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Water Quality Associated With Survival of Submersed Aquatic Vegetation Along an Estuarine Gradient

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ABSTRACT: The decline of submersed aquatic vegetation (SAV) in tributaries of the Chesapeake Bay has been associated with increasing anthropogenic inputs, and restoration of the bay remains a major goal of the present multi-state "Bay Cleanup" effort. In order to determine SAV response to water quality, we quantified the water column parameters associated with success of transplants and natural regrowth over a three-year period along an estuarine gradient in the Choptank River, a major tributary on the eastern shore of Chesapeake Bay. The improvement in water quality due to low precipitation and low nonpoint source loadings during 1985–1988 provided a natural experiment in which SAV was able to persist upstream where it had not been for almost a decade. Mean water quality parameters were examined during the growing season (May–October) at 14 sites spanning the estuarine gradient and arrayed to show correspondence with the occurrence of SAV. Regrowth of SAV in the Choptank is associated with mean dissolved inorganic nitrogen <10 µM; mean dissolved phosphate <0.35 µM; mean suspended sediment <20 mg l^{-1}; mean chlorophyll *a* in the water column <15 µg l^{-1}; and mean light attenuation coefficient (Kd) <2 m^{-1}. These values correspond well with those derived in other parts of the Chesapeake, particularly in the lower bay, and may provide managers with values that can be used as target concentrations for nutrient reduction strategies where SAV is an issue.

Introduction

The general observation that dense phytoplankton blooms are not compatible with significant populations of submersed macrophytes on the bottom of aquatic systems is now almost a century old (Koifoid 1903). Indeed, the relationship between increasing nutrients in the water column and the resulting stimulation of algal components of the system at the expense of submersed macrophytic vegetation is now among the classic "text book" paradigms (Wetzel 1975). Eutrophication severely limits the potential for the growth of submersed aquatic macrophytes not only by promoting planktonic algal blooms as Swingle (1947) demonstrated in fish pond management, but also by promoting excessive epiphytic and filamentous algal overgrowth (Phillips et al. 1978).

Evidence for the negative impacts of eutrophication on submersed macrophytes spans northern and southern hemispheres in marine as well as freshwater environments (Stevenson 1988). For example, nutrient loading of coastal salt ponds in New England has been shown to enhance marine macroalgae at the expense of seagrass species (Lee and Olsen 1985; Valiela and Costa 1988) and appears to be associated with a significant decline of seagrasses in Cockburn Sound, Australia (Shepherd et al. 1989). Despite these and laboratory studies (Gerloff and Krombholz 1966) establishing critical tissue nutrient concentrations for submersed aquatic vegetation (SAV), as well as dimensionless plots of responses of macrophytes to nutrients in lakes (Wetzel and Hough 1973; Phillips et al. 1978), there is a lack of quantification of nutrient concentrations associated with the demise (and/or regrowth) of macrophytes in coastal systems, including estuaries. This information gap is partially due to problems of assembling long-term concentration data in spatially and temporally variable environments where significant oscillations in abundances of submersed macrophytes are occurring.

The decline of SAV populations in Chesapeake Bay during the 1970s and early 1980s (Stevenson and Confer 1978; Orth and Moore 1983, 1984),

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followed by recent increases in abundance, make this an ideal estuarine system in which to look at temporal, as well as spatial, responses to water quality changes. Initial analysis of the causes for the Chesapeake Bay SAV decline in the 1970s suggests correlation with nonpoint source pollutants (Stevenson and Confer 1978). Later studies (Kemp et al. 1983; Staver 1984) focused attention on nutrient enrichment as the principal cause for the SAV decline.

Kemp et al. (1983) postulated that increased nutrient loadings of the Chesapeake in the 1970s enhanced growth of planktonic and epiphytic algal species which compete with SAV for light. In follow-up studies, Twilley et al. (1985) showed that even low levels of nutrient loading (0.42 g N m^2 d^{-1} and 0.09 g P m^2 d^{-1}) to mesocosms receiving ambient water from the lower Choptank estuary caused a 50% reduction in SAV biomass. Furthermore, at high nutrient concentrations, one species (*Ruppia maritima*) was virtually eliminated. Twilley et al. (1985) concluded that interception of light by epiphytic and filamentous algal species on leaf surfaces was the main factor for the SAV decline. Along with this scenario of nutrient effects, SAV productivity is reduced further when suspended sediment increases in the water column, exacerbating light attenuation problems in the bay (Wetzel and Penhale 1983; Kemp et al. 1984).

Microcosm experiments, used to further explore the effects of nutrient enrichment and light reduction on algal competitors and SAV (Staver 1984; Goldsborough and Kemp 1988), confirmed the importance of eutrophication in driving the SAV decline in Chesapeake Bay. Various models (Wetzel and Neckles 1986; Stevenson 1988) have been presented to illustrate the interaction of these factors and their effect on SAV. Although we now have extensive knowledge of the driving forces, it has been difficult to predict quantitatively with any degree of precision what nutrient regimes as well as what sediment and light levels in the estuary promote success or failure of SAV populations.

As an extension of previous experiments and modeling, we took an empirical field approach to this problem. Here we report the water column conditions along an estuarine gradient during low runoff years when water quality improved markedly and SAV populations expanded in the lower Choptank River estuary. We quantified changes in water quality moving upstream along the estuarine gradient into an environment where natural populations of SAV did not rebound and efforts at transplanting failed. In order to assess when these water quality parameters appear most important to SAV growth, we analyzed the historical phenological biomass distribution patterns of major species in mid-Chesapeake Bay when SAV populations were closer to their full potential than at present. Finally, we identified levels of key water column parameters associated with the success and failure of SAV in the shallows of a mid-Chesapeake Bay tributary.

Study Area

The Choptank River is the largest tributary on the eastern shore of Chesapeake Bay (estuarine surface area = 366 km^2), draining into the main stem approximately 185 km from the bay mouth (Fig. 1). It is a comparatively well-mixed (top to bottom), shallow tributary (Ward and Twilley 1986). Bathymetric data (Cronin and Pritchard 1975) indicate that 15,330 ha are potential SAV habitat (i.e., 3 m or less deep). There is a sill at the mouth of the Choptank which isolates it from anoxic subpycnocline waters of the mainstem Chesapeake during the summer (Sanford and Boicourt 1990), thus preserving its downstream water quality. The Choptank is ideal for this study because it has a well-developed longitudinal water quality gradient with progressively increasing concentrations of nutrients upriver (Ward and Twilley 1986), and a comparably well-documented history of SAV distribution (Stevenson and Confer 1978).

Annual surveys of SAV have been carried out by the Maryland Department of Natural Resources (Md.DNR) for more than a decade (L. Hindman personal communication) at 30 sites in the Choptank (Table 1). Additional surveys (Stevenson and Confer 1978) indicate that in the 1960s SAV was found 60 km upriver at Hog Island (Fig. 1). However, SAV declined rapidly in this region during the 1970s, becoming confined to restricted locations in the lower Choptank by 1975. These reduced populations remained relatively stable during the late 1970s, before declining severely in 1981, when less than 2% of the stations had SAV (Table 1).

