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LINKING WATER QUALITY TO LIVING RESOURCES IN A MID-ATLANTIC LAGOON SYSTEM, USA

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Abstract. The mid-Atlantic coastal bays are shallow coastal lagoons, separated from the Atlantic Ocean by barrier sand islands with oceanic exchanges restricted to narrow inlets. The relatively poor flushing of these lagoon systems makes them susceptible to eutrophication resulting from anthropogenic nutrient loadings. An intensive water quality and seagrass monitoring program was initiated to track ecological changes in the Maryland and Virginia coastal bays. The purpose of this study was to analyze existing monitoring data to determine status and trends in eutrophication and to determine any associations between water quality and living resources. Analysis of monitoring program data revealed several trends: (1) decadal decreases in nutrient and chlorophyll concentrations, followed by recently increasing trends; (2) decadal increases in seagrass coverage, followed by a recent period of no change; (3) blooms of macroalgae and brown tide microalgae; and (4) exceedance of water quality thresholds: chlorophyll *a* (15 μg/L), total nitrogen (0.65 mg/L or 46 μmol/L), total phosphorus (0.037 mg/L or 1.2 μmol/L), and dissolved oxygen (5 mg/L) in many areas within the Maryland coastal bays. The water quality thresholds were based on habitat requirements for living resources (seagrass and fish) and used to calculate a water quality index, which was used to compare the bay segments. Strong gradients in water quality were correlated to changes in seagrass coverage between segments. These factors indicate that these coastal bays are in a state of transition, with a suite of metrics indicating degrading conditions. Continued monitoring and intensified management will be required to avert exacerbation of the observed eutrophication trends. Coastal lagoons worldwide are experiencing similar degrading trends due to increasing human pressures, and assessing status and trends relative to biologically relevant thresholds can assist in determining monitoring and management priorities and goals.

Key words: chlorophyll; coastal bay; coastal lagoon; eutrophication; Maryland, USA; mid-Atlantic; nutrients; seagrass; Virginia, USA; water quality.

Introduction

Coastal lagoons worldwide, including the U.S. mid-Atlantic bays, are currently threatened by anthropogenic inputs of nutrients. Like other lagoon systems, the coastal embayments of Maryland and Virginia, USA, are highly susceptible to nutrient over-enrichment. These are seagrass-dominated coastal lagoons (Wazniak et al. 2005*a*, Orth et al. 2006), and due to the high light requirements of seagrass (Dennison et al. 1993), such systems are more rapidly impacted by nutrient addition than those dominated by other primary producers, such as phytoplankton (NRC 2000, Cloern 2001). The coastal bays also have low tidal flushing and limited freshwater inflow (Pritchard 1960, Lung 1994, Boynton et al. 1996), and they have been classified by a nationwide survey as highly susceptible to eutrophication (Bricker et al. 1999). Due to these factors, the Maryland Coastal Bays National Estuary Program, MCBP, identified nutrient over-enrichment as the primary threat to the Maryland coastal bays (MCBP 1999).

Several signs of ecosystem stress related to nutrient over-enrichment have become evident in the coastal bays. A recent study indicated elevated macroalgae biomass (Goshorn et al. 2001) in areas coinciding with seagrass increases, which provides some evidence for a transition to a more eutrophic state (Fourqurean and Rutten 2003). Another indicator of stress is an annual bloom of *Aureococcus anophagefferens*, the phytoplankton species that causes brown tide, especially in the southern coastal bays (Trice et al. 2004, Wazniak and Glibert 2004), as well as the presence of other harmful algal bloom species (Tango et al. 2005). These shallow, wind-dominated bays are not known to stratify, but seasonal hypoxic events do occur in localized areas

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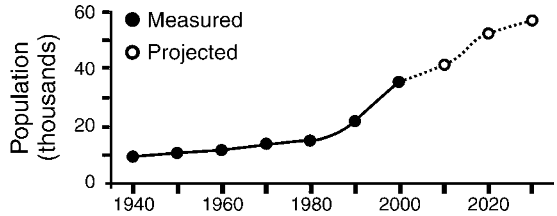

FIG. 1. Decadal human population rise in the Maryland coastal-bays watershed (Hager 1996).

(Maryland DNR 2002, Hall and Wazniak 2005; B. Sturgis, *unpublished data*). Even though there has been extensive expansion of seagrass acreage over the past three decades, this trend has leveled during recent years (Orth et al. 2006). In relation to this, seagrass-dependent bay scallops (*Argopecten irradians*), while present, are only found in low densities, suggesting the long-term viability of the population is in question (Tarnowski 2005). These observations suggest that these coastal bays are undergoing ecosystem change consistent with increasing anthropogenic nutrient addition.

Like many coastal areas, the local economy of the coastal bays is linked to the ecological health of the estuaries. As population increases, dependence on the estuary to support economic activities such as fisheries and tourism also increases. The coastal bays' watershed population has doubled between 1980 and 2000 and is expected to nearly double again by 2020 (Fig. 1; Hager 1996). Additionally, $>11 \times 10^6$ tourists visit the region annually, and tourist numbers are also projected to grow significantly (MCBP 2005). These population increases could exacerbate the biological impairment caused by anthropogenic disturbance, including nutrient enrichment, physical alteration, introduction of toxins, and changes in overall biotic structure (Day et al. 1989). Additionally, anthropogenic changes in hydrodynamics (through the long-term stabilization of inlets, dredging, and development on the barrier islands themselves) have altered the natural resilience of many of these systems. These factors may contribute to the deteriorating conditions of bay resources on which the economy depends and make attaining management goals more difficult.

The Comprehensive Conservation Management Plan for the Coastal Bays included decreasing nutrients, increasing seagrass populations, and maintaining viable fishery populations as key management goals (MCBP 1999). Hence, effectively assessing the health of the system in a relevant ecological framework and determining change over time among ecosystem components was needed to guide management decisions. However, defining and assessing the eutrophic status or health of lagoon systems, and coastal systems as a whole, continues to prove challenging to ecosystem managers. Current national assessments seek to compare a diversity of coastal systems using the same criteria (Bricker et al. 2003, U.S. Environmental Protection Agency [EPA] 2004). Furthermore, the application of "threshold risk levels" needs to be verified for each type of estuarine system.

Ensuring that metrics are chosen that can be linked to causes, allow future predictions, and are sensitive enough to assess changes resulting from management actions is essential (Suter 1993). Additionally, whether different groups of metrics (biotic, water quality, physical) should be combined (Ferreira 2000, Kiddon et al. 2003) and whether these combined indices should be weighted or not (Borja et al. 2004) are important factors to consider. Lastly, determining whether the combination of chosen parameters is related to indirect ecosystem responses (Phase II conceptual model of Cloern [2001]) will increase our understanding of different estuarine types.

To clarify these issues, the current paper addresses three key questions: Are conditions in the coastal bays suitable for key living resources? Are conditions improving or degrading? Are differences in water quality associated with status and trends in living resources?

