TIDAL POTOMAC INTEGRATIVE ANALYSIS PROJECT

A Series of Reports on the Water Quality and Living Resources Responses to Management Actions to Reduce Nutrients in the Potomac River Estuary

FINAL

Interstate Commission on the Potomac River Basin
Suite 300, 6110 Executive Blvd.
Rockville, Maryland 20852

August 1999

ICPRB Report 99-4

GOAL

The intent of this project was to use long-term monitoring data to quantify how recent, anthropogenic changes in nutrient loadings to the Potomac Estuary have altered water and habitat quality and subsequently changed key biological communities. Based on this analysis, project participants hope to establish Chesapeake Bay Program management expectations for responses to and understanding of nutrient load reductions in the Potomac Estuary.

ACKNOWLEDGMENTS

This project was accomplished by a team of scientists and resource managers familiar with the diverse Potomac Estuary monitoring data. Project participants and contributors included, at one time or another, the following individuals (asterisked individuals were authors of report sections or report appendices):

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The project was supported with funds or in-kind matches from the Chesapeake Bay Program, the Interstate Commission on the Potomac River, Maryland Department of the Environment / Department of Natural Resources, George Mason University, the U. S. Geological Survey, and the U. S. Environmental Protection Agency. Thanks are owed to the panel of experts who participated in the April 24-25, 1997 workshop to review and interpret project results to-date.

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TIDAL POTOMAC INTEGRATIVE ANALYSIS PROJECT

A Series of Reports on the Water Quality and Living Resources Responses to Management Actions to Reduce Nutrients in the Potomac River Estuary

EXECUTIVE SUMMARY

Revised February, 2000

The Potomac River Estuary represents one of our nation's emerging success stories in water quality restoration. The estuary begins at the Piedmont fall-line in Washington, D.C. where it receives about 90% of its freshwater surface flow, and it travels 113 miles (182 kilometers) across the Mid-Atlantic Coastal Plain to the Chesapeake Bay. Historical accounts indicate the estuary was once a diverse, productive ecosystem and it yielded some of the largest United States East Coast fish harvests for many years in the 19th century. Over-harvested fisheries and water quality degradation began to emerge as problems about 150 years ago. Nutrient enrichment escalated sharply and spread downriver in the 20th century and the estuary was grossly polluted and considered a national disgrace by the 1960s, despite sporadic efforts to treat waste from the rapidly growing human population. Abundant nutrients, especially phosphorus, nitrogen, and organic matter, caused excessive algal growth, low dissolved oxygen levels and high turbidity.

Management Actions

The estuary condition stimulated management action and became a catalyst of the nation's Clean Water Act of 1972. Regional jurisdictions expanded and upgraded wastewater treatment facilities to reduce phosphorus and organic matter, and to convert ammonium nitrogen to less toxic nitrate. They implemented a ban on phosphate detergents, encouraged protection of wetlands and riparian buffers, advocated soil and runoff controls in the basin and established caps on nutrient loads. Recently, a new technology to remove nitrogen from wastewater was implemented at Blue Plains, the largest treatment plant in Washington, D.C. The impetus for many of these activities came from enforcement programs established as a result of the Potomac Washington Area Enforcement Conferences in 1957, 1958, and 1969 - 1970, from the National Pollutant Discharge Elimination System (NPDES) established in the early 1970s, and from a regional agreement signed by the Executive Council of the newly formed Chesapeake Bay Program (CBP) in 1983 which called for all levels of government to cooperatively reduce pollution in Chesapeake Bay tributaries, including the Potomac River. The Chesapeake Bay Agreement was updated in 1987 to include a 40% reduction goal in "controllable" nitrogen and phosphorus loadings from the tributaries by the year 2000. Over the past four decades, regional commitments have broadened from simply cleaning up the fouled waters of the Potomac Estuary, to restoring the potential of the estuary ecosystem to provide abundant food and habitat for fish and wildlife.

Report Objective

This report assesses the available Potomac monitoring data in order to a) quantify how anthropogenic nutrient reduction strategies have changed nutrient loads and altered water and habitat quality, and b) determine if key biological communities have responded positively to these changes. Emphasis was placed on trying to account for natural variability in the ecosystem caused by flow, season and salinity in order to identify changes that could be related to management actions. The authors calculated trends in nutrient loadings and ambient water quality for the "CBP Years" (i.e. 1985 - 1998) and for longer time periods. These trends were then compared to the status and trends of ecologically important groups, including underwater grasses, plankton and benthos. The authors attempted to answer two general questions: *Will the current nutrient reduction policies successfully return the Potomac water quality to desirable and habitable levels? Will current nutrient reductions sufficiently improve food and habitat conditions so that the impacts of other stressors–habitat loss, over-harvesting, exotic species, toxic pollutants--are overcome and a balanced, productive ecosystem with abundant fish and wildlife populations is restored?* The entire report consists of this synthesis of the project team's analyses, followed by detailed reports written by individual team members.

Pollutant Loads Delivered to the Estuary at the Fall-line

Surface freshwater flows ranging between 2.6 trillion and 24.9 trillion cubic meters per year enter the Potomac Estuary at the Piedmont fall-line. Annual pollutant loads delivered to the estuary at the fall-line are directly related to the amount of freshwater discharged each year and the pollutant concentrations in the water. Daily flow rates and pollutant concentrations measured near the fall-line were used to generate estimates of actual fall-line loads. Trends in major landscape inputs were also used to infer historical fall-line loads.

Long-term Perspective

Measured and inferred 20th century increases in fall-line loads of sediment and nutrients reflect changing land uses in the upper basin, including a 3-fold population increase to 16.6 people/km^2, major shifts in agricultural practices, increased meat and poultry production, and increased atmospheric deposition. Estimated landscape inputs from wastewater, agriculture and atmospheric deposition alone suggest total phosphorus loads at the fall-line rose at least 6-fold and total nitrogen loads rose at least 5-fold between 1900 and the mid-1980s. Actual water quality measurements made near the fall-line show a comparable 5-fold rise in the concentration of nitrate (a component of total nitrogen) since 1913. Total residue (a measure of dissolved and particulate matter) and sulfate loads doubled, potassium and chloride loads rose 10-fold, and water became more alkaline in the same period.

CBP Years

A close succession of wet years in 1993, 1994, 1996 and 1998 caused a 71% rising trend in Potomac fall-line freshwater flow during the "CBP Years," from 1985 to 1998, and offset stable or improving trends in fall-line pollutant concentrations and loadings. Trends in total nitrogen (TN) and nitrate-nitrite (NOx) loadings increased significantly when they might have remained unchanged in an average-flow period, and trends in sediments, total phosphorus (TP) and dissolved inorganic phosphorus (DIP) loadings remained unchanged when they might have decreased (Figure 1). A return to more average flows should produce lower sediment and phosphorus loads. Annual average loading rates ranged from about 42,000 to 171,000 kilograms TN per day and from 2,000 to 19,400 kilograms TP per day in the CBP Years. The large % change in NOx fall-line loadings (Figure 1) indicates nitrogen composition is shifting towards a higher proportion of NOx, possibly the result of nitrification at upriver treatment plants. Downward trends (~35%) in fall-line concentrations of ammonia and organic nitrogen compounds support this suggestion.

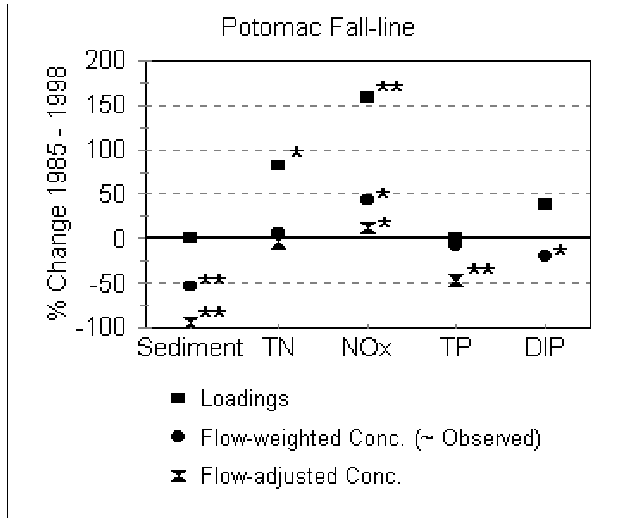

Figure 1. Comparison of % change in sediment, total nitrogen (TN), nitrate-nitrite (NOx), total phosphorus (TP) and dissolved inorganic nitrogen (DIP) at the Potomac fall-line station for the Chesapeake Bay Program years, 1985 - 1998. Freshwater flow at the fall-line increased 71% due to a close succession of wet years in 1993, 1994, 1996 and 1998 and offset improvements occurring in sediment and phosphorus concentrations and loadings. Loadings = total quantity entering tidal waters annually at the fall-line; flow-weighted concentration = concentration not adjusted for flow effects (similar to observed concentrations in tidal waters); flow-adjusted concentration = concentration adjusted to an average flow year in order to remove effects of variable flow. Trend significance: **, p<0.01; *, p<0.05; no asterisk, p$\geq$0.05. Analysis done by the U. S. Geological Survey.

Pollutant Loads Delivered to the Estuary from Sources below the Fall-line

Long-term Perspective

The majority of point sources discharging directly to Potomac tidal waters (e.g. wastewater treatment plants, industrial dischargers) are located in the Washington metropolitan area. The area population grew 13-fold during the 20th century and is now about 5 million. Organic carbon loads from wastewater more than tripled between 1913 and 1944, decreased 91% with better treatment over the next 40 years, and are now at pre-1913 levels. Phosphorus wastewater loads rose over 200-fold, peaked in the 1970s and decreased sharply (98%) before the mid-1980s in response to advanced treatment and the phosphate detergent ban. Loads are now roughly equal to those of the 1920s. Nitrification processes converted ammonia to nitrate but did not removed it. Hence, the estuary experienced an 8-fold rise in nitrogen wastewater loads during the 20th century (Figure 2).

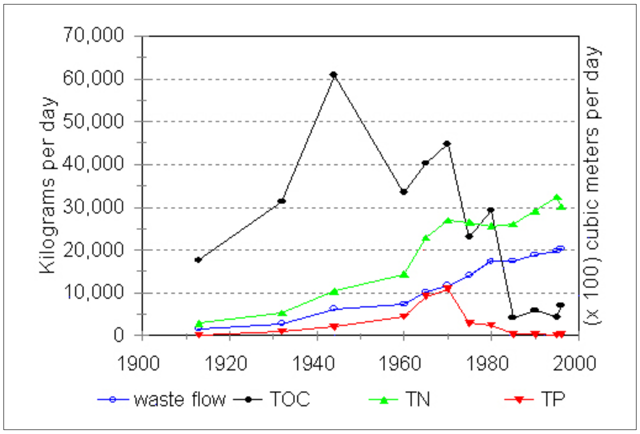

Figure 2. Nutrient loads from wastewater discharged into the upper Potomac estuary below the fall-line. The first Washington, D.C. sewers were built in the 1870s to better channel wastewater into the river. Sewage was untreated until completion of the Blue Plains Wastewater Treatment Plant (WWTP) in the 1930s. Advanced treatment and the construction of additional WWTPs removed much of the organic carbon and phosphorus in wastewater discharged to the river, despite a rapidly growing metropolitan population and a corresponding increase in wastewater flow. Meanwhile, nitrification processes converted ammonium to nitrate but did not removed it. Wastewater treatment upgrades at the Indian Head Naval Warfare Center in 1996 and implementation of a denitrification process at Blue Plains to remove nitrogen in late 1996 reversed the upward nitrogen loadings trend.

CBP Years

Water discharged from wastewater treatment plants (WWTPs) now accounts for about 11% (range: 2% - 39%) of the river flow below Washington, D.C. Nitrogen loads from WWTPs are roughly equal to 50% of fall-line loads due to the higher nitrogen concentrations in the wastewater. The century-long rise in nitrogen wastewater loads was reversed in the mid-1990s by upgrades to industrial dischargers and implementation of denitrification at Blue Plains WWTP. Blue Plains presently contributes 2/3 of the Washington area wastewater flow. Phosphorus and organic carbon loads from WWTPs are presently about 5% and 8% of fall-line loads, respectively, and show no overall trends because their biggest reductions were accomplished before the CBP years. Loads from combined stormwater overflows are probably significant but have not been monitored. Estimated loads of nutrient and sediment entering the estuary below the fall-line in runoff, groundwater, and atmosphere were not assessed.

Ambient Water Quality Status and Trends

The status of most pollution parameters is "fair" or "poor" in Potomac tidewaters relative to other Chesapeake tributaries, although bottom dissolved oxygen is generally "good" in the upper and middle estuary. The 1990s wet years caused an overall increasing trend in estuary flushing rates in the CBP Years (1985 - 1998), and relatively large amounts of nutrients and sediments were transported to Chesapeake Bay. The extent of the nutrient enriched tidal fresh zone was frequently expanded, displacing the oligohaline and mesohaline zones downriver and making the estuary less saline overall. Many of the observed Potomac Estuary water quality trends during the CBP Years can be explained by flow effects of the 1990s wet years.

Phosphorus

The Potomac Estuary water quality response to the drop in phosphorus wastewater loadings has been dramatic. Total phosphorus at monitoring stations in the upper and middle estuary decreased from average monthly concentrations that were at times greater than 0.6 mg/liter in the late 1960s and early 1970s, to concentrations that rarely exceeded 0.15 mg/liter after 1985. Long-term (1965 - 1996) trends in total phosphorus concentrations showed declines of 69% to more than 95%. Trends in total phosphorus and phosphate during the CBP Years were non-significant, presumably because most reduction efforts had been accomplished. Also, management actions that would have indirectly reduced non-point source phosphorus loadings in the CBP Years (e.g. storm ponds, best management practices) were offset by the large, sediment-bound phosphorus loads washed into the estuary during high flow events in the 1990s. Average ambient concentrations of total phosphorus are fairly uniform from the fall-line to the middle of the lower estuary. They are still too high to limit algal growth, except occasionally in the upper estuary during summer and in the middle estuary during winter. Phosphorus concentrations drop off abruptly in the lower estuary, reflecting dilution of Potomac waters by influxes of bottom waters from the Bay mainstem which have lower phosphorus concentrations. These bottom water influxes are greater when freshwater flows are high, and decreasing trends were in fact observed in the lower Potomac during the 1990s wet years. Phosphorus concentrations in the lower estuary are weakly limiting to algal growth during spring, but are rarely limiting during the summer maximum growth period.

Nitrogen

Average concentrations of organic nitrogen, ammonium, and nitrate--the predominant nitrogen forms--quickly increase as tidal fresh waters pass through Washington, D.C., and they remain high between Rosier Bluff in southeast Washington, D.C. and Indian Head, Maryland before finally tapering off in the middle estuary (Figure 3). Nitrogen concentrations are presently never low enough to limit algal growth in the upper and middle estuary, and excess nitrogen is transported downriver. Nitrogen is more completely utilized by the mesohaline phytoplankton populations of the lower estuary, and concentrations eventually drop to levels that control, or limit, phytoplankton growth during summer and fall (<0.07 mg/liter). Long-term increases in nitrogen loads to the Potomac Estuary are thought to be fueling the large summer peak in algal production in the lower estuary and, in the absence of significant grazing pressure by large-bodied consumers (see below), the subsequent algal decomposition is expanding and strengthening bottom water hypoxia and anoxia. Long-term trends (1965 - 1996) show total nitrogen concentrations have increased as much as 20%-40% in the upper Potomac Estuary and 73% in the lower estuary, reflecting steadily rising loads to the upper estuary and increasing transport of nitrogen to the lower estuary. Trends during the shorter-term CBP Years (1985 - 1998) countered the long-term trends: total nitrogen (TN) and dissolved inorganic nitrogen (DIN) trends showed declines as much as 33% and 53%, respectively, at stations in the upper and middle Potomac estuary, and no change at stations in the lower estuary. There are two probable causes of these opposing trends. First, nitrogen concentrations in fall-line waters are significantly lower than those in tidewaters, and the greater quantities of fall-line waters that entered the estuary during the 1990s diluted tidewater concentrations. Second, the tidewater TN and DIN declines may reflect the decreases in Washington, D.C. area point source nitrogen loadings that occurred after 1995. These point source declines should hold tidewater concentrations to lower levels when the dilution effect of fall-line freshwater flows diminishes in average or low flow years.

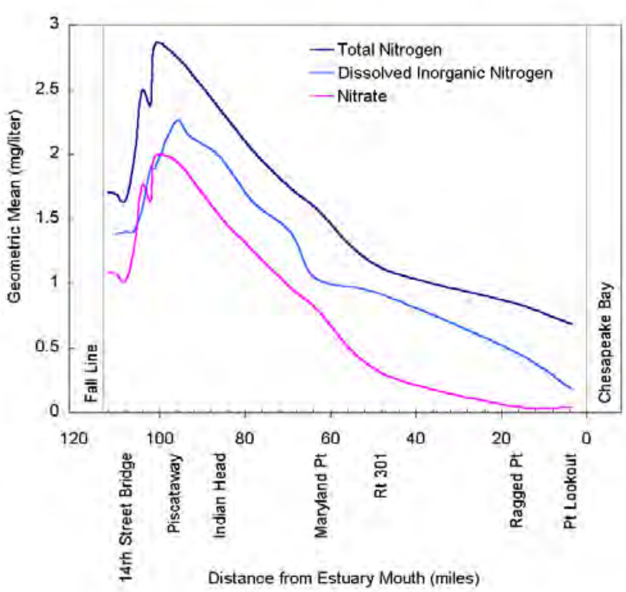

Figure 3. Long-term average concentrations of total nitrogen, dissolved inorganic nitrogen, and nitrate for 1990 - 1996 in the upper and middle Potomac Estuary. River landmarks: D.C. = District of Columbia; Pisc = Piscataway Creek; IH = Indian Head; MD = Maryland Point; Mor = Morgantown; Rag = Ragged Point; PtL = Point Lookout.

Total Organic Carbon

Long-term (1965 - 1996) trends in ambient concentrations of total organic carbon (TOC) fell between 61% (upper estuary) and 12% (lower estuary) to ~6 mg/liter as a result of wastewater treatments between 1940 and 1980 (Figure 2). Ambient TOC concentrations fluctuated sharply between 1976 and 1983, occasionally reaching levels as high as ~37 mg/liter. Massive summer die-offs of the Asiatic clam (*Corbicula fluminea*) are the likely cause of these fluctuations. The upper estuary population of this introduced species underwent extreme cycles in abundance in the mid-1970s and the early 1980s before stabilizing. Total organic carbon concentrations were stable at ~4 mg/liter during the CBP Years and fairly uniform distributed throughout the estuary.

Dissolved Oxygen

Long-term 1965 - 1996 trends for bottom dissolved oxygen show improvements as great as 25% in the upper estuary, despite a return to higher chlorophyll levels at the end of this time period (see below). Dissolved oxygen levels between Washington D.C. and Maryland Point rarely fell below the 5 mg/liter minimum requirement for a healthy habitat (1993 Chesapeake Bay Dissolved Oxygen Goals for Restoration of Living Resource Habitat) after 1980. This improvement can be credited in part to the reductions in phosphorus and organic carbon loads which lowered chlorophyll levels and overall biological oxygen demand (BOD) in the upper and middle estuary. Nutrient load reductions have not restored acceptable dissolved oxygen levels in the lower estuary. In this much larger, deeper segment of the Potomac Estuary, salinity and temperature stratification normally isolate bottom waters in summer and facilitate development of an anoxic/hypoxic zone. Bottom dissolved oxygen worsened between 1965 and 1985, but showed no significant trend during CBP Years.

Water Clarity and Suspended Solids

Water clarity is generally "fair" to "poor" throughout most of the Potomac estuary relative to other Chesapeake tributaries. Potomac tidewaters are naturally clouded by strong tidal mixing in the middle estuary, where salinities transition from fresh to brackish, and a turbidity maximum is expected there. However, water clarity is not much better above or below this maximum due to high concentrations of total suspended solids (sediments, detritus, and phytoplankton) in the water (Figure 4). The annual average secchi depth drops quickly from greater than 1 meter (0.1 - 3.6 meters range) to about 0.7 meter (0.2 - 1.8 meters range) as tidal fresh waters pass the short distance through Washington, D.C. Secchi depth averages (1984-1998) then vary between 0.4 and 0.7 meters the entire length of the upper and middle estuary segments before finally increasing to 1 - 2 meters in the lower estuary.

Water clarity in the upper estuary is presently so poor that it limits phytoplankton growth almost all the time (phosphorus was briefly limiting during the summers of 1991, 1992 and 1998). It also frequently fails CBP tidal fresh habitat criteria for submersed aquatic vegetation (>0.7 meter) between March and September. Water clarity attains the CBP oligohaline habitat criteria for submersed aquatic vegetation (>0.7 meter) about half the time in the middle estuary. However, it almost always limits algal growth. Water clarity in the lower estuary attains the mesohaline submersed aquatic vegetation criteria (>1.0 meter) most of the time and light levels rarely limit algal growth. The potential consequences of high suspended solids concentrations to animals, whether organic or inorganic, have not been assessed. They include deleterious effects on filter-feeding animals, a smothering of benthic organisms either directly with sediments or indirectly with increases in phytoplankton, and disruptions of light-dependent animal behaviors.

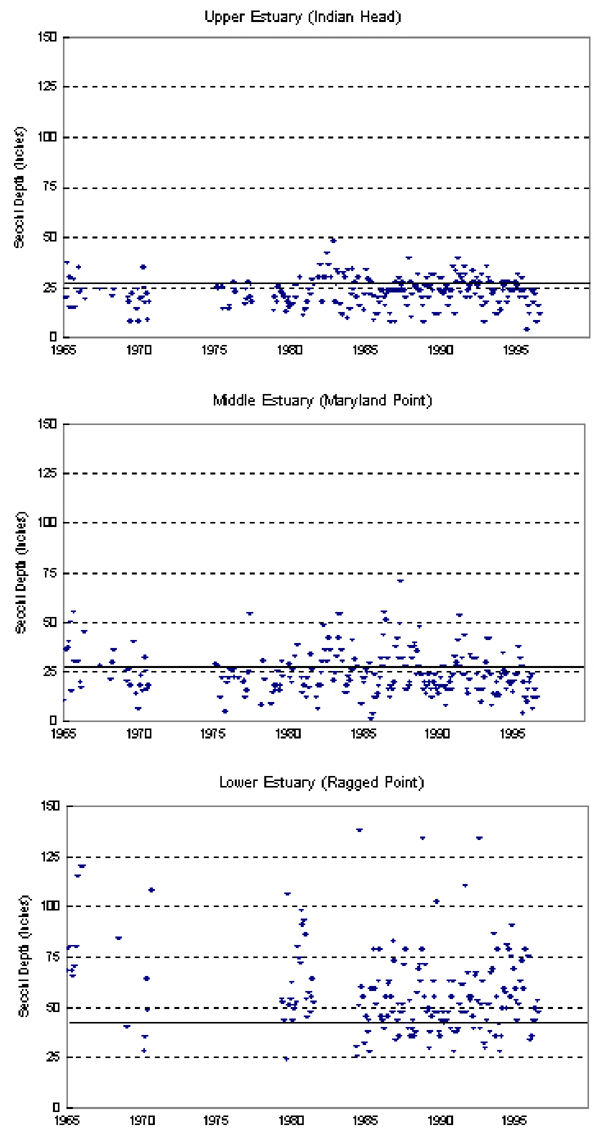

Figure 4. Secchi depth time series for the Indian Head station in the upper estuary, Maryland Point station in the middle estuary and Ragged Point station in the lower estuary, for 1965 - 1996. Solid lines are the submersed aquatic plant habitat criteria for secchi depth: 0.7m for the upper and middle estuary salinities, 1.0 m for the lower estuary salinity. Segment trends (all station data pooled by segment) during the CBP Years (1985 - 1998) were -23.9% (p<0.01) in the upper estuary, -20.9% (p=0.013) in the middle estuary and -25.3% (p<0.01) in the lower estuary.

Average sediment concentrations, and presumably the clarity, of fresh waters entering the estuary at the fall-line improved during the CBP Years (Figure 1) yet ambient concentrations of total suspended solids (TSS) in the upper estuary did not decline and water clarity did not increase. The lack of improving ambient trends was likely due to the observed increase in the phytoplankton component of TSS, and in particular the increase in bluegreen algae. Bluegreen algae are a taxonomic group adept at growing in turbid environments, and they were probably able to respond quickest to the declining sediment concentrations and improving water clarity of the incoming waters. This hypothesis cannot be explored because data on the organic and inorganic fractions of TSS in tidewaters are not available. Long-term (1965 - 1996) station trends in water clarity declined 10% in the middle estuary and up to 30% in the lower estuary. Declines were steeper during the CBP Years (Figure 4), apparently because of increasing trends in TSS and/or bluegreen algae biomass in those reaches of the estuary.

Biological Responses

Biological responses to changing nutrient loads and ambient concentrations are first detected in the microbial realm of bacteria and algae, and then in the submersed aquatic vegetation, benthic macroinvertebrates and zooplankton. All these groups have been studied in the Potomac Estuary during the 20th century and the latter four have been monitored intermittently since the 1960s.

Phytoplankton

Phytoplankton, the base of the aquatic food web, showed enormous community responses to the 20th century eutrophication of the Potomac Estuary. Extensive blooms of freshwater bluegreen algae species proliferated during the summer months of the 1960s, 1970s and early 1980s in a 60 kilometer expanse of the upper and middle Potomac Estuary, between Mt. Vernon and Maryland Pt. The blooms were often dominated by the colonial algae, *Microcystis aeruginosa,* a species of no nutritional value to most large-bodied grazers. Chlorophyll *a*, a quantitative indicator of phytoplankton biomass, frequently exceeded 100 micrograms per liter and phytoplankton were a major component of the suspended solids degrading water clarity. These undesirable characteristics of the phytoplankton community were responses to the high nutrient loads as well as favorable meteorological and flow conditions. Freshwater algae biomass generated in the upper and middle estuary was transported downriver, unconsumed, to die and decompose in the steep salinity gradient usually located between Maryland Pt. and the Route 301 Bridge. In the lower estuary, dinoflagellate algae species dominated in summer, and frequently formed "mahogany tides" or blooms.

Average surface chlorophyll *a* concentrations declined as much as 50% at middle estuary monitoring stations after loadings of the nutrient, phosphorus, were sharply reduced in the mid-1970s. Summer blooms continued to occur in the upper estuary, probably because phosphorus that had accumulated in bottom sediments was still being released to the water. The intensity and downriver extent of upper estuary summer blooms decreased after 1985 and chlorophyll *a* now rarely exceeds 100 micrograms per liter (Figure 5), suggesting that sediment phosphorus is becoming depleted. The upper and middle estuary zones are still extremely enriched, however. Bluegreen algae still dominate the summer populations, and phytoplankton are primarily limited by light and residence time rather than nutrients. Bluegreen algal densities and chlorophyll *a* rose during the 1990s

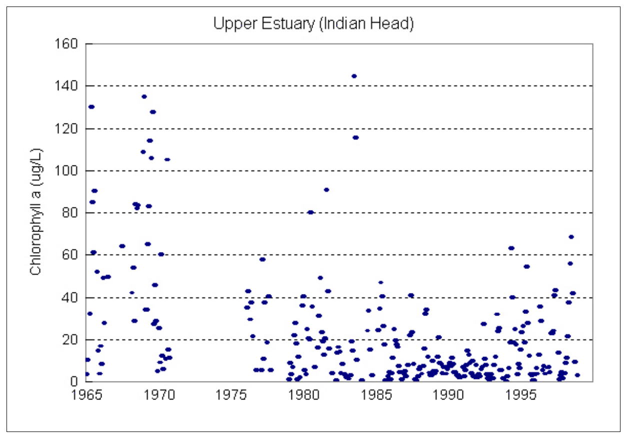

Figure 5. Surface chlorophyll *a* concentrations at the Indian Head station (TF2.3) in the upper Potomac Estuary, 1965 - 1998. Data from multiple sources.

wet years, possibly reflecting the effect of an enlarged freshwater zone and favorable summer weather conditions.

In the lower estuary, chlorophyll concentrations dropped after the 1970s phosphorus reductions but then showed no long-term (mid-1970s - 1996) trends. Chlorophyll concentrations increased slightly during 1990s wet years, and small, filamentous, salt-tolerant bluegreen algae have recently become dominant over estuarine diatoms and dinoflagellates in summer, climbing as high as 68% of the total phytoplankton biomass during August 1998. The shift in species dominance to bluegreen algae in mesohaline waters is unusual and could be related to a number of factors, including degrading water clarity trends, transport of higher nitrogen loads to the lower estuary in the 1990s wet years, or shifting ratios of dissolved inorganic nitrogen and phosphate. Nitrogen is more completely utilized by the mesohaline phytoplankton in the better light environment of the lower estuary, hence summer and fall nitrogen concentrations can become limiting but only after a large algal biomass has built up. Phosphorus concentrations weakly limit algal growth during spring and silicon can limit spring diatom growth in low flow years. Nitrogen reductions in the upper estuary are expected to improve poor dissolved oxygen levels in bottom waters of the lower estuary by directly limiting the spring diatom bloom and indirectly limiting the subsequent algal production and decomposition in summer.

Submersed aquatic vegetation

Flourishing beds of underwater grass, or submersed aquatic vegetation (SAV), once lined the entire Potomac Estuary shoreline and supported abundant fish, invertebrate and waterbird populations. In the 1930s, disease decimated the eelgrass (*Zostera marina*) beds in the lower Potomac Estuary while turbidity (dredging, severe storms) and the exotic SAV species water chestnut (*Trapa natans*) greatly diminished native SAV beds in the upper estuary. During the 1960s, worsening turbidity (algal blooms), another exotic SAV species Eurasian watermilfoil (*Myriophyllum spicatus*), and the cutting and chemical treatment of watermilfoil eventually eliminated all SAV except for narrow bands along the middle estuary shoreline and isolated patches elsewhere. Native SAV returned to the upper estuary only after several years of improved water clarity combined with below-average wind speed and above-average sunlight in 1983. This sudden return was accompanied by the eruption of a third exotic species, *Hydrilla verticillata*. Total SAV coverage quadrupled and distribution spread downriver after 1983 (Figure 6), however the SAV population is not stable. The region of greatest SAV coverage shifted rapidly downriver during the 1990s wet period, and coverage has varied substantially in the past 13 years (Figure 6). Total coverage of the SAV beds in the estuary today is only about 9% of their probable coverage in the early 20th century, and the full habitat and food value of the native species assemblage is still missing. Potomac SAV beds reached 57 % of the CBP Tier 1 Goal and 16% of the CBP Tier 2 Goal in 1998.

In the Potomac Estuary, attainment or non-attainment of the SAV habitat criteria established by the CBP has *generally* corresponded to year-to-year expansion or

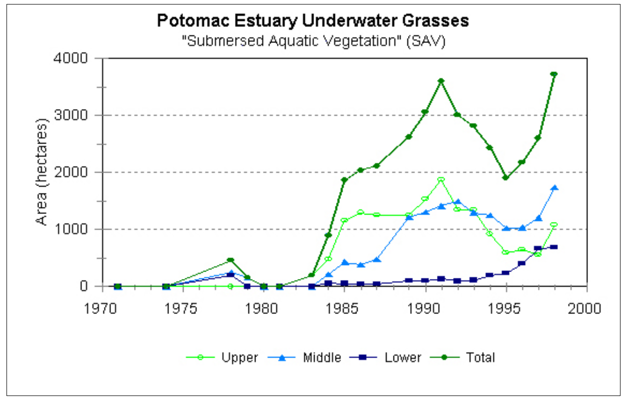

Figure 6. Coverage of submersed aquatic vegetation (SAV) in the upper, middle and lower Potomac Estuary (mainstem only). The upper zone is tidal fresh; the middle estuary is generally oligohaline; and the lower estuary is generally mesohaline. Data were obtained from USGS surveys and from the CBP SAV monitoring program, an aerial survey conducted in summer by the Virginia Institute of Marine Sciences.

contraction of SAV coverage, but does not explain all of the changes that have occurred in SAV coverage. Seasonal weather and flow patterns also are important.

Zooplankton

The microbial loop community which includes *bacterio*plankton and *micro*zooplankton (protozoans and rotifers) is thriving in the Potomac Estuary. Concentrations of dissolved organic material largely generated by phytoplankton are presently able to support unprecedented high numbers of bacterioplankton (30-50 million cells per milliliter). High levels of phytoplankton and bacteria in turn support abundant populations of ciliate protozoans and rotifers. Potomac rotifer numbers were among the highest in the Chesapeake system during the CBP Years. Analysis suggests these high abundances may reflect both the very abundant food sources and insufficient predation pressure by higher level consumers such as the copepod *Acartia tonsa*. This clearly demonstrates the continued eutrophic state of the Potomac Estuary.

*Meso*zooplankton abundances in the Potomac Estuary were variable, but usually low, before and during the CBP Years. Mesozooplankton (copepods, cladocera, invertebrate larvae) are an important link in food web pathways leading to fish. Upper and middle estuary abundances were often insufficient for larval anadromous fish in spring, and summer abundances were moderate to low when compared to other

Chesapeake tidal fresh and oligohaline areas. The work of several investigators suggests production rates of important mesozooplankton species are below their full potential in these areas. For example, the estimated productivity of *Eurytemora affinis*, a dominant copepod, during a 1977 study was lower than measured productivity under favorable laboratory conditions. Mesozooplankton are presently not limited by food quantity, however food quality in the upper and middle estuary - and particularly the high proportion of bluegreen algae in the summer phytoplankton - may be one cause of their lower rate of production. Predation pressure from summer planktivorous fish also appears to control mesozooplankton populations in the upper estuary, i.e. mesozooplankton were abundant when planktivores were sparse, and vice versa (Figure 7). In the lower estuary, seasonal abundances were low compared to abundances in other Chesapeake mesohaline areas, and may be controlled by several factors including high predation pressure by jellyfish and fish and poor underwater visibility. The number of mesozooplankton species (species richness) has remained fairly constant in tidal fresh and mesohaline waters since the 1970s indicating no improvement.

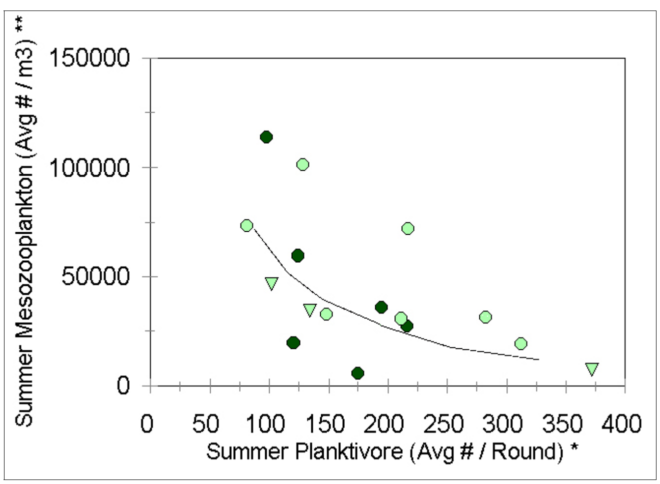

Figure 7. Regression between summer abundances of mesozooplankton and obligate finfish planktivores (i.e. fish that eat zooplankton their entire lives) at the Indian Head, tidal fresh monitoring station in the Potomac Estuary mainstem ($p<0.01$, $r^2=0.71$). Similar relationships were also found upriver in Gunston Cove and the Potomac mainstem adjacent to Gunston Cove. Maryland Summer Seine Survey fish data, historical zooplankton data (1973, 1981), and Chesapeake Bay Program Zooplankton Monitoring Program data (1984 - 1998). Light triangle: SAV absent or in low abundance; light circle: SAV moderately abundant; dark circle: SAV abundant. The presence of SAV in shallow waters may be associated with shifts in the predator-prey data towards one end of the regression curve or the other. SAV is know to significantly affect mesozooplankton and planktivorous minnow abundances in those waters.

However, it has increased 3- to 4-fold in the oligohaline reach of the Potomac, suggesting this middle reach of the estuary is responding more quickly than the others to management actions.

The comb jellyfish (ctenophores) and true jellyfish (cnidarians) are predators on mesozooplankton and fish larvae in mesohaline waters. Ctenophore densities increased in the lower estuary during the CBP Years and their predation rates on mesozooplankton are potentially large. Ctenophores may be increasing because a) lower salinities in the past decade separate them more from their cnidarian predators, and/or b) increasing turbidity (i.e. higher TSS, lower secchi depth) give them a competitive edge against visual predators such as finfish.

Benthos

Benthic invertebrates dwelling in "soft-bottom" areas, or sediments (muds, clays, sands), are another important link in the food web pathways leading to fish. A diverse soft-bottom benthic community inhabited the Potomac Estuary in the first half of the 20th century. *Rangia cuneata* (brackish-water clam) invaded the middle and lower Potomac estuary mid-century, and *Corbicula fluminea* (Asiatic clam) invaded the upper Potomac Estuary in the late 1970s. Both species climbed to very high densities and briefly dominated their benthic assemblages before declining. The benthos of the upper Potomac estuary were stressed by poor dissolved oxygen during the 1960s and 1970s but recovered as oxygen improved, and the present community has some of the highest measures of species richness, total abundance and biomass in the Chesapeake system. Closer examination suggests the upper estuary population is not completely restored since it is dominated by tubifex worms and other pollution-tolerant species. In the middle estuary, an area naturally stressed by salinity fluctuations, species richness is low but has increased during the CBP Years. The Benthic Index of Biological Integrity (BIBI), a multivariate scoring protocol for benthic macroinvertebrate communities, indicates this area occasionally meets the CBP restoration goals. In contrast to the upper and middle estuary, the benthos community in the lower Potomac estuary is the worst in the tidal Chesapeake system, along with the deep trench in the Bay mainstem, and consistently fails the BIBI. This dichotomy is primarily due to the different summer oxygen levels.

The food value of the soft-bottom benthos to higher trophic levels was not assessed, and it is unclear if the community provides sufficient food for benthivore fish predators and diving ducks. Historic benthos levels have not been recovered because submersed aquatic vegetation beds, a habitat inwhich benthos are known to concentrate, are still at only 9% of their historical levels.

Fish and Shellfish

Improvements to-date in habitat and food conditions have not directly stimulated long-lasting increases or complete recoveries in Potomac Estuary commercial fisheries, including oysters. Populations of other large-bodied, long-lived consumers are similarly unchanged, and the Potomac Estuary food web is still dominated by microbial pathways. Aggressive state and federal management has helped striped bass recover from overharvesting in the 1980s. The unstable recovery of submersed aquatic vegetation in the upper and middle estuary has encouraged some fish groups (e.g. minnows, large mouth bass) to increase.

Conclusions and Recommendations

Will the current nutrient reduction policies successfully return the Potomac water quality to desirable and habitable levels? Management actions to control organic carbon and phosphorus loadings, especially those from point sources, significantly reduced ambient concentrations of these parameters in the Potomac Estuary before the start of the Chesapeake Bay Program. These pre-1985 reductions were largely responsible for improving dissolved oxygen to habitable levels (>5 milligrams per liter) in the upper and middle estuary. Actions designed to further reduce phosphorus loadings during the CBP Years have not significantly changed ambient concentrations in the water column. Phosphorous concentrations in the upper and middle estuary still have not reached the desired, low levels that limit phytoplankton growth. However, declines in the extent and intensity of summer bluegreen algae blooms under favorable weather conditions suggest that the release of phosphorus stored in the sediments may be diminishing as the accumulation rate slows.

Recent management actions to reduce point and non-point sources of nitrogen are apparently reversing the century-long climb in nitrogen loads. The resulting decreases in ambient nitrogen concentrations are expected to strengthen nutrient-limitation as the primary factor controlling summer phytoplankton productivity in the lower estuary, and will reduce the amount of nitrogen exported to Chesapeake Bay. The reductions are also expected to lower chlorophyll levels and improve dissolved oxygen in the lower estuary, thereby alleviating the anoxic/hypoxic stress on mesohaline biological communities to some degree. Dissolved inorganic nitrogen concentrations in the upper and middle estuary are presently always above the 0.07 milligram N per liter limitation threshold for phytoplankton growth.

Existing nutrient load reduction strategies could possibly attain the desired, algal-limiting nutrient concentrations if management action is also taken to reduce turbidity. Poor water clarity presently limits algal growth and stresses submersed aquatic vegetation in the upper and middle estuary. As long as light environments remain poor in these areas, further nutrient load reductions will only minimally improve habitat conditions. Degrading trends in the lower estuary suggest the same association of poor water clarity, high nutrient levels and summer blue-green algal blooms is developing in this segment as well. Nutrient reduction strategies may now need to be augmented by strategies to increase light penetration in order to restore healthy phytoplankton and submersed aquatic vegetation communities. Under higher light levels, dissolved inorganic nutrients are more rapidly utilized and converted to particulate organic matter, namely phytoplankton biomass, resulting in faster nutrient turnover and possibly lower concentrations of dissolved nutrients. While higher phytoplankton biomass is not wanted, better light environments and lower nutrient concentrations should allow more desirable algal species to grow. This improved food base should encourage consumption (grazing) by zooplankton and benthos. Similarly, higher light levels should also support more stable, robust growth of submersed aquatic vegetation beds and result in higher abundances of benthic and zooplankton grazers associated with the beds.

Will current nutrient reductions sufficiently improve food and habitat conditions so that the impacts of other stressors are overcome and a balanced, productive ecosystem with abundant fish and wildlife populations is restored? Nutrient reductions to-date have not been sufficient to overcome the other, often mutually-reenforcing anthropogenic impacts to the estuary. In view of the estuary's present status, fully implemented nutrient reduction strategies should also not be expected to do so. This conclusion was expected, given the number and magnitude of anthropogenic impacts that have changed the estuary in addition to nutrient enrichment. Landscape changes after European settlement increased the variability of freshwater baseflows into the estuary by ~40%. Heavier sedimentation filled in many tributaries. Enormous (>98%) losses in many large-bodied, long-lived consumers of the food web, including American shad, river herring and sturgeon, profoundly changed food web dynamics in the 20th century. Major declines in two key "living" habitats (oysters and submersed aquatic vegetation) and blockages to migratory fish have substantially limited the physical habitats available. Exotic species were introduced and are now an integral part of the ecosystem. Bioaccumulation of toxic chemicals has become a persistent problem in the upper and middle estuary and in the Anacostia tributary.

Thus, in order to successfully restore a balanced, productive system with abundant living resources, management strategies for the Potomac Estuary need to become both more holistic and more region specific. For example, halting the degrading trends in the lower estuary may require reducing upriver nitrogen loadings *and* restoring oyster populations (for the purpose of filtering sediments and algae from the water). Actual restoration of the lower estuary may require additional management actions. Furthering the gains made to-date in the upper and middle estuary may require reducing *in*organic sediment loadings above and below the fall-line for the purpose of restoring the ecologically important submersed aquatic vegetation beds and reducing bluegreen algae dominance in the phytoplankton community.

Additional conclusions in this report that confirm or support the results of other Chesapeake Bay Program analyses include:

- Variability is an expected and natural feature of estuaries, and annual variability in freshwater flows can temporarily negate and even offset nutrient reduction efforts.

- Downward nitrogen trends observed in some western Chesapeake tributaries during the 1985 - 1998 period ("CBP Years") may be partly the result of greater dilution by relatively cleaner upstream waters during 1990s high flow years.

- Reductions in point and non-point source loadings of phosphorus may take years and even decades to be fully expressed as reductions in ambient phosphorus and chlorophyll concentrations in the upper and middle estuary.

- Biological populations in the middle of the open water food web currently do not appear to be closely linked to phytoplankton, the base of that food web.

- Populations higher in the food web take more time to recover because of their longer life cycles, and their restoration may require more effort than simply improving habitat conditions (e.g. fish stocking, managed harvests, physical habitat restoration).

- Living resources, especially submersed aquatic vegetation and oysters, may be crucial partners in successfully attaining good water quality and food conditions.

Several issues were raised by this project that can best be answered with further research or data analysis efforts, in order to direct management actions in more effective ways:

- Further investigate the downward trends in ambient total nitrogen concentration and shifts in the proportion of nitrate, to determine how much can be attributed to Biological Nitrogen Removal (BNR) at Blue Plains Treatment Plant.

- Explore the causes of the decreasing water clarity trends in all segments of the estuary.

- Explore the reasons why bluegreen algae have recently increased throughout the estuary, and examine the role of phytoplankton in light attenuation in the upper estuary.

- Investigate mesozooplankton rates of production to determine a) if growth rates are maximal and abundances are low because of predation pressure, flow or some other natural loss function, or b) if growth rates are in fact below the population's potential due to stressful habitat conditions and/or poor food quality. Determine what factors limit microzooplankton production rates and what factors control their total abundance.

- Determine the oyster population abundances required to reduce ambient sediment and nitrogen concentrations in the lower Potomac estuary by up to 40% of 1985 levels, and explore the potential role of restored oyster beds in destabilizing stratification and inhibiting summer anoxia development in the lower estuary.

Management strategies have emphasized nutrient reduction strategies while recognizing the importance of living resources conservation and restoration in achieving good water quality (e.g. CBP Submersed Aquatic Vegetation Restoration Goal, CBP Ecologically Valuable Species Strategy, CBP Oyster Reef Restoration Goal). To further restore the Potomac Estuary ecosystem to a productive, well-functioning state, jurisdictions need to continue to reduce nitrogen and phosphorus loadings and also:

- Improve water clarity.
- Restore two important "living" habitats, submersed aquatic vegetation beds in the upper estuary and oyster reefs in the lower estuary, and key migratory fish passages.
- Restore and/or protect key finfish top predators and mid-level prey species.

TIDAL POTOMAC INTEGRATIVE ANALYSIS PROJECT

A Synthesis of the Project Results

Claire Buchanan, William Romano and Richard Lacouture

INTRODUCTION

The Potomac Estuary stretches 113 miles from head of tide to mouth and holds more than 7.73 billion cubic meters of water, making it the fourth largest estuary on the United States East Coast and the 25th largest in the nation. The Nation's Capitol sits at the head of tide, giving the estuary added visibility and national importance. The estuary is riverine in its upper-most, tidal fresh segment. Freshwater surface inflows average 12,180 cubic feet per second and typically flush the segment in a matter of days. The fresh waters slow and begin to mix with incoming salt water as the estuary widens below Indian Head, forming a transition zone 30 - 65 miles below the head of tide. The estuary makes sharp bends at Maryland Point and Mathias Point, then enters the largest (6.24 billion cubic meters), widest segment approximately 65 miles below the head of tide. This lower estuary segment takes 2-3 months to flush, its salinities are usually mesohaline (5 - 18 parts per thousand) and its water column can stratify. The estuary empties into the Chesapeake Bay, contributing about 19% of the surface flow into that system. Historical accounts indicate the Potomac Estuary was once a diverse, productive ecosystem and yielded some of the largest United States East Coast fish harvests for many years in the 19th century (Tilp 1978). Evidently, the variable morphology (Figure 1) and hydrology (Figure 2) of the Potomac River estuary was capable of providing abundant food and favorable habitat to many fish species. Further descriptions of the Potomac Estuary geography, morphology and general physical and chemical characteristics are available in Lippson et al (1979), in the appendices of this report, and elsewhere.

40% Nutrient Reductions

The Potomac River was grossly polluted and a national disgrace by the 1960s. Over-harvesting, habitat losses, exotic species, and nutrient enrichment were major anthropogenic stresses on the ecosystem during the previous one and a half centuries (see below). The impacts of these stressors were modified and exacerbated by natural stresses, including floods, droughts, and disease. Restoring Potomac waters and recovering the estuary's potential to provide abundant food and habitat for fish has became a long-term commitment of the region's jurisdictions.

Studies conducted in the late 1960s and 1970s focused on the impact of excess nutrients, especially phosphorus, nitrogen, and organic matter which caused excessive algal growth, low dissolved oxygen levels and high turbidity in the upper estuary, or tidal river (Jaworski, 1990). Regional authorities implemented improved wastewater treatment, a phosphate ban, and non-point source nutrient management strategies. Nutrient reductions appear to be working to some degree and deteriorating trends in phosphorus and nitrogen concentrations in the waters flowing into the estuary are reversing. Other strategies endorsed by the Chesapeake Bay Program (CBP) Executive Council have recognized the importance of toxic chemical reductions, sediment controls, and living resources conservation and restoration in restoring water quality, however the Program has kept its emphasis on nutrient reduction strategies. The question now facing Potomac resource managers is: *Will the 40% nutrient reduction policy of the Chesapeake Bay Program successfully return Potomac water quality to a desirable status and benefit living resources?* It is possible that the 40% phosphorus and nitrogen reductions called for by the CBP to restore acceptable habitat space will be sufficient to encourage living resources to return to the system. It is also possible that this "build it and they will come" approach will *not* be sufficient, and additional efforts to restore habitats and living resources will be needed.

Purpose of Report

The purpose of this synthesis report is to assemble and assess the available monitoring data and a) quantify how anthropogenic nutrient reduction strategies have reduced nutrient loads and altered water and habitat quality to-date, and b) determine if key biological communities have responded positively to these changes. Following descriptions of historical changes in the Potomac estuary, the report is divided into three sections with different objectives. The first objective is to evaluate the linkages between nutrient loadings and ambient chemical and physical parameters. The second objective is to evaluate the linkages between ambient chemical / physical conditions and the major primary producers, submerged aquatic vegetation and phytoplankton. The third objective is to evaluate the linkages between ambient chemical / physical / biological conditions and the major lower trophic level consumers, zooplankton and soft-bottom benthos. Emphasis is placed on trying to account for natural variability in the ecosystem caused by flow, season and salinity in order to focus on changes that could be related to management actions. Flow corrected and uncorrected trends in ambient water quality conditions are analyzed at individual stations in the Potomac mainstem and in smaller embayments. Data are also grouped by ecologically significant segments. In order to better understand long-term trends, the hydrological and weather data were analyzed to tease out their relationships with and influence on, Potomac water quality and biological communities. Several time periods were analyzed and related to the ecologically important living resources groups including phytoplankton, submerged aquatic vegetation, zooplankton and benthos.

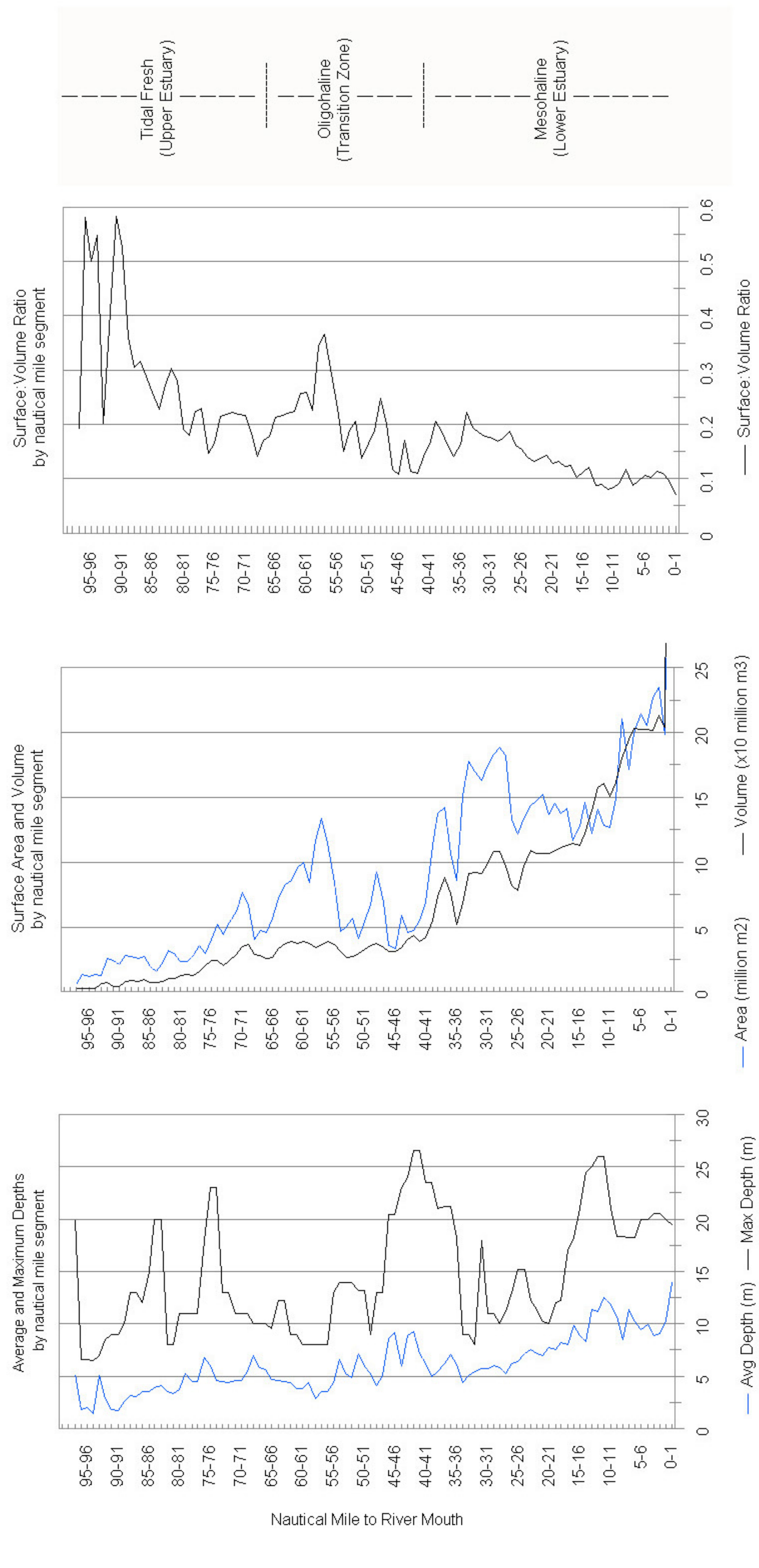

Figure 1. Average depth (meters), maximum depth (meters), surface area (million square meters), volume (10 million cubic meters) and ratio of surface area to volume for each nautical mile (= 1.852 kilometers or 1.151 statute miles) segment in the Potomac estuary. Adapted from Lippson et al, 1979. Data originally from Cronin and Pritchard, 1975.

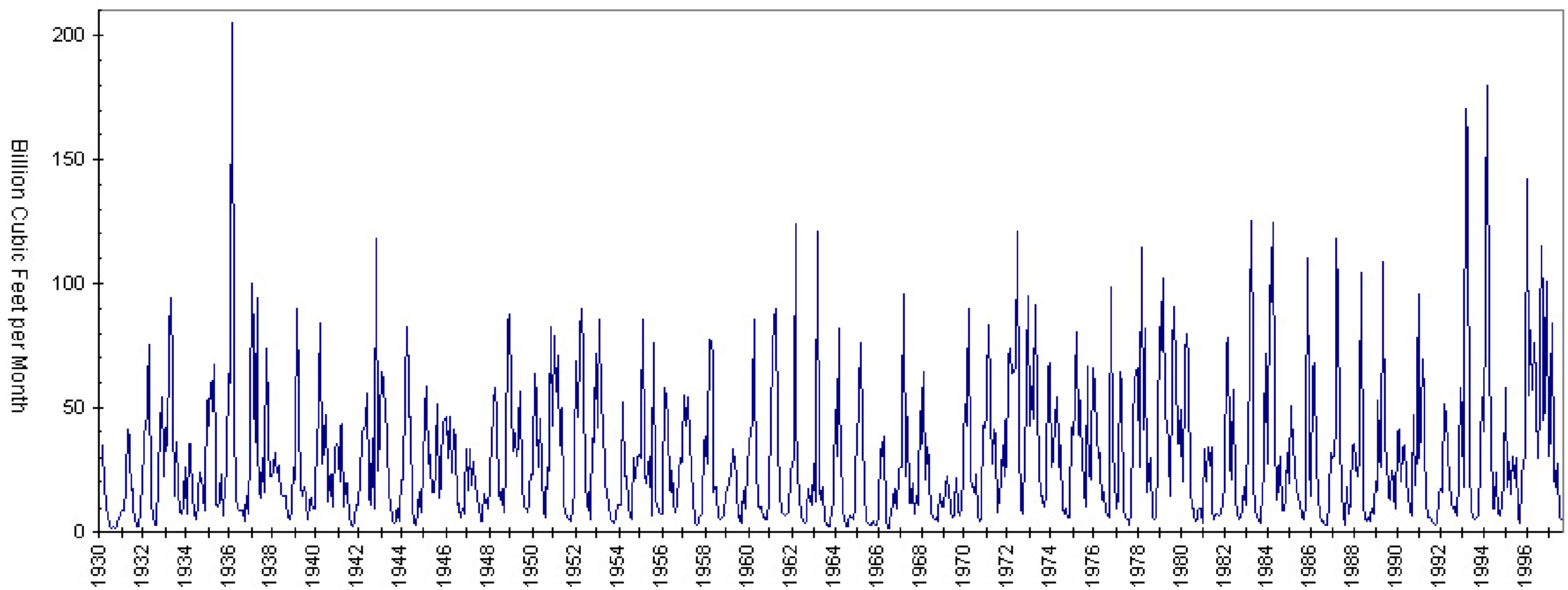

Figure 2. Total monthly freshwater flows at the Potomac River USGS gaging station 01646500 at the fall-line (Little Falls Dam, Montgomery County), for March 1930 - December 1997. The station is located at latitude 38°56'58" longitude 77°07'40",1 mile upstream from District of Columbia boundary line and at mile 117.4 from the mouth of the Potomac River estuary. Watershed drainage area is 11,560 square miles. Water supply diversions and releases from upstream reservoirs affect the natural flows. Extremes for period of record: maximum discharge, 484,000 ft^3/s, Mar. 19, 1936; minimum daily discharge observed at gaging station, 121 ft^3/s, Sept.9, 1966, does not include diversion of 489 ft^3/s for municipal use; minimum daily discharge (adjusted), 601 ft^3/s, Sept. 10, 1966, includes diversion of 449 ft^3/s for municipal use. Extremes outside period of record: flood of June 2, 1889, was of approximately the same magnitude as that of March 19, 1936 (from USGS).

ANTHROPOGENIC IMPACTS ON THE POTOMAC ESTUARY[1]

Measures (indicators) of the tidal Potomac water quality and biological communities suggest a poor level of ecosystem integrity now relative to the 1700s and early 1800s. Anecdotal reports of teaming fish and wildlife populations in that period indicate the Potomac Estuary was a highly productive ecosystem with many food web consumers. The most abundant fish near Washington, DC were several migratory species - shad, herring and sturgeon, and occasionally striped bass. George Washington's diary includes entries such as "caught 500 shad today" (~2,000 lbs) and "caught about 50,000 herring at a draught this afternoon" (~50,000 lbs), and he once complains about catching only 30,000 herring during a night. Newspaper accounts at the time refer to an occasion when 450 migrating striped bass averaging 60 pounds each were caught in a single seine "draft" at General Mason's Sycamore Landing near Washington, DC. Flourishing underwater grass beds lined the entire Potomac estuary shoreline and supported abundant fish, crab and waterbird consumers. Oyster reefs, so dense that they were frequent navigation hazards, crowded the shallower waters of the lower estuary. Nutrient inputs at this time would have been near the baseline levels associated with an entirely forested land cover, yet these low loadings were apparently enough to support the large and diverse estuarine biota. Reports of clear water with good visibility suggest primary production by phytoplankton was efficient and passed quickly to consumers. Epi-benthic algae would have flourished in the clear waters and they probably played an important role in the food web. The natural bounty and lack of restrictions fostered the attitude that supply was unlimited and lead to the intense use of Potomac living resources.

Finfish and Shellfish Declines

Harvest pressure on alosid species climbed to their highest levels in the 1830s, with recorded annual catches reaching ~75 million pounds for herring and ~90 million pounds for shad. The populations apparently were able to withstand the high harvest pressure at first, evidence of their original high numbers and resiliency. The fishery expanded to include other species and total Potomac commercial harvests became the largest of any East Coast river by the late 1800s (Tilp 1978). Fish populations, however, reached a turning point in the mid to late 1800s and annual harvests began to decline. This prompted public calls for restocking programs (e.g. The Evening Star of Saturday, June 17, 1876 quoted in The Evening Star, April 3, 1921) but not for a reduction in harvest pressure. Harvest pressure remained so great that at the turn of the century "it was scarcely an exaggeration to say that not a gallon of the water of the river flowed into Chesapeake Bay without being strained through the meshes of some net" (Tilp 1978).

Annual harvests of the heavily exploited herring and shad continued to sag in the 1900s, the long-lived sturgeon essentially disappeared from the Potomac, and intense fishing pressure shifted to other species. A 1921 newspaper article commented that "the Potomac 'fisheries' have receded downstream until they have virtually passed out of the river....and relatively small

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[1] The historical perspective was synthesized from information cited in several sources: Lippson et al. (1979), Tilp (1978), Chesapeake Bay Futures Report (in prep.), Mason, W. T. and K. C. Flynn, eds (1976), Richkus et al. (1994), and the US Commercial Fisheries Reports (1889, 1892). Numerous 1980s and early 1990s news articles collected by J. Cummins (Interstate Commission on the Potomac River Basin) were also reviewed.

numbers of fish reach the [spawning] rivers..." (The Evening Star, Washington, D.C., April 3, 1921). Fisheries managers at the time blamed the declines on local pollution (principally raw sewage from Washington DC), the "extreme wastefulness" of fish harvest methods, and overexploitation (The Evening Star, Washington, D.C., undated newspaper article, 1937). Fish blockages were recognized as an additional cause of the alosid (herring, shad) declines. Stocking programs for several species, including shad, were implemented at local fish hatcheries. While these efforts temporarily propped up some populations, shad and herring harvests in the Potomac sank into insignificance in the 1970s, showing a >98% decline over 140 years. By that time, heavy fishing pressure was focused on yellow perch, striped bass, spot, catfish, blue crab and especially menhaden, a phytoplankton-eating species that had been abundant in the 1940s and 1950s, declined precipitously in the 1960s, and became abundant again in the 1970s and 1980s. Blue crab currently accounts for most of the annual fish harvest. Fisheries independent surveys performed in the Potomac by the Maryland Department of Natural Resources (MDDNR) have documented population fluctuations in both commercial and non-commercial species since 1958. Many commercial species are trending downward (e.g. menhaden, spot) or have not recovered to previous highs (e.g. river herring, shad). Striped bass, and incidently white perch, were boosted by aggressive management and favorable spawning conditions in the 1990s, and returned to somewhat higher numbers (Hornick et al, 1997). The Atlantic States Marine Fisheries Commission (ASMFC), the Chesapeake Bay Program and state natural resource agencies are developing management plans to conserve and restore fish populations.

Harvest pressure on oysters rose to very high levels in the 18th and early 19th centuries, but then began to parallel finfish harvest declines in the late 19th century. According to the US Fish Commission of 1892, the Potomac estuary at that time had 42 square miles of oyster dredging grounds. This is approximately half of the mesohaline bottom areas with depths less than 18 feet, where oyster reefs typically grow. Harvests fell from ~9.6 million pounds in the 1880s to ~ 2 - 5.2 million pounds in the 1960s. The drop in harvest size reflects a long-term decline in Potomac oyster reefs during this period caused by over-exploitation, reef habitat destruction due to oyster dredging and tonging methods, the increasing extent and duration of low dissolved oxygen in the Potomac mainstem, and low, erratic reproduction in the mainstem (Beaven 1945, Haven 1976). An estimated 70% of the already depleted Potomac population was lost and never completely recovered when Tropical Storm Agnes brought record summer flows into the estuary in 1972 (Richkus et al 1994). Finally, intensifying oyster diseases have further decimated population levels in recent decades. Potomac harvests still continued and by the 1980s they amounted to less than 1.5 million pounds per year, a more than 84% decline in one century. The loss of this filter feeder was particularly important to the Potomac and the entire Chesapeake estuarine ecosystem. Historic oyster populations probably filtered the mesohaline waters of Chesapeake Bay in less than three days (Newell 1988). They removed phytoplankton and detritus from the water, keeping it clear. Present day low oyster populations take approximately one year to filter that much water and are no longer effective algal consumers.

The oyster declines brought about concurrent declines in communities dependent on the physical structure of the reefs. Reefs of living oysters support productive, diverse assemblages of epibenthic algae, invertebrate scrapers, grazers and predators, and small fish consumers. Heavy harvest pressure and the methods used to collect oysters eventually flattened most reefs in the Potomac, destroying the hard surfaces and vertical elevations needed by both the oysters and the reef community species (Lippson et al 1979).

The intense 19th century harvests and large 20th century declines in fish and shellfish consumers of the food web must have had a profound impact on the tidal Potomac ecosystem. Important food web links in the middle and upper trophic levels were reduced or removed, and the ecosystem became dependent on a less diverse assemblage of consumers. Ecological theory suggests this lower diversity would have weakened the Potomac ecosystem's resilience to natural stresses such as Tropical Storm Agnes in 1972. Biomass produced by the algal community, which by this time was responding to increasing nutrient inputs from a growing human population, tended to enter other trophic pathways. Although no direct monitoring data exists that documents this shift to other pathways, the redirected primary production would most likely have boosted the Potomac's bacterial, protozoan and rotifer consumers of the food web, and facilitated eutrophication of the tidal Potomac Estuary.

Eutrophication and the Clean Water Act

Eutrophication is the enrichment of nutrients, especially nitrogen and phosphorus, to a level that supports a dense population of algae. Algal growth under these conditions has surpassed the abilities of consumers to remove algal biomass, and the accumulating biomass eventually dies and decays, causing oxygen depletion. If oxygen levels become very low (hypoxia and anoxia), less nitrification occurs in the sediments and more phosphorus is released from bottom sediments, further enriching the water (Boynton, this report; 1982). In very eutrophic systems, algal growth becomes limited by the algal population itself which blocks underwater light ("self-shading") and by the increased turbidity caused by secondary results of eutrophication (e.g. bacterioplankton, organic detritus). Ambient nutrient concentrations under these conditions are not limiting and hence not fully utilized or incorporated into the food web. Shifts in algal community composition to undesirable species (e.g. blue-green algae) and changes in SAV occur. The comparatively long flushing time of the Potomac Estuary and the tendency of its water column to stratify make it especially sensitive to variations in nutrient fluxes and hence eutrophication (Walker et al, in press).

Nitrogen, phosphorus and sediment inputs from the landscape were minimal until the late 1700s when the use of the iron moldboard plow and a shift from tobacco to wheat and other grain crops began to affect the environment (Walker et al, in press, DeFries 1986). Rapid population growth resulted in a sharp climb in agriculture, associated infrastructure and demands on natural resources well into the 1800s. The Civil War (1861-1865) and its aftermath had varying impacts on the landscape of the Potomac estuary. Some farmland was abandoned and reclaimed by young forests, vast old-growth forests in the western basin were harvested, westward migrations intensified, and the industrial revolution encouraged the development of urban centers. Extreme nutrient enrichment from raw sewage occurred in the tidal river near Washington DC during the Civil War and thereafter, however because the metropolitan area's population was relatively small (less than half a million) pollution problems were local (Jaworski 1990).

In the 20th century, nitrogen and phosphorus inputs to tidal waters from the upper basin rose several fold as land uses continued to change and intensify. After World War II, the increased use of manufactured fertilizers became very evident in fall-line loadings (Appendix B). Nutrient pollution from sources below the fall-line intensified as human population numbers climbed steeply after 1930 and wastewater flows increased (Figure 3). Blue Plains WWTP came on-line in 1938 with primary treatment (removal of solids) but conditions continued to worsen and secondary treatment was initiated in 1958. The Capitol area "cesspool" reached crisis levels in the 1960s, and the floating algal mats, dead fish and stench of the upper tidal river were a major impetus for the Clean Water Act of 1972. Water quality managers from state, local and federal governments reacted to the poor 1960s condition of the Potomac estuary by further improving wastewater treatment facilities and implementing a ban on phosphate detergents in Maryland, Virginia and the District of Columbia. In 1983, the Executive Council of the Chesapeake Bay Program signed the Chesapeake Bay Agreement which called for a cooperative effort involving different states and levels of government to reduce pollution entering the Bay. In 1987, the Bay Agreement was updated to include a 40% reduction in nitrogen and phosphorus entering the Bay by the year 2000. Sewage treatments to remove solids, then phosphorus, and most recently nitrogen are alleviating the Capitol area eutrophication problems. Concentrations of phosphorus and recently nitrogen have also declined in the fresh waters flowing into the tidal Potomac at the fall-line. These gains, however, could be overwhelmed if the basin's human population continues to grow at its current, uncontrolled rate and animal husbandry operations continue to intensify.

Like the loss of fish and shellfish consumers in the food web, eutrophication had a profound impact on the tidal Potomac ecosystem. It fueled large, oxygen-depleting algal blooms in the tidal fresh river below Washington and posed serious health risks to humans. It greatly exacerbated the stratification-dependent hypoxic zone in the lower estuary. It facilitated the export of nutrients, and especially nitrogen, downstream to the lower Potomac and the Chesapeake Bay. It increased bacterioplankton and organic detritus in the water.

Decline of Underwater Grasses

While over-exploitation of fish stocks, habitat destruction and blockages, and eutrophication impacted open water food webs in the Potomac, another ecologically significant change was occurring in shallow waters. Eelgrass blight decimated SAV beds in the lower estuary during the 1930s and they are only now returning (Appendix E). The lush Potomac submerged aquatic vegetation (SAV) beds in the upper tributary were invaded in the 1930s by water chestnut, *Trapa natans*, and in the 1950s and 1960s by Eurasian water milfoil, *Myriophyllum spicatum* (Lippson et al 1979). Both exotic SAV species displaced the native species and then crashed, and the native species did not return after the second invasion. Their lack of success in re-colonizing the tidal river was probably related to the poor water quality conditions in the 1960s (Carter and Rybicki 1986). The advent of a hardy exotic species, *Hydrilla*, in the early 1980s has partially restored the SAV beds to the upper and middle Potomac River and helped some native species return. However, the aerial extent of SAV beds in the entire river today is only 9% of their probable former coverage (CBP Tier III goal). The full habitat and food value of the native species assemblage is still missing and some waterbird consumers (e.g. redheads, canvasbacks) have not returned.

The disappearance of the vast SAV beds, especially in the tidal fresh zone after the second exotic species invasion, must have also had a profound impact on the Potomac ecosystem. Substantial populations of invertebrate consumers associated with the SAV beds were lost. The important sediment-trapping function of SAV beds was lost and shoreline erosion was accelerated, adding to the river water's increasing turbidity. Important habitat structure for fish (especially young-of-year life stages) and crabs was lost, and waterbirds were displaced.

Chemical Contaminants

Chemical contaminants are a recognized problem in the upper and middle Potomac River estuary, especially in the tidal Anacostia River. At various locations in the mainstem and tidal tributaries, metals such as copper, zinc, nickel, cadmium, and lead, polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and/or pesticides are present in concentrations that probably have "adverse effects on living resources" (http://www.chesapeakebay.net/content/publications/cbp_12443.pdf).

Changing Hydrology Patterns

A more subtle impact on the Potomac estuary in the past 200 years has been the changes in basin land cover as a result of lumber harvesting, agriculture and urban growth. Land cover affects the hydrological cycle, and ultimately freshwater flows to the estuary, through its influence on soil moisture, groundwater replenishment, surface water runoff, erosion, and evapotranspiration. Freshwater flow regulates residence times of the estuary, determines nutrient and sediment fluxes into the estuary, moves salinity zones up and down the estuary's longitudinal axis, and alters the strength of vertical salinity gradients and hence anoxia at the mouth of the estuary.

Farming and lumbering reduced the original forest cover from >95% to about 40% in the Chesapeake Bay basin (includes the Potomac River basin) between the time of European settlement in the 1600s and the Civil War (Chesapeake Bay Program 1999). The Potomac River basin above the fall-line was not affected by significant land clearing until after the Revolutionary War, therefore the 1680 - 1800 period in this area represents a fairly undisturbed hydrology. Rapid land clearing in the basin occurred from 1800 to the Civil War (1861 - 1865), after which many farms were abandoned and forests gradually reclaimed up to 60% of their former extent. However, population in the region began to climb rapidly after the Civil War as a result of higher immigration rates, urbanization and industrial development. Populations climbed especially fast in and around metropolitan areas after 1930. Forest recovery slowed and eventually reversed about 1970. River flow patterns in the 1865 - 1998 period are therefore from a landscape with multiple, shifting land uses.

Land use changes during the 1800s and 1900s are associated with an increased variance in Potomac River baseflows, higher extreme (wet, dry) baseflow conditions, and more frequent dry baseflow events (Figure 4). Baseflow is the movement of groundwater into surface streams and rivers and it is the major source of freshwater inflow to the Potomac estuary between rain events and during droughts. The lowest river flow observed in a season, especially summer, is a good proxy for baseline flow since the influence of current and recent rainfall are usually minimal (Hagen et al 1999). Furthermore, there appears to be a close relationship between summer baseflow condition and the tree ring based Palmer Drought Severity Index (PDSI), a measure of historical soil moisture conditions during the summer growing season (Figure 4). Using the tree-ring based PDSI as a surrogate for summer baseflow condition, it is evident that average summer baseflow levels probably did not change significantly after 1800 but that variance in the baseflow might have increased by about 40% ($p=0.017$). Six post-1800 baseflows peaks were as much as 24% higher than the pre-1800 baseflow maximum value. Four post-1800 baseflow peaks were as much as 33% lower than the pre-1800 baseflow minimum value. Finally, very low baseflows (i.e. below 10^{th} percentile of all data points) were 38% more frequent after 1800.

Total surface flows (baseflow plus the short-term runoff amounts from rainfall and snow melt) entering the Potomac estuary at the fall-line gaging station at Little Falls Dam seem to have increased in the past three decades (Appendix A), but historical flow records only go back to 1930 at that gage. The longer Point of Rocks gaging record upstream of the metropolitan area, however, shows no long-term trend occurring since 1895. Multiple-century trends in total surface flows are not available, and prehistoric proxies or indicators were not analyzed for this report. The higher total flows in the 1990s have decreased the overall salinity of the estuary, reduced residence times, increased total nutrient and sediment loadings to the estuary and tended to intensify stratification in the lower estuary (Appendix A, Appendix C).

The enormous increases in water withdrawals above the fall-line after 1930, to supply potable water to the Washington metropolitan area, have significantly altered low flow levels in the approximately two mile stretch of river between the river water intake pipes and the head of tide. Much of the diverted water, however, returns to the upper estuary from waste water treatment plants (Steiner, personal communication). Three dams built in Potomac tributaries above the fall-line since 1950 to create water supply reservoirs and reservoir releases to bolster river flow during droughts have altered Potomac flow patterns somewhat, especially in periods of very low flow, however their impact on the estuary is probably minimal.

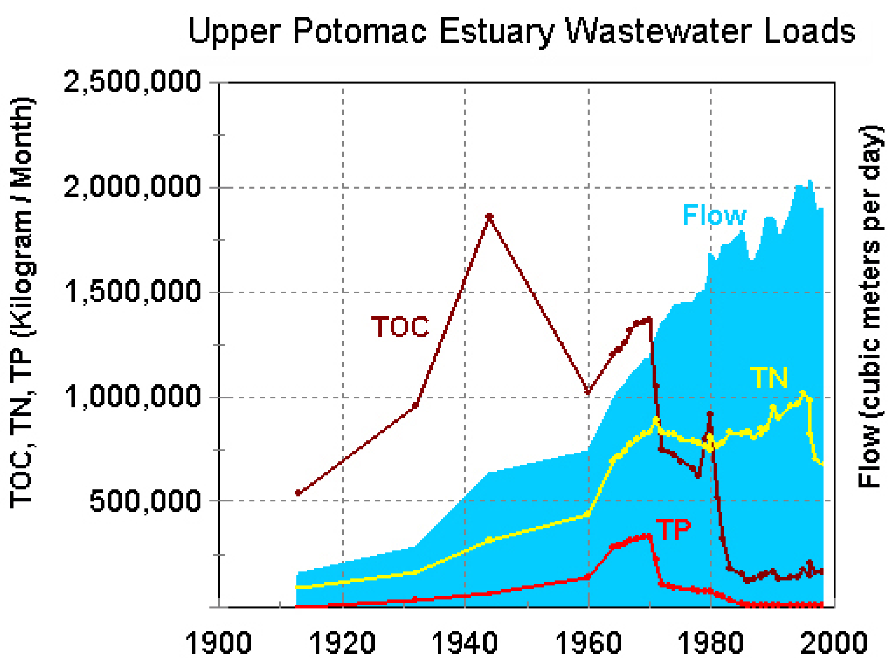

Figure 3. Wastewater flows and total nitrogen, total phosphorus and total organic carbon loadings into the upper Potomac Estuary since 1913 (data from Jaworski).

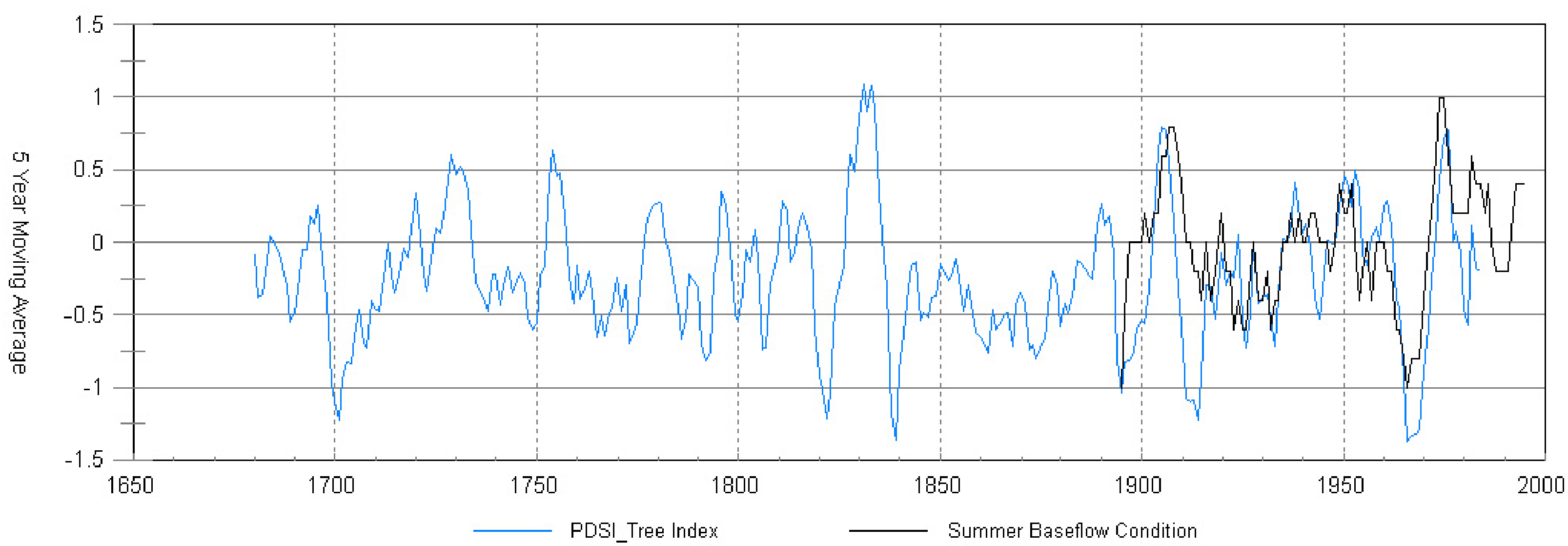

Figure 4. The five-year moving average of the tree ring based Palmer Drought Severity Index (PDSI) and the five-year moving average of a yearly summer baseflow condition index. PDSI Tree Index: values were obtained from Potomac Basin Grid Point 144 of the NOAA Paleoclimate Program Tree Ring Network (Cook et al 1999). PDSI is a meteorological drought index related to summer precipitation inwhich negative numbers denote dry conditions and positive numbers, wet conditions. Summer Baseflow Condition: the lowest summer flow value for each year was obtained from the 1895-1998 flow record at the Point of Rock gaging station. Hagen et al (1999) classifications of baseflow condition were used (wet = top third all flow gage; medium = middle third; dry = lowest third). The classifications were converted to a numeric index as follows: wet = +1, medium = 0, dry = -1.

NUTRIENT LOADS AND AMBIENT NUTRIENT CONCENTRATIONS (OBJECTIVE 1)

Introduction

This section of the report summarizes and updates two larger, more detailed sections that describe water quality of the Potomac River. The *Water Quality Section of the Potomac Integrated Analysis Report* (Appendix A) describes the field methods, sampling regime, laboratory methods, changes in detection limits over time, and the statistical methods that were used to adjust for flow and analyze the data. In addition, trend results for surface and bottom, observed and flow-adjusted data for two time periods (mid-1970s-1996 and 1985-1996) are described for the following variables: total phosphorus (TP), ortho-phosphate (PO_4), total nitrogen (TN), dissolved inorganic nitrogen (DIN), nitrite plus nitrate (NO_{23}), ammonium (NH_4), total suspended solids (TSS), Secchi depth (SECCHI), chlorophyll *a* (CHLA), dissolved oxygen (DO), pH, and total alkalinity (TALK). Trends for the above variables, time periods, and layers are described in the *Water Quality Section of the Potomac Integrated Analysis Report* for the following seasons: annual, March-May, April-June, June-September, July-September, April-October, October-November, and March-November. Water quality trends for a number of seasons were analyzed to allow living resources scientists to compare and link changes in water quality to changes in living resources. The *Water Quality Section of the Potomac Integrated Analysis Report* also examines changes in nutrients, flows, and loads at the Chain Bridge river input monitoring station in Washington, D.C., and at the major (0.5 millions of gallons day^{-1} or greater) wastewater treatment plants (WWTPs) that discharge to the upper tidal Potomac River.

A Historical Analysis of the Eutrophication of the Potomac River (Appendix B) describes changes in water quality, long-term landscape loadings, and trends. The *Historical Analysis* has two basic objectives. For the period 1900-1995, the *Historical Analysis* examines changes in: annual landscape loadings from animal and crop production, air deposition, and wastewater to the upper basin; water quality, average annual concentrations and riverine export fluxes of the Potomac River above Washington, D.C.; and, the annual nutrient loadings from WWTPs directly discharging to the upper estuary. The second objective, which covers the 1965-1996 time period, is to describe: the combined (upper basin and estuarine wastewater) monthly nutrient loadings and how control measures have affected the loadings; results of trend analyses in nutrient concentrations, DO, and SECCHI depth; the consequences of eutrophication in terms of low levels of bottom water DO, elevated concentrations of CHLA and total organic carbon (TOC), and decreased light penetration; and, conduct an analysis of how well the ambient estuarine nutrient concentrations can be predicted from the combined loading estimates.

The specific objectives of this section of the Potomac Integrated Analysis Report are to: summarize trends for three time periods, which include short-term (1985-1998) and long-term (mid-1970s-1996 and mid-1960s-1996); describe historical landscape loadings for 1900-1995; discuss changes that have occurred in fall-line loadings, flow, and nutrient concentrations; and, discuss changes that have occurred at the WWTPs.

Trend results for the 1985-1998 time period were taken from the Data Analysis Workgroup (DAWG) June 11-12, 1999 workshop (the DAWG is part of the Monitoring Subcommittee of the

U.S. Environmental Protection Agency Chesapeake Bay Program). Trend results from the DAWG workshop were added to this report to update the trends that appear in the *Water Quality Section of the Potomac Integrated Analysis Report* because in the time it has taken to finalize the larger report, two additional years of water quality data have become available. For the DAWG workshop, trend analyses were conducted on TP, filtered ortho-phosphate (PO_4F), TN, DIN, TSS, SECCHI, CHLA, and DO. Depending on station depth and salinity, either surface, surface mixed or bottom, bottom mixed data were analyzed (Surface mixed and bottom mixed layers were used for XDC1706, MLE2.2, and MLE2.3 because only these stations develop stratified layers that result from salinity differences with depth. To make description of the trends easier, layers will be referred to as either surface or bottom, even though mixed layers were used for the above three stations). Surface and bottom trends were analyzed for all variables except for SECCHI, which is neither surface nor bottom, and DO, for which only bottom data were analyzed. Stations analyzed for the 1985-1998 time period included XFB2470, XFB1433, XEA6596, and XEA1840 in the tidal fresh mainstem; PIS0033 and XFB1986 on Piscataway Creek; MAT0016 and MAT0078 on Mattawoman Creek; XDA4238 and XDA1177 in the oligohaline zone; and, XDC1706, MLE2.2, and MLE2.3 in the mesohaline zone (Note that trends for Piscataway and Mattawoman Creeks start in 1986 because data collection did not start until that year). The 1985-1998 trends were analyzed using both observed and flow-adjusted data.

Trend results for the mid-1970s-1996 were taken directly from the *Water Quality Section of the Potomac Integrated Analysis Report* and are reproduced here. The trends for the mid-1970s-1996 time period are based in part on data collected before the inception of the Chesapeake Bay Program, which started in July 1984. The pre-Bay Program data were collected under the State of Maryland Core-Trend monitoring program. The early Core-Trend data are available only for some Potomac mainstem stations, including XFB2470, XFB1433, XEA6596, XEA1840, XDA4238, XDA1177, XDB3321 (sampling stopped at this station in September 1990), and XDC1706. Variables analyzed for trends for the mid-1970s-1996 period included TP, whole ortho-phosphate (PO_4W), TN, DIN, TSS, SECCHI, CHLA, and field DO (DO_FLD) for surface only and bottom only trends using observed and flow-adjusted data. DO_FLD was analyzed because DO data, which is corrected for temperature, were not available in the mid-1970s data set.

Trends for the 1965-1996 time frame were analyzed for a limited number of stations because early data were not available for all monitoring locations. Stations analyzed for 1965-1995 included: XFB2470 (Piscataway), XEA6596 (Indian Head), XDA1177 (Maryland Point), XDC1706 (301 Bridge), MLE2.2 (Ragged Point), and MLE2.3 (Point Lookout). Variables analyzed included TN, DIN, TP, total organic carbon (TOC), CHLA, DO, and SECCHI. Surface data were analyzed for all variables except DO, for which only bottom layer trends were calculated. Only observed data were analyzed for trends for 1965-1996. The 1965-1996 data set was compiled from a number of sources including the U.S. Geological Survey, the U.S. Environmental Protection Agency, Johns Hopkins University, the State of Maryland Core-Trend Program, and the Chesapeake Bay Program. The complete list of data sources appears in *A Historical Analysis of Eutrophication of the Potomac Estuary*.

In preparing the 1965-1996 data set for trend analyses, the data were not censored to the highest detection limit over time because early detection limits were not know. As a result, there may be a "step" component in the magnitude of change that resulted from improvements in detection limits and not from changes in water quality. The potential for detection limit-related step trends in the data is assumed to be small, because most of the nutrients analyzed are "totals" which are generally present at detectable levels.

The Seasonal Kendall test of monotonic trend was used on observed data for all trends and on flow-adjusted water quality data if the number of censored flow-adjusted data did not exceed 5% (flow-adjusted trend results are not available for the 1965-1996 time period). In cases where the number of censored data exceeded 5%, trends in flow-adjusted long-term data (mid-1970s-1996) were analyzed using the Statistical Analysis System (SAS) LIFEREG procedure, an implementation of Tobit regression. For short-term flow-adjusted data (1985-1998), when the number of censored data exceeded 5%, the significance level is reported as "BDL" for below detection limit. Data were adjusted for flow to control for any statistically significant relationship with flow. Trends in observed data were considered significant if they were detected at the 99% confidence level ($p \leq 0.01$) and at the 95% level ($p \leq 0.05$) for flow adjusted trends.

General Conclusions

Numerous changes have occurred in the upper Potomac basin that have affected water quality in the estuary. Since the 1900s, population density has tripled to a current density of 16.6 people/km^2, milk, beef, and egg production have increased, and until the mid-1970s, emissions of NO_x steadily increased (Appendix B). These changes have caused increases in the loadings of nitrogen, phosphorus, and potassium, the major terrestrial nutrients, to the upper basin (see Figure 5, Appendix B). The 12-month running average TN concentrations for the WWTPs and the fall line are shown in Figure 6 (from Appendix B) for the 1964-1996 time frame. The figure shows that TN loadings at the river input station range from 400,000 kg/month in the late 1960s to 5,000,000 kg/month in 1996. The fluctuations in that time period indicate that the loads delivered to the upper estuary are primarily a function of flow. TN loads from the WWTPs range from 700,000 kg/month in 1964 to over 1,000,000 kg/month in 1995 and drop-off slightly thereafter. The drop-off in TN from the WWTPs may be in part the result of denitrification at the Blue Plains WWTP, which started a pilot project to remove nitrogen from the effluent in October 1996. Figure 6 clearly indicates that except for brief periods, fall line loads greatly exceed those from the WWTPs.

Trends in TN concentrations vary with the start and end dates of the analysis period. For the 1965-1996 time frame, TN increased at four of the six stations for which long term data are available (see Table 1, Appendix B). For the mid-1970s-1996 data set, TN trends are still increasing; however, for 1985-1998, the trends in TN concentrations have clearly reversed and are decreasing significantly. It is assumed that the reversal in TN trends is the result of upgrades at the WWTPs, since there is no clear pattern in the fall line loads.

There is also no clear pattern to the TP fall-line loads, which range from 30,000 kg/month to 500,000 kg/month, and it is again assumed that the loads are mostly a function of flow. TP loads from the WWTPs showed a slight increase from 1964 through the early1970s, when they reached 350,000 kg/month. After reaching their peak, TP WWTP loads show clear step trends with decreases occurring in the early1970s and mid-1980s, coinciding with the WWTP upgrades and the phosphate detergent ban, respectively (see Figure 7, Appendix B).

Trends in TP concentrations decreased dramatically at five of the six stations for which 1969-1996 data are available (see Table 1). Decreasing TP trends are also prevalent in the mid-1970s-1996 data set, but are less prevalent in the 1985-1998 data. The pattern in TP trends for the three time periods no doubt resulted from the timing of the interventions to control phosphorus being discharged from the WWTPs.

Trends in PO_4 are difficult to interpret because of changes in the analysis method that occurred in October 1990; however, it seems reasonable to assume that PO_4 trends would be similar to those of TP.

Trends in CHLA for the 1965-1996 time period are significantly decreasing at five of the six stations for which long-term data are available (see Table 1). Some decreasing and some increasing trends in CHLA were observed in the mid-1970s-1996 period, and mostly increasing trends were detected in the 1985-1998 data. The reasons for these patterns in CHLA trends are not entirely clear. Although the long-term trend in CHLA is decreasing, the decreases may have been driven by a few large blooms of blue-green algae that occurred early in the trend period as opposed to a consistent steady decrease over time. Another reason for the increases observed over the 1985-1998 time period is that although nutrients have been reduced in the estuary, the system is still nutrient saturated, which would allow chlorophyll concentrations to increase until self-shading by algae occurs.

Trends in TSS for the mid-1970s-1996 are significantly increasing. For the 1985-1996 time frame although there are still some increasing TSS trends, there are fewer than there were for the earlier time period. Perhaps that is a sign of improvement.

The trends described above for the tidal Potomac River were no doubt a result of the continuing improvements to wastewater treatment facilities, bans on phosphate detergents implemented in 1985 (Maryland) and 1986 (Virginia and the District of Columbia) and the initiation denitrification at Blue Plains. These improvements have led to substantially improved nutrient levels and organic matter in the Potomac Estuary.

Long-Term Water Quality Trends for Six Estuarine Sampling Stations (1965-1996)

Trend results for 1965-1996 are presented in Table 1. Over the 30-year period, TN concentrations increased by 21% at XEA6596 to 73% at MLE2.2. Other stations with significant TN trends include XDA1177 and XDC1706. Significant trends in DIN were detected at five of the six stations for which long-term data are available. DIN increased by 41% to 80% at the upper four stations (DIN trends at MLE2.2 were not significant). DIN decreased by 42% at MLE2.3, which is located at the mouth of the Potomac River. Some of the largest trends were observed for TP, which was the focus of early efforts to reduce nutrient pollution in the Potomac estuary. Decreases in TP ranged from 24% at MLE2.2 to over 95% at XFB2470. The only station that did not have a significant trend in TP was MLE2.3.

Significant decreasing trends were detected at all stations for TOC. The decreases in TOC ranged from 12% at MLE2.3 to 61% at XEA6596. Significant decreasing trends were detected at five stations for CHLA, which responded to decreases in phosphorus and carbon at the WWTPs. Decreases in CHLA ranged from 30% at XFB2470 to over 95% at XDA1177. The CHLA trend was not significant at MLE2.3. Conflicting trends were observed for bottom DO. DO increases were observed at two tidal fresh stations (60% at XFB2470 and 20% at XEA6596). Despite improving trends in TP, TOC, and CHLA, bottom DO decreased by 20% at XDC1706. Only one significant trend was detected in Secchi depth, a 30% decrease at MLE2.3.

Twenty Year Trends (mid-1970s-1996)

Surface Non-flow Adjusted Trends (mid-1970s-1996)

For the long-term trends, significant decreases in TP were detected in the tidal fresh and oligohaline zones. Decreases in the tidal fresh zone ranged from 27% to 39%. A 22% decrease in TP was observed at one oligohaline zone station. Significant trends in PO_4W were only detected in the tidal fresh zone where decreases ranged from 40% to 61%. Increasing trends in TN were observed in both the tidal fresh and oligohaline zones. Tidal fresh TN increases ranged from 19% to 38% for the four stations. TN increases of 19% and 35% were observed at two oligohaline zone stations. Significant increases ranging from 20% to 39% were detected at three tidal fresh stations for DIN. All three stations in the oligohaline zone had significant increasing DIN trends, which ranged from 24% to 43% (see Figure 8a).

Significant increases in TSS were observed in all three salinity zones. TSS increased by 50% and 60% at two tidal fresh stations, by 70% and 30% at two oligohaline zone stations and by 30% at the one mesohaline zone station with long-term data available. No significant trends in Secchi depth were detected in spite of the increases in TSS. CHLA decreased by 30% at one tidal fresh station and by 20% and 30% at two stations in the oligohaline zone. DO increased by 10% at the upper-most tidal fresh station and decreased by 10% at the lower-most. DO decreased by 10% at the mesohaline zone station (see Figure 8b).

Bottom Non-flow Adjusted Trends (mid-1970s-1996)

There were no significant bottom layer trends in TP. Decreases in PO_4W were observed at all four tidal fresh station and at two oligohaline zone stations. Decreases in the tidal fresh zone ranged from 37% to 57%. Decreases of 34% and 36% were detected at the two oligohaline zone stations. Trends in TN and DIN were not significant (see Figure 9a).

A 90% increase in TSS was detected at XEA6596 in the tidal fresh zone. No other TSS trends were significant. A 20% increase in CHLA was detected at XFB2470 in the tidal fresh zone, which is not where the increasing TSS trend was observed. DO increased by 10% at one tidal fresh station and decreased by 10% at another. A 10% decrease in DO was also observed at one oligohaline zone station (see Figure 9b).

Surface Flow Adjusted Trends (mid-1970s-1996)

Significant decreases ranging from 31% to 39% were detected in TP at the four tidal fresh stations. TP decreased by 25% at two stations in the oligohaline zone and by 23% in the mesohaline zone station. Four significant trends were also detected in PO_4W in the tidal fresh zone. The PO_4W tidal fresh zone trends were estimated using the SAS LIFEREG procedure and a percent change calculation that differs from that used for most other trends. As a result, the decreases, which ranged from 49% to 66%, may not be strictly comparable to other trends that were estimated using Sen's slope. No significant PO_4W trends were detected in the oligohaline zone, although a decrease of 46% was detected in the mesohaline zone. Significant increasing trends in TN were detected at all four tidal fresh stations and all three stations in the oligohaline zone. In the tidal fresh zone, TN increases ranged from 17% to 38%, while those in the oligohaline zone ranged from 22% to 46%. Increasing DIN trends were detected at all stations, including the mesohaline zone. Increases in the tidal fresh zone ranged from 20% to 40%, while those in the oligohaline zone ranged from 27% to 57%. DIN at the mesohaline zone station increased by 23% (see Figure 10a).

Three significant increases ranging from 40% to 60% were observed in the tidal fresh zone for TSS. Increasing TSS trends ranging from 30% to 80% were detected at the three oligohaline zone stations. The trend in TSS at the mesohaline zone station was increasing, though not significant. Even though large increases in TSS were observed at almost all stations, no significant trends in Secchi depth were observed. Significant trends in CHLA were observed at only two stations, one in the tidal fresh zone and one in the oligohaline zone. In the tidal fresh zone, CHLA decreased by 30%, whereas CHLA decreased by 50% in the oligohaline zone. DO increased by 10% at one tidal fresh station and decreased by 10% and 20% at two others. DO decreased by 10% at one oligohaline zone station and decreased by 20% at the mesohaline zone station (see Figure 10b).

Bottom Flow Adjusted Trends (mid-1970s-1996)

A significant decreasing TP trend of 38% was detected in the tidal fresh zone. No other significant TP trends were observed. Decreasing PO_4W trends ranging from 48% to 66% were detected at all four tidal fresh zone stations. It should be noted that the PO_4W trend at the upper-most tidal fresh station was estimated using the Seasonal Kendall test and Sen's slope and that trends and slopes at the lower three stations were estimated using SAS LIFEREG. As a result, the percent changes across the four stations may not be strictly comparable. Decreasing PO_4W trends were also found at the oligohaline zone stations. Decreasing PO_4W trends in the oligohaline zone ranged from 25% to 34% (PO_4W trends at stations XDA4238 and XDB3321 were also estimated using SAS LIFEREG and may not be comparable to other percent changes). Only two significant trends in TN were detected, one in the tidal fresh zone, where TN increased by 24% and one in the oligohaline zone, where a 15% increase was observed. DIN also increased at one tidal fresh zone station (18%) and at one oligohaline zone station (28%) (see Figure 11a).

One significant trend was detected in TSS, an increase of 50% at XDC1706. CHLA increased at the upper two tidal fresh stations by 30% and 20%. CHLA decreased by 70% at XDB3321 in the oligohaline zone, where sampling stopped in 1990. Conflicting DO trends were observed in the tidal fresh, where an increase of 10% was detected at the upper station and a 20% decrease was detected at the lower station. DO decreased by 10% at two oligohaline zone stations and decreased by 20% at XDC1706 in the mesohaline zone (see Figure 11b).

Recent Trends (1985-1998, 1986-1998)

Surface Non-flow Adjusted Trends (1985-1998) and (1986-1998)

One significant trend in TP was observed in the Potomac mainstem, where TP decreased by 24% at MLE2.2 in the mesohaline zone (TP also decreased at XDC1706, but the trend was not significant at $p \leq 0.01$). A statistically significant decrease was observed in PO_4F at XEA1840; however, the magnitude of the decrease is not reported because of the number of censored data. Both TN and DIN decreased at all tidal fresh stations. Decreases in TN were fairly consistent across the four tidal fresh stations, ranging from 27% to 31%. Decreases in DIN were in the same range for the upper three tidal fresh stations (26%-29%). The DIN decrease at the lower-most tidal fresh station (XEA1840) was 43%. TN decreased at one oligohaline zone station (XDA4238) by 27% and at one mesohaline station (XDC1706), but the trend was not significant at $p \leq 0.01$. DIN decreased by 34% at XDA4238. DIN also decreased at XDA1177 and XDC1706, but the trends were not significant at $p \leq 0.01$ (see Figure 12a).

A non-significant increase in TSS was observed at the lower-most tidal fresh station (XEA1840). TSS increased by 38% and 33% at XDA4238 and XDA1177, respectively in the oligohaline zone. An 82% increase in TSS was detected at XDC1706 and a non-significant increase was observed at MLE2.2. Secchi depth decreased at almost all stations where TSS increased; however, significant decreases in Secchi depth were detected at only two stations, XDC1706 (25%) and MLE2.3 (17%) in the mesohaline zone (there was no trend in TSS at MLE2.3). CHLA increased at all stations except MLE2.2 and MLE2.3 in the mesohaline zone. The CHLA increases were significant at three of the four tidal fresh stations (not significant at XEA6596), where changes ranging from 23% to 74% were detected. CHLA increased by 40% at XDA4238 in the oligohaline zone; the trend at the other oligohaline zone station was increasing but not significantly (see Figure 12b).

Trends of 13-year duration are presented in Figure 12c for stations on Piscataway and Mattawoman Creeks (sampling at these stations did not start until 1986). A decreasing trend of 33% was detected at MAT0078 for TP (trends in TP at the remaining three stations were not significant at $p \leq 0.01$). Conflicting trends were observed for PO_4F in the tidal creeks. PO_4F decreased by 31% at PIS0033 and increased, though not significantly, at MAT0016. Consistent decreases ranging from 25% at MAT0016 to 35% at both Piscataway Creek stations were detected in TN. DIN decreased on Piscataway Creek by 42% and 35% at PIS0033 and XFB1986, respectively. DIN also decreased at MAT0016, but the magnitude is not reported because of the number of censored data.

TSS increased at PIS0033 and at MAT0078. The trend magnitude at PIS0033 is not reported in Figure 12d because of the number of censored data. The trend at MAT0078 was increasing, but not significant at $p \leq 0.01$. A significant increasing trend in CHLA of 121% was detected at XFB1986 at the mouth of Piscataway Creek. CHLA also increased at the upper Piscataway Creek station, but the trend was not significant at $p \leq 0.01$.

Bottom Non-flow Adjusted Trends (1985-1998) and (1986-1998)

TP decreased at three stations on the Potomac mainstem, one each in the tidal fresh, oligohaline, and mesohaline zones. Trend magnitudes for TP are not reported in Figure 13a because the trends were not significant at $p \leq 0.01$. PO_4F decreased only at XEA1840 in the tidal fresh zone; however, the number of censored data exceeded 5% so the magnitude is not reported. Significant decreases in TN were detected at all four tidal fresh stations, where decreases ranged from 26%

to 32%. TN also decreased by 19% at XDA4238 in the oligohaline zone. No other significant TN trends were detected. Significant decreasing DIN trends were observed at the same stations where significant TN trends were observed. In the tidal fresh zone, decreases in DIN ranged from 26% to 45%. The decrease at XDA4238 was 33%.

TSS increased by 102% at MLE2.2 in the mesohaline zone, which was the only station with a significant TSS trend. CHLA increased at all mainstem stations except MLE2.3, which is located in the mesohaline zone. CHLA increases in the tidal fresh zone ranged from 43% to 96%. CHLA increased by 92% and 56% at XDA4238 and XDA1177, respectively in the oligohaline zone. An increase of 113% in CHLA was detected at XDA1706 in the mesohaline zone, the only mesohaline zone station with a significant trend. Increasing trends in DO were observed at all four tidal fresh stations and at one station in the oligohaline zone. Three of the four DO trends in the tidal fresh were significant at $p \leq 0.01$. The range of increases in DO for statistically significant trends was from 16% to 25%. DO increased by 11% at XDA1177 in the oligohaline zone (see Figure 13b).

Surface Flow Adjusted Trends (1985-1998)

TP decreased at three of four tidal fresh stations, both stations in the oligohaline zone, and two of three stations in the mesohaline zone (see Figure 14a). Of the three decreasing TP trends in the tidal fresh zone, only one (17% at XEA6596) was significant. Neither trend in the oligohaline zone was significant at $p \leq 0.05$, the significance level for flow-adjusted trends. In the mesohaline zone, TP decreased by 15% and 18% at XDC1706 and MLE2.2, respectively. PO_4F decreased at the lower three tidal fresh stations; however, trends at two stations were not significant at $p \leq 0.05$ and one station had an unacceptable number of censored data. PO_4F increased at MLE2.3 in the mesohaline zone, but the trend was not significant at $p \leq 0.05$.

Significant decreasing trends in TN were detected at all stations except for MLE2.2 and MLE2.3 in the mesohaline zone. In the tidal fresh zone, the decreases in TN ranged from 22% at the upper-most station (XFB2470) to 38% at the lower-most station (XEA1840). In the oligohaline zone, TN decreased by 32% and 22% at XDA4238 and XDA1177, respectively. TN decreased by 32% at XDC1706, the only mesohaline zone station with a significant TN trend. Significant DIN trends were detected at stations in all three salinity zones. Decreases in the tidal fresh zone ranged from 18% at XFB2470 to 54% at XEA1840. The decreases in the two oligohaline zone stations were in the same general range (46% and 41%). One significant and one non-significant trend in DIN were observed in the mesohaline zone (DIN decreased by 58% at XDC1706, while the trend at MLE2.3 was decreasing though not significant at $p \leq 0.05$).

Significant trends in TSS were only observed in the mesohaline zone, where increases of 53% and 43% were detected at XDC1706 and MLE2.2, respectively (see Figure 14b). A non-significant increasing trend in TSS was observed at MLE2.3. Four trends were observed in Secchi depth, one in the tidal fresh zone and three in the mesohaline zone. The trend in the tidal fresh zone (XEA1840) was decreasing, but not significant at $p \leq 0.05$. Of the three Secchi depth trends in the mesohaline zone, one was decreasing, but not significant (XDC1706), one was increasing, but not significant (MLE2.2) and one was decreasing and significant (17% at MLE2.3). Significant trends in CHLA were detected in the tidal fresh and the oligohaline zones. Substantial increases in CHLA of 209% and 165% were detected at the upper two tidal fresh stations (XFB2470 and XFB1433). CHLA trends at the two lower tidal fresh stations were also increasing, but were of lesser magnitude (73% and 37%). CHLA increased by 81% and 33% at XDA4238 and XDA1177 respectively, in the oligohaline zone.

Bottom Flow Adjusted Trends (1985-1998)

Four decreasing trends and one increasing trend were observed in TP (see Figure 15a). Two of the decreasing trends, one non-significant and one significant, were observed in the tidal fresh zone (TP decreased by 33% at station XFB2470). A significant decrease of 22% was detected at XDA1177 in the oligohaline zone. Conflicting, though non-significant trends in TP were observed in the mesohaline zone, where TP decreased at MLE2.2 and TP increased at MLE2.3. PO_4F concentrations decreased at two stations in the tidal fresh zone and one station in the mesohaline zone; however, the number of censored data preclude reporting the magnitude of change for any station. Significant TN trends were detected at all stations in the tidal fresh and oligohaline zones. The range of decreases in the tidal fresh zone ranged from 23% at XEA6596 to 36% at XEA1840. TN decreased by 27% and 15% at XDA4238 and XDA1177, respectively in the oligohaline zone. DIN decreased in the tidal fresh and the oligohaline zones. DIN decreases in the tidal fresh zone ranged from18% at the upper-most station (XFB2470) to 54% at the lower-most station (XEA1840). DIN decreased by 44% and 34% at XDA4238 and XDA1177, respectively in the oligohaline zone. One decreasing DIN trend was observed in the mesohaline zone at MLE2.3, but the trend was not significant at $p \leq 0.05$.

Only two trends were detected in TSS, one decreasing in the tidal fresh zone (36% at XFB2470) and one increasing in the mesohaline zone (81% at MLE2.3) (see Figure 15b). CHLA increased at all four tidal fresh zone stations and at both oligohaline zone stations. Increases in the tidal fresh zone ranged from 82% at XEA6596 to 165% at XFB1433. One significant and two non-significant trends in DO were observed. A significant decrease of 11% was detected at XDA4238 in the oligohaline zone. A non-significant increase in DO was observed at XFB2470 in the tidal fresh zone and a non-significant decrease was observed at MLE2.3 in the mesohaline zone.

Wastewater Treatment Plant Loads and Flow

Changes in TN Loads and Flows

Figure 16 shows the total nitrogen loads and flows from 1913 to 1996. For that time period, the lowest (estimated) TN load of 2,900 kg/day and the lowest flow of 1.8 m^3/sec occurred in 1913. The maximum TN load of 31,043 kg/day in occurred 1990 and the maximum flow of 23.2 m^3/sec was reached in 1996.

For the period of record, TN loads increased from 2,900 kg/day to 25,289 kg/day, an increase of 772%. During that period, flow increased from 1.8 m^3/sec to 23.2 m^3/sec, or by 1167%. From 1970 to 1996, TN loads increased from 23,449 kg/day to 25,289 kg/day, or by 8%. For the same period, flow increased from 13.7 m^3/sec to 23.2 m^3/sec, or by 70%. Between 1985 and 1996, TN loads actually decreased from 27,004 kg/day to 25,289 kg/day, or by 6%. The decrease in TN loads occurred despite an increase in flow from 19.2 m^3/sec to 23.2 m^3/sec or 21%. The disconnect between TN load and flow occurred in 1990, when TN load reached a maximum of

31,043 kg/day and flow was 21.4 m^3/sec. Between 1990 and 1996 TN loads decreased from 31,043 kg/day to 25,289 kg/day or by 19%; however, flow increased from 21.4 m^3/sec to 23.2 m^3/sec, or by 8% during that period.

Between 1995 and 1996 flow increased by 11%, but TN loads dropped by 14.6%. As stated above, biological nutrient removal (BNR) of nitrogen was initiated at Blue Plains in October 1996. It would be interesting if the drop in TN load could be attributed to the Blue Plains BNR pilot program. However, a comparison of TN concentrations for the major plants indicated that the change resulted from a sharp reduction in TN effluent concentrations between 1995 and 1996 at the Indian Head Naval Warfare Center.

Changes in TP Loads and Flows

TP loads and flows from 1913 to 1996 are also presented in Figure 16. As shown in the figure, the total phosphorus story is quite different from that described above for total nitrogen, since phosphorus control was historically the focus of the WWTP upgrades. TP loads increased from 500 kg/day in 1913 (estimated) to a maximum of 9,825 kg/day in 1970, an increase of over 1800%. For the same period, flow increased from 1.8 m^3/sec to 13.7 m^3/sec, or by approximately 648%. Between 1970 and 1996 TP loads decreased by 98%, or from 9,825 kg/day to 187 kg/day, despite an increase in flow of 70%. For the 1985 through 1996 time period, TP loads declined, from 332 kg/day to 187 kg/day, probably as a result of the phosphate detergent ban, even though flow continued to increase.

Changes in PO_4 Loads and Flows

Orthophosphate loads for 1985-1996 are provided in Figure 17. Unlike TN and TP, earlier data for PO_4 are not available. In addition, the plant records for PO_4 are not complete.

The highest PO_4 load (185 kg/day) for the 1985-1996 time period occurred in 1987 although the highest flow rate (23 m^3/sec) was not recorded until 1996. PO_4 loads decreased in 1988 and 1989, following the implementation of the phosphate detergent ban. Loads increased slightly to 164 kg/day in the following year and then gradually decreased through 1996. For the 1985 to 1996 period, PO_4 loads decreased by 36% despite an increase in flow of 20%.

River Input Station at Chain Bridge

Trend results for Chain Bridge are available only for 1985 through 1998. (Note that USGS trend results for the 1985-1998 time period may differ from those presented in the *Water Quality Section of the Potomac Integrated Analysis Report*. The reason for the potential differences is that the 1985-1998 trends presented herein were based on a different model run than the 1985-1996 trend results.) Recent trend analyses were performed by the U.S. Geological Survey (USGS) on data collected at Chain Bridge (see Table 2) (DAWG Workshop, June 11-12, 1999). Trends were analyzed by USGS for flow-adjusted concentrations, flow-weighted concentrations, and for loads. Trends in flow-adjusted concentrations were calculated to remove short- and long-term flow-related variability and assess the impact of management activities on water quality. Flow-weighted concentrations represent trends in data where the effects of flow are not removed and are therefore similar to the “observed” data collected below the fall-line. Loads are the product of flow and predicted concentrations.

The most consistent trend observed at the fall-line was for NO_{23}, where flow-adjusted concentrations, flow-weighted concentrations and loads all increased. Concentrations and loads of NO_{23} may have increased for a number of reasons, including: nitrification at wastewater treatment plants to convert ammonia to nitrate, the solubility of nitrate and the corresponding lag in concentration reduction due to contributions from groundwater, and the increase in stream flow (71%) that was observed for the time period. Flow-adjusted concentrations decreased for TP, which reflects the role of management actions to reduce phosphorus concentrations above the fall-line. Both flow-adjusted and flow-weighted concentrations decreased for suspended sediments; however, trends in suspended sediment loads were not significant. Load and flow data are presented in Figures 18, 19a, and 19b for TN, TP, and PO_4, respectively.

Table 1. Water quality trends for six extuarine sampling stations, 1965 - 1996. Station name (historic name/CBP name) is given in the left column; Piscataway is just south of Washington, DC, and Point Lookout is at the confluence of the Potomac River and Chesapeake Bay. TN, total nitrogen; DIN, dissolved inorganic nitrogen; TOC, total organic carbon; Chl, chlorophyll a; B_DO, bottom dissolved oxygen; Secchi, Secchi depth in meters. Trend significance: ns, non-significant; ↗, significant upward trend (p<0.01); ↘, significant downward trend (p<0.01).

Station	TN (mg/l)	DIN (mg/l)	TP (mg/l)	TOC (mg/l)	Chl (µg/l)	B-DO (mg/l)	Secchi (m)
Piscataway (XFB2470 / TF2.1)	ns	↗	↘	↘	↘	↗	ns
Indian Head (XEA6596 / TF2.3)	↗	↗	↘	↘	↘	↗	ns
Maryland Pt. XDA1177 / RET2.2	↗	↗	↘	↘	↘	ns	ns
301 Bridge (XDC1706 / RET2.4)	↗	↗	↘	↘	↘	↘	ns
Ragged Pt. (MLE2.2 / LE2.2)	↗	ns	↘	↘	↘	ns	ns
Pt. Lookout (MLE2.3 / LE2.3)	ns	↘	ns	↘	ns	ns	↘

Table 2. Trends in concentrations and loads at Chain Bridge in Washington, D.C., for 1985-1998 (from Data Analysis Workgroup Workshop, June 11-12, 1999).

	Flow-adjusted Concentration		Flow-weighted Concentration		Load	
Constituent	p-value	% Change	p-value	% Change	p-value	% Change
TN	0.2020	-4	0.2125	7	0.0170	82
NO23	**0.0059**	**12**	**0.0001**	**43**	**0.0002**	**158**
TP	**0.0000**	**-47**	0.6033	-8	0.1957	58
PO_4	NA[1]	NA	0.0165	-20	0.2682	38
SSED	**0.0000**	**>-90%**	**0.0080**	**-54**	0.6131	-22

[1]NA - Not available, percent below detect exceeds 20%.

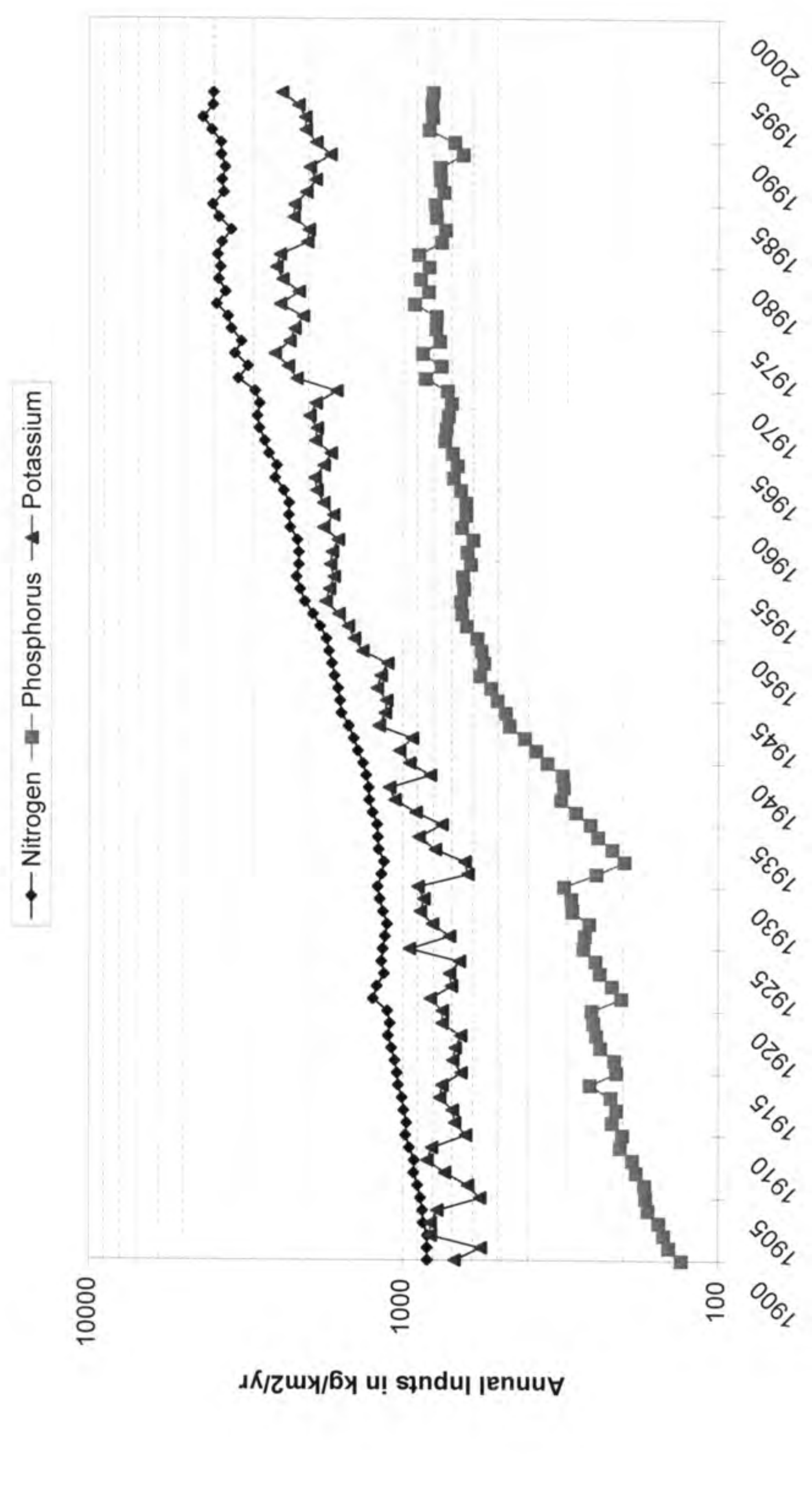

Figure 5. Upper Potomac River Basin Landscape Inputs

POTOMACAWSData7.5

9/7/99

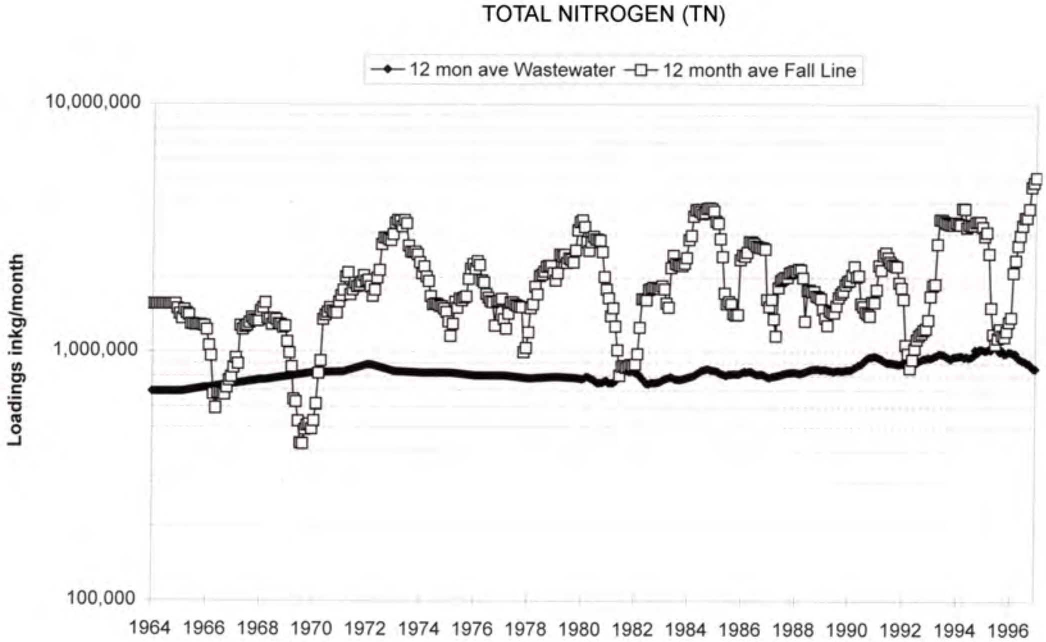

Figure 6. Fall Line & Wastewater TN Monthly Loadings

POTWWRiverfluxes5.5 9/7/99

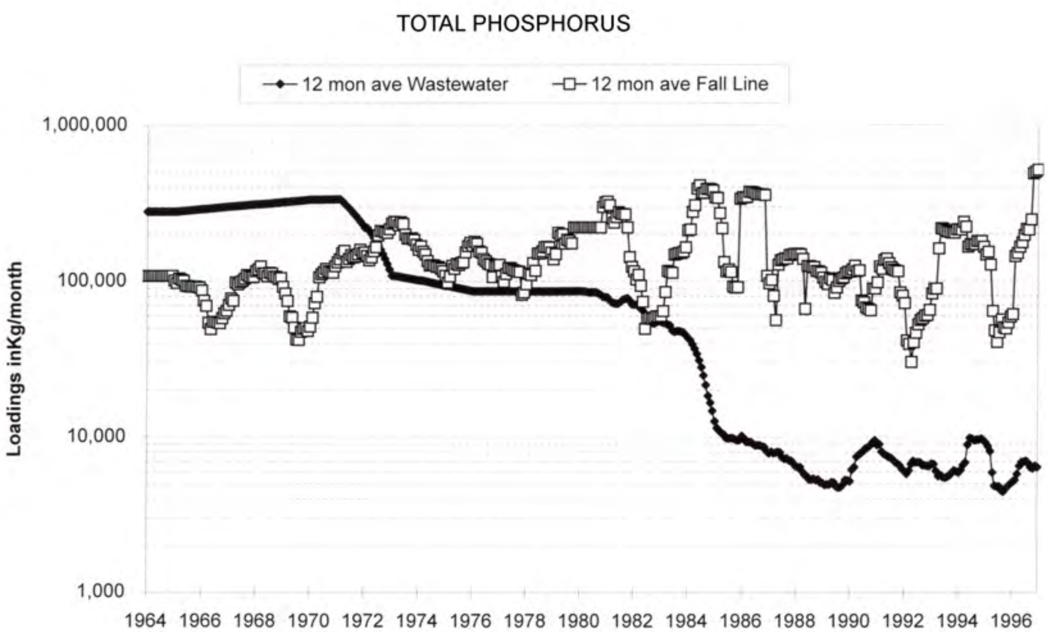

Figure 7. Fall Line & Wastewater TP Monthly Loadings

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Potomac River (mid-1970s-1996)

Twenty-five Year Surface Trends Non-flow Adjusted

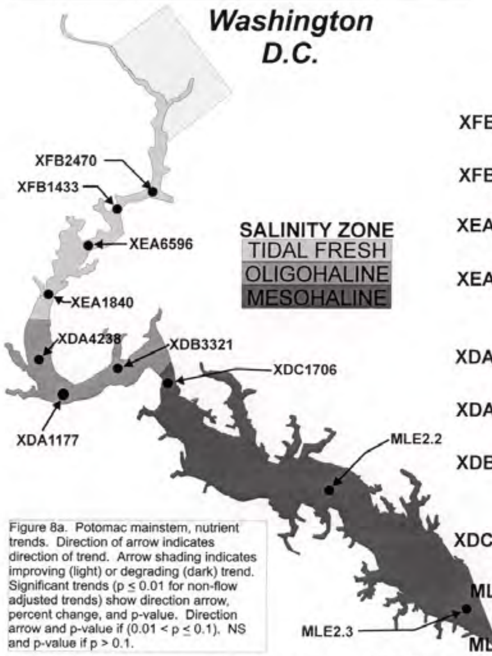


Figure 8a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

	TP	PO4W	TN	DIN
XFB2470	↓ 27% p=0.000	↓ 58% p=0.000	↑ 19% p=0.001	↑ NS p=0.017
XFB1433	↓ 29% p=0.000	↓ 61% p=0.000	↑ 15% p=0.002	↑ 20% p=0.001
XEA6596	↓ 30% p=0.000	↓ 50% p=0.000	↑ 28% p=0.000	↑ 29% p=0.000
XEA1840	↓ 39% p=0.000	↓ 40% p=0.000	↑ 38% p=0.000	↑ 39% p=0.000
XDA4238	↓ 22% p=0.001	NS p=0.207	↑ NS p=0.021	↑ 24% p=0.004
XDA1177	↓ NS p=0.039	NS p=0.191	↑ 19% p=0.001	↑ 36% p=0.003
XDB3321*	NS p=0.107	NS p=0.105	↑ 35% p=0.000	↑ 43% p=0.000
XDC1706	↓ NS p=0.016	↓ NS p=0.052	NS p=0.380	↑ NS p=0.024
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

NA - Data not available for this station. PO4W trends end in September 1990. *Station not sampled after September 1990.

Figure 8a.

Potomac River (mid-1970s-1996)

Twenty-five Year Surface Trends Non-flow Adjusted

Figure 8b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

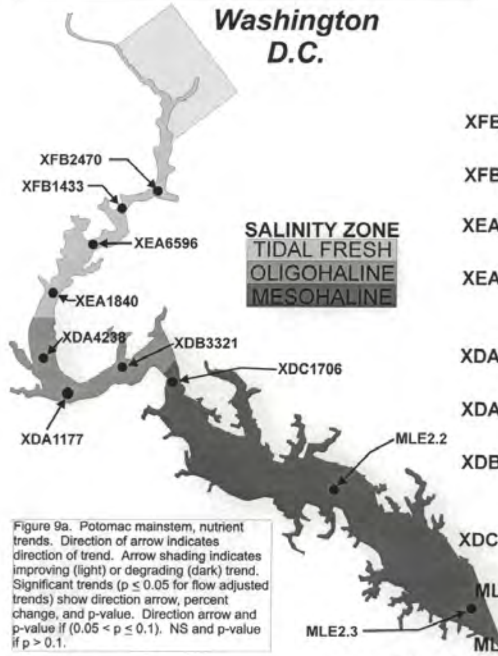
	TSS	SECCHI	CHLA	DO_FLD
XFB2470	↑ 50% p=0.000	↓ NS p=0.043	↑ NS p=0.021	↑ 10% p=0.004
XFB1433	↑ 60% p=0.000	NS p=0.354	NS p=0.830	NS p=0.710
XEA6596	↑ NS p=0.020	NS p=0.876	↓ NS p=0.088	↓ NS p=0.033
XEA1840	NS p=0.351	NS p=0.368	↓ 30% p=0.000	↓ 10% p=0.000
XDA4238	↑ 70% p=0.003	NS p=0.218	↓ 20% p=0.006	NS p=0.192
XDA1177	↑ 30% p=0.010	NS p=0.226	NS p=0.136	↓ NS p=0.069
XDB3321*	NS p=0.140	NS p=0.475	↓ 30% p=0.006	↓ NS p=0.046
XDC1706	↑ 30% p=0.004	NS p=0.446	NS p=0.171	↓ 10% p=0.000
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

NA - Data not available for this station. *Station not sampled after September 1990.

Figure 8b.

Potomac River (mid-1970s-1996)

Twenty-five Year Bottom Trends Non-flow Adjusted



Station	TP	PO4W	TN	DIN
XFB2470	NS p=0.807	↓ 57% p=0.002	NS p=0.228	NS p=0.621
XFB1433	↓ NS p=0.046	↓ 46% p=0.002	NS p=0.908	NS p=0.897
XEA6596	↓ NS p=0.065	↓ 48% p=0.000	NS p=0.219	NS p=0.112
XEA1840	NS p=0.305	↓ 37% p=0.010	↑ NS p=0.010	↑ NS p=0.056
XDA4238	NS p=0.234	↓ 34% p=0.010	NS p=0.424	NS p=0.592
XDA1177	NS p=0.989	↓ 36% p=0.008	NS p=0.175	NS p=0.312
XDB3321*	NS p=0.976	↓ NS p=0.083	NS p=1.000	NS p=0.466
XDC1706	NS p=0.333	NS p=0.330	NS p=0.584	NS p=0.956
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

Figure 9a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1.

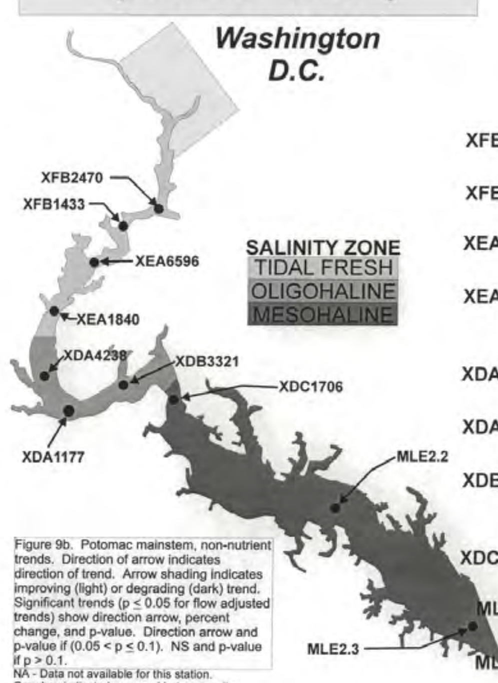
NA - Data not available for this station.

PO4W trends end in September 1990. *Station not sampled after September 1990.

Figure 9a.

Potomac River (mid-1970s-1996)

Twenty-five Year Bottom Trends Non-flow Adjusted



Station	TSS	CHLA	DO_FLD
XFB2470	↑ NS p=0.022	↑ 20% p=0.000	↑ 10% p=0.008
XFB1433	↑ NS p=0.012	↑ NS p=0.039	NS p=0.144
XEA6596	↑ 90% p=0.004	NS p=0.679	NS p=0.372
XEA1840	NS p=0.771	NS p=0.765	↓ 10% p=0.009
XDA4238	↑ NS p=0.070	↑ NS p=0.071	NS p=0.216
XDA1177	↑ NS p=0.049	NS p=0.543	↓ NS p=0.014
XDB3321*	NS p=1.000	↓ NS p=0.076	↓ 10% p=0.004
XDC1706	↑ NS p=0.011	↑ NS p=0.083	↓ NS p=0.050
MLE2.2	NA	NA	NA
MLE2.3	NA	NA	NA

Figure 9b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1.

NA - Data not available for this station.

Gray box indicated seasonal heterogeneity.

*Station not sampled after September 1990.

Figure 9b.

Potomac River (mid-1970s-1996)

Twenty-five Year Surface Trends Flow Adjusted

Station	TP	PO4W*	TN	DIN
XFB2470	↓ 31% p=0.000	↓ 66% p=0.000	↑ 23% p=0.000	↑ 20% p=0.004
XFB1433	↓ 32% p=0.000	↓ 64% p=0.000	↑ 17% p=0.003	↑ 21% p=0.001
XEA6596	↓ 34% p=0.000	↓ 62% p=0.000	↑ 28% p=0.000	↑ 31% p=0.000
XEA1840	↓ 39% p=0.000	↓ 49% p=0.000	↑ 38% p=0.000	↑ 40% p=0.000
XDA4238	↓ 25% p=0.001	NS p=0.210	↑ 22% p=0.001	↑ 27% p=0.000
XDA1177	↓ 25% p=0.022	NS p=0.265	↑ 24% p=0.001	↑ 36% p=0.000
XDB3321*	NS p=0.129	NS p=0.100	↑ 46% p=0.000	↑ 57% p=0.000
XDC1706	↓ 23% p=0.015	↓ 46% p=0.001	NS p=0.300	↑ 23% p=0.017
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

*Tidal fresh PO4W trends were estimated using LIFEREG. Percent change may not be comparable to Sen's method.

Figure 10a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1.

NA - Data not available for this station. PO4W trends end in September 1990. *Station not sampled after September 1990.

Figure 10a.

Potomac River (mid-1970s-1996)

Twenty-five Year Surface Trends Flow Adjusted

Station	TSS	SECCHI	CHLA	DO_FLD
XFB2470	↑ 60% p=0.005	NS p=0.176	NS p=0.110	↑ 10% p=0.002
XFB1433	↑ 40% p=0.006	NS p=0.929	NS p=0.641	NS p=0.726
XEA6596	↑ 50% p=0.011	NS p=0.388	NS p=0.102	↓ 10% p=0.040
XEA1840	NS p=0.181	NS p=0.250	↓ 30% p=0.000	↓ 20% p=0.000
XDA4238	↑ 80% p=0.024	NS p=0.305	↓ NS p=0.069	NS p=0.107
XDA1177	↑ 30% p=0.027	NS p=0.398	NS p=0.262	↓ 10% p=0.004
XDB3321*	↑ 40% p=0.019	NS p=0.315	↓ 50% p=0.001	↓ NS p=0.060
XDC1706	NS p=0.121	NS p=0.510	NS p=0.402	↓ 20% p=0.000
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

Figure 10b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1.

