Cultural eutrophication in the Choptank and Patuxent estuaries of Chesapeake Bay

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Abstract

?1   The Choptank and Patuxent tributaries of Chesapeake Bay have become eutrophic over the last 50–100 years. Systematic monitoring of nutrient inputs began in ~1970, and there have been 2–5-fold increases in nitrogen (N) and phosphorus (P) inputs during 1970–2004 due to sewage discharges, fertilizer applications, atmospheric deposition, and changes in land use. Hydrochemical modeling and land-use yield coefficients suggest that current input rates are 4–20 times higher for N and P than under forested conditions existing 350 yr ago. Sewage is a major cause of increased nutrients in the Patuxent; agricultural inputs dominate in the Choptank. These loading increases have caused three major water-quality problems: (1) increased nutrients, phytoplankton, and turbidity; (2) decreased submerged grasses due to higher turbidity and epiphyton shading; and (3) bottom-water hypoxia due to respiration of excess organic matter. Oxygen in the Patuxent is consistently <3 mg L^{-1} in bottom waters in summer, whereas oxygen in the Choptank has been decreasing for the last 20 yr and is now approaching 3 mg L^{-1} in wet years. The low N:P of sewage inputs to the Patuxent results in an N-limited, P-saturated system, whereas the Choptank is primarily limited by N, but with P limitation of phytoplankton during spring river flows. Insufficient action has been taken to improve the water and habitat quality of these estuaries, although reduced eutrophication in dry years suggests that both estuaries will respond to significant decreases in nutrients.

The coastal zone throughout much of the world has been adversely affected by human activities (Nixon 1995). Increasing human populations have caused land-cover changes from forest to agricultural and urban areas, and these changes in turn have resulted in greater freshwater flows and increased fluxes of particulates, nitrogen (N), and phosphorus (P) from watersheds to estuaries and coastal waters (Vitousek et al. 1997; Howarth et al. 2000; Cloern 2001). These increased inputs have promoted a widespread but not universal pattern of responses. Increased loading of particles promotes turbidity and sedimentation, and increased loading of N and P increases phytoplankton biomass and production, which also reduce water clarity. These changes have contributed to losses of submerged aquatic vegetation (SAV), augmented organic loading of bottom waters, and increased oxygen demand, leading in some cases to hypoxia or anoxia (Boynton et al. 1995). All of these responses have occurred in the Chesapeake Bay region of the eastern U.S.A., and similar responses have been observed elsewhere (e.g., Conley et al. 2000; Dennison et al. 2004). On the other hand, there are examples where anthropogenic changes in the watersheds have resulted in smaller estuarine effects, reflecting substantial differences in the aquatic processes that control ecological response to anthropogenic inputs (Eyre 1997).

Overexploitation of fish and shellfish populations has compounded water-quality problems. Overfishing results in diversity loss and progressive dominance by short-lived, pelagic species. The ecosystem consequences of overfishing are similar to those of nutrient enrichment (Caddy 1993) and can interact with enrichment in complex ways (e.g., Daskalov 2002). In a relatively simple example, overfishing of Chesapeake Bay oyster populations to their present 1% of historical levels has eliminated a major grazer on phytoplankton production (Newell 1988). Benthic suspension feeders still have a large effect on planktonic populations in

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many eutrophic estuaries, such as San Francisco Bay (Cloern 2001) and many Danish estuaries (Conley et al. 2000). Collectively, these anthropogenic effects on nutrient inputs and trophic structures are referred to as ''cultural eutrophication.''

Three main watershed characteristics determine the magnitude of increased nutrient export to coastal waters: (1) human population density (Peierls et al. 1991), (2) intensive agricultural production (Lee et al. 2001), and (3) the ratio of terrestrial drainage area to aquatic area (Caddy 1993). In rural areas with low population densities, septic systems discharge into groundwater, which traps much of the anthropogenic P in soils and may allow a hydrologic flow path long enough to provide opportunities for denitrification and plant uptake of anthropogenic N (e.g., Weiskel and Howes 1992; Lee et al. 2001). In contrast, dense human populations generate large volumes of nutrient-rich wastewater delivered by public sewer systems quickly and directly to aquatic systems. Installation of public sewers initially worsened coastal water quality in some cases, but advanced sewage treatment now accounts for some of the most successful efforts to reduce nutrient loads (Conley et al. 2002). Although tertiary treatment in sewage systems can reduce N and P concentrations to <35 fmol P L^{-1} and <250 fmol N L^{-1}, these technologies have not been promoted as aggressively in the U.S. as elsewhere (Conley et al. 2002; U.S. Environmental Protection Agency [US EPA] 2003).

The second cause of cultural eutrophication is the intensive production of food for human and animal populations. Agricultural production now depends heavily on the use of fertilizers for high crop yields from relatively small areas, and in the second half of the 20th century, applications of fertilizer to sustain or increase yields have greatly increased groundwater nitrate and soil P (e.g., Bohlke and Denver 1995; Sims et al. 1998). As a result, hydrologic fluxes through agricultural landscapes generate base flows in streams with high concentrations of nitrate derived from groundwater and storm flows with high concentrations of P derived from overland flow (e.g., Fisher et al. 1998).

The third cause of cultural eutrophication is the ratio of terrestrial drainage area to the area of coastal receiving waters. This ratio reflects the potential for impacts on aquatic systems by increases in human populations and intensive agriculture in adjacent terrestrial basins. Large terrestrial areas draining into small enclosed seas can potentially reduce salinity and increase nutrients and turbidity if the land is heavily populated and/or occupied by intensive agriculture. In contrast, small terrestrial areas draining into large coastal systems may have a much smaller influence even if such areas are heavily populated or occupied by agriculture.

To illustrate these processes, we present two case studies of cultural eutrophication. The Patuxent and Choptank basins are moderately sized terrestrial basins that discharge into estuarine tributaries of Chesapeake Bay. The Patuxent basin and its estuary have been used as a model for management in the Chesapeake region because they lie entirely within the state of Maryland, minimizing jurisdictional issues. The Choptank basin also lies mostly (86%) within the state of Maryland, with the remainder in Delaware. Both systems have a history of cultural eutrophication, but agriculture is

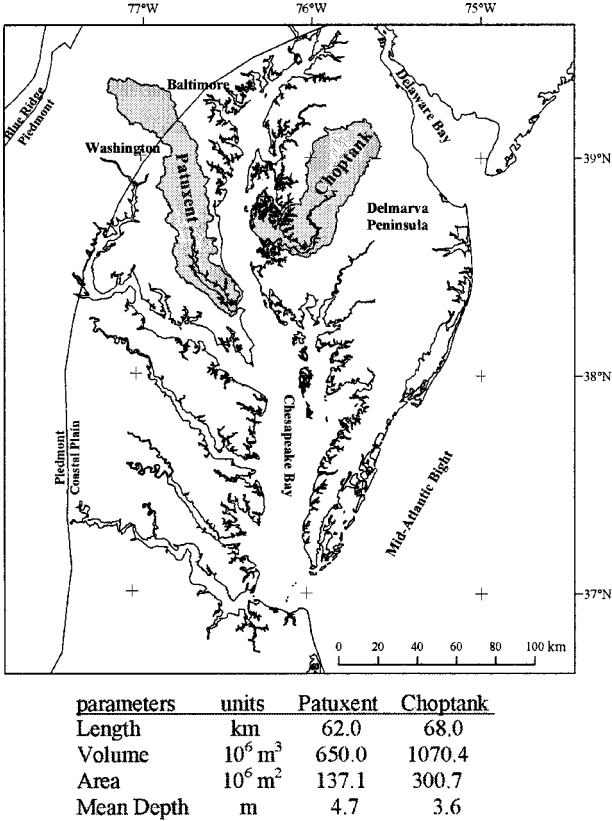| parameters	units	Patuxent	Choptank
Length	km	62.0	68.0
Volume	10^6 m^3	650.0	1070.4
Area	10^6 m^2	137.1	300.7
Mean Depth	m	4.7	3.6

Fig. 1.   Location of the Patuxent and Choptank basins and estuaries in the Chesapeake drainage of the mid-Atlantic region. The Choptank River basin lies in the Atlantic Coastal Plain province on the Delmarva peninsula (eastern shore of Chesapeake Bay), and the Patuxent River basin lies on the western shore of Chesapeake Bay, straddling the fall line, the border between the Piedmont and Coastal Plain provinces. Statistics contrasting the physical features of the estuaries are included in the table on the lower left.

the primary cause in the Choptank, whereas human wastewater is the primary cause in the Patuxent. Furthermore, the ratios of watershed area to estuarine area in these two systems differ by a factor of 3. In our exploration of these case studies, the main questions that we address are related to the historical record of increased nutrient export from the basins and ecological effects on their associated estuaries.