Several municipalities (Fig. 1) discharge sewage treatment plant (STP) effluent into the river, most notably the 4.1 million-gallons-per-day (MGD) plant at Cambridge (37 km upstream) and the 1.7 MGD plant at Easton (61 km upstream). Upstream of the confluence with the Tuckahoe Creek (72 km upstream), STP discharges are much smaller (e.g., 0.29 MGD at Denton and 0.12 MGD at Greensboro). Land use in the Choptank River watershed is 29% forested and 66% agricultural (Lomax and Stevenson 1982). The remaining area is in small municipalities and residential development, which has been recently increasing. Previous studies in which rates of nutrient loading were estimated (Lomax and Stevenson 1982), indicate that nonpoint sources dominate long-term nitrogen (N) and phos-

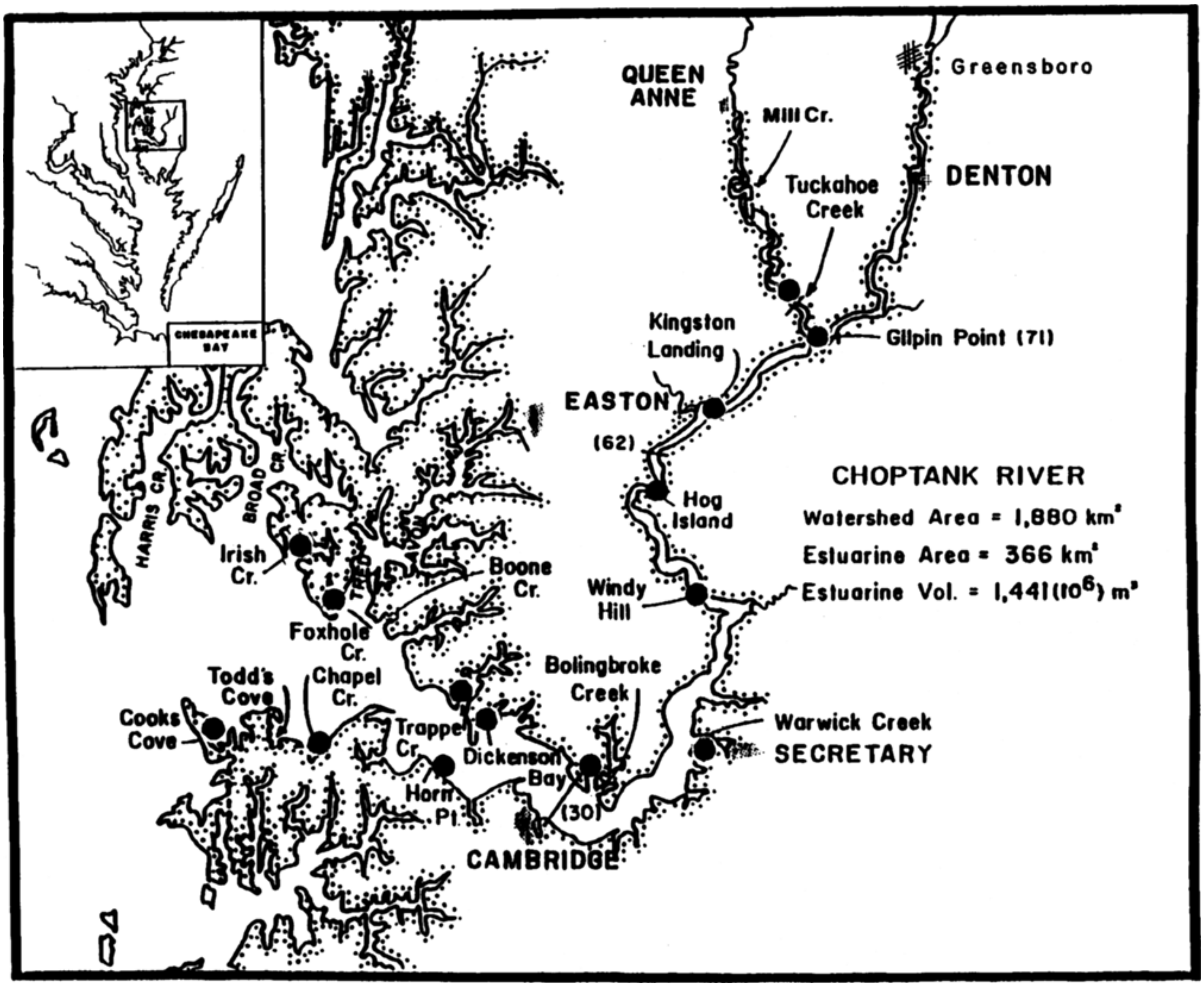

Fig. 1. Map of the Choptank Estuary showing distances upriver from the mouth (km in parentheses) and water quality sampling locations.

phorus (P) inputs, contributing 70–90% of the total N and 50–80% of total P in an annually variable precipitation-driven pattern (Stevenson et al. 1986).

Methods

SAV Biomass, Transplanting, and Distribution

The distribution of SAV over the range of 14 sampling stations was recorded during each growing season from 1985 to 1988 and was compared to the annual survey carried out by the Maryland Department of Natural Resources (L. Hindman unpublished data). We also utilized information from aerial surveys (Orth et al. 1986, 1987; Orth and Moore 1988; Orth and Nowak 1990) as well as oblique aerial photographs of selected portions of the shallows taken with low flying aircraft.

Experimental transplanting of SAV was initiated in order to insure that propagules were not limiting the establishment of viable populations along the estuarine gradient. Transplanting of SAV was carried out by removing plugs of sediment containing living plants (*Ruppia maritima, Potomogeton perfoliatus*, and *Potomogeton pectinatus*) from the brackish ponds at Horn Point and placing them in 15-cm diameter biodegradable pots. The pots were transported via small boats to shallow sites and planted 1 m apart on 11 m × 11 m grids. The corners were marked with wooden stakes so that survival could be determined in later months (and years). Transplanting was carried out in the spring and early summer months of 1985–1987 at (1) Horn Point, (2) Chapel Creek, (3) Todd's Cove, (4) Irish Creek, (5) Foxhole Creek, (6) Boone Creek, (7) Dickinson Bay, (8) Bolingbroke Creek, (9) Goose Creek, and (10) Warwick Creek (Fig. 2). Additional seeding of *Zostera marina* in the summer of 1988 was carried

TABLE 1. Mean Seasonal Freshwater Discharge ($m^3 s^{-1}$) at the head of the Choptank River (Greensboro United States Geological Survey Station) and estimated SAV abundance 1977–1988.

Year	Discharge at Greensboro Fall	Winter	Spring	Summer	Percent of stations vegetated[a]	Estimated Coverage (ha) from Aerial Photographs[b]
1977	1.93	2.62	1.30	0.40	25.8	
1978	3.85	11.63	3.95	2.39	28.3	1,740
1979	1.36	13.54	5.28	2.31	26.7	
1980	4.61	5.52	4.85	1.98	25.0	
1981	2.10	2.17	3.64	1.01	1.7	
1982	1.46	6.34	3.80	0.75	6.7	
1983	1.48	6.52	10.65	1.39	5.0	
1984	5.31	9.29	7.27	1.04	1.7	82
1985	0.82	2.64	1.42	1.27	11.7	1,528
1986	2.61	6.09	1.59	0.36	5.0	452
1987	2.52	8.00	2.70	0.31	15.0	
1988	0.48	3.49	3.00	0.57	33.3	

[a] Maryland Department of Natural Resources Data Files.
[b] Aerial Estimates (hectares) from Orth et al. 1986.

out near the mouth at (11) Cooks Point Cove and (12) Tilghman Island (Fig. 2).

The temporal biomass variability in natural populations was determined over the entire gradient in the Choptank in 1989. These data were compared to previous data of sequential harvests of *Myriophyllum spicatum* in Trappe Creek as well as *Potamogeton perfoliatus* and *Ruppia maritima* in Todd's Cove taken in 1977 as part of a study of nitrogen fixation associated with SAV (Lipschultz et al. 1979). Aboveground biomass was determined using three to five 0.25 m^{-2} quadrats; belowground biomass was estimated via 0.1 m^{-2} cores. All material was dried in a forced hot air oven at 60°C and weighed to the nearest 0.1 g.

