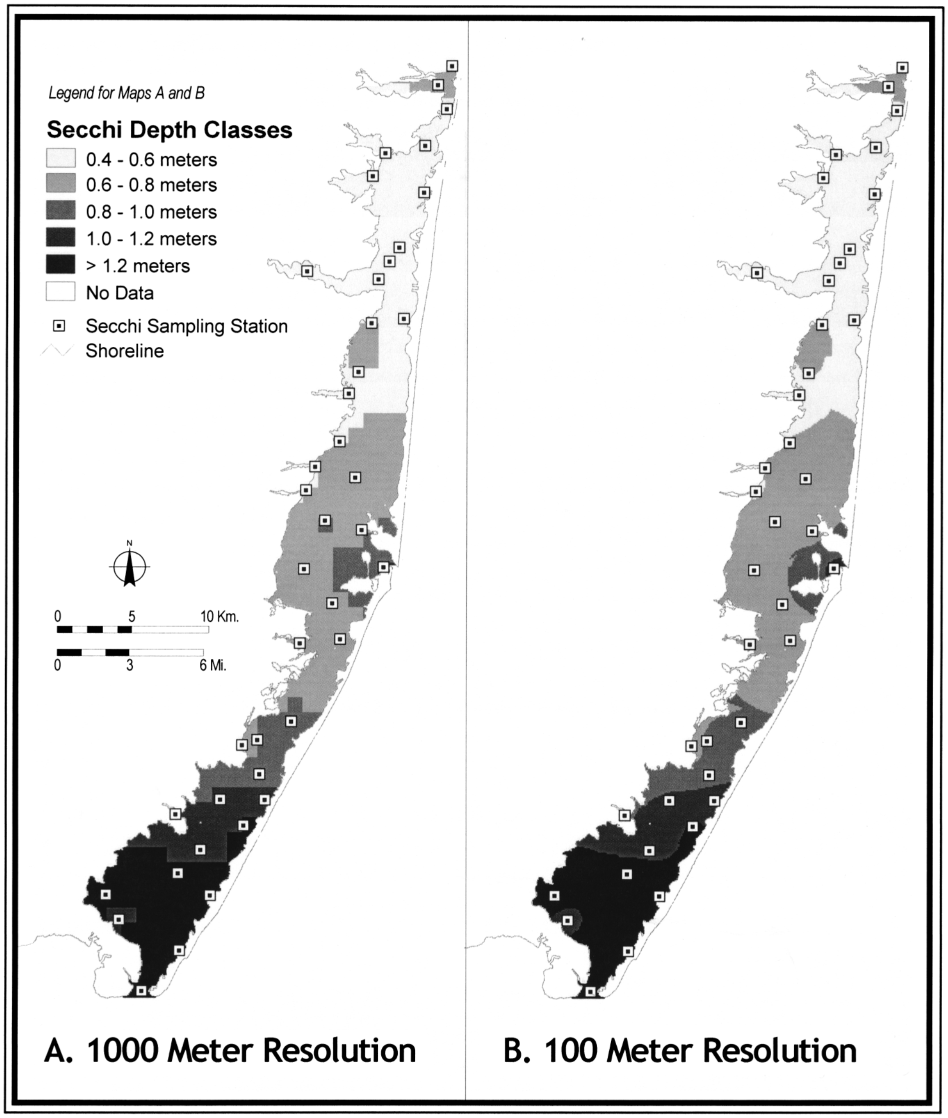

Fig. 2. Map of kriged S_d values for the summer data set with location of 41 sampling stations. A) 1,000 m resolution grid cell. B) 100 m resolution grid cell.

sures of accuracy including the percent correct, errors of omission and commission, and the Kappa statistic were calculated. The error of omission represents the ratio of the area incorrectly predicted as seagrass absent over the total area observed as seagrass present. The error of commission represents the ratio of the area incorrectly predicted as seagrass present over the total area predicted as seagrass present. The Kappa statistic is a measure of the difference between actual agreement between reference data and the model output and the chance agreement between reference data and a random assignment (Congalton 1991). The Kappa statistic varies between −1 and 1 with 1 representing a perfect agreement.

