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Chesapeake Bay Program Technical Studies: A Synthesis

FOREWORD

In recent years the well-being of Chesapeake Bay and its tributaries has been stressed by activities of the region's growing population. Concern for this national resource prompted Congress in 1976 to direct the U.S. Environmental Protection Agency (EPA) to conduct an intensive five-year study of the Bay's resources and water quality, and develop related management strategies. To address concerns of Congress, this study, known as the Chesapeake Bay Program (CBP), focused research on three principal problems in the Bay--the presence of toxic substances, nutrient enrichment, and the disappearance of valuable submerged aquatic vegetation. In addition to evaluating the severity of these problems and what they may indicate about the Bay's water quality, the CBP was directed to review current mechanisms of pollution control and suggest management strategies.

This document is the second of the Program's four final reports. It is intended to share the results and significance of the Chesapeake Bay Program's technical studies with managers, decision-makers, and citizens. The report integrates, or synthesizes, results of the many technical studies that have addressed Congress' concerns. This integration by key scientists in the three problem areas centered around a set of specific questions relevant to managers and decision-makers of the Bay region, and were developed by Program staff, and State and Federal environmental managers. In attempting to answer these questions with the best scientific information, the authors of the papers were not confined only to information derived from the projects. They drew on the research literature, personal communications, and their own rich knowledge of the Bay's ecology, as well as their extensive interaction with peer scientists. The conclusions of each paper, although based primarily on results from CBP research projects, reflect a mixture of scientific results and the best judgment of scientists responding to management questions.

The authors and contributors hope that this report will further knowledge of changes taking place in the Bay, so that together, we can manage Chesapeake Bay effectively.

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SUMMARY

As part of the five year study plan for the EPA Chesapeake Bay Program (CBP), EPA staff, officials from Maryland and Virginia, and citizens identified 10 areas as foremost water quality problems of the Bay, and agreed upon three as most critical for intensive investigation: Nutrient Enrichment, Toxic Substances, and the Decline of Submerged Aquatic Vegetation. The EPA then initiated research to study intensively these three problem areas. The following summary describes the findings from research projects funded by the Chesapeake Bay Program in those three technical areas. Two other CBP reports, "Characterization of Chesapeake Bay" and "Management Strategies for Chesapeake Bay" assess Bay-wide conditions and suggest management strategies.

NUTRIENT ENRICHMENT

Nutrients, both phosphorous (P) and nitrogen (N), are crucial to Bay life. Nutrient enrichment occurs when excessive additions of nitrogen and phosphorous compounds enter the water. Enrichment can lead to undesirable consequences such as phytoplankton blooms, depletion of oxygen, and changes in kinds of fish present. When an estuary, such as Chesapeake Bay, becomes nutrient-enriched, algae can thrive and accumulate in the water column. Their presence decreases light transparency, and, when they degrade, they use up dissolved oxygen that other plants and animals need.

Nutrient enrichment in Chesapeake Bay is evaluated by measuring a number of related factors including nutrient concentration and oxygen levels in the water, amounts of chlorophyll a, (a green pigment found in most algae), and transparency of the water (Secchi depth). Historical records of these measurements were gathered and analyzed during the Bay Program to look at trends in nutrients over the past 20 years. During this time, nutrient concentrations have increased, causing enrichment in some areas. Figure 1 shows areas of the Bay that are enriched. These include: most of the western tributaries such as the Patuxent, Potomac, and James; the northern and central main Bay; and some Eastern shore tributaries including the Chester and Choptank. These areas show high levels of nutrients and chlorophyll a, and reduced light transparency. The lower Bay, however, has remained relatively unaffected. An analysis that relates these trends to the health of fisheries in the Bay will be presented in the CBP report entitled "Characterization of Chesapeake Bay."

Sources of Nutrients

Phosphorus (P) and nitrogen (N) enter the Bay from several major sources or pathways: atmosphere, rivers, point sources, and sediments. The estimated percentage that each of the sources contributes to the Bay during a year is shown in Table 1.

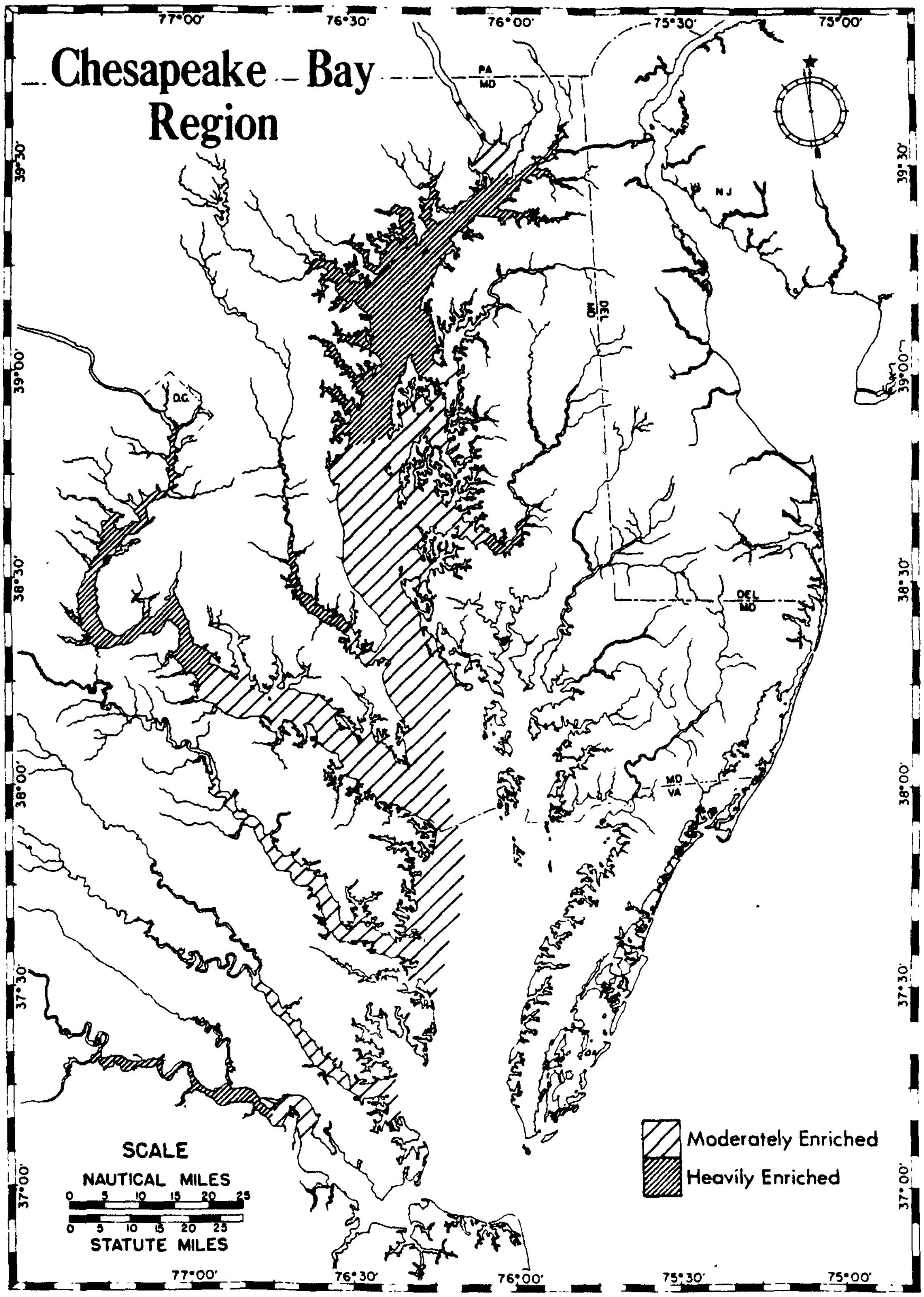

Figure 1. Map showing portions of Chesapeake Bay that are moderately or heavily enriched according to the criteria of Heinle et al. (1980).

TABLE 1. PERCENTAGE OF ANNUAL NUTRIENT LOADINGS FROM VARIOUS SOURCES(1)

Constituent	Atmospheric Sources	Riverine Sources	Point Sources	Sediment Sources
Total nitrogen	13	56	22	9
Total phosphorus	5	35	35	25

(1)Definition of Terms

Atmosphere: aerial input that directly lands on fluvial or tidal waters.

Riverine: mass loadings of nutrients to Bay from above the head of tide.

Point sources: nutient loads from industry and municipalites below the head of tide.

Sediment sources: nutrient releases or loads from the bottom sediment of Chesapeake Bay.

Riverine Sources--

Riverine sources are a major contributor of N and P to the Bay; approximately 56 percent of the total nitrogen loading comes from these sources. This loading ranges from 39 percent in summer to 64 percent in spring when river flows are highest. Riverine source loads for P are about 35 percent of the total annual input and range from 12 percent in summer to 57 percent in spring.

Of all the river sources, the Susquehanna River is the major contributor of P and N, as shown in Table 2. The Susquehanna River has by far the largest drainage area and annual flow discharge among the river sources. This at least partly accounts for the relatively higher contribution of N and P from the Susquehanna. This river carries about 70 percent of the total nitrogen and 56 percent of the total phosphorus delivered to the Bay each year from riverine sources. Most of these loads enter during the winter and spring.

The Susquehanna produces only about 40 percent of annual sediment load, because the particulate matter is trapped in reservoirs located on the lower 60 miles of the main stem of the river. Only a large flow, above 400,000 cubic feet per second (cfs), will transport sediment through the reservoir and deliver them to the Bay. Such flows occur only one percent of the time.

TABLE 2. ESTIMATED PERCENTAGE OF TOTAL ANNUAL RIVERINE NUTRIENT AND SEDIMENT LOADS FROM CHESAPEAKE BAY TRIBUTARIES

Constituent	Susquehanna	Potomac	James	Other Tributaries
Total nitrogen	70	19	6	5
Total phosphorus	56	22	16	6
Sediment	40	33	16	11

Major land uses in the Chesapeake Bay basin and their estimated contribution to riverine nutrient loads are shown in Table 3.

TABLE 3. MAJOR LAND-USES ABOVE THE FALL LINE AND THEIR ESTIMATED CONTRIBUTION TO RIVERINE NUTRIENT LOADS

Land Use	Percent In Basin	Percent of Riverine Nutrient Loads	
		TN	TP
Cropland	15-20	45-70	60-85
Pasture	8-12	4-13	3- 8
Forest	60-65	9-30	4- 8
Urban	3- 5	2-12	4-12

Riverine loadings can vary considerably among land uses. The highest riverine loading rates come from cropland, and lowest from forest sites. Agricultural land appears to produce the largest fraction of the riverine loads by at least a factor of three for both nitrogen and phosphorus, due to the high unit-area loadings and large percentages of land used for agriculture in this area. The CBP's Bay-wide watershed model has estimated the relative contributions of nutrients from all nonpoint sources. These results will be presented in the CBP report "Management Strategies for Chesapeake Bay."

Point Sources--

Most of the remaining nutrients in the Bay are contributed from point sources, such as sewage treatment plants and industries lying below the head of tide (see Table 1). These point sources account for about 22 percent of total nitrogen load and some 35 percent of total phosphorus input. The percentage of nutrient load from point sources ranges from 15 in spring to 29 in fall, while phosphorus percentages range from 59 percent in fall to 21 percent in summer.

Other sources include the atmosphere and bottom sediments. Atmospheric contribution constitutes about 13 percent of the total nitrogen and five percent of the annual phosphorus input, while bottom sediments make up about 10 percent of the annual nitrogen and 25 percent of the annual phosphorous load.

Seasonal Nature of Nutrient Loads

The largest portion of the annual nitrogen load enters the Bay during the winter and spring, while the highest portion of the annual phosphorus load enters during the spring and summer. These nutrient inputs support increases in algal standing crop. Since the relative abundance of nitrogen and phosphorus changes from spring to summer, so the potential limiting nutrient for the algal standing crop may change.

The limiting nutrient changes during the year in Chesapeake Bay as a result of three prominent events. The first is the substantial nitrate input with a spring runoff from the Susquehanna River. The second event occurs during mid-summer when very low oxygen concentrations in deeper Bay water permit release of phosphate from Bay sediments and accumulation of both phosphate and ammonium in the deep water. The third event is the fall nitrite maximum observed in both mid-Bay and in the lower Potomac River estuary. Thus, peak nitrogen availability occurs in spring, while peak phosphorus availability occurs in summer.

Consequently, phosphorus concentration is generally higher in deep water during summer. Addition of phosphorus during the other seasons could cause the standing crop of phytoplankton to increase, if nitrogen is available. Thus, phosphorous appears to be the biomass limiting, or regulating, nutrient for spring, fall, and winter. Nitrogen, however, is at its lowest levels and could be limiting in summer; additions at this time may cause phytoplankton to grow if phosphorous is available from the deep water due to recycling processes. An awareness of the response of phytoplankton to available nutrients is important when considering effects on Bay resources and how to control input. Because phytoplankton form the base of the Bay's food web, increases in their populations will create more food for other Bay inhabitants, to a point. Beyond this point (we feel that Figure 1 indicates what areas of the Bay are at this point) growth of phytoplankton can be detrimental to the Bay's water quality and its resources.

Management Implications

Management strategies to address the problem areas must take into account the seasonal patterns of nitrogen and phosphorous we have described and the degree to which each contributing source may be controlled, its relative costs to achieve this control, and trade-offs between point and nonpoint sources. The possible management strategies will be shown in the CBP report "Management Strategies for Chesapeake Bay".

TOXIC SUBSTANCES

Toxic substances constitute the second of three critical areas studied under the CBP. The research focused on determining the status of both metals and organic compounds in Chesapeake Bay, including their concentration in the water column, bed sediments, suspended sediments, and in some bivalves. Sources of metals and organic compounds were also investigated. A limited amount of research was performed on assessing the toxicity of point source effluents and Bay sediments.

Toxic substances are usually defined as chemicals or chemical compounds that can harm living plants and animals, including humans, or impair physical or chemical processes. The two general classes of toxic substances studied were inorganic and organic compounds. Inorganic materials are metals such as arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), and zinc (Zn). Many of the organic compounds are products of human activities and include pesticides, phthalate esters, polynuclear aromatic hydrocarbons (PNA's), and other chlorinated hydrocarbon compounds (PCBs, etc.).

Current Status

The highest concentrations of metals in Bay sediment occur in Baltimore Harbor and the Elizabeth River. In the main Bay, the highest metals concentrations in sediment occur in the northern Bay and particularly near the western shore where cadmium, cobalt, copper, manganese, nickel, lead, and zinc are enriched (elevated relative to natural levels) two to eight times above natural levels from the Susquehanna Flats to Baltimore Harbor region. At least half of the metal loads for chromium, cadmium, copper, and lead orginate from human sources.

Metals tend to partition with fine particulate matter such as detritus and silt. Consequently, highest concentrations of metals in suspended material (ug of metal per gram suspended material) occur in near-surface water in the central Bay where organic matter tends to be high. Cadmium, lead, copper, and zinc display the highest concentrations. Because this enriched zone is an area of high organic activity where organisms respire, reproduce, and grow, metals are available for uptake by phytoplankton and marine organisms. Once in the plankton, the metals can be passed through the food chain.

Like metals, organic compounds tend to cling to fine material that is suspended in the water. When this material settles, organic compounds will accumulate on the Bay floor. Concentrations of organic compounds in bottom sediments are highest in the northern Bay. They exhibit similar trends to metal enrichment, with highest concentrations occurring in the vicinity of Baltimore Harbor. Concentrations tend to increase up the Bay from the Potomac River mouth toward the Patapsco River. North of the Patapsco River, elevated concentrations are found to exist to the Susquehanna River mouth. It appears that many of these organic compounds may have entered from the Susquahanna River. In the southern Bay, the highest concentrations of organic compounds are found where the river estuaries enter the main Bay.

The sediments of the Patapsco River estuary show the highest concentrations of organic compounds. Highest levels occur near source locations. These sediments appear to be largely trapped within Baltimore Harbor.

Oysters collected from around the Bay and oyster-tissue extracts were examined for organic compound concentrations. These bivalves did accumulate some toxic compounds. There were 42 compounds detected whose individual concentrations exceeded 50 parts per billion. The mouth of the James River had 29 percent, and Baltimore Harbor 24 percent of these 42 compounds.

Sources

Riverine sources above the fall line, point sources below fall line, and atmospheric sources, contribute most of the metals to Chesapeake Bay as shown in Table 4. Of the three major rivers in which metal concentrations were measured (Susquahanna, Potomac, and James), the Susquahanna contributes the greatest amount of metals. These river loads include municipal, nonpoint, and industrial sources above the fall lines. The annual loadings of various metals of the three rivers are compared in Table

5. The concentration levels of metals in the three rivers are similar, however, the Susquehanna has greater loadings because of its higher flow. The Susquehanna River is also very significant to quality of water in the Bay proper, because the loads it delivers enter the Bay directly and are not trapped in the sub-estuaries like those from the James and Potomac.

Industrial and municipal input below the fall line are a major contributor of metals to the Bay (Table 4). For example, industrial loads account for 66 percent of total cadmium load. Municipal POTWs account for 19 percent of total chromium load. The distribution of these loadings for POTWs and industries below the fall line (Pennsylvania counties, thus, not included) by counties is shown in Table 6. The inputs of Cd, Cr, Cu, Fe, and Zn in Baltimore County and Baltimore City far exceed those from other counties. Substantial inputs from POTWs are also noted for Cr, Fe, and Zn in Richmond City; for Cr, Fe, and Zn from Norfolk City; and for Cr, Fe, and Zn at Hopewell City. The industrial load exceeds POTW loadings by two times. Loadings from urban runoff and atmospheric sources are also significant for several metals as shown in Table 4.

Results from the CBP show that sources of organic compounds to the Bay are human-related. In particular, organic compounds in northern-Bay sediments are probably from the Susquehanna River, and possibly some from the Patapsco. Concentrations of organic compounds in the Bay should be highest in areas of sedimentation near industrial regions and high population areas. The CBP is further investigating sources of toxic substances and will present the results in CBP report "Management Strategies for Chesapeake Bay".

TABLE 4. LOADINGS OF METALS FROM THE MAJOR SOURCES AND PATHWAYS TO CHESAPEAKE BAY (VALUES IN METRIC TONS/YEAR)

Source	Cr	Cd	Pb	Cu	Zn	Fe
Industry	200 (19)[1]	178 (66)	155 (22)	190 (22)	167 (6)	2,006 (1)
Municipal Wastewater	200 (19)	6 (2)	68 (10)	99 (12)	284 (10)	625 (1)
Atmospheric	---	3 (1)	34 (5)	28 (3)	825 (29)	87 (1)
Urban Runoff	10 (1)	7 (2)	111 (16)	9 (1)	63 (2)	977 (1)
Rivers	551 (53)	75 (28)	307 (43)	517 (59)	1444 (50)	199,682 (77)
Shore Erosion	83 (8)	1 (1)	28 (4)	29 (3)	96 (3)	57,200 (22)

[1]Values in parenthesis represent percent of total loading

In certain areas, present levels of toxic substances could threaten the health of organisms. Bioassay tests on bottom sediments from the Bay show that sediments from the Patapsco and Elizabeth Rivers and northern Bay are potentially more toxic than elsewhere. This toxicity is probably produced by a combination of high metal content and large loads of organic compounds. These tests on bottom sediments found concentrations that cause mortality. The highest mortalities occurred on samples from the upper reach of the Patapsco and Elizabeth Rivers, and the northern Bay. Tests performed on effluent from industrial plants around the Bay area revealed that up to half of effluents sampled killed test fish and invertebrates. The significance of these results and their relationship to Bay resources will be discussed in CBP report "Characterization of Chesapeake Bay".

TABLE 5. ESTIMATED AVERAGE ANNUAL LOADINGS FOR VARIOUS METALS FROM THE MAJOR TRIBUTARIES OF THE CHESAPEAKE BAY FOR 1979-1980 PERIOD* (VALUES IN METRIC TONS/YEAR) (FROM LANG AND GRASON 1980)

Parameter	Susquehanna @ Conowingo Dam	%	Potomac @ Chain Bridge	%	James @ Cartersville, Va.	%	Totals
Al-T	161,618	69	37,626	16	33,884	15	233,128
As-T	82	71	13	12	20	17	115
Cd-T	65	87	4	5	6	8	75
Co-T	59	40	39	27	48	33	146
Cr-T	383	70	105	19	63	11	551
Cu-T	390	75	86	17	41	8	517
Fe-D	1,844	57	839	26	567	17	3,250
Fe-S	192,422	65	76,227	26	27,783	9	296,432
Mn-T	14,469	77	1,933	10	2,327	13	18,729
Ni-T	229	57	109	27	64	16	402
Pb-T	174	57	102	33	31	10	307
Zn-T	837	58	322	22	285	20	1,444

*Values listed represent the mean of 1979 and 1980 calender year loadings. (Note: Percentages above are approximate numbers)

D - Dissolved
S - Suspended
T - Total

Management Implications

Managing toxic substances requires a priority, or ranking, framework that evaluates toxic material for its greatest potential to affect human and environmental health. As with nutrients, areas where environmental quality is severely degraded should be established, based on all available environmental quality data (sediment, biota, and water) and should be top

TABLE 6. POINT SOURCE LOADINGS OF METALS FROM INDUSTRIES[2] AND PUBLICLY OWNED TREATMENT WORKS (POTW'S)[1] IN COUNTIES BELOW THE FALL LINE FOR CR, CD, PB, CU, ZN, FE, IN METRIC PER YEAR

	Metal											
	Cr		Cd		Pb		Cu		Zn		Fe	
	POTW	I[3]	POTW	I	POTW	I	POTW	I	POTW	I	POTW	I
Arundel	7.3	0.7	0.2	0.2	2.4	3.1	3.4	2.4	9.9	1.3	21.8	9.4
Baltimore		59.5		24.1		17.5		88.5		59.1		225.1
Baltimore City	78.9	47.2	1.8	142.3	25.5	9.9	37.1	20.7	106.8	45.5	234.4	1729.6
Calvert		3.8		-		8.9		1.9		-		-
Caroline		0.0		0.0		0.0		0.0		0.0		0.0
Cecil	0.6	0.0	0.0	0.0	0.2	0.0	0.3	0.0	0.9	0.0		0.0
Charles		0.0		0.0		0.0		0.0		0.0		0.0
Dorchester	4.9	0.6	0.1	0.1	1.6	0.2	2.3	0.4	6.6	0.4	14.4	-
Harford	2.2	0.8	0.1	0.2	0.7	0.1	1.0	0.6	2.9	0.6	6.5	-
Howard		0.0		0.0		0.0		0.0		0.0		0.0
Kent	0.2	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.3	0.0	0.7	-
Prince Georges	12.8	1.8	0.3	0.0	4.1	4.1	6.0	0.9	17.3	0.0	37.9	0.1
Saint Mary's	0.0	0.0	0.9	0.0	0.3	0.0	0.4	0.0	1.2	0.0	2.6	-
Wicomico	1.8	0.1	0.0	0.0	0.6	0.0	0.9	0.0	2.5	0.1	5.4	0.1
Alexandria City	10.2	2.2	0.2	-	3.3	5.1	4.8	1.1	13.7	-	30.2	-
Chesterfield	2.5	9.6	0.1	0.1	0.8	17.5	1.2	4.0	3.3	2.4	7.3	0.3
Henrico		1.1		0.0		0.4		0.2		1.9		0.0
Hopewell City	14.6	7.6	0.3	0.3	4.7	2.9	6.9	1.4	19.7	13.9	43.3	-
Louisa		22.7		-		53.0		11.4		-		-
Newport News City	12.8	9.6	0.3	3.9	4.2	2.7	6.0	13.9	17.4	9.3	38.1	36.7
Norfolk City	16.7	2.0	0.4	0.9	5.4	0.4	7.8	3.0	22.6	2.0	49.5	5.9
Northampton		0.0		0.0		0.2		0.0		0.0		0.7
Portsmouth City	5.2	14.4	0.1	5.9	1.7	3.1	2.5	20.8	7.0	14.9	15.5	-
Prince William	13.4	2.0	0.3	-	4.3	4.7	6.3	1.0	18.1	-	39.7	-
Richmond City	22.6	1.1	0.5	0.6	7.3	2.5	10.6	12.8	30.5	6.7	67.1	-
Spotsylvania	0.2	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.3	0.0	0.6	-
Westmoreland	0.3	0.1	0.0	0.0	0.1	0.0	0.1	0.1	0.4	0.1	0.9	0.2
Williamsburg City	2.3	0.1	0.1	0.0	0.7	0.0	1.1	0.0	3.1	0.1	6.8	-
York		12.1		0.2		18.7		4.5		8.9		-
TOTAL	199.5	199.1	5.7	178.8	68.1	155.0	98.9	189.6	284.4	167.3	624.6	2008.2

[1] POTW loadings were calculated for facilities where flows were 0.5 MGD.

[2] Loadings computed from approximately 122 industrial dischargers.

[3] I = Industry

priority for cleanup. The priority areas will be examined in the CBP report "Characterization of Chesapeake Bay".

SUBMERGED AQUATIC VEGETATION

Pattern of Decline

Submerged aquatic vegetation (SAV) has, in the past, been very abundant throughout Chesapeake Bay. Our current evidence indicates a pattern of SAV decline that includes all species in all sections of the Bay. A marked decline has occurred throughout the estuary since the mid-1960's. Present abundance of Bay grasses is at its lowest level in recorded history.

Historical analysis of sediments on Bay-grass seeds and pollen indicates a continuous presence of Bay grasses from the 17th century. In the last 50 years, there have been several distinct periods and patterns where Bay grasses have undergone major changes. An outbreak of eelgrass wasting disease occurred in 1930's and reduced SAV populations, as did a watermilfoil outbreak in the late 1950's and early 1960's. However, a far more dramatic and Bay-wide decrease in SAV populations occurred in the 1960's and 1970's where, unlike the eelgrass and milfoil events, all species in almost all areas of the Bay were affected. The change is not attributable to disease.

Because there has not been a significant change in SAV distribution along the east coast of the United States comparable to the Chesapeake Bay decline, it is most likely that water quality problems affecting the distribution of grasses in Bay are regional and specific to the Bay, its tributaries, and their drainage basin. Recent international studies have found that SAV declines in other countries are highly correlated with changing water-quality conditions, such as decreasing water clarity resulting from increased eutrophication, as sewage, agricultural runoff, and suspended sediment inputs increase. CBP work suggests that sediment composition and light availability are the most important factors controlling the distribution of SAV within regions of the Bay. In addition, SAV decline parallels historical increases in nutrients and chlorophyll a concentrations in the upper Bay and major tributaries that occurred first in freshwater parts and have now moved "down-river".

Value

The severity of the decline is heightened by the importance of SAV to the vitality of the Bay. The Bay grasses are vitally important to the Bay because of their value as large primary producers, food sources for waterfowl, habitat and nursery areas for many commercially important fish, controls for shoreline erosion, and mechanisms to buffer negative effects of excessive nutrients.

Numerous studies have shown that the primary productivity of SAV communities is among the highest recorded for any aquatic systems. However, trends in SAV biomass production follows those of its distribution and abundance. The average biomass estimates for SAV in the Bay are low relative to other communities. For example, we have estimated that some 40 percent of primary production in Bay was attributable to SAV in 1963 while only six percent is attributable to SAV in 1975. These trends along with other results are indicative of stressed plants, particularly in the upper Bay.

SAV provides food and habitat for many species of birds and animals. The most definitive linkage is between SAV and waterfowl. Some types of SAV are excellent food for waterfowl. In recent years, the most important waterfowl wintering areas have also been the most abundantly vegetated areas. Waterfowl have adapted to the SAV decline primarily by wintering elsewhere in the Atlantic Flyway.

SAV beds in Chesapeake Bay support larger populations of most animals than nearby unvegetated bottoms, and provide significant protection from predators. Fish abundance in SAV communities in the upper Bay are among the highest ever recorded, indicating that SAV are sources of food either directly, or indirectly, to important Bay species. Few commercially-important finfish use SAV beds as significant nursery habitats. However, lower Bay beds do serve as a primary blue crab nursery, supporting a very large number of juvenile blue crabs throughout the year.

Work in the upper Chesapeake Bay has shown that SAV is important in stablizing suspended sediments. As turbid water enters SAV beds on rising tides, sediments are effectively removed, and light transparency increases. Sediment resuspension is reduced in proportion to SAV biomass.

SAV also reduces nutrient levels in the water. Our studies show that, at moderate loading rates, nutrient concentrations are consistently lower in SAV communities than in unvegetated sites. Ammonium concentrations were one to 10 times lower, nitrate two to 10 times lower, and orthophosphate generally two to four times lower in the SAV community than in deeper, offshore waters. When loading rates and nutrient concentrations reached high levels, SAV was no longer effective in reducing nutrient levels.

Cause of the Decline

During the Bay program, investigators looked at light reduction as a major cause of SAV decline. Overall, factors governing light energy availability to submerged aquatic vegetation are the principal control for growth and survival. Bay grasses are currently living in a marginal light environment, and water quality problems, such as increases in nutrients and chlorophyll a concentrations in major tributaries and the main stem of the Chesapeake Bay over the past several decades, are seriously affecting the distribution and abundance of grasses in the Bay region. Epiphyte communities, those organisms that directly attach to submerged aquatic plant blades, can also limit light availability.

Another important factor contributing to the stress of SAV in the Bay is the input of herbicides to the ecosystem. Our laboratory and field experiments indicate that herbicides are not generally available to SAV in toxic levels, and their presence alone probably did not cause the SAV decline. However, herbicide-induced impacts could, in concert with the other major stresses (such as those from light limitation), create intolerable conditions for SAV existence.

In summary, the SAV decline parallels a general increase in nutrients, chlorophyll a concentrations, and turbidity in the upper Bay and major tributaries. This decline first ocurred in freshwater portions, and has moved down-river. The upper-Bay, western-shore, and lower-Bay communities have been the most severely impacted. Light, restricted by organic and inorganic suspended particles from runoff and nutrient loads, and by changes in physical-chemical regimes (salinity and temperature), is the principal factor controlling Bay-grass growth and survival. Bay grasses are now living in a marginal light environment and will be adversely stressed if water quality in the Bay declines further. Management programs that minimize sediment and nutrient loads will have to be improved and expanded if SAV is to flourish again throughout the Bay.

The "Characterization" report will address relationships between SAV, other natural resources, and water quality trends; the "Management Strategies" report will suggest ways to protect and/or enhance these resources.

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Credits

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PART I

HOW WE STUDIED THE BAY: ASKING AND ANSWERING THE QUESTIONS

PART I

HOW WE STUDIED THE BAY: ASKING AND ANSWERING THE QUESTIONS

INTRODUCTION

At a singular gathering in October 1977, EPA staff, officials from Maryland and Virginia, and citizens developed a five-year study plan for the Chesapeake Bay Program (CBP). As part of the plan, they identified the ten foremost water quality problems of the Bay, and methods needed to research those areas. These ten problems were:

- wetlands alteration
- shoreline erosion
- water quality effects of boating and shipping
- hydrologic modification
- fisheries modification
- shellfish bed closures
- accumulation of toxic substances
- dredging and dredged material disposal
- nutrient enrichment
- submerged aquatic vegetation

By early the following winter, three critical areas were chosen from the ten for intensive investigation -- Nutrient Enrichment, Toxic Substances, and the decline of Submerged Aquatic Vegetation (SAV).

In all three areas, we wanted to improve our understanding of major changes taking place in the Bay. Increasing development within the Bay area has enriched major tributaries and parts of the main Bay with nutrients, resulting in loss of dissolved oxygen and large algae blooms. In the nutrients area, CBP has assessed the relationship between nutrients and water quality, and the potential for future enrichment. Until inception of the Program, much of the basic information needed to assess the presence of toxic material in the Bay and its effects on Bay communities was not available, or poorly known. To build an information base upon which future measures and effects can be compared, the CBP estimated the distribution and abundance of toxic substances throughout the Bay. The past ten years have also revealed sharp declines in the diversity and density of SAV. The CBP looked at the role and value of SAV in the Bay ecosystem and at some of the most probable causes of its decline.

With the completion of most of the technical studies in the summer of 1981, the CBP began to analyze and integrate results. Early in the program the staff, State managers, citizens, and researchers posed a series of questions pertinent to managing the Bay. These questions appear at the end of this part of the report. Using the Management Questions as a guideline, scientists in each of the three problem areas jointly wrote research papers that integrate results across the specific problem areas. To best answer the questions for managers, decision-makers, and citizens, the authors integrated into their papers not only data from specific projects, but information from other research and world literature. The papers were drafted prior to September 1981 and include data up to that point, except where noted. (Some later data have been incorporated into the CBP's characterization process as they were available.) Drafts of the synthesis papers were carefully reviewed by scientists outside of the Bay area as well as by CBP staff and State participants in the Program. The major State agencies involved in contributing to, and reviewing the synthesis papers include: The Virginia State Water Control Board; the Maryland Department of Health and Mental Hygiene; the Maryland Department of Natural Resources; and the Pennsylvania Department of Environmental Resources.

These papers not only respond to many of the Management Questions, but also support the rest of the phases of CBP's program - water quality and resources characterization, environmental quality classification, and management strategies. The papers, for example, provide a sound technical foundation for the CBP's characterization process, presented in the third final report. In this analysis, many important parameters used to assess water quality in parts of the Bay were identified from information in the synthesis papers. Furthermore, the last final report on management strategies builds on the management questions and answers in the synthesis papers to present the best options for managing Chesapeake Bay.

In overview, this report represents the most technically comprehensive product of the Program. A list of all of the products and their relationship to the synthesis papers includes:

- 40 final reports on individual research projects, with summaries of each report.

- Description of the Program's computer model of the Chesapeake Bay system.

- Chesapeake Bay: Introduction to an Ecosystem--explains important ecological relationships and serves as a reference for the synthesis report, the characterization report, and the CBP management alternatives.

- Chesapeake Bay Program Technical Studies: A Synthesis--summarizes and explains the technical knowledge gained from the research projects funded by this program in the areas of nutrient enrichment, toxic substances, and submerged aquatic vegetation. It provides an understanding of the processes which affect the quality of Chesapeake Bay.

- Characterization of Chesapeake Bay -- Assesses trends in water quality and living resources over time, and examines relationships between the two.

- Chesapeake Bay Program Management Study -- Identifies control alternatives for agriculture, sewage treatment plants, industry, urban runoff, and construction; estimates costs and effectiveness of different approaches to remedy "hot spots."

STUDYING THE BAY

The integrity of Chesapeake Bay begins far from the actual estuary. The Bay itself lies within the Atlantic Coastal Plain but draws water from a drainage basin of 64,000 square miles that includes five states and parts of the Piedmont and Allegheny Plateaus. The diverse rock types found in the plateaus affect the chemical make-up of the many tributaries running to the Bay. At the estuary, this chemically-varied riverine water meets and mixes with oceanic water to form a variety of physical and chemical environments. Since organisms living in water are suited to different ranges of temperature and chemical mixtures, how the mixture changes naturally, or unnaturally, influences the ability of the Bay to maintain a wide variety of life.

More than 2000 species of plants and animals inhabit the Bay. These plants and animals live in communities, such as in marshes or on the bottom, and depend on each other for food and shelter. Communities respond naturally to changes in the environment through changes in diversity and abundance. Some variations result from seasonal changes, others from long-term fluctuations; still others are caused by human influences. Assigning the cause of this biological variation to natural or human influences is one of the most difficult problems encountered in ecology.

To better understand the major processes governing the Bay and its inhabitants, and how they may be affected by continued input of pollutants, CBP devised Bay-wide research plans focusing on three study areas--nutrient enrichment, toxic chemicals, and submerged aquatic vegetation. State and CBP staff, together with EPA personnel, wrote plans of action and asked any interested scientists to respond with suggestions and proposals for doing research. These proposals were reviewed and modified, with selected ones chosen for funding during the spring and summer of 1978. The program spent nearly $17 million on 40 research projects, grants, cooperative agreements, and contracts. This approach to funding the Program's studies allowed a broad research community to take part in the investigations.

Scientists and institutes often cooperated in collecting and analyzing their data. They shared research vessels, used commmon sampling sites, and similar time periods. One of the largest cooperative efforts occurred during a Bay-wide, water quality survey. During this series of cruises aboard several research vessels, scientists from a dozen private research institutions, and State and Federal agencies collected information on nutrients, other important water quality factors, and hydrodynamics of the Bay and its tributaries.

To ensure that the diverse data collected and analyzed during the five years of investigation were credible, properly maintained, and analyzed accurately, CBP undertook a quality assurance program. In this program a computer and research staff made sure the data from research projects and historical sources were reliable. The staff also prepared the data for computer storage and analysis by devising a set of standardized names for variables and units. Statistical analyses were documented in directories and reviewed by CBP technical staff. In addition, inferences derived from the analyses were reviewed by both technical and computer staff to assure statistical validity and technical accuracy.

The synthesis papers are divided into three parts. The first presents a synthesis of information on nutrient enrichment -- what the enrichment problem is, what chemical, physical, and biological processes interact to sustain the problem, and what the major sources of nutrients to the Bay are. The second part covers the CBP toxic substances program. This section discusses our knowledge of toxic chemicals, sources, distribution, and concentration of metals and organic compounds in the water and sediment, and how geochemical and biological processes of the Bay can affect the character and distribution of toxic substances. The third part explains the results of CBP's SAV investigations in light of what factors caused its decline. The sections in this part discuss the distribution and abundance of SAV now and in the past, the value of SAV to the Bay ecosystem, the possibility of herbicides as a major factor in its decline, and light as the link between SAV and its decline.

MANAGEMENT QUESTIONS AND ANSWERS - NUTRIENT ENRICHMENT

1. The Nutrient Enrichment Problem

1.1. Where and how severe are nutrient enrichment problems in the Bay?

The upper Bay, upper Potomac, and upper James are nutrient enriched and are the sites of current and potential problems. The mid-Bay, other Western Shore tributaries and Susquehanna are less enriched, but could become nutrient enrichment problems.

1.2. What are the consequences of nutrient enrichment?

The consequences of nutrient enrichment are enhanced plant production and higher levels of organic matter in the water column. This organic matter may accumulate in deep water, where its degradation results in oxygen depletion. Mobile estuarine organisms leave the low oxygen water; stationary organisms succumb. However, it is possible that planktivorous organisms, like menhaden, could benefit from increased production of plankton.

Nutrient enrichment may also alter the species composition of phytoplankton, potentially causing changes in fisheries.

1.2.a. *What factors are required by phytoplankton for growth?*

Phytoplankton require light, nutrients, appropriate temperature, appropriate salinity, and innumberable other factors. Of the criteria listed above, only the nutrients, specifically nitrogen and phosphorus, are controllable by people. Any element can be limiting: phytoplankton cannot grow in inadequate light or in areas having inappropriate salinity.

1.3. what are the advantages and disadvantages of the commonly used criteria for evaluating a water quality problem related to nutrient enrichment?

Chlorophyll *a* levels are useful because they give a direct indication of phytoplankton density, which is one of the important consequences of nutrient enrichment. There is also a fairly good historical record for chlorophyll *a*. However, laboratory techniques have changed over the years, particularly in the mid-1970's, and there may be a problem in comparing current data with historical data. Another disadvantage is that it is possible to have high chlorophyll *a* levels in non-enriched situations, because of circulation and behavioral responses of phytoplankton. For this reason, chlorophyll *a* measurements should be repeated over time to corroborate their validity.

Secchi depth is a commonly used criterion because its determination is inexpensive, and it is reliably measured from person to person. It also has a long historical record. On the other hand, it is not sensitive to small changes in photic zone, which can reflect large changes in turbidity. It cannot distinguish between inorganic and organic sources of turbidity.

Measurement of inorganic nutrient concentrations is fairly simple, and methods have been reasonably uniform historically. However, while high inorganic nutrient levels indicate a problem, low levels give no indication of enrichment because nutrients can be tied up in organic forms. Measurement of total nutrient levels would make it possible to assess the total enrichment of the system, but is difficult to carry out and does not have good historical record.

Dissolved oxygen levels are tremendously useful to managers because oxygen depletion is the major consequence of enrichment. However, dissolved oxygen should be expressed as oxygen deficit (a term related to saturation level) and should account for season. The disadvantage is that short-term events, like wind, can affect dissolved oxygen levels.

Algal species shifts are a good indicator of nutrient enrichment in fresh waters, where blue-green algal blooms are known to occur under enriched conditions. However, in estuarine systems the "normal" algal flora is not well defined, so changes due to nutrient enrichment cannot at present be documented.

1.4. What techniques can be used to evaluate or predict nutrient enrichment problems?

Nutrient enrichment indices are desirable to managers because they give an assessment of enrichment stated in very simple terms. Their disadvantages are that they may not provide an adequate reflection of complex ecological conditions; they are not generally applicable from system to system, and they are subjective.

Computer-based mathematical models can quantify multiple combinations of processes and conditions that are beyond the capacity of human comprehension. They are valuable planning tools because they can project the response of an estuarine system to specific conditions. On the other hand, they are not generally applicable because specific pollutants and systems require specific models. Calibration and verification may be difficult because of gaps in data. Finally, people are inclined to expect models to provide final answers, perhaps not scruitinizing the modelling process or results sufficiently.

1.5. What are the historical trends in nutrient enrichment?

In some areas of the Chesapeake Bay system, chlorophyll *a* concentrations have increased from a pre-settlement level of less than 30 ug/liter to over 60 ug/l during the summer. These areas include the upper Bay, upper Potomac, and upper James and, for this reason such areas are considered to be heavily enriched. This question will be evaluated further by the Chesapeake Bay Program Characterization Report.

1.6 What are the greatest needs for further research?

The primary need in Chesapeake Bay research is long-term coordination. Many gaps need to be filled in basic research, and this can only be accomplished if areas needing further research are identified and a concerted, coordinated, long-term effort made to fill the gaps.

Nutrients research would be furthered by the development of better models for the estuarine system.

Finally, better understanding of processes like hydrodynamics, species composition, algal productivity, assimilative capacity, and effects on fisheries is needed.

2. Nutrient Processes

2.1 What nutrients are available, at what times, in the Chesapeake Bay system?

The availability of nutrients in Chesapeake Bay follows an annual cycle which has three prominent events. First, the spring freshet brings a substantial amount of nitrate into the Bay. Second, deoxygenation of deep water in summer results in phosphate release from the sediments and accumulation of both phosphate and ammonium in the deoxygenated region. Third, reoxygenation of deep waters in fall corresponds with the loss of phosphate from the water column and the oxidation of ammonium to nitrite and nitrate.

2.2 What is nutrient limitation? How does it regulate algal production?

Healthy algae require carbon, nitrogen and phosphorus in certain ratios. Algal production is regulated by the nutrient in least abundance relative to the algal requirement (assuming that other factors like light and salinity are adequate). The nutrient regulating algal production is referred to as the limiting nutrient; addition of the limiting nutrient stimulates algal production. (Taft pp 12-29)

2.3 When and where are phosphorus and nitrogen limiting?

The potential for phosphorus limitation in the tidal fresh regions of the Bay exists throughout the year. This is because blue-green algae, major constituents of fresh-water systems, are not limited by nitrogen due to their ability to fix this nutrient from the atmosphere. (P limitation is expressed as a potential because light may actually limit growth.) Phosphorus is limiting in the Bay stem during spring, because this is the major period of nitrogen influx from the tributaries, while phosphorus is still largely bound to the sediments because of oxygenated conditions. (In the maximum turbidity zone light may in fact be limiting, so the potential for phosphorus limitation may not be expressed.)

Nitrogen is limiting over most of the main Bay in summer (with the exception of the maximum turbidity zone, where the potential for nitrogen limitation exists but growth may actually be limited by light). During the summer phosphorus is provided from the sediments because of anoxic conditions, while there is no major influx of nitrogen.

2.4 How does light regulate phytoplankton production? When and where is light limiting?

Light may limit algal production when turbidity is high due to sediment accumulation in the water column. This happens particularly in spring in the upper Bay when sediment influx is extensive. It may happen in maximum turbidity zones year-round.

Chesapeake Bay Program research indicates that physical processes may lift phytoplankton from dark subsurface layers into the surface waters, overcoming the potential for light limitation for these algae. Light limitation will also not be important where adequate mixing brings phytoplankton to the surface regularly.

2.5 How does nutrient enrichment affect algal production?

Whether nutrient enrichment increases algal production depends on whether the nutrient is limiting, whether luxuriant uptake occurs, and whether the nutrient is in its "preferred" form.

Where a nutrient is limiting, its addition will increase algal production. Addition of a non-limiting nutrient may also ultimately increase biomass because of luxuriant uptake, in which phytoplankton take up a nutrient but do not immediately utilize it. Internal stores of the nutrient are created, which can be drawn from later if there is a shift in the limiting nutrient.

Addition of nitrate or nitrite will not stimulate phytoplankton growth in the presence of a threshold level of ammonium. Phytoplankton preferentially take up ammonium and will not utilize added nitrate in the presence of ammonium. The phenomenon was confirmed as a result of Bay Program research, and is particularly significant in the spring when the large inputs of nitrate appear to pass through into the lower areas of the Bay, unutilized because of the presence of ammonium.

2.6 How does nutrient enrichment affect species composition, diversity and trophic relationships of phytoplankton?

Shifts to blue-green algal dominance in the tidal fresh regions have been a well-documented response to nutrient enrichment. Such compositional shifts have not been shown in the higher salinity areas of the estuary. Where blooms clearly do occur in response to nutrient enrichment, they result in a decline in diversity and stability of the phytoplankton community. Thus rapid growth can be followed by rapid declines, leading to unaesthetic conditions, de-oxygenation, and other consequences.

Nutrient enrichment probably affects trophic relationships, because blue-green algae are inedible for most plankton feeders. When blue-green algal blooms occur and little other phytoplankton is available in the rivers, plankton feeders must find some other food source (switch-feeding) or decline.

2.7 How might higher trophic levels be affected by the changes described in 2.6?

When species shifts occur, the dominant organism may not provide complete nutrition to grazers, which may shift the grazing population. Planktivorous species may be favored by increased phytoplankton production and species shifts; trends in menhaden populations may show this.

2.8 What are the major water column nutrient cycling processes?

Important processes contributing to water column nutrient dynamics are hydrodynamics, grazing, decomposition, and bacterial transformations of inorganic nutrients.

Grazing by predators (plankton feeders, etc.) and decomposition by bacteria and fungi are the regeneration mechanisms. Nutrient regeneration is important because phytoplankton can use primarily inorganic nutrients. Regeneration can be a major source of nutrients to phytoplankton during certain periods.

Grazing of phytoplankton by predators yields production of feces by the grazers, as well as release of materials from the phytoplankton cells. This facilitates bacterial degradation of phytoplankton organic material.

Bacteria and fungi decompose dead organic matter, converting complex organic molecules into simple inorganic molecules like nitrate, ammonia, phosphate, nitrite. They also transform inorganic nutrients in nitrification and denitrification. New data from CBP research indicates that nitrification is important in the fall, resulting in a nitrite maximum then. Nitrogen fixation may be important in the tidal fresh portions but is insignificant in the rest of the Bay.

Hydrodynamic processes like circulation, wind, and tides transport and dilute nutrients. Increased stratification in summer results in nutrients being held below the pycnocline. Important vertical exchange processes are dilution and tidal or wind mixing. These processes, combined with chemical and biological events, tend to retain nutrients in two-layered estuaries like the Bay.

2.9 What are the major sediment nutrient cycling processes, and how do these contribute to nutrient enrichment?

The important sediment nutrient processes are flux from the sediments into the water column and vice versa, nutrient cycling, and binding of phosphorus by iron and manganese oxides.

Phosphorus flux rates depend on oxidation state: under anaerobic conditions phosphorus is released from the sediments. This is an important process in deeper Bay waters in mid-summer, which may then be anoxic. During the rest of the year waters are oxic, and iron and manganese compounds retain phosphorus in the sediments.

Ammonia flux rates vary with interstitial water concentration. In mid-summer, ammonia is not readily oxidized and accumulates in bottom waters.

CBP research indicates that water column nutrient recycling yields 5 to 10 times as much available nutrient as sediment processes.

2.10 What factors affect levels of dissolved oxygen in Bay waters and sediments?

Oxygen is produced by photosynthesis. It is also added by re-aeration, resulting from diffusion of oxygen from air into upper waters. Its rate depends on wind, temperature, and the oxygen gradient in the water.

Oxygen is utilized by respiration, especially in summer. Respiration is carried out by phytoplankton, microbes, and animals. Oxygen is also utilized by microbes as they oxidize reduced chemical species like ammonia. These processes result in BOD (biochemical oxygen demand) and SOD (sediment oxygen demand).

In summer, respiration rates are high because of elevated temperatures and high production. Respiration of detritus in bottom waters depletes oxygen there, and stratification prevents re-aeration. In some areas of the Bay this summer anoxia is probably natural, but it is aggravated by nutrient enrichment.

2.11 Which processes dominate seasonally?

In spring, the major event is the nitrate influx and the effect of freshwater on stratification.

In summer, bottom waters are depleted of oxygen by respiration; replenishment is prevented by stratification. Phosphorus is released by the sediments, and ammonia accumulates.

In fall, stratification is reduced, and the water is reaerated. Nutrients are biologically and chemically transformed as a result of the newly available oxygen.

In winter, the water is well oxygenated. Low temperatures and light levels reduce system productivity; nutrients may be present in measurable quantities.

3. Nutrient and Sediment Loads[1]

3.1 What is the atmospheric contribution to nutrient input?

The atmospheric nutrient contribution that directly enters tidal waters is at least 40 million pounds of nitrogen and 1.6 million pounds of phosphorous each year [Table VIII.1(a)]. This load constitutes about 13 percent of the annual nitrogen, and five percent of the annual phosphorous input budgets [Table VIII.1(b)]. Seasonally, atmospheric sources may make up as much as 20 percent of the seasonal total nitrogen (winter) input and five percent of the seasonal total phosphorous (summer) input and as little as seven percent of the total nitrogen load and three percent of the total phosphorous load in the winter and spring [Tables VIII.2(b) to VIII.5(b)].

3.2. What percentage of the nutrients is from point sources?

On an annual basis, about 20 to 25 percent of the total nitrogen load entering tidal waters comes from point sources basin-wide [Table VIII.1(b)]. This percentage range would hold even if <u>all</u> of the point sources load discharged above the fall line were transported directly to the tidal system (a very conservative assumption since losses undoubtedly occur in transport, especially during the summer). The proportions are relatively invariant throughout the year, reaching the lower end of the range in the spring and the upper end in the summer and fall.

To make a reasonable estimate of the percentage of the phosphorous load deriving from point sources, some manipulations of the riverine loading models developed in Chapter III were performed. Low flow values were chosen for each of the major tributaries[2], and the total phosphorous load expected to occur at these flows was computed. This total flow (sum of all three tributaries) was about 9660 cubic feet per second. Note from Table IV.12(a) that the total point source flow entering above the fall line is about 688 cfs. The total phosphorus load computed to be carried to the tidal system at a stream discharge of 9660 cfsd is about 1950 lbs./day or about 0.7 million pounds per year. If the extremely conservative assumption is made that all of this load derives from point source discharges and is summed with the 10.8 million pounds of point source phosphorous discharged per year below the fall line, the total point source contribution of phosphorous is computed to be about 40 percent of the total annual phosphorous input budget of around 11 million pounds per day. Seasonally, the point source contribution of phosphorous makes up as much as 65 percent of the fall total phosphorus input budget and as little as 25 percent of the summer total phosphorous input budget.

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[1]Answers to all questions in the following section are based on Chapter 3 of Part II in "Chesapeake Bay Program Technical Studies: A Synthesis."

[2]The "daily discharge that is greater than or equal to the flows that occur 10 percent of the time" was computed for each major tributary. They are: Susquehanna, 6640 cfsd; Potomac, 1690 cfsd; James, 1330 cfsd.

3.3. What percentage of nutrients is from nonpoint sources and how do they vary over time?

To discuss nonpoint sources within the structure of this paper, we define three categories of diffuse sources. They are:

i) Atmospheric contributions

ii) Land runoff/base flow contributions

iii) Bottom contributions

Categories i) and ii) are covered separately elsewhere in this section of the Management Questions. To answer this Management Question, we define nonpoint sources as the sum of land runoff and base flow (groundwater discharge) carried by fluvial processes to the tidal Bay system. Contributions from the coastal plain are not considered.

On an annual basis, the mean total nonpoint source nitrogen loading is about 50 to 55 percent of the total input budget, or about 160 to 177 million pounds of nitrogen per year [Tables VIII.1(a) and VIII.1(b)], making this the single largest external source of nitrogen loading to the Bay. Seasonally, the variation in the nonpoint source nitrogen loading is quite dramatic, ranging from about 23-25 million pounds in the summer (36 - 39 percent of the total source load) to around 69 - 71 million pounds in the spring (63 - 66 percent of the total spring nitrogen load). The dominant species of nonpoint source nitrogen at the fall line is always nitrite-nitrate, making up consistently between 62 and 64 percent of the total nitrogen from this source.

On an average annual basis, the nonpoint source loading of phosphorous is about 30 to 34 percent of the total phosphorous input budget, ranging from around 9 to 10 million pounds per year. As much as 65 to 70 percent of this load on an annual basis is in the suspended phase, meaning most of the phosphorous being carried to the Bay is associated with particulate matter and therefore, not immediately available for phytoplankton utilization. Seasonally, the nonpoint phosphorous contribution probably varies from about 1.2 to 1.4 million pounds (only about 10-11 percent of the summer total phosphorous budget) in the summer to about 4 million pounds in the spring or 55 percent of the total spring input budget of phosphorous from all sources. The very low percentage of the load coming from fluvial sources in the summer is due mainly to the dominant effect of bottom sources of phosphorous released in that season.

3.4. What are the pollutant runoff rates for particular land uses?

The information upon which this answer is based may be found in the EPA Chesapeake Bay Program Information Series Nutrient Summary 3: "Assessment of Nonpoint Source Discharge to Chesapeake Bay" (unpublished). The data presented in that report are the results of a preliminary analysis of the data from the Chesapeake Bay Program Intensive Watershed Studies (IWS).

The analysis performed on the data used the volume-weighted mean concentrations of storm event runoff, computed for the CBP studies (Hartigan, 1981) along with some typical expected average annual runoff volumes for various land use/soil texture combinations, to generate generalized annual pollutant loadings for various classes of land use. These data are presented in Table VIII.6. Although the analysis, in its preliminary state, necessarily produced overlapping ranges of runoff loading rates among land uses, the data in Table VIII.6 allow us to assign order of magnitude rankings for the land uses studied by areal loading rate. The generalized rankings are shown in Table VIII.7.

In all cases, the highest unit area loading rates were generally exhibited by cropland sites and the lowest by forest sites. (N.B. The rankings shown in Table VIII.7 are a very broad generalization!)

TABLE VIII.7 CONCENTRATION, MG/L (TOP LINE), AND LOADING RATES, LBS/AC/YR (BOTTOM LINE), FOR TOTAL SUSPENDED SOLIDS, TOTAL PHOSPHORUS, ORTHOPHOSPHATE, TOTAL NITROGEN, AND NITRITE-NITRATE FROM VARIOUS USES OF LAND(1)(2)

Land Use	SED	TP	OP	TN	NO23
Cropland(3)	46.5-3202.8	0.21-12.49	0.01-2.77	1.3-22.2	0.02-16.20
	10.54-2460.83	0.05-9.78	0.01-2.20	0.75-17.59	0.02-12.90
Pasture	145.20-669.70	0.38-1.12	0.06-0.14	2.20-6.20	0.30-1.71
	16.45-303.50	0.04-0.51	0.01-0.06	1.25-2.81	0.03-0.78
Forest	9.40-71.5	0.06-0.23	0.00-0.04	0.40-1.10	0.01-0.48
	0.53-48.60	0.00-0.16	0.00-0.03	0.02-1.00	0.00-0.33
Residential	38.00-634.4	0.10-1.66	0.02-0.17	0.70-2.8	0.26-0.90
	47.40-2395.1	0.13-5.22	0.03-0.54	0.87-8.82	0.32-2.84

(1)Volume-weighted concentration data from preliminary analysis by NVPDC, concentration in milligrams per liter. Personal Communication: "Volume-Weighted Mean Concentrations of Storm Event Runoff from EPA/CBP Test Watersheds," J.P. Hartigan, Regional Resources Division, Norther Virginia Planning District Commission, Falls Church, VA, October 13, 1981.

(2)Loading rate computed by CBP staff, in lbs./ac./year.

(3)Cropland includes primarily conventional and minimum till with some no-till land practices.

TABLE VIII.8 GENERALIZED RANKING OF LAND USES BY UNIT AREA RUNOFF LOADING RATE (1 = HIGHEST RATE, 4 = LOWEST RATE)

Land Use	TN	NO23	TP	OP	SED
Cropland	1	1	1	1	1
Residential	2	2	2	2	2
Pasture	3	3	3	3	3
Forest	4	4	4	4	4

For instance, one of the cropland sites in the southern portion of the western shore produced less nitrite-nitrate per acre than one of the forest sites on the upper Eastern Shore. Although this example may be anomalous, it illustrates that there is overlap in the data and that the rankings shown are general in nature and by no means apply to all sites on all soil types. They are intended to give indications of which land uses, in general, have the highest loading rates and which uses have the lowest rates, relative to one another.

Within the class of developed land use types such as residential and commercial uses, it has been shown (Smullen, Hartigan, and Grizzard, 1978; Smullen 1979, NVPDC 1979) that there is a direct relationship between intensity of land use, often measured as the imperviousness of a site, and the unit area loading rate yield of nutrients. A ranking of the urban uses by loading rate is shown in Table VIII.8.

TABLE VIII.9 RANKING OF URBAN LAND USES BY UNIT AREA LOADING RATE[1] FOR NUTRIENTS (HIGHEST LOADING RATE = 1, LOWEST LOADING RATE = 7)

Land Use	Ranking
Central Business District	1
Shopping Center	2
High-Rise Residential	3
Multiple Family Housing	4
High Density Single Family Housing	5
Medium Density Single Family Housing	6
Low Density Single Family Housing	7

In general, urban uses exhibit higher unit area loading rates of nutrients than forest or pasture uses and lower rates than cropland uses. Exceptions to this "rule of thumb" are that pasture typically will yield slightly higher rates than the very low-density residential uses and that well-managed, low-tillage cropland uses on pervious soils can yield lower rates than some of the more intensive urban uses.

3.5. What percentages of nonpoint source nutrient loadings can be attributed to particular land uses?

To answer this question with any level of precision, we first must accept two basic assumptions to facilitate the estimate, and they are: (1) that the land uses are homogeneously distributed above the fall line; and (2) that baseflow loadings (groundwater contributions) of nutrients may be considered a constant background load, and the nonpoint load is measured as surface runoff

[1] (Smullen, Hartigan, and Grizzard 1977)

and interflow[1] nutrient loadings. The homogeniety of land use assumption is considered reasonable because most of the urban population resides on the coastal plain (below the fall line) and, with the exception of the mountainous areas, the agricultural and forest lands in the basin are fairly evenly distributed. This assumption is necessary because the closer a source is to the Bay, the more effect its loading will have on the water quality of the system. Thus, it is important that no large mass of a particular land use type above the fall line be closer than any other type or there would be a skew of the loadings at the fall line reflecting that skew in the land use distribution. The second assumption is necessary because we do not intuitively understand the functional relationship between land use and the quality of groundwater discharge on basins the size of the Potomac, James, and Susquehanna.[2] We do know isolated facts -- such as, the more fertilizer applied, the greater the opportunity for increasing groundwater nitrate levels and the resulting baseflow nitrate loadings in the stream. For the purpose of this analysis, it is enough to accept that for land uses that do not involve a lot of impervious cover, the baseflow loadings will move reasonably well with the runoff loadings. That is to say, that land uses exhibiting higher nutrient runoff loadings will produce groundwater discharge loadings equal to or greater than those from uses exhibiting lower runoff nutrient loadings.

The land uses above the fall line of the Chesapeake basin are about: 60-65 percent forested, 15-20 percent cropland, 8-12 percent pasture, 3-5 percent urban/suburban, and 2-14 percent other. These are rough estimates made from existing land use maps and will adequately serve the purpose of this "order-of-magnitude" analysis. Land use/nutrient loading rate relationships developed locally within the Chesapeake basin (Smullen, Hartigan, and Grizzard 1978, Smullen 1979, NVPDC 1979) used for this analysis are shown below:

Land Use	Percent in Basin	Estimated Loading Rate (lbs./ac./yr.)	
		TN	TP
Cropland	15-20	8-18	1.5-5
Pasture	8-12	2-6	.3-.5
Forest	60-65	.5-2	.05-.1
Urban/Suburban	3-5	4-10	1-2

[1]Interflow is the lateral movement of water through soils to streams at shallow soil depths during and directly after storm events. It is of short duration and, for our purposes, can be considered to be part of the runoff hydrograph.

[2]This is a good example of why assessments such as this are best made with mathematic models. They facilitate the orderly sorting out of base flow, runoff, and interflow and allow the analyst to handle groundwater contributions by inspection of observed flow data.

The unit area loading rates shown above were weighted by the fractions of the land areas in each use and the following ranges of loading fractions were obtained:[1]

Land Use	Percent of Nonpoint Source Load TN	Percent of Nonpoint Source Load TP
Cropland	45-70	60-85
Pasture	4-13	3-8
Forest	9-30	4-8
Urban	2-12	4-12

In summary, agricultural cropland appears to produce the largest fraction of the nonpoint source load from above the fall lines by at least a factor of two for both nitrogen and phosphorous. This is partly due to a high unit area loading rate for cropland and mostly due to the large percentage of the land area in this use. Forest loadings of nitrogen are the next highest percentage and this is entirely due to the large fraction of the watershed still being in forest land. Urban/suburban and pasture lands above the fall line produce approximately equal loads.

By inspection, the percentages shown above would change very little if the Coastal Plain areas were included. Although the three major metropolitan areas (Washington, D.C., Richmond, Virginia, and Baltimore, Maryland) would increase the total amount of urban land area, this increase would probably be offset by the large rural land areas of the eastern and western shore portion of the Coastal Plain. At any rate, even if the proportion of urban area doubled, cropland would still be the largest nonpoint source nutrient load by an approximate factor of three.

3.6. What are the nutrient loadings from the fall line?

The nutrient loadings from the fall line are shown in Tables III.10 and again in Tables VIII.2 through VIII.5 in Chapter 3 of Part II in this report. The values for total nitrogen and total phosphorus are shown again below in millions of pounds.

	Annual	Winter	Spring	Summer	Fall
TN	178.1	51.4	72.2	25.1	27.9
TP	10.3	2.97	4.29	1.42	0.47

The percentage of the annual above fall line load produced in each season are shown below:

[1]Some best and worst case assumptions were used along with some common sense judgment. For example, the lower range of cropland loading was produced by assuming the lowest loading rate/percent land use combination for cropland and the middle value of the ranges for all other uses.

	Winter	Spring	Summer	Fall
TN	28.9	40.5	14.1	15.7
TP	28.8	41.7	13.8	4.6

From the data presented, it can be seen that the largest fraction of the fluvial nutrient load (40 percent of both nitrogen and phosphorus) is discharged to the tidal system during the spring. Observation of the data in Table III.10 shows that a large fraction of these spring loads are in forms of nutrients that are readily available for aquatic plant uptake[1], with 68 percent of the nitrogen as ammonia or nitrite-nitrate and 34 percent of the phosphorus as orthophosphorous. This is important since the spring is the critical start-up period for the phytoplankton growing season, the aquatic plant growth that will dominate, in part, the dissolved oxygen and chlorophyll conditions in the Bay through the summer and into the early fall. As noted elsewhere in this chapter, the predominant upstream source of the riverine transported spring nutrient load is probably runoff and groundwater discharge from agricultural lands. The next most important source of nitrogen (but probably lower by almost an order of magnitude) in spring river discharge from above the fall line is probably runoff and groundwater discharge from the melting of the snow-pack in the physiographic provinces upstream of the Piedmont (see Figure III.2).

The summer is the period during which the plankton growth in the Bay reaches the annual maximum (see Chapter 2 of this part). The fluvial transported nutrients play a lesser role during this period providing only about 39 percent of the readily available nitrogen forms of plant nutrients and only about 5 percent of the readily available phosphorus. Plankton communities flourish during this period primarily by recycling nutrients already in the water column (put there in part by the spring fluvial process) as noted in Chapter VII (Table VII.5); and secondarily by the supply of nitrogen from atmospheric, point and bottom sources and by the supply of phosphorus from point and bottom sources.

3.7. What do the bottom sediments contribute to nutrient inputs?

On an annual basis, bottom sediments contribute 32 million pounds of nitrogen and seven million pounds of phosphorus [Table VIII.1(a)]. This makes up about 10 and 25 percent of the annual nitrogen and phosphorous budgets, respectively [Table VIII.1(b)]. However, the nitrogen contributed from the bottom source is predominately ammonia and makes up about 45 percent of the total annual Bay-wide contribution of this nitrogen species, which is most preferred by aquatic plants. More than 50 percent of the externally supplied water column ammonia produced during the spring and summer comes from the benthos.

The sediments have their most dramatic effect on the nutrient input budget as a source of phosphorous in the summer. As discussed in Section V, most of phosphorous migrating up through the sediments via the pore waters probably is fixed chemically by iron in the overlying oxygen-rich waters and held in a fluff layer as a small particle, or floc. This process occurs during most of the year (late fall, winter, spring). However, when the oxygen in the lower layers of the Bay waters is depleted for periods during the summer, most of the phosphorus incorporated or stored during the rest of the year probably is released over a very short period of time. The result is that as much as 62 percent of the phosphorous input to the Bay in the summer could come from this source. Other than recycling, the bottom source is probably the single largest factor in the supply of phosphorous for summer primary productivity.

3.8. What are the flux rates of nutrients from the bottom sediments and how do they vary seasonally?

The bottom flux rates for nitrogen range from as low as 0.5 pounds of N per square foot per day in portions of the upper Bay in the spring to as high as 5 pounds of N per square foot per day in portions of the upper Bay in the spring and summer. The annual seasonal Bay-wide average flux rates for nitrogen are shown below:

	Nitrogen Benthic Flux of Nitrogen (Thousands of Pounds per day)	Percent of Annual Average
Winter	88.1	100
Spring	75.3	85
Summer	98.4	111
Fall	91.2	103
Annual Average	88.3	

As can be seen above, the summer period exhibits the highest flux rate of nitrogen from the sediments, and the spring the lowest. The nitrogen is moving out of the sediments fastest when the standing crop of phytoplankton is largest, and being produced in a form readily available for plant uptake.

As discussed previously, the seasonal variation of phosphorous flux from the sediment to the water column is severe, with about 85 percent of the total annual input being released rapidly sometime from late May to mid-June, with most of the other 15 percent released from that time through late summer.

The maximum Bay-wide phosphorous release rate might be as high as one-half million pounds a day during the period of the rapid onset of bottom-water anoxia. This rate probably levels off to about 16,000 pounds per day by late summer and down to near zero by sometime in late fall.

3.9. Given the estimated loadings of nutrients for each of the sources, which will be the most important in terms of their effects on the Bay system?

This is a difficult question to answer, because there are so many potential effects on the Bay system that could result from variations in nutrient loadings. Some effects are understood well; some not so well, and some are unknown. However, to provide an answer to this question, we will consider the potential effects on Bay-wide primary production which might result from variations in the amounts of nutrients entering from various sources.

On an annual basis (Table VII.2), probably only about 20 to 30 percent of primary production in the Bay proper is supported by nitrogen and phosphorous entering the water column from external sources. We will assume that nutrient recycling rates by phytoplankton would vary only moderately in response to changes in external nutrient supply. Given this assumption, it can be seen from the data in Table VII.2 that even as much as a 50 percent reduction in both point and nonpoint source annual nutrient loadings may result in only a 10 percent reduction in Bay-wide primary production. Seasonally, this effect could decrease to a 5 percent reduction in summer production in response to a 50 percent reduction of summer point/nonpoint nutrient loading. If these loading reductions were sustained, production would probably decrease futher as the nutrient reservoir in the sediments depleted over time. These estimated decreases of primary production in the short-term approach the detection limit of our ability to assess such reductions.

The important point in this discussion is that changes in lower Bay water quality (essentially meaning the great majority of the Bay that lies below the mouth of the Patuxent) in response to changes in nutrient inputs would probably take place slowly over decades. However, the upper portions of the Bay and the tidal tributaries would be much more responsive to change in nutrient loads than the main Bay. The nutrient loads that the main Bay receives must travel through these smaller, heavily impacted areas of the system.

The nutrient inputs are diluted as they move towards the lower Bay as a function of ever increasing volume. In addition, the surface area available for contributing nutrients from the sediments is much greater in the main Bay than in the upper portions of the system, resulting in much larger bottom releases of nutrients. These factors and others create a situation in the main Bay that tends to buffer or dampen water quality response to changes in anthropogenic nutrient loadings. It is, therefore, reasonable to expect the water quality of the upper areas (tidal fresh areas) of the system to respond more quickly to load reductions than the areas of the lower main Bay.

The apparent improvement in the water quality of the upper Potomac in response to decreased nutrient loadings over the last decade would seem to support this concept. Even though some unknown amount of that improvement probably results from differing climatic conditions over the last ten years, some degree of the improvement is most likely due to the decreases in the external nutrient supply from POTW's. We would not expect to see immediate changes in lower Bay water quality due to that reduction of loading and, in fact, have not. Such a change could only be seen over a much longer period of time and to a lesser (diluted) extent. This situation would seem to support the concept that if we manage the local ("near field") problems, the main Bay ("far field") will, in time, respond in kind. An aggressive policy of water quality improvement in currently adversely impacted areas should insure the maintenance of a nondegradation condition in the main Bay.

MANAGEMENT QUESTIONS AND ANSWERS - TOXIC SUBSTANCES

1. Is there a toxic chemical problem in the Chesapeake Bay?

There are trends of general concern and specific problem areas.

There is concern that grass, shad, and bass have declined in the last three decades and that oyster reproduction has diminished. In the James River, chlorine is strongly suspected of causing massive fish kills and Kepone has resulted in closure of the estuary to fishing for years. At the same time, there is an increase in the number of potentially toxic chemicals synthesized, produced, and used in the region. Analysis of a sediment core from the northern Bay, for example, reveals an upward increase in metal content of Cu and Zn with time. Enrichment factors range from 3 to 20.

Although it is recognized that toxicants accumulate in certain biota many thousandfold more than ambient concentrations, the link between cause and effect still eludes scientists. Toxic chemicals, however, are strongly suspected of being partly responsible for the decline of essential biotic components. The fact that many compounds are carcinogenic to mammals is cause for concern.

Major problem areas are Baltimore Harbor - Patapsco River - and Norfolk Harbor - Elizabeth River - which are sources of industrial/municipal discharge and shipping activity. Because of their limited circulation, these areas are natural "sinks" for toxics adsorbed on fine sediment. Concentrations of metals, for example, are 2 to 50 times greater than in mid-Chesapeake Bay. Zones of metals enrichment in Baltimore Harbor are associated with disrupted bottom communities. Bioassays of fish, invertebrates and bacteria indicate effluents have moderate to high toxicity. The greatest number of organic compounds detected per oyster and the highest concentrations were recorded off the James River and Baltimore Harbor.

1.1 What toxic chemicals are present and what is the concentration of them in the estuary?

Two classes of chemicals pose a threat to the Bay; 1) inorganic compounds, mainly trace metals like As, Cd, Cr, Cu, Hg, Sn and Zn; 2) organic compounds including pesticides, phthalate esters, polynuclear aromatic hydrocarbons (PAHs), polychlorinated hiphenyls (PCBs) and many other chlorinated hydrocarbon compounds. Many of these chemicals are produced naturally or synthetically. Approximately 300 organic compounds were found in the Bay's sediment, the majority of these compounds were PAHs.

The trace metals are found in several phases; 1) dissolved, and 2) solids, either sorbed to suspended sediment or bed sediment. Although concentrations may reach high values in biota, the bed sediments contain the greater mass and thus constitute the main toxic reservoir. Because sediments have a longer residence time in the Bay than water, bottom filter feeders like oysters are more exposed to contaminated sediment than water.

SUMMARY OF MEAN METAL CONCENTRATIONS IN BOTTOM SEDIMENT, SUSPENDED MATTER, AND DISSOLVED PHASES IN THE BAY

Metal	Bottom [1] Sediment ug/g	Suspended [2] Sediment ug/g	Dissolved[3] Water Column ug/l
As	3.9	13.0	
Cd	0.4	14.16	0.05
Co	12.8	--	0.07
Cr	28.9	---	0.17
Cu	21.6	127.96	0.66
Fe	24,250.0	3.11%	3.12
Hg	0.1	3.89	---
Mn	848.0	2.88	13.88
Mo	---	---	3.26
Ni	26.1	95.80	1.21
Pb	29.4	160.30	0.11
Sc	---	---	---
Sn	0.7	17.97	0.86
Th	---	---	---
U	---	---	0.93
Zn	157.0	0.75	1.19

1 - Means from combined Nichols and Helz (1981) data.

2 - From Nichols (1981)

3 - From Kingston (1981)

Summary of mean concentrations of various PAH organic compounds in Bay sediments listed on EPA's priority pollutant list.

Compound	Mean Concentration (ppm)
Phenanthrene	575
Pyrene	758
Floranthene	962
Benz (a) anthracene	310
Chrysene	448
Benzo (a) pyrene	440
Benzo (ghi) perylene	271

1.2 What are toxic chemicals associated with?

Most toxic materials tend to partition with sediment. Organic compounds and metals tend to partition to suspended material and then are deposited on the bottom as the suspended sediment is deposited. Because of polarity, some organics may be dissolved in the water column and exist below the detection limit of present day instrumentation.

1.3 Do toxic substances entering the system accumulate?

Most toxic chemicals of all classes entering the system accumulate in the sediment; others degrade, and some accumulate in the biota or are flushed out of the system. The degradation process occurs under changing physical/chemical conditions. Suspended sediment is particularly important in the accumulation of toxic materials, because metals are adsorbed, found and precipitated on suspended material. In this form they can be picked up by filter-feeding organisms or metabolized by plankton and reach high concentrations.

Fluid mud, dense suspensions of sediment, lies in fluid masses near the bottom of the Bay. It serves both as a reservoir for potentially toxic metals and as a medium for chemical transfer between the mud and overlying water.

Analysis of selected sediment cores demonstrate that Cu, Zn, Pb and Co increase dramatically near the sediment-water interface indicating that sediments are an important reservoir of metals and that the origin of these metals is man's activity.

1.4 Is the Bay regionally contaminated with trace metals?

Metal content of bed sediments from the main northern Bay is enriched 4 to 6 times in Mn, Pb and Zn compared to average shale. Sediment cores show an upward increase of more than two times. The distribution of enrichment factors in the main Bay is controlled by sediment type and deposition processes rather than nearness to sources of contamination.