Study site descriptions

The Choptank and Patuxent basins lie primarily in the Mid-Atlantic Coastal Plain and within the Chesapeake drainage (Fig. 1). The Choptank basin (1,756 km^2) is rural and dominated by agriculture (62%) on the eastern side of Chesapeake Bay; only 5% of the basin is urban (Table 1). The basin surrounds a 300-km^2 estuarine tributary, with a mean depth of 3.6 m and a land:water ratio of 5.8 (Table 1). In contrast, the Patuxent basin (2,260 km^2), located between two major cities on the western side of Chesapeake Bay (Fig. 1), is dominated by forest (64%), with a significant urban component (16%). Agriculture accounts for only 20% of the land use (Table 1). The basin encloses a 140-km^2 estuarine

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Table 1. Physical and biogeochemical properties of the Choptank and Patuxent River basins. Land cover data for the Choptank basin are based on Lee et al. (2001) for 1990. Land cover data for the Patuxent basin are derived from Landsat ETM+ imagery for 2000 (S. Prince unpubl. data). Abbreviations: Agri., agriculture; Inter. wet., intertidal wetlands; Pop. den, population density.

| Basin | Land area (km^2) | % land cover | | | | Pop. den. (No. km^{-2}) | Sewage (10^6 L d^{-1}) | Estuary area (km^2) | Land/water |
		Forest	Agri.	Urban	Inter. wet.				
Choptank	1,756	30.9	61.5	5.4	2.2	59	22.7	301	5.83
Patuxent	2,260	63.5	20.3	15.7	0.4	262	235	137	16.5

tributary, with a mean depth of 4.7 m and a land : water ratio of 16, three times higher than the Choptank basin (Table 1).

The history of land use in these basins was similar until about 1900. Both basins were settled by Europeans in the 17th century and were dominated by agriculture throughout the 19th century (Benitez and Fisher 2004). The Choptank basin remained rural and dominated by agriculture and for-
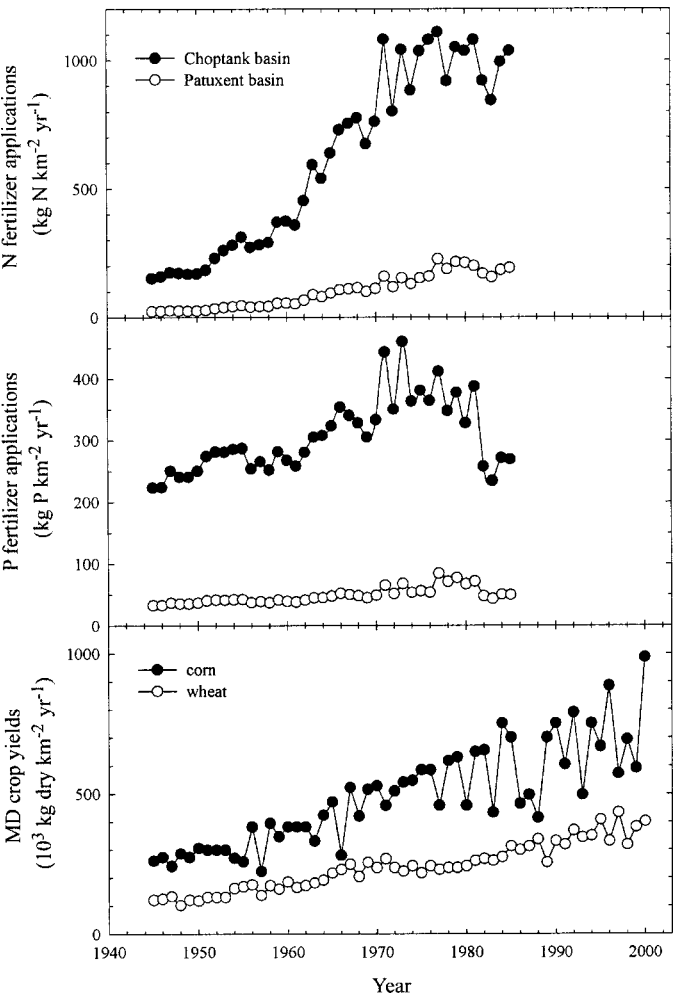

Fig. 2.   Fertilizer sales in the Choptank and Patuxent River basins and Maryland crop yields for corn and wheat. County-level information on sales of N and P fertilizer were obtained from the U.S. Geological Survey (USGS) website: http://water.usgs.gov/pubs/of/ofr90130/data.html. An area-based proportion of county fertilizer sales was added to obtain total basin values. Data sources: USGS, Benitez (2002).

est, with low population density throughout the 20th century (Benitez 2002). Although human populations doubled after 1950, the population density in the Choptank basin remains low (59 km^{-2}; Table 1). In contrast, the Patuxent basin lost much of its agriculture in the 20th century and underwent reforestation and suburbanization as urban areas expanded and population within the basin increased to 262 km^{-2}. The increase in human population has been particularly large in the upper Patuxent basin, where 10-fold changes have occurred since 1950.

Another important difference between the two basins is the magnitude of fertilizer applications. Because of the greater amount of agricultural land in the Choptank basin, fertilizer sales and use are much greater than in the Patuxent basin (Fig. 2). As in all agricultural areas in the U.S., fertilizer use on croplands increased dramatically after 1950, causing large increases in crop yields (Benitez 2002; Fig. 2). In 1990, basin-averaged applications (fertilizer sales divided by basin area) in the Choptank were >1,000 kg N km^{-2} yr^{-1}, but <200 kg N km^{-2} yr^{-1} in the Patuxent, reflecting the lower prevalence of croplands, not applications rates to croplands, which are similar in each basin, typically 5,000–15,000 kg N and 1,000–3,000 kg P km^{-2} yr^{-1}. Likewise, sales of P fertilizers increased by ~50% in the last half of the 20th century, and basin-averaged P applications in the Choptank were also considerably greater than those in the Patuxent (~300 kg P km^{-2} yr^{-1} vs. ~50 kg P km^{-2} yr^{-1}, respectively; Fig. 2). Expressing fertilizer application rates per unit area of basin enables comparison between basins and with diffuse source losses from the basin.

The estuaries associated with these two basins also have important physical differences. The salt-intruded lengths are similar (60–70 km), but the Choptank estuary has a larger area (300 km^2 vs. 140 km^2) and volume (1.1 km^3 vs. 0.65 km^3), a shallower mean depth (3.6 m vs. 4.7 m), and smaller area ratio of basin to estuary (5.8 vs. 16). These differences result in a wider, shallower Choptank estuary compared to the Patuxent, with a much smaller volume below the pycnocline, making the Choptank estuary less sensitive to hypoxia. Furthermore, the deepest region of the Choptank estuary (~26 m) lies 15–20 km from Chesapeake Bay and is isolated from the bay by a shallow sill <10 m in depth. In contrast, the deepest region of the Patuxent estuary (~36 m) is only 10 km from Chesapeake Bay, without a significant isolating sill, increasing the probability that hypoxic bottom waters from Chesapeake Bay can enter the Patuxent estuary.

Below we review four topics related to the eutrophication of these two systems. First, we evaluate four potential causes of cultural eutrophication. Second, we evaluate the magni-

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tude of cultural eutrophication. Third, we describe the effects of cultural eutrophication on the estuarine systems. Fourth, we review progress towards restoration of the estuaries.

Results and discussion

Causes of cultural eutrophication—We consider four potential causes of cultural eutrophication in these two systems: (1) fertilizer applications on agricultural lands, (2) wastewater from the increasing human populations, (3) N deposition from the atmosphere, and (4) advection of Chesapeake Bay water into each tributary.