Results and Discussion

Spatial and Temporal Trends in Seagrass Distribution

While we were fortunate in having a series of SAV surveys dating back over thirty years, the great difference in mapping methods and the poor quality of some of the hard-copy mapped products, made a rigorous comparison of the surveys problematic. The 1970s survey (Macomber and Allen 1979) relied on aerial photography complemented by float plane-assisted field checking. The 1980s survey (Joseph et al. 1992) relied on a boat-based systematic grid sampling (one-quarter mile interval) while the 1990s survey (McLain and McHale

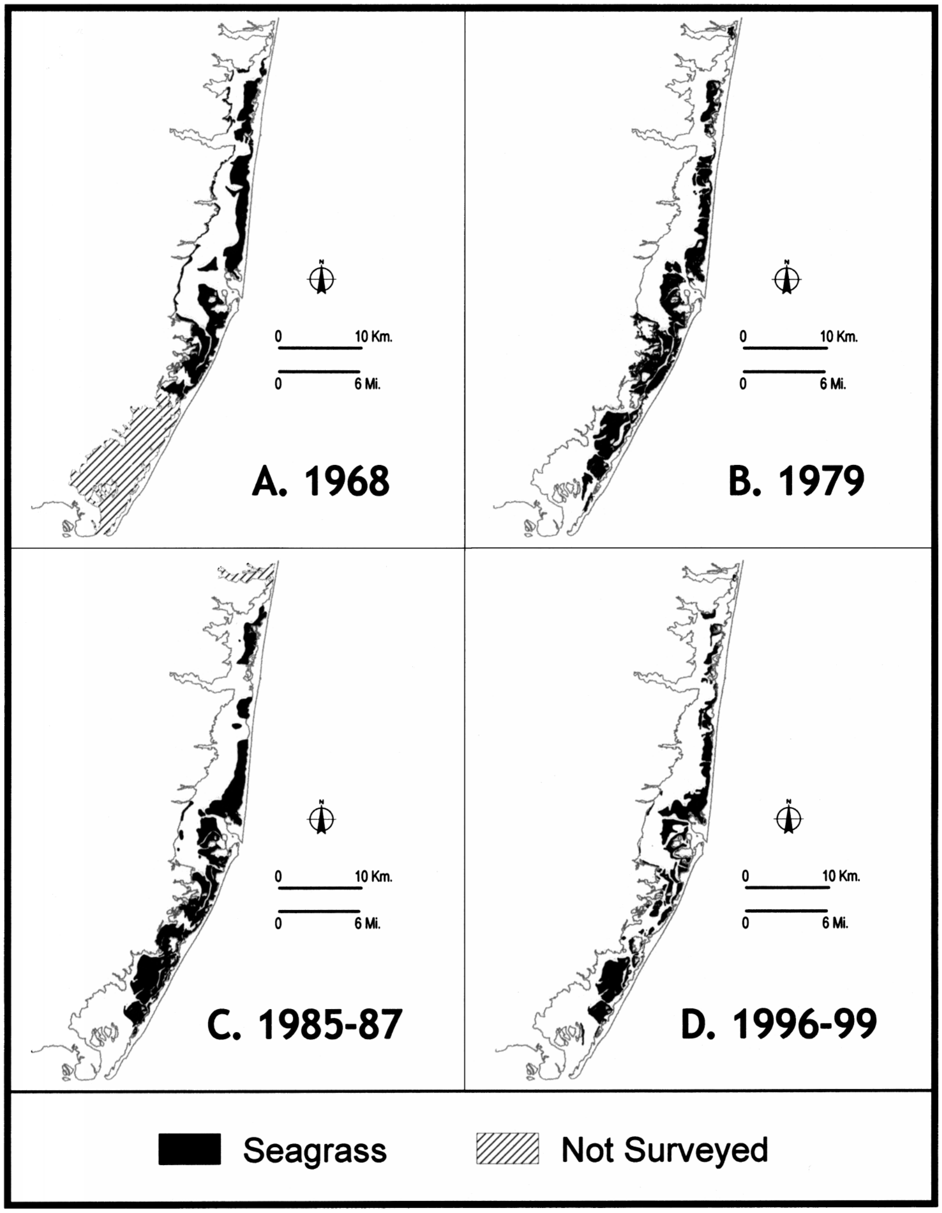

Fig. 3. Map of seagrass distribution for Barnegat Bay and Little Egg Harbor over the past four decades. Sources: 1968 map (U.S. Army Corps of Engineers 1976), 1979 map (Macomber and Allen 1979), 1985–1987 (Joseph et al. 1992), and 1996–1999 map (McClain and McHale 1996; Bologna et al. 2000).

1996; Bologna et al. 2000) was a boat-based survey that traced the outer boundaries of individual beds. These surveys allow us to examine general temporal and spatial trends in seagrass distributions in Barnegat Bay.

The 1968 survey (U.S. Army Corps of Engineers 1976) mapped approximately 6,800 ha of seagrasses in the north and central portions of Barnegat Bay (the survey excluded southern Barnegat Bay-Little Egg Harbor) (Fig. 3a). The 1970s SAV survey (Macomber and Allen 1979) mapped 8,053 ha as dominated (> 80%) by either *Z. marina* or *R. maritima* in the main body of Barnegat Bay (excluding associated tidal creeks and subtidal ponds; Fig. 3b). The 1980s survey (Joseph et al. 1992) mapped approximately 8,799 ha as eelgrass-dominated SAV beds (Fig. 3c). The combined 1990s survey (McLain and McHale 1996; Bologna et al. 2000) mapped only 6,083 ha of eelgrass or widgeongrass-dominated SAV bed (Fig. 3d). Table 1 shows the results of all four mapped surveys broken down by zone (see Fig. 1). These geographic zones were

TABLE 1. Comparison of mapped seagrass area (ha) for the four time periods.