METHODS

Site description

The coastal embayments of Maryland and Virginia, USA, are temperate systems located on the Delmarva Peninsula (the coastal plain peninsula between the Chesapeake Bay and the Atlantic Ocean; Fig. 2). These

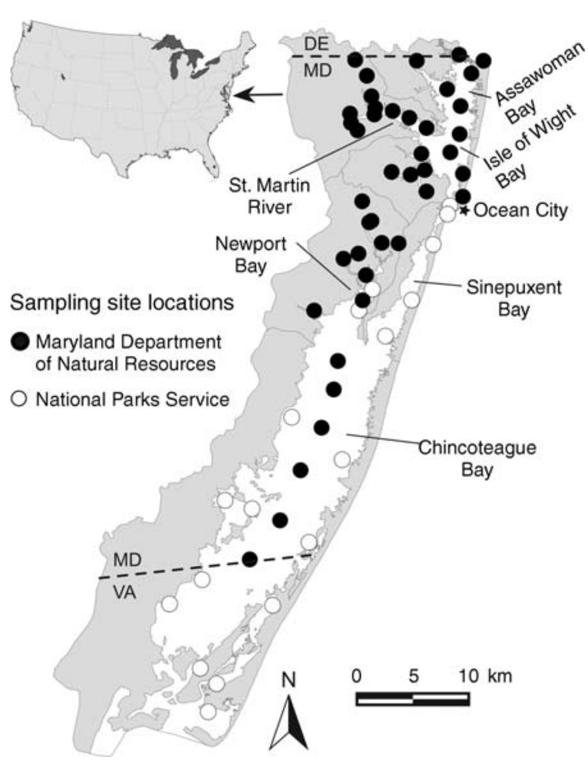

FIG. 2. General location of the Maryland coastal bays along the east coast of the United States. The watershed area of each of the bay segments and the water quality monitoring sites are shown.

coastal bays comprise a series of estuarine lagoons that were divided into six segments (i.e., subwatersheds), based on major drainage and inlet characteristics, for this analysis (Fig. 2). The total watershed is small (452 km^2) with a watershed-to-water-surface-area ratio of 1.6. Due to relatively low and constant elevation as well as sandy soils, groundwater is the major pathway of freshwater input to the bays, with the St. Martin River providing the largest stream input (Lung 1994). Salinities range from polyhaline (30–35‰) in the open bays to fresh in some tributaries. Circulation is wind- and tide-dominated: tidal exchange with the Atlantic Ocean is limited to two inlets, one at Ocean City and the other at the south of Chincoteague Bay (Fig. 2). Flushing is slow compared with other estuaries (e.g., Chincoteague flushing rate is 62 days), causing protracted residence times for contaminants, including nutrients (Pritchard 1960, Bricker et al. 1999). Nutrient loads to the coastal bays are dominated by nonpoint sources (e.g., surface runoff, groundwater, atmospheric deposition, and shoreline erosion) (Boynton et al. 1996, Wells et al. 2004) with recent estimates suggesting one-half to two-thirds of nutrients entering the bays come from agricultural sources, the dominant land use in the area (Boynton et al. 1996).

Data collection and sample analysis

The National Park Service (NPS) at Assateague Island National Seashore and the Maryland Department of Natural Resources (DNR) water quality monitoring programs conducted routine monthly sampling and provided the data used in this paper. The NPS measured water quality parameters monthly at 18 stations in the southern coastal bays since 1987 or 1991, depending on the station (Fig. 2). The NPS sites were not monitored over winter months during the first five years. The DNR measured water quality monthly at 28 sites in the St. Martin River, Isle of Wight Bay, and Newport Bay segments since 1999 and 17 sites in Assawoman Bay, Isle of Wight Bay, and Chincoteague Bay since 2001 (Fig. 2). All stations were tidal, except for five DNR stations, and all were monitored in accordance with U.S. EPA-approved quality assurance plans (CBP 2001, Sturgis 2001) and in conjunction with the Maryland Coastal Bays Program's Eutrophication Monitoring Plan (Wazniak 1999).

Both programs recorded on-site water quality parameters (including dissolved oxygen [DO]) and collected samples to send to laboratories for nutrient and chlorophyll *a* analyses. Whole water samples were collected just below the surface and stored on ice and in the dark until being filtered (less than four hours). The DNR samples were analyzed by the University of Maryland Center for Environmental Science (UMCES) Chesapeake Biological Laboratory for total dissolved nitrogen (D'Elia et al. 1977, Valderrama 1981), particulate nitrogen (U.S. EPA 1997), total dissolved phosphorus (Valderrama 1981), and particulate phosphorus (Aspila et al. 1976). Chlorophyll *a* concentration was determined by the Maryland Department of Health and Mental Hygiene (APHA 1985). Total nitrogen (TN) and total phosphorus (TP) were calculated by adding the total dissolved fractions to the particulate fractions. The NPS samples were analyzed by UMCES Horn Point Laboratory for TN and TP, using persulfate digestion (Valderrama 1981), and chlorophyll *a*, using high-performance liquid chromatography (Van Heukelem et al. 1994, Van Heukelem and Thomas 2001). All three laboratories were evaluated as part of the Chesapeake Bay Program quality assurance protocol and found to not differ significantly (Zimmerman and Keefe 2004).

Seagrass coverage

Annual black-and-white aerial photography, 1986–2003, was obtained by the Virginia Institute of Marine Sciences (VIMS) at a scale of 1:24 000 from three flight lines and examined to identify all visible seagrass beds (Orth et al. 2004). Ground survey information was also collected by VIMS, tabulated, and entered into the seagrass geographic information system (GIS). This provided a digital database for analysis of bed areas and locations. Current seagrass habitat is defined as a composite of the past three years (2001–2003) of observed seagrass coverage. Potential seagrass habitat was identified, in the Maryland portion only due to data limitations, as being <1.5 m in mean water depth (based on a composite seagrass layer showing 93% of all seagrasses digitized between 1986–2003 were ≤ 1.3 m deep) and having a silt/clay content of $<35\%$ (E. M. Koch, *unpublished data*).

Determination of biologically relevant thresholds

The current study used historically collected data to assess system health by summarizing four common water quality indicators (TN, TP, chlorophyll *a* [algae: chl *a*], and DO) and compared these to biologically relevant thresholds, established for maintenance of seagrass (Stevenson et al. 1993, Kemp et al. 2004) and fish communities (e.g., Breitburg 2002). The use of nutrients, phytoplankton, and dissolved oxygen is common in evaluating estuarine systems (Bricker et al. 1999, Kiddon et al. 2003, Jones et al. 2004, U.S. EPA 2004) and, specifically, habitat quality for submerged aquatic vegetation (Valdes-Murtha 1997, Lea et al. 2003, Kemp et al. 2004). An evenly weighted water quality index of these parameters was used to assess eutrophic status. Water quality status was then related to the current distribution of seagrasses within the bays.

Since seagrasses and fisheries are important biological components of the coastal bays, they are the basis of management focus (MCBP 1999). Biologically relevant threshold values for variables used to assess eutrophication (Table 1) were set based on literature values for seagrass habitat requirements (Dennison et al. 1993, Stevenson et al. 1993, Valdes-Murtha 1997, Lea et al. 2003) as well as dissolved oxygen requirements for fish (Howell and Simpson 1994, Diaz and Solow 1999, Breitburg et al. 2001, Breitburg 2002) and benthic

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 March–November (SAV growing season). "STAC" stands for Scientific and Technical Advisory Committee.

communities (Baden et al. 1990, Pihl et al. 1991, 1992, Smith and Dauer 1994, Ritter and Montagna 1999) (Table 2). Five threshold categories were determined for status analyses while a single biologically relevant threshold was used to develop a water quality index to compare coastal-bay segments.