Enrichment of suspended material in near-surface water of the central Bay in Cd, Cu, Ni, Pb, and Zn is related to high organic content. Enrichment exceeds natural concentrations of metals in oceanic plankton 9 to 19 times.

1.5 Is the Bay regionally contaminated with organic compounds?

Although concentrations are variable, some areas of the Bay have extremely high concentrations of toxic organic compounds. Approximate maximum concentrations of various organic compounds measured in Bay sediments are:

Compound	Max. Concentration (ppm)
Phenanthrene	100
Pyrene	150
Floranthene	200
Benz (a) anthracene	70
Chrysene	90
Benzo (a) pyrene	90
Benzo (ghi) perylene	70

With the magnitude of these concentrations, regional contamination is very obvious and at alarming levels in some areas.

1.6 Do levels of toxic chemicals found in the environment present a risk to the ecosystem?

Certain compounds including PAH's, PCB's, phthalate esters, DDT, As, Cd, Cr, Pb, Hg, Zn, may represent a risk to the ecosystem. However, to evaluate the risk associated with these chemicals is a complex problem. Each specific compound has a different effect on various species. Likewise, each species has a different reaction to specific compounds. To make the problem more complex, the synergistic effects and the stress which toxic material places on organisms are nearly impossible to quantify. For the most part, the observed dissolved metals concentrations do not exceed risk levels. For organic compounds, we have very little information on concentrations in the water column from which to make an evaluation.

Bioassays performed on specific sediment samples can indicate relative toxicity of the sediment. These tests indicate that the sediments in the Bay and several tributaries are generally more toxic than a west coast estuary. Also, an assessment of biological indices of the bottom biota in the Baltimore Harbor indicate that there are stressed and impacted conditions existing there.

2.0 What is the distribution of toxic chemicals in the Bay?

In suspended material, metal content per gram of As, Cd, Cu, Pb, Hg, Ni, Sn and Zn are maximal in near-surface water of the central Bay. These concentrations most likely are bioaccumulated by plankton. On the other hand, per liter of water, metal concentrations are highest in the northern Bay where suspended sediment concentrations are high - a zone called the turbidity maximum.

In bed sediment, metal content of Cr, Mn, Fe, Co, and Ni is highest in fine sediment of the northern Bay. Concentrations of most metals are maximal in the zone from the Susquehanna mouth to the Patapsco mouth where fine sediment is entrapped. Concentrations of Cr, Pb, and Zn are maximal in Baltimore Harbor and concentrations are not elevated in the main Bay off Baltimore. Concentrations of metals are relatively low throughout the southern Bay. Organic compounds are highest in fine, bed sediment from the Bay between the Susquehanna River and the Patapsco River. They generally decrease further seaward to the Potomac River; but in the southern Bay, locally high concentrations are found in sediment from estuary entrances.

The distribution of both metal and organic compounds is associated with the distribution of fine sediment and moderate to fast sedimentation.

2.1 What parts of the Bay are most susceptible to contamination?

The greatest enrichment may be expected in zones where: 1) the source supply is high and entrapment is good; 2) fine sediment accumulates; and 3) where rates of sedimentation are moderate to fast. Contamination of near source areas is common to tributaries near treatment plants and industrial facilities. Contamination that follows the fine sediment and fast rates of sedimentation is common to the main Bay, the zone of deep water in the central Bay. Sediment water content and fluid mud thicknesses are greatest in this region. This zone holds atmospheric contaminates as well as water-borne contaminates settled from overlying water or dispersed a great distance from their source.

Identifying locations of accumulation shows the distribution of fine grain sediments to which toxic chemical attach. Locations in the Bay accumulate sediments at variable rates from negative values because of erosion, to several m/century. In the upper Bay, fine grain sediment accumulates N to S generally down the Bay, especially between Baltimore and mouth of the Chester River. These accumulations are small, amounting to .5 to 3 m/century.

In the lower Bay, accumulation is again N to S in three main regions. The average rate is 0.5 m/century. The first region is in the deep channel down the stem of the Bay and where the channel flairs, just above the Rappahannock River. As much as 1.5 to 2 m/century accumulates at this location and sediment here is mostly silt/clay. The second region is just north of the York River; locally rates are as great as 2.5 m/century on the eastern flank of the Cape Charles deep opposite Old Plantation Flats. Sediment here is very fine sand. That same latitude, 37°20: shows similar accumulation on the western side of the channel.

2.2 What role do the biota play in the transport of toxic substances from the sediment to the water column?

Generally, benthic animals living in or on bottom sediments can reintroduce chemicals from the sediments to the water column. In addition, fish migrating to other parts of the Bay or Atlantic Ocean can transport chemicals with them. The main activities of benthic animals that can transport chemicals are:

- mixing - causing newly arrived surface material to be quickly buried or resurfacing older material.
- ventilation - increasing the exchange between interstitial water and the water column.
- increasing sediment stability - decreasing the probability that buried material will be resurfaced.
- decreasing sediment stability - increasing the probability that buried material will be resurfaced.
- causing rapid sedimentation - through pellitization of fine suspended particles.
- causing erosion - by making sediment more easily transported.
- bioaccumulation

2.3 What other processes (physical or chemical) exist which can cause remobilization of toxic chemicals into the water column?

Materials in the bottom sediment may be reintroduced to the surface environment and water column by two groups of processes.

Physical disturbance of the sediment can reintroduce toxic substances by storms, biologic activity (bioturbation), dredging and other engineering projects, propeller wash, harvesting of bottom organisms by dredging (e.g., clams, oysters).

Important chemical processes leading to remobilization might include diffusion driven by concentration differences, and life processes of benthic organisms such as irrigation of burrows and benthic feeding.

Physical disturbances are episodic occurrences whereas diffusion is a continuously operating process. Exhumation and resuspension of sediment by physical processes can re-expose material that had previously been buried and out of direct contact with the surface environment. Interstitial water, the water trapped in the voids between sediment particles as the sediment accumulates in the subaqueuous environment, is the vehicle through which chemical constituents in the sediment are continuously remobilized and transported within the sediment and across the sediment-water interface.

3. What are the sources and loadings of the pollutants of concern?

3.1 What is the direct contribution of toxic material from point sources?

	Metric tons per year				
	Cr	Cd	Pb	Cu	Zn
Municipal Wastewater	200	6	68	99	284
Industrial Discharge	199	178	155	190	167

3.2 What is the direct contribution from nonpoint sources?

	Metric tons per year				
	Cr	Cd	Pb	Cu	Zn
Shore Erosion	83	1	28	29	96
Atmosphere	189	99	582	95	?

3.3 What are the loadings from the major tributaries?

	Metric tons per year				
	Cr	Cd	Pb	Cu	Zn
Urban Runoff	1				
Rivers	100	4	180	220	1500

3.5 Are there other sources of toxic substances?

The massive reservoir of materials contained in the bottom sediments of estuaries have largely been ignored as a potential source of nutrients and trace elements until recent years. On the basis of interstitial water chemistry investigations, it is apparent that there is a very substantial contribution of these substances from the sediment to the water column. By far, the largest source of Pb is the atmosphere, and the largest source of Cd is industrial effluents.

MANAGEMENT QUESTIONS AND ANSWERS - SUBMERGED AQUATIC VEGETATION

1. Is there a problem in Chesapeake Bay related to SAV?

Yes, because SAV is declining, and because it has an important ecological role and economic value.

1.1. Are the current distribution and abundance of SAV unusually low ?

Yes, probably lower than every recorded in the Bay's history.

1.1.1. What is the current distribution and abundance of SAV in Chesapeake Bay?

About 16,000 hectares, or 5 percent of the portion of the Bay less than two meters deep is vegetated by SAV. (Sediment type and exposure to winds and currents make much of this shallow area unsuitable for SAV.) Most SAV is concentrated in four regions of the Bay: (1) the middle stretch of Maryland's Eastern Shore, including the Chester River, Eastern Bay, and the Choptank River, (2) the shoals between Smith and Tangier Islands, (3) behind sand bars along Virginia's Eastern Shore, (4) around the mouth of the York River from Mobjack Bay to Back River.

1.1.2. Have the distribution and abundance of SAV recently declined?

Yes, dramatic declines have occurred since the 1960's. In limited sampling between 1967 and 1969 along Maryland's Eastern shore from near the head of the Bay to Pocomoke Sound, most areas had 70 to 100 percent of their sampling stations vegetated by SAV. Only one area had less than a third of its stations vegetated. An annual summer survey by the U.S. Fish and Wildlife Service and Maryland's Department of Natural Resources shows that only 28.5 percent of their sampling stations in Maryland was vegetated in 1971, and only 10.5 percent was vegetated in 1973. Smaller fluctuations have occurred since 1973, and the percentage of vegetated stations now stands at an all-time low of eight. Archival aerial photography of six locations in the lower Bay reveals that five of them experienced declines since 1960s ranging from 45 to more than 99 percent.

1.1.3. Have all areas and species experienced declines at the same time and to the same degree? Have the declines been gradual, or sudden events occurring between periods of relative stability?

All areas and species have been affected, but not to the same degree, nor at precisely the same time. The areas mentioned in 1.1 as currently having most of the SAV have been the least affected. The head of the Bay, Maryland's lower Eastern Shore from Taylors Island to Pocomoke Sound, and the major Western Shore Rivers have been the most affected. Overall, during the last 15 years, declines have been a combination of sudden drops superimposed on an uneven but continuing downward trend. The Potomac and Patuxent Rivers experienced large declines between 1965 and 1970. In 1907, the Potomac had dense beds of SAV along both shores, but by 1970, only scattered pockets of vegetation remained. Large declines along Maryland's Eastern Shore occurred between 1969 and 1971. Further big declines occurred in the upper Bay in 1972, the year of tropical storm Agnes. In the Susquehanna Flats during the early 1960's, European milfoil displaced native species to a great extent. When milfoil declined in the mid-1960's, the native species recovered about two-thirds of their former abundance before decreasing slightly in the late 1960's. In 1972, there was a dramatic decrease in SAV abundance. Virginia's Eastern Shore had major declines between 1972 and 1974.

1.1.4. Have deeper areas been affected more than shallower areas, thus implicating turbidity as a cause of decline?

There is not enough evidence to say conclusively that deeper areas have been affected more than shallower areas, but limited evidence from archival photography suggests that this may be the case, at least in some areas.

1.1.5. Does the biostratigraphic record indicate that a decline as severe as the one of the last decade ever occurred before, or that cyclic changes have occurred?

No, limited evidence from the Susquehanna Flats reveals a continuous seed record until the top of the core. The seedless layer at the top corresponds to the time since tropical storm Agnes. There is no evidence of cycles in SAV abundance.

1.1.6. Has the recent decline of SAV in Chesapeake Bay been paralleled by declines in estuarine and marine ecosystems in other parts of the world, especially along the Atlantic coast of North America?

Declines that have occurred around the world have been near population centers. Localized declines, especially in Florida, have occurred along the Atlantic coast of North America, but generally the extensive declines in Chesapeake Bay stand in marked contrast to trends along the rest of the Atlantic coast.

1.2. Does SAV have a significant ecological role and economic value?

Yes.

1.2.1. Is SAV a direct or indirect source of food for animals, including economically important species?

Before 1960, SAV constituted more than half the food of at least six species of waterfowl (canvasbacks, ring-necked ducks, redheads, American wigeon, gadwalls, and whistling swans). Canvasbacks were an especially important species, attracting many hunters to Chesapeake Bay. With the decline in SAV, whistling swans and canvasbacks have switched to other foods, while redheads and wigeon have found other wintering areas. SAV also contributes to the detritus-based food web.

1.2.2. Does SAV provide habitat, especially for economically important species?

SAV beds currently support two to five times more finfish and invertebrates than nearby bare areas. SAV beds in Virginia are important nurseries for blue crabs. In Chesapeake Bay, in contrast to other regions, there is insufficient evidence to support the idea that SAV beds are nurseries for commercially important finfish; however, there is good evidence that numerous fish of ecological, but not economic importance occur in SAV beds.

1.2.3. Does SAV play an important role in nutrient dynamics?

SAV may act as a nutrient buffer, potentially taking up large quantities of nutrients during the spring growth period. In comparison to algae, SAV releases nutrients more slowly, and exerts a lower oxygen demand during decomposition after autumn die-back. CBP research has demonstrated the ability of SAV to rapidly take up nutrients from the water column, as well as from sediments.

1.2.4. Does SAV play an important role in sediment dynamics?

SAV roots and rhizomes can stabilize sediments, and SAV shoots can slow water currents and dissipate waves, thus allowing suspended material to settle to the bottom. CBP research at sites in Eastern Bay and the Choptank River has documented that suspended sediments are removed from water moving into SAV beds.

2. If there is a problem regarding SAV, what caused it?

Different combinations of factors were probably important in different localities.

2.1. Have herbicides been a factor in the decline of SAV?

They have probably not been the major Bay-wide factor. Extensive research on atrazine and linuron indicate that these pesticides may have been a contributing cause of decline of SAV already stressed by other factors, but this would be true only for SAV beds near sites of herbicide application, and in years when precipitation occurred soon after application.

2.1.1. What effects do herbicides have on SAV, and at what concentrations are these effects produced?

Atrazine and linuron concentrations of 50-100 ppb consistently cause significant reductions in photosynthesis in several species of SAV. Five to 10 ppb sometimes produce harmful effects. One ppm can kill SAV. Sublethal effects can last several days after exposure times of one to a few hours. Generally, full recovery occurs after exposures of less than

100-500 ppb. Experiments have not been done on toxicity of degradation products to SAV, but for agricultural weeds, degradation products of atrazine are far less toxic than the parent compound.

2.1.2. How do herbicides enter SAV?

They are taken up from the water column through the leaves. Root uptake can also occur, but is probably much less important because herbicide availability in the sediment is low.

2.1.3. To what amounts of herbicides is SAV exposed, and for how long?

Observed high concentrations of atrazine were 4 ppb in the mainstem of the Bay, 7 ppb in the primary tributaries, 20 ppb in secondary bays and coves, and 100 ppb in drainage creeks adjacent to agricultural fields. Exposure concentrations declined from these highs to about 20 ppb in a few hours in drainage creeks, to about 7 ppb in a few days in secondary bays and coves, to about 4 ppb in a few weeks in the primary tributaries, and to near zero ppb in a few weeks in the mainstem of the Bay.

2.1.4. What physical and chemical processes are involved in the transfer of herbicides from agricultural fields to SAV? What degradation rates and sorption constants do herbicides have?

Atrazine applied to agricultural fields can adsorb to sediment particles and colloidal material, or dissolve in water. Sorption coefficients for colloids are about 10 times higher than those for sediments, and sorption to sediments is about 10 times greater than solubility in water. However, over 90 percent of the atrazine in estuaries is in the unfilterable component of the water column. Herbicides are transported to the estuary mainly by runoff, although transport by subsurface drainage is also possible. Half lives of atrazine due to degradation are a few days to a few weeks in estuarine water, a month or more in estuarine sediments, and up to a year in agricultural soils.

2.2. Has the decline of SAV been caused by inadequate light reaching SAV leaves?

Inadequate light may be the most important proximate cause of SAV decline.

2.2.1. How does SAV respond to different amounts of photosynthetically active radiation (PAR)?

As the amount of PAR increases, net photosynthesis increases to a maximum. At this point net photosynthesis is light saturated. Above this point, SAV may become inhibited by too much lihgt. Below the saturation point, there is a compensation point at which gross photosynthesis equals respiration, and net photosynthesis is zero. Community compensation points are on the order of 200-300 microeinsteins per square meter per second.

These rates can vary considerably depending on periphyton density and other factors. Compensation points for individual species are on the order of 30 to 50 microeinsteins per square meter per second. Maximum rates of net daytime photosynthesis are on the order of 1.1 to 1.3 mg C $g^{-1}hr^{-1}$. Upper Bay species are generally not light saturated (i.e., they are light limited), and that their photosynthetic efficiency does not change seasonally. In the lower Bay, Zostera marina is light limited during both its spring and fall growing seasons, and appears to undergo acclimitization.

2.2.2. How do light and herbicides act together to affect SAV photosynthesis?

Although other research indicates that herbicides have a diminished relative effect at lower light levels, CBP research does not convincingly support such a conclusion.

2.2.3. What is the quantity and spectral distribution of light at different depths in SAV beds, bare areas, and areas that recently have lost their vegetation, and how do they vary seasonally?

Light penetration is greatest in the green and least in the blue region of the spectrum. Studies in a limited region of the lower Bay indicated no significant difference between spectral distributions in bare and vegetated areas. The attenuation coeffiecient for PAR ranged from 0.5 m^{-1} to 1.6 m^{-1}, and increased significantly from April to July at most sites. No clear pattern of difference occurred between vegetated and nearby bare areas in the lower Bay. In the upper Bay, attenuation was usually less in SAV beds than in bare areas.

2.2.4. What are the sources of turbidity, and what is their relative importance?

Suspended sediments and phytoplankton are the major contributors to turbidity. Their relative importance varies seasonally and between localities.

2.3. Has the decline in SAV been caused by changes in nutrient levels in the Bay?

Nutrient enrichment, through its stimulation of phytoplankton and periphyton, is a factor controlling SAV, and may have contributed to its decline.

2.3.1. To what levels of nitrogen is SAV exposed, and how do they vary seasonally?

Nitrate concentrations in the water column range from near zero to 100 micromolar. Nitrite concentrations range from near zero to 2 micromolar. Ammonium concentrations range from near zero to 20 micromolar. Nitrate concentrations are highest in spring and decline to a low in summer. In the upper Bay, interstitial concentrations of ammonium ion in the rooting zone (down to 15 to 20 cm) are about 80 micromolar.

2.3.2. How do nitrogen levels indirectly affect SAV?

Nutrient enrichment can stimulate the growth of phytoplankton, which can contribute to attenuation of light in the water column. Phytoplankton can also stimulate the growth of filter feeding animals that live attached to SAV leaves. These filter feeders can form a crust that blocks light and depresses photosynthesis. Nutrient enrichment can also stimulate epiphytic algae, which can block light. Epiphytic algae may also be controlled by animals that graze on the surface of SAV leaves. One such grazer that is found in Virginia, the snail *Diastoma*, has been shown under experimental conditions to dramatically decrease the density of periphyton on *Zostera*. The western shore population of *Diastoma* may have been virtually eliminated by the low salinities resulting from flooding at the time of tropical storm Agnes in 1972. The loss of *Diastoma* may be an important cause of SAV decline in certain localities along Virginia's western shore.

2.4. In summary, what are the most likely principal causes of SAV decline during the last 20 years?

SAV can be stressed by many factors whose relative importance can vary spatially, seasonally, and yearly. Some of these stresses include light attenuation in the water column caused by suspended sediment and phytoplankton, light attenuation by periphyton, herbicides, unusually high salinities, physical damage by storms, eating by whistling swans, uprooting by cownose rays, and biotic interactions that are not fully understood. Underlying factors may control one or more of these stresses. For instance, nutrient enrichment can stimulate both phytoplankton and periphyton. These multiple stresses, and the complex time-space patterns they can exhibit, must be considered against the background of the history of SAV distribution and abundance in the Bay. Historically, Chesapeake Bay probably had much more SAV than now. In 1907, extensive beds of SAV occurred along the length of the Potomac River estuary. It is reasonable to expect that the same was true of other parts of the Bay. Precipitous declines have occurred throughout most of the Bay since 1969, but not all species or areas have been equally affected. Disease cannot, by itself, explain the declines because it probably would not affect all species equally. The pattern of decline does not support the idea that point sources of pollution are the single cause of decline. The biostratigraphic record does not support the concept of entrained cycles in SAV populations, and tropical storm Agnes, although probably an important factor, is not the single cause of decline, again because the pattern of decline is not consistent with such an hypothesis. Interestingly, there is a positive correlation between SAV decline and potential diffuse loading (the ratio of the drainage area of a river to the river's volume). These facts suggest that a Bay-wide decline can be demonstrated. In particular, herbicides, although potential stresses, are not the sole cause of decline. Nutrient enrichment, and its effects of light attenuation, may be the most important contributing causes.

2.5. What are the minimum requirements for SAV growth?

Because factors may interact in a complex way, the minimum requirement for one factor depends on current levels of other factors. The following levels represent very rough approximations that cannot be well substantiated by current information. Light: above 200-300 uE $m^{-2}s^{-1}$ measured in the water column of SAV beds. Herbicides: below 5 ppb measured in water column.

PART II

NUTRIENT ENRICHMENT

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INTRODUCTION

The nutrients portion of this synthesis report presents the integrated findings of the Nutrients Program of the Chesapeake Bay Program (CBP). More than 10 individual research projects (listed in Appendix A), funded under the CBP, contributed to the three chapters of this part. Additional literature, other data bases, and many individuals also contributed valuable information for completing the synthesis of our knowledge of nutrient enrichment in Chesapeake Bay.

The CBP studied nutrients, because the natural process of nutrient enrichment, or eutrophication, is being hastened by anthropogenic (or human-related) contributions of primarily nitrogen and phosphorus compounds. Though needed by Bay organisms to grow, excesses of these nutrients can deteriorate the water quality.

Inorganic nitrogen and phosphorus compounds, such as nitrite, nitrate, ammonia, and phosphate, are referred to as "nutrients" because they are required by plants for growth. In an estuary like the Chesapeake Bay, nutrients support the growth of phytoplankton, submerged aquatic vegetation, and emergent marsh grasses. This plant material, in turn, supports the rest of the many organisms in the Bay.

When nutrients are introduced into an estuary in excessive amounts (nutrient enrichment) detrimental effects may result. Growth of phytoplankton may be stimulated, causing dense and unesthetic blooms. Or, a few species may dominate, resulting in declines of other types and loss of species diversity. Although phytoplankton blooms produce photosynthetic oxygen as they develop, as they die, respiration may exceed photosynthesis. Oxygen will be depleted from the water as a result. In addition, grazers and decomposers deplete oxygen by respiration as they process the phytoplankton. Consequently, oxygen depletion from the water is a common corollary to nutrient enrichment. The severity of these effects depends on season, rainfall, circulation, and the availability of phytoplankton seed stock.

The relationship between nutrient enrichment, phytoplankton growth, and oxygen depletion is fairly direct and well-documented. Less accepted are indirect relationships between nutrient enrichment and higher trophic levels, particularly commercial fisheries. Yet, in Chesapeake Bay, declines of important fisheries like striped bass, American shad, blue crab and oyster have been observed; it would be of great interest to know whether these declines have resulted from anthropogenic nutrient inputs. Although satisfying conceptual models can be developed in which nutrient enrichment, algal species composition, and competitive/predative fisheries interactions are related, data for calibration and verification are scarce. There appears to be a relationship between nutrient enrichment and another resource, submerged aquatic grasses. The value of Bay grasses and the relationship between their decline and nutrient enrichment will be discussed in Part IV on Submerged Aquatic Vegetation.

The Chesapeake Bay Program has assessed the nutrients problem in the Bay from three perspectives: analysis of historical trends, assessment of sources, and understanding of processes. With three separate but related approaches, the Program can determine the extent and nature of the nutrients problem and what should be done to alleviate it.

Analysis of historical trends in nutrient enrichment is presented in the first chapter of this part. Such an evaluation can help assess whether a problem exists because it can establish an historical baseline against which to compare present levels. Ideally, it is desirable to have a "pristine" baseline, nutrient levels existing before human settlement. However, it is obviously not possible to obtain such data. One must settle for the earliest period for which good data exists, and anecdotal data for earlier periods.

For Chesapeake Bay, the earliest large data base is that developed by CBI in 1949-51. This provides us a baseline for analysis of trends in the past 30 years. Within the Chesapeake Bay Program, these trends have been analyzed by Heinle et al. and will be discussed in the first chapter.

In assessing historical trends for a large system like Chesapeake Bay, it is important that a regional approach be taken. For example, trends in the Potomac River are quite different from those of the upper Bay. It is also important that nutrient levels be assessed on a seasonal basis, because nutrient processes are highly dependent on season (discussed more fully in the Processes section). Finally, fresh-water inflow must be accounted for, as this can greatly affect runoff rates and dilution (to be discussed under Sources).

Besides establishing a baseline, developing historical trends in nutrient enrichment provides a source of comparison with trends in resources like fisheries, submerged aquatic vegetation, etc. If declines in resources can be correlated with increases in nutrients, it is possible to begin investigating the causal relationships, if such exist, behind the correlations. Comparisons of historical trends are being investigated in the Bay Program's Characterization process.

The movements and transformations of nutrients in an estuary, called nutrient processes, are directly related to their potential negative effects. Understanding these processes is critical to developing appropriate nutrient controls. Nutrient processes vary in space and time (specific examples are discussed in the Processes section), and control strategies must account for regional and seasonal factors. Major nutrient processes include phytoplankton nutrient uptake, nutrient cycling, and circulation and are discussed in the second chapter of this part.

The primary negative effect of nutrient enrichment is overgrowth of phytoplankton. Whether phytoplankton growth occurs as a result of nutrient addition, and the extent of this growth, depend on whether the phytoplankton take up the nutrient and are able to grow. A number of factors affect phytoplankton uptake and growth. For example, all other growth requirements of the phytoplankton must be satisfied, such as temperature and light. In general, addition of a nutrient will stimulate growth only if that nutrient is limiting. Furthermore, some nutrients are "preferred" by phytoplankton over others and will be taken up first.

Uptake of nutrients by phytoplankton does not always result in growth. Under some conditions "luxury uptake" occurs, in which nutrient is taken up and stored within the cell. As a result, nutrient depletion from the water column may be observed without concomitant increase in phytoplankton biomass.

Finally, ambient nutrient levels do not always correlate with high phytoplankton biomass (as determined by chlorophyll *a* levels). Nutrients may be taken up by the cells so rapidly that they never accumulate in the water column. For this reason, measurement of ambient nutrient levels may not provide a good indication of eutrophication.

Nutrient cycling is the general term for the many biological, geological, and chemical processes by which nutrients change form. In the Chesapeake Bay, the most important of these are grazing and decomposition, nitrification and denitrification, and phosphate binding in the sediments.

Grazing of phytoplankton by predators is important because it prevents accumulation of phytoplankton biomass and may increase productivity of higher trophic levels. Thus, it can prevent the negative effects of nutrient enrichment. Because of grazing, high nutrient levels may not lead to high levels of chlorophyll a. Whether grazing occurs depends in part on the availability of grazers and on the palatability of the phytoplankton (e.g., blue-green algae are generally inedible). Decomposition of dead phytoplankton, animals and other organic matter by bacteria and fungi converts nutrients from their organic to inorganic forms, making them again available for phytoplankton and other plant uptake. It is an important part of the eutrophication process, because the respiration required depletes oxygen from the water column.

Nitrification and denitrification are bacterial transformations of inorganic nitrogen forms. Nitrification is the conversion of ammonia to nitrite and thence to nitrate. These conversions require the presence of oxygen; thus, under conditions of oxygen depletion, ammonia and/or nitrite may accumulate. Denitrification, the conversion of nitrate to nitrite and thence to nitrogen gas, occurs under anerobic conditions and may be an important mechanism for ridding the system of excess nitrogen.

The availability of phosphate in the water column depends in part on processes in the sediments. Under aerobic conditions, phosphate complexes with iron and manganese and precipitates to the sediments. Under anerobic conditions, however, phosphate is released from the sediments into the water column. Clearly, the cycling of nitrogen and phosphorus compounds is dependent on oxygenation, particularly of bottom waters and sediments. As a result, nutrient activities will be very different in the winter, when water is well oxygenated, than in the summer when oxygen depletion of bottom waters occurs.

Circulation is a critical component of nutrient processes. It determines the spatial distribution of nutrients, as well as that of the phytoplankton that would utilize them. It also affects the availability of oxygen to the bottom waters, through the processes of stratification, mixing, and turnover. Circulation is discussed in Processes.

In addition to understanding processes, assessing nutrient sources is critical to developing effective controls. Sources of nutrients to Chesapeake Bay include municipal sewage effluents and industrial nutrient effluents (point sources), as well as agricultural, urban and other land runoff (nonpoint sources), and atmospheric sources (precipitation). The third chapter of this part addresses these sources.

Assessment of nutrient sources is generally accomplished by a combination of monitoring and modeling. Point sources are routinely monitored; modeling is used when projections of future point source loads are made, or when the number of point source effluents is too great for frequent routine monitoring, and expected loads must be calcuated.

Nonpoint source nutrient loads are much more difficult to quantify. An estimate can be made by monitoring nutrient levels in the major tributaries entering the Bay (e.g., Susquehanna, Potomac, James Rivers). However, this gives no indication of nutrient loads from specific land uses within the watershed. That must be determined by monitoring nutrient levels in small tributaries draining single land use types, and extrapolating to the entire Bay watershed. Such nonpoint source modeling is an important technique in understanding which land uses are the greatest sources of nutrients, and where controls would be most effective.

The following chapters were prepared by CBP project investigators and staff to describe findings of the Nutrients Program and their implications. Nutrient Problems (author: Christopher D'Elia) discusses historical trends in nutrient enrichment and its effects. In Nutrient Processes, Jay Taft discusses the major nutrient processes and how these events affect the outcome and management of nutrient enrichment. Finally, James Smullen and Joe Macknis discuss the relative importance of nutrient sources. A final chapter relates these findings to implications for managing nutrient contributions to the Bay.

The chapters are organized around a set of management questions:

1. Is there a problem with nutrient levels in Chesapeake Bay?
2. What are the important processes interacting to create the problem?
3. What are the sources, loadings, and losses of the pollutants of concern?

A more detailed list of the questions can be found at the end of the Nutrients part.

Technical Glossary

aerobic: Environmental condition characterized by presence of oxygen.

albedo: Relation between amount of light sent back from a dark or unpolished surface and the amount falling on it, measuring its power of reflection.

allocthonous: Material coming from outside; not produced internally.

anaerobic: Environmental condition lacking oxygen.

autocthonous: Originating in location where found, e.g., Bay phytoplankton vs. river plankton washed into the Bay.

autotrophic: (Of plant) building up its food from simple chemical substances, not using or not dependent on ready-made plant substances, living or dead.

bioassay: The measuring of power of substances by their effects on organisms, e.g., toxic power of a heavy metal or organic pesticide.

biomass: The total mass or amount of living organisms in a particular area or volume.

biostratigraphy: Method used by geologists to analyze layers and fossil remains.

brackish: Somewhat salty, as the waters of some marshes near the sea or waters near the head of the Bay.

chironomids: Midges, a class of mosquito-like insects; typically refers to their larvae found in fresh to brackish Bay sediments.

coprophagy: The act of taking excrement as food.

electronic planimeter: Instrument for measuring the areas of plane curved forms.

epiphytes: Plant fixed to another plant but not dependent on it for food, using the host plant primarily as a substrate.

etiolation: Condition of a plant which is feeble and without normal green color through not getting enough light.

euphotic zone: The upper zone of a sea or lake into which sufficient light can penetrate for active photosynthesis to take place.

eutrophication: Natural or artificial addition of nutrients to bodies of water that results in increased plant biomass and typically low levels of dissolved oxygen during advanced stages.

Fickian diffusion: Diffusion of a substance through a unit area at a rate dependent upon concentration differences over a defined distance.

Gelbstoff: "Yellow substance" found dissolved in seawater, believed derived from decomposition products of plants, especially carbohydrates, in the presence of amino acids to form humic materials.

ground truth surveys: Technique to verify photographic interpretations.

Hill reaction: Part of photosynthesis involving light reactions within the chloroplast; fundamentally, splitting of a water molecule resulting in the evolution of oxygen through action of light on plant chloroplast. First stage in photosynthesis named after discoverer.

isopod: Crustacean without a hard cover, having a body commonly flat and made up of six or more divisions with legs used for walking, and eyes with fixed or no stems. Typically small in length (5 - 20 mm), living on and in sediments.

littoral zone: That part of the edge of the sea between high- and low-water mark or a little further out, as the living place of certain sorts of animals and plants.

meristic: Involving variation in number or geometrical relation of body parts, e.g., a variation in flower petals.

oligochaetes: Animal without a clearly marked head and with only a small number of chaetae on every body division, hermaphrodite, and living in earth or inland water, for example, the earthworm.

phytoplankton: Plants, most of which are very small, living in the water of seas, rivers, etc., chiefly near the top, and moving freely with it but having little or no power of swimming.

plastoquinone: Lipoidal compound localized in subcellular organelles and functioning as coenzymes in electron transport.

polychaetes: Animals having a great number of stiff hairs, a well-marked head with special outgrowths. Sea animals in which the sexes are seaparate and the uniting of sex cells takes place outside the body.

post-veliger larvae: Characteristic ciliated larvae whose free-swimming existence has changed to one of settlement to the bottom and attachment to a firm surface.

regression analysis: Mathematical method of fitting an equation to data, usually expressed as the change in a y - variable (dependent) relative to unit change in an x - variable (independent).

spectral attenuation coefficient: A number multiplier that expresses the diminuation of part of the light spectrum as the light energy passes through water.

spectrophotometer: An instrument used for measuring the intensities of light of different wave-lengths in a spectrum.

substrate: That substance on which an enzyme has the power of acting.

topographic quadrangles: A section of a topographic map seven and a half by seven and a half minutes, at a 1:24,000 scale.

2-4D: 2,4, dichlorophenoxyacetic acid: synthetic compound used as a weed killer in agriculture.

TECHNICAL SYMBOLS

BOD	-- biological oxygen demand
C	-- carbon
CFSD	-- cubic feet per second daily
chl *a*	-- chlorophyll *a*
COD	-- carbon oxygen demand
d	-- day
DIP	-- dissolved inorganic phosphate
DN	-- dissolved nitrate
DP	-- dissolved phosphorus
h	-- hour
K_s	-- half saturation value
L	-- liter
m	-- meter
ug	-- microgram
ug atom	-- microgram atom
MGD	-- million gallons per day
ug/L	-- micrograms per liter
um	-- micrometer
N	-- nitrogen
NH_4	-- ammonium
$NH_{3,4}$	-- total ammonia nitrogen
NO_2	-- nitrite
NO_3	-- nitrate
$NO_{2,3}$	-- total nitrite plus total nitrate nitrogen
O_2	-- oxygen
P	-- phosphorus
PO_4	-- phosphate
POTWs	-- publicly owned treatment works
ppm	-- parts per million
ppt	-- parts per thousand
OP	-- orthophosphorous
Q	-- mean daily discharge
RQ	-- respiratory quotient
sec	-- second
SED	-- suspended sediment
TKN	-- total Kjeldahl nitrogen
TN	-- total nitrogen
TP	-- total phosphorus
V_{max}	-- maximum uptake velocity

CHAPTER 1

NUTRIENT ENRICHMENT OF CHESAPEAKE BAY: AN HISTORICAL PERSPECTIVE

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CONTENTS

FIGURES

FIGURES (Continued)

Figures marked with an asterisk (*) are originals presented in the report by Heinle et al. (1980).

TABLES

SECTION 1

INTRODUCTION

The following paper deals with historical changes in the nutrient enrichment of Chesapeake Bay and its tributaries. In the present context, "historical changes" refer to those changes that have occurred primarily in the last several decades during which we have data. "Nutrient enrichment" refers to the addition of nitrogen and phosphorus compounds to bodies of water, and in excess can lead to phytoplankton blooms, loss of oxygen, and changes in fisheries species composition. Each section of the report contains an important topic relative to nutrient enrichment and discussions of the following Chesapeake Bay Program (CBP) management questions:

- Where and how severe are nutrient enrichment problems in the Bay?
- What are the consequences of nutrient enrichment?
- What are the commonly used criteria for evaluating a water quality problem related to nutrient enrichment, and what are their advantages and disadvantages?
- What techniques can be used to evaluate or predict nutrient enrichment problems?
- What are the historical trends in nutrient enrichment?
- What addditional research needs to be done?

This paper draws heavily on a previous report to the EPA/CBP by Heinle, D'Elia, Taft, Wilson, Cole-Jones, Caplins, and Cronin (1980). That report, entitled "Historical Review of Water Quality and Climatic Data from Chesapeake Bay with Emphasis on Effects of Enrichment," should be consulted by readers interested in greater detail about historical changes in water quality as they relate to anthropogenic and natural causes.

OVERVIEW OF NUTRIENT ENRICHMENT IN CHESAPEAKE BAY

There is little doubt that there are nutrient enrichment problems in Chesapeake Bay and its tributaries; however, there is doubt as to how extensive the problems are, and how rapidly environmental degradation is occurring. Human population growth in the Chesapeake Bay area has resulted in increased nutrient loadings from point (sewage) and non-point (runoff) sources. These increased loadings have had their greatest effects in the tributaries nearest the centers of demographic development, such as the tidal freshwater portions of the Potomac River near Washington, DC, where point source loadings from municipal sewage treatment plants had noticeable effects early in this century (Cumming 1916, Cumming et al. 1916). Although earliest concerns focused on problems of human health and sanitation, it was nonetheless recognized that the input of untreated sewage to the Potomac caused oxygen depletion in receiving waters. Blue-green algal blooms were observed in the upper Potomac estuary as early as 1916 (Cumming et al. 1916). By the mid-1960's sewage inputs in the tidal freshwater portion of the Potomac sufficiently enriched the water with nutrients, and blue-green algae became a serious problem (Jaworski et al. 1971b).

Other tributaries of the Chesapeake also show signs of nutrient enrichment. The upper Bay near Baltimore, MD, and the upper James near Richmond, VA, are quite enriched. Also enriched, but to a lesser extent, are the York, Rappahannock, Patuxent, and Susquehanna Rivers. Of these moderately enriched tributaries, the greatest data base exists for the Patuxent River and estuary. This excellent data base extends back to the mid-1930's and is one of the older and more complete data bases for any estuary in the world. For that reason, and because the estuary seems to be undergoing continuing change (dissolved inorganic nutrient levels are rising, and transparency and deep water dissolved oxygen concentrations are decreasing), much of the following data analysis and discussion deals with the Patuxent River. Furthermore, changes occurring in the Patuxent River could also occur in the main Bay and other tributaries if enrichment in these areas increases. The Patuxent River can be seen as an analog of the main Bay and of other western shore tributaries (Klein, unpublished).

Background information on nutrient enrichment and its relationship to algal growth is provided below to help underscore why the problem of nutrient enrichment is complex and difficult to assess in light of data gaps in the historical record. This report avoids the use of the term "eutrophication" because its meaning can be ambiguous and unclear.

SOURCES OF NUTRIENTS

Nutrient inputs to estuaries come from "point" sources, such as sewage treatment plant effluents, and "nonpoint" or "diffuse" sources such as runoff from the land. Increases in loadings from both point and non-point sources have occurred in the Chesapeake Bay region. As population increased and urbanization occurred, particularly in the last two decades, sewage treatment plants were constructed. Nutrients that would otherwise have been applied over the land or contained in home septic systems were combined and discharged at points along the rivers. Sewage treatment plant construction was accelerated after the grant program established in the 1972 Federal Water Pollution Control Act (PL92-500) was adopted. As a result of the move toward centralized treatment, large increases in total amounts of nitrogen (N) and phosphorus (P) from human wastes discharged to the Chesapeake Bay system have occurred. Brush (1974) summarized the sewage discharges to the Bay in 1973, and EPA/CBP recently completed a revised inventory. Details of the CBP inventory of sewage discharges are found in the last chapter of this part.

In contrast to point sources that are solely attributable to human activities, diffuse sources may be natural, or result from human activities. Native, undisturbed ecosystems, such as forests, are natural nonpoint sources. Agricultural or urban runoff accounts for much of the anthropogenic diffuse loadings. The importance of nonpoint sources depends on season. For example, in the spring, loadings from nonpoint sources are by far the dominant source of nitrogen to the Bay system (Smullen et al. 1982). The CBP Modeling Study (Hartigan, unpublished) relates land use to nonpoint source loads. Sources of nutrients and historical changes in loadings are discussed in depth by Heinle et al. (1980) and in the last chapter of this part.

SECTION 2
CONSEQUENCES OF NUTRIENT ENRICHMENT

The task of enumerating the most important consequences of enrichment in estuarine systems is yet incomplete, because the consequences of nutrient enrichment of freshwater environments are much better understood than for brackish or saline ones. The theme of a recent symposium (Neilson and Cronin 1981) was the enrichment of estuaries with nutrients. Many of the papers in the symposium deal directly with Chesapeake Bay. For example, Webb (1981) formulated a conceptual model of an estuary's response to nutrient enrichment in his review paper. His conclusions state that small additions of nutrients increase overall production, with increased biomass showing up at any trophic level. Large increases produce changes in species composition at all trophic levels. Interested readers should consult Webb's review for further details.

The consequences of enrichment in estuaries are more difficult to assess than those in fresh waters, because estuaries are generally subject to more complex hydrodynamic processes. Also, the effects of salt on biological and chemical processes have no analogues in fresh waters. However, there appear to be certain consequences that are at least qualitatively similar for all water bodies that are nutrient enriched. Figure 1 presents a scheme of probable consequences. As shown, one consequence is that plant productivity is enhanced by higher concentrations of nutrients in the water. Levels of organic matter contained in the water column in turn often increase, although enhancement in the rates of other processes may counterbalance the increase to some extent. Organic matter produced in the water column may accumulate in deep water where its degradation results in an oxygen deficit that is not balanced by atmospheric input. The Chesapeake Bay is characteristically two-layered; there is a natural, seasonal isolation of deep water from potential atmospheric oxygen inputs. Oxygen consumed during the oxidation of extra organic matter produced by enrichment may not be replaced in the lower, isolated layer, resulting in an oxygen imbalance uncharacteristic to the natural system. Most estuarine organisms of direct interest to humans--fish and shellfish, for example, require oxygen; they will either swim away from uncharacteristically low oxygen water or will perish.

FATE OF ADDED NUTRIENTS

An aquatic system can respond to nutrient enrichment in a variety of ways. If one views such a system as a compartment or a series of compartments, as mathematical modelers often do, one can more easily conceive of the ways in which responses might occur. Figure 2 shows a simple compartmental representation of an estuarine system: a single compartment with a series of exchanges or fluxes across the compartment boundaries. The simplest approach to understanding the nutrient mass balance of such a system, is to measure the amount of a given nutrient within the compartment and estimate the fluxes and exchanges of that nutrient between the compartment and the outside. The amount of nutrient contained in the box after a given time interval is a function of the amount, or "standing stock," of nutrient in the box at the beginning of the interval, less the amount lost over the interval, plus the amount gained

INCREASED
NUTRIENT INPUTS

↓

INCREASED
NUTRIENTS
IN WATER COLUMN

↓

INCREASED
ALGAL GROWTH
IN WATER COLUMN

↓

DECREASED CLARITY AND
INCREASED PARTICULATE ORGANIC
LEVELS IN WATER COLUMN

↓

SETTLING OF PARTICULATE ORGANIC
MATERIAL TO DEEP WATER

↓

DECAY OF PARTICULATE ORGANIC
MATERIAL AND DECREASE IN
OXYGEN LEVELS
IN DEEP WATER

Figure 1. Scheme of possible effects of enrichment in a stratified water column.

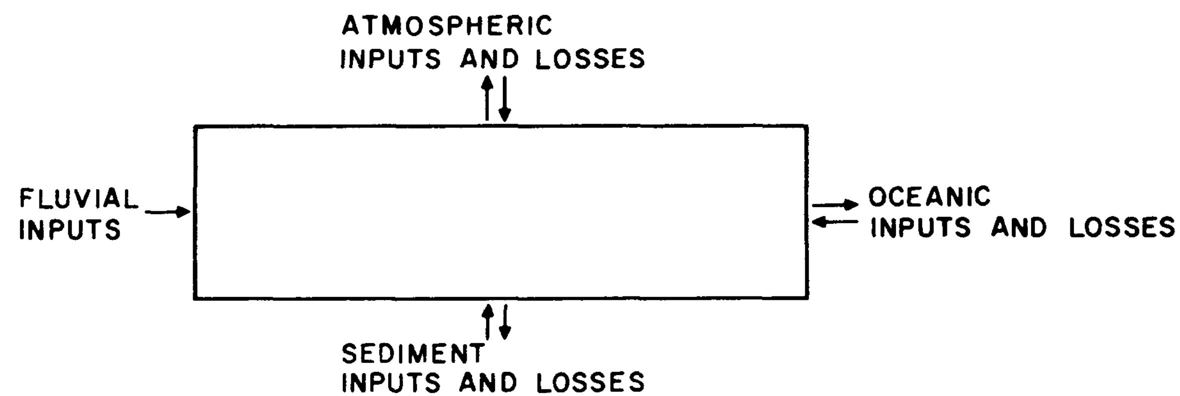

Figure 2. Simple one compartment box model of an estuary.

over the interval. Clearly this "black box" approach to understanding an estuary has a number of deficiencies. For example, it tells us nothing about the internal partitioning of specific nutrients of interest, or about the biological response in the estuary to a change in a nutrient input or loss. We know only whether the total amount of nutrient in the compartment changes.

Understanding internal nutrient partitioning is essential if we are to increase the complexity of our model to account for internal responses of an estuarine system to changes in loadings or losses. Only in the last few years was any attempt made to assess sediment-nutrient exchanges in the Chesapeake. It is now known that they are of appreciable importance. For example, during the summer, the sediments are the greatest source of phosphorus in most of the Bay system (Smullen et al. 1982). Until recently, very little emphasis was placed on collecting any information other than on internal compartmental nutrient concentrations. The focus on point-in-time measurements leaves the historical record grossly deficient in process-oriented measurements of fluxes, exchanges, and transformations. Although it is possible to infer from differences among point-in-time measurements that changes occurred in the compartment, it is difficult to attribute those changes to a specific cause, unless exchanges that were not measured are assumed to remain constant during the interval between measurements.

Chesapeake Bay bears little resemblance to the simple one- compartment system represented in Figure 2. An estuary by definition is a place where sea water and fresh water mix to produce a range of intermediate salinities. Provisions must be made in a model to account for this characteristic, and to understand how Chesapeake Bay might respond to continued increases in nutrient loadings. Model complexity increases greatly when one attempts to include provisions for time-varying phenomena such as intra- and interannual changes in loadings, losses, and hydrodynamics. Modelers dealing with the Bay and other estuarine systems have been struggling to determine what level of complexity is necessary to include in their models (e.g., Harleman 1977; O'Connor 1981).

SYSTEM RESPONSES TO INCREASED LOADS

What are the possible responses of the Chesapeake Bay system to increased nutrient loads? Figure 3 presents a chart showing the possible response of the water column to increased levels of nutrients. For simplicity, we can assume that the single compartment conceptual model given in Figure 2 represents the water column to which additional loadings are applied. Figure 3a expands the single compartment into subcompartments, reflecting partitioning at four levels. The partitioning scheme is given to show the major pools into which added nutrients must go, or pass through. Increased loading would manifest itself at level "i" as higher levels of nutrients in the water column, as enhanced rates of nutrient loss from the system, or as a combination of both. It is possible for the internal nutrient content of any compartment to remain constant over a period of years in the face of increased nutrient loadings, if increases in net losses from the compartments keep pace with increases in net inputs. There is no assurance that increased loadings will necessarily

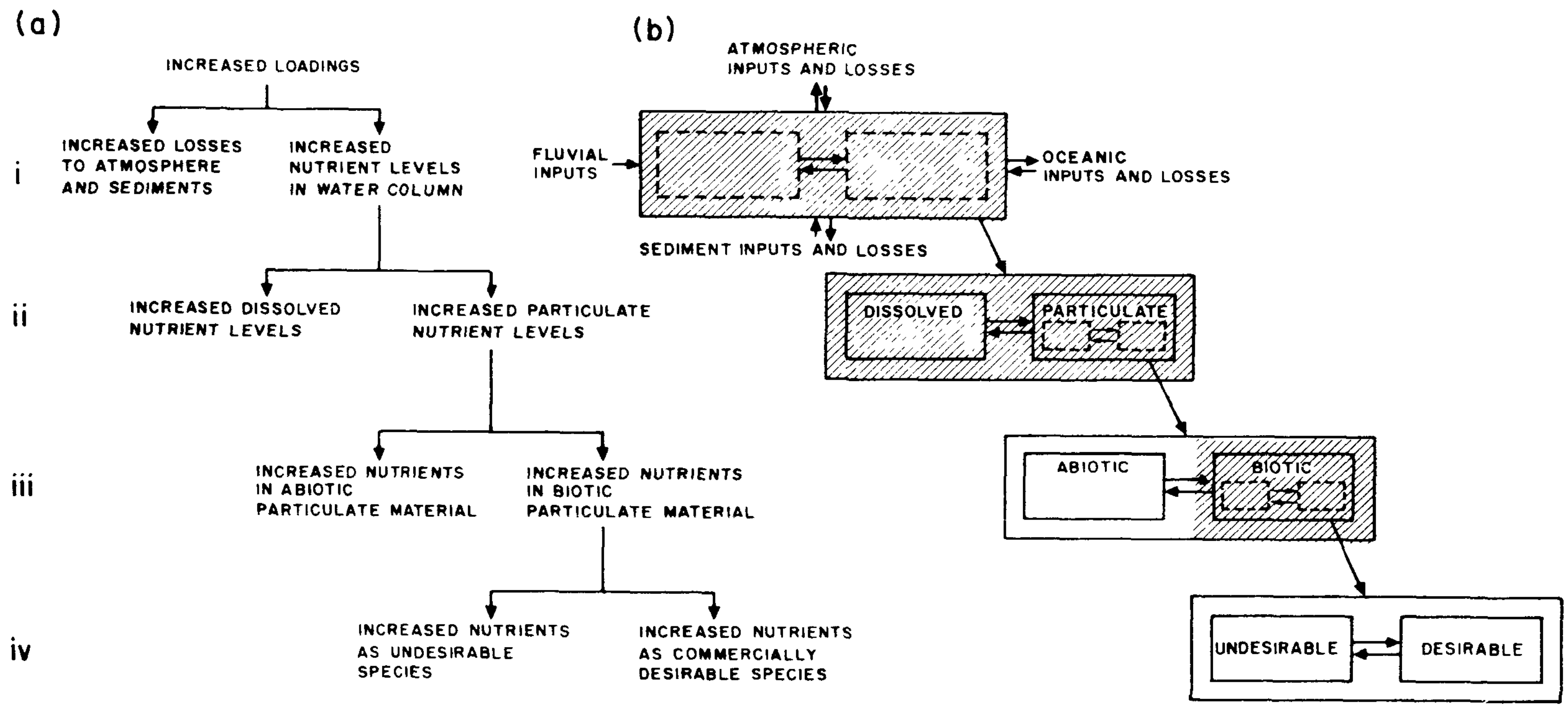

Figure 3. Binary dendogram showing possible responses of the water column to increased nutrient loadings.

be manifested in increases of the contents of any particular compartment. Although the scheme developed in Figure 3 is constructed as a binary dendrogram, an "either....or" situation is not necessarily implied, as added nutrients may result in either of the binary choices or in some intermediate of the two.

The historical record for Chesapeake Bay lacks information on a number of the pools shown in Figure 3, and on the specific transformation processes and rates affecting them. However, quite good catch records have been kept by local authorities on harvestable fish species, providing some information on partitioning of fisheries between commercially desirable and undesirable species (level iv). From these records, Heinle et al. (1980) present evidence that the partitioning to commercially less desirable and undesirable species has increased in recent years. Unfortunately, little is known, even now, about food chains leading to the production of desirable species. Factors such as climate can play an important role regulating the abundance of estuarine fish stocks.

To explain the changes in fisheries partitioning between desirable and undesirable species, it is necessary to examine the previous hierarchical level (level iii), the partitioning of added nutrients between biotic (here signifying "living") and abiotic particulate material. The historical record is poor on both the absolute quantities and the partitioning ratios of nutrients in particulate material. Fortunately, however, the historical record does include a considerable amount of information on transparency of the water as determined by Secchi disk. Transparency is affected by the amount of particulate matter in the water. This particulate matter is composed of inorganic material (clays, silts, etc.), non-living organic detrital material, and living material. There is convincing evidence that in certain places on the Bay, such as near the mouth of the Patuxent River, transparency as measured by Secchi disk has declined in the last 40 years. This suggests that the amount of organic material and, by inference, organically bound N and P in the water column, have increased. Turbidity derived from inorganic material may have increased also.

Increased biotic particulate material results from increased nutrient levels in the water column partitioned between the dissolved and particulate forms. Because there is no way to measure how much N and P are contained in living material relative to detrital material, it is of interest to examine the partitioning of nitrogen and phosphorus compounds (level ii). The total phosphorus in the water column is composed of dissolved inorganic phosphorus, dissolved organic phosphorus, and particulate phosphorus. Of these, the historical record contains substantial information on dissolved inorganic phosphorus only. Total nitrogen is composed of dissolved inorganic nitrogen (nitrate plus nitrite plus ammonium), dissolved organic nitrogen, and particulate nitrogen. Analytical techniques for the identification of all forms of nitrogen existed in the 1930's but were unreliable, especially for ammonium. Heinle et al. (1980) found no data on levels of particulate or dissolved organic nitrogen anywhere in the Bay prior to the 1950's. This represents an enormous gap in the data record for these forms of nitrogen. For this reason we have very little understanding of what the historical partitioning of dissolved versus particulate nutrients in the Bay was, and how this may have changed in response to increased loads.

Increased loading will manifest itself as higher levels of nutrients in the water column, as enhanced rates of nutrient loss from the system, or as a combination of both. Increased nutrient loadings to the water column of the Bay, not accompanied by increased nutrient losses to the atmosphere and sediments, will result in increased nutrient levels in the water column. It appears that, on an annual basis, sediments do not absorb more nutrients than they release (Smullen et al. 1982). We can assume that additional loadings will not result in equivalent additional losses, and that nutrients entering the water column will remain there and be manifested as a corresponding increase in dissolved nutrients, particulate nutrients, or some other compartment shown in Figure 3.

Although quantity and partitioning data can yield information about historical changes in the distributions and standing stocks of nutrients in the system, they yield little knowledge about the internal dynamics of the system that cause the changes. The next section addresses the internal dynamics briefly; for more information refer to the following chapter by Taft.

NUTRIENT ENRICHMENT AND ALGAL GROWTH

The addition of nutrients to an aquatic system frequently enhances algal "specific growth rates" (increase in biomass per unit biomass). "Nutrient sufficiency" occurs when algal specific growth rate is not stimulated by further nutrient addition; "nutrient limitation" occurs when algal specific growth rate is restricted by the availability of nutrients. "Algal productivity," that is, the rate at which new organic material is being produced per m^2 by plants, is a function of both specific growth rate and biomass. Systems can exhibit very high specific growth rates, and the algae can be nutrient-sufficient, although the productivity per unit area is low (systems can exhibit very high rates of productivity without producing nuisance levels of algal biomass). Implicit in this is that the biomass or standing stock of algae, although growing at a very fast rate, is low, resulting in a low level of production. In other words, what material is present is growing fast, but there is not very much of it present to grow. The converse is also true. A rather high level of production (that is, increase in algal biomass) occurs when large quantities of slow-growing algae are present. The situation is reminiscent of a bank account earning interest. The interest rate is analgous to the growth rate, and the principal is the biomass. The increase in principal per time is the analogue of productivity--the highest rate of increase in principal will occur when the interest rate and the principal are both high.

An important distinction to make is that between "net" and "gross" productivity. Gross primary productivity is the total rate of organic production by photosynthesis, irrespective of accompanying consumption of organic material by respiration. Net primary productivity is the rate of accumulation of organic material in excess of its consumption by respiration. In the bank account analogy, the principal in the account will only grow at its fastest rate when no withdrawals are made, and there are no bank charges. When the withdrawal rate or bank charges equal the interest rate, principal does not grow. Unlike bank accounts in which we want the highest possible increase in principal with time, in aquatic systems there comes a point at which the accumulation of organic matter becomes dangerously high. Nuisance levels of organic matter build up only when the rate of production of organic matter exceeds its rate of consumption. Standing stocks of algae can be held at continuously low levels and still exhibit high productivity if what is produced is consumed as quickly as it is produced.

Systems where nutrient enrichment problems are greatest are usually those in which the levels of production are greatest and out of balance with consumption. Implicit in this is that high levels of biomass accumulate, and what is produced is not removed quickly. Large accumulations of biomass--organic matter representing high biochemical oxygen demand (BOD)--are often responsible for oxygen depletion from the water column and other negative effects we associate with over-enrichment by nutrients.

Systems that exhibit high rates of productivity, but in which little organic material or biomass accumulate, also exhibit high rates of nutrient recycling or throughput. In such systems, N and P atoms resident in the systems may "turn over," or pass through organisms in a matter of hours. There are very few "new" atoms of N and P entering the system from outside. On the other hand, systems that quickly accumulate organic material or biomass, exhibit low rates of turnover and generally high rates of nutrient addition, without correspondingly high rates of removal. In its pristine state, Chesapeake Bay probably fell more into the category of a high productivity system in which standing stocks of organic material or biomass did not accumulate as much as they do now. Decreases in transparency as represented by Secchi depth probably signify the accumulation of organic matter, and it is this organic matter that can decay and use up oxygen to create a problem.

The important point to note from the above discussion is that increased loadings may result in greater rates of algal production or nutrient uptake without increasing standing stock of either; that is, the algae or additional nutrients are removed from the system as fast as they are produced or added. A truly adequate historical assessment of nutrient enrichment effects should assess both standing stocks and process-oriented, flux rate measurements. Unfortunately, in the present case, the historical record is heavily weighted toward measurements of individual parameters of standing stock, and it will not be possible to adequately consider the changing dynamics of the system. Therefore, enrichment- related changes in the system not observably affecting standing stocks will not be discernible.

SECTION 3

EVALUATING NUTRIENT ENRICHMENT PROBLEMS

INDICATORS OF NUTRIENT ENRICHMENT

The traditional approach to assessing enrichment in an estuary relies on both primary and secondary indicators of nutrient enrichment. "Primary indicators" of nutrient enrichment are typically the first indicators used in assessing enrichment. They are not necessarily the best indicators, but are the ones for which this historical record is most complete. "Secondary indicators" are those that have potential value in assessing enrichment, but are secondarily used. Some of the major primary indictors are nutrient concentration, O_2 concentration, Secchi depth, chlorophyll a, and algal species shift. Secondary indicators include measurements of dynamic processes such as primary productivity and nutrient flux rates, and other nutrient concentrations, pH, bacteria, BOD, and COD. These indicators will not be discussed in this section.

Primary Indicators

Nutrient Concentrations--

Virtually any water quality assessment program will include determination of nutrient concentrations in the system of interest. The most commonly measured nutrient forms are nitrate, nitrite, ammonium, and phosphate. They are of analytical interest, because they are the "fertilizer" nutrients most often responsible for the growth of aquatic plants. They also indicate the amount of N and P in the water column readily available to support algal growth.

Oxygen Concentrations--

Most water quality assessment programs also provide for dissolved oxygen determinations. Oxygen is probably the most crucial water quality parameter. Low oxygen tensions occur as the result of the oxidation of organic material without adequate physical means of oxygen resupply (that is, reaeration). Because commercially important species require oxygen, we are concerned with the effect of the accumulation of organic matter released from sewage outfalls, or produced by algae in response to nutrient enrichment on oxygen concentration. Fortunately, for analysis of trends, the historical record for oxygen concentrations in Chesapeake and its tributaries is good, particularly for the Patuxent River.

Secchi Depth--

The Secchi disk has been used for decades to measure the transparency of water bodies and to make inferences about levels of organic material and algae present in the water. Secchi depth (the depth to which the disk can be lowered and still be visible) is greatest in water of the greatest transparency. Secchi depth tends to be reliably determined from operator to operator, and the historical data record is quite good, although Secchi depth does not differentiate between turbidity from algae and other materials present such as suspended sediments, detritus, and other particulates. Also, Secchi measurements are rather imprecise in extremely turbid systems.

Chlorophyll *a* Concentrations--

Chlorophyll *a* is a reliable indicator of algal biomass and can give a general indication of the standing stock of phytoplankton present in the water column. The measurement of chlorophyll did not come into wide practice until the early 1960's, and since then, methods for measurement have evolved considerably. Measurement of chlorophyll levels over the next several decades will probably be more widely used in documenting changes than it has been over the last several decades.

Algal Species Shifts--

Many water quality studies have also involved collecting phytoplankton samples for identification. In fresh waters, under highly enriched conditions, the species composition often changes toward a dominance by blue-green algae. Such shifts have been observed in the upper Potomac River near Washington, DC, but are not generally observed in the saline waters of the Bay. It is not widely appreciated that marine and estuarine nutrient enrichment does not involve a shift in species composition toward blue-greens. Therefore, blue-green algae are not considered good indictors of nutrient pollution in saline systems.

A great difficulty encountered when attempting to examine the historical data record for shifts in phytoplankton-species composition, is the evolution of sampling and counting methods for phytoplankton. In the 1950's oceanographers began to appreciate that 35 to 50 um mesh (or greater) nets traditionally used to sample for phytoplankton in the ocean, were not catching the smaller-diameter algae responsible for the bulk of photosynthesis. Phytoplankton sampling on Chesapeake has been no exception. Early workers used nets and, therefore, their results do not include counts on important, smaller species. McCarthy et al. (1974) verified that the smaller phytoplankton on Chesapeake do indeed account for most of the primary productivity. Thus, comparison of phytoplanktonic-species composition with time must be done carefully.

TECHNIQUES FOR EVALUATING ENRICHMENT OF ESTUARIES

Evidence in Chesapeake Bay historical data base indicates that changes have occurred in nutrient concentrations, oxygen levels, and Secchi depths in parts of the Bay. These changes seem to have resulted from increased nutrient loadings in the last twenty years. One must understand that "historical" here refers to relatively recent history, that is, the last several decades for which we have data. Anthropogenic changes in the system may have occurred prior to collecting and recording of detailed data. Other kinds of ecological evidence, particularly on rates of production, consumption of organic matter, nutrient exchanges, and other factors, would also be useful in assessing enrichment effects. Such data, however, are obtained by relatively modern techniques and are difficult to compare because of inconsistent methodologies. This section contains a discussion of techniques developed previously for evaluating enrichment in fresh waters; these techniques evaluate the state of enrichment and predict changes in water quality in response to nutrient loadings.

Water Quality Indices

Managers, when faced with the responsibility of evaluating and improving the "water quality" of an estuary, often turn to water quality indices to assess the current water quality. What is an index? Thomas (1972) describes an index as a "composite value for an environmental component for which we have more than one indicator." Ott (1978) defines an index as any "mathematical approach which aggregates data on two or more water quality variables to produce a single number." Pikul et al. (1972) consider an index "a mathematical combination of two or more parameters which has utility in an interpretive sense." McErlean and Reed (1981) have reviewed the use and application of indices to estuaries, and have concluded that lack of success of transferring counterpart freshwater indices to estuaries is attributable to three reasons: (a) the lack of an exact and widely accepted definition of estuarine "eutrophication"; (b) a basic lack of knowledge of nutrient limitation and cycling in estuaries; and (c) possible fundamental differences between estuaries and other water bodies which invalidate transfer attempts. Other scientists have been critical of indices because they oversimplify complex ecological properties; they are biased in their formulation; and they do not clearly associate cause and effect between nutrient enrichment and response of plants and ecosystem level changes.

Two projects supported by the EPA/CBP were conducted to review the applicability of existing indices and to develop new water quality indices for the Bay. McErlean and Reed (1979) proposed the use of five indices in estuaries. Four were selected from the available literature, and one was developed by the authors. The four previously developed indices were the National Sanitation Foundation Index (NSFI) by Brown et al. (1970), the Minimum Operator (MO) or Water Pollution Index (Ott 1978), the Principal Nutrient Index (PNI) by Olinger et al. (1975), and the Beta Function Index (BFI) developed by the State of Illinois. McErlean and Reed's index is entitled "Estuarine Index of Enrichment" or EIE. The second project was that of Neilson (1981), who developed a use-oriented rather than a general index of enrichment. Table 1, as an example, presents the simple indicator criteria that constitute Neilson's index. Table 2 shows the use-related interpretations of indicator values that he has employed.

The use of indices in summarizing data from monitoring programs may serve to identify areas that are changing, or are in need of closer study. Indices may be of great value in the indentification of danger zones where close scrutiny by scientists and managers is required.

Water Quality Models

Another approach to dealing with environmental problems associated with excessive nutrient enrichment is to formulate and develop models that are mathematical constructs attempting to represent numerically some key features (for example, chlorophyll, oxygen concentration, nutrient concentration) of ecological systems affected by nutrient enrichment. EPA/CBP is making extensive use of such models.

TABLE 1. CLASSIFICATION SCHEME FOR NUTRIENT ENRICHMENT IN ESTUARIES (FROM NEILSON 1981)

Level of Nutrient Enrichment	Total Nitrogen mg/l	Total Phosphorus mg/l
0	0.003	0.0004
1	0.010	0.001
2	0.032	0.004
3	0.100	0.014
4	0.320	0.044
5	1.000	0.140
6	3.200	0.440
7	10.000	0.400
8	32.000	4.400
9	100.000	13.800
10	320.000	44.000

Numerical water quality models have proved to be successful in representing the operation of some sewage treatment plants and in representing rivers, streams, and lakes in which hydrodynamic factors are relatively simple and easy to simulate mathematically. Such models have typically been "steady-state," that is, those in which boundary conditions and inputs remain constant through a given model run, in contrast to time-varying or "real-time" models where such parameters are not held constant. In estuarine systems, the complexities of the non-steady-state hydrodynamics greatly complicate nutrient cycles and distributions, oxygen exchanges, and the growth and distribution of organisms (Harleman 1977, D'Elia et al. 1981). Mathematical modeling of estuaries is considerably more challenging. Below, some of the strengths and weaknesses of water quality models as tools of the scientist and manager are briefly reviewed.

Since estuaries are complex time-varying systems in the hydrodynamic sense, models may be constructed for different pollutants yet contain similar hydrodynamic representation. However, factors not related to hydrodynamics but that affect pollutant chemical specification and transformation, will probably be pollutant-specific and, thus, require different modeling formulation. Virtually any water-quality, numerical model must be designed with the system and pollutants of interest in mind.

As in the case of water quality indices, data gaps can be problematical with numerical water quality models. Standard procedure in developing such models involves calibration and verification. Once a numerical model is formulated it is "fine tuned" with a set of environmental data, so that an appropriate set of inputs will reproduce a set of data actually collected in the environment. At that point, it is calibrated. The model is next verified by seeing if it can reproduce another set of "real" data collected under different conditions. Data collected must be appropriate to provide for rigorous calibration and verification. Time and space intervals used in obtaining data for these processes must reflect the scales that the model is designed to resolve, and the model, in turn, should be designed to

TABLE 2. NEILSON'S (1981) CHART SHOWING IMPACTS OF NUTRIENT ENRICHMENT ON WATER USES

Level of Nutrient Enrichment	Public Drinking Water Supply	Livestock Drinking	Irrigation	Freshwater Aquatic Life	Marine Aquatic Life	Recreation and Aesthetics	Industry	Commercial Shipping
0	Acceptable	A	Acceptable	Acceptable (oligotrophic)			A C	
1		C C			Acceptable (oligotrophic)	Acceptable	C	A C
2	Minor Purification Required	E P	Increased Nutrient Levels Could Enhance Usefulness	(mesotrophic) Problems Arise Periodically			E P	C E
3	More Extensive Treatment Needed	T A			Problems arise Infrequently or due to Local Conditions	Infrequent Episodes when not Acceptable	T A	P T
4		B L		(eutrophic) Marginally Acceptable	(mesotrophic) Marginally Acceptable		B L	A B
5		E			(eutrophic) Marginally Acceptable	Frequent Episodes when not Acceptable	E	L E
6	Marginally Acceptable and Sometimes not Acceptable	Algae May Clog Intake Pipes	Generally Acceptable But Algae Could Clog Pipes and Pumps and There Could be Nitrate Build-up in Ground Water				Algae May Clog Intake Pipes	
7				Generally not Acceptable				
8	Not Acceptable	Marginally Acceptable			Generally Not Acceptable	Not Acceptable		Problems May Arise
9		Not Acceptable					Not Acceptable for some Purposes	Hydrogen Sulfide
10								

reflect time and space scales of importance in nature.

Mathematical models often intimidate non-mathematicians who, therefore, often find it difficult to evaluate the utility of models as management tools. However, models' ability to predict or project water quality conditions is not their only role in aiding managers. Clearly, mathematical models are of special benefit in developing conceptual formulations of nutrient enrichment responses and in identifying where additional research and data collection are needed (cf. O'Connor et al. 1981).

Other Techniques

Other methods for evaluating the current state of nutrient enrichment that have been less intensively utilized in the CBP. Two are presently being incorporated in ongoing and incomplete studies of the Bay.

Considerable effort has been paid, in particular, to developing an assessment methodology for determining available forms of phosphorus in fresh waters. Relatively simple, statistical models have been developed to relate phosphorus loading, hydraulic residence times, and algal biomass in a number of lakes. Leaders in this area of endeavor have included Vollenweider (1976), Schindler (1977), and Lee et al. (1978). Such an approach would be difficult to accomplish for Chesapeake Bay, because both phosphorus and nitrogen seem to play roles as limiting nutrients at different seasons and in different places, and because loading levels are not adequately quantified. However, Lee and Jones (1981) have developed a preliminary statistical model applicable to Chesapeake Bay.

Bay area scientists have also devoted some attention to the use of salinity-dilution diagrams for nutrients. This method may help identify localities of abundance and depletion of nutrients (Boynton and Kemp, unpublished; Taft, unpublished; D'Elia, unpublished; Webb, unpublished). The idea behind this approach is simple: when nutrients are supplied primarily in freshwater inputs and diluted down-estuary by saline waters low in nutrients, the concentration of a given nutrient in the water column will be in proportion to the salinity, unless there are sinks or sources of nutrients along the way. The statistical modeling of nutrient-loading responses, and the diagraming of salinity-dilution relationships will probably receive much greater attention in future evaluation of nutrient enrichment of Chesapeake and its tributaries.

SECTION 4

HISTORICAL TRENDS IN NUTRIENT ENRICHMENT

The review of historical trends in nutrient enrichment of Chesapeake Bay (Heinle et al. 1980) concluded that nutrient enrichment problems were greatest in the low salinity areas (less than 8 to 12 ppt), where summer chlorophyll a concentrations often reach or exceed 60 ug L^{-1}. Such areas are those where the tributaries pass through the more populous areas; prior to population growth, chlorophyll levels in those areas may have rarely exceeded 20 ug chlorophyll a L^{-1}. Climatic and other natural factors strongly affect ecological expression of nutrient enrichment. This is now of concern, particularly in the main stem of the Bay where relatively unenriched seawater dilutes the nutrient content of the enriched river water.

TREND EVALUATION

Separating human-induced changes from natural cycles is often the crux in both scientific assessment of the state of the Bay and management decisions in preserving (or improving) Bay environmental quality. The obvious importance and weight of these determinations lead scientists and managers to examine the ability to determine accurately a trend or change in the presence of noise or large variation. Without resorting to the formation of signal theory the problem is: can we be assured that a trend or change we observe over time is not simply part of a natural cycle of change, whose period is considerably longer than our viewing time? One solution to this uncertainty is to observe the Bay over a time much longer than the longest period of expected variablility. The difficulty here, however, is that natural cycles of climate and runoff can vary over periods greater than 10 years (Table 3). If time-series analysts were strict and required many cycles for accurate determination, then they would demand records of observations that were longer than all but a few available from the Chesapeake Bay system. Table 3 lists some of the cycles that are expected to affect the Bay's ecosystem. The shorter-period cycles (with variation on the order of one year or less) can serve as guides for the design of observational programs that ensure that the record length will encompass the variability. The longer-period cycles offer a test of a record's ability to separate trend from cycle.

One rule of thumb for time-series analysis is that a record should comprise on the order of 10 cycles for proper resolution. If, for instance, the Bay ecosystem responded to the six-year cycle in rainfall, then a 60-year time series would be desirable. Few natural systems have been observed with even simple measures for such a long time.

The situation is not hopeless, however, for scientists and managers who are forced to assess trends or changes on the basis of time-series with much shorter lengths. This assessment can often be made with acceptable certainty if additional information, such as cause and effect, is considered. For comparatively simple relationships, such as the effect of runoff on estuarine salinity, the separation between trend and cycle can be achieved despite shorter record length. A numerical model predicting

TABLE 3. EXAMPLES OF NATURAL CYCLES AFFECTING CHESAPEAKE BAY'S ECOSYSTEM

Cycle Period	Type of Cycle
12 to 42 hours	Semidiurnal tide
24 hours	Diurnal light cycle, sea breeze, etc.
4 to 8 days	Passage of low-pressure systems
14 days	Spring-neap tidal range progression
1 month	Monthly tidal variation
1 year	Seasonal climatic cycle
6 years	Climatic (rainfall, runoff)
11 to 12 years	Climatic (rainfall, runoff)
20 years	Climatic (sunspot activity cycle, rainfall, runoff)

salinities from runoff data would provide the necessary additional information here. For relationships that are derivative and not direct, or that depend on multiple causes having cycles of differing periods, separations between trends and cycles are difficult, even if long records are available. Time-series analysis techniques can help refine the statements on variability, but they cannot provide information that is not on the record itself. In spite of these warnings and difficulties, the history of an indicator of Bay environmetal quality is the necessary starting place for an assessment of change.

TRENDS BY REGION

For purposes of contrast and comparison in the ensuing discussion, the Bay is divided into four geographical regions (Figure 4). These regions are: (1) the upper Bay and western shore tributaries, characterized by the highest fluvial inputs; (2) the middle Chesapeake Bay; (3) the eastern shore tributaries, characterized by low fluvial and sewage but high agricultural nonpoint source inputs; and (4) the southern Chesapeake Bay. The geographical regions reflect, in a general sense, the segmentation approach to the Bay adopted by the EPA/CBP. For example, the main Bay and western shore tributaries can be considered to be analagous (Klein, unpublished). However, the EPA/CBP segmentation scheme is more detailed, allowing for close examination of individual portions of the Bay and for modeling purposes. The EPA/CBP segmentation approach, therefore, provides for more resolution than is necessary for purposes of this paper. Readers who wish to learn in greater detail about historical changes in specific localities should consult Heinle et al. (1980).