NA - Data not available for this station. *Station not sampled after September 1990.

Figure 10b.

Potomac River (mid-1970s-1996)

Twenty-five Year Bottom Trends Flow Adjusted

*Lower three tidal fresh, XDA4238, and XDB3321 trends were estimated using LIFEREG. Percent change may not be comparable to Sen's method.

Figure 11a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1.

Station	TP	PO4W*	TN	DIN
XFB2470	NS p=0.276	↓ 48% p=0.000	↑ NS p=0.065	↑ 18% p=0.033
XFB1433	↓ 38% p=0.022	↓ 66% p=0.000	NS p=0.555	NS p=0.818
XEA6596	↓ NS p=0.057	↓ 66% p=0.000	NS p=0.127	NS p=0.174
XEA1840	NS p=0.123	↓ 49% p=0.000	↑ 24% p=0.012	NS p=0.108
XDA4238	NS p=0.182	↓ 34% p=0.017	NS p=0.256	NS p=0.735
XDA1177	NS p=0.692	↓ 25% p=0.035	↑ 15% p=0.042	NS p=0.299
XDB3321*	NS p=0.953	↓ 30% p=0.031	NS p=0.465	↑ 28% p=0.033
XDC1706	NS p=0.278	↓ NS p=0.071	NS p=0.537	NS p=0.550
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

NA - Data not available for this station. PO4W trends end in September 1990. *Station not sampled after September 1990.

Figure 11a.

Potomac River (mid-1970s-1996)

Twenty-five Year Bottom Trends Flow Adjusted

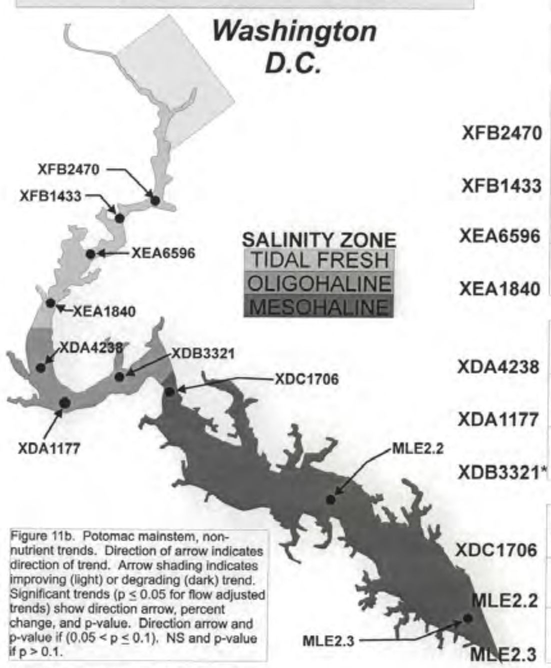


Figure 11b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1.

Station	TSS	CHLA	DO_FLD
XFB2470	NS p=0.462	↑ 30% p=0.000	↑ 10% p=0.003
XFB1433	NS p=0.310	↑ 20% p=0.003	NS p=0.543
XEA6596	NS p=0.146	NS p=0.452	NS p=0.210
XEA1840	NS p=0.891	NS p=0.867	↓ 20% p=0.001
XDA4238	NS p=0.182	NS p=0.197	NS p=0.128
XDA1177	↑ NS p=0.027	NS p=0.924	↓ 10% p=0.006
XDB3321*	NS p=0.972	↓ 70% p=0.001	↓ 10% p=0.028
XDC1706	↑ 50% p=0.040	NS p=0.371	↓ 20% p=0.002
MLE2.2	NA	NA	NA
MLE2.3	NA	NA	NA

NA - Data not available for this station. *Station not sampled after September 1990.

Figure 11b.

Potomac River (1985-1998)

Fourteen Year Surface Trends Non-flow Adjusted

Figure 12a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1. Mesohaline zone stations are surface mixed. Others are surface only.

Station	TP	PO4F	TN	DIN
XFB2470	NS p=0.933	NS p=1.000	↓ 29% p=0.000	↓ 29% p=0.000
XFB1433	NS p=0.521	NS p=0.326	↓ 30% p=0.000	↓ 26% p=0.000
XEA6596	NS p=0.639	NS p=0.238	↓ 27% p=0.000	↓ 28% p=0.000
XEA1840	NS p=0.676	↓ BDL p=0.002	↓ 31% p=0.000	↓ 43% p=0.000
XDA4238	NS p=0.297	NS p=0.229	↓ 27% p=0.000	↓ 34% p=0.000
XDA1177	NS p=0.566	NS p=0.147	NS p=0.146	↓ NS p=0.034
XDC1706	↓ NS p=0.015	NS p=0.375	↓ NS p=0.046	↓ NS p=0.090
MLE2.2	↓ 24% p=0.006	NS p=0.502	NS p=0.877	NS p=1.000
MLE2.3	NS p=0.791	NS p=0.692	NS p=0.445	NS p=0.312

BDL - Number of below detection limit values exceeds 5%.

PO4F trends start in October 1990.

Figure 12a.

Potomac River (1985-1998)

Fourteen Year Surface Trends Non-flow Adjusted

Figure 12b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1. Mesohaline zone stations are surface mixed. Others are surface only.

Station	TSS	SECCHI	CHLA
XFB2470	NS p=0.455	NS p=0.623	↑ 74% p=0.005
XFB1433	NS p=0.174	NS p=0.973	↑ 64% p=0.002
XEA6596	NS p=0.159	NS p=0.151	↑ NS p=0.016
XEA1840	↑ NS p=0.033	↓ NS p=0.011	↑ 23% p=0.003
XDA4238	↑ 38% p=0.008	↓ NS p=0.056	↑ 40% p=0.005
XDA1177	↑ 33% p=0.010	↓ NS p=0.011	↑ NS p=0.075
XDC1706	↑ 82% p=0.000	↓ 25% p=0.002	↑ NS p=0.040
MLE2.2	↑ NS p=0.016	NS p=0.675	NS p=0.489
MLE2.3	NS p=0.141	↓ 17% p=0.003	NS p=0.153

Figure 12b.

Potomac River (1986-1998)

Thirteen Year Surface Trends Non-flow Adjusted

	TP	PO4F	TN	DIN
PIS0033	NS p=0.895	↓ 31% p=0.003	↓ 35% p=0.000	↓ 42% p=0.000
XFB1986	NS p=0.685	NS p=0.204	↓ 35% p=0.000	↓ 35% p=0.000
MAT0078	↓ 33% p=0.002	NS p=0.199	↓ 31% p=0.000	NS p=0.121
MAT0016	NS p=0.896	↑ NS p=0.097	↓ 25% p=0.000	↓ BDL p=0.001

PO4F trends start in October 1990.

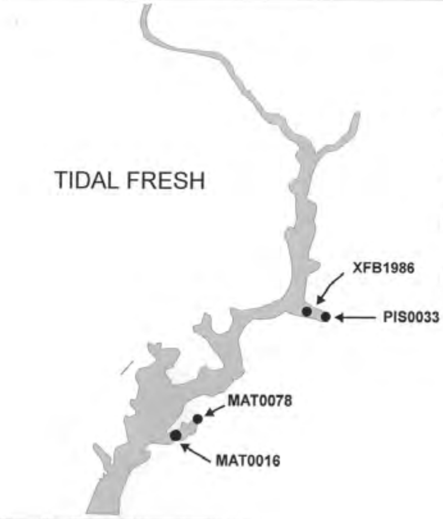


Figure 12c. Potomac tidal creeks nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

Figure 12c.

Potomac River (1986-1998)

Thirteen Year Surface Trends Non-flow Adjusted

	TSS	CHLA
PIS0033	↑ BDL p=0.021	↑ NS p=0.045
XFB1986	NS p=0.116	↑ 121% p=0.000
MAT0078	↑ NS p=0.034	NS p=0.892
MAT0016	NS p=0.942	NS p=0.140

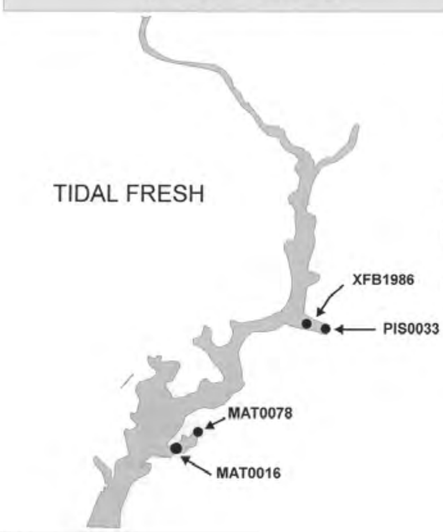


Figure 12d. Potomac tidal creeks nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

Figure 12d.

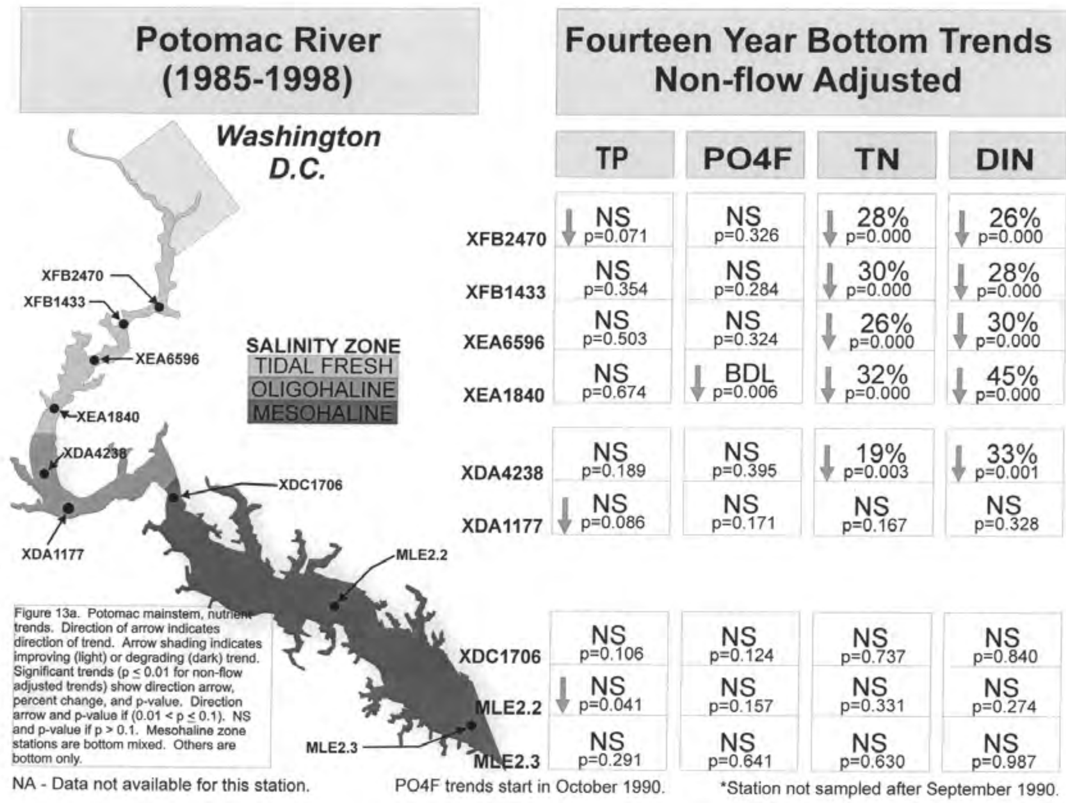


Figure 13a.

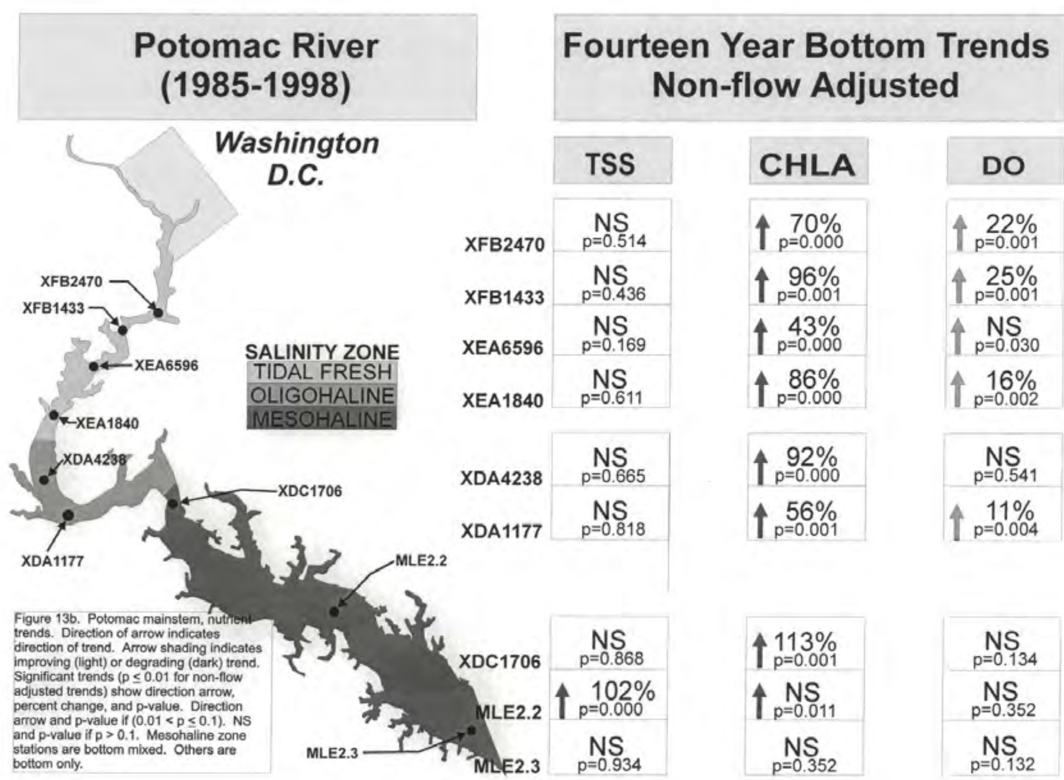


Figure 13b.

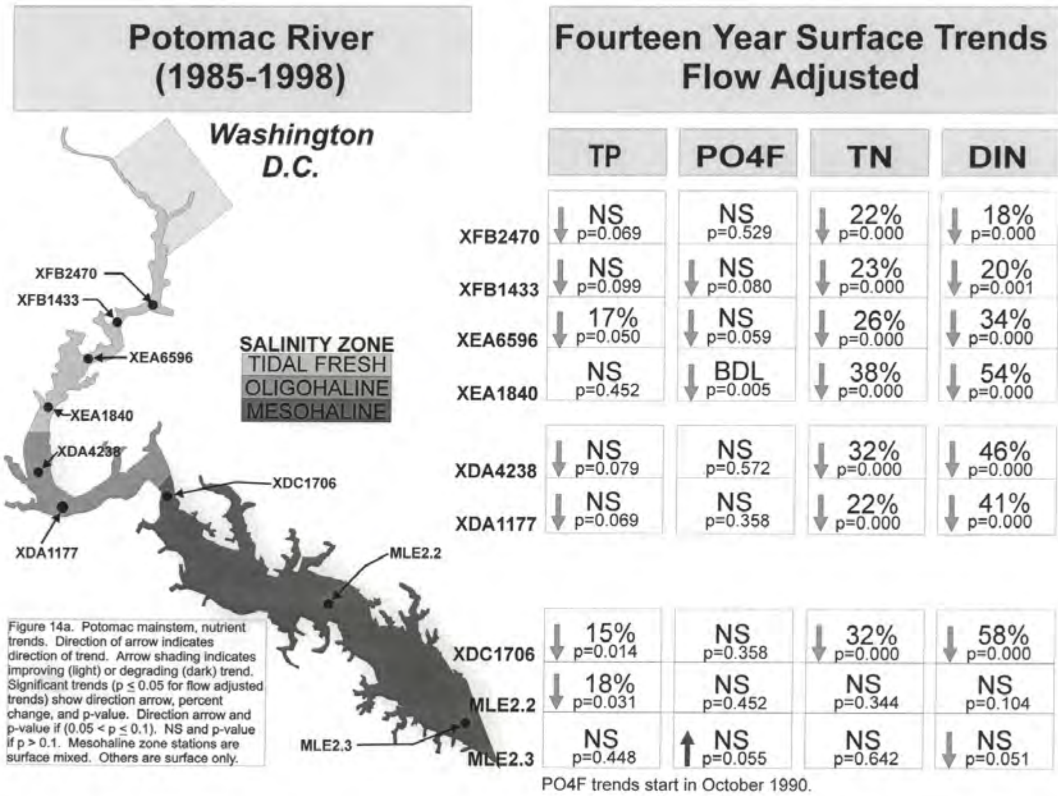


Figure 14a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1. Mesohaline zone stations are surface mixed. Others are surface only.

Figure 14a.

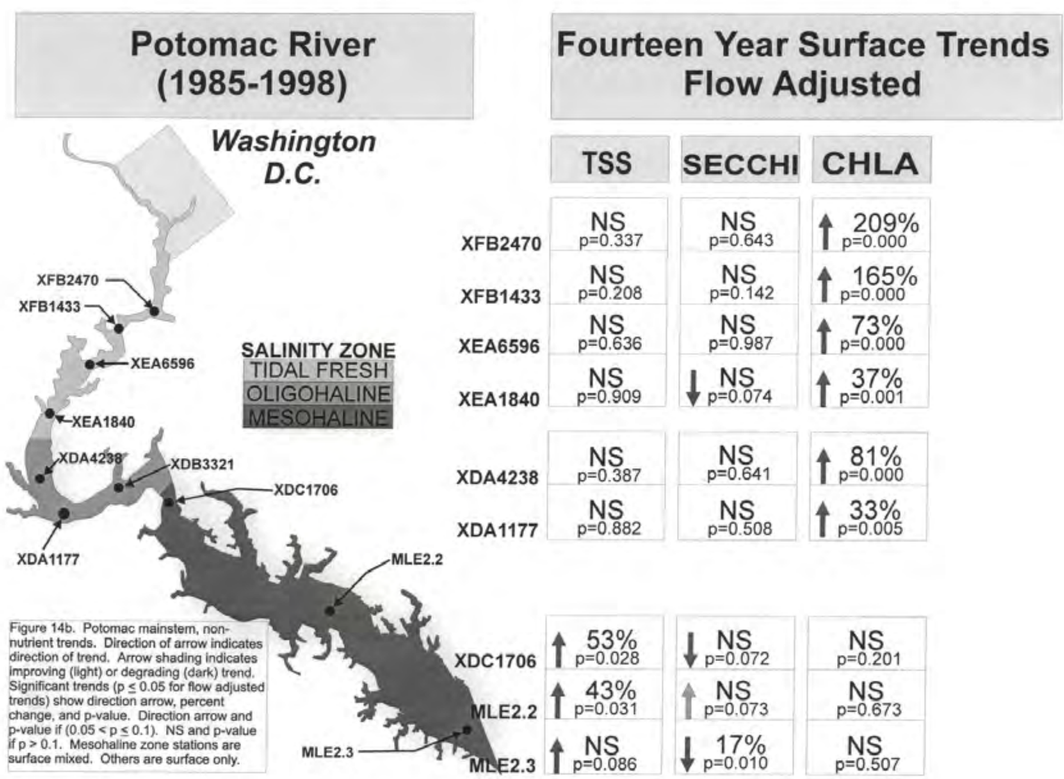


Figure 14b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1. Mesohaline zone stations are surface mixed. Others are surface only.

Figure 14b.

Potomac River (1985-1998)

Fourteen Year Bottom Trends Flow Adjusted

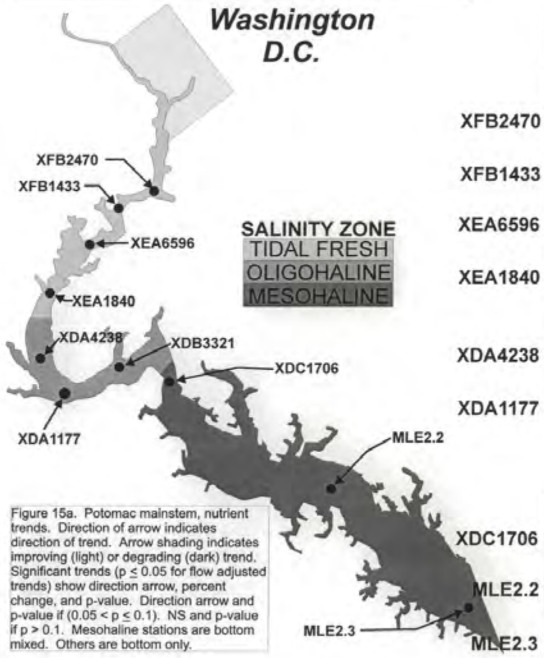


Figure 15a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1. Mesohaline stations are bottom mixed. Others are bottom only.

	TP	PO4F	TN	DIN
XFB2470	↓ 33% p=0.014	NS p=0.124	↓ 25% p=0.000	↓ 18% p=0.001
XFB1433	↓ NS p=0.092	↓ BDL p=0.046	↓ 24% p=0.000	↓ 22% p=0.000
XEA6596	NS p=0.262	NS p=0.107	↓ 23% p=0.000	↓ 36% p=0.000
XEA1840	NS p=0.411	↓ BDL p=0.028	↓ 36% p=0.000	↓ 54% p=0.000
XDA4238	NS p=0.120	NS p=0.437	↓ 27% p=0.000	↓ 44% p=0.000
XDA1177	↓ 22% p=0.018	NS p=0.350	↓ 15% p=0.008	↓ 34% p=0.004
XDC1706	NS p=0.170	NS p=0.138	NS p=0.416	NS p=0.322
MLE2.2	↓ NS p=0.100	NS p=0.371	NS p=0.424	NS p=0.338
MLE2.3	↑ NS p=0.092	↓ BDL p=0.035	NS p=0.740	↓ NS p=0.099

PO4F trends start in October 1990.

Figure 15a.

Potomac River (1985-1998)

Fourteen Year Bottom Trends Flow Adjusted

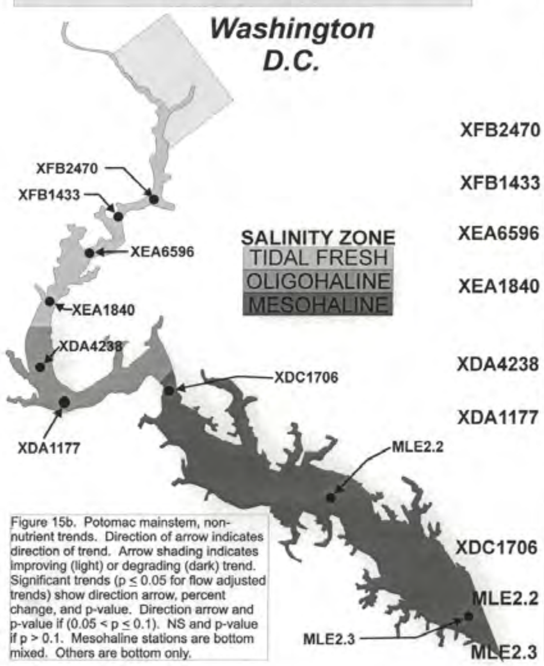


Figure 15b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1. Mesohaline stations are bottom mixed. Others are bottom only.

	TSS	CHLA	DO
XFB2470	↓ 36% p=0.040	↑ 140% p=0.000	↑ NS p=0.100
XFB1433	NS p=0.529	↑ 165% p=0.000	NS p=0.143
XEA6596	NS p=0.960	↑ 82% p=0.000	NS p=0.584
XEA1840	NS p=0.806	↑ 117% p=0.000	NS p=0.393
XDA4238	NS p=0.244	↑ 88% p=0.001	↓ 11% p=0.038
XDA1177	NS p=0.459	↑ 54% p=0.012	NS p=0.208
XDC1706	NS p=0.334	NS p=0.333	NS p=0.626
MLE2.2	↑ 81% p=0.003	NS p=0.749	NS p=0.106
MLE2.3	NS p=0.391	NS p=0.791	↓ NS p=0.080

Figure 15b.

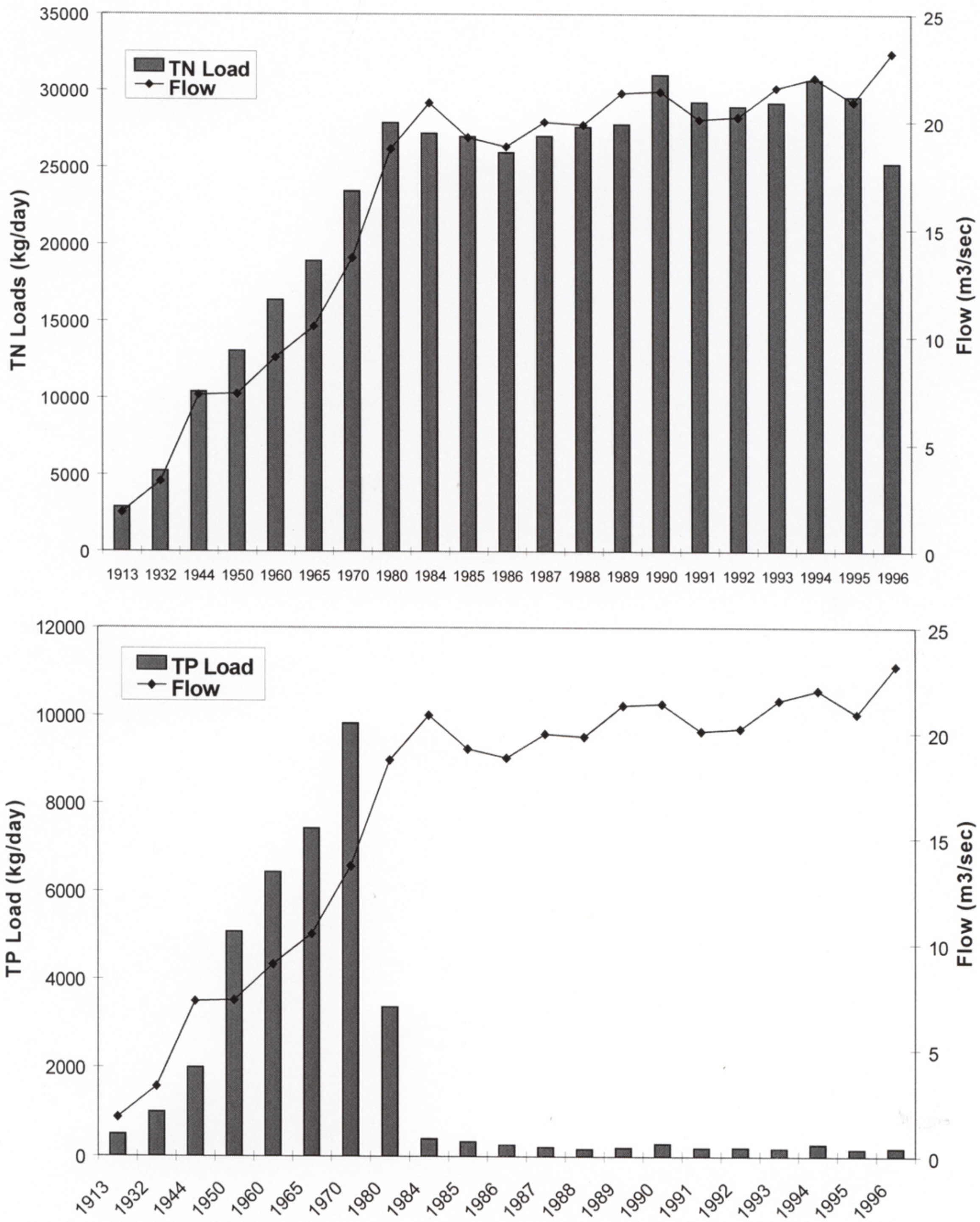


Figure 16. Historical total nitrogen and total phosphorus loads for the major wastewater treatment plants (permitted flow of 0.5 MGD or greater) on the Potomac River for 1913-1996. Data for 1913, 1932, and 1944 from Jaworski, 1990. Data for 1950 through 1980 from EPA Chesapeake Bay Program. Data from 1984 through 1996 from Metropolitan Washington Council of Governments.

WWTPs.CDR

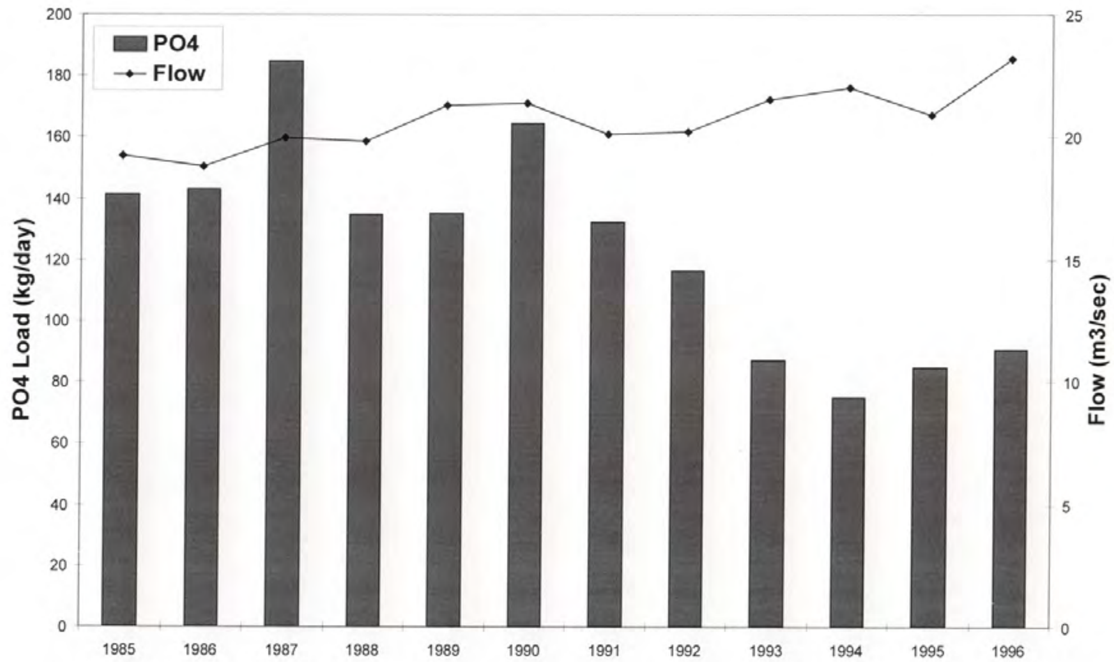


Figure 17. Recent orthophosphate loads for the major wastewater treatment plants (permitted flow of 0.5 MGD or greater) on the Potomac River for 1985 through 1996. Orthophosphate data are not as complete as those for TN and TP. The data record is complete for Blue Plains, Lower Potomac, Mattowoman Creek, Piscataway Creek, and Upper Occoquan. No data are available for Quantico. Data for the remaining plants are intermittent. WWTPs.CDR

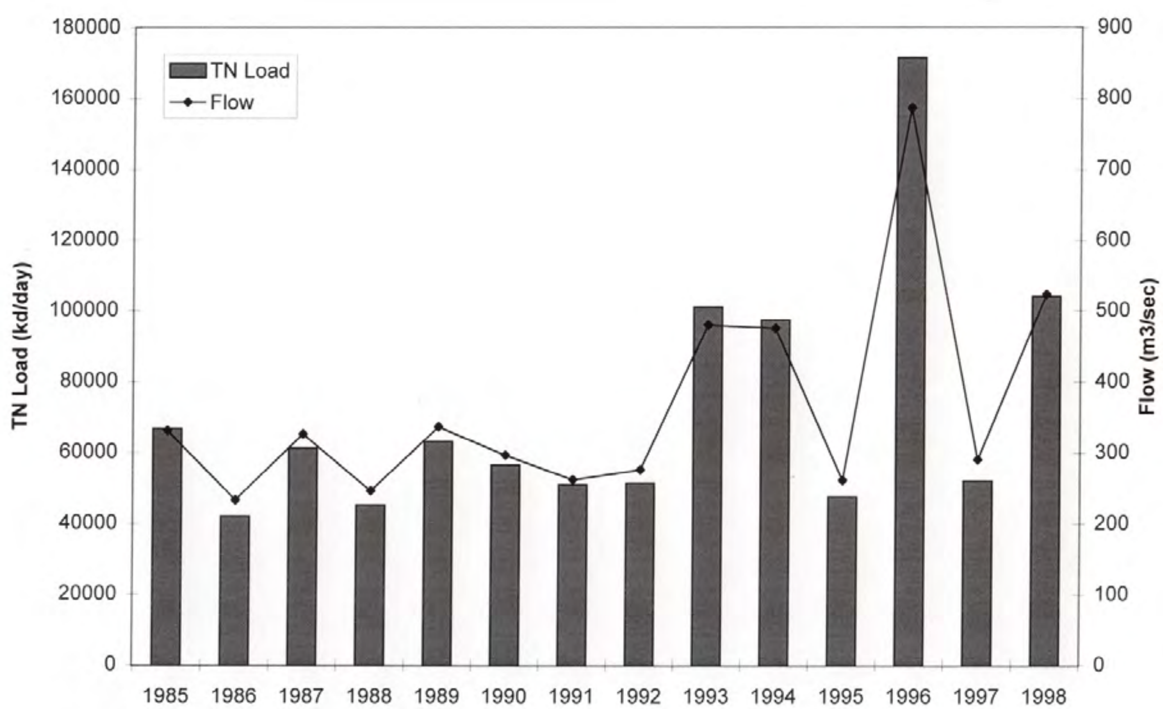


Figure 18. TN load and flow at Chain Bridge.

POTLOAD.XLS 9/7/99

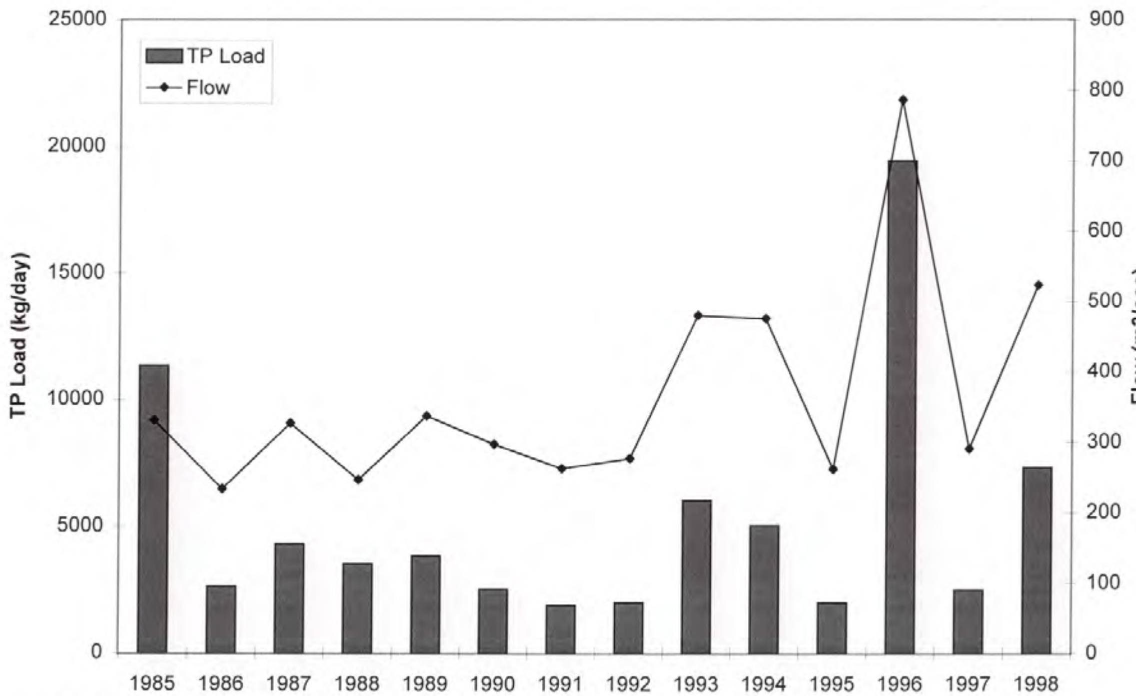


Figure 19a. TP load and flow at Chain Bridge.

POTLOAD.XLS 9/7/99

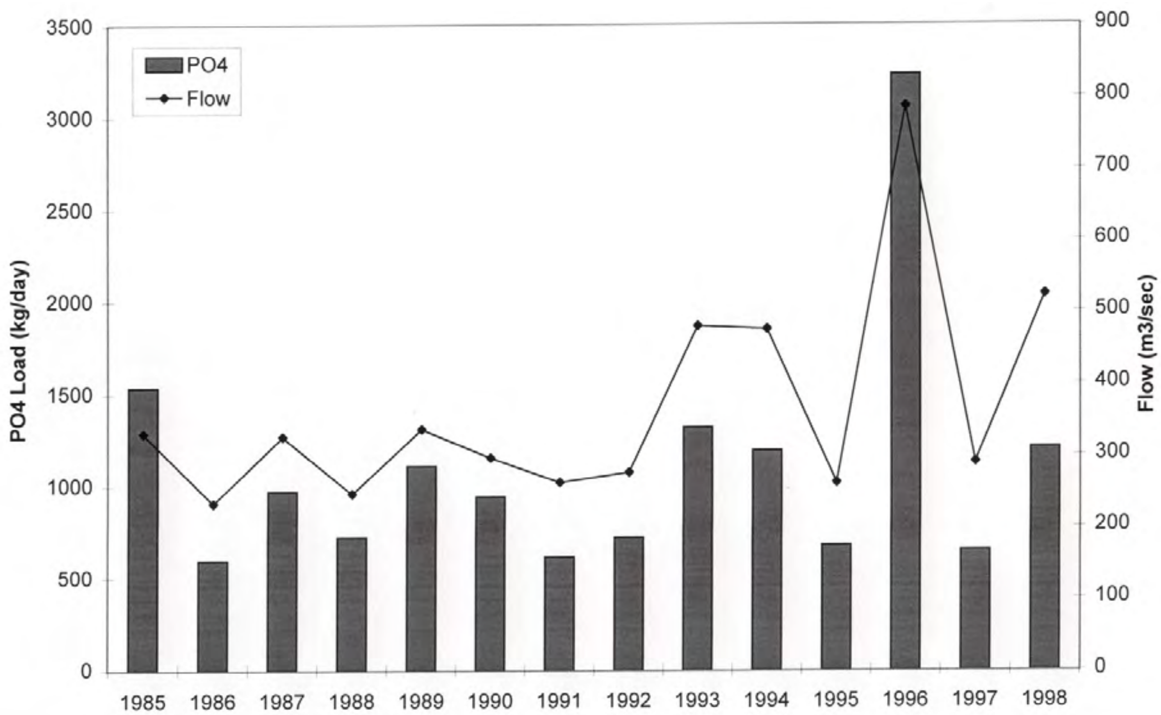


Figure 19b. PO4 load and flow at Chain Bridge.

POTLOAD.XLS 9/7/99

PRIMARY PRODUCERS: THE SUBMERGED AQUATIC VEGETATION AND THE PHYTOPLANKTON (OBJECTIVE 2)

The Potomac River primary producers - the microscopic algae or phytoplankton, and the seagrasses or submerged aquatic vegetation (SAV) - were largely responsible for the U.S. Environmental Protection Agency Chesapeake Bay Program's decree in 1983 that the Chesapeake Bay ecosystem was detrimentally impacted by nutrient loading and enrichment (USEPA, 1982). The dramatic cyanobacteria (blue-green algae) blooms which occurred during the summer months of the late 1970's and early 1980's in a substantial portion of the tidal fresh region of the Potomac Estuary combined with the loss of SAV throughout the estuary and the expanding anoxic zone in the bottom layer of the lower estuary were indicators of the eutrophication which was common in the Potomac River as well as other parts of the Chesapeake Bay ecosystem. The proximity of these indicators of a stressed aquatic system to our nation's capital contributed to the efficacy of the implementation of a nutrient reduction management plan designed to reverse the degree of the eutrophication process which was 'choking' the estuary.

The Recent Past

Pheiffer (1976), Jaworski (1971), Lear and Smith (1976), Seitzinger (1987), and others have described the nuisance bluegreen algal blooms at their peak in the 1970s and early 1980s, and their impacts on the river. A fifty mile reach of the river stretching from Mt. Vernon just below the Washington, D.C. area to Maryland Point where the estuary turns sharply and enters the mesohaline segment regularly exhibited intense blooms of bluegreen algae. *Microcystis aeruginosa* (a.k.a. *Anacystis cyanea*) dominated the bloom which began in the early summer and could last well into fall. In the sharp salinity gradient between Maryland Point and the Route 301 Bridge which crossed the Potomac near Morgantown, MD, the bloom died and this stretch of the river was "often characterized by rotting bluegreen algae, large patches of foam, and in general seemed to be a large heterotrophic 'kettle'" (Lear and Smith 1976). In the lower estuary, "red water" or "mahogany tides" caused by high densities of dinoflagellates were common.

By the 1970s, the once lush Potomac submerged aquatic vegetation (SAV) beds throughout the tributary had disappeared. The combined impact of disease, exotic species invasions and poor water quality were responsible for the losses (see "Anthropogenic Effects"). The advent of a hardy exotic species, *Hydrilla*, in the early 1980s partially restored the SAV beds of the upper and middle Potomac River and helped some native species return.

Factors Affecting Primary Producers

Several factors have a major influence on primary producers in the Potomac Estuary, including freshwater flow rate and the closely related residence time of water, and nutrient and sediment concentrations. It is important to be aware of trends in these factors and the impacts of varied meteorological conditions, population growth and changes in land use when assessing the changes in Potomac primary producers. The first part of this report section will summarize key water quality trends from a historical perspective (early-1900's-1998) and from a post nutrient reduction management strategy perspective (1984-1998). These two distinct time series approaches enables one to perceive how the most recent changes in water quality parameters fit into a larger perspective, thus providing a different scale to assess the progress of man's management actions. The remainder of this section will summarize changes that have occurred in the primary producers and attempt to link these changes to some of the water quality and hydrodynamic conditions in the Potomac River between 1985-1998.

Flow effects

Interpretations of man's progress in reversing the eutrophication process of the estuary is confounded by the annual variability of the freshwater flow entering the estuary. Fresh water entering the estuary from tributary rivers transports significant percentages of the nutrients which are delivered annually to the ecosystem. Approximately 61% of the total nitrogen and 71% of the total phosphorus are delivered to the Bay from diffuse sources by the portion of the river above the fall line (Boynton et al. 1995). Total annual loads of nutrients from riverine sources is directly related to the amount of freshwater being discharged and can vary by factors of 2 and 4 for TN and TP, respectively (Boynton, et al. 1995). A flow normalization technique has been used in the statistical analysis of trends in the data in an attempt to assess the changes in water quality relative to management actions. The bulk of the management actions have targeted point sources of nutrients such as waste water treatment plants and other industries which discharge nutrient-enriched water. Point sources of nutrients contribute approximately 34% of TN and 26% of TP to the Potomac River (Boynton et al., 1995). The factor which confounds management goals in decreasing nutrient loads from point sources is the consistent increase in population in the Chesapeake Bay watershed and the resulting increase in wastewater flow (Figure 3). Population in the Washington, D.C. metropolitan area has increased from 200,000 people in the early-1900's to over 3 million people in the early-1980's (Callender et al., 1984).

Nutrient and suspended sediment effects

Jaworski and Romano (Appendix B) focus on changes in nutrient loadings, nutrient concentrations, suspended sediments, chlorophyll (a measure of phytoplankton biomass) and dissolved oxygen for the 1965-1996 period. During this period, loadings of phosphorus have decreased by 56% while TN loadings have increased by approximately 95%. In association with these changes in the loads of nitrogen and phosphorus, concentrations of TP have decreased by an average of 70% throughout the river, while TN concentrations increased by 42% when averaged at four stations in the tidal fresh and transition zones of the river. The concentrations of chlorophyll have decreased by an average of 60% at five stations throughout the river's reach between 1965-1996. Light penetration (secchi depth) has not changed significantly in the tidal fresh river or transition zone since 1965 but has decreased near the mouth of the Potomac Estuary. Summer bottom dissolved oxygen concentrations have improved in the tidal fresh zone while summer bottom concentrations were consistently hypoxic-anoxic in the channel of the lower portion of the river. Results for this extended period of time indicate improving conditions for phosphorus and chlorophyll, generally deteriorating conditions for nitrogen and light penetration and mixed signals for bottom dissolved oxygen concentrations.

Romano et al. (Appendix A) focuses on the period following implementation of nutrient reduction strategies, 1985-1998, and shows considerably different trends in water quality parameters than the longer period. Concentrations of TP have increased at stations in each of the major salinity zones of the river. TN concentrations have decreased in the tidal fresh and upper oligohaline portions of the river and increased in the lower oligohaline and river mouth. Chlorophyll concentrations have increased throughout the water column in the tidal fresh and oligohaline stations and in the bottom layer of the mesohaline stations. Light penetration (secchi depth) has decreased at stations throughout the river. Bottom dissolved oxygen concentrations have increased significantly in the upper tidal fresh and mesohaline areas.

Nutrient and sediment loads decrease water clarity which has an adverse affect on SAV growth. Light is the major factor which impacts the distribution and growth of SAV in the Potomac River (Carter, et al., 1994). The amount of light reaching the leaves of the SAV is dependent initially upon the available sunshine at the surface of the water and the amount of scattering in the water column which results from the particulate load. This particulate load in the water column consists of inorganic material such as silt and clay and organic particles consisting largely of phytoplankton and marsh plant detrital material. Calculations made by Jones (pers. comm.) in the tidal fresh portion of the Potomac River during the summer indicate that ~ 30% of the total suspended particulate load is composed of living phytoplankton cells. This calculation was based on weight which could conceivably be an underestimation of the contribution of phytoplankton to the total load of suspended particles. The other factor which reduces light availability for SAV growth is the epiphytic growth of bacteria and phytoplankton on the leaves of the plants. The nutrient load to the watershed thereby impacts SAV growth with a double-edged sword, the scattering and reduction of light by the phytoplankton floating in the water column and by the growth of phytoplankton directly upon the leaf surface. Between 1983-1996, analysis indicates that the ratio of total suspended solids (TSS) to chlorophyll (a surrogate measure of the organic fraction of the total load) has increased at all stations in the river except near the mouth. This increase in the relative proportion of sediment particles to phytoplankton may be the result of high freshwater flow years in 1993, 1994 and 1996 transporting large sediment loads to the river. The cumulative area of SAV coverage in the Potomac River in 1996, ~2181 hectares, was comparable to the coverage in 1987, ~2106 hectares, but the distribution within the river was very different. In 1987, the upper tidal fresh area had the greatest coverage (~1465 ha) with a smaller area in the oligohaline (~485 ha) and minimal coverage in the lower tidal fresh (~113 ha) and mesohaline (~43 ha) areas. In 1996, the oligohaline was characterized with the greatest SAV coverage (~1307 ha) followed by the lower tidal fresh (~470 ha), mesohaline (~402 ha) and upper tidal fresh regions.

Tidal Fresh River (Upper Estuary)

Phytoplankton biomass in the tidal fresh reach of the Potomac River is characterized by a steady increase from the winter months to a peak during the middle of the summer and a subsequent steady decline thereafter (Figure 20). Peak chlorophyll *a* concentrations occur in early August when the mean concentration for that time during the study period was 62.7 µg/l (3.6 - 106.7 µg/l). This concentration is twice as high as that for any month at the oligohaline or mesohaline plankton stations. The majority of the phytoplankton biomass during the summer months is composed of cyanobacteria (mean= 40%; range = 29.9-54.3%) (Figure 21). The summer bloom of cyanobacteria is composed of numerous colonial and filamentous taxa and historically has been dominated by the colonial form, *Microcystis aeruginosa* (Jaworski and Hetling, 1970; Thomann et al., 1985). One of the peak cyanobacteria blooms took place in 1983 when a thirty mile stretch of the tidal fresh reach was impacted by a bloom which was dominated by *M.*

aeruginosa and which was characterized at its peak in August by chlorophyll concentrations > 200 µg/l (Thomann et al., 1985). During only one year of the current study period, 1985, was there a significant bloom of *M. aeruginosa*, yet the temporal and spatial scales of this event were considerably less than the bloom which took place in 1983 (Figure 22). During 1988 and 1998, there were periods of high densities of *M. aeruginosa* at station TF2.3 off of Indian Head but these concentrations were considerably less than those which persisted for months during 1983. In addition, chlorophyll concentrations exceeded 200 µg/l only once during the study period, in July, 1988. In spite of the decline of *M.aeruginosa* densities from the tidal fresh portion of the river, there continue to be significant blooms of other taxa of cyanobacteria. Specifically, other forms of Microcystis, Agmenellum, Oscillatoria, Raphidiopsis , Anabaena and a thin unidentified filament occur in great densities during the summer (Figure 23). Spatially, this bloom typically begins in the vicinity of Mattawoman Creek and peaks off of Indian Head up to Gunston Cove (the upriver extent of the sampling of this program), but this is subject to variability depending on the freshwater flow which occurs during the summer months (Figure 24). Phytoplankton biomass also tends to be higher in the coves (Gunston Cove and Mattawoman Creek) and partially enclosed bays (Occoquan Bay) of the upper portion of the river (Figure 25). With the exception of 1996, the overall bloom of cyanobacteria has increased since August, 1992 and indicates a significantly increasing trend during the overall study period (Figure 26). Partially in response to this increase in summer cyanobacteria biomass, trends in chlorophyll and primary productivity have increased between 1985-1998 at station TF2.3 (Figures 27,28,29&30).

Following the 1983 cyanobacteria bloom, data analysis and modeling suggested that the initial cause of the bloom was a combination of nutrient enrichment and meteorological and hydrological factors (high sunshine, low winds, high temperature and water column stability) (Thomann et al., 1985; Jones, 1999). This study further reveals that the bloom was intensified and persisted as a result of the release of phosphorus from the sediment, driven by elevated pH levels caused by the initial pulse of the bloom (Seitzinger, 1987). The current study indicates that residence time (an inverse relationship with flow) with a seven day lag from the gaging station shows a fairly strong positive relationship with cyanobacteria biomass during July-August ($r^2 = 0.48$; Figure 31). Orthophosphate concentrations also are positively coupled with cyanobacteria biomass during the summer months, July-September ($r^2 = 0.28$; Figure 32). Conversely, nitrogen concentrations as dissolved inorganic nitrogen, have an inverse relationship with cyanobacteria carbon ($r^2 = 0.20$) and with total phytoplankton biomass ($r^2 = 0.48$) during July-September (Figures 33&34). These factors indicate that freshwater flow and more specifically residence time (the amount of time that a parcel of water remains in the tidal fresh zone) and phosphorus are very important factors in perpetuating the cyanobacteria bloom in the upper Potomac River. The initiation of the bloom during the early summer may be driven by nitrogen concentrations delivered to the system during the high flows of spring. This point was suggested by Thomann et al. as TN concentrations were approximately 2 mg/l throughout the upper river during the early summer of 1983. Nutrient bioassay studies during the period 1990-1997 at a variety of sampling sites throughout the Maryland portion of the Chesapeake Bay have provided indicators of nutrient and light limitation for phytoplankton growth (Fisher and Gustafson, 1998). When applied to the tidal fresh Potomac River nutrient data, these indicators reveal that nitrogen is rarely in concentrations which are close to being limiting to phytoplankton growth and that phosphorus was in concentrations that were >50 % probability of being limiting in the summer only in 1991 and 1992 (Figures 35&36). With these facts in mind, it is hypothesized that nutrient conditions remain satisfactory for the initiation of the cyanobacteria bloom and that should meteorological and hydrological conditions be favorable during a given summer, the bloom will persist. The one condition that is absent from the equation set forth in the Potomac Eutrophication Model is the pH levels which rarely have exceeded the threshold value of 9.0 during the study period. It is also questionable that a pool of phosphorus remains in the sediments of the upper Potomac River in large enough concentrations to fuel the proliferation of a bloom comparable in size to that of 1983. I make this statement based upon the water column concentrations of phosphorus which have declined significantly between 1985-1998 (Figure 37).

In the upper tidal fresh portion of the river SAV coverage peaked in 1985-1988 (~1500 ha), declined in 1989 and has remained relatively stable through 1996 (~500 ha). In the lower tidal fresh reach, SAV coverage was virtually non-existent in 1983-1985, possibly as a result of the cyanobacteria blooms in that portion of the river, returned slightly in 1986-1988, increased dramatically in 1989, peaked in 1991 (~1100 ha), and declined steadily in 1994 and 1995 (low of ~250 ha). The increase of SAV in 1989-1992 coincided with a decline in phytoplankton biomass, specifically the cyanobacteria bloom, in that segment (Figure 23). SAV coverage was negatively correlated with chlorophyll especially during the summer and during low flow years, which emphasizes the relationship between flow or residence time and the development of the cyanobacteria bloom. TSS (1995-1996) has replaced chlorophyll (1984-1985) as the major SAV habitat criterion not met in order to sustain growth.

Transition Zone (Middle Estuary)

The oligohaline or transition zone of the Potomac River is not adaquately characterized by this study since there is only one plankton sampling station in the zone and the dynamic chlorophyll maximum region of the river which is located in the lower oligohaline - upper mesohaline area is not sampled except for chlorophyll. Nonetheless, trends in chlorophyll concentrations in this chlorophyll maximum region of the river show a significant increase in the bottom layer of the upper mesohaline region during 1985-1998 (Romano et al., Appendix A). The plankton station in this portion of the river, RET2.2, is located off of Maryland Point, in the upper portion of this transition zone. This station is characterized by relatively low phytoplankton biomass with a summer peak composed of diatoms, dinoflagellates and cyanobacteria transported from the bloom upriver (Figures 20&38). This station is located within the turbidity maximum portion of the river and is therefore subject to light limitation as evidenced by the integrated carbon fixation values relative to those values at the other two plankton stations in the river (Figure 39). Seasonal Kendall trend analysis indicates significantly increasing chlorophyll concentrations throughout the water column and increases in cyanobacteria in the surface mixed layer (Figures 26,27&40). The increase in chlorophyll levels is largely driven by summer peaks which are partially a result of high flows lowering the salinity in this area. This lowered salinity and/or the transport of particles from upriver enables the cyanobacteria bloom to expand to this region of the river.

As one moves downriver, nitrogen concentrations decrease but not to the point that cause limitation in phytoplankton growth (Figure 41). Phosphorus limitation is a rare event in this portion of the estuary and the few times during which it is likely to occur is during the winter months (Figure 42). The lack of nutrient reduction in this area combined with the relatively low chlorophyll and primary production levels indirectly indicates that light is the limiting resource to phytoplankton growth. The hydrodynamics of a partially mixed estuary give rise to the turbidity maximum region and the idea that reducing sediment loads to the river will enable enhanced phytoplankton growth is probably not applicable to this region.

SAV coverage in the oligohaline was relatively low (~400 ha) during 1984-1988, increased dramatically in 1989 (~ 1100 ha), peaked in 1992 (~1400 ha) and declined slightly in 1995-1996 (~1000 ha). TSS , apart from chlorophyll, did not meet the SAV habitat criteria for suitable growth conditions most often in the oligohaline. This is most likely due to the fact that the turbidity maximum zone of this partially mixed estuary is within this segment and possibly due in part to resuspension of bottom sediments since the axis of the river is largely north-south in this area and subject to the prevailing winds. This section of the river showed the most lasting improvements in SAV coverage of the entire river.

Mesohaline (Lower Estuary)

The lower portion of the estuary, the mesohaline zone, is impacted by loads transported downriver as well as down the mainstem of the Chesapeake Bay and is also subject to influences from the marine environment as a result of bottom water being transported upbay. The sediment-water interface becomes an important boundary in the mesohaline as both a sink for particulate forms of nutrients and the source for re-worked dissolved nutrients. With reduced light limitation in this portion of the river, inorganic nutrients are rapidly converted to particulate and organic matter. This organic matter serves as the substrate for bacterial metabolism which during the summer drives the hypoxic conditions which are common to the deeper areas of this region.

The annual phytoplankton cycle in the mesohaline portion of the Potomac River is dominated by a bloom of diatoms during the early winter (December), another more substantial bloom of diatoms during the early spring (March-April), a bloom of the dinoflagellate, *Prorocentrum minimum* transported upbay to the area in the bottom waters (Tyler and Seliger, 1978) during May and a summer bloom of dinoflagellates (Figures 20&43). When integrated vertically over the entire water column and considering the volume of water in this reach of the river, the mesohaline is the most productive section of the river. Trend analysis reveals that chlorophyll concentrations are increasing in the bottom layer below the pycnocline and that cyanobacteria biomass is increasing significantly in the surface mixed layer (Figures 26,27,44&45). This increase in cyanobacteria biomass is a result of the proliferation of a filamentous form of cyanobacteria which began during the summer, 1994 and peaked during summer, 1997 and 1998 (Figure 46). This taxon is apparently a brackish-marine form that can tolerate moderate salinity concentrations. It has reached peak densities during August-September and in 1997 and 1998 comprised 34% and 68% of the total phytoplankton biomass, respectively during early August.

Nitrogen is the controlling nutrient in regards to phytoplankton growth in the mesohaline portion of the Potomac River (Appendix C). During the study period, it frequently limited algal growth during the summer-fall (Figure 47). This is an important fact for managers to realize when they are deciding which nutrients to attempt to reduce at point sources in the upper river. Nitrogen may not play a controlling role to the freshwater phytoplankton populations in the vicinity of the point sources, but the transport of this nutrient downriver has a significant impact on controlling the phytoplankton populations in the mesohaline. The spring diatom bloom which is largely responsible for bottom dissolved oxygen conditions in the mesohaline shows an inverse relationship with DIN ($r^2 = 0.15$) indicating the transformation of the dissolved nutrient into a particulate, organic form (Figure 48). This particulate nitrogen sinks to the bottom and is regenerated to the water column during the summer months, fueling the peak in primary production which occurs during the summer in the mesohaline. The reduction of nitrogen in the watershed plays a very important role in improving water quality by directly limiting the spring diatom bloom and indirectly controlling the subsequent summer peak in primary production. The other nutrient which plays a significant role in the dynamics of the spring diatom bloom is silicon. During high moderate-high flow years there is sufficient silicon transported downriver, but during low flow springs (1987) silicate concentrations are limiting (Conley and Malone, 1993). Phosphorus is very weakly related to phytoplankton growth in the mesohaline ($r^2 < 0.1$ for spring and summer chlorophyll concentrations) and never produces a probability > 50% for phytoplankton limitation using Fisher and Gustafson's indicator model (Figure 49) (Fisher and Gustafson, 1998).

Coverage of SAV in the mesohaline remained sparse during the study period, 1983-1996, but did show the greatest improvement from 1993-1996 to a peak value of ~ 400 hectares in 1996. Light is not limiting to SAV growth in this portion of the river. Chlorophyll and TSS concentrations, as well as secchi measurements, met the SAV habitat criteria for the study period. DIN concentrations often failed the habitat criteria set forth for SAV from the standpoint of potentially limiting the growth of the seagrasses. The available nitrogen is utilized by the phytoplankton which in turn has an adverse effect upon the light availability to the SAV. It is also possible that there has been an inadequate supply of propagules to allow for expansion of coverage by the SAV in the mesohaline portion of the river.

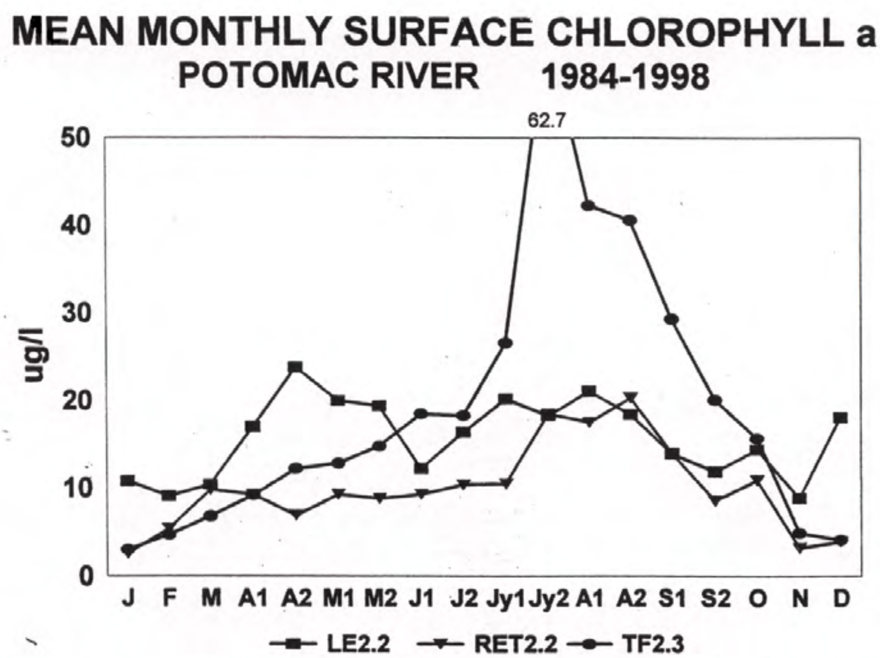


Figure 20: Mean monthly surface chlorophyll a concentrations at 3 stations in the Potomac River, 1984-1998.

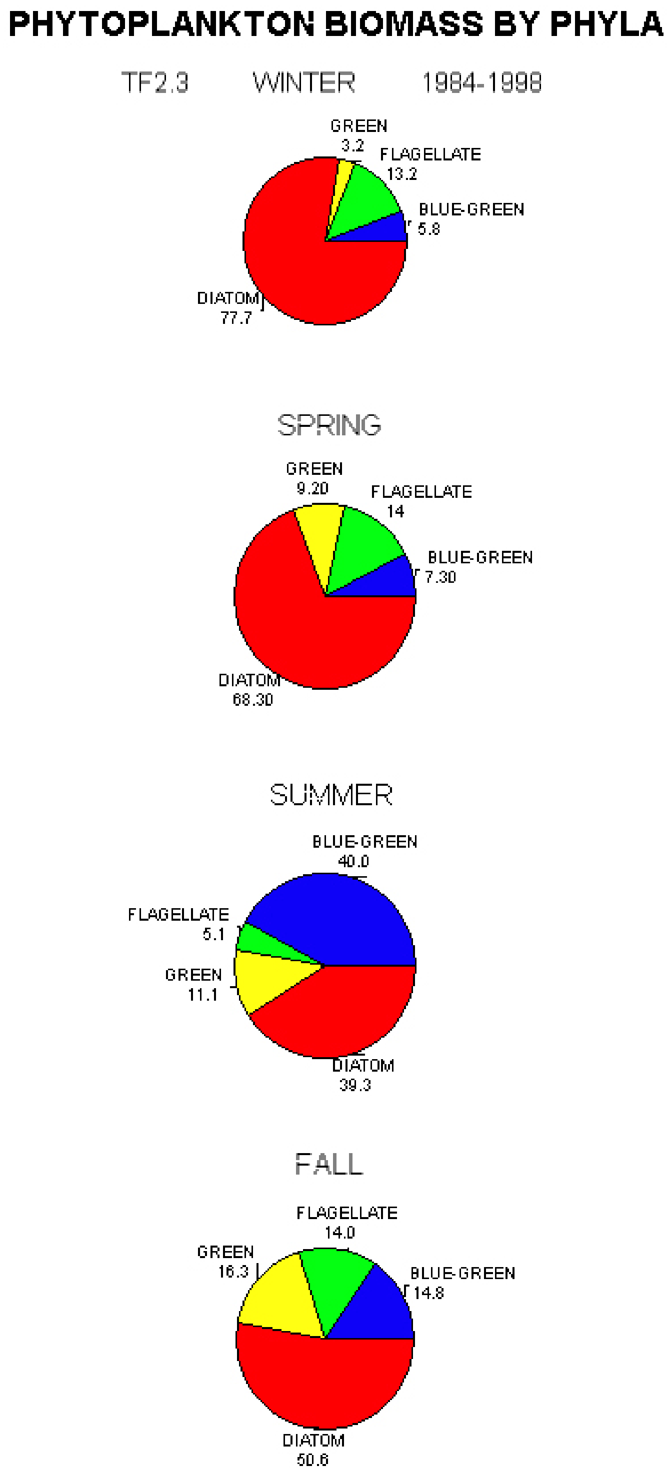


Figure 21. Mean seasonal phytoplankton biomass by phylum at station TF2.3, 1984-1998.

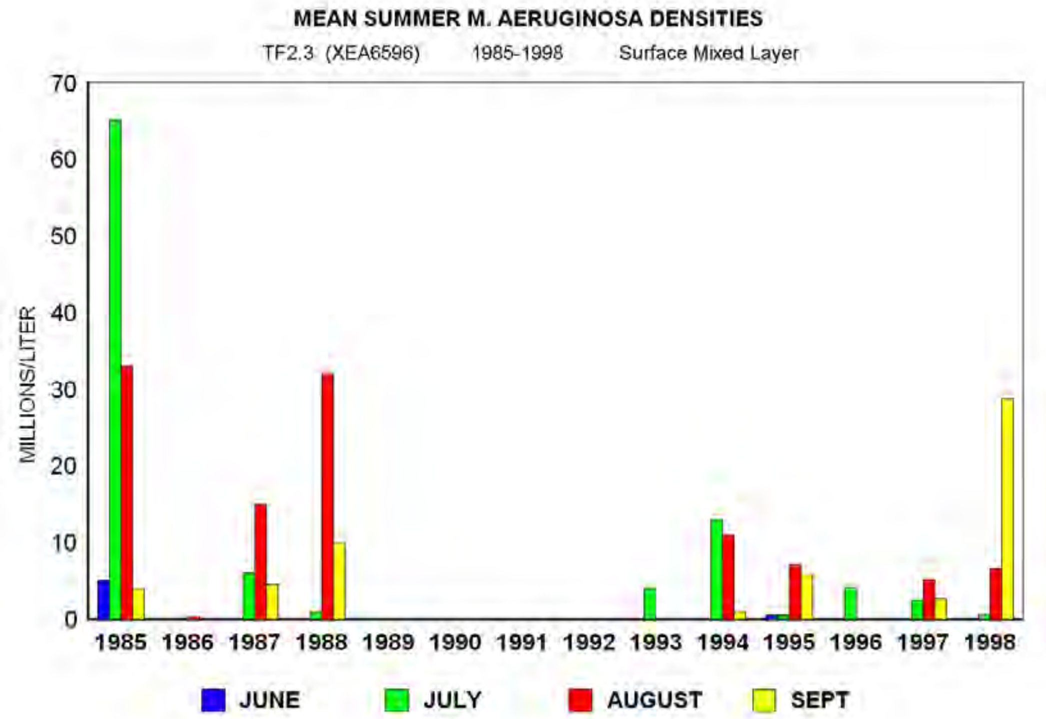


Figure 22. Mean monthly summer densities of *Microcystis aeruginosa* in the surface mixed layer at station TF2.3, 1985-1998.

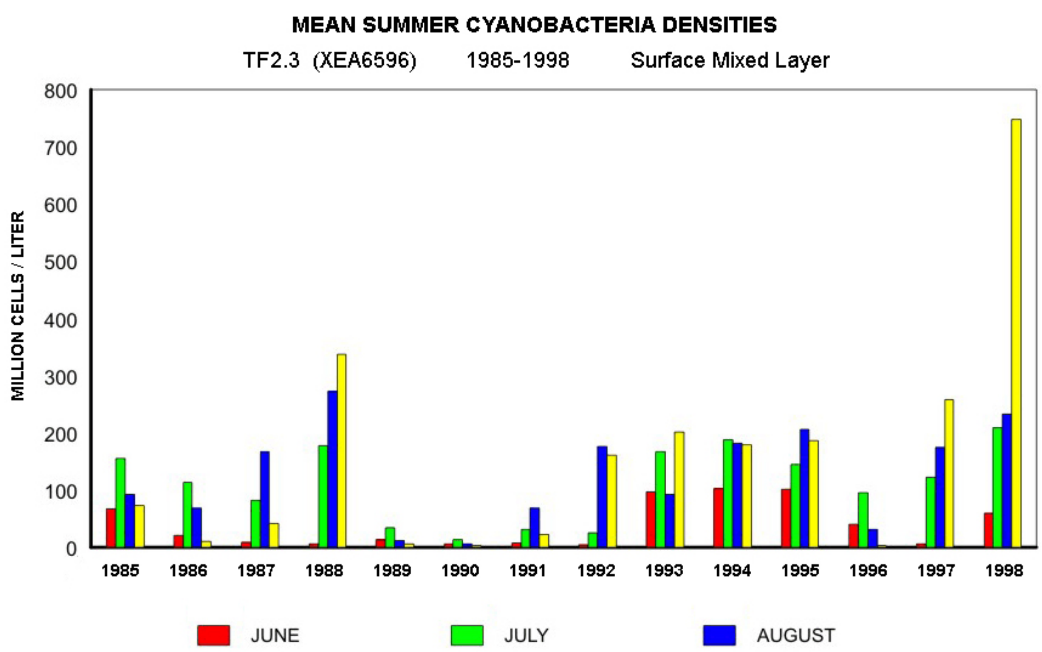


Figure 13. Mean monthly summer densities of cyanobacteria (blue-green algae) at station TF2.3, 1985-1998.

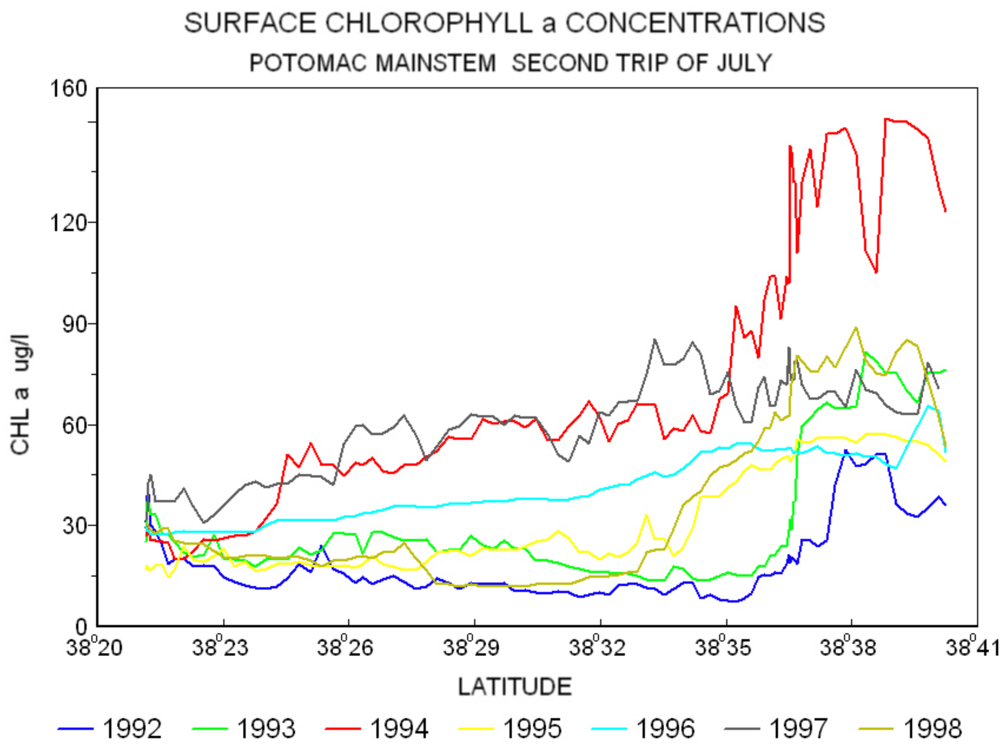


Figure 24. Surface chlorophyll a measured during a longitudinal transect between Maryland Point (left side) and Gunston Cove (right side), 1992-1998.

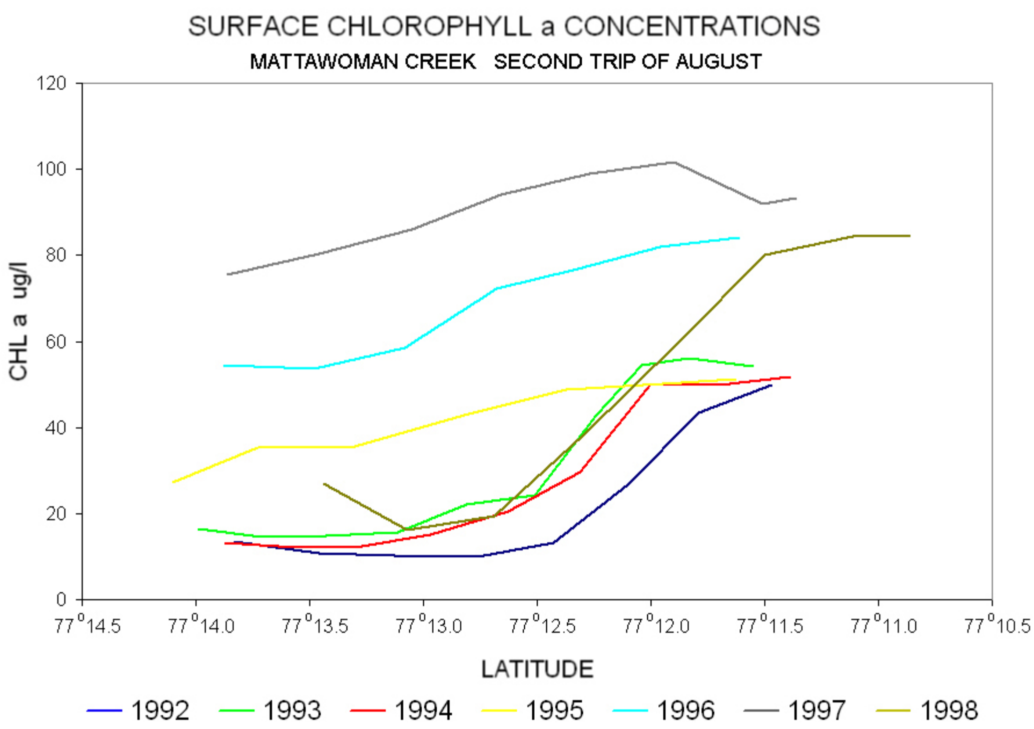
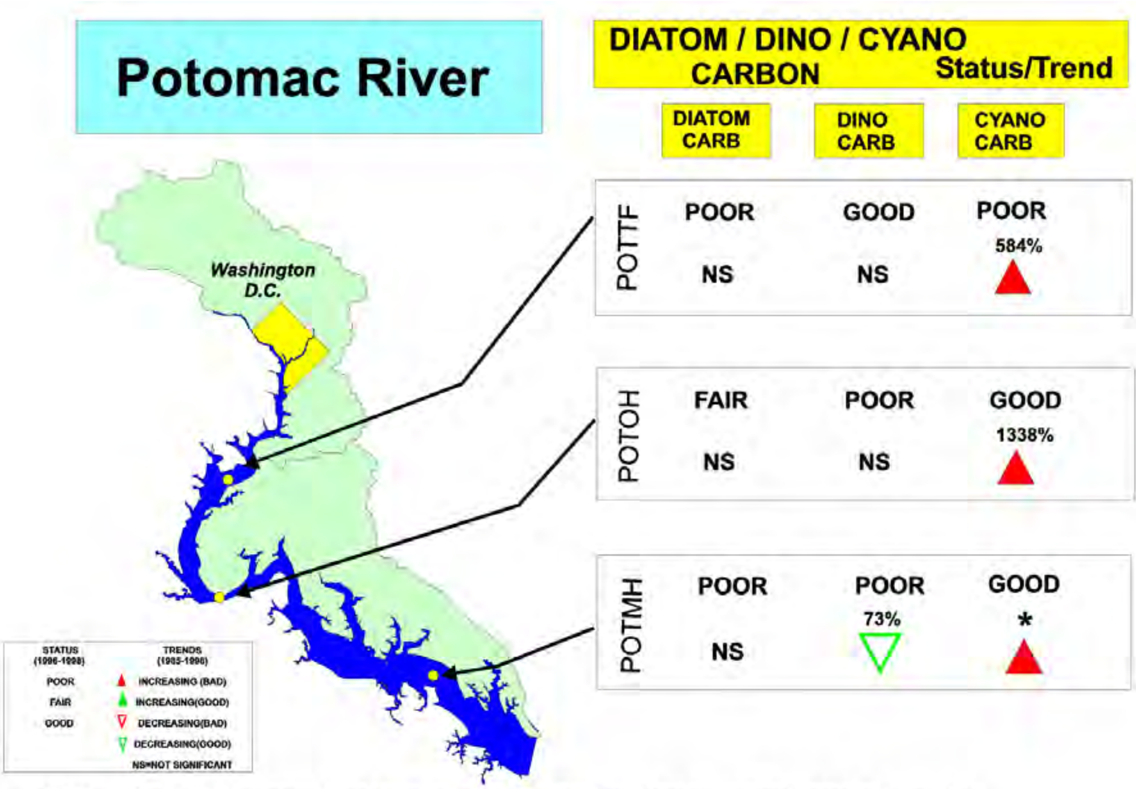


Figure 25. Surface chlorophyll a along a transect from mid-channel Potomac River (left side) into Mattawoman Creek (right side), 1992 - 1998.



NOTE: * = Ecologically Insignificant Value - extremely low densities during the early part of the study result in extremely high percent change values.

Figure 26. Trends, status, and percent change for total diatom, dinoflagellate, and cyanophyte carbon in the Potomac River.

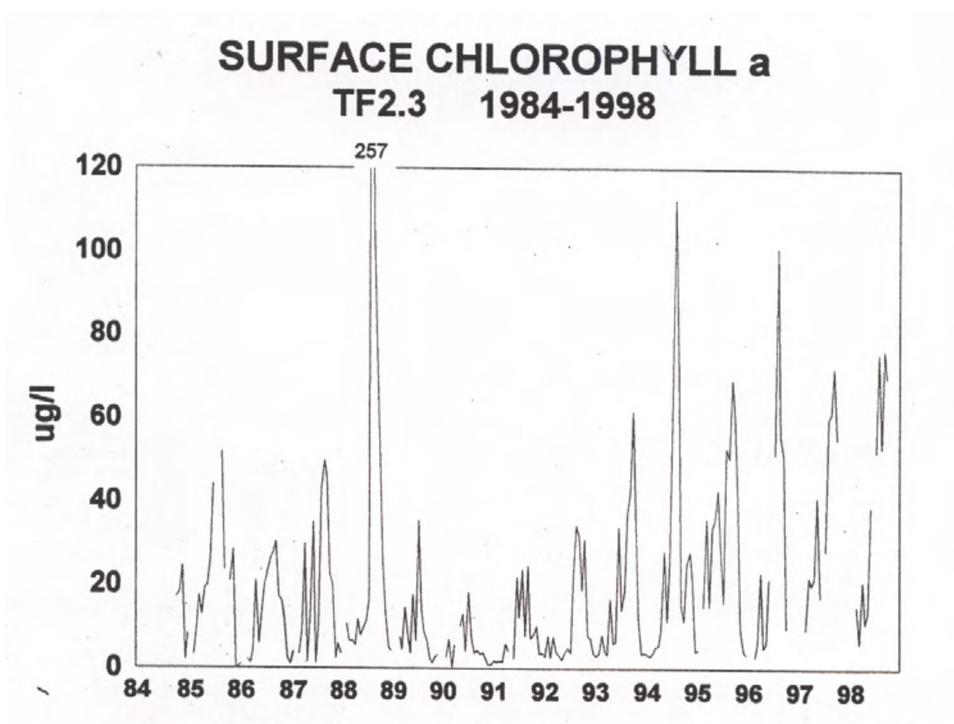


Figure 27: Surface chlorophyll a concentrations at station TF2.3 - 1984-1998.

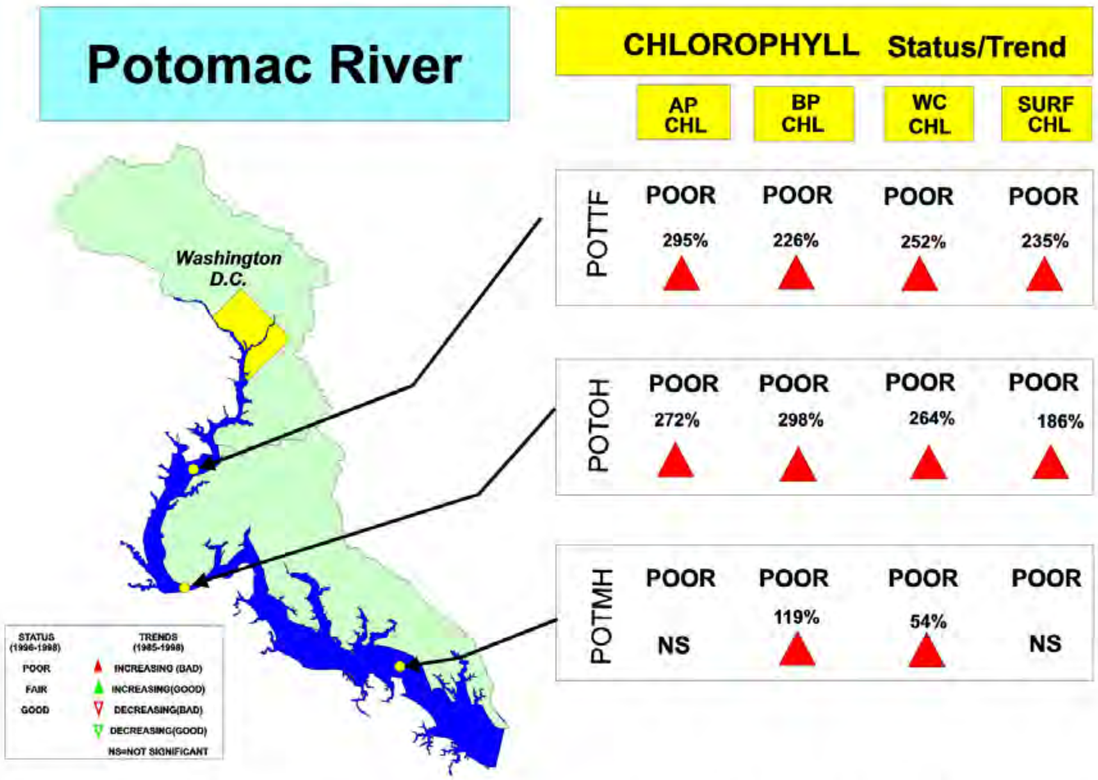


Figure 28. Trends, status, and percent change for chlorophyll in the Potomac River. AP, above pycnocline; BP, below pycnocline; WC, entire water column; and SURF, 0.5 meters.

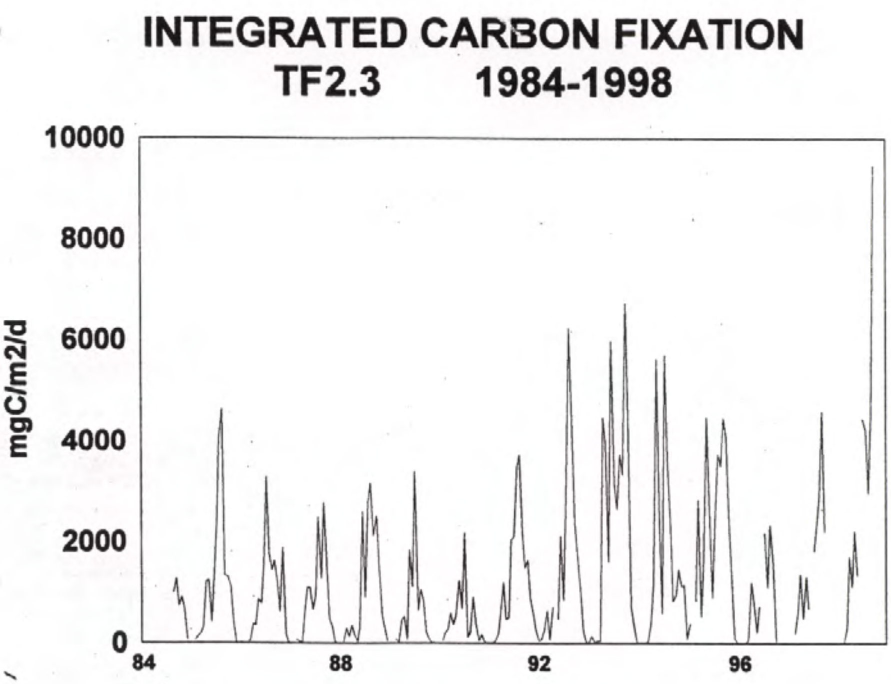


Figure 29: Integrated carbon fixation at station TF2.3 - 1984-1998.

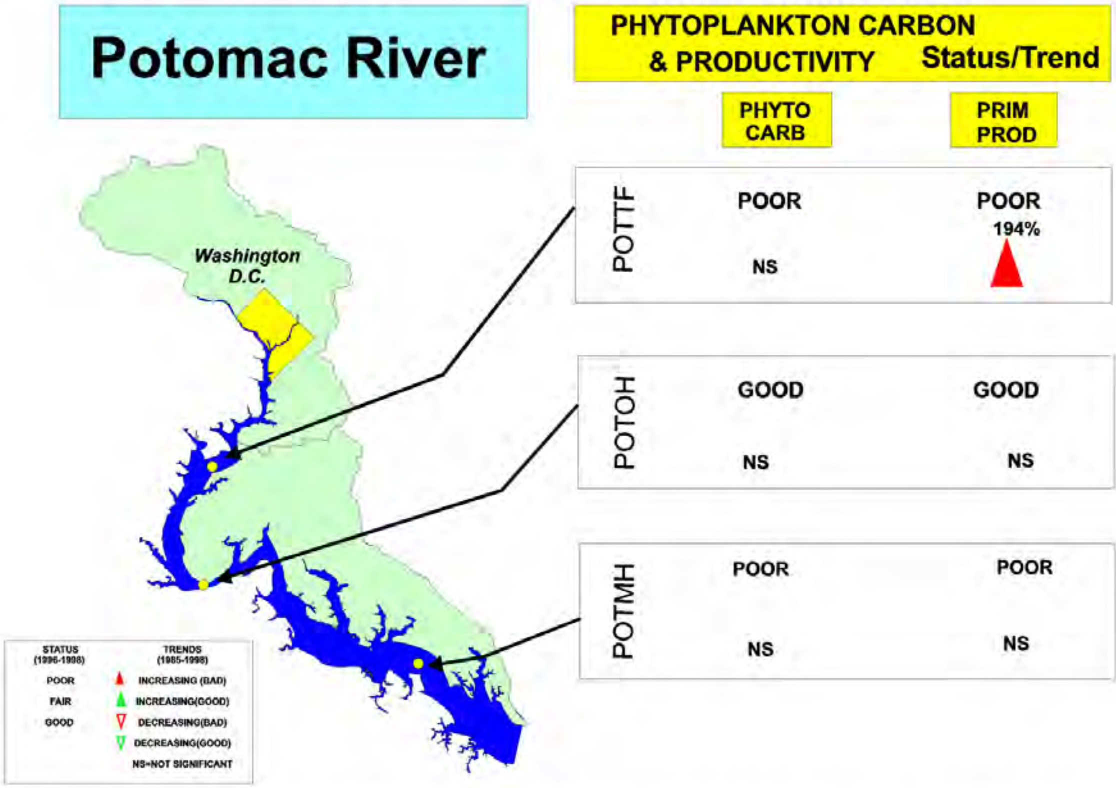


Figure 30. Trends, status, and percent change for phytoplankton carbon and primary productivity in the Potomac River.

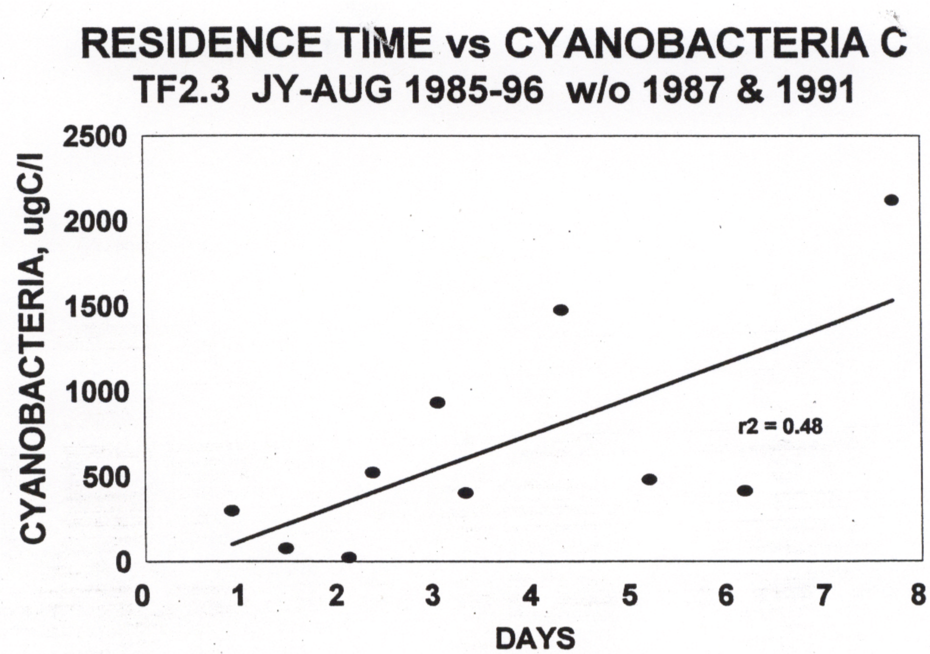


Figure 31: Relationship of residence time to cyanobacteria biomass at station TF2.3 during July-August, 1985-1996. Note: 1987 & 1991 not included due to different gaging station used for flow measurements.

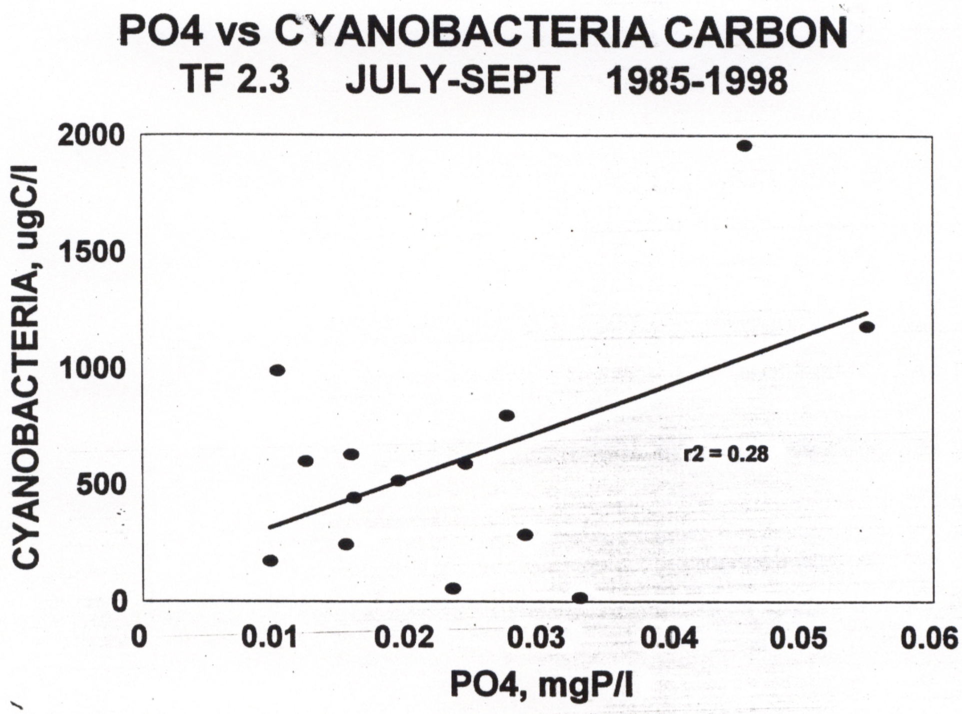


Figure 32: Relationship between PO4 concentration and cyanobacteria biomass at station TF2.3 during July-September, 1985-1998.

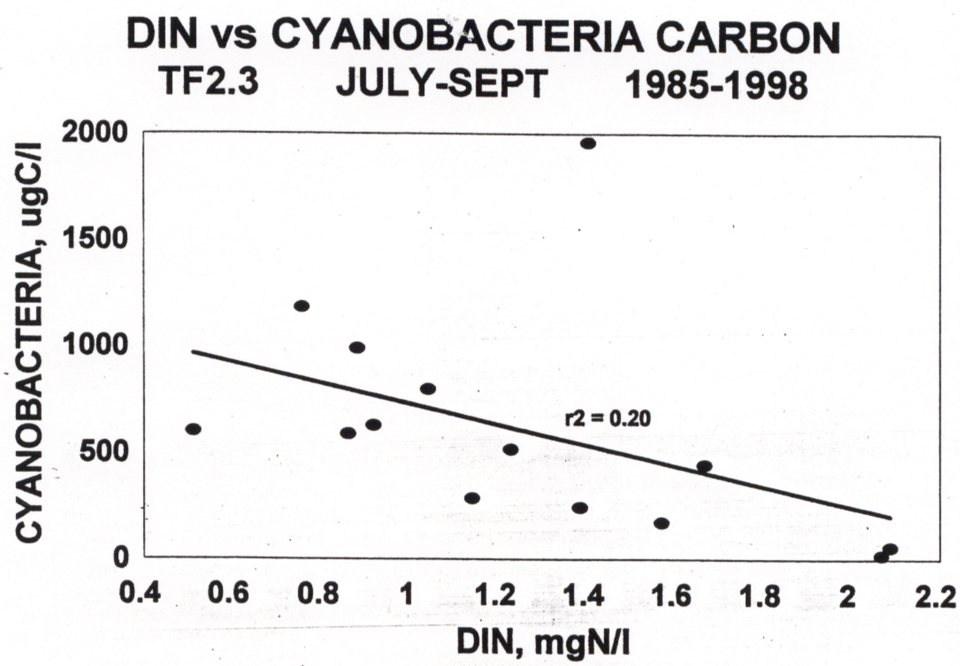


Figure 33: Relationship between DIN concentration and cyanobacteria biomass at station TF2.3 during July-September, 1985-1998.

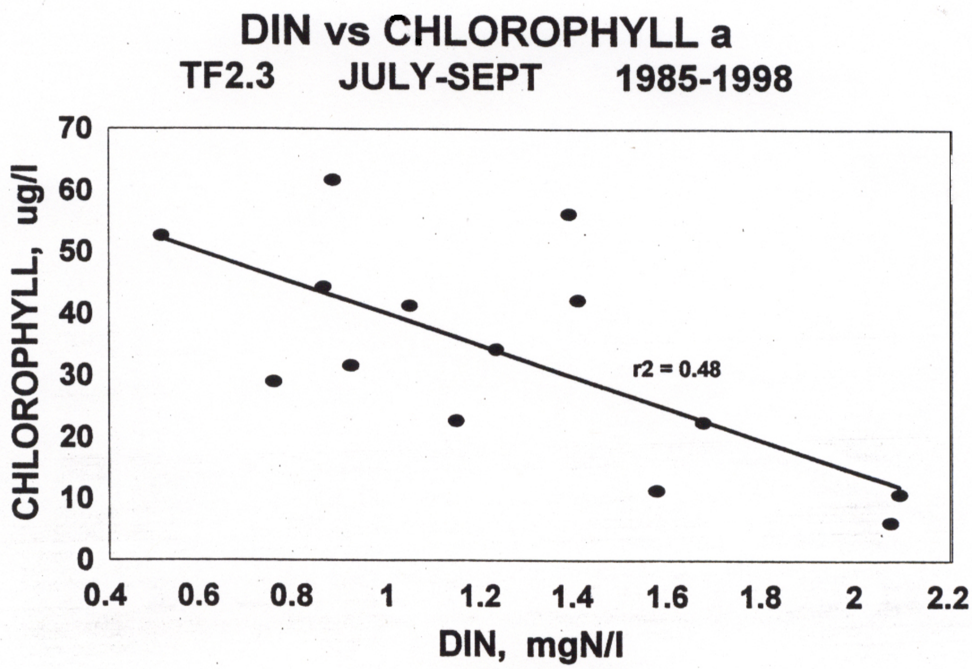


Figure 34: Relationship between DIN concentration and phytoplankton biomass at station TF2.3 during July-September, 1985-1998.

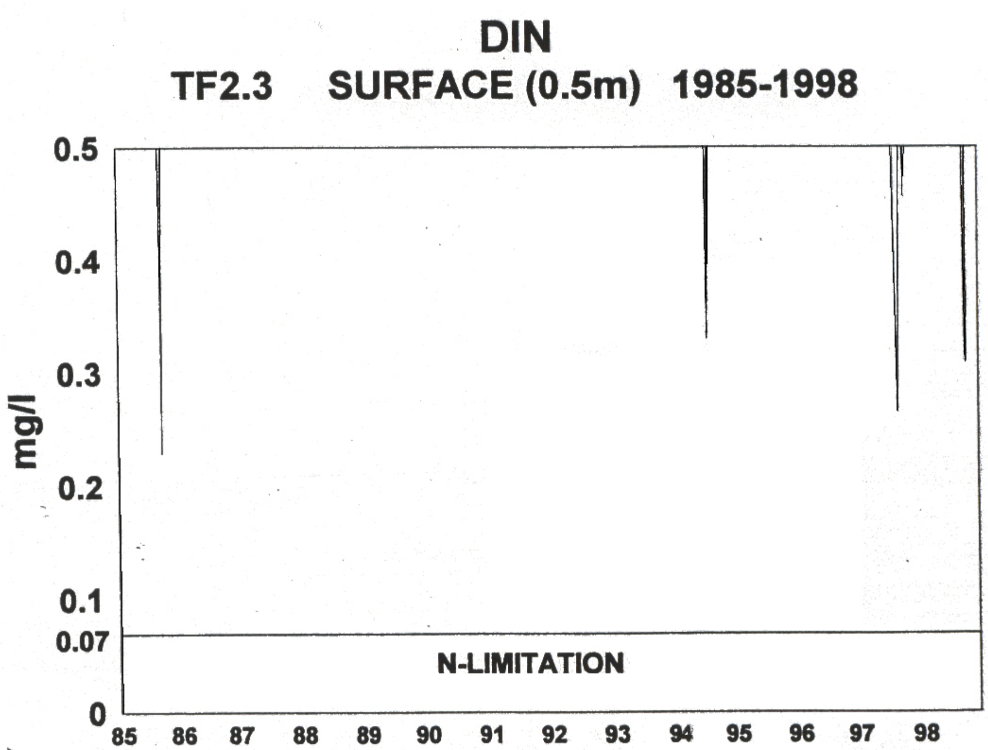


Figure 35: Surface DIN concentrations at station TF2.3 indicating limiting concentration of 0.07 mg/l - 1985-1998.

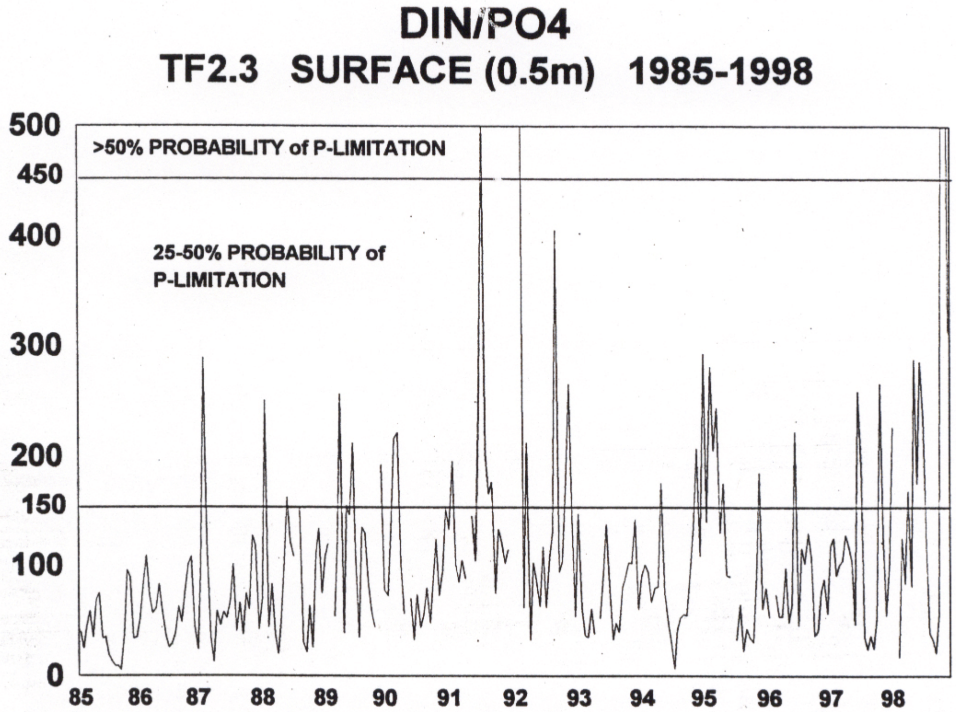


Figure 36: Ratio of surface DIN/PO4 at station TF2.3 indicating the probability of phosphorus limitation -1985-1998.

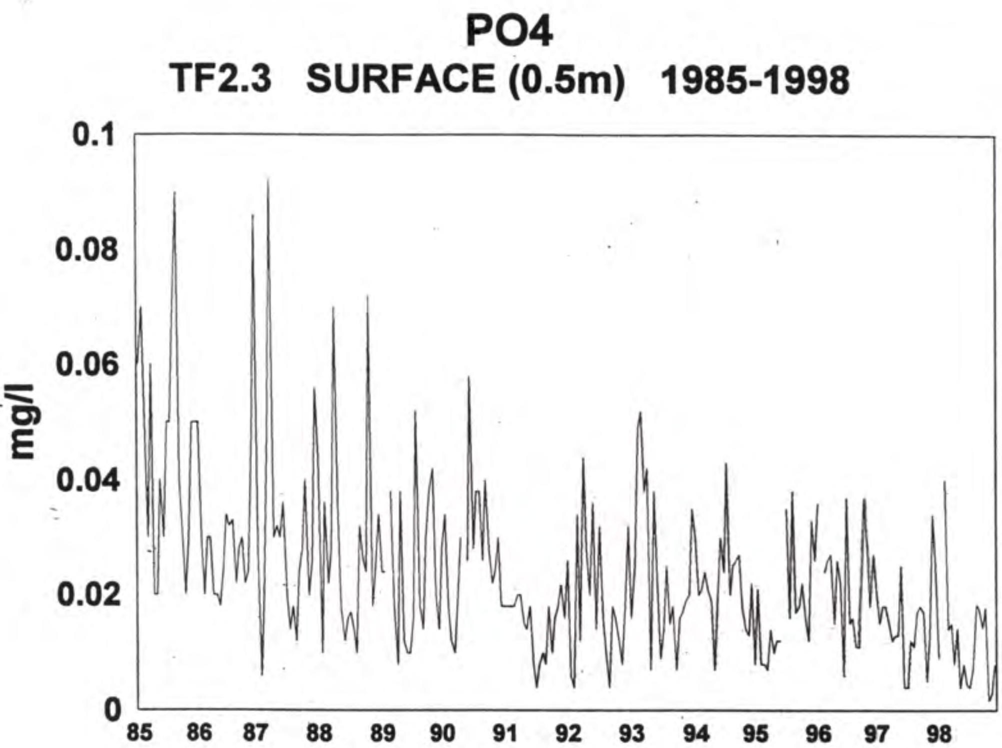


Figure 37: Surface concentrations of PO4 at station TF2.3 - 1985-1998.

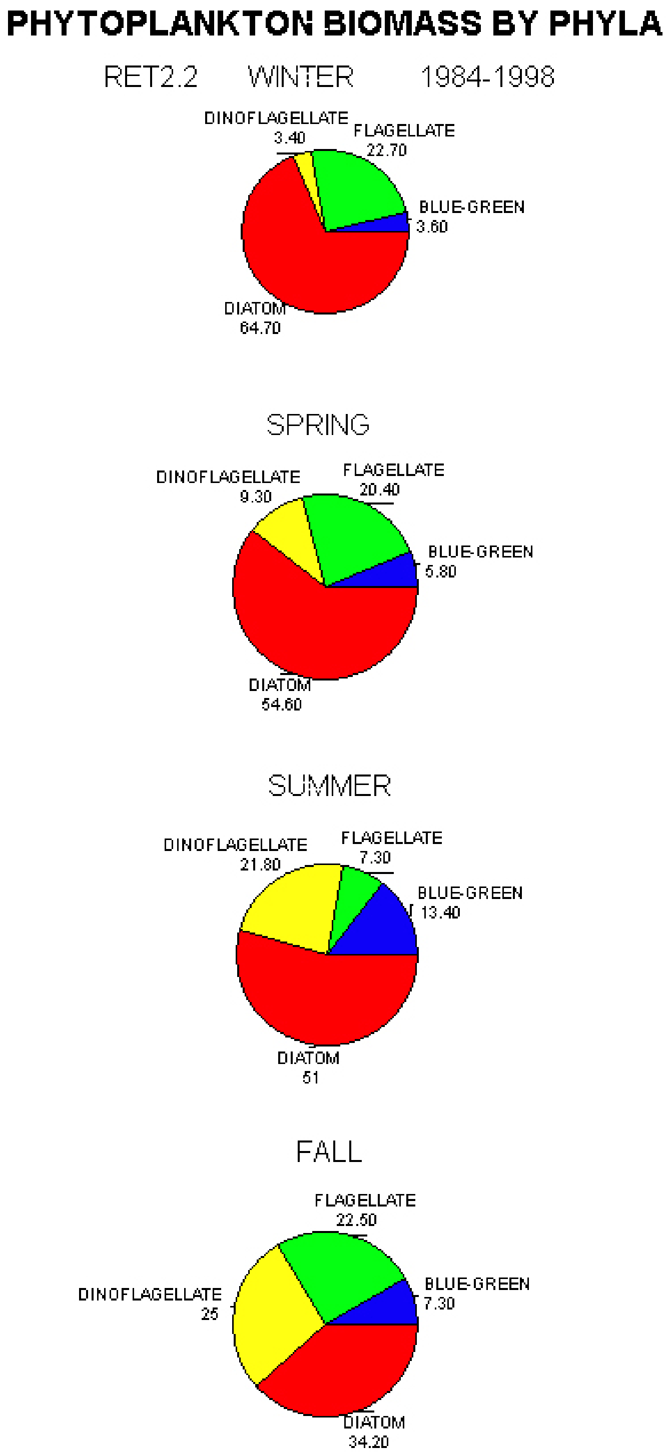


Figure 38. Mean seasonal phytoplankton biomass by phylum at station RET2.2, 1984 - 1998.

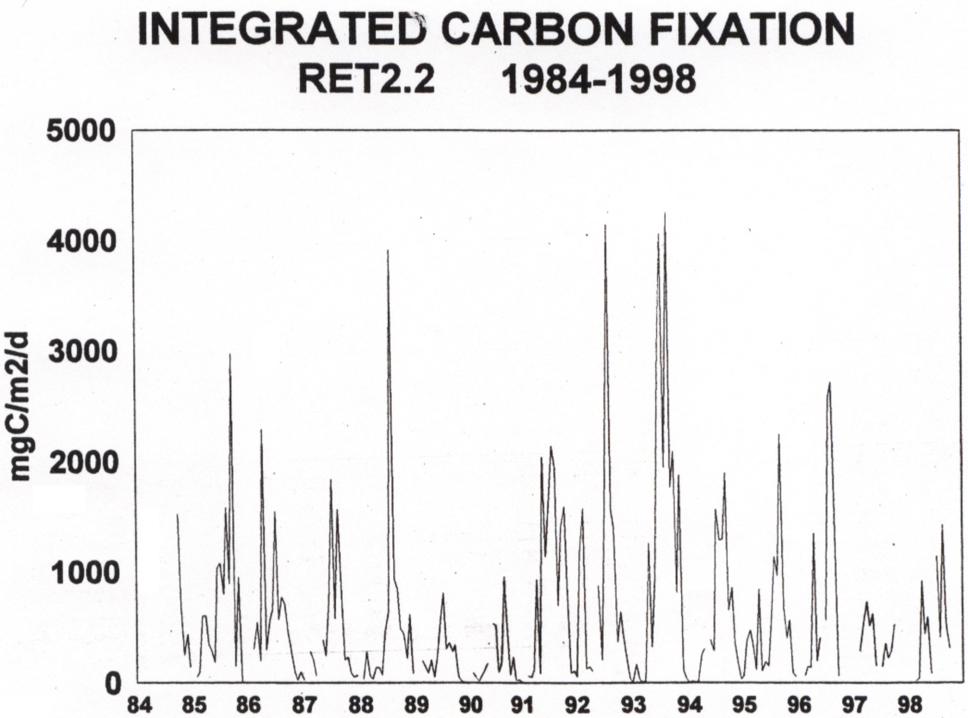


Figure 39: Integrated carbon fixation at station RET2.2 - 1984-1998.

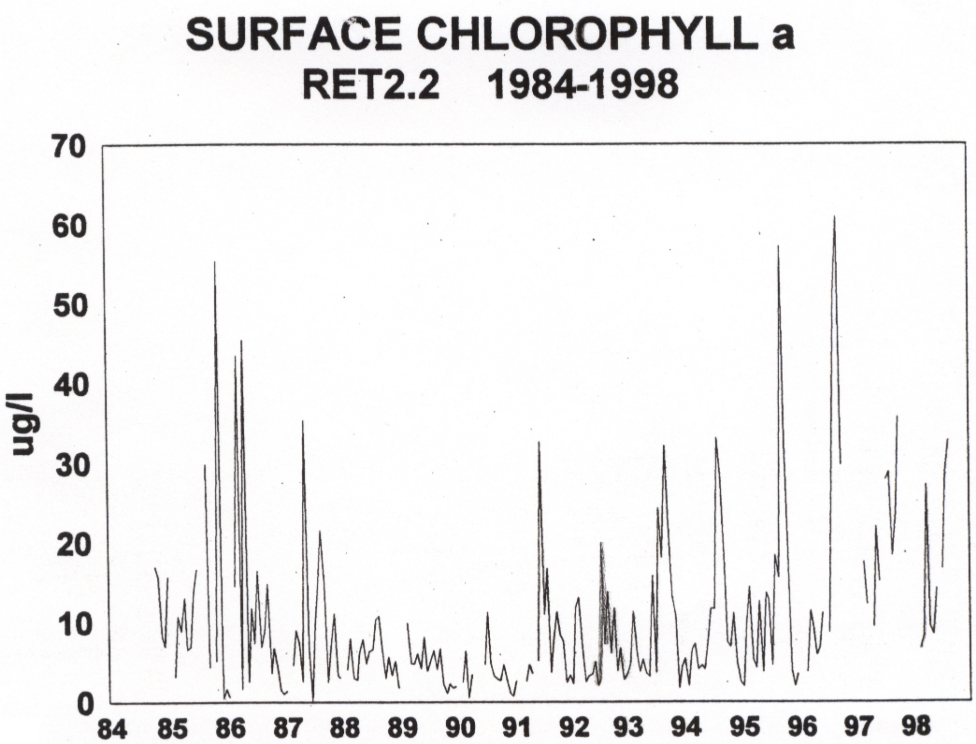


Figure 40: Surface chlorophyll a concentrations at station RET2.2 - 1984-1998.

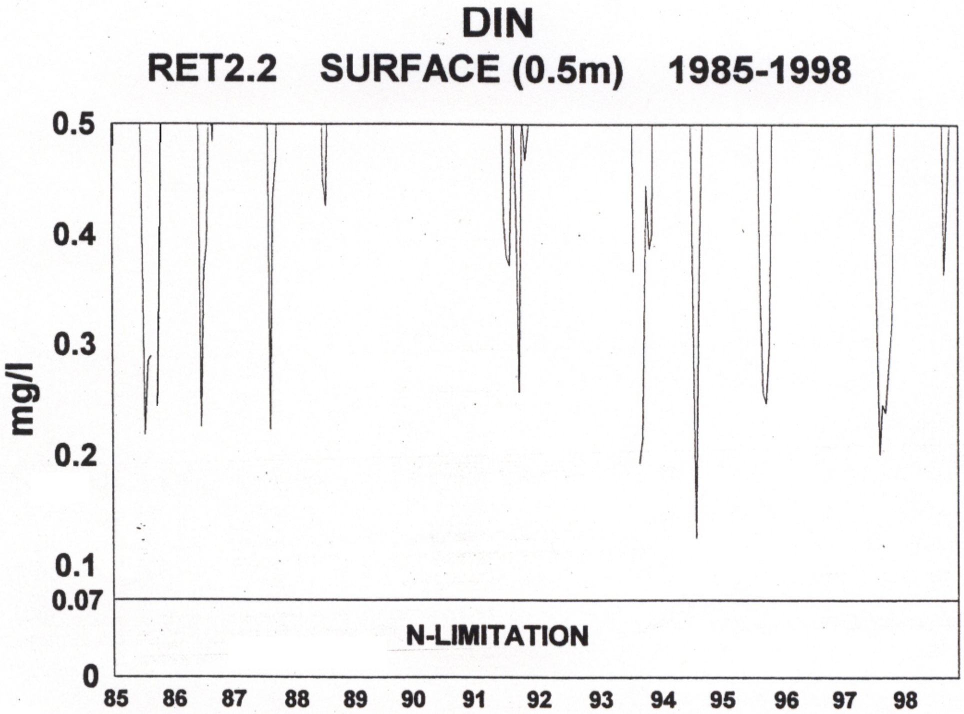


Figure 41: Surface DIN concentrations at station RET2.2 indicating a limiting concentration of 0.07 mg/l - 1985-1998.

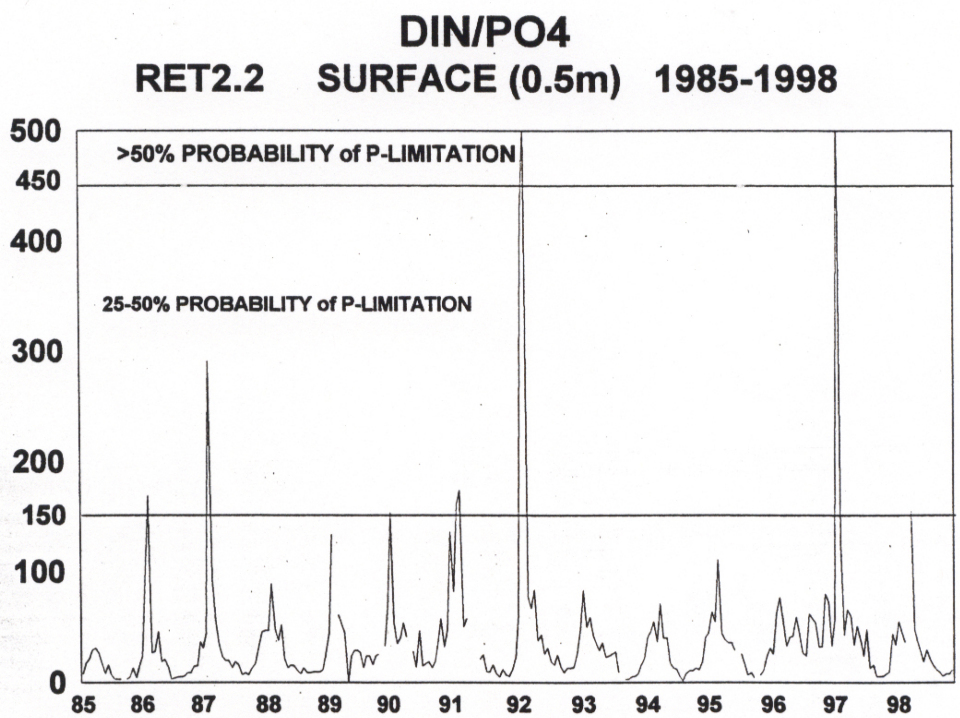


Figure 42: Ratio of surface DIN/PO4 at station RET2.2 indicating the probability of phosphorus limitation - 1985-1998.

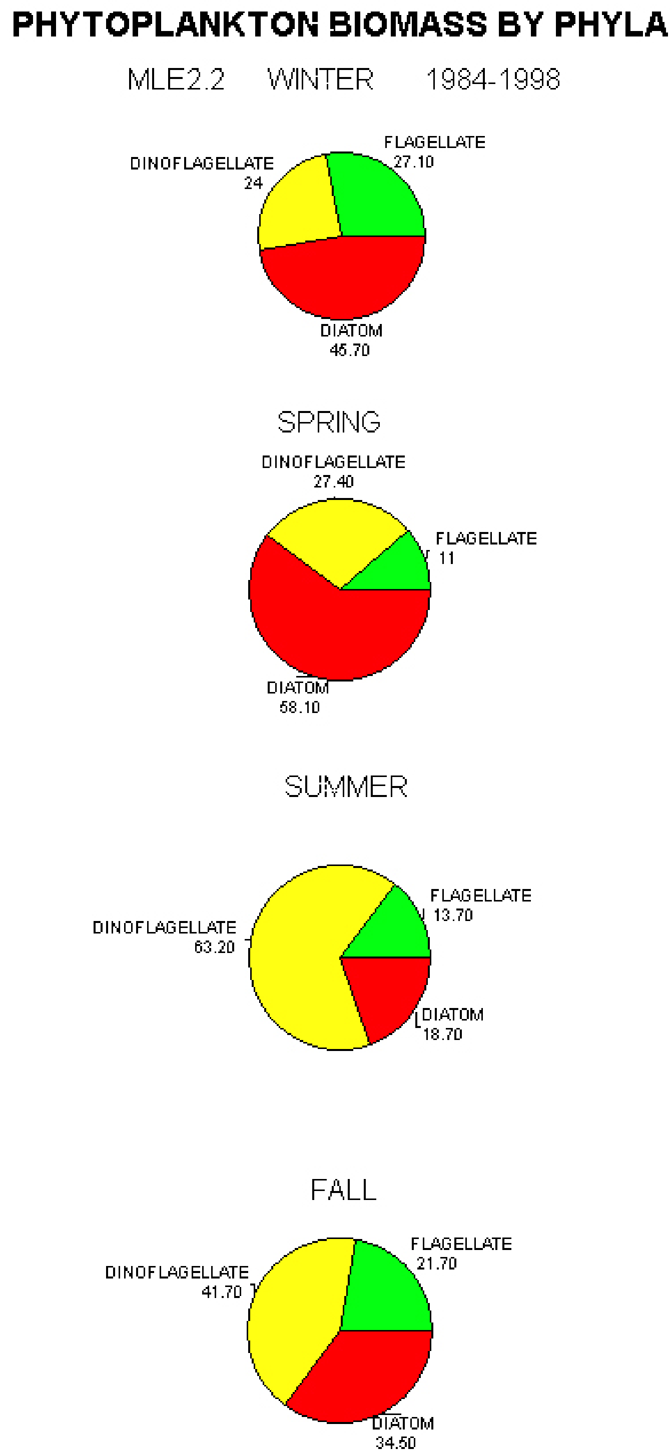


Figure 43. Mean seasonal phytoplankton biomass by phylum at station LE2.2, 1984 - 1998.

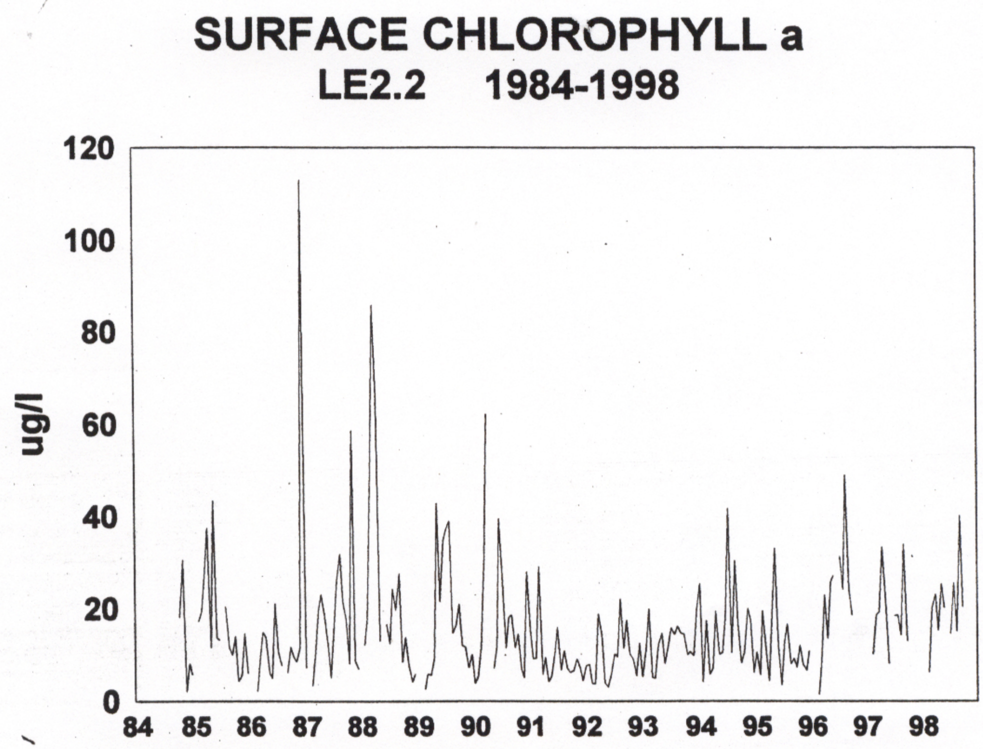


Figure 44: Surface chlorophyll a concentrations at station LE2.2 - 1984-1998.

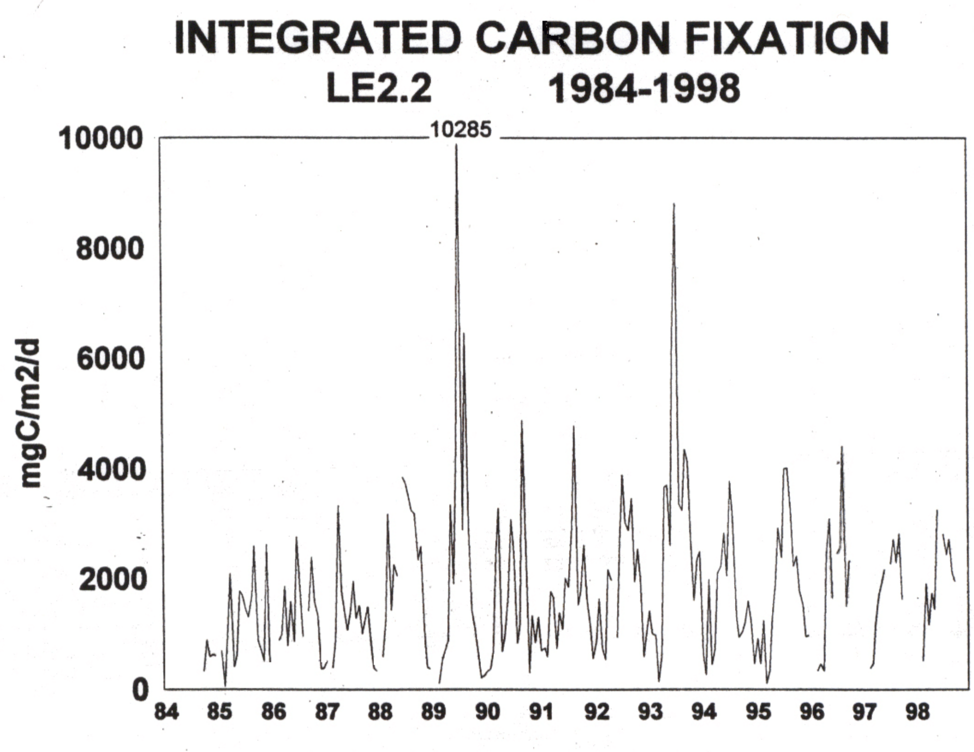


Figure 45: Integrated carbon fixation at station LE2.2 - 1984-1998.

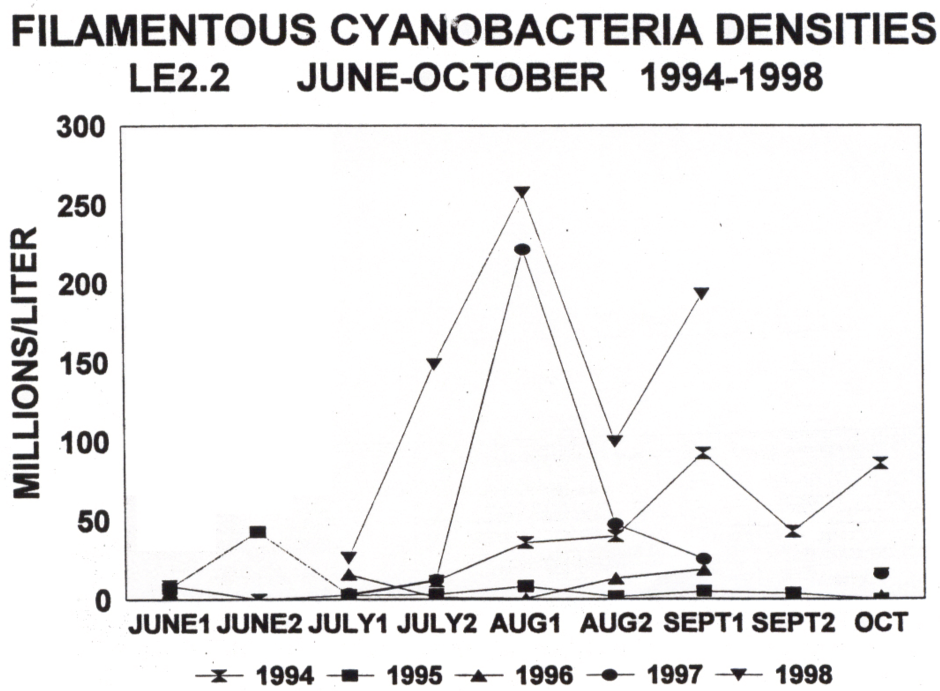


Figure 46: Densities of a filamentous cyanobacteria at station LE2.2 during June-October, 1994-1998.

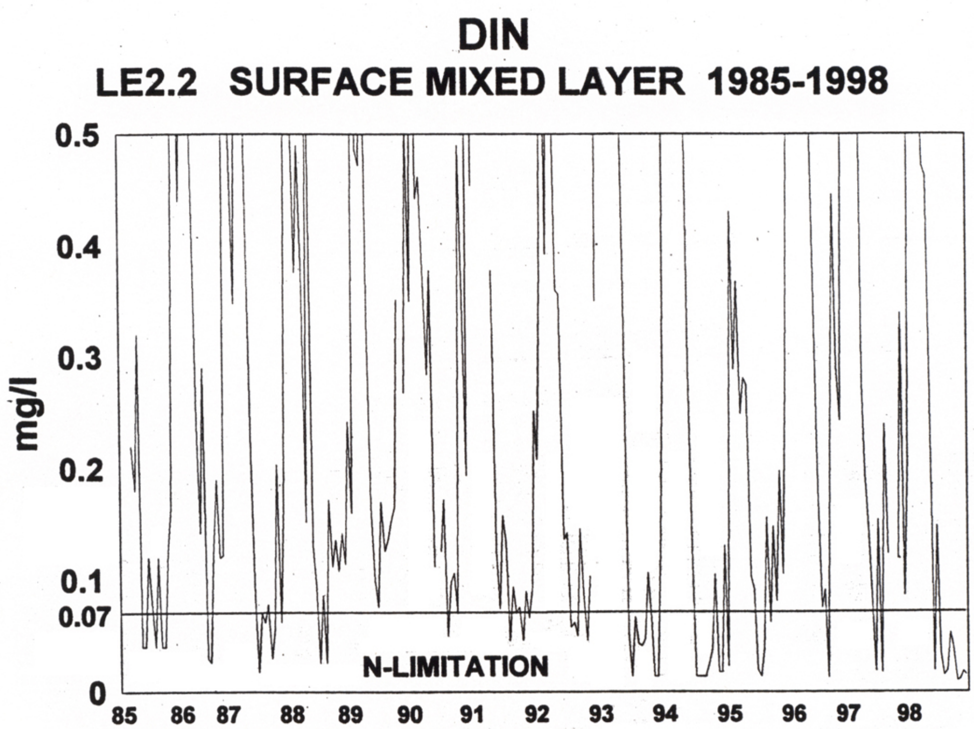


Figure 47: Surface DIN concentrations at station LE2.2 indicating a limiting concentration of 0.07 mg/l - 1985-1998.

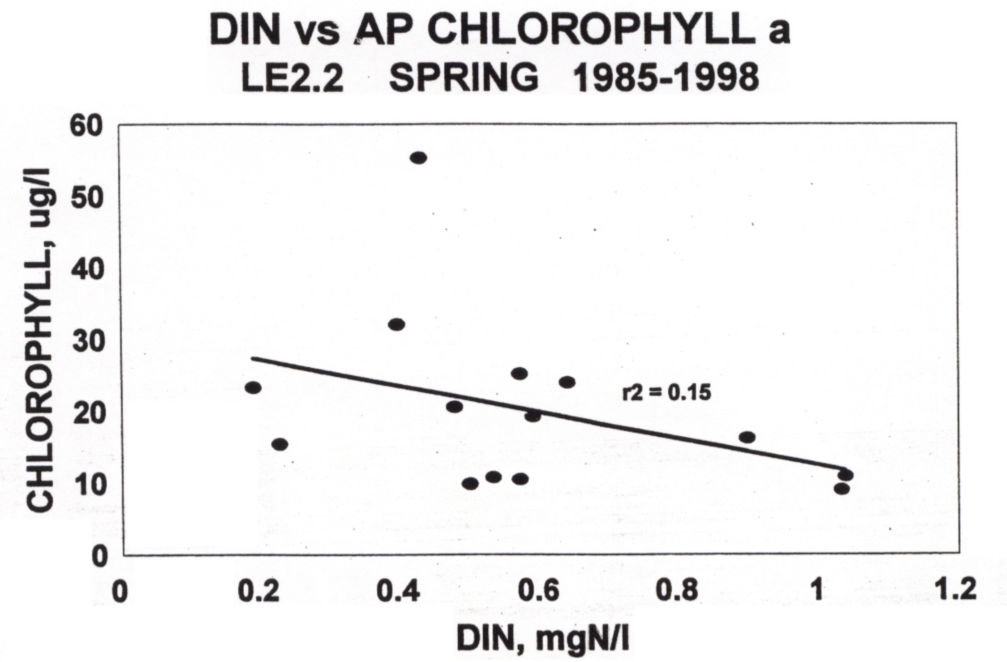


Figure 48: Relationship between DIN concentration and phytoplankton biomass at station LE2.2 during March-May, 1985-1998.

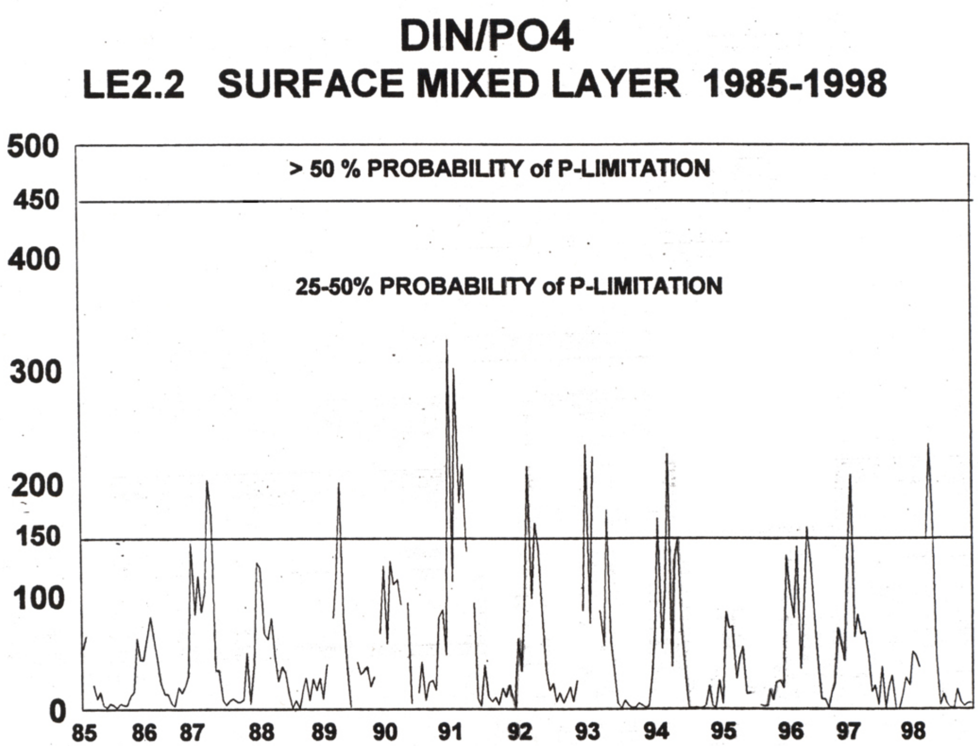


Figure 49: Ratio of surface DIN/PO4 at station LE2.2 indicating the probability of phosphorus limitation - 1985-1998.