Measures of total nutrients (rather than dissolved inorganics) were used to reduce variability associated when measuring dissolved nutrients only. Additionally these coastal bays have been shown to have high regional organic nutrient concentrations (Glibert et al. 2001) that are at least partially bioavailable (Seitzinger and Sanders 1999, Seitzinger et al. 2002, Mulholland et al. 2004, Glibert et al. 2006; Wiegner et al. 2006; M. R. Mulholland, G. Boneillo, and E. C. Minor, *unpublished manuscript*). Therefore, total nutrients were determined to be better than dissolved inorganic nutrients as indicators of relative nutrient availability in this system that is known to have high organic inputs, as well as long residence times. Maryland State guidelines were used to set the borderline threshold at 5 mg/L DO (COMAR 1995), although the references above support this value for healthy fish and benthic communities in shallow-water ecosystems.

Water quality index assessment

An evenly weighted water quality index was developed to compare coastal-bay segments. A single, biologically relevant threshold value for each indicator was chosen from those used for status analysis and data condensed into a single index of water quality. Threshold values were: chlorophyll *a*, <15 μg/L; total nitrogen, <0.65 mg/L or <46 μmol/L; total phosphorus, <0.037 mg/L or <1.2 μmol/L; and dissolved oxygen, >5 mg/L (Dennison et al. 1993, Stevenson et al. 1993, Ritter and Montagna 1999, Breitburg 2002).

For the 60 sampling stations with at least 10 records for all variables between 2001 and 2003, median values for TN, TP, and chl *a* and a second percentile for DO were calculated. Calculated values were then compared to established threshold values and scored as one (met criteria) or zero (failed to meet criteria). These scores were summed for all four variables and divided by the number of variables to result in an index value ranging from zero to one for each station. Therefore, an index value of zero indicated that a station met none of the water quality criteria and would not be expected to support seagrasses or fisheries, while a score of one indicated a station met all water quality criteria and should support ecosystem services. Intermediate values indicated the system was variable and that some ecosystem functions (seagrass beds or fisheries) would be expected to be present periodically.

Once an index value was calculated for each sampling station, a mean of the individual index variables for all stations within each segment was calculated and these are presented by measured variable (Table 3) and combined segment index values. Standard errors associated with mean index values represent spatial variation between sites within a segment only and do not include temporal variability.

Statistical analyses

Three types of statistical analyses were performed on the data sets, including three-year water quality status, linear trends, and nonlinear trends. Status analysis compared three-year medians of water quality indicators to biologically relevant thresholds. This analysis provided a current assessment of eutrophication effects. To determine whether or not these effects were changing over time, linear seasonal trend analysis and nonlinear trend analyses that could detect trend reversals were performed on data collected over the past 13–17 years. Potential implications of these trends in water quality on seagrasses were also investigated.

For status analysis, chl *a* was assessed during the putative seagrass growing season of March through

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. Breakdown of water quality index variables by segment. Each variable shows the proportion of sites meeting threshold (mean with SE in parentheses).

Segment	Chlorophyll	Total N	Total P	Dissolved oxygen
Assawoman	0.33 (0.21)	0.00 (0.00)	0.00 (0.00)	0.83 (0.17)
St. Martin	0.36 (0.14)	0.00 (0.00)	0.00 (0.00)	0.09 (0.09)
Isle of Wight	0.89 (0.11)	0.33 (0.17)	0.00 (0.00)	0.33 (0.17)
Sinepuxent	1.00 (0.00)	1.00 (0.00)	0.40 (0.24)	0.80 (0.20)
Newport	0.43 (0.14)	0.08 (0.08)	0.08 (0.08)	0.42 (0.15)
North Chincoteague	1.00 (0.00)	0.64 (0.15)	0.18 (0.12)	0.73 (0.11)
South Chincoteague	1.00 (0.00)	1.00 (0.00)	0.33 (0.21)	1.00 (0.00)

November. This was decided because the two species present, *Zostera marina* and *Ruppia maritima*, overlap in growing seasons. *Zostera marina* grows during two seasons in a typical year (March through May and October through November), while *R. maritima* tends to grow during a single protracted season (April through October) (Moore et al. 2000, Kemp et al. 2004). Dissolved oxygen was assessed during the summer season (June through September) since most low DO events (sustained concentrations below 5 mg/L) tended to occur during these months (Funderburk et al. 1991, Wazniak et al. 2005*b*). Nutrient loading was a probable factor in high algal production and subsequent drops in DO in the coastal bays (Boynton et al. 1996). Since nutrient loading took place throughout the year (albeit at varying seasonal rates), TN and TP were evaluated based on the entire year. For each indicator at each station, medians were calculated for chl *a*, TN, and TP, and a lower second percentile was calculated for DO within the delineated seasons over three years (2001–2003). Medians were used for chl *a*, TN, and TP instead of means to insure that the highest and lowest monthly concentrations had smaller effects on central tendency. The second percentile was used for DO to most critically evaluate the available daytime data while excluding extreme values that may not be typical for the time period. Three years were chosen to minimize natural interannual variability (e.g., differences in flow). Each three-year station median was then tested against all five threshold values by a Wilcoxon rank sum test ($\alpha = 0.01$ to protect against Type I error), a nonparametric test of position. The five threshold values formed five threshold categories as depicted in Tables 1 and 2. If the Wilcoxon test was significant (the median was significantly different from all of the thresholds) then the median was said to be significant (there was confidence in which threshold category the median fell). If the Wilcoxon test was not significant, then the actual status of the median was not as certain (i.e., there was a question regarding in which of two adjacent threshold categories the median fell).

Trends, defined as the amount an indicator has changed over time, were calculated for chl *a*, TN, and TP. Dissolved oxygen was not tested for trend due to natural diel variability of concentrations and variation in the time of day that measurements were taken. Only stations with 10 years or longer of data collection were analyzed for trends, meaning only the 18 NPS stations south of the Ocean City inlet were used (Fig. 2). This data set was further trimmed to exclude winter months (November through March), as these were not uniformly sampled, especially in earlier years. Half of the 18 stations were sampled beginning in 1987 and the other half began sampling in 1991 (Fig. 6). The seasonal Kendall test was used to identify trends, and Sen's slope estimator was used to test the magnitude of change over time when a significant trend was present (Hirsch et al. 1982, VanBelle and Hughes 1984, Ebersole et al. 2002). The seasonal Kendall test was derived from Kendall's tau, a nonparametric correlation statistic, and was used to provide a distribution-free test of trend unaffected by seasonality (month effects over years, in this case) (Hirsch et al. 1982, Van Belle and Hughes 1984). The seasonal Kendall test for trend was considered significant at $\alpha < 0.01$. Confidence intervals were calculated around each Sen slope as a general retrospective test of power (Thomas 1997), though this method cannot directly evaluate study design.

Observations of raw data plots revealed potential reversals in trend direction in the water quality data. Quadratic (nonlinear) regression was used to evaluate these potential trends. Total nitrogen, TP, and chl *a* were each regressed using

$$\begin{aligned}&\log[\text{indicator(TN, etc.)}]\\&\quad= \beta_0 + \beta_1(\text{time}) + \beta_2(\text{time})^2\\&\qquad + (\beta_3 \cdots \beta_{10})(\text{individual months}) + \text{error}. \qquad (1)\end{aligned}$$

Month terms were included in the model to reduce seasonality effects. The data were first standardized to a centralized date. This caused the time and the time^2 terms to become orthogonal, reducing the chances of misinterpreting correlated regression terms (E. Perry, *personal communication*). Those stations with significant quadratic (time^2) terms (i.e., trend reversals) at $\alpha < 0.05$ were analyzed further. The direction of the trend after reversal determined whether concentrations were increasing (parabolic up) or decreasing (parabolic down). The magnitude of this post-reversal trend was evaluated by examining 95% confidence limits around the curve. If the ordinate value of the point at which the trend reversal occurred was encompassed by the confidence limits, the trend was significant. If this condition was not met, the trend was effectively asymptotic at the ordinate trend reversal value (inflection point). Confidence limits also served as an indirect test of retrospective power (Thomas 1997).