Upper Bay and Western Shore Tributaries

This region has been most severely affected by anthropogenic, nutrient enrichment. The enrichment problem is greatest in the summer when water-residence times, light availability, and temperatures are also

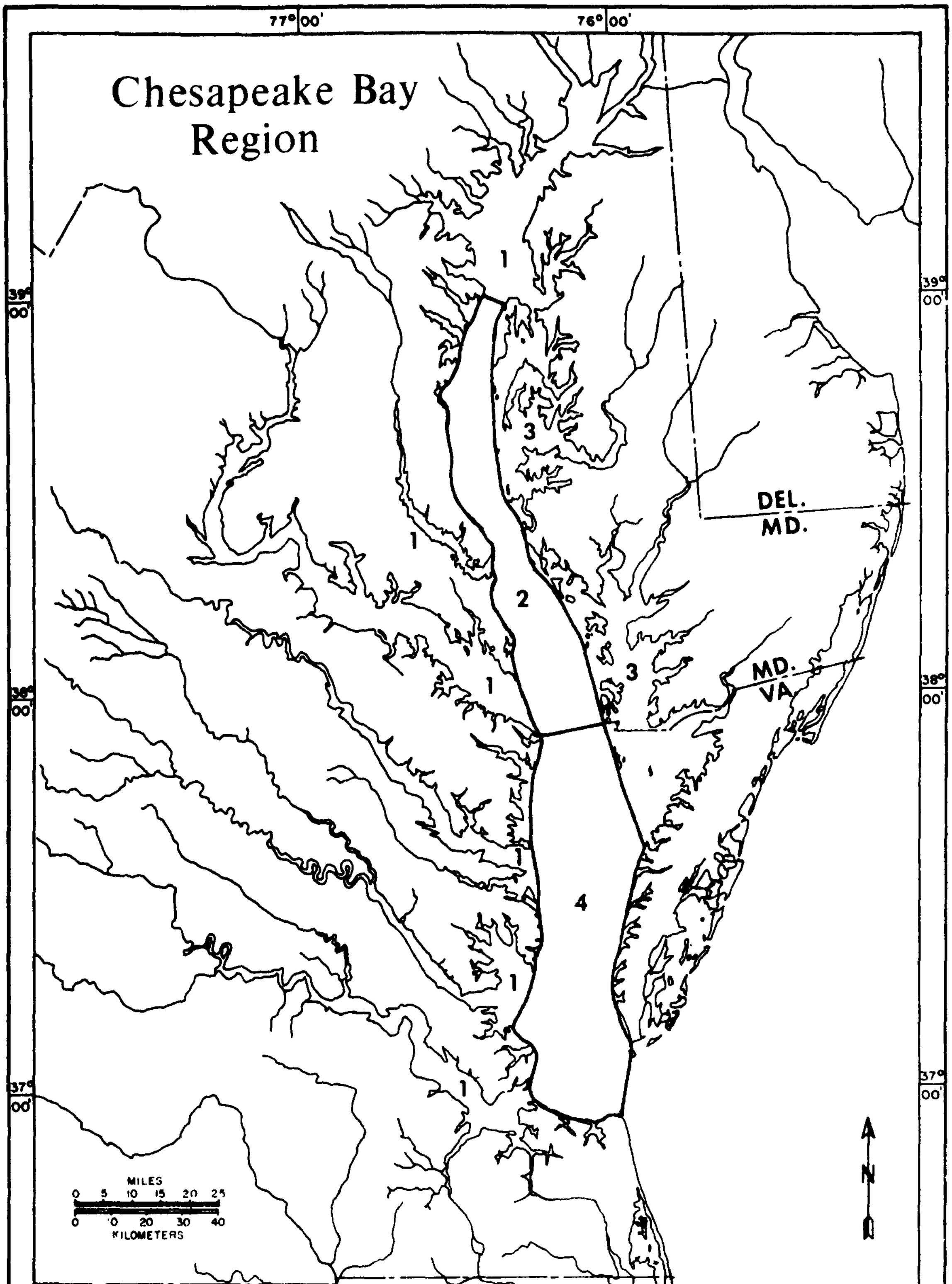

Figure 4. Regions of Chesapeake Bay.

greatest. Historical evidence for enrichment-related effects is most substantial in this region, with long-term trends clearly distinguishable from short-term variations. The seasonality of the nutrient cycle is very evident and quite complex. Nutrient inputs through the tributaries are greatest during the high-flow period of the year, typically in March through May. These inputs are characterized by high N:P ratios; that is, N is in excess of P relative to the ratio normally required by phytoplankton (about 16 atoms of N per atom of P). The amount of N from fluvial sources, during this period, is high relative to the amount of N coming into the system from point sources--sewage treatment plants. High flow, nonpoint source fluvial inputs are also highly oxidized. In other words, nitrate is the primary form in which the N is found. There is some evidence, especially for the nitrate input at high flow from the Susquehanna, the largest volume tributary to the Bay, that much of this nitrate passes through the Bay unassimilated, because of short residence times of this nitrate relative to the seasonally slow uptake rates of the plankton for nitrate (Taft 1982). A similar condition may exist in other tributaries, and it is important to scale the importance of this N in annual input budgets that have, as their goal, the development of input ratios for steady-state mathematical models.

In the summer, when river flows decrease, point-source inputs to the tributaries become the predominant input-source of new N and P to the system. The N:P ratio of point-source inputs is much lower; however, regeneration of N and P under oxic conditions from the stored reserves in the sediments (in effect, a nonpoint source to the water column) may counterbalance this to some extent. Chlorophyll levels in the water column increase in response to greater hydraulic detention times and higher algal growth rates. Oxygen concentrations in the water column are high in the daytime when algal photosynthesis is high and are low at night when planktonic respiration is not counterbalanced by photosynthetic oxygen production that cannot occur without light. Fortunately, dissolved oxygen levels in upstream waters rarely get critically low, because the water column is typically shallow and unstratified and can easily mix and reaerate.

Upper Bay--

Early data from the upper Bay exhibited a pattern of maximum dissolved inorganic phosphate (DIP) concentrations in the spring and fall with minimal concentrations in the winter, and especially in the summer; more recent data suggest that relatively uniform concentrations exist all year (Figure 5). For example, in 1949-1951 and 1964-1966, values in June, July, and August did not exceed 0.645 ug atoms L^{-1}. In contrast, values in 1969-1971 for those months exceeded 1 ug atom L^{-1}. The upper Bay differs from the western shore tributaries that apparently can reach much higher levels of DIP. Nonetheless, the upper Bay appears to show some increase in annual DIP abundance.

The concentration of nitrate plus nitrite-N, the only parameter for which we have reliable data back to the early 1950's, does not appear to have changed in the upper Bay (Figure 6). For example, in March, the period of greatest influx, values in 1964 to 1966 ranged from 20 to over 100 ug atoms L^{-1}. From 1969 to 1971, March values ranged from 28 to 83 ug atoms L^{-1}. The seasonal pattern is typical for most of the Bay, with

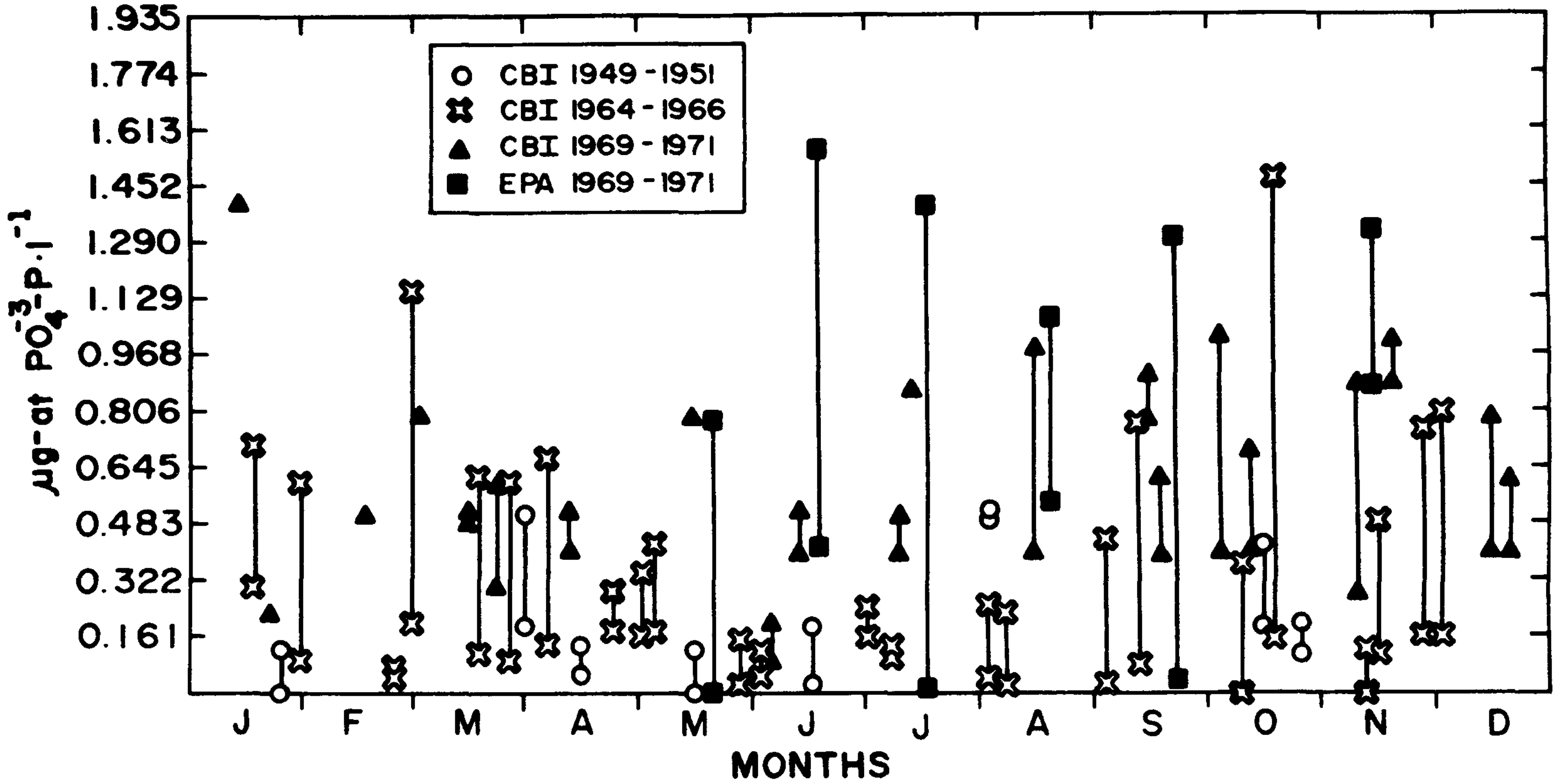

Figure 5. Ranges of concentrations of orthophosphate-P observed during studies of the upper Chesapeake Bay (Heinle et al. 1980).

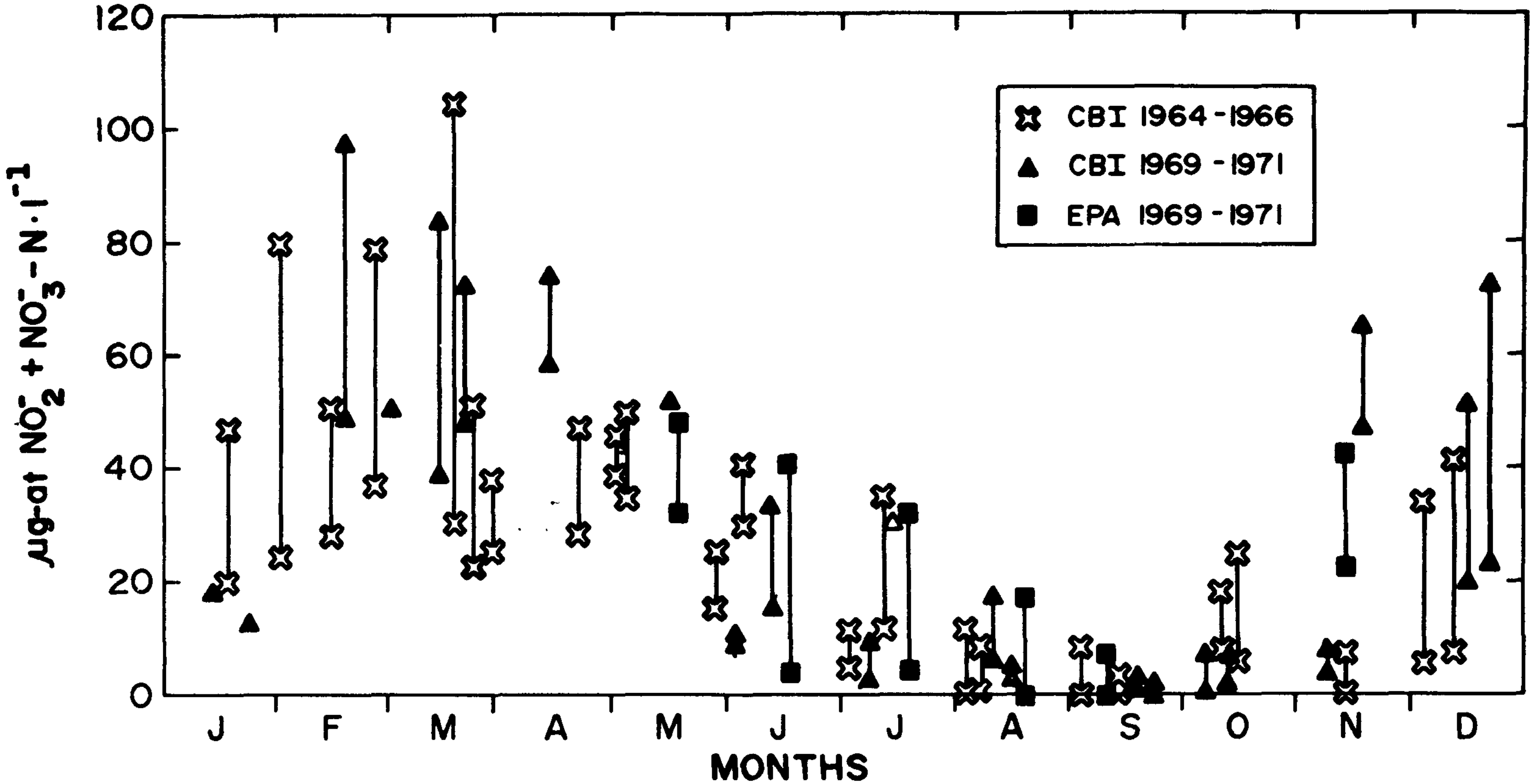

Figure 6. Ranges of concentrations of nitrite plus nitrite-N observed during studies of the upper Chesapeake Bay.

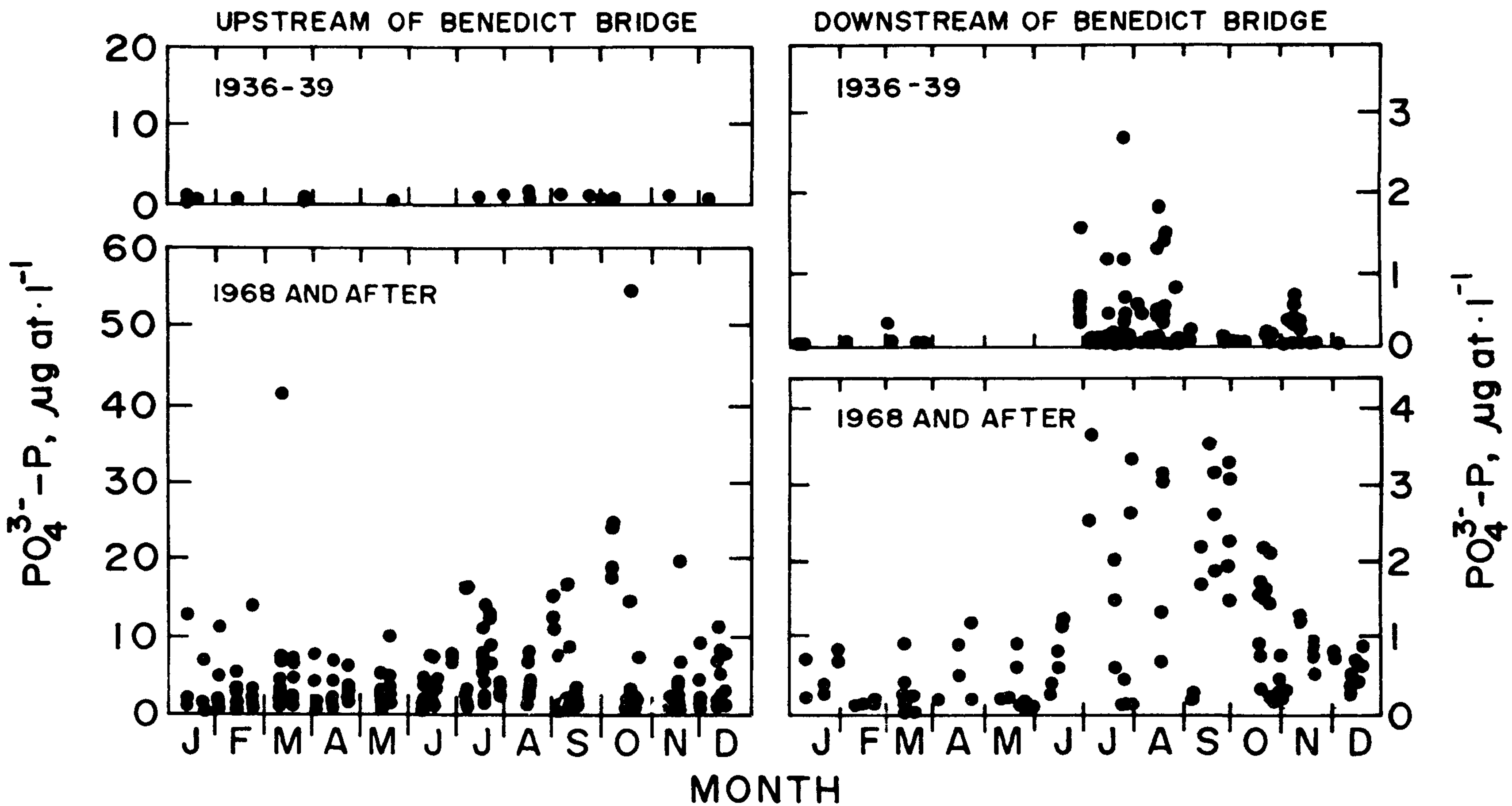

Figure 7. Concentrations of orthophosphate-P in surface waters of the Patuxent River upstream and downstream of Benedict Bridge versus time of year.

the nitrate maximum occurring during the high-flow period, and the nitrate minimum during the low-flow period. The upper Bay is so dominated by the flow of the Chesapeake's most important tributary, the Susquehanna River, that it is not surprising that nitrate availability in the water column strongly reflects nitrate input to the upper Bay by that river. Additional information about the dominance of the Susquehanna is presented by Smullen et al. (1982).

The effect of enrichment on chlorophyll levels in the upper Bay is unclear. Evidence for increased concentrations of chlorophyll in the upper Bay is inconclusive based on the Heinle et al. (1980) historical data base, whereas the data presented by Salas and Thomann (1978) appear to indicate conclusively that an increase in chlorophyll occurred. Although concentrations of chlorophyll in the early CBI data never exceeded 10 ug L^{-1}, no measurements of chlorophyll a were made during August and September, the months in which annual chlorophyll maxima are often achieved. Productivity may have increased in response to nutrient enrichment without an accompanying increase in plant biomass, providing that the turnover of plant material increased accordingly.

As in the Patuxent River where the issue of what nutrient, if any, limits productivity is complex, the issue of what limits phytoplankton production in the upper Bay is also complex. Salas and Thomann (1978) and Jaworski (1981) concluded that P limitation predominates, but Clark et al. 1973 concluded that N limits phytoplankton growth. Without a complete understanding of dissolved inorganic nitrogen inputs (other N forms such as ammonium must be taken into account also) and without knowing the seasonal breakdown on N:P input ratios, the question of whether N or P limits productivity is very difficult to assess (Taft 1982).

Patuxent River--

The Patuxent River has an excellent historical record, and it appears that it provides an analog of the main Bay and western shore tributaries. (However, correlations evaluating the relationship are being made in the CBP's characterization analysis). The Patuxent has been increasingly enriched in recent years; detrimental effects of this enrichment could be expected to occur in analogous segments of the Bay system if they were equally enriched.

The Patuxent River shows a somewhat different pattern in DIP abundance than does the upper Bay. Figure 7 shows the rather striking historical changes that have occurred in DIP concentrations in surface waters there, probably in response to increased point source loadings. Maximum concentrations of DIP have clearly increased upstream of the Benedict Bridge, where salinities are typically less than nine ppt. Downstream of the bridge, where salinities range from about eight to 18 ppt, surface DIP concentrations are significantly lower than those observed upstream (note scale change between panels in Figure 7), presumably as a result of the dilution of phosphate-rich fresh water by less enriched saline water. There appears to have been an increase in DIP levels since the 1930's in this region of the river also. This increase is most pronounced in the summer. Such a summer phosphate maximum is characteristic of Chesapeake Bay and other estuaries (Taft and Taylor 1967a, 1967b); it may result from surfacing of water rich in phosphate, produced by enhanced rates of benthic regeneration at higher summer temperatures, and by increased phosphate solubility at lower oxygen concentrations below the halocline. Because we do not know the effect of enrichment on total phosphorus levels, we cannot rule out a change in partitioning of water-column-total P resulting in higher DIP levels.

On the basis of the 1968-to-present data set, phosphorus limitation seems unlikely anywhere on the Patuxent River throughout most of the season when severe oxygen deficits occur (late spring through fall). Light limitation seems more probable (O'Connor et al. 1981). Phosphorus limitation may have been present prior to the late 1960's, when P loadings from sewage treatment plants were considerably lower, but this is not unquestionable.

The concentrations of nitrate plus nitrite-N in the Patuxent exhibit the same seasonal cycle of abundance that has been reported for the upper Bay; moreover, there appears to have been an increase in nitrate content of the water since the late 1930's (Figure 8). Most of the increase appears to have occurred later than 1965, coinciding with the beginning of extensive development of the Patuxent River basin. The source of this nitrate is probably nonpoint; most sewage treatment plants are not discharging fully oxidized effluents--most inorganic N is usually in the ammonium, not nitrate form. As for DIP, less nitrate is found in the water south of Benedict Bridge, reflecting the dilution of nutrient-rich fresh water by less enriched saline water.

The historical record does not include adequate data on ammonium. This is unfortunate, because ammonium is taken up preferentially by phytoplankton relative to most other N forms. We know from previous work (Boynton et al. 1980) and work in progress at CBL, that the regeneration of ammonium by the Patuxent riverbed occurs at some of the highest rates ever recorded anywhere. This regenerated ammonium can drive internal recycling processes in the absence of added nutrients (Nixon 1981). We know also that this ammonium accumulates below the halocline and diffuses across that boundary often at rates lower than those at which it is removed from the water column above. Boynton (personal communication) does not consider the sediment nitrogen reserves to be adequate for more than a few weeks' supply of regenerated ammonium, and there appears to be rapid settlement and mineralization of nitrogen on the benthos. Rapid recycling of nitrogen occurs between the water column and the riverbed. A productive system could be maintained for some time in the absence of added nutrients; the effects of nutrient controls might not be immediately apparent.

Although the analytical procedure for determination of nitrite has remained essentially the same over the last 50 years, relatively little attention has been paid to its measurement. This is because it rarely achieves significant concentrations in the water column. A number of investigators have observed periodic accumulations of nitrite in Chesapeake Bay waters (McCarthy et al. 1977; Webb and D'Elia 1980; Academy of Natural Sciences of Philadelphia, unpublished). This nitrite accumulation occurs in the late summer and early fall and is probably a consequence of ammonium oxidation (nitrification, step one). In a synoptic sampling program conducted in the fall of 1981, Taft et al. (unpublished) observed elevated nitrite concentrations throughout the Bay. In the Patuxent River, levels exceeded 15 uM nitrite-N. It is unknown whether this phenomenon occurred historically, but Webb (1981) suggests that the magnitude of the nitrite accumulation is a function of the degree of nutrient enrichment of the Bay

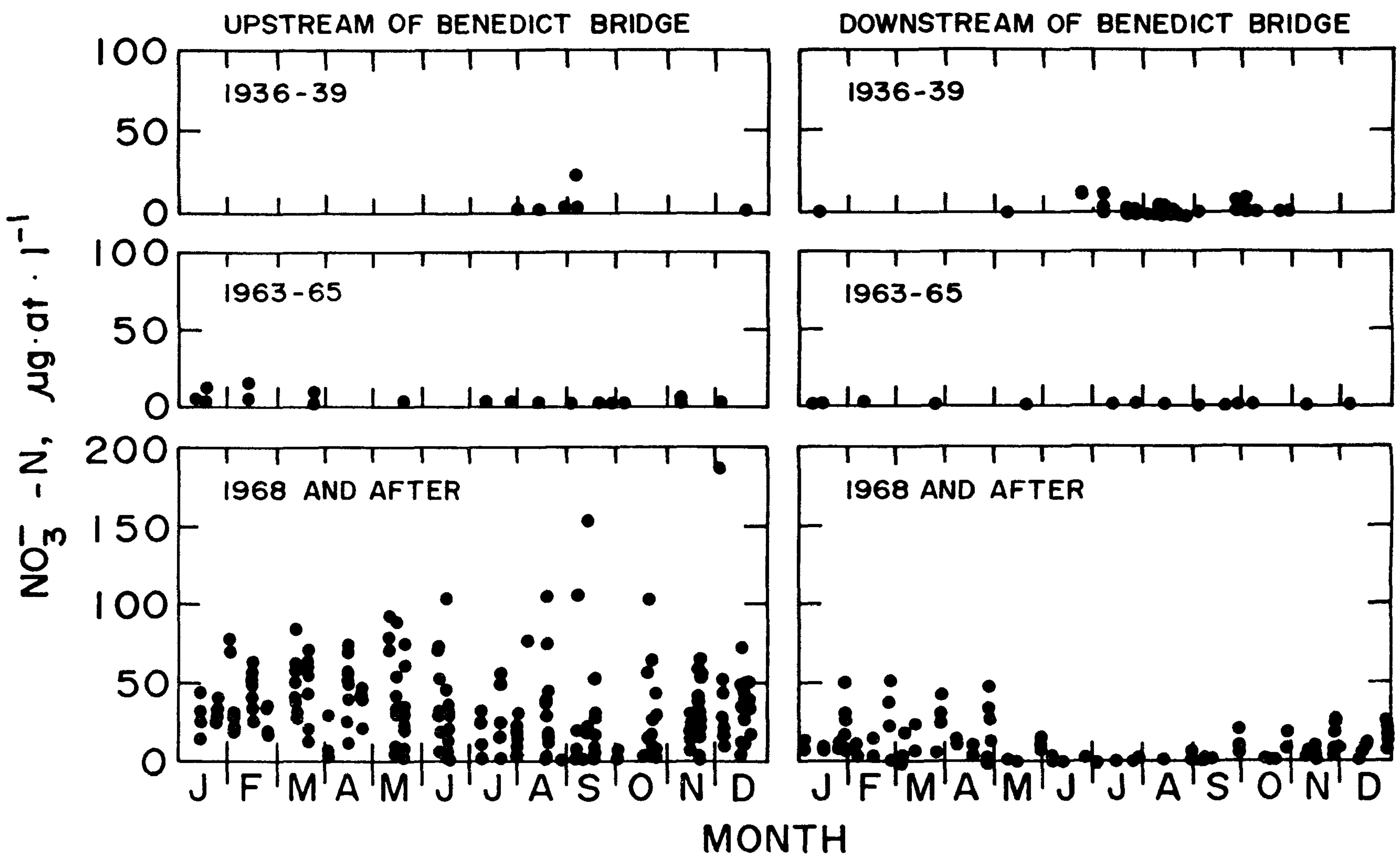

Figure 8. Concentrations of nitrate in surface waters of the Patuxent River upstream and downstream of Benedict Bridge versus time of year (Heinle et al. 1980).

during the summer, and he recommends continued monitoring of the nitrite maximum.

There is strong evidence that nutrient enrichment stimulated increased phytoplankton production and an accumulation of plant biomass in the lower Patuxent River. This conclusion is based mainly on Secchi-depth data rather than chlorophyll-concentration data, because the historical data base for chlorophyll on the Patuxent is less complete (but suggests the same trends). Figure 9 shows Secchi data from July, 1937 to July, 1978, normalized against surface salinity (to account for variations in river flow). The inability of the Secchi disk to resolve differences in transparency when transparency is low, means that little can be said about historical changes at surface salinities below about eight ppt. Such low salinity regimes are also subject to high levels of because of turbidity because of inorganic sediment. However, at the greater transparencies found at higher salinities, the resolving power of the Secchi disk is good, and inorganic sediment loads, particularly at lower flow times of the year such as July, are less appreciable.

Transparencies of the water in the lower estuary during 1963 were similar to those observed during 1936 to 1940 (Figure 9). Heinle et al. (1980) felt that the decreased Secchi depths in the lower estuary during the summer in recent years reflect increased standing stocks of algae and probably also of organic detritus; an alternate explanation is that small particle sediment levels have increased. Increases in algal standing stocks imply that algal production has increased to a rate greater than that of its consumption, and that a concomitant increase in BOD has also occurred. This is of concern because, in the lower Patuxent estuary, which is often stratified in the summer, oxygen concentrations are quite low in the earliest data, and they may be driven lower by the settling of organic matter with high BOD produced in surface waters. Still unresolved is how great a role is played by nutrient rich-oxygen poor deep water advected into the river from the Bay. Clearly, inputs from the Bay are important; likewise, nutrient inputs to the lower river from upstream sources may stimulate organic production in the lower river and increase BOD. This increased BOD may further depress deep-water oxygen concentrations. Oxygen concentration and factors that affect it in the lower Patuxent are discussed in greater detail below.

One of the more common effects of excessive enrichment is increased variation in diurnal and nocturnal dissolved oxygen concentration in the water column, in response to greater levels of community metabolism. This represents a particularly serious problem when nighttime consumption of oxygen by respiration becomes great enough to lower oxygen tension to a point where it jeopardizes the viability of aerobic organisms in the community. Under such conditions, we observe the nuisance conditions most often associated with excessive nutrient enrichment or as many refer to it "eutrophication." There is evidence that day/night deflections in oxygen concentration in the upper Patuxent are increasing, although the problem, at least at Benedict where the measurements have been made, has yet to reach crisis proportions. Cory (1974) and Cory and Nauman (1970) noted evidence for such changes in the Patuxent at Benedict during the period from 1963 through 1969. They observed greater extremes in concentration of dissolved oxygen and a reduced ratio of production in respiration during that period, suggesting that increased levels of heterotrophy are

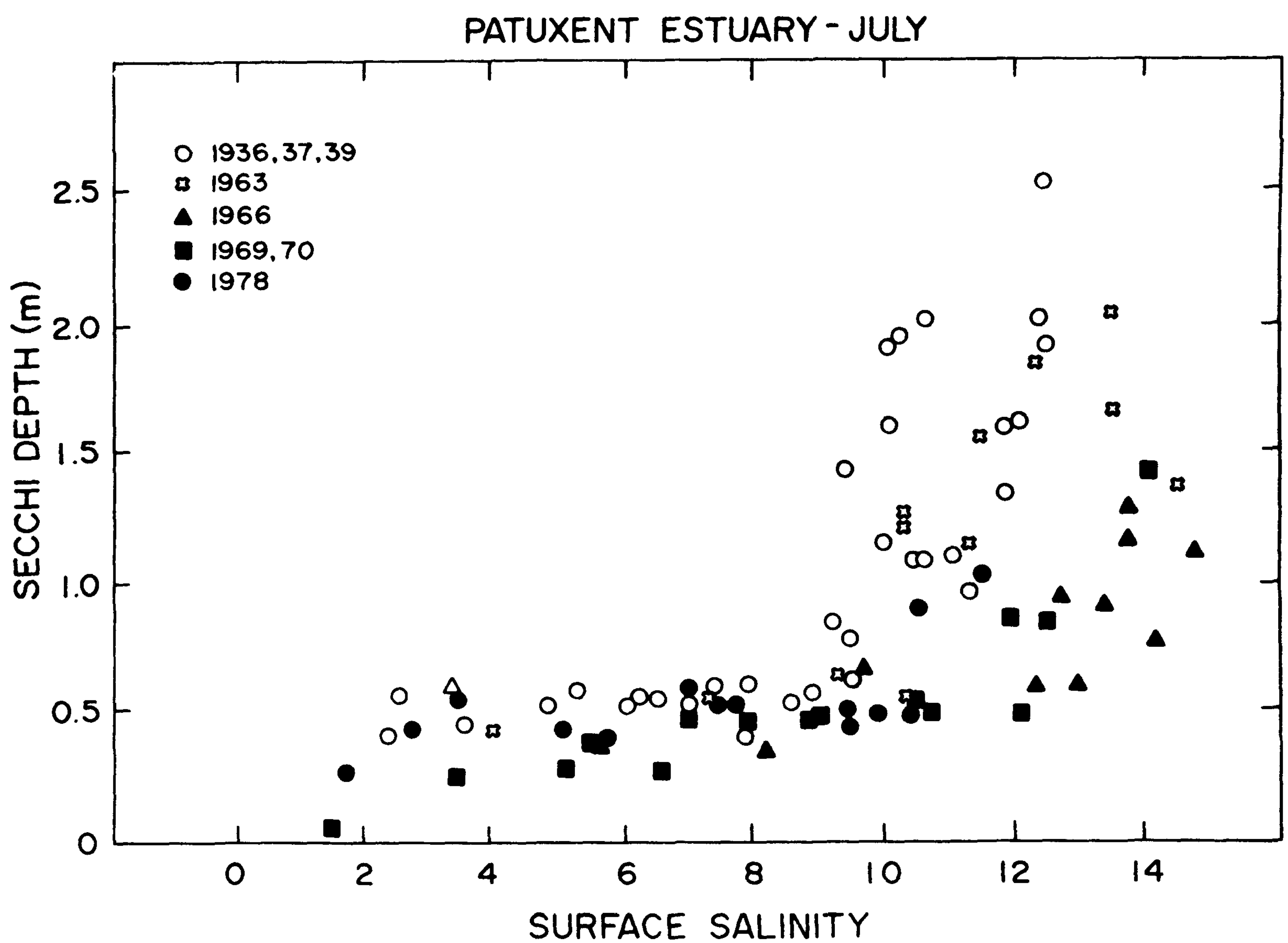

Figure 9. Secchi depth during July in the Patuxent estuary versus salinity (Heinle et al. 1980).

occurring. In a later unpublished study from which Cory made his data available to Heinle et al. (1980), it appears that continued changes have occurred between 1969 and 1977. Figure 10 shows weekly maximum and minimum concentrations of dissolved oxygen during May through August near the surface at Benedict Bridge. Minimum concentrations observed (about 2 mg O_2 L^{-1}) are fortunately, transient, but are nonetheless approaching dangerously low values. The increased range of values in 1977 over that of 1964 is clearly evident in Figure 10.

The greatest ecological concern in the Patuxent River does not rest in oxygen concentrations nor in aesthetic deterioration by enhanced turbidity in upstream waters but, instead, in the oxygen concentration in the deep waters of the lower estuary. In a stratified body of water such as the Patuxent estuary, increased productivity in the surface waters can cause decreased oxygen concentrations in deeper waters as organic matter settles in the water column and decomposes. Sustained oxygen depletion (perhaps by this mechanism) is known to occur naturally in the central part of the Bay (Newcombe and Horne 1938, Taft et al. 1980). On the basis of present information, the extent of this low-oxygen water is increasing with time.

Nash (1947) observed that the differences between surface and bottom concentrations of dissolved oxygen were greater at times of greater stratification, and he postulated that the degree of stratification was an important determinant of bottom-dissolved-oxygen concentration. D'Elia and Farrell (unpublished manuscript) have plotted bottom-dissolved-oxygen content of lower Patuxent waters, versus an index of stratification, surface to bottom salinity difference, over a period of three summers (Figure 11). They have verified Nash's observations that stratification strength is a critical consideration. Bottom-oxygen levels decrease with increasing stratification, because mixing with aerated upper waters is prevented. Similar results have been observed for the mainstem of Chesapeake Bay (Taft et al. 1980) and for the lower York River (Webb and D'Elia 1980). This greatly complicates the interpretation of nutrient enrichment effects, and it is not surprising that bottom-dissolved-oxygen content in the historical data base shows a wide variation within a given year (Figure 12).

The long-term decrease in mean oxygen content of deep waters in the lower Patuxent is one of the more striking examples of an enrichment-related phenomenon in the mesohaline regions of Chesapeake Bay. Figure 12 shows that recent, bottom-dissolved-oxygen content in the lower Patuxent is considerably lower on the average than it was in earlier years. The highest concentrations observed in the deep water south of Benedict do not exceed about six mg L^{-1} in the recent data, whereas in the 1936 to 1940 data, deep water oxygen concentration maxima were twice that, reaching supersaturation at 12 mg L^{-1}. Heinle et al. (1980) noted that the Winkler oxygen method has remained essentially the same for decades and, after checking and verifying the accuracy of the notes and calculations of the original analyst, concluded that the early data were reliable.

Concentrations of dissolved oxygen in bottom waters between Benedict and Broomes Island appear to be affected by in situ respiration and decomposition of organic matter produced within the Patuxent estuary and by intrusion of Bay waters naturally low in dissolved oxygen. The relative roles of these two causes of oxygen depletion are not certain. Low concentrations of dissolved oxygen are often observed downstream of Broomes

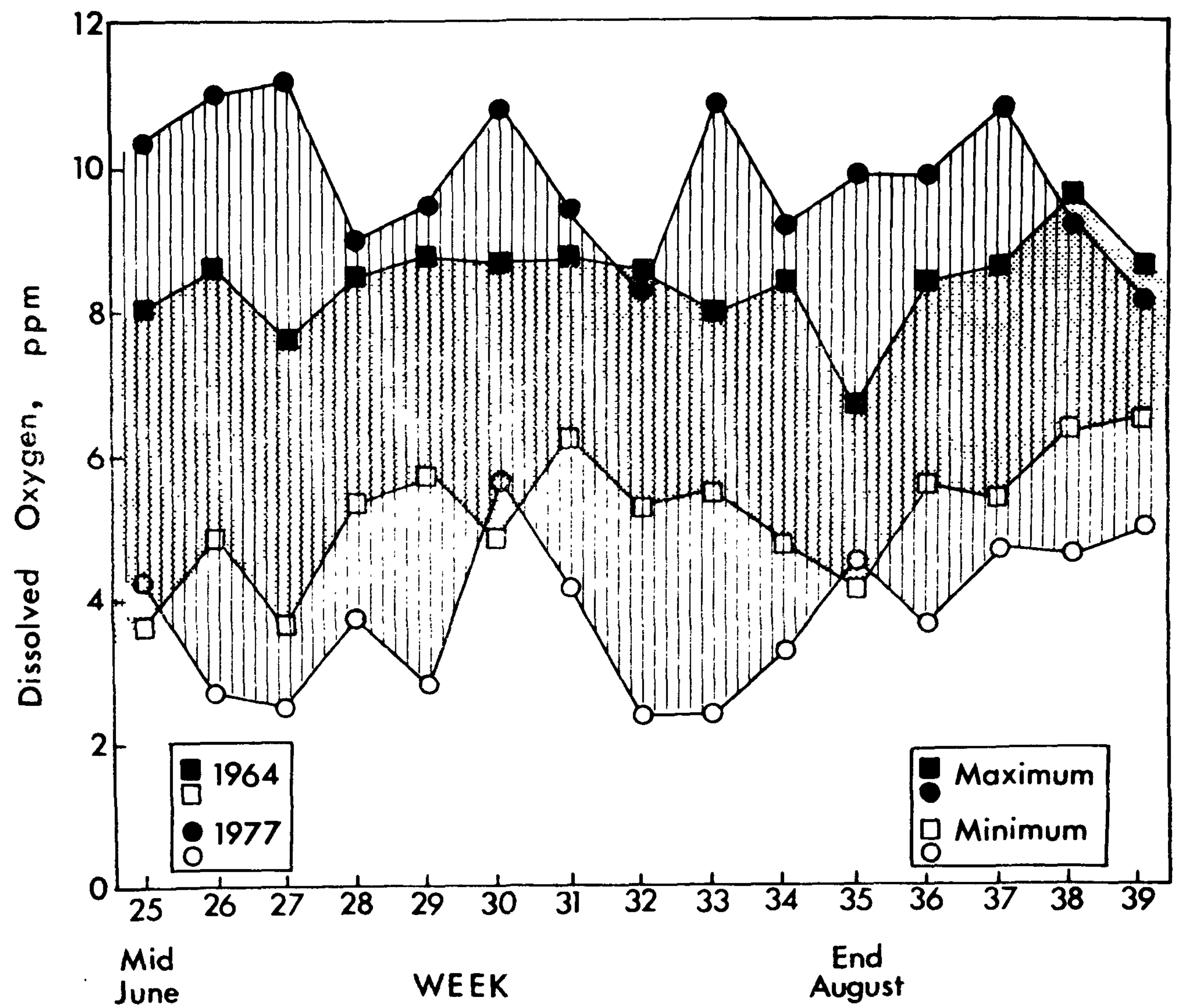

Figure 10. Weekly maximum and minimum concentrations of dissolved oxygen at Benedict Bridge in the Patuxent estuary during 1964 and 1977 (courtesy of R. L. Cory, U.S. Geological Survey).

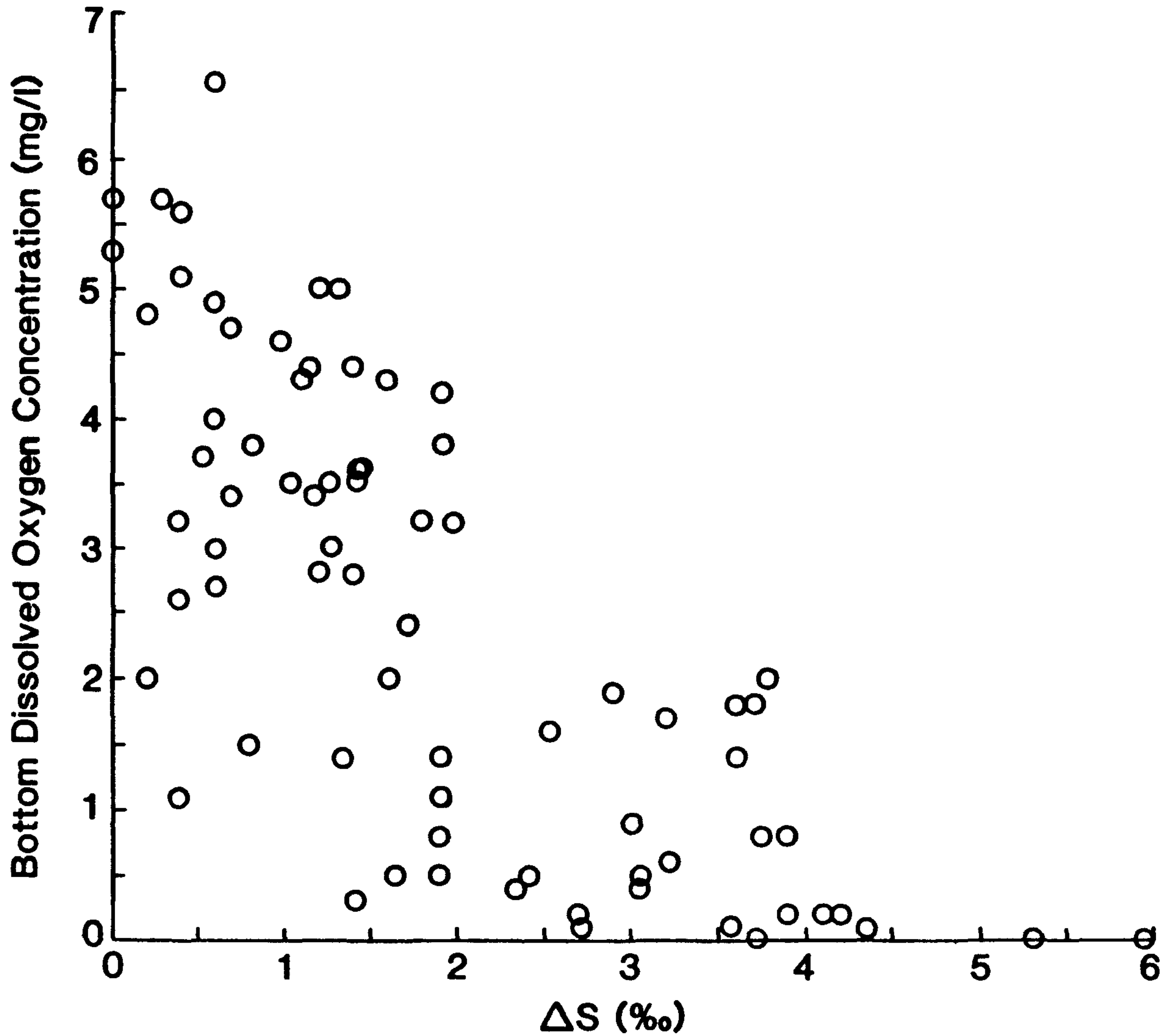

Figure 11. Bottom dissolved oxygen concentration in the lower Patuxent River during July and August 1978-1980 versus a stratification parameter (ΔS -- surface to bottom salinity difference) (D'Elia, unpublished).

Island. On occasions when concentrations are low upstream of Broomes Island, they are not so low downstream near the mouth of St. Leonard's Creek (Figure 12). This suggests that the Bay is not the sole source of the very low dissolved oxygen water.

Potomac River--

The Potomac River has been studied with varying degrees of intensity since 1913, yet the early data set for the Potomac is not so extensive as it is for the Patuxent, where CBL scientists were conducting some of the first intensive basic research on the nutrient distribution and dynamics of an estuary. Wolman (1971) reviewed the history of the effects of a growing population on continuing efforts toward improvement of water quality in the Potomac. Jaworski et al. (1972) also discuss the changes that have occurred there. The USGS is presently conducting comprehensive studies on the water column and sediments of the river, expecting to produce detailed reports on their studies in the next year. There is an excellent environmental atlas of the Potomac River (Lippson et al. 1979) that should also be consulted for further details.

Because the most serious problems in the Potomac occur near the head of tide near Washington, DC, most scientific and monitoring efforts have dealt with that region of the river. Yet even now, with concern growing about the higher salinity regions farther south in the river, most debate and study of water quality still center on upriver regions. Cumming et al. (1916) apparently measured nutrient and dissolved oxygen concentrations in the lower estuary during 1913, but Heinle et al. (1980) could not locate the data. Although CBI did conduct some sampling in 1949 to 1951 (Hires et al. 1963, Stroup and Wood 1966), the first intensive studies of water quality that encompassed the length of the estuary were those of CBI during 1965 to 1966 (Carpenter et al. 1969). What data do exist for the Potomac estuary suggest that slightly higher concentrations of phosphorus and considerably higher concentrations of chlorophyll *a* occur during the summer in the lower Potomac. By the time of the CBI studies, quite elevated chlorophyll *a* concentrations of 80 to 100 ug L^{-1} were common in the portion of the estuary up to 20 miles or more downstream from Washington, DC. Dissolved oxygen levels frequently reached low concentrations and there were substantial blooms of blue-green algae (Jaworski et al. 1971b, 1972). Since that time, plans were made to limit both the N and P levels in the effluent from the largest single point source to the Bay, the Blue Plains Sewage Treatment Plant. However, N controls were never instituted. A battle still rages over the effectiveness of the single nutrient advanced wastewater treatment strategy in force; there have been hearings held in front of administrative law judges in the past year.

The floating mats of blue-green algae that were prominent during the 1960's were not observed in the more recent studies, and this has prompted EPA officials to regard the present Blue Plains effluent limitations as effective. Proponents of the opposite point of view argue, instead, that flow regimes and hydraulic detention times, characteristic during the periods of the worst problems with blue-greens, have not occurred in recent years. Irrespective of the outcome of the controversy, it is apparent that some nutrient control strategy will be necessary to prevent future problems.

An interesting contrast occurs between the Potomac, where extensive blue-green algal blooms are seen, and other tributaries to the Bay, where

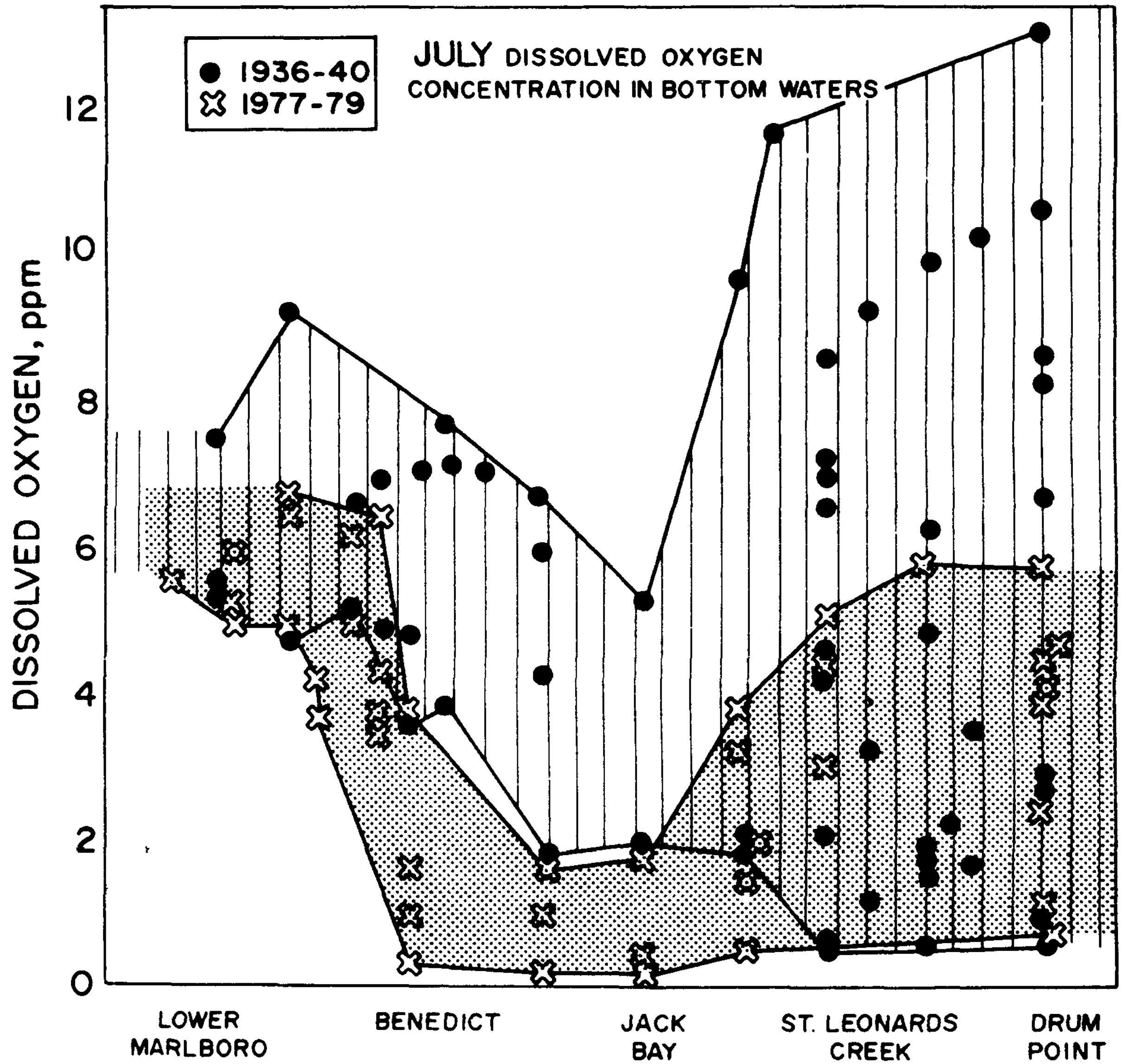

Figure 12. Concentrations of dissolved oxygen in bottom waters of the Patuxent River estuary during July 1936-1940 and July 1977-1979.

they are not. Blue-greens are rarely dominant in the water columns of saline environments of any tributary, including the Potomac; in the freshwater parts of the Potomac, N:P input ratios and characteristic hydraulic features probably account for the blue-green blooms.

James River--

Heinle et al. (1980) were unable to locate substantial early data for the James River. The first useful data on the James River were obtained by CBI in 1950, followed by a more complete study by Brehmer and Haltiwanger (1966), who sampled farther upstream than previous workers. By the time they began their study, the upper James appeared to have already been affected by enrichment. Summer chlorophyll concentrations of 50 to 80 ug L^{-1} were common at their upriver stations in the tidal-freshwater portion of the estuary, and 20 to 50 ug L^{-1} were observed at their midriver stations. Prior to enrichment, annual chlorophyll maxima in the low salinity regions of all of the western tributaries probably rarely exceeded 30 to 40 ug chlorophyll *a* L^{-1}.

DIP concentrations upriver show no seasonal or longitudinal patterns in the data of Brehmer and Haltiwanger (1966); typical values are less than 1.0 ug-atom L^{-1}. Downriver, a slight summer-concentration maximum is apparent as is characteristic for the Chesapeake estuary (Taft and Taylor 1976a, 1976b) (Figure 13). Data collected in the 1970's (Adams et al. 1975) show markedly higher concentrations of DIP through most of the year than in the earlier data (Figure 13). There have also been significant increases in nitrate and nitrite in the lower James estuary (Figure 14). Earlier data evidenced the spring seasonal maximum characteristic in Bay tributaries; in the latter study nitrate levels were high year-round.

In spite of the high ambient levels of both N and P in the lower James, concentrations of chlorophyll *a* have apparently not increased (Figure 15). The explanation for this apparent lack of response to enrichment is uncertain, but may simply relate to an increased turnover rate, but not to standing stock of plant material or to inadequate data availability.

York and Rappahannock Rivers--

In recent years, the York and Rappahannock have also exhibited increased levels of chlorophyll and nutrients; changes are comparable to those observed in other tributaries, so they will not be reviewed in detail here. There are some interesting hydrographic aspects of the York, James, and Rappahannock Rivers that have bearing on the water quality of those estuaries. Haas (1977) noticed that there was a striking correlation on those rivers between the occurrence of high spring tides and destratification of the water column. Since then, in more detailed studies of this predictable occurrence, it has been learned that the water quality characteristics are affected greatly by this cycle (Webb and D'Elia 1980; D'Elia et al. 1981). As for the Patuxent River, the bottom-dissolved-oxygen concentration of the York (and presumably the James and Rappahannock) River is generally highest under conditions of destratification. Thus, the water quality of the river, as reflected in oxygen content of the bottom water, can alternate rapidly between acceptable and low values. Nutrients are also affected. Short-term phenomena like this greatly complicate the evaluation of enrichment in an estuary and cause considerable range in the values of water quality

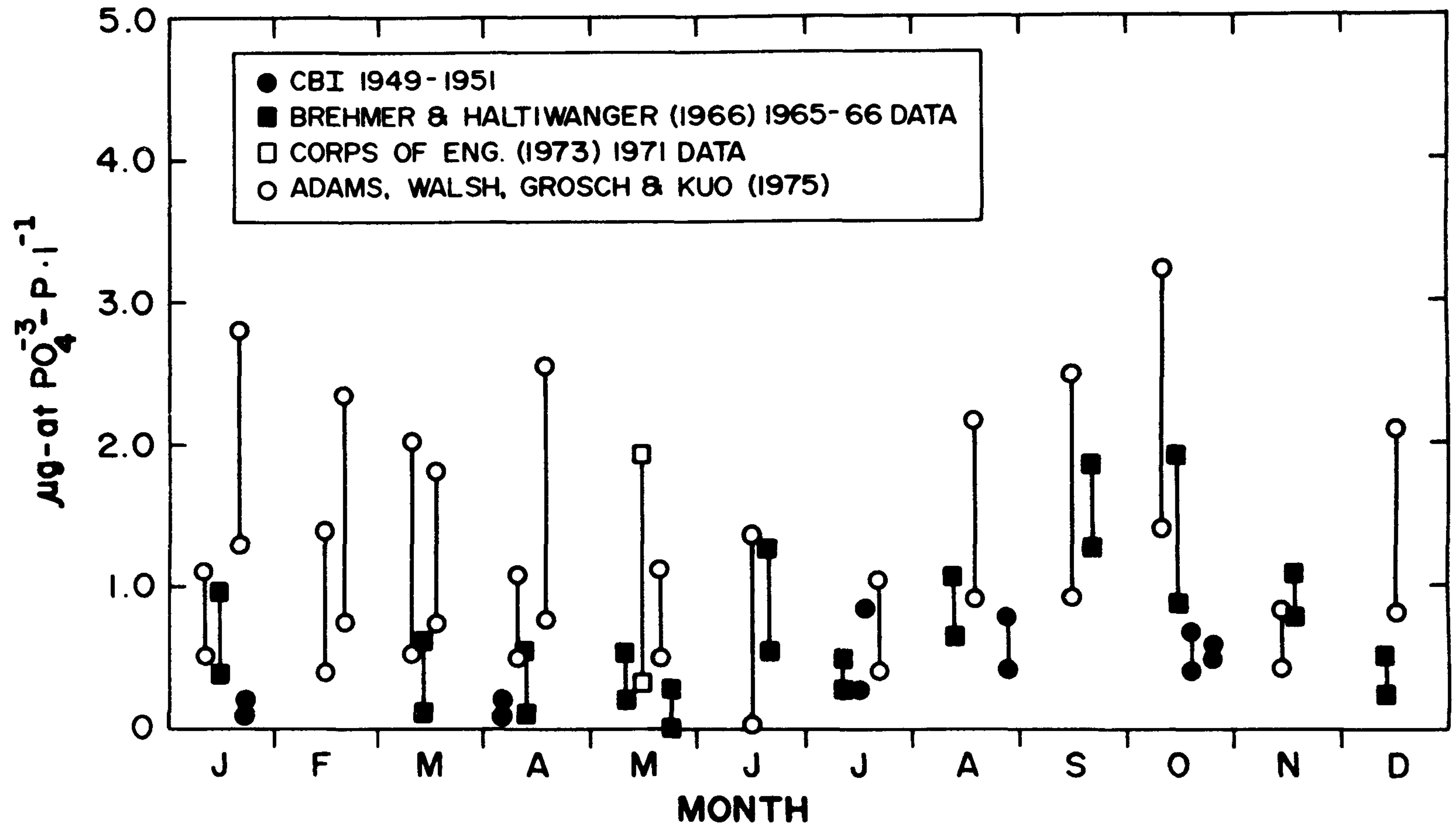

Figure 13. Concentrations of orthophosphate-P by month in the lower James River, Virginia.

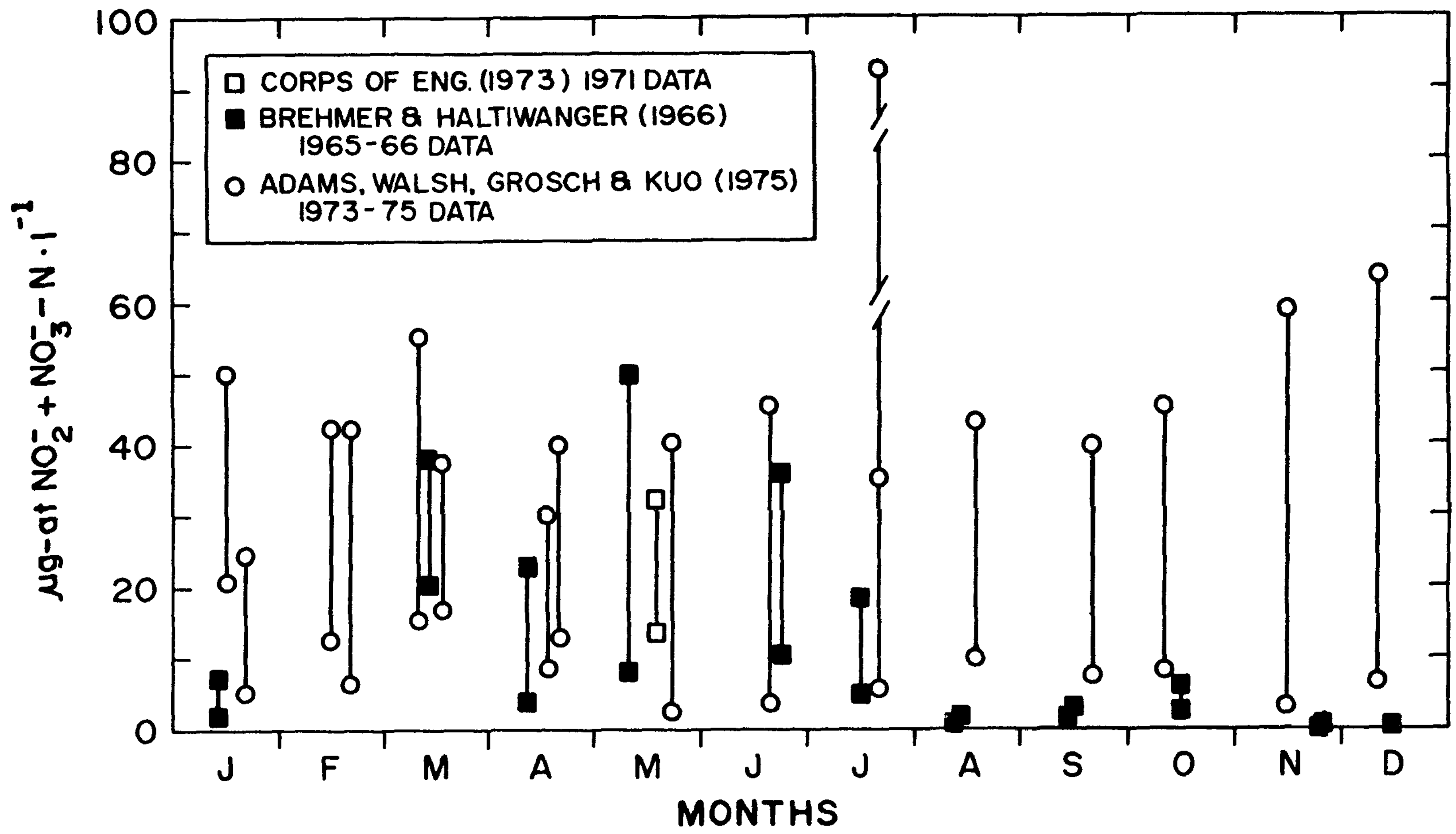

Figure 14. Concentrations of nitrate-N by month in the lower James River, Virginia.

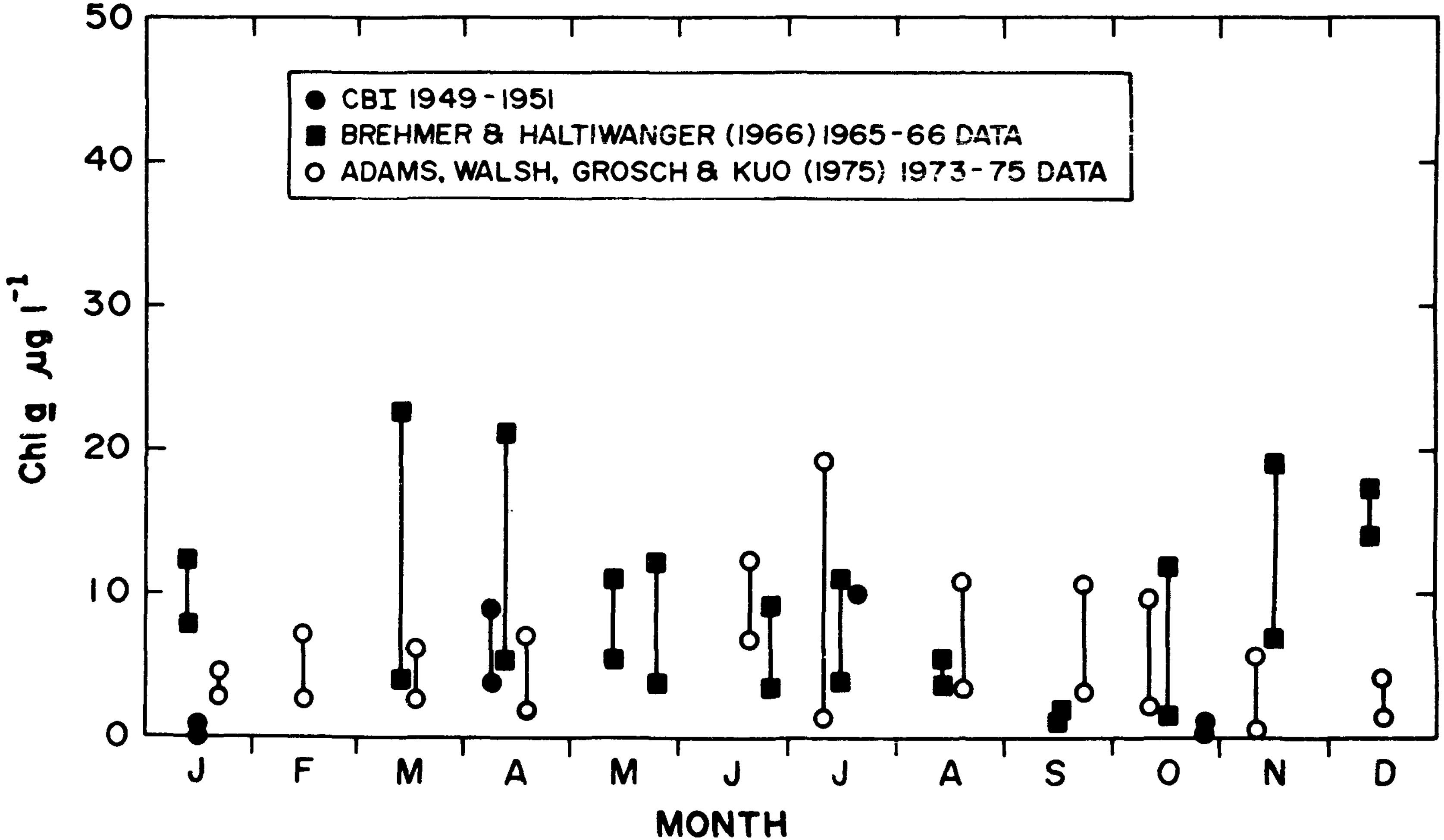

Figure 15. Concentrations of chlorophyll a by month in the lower James River, Virginia.

parameters measured. Such phenomena also seem to indicate that the steady-state assumption often used in water quality models may be a risky, one, if realistic model results are to be obtained.

Middle Chesapeake Bay

The mid-Bay is showing evidence of nutrient enrichment, but it is not as severely affected as are the western shore tributaries and the upper Bay, probably because of the sheer volume of this region and because of the ameliorating affects of dilution by low-nutrient sea water. A fair amount of early work at CBL was conducted in the mid-Bay off the mouth of the Patuxent River (Newcombe 1940, Newcombe and Brust 1940, Newcombe and Lang 1939); there is a reasonably extensive set of older data for this area.

DIP was comparable at all depths from 1936 to 1951, with values ranging from undetectable to 1.3 ug atom L^{-1} (Figure 16). By 1964 to 1966, maximum values increased to two ug atom L^{-1} and, by the mid 1970's, values of 2.5 ug atoms L^{-1} were observed (Figure 16). Chlorophyll a data show some increases in the mid-Bay between 1951 and 1964 to 1966. Peak values in the euphotic zone (upper 10 m) are less than 25 ug L^{-1} (Figure 17). The highest values were observed in the deep water, usually in winter or spring.

The data for nitrogen are less complete than for phosphorus. As in the tributaries, nitrate tends to be the dominant inorganic form in the winter and spring and is associated with high runoff. Salinity-dilution diagrams of the main stem of the Bay prepared by Taft (1982) indicate that this nitrate is conservatively diluted by seawater. This suggests that most of this nitrogen is passing through the mid-Bay unassimilated. Ammonium is more abundant in the summer and fall, but the lack of old historical data for ammonium leaves no basis for comparison. As in the tributaries, there is a late-summer, early-fall nitrite maximum in the mid-Bay (McCarthy et al. 1977, Taft et al., unpublished); this nitrite is probably derived from the oxidation of ammonium by nitrifying organisms (McCarthy, unpublished).

Phosphorus probably limits biomass in the spring when inorganic nitrogen is abundant (Taft et al. 1975; Taft and Taylor 1976a, 1976b). However, there are too few data to establish clearly a limiting nutrient in other seasons. Flemer and Biggs (1971) have noted that "the suspended particulate organic material in [that region] is suffering a relative loss of nitrogen with respect to carbon," and it may be that there is temporal variation in the limiting nutrient.

The range of dissolved oxygen values for surface waters is comparable in the earliest and latest data sets available (Figure 18). Oxygen concentrations in the deep water, however, seem to be depressed for longer periods in the summer and over wider regions of the mid-Bay. There is some concern that low-oxygen, high-nutrient water masses advected from the deep mid-Bay into the lower tributaries such as the Patuxent exacerbate present enrichment problems there.

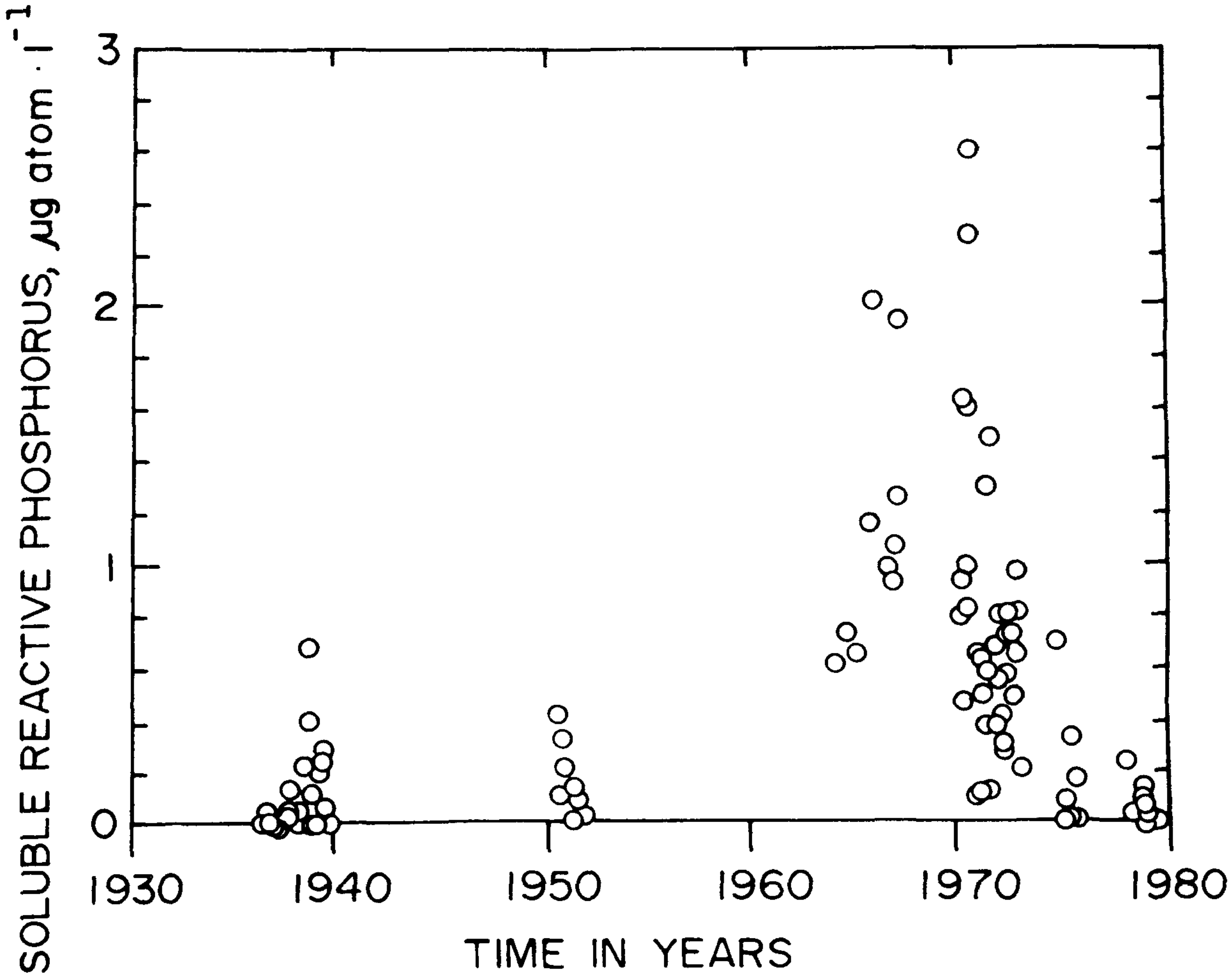

Figure 16. Monthly mean concentrations of orthophosphate-P (soluble reactive phosphorus) in the surface at 10 m at mid-Bay versus time.

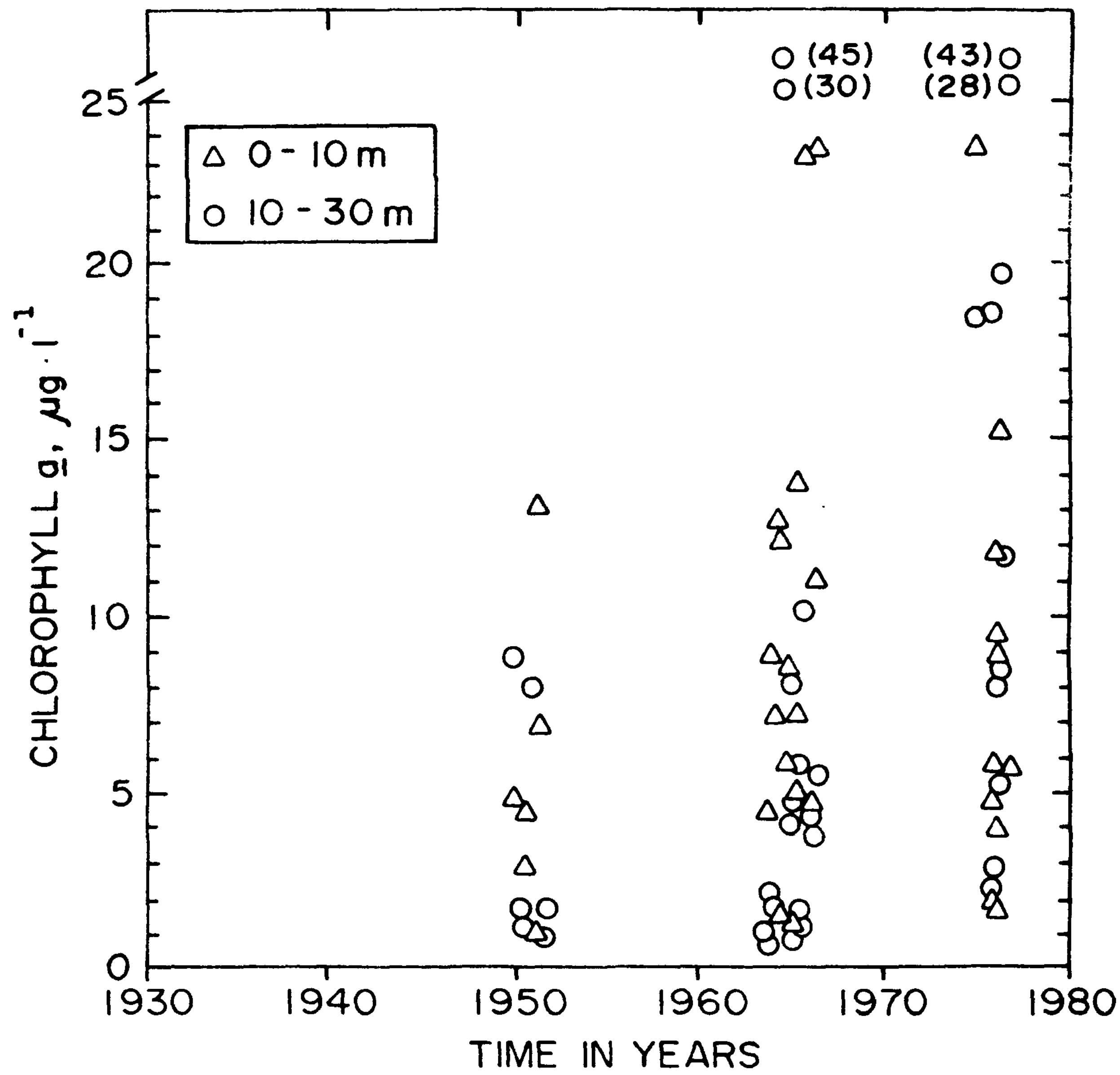

Figure 17. Concentrations of chlorophyll a at 0-10 m and 0-30 m depth versus time.

