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A Gomparative Analysis of Eutrophication Patterns in a Temperate Coastal Lagoon

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ABSTRACT: The coastal bays and lagoons of Maryland extend the full length of the state's Atlantic coast and compose a substantial ecosystem at the land-sea margin that is characterized by shallow depth, a well-mixed water column, slow exchange with the coastal ocean, and minimal freshwater input from the land. For at least 25 years, various types of measurements have been made intermittently in these systems, but almost no effort has been made to determine if water quality or habitat conditions have changed over the years or if distinctive spatial gradients in these features have developed in response to changing land uses. The purpose of this work was to examine this fragmented database and determine if such patterns have emerged and how they may he related to land uses. Turbidity, dissolved inorganic phosphate, algal biomass, and primary production rates in most areas of the coastal bays followed a regular seasonal pattern, which was well correlated with water temperature. Nitrate concentrations were low (<5 μM), and only modestly higher in tributary creeks (<20 μM). Additionally, there was little indication of the spring bloom typical of river-dominated systems. There does appear to be a strong spatial gradient in water quality conditions (more eutrophic in the upper bays, especially in tributary creeks). Comparisons of water quality data collected between 1970 and 1991 indicate little temporal change in most areas and some small improvements in a few areas, probably related to decreases in point-source discharges. Seagrass communities were once extensive in these systems but at present are restricted to the eastern portion of the lower bays where water clarity is sufficient to support plant survival. Even in these areas, seagrass densities have recently decreased. Examination of diel dissolved oxygen data collected in the summer indicates progressively larger diel excursions from lower to upper bays and from open bays to tributary subsystems; however, hypoxic conditions (<2 mg l-') were rarely observed in any location. Nitrogen input data (point, surface runoff, groundwater, and atmospheric deposition to surface waters) were assembled for seven regions of the coastal bay system; annual loading rates ranged from 2.4 g N m^{-2} yr^{-1} to 39.7 g N m^{-2} yr^{-1}. Compared with a sampling of loading rates to other coastal systems, those to the upper and lower bays were low while those to tributaries were moderate to high. Regression analysis indicated significant relationships between annual nitrogen loading rates and average annual total nitrogen and chlorophyll a concentrations in the water column. Similar analyses also indicated significant relationship between chlorophyll a and the magnitude of diel dissolved oxygen changes in the water column. It is concluded that these simple models, which could be improved with a well-designed monitoring program, could be used as quantitative management toots to relate habitat conditions to nutrient loading rates.

Introduction

The shallow marine lagoons and bays lying behind barrier islands and sand spits are a conspicuous physiographic feature of continental land margins around the world. Globally, these coastal features occupy some 13% of the coast, with the percentage being even larger for tropical coastlines bordered by mangrove forests and coral reefs (Nixon 1982). These marine ecosystems are generally characterized by their shallow depth (mean depth <3 m; Kjerfve 1986) and limited exchange with the adjacent oceans (Lankford 1977). Watersheds associated with lagoonal systems tend to be

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small compared to those of river-dominated estuaries. This results in relatively low freshwater delivery to the embayments, which in conjunction with high evaporation rates leads to salinities typically approaching or exceeding that of seawater (Mee 1978).

As with estuaries and other coastal environments, lagoons are among the most productive ecosystems in the world. In most of these relatively shallow systems, phytoplankton photosyothesis is supplemented with primary production from a rich assemblage of seagrasses and benthic ma—— algae (Lee and Olsen 1985). A relatively large fraction of total ecosystem primary production in these shallow lagoonal systems is deposited to the sediment surface (Hargrave 1973), causing bent c respiration to exceed planktonic respiration (Kemp et al. 1992). Thus, it is likely that the majority of secondary production in these systems is associated with benthic food-chains, as opposed to the plankton. In fact, many migrating demersal nektonic species (e.g., shrimp, crabs, spot, flounder) depend on shallow lagoonal habitats as nura ery areas for early development (Day et al. 1989).

As a result of the relatively low freshwater runoff into coastal lagoons, nutrient inputs to these systems are generally lower than those of other estuaries (Nixon 1982). Atmospheric deposition (especially, on the eastern margins of industrialized continents, such as North America, Paerl et al. 1990) and advection from adjacent upwelling coastal waters (especially on western continental margins, Smith et al. 1991) may be more important in these, systems than in others. The systems whose ecological structures make them productive despite low nutrient inputs may be more adversely affected by nutrient enrichment than other more nutrient-replete estuarine systems,

As a result of continued industrialization and growth of human populations worldwide, eutrophication is becoming more common and more rapid in coastal ecosystems of all classes (Nixon et al. 1986). While substantial discussion has focused on the effects of nutrient enrichment on river-dominated systems such as Chesapeake Bay (Kemp et al. 1983; Officer et al. 1984), little information has been published about similar processes occurring in the nearby coastal lagoonal systems (Maryland coastal bays), In general, surprisingly little is known about the responses of lagoonal ecosystems to eutrophication. This could be important because some processes occurring in deeper systems may be modified in these shallow systems. In particular, the overriding importance of water-column stratification in the deoxygenation of bottom waters (Turner et al. 1987) suggests that hypoxia may not be a problem in these shallow, well-mixed systems. In fact, the more elusive problem of episodic and diel oxygen depletion may replace the seasonal hypoxia observed in other estuaries. However, the declines in seagrass abundance associated with benthic diatoms assuming an epiphytic growth pattern may be intensified in these systems by the natural dominance of benthic algae (Sand-Jensen and Søndergaard 1981).

The purpose of this paper is to present some historical information on nutrient inputs and water quality conditions in Maryland's coastal lagoons. Information is presented regarding key temporal and spatial patterns for these variables. Data are analyzed in terms of nutrient budgets and correlations between regional nutrient loading rates and water quality conditions. The implications of these analyses for ecological processes and nutrient waste management strategies in this lagoonal system are discussed.

The Study Area

The Maryland coastal bay system extends along the entire Atlantic coast of the state behind the barrier islands of Assateague and Eenwick (Fig. 1). The system consists of five major bays: Chincoteague, Newport, and Sinepuxent bays to the south ("lower bays"), and Isle of Wight and Assawoman bays to the north ("upper bays"). Several smaller subsystems are associated with these bays, including the St. Martin River, Bishopville Creek, and Turville Creek in the upper bays, and Trappe Creek in the lower bays. The majority of the creek systems associated with the bays are tidal and have low rates of freshwater discharge (Cerco et al. 1978). River gauging stations on the branches of the St. Martin River indicate low flows (0.02–0.03 $m^3\ s^{-1}$) even for this major tributary (Cushing et al. 1973). The bays and associated tributaries are shallow, with an average water depth of about 1 m (Table 1).