PRIMARY CONSUMERS: ZOOPLANKTON AND SOFT-BOTTOM BENTHOS (OBJECTIVE 3)

Most managers consider a productive, healthy aquatic ecosystem to be one in which biomass produced by plants moves efficiently through the food web and accumulates in long-lived, large-bodied consumers which are grazers and predators such as fish, shellfish and water birds. These food web pathways are characteristic of relatively undisturbed aquatic ecosystems. The contrast between the enormous Potomac finfish and oyster harvests in the 19th century and the low abundances observed in the many fisheries species and non-commercial finfish species now (see Anthropogenic Impacts above) is evidence that these food web pathways have been considerably weakened. Major losses in two key "living" habitats, the SAV beds and oyster reefs, during the 20th century significantly reduced the amount of structural habitat available in the estuary. We can infer that fish species inhabiting SAV beds and oyster reefs also declined, and that this further weakening food web pathways leading to long-lived, large-bodied consumers. Finally, the spread of eutrophication downstream from the rapidly growing Washington metropolitan area caused two important changes: a) phytoplankton buildups and shifts to undesirable species (bluegreen algae) which altered "bottom-up" controls on the food web, and b) reduced water clarity and depleted bottom oxygen which diminished favorable open water habitats. Eutrophication, in effect, exacerbated losses already occurring at the middle and top of the food web.

What happened to "primary consumer" populations of the tidal Potomac food webs in this 20th century period of declines and degradation? These are the zooplankton and the soft-bottom benthic macroinvertebrates which consume phytoplankton and suspended organic material. They comprise the principal food source for most forage fish species and represent an "interface" in the food web where nutrient enrichment impacts at one end and heavy harvest pressure at the other meet. Like fish and birds, zooplankton and benthos are affected by changes in habitat quality and quantity. It could be supposed that populations of these consumer taxa flourished in the early 20th century as their algal food supply increased and their many predators declined. However, habitat deterioration and loss in the same time period are just as likely to have depressed their population abundances.

Historical Populations

We can try to reconstruct what historical (pre 1900s) populations were like from circumstantial evidence. Zooplankton and benthic macroinvertebrate production, for example, would have to have been high in order to support the abundant finfish populations documented in fisheries records of the 1800s. Using 1830s harvest records of the two major planktivores, herring and shad, we can roughly estimate the daily zooplankton biomass production needed to support just these harvested individuals when they were juveniles in the upper and middle estuary. Seventy-five million pounds of herring and ninety million pounds of shad translate to approximately 75 million adult herring and 22.5 million adult shad. Assuming a moderate growth rate of 10 mg dry weight/individual/day (from Klauda et al 1991), a metabolic efficiency of 0.1, and a total habitat volume of 1 billion cubic meters (volume of the tidal fresh and oligohaline zones, including tributaries), these individuals alone consumed roughly 10 mg zooplankton dry weight

m^{-3} day^{-1} as juveniles in the upper Potomac. These harvested individuals represented a fraction of their original juvenile cohort, and their species were just two of many that fed on zooplankton. Total finfish consumption of zooplankton thus could conceivably have been equal to or greater than1 g zooplankton dry weight m^{-3} day^{-1}. Total zooplankton production in the upper and middle estuary would have to have been higher than this level in order to counter both finfish and benthos predation.

Historical species compositions were very likely different than those found today. Zooplankton samples collected in the fairly pristine upper Chesapeake Bay in the early 1900s (Wilson 1932) suggest that large-bodied, long-lived species capable of resisting tidal mixing and able to undergo daily vertical migrations could have dominated in the 18th and 19th century Potomac estuary. Furthermore, species sensitive to bluegreen algal toxins or unable to utilize bluegreens as food are often absent from present day assemblages in the upper Potomac, but could possibly have been present in the past. Favorable dissolved oxygen conditions probably allowed large-bodied, long-lived bivalve species to dominate the soft-bottom benthos community as they do today (Weisberg et al 1997). References to abundant oyster reefs and bottom dwelling fish such as sturgeon indicate decomposition by bacterioplankton in the mesohaline water column was never sufficient to create the extensive, anoxic "dead-zones"observed there today. Abundant oyster reefs would have also actively interfered with water column stratification (e.g. Lenihan et al 1999) and slowed oxygen depletion in bottom waters.

No information is available on historical abundances of zooplankton and benthos but considering the heavy predation pressure on these populations, their overall abundances may have been modest or even near the food thresholds limiting fish growth rates at times. Overall abundances still would have to have been higher than present-day abundances in order to produce the biomass needed to support the abundant fish populations of this era. We can infer that the abundant, shallow water communities of zooplankton and benthic macroinvertebrates associated with SAV beds were lost when SAV declined in the 1960s.

Intermittent monitoring of zooplankton and benthos was done prior to the mid 1980s in the Potomac estuary and regular, long-term monitoring data were collected afterward. The rest of this chapter will discuss zooplankton and benthos status and trends since 1965 and how they appear to have responded to management-related water quality changes in that time period.

Tidal Fresh River (Upper Estuary)

There have been several management-related water quality improvements in the river-like, tidal fresh region of the Potomac estuary over the past 30+ years that would have directly affected zooplankton and benthos communities. They are a) an increase in the minimum levels of summer bottom dissolved oxygen above the minimum requirement of 5 mg/l for a healthy habitat, b) a significant decrease in ammonia to concentrations below EPA 1999 chronic continuous criteria for toxicity, c) a 30% - 50% long-term decline in surface chlorophyll concentration, and d) a diminishing extent and intensity in summer bluegreen algal blooms (from Appendix A and B; W. Romano, pers. comm.). Total ammonia concentrations in 1969-1974 were approximately 0.8 mg l^{-1} and ranged between <0.1 and levels as high as 3.2 mg l^{-1} (Pheiffer 1976). The highest values were above EPA 1999 chronic continuous criteria for toxicity (US EPA 1999). The 1985-1986 values with annual medians of 0.23 - 0.15 mg l^{-1} (W. Romano, pers.

comm.) were usually below EPA chronic continuous toxicity criteria. Total suspended solids and Secchi depth still appear more related to flow rate than to management actions. Levels of both have fluctuated over the past 30+ years. Present levels are slightly more degraded than the late 1970s levels. Chlorophyll levels still increase when high flows bring large nutrient loads to the upper estuary or when very low flows lengthen the residence time. Bluegreen algal blooms still occur when summer environmental conditions are favorable, and their frequency and extent have increased in the 1990s (see above). Habitat changes not related to intentional management actions also affected zooplankton and benthos during the past 30+ years. They include the introduction of the exotic uniod bivalve, *Corbicula fluminea* (Asiatic clam), in the late 1970s, b) the abrupt appearance of the exotic SAV *Hydrilla* near Washington in the early 1980s, its spread downstream and its gradual subsidence, and c) more frequent high-flow events in the 1990s which broadened the extent of the estuary's freshwater zone and negated reductions made in sediment inputs above the fall-line.

Taxa richness

Overall there has been little change in taxa richness, a measure of species diversity, in open water habitats of the upper Potomac estuary during the past 25 years. Surveys in the 1970s found more than 70 freshwater macrobenthic invertebrate species (Pfitzenmeyer 1976, Lippson et al 1979). The CBP monitoring program found most of the same species between 1984 and 1996, and observed no trend in species richness (Ranasingha 1999). Community dominants during the later time period were immature Tubificidae worms (mainly *Limnodrilus hoffmeisteri*) which comprised 38.9% of the population, followed by *Corbiula fluminea* (Asiatic clam) at 5.1%, *Quistadrilus multisetosus* at 4.5%, *Musculium transversum* at 4.3%, *Gammarus* spp. at 3.7% (sometimes found in the plankton), and *Branchiura sowerbyi* at 2.3%.

Surveys in the 1970s found 19 copepod and 10 cladoceran "mesozooplankton" species (Sage et al 1976). The smaller "microzooplankton" species (rotifers, protozoans) were not well studied. The CBP monitoring program found similar numbers of copepod and cladoceran species, and observed no change in the numbers of copepod or cladoceran species between 1984 and 1996 (Figure 50). Seasonal dominants over the entire period included the rotifers *Brachionus calyciflorus* and *Keratella* spp., the cladocerans *Bosmina longirostris, Moina micrura* and *Diaphanosoma brachyurum*, and the copepods *Eurytemora affinis, Cyclops vernalis* and *Mesocyclops edax.*

Abundance

Multiple monitoring data sets collected in the upper Potomac since 1981 show mixed status and trend patterns for zooplankton and benthos abundances for the past 15 - 20 years. The initial ecosystem response to the phosphate ban and subsequent phosphate reductions had occurred, and this was a period of good dissolved oxygen, high chlorophyll, poor water clarity and generally increasing freshwater flows.

Benthos abundances have increased since the 1960s - 1970s period when a summer "oxygen sag" impacted the between Washington, D.C. and Indian Head (Pfitzenmeyer 1976), but in the absence of the once lush SAV beds they probably have not returned to historical levels. Total abundances climbed sharply in the late 1970s when *Corbicula fluminea*, the asiatic clam, was introduced to the Potomac. Its populations peaked in the early 1980s, and declining trends in total benthos abundance and biomass during the 1980s and 1990s reflect *Corbicula* declines (Figure 5-5, Appendix G); there were no significant trends in interface feeders, deposit feeders, pollution-tolerant taxon, or pollution-sensitive taxon abundance (Appendix G). The Benthic Index of Biotic Integrity (Weisberg et al 1997) was not thoroughly developed for low salinity waters of the Chesapeake system when this report was drafted, however Richkus et al (1994) found the tidal fresh - oligohaline Potomac segments have the highest species richness (# species per sample), the third highest total abundance (# m^{-2}) and third highest total biomass (grams ash free dry weight per m^{-2}) of the tidal fresh stations sampled. This would suggest that the Potomac has a healthy benthic macroinvertebrate population relative to present-day Chesapeake populations, however a large proportion of the population still consists of pollution-tolerant species such as tubifex worms.

Between 1985 and 1998, mesozooplankton abundances rose and fell from year to year and did not reflect significant long-term trends (Appendix F). Spring and summer mesozooplankton abundances near Indian Head, for example, rose and declined in the 1980s and then rose again in the mid 1990s. An upward trend in summer mesozooplankton abundance recently became evident (Appendix F) but is probably related to the flow-related expansion of the freshwater zone. Seasonal mesozooplankton abundances are moderate to low relative to those in other Chesapeake tidal fresh areas. Absolute values of spring abundances were frequently "below minimum" (5,000 - 15,000 per m^3) or "poor" (<5,000 per m^3) for normal larval striped bass growth prior to 1995 but have climbed to "minimum" (15,000 - 25,000 per m^3) and occasionally "optimal" (> 25,000 per m^3) levels in the late 1990s, probably as a result of the higher spring flows (Jacobs et al 1998). Rotifer abundances are relatively high compared to other Chesapeake subestuaries. Rotifer biomass has shown no significant long-term trend in any season but overall has increased since 1985 (Lacouture et al 1999, Appendix F).

Zooplankton abundances in open waters of the upper Potomac estuary are highly affected by flushing time. Abundances are typically highest in the larger, slowly flushing tributaries (e.g. Anacostia River, Quantico Creek) and in the broad mainstem segment near Indian Head, and they are lowest in the riverine mainstem below Chain Bridge (Buchanan and Schloss 1983, Jones and Kelso 1988, Storms 1981, Aurand 1984). Zooplankton abundances in the upper Potomac estuary also appear to be affected by abundances of their finfish predators. Buchanan and Vaas (1993) found an inverse relationship ($r^2 = 0.48$; $p<.03$) between the average summer mesozooplankton abundance at CBP biomonitoring station XEA6596 and the average summer zoo-planktivore finfish abundances at adjacent Maryland Estuarine Juvenile Finfish Survey stations. The inverse relationship suggests that as predator (finfish) abundance increases, prey (zooplankton) abundance decreases.

Transition Zone (Middle Estuary)

There was one management-related improvements in water quality that could have directly affected zooplankton and benthos communities in the transition (oligohaline) zone: a >95% decline in average chlorophyll concentrations over the past 30+ years. Other eutrophication parameters have not changed significantly or were ecologically insignificant. Summer bottom dissolved oxygen frequently dips to 4 - 5 mg/liter, and average dissolved oxygen concentrations in the water column have trended downward (Appendix A and B). Secchi depth and total suspended solids concentrations are altered by high-flow events and droughts, but otherwise have remained at fairly consistent levels with secchi depth ~ 0.6 meters and total suspended solids ~ 25 mg/liter. Ammonia concentrations in 1969-1974 were approximately 0.3 mg l^{-1} and ranged from <0.05 to 1.5 mg l^{-1} (Pheiffer 1976); 1985-1986 annual medians for ammonia were 0.14 - 0.093 mg l^{-1} (W. Romano; pers. comm.). Total ammonia levels were occasionally above EPA 1999 chronic continuous criteria for toxicity (US EPA 1999) in the earlier period but below the criteria in the latter period. Habitat changes not directly related to management efforts in the past 30+ years that possibly affected the zooplankton and benthos include a) the spread of the exotic SAV *Hydrilla* to the oligohaline zone in 1989, and b) the more frequent high-flow events in the 1990s which reduced salinities in the zone.

Taxa richness

Despite the lack of improvement in *eutrophication* parameters, zooplankton taxa richness improved steadily in the Potomac oligohaline open water habitats during the past 24 years (1975 - 1998), over several wet and dry cycles. The number of cladoceran zooplankton species rose ~ 3-fold and the number of copepod zooplankton species rose ~ 4-fold (Figure 50). Seasonal dominants consist of a mixture of freshwater and brackish water species, including most of the freshwater dominants listed above as well as the euryhaline copepods *Acartia tonsa* and *Ectinosoma curticorne* and barnacle nauplii.

The number of benthic macroinvertebrate taxa per sample also rose significantly in the oligohaline (Appendix G). Dominant taxa consisted of six euryhaline taxa (the bivalve *Macoma balthica*, the amphipods *Leptocheirus plumulosus* and *Gammarus*, and the polychaetes *Polydora cornuta*, *Streblospio benedicti*, and *Marenzelleria viridis*) and four freshwater taxa (oligochaete worms, the bivalve *Rangia cuneata*, chironomid larvae, and the isopod *Cyathura polita*).

Abundance

Seasonal trends (1984 - 1998) in benthic macroinvertebrate, microzooplankton and mesozooplankton abundances and/or biomasses were insignificant or negligible in this salinity zone. Seasonal and annual means of rotifer biomass were high relative to those in other Chesapeake oligohaline segments (S. Sellner, personal communication); seasonal mesozooplankton abundances were moderate to low (Appendix F). The interesting juxtaposition of increasing copepod and cladoceran species richness, a lack of trends in zooplankton abundance or biomass, and moderate to low abundances of mesozooplankton suggests factors other than eutrophication are affecting populations.

Mesohaline (Lower Estuary)

Management-related changes in water quality that would have directly affected mesohaline zooplankton and benthos communities in the lower estuary over the past 30+ years include a) 50% - 60% decreases in chlorophyll (except at the mouth of the estuary), b) 12% - 57% decreases in total organic carbon, c) slight declines in secchi depth, d) slight declines in bottom dissolved oxygen (Appendix A and B). In addition, increases in total suspended solids were seen in the mid-1970s - 1996 timeframe. Ammonia declined from approximately 0.15 mg l^{-1} (range: <0.05 - 1.0 mg l^{-1}) to 1985-1986 annual medians of 0.075 - 0.02 mg l^{-1}, but levels rarely exceeded the

EPA 1999 chronic continuous criteria for toxicity (US EPA 1999) in either time period.

Taxa richness

Numbers of benthic macroinvertebrates species rose significantly in low mesohaline (5 - 10 ppt) mud habitats and in sand mesohaline habitats but remained mostly unchanged in other mesohaline habitats (Appendix G). Benthic macroinvertebrate taxa richness in deep waters of the Potomac mesohaline zone is among the poorest in the Chesapeake system (Richkus et al 1994). *Rangia cuneata* invaded the middle and lower Potomac estuary in the mid 20th century, probably from the James River system (Pfitzenmeyer 1976), and apparently dominated the community before gradually declining to more stable levels. In the water column, numbers of cladoceran and copepod mesozooplankton species have remained mostly unchanged in the past 15 years (Figure 50).

Abundance

Benthic macroinvertebrate abundances have shown increasing trends in some mesohaline habitats but no improvement in the deep, mud habitats of the mainstem that become anoxic each summer. The Benthic IBI for the lower Potomac deep stations is the worst in the Chesapeake Bay system (Appendix G). Benthic macroinvertebrate abundances and biomasses in low mesohaline (5-10 ppt) open water habitats and high mesohaline (10 - 18 ppt) shallow water habitats were moderate relative to the range of values found in other Chesapeake tributaries, and extremely depauperate in the deep, mud habitat (Richkus et al 1994).

Total mesozooplankton abundances in a 1974 spring survey were 2,000 - 15,000 m^{-3} in the lower Potomac estuary (Sage et al 1976). These spring abundances were similar to abundances in a 1976 - 1980 spring survey in the middle Chesapeake Bay mainstem near Calvert Cliffs (Olson 1987), indicating these two adjacent mesohaline areas were comparable at that time. Thus Potomac summer abundances could be expected to have had geometric means of 10,000 - 34,000 m^{-3} in the late 1970s (from Olson 1987). Since 1985, however, the CBP monitoring program has observed total abundances in the lower Potomac estuary that are low relative to other Chesapeake tributary mesohaline zones, with summer geometric means less than 12,000 m^{-3} and usually closer to 5,000 m^{-3} (Appendix F). A similar decline occurred after 1990 in the middle Chesapeake Bay mainstem mesozooplankton populations. Potomac mesozooplankton abundances have varied since 1985, with a small peak in 1990, but have shown no long-term trends. Nauplii abundances, a rough measure of reproduction rates in this copepod-dominated salinity zone, were high compared to other mesohaline regions of the Chesapeake system suggesting that copepod productivity was relatively high. Rotifers were not observed in 1974 spring survey of the Potomac lower estuary (Sage et al 1976), possibly because coarse sized nets (>150µ) were used. Counts from whole water samples, however, suggest rotifers were in fact not very abundant. Rotifer abundances in the lower Potomac estuary are presently at moderate levels relative to other Chesapeake mesohaline areas (S. Sellner, pers communication).

Continuing Eutrophication Impacts

Nutrient reductions have obviously alleviated some eutrophication-related problems, in particular the low dissolved oxygen levels and extreme Cyanobacteria (bluegreen algae) summer blooms in the upper estuary. There appears to be a need to further restore habitat conditions and food quality before healthy, productive populations of zooplankton and benthic macroinvertebrates can be restored. In the following sections, we discuss the continuing impacts of eutrophication on habitat condition and food quality for zooplankton and benthic macroinvertebrates.

Low dissolved oxygen

The impact of low dissolved oxygen on the tidal river immediately below Washington, DC. during the 1970s was evident in the depressed zooplankton and benthos population abundances. Populations rebounded as oxygen improved further downstream and the river entered the oligohaline section (Pfitzenmeyer 1976, Sage et al 1976). *Corbicula fluminea* reached very high abundances in tidal fresh waters shortly after it became established in the late 1970s, and was able to ameliorate the phytoplankton-driven oxygen sag below Washington (Cohen et al 1984) before its abundances declined in 1985. The clam, in combination with favorable flow conditions, may have also helped improve water clarity to the extent that the SAV exotic species, *Hydrilla*, could become established (Phelps 1994). The oxygen sag continued to diminish in the mid 1980s, presumably because of long-term improvements in chlorophyll concentrations in the area (see above). Presently, summer dissolved oxygen concentrations usually - but not always - meet the 5 mg/liter minimum level required by benthos, zooplankton and fish communities.

Summer hypoxia-anoxia in the lower Potomac estuary continues to severely impact the benthic macroinvertebrate community, causing the Benthic Index of Biotic Integrity for this area to score very low relative to other Chesapeake areas (Appendix G). The impact of low dissolved oxygen on the zooplankton community is not as clear cut. A summer diel vertical migration (DVM) study in 1985 demonstrates that mesozooplankton populations in the lower Potomac estuary do migrate into the low oxygen bottom layer (Buchanan, in prep.) which suggests they can employ physiological mechanisms to temporarily cope with low dissolved oxygen environments. Diel vertical migration is an important, light-driven behavior which normally serves to maintain the longitudinal position of mesozooplankton populations in estuaries and to reduce their vulnerability to visual predators (finfish) during the day. Proximity to the hypoxic bottom water, however, can be detrimental. White and Roman (1992) observed high mortalities in newly laid *Acartia* eggs sinking into the low oxygen bottom layer of the Chesapeake mainstem. In summary, summer low dissolved oxygen in the lower Potomac estuary heavily impacts the benthic macroinvertebrates and probably impacts mesozooplankton to some degree. Mesozooplankton can benefit from DVM behavior but potentially lose young and/or individuals to anoxic-induced mortality.

Suspended solids

Estuaries are normally turbid environments (Day et al.1989) but very high turbidity impairs normal functioning of estuarine ecosystems. Excess concentrations of dissolved and suspended particulate matter reduce light penetration (secchi depth) to a point where photosynthesis of SAV and phytoplankton is “light-limited,” dominance of bluegreen algal species is favored, light-driven behaviors of zooplankton and benthic macroinvertebrates are weakened, and visual acuity of fish is impaired. Suspended particulate matter, or solids, can also clog the filtering apparatuses of estuarine animals, dilute the phytoplankton food of many filter feeders with non-food particles, and smother bottom communities. Analysis of water quality monitoring data indicate SAV light requirements are frequently not met in the upper and middle segments of the Potomac (Appendix E) and bioassay results demonstrate that phytoplankton are most frequently light-limited in the upper and middle segments of the Potomac (see Primary Producers above).

The impacts of present-day levels and compositions of suspended sediments on zooplankton and benthic macroinvertebrate populations have not been well documented for the Potomac estuary. We know that mesozooplankton could respond to light stimuli and undergo diel vertical migrations in the upper and lower estuary during the early 1980s (Buchanan and Schloss 1983; Buchanan, unpublished data). There is an apparent relationship between high turbidity levels, low dissolved oxygen and the increasing dominance of ctenophores and cnidarians (zooplankton predators) in mesohaline areas adjacent to the lower Potomac (Breitburg et al. unpublished data; Breitburg et al. in review; Breitburg et al. 1997; Breitburg et al. 1994). Suspended sediment impacts on Potomac zooplankton and benthos are potentially significant since the Potomac has high total suspended solids (TSS) concentrations in all of its bottom waters except near the mouth, and in most of its surface waters, especially in the two turbidity maxima, relative to TSS concentrations in other Chesapeake sub-estuaries. The monitoring data indicate one Potomac maximum occurs as the tidal freshwater river segment opens up into the oligohaline segment near Indian Head and another occurs as the oligohaline empties into the mesohaline lower estuary at the Route 301 Bridge near Morgantown.

Sparse submersed aquatic vegetation

The zooplankton and benthic macroinvertebrate communities associated with SAV beds presumably made a partial recovery as SAV returned to the upper Potomac in the mid 1980s. In her thesis, Monk (1988) found that zooplankton communities in SAV beds were more abundant and diverse than in nearby unvegetated sites where water movement was faster. SAV are apparently able to hold or trap zooplankton because they slow water movement. Also, the structural complexity of SAV beds seems to promote higher zooplankton diversity, and the more complex the SAV community (i.e. *Hydrilla* only vs mixed species beds), the greater the zooplankton diversity. Several zooplankton taxa were found almost exclusively in SAV beds, whereas the reverse was not true and open water zooplankton taxa were often found in SAV beds. The recent SAV declines in the tidal river which occurred as habitat quality declined in the 1990s (Appendix E) presumably caused a similar reversal in the SAV-associated zooplankton and benthic macroinvertebrate populations. This has not been confirmed.

Buchanan and Vaas (1993) noted the coincidence of a) the return of SAV, b) the resurgence of a top predator species, largemouth bass, which relies heavily on SAV for nesting and nursery habitat, c) a decline in total zoo-planktivorous fish which are prey to largemouth bass (i.e. primarily silversides, anchovies, and shiners but also including herring, shad and menhaden), and d) a peak in mesozooplankton abundances around 1987 - 1988. They proposed a possible trophic cascade effect initiated by the return of SAV, whereby an increase in the top predator population resulted in higher open water mesozooplankton abundances. These associations were made on monitoring data collected prior to 1993. Analysis of the associations with data updated through 1998 are underway but not completed.

Food quantity and quality

Nutrient reduction strategies have prompted management concerns that subsequent phytoplankton chlorophyll reductions will lessen the amount of organic matter reaching the bottom. While this will reduce bacterioplankton decomposition and hypoxia-anoxia in summer, it may also cause food shortages for benthic macroinvertebrates. (Presumably, turbidity would still be too high in this situation to allow the restoration of an epibenthic algal community.) Conversely, improving water quality conditions could improve food quality. In the Potomac estuary, six of seven stations showed declines in sediment carbon concentrations. The highest rates of decline were found in the tidal river; only the mesohaline mud habitat farthest down the river displayed no significant trend. There were no significant responses in total benthic abundance or biomass to the observed decreases in sediment carbon concentrations between 1982 and 1995, suggesting that the benthos are not yet food limited or enhanced by better foods. There also was no change anywhere in the relative proportion of organisms classified as pollution-sensitive or pollution-indicative (see Appendix G for more detail).

Management is similarly concerned that phytoplankton reductions will impact food supplies for zooplankton. Analysis of the monitoring data indicate that phytoplankton biomass and cell concentrations are typically well above levels considered limiting to mesozooplankton consumers (Buchanan and Schloss 1983; Sellner and Jacobs, undated report; Lacouture, pers. com.). Using multivariate regression methods, Sellner and Alden (1993) searched for but did not find direct, statistically significant relationships between phytoplankton and mesozooplankton parameters which would have indicated phytoplankton food quantity is limiting mesozooplankton growth. Furthermore, the dominant copepod, *Acartia tonsa*, is known to be an opportunistic omnivore and will ingest rotifers and protozoans as well has phytoplankton cells (Lonsdale et al 1979, Roman 1984). *Eurytemora affinis* can utilize the microzooplankton associated with detritus (Heinle and Flemer 1975).

Unlike the mesozooplankton, microzooplankton (especially rotifers) will probably decline as phytoplankton biomass decreases. Close positive correlations between phytoplankton and microzooplankton parameters have been found in the tidal fresh zone (Alden and Sellner 1996) which suggests that microzooplankton are controlled in part by their food supplies ("bottom-up" control). Historical monitoring data suggests that rotifers did not become common until eutrophication had built up chlorophyll levels (see above).

Despite the apparent abundance of phytoplankton food, eutrophication may be degrading food quality and thus impacting growth rates and productivity of the primary consumers. Low light levels (high turbidity), excess nutrients and toxic pollutants all favor the dominance of small-sized cells ("picoplankton") and cyanobacteria (bluegreen algae) in phytoplankton assemblages and encourage the development of a microbial food web (Sellner 1988). "Picoplankton" are frequently too small for larger-bodied mesozooplankton and benthic macroinvertebrates to consume; bluegreen algae can become toxic and are generally not nutritious (Fulton and Paerl 1987). The CBP monitoring program has in fact documented a trend toward smaller phytoplankton average cell size as turbidities have increased since 1985 (Lacouture et al 1999). In the 1984 - 1998 winter-spring-summer-fall seasons, bluegreen algae comprised averages of 5.8% - 7.3% - 40.0% - 14.8% of the phytoplankton biomass in the tidal fresh river, 3.6% - 5.8% - 13.4% - 7.3% in the transition zone, and less than 1% to 3% in the mesohaline zone (from "Primary Producers" above). Bluegreen algae summer proportions in the tidal river and oligohaline were much higher during the 1970s and early 1980s (Lear and Smith 1976).

Recently, the CBP monitoring programs have found a very significant increasing trend in a small, filamentous bluegreen algae in the mesohaline zone. This may be impacting zooplankton food quality in the mesohaline zone.

Historic and present-day species compositions of the zooplankton and benthos assemblages in tidal fresh and oligohaline waters tend to confirm the thesis that the presence of bluegreen algae degrades food quality of phytoplankton. The herbivorous species that have dominated the zooplankton assemblages in the last three decades (see above) are species with known behaviors or mechanisms allowing them to cope with abundant toxic or non-nutritious bluegreen algal cells in their food supply. For example, *Brachionus calyciflorous* (rotifer) and *Bosmina longirostris* (cladoceran) are resistent to *Microcystis aeruginosa* (bluegreen) toxins and have some ability to utilize bluegreen algae biomass as food; most copepods, including *Eurytemora affinis*, actively avoid bluegreen cells and can selectively feed on the microzooplankton associated with bluegreen blooms; some cladocerans such as *Diaphanosoma* and *Moina* are physically incapable of ingesting colonial forms of the common bluegreen species and avoid eating them by default (Fulton and Paerl 1987, Fulton 1988a, 1988b, 1991). Taxa that cannot employ these behaviors or mechanisms such as *Daphnia* suffer reduced growth and eventual mortality in the presence of bluegreens.

Similarly, several of the dominant benthic taxa in the tidal fresh and oligohaline zones are suspension feeders, a feeding guild that can frequently avoid ingesting undesirable particles or can discharge them undigested as pseudofeces, i.e. *Corbicula fluminea*, *Rangia cuneata*, *Musculium transversum* (Lippson et al 1979). Others are deep deposit feeders which can selectively feed on desirable particles in the sediments.

Productivity

Stressful habitat conditions and/or poor food quality are common causes of below-potential growth rates in animals (e.g. Day et al 1989). While nutrient reductions have alleviated a few eutrophication-related problems in the Potomac estuary (see above), other eutrophication-related problems such as poor food quality could be affecting productivity of the zooplankton and benthos. This would indicate a need to further restore habitat conditions and phytoplankton food quality.

Mesozooplankton and benthos species productivities have not been measured in the Potomac estuary in recent years but many are suspected of being low, and thus partly responsible for the low abundances of certain taxa. Productivity is the rate at which new biomass is produced through reproduction and growth, and it is a function of both the initial abundance of a population and the instantaneous growth rates of its individuals. Stressful habitat conditions and/or poor food quality are frequent causes of below-potential growth rates. Hypoxic- anoxic summer conditions are clearly responsible for the extremely low benthic macroinvertebrate standing stock in deep waters of the lower estuary (Appendix G), and the inference is that in addition to the death rate being high, benthic productivity is low. High benthic macroinvertebrates abundances in the tidal river suggest productivity there is high, although the dominance of pollution-tolerant species such as *Corbicula* and tubifex worms may not be the most desirable species composition. The relatively low abundances of large-bodied mesozooplankton throughout the Potomac suggest that their productivities are currently low while the moderate to high abundances of rotifers suggest their productivities are high.

A 1977 study supports the thesis that mesozooplankton productivity may presently be low in the upper Potomac estuary while rotifer productivity is high. Heinle et al (1979) estimated the combined production rate for copepods, cladocera and rotifers was usually less than 100 mg dry weight m^{-3} day^{-1} between March and May, the peak zooplankton "bloom" period in the upper Potomac estuary. They concluded that rotifer production was very close to it maximum potential whereas production of the dominant copepod *Eurytemora affinis* was well below its potential and probably affected by poor food quality (rather than total phytoplankton abundance which was high, i.e. 4.4 - 61.1 µg chl a per liter). Feeding experiments (Fulton and Paerl 1987a, 1987b) later showed that *E. affinis* feeding behavior allows it to avoid ingesting non-nutritious and/or toxic cells associated with eutrophication (i.e. bluegreen algae which co-dominate the summer phytoplankton), but other impacts of eutrophication on food quality are possible (see below). Heinle's estimates of zooplankton production rates in 1977 were below the 1 g zooplankton dry weight m^{-3} day^{-1} estimated as being needed to support, for example, the 1830s zoo-planktivore populations in the upper Potomac estuary (see above).

Summer growth rates of the dominant copepod, *Acartia tonsa*, in the mesohaline region of the nearby Patuxent Estuary in 1964 were close to maximal values observed elsewhere in the laboratory and field (Heinle 1966). Conservative estimates of *Acartia tonsa* production for a 2 month summer period were 60 - 67 mg dry weight m^{-3} day^{-1}, or 1.61 - 1.78 pounds $acre^{-1}$ day^{-1}, for *Acartia tonsa* nauplii, copepodite and adult abundances averaging 61,270 m^{-3}, 14,800 m^{-3} and 2,670 m^{-3}, respectively. *Acartia* comprised 60% - 90% of the summer mesozooplankton at that time, thus total mesozooplankton productivity was somewhat higher. We can postulate that summer *Acartia tonsa* growth rates in the lower Potomac Estuary at that time were comparable to Patuxent River summer growth rates, and since abundances were similar, total mesozooplankton productivity was equally high. Present-day levels of total mesozooplankton productivity have declined from the presumed 1970s Potomac levels. Even if maximal growth rates are still found, the lower mesozooplankton abundances (see above) very likely preclude high production levels.

Actual productivity measurements for benthic macroinvertebrates, mesozooplankton and microzooplankton during spring and summer, the expected peak production periods in the upper and lower estuary, would help determine if a) growth rates are maximal and abundances are low because of predation pressure, flow or some other natural loss function, or b) growth rates are in fact below the population's potential.

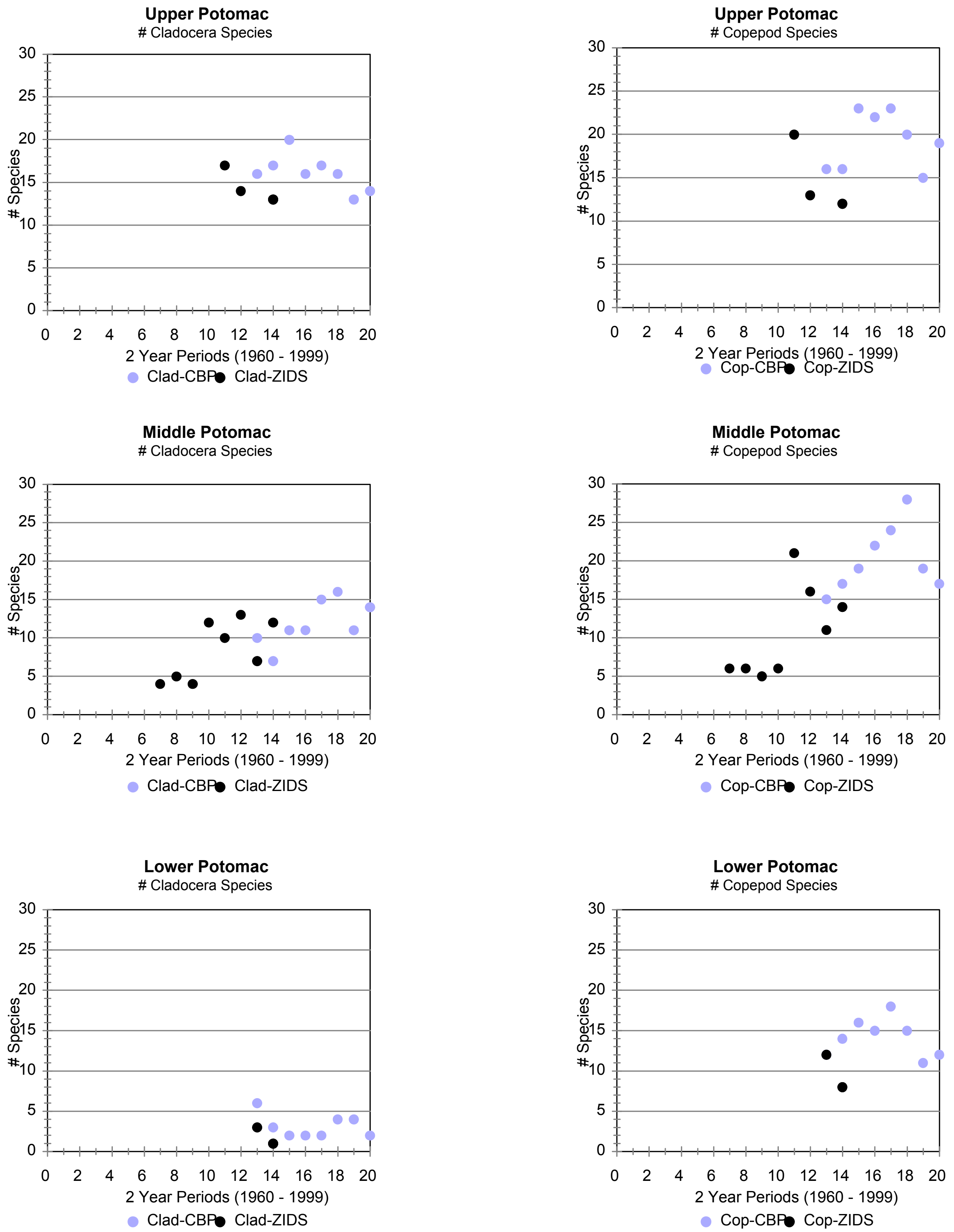


Figure 50. Species richness of cladoceran ("Clad") and copepod ("Cop") taxa in the Potomac Estuary. CBP: Chesapeake Bay Program monitoring data for the Potomac Estuary, collected by Versar, Inc. ZIDS: Zooplankton Indicator Database of non-CBP monitoring data compiled by ICPRB Period 1 is 1960-1961 .... Period 6 is 1970-1971 .... Period 11 is 1980-1981 ..... Period 16 is 1990-1991...

DISCUSSION AND CONCLUSIONS

The current status of the Potomac ecosystem, expressed as measures (indicators) of its water quality and biological communities, suggests a very poor level of ecosystem integrity relative to what it must have been at the time of European colonization. The lack of multi-faceted monitoring information prior to the 1970s will make it impossible to retrospectively determine which anthropogenic impacts to the habitat and food web were primarily responsible: the extreme losses to the upper trophic levels caused by overfishing, the enormous physical habitat losses when SAV and oyster reefs receded, or the profound changes to water quality caused by eutrophication and toxic chemical pollution. It is reasonable to presume that each of these impacts moved the ecosystem past critical thresholds and eventually began to synergistically reenforce the effects of the others, hastening the degradation of the Potomac estuary. The question now is whether the 40% nutrient reduction policies of the Chesapeake Bay Program can successfully return the Potomac ecosystem to a desirable status which benefits living resources, and if this can be done without the restoration of significant food web consumer populations.

General conclusions from this report that confirm and support the results of other Chesapeake Bay Program analyses include:

- Ambient nutrient concentrations in waters of the upper and middle Potomac estuary are still above thresholds that would limit phytoplankton growth.

- Variability is an expected and natural feature of estuaries, and annual variability in freshwater flows can temporarily negate and even countervail nutrient reduction efforts in the Potomac basin.

- Reductions in point and non-point source loadings of nitrogen and phosphorus may take years and even decades to be expressed as reductions in ambient nitrogen and phosphorus concentrations.

- Populations high in the food web take more time to recover because of their longer life cycles, and their restoration may require more effort than simply improving habitat conditions.

- Living resources may be crucial partners in accomplishing man's efforts to restore good water quality and habitat conditions.

Tidal Fresh River (Upper Estuary)

Nutrient load reductions to-date have diminished eutrophication impacts in the Potomac tidal freshwater river segment but habitat conditions and food quality have not improved enough to restore many ecosystem functions. Phosphorus and chlorophyll declined significantly since 1965 despite recent flow-related increases, and dissolved oxygen has improved. However, high total suspended solids still limit phytoplankton and submerged aquatic vegetation (SAV) photosynthesis. While the high concentrations of nitrogen are not primarily responsible for the abundant phytoplankton populations in this salinity zone (i.e. nitrogen is not a limiting nutrient), nitrogen is transported downstream where it eventually fuels the development of anoxia in the lower estuary. Summer cyanobacteria blooms still occur frequently but the fact that the extent and duration of recent blooms, including the 1999 bloom (W. Butler, pers. comm.; R. Lacouture, pers. com.), were not as severe as blooms prior to 1985 under similar conditions suggests that the pool of phosphorus stored in the sediments is diminishing. Zooplankton populations in spring are frequently insufficient for normal growth of larval anadromous fish.

Poor food quality caused by high proportions of bluegreen algae in the phytoplankton is perhaps the biggest deterrent to a restored zooplankton population in the upper estuary. Habitat conditions, including low light levels and high nutrients, still favor development of summer bluegreen algae blooms. Better food quality would encourage larger, healthier populations of mesozooplankton and possibly benthic macroinvertebrates. This in turn would increase grazing pressure on the phytoplankton and movement of carbon through the food web to desired, large-bodied consumers.

Mesozooplankton and benthic macroinvertebrate grazing pressure on algal populations is presently not very strong in tidal fresh waters and does not exert significant control on the phytoplankton population. Grazing estimates indicate present-day populations of mesozooplankton filter <50% of the river water volume per day in summer, and often <25% (Buchanan and Schloss 1983, Sellner and Jacobs undated report, Jones and Kelso 1988, Hyde 1989). Grazing pressure is higher in spring, sometimes briefly reaching 200% per day (Hyde 1989). Although significant, these rates are not sufficient to control phytoplankton populations, as witnessed by the fact that there are no close inverse correlations between phytoplankton and mesozooplankton parameters (Sellner and Alden 1993). Benthic macroinvertebrate grazing pressure was significant enough to seasonally affect the phytoplankton populations when *Corbicula* were abundant in the early 1980s (Cohen et al 1984), but has since diminished as *Corbicula* abundances declined. Overall, mesozooplankton and benthic macroinvertebrates presently are not influential consumers of the phytoplankton in the tidal fresh or oligohaline Potomac. Microzooplankton also have the potential to exert a substantial grazing pressure on phytoplankton (Sellner and Jacobs, undated report). Furthermore, they are important grazers of the bacterioplankton (Day et al 1989) which presently have summer abundances that reach 30 - 50 million cells per milliliter in the Potomac estuary (R. Jonas, pers. comm.). However, in the upper Potomac estuary during the 1984 - 1991 time period, large microzooplankton (>44 - 202µ) apparently only cleared on average 6% - 17.2% of the water column per day (Sellner and Jacobs, undated report). Estimates of clearance rates for the extremely abundant ciliate populations were not possible because these were not monitored at the time. Enhancing populations of mesozooplankton and benthic macroinvertebrates will be critical in restoring ecosystem function in the upper tributary because a) the grazing pressure on phytoplankton will increase, and b) larger-bodied invertebrates in the plankton and benthos are better fish food.

The challenge of improving light attenuation in the upper estuary to levels that encourage better phytoplankton food quality will involve reducing internal and external loadings of inorganic sediments as well as nutrient loadings. Shading by high phytoplankton populations has been blamed as a major cause of low light penetration in Chesapeake systems (e.g. Batiuk et al 1992). However, the very large decreases in chl coupled with the lack of trends in secchi depth since 1965 suggests chlorophyll concentrations are not the primary factor in light penetration. Analyses by R. C. Jones (pers. comm.) suggest that inorganic particles make up ~ 60% of the total suspended solids whereas living algal cells only make up ~ 30% in the tidal fresh river. The proportions of living organic particles (especially phytoplankton and bacterioplankton cells), dead organic particles (detritus) and inorganic particles (silt, clay, etc.) in TSS vary with season and salinity zone and have been quantified by R. C. Jones (pers comm.) for the tidal fresh Potomac River off of Gunston Cove. In the Potomac tidal river in summer, presently one of the most turbid areas and seasons, total organic material makes up about 40% (range: 10% - 90%) of the TSS. Living algal cells can constitute up to 100% of the total organic material, but usually account for less. Thus, on average, living algal cells comprise only about 30% of the total suspended solids, dead organic material about 10% and inorganic particles about 60%.

Reducing suspended inorganic sediments is perhaps best achieved by encouraging the growth of submerged aquatic vegetation (SAV). Beds of SAV stablize shorelines and reduce erosion. While habitat requirements for native SAV are still substandard in the upper Potomac, the exotic species *Hydrilla* is apparently able to populate the region and, in turn, encourages the growth of native species.

Transition Zone (Middle Estuary)

Eutrophication of the oligohaline zone has reversed and this reach of the estuary may be poised for a recovery. Phosphorus and chlorophyll concentrations in the oligohaline zone have shown the largest declines in the estuary since 1965, despite recent increases caused by the 1990s high flow conditions. While present in high concentrations, nitrogen is not a limiting nutrient in this salinity zone and thus is not spurring phytoplankton growth. It continues be transported through the oligohaline zone with little impact and moves into the mesohaline segment where it has a large impact. Turbidity and secchi depth show no trends, but this should not be surprising because one turbidity maximum of the estuary falls in the oligohaline zone and suspended sediment concentrations are normally high. Oxygen levels occasionally dip below 4 - 5 mg liter^{-1} but generally meet living resource requirements. The exotic species, *Hydrilla*, migrated to this reach in 1989 and has since established a stable population which is encouraging the return of native SAV species. The species richness of both mesozooplankton and benthic macroinvertebrates has increased since the 1970s. Total abundances of benthos, mesozooplankton and microzooplankton have remained fairly stable. It appears as if the most severe eutrophication impacts have retreated upstream of the oligohaline zone and no longer spill directly into the mesohaline zone except in very unusual conditions (e.g. 1998-1999 drought). Similarly, the anoxia problem and the deteriorating water quality conditions of the lower estuary are not intruding upstream. Thus the oligohaline zone in effect now separates two regions with lingering problems.

Mesohaline Zone (Lower Estuary)

Mesohaline water quality, habitat conditions and the zooplankton and benthos communities in the lower Potomac estuary have not recovered significantly and in some cases have continued to degrade during the 1985 - 1998 time frame. Nitrogen and TSS have increased. Light penetration (secchi depth) and dissolved oxygen have declined slightly. The “improving” (declining) trend in chl should be viewed with scepticism considering the increasing trends in TSS and nitrogen and decreasing trend in secchi depth.

Improving trends in the upper and middle estuary segments suggest that all of the tidal river eutrophication problems of the 1960s no long spill directly into the mesohaline zone of the lower estuary. High nitrogen loads still spiral unutilized downstream and are a likely cause of the high algal concentrations and the development of summer anoxia in the lower estuary. Causes of the degrading TSS and secchi trends in the lower estuary, however, are probably internal to the mesohaline. This conclusion is supported by studies of sedimentation rates in the Potomac. Defries (1986) concluded from her analysis of sediment cores that initial European settlement apparently did not affect sedimentation rates near the mouth of the estuary, and even intensive construction activities starting in the 1950s did not affect sedimentation rates throughout much of the estuary. Subestuaries are depositional areas for much of the sediments (~75%) associated with accelerated erosion from land use, and most sedimentation impacts are local. Therefore, suspended sediments in the lower estuary are likely to have originated within the lower estuary through wind resuspension, bioperturbation or other natural mechanisms.

The biggest factor preventing the lower estuary from recovering may not be the high nitrogen loadings but rather the depauperate oyster reefs and the food web's inability to both remove sediments from the water column and direct phytoplankton biomass into pathways leading to higher trophic levels. Suspended particles in the water column, including inorganic sediments as well as detritus and living algal cells, were removed quickly by the historically large populations of oysters as well as benthic macroinvertebrate filter-feeders in the mesohaline. In the absence of large oyster and benthic populations, this important mechanism for removing sediments and algae is weak. Since a large proportion of the TSS is inorganic, the CBP strategy of indirectly reducing turbidity by reducing nutrients and, thus, phytoplankton populations is not effective or even successful in this portion of the river. Shoreline stabilization by SAV and oyster reef are more effective in reducing resuspension of sediments, and a better approach to improving water quality in the lower estuary might be enhanced SAV and oyster restoration efforts.

Nutrient Reductions

Phosphorus and raw sewage treatment efforts implemented prior to the Chesapeake Bay Program had already lowered phosphorus, ammonia and total organic carbon concentrations throughout the estuary and improved dissolved oxygen conditions in the upper estuary. Efforts since 1985 to reduce phosphorous and nitrogen loadings another 40% are continuing the downward phosphorus trends and stopping the upward nitrogen trends. The improvements are apparently benefitting SAV and benthic macroinvertebrates in the middle and upper estuary and reducing the intensity and extent of the bluegreen algal blooms despite recent hydrologic extremes. However, the improvements to-date have not significantly reduced the bluegreen algae component of the phytoplankton community, or stabilized and maintained a diverse SAV community, or encouraged production of sufficient zooplankton food for anadromous fish larvae in the upper estuary. Furthermore, the improvements apparently had little influence on biological trends in the lower estuary.

Continuation of on-going efforts to reduce nitrogen loadings and maintain low phosphorus loadings should extend the improving water quality trends to the lower estuary. Excess nitrogen is presently shunted through the tidal fresh and oligohaline zones because it is poorly utilized by the primarily light-limited phytoplankton populations. Thus, large nitrogen loads still exported from the upper estuary to the lower estuary. When these loads decline, algal growth in the lower estuary can be expected to abate and dissolved oxygen improve.

Maintaining low nutrient loadings will prove more and more difficult as the basin's human population - and human water consumption - continues to grow rapidly. Washington metropolitan wastewater flows have increased more than 70% in the past 30 years (Appendix A). More urban development will tend to increase the flashiness and erosion strength of Potomac tributaries high flows, and heighten sediment and nutrient loads. In the future, the present level of effort to reduce nutrients will need to increase simply to maintain the water quality improvements that have been gained.

Zooplankton and Benthos Predators

Will the 40% nutrient reduction policy of the Chesapeake Bay Program successfully return Potomac water quality to a desirable status and benefit living resources? Although there is no sure answer to this question, this report results suggest that a 40% nutrient reduction strategy will not be sufficient to restore the Potomac estuary. If high level consumers remain absent or sparse, phytoplankton biomass in the still highly productive Potomac estuary will continue to move into "microbial loop" pathways (bacteria-protozoans-rotifers consumers), regardless of how much nutrient inputs are reduced. These pathways will carry on the present trends in poor water clarity and low dissolved oxygen, further exacerbating poor habitat conditions for fish, oysters and SAV as well as for their benthic and planktonic prey.

Natural predation is one of several possible loss functions that counter population gain functions or production (growth, reproduction, immigration). The balance of losses and gains results in the observed "standing stock" or population abundance. Declines of important finfish species and the oyster reefs in the 19^{th} century and early 20^{th} century reduced or removed food web links to higher trophic levels, making the Potomac ecosystem dependent on a less diverse assemblage of large-bodied consumers and tending to redirect primary production into other pathways. Biomass produced by the algal community, which by this time was affected by intensifying eutrophication in the upper estuary, accumulated in lower trophic levels or washed downstream unused to sink and decompose. This redirected primary production boosted the Potomac's bacterial and protozoan fauna to their present day extremely high levels. Long-lasting increases and complete recoveries have not occurred in most of the Potomac fisheries populations since water quality improvements were initiated by the Clean Water Act of 1972 and restoration / conservation efforts began in the 1980s and 1990s. Recruitment in anadromous species responded positively to several high flow years in the 1990s but, with the possible exception of striped bass, have not produced recovered populations of these species in the Potomac. Bay anchovy and other non-commercial, resident species show a similar lack of improvement. Menhaden populations are declining baywide. One favored species has come back. The large mouth bass, a versatile, high level consumer (piscivore) in freshwater food webs, rebounded with the return of SAV habitat in the upper Potomac. However, an undesirable zooplankton predator, the ctenophores or comb jellies (principally *Mnemiopsis*), appears to be increasing in the lower estuary and its populations is a possible cause of the recent declines in the lower Potomac.

Restoring a healthy, diverse community of large-bodied, top level consumers to the Potomac food web is becoming critical to completing the restoration of a productive, well-functioning ecosystem that is desirable to man. Efforts to remove fish blockages, restore and manage certain finfish populations, rebuild oyster reefs and SAV beds, and reduce toxic inputs are already underway to varying degrees. The important role of these efforts in reestablishing ecosystem functions in the Potomac should be recognized and encouraged.

RECOMMENDATIONS

Efforts to control and reduce total organic carbon and phosphorus loadings, especially those from point sources, had significantly reduced ambient concentrations of these parameters in the Potomac Estuary before 1985, the start of the Chesapeake Bay Program 40% nutrient reduction goal. Since that time, efforts to further reduce phosphorus loadings have not significantly changed ambient concentrations in the water column. However, declines in the extent and intensity of the summer bluegreen algae blooms suggest phosphorus releases from sediments may be diminishing as the rate of accumulation slows. Recent efforts to reduce point and non-point sources of nitrogen may have begun to reverse the steady, century-long climb in ambient nitrogen concentrations. Further nutrient reductions should eventually replace light-limitation as the factor controlling phytoplankton productivity in the upper and middle estuary. They are also likely to improve phytoplankton species assemblages and food value, which should benefit the zooplankton and benthic macroinvertebrates.

Once 40% nutrient reductions in point and non-point freshwater inflows are achieved, will these concentrations be sufficient to restore a balanced, productive ecosystem with abundant living resources? Nutrient reductions to-date have not been sufficient to remedy the other, often mutually-reenforcing anthropogenic impacts to the estuary, nor should they be expected to restore a balanced, productive ecosystem with abundant living resources. This conclusion is not unexpected, given the magnitude of the anthropogenic impacts that have affected the estuary in the past in addition to nutrient enrichment (eutrophication). Land uses in the Potomac basin after European settlement increased the variability and "flashiness" of freshwater baseflow to the estuary. Sedimentation rates, especially in the tributaries, have increased significantly. Enormous losses in many large-bodied, long-lived consumers in the Potomac food web, including American shad, river herring and sturgeon, profoundly changed the food web in the 20th century. Major declines in two key "living" habitats, the SAV and oyster reef communities, greatly altered the structural habitats available in the estuary. Exotic species were introduced and are now an integral part of the Potomac ecosystem. Bioaccumulation of toxic chemicals has become a persistent problem in the upper two thirds of the estuary and in the Anacostia tributary. Damage done to the Potomac ecosystem structure and function by these other impacts needs to be alleviated or corrected before a balanced, productive system with abundant living resources can be restored.

Thus, management strategies for the Potomac estuary need to become both more holistic and more region specific. For example, halting degradation of the lower estuary may require reducing nitrogen loadings from upstream *and* restoring oyster populations for the purpose of filtering sediments and algae from the water. Actual restoration of the lower estuary living resources and the food web they depend on may require additional management actions. Furthering the improvements gained to-date in the upper estuary may require reducing *in*organic sediment loadings for the purpose of restoring the ecologically important SAV beds.

Several questions were raised by this project effort that could best be answered with further research and data analysis efforts. These efforts could potentially direct management actions in more effective ways. The following recommendations address these issues

- Explore why chlorophyll is increasing in the light-limited upper and middle Potomac estuary.

- Investigate the role of the dissolved nitrogen and phosphorus ratio (DIN/DIP) in phytoplankton growth and the change in the relative importance of fall-line and wastewater treatment plant loads as biological nutrient removal is implemented.

- Investigate mesozooplankton rates of production to determine if a) growth rates are maximal and abundances are low because of predation pressure, flow or some other natural loss function, or b) growth rates are in fact below the population's potential due to stressful habitat conditions and/or poor food quality. Determine what factors limit microzooplankton production rates and what factors control their total abundance.

- In conjunction with other nutrient reduction strategies, what population abundances of oysters are needed to reduce ambient sediment and nitrogen concentrations in the lower Potomac estuary by up to 40% of 1985 levels?

- Explore the potential role of restored oyster beds in destabilizing stratification and inhibiting summer anoxia development in the lower estuary.

- Explore innovative and environment-friendly bank stabilization techniques which allow the migration of wetlands as sea level rises.

Management strategies since the 1987 Bay Agreement have emphasize nutrient reduction strategies while recognizing the importance of living resources conservation and restoration in achieving good water quality (e.g. SAV restoration goal, ecologically valuable species strategy, oyster reef restoration goal). To completely restore the Potomac estuary to a productive well-functioning ecosystem, we may need to implement additional measures. They include:

- Continue to reduce nitrogen loadings to the estuary.

- Reduce runoff of inorganic sediments.

- Further restore two important "living" habitats, SAV beds in the upper estuary and oyster reefs in the lower estuary, in order to reduce inorganic suspended sediments.

- Restore and/or protect key finfish top predators and mid-level prey species.

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APPENDIX A

WATER QUALITY SECTION OF THE POTOMAC INTEGRATED ANALYSIS REPORT

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WATER QUALITY SECTION

OF THE

POTOMAC INTEGRATED ANALYSIS REPORT

December 1998

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EXECUTIVE SUMMARY

The Potomac River, one of the major estuaries of the Chesapeake Bay in Maryland, drains 14,670 square miles of diverse forested, agricultural, and highly urbanized land in the coastal plain, Piedmont, and Blue Ridge provinces of Maryland, Virginia, and West Virginia. The estuarine portion of the river begins near the western boundary of the District of Columbia and continues 113 miles across the coastal plain. The importance of the Potomac River in terms of the health of the Bay is underscored by the fact that the Potomac contributes 18% of the fresh water flow to the Bay (second to the Susquehanna River), 28% of the total nitrogen load and 33% of the total phosphorus load from the non-tidal part of the Bay (USGS, 1998).

This report describes short-term (1985-1996) and long-term (mid-1970s-1996) trends in the water quality of the tidal Potomac River. The Seasonal Kendall test of monotonic trend was used on observed and flow-adjusted data if the number of censored flow-adjusted data did not exceed 5%. In cases where the number of censored data exceeded 5%, trends in flow-adjusted data were analyzed using the Statistical Analysis System (SAS) LIFEREG procedure, an implementation of Tobit regression (Tobin, 1958). Data were adjusted for flow to control for any statistically significant relationship with flow. Trends in observed data were considered significant if they were detected at the 99% confidence level ($p \leq 0.01$) and at the 95% level ($p \leq 0.05$) for flow adjusted trends.

Trends were analyzed for the following water quality indicators: total phosphorus, orthophosphate, total nitrogen, dissolved inorganic nitrogen, nitrite plus nitrate, ammonium, total suspended solids, Secchi depth, active chlorophyll *a*, dissolved oxygen, and total alkalinity. In addition to analyzing the observed data for annual trends (flow-adjusted data were only analyzed for annual trend), the following seasonal trends were analyzed: March-May, April-June, June-September, July-September, April-October, October-December, March-November. Seasonal trends were analyzed to allow comparisons to living resources data.

Trends were analyzed at four mainstem tidal fresh stations (XFB2470, XFB1433, XEA6596, and XEA1840), three oligohaline zone stations (XDA4283, XDA1177, and XDB3321), and three mesohaline zone stations (XDC1706, MLE2.2, and MLE2.3). In addition, trends were analyzed at four stations located in tidal creeks (XFB1986, PIS0033, MAT0016, and MAT0078) and two stations in Gunston Cove (stations 7 and 9). Annual trend results for the water quality indicators or primary concern are described below.

Surface Non-flow Adjusted Trends (1985-1996) and (1986-1996)

There were no significant trends for either TP or PO_4F at any Potomac mainstem station. Significant decreases of 20% and 13% were detected for TN at two tidal fresh stations. TN increased by 32% at XDB3321 (sampling stopped at this station in 1990). TN also increase at one mesohaline zone station (MLE2.3) by 15%. DIN increased by 69% at XDB3321. Significant increases in TSS were detected at several mainstem stations. TSS increased by 38%

at one tidal fresh station. Increases of 51% and 45% were observed at two oligohaline zone stations. Dramatic increases of 101% and 87% were detected in TSS at two mesohaline zone stations. Although TSS increased at a number of mainstem stations, only one significant decrease in Secchi depth (19%) was observed in the mesohaline zone. CHAA increased at only one station, XFB2470 in the tidal fresh zone, by 121%. Conflicting trends were observed in DO. DO increased by 18% and 10% at two tidal fresh stations and decreased at one mesohaline zone station by 8%. The surface non-flow adjusted trends described above are presented in Figures ES-1a and ES-1b.

Piscataway Creek, Matawoman Creek, and Gunston Cove

TP decreased by 42% at one Gunston Cove station, no significant TP trends were observed in the Potomac tidal creeks. PO_4F increased by 179% at MAT0016 on Mattawoman Creek, no other significant PO_4W trends were observed on the non-mainstem stations. Significant decreases in TN ranging from15% to 27% were detected at all four tidal creek stations. TN decreased by 35% at one Gunston Cove station (GUN07). DIN decreased by 26% and 30% at the Piscataway Creek stations and by 39% and 26% at the Gunston Cove stations. A significant increase (60%) in TSS was observed at one station located in Piscataway Creek. Secchi depth increased by 60% at GUN07. CHAA increased by 140% at one Piscataway Creek station and by 110% at GUN09. Conflicting trends in DO were observed in Piscataway Creek, where the upper station had a 10% decrease and the lower station had a 20% increase. DO increased by 10% at GUN09. Surface non-flow adjusted trends for Piscataway and Matawoman Creeks and Gunston Cove are presented in Figures ES-1c and ES-1d.

Bottom Non-flow Adjusted Trends (1985-1996) and (1986-1996)

No significant TP or PO_4F trends were observed on the Potomac mainstem. Decreases in TN ranged from 13% to 16% in the tidal fresh zone. No other significant TN trends were observed. Only one DIN trend, an increase of 72%, was observed along the Potomac mainstem at XDB3321. TSS increased by 120% at MLE2.2. Three significant increases were detected in CHAA. CHAA increased by 132% and 105% at two tidal fresh stations and by 61% at one oligohaline zone station. Bottom DO increased by 13% at XFB2470 and by 16% at XDC1706. Bottom layer non-flow adjusted trends for the Potomac mainstem stations are presented in Figures ES-2a and ES-2b.

Piscataway Creek, Matawoman Creek, and Gunston Cove

Bottom layer samples are not collected at the Potomac tidal creeks. TP decreased by 36% at GUN07. TN and DIN also decreased at GUN07 by 35% and 38%, respectively. DIN decreased by 27% at GUN09. Bottom DO increased by 20% at GUN09. Bottom layer non-flow adjusted trends for Piscataway and Matawoman Creeks and Gunston Cove are presented in Figures ES-2c and ES-2d.

Surface Flow Adjusted Trends (1985-1996)

No significant TP or PO_4F trends were detected along the mainstem stations. Decreases in TN were observed in all three salinity zones. TN decreases in the tidal fresh zone ranged from 9% to 23%. TN decreased by 22% and 12% at two oligohaline zone stations and by 19% at one mesohaline zone station. Decreases in DIN were also observed at all three salinity zones along the mainstem. DIN decreased by 12% and 21% at two tidal fresh stations, by 26% at one oligohaline zone station, and by 42% and 29% at two mesohaline zone stations. DIN increased by 50% at XDB3321, where sampling stopped in 1990.

Significant trends in TSS were only observed in the mesohaline zone, where increases of 60% and 80% were detected. Conflicting trends in Secchi depth were detected in the mesohaline zone. Secchi depth increased by 10% at the station where TSS increased by 80% and decreased by 20% at a station where the trend in TSS was not significant. Increasing trends in CHAA were observed only in the tidal fresh and oligohaline zones. CHAA increases in the tidal fresh ranged from 80% to 260%. CHAA increased by 60% and 30% at two oligohaline zone stations. One increasing and four decreasing trends were detected in DO in the Potomac mainstem. DO increased by 10% at one tidal fresh station and decreased by 10% at another. Decreases of 10% were also observed at one oligohaline zone station and one in the mesohaline. DO decreased by 20% at another station in the mesohaline zone. Surface flow adjusted trends for the Potomac mainstem stations are presented in Figures ES-3a and ES-3b.

Bottom Flow Adjusted Trends (1985-1996)

One increasing trend was observed for TP and one for PO_4F, both in the mesohaline zone. TP increased by 38% at MLE2.3 and PO_4F increased by 119% at MLE2.2. TN decreased by 12% and 16% at two tidal fresh stations and by 14% at one oligohaline zone station. TN increased by 23% at MLE2.2 in the mesohaline zone. Three significant trends were detected for DIN in the mainstem Potomac. A 32% decrease was observed at one oligohaline zone station and a 33% decrease was observed at one mesohaline zone station. A 22% increase in DIN was detected at MLE2.2 in the mesohaline zone.

TSS decreased by 60% at XDB3321 and increased by 100% at MLE2.2. CHAA increased at all four tidal fresh stations and at one station in the oligohaline zone. Tidal fresh increases ranged from 40% to 220%. The increase at the oligohaline zone station was 60%. Decreases in DO of 10% were observed at two tidal fresh stations and one oligohaline zone station. DO decreased by 20% at each of two stations in the mesohaline zone. Flow adjusted bottom layer trend for 1985-1996 are summarized in Figures ES-4a and ES-4b.

Surface Non-flow Adjusted Trends (mid-1970s-1996)

For the long-term trends, significant decreases in TP were detected in the tidal fresh and oligohaline zones. Decreases in the tidal fresh zone ranged from 27% to 39%. A 22% decrease in TP was observed at one oligohaline zone station. Significant trends in PO_4W were only detected in the tidal fresh zone where decreases ranged from 40% to 61%. Increasing trends in TN were observed in both the tidal fresh and oligohaline zones. Tidal fresh TN increases ranged from 19% to 38% for the four stations. TN increases of 19% and 35% were observed at two oligohaline zone stations. Significant increases ranging from 20% to 39% were detected at three tidal fresh stations for DIN. All three stations in the oligohaline zone had significant increasing DIN trends, which ranged from 24% to 43%. Significant increases in TSS were observed in all three salinity zones. TSS increased by 50% and 60% at two tidal fresh stations, by 70% and 30% at two oligohaline zone stations and by 30% at the one mesohaline zone station with long-term data available. No significant trends in Secchi depth were detected in spite of the increases in TSS. CHAA decreased by 30% at one tidal fresh station and by 20% and 30% at two stations in the oligohaline zone. DO increased by 10% at the upper-most tidal fresh station and decreased by 10% at the lower-most. DO decreased by 10% at the mesohaline zone station. Long-term surface trends for non-flow adjusted data are presented in Figures ES-5a and ES-5b.

Bottom Non-flow Adjusted Trends (mid-1970s-1996)

There were no significant bottom layer trends in TP. Decreases in PO_4W were observed at all four tidal fresh station and at two oligohaline zone stations. Decreases in the tidal fresh zone ranged from 37% to 57%. Decreases of 34% and 36% were detected at the two oligohaline zone stations. Trends in TN and DIN were not significant. A 90% increase in TSS was detected at XEA6596 in the tidal fresh zone. No other TSS trends were significant. A 20% increase in CHAA was detected at XFB2470 in the tidal fresh zone, which is not where the increasing TSS trend was observed. DO increased by 10% at one tidal fresh station and decrease by 10% at another. A 10% decrease in DO was also observed at one oligohaline zone station. The results of the long-term, non-flow adjusted bottom trends are presented in Figures ES-6a and ES-6b.

Surface Flow Adjusted Trends (mid-1970s-1996)

Significant decreases ranging from 31% to 39% were detected in TP at the four tidal fresh stations. TP decreased by 25% at two stations in the oligohaline zone and by 23% at the mesohaline zone station. Four significant trends were also detected in PO_4W in the tidal fresh zone. The PO_4W tidal fresh zone trends were estimated using the SAS LIFEREG procedure and a percent change calculation that differs from that used for most other trends. As a result, the decreases, which ranged from 49% to 66%, may not be strictly comparable to other trends that were estimated using Sen's slope. No significant PO_4W trends were detected in the oligohaline zone, although a decrease of 46% was detected in the mesohaline zone. Significant trends in TN were detected at all four tidal fresh stations and all three stations in the oligohaline zone. In the tidal fresh zone, TN increases ranged from 17% to 38%, while those in the oligohaline zone ranged from 22% to 46%. Increasing DIN trends were detected at all stations, including the mesohaline zone. Increases in the tidal fresh zone ranged from 20% to 40%, while those in the oligohaline zone ranged from 27% to 57%. DIN at the mesohaline zone station increased by 23%.

Three significant increases ranging from 40% to 60% were observed in the tidal fresh zone for TSS. Increasing TSS trends ranging from 30% to 80% were detected at the three oligohaline zone stations. The trend in TSS at the mesohaline zone station was not significant. Even though large increases in TSS were observed at almost all stations, no significant trends in Secchi depth were observed. Significant trends in CHAA were observed at only two stations, one in the tidal fresh zone and one in the oligohaline zone. In the tidal fresh zone, CHAA decreased by 30%, whereas CHAA decreased by 50% in the oligohaline zone. DO increased by 10% at one tidal fresh station and decreased by 10% and 20% at two others. DO decreased by 10% at one oligohaline zone station and decreased by 20% at the mesohaline zone station. The long-term flow adjusted surface trend results are presented in Figures ES-7a and ES-7b.

Bottom Flow Adjusted Trends (mid-1970s-1996)

A significant decreasing TP trend of 38% was detected in the tidal fresh zone. No other significant TP trends were observed. Decreasing PO_4W trends ranging from 48% to 66% were detected at all four tidal fresh zone stations. It should be noted that the PO_4W trend at the uppermost tidal fresh station was estimated using the Seasonal Kendall test and Sen's slope and that trends and slopes at the lower three stations were estimated using SAS LIFEREG. As a result, the percent changes across the four stations may not be strictly comparable. Decreasing PO_4W trends were also found at the oligohaline zone stations. Decreasing PO_4W trends in the oligohaline zone ranged from 25% to 34%. PO_4W trends at stations XDA4238 and XDB3321 were also estimated using SAS LIFEREG and may not be comparable to other percent changes. Only two significant trends in TN were detected, one in the tidal fresh zone, where TN increased by 24% and one in the oligohaline zone, where a 15% increase was observed. DIN also increased at one tidal fresh and one oligohaline zone station. DIN increased by 18% at the tidal fresh station and increased by 28% at the oligohaline zone station.

One significant trend was detected in TSS, an increase of 50% at XDC1706. CHAA increased at the upper two tidal fresh stations by 30% and 20%. CHAA decreased by 70% at XDB3321, where sampling stopped in 1990. Conflicting DO trends were observed in the tidal fresh, where an increase of 10% was detected at the upper station and a 20% decrease was detected at the lower station. DO decreased by 10% at two oligohaline zone stations and decreased by 20% at the mesohaline zone station. Long-term bottom layer flow adjusted trend results are presented in Figures ES-8a and ES-8b.

Wastewater Treatment Plant Loads and Flow

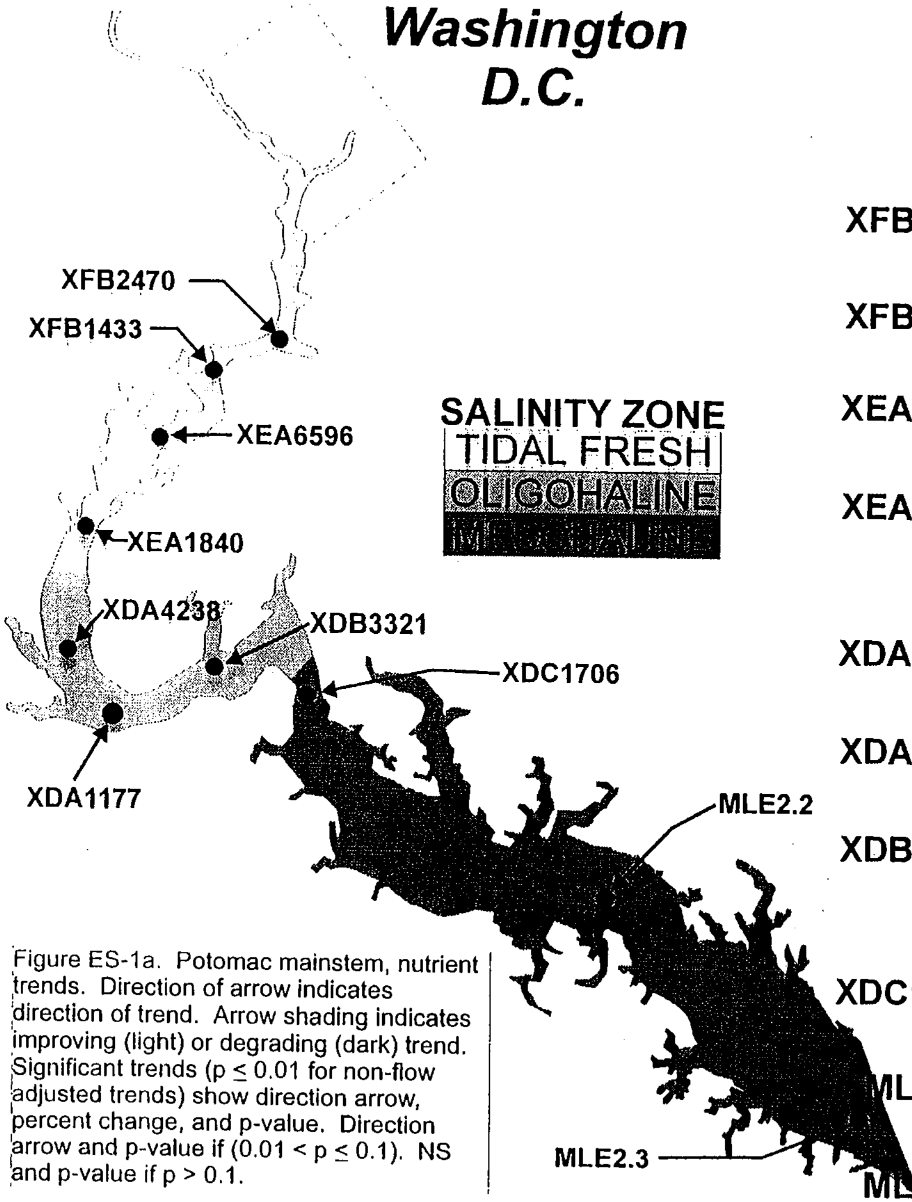
Based on a comparison of the average TN loads for 1970 to those in 1996, TN loads at the major WWTPS decreased by 8% even though flow increased by 70%. For the 1985 to 1996 time period, TN loads actually decreased by 6% despite a 21% increase in flow. TP loads decreased dramatically from 1970 to 1996, since efforts to reduce phosphorus were initially the focus of the Potomac restoration program. As a result of upgrades at Blue Plains and the phosphate detergent ban, TP loads decreased by 98% for the 1970-1996 time period. For 1985-1996, TP loads decreased by 44%. The data record for PO_4 at the WWTPs is less complete than it is for TN and TP. Loads for PO_4 decreased by 36% between 1985 and 1996.

River Input Station at Chain Bridge

Concentration and load data for the river input station at Chain Bridge are only available back to 1978, and trend results are only available for the 1985-1996 time period. The trend in flow adjusted TN concentrations for 1985-1996 was not significant. TP concentrations at the fall-line station decreased by 50% for 1985-1996. PO_4 concentrations for 1985-1996 decreased by 36%.

The trends described above for the tidal Potomac River were no doubt a result of the continuing improvements to wastewater treatment facilities, and bans on phosphate detergents implemented in 1985 (Maryland) and 1986 (Virginia and the District of Columbia). These improvements have led to substantially improved nutrient levels, dissolved oxygen, organic matter, and turbidity in the Potomac Estuary (MWCOG, 1989). In the report that follows, the results of long-term water quality monitoring programs are compiled and statistically analyzed to assess what affect these reductions are having on actual environmental conditions in the Potomac River.

Potomac River (1985-1996)



Twelve Year Surface Trends Non-flow Adjusted

Station	TP	PO4F	TN	DIN
XFB2470	↑ NS p=0.021	↑ NS p=0.072	↓ NS p=0.012	↓ NS p=0.043
XFB1433	NS p=0.240	NS p=0.100	↓ 20% p=0.002	↓ NS p=0.088
XEA6596	NS p=0.608	NS p=0.199	↓ 13% p=0.004	NS p=0.115
XEA1840	↑ NS p=0.063	NS p=0.376	↓ NS p=0.021	NS p=0.287
XDA4238	NS p=0.764	NS p=0.496	↓ NS p=0.085	NS p=0.232
XDA1177	↑ NS p=0.041	NS p=0.752	NS p=0.535	NS p=0.851
XDB3321*	NS p=0.143	NA	↑ 32% p=0.001	↑ 69% p=0.000
XDC1706	NS p=0.914	NS p=0.319	NS p=0.690	NS p=0.883
MLE2.2	NS p=0.745	NS p=0.379	NS p=0.129	NS p=0.492
MLE2.3	↑ NS p=0.011	NS p=0.274	↑ 15% p=0.010	NS p=0.511

Figure ES-1a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.01$ for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.01 < p \leq 0.1$). NS and p-value if $p > 0.1$.

NA - Data not available for this station.

PO4F trends start in October 1990.

*Station not sampled after September 1990.

Potomac River (1985-1996)

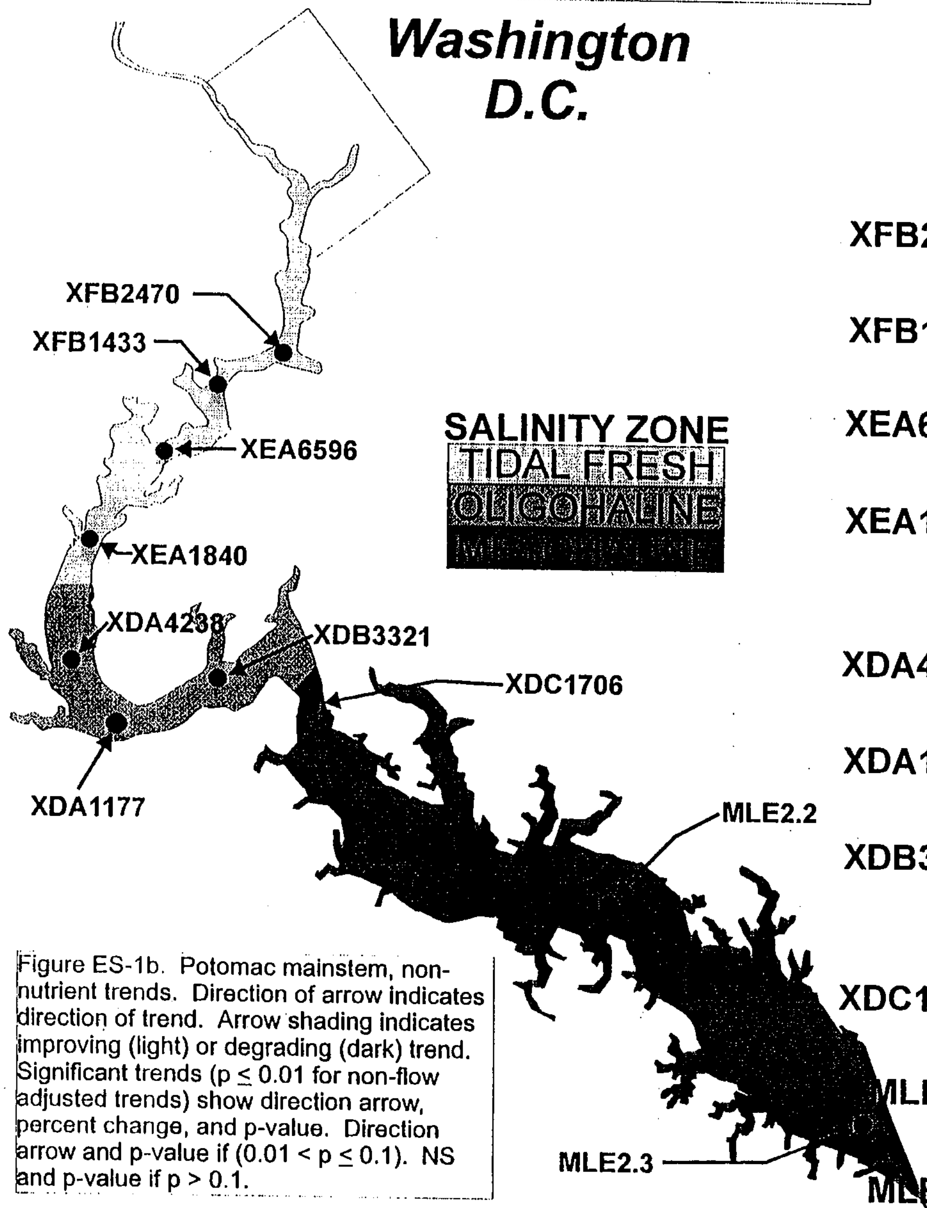


Figure ES-1b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.01$ for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.01 < p \leq 0.1$). NS and p-value if $p > 0.1$.

Twelve Year Surface Trends Non-flow Adjusted

Station	TSS	SECCHI	CHAA	DO_FLD
XFB2470	↑ NS p=0.019	NS p=0.152	↑ 121% p=0.000	↑ 18% p=0.000
XFB1433	↑ 38% p=0.006	NS p=0.862	↑ NS p=0.041	↑ 10% p=0.002
XEA6596	↑ NS p=0.043	NS p=0.719	NS p=0.670	↑ NS p=0.017
XEA1840	↑ NS p=0.021	↓ NS p=0.085	NS p=0.538	NS p=0.771
XDA4238	↑ 51% p=0.004	↓ NS p=0.061	↑ NS p=0.090	NS p=0.176
XDA1177	↑ 45% p=0.001	↓ NS p=0.018	NS p=0.144	NS p=0.384
XDB3321*	NS p=0.389	NS p=1.00	NS p=0.248	NS p=0.458
XDC1706	↑ 101% p=0.000	↓ NS p=0.084	NS p=0.408	NS p=0.677
MLE2.2	↑ 87% p=0.000	NS p=0.323	NS p=0.656	NS p=0.884
MLE2.3	NS p=0.769	↓ 19% p=0.006	NS p=0.309	↓ 8% p=0.005

*Station not sampled after September 1990.

Potomac River (1974-1996)

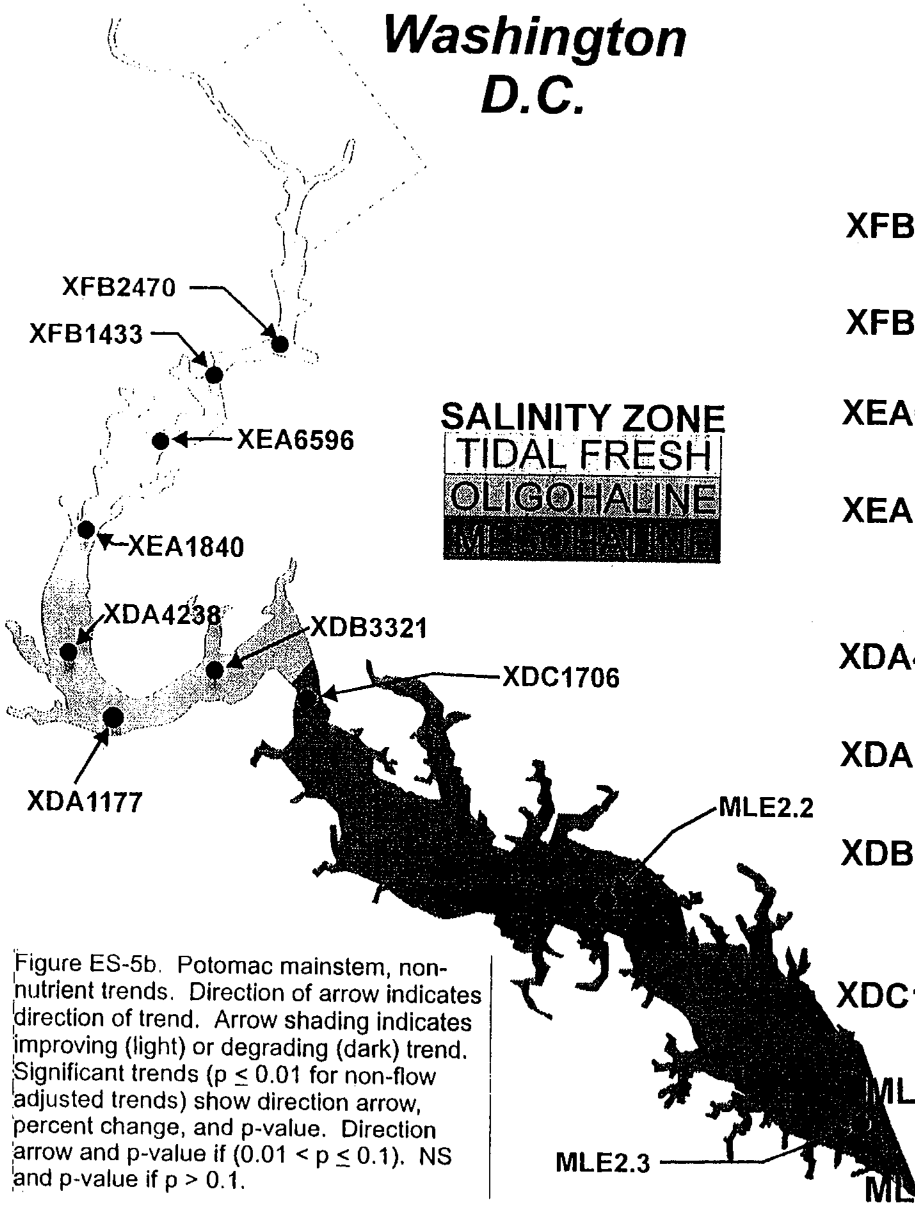


Figure ES-5b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

NA - Data not available for this station.

Twenty-five Year Surface Trends Non-flow Adjusted

	TSS	SECCHI	CHAA	DO_FLD
XFB2470	↑ 50% p=0.000	↓ NS p=0.043	↑ NS p=0.021	↑ 10% p=0.004
XFB1433	↑ 60% p=0.000	NS p=0.354	NS p=0.830	NS p=0.710
XEA6596	↑ NS p=0.020	NS p=0.876	↓ NS p=0.088	↓ NS p=0.033
XEA1840	NS p=0.351	NS p=0.368	↓ 30% p=0.000	↓ 10% p=0.000
XDA4238	↑ 70% p=0.003	NS p=0.218	↓ 20% p=0.006	NS p=0.192
XDA1177	↑ 30% p=0.010	NS p=0.226	NS p=0.136	↓ NS p=0.069
XDB3321*	NS p=0.140	NS p=0.475	↓ 30% p=0.006	↓ NS p=0.046
XDC1706	↑ 30% p=0.004	NS p=0.446	NS p=0.171	↓ 10% p=0.000
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

*Station not sampled after September 1990.

Potomac River (1974-1996)

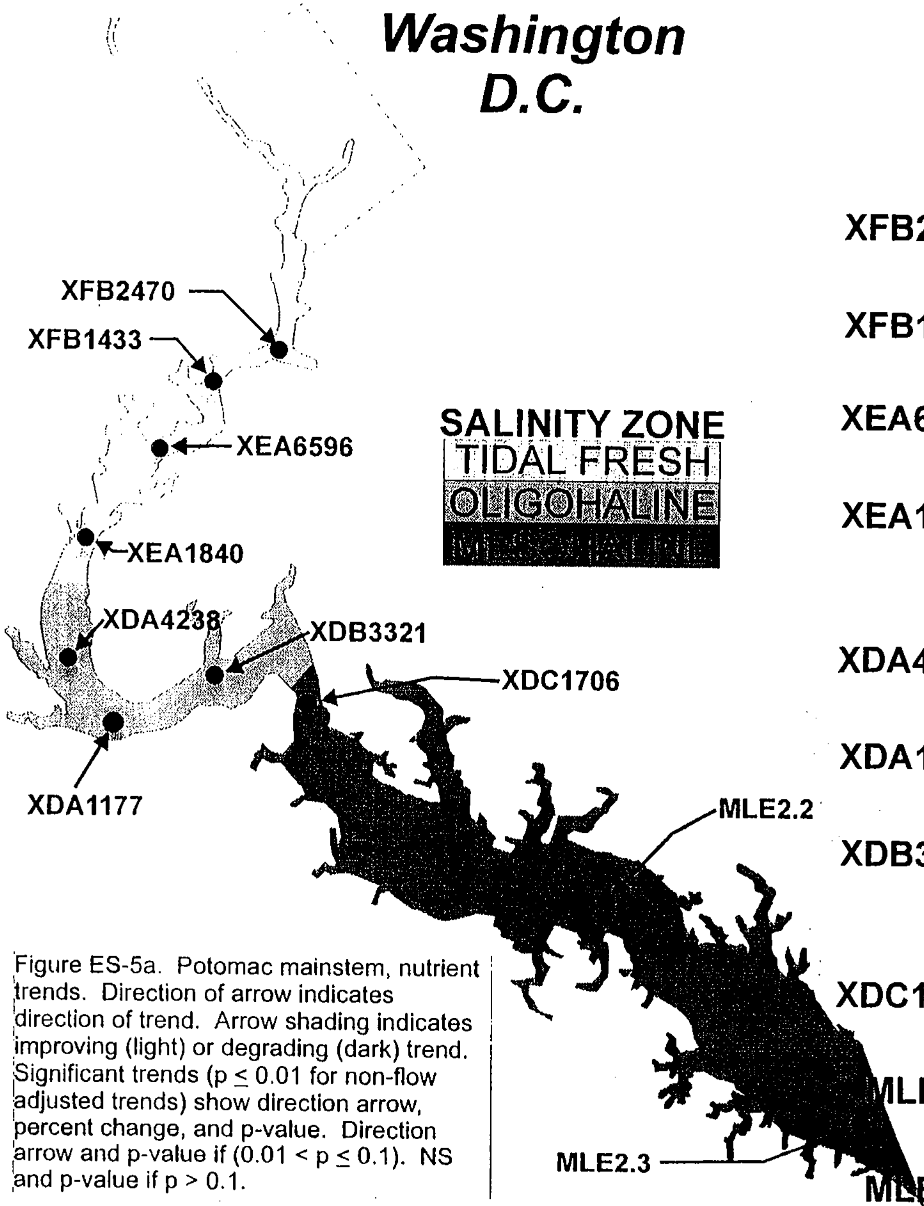


Figure ES-5a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.01$ for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.01 < p \leq 0.1$). NS and p-value if $p > 0.1$.

Twenty-five Year Surface Trends Non-flow Adjusted

	TP	PO4W	TN	DIN
XFB2470	↓ 27% p=0.000	↓ 58% p=0.000	↑ 19% p=0.001	↑ NS p=0.017
XFB1433	↓ 29% p=0.000	↓ 61% p=0.000	↑ 15% p=0.002	↑ 20% p=0.001
XEA6596	↓ 30% p=0.000	↓ 50% p=0.000	↑ 28% p=0.000	↑ 29% p=0.000
XEA1840	↓ 39% p=0.000	↓ 40% p=0.000	↑ 38% p=0.000	↑ 39% p=0.000
XDA4238	↓ 22% p=0.001	NS p=0.207	↑ NS p=0.021	↑ 24% p=0.004
XDA1177	↓ NS p=0.039	NS p=0.191	↑ 19% p=0.001	↑ 36% p=0.003
XDB3321*	NS p=0.107	NS p=0.105	↑ 35% p=0.000	↑ 43% p=0.000
XDC1706	↓ NS p=0.016	↓ NS p=0.052	NS p=0.380	↑ NS p=0.024
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

NA - Data not available for this station.

PO4W trends end in September 1990.

*Station not sampled after September 1990.

Potomac River (1985-1996)

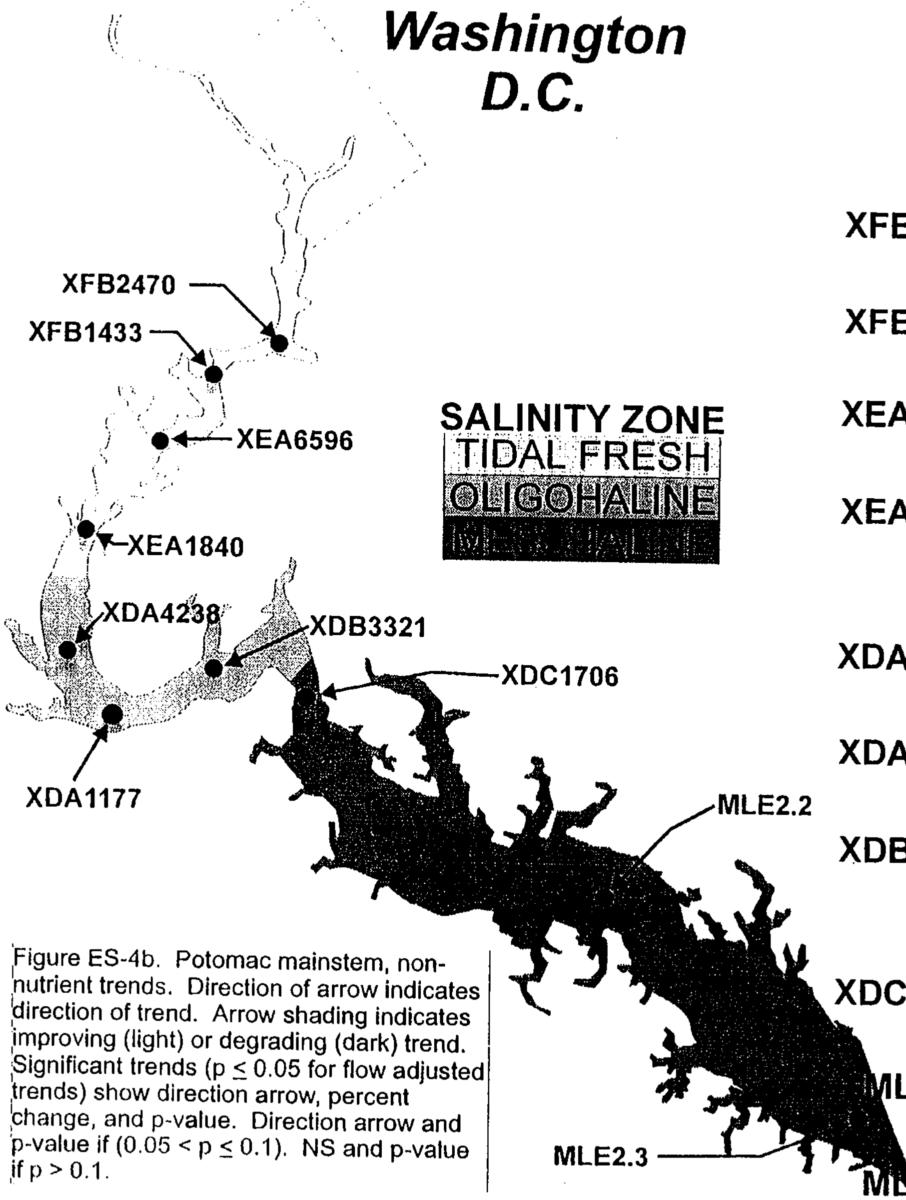


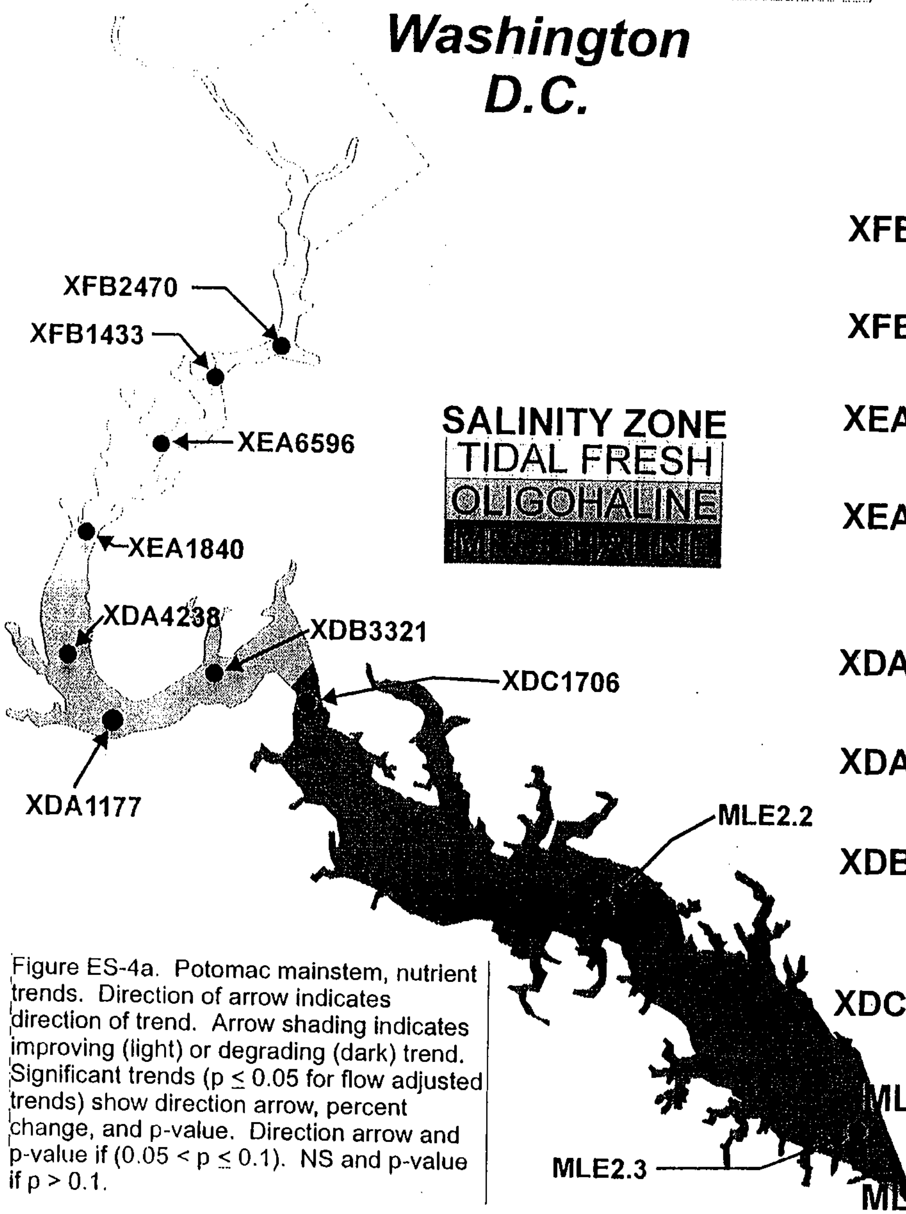
Figure ES-4b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

Twelve Year Bottom Trends Flow Adjusted

	TSS	CHAA	DO_FLD
XFB2470	NS p=0.140	↑ 220% p=0.000	NS p=0.818
XFB1433	NS p=0.491	↑ 180% p=0.003	NS p=0.723
XEA6596	NS p=0.866	↑ 90% p=0.009	↓ 10% p=0.030
XEA1840	NS p=0.433	↑ 40% p=0.023	↓ 10% p=0.000
XDA4238	NS p=0.367	↑ 60% p=0.006	↓ 10% p=0.001
XDA1177	NS p=0.650	↑ NS p=0.080	NS p=0.147
XDB3321*	↓ 60% p=0.046	NS p=0.151	NS p=0.262
XDC1706	NS p=0.319	NS p=0.865	NS p=0.934
MLE2.2	↑ 100% p=0.000	NS p=0.799	↓ 20% p=0.011
MLE2.3	↓ NS p=0.082	NS p=0.132	↓ 20% p=0.001

*Station not sampled after September 1990.

Potomac River (1985-1996)



Twelve Year Bottom Trends Flow Adjusted

	TP	PO4F	TN	DIN
XFB2470	NS p=0.798	NS p=1.000	↓ NS p=0.086	NS p=0.701
XFB1433	NS p=0.495	NS p=0.743	↓ 12% p=0.026	↓ NS p=0.070
XEA6596	NS p=0.949	NS p=0.709	NS p=0.113	↓ NS p=0.054
XEA1840	NS p=0.702	NS p=0.876	↓ 16% p=0.017	↓ NS p=0.058
XDA4238	NS p=0.497	NS p=0.269	↓ 14% p=0.005	↓ 32% p=0.001
XDA1177	NS p=0.344	NS p=0.495	NS p=0.287	NS p=0.348
XDB3321*	NS p=0.370	NA	NS p=0.187	↑ NS p=0.051
XDC1706	NS p=0.600	NS p=0.297	NS p=0.270	NS p=0.785
MLE2.2	NS p=0.545	↑ 119% p=0.002	↑ 23% p=0.024	↑ 22% p=0.037
MLE2.3	↑ 38% p=0.001	NS p=0.679	NS p=0.597	↓ 33% p=0.025

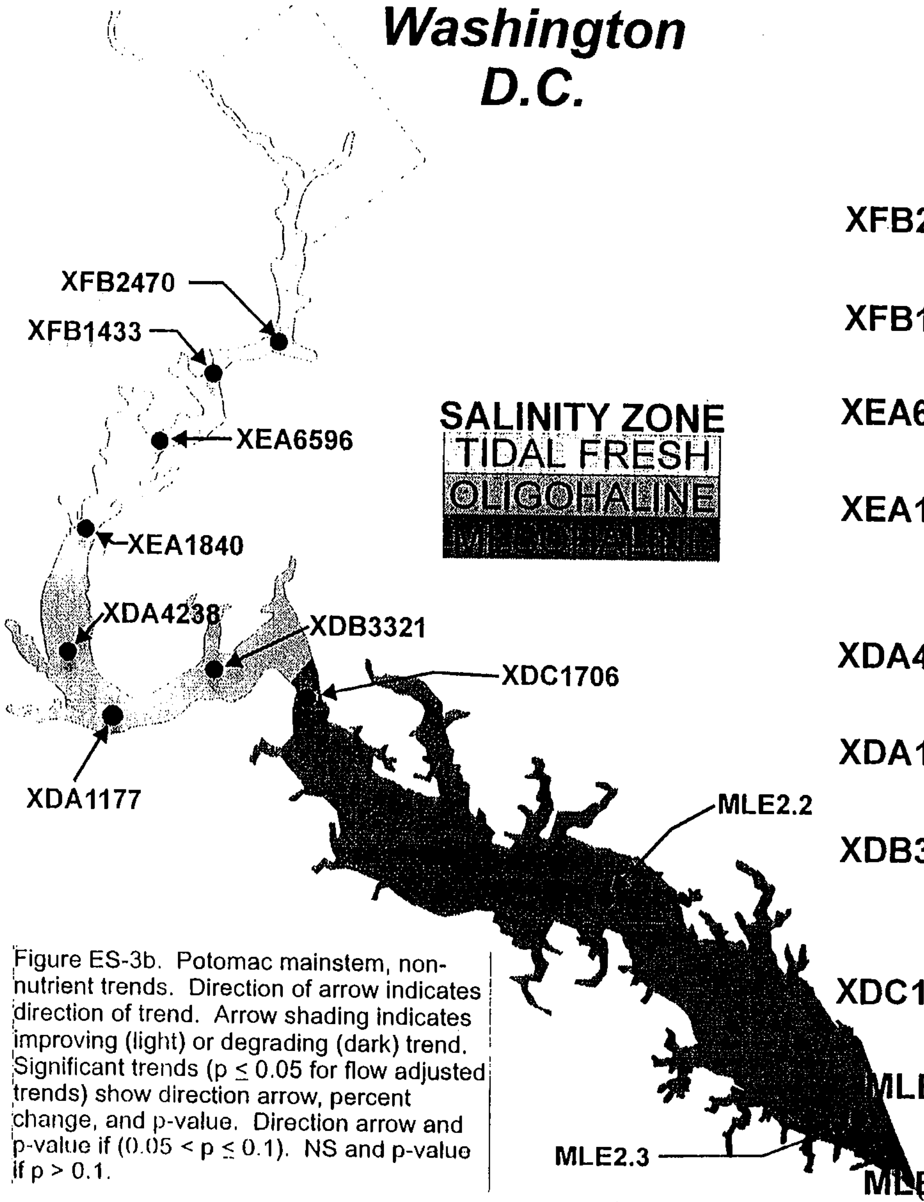
Figure ES-4a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

NA - Data not available for this station.

PO4F trends start in October 1990.

*Station not sampled after September 1990.

Potomac River (1985-1996)



Twelve Year Surface Trends Flow Adjusted

	TSS	SECCHI	CHAA	DO_FLD
XFB2470	NS p=0.818	NS p=0.851	↑ 260% p=0.000	↑ 10% p=0.026
XFB1433	NS p=0.786	↑ NS p=0.088	↑ 130% p=0.000	NS p=0.545
XEA6596	NS p=0.984	NS p=0.229	↑ 80% p=0.014	NS p=0.756
XEA1840	NS p=0.967	NS p=0.483	NS p=0.718	↓ 10% p=0.007
XDA4238	NS p=0.299	NS p=0.967	↑ 60% p=0.001	NS p=0.110
XDA1177	NS p=0.967	NS p=0.984	↑ 30% p=0.014	↓ 10% p=0.012
XDB3321*	NS p=0.540	↑ NS p=0.066	NS p=0.158	NS p=0.326
XDC1706	↑ 60% p=0.008	NS p=0.755	NS p=0.766	↓ 10% p=0.000
MLE2.2	↑ 80% p=0.000	↑ 10% p=0.025	NS p=0.832	NS p=0.100
MLE2.3	NS p=0.615	↓ 20% p=0.023	NS p=0.776	↓ 20% p=0.000

*Station not sampled after September 1990.

Figure ES-3b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

Potomac River (1985-1996)

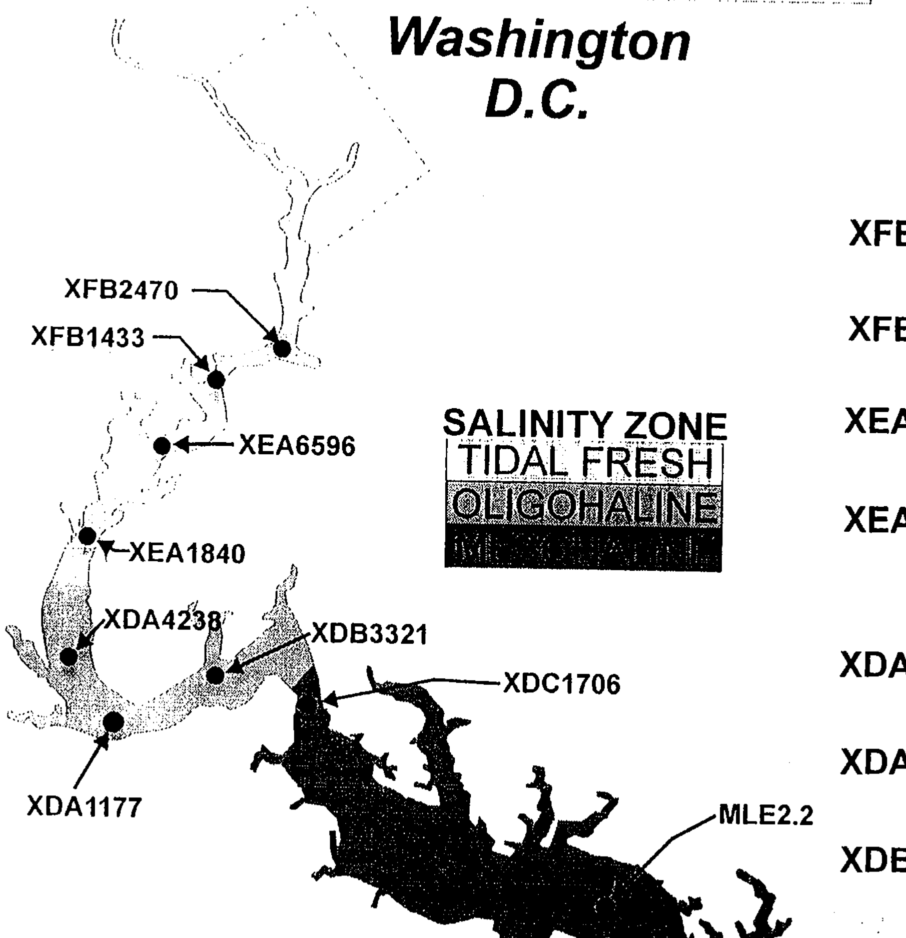


Figure ES-3a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

Twelve Year Surface Trends Flow Adjusted

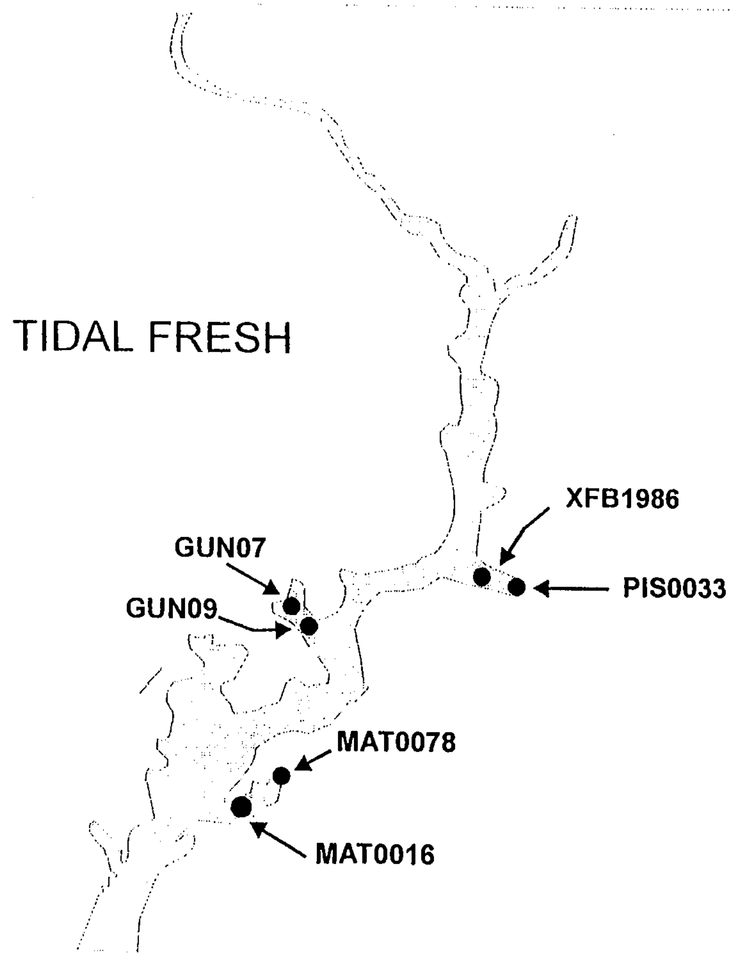
	TP	PO4F	TN	DIN
XFB2470	NS p=0.864	NS p=0.630	NS p=0.120	NS p=0.533
XFB1433	NS p=0.915	NS p=0.750	↓ 9% p=0.045	NS p=0.206
XEA6596	NS p=0.200	NS p=0.957	↓ 12% p=0.030	↓ 12% p=0.019
XEA1840	NS p=0.833	NS p=0.533	↓ 23% p=0.001	↓ 21% p=0.026
XDA4238	NS p=0.607	NS p=0.834	↓ 22% p=0.001	↓ 26% p=0.004
XDA1177	NS p=0.798	NS p=0.496	↓ 12% p=0.026	↓ NS p=0.057
XDB3321*	NS p=0.143	NA	NS p=0.141	↑ 50% p=0.001
XDC1706	NS p=0.879	NS p=0.114	↓ 19% p=0.018	↓ 42% p=0.017
MLE2.2	NS p=0.634	NS p=0.154	NS p=0.382	NS p=0.098
MLE2.3	NS p=0.235	NS p=0.665	NS p=0.555	↓ 29% p=0.029

NA - Data not available for this station.

PO4F trends start in October 1990.

*Station not sampled after September 1990.

Potomac River (1986-1996)



Eleven Year Bottom Trends Non-flow Adjusted

	TSS	CHAA	DO
PIS0033	NA	NA	NA
XFB1986	NA	NA	NS p=0.871
MAT0078	NA	NA	NA
MAT0016	NA	NA	NS p=0.121
GUN07	NS p=0.674	NA	↑ NS p=0.033
GUN09	NS p=0.297	NA	↑ 20% p=0.000

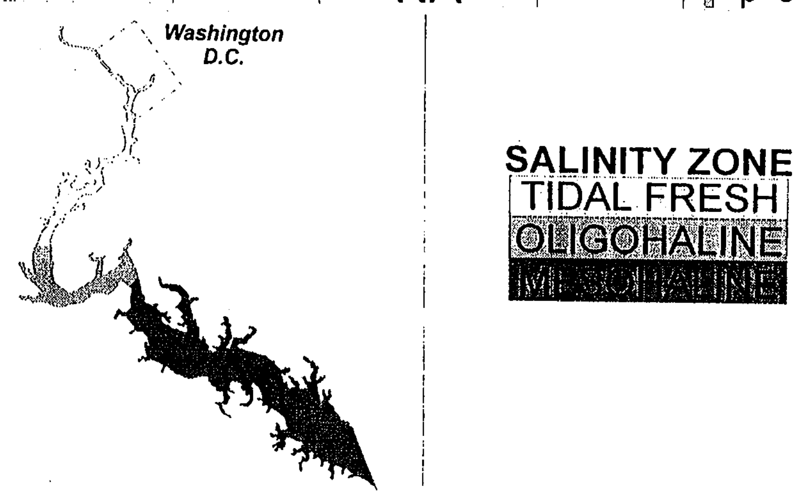


Figure ES-2d. Potomac tidal creeks and Gunston Cove nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.01$ for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.01 < p \leq 0.1$). NS and p-value if $p > 0.1$.

NA - Data not available for this station.

Potomac River (1986-1996)

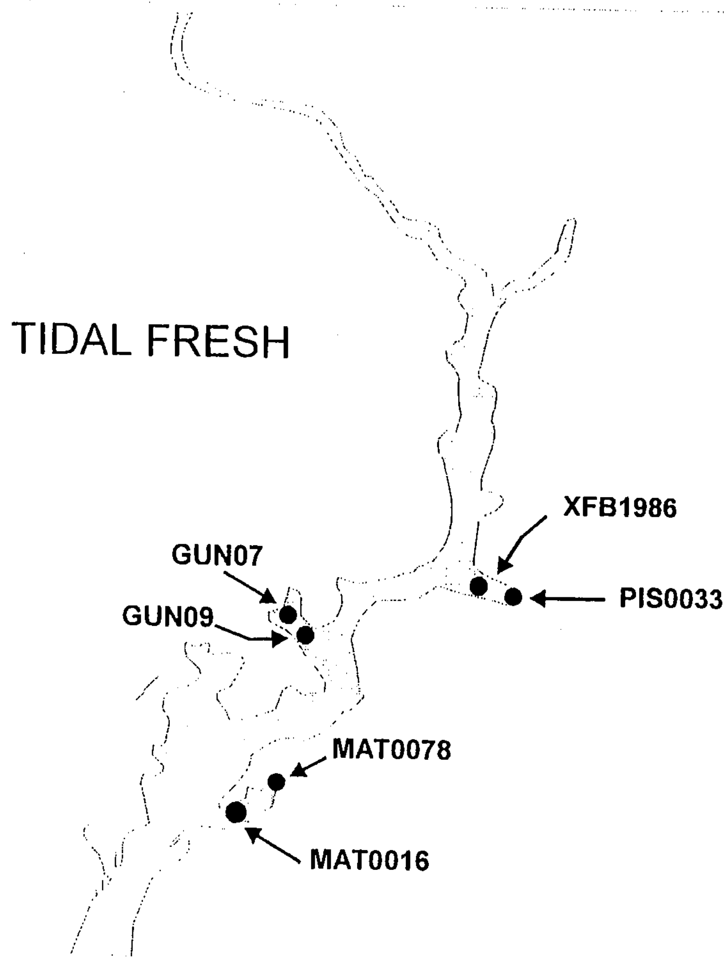
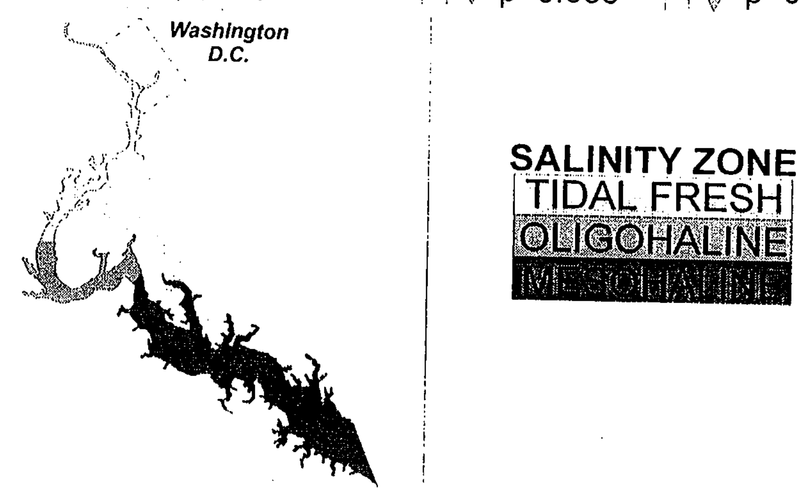


Figure ES-2c. Potomac tidal creeks and Gunston Cove nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

NA - Data not available for this station.

Eleven Year Bottom Trends Non-flow Adjusted

	TP	PO4F	TN	DIN
PIS0033	NA	NA	NA	NA
XFB1986	NA	NA	NA	NA
MAT0078	NA	NA	NA	NA
MAT0016	NA	NA	NA	NA
GUN07	36% p=0.001	NS p=0.182	35% p=0.000	38% p=0.005
GUN09	NS p=0.369	NS p=0.605	NS p=0.035	27% p=0.001



Potomac River (1985-1996)

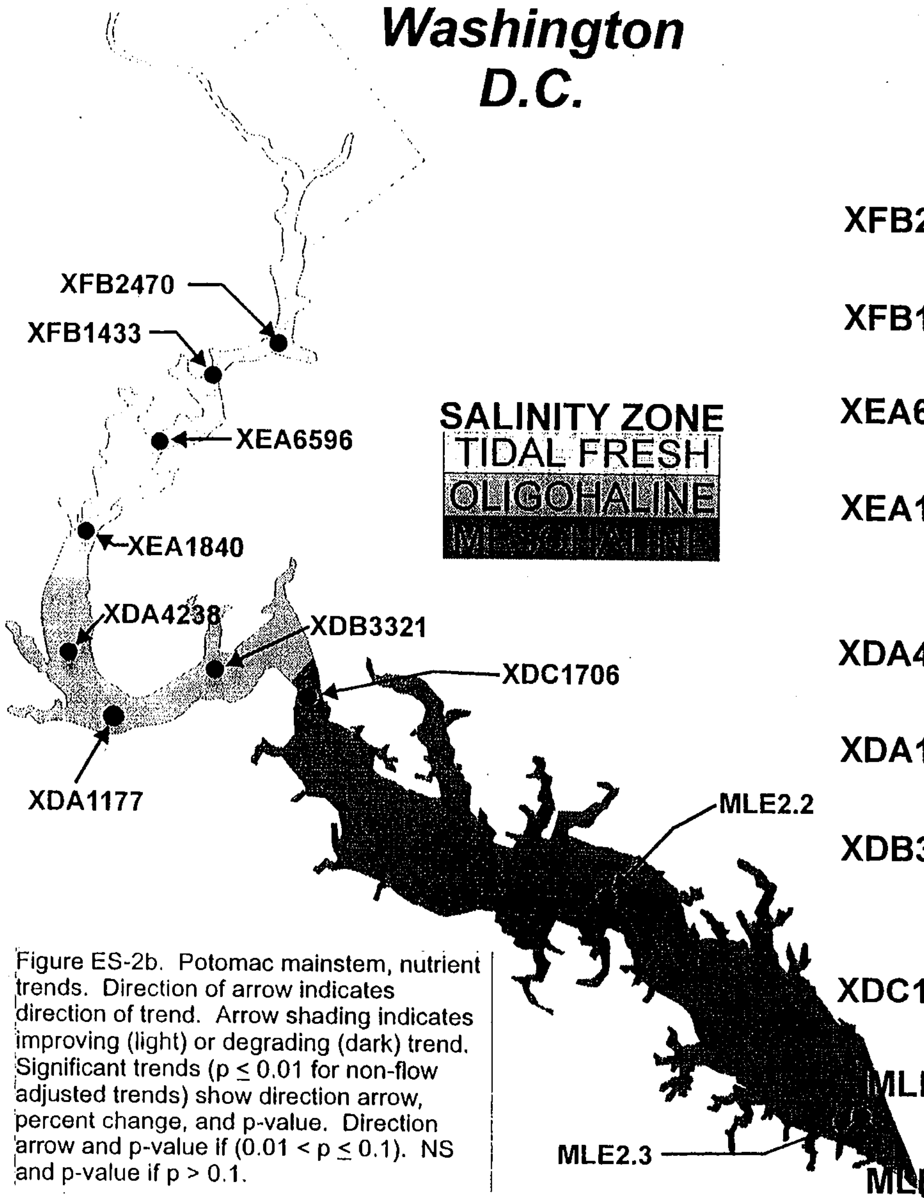


Figure ES-2b. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

Twelve Year Bottom Trends Non-flow Adjusted

Station	TSS	CHAA	DO_FLD
XFB2470	NS p=0.816	↑ 132% p=0.000	↑ 13% p=0.002
XFB1433	NS p=0.277	↑ 105% p=0.003	↑ NS p=0.011
XEA6596	↑ NS p=0.060	NS p=0.119	NS p=0.189
XEA1840	NS p=0.756	↑ NS p=0.068	NS p=0.443
XDA4238	NS p=0.705	↑ 61% p=0.001	NS p=0.570
XDA1177	NS p=0.694	↑ NS p=0.013	NS p=0.303
XDB3321*	↓ NS p=0.070	NS p=1.000	↑ NS p=0.020
XDC1706	NS p=1.000	↑ NS p=0.013	↑ 16% p=0.003
MLE2.2	↑ 120% p=0.000	↑ NS p=0.054	NS p=0.211
MLE2.3	NS p=0.529	NS p=0.404	NS p=0.422

*Station not sampled after September 1990.

Potomac River (1985-1996)

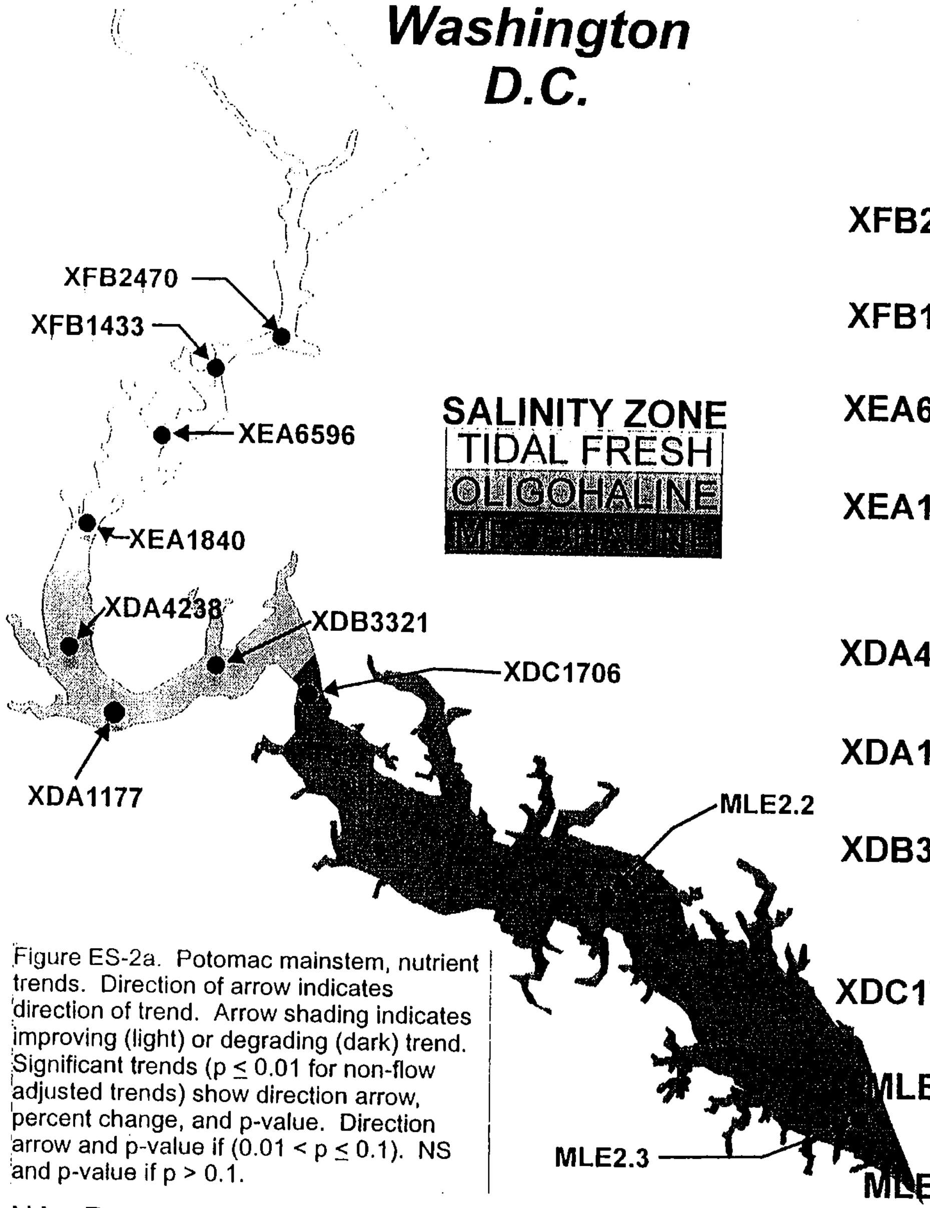


Figure ES-2a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.01 for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.01 < p ≤ 0.1). NS and p-value if p > 0.1.

NA - Data not available for this station.

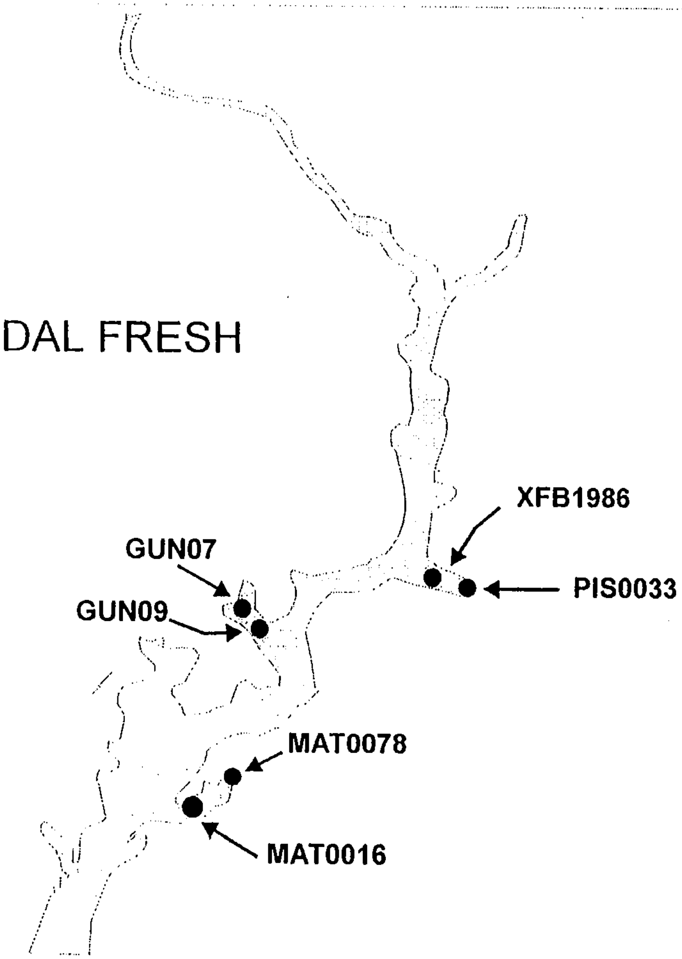
Twelve Year Bottom Trends Non-flow Adjusted

	TP	PO4F	TN	DIN
XFB2470	NS p=0.296	NS p=0.244	↓ 15% p=0.010	↓ NS p=0.020
XFB1433	NS p=0.669	NS p=0.176	↓ 16% p=0.001	↓ NS p=0.020
XEA6596	NS p=0.608	NS p=0.199	↓ 13% p=0.004	NS p=0.115
XEA1840	NS p=0.702	NS p=0.499	NS p=0.142	NS p=0.440
XDA4238	NS p=0.793	NS p=0.958	NS p=0.167	NS p=0.170
XDA1177	NS p=0.813	NS p=0.875	NS p=0.661	NS p=0.491
XDB3321*	NS p=0.404	NA	↑ NS p=0.067	↑ 72% p=0.002
XDC1706	NS p=0.498	↑ NS p=0.094	NS p=0.143	NS p=0.294
MLE2.2	NS p=0.681	↑ NS p=0.047	↑ NS p=0.016	NS p=0.351
MLE2.3	↑ NS p=0.035	NS p=0.849	NS p=0.173	NS p=0.886

PO4F trends start in October 1990.

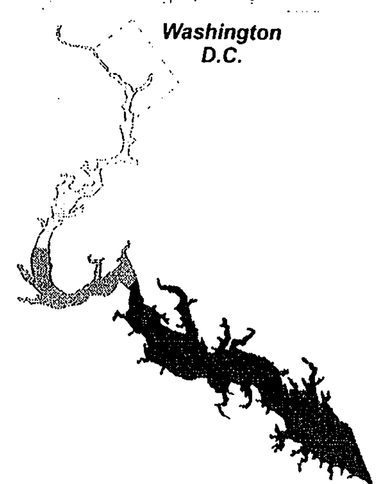
*Station not sampled after September 1990.

Potomac River (1986-1996)



Eleven Year Surface Trends Non-flow Adjusted

	TSS	SECCHI	CHAA	DO
PIS0033	↑ NS p=0.020	NA	NS p=0.206	↓ 10% p=0.008
XFB1986	↑ 60% p=0.001	↓ NS p=0.054	↑ 140% p=0.000	↑ 20% p=0.000
MAT0078	↑ NS p=0.027	NA	NS p=0.396	NS p=0.116
MAT0016	NS p=0.560	NS p=0.822	↓ NS p=0.822	NS p=0.771
GUN07	↓ NS p=0.056	↑ 60% p=0.000	NS p=0.737	↑ NS p=0.067
GUN09	NS p=0.765	NS p=0.204	↑ 110% p=0.000	↑ 10% p=0.002

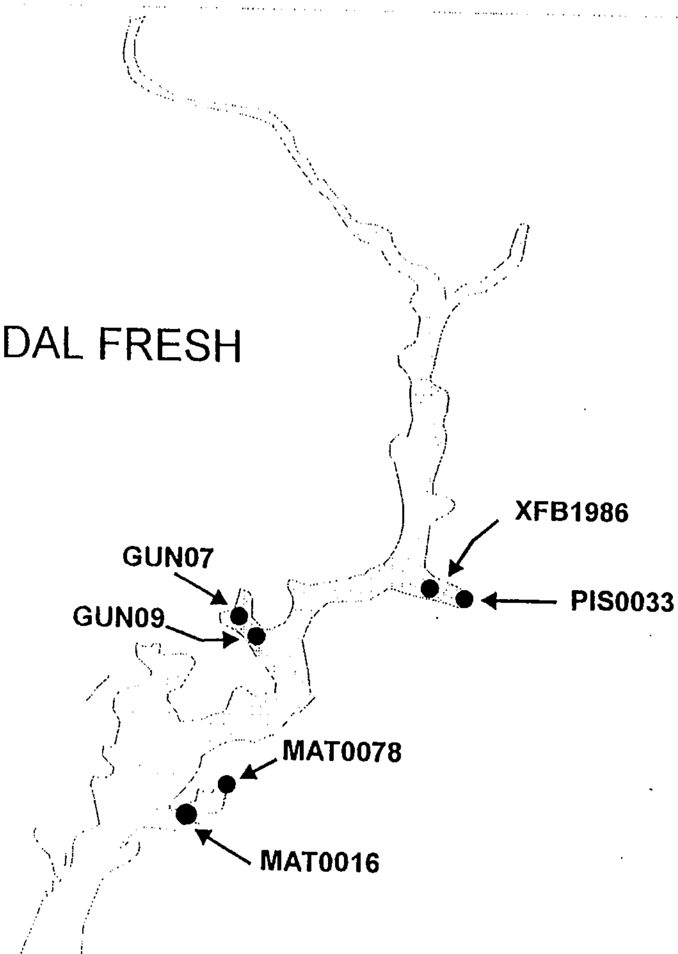


SALINITY ZONE
TIDAL FRESH
OLIGOHALINE
MESOHALINE

Figure ES-1d. Potomac tidal creeks and Gunston Cove nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.01$ for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.01 < p \leq 0.1$). NS and p-value if $p > 0.1$.
NA - Data not available for this station.
Gray box indicates seasonal heterogeneity.