Lower Bay

Data for the lower Bay are available from CBI cruises of 1949 to 1951, 1961, and 1969 to 1971. Other data Heinle et al. (1980) used in comparison came from Smith et al. (1977), from Patten et al. (1963), and from Fleischer et al. (1977). Sufficient data exist to show that the lower Bay has remained relatively unaffected by nutrient enrichment upestuary. Because little change is evident, the data will not be reproduced here but instead will highlight the major features characteristic of the nutrient regime in the lower Bay.

DIP concentrations, throughout the lower Bay, were low historically and continue to be so. The summer maximum of phosphate reaches or exceeds slightly 1.0 ug atom L^{-1}, but is generally half that or less in other seasons. Nitrogen is not well represented in the historical data base so historical changes cannot be assessed. Recent data showed that nitrate availability in the lower Bay is similar to its availability in the central Bay--high-flow nitrate maxima are observed, and most of this nitrate probably passes out the Bay mouth unassimilated. Maxima in the spring may approach, or even rarely exceed, 25 ug atom L^{-1}. McCarthy et al. (1977) provide a detailed summary, by season, of nitrogen dynamics and the plankton of the lower Bay. Spring maxima in chlorophyll levels occur that exceed 20 ug L^{-1}; however, for the rest of the year, concentrations are generally below 13 ug L^{-1}, and are characteristic of a relatively unenriched system.

Eastern Shore Tributaries

The flows associated with eastern shore tributaries are trivial with respect to those of the western shore. Historical data suggest that moderate effects of enrichment can be observed in eastern shore tributaries. The earliest data were again obtained by CBI in the late 1940's. Early data show chlorophyll levels of less than six ug L^{-1} in the Chester, Choptank, and Miles Rivers, and low DIP levels as well (<0.6 ug atom L^{-1}). More recent observations show chlorophyll levels exceeding 25 ug atom L^{-1}.

Recent studies on SAV conducted for the EPA show that nitrogen is likely to be severely limiting on the Choptank River during the summer. Figure 19 presents results reported by Twilley et al. (1981) on dissolved inorganic nitrogen:dissolved inorganic phosphorus (DIN:DIP) ratios in the water column from April through September of 1980. There is a progression, from a condition in which DIN is far more abundant than DIP in April, to a condition in which the opposite is true in September. When N:P is less than 15, nitrogen limitation may occur; Figure 19 shows nitrogen becoming potentially limiting in July. DIN:DIP ratios shown for September are below 2.0 and are among the lowest values reported for the Chesapeake.

Most of the nutrients responsible for the observed enrichment of eastern shore tributaries undoubtedly derive from nonpoint source inputs associated with agricultural runoff. With the cost of fertilizers going up, more judicious and parsimonious application may occur, reducing loadings. Increased awareness of minimum tillage practices and wiser land use may also reduce nonpoint source inputs somewhat. Future nutrient enrichment problems will result more from population increases and associated point source loadings than from increases in diffuse sources.

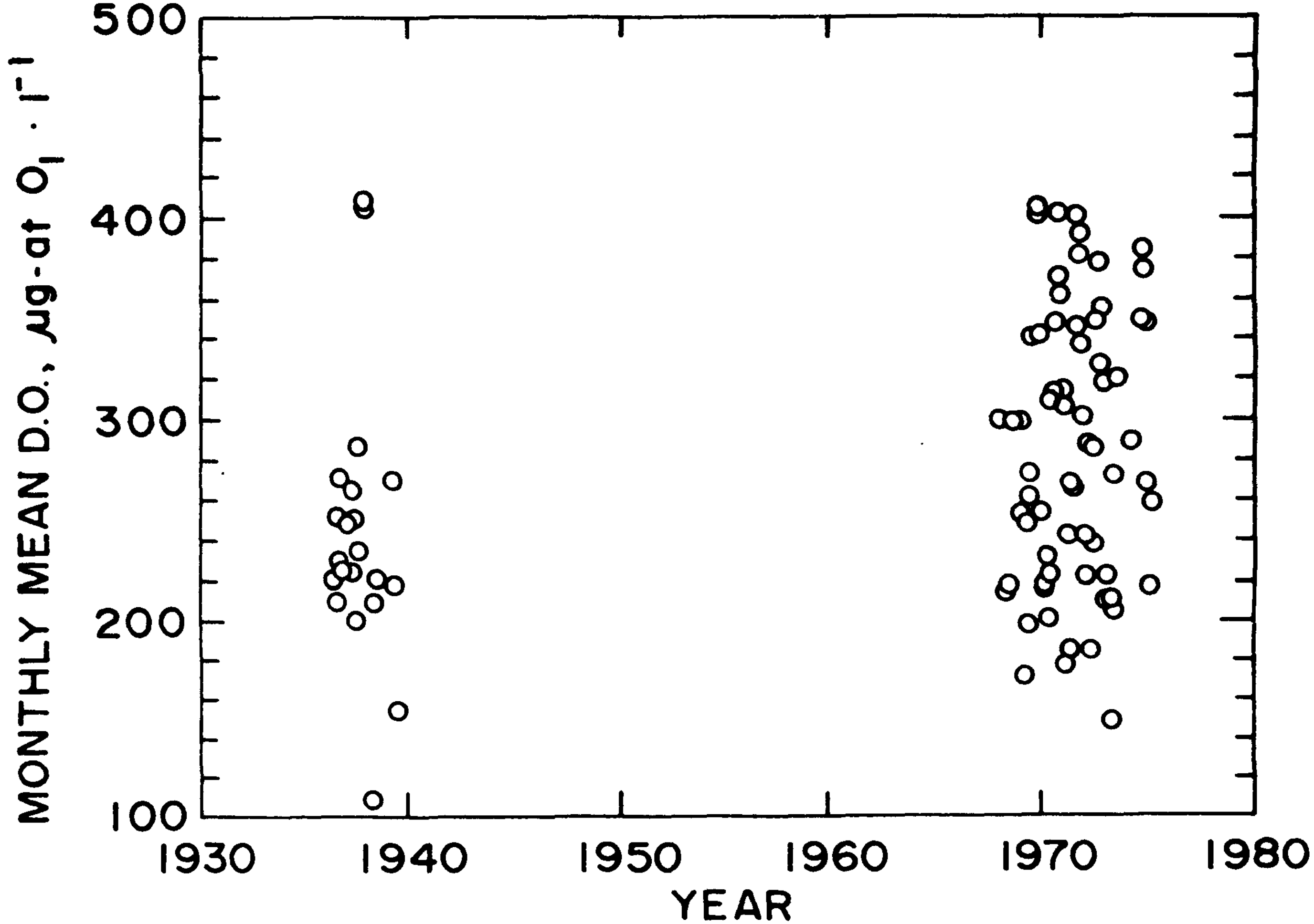

Figure 18. Daytime concentrations of dissolved oxygen (D.O.) in surface waters at mid-Bay during two selected time intervals.

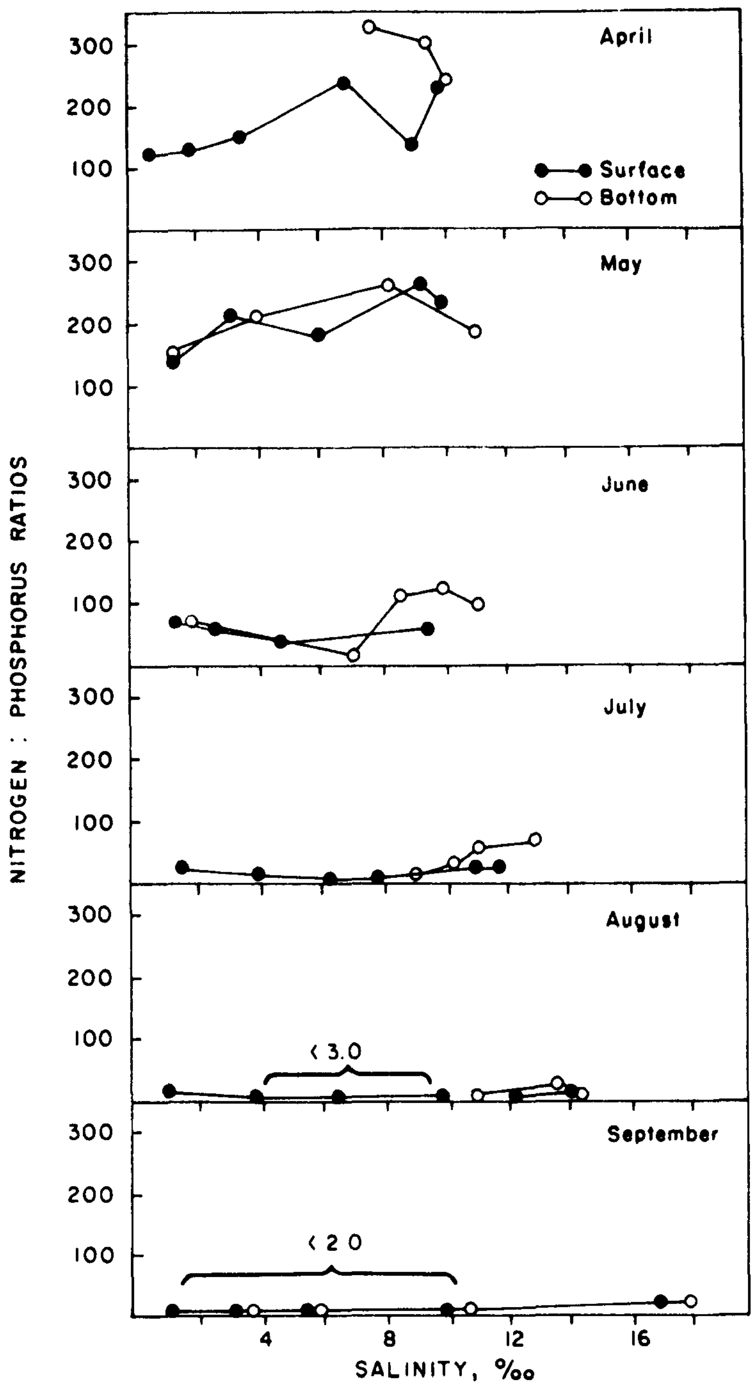

Figure 19. Nitrogen:Phosphorus ratios (dissolved inorganic nutrients only) in surface and bottom waters of the Choptank River, 1980.

The hypothesis has been advanced that the disappearance of submerged aquatic vegetation (SAV), once abundant in the shallow waters of the Eastern Shore and its tributaries, is because of turbidity related reduced light levels to a point below which SAV can survive. The historical data base on chlorophyll levels for eastern shore tributaries is consistent with this hypothesis. Since nutrient loadings to this area of the Bay are primarily from nonpoint sources, the prospect of controlling enrichment and associated plant biomass induced turbidities seems poor.

SECTION 5

SUMMARY AND CONCLUSIONS

Chesapeake Bay and its tributaries have undergone increased nutrient input over the last several decades. The most severe effects of this input that can be discerned with reasonable assurance have occurred in the tributaries. Particularly affected are the low-salinity regions near large urban centers and sewage treatment facility effluents. Figure 20, taken from the Heinle et al. (1980) report, gives approximate locations of the moderately and heavily enriched areas in the Bay and its tributaries. The criteria used in developing this figure are as follows: In the low salinity areas (less than 8 to 12 ppt), pre-enrichment concentrations of chlorophyll _a_ were believed, by those authors, to be less than 30 ug L^{-1}; hence values between 30 and 60 ug L^{-1} during the summer months were taken to indicate moderate enrichment. Concentrations over 60 ug L^{-1} were taken to indicate high enrichment. In the high salinity areas (greater than 8 to 12 ppt), where historical data suggest that concentrations of chlorophyll rarely exceeded 20 ug L^{-1} during the summer, concentrations of 20 to 40 ug L^{-1} were considered to represent moderate enrichment; values exceeding that, great enrichment. Although Heinle et al. (1980) recognized that chlorophyll levels per se were not necessarily bad, the relatively great change in chlorophyll concentrations, over apparent pristine levels, was considered a harbinger of enrichment problems. This is especially true when the chlorophyll levels, now encountered, represent the presence of an amount of organic material that when oxidized could account for depletion of oxygen from the water column in summer months. Heinle et al. (1980) emphasize that it is excessive oxygen depletion that most laymen and professionals regard to be the most severe result of over-enrichment of natural waters. Oxygen depletion problems in the Bay are discussed further below, but first some of the important regional concerns represented in Figure 20 will be summarized.

A good and well-known example of a severely affected location is the Potomac River near Washington, DC. Although other localities on the Bay and its tributaries are not yet considered to exhibit such serious symptoms of over-enrichment, effects of increased nutrient loadings have been noticed. For example, the upper Patuxent River in Maryland, for which an excellent historical data record exists, has shown signs of decreased transparency and increased nutrient concentrations and standing stocks of algae. The upper James River in Virginia can be considered similarly enriched.

There is concern in the lower Patuxent River that increased production of organic matter, as a result of increased nutrient loadings, may ultimately lead to lower dissolved oxygen concentrations, particularly in deep water, through the decay of organic matter. But because the nutrient dynamics and trophic structure of this estuary are not adequately understood it is difficult to predict or project through modeling exactly how the estuary will respond to increased loadings. The CBP's characterization analysis will discuss these responses further.

The other lightly shaded areas shown in Figure 20, like the lower Patuxent, are the middle salinity zones that are considered areas of prime concern. Figure 21 shows portions of Chesapeake Bay where Heinle et al.

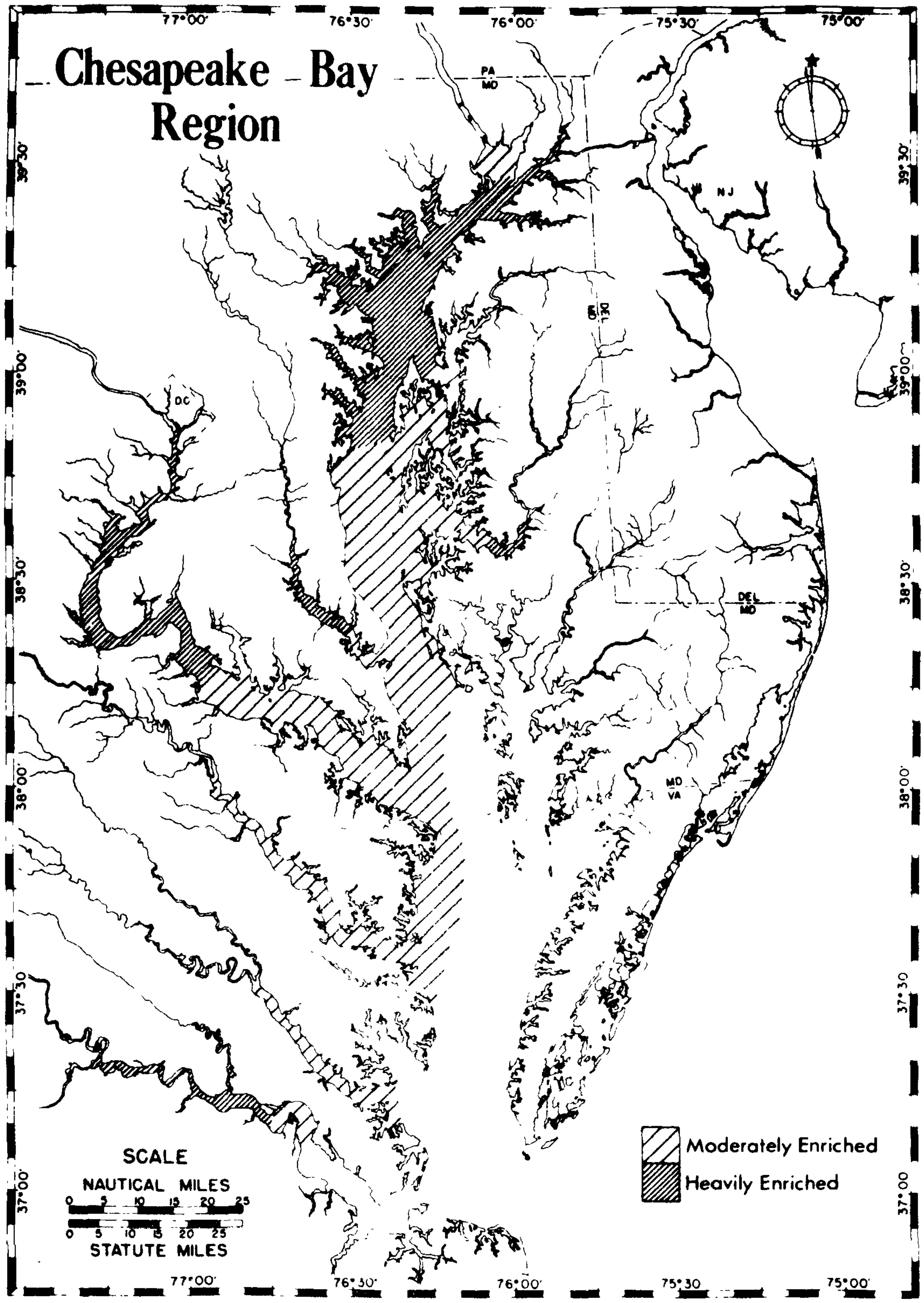

Figure 20. Map showing portions of Chesapeake Bay that are moderately or heavily enriched according to the criteria of Heinle et al. (1980).

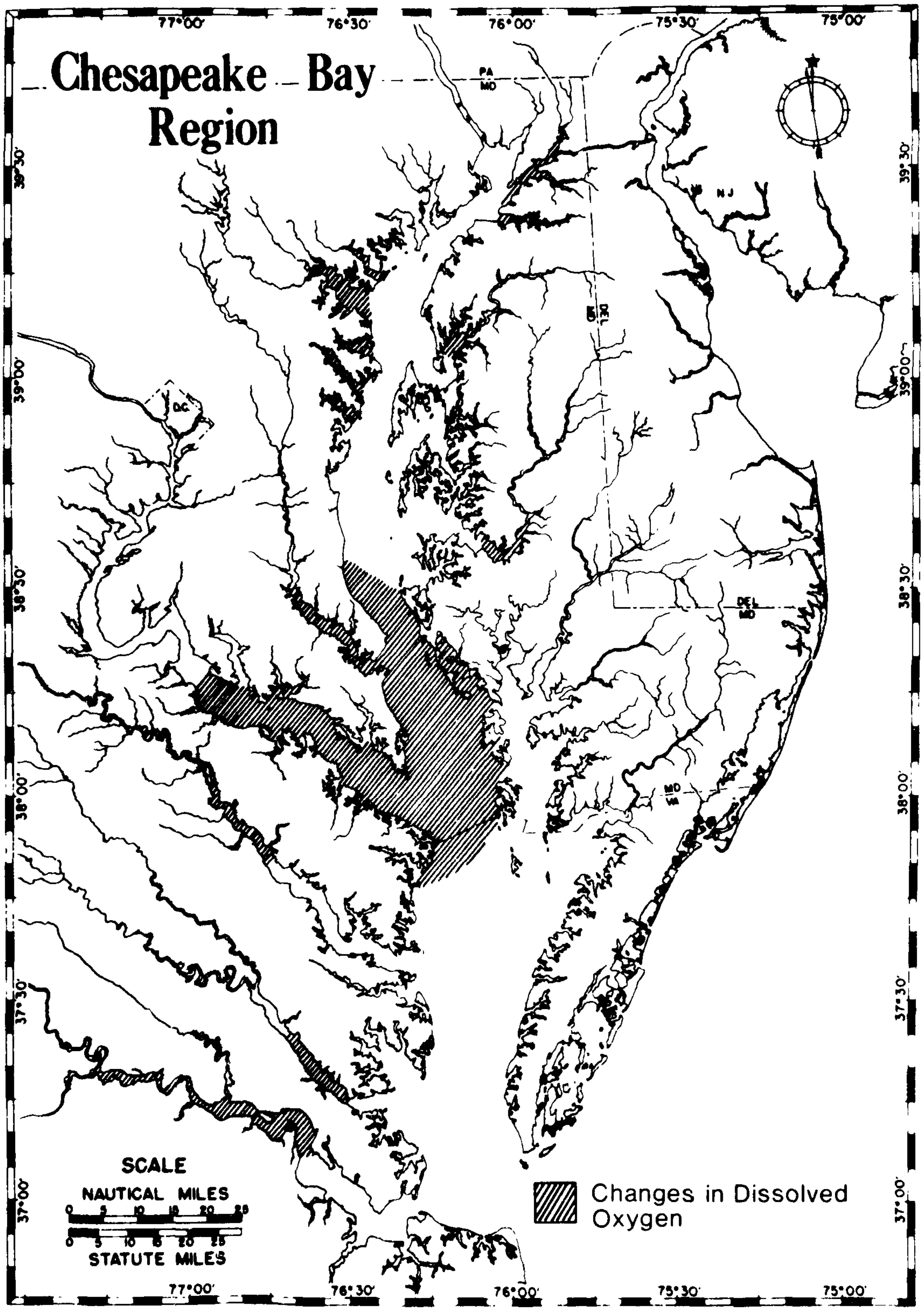

Figure 21. Map showing portions of Chesapeake Bay where natural regimes of dissolved oxygen appear to have changed.

felt that worrisome alterations in oxygen regime of the deep water in particular have apparently occurred in response to enrichment. Yet to be learned is whether the most important causes of oxygen depletion in these areas are enhanced productivities in local surface waters, periods of high freshwater flow and resultant poor vertical mixing and reaeration, or import and decay of organic material produced upstream. The interaction of these factors is not completely understood, and the historical data base is not comprehensive enough to allow us to analyze it adequately. However, apparent changes in oxygen regime in the mid-Bay must be viewed as tentative, but probable.

Because too little is presently known to manage the trophic structure of the Bay to result in increased fisheries yields from additional nutrient input, sensible efforts to control inputs should continue. The indication that nitrogen is often limiting in the lower and middle reaches of the Bay suggests that affordable advanced technologies for N enrichment control should be sought and given due consideration for implementation. However, other considerations are important; for instance, it will make little sense to implement nutrient-removal processes that will ultimately prove too costly to operate or too complex to manage properly. Workable management programs for the future will certainly involve better land use practices and control of nonpoint-source N inputs, particularly in the summer months when hydraulic residence times are longest. Unconventional or unpopular sewage treatment processes such as land application may prove important in controlling enrichment.

Continued scientific evaluation of the trophic structure and of the nutrient dynamics of the Bay will prove important if we are to assess adequately future changes and the efficiency of control strategies. Routine monitoring programs should be adopted and supplemented by more basic research into effects of enrichment on algal productivity, species composition, and the natural assimilative capacity of the environment for nutrients. An inventory of point source inputs should be established and kept up-to-date. These and other data are useful to environmental scientists. The partitioning of the carbon, fixed by algal photosynthesis among species at higher trophic levels, remains a poorly understood but critical area for research. Dose-response studies, such as those sponsored by the EPA in Narragansett Bay, Rhode Island, may prove extremely helpful in this regard. Scientists, modelers, and managers should work closely to develop models of hydrodynamics and of dose responses to nutrient addition. This information will help identify gaps in understanding the Bay's ecology and in locating problem areas.

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CHAPTER 2

Nutrient Processes in Chesapeake Bay

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FIGURES

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SECTION 1

INTRODUCTION

A principal characteristic of Chesapeake Bay (Figure 1) is that, like other partially mixed estuaries, its two-layer circulation pattern enhances the retention of nutrients. Although the flushing time for water in Chesapeake Bay is about one year (based on the basin volume and annual river flow), nutrients are not flushed out to sea in direct proportion. Instead, soluble nitrogen and phosphorus are incorporated into particles, such as phytoplankton, which sink from the seaward-flowing, surface water into the landward-flowing, deep water. In this way, nutrients entering from the tributaries are carried part way down the estuary, sink toward the bottom, and are carried back upstream. This accumulation phenomenon is the mechanism for desirable, high production on the one hand, and undesirable, over-enrichment on the other.

This chapter deals with the shaded portion of the binary diagram in Figure 2. In this portion, dissolved nutrients become particulate (biotic component). (Abiotic particulate nutrients, such as phosphate flocculants, are not discussed in this chapter.) The dissolved, and biotic particulate nutrient compartments are expanded in Figure 3a to show the different categories of dissolved and particulate constituents that will be discussed in the following sections. The soluble forms of inorganic nitrogen and urea are illustrated in Figure 3b. The transformations among these constituents are generally mediated by bacteria, but all four forms may be taken up and utilized by phytoplankton in Chesapeake Bay (McCarthy et al. 1977). Inorganic phosphorus, on the other hand (Figure 3c), is present primarily as orthophosphate, which may interact with adsorbing minerals such as iron oxyhydroxides under certain chemical conditions (Taft and Taylor 1976a).

This chapter has three purposes. First, it is intended to acquaint the non-scientist with fundamental concepts of the major estuarine processes related to water quality. Second, it illustrates the concepts with data from Chesapeake Bay or its tributaries. Finally, it relates the processes to management concerns with the hope that decision-makers will gain insight into the relations between water quality, the controlling estuarine dynamics, and potential management options.

The uptake of nitrogen and phosphorus by phytoplankton is a major pathway in the nutrient retention scheme in Chesapeake Bay. For this reason, Section 2 will discuss pertinent details of phytoplankton physiology, including patterns of nutrients available to phytoplankton and factors affecting their growth and productivity.

Another major pathway, also discussed in Section 2, is phytoplankton consumption by zooplankton. Zooplankton recycle some of the nutrients in the phytoplankton back into the water, assimilate some into body tissue, and release the remainder as particulate material which sinks to the bottom. This material, comprising detritus, is colonized and further degraded by bacteria, forming a third pathway that returns nutrients to deep water flowing back upstream. Nutrient recycling from the organic forms to the soluble inorganic forms requires oxygen utilization. Section 3 discusses oxygen sources and plankton respiration rates in relation to nutrient retention and recycling.

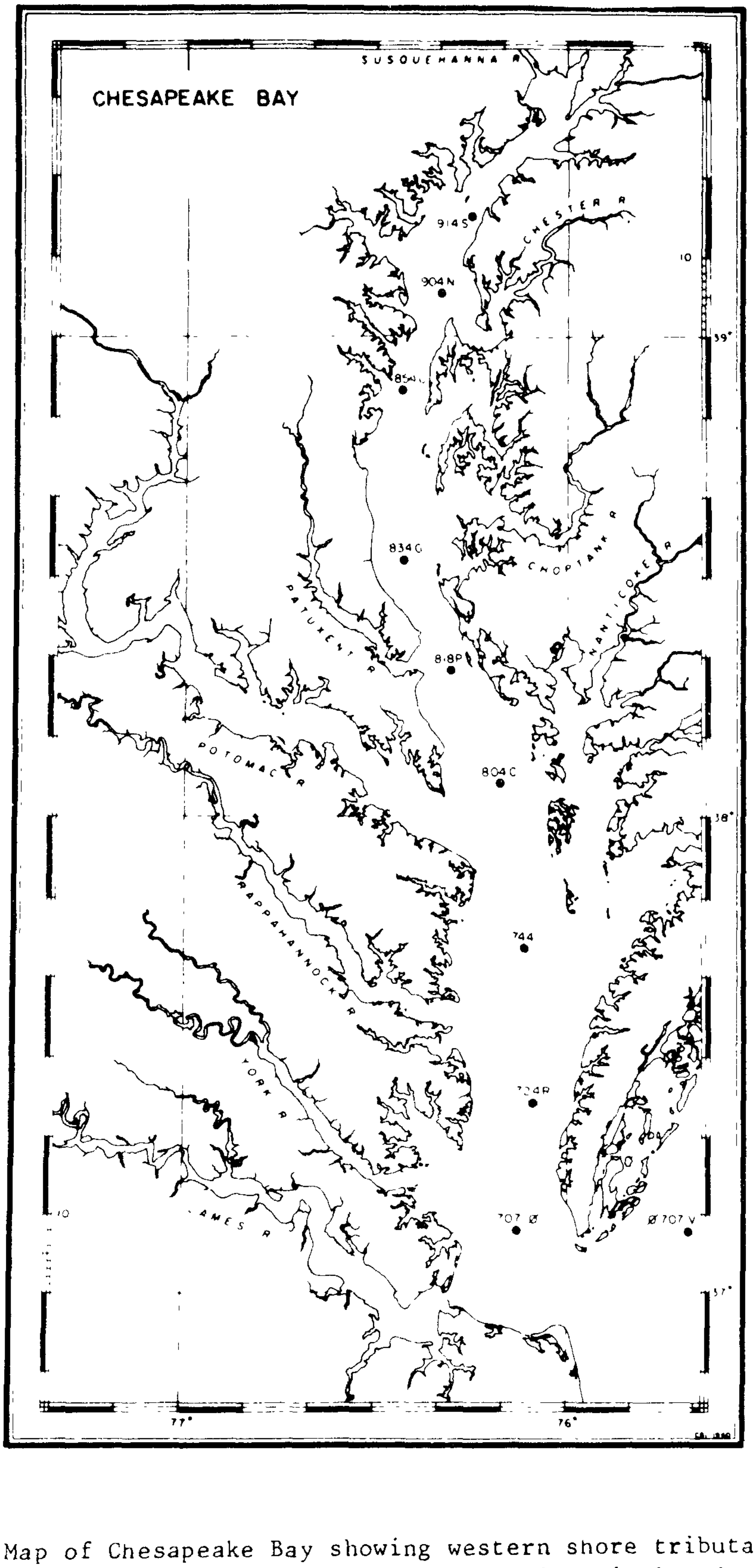

Figure 1. Map of Chesapeake Bay showing western shore tributaries and stations routinely sampled for biological and chemical data.

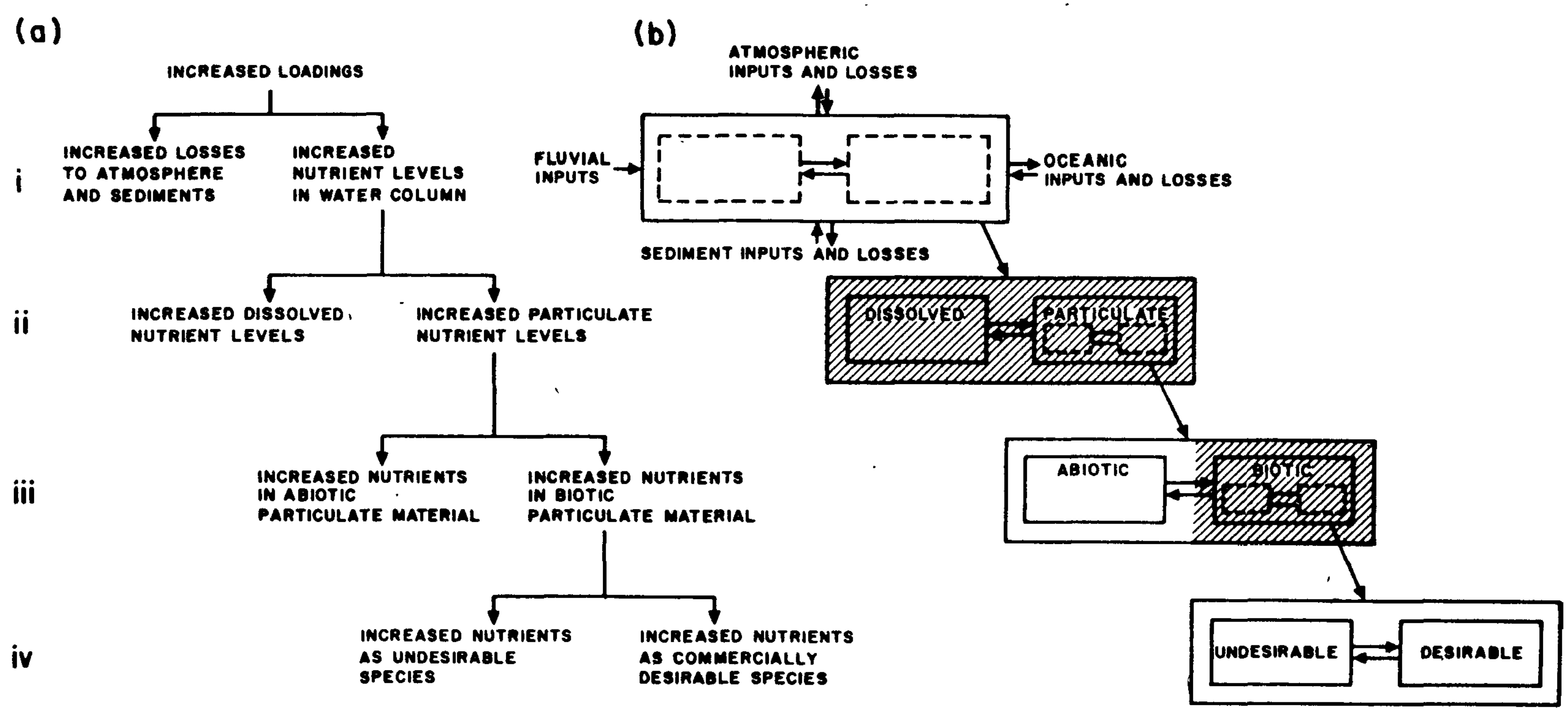

Figure 2. Binary dendogram showing possible response of the water column to increased nutrient loadings.

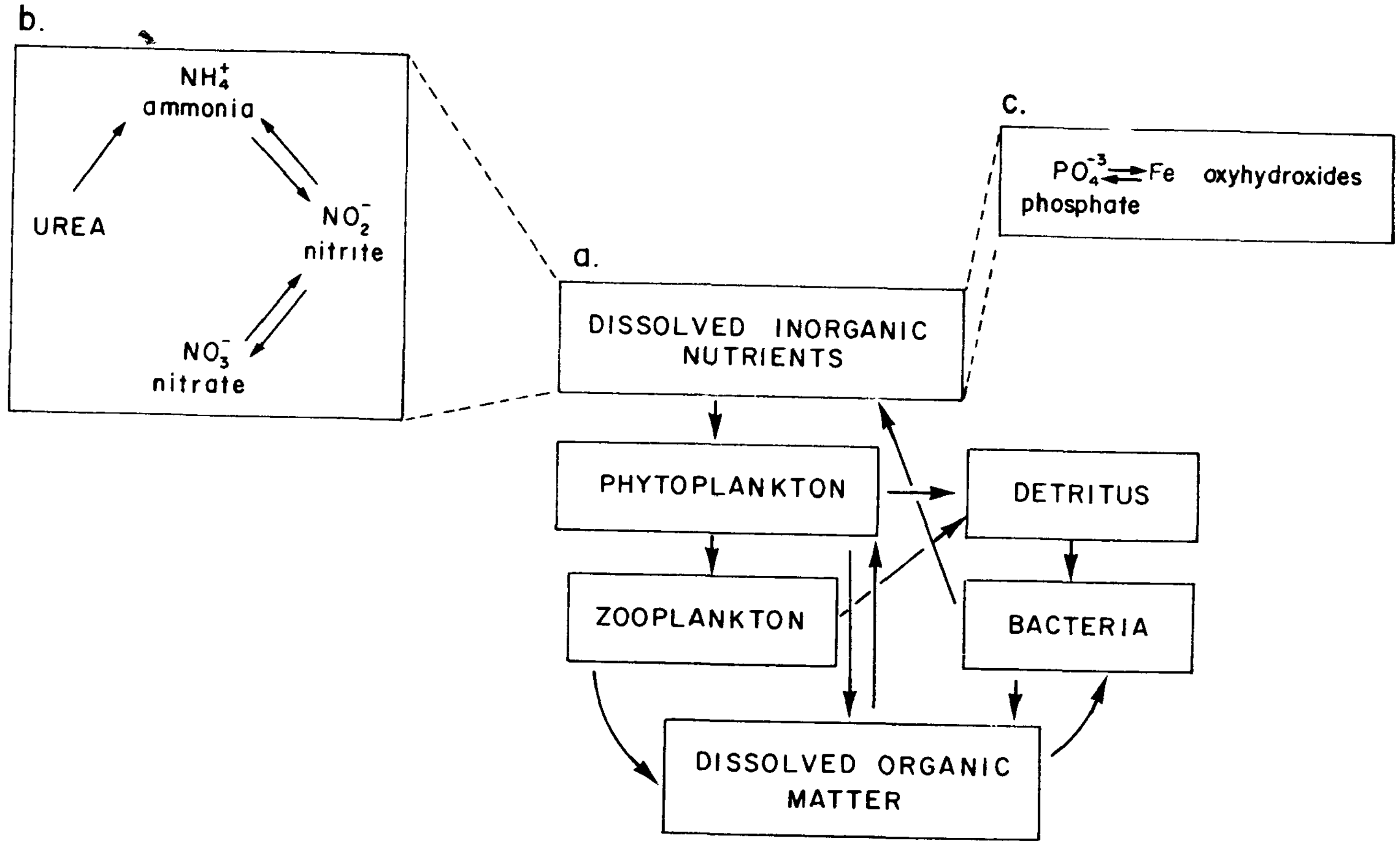

Figure 3. (a) Schematic diagram of the basic nutrient cycle in aquatic systems, (b) inorganic nitrogen cycle transformations mediated primarily by bacteria, and (c) interaction of orthophosphate with iron oxyhydroxides which occurs in the sediment and may occur in the water column or under certain conditions.

Since the readership of this paper may vary from citizen to scientist, the format will try to accommodate a range of technical expertise. The indented sections explain, in less technical terms, important concepts the reader should understand. The concepts are illustrated as much as possible with data from the Chesapeake Bay estuarine system.

SECTION 2

NUTRIENT AVAILABILITY AND PHYTOPLANKTON PHYSIOLOGY

PATTERNS OF NUTRIENT AVAILABILITY

Open Bay

The annual nutrient cycle in Chesapeake Bay is marked by three prominent events. The first is the substantial nitrate input with winter and spring runoff from the Susquehanna River (Carpenter et al. 1969). The source of nitrate in the runoff is partly ground water and partly atmospheric. Rain and snow contain nitrate concentrations of up to one-third to one-half those in the runoff (Smullen 1982). Dilution in the upper Bay (Figure 4 and Figure 5) followed by phytoplankton uptake in the mid- and lower Bay depletes this nitrate from about 40 to 100 ug atom L^{-1} to less than one ug atom L^{-1} by midsummer. Figure 4 shows how nitrate is depleted toward the Bay mouth; Figure 5 shows its seasonal presence. The bottom diagram shows nitrate present in May, but undetectable in August (not shown in Figure). In contrast to the heavy input of nitrate, orthoposphate is undetectable throughout spring (top diagrams).

The second important event occurs during midsummer when very low oxygen concentrations in deeper Bay water permit release of phosphate and accumulation of both phosphate and ammonium there (Taft and Taylor 1976a, 1976b). Some of these nutrients are transported by diffusion and advection to the upper layers where they are incorporated into phytoplankton. The annual maximum for total phosphorus in the surface layer of the Bay usually occurs in summer, because phosphorus availability is greatest then. However, not all of the deep water phosphorus reaches the upper layer. New information suggests that some phosphorus may be precipitated by iron-rich minerals at the boundary between the upper and lower water layers (Figure 3c). This natural control of phosphorus at the boundary may, at times, prevent all of the nutrient from being available to the many non-motile phytoplankton. Strong swimmers such as the dinoflagellates, however, may migrate down to the nutrient-rich layer at night and up into the sunlight during the day. As a result, their growth is not limited by phosphorus availability.

The third event is the fall nitrite maximum observed in both mid-Bay (McCarthy et al. 1977) and in the lower Potomac River estuary (Taft, unpublished data). At present, ammonium oxidation appears to be the most probable mechanism to explain these observations (Figure 3b). An experiment was conducted as part of the Chesapeake Bay Program to measure the rates of this important process; results are discussed in Section 3.

Although several studies have examined the longitudinal (vertical) nutrient distributions in the Bay, none have explored the lateral distributions. Since lateral integration of parameters is a common feature in one and two dimensional models, it is necessary to show that lateral changes are small compared to longitudinal, or vertical changes. When such lateral measurements were made during April, 1977 they revealed an interesting picture. A layer of ammonium was observed at mid-depth (Figure

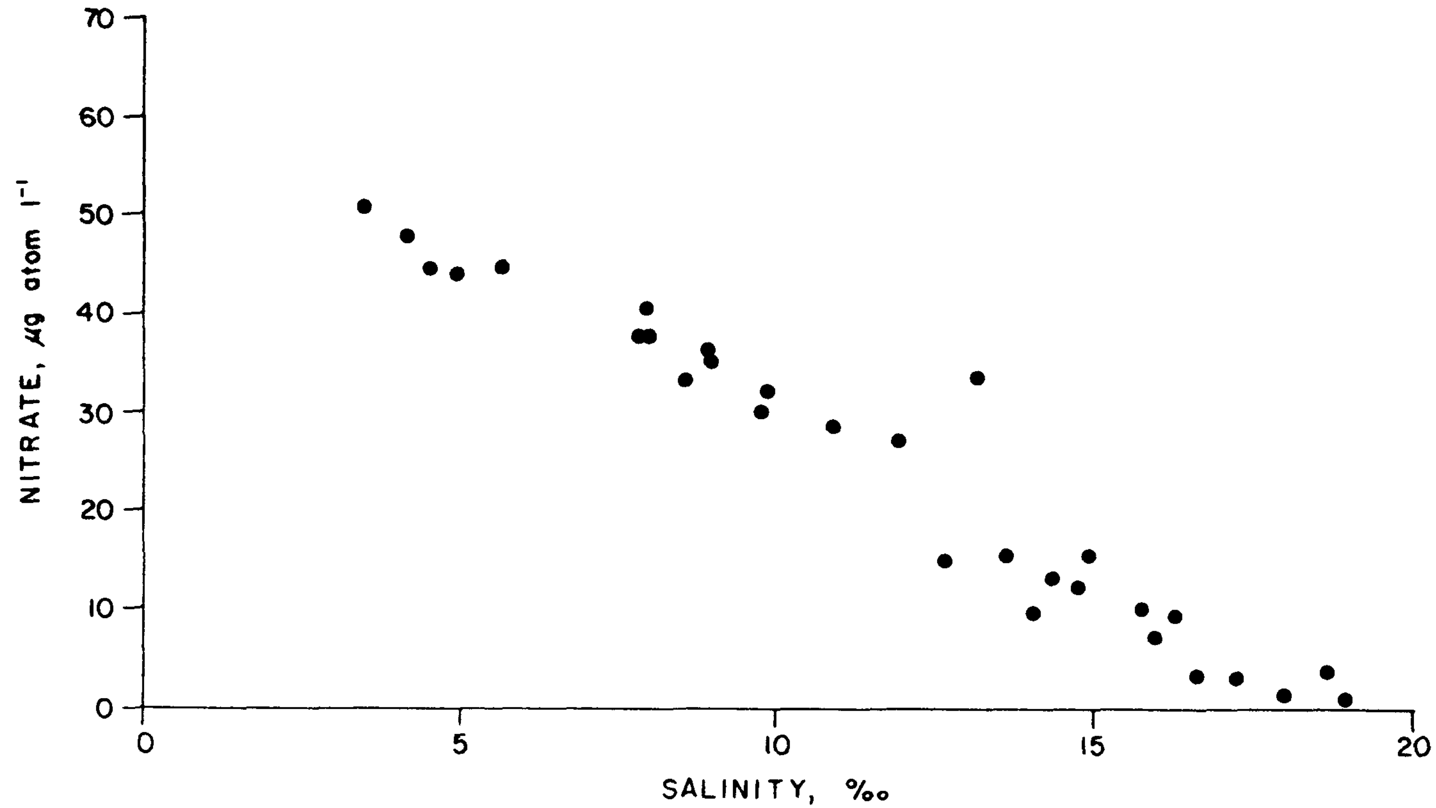

Figure 4. Graph of nitrate versus salinity in the upper Bay showing conservative dilution of nutrient from the Susquehanna River with low nitrate seawater.

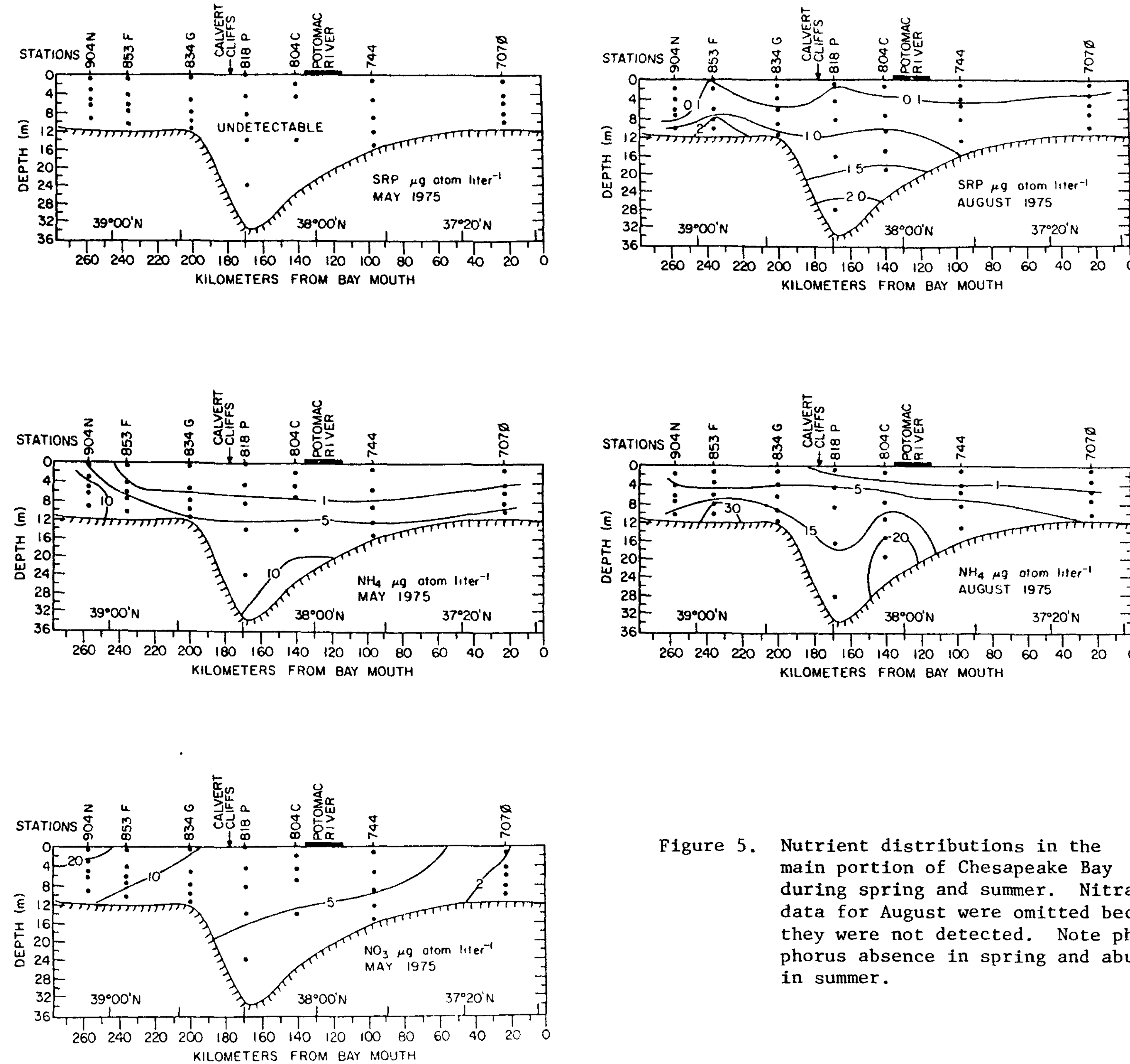

Figure 5. Nutrient distributions in the main portion of Chesapeake Bay during spring and summer. Nitrate data for August were omitted because they were not detected. Note phosphorus absence in spring and abundance in summer.

6), extending over much of the northern half of the Bay. This feature was not observed during subsequent summer and winter cruises. At present, the best explanation is that ammonium-enriched deep water is displaced westward and upward by sea water, flowing in along the bottom on the eastern shore. This view is particularly well supported by Figure 6a in which the maximum value of nine ug atom NH_4-N L^{-1} is found on the bottom along the eastern shore sill (station #834A), but is at mid-depth in the eastern channel (station 834C) where it has been displaced upward. Thus, this action pushes nutrient-rich water upward to the photic zone where it is available to phytoplankton.

Tributaries

The Potomac River was selected as a representative tributary because of the extensive data on nutrient processes available. The analogue between the Potomac and Bay is further described in the forthcoming "Characterization of Chesapeake Bay" report. The patterns of nutrient availability in the Potomac River have been studied extensively for the last 20 years. This interest was stimulated by the necessity to discharge sewerage from Washington, DC into the river near the head of tide. Carpenter et al. (1969), Jaworski et al. (1972), McElroy et al. (1978), and others have examined nutrient dynamics and budgets. Najarian and Harleman (1977) and Najarian and Taft (1981) have modeled nitrogen dynamics using data from the Potomac. Much of the following discussion is true not only for the Potomac, but for the main Bay.

The Potomac River is somewhat similar to the main Bay with respect to the availability of nutrients. The lower Potomac displays the same summer release of phosphorus and the fall nitrite maximum (Taft, unpublished data) as described for the main Bay. There is not the same extensive spring nitrate influx, however. The sewage effluent from the Blue Plains Treatment Plant is a major source of nutrients to the Potomac; its effect on the availability of nutrients in the Potomac is discussed in the following paragraph.

Data are presented here for June 1977 to orient the reader; this is not intended as a comprehensive treatment. Figure 7, Figure 8, and Figure 9 show longitudinal distributions of salinity, dissolved oxygen, and chlorophyll a in the Potomac River. Figure 10 depicts surface nutrient concentrations. Ammonium entering the river from the Blue Plains Sewage Treatment Plant is diluted as it moves downstream but is also oxidized to nitrite and then to nitrate. The nitrite peaks at mile 80, and the nitrate peaks slightly farther downstream from there. Thus, nitrogen from Blue Plains is detectable in one form or another for 30 miles from the discharge. Phosphate, likewise, was detectable from mile 90 down to mile 60. However, unlike the nitrogen forms, phosphate increased again in the turbidity maximum region of the river, possibly because of release from the sediments (Boynton et al. 1980). The location of the turbidity maximum, a region where sediment and associated phosphate is continually resuspended, is shown in Figure 11, between river miles 55 and 65.

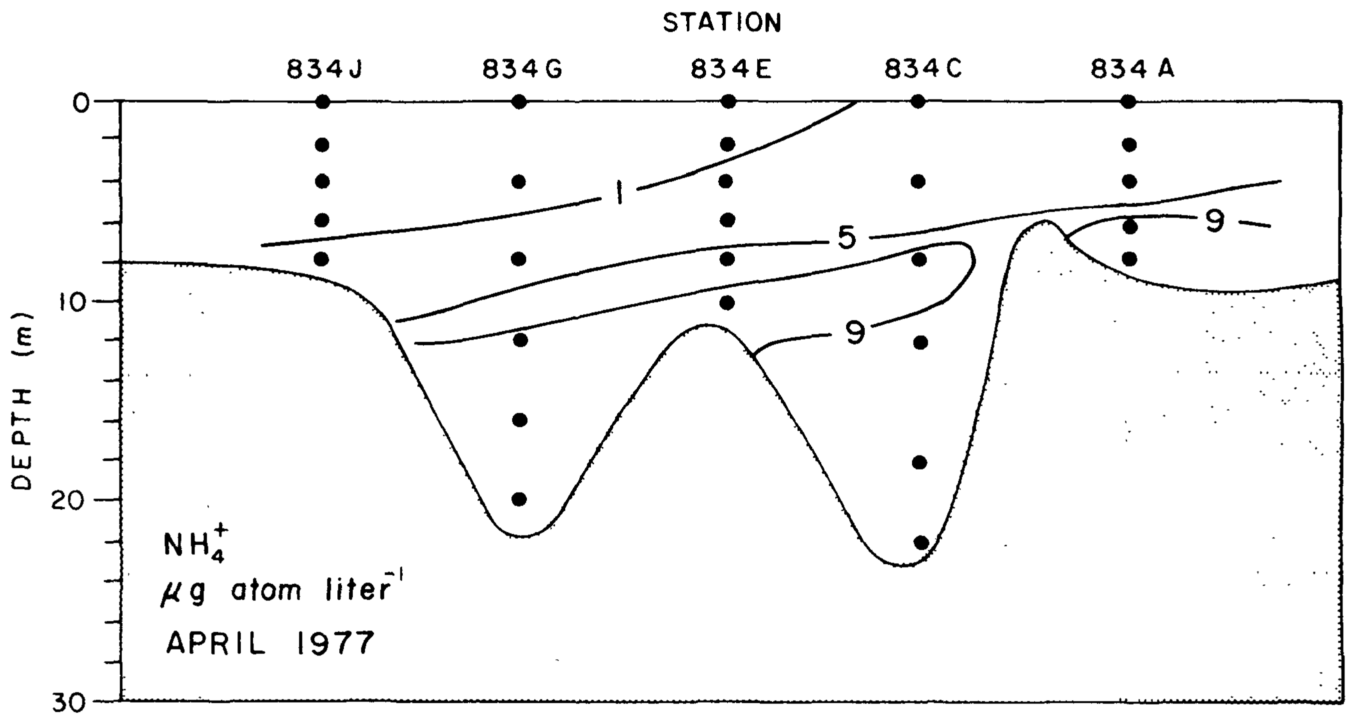

Figure 6a. Distribution of ammonium along a transect at 38°34'N, showing possible lateral transport of ammonium.

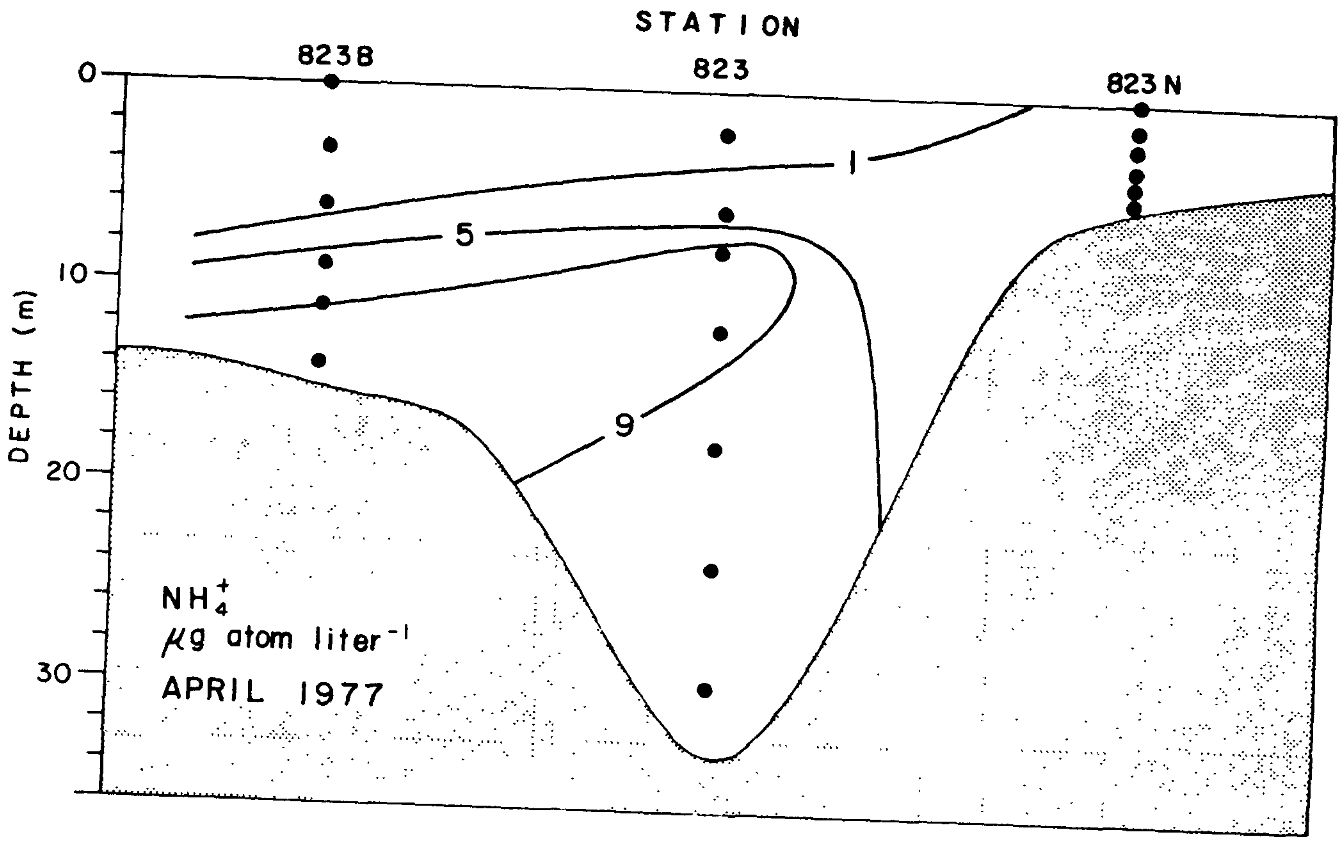

Figure 6b. Distribution of ammonium along a transect at 38°23'N, showing possible lateral transport of ammonium.

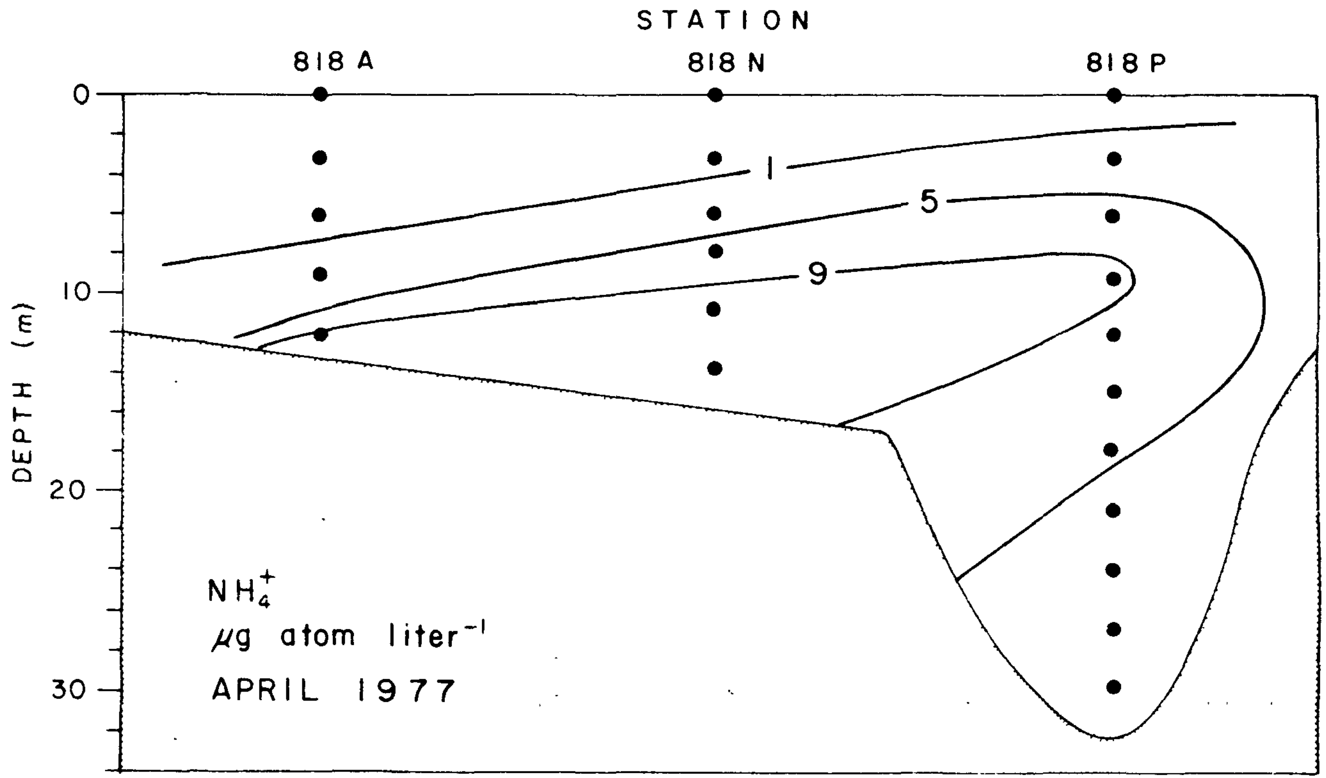

Figure 6c. Distribution of ammonium along a transect at 38°18'N, showing possible lateral transport of ammonium.

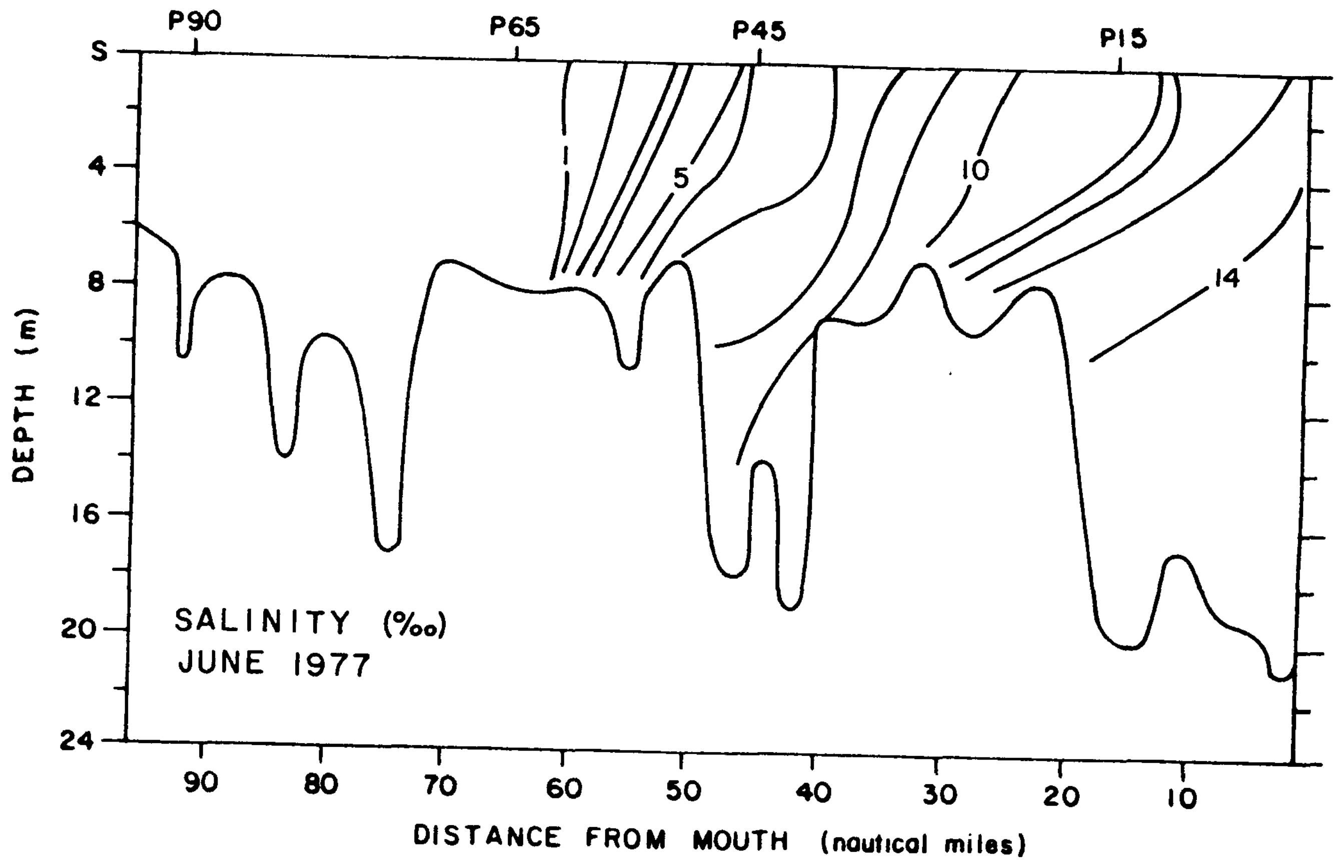

Figure 7. Salinity in the Potomac River.

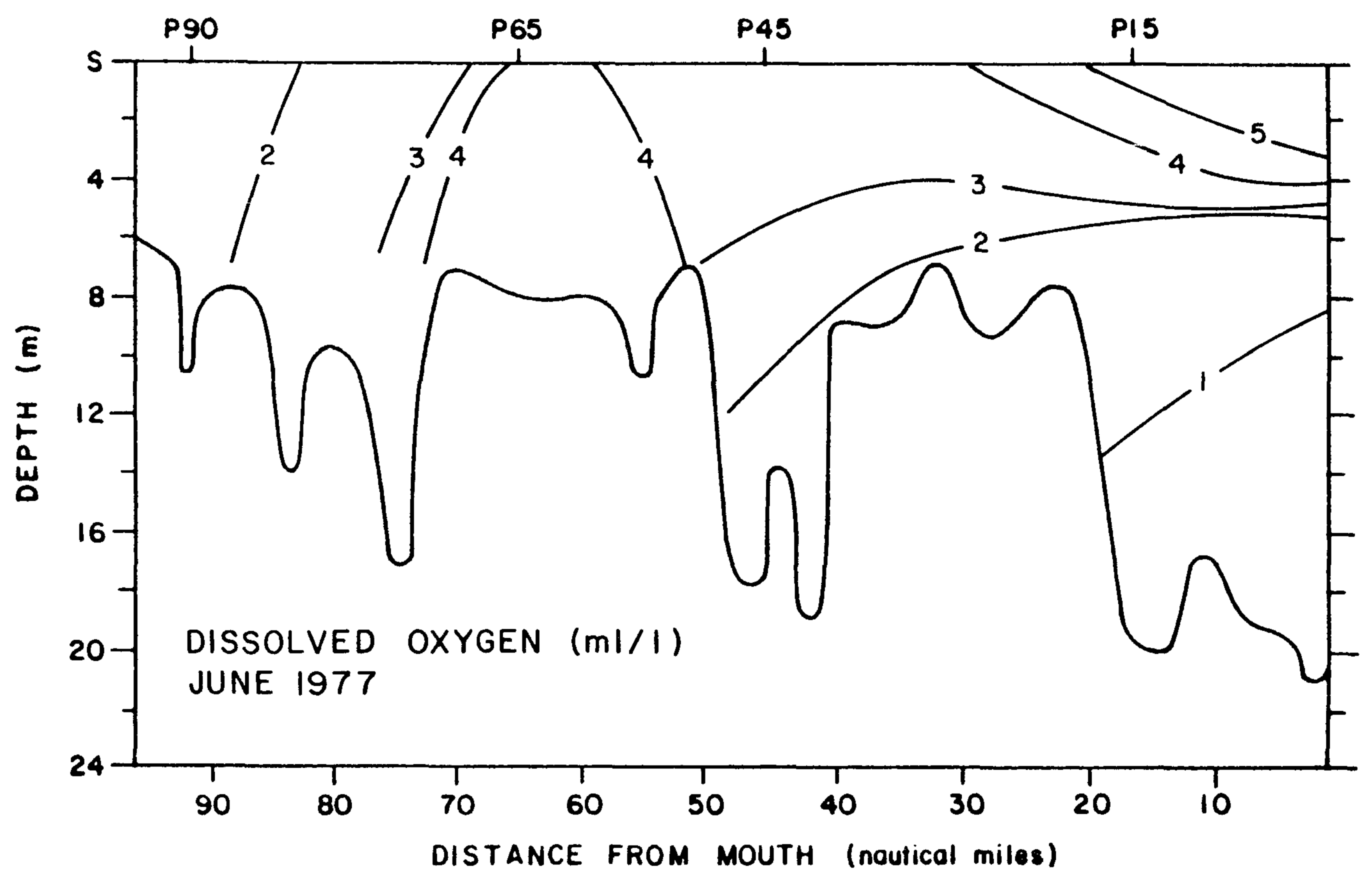

Figure 8. Dissolved oxygen in the Potomac River.

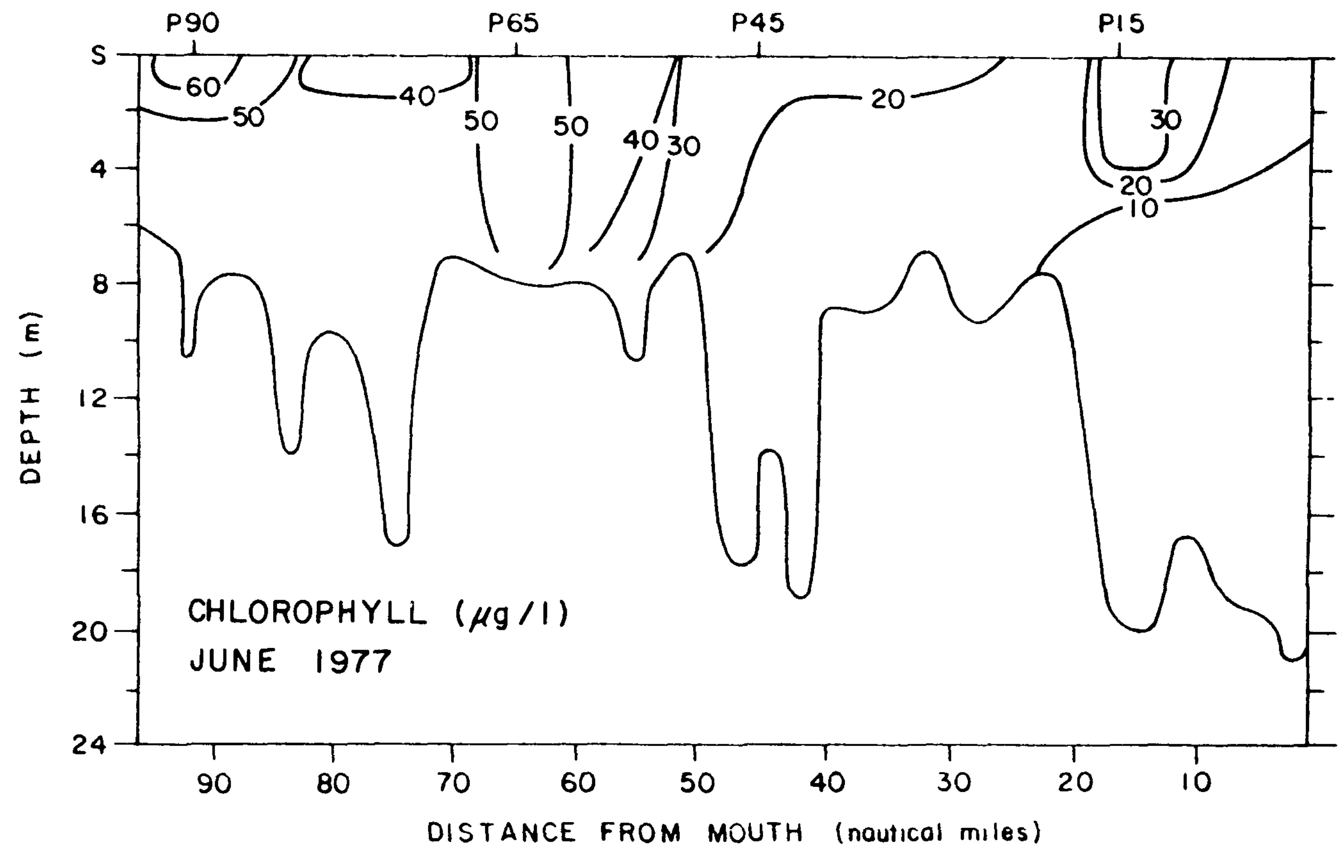

Figure 9. Chlorophyll a in the Potomac River.

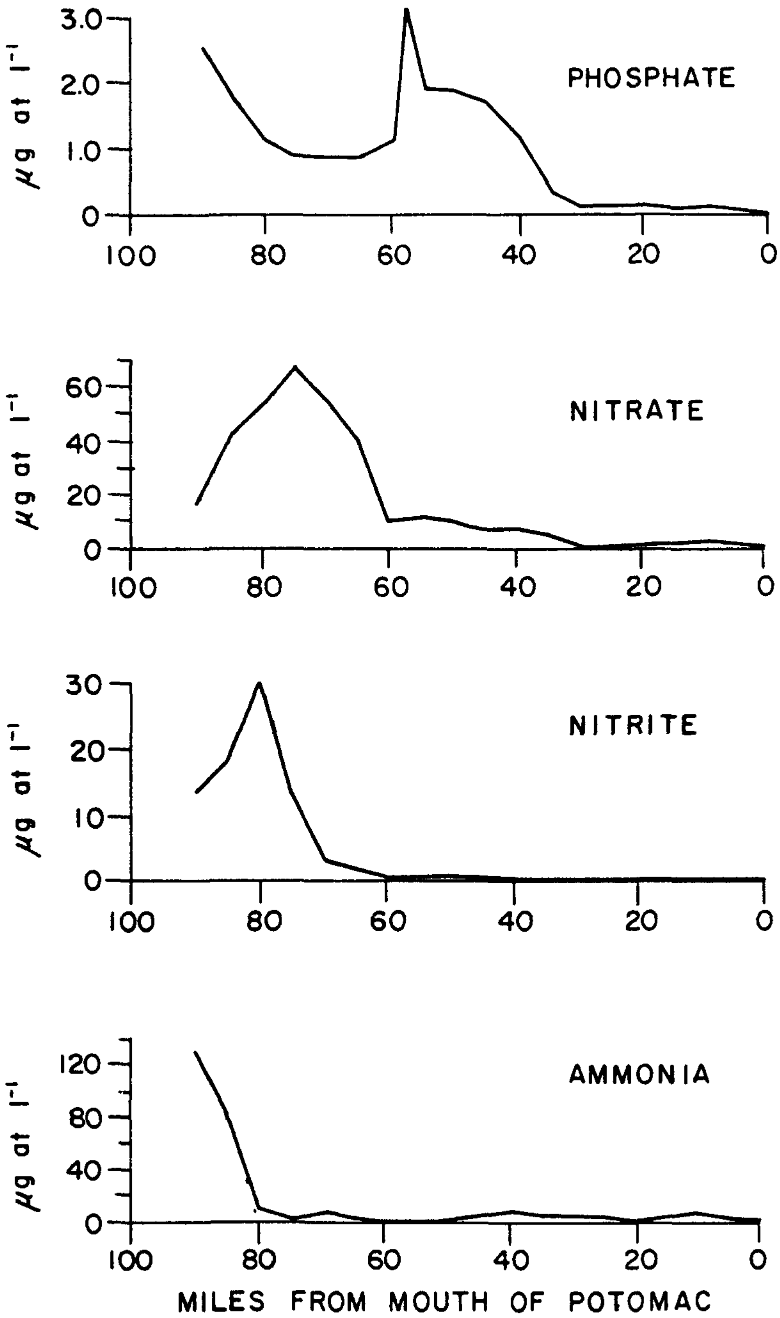

Figure 10. Nutrients in surface water of the Potomac River.