Bay Zone	1968	1979	1985–1987	1996–1999
1	1,289	767	723*	437
2	5,536	5,126	5,340	3,700
3	No data	2,160	2,736	1,946
Total	6,825**	8,053	8,799	6,083

* Does not include the Metedeconk River portion of the bay.
** The 1968 survey did not map Zone 3.

chosen to represent distinct regions within Barnegat Bay-Little Egg Harbor as well as the gradient of watershed development (i.e., Zone 1 has the highest (35%) and Zones 2 and 3 have the lowest (12%) percentage of developed land in the upland watershed; Lathrop and Bognar 2001). A spatial comparison of the changes between the 1970s and 1980s surveys reveals minor shifts in the spatial distribution that might be due to real changes in distribution or purely artifacts of differences in the survey and mapping methodologies. While the spatial distribution of many seagrass beds has remained relatively stable over the time period, comparison of the 1970s and 1980s maps with the 1990s survey shows an overall decrease of over 2,000 ha in total seagrass area (Table 1).

The 1990s surveys suggest that there has been a loss of seagrasses in the deeper waters of the bay, resulting in the contraction of the beds to the shallower subtidal flats (< 1 m depth) between the 1970s–1980s and the 1990s. In the northern portion of the bay (north of Toms River) the outright loss of many beds does appear by the 1990s survey. Due to the great difference in mapping methods, we must be cautious in directly attributing the decrease in eelgrass acreage to a large-scale dieback of eelgrass. This is especially true at the deep water edge of seagrass beds where accurate mapping becomes problematic and the ability to conclusively detect change becomes difficult. While we can not conclusively establish that there has been a major dieback and loss of eelgrass acreage, there is reason for concern over the status of eelgrass beds in Barnegat Bay. To gain further insight into the spatial and temporal patterns of seagrasses in the bay, the degree to which water transparency can explain the current distribution of seagrasses in the bay was explored.