Potomac River (1986-1996)

Eleven Year Surface Trends Non-flow Adjusted



	TP	PO4F	TN	DIN
PIS0033	NS p=0.558	↓ NS p=0.045	↓ 25% p=0.004	↓ 26% p=0.003
XFB1986	↑ NS p=0.017	NS p=0.080	↓ 27% p=0.000	↓ 30% p=0.000
MAT0078	NS p=0.528	NS p=0.567	↓ 23% p=0.000	NS p=0.557
MAT0016	NS p=0.205	↑ 179% p=0.001	↓ 15% p=0.003	NS p=0.504
GUN07	↓ 42% p=0.000	NS p=0.673	↓ 35% p=0.000	↓ 39% p=0.003
GUN09	NS p=0.672	NS p=0.061	↓ NS p=0.088	↓ 26% p=0.001

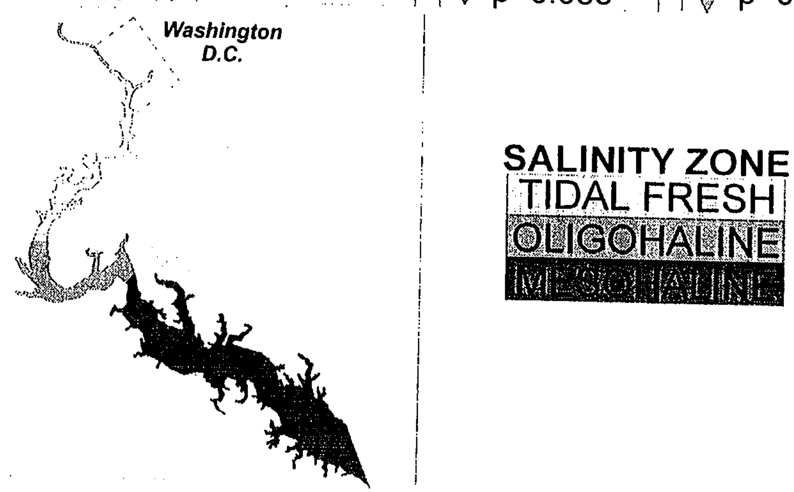


Figure ES-1c. Potomac tidal creeks and Gunstone Cove nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.01$ for non-flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.01 < p \leq 0.1$). NS and p-value if $p > 0.1$.

PO4F trends for the Potomac tidal creeks start in October 1990.

Potomac River (1974-1996)

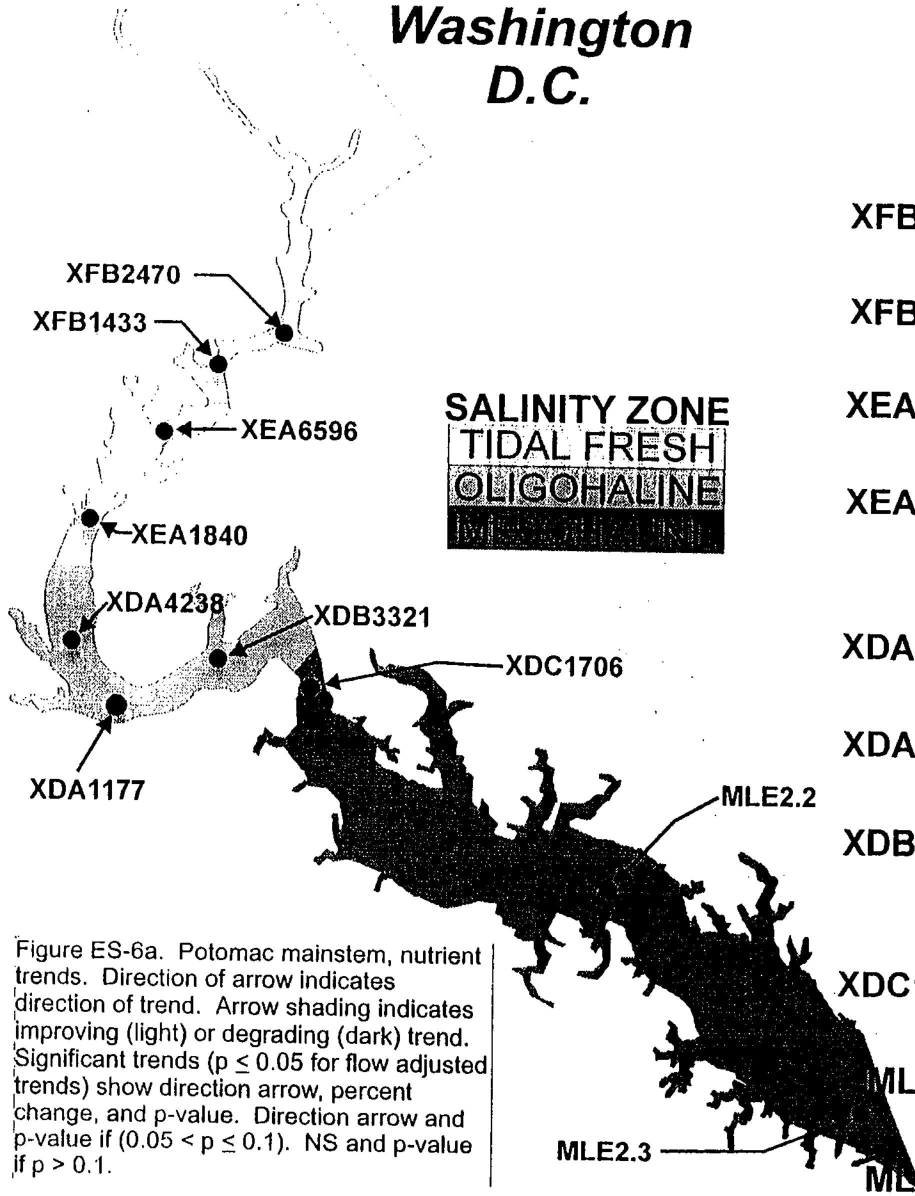


Figure ES-6a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends (p ≤ 0.05 for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if (0.05 < p ≤ 0.1). NS and p-value if p > 0.1.

NA - Data not available for this station.

Twenty-five Year Bottom Trends Non-flow Adjusted

	TP	PO4W	TN	DIN
XFB2470	NS p=0.807	↓ 57% p=0.002	NS p=0.228	NS p=0.621
XFB1433	↓ NS p=0.046	↓ 46% p=0.002	NS p=0.908	NS p=0.897
XEA6596	↓ NS p=0.065	↓ 48% p=0.000	NS p=0.219	NS p=0.112
XEA1840	NS p=0.305	↓ 37% p=0.010	↑ NS p=0.010	↑ NS p=0.056
XDA4238	NS p=0.234	↓ 34% p=0.010	NS p=0.424	NS p=0.592
XDA1177	NS p=0.989	↓ 36% p=0.008	NS p=0.175	NS p=0.312
XDB3321*	NS p=0.976	↓ NS p=0.083	NS p=1.000	NS p=0.466
XDC1706	NS p=0.333	NS p=0.330	NS p=0.584	NS p=0.956
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

PO4W trends end in September 1990.

*Station not sampled after September 1990.

Potomac River (1974-1996)

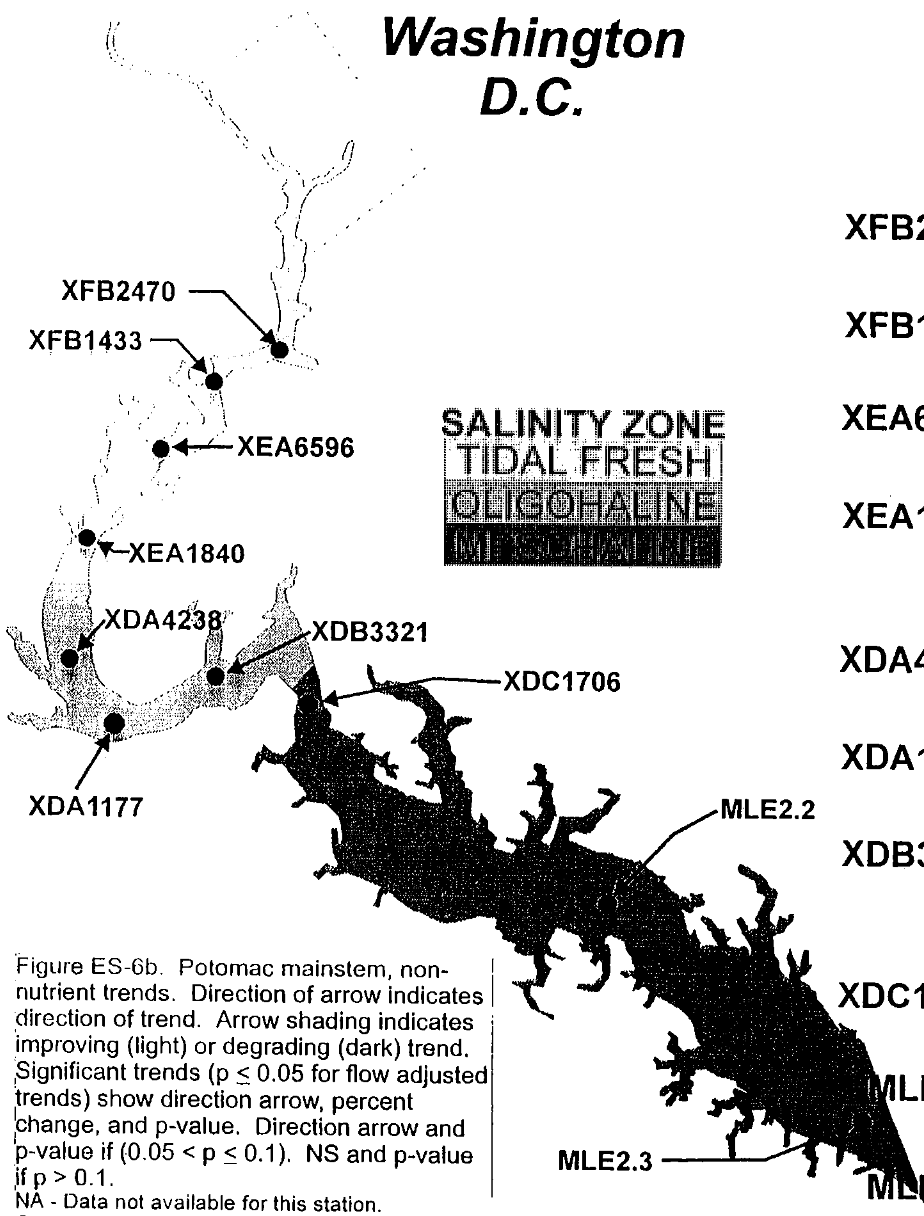


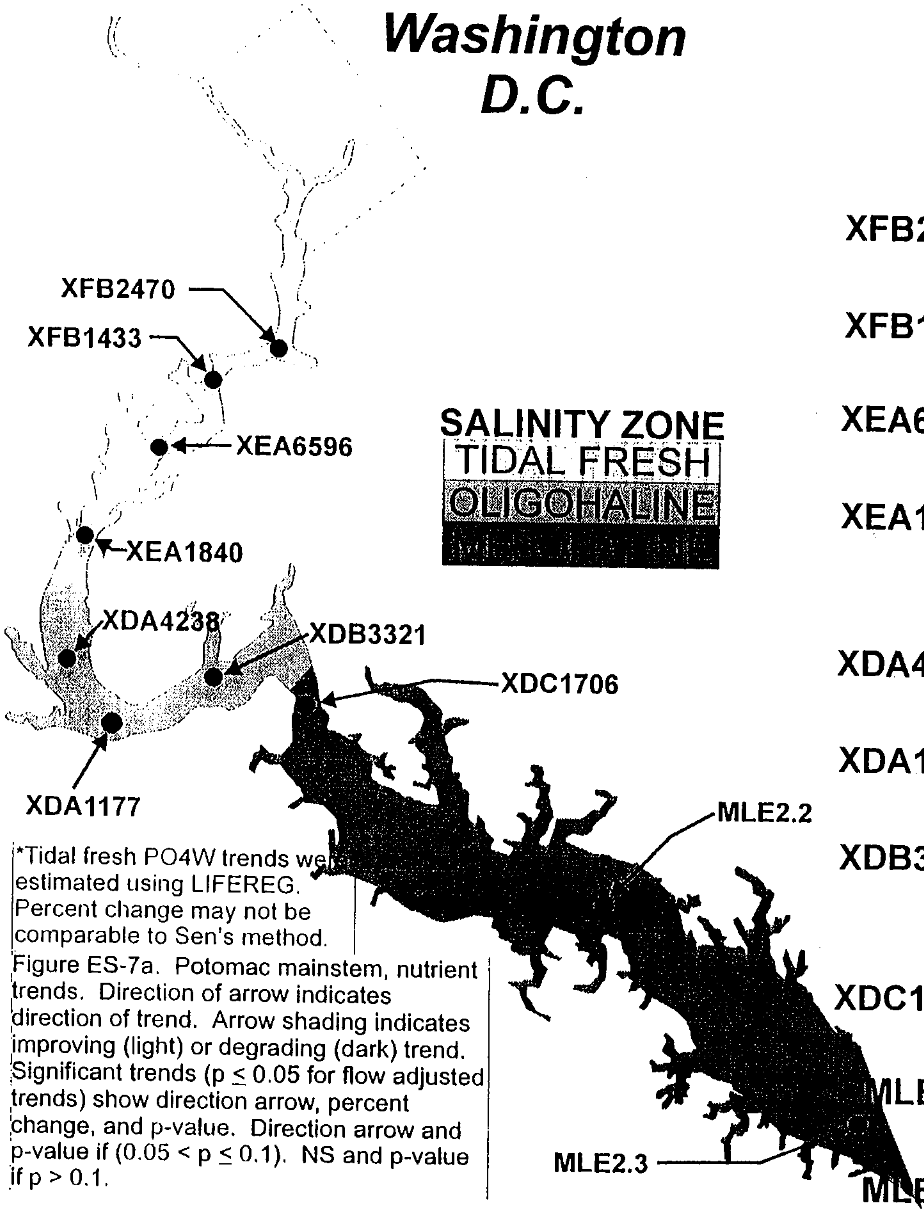
Figure ES-6b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.
NA - Data not available for this station.
Gray box indicated seasonal heterogeneity.

Twenty-five Year Bottom Trends Non-flow Adjusted

	TSS	CHAA	DO_FLD
XFB2470	↑ NS p=0.022	↑ 20% p=0.000	↑ 10% p=0.008
XFB1433	↑ NS p=0.012	↑ NS p=0.039	NS p=0.144
XEA6596	↑ 90% p=0.004	NS p=0.679	NS p=0.372
XEA1840	NS p=0.771	NS p=0.765	↓ 10% p=0.009
XDA4238	↑ NS p=0.070	↑ NS p=0.071	NS p=0.216
XDA1177	↑ NS p=0.049	NS p=0.543	↓ NS p=0.014
XDB3321*	NS p=1.000	↓ NS p=0.076	↓ 10% p=0.004
XDC1706	↑ NS p=0.011	↑ NS p=0.083	↓ NS p=0.050
MLE2.2	NA	NA	NA
MLE2.3	NA	NA	NA

*Station not sampled after September 1990.

Potomac River (1974-1996)



*Tidal fresh PO4W trends were estimated using LIFEREG. Percent change may not be comparable to Sen's method.

Figure ES-7a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

Twenty-five Year Surface Trends Flow Adjusted

	TP	PO4W*	TN	DIN
XFB2470	↓ 31% p=0.000	↓ 66% p=0.000	↑ 23% p=0.000	↑ 20% p=0.004
XFB1433	↓ 32% p=0.000	↓ 64% p=0.000	↑ 17% p=0.003	↑ 21% p=0.001
XEA6596	↓ 34% p=0.000	↓ 62% p=0.000	↑ 28% p=0.000	↑ 31% p=0.000
XEA1840	↓ 39% p=0.000	↓ 49% p=0.000	↑ 38% p=0.000	↑ 40% p=0.000
XDA4238	↓ 25% p=0.001	NS p=0.210	↑ 22% p=0.001	↑ 27% p=0.000
XDA1177	↓ 25% p=0.022	NS p=0.265	↑ 24% p=0.001	↑ 36% p=0.000
XDB3321*	NS p=0.129	NS p=0.100	↑ 46% p=0.000	↑ 57% p=0.000
XDC1706	↓ 23% p=0.015	↓ 46% p=0.001	NS p=0.300	↑ 23% p=0.017
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

NA - Data not available for this station.

PO4W trends end in September 1990.

*Station not sampled after September 1990.

Potomac River (1974-1996)

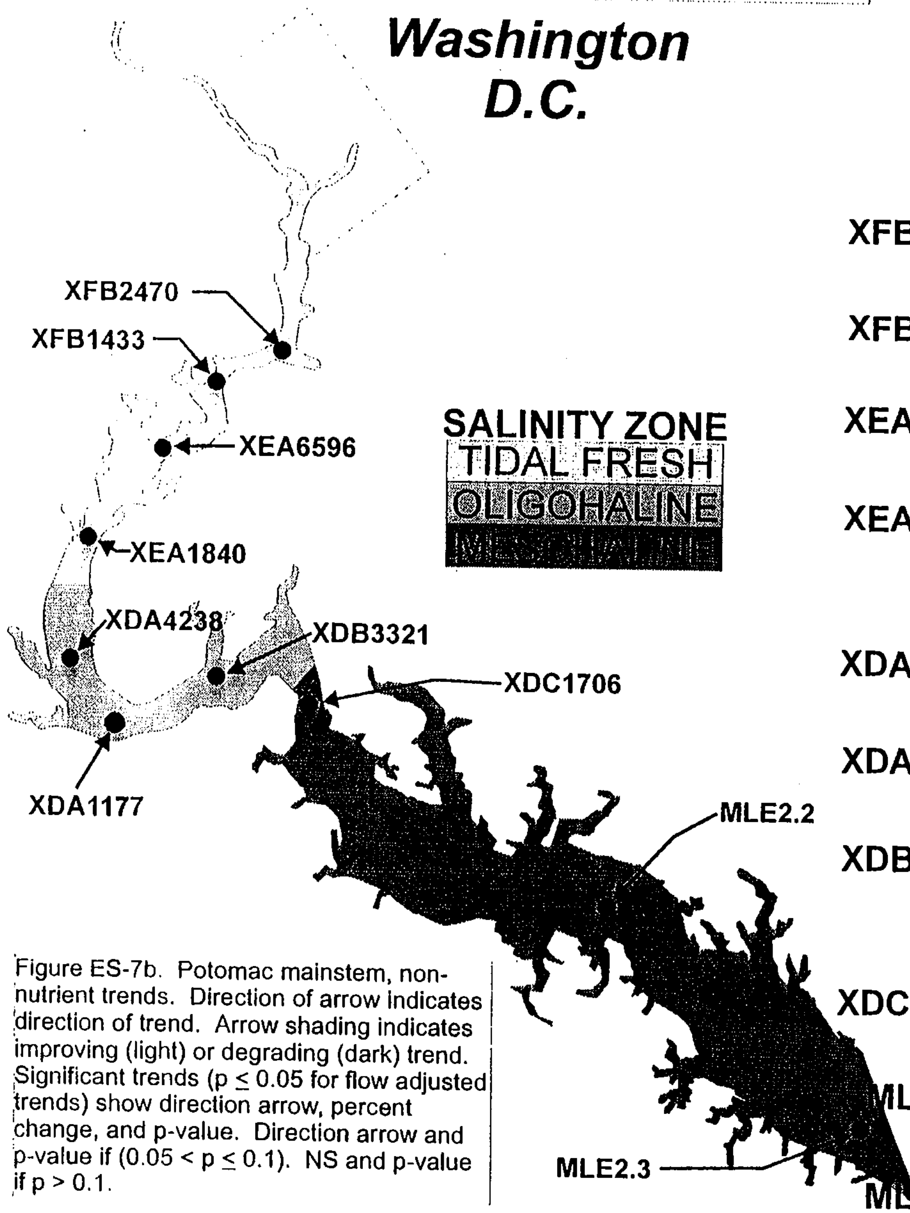


Figure ES-7b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

NA - Data not available for this station.

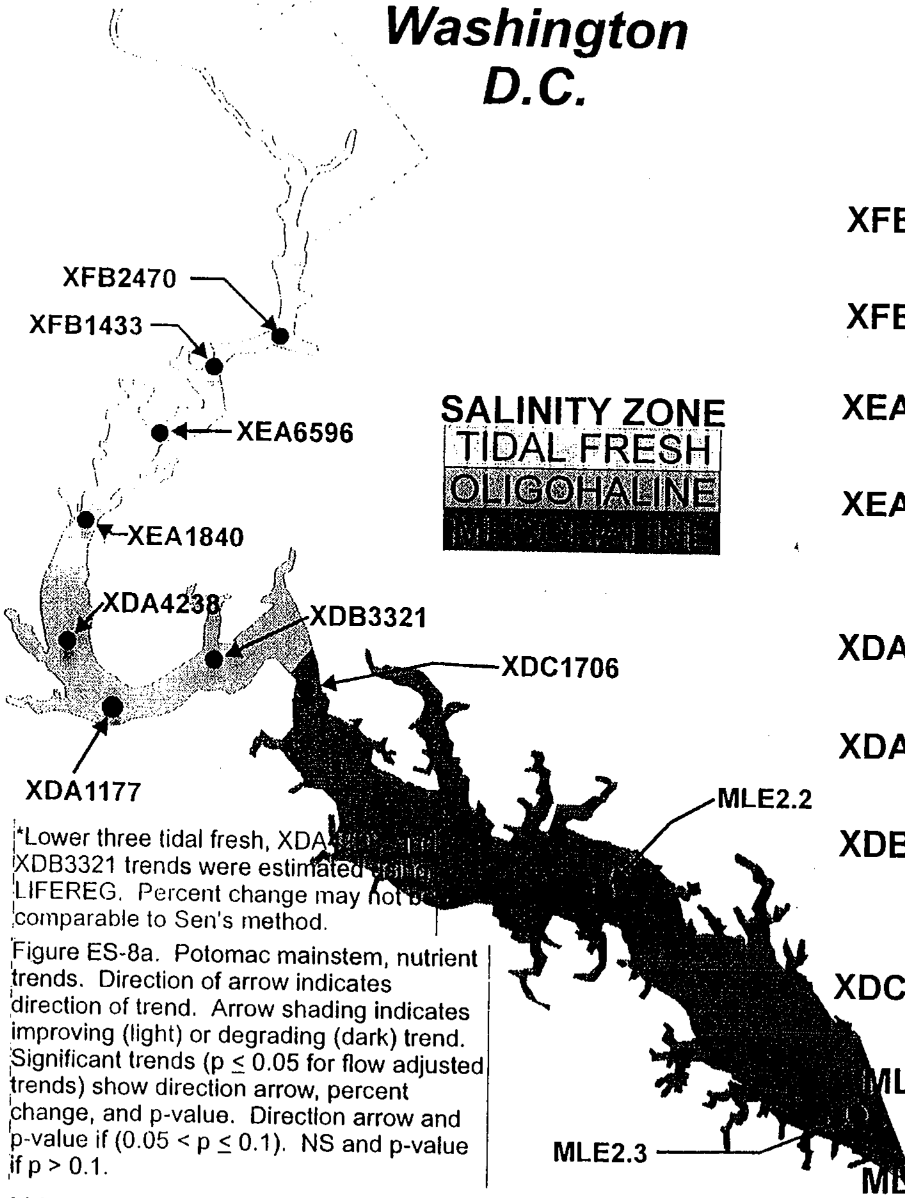
Twenty-five Year Surface Trends Flow Adjusted

	TSS	SECCHI	CHAA	DO_FLD
XFB2470	↑ 60% p=0.005	NS p=0.176	NS p=0.110	↑ 10% p=0.002
XFB1433	↑ 40% p=0.006	NS p=0.929	NS p=0.641	NS p=0.726
XEA6596	↑ 50% p=0.011	NS p=0.388	NS p=0.102	↓ 10% p=0.040
XEA1840	NS p=0.181	NS p=0.250	↓ 30% p=0.000	↓ 20% p=0.000
XDA4238	↑ 80% p=0.024	NS p=0.305	↓ NS p=0.069	NS p=0.107
XDA1177	↑ 30% p=0.027	NS p=0.398	NS p=0.262	↓ 10% p=0.004
XDB3321*	↑ 40% p=0.019	NS p=0.315	↓ 50% p=0.001	↓ NS p=0.060
XDC1706	NS p=0.121	NS p=0.510	NS p=0.402	↓ 20% p=0.000
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

*Station not sampled after September 1990.

Potomac River (1974-1996)

Twenty-five Year Bottom Trends Flow Adjusted



	TP	PO4W*	TN	DIN
XFB2470	NS p=0.276	↓ 48% p=0.000	↑ NS p=0.065	↑ 18% p=0.033
XFB1433	↓ 38% p=0.022	↓ 66% p=0.000	NS p=0.555	NS p=0.818
XEA6596	↓ NS p=0.057	↓ 66% p=0.000	NS p=0.127	NS p=0.174
XEA1840	NS p=0.123	↓ 49% p=0.000	↑ 24% p=0.012	NS p=0.108
XDA4238	NS p=0.182	↓ 34% p=0.017	NS p=0.256	NS p=0.735
XDA1177	NS p=0.692	↓ 25% p=0.035	↑ 15% p=0.042	NS p=0.299
XDB3321*	NS p=0.953	↓ 30% p=0.031	NS p=0.465	↑ 28% p=0.033
XDC1706	NS p=0.278	↓ NS p=0.071	NS p=0.537	NS p=0.550
MLE2.2	NA	NA	NA	NA
MLE2.3	NA	NA	NA	NA

*Lower three tidal fresh, XDA4238, and XDB3321 trends were estimated using LIFEREG. Percent change may not be comparable to Sen's method.

Figure ES-8a. Potomac mainstem, nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

NA - Data not available for this station.

PO4W trends end in September 1990.

*Station not sampled after September 1990.

Potomac River (1974-1996)

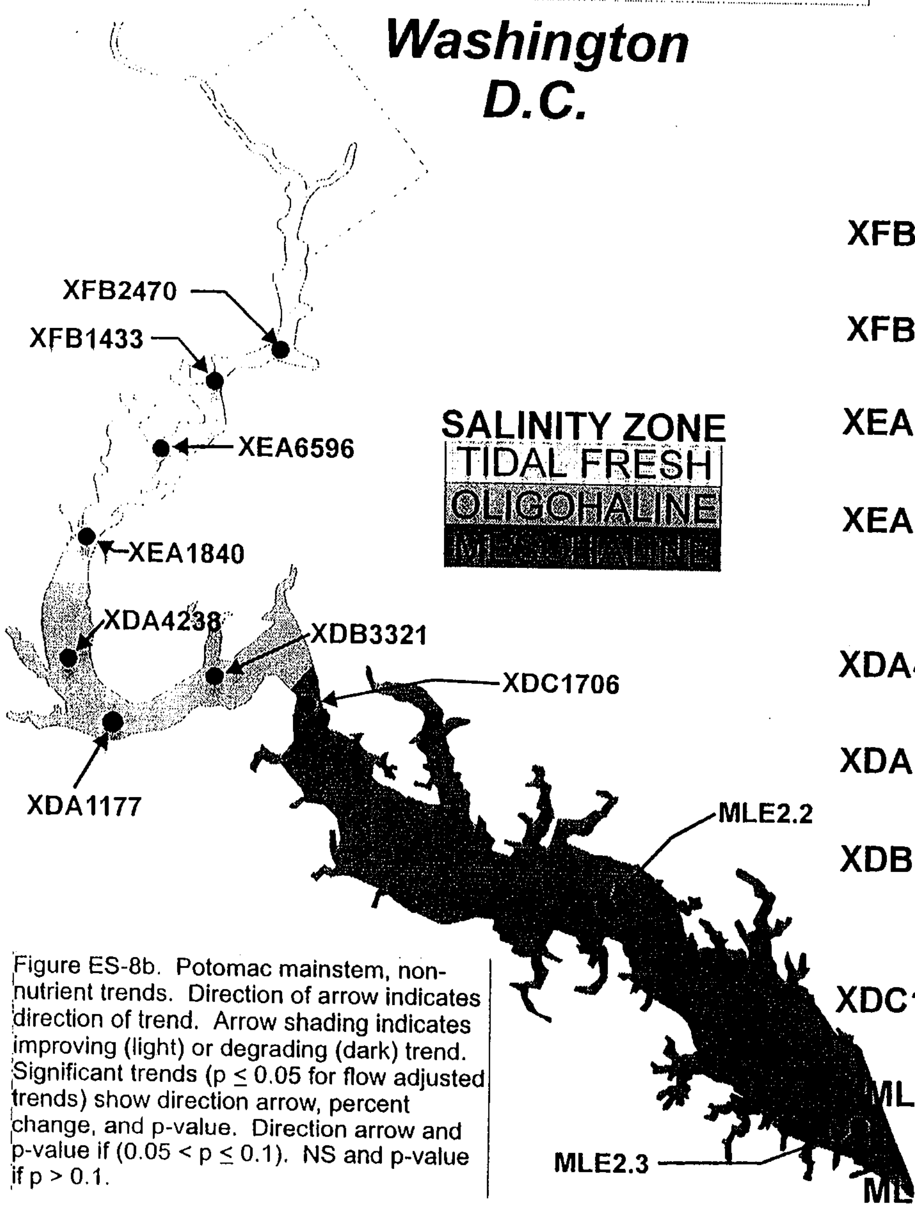


Figure ES-8b. Potomac mainstem, non-nutrient trends. Direction of arrow indicates direction of trend. Arrow shading indicates improving (light) or degrading (dark) trend. Significant trends ($p \leq 0.05$ for flow adjusted trends) show direction arrow, percent change, and p-value. Direction arrow and p-value if ($0.05 < p \leq 0.1$). NS and p-value if $p > 0.1$.

NA - Data not available for this station.

Twenty-five Year Bottom Trends Flow Adjusted

Station	TSS	CHAA	DO_FLD
XFB2470	NS p=0.462	↑ 30% p=0.000	↑ 10% p=0.003
XFB1433	NS p=0.310	↑ 20% p=0.003	NS p=0.543
XEA6596	NS p=0.146	NS p=0.452	NS p=0.210
XEA1840	NS p=0.891	NS p=0.867	↓ 20% p=0.001
XDA4238	NS p=0.182	NS p=0.197	NS p=0.128
XDA1177	↑ NS p=0.027	NS p=0.924	↓ 10% p=0.006
XDB3321*	NS p=0.972	↓ 70% p=0.001	↓ 10% p=0.028
XDC1706	↑ 50% p=0.040	NS p=0.371	↓ 20% p=0.002
MLE2.2	NA	NA	NA
MLE2.3	NA	NA	NA

*Station not sampled after September 1990.

I. INTRODUCTION

A. Background

The EPA Chesapeake Bay Study (EPA, 1982) concluded that deteriorating conditions in the Chesapeake Bay, including poor summer bottom dissolved oxygen conditions and declines in living resources, resulted largely from problems associated with nutrient over-enrichment. The Bay Study provided the scientific basis for the 1983 EPA publication *Chesapeake Bay: A Framework for Action* that recommended a reduction in point and non-point source nutrient loading to the Bay. Following the publication of these recommendations, the 1983 Chesapeake Bay Agreement was signed by the Chesapeake Bay Executive Council that includes the Governors of Maryland, Virginia, and Pennsylvania, the Mayor of the District of Columbia, the EPA Administrator, and the Chairman of the Chesapeake Bay Commission. This historic agreement began a unique cooperative effort to restore, protect, and manage the Bay. The subsequent 1987 Bay Agreement called for a 40% reduction in controllable nitrogen and phosphorus loads to the Bay from the 1985 baseline by the year 2000. "Controllable" loads are those resulting from human activity, excluding atmospheric emissions. The 40% reduction goal was reaffirmed in the 1992 Amendments to the Bay Agreement. These amendments emphasized the importance of a watershed-based approach to nutrient reduction. A new focus was placed on the tributaries, and a commitment was made to design and implement basin-specific nutrient reduction plans known as Tributary Strategies. In addition, it was stipulated that nutrient loads 40% below 1985 levels are not only goals, but caps to future allowable load limits, even as populations and development increase.

Tremendous progress has been made toward meeting the 40% goal for nutrient reduction. In Maryland, practices already implemented to reduce phosphorus loads are projected to reduce controllable phosphorus loadings to 38% below the 1985 baseline, for a total phosphorus load reduction of 34% from baseline. Much of the improvement can be traced to December, 1985, when Maryland banned phosphate detergents. Similar bans in Pennsylvania, Virginia, and the District of Columbia, along with upgrades to waste-water treatment plants (WWTPs) further curtailed point-source loadings of phosphorus. Practices implemented to decrease nitrogen loads are projected to reduce controllable nitrogen loadings to 23% below the 1985 baseline, for a total nitrogen load reduction of 17% from baseline. As with phosphorus, much of this progress derives from improvements in WWTPs. Additional reductions were made in non-point sources of both nitrogen and phosphorus. Successes include Statewide efforts to maintain stream-side grass or forest buffer strips, apply agricultural Best Management Practices, and effectively manage urban storm water.

Ambient water quality conditions, not loads, determine the suitability of the Bay as habitat for aquatic plants and animals. Analysis of long-term trends in ambient water quality, therefore provides a measure of our "real" progress toward the goal of a healthier Bay. Trend tests allow us to determine if and where Bay water quality is improving. These analyses provide managers with information on areas where current management strategies are working and where changes are necessary.

The objectives of this project were (1) to test for long-term trends in indicators of nutrient enrichment in tidal portions of the Potomac river, and (2) to relate trend information to changes in nutrient loads that have resulted from management actions.

B. Water Quality Variables Evaluated For Trends

The following 12 water quality variables were evaluated for trends:

1) total phosphorus (TP),
2) orthophosphate (PO_4),
3) total nitrogen (TN),
4) dissolved inorganic nitrogen (DIN),
5) nitrate plus nitrite (NO_{23}),
6) ammonium (NH_4),
7) total suspended solids (TSS),
8) Secchi depth (SECCHI),
9) active chlorophyll a (CHLA),
10) field dissolved oxygen (DO_FLD),
11) field pH (PH_FLD), and
12) total alkalinity (TALK)

The nutrient variables included are TP, PO_4, TN, DIN, NO_{23}, and NH_4. Both nitrogen and phosphorus are required for the metabolic processes of phytoplankton. Total nitrogen and total phosphorus were examined because they provide measures of all components of these nutrients in the water column, dissolved and particulate in both inorganic and organic forms. Concentrations of the various species of these nutrients cycle seasonally through physical and biological processes. Dissolved inorganic nitrogen, which consists of nitrate, nitrite and ammonium, is the most available form for uptake by phytoplankton. Depending on season and location, DIN can comprise less than 5% to greater than 95% of the total nitrogen pool. Relative concentrations of DIN are greatest in the tidal fresh zone during cold, high volume flows, and are lowest in high-salinity zones during warm low-flow periods (Boynton *et al.*, 1995).

Unlike nitrogen, inorganic phosphorus adheres readily to suspended matter and is also stored within phytoplankton cells, so that the largest fraction of the phosphorus pool is generally particulate rather than dissolved. However, orthophosphate and other dissolved forms are very important to the ecosystem because they represent phosphorus immediately available for uptake by plants. As such, this fraction may cycle very rapidly through the plankton community (Boynton *et al.*, 1995; Wetzel, 1983).

When available in high concentrations, nitrogen and phosphorus can promote excessive growth of algae. In low concentrations, they can be limiting to growth. Both DIN and PO_4 are important in establishing habitat requirements for submerged aquatic vegetation (SAV) (EPA, 1992). Growing season (April to October) median concentrations of DIN $\leq$ 0.15 mg/l in the mesohaline, $PO_4 \leq$ 0.02 mg/l in the tidal fresh and oligohaline, and $PO_4 \leq$ 0.01 mg/l in the mesohaline have been correlated with the presence of SAV in areas suitable for SAV growth (EPA, 1992; Dennison *et al.*, 1993).

Secchi depth is a commonly used visual estimation of light attenuation. Light attenuation determines the depth at which aquatic plants, both phytoplankton and SAV, can photosynthesize. Light attenuation, and therefore Secchi depth, is affected by dissolved or suspended materials. Total suspended solids (TSS) include all organic and inorganic particulate matter in the water column. High TSS concentrations, which are normally comprised of inorganic particles from runoff, dense phytoplankton communities, or both, increase light attenuation and lower Secchi depth. Concentrations of TSS and light attenuation are addressed in the SAV habitat requirements (EPA, 1992). Median concentrations of TSS during the growing season should normally be less than 15 mg/l for all salinity zones; light attenuation coefficients (K_d) of less than 2 m^{-1} for the tidal fresh and oligohaline, and less than 1.5 m^{-1} for the mesohaline are correlated with SAV presence (EPA, 1992; Dennison *et al.*, 1993). These light attenuation habitat requirements correspond to Secchi depths of 0.7 m (tidal fresh and oligohaline) and 1.0 m (mesohaline).

Active chlorophyll *a* is one of the most commonly used measures of phytoplankton abundance. High chlorophyll concentration is an indication of eutrophic conditions. Chlorophyll is also addressed in the SAV habitat requirements because a growing season median less than 15 mg/l has been correlated with SAV presence (EPA, 1992; Dennison *et al.*, 1993).

Dissolved oxygen is vital to the survival of all organisms that respire aerobically, and its depletion in the Bay has long been identified as a principal impact of nutrient enrichment in the Chesapeake Bay (EPA, 1982). Oxygen enters the estuary from the atmosphere and from the photosynthetic activity of aquatic plants including phytoplankton. At the surface, dissolved oxygen levels are generally more than adequate because of mixing with the atmosphere. In deeper waters, the level of dissolved oxygen is dependent on the rate of oxygen consumption, the amount of mixing with aerated surface waters, and lateral or longitudinal hydrodynamic transport. Bottom dissolved oxygen levels become depressed when mixing with surface waters is minimal and bacterial decomposition of organic matter is high (Day *et al.*, 1989). Bottom DO concentrations greater than 5 mg/l are considered healthy for most living resources. Concentrations below 5 mg/l may be biologically stressful for aquatic animals, and DO concentrations less than 1 mg/l are considered unsuitable habitat (Jordan *et al.*, 1993).

pH is a measure of the hydrogen ion concentration $[H^+]$ and quantifies the acidity or alkalinity of a system. In natural waters pH can range from <2 to14 with a pH of 7 considered neutral (Wetzel, 1983). In estuaries, pH averages from 7.0 to 7.5 in fresh areas and from 8.0 to 8.6 in more saline waters. Short-term swings and long-term changes in pH can be very harmful to aquatic organisms, which in marine systems prefer conditions with pH ranging from 6.5 to 8.5. Survival of many aquatic plants and animals may be affected at pH less than 5.0 or greater than 9.0 (EPA, 1993). Because pH is defined in a logarithmic scale, a change in pH of one unit corresponds to a ten-fold change in hydrogen ion concentration.

Alkalinity refers to the quantity and types of compounds which can change the pH of a system from neutral to alkaline and represents the sum of all the titratable bases present in the system. Because in many surface waters alkalinity is primarily a function of carbonate, bicarbonate, and hydroxide concentrations, alkalinity is expressed in terms of $CaCO_3$. In aquatic systems, alkalinity may also include the contribution from phosphates, silicates, and borates. The buffering capacity of aquatic systems is important in maintaining a pH range suitable for reproduction and growth.

II. METHODS

A. Water Quality Monitoring Stations Included in the Study

Water quality stations analyzed for trends included ten located in the Potomac mainstem, four on tidal creeks which discharge into the Potomac from the Maryland side and two stations in Gunston Cove, which is located in Virginia. The ten mainstem and four tidal creek stations are monitored under the U.S. EPA Chesapeake Bay Program by Maryland Department of Natural Resources (DNR). Gunston Cove is monitored under the Gunston Cove Study by the Biology Department of George Mason University (GMU) located in Fairfax County, Virginia and funded by the Fairfax County Department of Public Works.

The ten mainstem stations include: XFB2470 (at the mouth of Piscataway Creek); XFB1433 (near Marshall Hall); XEA6596 (near Indian Head); XEA1840 (between Stump Neck and Moss Point); XDA4238 (south of Smith Point); XDA1177 (Maryland Point); XDB3321 (at the mouth of Nanjemoy Creek); XDC1706 (Morgantown/301 Bridge); MLE2.2 (Ragged Point); and, MLE2.3 (Point Lookout at the mouth of the Potomac River). The four stations in Maryland tidal creeks are: XFB1986 (Piscataway Creek near Buoy 6); PIS0033 (Piscataway Creek near Route 210); MAT0016 (Matawoman Creek across from Sweden Point); and, MAT0078 (Matawoman Creek near Route 225). Although a number of stations in Gunston Cove are sampled, only two were selected for this report based on location and completeness of the data record. The two Gunston Cove stations included: Gunston 7 (center of Gunston Cove); and, Gunston 9 (Potomac River channel at the mouth of the Cove). Station names, locations and identifying landmarks are provided in Fig. II-1.

B. Field Methods and Sampling Regime

Stations monitored by DNR were sampled twenty times per year including one sampling date per month during cool months (November through February) and two sampling dates per month during the growing season (March through October). This level of sampling allows for the resolution of seasonal cycles that appear in many water quality variables and provides sufficient statistical power to detect trends (MDE, 1993a). Gunston Cove stations are monitored twice per month except during March and in the Fall, when they are monitored once per month.

The goal of the depth sampling protocol for nutrients is to characterize water quality in the surface mixed layer and the bottom mixed layer in areas of density stratification. In areas where density gradients are minimal or non-existent samples are collected at surface (0.3-0.5 meter depth) and bottom (1 meter less than the total depth). Four samples (surface, above-pycnocline, below-pycnocline, and bottom) are collected in areas that demonstrate appreciable salinity stratification. For all parameters evaluated in this report, data from the surface or surface mixed and bottom or bottom mixed layers were analyzed depending on station depth and the presence or absence of salinity stratification.

Mainstem stations from XFB2470 to XDB3321 are located in tidal fresh or transition zone waters. Because waters in the upper Potomac estuary lack the salinity to become stratified, all parameters for the upper seven mainstem stations were analyzed as surface or bottom data (unmixed). Data for the lower three mainstem stations, from XDC1706 to MLE2.3, were analyzed after being combined into a surface-mixed zone and a bottom-mixed zone. The surface- and bottom-mixed zones were created by calculating the mean of the surface and above pycnocline data and the mean of the bottom and below pycnocline data, respectively. Note that multiple layer data for XDC1706 were not available until October 1990, following a revision of the Potomac River monitoring protocol to have it more closely match the Bay program protocol; consequently, XDC1706 data for trend analyses prior to October 1990 were either surface or bottom (unmixed). Multiple layer data for MLE2.2 and MLE2.3 were available since the start of sampling under the U.S. EPA Chesapeake Bay monitoring program in July 1984.

Data for Piscataway and Matawoman Creek stations were unmixed since all four are located in tidal fresh waters. For the lower two stations (XFB1986 and MAT0016) both surface and bottom data were analyzed. Only surface data were analyzed for the upper two stations (PIS0033 and MAT0078) since they are located in shallow water and "bottom" data are not available. Finally, only surface and bottom (unmixed) samples were analyzed for Gunston Cove, which is located in tidal fresh water and hence does not undergo salinity stratification.

Physical variables, including PH_FLD, dissolved oxygen, and SECCHI depth, are measured *in situ*. PH_FLD and dissolved oxygen are measured using Hydrolab System Water Quality Instrumentation (HSWQIM, 1984). SECCHI depth is measured with a 20 centimeter Secchi disk (MDE, 1993b).

Systematic water quality sampling on the Potomac River has been underway for several decades. The original field sampling protocol on the Potomac called for the delivery of whole water samples to the laboratory for nutrient analyses. The whole water sampling protocol was maintained on the Potomac following the inception of the U.S. EPA Chesapeake Bay monitoring program despite the implementation of filtered sampling analyses for all other tidal tributaries and the Chesapeake Bay mainstem. In October 1990 the field sampling protocol on the Potomac was changed to incorporate filtering samples in the field thus matching the Bay Program methodology. Starting in October 1990, samples for laboratory analysis were vacuum-filtered using 0.7 μ filters for all tidal DNR-monitored stations except MLE2.3, which was collected and analyzed under the Bay Program since July 1984. The change from whole to filtered water analyses potentially affected the following nutrients evaluated in this report: PO_4, NH_4, NO_{23}, TN, and DIN.

C. Laboratory Methods

Four laboratories were involved in the nutrient analyses of water samples during the period of this study (Fig. II-2). Samples taken prior to 1980 were analyzed at Maryland's Water Resources Administration (WRA) laboratory in Annapolis. A transition period occurred during 1980-1981 when responsibility for analysis of nutrient samples was transferred from WRA to Maryland's Department of Health and Mental Hygiene (DHMH) laboratory in Baltimore, Maryland. The chemical methodologies used did not change during this transition. Samples from all Potomac River stations, except the lower Potomac station MLE2.3, continue to be analyzed at the DHMH laboratory. From June 1984 - May 1985, samples from station MLE2.3 (located at the mouth of the Potomac and collected as part of the Chesapeake Bay mainstem monitoring program) were analyzed by Maryland's Office of Environmental Program staff at the EPA Central Regional Laboratory (CRL) in Annapolis, Maryland. Beginning May 16, 1985, samples from MLE2.3 were analyzed by CBL.

Chemical analysis methods differed at the various laboratories (Table II-1). Total phosphorus (TP) was measured directly at the DHMH Laboratory and was measured directly at CBL from October 1986 - September 1987. All other CBL TP values are calculated as the sum of particulate phosphorus (PP) and total dissolved phosphorus (TDP). Orthophosphate was measured directly at all laboratories, and DIN was always calculated as the sum of nitrite and nitrate (NO_2+NO_3) and ammonium (NH_4). Total nitrogen (TN) is calculated either as the sum of unfiltered total Kjeldahl nitrogen (TKNW) and NO_2+NO_3, or the sum of particulate nitrogen (PN) and total dissolved nitrogen (TDN). Changes in detection limits (DL) which occurred during the study period are detailed in Tables II-2 and II-3.

D. Statistical Methods

i. Initial Treatment of Data

Because detection limit changes occurred during the study period, censoring the data before trend analysis was necessary to avoid incorporating method-related step trends. To preclude the introduction of method-related step trends, values for TP, PO_4, NO_{23}, NH_4, and TSS measured below the highest detection limit were set to one-half the highest detection limit for the effected data within the study period (MDE, 1993b). The decision to use one-half the detection limit for data recorded as below detect is based on the assumption that all measurements between zero and the detection limit are equally likely to occur. When the data between zero and the level of detection are uniformly distributed, the mean of the data is unbiased (Gilbert, 1987). Component parameters (NH_4, NO_{23}, and TKNW) measured below the highest detection limit were set to half the highest detection limit prior to the calculation of DIN or TN.

As described in Laboratory Methods, samples for all of the Maryland tidal Potomac stations except MLE2.3 are analyzed at the DHMH laboratory (samples for MLE2.3 are analyzed at CBL). Due to differences in the detection limits between the two laboratories, CBL detection limits are lower, data for the mesohaline zone stations (XDC1706, MLE2.2, and MLE2.3) could not be combined without losing a great deal of information at MLE2.3. In addition, since the focus of this report is individual station trends, as opposed to trends across stations within a given salinity zone, samples analyzed at CBL for MLE2.3 were not censored to the higher detection limits of DHMH analyzed samples. The potential for method-related trends within a salinity zone would only have affected stations in the mesohaline zone because stations in the tidal fresh and oligohaline zones, and XDC1706 and MLE2.2 in the mesohaline zone, are analyzed by DHMH.

Because multiple observations are available per time period (month) in the trend analysis data set, a summary statistic was needed. Options for a summary statistic would include selecting the data point closest to the mid-point of the time interval, or selecting the mean, trimmed mean, or the median of the observations. Although choosing the observation from the middle of the time period would be the simplest solution, doing so would result in a loss of information from the unused data. In general, the median is recommended when the number of data points within each time period is essentially the same (Reckhow, et al., 1992), which is the case with the monitoring program data set.

For the purpose of scoring monthly medians with detection limit problems, a monthly median was considered affected if $\geq$ 50% data collected during the month were below the highest detection limit of the study period. Calculated variables (TN and DIN) were considered to have detection limit problems only when the components measured below detection comprised at least 10% of the value of the parameter.

Prior analyses of the monitoring program data indicated that total phosphorus values submitted by DHMH between July 1984 and December 1989 were consistently high relative to CBL, Virginia Institute of Marine Science (VIMS), and Old Dominion University (ODU), the other laboratories that participate in the Chesapeake Bay Coordinated Split Sample Program. Following a review of the split sample data, a problem was discovered in the method of calculating DHMH TP values. Subsequently, a computer program was developed that applied a general correction factor to compensate for this bias. This general correction factor adjusted the data to be more in line with the CBL TP values, but was found less reliable at low concentrations than at average or high concentrations. A project to recalculate each individual value from that period using the original data, calibration standards, blank results, and regression curves was completed in the fall of 1996 and the correct data were entered into the DNR water quality data base. Due to record keeping problems, not all of the data in the DNR data base were corrected as part of the re-calibration project. Data for July 1984 through December 1984 could not be re-calibrated for any of the Potomac mainstem stations analyzed at DHMH. In addition, for some stations (XDA1177, XEA6596, and XFB2470) data for all of 1985 could not be re-calibrated. In an attempt to have as complete a data set as possible, the decision was made to use the old TP correction program to adjust the DHMH-analyzed data not corrected as part of the re-calibration project. TP data for MLE2.3 were not affected by the difficulties at the DHMH lab since the MLE2.3 data used in this report were analyzed by CBL. In addition, all the TP data from the four Maryland tidal creek stations were recalculated as part of the re-calibration project.

As described above under Field Methods and Sampling Regime, the field team started filtering samples for nutrient analyses on the Potomac River in October 1990 to match the sampling protocol used on the other tidal monitoring stations and the Chesapeake Bay mainstem.

Unfortunately, the potential effects of changing from whole to filtered water analyses on performing trend tests were not considered when the transition was made (the most likely effects would be a lower mean and smaller standard deviation in the filtered samples). By not taking the change from whole to filtered water sampling into account, a method-related step trend in the data could be misinterpreted as a true time trend.

The potential effects on trend analyses of using a data set consisting of whole and filtered water samples was assessed by comparing split-sample means of several nutrients collected in the tidal fresh, oligohaline, and mesohaline zones. Water samples were collected on April 22, 1997 and split into two groups. One group was filtered using a 0.7 µ filter per the current sampling methodology, and the other group was unfiltered to correspond with the procedures in place on the Potomac prior to October 1990. Both the filtered and unfiltered samples were then placed on ice and shipped to CBL for processing. The means of the two groups were then compared using a paired T-test for the following nutrients: NO_{23}, PO_4, NO_2, and NH_4. The results of the T-tests indicated that only whole and filtered PO_4 differed significantly at the 0.01 level. Filtering had no significant effect on the mean concentration of the remaining nutrients.

Based on the above results, DNR decided to conduct separate trend analyses on PO_4 using October 1990, the date when filtering began, to split the whole and filtered data. For the remaining variables and calculated variables such as TN and DIN, trends were conducted on the entire data set.

ii. Flow Correction

Because many water quality variables are strongly influenced by flow either directly or indirectly, flow adjusted data were tested for trends in addition to the raw (unadjusted) data. As part of an on-going effort to reduce the variability in water quality data that can be attributed to flow, and therefore better discern the effects of management actions to improve water quality, the Data Analysis Workgroup (DAWG) of the Chesapeake Bay Program's Monitoring Subcommittee funded a study to develop statistical flow adjustment models, which are described below.

The first part of the flow adjustment project entailed selecting an appropriate "averaging window" for flow *i.e.*, finding the number of days preceding the water quality sampling date that best summarized flow conditions for the period likely to influence the data. The study group evaluated models associated with 3-, 7-, 14-, and 28-day averaging times. Following an analysis of r values (correlation coefficients) for the four averaging times, it was concluded that since they were quite similar, averaging time was not highly correlated with flow. The study group decided that "Based on these results and considerations of the flow averaging technique, a flow averaging value of seven (7) days was deemed appropriate for all flow corrections" (Alden, *et al.*, unpublished).

It was observed by the study group that because the U.S. Geological Survey (USGS) gaging stations, where flow is measured, are often located several river miles up-stream of the nearest water quality monitoring stations and tens of miles from the farthest, the effect of flow on water quality at down-stream monitoring stations could be delayed by several days to weeks. In an attempt to capture the potential delayed effects as pulses of fresh water move downstream, flow lag models of up to 100 days were evaluated. The lag time in the flow-concentration relation that provided the best fit was determined by plotting correllograms of each of the averaging times described above for selected transformations of flow. The correlation coefficient between flow and concentration for one-day lags up to 100 days was calculated and the r value was plotted against the lag time to determine when in the lag series the best fitting model, judged from the highest r value, occurred. With regard to lag times, the study concluded that "The upper estuary lag times are generally accommodated by a lag of 4-7 days. Down estuary lag times are on the order of 4 to 7 weeks" (Alden, *et al.*, unpublished). Following discussions with the entire DAWG, it was recommended that the following lag times be used in the flow adjustment program: tidal fresh zone - 3- and 7-day lags; oligohaline zone - 3-, 7-, and 14-day lags; mesohaline and polyhaline zones - 7-, 14-, and 21-day lags.

In *A Study of Trends in Total Phosphorus Measurements at NASQAN Stations* (Smith *et al.*, 1982) 11 concentration-flow models were described, including the following:

Linear (Lin)	$P = F$
Logarithmic (Log)	$P = \log e\ (F)$
Inverse (Inv)	$P = 1 / F$
Hyperbolic	$P = 1 / (1 + 10^{x\beta^*} \times F)$

where:

P = the water quality parameter
F = the freshwater discharge
x = a value from -2.5 to 1.5 at increments of 0.5
β^* = the integer value of the $\log_{10}$ transformed grand mean flow for the station in question.

Eight separate hyperbolic flow transformations were evaluated (denoted as Hyp-1 to Hyp-8), one for each increment.

The DAWG study group compared the 11 models by calculating an F-statistic using the following formula:

$$F = (SSR1 - SSRa) / (SSE1 / d.f.)$$

where:

SSR1 = the regression sum of squares from the model producing the highest R^2
SSE1 = the error sum of squares from the best model
d.f. = the degrees of freedom

With regard to the 11 potential transformations of flow, the study group found that:

> Within each of the four salinity regimes, the selection frequency was highest for either the Hyp-4 or Log transformations. A total of 72% of the station/parameter combinations analyzed had significant relationships with either the Hyp-4 or Log transformations of flow. The selection frequency was calculated a second time excluding all station/parameter combinations which had the Hyp-4 or Log transformations selected as the best or an alternative transformation. For these cases, the selection frequency was highest for either the Hyp-3 or Hyp-5 transformations. Nearly 85% of the remaining station/parameter combinations with significant relationships to flow had either the Hyp-3 or Hyp-5 as the best or an alternative transformation. Therefore, the Hyp-3, Hyp-4, Hyp-5, and Log transformations were selected as either the best or an alternative model for over 90% of the station/parameter combinations with significant relationships to flow (Alden, *et al.*, unpublished).

Based on these results, the study group recommended that only the Hyp-3, Hyp-4, Hyp-5, and natural log transformations be used for flow adjusting the data.

A Statistical Analysis Software (SAS) program was written that calculated R^2 statistics for each averaging time, lag, and flow transformation appropriate for the stations in a given salinity zone. The flow adjusted residuals from the model that produced the highest R^2 were then output to a data set and evaluated for trends.

iii. Trend Tests

The seasonal Kendall test for trend was used for all trend analyses (Helsel and Hirsch, 1992; Hirsch and Slack, 1984; Gilbert, 1987). This test, adopted by the U.S. Geological Survey (USGS, 1993a), is a non-parametric test that addresses the confounding effects of seasonal differences in water quality data. Use of a non-parametric test of trend for water quality data was preferred since the data tend not to be normally distributed (often positively skewed). The test was run on monthly medians. To compensate for the potential presence of serial correlation in the data sets (which would increase the probability of Type I error, *i.e.*, falsely rejecting the hypothesis of no trend), a conservative p-value was chosen for evaluation of trends (Alden *et al.*, 1992). Trends are reported as statistically significant at the $p \leq 0.01$ level for non-flow adjusted data and $p \leq 0.05$ for flow adjusted data.

In addition to evaluating the presence of a significant trend in the water quality data, it is also of interest to estimate the magnitude of change for those cases where a significant trend has been discerned. The Sen estimate of slope, a non-parametric slope estimator, is used with the Seasonal Kendall test to assess the magnitude of change in the data series. The Sen estimate of slope is the median of all the differences that are calculated in the Seasonal Kendall trend test. Although the Seasonal Kendall test and the direction of the Sen estimate of slope are robust to data sets with large numbers of censored data, the magnitude of the slope estimate is likely to be biased when the level of censoring is high. Note that the presence of a limited number of censored data points (less than approximately 5%) is not likely to affect the magnitude of slope to any significant degree (Hirsch, *et al.*, 1991).

The percent change for statistically significant trends was calculated using the equation:

Percent change = (Sen Slope Estimate * Years) / (Initial Median Concentration) * 100.

"Initial" values reported here are medians of the first two years of data in the time series. "Median" values reported here were calculated by determining the monthly medians for the time period in question, then taking the median of this reduced data set.

Assessing the significance level, and the direction and magnitude of trend in flow adjusted data is a straightforward process that uses the procedures described above for non-flow adjusted data provided that the data record is un-censored or contains only a limited number of censored data points. Currently, there is no non-parametric test for assessing monotonic trends in censored flow adjusted data. The difficulty with using the Seasonal Kendall test to estimate the significance of trend in censored flow adjusted data concerns how "ties" are broken when the adjustment for flow is made. In the calculation of variance, censored data are treated as ties, thus reducing the amount of information in the data set. Adjusting for flow breaks the ties, thus giving the false impression that the data set contains more information than is actually the case. The reduction in the variance associated with breaking the ties may lower the trend p-value leading the analyst to falsely reject the null hypothesis of no trend.

To solve the problem of assessing trends in censored flow adjusted data, the DAWG decided to use Tobit regression (Tobin, 1958) in those cases where the number of censored data points exceeded 5%. Tobit regression estimates the parameters of a linear regression model that relates the response variable (*e.g.*, TN, SECCHI depth) to explanatory variables such as time, season, and flow using maximum likelihood estimation (MLE) (Hirsch *et al.*, 1991). Tobit regression was performed using the SAS implementation PROC LIFEREG, which fits a parametric model to left- or right-censored data (water quality data may be left-censored). Because PROC LIFEREG fits a parametric model the data, it was recommended that the response variable first be transformed using the natural log transformation to better meet the normality assumption of the model (Elgin Perry, personal communication). The data were transformed by specifying the LNORMAL option in the MODEL statement.

The percent change from log transformed data was calculated by estimating the difference between the logarithm of the final parameter estimate and the logarithm of the baseline parameter estimate using the following equation:

$$(e^{\beta_2(t_2 - t_1)} - 1) \times 100$$

where:

β_2 = the slope estimate for time
t_2 = the end of the period of observation
t_1 = the beginning of the period of observation

iv. Tests of Homogeneity

The Seasonal Kendall test has the implicit assumption that the trend is homogeneous across the seasons of interest (months). That is to say, the trend for all of the months will be moving in the same direction, or at least the direction will not vary from month to month to any significant degree. In some cases however, the trend in a water quality variable may be increasing for some months and decreasing for others. When there is heterogeneity between months in a test of annual trend the results may be misleading. In addition, for the Potomac River, where multiple stations are located in each salinity zone, global statements about water quality may also be of interest. Basin wide statements about trends can be made if the trend is homogeneous across seasons and stations for all seasons and stations included in the test.

Tests of homogeneity for season, station, and season-station interaction have been developed by combining the results of the Seasonal Kendall test into chi-square statistics following the procedure of van Belle and Hughes (van Belle and Hughes, 1984). In cases where the season, station, and season-site interaction chi-square statistics are not significant (*i.e.*, the null hypothesis of homogeneity is not rejected), then a statement about a global or basin wide trend can be made. If seasons are heterogeneous, but the stations are homogeneous, trends can be analyzed for individual seasons. For cases where sites and seasons are heterogeneous or a significant site-season interaction exists, then the only meaningful trend test will be for individual site-seasons (van Belle and Hughes, 1984). Although the focus of this report is trends at individual stations, significant trends within a salinity zone will be described based on the results of the chi-square homogeneity statistics.

v. Seasons Evaluated for Trends

Large-scale annual trend analyses for the tidal Potomac have been completed as part of other investigations (Tributary Strategy Teams and State of the Bay reports); however, to date there have been no attempts to evaluate trends for multiple seasons. One component of the current project involves examining a variety of temporal scales and seasons to establish links between changes in water quality and changes in living resources. To achieve that goal, water quality trends corresponding to a variety of living resources seasons were evaluated. The seasons, salinity zones, and rationale for selecting the season are provided in Table II-4.

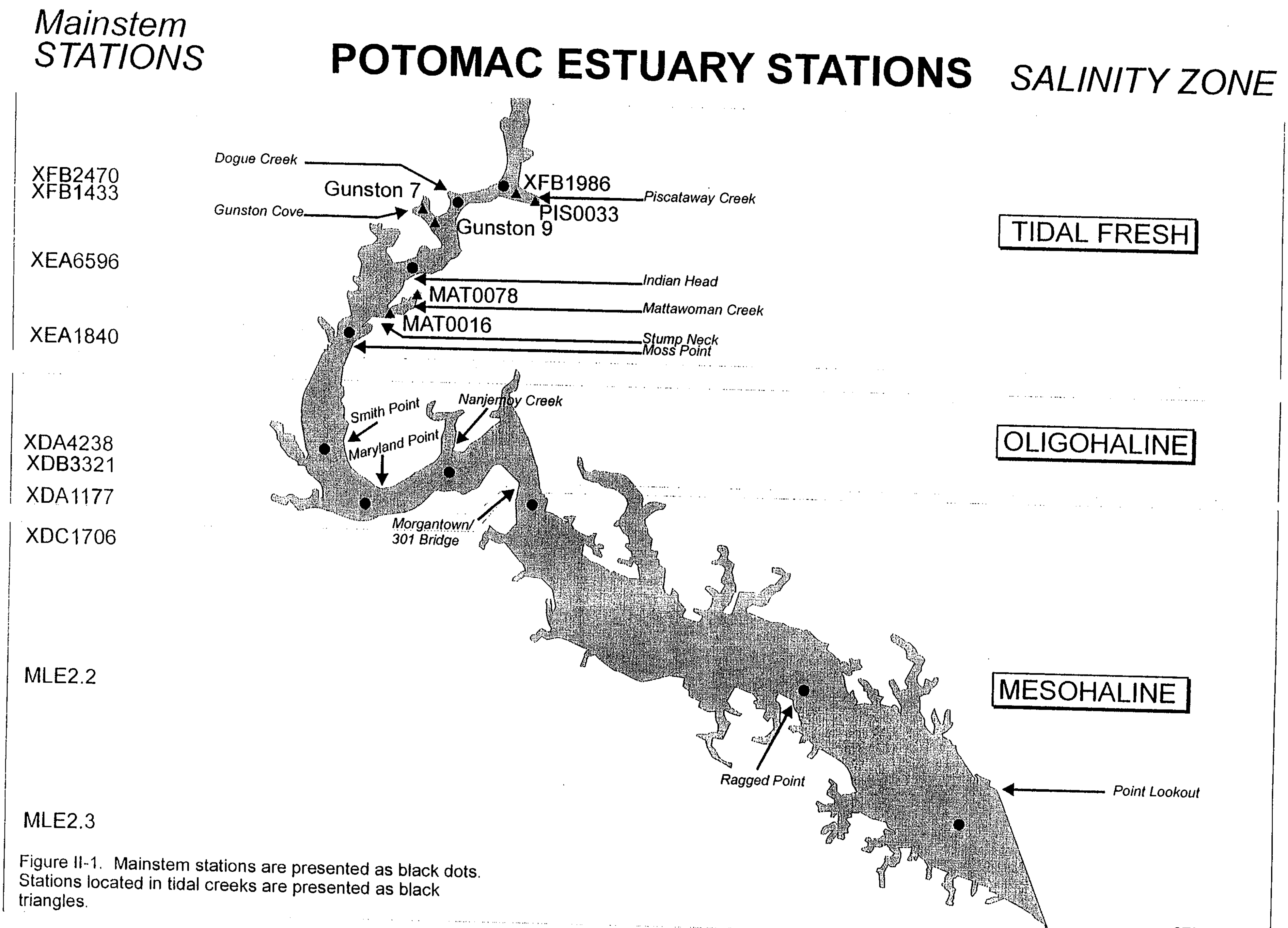


Figure II-1. Mainstem stations are presented as black dots. Stations located in tidal creeks are presented as black triangles.

STATMAP.CDR

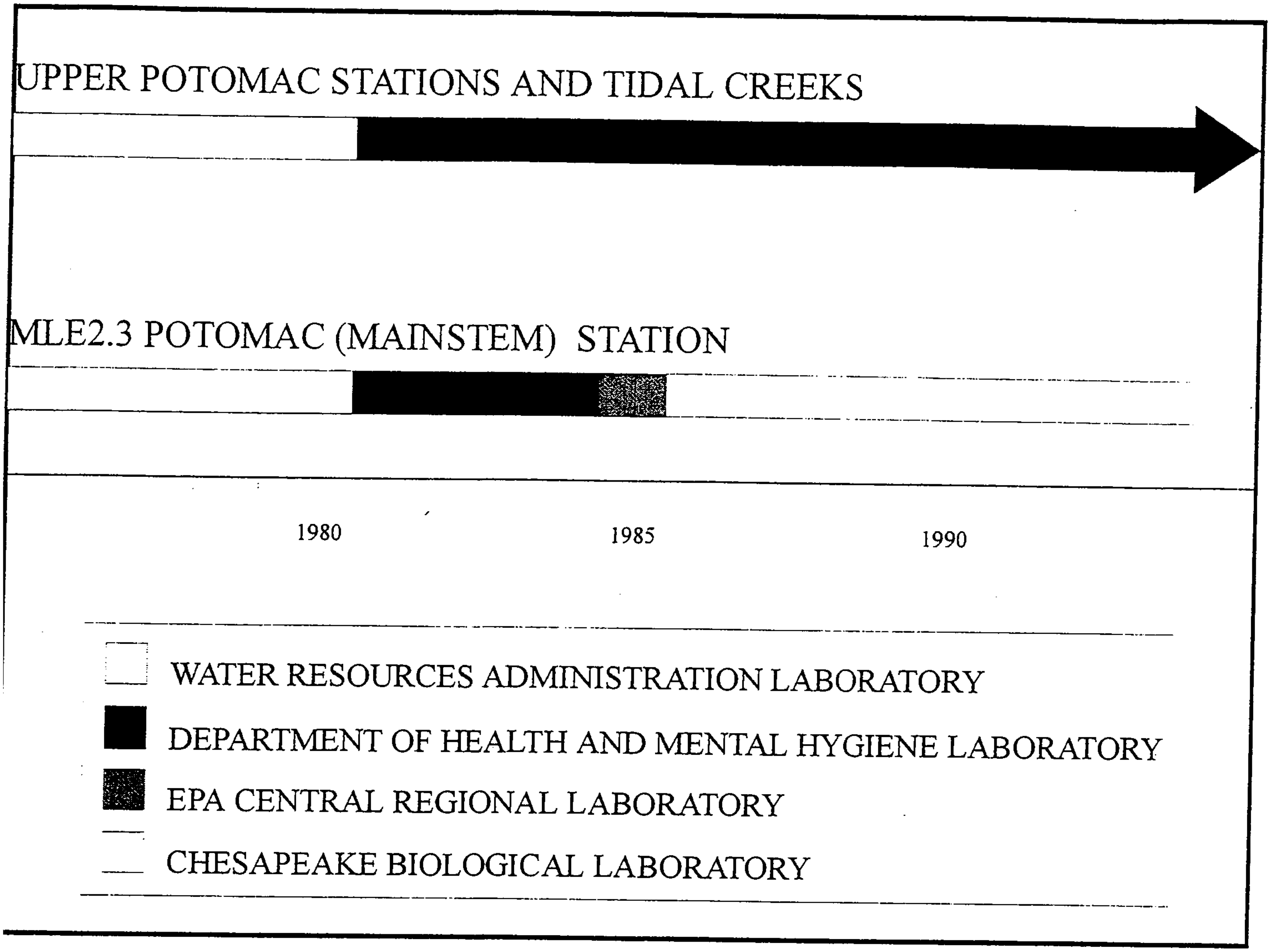


Figure II-2. Succession of laboratories that have analyzed water quality samples from the Potomac mainstem and Maryland tidal creeks.

FIGII2.CDR

Table II-1. Laboratory Analysis of Water Quality Parameters.

Parameter	Component	Laboratory	Method
TP	TP	DHMH	block digestion EPA, 1979, method 365.4
TP	TP	CBL (Oct 86 - Sep 87)	acid persulfate EPA, 1979, method 365.4
TP	PP	CBL	Aspila *et al.*, 1976
TP	TDP	CBL	D'Elia *et al.*, 1977 persulfate oxidation
PO4	PO4	ALL LABS	EPA, 1979. Method 365.1; colorimeric; automated ascorbic acid
TN	TKNW	CRL	helix digestion, EPA, 1979, method 351.2; colorimetric automated phenate
TN	TKNW	DHMH	block digestion, EPA, 1979, method 351.2
TN	TKNW	CBL (Oct 86 - Sep 87)	EPA, 1979, method 351.1
TN	NO23	ALL LABS	EPA, 1979, method 353.2
TN	PN	CBL	Leeman Labs Inc., 1988
TN	TDN	CBL	D'Elia *et al.*, 1977, persulfate oxidation
DIN	NO23	ALL LABS	EPA, 1979, method 353.2
DIN	NH4	ALL LABS	EPA, 1979, method 350.1
TSS	TSS	ALL LABS	APHA, 1981. (Sect. 209D, p. 94., gravimetric)
CHLA	CHLA	DHMH	APHA, 1981. (Sect. 1002G, pp. 50-54. spectrophotometric; trichromatic)

Table II-2. Chronology of Detection Limits for Water Column Parameters in Chesapeake Bay Tributary Monitoring Program.

Parameter	18 Jun 84 - 31 May 86	01 Jun 86 - Present
Ammonium (mg/l as N)	0.02 mg/l	0.008 mg/l
Nitrate + Nitrite (mg/l as N)	0.02 mg/l	0.02 mg/l
Nitrite (mg/l as N)	0.002 mg/l	0.002 mg/l
Total Kjeldahl Nitrogen unfiltered mg/l as N	0.1 mg/l	0.1 mg/l
Total Dissolved Phosphorus (mg/l as P)	0.01 mg/l	0.01 mg/l
Orthophosphate (PO_4) (mg/l as P)	0.01 mg/l	0.004 mg/l
Total Phosphorus (mg/l as P)	0.01 mg/l	0.01 mg/l
Total Suspended Solids (mg/l)	1.0 mg/l	1.0 mg/l
Chlorophyll *a* (mg/l)	0.01 µg/l	0.01 µg/l

Table II-3. Chronology of Detection Limits for Water Column Parameters Applicable to Station MLE2.3

Parameter	18 Jun. 84 - 28 Feb. 85	01 Mar. 85 - 15 May. 85	16 May. 85 - 30 Sep. 86	01 Oct. 86 - 31 Sep. 87	01 Oct. 87 - 19 Sep. 88	20 Sep. 88 - Present
Ammonium (mg/l as N)	0.02 mg/l 0.04 mg/l (Feb. 85 only)	0.003 mg/l	0.003 mg/l	0.003 mg/l	0.003 mg/l	0.003 mg/l
Nitrate + Nitrite (mg/l as N)	0.04 mg/l	0.0009 mg/l	0.0009 mg/l	0.0009 mg/l	0.00015 mg/l	0.0002 mg/l
Nitrite (mg/l as N)	0.01 mg/l	0.0005 mg/l	0.0005 mg/l	0.0005 mg/l	0.00015 mg/l	0.0002 mg/l
Total Kjeldahl Nitrogen (TN) unfiltered mg/l as N	0.2 mg/l	0.2 mg/l	n.a.	0.2 mg/l - 5.0 mg/l	n.a.	n.a.
Particulate Nitrogen (mg/l as N)	*c	*c	0.001 mg/l	*c	0.0001 mg/l	0.0105 mg/l
Total Dissolved Nitrogen (mg/l as N)	*c	*c	0.03 mg/l	*c	0.02 mg/l	0,02 mg/l
Total Dissolved Phosphorus (mg/l as P)	0.12 mg/l 0.010 mg/l (Feb. 85 only)	0.005 mg/l	0.005 mg/l	0.012 mg/l	0.001 mg/l	0.001 mg/l
Orthophosphate (P04) (mg/l as P)	0.007 mg/l 0.012 mg/l (Feb. 85 only)	0.0016 mg/l	0.0016 mg/l	0.0016 mg/l	0.001 mg/l	0.001 mg/l
Total Phosphorus (TP) (mg/l as P)	0.012 mg/l 0.010 mg/l (Feb. 85 only)	0.005 mg/l	*c	0.012 mg/l	*c	*c
Participate Phosphorous (mg/l as P)	*c	*c	.0013 mg/l	*c	0.0012 mg/l	0.0012 mg/l
Total Suspended Solids (mg/l)	4 mg/l	4 mg/l	1 mg/l	1 mg/l	1.98 mg/l	1.5 mg/l
Chlorophyll a (ug/l)	0.01 ug/l	0.01 ug/l	0.01 ug/l	0.01 ug/l	0.01 ug/l	0.01 ug/l

*c = calculated value
n.a. = not analyzed for and not calculated

Table II-4. Trend Analysis Seasons and Salinity Zones.

Season	Salinity Zone	Rationale
Annual	Tidal fresh	Management actions
	Oligohaline	Management actions
	Mesohaline	Management actions
March - May	Tidal fresh	Microzooplankton, phytoplankton
	Oligohaline	Microzooplankton, phytoplankton
	Mesohaline	Microzooplankton, phytoplankton
April - June	Tidal fresh	Mesozooplankton
June - September	Tidal fresh	Phytoplankton
July - September	Tidal fresh	Mesozooplankton, fish, phytoplankton
	Oligohaline	Mesozooplankton, fish, phytoplankton
	Mesohaline	Mesozooplankton, fish phytoplankton
April - October	Tidal fresh	Submerged aquatic vegetation
	Oligohaline	Submerged aquatic vegetation
October - December	Tidal fresh	Phytoplankton
	Oligohaline	Phytoplankton
	Mesohaline	Phytoplankton
March - November	Mesohaline	Phytoplankton

III. TRENDS IN NUTRIENTS AND BIOLOGICALLY MEDIATED WATER QUALITY VARIABLES

The Potomac estuary is conceptually divided into three zones based on long-term salinity measurements according to the Venice System (Symposium, 1959). The upper-most region is termed tidal fresh because the river is tidally influenced and has a salt concentration of less than 0.5 ppt. Salt concentrations in the transition or oligohaline zone range from 0.5 ppt to less than 5 ppt. The lower estuarine or mesohaline zone has salt concentrations that range from 5 to 18 ppt. Water quality monitoring stations are located in each of the three salinity zones of the Potomac estuary. The station name, longitude and latitude, location, and depth of the tidal fresh, oligohaline, and mesohaline stations are presented in Table III-1.

As an aid to visualizing changes in water quality over the period of record, monthly medians of surface data are presented for the more important water quality indicators, including: TP, PO_4, TN, DIN, TSS, Secchi depth, and CHAA. Bottom DO values are also presented. Figures are presented for the following stations: XFB2470 (Figures III-1a and III-1b), XFB1433 (Figures III-2a and III-2b), XEA6596 (Figures III-3a and III-3b), XEA1984 (Figures III-4a and III-4b), XFB1986 (Figures III-5a and III-5b), PIS0033 (Figures III-6a and III-6b), MAT0016 (Figures III-7a and III-7b), MAT0078 (Figures III-8a and III-8b), Gunston 7 (Figures III-9a and III-9b), Gunston 9 (Figures III-10a and III-10b), XDA4238 (Figures III-11a and III-11b), XDA1177 (Figures III-12a and III-12b), XDB3321 (Figures III-13a and III-13b), XDC1706 (Figures III-14a and III-14b), MLE2.2 (Figures III-15a and III-15b), MLE2.3 (Figures III-16a and III-16b).

Please note that in the following discussion of trends, "significant" means statistically significant at $p \leq 0.01$ for non-flow adjusted trends and at $p \leq 0.05$ for flow adjusted trends.

A. Non-flow Adjusted Surface Trends for 1985-1996

The following sections describe surface, non-flow adjusted trends for the 1985-1996 time period for the stations located in the mainstem, Piscataway Creek, Matawoman Creek, and Gunston Cove. Detailed information on surface, non-flow adjusted trends for 1985-1996 is presented in Appendix A.

i. Tidal Fresh Zone

a. Mainstem Stations

Significant trends in TP were observed at two of the four tidal fresh mainstem stations (XFB2470 and XFB1433). Although no annual trends were detected, significant seasonal trends were found for June-September, July-September, and April-October. The increasing seasonal trends ranged from 36% to 45%. Despite the lack of significant annual trends at individual stations, a significant segment-wide trend was observed. There were no significant annual or seasonal trends for PO_4W. A segment-wide significant PO_4W trend was detected; however, the null hypothesis of homogeneity for season was rejected (trends across the stations are decreasing in some months while in other months trends are increasing). This would indicate that a segment-wide statement about trends would not be appropriate. Since the seasons are heterogeneous and the stations are homogeneous, the focus of a segment-wide trend analysis should be on individual months and not annual trends. For PO_4F, there were no significant annual, seasonal, or segment-wide trends.

Significant annual trends in TN were detected at XFB1433 (-20%) and XEA6596 (-13%). A decrease of 19% was observed for the April-October season at XFB2470. The segment-wide TN trend was significant and homogeneous for station, season, and station-season interaction. Despite the absence of annual and seasonal trends in DIN, a significant segment-wide trend was observed that was homogeneous for station, season, and station-season interaction.

No annual, seasonal, or segment-wide trends were detected for NO_{23}, a component of both TN and DIN. For NH_4, significant decreasing annual trends were observed at all four tidal fresh mainstem stations and significant decreasing seasonal trends were observed at three. The decreases ranged from 48% at XEA6596 to 70% at XFB2470 and XFB1433. Seasons with significant trends include March-May, April-June, April-October, and October-December. Of the 13 overall significant NH_4 trends, seven were associated with greater than 5% below detection limit data censoring. Although the direction of trend is unaffected by large numbers of censored data, the magnitude of the slope estimate is likely to be in error and therefore the percent change should be interpreted with caution (Hirsch, *et al.*, 1991). The large number of significant station trends resulted in a significant segment-wide trend that was homogeneous for season, station, and station-season interaction.

Five significant trends were detected for TSS in the tidal fresh mainstem, two at XFB2470 (June-September and July-September) and three at XFB1433 (annual, June-September, and April-October). The percent changes were all positive and ranged from 40% to 60%. There was a significant segment-wide annual trend, however seasonal heterogeneity was present. As a result, segment-wide trend analyses should be conducted across stations on a monthly as opposed to an annual basis. No significant trends in SECCHI depth were observed in the tidal fresh zone in spite of the significant increases in TSS.

There were numerous significant trends in CHAA, all of which were positive and some of which were quite large. Annual and numerous seasonal trends were significant at the upper-most station (XFB2470), which had trends ranging from 120% (annual) to190% (October-December). Other significant seasonal trends at XFB2470 included: June-September, July-September, and April-October. An increase of 290% occurred during the October-December season at XFB1433. No significant trends were observed at the lower two stations (XEA6596 and XEA1840). The CHAA segment-wide trend was significant and homogeneous for season, station, and station-season interaction in the tidal fresh zone.

Significant increasing annual and seasonal trends of either 10% or 20% were observed in surface

DO_FLD at the upper two tidal fresh stations. For XFB2470, the seasons in which DO_FLD trends were observed closely matched by those of CHAA. Only the annual and March-May trends were significant at XFB1433. The segment-wide DO_FLD trend was significant and homogeneous for season, station, and station-season interaction.

Significant trends in PH_FLD were observed at the upper three tidal fresh stations. Increasing trends of 10% were detected at XFB2470 for the annual trend and all except one seasonal trend (October-December), which had a percent change of 0 due to rounding. Increasing trends of 10% were also observed at XFB1433 for the annual trend and all seasons except July-September and October-December. Trends at XEA6596 were significant but have a percent change of 0 due to rounding. The segment-wide annual trend was significant for PH_FLD; however, the null hypothesis of homogeneity was rejected for season. As a result, segment-wide trend analyses should be conducted across the stations on a monthly and not annual basis.

TALK increased seasonally at three stations and annually at one. The increases ranged from 10% to 60%. An increase of 10% occurred during April-October at XFB1433, which was the only significant change at that station. The only annual increase (20%) was observed at XEA6596. The largest increase (60%) occurred at XEA1840 during July-September. The segment-wide trend in TSS was significant and homogeneous for season, station, and station-season interaction.

b. Piscataway Creek

Only one significant TP trend was observed at Piscataway Creek, a 58% increase at XFB1986 for April-October. No significant trends were detected for PO_4W at either Piscataway Creek station; however, three significant seasonal trends were observed at PIS0033, the upper station, for PO_4F. The seasonal decreases, which occurred during June-September, July-September, and April-October, were 64%, 74%, and 50%, respectively. Segment trends were not significant for either whole or filtered PO_4.

Significant decreasing annual TN trends were detected at both Piscataway stations. Seasonal trends (June-September, July-September, and April-October) were only observed at XFB1986. TN decreases in Piscataway Creek ranged from 25% to 33%. The significant annual segment-wide TN trend was homogeneous for season, station, and station-season interaction. Significant annual and seasonal DIN trends were observed at both stations. The seasonal trends were observed during June-September, July-September, and April-October at PIS0033 and XFB1986. The decreases in DIN ranged from 26% to 65%. The basin-wide trend was significant and homogeneous for season, station, and station-season interaction.

Significant annual and seasonal trends in NO_{23} ranging from -21% to -41% were observed only at XFB1986. Seasonal trends occurred during June-September, July-September, and April-October. The segment-wide trend statistic was also significant and homogeneous, despite having an annual trend at only one station. Both Piscataway Creek stations had significant annual and seasonal trends in NH_4. Decreases in NH_4 ranged from 39% to 88%. High numbers of censored data may have resulted in a biased slope estimate at one station (XFB1986) for April-October trends. Other seasons with significant trends include: April-June, June-September, and July-September. Segment-wide trends for NH_4 were significant and homogeneous for season, station, and station-season interaction.

Significant annual and seasonal trends in TSS were observed at XFB1986. TSS increased by 60% for annual and two seasonal (June-September and April-October) trends and by 70% for the July-September season. Annual segment-wide trends were also significant and homogeneous despite the absence of a significant annual trend at PIS0033 ($p = 0.02$). PIS0033 is located along a portion of Piscataway Creek that is too shallow to measure SECCHI depth. Although SECCHI depths decreased for annual and most seasonal trends at the remaining Piscataway Creek station (XFB1986) no significant trends, either annual or seasonal, were observed.

CHAA dramatically increased at XFB1986 (no trends were detected at PIS0033). The increases ranged from 140% for the annual trend to 260% for the April-October season. It is interesting to note that the time periods in which CHAA increased matched those where increasing TSS trends were found. The segment-wide CHAA trend was significant and homogeneous for season, station, and station-season interaction.

Conflicting significant trends were observed in Piscataway Creek for DO_FLD. DO_FLD decreased by 10% for annual, March-May, and April-June time periods at PIS0033. Conversely, DO_FLD increased by 20% to 40% at XFB1986. Increases in DO_FLD occurred during annual, June-September, July-September, and April-October time periods. PH_FLD increased by 10% for several time periods at XFB1986 (no significant trends were detected at PIS0033). In additional to increasing annual trends in PH_FLD at XFB1986, significant increases were also observed for the following seasons: April-June, June-September, July-September, April-October. Although the segment-wide trend for PH_FLD was significant, the null hypothesis of homogeneity across stations was rejected. As a result, annual trends should be conducted at individual stations.

Significant increases of 60% for TALK were detected at PIS0033 for March-May and April-June. The segment-wide annual trend was also significant and homogeneous for season, station, and station-season interaction despite the absence of any significant trends at XFB1986.

c. Mattawoman Creek

No significant annual or seasonal TP trends were observed at either Matawoman Creek station. A significant annual trend of 67% was detected for PO_4W at MAT0078. No other PO_4W trends were significant. PO_4F increased dramatically at MAT0016 for the annual trend (179%) and for the October-December season (417%). Please note that the number of censored data points for both PO_4F trends exceeded 5%. As a result, although the direction of trend is unaffected, the magnitude may be in error.

Significant annual decreases of 15% and 23% were observed in TN at MAT0016 and MAT0078, respectively. TN also decreased at MAT0078 for the July-September (34%) and April-October (22%) seasons. The segment-wide annual TN trend was significant and homogeneous for season, station, and station-season interaction. No significant annual or seasonal trends were detected for DIN or NO_{23} at either Matawoman Creek station.

Decreasing NH_4 trends of 30% and 35% were detected for the annual and April-October time periods at MAT0016. In spite of the absence of a significant annual trend for MAT0078, a significant homogeneous segment-wide trend was observed. No significant trends were observed at either station for the following variables: TSS, SECCHI, CHAA, DO_FLD, and PH_FLD.

Increasing TALK trends of 40% were detected at MAT0016 for the annual and April-October time periods. The basin-wide annual TALK trend was also significant and homogeneous, despite the absence of a significant annual trend for MAT0078.

d. Gunston Cove

Decreasing trends in TP were observed at Gunston 7 for all time periods except October-December. The TP decreases ranged from ranged from 24% to 60%. No significant TP trends were detected at Gunston 9, which is located at the mouth of the Cove, near the Potomac mainstem. Although the segment-wide TP trend was significant, the null hypothesis of homogeneity was rejected for station. Consequently, annual trends should analyzed for individual stations only. No significant trends in PO_4 were observed at either Gunston Cove station.

Significant TN changes of from -34% to -45% were observed at Gunston 7 for four time periods (annual, April-June, June-September, and April-October). Despite the absence of significant TN trends at Gunston 9, the overall segment-wide trend was significant and homogeneous. Significant annual DIN trends of -39% and -26% were detected at Gunston 7 and Gunston 9, respectively. The remaining significant trend of -56% occurred at Gunston 7 during the April-October time period. The segment-wide DIN trend was significant and homogeneous for station, season, and station-season interaction.

Significant NO_{23} trends were found only at Gunston 9, the near-mainstem station. NO_{23} trends for the annual and June-September time periods were -19% and -31%, respectively. Significant decreasing NH_4 trends of 81% and 85% were detected at Gunston 9 for the March-May and April-June time periods, respectively. The magnitude of both NH_4 trends may be biased because the number of detection limited data exceeded 5%. No other significant NH_4 trends were observed.

A decreasing TSS trend of 30% was observed at Gunston 7 for the April-October season. No other significant TSS trends were found. A number of significant SECCHI depth trends were observed at Gunston 7 including an annual trend of 60% and several seasonal trends. The annual trend should be interpreted with caution since it is not homogeneous across all months (most months SECCHI depth increased, but for some it decreased). Increases ranging from 70% to 110% were observed for four living resources seasons (June-September, July-September, April-October, and October-December. Despite the absence of significant trends at Gunston 9, the segment-wide trend was significant, but heterogeneous for season. Rejection of the null hypothesis of homogeneity for season indicates that segment-wide trends should be analyzed for individual months.

Three increasing CHAA trends (annual, April-October, and October-December) were detected at Gunston 9; no CHAA trends were observed at Gunston 7. Increases ranged from 110% for annual and April-October to a large 590% increase during October-December. Although CHAA trends were absent at Gunston 7, the segment-wide trend was significant and homogeneous for season, station, and station-season interaction.

Three increasing trends were also observed for DO (annual, March-May, and April-June) at Gunston 9. DO increased by 10% for the annual trend and by 20% for the living resources seasons. The DO segment-wide trend was significant and homogeneous, despite the absence of significant DO trends at Gunston 7. No significant trends were detected for either pH or TALK at either Gunston Cove station.

ii. Oligohaline Zone

No significant trends were observed for TP at the oligohaline stations. PO_4W decreased for the annual and April-October time periods by 44% and 40%, respectively. Although significant PO_4W trends were observed only at one of three oligohaline zone stations, the segment-wide trend statistic was significant and homogeneous. No significant trends were observed for PO_4F at either oligohaline zone station where filtered samples are collected (sampling at XDB3321 ceased in September 1990, just prior to the analysis of filtered samples).

Significant TN trends were detected only at XDB3321 for the annual, March-May, July-September, and April-October time periods. TN increases over the approximately 5-year (1985-1990) period ranged from 19% to 62%. DIN also increased at XDB3321 for annual trend (69%), July-September (279%), and April-0ctober (125%). Note that the magnitude of the 279% DIN increase may be biased because the number of censored data points exceeded 5%. NO_{23}, a major component of both TN and DIN, increased at XDB3321 for the same time periods described above for DIN. Increases ranged from 74% to 283%.

Significant decreasing annual trends of 45% and 30% in NH_4 occurred at XDA4238 and XDA1177, respectively. Seasonal trends ranging from -40% to -79% were observed at XDA4238 for July-September, April-October, and October-December. All significant trends had high numbers of censored data; therefore, although the trend direction is accurate, the magnitude of change may be biased. The basin-wide trend statistic for the oligohaline zone was significant, but non-homogeneous for station. Rejection of the null hypothesis of homogeneity for station indicates that trends should be run on individual stations only. Non-homogeneity across stations probably resulted from the absence of data for XDB3321 after September 1990.

Significant annual and seasonal (April-October) trends were observed at both XDA4238 and XDA1177. The annual trends were 50% and 40% for XDA4238 and XDA1177, respectively. Both seasonal trends were increases of 50%. The segment-wide TSS trend was significant and homogeneous. Although significant increasing TSS trends were observed in the oligohaline zone, trends in SECCHI depth were not significant. A significant CHAA trend of 100% was observed for the July-September time period at XDA4238.

No significant trends in DO_FLD were found in the oligohaline zone. PH_FLD increased during the annual, July-September, and April-October time periods at both XDA4238 and XDA1177. The percent changes were either 0% (due to rounding) or 10%. The segment-wide trend statistic was significant; however, the trend was heterogeneous for season (some months increased and others decreased). As a result, segment-wide trends results should be examined on a monthly basis.

Time periods and stations with increasing TALK trends correspond to those with increasing PH_FLD trends. The annual TALK trends increased by 30% at both XDA4238 and XDA1177. The July-September trends increased by 70% and the April-October trends increased by 40%. The segment-wide statistic for annual trend was significant, but heterogeneous for station. Consequently, trends should be analyzed at individual sites.

iii. Mesohaline Zone

There are no significant trends in TP or PO_4W at any mesohaline zone station (note that PO_4W is not available at MLE2.3 because data for the1985-1996 time period were analyzed at CBL under the Bay Program sample analysis protocols which specified filtration). Only one significant PO_4F trend (-41%) was observed in the mesohaline zone at MLE2.3 for the October-December living resource season.

TN increased by 19% for March-November at MLE2.3. DIN decreased by -60% at MLE2.3 for the October-December season; however, the number of censored data (5.6%) may have resulted in a slightly biased trend magnitude. No significant trends were detected for NO_{23} in the mesohaline zone. Three significant trends were detected for NH_4 in the mesohaline zone, all at MLE2.3. Decreases of 73% were observed for annual and March-November and a decrease of 67% occurred during October-December. All three trends were associated with high numbers of censored data, so the trend magnitudes may be biased.

Several significant TSS trends were observed at XDC1706 and MLE2.2; there were no significant TSS trends at MLE2.3. TSS changes at XDC1706 ranged from 90% for March-May to 110% for July-September. Significant trends were also observed for the annual and March-November time periods. At MLE2.2, TSS increases ranged from 70% for March-November to 170% for October-December. For annual trend, TSS increased at MLE2.2 by 90%. Two significant trends (annual and March-November) in SECCHI depth were observed in the mesohaline zone, both of -20% at MLE2.3. There were no significant trends in CHAA in the mesohaline zone. The absence of CHAA trends and the prevalence of TSS trends could indicate that increased TSS concentrations may be attributable to the inorganic fraction.

A 10% decrease in DO_FLD was observed at MLE2.3 for the annual time period. Significant changes, of either 0% or 10% (due to rounding), in PH_FLD were detected at MLE2.2 (no other mesohaline zone station had significant PH_FLD trends). Time periods during which significant trends were observed include: annual, July-September, October-December, and March-November. Despite the absence of significant annual PH_FLD trends at the remaining mesohaline zone stations, the segment-wide trend statistic was significant, though not homogeneous for season. As a result, segment-wide trends should be performed on individual months.

TALK analyses are only performed at XDC1706, where significant trends of 20% and 30% were observed. Time periods with significant trends included annual, October-December, and March-November.

B. Non-flow Adjusted Bottom Trends for 1985-1996

The following sections describe bottom, non-flow adjusted trends for the 1985-1996 time period for the stations located in the mainstem, Piscataway Creek, Matawoman Creek, and Gunston Cove. Detailed information on bottom, non-flow adjusted trends for 1985-1996 is presented in Appendix A.

i. Tidal Fresh Zone

a. Mainstem Stations

There are no significant annual or seasonal bottom trends for TP, PO_4W or PO_4F. Seasonal or annual TN trends were detected at three of the four tidal fresh mainstem stations (XFB2470, XFB1433, and XEA6596). Changes ranged from -14% at XEA6596 (April-June) to -19% at XFB1433 (April-June). Annual decreases for XFB2470 and XFB1433 were 15% and 16%, respectively. The remaining seasonal TN trend, a 16% decrease, occurred at XFB1433 during April-October. The segment-wide trend statistic was also significant and homogeneous for station and season. Although no DIN trends were significant at $p \leq 0.01$ (criterion for non-flow adjusted trends), a significant, homogeneous segment-wide annual trend was observed.

No significant NO_{23} trends were found at any tidal fresh station; however, significant annual and seasonal trends for NH_4 were found at all tidal fresh stations. The decreases in NH_4 ranged from a 50% October-December trend to an 88% July-September trend, both at XEA1840. Additional significant trends were detected at XFB2470 (annual, March-May, April-June, and April-October); XFB1433 (annual, March-May, April-June, April-October); XEA6596 (annual, March-May, April-June, June-September, April-October, October-December); and, XEA1840 (annual, June-September, April-October). The segment-wide statistic for annual trend was also significant and homogeneous for station and season. Note that seven of the seasonal NH_4 trends, including the 88% decrease for July-September at XEA1840, were affected by censored data in excess of 5%.

Although no significant changes were observed for TSS in the tidal fresh zone, increases in CHAA, a component of TSS, were evident. The majority of the significant trends (annual, April-June, June-September, April-October), occurred at XFB2470, the upper-most tidal fresh station, where increases ranged from 110% to 130%. The remaining two significant trends occurred at XFB1433, which had increases of 100% (annual) and 190% (October-December). The segment-wide annual trend was also significant and homogeneous for season and for station, despite only two of the four stations exhibiting significant trends.

Significant trends were recorded for DO_FLD, PH_FLD, and TALK in the tidal fresh zone. DO_FLD trends were significant and homogeneous for segment-wide trend although on a station-by-station basis, trends were only observed at XFB2470 (annual, June-September, July-September, April-October) where increases of either 10% or 20% occurred. Numerous significant PH_FLD trends, all of which were 10% due to rounding, were observed at the tidal fresh zone stations. At XFB2470 and XFB1433, significant trends occurred for annual, March-May, April-June, June-September, July-September, and April-October. Annual, March-May, and April-October trends were observed at XEA6596 and XEA1840. Additional seasonal trends (June-September, July-September) were noted in XEA1840. The segment-wide trend for PH_FLD, was significant for trend and homogeneous for station, season, and station-season interaction. Fewer significant trends were found for TALK. One seasonal TALK trend (July-September) was observed at both XFB1433 and XEA6596, which had increases of 30% and 40%, respectively. Three seasonal trends (June-September, July-September, and April-October) ranging from 30% to 60% were detected at XEA1840. The segment-wide annual trend statistic for TALK was significant, but the null hypothesis of homogeneity for season was rejected. As a result, statements about segment-wide TALK trends would be misleading. Given the seasonal heterogeneity, segment-wide trend analyses could be performed across stations on a monthly as opposed to an annual basis.

b. Piscataway Creek

The only bottom data available for Piscataway Creek are for DO_FLD and PH_FLD at XFB1986 (PIS0033 is located in a very shallow area). No significant trends were found for either variable.

c. Mattawoman Creek

Mattawoman Creek also has only DO_FLD and PH_FLD data for the deeper of its two stations, MAT0016, where a 20% decreasing trend in DO_FLD was observed for the October-December season.

d. Gunston Cove

Significant decreasing trends in TP were observed at Gunston 7 for the annual, June-September, July-September, and April-October time periods. The decreases in TP ranged from 36% for the annual trend to 43% for the July-September trend. No TP trends were detected at Gunston 9. Despite of the absence of significant TP trends at Gunston 9, the segment-wide trend statistic was significant and homogeneous. No significant trends in PO_4 were detected at either Gunston Cove station.

Annual, June-September, and April-October TN trends were significant at Gunston 7, where decreasing TN trends ranged from 35% to 37%. Although no significant TN trends were observed at Gunston 9, the segment-wide trend statistic was significant and homogeneous. Significant decreasing trends in DIN were detected at both Gunston 7 and Gunston 9. For annual trends, DIN decreased by 38% and 27% at Gunston 7 and Gunston 9, respectively. Additional DIN trends were observed for the June-September (-35%), July-September (-33%), and April-October (-28%) time periods at Gunston 9. The segment-wide trend statistic for DIN was significant and homogeneous.

Significant trends in NO_{23} were only observed at Gunston 9, where decreases ranged from 24% for the annual trend to 43% for the July-September season. Additional significant seasonal trends were detected for the June-September (-37%) and April-October (-27%) time periods. Three significant NH_4 trends were observed in Gunston Cove. NH_4 decreased by 60% at Gunston 7 for the annual trend and by 93% and 75% for March-May and April-June, respectively at Gunston 9. Note that all three trends may be biased due to the large number of below detection limit values. The segment-wide trend for NH_4 was also significant.

No significant TSS trends were observed at either Gunston Cove station. Significant DO trends were detected only at Gunston 9 for annual, April-June, June-September, July-September, and April-October. Due to rounding, trends of either 20% or 30% were observed. The segment-wide trend statistic for DO was significant and homogeneous.

Two significant trends in PH_FLD were observed at Gunston 9. The annual trend was slight and rounded to 0. An increase of 10% was detected for the April-October season. No significant trends were observed for TALK at either Gunston Cove station.

ii. Oligohaline Zone

The only significant phosphorus trends in the oligohaline zone were decreasing annual trends for PO_4W of 38% and 30% at XDA4238 and XDA1177, respectively, indicating that orthophosphate decreased during the 1985 through September1990 period. The segment-wide annual trend statistic indicates that PO_4W trends were significant and homogeneous for station, season, and station-season interaction. No significant trends were observed for either TP or PO_4F.

One significant TN trend and three significant DIN trends were detected in the oligohaline zone. The significant TN trend (-22%) occurred at XDA4238 for the March-May season. The three increasing trends in DIN, which ranged from 72% to 155%, were observed at XDB3321 for the annual, July-September, and April-October time periods.

Three increasing NO_{23} trends were observed at XDB3321. The NO_{23} trends, which were for the same seasons as those for DIN, ranged from 95% to 265%. Significant annual and seasonal NH_4 trends were observed at XDA4238 and XDA1177. Trends at XDA4238 ranged from -42% (annual and October-December) to -69% (April-October). A decreasing trend was also observed during July-September at XDA4238. Decreasing trends of 41% and 48% were found at XDA1177 for the annual and April-October time periods, respectively. Note that all NH_4 trend magnitudes may be biased as a result of having large numbers of censored data points. The segment-wide NH_4 trend was significant; however, the null hypothesis of homogeneity for station was rejected. As a result, it would be misleading to make segment-wide statements about trends. Trend results would have to be interpreted on a station-by-station basis.

Despite the absence of any TSS trends in the oligohaline zone, increasing trends in CHAA were observed at XDA4238 for the annual (60%), July-September (200%), and April-October (100%) time periods. The segment-wide trend was significant for CHAA, but was heterogeneous for season. Consequently, no segment-wide statements should be made about annual CHAA trends. Trend analyses could be performed across stations, but only on a monthly basis. There were no significant trends for DO_FLD in the oligohaline zone.

Increasing trends of 10% in PH_FLD were detected at XDA4238 and XDA1177 for the following time periods: annual, March-May, July-September, and April-October. Significant PH_FLD trends were also observed at XDB3321, but were so slight as to be rounded to zero. The segment-wide PH_FLD annual trend statistic was significant and homogeneous for season, station, and station-season interaction. Annual and seasonal TALK trends ranging from 20% to 70% were detected in the oligohaline zone. Trends at XDA4238 and XDA1177 included annual, July-September, and April-October time periods. The segment-wide statistic was significant, but not homogeneous for station. As a result, statements about segment-wide trends would be misleading. Trends should be examined on a station-by-station basis.

iii. Mesohaline Zone

One phosphorus trend, a 250% increase in PO_4F at MLE2.2 during October-December, was detected in the mesohaline zone; however, the trend magnitude is likely quite biased since one-half of the data were censored. No significant trends were observed in TP or PO_4W in the mesohaline zone.

An increasing TN trend of 50% was observed at MLE2.2 for the March-November season. DIN decreased by -61% at MLE2.3 during the October-December season. No significant NO_{23} trends were found in the mesohaline zone. For NH_4, decreasing trends of -29% and -83% were found at MLE2.3 for the annual and October-December time periods, respectively. The annual trend may be suspect because the trend is not homogeneous for season and because the number of censored data points exceeded 5%.

MLE2.2 is the only mesohaline zone station that had significant TSS trends, where percent changes ranged from 60% to 150%. Significant changes in TSS were observed for the following time periods: annual, March-May, July-September, and March-November. Significant trends in CHAA were detected at XDC1706 for July-September (80%) and at MLE2.3 for October-December (180%). The segment-wide CHAA trend was also significant and homogeneous for season, station, and station-season interaction.

Two significant trends were observed for DO_FLD, both at XDC1706. The annual and March-November DO_FLD trends were both 20%. Two significant, though 0%, seasonal trends (July-September and March-November) were observed for PH_FLD at XDC1706. The segment-wide PH_FLD trend statistic was significant and homogeneous. There were no TALK trends for XDC1706, the only mesohaline zone station where TALK is analyzed.

C. Flow Adjusted Surface Trends for 1985-1996

Unlike the previous two sections, where non-flow adjusted trends were described for annual and a variety of living resources seasons, Sections C and D present only flow adjusted annual trends. Since living resources are affected by in situ water quality, correcting for flow would distort the conditions that living resource are exposed to and thus be misleading. Annual flow adjusted trends are summarized below to assess the impacts of management actions on water quality. Removing flow effects from the data is based on the assumption that flow is a major forcing function and source of inter-annual variability. Note that the criterion for statistical significance for flow adjusted trends is $p \leq 0.05$. The significance level was "relaxed" for flow adjusted trends since the flow adjustment reduces a major source of variability in the data. Finally, only mainstem station trend results are presented in this and the next section because of uncertainty regarding the effect of flow on stations outside of the main channel. Detailed information on surface, flow adjusted trends for 1985-1996 is presented in Appendix B.

i. Tidal Fresh Zone

There were no significant TP trends in the tidal fresh zone. PO_4W, which was significantly related to flow, decreased by 32% at XEA6596. No significant trends in PO_4F were observed.

Significant decreasing TN trends ranging from 9% to 23% occurred at XFB1433, XEA6596, and XEA1840. TN concentration was significantly related to flow at XFB1433 and XEA1840, and not related to flow at XEA6596. A significant segment-wide TN trend was observed. DIN was significantly related to flow at all four tidal fresh stations. Despite the significant DIN-flow relation at all tidal fresh stations, significant DIN trends of -12% and -21% were observed only at XEA6596 and XEA1840, respectively. A significant segment-wide, homogeneous DIN trend was observed for the tidal fresh zone.

NO_{23} was significantly related to flow at three tidal fresh stations; however, no significant time trends were observed. Significant NH_4 trends were detected at all four tidal fresh stations where decreases ranged from 43% to 62%. NH_4 was significantly related to flow at the lower three tidal fresh stations. Because of the high numbers of censored data for NH_4, trends at all four stations were analyzed using SAS PROC LIFEREG, which employs Tobit regression.

Significant flow adjustment models were found for TSS and SECCHI depth at all four tidal fresh stations; however, the time trends were not significant for either variable. Significant flow adjustment models were observed for CHAA at all four tidal fresh stations. CHAA increased at the upper three tidal fresh zone stations by between 80% and 260%. Although the segment-wide CHAA trend was significant, the trend was not homogeneous for station. As a result, segment-wide statements about trends would be misleading.

DO_FLD was significantly related to flow at all four tidal fresh stations; however, significant time trends were observed at only XFB2470 (10%) and XEA1840 (-10%). The segment-wide trend statistic for DO_FLD was not significant. Significant flow adjustment models were observed for PH_FLD at XFB2470 and XEA1840. Significant increases of 10% were observed for PH_FLD at XFB2470 and XFB1433. A significant trend in PH_FLD was also observed at XEA6596, but the change was so small as to be rounded to zero. The segment-wide trend statistic for PH_FLD was significant, but not homogeneous for season. As a result, segment-wide trends in PH_FLD should be examined on a monthly basis. Significant flow models and increasing TALK trends of 20% were observed at XFB2470, XFB1433, and XEA6596. The segment-wide TALK trend was significant and homogeneous.

ii. Oligohaline Zone

Significant flow models were found at two stations for TP in the oligohaline zone although the trends were not significant. Flow was significantly related to PO_4W at all three oligohaline zone stations. Significant decreasing trends of 42% and 18% were found at XDA4238 and XDB3321, respectively. The segment-wide trend for PO_4W was also significant and homogeneous for season, station, and station-season interaction. Flow models for PO_4F were significant, but the time trends were not.

Flow models were significant for TN at all three oligohaline stations. TN decreased by 22% and 12% at XDA4238 and XDA1177, respectively. Significant flow adjustment models were detected for DIN at all three oligohaline zone stations. Conflicting trends were found for DIN in the oligohaline zone. DIN decreased by 26% at XDA4238 and increased by 50% at XDB3321 (note that data collection at XDB3321 ceased in September 1990). The reversal in trends could indicate that reductions in DIN occurred after 1990. DIN flow adjustment models were significant at all three stations.

A pattern similar to that described for DIN also occurred for NO_{23}, where a decrease of 26% and an increase of 51% were observed for XDA4238 and XDB3321, respectively. NO_{23} flow adjustment models were significant at all three stations. For NH_4, the other component of DIN, two decreasing trends of 61% and 40% occurred at XDA4238 and XDA1177, respectively. Note that NH_4 trends were estimated using LIFEREG due to the high percentages of censored data. Flow models were significant at all three stations.

Although flow was significantly related to both TSS and to SECCHI depth, significant trends were not observed for either variable in the oligohaline zone. CHAA increased by 60% and 30% at XDA4238 and XDA1177, respectively. Neither the flow models nor the segment-wide trend were significant. Flow adjustment models at all three stations were significant for DO_FLD. Despite having a significant trend (a decrease of 10%) at only one oligohaline zone station (XDA1177), a significant, homogeneous segment-wide trend was observed.

Significant flow adjustment models were observed for PH_FLD at all three oligohaline zone stations. A significant PH_FLD time trend occurred only at XDA4238; however, the magnitude of change was negligible due to rounding. The segment-wide trend was significant but the null hypothesis of homogeneity for season; therefore, segment-wide trends should be analyzed on a monthly basis. TALK increased by 30% at both XDA4238 and XDA1177; however, a significant flow adjustment model was observed only at XDA4238. The segment-wide trend was significant, but not homogeneous for station; consequently, segment-wide statements regarding trends would be misleading.

iii. Mesohaline Zone

As previously described, nutrient samples for the mesohaline zone were analyzed at two different laboratories; XDC1706 and MLE2.2 at DHMH and MLE2.3 at CBL. One of the consequences of using different labs is having different detection limits. The data for the mesohaline were not censored to the higher detection limits of the DHMH laboratory since this would have resulted in a great loss of information at MLE2.3. Because trend comparisons cannot be made across stations with different detection limits, homogeneity statistics are not provided for nutrients in the mesohaline zone.

Although significant flow models were observed in the mesohaline zone for TP, PO_4W, and PO_4F, the time trends were not significant.

Significant decreasing trends of 19% and 42% were detected for TN and DIN, respectively at XDC1706. A decreasing trend in DIN (29%) was also detected using the LIFEREG procedure at MLE2.3. Some of the TN and DIN decreases at XDC1706 can be attributed to a significant decrease of 41% in NO_{23}. A 75% decreasing trend in NH_4 was detected at MLE2.3 using the

LIFEREG procedure. The strong effect of flow on nitrogen concentration in the mesohaline zone was apparent in that all of the flow adjustment models were significant for all four nitrogen species analyzed.

TSS increased by 60% and 80% at XDC1706 and MLE2.2, respectively, although TSS and flow were only significantly related at XDC1706. In view of the TSS trends, trends in SECCHI depth, which are significantly related to flow at XDC1706 and MLE2.2, are difficult to explain. There was no significant change in SECCHI depth at XDC1706, where TSS increased by 60%. SECCHI depth increased by 10% at MLE2.2, despite an 80% increase in TSS. SECCHI depth decreased by 20% at MLE2.3, although TSS was unchanged at that station. There were no significant changes in CHAA in the mesohaline zone, where CHAA was significantly related to flow at only MLE2.3.

Significant decreases in DO_FLD of 10% and 20% occurred at XDC1706 and MLE2.3, respectively. The segment-wide trend was significant and homogeneous despite a non-significant trend at MLE2.2. DO_FLD was significantly related to flow at all three mesohaline zone stations. A significant trend in PH_FLD was observed at MLE2.2; however, the slope was so small that the percent change was zero. Flow models were significant for PH_FLD at all three mesohaline zone stations. TALK increased by 20% at XDC1706, the only mesohaline zone station where it is measured. The flow model for TALK at XDC1706 was significant.

D. Flow Adjusted Bottom Trends for 1985-1996

The following sections describe bottom, flow adjusted trends for the 1985-1996 time period for the stations located in the mainstem. P-values, trend directions, and percent changes for annual trends are summarized for the mainstem stations in Figures III-4a (nutrients) and III-4b (non-nutrients). Detailed information on bottom, flow adjusted trends for 1985-1996 is presented in Appendix B.

i. Tidal Fresh Zone

Although flow models for TP were significant for all four tidal fresh stations none of the time trends achieved the $p \leq 0.05$ significance level. PO_4W decreased by 26% at XEA6596 for the 1985 - September 1990 time period. The segment-wide PO_4W trend was also significant and homogeneous, despite having a significant trend at only one station. PO_4W-flow models were significant at all four tidal fresh stations. PO_4F time trends were not significant at any tidal fresh station, although all four flow correction were significant at $p \leq 0.01$.

Reductions of 12% and 16% in TN were observed at XFB1433 and XEA1840, respectively. Despite having significant TN trends at only two of four tidal fresh stations, a significant, homogeneous segment-wide trend was observed. Flow models for TN were significant at XFB2470, XFB1433, and XEA1840. Flow models for DIN at all four stations and the segment-wide trend statistic were significant, although DIN trends at individual stations were not (p-values ranged from 0.05 to 0.07). For NO_{23}, flow models were significant at all tidal fresh stations; however, significant time trends were not observed.

All four tidal fresh stations had significant decreasing NH_4 trends, to of which were estimated using LIFEREG (LIFEREG was used at XFB1433 and XEA1840 where NH_4 decreased by 54% and 59% respectively). Decreasing NH_4 trends of 48% at XFB2470 and 71% at XEA6596 were detected using the Seasonal Kendall test. Flow correction models for NH_4 were significant only at XFB1433 and XEA6596. Because not all NH_4 trends were estimated using the Seasonal Kendall test, on which the homogeneity statistics are based, segment-wide trends could not be assessed.

TSS was significantly related to flow at all four tidal fresh stations; however, neither individual station trends nor a segment-wide trend were detected. Significant increases in CHAA were observed at all four tidal fresh stations and for the segment overall. The increases ranged from 40% at XEA1840 to 220% at XFB2470, the upper-most tidal fresh station. Significant CHAA-flow models were observed at XFB2470, XFB1433, and XEA6596.

A significant decrease of 10% was observed in bottom DO_FLD at both XEA6596 and XEA1840. In addition, the flow adjustment models at all four stations and the segment-wide trend were also significant. Significant trends in PH_FLD were detected at all four stations, but at two stations (XEA6596 and XEA1840) the slopes were so small that the percent changes were rounded to zero. Increases of 10% were observed at XFB2470 and XFB1433, the upper two stations. The flow models for PH_FLD at all four stations and the segment-wide trend were also significant. Significant flow models and increases of 20% were observed for TALK at XFB2470, XFB1433, and XEA6596. The segment-wide trend was also significant and homogeneous.

ii. Oligohaline Zone

There were no significant TP trends in the oligohaline zone. One significant TP flow adjustment model was detected at XDA1177. Significant decreases of 38% and 25% in PO_4W were observed at XDA4238 and XDA1177, respectively. Significant PO_4W flow adjustment models were observed at all three oligohaline zone stations. The segment-wide trend statistic for PO_4W was also significant. PO_4F trends were not significant at either oligohaline zone station for which data are available; however, flow adjustment models at both stations were significant.

TN decreased by 14% at XDA4238, the only station with a significant trend. Flow adjustment models were significant at all three stations. A pattern similar to that observed for TN was also found for DIN, which decreased by 32% at XDA4238. Flow adjustment models for DIN at all three stations were significant. Conflicting NO_{23} trends were observed in the oligohaline zone where a decrease of 30% was observed at XDA4238 and an increase of 68% was observed at XDB3321. NO_{23} flow adjustment models at all three stations were significant. The dichotomy in the trends for these stations probably resulted from the time periods analyzed. Data are available for both stations starting in January1985; however the ending dates differ. Data for XDA4238 ends in December 1996, whereas data for XDB3321 ends in September 1990. The differing trend directions for the two stations/time periods indicates that NO_{23} increased during the first period and then decreased from 1990 onward. Significant decreasing trends in NH_4 of 60% and 53% were observed at XDA4238 and XDA1177, respectively. Both trends were estimated using the LIFEREG procedure due to the high numbers of censored data. Flow adjustment models for NH_4 were significant at all three oligohaline zone stations.

A significant decreasing trend of 60% was detected for TSS at XDB3321. Significant flow adjustment models for TSS were observed at XDA4238 and XDA1177. CHAA increased by 60% at XDA4238. The only significant flow adjustment model for CHAA was observed at XDB3321.

DO_FLD decreased by 10% at XDA4238, the only oligohaline zone station with a significant trend; however, DO_FLD flow adjustment models were significant at all three stations. Significant, though negligible changes in PH_FLD were observed at all three oligohaline zone stations. Although the slopes for PH_FLD were positive, they were so slight that rounding for the percent change calculation yielded a result of zero. Significant flow adjustment models were observed for PH_FLD at XDA4238 and XDA1177. The segment-wide trend was significant and homogeneous. Increases of 30% and 20% were found at XDA4238 and XDA1177, respectively for TALK. The flow adjustment models were not significant.

iii. Mesohaline Zone

TP increased by 38% at MLE2.3, which is also the only station with a significant flow adjustment model. PO_4W trends were not significant at either XDC1706 or MLE2.2, the only stations in the mesohaline zone for which whole water samples were analyzed (whole water samples were not analyzed at MLE2.3 during the study period). Flow adjustment models for PO_4W at both XDC1706 and MLE2.2 were significant. PO_4F increased by 119% at MLE2.2, the only mesohaline zone station with a significant time trend. Due to the large number of censored data points, the PO_4F trend at MLE2.2 was estimated using the LIFEREG procedure. PO_4F flow adjustment models were significant at all three stations.

A significant increasing TN trend (23%) was detected at only one mesohaline zone station (MLE2.2); however, flow adjustment models at all three stations were significant. Conflicting trends were observed for DIN in the mesohaline zone. DIN decreased by 33% at MLE2.3 and increased by 22% at MLE2.2. Two additional points should be made regarding the DIN trends for these two stations. First, the null hypothesis of seasonal homogeneity was rejected for DIN at MLE2.3, which indicates that for most months DIN decreased, but for some DIN increased. Second, the trend at MLE2.2 was estimated using the LIFEREG procedure. Flow adjustment models for DIN were significant at all three stations. Although flow adjustment models for NO_{23} were significant in the mesohaline zone, the trends were not significant. Significant decreases in NH_4 were detected using the LIFEREG procedure at XDC1706 (49%) and MLE2.3 (45%). The only significant flow adjustment model for NH_4 was observed at MLE2.3.

TSS doubled (100% increasing trend) at MLE2.2, the only mesohaline zone station with a significant trend. Flow adjustment models for TSS were significant at all three stations. Flow adjustment models for CHAA were also significant at all three stations; however, no significant trends were observed. Bottom DO_FLD decreased by 20% at MLE2.2 and MLE2.3. The flow adjustment models were significant at all three stations. The segment-wide trend statistic was significant, but the null hypothesis of seasonal homogeneity was rejected, indicating that during most months DO_FLD is decreasing, but for some months, DO_FLD is increasing. As a result, any segment-wide trend analyses on DO_FLD should be conducted across stations on individual months. A significant PH_FLD trend was detected at XDC1706 although significant flow adjustment models were observed at the other two mesohaline zone stations. The change in PH_FLD at XDC1706 was positive, but so small that the trend was rounded to zero during the percent change calculation. TALK increased by 10% at XDC1706, the only mesohaline zone station where alkalinity is measured. The flow adjustment model was also significant.

E. Non-Flow Adjusted Surface Trends for mid-1970s-1996

The next four sections describe trends that incorporate the "historic" data that was collected from the mid-1970s through the inception the Chesapeake Bay Program in July 1984. Data from the mid-1970s through July 1984 were collected under the State's Basic Water Monitoring Program (BWMP) which was funded through the EPA Clean Water Act. Data were not collected on as regular a basis under the BWMP as they currently are under the Chesapeake Bay Program. As a result, the start dates are not as consistent for the BWMP data set as they are for the 1985-1996 data set. In addition, data for half of 1977 and all of 1978 are missing from the BWMP data set due to laboratory errors.

Long-term trends are not presented for MLE2.2 and MLE2.3, two of the mesohaline zone stations, because of large gaps in the data set (data were collected at those stations on an infrequent basis under the BWMP). In addition, only annual trends appear below since the living resources programs that would correlate with the seasonal trends in the water quality data were not in place until after the start of the Bay Program.

The following sections describe surface, non-flow adjusted trends for the mid-1970s-1996 time period for the stations located in the mainstem. Detailed information on surface, non-flow adjusted trends is presented in Appendix C.

i. Tidal Fresh Zone

TP trends at all four tidal fresh stations were significant. Decreases ranged from 27% at XFB2470 to 39% at XEA1840. The segment-wide trend was significant and homogeneous for season, station, and station-season interaction. Decreasing trends were also observed for PO_4W (data were not filtered under the BWMP so PO_4F trends are not available). Percent changes for

PO_4W ranged from -40% at XEA1840 to -61% at XFB1433. It should be noted that all stations had greater than 5% censored data so the magnitudes of changes may be biased. The PO_4W segment-wide trend was significant and homogeneous across all stations and seasons.

Unlike TN trends for the 1985-1996 period, which for the most part were decreasing, TN trends for the longer time period are significantly increasing at all four tidal fresh stations. TN increases for the long-term data set ranged from 15% at XFB1433 to 38% at XEA1840. Although the TN segment-wide trend was significant, the null hypothesis of homogeneity for season was rejected. Consequently, any segment-wide trend analyses should be performed on a monthly not an annual basis. Increasing trends were also observed for DIN, the species of nitrogen most readily used by living resources. Significant increases in DIN were observed at three of the tidal fresh stations; the trend at XFB2470 was not significant. DIN increases ranged from 20% at XFB1433 to 39% at XEA1840. The DIN segment-wide trend was significant and homogeneous across all stations and seasons.

Trends in NO_{23}, a component of both TN and DIN, increased significantly at all four tidal fresh stations. The increases ranged from 63% at XEA6596 to 75% at both XFB2470 and XEA1840. The segment-wide NO_{23} trend was significant; however, the null hypothesis of seasonal homogeneity was rejected. As a result, any segment-wide trends would have to be conducted on individual months (seasons). Unlike the other nitrogen species, significant decreases in NH_4 occurred at three stations in the tidal fresh zone. Decreases in NH_4 ranged from 32% at XEA6596 to 65% at XFB2470. The magnitudes of the percent changes may be biased due to the high numbers of censored data points. Although the segment-wide trend for NH_4 was significant, the null hypothesis of homogeneity was rejected for season and station. Heterogeneity in station and season would indicate the only meaningful trends are for individual site-seasons.

TSS increased by 50% and 60% at XFB2470 and XFB1433, respectively. Despite having significant trends at only two stations, the segment-wide trend was significant for the tidal fresh zone. The null hypothesis of homogeneity was rejected for season, indicating that for most seasons (months) TSS was increasing, but for some TSS was decreasing. Segment-wide TSS trends should therefore be conducted on individual months. Even though significant increases were observed for TSS at two tidal fresh stations, there were no detectable effects on SECCHI depth, which remained unchanged (SECCHI depth slopes for the four stations are zero). A significant decreasing trend of 30% was detected at XEA1840 for CHAA.

Conflicting tends were detected for DO_FLD in the tidal fresh zone. A 10% increase was observed at XFB2470, the upper-most station, while a 10% decrease was observed at XEA1840, the lower-most station. Significant increases in PH_FLD were observed at three stations in the tidal fresh zone. Two stations, XFB2470 and XFB1433 had increases of 10%, while the slope at the remaining station, XEA6596, was so small that the percent change was rounded to zero. Although the segment-wide trend for PH_FLD was significant, the null hypothesis of homogeneity was rejected for station and season; therefore, segment-wide trends are inappropriate and the only meaningful trend tests are for individual sites and seasons.

ii. Oligohaline Zone

Total phosphorus decreased by 22% at XDA4238, the only oligohaline zone station with a significant trend. Despite the lack of significant TP trends at the other two oligohaline zone stations, the segment-wide trend was significant and homogeneous for trend. There were no significant PO_4W trends at individual stations. Despite the absence of significant trends at individual stations, the segment-wide PO_4W trend was significant, but heterogeneous for season.

TN increased by 19% at XDA1177 and by 35% at XDB3321, where sampling stopped in September 1990. The increasing TN trend was significant and homogeneous for the segment. Significant increasing DIN trends were detected at all three oligohaline zone stations. DIN increases ranged from 24% at XDA4238 to 43% at XDB3321. The DIN trend was significant and homogeneous for the segment. NO_{23}, a major component of both TN and DIN, also had significant increasing trends at all three oligohaline zone stations. NO_{23} trends in the oligohaline zone, which ranged from 46% to 65% at XDA4238 and XDB3321, respectively were significant and homogeneous for the segment. NH_4, which is a minor component of DIN, decreased by 27% at XDA4238 and by 32% at XDA1177. The NH_4 trend magnitudes may be biased, since the numbers of censored data points greatly exceeded 5%. Despite the absence of a significant NH_4 trend at XDB3321, a significant and homogeneous segment trend was observed.

Increasing TSS trends were observed at XDA4238 (70%) and XDA1177 (30%). The segment-wide trend was significant, but heterogeneous for season. Therefore, segment-wide statements about annual trends would be misleading. Any trend analyses across stations should be conducted on a monthly basis. Although increasing TSS trends were observed at the upper two stations, significant changes in SECCHI depth were not detected. CHAA decreased by 20% and 30% at XDA4238 and XDB3321, respectively. The pattern in heterogeneity described for TSS also occurred for CHAA, in that the trend was significant for the segment, but the null hypothesis for homogeneity across season was rejected. It appears that the changes observed in TSS are probably a result of increases in the inorganic fraction that are capable of off-setting the decreasing trends in CHAA. Significant trends for DO_FLD and PH_FLD were not detected in the oligohaline zone.

iii. Mesohaline Zone

Trend analyses were only conducted for one station (XDC1706) in the mesohaline zone because data collection at MLE2.2 and MLE2.3 from the mid-1970s until the start of sampling under the Chesapeake Bay Program was sporadic at best. Only three significant trends were detected at XDC1706; NO_{23} increased by 35%, TSS increased by 30% and DO_FLD decreased by 10%.

F. Non-Flow Adjusted Bottom Trends for mid-1970s-1996

The following sections describe bottom, non-flow adjusted trends for the mid-1970s-1996 time period for the stations located in the mainstem. Detailed information on surface, non-flow adjusted trends is presented in Appendix C.

i. Tidal Fresh Zone

There were no significant TP trends at individual stations; however, the overall segment-wide trend was significant and homogeneous for season, station, and station-season interaction. Long-term trends in PO_4W were significant at all four tidal fresh stations. Changes ranged from -37% at XEA1840 to -57% at XFB2470. The number of censored data points exceeded 5% at XFB1433, XEA6596, and XEA1840, thus trend magnitudes at those stations may be biased. The segment-wide PO_4W trend was significant and homogeneous for station, season, and station-season interaction.

Non-flow adjusted, bottom trends for TN were not significant at individual stations or segment-wide. Similarly, trends for DIN were not significant at individual stations or segment-wide. Despite the absence of trends in either TN or DIN, significant trends were detected in NO_{23}, a major component of both species. NO_{23} trends ranged from 41% at XEA1840 to 54% at XEA6596. The overall, segment-wide NO_{23} trend was significant and homogeneous for station, season, and station-season interaction. In contrast to the increasing NO_{23} trends observed in the tidal fresh zone, trends in NH_4 were decreasing (probably as a result of nitrification at sewage treatment plants). Decreasing NH_4 trends ranged from 30% at both XEA6596 and XEA1840 to 58% at XFB2470. Although the segment-wide NH_4 trend was significant, the null hypothesis of homogeneity for season was rejected, indicating that during most seasons NH_4 decreased, but during some seasons it increased. As a result, trend analyses for NH_4 across stations should be conducted on a monthly basis.

XEA6596 is the only tidal fresh station that had a significant bottom trend for TSS. Although only one significant increase (90%) was observed on an individual station basis, the segment-wide trend was significant; however, the null hypothesis of homogeneity for season was rejected. CHAA increased by 20% at XFB2470, the only tidal fresh station with a significant CHAA trend.

Conflicting trends in bottom DO_FLD were detected in the tidal fresh zone. DO_FLD increased by 10% at XFB2470 and decreased by 10% at XDA1840. The null hypothesis of seasonal homogeneity at XFB2470 was rejected, which indicates that although DO_FLD increased during most months it decreased during others. Significant trends in PH_FLD were detected at three tidal fresh stations. PH_FLD increased by 10% at XFB2470 and XFB1433 and by 0% at XEA6596 (although the slope was positive at XEA6596, it was so slight that percent change was rounded to zero). The trend in PH_FLD was significant and homogeneous across all stations in the tidal fresh zone.

ii. Oligohaline Zone

TP trends in the oligohaline zone were not significant. Decreasing PO_4W trends of 34% and

36% were detected at XDA4238 and XDA1177, respectively. The magnitude of trend at XDA4238 may be biased due to the high number of censored data points. Although the PO_4 trend was significant for the segment, the null hypothesis of homogeneity for season was rejected.

Trends for both TN and DIN were not significant. Despite the absence of individual station trends for NO_{23} (p-values were in the 0.02-0.03 range), a significant, homogeneous segment-wide trend was detected. Significant decreasing trends in NH_4 were detected at XDA4238 (-88%) and XDA1177 (-46%); however, the magnitude of trend at both stations may biased due to the high number of censored data points. The NH_4 trend was significant for the segment, but the null hypothesis of homogeneity for station was rejected, probably as a result of the absence of significant decreasing trend at XDB3321. Consequently, analyzing NH_4 trends across stations would be misleading. Trends in TSS and CHAA were not significant in the oligohaline zone.

A 10% decreasing trend in DO_FLD was detected at XDB3321, where sampling stopped in September 1990. Although trends at the remaining two oligohaline zone stations were not significant, the segment-wide trend in DO_FLD was significant. Segment-wide DO_FLD trends should be analyzed on a monthly basis since the null hypothesis of homogeneity for season was rejected. Significant PH_FLD trends were observed at all three stations; however, the slopes were so slight that the percent changes were rounded to zero.

iii. Mesohaline Zone

Data for one station (XDC1706) was analyzed in the mesohaline zone due the lack of historical data at the remaining two stations. The only variable with a significant trend at XDC1706 is PH_FLD, where the slope was so small that the percent change was rounded to zero.

G. Flow Adjusted Surface Trends for mid-1970s-1996

The following sections describe surface, flow adjusted trends for the mid-1970s-1996 time period for the stations located in the mainstem. Detailed information on surface, flow adjusted trends is presented in Appendix D.

i. Tidal Fresh Zone

TP trends and flow adjustment models were significant at all four tidal fresh zone stations. The decreasing TP trends ranged from 31% at XFB2470 to 39% at XEA1840. The trend in TP was significant and homogeneous for the segment. Trends in PO_4W were substantial and negative at all four tidal fresh zone stations. Due to the large numbers of censored data points, PO_4W trends at all four stations were estimated using the LIFEREG procedure. The trends, all of which were decreasing, ranged from 49% at XEA1840 to 66% at XFB2470. Flow-adjustment models were significant at the upper three tidal fresh stations (XFB2470, XFB1433, and XEA6596).

TN trends and flow adjustment models were significant at all four tidal fresh zone stations. The trends, which were all increasing, ranged from 17% at XFB1433 to 38% at XEA1840. The TN trend was significant for the segment, although not homogeneous for season. As a result, segment-wide trends for TN should be conducted on a monthly basis. Trends and flow adjustment models were also significant at all four stations for DIN, where increases ranged from 20% at XFB2470 to 40% at XEA1840. The DIN trend was significant and homogeneous for the segment.

The increases in TN and DIN are in part attributable to increases in NO_{23}, a major component of both species. NO_{23} increases ranged from 62% at XEA6596 to 74% at both XFB1433 and XEA1840. The NO_{23} trend was significant and homogeneous for the segment. Flow adjustment models for NO_{23} were significant at three of the four stations. In contrast to the increasing NO_{23} trends, significant decreasing trends in NH_4 were detected in the tidal fresh zone. Decreasing trends of 79% were detected at both XFB2470 and XFB1433, while a decreasing trend of 63% was detected at XEA6596. Decreasing trends in NO_{23} and increases in NH_4 in the tidal fresh zone may have resulted from nitrification at the sewage treatment plants. The NH_4 trends were estimated using the LIFEREG procedure due to the high number of censored data points. NH_4 flow adjustment models were significant only at XEA6596 and XEA1840.

Increasing TSS trends were detected at XFB2470 (60%), XFB1433 (40%) and XEA6596 (50%). Flow adjustment models were significant at all four tidal fresh. The TSS segment-wide trend was significant; however, the null hypothesis of homogeneity for season was rejected. As a result, segment-wide trends should be conducted on a monthly basis. Despite significant increases in TSS, and significant flow-adjustment models for SECCHI depth, no significant SECCHI depth trends were observed. Significant flow adjustment models for CHAA were detected at all four tidal fresh stations; however, only one significant trend (-30%) was observed at XEA1840.