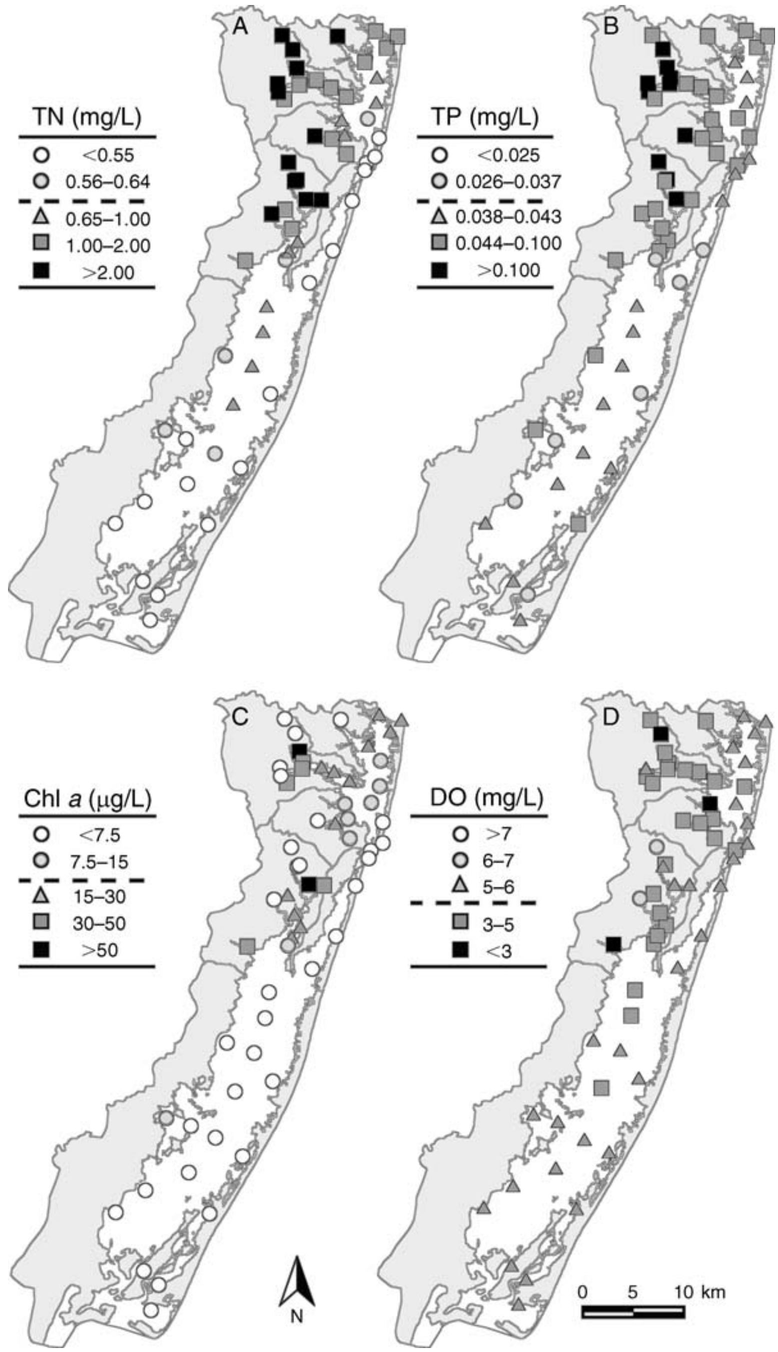

FIG. 3. Water quality status maps for 2001–2003: (A) TN, median total nitrogen (annual); (B) TP, median total phosphorus (annual); (C) Chl *a*, median chlorophyll *a* (March–November); and (D) DO, dissolved oxygen, second percentile (June–September).

RESULTS

Are conditions in the coastal bays suitable for key living resources?

Nutrient status.—The upper tributaries, mostly in the northern coastal bays, and Newport Bay were severely nitrogen enriched (Fig. 3A). This is expected since these areas have the most intense development and the largest streamflow and surface runoff inputs. Conversely, the southern coastal bays, Sinepuxent and Chincoteague, had the lowest TN concentrations. These areas have smaller watersheds, large wetland areas, and are groundwater-dominated. Overall, 61% of the stations failed the TN seagrass threshold. Phosphorus enrichment appeared to be even more widespread (Fig. 3B), with only the Sinepuxent and Chincoteague Bay

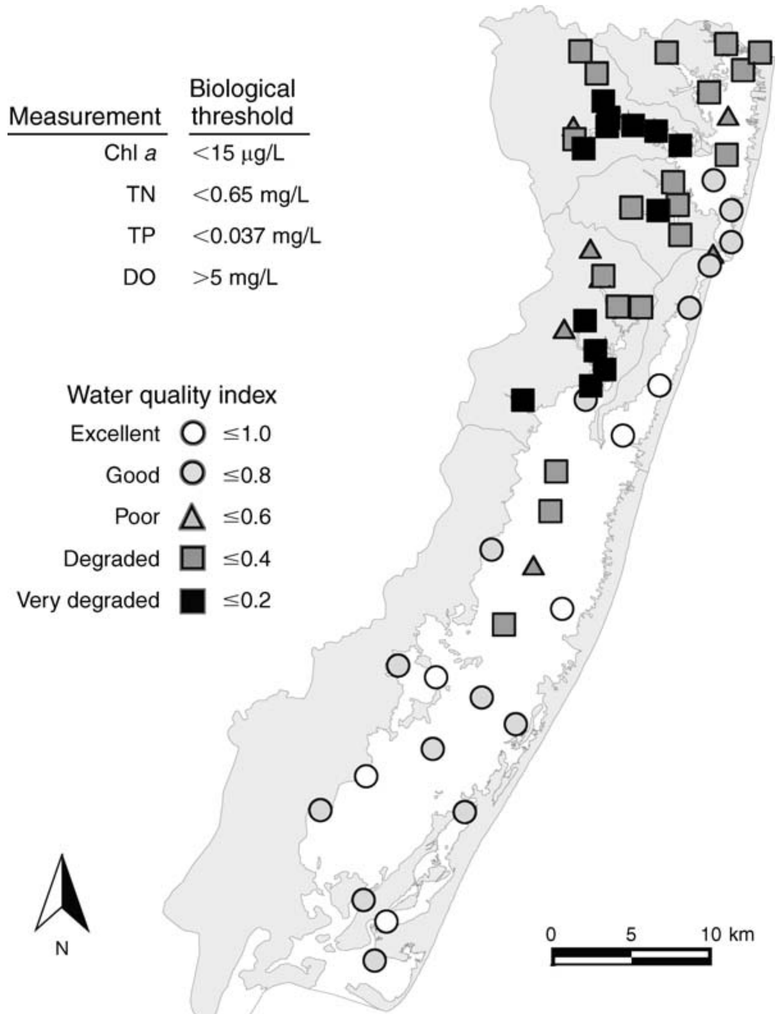

FIG. 4. Results of water quality index and biological thresholds used. The molar equivalents of the biological threshold for TN and TP are 46 mmol/L and 1.2 mmol/L, respectively.

segments maintaining phosphorus levels suitable for seagrass growth. Overall, 88% of the stations failed the TP seagrass threshold.