Comparison of Predicted and Mapped Distribution of Seagrasses

Areas of potential seagrass habitat were mapped using Eq. 3 under both spring and summer water column turbidity conditions and compared to mapped distribution of seagrass during summer in the bay (Fig. 4). The model correctly predicts seagrass presence-absence over two-thirds of the time (Table 2). The summer model showed a slightly higher overall accuracy than that obtained for the spring model, 68% versus 66%, respectively. Due to lower water transparency (i.e., observed lower Secchi depths) during the summer months, the summer model is more restrictive predicting 9,573 ha as compared to 12,673 ha for the spring model, a decrease of 25%. The lower water transparency during the summer months appears to be more of a constraining factor than water transparency during the spring. The higher omission error (i.e., model does not predict seagrass but seagrass was observed) during the summer (Table 2) suggests that eelgrass might become established in areas with a suitable light environment during the spring that then become marginal during the summer.

The majority of the model error was due to errors of commission (i.e., the model predicts that the water transparency-light environment was suitable but no seagrass was found). The model overestimates the total amount of seagrass by over 55% (predicted = 9,573 ha versus observed = 6,083 ha) using the summer model (Table 2). If the omission error was greater, it might suggest that the light extinction model was too conservative (i.e., the model does not predict seagrass occurrence where it was observed to occur and thus underestimates eelgrass occurrence). The Kappa statistic for the spring and summer models was 0.31 and 0.26, respectively. This represents a comparatively weak fit, i.e., a Kappa statistic of 0.30 implies that the model was avoiding only 30% of the errors that a completely random assignment would generate (Congalton 1991).

Discrepancies between the observed and modeled seagrass distribution may be due to a number of factors in both the mapping and modeling components including limitations in field methods, errors associated with the seagrass surveys and mapping, difficulties in the scaling of point measurements to 2-dimensional mapped surfaces at the scale of the entire bay including the interpolation and the selection of a grid cell size of 1 ha as the areal unit of analysis, the use of mean low water depth for bathymetry, and the exclusion of other factors that may be affecting the distribution of seagrasses (e.g., sediment type, storm events, human disturbance, nutrient limitations, etc.). In addition, a modification of Barnegat Inlet (in Zone 2, see Fig. 1) by the Army Corps of Engineers in 1990–1992 resulted in a slight change in the tidal amplitude of Barnegat Bay since the bathymetry data used in this study was charted. The mean tidal range increased by approximately 6 cm in the central and northern portions of Barnegat Bay (Kennish personal communication) over a background