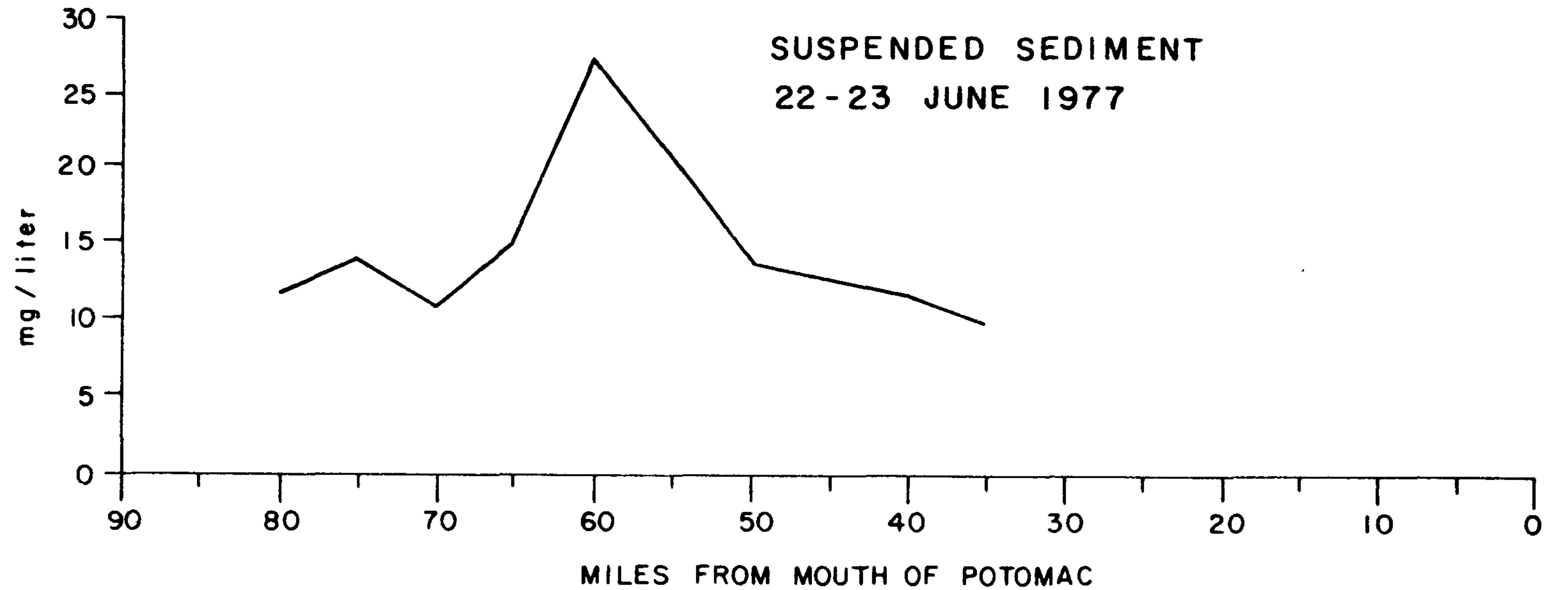

Figure 11. Suspended sediment in the Potomac River.

FACTORS AFFECTING PHYTOPLANKTON GROWTH AND PRODUCTIVITY

Background: The Requirements of Phytoplankton

Phytoplankton photosynthesis, or primary productivity, requires adequate light and nutrients. These nutrients stimulate phytoplankton growth which, in turn, supports the remarkable productivity characteristics of Chesapeake Bay. Phytoplankton growth occurs as a result of photosynthesis, the process by which phytoplankton use light to produce energy-rich molecules that are used, in light or dark, to convert carbon dioxide to carbohydrates and oxygen.

Concept: Nutrient Limitation

The nutrition of higher trophic-level organisms in Chesapeake Bay ultimately depends on the phytoplankton and, to a lesser extent, on the macrophytes. The availability of nutrients, in turn, regulates plant standing crop, or biomass. The notion that standing crop could be regulated by a single factor was expressed in 1840 by Justis Liebig, who stated that "growth of a plant is dependent on the amount of foodstuff which is presented to it in minimum quanity" (quote from Odum 1971).

In other words, a plant needs a certain amount of nitrogen and a certain amount of phosphorus to grow at the maximum rate. If nitrogen is scarce, but phosphorus is abundant, growth is nitrogen-limited. If the reverse is true, the plant is phosphorus-limited. For maximum growth, both elements in their correct proportions are needed. Nutrient regulation of phytoplankton standing crop in Chesapeake Bay is established by the natural annual cycles in nutrient inputs from the rivers, direct land runoff, and the sediments. Minimum phosphorus availability occurs during spring and fall in the main portion of the estuary. Thus, natural cycles cause the limiting nutrient to change over the year.

Although photosynthesis requires nutrients, it appears that the specific rate of primary productivity of the biomass is not directly influenced by nutrient concentrations, because high productivity often coincides with low inorganic phosphorus and nitrogen concentrations in the euphotic zone. The rate may be influenced more by intracellular nutrient pool size and nutrient supply rates from external sources (tributaries, sediments, recycling) than by extracellular concentration.

Although biomass is nutrient-regulated in the sense of Liebig's statement, the specific rate of primary productivity of the biomass is not controlled by the nutrient concentrations found in the water. Instead, primary productivity is directly regulated by intracellular nutrient pools, and, of course, light. The rate of internal nutrient replacement is controlled primarily by the rate of nutrient supply to the environment, the recycling rate. Even under conditions of nutrient limitation of phytoplankton biomass, recycling supports a healthy, productive ecosystem.

Nutrients recycled in the water column may be considered as "regenerated." Those recycled in the sediments and those entering from the land may be considered as "new," since they are being added to the water column. Comparison of nutrient flux estimates with productivity indicates that "new" nutrients could significantly support phytoplankton productivity north of 39°N latitude (Chesapeake Bay Bridge) because of their greater relative availability. "New" nutrients are less available and, thus, have diminished importance to the south, where recycling seems to be the dominant process providing nutrients for phytoplankton primary productivity. "New" nutrients provided primarily by benthic biological and chemical activity may have the dominant role in supporting phytoplankton biomass increases in Chesapeake Bay as a whole.

In summary, the potential for phosphorus limitation in the tidal fresh regions of Bay tributaries exists throughout the year. This is because blue-green algae, common in fresh water, can utilize nitrogen gas, so that nitrogen cannot become the limiting nutrient. The term "potential" is used, because light may also limit biomass in high turbidity regions. Phosphorus is limiting to biomass in the main portion of the Bay during spring and fall. Nitrogen is limiting in summer. In winter, light or phosphorus may be the limiting factor depending on inflow and cloud cover.

Concept: Regulation of photosynthesis by light

In the presence of adequate nutrients, photosynthesis is controlled by both light quantity and quality. The net rate of photosynthesis is not constant, even during daylight hours. Different organisms seem to maximize photosynthetic efficiency during different times of the day. This means that results of experiments designed to determine the photosynthetic rate are influenced by light quantity, by light quality as effected by scattering and absorption in the water, and by time of day.

How Phytoplankton Respond to Nutrients

Occasionally, the production of phytoplankton biomass sufficiently exceeds its loss through sinking, grazing, and flushing to permit algal biomass accumulation in the main portion of the estuary (Loftus et al. 1972). But most of the year, phytoplankton standing crop falls in the range of five to 30 ug chl a L^{-1}, with the higher numbers in the upper layer during cold weather. Phytoplankton nutrition, as indicated by particulate C:N:P atom ratios, reflects seasonal changes in nutrient dynamics.

Concept: Particulate Nutrient Ratios

Well nourished phytoplankton contain optimum amounts of the nutrient elements, carbon, nitrogen, and phosphorus. Field and laboratory experiments indicate that the ratio of atoms of these elements under optimum conditions, is approximately 106 atoms carbon to 16 atoms nitrogen to one atom phosphorus. This specific configuration is called the Redfield ratio after the oceanographer who first suggested it as a characteristic of well-nourished phytoplankton cells (Redfield et al. 1963). Departures from the Redfield ratio provide information about depleted intracellular nutrient stores.

Particulate samples collected on 12 Chesapeake Bay Institute (CBI) cruises in the main Bay during 1972 to 1976 give particulate N:P atom ratios in spring usually between 30:1 and 45:1, suggesting phosphorus deficiency with respect to phytoplankton nitrogen. Some of these data are depicted in Figure 12. The rate of organic phosphorus utilization by phytoplankton, a response to inorganic phosphorus deficiency, also peaks in spring, supporting this interpretation (Taft et al. 1977). In summer most N:P ratios drop below 30, and organic phosphorus degradation is reduced. The atom ratio of ammonium nitrogen to phosphate-phosphorus in the deep water is about four to one (Taft and Taylor 1976b), and the soluble nitrogenous nutrients in the euphotic zone are insufficient to allow the biomass of the phytoplankton present to double (McCarthy et al. 1975). This evidence suggests that the biomass of phytoplankton is controlled by nitrogen in summer -- a shift from the spring situation of biomass control by phosphorus.

In a nutrient-limited system, phytoplankton biomass is controlled by the concentration of nutrients (assuming grazing, flushing, and sinking do not occur). Phytoplankton productivity, however, appears to be fairly independent of nutrient concentration. Thus, although the rate at which an individual phytoplankter is productive is relatively independent of nutrient concentration, increase in population biomass does depend primarily on nutrient concentration.

A relationship between phytoplankton productivity and nutrient concentration in the Bay is fairly difficult to demonstrate for several reasons. First, the highest production rates coincide with very low extracellular concentrations of one or more nutrients. In contrast to Fournier's (1966) results for the lower York River, adding nutrients singly, or in combination usually failed to stimulate primary productivity in experimental incubations with natural Chesapeake Bay phytoplankton assemblages (Taylor and McCarthy 1972). Second, phytoplankton exhibit preferences for certain forms of nutrients over others.

Concept: Nutrient Preferences

It is energetically advantageous for a cell to take up reduced nitrogen in the ammonium form, because it can be incorporated into amino acids and proteins directly. At ammonium concentrations below a threshold value, usually 1.0 to 1.5 ug-at L^{-1}, oxidized nitrogen as nitrite and nitrate are taken up as well (McCarthy et al. 1975, 1977). The cell must expend more energy to reduce these ions to ammonium, but the expenditure is justified. Similarly, phytoplankton incorporate orthophosphate alone until concentrations fall below threshold. Then cells degrade simple organic phosphates to supplement cellular phosphorus nutrition. Convincing evidence indicates that, because of phytoplankton preferences, much of the nitrate entering in spring from the Susquehanna River passes through the upper Bay, because ammonium concentrations are above threshold, to be utilized in the lower Bay where ammonium concentrations are below threshold. The abundance of nitrogen allows orthophosphate concentrations to drop below threshold, and degradation of simple organic phosphates to be stimulated.

Ammonium is selected preferentially over nitrate. Although the orthophosphate ion is generally the phosphorus source preferred by phytoplankton, some species will grow equally well in culture with an organic mono-ester as the phosphorus source (Kuenzler 1965; Taft, unpublished data). However, like orthophosphate, mono-ester concentrations

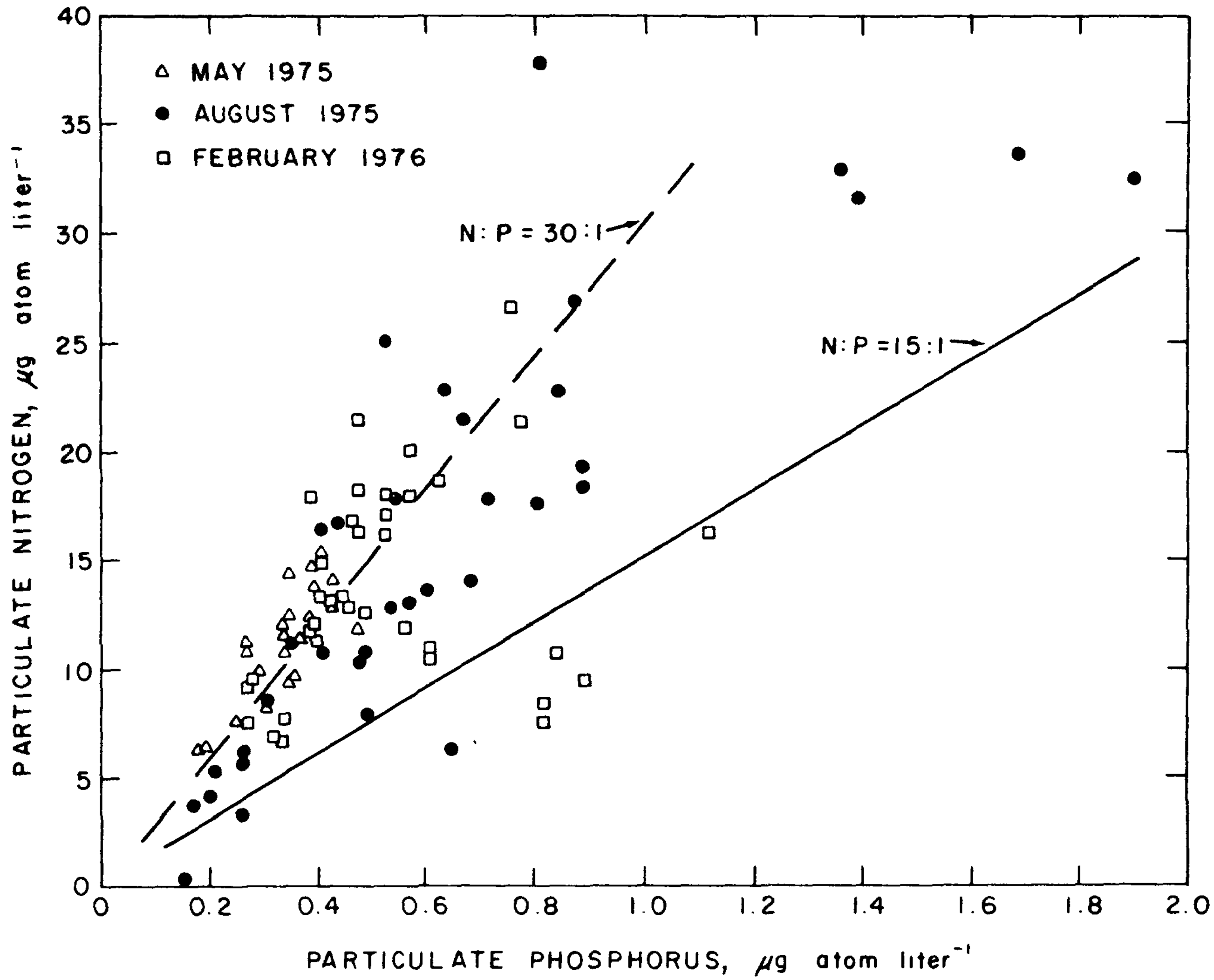

Figure 12. Nitrogen and phosphorus content of particles in Chesapeake Bay on three cruises from the Bay Bridge to the mouth. The solid line is the ratio for well nourished phytoplankton. The dashed line bisects the bulk of the data at an N:P ratio of 30:1.

usually approach detectability limits; no transition concentration (as exists for preferred nitrogen species) is obvious for preferred phosphorus species.

Third, if nutrient concentrations do exert a regulating influence on carbon fixation, it most likely occurs intracellularly near the enzyme systems affecting final nutrient utilization. As one nutrient molecule is removed by enzyme activity from the substrate pool, it could be replaced from the dilute extracellular medium to retain necessary high concentrations near the enzyme. This hypothesized sequence leads to the notion that coupling between carbon incorporation rates and nutrient incorporation rates should be very close; this does occur under conditions of high productivity (growth rate) and low nutrient concentrations in chemostats.

When growth rates are low, carbon fixation and nutrient uptake may become uncoupled. Eppley and Renger (1974) observed (for the diatom Thalassiosira pseudonana) increased maximum uptake velocities for nitrate and ammonium as growth rate decreased. Orthophosphate uptake by P deficient phytoplankton is frequently much more rapid than short-term growth in culture (Ketchum 1939) or photosynthetic rate in Chesapeake Bay (Taft et al. 1975). The potential for one phytoplankter to incorporate nutrients rapidly at low external concentration leads to the conclusion that nutrient uptake should never be concentration-limited (Kuenzler and Ketchum 1962). Inability to demonstrate continuous, close-coupling between carbon fixation and nutrient uptake, and elimination of the phosphate and ammonium uptake steps as productivity regulating factors, also complicates direct demonstration of nutrient regulation of phytoplankton primary productivity.

These observations lead us to conclude that neither ambient nutrient concentrations, nor increased uptake potential resulting solely from elevated nutrient concentrations, have Bay-wide significance in regulating open water phytoplankton productivity. Therefore, static measurements of nutrient concentrations and other water quality parameters do not convey enough information about the dynamic events taking place. Optimal water quality management requires information about processes and their rates.

How Phytoplankton Respond to Physical Processes

Concept: Phytoplankton are Distributed Unevenly in Space and Time

The term "phytoplankton" implies a plant cell that has limited mobility; it is transported more by water movement than by swimming. The most advantageous use of swimming by phytoplankton is exhibited by the dinoflagellates that can travel vertically. In the two-layered estuary, they have the capability to move from the seaward-flowing surface layer to the landward flowing deep layer and, thus, stay in the estuary. They can also migrate from nutrient-poor, surface water to nutrient sources in the deep water or sediments. Weaker swimming organisms and those, such as diatoms, which don't swim at all, depend on buoyancy and water movement to keep them in a suitable environment.

The interaction of phytoplankton buoyancy or swimming with water motion produces a spatially patchy distribution of organisms. The upward motion of cells against downward-flowing water can result in the accumulation of organisms near the surface of a so-called frontal region. Growth of surface organisms can be stimulated by the upward motion of nutrient-rich deep water to the surface, so that biomass increases in one area relative to nearby regions where such motion does not exist.

Spatial distribution is also influenced by salinity of the water. Some species can adapt to a wide range of salinities and may be found throughout the estuary. But many riverine and marine forms have very narrow salt tolerances so their occurrence is limited.

The temporal distribution of phytoplankton species depends primarily on water temperature; some are considered summer species, and others are winter species. If both temperature and salinity regimes are acceptable, the organisms survive long enough to be transported by circulation.

The use of phytoplankton distributions as indicators of water movement has been demonstrated as a useful technique in Chesapeake Bay. Moreover, the significance of coupling between phytoplankton ecology and physical processes in the estuary has been clearly established for one dinoflagellate species (Tyler and Seliger 1978). This research reemphasizes the necessity of examining estuarine processes in detail to understand the system. Further, research indicates that the tributaries are very important sources of phytoplankton that may achieve local numerical dominance, and in some cases, biomass dominance in the main portion of Chesapeake Bay. Thus, the ecology of these organisms is closely coupled to physical processes in the estuarine system.

Movement of phytoplankton through the estuary can be roughly estimated using a box model with particulate organic carbon (POC) representing the phytoplankton. Figure 13 shows the flux estimates for (a) February, (b) May and (c) August 1975, and (d) February 1976. Units are 10^5 ug atom C sec^{-1}. Net POC flux was greater during the two winter periods. Vertical transport of phytoplankton was dominated by upward movement over much of the Bay. This upward movement was due to minimum stabilization of the water column, which created high potential for mixing both salt ions and particles upward from the deep layer. The source and sink terms, shown in small boxes, represent nonconservative gains and losses of POC such as growth, grazing, sinking, and disruption. The net values of these processes were also higher over most of the Bay during winter than during spring or summer. This information can help locate areas and times of high activity that would, subsequently, increase phytoplankton biomass.

Kinetic Measurements of Nutrient Uptake by Phytoplankton

Environmental biologists began making kinetic measurements of nutrient uptake by phytoplankton to obtain physiological information and predict changes in species composition from changes in nutrient concentrations. Nutrient uptake by phytoplankton proceeds at rates that are concentration-dependent. Uptake rate increases with increasing concentration up to some maximum rate, beyond which it is constant regardless of concentration (Figure 14). The relation between substrate concentration and uptake rate is usually expressed mathematically as a rectangular hyperbola. Two characteristic parameters of this form are the half saturation value (K_s) and the maximum uptake velocity (V_{max}). K_s is the substrate

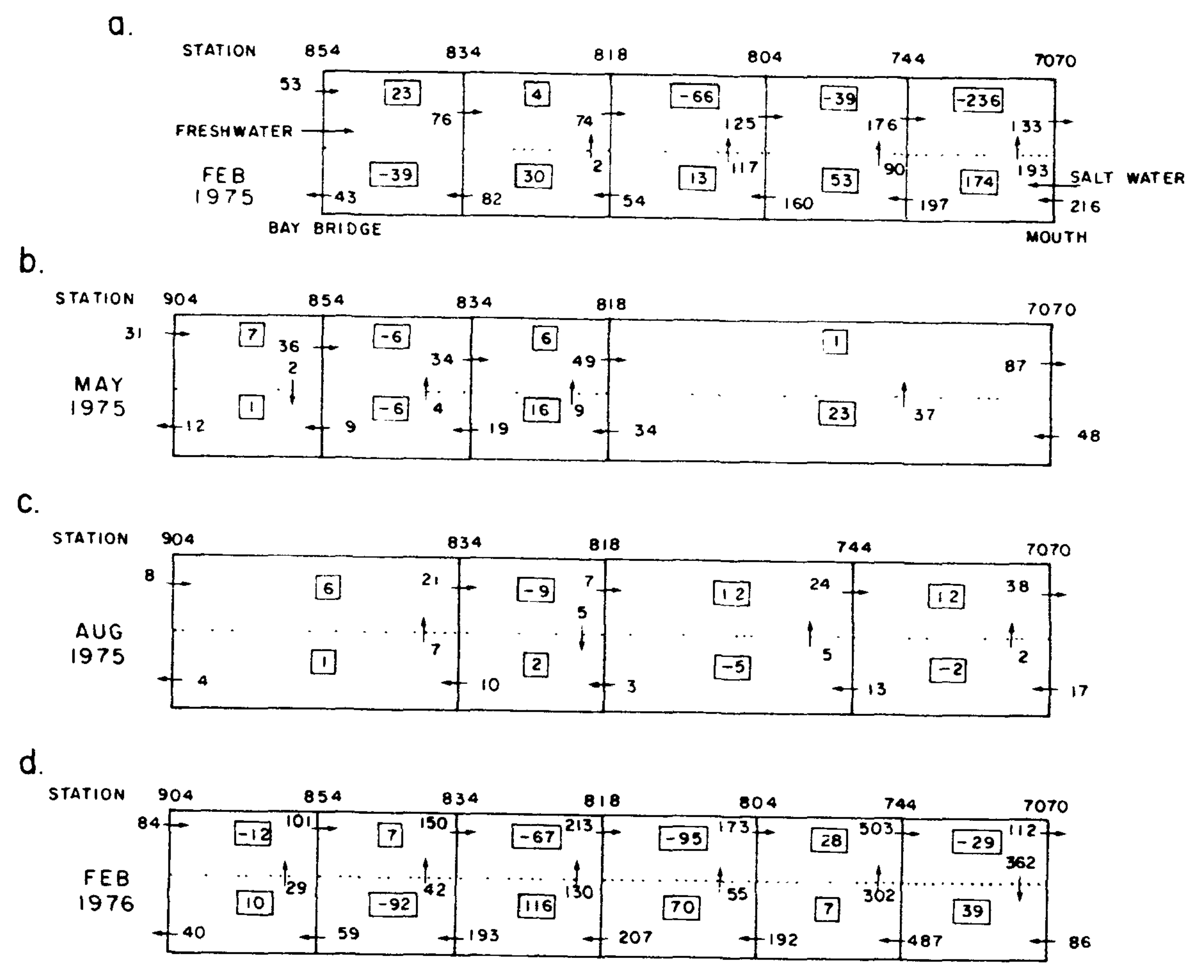

Figure 13. Flux of particulate organic carbon through Chesapeake Bay over seasons. (a) February, (b) May, (c) August, and (d) February (1976). Numbers in boxes reflect net amount of carbon in the water column in either the upper (above dotted line), or lower (below dotted line) estuarine layer. Arrows indicate direction of carbon flux. Units of 10^5 ug atom C sec^{-1}.

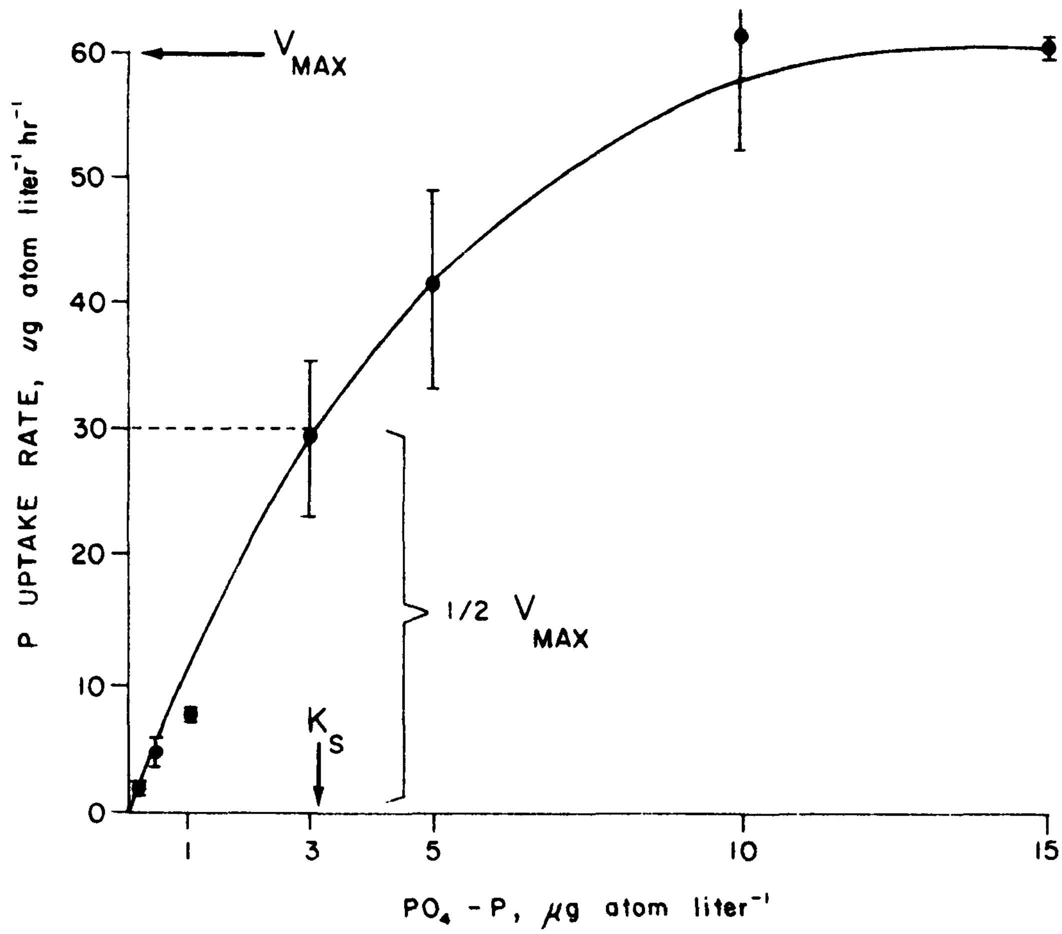

Figure 14. Phosphate uptake kinetics for a natural phytoplankton population containing primarily one dinoflagellate species.

concentration at which uptake velocity reaches one-half the maximum rate. In this way, nutrient uptake is treated as an analog of enzyme kinetics, reflecting the participation of enzymes and carrier molecules in the uptake process.

Concept: Kinetic Parameters are not Constant

The kinetic parameters K_s and V_{max} are coefficients used in mathematical models. However, they should not be considered "constants" because they are subject to variation, even within species, depending on environmental conditions, the organisms' recent history, and the types of organisms present in a natural population. Kinetic parameters determined with pure cultures can be employed in models of natural systems if the modeler recognizes that a factor of ten range in the values is not unusual. Table 1 shows the range of K_s and V_{max} values observed in Chesapeake Bay and Potomac River. These ranges indicate differences in phytoplankton physiology. As the table shows, nitrogen values can vary by a factor of two or more.

TABLE 1. HALF-SATURATION VALUES (K_s) AND MAXIMUM UPTAKE VELOCITIES (V_{max}) FOR NUTRIENTS IN THE CHESAPEAKE BAY ESTUARINE SYSTEM

Nutrient	K_s ug atom·L	V_{max} ug atom chl $\underline{a}$ h^{-1}
Chesapeake Bay		
Phosphate	0.09 to .172	0.004 to 0.160
Ammonium	1 to 2	---
Nitrate	2 to 4	---
Potomac River		
Phosphate	0.2 to 0.4	0.0005 to 0.0015
Ammonium	1.5 to 1.7	0.003 to 0.017
Nitrate	1.2	0.005 to 0.039

K_s and V_{max} are often considered constants for a particular phytoplankton species for mathematical modeling purposes and for comparing one species with another. K_s is an indicator of the affinity between the nutrient and the cell's uptake system; the smaller K_s, the greater the affinity. It has been a popular concept that a species with a lower K_s can dominate when nutrient concentrations are low because of greater affinity for the nutrient; a species with higher K_s can dominate only when nutrients are high. As a generality, this concept is acceptable. However, K_s is not a true constant. Modifications in the uptake system or the membrane to which it is bound on the cell alter K_s. Such modifications may be related to the relative amounts of saturated and unsaturated lipids in the cell membrane, to the cell's immediate history, and to the intracellular nutrient supply. Similarly, V_{max} is not a true constant. It may be changed by membrane alterations or changes in the number of uptake sites per cell. Since these kinetic parameters are not constant, predictions of species shifts with changes in nutrient concentrations have had limited success. At best, shifts between green and blue-green algae in fresh waters can be described based on nutrient loading ranges. Resolution beyond this remains to be developed.

It is possible, however, to describe a hypothetical relationship between nutrients and commercial species based upon results from culture experiments with a variety of organisms. From these experiments, it is known that not all phytoplankton species have equal nutritional value for the planktivores that graze them. High diversity of phytoplankton species, in a natural population favors a balanced diet for the grazers. Modifications of the nutrient regime, which cause species shifts and reduce population diversity, may increase the potential for deficiencies in the grazer diet. Thus, the yield of filter-feeding commercial species, such as oysters and menhaden, could be influenced indirectly by nutrient inputs to the system.

Summary

In summary, the best Bay management requires an understanding of the major processes affecting growth and reproduction of phytoplankton because the ecology of phytoplankton is closely coupled to physical processes of the estuary. These influences include the effect of light, nutrients, and physical and chemical processes on phytoplankton, and how quickly phytoplankton assimilates nutrients.

In the presence of adequate nutrients, photosynthesis is regulated by light. Specifically, the quality and quantity of light affect the rate of photosynthesis in phytoplankton. However, in a nutrient-limited system, such as the Bay, the presence of P or N in the smallest amount regulates phytoplankton standing crop. Phosphorus is limiting in the main Bay in spring and fall, with N limiting during summer. In winter, light or P can be the limiting factor. The availability of these nutrients is controlled by the recycling rate, or the rate of nutrient supply to the environment. "New" nutrients, or those recycled in the sediments and entering by land, provide the major source to phytoplankton and probably are the causes of increases in biomass of Chesapeake Bay as a whole.

The uneven distribution of phytoplankton in the Bay results from their responses to physical and chemical processes. Mobility of some phytoplankton species enables them to overcome circulation patterns. They can move vertically between layers of the Bay and migrate to nutrient-rich areas. Circulation of the Bay brings, to certain areas, upward-moving, rich waters; in these areas, growth of surface phytoplankton is stimulated. Salinity limits the distribution of some phytoplankton, but others dependent on water temperature will only persist at certain times of the year.

Measuring the rate of nutrient uptake by phytoplankton can indicate species shifts in phytoplankton and consequences on organisms higher in the food chain. A certain species of phytoplankton with a slow uptake rate will produce less biomass in a given time than a phytoplankton with a faster uptake rate. The latter species would dominate, perhaps causing a bloom. Diversity in phytoplankton favors a balanced diet for grazers, which may ultimately influence the yield of filter-feeding Bay resources such as oysters and menhaden.

SECTION 3

NUTRIENT CYCLING

INTRODUCTION

The flow rate of nutrient molecules through remineralizing processes to the inorganic forms may be an important factor regulating productivity. These recycling processes include (1) grazing of the phytoplankton standing crop by macro- and micro-zooplankton, followed by nutrient release through excretion or heterotrophic bacterial activity; (2) activity of free-living aquatic bacterial heterotrophs; (3) nutrient release from the sediments by both biological and chemical processes; and (4) nutrient transport by physical processes into contact with phytoplankton able to assimilate the nutrient.

The basic elements in a living system are carbon, oxygen, hydrogen, nitrogen, and phosphorus. Carbon, oxygen, and hydrogen are readily available to plants in the biologically usable compounds, oxygen gas, carbon dioxide, bicarbonate ion, and water. Nitrogen and phosphorus, however, are not so abundant in a useable form. The cycling of these nutrients in surface waters from the dissolved inorganic form to the living form and back to the dissolved inorganic form, is the mechanism by which the photosynthetic conversion of the more abundant elements into metabolically useful compounds is maximized in the aquatic environment.

About 90 percent of the primary productivity in Chesapeake Bay is accomplished by planktonic algae that pass through a 35 um mesh (McCarthy et al. 1974). These organisms are the principal food of zooplankton in the Bay, which become food for higher organisms. Studies of the larger zooplankton in Chesapeake Bay reveal that 50 to 70 percent of the animals caught on 103 um mesh are copepods of the genus *Acartia* (Rupp 1969). *Acartia tonsa* is also abundant in the Patuxent River estuary (Heinle 1966). Copepods of the genus *Eurytemora* are seasonally abundant and are of the same size as *Acartia*.

Previous studies reveal that grazing macro-zooplankton (adult copepods) consumed only about 10 percent of the daily phytoplankton productivity in Chesapeake Bay (Storms 1974). Therefore, the role of micro-zooplankton in grazing phytoplankton and in returning nutrients to the water was examined. It now appears that the most significant role of the micro-zooplankton is to respond, through rapid growth, to graze blooms of phytoplankton that occur periodically in the Bay (Heinbokel, unpublished data). Data are now becoming available on protozoa and metazoa, that are smaller than the common copepods. As a group, the zooplankton inhabiting the estuary south of the Bay Bridge play a major role in regenerating nitrogen and phosphorus to meet the requirements of the phytoplankton on which they feed.

Concept: Nutrient Cycling

Nitrogen and phosphorus are converted from the inorganic, to living, organic forms and back again on varying time scales in the Chesapeake Bay estuary. For simplicity of illustration, here, the time scales are divided into short-term (minutes to weeks) and long-term (months to years). It is also convenient to conceptualize that short-term cycling occurs primarily in the water and the surface sediments and long-term cycling takes place primarily in the deep sediments (more than two cm below the surface).

WATER COLUMN PROCESSES

Respiration

Respiration rate is measured as the rate of oxygen consumption of a water sample incubated in a darkened container. It is possible, by measuring the plankton respiration rate accurately and precisely, to estimate C, N, and P regeneration in the water column, using a suitable respiratory quotient and particulate C:N:P ratio.

Results presented here are from two experiments at natural plankton densities performed by Dr. Eric Hartwig with a sensitive photoelectric oxygen titrator. A suitable respiratory quotient (RQ) must be used to convert oxygen consumption to the amount of organic carbon degraded. The respiratory quotient is the ratio of carbon dioxide produced to oxygen consumed by an organism. Commonly determined RQ values range from 0.27 in an intertidal sand flat to 1.6 for *Chlorella* using nitrate as the nitrogen source (Teal and Kanwisher 1961, Pamatmat 1968). For the purposes of this report, an RQ of 0.85 will be assumed, with the realization that deviation in RQ of ± 0.20 encompasses most RQ measurements found in the literature and yields a ± 25 percent variability, which is within acceptable limits. The incorporation of C:N:P atom ratios, into the calculation, yields estimates of inorganic nitrogen phosphorus regeneration rates. The atomic ratio of Redfield et al. (1963) (106 C to 16 N to 1 P) will be used.

The winter respiration rates given in Table 2 and Table 3 were 63 percent of the summer rates (August). The water temperature difference between February and August was approximately 25°C (77°F). If the respiration rate of the organisms present doubled for each 10°C (50°F) temperature change ($Q_{10} = 2$), the February rates would only be 20 percent of the August rates, other factors being equal. This implies that the thermal regime of Chesapeake Bay exerts a selection pressure on microbial communities so that bacterial species change during the year as the temperature changes. Temperature changes of the magnitude existing in Chesapeake Bay were found by Sieburth (1967) to cause shifts in the thermal types of microbes present in Narragansett Bay, Rhode Island. Thermal selection of bacterial species, adapted to either warm or cold temperatures, may be a factor permitting maximum utilization of organic substrates throughout the year.

TABLE 2. AUGUST RESPIRATION AND REGENERATION RATES FOR TOTAL PLANKTON (TP), PLANKTON PASSING THROUGH 35 um MESH (<35 um) AND PLANKTON PASSING THROUGH 3 um FILTERS (<3 um)

Station	Sample	Respiration rate ug atom O_2 L^{-1} h^{-1}	Estimated N regeneration rate ug atom N $L^{-1}h^{-1}x10^{-1}$	Estimated P regeneration rate ug atom P L^{-1} $h^{-1}x10^{-2}$
904N	Surface TP	4.9	6.3	3.9
	7 m TP	5.2	6.7	4.2
853F	Surface TP	3.4	4.4	2.7
	<3um	2.7	3.5	2.2
	6 m TP	7.1	9.1	5.7
834G	Surface TP	4.6	5.9	3.7
818P	Surface TP	3.6	4.6	2.9
744	Surface TP	4.6	5.9	3.7
	<35 um	4.1	5.3	3.3
	< 3 um	2.3	3.0	1.8
7070	Surface TP	2.4	3.1	1.9
	10 m TP	2.1	2.7	1.7
Potomac Estuary	Surface TP	2.5	3.2	2.0
	< 3 um	1.2	1.5	0.96

TABLE 3. FEBRUARY RESPIRATION AND REGENERATION RATES FOR TOTAL PLANKTON SAMPLES

Station	Depth	Respiration rate ug atom O_2 $L^{-1}h^{-1}x10^{-3}$	Estimated N regeneration rate ug atom N $L^{-1}h^{-1}x10^{-1}$	Estimated P regeneration rate ug atom P L^{-1} $h^{-1}x10^{-2}$
904N	2m	2.1	2.7	1.7
	2m	2.5	3.2	2.0
	11m	0.85	1.1	0.68
834G	2m	3.3	4.2	2.7
	9m	1.3	1.7	1.0
804C	4m	2.3	3.0	1.9
	20m	1.4	1.8	1.1
744	2m	1.7	2.2	1.4
	10m	1.3	1.7	1.0
7070	1m	3.5	4.5	2.8
	10m	2.8	3.6	2.3
Calvert Cliffs	Intake	1.3	1.7	1.0
Nuclear Power Plant	Discharge	2.2	2.8	1.8

An important aspect of the water-column-nutrient-regeneration rate concerns its coupling with the nutrient supply required for primary productivity. Estimates of upper Bay productivity values for August [4 ug atom $C(L \cdot h)^{-1}$] and February [0.6 ug atom $C(L \cdot h)^{-1}$] (Taylor, personal communication), with a C:N:P assimilation ratio of 106:16:1, yield a requirement for 0.6 ug atom N $(L \cdot h)^{-1}$ and 0.04 ug atom P $(L \cdot h)^{-1}$ in August and a requirement for 0.09 ug atom N $(L \cdot h)^{-1}$ and 0.005 ug atom P $(L \cdot h)^{-1}$ in February. Table 2 and Table 3 show estimates of regenerated N and P, based on respiration measurements (Taft et al. 1980). These data indicate that respiration could regenerate most of the nutrient requirement for the upper Bay (stations 904N, 853G, 818P, 804C) in August, and an excess of nutrients in February. As a result, addition of further nutrients in August would increase biomass, but addition of nutrients in February would result in nutrient accumulation in the water column.

Grazing

Grazing is the process by which herbivores, such as copepods, consume primary producers (phytoplankton). The grazers in Chesapeake Bay span the range from small ciliates, to rotifers and copepods, all the way up to crustacean and fish larvae, and adult planktivorous fish such as menhaden. The ecological role of these grazers is to transfer the organic material and energy fixed by the phytoplankton through the food web. The grazers themselves are consumed by higher predators such as the carnivorous fishes, waterfowl, and humans. Grazing keeps estuaries in balance by restricting phytoplankton populations.

However, not all of the primary production is assimilated into animals. Some nutrients are released back into the water, directly by excretion, or indirectly, by bacterial degradation of dead cells or animal fecal material. In this way, the grazers help keep the estuary productive by grazing the phytoplankton standing crop and supplying nutrients for continued phytoplankton growth (Table 4). Thus, nutrients entering the estuary are distributed throughout the food web and may be cycled through the planktonic ecosystem several times each year. One of the major goals of biological studies in Chesapeake Bay is to quantify recycling rates, including the contributions from grazers.

TABLE 4. THE MAJOR PHYTOPLANKTON GRAZERS AND PERCENTAGE OF DAILY PHYTOPLANKTON PRODUCTION USED

Animal	percent daily phytoplankton production used
Copepods	up to 15
Microzooplankton	15
Other	70
larval stages of small biota	
planktivorous fish	

Copepods--

Copepods are small crustaceans inhabiting the estuary. Common species are Acartia, Eurytemora, Temora, and Centropages. Copepods have a somewhat complex life cycle. Like most crustaceans, they exhibit several life stages from nauplius to copepodite to adult. All of these stages graze phytoplankton. The grazing may be accomplished by encounter feeding, active hunting of single cells, and filter feeding. The adults have specialized feeding appendages that sweep through the water, directing the phytoplankton cells to the mouth. Some copepods seem to graze selectively upon particular size cells, especially if one size range contains a large fraction of the standing crop.

The copepods found in Chesapeake Bay, particularly the adults, have been fairly well studied. The adults usually number from one to ten per liter in the main portion of the estuary. They graze one to 15 percent of the daily phytoplankton production. Less is known about grazing by early life stages but, by analogy to studies of oceanic copepods, it is accepted that naupliar stages may graze three to five times the adult rate per unit of body weight.

Nutrient cycling rates by copepods can be estimated by assuming 30 percent assimilation efficiency, 60 percent incorporation into fecal material, and 10 percent direct excretion. If copepods grazed 10 percent of the daily phytoplankton production, three percent of the daily production would be assimilated into copepod tissue (30 percent of 10 percent), six percent would be released as particulate fecal material, and one percent would be directly excreted. Nutrients would be similarly distributed, with about one percent of the phytoplankton nitrogen and phosphorus returned directly to the water, and about one-half of the six percent fecal nutrients returned by bacterial activity.

It is clear, by comparing phytoplankton growth with copepod recycling of nutrients, that copepods are a small component of the nutrient cycling system. The phytoplankton grow and divide about once every one or two days in spring, summer, and fall, but the phytoplankton standing crop does not double each day, indicating that the loss due to grazing approximately equals the phytoplankton growth rate. Since copepods are only eating one percent to 15 percent of the daily production, other organisms must consume the remaining 85 to 99 percent.

Micro-Zooplankton--

The micro-zooplankton are those grazers whose size approaches that of the phytoplankton cells. The ciliates and small rotifers may be included in this group. In addition to size similarity, the ciliates have growth rates and generation times similar to phytoplankton, whereas rotifers and copepods have long generation times compared to the phytoplankton.

Less is known about micro-zooplankton abundance distribution in Chesapeake Bay. Recent studies at CBI reveal that ciliates consume about 15 percent of the daily production Bay-wide. Therefore, their contribution to grazing pressure and the recycling of nutrients is probably only slightly greater than that of the copepods. Experiments conducted as part of the Chesapeake Bay Program (CBP) indicate that nitrogen is recycled by micro-zooplankton at the rate of about 0.05 ug atom NH_4-N $L^{-1}h^{-1}$. This represents about 10 percent of the phytoplankton requirement for nitrogen.

Bacterial Activity

Bacteria are heterotrophic organisms that are probably numerically dominant in the estuary. Bacterial abundance is thought to be one to ten million cells per milliliter of water. A major role of these organisms is to metabolize organic material during which some inorganic carbon, nitrogen, and phosphorus are recycled. These metabolic processes consume oxygen and, at times, may be the dominant oxygen-consuming process.

Important groups of bacteria involved in nutrient cycling are *Nitrosomonas* and *Nitrobacter*. The genus *Nitrosomonas* oxidizes ammonium to nitrite. *Nitrobacter* then further oxidizes nitrite to nitrate. These reactions probably occur in oxygenated sediments year round, but are most conspicuous during the late summer and fall in Chesapeake Bay when ammonium-rich deep water is re-oxygenated. The ammonium is rapidly oxidized to nitrite that reaches relatively high concentrations throughout the Bay. The second oxidation step to nitrate has been observed less frequently than the first.

An experiment was conducted under the CBP to specifically examine this phenomenon. Ammonium was oxidized at the rate of 0.05 ug atom NH_4-N L^{-1} h^{-1} by planktonic bacteria. During the process, about one percent of the NH_4-N was converted to gaseous N_2O.

A considerable amount of research has been done on a few kinds of bacteria in Chesapeake Bay, such as the shellfish pathogens, but little has been done quantitating the role of bacteria in nutrient recycling during the winter, fall, and spring. The bacterial contribution to recycling has usually been estimated from oxygen consumption, or obtained by difference calculation rather than measured directly. At best, there are fractionation studies wherein water samples are passed through various size filters, and the oxygen consumption of each fraction is measured. The results of three such experiments are shown in Table 2 for samples passing through a 3 um filter (labeled $<$ 3 um). Based on the respiration of the fraction containing the bacteria, these organisms have the potential to recycle 10 to 100 times more nitrogen and phosphorus than the copepods or the micro-zooplankton. However, under conditions favorable to bacterial growth, bacteria may incorporate nutrients rather than recycle them to the water.

SEDIMENT PROCESSES

The sediments are an integral component of the nutrient cycling system in Chesapeake Bay. Nutrients accumulate in estuaries, because they are incorporated into particles that sink to the bottom. These organic particles may remain on the surface or be mixed down into the sediment by benthic animals. Benthic animals and bacteria degrade the organic particles, incorporating some of the material into their own structure and regenerating some as inorganic carbon (CO_2), nitrogen (NH_4+), and phosphorus (PO_4^{-3}). If regeneration occurs on the sediment surface, the nutrients will be returned directly to the water. However, if regeneration occurs deeper in the sediments, the nutrients may be dissolved in the sediment interstitial waters. This results in interstitial waters having relatively high concentrations, several hundred micromolar, of ammonium and phosphate.

Nutrient Flux

The flux of nutrients out of the sediments is largely dependent on two processes. One is degradation directly on the sediment surface. The other is diffusion of nutrients out of the sediments, based upon the concentration gradient in the interstitial water. Direct release to the water can be estimated from the sediment-oxygen demand, as is done for water-column- regeneration estimates.

Diffusion of nutrients out of the sediment can be calculated from the concentration gradient and the physical characteristics of the sediment. These calculated values are minimum values for nutrient flux out of the sediments. Several other factors, difficult to quantify, have roles in moving nutrients out of the sediment. One of these factors is stirring of the sediments by benthic animals. Another is the lateral diffusion of nutrients into animal burrows, followed by turbulent diffusion or advection up the burrow to the sediment-water interface. A third possible factor is hydrostatic pumping of water as a surface wave passes. Theoretically, large hydrostatic pressure under a wave crest could pump water out of the sediment under an adjacent wave trough, where hydrostatic pressure is less.

Another way to determine nutrient flux from sediments is by placing a chamber on the bottom to isolate a portion of the sediment-water interface. Flux rate is then determined from nutrient concentration changes in the water contained by the chamber. Since this method may also introduce some artifacts, we consider flux rates obtained in this way to be maximum potential values.

During the course of the CBP, nutrient flux from the sediments was studied by both the diffusion and chamber method. Table 5 summarizes the results. For ammonium, the diffusion value is usually about one-fourth to one-half of the measured chamber value. This is because processes at the sediment surface (bioturbation and other biological processes) increase the flux of nutrients over that permitted by diffusion alone. The actual ammonium flux from undisturbed sediments lies between these two values and is probably closer to the higher one. At present, this is the best estimate that can be made from this data.

TABLE 5. AMMONIUM AND PHOSPHATE FLUX FROM CHESAPEAKE BAY SEDIMENTS EXPRESSED AS ug atom m^{-2} h^{-1}. POSITIVE IS OUT OF THE SEDIMENT

	Ammonium		Phosphate	
Location	Diffusion	Chamber	*Diffusion	Chamber**
Worton Creek	50	177	1.1	-4.2
Hart-Miller Island	52	102	0.9	2.3
Sharp's Island	171	455	16.3	0
Kenwood Beach	184	670	15.3	40
Todds Cove	37	410	3.8	16
Gwynn's Island	68	262	5.1	10
Pocomoke Sound	93	430	9.6	-3.2

* TDP
** DIP (Boynton)

In contrast, the phosphate flux values from diffusion calculations and chamber measurements are more similar than the ammonium values, but negative values are sometimes observed. This reflects the uptake of phosphate by sediments, a process which partly results from sorption by sediment particles.

The magnitude of the sediment flux of ammonium and phosphate can be illustrated by comparing the amount of nutrient being added with the total amount already in the water. Consider, for ammonium in the water column 1 $m^{-2} \cdot h^{-1}$, the ammonium concentration in the water would increase by

$$\frac{100 \text{ ug atom} \cdot h^{-1}}{10 \text{ m}^3} = 10 \text{ ug atom} \cdot m^{-3} \cdot h^{-1}$$

or 0.01 ug atom L^{-1} h^{-1} each day. The water would gain 0.24 ug atom $L^{-1} \cdot d^{-1}$. If the total nitrogen concentration is 25 ug atom L^{-1}, the benthic flux increases the nitrogen content of the water by about one percent per day. (Sedimentation removes nitrogen from the water so a cycle is maintained.) This calculation can be used to evaluate the nitrogen concentration in water from the flux measurements in Table 5.

Sorption - Desorption Reactions

Whereas ammonium leaves the sediments continually during the annual cycle, phosphate release takes place primarily in the summer. This results from the interaction of phosphate with iron at the sediment-water interface. In the presence of oxygen, iron is present on the sediment surface as solid iron oxyhydroxides. The phosphate diffusing upward in the interstitial water is apparently adsorbed onto these solids that block phosphate flux into the overlying water.

When the overlying water becomes anoxic in the deeper parts of the estuary during summer, a two-step release process occurs. First, the iron oxyhydroxides are reduced and dissolved, releasing both iron and phosphate into the water in a pronounced pulse. Second, with the block removed, phosphate in the interstitial water diffuses freely into the overlying water, but more slowly than the initial release. When the deep water is reoxygenated in late summer, the phosphate concentration declines rapidly. We hypothesize that sorption reactions involving newly-formed iron oxyhydroxides are responsible for a significant fraction of this phosphorus removal.

The residence time of nitrogen and phosphate in the sediments cannot be measured directly. From the relatively high interestitial water concentrations, we estimate a rather long residence time for some fraction of the nutrients. As much as one-third of the nitrogen could be permanently buried in the sediments. For the remaining 70 percent of the nitrogen and for much of the phosphorus, the residence time may be rather short, on the order of months to a few years. Additional research is required to further test ideas about processes influencing the residence time, and about the size (depth) of the considered sediment reservoir.

Geochemical Reactions

Phosphate participates in geochemical processes in the sediments. A detailed evaluation is beyond the scope of this discussion. The reader is referred to Bricker and Troup (1975) and Bray et al. (1973) for information on the equilibrium chemistry of phosphate minerals in sediments.

Marshes and Bay Grasses

The marshes and Bay grasses along the shorelines of the Bay and its tributaries serve as both sources and sinks for nutrients. At present, there is not complete agreement as to whether marshes are net sources or net sinks of nutrients. During the growing season, marsh plants assimilate nutrients from the water. Nitrogen fixation by some species may be significant when the water is nitrogen deficient. In winter, organic material may be periodically flushed out of the marshes to the adjacent open waters. Similarly, submerged aquatic vegetation (SAV) absorb nutrients during the growing season and contribute organic material to the system during the winter.

SECTION 4

DISSOLVED OXYGEN IN THE ESTUARY

OXYGEN SOURCES

Dissolved oxygen is of primary interest to water quality managers, because it directly affects the well-being of aquatic life. Sources of oxygen include diffusion from the surface, photosynthesis, and reduction of oxidized chemical species. Oxygen is lost from the water through respiration and oxidation of reduced chemical species.

Oxygen gas enters the water by two major mechanisms; diffusion and bubble entrainment at the air-water interface transfer oxygen to the water. The rate of transfer depends on temperature, sea state, wind velocity, and oxygen concentration in the water. Surface turbulence and low temperature enhance both the exchange and the solubility of oxygen in estuarine water. Salinity also exerts an influence, but it is small compared to the other parameters.

Photosynthesis is the second mechanism by which gaseous oxygen enters the system. Oxygen is a product of photosynthesis and is evolved during daylight by the phytoplankton. This is an important mechanism for aeration during summer when warm temperatures and calm weather minimize oxygen solubility and transfer from the atmosphere. However, the same organisms that produce oxygen during the day consume it at night. This results in a daily fluctuation in oxygen concentration, with the minimum value just before sunrise. Therefore, in regions of the estuary where oxygen concentrations are critical, measurements should be made at sunrise for comparison with the desired level of oxygen concentration.

A third oxygen input is the oxygen combined in sulfate and nitrate. Major groups of heterotrophic bacteria fulfill their oxygen requirements by reducing sulfate to sulfide. If gaseous oxygen is not mixed with the sulfide to permit its reoxidation to sulfate, the sulfide accumulates. Sediment interstitial water is characteristically sulfide rich as is the Bay deep water during the summer. Some bacteria reduce nitrate to ammonium and utilize the oxygen liberated. Although nitrate reduction preceeds sulfate reduction, it is not so significant as sulfate reduction because of the large sulfate concentration in sea water. However, this process does result in the sediments consuming nitrate from the overlying water, when nitrate concentrations are moderate to high.

OXYGEN UTILIZATION

Oxygen added to the water by processes just described is consumed by both biological and chemical reactions. The sites for these reactions may be susupended in the water, or contained in or on the sediments. Respiration as a means of regenerating nutrients was discussed in Section 3. Now consider respiration as process by itself.

Respiration is the biological reaction coupling oxygen to reduced substances, usually carbon, to release energy for other intracellular processes. Respiration removes oxygen from the water. The amount of respiration occurring in a body of water is usually not critical to water quality if the oxygen is replaced from the atmosphere as quickly as it is consumed. However, at times, oxygen replenishment lags utilization, so that undesirable conditions of low-oxygen concentration are reached. Since reaeration depends on climate and meterology (factors out of man's control), people have tried to control the addition of oxygen-consuming organic material and stimulants (nutrients) for the formation of organic material to natural waters. Attaining desirable oxygen levels year-round has been a major criterion for wastewater treatment in the United States.

Respiration occurs both in the water and in the sediments. In the main portion of Chesapeake Bay, water-column respiration in the spring removes oxygen faster than it is replenished, so that oxygen concentration declines to zero by May or June. Although oxygen depletion can be accounted for entirely by water-column respiration, the sediment demand is substantial. Its importance is probably expressed more in shallow areas where the amount of oxygen contained in the overlying water column is less, because the amount of water is less. In well mixed shallows, high sediment respiration can be sustained without undesirable oxygen depletion because reaeration keeps pace with utilization.

IMPLICATIONS

The net result of these interacting processes is dissolved oxygen depletion during summer in Bay waters deeper than about 10 m. Taft et al. (1980) suggest that a major proportion of organic matter driving oxygen depletion comes from primary production of the previous year. The remainder could be delivered with the spring freshet of the Susquehanna River. Since the oxygen decline has started as early as February when temperatures are still low, it seems unlikely that winter/spring production in the Bay itself contributes a significant organic load to the oxygen demand.

SECTION 5

SUMMARY AND CONCLUSIONS

Either nitrogen or phosphorus flux into phytoplankton cells can be limiting, according to season and to position in the estuary. The absolute values for nutrient concentrations in the water and for phytoplankton biomass, are the results of several processes tending to add or subtract from the standing crop. High concentrations of nutrients or cells do not necessarily indicate high turnover rates. If anything, the reverse tends to be true.

Nutrients in Chesapeake Bay participate in complex cycles, involving both biological and chemical interactions. Nitrogen and phosphorus have different annual cycles in the open Bay, resulting in nitrogen limiting biomass in summer, and phosphorus limiting it for most of the remainder of the year. The peak in nitrogen availability occurs in spring, because of large nitratè inputs from the tributaries in addition to ambient recycling in water and sediments. The peak in phosphorus availability occurs in summer and is linked with oxygen depletion in water deeper than ten meters. Oxidized iron compounds may play a key role in removing some phosphorus from the water, thus acting as a natural control mechanism where iron is abundant.

Phytoplankton, the major nutrient consumers in the system, have preferences for ammonium nitrogen and phosphate phosphorus. From modeling and from experiments in the Bay, it appears that much of the nitrate entering the upper Bay in spring passes through to the lower Bay, because the phytoplankton are consuming ammonium in preference to the nitrate. Analogs may exist in tributary estuaries.

The high productivity of Chesapeake Bay is sustained by rapid recycling of nutrients in the water column and in the sediments. It appears that the total plankton biomass in the system may be limited by nitrogen or phosphorus at different times, but that the rate of phytoplankton growth is not nutrient-limited because of rapid recycling.

The sediments are critical in nutrient processes, as both a source and sink for different compounds. Progress is being made in quantifying the rates of nutrient flux into and out of the sediments, but this area requires additional research.

Environmental decision-makers should grasp the important characteristics of the estuary discussed herein. In evaluating alternatives for controlling inputs to the system, managers should consider the amount of nutrient to be added to the system compared to what is already there. They should also consider its form. For example, nitrogen added in spring to the upper Bay as nitrate will probably not adversely affect the upper Bay. Phosphate added to the system at any time could increase phytoplankton standing crop, if the controlling influence of iron compounds is exceeded.

Since this estuary, by nature, accumulates nutrients, most nutrients and organic carbon added to the system will remain in it. Thus, once degraded, the lower Bay whole would probably take a fairly long time to recover. However, non-degradation is realistic and achievable. Nevertheless, since the estuary tends to trap nutrients, common sense suggests that increases in the total amount of nutrients should be kept to a minimum. If scientists can provide accurate information on process rates and outputs from the system, it may be possible for managers to regulate inputs to the level of the outputs. The key is quantifying the outputs to the ocean, sediments, and commercial catches, something which so far has proven to be difficult. Chapter 3 of this part of the CBP Synthesis Resport discusses those outputs further. Understanding processes may help humans overcome the estuary's tendency to accumulate nutrients, by finding positive ways to utilize them within the coastal system.

Management agencies supporting research should consider studying processes along with the traditional monitoring of nutrient concentrations. Both kinds of information are important in assessing the progress of chosen management strategies.

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CHAPTER 3

NUTRIENT AND SEDIMENT LOADS TO THE TIDAL CHESAPEAKE BAY SYSTEM

by

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SECTION 1

INTRODUCTION

An important area of concentration of the Management Questions within the CBP Nutrients Program is the excessive fertilization (over-enrichment) of the Bay system. Nutrient over-enrichment can cause excessive algal blooms and oxygen depletion. Although this fertilization process, called eutrophication, occurs naturally (i.e., runoff from the land and atmospheric deposition processes have always carried plant nutrients to receiving waters), the cultural activities of man accelerate it. When coupled with the complicating problem of increased sedimentation rates due to cultural activities, the result can be the shortening of the life of the estuary and a decrease in the value of the system and its resources. This two-pronged problem of nutrient over-enrichment and increased sediment yield has come to be known as "cultural eutrophication." The aquatic plant nutrients considered in this paper are the various forms (species) of nitrogen and phosphorous.

To understand and manage potential cultural eutrophication problems in the Bay system, we need tc answer a number of questions concerning sources of nitrogen, phosphorus, and sediment to the Bay and its tidal tributaries. This paper seeks to synthesize available research findings on these sources by answering the following Management Questions:

1. What is the atmospheric contribution to nutrient input?
2. What percentage of the nutrients is from point sources?
3. What percentage of the nutrients is from nonpoint sources? How do they vary over time?
4. What are the pollutant runoff rates for particular land uses?
5 What percentage of nonpoint sources can be attributed to particular land uses?
6. What are the nutrient loadings from the Fall Line?
7. What do the bottom sediments contribute to nutrient inputs?
8. What are the flux rates of nutrients from the bottom sediments, and how do they vary seasonally?
9. Given the estimated loadings of nutrients for each of the sources, which will be the most important in terms of their effects on the Bay system.

To answer the management questions, we synthesized available information to understand the components of a nutrient budget. In this paper, the approach taken for determining the nutrient budget centers on a simplified consideration of the Bay as a container, or box, into which flow nutrient-laden waters from various sources (see Figure I.1). This box model approach allows the reader to visualize nutrient sources simultaneously as a simple schematic diagram or picture. We considered five external sources expressed both as annual and seasonal loadings. These sources are shown in Figure I.2 and include:

- Atmospheric Sources, defined for the purposes of this paper as nitrogen and phosphorus falling onto tidal Chesapeake Bay system waters in precipitation and nitrogen lost to the atmosphere as nitrous oxide and gained through nitrogen fixation. No estimate is made of denitrification losses.

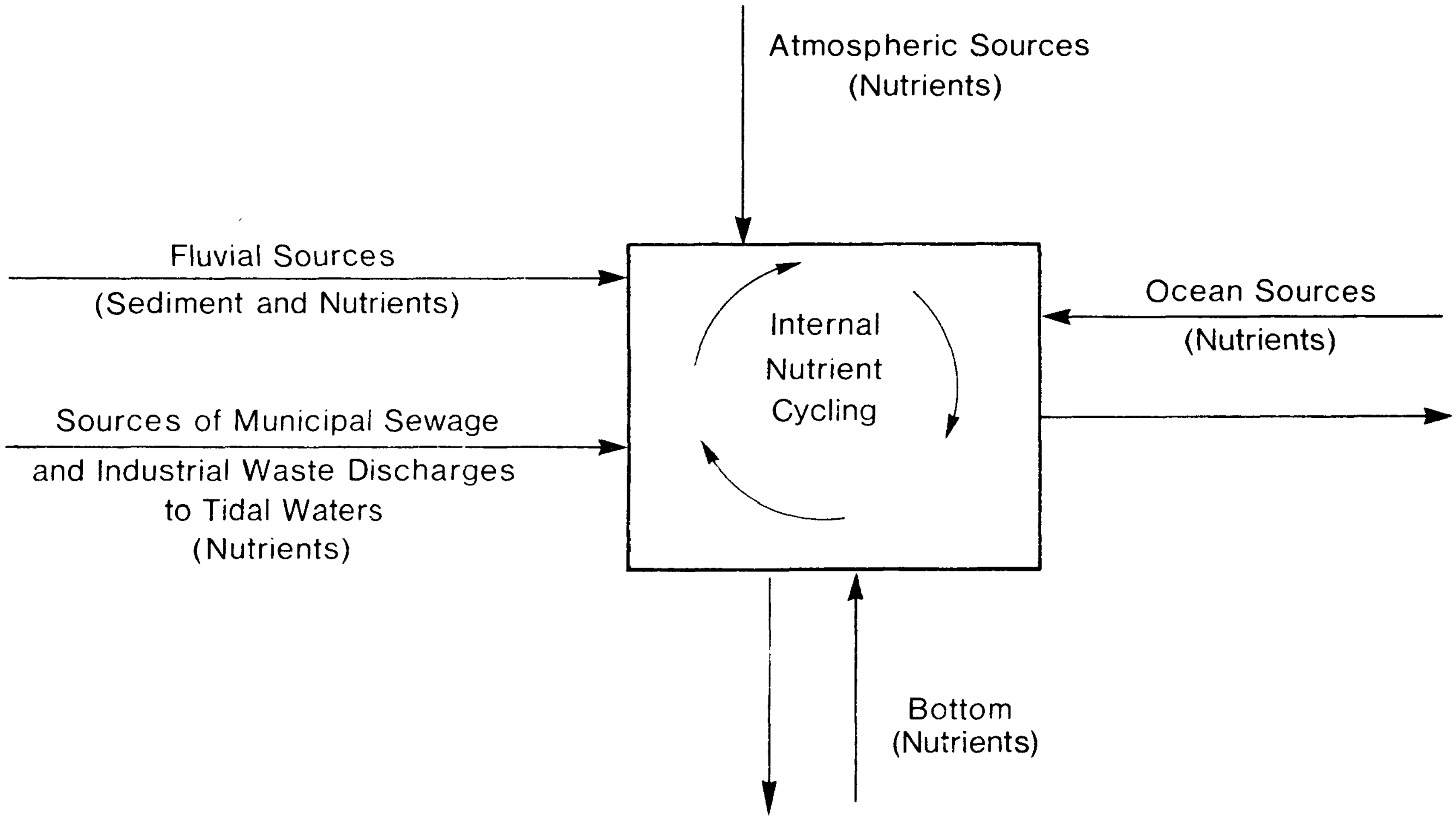

Figure I.1. Box model of nutrient and sediment input sources to the Chesapeake Bay system.

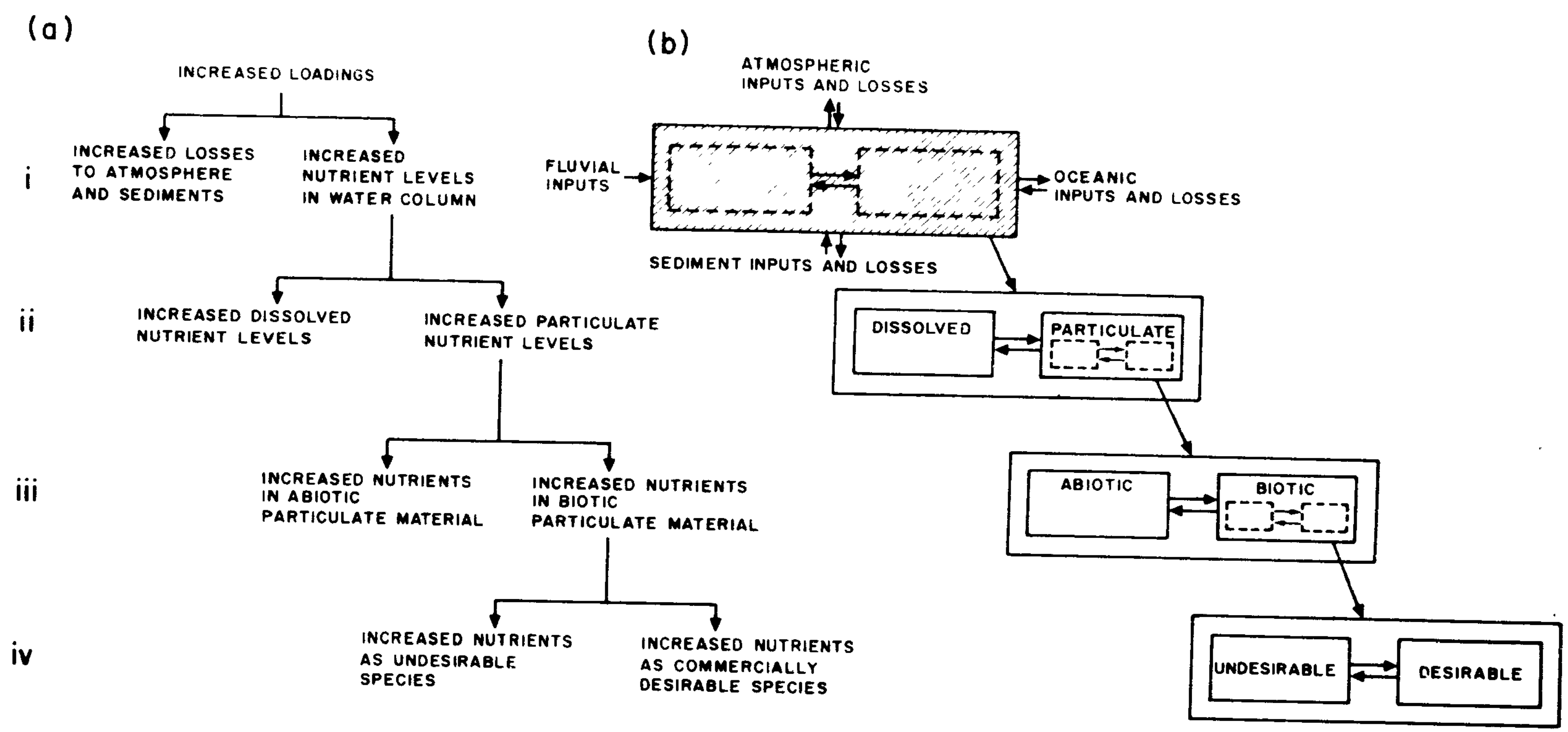

Figure I.2. Binary dendogram showing possible responses of the water column to increased nutrient loadings.

- Riverine-Transported Sources, defined as nitrogen, phosphorous, and sediment, which derive from above the head of tide or the Fall Line[1]. This source includes loadings from surface land runoff, atmospheric loadings falling upon upland waters, groundwater contributing as baseflow, municipal sewage treatment plant discharges from above the Fall Line and industrial waste discharges from above the Fall Line as measured at the Fall Line water quality monitoring stations. No estimate is made of land runoff or groundwater loadings deriving from below the Fall Line.
- Point Sources to Tidal Waters, defined as nutrient loadings from publicly owned sewage treatment works discharges and industrial waste discharges that enter the tidal waters of the Bay system directly. For the purposes of this paper (see definition of "functional" fall line in Section IV) all such discharges are defined to be those that enter downstream of the head of tide of the Susquehanna, Potomac and James rivers;
- Bottom Sources, defined as fluxes of nitrogen and phosphorous between the bottom sediments of the Bay and the water column. Because of the lack of wide spread tributary benthic flux data, fluxes are computed only for the Bay proper;
- Ocean Sources, defined as the net flux of nutrients between the Bay and the Atlantic Ocean.

In addition to these five sources, the internal or re-cycled nutrients as a source will also be discussed. The only sediment source considered is riverine, included as described above under Riverine Transported Sources. No estimates were made of potential sediment loads entering the system from the ocean or shore erosion, nor of the contribution because of phytoplankton production of skeletal material. The net sedimentation of nutrients was determined by difference. A schematic diagram of the box model is shown in Figure I.1, and the nutrient budgets appear in Section VIII. Nonpoint source nutrient contributions below the fall line have not been included in this paper because the data were not available at the time of writing. Estimates from below the fall line are being currently made through a computerized model and will be available in the near future. The eventual inclusion of these data in the nutrient budget will tend to reduce the percentages of nutrient loads shown here.

In summary, the authors of this paper have assembled available information concerning the most important nutrient sources and attempted to answer the pertinent management questions. Each of the sources is discussed in separate sections, with a comparison of these sources made in Section VIII. Conclusions and answers to the management questions are also found in the last section.

[1]Wolman (1968) describes a broad area trending from Southeast to Northeast which defines the head of tide (and the head of navigation) as the contact between the hard crystalline basement rocks and the unconsolidated sediments of the Coastal Plain. This demarcation he calls the Fall Line or Fall Zone.

SECTION II

ATMOSPHERIC SOURCES OF NUTRIENTS

Precipitation and dust/dirt dryfall are the major mechanisms that return to the earth's surface gaseous and particulate materials that are injected into the atmosphere from natural and man-induced sources. To determine the relative importance of the atmospheric source to the overall nutrient input budget, we estimated the mass of nitrogen and phosphorous carried into the Bay by rainfall. No estimate of the dryfall portion of the atmospheric source has been made because of the paucity of available data and the uncertainty of existing dryfall sampling techniques[1].

Although about 10 percent of annual areal precipitation is made up of forms other than rainfall (i.e., snow or ice), data on concentrations of nutrients in these forms are lacking. Therefore, for the purpose of this paper, the concentrations computed for rainfall will be assigned to the total precipitation budget.

NUTRIENT CONCENTRATIONS IN PRECIPITATION

Rainfall quality data were chosen from studies conducted within the Chesapeake Bay drainage basin. We chose this method to ensure that the data reflect the natural and man-induced surface sources peculiar to the region. The size of the available data base made the regional restriction feasible. The data included interim reports, draft final reports, completion reports or personal communications of six major regional nonpoint pollution and rainfall quality studies (Northern Virginia Planning District Commission [NVPDC] and Virginia Polytechnic Institute and State University [VPI&SU] 1977, Bostater[2] 1981, Wade & Wong 1981, Correll et al. 1978, Ward and Eckhardt 1979, Lietman[3] 1981, VPI & SU 1981, Weand[4] 1981). The general locations of these study areas are shown in Figure II.1.

The assembled data base consists of bulk precipitation samples from as many as 125 storm events collected at up to 18 sampling locations for all seasons from 1976 through 1981. In most cases, a raingage within, or near the sample collection areas, recorded precipitation volumes. For most storm events, composited samples were analyzed for ammonia nitrogen, nitrite + nitrate nitrogen, total Kjeldahl nitrogen, orthophosphorus and

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[1]The authors note that although some dryfall deposition data collected within Chesapeake Bay sub-basins are available (Correll et al. 1978, Virginia Polytechnic Institute & State University 1978), too little was available to make reliable Bay-wide estimates.

[2]Personal Communication: "Patuxent River Park Rainfall Quality Data," C. Bostater, Department of Natural Resources, State of Maryland, 1981.

[3]Personal Communication: "Pequea Creek Watershed Rainfall Quality Data," P. Lietman, Harrisburg Sub-district, U.S. Geological Survey, Harrisburg, PA, October, 1981.