Conflicting DO_FLD trends were observed in the tidal fresh zone. DO_FLD increased by 10% at XFB2470 and decreased by 10% and 20% at XEA6596 and XEA1840, respectively. Flow adjustment models for DO_FLD were significant at all four stations. Significant trends in PH_FLD were detected at three tidal fresh zone stations. The upper two stations, XFB2470 and XFB1433 had increasing trends of 10%. The trend at XEA6596 was also significant; however, the slope was so negligible that the percent change was rounded to zero. The segment-wide trend for PH_FLD was significant; however, the null hypotheses for station and seasonal homogeneity were rejected. As a result, meaningful trend tests would be for individual stations-seasons.

ii. Oligohaline Zone

A significant TP decrease of 25% was detected at both XDA4238 and XDA1177. A significant TP flow correction model was observed at XDA1177. Although the TP trend at XDB3321 was not significant, a significant, homogeneous segment-wide TP trend was observed. PO_4W decreased at all three stations; however, there were no significant PO_4W trends or flow correction models in the oligohaline zone. Despite the absence of significant PO_4W trends at individual stations, the segment-wide trend was significant and homogeneous.

Significant trends and flow adjustment models were detected for TN at all three stations, where increases ranged from 22% at XDA4238 to 46% at XDB3321. The segment-wide TN trend was also significant and homogeneous for season, station and station-season interaction. Significant trends and flow adjustment models were also observed for DIN at all three stations. DIN increases ranged from 27% at XDA4238 to 57% at XDB3321. The segment-wide trend statistic was significant, but the null hypothesis of homogeneity for season was rejected. Consequently, segment-wide statements about trends in DIN would be misleading.

Trends in NO_{23}, a major component of both TN and DIN increased significantly at all three stations. The increases ranged from 48% at XDA4238 to 65% at XDB3321. NO_{23} flow adjustment models were also significant at all three stations. Although a significant trend statistic was observed for the segment, the null hypothesis of homogeneity for season was rejected. NH_4 trends at all three stations were analyzed using the LIFEREG procedure due to the large number of censored data points. Significant trends in NH_4 were observed at XDA4238 and XDA177, where NH_4 decreased by 61% and 48%, respectively. Flow adjustment models for NH_4 were significant at all three stations. Homogeneity statistics are not available for NH_4 since the trends were not estimated using the Seasonal Kendall test.

Significant trends and flow adjustment models were observed in the oligohaline zone for TSS, where increases ranged from 30% at XDA1177 to 80% at XDA4238. The TSS segment-wide trend was significant and homogeneous for station, season, and station-season interaction. Despite sharp increases in TSS at all three stations in the oligohaline zone, and significant flow adjustment models for SECCHI depth, no significant SECCHI depth trends were observed. A significant 50% decrease in CHAA was observed at XDB3321, the only oligohaline zone station with a significant trend. Despite the absence of significant trends at the remaining two stations, the segment-wide trend was significant, but not homogeneous for season.

Although significant flow adjustment models were observed at all three oligohaline zone stations for DO_FLD, a significant trend was observed only at XDA1177, where DO_FLD decreased by 10%. Although only one station had a significant trend in DO_FLD, the segment-wide trend was significant and homogeneous for station, season, and station-season interaction. Flow adjustment models were significant at two stations for PH_FLD; however, no significant trends were observed.

iii. Mesohaline Zone

A trend of -23% was observed for TP at XDC1706. The flow adjustment model was not significant. Due to the high number of censored data points for PO_4W at XDC1706, the trend, which was significant and decreasing (29%), was estimated using the LIFEREG procedure. The PO_4W flow adjustment model was also significant. A significant increasing trend of 23% and a significant flow adjustment model were observed for DIN at XDC1706. The TN trend was not significant. Part of the increase in DIN can be attributed to a significant 38% increase in NO_{23}, which also had a significant flow adjustment model. NH_4, a minor component of DIN, had a significant decrease of 38% at XDC1706. The NH_4 flow adjustment model was also significant. Significant flow adjustment models were observed for TSS and for SECCHI depth; however, the trends were not significant for either variable. Neither a significant flow adjustment model nor a significant trend were detected at XDC1706 for CHAA. Significant flow adjustment models were observed for DO_FLD and for PH_FLD; however, a significant trend (-20%) was detected only for DO_FLD.

H. Flow Adjusted Bottom Trends for mid-1970s-1996

The following sections describe bottom, flow adjusted trends for the mid-1970s-1996 time period for the stations located in the mainstem. Detailed information on the bottom, flow adjusted long-term trends is presented in Appendix D.

i. Tidal Fresh Zone

Significant flow adjustment models were observed at XFB2470 and XFB1433 for TP. A significant decreasing trend of 38% was detected in TP at XFB1433. Although only one significant TP trend was observed in the segment, the segment-wide trend was significant and homogeneous for station and for season. Flow adjustment models for PO_4W were significant at the upper three stations in the tidal fresh zone. Trends in PO_4W were significant at all four stations, three of which were estimated using the LIFEREG procedure due to high numbers of censored data points. Decreasing trends of 66% were detected using LIFEREG at XFB1433 and XEA6596. A decreasing trend of 49% was detected at XEA1840, which was also evaluated with LIFEREG. The PO_4W trend at XFB2470, which was estimated using the Seasonal Kendall test and Sen's slope estimator, was -48%.

Flow adjustment models for TN were significant at three tidal fresh stations. A significant increasing TN trend of 24% was observed at XEA1840. Although a significant trend was observed at only one station, the segment-wide trend was significant and the null hypotheses of homogeneity were not rejected. Significant flow adjustment models were observed at all four tidal fresh stations for DIN; however, only XFB2470 had a significant trend, where an increase of 18% was observed. Despite only one tidal fresh station having a significant trend in DIN, the segment-wide trend was significant and homogeneous.

Significant flow adjustment models and trends in NO_{23} were detected at all four tidal fresh stations. Changes in NO_{23} ranged from 32% at XFB1433 to 56% at XFB2470. In addition to significant trends at individual stations, a significant, homogeneous segment-wide trend in NO_{23} was also observed. Significant flow adjustment models for NH_4 were detected at the two lower tidal fresh stations; however, significant trends were detected at all four stations. A significant decrease of 53% was detected in NH_4 at XFB2470 using the Seasonal Kendall test. NH_4 trends using the LIFEREG procedure ranged from -47% at XEA1840 to -76% at XFB1433.

Although significant flow adjustment models were observed for TSS at all four tidal fresh stations, the trends were not significant. Flow correction models for CHAA were significant for the upper three tidal fresh stations. Significant trends in CHAA of 30% and 20% were detected at XFB2470 and XFB1433, respectively. In addition to the significant trends at two stations, a significant, homogeneous segment-wide trend in CHAA was also observed.

Flow adjustment models for DO_FLD were significant at all four tidal fresh stations. The direction of DO_FLD trends in the tidal fresh zone differed between the upper station (XFB2470), which had an increasing trend of 10%, and the lower station (XEA1840), which had a decreasing trend of 20%. Significant flow adjustment models and trends for PH_FLD were observed at three tidal fresh stations. Significant trends were detected at the upper three stations, whereas significant flow adjustment models occurred at the upper two and the lower-most stations. The PH_FLD trends at XFB2470 and XFB1433 were both 10%. The trend at XEA6596 was so slight that the percent change was rounded to zero. In addition to the significant PH_FLD trends at individual stations, the segment-wide trend was significant, though the null hypothesis of homogeneity for season was rejected. Therefore, trends across stations should be analyzed on a monthly basis.

ii. Oligohaline Zone

A significant TP flow correction model was observed at one station in the oligohaline zone. No significant TP trends were detected. PO_4W trends were significant at all three stations. A decrease of 25% was detected at XDA1177 using the Seasonal Kendall test. Decreases of 34% and 30% were detected at XDA4238 and XDB3321, respectively using the LIFEREG procedure. PO_4W trends at two of the stations were estimated using LIFEREG due to the high number of censored data points. A significant flow adjustment model for PO_4W was observed only at XDA1177.

Significant flow adjustment models for TN were detected at all three stations; however, a significant trend was only observed at XDA1177, where TN increased by 15%. Significant flow adjustment models for DIN were also observed at all three stations. The only station with a significant DIN trend was XDB3321, where DIN increased by 28%. NO_{23} also had significant flow adjustment models at all three stations and a significant trend of 49% at XDB3321. Despite only having one significant station trend, the overall segment-wide NO_{23} trend was significant and homogeneous. Trends and flow adjustment models for NH_4 were significant at all three oligohaline zone stations. Due to the large number of censored data points the NH_4 trends, which ranged from -39% at XDB3321 to -70% at XDA4238, were estimated using LIFEREG.

A significant 50% increase in TSS was detected at XDA1177. Significant flow adjustment models were detected at XDA4238 and XDA1177. A significant trend (-70%) and flow adjustment model were observed for CHAA at XDB3321.

Significant flow adjustment models were observed at all three oligohaline zone stations for DO_FLD. Significant decreasing trends of 10% were detected at XDA1177 and XDB3321. The segment-wide DO_FLD trend was also significant; however, the null hypothesis of homogeneity for season was rejected. Trends in PH_FLD were significant at all three stations. At the upper two stations, the slopes were small and positive while at the lower station (XDB3321) the slope was small and negative. The slopes at all three stations were so small that the trends were rounded to zero during the calculation of percent change. Significant flow adjustment models were observed at only the upper two stations.

iii. Mesohaline Zone

Neither the trend nor the flow adjustment model for TP were significant at XDC1706. A significant flow adjustment model was observed for PO_4W; however, the trend, estimated using LIFEREG, was not significant. Similarly, for TN and DIN flow adjustment models were significant, but the trends were not. Trends and flow adjustment models for NO_{23} and NH_4 were significant at XDC1706. The trend directions for NO_{23} and NH_4 differed; NO_{23} increased by 32%, while NH_4, estimated using LIFEREG, decreased by 45%.

Both the flow adjustment model and the trend (50%) in TSS were significant. Despite having a significant flow adjustment model, the trend in CHAA was not significant. A significant decreasing trend of 20% and a significant flow adjustment model were observed for DO_FLD. The change in PH_FLD was significant, but so slight that during the calculation of percent change the trend was rounded to zero. The flow adjustment model was not significant.

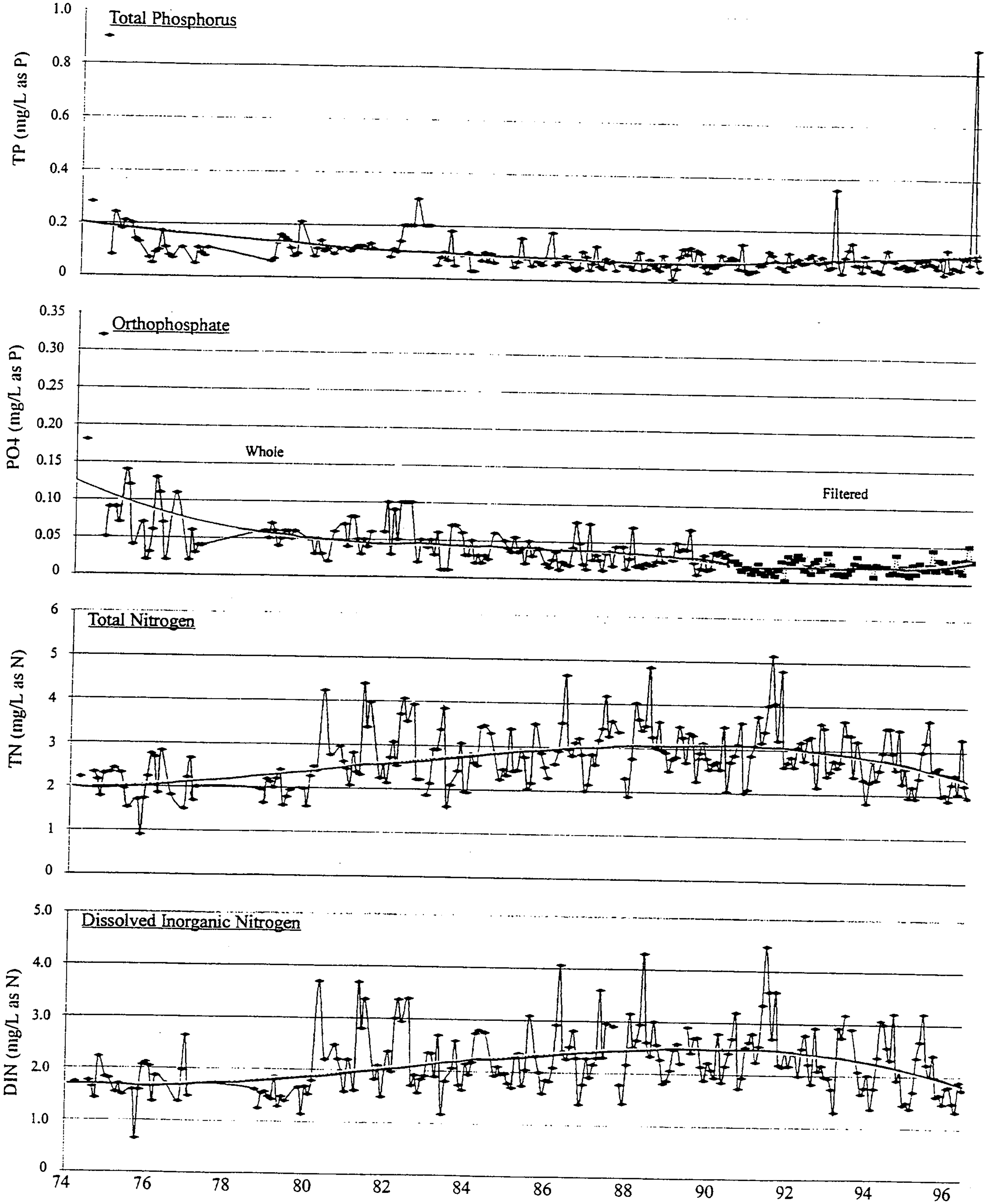


Figure III-1a. Monthly medians of surface nutrient water quality variables for XFB2470. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XFB2470.CDR

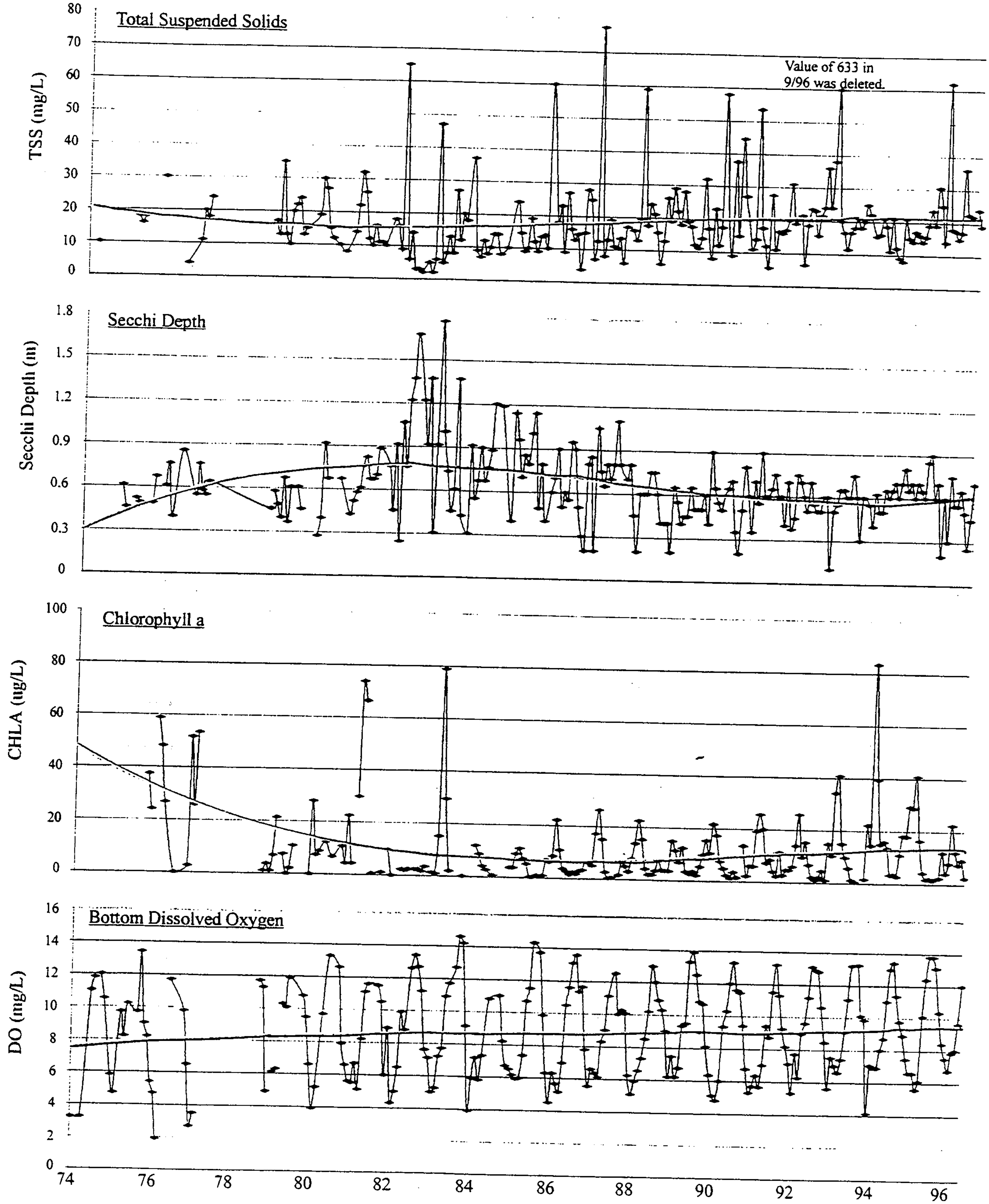


Figure III-1b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XFB2470. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XFB2470.CDR

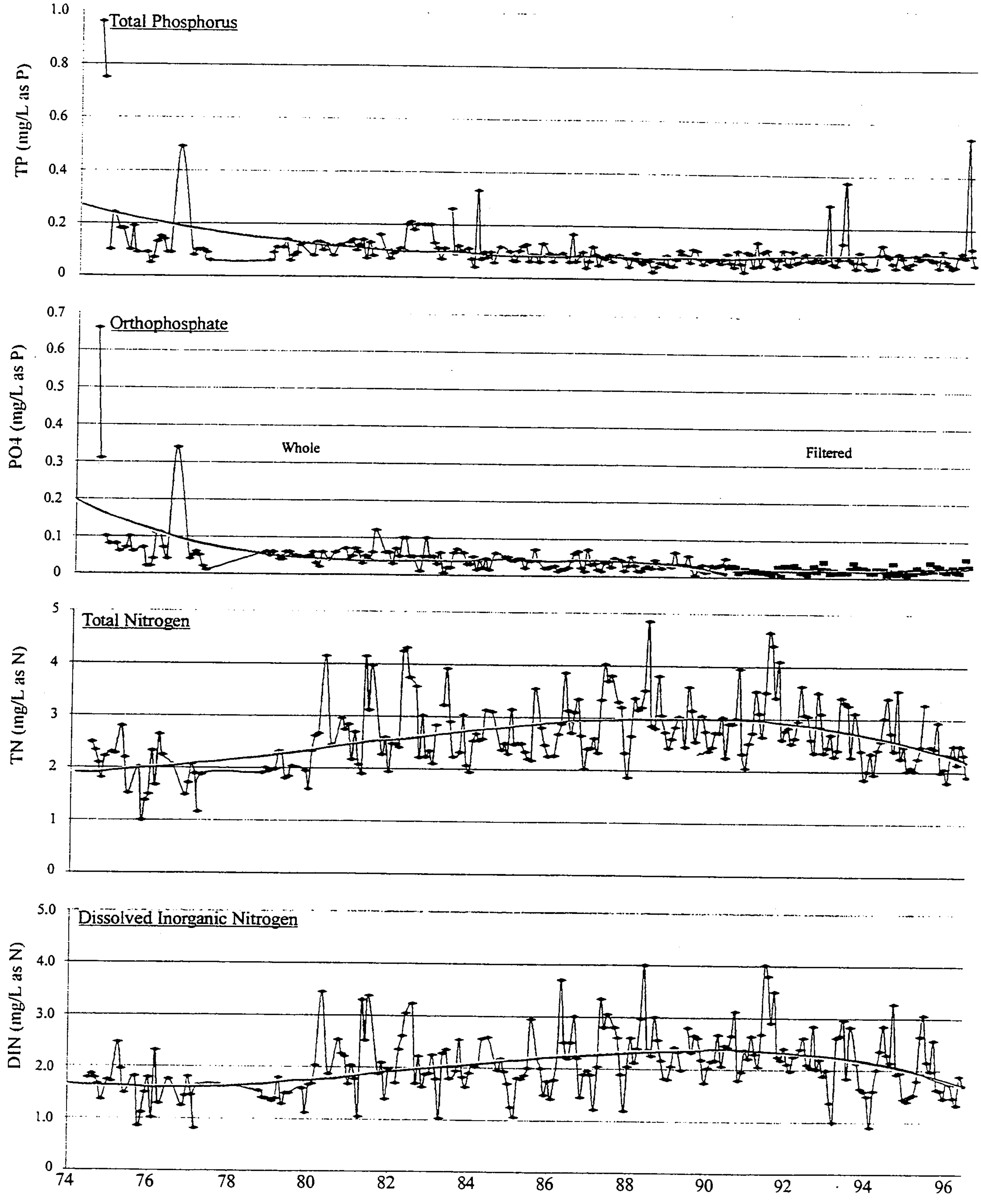


Figure III-2a. Monthly medians of surface nutrient water quality variables for XFB1433. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XFB1433.CDR

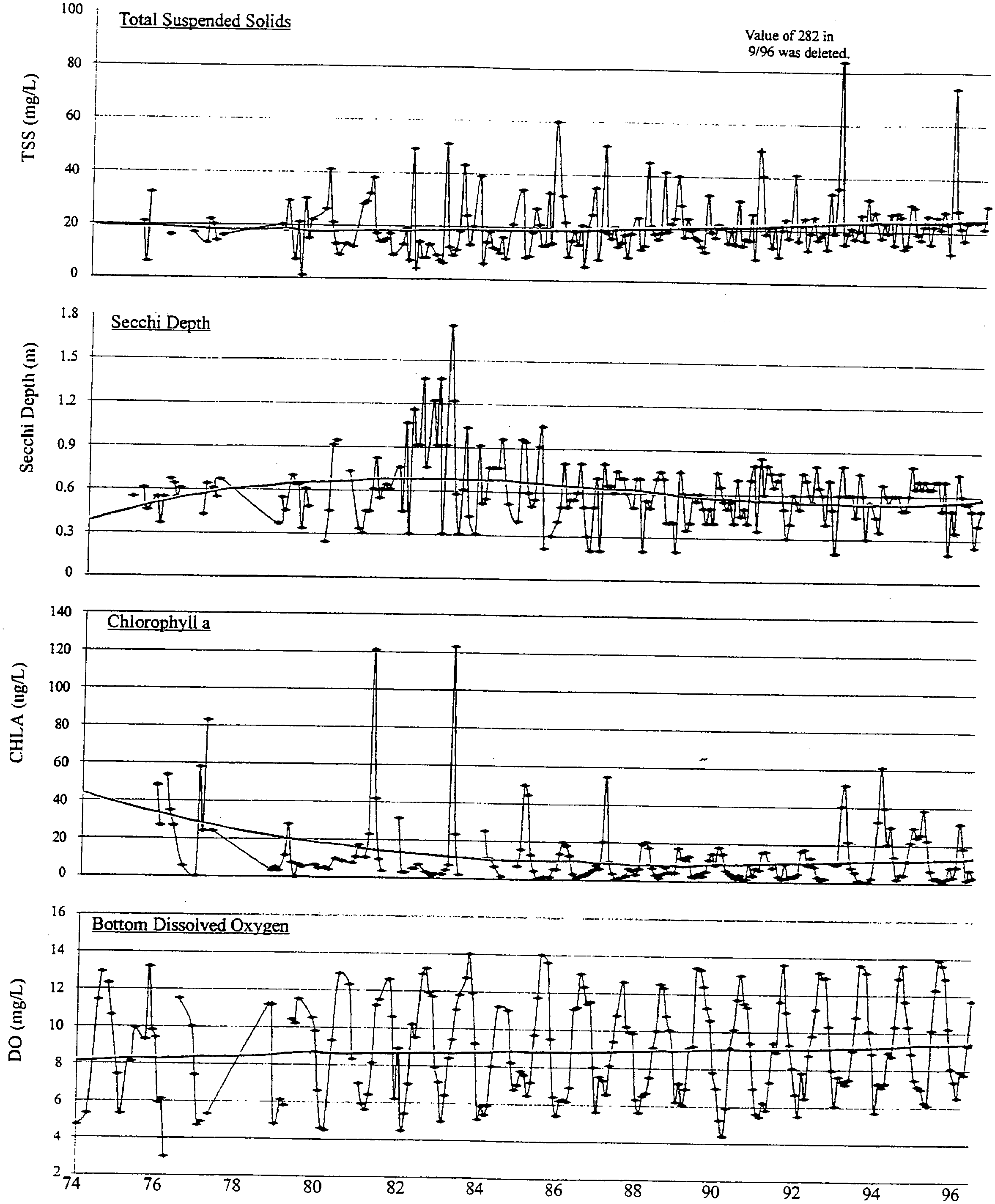


Figure III-2b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XFB1433. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XFB1433.CDR

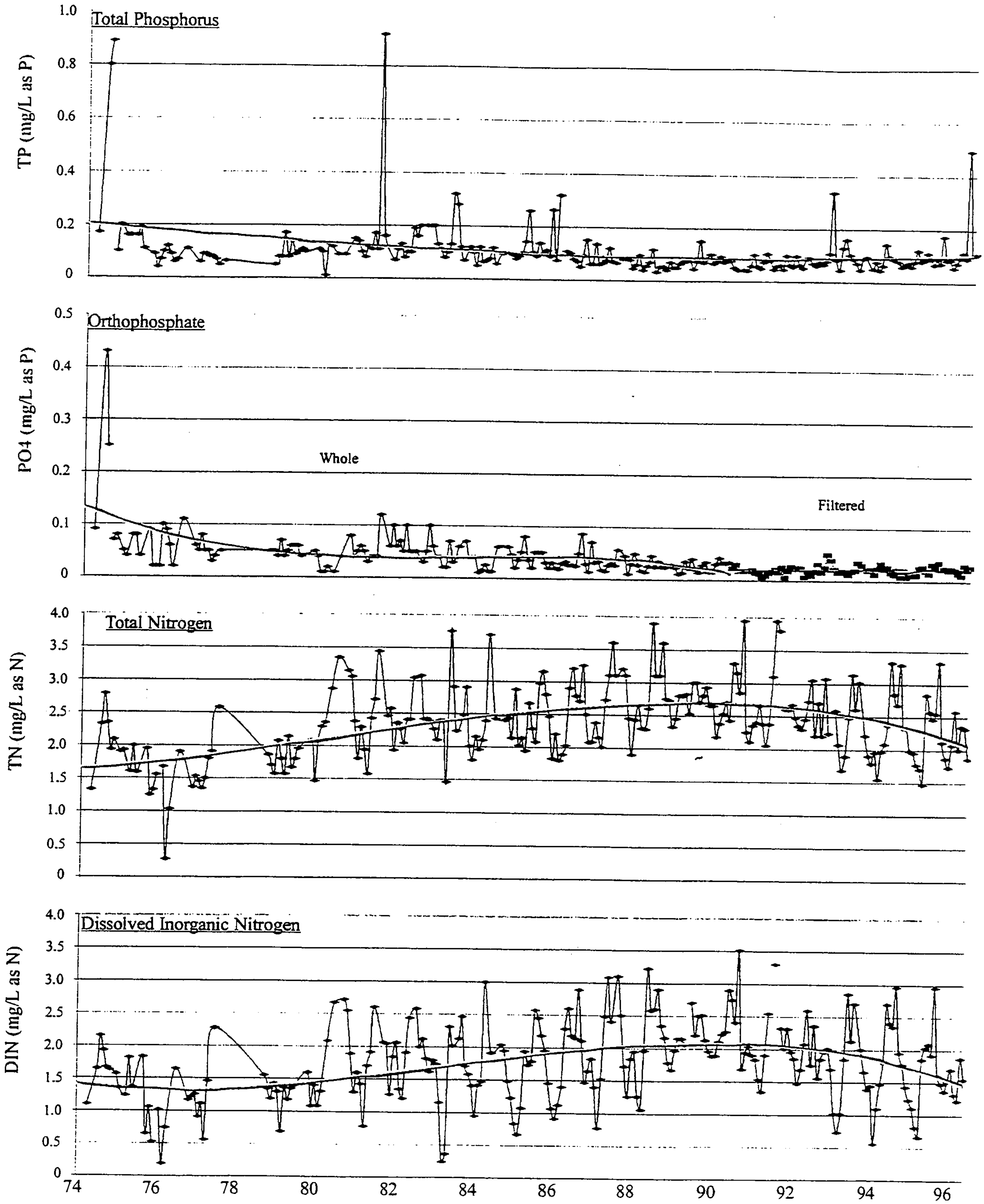


Figure III-3a. Monthly medians of surface nutrient water quality variables for XEA6596. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XEA6596.CDR

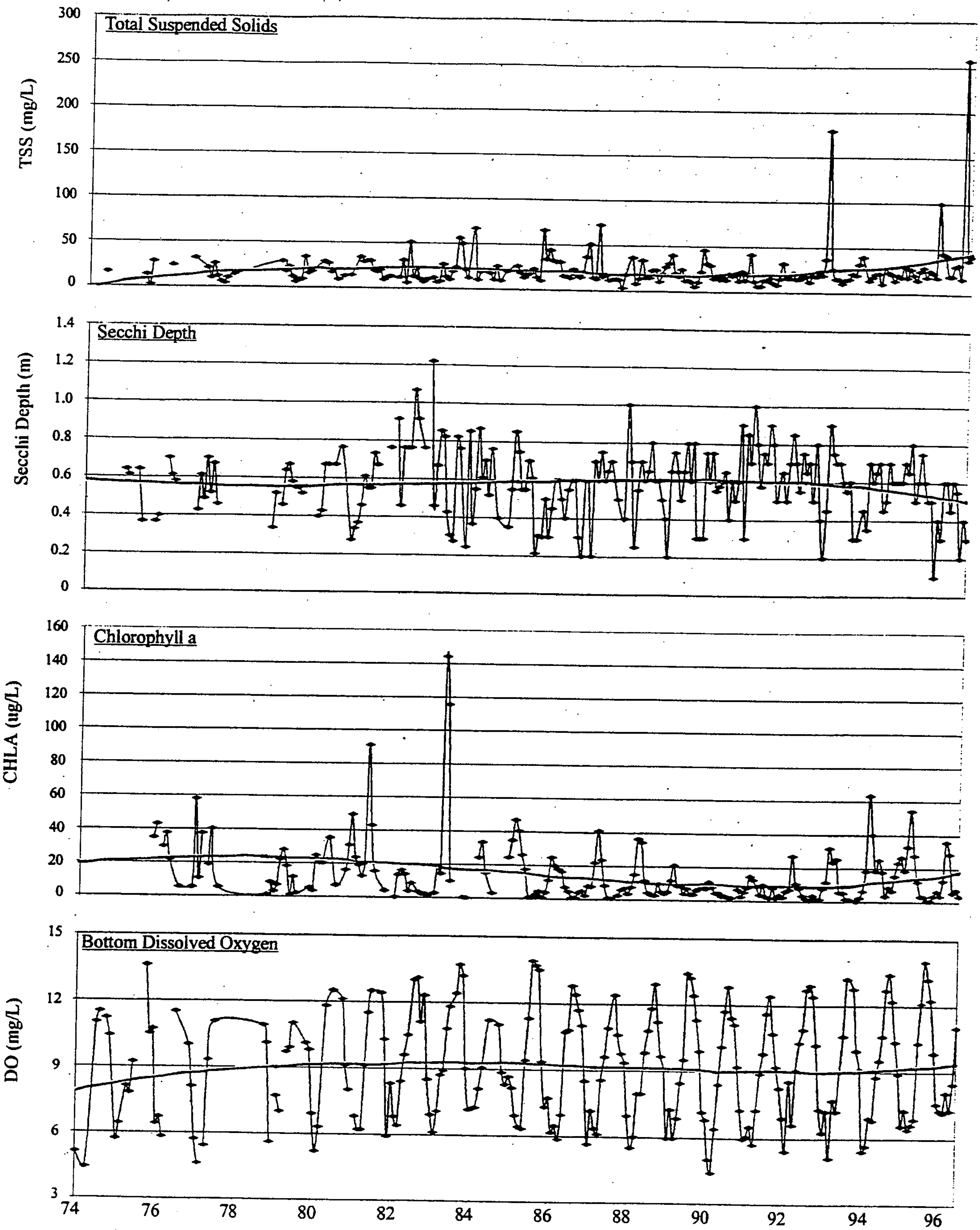


Figure III-3b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XEA6596. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XEA6596.CDR

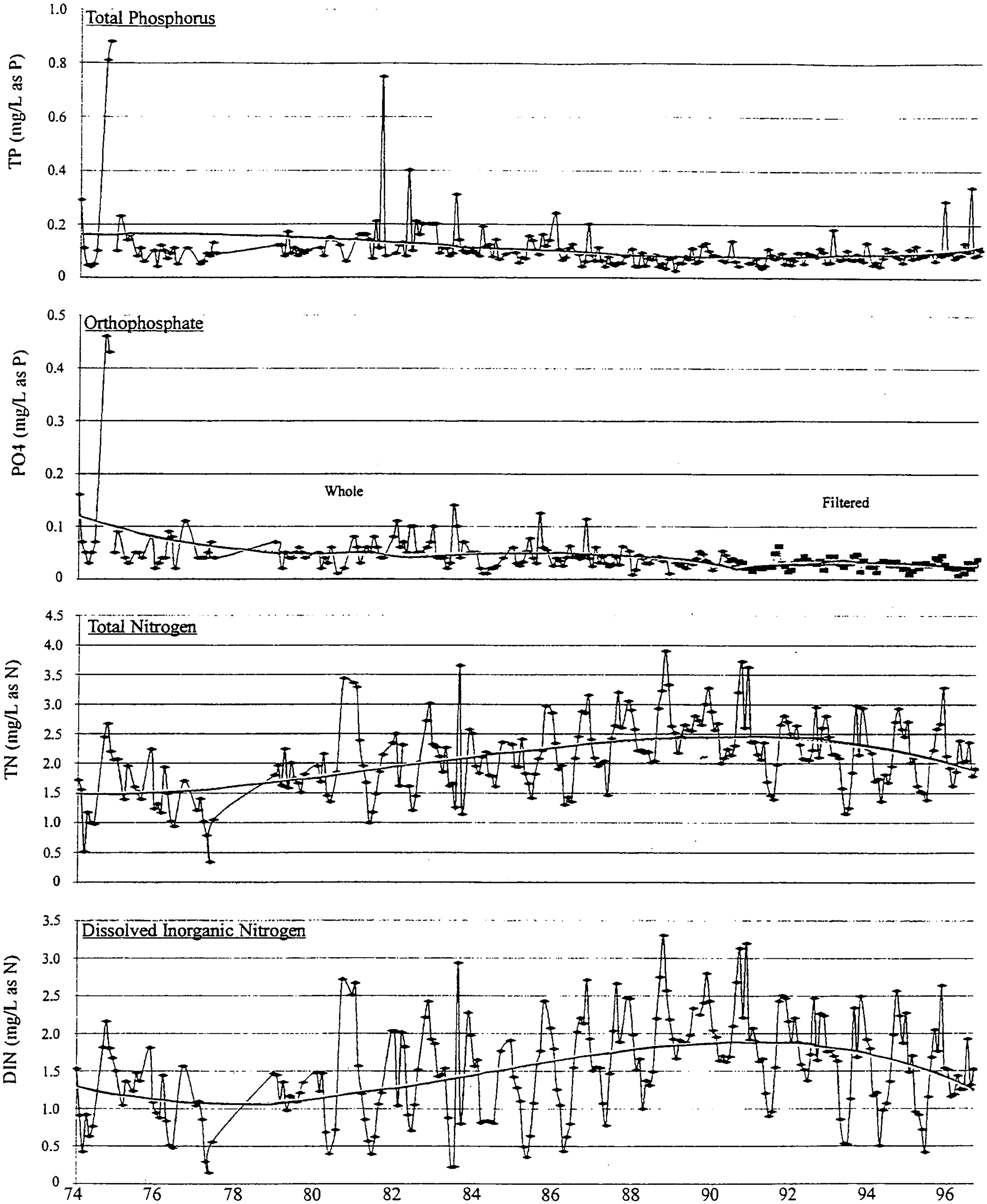


Figure III-4a. Monthly medians of surface nutrient water quality variables for XEA1840. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XEA1840.CDR

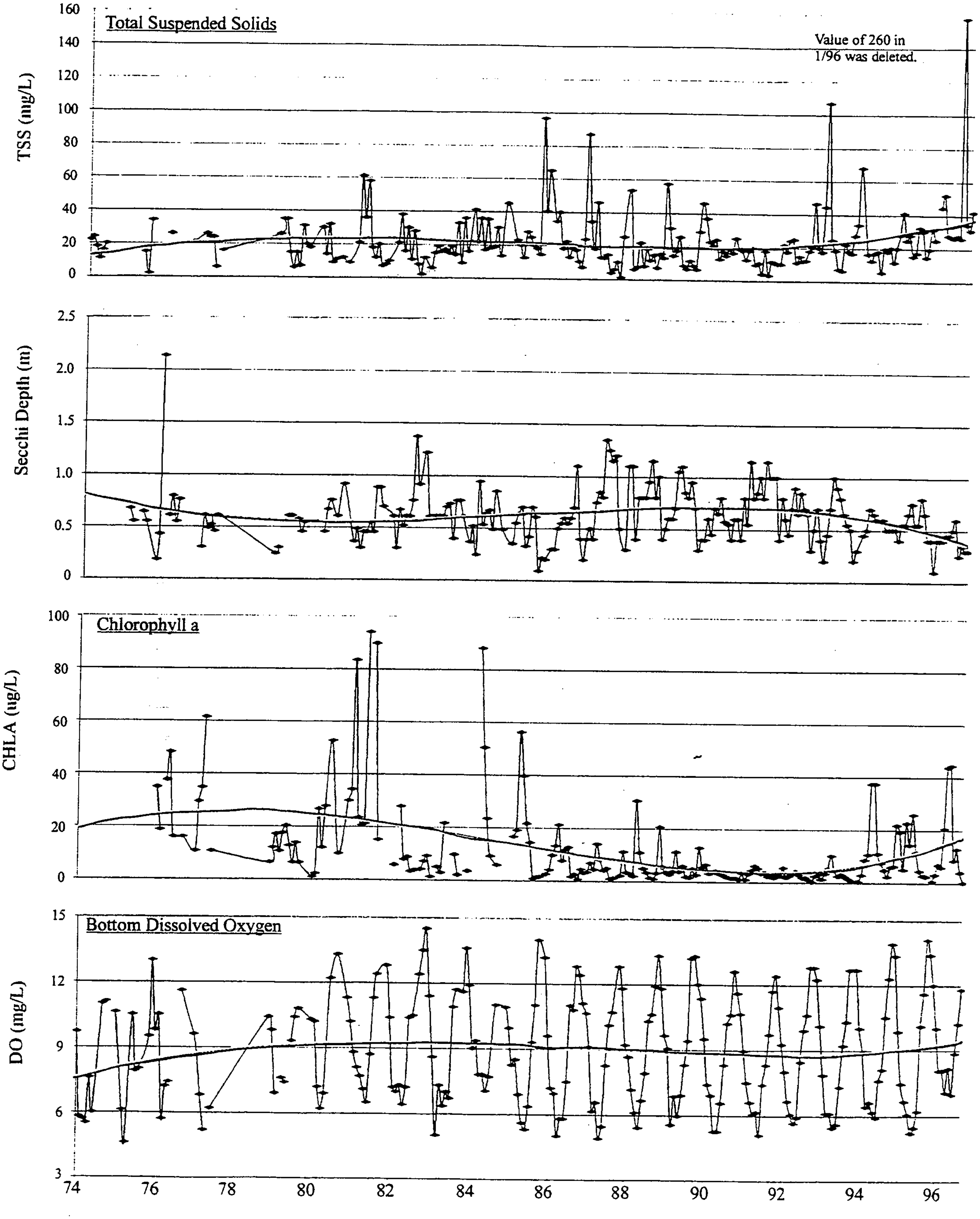


Figure III-4b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XEA1840. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XEA1840.CDR

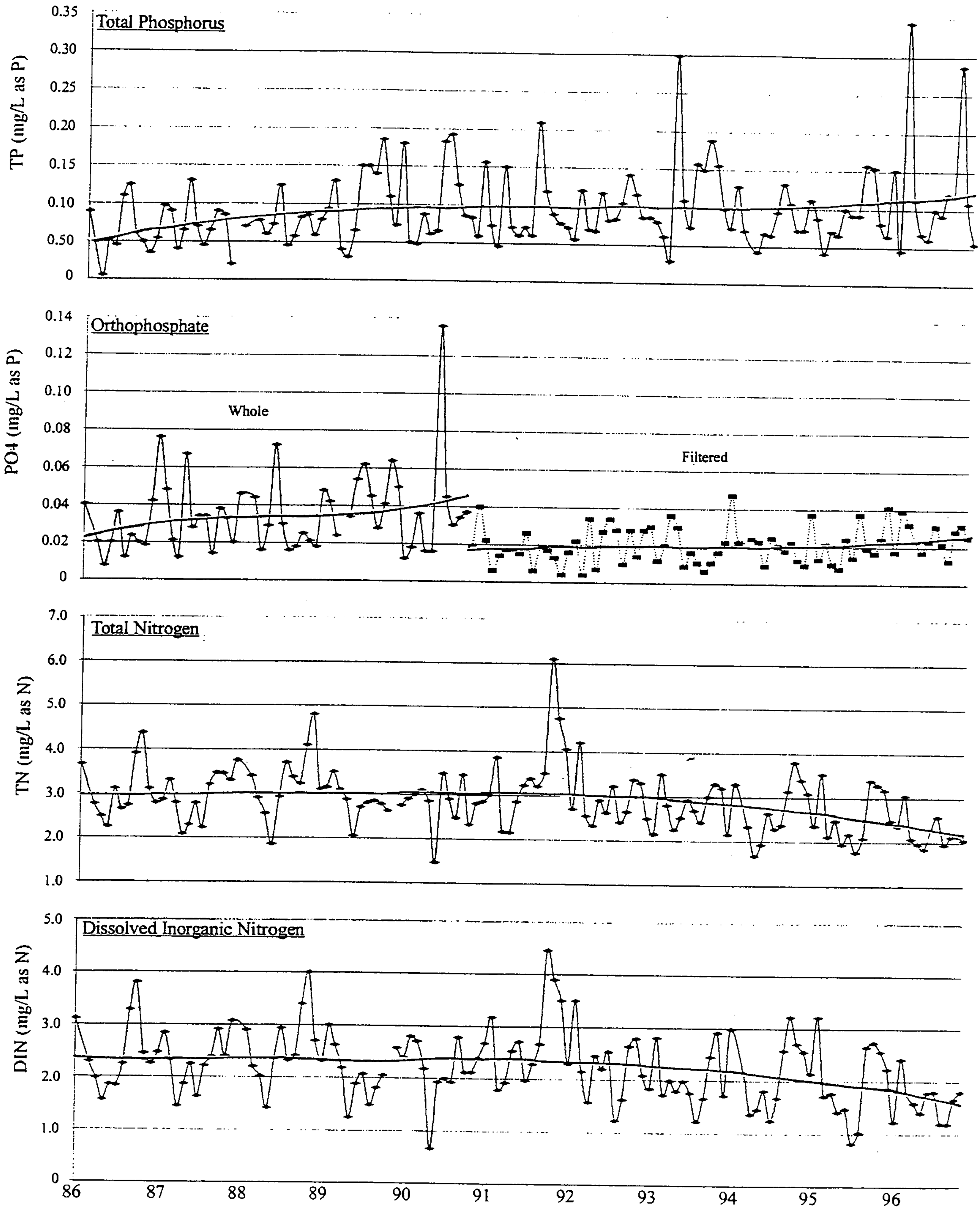


Figure III-5a. Monthly medians of surface nutrient water quality variables for XFB1986. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XFB1986.CDR

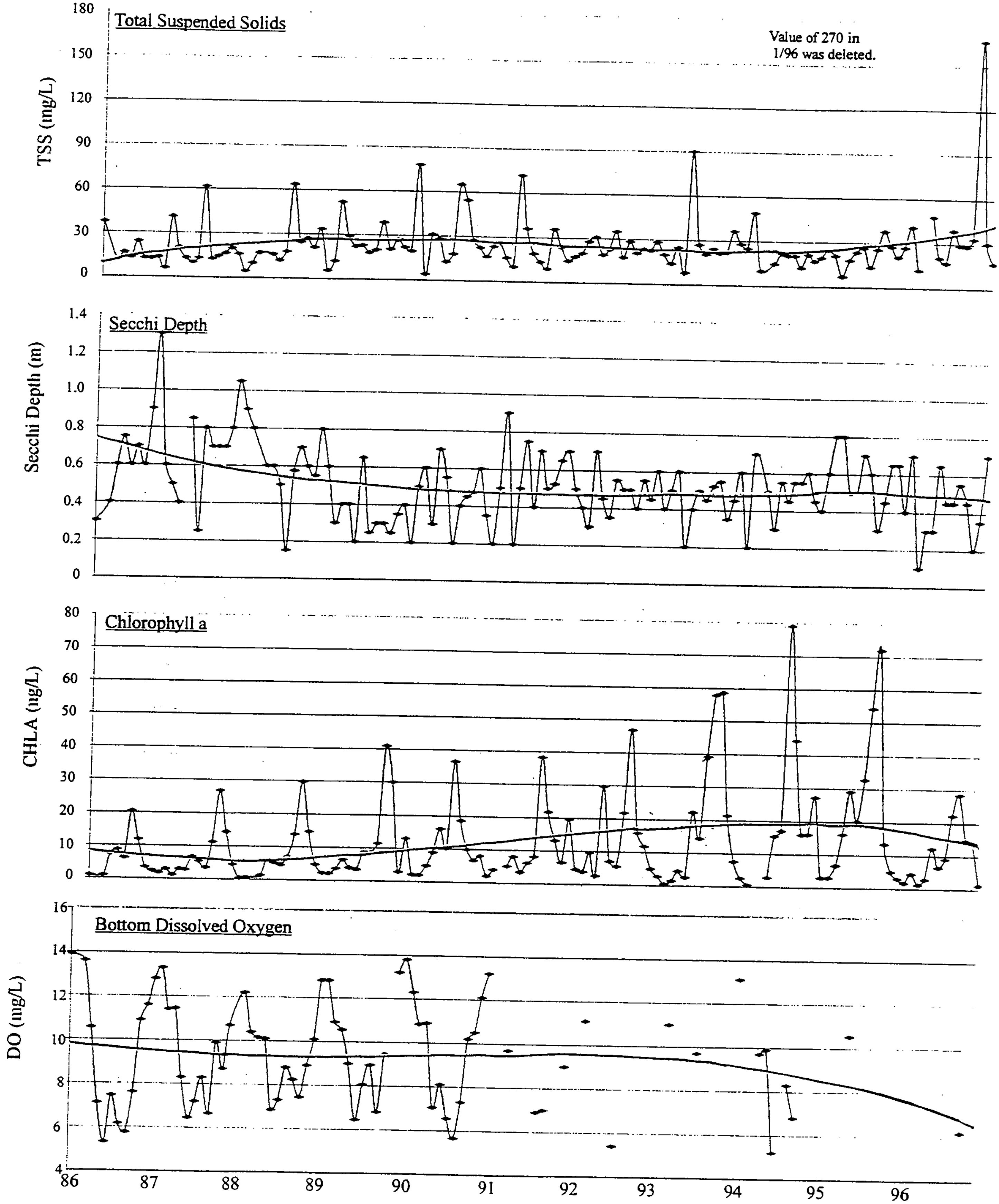


Figure III-5b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XFB1986. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XFB1986.CDR

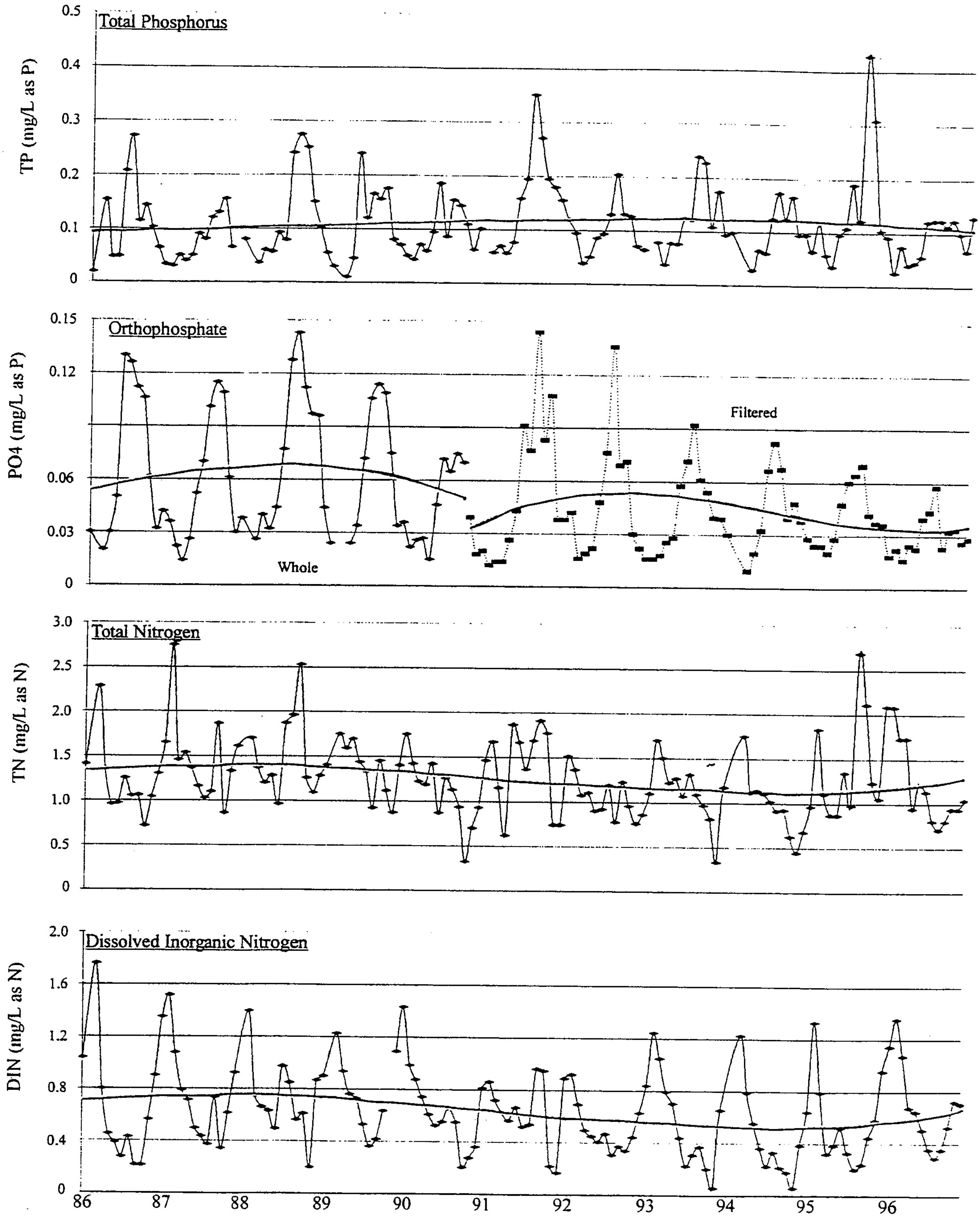
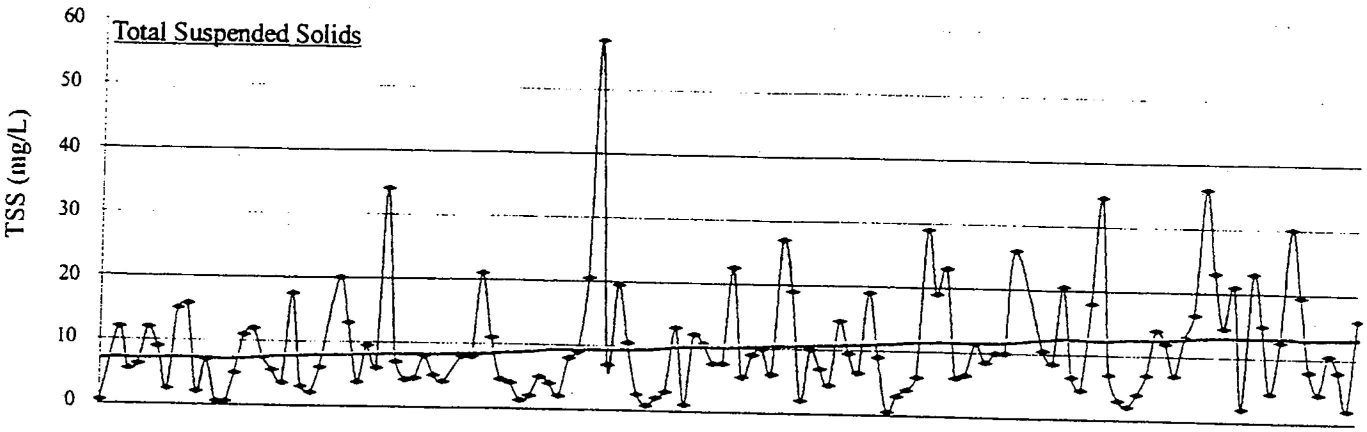


Figure III-6a. Monthly medians of surface nutrient water quality variables for PIS0033. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

PIS0033.CDR



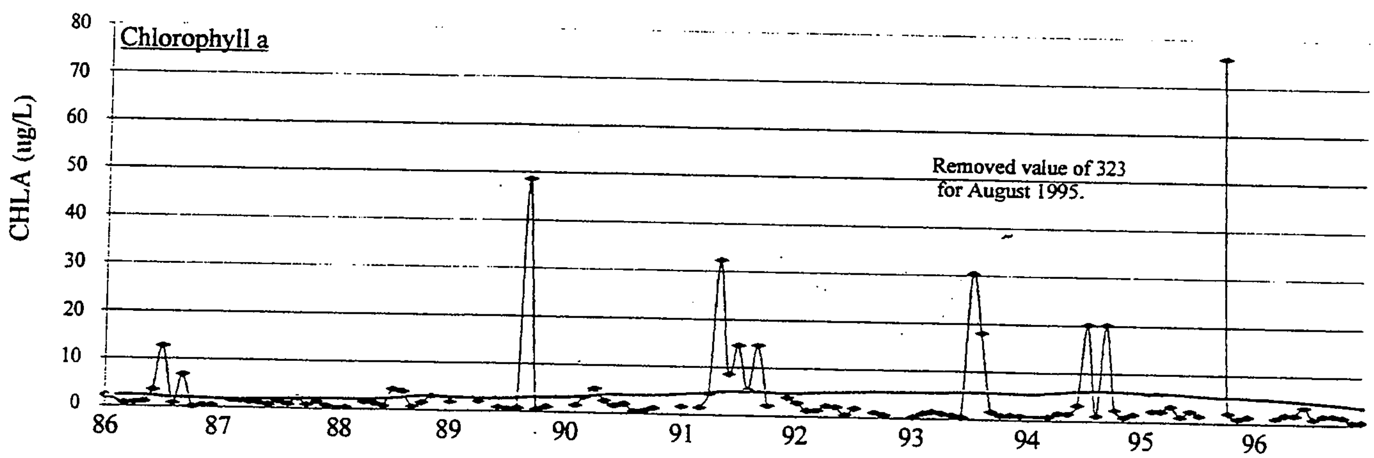


Figure III-6b. Monthly medians of surface non-nutrient water quality variables for PIS0033. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change. Station too shallow to monitor Secchi depth or bottom dissolved oxygen.

PIS0033.CDR

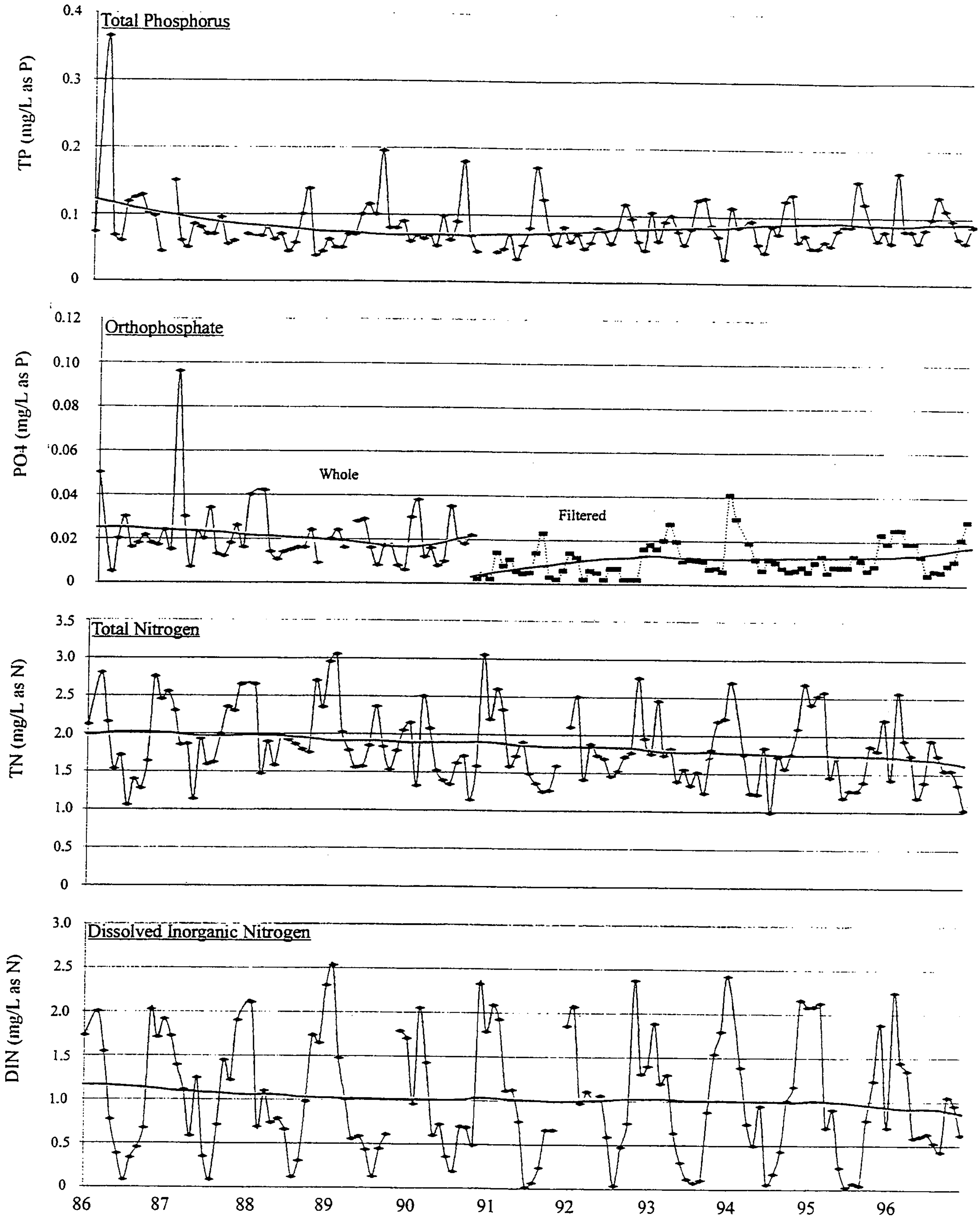


Figure III-7a. Monthly medians of surface nutrient water quality variables for MAT0016. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

MAT0016.CDR

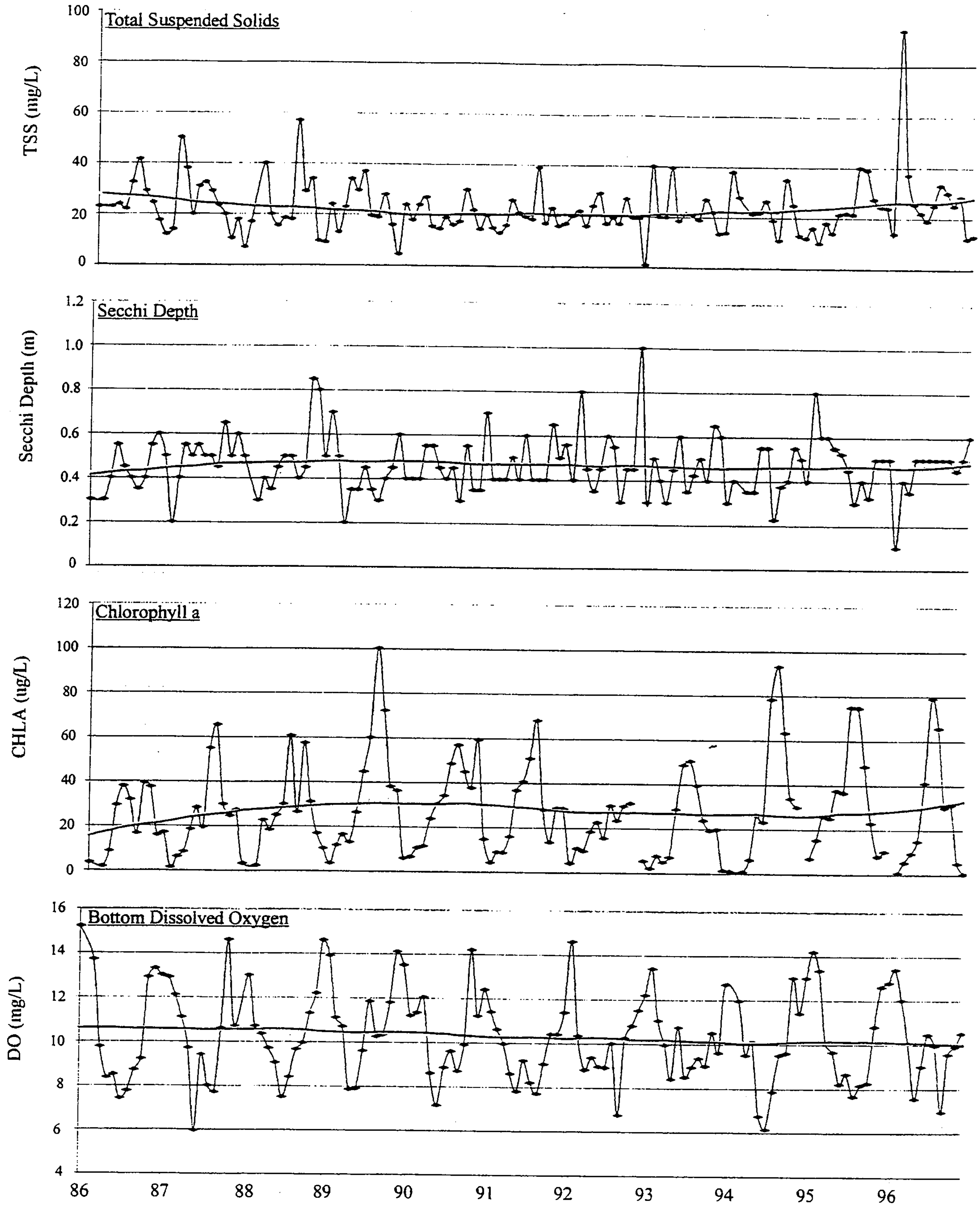


Figure III-7b. Monthly medians of surface non-nutrient water quality variables and bottom DO for MAT0016. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

MAT0016.CDR

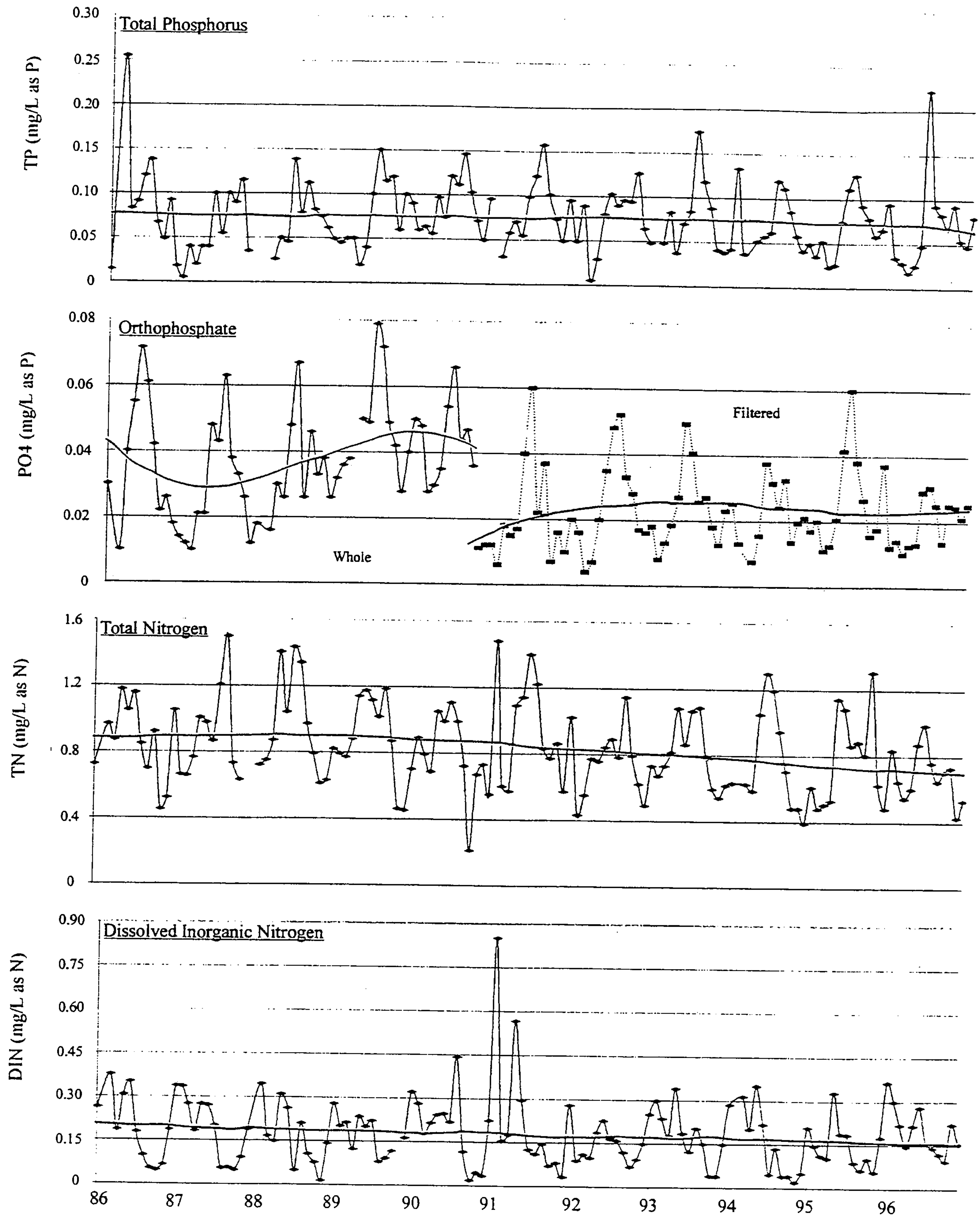


Figure III-8a. Monthly medians of surface nutrient water quality variables for MAT0078. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

MAT0078.CDR

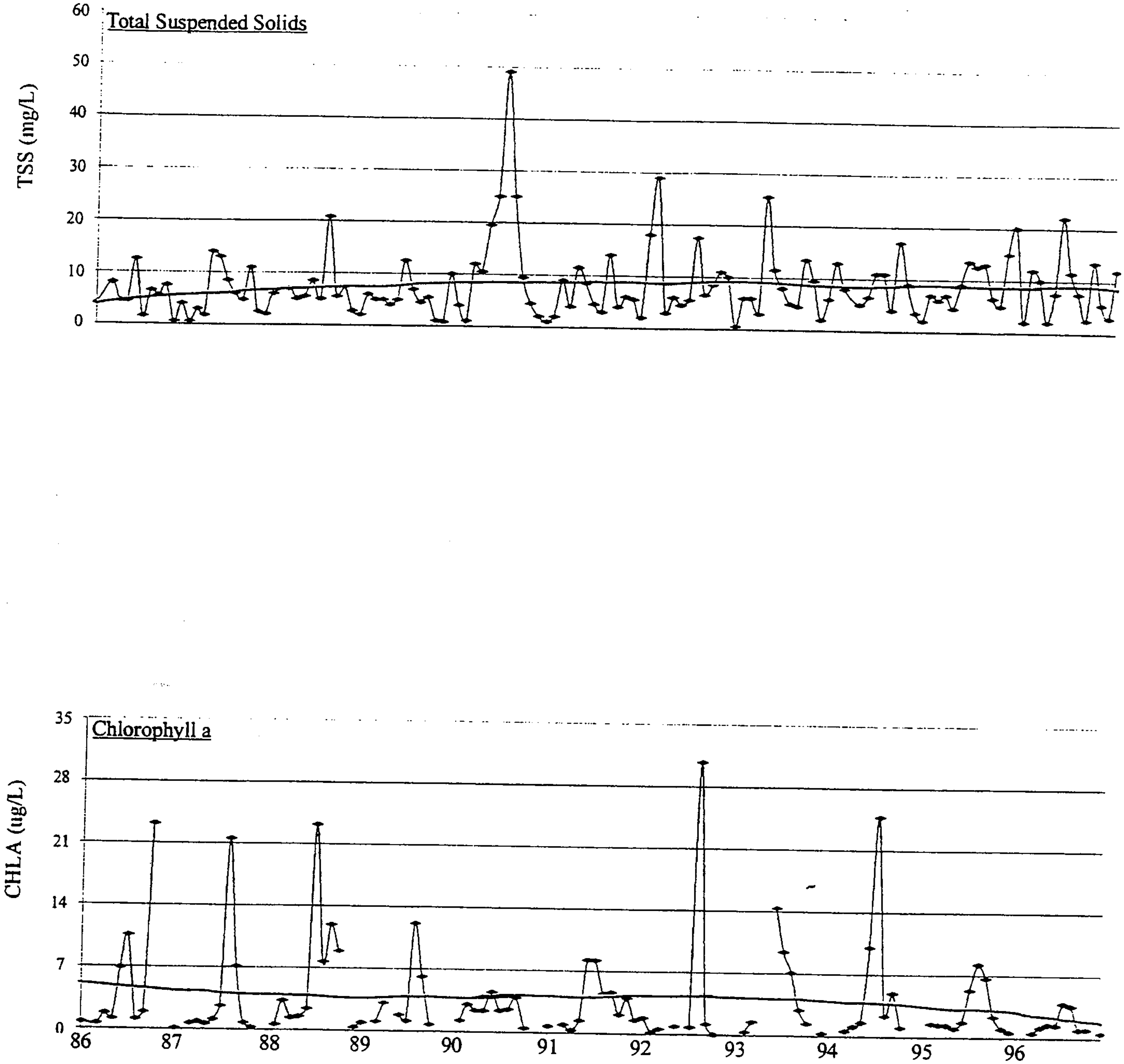


Figure III-8b. Monthly medians of surface non-nutrient water quality variables for MAT0078. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change. Station too shallow to monitor Secchi depth or bottom dissolved oxygen.

MAT0078.CDR

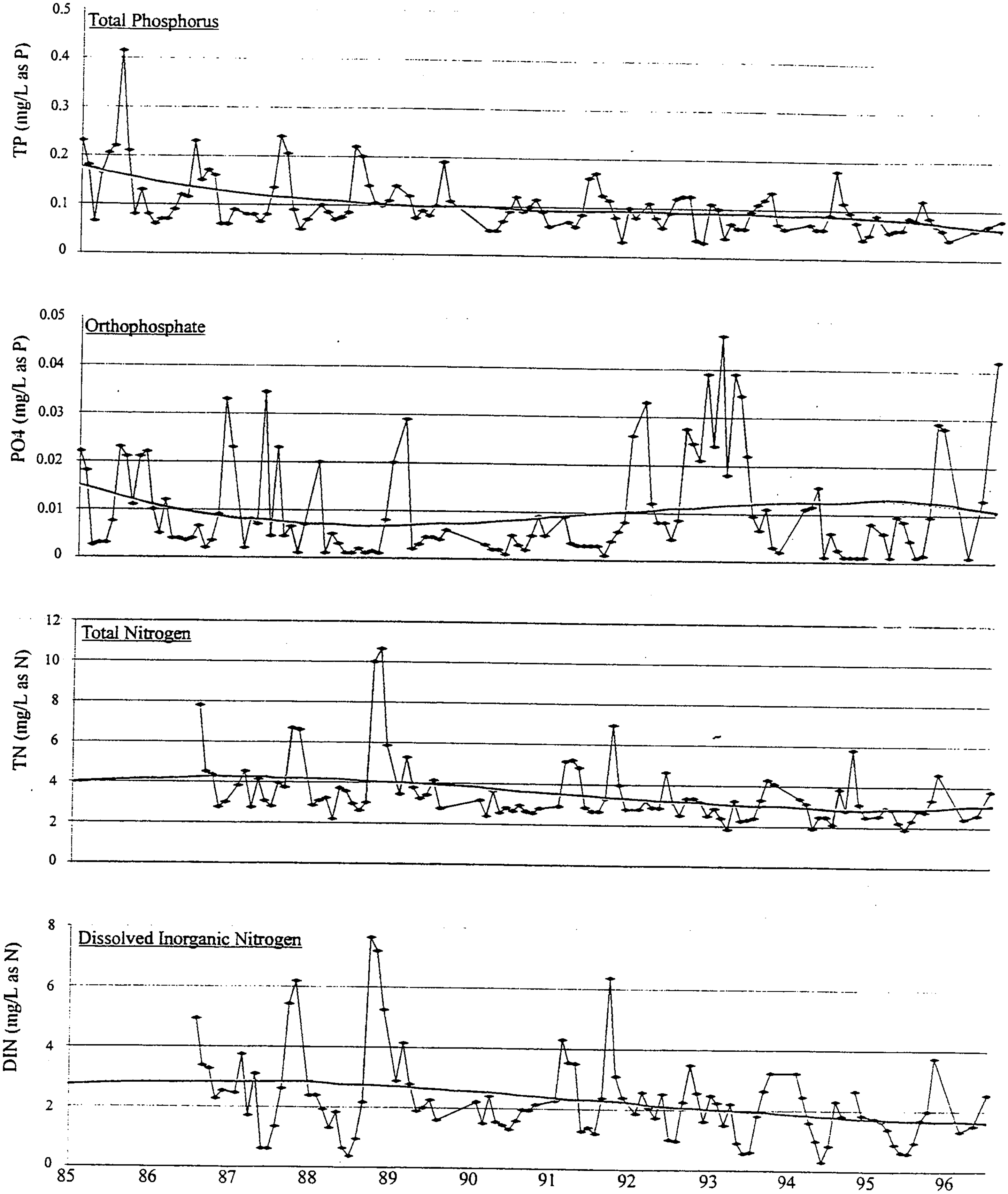


Figure III-9a. Monthly medians of surface nutrient water quality variables for Gunston 7. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change. Laboratory data were censored to highest detection limit of the time period.

GUN7.CDR

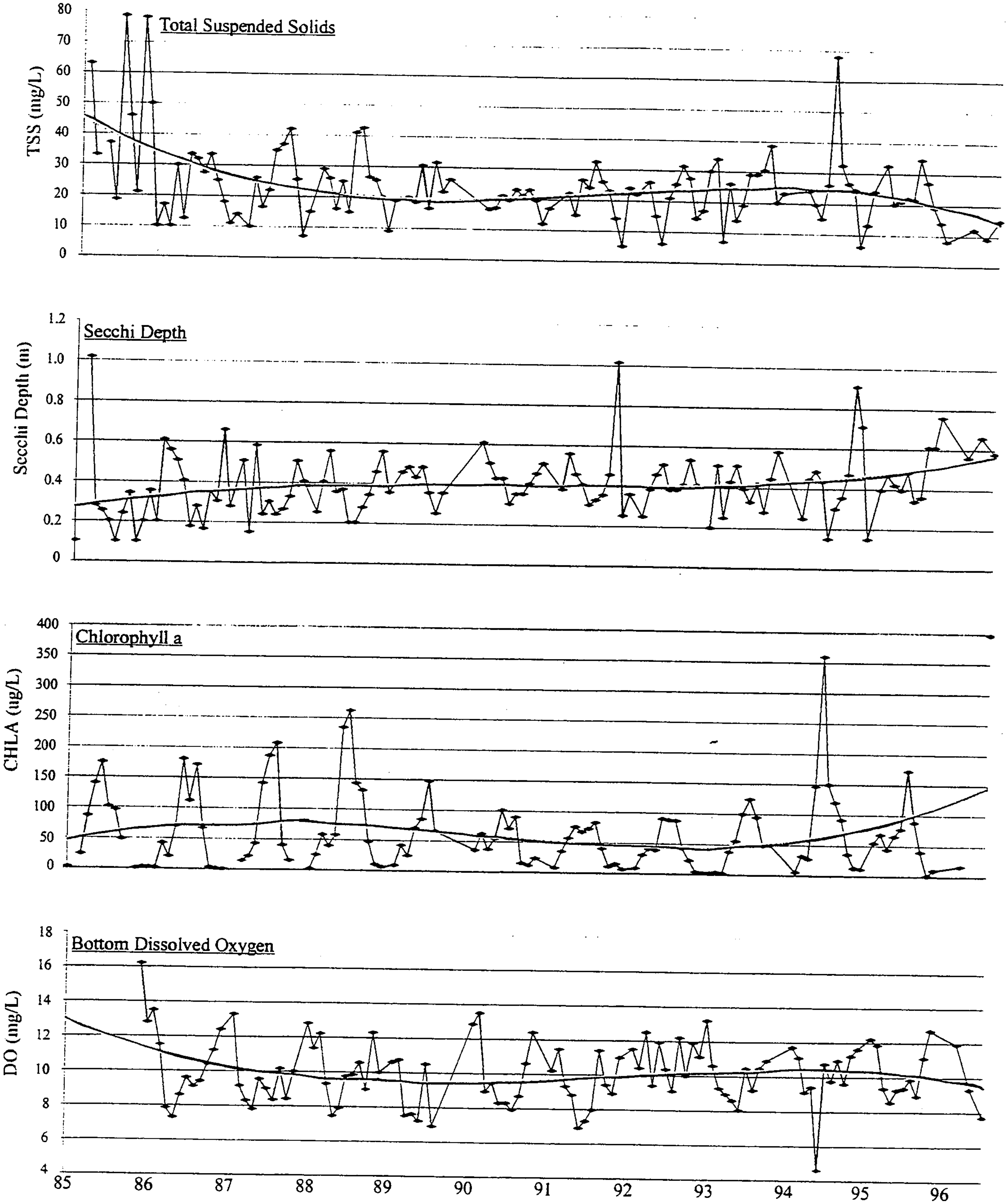


Figure III-9b. Monthly medians of surface non-nutrient water quality variables and bottom DO for Gunston 7. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change. Laboratory data were censored to highest detection limit of the time period.

GUN7.CDR

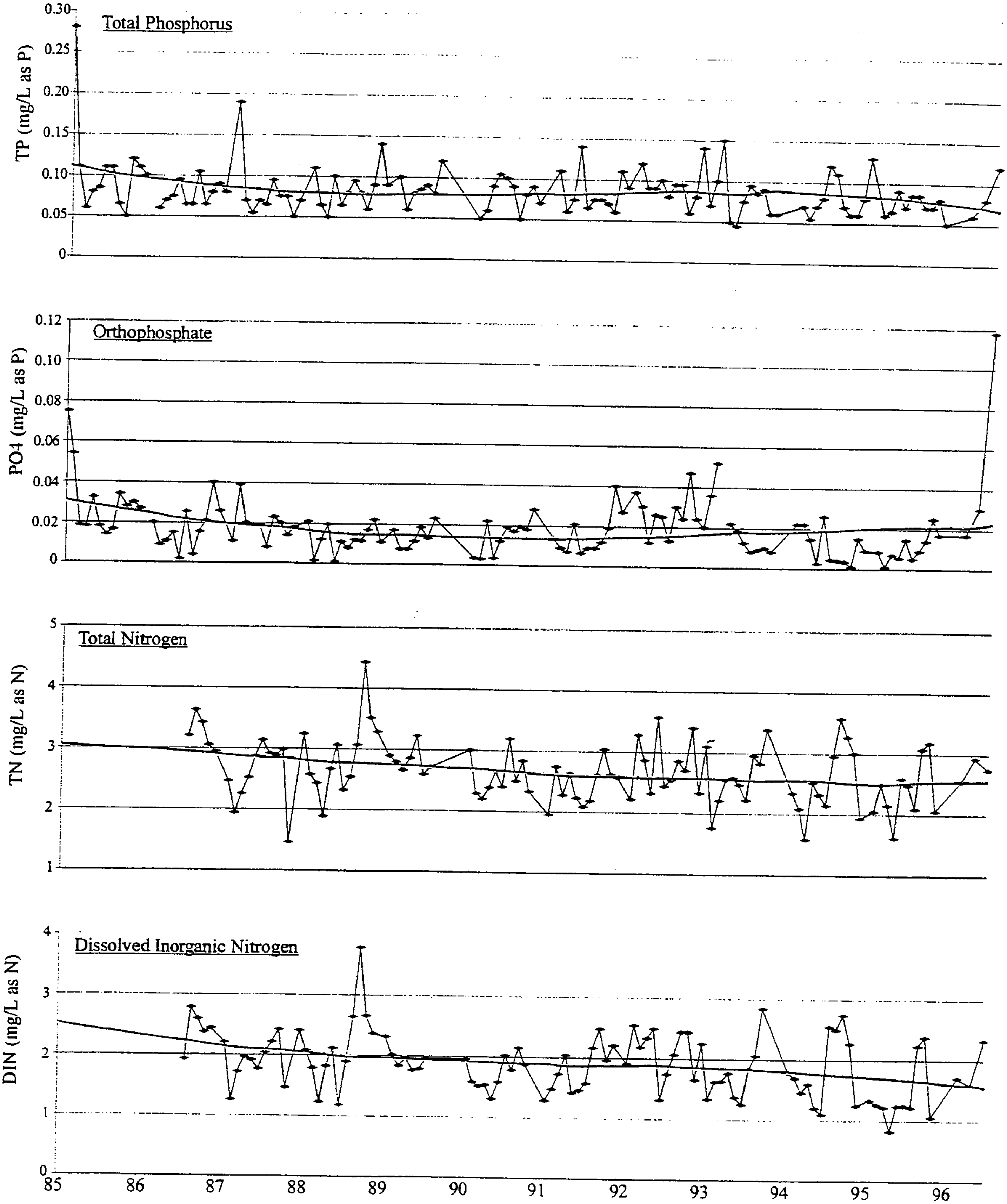


Figure III-10a. Monthly medians of surface nutrient water quality variables for Gunston 9. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change. Laboratory data were censored to highest detection limit of the time period.

GUN9.CDR

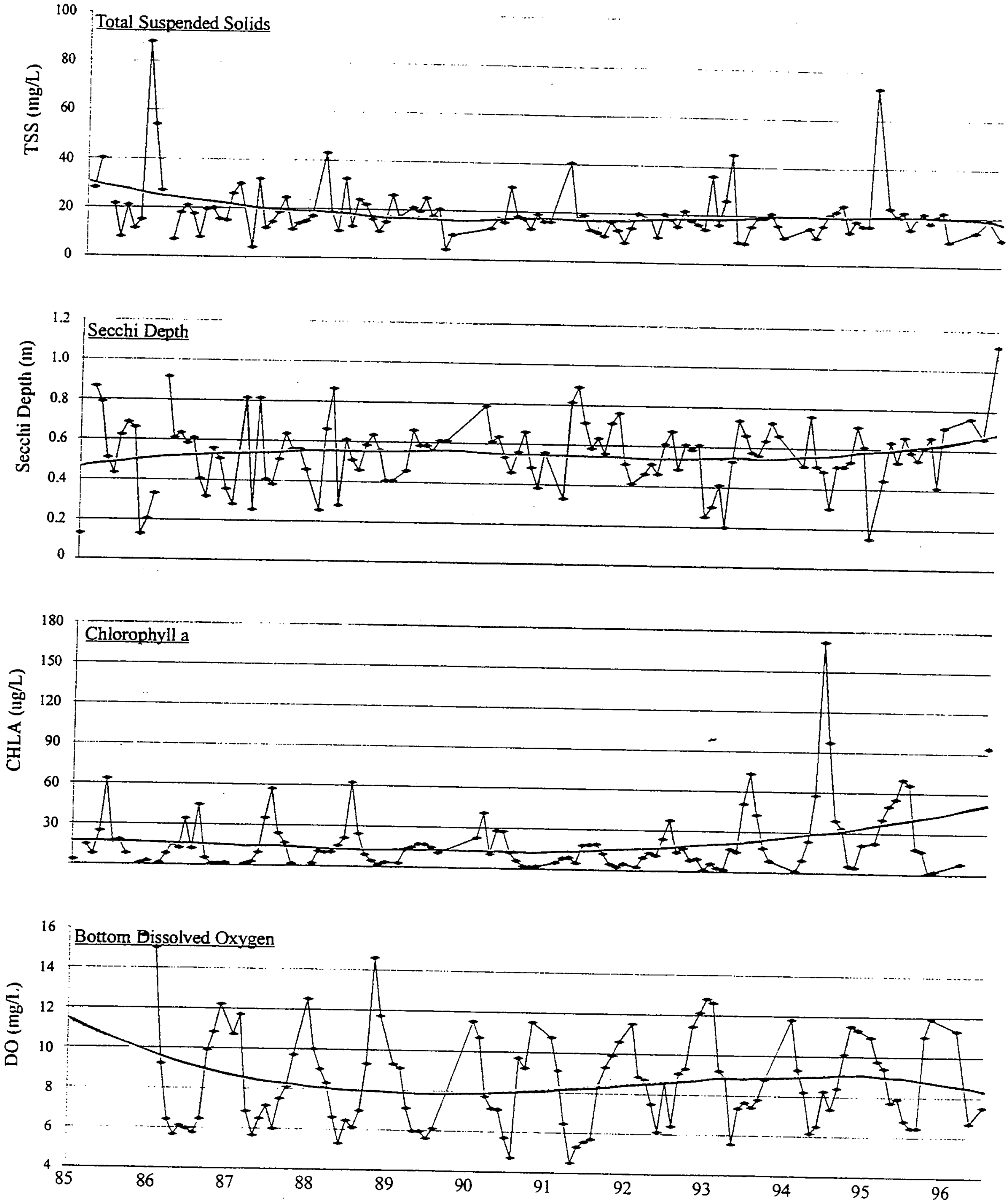


Figure III-10b. Monthly medians of surface non-nutrient water quality variables and bottom DO for Gunston 9. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change. Laboratory data were censored to highest detection limit of the time period.

GUN9.CDR

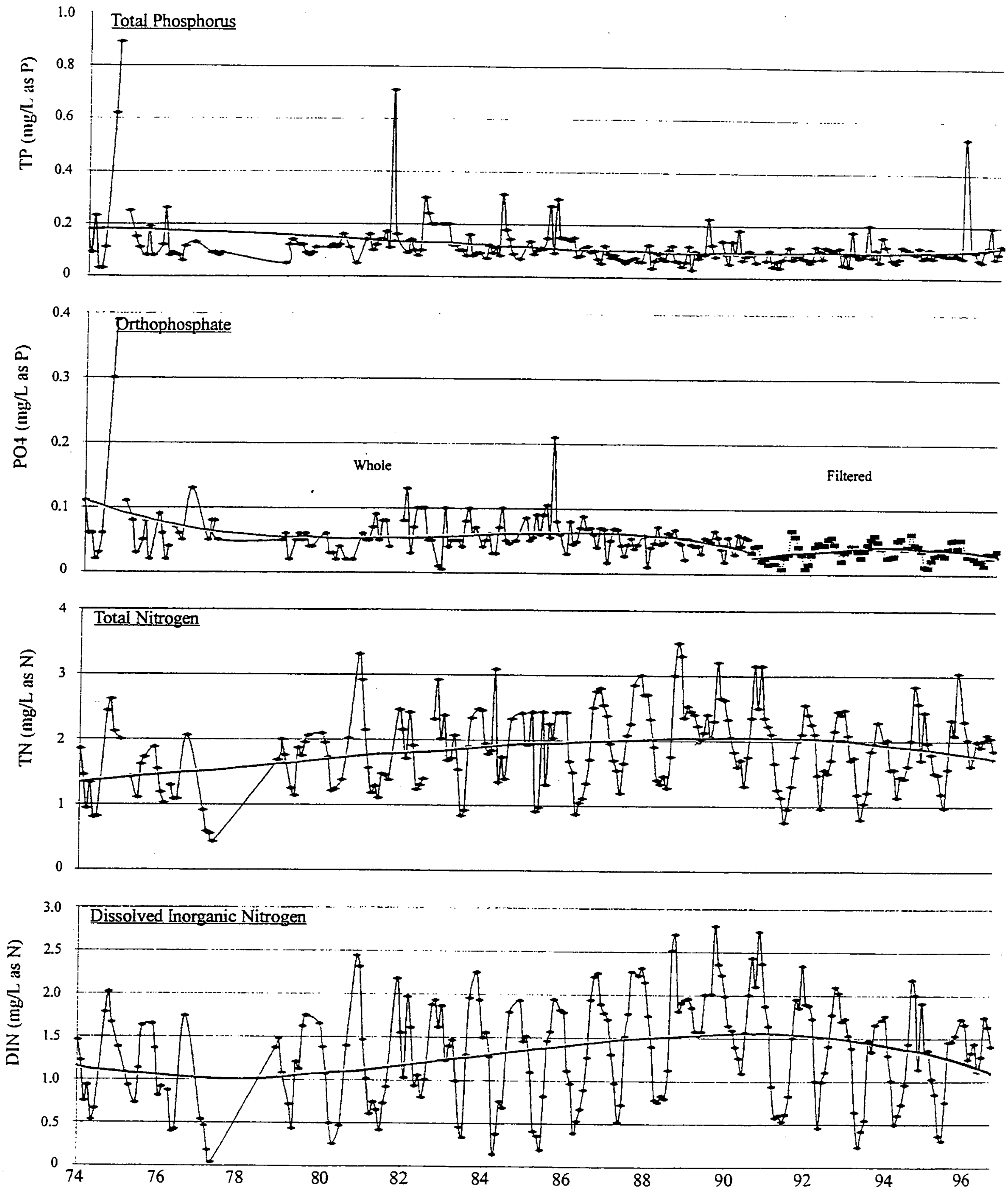


Figure III-11a. Monthly medians of surface nutrient water quality variables for XDA4238. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDA4238.CDR

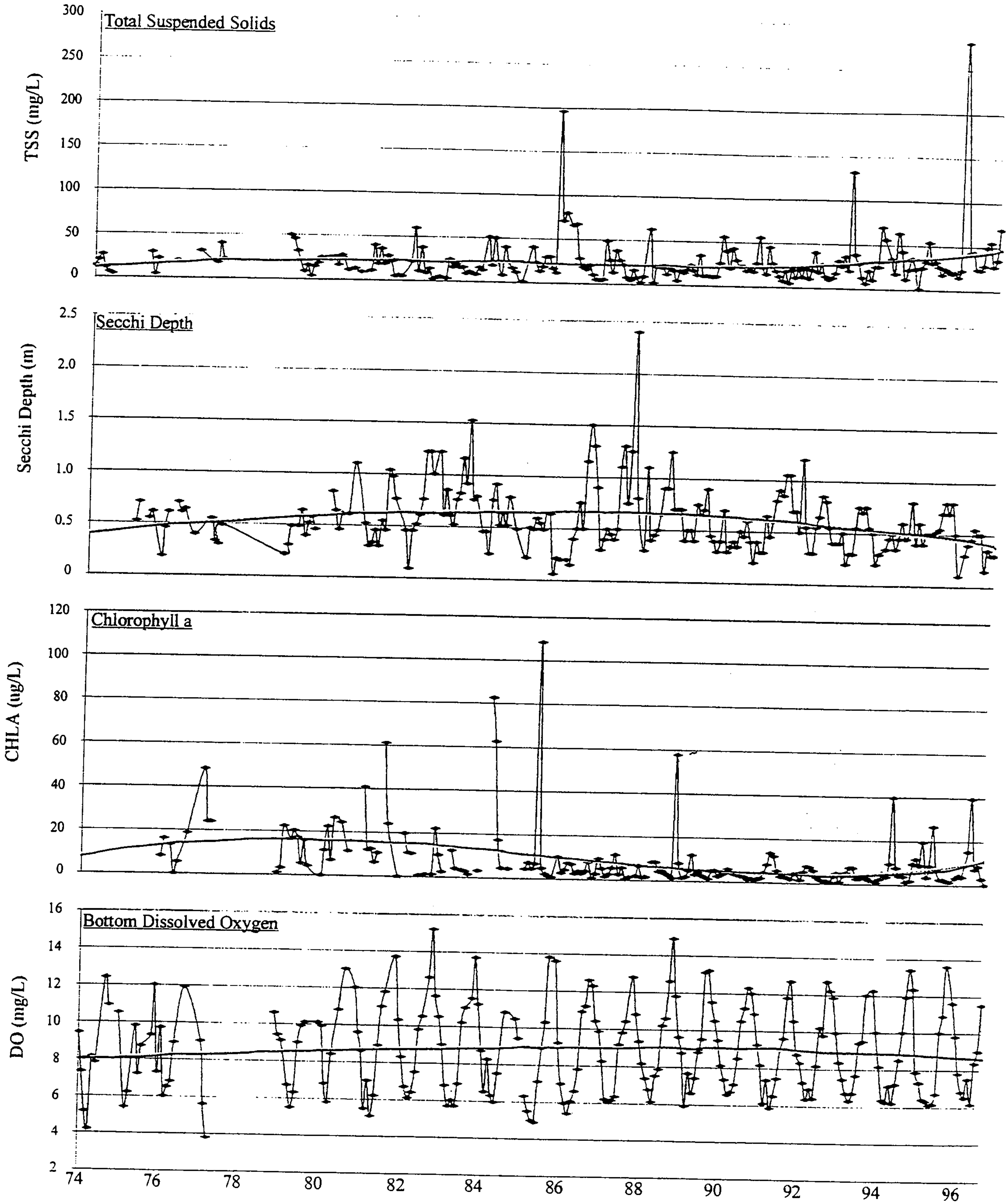


Figure III-11b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XDA4238. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDA4238.CDR

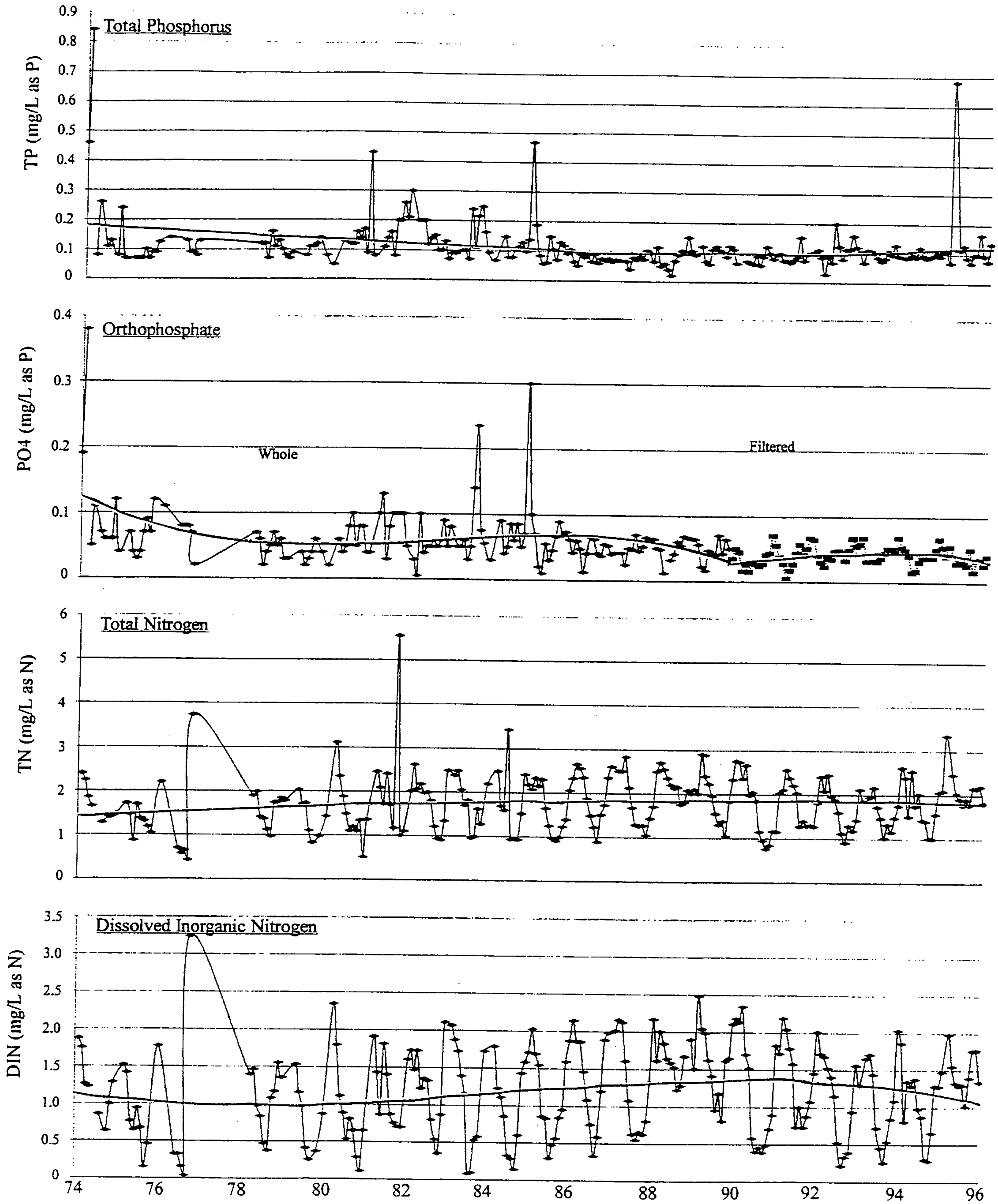


Figure III-12a. Monthly medians of surface nutrient water quality variables for XDA1177. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDA1177.CDR

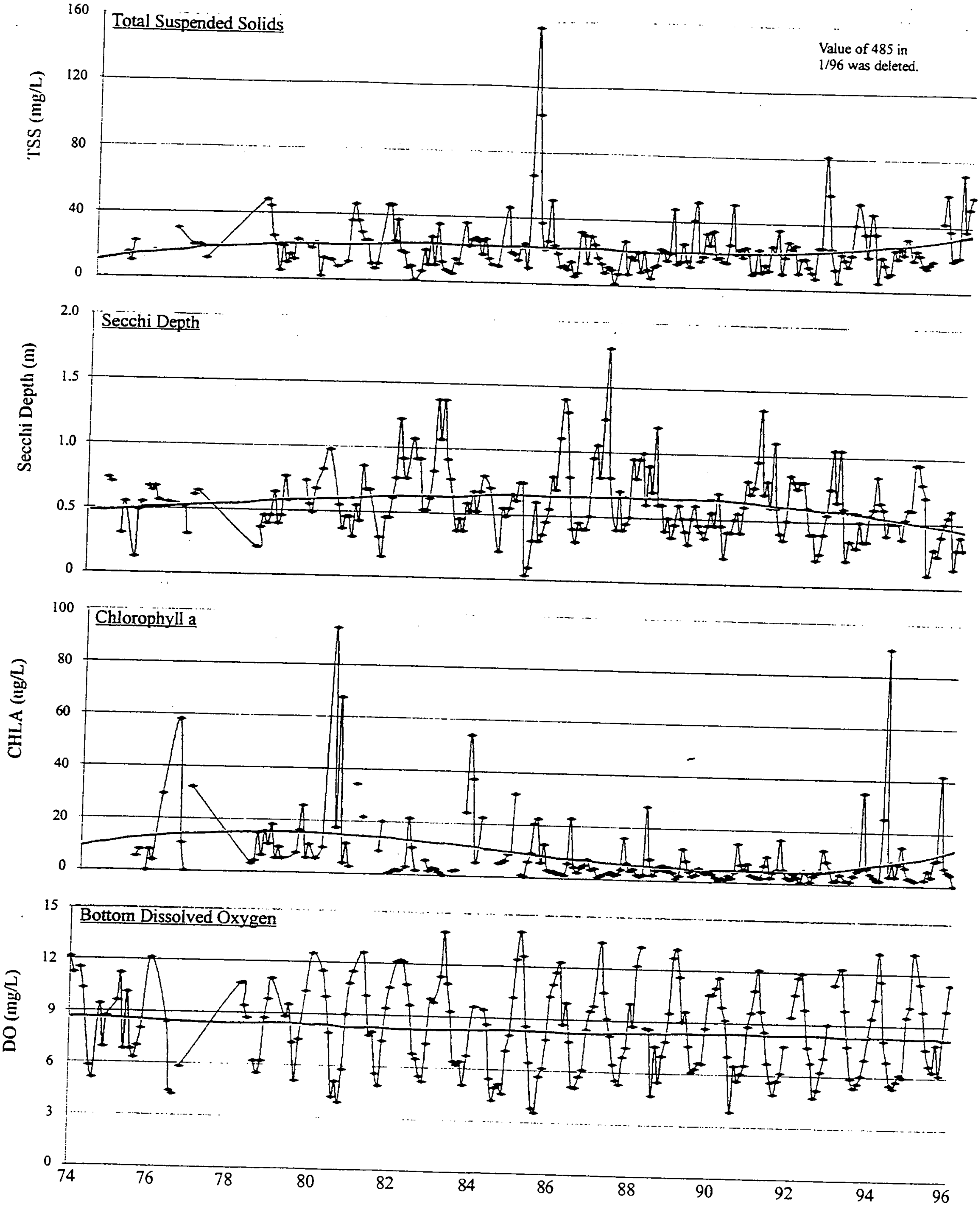


Figure III-12b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XDA1177. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDA1177.CDR

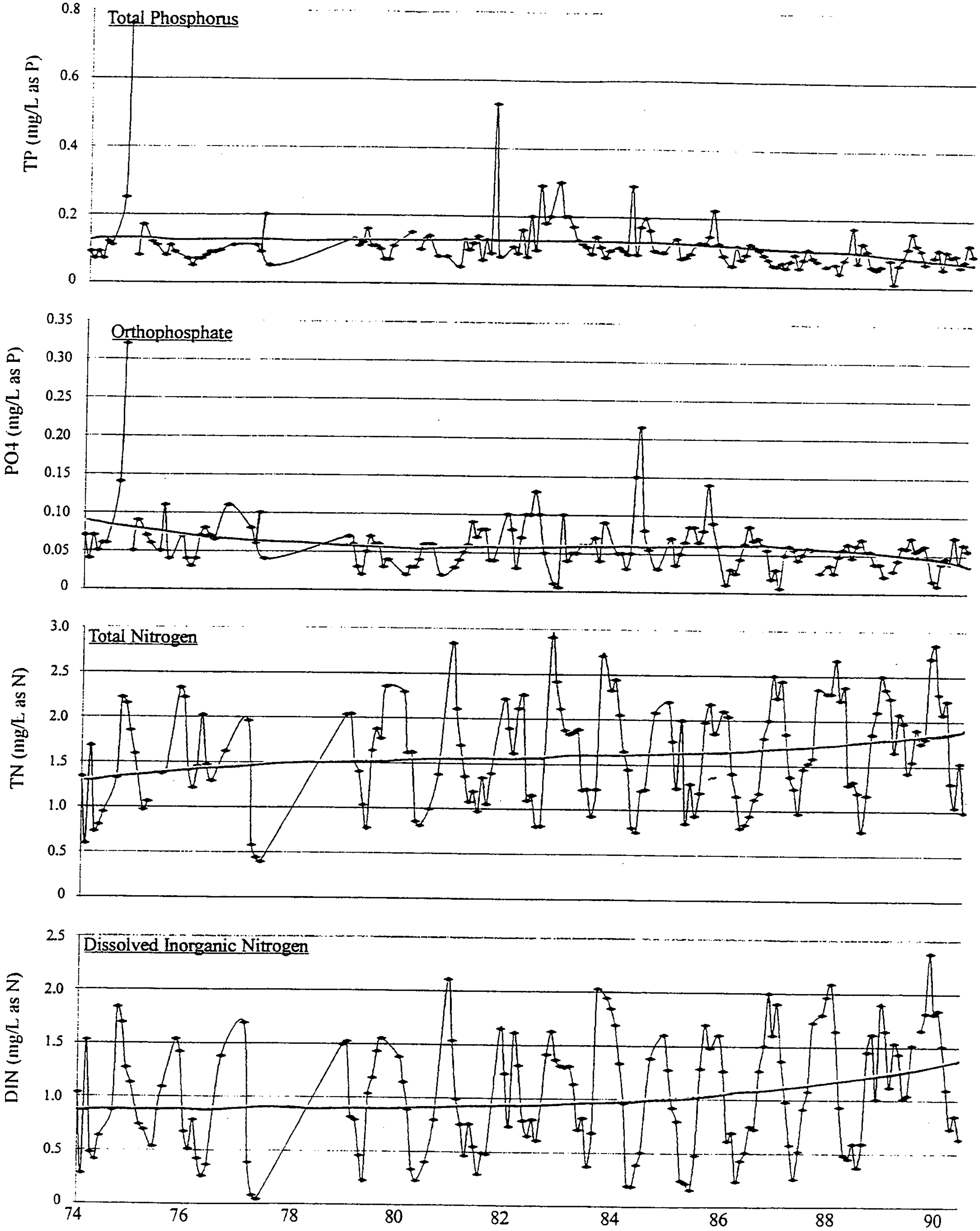


Figure III-13a. Monthly medians of surface nutrient water quality variables for XDB3321. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDB3321.CDR

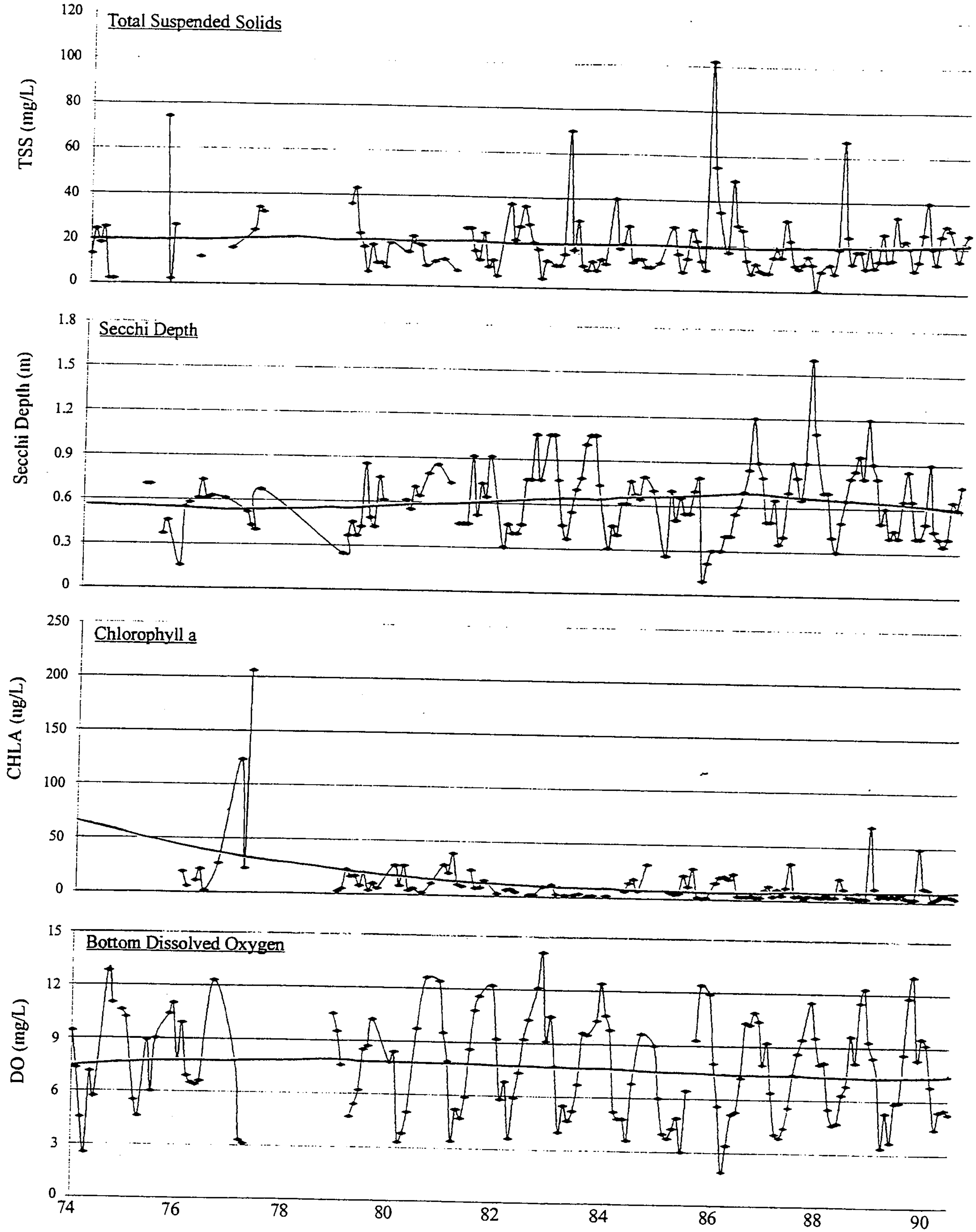


Figure III-13b. Monthly medians of surface non-nutrient water quality variables and bottom DO for XDB3321. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDB3321.CDR

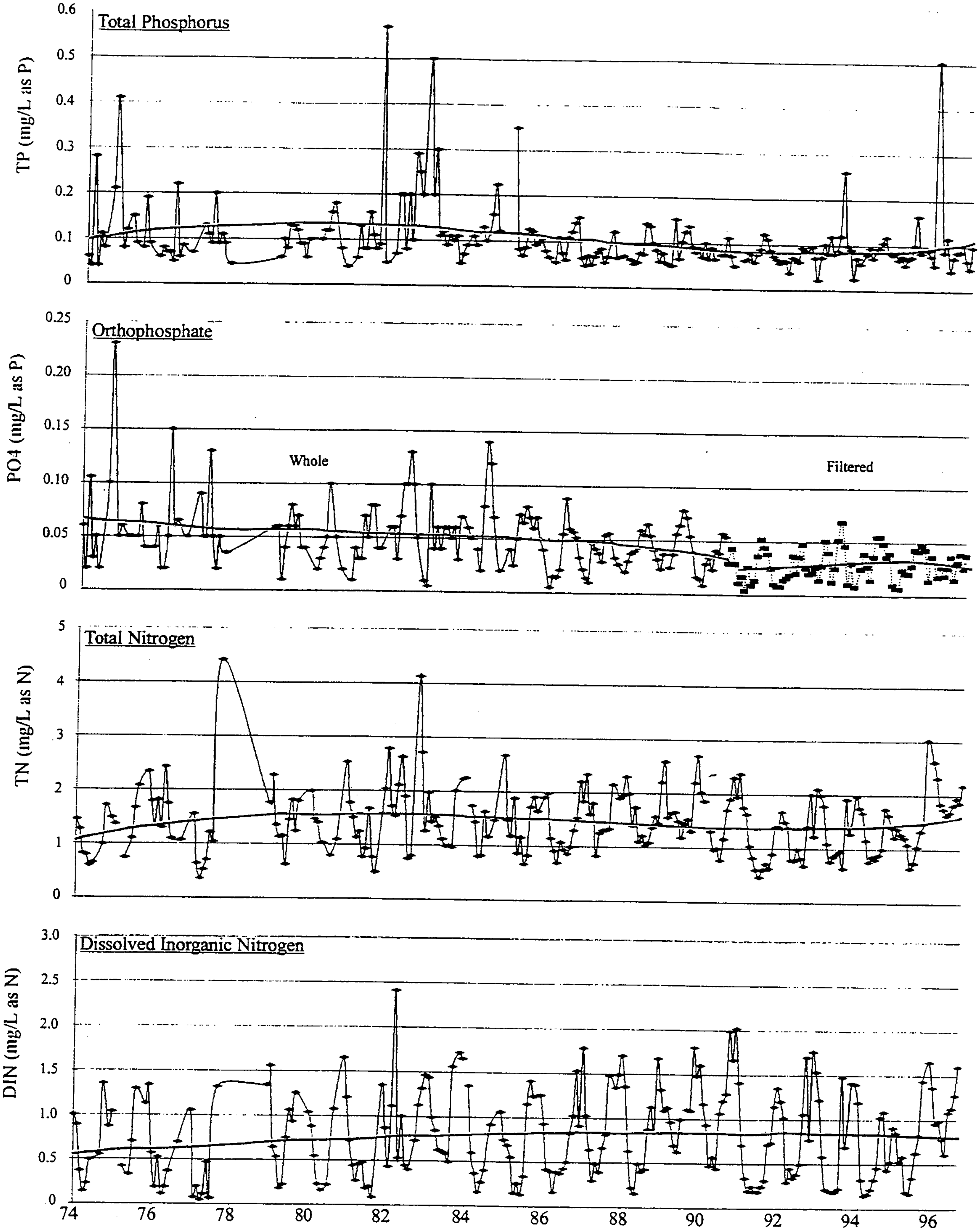


Figure III-14a. Monthly medians of surface nutrient water quality variables for XDC1706. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDC1706.CDR

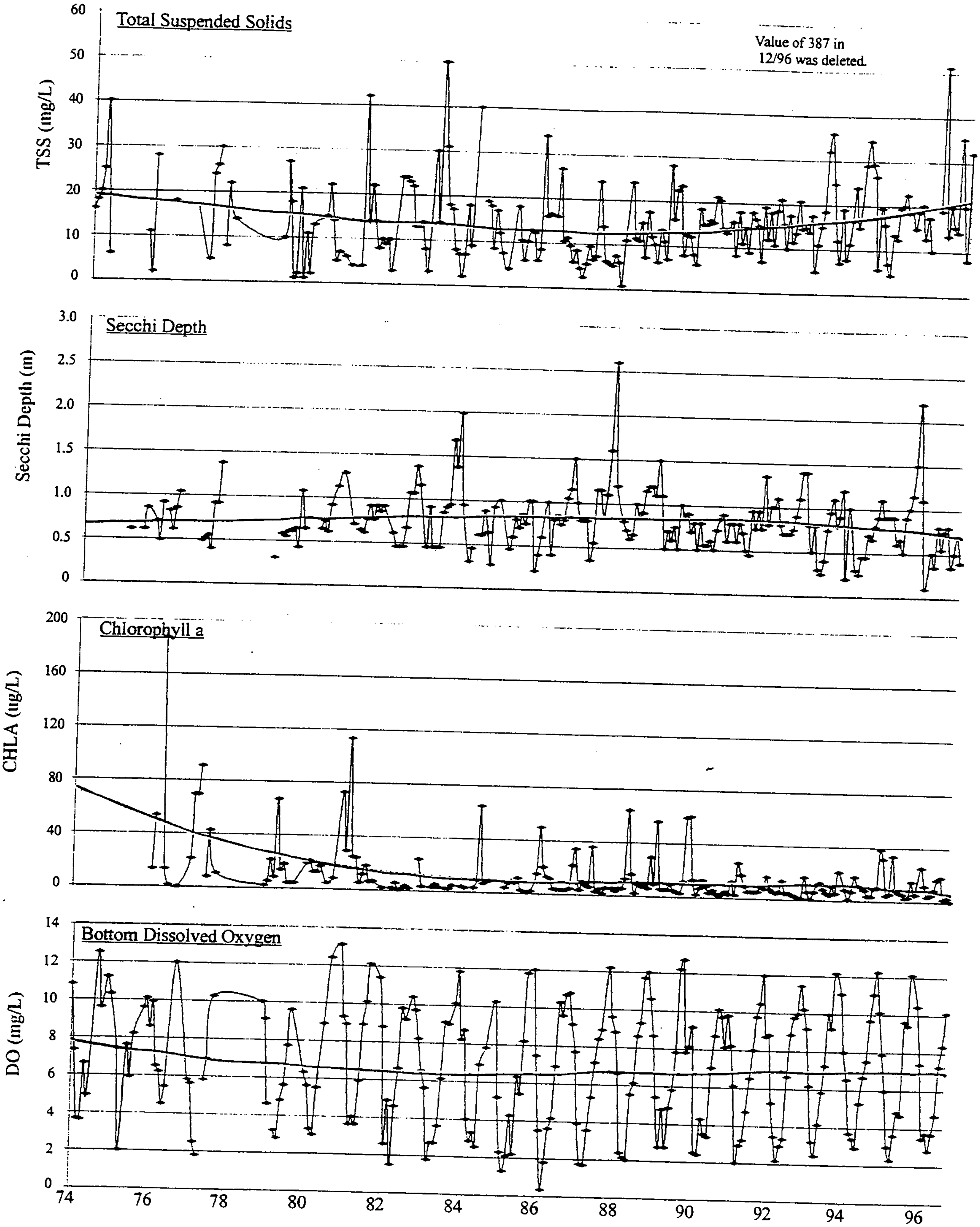


Figure III-14b. Monthly medians of surface nutrient water quality variables and bottom DO for XDC1706. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

XDC1706.CDR

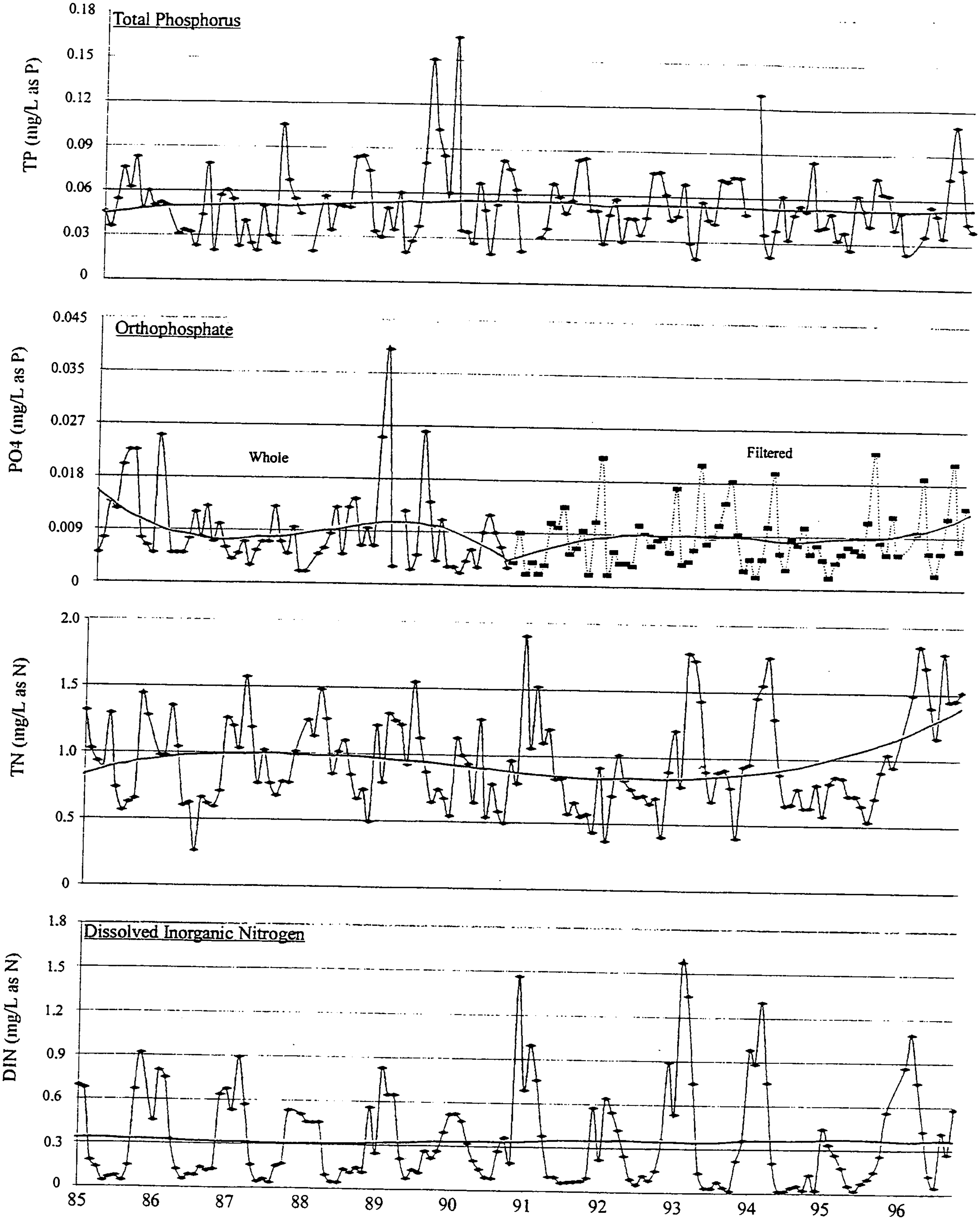


Figure III-15a. Monthly medians of surface nutrient water quality variables for MLE2.2. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

MLE22.CDR

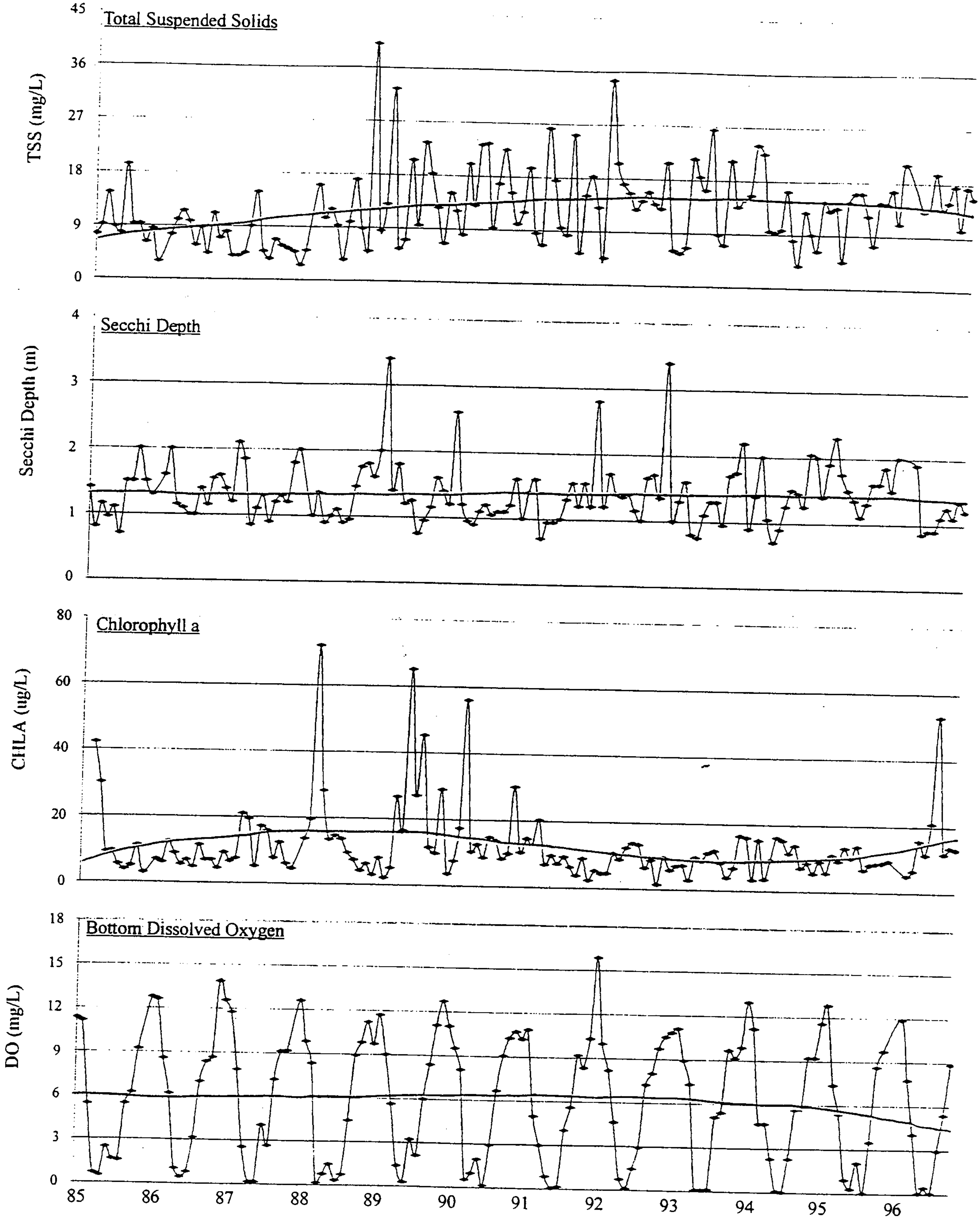


Figure III-15b. Monthly medians of surface non-nutrient water quality variables and bottom DO for MLE2.2. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

MLE22.CDR

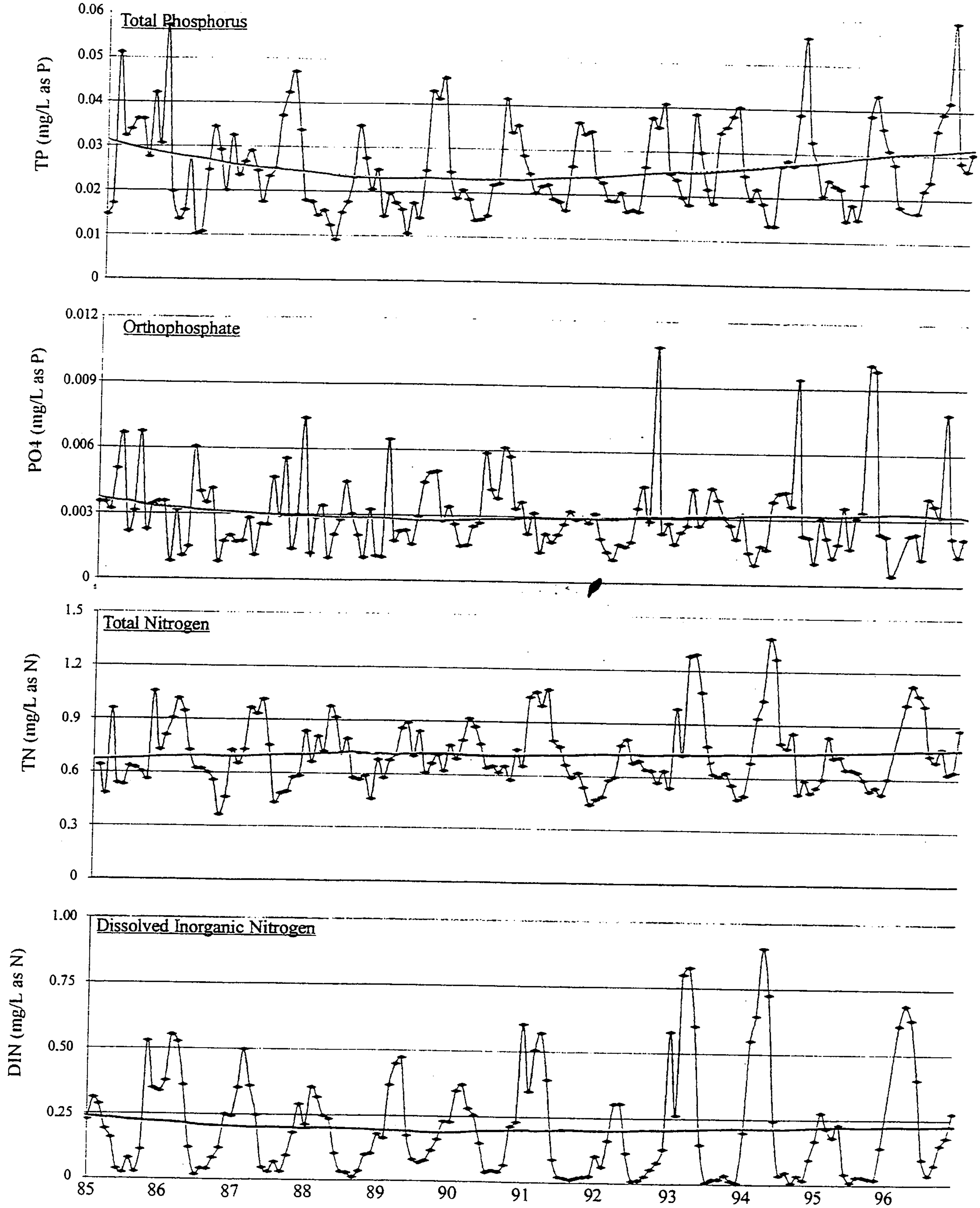


Figure III-16a. Monthly medians of surface nutrient water quality variables for MLE2.3. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

MLE23.CDR

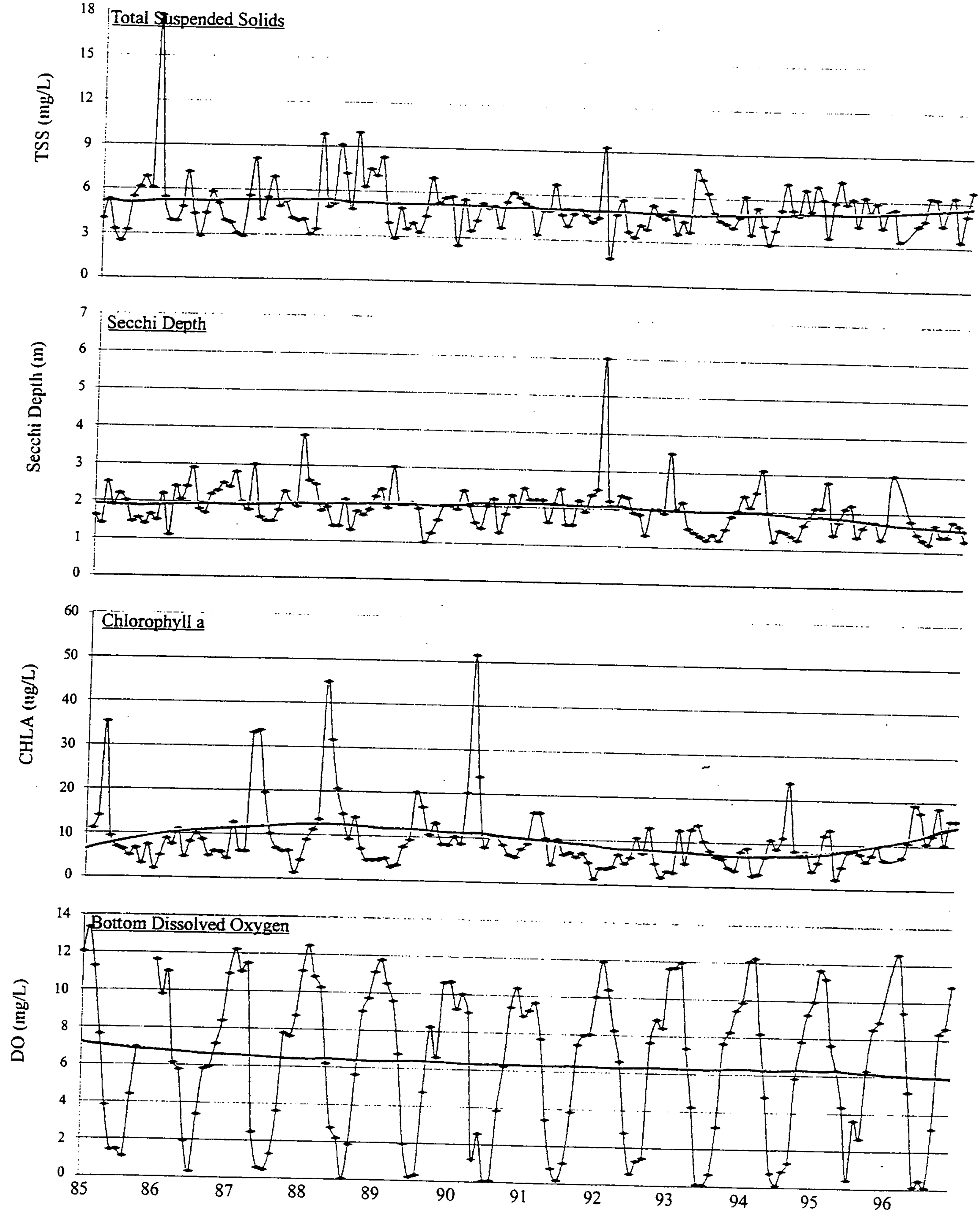


Figure III-16b. Monthly medians of surface non-nutrient water quality variables and bottom DO for MLE2.3. The trend line, fitted using third order polynomial regression, is presented only as an aid to visualization and is not the trend line used to calculate percent change.