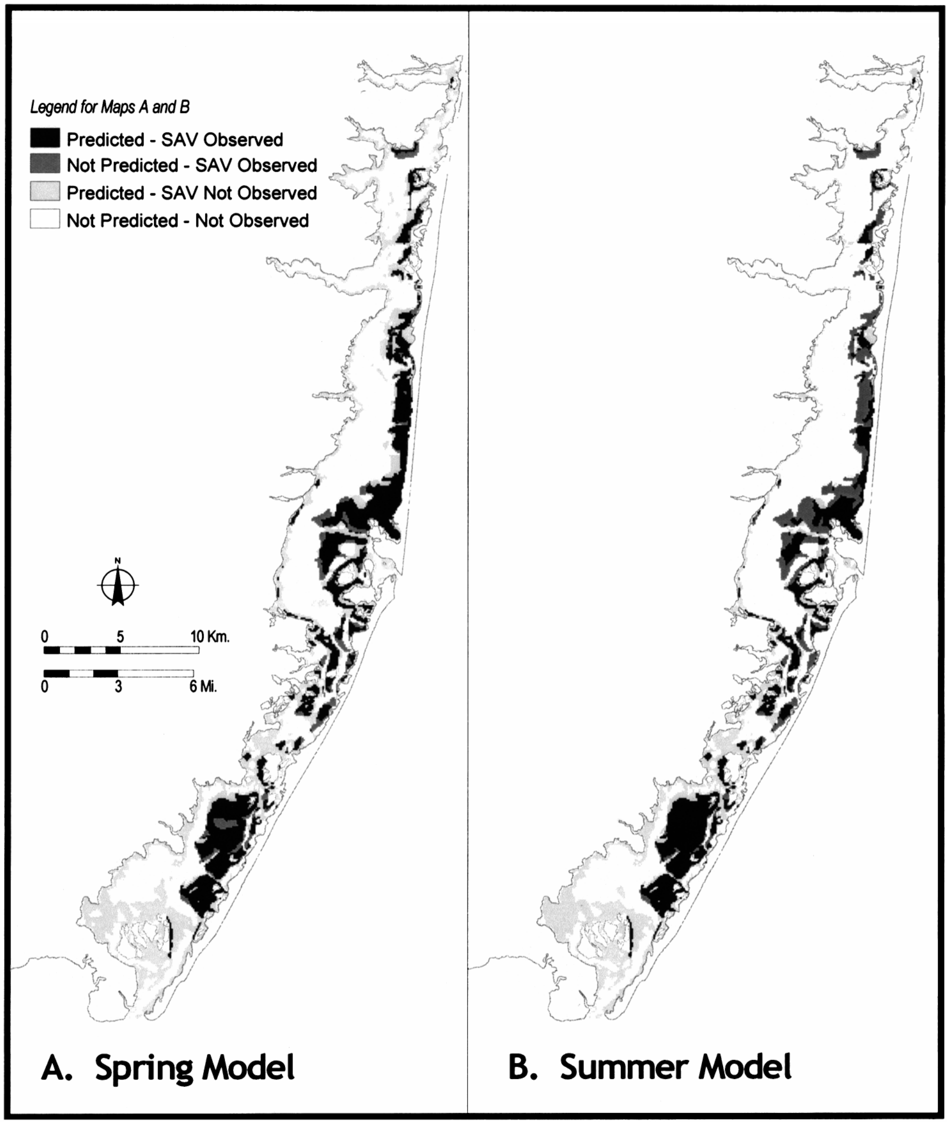

Fig. 4. Map of the predicted versus observed distribution of seagrass for the 1990s comparisons. A) 1990s versus Spring Model. B) 1990s versus Summer Model.

mean tidal range of 15 cm (Chizmadia et al. 1984). The extent to which this increase in tidal amplitude has affected seagrass distributions, especially in areas adjacent to the inlet, is uncertain.

Visual examination of the predicted versus observed distribution for the 1990s (Fig. 4) shows a significant amount of the commission error is accounted for by two areas: the shallow shoreline areas fringing the western shoreline of the bay and the extreme southern end of the study area. These two areas of the bay together account for over half of the commission errors. Over one-quarter of the commission errors in the 1990s comparison (29.9% for the spring, 26.9% for the summer model) are due to the shallow shoreline areas fringing the mainland shoreline of the bay. This commission error may occur because the model correctly predicts eelgrass occurrence but the mapped reference data is inadequate, or because the model correctly predicts an adequate light environment, but other factors such as bottom substrate are limiting eelgrass presence. While eelgrass is known to occur in sparse amounts (e.g., in patches with only 30% cover) along this mainland shoreline (Macomber and Allen 1979), the 1990s seagrass survey concentrated on mapping only the larger, high percent cover beds in the central and eastern portions of the bay and less effort was expended in surveying the bay's western shoreline (McClain and McHale 1996). The bottom substrate along

TABLE 2. Comparison of predicted versus observed distributions (in ha) of eelgrass for the 1990s survey (area in ha). For spring: * % correct = 17,194/26,070 = 66.0%, ** Error of commission = 7,733/12,673 = 61.0%, *** Error of omission = 1,143/6,083 = 18.8%, and Kappa statistic = 0.31. For summer: * % correct = 17,848/26,070 = 68.5%, ** Error of commission = 5,856/9,573 = 61.2%, *** Error of omission = 2,366/6,083 = 38.9%, and Kappa statistic = 0.26.