[4]Personal Communication: "Occoquan Watershed Rainfall Quality Data," B. Weand, Occoquan Watershed Monitoring Laboratory, Manassas, VA, November, 1981.

total phosphorus were assembled (total nitrogen was computed as the sum of total Kjeldahl nitrogen plus nitrite + nitrate nitrogen). Concentrations were recorded as milligrams per liter (mg/L) or converted to mg/L from data expressed as molar concentration. All concentrations are expressed as elemental nitrogen (N) or phosphorous (P).

Many investigators have shown that constituent concentrations in rainfall are strongly related to precipitation amounts (Stensland 1980). For example, given similar antecedent rainfall conditions, a smaller rainfall event would probably have higher nutrient concentrations than a larger rainfall event occurring in the same geographical area. This is due primarily to the tendency for rainfall pollutant loadings to exhibit "first-washout" or "first-flush" effects (NVPDC and VPI & SU 1978; Gambell and Fisher; Uttormark, Chapin, and Green 1974). The first-flush effect is characterized by concentrations of rainfall constituents reaching a maximum value early in a storm event and declining rapidly thereafter. To compensate for this effect it is common to report rainfall constituents as volume-weighted average concentrations[1] rather than as arithmetic average concentrations (Stensland 1980). For this reason, all concentrations reported in this chapter have been computed as volume-weighted averages.

For this analysis, we used the volume weighted mean of the nutrient concentrations for data collected in each geographic area (shown in Figure II.1), for each season. An equal-weight average of the means of data collected at each of the geographic locations by season was computed. We took this approach to reduce the potential for those studies with the most data to skew the means in favor of one particular geographic area. The results of these computations are reported in Table II.1.

Lang and Grason (1980) reported mean monthly precipitation totals (based upon NOAA records, 1941-1970) at three sites within the Chesapeake Bay Basin, including Richmond, VA, College Park, MD, and Harrisburg, PA. An average of the mean monthly totals over the three stations was computed to represent a Bay-wide average mean monthly precipitation and is reported in Table II.2. Also shown in this Table are the computed seasonal totals and the annual average of 39.6 inches of precipitation. It can be seen from the data in Table II.2 that, on the average, the Bay receives 21.6, 25.3, 24.8, and 23.3 percent of its annual precipitation in winter, spring, summer and fall respectively. These percentages were used as the weighting factors to compute volume-weighted annual nutrient concentrations (from the seasonal concentrations), shown on the last row of Table II.1.

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[1] $$\frac{\sum_{i=1}^{n} c_i q_i}{\sum_{i=1}^{n} q_i}$$

c_i = concentration

q_i = volume

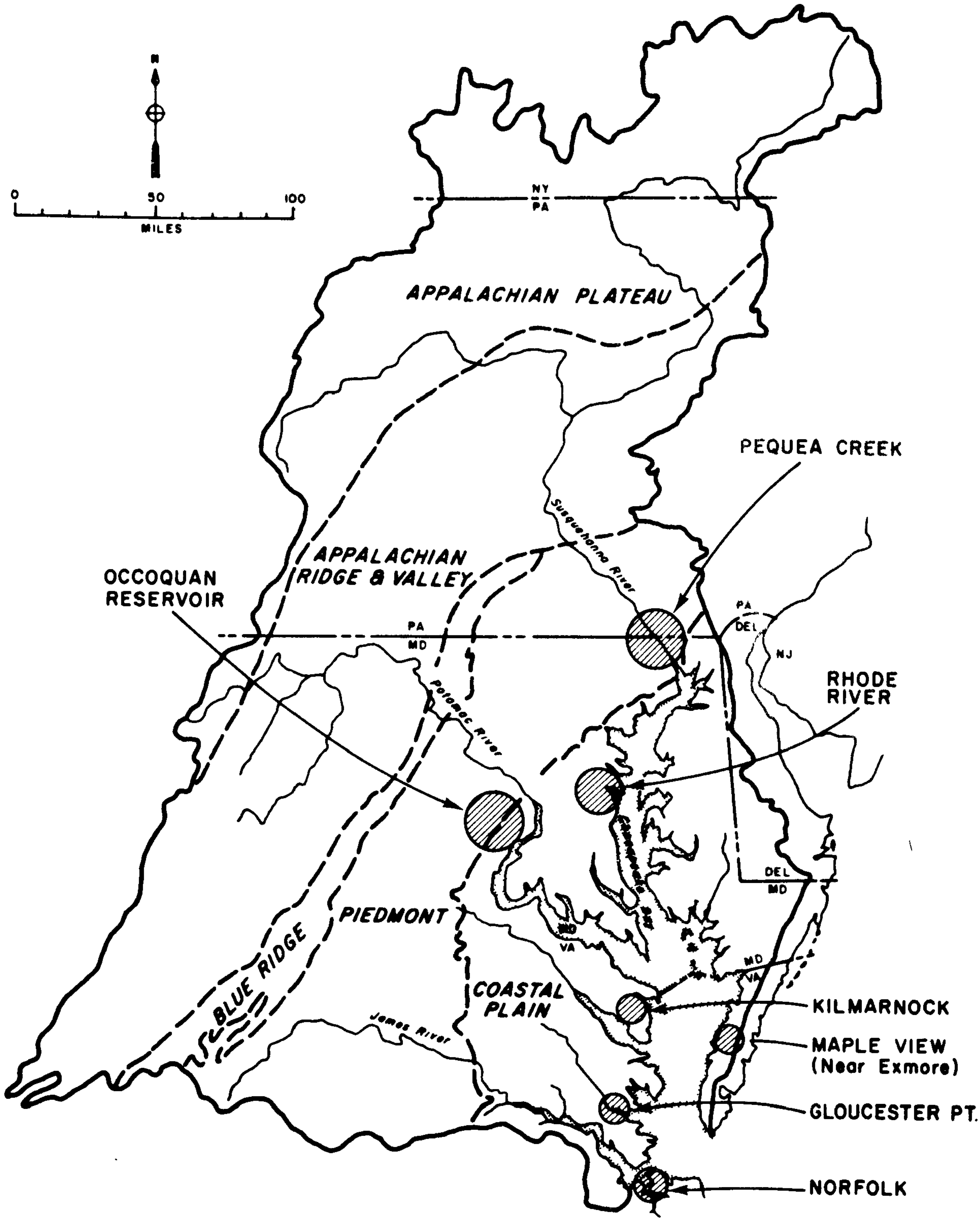

Figure II.1. Locations of rainfall sampling.

TABLE II.1. SEASONAL AND ANNUAL VOLUME-WEIGHTED MEAN NUTRIENT CONCENTRATIONS OBSERVED IN BAY AREA RAINFALL (REPORTED AS mg/L OF ELEMENTAL MATERIAL)

		NH_3-N mg/L	NO_2+NO_3-N mg/L	TKN mg/L	TN (NO_2+NO_3+TKN) mg/L	Ortho P mg/L	Total P mg/L
WINTER	X(1)	0.376	0.540	0.586	1.126	0.016	0.038
	SD	0.169	0.271	0.545	---	0.013	0.030
	A,B	4, 39	6, 50	5, 33	---	4, 37	6, 51
SPRING	X	0.537	0.731	1.783	2.514	0.015	0.080
	SD	0.204	0.304	0.852	---	0.009	0.092
	A,B	3, 98	5, 125	4, 96	---	3, 91	5, 122
SUMMER	X	0.250	0.621	0.988	1.609	0.014	0.079
	SD	0.141	0.525	0.426	---	0.015	0.098
	A, B	3, 22	5, 40	4, 35	---	5, 21	5, 41
FALL	X	0.256	0.360	0.642	1.002	0.021	0.050
	SD	0.232	0.193	0.662	---	0.013	0.036
	A, B	3, 22	4, 33	4, 33	---	3, 16	4, 31
ANNUAL		0.351	0.571	1.022	1.593	0.016	0.064
		--	--	--	--	--	--
		--	--	--	--	--	--

[1] Legend - X = Equal Weight Mean (mg/L)
SD = Standard Deviation
A = # studies from which data were taken for computation
B = # station storms sampled (n)

Nutrient Loads in Precipitation

Seasonal and annual nutrient loadings were estimated based on precipitation falling upon the water areas of Chesapeake Bay and its tidal tributaries. Nutrient loadings, from precipitation falling upon the water and land surfaces of the Bay watershed above the head of tide of the Susquehanna, Potomac, and James Rivers, are accounted for in the fluvial loadings computed in Chapter III of this paper. The mean annual and seasonal precipitation values shown in Table II.2 were used to compute expected annual and seasonal volumes of precipitation input to the 4412.1 square mile Chesapeake Bay tidal system. These volumes were applied to the concentrations in Table II.1 to produce the seasonal and annual nutrient loading estimates that are shown in Table II.3.

TABLE II.2. BAY-WIDE MEAN MONTHLY AND SEASONAL PRECIPITATION, IN INCHES, COMPUTED FROM MONTHLY AVERAGES OF NOAA STATIONS[1]

Month	Mean Monthly Total (in.)	Season	Mean Seasonal Total (in.)	Seasonal % of Annual Total
December	3.18	----		
January	2.72	----		
February	2.65	Winter	8.55	21.6
March	3.14	----		
April	2.95	----		
May	3.69	Spring	10.04	25.3
June	3.79	----		
July	3.61	----		
August	4.42	Summer	11.82	29.8
September	3.21	----		
October	2.79	----		
November	3.21	Fall	9.21	23.3

Average Annual Total = 39.62 in.

[1]Monthly totals shown are the average of the mean monthly totals at each of three NOAA stations (Richmond, VA, College Park, MD, Harrisburg, PA) based on precipitation records 1941-1970 as reported by Lang and Grason, 1980.

TABLE II.3. SEASONAL AND ANNUAL NUTRIENT LOADS FROM PRECIPITATION TO THE TIDAL CHESAPEAKE BAY SYSTEM (MILLIONS OF POUNDS)

	Precipitation Volume (inches)	Ammonia-N	Nitrite + Nitrate-N	Total Kjeldahl N	Total Nitrogen-N	Ortho-Phosphorus P	Total Phosphorus P
Winter	8.55	2.06	2.95	3.21	6.16	0.088	0.208
Spring	10.04	3.45	4.70	11.46	16.15	0.096	0.514
Summer	11.82	1.89	4.70	7.47	12.17	0.106	0.598
Fall	9.21	1.51	2.12	3.78	5.91	0.124	0.295
Annual	39.62	8.91	14.47	25.92	40.39	0.399	1.64

The compilations in the Tables indicate that both concentrations and areal loading rates in rainfall are significant in comparison to other sources (Section VIII). Nitrogen and phosphorus concentrations shown are similar to those found in other studies conducted in the northeastern United States

(Ward and Eckhart 1977, Stensland 1980). The seasonal and annual rainfall concentrations of total nitrogen exceeded the volume-weighted mean concentrations observed in runoff from forested sites studied under the Chesapeake Bay Programs's Intensive Watershed Studies Project[1]. (See Table VIII.6 for these concentrations.) Orthophosphorus concentrations in precipitation are of the same order of magnitude, but generally less than those observed in forested land runoff. Concentrations of most constituents are typically much less than those in runoff from other land uses.

Comparisons between atmospheric and other sources are made in Section VIII. For example, it can be seen in Tables VIII.3(b) and VIII.4(b) that precipitation is a major contributor of TKN in spring and summer. This could be particularly important in summer, when nitrogen limits phytoplankton biomass in much of the Bay (Chapter 2 of this part).

OTHER ATMOSPHERIC NUTRIENT INTERACTIONS

Other nutrient processes involving gains of nitrogen from the atmosphere and losses of nitrogen to the atmosphere were considered. The nitrogen input to the Bay by nitrogen fixation is not well known, but it should be small compared to other inputs since nitrogen fixation rates in the water are vanishingly small. We estimate 25,000 pounds per year net input from marshes. The nitrogen loss to the atmosphere as N_2O and NH_3 gas is also probably small. Few measurements have been made from which we estimate an annual loss of 40,000 pounds per year from the estuary. We hope, future research will refine these estimates. Losses due to denitrification were not estimated, but were considered to be small relative to the sources (i.e., precipitation, riverine, etc.).

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[1]Personal Communication: "Volume-Weighted Mean Concentrations of Storm Event Runoff from EPA/CBP Test Watersheds," John P. Hartigan, Northern Virginia Planning District Commission, Falls Church, VA, October 13 1981.

SECTION III

RIVERINE-TRANSPORTED SOURCES OF NUTRIENTS AND SEDIMENT

A major objective of the EPA Chesapeake Bay Program was to assess the loadings of nutrients, sediment and other water quality constituents from the watersheds of the Bay to the tidal system. The approach taken included developing an intensive data base of the nutrient and sediment loadings entering the Bay from fluvial sources over a period of several years, and then developing a methodology to extrapolate that data to produce reliable estimates of the expected long-term loadings from the upstream sources. In this chapter, we estimate seasonal and annual total mass flux of nutrients to the tidal waters of the Bay system from above the head of tide, or the fall line. The section is subdivided into three sub-sections. The first describes a fairly rigorous development of loadings from the three major tributaries (Susquehanna, Potomac, and James Rivers) based on data collected as part of the EPA Chesapeake Bay Program. The second section contains estimates of minor tributary (Patuxent, Rappahannock, Mattaponi, Pamunkey, and Chickahominy Rivers) loadings based, in part, upon a field study performed by Guide and Villa in 1969 through 1970. In the third section, the total annual and seasonal fluvial-loading estimates are presented.

NUTRIENT INPUTS FROM THE MAJOR TRIBUTARIES

To determine the nutrient contributions from the major watersheds of the Chesapeake Bay drainage area, the Bay Program established a fall line monitoring project. This project, performed by the U.S. Geological Survey, monitored water quality of three major tributaries of the Bay. The sites monitored were: Susquehanna River at Conowingo, MD; Potomac River at Chain Bridge, Washington, DC; James River at Cartersville, VA. Together, the three rivers drain about 70 percent of the approximately 64,000 square mile Chesapeake Bay drainage basin and account for about 80-85 percent of the long-term average discharge Bay-wide (Wolman 1968) (see Figure III.1). Previous work by Guide and Villa (1972) indicated that these three tributaries were the primary riverine sources of nutrient loads to the tidal Chesapeake Bay system. They found that these tributaries contributed as much as 94 percent of the total phosphorus load and 95 percent of the total nitrogen load emanating from the eight major Bay tributaries.[1]

The USGS began the sampling program in January of 1979 and continued through April of 1981, a period of 28 months.[1] Base flow water quality was monitored every two weeks at the Conowingo station on the Susquehanna, and once a month on the Potomac and James. Samples were also taken at high flows on all stations to better understand the mechanisms affecting water quality during these critical periods of high-mass transport. Samples were analyzed for major ions, suspended sediment, selected nutrient species, and trace metals.

[1]These are Susquehanna, Patuxent, Potomac, Rappahannock, Pamunkey, Mattaponi, James, and Chickahominy Rivers.

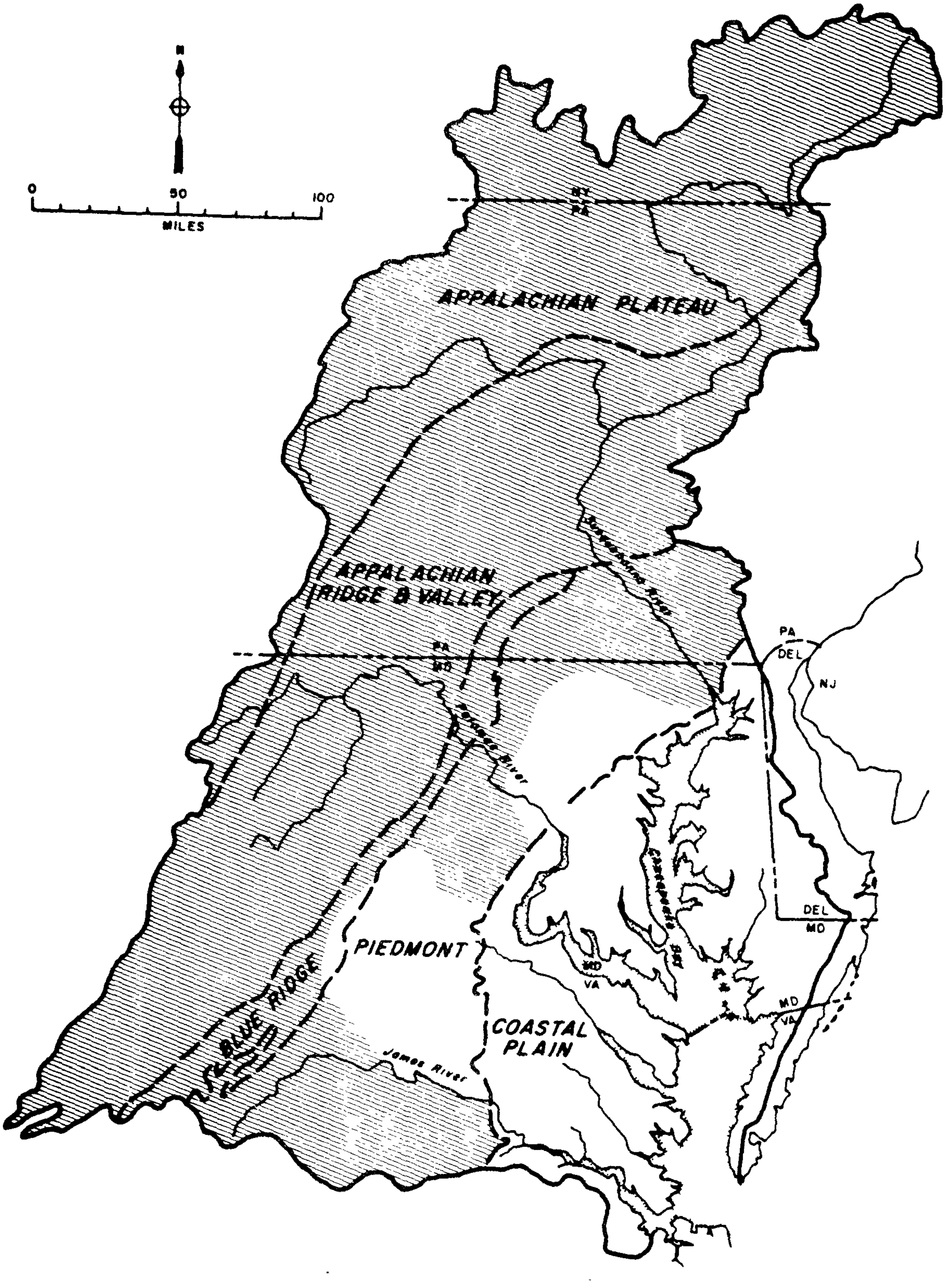

Figure III.1. Physiographic provinces of Chesapeake Bay basin. Shaded areas drain into the fall line areas in the Susquehanna, Potomac, and James Rivers.

An interim basic data report by Lang and Grason (1980) describes the first year of this project, and a draft final report (Lang 1982) will soon be completed.

Description of the Major Tributary Drainage Areas

Susquehanna: The Susquehanna River Basin has a drainage area of about 27,510 square miles of which 27,100 are above the fall line monitoring station. The basin is about 250 miles long and about 170 miles wide. The drainage lies within four physiographic provinces: the Appalachian, the Ridge and Valley, the Piedmont, and the Blue Ridge. The land-use is predominately forest and agriculture with no major urban areas.

Potomac: The Potomac River Basin has a drainage area of about 14,670 square miles of which 11,560 lie above the monitoring station. The basin is made up of eight major sub-basins with the main stem approximately 280 miles in length. The drainage lies within five physiographic provinces: the Applachian, the Ridge and Valley, the Piedmont, the Blue Ridge, and the Coastal Plain (Figure III.2). The Coastal Plain portion of the basin does not lie within the monitored area. The land use is predominately agriculture and forest with the Washington, DC area draining below the monitoring gage site.

James: The James River Basin has a drainage area of about 10,000 square miles of which 6,257 drain to the sampling station. The basin is about 400 miles in length and drains four physiographic provinces: the Ridge, and Valley, the Piedmont, the Blue Ridge, and the Coastal Plain (Figure III.2). None of the Coastal Plain portion of the watershed lies within the monitored drainage. The basin is mostly agricultural and forested with the Richmond metropolitan area draining below the monitoring point.

The mean annual and seasonal discharges for each basin are presented in Table III.1. These data were computed based upon records retrieved from the USGS stream discharge stored on the EPA STORET system. Water year 1981 data were retrieved as provisional data, subject to revisions.

[1]Although the monitoring period reported by Lang (1982) covers only 28 months, other collection efforts at the three sites resulted in data being available for this analysis, beginning in October 1978 and running through as late as November of 1981. Some of these data were collected as part of the ongoing EPA/USGS National Stream-Quality Accounting Network (NASQUAN).

TABLE III.1. ANNUAL AND SEASONAL[1] MEAN DAILY DISCHARGES AND DRAINAGE AREAS OF THE MAJOR BASINS MONITORED: SUSQUEHANNA, POTOMAC, AND JAMES RIVERS

BASIN	DRAINAGE AREA (Square miles)	MEAN DAILY DISCHARGE[2] (Cubic Feet Per Second-Per Day) (CFSD)		PERIOD OF RECORD Beginning year-Ending year (Number of years/number of days)
Susquehanna River @Conowingo, MD (01578310)	27,100	Annual	43,286.8	Oct. 1967-Sept.1981 (14 yrs/5072 days)
		Winter	50,109.6	(1,264 days)
		Spring	68,011.5	(1,288 days)
		Summer	25,193.3	(1,277 days)
		Fall	29,317.5	(1,243 days)
Potomac River near Washington, DC (01646500)	11,560	Annual	10,953.9	March 1930-Sept.1981 (51 yrs./18,842 days)
		Winter	13,286.9	(4,603 days)
		Spring	18,466.8	(4,784 days)
		Summer	6,147.2	(4,784 days)
		Fall	5,883.3	(4,671 days)
James River @Cartersville, VA (02035000)	6,257	Annual	6,879.1	Oct.1924-Sept.1981 (57 yrs./20,505 days)
		Winter	8,811.7	(5,060 days)
		Spring	10,209.3	(5,156 days)
		Summer	4,248.2	(5,155 days)
		Fall	4,271.7	(5,134 days)

(1) Winter = December, January, February
Spring = March, April, May
Summer = June, July, August
Fall = September, October, November
Statistical comparisons between the Conowingo Station data and Harrisburg Station data indicate that the 1967 through 1981 periods are representative of the long term stream flow characteristics.

(2) Discharges shown were computed from records retrieved through the USEPA-STORET data bank as transferred from the USGS-WATSTORE system. Water Year 1981 records used are provisional and subject to revision. No adjustments (i.e., for diversions) have been made to the discharges. Computations were made using the Statistical Analysis System Procedure MEANS (SAS Institute 1979).

Computation of Riverine-Transported Nutrient and Sediment Loads

To predict the statistically significant, expected value of the daily nutrient loading from each of the major tributaries, we extrapolated the nutrient data collected during the fall line monitoring project through a series of linear and log-linear regression models. These models relate nutrient concentration, or nutrient loading rate, with mean daily discharge for each of the monitored tributaries. In all cases, models were developed using bivariate least squares regression techniques.

The independent, or predictor variable (X, in equation III-2), is the mean daily discharge of the flow-monitoring station adjacent to the water quality monitoring site. We eliminated from consideration other potential independent variables such as instantaneous flow, specific conductance, or sediment concentration (Lang and Grason 1980, Lang 1982) in either univariate or multivariate models, because there is no available long-term record of occurrences of these water qualilty constituents.

The models tested in this analysis were either linear, or linearized through transformations of the variables. This infers a direct relationship between the frequency-duration distribution of the independent variable (mean daily discharge) and that of the response variable (e.g., daily nutrient loads). It is implied that the response variable has the identical (but perhaps transformed) frequency-duration distribution of the predictor variable. Only a parameter with a long-term period of record sufficient to develop a reliable frequency-duration relationship should be utilized as a predictor variable. This limitation restricted the model formulation process to use of mean daily discharge (Q) as the independent variable in all models.

The period of record and the number of years of daily discharge data available at each site are shown in Table III.1. For further detail on the model development methodology, see Appendix A. This appendix contains the equations used to normalize storm events. Development of concentration and loading rate models using regression analysis is also explained.

Regression Analysis Results

As mentioned above, the development of the regression equation and the model selection methodology can be found in Appendix A. The models chosen, along with the appropriate regression statistics, are shown in Table III.2, III.3, and III.4 for the Susquehanna, Potomac, and James stations respectively. For example, the variance-stabilizing transformations were selected for the Susquehanna River for the variables TN, DN, NO23, TKN, and DP (Table III.2). These models either had r^2 values below 0.65 for NH34, TP,OP, and SED (Table III.3), or the examination of scatter plots did not support the use of a variance stabilizing transformation and, therefore, loading rate models were selected for these variables.

TABLE III.2. REGRESSION MODELS CHOSEN FOR THE SUSQUEHANNA RIVER AT CONOWINGO, MD (1578310)

Model Chosen	Water Quality Constituent	Regressed Intercept (B_0)	Regressed Slope (B_1)	Pr [(1)] value t (slope)	Coefficient of Determination (r^2)	Degrees of Freedom
ln(C/Q) vs ln(1/Q)	TN	-0.318	0.937	0.0001	0.92	86
ln(C/Q) vs ln(1/Q)	DN	0.0762	0.982	0.0001	0.93	66
ln(C/Q) vs ln(1/Q)	NO23	-0.600	0.948	0.0001	0.87	86
ln(LR) vs ln(Q)	NH34	-2.48	1.15	0.0001[(2)]	0.71	86
ln(C/Q) vs ln(1/Q)	TKN	-1.732	0.921	0.0001	0.81	87
ln(LR) vs ln(Q)	TP	-5.74	1.42	0.0001[(2)]	0.89	87
ln(C/Q) vs ln(1/Q)	DP	-4.42	0.972	0.0001	0.78	85
ln(LR) vs ln(Q)	OP	-3.40	1.11	0.0001[(2)]	0.73	66
ln(LR) vs ln(Q)	SED	-1.19	1.56	0.0001[(2)]	0.66	93

[(1)] Students "t" test for H_0: slope = 0. The probability value shown answers the question "If the parameter is really equal to zero, what is the probability of getting a larger value of it?" A very small value for this probability indicates that the slope is not likely to equal zero and, therefore, that flow (or the indicated transformed flow) contributes significantly to the model (SAS 1979, Procedure: General Linear Models).

[(2)] NB: The relationship implied by this model may be biased and, therefore, may limit the usefulness of the student's "t" test (see text).

TABLE III.3. REGRESSION MODELS CHOSEN FOR THE POTOMAC RIVER AT CHAIN BRIDGE, WASHINGTON, DC (1646580)

Model Chosen	Water Quality Constituent	Regressed Intercept (B_0)	Regressed Slope (B_1)	Pr [(1)] value t (slope)	Coefficient of Determination (r2)	Degrees of Freedom
ln(C/Q)vs.ln(1/Q)	TN	-0.942	0.857	0.0001	0.86	64
ln(C/Q)vs.ln(1/Q)	DN	-1.49	0.827	0.0001	0.84	63
ln(C/Q)vsln(1/Q)	NO23	-1.22	0.881	0.0001	0.80	64
ln(LR)vs.ln(Q)	NH34	-3.53	1.23	0.0001[(2)]	0.71	61
ln(C/Q)vs.ln(1/Q)	TKN	-2.53	0.807	0.0001	0.72	80
ln(LR)vs.ln(Q)	TP	-3.87	1.33	0.0001[(2)]	0.85	79
ln(C/Q)vs.ln(1/Q)	DP	-4.67	0.885	0.0001	0.72	77
ln(LR)vs.ln(Q)	OP	-2.58	1.08	0.0001[(2)]	0.70	47
ln(LR)vs.ln(Q)	SED	-4.76	2.06	0.0001[(2)]	0.88	60

[(1)] Students "t" test for H_0: slope = 0. The probability value shown answers the question "If the parameter is really equal to zero, what is the probability of getting a larger value of it?" A very small value for this probability indicates that the slope is not likely to equal zero and, therefore, that flow (or the indicated transformed flow) contributes significantly to the model (SAS 1979, Procedure: General Linear Models).

[(2)] NB: The relationship implied by this model may be biased and, therefore, may limit the usefulness of the student's "t" test (see text).

TABLE III.4. REGRESSION MODELS CHOSEN FOR THE JAMES RIVER AT CARTERSVILLE, VA (2035000)

Model Chosen	Water Quality Constituent	Regressed Intercept (B_0)	Regressed Slope (B_1)	Pr [1] value t (slope)	Coefficient of Determination (r^2)	Degrees of Freedom
ln(C/Q) vs.ln(1/Q)	TN	-2.11	0.802	0.0001	0.82	54
ln(C/Q) vs.ln(1/Q)	DN	-1.09	0.957	0.0001	0.89	38
ln(C/Q) vs.ln(1/Q)	NO23	-2.20	0.902	0.0001	0.76	56
ln(C/Q) vs.ln(1/Q)	NH34	-4.09	0.909	0.0001	0.65	49
ln(LR) vs.ln(Q)	TKN	-1.72	1.28	0.0001[2]	0.86	55
ln(C/Q) vs.ln(1/Q)	TP	-3.18	0.885	0.0001	0.67	56
ln(C/Q) vs.ln(1/Q)	DP	1.07	1.45	0.0001	0.91	56
ln(C/Q) vs.ln(1/Q)	OP	0.0494	1.35	0.0001	0.82	46
ln(LR) vs.ln(Q)	SED	-5.06	2.12	0.0001[2]	0.90	71

(1) Students "t" test for H_0: slope = 0. The probability value shown answers the question "If the parameter is really equal to zero, what is the probability of getting a larger value of it?" A very small value for this probability indicates that the slope is not likely to equal zero and, therefore, that flow (or the indicated transformed flow) contributes significantly to the model (SAS 1979, Procedure: General Linear Models).

(2)<u>NB</u>: The relationship implied by this model may be biased and, therefore, may limit the usefulness of the student's "t" test (see text).

Computation of Nutrient and Sediment Loads from the Major Tributaries

Each of the regression equations shown in Tables III.2, III.3, and III.4 were encoded in a Statistical Analysis System (SAS Institute 1979) program. This program computed a daily load for each day in the period of record of the flow data (Table III.1), multiplied the individual daily load by the relative frequency of the day's flow (relative frequency = 1/ number of days in period of record), and summed the product over the period of record. In this way, the program computed the area under the loading-frequency curve of one-day duration, which is equivalent to the expected value of daily loading. This technique ensured proper computation of expected values whether the model being used was linear or non-linear. Computations were performed for annual and seasonal discharge-frequency distributions of the three major tributaries for the parameters shown in Table A-1. The results of these computations for annual, winter, spring, summer, and fall seasons are presented in Tables III.5(a), III.6(a), III.7(a), III.8(a), and III.9(a) respectively. Percentage breakdowns for each source are listed for annual, winter, spring, summer, and fall seasons in Tables III.5(b), III.6(b), III.7(b), III.8(b), and III.9(b) respectively.

Not computed here are flow/load ratios which would show that although the Susquehanna River has the largest flows, the ratios of material to flow, and material to drainage area, are no greater than in the other tributaries. In fact, these ratios are less than those in several of the other Bay tributaries.

TABLE III.5(a). ESTIMATED ANNUAL MEAN DAILY NUTRIENT AND SEDIMENT LOADS TO THE CHESAPEAKE BAY SYSTEM FROM SOURCES TRANSPORTED BY RIVERS (ALL VALUES x 10^3 LBS/DAY UNLESS OTHERWISE INDICATED)

Constituent	Susquehanna	Potomac	James	Other Tribs.	Total Fluvial Load to the Bay System
TN	342.84	95.27	29.11	20.39(1)	468.61
DN	307.81	74.38	18.65	14.49(2)	415.33
NO23	228.70	56.75	10.31	9.15	304.91
NH34	19.41	3.18	1.46	0.74	24.79
TKN	99.70	32.14	17.49	11.24	160.57
TP	15.65	6.26	4.54	1.69	28.14
DP	3.84	1.73	1.80	0.57(3)	7.94
OP	4.93	1.83	1.58	0.54(4)	8.88
SED	7,263.44	5,986.22	2,979.81	1,925.5(5)	18,155.00
Discharge	43,286.8cfsd	10,953.9cfsd	6,879.1cfsd	3,525cfsd(6)	64,644.8cfsd

(1)Computed as the sum of $NO_{2,3}$ + TKN

(2)Estimated by computing the mean of DN:TN ratios for the Potomac and James and applying to the estimated TN loading rate for the 'Other Tribs.' The Susquehanna was excluded from this calculation because it is a regulated (i.e. reservoirs) system; mean DN:TN = 0.711, sd. = 0.10

(3)Same method as in footnote 2 above; mean DP:TP = 0.336, sd. = 0.08.

(4)Same method as in footnote 2 above; mean OP:TP = 0.320, sd. = 0.004.

(5)Computed by applying the mean unit area sediment load from the Potomac and James (497.0 lbs/mi^2/day) to the total drainage area of the minor tributaries (3874 mi^2), as measured at the USGS gauges used by Guide and Villa (1972). The standard deviation of mean areal loading rate = 29.4.

(6)Approximate annual mean daily flow for the Rappahannack, Mattaponi, Paumunkey, Patuxent, and Chickahominy Rivers from various USGS Water Resources Data Publications. The drainage area above the collected gauges is about 3874 square miles.

TABLE III.5(b). ESTIMATED PERCENTAGE OF TOTAL ANNUAL NUTRIENT AND SEDIMENT LOADS FROM CHESAPEAKE BAY TRIBUTARIES

Constituent	Susquehanna	Potomac	James	Other Tribs.
TN	73.2	20.3	6.2	4.4
DN	74.1	17.9	4.5	3.5
NO23	75.0	18.6	3.4	3.0
NH34	78.3	12.8	5.9	3.0
TKN	62.1	20.0	10.9	7.0
TP	55.6	22.2	16.1	6.0
DP	48.4	21.8	22.7	7.2
OP	55.5	20.6	17.8	6.1
SED	40.0	33.0	16.4	10.6
Discharge	67.0	16.9	10.6	5.5

TABLE III.6(a). ESTIMATED WINTER MEAN DAILY NUTRIENT AND SEDIMENT LOADS TO THE CHESAPEAKE BAY SYSTEM FROM SOURCES TRANSPORTED BY RIVERS (ALL VALUES x 10^3 LBS/DAY UNLESS OTHERWISE INDICATED)

Constituent	Susquehanna	Potomac	James	Other Tribs.	Total Fluvial Load to the Bay System
TN	397.20	95.27	29.11	20.09(1)	571.29
DN	356.44	90.58	24.00	14.18(2)	485.2
NO23	264.94	69.09	13.34	10.74	358.11
NH34	22.47	3.87	1.89	0.87	29.10
TKN	115.51	39.14	22.93	9.35	186.93
TP	17.83	7.59	5.89	1.65	32.96
DP	4.44	2.11	2.13	0.53(3)	9.21
OP	5.72	2.23	1.91	0.51(4)	10.37
SED	8,121.18	6,295.99	3,616.87	2,174.65(5)	20,208.69
Discharge	50,109.6cfsd	13,286.9cfsd	8,811.7cfsd		

(1) Computed as the sum of $NO_{2,3}$ + TKN.

(2) Estimated using methodology shown in TABLE III.9(a), footnote (2). Winter mean DN:TN = 0.706, SD = 0.11.

(3) Estimated as in TABLE III.9(a), footnote (3). Potomac and James Winter mean DP:TP = 0.320, SD = .02.

(4) Estimated as in TABLE III.9(a), footnote (4). Potomac and James Winter mean OP:TP = 0.304, SD = .02.

(5) Estimated as in TABLE III.9(a), footnote (5). Potomac and James Winter mean areal sediment loading rate = 561.34 lbs/mi^2/day), SD = 23.6.

TABLE III.6(b). ESTIMATED PERCENTAGE OF WINTER NUTRIENT AND SEDIMENT LOADS LOADS FROM CHESAPEAKE BAY TRIBUTARIES

Constituent	Susquehanna	Potomac	James	Other Tribs.
TN	69.5	20.3	6.6	3.5
DN	73.5	18.7	4.9	2.9
NO23	74.0	19.3	3.7	3.0
NH34	77.2	13.3	6.5	3.0
TKN	61.8	20.9	12.3	5.0
TP	54.1	23.0	17.9	5.0
DP	48.2	22.9	23.1	5.7
OP	55.2	21.5	18.4	4.9
SED	40.2	31.2	17.9	10.8

TABLE III.7(a). ESTIMATED SPRING MEAN DAILY NUTRIENT AND SEDIMENT LOADS TO THE CHESAPEAKE BAY SYSTEM FROM SOURCES TRANSPORTED BY RIVERS (ALL VALUES x 10^3 LBS/DAY UNLESS OTHERWISE INDICATED)

	Total Fluvial				
Constituent	Susquehanna	Potomac	James	Other Tribs.	Load to the Bay System
TN	546.10	166.84	44.50	27.58(1)	785.02
DN	485.60	131.18	27.89	19.50(2)	664.17
NO23	363.46	98.80	15.55	14.78	492.59
NH34	31.42	5.69	2.20	1.22	40.53
TKN	159.32	56.94	26.94	12.8	256.00
TP	26.08	11.39	6.87	2.33	46.67
DP	6.07	3.01	2.36	0.71(3)	12.15
OP	7.93	3.16	2.15	0.69(4)	13.93
SED	12,110.89	11,556.71	4,232.68	3,247.76(5)	31,148.04
Discharge	68,011.5cfsd	18,466.8cfsd	10,209.3cfsd		

(1)Computed as the sum of $NO_{2,3}$ + TKN.

(2)Estimated using methodology shown in TABLE III.9(a), footnote (2). Spring mean DN:TN = 0.707, SD = 0.11.

(3)Estimated as in TABLE III.9(a), footnote (3). Potomac and James Spring mean DP:TP = 0.304, SD = .06.

(4)Estimated as in TABLE III.9(a), footnote (4). Potomac and James Spring mean OP:TP = 0.295, SD = .03.

(5)Estimated as in TABLE III.9(a), footnote (5). Potomac and James Spring mean areal sediment loading rate = 838.09 lbs/mi^2/day), SD = 228.6.

TABLE III.7(b). ESTIMATED PERCENTAGE OF SPRING NUTRIENT AND SEDIMENT LOADS FROM CHESAPEAKE BAY TRIBUTARIES

Constituent	Susquehanna	Potomac	James	Other Tribs.
TN	69.6	21.3	5.7	3.5
DN	73.1	19.8	4.2	2.9
NO23	73.8	20.1	3.2	3.0
NH34	77.5	14.0	5.4	3.0
TKN	62.2	22.2	10.5	5.0
TP	55.9	24.4	14.7	5.0
DP	50.0	24.8	19.4	5.8
OP	56.9	22.7	15.4	4.9
SED	38.9	37.1	13.6	10.4

TABLE III.8(a). ESTIMATED SUMMER MEAN DAILY NUTRIENT AND SEDIMENT LOADS TO THE CHESAPEAKE BAY SYSTEM FROM SOURCES TRANSPORTED BY RIVERS (ALL VALUES x 10^3 LBS/DAY UNLESS OTHERWISE INDICATED)

Constituent	Susquehanna	Potomac	James	Other Tribs.	Total Fluvial Load to the Bay System
TN	195.66	49.07	16.83	11.38(1)	272.94
DN	178.06	37.66	11.32	8.19(2)	235.23
NO23	130.92	29.64	6.14	5.16	171.86
NH34	10.87	1.56	0.87	0.41	13.71
TKN	56.66	16.10	9.94	6.22	88.92
TP	8.92	2.92	2.69	0.93	15.46
DP	2.21	0.91	1.38	0.38(3)	4.88
OP	2.78	0.98	1.16	0.36(4)	5.28
SED	4,398.53	2,531.19	2,318.98	1,142.02(5)	10,390.72
Discharge	25,193.3cfsd	6,147.2cfsd	4,248.2cfsd		

(1)Computed as the sum of $NO_{2,3}$ + TKN.

(2)Estimated using methodology shown in TABLE III.9(a), footnote (2). Summer mean DN:TN = 0.720, SD = 0.07.

(3)Estimated as in TABLE III.9(a), footnote (3). Potomac and James Summer mean DP:TP = 0.412, SD = 0.142.

(4)Estimated as in TABLE III.9(a), footnote (4). Potomac and James Summer mean OP:TP = 0.383, SD = 0.07.

(5)Estimated as in TABLE III.9(a), footnote (5). Potomac and James Summer mean areal sediment loading rate = 294.79 lbs/mi^2/day), SD = 107.24

TABLE III.8(b). ESTIMATED PERCENTAGE OF SUMMER NUTRIENT AND SEDIMENT LOADS FROM CHESAPEAKE BAY TRIBUTARIES

Constituent	Susquehanna	Potomac	James	Other Tribs.
TN	71.7	18.0	6.2	4.2
DN	75.7	16.0	4.81	3.5
NO23	76.2	17.3	3.6	3.0
NH34	79.3	11.4	6.4	3.0
TKN	63.7	18.1	11.2	7.0
TP	57.7	18.9	17.4	6.0
DP	45.3	18.6	28.3	7.9
OP	52.7	18.6	22.0	6.8
SED	42.3	24.4	22.3	11.0

TABLE III.9(a). ESTIMATED FALL MEAN DAILY NUTRIENT AND SEDIMENT LOADS TO THE CHESAPEAKE BAY SYSTEM FROM SOURCES TRANSPORTED BY RIVERS (ALL VALUES x 10^3 LBS/DAY UNLESS OTHERWISE INDICATED)

Constituent	Susquehanna	Potomac	James	Other Tribs.	Total Fluvial Load to the Bay System
TN	228.17	48.85	17.23	12.79(1)	307.04
DN	207.42	37.86	11.44	9.02(2)	265.92
NO23	152.66	29.29	6.23	5.82	194.00
NH34	12.62	1.60	0.88	0.47	15.57
TKN	66.06	16.29	10.22	6.97	99.54
TP	9.56	3.11	2.74	0.98	16.39
DP	2.58	0.89	1.34	0.38(3)	5.19
OP	3.42	0.96	1.13	0.36(4)	5.87
SED	4,311.54	3,514.34	1,757.22	1,132.85	10,715.95
Discharge	29,317.5cfsd	5,883cfsd	4,271.7cfsd		

(1)Computed as the sum of $NO_{2,3}$ + TKN.

(2)Estimated using methodology shown in TABLE III.9(a), footnote (2). Fall mean DN:TN = 0.719, SD = 0.08.

(3)Estimated as in TABLE III.9(a), footnote (3). Potomac and James Fall mean DP:TP = 0.388, SD = 0.14.

(4)Estimated as in TABLE III.9(a), footnote (4). Potomac and James Fall mean OP:TP = 0.361, SD = 0.07.

(5)Estimated as in TABLE III.9(a), footnote (5). Potomac and James Fall mean areal sediment bed = 292.43 lbs/mi^2/day), SD = 16.38

TABLE III.9(b). ESTIMATED PERCENTAGE OF FALL NUTRIENT AND SEDIMENT LOADS FROM CHESAPEAKE BAY TRIBUTARIES

Constituent	Susquehanna	Potomac	James	Other Tribs.
TN	74.3	15.9	5.6	4.2
DN	78.0	14.2	4.3	3.5
NO23	78.7	15.1	3.2	3.0
NH34	81.1	10.3	5.7	3.0
TKN	66.4	16.4	10.3	7.0
TP	58.3	19.0	16.7	6.0
DP	49.7	17.2	25.8	7.3
OP	58.3	16.4	19.3	6.1
SED	40.2	32.8	16.4	10.6

NUTRIENT INPUTS FROM SELECTED MINOR TRIBUTARIES

Guide and Villa (1972) reported seasonal and annual nutrient loadings from five of the next largest tributaries (after the Susquehanna, Potomac, and James) of the western shore of Chesapeake Bay. These basins are the Patuxent, Rappahannock. Mattaponi, Paumunkey, and Chickahominy Rivers, that together, drain parts of the Blue Ridge, Piedmont, and Coastal physiographic provinces (Figure III.2). Loading estimates by Guide and Villa were made for the areas of the tributaries that drain above USGS discharge monitoring stations. The land-area contributing to these estimates totaled 3,874 square miles, or about 6.1 percent of the total Bay drainage basin, with an accummulated mean daily discharge of about 3,535 cfsd.

Guide and Villa observed, for the period June, 1969, through August, 1970, that these five basins generally contributed about five percent or less of the nutrient loading of various nitrogen and phosphorus species. They found that for the entire period of observation those minor tributaries contributed six percent, seven percent, three percent, and three percent of TP, TKN, $NO_{2,3}$, and $HN_{3,4}$ loads respectively [see Table III.5(b)]. Similarly, they found that for the winter and spring months, these basins contributed five percent, five percent, three percent, and three percent of TP, TKN, $NO_{2,3}$, and $NH_{3,4}$ loads respectively. All loading estimates in that study were performed using log-linear models of loading rate versus mean daily discharge developed with bivariate least squares techniques. These methods were very similar to those used in the previous section of this chapter and described in Appendix A.

Estimates of nutrient loadings were developed from the minor western shore tributaries by utilizing the percentages reported by Guide and Villa (1972) in conjunction with the estimate made in the previous section of the loadings from the three major tributaries. For example, the annual mean daily TP load from the three major tributaries is estimated in Table III.5(a) to be 2.64×10^4 pounds. If it is assumed, after Guide and Villa (1972), that 6 percent of the total phosphorus load comes from the minor tributaries, then the 2.64×10^4 pounds of phosphorus should be about 94 percent of the total load. The total TP load, therefore, should be about 2.81×10^4 pounds per day, and by difference, the load from the minor tributaries about 1.69×10^3 pounds per day.

The annual and seasonal expected daily nutrient loadings from the minor tributaries have been computed in the manner described above and are presented in the fifth column of Tables III.5(a), III.6(a), III.7(a), III.8(a), and III.9(a). The estimates shown in these tables for the minor-basin DN, DP, and OP loadings were made, based upon the mean of the DN:TN, DP:TP, and OP:TP ratios of the Potomac and James[1]. The estimates for the minor tributaries' sediment loads were made by computing the mean areal (per unit area) sediment loads on the Potomac and James [1] and applying

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[1]The Susquehanna loading ratios and areal sediment yield rates were not used because that river system is regulated by the reservoirs in the downstream main stem. Lang (1982) notes that the Susquehanna reservoirs cause sediment deposition and transformation among nutrient species. These transformations would not normally occur in free flowing streams like the minor tributaries under consideration.

them to the 3874 square mile drainage area of the five small tributaries. The calculations used for both these methods are shown in the footnotes of each table.

COMPUTATION OF TOTAL NUTRIENT AND SEDIMENT LOADS FROM RIVERINE TRANSPORTED SOURCES

The loading rates for each of the major tributaries and the estimates of those from the minor tributaries are included in the 'a' parts of Tables III.5 through III.9. These sources are summed to form the sixth column of those tables for the annual and for each of the four seasonal loading rates. The percentage that each of these sources contributes to the total load has been computed and is included as the 'b' parts of Tables III.5 through III.9.

Inspection of the loading percentages shown in Table III.5(b) reveals that the Susquehanna probably carries about 70 percent of the total nitrogen and 56 percent of the total phosphorus delivered to the Bay each year from riverine-borne sources. Most of these loadings are carried during the winter and spring seasons [Table III.6(a) and III.7(a)]. The predominent form of fluvial-transported nitrogen entering the system is nitrate + nitrite, with this effect most pronounced in the spring [Table III.7(a)]. Phosphorus enters the system from riverine-transported sources primarily in the suspended phase.

The Susquehanna produces a much smaller fraction of the total riverine-borne phosphorus load than that of nitrogen, contributing about 50 percent in the winter and 58 percent of the total load in the summer. The same trend occurs when considering the sediment loads to the Bay, with the Susquehanna producing only about 40 percent in any season -- usually less than that transported by the Potomac and James taken together. The small fraction of both the phosphorus and sediment loads produced by the Susquehanna relative to its drainage area and flow, no doubt, is due to the trapping of particulate matter in the reservoirs located on the lower sixty miles of the main stem of the river. For example, in an average spring season, the Potomac and James taken together contribute daily about 840 pounds of suspended sediment per square mile of drainage while the Susquehanna would produce only about 447 pounds per square mile, or roughly about half as much[1].

The data included in Tables III.5(a) through III.9(a) were used to generate total expected seasonal and annual riverine-borne mass loadings of nutrients and sediments to the Bay system. The results of these computations are found in Table III.10.

Fluvial transported loadings are compared with other nutrient sources in Chapter VIII. For example, in Table VIII.3(b) shows that riverine transported sources provide the largest proportion of all nutrients entering the Bay system in spring, with the exception of orthophosphate.

In summary, stream-transported loading estimates have been computed for the Chesapeake Bay system. These estimates are well within order of magnitude accuracy and are suitable for comparison with the estimated loads from other sources discussed in this paper.

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[1]Lang (1982) notes that the Susquehanna system probably begins to scour (deliver to the Bay) the sediment stored in the reservoirs at flows above 400,000 cfs. Flows that large occur only less than one percent of the time, however. Most of the time the reservoirs act as an efficient sediment trap.

TABLE III.10. SEASONAL AND ANNUAL NUTRIENT AND SEDIMENT LOADS TRANSPORTED BY RIVERS TO THE TIDAL CHESAPEAKE BAY SYSTEM, (MILLIONS OF POUNDS UNLESS OTHERWISE NOTED)

Constituent	Winter	Spring	Summer	Fall	Annual
TN	51.4	72.2	25.1	27.9	178.1
DN	43.7	61.1	21.6	24.2	151.7
NO23	32.2	45.3	15.8	17.7	111.47
NH34	2.62	3.73	1.26	1.42	9.06
TKN	16.8	23.6	8.18	9.06	58.6
TP	2.97	4.29	1.42	1.49	10.3
DP	0.829	1.12	0.449	0.472	2.907
OP	0.933	1.28	0.486	0.534	3.24
SED	1.83×10^9	2.87×10^9	1.07×10^9	9.75×10^8	6.63×10^9

The Tables (A-2, A-3, and A-4), presented in Appendix A, show that poor fits ($r^2 < 0.50$) were found in almost all cases, for the concentration-versus-discharge models. In most cases visual inspection of scatter diagrams allows a case to be made for heterosodasticity and, for this reason, the variance-stabilizing transformation was favored in selecting appropriate models.[1] Only when correlation coefficients were significantly below 0.65, or 't' tests ($H_0:B_1 = 0$) indicated that B_1, the slope, was not significantly different from zero at the 95 percent confidence level, was a loading rate model chosen.

[1]During the course of examination of the concentrations, predicted by each of the models over the range of flow observed in the period of record, it was determined that the arithmetic form of the variance-stabilizing transformation (C/Q versus 1/Q) yielded unrealistically high values for discharges, in excess of those observed during the period of the monitoring program. The log-log transformation of this model [ln(C/Q) versus ln(1/Q)] proved to be much better behaved in predicting concentrations for these higher flows. The curves produced with this transformation 'flatten out' very quickly, as flows approach those at the upper limit of the discharge data, observed during the field program. Therefore, only the log transformed versions of the variance-stabilizing transformation were considered for cases exhibiting heterosodasticity.

SECTION IV

POINT SOURCE LOADINGS OF NUTRIENTS

Water quality managers and researchers typically divide sources of water pollution into two broad categories: point and nonpoint. Although the distinction between them is not always clear, point sources are generally described as those discharging to a water body from a discrete pipe or ditch. Examples of point sources are municipal sewage treatment plant discharges, industrial discharges, and combined sewer overflows. Nonpoint sources arise from multiple causes and can be dramatically affected by rainfall and storms. Examples of nonpoint sources are runoff from urban and suburban storm sewers, agricultural activities, forestry activities, and atmospheric deposition.

The objective of this Section is to estimate the load of nutrients discharged to the Bay system from point sources. Table IV.1 lists the nutrients analyzed for estimating loads. Municipal and industrial point source loadings are estimated separately. Estimations are made above and below the head of tide, or fall line, for the river systems (major/minor) discharging to Chesapeake Bay. Loadings below the fall line represent point source loads to the tidal Bay system in excess of what was computed in riverine loads of Section 3.

The river basins that make up the Bay's 64,000 square mile drainage area are delineated by EPA in its STORET data system and are illustrated in Figure IV.1. (STORET is a computerized data base maintained by EPA for the storage and retrieval of parametric data, relating to the quality of the waterways of the United States.) The "fall line"[1] is delineated by USGS hydrologic units (USGS, office of Water Data Coordination in consultation with the U.S. Water Resource Council) and is also illustrated in Figure IV.1. Section III discussed point sources of nutrients discharging above the fall line, reflected in the fluvial loads computed at the fall line monitoring stations of the Bay's three major tributaries. Therefore, for this section, it was important to know which point sources discharge above the line and which discharge below, so that a double counting could be avoided. For this analysis, loads generated above the fall line are included in the fall line monitoring data, but those generated below the fall line are

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[1]The "fall line" defined for the purpose of this Section is not the true geologic Fall Line as defined in Section I. The functional definition of the fall line used in this paper is the line of demarcation below the drainage of the three major tributaries' monitoring stations (Susquehanna at Conowingo, MD; Potomac at Chain Bridge, DC; James at Cartersville, VA) described in Chapter III. All point sources discharging downstream of this line are considered not to have been accounted for in the loads computed in the previous Chapter. This definition assumes that all the point source loadings from the Rappahannock and York Rivers (Pamunkey and Mattaponi) are discharged below the monitoring stations employed by Guide and Villa (1972), or that they have begun discharging since 1971 and were not incorporated in the loads monitored during that study. In any event, the potential for a double counting error is small as can be seen from the data in Tables IV.12(a) and IV.12(b).

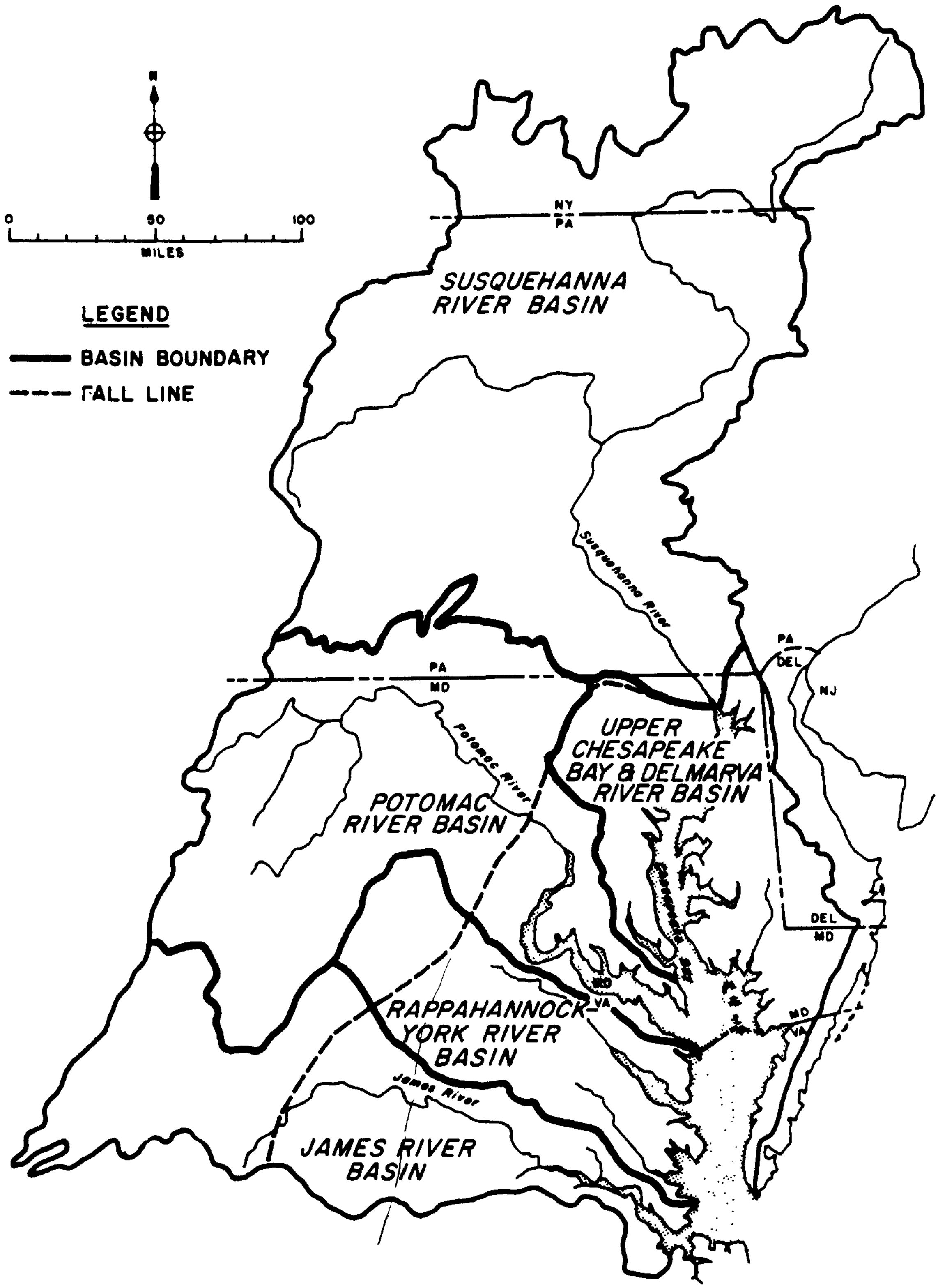

Figure IV.1. River systems discharging to Chesapeake Bay. Dashed line indicates the USGS fall line.

considered to discharge directly to the tidal waters of the Bay. Table IV.1 identifies the USGS hydrologic units included below the fall line of each major drainage basin. Hydrologic units, defined by the U.S. Geological Survey, in cooperation with the U.S. Water Resources Council, delineate the hydrographic boundaries of major river basins in the United States. They provide a standard geographical framework for detailed, water-related planning and serve as an aid to organizing and disseminating data. Once point source loadings above and below the fall line for each drainage basin are calculated, they can be compared to nonpoint source loadings, and the relative contribution of each determined.

TABLE IV.1. WATER QUALITY VARIABLES

Water Quality Variable	Variable Code
Total Nitrogen (as N)	TN
Total Kjeldahl Nitrogen (as N)	TKN
Total Nitrite plus Total Nitrate Nitrogen (as N)	NO2 + NO3 or NO23
Total Ammonia Nitrogen (as N)	NH34
Organic Nitrogen	ORGN
Total Phosphorus (as P)	TP
Total Orthophosphorous (as P)	OP

TABLE IV.2. USGS HYDROLOGIC UNITS BELOW THE FUNCTIONALLY-DEFINED FALL LINE OF THE CHESAPEAKE BAY DRAINAGE BASIN

Drainage Basin	USGS hydrologic units included below fall line
Susquehanna 0212	
Upper Chesapeake Bay & Delmarva 0213	02 - 06 - 00 - 02
	02 - 06 - 00 - 03
	02 - 06 - 00 - 04
	02 - 06 - 00 - 05
	02 - 06 - 00 - 06
	02 - 06 - 00 - 07
	02 - 06 - 00 - 08
	02 - 06 - 00 - 09
	02 - 08 - 01 - 09

(continued)

TABLE IV.2. (continued)

Potomac	02 - 07 - 00 - 10
0214	02 - 07 - 00 - 11
Rappahannock/York	02 - 08 - 01 - 02
0215	02 - 08 - 01 - 03
	02 - 08 - 01 - 04
	02 - 08 - 01 - 05
	02 - 08 - 01 - 06
	02 - 08 - 01 - 07
James	02 - 08 - 02 - 05
0216	02 - 08 - 02 - 06
	02 - 08 - 02 - 07
	02 - 08 - 02 - 08
	02 - 08 - 01 - 08

Estimation of Nutrient Loads from Municipal Point Sources

The basic strategy for estimating nutrient loads from municipal point sources or publicly owned treatment works (POTWs), called for merging computerized data bases and accessing state and facility effluent monitoring data. The data bases merged included the EPA 1980 Needs Survey (Needs) and the Industrial Facilities Dischargers (IFD) file. The 1980 Needs Survey is performed in compliance with the provisions of Sections 205 (a) and 516 (b)(2) of the Clean Water Act Amendments of 1977, PL 95-217. The Survey collects technical and administrative data on new and existing POTWs, which then serve as a basis for Congressional allotment of construction grant funds among the states. The Needs data base provided an inventory of existing and projected flows, and of levels of treatment for POTWs. The IFD file is a comprehensive data base on municipal and industrial dischargers assembled by the Monitoring and Data Support Division of EPA. It was used to verify the Needs file and furnished valuable locational information.

Although the merging of these data bases generated an inventory of POTWs and provided a substantial amount of information concerning their flow, level of treatment, and location, it did not provide information concerning the concentration of nutrients in effluents. To obtain this information, we began a systematic analysis of the CBP-generated data base. This analysis determined the percentage of total flow contributed by POTWs in different flow (size) categories. It indicated that there are 580 POTWs located within the Chesapeake Bay drainage basin having a combined flow of 1350 million gallons/day (MGD). Further analysis revealed that 96 percent of this flow is contributed by the 197 POTWs larger than 0.5 MGD, and 78 percent of the flow by the 47 POTWs larger than 5.0 MGD. Based on this analysis and existing data needs, we requested necessary information from the Maryland Department of Health and Mental Hygiene, Office of Environmental Programs (OEP), the Virginia State Water Control Board (VSWCB), and the Pennsylvania Department of Environmental Resources (DER).

Each State staff was requested to provide 1980 data on operational flow, total nitrogen (TN), total phosphorus (TP), five-day biological oxygen demand (BOD5), and total suspended solids (SED) concentration for the POTWs larger than 5.0 MGD within their political boundaries. In addition, the tasks were intended to determine the status of certain dischargers that were in question. State expertise in these areas was invaluable. In the meantime, estimates of the concentration of these pollutants in wastewater after different levels of treatment were made (Barth 1981)[1] and included in the CBP data base. They are presented in Table IV.3. This table shows that for BOD, a lot of difference exists between primary and secondary treatment. However, to obtain decreases in N and P, tertiary treatment must probably be used. Later, program requirements dictated that the TN and TP estimates be broken down into various species. This information (Barth 1981)[1] is presented in Table IV.4. Again, not until AWT is used, will any significant decreases in N and P occur.

The response from States staffs was very good, and we updated the CBP data base with the provided information. In addition, we contacted several POTW operators and requested actual data or estimates of nutrient concentrations in their effluent. This information was also added to the CBP data base and is presented in Table IV.5.

Using the updated CBP data base and our functional definition of the fall line, we calculated nutrient loads from POTWs above and below the fall line. This information is presented in Table IV.6 and is currently undergoing final review by the states' staffs. This table shows that the largest loadings above the fall line are discharged within the Susquehanna drainage basin and, with the exception of TN (Potomac 57,489 lbs/day vs. James 43,770 lbs/day), the largest loadings below the fall line are discharged within the James drainage basin. The smallest loadings above the fall line are discharged within the Upper Chesapeake Bay - Delmarva drainage basin and below the fall line within the Rappahanock - York drainage basin. The large loadings from the Susquehanna indicate that its basin is largely located above the fall line, but the small loadings from the Upper Chesapeake Bay - Delmarva indicate it is mostly located below the fall line. The small load from the Rappahannock/York is due to lack of development. It is interesting to note that although the Potomac drainage basin receives the greatest total volume of treated wastewater (589 MGD), its total TP load (9,583 loads/day) is less than that from the Susquehanna (16,052 lbs/day) and James (11,920 lbs/day). This results from the large volume of wastewater undergoing phosphorus removal at the Blue Plains POTW.

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[1] Personal Communication: "Fractions of Nitrogen and Phosphorous in Effluents," and others, E.F. Barth, Biological Treatment Section, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, Cincinnati, OH, 1981.

TABLE IV.3. RANGE OF POTW CONSTITUENTS CONCENTRATIONS BASED ON LEVEL OF TREATMENT (MG/L) (SOURCE: BARTH 1981)

Treatment Level	BOD_5	SED	TN	TP
None (Raw Discharge)	210 - 300	230 - 300	15 - 30	9 - 11.5
Primary[1]	130 - 140	100 - 130	13.5 - 28	9 - 10
Advanced Primary[2]	60 - 65	40 - 52	12 - 25	8 - 9
Secondary[3]	20 - 30	20 - 30	12 - 25	7 - 9
Advanced Secondary (Nitrification)	10 - 20	10 - 20	10 - 20	1 - 2
Tertiary (Nitrogen removal and P removal)	5 - 10	5 - 10	3 - 10	0.1 - 2

[1]Preliminary treatment (bar screen and grit removal) and primary sedimentation.

[2]Primary treatment with post aeration.

[3]Activated sludge, rotating biological contactors, or low-rate trickling filters.

TABLE IV.4. ESTIMATE OF DISTRIBUTION OF POTW NITROGEN AND PHOSPHORUS INTO VARIOUS FRACTIONS ACCORDING TO SELECTED TREATMENT PROCESS (MG/L) (SOURCE: BARTH 1981) THE TN AND TP VALUES IN THIS TABLE REPRESENT THE AVERAGE OF THE RANGE OF TN AND TP CONCENTRATIONS FROM TABLE IV.3

	Nitrogen Fractions					Phosphorus Fractions		
Treatment	Org-N	TKN	NH34	NO23	TN	Insol + Poly	OP	TP
None (Raw discharge)	9	22.5	13.5	0	22.5	6.75	3.5	10.25
Primary	7.0	20.75	13.75	0	20.75	5.25	4.25	9.5
Advance Primary	6.5	18.5	12	0	18.5	4.25	4.25	8.5
Secondary	3.0	16.5	13.5	2.0	18.5	1.2	6.8	8.0
AST	2.0	3.0	1.0	12	15	0.5	1.0	1.5
AWT	1.5	2.5	1.0	4.0	6.5	-	1.05	1.05

TABLE IV.5 Updated nutrient data on municipal point sources
(All values in mg/L except Flow in MGD)

Drainage Basin	State	NPDES	Facility Name	Flow (80)	TP	OP	TN	TKN	NH34	NO23
James	VA	0025208	Army Base W P C F	12.38	5.60	2.20	-	23.20	17.50	-
James	VA	0025283	Boat Harbor W P C F	19.06	3.50	1.60	-	20.90	17.70	-
James	VA	0025275	Chesapeake-Elisabeth WCPF	23.12	6.10	5.30	-	27.00	22.20	-
James	VA	0025640	Hopewell STP	36.30	-	-	-	-	45.00	-
James	VA	0025241	James River W P C F	14.27	7.40	6.60	-	7.50	6.70	4.17
James	VA	0025259	Lamberts Point W P C F	20.63	4.50	1.34	-	23.20	18.40	-
James	VA	0025437	Petersburg STP (Va)	11.44	6.21	-	-	20.06	19.00	-
Potomac	VA	0025160	Alexandria STP (Va)	26.96	0.92	-	-	-	-	-
Potomac	VA	0025143	Arlington Co W P C P	22.27	3.08	2.80	-	-	-	-
Potomac	DC	0021199	Blue Plains STP	317.00	1.20	-	-	5.70	4.16	6.46
Potomac	DC		Blue Plains Bypass	27.60	4.60	-	-	16.94	-	-
Potomac	MD	0021598	Cumberland W W T P	9.10	4.13	-	15.35	-	0.50	13.60
Potomac	MD	0021610	Frederick City W W T P	4.40	7.62	-	22.60	-	16.60	0.05
Potomac	MD	0021776	Hagerstown W P C F	4.50	4.56	-	28.30	-	11.30	2.10
Potomac	MD	0020214	Halfway Subdist No 1 W W T P	0.90	8.49	-	18.08	-	10.50	2.63
Potomac	MD	0021733	Horsepen Sewage System		-	-	-	1.17	-	9.96
Potomac	MD	0020524	La Plata W W T P	0.20	3.26	-	5.83	-	0.40	4.40
Potomac	MD	0024767	Leonardtown STP	0.25	6.33	-	25.02	-	14.00	0.30
Potomac	VA	0025364	Lower Potomac STP	17.86	2.29	-	-	-	-	-
Potomac	MD	0023957	Md Correctional Institute	0.36	-	-	-	1.25	0.07	7.39
Potomac	MD	0021725	Parkway STP	4.90	1.61	-	13.00	6.13	0.50	6.87
Potomac	MD	0021539	Piscataway W W T P	15.00	0.72	-	22.40	-	16.00	1.00
Potomac	MD	0021491	Seneca Creek Interm AWT	4.70	0.50	-	17.22	3.50	0.04	16.60
Potomac	MD	0020672	Taneytown STP	0.24	8.00	-	18.94	-	2.40	14.90
Potomac	MD	0021121	Thurmont W W T P	0.40	4.80	-	17.53	-	10.70	1.06
Potomac	VA	0024988	Upper Occoquan	7.25	0.03	-	18.12	0.40	-	-
Potomac	MD	0021687	UPRC W W T P	22.40	0.88	-	-	-	-	-
Potomac	MD	0021741	Western Branch W W T P	12.60	1.90	5.00	11.13	4.20	21.20	8.50
Potomac	VA	0025399	Westgate Pumpover	8.39	4.60	-	-	-	-	-
Potomac	MD	0021831	Westminster W W T P	1.60	3.79	-	7.53	-	2.20	4.50
UP CB Delmarva	MD	0021563	Aberdeen W W T P	1.00	7.45	-	24.17	-	17.80	1.05
UP CB Delmarva	MD	0021814	Annapolis City STP	4.70	0.45	-	20.80	-	16.00	1.40
UP CB Delmarva	MD	0021555	Back River W W T P	80.60	5.60	-	19.44	-	-	-
UP CB Delmarva	MD	0021628	Bowie W W T P	2.50	8.80	-	27.80	-	18.90	0.15
UP CB Delmarva	MD	0021644	Broadneck W W T P	2.10	4.53	-	15.62	-	12.80	1.65
UP CB Delmarva	MD	0024350	Broadwater W W T P	0.14	0.57	-	14.76	-	1.75	11.75
UP CB Delmarva	MD	0021636	Cambridge W W T F	4.40	4.59	-	11.94	-	6.80	3.16
UP CB Delmarva	MD	0020834	Centreville W W T P	0.22	3.96	-	12.00	-	5.80	4.20
UP CB Delmarva	MD	0021661	Cox Crk W W T P	6.40	7.93	-	27.40	-	20.50	0.06
UP CB Delmarva	MD	0020001	Crisfield W W T P	0.76	4.40	-	6.86	-	1.00	3.70
UP CB Delmarva	MD	0020273	Easton W S L	1.80	8.60	-	10.22	-	1.20	0.30
UP CB Delmarva	MD	0020681	Elkton W W T P	0.80	7.05	-	24.81	-	13.80	1.56
UP CB Delmarva	MD	0020249	Federalsburg W W T F	0.22	1.75	-	6.14	-	0.20	4.70
UP CB Delmarva	MD	0021512	Freedom District W W T P	0.80	6.75	-	6.53	-	3.20	2.30
UP CB Delmarva	MD	0022446	Hampstead W W T P	0.15	-	-	15.47	-	9.00	3.00
UP CB Delmarva	MD	0022730	Hurlock W W T F	1.10	3.50	-	13.50	-	3.67	0.80
UP CB Delmarva	MD	0023132	Md City W W T P	0.70	9.60	-	21.79	17.80	11.98	7.26
UP CB Delmarva	MD	0021601	Patapsco W W Trtmt Plant	19.50	6.48	-	16.50	-	15.00	0.12
UP CB Delmarva	MD	0021652	Patuxent W W T P	3.60	5.53	-	20.96	18.80	16.41	1.10
UP CB Delmarva	MD	0021679	Pine Hill Run W W T P	2.20	5.67	-	18.24	15.50	11.50	2.56
UP CB Delmarva	MD	0020656	Princess Anne W W T P	0.27	7.95	-	20.64	-	13.14	0.46
UP CB Delmarva	MD	0023370	Queenstown W W T P	0.06	4.43	-	16.60	-	11.80	1.70
UP CB Delmarva	MD	0021571	Salisbury W W T F	3.50	5.58	-	24.64	-	19.30	1.80
UP CB Delmarva	MD	0021547	Savage W W T P	8.50	9.60	-	22.57	18.40	11.30	1.76
UP CB Delmarva	MD	0022764	Snow Hill W W T P	0.40	5.81	-	18.50	-	10.60	0.34
UP CB Delmarva	MD	0021709	Sod Run W W T P	2.90	1.50	-	20.40	-	12.10	1.93
UP CB Delmarva	MD	0023051	Woodland Beach W W T P	0.56	6.37	-	17.6	-	4.2	11.1

TABLE IV.6. ESTIMATES OF NUTRIENT LOADS FROM MUNICIPAL POINT SOURCES ABOVE AND BELOW THE FUNCTIONALLY-DEFINED FALL LINE (ALL VALUES IN LBS/DAY EXCEPT FLOW IN MGD)

Drainage Basin	Water Quality Parameter	Above	Below	Total
	BOD5	105899	134	106033
	TP	16018	34	16052
Susquehanna	OP	11504	29	11533
	TN	48098	85	48183
(0212)	TKN	33502	71	33573
	NO23	14596	14	14610
	NH34	24353	58	24411
	ORGN	9149	13	9162
	FLOW	329	.6	330
	BOD5	64	54824	54888
Upper	TP	7	8224	8231
Chesapeake	OP	4	5781	5785
Bay and	TN	41	26406	26447
Delmarva	TKN	16	12916	12932
	NO23	25	9482	9507
(0213)	NH34	8	10404	10412
	ORGN	7	4303	4310
	FLOW	.3	164	164.3
	BOD5	29972	50277	80249
	TP	2883	6700	9583
Potomac	OP	2555	5251	7806
	TN	13089	57489	70578
(0214)	TKN	8616	26764	35380
	NO23	3760	30298	34058
	NH34	6049	23445	29494
	ORGN	3014	9263	12277
	FLOW	87	502	589
	BOD5	355	2675	3030
	TP	55	576	631
Rappahannock/	OP	42	458	500
York	TN	310	1542	1852
	TKN	112	1196	1308
(0215)	NO23	198	346	544
	NH34	69	922	991
	ORGN	43	274	317
	FLOW	2.4	10.4	12.8

(continued)

TABLE IV.6. (continued)

	BOD5	7349	74688	82037
	TP	1574	10346	11920
James	OP	1218	7237	8455
	TN	3730	43770	47500
(0216)	TKN	3280	39303	42583
	NO23	450	7216	7666
	NH34	2567	32991	35558
	ORGN	713	6277	6990
	FLOW	24	231	255

Estimation of Nutrient Loads from Industrial Point Sources

Types of industrial activity with the potential to discharge the nutrients TP, TN, TKN, and $NH_{3,4}$ were identified through discussion with State and EPA officials. The Standard Industrial Classification (SIC) system, which classifies industries by their economic activity, was used to assign codes to these discharges. For example, industries engaged in the preparation of fresh or frozen packaged fish and seafoods were assigned SIC code 2092, the code corresponding to that particular economic activity. For industries engaged in petroleum refining, the SIC code assigned was 2911, the code denoting petroleum refinering as the primary economic activity. Table IV.7 lists the industrial economic activities considered to be nutrient generators and their corresponding SIC codes. The advantage of SIC codes is the speedy identification of all dischargers engaged in a particular economic activity.