Fertilizer applications on agricultural lands—Applications of fertilizers to croplands increased substantially after 1950 (Bohlke and Denver 1995). As application rates increased through the 1960s and 1970s, one of the results was dramatically increased crop yields (Benitez 2002; Fig. 2), greatly enhancing both the productivity and efficiency of large-scale agriculture. An unintended consequence of fertilizer applications was enrichment of surface unconfined aquifers in agricultural areas with nitrate (NO_3^-). Infiltration of rain through the N-enriched root zone of cropland soils (typically the top 0.5 m) is responsible for the high nitrate in groundwater on the Mid-Atlantic Coastal Plain (Bohlke and Denver 1995; Staver and Brinsfield 1998). The same pattern has been observed widely in Europe (Forsberg 1994; Iversen et al. 1998) and elsewhere. The root zone of croplands is relatively isolated hydrologically during the summer; high temperatures at this time of year result in evapotranspiration rates that typically exceed rainfall rates and reduce soil moisture levels. However, in the fall after crop harvests, as decreasing temperatures and evapotranspiration rates permit increased water storage in the root zone, rain events infiltrate deeper than the root zone and percolate to shallow groundwaters. On croplands without winter cover crops, nitrate concentrations in initial root zone leachate may reach concentrations between 3 mmol L^{-1} and 14 mmol L^{-1}, gradually declining to 0.14–1.40 mmol L^{-1} during the winter as excess N is flushed from the root zone (Staver and Brinsfield 1998). As a result, shallow groundwaters beneath fertilized crop fields on the Mid-Atlantic Coastal Plain are typically enriched with nitrate to concentrations of 0.6–2.1 mmol NO_3 L^{-1} (Bohlke and Denver 1995; Staver and Brinsfield 1998). These concentrations render shallow groundwaters in most agricultural areas undrinkable according to the 0.71-mmol NO_3 L^{-1} (= 10 mg NO_3–N L^{-1}) drinking water standard used by the US EPA (Title 40 of the U.S. Code of Federal Regulations, part 141 section 62). Similarly, average NO_3 concentrations in shallow groundwaters in Denmark are 0.5–1.0 mmol L^{-1}, making shallow groundwater in many areas unsuitable for drinking (Iversen et al. 1998).

Nitrate-rich groundwater imposes a distinct agricultural signature on stream chemistry on the Mid-Atlantic Coastal Plain and Piedmont provinces. Groundwater typically contributes >50% of streamflow as baseflow (e.g., Lee et al. 2001). Even allowing for losses of nitrate as groundwater transits to streams, the percentage of agricultural land use within a basin can be used to predict the total N concentration of a stream draining from that basin (e.g., Jordan et al.
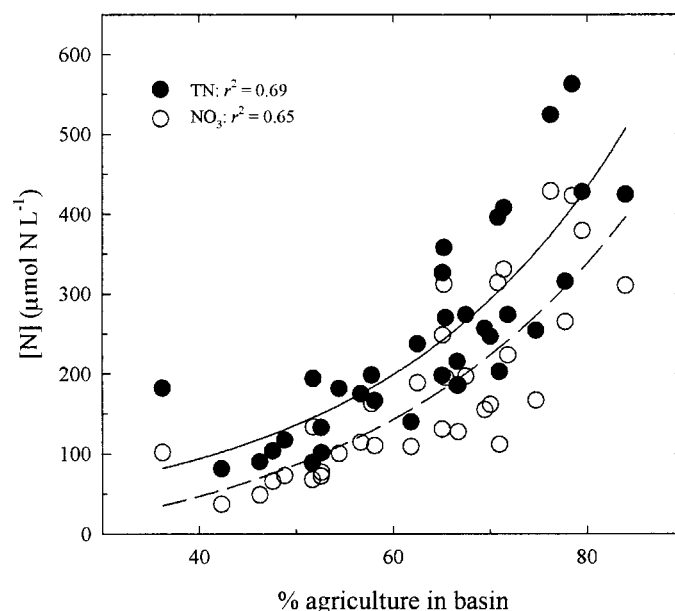
Fig. 3.   The effect of agricultural land use on annual average concentrations of nitrate (NO_3) and total N (TN) in second- and third-order streams draining small watersheds (6–55 km²) within the Choptank River basin.

1997; Weller et al. 2003). The total N in agriculturally dominated basins is typically >75% nitrate (e.g., Fisher et al. 1998), which is largely derived from fertilizer applications on croplands (Fig. 3).

Application of fertilizers to soils has also increased soil P content and crop yields (Sims et al. 1998). In addition to fertilizers, soils may be amended with sewage sludge or manures from animal feeding and rearing operations (e.g., poultry, hogs). Due to the high cost of transport, cropland soils near facilities used for animal husbandry are often used for disposal of excess animal waste (Staver and Brinsfield 2001), leading to soil P levels that may greatly exceed those required to saturate plant uptake (Sims et al. 1998).

Erosion of P-rich soils results in episodic transport of P. In late winter and spring, when soils are usually water saturated, rain events on fallow croplands without winter cover crops may produce erosive overland flows that carry high P concentrations to streams. Crop residues, applications of manure, and no-till agricultural practices exacerbate this effect by concentrating soluble and erodible P in surface soils (Staver and Brinsfield 1998, 2001). Following overland flows, stream P concentrations increase dramatically (Fig. 4). Particulate and organic N may increase during rain events, but nitrate usually declines due to dilution of groundwater nitrate (Fisher et al. 1998). The high flows and high P concentrations during storm events result in episodic transport of P, compared to the more constant delivery of N primarily in baseflow.

Increases in human populations and wastewater inflows—The increasing human populations in these basins generate increasing volumes of wastewater (Fig. 5). In the town of Easton, Maryland, in the Choptank basin, wastewater flows have increased slowly but significantly over the last 20 yr
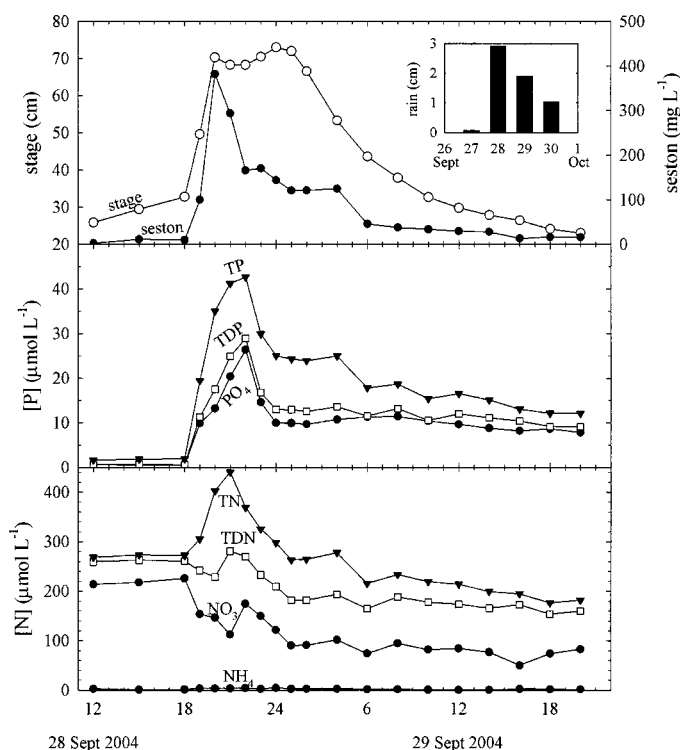
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Fig. 4.   Hydrograph from Norwich Creek in the Choptank River basin during a 5.9-cm rain event (inset graph) following an 8-d period without precipitation. Half of the rain occurred on 28 Sep 2004, resulting in the highest stage at the end of that day and in the early hours of 29 Sep 2004. P concentrations increased rapidly on the rising limb due to leaching and erosion by overland flow, whereas N concentrations decreased as high-nitrate groundwater was diluted by the lower N in overland flow. Data source: Suttton.

(Fig. 5F; obscured by common *y* axis for left and right panels); however, the addition of an overland-flow, tertiary-treatment system in 1987 significantly decreased P concentrations (Fig. 5B). As a result, fluxes of N and P have remained approximately steady or declined slightly, despite increasing volumes of wastewater. Similar management actions at other wastewater treatment plants in the Choptank basin have also resulted in reduced concentrations and approximately constant fluxes of N and P.

In the Patuxent basin, wastewater flows are both larger and increasing faster than in the Choptank basin (Fig. 5E). However, more aggressive and earlier implementation of P removal in 1986 and biological N removal (BNR) in 1991 have greatly reduced wastewater P fluxes and seasonally reduced N fluxes at many plants. At the Western Branch wastewater plant (Fig. 5C), representing 32% of the permitted wastewater flows into the Patuxent, BNR has not been implemented during winter months, when the treatment is slower and less efficient at low temperatures. However, if present discharge trends continue, fluxes from this plant with seasonal BNR will return to 1985 levels within 5 yr.