Water Quality Sampling and Analysis

A series of 14 sampling stations was selected along the axis of the Choptank River, from Cook's Cove, near the mouth, to a point in Tuckahoe Creek, approximately 80 km upstream (Fig. 1). The sites were located along the margins of the river at water depths <3 m in areas of historical or potential SAV habitat. Stations in the lower part of the river were selected in protected coves while those in the upper river (where coves are lacking) were located in shallow areas adjacent to shore but in close proximity to the main channel. All stations were sampled by boat on a monthly basis, with bimonthly sampling

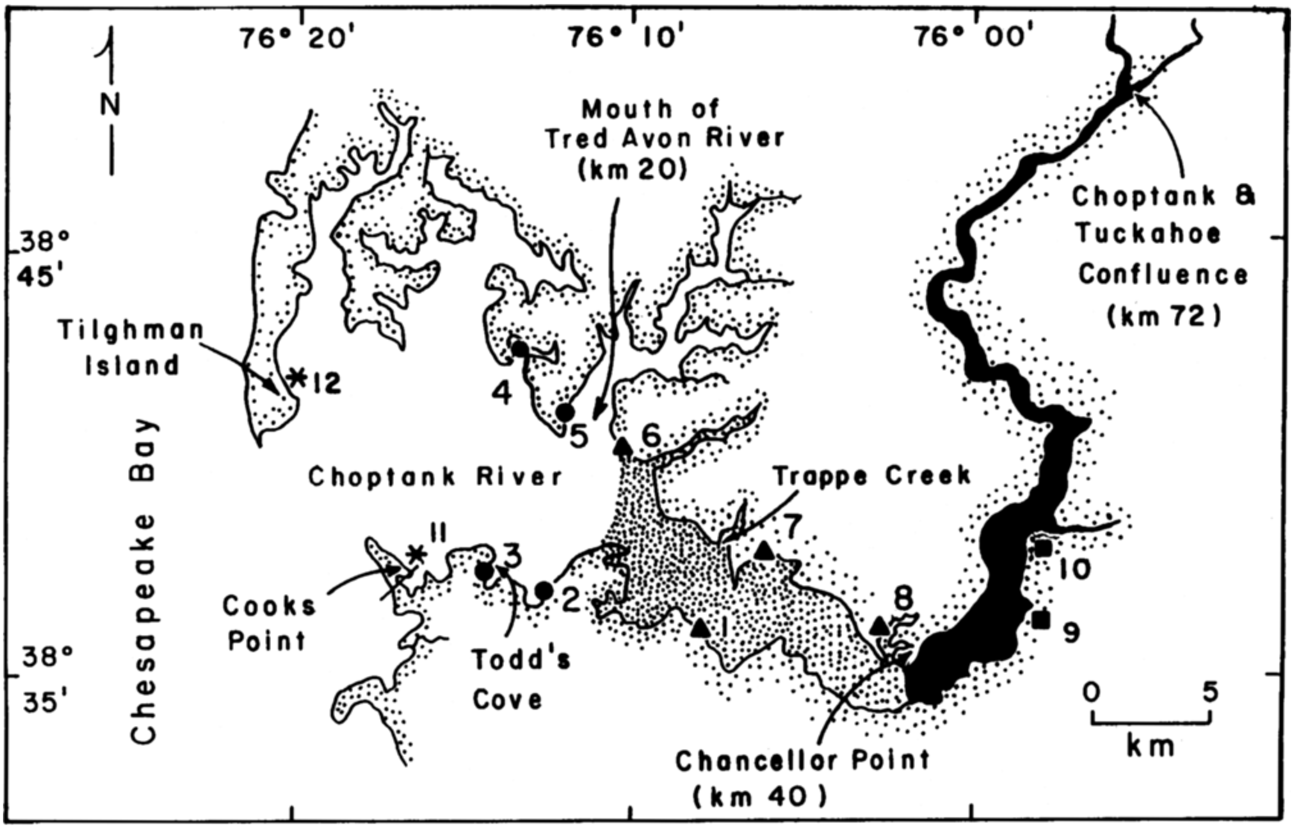

Fig. 2. SAV transplant sites in the Choptank Estuary. Squares indicate no survival, triangles indicate marginal survival beyond several months, and dots indicate no survival at the end of one growing season. Asterisks indicate sites at the mouth of estuary where seeds of *Zostera marina* were planted. Stippled area indicates marginal zone where natural regrowth of SAV was sporadic and blackened area is the zone where no regrowth was observed.

during winter months when access is often restricted by ice.

Nutrient concentrations measured at each station included nitrate (NO_3^-), nitrite (NO_2^-), ammonium (NH_4^+), phosphate (PO_4^{3-}), total nitrogen (TN), total dissolved nitrogen (TDN), total phosphorus (TP), and total dissolved phosphorus (TDP). Water samples for dissolved nutrient analysis were filtered in the field through a Whatman GFC glass-fiber filter (nominal pore size 1.2 μ). Both filtrate and filters were placed on ice, then frozen upon return to the laboratory for later determination of dissolved nutrients and chlorophyll *a*, respectively. Dissolved and total nutrient analyses were made on a Technicon AutoAnalyzer II system. Chlorophyll *a* concentrations were determined fluorometrically (Parsons et al. 1984) on a Turner fluorometer, model 111. Raw water samples were placed on ice in the field and processed within 24 h for total N and P (persulfate digestion, Valderrama 1981). Suspended particulate material (SPM) was measured using a modified gravimetric determination (Banse et al. 1963). Salinity was determined in the field via a Reichert T/C refractometer (Model 10419) and conductivity was later measured in the laboratory with a YSI (model 34) conductance-resistance meter.

Following filtration and SPM determination, particulate C and N were determined by combustion of the precombusted Whatman GFC filter in a Control Equipment Corporation modified Perkin-Elmer 240B C-H-N analyzer. Light extinction coefficients were determined by measuring photosynthetically active radiation (PAR) at depths of 1–2 cm and 1.0 m below the surface with a Li-cor LI-1000 datalogger equipped with a LI-1925A underwater quantum sensor. Dissolved oxygen and temperature were determined in the field with a Nestor portable DO meter (model 8500) equipped with a field probe, and pH was measured with a Beckman ϕ 31 pH meter with a gel-filled combination electrode. Time-space contour plots of the data along the gradient in the Choptank were generated using "Minimum Curvature Method" (Surfer, vers. 4, software) on an IBM P/S 2 Model 60 PC.

Results and Discussion

Trends in SAV Regrowth and Transplant Survival

The years from 1985 through 1988 were among the driest on record in the mid-Chesapeake region, with very little agricultural runoff from eastern shore watersheds (Staver et al. 1988) compared to the late 1970s and early 1980s (Stevenson et al. 1986). The spring discharge in 1985–1988 from the upper watershed into the Choptank River at the United States Geological Survey Greensboro gauging station (Fig. 1) ranged from below 1.4 $m^3 s^{-1}$ to 3.0 $m^3 s^{-1}$; whereas during the previous seven years, it exceeded 3.6 $m^3 s^{-1}$, sometimes rising above 10 $m^3 s^{-1}$ (Table 1) during the comparatively wet spring of 1983 when Ward and Twilley (1986) observed a significant freshet.

The upper Chesapeake loadings are highly dependent on nonpoint source inputs which dramatically change according to precipitation and runoff (United States Environmental Protection Agency 1983). Consistent with the hypothesis that reduced runoff would be beneficial to SAV populations in the bay, there was an immediate resurgence of *Zannichellia palustris* and *Ruppia maritima* in 1985, two "r" species noted for their colonizing abilities (Stevenson 1988). Regrowth was prolific during the spring and early summer of 1985 in coves of the lower Choptank; over 10% of the Maryland Department of Natural Resources (Md.DNR) stations were vegetated compared to 1.7% in 1984 (Table 1). *Ruppia* seedlings were again abundant in 1986 and sediment excavations early in the growing season indicated emergence both from germinating seeds and vegetatively via underground rhizomes. The region of natural recolonization was confined to stations below the entrance of the Tred Avon River, approximately 20 km from the mouth (Fig. 2). Delineation of beds from aerial photographs taken near peak growth showed that the SAV coverage in the Choptank had expanded from an estimated 82 ha in 1984 to 1,528 ha in 1985, but was reduced to 452 ha in 1986 (Orth et al. 1986). Although there were problems with the aerial photography in 1987 and 1988, the Md.DNR survey confirmed the 1984–1986 trend and indicated continued recovery to 15% in 1987 and 33% in 1988 (Table 1).