The drainage basins of the coastal bays are relatively small compared to open water areas (45,246 ha or about 1.7 times the area of the bays) and are many times smaller than those associated with the adjacent Chesapeake Bay system. These small, flat watersheds generate low rates of freshwater flow into the bays. Limited freshwater input and constricted ocean inlets results in very slow water replacement times. For example, Pritchard (1960) estimated a flushing rate for Chincoteague Bay of only 7.5% per day. It appears that sediments, nutrients, pathogens, and toxic materials are effectively retained in the bay systems, in part because of the poor flushing characteristics. The National Oceanic and Atmospheric Administration-United States Environmental Protection Agency Team on Near Coastal Waters reached a similar conclusion

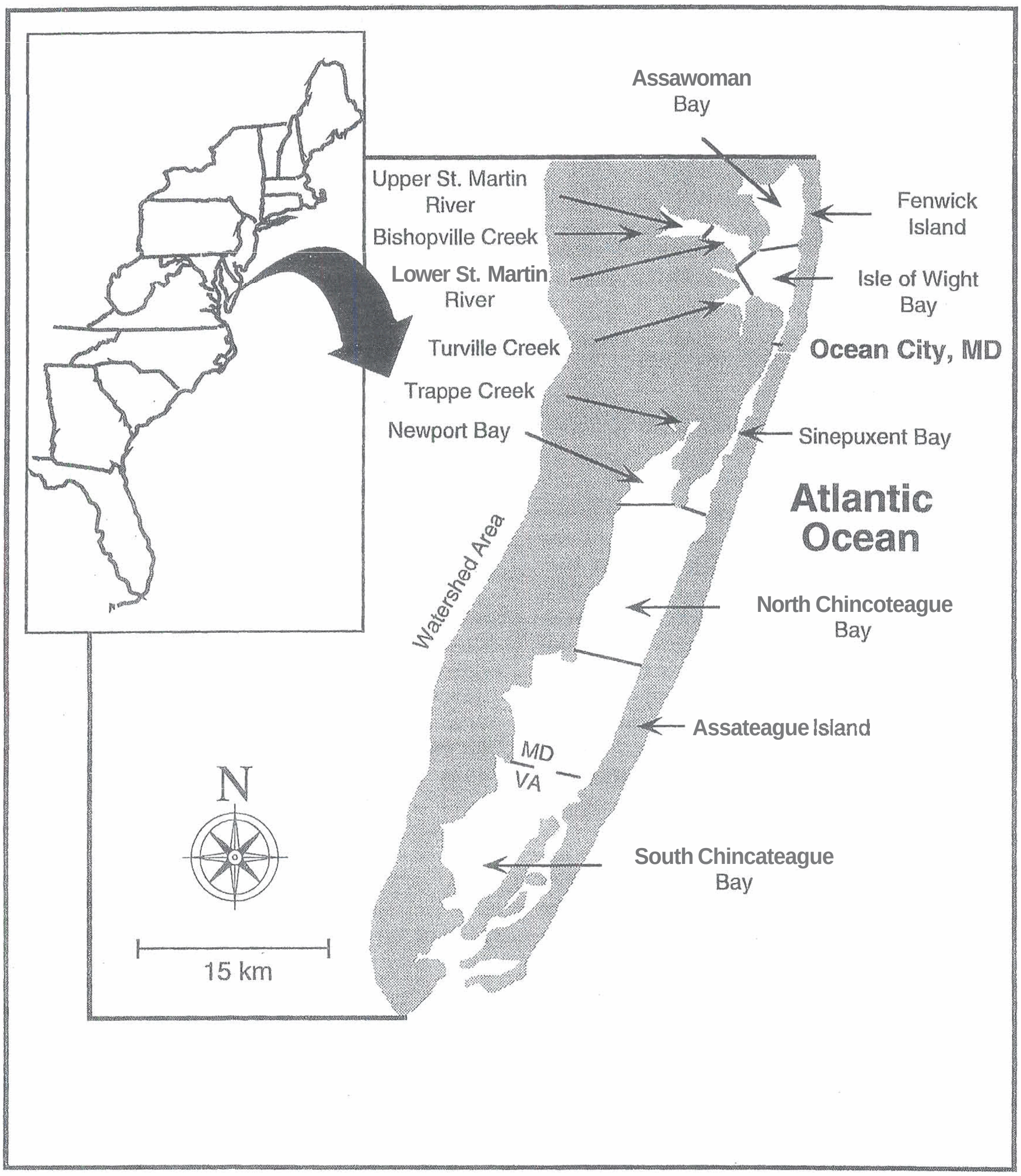

Fig. 1. **Map of the Maryland coastal bays complex indicating the boundaries of the watershed and of subsystems for which nitrogen inputs were estimated.**

and rated the Maryland coastal bays as being highly susceptible to the effects of increased inputs of materials (Quinn et al. **1989).**

Past **(1973, 1990)** and projected **(2005)** land uses in the major basins of the coastal bays are summarized in Table 2. In all the basins, natural land uses (forest and wetland) dominated during **1973** and **1990.** Agricultural uses were also substantial, while feeding and developed uses were less prevalent. Feeding operations, while not important on an areal basis, are the largest contributor to nitrogen loading because of their very high loading coefficients (Jacobs et al. **1993).** One of the major features of land use in the region was the relatively small changes that have occurred during the 17-yr period between **1973** and **1990.** Approximately **4%** of the watershed was converted from agriculture to developed land uses despite a pop-

TABLE 1. **Water surface area, average depth and volume, and drainage area for the Maryland coastal bays system. Sources for data are footnoted. Boundaries for the coastal bay locations are shown in Fig.** 1.

Coastal Bay Locauon	Surface Area[1] (m^2 10^6)	Average Depth[2] (m)	Volume[3] ($m^3 \cdot 10^6$)	Drainage Area ($m^2 \cdot 10^6$)
Assawoman Bay	22.5	1.20	27.0	24.7
isle of Wight Bay	15.8	1.22	19.3	17.5
St. Martin River	8.40	0.67	5.63	95.5
Turville Creek	5.30	0.67	3.55	34.3
Sinepuxent Bay	24.6	0.67	16.5	26.7
Newport Bay	15.9	1.22	19.4	113
Chincoteague Bay (Maryland portion)	189	1.22	231	141
Totals	282		322	452

[1] **Data from Jacobs et** al. (1993).
[2] **Data from Boynton** (1993).
[3] **Volumes calculated by multiplying surface areas by average depths.**

ulation growth of 43% over the same period (Table 3), reflecting an increase in only the density of development. While it is always difficult to judge the utility of projected land uses, Jacobs et al. (1993) concluded that substantial conversion of natural and agricultural land uses to developed uses would occur by 2005 in the basins adjacent to the resort area of Ocean City, Maryland. Based solely on land use types it would appear that nutrient and sediment inputs to the coastal bays did not change greatly between 1970 and 1990. The potential for change during the next decade appears to be considerably greater.

The population of Worcester County has increased slowly (1.5% per year) since 1970 and is now about 35,000 (Table 3). Relative to some basins of the Chesapeake, population density is low (0.8 persons ha^{-1}) in the coastal bays region. Population is projected to increase about 15% by the year 2000 (Andriot 1980). One of the major features of the coastal bays is the immense seasonal change in population; during the summer months the population of Ocean City swells to about 300,000 persons or almost 60 times the resident population. Most of this impact appears to be directed toward the ocean side of Fenwick Island because of the focus on beach activities and an ocean outfall for city sewage.