The sequential application of P removal and BNR have caused shifts in the molar N:P ratio of nutrient inputs from all sources to the Patuxent estuary. In the 1970s, when sewage dominated the inputs, the N:P was ~8:1. P removal in

the 1980s pushed the N:P to ~40:1, but BNR application in the 1990s resulted in N:P <20:1 (D'Elia et al. 2003). Although the Patuxent estuary has been characterized as P saturated, this conclusion was based on nutrient limitation bioassays obtained after 1990 (Fisher et al. 1999), when the N:P was relatively low. Earlier data obtained by D'Elia et al. (1986), when the N:P was higher, showed winter P limitation. This suggests that in the 1980s this estuary was at the threshold of experiencing seasonal P limitation when diffuse sources with high N:P dominated the inputs. In fact, a recent nutrient budget for the Patuxent estuary (Jordan et al. 2003) showed that reductions in point source inputs (Fig. 5) have caused diffuse sources of N and P to be of equal or larger magnitude, especially as forest is converted to residential areas. If N reductions continue in the Patuxent without additional P removal, excess P in the estuary can be expected to increase, with possible export to Chesapeake Bay.

Atmospheric deposition—The role of atmospheric deposition in cultural eutrophication has been extensively investigated. Once ignored in most nutrient budgets, studies by Paerl and colleagues clearly showed that rain could stimulate the growth of phytoplankton in marine areas where N was depleted (e.g., Paerl et al. 1999). However, for inshore coastal waters and land, other inputs are often more important than atmospheric deposition. In the mid-Atlantic region, atmospheric N deposition is ~1,000 kg km^{-2} yr^{-1} (Lee et al. 2001) and is small compared to other sources. Whereas atmospheric N deposition is distributed evenly at rates of ~1000 kg N km^{-2} yr^{-1}, N fertilizer applications are concentrated on croplands at rates of 5,000–15,000 kg N km^{-2} yr^{-1} (e.g., Staver and Brinsfield 1998). These areas, and areas receiving human and animal wastes, export N at rates usually >1,500 kg N km^{-2} yr^{-1}, with nitrate concentrations in groundwater exceeding 500 fmol NO$_3$ L^{-1}. In contrast, forests and natural grasslands, which usually receive only atmospheric deposition as an N input, typically export small amounts of N (<200 kg N km^{-2} yr^{-1}; e.g., Beaulac and Reckhow 1982; Clark et al. 2000) and have low nitrate in groundwater (<7 fmol NO$_3$ L^{-1}; Bohlke and Denver 1995). This implies that, in the absence of other anthropogenic inputs, forest vegetation and soils intercept much of the atmospheric inputs via plant uptake and/or soil denitrification. Exceptions include portions of northwestern Europe with atmospheric deposition rates 2–3 times higher (Forsberg 1994) and forested ridges subject to insect defoliation (Eshleman et al. 1998). These conditions lead to N saturation and high N export (Aber et al. 1998). Since atmospheric N deposition in the mid-Atlantic region occurs at low rates compared to fertilizer applications on croplands and is captured by most forests, atmospheric N sources are most important in urbanized watersheds, where impervious surfaces can deliver N to surface waters with relatively little loss. However, even in the relatively urbanized Patuxent watershed, impervious surfaces in urban areas account for only 3–6% of the basin.

Atmospheric N deposition directly to estuarine waters has been shown to account for <10% of N sources in several Chesapeake tributaries (Boynton et al. 1995; Lee et al. 2001; Jordan et al. 2003). Direct experimental evidence has also
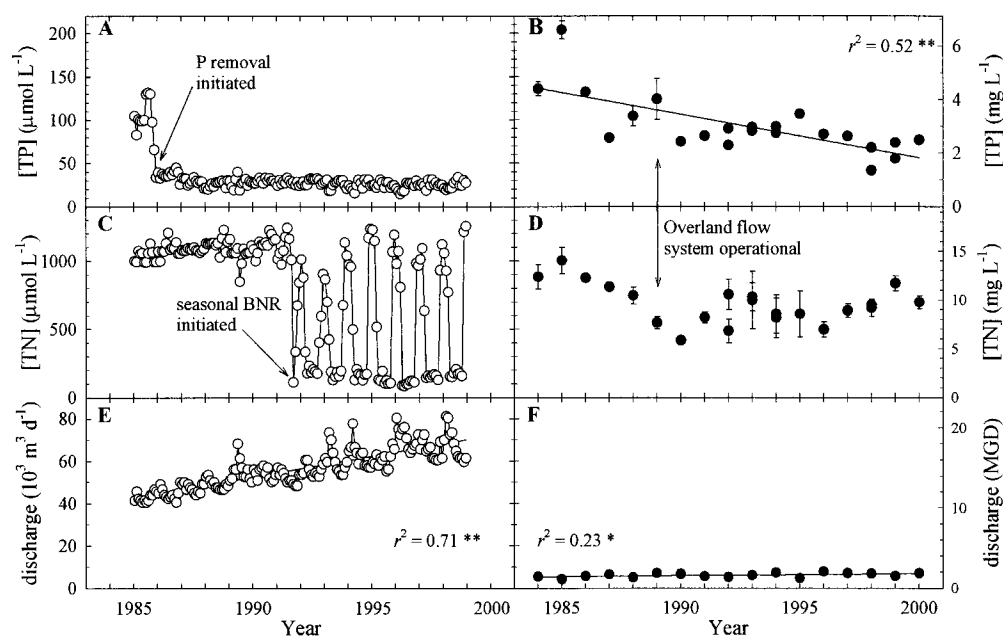
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Fig. 5.   Discharge rates (E) and concentrations of TN (C) and TP (A) in wastewater from the Western Branch wastewater treatment plant in the Patuxent River basin and from the Easton, Maryland, wastewater treatment plant in the Choptank River basin (B, D, F). Panels with lines indicate significant trends, with r^2 values shown. Significance is indicated as NS ($p > 0.05$), * ($0.05 > p > 0.01$), and ** ($p < 0.01$). Data sources: D'Elia et al. (2003); T. R. Fisher (unpubl. data); Easton, Maryland, Utilities Commission.

shown that such inputs to estuarine waters have limited potential to stimulate phytoplankton production. At five stations spanning the main salinity gradient of Chesapeake Bay, experimental rain additions in monthly bioassays were scaled to mimic the effects of a relatively large storm event (4 cm) on the upper mixed layer of these stations with depths of 2–7 m. These scaled rain additions provided a small stimulus to phytoplankton growth in August when N was most depleted and most limiting, but had no measurable effect at other times of year (Fisher et al. 2001). These results in August were similar to those of Paerl et al. (1999); however, the effect of a large storm event in August was small and of short duration compared to nutrient additions simulating the larger input of N from rivers (Fisher et al. 2001).

Much less information is available for atmospheric P deposition, which can be locally important, especially if dust sources are nearby (Mignon and Sandroni 1999). However, most measurements outside of regions with significant dust sources (e.g., the mid-Atlantic region) have P concentrations in precipitation of <0.5 fmol L^{-1} and deposition rates <10 kg P km^{-2} yr^{-1} (Lee et al. 2001). In contrast, P fertilizer and manure application rates in the mid-Atlantic region are 2,000–5,000 kg P km^{-2} yr^{-1} (Sims et al. 1998; Staver and Brinsfield 2001). We conclude that atmospheric deposition is a minor source of P compared to other inputs in these estuaries, and that the importance of atmospheric N deposition depends on local land use.

Exchange with Chesapeake Bay—Water quality in the Patuxent and Choptank estuaries is potentially affected by exchanges with Chesapeake Bay. Although flushing may be

seen as likely to remove nutrients and organic matter from estuaries, this need not be the case when the body of water at the seaward margin is also eutrophic. Both nutrients and organic matter may be imported at the seaward margin of an estuary if gradients in concentration favor that direction of exchange, as illustrated by the range of mixing profiles observed by Eyre (2000) in Australian estuaries. In the Patuxent and Choptank estuaries, net landward exchange of nutrients is relatively more likely because of the long-term increase in nutrient inputs to Chesapeake Bay via the Susquehanna River (Hagy et al. 2004). Thus, while it is likely that local inputs are the most important contributor to long-term change in the tributaries, the potential effect of exchanges with Chesapeake Bay should not be ignored.