MLE23.CDR

Table III-1. Potomac River Sampling Stations. River kilometer is defined as the distance from the river mouth. Fresh is defined as 10-year median salinity <0.5 ppt. oligohaline as salinity > 0.5 ppt and < 5.0 ppt., and mesohaline as salinity > 5.0 ppt. These correspond to the limnetic, oligohaline and mesohaline zones of the Venice System (Symposium, 1959).

Station/ CBP Segment	Latitude/ Longitude (NAD 27)	Location/Depth	River Km	Salinity Zone
XFB2470 POTFI	38°42'26" 77°02'57"	At FL buoy 77 off mouth of Piscataway Creek; 19m.	143.0 km	Tidal Fresh
XFB1433 POTTF1	38°41'26" 77°06'31"	Bouy 67 off mouth of Dogue Creek; 8m.	137.4 km	Tidal Fresh
XEA6596 POTTF1	38°36'29" 77°10'27"	Bouy N 54 mid-channel off Indianhead; 15m.	123.6 km	Tidal Fresh
XEA1840 POTTF1	38°31'47" 77°15'56"	Bouy 44 between Possum Pt. and Moss Pt.; 9m.	111.7 km	Tidal Fresh
PIS0033 POTTF3	38°41'54" 76°59'13"	At route 210 overpass; 0.5 m.	139.7 km	Tidal Fresh
XFB1986 POTTF3	38°41'51" 77°01'24"	Piscataway embayment, SE of buoy 6	142.9 km	Tidal Fresh
MAT0078 POTTF2	38°35'18" 77°07'08"	At route 225 overpass; 0.5 m.	113.3 km	Tidal Fresh
MAT0016 POTTF2	38°33'42" 77°13'09"	At FL buoy 5 in Mattawoman embayment.	113:3 km	Tidal Fresh
GUN07	38°39'22" 77°07'35"	Center of Gunston Cove.	133.7 km	Tidal Fresh
GUN09	38°39'22" 77°07'35"	Mouth of Gunston Cove at mainstem	133.7 km	Tidal Fresh
XDA4238 POTOH	38°24'12" 77°16'10"	Buoy 27 SW of Smith Pt.; 8m.	99.1 km	Oligohaline
XDA1177 POTOH	38°21'07" 77°12'17"	Bouy 19 mid-channel off Maryland Pt.; 11m.	88.5 km	Oligohaline
XDB3321 POTOH	38°23'17" 77°07'51"	Bouy 13 off mouth of Nanjemoy Creek; 9m.	78.5 km	Oligohaline
XDC1706 POTMH	38°21'45" 76°59'27"	Mid-channel at Morgantown Bridge (U.S. Rt. 301); 19m.	64.4 km	Mesohaline
MLE2.2 POTMH	38°10'00" 76°44'00"	Potomac River off Ragged Pt. at bouy 51B; 10m.	19.1 km	Mesohaline
MLE2.3 POTMH	38°01'18" 76°21'00"	Mouth of Potomac River; 19.8m.	0 km	Mesohaline

IV. CHANGES IN LOADINGS AND FLOW AT THE WWPTs AND THE FALL-LINE

A. Wastewater Treatment Plant Loadings and Flow

i. Historical Perspective on WWTPs

In the 1860s, well before the construction of the first WWTPs in the Washington, D.C., area, raw sewage was dumped into canals that led from the city to the Potomac River. In 1870, the first sewers were constructed to convey sanitary wastes to the Potomac downstream of Washington, D.C. A 1913 U.S. Public Health Service survey of the bacterial quality of the Potomac River concluded that "(1) at no point below Washington, D.C., is the water of the Potomac River safe for use as a public water supply without reasonable treatment; (2) the portions of the main of Georgetown Channel between Chain Bridge and the junction of the main Channel with the Anacostia River and the Washington Channel are so heavily polluted that the water is unsafe for bathing purposes; (3) the conditions of the areas in the Anacostia River in the neighborhood of the sewage pumping stations and at the junction of the three channels are poor during hot summer weather and at times constitute a public health nuisance" (Jaworski, 1990).

In 1920, Washington, D.C., city officials agreed that sewage should be treated before being discharged to the Potomac River. In 1938 the Blue Plains WWTP came on-line with primary treatment, which removed a portion of the suspended solids and organic matter. Primary treatment, the most rudimentary level of treatment, was achieved with physical operations such as screening and sedimentation. Treated effluent still contained a substantial amount of organic matter and would have had a comparatively high biochemical oxygen demand (BOD) (Tchobanoglous and Schroeder, 1985).

Despite the primary treatment of 5.7 m^3/sec (130 millions-of-gallons per day) of effluent at the Blue Plains WWTP (the 1938 plant capacity), water quality of the Potomac River continued to deteriorate. In 1957, the Potomac Washington Metropolitan Area Enforcement Conference was convened in response to the continued degrading water quality conditions. The Conference, which initially met in 1957 and again in 1958, developed a program requiring a minimum wastewater treatment of 80% BOD removal, effluent disinfection, and storm water overflow control by 1966. Secondary treatment of effluent, to remove residual organic matter and suspended material, was initiated at Blue Plains in 1958 in response to the 1957 Conference.

A third session of the Conference was convened in 1969 in response to the continued degradation of water quality conditions in the Potomac River. Several conclusions were reached at the 1969 Conference, among the more important: (1) waste water discharges to the Potomac should be based on specific loads as opposed to percent removal limits; (2) 96% of BOD and phosphorus and 95% of nitrogen should be removed; (3) the technology existed for a high degree of BOD, phosphorus, and nitrogen removal; (4) construction should be completed by 1977 (Jaworski, 1990).

The third Conference was reconvened in 1970 and a Memorandum of Understanding (MOU) between Maryland, Virginia, and Washington, D.C., was adopted. The MOU set specific loadings limits for the upper Potomac estuary for ultimate oxygen demand, nitrogen, and phosphorus, which essentially set limits on the capacity of Blue Plains. Limiting the capacity of Blue Plains resulted in the need to site several smaller regional plants to accommodate the needs of a growing population.

Water quality improvements in the Potomac River occurred following the signing of the Clean Water Act in 1972 and the creation of the National Pollutant Discharge Elimination System (NPDES). By 1974, permits limiting the discharge of BOD, suspended solids, phosphorus, and nitrogen were issued to dischargers in the Potomac Estuary. The Clean Water Act resulted in funding by the U.S. Environmental Protection Agency and state and local governments to implement the guidelines of the second Potomac Enforcement Conference. Upgrades to the WWTPs included the addition of secondary treatment and advanced treatment to further reduce nutrient discharges (Michael, 1989).

An Adjudicatory Hearing was held in 1976 to reconsider the limits on nitrogen discharges at Blue Plains. Based on the cost of energy in the mid-1970s and the lack of nitrogen removal technology, the nitrogen removal requirements were relaxed. Phosphorus removal and nitrification of wastewater to reduce nitrogenous BOD continued at Blue Plains, but nitrogen removal was not required.

In a continuing effort to reduce phosphorus loads to Maryland waters, including the Potomac River, the Maryland legislature passed a law in 1985 banning phosphates in detergents. The Maryland phosphate detergent ban went into effect the following year. Washington, D.C., initiated a phosphate detergent ban in 1986 and the Commonwealth of Virginia followed suit in 1987. In September 1996 a pilot program was initiated at Blue Plains to denitrify the effluent in the hope of reducing ambient nitrogen concentrations in the Potomac River.

Currently, there are 13 major (permitted flow level of 0.02 m^3/sec or greater) WWTPs and one major industrial discharger located in the Washington, D.C., metropolitan area. The largest WWTP, Blue Plains, is located in Washington, D.C.; two WWTPs (Mattawoman and Piscataway) and the large industrial discharger (Indian Head Naval Warfare Center) are located in Maryland and the remaining ten WWTPs (Alexandria, Arlington, Aquia, Dale City #1, Dale City #8, Little Hunting Creek, Lower Potomac, Mooney, Quantico, Upper Occoquan) are located in Virginia (Fig. IV-1) (Note that Little Hunting Creek went off-line in early 1992. Subsequent to the plant closure, wastewater was sent to the Lower Potomac Plant, which has since been renamed the Noman M. Cole Water Pollution Center). Because of the population density, point source loads from WWTPs have historically been the primary source of nutrient pollution. During the 1970s and 1980s, aggressive pollution abatement programs described above resulted in substantial up-grades to the major plants. The plant names, year when the plant went on-line, the receiving stream, and the salinity zone of the Potomac River into which the receiving stream flows are presented in Table IV-1.

ii. Changes in TN Loads and Flows

Figure IV-1 shows the total nitrogen loads and flows from 1913 to 1996. For that time period, the lowest (estimated) TN load of 2900 kg/day and the lowest flow of 1.8 m^3/sec occurred in 1913. The maximum TN load of 31043 kg/day in occurred 1990 and the maximum flow of 23.2 m^3/sec was reached in 1996.

For the period of record, TN loads increased from 2900 kg/day to 25289 kg/day, an increase of 772%. During that period, flow increased from 1.8 m^3/sec to 23.2 m^3/sec, or by 1167%. From 1970 to 1996, TN loads increased from 23449 kg/day to 25289 kg/day, or by 8%. For the same period, flow increased from 13.7 m^3/sec to 23.2 m^3/sec, or by 70%. Between 1985 and 1996, TN loads actually decreased from 27004 kg/day to 25289 kg/day, or by 6%. The decrease in TN loads occurred despite an increase in flow from 19.2 m^3/sec to 23.2 m^3/sec or 21%. The disconnect between TN load and flow occurred in 1990, when TN load reached a maximum of 31043 kg/day and flow was 21.4 m^3/sec. Between 1990 and 1996 TN loads decreased from 31043 kg/day to 25289 kg/day or by 19%; however, flow increased from 21.4 m^3/sec to 23.2 m^3/sec, or by 8% during that period.

Between 1995 and 1996 flow increased by 11%, but TN loads dropped by 14.6%. As stated above, biological nutrient removal (BNR) of nitrogen was initiated at Blue Plains in September 1996. It would be interesting if the drop in TN load could be attributed to the Blue Plains BNR pilot program. However, a comparison of TN concentrations for the major plants indicated that the change resulted from a sharp reduction in TN effluent concentrations between 1995 and 1996 at the Indian Head Naval Warfare Center.

iii. Changes in TP Loads and Flows

TP loads and flows from 1913 to 1996 are also presented in Figure IV-1. As shown in the figure, the total phosphorus story is quite different from that described above for total nitrogen, since phosphorus control was historically the focus of the WWTP upgrades. TP loads increased from 500 kg/day in 1913 (estimated) to a maximum of 9825 kg/day in 1970, an increase of over 1800%. For the same period, flow increased from 1.8 m^3/sec to 13.7 m^3/sec, or by approximately 648%. Between 1970 and 1996 TP loads decreased by 98%, or from 9825 kg/day to 187 kg/day, despite an increase in flow of 70%. For the 1985 through 1996 time period, TP loads declined, from 332 kg/day to187 kg/day, probably as a result of the phosphate detergent ban, even though flow continued to increase.

iv. Changes in PO_4 Loads and Flows

Orthophosphate loads for 1985-1996 are provided in Figure IV-2. Unlike TN and TP, earlier data for PO_4 are not available. In addition, the plant records for PO_4 are not complete. Data for Blue Plains, Lower Potomac, Mattawoman, Piscataway, and Upper Occoquan are available for 1985 through 1996. No PO_4 data are available for Quantico. Limited data are available for the remaining plants.

The highest PO_4 load (185 kg/day) for the 1985-1996 time period occurred in 1987 although the highest flow rate (23 m^3/sec) was not recorded until 1996. PO_4 loads decreased in 1988 and 1989, following the implementation of the phosphate detergent ban. Loads increased slightly to 164 kg/day in the following year and then gradually decreased through 1996. For the 1985 to 1996 period, PO_4 loads decreased by 36% despite an increase in flow of 20%.

v. Loads Contributed by Individual Plants

Figure IV-3 shows the percent of the TN load contributed by each plant for the 1985-1996 period. The highest percentage of the TN load (60%) comes from Blue Plains, with Alexandria a distant second at 11%. Lower Potomac contributes 10%, with Arlington and Upper Occoquan contributing 5% and 4%, respectively. Indian Head Naval Warfare Center, and Piscataway each contribute 3%.

Blue Plains also contributes the highest percentage (57%) of the TP load (see Figure IV-4). Indian Head Naval Warfare Center and Mattawoman each contribute 8% of the TP load, and Arlington and Lower Potomac contribute 7% and 6%, respectively.

The percent of the PO_4 load contributed by each plant is presented in Figure IV-5. As discussed above, the PO_4 data set is not as complete as the TN and TP data sets. Consequently, the plant contributions described herein may not be entirely accurate. As in the case of both TN and TP, Blue Plains contributes the highest percentage of PO_4 at 57%. The Mattawoman and Lower Potomac Plants contribute 11% and 10%, respectively.

The percent contribution of the total WWTP flow on a plant-by-plant basis is presented in Figure IV-6. As expected, at 66% Blue Plains contributes the highest flow of all the plants. Alexandria contributes the second highest amount at 9% of the total. Lower Potomac and Arlington contribute 8% and 6%, respectively and the remaining plants contribute 4% or less.

The importance of WWTP flow, and hence load, in the upper Potomac Estuary is depicted in Figure IV-7, which shows WWTP flow as a percentage of total river flow. Although WWTP flow is relatively constant throughout the year, river flow varies seasonally, thus accounting for the changes in the percentages. As expected, during the Spring when river flow is highest, WWTP flow makes up a relatively small percentage of the total river flow. On average, during the month of March WWTP flow comprises the smallest percentage of total river flow at 4% and has one of the smallest ranges (from 1% to 9%). However, during the summer months, when precipitation and therefore river flow have decreased and are more variable, WWTP comprises a greater percentage of the total river flow. During the month of September WWTP flow averages 22% of the total river flow and ranges from 2% to 39%. The link between WWTP flow and population is clearly shown in Figure IV-8. Although the data record is incomplete, the figure shows that population and WWTP flow are highly and directly related.

B. Changes in Loads and Concentrations at the Fall-line

i. Changes in Loads for 1978-1996

Water quality in the upper Potomac estuary is influenced by nutrient inputs from above the fall-line. Fall-line loads represent the sum of point source loads from the WWTPs and industries along the riverine section plus non-point source loads from the surrounding farm fields and forests located above the fall-line. Monitoring water quality at the fall-line is important because 30,562 km^2 (11,800 mi^2) of the 37,995 km^2 (14,670 mi^2) of the drainage basin, roughly 80%, lies above the fall-line.

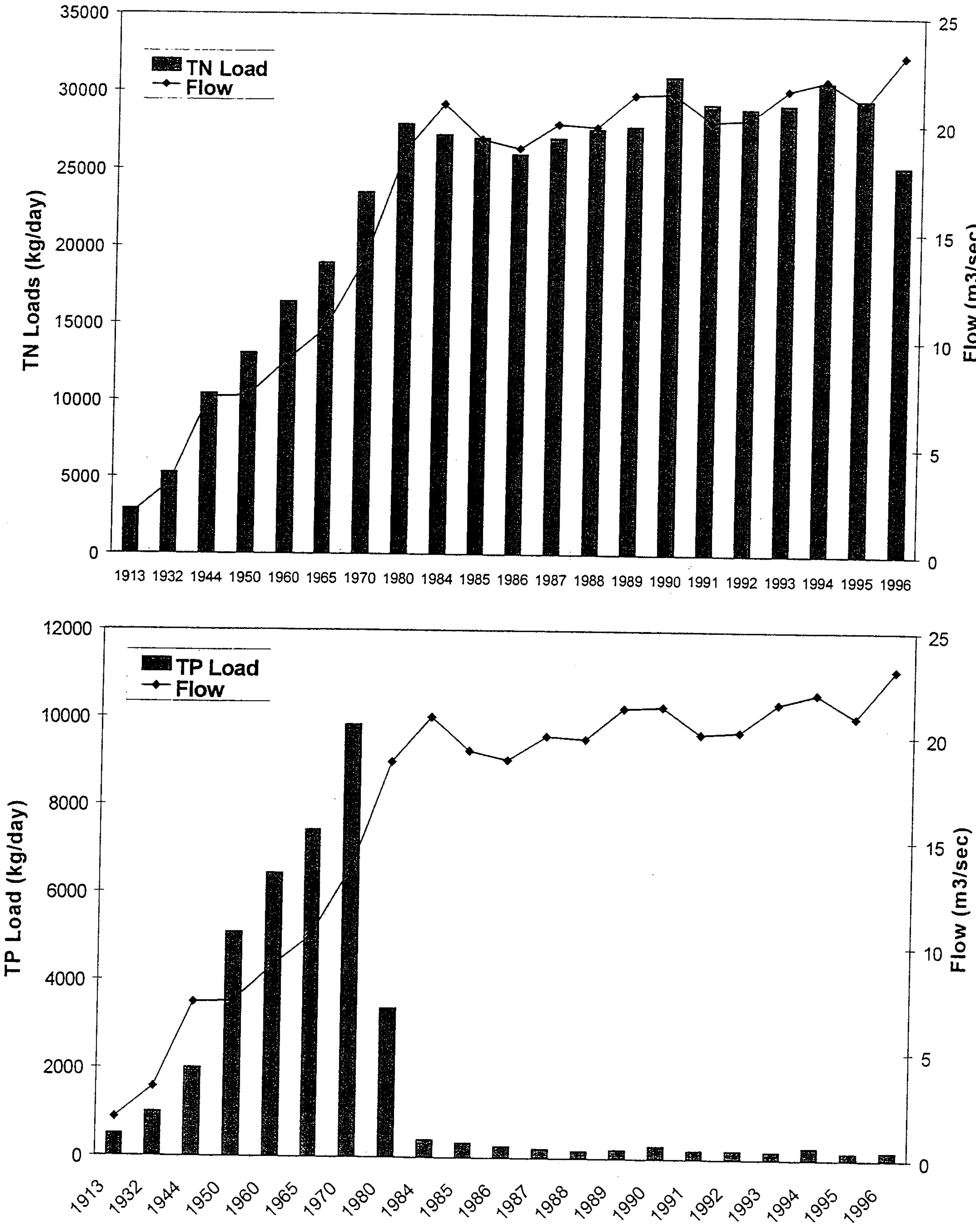
The fall-line monitoring program is conducted by the U.S. Geological Survey (USGS), which began collecting water samples at Chain Bridge in Washington, D.C., in 1978 and the Metropolitan Washington Council of Governments (MWCOG). USGS collects base-flow water samples bi-weekly and collects samples during storm-flow conditions on a seasonal basis (generally three storms per season). The USGS monitoring program emphasizes collecting samples during periods of high flow, since most of the nutrient and sediment load is delivered during storm events. USGS estimates nutrient and sediment loads using a 7-parameter linear regression model. Trends in the concentration data are taken directly from the model coefficients (USGS, 1993b).

Fall-line loadings and flow data for the period 1978 through 1996 are presented in Figure IV-9 for TN and TP, Figure IV-10 for NH_4 and PO_4, and Figure IV-11 for TSS and NO_{23}. Given that fall-line nutrient and suspended solids concentrations are comparatively low and that loads are a function of concentration and flow, it is no surprise that the loads presented in Figures IV-9 through IV-11 closely match the flow data. Unlike the TP loads presented in Figure IV-1 for the WWTPs, TP as well as other fall-line loads, are not distinctly different from flows. That fall-line loads are so similar to flows could be interpreted to as an indication that non-point loads are much more difficult to control than point source loads.

ii. Trends in Concentrations at the Fall-line for 1985-1996

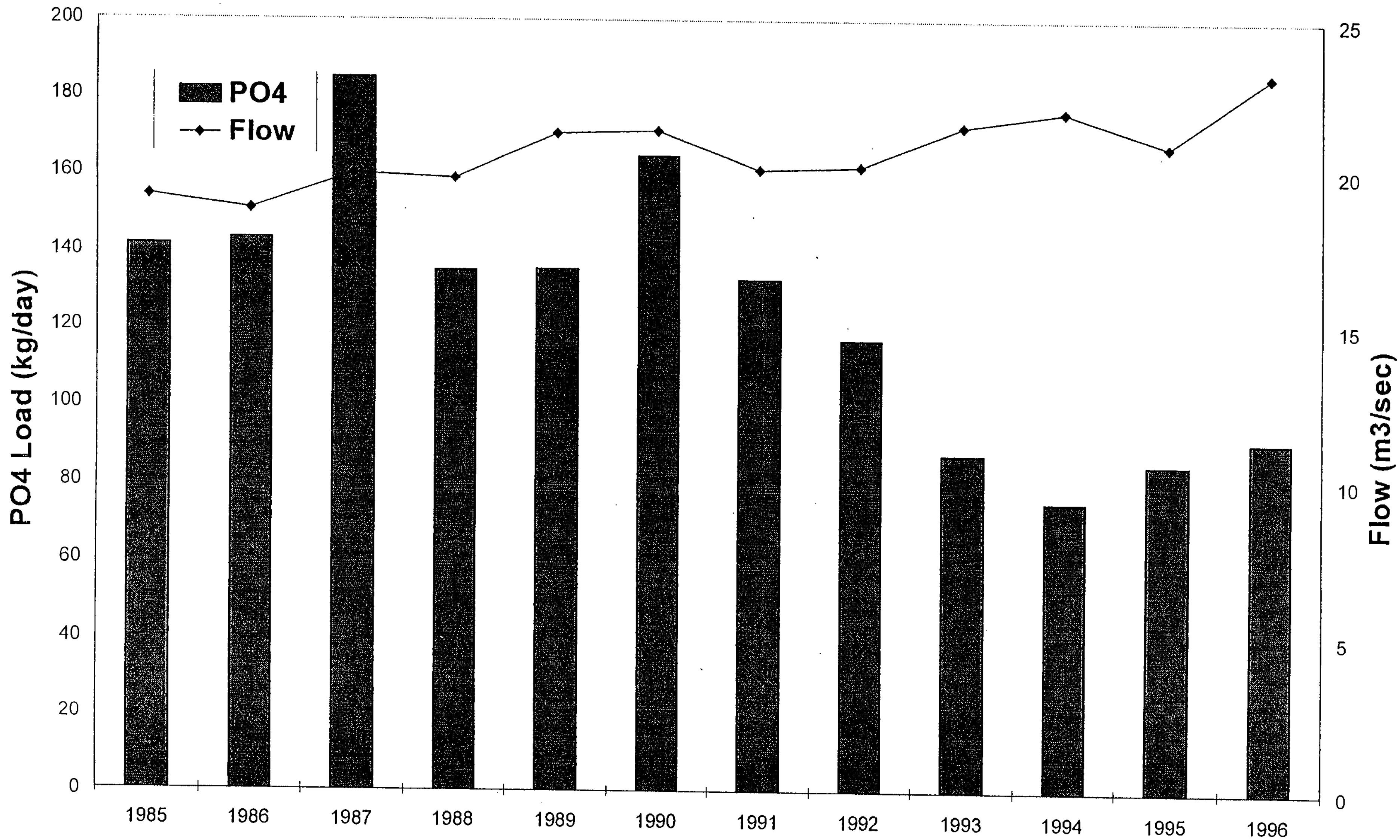
Only flow adjusted trends in concentration are analyzed under the river input program since flow is accounted for in the 7-parameter linear regression model that is used to predict concentrations. Unlike many tidal fresh monitoring stations, the TN trend at the river input station was not statistically significant, despite a 10% decrease in point sources of TN above Chain Bridge. The absence of a significant trend (significant is defined as $p \leq 0.05$) in TN at the river input station may have been the result of conflicting trends in total Kjeldahl nitrogen (TKNW) and NO_{23}, the constituents of TN. Trends in TKNW, which is comprised of NH_4 and organic nitrogen species, decreased significantly by approximately 48%, whereas NO_{23} increased significantly by approximately 71%. The decreases in TKNW may have resulted from nitrification (conversion of NH_4 to NO_{23}) at WWTPs or from the implementation of best management practices (BMPs) above the fall-line, which would have lowered the concentrations of both NH_4 and organic forms of nitrogen. The upward trend in NO_{23} may have been the result of increased NO_3 in discharges from groundwater, atmospheric deposition, and from nitrification at WWTPs, which led to decreases in NH_4.

Flow adjusted trends in TP and PO_4 decreased by 50% and 36%, respectively between 1985 and 1996. The decreasing TP trend coincides with a 33% decrease in TP from point sources above Chain Bridge, which is believed to have resulted from the phosphate detergent ban. Suspended sediment, which is not to be confused with TSS, decreased by 65% (USGS, 1998).

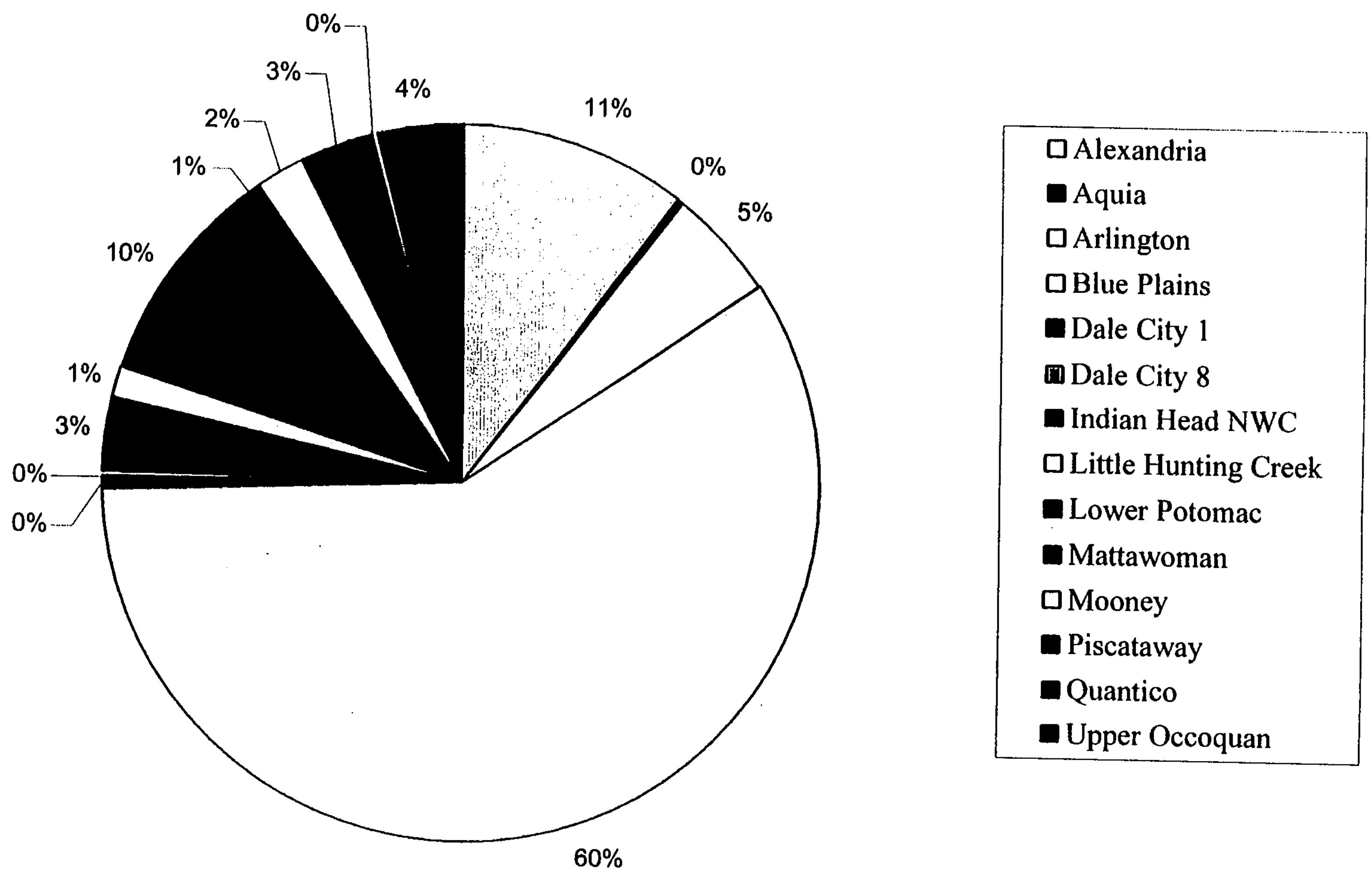


FigureIV-1. Historic total nitrogen and total phosphorus loads for the major wastewater treatment plants (permitted flow of 0.5 MGD or greater) on the Potomac River for 1913-1996. Data for 1913, 1932, and 1944 from Jaworski, 1990. Data for 1950 through 1980 from EPA Chesapeake Bay Program. Data from 1984 through 1996 from Metropolitan Washington Council of Governments.

WWTPs.CDR



FigureIV -2. Recent orthophosphate loads for the major wastewater treatment plants (permitted flow of 0.5 MGD or greater) on the Potomac River for 1985 through 1996. Orthophosphate data are not as complete as those for TN and TP. The data record is complete for Blue Plains, Lower Potomac, Mattowoman Creek, Piscataway Creek, and Upper Occoquan. No data are available for Quantico. Data for the remaining plants are intermittent.



FigureⅣ -3. Percent of average daily total nitrogen load contributed by the major wastewater treatment plants on the Potomac River (1985-1996). Data were obtained from the Metropolitan Washingon Council of Governments.

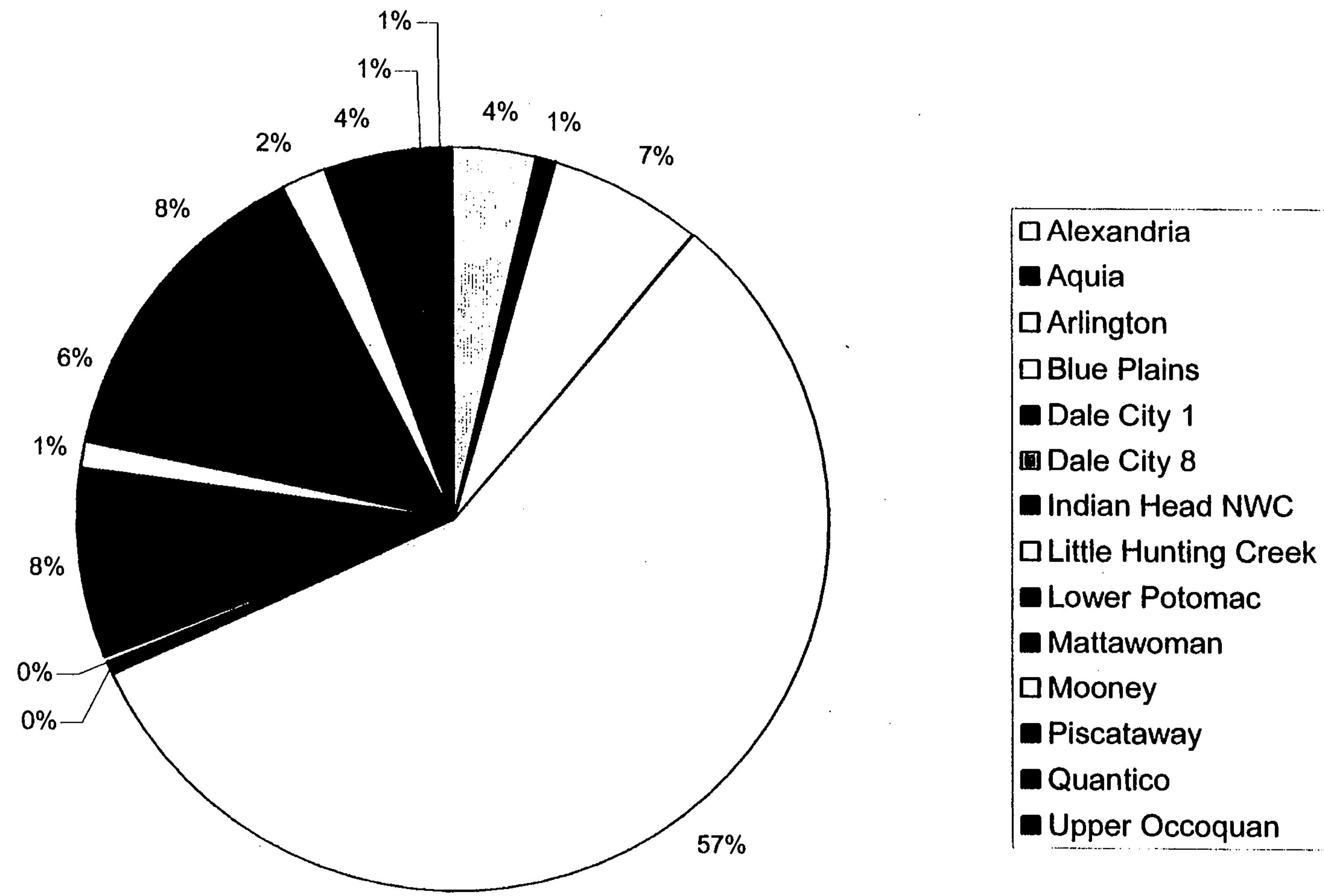
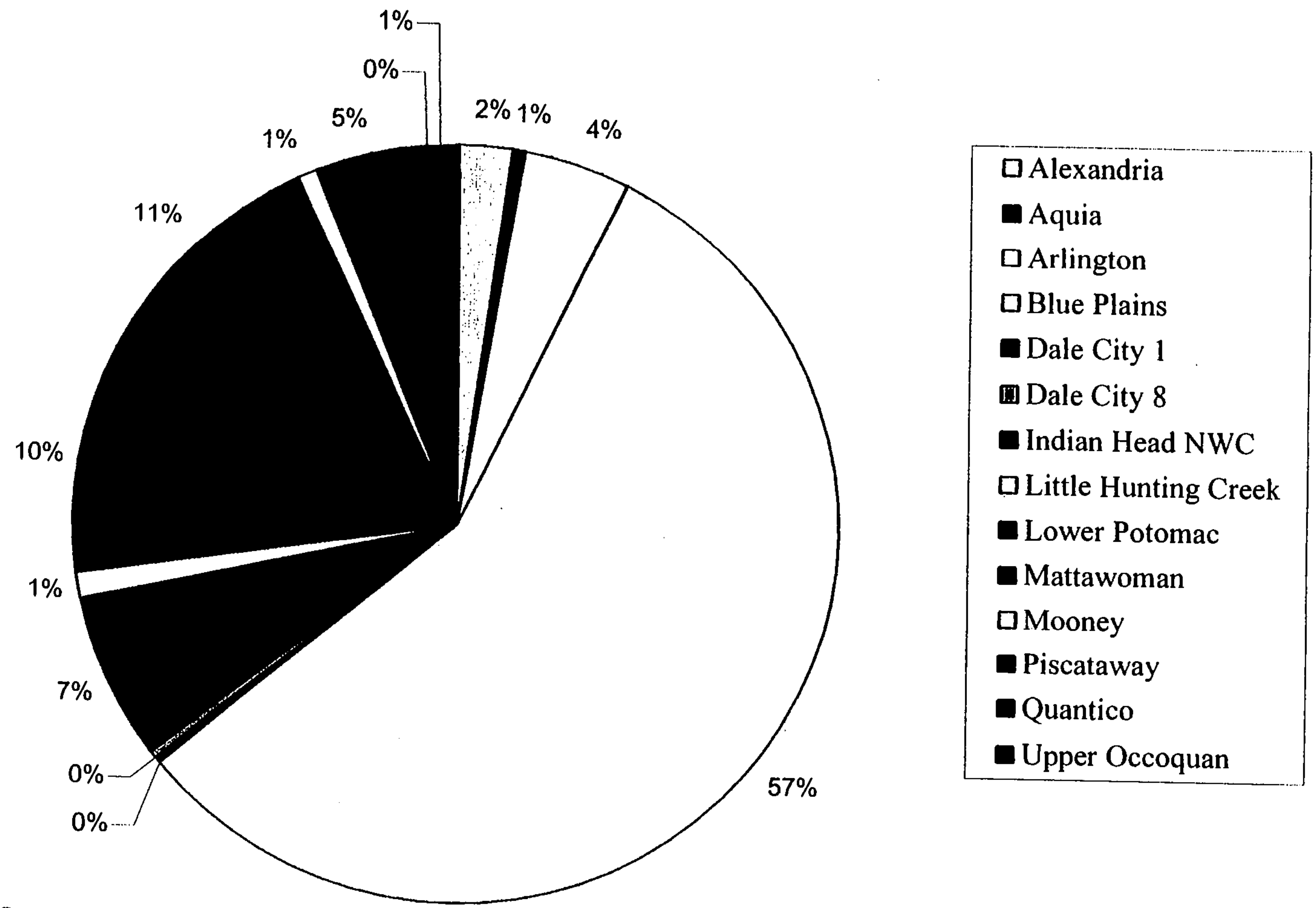
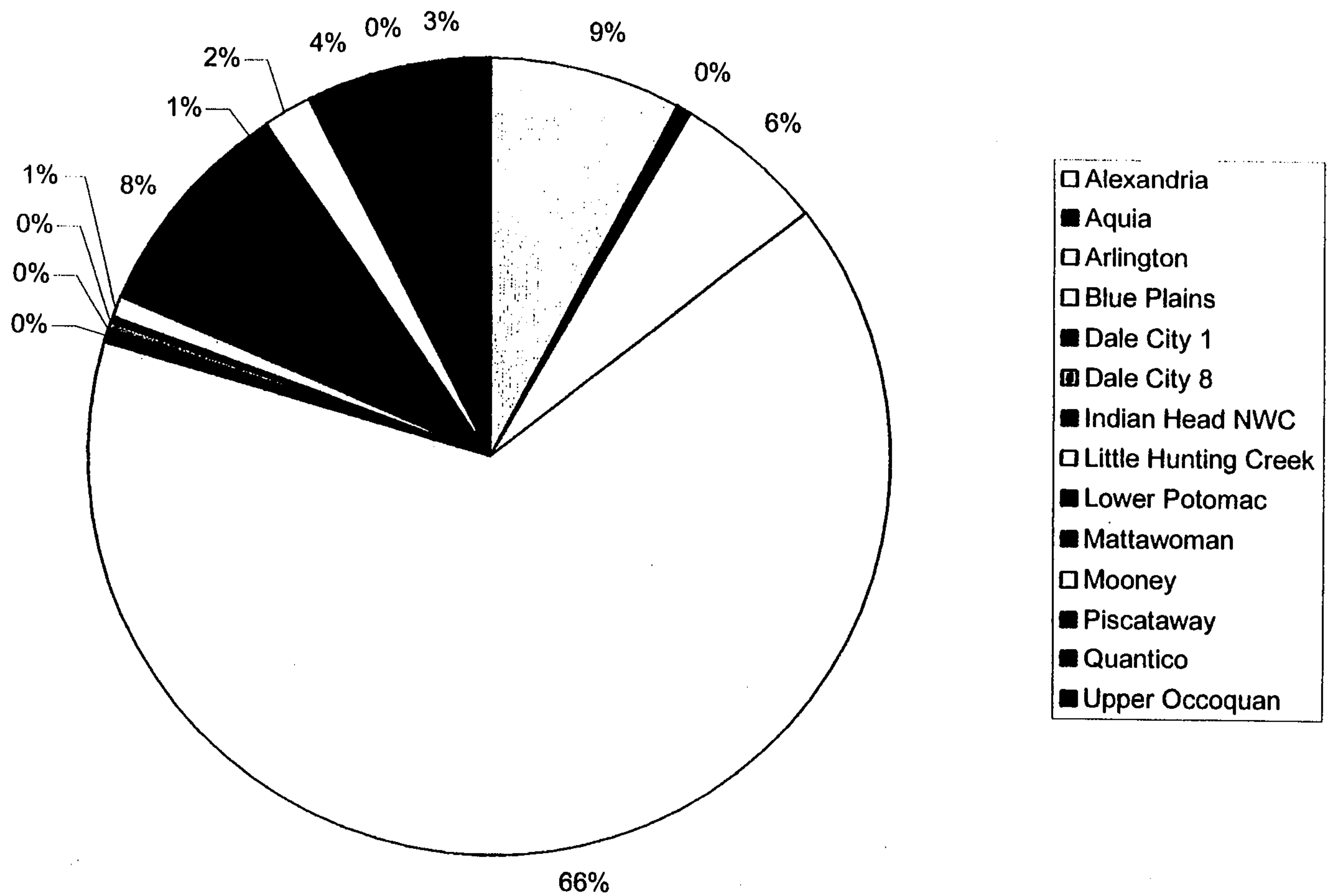


Figure IV-4. Percent of average daily total phosphorus load contributed by the major wastewater treatment plants on the Potomac River (1985-1996). Data were obtained from the Metropolitan Washington Council of Governments.



FigureIV!-5. Percent of average daily orthophosphate load for 1985-1996. Data record is incomplete for most stations. Data record for Blue Plains, Lower Potomac, Matawoman Creek, Piscataway Creek, and Upper Occoquan is complete. No data available for Quantico. Incomplted data records for the remaining plants. Data were obtained from the Metropolitan Washington Council of Governments.



FigureIV.-6. Percent of total wastewater treatment plant flow contributed by major individual plants on the Potomac River (1985-1996). Data were obtained from the Metropolitan Washington Council of Governments.

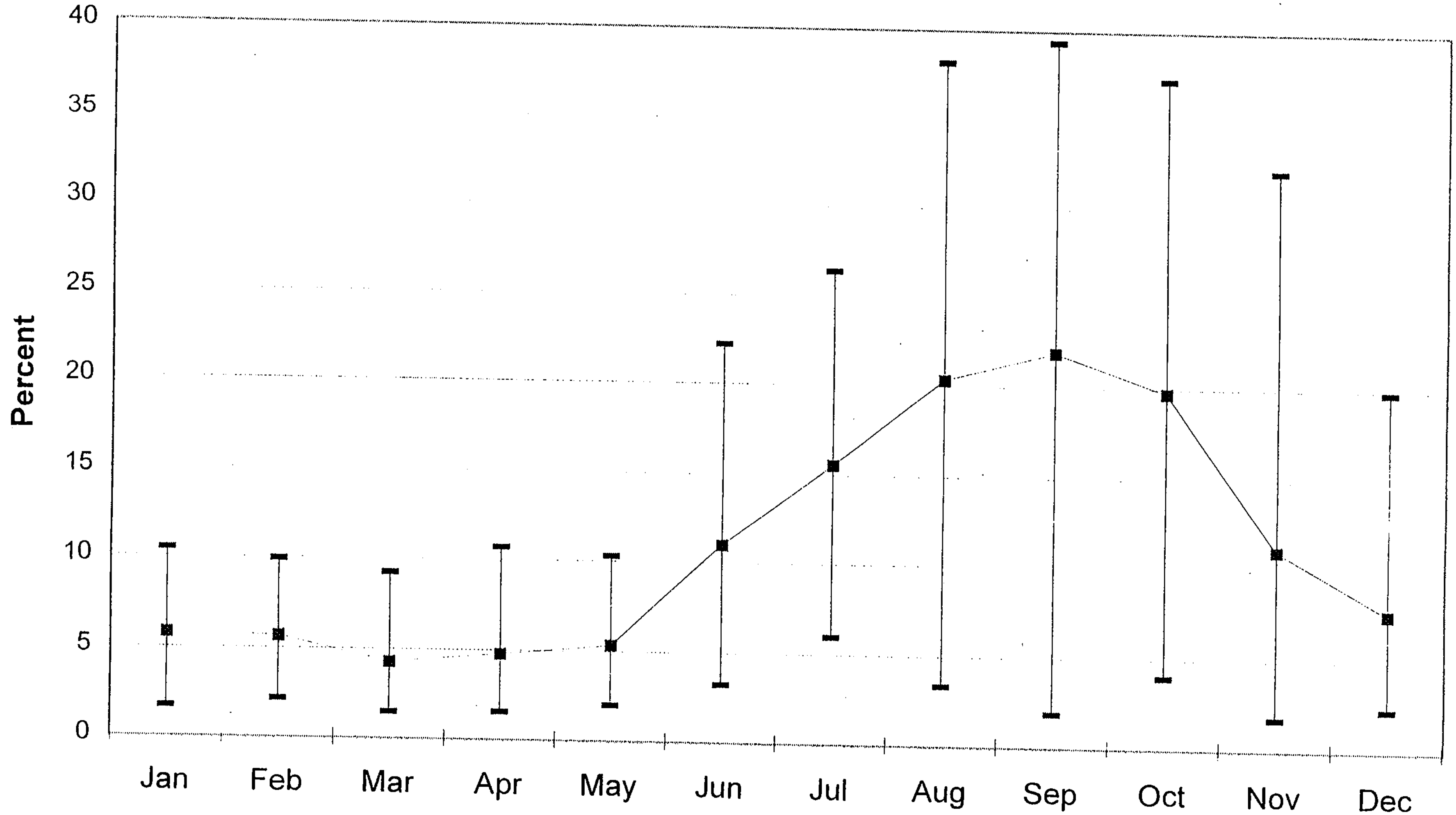


Figure IV-7. Minimum, average, and maximum wastewater treatment plant flow as a percentage of total (river plus treatment plant) flow for 1985-1996. Data were obtained from the Metropolitan Washington Council of Governments.

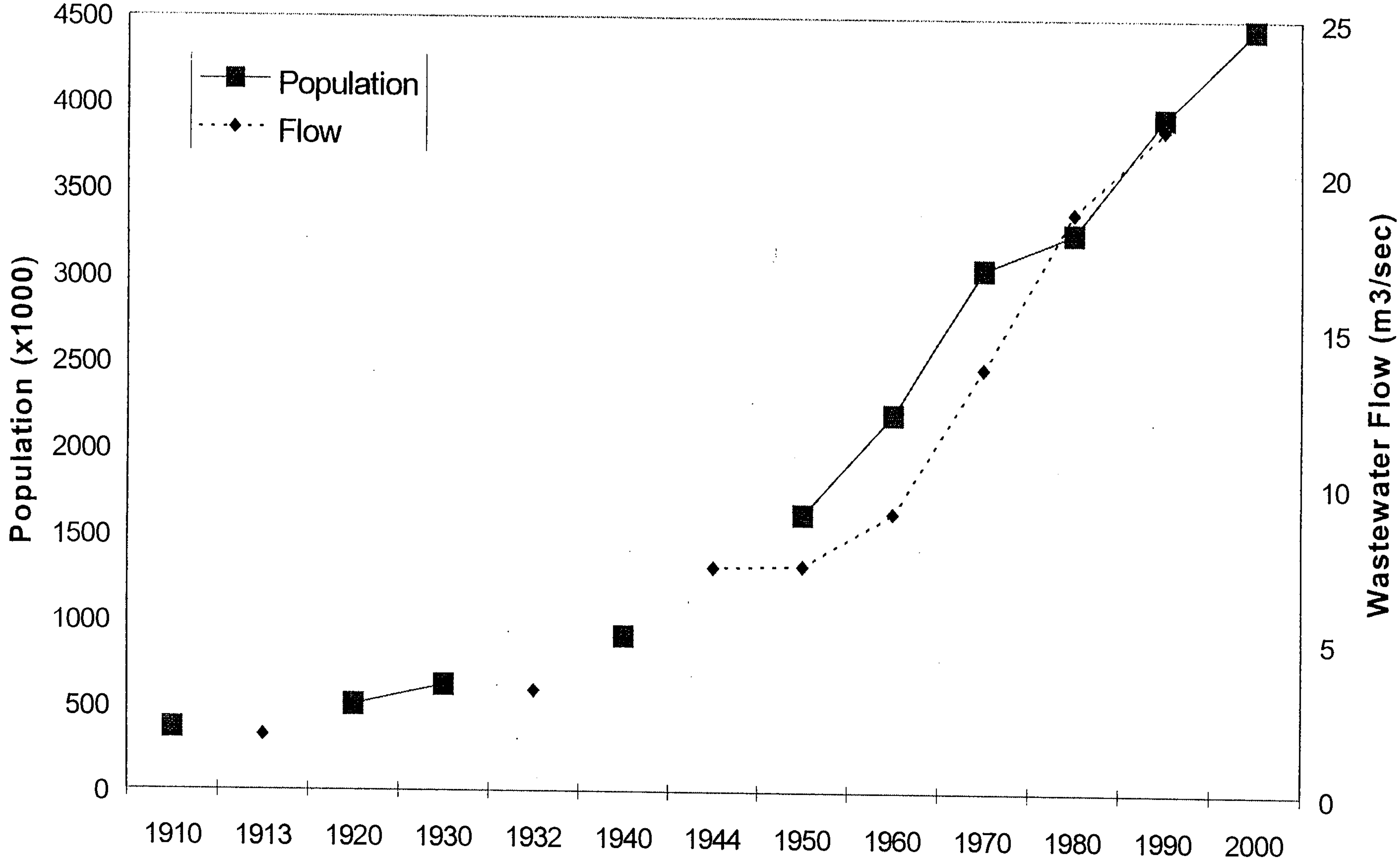
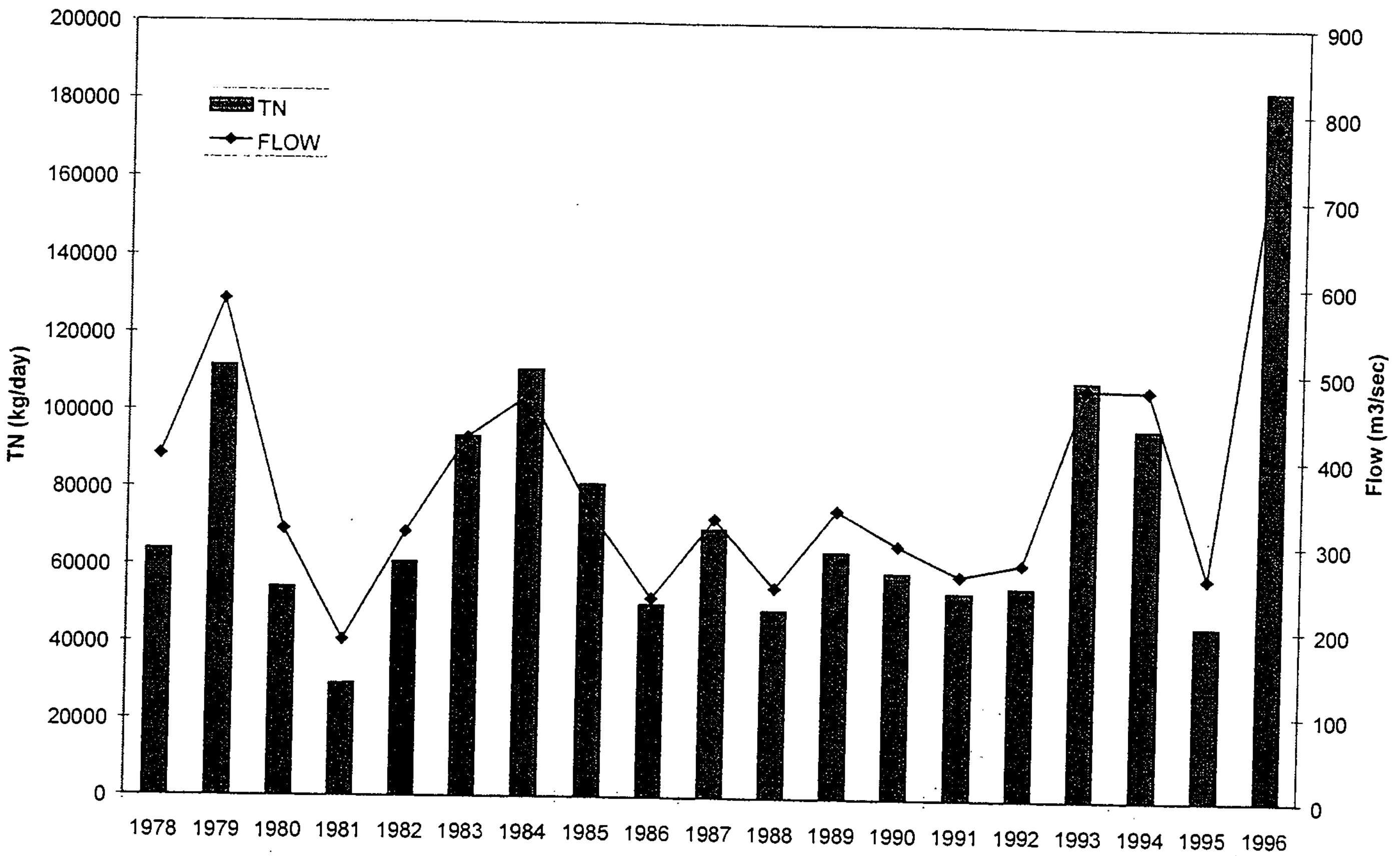
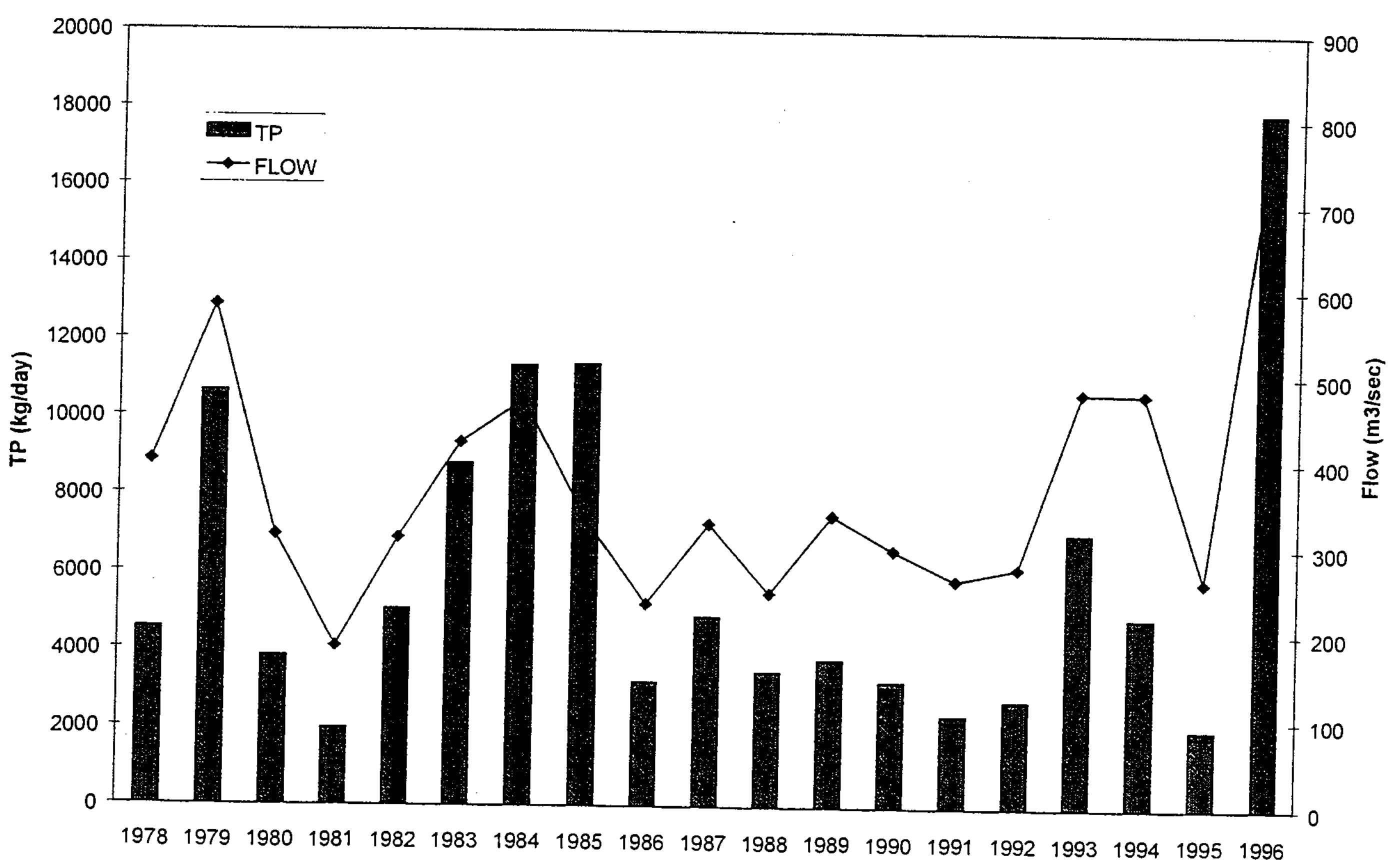


Figure IV -8. Increases in population and wastewater treatment plant flow for the Washington, D.C., Metropolitan area. Population data for 1910 through 1940 are from U.S. Census Bureau, 1998. Population data from 1950 through 2000 are from Metropolitan Washington Council of Governments (year 2000 was projected). Flow data from 1913 through 1980 were from Jaworski, 1990. Flow for 1990 was from Metropolitan Washington Council of Governments.

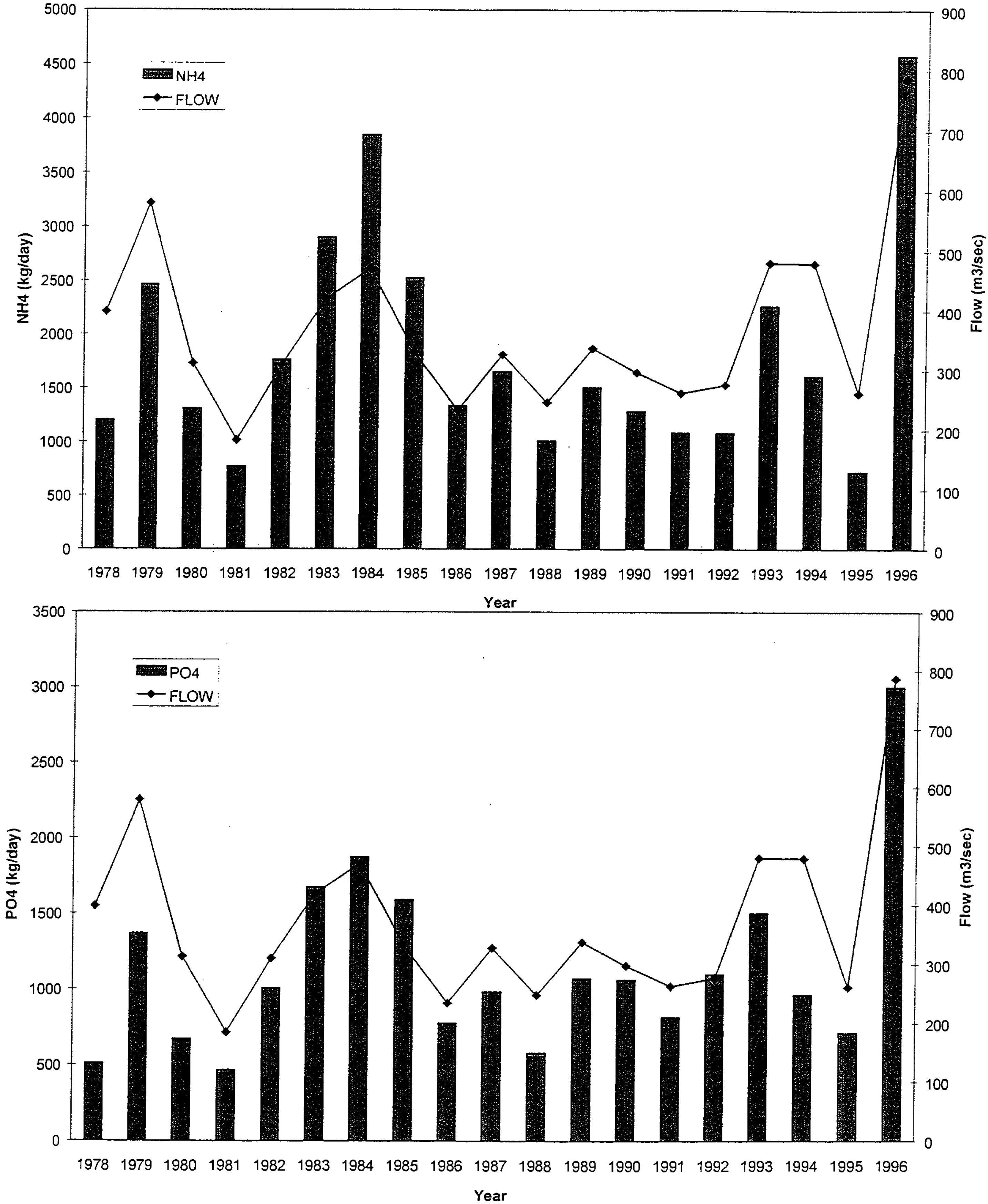
WWTPs.CDR





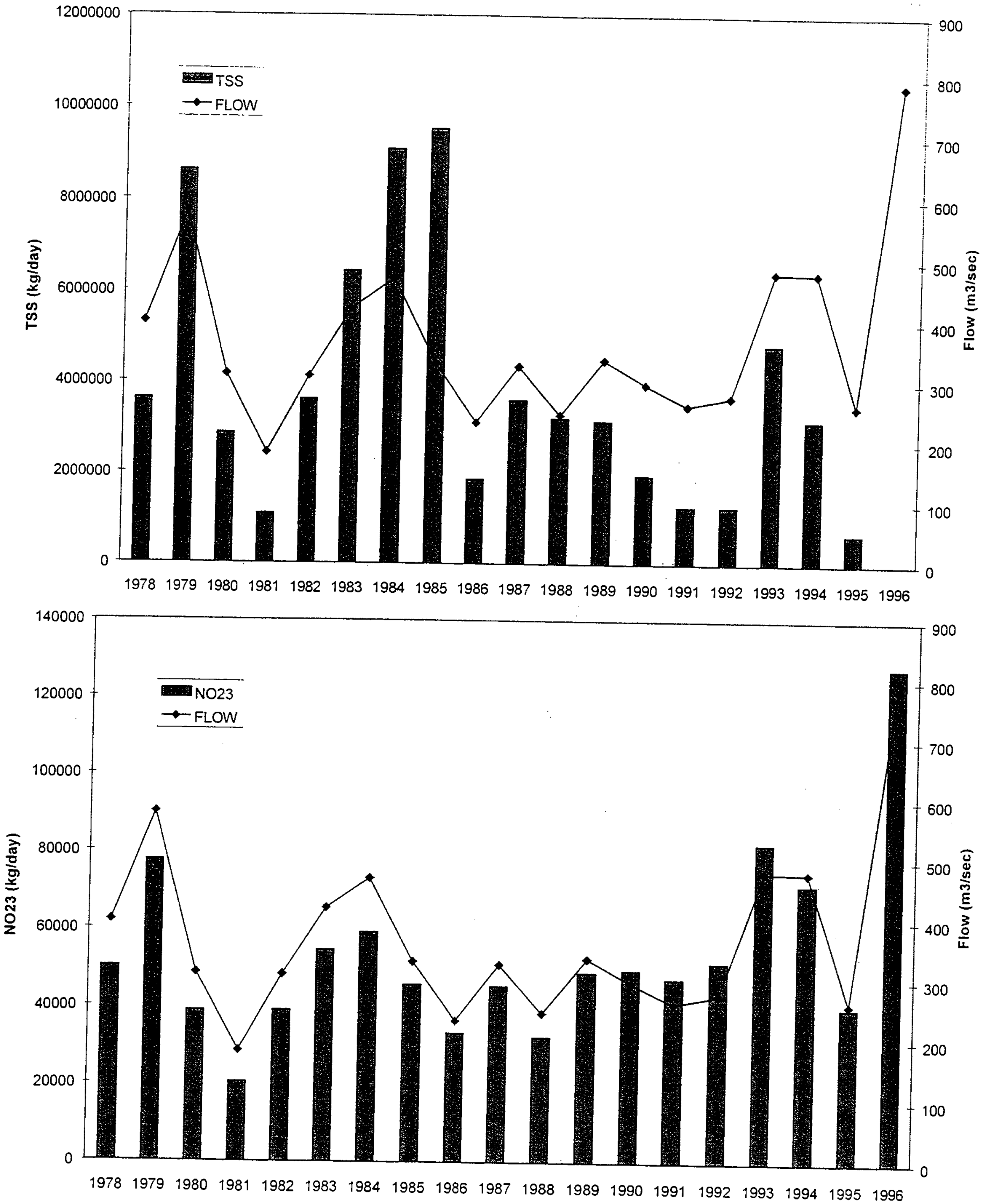
FigureIV -9. Total nitrogen and total phosphorus loads for the fall-line monitoring station at Chain Bridge in Washington, D.C., for 1978-1996. Data obtained from the U.S. Geological Survey.

POTLOAD.CDR



FigureIV -10. Ammonium and orthophosphate loads for the fall-line river input monitoring station at Chain Bridge in Washington, D.C., for 1978-1996. Data obtained from the U.S. Geological Survey.

POTLOAD.CDR



FigureIV -11. Total suspended solids and nitrite plus nitrate loads for the fall-line river input monitoring station at Chain Bridge in Washington, D.C., for 1978-1996. Data obtained from the U.S. Geological Survey. Note that TSS loads may be underestimated since they do not include base flow samples.

POTLOAD.CDR

Table IV-1. Year On-Line, Receiving Stream, and Salinity Zone for the Major Wastewater Treatment Plants Discharging on the Potomac River.

Plant	Year On Line	Receiving Stream	Salinity Zone
Alexandria	1956	Hunting Creek	Tidal Fresh
Aquia	1980	Aquia Creek	Oligohaline
Arlington	1936	Fourmile Run	Tidal Fresh
Blue Plains	1938	Potomac River	Tidal Fresh
Dale City #1	1975	Neabsco Creek	Tidal Fresh
Dale City #8	1980	Neabsco Creek	Tidal Fresh
Indian Head NOS	1978	Mattawoman Creek	Tidal Fresh
Little Hunting Creek	1959	Little Hunting Creek	Tidal Fresh
Lower Potomac	1970	Powhick Creek	Tidal Fresh
Mattawoman	1979	Mattawoman Creek	Tidal Fresh
Mooney	1981	Neabsco Creek	Tidal Fresh
Piscataway	1967	Piscataway Creek	Tidal Fresh
Quantico	1975	Chopawamsic Creek	Oligohaline
Upper Occoquan	1978	Occoquan Creek	Tidal Fresh

V. LINKS BETWEEN TIDAL FRESH WATER QUALITY, THE WWTPs, AND THE FALL-LINE

A. Total Nitrogen

The direction and significance of TN trends depend on the time period of interest. Significant annual TN trends were not detected for either observed or flow adjusted data at XFB2470 for 1985-1996. Decreasing annual trends in observed data of 20% and 13% were detected at XFB1433 and XEA6596, respectively for 1985-1996. Significant decreasing trends in flow adjusted data ranged from 9% to 23% for the lower three tidal fresh stations. TN loads at the WWTPs decreased by 6% between 1985-1996, which may explain the source of the decreasing trends in the tidal fresh concentrations at the ambient monitoring stations. The fall-line TN trend was not significant for 1985-1996.

TN trends at the monitoring stations were significant and positive for observed and flow adjusted data for the mid-1970s-1996 time period. Trends in the observed data ranged from 15% to 38%. Flow adjusted trends ranged from 17% to 38%. WWTP TN loads increased by 8% from the 1970s to 1996. Long-term trends at the fall-line are not available.

TN concentrations for the fall-line, the WWTPs, and the tidal fresh ambient monitoring stations are presented with river flow in Figure V-1. The WWTP concentration data were averaged by year and month across the plants listed in Table V-1. The ambient monitoring station data were similarly averaged across the four stations located in the tidal fresh zone. Although the WWTP data begin in 1983, most of the early data were "estimated" and are clearly higher than the actual concentrations that have been measured since late 1989.

As seen in Figure V-1, TN effluent concentrations from mid-1992 through 1995 are highly variable then drop quite sharply in 1996. The 1996 decrease is presumably a result of the sharp decline in TN concentrations at the Indian Head Naval Warfare Center described in Section VI. Similar variability and a similar decline in the 1996 data can also be observed at the monitoring stations, which suggests that two may be linked. The figure also shows that TN effluent concentrations are roughly 10-fold higher than the those of the river. It would therefore appear that substantial improvements to ambient water quality should result from decreased TN effluent loads.

Fall-line TN concentrations are somewhat similar to the ambient concentrations for most of the record. It is interesting to note the break in the pattern that occurs during the high flow year of 1996. Note that flows were quite high for all of 1996 as were the fall-line concentrations, but that TN for the monitoring stations decreased. This might suggest that TN concentrations in the tidal fresh zone are controlled more by the WWTPs than by non-point sources above the fall-line.

B. Total Phosphorus

Total phosphorus concentrations for the fall-line, the WWTPs, and the tidal fresh stations and river flow are presented in Figure V-2. For 1985-1996, there were no significant trends in TP at any tidal fresh station for either observed or flow adjusted data. This result is difficult to explain because a decrease of 44% was observed in the TP loads from the WWTPs and a 50% decrease in flow adjusted concentrations was detected at the fall-line. With such large decreases from the WWTPs and at the fall-line, one might have expected to see some decreases in the tidal fresh zone stations during the 1985-1996 time period.

Links between the tidal fresh ambient monitoring stations, the fall-line, and the WWTPs are more difficult to make for the longer-term (mid-1970s through 1996) data, because long-term trends are not available for the fall-line, and the WWTP and ambient monitoring data start times do not exactly match those of the monitoring program. Nevertheless, it is interesting to note that significant trends were detected in both observed and flow adjusted data at all four tidal fresh stations for TP. For observed data, the decreasing trends ranged from 31% at XFB2470 to 39% at XEA1840. For the flow adjusted data, the decreasing trends ranged from 27% at XFB2470 to 39% at XEA1840. Significant decreasing trends in TP in the tidal fresh zone may be explained by the large decreases in historical TP loads from the WWTPs, which declined by approximately 98% from the 1970s to the present.

C. Orthophosphate

PO_4 concentration and flow data are presented in Figure V-3. Trends in PO_4 are difficult to analyze in the Potomac River because of the 1990 change in methodology. The only significant PO_4 trend in the tidal fresh zone for 1985-1990 was a 32% decrease in flow adjusted data at XEA6596. For 1985-1996, WWTP PO_4 loads decreased by 36%. Flow adjusted PO_4 concentrations at the fall-line also decreased by 36%. Perhaps more significant trends would have been detected at the tidal fresh monitoring stations if the entire record could have been used, instead of only through September 1990.

Unfortunately, long term WWTP loadings and fall-line concentration data for PO_4 are not available. As a result, it is not possible to directly explain the dramatic decreases in PO_4 that were observed at the tidal fresh zone monitoring stations between 1970 and 1990. For 1970-1990, significant trends in observed and flow adjusted data were detected at all four tidal fresh stations. Decreases in observed data ranged from 49% at XEA1840 to 66% at XFB2470, while decreases in flow adjusted data ranged from 40% at XEA1840 to 58% at XFB2470.

Although it cannot be demonstrated statistically, it is difficult to believe that the management activities from the mid-1970s through the phosphate detergent ban in the mid-1980s to implementation of tertiary treatment in the 1990s have not had a direct effect on improving PO_4 trends in the Potomac River.

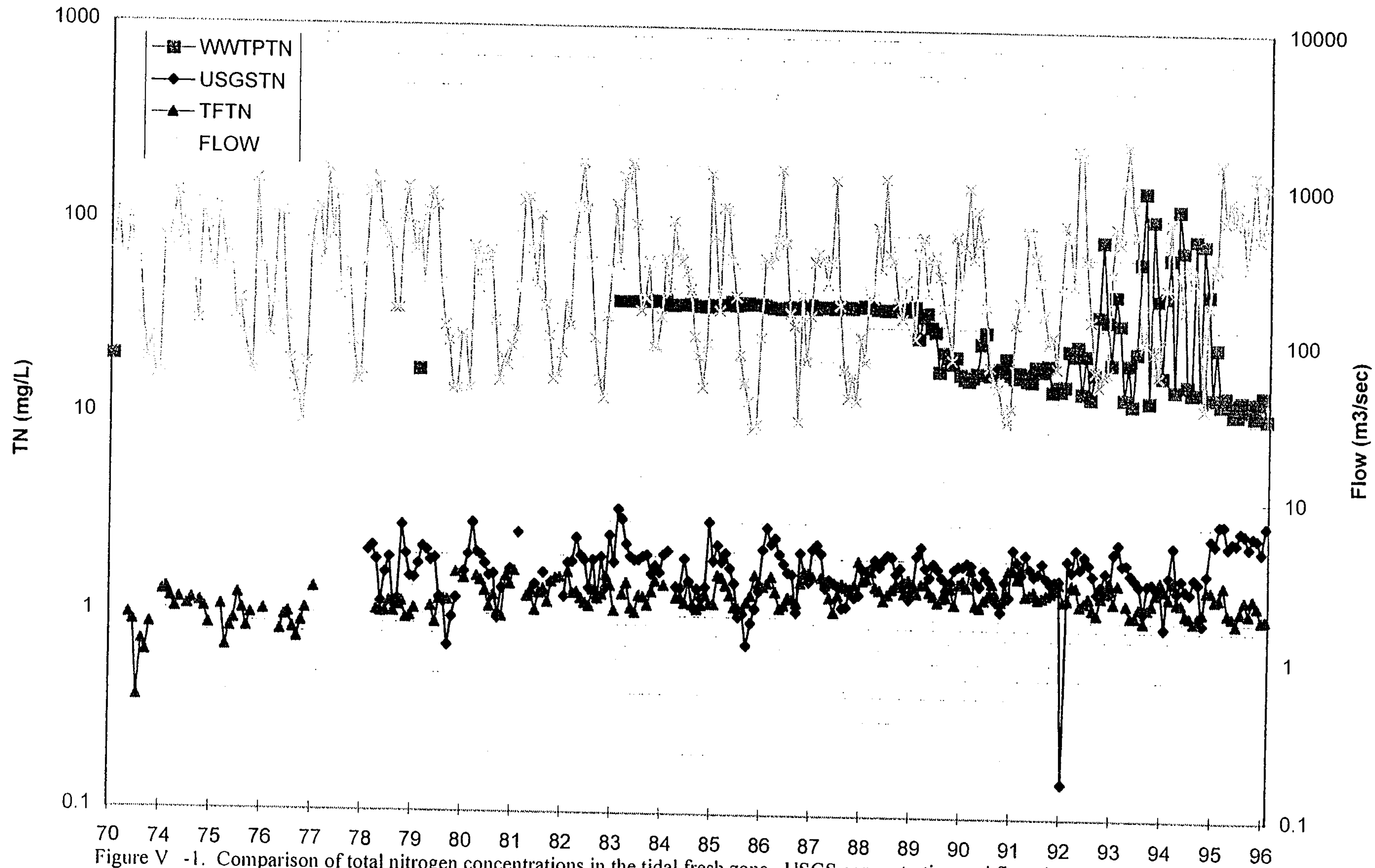


Figure V -1. Comparison of total nitrogen concentrations in the tidal fresh zone. USGS concentration and flow data were measured at Chain Bridge. WWTP data are monthly average concentrations for the major plants. Tidal fresh TN concentrations were averaged across the four tidal fresh zone stations.

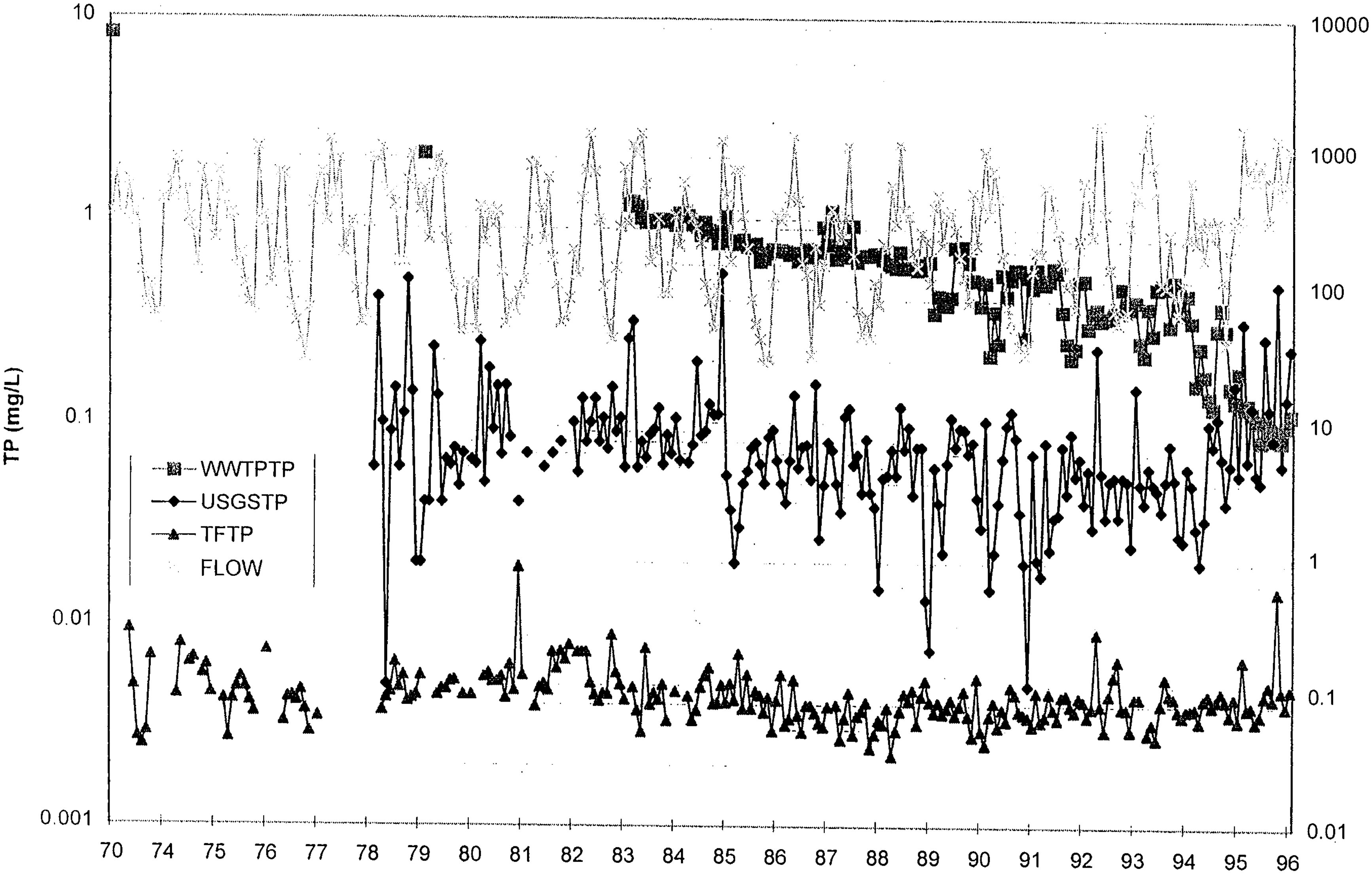


Figure V''-2. Comparison of total phosphorus concentrations in the tidal fresh zone. USGS concentration and flow data were measured at Chain Bridge. WWTP data are monthly average concentrations for the major plants. Tidal fresh TP concentrations were averaged across the four tidal fresh zone stations.

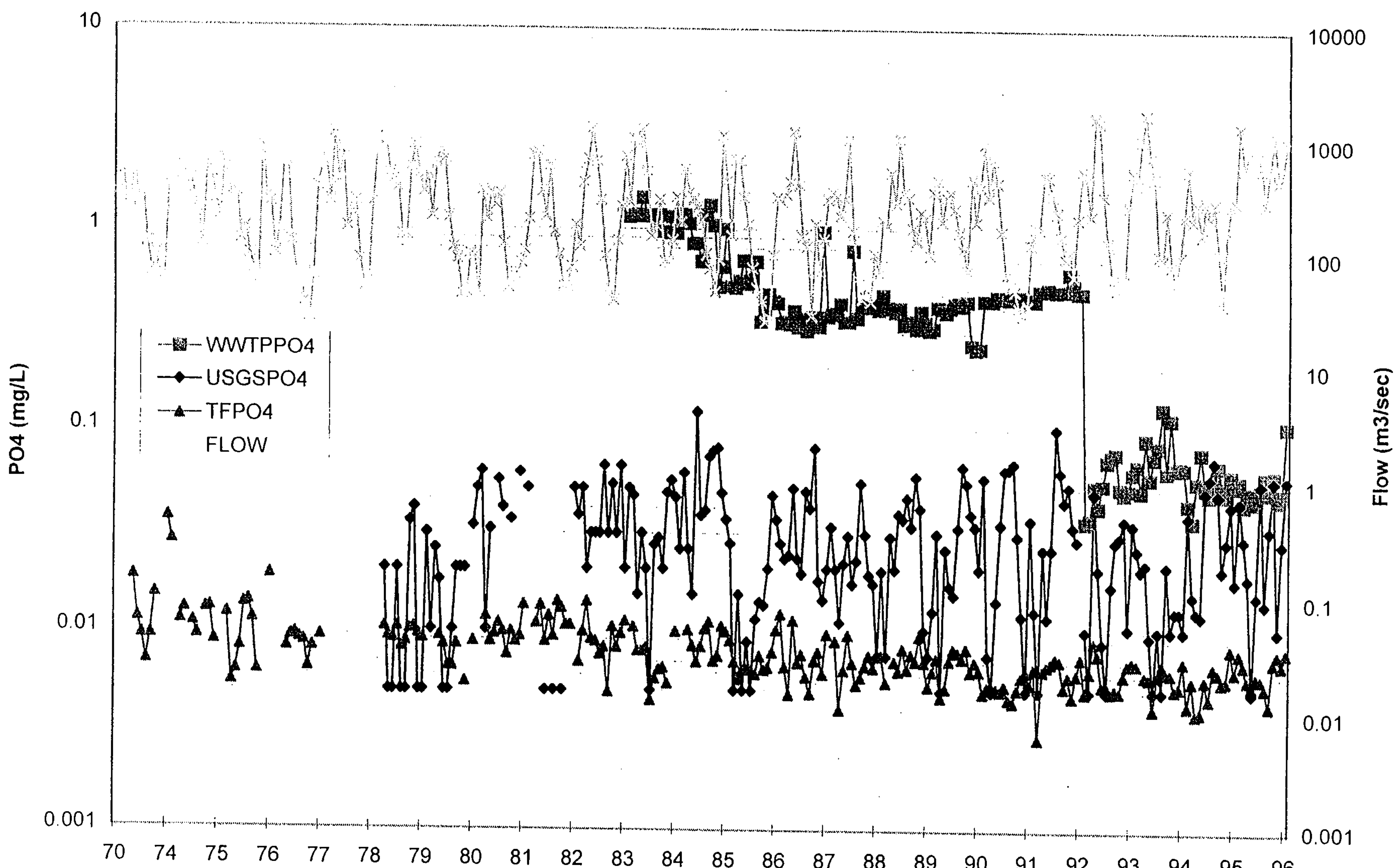


Figure VI-3. Comparison of orthophosphate concentrations in the tidal fresh zone. USGS concentrations and flow data were measured at Chain Bridge. WWTP data are monthly average concentrations for the major plants. Tidal fresh PO4 concentrations were averaged across the tidal fresh zone stations.

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APPENDIX B

A HISTORICAL ANALYSIS OF THE EUTROPHICATION OF THE POTOMAC ESTUARY

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A Historical Analysis of the Eutrophication of the Potomac Estuary

By Norbert A. Jaworski and William D. Romano

Introduction

The geology, physiography, and hydrology of the Potomac River Basin have been well-characterized (6), (8), (16), (21). Numerous studies (3), (9), (18) have been conducted on the water quality problems of the Potomac Estuary as part of enforcement programs in the 1950s and 1960s and then as part of the Chesapeake Bay Program. A retrospective study of the water quality issues of the Upper Potomac Estuary was published in 1990 (10). Major water quality assessments and nutrient loading synthesis studies have been recently completed for the entire Chesapeake Bay in 1995 (15) and for the Potomac River Basin in 1996 (2) and 1998 (1).

The Potomac Integrative Analysis Project was recently undertaken to focus on the potential water quality and living resources responses to a 40% reduction in nitrogen and phosphorus in the Potomac Estuary. The project uses as its analysis period the 1985-1996 time frame. The principal goal of this paper is to provide a broader perspective by undertaking a longer historical analysis of the nutrient management issues needed to understand the effects of reduced nutrient load on the eutrophication of the Potomac Estuary. The longer historical analysis period contains the low stream flows of the mid-1960s and the high stream flows of 1996, thus providing the authors a unique opportunity to quantify what changes have occurred over a very wide range of stream flow conditions and enhanced wastewater treatment.

Specific objectives of this paper:

(a) For the period 1900-1995, describe the changes in: (1) the annual landscape nutrient loadings from animal production, crop production, air deposition and domestic wastewater to the upper basin, (2) the water quality, average annual concentrations and riverine export fluxes of the Potomac River above Washington, DC, and (3) the annual nutrient loadings from domestic wastewater directly discharged to the upper estuary and,

(b) For the period 1965-1996, present: (1) the combined (upper basin and estuarine wastewater) monthly nutrient loadings and describe how the control measures affected the loadings, (2) the results of trend analyses in nutrient concentrations of the estuary, and (3) the sampling data to show the consequences of eutrophication i.e., low Dissolved Oxygen (DO) in the bottom waters, elevated levels of chlorophyll and of Total Organic Carbon (TOC), and reduced light penetration (Secchi disk). Finally, conduct an analysis of how well the ambient estuarine nutrient concentrations can be predicted from the combined loading estimates.

Data Sources and Analysis Methods

Data were obtained from numerous sources as presented in Appendix A. All the historical data have been compiled on computerized spreadsheets to facilitate the analysis and to serve as documentation of all data and analysis methods.

The US Department of Agriculture's methods for estimating nutrients available from animal production (23) and used in crop production (14) along with other nutrient conversion data (19), (24) were adopted to determine annual nutrient loadings to the upper basin landscape from agricultural practices. The recent nitrogen landscape loadings from atmospheric deposition were from the National Atmospheric Deposition Program's electronic data base at Colorado State University, Fort Collins, CO. The historic nitrogen loadings were estimated from historical nitrogen emission data (12). In that current wastewater loadings are known, the historical annual loadings from domestic wastewater discharges for the upper basin were assumed to be proportional to the historical population.

The major source of historical water quality data for the Potomac River above Washington, DC, is from the drinking water supply monitoring program of the US Army Corps of Engineers' (USCOE) Delecarlia Treatment Plant. Except for a few years in the 1920s when the data were lost, monthly summaries of weekly chemical analyses of the "raw water" from the Potomac River above Great Falls have been compiled and annual summaries calculated.

In response to continuing degradation of the water quality of the upper estuary, the first Potomac Washington Area Enforcement Conference was convened in 1957 by the US Department of Health, Education, and Welfare (10). In addition to high fecal coliform counts and low dissolved oxygen levels, the upper estuary was highly eutrophic. As a result of the enforcement programs, nutrient water quality monitoring of the Upper Potomac River Basin was established in 1966 by the Federal Water Pollution Control Agency (FWPCA) (8) and in 1969 by the Environmental Protection Agency (EPA) (4). The upgrading of the wastewater treatment facilities to include phosphorus removal and nitrification was a result of the enforcement programs. The District of Columbia's upper estuary monitoring was expanded to include nutrients and annual reports were published by the Metropolitan Washington Council of Governments (MWCOG) (18). A major current water quality monitoring program is a cooperative effort between MWCOG, the US Geological Survey and the State of Maryland's Department of Natural Resources (20).

The historical annual export fluxes of nitrate nitrogen and other chemicals from the upper basin in kilograms per square kilometer per year ($kg/km^2/yr$) have been compiled based on the USCOE's monthly chemical analyses and USGS' river discharge measurements. For the period 1965 to 1977, estimates of monthly nutrient export fluxes were based on predictive models derived using regression analysis, which describe nutrient fluxes-river discharge logarithmic relationships by FWPCA (8) and EPA (4). Using daily USGS river discharge measurements (J.M. Landwehr, personnel communication) and regression relationships, monthly summaries were compiled for the 1964-1977 period. The nutrient export fluxes from 1978 to the present were provided by the USGS as part of the Chesapeake Bay Program. The export fluxes were based on the USGS' ESTIMATOR program which is a statistical load estimation technique (15).

The historical nutrient loadings from wastewater treatment plants discharging directly into the estuary have been compiled from numerous sources (4), (8), (18). For the period 1985-1996, monthly wastewater effluent flow and loadings data were obtained from MWCOG and the Maryland Department of the Environment.

Data for six water quality monitoring stations, which where sampled by various state and federal agency programs since 1965 (See Appendix A), have been compiled for analysis. The six stations and their distances from confluence with Chesapeake Bay are:

Station	Distance
Piscataway	158 kilometers
Indian Head	139 kilometers
Maryland Point	103 kilometer
301 Bridge	64 kilometers
Ragged Point	19 kilometers
Point Lookout	At confluence

Surface water quality data were averaged for each month in which data were obtained except for DO which included both surface and bottom waters. In that the early eutrophication problems addressed were in the upper estuary, the upper three stations have more months which have data than the three stations in the lower estuary. The period of 1978 to 1996 has the most complete water quality data record throughout the 12 months of a given year. Although data from the various monitoring programs have been merged into one spreadsheet for the six stations for +31 years of data, it remains possible to identify the original data sources. The statistical trends were analyzed using the Seasonal Kendall test (7). In merging data from various sources, the authors recognize that there can be differences in analytical methods, especial in the detection limits. The method used by various agencies is being documented.

Changes in the Upper Basin Landscape 1900-1996

Since the turn of the century, the population density of the upper basin has about tripled to a current density of 16.6 people/km^2. During that same period, major agricultural changes have occurred in the upper basin centered around the production of meat (beef, poultry, and pork), eggs, and milk. An example of these changes is in the cattle densities for the states of Maryland (MD) and Virginia (VA). As shown below, the cattle densities (cattle/km^2) for VA have increased dramatically, with beef production resulting in the highest density in 1996.

	VA			MD		
	1900	1955	1996	1900	1955	1996
Dairy	2.6	4.4	1.3	5.8	10.0	4.0
Beef	4.6	8.6	14.7	4.2	11.0	8.0
Total	7.2	13.0	16.0	10.0	21.0	12.0

For MD, both beef and dairy cattle densities were the highest in the 1950s. While the number of dairy cattle have decreased since the 1950s, milk production has gone from about 4,000 lb/yr/cow (22) at the turn of the century to current production levels of approximately 16,000 lb/yr/cow. In that milk production is directly related to animal food requirements, corn silage became a major crop for milk production. To produce the silage corn, large quantities of plant fertilizer were applied at increasing rates following World War II. Since the 1970s, nitrogen fertilizer application rates have leveled off at +2000 $kg/km^2/yr$ for MD and +1000 $kg/km^2/yr$ for VA.

Significant increases in poultry production have occurred, initially in the Shenandoah sub-basin and more recently in the Capapon and South Branch Potomac sub-basins. Even with increased animal food crop production, it has become necessary to import animal feed from the mid-west for meat, eggs, and milk production. It has been estimated that about 900 $kg/km^2/yr$ of TN is imported into the Mid-Atlantic and Northeast USA (13). The nutrients in manure from these animals often exceed the crop nutrient requirements (14).

Since the 1900s, the NOx emissions from combustion sources for the states in which the Potomac Basin lies, have increased from about 500 $kg/km^2/yr$ to a maximum of 2500 $kg/km^2/yr$ in 1970s and have since decreased to about 2100 $kg/km^2/yr$. The current estimate is that about 1000 $kg/km^2/yr$ of nitrogen is deposited onto the basin from wet and dry atmospheric deposition.

The landscape loadings from the above mentioned sources for all three major terrestrial nutrients i.e., nitrogen, phosphorus and potassium, have significantly increased since the turn of the century (Figure 1). Nitrogen loadings have increase by a factor +5.0, from 800 to 4000 $kg/km^2/yr$. Likewise, potassium has increased by a factor of +3.3 and phosphorus, +7.3. For the period 1965-1996, the TN landscape loadings have increased by approximately 37%.

Impact on Riverine Water Quality and Export Fluxes

The impacts of man's activities in the upper basin on the water quality above DC have been dramatic (Figure 2). The chloride and nitrate riverine concentrations have increased about 10 fold. The total residue, which is a measure of dissolved and particulate matter, has doubled. The river water, which is the major source of drinking water for the DC area, has also become harder and more alkaline.

The amount of nutrients, chlorides, and sulfates, in flux units of $kg/km^2/yr$ that were exported from the upper basin to the upper estuary, are presented in Figure 3. The nitrate export has increased from about 100 $kg/km^2/yr$ in the 1900s to over 500 $kg/km^2/yr$ in the 1990s. Potassium and chlorides export fluxes have increased by a factor of 10. Since the 1920s, sulfate export fluxes, for which acid rain is a significant source, doubled by the 1970s and later began to decrease, primarily in response to a reduction in atmospheric deposition of SOx. The phosphorus export fluxes peaked about 1975-1985 and began to decrease in response to reductions in fertilizer use and the implementation of the various phosphate detergent bans (17). The large TP export fluxes in 1996 were a result of high stream flows.

Upper Potomac Estuary Wastewater Historical Loadings

Most of the domestic wastewater from the 4.6 million inhabitants of the DC Metropolitan Area receives advanced treatment from numerous facilities in DC, VA, and MD, and is subsequently discharged into the fresh water portion of the upper estuary. While the first sewers were built in the 1870s, there was no wastewater treatment until 1930s, when the Blue Plains plant started to provide primary treatment. The historical loading data (Table 1) show that the flow volume and nitrogen loading have increased over 10 fold since 1913. The TOC and TP loadings reached a maximum in the 1970s and have been reduced by about 90% and 98%, respectively, by the implementation of advanced wastewater treatment. While the advanced wastewater treatment plants included nitrification, which converts the ammonia to nitrates, only a small amount of the total nitrogen was removed from the wastewater discharges. In October 1996, the Blue Plains plant added the denitrification process and began to remove nitrogen.

Total Nutrient Loadings to Upper Estuary for the period 1965-1996

The contributions from the upper basin and from the upper estuary wastewater treatment plants are presented for TN, TP, and TOC in Figures 4, 5, and 6, respectively. Currently about 50% of the TN is from the upper basin and the remaining 50% from the wastewater treatment plants. With the application of advanced wastewater treatment technology, the 1990s TOC and TP loadings from the estuary discharges were 5% and 8%, respectively of the loadings from the upper basin. Prior to 1972, about two-thirds of the TP load was from wastewater discharges in the DC area. The low loadings observed in 1966, 1969, 1981, 1992 and 1995 occurred during drought conditions whereas the high loadings observed in 1973, 1980, 1985, 1994 and 1996 occurred during high stream flow periods.

The TN, TP, and TOC monthly loadings in kg/month from both the upper basin and upper estuary wastewater discharges for the first and last 6-year periods of the 1965-1996 time frame are summarized below:

	TN	TP	TOC
1965-1970	1,899,200	414,300	4,654,800
1991-1996	3,681,400	181,700	6,016,400
% Change	+94%	-56%	+29%

Excluding the loadings for 1996, which was a very high flow year, the loading changes were +68%, -70%, and +20% for TN, TP, and TOC respectively.

The TN, TP, and TOC "Point-of Entry" concentrations were estimated using the combined upper estuary wastewater and the upper basin export loadings to assess the effects on the biological communities (Figure 7). Given that the upper estuary has a short retention time, three month moving-average Point-of-Entry concentrations were calculated using the combined wastewater and fall-line loadings and the combined wastewater flow and river discharge data.

Over the +31 year period, the estimated upper estuarine ambient TP Point-of-Entry concentrations (Figure 7) went from an average of about 0.943 mg/l in the 1960s (dry period) to about 0.116 mg/l in the 1990s (wet period) for a reduction of 88%. The estimated ambient TOC levels were reduced by about 40%. During the +31 year period, the estimated ambient TN Point-of-Entry concentrations increased by about 12%.

Estuarine Nutrient Concentration Trends 1965-1996

The water quality trends for all six stations for TN, dissolved inorganic nitrogen (DIN), TP, TOC, chlorophyll, bottom DO and Secchi disk were analyzed using the Seasonal Kendall test and are presented in Table 2. Inorganic phosphorus was not included because there were significant chemical analysis changes in 1990, which precluded examining the entire record.

In Figures 8 and 9, surface TP and TN concentration data for the period 1965-1996 are presented for three Potomac Estuary sampling stations: Indian Head, 301 Bridge and Point Lookout. At Indian Head, the TP levels have dropped from the late 1960s concentrations of 0.2-0.4 mg/l to 1990s levels of 0.08 to 0.1 mg/l (Figure 8). Statistically significant reductions occurred at the upper five stations. It is not clear if there has been a significant reduction at Point Lookout in that the early data reflects mainly low river discharge conditions and that station may be more influenced by the ambient levels in the Chesapeake Bay. The increased levels of TP in the early 1980s at all three stations were not due to the increased wastewater or riverine input. In fact, the two loadings decreased during this period (Table 1 and Figure 6). The increases in concentrations were from the sediments mainly in upper and middle reaches due to a loss in buffering capacity(9).

Statistically significant increasing trends in TN were detected at all stations except Piscataway Creek and Point Lookout (Table 2); trends at Piscataway and Point Lookout were not statistically significant. At Indian Head, Maryland Point, 301 Bridge, and Ragged Point the trends were statistically significant with increases of 21%, 22%, 38%, and 73%, respectively. There were also statistically significant upward trends in DIN at the four upper stations, which ranged from 41% to 80%. The larger increase of 80% at the 301 Bridge station rather than the +40% increases at upper three stations was probably due to lower alga uptake in the upper estuary and greater down stream transport. A statistically significant decrease of 42% was observed in DIN at Point Lookout, which may be more influenced by Chesapeake Bay water quality than by changes in the upper estuary. During the high stream flows that occurred during all of 1996, the TN levels at the upper five stations were relatively constant at approximately 2.5 mg/l.

Consequences of Eutrophication

We selected indicators of the consequences of over enrichment to be increased chlorophyll and TOC levels, reduced light penetration, and reduced dissolved oxygen concentrations in the bottom waters. The trend analysis results for these indicators are presented in Table 2 for all six stations and plots of the data are presented in Figures 10, 11, 12, and 13 for Indian Head, 301 Bridge, and Point Lookout.

The TOC concentrations (Figure 10 and Table 2) have also statistically significantly decreased at the five upper stations by 26% to 61%, in a fashion similar to TP. Since 1985, the TOC levels at all stations were very similar ranging from 3 to 6 mg/l or about two fold greater than at the fall line. The cause of the elevated levels of TOC in the late 1970s and early 1980s is unknown in that the wastewater and riverine inputs were not higher than normal.

Statistically significant decreases in chlorophyll were detected at all stations except Point Lookout. Chlorophyll concentrations for the upper five stations have decreased by an average of 60% (Table 2).

Statistically significant improving trends in bottom dissolved oxygen were observed at Piscataway Creek and Indian Head. A statistically significant degrading trend in bottom DO was detected at 301 Bridge; DO was unchanged at Maryland Point, Ragged Point, and Point Lookout. As a result of reduced TP concentrations, chlorophyll levels decreased and there were fewer occurrences of noxious alga blooms (10). As described earlier, the higher levels of chlorophyll in the late 1970s and early 1980s appear to have occurred in response to elevated levels of TP.

Although the changes were not statistically significant, light penetration in the upper estuary has improved from approximately 20 inches to 30 inches, except during 1996 when light penetration fell below 20 inches (Figure 12). The decrease in light penetration observed during 1996 may have resulted from elevated levels of suspended particles in the upper basin, which occurred during high stream flow periods. Improved light penetration in the upper basin is attributed to advanced wastewater treatment.

At 301 Bridge and Ragged Point, light penetration also appeared to have improved, again except for 1996. At Point Lookout, light penetration declined by a statistically significant 30%. The reduced light penetration at most of the stations during the 1975-1985 period appears to have been in response to increased chlorophyll levels.

The bottom water DO levels in the upper estuary at the Piscataway Creek and Indian Head stations have improved mainly in response to the greater removal of Biochemical Oxygen Demand (BOD) in the wastewater treatment plants (Figure 13 and Table 2). The bottom water DO concentrations for the 301 Bridge, Ragged Point, and Point Lookout stations have not improved over the 1965-1996 time period and were often less than 2.0 mg/l during summer time conditions. At the Piscataway Creek station, there is a more complete picture of surface DO for the 31 year period as shown in Figure 14. The improvement in DO was not only due to the greater removal of carboneous BOD, but also resulted from the reduction of the nitrogenous BOD when the wastewater treatment plants began to nitrify as part of the advanced treatment process. This additional treatment process increased the DO at Piscataway Creek by about 2.0 mg/l. Since 1985, except for one month, the DO has been over 5.0 mg/l.

Comparison of Observed Estuarine Nutrient Concentrations with the Calculated Combined Loading Point-of Entry Concentrations.

Estimates were made of the TN, TP, and TOC levels of the upper estuary for the combined wastewater and upper basin export loadings (Figure 7). These Point-of-Entry concentrations, which were calculated from the combined fall-line and upper basin loadings and from the combined wastewater flow and river discharge data, represent the nutrient content of an upper estuary prior to tidal dilution and dispersion. The Point-of-Entry concentrations (Figure 7) are compared to the observed estuarine nutrient concentrations at Piscataway Creek (Figures 15, 16, and 17).

The observed and calculated Point-of-Entry estuarine concentrations at Piscataway Creek for all three nutrients were remarkably similar for the +31-year study period. The predicted TOC (Figure 15) and TP (Figure 16) levels reflect the improvements in wastewater treatment except for the 1975-1985 period where the predicted TP concentrations were much higher than the observed and the TOC concentrations were lower. This period was when many of the advanced wastewater treatment units went on line and some may have had start-up difficulties (9).

For TN (Figure 17), the observed and calculated estuarine concentrations at Piscataway Creek were also remarkably similar, except during low stream flow periods when the predicted concentrations were much higher throughout the +31 years. When one examines the nitrate concentrations at Piscataway Creek and at Great Falls (Figure 18), it can be seen that prior to the initiation of nitrification at most major wastewater plants, which occurred around 1979-80, the nitrate levels were around 1.0 mg/l. Since the early 1980s, with higher nitrate concentrations being discharged from the treatment facilities and from the upper basin, the nitrate level at Piscataway Creek has more than doubled to +2.5 mg/l .

A scatter plot showing the relationship between the predicted Point-of-Entry TN concentration data at the Piscataway Creek station and the observed levels has a slope of 0.37 and an intercept of 1.4 (Figure 19). This suggests that if the concentration of the combined inputs was reduced by 1.0 mg/l, the concentration at Piscataway Creek, after dilution and tidal dispersion, would be reduced by approximately 0.37 mg/l. Thus, the regression relationship can be used to gain some insight as to how the concentration of TN at Piscataway Creek might respond to either upper basin or wastewater management of nitrogen.

Summary

Landscape loadings to the upper Potomac River Basin have increased over the past +90 years. TN loadings having increased by a factor of +5.0 during this period. More recently, for the 1965-1996 period, the TN landscape loadings have increased by approximately 37%.

Water quality at Great Falls on the upper Potomac River as measured by total residue, hardness, alkalinity, chlorides, and nitrates, has significantly degraded over the past +90 years. The total residue has doubled and the river water has become harder and more alkaline. The chloride and nitrate riverine concentrations have increased by approximately +8 fold.

Riverine export of nitrates and potassium from the upper basin into the upper estuary has increased +8 fold. Currently about 50% of the TN is from the upper basin and the remaining 50% is from the wastewater treatment facilities whose discharges are to the upper estuary.

Major improvements in wastewater treatment in the mid-1970s and early 1980s have significantly reduced the TP and TOC loadings from municipal wastewater treatment plants discharging into the upper estuary. TN loadings for the +31 year period from the treatment plants increased as wastewater flows increased.

Since 1965, with the treatment improvements, the total loadings of TP and TOC to the upper estuary have decreased by 56% and 29%, respectively. However, TN loadings from all sources to the upper estuary have increased by approximately 95% over the +31 year period.

In response to reductions in total loads, TP and TOC concentrations for the upper five stations in the estuary decreased during the 1965-1996 period by an average of 70% and 50%, respectively. Although a statistically significant decrease of 12% in TOC was observed at Point Lookout, the trend in TP, although decreasing, was not significant. Statistically significant increases were observed for DIN at the upper four stations, which changed on average by 54% during the 1965-1996 period. DIN decreased by 42% at Point Lookout.

The chlorophyll levels at all stations except Point Lookout have decreased in response to wastewater treatment plant improvements. Chlorophyll concentrations for the upper five stations have decreased by an average of 60%.

Light penetration increased in the upper estuary except for 1996, but has decreased in the middle and lower zones during the 1965-1996 period. There was a statistically significant 30% decrease in light penetration at Point Lookout.

Bottom DO concentrations have improved in the upper estuary with the Piscataway Creek station levels usually above 5.0 mg/l during the summer months. However, the DO in the bottom waters of the deep channels of the lower estuary during the 1965-1996 period were hypoxic with summer time concentrations often near 0 mg/l. At Point Lookout, there has been no discernible change in DO in the bottom water of the deep channels during the past +31 years.

The comparison of the nitrate nitrogen concentrations at Piscataway Creek with those at Great Falls for the +31 years reflects the initiation of nitrification at most major wastewater treatment plants, which was around 1979-80. The nitrate levels were around +1.0 mg/l. Since the early 1980s, with higher nitrate concentrations being discharged from the treatment facilities and from the upper basin, the nitrate levels at Piscataway have more than doubled to +2.5 mg/l.

Estimates were made of the TN, TP, and TOC Point-of-Entry of the upper estuary from the combined wastewater and upper basin export loadings compared favorably to the observed estuarine nutrient concentrations at Piscataway Creek.

The relationship between the calculated Point-of-Entry concentrations of the combined TN inputs and the observed levels at the Piscataway Creek station illustrates the effect of dilution and tidal dispersion. This statistical relationship can be used to gain some insight as to how the concentration of TN in the upper estuary at Piscataway Creek may respond to changes in either upper basin and/or upper estuary wastewater nitrogen loadings.

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Table 1. Upper Potomac Estuary Wastewater Loadings

Year	Waste Flow	TOC	TN	TP
	(million liters/day)	(kg/day)	(kg/day)	(kg/day)
1913	158	17620	2900	50
1932	283	31360	5270	1000
1944	630	60900	10400	2000
1960	737	33430	14400	4500
1965	1020	40250	22960	9220
1970	1180	44800	27040	10930
1975	1410	23040	26500	2890
1980	1730	29268	25670	2480
1985	1750	4200	26080	320
1990	1890	5940	29190	260
1995	1970	4350	32430	160
1996	2030	7080	30200	230

Table 2. Water Quality Trends for Six Estuarine Sampling Stations, 1965-1996

Station		Total Nitrogen mg/l	Inorganic N mg/l	Total Phosphorus mg/l	T Organic Carbon mg/l	Chlorophyl ug/l	Bottom DO mg/l	Secchi Disk inches
Piscataway	Initial Median	2.389	1.47	0.169	5.8	36.4	5.7	18
(XFB2470)	% Change	9	**46**	**>-95**	**-50**	**-30**	**60**	0
	p-value	0.0404	0.0000	0.0000	0.0000	0.0000	0.0000	0.3843
Indian Head	Initial Median	1.8	1.368	0.168	5.86	32	5.7	21.2
(XEA6596)	% Change	**21**	**48**	**-83**	**-61**	**-50**	**20**	10
	p-value	0.0004	0.0000	0.0000	0.0000	0.0000	0.0003	0.1891
Maryland Point	Initial Median	1.477	1.05	0.065	5.24	6.2	6.8	30
(XDA1177)	% Change	**22**	**41**	**-69**	**-60**	**>-95**	0	-10
	p-value	0.0005	0.0000	0.0001	0.0000	0.0000	0.5727	0.0750
301 Bridge	Initial Median	0.926	0.393	0.099	5.06	15.3	5.9	40
(XDC1706)	% Change	**38**	**80**	**-36**	**-57**	**-60**	**-20**	0
	p-value	0.0007	0.0000	0.0000	0.0000	0.0000	0.0025	0.1904
Ragged Point	Initial Median	0.504	0.081	0.093	5.45	13.2	5.2	79.5
(MLE2.2)	% Change	**73**	86	**-24**	**-26**	**-50**	0	-20
	p-value	0.0002	0.0441	0.0018	0.0070	0.0066	0.7631	0.0357
Point Lookout	Initial Median	0.394	0.174	0.031	10.57	5.8	5.9	120
(MLE2.3)	% Change	22	**-42**	-10	**-12**	20	-10	**-30**
	p-value	0.3652	0.0035	0.1693	0.0004	0.4698	0.1374	0.0000

Note: All trends are based on 31.58 years of data except for TOC which has 27.9 years except for Point Lookout which has 14.5 years.
Bold type indicates trends are statistically significant

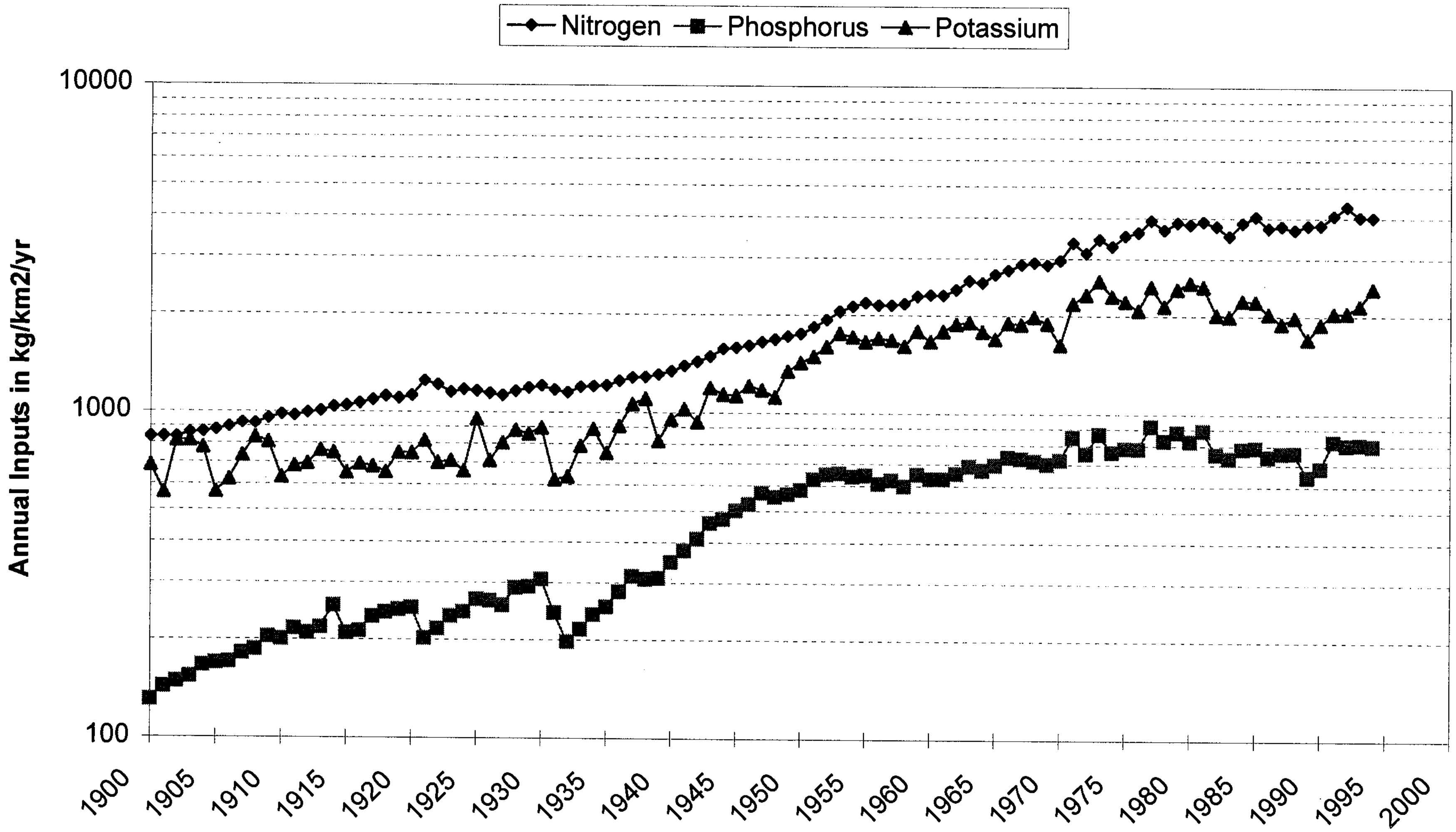


Figure 1. Upper Potomac River Basin Landscape Inputs

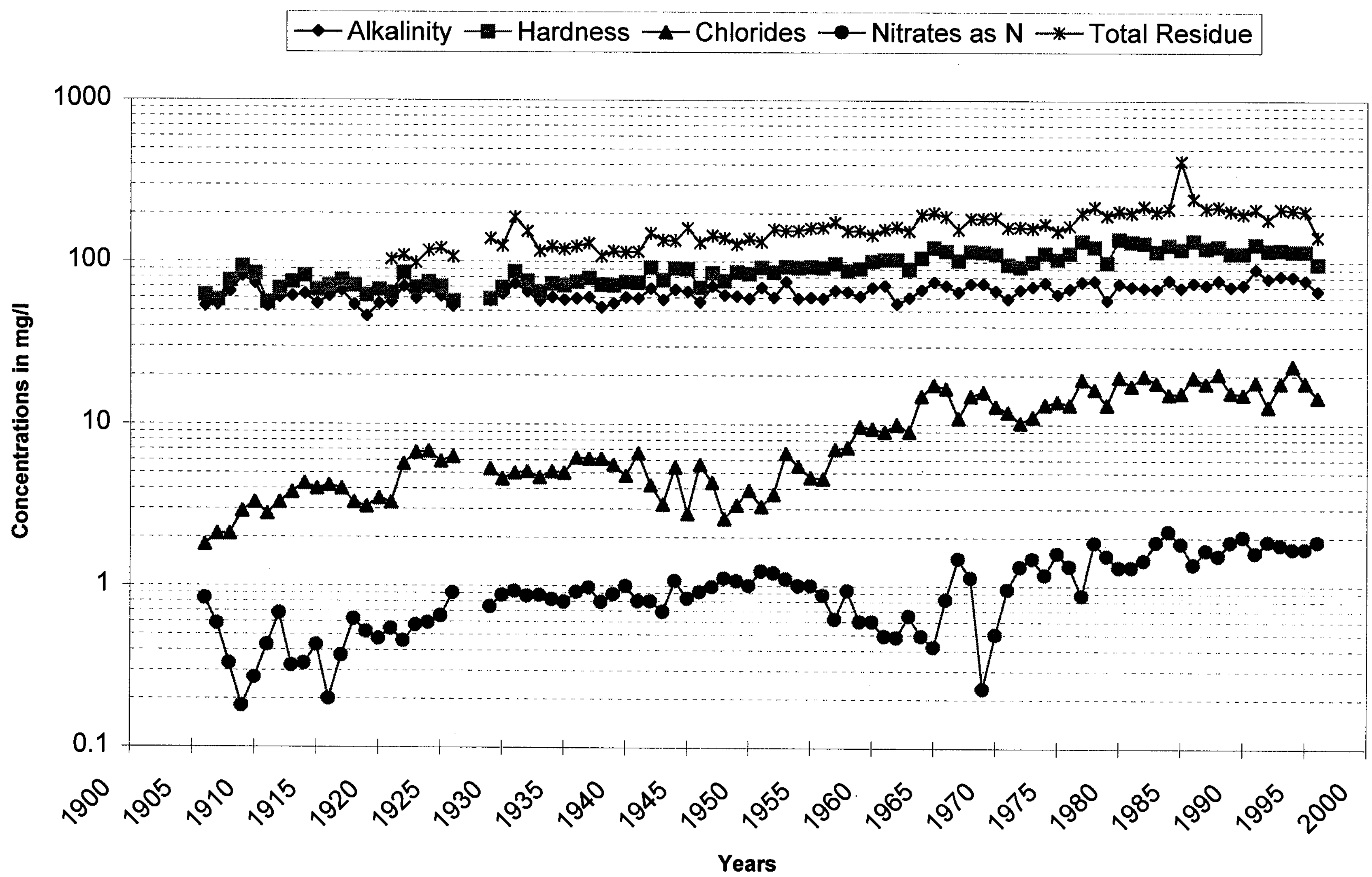


Figure 2. Potomac River Water Quality Trends

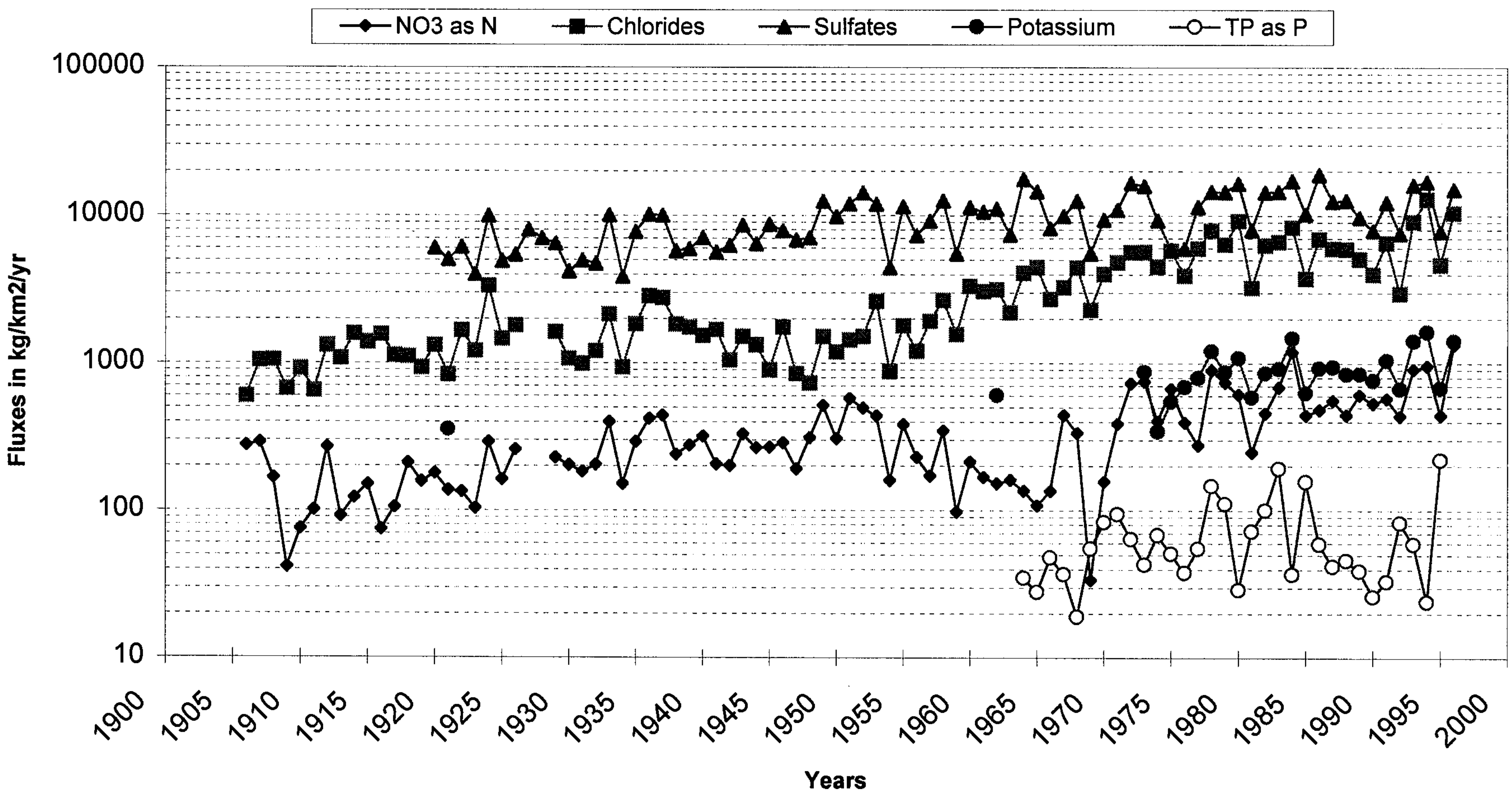


Figure 3. Potomac River Fluxes

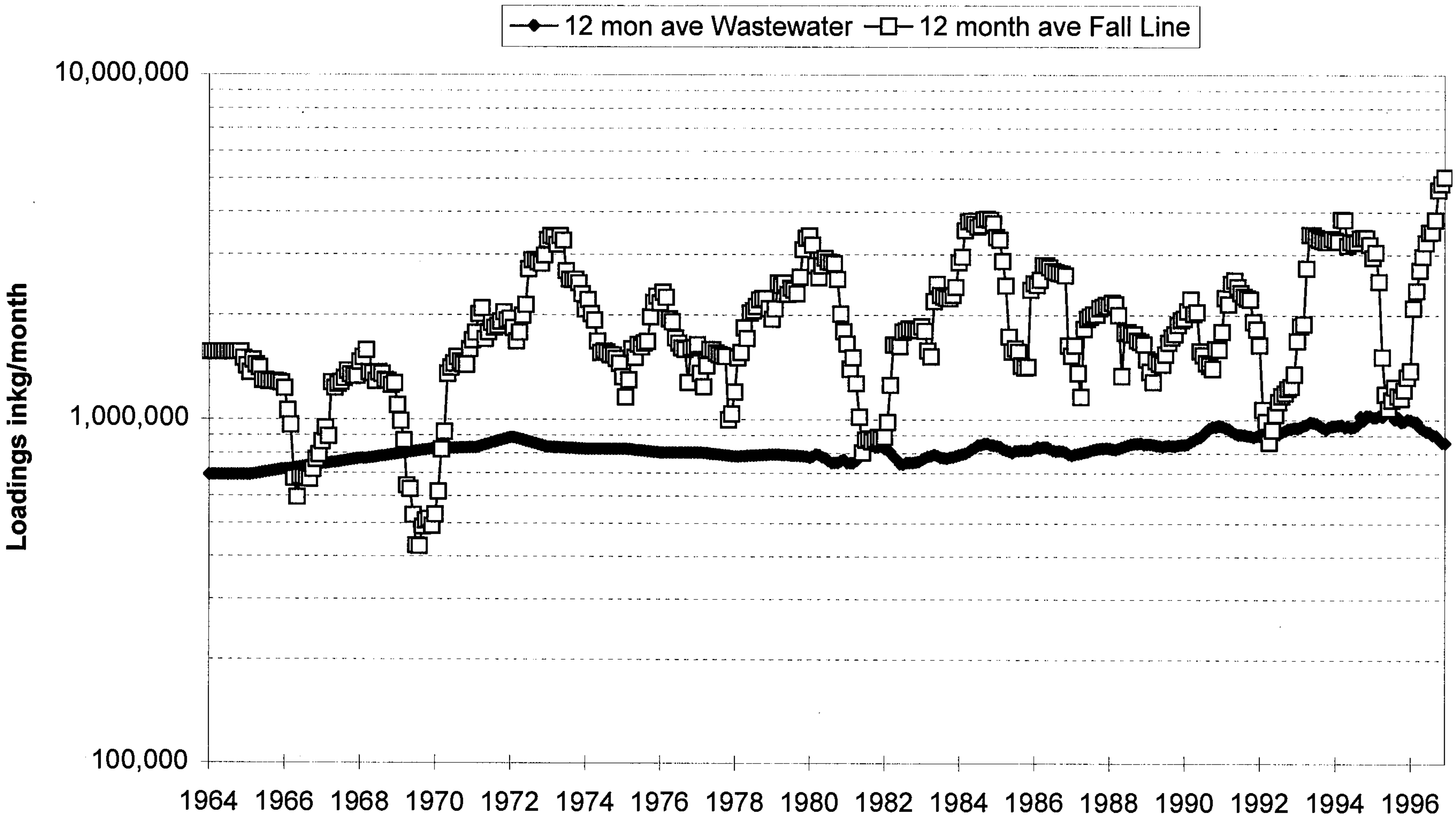


Figure 4. Fall Line & Wastewater TN Monthly Loadings

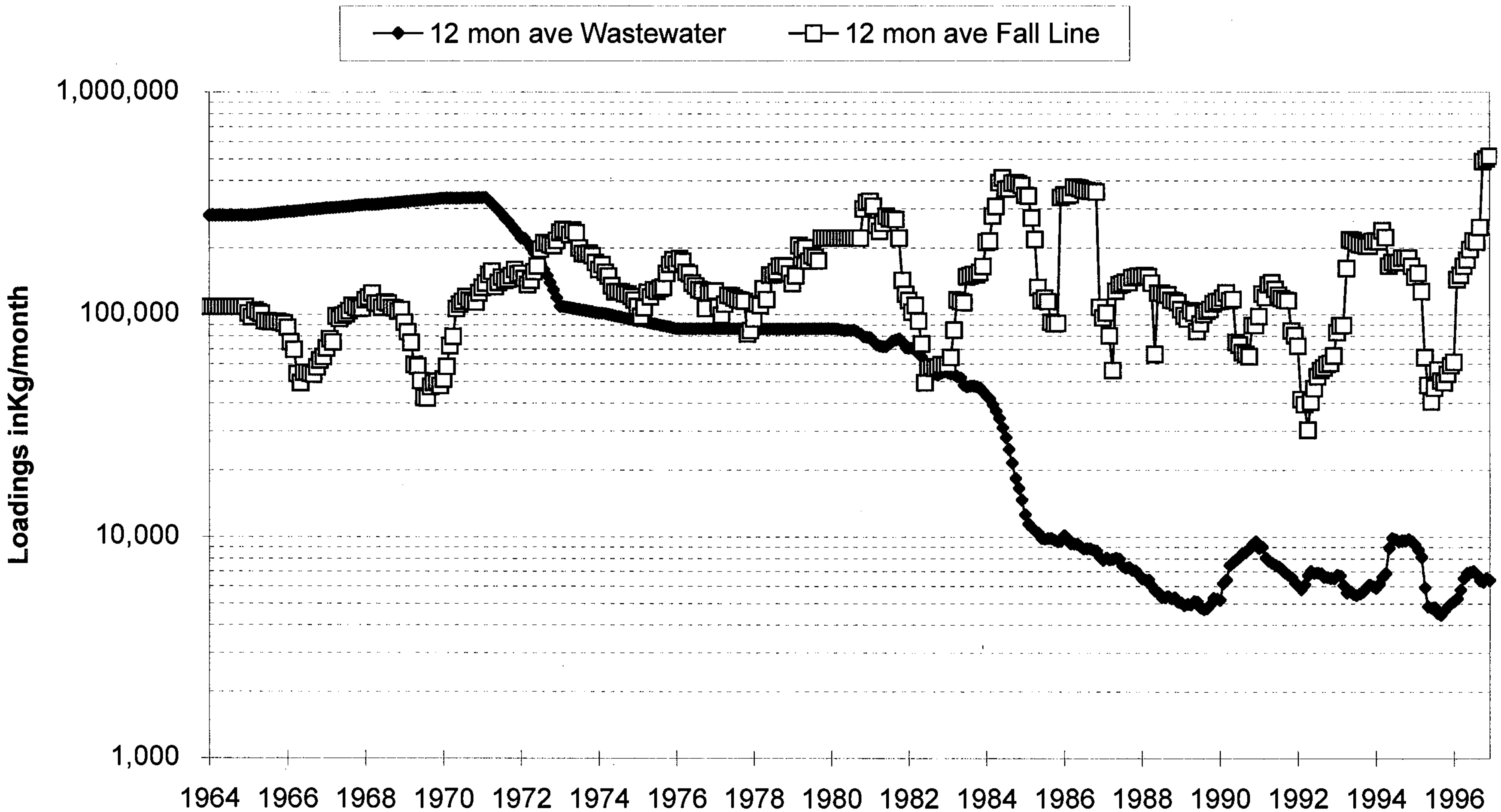


Figure 5. Fall Line & Wastewater TP Monthly Loadings

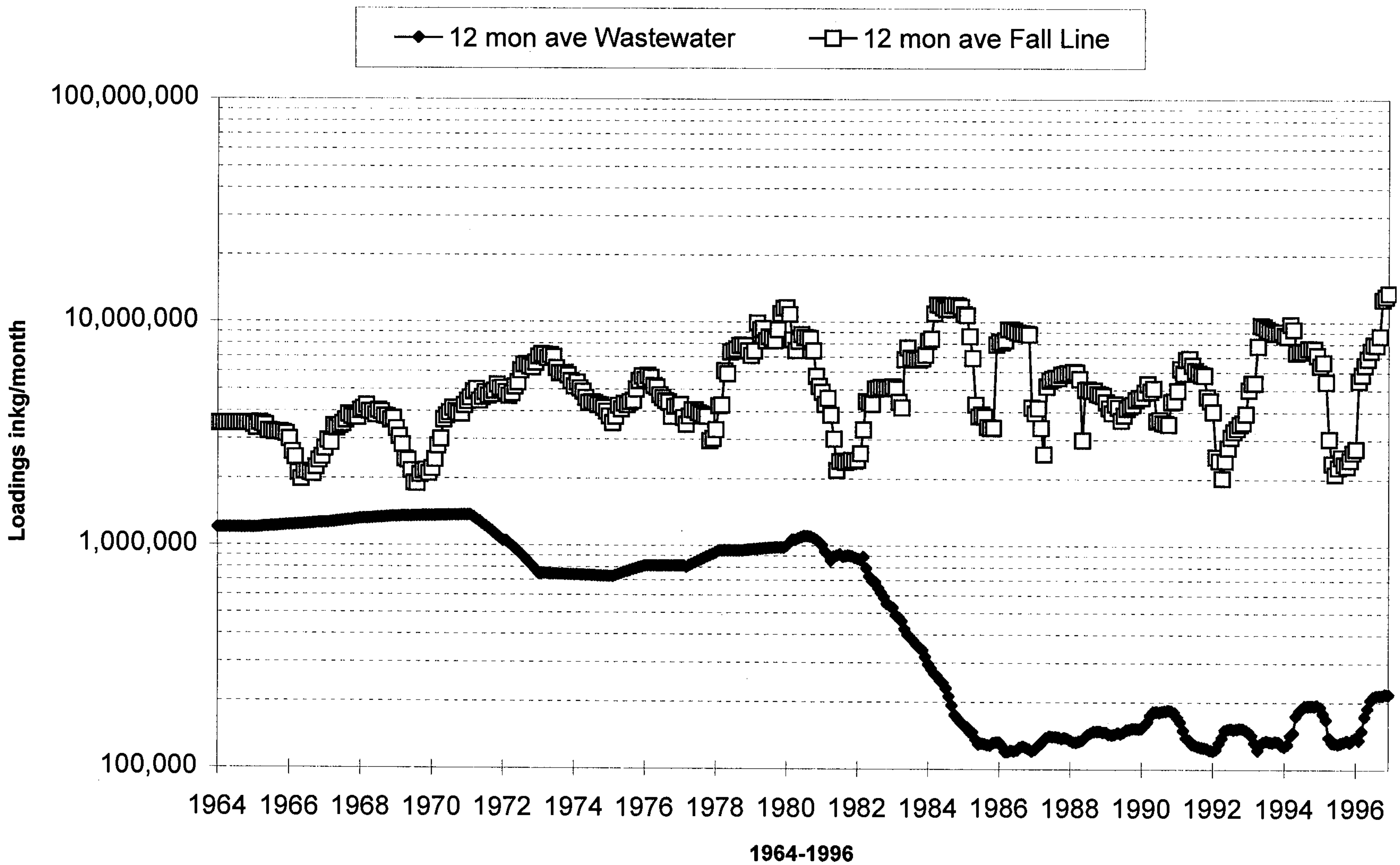


Figure 6. Fall Line & Wastewater TOC Monthly Loadings

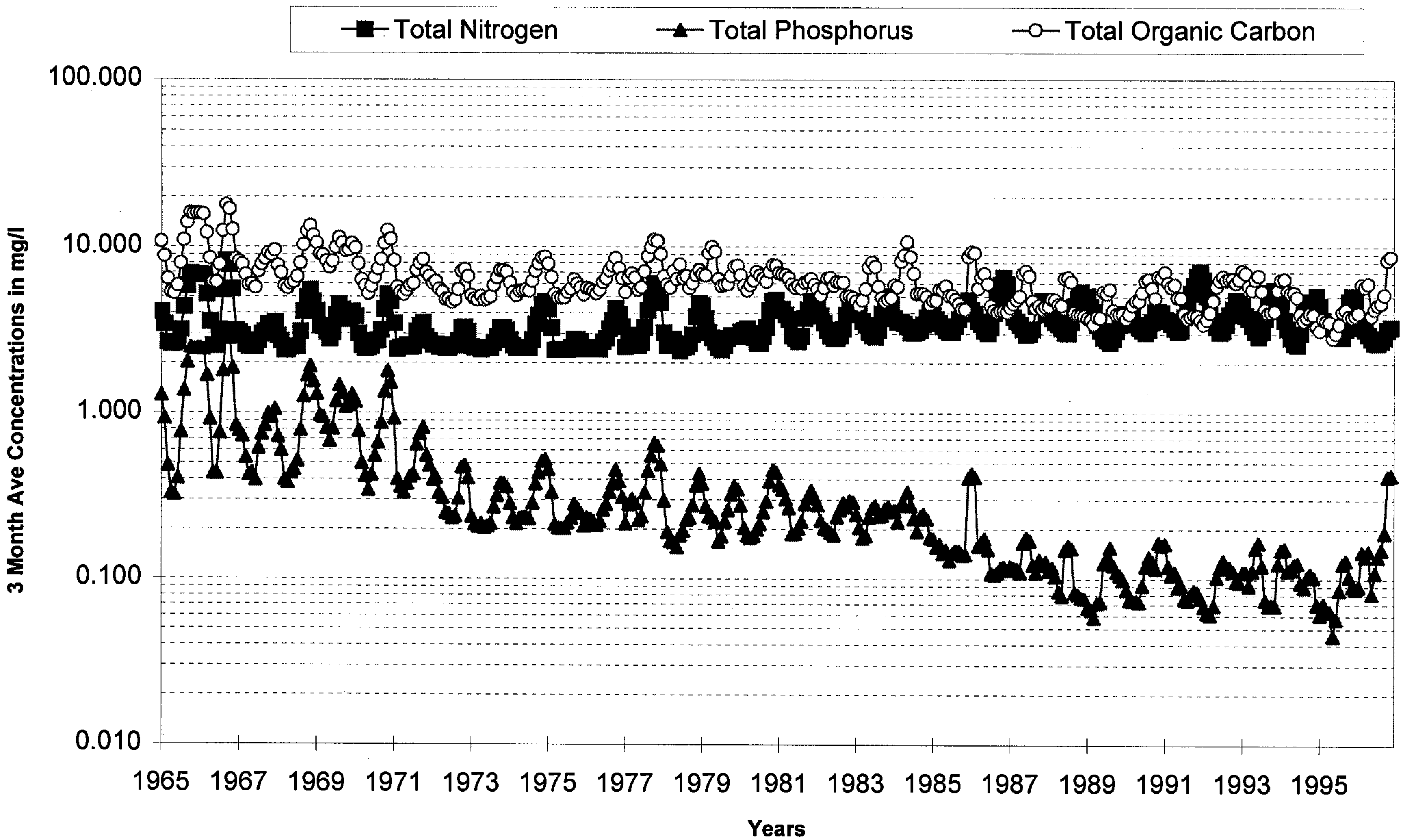
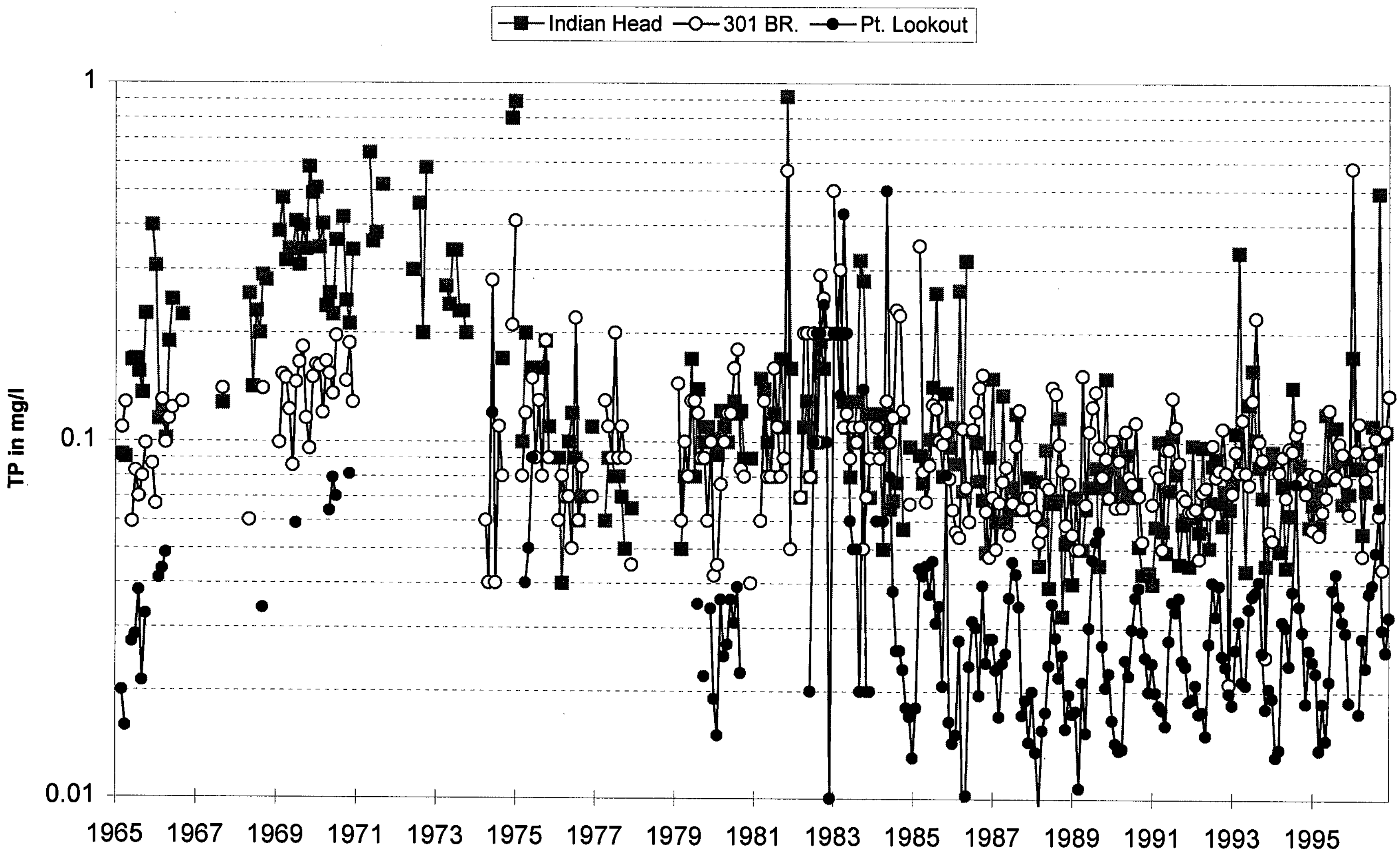