Chlorophyll a status.—Despite many areas failing nutrient thresholds, chl *a* concentrations were generally low in the open bays and also in the nontidal stations (Fig. 3C). The higher chlorophyll values that were observed in the main tributaries, where development and surface runoff are concentrated, was expected. The open bays are downstream of the turbidity maximum and may be light-limited from natural resuspension and, hence, have lower chlorophyll values. A total of 66% of the sites passed the chl *a* threshold for seagrass growth (three-year medians for the other 34% of the sites fell between 15.2 and 61.3 μg/L with 14% of sites having chlorophyll values >30 μg/L).

Dissolved oxygen status.—The coastal bays are shallow (<2 m) lagoons, which typically do not vertically stratify. Status analysis of the second percentile of DO over the three summer periods indicated that 40% of the stations did not meet summer oxygen objectives (Fig. 3D). Dissolved oxygen fell below 5 mg/L in the upper St. Martin River, most of Newport Bay, areas of Chincoteague Bay, and in tributaries to Isle of Wight and Assawoman Bays (Fig. 3D). Analysis of daytime minimum values revealed that 70% of stations did not meet summertime oxygen objectives. Areas that had <5 mg/L DO during the day may stress living organisms at night when respiration further reduces DO levels.

Water quality index.—Overall, the coastal bays had generally poor or degraded water quality in or close to tributaries and good or excellent water quality in better flushed, open-bay regions (Fig. 4). Water quality index values at upstream stations that rated better than downstream values were due to lower chl *a* values in fresh nontidal stations.

Segment summaries identified that Sinepuxent and south Chincoteague exhibited excellent water quality, north Chincoteague had good water quality, Isle of Wight had poor water quality, and Assawoman, St. Martin, and Newport all displayed degraded water quality (Fig. 5). Variations in water quality between segments reflect variation in nutrient concentrations (Fig. 5). However, many stations throughout the system displayed effects of high phytoplankton biomass and reduced dissolved oxygen.

Is water quality improving or declining?

Trends.—Linear trend results showed mostly improving conditions (17% of sites for TN, 50% for TP, and 33% for chl *a*) with just two stations having increasing trends (Fig. 6A). When the same 18 stations were tested for polynomial trends, 89% had significant quadratic trends (concave or U-shaped) in TN, 78% for TP, and 50% for chl *a* (Table 4, Fig. 6B). All of these significant quadratic trends were from a decreasing condition to either an increasing or not significant (asymptotic) condition. No stations were found to have quadratic trends going from increasing to decreasing conditions. For example, station 2 had significant quadratic trends in TN and TP, but not for chl *a* (Fig. 7). Further, only TN had a significant trend after the inflection point at this station (i.e., was currently increasing; Fig. 7A). Critical inflection points for all significant quadratic trends ranged between the years 1995 and 2000, which was apparently a time of vital change for the estuary.

Total nitrogen concentrations were found to be significantly decreasing at three stations and significantly increasing at one station over the sampling period, based on linear seasonal Kendall tau trend analyses (Fig. 6A). Conversely, 89% of stations had significant quadratic trends and 83% were significantly degrading (currently increasing TN) after their inflection point in TN.

Fifty percent of the stations sampled were found to have significantly decreasing linear TP trends, while 11% of stations had significantly increasing trends. These stations were also above the seagrass threshold for three-year median TP. Nonlinear trend results indicated 61% of stations were significantly increasing post-inflection point for TP (Fig. 6B).

Linear chl *a* concentrations were found to be significantly decreasing at one-third of the stations and significantly increasing at only one station (Fig. 6A). In contrast, one-third of the stations were significantly degrading post-inflection in the nonlinear analysis (Fig. 6B).

Several stations exhibited significant linear and nonlinear trends for individual parameters as a result of ending concentrations being significantly less than beginning concentrations (Fig. 8).

Are differences in water quality associated with status and trends in living resources?

Status and trends of seagrass coverage.—Throughout the coastal bays, seagrass abundance increased steadily since monitoring began, with an approximate threefold increase since 1986 (Fig. 9). However, these increases leveled over the last several years (Orth et al. 2004, 2006; Fig. 9). Currently, seagrass coverage is estimated to occupy 67% of the potential habitat in the Maryland portion of the bays with the greatest percentage of seagrass habitat occupied in Sinepuxent and Chincoteague Bays (Fig. 5). Between bay segments, a positive correlation was observed between seagrass cover (as a percentage of potential habitat occupied) and the synthetic water quality index (Fig. 10).

Discussion

Are conditions suitable for living resources?

Large areas of the coastal bays can be classified as eutrophic. This is most evident near the major tributaries, but even the relatively pristine open bay areas are showing signs of ecosystem change. Phosphorus enrichment is more widespread than nitrogen enrichment, but both macronutrients are above seagrass survival thresholds throughout large areas of the bays. Dissolved oxygen measurements also show clear indications of system degradation throughout, while only about half of the sampled sites had chl *a* concentrations below the threshold for seagrass survival. Although the coastal bays are shallow lagoons that typically do not stratify, oxygen values were frequently low in some areas. Observed low DO values may be due to the respiration of large algae blooms (caused by elevated nutrient levels), high sediment oxygen demand from organically enriched sediments in many areas (Wells and Conkwright 1999, Bailey et al. 2004), and the decay of phytoplankton, macroalgae, seagrasses, and/or marsh vegetation. These effects are all compounded by limited circulation within the system, especially at stations far removed from the two inlets.

Despite many areas exceeding nutrient thresholds in the coastal bays, chl *a* concentrations were generally low in the open bays. Probable reasons for this include that (1) the majority of the algal biomass (organic matter) produced in the tributaries is deposited within these areas (Wells and Conkwright 1999) or (2) nutrients are sequestered in or utilized by other forms such as benthic microalgae (microphytobenthos), benthic macroalgae, and seagrasses instead of water column phytoplankton. Although current chl *a* status has not surpassed threshold risk levels, degrading trends are an indicator they will be surpassed in time if nutrient reduction strategies are not implemented.

Water transparency is not thought to be a good response variable in shallow lagoon systems due to naturally high turbidity and temporal variability. In addition to the previously mentioned factors, coloration of water due to tannins in some parts of the coastal bays and generally limited data availability (monthly) has led to no agreement on light thresholds. Therefore, this was not included as a key variable.

The water quality index was used to rank the bays health from best to worst as follows: Sinepuxent Bay,