TABLE 3. **Population growth of Worcester County and selected cities** (1970–1990) **and projected Year** 2000 **population for** the **county.**

Year	County Total	Berlin	Ocean City[1]	Pocomoke City	Snow Hill
1970	24,442	1,942	1,493	3,573	2,201
1980	30,889	2,162	4,946	3,558	2,192
1990	35,028	2,616	5,146	3,922	2,217
2000	40,350				

[1]300,000 **(summer population), population data from Worcester County Development Office (personal communication).**

Methods and Data Sources

Data for analyses presented here are taken from a variety of unpublished contract reports and agency data files, each of which is available upon request. Primary data sources include four studies describing physical, chemical, and biological properties in the bay waters and studies compiling information on land use, human activities in the region, and nutrient loadings to the bays. Details of field and laboratory methods are contained in the cited reports, and here we provide only brief descriptions of techniques.

Water Quality Studies

Four water quality studies conducted in the Maryland coastal bays were used extensively in this analysis. The most spatially and temporally extensive of these was conducted by the Virginia Insti-

TABLE 2. **Changes in land use (percent of total for each watershed area) for the coastal bays from** 1973 **to** 1990 and **projected for** 2005. **Data for** 1973 **are from Cerco et** al. (1978); **data for** 1990 **and projected** 2005 **estimates are from Jacobs et** al. (1993).

Watershed	Yea3	Forest	Wetland	Agriculture	Feeding	Development
Assawoman	1973	26	27	38	2	7
	1990	23	25	26	2	24
	2005	18	25	27	3	27
Isle of Wight	1973	36	7	40	2	15
	1990	37	4	40	2	15
	2005	22	1	25	0	52
Sinepuxent	1973	32	35	26	1	6
	1990	29	33	19	0	9
	2005	0	15	7	0	78
Newport	1973	40	15	34	2	9
	1990	42	14	34	1	7
	2005	34	6	20	1	39
Chincoteague	1973	46	19	32	2	1
	1990	40	31	25	1	1
	2005	35	30	25	1	6

tute of Marine Science (VIMS) and included one intense diel study (August 1975) and monthly slack water surveys during 1975–1976 (Fang et al. 1977a, b). Both the diel and slack water surveys included measurements of physical conditions (depth, tidal stage, temperature, Secchi depth, and turbidity) and chemical characteristics (dissolved nutrient concentrations, salinity, dissolved oxygen, and chlorophyll a concentrations) at approximately 24 and 29 stations in the lower and upper bays, respectively. The Maryland Department of Health and Mental Hygiene (1985) conducted a second study in 1983. This study focused on examining water quality of the upper bays during summer months only; measurements included chlorophyll *a*, dissolved nutrient concentrations, salinity, and temperature. The National Park Service (1991) conducted another water quality survey from 1987 through 1991. Nine stations, located in the Maryland portion of the lower bays, were sampled from early spring (March–April) through fall (October). Variables measured included temperature, salinity, dissolved oxygen, chlorophyll a, and dissolved nutrients. In addition, nutrient and chlorophyll a concentrations, temperature, salinity, and turbidity characteristics and water column production and respiration were measured on a monthly basis at three locations in Chincoteague Bay during 1970 (Boynton 1973).

The analytical methods used are briefly described as follows. Temperature and salinity were analyzed with standard in situ conductivity and temperature probes; water clarity was monitored via Secchi disk observations; dissolved nutrients were measured with oceanographic colorimetric techniques; chlorophyll a analyses were done using either spectrophotometric or fluorescence methods (Strickland and Parsons 1972); dissolved oxygen concentrations were analyzed using either Winkler titrations or polarographic (Clark-type) electrodes; and rates of plankton community production and respiration were measured using the light-dark bottle oxygen technique (Strickland and Parsons 1972). While nutrient analyses have become more automated through time, the nature of the method has remained the same. There is no indication that the detection limits of the chemical analyses have changed in a meaningful fashion.

LAND-USE AND NUTRIENT LOADING STUDIES

Two studies developed estimates of annual nutrient loads to the bay system from point and diffuse sources (Cerco et al. 1978; Jacobs et al. 1993). In the first of these, the standard United States Army Corps of Engineers STORM model was used. To calibrate the model, loading rate estimates from the Chincoteague Bay watershed were obtained from field data on small watersheds (10–100 acres). Point-source data were also collected during this study. In the second evaluation, point-source loads were collected from National Pollution Discharge Elimination System records for the coastal bays. Nutrient loads from surface runoff were estimated using unit area pollutant load modeling for 15 different land-use categories; groundwater nitrogen loads were estimated using groundwater concentrations coupled to a groundwater flow model for the coastal bays region. These calculations were completed for 23 subwatersheds.

Results and Discussion

SEASONAL AND INTERANNUAL PATTERNS OF WATER QUALITY

The most complete description of annual cycles in water quality and plankton community variables for the coastal bays is provided by a dataset that includes measurements from several stations in the central portion of Chincoteague Bay during 1970 (Boynton 1973; Fig. 2). Several distinctive patterns were evident. Strong unimodal annual cycles (well correlated with water temperature) were observed for primary production, chlorophyll a, dissolved inorganic phosphate concentration, and water column turbidity. Consistently low concentrations of nitrate (<1 μM) indicate that diffuse-source nitrogen inputs from adjacent watersheds were relatively unimportant, while modest ammonium concentrations (1–4 μM) peaking in warmer months suggest the importance of nutrient recycling. A remarkably low N:P ratio (dissolved inorganic nutrients) results mainly from very high phosphate concentrations during warm portions of the year. While temporal sampling in other water quality programs in the coastal bays was not as intense, seasonal patterns similar to those observed in Chincoteague Bay emerged for the upper bays. These cycles are different than those observed for river-dominated estuaries (e.g., Chesapeake Bay and tributaries), where nitrate from land sources is very high following the spring freshet, algal biomass is typically at a maximum in spring (associated with a diatom bloom), and dissolved inorganic phosphate concentrations are low in surface waters for most of the year (Boynton et al. 1982).

There are also predictable spatial differences in water quality patterns related to location in the coastal bay system (Fig. 3). For example, occasional high nitrate concentrations are associated with the St. Martin River, which is one of the few substantial sources of freshwater to these systems. Similarly, dissolved inorganic phosphate and chlorophyll a concentrations were higher in this subsystem, slightly enhanced in the adjacent Isle of Wight Bay,