Figure 7. Calculated Point-of-Entry for the Upper Estuarine Nutrient Concentrations from Combined Wastewater and Riverine Inputs



1965-1996

Figure 8. Potomac Estuary TP Surface Concentrations

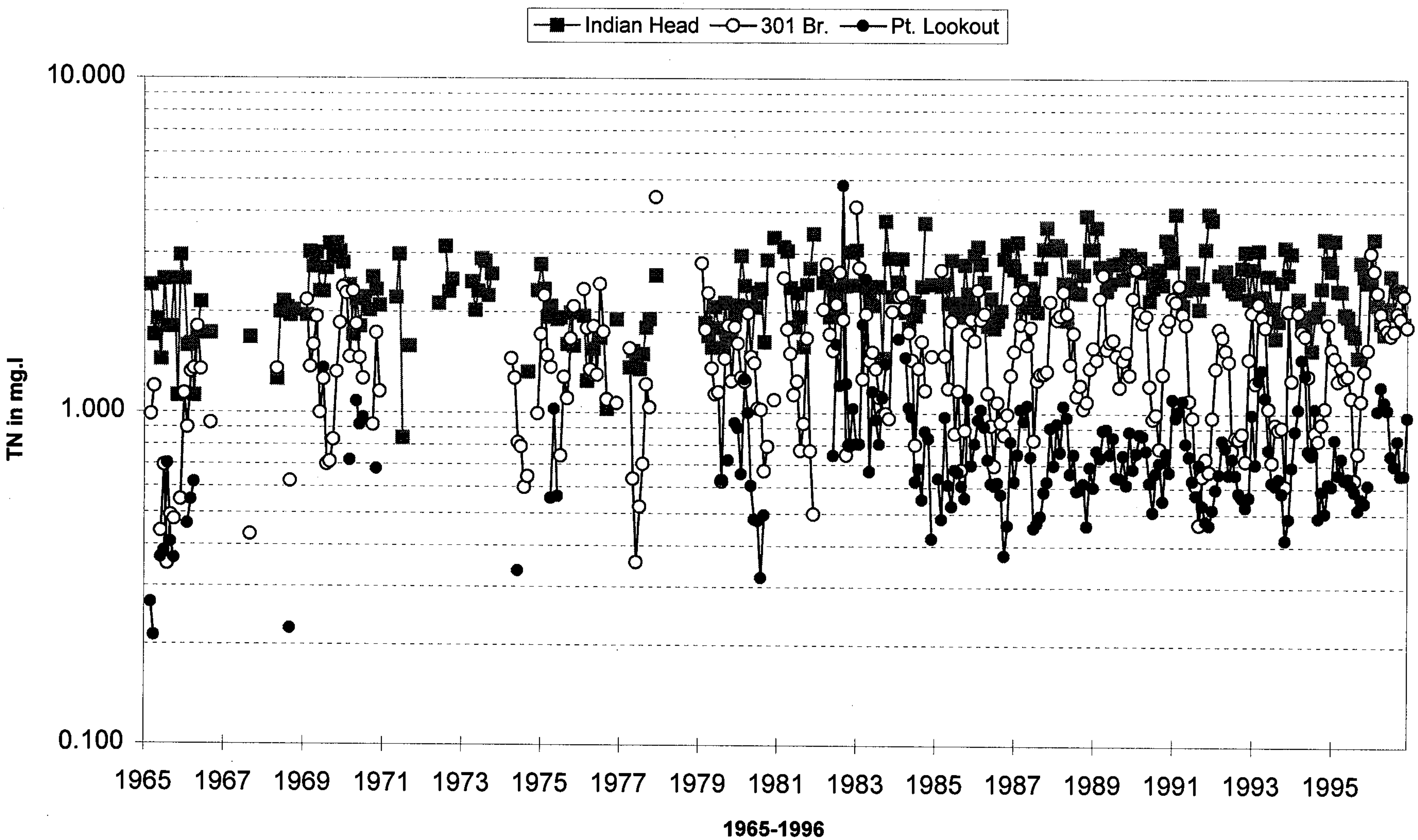


Figure 9. Potomac Estuary TN Surface Concentrations

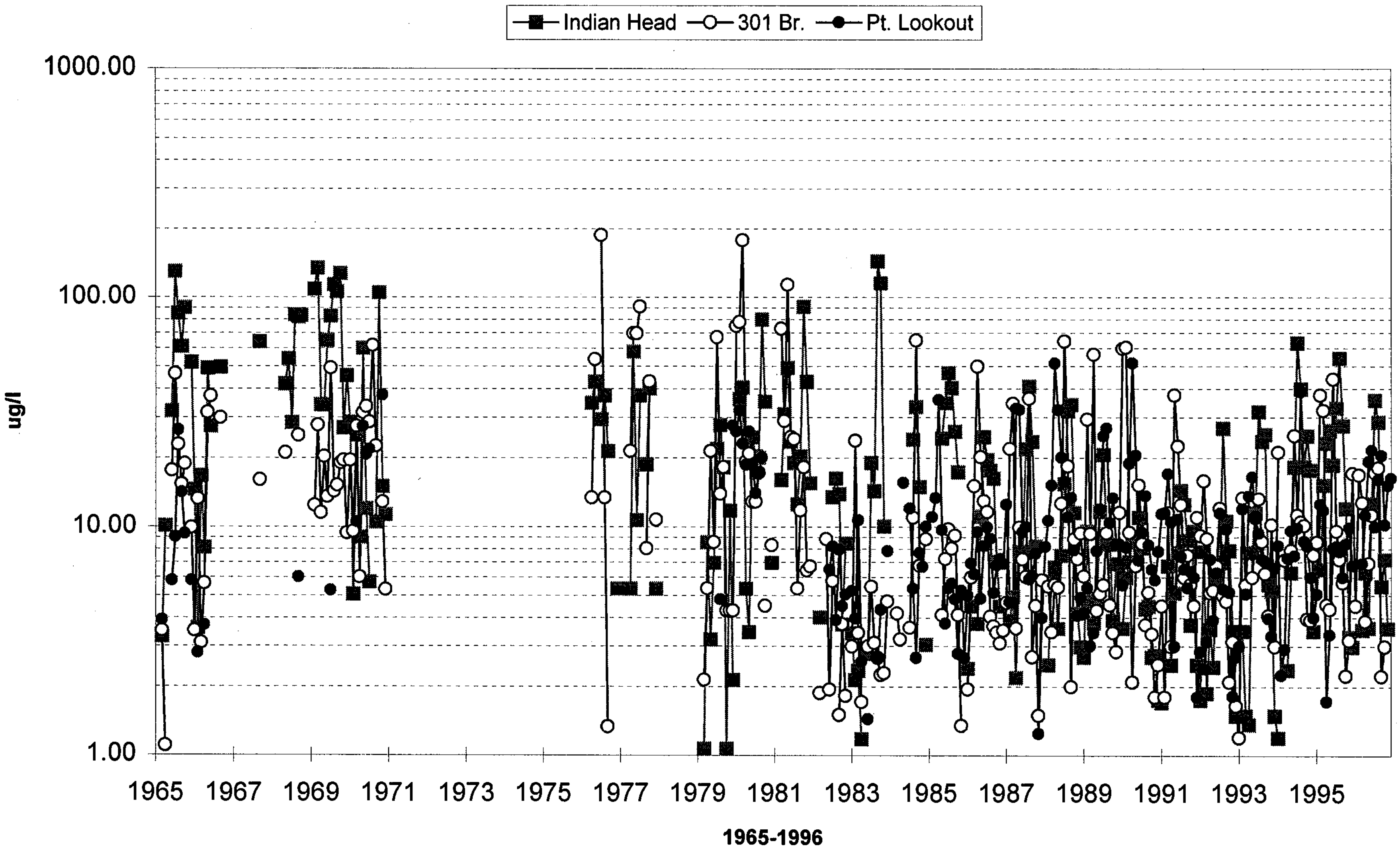


Figure 10. Potomac Estuary Chlorophyll Surface Concentrations

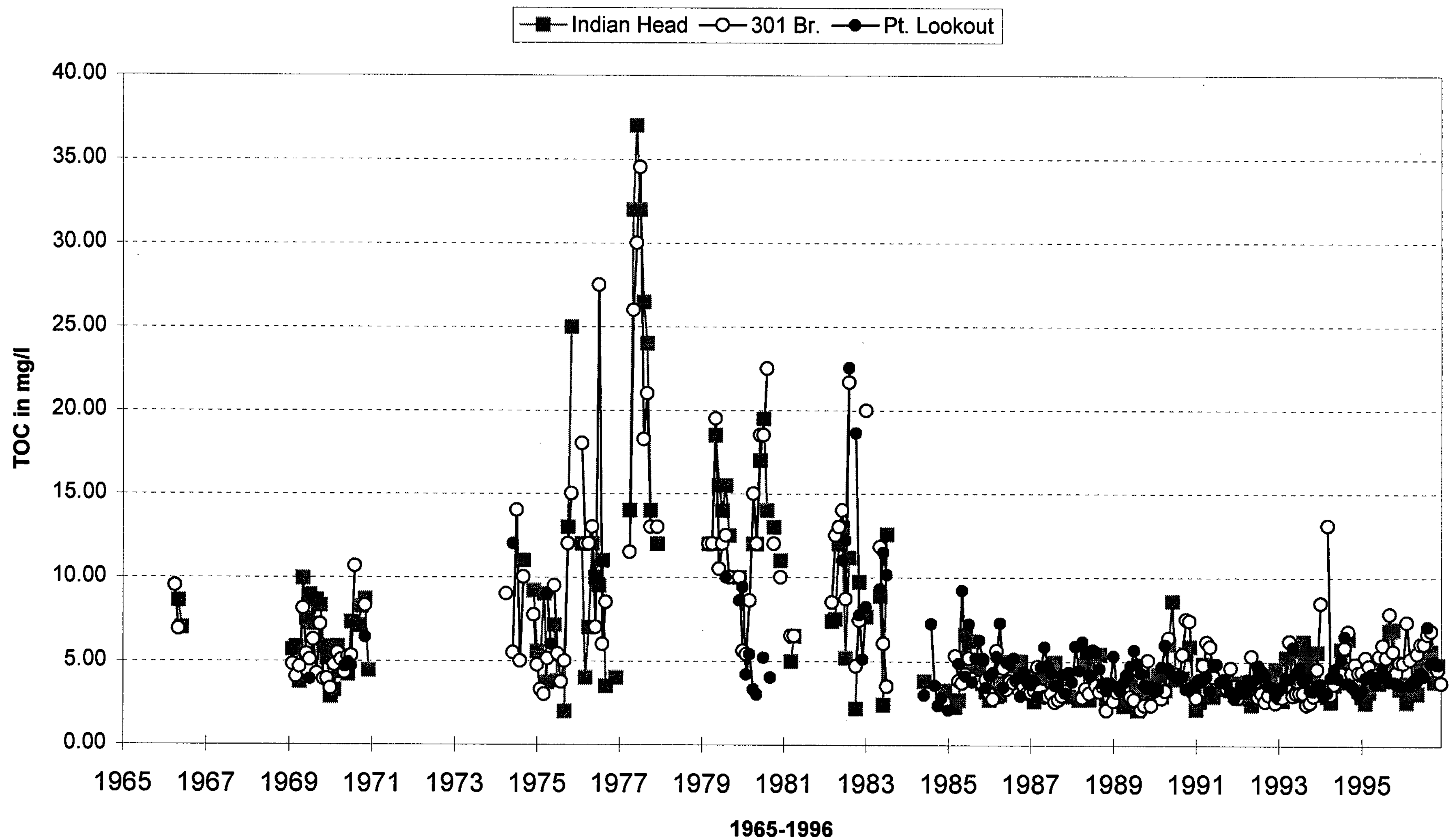


Figure 11. Potomac Estuary TOC Surface Concentrations

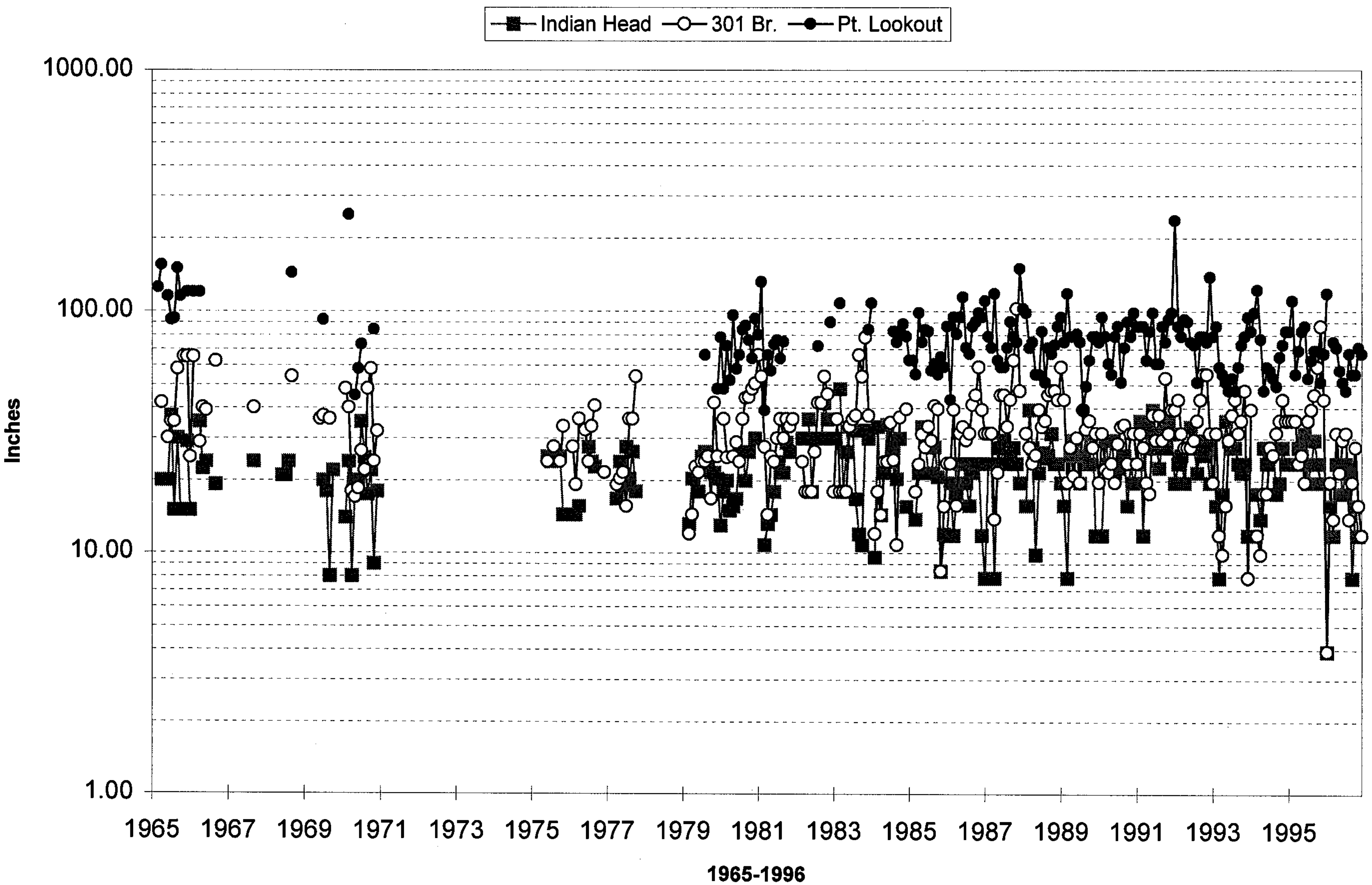


Figure 12. Potomac Estuary Secchi Disk Depths

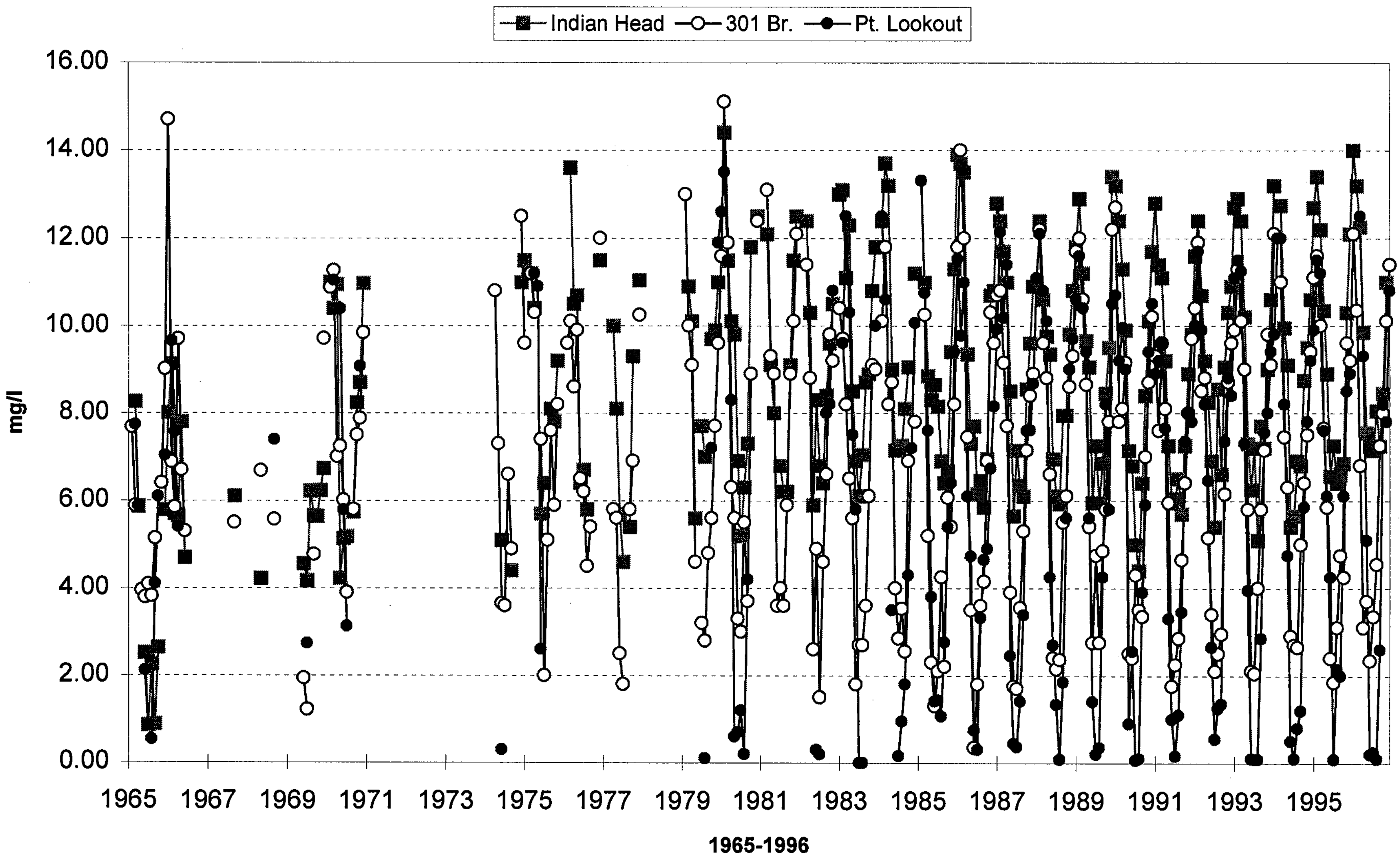


Figure 13. Potomac Estuary Bottom Dissolved Oxygen

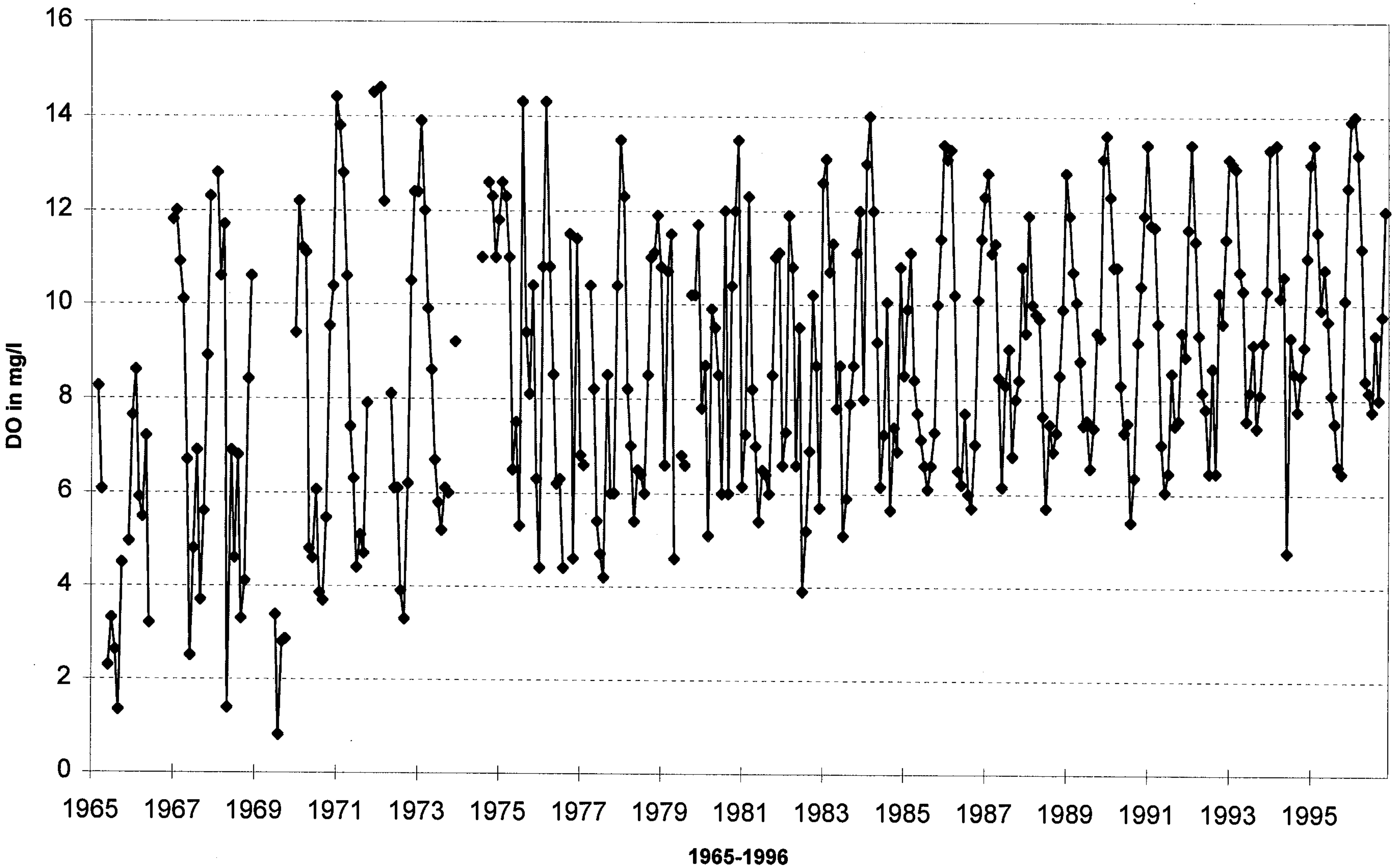


Figure 14. Dissolved Oxyen Concentrations of Potomac Estuary At Piscataway

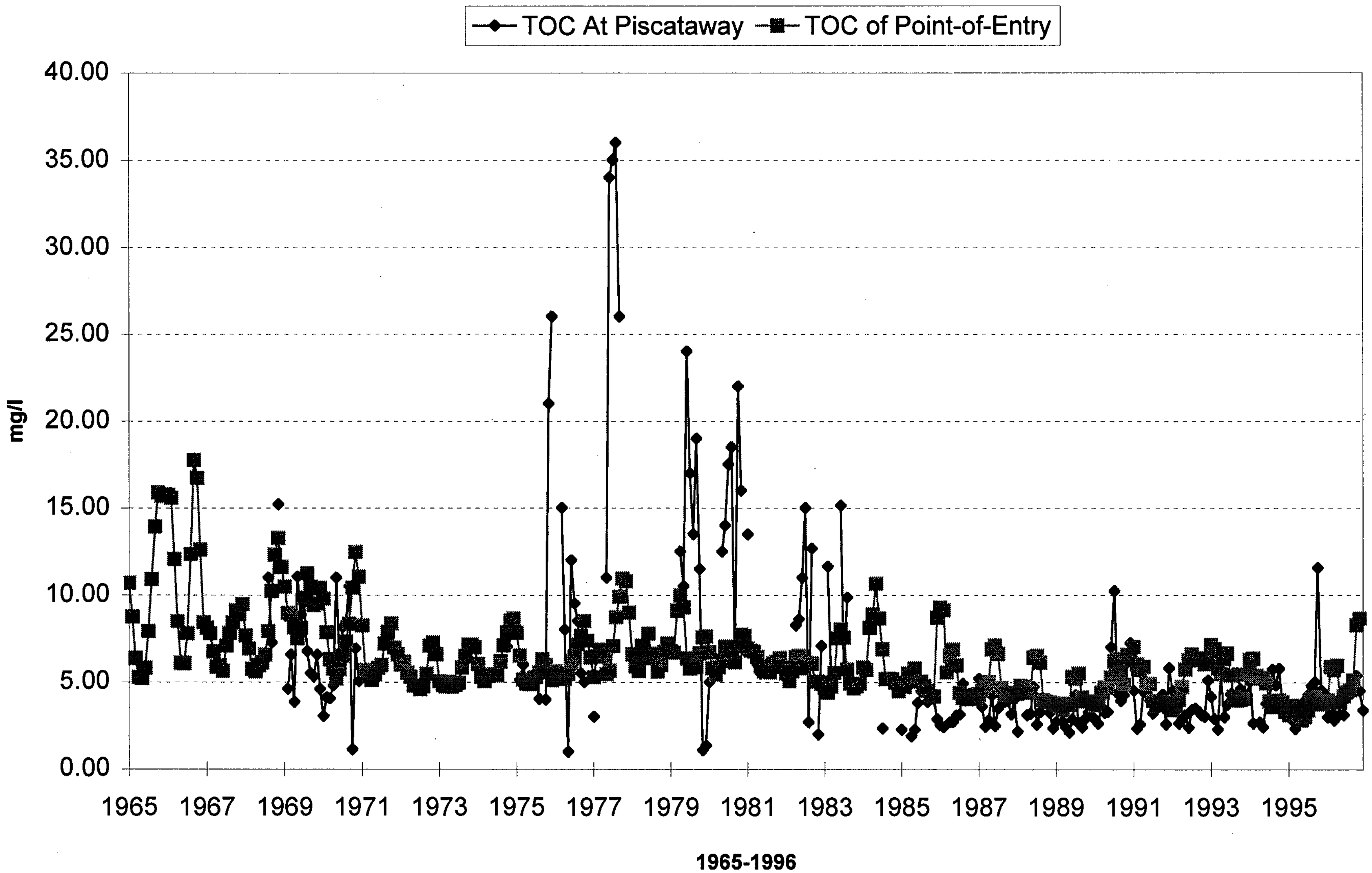


Figure 15. Total Organic Carbon Concentration Trends

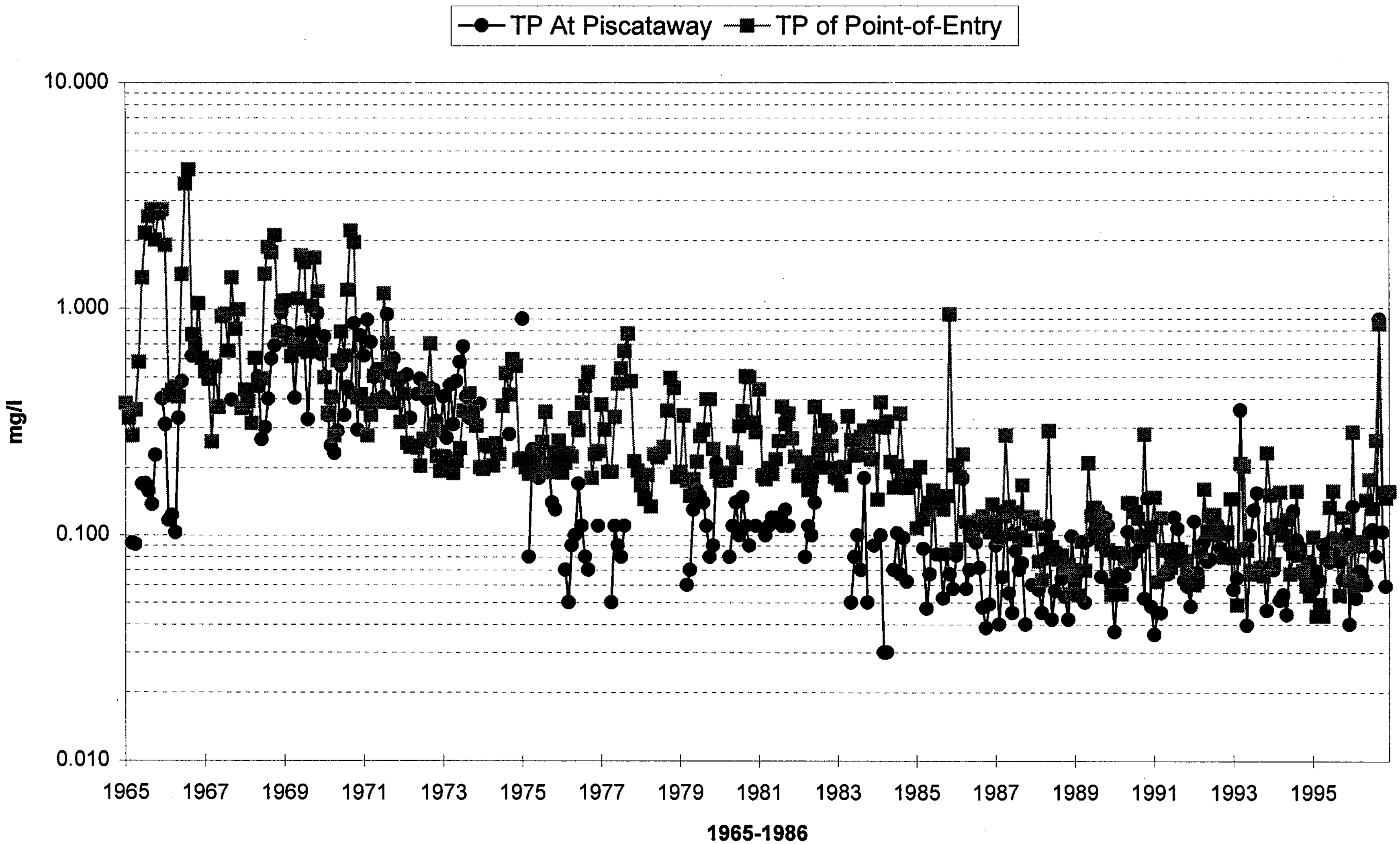


Figure 16. Total Phosphorus Concentration Trends

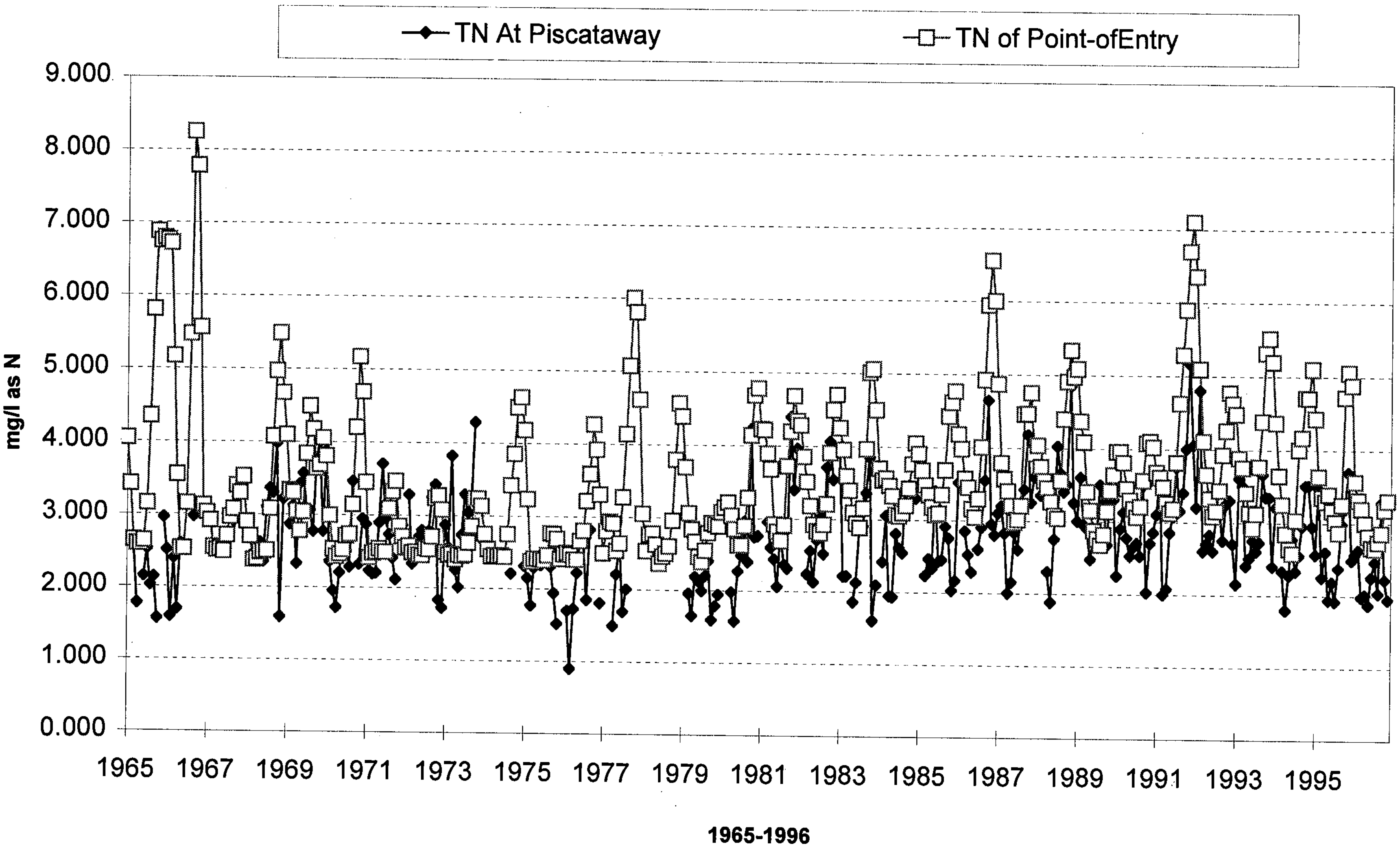


Figure 17. Total Nitrogen Concentration Trends

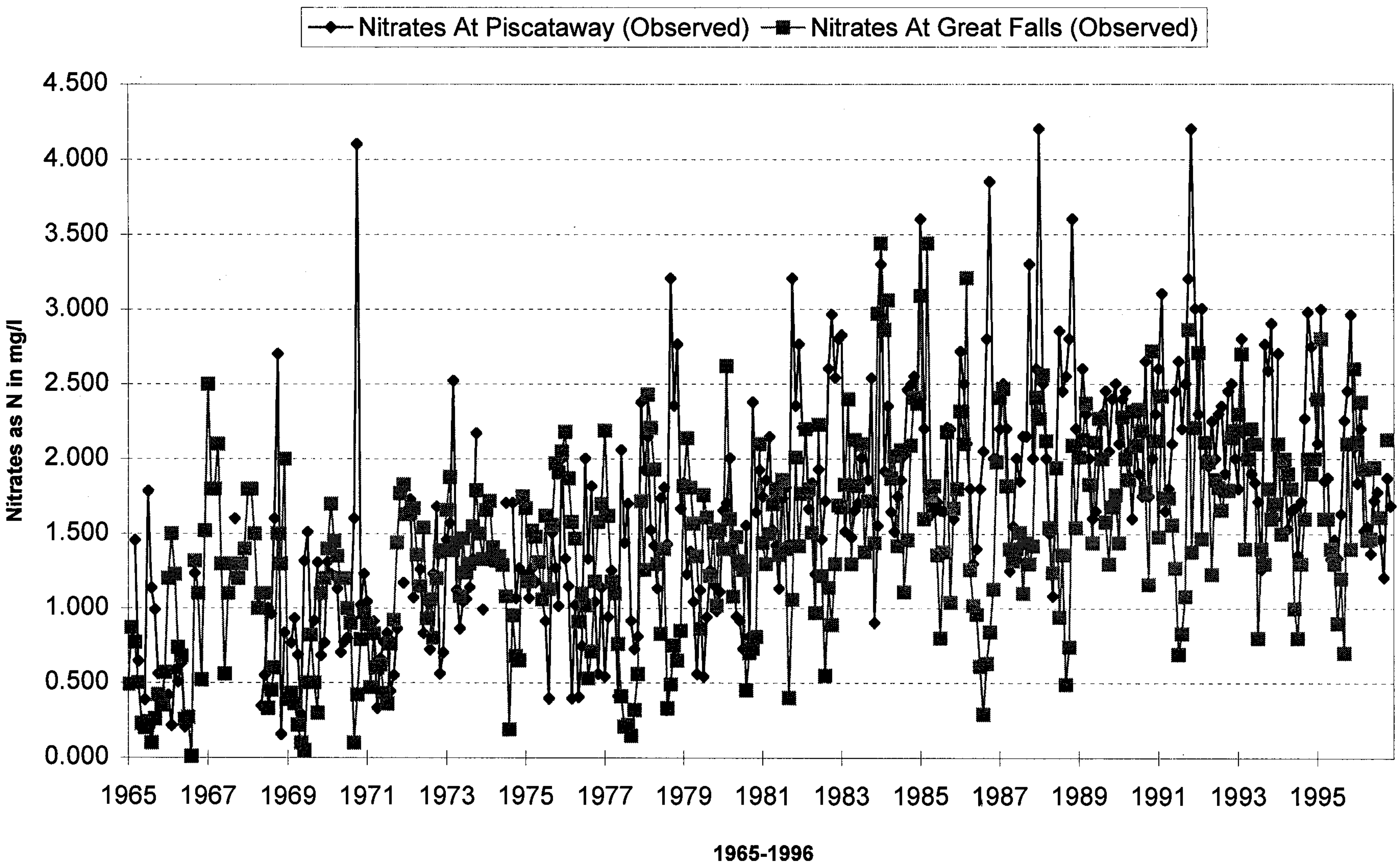


Figure 18. Nitrate Concentration Trends

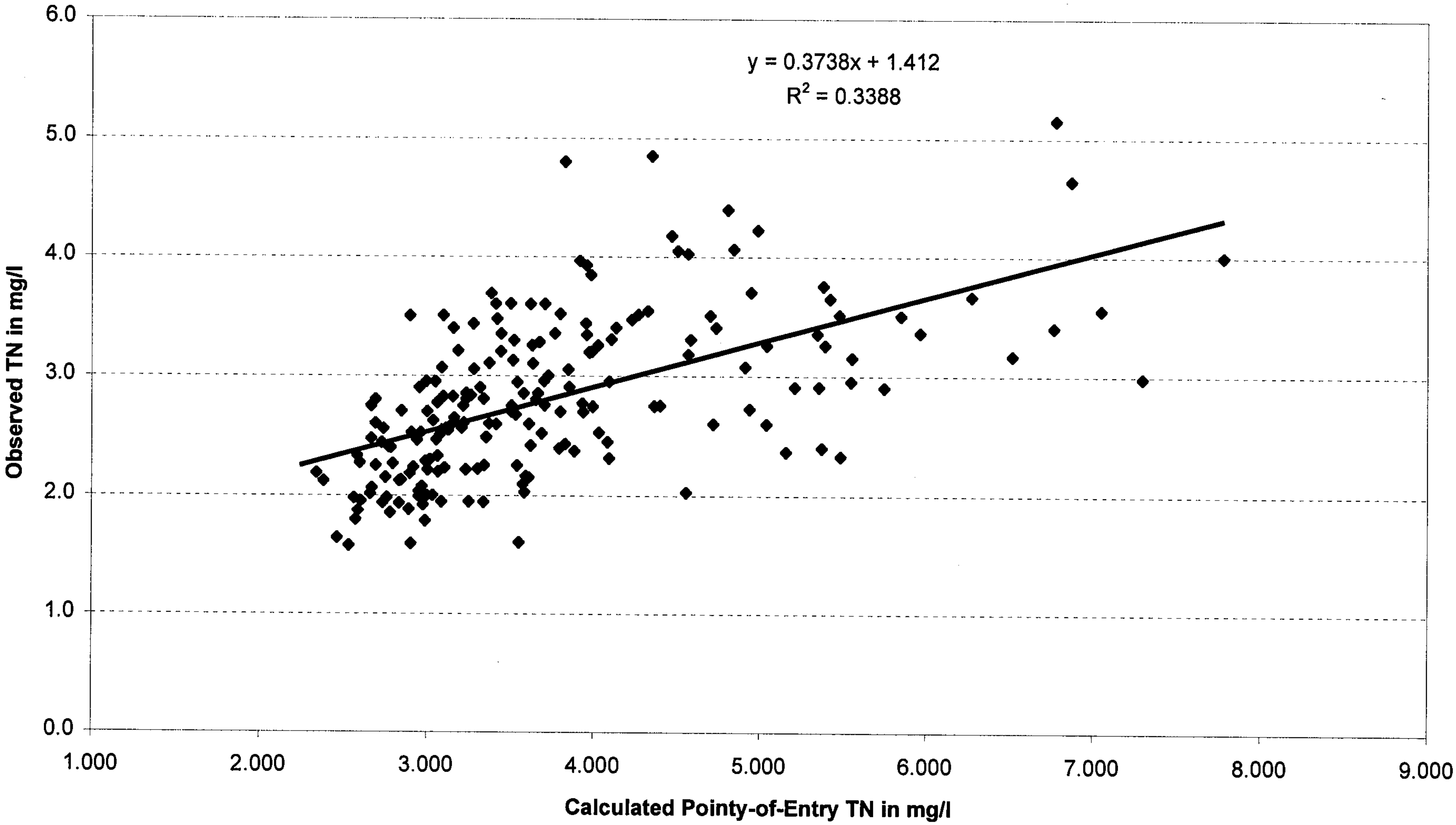


Figure 19. Calculated Point-of Entry TN Versus Observed Concentrations for Potomac Estuary at Piscataway Station 1980--1996

Tidal Potomac Integrative Analysis Project, A Series of Reports on the Water Quality and Living Resources Responses to Management Actions to Reduce Nutrients in the Potomac River Estuary, Final Draft.

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Prepared for the Chesapeake Bay Program.
ICPRB Report 99-4, 268 pp.

APPENDIX C

RIVER FLOW AND NUTRIENT LOAD CHARACTERISTICS; WATER QUALITY CONDITIONS IN THE LOWER POTOMAC (MLE2.2); AND SEDIMENT-WATER FLUXES IN THE POTOMAC ESTUARY

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POTOMAC RIVER INTEGRATED ANALYSIS PROJECT

- RIVER FLOW AND NUTRIENT LOAD CHARACTERISTICS
- WATER QUALITY CONDITIONS IN THE LOWER POTOMAC (MLE 2.2)
- SEDIMENT-WATER FLUXES IN THE POTOMAC ESTUARY

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July, 1997

INTRODUCTION

The Maryland Chesapeake Bay Water Quality Monitoring Program (Magnien et al. 1987) has multiple component programs, one of which is the Ecosystem Processes Component. This component of the monitoring program has been making measurements of sediment-water nutrient and oxygen exchanges at a number of sites (8-10) in mainstem Chesapeake Bay and tributary rivers during summer periods since 1985. In addition, this component of the monitoring program has been active in exploring the monitoring data base for relationships among water quality variables. Of particular interest are relationships between inputs of water, organic matter and nutrients to Chesapeake Bay, and sub-systems such as the Potomac River, and estuarine responses in terms of algal biomass, deep water oxygen conditions and sediment-water oxygen and nutrient exchanges.

Estuarine eutrophication

During the past decade much has been learned about the effects of both natural and anthropogenic nutrient inputs (*e.g.*, nitrogen, phosphorus, silica) on such important estuarine features as phytoplankton production, algal biomass, seagrass abundance and oxygen conditions in deep waters (Nixon, 1981, 1988; Kemp *et al.*, 1983 ; D'Elia *et al.*, 1983; Malone, 1992; and Kemp and Boynton, 1992). While our understanding is not complete, important pathways regulating these processes have been identified and related to water quality issues. Of particular importance here, it has been determined that (1) algal primary production and biomass levels in many estuaries (including Chesapeake Bay) are responsive to nutrient loading rates, (2) high rates of algal production in surface waters and low dissolved oxygen conditions in deep waters are sustained through summer and early fall periods by sediment recycling of essential nutrients and sediment oxygen consumption, respectively and (3) deposition of organic matter from surface to deep waters links these processes of production and consumption (Boynton *et al.*, 1982a ; Garber *et al.*, 1989).

Research conducted in Chesapeake Bay and other estuaries indicates that estuarine sediments act as important storage sites for nutrients as well as sites of intense organic matter decomposition and oxygen consumption (Kemp and Boynton, 1984). For example, during summer periods in the Choptank and Patuxent estuaries, 40-70% of the total oxygen utilization was associated with sediments and 25-70% of algal nitrogen demand was supplied from estuarine sediments (Boynton *et al.*, 1982b). Processes of this magnitude have a pronounced effect on estuarine water quality and habitat conditions. Sediments in much of Chesapeake Bay, especially the upper bay and tributary rivers, contain significant amounts of carbon, nitrogen, phosphorus and other compounds (Boynton *et al.*, 1995). A large percentage of this material appears to reach sediments following the termination of the spring bloom and again after the fall bloom. A portion of this material is available to regenerative processes and once transformed into inorganic nutrients again becomes available for algal utilization. Nutrients and other materials deposited or buried in sediments represent the potential "water quality memory" of the bay.

Nutrients enter the bay from a variety of sources, including sewage treatment plant effluents, fluvial inputs, local non-point drainage and direct rainfall on bay waters. Dissolved nutrients are rapidly incorporated into particulate matter via biological, chemical and physical mechanisms. Much of this particulate material then sinks to the bottom and is potentially available for remineralization. Essential nutrients released during the decomposition of organic matter may then again be utilized by algal communities. A portion of this newly produced organic matter sinks to the bottom, contributing to the development of anoxic conditions and loss of habitat for important infaunal, shellfish and demersal fish communities. The regenerative capacities and the potentially large nutrient storages in bottom sediments ensure a large return flux of nutrients from sediments to the water column and thus sustain continued phytoplankton growth. Continued growth supports deposition of organics to deep waters, creating anoxic conditions typically associated with eutrophication of estuarine systems (Figure 1). To a considerable extent, it is the magnitude of these processes which determines nutrient and oxygen water quality conditions in many zones of the bay. Ultimately, these processes are driven by inputs of organic matter and nutrients from both natural and anthropogenic sources. If water quality management programs are instituted and loadings decrease, changes in the magnitude of sediment processes will serve as a guide in determining the effectiveness of strategies aimed at improving bay water quality and habitat conditions.

Scope of this work

The schematic diagram in Figure 2 summarizes this conceptual eutrophication model where increased nitrogen (N) and phosphorus (P) loads result in a water quality degradation trajectory and reduced nitrogen and phosphorous loads lead to a restoration trajectory. Sediment processes play a prominent role in both trajectories. The working hypothesis is that if nutrient and organic matter loading to the bay decreases then the cycle of intense algal blooming, deposition of algal detritus to sediments, sediment oxygen demand resulting in poor deep water dissolved oxygen conditions, release of sediment nutrients and continued high algal production based on these recycled nutrients will also decrease. Because loads and water and sediment quality processes are linked, as described above, all three are considered in this component of the analysis. This work focuses on water and sediment quality conditions in the mesohaline portion of the Potomac River estuary (water quality monitoring station MLE 2.2) because this is the only site in the Potomac where both water quality and sediment process work has been conducted routinely for a decade. Work reported here is based on data collected during the period January 1986 through December 1995. Both monthly, seasonally averaged and annual data are examined.

CHARACTERISTICS OF RIVER FLOW AND NUTRIENT LOADS (1986-1995)

Annual patterns and trends

On an annual average basis, river flow ranged between 8000 to 12000 cfs during all years but 1993 and 1994 when annual average flows were much higher (16000 to 17000 cfs; Figure 3). Annual average falline plus Blue Plains nutrient loads of TN and TP generally followed river flow patterns with higher loads in wet years and lower loads in drier years. Loads for TN ranged between 60,000 and 75,000 kg per day except during 1993 and 1994 when TN loads averaged between 110,000 and 125,000 kg per day. Lowest TN loads occurred in 1986 and 1995, both low flow years. Interannual patterns of TP loading were similar. During the decade for which loads were examined there did not appear to be any strong trends of either increasing or decreasing TN or TP loads (Figure 4).

Monthly patterns and trends 1988

Monthly patterns of river flow are richer and more suggestive of differing ecological effects among years (Figure 5). For example, there were 2 very high flow years (1993 and 1994) and 4 very low flow years (1986, 1990, 1992 and 1995). Other years in this record were intermediate. In addition, the peak flows, which might reasonably be expected to have more influence on estuarine conditions than low flows, varied considerably among years in terms of when they occurred. There were four years with winter maximum flows, four with early spring maximum flows and two with late spring (May) maximum flows. The water quality consequences of winter versus spring inputs appear to be considerable, as will be discussed later. The monthly pattern in TN and TP load reflects river flow conditions and, because of this, there is substantial interannual variability in loads (> 2x for TN; > 3x for TP). Possibly the most important point is that the seasonal timing of maximum loads differs among years and this may have strong water quality consequences.

The N:P ratio of the falline load was calculated (on a molar basis) and is shown in Figure 6. This ratio has some utility as an indicator of potential N or P limitation of phytoplankton. Several interesting points emerged from inspection of this time series plot. First, there is great seasonal variability in the fall line load ratio which ranges from a low of about 22 to a high of 122. In most years the ratio is highest in winter-spring because diffuse source nutrient loads are typically much richer in N than P (nitrogen compounds, especially nitrate which is a major component of diffuse source nitrogen, is very soluable and moves readily with runoff). In terms of loading, the system is always rich in nitrogen relative to phosphorus. This is similar to other portions of Chesapeake Bay and tributaries that have important diffuse source nutrient loads. Surface water N:P ratios (atomic basis; DIN/PO4) were also computed for a mesohaline site in the Potomac River estuary (Sta MLE 2.2). Ratios were high in winter-spring and low in summer-fall and exhibited a greater range than in the fall line load ratio. The traditional Redfield value for balanced nutrient conditions is about 16:1(shown as a range between 10 and 20 in Figure 6). Note that values were seldom this low (~20% of all observations). However, Fisher et al (1997) have found, using a bioassay approach, that strong P limitation of phytoplankton growth does not generally emerge until this ratio is about 100-150. Of the 120 observations shown in Figure 6, only 36 have N:P values greater than 100 and all occurr in winter or early spring. This suggests summer through fall N limitation of phytoplanktonic growth, consistant with the work of Fisher et al (1997).

CHLOROPHYLL-A CONDITIONS IN THE LOWER POTOMAC (Station MLE 2.2)

Monthly average water column chlorophyll concentrations exhibited large ranges during the period of observation at the water quality station in the mesohaline region of the Potomac River estuary (Figure 7). Of particular interest are the large spikes of chlorophyll-a during 1988, 1989 and 1990. The remaining years did not exhibit large spikes although seasonal patterns were evident. In general, maximum concentrations were observed in the late winter or spring; only in 1989 did high concentrations of chlorophyll-a persist through the summer period. No long term trend was obvious from inspection of the chlorophyll-a time series at this mesohaline site.

One of the main issues to be addressed for the Potomac River, as well as other locations in the Bay region, is establishing a nutrient loading rate that allows for resource restoration including a decline in algal blooming (as indicated by chlorophyll-a concentrations), restoration of higher bottom water dissolved oxygen concentrations and re-establishment of seagrass communities. Possible relationships between chlorophyll-a and nutrient loading rates were examined for the mesohaline region using data from water quality station MLE 2.2. TN load was plotted versus water column chlorophyll concentrations (monthly averages for both) and results are shown in Figure 8. No significant relationship emerged, as expected, from this simple treatment of the data. Considerably more interesting results emerged from two different groupings of data which included time lags between nutrient inputs at the fall line and chlorophyll-a responses in the mesohaline estuary (Figure 9). In the first case (top panel) peak chlorophyll-a concentrations which occurred in each year (in some cases a single monthly value if the peak was so defined; in cases of a protracted bloom several months were averaged to estimate peak concentrations) were plotted as a function of the monthly TN load 1-2 months prior to the time when peak chlorophyll conditions were observed. In this case the lag was introduced to account for the time required for nutrients (TN in this case) to be transported from the fall line to the mesohaline estuary and to then simulate phytoplanktonic growth. This sort of lag produces an unward sweeping curve that fits 9 of the 10 years of observation quite well. The 1990 chlorophyll-a data were strongly divergent wherein chlorophyll-a concentrations were much higher than would be predicted from lagged TN loads alone. The reason for this divergence is not clear at this time. Another approach used to examine the data base for load-chlorophyll-a relationships used the monthly TN load associated with the freshet (whenever it occured) in each year versus the peak water column chlorophyll concentrations that subsequently developed. In this case a humped curve results that has some interesting characteristics (Figure 9; lower panel). First, highest chlorophyll concentrations developed in association with late spring TN loads, as was the case in 1988 and 1989. These were not huge load years but they did develop very high chlorophyll concentrations. Very high load years (1987, 1993 and 1994) in which the peak load entered in March-April generated modest to low chlorophyll concentrations in the mesohaline estuary. This suggests that low temperature, limited sunlight and reduced residence times (because of high flow) combined to limit the development of algal stocks; inorganic nutrients may have been transported out of the lower Potomac to the Bay under these conditions. At the other extreme, small loads (whenever they occurred) produced low chlorophyll concentrations (except during 1990). Nutrients entering the system after some warming has occurred (i.e. May-June) have the ability to generate very large algal standing stocks.

DISSOLVED OXYGEN CONDITIONS IN THE LOWER POTOMAC (Station MLE 2.2)

Bottom water dissolved oxygen conditions at this station in the mesohaline portion of the estuary are very poor during the summer months (Figure 10). During some months, average dissolved oxygen concentration in the bottom 6 m of the water column was just a few tenths of a milligram per liter. Despite a large range in river flows (which influences the strength of water column stratification and reaeration of deeper waters) and nutrient loads (which provide essential elements supporting algal biomass, the decomposition of which utilizes dissolved oxygen), very low dissolved oxygen concentrations were observed at this site in all years. Even under the lowest river flow and nutrient load conditions observed between 1996 and 1995 (which would favor higher bottom water dissolved oxygen conditions) low dissolved oxygen concentrations were always observed during summer periods.

An estimate of the extent of hypoxia (water having less than 2 mg/l of dissolved oxygen) present in the mesohaline Potomac was also estimated. In the top panel of Figure 11 the cross-sectional area of the Potomac at Station MLE 2.2 having dissolved oxygen concentrations less than 2 mg/l was computed for each year. In the lower panel the result of intergrating the darken areas of the top panel is indicated. The hypoxic water mass has units of m^2-days rather than volume-days because there was only a single station in this region of the Potomac. If data from two mesohaline stations had been available, the more commonly computed hypoxic volume-days contained between two cross sections would have been reported . Hypoxic areas increased from 1986 through 1988, dropped sharply in 1989 and gradually increased to the present time. It is important to note that the interannual range in hypoxic cross-sectional days is relatively small compared to the interannual ranges in variables causing hypoxic conditions (river flow, nutrient loading rates, in-situ chlorophyll-a concentrations) suggesting an attenuated response of hypoxia, at least in this section of the estuary.

A number of regression analyses were completed to explore the data set for relationships between hypoxia in the mesohaline estuary and features causing hypoxia. Most did not yield significant results. However, in most years hypoxic conditions can be reasonably predicted as a function of average chlorophyll-a concentrations in the mesohaline estuary during the winter-spring period. In effect, summer season hypoxia can be predicted based on winter-spring season chlorophyll-a conditions in the water column (Figure 12). The basis for this model includes the following. In many areas of the bay there is an algal bloom (dominated by diatoms; the "spring bloom") which generally follows the late winter - early spring freshet. It appears that most of the algal material generated by this bloom is not grazed by herbivoures while still in the water column; rather, these algal cells sink to deep waters and sediments and when bottom water temperatures increase in late spring and summer is consumed by benthic hetertrophs, the activity of which creates hypoxic conditions. However, algal blooms which occur in late spring and summer seem to have less of an impact on deep water oxygen conditions based on examination of data from Sta MLE 2.2. It is probable that algal biomass generated from these blooms is consumed while still in surface waters rather than being first deposited to deep waters prone to oxygen depletion (Smith and Kemp 1995).

SEDIMENT-WATER FLUXES IN THE POTOMAC RIVER ESTUARY

Characteristics of sediment-water fluxes

Average monthly sediment-water exchanges of ammonium and phosphorus measured at five locations in the Potomac River estuary are summarized as a series of bar graphs in Figures 13a and 13b. Sediment-water exchange measurements are not available from sites in the Potomac River estuary during cooler periods of the year (November - April) but measurements made in other sections of Chesapeake Bay and tributary rivers all indicate that exchanges are low to very low during the cooler months (Boynton et al 1980). It is reasonable to assume that this is also the case in the Potomac and that the data displayed in Figure 13 encompass the period of the year when these processes have water quality significance.

Ammonium fluxes tended to be highest during the warmest month (August) at most stations. However, seasonal patterns are not very clear at the three most upriver stations because the sampling frequency was low (May and August at Anacostia River stations; May, July, August and October at Hedge Neck and Gunston Cove stations). The pattern observed at Ragged Point is based on monthly sampling between May and October for multiple years and probably represents the pattern which would emerge for other stations. Perhaps the most striking feature of the ammonium flux data is the decrease in the magnitude of fluxes from the tidal fresh region to the lower mesohaline region. We suspect that the reason for this pattern is that sediments in the upper estuary are exposed to high deposition rates of phytoplanktonic detritus which is labile and subject to rapid decomposition and release of ammonium under both aerobic and anaerobic sediment conditions. The station in the mesohaline region is exposed to generally lower organic matter deposition rates and is deeper and therefore more organic matter can be remineralized prior to reaching the sediment surface. Overall, rates of sediment ammonium release were high to very high compared to other sites in Chesapeake Bay and tributary rivers (Boynton et al 1996).

Phosphorus fluxes exhibited a very different pattern. Fluxes were uniformly low in the tidal fresh and low salinity portions of the estuary ($< 15\ \mu mol\ m^{-2}\ hr^{-1}$) and very high ($> 30 \mu mol\ m^{-2}\ hr^{-1}$) in the mesohaline region. The probable mechanism responsible for this pattern involves the adsorbtion of dissolved sediment phosphorus to ferric oxides under aerobic conditions and the release of adsorbed phosphorus under anaerobic conditions (Kemp and Boynton 1992). It appears that there is ample phosphorus in sediments throughout the estuary to support high sediment exchanges (Boynton et al 1995) but the generally oxidized surface sediments of the shallower upper estuary prevent phosphorus release while the hypoxic conditions of the deeper mesohaline portion of the estuary promotes these releases. An example of the sensitivity of sediment phosphorus fluxes to dissolved oxygen conditions is shown in Figure 14; it appears that if bottom water dissolved oxygen concentrations fall below about 3 mg/l fluxes increase and if dissolved oxygen conditions decrease below 1 mg/l very large sediment phosphorus releases can occur.

It is also important to note that low dissolved oxygen levels is not the only mechanism that can solubilize phosphorus bound to sediment particles. In the relatively poorly buffered upper estuary, increases in water column pH (> 9.0) which can be caused by intense algal blooms (depleting inorganic carbon from the water column) will also lead to sediment phosphorus releases. This mechanism apparently played a role in sustaining a large algal bloom in the upper estuary during the 1980's (Seitzinger pers comm). Finally, it was noted earlier that the relative abundance of dissolved nitrogen to dissolved phosphorus in the water column in the mesohaline estuary during the warmer seasons of the year were much lower than the nitrogen to phosphorus ratio of nutrient inputs at the fall line. One reason for this large shift in the water column N:P ratio is that sediment releases of phosphorus in this zone of the estuary are particularly large and would have the effect of lowering the ratio in the mesohaline region and ultimately contribute to nitrogen limitation of phytoplankton growth.

Status of Potomac fluxes relative to other portions of the Bay

A standardized protocol has been developed for scaling sediment-water flux data in order to compare fluxes from different regions of the Bay and tributary rivers (Figure 15). Several versions of this approach have evolved and the version described below has been adopted by the Chesapeake Bay Monitoring Program (Alden and Perry, 1997). The status bar for each sediment-water flux variable comprises a benchmark (with a gradient scale) and a pointer which indicates the current status or condition of sediment-water flux along the benchmark scale at a particular sampling site.

The complete sediment-water exchange data set (collected at eight stations in the Bay and tributary rivers from 1985 through 1996) was used to create a status bar for each parameter (*e.g.* sediment oxygen consumption [SOC]). Using all sediment-water exchange data assured that the widest observable range of variability due to factors such as river flow and nutrient loading rates were included. The 5th and the 95th percentile values were calculated for each exchange variable and were used to indicate the end points of the gradient scale. An additional two centiles, the 35th and 65th centiles, were used to scale the final benchmark such that it was delineated into three categories: poor, fair and good. A linear quantitative scale with "good" and "poor" end points was thus developed. The annual median for each sediment-water exchange variable at each station was calculated and placed on the status bar as a vertical arrow. Data from the Ragged Point station in the mesohaline Potomac River was used in developing the status bars for each sediment-water exchange variable; other stations in the Potomac were not used in developing the status bars because observations at these sites were too limited. Again, at the Ragged Point station, the current status (vertical arrow) of each sediment-water exchange variable was calculated as the average value for the years 1994, 1995 and 1996. This averaging of the last three years of data has the effect of eliminating the influence of extreme climatic conditions (*i.e.* very wet or very dry years) since such extremes do not usually occur for several years in succession. Average values were calculated for other stations in the Potomac using whatever sediment-water exchange data that were available (usually a single year).

i Sediment Oxygen Consumption (SOC)

The current status of sediment oxygen consumption (SOC) fluxes at the five Potomac River estuary stations is indicated in Figure 15a. It seems appropriate to judge higher values of SOC as good in the context of this evaluation for several reasons despite the fact that high SOC rates indicate that sediments are using dissolved oxygen. The main reason for adopting this approach is that SOC rates are responsive to DO concentrations in the water. When dissolved oxygen concentrations in the water are high, SOC rates can potentially be high. Conversely, when dissolved oxygen concentrations in the water are low, SOC rates will always be low. Since restoration of increased dissolved oxygen in bottom waters is a goal of the management program we have adopted the position of treating higher SOC rates as indicative of healthy sediments in aerobic environments. Among the Potomac River estuary stations, three were considered to have SOC rates in the good range, one was in the fair range and one was in the poor range. The pattern of SOC fluxes in the provides substantiation that the benchmark is appropriate. SOC fluxes progress from good up-river to poor down-river. This pattern largely results because the water column is well mixed

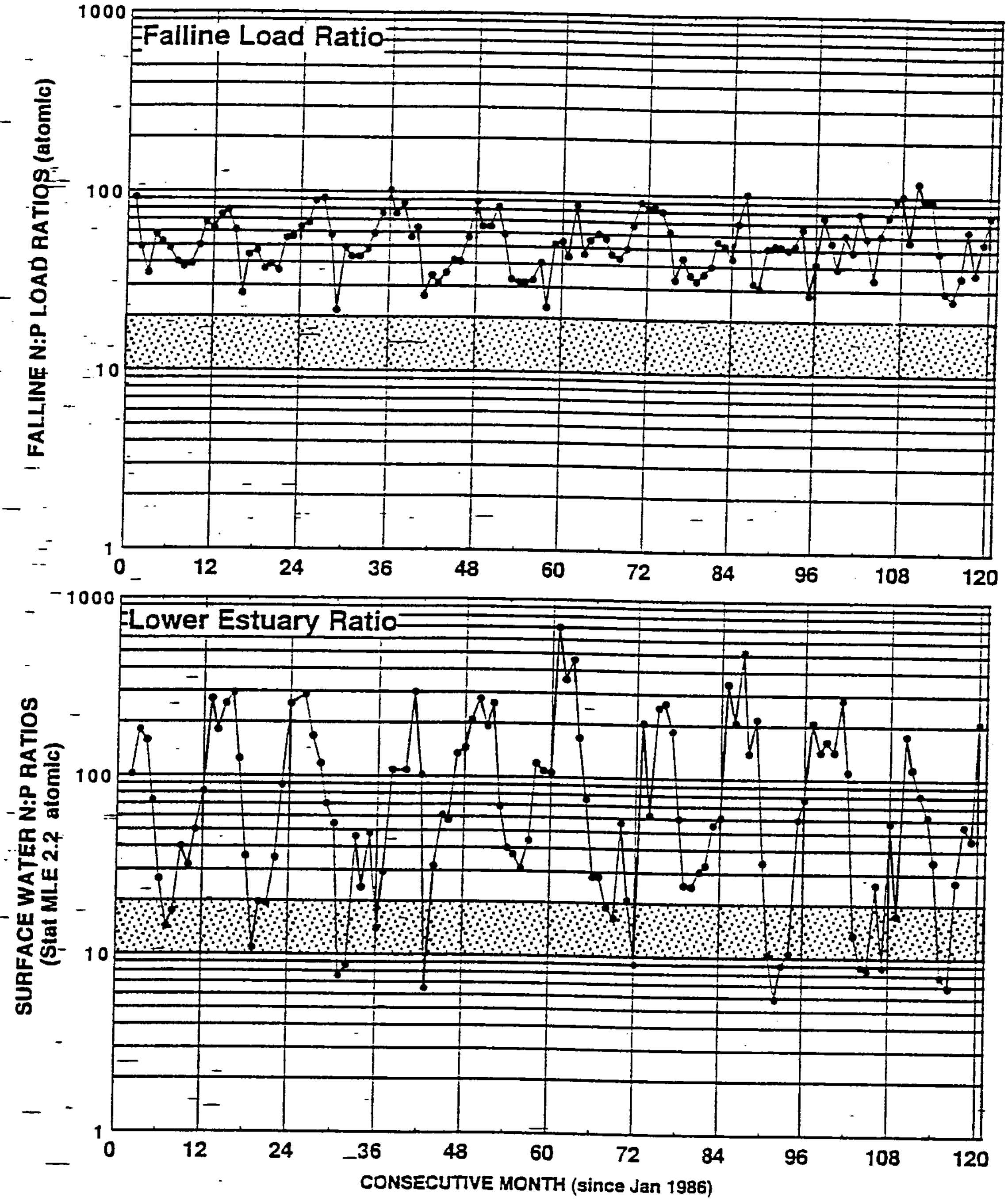


Figure 6. Average monthly ratios of total nitrogen to total phosphorus (TN : TP) of the fall line load and of surface waters in the mesohaline region (Sta. MLE 2.2) of the Potomac River estuary. Data were collected between January, 1986 and December, 1995 by the Maryland Chesapeake Bay Water Quality Monitoring Program. The shaded area on each figure indicates the approximate range of TN : TP ratios where potential N or P limitation is unlikely; values below 10 suggest possible N limitation and values above 20 suggest possible P limitation of phytoplankton communities.

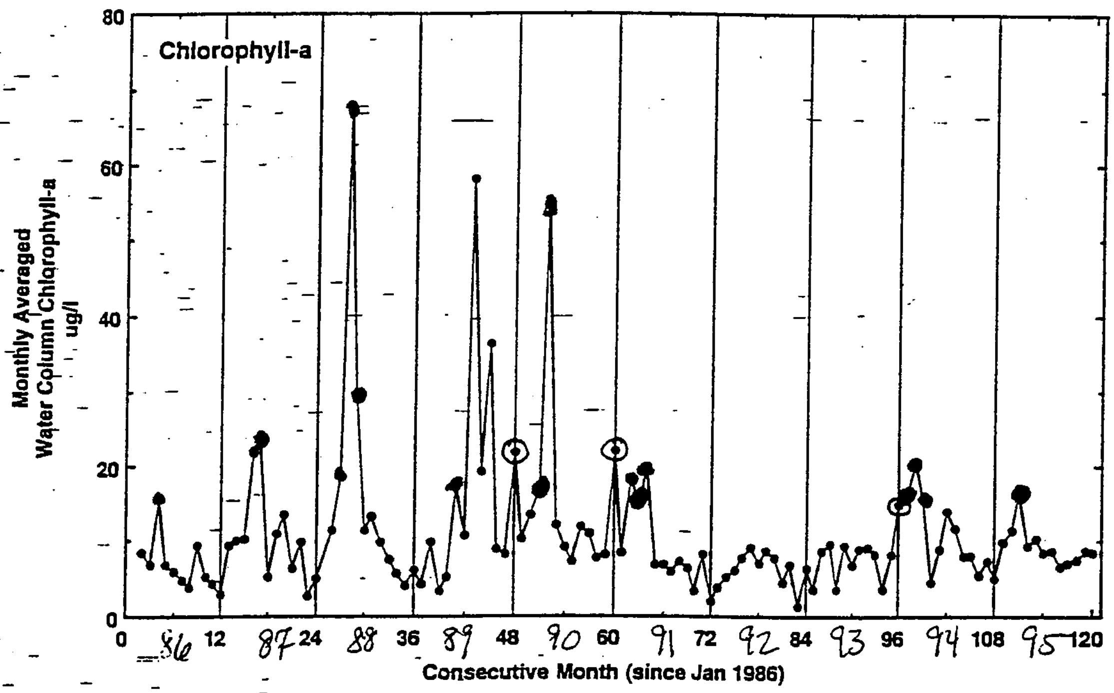


Figure 7. A plot of water column averaged chlorophyll-a concentrations at Sta MLE 2.2 located in the mesohaline region of the Potomac River estuary. Data were collected by the Maryland Chesapeake Bay Water Quality Monitoring Ptogram from January, 1986 through December, 1995.

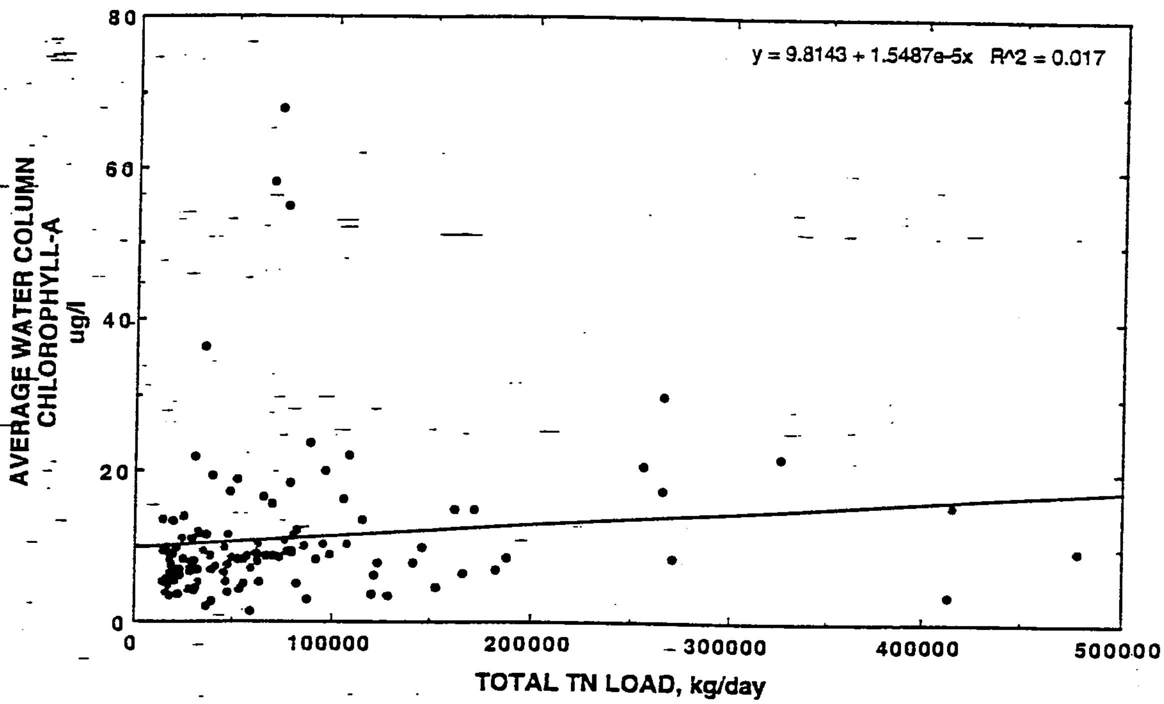


Figure 8. A scatter plot of water column averaged chlorophyll-a at a mesohaline station (MLE 2.2) versus total nitrogen (TN) loading rate measured at the fall line of the Potomac River. In this plot paired load and chlorophyll-a data were collected in the same month; there are no time lags between load and chlorophyll-a in this plot. Data are from the Maryland Chesapeake Bay Water Quality Monitoring Program.

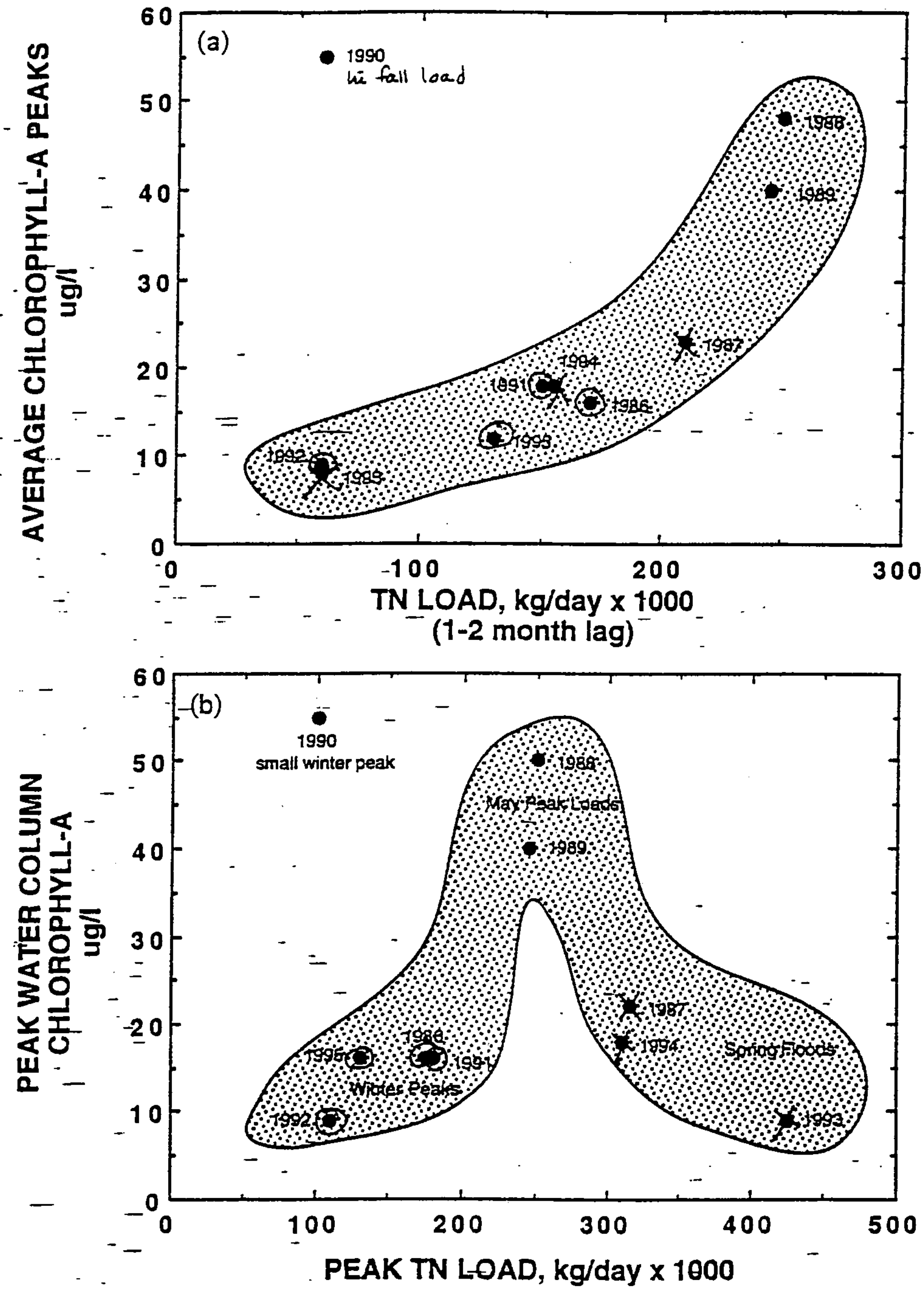


Figure 9. Scatter plots of water column averaged chlorophyll-a at a mesohaline station (MLE 2.2) versus several different functions of total nitrogen (TN) loading rate measured at the fall line of the Potomac River estuary. In (a) average peak chlorophyll-a concentrations were regressed against TN load measured 1-2 months prior to the chlorophyll-a peak. In (b) peak water column chlorophyll-a concentrations were regressed against peak TN load occurring prior to the chlorophyll-a peak. Data are from the Maryland Chesapeake Bay Water Quality Monitoring Program.

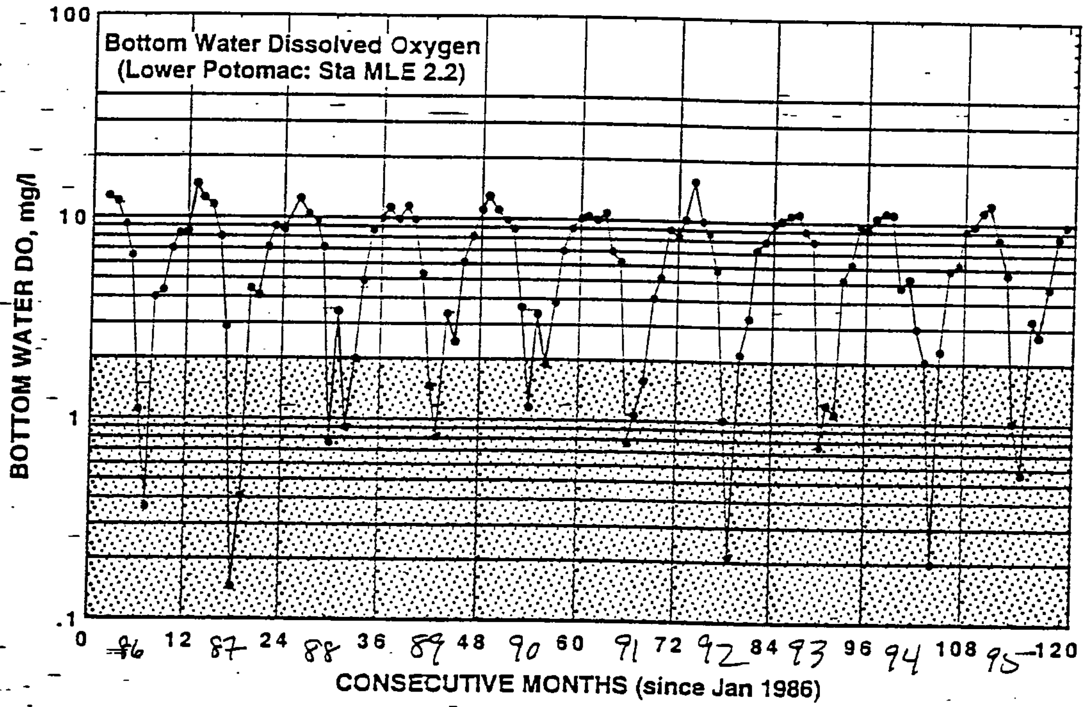


Figure 10. A summary of average monthly bottom water dissolved oxygen concentrations at a station in the mesohaline region of the Potomac River estuary (Sta MLE 2.2). The stippled area of the figure indicates observations with dissolved oxygen concentrations less than 2 mg/l. Data are from the Maryland Chesapeake Bay Water Quality Monitoring Program.

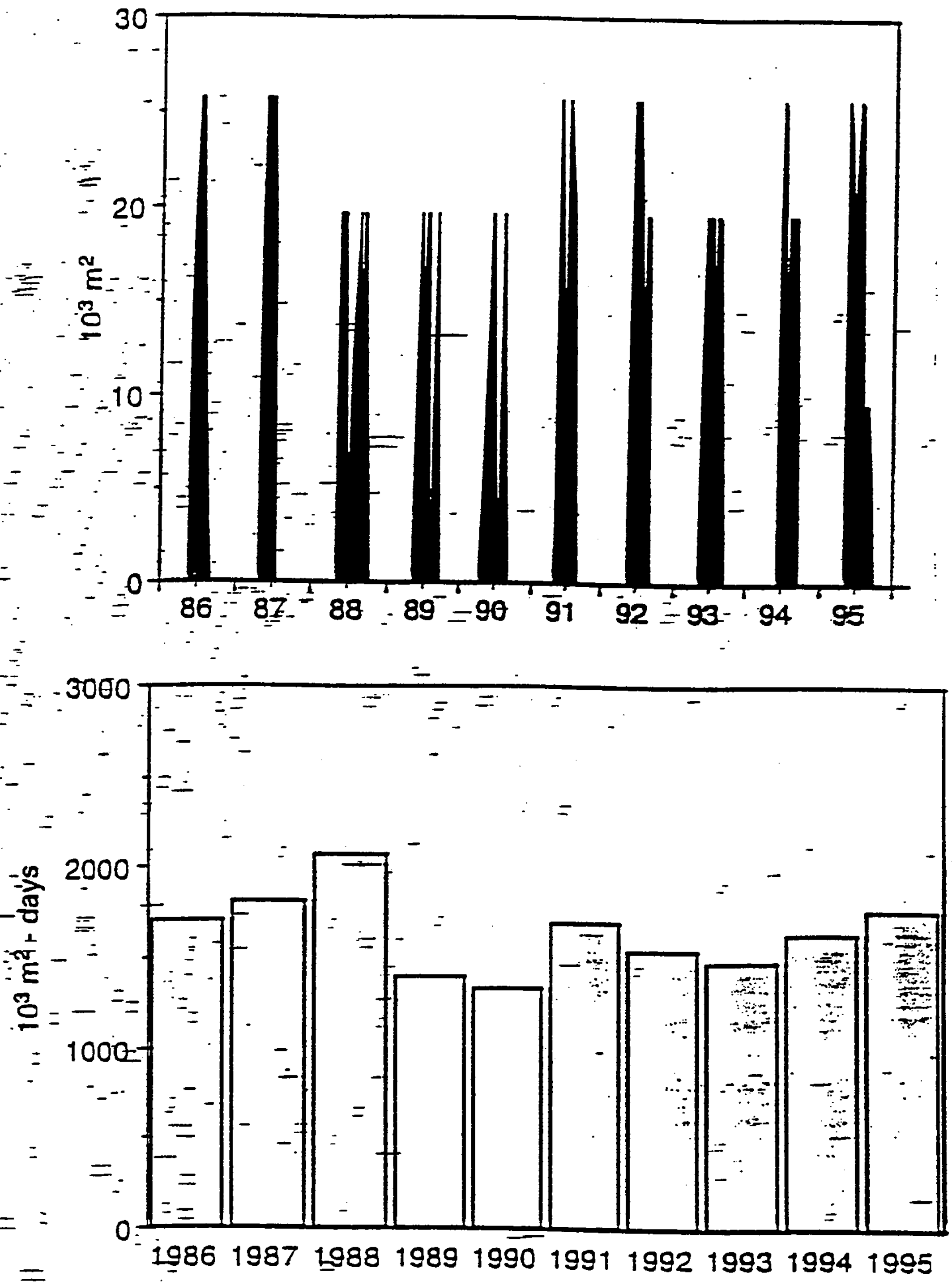


Figure 11. Annual estimate of the hypoxic (DO < 2.0 mg/l) cross-sectional area at a mesohaline site (river cross-section at Sta MLE 2.2) in Potomac River estuary and hypoxic cross-sectional area days at the same location. Data are from the Maryland Chesapeake Bay Water Quality Monitoring Program.

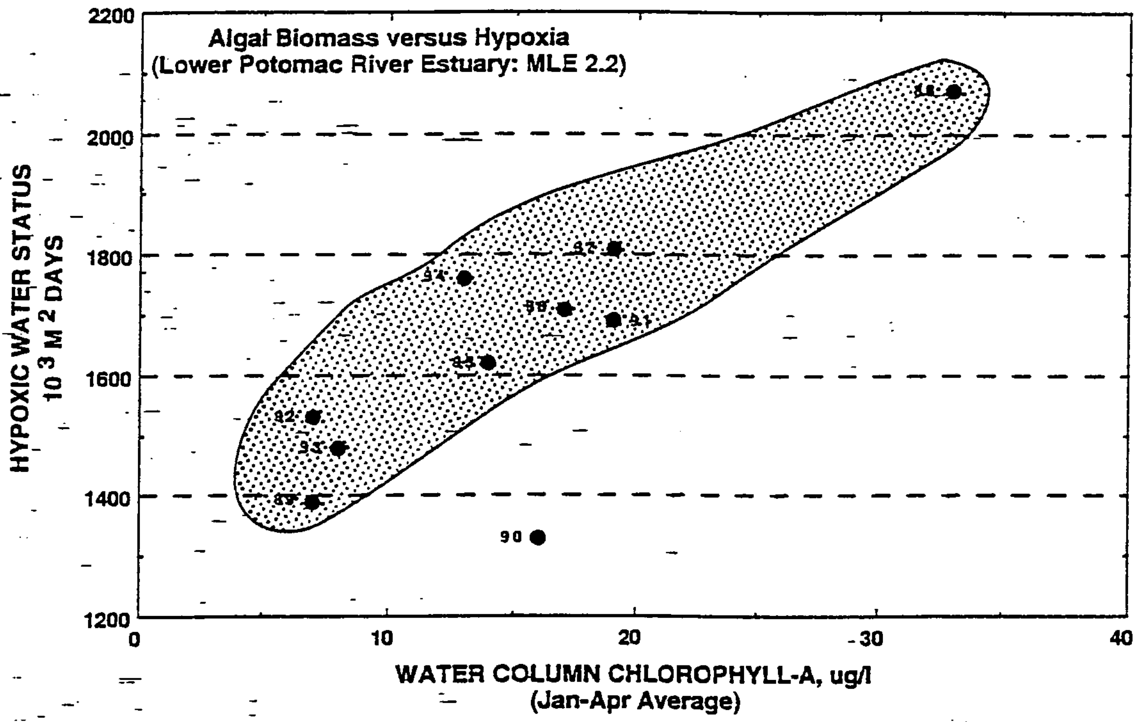


Figure 12. A scatter plot of annual hypoxic (< 2 mg/l) water status versus seasonal (average of January - April) water column chlorophyll-a concentrations at a station in the mesohaline region of the Potomac River estuary. Data are from the Maryland Chesapeake Bay Water Quality Monitoring Program. Hypoxic water status represents the product of the cross-sectional area of the Potomac River at station MLE 2.2 which had dissolved oxygen concentrations below 2 mg/l multiplied by the number of days this condition persisted.

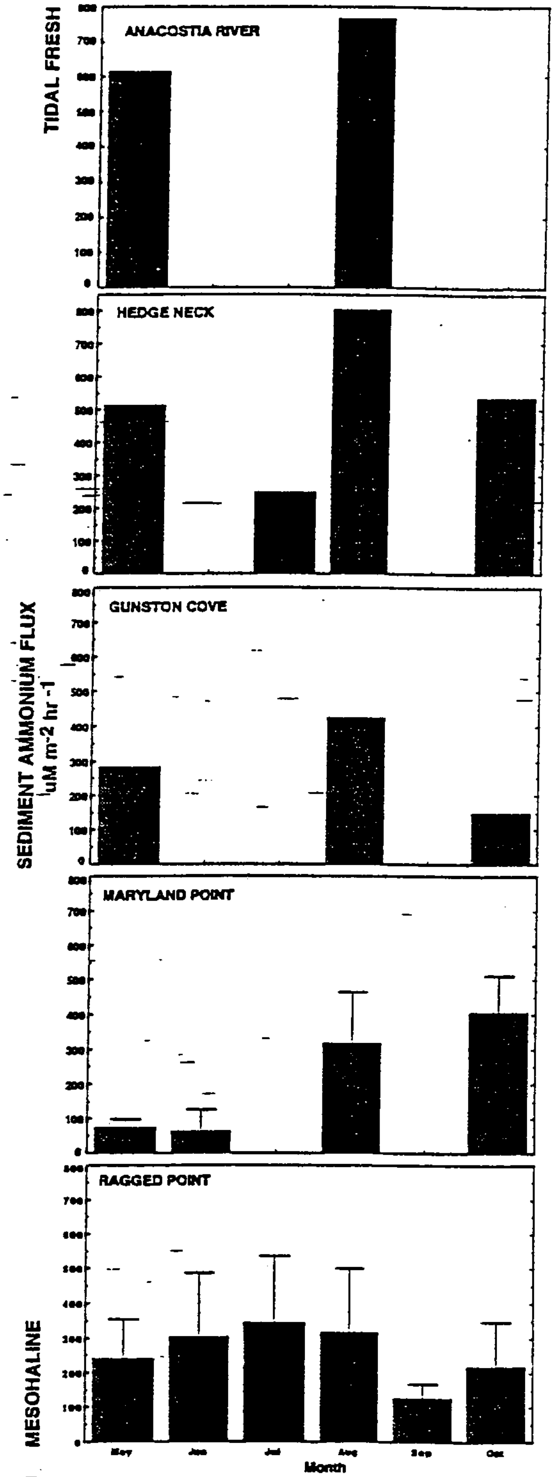


Figure 13 a. Monthly average sediment-water exchanges of ammonium at stations located along the Potomac River estuary from tidal fresh to mesohaline regions. Average monthly fluxes at Ragged Point were based on measurements collected between 1985 and 1996; average fluxes at Maryland Point were based measurements collected between 1985 and 1988 and 1992; fluxes at remaining stations were based on a single year of observation. Data are from the Maryland Chesapeake Bay Water Quality Monitoring Program.

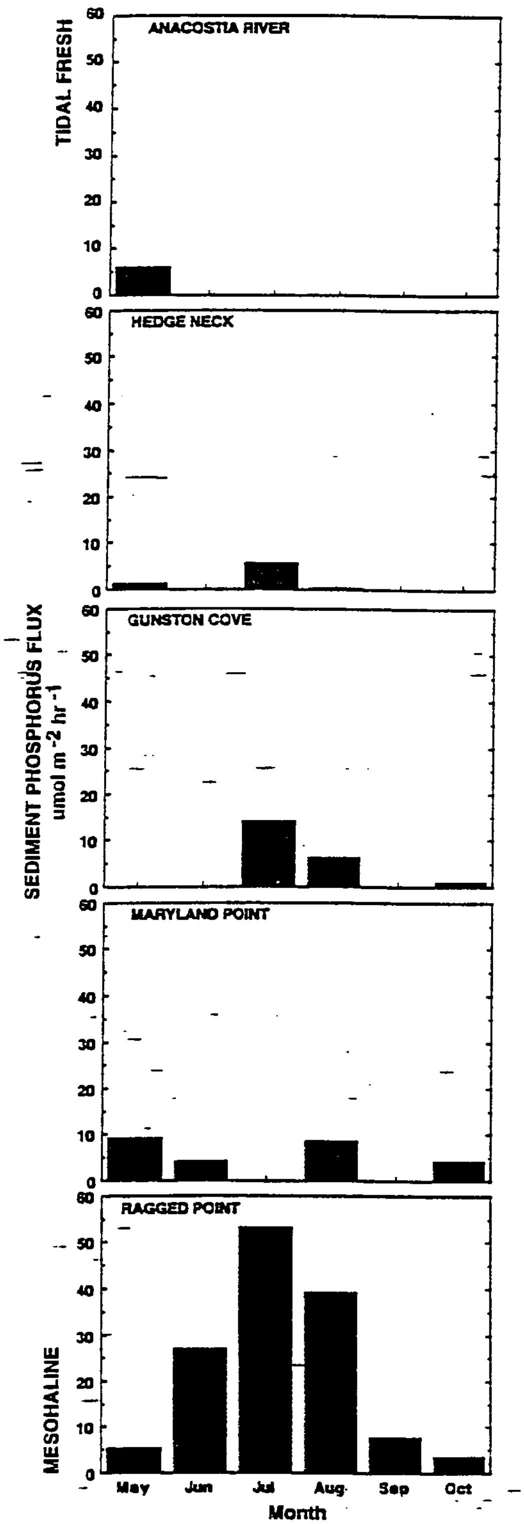


Figure 13 b. Monthly average sediment-water exchanges of phosphorus at stations located along the Potomac River estuary from tidal fresh to mesohaline regions. Average monthly fluxes at Ragged Point were based on measurements collected between 1985 and 1996; average fluxes at Maryland Point were based measurements collected between 1985 and 1988 and 1992; fluxes at remaining stations were based on a single year of observation. Data are from the Maryland Chesapeake Bay Water Quality Monitoring Program.

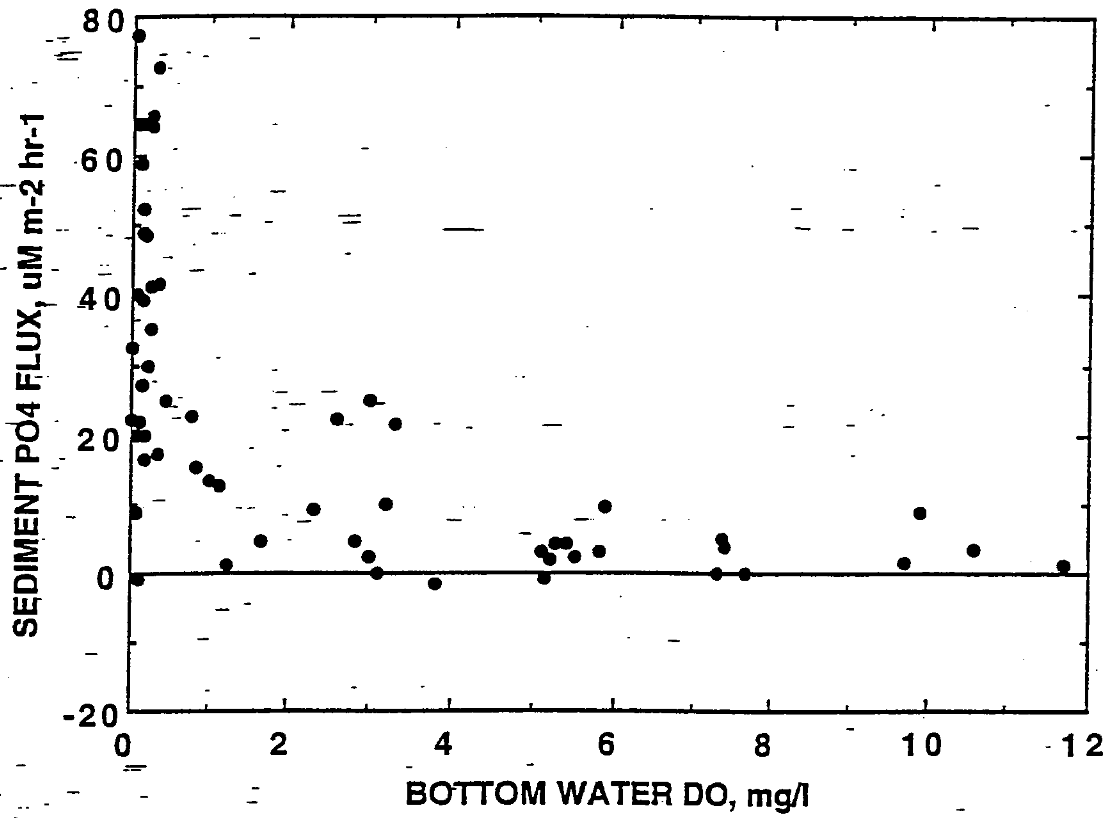
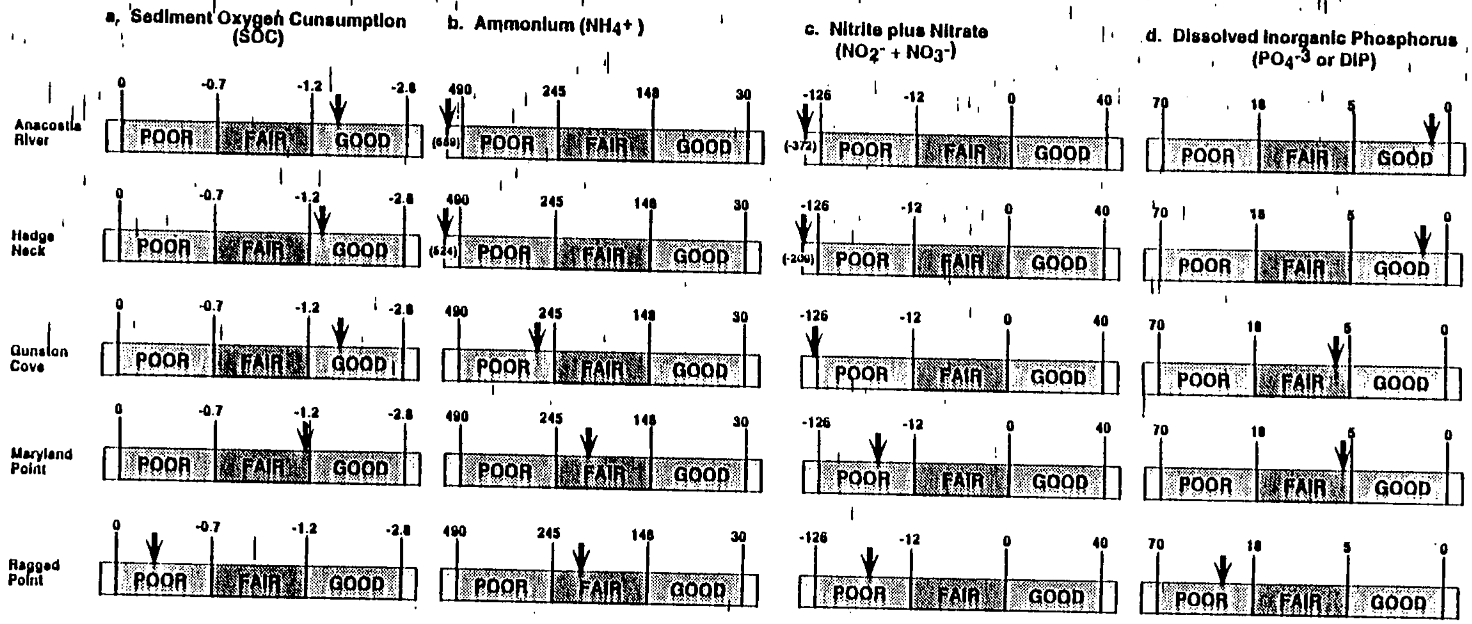


Figure 14. A scatter plot of sediment-water phosphorus fluxes versus bottom water dissolved oxygen concentrations from a station in the mesohaline region of the Potomac River estuary (Sta. MLE 2.2). Positive and negative values on the Y-axis represent fluxes from sediments to water and water to sediments, respectively. All of the larger phosphorus fluxes (> 10 umol m^{-2} hr^{-1}) occurred under hypoxic conditions.



Sediment Oxygen Consumption (SOC):
High SOC rates are judged to be good because SOC rates are responsive to DO concentrations in the water. When dissolved oxygen concentrations are high SOC rates can be high; when dissolved oxygen concentrations are low, SOC rates will be low. Higher SOC rates are are judge to be indicative of healthy sediments in aerobic environments.

Ammonium (NH_4^+):
High ammonium fluxes are judged to be poor because of the well established linkage between ammonium availability and excessive phytoplankton biomass accumulation.

Nitrite plus Nitrate ($NO_2^- + NO_3^-$):
High nitrite plus nitrate fluxes (positive values) are judged to be good because of the well established linkage between nitrite plus nitrate evolution from sediments via nitrification and oxidized sediment conditions.

Dissolved Inorganic Phosphorus (PO_4^{-3} or DIP):
High phosphorus values are judged to be poor because of the well established linkage between phosphorus availability and excessive phytoplankton biomass accumulation.

Figure 15. A summary of the status of sediment-water oxygen and nutrient exchanges for several locations in the Potomac River estuary. Stations are arranged from up-river (top of diagram) to down-river (bottom of diagram). Details concerning the development and scaling of status bars is provided in the text. In brief, for each flux variable (e.g. SOC) all data collected at 8 regularly sampled (1985 - 1996) stations in Chesapeake Bay (including one station in the lower Potomac River; Ragged Point) were combined and the 5th, 95th, 35th and 65th percentiles determined. In each status bar these are indicated by vertical lines separating the categories of poor, fair and good and the numerical value of the flux separating each catagory is indicated. Downward pointing arrows on each bar represent the current status of a sediment-water flux variable. The status at Ragged Point is based on the mean of fluxes observed during the period 1994-1996; at Maryland Point status is based on the mean of fluxes measured from 1985-1988 and 1992; status at the remaining stations is based on measurements made during a single year. In a few cases fluxes measured in the Potomac exceeded the range of previously measured values and in these cases the actual flux value is indicated in parentheses at one end of the status bar. At the base of the figure the general criteria for judging flux status as good, fair or poor is provided and further discussed in the text. The development of these status bars exactly follows the procedure adopted by the Chesapeake Bay Program; determination of current status at Ragged Point also exactly follows that procedure but status at the remaining stations was determined using data from any year in which measurements were available.

Table 1a. Summary of seasonal Kendall test statistics (not flow adjusted) based on data collected at a station in the mesohaline portion of the Potomac River (Ragged Point) for six sediment-water flux variables. Data were collected from 1985 - 1996 and 4 to 6 measurements were made during each year. The term "not flow adjusted" indicates that flux data were not modified prior to statistical testing in any way because of freshwater flow conditions during the period of measurement. Significance: + p = 0.10; * p = 0.05; ** p = 0.01; *** = 0.001

a. Sediment Oxygen Consumption (SOC; [g O_2 m^{-2} day^{-1} yr^{-1}])

	April	May	June	July	August	September	October	November	Annual
Ragged Point (RGPT)									
Sign	1	6	-9	-13	-22	2	-3	1	-37
p value		0.55	0.53		0.08+		0.77		0.11
n	3	8	11	8	11	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-42	-44	-35
p value	0.04*	0.03*	0.02*

b. Ammonium (NH_4; [μM N m^{-2} hr^{-1} yr^{-1}])

	April	May	June	July	August	September	October	November	Annual
Ragged Point (RGPT)									
Sign	-1	-4	-2	-8	-12	5	-1	-3	-26
p value		0.72	0.95	0.40	0.45	0.47	1.00		0.32
n	3	8	12	8	12	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-17	-22	-20
p value	0.48	0.34	0.25

c. Nitrite (NO_2; [μM N m^{-2} hr^{-1} yr^{-1}])

	April	May	June	July	August	September	October	November	Annual
Ragged Point (RGPT)									
Sign	0	-5	-6	10	-1	1	5	0	4
p value		0.47	0.56	0.28		1.00	0.47		0.86
n	1	6	8	8	8	6	6	1	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	4	3	9
p-value	0.84	0.87	0.48

Table 1a (continued). Summary of seasonal Kendall test statistics (not flow adjusted) based on data collected at a station in the mesohaline portion of the Potomac River (Ragged Point) for six sediment-water flux variables. Data were collected from 1985 - 1996 and 4 to 6 measurements were made during each year. The term "not flow adjusted" indicates that flux data were not modified prior to statistical testing in any way because of freshwater flow conditions during the period of measurement. Significance: + p = 0.10; * p = 0.05; ** p = 0.01; *** = 0.001.

d. Nitrite plus nitrate ($NO_2 + NO_3$; [µM N m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)+	April	May	June	July	August	September	October	November	Annual
Sign	-3	2	-28	-10	-21	8	1	-1	-52
p value		0.90	0.06+	0.28	0.17		1.00		0.04*
n	3	8	12	8	12	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-51	-59	-31
p value	0.03*	0.01**	0.07+

e. Dissolved Phosphorus (PO_4^-; [µM P m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	3	-6	4	-8	-6	7	11	-1	4
p value		0.55	0.84	0.40	0.73	0.27	0.14		0.91
n	3	8	12	8	12	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-3	-10	-14
p value	0.93	0.68	0.44

f. Silicate ($Si(OH)_4$; [µM Si m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	-3	-2	7	-5	31	2	-3	-1	24
p value		0.90	0.64	0.47	0.005**	0.75	0.77		0.28
n	3	8	11	6	10	4	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	35	33	26
p-value	0.06+	0.07+	0.04*

Table 1b. Summary of seasonal Kendall test statistics (flow adjusted) based on data collected at a station in the mesohaline portion of the Potomac River (Raggec Point) for six sediment-water flux variables. Data were collected from 1985 - 1996 and 4 to 6 measurements were made during each year. The term "flow adjusted' indicates that flux data from each month and for groups of months were regressed against river flow averaged for the January through April period of the same year Residuals from this analysis were then used in the seasonal Kendall test. Flow correction represents an attempt to exclude natural effects (river flow variability) anc test for trends related to management actions (nutrient load reduction). Significance: + p = 0.10; * p = 0.05; ** p = 0.01; *** = 0.001

a. Sediment Oxygen Consumption (SOC; [g O_2 m^{-2} day^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	-1	8	-9	-12	-16	7	-3	-1	-27
p value		0.40	0.53	0.18	0.24	0.27	0.77		0.26
n	3	8	11	8	11	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-30	-37	-28
p value	-0.16	0.07+	0.07+

b. Ammonium (NH_4; [µM N m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	-1	-8	-12	-8	-12	9	-1	-1	-34
p value		0.40	0.45	0.40	0.45	0.14	1.00		0.19
n	3	8	12	8	12	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-23	-32	-20
p value	0.33	0.16	0.25

c. Nitrite (NO_2; [µM N m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	0	-3	2	4	-8	-3	-9	0	-17
p value		0.72	0.90	0.72	0.40	0.72	0.14		0.34
n	1	6	8	8	8	6	6	1	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-5	-2	-4
p-value	0.79	0.94	0.79

Table 1b (continued). Summary of seasonal Kendall test statistics (flow adjusted) based on data collected at a station in the mesohaline portion of the Potomac River (Ragged Point) for six sediment-water flux variables. Data were collected from 1985 - 1996 and 4 to 6 measurements were made during each year. The term "flow adjusted" indicates that flux data from each month and for groups of months were regressed against river flow averaged for the January through April period of the same year. Residuals from this analysis were then used in the seasonal Kendall test. Flow correction represents an attempt to exclude natural effects (river flow variability) and test for trends related to management actions (nutrient load reduction). Significance: + p = 0.10; * p = 0.05; ** p = 0.01; *** = 0.001.

d. Nitrite plus nitrate ($NO_2 + NO_3$; [µM N m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	1	6	-24	-6	-26	9	-3	-1	-46
p value		0.55	0.11	0.55	0.09+	0.14	0.77		0.07+
n	3	8	12	8	12	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	-47	-56	-32
p value	0.04*	0.01**	0.06+

e. Dissolved Phosphorus (PO_4^-; [µM P m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	1	-8	-4	-2	8	9	9	-1	2
p value		0.40	0.84	0.90	0.95	0.14	0.24		0.97
n	3	8	12	8	12	6	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	1	-8	-4
p value	1.00	0.75	0.86

f. Silicate ($Si(OH)_4$; [µM Si m^{-2} hr^{-1} yr^{-1}])

Ragged Point (RGPT)	April	May	June	July	August	September	October	November	Annual
Sign	-1	-4	57	-5	27	2	-3	-1	20
p value		0.72	0.76	0.47	0.02*	0.75	0.77		0.37
n	3	8	11	6	10	4	7	3	

	Jun-Sep	Jun-Aug	Jul-Aug
Sign	29	27	22
p-value	0.12	0.15	0.09+

Tidal Potomac Integrative Analysis Project, A Series of Reports on the Water Quality and Living Resources Responses to Management Actions to Reduce Nutrients in the Potomac River Estuary, Final Draft.

Buchanan, C. [ed.] 1999.
Prepared for the Chesapeake Bay Program.
ICPRB Report 99-4, 268 pp.

APPENDIX D

LONG-TERM TRENDS IN SUMMER PHYTOPLANKTON CHLOROPHYLL A IN THE TIDAL FRESHWATER POTOMAC RIVER, USA: RELATIONSHIP TO CLIMATIC AND MANAGEMENT FACTORS

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Long-term Trends in Summer Phytoplankton Chlorophyll a in the Tidal Freshwater Potomac River, USA: Relationship to Climatic and Management Factors

R. Christian Jones

Keywords: phytoplankton, climate, tidal freshwater, interannual trends

Introduction

The management of freshwater systems has been prefaced on the belief that nutrient reduction from point sources is an effective way to control eutrophication. This paradigm is based on results from a number of studies relating dramatic reductions in algal standing crops which have occurred when loading of nutrients, notably phosphorus, has been substantially reduced. However, in any given situation the response to point source nutrient control may be influenced by a number of other factors such as flushing, turbidity, or temperature which may limit algal growth for at least part of the year. Furthermore, the system may also be influenced by non-controlled sources such as runoff and sediment release. Since these factors may vary in importance from year-to-year, interannual trends and relationships to nutrient loading changes may be difficult to decipher. These factors may be especially important in tidal freshwater.

The Potomac River is the second largest tributary of Chesapeake Bay and one of the largest rivers on the Atlantic drainage of North America. The tidal freshwater section extends some 60 km from the head of tide at the western edge of the Washington, D.C. to near Quantico, Virginia. River inflow varies seasonally resulting in much longer residence times in summer and fall than in winter and spring. Over 1 million cubic meters per day of treated sewage is discharged into the tidal freshwater Potomac which at low flow may represent roughly half of the freshwater input to the river. Further description of the tidal Potomac may be found in JONES (1991) and LIPPSON et al. (1979). The expenditure of large amounts of public funds to upgrade sewage treatment in the region has resulted in strong interest in the success of these efforts in controlling excess algal growths.

Since 1984 algal populations have been monitored in the Gunston Cove region of the tidal freshwater Potomac River as part of a broad ecological study (JONES et al. 1992). Gunston Cove is a moderately large (4 km long x 1 km wide), shallow ($z \leq 2$ m) embayment of the river located about 35 km downstream from the head of tide. The cove receives 204×10^3 $m^3 \cdot day^{-1}$ of treated sewage daily from the Noman Cole Pollution Control Plant through one of its tributaries. The river mainstem is about 2 km wide in this area with $z_{max}=15$ m. The tidal Potomac is typified by semidiurnal tides with an amplitude of about 0.5 m. Phytoplankton populations show a strong seasonal trend with highest densities during the summer months from mid June through mid September (JONES et al. 1992). High algal densities in summer have been identified as a major water quality concern in the tidal freshwater Potomac River.

The objective of this paper is to examine interannual trends in summer phytoplankton chlorophyll *a* and relate them to the success of management actions. To achieve this goal, variability attributable to significant relationships with climatic variables was removed by regression and the residuals from these regressions examined for trends that could be related to management efforts.

Methods

Samples were collected on a typically biweekly basis from March through December during the years 1984 to 1997. Stations were sited to characterize both the shallow embayment habitat (cove stations) and the deeper river channel area (river stations); up to six stations were located in each area. Only samples collected between June 15 and September 15 are utilized in this paper. Depth-integrated samples were formed by combining equal amounts of water collected with a submersible pump from three depths: 0.3 m, mid-depth, and 1.0 m above the bottom. Water was maintained at *in situ* temperature in the dark and returned to the lab for filtration within six hours. Aliquots were filtered through 0.45 μm membrane filters which were extracted in DMSO-acetone overnight followed by fluorometric assay (JONES 1998). Total phosphorus was determined by stannous chloride method following persulfate digestion, nitrate by cadmium reduction method, and ammonia and total kjeldahl nitrogen by automated phenate method.

Mean daily air temperature, mean wind speed and direction, and total daily sunshine data for Reagan National Airport located 23 km north of the study area were obtained from the U.S. National Climatic Data Center. Previous work established a strong relationship between minutes of sunshine at Reagan National Airport and photosynthetically active radiation in the study area (Jones 1998). Daily river and tributary flow data were furnished by the U.S. Geological Survey for the Potomac River at Little Falls (representing mainstem flow into the tidal Potomac) and Accotink Creek at Braddock Road (representing a portion of local tributary flow). Average values for the climatic parameters were computed for a range of time periods preceding and including the date of sampling (3-4 days, 7 days, 14 days, 28 days, and in some cases 42 days and 56 days). Correlations of log of daily chlorophyll *a* with the suite of weather and flow variables were calculated. For those variables that showed a significant correlation, regression was attempted with the chlorophyll variables. The residuals of the most significant regression at each station were plotted as a yearly time series to determine if any consistent trends were visible. Trend lines were established by LOWESS (locally weighted regression) with a tension of 0.5.

Results

Chlorophyll *a* values in Gunston Cove demonstrated a consistent increase from 1984 through 1987 (Figure 1). The LOWESS trend line rose from 70 μg/L in 1984 to about 150 μg/L in 1987 and 1988. Beginning in 1989 and continuing through 1997 a steady decline in summer chlorophyll a has occurred at cove stations resulting in the trend line reaching about 60 μg/L in 1997. Correlations with flow parameters revealed that cove summer chlorophyll was more strongly related to Potomac River discharge at Little Falls than with local inputs from Accotink Creek. Log of average Potomac flow for the 28 days preceding sampling was the most highly correlated flow variable ($r=-0.517$, $n=309$). Temperature demonstrated significant positive correlations with chlorophyll, the highest being with 28 ($r=0.386$) and 42 day ($r=0.409$) averages. Two wind parameters were significantly correlated with summer phytoplankton chlorophyll: wind direction ($r=-0.288$) and 4-day average wind velocity ($r=0.208$). No solar radiation parameter exhibited significant correlation.

Chlorophyll *a* values at the river stations were lower, but demonstrated a less consistent interannual pattern (Figure 2). The LOWESS line rose slightly from 1984-1987 and then dropped about the same amount from 1987 to 1990. This was followed by another small rise from 1990 until 1993 with little change thereafter. Average Potomac flow for the 28 days preceding sampling (not logged) was the most significantly correlated flow parameter ($r=-0.481$, $n=224$). Temperature with a 28 day averaging period was very strongly correlated with river summer chlorophyll a ($r=0.510$). Solar radiation was correlated to chlorophyll with a 28 day averaging period being most signficant ($r=0.402$). Wind parameters failed to demonstrate significant correlation to river chlorophyll.

The most significant regression between chlorophyll and climatic parameters in the cove utilized log Potomac flow (28-day average), air temperature (28-day average), wind direction (degrees from north), and wind velocity (4-day average). This regression was highly significant, accounting for 38.4% of the variance in summer chlorophyll *a* in the cove. Residuals from this regression were plotted against years to determine trends independent of significant climatic relationships (Figure 3). The residuals demonstrate the same basic interannual pattern as was found in the raw data (Figure 1). An increase was observed through 1987 followed by a consistent decline through 1997. The main effect of removing the climatic relationships appears to be a dampening of the late 1980's peak in chlorophyll.

At river stations the most significant regression involved Potomac flow (28-day average), air temperature (28-day average), and solar radiation (minutes of sunshine, 28-day average). This regression was highly significant, accounting for 45% of the variance in summer chlorophyll in the river. Residuals from this regression demonstrated a slightly different trend than those from the raw data. The early 1980's increase was not apparent. The late 1980's decline was accentuated. The gradual increase and stabilizing of river chlorophylls during the 1990's was still apparent.

Trends in nutrient concentrations were examined to determine if they could help explain the chlorophyll trends. Total phosphorus at cove stations demonstrated a pattern very similar to that observed in both the raw and residual cove chlorophyll values (Figure 4). The LOWESS trend line suggests increasing amounts of chlorophyll through 1986 followed by a slow gradual decline in phosphorus through 1997. The trend line in 1997 reached 0.1 mg/L whereas nearly double that amount was suggested for 1986. Total nitrogen followed a similar pattern, but with a much smaller relative decline. Total inorganic nitrogen remained constant at about 2 mg/L through 1990 and then decreased steadily falling below 1 mg/L in 1997.

At river stations there was virtually no change in total phosphorus over the study period with the trend line remaining near 0.1 mg/L. Total nitrogen showed a consistent drop from about 3 mg/L during the period with the decline accelerating in the late 1990's and dropping below 2 mg/L in 1997. Total inorganic nitrogen declined from values near 2 mg/L in 1984 to about 1.5

mg/L by 1990 and more rapidly to about 0.6 mg/L by 1997.

Discussion

Significant relationships were found between summer phytoplankton chlorophyll *a* and various climatic variables. At both sites freshwater inflow was negatively correlated, not surprising in importance of flushing and dilution in the tidal freshwater environment. Temperature was also a consistent correlate consonant with its effect on metabolism. The importance of solar radiation in the river and its lack of importance in Gunston Cove is probably related to the relative depths in each area. The shallow cove is less likely to be light limited than the deeper river, particulary since water transparency does not differ greatly between the two areas. Winds from the east and southeast were correlated with higher chlorophyll levels possibly by slowing tidal flushing of the cove.

The persistence of interannual trends in the cove after removal of significant climatic variables suggests a relationship of these trends to management factors. In the cove the correspondence between the chlorophyll a trend and that of total phosphorus suggests a direct link between the two. While nitrogen followed the same general pattern, the percent decline was small and nitrogen remained at non-limiting values with N:P ratio generally above 20 (by weight) and almost never below 10. Interestingly, the major decline in phosphorus loading to the cove occurred in 1983 when the Noman Cole plant began removing phosphorus in its discharge to a level of 0.18 mg/L. Water column phosphorus continued to increase for several more years apparently due to sediment release of phosphorus stored during years of higher loading (CERCO 1988). Thus, the chlorophyll trends observed in the cove are consistent with a response to management actions in the form of enhanced phosphorus removal.

The main trend in summer phytoplankton in the river was the decline in chlorophyll centered in 1990. This trend does not seem to be attributable to climatic factors since it remained when significant climatic variables were removed. No simple relationship to nutrient concentrations was found.

Acknowledgments

These studies have been funded by the Department of Public Works of Fairfax County, Virginia. Special thanks there goes to Elaine Schaeffer and Jimmie Jenkins. The assistance of numerous students and technicians is gratefully acknowledged. My colleague Don Kelso has been a continuing source of help and encouragement throughout this project. I thank Ann Powel for her careful reading of this manuscript.

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Figure Legends

Figure 1. Depth-integrated summer chlorophyll *a* concentrations for Gunston Cove stations from 1984 to 1997. Line is LOWESS smoothing function with a tension of 0.5.

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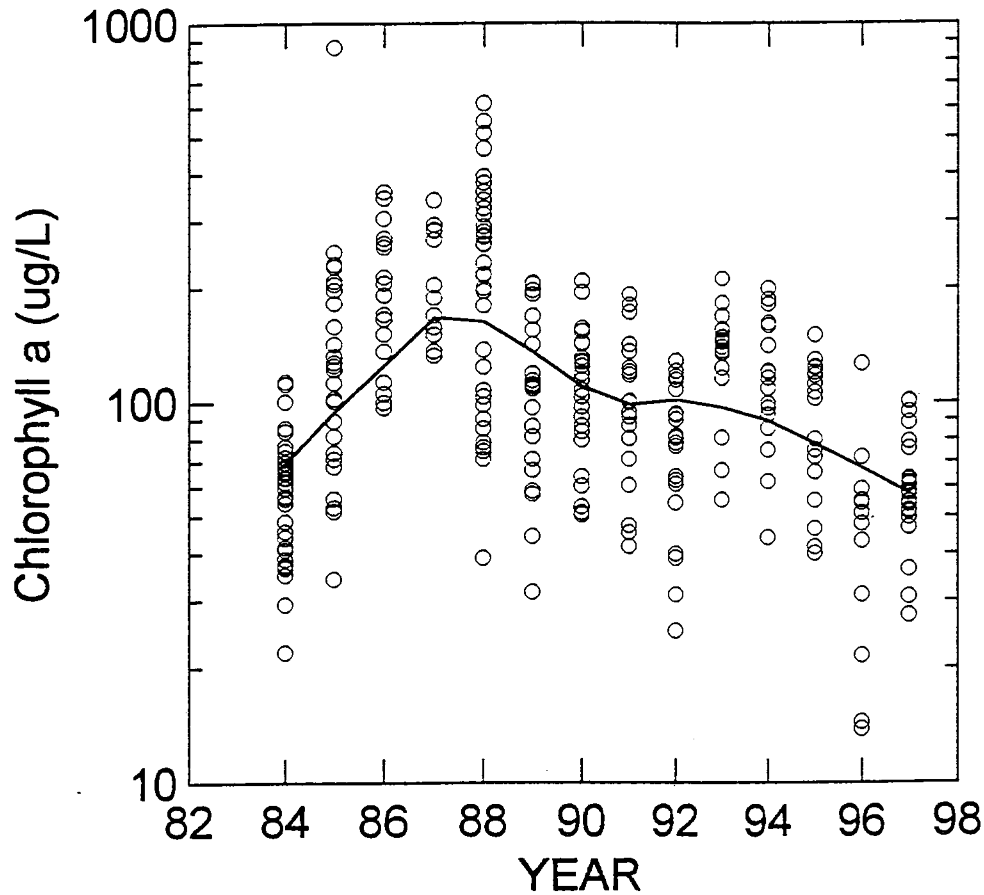
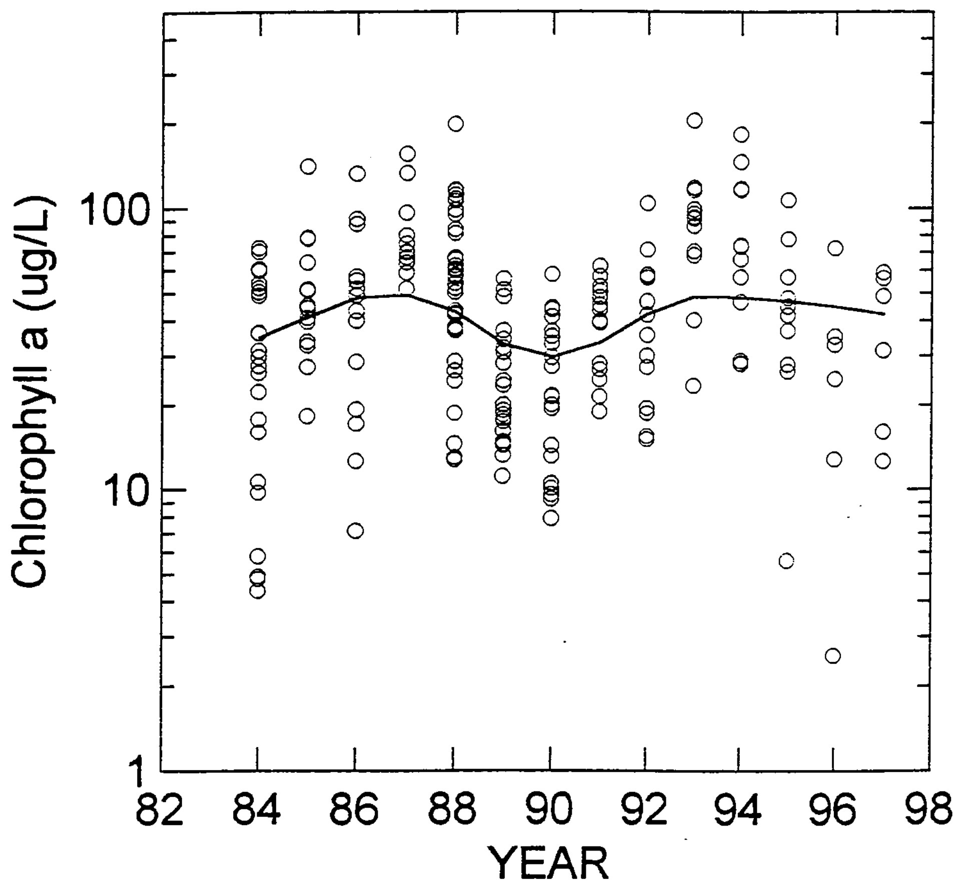
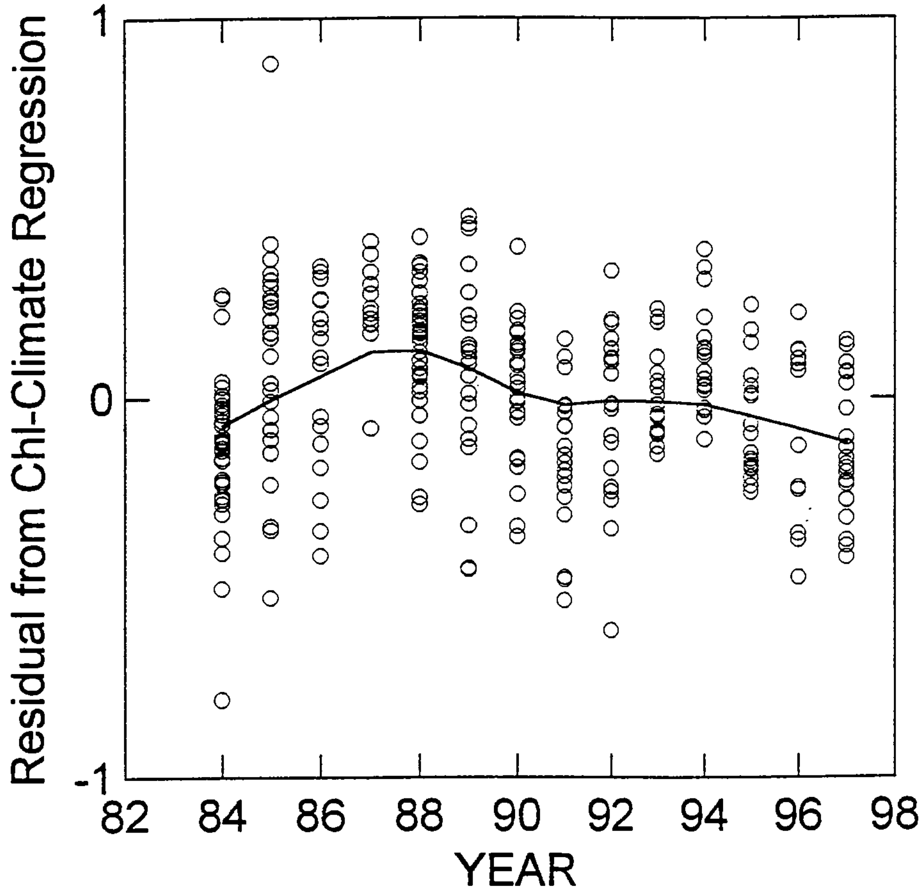
Cove Stations: June 15-Sept 15



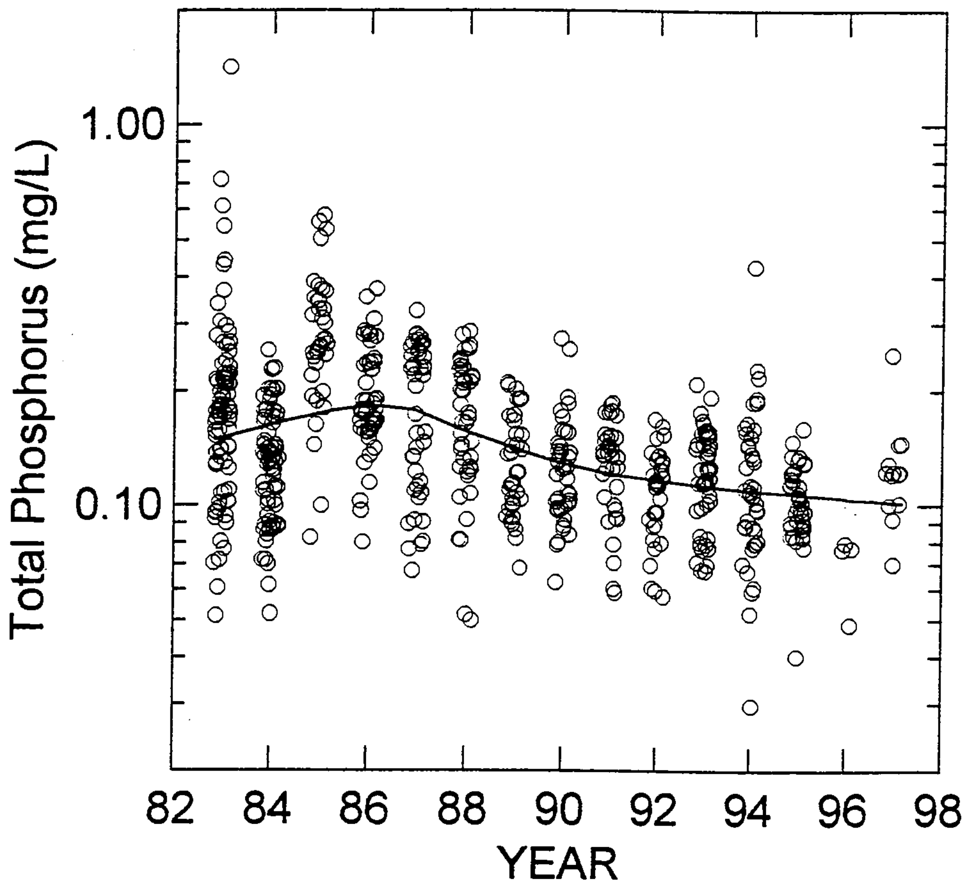
River Stations: June 15-Sept 15



Cove Stations: June 15-Sept 15



Cove Stations: June 15-Sept 15



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Buchanan, C. [ed.] 1999.
Prepared for the Chesapeake Bay Program.
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APPENDIX E

CHESAPEAKE BAY HABITAT CRITERIA SCORES AND THE DISTRIBUTION OF SUBMERSED AQUATIC VEGETATION IN THE TIDAL POTOMAC RIVER AND POTOMAC ESTUARY, 1983-1997

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This Appendix was published as USGS Open-File Report 99-219:

Landwehr, J. M., J. T. Reel, N. B. Rybicki, H. A. Ruhl, and V. Carter. 1999. Chesapeake habitat criteria scores and the distribution of submersed aquatic vegetation in the tidal Potomac River and Potomac estuary, 1983 - 1997. USGS Open-File Report 99-219. Available online at http://pubs.usgs.gov/of/1999/of99-219/.