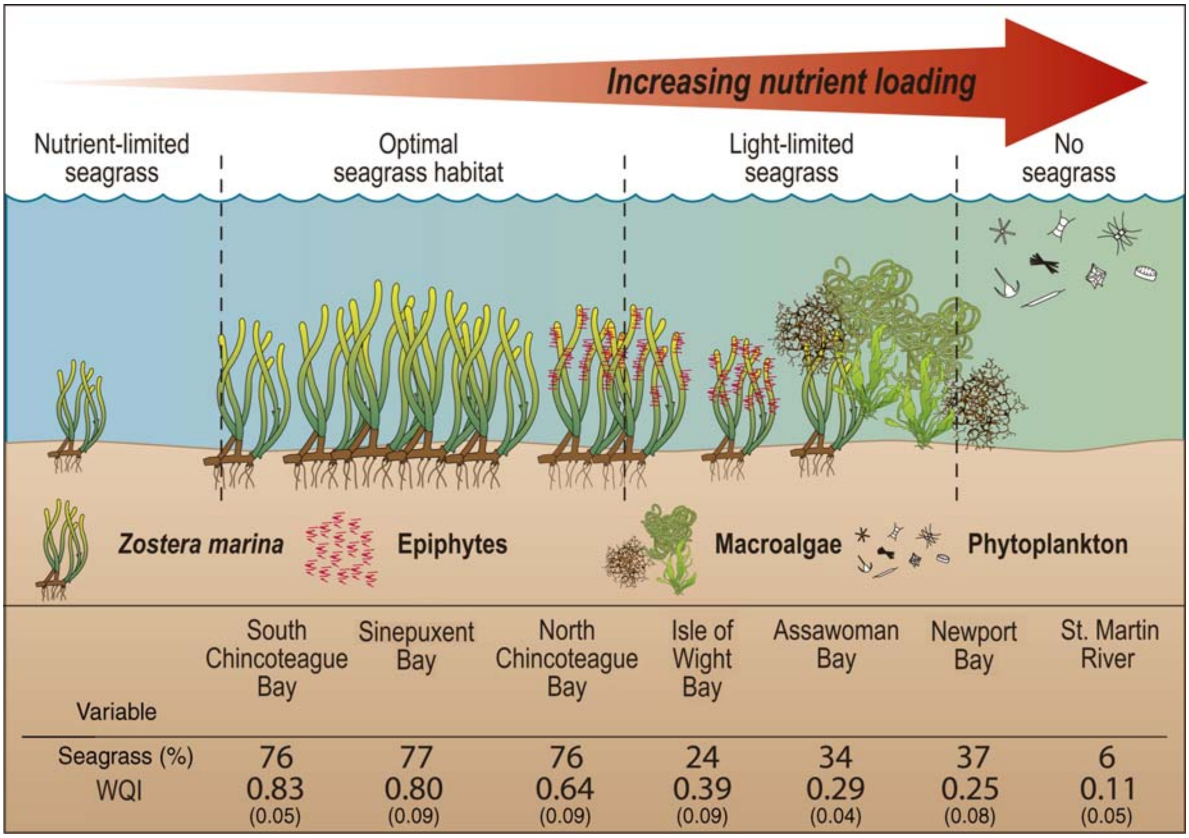

FIG. 5. Conceptual diagram of impacts of increasing nutrient loads. Water quality index (WQI) values are given as mean with SE in parentheses.

Chincoteague Bay, Isle of Wight Bay, Assawoman Bay, Newport Bay, and St. Martin River (Fig. 5). The index results indicated tributaries were generally poor to very degraded due to high nutrient inputs, while the open bays had good to excellent water quality. Regional differences in the index have implications for aquatic communities, suggesting that many regions within the coastal bays do not provide suitable habitat for seagrass and/or fish. The water quality index was useful in communicating scientific results to the public (Wazniak et al. 2004) and for comparing bay segments in relative terms and may be useful for tracking system changes over time. Nontidal stations that had better index scores than downstream sites were not truly representative of improved water quality in these areas. These stations were located above the chlorophyll maximum for streams, thereby meeting chl *a* thresholds. Future analyses will reevaluate chl *a* threshold levels for nontidal stations.

The techniques described here built upon past methods for assessing eutrophication. Historical concentrations, narrative statements, or comparisons to reference stations, on their own, have not been effective at communicating regional differences or the connections between water quality and biota. The use of biologically relevant thresholds provides a link between water quality and aquatic organisms that depend on these waters for survival. The link to changes in seagrass coverage indicated that these are meaningful indicators of ecological changes for the system under consideration (Fig. 10). Further, by synthesizing individual metrics into an overall index, general patterns in eutrophication emerged (Fig. 5).

Integrated assessments should ideally use as many independent parameters measuring system state as possible (e.g., nutrients, dissolved oxygen) as well as primary (e.g., phytoplankton abundance) and secondary ecological responses (e.g., seagrass occurence). This combination of diverse measures ensures that the approach is broadly applicable along the typical eutrophication transition within a system from benthic-dominated to water-column-dominated primary production (Fig. 5). Different parameters will provide information about changes within the system at different times in this process. This was observed in the relatively unimpacted Chincoteague Bay, where seagrass abundance is high and phytoplankton often low, but increases in nutrients indicate a potential eutrophication transition (Figs. 3 and 5). Once secondary ecological features are impaired (e.g., seagrass loss), measuring nutrients will provide information on whether management initiatives are effective in reducing nutrient loads

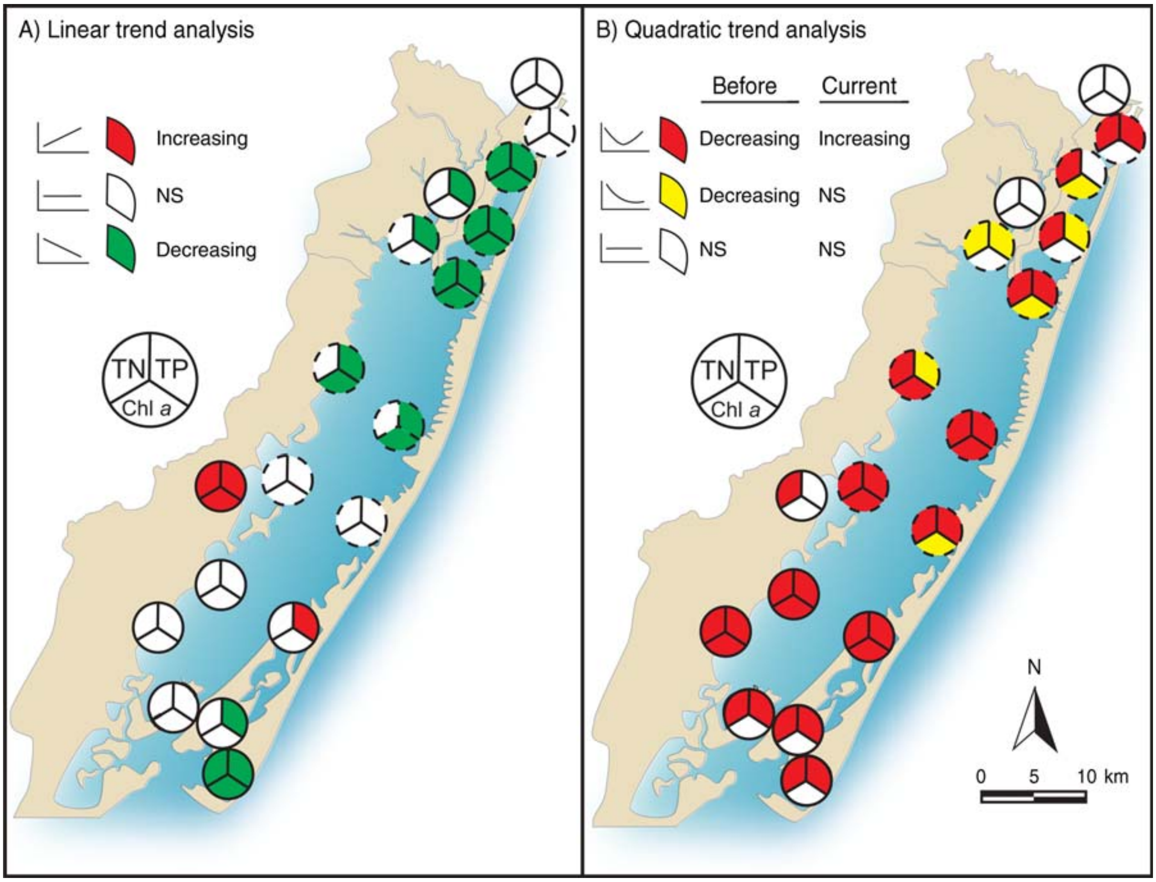

FIG. 6. (A) Linear and (B) nonlinear water quality trend analyses (NS indicates no significant change) for total nitrogen (TN), total phosphorus (TP), and chlorophyll *a* (Chl *a*). Panel (B) gives trends for two periods: "before" and "current." Trends cover the periods 1987–2004 (dashed circles) or 1991–2004 (solid circles).

and ultimately whether biotic communities respond to these reductions. For these reasons, it is ideal to also include further metrics on macroalgae, fisheries, and wetlands in a fully integrated assessment for tracking ecosystem status.