Observed	Predicted Seagrass Absent	Predicted Seagrass Present	Total
Spring Model			
Seagrass Absent	12,254*	7,733**	19,987
Seagrass Present	1,143***	4,940*	6,083
Total	13,397	12,673	26,070
Summer Model			
Seagrass Absent	14,131*	5,856**	19,987
Seagrass Present	2,366***	3,717*	6,083
Total	16,497	9,573	26,070

these mainland shoreline areas may not be suitable for eelgrass in many areas. Sediments along the barrier island side of the bay are primarily fine to medium sands, with muddy sand and silt-clay sediments in the deeper regions and along the mainland side of the bay (Chizmadia et al. 1984). The only places where extensive eelgrass beds were documented to occur were westward of Barnegat Inlet, where the substrate has a higher sand component.

An additional area of major disagreement for both the 1990s comparison was identified in the extreme southern end of the study area in the southern third of Little Egg Harbor (Fig. 4). Over one-quarter of the commission errors in the 1990s comparison (26.6% for the spring, 34.4% for the summer model) are due to this area. If this area is excluded, the percent correct increases for the remainder of the bay to 69.5% and 72.2% (Kappa statistic = 0.37 and 0.35) for the spring and summer models, respectively. Although there appears to be an adequate light environment, this area has not supported extensive eelgrass beds during the 1970s, 1980s, or 1990s. It is interesting to note that Great Bay, the next estuary south, does not support eelgrass either. These two estuarine systems are closely connected with water flow as they share the same ocean inlet, Little Egg Inlet, as well as numerous tidal creek connectors. It is unclear what is limiting the eelgrass distribution in this region.

Though the Secchi disk model is relatively simplistic, it does explain over two-thirds of the variability in the spatial distribution of seagrass. While this does not conclusively prove that water transparency is the causal factor, it does show a strong association. The majority of the model error is due to errors of commission, i.e., the model predicts seagrass occurrence where it was not observed to occur. Most of this commission error is located in specific geographic areas (i.e., southern third of Little Egg Harbor and the western shoreline of the bay). The model is useful in highlighting areas of disagreement between the predicted and observed distributions where factors other than water transparency may be a controlling influence.

Evaluation of Alternative Water Column Turbidity Scenarios

Water column turbidity is generally higher in the northern bay (Zone 1, mean Secchi depth is 0.6 m) than the southern bay (Zones 2 and 3, mean Secchi depth 0.8 m) during summer when phytoplankton biomass is highest (Styles et al. 2001). This pattern of higher water turbidity in the northern bay follows the pattern of greater watershed development (Zone 1 watershed is 35% developed as compared to 12% developed for Zones 2 and 3 watershed; Lathrop and Bognar 2001) as well as greater nutrient loading (Moser 1997; Hunchak-Kariouk et al. 1999) in the northern bay. To illustrate the possible ramifications of changes in watershed development on seagrass distribution, we used the above model to examine the change in seagrass distributions under two alternative water quality scenarios.

In the first scenario, the summer Secchi depths north of Toms River (Zone 1) were increased by 0.2 m. This scenario was designed to simulate a change in land use/human activities in the northern bay watershed (Zone 1) that would decrease water column turbidity in the northern bay (current average Secchi depth 0.6 m) to levels that are similar to the current southern bay (Zones 2 and 3; average Secchi depth 0.8 m; Fig. 1). In the second scenario, we increased the water column turbidity throughout the bay by applying a Secchi depth of 0.4 m to all regions. These Secchi depths are 0.2 and 0.4 m lower (on average) than current summer levels in the northern bay and southern bay, respectively. This decreased the Secchi depths in the whole bay to the lowest levels currently reported in the northern bay during summer. A Secchi depth of 0.4 m is similar to those reported for summer in Rehoboth Bay, Delaware (one of the Delaware Inland bays) (Lacouture and Sellner 1988), a region that has experienced total loss of eelgrass (Orth and Moore 1988).

Under Scenario 1 with improved summer water transparency in Zone 1, the model estimates an increase in seagrass area, from 9,573 to 12,900 ha (Table 3). Visual analysis of the model results (Fig. 5a) shows a spatial distribution in the northern bay similar to that expected under existing spring season conditions (Fig. 4a). The declining water quality of Scenario 2 (baywide decrease of summer Secchi depth to 0.4 m) results in a major decrease in predicted seagrass area to 4,580 ha (Table 3). This

TABLE 3. Area estimates of seagrass distribution by zone for the spring, summer, and scenario 1 and 2 models, along with the observed distribution (based on summer data) during 1990s surveys (area in ha). Analysis grid cell size = 1 ha.