The EPA/CBP computerized data base was accessed to retrieve the industries within the selected SIC - defined categories and located within the Chesapeake Bay drainage basin. This EPA data base includes: the Management Information Control System (MICS) - EPA Region III's (Philadelphia) computerized system containing basic information on all NPDES permitees; the Virginia NPDES permit file - the Virginia computerized system containing NPDES permit conditions, facility information, and discharge monitoring report (DMR) data; the National Enforcement Investigations Center (NEIC) system - an EPA data base generated by EPA's effort to define Major/Minor dischargers on a uniform national basis; and the already discussed IFD file.

Concentrations of nutrients expected to be found in the effluent from dischargers within a selected SIC category were obtained from EPA's Effluent Guideline Division (EGD) and the literature. Maryland 1979 NPDES permit compliance monitoring data and Virginia DMR's were also reviewed for observed nutrient data. Table IV.8(a) presents the nutrient concentrations estimated for the various SIC categories when observed data were absent. Table IV.8(b) identifies the source of the estimated concentrations.

Most flow data from the dischargers of interest were based on state DMR's or from NPDES permits. In some cases, flow data were not available from the sources and so were estimated from a particular industrial activity. Loads generated in this manner, however, constitute only a small percentage of the total loading from industrial sources and are identified as 'estimated' load (17 percent of TP and seven percent of TN). Table IV.9 presents assigned flows and the information source. Table IV.9 also identifies the data base providing observed/permitted flows.

Anomalies in the nutrient loadings computed with this approach were corrected by review of assigned concentrations and flows. In many cases, this resulted in close examination of an individual discharger and the assignment of more accurate nutrient loadings based on observed data. Table IV.10 lists these dischargers and their assigned nutrient loads. In addition, State officials familiar with dischargers within their jurisdiction have reviewed the loadings assigned to specific dischargers for reasonableness and completeness. The industrial point source loadings calculated in this manner for each drainage basin above and below the fall line are presented in Tables IV.11(a), (b), and (c). These tables reveal several trends. Table IV.11(b) indicates that the largest TP and TN contributions from industrial point sources occur within the James drainage basin. Most of the TP load is contributed by several large meat rendering, poultry processing, and food processing plants. The large TN load is attributable to these same dischargers, and to the presence of petroleum refineries and a fertilizer manufacturer in the basin. For comparative purposes, the largest industrial load of TP (1906 lbs/day in the James) constitutes only 15.5 percent of the total industrial and municipal TP load below the fall line in the James. From this we can conclude that industrial point sources are relatively minor contributors of nutrients.

TABLE IV.7. SIC CODE AND ECONOMIC ACTIVITY

SIC Code	Economic Activity
2011	Meat Packing & Rendering
2016	Poultry Processing
2023	Condensed and Evaporated Milk
2024	Ice Cream and Frozen Desserts
2026	Fluid Milk
2033	Canned Fruits, Preserves and Jams
2035	Pickled Fruits and Vegetables
2037	Frozen Foods
2038	Frozen Specialties
2077	Animal and Marine Fats and Oils
2091	Canned and Cured Fish and Seafoods
2092	Fresh or Frozen Packaged Fish and Seafoods
2812	Industrial Inorganic Chemicals - Alkalines and Chlorine
2813	Industrial Inorganic Chemicals - Industrial Gases
2816	Industrial Inorganic Chemicals - Inorganic Pigments
2819	Industrial Inorganic Chemicals - Not Elsewhere Classified
2821	Plastics Materials, Synthetic Resins, & Elastomers
2822	Synthetic Rubber
2823	Synthetic Organic Fibers
2824	Industrial Organic Chemicals - Cyclic Crude & Pigments
2833	Medicinal Chemicals & Botanical Products
2869	Industrial Organic Chemicals - Not Elsewhere Classified
2873	Manuf. of Nitrogenic Fertilizers
2874	Manuf. of Phosphatic Fertilizers
2879	Pesticides & Agricultural products
2891	Adhesives & Sealants
2892	Explosives Manufacture
2893	Printing Ink
2911	Petroleum Refineries
3111	Leather Tanning and Finishing
3312	Blast Furnaces, Steel Works
3321	Gray Iron Foundries
3322	Malleable Iron Foundries
3411	Metal Can Manufacture
3471	Electroplating
3612	Power Distribution & Specialty Transformers
3621	Electrical Industry Apparatus
3644	Electric Lighting & Equipment
3674	Semiconductors & Related Devices
3679	Electronic Components
3662	Radio Detection Equipment & Apparatus
3731	Ship Building & Repair
3861	Photographic Equipment & Supplies
6515	Mobile Home Site Operators
7011	Hotels, Motels, and Tourist Courts
7215	Coin-Operated Laundries and Dry Cleaning
8211	Elementary and Secondary Schools
8221	Colleges and Universities

TABLE IV.8(a). SIC CODE AND ESTIMATED CONCENTRATIONS OF WATER QUALITY CONSTITUENTS (mg/L)

SIC Code	SED	BOD5	TP	TN	NH_3	TKN
2011	67	68	20			8.6
2016	90.36	130.9	7.67	43.61	21.2	42.9
2023	157	338	10.8			61.2
2024	157	338	10.8			61.2
2026	157	338	10.8			61.2
2033	302	503	190.8			18
2037	302	503	190.8			18
2038	302	503	190.8			18
2077		18.8	7.1			8.5
2091	520	942.6	12.02		6.8	94.1
2092	520	942.6	12.02		6.8	94.1
2812	18.5		.183			3.61
2813	18.5		.183			3.61
2816	18.5		.183			3.61
2819	18.5		.183			3.61
2821	30.1	22.7				11.3
2822	30.1	22.7				11.3
2823	30.1	22.7				11.3
2824	30.1	22.7				11.3
2865	30.1	22.7				11.3
2869	30.1	22.7				11.3
2873					15	
2874					15	
2879	31		19.2			.85
2891	18.5					3.63
2892					15	
2893					15	
2911	30.1	22.7			11.3	
3111	27.16	27.5				8.33
3312	27.16	27.5				8.3
3321	27.16	27.5				8.3
3322	27.16	27.5				8.3
3411	25	7.25	.35			1.15
3471	25	7.25	.35			1.15
3612	25	7.25	.35			1.15
3621	25	7.25	.35			1.15
3644	25	7.25	.35			1.15
3674	25	7.25	.35			1.15
3679	25	7.25	.35			1.15
3662	25	7.25	.35			1.15
3731	25	7.25	.35			1.15
3861	25	7.25	.35			1.15
6515	40	40	9	20		
7011	40	40	9	20		
7215	43	43	9			
8211	40	40	9	20		
8221	40	40	9	20		

TABLE IV.8(b). SIC CODE AND SOURCE OF ESTIMATED CONSTITUENT CONCENTRATIONS

SIC Code	Source of value
2011	RFF Pollution Matrix Lookup Routine
2016	Average of data collected in 1979 MD. Compliance monitoring (3 plants)
2023	RFF Pollution Matrix Lookup Routine
2024	RFF Pollution Matrix Lookup Routine
2026	RFF Pollution Matrix Lookup Routine
2033	RFF Pollution Matrix Lookup Routine
2037	RFF Pollution Matrix Lookup Routine
2038	RFF Pollution Matrix Lookup Routine
2077	RFF Pollution Matrix Lookup Routine
2091	"Waste Treatment & Disposal From Seafood Processing Plants" EPA-600/2-77-157, August 1977
2092	"Waste Treatment & Disposal From Seafood Processing Plants" EPA-600/2-77-157, August 1977
2812	RFF Pollution Matrix Lookup Routine
2813	RFF Pollution Matrix Lookup Routine
2816	RFF Pollution Matrix Lookup Routine
2819	RFF Pollution Matrix Lookup Routine
2821	RFF Pollution Matrix Lookup Routine
2822	RFF Pollution Matrix Lookup Routine
2823	RFF Pollution Matrix Lookup Routine
2824	RFF Pollution Matrix Lookup Routine
2865	RFF Pollution Matrix Lookup Routine
2869	RFF Pollution Matrix Lookup Routine
2873	EPA Effluent Guidline Division
2874	EPA Effluent Guidline Division
2879	RFF Pollution Matrix Lookup Routine
2891	RFF Pollution Matrix Lookup Routine
2893	RFF Pollution Matrix Lookup Routine
2911	RFF Pollution Matrix Lookup Routine
3111	RFF & Maryland NDPES permit compliance data
3312	RFF & Maryland NDPES permit compliance data
3321	RFF & Maryland NDPES permit compliance data
3322	RFF & Maryland NDPES permit compliance data
3411	RFF Pollution Matrix Lookup Routine
3471	RFF Pollution Matrix Lookup Routine
3612	RFF Pollution Matrix Lookup Routine
3621	RFF Pollution Matrix Lookup Routine
3644	RFF Pollution Matrix Lookup Routine
3674	RFF Pollution Matrix Lookup Routine
3679	RFF Pollution Matrix Lookup Routine
3662	RFF Pollution Matrix Lookup Routine
3731	RFF Pollution Matrix Lookup Routine

(continued)

TABLE IV.8(b). (continued)

3861	RFF Pollution Matrix Lookup Routine
6515	Barth - EPA, MERL, Cincinnati
7011	Barth - EPA, MERL, Cincinnati
7215	RFF Pollution Matrix Lookup Routine
8211	Barth - EPA, MERL, Cincinnati
8221	Barth - EPA, MERL, Cincinnati

TABLE IV.9. SIC CODE, ASSIGNED FLOW, AND SOURCE OF VALUE (MGD)

SIC Code	Assigned Flow	Source
2011	.09	Average of Maryland NPDES "fact sheet data"
2016	.34	Average of Mayland 1979 NPDES compliance monitoring data
2023	.001	Author's Best judgement
2024	.001	Author's Best judgement
2026	.001	Author's Best judgement
2033	.001	Author's Best judgement
2035	.05	Author's Best judgement
2037	.001	Author's Best judgement
2038	.001	Author's Best judgement
2077	.001	Author's Best judgement
2091	.001	State official recommendation
2092	.001	State official recommendation
2812		IFD NEIC data bases
2813		IFD NEIC data bases
2816		IFD NEIC data bases
2819		IFD NEIC data bases
2821		IFD NEIC data bases
2822		IFD NEIC data bases
2823		IFD NEIC data bases
2824		IFD NEIC data bases
2865		IFD NEIC data bases
2869		IFD NEIC data bases
2873	.05	Author's Best judgement
2874	.05	Author's Best judgement
2879		IFD NEIC data bases
2891		IFD NEIC data bases
2892		IFD NEIC data bases
2893		IFD NEIC data bases
2911		IFD NEIC data bases
3111		IFD NEIC data bases
3312		IFD NEIC data bases
3321		IFD NEIC data bases
3322		IFD NEIC data bases
3411		IFD NEIC data bases
3471		IFD NEIC data bases
3612		IFD NEIC data bases
3621		IFD NEIC data bases
3644		IFD NEIC data bases
3674		IFD NEIC data bases
3679		IFD NEIC data bases
3662		IFD NEIC data bases
3731		IFD NEIC data bases
3861		IFD NEIC data bases
6515	.05	Author's Best judgement
7011	.01	Author's Best judgement
7215	.1	Author's Best judgement
8211	.015	Average of MD "fact sheet" data
8221	.039	Average of MD "fact sheet" data

TABLE IV.10. ASSIGNED INDUSTRIAL FACILITIES NUTRIENT LOADINGS FROM OBSERVED DATA[1] (LBS./DAY) IN THE ABSENCE OF THESE KINDS OF DATA, LOADS WERE CALCULATED FROM CONCENTRATIONS SHOWN IN TABLE IV.8(a)

Basin	State	NPDES (permit #)	Facility Name	TP	NH34	TKN	SIC
	VA	1856	Thiokol Fibers Div	1.76	6.4	7.92	2297
	VA	248	Radford Army Ammunition plant		129.8		2892
	VA	1651	Burlington Ind. Inc. Clarksville	36.4	10.10	157.6	2269
0214	VA	1899	Crompton-Shenandoan Company		0.10		2016
0214	VA	1902	Rocco Farms Foods Edinburg		36.0		2016
0214	VA	1961	Rockingham Poultry Market Co. Inc.		12.0	25.0	2016
	VA	4782	Wright Chemical Corp. Waverly		0.66		2891
0214	VA	2160	Dupont Waynesboro		92.4	347.0	2821
0214	VA	2178	Merck Co. Inc. Stonewall Pl.	710.0	1744.0		2835
0214	VA	2313	Wampler Food Hinton		3.6		2016
	VA	27871		1.2			
0216	VA	3387	Virginia Chemicals Inc.	244.0			2819
0214	VA	4031	Holly Farms Glen Allen	57.0	90.0		2016
0214	VA	2402	General Electric Waynesboro	5.4			3471
	VA	29416		0.26			
0213	MD	1201	Bethlehem Steel Sparrows Point	1660.0	1488.0	3365.0	3312
0213	MD	299	FMC Corp. Organic Chem Div	400.0			2869
0216	VA	5291	Allied Chem. Corp Hopewell		1637.0		2869
0213	MD	311	WR Grace Davidson Chem. Div.		2203.0		2819
0214	VA	2208	Avtex Fibers Inc.			472.0	282
0214	VA	2267	Virginia Oak Tannery		15.0		3111
0214	VA	4669	Dupont Spruance			94.0	2821
0215	VA	3115	Chesapeake Corporation	403.0		631.0	2621

[1] 305b reports, DMRs, and facility representatives

TABLE IV.11(a). ESTIMATES OF NUTRIENT LOADS FROM INDUSTRIAL POINT SOURCES FROM ABOVE THE FUNCTIONALLY-DEFINED FALL LINE (mg/L)

Drainage Basin	Water Quality Parameter	Above the fall line[1] Estimated	Measured	Total
	BOD5	799	5718	6517
	TP	183	214	397
Susquehanna	TN	386	2334	2720
(0212)	NH34		540	540
	TKN	2	2318	2320
	BOD5		0	0
Upper	TP		0	0
Chesapeake	TN			
Bay and Delmarva	NH34			
(0213)	TKN		0	0
	BOD5	132	2589	2721
	TP	24	95	119
Potomac	TN	42	5194	5236
(0214)	NH34		1917	1917
	TKN	.5	3870	3871
	BOD5		.29	.29
Rappahannock/	TP		.01	.01
York	TN		.05	.05
(0215)	NH34			
	TKN		.05	.05
	BOD5	36	34	71
James	TP	1	2	3
	TN		7.4	7.4
(0216)	NH34		.01	
	TKN		7.4	7.4

(1) 'Estimated' and 'measured' refer to how flow values for individual dischargers or types of dischargers were determined. Estimated flows are unmeasured flows and are based on averages of similar dischargers or best judgement. Measured flows are recorded flows from NPDES 'fact sheet' files or assessed data bases. Estimated and measured flows were then multiplied by expected concentrations of nutrients in wastewater to calculate loads. These loads, in turn, were designated as estimated or measured.

TABLE IV.11(b). ESTIMATES OF NUTRIENT LOADS FROM INDUSTRIAL POINT SOURCES FROM BELOW THE FUNCTIONALLY-DEFINED FALL LINE (mg/L)

Drainage Basin	Water Quality Parameter	Below the fall line Estimated	Measured	Total
	BOD5	609		609
	TP	133		133
Susquehanna	TN	294		294
(0212)	NH34			
	TKN	5		5
	BOD5	1192	9760	9271
Upper	TP	301	488	789
Chesapeake	TN	296	6561	6857
Bay and Delmarva	NH34	7	3815	3822
(0213)	TKN	45	6557	6602
	BOD5	477	507	984
	TP	68	747	815
Potomac	TN	153	1513	1666
(0214)	NH34	1	203	204
	TKN	16	1993	2009
	BOD5	1115	1422	2537
Rappahannock/	TP	71	24	95
York	TN	248	575	823
(0215)	NH34	90	255	345
	TKN	245	445	690
	BOD5	91	17880	17971
James	TP	16	1890	1906
	TN	10	3755	3765
0216)	NH34	0	2295	2295
	TKN	10	4159	4169

TABLE IV.11(c). ESTIMATES OF NUTRIENT LOADS FROM INDUSTRIAL POINT SOURCES FROM ABOVE AND BELOW THE FUNCTIONALLY-DEFINED FALL LINE (LBS/DAY)

Drainage Basin	Water Quality Parameter	Above and Below the fall line Estimated	Measured	Total
	BOD5	1409	5718	7127
	TP	316	214	530
Susquehanna	TN	680	2334	3014
(0212)	NH34		540	540
	TKN	7	2318	2325
	BOD5	1192	9760	11952
Upper	TP	301	448	749
Chesapeake	TN	296	6561	6857
Bay and Delmarva	NH34	7	3815	3822
(0213)	TKN	45	6557	6602
	BOD5	609	3096	3705
	TP	92	842	934
Potomac	TN	195	6707	6902
(0214)	NH34	1	2120	2121
	TKN	17	5863	5880
	BOD5	1115	1422	2537
Rappahannock/	TP	71	24	95
York	TN	248	575	823
(0215)	NH34	90	255	345
	TKN	245	445	690
	BOD5	126	17914	18040
James	TP	17	1892	1909
	TN	10	3762	3772
0216)	NH34	0	2295	2295
	TKN	10	4166	4176

Estimated Total Point Source Nutrient Loading

The total estimated municipal point source loading data (Table IV.6) and the total estimated industrial point source loading data [Tables IV.11(a), (b), and (c)] have been summed to generate a table of estimated total nutrient loadings from all point sources to the Bay system [Tables IV.12(a), (b), and (c)]. The data presented in Table IV.12 are broken out to delineate loadings from above and below the functional fall line.

Tables IV.12(a), (b), and (c) indicate the relative magnitude of pollutant loadings from municipal and industrial point sources. By changing these loads to percentages, it can be seen from Table IV.12 that, above the fall line, the industrial contribution of total TP ranges from two percent in the James to four percent in the Potomac. For total TN, the industrial contribution ranges from two percent in the James to 28.5 percent in the Potomac. Calculations on data in Table IV.12(b) indicate that the industrial contribution of TP below the fall line ranges from 8.7 percent in the Upper Chesapeake Bay Delmarva drainage basin to 79.6 percent in the Susquehanna. However, it should be pointed out that very little of the Susquehanna is below the fall line. More representative of industrial point source nutrient contributions below the fall line is the Potomac with industrial point sources contributing 10.8 percent of the total TP to its drainage basin and 14 percent to the Rappahannock/York drainage basin. Without the Susquehanna, the industrial contribution to the TN load, below the fall line, ranges from 2.8 percent in the Potomac to 34.7 percent in the Rappahannock/York. In the James River, industrial point sources contribute 12.7 percent and, in the upper Bay, Delmarva 20.6 percent.

Table IV.12(c) presents the industrial and municipal contribution to the total drainage basin load. The industrial contribution to the total TP load ranges from 3.2 percent in the Susquehanna to 13.8 percent in the James. The TN industrial load ranges from 5.8 percent in the Susquehanna to 30.7 percent in the Rappahannock/York. From this information, it can be concluded that although industrial point sources of nutrients may be significant in local areas, overall their relative contribution to the Bay is minor in comparison to the loadings from municipal point sources.

The loadings indicated as being below the fall line are intended to represent the point source load to the tidal Bay system in excess of that already included in the computations of Section III. These data were employed to compute the total estimated seasonal and annual mass loading of nitrogen and phosphorus species to the tidal Chesapeake Bay system. They are shown in Table IV.13. This table shows that nitrogen and nitrogen species constitute the largest proportion of nutrients reaching the tidal Bay from point sources. Because each season contains approximately equal numbers of days, seasonal loads (based on daily flows) do not reveal large differences. Climatelogical and other influences (e.g., infiltration/inflow) were not considered in breaking out seasonal loads in this analysis.

TABLE IV.12(a). ESTIMATES OF NUTRIENT LOADS FROM MUNICIPAL AND INDUSTRIAL POINT SOURCES FROM ABOVE THE FUNCTIONALLY-DEFINED FALL LINE (LBS/DAY EXCEPT FLOW IN MGD)

Drainage Basin	Water Quality Parameter	Above the fall line Municipal	Industrial	Total
	BOD5	105899	6517	112416
	TP	16018	397	16415
Susquehanna	OP	11504		
	TN	48098	2720	50818
(0212)	TKN	33502	2320	35822
	NO23	14596		
	NH34	24353	540	24893
	ORGN	9149		
	FLOW	329		
	BOD5	64		64
Upper	TP	7		7
Chesapeake	OP	4		4
Bay and	TN	41		41
Delmarva	TKN	16		16
	NO23	25		25
(0213)	NH34	8		8
	ORGN	7		7
	FLOW	.3		0.3
	BOD5	29972	2721	32693
	TP	2883	119	3002
Potomac	OP	2555		
	TN	13089	5236	18325
(0214)	TKN	8616	3871	12487
	NO23	3760		
	NH34	6049	1917	7966
	ORGN	3014		
	FLOW	87		
	BOD5	355		355
	TP	55		55
Rappahannock/	OP	42		
York	TN	310		310
	TKN	112		112
(0215)	NO23	198		
	NH34	69		69
	ORGN	43		
	FLOW	2.4		

(continued)

TABLE IV.12(a). (continued)

Drainage Basin	Water Quality Parameter	Above the fall line Municipal	Industrial	Total
	BOD5	7349	71	7420
	TP	1574	3	1577
James	OP	1218		
	TN	3730	7.4	3737
	TKN	3280	7.4	3287
(0216)	NO23	450		
	NH34	2567		2567
	ORGN	713		
	FLOW	24		

TABLE IV.12(b). ESTIMATES OF NUTRIENT LOADS FROM MUNICIPAL AND INDUSTRIAL POINT SOURCES BELOW THE FUNCTIONALLY-DEFINED FALL LINE (LBS/DAY EXCEPT FLOW IN MGD)

Drainage Basin	Water Quality Parameter	Below the fall line Municipal	Industrial	Total
	BOD5	134	609	743
	TP	34	133	167
Susquehanna	OP	29		
	TN	85	294	379
(0212)	TKN	71	5	76
	NO23	14		
	NH34	58		
	ORGN	13		
	FLOW	.6		
	BOD5	54824	9271	64095
Upper	TP	8224	789	9013
Chesapeake	OP	5781		
Bay and	TN	26406	6857	33263
Delmarva	TKN	12916	6602	19518
	NO23	9482		
(0213)	NH34	10404	3822	14226
	ORGN	4303		
	FLOW	164		
	BOD5	50277	984	51261
	TP	6700	815	7515
Potomac	OP	5251		
	TN	57489	1666	59155
(0214)	TKN	26764	2009	28773
	NO23	30298		
	NH34	23445	204	23649
	ORGN	9263		
	FLOW	502		
	BOD5	2675	2537	5212
	TP	576	95	671
Rappahannock/	OP	458		
York	TN	1542	823	2365
	TKN	1196	690	1886
(0215)	NO23	346		
	NH34	922	345	1267
	ORGN	273		
	FLOW	10.4		

(continued)

TABLE IV.12(b). (continued)

Drainage Basin	Water Quality Parameter	Below the fall line Municipal	Industrial	Total
	BOD5	74688	17971	92659
	TP	10346	1906	12252
James	OP	7237		
	TN	43770	6044	47535
(0216)	TKN	39303	4169	43472
	NO23	7216		
	NH34	32991	2295	35286
	ORGN	6277		
	FLOW	231		

TABLE IV.12(c). ESTIMATES OF NUTRIENT LOADS FROM MUNICIPAL AND INDUSTRIAL POINT SOURCES TOTALED ABOVE AND BELOW THE FUNCTIONALLY-DEFINED FALL LINE (LBS/DAY EXCEPT FLOW IN MGD)

Drainage Basin	Water Quality Parameter	Above and below the fall line Municipal	Industrial	Total
	BOD5	106033	7127	113160
	TP	16052	530	16582
Susquehanna	OP	11533		
	TN	48183	3014	51197
(0212)	TKN	33573	2325	35898
	NO23	14610		
	NH34	24411	540	24951
	ORGN	9162		
	FLOW	330		
	BOD5	54888	11952	66840
Upper	TP	8231	749	8980
Chesapeake	OP	5785		
Bay and	TN	26447	6857	33304
Delmarva	TKN	12932	6602	19534
	NO23	9507		
(0213)	NH34	10412	3822	14234
	ORGN	4310		
	FLOW	164.3		
	BOD5	80249	3705	83954
	TP	9583	934	10517
Potomac	OP	7806		
	TN	70578	6902	77480
(0214)	TKN	35380	5880	41260
	NO23	34058		
	NH34	29494	2121	31615
	ORGN	12277		
	FLOW	589		
	BOD5	3030	2537	5567
	TP	631	95	726
Rappahannock/	OP	500		
York	TN	1852	823	2675
	TKN	1308	690	1998
(0215)	NO23	544		
	NH34	991	345	1336
	ORGN	317		
	FLOW	12.8		

(continued)

TABLE IV.12(c). (continued)

Drainage Basin	Water Quality Parameter	Above and below the fall line Municipal	Industrial	Total
	BOD5	82037	18040	100077
	TP	11920	1909	13829
James	OP	8455		
	TN	47500	3772	51272
(0216)	TKN	42583	4176	46759
	NO23	7666		
	NH34	35558	2295	37853
	ORGN	6990		
	FLOW	256		

TABLE IV.13. TOTAL ESTIMATED AVERAGE SEASONAL AND ANNUAL NUTRIENT LOADINGS FROM POINT SOURCES TO THE TIDAL PORTIONS(1) OF THE CHESAPEAKE BAY SYSTEM

Constituent	Daily (Thousands of Pounds)	Winter	Spring	Summer	Fall	Annual
		(Millions of Pounds)				
TN	142.7	12.8	13.1	13.1	13.0	52.1
NO23	47.4	4.26	4.36	4.36	4.31	17.3
NH34	74.5	6.70	6.85	6.85	6.78	27.2
TKN	93.7	8.44	8.62	8.62	8.53	34.2
TP	29.6	2.67	2.72	2.72	2.70	10.8
OP	18.8	1.69	1.73	1.73	1.71	6.85

(1)Discharges entering the system downstream of the functional fall line as described in this Section.

SECTION V

BOTTOM FLUXES OF NUTRIENTS

CONCEPTUAL FRAMEWORK

The bottom sediments of Chesapeake Bay and its tributaries constitute a large reservoir of nutrients available for potential release to the water column. The nutrients enter the sediments primarily as organic and inorganic particulates that settle out of the overlying water. Biological and chemical reactions convert the organicallly-bound nutrients to inorganic forms, reaching an equilibrium distribution between a soluble phase in sediment interstitial (pore) water and a particulate phase adsorbed onto the sediment solids. Thus, the sediments represent a sink capable of retaining a portion of the nutrients settling out of the water column. But they also represent a source because part of the remineralized nutrients diffuse out of the sediments through the pore water, part is advected out via sediment disturbance by burrowing animals or physical resuspension; and those remineralized on the sediment surface escape directly to the overlying water.

The sediments are a complex environment so, for analytical purposes, we adopted a simplified conceptual framework, shown in Figure V.I. We will consider the sediments to have discrete layers distinguishable by the chemical and biological processes occurring in each. Figure V.1 diagrams a vertical section of sediment. The organic fluff layer is composed of colloidal material and fine particles that are unconsolidated, have a density near that of water, and may be resuspended and transported by near-bottom currents. The underlying, compacted surface layer is somewhat more consolidated material that is not readily resuspended in the water column, and its surface is oxidized when the overlying water contains oxygen. If the overlying water becomes anoxic, so does the compacted surface layer. The largest portion of the sediments is the compacted, anoxic layer, which is subject to biological processes in the upper 15 inches or so. Various parts of this Section will refer to this conceptualization.

Two methods will be used to estimate the rate of nutrient release from the sediments to the water column. The first makes use of the sediment gravity core samples taken during the course of the Chesapeake Bay Program (Hill and Conkwright 1981, Tyree et al. 1981, Bricker[1] 1981) as well as those from the U.S. Geological Survey's Potomac River Project. The second approach uses measurements of nutrient release into domes placed on the bottom as part of the Bay Program's nutrient dynamics study. These two methods will be used to compute ranges of potential nitrogen and phosphorus flux.

PORE WATER STUDIES

The lower limit for potential nutrient flux out of the sediment is estimated from the pore water studies; other factors, like bioturbation, may

[1]Personal Communication: "Benthic Flux of Nutrients from Pore Water Studies," O.P. Bricker, Northeast Research, U.S. Geological Survey, Reston, VA, October, 1981.

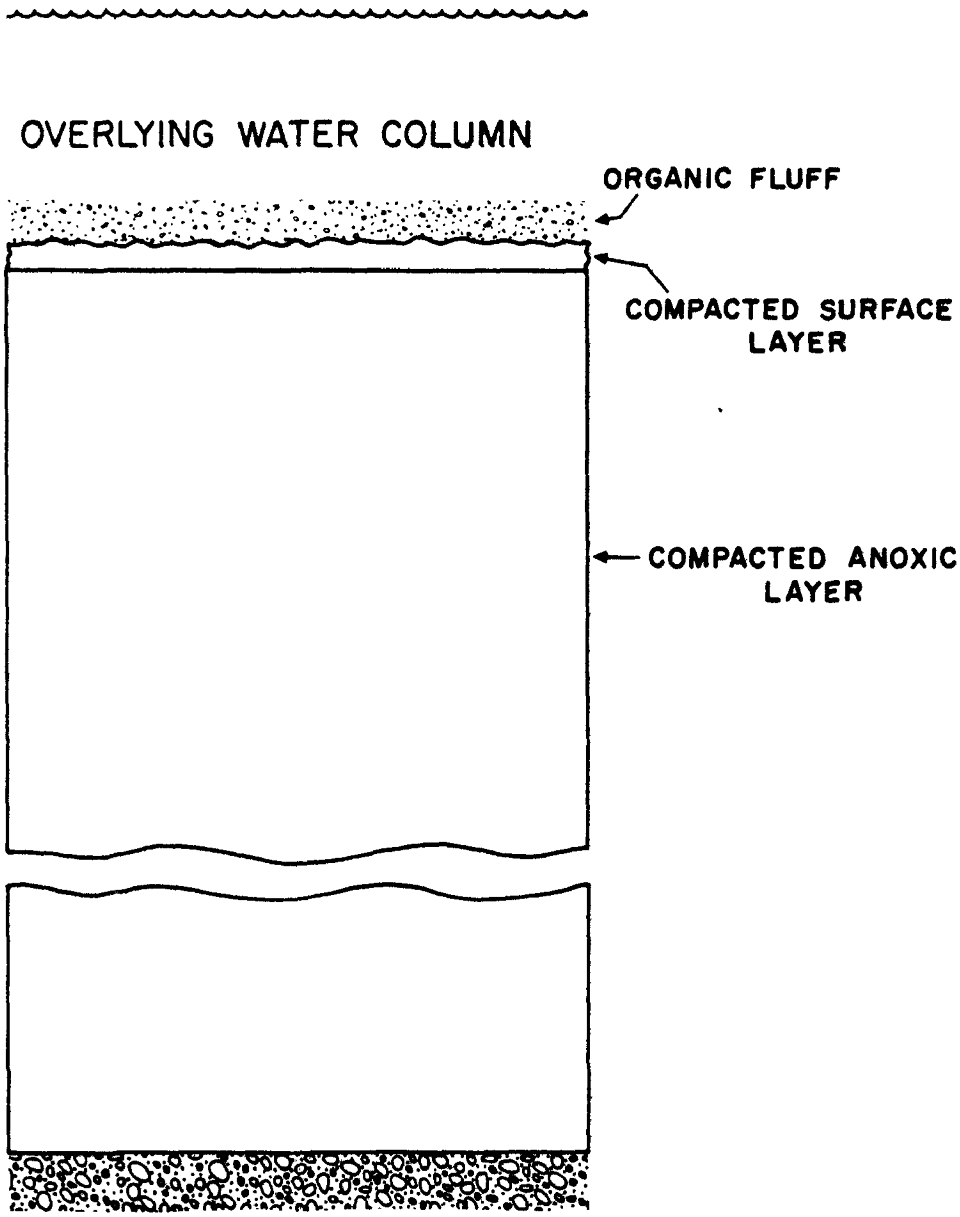

Figure V.1. Conceptual diagram of estuarine sediment column.

increase the actual nutrient flux over that described by pore water study. Over 100 cores taken in the main Bay were analyzed for nutrient content. The rates of potential nutrient diffusion out of the sediment were then calculated for each CBP segment (see Figure V.1) based on the concentration gradients in the sediment, the porosity of the sediment, and a characteristic coefficient for molecular diffusion determined by ion activity and the diffusing characteristics of the molecules. The results appear in Table V.1. Nitrogen is released primarily as ammonium at calculated rates of 0.5 to 8.8 millionth of a pound of nitrogen per square foot per day. The winter values are the arithmetic mean of values for the other seasons, since no cores were taken in winter. This information was then extrapolated to the entire segment by multiplying the values in Table V.1 by the area of the Bay bottom in each segment composed of more than 50 percent organically-enriched mud (functionally, areas that have less than 50 percent sand). The total daily potential release per segment was then multiplied by the number of days per season to obtain the seasonal input to each segment from the sediments as shown in Table V.2. The total nitrogen and phosphorus input from the sediments for each season appears in the right hand column, and the total annual input for each segment appears at the bottom of Table V.2. The minimum potential annual inputs from the sediment are 32.2 million pounds (1045.9×10^{12} micro moles) of nitrogen and 7.44 million pounds (100×10^{12} micro moles) of phosphorus.

TABLE V.1. POTENTIAL NITROGEN AND PHOSPHORUS UNIT AREA DIFFUSION FROM SEDIMENT PORE WATERS (UNITS ARE 10^{-6} POUNDS PER SQUARE FOOT PER DAY AS N OR P)

Segment	CB-1	CB-2	CB-3	CB-4	CB-5	CB-6	CB-7	CB-8
<u>Spring</u>								
NH_4^+	4.5	0.5	1.6	5.2	3.7	2.8[1]	1.0	2.8[1]
PO_4^{-3}	0.006	0.27	0.55	1.1	1.4	0.58[1]	0.21	0.58[1]
<u>Summer</u>								
NH_4^+	1.4	1.8	3.5	5.0	3.5	3.2[1]	4.3	3.2[1]
PO_4^{-3}	0.55[1]	0.43	0.77	0.53	0.85	0.10 [1]	0.68	0.55[1]
<u>Fall</u>								
NH_4^+	6.3	1.4	1.6	1.3	3.7	3.6	8.8	2.3
PO_4^{-3}	0.082	0.095	0.78	1.5	0.84	0.18	0.14	0.16
<u>Winter</u>[2]								
NH_4^+	4.1	1.3	2.2	3.8	3.6	3.21	4.7	2.8
PO_4^{-3}	0.21	0.27	0.70	1.0	1.0	0.29	0.35	0.43

[1]Value calculated as mean of other measurements taken is same season (across columns)

[2]Winter values calculated as mean for other seasons in each segment (down columns) because no winter data were taken.

TABLE V.2. POTENTIAL NITROGEN AND PHOSPHORUS MASS DIFFUSION FROM SEDIMENT PORE WATERS FOR EACH SEGMENT (UNITS ARE THOUSANDS OF POUNDS N OR P)

Segment[1]	CB-2	CB-3	CB-4	CB-5	CB-6	CB-7	CB-8	Total Bay
Spring								
NH_4^+	80	647	2710	2279	770	400	43	6929
PO_4^{-3}	41	211	559	818	164	89	7	1889
Summer								
NH_4^+	283	1355	2618	2156	893	1724	22	9051
PO_4^{-3}	68	300	280	518	27	280	7	1480
Fall								
NH_4^+	222	616	647	2279	986	3511	37	8298
PO_4^{-3}	14	300	750	505	48	55	3	1675
Winter								
NH_4^+	191	862	1940	2187	862	1848	43	7933
PO_4^{-3}	41	266	518	614	818	136	7	2400
Total Annual								
NH_4^+	776	3480	7915	8901	3511	7483	145	32211
PO_4^{-3}	164	1077	2107	2455	1057	560	24	7444

(1) No values are indicated for segment CB-1 because no substantial area in that segment is composed of organically-enriched mud (less than 50 percent sand).

On a seasonal basis, ammonium and phosphorus behave differently. Ammonium is released year-round, regardless of whether the overlying water and compacted surface layer are oxygenated or anoxic (Taft 1982). Phosphate, however, seems to be trapped by the compacted surface layer when it is oxygenated, and released rapidly when it becomes anoxic. Therefore, phosphate release by pore diffusion should be most significant during summer in regions of the Bay where the overlying water is anoxic. Moreover, release should be a two-step event. In step one, a large mass of phosphorus, approximately equivalent to that which has accumulated in the compacted surface layer during the previous nine months of oxygenated conditions, is released rather quickly. In step two, diffusion out of the pore water continues at a slower rate, governed by concentration gradients and sediment characteristics, for the period of anoxia in the deep water.

This concept can be tested by calculating the amount of phosphate that would be trapped in the compacted surface layer during fall, winter, and spring within the bottom region subjected to anoxia. If this amount of phosphate were released at once into the volume of anoxic water, it would produce a phosphate concentration of 0.22 mg/L. The observed phosphate concentration shortly after the onset of deep water anoxia is about 0.124

mg/L (Taft 1982, Figure 5). So our estimate exceeds, but is reasonably close, to the observed values. This result suggests that the concept is basically correct, but that the system is not operating as a simple on-off release mechanism. Also, our calculatioan does not account for transport out of the deep water to the surface layer, which does occur and would make the observed values less than the calculated ones.

In light of this behavior, and for the purpose of the seasonal comparison of various phosphorus sources made in Section VIII, the assumption that the calculated annual flux of phosphorus from the pore waters is released in the summer months appears to be reasonably well supported.

DOME STUDIES

Direct measurements of nutrient release from the sediments were made with diver-installed domes in five locations in the main portion of the Bay during August 1980 and May 1981. The dome technique measures both diffusion of nutrients (primarily ammonium since the domes were placed in oxygenated bottom water) and remineralization on the sediment surface. It could also include nutrient release caused by burrowing animals if they were covered by the dome. Thus dome measurements give the upper limit for potential nutrient release from the sediments.

The dome results appear in Table V.3. The spring values for both nutrients are less than the summer values with phosphate flux being zero in all but the northern most segment. Although the compacted surface layer was oxygenated, phosphate release was observed in summer but not in spring. This result suggests that diffusion is blocked by, and remineralization is minimal in, the compacted surface layer and the organic fluff layer during spring. The latter may be due to low temperatures and correspondingly low biological activity. Both nutrients show marked flux rates in summer reflecting increased biological activity in the surface layers probably stimulated by warmer temperatures.

The magnitude of nutrient remineralization in the two surface layers can be obtained as the dome release minus the calculated diffusion from pore water. With the use of the data in Tables V.1 and V.3, remineralization in the surface-sediment layers accounts for 80 to 90 percent of the nitrogen release and for 30 to 90 percent of the phosphorus release in summer (except for segment CB-3, which has a lower dome rate than diffusion rate). For purposes of this analysis, we consider that the processes by which nutrients are remineralized on the sediment surface are similar to those operating in the water column. Sorption of remineralized nutrients onto sediment particles could occur, but would not influence our conclusions, because the dome flux rates were determined from measured nutrient concentration changes.

TABLE V.3. NUTRIENT RELEASE FROM THE SEDIMENTS MEASURED UNDER DOMES (UNITS ARE 10^{-6} POUNDS PER SQUARE FOOT PER DAY)

Segment	CB-2	CB-3	CB-4	CB-5	CB-6	CB-7	CB-8
Spring							
NH_4^+	3.3	4.1	4.2	5.5	5.5	5.5	5.5
PO_4^{-3}	0.5	0	0	0	0	0	0
Summer							
NH_4^+	12	7.0	46	18	18	18	18
PO_4^{-3}	0.6	0.5	6.1	1.5	1.5	1.5	1.5

TABLE V.4. NUTRIENT RELEASE IN EACH SEGMENT CALCULATED FROM DOME STUDIES (UNITS ARE THOUSANDS OF POUNDS)

Segment	CB-2	CB-3	CB-4	CB-5	CB-6	CB-7	CB-8	Total Bay
Spring								
NH_4^+	524	1602	2187	3357	1509	2218	92	11489
PO_4^{-3}	68	0	0	0	0	0	0	68
Summer								
NH_4^+	1910	2741	23900	11026	4959	7300	308	52144
PO_4^{-3}	95	177	3137	955	409	614	20	5407

SUMMARY

The upper and lower limits for nutrient release from the sediments have been established for the main portion of Chesapeake Bay. The lower limits, from diffusion calculations (Table V.2), are about 32 million pounds of nitrogen (as ammonium) and 7.4 million pounds of phosphorus (as phosphate) per year. The upper limits can be estimated from the spring and summer dome studies (Table V.4) by multiplying the spring values by three, to account for winter and fall, and adding the product to the summer values. The result is 86.6 million pounds of nitrogen and 5.6 million pounds of phosphorus. The difference in the nitrogen values (54.6 million pounds) represents regeneration in the unconsolidated sediment layer. The similarity of values for phosphorus suggests suppression of diffusive flux and dominance of regeneration in the unconsolidated layer during experimental measurements. There is clearly a need for more field studies on sediment processes.

The relationship between benthic and other nutrient sources is shown in Chapter VIII. For example, during the summer the bottom is the major source of ammonium and orthophosphorus (Table VIII.4b).

SECTION VI

NUTRIENT FLUXES AT THE MOUTH OF CHESAPEAKE BAY

Since Chesapeake Bay receives ocean water at its mouth, it also receives nutrients from the ocean. However, it is not clear whether there is net gain or loss of nutrients at the ocean boundary. Nutrient transport at the mouth is dependent on the direction and magnitude of water flow, and on the nutrient concentrations in the water. The long term net flow is out of the Bay and is equal to the riverflow minus evaporation, plus precipitation input. However, over any short time interval, meteorological conditions can drive water into or out of the Bay continually for several days at a time. These short term variations make it difficult to calculate long term nutrient transport.

Calculations are also complicated by the structure of water flow at the mouth. Within the Bay, the fresher, lighter river water overlays the saltier, heavier ocean water. At the mouth, however, the basin geometry and the earth's rotation interact so that the ocean water inflow often occurs at all depths on the north side with outflow on the south side of the mouth. Thus, the two-layer structure is side by side rather than top and bottom.

As part of the Chesapeake Bay Program, an intensive study of the mouth region was made in July 1980. Current measuring devices were deployed at five locations across the mouth for 38 days. Nutrient measurements were made at each current meter location for eight consecutive days during the deployment.

When the current meter data are averaged over the 38 days beginning June 23, 1980, the net flow, less tidal currents, is obtained. Figure VI.1 shows net flow along the bottom into the Bay on both the south (left) and north side. (Positive velocity equals inflow.) Net outflow occurred at the surface all across the mouth and from the surface to the bottom near the middle of the mouth. These results differ somewhat from the flow structure expected from previous work (Boicourt, in progress), but we will use them for flux calculations, because nutrient data were collected concurrently with the flow measurements.

The nutrient fluxes were calculated from nutrient concentrations measured within isotachs shown in Figure VI.1. The measured concentrations were integrated over the area between isotachs to give nutrient fluxes for each range of current velocity. These values were then summed to give fluxes into and out of the Bay.

If the net water fluxes are multiplied by nutrient concentrations, the nutrient fluxes are obtained. Table VI.1 shows the nutrient fluxes calculated in this way. The net fluxes of organic carbon and total nitrogen were out of the Bay, whereas total phosphorus and suspended solids fluxes were into the Bay for this period. For the reasons mentioned above, it is difficult to extrapolate this information to seasonal or annual fluxes, but a comparison with another kind of information can be made. Table VI.2 shows the fluxes calculated from a very simple box model approach (Taft et al. 1978), using unpublished data collected during several periods in 1975-1976. It can be seen that the flux of particulate nutrients for 1975-1976 was generally out of the Bay. The values for the

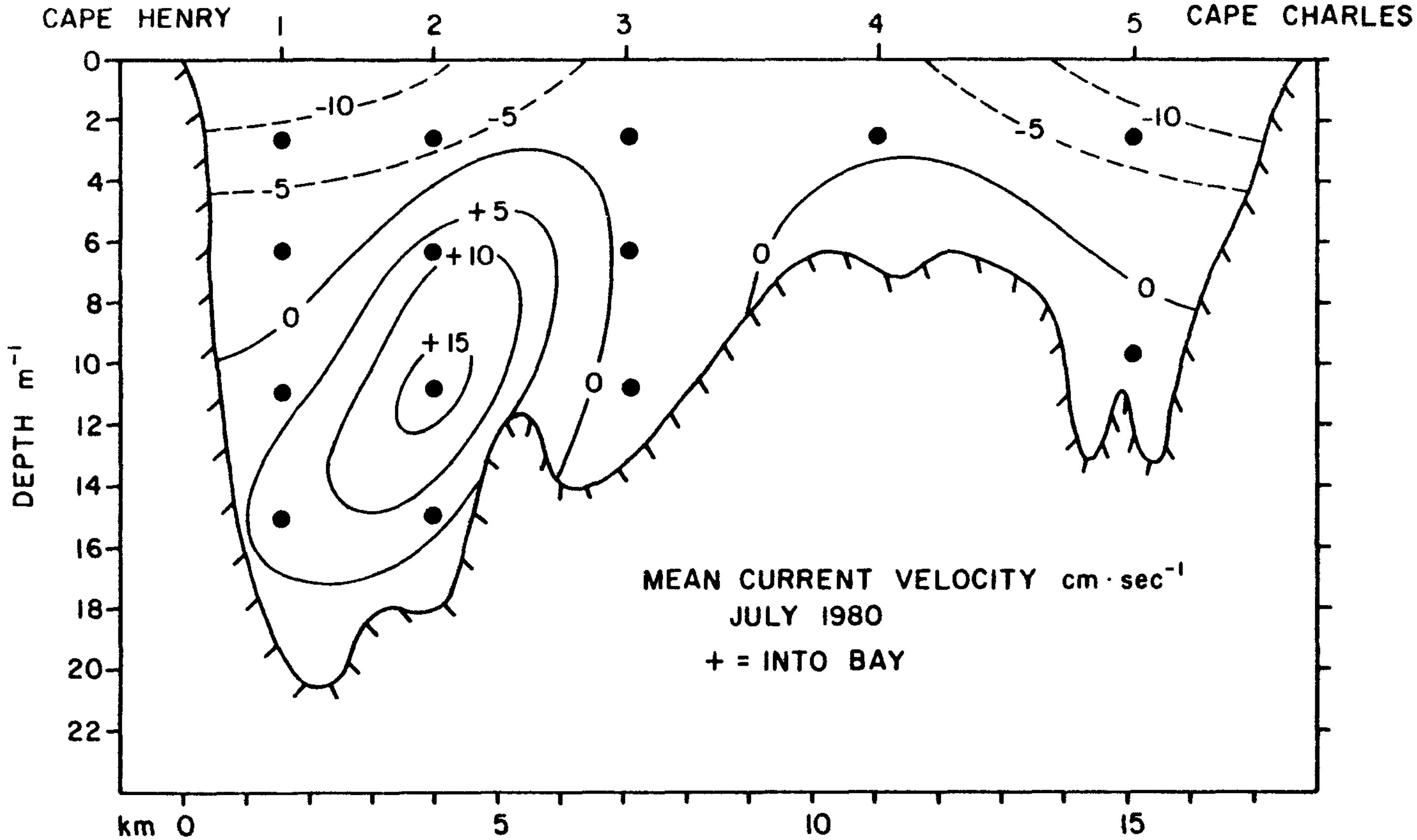

Figure VI.1. Net flows at the mouth of Chesapeake Bay in July 1980 as viewed from the ocean looking into the Bay. Solid circles are current meter locations.

particles alone are much higher than both particle and total fluxes for July 1980. Comparing the suspended solids flux for July 1980 of +55,000 lbs./day (+290 $g \cdot s^{-1}$) (Table VI.I) with the total particulate organic fluxes for August 1975 and 1976 (Table VI.2) shows a difference by about a factor of ten. The net flux of suspended solids is into the Bay, whereas the net particulate organic flux is out of the Bay. This could be reasonably explained if incoming material were enriched with inorganic particles, and if outgoing material were enriched with organic particles such as phytoplankton.

Assume that the values in Table VI.2 are too high, because they are derived from measurements made ten miles inside the Bay rather than at the mouth. Further assume, however, that the net flux is out and the relative differences among the seasons represented in Table VI.2 are approximately correct. That is, the spring flux of organic carbon is higher than the summer flux by about 1.5 times. We can than construct an approximate flux of total carbon out of the Bay by multiplying the total organic carbon flux of -695,250 lbs./day (-3650 $g \cdot s^{-1}d$) by 1.5 for spring. The winter flux is likewise taken as 1.5 times -695,250 lbs./day (-3650 $g \cdot s^{-1}$). We can assume the fall flux equals the summer values for lack of better information.

The average annual flux then is:

Summer	-695,250 lbs./day (-3650 $g \cdot s^{-1}$)
Fall	-695,250 lbs./day (-3650 $g \cdot s^{-1}$)
Winter	-1,040,250 lbs./day (-5475 $g \cdot s^{-1}$)
Spring	-1,040,250 lbs./day (-5475 $g \cdot s^{-1}$)
TOTAL	-3,471,000 lbs./day (-18,250 $g \cdot s^{-1}$)
AVERAGE	- 867,750 lbs./day (- 4,562 $g \cdot s^{-1}$)

Thus, the net flow of organic carbon out of the Bay is estimated to be 867,750 lbs./day (4562 $g \cdot s^{-1}$) for a full year or 316 million pounds per year (1440×10^8 $g \cdot yr^{-1}$). If we then apply the same reasoning to the other nutrients, we calculate the net outflow of nitrogen to be 3 million pounds per year (12.6×10^8 $g \cdot yr^{-1}$), and the net inflow of phosphorus to be 1.7 million pounds per year (7.9×10^8 $g \cdot yr^{-1}$).

The difference in sign between the suspended solids and total phosphorus fluxes, on the one hand, and the remaining nutrient fluxes, on the other, is interesting and can be explained. The suspended solids data contain both organic and inorganic particles. Since the net flux of organic particles seems to be out of the Bay, the observed inflow must be due to inorganic sediments entering the Bay from the ocean. This interpretation is consistent with ideas put forth by Schubel concerning net sediment transport into Chesapeake Bay from the ocean. The net inflow of phosphorus from the ocean to the Bay is consistent with the notion that nitrogen is limiting to phytoplankton biomass in the ocean, so that phosphorus may be present in excess in the ocean water entering the Bay. It may also be sorbed onto suspended sediment particles entering the Bay.

It should be clear to the reader from this summary of the available data that our understanding of transport through the Bay mouth is still quite rudimentary.' The calculations made here should be used to form additional scientific questions focused on improving insight into this important aspect of nutrient dynamics in Chesapeake Bay.

At any rate, the net flux of nutrients to the Bay from the ocean appears to be small enough related to the other sources that it can be ignored for calculating nutrient sources to the Bay system without the introduction of a major error. Although minor on a Bay-wide scale, however, oceanic flux of nutrients may be of local importance.

TABLE VI.1. NUTRIENT FLUXES ACROSS THE MOUTH OF CHESAPEAKE BAY IN JULY 1980 (UNITS ARE THOUSANDS OF POUNDS PER DAY) POSITIVE VALUES INDICATE FLUX INTO THE BAY

	Flux In	Flux Out	Net Flux
Total Organic Carbon	+ 1170	- 1846	- 676
Total Nitrogen	+ 184	- 190	- 6
Total Phosphorus	+ 14	- 13	+ 1
Total Suspended Solids	+ 3,969	- 3,914	+ 55
Particulate Organic Carbon	+ 247	- 348	- 101
Particulate Organic Nitrogen	+ 33	- 49	- 16

TABLE VI.2. FLUXES OF PARTICULATE MATERIAL AT THE BAY MOUTH CALCULATED WITH A BOX MODEL (UNITS ARE THOUSANDS OF POUNDS PER DAY) POSITIVE IS INTO THE BAY (1)

Time	C	N	P	Chl *a*	Total
February 1975	+1890	+ 265	+13	+4	+ 2,168
May 1975	- 874	- 122	-6	-7	- 1,002
August 1975	- 461	- 76	-4	-0.6	- 541
February 1976	-6016	- 67	-3	-1.4	- 6,086
April 1976	- 446	- 63	-3	-1.1	- 512
August 1976	- 373	- 60	-3	-1.1	- 436

(1) Information in table from unpublished data (Taft). Box model concept to be published in Spring 1982.

The minimal flux of nutrients out of the Bay has profound implications for management. Nearly all of the materials that enter the Bay remain there; nutrients trickle out of the Bay mouth at a very slow rate. Thus, even if nutrient loads were dramatically reduced, Bay-wide improvement of water quality would be very slow. It would take many years for the accumulated mass of nutrients to leave the system.

SECTION VII

PRIMARY PRODUCTIVITY IN CHESAPEAKE BAY

Primary productivity is the rate of organic carbon production from inorganic carbon by plants and constitutes an important source of nutrition to Chesapeake Bay. For purposes of this Section, only phytoplankton productivity is considered. The productivity by submerged aquatic vegetation (SAV) is covered in the synthesis paper on SAV (part III).

The basic data set used here for primary productivity calculations was collected on bi-monthly cruises during 1972 and 1973 (Taylor 1982). Values measured at single stations have been averaged over the regions shown on the map in Figure VII.1. Further, the measurements have been integrated over various depths of euphotic zone according to location in the Bay.

The single station measurements and multiplying factors for surface area and euphotic zone depth are shown in Table VII.1. As one might expect productivity is generally greater in summer than in winter by factors of five to 20, depending on region, as well as on higher light levels and temperatures in the summer. Also, annual average productivity per square foot is higher in the upper Bay than lower, because of the greater availability of nutrients. However, owing to the proportionally greater area of the lower Bay, total productivity is greater in the lower Bay regions. Productivity is about equally divided between the states with 30 x 10^8 lbs. C/yr (14 x 10^{11} gC/yr) in Maryland (Table VII.1, Regions I-VII) and 32 x 10^8 lbs. C/yr (15 x 10^{11} gC/yr) in Virginia (Regions VIII-IX) for a total of 62 x 10^8 lbs.C/yr. (29 x 10^{11} gC/yr).

The amount of nitrogen and phosphorus required to support this amount of productivity can be estimated from the ratio of C:N:P in phytoplankton. This ratio is commonly taken to be 106:16:1 by atoms (Redfield ratio).

The nitrogen requirement estimated from the Redfield ratio is 11 x 10^8 lbs. N/yr. (5.2 x 10^{11} gN/yr), and the phosphorus requirement is 1.5 x 10^8 lbs. P/yr. (0.7 x 10^{11} gP/yr). These requirements are met, in part by inputs from rivers, the atmosphere, point sources, and the sediments and, in part, by recycling of organic materials into inorganic nutrient forms. Table VII.2 shows the annual total nitrogen and phosphorus inputs compared with the amount required to support the observed phytoplankton primary productivity. The annual inputs are 302.8 million pounds of nitrogen and 30.2 million pounds of phosphorus. Accounting for the estimated net flux at the mouth and the nutrient "stored" in the water column yields 380.0 million pounds of nitrogen and 38.4 million pounds of phosphorus either in the Bay or entering it annually. The requirements to support phytoplankton primary productivity are, as a minimum, three times greater than the supply for nitrogen and four times greater for phosphorus. This additional amount of nutrient must be supplied by recycling in the water through the mechanisms discussed in Chapter 2 of this part, including grazing and decomposition of organic matter.

The seasonal relationships between phytoplankton productivity (Table VII.3) and nutrient inputs is shown in Tables VII.4 through VII.7. In the winter (Table VII.4), nitrogen entering the Bay potentially supports about seven-tenths of the productivity in winter. This is shown by dividing nutrients in, or entering the Bay, by those required to support primary

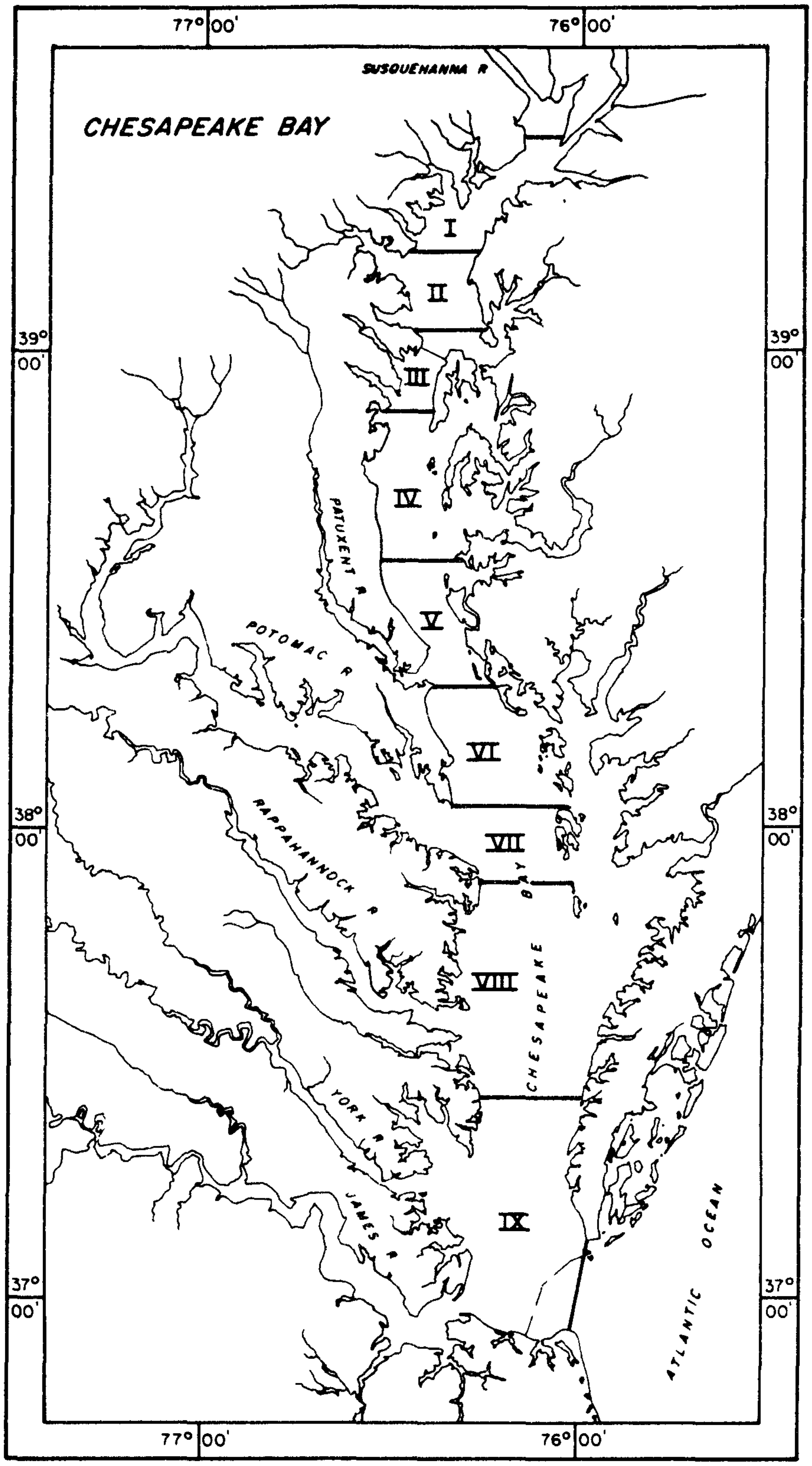

Figure VII.1. Map of Chesapeake Bay showing regions in which primary productivity measurements have been averaged.

productivity. Nitrogen supports about one-half of the productivity in spring (Table VII.5), about one-tenth in summer (Table VII.6), and about one-fifth in the fall (Table VII.7). Incoming phosphorus potentially supports about two-fifths of the productivity in winter, about one-quarter in spring, about one-fifth in summer, and one-eighth in the fall. Nutrients entering and leaving the system as migrating finfish could not be evaluated. The nutrients in fish caught, amounts to about eight million pounds N and one million pounds P annually, but these values are not included in the Tables.

The nutrient estimates were made assuming that all of the inputs are thoroughly mixed in the Bay, an incorrect assumption. Most nutrients are probably retained in the tributaries for a considerable length of time. Moreover, most of the incoming nutrients seem to enter the sediments rather quickly. Note also that the estimates of production are only for the Bay proper; the tributaries have not been included.

TABLE VII.1. PRIMARY PRODUCTIVITY MEASUREMENTS AND FACTORS USED TO CALCULATE ANNUAL AVERAGE PRODUCTIVITY FOR CHESAPEAKE BAY DEVELOPED FROM DATA BY FLEMER 1970 AND TAYLOR[1] 1973

	2/73	4/73	6/73	8/73	10/73	12/73	Av.	SURFACE AREA (10^6) (ft^2)	EUPHOTIC ZONE DEPTH (ft.)	EUPHOTIC ZONE VOLUME (10^9) (ft^3)	ANNUAL AVERAGE PRODUCTION (10^8) (lbs-C)
I	1.8×10^{-6} pounds C/ft^2/day						6273	--	--		1.0
II	374	677	6237	6653	2376	386	2784	3809	15	57	5.7
III	344	5049	2257	6118	2257	1960	2998	2066	15	31	3.4
IV	611	1366	1722	3089	1541	1485	1636	5853	15	88	5.3
V	481	891	1129	4752	1188	499	1490	3680	18	66	3.6
VI	339	891	1426	2317	1960	653	1264	7371	18	133	6.1
VII	320	891	1960	2851	1426	594	1340	4186	20	84	4.1
VIII	-	713	1188	2079	1307	1307	1319	14246	20	285	13.7
IX	315	-	1010	1485	1485	653	990	22284	24	535	19.3
										Total	62.2

[1] Personal Communication: "Primary Production Data for the Chesapeake Bay, 1973," W.R. Taylor, Chesapeake Bay Institute, Shady Side, MD, January, 1982.

TABLE VII.2. RELATION BETWEEN ANNUAL PLANKTON PRODUCTIVITY AND ANNUAL NUTRIENT INPUTS

	Total N	Total P
	Millions of Pounds	
Required to support[1] Primary Productivity	1100	150
Annual input from [2]		
Atmosphere	40.4	1.64
Fluvial sources	178.1	10.3
Point sources	52.1	10.8
Sediments	32.2	7.44
Total inputs	302.8	30.2
Net Flux at the mouth [3]	- 3.0	+ 1.7
Net inputs	299.8	31.9
Total Nutrient in the water[4]	80.2	6.5
Nutrients in or entering the Bay annually	380.0	38.4
Nutrients recycled[5]	720.0	111.6
% Productivity supported by available nutrients	34.6%	25.6%
% Productivity supported by recycling	65.5%	74.4%

[1] Calculated from Table VII.1

[2] From Table VIII.1

[3] From Chapter VI

[4] Estimated as the product of average concentrations of readily-available algal nutrients and water volume. The nutrient forms are: available nitrogen = nitrate and ammonium
available phosphorus = soluble reactive phosphorus

[5] Inorganic nutrient forms regenerated from organic forms by grazing, decomposition, and other processes.

TABLE VII.3. SEASONAL PRIMARY PRODUCTIVITY IN CHESAPEAKE BAY

Season	% Annual productivity	10^8 pounds C/season
Spring	20	12.4
Summer	45	28.0
Fall	25	15.6
Winter	10	6.2

TABLE VII.4. RELATION BETWEEN WINTER PHYTOPLANKTON PRODUCTIVITY AND NUTRIENT INPUTS

	Total N	Total P[1]
	Millions of Pounds	
Required to support Primary Productivity	110	15
Input from		
Atmosphere	6.2	0.2
Fluvial sources	51.4	3.0
Point sources	12.8	2.7
Sediments	7.9	
Total inputs	78.3	5.9
Net Flux at the mouth	- 0.9	+ 0.3
Net inputs	77.4	6.2
Total Nutrient in the water	18.2	0.5
Nutrients in or entering the Bay	95.6	6.7
Nutrients recycled	14.4	8.3
% Productivity potentially supported by available nutrients	86.9	44.7%
% Productivity supported by recycling	13.1%	55.3%

(1) Source is Tables VII.2 and VII.3 for Total N and Total P.

TABLE VII.5. RELATION BETWEEN SPRING PHYTOPLANKTON PRODUCTIVITY AND NUTRIENT INPUTS

	Total N	Total P
	Millions of Pounds	
Required to support Primary Productivity	220	30
Input from		
Atmosphere	16.2	0.51
Fluvial sources	72.2	4.21
Point sources	13.1	2.72
Sediments	6.9	
Total inputs	108.4	7.4
Net Flux at the mouth	- 0.9	+ 0.3
Net inputs	107.5	7.7
Total Nutrient in the water	18.2	0.5
Nutrients in or entering the Bay	125.7	8.2
Nutrients recycled	94.3	21.8
% Productivity supported by available nutrients	57.1%	27.3%
% Productivity supported by recycling	42.9%	72.7%

TABLE VII.6. RELATION BETWEEN SUMMER PHYTOPLANKTON PRODUCTIVITY AND NUTRIENT INPUTS

	Total N	Total P
	Millions of Pounds	
Required to support Primary Productivity	497	68
Input from		
Atmosphere	12.2	0.6
Fluvial sources	25.1	1.4
Point sources	13.1	2.7
Sediments	9.1	7.4
Total inputs	59.5	12.1
Net Flux at the mouth	- 0.6	+ 0.2
Net inputs	58.9	12.3
Total Nutrient in the water	22.8	5.0
Nutrients in or entering the Bay	81.7	17.3
Nutrients recycled	415.3	50.7
% Productivity supported by available nutrients	16.4%	25.4%
% Productivity supported by recycling	83.6%	74.6%

TABLE VII.7. RELATION BETWEEN FALL PHYTOPLANKTON PRODUCTIVITY AND NUTRIENT INPUTS

	Total N	Total P
	Millions of Pounds	
Required to support Primary Productivity	277	38
Input from		
Atmosphere	5.9	0.3
Fluvial sources	27.9	1.5
Point sources	13.0	2.7
Sediments	8.3	
Total inputs	55.1	4.5
Net Flux at the mouth	- 0.6	+ 0.2
Net inputs	54.5	4.7
Total Nutrient in the water	21.0	0.5
Nutrients in or entering the Bay	75.5	5.2
Nutrients recycled	201.5	32.8
% Productivity supported by available nutrients	27.3%	13.7%
% Productivity supported by recycling	72.7%	86.3%

SECTION VIII

SUMMARY AND CONCLUSIONS: THE MANAGEMENT QUESTIONS ANSWERED

This section is divided into two sub-sections. The first synthesizes the results of Chapters II through VI, presenting annual and seasonal loadings from all sources, and computing the total Bay-wide nutrient and sediment input budgets. The second half of the chapter contains a restatement of the pertinent Management Questions listed in the Introduction (Section I). Following each question is a statement of answers that draws upon the data presented here as well as upon other sources that are as complete, technically correct, and editorially succinct as possible within the limitations of the authors capabilities.

ANNUAL AND SEASONAL LOADINGS OF NUTRIENTS TO THE BAY FROM MAJOR SOURCES

The nutrient loading estimates from each source have been accumulated and, in some cases reformatted to develop estimates of the total nutrient inputs to the tidal Chesapeake Bay system. The results have been depicted in terms of the total mass flux into the tidal system for the year and each of the seasons. As previously stated, the months included in each season are as follows:

WINTER:	December, January, February	(90 days)
SPRING:	March, April, May	(92 days)
SUMMER:	June, July, August	(92 days)
FALL:	September, October, November	(91 days)
ANNUAL:	December - November	(365 days)

The sources included in the synthesis are Atmospheric, Fluvial, Point (below fall line), and Bottom. As mentioned at the end of Chapter VI, the ocean has been eliminated from consideration as a source for the purposes of this paper because the net flux was insignificant. The annual and seasonal nutrient input budgets are presented in Table VIII.1(a) through VIII.5(a).

The fraction that each source represents of the annual (or seasonal) total for each constituent has been computed, expressed as a percentage, and included as the "b" section of each Table [Tables VIII.1(b) through VIII.5(b)].

TABLE VIII.1(a). AVERAGE ANNUAL NUTRIENT AND FLUVIAL SEDIMENT INPUT TO THE WATER COLUMN OF THE TIDAL CHESAPEAKE BAY SYSTEM (MILLIONS OF POUNDS)

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources	Total
Total Nitrogen-N (TN)	40.4	178.1	52.1	32.2	302.8
Nitrite + Nitrate Nitrogen-N (NO23)	14.5	111.5	17.3		143.3
Ammonia Nitrogen-N (NH34)	8.91	9.06	27.2	32.2	77.4
Total Kjeldahl Nitrogen-N (TKN)	25.9	58.6	34.2	32.2	150.9
Total Phosphorous-P (TP)	1.64	10.3	10.8	7.44	30.2
Orthophosphorus-P (OP)	0.40	3.24	6.85	7.44	17.9
Sediment (SED)		6630.			6630.

TABLE VIII.1(b). PERCENTAGES OF <u>ANNUAL</u> NUTRIENT LOADINGS FROM VARIOUS SOURCES

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources
Total Nitrogen-N (TN)	13.3	58.8	17.2	10.6
Nitrite + Nitrate Nitrogen-N (NO23)	10.1	77.8	12.1	
Ammonia Nitrogen-N (NH34)	11.5	11.7	35.2	41.6
Total Kjeldahl Nitrogen-N (TKN)	17.2	38.8	22.7	21.3
Total Phosphorous-P (TP)	5.4	34.1	35.8	24.7
Orthophosphorus-P (OP)	2.2	18.1	38.2	41.5

TABLE VIII.2(a). AVERAGE <u>WINTER</u> NUTRIENT AND FLUVIAL SEDIMENT INPUT TO THE WATER COLUMN OF THE TIDAL CHESAPEAKE BAY SYSTEM (MILLIONS OF POUNDS)

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources	Total
Total Nitrogen-N (TN)	6.16	51.4	12.8	7.93	78.3
Nitrite + Nitrate Nitrogen-N (NO23)	2.95	32.2	4.26		39.4
Ammonia Nitrogen-N (NH34)	2.06	2.62	6.70	7.93	19.3
Total Kjeldahl Nitrogen-N (TKN)	3.21	16.8	8.44	7.93	36.4
Total Phosphorous-P (TP)	0.21	2.97	2.67		5.85
Orthophosphorus-P (OP)	0.09	0.933	1.69		2.71
Sediment (SED)		1830.			1830.

TABLE VIII.2(b). PERCENTAGES OF <u>WINTER</u> NUTRIENT LOADINGS FROM VARIOUS SOURCES

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources
Total Nitrogen-N (TN)	7.9	65.7	16.3	10.1
Nitrite + Nitrate Nitrogen-N (NO23)	7.5	81.7	10.8	
Ammonia Nitrogen-N (NH34)	10.7	13.6	34.7	41.1
Total Kjeldahl Nitrogen-N (TKN)	8.8	46.1	23.2	21.8
Total Phosphorous-P (TP)	3.6	50.8	45.6	
Orthophosphorus-P (OP)	3.3	34.4	62.3	

TABLE VIII.3(a). AVERAGE SPRING NUTRIENT AND FLUVIAL SEDIMENT INPUT TO THE WATER COLUMN OF THE TIDAL CHESAPEAKE BAY SYSTEM (MILLIONS OF POUNDS)

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources	Total
Total Nitrogen-N (TN)	16.2	72.2	13.1	6.93	108.4
Nitrite + Nitrate Nitrogen-N (NO23)	4.70	45.3	4.36		54.4
Ammonia Nitrogen-N (NH34)	3.45	3.73	6.85	6.93	21.0
Total Kjeldahl Nitrogen-N (TKN)	11.5	23.6	8.62	6.93	50.6
Total Phosphorous-P (TP)	0.51	4.29	2.72		7.52
Orthophosphorus-P (OP)	0.10	1.28	1.73		3.11
Sediment (SED)		2870.			2870.

TABLE VIII.3(b). PERCENTAGES OF SPRING NUTRIENT LOADINGS FROM VARIOUS SOURCES

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources
Total Nitrogen-N (TN)	14.9	66.6	12.1	6.4
Nitrite + Nitrate Nitrogen-N (NO23)	8.6	83.3	8.0	
Ammonia Nitrogen-N (NH34)	16.5	17.8	32.7	33.1
Total Kjeldahl Nitrogen-N (TKN)	22.7	46.6	17.0	13.7
Total Phosphorous-P (TP)	6.8	57.0	36.2	
Orthophosphorus-P (OP)	3.2	41.2	55.6	

TABLE VIII.4(a). AVERAGE SUMMER NUTRIENT AND FLUVIAL SEDIMENT INPUT TO THE WATER COLUMN OF THE TIDAL CHESAPEAKE BAY SYSTEM (MILLIONS OF POUNDS)

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources	Total
Total Nitrogen-N (TN)	12.2	25.1	13.1	9.05	59.5
Nitrite + Nitrate Nitrogen-N (NO23)	4.70	15.8	4.36		24.9
Ammonia Nitrogen-N (NH34)	1.89	1.26	6.85	9.05	19.05
Total Kjeldahl Nitrogen-N (TKN)	7.47	8.18	8.62	9.05	33.4
Total Phosphorous-P (TP)	0.60	1.42	2.72	7.44	12.2
Orthophosphorus-P (OP)	0.11	0.49	1.73	7.44	9.77
Sediment (SED)		955.9			955.9

TABLE VIII.4(b). PERCENTAGES OF SUMMER NUTRIENT LOADINGS FROM VARIOUS SOURCES

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources
Total Nitrogen-N (TN)	20.5	42.2	22.0	15.2
Nitrite + Nitrate Nitrogen-N (NO23)	18.9	63.6	17.5	
Ammonia Nitrogen-N (NH34)	9.9	6.6	36.0	47.5
Total Kjeldahl Nitrogen-N (TKN)	22.4	24.5	25.8	27.1
Total Phosphorous-P (TP)	4.9	11.7	22.3	61.1
Orthophosphorus-P (OP)	1.1	5.0	17.7	76.2

TABLE VIII.5(a). AVERAGE FALL NUTRIENT AND FLUVIAL SEDIMENT INPUT TO THE WATER COLUMN OF THE TIDAL CHESAPEAKE BAY SYSTEM (MILLIONS OF POUNDS)

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources	Total
Total Nitrogen-N (TN)	5.91	27.9	13.0	8.30	55.1
Nitrite + Nitrate Nitrogen-N (NO23)	2.12	17.7	4.31		24.1
Ammonia Nitrogen-N (NH34)	1.51	1.42	6.78	8.30	18.0
Total Kjeldahl Nitrogen-N (TKN)	3.78	9.06	8.53	8.30	29.7
Total Phosphorous-P (TP)	0.30	1.49	2.70		4.49
Orthophosphorus-P (OP)	0.12	0.53	1.71		2.36
Sediment (SED)		975.			975.