In the area between 20 km and 40 km upstream, SAV regrowth was marginal and much more limited to small patches in isolated locations. Within this marginal area, *Zannichellia palustris* was found sporadically at the beginning of the growing season but declined precipitously by early summer. In 1988, peak biomass of the dominant species, *Ruppia maritima* ranged from 120–294 g m^{-2} downriver to 54–135 g m^{-2} in the marginal area. In this

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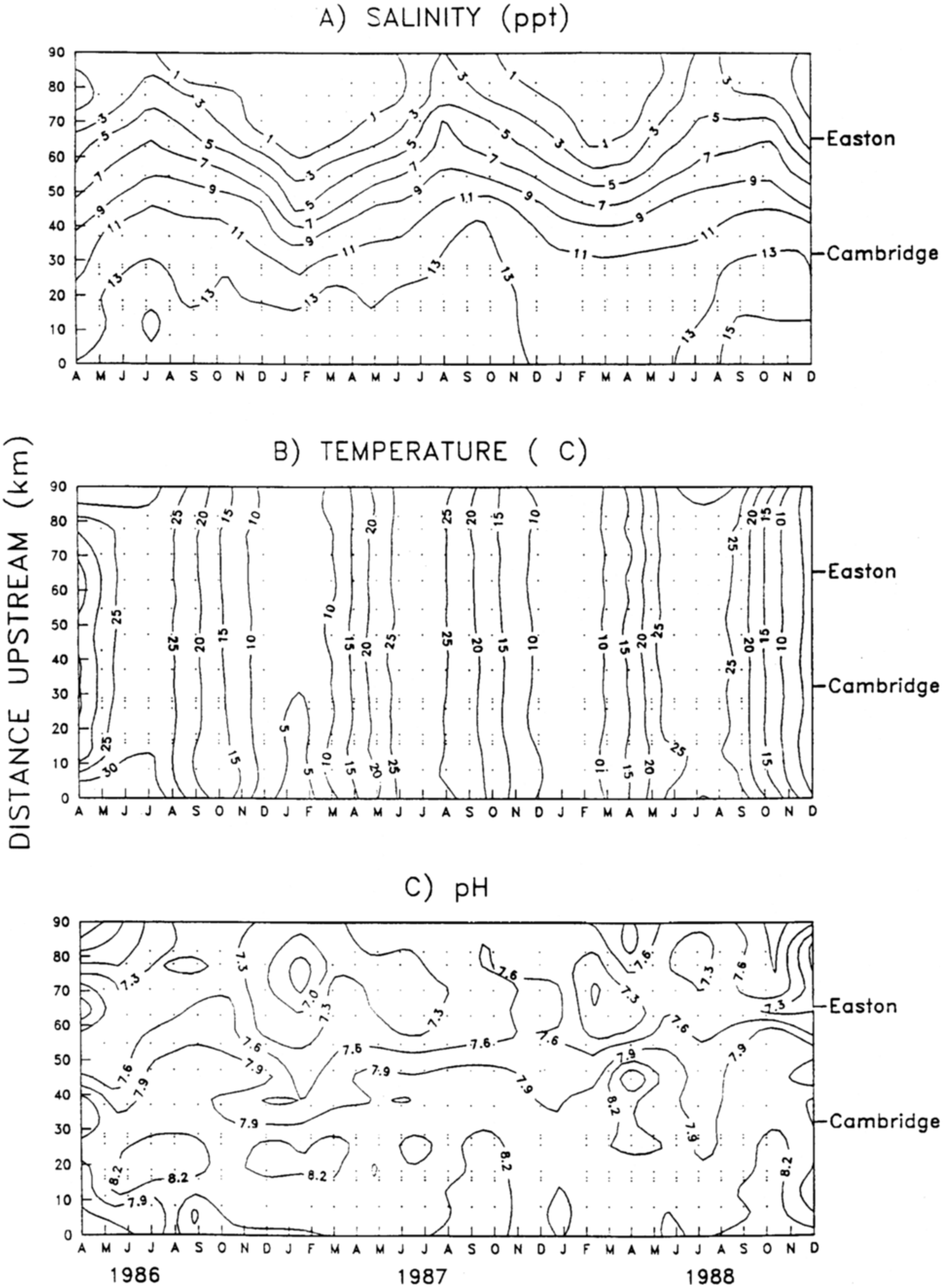
Fig. 3. Contour plot showing distributions of (A) salinity, (B) temperature, and (C) pH in the Choptank Estuary, 1986–1988.


A) SALINITY (ppt)
B) TEMPERATURE (C)
C) pH
DISTANCE UPSTREAM (km)
Easton
Cambridge
A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D
1986
1987
1988

last region, the shallow protected areas were often dominated by floating macroalgae: *Ulva curvata*, *Enteromorpha prolifera* and *Cladophora dalmatica*, which develop on the oyster beds of this area (Connor 1979), especially after dieback of *Zannichellia* in early summer. The macroalgae were often caught in the beds of *Ruppia maritima*, which persisted at a few locations in the marginal region (Fig. 2) throughout the growing season.

Experimental transplants had near 100% survival rates downstream and by mid-summer were indistinguishable from surrounding dense vegetation. However, in the marginal zone of the river (between km 20 and km 40), success was limited. Some transplants survived the first month but all declined by late summer. Transplants further upriver (>40 km) at Warwick and Goose creeks proved completely unsuccessful, and upriver from Windy Hill no SAV were ever detected on any of the surveys. Thus, three regions of the Choptank could be distinguished in terms of SAV survival during the 1985–1988 low flow period (Fig. 2): the lower Choptank, 0–20 km upstream from the mouth, where SAV was most abundant; the mid-Choptank, an area of marginal regrowth 20–40 km upstream; and the upper Choptank, >40 km upstream where SAV regrowth was not observed and transplants have been unsuccessful. The pattern of SAV survival in these three regions of the Choptank can then be used as a perspective to characterize water quality parameters relevant to macrophyte growth.

The Estuarine Gradient

Salinity, Temperature, and pH

During the dry years of 1986–1987 there were strong lower-layer salinity intrusions documented at the mouth of the Choptank (Sanford and Boicourt 1990). Elevated salinities characterize the entire estuary during the study period, stretching well into Tuckahoe Creek (usually fresh water), which empties into the Choptank at km 72 (Fig. 2), where salinities reached 3‰ (Fig. 3A). Salinities near the mouth entered the range where *Zostera marina* can survive (15‰); although the *Zostera* seeds germinated, they did not persist long enough to promote extensive colonization of *Zostera marina* despite the high salinities which were within the tolerance range of this species. Elevated salinity is not assumed to have been instrumental in the return of the dominant species, *Ruppia maritima*, since this species tolerates the widest salinity range of any submersed macrophyte in the world (Jagels and Barnabas 1989), spanning 0‰ to 60‰ (Stevenson and Confer 1978).

Moreover, other physical/chemical factors such as temperature and pH were comparatively stable along the estuarine gradient and thus appear to have contributed little to the observed change in SAV distribution. Despite the large seasonal range of temperatures from near zero to above 30°C in this estuary (Fig. 3B), differences from site to site are minimal and thermal tolerances of most temperate SAV species appear to be very broad (Pip 1989). Although the mid-Chesapeake region has precipitation very low in pH (values of 3–5 being commonly encountered during rainfall events), values in the Choptank River are circumneutral even in the freshwater upstream segments (Fig. 3C), most likely because of the moderately high alkalinity (15–65 mg l^{-1} $CaCO_3$). It therefore appears unlikely that either pH or temperature were significant factors in determining the distribution of SAV in this estuary during 1986–1988.

Chlorophyll a, Seston, and Light Attenuation

Chlorophyll *a* (Chl *a*) is effective at removing photosynthetically active radiation from the mixed layer of Chesapeake Bay (Rivkin and Voytek 1985), thereby limiting benthic productivity. The Choptank is similar to the mainstem of the Chesapeake, where phytoplankton biomass (on a volumetric basis) increases with decreasing salinity (Fisher et al. 1988). The lowest planktonic biomass is consistently found in the stations near the mouth of the Choptank. Downriver (0–20 km) Chl *a* concentrations suggest a slight tendency for a weak late winter bloom (>5 $\mu g\ l^{-1}$), with peak values in August in the range of 5–10 $\mu g\ l^{-1}$ (Fig. 4A).