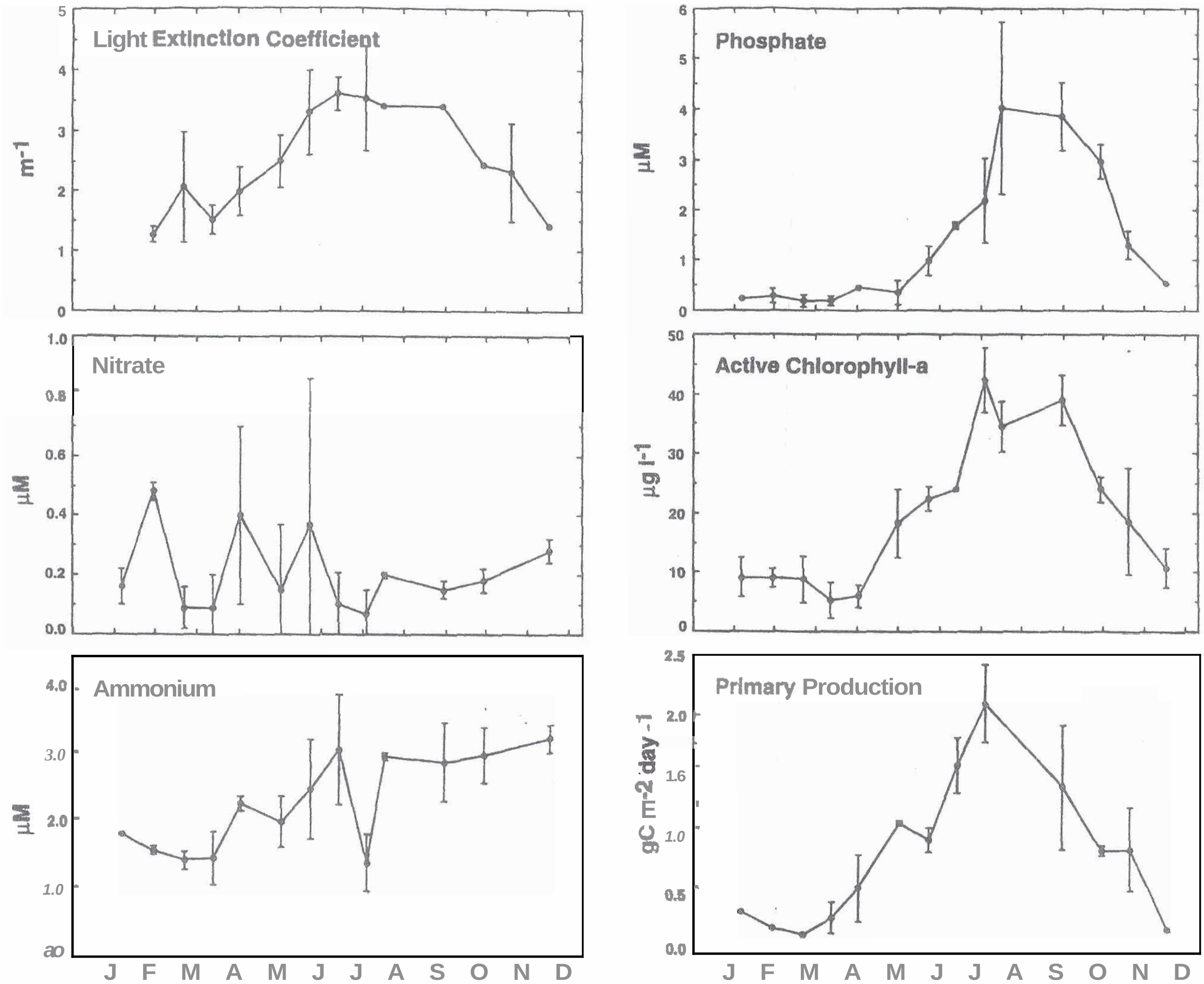

Fig. 2. Annual patterns (mean and standard deviation) for a selection of water quality variables (light extinction coefficient, nitrate, ammonium, phosphate, chlorophyll a, and phytoplankton primary production rates) based on samples collected in the central portions of Chincoteague Bay during 1970 (Boynton 1973).

and lower in Chincoteague Bay. In general, there appeared to be water quality gradients related to spatial proximity to nutrient source areas. In all of the coastal bays, dissolved inorganic N:P ratios (e.g., Fig. 3) suggest strong nitrogen limitation for primary production, with values generally below 10 throughout the year and below 5 during summer. It is unclear what aspects of nutrient loading and internal cycling processes contribute to this pattern, but these conditions are very different from those reported for Chesapeake Bay and other river-dominated estuaries (e.g., Boynton et al. 1982, 1995).

Despite the fact that water quality measurements have been made in the coastal bays region for several decades, it is not possible to develop a rigorous trend analysis of such variables because there is little temporal or spatial continuity between the datasets. However, a more qualitative assessment can be made using chlorophyll a data collected during summer periods from the upper and lower bays (and associated subsystems) during 1975, 1983, and 1991 (Fig. 4). It appears that the highest chlorophyll a concentrations have occurred in u p per bay areas adjacent to nutrient sources. Further, there is an indication that summer phytoplankton biomass is slightly lower in more recent years than during the mid-1970s, although the mean concentrations are not significantly different in most cases. As indicated below, spatial and temporal patterns exhibited by chlorophyll a and other water quality indicators (such as diel dissolved oxygen

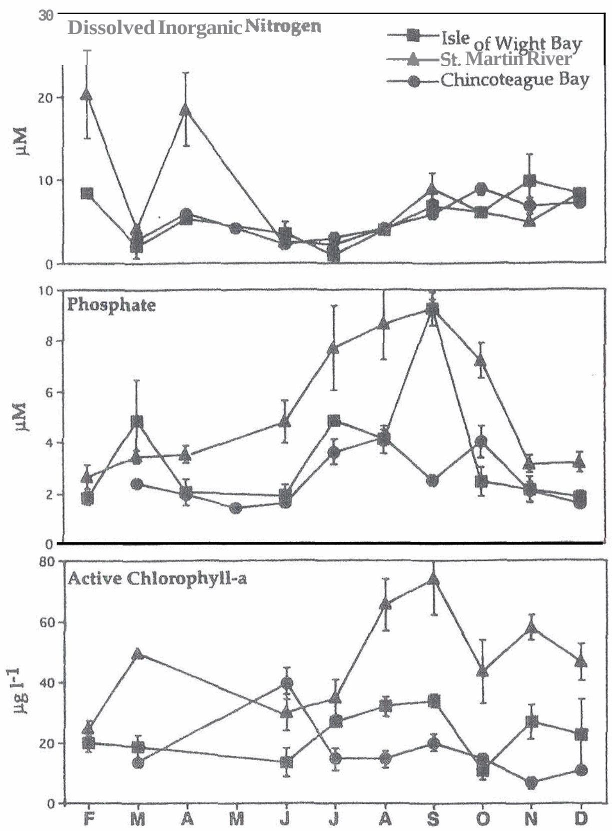

Fig. 3. Annual patterns (mean and standard deviation) for a selection of water quality variables (dissolved inorganic nitrogen, phosphate, and chlorophyll a) based on samples collected at clusters of stations (n = 4–7) in Isle of Wight Bay, St Martin River, and Chincoteague Bay during 1975–1976 (Fang et al. 1977a).