Is water quality improving or degrading?

Trend analyses allowed us to track changes over time to determine whether management actions were helping to improve conditions in the bays (Fig. 6). Linear trends were the historic method for analyzing change in the coastal bays. This method showed a majority of stations with no significant trend and among those with a trend more were significantly decreasing in nutrients (Fig. 6A). Only two stations were significantly increasing in nutrients, both in Chincoteague Bay. These two stations had no proximate point source discharges, so increases in nutrients and, subsequently, chl *a* concentrations, may have been due to groundwater intrusion or another local nonpoint source. Recent nitrogen isotope ratio studies revealed sources of highly processed nitrogen in the areas of these two stations, which may indicate sewage/septic inputs (Jones et al. 2004).

TABLE 4. Comparison of the percentage of sites with significant trends based on linear trend analyses vs. nonlinear trend analyses.

	Linear trend analyses		Quadratic trend analyses		
Parameter	Significantly improving	Significantly degrading	Significant parabola	Currently improving	Currently degrading
Chlorophyll *a*	22	6	61	0	39
Total N	11	6	94	0	89
Total P	44	11	78	0	50

Note: The "Significant parabola" column reports the percentage of sites that have a significant quadratic term (trend reversal), while the "Currently degrading" quadratic results are the percentages of sites that had trends significantly increasing after the inflection point (see Fig. 7).

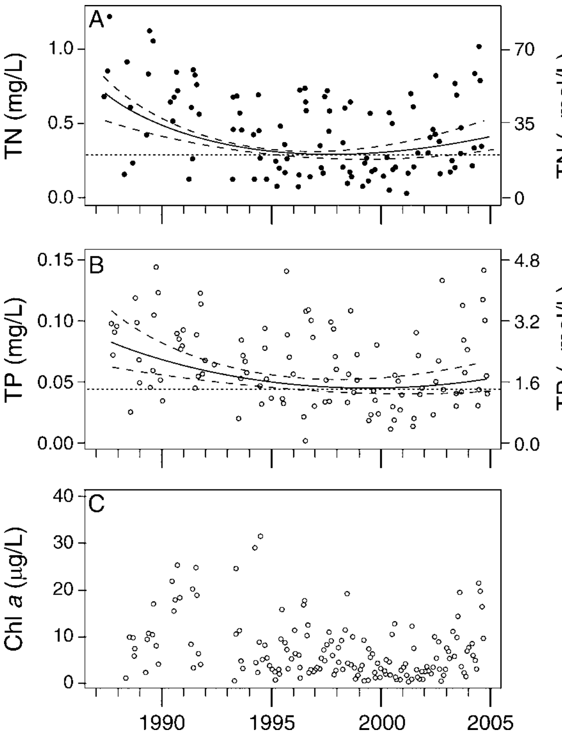

FIG. 7. Nonlinear trend analysis results for the years 1987–2004 representing the three colors in Fig. 6B. Data are from the National Parks Service station 2 in Sinepuxent Bay. (A) The solid black line is the quadratic curve fit of total nitrogen (TN) over time with surrounding dashed lines and dotted lines representing upper and lower 95% confidence intervals. The horizontal dotted line is the critical inflection value, or the hypothetical date when trend reversal occurred. Toward the end of the date range, the critical value is not encompassed by the confidence limits, so the current trend in TN can be said to be significantly increasing (red in Fig. 6). (B) Similar to (A), except that the critical value for total phosphorus (TP) is encompassed by the confidence limits. Therefore, the post-critical-value upward trend in TP is not significant (yellow in Fig. 6). Total nitrogen and total phosphorus are presented in both mg/L (left axes) and μmol/L (right axes). (C) No trends were found in the chlorophyll *a* (Chl *a*) data for this station (white in Fig. 6).

Nonlinear trend analyses provided vastly different results to the linear analyses, showing that a majority of stations have undergone significant trend reversals (from decreasing to increasing concentrations) and are transitioning to a more eutrophic state (Fig. 6). Currently degrading nutrient conditions were prevalent in the open bays, which are rated as good habitat for seagrasses. Significant chl *a* trends were focused in the middle of Chincoteague Bay and not apparent closer to the inlets. This is likely reflecting higher flushing frequencies in these areas that are greater than the doubling time of the phytoplankton. Degrading nutrients were not apparent in the more impacted Newport Bay, which may be a sign that management actions to reduce nutrient inputs are at least maintaining a set level in this embayment (MDE 2003).

The widespread distribution of currently degrading trends throughout the southern bays indicates a large-scale nonpoint source impacting water quality. Land cover in these watersheds is predominantly forest, agriculture, and wetlands. Groundwater inputs from agriculture or increased septic inputs, as well as increases in atmospheric deposition, may explain the currently increasing nutrients. Since delivery of groundwater to the bays is much slower than surface runoff (several years to decades compared to hours or days), nutrients currently entering the bays may have been applied to the land many years ago (Dillow et al. 2002). However, dissolved organic nitrogen, DON, is believed to be driving the currently increasing TN trends in the southern bays (Glibert et al. 2007).

Dissolved organic nutrients have been shown to be bioavailable in these bays (Glibert et al. 2001, Mullholland et al. 2004; M. R. Mulholland, G. Boneillo, and E. C. Minor, *unpublished manuscript*). Current sources may include atmospheric deposition as well as input from large animal operations located near streams (poultry manure is a common fertilizer in the region). Recent research indicates that increased use of organic nitrogen fertilizers (urea) as well as non-fertilizer uses coupled with sewage impacts over the past decade may be polluting coastal areas (Glibert et al. 2006). Direct runoff of urea is suspected since urease inhibitors are increasingly being added to manure to reduce ammonia emissions (Glibert et al. 2006). Additionally, phosphorus has become mobile in sediments due to the longtime application of phosphate-rich poultry manure. Current trend reversals are a warning sign that more needs to be done to protect this fragile ecosystem.

Impacts to seagrass

Monitoring the effects of degrading water quality trends on living resources is the ultimate assessment in evaluating eutrophication. Can threshold failures and trend analyses predict changes in organismal populations? Here, the key living resource examined was seagrass. Water quality status analyses show that many stations, even those in Chincoteague Bay, currently fail seagrass thresholds for one or both nutrients (Fig. 3A, B). The variability of seagrasses among segments (6% available habitat occupied in St. Martin River vs. 77% in Sinepuxent Bay) is correlated to the regional water quality index summaries (Fig. 10).

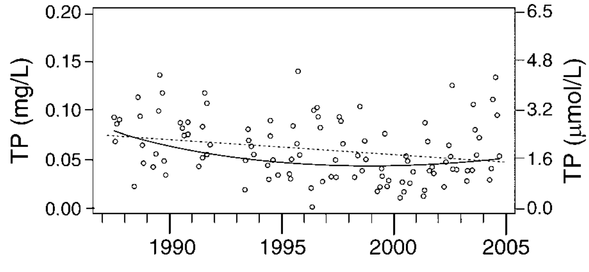

FIG. 8. Comparison of linear (dotted line) vs. nonlinear (solid line) trend analyses plots for total phosphorus (TP) at a single station for the years 1987–2004.