Proceeding upstream, Chl *a* increases substantially 40–50 km above the mouth where the main channel narrows considerably and extensive meander marshes are found. This region also has two STP outfalls (Fig. 1), one at Warwick Creek (Secretary) and the other at Hog Island (Easton), which may help account for the increase in phytoplankton biomass. Chl *a* reached a maximum of 50 $\mu g\ l^{-1}$ in later summer 1986 (Fig. 4A); in the summers of 1987 and 1988 the values approached the concentrations encountered by Ward and Twilley (1986) in 1983 (40 $\mu g\ l^{-1}$). Elevated Chl *a* levels significantly reduce the ambient light available for SAV. For example, assuming a molar absorption

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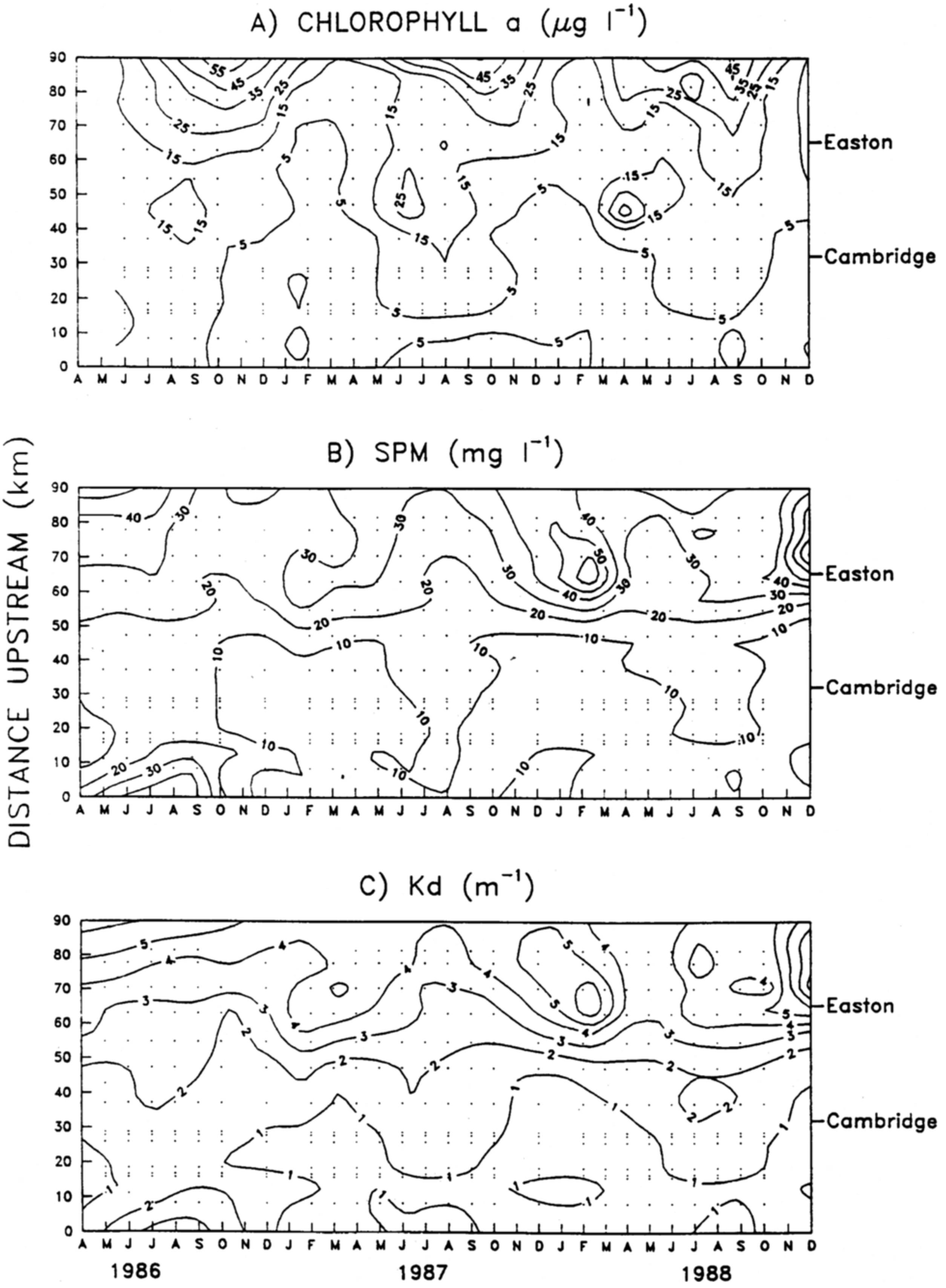
Fig. 4. Contour plot showing distributions of (A) Chlorophyll *a*, (B) suspended particulate material (SPM), and (C) light (photosynthetically active radiation) extinction coefficient (Kd) in the Choptank Estuary, 1986–1988.


A) CHLOROPHYLL a (μg l^{-1})
B) SPM (mg l^{-1})
C) Kd (m^{-1})
DISTANCE UPSTREAM (km)
Easton
Cambridge
A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D
1986
1987
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coefficient of 0.016 m^2 μg^{-1} (Kirk 1983), a Chl *a* concentration of 50 $\mu g\ l^{-1}$ can attenuate 90% of the light at 2.8 m in depth. Only species such as *Hydrilla verticillata* have been reported to withstand light levels below 10% of ambient, which renders this a difficult environment for all but exotic species of SAV.

Phytoplankton undoubtedly plays a role in limiting SAV growth in the upper Choptank, but the presence of additional suspended particulate material (SPM) makes its survival virtually impossible. SPM can exceed 50 mg l^{-1} as it did in February 1988 approximately 70 km upstream from the mouth (Fig. 4B). However, the SPM peaks we encountered were considerably lower than those of Ward and Twilley (1986), who recorded 80 mg l^{-1} during the strong, early spring freshet in 1983. This demonstrates the marked difference that runoff can make in this estuary, with SPM peaks doubling during wet years. Typically, SPM peaks coincide with periods of high freshwater discharge and therefore occur between late fall and early spring (i.e., nongrowing season months of the year) (Fig. 4B).

Combining high SPM in spring with the substantial Chl *a* levels later in the growing season leads to continuously high light attenuation coefficients, particularly in the upper part of this estuary. Wetzel and Penhale (1983) indicated that low light availability inhibits photosynthesis of SAV in regions of the lower Chesapeake Bay, where the light attenuation coefficient (Kd) is characteristically less than 2 m^{-1}. At 40 km above the mouth of the Choptank, Kd values of 2 and above were common throughout the year (Fig. 4C). Extreme Kd values above 5 were encountered several times at the head of the estuarine portion of the Choptank and in Tuckahoe Creek (Fig. 4C), during nongrowing season periods, primarily the result of elevated SPM during increased freshwater discharge. These Kd values are as high as Champ et al. (1980) reported in the "turbidity maximum" at the head of Chesapeake Bay.

As at the head of the Chesapeake, the phytoplankton biomass in the Choptank is much less a factor in reducing light than is SPM. The maximum Chl *a* level of 50 $\mu g\ l^{-1}$ in the upper Choptank yields a light attenuation of 0.8 m^{-1}, so the preponderance of light attenuation in the upper Choptank is due to SPM rather than Chl *a*. Below 20 km, attenuation coefficients generally remain below 1.75 m^{-1}, except during ephemeral wind events when sediment resuspension in the shallows can be extensive (Ward et al. 1984). The high light attenuation in the upper Choptank contrasts with the shallows at the head of other Chesapeake Bay tributaries such as the Potomac River and the edge of Susquehanna Flats where SAV are growing. These areas are much less turbid and had greater SAV abundance during the mid-1980s (Carter and Rybicki 1986; Staver 1986). One pronounced feature of the Choptank is the extent of agricultural land in close proximity to the upper estuary, which can contribute large amounts of sediment on storm events, especially if they occur shortly after plowing of the fields. This new sediment coupled with resuspension of "old" from the bottom during windy periods in the Choptank (Yarbro et al. 1983) produces a highly turbid environment in open areas, which seriously reduces SAV growth potential.