Data analysis supporting this appendix can be found in USGS Open-File Report 98-657:

Carter, V., N. B. Rybicki, J. M. Landwehr, J. T. Reel, and H. A. Ruhl. 1998. Summary of correlations among seasonal water quality, discharge, weather, and coverage by submersed aquatic vegetation in the tidal Potomac River and Potomac estuary, 1983-1996. USGS Open-File Report 98-657. Available online at http://water.usgs.gov/nrp/proj.bib/sav/ofr98/.

U. S. Department of the Interior
U. S. Geological Survey

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CHESAPEAKE BAY HABITAT CRITERIA SCORES AND THE DISTRIBUTION OF SUBMERSED AQUATIC VEGETATION IN THE TIDAL POTOMAC RIVER AND POTOMAC ESTUARY, 1983-1997

By Jurate M. Landwehr, Justin T. Reel, Nancy B. Rybicki, Henry A. Ruhl and Virginia Carter

U.S. Geological Survey

Open-File Report 99-219

Reston, Virginia

1999

U. S. DEPARTMENT OF THE INTERIOR
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by

Jurate M. Landwehr, Justin T. Reel, Nancy B. Rybicki,

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ABSTRACT

The Chesapeake Bay Program has identified habitat requirements for the restoration of submersed aquatic vegetation (SAV) in the Chesapeake Bay estuary and tidal reaches of contributing river systems conditioned on the salinity regime of a specific location. The tidal Potomac River and Potomac Estuary is an important component of the Chesapeake Bay system to which these requirements can be applied. The SAV habitat requirements are formulated as threshold criteria that certain critical water-quality characteristics must satisfy during the SAV growing season. A multivariate scoring system based on these criteria was developed in order to synopsize water quality conditions during the 1983-1997 SAV growing seasons. Chesapeake Bay habitat criteria scores are displayed relative to annual SAV coverage for each Potomac River and Potomac Estuary segment. It is seen that although there is some correspondence in the inter-annual expansion or contraction of SAV coverage and compliance with Chesapeake Bay SAV habitat criteria, individual criteria provide neither necessary nor sufficient conditions to explain inter-annual dynamics of SAV coverage, especially in the Potomac Estuary .

INTRODUCTION

The objective of the U.S. Geological Survey's (USGS) Chesapeake Bay Ecosystem Program is to provide information on ecosystem structure responses to changes in water quality , especially nutrients, and to climate variability. This information is used by the broad community of policy makers, resource managers, scientists, and private citizens working on the environmental restoration of the Chesapeake Bay, including the Chesapeake Bay Program (CBP) – which is coordinated by the U.S. Environmental Protection Agency (EPA). During this century, the ecosystem of the Chesapeake Bay, the Nation's largest estuary, has been adversely affected by the loss of submersed aquatic vegetation (SAV) throughout the system. These primary producers form the base of the food chain and provide critical habitat for many of the living resources of the estuary, such as finfish, shellfish, and waterfowl. Decline in SAV has been attributed primarily to decreased water clarity, in response to increases in nutrient and sediment loads that have accompanied regional population growth.

As part of the USGS mission, USGS scientists are collecting and analyzing data related to current and historical nutrient and sediment loads in the drainage basin of the Chesapeake Bay and determining linkages between water quality and the distribution and abundance of SAV in the Potomac River drainage basin. In the tidal Potomac River, since the early 1980's, there has been both a dramatic resurgence as well as a retreat of SAV. The resurgence has been attributed primarily to improved waste-water treatment leading to improved water-column clarity over this time period. (Carter and Rybicki, 1986.) At the same time, there has been a minimal but consistently positive trend in the reemergence of SAV in the Potomac Estuary . This report provides information about the variations in the areal coverage of SAV in the tidal Potomac River and Potomac Estuary in relation to variations in water-quality as defined by CBP criteria for the period 1983 through 1997.

TIDAL POTOMAC RIVER AND POTOMAC ESTUARY SEGMENTATION AND COVERAGE BY SUBMERSED AQUATIC VEGETATION

The tidal Potomac River and Estuary extends 183 km from Little Falls near Chain Bridge in Washington, D.C., down to the river's mouth at the Chesapeake Bay. For its studies, the CBP has divided the Potomac River and Estuary into three segments by salinity regimes -- tidal fresh, oligohaline and mesohaline. Before 1997, the CBP defined these segments as TF2, RET2 and LE2, respectively, but they have since been redefined and renamed POTTF, POTOH, and POTMH, respectively. (Table 1 summarizes the various abbreviations used throughout this report.) In this report, data are reported by the three segments TF2, POTOH, and POTMH, as shown in Figure 1. Differences in realignment between the oligohaline (RET2 versus POTOH) and mesohaline (LE2 versus POTMH) segments were significant. Realignment within the freshwater tidal regime (TF2 versus POTTF) consisted primarily of the exclusion of two creek areas from the TF2. This designation of POTTF impeded historical comparisons, so that TF2, rather than POTTF, was used in this study .

For USGS study purposes, the tidal fresh segment, TF2, has been further subdivided into two segments, an upper tidal fresh (UTR) and lower tidal fresh (LTR) segment. (Carter and Rybicki, 1986; Carter and Rybicki in Batiuk and others, 1992.) Studies have indicated that historical patterns of SAV growth in UTR and LTR segments have been different (Landwehr and others, 1997). With respect to the TF2 versus POTTF designation, the POTTF designation would exclude SAV-contributing areas from Piscataway Creek in the UTR and from Mattowoman Creek in the LTR. The areal coverage of SAV in UTR and in LTR sum to that in the TF2; hence, to facilitate time series comparisons, we have chosen to retain the TF2 segment.

The growing season for SAV in the Potomac River extends from April through October. Using aerial photographs and on-site assessments, the Virginia Institute of Marine Science (Orth and others, 1997; and internet site http://www.vims.edu/bio/sav/) provides estimates of the annual SAV coverage in hectares for each of these CBP segments. USGS researchers, working with researchers at the Virginia Institute of Marine Science, have further partitioned the TF2 SAV coverage into coverage within the UTR and LTR segments. A summary of SAV coverage by river segment for the period 1983 through 1997 is given in Table 2.

The Maryland Department of Natural Resources (MD DNR) routinely collects water-quality data in the Potomac River and Potomac Estuary at nine locations, including four stations in TF2 with two each in UTR and LTR, two stations in POTOH, and three stations in POTMH, as shown in Figure 1. Because the areas of the salinity regime-based river segments that have been adopted by the CBP are fairly large, we have also considered smaller river segments around each of the nine water-quality monitoring sites. Each river segment consists of the six-kilometer river-length area which encompasses the water-quality site; that is, the segment comprises the three-kilometer river-length areas immediately above and downstream from a water-quality monitoring site. The SAV coverage for each of these monitoring station segments was determined jointly by workers at the USGS and at the Virginia Institute of Marine Science. SAV coverage in hectares by station segment and year is given in Table 3.

WATER-QUALITY DATA

The MD DNR routinely analyzes water samples for a suite of water-quality parameters. Generally, two samples per month are collected during the SAV growing season (April through October). According to Batiuk and others (1992), five water quality parameters are considered to be particularly relevant for SAV habitat restoration and SAV survival -- Secchi depth (SECCHI), total suspended solids (TSS), chlorophyll-a concentration (CHLA), dissolved inorganic phosphorus (DIP), and dissolved inorganic nitrogen (DIN). We obtained data sets for these five parameters from the MD DNR Sampling Program for April through October for the period 1983 through 1997.

The following quality control procedures were followed in preparing the data for analysis. First, data obtained from the MD DNR was examined for outliers and any extraordinary values were discussed with MD DNR staff. In a very few cases, numbers were adjusted, but only in accordance with recommendations by MD DNR staff. Second, a small number of measurements were reported to be below the detection limit of the respective analytical procedure. After examination of the distribution of the seasonal data for each respective parameter for any season during which a value below the detection limit was reported, we decided to set those values equal to the detection limit because the seasonal distributions were not markedly affected. Third, in several cases, replicate samples were analyzed and two measurements were reported for the same parameter on the same day at the same time and location. In these cases, the two measurements were averaged and the average was used in the analysis, unless one of the measurements was reported as below the detection limit, in which case only the measurement not below the detection limit was used.

In discussing the data with staff at MD DNR, we discovered that from 1991 forward, analyses for DIP and DIN were run on filtered water samples, whereas prior to 1991, all analyses were performed on unfiltered samples. Unfortunately, no overlap sampling period existed during which both filtered and unfiltered samples were analyzed. Thus, it is possible that our results reflect changes in sampling procedures in addition to variations in river column conditions before and after 1990.

The median seasonal value for the set of measurements for each parameter for each year was computed. For the CBP segments, measurements made at each water-quality monitoring site within the respective segment were pooled and the season median was computed for the entire data set. Median values for the SAV growing season for the pertinent parameters for years 1983 through 1997 are given in Tables 4-8 for the UTR, LTR, TF2, POTOH and POTMH segments, respectively. Median values for the SAV growing season for the five parameters for years 1983 through 1997 for the nine water-quality monitoring station segments are given in Tables 9-17. Again, samples for DIP and DIN were analyzed differently after the 1990 season, and the contents of the tables are qualified by this observation.

CHESAPEAKE BAY HABITAT CRITERIA AND SCORES

The CBP identified salinity-regime based criteria for certain critical water-quality parameters in order to assure the restoration of SAV. These parameters were identified in Table IV-1 (p.27) of the Chesapeake Bay Technical Synthesis Report (Batiuk and others, 1992). A DIP criterion specifically for the tidal Potomac River and Potomac Estuary was given in Table V-10 (page 74) of that same report. These criteria are presented in Table 18, expressed as bounds on the median values of samples collected during the SAV growing season (April-October). Note that the criterion for the light attenuation coefficient is expressed in terms of the more commonly made measurement of Secchi depth (SECCHI) by using the conversion factor assumed for the Chesapeake Bay by the CBP, namely SECCHI=1.45/(light attenuation coefficient); to be conservative, terms are rounded up. Note also that the criterion is equivalent to one expressed as per cent (%) light saturation in reference to a specified depth in the water column, for example, the one-meter restoration goal . The growing season (April-October) medians were calculated for the period 1983 through 1997. These are shown by salinity regime segment in Tables 5-11, and by monitoring station in Tables 12-20.

In order to summarize for management purposes whether or not the criteria have been historically satisfied, the median seasonal sample values for the water quality parameters for the several tidal Potomac River and Estuary segments can be converted into “achievement scores”, which are here called “Chesapeake Bay habitat criteria scores”. A score is assigned for each water-quality parameter by river segment for each year studied by the steps outlined in 1-3 below.

1. Consider a river segment **r** which is in salinity regime **s** (fresh, oligohaline or mesohaline). For each water-quality parameter **i,** the median value of measurements made for samples taken during the SAV growing season (April-October) in year **t** is represented as median **(i,r,t)**. Table 18 presents the criterion **(i,s)** to which the median in salinity regime s for water quality parameter **i** is compared.

2. For **i** = TSS, CHLA, DIP and DIN, the criteria define an upper bound which the seasonal median should not exceed. The Chesapeake Bay habitat score is computed to be:

Score **(i,r,t)** = 1- [median (**i,r,t**) / criterion (**i,s**)] ;

but

if Score (**i,r,t**) < -1, then set Score **(i,r,t)** = -1.

3. For **i** = SECCHI, the criterion defines a lower bound for the seasonal median. The Chesapeake Bay habitat score is computed as follows.

Score **(i,r,t)** = [median (**i,r,t**) / criterion (**i,s**)] – 1 ;

but

if Score (**i,r,t**) > +1, then set Score **(i,r,t)** = +1.

Thus, for each water quality characteristic, the score is bounded; that is,

-1 ≤ Score **(i,r,t)** ≤ +1**.**

When a criterion is satisfied for a particular year, Score **(i,r,t)** ≥ 0, but when a criterion is not satisfied during a particular season, Score **(i,r,t)** < 0. If Score **(i,r,t)** = 0, then the criterion has just been met.

Figures 2 through 15 include bar charts to graphically depict the scores for each water quality parameter and river segment derived from the seasonal median values presented in Tables 4 through 17, respectively , and the criteria in Table 18.

CHESAPEAKE BAY HABITAT CRITERIA SCORES AND COVERAGE BY SUBMERSED AQUATIC VEGETATION

SAV coverage given in Tables 2 and 3 is also graphically displayed in Figures 2 through 15; SAV coverage for each of the CBP and USGS segments are shown in Figures 2 through 6, and for each of the water quality monitoring station segments in Figures 7 through 15. Also denoted on these figures is an indication when analytical procedures changed and filtered rather than unfiltered samples were used for analysis.

In the freshwater tidal Potomac River segments UTR, LTR and TF2, SAV coverage predominantly corresponds to fluctuations of the Chesapeake Bay habitat scores: SAV coverage tended to increase between years if the scores were predominantly positive and decrease as scores became negative. In the oligohaline Potomac Estuary segment POTOH, the correspondence is more mixed, possibly confounded by the change in analytical procedures. In the mesohaline Potomac Estuary segment POTMH, there appears to be little correspondence to either the sign or the magnitude of the scores. SAV coverage remains minimal, albeit trending positively, throughout this period even though the preponderance of scores is positive. Ongoing propagule studies suggest that lack of seed materials, rather than water-quality conditions, are limiting the regrowth of SAV in this portion of the river.

Patterns in each of the water-quality monitoring station segments shown in Figures 7 through 15 correspond generally to the observations made for the larger salinity zone segments. However, the inter-annual pattern of SAV coverage in the tidal Potomac River station segments suggest that peak growth years show a distinct upstream-downstream gradient, suggested a strong relation to in-stream conditions, rather than external forcing.

The patterns in the Chesapeake Bay habitat criteria scores in relation to SAV coverage in each respective segment indicate that satisfaction of the SAV habitat criteria independently present neither necessary nor sufficient conditions for SAV to occur.

SUMMARY

Potomac River segments were assigned an annual (growing season median) score in reference to how well the water-quality conditions satisfied the SAV habitat criteria that were established for the Chesapeake Bay in 1992. Parameters included in the criteria are light attenuation, dissolved inorganic nitrogen, dissolved inorganic phosphorus, total suspended solids, and chlorophyll-a. SAV was generally present when SAV habitat criteria were met, but these criteria were not the only determining factors for fluctuations

in SAV areal coverage. For example, SAV habitat criteria were met in the lower Potomac Estuary for 1983-1997, yet SAV areal coverage was minimal.

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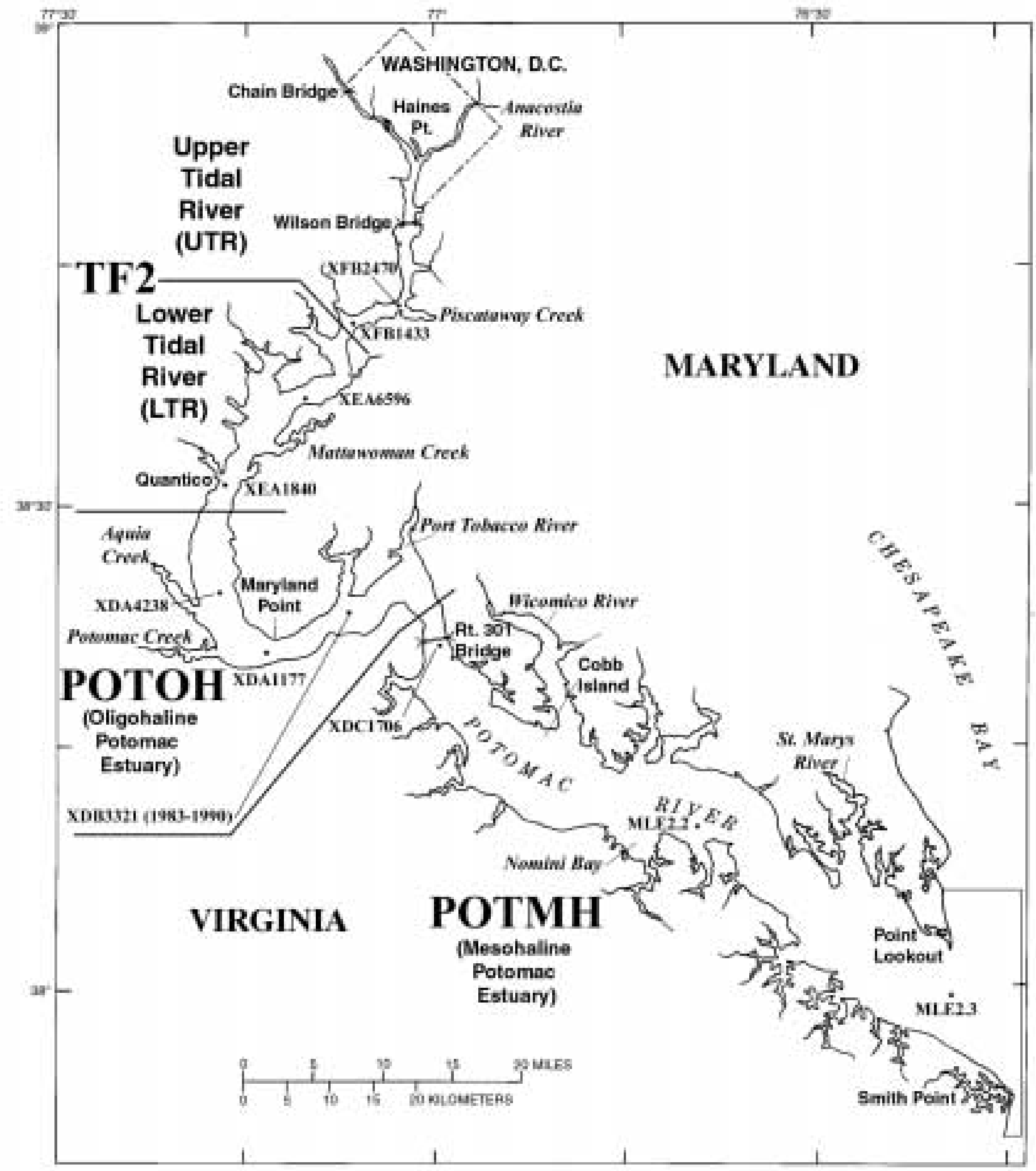

Figure 1. Map – Chesapeake Bay Program segments and stations for the tidal Potomac River and Potomac Estuary. Station numbers correspond to State of Maryland Department of Natural Resources mainstem monitoring stations.

Table 1. Abbreviations for terms used throughout the report

Term	Abbreviation
Secchi depth, in meters (m)	SECCHI
Total suspended solids, in milligrams per liter (mg/l)	TSS
Chlorophyll-a, in micrograms per liter (µg/l)	CHLA
Dissolved inorganic phosphorus, in milligrams per liter (mg/l)	DIP
Dissolved inorganic nitrogen, in milligrams per liter (mg/l)	DIN
Submersed aquatic vegetation	SAV
Freshwater tidal Potomac River segment	TF2
Upper tidal Potomac River segment	UTR
Lower tidal Potomac River segment	LTR
Oligohaline Potomac Estuary segment	POTOH
Mesohaline Potomac Estuary segment	POTMH
State of Maryland Department of Natural Resources	MD DNR
Chesapeake Bay Program	CBP

Table 2. Surface area, in hectares, covered by submersed aquatic vegetation in the tidal Potomac River and Potomac Estuary for the period 1983 through 1997, for USGS segments UTR and LTR, and Chesapeake Bay Program segments TF2, POTOH, and POTMH (n.d. = no data; see Figure 1 for location of river segments)

YEAR	UTR	LTR	TF2	POTOH	POTMH
1983	200.00	0.00	200.00	n.d.	n.d.
1984	619.51	0.00	619.51	217.09	59.90
1985	1375.50	0.41	1375.91	431.80	55.61
1986	1554.20	56.54	1610.74	383.87	43.12
1987	1465.06	113.17	1578.23	484.95	49.64
1988	1267.00	140.00	1407.00	434.00	n.d.
1989	586.75	717.55	1304.30	1216.19	100.83
1990	627.81	1010.95	1638.75	1308.19	108.77
1991	837.09	1207.11	2044.20	1414.87	136.75
1992	435.99	976.42	1412.41	1501.17	96.58
1993	479.01	933.85	1412.86	1296.74	110.07
1994	412.29	569.97	982.26	1255.18	194.55
1995	419.17	224.45	643.61	1023.53	239.17
1996	272.40	470.28	742.69	1036.66	402.40
1997	364.41	363.23	727.64	1206.26	666.84

Table 3. Surface area, in hectares, covered by submersed aquatic vegetation in the tidal Potomac River and Potomac Estuary for the period 1983 through 1997, for water-quality monitoring station segments (n.d. = no data; see Figure 1 for location of monitoring stations)

YEAR	XFB2470	XFB1433	XEA6596	XEA1840	XDA4238	XDA1177	XDC1706	MLE2.2	MLE2.3
1983	0.00	0.00	0.00	0.00	n.d.	n.d.	n.d.	n.d.	n.d.
1984	254.82	30.19	0.00	0.00	0.00	17.13	4.48	0.00	0.00
1985	499.35	108.99	0.41	8.99	5.41	22.35	5.61	0.00	2.18
1986	647.21	171.63	2.14	10.31	7.08	16.41	7.56	0.00	0.69
1987	618.33	136.62	10.05	33.59	4.84	12.13	19.97	0.00	0.00
1988	540.00	146.00	9.00	94.00	15.00	4.00	n.d.	n.d.	n.d.
1989	81.15	118.07	31.77	232.72	55.96	49.09	54.65	0.00	0.00
1990	101.92	113.28	42.55	333.04	207.70	49.17	54.70	0.00	0.00
1991	222.23	156.97	74.44	342.48	201.22	61.03	53.33	0.00	0.00
1992	45.63	45.61	60.09	337.19	163.86	71.16	49.52	0.00	0.00
1993	39.47	45.54	71.74	351.54	172.17	75.28	50.86	0.00	0.00
1994	40.11	15.24	21.68	298.49	157.61	76.87	59.35	0.00	0.00
1995	46.53	16.06	16.77	71.03	78.94	82.14	57.51	21.69	0.00
1996	49.70	24.19	37.89	87.60	64.20	58.05	60.63	49.51	0.00
1997	122.41	30.62	36.16	24.25	49.46	35.06	71.97	63.74	0.00

Table 4. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters in freshwater tidal Potomac River segment UTR for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)
1983	0.66	12.5	5.56	0.0300
1984	0.76	10.0	8.26	0.0200
1985	0.91	14.5	7.98	0.0400
1986	0.60	15.0	7.35	0.0200
1987	0.70	14.0	8.37	0.0300
1988	0.60	19.0	7.03	0.0220
1989	0.50	15.5	6.88	0.0380
1990	0.60	18.0	8.52	0.0320
1991	0.70	15.5	8.97	0.0160
1992	0.60	18.0	7.85	0.0230
1993	0.60	19.5	12.71	0.0130
1994	0.60	17.0	18.44	0.0205
1995	0.70	20.0	19.94	0.0160
1996	0.60	23.0	8.65	0.0215
1997	0.60	20.5	20.50	0.0125

Table 5. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters in freshwater tidal Potomac River segment LTR for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)
1983	0.61	18.0	9.68	0.0400
1984	0.67	18.0	25.66	0.0200
1985	0.61	15.0	24.44	0.0375
1986	0.50	17.5	12.79	0.0320
1987	0.75	14.0	7.73	0.0290
1988	0.75	15.7	6.48	0.0290
1989	0.80	11.5	5.21	0.0250
1990	0.60	17.0	4.38	0.0300
1991	0.80	9.0	4.80	0.0185
1992	0.70	15.0	3.99	0.0300
1993	0.70	15.5	4.98	0.0255
1994	0.60	19.3	12.96	0.0255
1995	0.60	22.0	17.57	0.0233
1996	0.50	27.5	11.04	0.0185
1997	0.45	26.3	25.08	0.0148

Table 6. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters in freshwater tidal Potomac River segment TF2 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)
1983	0.61	16.5	5.94	0.0400
1984	0.75	14.0	12.03	0.0200
1985	0.70	15.0	18.04	0.0400
1986	0.60	16.0	9.87	0.0240
1987	0.70	14.0	7.78	0.0300
1988	0.70	17.0	7.03	0.0240
1989	0.60	13.5	5.68	0.0325
1990	0.60	18.0	5.38	0.0300
1991	0.75	11.8	6.12	0.0160
1992	0.70	17.0	5.23	0.0260
1993	0.60	17.5	10.02	0.0165
1994	0.60	18.0	15.20	0.0225
1995	0.70	20.8	18.89	0.0190
1996	0.50	25.8	10.02	0.0210
1997	0.50	21.3	24.30	0.0130

Table 7. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters in oligohaline Potomac Estuary segment POTOH for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)
1983	0.82	10.5	2.67	0.0500
1984	0.66	16.0	26.62	0.0600
1985	0.61	18.0	6.81	0.0600
1986	0.60	15.5	3.61	0.0640
1987	0.65	15.0	3.89	0.0475
1988	0.70	13.0	4.19	0.0540
1989	0.50	15.5	3.74	0.0480
1990	0.45	21.0	3.42	0.0500
1991	0.80	12.0	5.78	0.0300
1992	0.60	16.3	3.36	0.0470
1993	0.60	19.3	4.02	0.0448
1994	0.50	21.0	4.86	0.0463
1995	0.58	20.8	6.15	0.0375
1996	0.40	30.5	8.83	0.0258
1997	0.50	25.3	11.85	0.0270

Table 8. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters in mesohaline Potomac Estuary segment POTMH for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (μg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.88	7.5	2.83	0.0400	0.5860
1984	0.87	6.0	8.03	0.0400	0.1895
1985	1.15	8.1	7.06	0.0200	0.1195
1986	1.20	6.0	7.33	0.0100	0.2100
1987	1.20	6.0	9.12	0.0070	0.2500
1988	1.15	8.0	13.46	0.0080	0.1495
1989	1.10	8.0	10.39	0.0090	0.4035
1990	1.00	11.0	8.82	0.0080	0.2700
1991	1.25	7.0	7.18	0.0045	0.1390
1992	1.40	12.0	5.98	0.0090	0.2330
1993	1.20	8.0	8.01	0.0100	0.1400
1994	1.15	7.6	9.12	0.0065	0.1760
1995	1.40	12.5	7.18	0.0080	0.1435
1996	1.00	12.0	13.23	0.0090	0.6320
1997	1.00	13.0	11.10	0.0070	0.2155

Table 9. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at freshwater tidal Potomac River monitoring station XFB2470 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (μg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.76	12.0	9.33	0.0300	1.9040
1984	0.87	10.0	4.97	0.0300	2.1335
1985	1.01	12.5	6.74	0.0400	2.0330
1986	0.70	13.0	6.76	0.0180	1.8540
1987	0.80	14.0	6.88	0.0310	2.0680
1988	0.65	21.5	6.06	0.0210	2.5440
1989	0.50	15.0	6.80	0.0380	2.1120
1990	0.60	18.0	9.57	0.0320	2.0520
1991	0.60	14.5	8.72	0.0160	2.5720
1992	0.60	19.0	7.85	0.0250	2.4120
1993	0.60	19.0	14.45	0.0120	2.1000
1994	0.60	15.5	14.95	0.0215	1.9045
1995	0.70	19.5	18.19	0.0160	1.7640
1996	0.60	22.5	9.79	0.0225	1.6395
1997	0.65	19.0	18.40	0.0140	1.6070

Table 10. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at freshwater tidal Potomac River monitoring station XFB1433 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (μg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.66	18.0	5.56	0.0300	1.8690
1984	0.76	11.0	9.71	0.0200	1.9840
1985	0.85	15.0	16.27	0.0400	1.7940
1986	0.60	15.0	8.37	0.0200	1.7520
1987	0.60	15.0	8.97	0.0300	1.9610
1988	0.60	17.5	8.15	0.0230	2.2200
1989	0.55	16.0	8.67	0.0380	2.0760
1990	0.55	18.0	7.48	0.0280	2.1100
1991	0.70	16.0	9.22	0.0140	2.2600
1992	0.65	17.5	8.22	0.0210	2.1720
1993	0.60	20.0	11.96	0.0140	2.0100
1994	0.60	18.5	20.19	0.0190	1.6230
1995	0.70	25.5	22.18	0.0160	1.6550
1996	0.55	23.0	7.92	0.0215	1.6095
1997	0.50	22.5	28.60	0.0120	1.1480

Table 11. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at freshwater tidal Potomac River monitoring station XEA6596 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (μg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.53	24.0	16.68	0.0400	1.6110
1984	0.70	15.5	22.13	0.0200	1.4420
1985	0.61	14.0	24.55	0.0300	1.4200
1986	0.50	16.0	12.86	0.0280	1.3880
1987	0.65	14.5	9.79	0.0290	1.5860
1988	0.65	19.0	14.35	0.0230	1.7760
1989	0.65	12.0	8.01	0.0180	1.9680
1990	0.60	16.0	5.68	0.0280	2.0600
1991	0.80	9.5	5.48	0.0145	1.8520
1992	0.70	15.0	6.98	0.0160	1.7160
1993	0.60	16.5	14.08	0.0175	1.4500
1994	0.60	20.0	21.93	0.0225	1.4715
1995	0.60	23.0	22.13	0.0160	1.1295
1996	0.50	22.5	9.57	0.0200	1.4750
1997	0.45	27.5	23.00	0.0130	1.1155

Table 12. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at freshwater tidal Potomac River monitoring station XEA1840 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.66	16.0	3.96	0.0400	1.4210
1984	0.67	21.0	32.72	0.0200	0.8440
1985	0.61	15.0	19.11	0.0400	0.8908
1986	0.50	17.8	10.73	0.0405	1.0150
1987	0.80	11.0	6.06	0.0303	1.4200
1988	0.90	13.0	4.26	0.0305	1.4560
1989	0.80	11.0	3.81	0.0290	1.9410
1990	0.60	17.8	3.19	0.0320	1.7450
1991	1.00	8.0	4.26	0.0240	1.4820
1992	0.70	15.0	2.49	0.0365	1.6470
1993	0.70	14.5	3.64	0.0333	1.1235
1994	0.60	17.8	9.78	0.0295	1.1710
1995	0.65	20.0	14.32	0.0280	0.9438
1996	0.45	27.8	12.75	0.0173	1.4240
1997	0.45	23.0	25.40	0.0200	0.6940

Table 13. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at oligohaline Potomac Estuary monitoring station XDA4238 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.82	11.0	3.69	0.0500	1.2300
1984	0.66	16.0	20.69	0.0600	0.7580
1985	0.49	19.5	6.36	0.0600	0.8240
1986	0.55	18.5	3.84	0.0680	0.8530
1987	0.70	15.0	3.14	0.0475	1.1980
1988	0.60	13.0	4.34	0.0540	0.8700
1989	0.60	12.0	2.99	0.0450	1.8480
1990	0.40	21.0	3.42	0.0500	1.5040
1991	0.80	11.5	7.10	0.0260	0.6460
1992	0.55	16.0	4.17	0.0440	1.3100
1993	0.60	20.5	3.46	0.0380	0.7200
1994	0.45	25.0	4.86	0.0465	0.9550
1995	0.63	19.5	6.48	0.0340	0.9335
1996	0.40	28.0	8.22	0.0235	1.3705
1997	0.45	26.0	13.50	0.0270	0.5865

Table 14. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at oligohaline Potomac Estuary monitoring station XDA1177 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.82	10.0	2.46	0.0500	1.2200
1984	0.62	22.0	28.87	0.0700	0.5150
1985	0.61	17.0	8.75	0.0600	0.6590
1986	0.60	13.6	3.29	0.0590	0.7230
1987	0.55	15.0	4.49	0.0470	1.0070
1988	0.80	13.0	3.74	0.0540	0.6720
1989	0.50	18.0	4.34	0.0540	1.5240
1990	0.50	20.0	3.36	0.0500	1.2460
1991	0.75	13.5	5.63	0.0330	0.4910
1992	0.60	17.0	3.27	0.0505	1.0850
1993	0.60	18.5	4.42	0.0475	0.5590
1994	0.55	17.5	4.24	0.0460	0.7180
1995	0.58	21.0	5.00	0.0430	0.8075
1996	0.35	34.3	8.83	0.0275	1.3613
1997	0.50	25.3	7.85	0.0275	0.5023

Table 15. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at mesohaline Potomac Estuary monitoring station XDC1706 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)	DIN (mg/l)
1983	0.88	8.0	3.05	0.0600	0.8430
1984	0.64	10.0	7.38	0.1000	0.2870
1985	0.79	9.8	6.01	0.0600	0.3165
1986	0.80	9.8	5.46	0.0495	0.4095
1987	0.95	7.0	5.05	0.0455	0.6620
1988	0.90	11.0	6.23	0.0480	0.4920
1989	0.80	12.5	5.57	0.0575	1.0460
1990	0.70	14.5	4.86	0.0380	0.5350
1991	0.80	13.5	7.20	0.0310	0.2400
1992	0.90	14.0	5.67	0.0270	0.5120
1993	0.80	14.0	7.58	0.0475	0.3200
1994	0.60	18.5	7.73	0.0445	0.3240
1995	0.80	17.0	5.58	0.0320	0.3865
1996	0.70	19.0	9.80	0.0330	1.0190
1997	0.80	19.0	11.95	0.0165	0.2540

Table 16. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at mesohaline Potomac Estuary monitoring station MLE2.2 for the period 1983 through 1997

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)	DIN (mg/l)
1984	0.76	4.0	11.91	0.0250	0.1520
1985	1.20	10.0	10.38	0.0200	0.0800
1986	1.10	8.0	7.63	0.0100	0.1160
1987	1.10	7.0	14.20	0.0070	0.0800
1988	1.00	8.0	14.50	0.0080	0.1080
1989	1.20	13.0	22.33	0.0100	0.1560
1990	1.05	13.0	13.76	0.0060	0.1560
1991	1.15	13.0	6.95	0.0040	0.0640
1992	1.40	16.5	10.77	0.0070	0.1250
1993	1.25	11.5	8.26	0.0130	0.0490
1994	1.30	9.5	10.89	0.0075	0.0560
1995	1.60	14.5	8.75	0.0080	0.0830
1996	1.10	15.0	16.22	0.0090	0.3820
1997	1.30	19.3	12.00	0.0080	0.1735

Table 17. Seasonal (April through October) median values for submersed aquatic vegetation habitat criteria parameters at mesohaline Potomac Estuary monitoring station MLE2.3 for the period 1983 through 1997 (n.d. = no data)

YEAR	SECCHI (m)	TSS (mg/l)	CHLA (µg/l)	DIP (mg/l)	DIN (mg/l)
1983	n.d.	6.0	2.62	0.0100	0.4680
1984	2.15	4.0	7.68	0.0070	0.0760
1985	1.75	4.8	4.19	0.0028	0.0607
1986	2.20	4.4	7.48	0.0021	0.0892
1987	1.90	4.3	9.42	0.0033	0.0526
1988	1.60	7.6	13.91	0.0021	0.0336
1989	1.60	5.4	12.46	0.0032	0.1175
1990	2.00	5.2	9.72	0.0042	0.0564
1991	1.80	4.3	7.14	0.0028	0.0252
1992	2.00	4.4	5.79	0.0023	0.0328
1993	1.40	4.8	8.35	0.0028	0.0261
1994	1.40	5.9	9.12	0.0031	0.0316
1995	1.50	5.0	6.43	0.0024	0.0318
1996	1.40	4.8	13.61	0.0029	0.1510
1997	1.60	6.7	10.70	0.0028	0.0175

Table 18. Chesapeake Bay habitat criteria for the tidal Potomac River and Potomac Estuary based on Chesapeake Bay submersed aquatic vegetation (SAV) habitat requirements for one meter restoration as established by the Chesapeake Bay Program

Water-Quality Parameter	**Criteria by Salinity Regime** The median value of measurements made during the SAV growing season (April through October) for each water-quality parameter must satisfy the criterion for the specific salinity regime of the sampling site.		
	Freshwater	**Oligohaline**	**Mesohaline**
SECCHI	≥ 0.7 m *	≥ 0.7 m *	≥ 1.0 m **
TSS	≤ 15 mg/l	≤ 15 mg/l	≤ 15 mg/l
CHLA	< 15 µg/l	< 15 µg/l	< 15 µg/l 1
DIP	≤0.04 mg/l	<0.07 mg/l	<0.01 mg/l
DIN	(none)	(none)	<0.15 mg/l

* This is equivalent to a criterion for the seasonal median value of the light attenuation coefficient to be ≤ 2, and for the seasonal median percent light to be ≥12.6% at 1-meter depth in the water column, assuming the Chesapeake Bay Program conversion factor, light attenuation coefficient = 1.45/SECCHI.

** This is equivalent to a criterion for the seasonal median value of the light attenuation coefficient to be ≤ 1.5, and for the seasonal median percent light to be ≥23.5% at 1-meter depth in the water column, assuming the Chesapeake Bay Program conversion factor, light attenuation coefficient = 1.45/SECCHI.

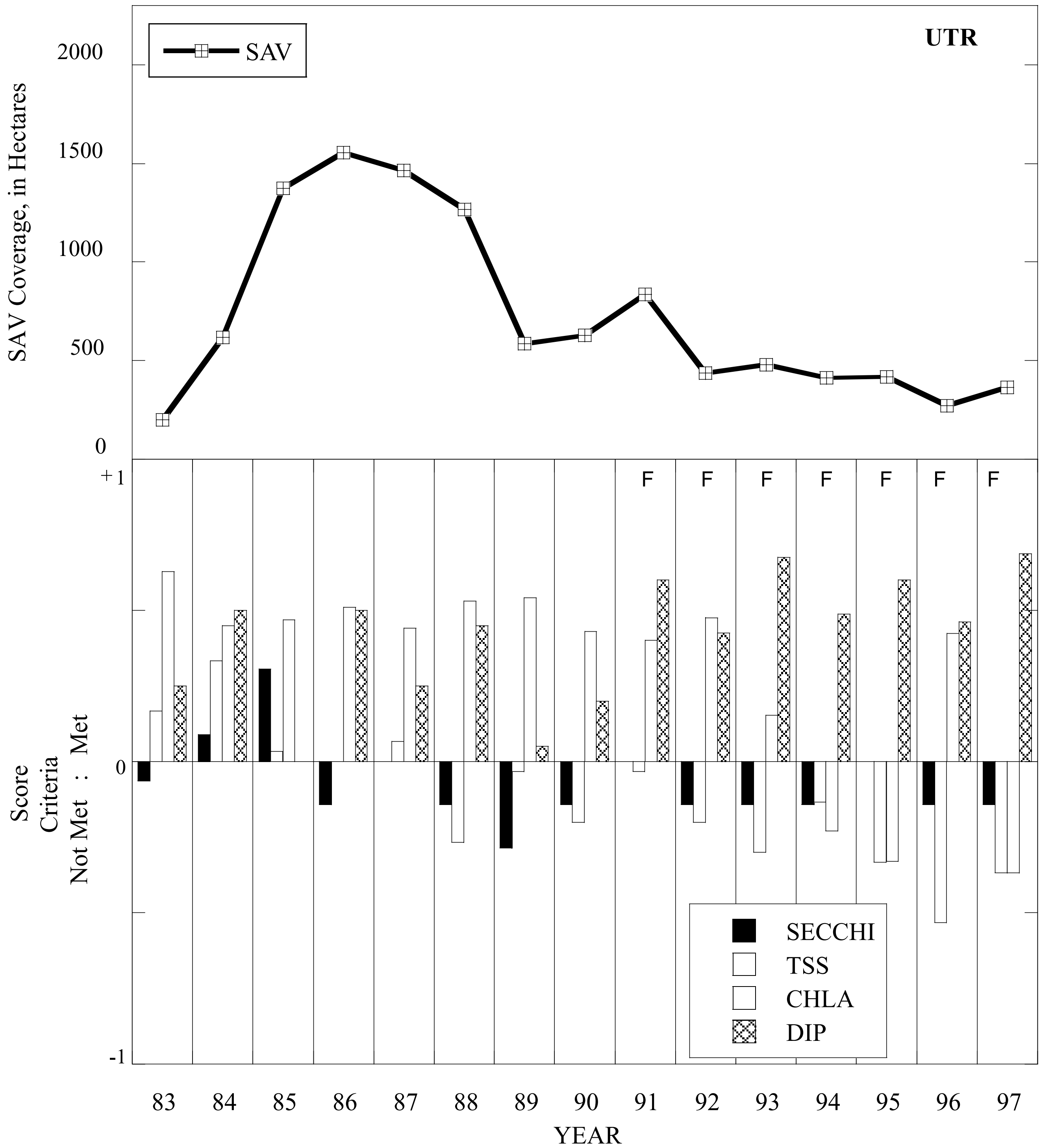

Figure 2. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores in freshwater tidal Potomac River segment UTR for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

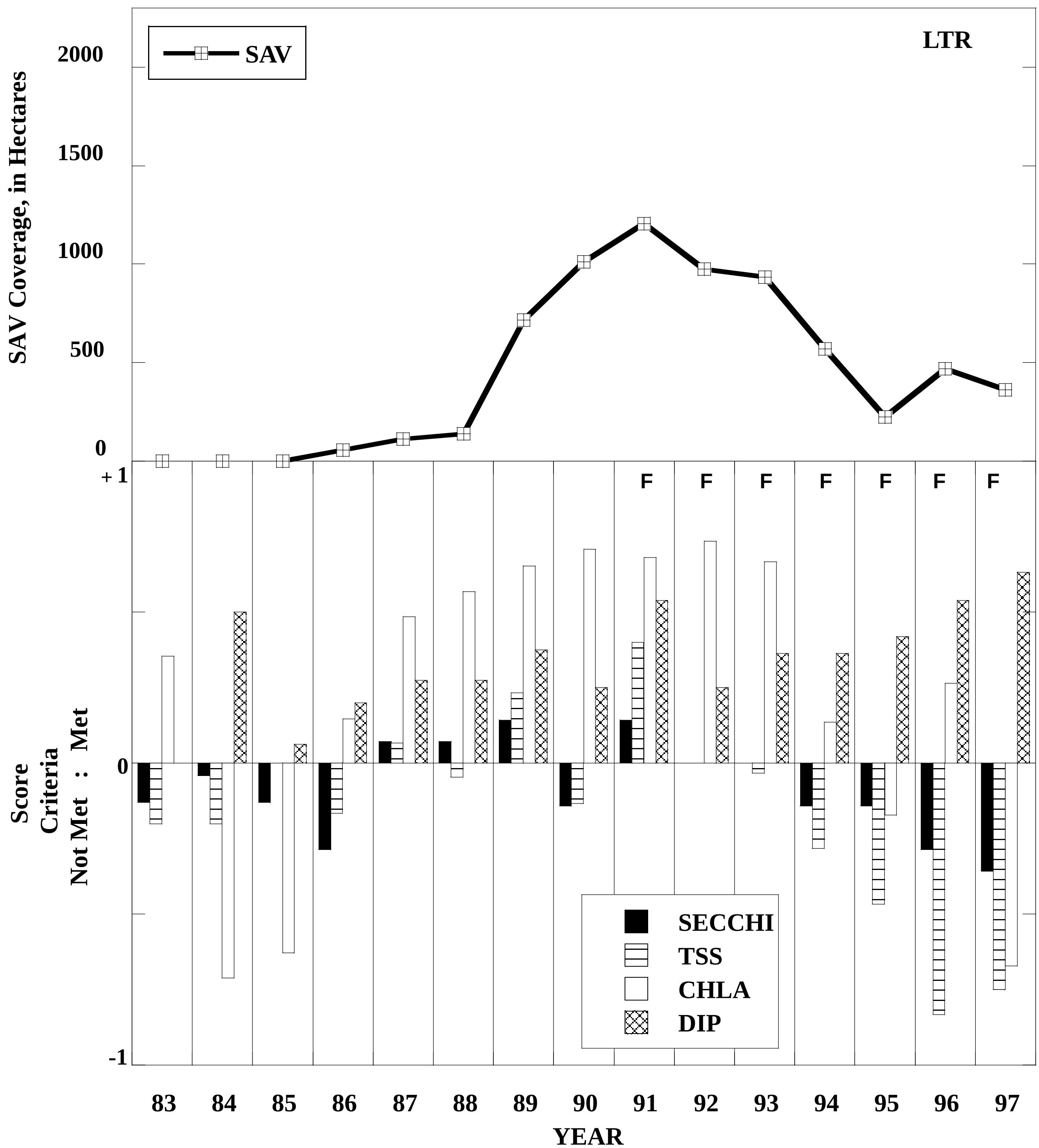

Figure 3. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores in freshwater tidal Potomac River segment LTR for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

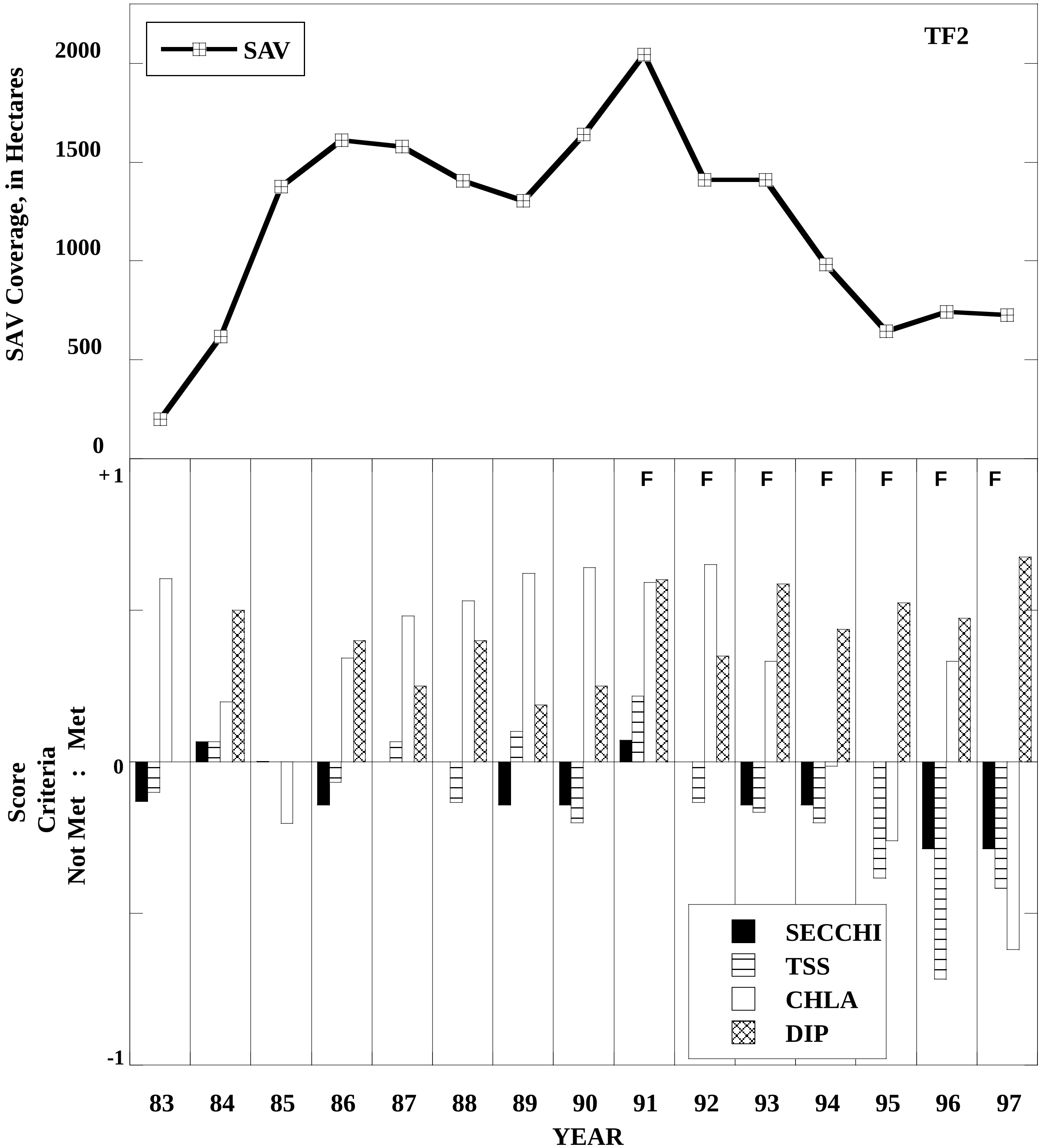

Figure 4. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores in freshwater tidal Potomac River segment TF2 for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

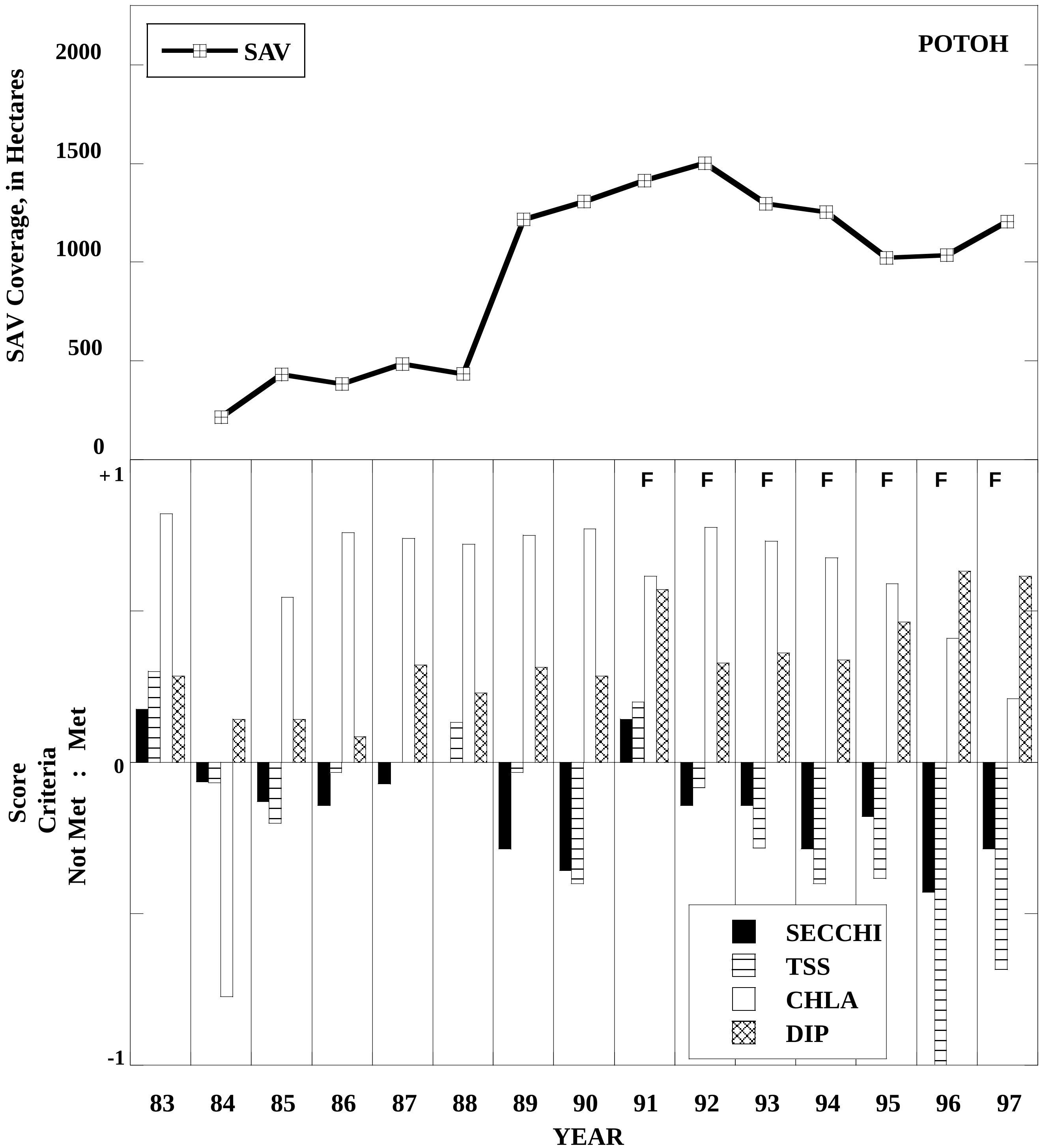

Figure 5. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores in oligohaline Potomac Estuary segment POTOH for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

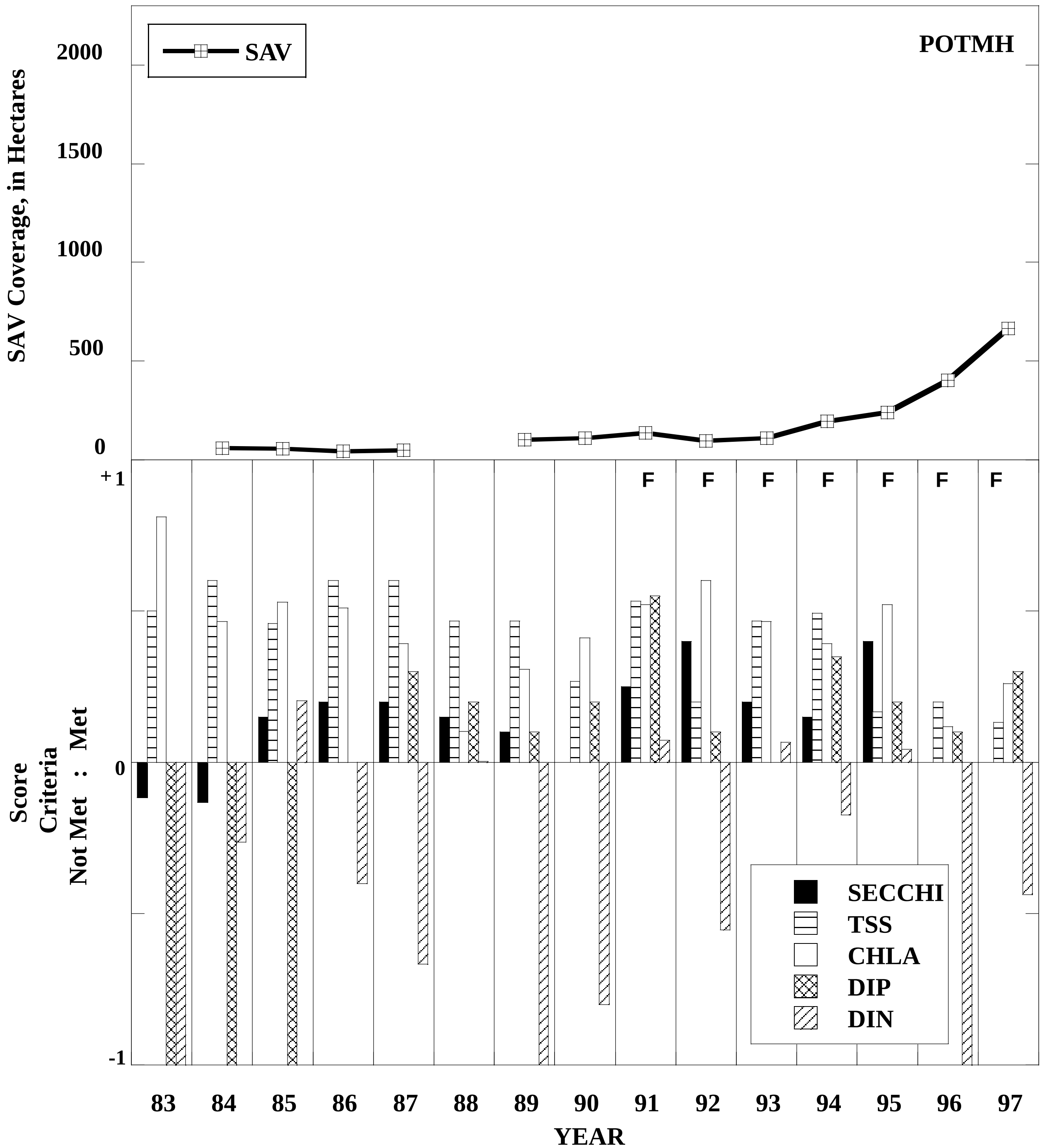

Figure 6. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores in mesohaine Potomac Estuary segment POTMH for the period 1983 through 1997. (F denotes use of filtered water samples for DIN and DIP.)

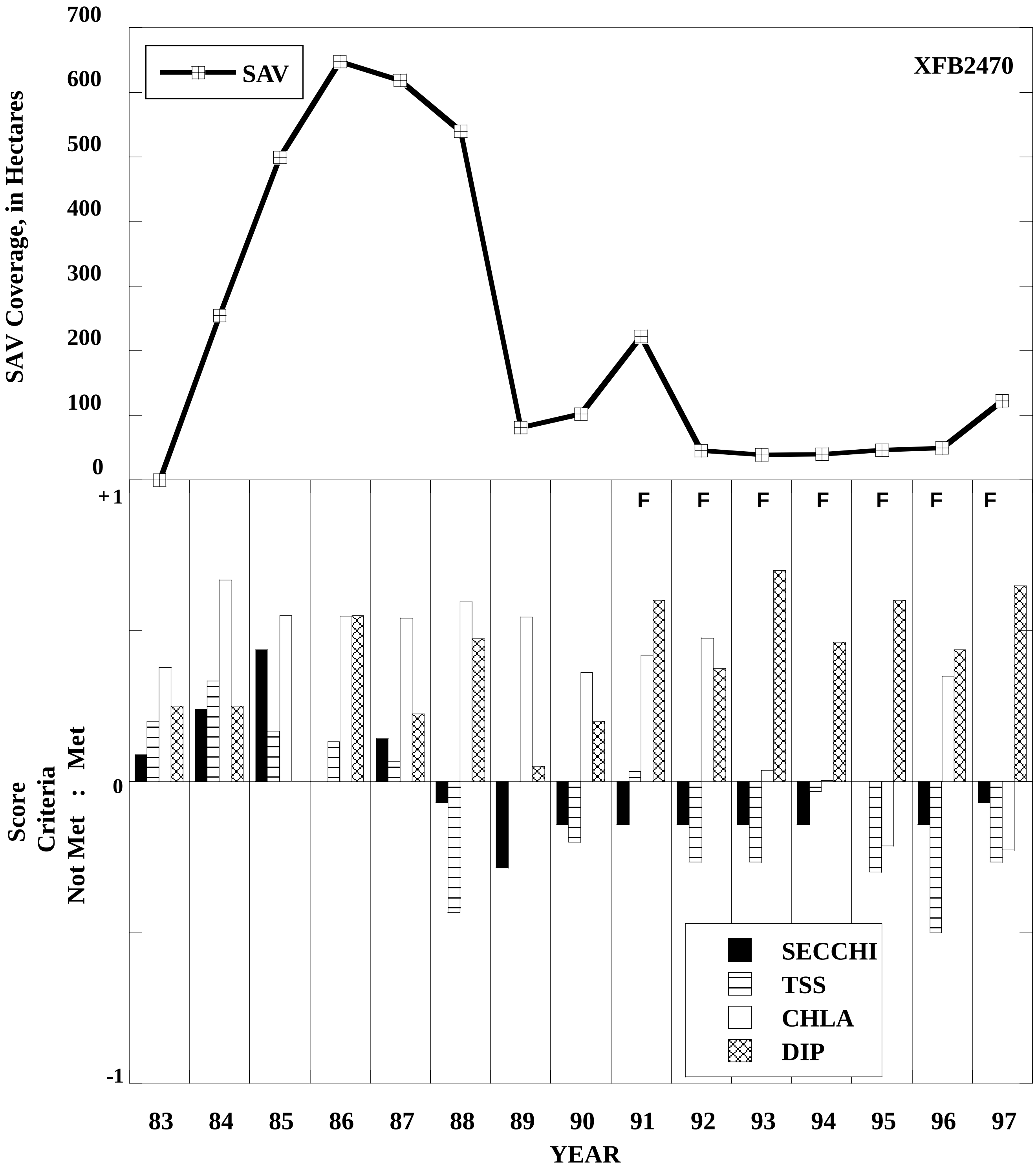

Figure 7. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for freshwater tidal Potomac River monitoring station XFB2470 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

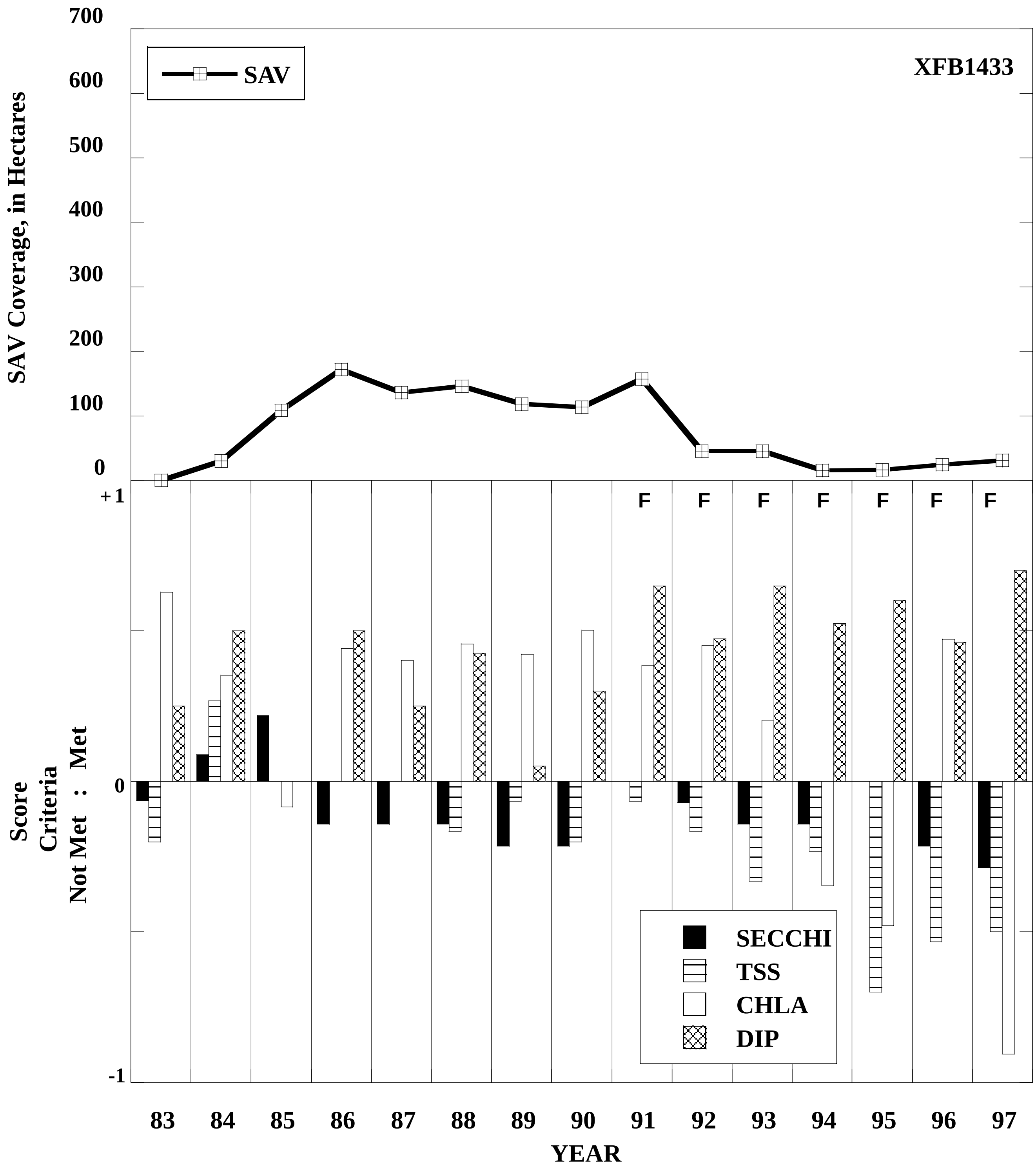

Figure 8. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for the freshwater tidal Potomac River monitoring station XFB1433 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

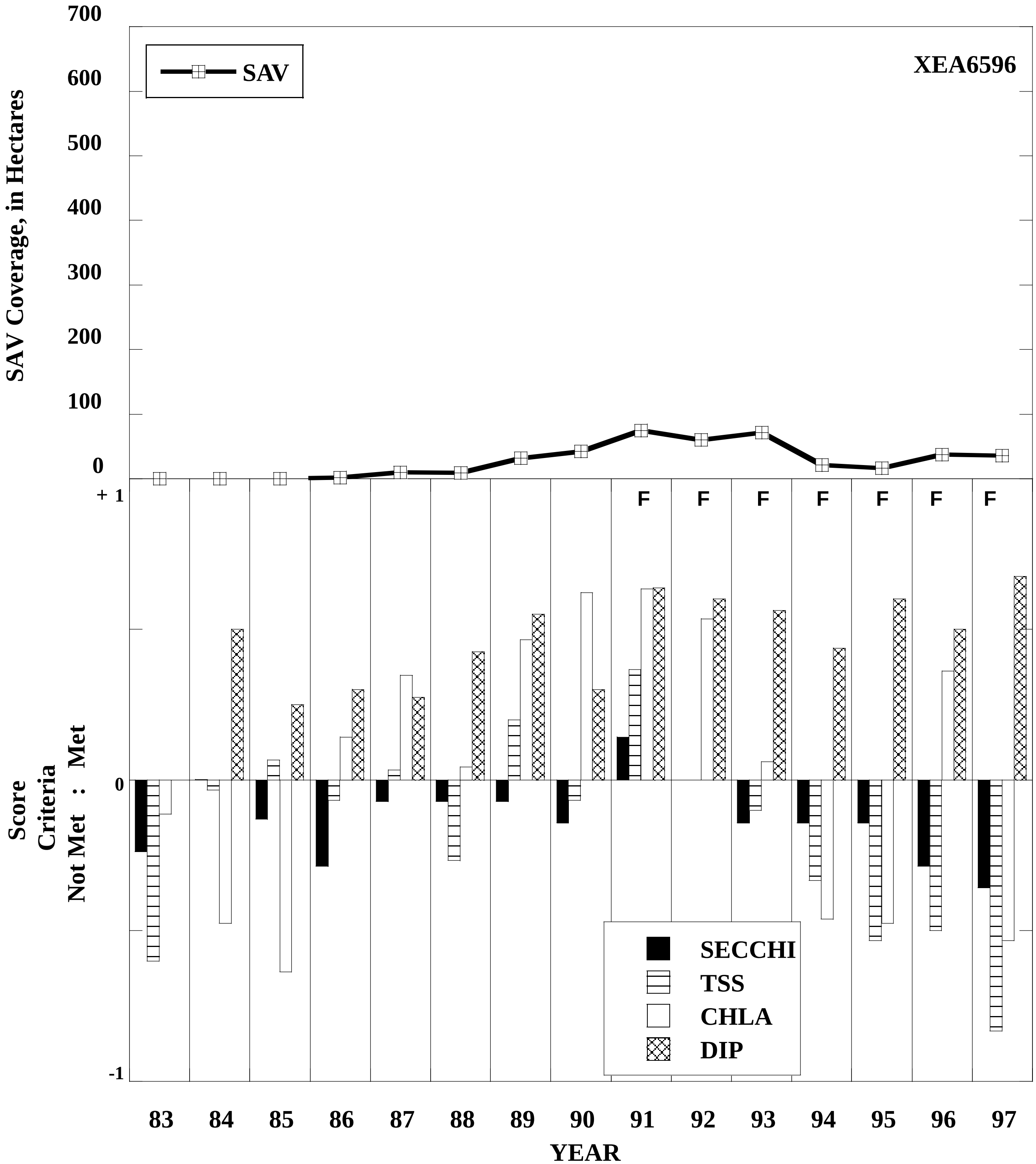

Figure 9. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for freshwater tidal Potomac River monitoring station XEA6596 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

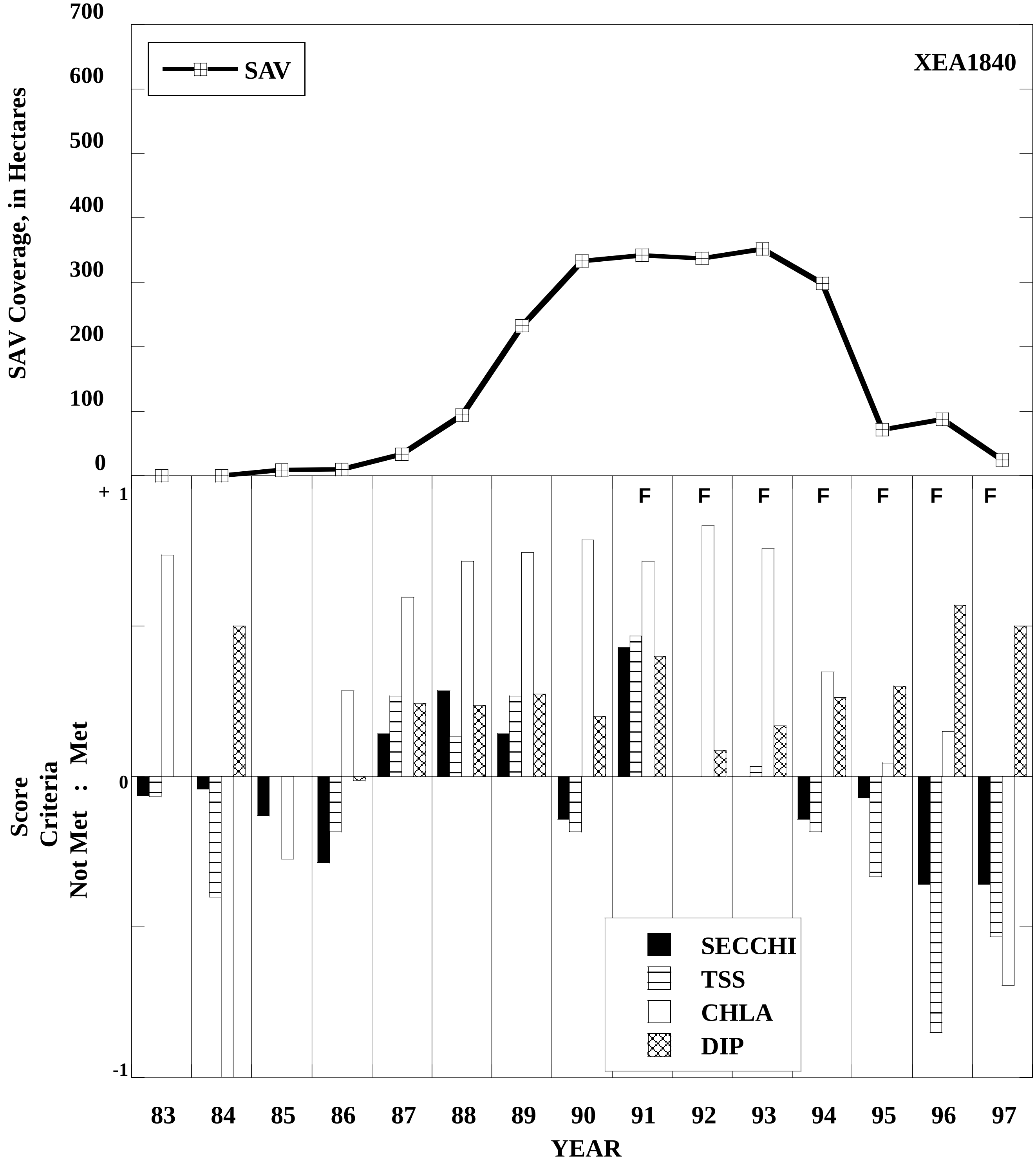

Figure 10. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for freshwater tidal Potomac River monitoring station XEA1840 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

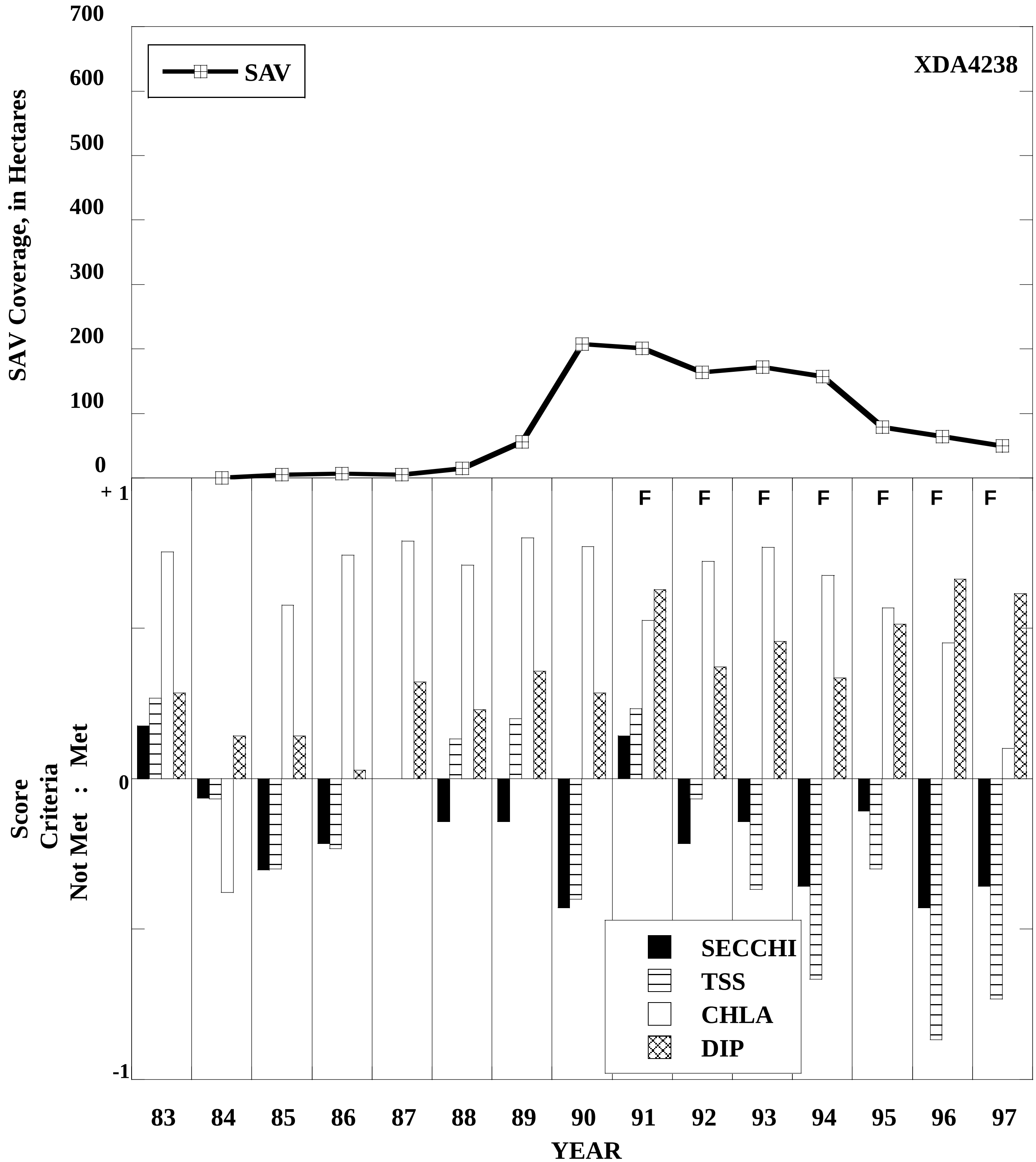

Figure 11. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for oligohaline Potomac Estuary monitoring station XDA4238 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

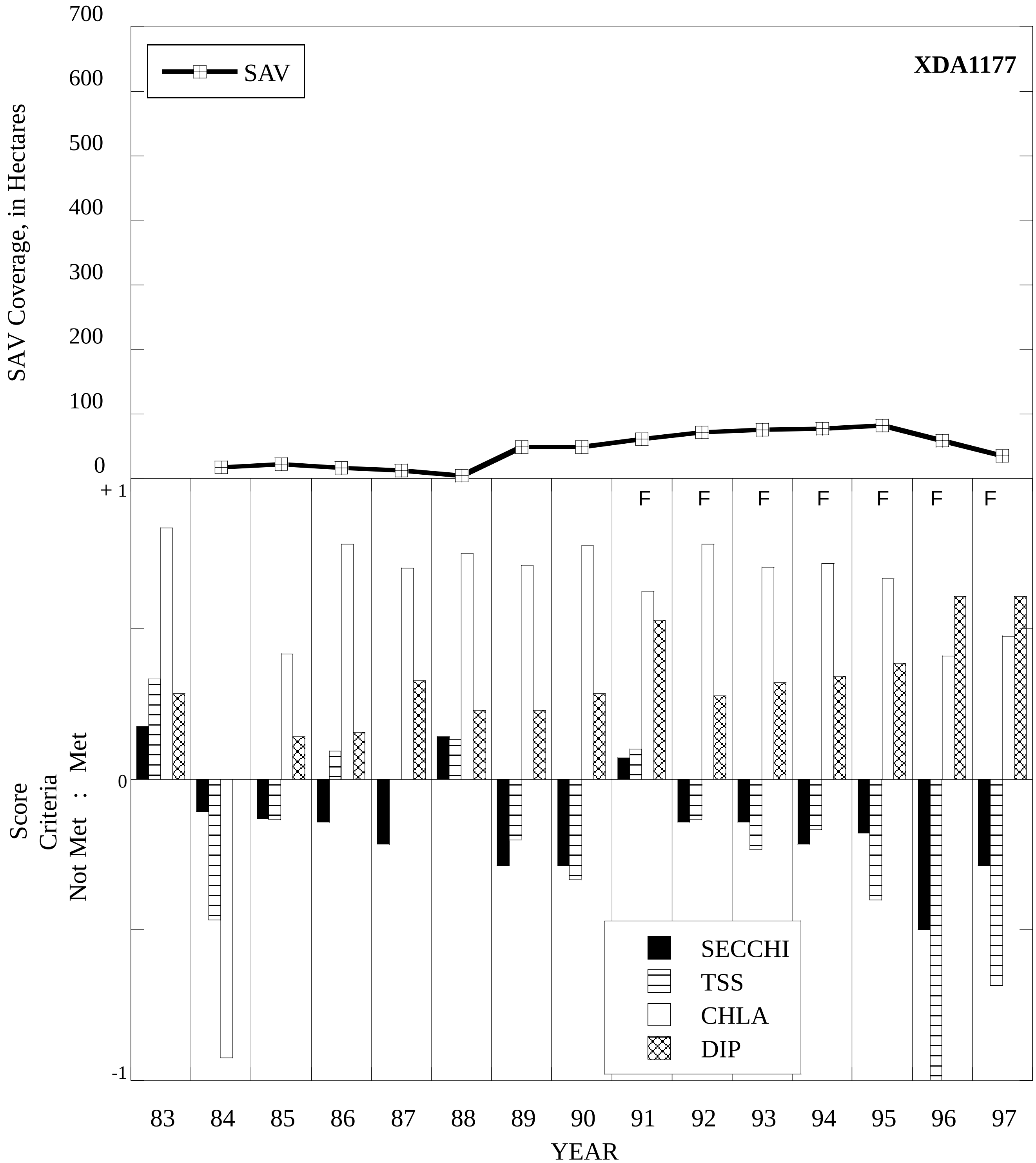

Figure 12. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for oligohaline Potomac Estuary monitoring station XDC1177 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIP.)

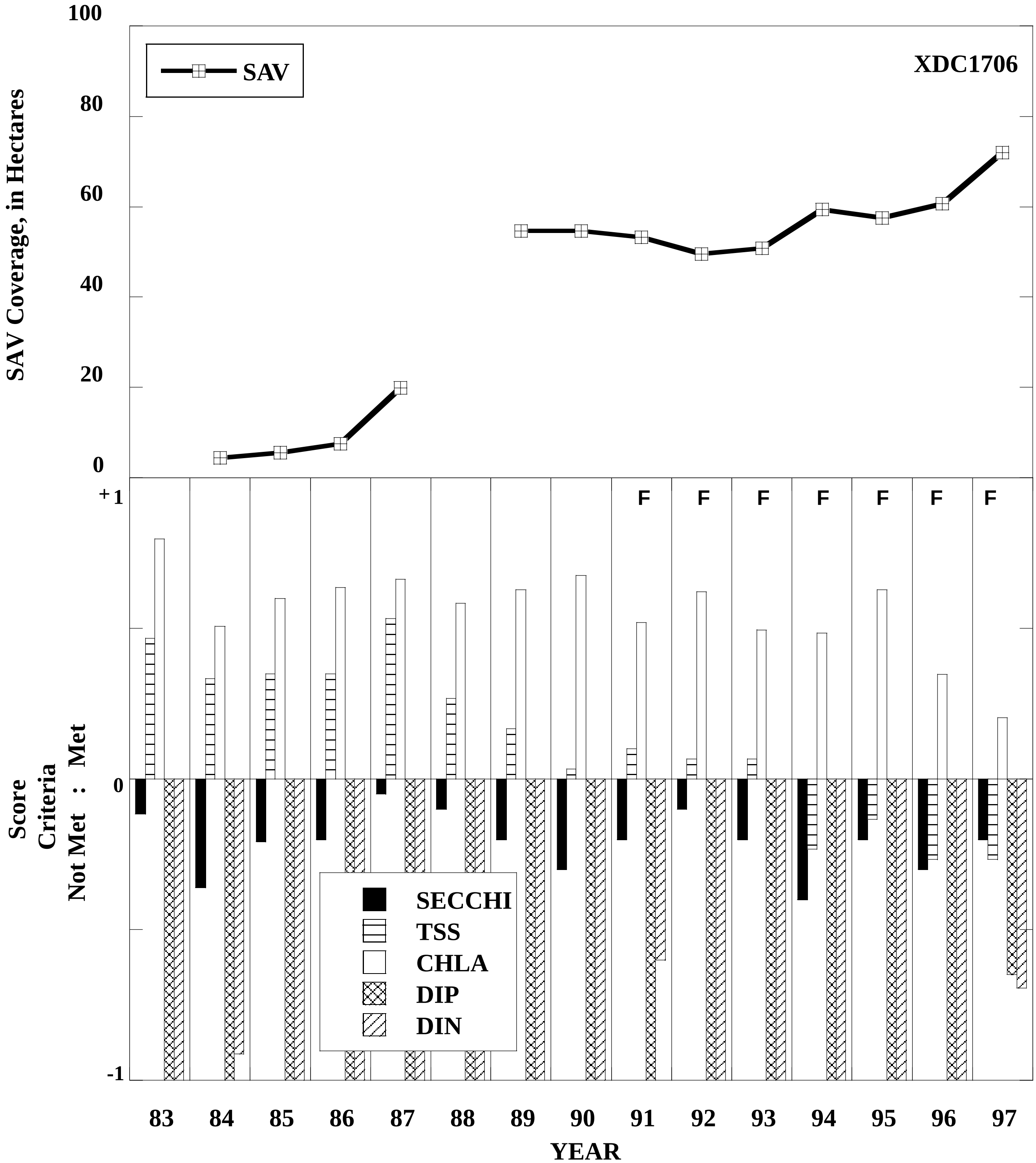

Figure 13. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for mesohaline Potomac Estuary monitoring station XDC1706 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIN and DIP.)

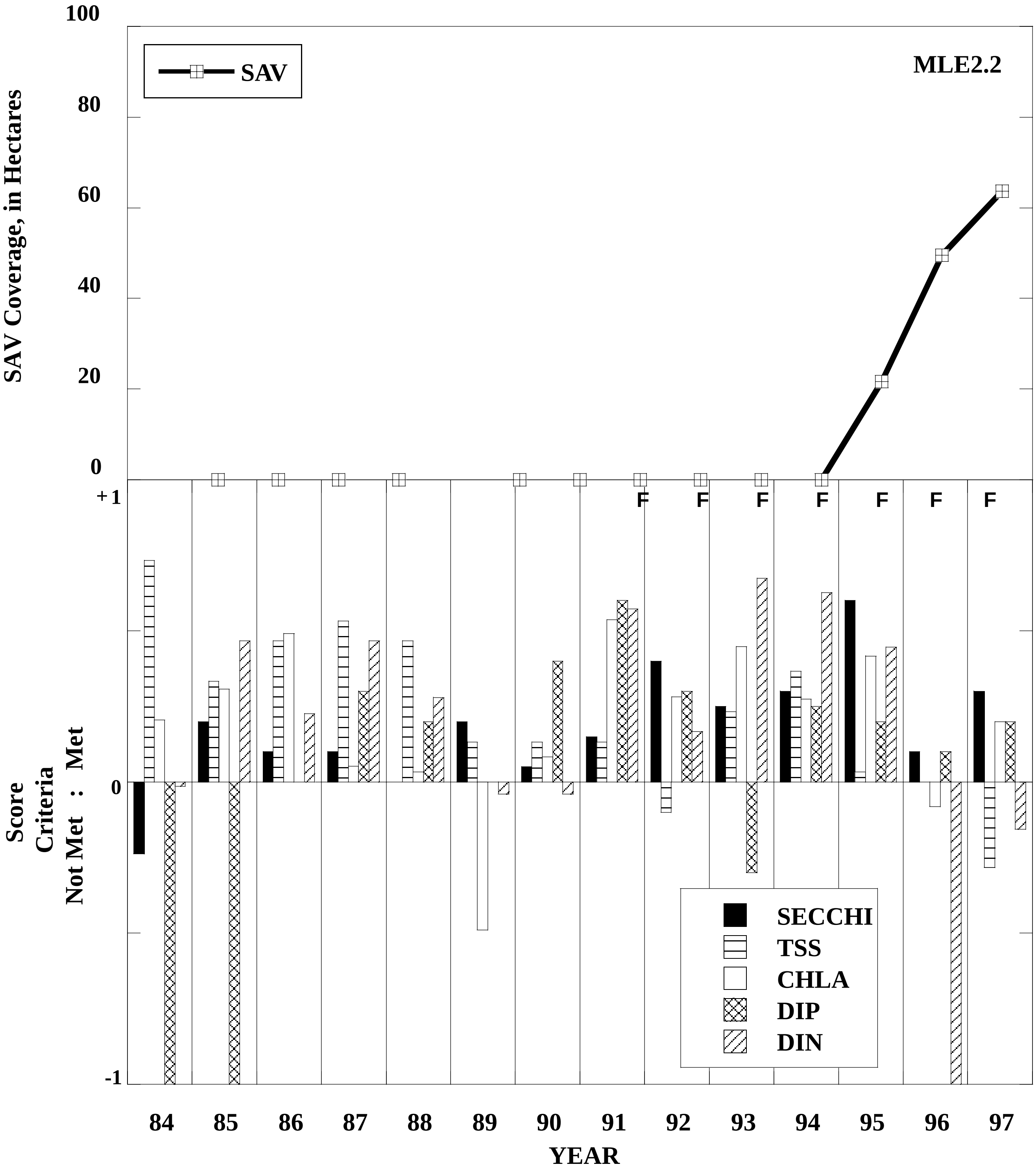

Figure 14. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for mesohaline Potomac Estuary monitoring station MLE2.2 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIN and DIP.)

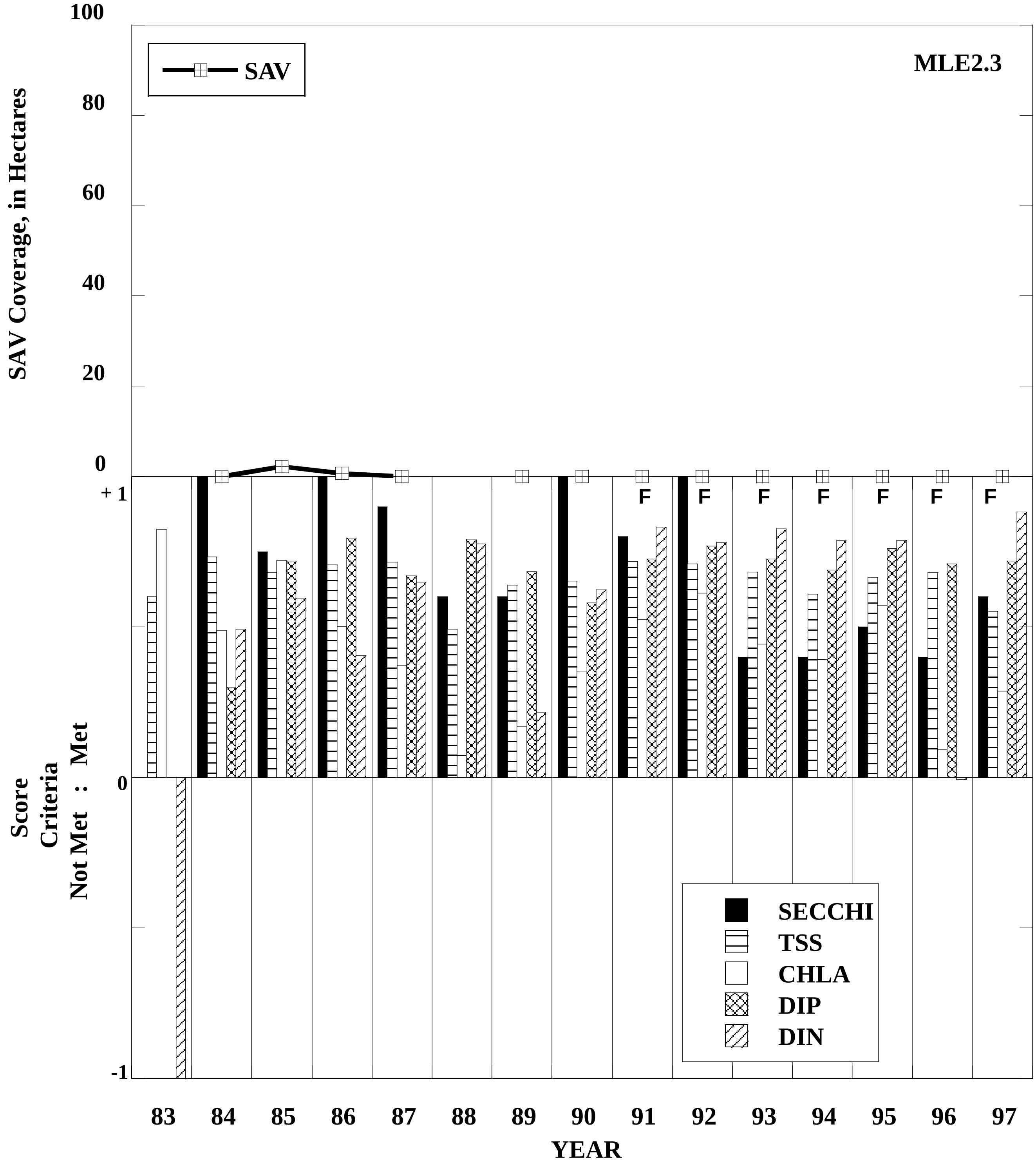

Figure 15. Surface area, in hectares, covered by submersed aquatic vegetation and Chesapeake Bay habitat criteria scores for mesohaline Potomac Estuary monitoring station MLE2.3 segment for the period 1983 through 1997. (F denotes use of filtered water samples for DIN and DIP.)

Tidal Potomac Integrative Analysis Project, A Series of Reports on the Water Quality and Living Resources Responses to Management Actions to Reduce Nutrients in the Potomac River Estuary, Final Draft.

Buchanan, C. [ed.] 1999.
Prepared for the Chesapeake Bay Program.
ICPRB Report 99-4, 268 pp.

APPENDIX F

CHESAPEAKE BAY WATER QUALITY MONITORING PROGRAM: 1995 MESOZOOPLANKTON COMPONENT

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CHESAPEAKE BAY WATER QUALITY MONITORING PROGRAM 1995 MESOZOOPLANKTON COMPONENT

Prepared for

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June 1996

3.2. Potomac River

The current status (1993-1995) of spring zooplankton density in the Potomac River appears to be somewhat lower than that for the spring baseline period (1985-1986) at the upstream and transition zone stations (Figure 3-4). Trends in abundance over time, however, were not significant at either location. At the downstream station, current zooplankton density is extremely high, when compared to the 1985-1986 baseline period. Furthermore, Kendall's T test indicated a significantly increasing long term trend in abundance (113%) at the downstream station.

There appeared to be a significant downward trend in TP (49%) and Chl a (40%) in spring at the upstream station. However, as stated above, there was not a corresponding trend in zooplankton abundance. An 82% reduction in Chl a at the transition Potomac station was the only other parameter examined that exhibited a significant trend in the spring.

The current status of summer zooplankton density appears to be somewhat greater at all three Potomac River stations when compared to the baseline period (Figure 3-5). However, none of the observed increases were statistically significant, indicating a general lack of a long term trend over the eleven year period. There were no significant long term trends in the nutrient data at any of the three stations in summer.

The Potomac fall zooplankton data indicated an increase in the current status at the upstream and downstream stations and a decrease in current status from the baseline period at the transition zone station (Figure 3-6). However, conclusions drawn from these findings should be viewed with caution, as none of the three stations showed a significant long term trend of increasing or decreasing abundance. The findings merely indicate that the current period exhibits higher densities than the baseline period. Because there was a lack of a consistent trend, it can be surmised that data in the middle years (1987-1992) were more randomly distributed about some mean value for the entire time period. Except for a significant increase in Chl a at the transition zone station, the nutrient data for the fall did not indicate any other significant trends.

The Potomac River, unlike the Patuxent, has shown little in the way of consistent long term trends in seasonal abundance. While the current status of zooplankton density may be higher or lower than the baseline period, in most cases this was not reflective of a significant trend. For example, the current status of the spring zooplankton data at the upstream station was lower than the baseline period. Examination of the historical data indicated that in both 1985 and 1986, spring zooplankton densities were relatively high, when compared to data from most subsequent springs. Data collected from 1993 and 1994 were more consistent with those from the 1987-1992 period, than from the baseline period.

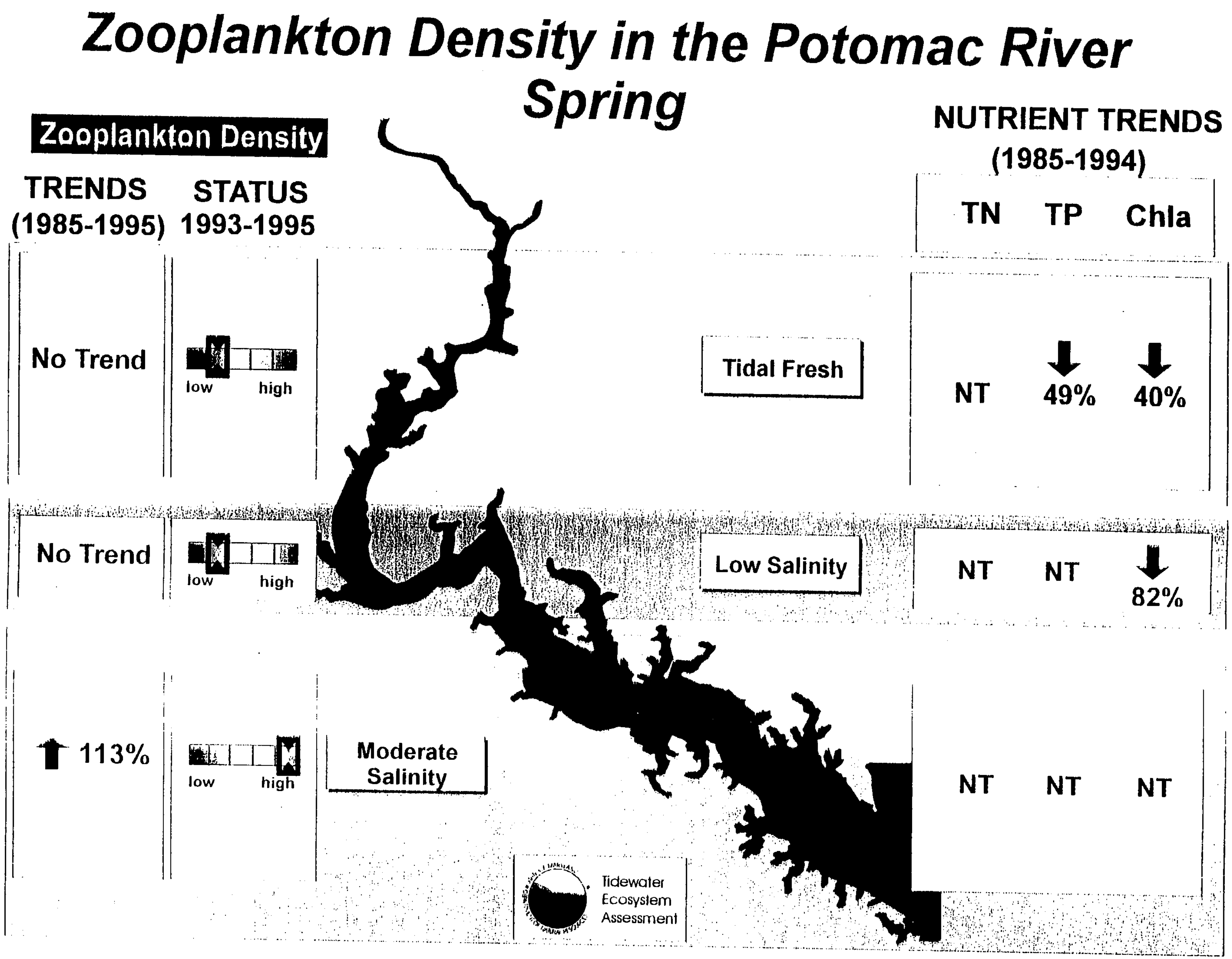

Figure 3-4. Zooplankton Status and Trends for the Potomac River, spring.

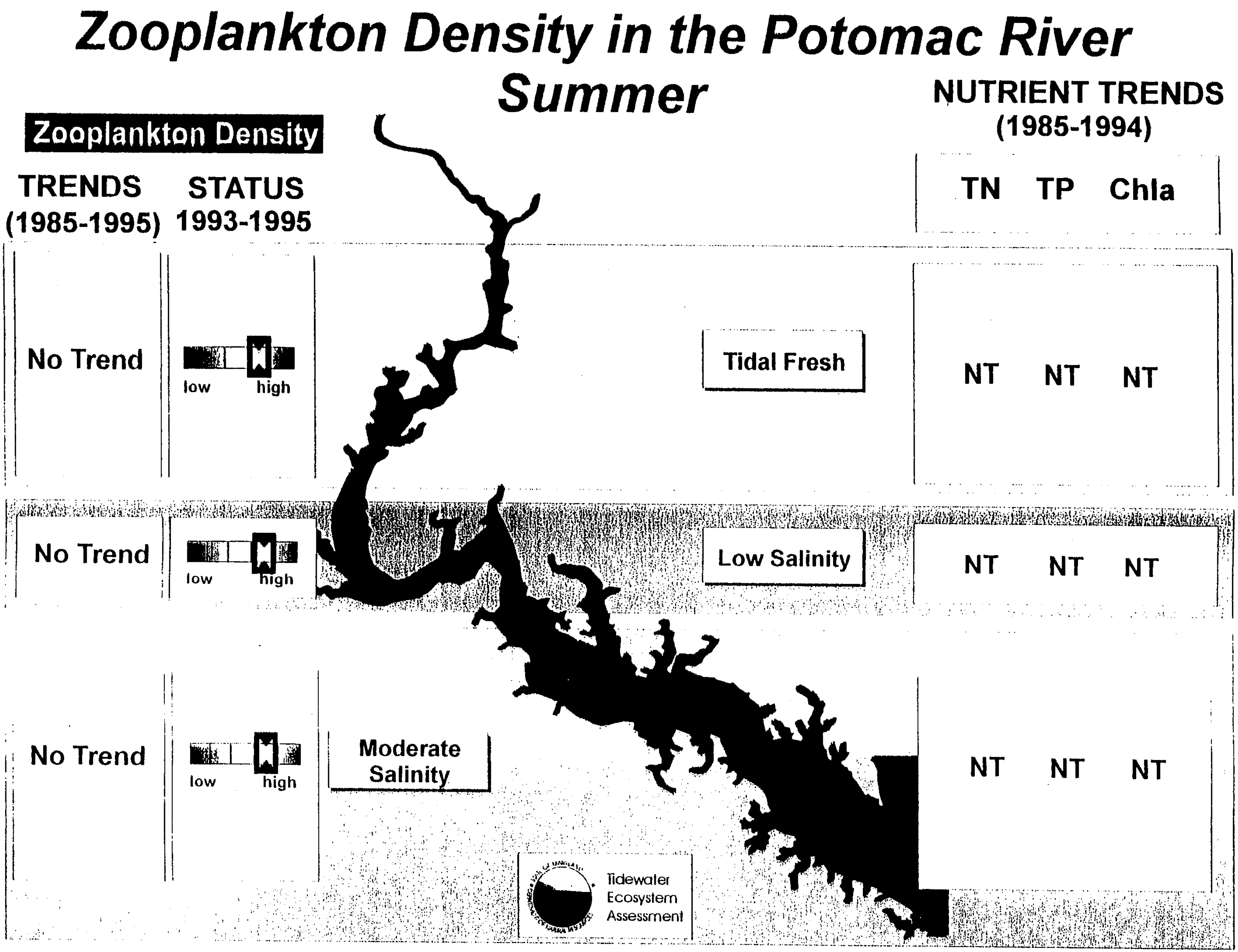

Figure 3-5. Zooplankton Status and Trends for the Potomac River, summer.

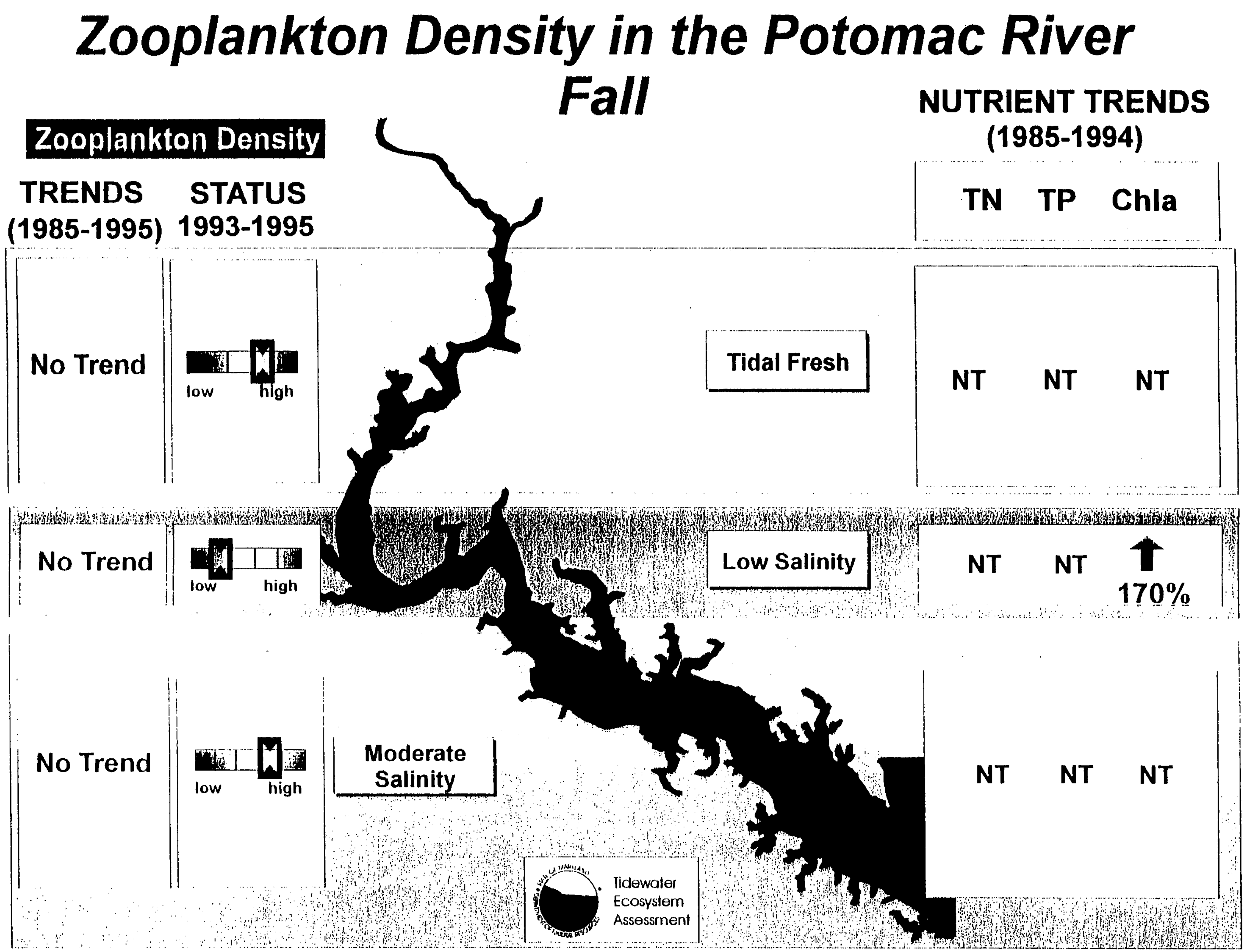

Figure 3-6. Zooplankton Status and Trends for the Potomac River, fall.

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Buchanan, C. [ed.] 1999.
Prepared for the Chesapeake Bay Program.
ICPRB Report 99-4, 268 pp.

APPENDIX G

VERSAR FINAL REPORT FOR THE MARYLAND DEPARTMENT OF NATURAL RESOURCES TIDEWATER ASSESSMENTS

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EXCERPTED FROM

VERSAR FINAL REPORT FOR THE
MARYLAND DEPT. NATURAL RESOURCES
TIDEWATER ASSESSMENTS

1997

5.0 POTOMAC RIVER TRENDS

5.1 INTRODUCTION

Billions of dollars have been spent since the 1960s to improve wastewater treatment facilities (Bennett et al. 1986) and reduce non-point source nutrient inputs to the Potomac river. These efforts have been successful in reducing pollutants entering the river. Total Biological Oxygen Demand from point source loadings decreased by 92% between 1971 and 1985, and phosphorus loadings were reduced by 97% from 1972 to 1985 (Tsai et al., 1991). These changes resulted largely from advanced chemical treatment instituted at the Blue Plains facility in the early 1970s; Blue Plains is the largest treatment facility on the river and accounts for over 77% of its waste water loadings.

Some aspects of Potomac River ambient water quality have improved since the 1970s, particularly in low mesohaline and tidal freshwater habitats. The State of Maryland Department of Natural Resources (DNR) has maintained monthly monitoring of ambient water quality at eight Potomac River stations since 1974 and a ninth since 1984 (Figure 5-1). Phosphorus concentrations have decreased at six, and chlorophyll *a* concentrations have decreased at five of these nine stations (Table 5-1, Skelly et al. in press). Surface chlorophyll *a* concentrations decreased by 25 to 44%. Not all water quality measures have improved, however. Nitrogen concentrations increased significantly at six of the nine stations and dissolved oxygen (DO) concentrations hardly changed at all (Table 5-1). No trend in DO was detected at six of the nine stations. Of the other three, an increasing DO trend was detected at the uppermost tidal freshwater station while decreasing DO trends were detected at two mesohaline stations (Table 5-1).

The water pollution abatement efforts were intended to improve the quality and quantity of living resources in the river and they have been successful in improving the quality of plant populations. The frequency of nuisance blue-green algal (*Microcystis aeruginosa*) blooms has decreased since the early 1980s (Lacouture and Brownlee, 1995). There has also been a resurgence of submersed macrophytes in the river since 1983 (Carter et al. 1986).

The objective of this chapter is to assess whether improvements in living resources further along the food chain, specifically in Potomac River benthic assemblages, have also occurred during this period. The analyses presented here are for data collected at fixed sites analyzed in Chapter 3 (the seven current Potomac fixed sites), but these analyses go beyond those of Chapter 3 in several ways. First, the data record was extended by incorporating compatible historic data collected prior to the initiation of LTB in 1984. Second, Chapter 3 focused on a single indicator, the Benthic Index of Biological Integrity (B-IBI), whereas several assemblage measures and abundance of dominant taxa are analyzed in the present chapter. Finally, spring and summer data are analyzed here; Chapter 3 presented only summer data because the B-IBI has been developed only for summer.

Table 5-1 Trends in selected water quality measures at Maryland Department of Natural Resources Potomac River monitoring stations (from Skelly et al. in press). Results are for 20 years (1974-1994) except Station MLE2.2 (ten years; 1984-1994). *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$; NS: not significant; NR: not reported.

Habitat	Station	Total Nitrogen	Total Phosphorus	Chlorophyll *a*	Bottom Dissolved Oxygen Concentration
Tidal Freshwater	XFB2470	37% ***	-29% ***	NS	39% **
	XFB1433	32% ***	-29% ***	NS	NS
	XEA6596	42% ***	-29% ***	-27% **	NS
	XEA1840	46% ***	-60% ***	-43% ***	-33% **
Oligohaline	XDA4238	NS	-36% ***	-25% ***	NS
	XDA1177	18% *	-22% **	-34% ***	NS
Low Mesohaline	XDB3321	30% ***	NS	-44% *	NS
	XDC1707	NS	NR	NS	-31% **
High Mesohaline	MLE2.2	NS	NR	NS	NS

5.2 METHODS

Data from the seven LTB fixed sites in the Potomac (Figure 5-2, Table 2-1) were used to test for trends in benthic condition in order to analyze them for trends. Six of the seven stations had been sampled prior to initiation of LTB in 1984 as part of DNR's Power Plant Siting Program (Table 5-2). Those data, which commenced in 1981, were added to the LTB data base for these analyses. No pre-LTB data were available for tidal freshwater Station 36. Sample collection and laboratory processing techniques for the added data were the same as for LTB (Table 5-2); they are described in detail in Chapter 2.

Table 5-2. Habitat, sampling period, and sampling devices used at the seven LTB fixed stations located in the Potomac River. Mud > 40%, sand < 40% silt/clay sized particles.

Habitat	Station	Depth (m)	Sampling Period	Sampling Device
Tidal Freshwater Mud	36	3	July 1984 to August 1995	Hand-operated box corer: July 1984 to September 1989 Wildco box-corer: October 1989 to August 1995
Oligohaline Mud	40	8	July 1981 to August 1995	Petite ponar grab: July 1981 to June 1989 Wildco box-corer: July 1989 to August 1995
Mesohaline Sand	43	2	July 1981 to August 1995	Hand-operated box corer: July 1981 to August 1995
	47	2	July 1981 to August 1995	Hand-operated box corer: July 1981 to August 1995
	51	2	July 1981 to August 1995	Hand-operated box corer: July 1981 to August 1995
Low Mesohaline Mud	44	14	July 1981 to August 1995	Petite ponar grab: July 1981 to June 1989 Wildco box-corer: July 1989 to August 1995
High Mesohaline Mud	52	11	July 1981 to August 1995	Petite ponar grab: July 1981 to June 1989 Wildco box-corer: July 1989 to August 1995

5.3 RESULTS

5.3.1 Sediment Carbon Content

Sediment carbon content decreased significantly at six of the seven stations over the study period (Table 5-3). At mesohaline sand Stations 43,47 and 51, sediment carbon content declined from values always above 0.5% and often in the 1-2% range in the early 1980s, to values which were always below 0.5% in the late 1980s and 1990s (Figure 5-3). At the three mud stations furthest up the river (tidal freshwater Station 36, oligohaline Station 40, and low mesohaline mud Station 44) sediment carbon values decreased from highs over 3% in the early 1980s to values consistently from 2-2.5% in the late 1980s and 1990s (Figure 5-4). The highest median rate of decline, 1% every five years, was observed at Station 36, which was farthest up the river, in spite of its shorter, ten year, data record. Only high mesohaline mud Station 52, which was farthest down the river, displayed no significant carbon content trend.

data gap unfortunate

Table 5-3. Median annual rate of change (Sen 1968) and results of van Belle and Hughes (1984) trend test on sediment carbon content. Results are for spring and summer combined trends. *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$.

Habitat	Station	Median rate of carbon content change (%/year) and trend analysis result
Tidal freshwater	36	-0.20 **
Oligohaline	40	-0.06 *
Mesohaline sand	43	-0.13 ***
	47	-0.19 ***
	51	0.10 ***
Low mesohaline mud	44	-0.15 ***
High mesohaline mud	52	-0.03

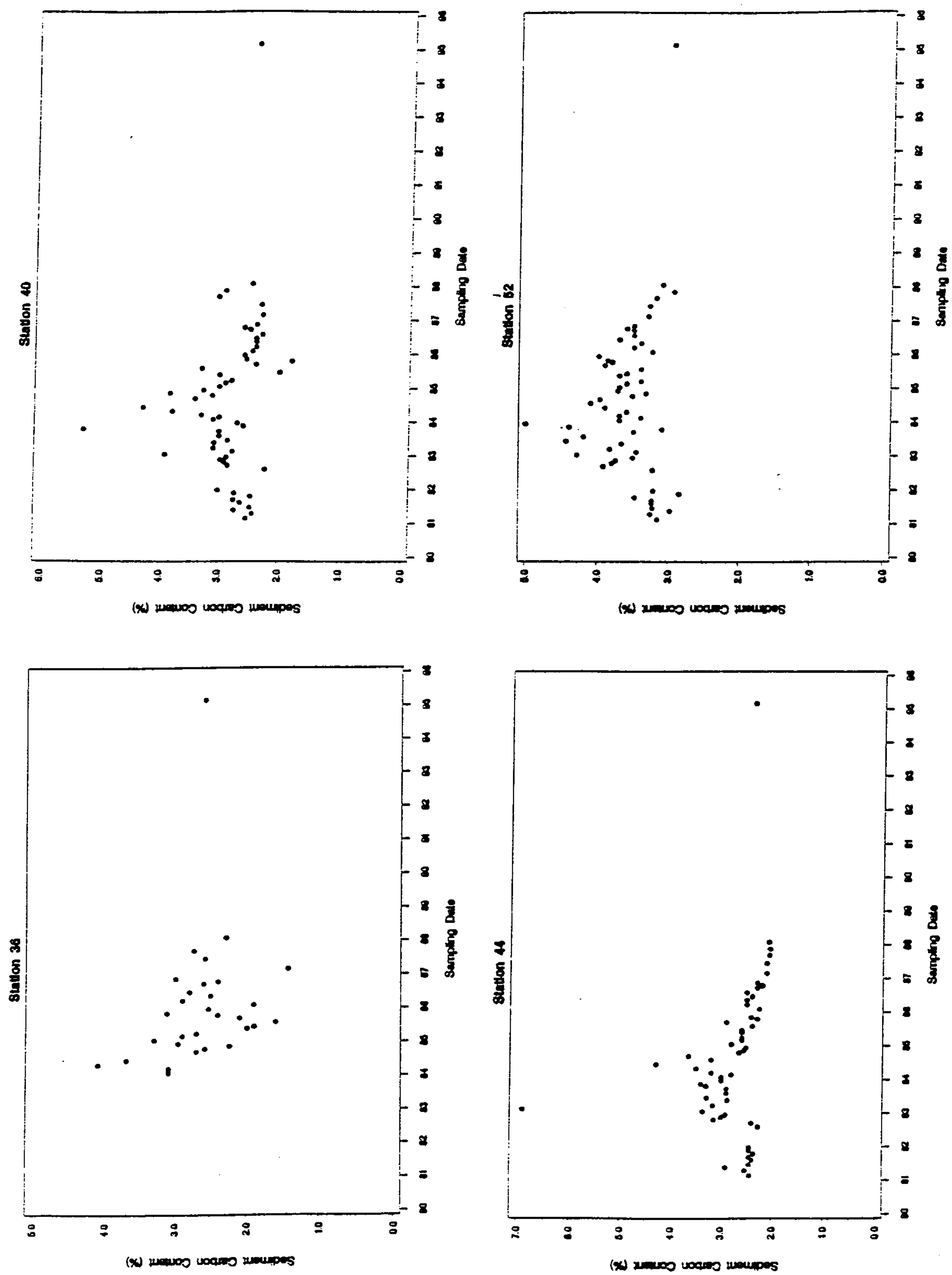

Figure 5-4. Sediment carbon content at tidal freshwater Station 36, oligohaline Station 40, low mesohaline mud Station 44, and high mesohaline mud Station 52

Table 5-5. Median annual rate of change (Sen 1968) and results of van Belle and Hughes (1984) trend test on benthic assemblage measures. Results are for spring and summer combined trends. +: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$; N/A: Not applicable.

	Tidal Freshwater Habitat	Oligohaline Habitat	Mesohaline Sand Habitat			Low Mesohaline Mud Habitat	High Mesohaline Mud Habitat
Measures	Station 36	Station 40	low Station 43	low Station 47	high Station 51	Station 44	Station 52
Benthic Index of Biotic Integrity	N/A	-0.03	-0.03 *	-0.03 +	0.00	0.03	-0.04 *
Number of Taxa (#/sample)	0.00	0.17 +	0.08 *	0.33 +	0.24 **	0.22 ***	0.00
Total Abundance (#/m^2)	-220.00	-30.91	0.00	204.44 *	104.76 *	30.85	8.76
Suspension Feeder Abundance (#/m^2)	-191.11 *	-2.00	12.44 *	15.00 +	3.64	0.00	0.00
Interface Feeder Abundance (#/m^2)	0.00	10.15	18.89	106.67 +	86.11 *	37.44 *	13.08
Deep Deposit Feeder Abundance (#/m^2)	-48.18	-13.33 ***	25.11 *	45.33	-5.71	0.85	0.00
Pollution-Sensitive Taxon Abundance (#/m^2)	0.00	-8.28	-1.94	56.67	10.00 +	0.91	0.00
Pollution-Tolerant Taxon Abundance (#/m^2)	0.00	0.00 *	1.44	0.00	12.22	8.89	3.15
Total Biomass (g/m^2)	-9.90	-0.08	4.32 +	3.32	-0.26	0.01	0.00
Mollusc Biomass (g/m^2)	-9.99	-0.07 *	3.67	2.71	0.02	0.00	0.00

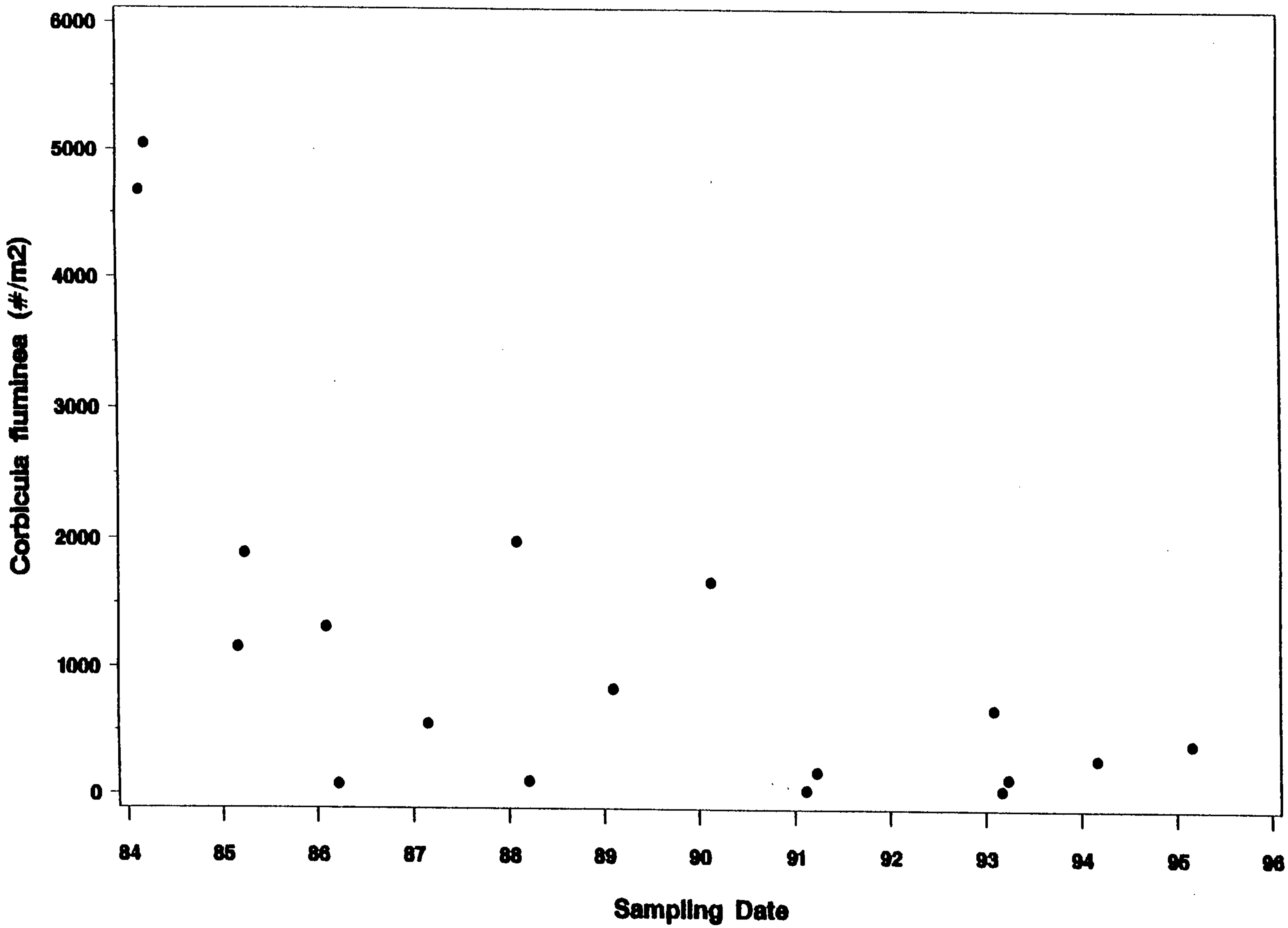

Figure 5-5. Summer abundance of *Corbicula fluminea* at Station 36

were among the ten abundance dominants at all three stations and together accounted for 67.7%, 75.4%, and 75.3% of the total abundance, respectively (Tables 5-7, 5-8, and 5-9). Stations 43 and 47 had identical abundance dominant lists, the other four taxa being the low salinity species *Rangia cuneata* and *Cyathura polita* and the omnivorous *Neanthes succinea* and *Laeonereis culveri*. At Station 51 the other dominants were higher salinity taxa.

Table 5-7. Median annual rate of change (Sen 1968) and results of van Belle and Hughes (1984) trend test on abundances of the ten most abundant taxa at mesohaline sand Station 43. Results are for spring and summer combined trends. +: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$.

Taxon	Feeding Guild [a]	Contribution to Total Abundance (1981-1995)	Trend
Marenzelleria viridis	Interface	42.2	-14.44
Heteromastus filiformis	Deep Deposit	16.5	21.82 +
Leptocheirus plumulosus	Interface	9.3	6.67
Rangia cuneata	Suspension	5.0	14.00 ***
Macoma mitchelli	Suspension	4.7	14.44 **
Laeonereis culveri	Carnivore/Omnivore	4.8	-0.07
Cyathura polita	Carnivore/Omnivore	4.0	0.77
Streblospio benedicti	Interface	2.4	1.67
Neanthes succinea	Carnivore/Omnivore	1.9	1.11
Macoma balthica	Interface	1.9	-3.64 +

Significant increasing trends in the number of taxa per sample were detected at all three stations (Table 5-5). On average, a taxon would be added to each sample every 12 years at Station 43, every three years at Station 47, and every four years at Station 51. Significant decreasing trends in B-IBI were detected at Stations 43 and 47, although at a low rate. In 1995, Stations 43 and 47 met the Restoration Goals, although Station 51 did not.

Increasing trends in benthic abundance were detected at all three stations; they were statistically significant at Stations 47 and 51 (Table 5-5). Strong, significant, increasing trends in interface feeder abundance were detected at Stations 47 and 51, although increasing trends were

Table 5-9. Median annual rate of change (Sen 1968) and results of van Belle and Hughes (1984) trend test on abundances of the ten most abundant taxa at mesohaline sand Station 51. Results are for spring and summer combined trends. +: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$.

Taxon	Feeding Guild (a)	Contribution to Total Abundance (1981-1995)	Trend
Heteromastus filiformis	Deep Deposit	27.4	-2.50
Streblospio benedicti	Interface	16.6	5.69
Macoma mitchelli	Suspension	15.7	31.87 *
Marenzelleria viridis	Interface	11.3	3.33
Micrura leidyi	Carnivore/Omnivore	3.0	0.00
Lepidactylus dytiscus	Interface	2.6	3.33 ***
Mulinia lateralis	Suspension	2.3	0.00
Leptocheirus plumulosus	Interface	2.3	0.00
Carinoma tremaphoros	Carnivore/Omnivore	2.0	5.93 ***
Macoma balthica	Interface	2.0	0.00

5.3.5 Low Mesohaline Mud (Station 44) Benthos

The assemblage encountered at the low mesohaline mud station was quite similar to the mesohaline sand assemblage. It shared all six of the top ten abundance dominants that were common among the three mesohaline sand stations (Table 5-10).

A significant increasing trend in number of taxa per sample, was detected with the increase equal to about one taxon every 4.5 years detected (Table 5-5). No significant trend in B-IBI was present. Station 44 did not meet the Restoration Goals in 1995.

No significant trend in total abundance was detected, and an increasing trend in interface feeder abundance was the only significant abundance trend detected. However, all the abundance measure rates of change were in a positive direction.

Significant increasing trends were observed for five of the ten most abundant taxa (Table 5-10). However, the median rates of change were insignificant.

Table 5-11. Median annual rate of change (Sen 1968) and results of van Belle and Hughes (1984) trend test on abundances of the seven most abundant taxa at high mesohaline mud Station 52. Results are for spring and summer combined trends. +: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$.

Taxon	Feeding Guild	Contribution to Total Abundance (1981-1995)	Trend
Streblospio benedicti	Interface	82.4	14.55
Mulinia lateralis	Suspension	3.8	0.00
Neanthes succinea	Carnivore/Omnivore	3.6	0.00
Hypereteone heteropoda	Carnivore/Omnivore	2.2	0.00
Paraprionospio pinnata	Interface	1.5	0.00
Macoma mitchelli	Suspension	1.2	0.00
Heteromastus filiformis	Deep Deposit	1.0	0.00 +

No trend in number of taxa per sample was detected (Table 5-5). A decreasing trend in B-IBI was detected and Station 52 failed the Restoration Goal. On the scale used to classify Baywide random samples (Chapter 2), it was classified as severely degraded.

No significant trends in total abundance, or feeding guild or pollution sensitive or pollution indicative taxon abundance (Table 5-5) were detected. The median rates of change were also insignificant. Similarly, no significant dominant taxon abundance trends (Table 5-11) or biomass trends (Table 5-5) of any magnitude were detected.

5.4 DISCUSSION

Measurable improvements in water quality of the Potomac River have been documented by several researchers. One of the major water quality improvements has been a reduction in chlorophyll *a* concentrations. Reduction in chlorophyll *a* in the water column can affect benthic habitats in at least two ways. First, the amount of decaying matter reaching the bottom may decrease sediment carbon concentrations. Second, a decline in organic matter may reduce metabolism and increase oxygen concentrations. Improvements in bottom water dissolved oxygen concentrations have not been reported, but the results of our trend analysis on sediment carbon concentrations show significant decreases at six of the seven stations.

enough, or did not cross a significant enough threshold, to effect a benthic response. Benthos at many of the sites were meeting Restoration Goals prior to the carbon decline and hypoxic stress was apparent at the sites that were not. While the effects of high carbon concentrations on benthos are well documented, the thresholds at which benthic response begins to occur is not.

Sediment carbon concentration is a potentially important mechanism linking nutrient reduction efforts in Chesapeake Bay with living resources. It is also potentially useful as an integrator of nutrient reduction efforts since it is less temporally variable than water column nutrient or chlorophyll *a* measures. The probability based element of LTB provides a robust mechanism for measuring the effectiveness of nutrient controls in reducing bay-wide sediment carbon concentrations. Future LTB efforts will investigate the relationship between carbon levels and benthic response to better define relevant carbon thresholds for interpreting trends in benthic data and which can serve as a means for establishing management goals in nutrient reduction efforts.