Boynton et al. (1995) estimated exchanges of N and P at the seaward margins of the Patuxent and Choptank estuaries with a mass balance approach. This approach estimates the seaward exchange as inputs from land (diffuse and point sources) and atmosphere minus internal storage and losses (burial, denitrification, and fish harvests). We have updated the results for the Choptank estuary in Table 2 with the inputs estimated by Lee et al. (2001). This updated approach shows a consistent pattern of net export of N (11–35% of inputs) from both estuaries to Chesapeake Bay and net import of P (42–124% of inputs) into each estuary from Chesapeake Bay. This mass balance approach suggests that substantial amounts of P (approximately equivalent to basin and atmospheric inputs) are imported from the eutrophic Chesapeake Bay into each tributary. The mass balance approach, while potentially useful, is subject to error propagation, and the seaward exchange is the small difference between two

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Table 2. Nutrient mass balances in the Choptank and Patuxent estuaries. The Patuxent data and all estuarine loss data are the same as those reported by Boynton et al. (1995); however, terrestrial inputs of N and P for the Choptank estuary have been updated with the modeling estimates of the hydrochemical model Generalized Watershed Loading Functions applied to the Choptank (Lee et al. 2001). The net residual equals total inputs minus total losses, both in absolute units and expressed as a percentage of total inputs.

| | Estuary | | | |
| | Choptank | | Patuxent | |
Process	10^6 kg N yr^{-1}	10^3 kg P yr^{-1}	10^6 kg N yr^{-1}	10^3 kg P yr^{-1}
Terrestrial diffuse	2.23	28.4	1.55	115
Atmospheric deposition	0.16	1.1	0.16	9.0
Point sources	0.10	28.1	0.64	53.0
Total inputs	2.49	57.6	2.36	177
Denitrification	1.37		0.54	
Burial	0.73	120	0.92	250
Fisheries	0.21	9.0	0.06	2.0
Total losses	2.31	129	1.52	252
Net residual	0.27	−71.4	0.835	−75
% of inputs	11	−124	35	−42

large values (inputs − losses), each with potentially large errors. An additional shortcoming is that this approach describes N and P in aggregate over a time scale of 1 yr or more. Seaward exchanges of N and P may vary seasonally and by chemical species (e.g., inorganic vs. organic).

Nutrient exchanges may be quantified on shorter time scales and by chemical form through the use of box models. These box models use the observed salinity distribution to infer physical transport, which is then applied to the ob-

served concentrations of N and P. This has been done here only for the Patuxent estuary, with the box model described by Hagy et al. (2000). The box model results indicated that organic N was consistently exported from the Patuxent estuary to Chesapeake Bay, while dissolved inorganic nitrogen (DIN) was generally imported into the Patuxent (Fig. 6). Ammonium (NH_4^+) was primarily imported in summer, and nitrate was predominantly imported in cooler months. Because the outward organic N fluxes were larger than those of DIN, total nitrogen (TN) exchanges were primarily outward, consistent with the mass balances in Table 2. Annual TN export from the Patuxent estuary was estimated to be 0.37×10^6 kg N yr^{-1}, equivalent to an export of 16% of the estimated inputs from the watershed and atmosphere, which is similar to the value estimated with mass balances (Table 2).

The box model results for P exchanges at the mouth of the Patuxent estuary also showed consistent seasonal export of all forms of P. The estuary exported both total phosphorus (TP) and PO_4 throughout the year (Fig. 6), with most fluxes occurring during fall, when the estuary is commonly flushed rapidly by higher salinity water from Chesapeake Bay (Hagy et al. 2000). On an annual basis, the TP efflux was estimated to be 0.064×10^6 kg P yr^{-1}, equal to 36% of inputs from the watershed and atmosphere (Table 2), of which the PO_4 efflux contributed 0.021×10^6 kg P yr^{-1}. These results conflict with the P mass balance in Table 2, which predicts net import of P into the Patuxent estuary; however, the results are consistent with the nutrient limitation results of Fisher et al. (1999), who showed P saturation in the Patuxent with respect to phytoplankton growth.

Overall, these results suggest that tributaries of Chesapeake Bay act as biological reactors. DIN and phosphate appear to be imported seasonally into the Patuxent estuary, with outwelling of organic N and P into Chesapeake Bay. We suspect the same may be true of the Choptank estuary. These mass balance and box modeling results illustrate the unresolved difficulties in estimating how the lateral tributaries of Chesapeake Bay interact with the main bay. However, the estimated magnitude of the net influxes of DIN into the
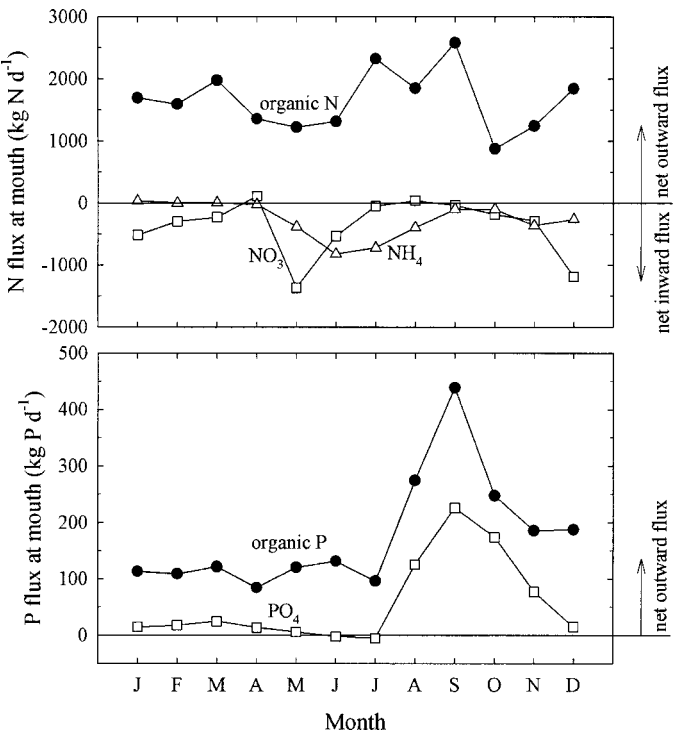

Fig. 6.   Average fluxes of N and P at the mouth of the Patuxent River estuary (Sta. LE1.4) for 1984–2000 computed using the box model approach of Hagy et al. (2000). Positive flux values are outward into Chesapeake Bay; negative fluxes are landward into the Patuxent River estuary.

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Patuxent estuary that were obtained from the box model results (Fig. 6) are small (0.18×10^6 kg N yr^{-1}) compared to the average loading rates of 2.5×10^6 kg N yr^{-1} from all terrestrial and atmospheric sources (Table 2). We therefore conclude that exchanges of these tributaries with Chesapeake Bay result in relatively small inputs of inorganic N and somewhat larger exports of organic N and P.

Magnitude of cultural eutrophication

Agricultural activities and disposal of human waste appear to be the dominant sources of N and P that cause cultural eutrophication in these two estuarine systems. We explore the magnitude of the anthropogenic supply of N and P compared to undisturbed conditions in order to estimate how much anthropogenic inputs have to be reduced to achieve water-quality goals in these regions. These estimates can be difficult to obtain because monitoring data are almost always inadequate. Monitoring programs typically begin after water-quality problems occur, and initial conditions are usually poorly characterized. However, we can estimate the magnitude of cultural eutrophication indirectly by using nutrient yield coefficients and hydrochemical modeling.

Nutrient yield or export coefficients represent land use–specific losses of N and P per unit area and time (kg N or P km^{-2} yr^{-1}). Forests have the lowest export coefficients compared to other land uses, and more intensive land uses such as urban, agriculture, and animal feedlots export N and P at rates that are 10–1,000 times higher than those of forests (Beaulac and Reckhow 1982; Fisher et al. 1998; Clark et al. 2000). While the highest rates may occur only over small areas of a watershed (e.g., a feedlot), the effects can still be large because of the magnitude of the nutrient yields compared to less disturbed conditions.

Agricultural, urban, and forested land uses account for most of the area of the Choptank and Patuxent basins. Using estimated nutrient yield coefficients for each land use provides a quantitative estimate of the effect of anthropogenic diffuse sources (not including point sources). Forests export N and P at rates of ~100 kg N km^{-2} yr^{-1} and ~10 kg P km^{-2} yr^{-1}, respectively (Clark et al. 2000), whereas urban lands export N and P at rates of ~1,000 kg N km^{-2} yr^{-1} and ~200 kg P km^{-2} yr^{-1}, respectively, and agricultural land uses export 500–1,600 kg N km^{-2} yr^{-1} and 100–400 kg P km^{-2} yr^{-1}, respectively (Beaulac and Reckhow 1982; Fisher et al. 1998). This comparison suggests that anthropogenic land uses export diffuse sources of N and P at rates that are 10–20 times higher than those of forests.