Nitrogen in the Water Column

The agricultural input of N is characterized by large annual pulses of NO_3^-, which are flushed into the upper reaches of the estuary in winter from the surrounding agricultural land (Stevenson et al. 1986). By summer, NO_3^- concentrations fall two orders of magnitude (Fig. 5A), reflecting both reduced diffuse source inputs, as well as assimilatory and dissimilatory processes (denitrification) associated with shallow embayments in this region (Shenton-Leonard 1982). For much of the growing season, nitrate levels were well below 10 μM throughout the estuary.

Nitrite (NO_2^-) normally comprises only a small fraction of the N in this estuary; peak values approached 2 μM during the summers of 1986 and 1987 (Fig. 5B). These peaks occurred 60 km to 70 km upstream and suggest nitrification activity in this region of the estuary during these years. In this respect the Choptank is very similar to the mainstem of Chesapeake Bay (Fisher et al. 1988). The NH_4^+ concentrations are higher than NO_2^-, and two peaks exceeding 20 μM were measured during our 1986–1987 cruises (Fig. 5C). The downstream NH_4^+ peak during July 1986 was undoubtedly associated with the previously documented entrainment of hypoxic bottom water from the mainstem of the bay into the lower Choptank (Sanford and Boicourt 1990). During this same period in 1986, ammonium concentrations were in the range of 20–30 μM below the pycnocline in the adjacent bay mainstem (Malone et al. 1988).

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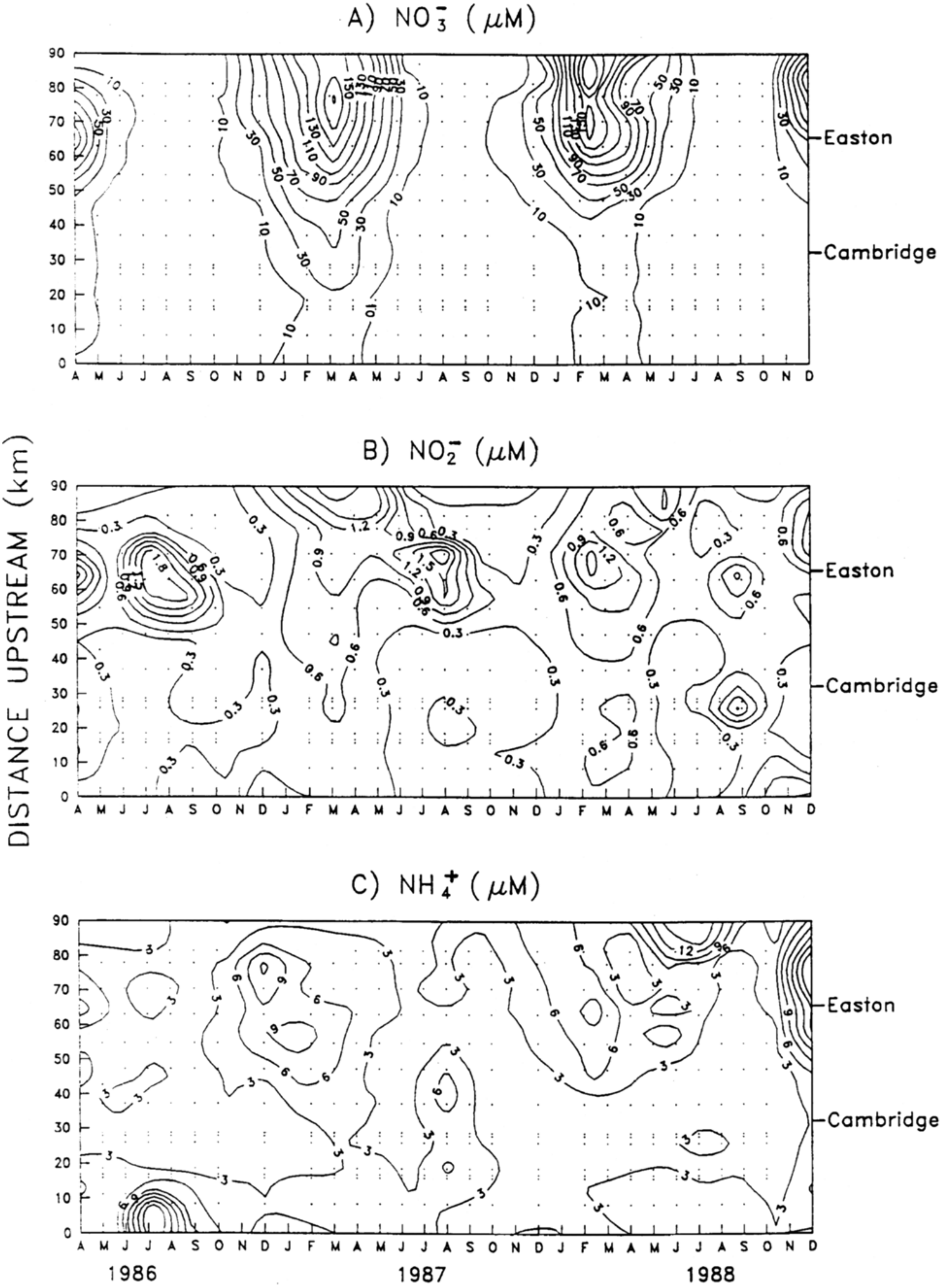
Fig. 5. Contour plot showing distribution of (A) nitrate, (B) nitrite, and (C) ammonium in the Choptank Estuary, 1986–1988.


A) NO_3^- (μM)
B) NO_2^- (μM)
C) NH_4^+ (μM)
DISTANCE UPSTREAM (km)
Easton
Cambridge
A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D
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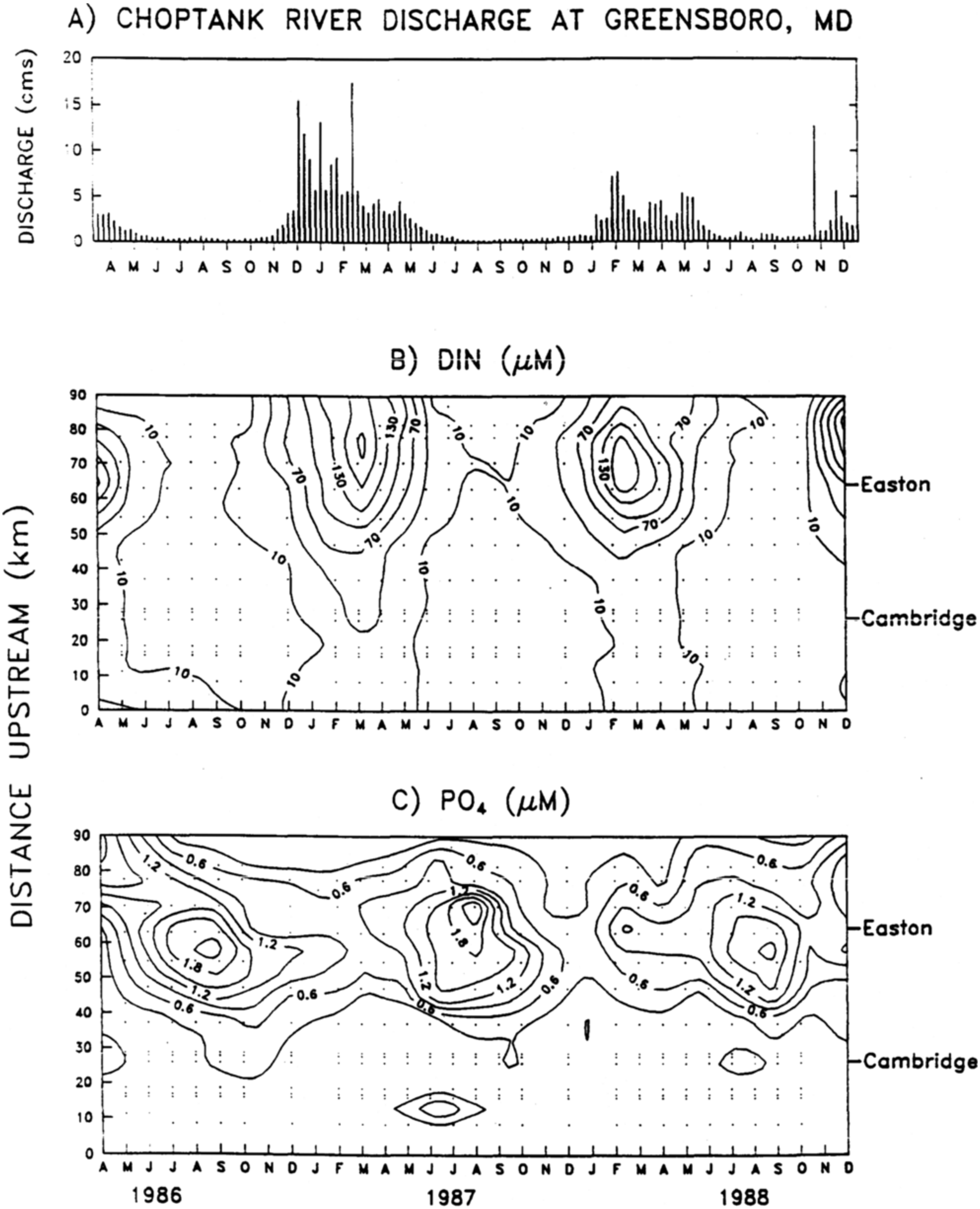