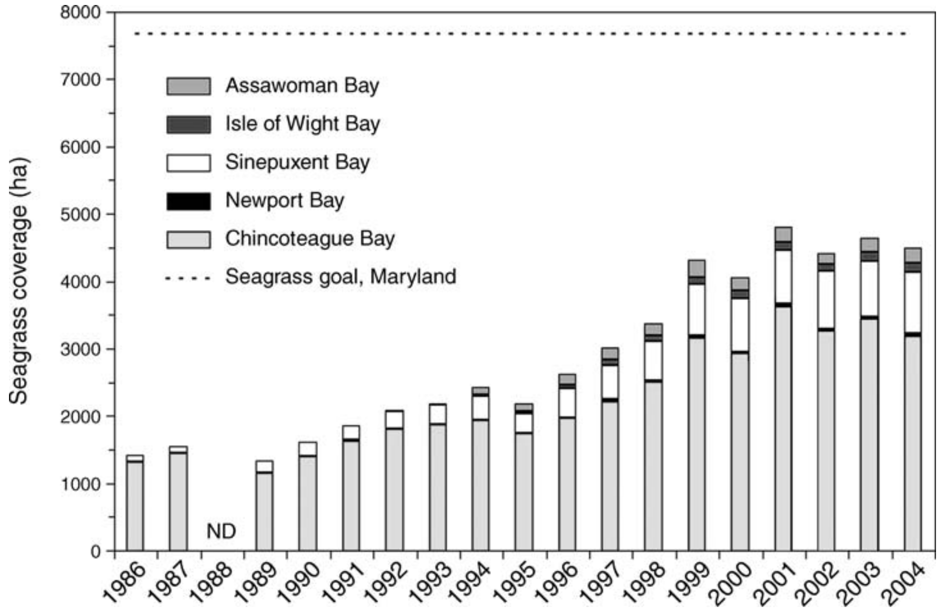

FIG. 9. Annual seagrass coverage in the Maryland coastal bays. A seagrass goal (dashed line) of 7669 ha was identified as potential seagrass habitat in the Maryland portion of the bays based on depth and sediment type. Data are taken from the Virginia Institute of Marine Sciences ⟨www.vims.edu/bio/sav/⟩. ND indicates no data.

Throughout the coastal bays seagrass abundance has been increasing since monitoring began in 1986, with an overall 320% increase that may be related to historically improving nutrient trends. However, these increases have leveled off over the past several years (Fig. 9; Orth et al. 2004, 2006). One possible explanation is that seagrasses have occupied all viable habitat. However, recent analyses estimate that current seagrass coverage occupies 67% of the total potential habitat (area of suitable depth and sediment type) in the Maryland portion of the bays. Another hypothesis is that the leveling of seagrass abundance coincides with the inflection period of nutrient and chlorophyll *a* trend reversals, suggesting that seagrass coverage may also be in an inflection period and may begin moving toward a decline if current trends continue.

Impact to fisheries

While no estimates of fish health in relation to the DO threshold have been calculated in this study, the State of Maryland criteria for oxygen was codified to protect fish and other aquatic resources (COMAR 1995). Recent analyses of a long-term fishery data set collected in the coastal bays used nonmetric multidimensional scaling to assess relatedness of fish communities within the bays. Discrete groups were associated with the same bay segments defined for water quality, and the groupings clustered similarly to the ranking of embayments by the water quality index (Murphy and Secor 2006).

Murphy and Secor (2006) also reported higher relative abundances of fish in the northern bays (Assawoman and Isle of Wight Bays) than in the southern bays (Chincoteague or Sinepuxent Bays). The northern bays were most affected by anthropogenic eutrophication, but also exhibited higher connectivity to the ocean inlet. The influence of increased eutrophication was a likely explanation for increased productivity levels of fish (Nixon and Buckley 2002) that were observed in the northern bays, particularly given that lower levels of seagrass and reduced water quality also occurred in the northern bays. Fishery abundance and diversity was shown to peak in early summer (June, July) compared to other seasons but decreased significantly in late summer (August, September; Murphy and Secor 2006). This trend tracked the seasonal rise and fall of DO in the coastal bays and provided evidence to support the DO threshold. However, a greater understanding of the interaction between DO, temperature, flushing, and fish communities is required.

Future monitoring and analyses

The current monthly monitoring effort is essential for assessing the effectiveness of management practices.

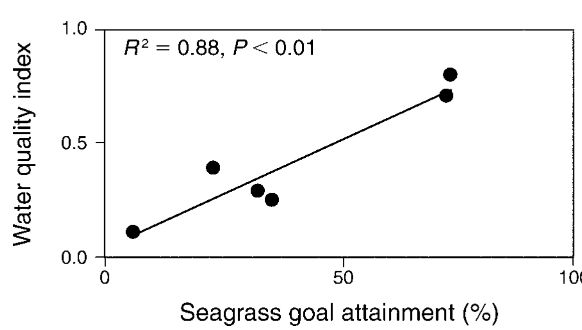

FIG. 10. Water quality index vs. percentage of potential seagrass habitat occupied.

Long-term trend analyses allow managers to evaluate past nutrient control efforts and project future needs in a region. However, this type of monitoring does not differentiate sources of nutrient pollution. Stable-isotope studies are currently underway to evaluate the effectiveness of this method for source detection. Another drawback, especially pertinent to DO measurements where minimum values are observed between the hours of 05:00 and 07:00, is that single monthly samples may not truly represent average conditions. Spatial coverage may also be inadequate for some analyses. These issues of temporal and spatial scale are being addressed in the Maryland coastal bays by the placement of in situ continuous monitors and intensive spatial mapping, respectively. Generally speaking, spatial variability is not as great in the Maryland coastal bays as temporal variability. More frequent measures of oxygen are essential in shallow, unstratified systems, and diel measurements are recommended to better assess eutrophication impacts.

The inclusion or exclusion of indicators can be a contentious issue in water quality management. The water quality and biotic indicators used in this study work well for this system and may be useful in other shallow coastal lagoons. Since these systems are naturally dominated by seagrass, the relationship of the water quality index to seagrass abundance should be further explored in other lagoons. However, as nutrient load increases and resulting higher chlorophyll levels lead to declines in seagrasses due to light attenuation, seagrass-dominated production gives way to macroalgae-dominated production and, finally, to phytoplankton-dominated production (Fig. 5; Duarte 1995, Taylor et al. 1995, Valiela et al. 1997). Therefore, additional monitoring of other primary producers (i.e., epiphytes, microphytobenthos, macroalgae) would be a useful indicator of eutrophication severity level. Better quantification of diffuse nutrient sources, especially atmospheric and groundwater inputs, are needed to assess the influences on eutrophication. And, finally, a more detailed look at changes in nutrient components would provide insights on nutrient sources as well as biotic changes (see Glibert et al. 2006, 2007).

Conclusion

Primary symptoms of water quality degradation in the Maryland coastal bays are increasing nutrient and chlorophyll levels. Secondary symptoms include low dissolved oxygen, leveling of seagrass area, and increased harmful algae blooms (Glibert et al. 2007).

Are conditions in the coastal bays suitable for key living resources? Overall, no. Large areas of the bays exhibited nutrient enrichment above threshold levels needed to maintain biotic communities. Are conditions improving or degrading? Generally degrading. Large areas of what was thought to be pristine habitat showed significantly degrading water quality trends and living resource impacts. Are differences in water quality associated with status and trends in living resources? Yes, the water quality index was related to seagrass habitat, and water quality trends were also related to seagrass trends.

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