Similar conclusions can be drawn from application of the hydrochemical model GWLF (generalized watershed loading functions) to the Choptank basin. GWLF was validated in the Choptank basin by Lee et al. (2001) and has been applied to historical records of land use, human populations, and fertilizer applications in the Choptank for 1850–2000 (Benitez 2002). At the decadal time scale, the model has cumulative validation errors of 1–3% for stream flow, 5–13% for total N export, and 36–38% for total P export (Lee et al. 2001).

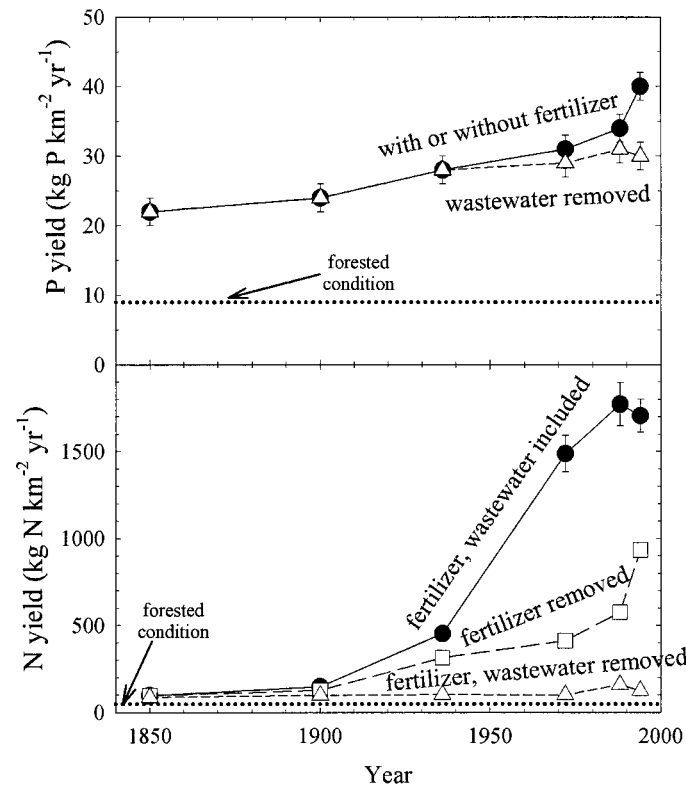To estimate the magnitude of the historical changes in

Fig. 7.   Results from the hydrochemical model GWLF applied to Tilghman Island, an urbanizing area in the Choptank River basin. Model scenarios in which fertilizers or human waste were set to zero in the model were used to estimate the historical effects of these anthropogenic processes on export on N and P. Data sources: Benitez (2002), Benitez and Fisher (2004).

export, we use the results from Tilghman Island, a part of the Choptank basin between the Choptank estuary and Chesapeake Bay. This part of the Choptank was initially dominated by agriculture (~70%) after colonization in the 17th century; however, urbanization claimed 29% of the island in the 20th century, and agriculture continues to dominate land use (46%; Benitez and Fisher 2004). As a result of these historical changes, Tilghman Island provides an example of growing small towns driving human population increases. The N and P export coefficients estimated by GWLF for conditions prior to colonization (50 kg N km^{-2} yr^{-1} and 9 kg P km^{-2} yr^{-1}) are similar to those estimated by Clark et al. (2000) for forests (100 kg N km^{-2} yr^{-1} and 10 kg P km^{-2} yr^{-1}). The model results using historical land use, human populations, and fertilizer applications for 1850–2000 are shown in Fig. 7. The upper solid line in each panel represents the 10-yr mean of model results averaged over hydrologic variability for the conditions in each time period; we have also run model scenarios in which fertilizers and human waste were set to zero within the model. These model scenarios included all anthropogenic effects or eliminated the application of fertilizers and/or production of human wastewater in the model, while leaving land use, soil, and human population patterns intact.

The urbanizing Tilghman Island, with its current human population density of 118 km^{-2}, showed strong effects of
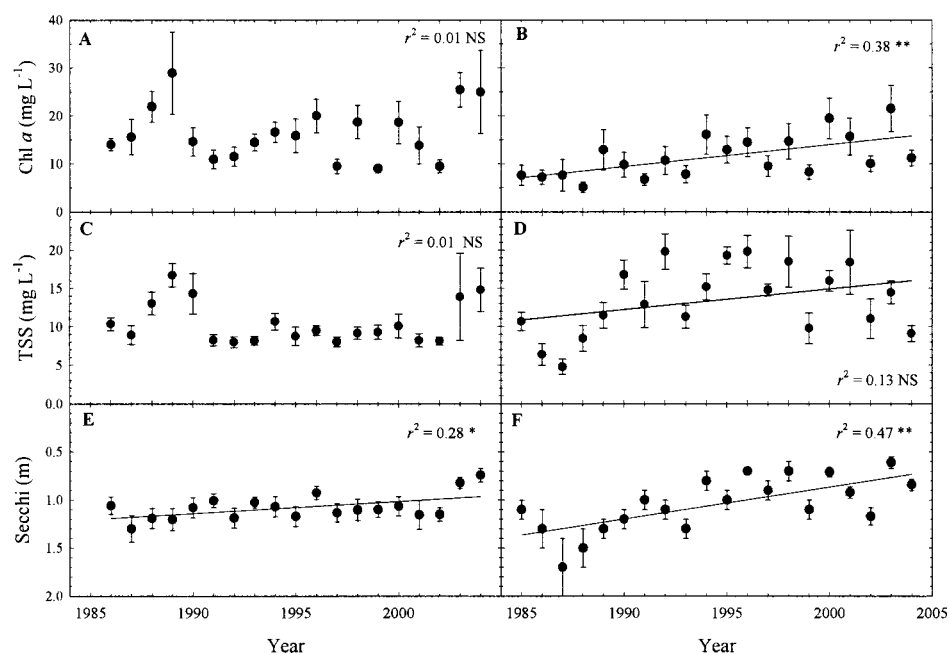
Fisher et al.

Fig. 8.   Annual average values of chlorophyll a (Chl *a*), total suspended solids (TSS), and Secchi depth (m; note the inverted scale) in the mesohaline Patuxent estuary (Sta LE1.1; panels A, C, E) and the mesohaline Choptank estuary (Sta. ET5.2; panels B, D, F). Significant interannual trends are shown as lines; significance is indicated as in Fig. 5. Data source: EPA Chesapeake Bay Program.

both wastewater and fertilizer applications. Wastewater production strongly influenced P export, although P fertilizer effects were essentially undetectable in the model results. From 1850 to 2000, P export appears to have doubled and is currently ~4 times the estimated forested condition (10 kg P km^{-2} yr^{-1}; Fig. 7). For N export, withholding N fertilizer applications and human waste in the model resulted in large reductions in the N yield coefficients, reducing contemporary N export from ~1,800 kg N km^{-2} yr^{-1} to ~100 kg N km^{-2} yr^{-1}. This is an anthropogenic effect of ~18, similar to the anthropogenic effect estimated using empirical nutrient yield data above (10–20). For both N and P, current nutrient yields estimated by GWLF are 20 and 4 times higher (respectively) than those from forests. Fertilizer had little influence on P export coefficients, although N export was strongly affected.

Effects of eutrophication on the estuaries

Turbidity and phytoplankton—Enhanced inputs of N and P are known to increase the turbidity and phytoplankton populations in estuarine systems (Vitousek et al. 1997). In the Choptank estuary, historical increases in N and P inputs are indicated in model results (Fig. 7), as well as in the shorter record of observations (Fisher et al. 1998). Since 1985, annual average chlorophyll *a* (Chl *a*) has doubled and water clarity has decreased in the mesohaline region (Fig. 8, right panels). Furthermore, much of the variability shown in the right panels of Fig. 8 is correlated with river flow ($r^2 = 0.6$), which suggests that river-borne nutrient inputs have a prominent role in the increasing Chl *a* in this system. These trends illustrate the slow, variable rates of eutrophication that re-

duce water quality. Without detailed monitoring data such as those in Fig. 8, these slow and erratic trends are noticed by local observers only after one or two human generations.

Comparable data for a mesohaline Patuxent station show few significant trends (Fig. 8, left panels). Chl *a* and total suspended solids (TSS) were variable, with a maximum in 1989, when an unusually large discharge of river water occurred in late spring. Lower values of Chl *a* and TSS occurred in the early 1990s, perhaps in response to seasonal N reductions from wastewater plants (Fig. 5), but annual average values of Secchi depth have been slowly decreasing since 1986. Although the reductions in N concentrations of point sources may have had an effect initially in the early 1990s as Chl *a* declined, these short-term improvements may have been lost because of continued increases in urbanization and wastewater flows (Fig. 5).