Fig. 6. (A) Stream runoff ($m^3 s^{-1}$) into the head of the Choptank Estuary and contour plots showing distribution of (B) dissolved inorganic nitrogen and (C) phosphate in the Choptank Estuary, 1986–1988.

The second NH_4^+ peak occurred in the vicinity of the Easton STP in December 1988.

Although algal communities competing with SAV generally take up NH_4^+ preferentially (Morris 1972), they are capable of utilizing all three forms of N we measured: NO_3^-, NO_2^-, and NH_4^+ (Antia et al. 1975). Therefore, for this discussion we consider them together as dissolved inorganic nitrogen (DIN) for evaluating their potential stimulation of periphyton communities to the detriment of SAV populations. When DIN is plotted along with river discharge (Figs. 6A and B), a clear relationship emerges. DIN coincides with discharge, with peak concentrations close to 200 μM in the upper Choptank in the winters of 1986 and 1987, followed by very low concentrations throughout the estuary in summers. Less than 10 μM was common in the lower Choptank during the growing season when SAV was thriving.

Phosphorus in the Water Column

Phosphorus (PO_4^{3-}) was often substantially less than 0.5 μM in the lower Choptank during the entire period of study (Fig. 6C). The highest PO_4^{3-} concentrations occurred during summer months at approximately 60–70 km upriver. As noted earlier, the Easton STP outfall is located 61 km above the mouth and may be the chief cause of the elevated PO_4^{3-} levels in this section of the estuary, along with regeneration from particulates in the

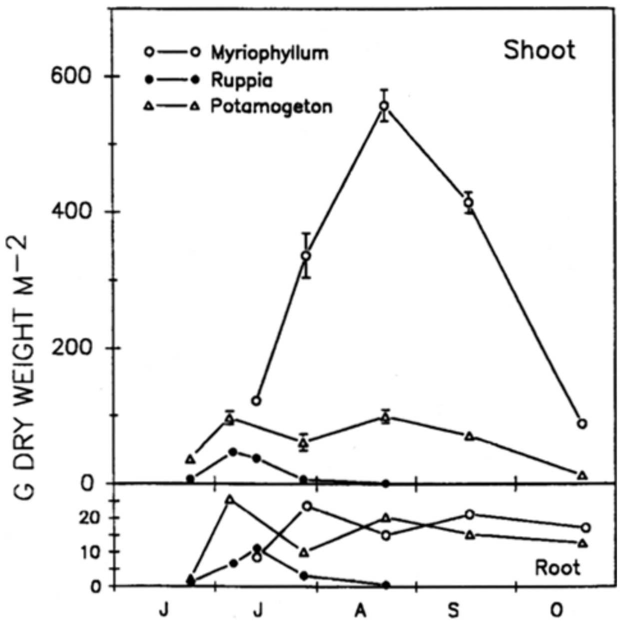

Fig. 7. Shoot and root biomass of three species of SAV in the lower Choptank River in 1977.

TABLE 2. Peak aboveground biomass of *Ruppia maritima* in the Choptank Estuary, July 1988.

Site	Date	Salinity (‰)	Biomass g dw m^{-2} (AF g dw m^{-2})
Cook's Pt. Cove	July 15, 1988	14	240 ± 20 (143 ± 12)
Chapel Creek	July 15, 1988	12	120 ± 25 (61 ± 14)
Irish Creek	July 29, 1988	12	294 ± 44 (137 ± 17)
Foxhole Creek	July 29, 1988	12	81 ± 38 (11 ± 3)
Trappe Creek	July 15, 1988	12	54 ± 25 (31 ± 14)
Dickinson Bay	July 8, 1988	12	135 (74)*

* Single sample.

sediments in the summer. Curiously, Ward and Twilley (1986) did not detect any clear PO_4^{3-} pattern, suggesting nonpoint source inputs. However, their study was conducted in a year that included a spring season with high rainfall and a large freshet, which may have obscured the STP inputs. The dry years during which our study was conducted may have made inputs from the STPs and any sediment phosphate releases more apparent due to reduced nonpoint source inputs and flushing rates.

SAV Phenology and Seasonality of Water Quality Parameters.

Since there was a marked difference between winter and summer nutrient patterns (especially DIN) and since the plants die back over the late fall and winter, we chose to average water quality values over the growing season to characterize the three regions of the estuary. In order to determine the potential time period during which SAV are sensitive to changes in water quality, we analyzed their phenology from data which were collected in 1977. These data were chosen as the earliest available quantitative harvesting in this estuary at a time of relative SAV abundance during a dry year (Table 1) and before the catastrophic decline in the early 1980s. The seasonal biomass distribution for three SAV species in the Choptank (Fig. 7) reveals that growth is well established by June and that it continues through October, after which it dies back. Interestingly, the peak biomass we obtained (294 g m^{-2}) during 1988 (Table 2) was only half of the maximum in 1977 (Fig. 7), when more stations were reported to have SAV (Table 1).

These data, along with those collected nearby (Lubbers et al. 1990), suggest that May is the month that SAV growth normally begins in the Choptank. Thus, we defined the growing season as the six-month period from May to October, and averaged water quality measurements during that period for each year of the study (Table 3). These mean values were arrayed along with the spatial distribution during the SAV resurgence from 1986 to 1988 in the Choptank (Fig. 8).