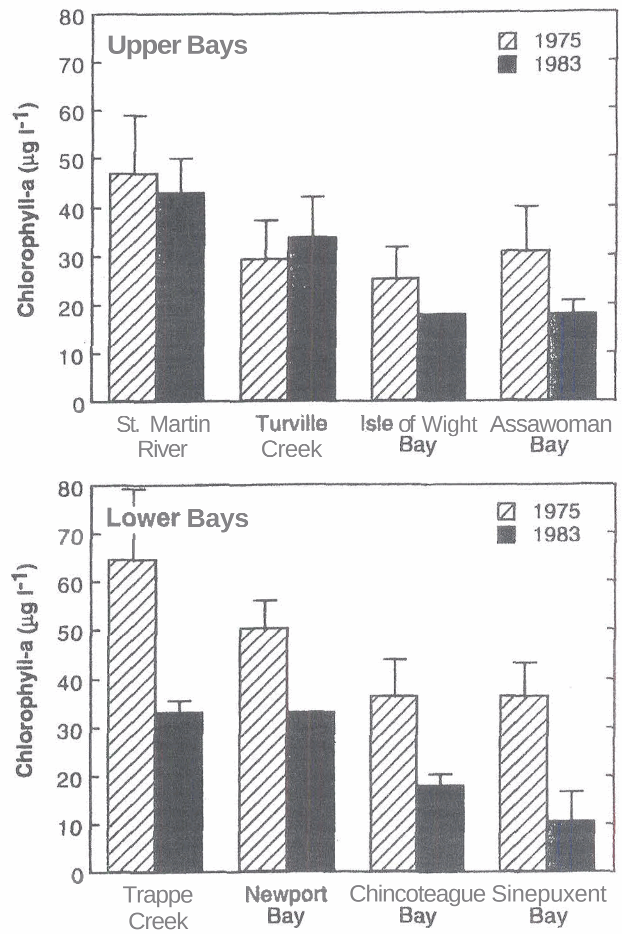

Fig. 4. Summer average chlorophyll *a* concentrations for representative regions of the Maryland coastal bays based on samples collected during 1975, 1983, and 1991. Data are from Fang et al. (1977a, b), Maryland Department of Health and Mental Hygiene (1985), and National Park Service (1991).

patterns and presence of submerged seagrasses) are related to each other and to differences in local nutrient loading rates.

Diel Cycles of Dissolved Oxygen Related to Algal Biomass

One of the common characteristics of coastal systems undergoing eutrophication is the development of hypoxic or anoxic bottom waters. This is a primary concern for management of estuarine and coastal marine ecosystems. Among the numerous impacts of depressed oxygen conditions are mortality of benthic infauna (Officer et al. 1984) and reductions in key biogeochemical processes such as coupled nitrification-denitrification (Kemp et al. 1990).

Hypoxic or anoxic bottom waters occur most commonly in systems that are seasonally or permanently stratified (Turner et al. 1987), and depressed oxygen conditions in such systems often persist for weeks to months (Malone et al. 1986; Magnien et al. 1990). The shallow, well-mixed water–column typical of the coastal bays would tend to prevent the development of hypoxic or anoxic conditions, at least on time scales of weeks or longer, because of reoxygenation of the water from the atmosphere. In metabolically active aquatic systems, however, a diel cycle in dissolved oxygen can develop in spite of the buffering influence of air-water exchange. In some cases, hypoxic conditions can develop on diel time scales, generally in the hours just before sunrise.

To investigate this possibility for the coastal bays, we examined diel oxygen data collected by Fang et al. (1977a) during late August 1975 (Fig. 5). Approximately 30 of 80 stations throughout the coastal bays complex had sufficient data to characterize a diel pattern. In some cases, dissolved oxygen declined substantially between dusk and dawn; however, hypoxic conditions (dissolved oxygen <2 mg

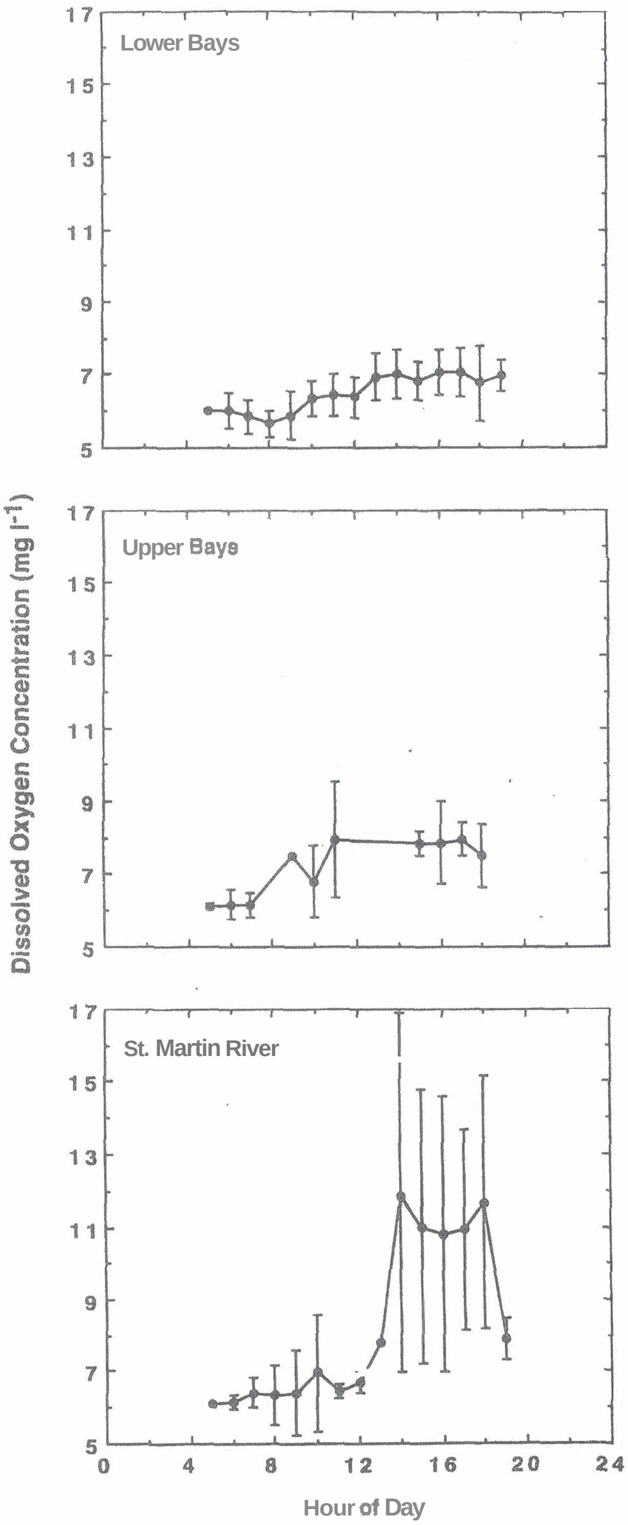

Fig. 5. Diel patterns (mean and standard deviation) of dissolved oxygen for three regions of the Maryland coastal bays. Samples were collected at multiple stations (n = 4-7) in the lower and upper bays and in the St. Martin River during several days in August 1975.

l^{-1}) were not observed for these stations. In fact, concentrations rarely declined below 85% of the oxygen solubility. Patterns in the magnitude of diel changes in dissolved oxygen concentrations were observed, however. These fluctuations became progressively larger (as did variability among stations) from the lower to the upper bays, and diel changes were particularly large at stations in the St. Martin River where afternoon oxygen concentrations reached over 160% of the saturation concentration.