Loss of submerged aquatic vegetation—Aerial photographs from 1938, 1952, and 1964 show that SAV was once abundant and widely distributed in the Patuxent estuary (Stankelis et al. 2003). By 1978, SAV coverage in the Patuxent estuary had already declined to <10% of its former abundance. SAV coverage prior to 1978 has not been quantified for the Choptank estuary, but visual inspection of 1938 aerial photographs of the Choptank showed SAV to be as widespread and abundant as it was in the Patuxent.

Annual surveys since 1984 reveal low and variable abundance of SAV in the two estuaries. A common external cause could be interannual changes in freshwater inflows and nutrient inputs, although SAV loss was much more severe in the Patuxent estuary than in the Choptank. The largest areas of SAV in recent years in the Choptank have been about
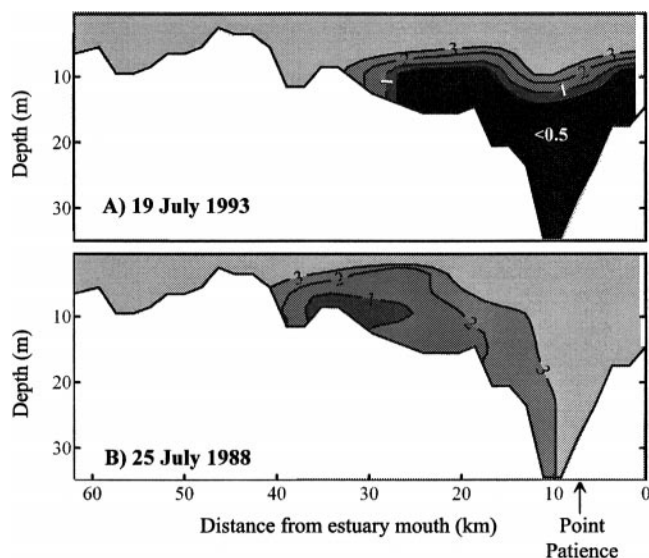
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Fig. 9.   Distribution of dissolved oxygen (DO; mg O_2 L^{-1}) along the major axis of the Patuxent River estuary on two dates that illustrate the two kinds of characteristic DO distributions frequently observed during 1986–2001. The DO distribution on 19 July 1993 illustrates the anoxia which is associated with occasional intrusions of hypoxic water from Chesapeake Bay into the Patuxent River estuary. The distribution on 25 July 1988 illustrates the more common pattern, in which hypoxia develops locally in the landward-flowing bottom. Data source: EPA Chesapeake Bay Program and interpolation algorithms described by Hagy et al. (2000).

27% of the Chesapeake Bay Program's restoration goal, while only 1.4% has been realized in the Patuxent because of water-column light attenuation in the upper estuary and light attenuation by epiphyton in the lower estuary (Stankelis et al. 2003).

Oxygen depletion in bottom waters—Depletion of dissolved oxygen (DO) from bottom waters is common during summer in Chesapeake Bay and its tributary estuaries (Kemp et al. 1999; Hagy et al. 2004). Hypoxia (<2 mg DO L^{-1}) develops locally in the Patuxent in the landward-flowing bottom layer in the middle of the estuary (Fig. 9B) due to respiration of excess, labile organic matter sedimented from surface waters that are enriched by local nutrient inputs from the Patuxent basin. Less frequently, hypoxic water is advected into the estuary from Chesapeake Bay. The latter events can be identified by horizontally uniform distributions of temperature and salinity, which characterize strong inflows of bay water (Hagy et al. 2000), and by hypoxic water extending from the mouth of the estuary into the middle estuary (Fig. 9A). These events are associated with upwelling on the western side of Chesapeake Bay caused by strong southerly winds (Breitburg 1990). Inflowing hypoxic bay water was observed on 26% of summer sampling dates (27 of 105) during 1985–2000, whereas hypoxia was observed in the middle estuary on 72% of summer sampling dates (75 of 104 observations).

Much of the Patuxent estuary's bottom waters are seasonally hypoxic. Approximately 25% of the volume of the Patuxent estuary is in the bottom layer, effectively isolated by
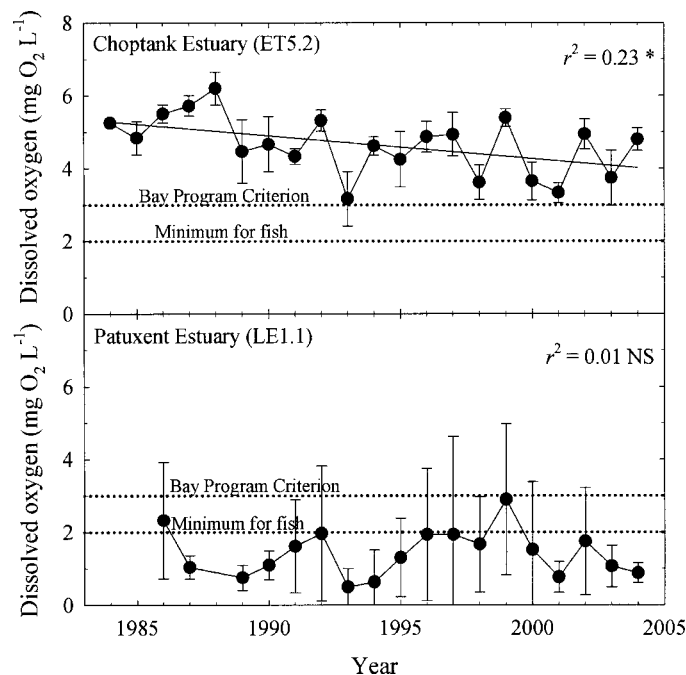

Fig. 10.   Summer (June–August) averages of dissolved oxygen (DO) concentrations in bottom waters of mesohaline Sta. ET5.2 in the Choptank River estuary and mesohaline Sta. LE1.1 in the Patuxent River estuary. The dotted lines at 2 and 3 mg O_2 L^{-1} represent the minimum DO required for survival of fish and the Chesapeake Bay criterion for 30-d average DO concentration in waters of the bay to which the deep-water seasonal fish and shellfish designated use applies. This designated use is established to protect bottom-feeding fish, crabs, oysters, and some other important species (US EPA 2003). Data source: EPA Chesapeake Bay Program.

the pycnocline from DO replenishment. During June–August, hypoxic water (DO <2.0 mg O_2 L^{-1}) averages 63% of bottom-water volume. Hypoxia is clearly the norm for bottom waters in the mesohaline Patuxent in summer, whether it is internally generated or advected from the main bay (Fig. 9). Although the Chesapeake Bay mainstem experiences the most severe, frequent, and extensive hypoxia, the Patuxent estuary is the second most hypoxic of the tributary estuaries, after the Potomac (Kemp et al. 1999; Hagy et al. 2004).

In contrast, hypoxia is virtually absent from the Choptank estuary. The few hypoxic events that have been observed in the Choptank have been attributed to advection of deep, hypoxic water from Chesapeake Bay over the relatively shallow sill at the mouth of the Choptank. This requires strong northerly winds and other conditions which have not yet been identified (Sanford and Boicourt 1990). Bottom DO <2 mg L^{-1} was found only 5% of the time at two monitoring stations in the Choptank estuary measured biweekly during summer since 1984. Near the mouth of the estuary, DO was most often near saturation (6–7 mg O_2 L^{-1}), whereas in deeper water (~12 m) further up the estuary, DO was 4–6 mg O_2 L^{-1}, indicating a saturation deficit. A time-series analysis shows a significant decline in bottom-water DO during the last 20 yr (Fig. 10), consistent with the declining water quality of surface waters (Fig. 8). If this trend continues, within a decade the average summer DO will be 2–3 mg O_2 L^{-1}.
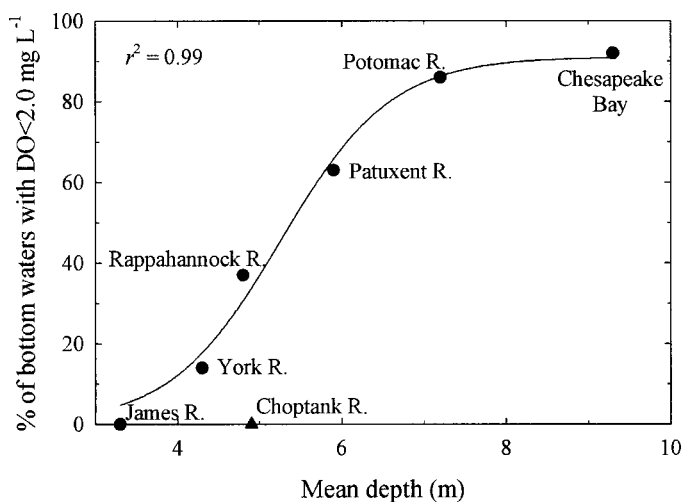
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Fig. 11.   The relationship between hypoxic volume of bottom waters and mean depth in regions of the Chesapeake Bay. Mean depths were computed for smaller, salt-intruded areas to compare local differences. The sigmoid curve and correlation coefficient shown in this figure exclude the Choptank; including the Choptank decreases r^2 to 0.91 ($p < 0.01$). Data source: Kemp et al. (1999).