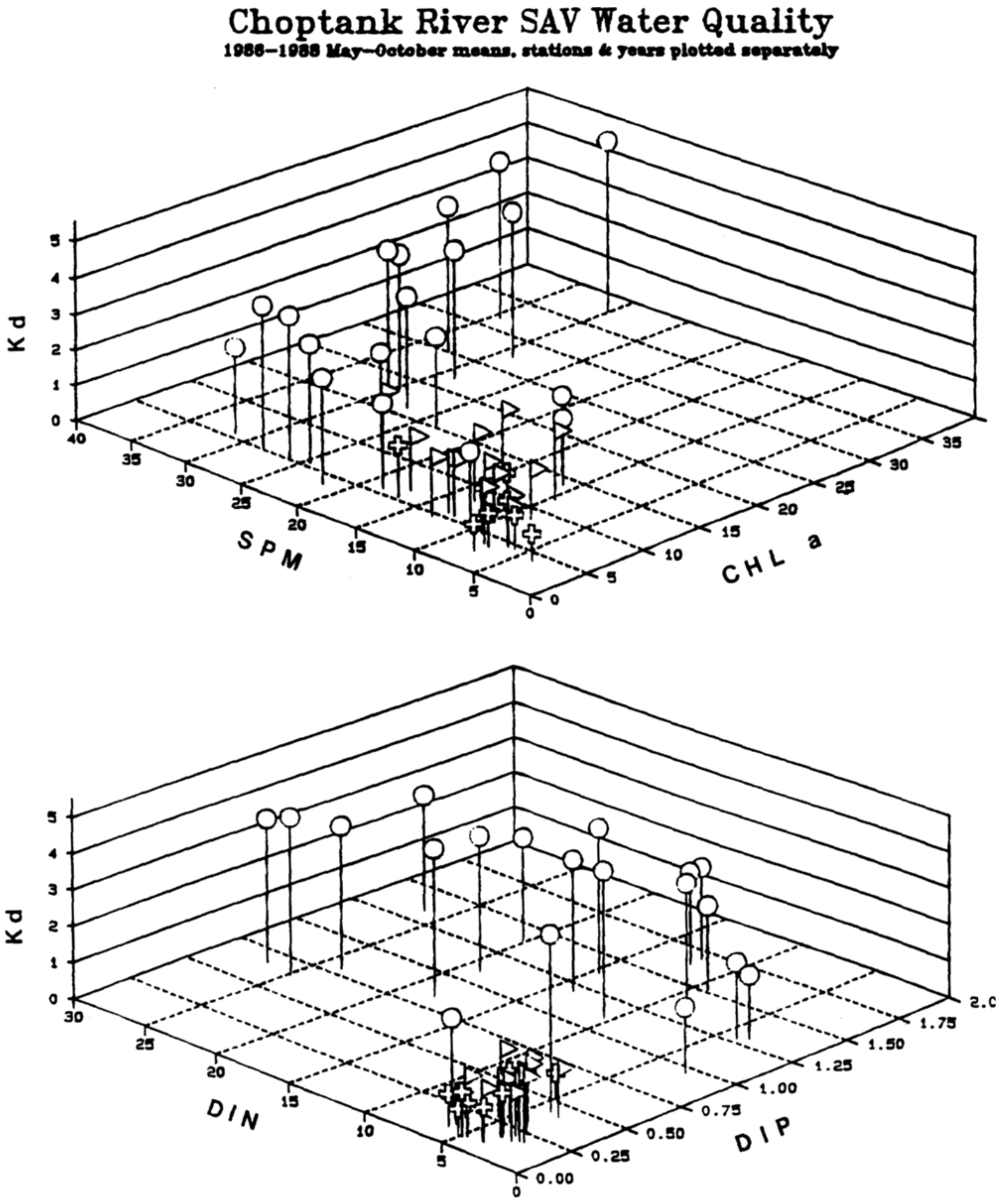

Fig. 8. (top) Relationship between light attenuation coefficient (Kd) in m^{-1}, suspended particulate material in mg l^{-1} (SPM), and Chlorophyll *a* in $\mu g\ l^{-1}$ (Chl *a*); (bottom) relationship between Kd, dissolved inorganic nitrogen in μM (DIN), and phosphorus in μM (DIP) to SAV occurrence in the Choptank Estuary 1986–1988. Circles indicate no survival, flags indicate marginal survival, and crosses indicate long-term survival.

TABLE 3. Values of indicator parameters during the growing season (May-October) in the Choptank Estuary, 1986–1988.

	Year	DIN (μM)	PO_4^{3-} (μM)	Chl *a* (μg l^{-1})	SPM (mg l^{-1})	Kd (m^{-1})
upper Choptank	86	10.46 ± 1.73	1.38 ± 0.12	18.51 ± 2.88	22.40 ± 2.28	2.98 ± 0.37
(>40 km)	87	18.31 ± 3.96	1.17 ± 0.14	20.02 ± 1.63	21.06 ± 2.44	3.03 ± 0.37
	88	16.29 ± 3.69	1.04 ± 0.12	20.30 ± 2.73	27.27 ± 3.44	3.52 ± 0.38
	x̄	15.02 ± 2.35	1.20 ± 0.10	19.61 ± 0.56	23.58 ± 1.89	3.18 ± 0.17
mid/lower Choptank	86	4.46 ± 4.46	0.25 ± 0.03	9.26 ± 1.13	14.18 ± 1.88	1.51 ± 0.09
(<40 km)	87	5.25 ± 0.90	0.17 ± 0.03	8.84 ± 1.14	8.92 ± 1.04	1.22 ± 0.10
	88	4.32 ± 0.61	0.16 ± 0.04	6.13 ± 0.59	9.02 ± 1.18	1.21 ± 0.13
	x̄	4.85 ± 0.32	0.19 ± 0.03	8.08 ± 0.98	10.71 ± 1.74	1.31 ± 0.10

Water Quality Thresholds Associated With SAV Regrowth

The distribution patterns of SAV during regrowth years and corresponding values of dissolved inorganic N and P (DIN and DIP), Chl *a*, and Kd show remarkably tight groupings (Fig. 8). Regrowth of SAV was substantial at DIN concentrations below 10 μM and DIP concentrations below 0.35 μM. In addition, the N:P ratio must also be considered. Survival of SAV may occur if the concentration of one nutrient is very high but the other is low enough to be limiting to algal components, which do not have access to sediment pools of nutrients (Barko and Smart 1981). These conditions are met when the N:P ratio is either very high (>100) or low (close to 1). The former appears to be the case at the head of the bay where SAV is abundant despite NO_3^- concentrations that can exceed 100 μM during the growing season (Staver 1986). One of the key features of the freshwater tidal system at the head of the Chesapeake is that planktonic and epiphytic algae are comparatively less abundant, possibly due to P limitation. On the other hand, where SAV has access to a large sediment pool of P at the edges, biomass reaches 1 Kg m^{-1} (Staver 1986).

The mid- and lower Choptank, zones of marginal and abundant SAV colonization, respectively, were characterized by mean growing season Chl *a* concentrations <15 μg l^{-1} and SPM concentrations below 20 mg l^{-1} (Fig. 8). The relatively high SPM at Cook's Cove resulting from rapidly eroding shorelines (as well as occasional NH_4^+ intrusions from bottom waters of the mainstem) helps explain why SAV growth is limited there compared to locations immediately upstream. Both Chl *a* and SPM affect the Kd value, which averaged below 1.5 m^{-1} in the lower Choptank (Table 3) during the growing seasons of 1986 to 1988.

Thus it appears that mean levels of Chl *a*, SPM, light attenuation (Kd), DIN, and DIP during the growing season can be used in concert to define the water quality associated with SAV regrowth in this mid-Chesapeake Bay estuary. While this approach may be useful for setting management goals for reducing nutrient inputs into estuaries like the Choptank, it does have limitations due to covariance problems. For example, it cannot adequately predict what happens to SAV if a single factor changes independently of the others. Also, the question of whether SAV could survive if nutrient and sediment loadings are delayed until after the growing season cannot be resolved with our data set, although the current pattern of higher loadings in nongrowing season months via freshwater discharge indicates that this strategy may have limited applications.

We assumed that SAV responses are integrated over the growing season, and that SAV responds to mean values of each parameter. This works well in the Choptank estuary where nutrient concentrations in the shallows appear to be relatively stable during the growing season (T. Jones and R. Newell unpublished data). The use of alternative statistical descriptors, such as medians, might be useful in more pulsed environments when survival of macrophytes might be an issue.

Finally, it should be emphasized that a hysteresis effect may exist in relation to the loss versus reestablishment of SAV. Extensive SAV populations can remove considerable quantities of nutrients from the water column, well beyond their own growth needs (Gerloff and Krombholz 1966). They also modify their environment by damping wave action and promoting sedimentation (Ward et al. 1984), creating lower water column SPM and Chl *a* concentrations and decreased Kd values. In addition, some SAV (e.g., *Potomogeton perfoliatus*) can form canopies when healthy and adjust to short periods (few weeks) of lowered light—at least in nonturbulent waters (Goldsborough and Kemp 1988).

Therefore we hypothesize that threshold concentrations associated with the *decline* of well-established SAV populations would be higher than those required for colonizing populations. In addition, Kd values lower than 2.0 m^{-1} are necessary for SAV to persist in deeper waters and even perhaps in the shallows for other species such as *Pot-*

amogeton perfoliatus which did not return to the Choptank in the 1980s. Despite these caveats, it is encouraging that concurrent work in a higher salinity range (Orth et al. 1987; Orth and Moore 1991; Dennison et al. 1993), using a different approach, indicates similar water quality values for predicting survival patterns of *Zostera marina* in the York River Estuary in Virginia. It remains to be seen how well these values can be applied to other coastal systems.

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