Bay Zone	Spring Model	Summer Model	Scenario 1	Scenario 2	Observed
1	2,101	1,009	2,539	736	437
2	5,889	3,691	5,443	1,966	3,700
3	4,680	4,873	4,918	1,878	1,946
Total	12,673	9,573	12,900	4,580	6,083

equates to a decrease of over 50% compared to the prediction under present conditions (9,573 ha) and a decrease of 25% over that actually mapped in the 1990s surveys. Visual analysis of the model results (Fig. 5b) shows that the northern half of Barnegat Bay (north of Barnegat Inlet) would be largely devoid of seagrass and the expansive beds in the southern bay (central and northern Little Egg Harbor) would be drastically reduced in extent.

Conclusions

Comparison of mapped surveys of seagrass distribution over the past four decades indicates that the spatial distribution of many seagrass beds has remained relatively stable over the time period. The most recent surveys in the mid-late 1990s, show loss and diminution of seagrass beds in some areas of the bay. While comparison of the 1970s

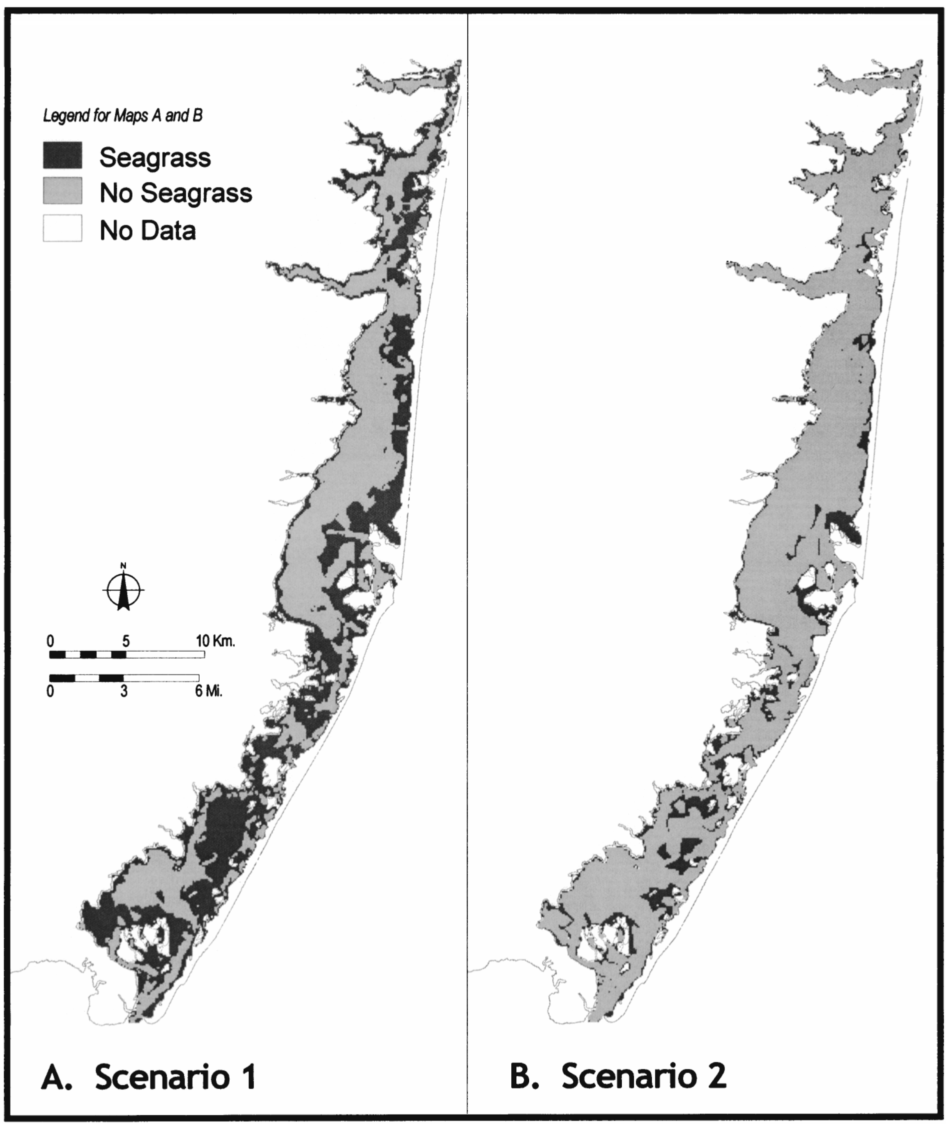

Fig. 5. Map of the predicted seagrass distribution under different water quality scenarios. A) Scenario 1 of increased water transparency in Zone 1. B) Scenario 2 of decreased water transparency to a Secchi depth of 0.4 m.

and 1980s maps with the 1990s survey shows an overall decrease of over 2,000 ha in total seagrass area, we are cautious in directly attributing the decrease in mapped eelgrass acreage to a large-scale dieback due to the great difference in mapping methods used for each of the surveys. To conclusively demonstrate changes in seagrass distribution requires well documented, consistent mapping techniques such as the photogrammetric protocols advocated by the NOAA Coastal Change Analysis program (Dobson et al. 1995) repeated over time.

Increased water column turbidity is thought to be a major contributor to observed declines in seagrasses worldwide (Short and Wyllie-Echevarria 1996). To examine the influence of water transparency on the spatial distribution of seagrasses in Barnegat Bay, we combined data on Secchi depth throughout the bay with an existing model of the relationship between *Z. marina* compensation depths and light attenuation coefficients to predict the presence-absence of seagrass. Our objective was to develop a simple but robust model that could be feasibly applied across an entire estuary area. When compared with mapped seagrass distribution in the bay, the model correctly predicts seagrass presence-absence over two-thirds of the time. The