The differences in bottom-water DO concentrations between these estuarine systems are related to nutrient loading rates and physical characteristics (Tables 1 and 2). Kemp et al. (1999) described the extent of hypoxia in Chesapeake Bay and six tributary estuaries, and the Choptank estuary had the lowest areal N and P loading rates (Table 1) and the least hypoxia. The James estuary, however, which also has very little hypoxia, has N and P loading rates comparable to those of the Patuxent estuary, but a shallow mean depth. Within the Chesapeake data reported by Kemp et al. (1999), the mean depth predicted the extent of hypoxia among these estuaries much better than nutrient loading rates (Fig. 11). Although the Choptank still has less hypoxia than estuaries of similar average depth (e.g., the Rappahannock), the decreasing DO in bottom waters (Fig. 10) suggests that the Choptank estuary could approach hypoxic volumes suggested by its physical dimensions in a decade.

There are several physical factors that contribute to lower incidence of hypoxia in the Choptank. The deepest region ($\sim$26 m) is separated from the mainstem bay by a broad, shallow region near the mouth which is usually mixed by winds and tides. This isolates the estuary from high-salinity bay water, reducing the potential for strong salinity stratification and hypoxic inflows. In contrast, the Patuxent estuary has a deeper, narrow region in close proximity to the bay. Therefore, not only does the Patuxent estuary have higher nutrient loading compared to the Choptank estuary (Table 2), but the Patuxent also is physically configured to have more isolated bottom waters with high potential for local hypoxia and hypoxic inflows.

Restoration progress and future management

There is a history of efforts to restore the Patuxent, as well as estuaries elsewhere in the world. The earliest warn-ings of cultural eutrophication in the Patuxent occurred in the 1970s, based on changes in water quality and habitat conditions (Mihursky and Boynton 1978). By the early 1970s, summer algal blooms were common, SAV communities had largely disappeared, and hypoxia was a regular feature. Despite the warnings, development continued in the basin, and sewage discharges increased. In the early 1980s, the southern counties of the basin brought suit against the State of Maryland and the U.S. Environmental Protection Agency, claiming that water-quality management plans were flawed and would result in further degradation. The southern counties won this court action, and sewage treatment plants were upgraded to remove first P (1986) and later N (1991–1993; *see Fig. 5*). N and P loads at the head of the estuary were reduced as a result of these actions by <20% (D'Elia et al. 2003), far short of the 40% goal (US EPA 2003).

The wastewater load reductions have been counteracted by continued development and associated land-use change within the basin. These changes have resulted in increasing volumes of sewage discharge leading to higher N and P loads despite decreasing concentrations as a result of P removal and BNR (Fig. 5). Furthermore, there have been increased diffuse sources of N and P during the wet 1990s. These processes largely erased any effects of point source reductions (Jordan et al. 2003), and computations of nutrient transport from the upper to the lower estuary have remained largely unchanged.

As a result of these counteracting processes, there was little indication of improved water or habitat quality in the Patuxent estuary. Water-column Chl *a,* TSS, Secchi depth, and deep-water DO have remained largely unchanged despite the partial institution of advanced wastewater treatment (Figs. 8, 10). Efforts to restore SAV in the mesohaline portion of the estuary with plantings of shoots and dispersal of seeds have largely failed. Stankelis et al. (2003) reported that, in most cases, severe epiphytic fouling of macrophyte leaves, leading to inadequate light, was the primary cause of plant mortality. In addition to turbidity levels that are still too high to support adequate growth of SAV, intensive grazing by invasive waterfowl (mute swans) and resident species (skates and rays) also contributed to these restoration failures. The current state of affairs has been characterized as "holding the line," but failing to achieve nutrient and sediment reductions required for clear and sustained improvements (Horton 2003).

The single, clear success in the Patuxent is a resurgence of SAV in the upper portion of the estuary (Fig. 12). A rapid expansion of macrophytes occurred following reductions in N point sources, but there was little response to the earlier P reductions. SAV in the turbid upper portion of the estuary has a slim margin for survival; however, these communities have colonized the very shallow margins of the tidal river at depths <0.5 m. At these shallow depths, a reduction in epiphytic fouling due to N-load reductions may have been enough to improve light availability, favoring survival and expansion. This observation emphasizes the responsiveness of these estuarine systems if sufficient water-quality changes occur.

In the Choptank estuary, the trajectory of eutrophication is similar but at an earlier stage. N inputs to both estuaries
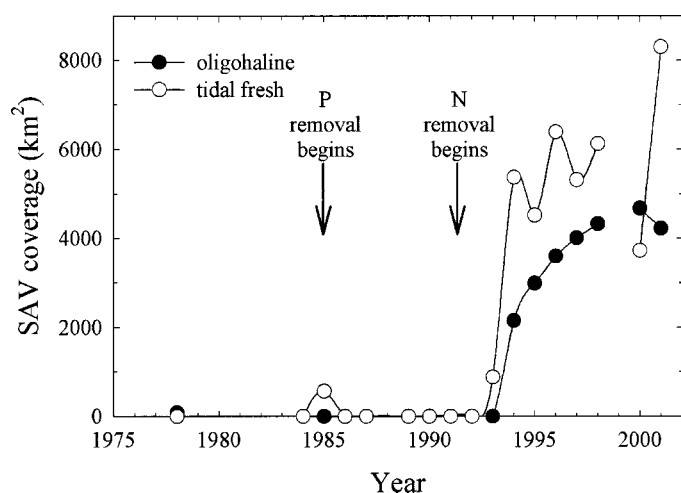
Cultural eutrophication of estuaries

Fig. 12. Interannual variations in abundance of submerged aquatic vegetation (SAV) in the upper Patuxent River estuary for 1978–2001. Data source: Boynton et al. (unpubl. data).

are currently very similar, while P inputs to the Patuxent are three times higher than those of the Choptank (Table 2). However, the greater surface area, greater volume, and smaller deep volumes of the Choptank estuary have limited the effects of these inputs (Fig. 11). Nonetheless, water-column Chl *a* and turbidity are increasing (Fig. 8), and summer DO in bottom waters is decreasing (Fig. 10). SAV abundance is higher than in the Patuxent, but lower than the potential habitat allows in this shallow estuary. These trends indicate that current efforts to improve water quality in the Choptank are failing to prevent further degradation.

These estuarine responses to load reductions indicate that further reductions in nutrient inputs, targeting diffuse as well as point sources, are required to achieve water-quality goals in both estuaries. Diffuse sources in particular will need to be addressed more aggressively, particularly in the Choptank but also in the Patuxent, a basin with substantial point sources. Progress to date has been slow and insufficient, and Ernst (2003) has articulated the frustration concerning slow progress and marginal efforts, suggesting that the cures for most of these water-quality and habitat problems are well known and that the public has lacked the social and political will to implement them.

We have shown above that nutrient inputs to these systems have increased by a factor of 4–20 during the last 100 yr. For comparison, recent reductions in the Patuxent point sources represent ~18% of the total basin inputs, and inputs to the Choptank are still increasing (Fisher et al. 1998), despite the commonly quoted goal of the Environmental Protection Agency Chesapeake Bay Program for 40% reductions of nutrient inputs. It is clear that considerably more effort and resources should be directed towards reductions in nutrient inputs in order to improve water clarity, restore SAV, and reduce the volume of hypoxic bottom waters. We suggest widespread implementation of winter cover crops, restoration of forested stream buffers, and better animal manure management to reduce inputs from agricultural areas (Staver and Brinsfield 1998) and year-round, advanced wastewater treatment at all sewage facilities within the Ches-

apeake basin. Local communities have never paid the full price for living in this region—depending on free ecosystem services to handle waste flows—and a degrading series of estuarine systems is the result.

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