Because levels of autotrophic and heterotrophic activity (rate of oxygen production and consumption) are often directly correlated with the magnitude of the algal stock (e.g., Malone et al. 1986), it was hypothesized that diel fluctuations in dissolved oxygen concentrations would be related to algal biomass, measured as chlorophyll a concentrations. This possibility was investigated using the same data as above (Fang et al. 1977a). There were 80 stations where both oxygen and chlorophyll a concentrations were measured near dawn (0530–0830 h) and near dusk (1730–1930 h). Oxygen concentrations at dusk were subtracted from those at dawn, and the difference was divided by the time between measurements to develop an oxygen rate of change for each station. The dawn and dusk chlorophyll a concentrations were simply averaged to obtain an estimate of algal biomass during the diel period. Several data points were discarded because dissolved oxygen increased overnight. This was assumed to have resulted from advection of water masses rather than biological processes. Linear regression analysis using the remaining data indicated a highly significant relationship between the variables ($r^2 = 0.633$, $p < 0.01$) (Fig. 6). The slope of the regression line indicates that the diel rate of dissolved oxygen change increases by 0.05 mg l^{-1} h^{-1} for every 10 μg l^{-1} increase in chlorophyll a concentration. These patterns in diel dissolved oxygen changes may be related to human activities in watersheds via nutrient loading rates and their relation to mean chlorophyll a concentrations.

Nitrogen Loads to the Coastal Bays and Other Coastal Systems

Three classes of total nitrogen inputs are provided in Table 4 for a typical year (i.e., with average rainfall) and include point, diffuse, and atmospheric sources. The diffuse-source value includes surface water inputs, groundwater inputs, and inputs from chicken-rendering plants, which are common in the drainage basin. Atmospheric deposition includes only wet-fall deposition to surface waters of the bays. The fraction of atmospheric nitrogen deposition to watersheds which reaches

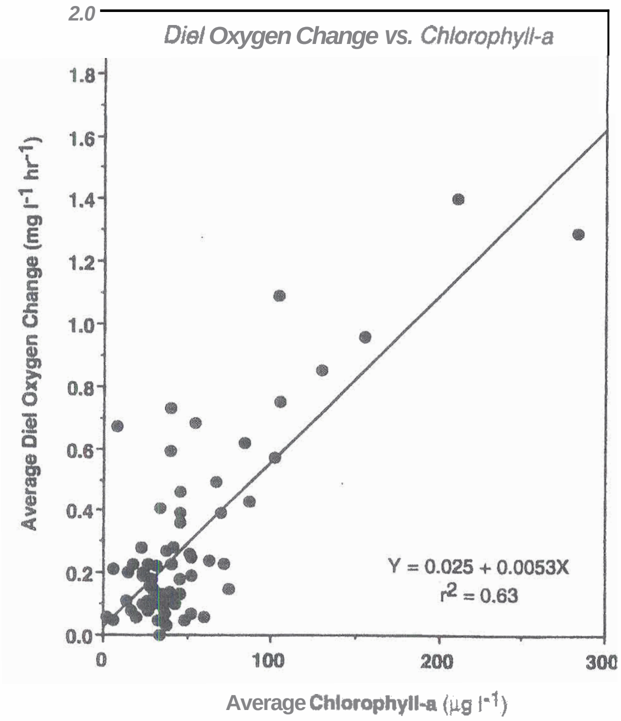

Fig. 6. A scatter plot of average daily chlorophyll a concentranons (n = 2–5) versus diel dissolved oxygen rate of change (based on differences between dawn and dusk dissolved in oxygen measurements). Measurements are from stations in the lower and upper bays, Turville Creek, and St. Martin River. All data are from Fang et al. (1977a, b).

streams is included in the diffuse-source values (Fisher and Oppenheimer 1991). With the exception of atmospheric inputs, all nutrient source estimates were made by Jacobs et al. (1993). These budgets were evaluated for an annual time period; it is assumed that this time-scale captures most of the important features characterizing nutrient impacts on these systems, The specific sections of the coastal bays for which nitrogen loads were calculated are shown in Fig. 1. The downstream boundary of each segment was determined by the natural morphology of the segment rather than by some other criteria (e.g., salinity zones). The surface areas of the segments varied by a factor of over 60, with the Maryland portion of Chincoteague Bay being the largest and the Turville Creek complex the smallest. The range in volumes was somewhat greater because the smallest systems were also slightly shallower. Nitrogen input budgets were developed for seven segments of the coastal bays complex. These input budgets also included best estimates of nitrogen export from one subsystem to another.

Overall, diffuse sources and atmospheric deposition of nitrogen to the surface waters of these systems were the most important sources of total nitrogen. Point sources of nitrogen represented only a minor source (4%) to the entire system of coastal bays. Areal loading rates for each of the coastal bay segments span a little more than an order of magnitude, ranging from 2.4 g N m^{-2} yr^{-1} in Sinepuxent Bay to 39.7 g N m^{-2} yr^{-1} in the St. Martin River. Areal loading rates for the Turville Creek complex and Newport Bay were about half those of St. Martin River. Rates for Assawoman, Isle

TABLE 4. Total nitrogen loading for regions of the Maryland coastal bays. Data sources and calculations used in developing the input budgets are provided in the footnotes. The inputs conJned in this budget are generally representative of average annual rainfall conditions rather than any specific recent annual period.

Coastal Bay Location	Point[1] Sources (Kg N yr^{-1})	Diffuse[2] Sources (Kg N yr^{-1})	Atmospheric[3] Sources (Kg N yr^{-1})	Total[4] Loading Rate (Kg N yr^{-1})	Aerial[5] Loading Rate (g N m^{-2} yr^{-1})
Assawoman Bay	0	52,091	39,800	91,891	4.10
Isle of Wight Bay	0	12,969	27,949	40,918	6.50
St. Martin River	18,290	302,867	12,382	333,539	39.7
Turville Creek	0	78,249	4,953	83,202	15.7
Sinepuxent Bay	10	22,566	35,820	58,396	2.40
Newport Bay	36,939	220,842	20,342	278,123	17.5
Chincoteague Bay (Maryland portion)	29	258,038	318,403	576,470	3.10

[1] Point sources of nitrogen were dweloped by Jacobs et al. (1993) based on information from Maryland Department of Environment. Data are for the 1990–1991 period.

[2] Nonpoint sources of nitrogen were developed by Jacobs et al. (1993) based on land uses and land-usespecific run-off coefficients. The diffuse-source loading rates used in these budgets reflect average conditions.

[3] Atmospheric inputs represent total nitrogen (TN) deposition in wet-fall directly to the surface of bay waters. Concentrations of TN are from Smullen et al. (1982). Rainfall data are from stations located in Snow Hill and Assateague Island National Seashore and were collected by National Oceanic and Atmospheric Administration, annual Climatological Summary (1980–1991). Average annual rainfall was taken to be 43.8 inches per year.

[4] Sum of point, nonpoint, and atmospheric TN sources.

[5] Aerial TN loads were calculated by dividing total loads by the surface area of specific regions of the coastal bays.

TABLE 5. A summary of annual areal total nitrogen loading rates for a sampling of estuanne and coastal systems. Data are from Boynton et al. (1995).

Location	Total Nitrogen Loading Race (g N m^{-2} yr^{-1})
Kaneohe Bay, Hawaii	2.2
Maryland coastal bays (lower bays)	2.4–3.1
Baltic Sea, Sweden	3.0
Choptank River, Maryland	4.3
Maryland coastal bays (upper bays)	4.1–6.5
Albermarle Sound. North Carolina	7.1
Apalachicola Bay, Florida	7.8
North Sea	9.4
Pamlico River, North Carolina	12.0
Patuxent River, Maryland	12.7
Mobile Bay, Alabama	17.9
Delaware Bay, Delaware	18.2
Mainstem Chesapeake Bay, Maryland	20.5
S. San Francisco Bay, California	22.6
Narragansett Bay, Rhode Island	27.6
Maryland coastal bays (tributaries)	15.7–39.7
Pocomac River, Maryland	29.3
Patapsco River, Maryland	49.0
Tokyo Bay, Japan	89.1

of Wight, and Chincoteague bays were slightly larger than those for Sinepuxent Bay.