Secchi depth-based model tends to overestimate the observed presence of seagrass, especially in two subareas of the bay: the extreme southern end of the bay and the shallow shoreline areas fringing the mainland shoreline of the bay. These areas of commission error serve a useful purpose in raising questions as to why the model doesn't fit well in these locations. What factors other than water transparency are limiting seagrass in these areas of the bay?

There is a large and expanding literature on seagrass ecology. However, it is ''dominated by descriptive research (> 60% of papers), with a paucity of efforts to synthesis results and derive general relationships'' resulting in ''a present lack of predictive ability, and scientific basis for the management of seagrass ecosystems'' (Duarte 1999, p. 7). In the current study we used a modified version of a model that was developed using data collected globally (Duarte 1991) and combined it with local environmental data using a GIS approach. While we recognize that the Secchi depth-based seagrass model is comparatively simplistic, we feel it has value as a management tool to evaluate the potential changes in the spatial distribution of seagrass under different water quality scenarios. The spatially distributed GIS approach allows us to account for the spatial heterogeneity of environmental conditions across the bay and predict the local presence or absence of seagrass. Based on our model, we predict that a decrease in water transparency to match levels found in the Delaware Inland Bays (Scenario 2 above) would decrease the spatial distribution of seagrass by over 50% of that predicted under present conditions and by 25% over that actually mapped in the latest 1990s surveys. The few field studies that have been conducted indicate that due to recurring disease, nuisance algal blooms, and periphyton infestations leading to seasonal dieback (McClain and McHale 1996; Bologna et al. 2000), the long-term status of eelgrass beds in Barnegat Bay is cause for concern. The extent to which environmental conditions may control these contributing factors to eelgrass decline in Barnegat Bay (i.e., thereby facilitate modeling) is presently unknown. While a number of factors may be affecting the spatial distribution and health of seagrasses in Barnegat Bay, continued maintenance of existing water transparency levels, at a minimum, is critical to the long-term sustainability of these vital seagrass habitats.

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