To place estimated total nitrogen loading rates to the coastal bays in perspective, rates for several coastal and estuarine systems were collected from literature sources (Table 5). This summary is not intended as a thorough synthesis of loading rates to coastal systems but rather as a means to compare local loading rates to those in a few other well-studied systems. There is about a factor of 17 dii ference between the highest and lowest nitrogen loading rates among the Maryland coastal bay systems and a factor of about 40 for the sample group of coastal systems. Compared to other estuaries in this analysis, it appears that loading rates to the major bays are low and loads to the tributary subsystems are intermediate to high.

Comparable nutrient loading rates in different systems do not necessarily evoke comparable ecosystem responses, however. For example, total nitrogen loading rates to the Potomac River and Narragansett Bay are similar, but nutrient and chlorophyll a concentrations in the Potomac are high compared to those in Narragansett Bay (Nixon et al. 1986). On the other hand, loading rates to the Baltic Sea are much lower than those of some of the coastal bay systems, but hypoxic and anoxic conditions are now common in the subpycnocline waters in the Baltic (Larsson et al. 1985). Basin morphology and circulation undoubtedly have a strong influence on the relative impact of loading rates (Wulff et al. 1990).

Water Quality Conditions Related to Nutrient Loading Rates

Vollenweider (1976) and his colleagues established relationships between nutrient loading rates and lake ecosystem responses. The earliest of these efforts developed strong relationships between algal biomass (chlorophyll a concentration in the upper mixed layer of lakes) and the total phosphorus loading rate for lakes. In these regression models, P loading rates were scaled for each lake by the surface area, the depth, and the freshwater turnover time. The resulting relationships indicated the trophic status of lakes (oligotrophic, mesotrophic, or eutrophic) and further indicated how much the P loading rate would have to be modified for the lake to change from one trophic status to another.

In this paper, we developed a similar series of analyses utilizing data available for the Maryland coastal bays and using N loading instead of P loading. Nitrogen loading rates normalized to surface area (g N m^{-2} yr^{-1}) were determined for each of the units of the coastal bays; however, these rates were not further adjusted for depth or freshwater fill time as in the Vollenweider model. An adjustment for mean depth would have made little difference because all of the coastal bay segments are about the same depth. No adjustment was made for residence time because freshwater fill time would not be a meaningful scaler and good estimates of tidal flushing could not be made without additional information. As will be noted below, an adjustment for residence time should be made and could potentially improve the model. Water column concentrations of total nitrogen and chlorophyll a were taken primarily from the Fang et al. (1977b) slack water surveys because that study encompassed all regions of the coastal bays and included data for almost all months of the year.

Broadly speaking, the resulting relationships (Figs. 7 and 8) indicate that different regions of the coastal bays respond in a reasonably consistent fashion to variations in N loading rates. Over small changes in nutrient loading rates, however (e.g., South Chincoteague Bay versus Assawoman Bay), other factors such as flushing characteristics are clearly important. The regression slope indicated that annual average total nitrogen concentration increased by about 0.5 μM for every unit increase in total nitrogen load. Chlorophyll a levels were in the range of 40–60 μg l^{-1} at loading rates of 30–60 g N m^{-2} yr^{-1} and decreased to 15–20 μg l^{-1} at loading rates of 2-6 g N m^{-2} yr^{-1}. The slope of the best fit regression line indicates that chlorophyll a concentrations increase on average by about 0.7 μg

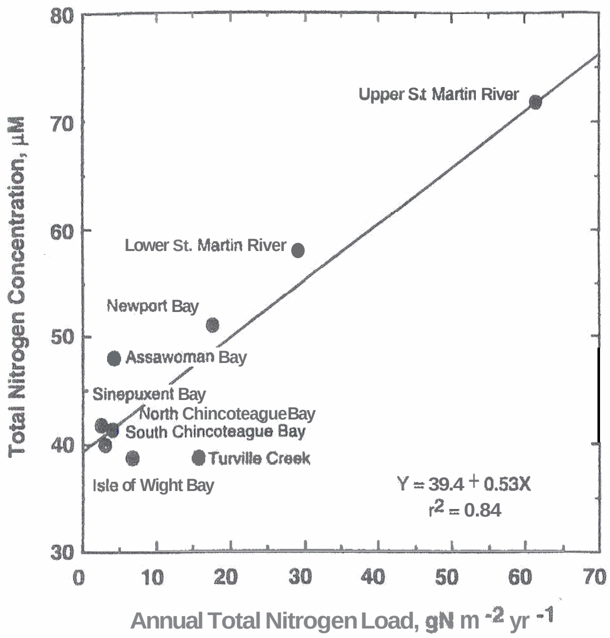

Fig. 7. A scatter plot relating annual areal total nitrogen loads to annual average total nitrogen concentrations for several regions of the Maryland coastal bays. Loading data are from Jacobs et al. (1993) and total nitrogen concentrations are from Fang et al. (1977a).

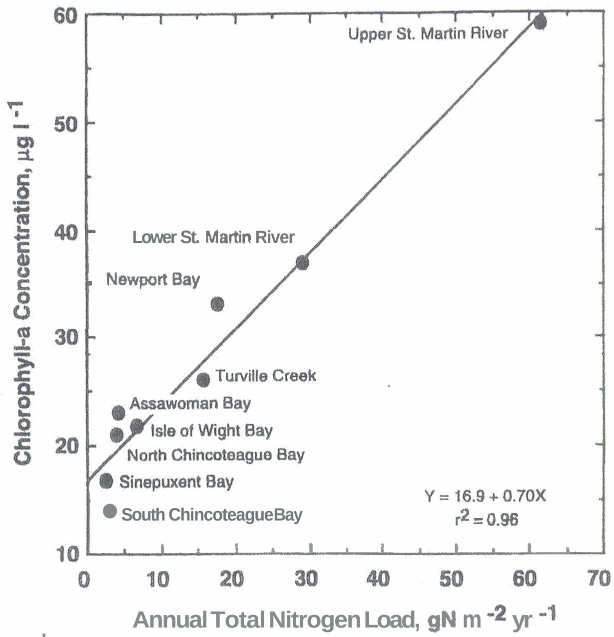

Fig. 8. A scatter plot relating annual areal total nitrogen loads to annual average chlorophyll a concentrations for several regions of the Maryland coastal bays. Loading data are from Jacobs et al. (1993) and total nitrogen concentrations are from Fang et al. (1977a).

l^{-1} for every unit increase (1 g N m^{-2} yr^{-1}) of N loading.

The strength af these relations suggests this approach is a simple and tractable tool for management of nutrient wastes in these coastal bays. In effect, the analysis allows prediction of chlorophyll a concentrations based on nitrogen loading rates and, by extension, indicates the magnitude of nitrogen loading rate reductions needed to achieve lower chlorophyll *a* levels. While indications from the present analysis are very promising, the data used in the analysis are incomplete. The chlorophyll a dataset is generally limited to the warmer seasons, and some areas of the coastal bays were sampled more intensively than others. In addition, in this preliminary analysis, we have related total N loading rates developed for 1990 land uses to measurements of total nitrogen and chlorophyll a concentrations from 1975 to 1977. Data limitations necessitated this approach. The fact that N loading rates to Chincoteague Bay estimated for 1977 (Cerco et al. 1978) were only 13% higher than those estimated for 1990 (Jacobs et al. 1993) suggests that the magnitude of this discrepancy may be small. In addition, plankton chlorophyll a concentrations were actually slightly higher during the earlier period, consistent with nitrogen loading rate estimates (Fig. 4). Furthermore, land-use changes thoughout the coastal bays system have been remarkably small, as noted above. Obviously, this problem could be resolved by establishing a well-designed ecological-water quality monitoring program in this coastal bay system.

SUBMERGED AQUATIC VEGETATION IN THE COASTAL BAYS

Biological indicators of eutrophication are another approach to characterizing eutrophication patterns in these coastal bays. In particular, the presence, absence, and vitality of seagrass beds in areas which have historically supported beds is a reasonable indicator of eutrophication, While the historical distribution of seagrasses in the Maryland coastal bays is not known precisely, it is likely that these shallow systems once supported, extensive beds. Historical preferences to the coastal bays indicate seagrasses were far more abundant than they are at present, they declined during the wasting disease of the 1930s, and they recovered to some extent following this period [Anderson 1970). Fresently, seagrasses are mainly limited to the lower bays (Chincoteague and Sinepuxent bays), where two species of seagrasses, *Zostera marina* and *Ruppia maritima,* occur mainly on the eastern shore in water shallower than 1 m (Orth et al. 1991). The total coverage has increased from 2,310 ha in 1987 to 2,494 in 1990, or from about 6.6% to 8.0% of bottom area during the 4-yr period

(Orth et al. 1986, 1989, 1991). In the upper bays, seagrass abundance and distribution has been severely limited. *Ruppia maritima* has been sighted in the Isle of Wight Bay in shallow waters near the Ocean City Inlet (Orth et al. 1991) where water quality and light conditions are probably most favorable within this region. There were no other sightings of this species, or of *Z. marina,* for the upper bays in these surveys.

Despite the small increases in coverage in the lower bays, the density of plants in these beds is declining. Orth et al. (1986, 1991) reported a decline in the percentage of "dense" beds from 51% in 1986 to 10% in 1990. Anderson (1970) reported an average biomass of about 249 g dw m^{-2} for *Z. marina* in Chincoteague Bay but, more recently, Dennison (unpublished data) found an average of 140 g dw m^{-2} at the same site and during the same time of year. These less dense beds also occupy shallower areas, suggesting the plants are generally light-limited. This may result from shading by phytoplankton, suspended sediments, and epiphytic algae attached to seagrass leaves (e.g., Kemp et al. 1983). In the mid-Atlantic region in general, the decline of seagrass communities has been attributed to declines in water quality, and particularly to nitrogen enrichment of the water column and decreased light availability (Izumi et al. 1982; Kemp et al. 1983; Dennison 1987).

The recent intensive investigations of seagrass ecology in Chesapeake Bay have resulted in a series of seagrass habitat criteria that indicate these plant communities in the Chesapeake are healthy when light extinction coefficients are less than 1.5 m^{-1}, chlorophyll *a* concentrations are less than 15 $\mu g\ l^{-1}$, and total nitrogen concentrations are less than 10 μM (Dennison et al. 1993; Stevenson et al. 1993). In most regions of the coastal bays, these criteria are not met, especially in the tributary subsystems. Sinepuxent and Chincoteague bays, where nutrient loading rates are the lowest of any in the coastal bay region (2–3 g N $m^{-2}\ yr^{-1}$; Table 4), are an exception. In Chincoteague Bay, annual mean concentrations of dissolved inorganic nitrogen and chlorophyll a are well below the seagrass habitat criteria (Figs. 3 and 4), although chlorophyll *a* levels did exceed the criterion in mid summer. It is in these areas that seagrass communities still exist. To the extent that criteria developed for the Chesapeake are applicable to the coastal bays, it would appear from the above regression analyses (Figs. 7 and 8) that nutrient loading rates would have to be decreased to between 2 g N $m^{-2}\ yr^{-1}$ to 5 g N $m^{-2}\ yr^{-1}$ before seagrass communities could be expected to flourish, a substantial reduction in the tributary subsystems.

Although these analyses should be considered preliminary, an empirical approach offers a simple management tool that applies ecological information more directly to management objectives than do conventional waterquality modeling approaches. As such, this simple, inexpensive tool provides a framework for rapidly synthesizing monitoring data into a form useful for management purposes. Coupled with "habitat criteria" as explicit management goals, this approach specifically and directly addresses relations between nutrient (N) loading and algal blooms, and associated potential for seagrass survival and diel hypoxia.

Summary and Management Implications

This analysis of the data for the Maryland coastal bays system suggests several ecological and management-oriented conclusions as well as several areas of uncertainty.

Eutrophication of coastal bay waters appears to be most severe in the upper bays, particularly the tributary subsystems. Concentrations of dissolved inorganic nutrients, chlorophyll *a*, and dissolved oxygen all suggest incipient eutrophication in several upper bay tributaries. Relatively high phosphorus concentrations throughout the coastal bays indicate a general pattern of strong nitrogen limitation for algal growth. Long-term interannual trends in water quality are unclear, with some data suggesting slight improvements and others indicating deterioration. It is clear that a welldesigned long-term water-quality monitoring program would provide the data needed to resolve these uncertainties. The program should have stations located in all regions of the coastal bays, including upper and lower bay areas of both open and restricted circulation. Special attention needs to be given to the temporal scales of measurement considering the large diel variability in dissolved oxygen concentrations typical of these shallow ecosystems.

Seagrass communities are limited to the eastern regions of the lower bay, which are the areas of lowest nutrient and chlorophyll a concentrations. A decrease in plant density and biomass in these sites during the last two decades suggests that water quality conditions are marginal for seagrass survival, Throughout the upper bays, water column nutrient and chlorophyll a concentrations exceed habitat criteria established from Chesapeake Bay research. Since the shallow depths in these bays suggest that seagrasses should be a major ecological feature, establishment of attainable restoration goals should be a priority.

There is still uncertainty in the quantitative rates of nutrient loading to the coastal bay system. Rates of nutrient input from near-surface groundwater directly entering the lagoons (saline water) need to be better documented and understood. Because of the low ratio of watershed area to water surface area for these bays, atmospheric deposition appears to a relatively major source of nitrogen to these systems. Inclusion of atmospheric dry-fall could substantially increase estimates of total N loading to these bays. The fundamental importance of obtaining high quality measurements of nutrient inputs to these systems for future management activities is obvious. There is virtually no information available from the coastal bays concerning the fate of nutrients once they enter the bays. Pollution dispersion processes (e.g., tidal flushing) should be estimated reliably. Since natural nutrient sinks such as sediment burial and denitrification serve to mitigate partially the need for nutrient control programs, it is useful to know the magnitude of these rates and, if they are Iarge, to manage them in such a fashion as to promote them (Boynton et al. 1995).

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