TABLE VIII.5(b). PERCENTAGES OF FALL NUTRIENT LOADINGS FROM VARIOUS SOURCES

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources
Total Nitrogen-N (TN)	10.7	50.6	23.6	15.1
Nitrite + Nitrate Nitrogen-N (NO23)	8.8	73.4	17.9	
Ammonia Nitrogen-N (NH34)	8.4	7.9	37.7	46.1
Total Kjeldahl Nitrogen-N (TKN)	12.7	30.5	28.7	27.9
Total Phosphorous-P (TP)	6.7	33.2	60.1	
Orthophosphorus-P (OP)	5.1	22.5	72.5	

The reader should be cautioned that the sum of the individual seasonal totals (Tables VIII.2(a) - VIII.5(a)) will not always agree exactly with the annual totals shown in Table VIII.1(a). The reason for this is that the annual load shown for the fluvial sources column of Table VIII.1(a) represents the results of the regression model equations applied in Section III for annual loads that are developed independently of the individual seasonal models. The annual loads are not simply the sum of the four seasonal model outputs, therefore, and any differences between the computations using the annual model and the sum of the individual seasons are an artifact of the regression analysis. The difference between presented annual values and the sum of the four seasons is usually less then one percent.

Examination of the data presented in Tables VIII.1(a) through VIII.5(b) reveals some interesting, if not new, information about the loading mechanisms that effect the Bay system. The need to look beyond annual loadings into the seasonality of loading patterns is evident. Shown below are the deviations from the seasonal loadings for nitrogen and phosphorous that would be "expected" if the annual loads (Table VIII.1) were distributed evenly into seasons.

TABLE VIII.6. SEASONAL DISTRIBUTION OF NUTRIENT LOADINGS

Nutrient	Season	"Expected" Value	Actual Value	Percent of Expected Value	% Deviation
TN	Winter	74.7	78.3	104.8	+4.8
TN	Spring	76.3	108.4	142.1	+42.1
TN	Summer	76.3	59.5	78.0	-22.0
TN	Fall	75.5	55.1	73.0	-27.0
TP	Winter	7.45	5.85	78.5	-21.5
TP	Spring	7.61	7.52	98.8	-2.0
TP	Summer	7.61	12.2	160.3	+60.3
TP	Fall	7.53	4.49	59.6	-40.4

In other words, the expected loads are defined as those that would be computed by applying an average annual loading rate expressed as daily loads (lbs/day) to the number of days in each season. These values are useful as a device to elucidate the importance of seasonal considerations of nutrient loading dynamics.

An immediate point of interest when studying Table VIII.6 of seasonal deviations is that while the spring freshet carries a large nitrogen and phosphorous load, a disproportionately large amount of the annual nitrogen budget is delivered [Table VIII.3(a) and VIII.3(b)]. The reason that phosphorous appears to remain close to the "expected" value is because the effect of the spring freshet load is offset by the very large pulse of phosphorous (seven million pounds) released from the sediments during the period of bottom-water anoxia in early summer [Table VIII.4(a)]. A secondary reason for this effect is that the ratio of total nitrogen to total phosphorous in runoff is in excess of three times greater then the TN:TP ratio of the point source load.[1] Point source loadings of total phosphorous are usually double those in runoff in summer and fall [Table VIII.4(a) and VIII.5(a)], about equal in the winter [Table VIII.2(a)], and about half as

[1] TN:TP for runoff varies from 17 to 19 while the TN:TP for point sources is 4.8.

great as the runoff TP load during the spring [Table VIII.3(a)] because of the effect of the freshet. In contrast, nitrogen from runoff always exceeds that from point sources with the greatest deviation occurring in the spring [Table VIII.3(a)] when the nonpoint source nitrogen flux is probably more than four times greater than the point source nitrogen flux. In summary, the largest portion of the annual nitrogen loading budget enters the tidal system during the winter-spring period, and the largest portion of the annual phosphorous budget enters during the spring-summer period. This seems to support Taft's observation that biomass within the euphotic zone in the Bay is most likely controlled (limited) by phosphorous in the spring and nitrogen in the summer (See Chapter 2 of this part).

The nitrogen being discharged from both fluvial and point sources is predominately nitrite-nitrate. However, a larger portion of the total nitrogen load from point sources is in the ammonia phase than for nonpoint sources [Table VIII.1(b)]. In fact, point sources discharge much more ammonia than fluvial sources every season, even during the freshet [Tables VIII.2(b) to 5(b)]. This load of ammonia, plus the input from the bottom would support the hypothesis that nitrate is transported conservatively (without changes in form) through the upper Bay in the spring because of phytoplankton preferences for ammonia (see Chapter 2). If phytoplankton growth in the upper Bay has sufficient ammonia-nitrogen for support of the population then nitrate-nitrogen will transport to the lower Bay without being utilized. With large fluvial loads occurring in the late spring, we can expect the lower Bay to receive these loads in a form readily available for algal assimilation, a condition which is apparent from field monitoring data.

NUTRIENT BUDGETS

With the information assembled in Tables VII.2 and VIII.1, it is possible to construct annual budgets for nitrogen and phosphorus transport. Such budgets, of course, suffer from uncertainties in the data, but are useful for visualizing the relative importance of sources and sinks for nutrients. The greatest uncertainty in our budgets occurs in the exchange between the Bay and the ocean. Since data are scanty, our estimates are based on defendable, but imperfect, assumptions. The amount of nutrient loss to the sediments in each budget was determined by subtracting the difference between the sum of the inputs and the sum of the outputs. Therefore, it has an uncertainty equal to, or greater than, the uncertainty in the ocean exchange estimate. Even with the uncertainties, the budgets reflect what happens in the estuary: It is filling with sediments; it is trapping nutrients.

Figure VIII.1 depicts the annual nitrogen and phosphorus budgets for Chesapeake Bay. Two important features, as discussed above, are exchange at the ocean boundary and the net amount of nutrient removal by sedimentation. For both nutrients, the transport across the boundary with the ocean is approximately balanced. This means that most of the nitrogen and phosphorus, entering from the land and the atmosphere, remain in the system. Some nutrient is stored in the water but, since water column concentrations do not increase dramatically from one year to the next, most incoming nutrient must go to the sediments during the annual cycle. The sedimentation values in Figure VIII.1 are net rates, indicating permanent burial of about 300 million

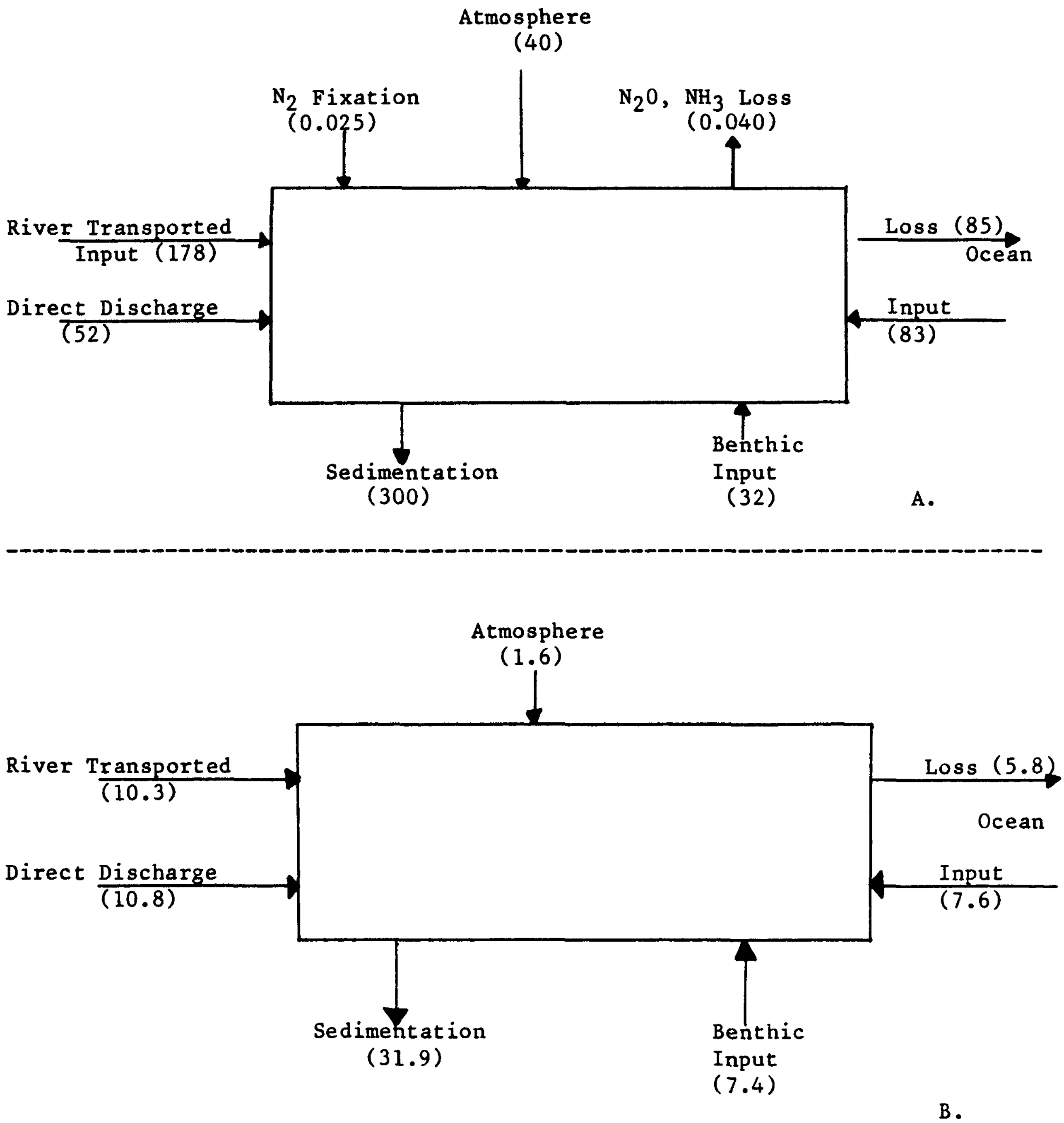

Figure VIII.1. Annual (a) nitrogen and (b) phosphorus budgets for Chesapeake Bay. (In millions of pounds)

pounds of nitrogen and 31.9 million pounds of phosphorus annually. About 10 percent of the nitrogen and 23 percent of the phosphorus is returned to the water column from the sediments.

The nitrogen input to the Bay by nitrogen fixation is not well known, but it should be small compared to other inputs since nitrogen fixation rates in the water are vanishingly small. We estimate 25,000 pounds per year net input from marshes. The nitrogen loss to the atmosphere as N_2O and NH_3 gas is also probably small. Few measurements have been made from which we estimate an annual loss of 40,000 pounds per year from the estuary. We hope future research will refine these estimates.

Neither budget accounts for nutrient gains or losses as fish, crabs, and birds migrate through the system. In the absolute sense, the numbers are no doubt large, but relative to the other inputs and losses, they should be small. By inspection, if all excess nutrients were leaving in the form of a harvestable fishery, eutrophication would not be the problem it is becoming in Chesapeake Bay.

Management Questions and Answers

Below and on the following pages are a restatement of the nine Management Questions and the answers as could best be derived from the available information.

1. What is the atmospheric contribution to nutrient input?

The atmospheric nutrient contribution that enters directly upon tidal waters is at least 40 million pounds of nitrogen and 1.6 million pounds of phosphorous each year [Table VIII.1(a)]. This load constitutes about 13 percent of the annual nitrogen, and five percent of the annual phosphorous input budgets [Table VIII.1(b)]. Seasonally, atmospheric sources may make up as much as 20 percent of the seasonal total nitrogen (winter) input and five percent of the seasonal total phosphorous (summer) input and as little as seven percent of the total nitrogen load and three percent of the total phosphorous load in the winter and spring [Tables VIII.2(b) to VIII.5(b)].

2. What percentage of the nutrients is from point sources?

On an annual basis, about 20 to 25 percent of the total nitrogen load entering tidal waters comes from point sources basin-wide [Table VIII.1(b)]. This percentage range would hold even if all of the point sources load discharged above the Fall Line were transported directly to the tidal system (a very conservative assumption since losses undoubtedly occur in transport, especially during the summer). The proportions are relatively invariant throughout the year, reaching the lower end of the range in the spring and the upper end in the summer and fall.

To make a reasonable estimate of the percentage of the phosphorous load deriving from point sources, some manipulations of the riverine loading models developed in Chapter III were performed. Low flow values were chosen for each of the major tributaries[1], and the total phosphorous load expected to occur at these flows were computed. This total flow (sum of all three tributaries) was about 9660 cubic feet per second. Note from Table IV.12(a) that the total point source flow entering above the Fall Line is about 688 cfs. The total phosphorus load computed to be carried to the tidal system at a stream discharge of 9660 cfsd is about 1950 lbs./day or about 0.7 million pounds per year. If the extremely conservative assumption is made that all of this load derives from point source discharges and is summed with the 10.8 million pounds of point source phosphorous discharged per year below the Fall Line, the total point source contribution of phosphorous is computed to be about 40 percent of the total annual phosphorous input budget of around 11 million pounds per day. Seasonally, the point source contribution of phosphorous makes up as much as 65 percent of the fall total phosphorus input budget and as little as 25 percent of the summer total phosphorous input budget.

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[1] The "daily discharge that is greater than or equal to the flows that occur 10 percent of the time" was computed for each major tributary. They are: Susquehanna, 6640 cfsd; Potomac, 1690 cfsd; James, 1330 cfsd.

3. What percentage of nutrients is from nonpoint sources and how do they vary over time?

To discuss nonpoint sources within the structure of this paper, we define three categories of diffuse sources. They are:

i) Atmospheric contributions

ii) Land runoff/base flow contributions

iii) Benthic contributions

Categories i) and ii) are covered separately elsewhere in this Section under the discussions of other Management Questions. For the purpose of answering this Management Question, we define nonpoint sources as the sum of land runoff and base flow (groundwater discharge) which is carried by fluvial processes to the tidal Bay system. Contributions from the coastal plain are not considered.

On an annual basis, the mean total nonpoint source nitrogen loading is about 50 to 55 percent of the total input budget, or about 160 to 177 million pounds of nitrogen per year [Tables VIII.1(a) and VIII.1(b)], making this the single largest external source of nitrogen loading to the Bay. Seasonally, the variation in the nonpoint source nitrogen loading is quite dramatic, ranging from about 23-25 million pounds in the summer (36 - 39 percent of the total source load) to around 69 - 71 million pounds in the spring (63 - 66 percent of the total spring nitrogen load). The dominant species of nonpoint source nitrogen at the Fall Line is always nitrite-nitrate, making up consistently between 62 and 64 percent of the total nitrogen from this source.

On an average annual basis, the nonpoint source loading of phosphorous is about 30 to 34 percent of the total phosphorous input budget, ranging from around 9 to 10 million pounds per year. As much as 65 to 70 percent of this load on an annual basis is in the suspended phase, meaning most of the phosphorous is being carried to the Bay associated with particulate matter and, therefore, not immediately available for phytoplankton utilization. Seasonally, the nonpoint phosphorous contribution probably varies from about 1.2 to 1.4 million pounds (only about 10-11 percent of the summer total phosphorous budget) in the summer to about 4 million pounds in the spring, or 55 percent of the total spring input budget of phosphorous from all sources. The very low percentage of the load eminating from fluvial sources in the summer is mainly due to the dominant effect of benthic sources of phosphorous released in that season.

4. What are the pollutant runoff rates for particular land uses?

This is the only management question to be answered in the paper for which the source information upon which the answer is based is not contained within the text. The information upon which this answer is based may be found in the EPA Chesapeake Bay Program Information Series Nutrient Summary 3: "Assessment of Nonpoint Source Discharge to Chesapeake Bay" (unpublished). The data presented in that report are the results of a preliminary analysis of the data from the Chesapeake Bay Program Intensive Watershed Studies (IWS).

The analysis performed on the data used the volume-weighted mean concentrations of storm event runoff, computed for the CBP studies (Hartigan, 1981) along with some typical expected average annual runoff volumes for various land use/soil texture combinations, to generate generalized annual pollutant loadings for various classes of land use. These data are presented in Table VIII.6. Although the analysis, in its preliminary state, necessarily produced overlapping ranges of runoff loading rates among land uses, the data in Table VIII.6 allow us to assign order of magnitude rankings for the land uses studied by areal loading rate. The generalized rankings are shown in Table VIII.7.

In all cases, the highest unit area loading rates were generally exhibited by cropland sites and the lowest by forest sites.
(N.B. The rankings shown in Table VIII.7 are a very broad generalization!)

TABLE VIII.7. CONCENTRATION, MG/L (TOP LINE), AND LOADING RATES, LBS/AC/YR (BOTTOM LINE), FOR TOTAL SUSPENDED SOLIDS, TOTAL PHOSPHORUS, ORTHOPHOSPHATE, TOTAL NITROGEN, AND NITRITE-NITRATE FROM VARIOUS USES OF LAND(1)(2)

Land Use	SED	TP	OP	TN	NO23
Cropland(3)	46.5-3202.8	0.21-12.49	0.01-2.77	1.3-22.2	0.02-16.20
	10.54-2460.83	0.05-9.78	0.01-2.20	0.75-17.59	0.02-12.90
Pasture	145.20-669.70	0.38-1.12	0.06-0.14	2.20-6.20	0.30-1.71
	16.45-303.50	0.04-0.51	0.01-0.06	1.25-2.81	0.03-0.78
Forest	9.40-71.5	0.06-0.23	0.00-0.04	0.40-1.10	0.01-0.48
	0.53-48.60	0.00-0.16	0.00-0.03	0.02-1.00	0.00-0.33
Residential	38.00-634.4	0.10-1.66	0.02-0.17	0.70-2.8	0.26-0.90
	47.40-2395.1	0.13-5.22	0.03-0.54	0.87-8.82	0.32-2.84

(1)Volume-weighted concentration data from preliminary analysis by NVPDC, concentration in milligrams per liter. Personal Communication: "Volume-Weighted Mean Concentrations of Storm Event Runoff from EPA/CBP Test Watersheds," J.P. Hartigan, Regional Resources Division, Norther Virginia Planning District Commission, Falls Church, VA, October 13, 1981.

(2)Loading rate computed by CBP staff, in lbs./ac./year.

(3)Cropland includes primarily conventional and minimum till with some no-till land practices.

TABLE VIII.8. GENERALIZED RANKING OF LAND USES BY UNIT AREA RUNOFF LOADING RATE (1 = HIGHEST RATE, 4 = LOWEST RATE)

Land Use	TN	NO23	TP	OP	SED
Cropland	1	1	1	1	1
Residential	2	2	2	2	2
Pasture	3	3	3	3	3
Forest	4	4	4	4	4

For instance, one of the cropland sites in the southern portion of the western shore produced less nitrite-nitrate per acre than one of the forest sites on the upper Eastern Shore. Although this example may be anomalous, it illustrates that there is overlap in the data and that the rankings shown are general in nature and by no means apply to all sites on all soil types. They are intended to give indications of which land uses, in general, have the highest loading rates and which uses have the lowest rates, relative to one another.

Within the class of developed land use types such as residential and commercial uses, it has been shown (Smullen, Hartigan, and Grizzard 1978; Smullen 1979, NVPDC 1979) that there is a direct relationship between intensity of land use, often measured as the imperviousness of a site, and the unit area loading rate yield of nutrients. A ranking of the urban uses by loading rate is shown in Table VIII.8.

TABLE VIII.9. RANKING OF URBAN LAND USES BY UNIT AREA LOADING RATE[1] FOR NUTRIENTS (HIGHEST LOADING RATE = 1, LOWEST LOADING RATE = 7)

Land Use	Ranking
Central Business District	1
Shopping Center	2
High-Rise Residential	3
Multiple Family Housing	4
High Density Single Family Housing	5
Medium Density Single Family Housing	6
Low Density Single Family Housing	7

In general, urban uses exhibit higher unit area loading rates of nutrients than forest or pasture uses and lower rates than cropland uses. Exceptions to this "rule of thumb" are that pasture typically will yield slightly higher rates than the very low-density residential uses and that well-managed, low-tillage cropland uses on pervious soils can yield lower rates than some of the more intensive urban uses.

5. <u>What percentages of nonpoint source nutrient loadings can be attributed to particular land uses?</u>

Although it was relatively easy to sort out the point source from nonpoint source loadings in answering questions 2 and 3, it is more difficult to determine, with any level of precision, the fraction each land use contributes to the overall nonpoint load. We first must accept two basic assumptions to facilitate the estimate, and they are: (1) that the land uses are homogeneously distributed above the fall line; and (2) that baseflow loadings (groundwater contributions) of nutrients may be considered a constant background load, and the nonpoint load is measured as surface runoff and

[1] (Smullen, Hartigan, and Grizzard 1977)

interflow[1] nutrient loadings. The homogeniety of land use assumption is considered reasonable because most of the urban population resides on the coastal plain (below the Fall Line) and, with the exception of the mountainous areas, the agricultural and forest lands in the basin are fairly evenly distributed. This assumption is necessary because the closer a source is to the Bay, the more effect its loading will have upon the water quality of the system. Thus, it is important that no large mass of a particular land use type above the Fall Line be closer than any other type or there would be a skew of the loadings at the Fall Line reflecting that skew in the land use distribution. The second assumption is necessary because we just don't intuitively understand the functional relationship between land use and the quality of groundwater discharge on basins the size of the Potomac, James, and Susquehanna.[2] We do know isolated facts -- such as, the more fertilizer applied, the greater the opportunity for increasing groundwater nitrate levels and the resulting baseflow nitrate loadings in the stream. For the purpose of this analysis, it is enough to accept that for land uses that don't involve a lot of impervious cover, the baseflow loadings will move reasonably well with the runoff loadings. That is to say, that land uses exhibiting higher nutrient runoff loadings will produce groundwater discharge loadings equal to or greater than those from uses exhibiting lower runoff nutrient loadings.

The land uses above the Fall Line of the Chesapeake basin are about: 60-65 percent forested, 15-20 percent cropland, 8-12 percent pasture, 3-5 percent urban/suburban, and 2-14 percent other. These are rough estimates made from existing land use maps and will adequately serve the purpose of this "order-of-magnitude" analysis. Land use/nutrient loading rate relationships developed locally within the Chesapeake basin (Smullen, Hartigan, and Grizzard 1978, Smullen 1979, NVPDC 1979) used for this analysis are shown below:

Land Use	Percent in Basin	Estimated Loading Rate (lbs./ac./yr.)	
		TN	TP
Cropland	15-20	8-18	1.5-5
Pasture	8-12	2-6	.3-.5
Forest	60-65	.5-2	.05-.1
Urban/Suburban	3-5	4-10	1-2

[1]Interflow is the lateral movement of water through soils to streams at shallow soil depths during and directly after storm events. It is of short duration and, for our purposes, can be considered to be part of the runoff hydrograph.

[2]This is a good example of why assessments such as this are best made with mathematic models. They facilitate the orderly sorting out of base flow, runoff, and interflow and allow the analyst to handle groundwater contributions by inspection of observed flow data.

The unit area loading rates shown above were weighted by the fractions of the land areas in each use and the following ranges of loading fractions were obtained:[3]

Land Use	Percent of Nonpoint Source Load TN	Percent of Nonpoint Source Load TP
Cropland	45-70	60-85
Pasture	4-13	3-8
Forest	9-30	4-8
Urban	2-12	4-12

In summary, agricultural cropland appears to produce the largest fraction of the nonpoint source load from above the fall lines by at least a factor of two for both nitrogen and phosphorous. This is partly due to a high unit area loading rate for cropland and mostly due to the large percentage of the land area in this use. Forest loadings of nitrogen are the next highest percentage, and this is entirely due to the large fraction of the watershed still being in forest land. Urban/suburban and pasture lands above the Fall Line produce approximately equal loads.

By inspection, the percentages shown above would change very little if the Coastal Plain areas were included. Although the three major metropolitan areas (Washington, D.C., Richmond, Virginia, and Baltimore, Maryland) would increase the total amount of urban land area, this increase would probably be offset by the large rural land areas of the eastern and western shore portion of the Coastal Plain. At any rate, even if the proportion of urban area doubled, cropland would still be the largest nonpoint source nutrient load by an approximate factor of three.

6. What are the nutrient loadings from the Fall Line?

The nutrient loadings from the Fall Line are shown in Tables III.10 and again in Tables VIII.2 through VIII.5. The values for total nitrogen and total phosphorus are shown again below in millions of pounds.

	Annual	Winter	Spring	Summer	Fall
TN	178.1	51.4	72.2	25.1	27.9
TP	10.3	2.97	4.29	1.42	0.47

The percentage of the annual above fall line load produced in each season are shown below:

	Winter	Spring	Summer	Fall
TN	28.9	40.5	14.1	15.7
TP	28.8	41.7	13.8	4.6

[1]Some best and worst case assumptions were used along with some common sense judgment. For example, the lower range of cropland loading was produced by assuming the lowest loading rate/percent land use combination for cropland and the middle value of the ranges for all other uses.

From the data presented, it can be seen that the largest fraction of the fluvial nutrient load (40 percent of both nitrogen and phosphorus) is discharged to the tidal system during the spring. Observation of the data in Table III.10 shows that a large fraction of these spring loads are in forms of nutrients that are readily available for aquatic plant uptake , with 68 percent of the nitrogen as ammonia or nitrite-nitrate and 34 percent of the phosphorus as orthophosphorous. This is important since the spring is the critical start-up period for the phytoplankton growing season, the aquatic plant growth that will dominate, in part, the dissolved oxygen and chlorophyll conditions in the Bay through the summer and into the early fall. As noted elsewhere in this chapter, the predominant upstream source of the riverine transported spring nutrient load is probably runoff and groundwater discharge from agricultural lands. The next most important source of nitrogen (but probably lower by almost an order of magnitude) in spring river discharge from above the fall line is probably runoff and groundwater discharge from the melting of the snow-pack in the physiographic provinces upstream of the Piedmont (see Figure III.2).

The summer is the period during which the plankton growth in the Bay reaches the annual maximum (see Chapter 2 of this part). The fluvial transported nutrients play a lesser role during this period, providing only about 39 percent of the readily available nitrogen forms of plant nutrients and only about 5 percent of the readily available phosphorus. Plankton communities flourish during this period primarily by recycling nutrients already in the water column (put there in part by the spring fluvial process) as noted in Chapter VII (Table VII.5); and secondarily by the supply of nitrogen from atmospheric, point and benthic sources and by the supply of phosphorus from point and bottom sources.

7. <u>What do the bottom sediments contribute to nutrient inputs?</u>

On an annual basis, bottom sediments contribute 32 million pounds of nitrogen and seven million pounds of phosphorus [Table VIII.1(a)]. This makes up about 10 and 25 percent of the annual nitrogen and phosphorous budgets, respectively [Table VIII.1(b)]. However, the nitrogen contributed from the benthic source is predominately ammonia and makes up about 45 percent of the total annual Bay-wide contribution of this nitrogen species, which is most preferred by aquatic plants. More than 50 percent of the externally supplied water column ammonia produced during the spring and summer comes from the benthos.

The sediments have their most dramatic effect on the nutrient input budget as a source of phosphorous in the summer. As discussed in Chapter V, most of phosphorous migrating up through the sediments via the pore waters is probably chemically fixed by iron in the overlying oxygen-rich waters and held in a fluff layer as a small particle, or floc. This process occurs during most of the year (late fall, winter, spring). However, when the oxygen in the lower layers of the Bay waters is depleted for periods during the summer, most of the phosphorus incorporated or stored during the rest of the year is probably released over a very short period of time. The result is that as much as 62 percent of the phosphorous input to the Bay in the summer could come from this source. Other than recycling, the bottom source is probably the single largest factor in the supply of phosphorous for summer primary productivity.

8. What are the flux rates of nutrients from the bottom sediments and how do they vary seasonally?

The benthic flux rates for nitrogen range from as low as 0.5 pounds of N per square foot per day in portions of the upper Bay in the spring to as high as 5 pounds of N per square foot per day in portions of the upper Bay in the spring and summer. The annual seasonal Bay-wide average flux rates for nitrogen are shown below:

	Nitrogen Benthic Flux of Nitrogen (Thousands of Pounds per day)	Percent of Annual Average
Winter	88.1	100
Spring	75.3	85
Summer	98.4	111
Fall	91.2	103
Annual Average	88.3	

As can be seen above, the summer period exhibits the highest flux rate of nitrogen from the sediments, and the spring the lowest. The nitrogen is moving out of the sediments the fastest when the standing crop of phytoplankton is the largest, and it is being produced in a form readily available for plant uptake.

As discussed previously, the seasonal variation of phosphorous flux from the sediment to the water column is severe, with about 85 percent of the total annual input being released rapidly sometime from late May to mid-June, with most of the other 15 percent released from that time through late summer.

An educated guess at the maximum Bay-wide phosphorous release rate is that it might be as high as one-half million pounds a day during the period of the rapid onset of bottom-water anoxia. This rate probably levels off to about 16,000 pounds per day by late summer and down to near zero by sometime in late fall.

9. Given the estimated loadings of nutrients for each of the sources, which will be the most important in terms of their effects on the Bay system?

This is a difficult question to answer because there are so many potential effects on the Bay system that could result from variations in nutrient loadings. Some effects are understood well; some not so well, and some are unknown. However, to provide an answer to this question, we will consider the potential effects on Bay-wide primary production which might result from variations in the amounts of nutrients entering from various sources.

On an annual basis (Table VII.2), probably only about 20-30 percent of the Bay proper primary production is supported by nitrogen and phosphorous entering the water column from external sources. We will assume, for this exercise, that nutrient recycling rates by phytoplankton would vary only moderately in response to changes in external nutrient supply. Given this assumption, it can be seen from the data in Table VII.2 that even as much as a 50 percent reduction in both point and nonpoint source annual nutrient loadings may result in as little as a 10 percent reduction in Bay-wide primary production. Seasonally, this effect could decrease to only a 5 percent reduction in summer production in response to a 50 percent reduction of summer point/nonpoint nutrient loading. If these loading reductions were sustained, production would probably decrease futher as the nutrient reservoir in the sediments depleted over time. These estimated decreases of primary production in the short-term approach the detection limit of our ability to assess such reductions.

The important point in this discussion is that changes in lower Bay water quality (essentially meaning the great majority of the Bay that lies below the mouth of the Patuxent) in response to changes in nutrient inputs would probably take place slowly over decades. However, the upper portions of the Bay and the tidal tributaries would be much more responsive to change in nutrient loads than the main Bay. The nutrient loads that the main Bay receives must travel through these smaller, heavily impacted areas of the system.

The nutrient inputs are diluted as they move towards the lower Bay as a function of ever increasing volume. In addition, the surface area available for contributing nutrients from the sediments is much greater in the main Bay than in the upper portions of the system, resulting in much larger bottom releases of nutrients. These factors and others create a situation in the main Bay that tends to buffer or dampen water quality response to changes in anthropogenic nutrient loadings. It is, therefore, reasonable to expect the water quality of the upper areas (tidal fresh areas) of the system to respond more quickly to load reductions than the areas of the lower main Bay.

The apparent improvement in the water quality of the upper Potomac in response to decreased nutrient loadings over the last decade would seem to support this concept. Even though some unknown amount of that improvement is probably due to differing climatic conditions over the last ten years, some degree of the improvement is most likely due to the decreases in the external nutrient supply from POTW's. We would not expect to see immediate changes in lower Bay water quality due to that reduction of loading and, in fact, have not. Such a change could only be seen over a much longer period of time and to a lesser (diluted) extent. This situation would seem to support the concept that if we manage the local ("near field") problems, the main Bay ("far field") will, in time, respond in kind. An aggressive policy of water quality improvement in currently adversely impacted areas should insure the maintenance of a nondegradation condition in the main Bay.

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APPENDIX A

METHODOLOGY FOR COMPUTATION OF NUTRIENT AND SEDIMENT LOADS TRANSPORTED TO THE BAY BY RIVERS

The dependent or response variable in each of the models is based upon the flow - weighted daily mean concentration of the constituent for which the model is being formulated. This was done to normalize the effects of observations taken on the rising versus falling limb of a storm hydrograph. To determine flow-weighted concentrations for days when multiple observations were collected, the products of the concentration for an observation and the instantaneous flow recorded for that observation were summed over all the observations in a day, and that sum was divided by the sum of the instantaneous flows. This is shown in equation A-1:

$$C_J = \frac{\sum_{i=1}^{n} C_i \, q_i}{\sum_{i=1}^{n} q_i} \qquad \text{(eq. A-1)}$$

where C_J = flow-weighted mean daily constituent concentration
C_i = individual constituent concentration observation (mg/l)
q_i = instantaneous discharge at time of observations 'c' (CFS)
n = number of observations in day 'J'.

For the special case of n = 1, that is only one observation taken on a particular day (e.g., a base flow observation), the mean daily concentration is simply set equal to the observed concentration, or $C_J = C_i$, after equation A-1.

Model Formulations: Models were developed using the basic least squares regression normal error model (Neter and Wasserman, 1974) stated as:

$$Y_i = B_o + B_1 X_i + e_i \qquad \text{(eq. A-2)}$$

where B_0 and B_1 are parameters
Y_i and X_i are known constants (dependent and independent variables)
e_i are independent $N(0,S^2)$
(The dependent variable Y_i is based on C_J, Equation A-1)

All model formulations attempted are based upon the simple linear regression model (eq. A-2) or the use of some remedial measure involving transformations of the data. Transformations were chosen either to linearize the regression function (semi-logarithimic or fully-logarithmic) or to stabilize the error term variance. Full descriptions of the methodologies and validity of these approaches can be found in Neter and Wasserman (1974) or most basic linear statistical models texts.

In all, twelve separate models were tested, made up of three sub-groups with arithmatic, semi-log transformation by each axis, and log-transformations on both axes performed within each sub-group. All logarithms are taken as Napierian logarithms. The model formulations are described below.

Concentration Models: The basic form of this model sub-group is the relationship between the mean daily, flow-weighted concentration of a constituent (C, mg/l) versus the mean daily discharge. (Q in cubic feet per second per day [Cfsd]). The four models investigated are shown below:

i) C versus Q
ii) ln(C) versus Q
iii) C versus ln(Q)
iv) ln(C) versus ln(Q)

Loading Rate Models: The basic form of this model sub-group is the relationship between daily constituent loading rate (LR, lbs/day), computed as the product of the flow-weighted constituent concentration for the day, the mean daily discharge for the day, and a conversion factor, versus the mean daily discharge (Q, cfsd)[1]. The four transformations investigated are shown below:

i) LR versus Q
ii) ln(LR) versus Q
iii) LR versus ln(Q)
iv) ln(LR) versus ln(Q)

It is noted that a functional relationship exists between the mass of pollutant washed off and discharge that is inherent in the determination of the dependent variable term for this model. It follows that the use of the least squares method may not result in the best linear unbiased estimation of the data in the Gauss-Markov theorem sense.[2] Although other biased or nonlinear estimation approaches such as distribution-free or non-parametric statistics might yield smaller variances, the least squares approach was chosen for its simplicity and ease of application. It is also noted that although the coefficients of determination developed from these models remain useful for comparison with other models, the 't' tests for the slope may not be useful for comparison with the other models because of the suspected bias in the relationships.

Variance-Stabilizing Transformation Models: The basic form of this sub-group involves a transformation to stabilize error variances. Residual analysis through scatter plot observations of the C versus Q type models (above) suggested that in many cases the variance of the error was increasing with the volume of discharge. That is, the relationships appeared to have heterosadastic tendencies, exhibiting non-constant variance over the range of observed flows. Therefore, the estimator B_o

---

[1]The nutrient loading rate is computed as:

$$LR = K \times C \times Q$$

where LR = Nutrient loading rate (lbs/day)
K = Conversion factor equal to 5.38 (liter-sec.-lb/mg.-ft^3-day)
C = Nutrient concentration in mg/l
Q = Mean daily discharge in cubic feet per second

[2]That is to say, the least squares estimator may not have a minimum variance within the class of linear, unbiased estimators.

and B_1 (eq. A-2), though still unbiased, are no longer minimum variance unbiased estimators. Neter and Wasserman (1974) suggest a transformation of the form:

$$Y' = \frac{Y}{X} \quad \text{and} \quad X' = \frac{1}{X} \qquad \text{(eq. III-3, III-4)}$$

to minimize the variance. The general form, then, of this group of model is as follows:

i) (C/Q) versus (1/Q)
ii) ln(C/Q) versus (1/Q)
iii) (C/Q) versus ln(1/Q)
iv) ln(C/Q) versus ln(1/Q)

Neter and Wasserman point out that this transformation is really equivalent to using weighted least squares and further indicate that the relationship remains unbiased. Regression statistics ('t' tests) remain fully useful.

Regression Analysis

The correlation coefficient for each of the models described above were computed at each site for the water quality parameters listed in Table A.1. The coefficients of determination for the Susquehanna, Potomac and James models are shown in Table A.2, A.3 and A.4. Only models exhibiting coefficients in excess of 0.50 are shown.

TABLE A.1. WATER QUALITY VARIABLES INCLUDED IN REGRESSION ANALYSIS

Water Quality Parameter	STORET No.	Variable Name
Total Nitrogen (as N) (Particulate & dissolved)	600	TN
Dissolved Nitrogen (as N)	602	DN
Total Kjeldahl Nitrogen (as N)	625	TKN
Total Nitrite plus Total Nitrate Nitrogen (as N)	630	$NO_2 + NO_3$ or NO23
Total Ammonia Nitrogen (as N)	610	$NH_3 + NH_4$ or NH34
Total Phosphorus (as P) (Particulate & dissolved)	665	TP
Dissolved Phosphorus (as P)	666	DP
Total Orthosphosphorus (as P)	70507	OP
Suspended Sediment	80154	SED

The Tables (A.2, A.3, and A.4) show that poor fits (r^2 0.50) were found in almost all cases for the concentration-versus-discharge models.

In most cases, visual inspection of scatter diagrams allows a case to be made for heteroodasticity and, for this reason, the variance-stabilizing transformation were favored in selecting appropriate models.[1] Only when correlation coefficients were significantly below 0.65 or 't' tests ($H_o:B_1 = 0$) indicated that B_1, the slope, was not significantly different from zero at the 95 percent confidence level was a loading rate model chosen.

[1]During the course of examination of the concentrations predicted by each of the models over the range of flow observed in the period of record, it was determined that the arithmetic form of the variance-stabilizing transformation (C/Q versus 1/Q) yielded unrealistically high values for discharges in excess of those observed during the period of the monitoring program. The log-log transformation of this model [ln(C/Q) versus ln(1/Q)] proved to be much better behaved in predicting concentrations for these higher flows. The curves produced with this transformation "flatten out" very quickly as flows approach those at the upper limit of the discharge data observed during the field program. Therefore, only the log transformed versions of the variance-stabilizing transformation were considered for cases exhibiting heterosodasticity.

TABLE A.2. REGRESSION MODEL RESULTS FOR THE SUSQUEHANNA RIVER AT CONOWINGO, MD (1578310) RESULTS DISPLAYED ONLY FOR MODELS EXHIBITING COEFFICIENTS OF DETERMINATION IN EXCESS OF 0.50

Water Quality Constituent	Model	r^2	d.f.
TN	C/Q vs. 1/Q	.948	86
	C/Q vs ln (1/Q)	.503	86
	ln (C/Q) vs. ln (1/Q)	.916	86
	LR vs. Q	.852	86
	ln (LR) vs.Q	.722	86
	LR vs. ln (Q)	.611	86
	ln (LR) vs. ln (Q)	.933	83
DN	C/Q vs. 1/Q	.972	66
	C/Q vs ln (1/Q)	.518	66
	ln (C/Q) vs. ln (1/Q)	.926	66
	LR vs. Q	.832	66
	ln (LR) vs.Q	.742	66
	LR vs. ln (Q)	.618	66
	ln (LR) vs. ln (Q)	.931	66
NO23	C/Q vs. 1/Q	.927	86
	ln (C/Q) vs 1/Q	.382	86
	ln (C/Q) vs. ln (1/Q)	.887	86
	LR vs. Q	.859	86
	ln (LR) vs.Q	.668	86
	LR vs. ln (Q)	.694	86
	ln (LR) vs. ln (Q)	.906	86
NH34	ln (C/Q) vs. ln (1/Q)	.575	87
	LR vs. Q	.558	87
	ln (LR) vs.Q	.611	86
	ln (LR) vs. ln (Q)	.713	86
TKN	C/Q vs. 1/Q	.916	87
	C/Q vs ln (1/Q)	.551	87
	ln (C/Q) vs. ln (1/Q)	.805	87
	LR vs. Q	.644	87
	ln (LR) vs.Q	.707	87
	ln (LR) vs. ln (Q)	.850	87

(continued)

TABLE A.2. (continued)

Water Quality Constituent	Model	r^2	d.f.
TP	C vs. Q	.518	87
	ln (C) vs. Q	.503	97
	C/Q vs. 1/Q	.863	87
	ln (C/Q) vs. ln (1/Q)	.565	87
	LR vs. Q	.696	87
	ln (LR) vs.Q	.792	87
	ln (LR) vs. ln (Q)	.885	87
DP	C/Q vs. 1/Q	.565	88
	ln (C/Q) vs. ln (1/Q)	.778	85
	LR vs. Q	.597	88
	ln (LR) vs.Q	.612	85
	ln (LR) vs. ln (Q)	.797	85
OP	ln (C/Q) vs 1/Q	.512	60
	ln (C/Q) vs. ln (1/Q)	.639	66
	ln (LR) vs.Q	.600	66
	ln (LR) vs. ln (Q)	.730	66
SED	ln (C) vs. Q	.677	96
	C/Q vs. 1/Q	.542	93
	ln (C/Q) vs 1/Q	.667	93
	LR vs. Q	.550	93
	ln (LR) vs.Q	.741	93
	ln (LR) vs. ln (Q)	.665	93

TABLE A.3. REGRESSION MODEL RESULTS FOR THE POTOMAC RIVER AT CHAIN BRIDGE, WASHINGTON, DC (1646580) RESULTS DISPLAYED ONLY FOR MODELS EXHIBITING COEFFICIENTS OF DETERMINATION IN EXCESS OF 0.50

Water Quality Constituent	Model	r^2	d.f.
TN	C/Q vs. 1/Q	.847	64
	ln (C/Q) vs 1/Q	.551	64
	C/Q vs ln (1/Q)	.776	64
	ln (C/Q) vs. ln (1/Q)	.856	64
	LR vs. ln (Q)	.571	64
	ln (LR) vs. ln (Q)	.914	64
DN	C/Q vs. 1/Q	.704	63
	ln (C/Q) vs 1/Q	.597	63
	C/Q vs ln (1/Q)	.681	63
	ln (C/Q) vs. ln (1/Q)	.840	63
	LR vs. Q	.721	63
	ln (LR) vs.Q	.717	63
	LR vs. ln (Q)	.592	63
	ln (LR) vs. ln (Q)	.913	63
NO23	C/Q vs. 1/Q	.637	64
	C/Q vs ln (1/Q)	.618	64
	ln (C/Q) vs. ln (1/Q)	.804	64
	LR vs. ln (Q)	.637	64
	ln (LR) vs. ln (Q)	.869	64
NH34	ln (LR) vs. ln (Q)	.707	61
TKN	C/Q vs. 1/Q	.682	80
	ln (C/Q) vs 1/Q	.580	80
	C/Q vs ln (1/Q)	.565	80
	ln (C/Q) vs. ln (1/Q)	.719	80
	ln (LR) vs.Q	.531	80
	ln (LR) vs. ln (Q)	.849	80
TP	C/Q vs. 1/Q	.510	80
	ln (C/Q) vs. ln (1/Q)	.588	79
	ln (LR) vs.Q	.549	79
	ln (LR) vs. ln (Q)	.847	79
DP	ln (C/Q) vs 1/Q	.576	77
	ln (C/Q) vs. ln (1/Q)	.718	77
	ln (LR) vs. ln (Q)	.801	77

(continued)

TABLE A.3. (continued)

Water Quality Constituent	Model	r^2	d.f.
OP	C/Q vs. 1/Q	.583	56
	ln (C/Q) vs. ln (1/Q)	.622	47
	ln (LR) vs.Q	.573	47
	ln (LR) vs. ln (Q)	.696	47
SED	C vs. Q	.515	60
	ln (C) vs. Q	.512	60
	ln (C) vs ln (Q)	.658	60
	LR vs. Q	.796	60
	ln (LR) vs.Q	.640	60
	ln (LR) vs. ln (Q)	.879	60

TABLE A.4. REGRESSION MODEL RESULTS FOR THE JAMES RIVER AT CARTERSVILLE, VA. (2035000) RESULTS DISPLAYED ONLY FOR MODELS EXHIBITING COEFFICIENTS OF DETERMINATION IN EXCESS OF 0.50

Water Quality Constituent	Model	r^2	d.f.
TN	C/Q vs. 1/Q	.769	54
	ln (C/Q) vs 1/Q	.679	54
	C/Q vs ln (1/Q)	.633	54
	ln (C/Q) vs. ln (1/Q)	.817	54
	LR vs. Q	.842	54
	ln (LR) vs.Q	.728	54
	LR vs. ln (Q)	.523	54
	ln (LR) vs. ln (Q)	.909	54
DN	C/Q vs. 1/Q	.861	38
	ln (C/Q) vs 1/Q	.728	38
	C/Q vs ln (1/Q)	.713	38
	ln (C/Q) vs. ln (1/Q)	.894	38
	LR vs. Q	.720	38
	ln (LR) vs.Q	.713	38
	ln (LR) vs. ln (Q)	.909	38
NO23	C/Q vs. 1/Q	.530	56
	ln (C/Q) vs 1/Q	.509	56
	ln (C/Q) vs. ln (1/Q)	.759	56
	LR vs. Q	.816	56
	ln (LR) vs.Q	.568	56
	LR vs. ln (Q)	.614	56
	ln (LR) vs. ln (Q)	.824	56
NH34	ln (C/Q) vs. ln (1/Q)	.646	49
	LR vs. Q	.525	56
	ln (LR) vs.Q	.551	49
	ln (LR) vs. ln (Q)	.725	49
TKN	C/Q vs. 1/Q	.653	55
	ln (C/Q) vs 1/Q	.617	55
	C/Q vs ln (1/Q)	.509	55
	ln (C/Q) vs. ln (1/Q)	.665	55
	LR vs. Q	.786	55
	ln (LR) vs.Q	.727	55
	ln (LR) vs. ln (Q)	.862	55

(continued)

TABLE A.4. (continued)

Water Quality Constituent	Model	r^2	d.f.
TP	C/Q vs. 1/Q	.781	56
	ln (C/Q) vs 1/Q	.718	56
	C/Q vs ln (1/Q)	.522	56
	ln (C/Q) vs. ln (1/Q)	.669	56
	LR vs. Q	.715	56
	ln (LR) vs.Q	.758	56
	ln (LR) vs. ln (Q)	.762	56
DP	C/Q vs. 1/Q	.873	56
	ln (C/Q) vs 1/Q	.754	56
	C/Q vs ln (1/Q)	.588	56
	ln (C/Q) vs. ln (1/Q)	.908	56
	LR vs. Q	.626	56
	ln (LR) vs.Q	.526	56
	ln (LR) vs. ln (Q)	.581	56
OP	C/Q vs. 1/Q	.809	48
	ln (C/Q) vs 1/Q	.725	46
	C/Q vs ln (1/Q)	.537	48
	ln (C/Q) vs. ln (1/Q)	.815	46
	ln (LR) vs. ln (Q)	.511	46
SED	C vs. Q	.631	71
	ln (C) vs. Q	.539	71
	C vs. ln (Q)	.535	71
	ln (C) vs ln (Q)	.723	71
	LR vs. Q	.841	71
	ln (LR) vs.Q	.647	71
	ln (LR) vs. ln (Q)	.903	71

SUMMARY AND CONCLUSIONS

Increased population and changed land uses (deforestation, agriculture, urbanization) in the Chesapeake Bay watershed have caused increases in nutrient and sediment loads to the Bay system over the past 50 years. The challenge facing water quality managers is to restrict excessive nutrient addition as efficiently as possible, at the most effective times and locations, and for the least expense. The Nutrients projects of the Chesapeake Bay Program, discussed in this Synthesis Paper, were designed to help managers meet this challenge by increasing their understanding of the nutrients problem, its sources, and important processes.

THE NUTRIENT ENRICHMENT PROBLEM

Many areas of the Chesapeake Bay system have a serious nutrient enrichment problem. Our studies indicate that the upper Bay, between Turkey Point and the Bay Bridge, upper Potomac, upper Patuxent, and upper James are heavily enriched; the mid-Bay, lower Patuxent, lower Potomac, Rappahannock, York and middle James Rivers are moderately enriched. These areas are considered to be enriched, because chlorophyll _a_ levels are elevated over historical levels.

Nutrient enrichment results in enhanced phytoplankton production and higher levels of organic matter in the water column. Decomposition of excessive organic matter results in depletion of oxygen from deeper waters, posing a hazard for bottom-dwelling animals.

Nutrient enrichment also alters the composition of phytoplankton species. Such changes have been shown to affect fisheries species composition in other systems, and may have the same effect in Chesapeake Bay.

The increased turbidity resulting from nutrient enrichment decreases the amount of light available for submerged aquatic vegetation (SAV), and has been implicated in their decline.

Solving the Problem: The Importance of Nutrient Processes

The extreme solution to nutrient enrichment problems is to eliminate the entry of excess nutrients to the Bay system. However, because of social and economic restraints, this is not possible. Instead, we must restrict the particular nutrients most responsible for the problems, at the most effective times and places. To do this requires an understanding of nutrient processes.

Seasonal factors determine the times at which nutrient restriction is the most effective. The availability of nutrients follows an annual cycle. In spring, heavy flows bring a substantial amount of nitrate into the Bay. In summer, deoxygenation of deep water results in release of phosphate from bottom sediments and the accumulation of phosphate and ammonium in bottom waters. In the fall, re-oxygenation of deep waters results in loss of phosphate from the water column and oxidation of ammonium to nitrite and nitrate.

Nutrient limitation of phytoplankton growth is a function of nutrient availability and the intrinsic requirements of phytoplankton. Healthy phytoplankton require carbon, nitrogen, and phosphorus in certain ratios. The nutrient in least supply with respect to the requirements of phytoplankton will limit their growth. Nutrient limitation occurs only if other environmental conditions (like light availability) are satisfactory; when too little light is available, for example, light becomes the limiting factor. When a nutrient is limiting, addition of that nutrient will stimulate phytoplankton growth.

Phosphorus is potentially limiting in the tidal-fresh reaches of the Bay throughout the year (sediments of tidal-fresh segments do not become anoxic, so phosphorus is not released from them). In the remainder of the Bay system, phosphorus is potentially limiting in spring and fall; nitrogen is potentially limiting in summer. Light is limiting in winter and in situations of high turbidity.

Whether increases in algal production result in problems depends, in part, on nutrient cycling. Water column nutrient cycling processes, such as hydrodynamics and grazing, help remove excess plankton biomass from the system. Decomposition, on the other hand, depletes the system of oxygen. Regeneration of inorganic nutrients through these processes provides a source of nutrients for phytoplankton growth.

Nutrient cycling processes in the sediments affect levels of nutrients in the water column. Phosphorus is removed from the water column by adsorption to iron and manganese compounds, to be released in summer from anoxic areas. Ammonium fluxes into, or out of, the sediments, depending on pore water concentrations and oxidation state.

Marshes and Bay grasses contribute to nutrient recycling by taking up nutrients during their growing season (periods of peak availability) and releasing nutrients during the winter through decomposition. Thus, they act as nutrient buffers.

Once the role of specific nutrients in specific times and places is understood, nutrient sources must be known before exact and economical solutions to nutrient problems can be developed.

Nutrient Sources: The Key to Comprehensive Control

On an annual basis, atmospheric contribution to the tidal waters of the Bay system make up about 13 percent of the nitrogen and five percent of the phosphorus. In winter, up to 20 percent of the nitrogen may come from atmospheric sources.

Point sources contribute up to 25 percent of the nitrogen and 40 percent of the phsophorus annually. While the nitrogen contribution varies little during the year, the phosphorus contribution may be as much as 65 percent in the fall.

Nonpoint sources (land runoff/base flow) contribute up to 55 percent of the nitrogen annually, the largest source of this nutrient to the Bay system. In spring, nonpoint sources contribute up to 66 percent of the total nitrogen. Nonpoint sources of phosphorus make up about 34 percent of the annual total; in spring the contribution is about 55 percent. The land-use contributing the most to these percentages, both on a unit area basis and as percentage of the total, is cropland.

Bottom sediments contribute about 10 percent of nitrogen and 25 percent of phosphorus annually. The nitrogen contribution, primarily as ammonia, is more than 50 percent in spring and summer. The bottom of the Bay is the largest contributor of phosphorus in the summer, supplying as much as 62 percent of this nutrient.

Appendix A

CBP Nutrient Enrichment Projects

Definition of Chesapeake Bay Problems of Excessive Enrichment or Eutrophication	L. Eugene Cronin Bruce Nielson Andrew McErlean Donald Heinle Kenneth Webb Jay Taft	Chesapeake Research Consortium
Evaluation of Management Tools in Two Chesapeake Bay Watersheds in Virginia	Robert V. Davis Thomas Grizzard Bruce Nielson	Virginia State Water Control Board
Evaluation of Water Quality Management Tools in the Chester River Basin	Howard Wilson Charles Bostater	Maryland Water Resources Administration
Intensive Watershed Study (Patuxent River Basin)	Howard Wilson Charles Bostater	Maryland Water Resources Administration
An Assessment of Nonpoint Source Discharge, Pequea Creek Basin, Lancaster County, Pennsylvania.	Robert J. Bielo Janice Ward	Susquehanna River Basin Commission
Modeling Philosophy and Approach for Chesapeake Bay Program Watershed Studies	Robert Ambrose	U.S. EPA Environmental Research Laboratory, Athens, Georgia
Fall Line Monitoring of the Potomac, Susquehanna, and James Rivers	David Grason David Lang	Water Resources Division, U.S. Geological Survey
Land Use and Point Source Nutrient Loading in the Chesapeake Bay Region	Benjamin J. Mason	Geomet, Incorporated
Chesapeake Bay Circulation Model	Robert Shubinski	Water Resources Engineers, Inc.
Water Quality Laboratory for Chesapeake Bay and its Subestuaries at Hampton Institute	Larry T. Cheung	Hampton Institute
Chesapeake Bay Nutrient Dynamics	Jay Taft	Chesapeake Bay Institute

PART III

TOXIC SUBSTANCES

by

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FIGURES

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Technical Glossary

aerosol:	A colloidal solution in which a substance in which the other is dispersed, is a gas.
anoxic:	Total deprivation of oxygen.
anthropogenic:	Of human origin and development.
As	arsenic
bioecology:	The science that deals with the interrelations of communities of animals and plants with their environment.
Cd	cadmium
Ce	cesium
Co	cobalt
Cr	chromium
Cu	copper
diagenesis:	Physical and chemical changes occurring to sediments during and after the period of decomposition up until the time of consolidation.
dpm cm^{-2}	distintegrations per minute per square centimeter
Eh	oxidation-reduction potential
fall line:	Geographical line indicating the beginning of a plateau, usually marked by many waterfalls and rapids.
Fe	iron
ft^3/sec	cubic feet per second
Hg	mercury
interstitial:	Of, forming, or occurring in small or narrow spaces between things or parts.
lithology:	Science of rock structures.
loads:	Quantity of a constituent per unit per time.
Mn	manganese
Mo	molybdenum

mT	metric tons
Ni	nickel
oxic:	applied to a soil layer from which much of the silica that was combined with iron and alumina has been leached.
Pb	lead
ppt	parts per thousand
$ps^=$	expression of sulfur ion content
Sc	scandium
Sn	tin
synergism:	The property or condition of working together, such as muscles together effecting a certain motion, or of hormones, or medical substances.
Th	thorium
U	uranium
ug cm^{-2}	micrograms per centimeter
Zn	zinc

SECTION 1

INTRODUCTION

This part of the CBP Synthesis Report summarizes and integrates the research findings and recommendations of 13 projects of the Chesapeake Bay Toxic Substances Program performed between July 1978 and October 1981. The following sections describe research on potentially toxic substances, or toxicants, in water-sediments and selected biota. The subjects considered include a brief review of metals, their sources, distribution and behavior, and then a review of sources and distribution of organic chemicals. Finally, information concerning the significance of toxicants in the Bay and their pattern of enrichment is provided. Most information synthesized in this report can be traced to its origin in scientific project reports listed in Appendix A.

The last three decades have witnessed some disturbing changes in Chesapeake Bay. Some biotic components are less abundant than in the past and are below natural levels. Oyster reproduction has diminished throughout the Bay. Of particular concern is the virtual disappearance of rooted aquatic plants over a large portion of the Bay floor. Fish, such as shad and striped bass, once spawned in astronomical numbers; but in recent years, they have declined severely (Cronin 1977). Taken together, these changes are cause for concern.

An understanding of what is happening, and why, to grass, bass, shad, and oysters still eludes scientists, though toxic substances are strongly suspected to be at least partially responsible. The lessons learned from DDT and PCB contamination show that toxicants can cause substantial ecological damage, ranging from reproductive failure in fish and birds to inhibition of photosynthesis in phytoplankton. The outbreak of neurological illness with 52 deaths caused by mercury (Hg) poisoning of shellfish in Japan amplifies the fact that toxic contamination in seafood resources can reach humans. Release of Kepone into the James River in Virginia, resulted in closure of the estuary to fishing for years, with an enormous economic loss and a need for a large-scale, expensive cleanup. Chlorine, a widely used biocide in sewage treatment plants, is strongly suspected of causing massive fish kills in the James River in 1973 (Douglas 1979).

Toxic substances are usually defined as chemicals or chemical compounds that can poison living plants and animals, including humans, or impair physical or chemical processes. Two classes of toxic substances pose a threat to the Bay environment: inorganic and organic compounds. The inorganic materials are the metals. They can be produced and delivered to the Bay by natural processes as well as by human activities. Potentially toxic metals include arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), tin (Sn), and zinc (Zn). Many of the organic compounds are products of human activities. However, a few polynuclear aromatic compounds (PNAs) can occur naturally, and thus augment the synthetic compounds. The main classes of organic compounds are pesticides, phthalate esters, polynuclear aromatic hydrocarbons, metalorganic compounds, alkyl-benzines, plasticisers, polychlorinated biphenyls (PCBs), and other halogenated hydrocarbon compounds.

Assessing the effects of toxic substances on biota has always been a difficult task. Effects range from rapid death, or acute toxicity, to gradual reductions in spawning success, or chronic toxicity. Months, or years of careful observations may be required to determine chronic effects for one chemical on one species. Effects of chemical mixtures on several species or a community are even more difficult to detect. The environment may also experience synergistic and antagonistic effects through exposure to two or more chemicals. In addition, toxic effects can be masked by wide fluctuations in natural conditions. In the laboratory, scientists have attempted to simulate effects of chemicals on the natural environment by subjecting single organisms, or a limited number of organisms, to toxicants, and observing the cause-and-effect relationships. But transfer of this information to interpret changes in entire faunal communities, with their wide variability within species, has achieved only limited success.

Because it is difficult to specify cause-and-effect relationships between toxicants and Bay resources, we attempted, during the Chesapeake Bay Program, to determine areas where levels of toxicants are high (above standards or threshold levels), and then relate these levels to known toxic effects. This evaluation will give us some insight into the existence of toxicity problems.

In summary, some trends recognized at the onset of the Program caused us to believe that the status of toxic substances in the Bay should be studied. These trends included: (1) decline of biotic components in the past three decades (Cronin et al. 1977); (2) increases in the number of potentially toxic chemicals being synthesized, produced, and used in the region (Huggett et al. 1977); (3) discharge of large amounts of potentially toxic substances (Brush 1974); (4) increase in population growth and industrial activity; (5) accumulation of toxicants in the sediments and biota, including commercial food species, many thousand-fold more than in ambient concentrations in the water (Huggett et al. 1974b, Huggett et al. 1977); and (6) carcinogenic nature of many organic compounds found in the Bay. At the same time the Bay is an important environmental resource for fisheries, wildlife, and recreation. Since controlling the threat of toxic substances to viable ecological resources requires new knowledge of their sources, distribution, and fate in the Bay ecosystem, we studied these factors.

Before the initiation of the CBP, information on metals and organic compounds was scarce. Data on the existence of metals were limited to the distribution and abundance of some trace metals in the northern Bay and several western tributaries. Likewise, available information on organic compounds consisted of levels of some chlorinated hydrocarbons (DDT, PCBs) and Kepone found in selected bivalves, fish, phytoplankton, and sediments of some parts of the Bay and tributaries. The CBP studies not only support and systematically expand this knowledge, but add information on sources of metals and organic compounds to the Bay, their behavior in the estuary, and impacts on resources.

Published information on potentially toxic metals in the Bay prior to the Chesapeake Bay Program, and from other studies, is summarized in Appendix B. Of note are studies of the Cu and Zn in oysters and sediments of the James and Rappahannock Rivers (Huggett et al. 1974a) that indicate differences in concentration gradients of the metals between sediments and oysters. Additionally, Carpenter et al. (1975) revealed marked temporal variations of the dissolved and suspended metals, Fe, Mn, and Zn, discharged over an annual cycle by the Susquehanna River. Our studies support these findings as discussed in Section 2. Villa and Johnson (1974) and Johnson and Villa (1976) reported high concentrations of metals in Baltimore Harbor and the Elizabeth River. By using a mass balance of metals for Northern Chesapeake Bay, Helz (1976) found that at least half of the Cd, Cr, Cu, and Pb input comes from human sources. Further assessment of contributions from human sources is presented in Section 2. Goldberg et al. (1978a), in a study of northern and central Chesapeake Bay, revealed anthropogenic fluxes of metal concentrations in upper parts of sediment cores. Since this study showed that sediment puts material into the system, we assessed sediments as a source. (See Section 2 for discussion of our results). The status of knowledge on biological effects of metals is presented by Frazier 1972, Cronin et al. 1974, Hansen et al. 1974, and Tsai et al. 1979. These studies indicate a biological toxicity problem that was cursorily studied by the CBP (see Section 6).

Prior information on synthetic organic compounds in the Bay is scant. Many synthetic compounds have been only newly created, with the necessary analytical instruments to detect them only recently developed. Of note (Appendix C) is the EPA National Estuarine Monitoring Program between 1965 and 1972, utilizing oysters (Munson and Huggett 1972). Additionally, Munson (1973) found that Chester River bed sediments, suspended sediment, and shellfish stocks contained chlorinated hydrocarbons derived from Chesapeake Bay. The Upper Bay Survey (Munson 1975) provided data on chlorinated hydrocarbon (PCBs, Chlordane, and DDT) sources and concentrations in suspended material and bed sediments as well as in shellfish and zooplankton. This study gave insights into routes and rates of transfer. Section 2 of this paper expands on this information. A consolidated listing of toxicants found in Chesapeake biota, water, and sediment, and a listing of toxicant data files is provided by CRC (1978). The intensive studies of Kepone in the James Estuary after 1975 provide detailed data for a single toxicant in a single tributary estuary. They cover studies of biota (Roberts and Bendl 1980, Huggett et al. 1980, Huggett and Bender 1980) and sediments (Trotman and Nichols 1978, Lunsford 1981, Nichols et al. 1979).

Brush's (1974) inventory of sewage treatment plants lists information on sources of toxicants. Additionally, the EPA-States National Pollutant Discharge Elimination System (NPDES), which began in 1973, contains extensive file data on metals and a few organic compounds discharged from point sources such as industrial effluents and sewage treatment plants.

In 1978 the CBP initiated research on toxic substances, aiming to provide new information and the data base necessary to manage toxic inputs to the Bay. It is the first comprehensive effort to address problems of potentially toxic substances in the Bay on a regional scale. Specifically, we attempted to:

- determine the present distribution and concentration of selected toxic substances in Bay sediments, water, and biota;
- assess the present input rates of potentially toxic substances to the Bay, their location, and composition;
- identify the major transport paths for toxic substances, their chemical behavior, and sites of accumulation; and

o determine the impacts of potentially toxic substances on the Bay ecosystem.

The chief studies were of four main types:

(1) Baseline Inventory
An assessment of the spatial distribution of sediments, biota, water characteristics, and toxic substances, (what toxic substances are present? where are they located?) and in what form or state (organic, inorganic, dissolved, particulate?) (Are they a problem?)

(2) Source Assessment
An identification of sources and estimation of the potential toxic inputs discharged by industry, sewage treatment plants, and the atmosphere.

(3) Behavior and Fate
An assessment of the mechanisms and routes of transport, sites of accumulation, chemical behavior, and likely biological impacts.

(4) Synthesis
A summary of research findings and integration of toxic substances with system components.

The program elements are interrelated scientifically by treating the Bay as a geochemical system with reservoirs. Sources, sinks, and pathways of material transports (such as air, water, and sediments) are the principal reservoirs inventoried; dissolved materials and biota are the main interacting components. As toxic substances are transferred between reservoirs and components, and from sources to sinks, they proceed along characteristic pathways, undergo transformation, and accumulate in viable and sedimentary constituents.

Research plans focused on toxic substances in the sediment reservoir, because toxicants have a great affinity for fine-grained sediment (which has a large surface area for sorption per unit mass). Levels in the water column may, in some cases, be important, but our work concentrated on sediment reservoirs because toxicants have a long residence time in sediments, build up to high concentrations, and are easily detected. Although toxic substances discharged in dissolved form can have a direct impact, their effect is believed to be short-lived because of rapid water movement and constant dilution. Consequently, sediments have a longer residence time in the Bay than dissolved substances. Thus, they can build up high concentrations of toxicants.

Growth of the region has increased the supply of sediment delivered to the Bay and, when combined with toxic substances, poses a significant problem to the Bay environment. Clearing land for agriculture and development has accelerated watershed erosion (Wolman 1967) and increased loads of suspended sediment (Schubel and Meade 1974). Suspended sediment creates turbidity which can decrease light penetration and adversely affect aquatic plants and primary production. As sediment fills channels and harbors, it creates a need for dredging and for disposal of contaminated sediment.

As in the previous part on nutrient enrichment, this section was written around several questions relevant to those interested in managing water quality of the Bay. The three basic questions addressed in this paper are:

Is there a toxic chemical problem in the Bay?
What is the distribution of toxic chemicals in the Bay?
What are the sources and loadings of pollutants of concern?

A more detailed list of these questions, with their answers, appears as the final section of this paper. The answers are drawn from the paper and serve as a summary of the technical material from a manager's perspective. They should concisely support Section 6, Conclusions and Research Needs.

SECTION 2

FINDINGS FROM STUDIES ON METALS

This chapter explains the results from CBP research on sources of metals to the Bay and their distribution and concentration in the estuary. The first part on sources discusses inputs of metals from industries and publicly owned treatment works (POTWs), the atmosphere, urban runoff, and three principal tributaries of Chesapeake Bay. The remaining sections summarize results from CBP studies on the concentration of metals in the Bay. Once in the estuary, the behavior of metals depends on how they respond to the Bay's chemical, biological, and physical processes. Some metals, for example, will become dissolved in the estuarine water. Others will associate with suspended matter, while certain amounts and types will be found in bottom sediments and interstitial water. This section deals with metals partitioned in all of their phases.

SOURCES

The CBP initiated studies to assess the input of metals from several major sources to the Bay. These sources are: industries and POTWs, atmospheric deposition, urban runoff, and three of the Bay's principal tributaries. Approximate loadings were computed for these sources to provide an estimate of the relative contributions each source makes.

Industries and POTWs Below the Fall Line

Rates of metal input from point sources in the Bay drainage basin were estimated for industries and POTWs below the fall line from data obtained between 1974 and 1980. Information from the National Enforcement Investigations Center (NEIC) of the U.S. Environmental Protection Agency (EPA) was used to place in priority the toxic dischargers from the approximately 5000 point source dischargers in the entire Chesapeake Bay basin. It was determined that there are 1000 major toxic dischargers, of which 122 are located below the fall line. For these 122 industries, loading estimates were computed for selected metals we found in relatively high concentrations in Bay sediments.

Concentration of metals in various industrial effluents was obtained from EPA effluent sampling data from Resources for the Future in the "Pollution Matrix Lookup Routine." Concentration values were assigned based on the industry's Standard Industrial Classification (SIC) code. The discharge rates for each industry were obtained from data collected for an EPA project referred to as the "Industrial Facilities Discharger File" (IFD). Loadings of metals in metric tons per year were computed by multiplying the effluent discharge rate (in millions of gallons per day [MGD]) by the concentration of the various metals milligrams per liter (mg/L), applying the appropriate conversion factors. However, when assigning effluent concentration values, the industries discharging cooling water were assigned concentrations representative of cooling water, not waste water. Those industries discharging cooling water and process

TABLE 1. POINT SOURCE LOADINGS OF METALS FROM INDUSTRIES[2] AND PUBLICLY OWNED TREATMENT WORKS (POTW'S)[1] IN COUNTIES BELOW THE FALL LINE FOR CR, CD, PB, CU, ZN, FE, IN METRIC TONS PER YEAR

	Metal											
	Cr		Cd		Pb		Cu		Zn		Fe	
	POTW	I[3]	POTW	I	POTW	I	POTW	I	POTW	I	POTW	I
Arundel	7.3	0.7	0.2	0.2	2.4	3.1	3.4	2.4	9.9	1.3	21.8	9.4
Baltimore		59.5		24.1		17.5		88.5		59.1		225.1
Baltimore City	78.9	47.2	1.8	142.3	25.5	9.9	37.1	20.7	106.8	45.5	234.4	1729.6
Calvert		3.8		-		8.9		1.9		-		-
Caroline		0.0		0.0		0.0		0.0		0.0		0.0
Cecil	0.6	0.0	0.0	0.0	0.2	0.0	0.3	0.0	0.9	0.0		0.0
Charles		0.0		0.0		0.0		0.0		0.0		0.0
Dorchester	4.9	0.6	0.1	0.1	1.6	0.2	2.3	0.4	6.6	0.4	14.4	-
Harford	2.2	0.8	0.1	0.2	0.7	0.1	1.0	0.6	2.9	0.6	6.5	-
Howard		0.0		0.0		0.0		0.0		0.0		0.0
Kent	0.2	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.3	0.0	0.7	-
Prince Georges	12.8	1.8	0.3	0.0	4.1	4.1	6.0	0.9	17.3	0.0	37.9	0.1
Saint Mary's	0.0	0.0	0.9	0.0	0.3	0.0	0.4	0.0	1.2	0.0	2.6	-
Wicomico	1.8	0.1	0.0	0.0	0.6	0.0	0.9	0.0	2.5	0.1	5.4	0.1
Alexandria City	10.2	2.2	0.2	-	3.3	5.1	4.8	1.1	13.7	-	30.2	-
Chesterfield	2.5	9.6	0.1	0.1	0.8	17.5	1.2	4.0	3.3	2.4	7.3	0.3
Henrico		1.1		0.0		0.4		0.2		1.9		0.0
Hopewell City	14.6	7.6	0.3	0.3	4.7	2.9	6.9	1.4	19.7	13.9	43.3	-
Louisa		22.7		-		53.0		11.4		-		-
Newport News City	12.8	9.6	0.3	3.9	4.2	2.7	6.0	13.9	17.4	9.3	38.1	36.7
Norfolk City	16.7	2.0	0.4	0.9	5.4	0.4	7.8	3.0	22.6	2.0	49.5	5.9
Northampton		0.0		0.0		0.2		0.0		0.0		0.7
Portsmouth City	5.2	14.4	0.1	5.9	1.7	3.1	2.5	20.8	7.0	14.9	15.5	-
Prince William	13.4	2.0	0.3	-	4.3	4.7	6.3	1.0	18.1	-	39.7	-
Richmond City	22.6	1.1	0.5	0.6	7.3	2.5	10.6	12.8	30.5	6.7	67.1	-
Spotsylvania	0.2	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.3	0.0	0.6	-
Westmoreland	0.3	0.1	0.0	0.0	0.1	0.0	0.1	0.1	0.4	0.1	0.9	0.2
Williamsburg City	2.3	0.1	0.1	0.0	0.7	0.0	1.1	0.0	3.1	0.1	6.8	-
York		12.1		0.2		18.7		4.5		8.9		-
TOTAL	199.5	199.1	5.7	178.8	68.1	155.0	98.9	189.6	284.4	167.3	624.6	2008.2

[1] POTW loadings were calculated for facilities where flows were 0.5 MGD.

[2] Loadings computed from approximately 122 industrial dischargers.

[3] [illegible] Ind[illegible]y

waste water were assigned concentration values approximately 85 percent less than those industries in the same SIC code but discharging all process waste water.

Loadings from POTWs were computed by multiplying discharge flow rates (MGD), obtained from the EPA 1980 Needs Survey, by concentration values obtained from results of pilot-scale studies conducted by the EPA Municipal Environmental Research Laboratory (MERL) (Petrasek 1980). Discharge flow rates are compiled in the Needs Survey for use in Congressional allotment of construction grant funds to upgrade or expand existing POTWs.

Computation of loadings showed that discharge of metals is greatest in areas of high industrial activity and large population centers. With the exception of Fe, all of the metals listed in Table 1 have established criteria levels. These levels vary for each metal and for chronic versus acute toxicity. In localized areas, such as Baltimore Harbor and Elizabeth River, the quantities of metals discharged create situations with a strong potential for high aquatic toxicity. For example, in Baltimore Harbor, metals are discharged in moderate amounts; but because of low flushing rates (10 percent renewal rate) (Sinex and Helz, unpublished), these metals concentrate in Harbor waters. Although we have no data to demonstrate the severity of the problem in the water column, Sinex and Helz (unpublished) have shown from bottom sediment samples that the bulk of metals discharged in the Baltimore Harbor does, in fact, remain in the Harbor.

The distribution of metal loadings for POTWs and industries (Table 1) shows that discharges of Cd, Cr, Cu, Fe, and Zn from POTWs and industries in Baltimore County and Baltimore City far exceed those from other counties. Lead from POTWs in Baltimore City is higher than in other counties. Substantial inputs from POTWs are also noted for Cr, Fe, and Zn in Richmond City, Norfolk City, and Hopewell City. Lead is notably large in industrial discharge from Louisa County. Taken as a whole, industrial loadings are more than twice as large as treatment plant loadings.

Atmospheric Sources

Pollutants from the atmosphere can deposit directly as dryfall (dust) and as dissolved constitutents in precipitation (rain, snow, hail). Because we lacked data on the dryfall component of atmospheric deposition, no estimate of dryfall loading to the Bay is made in this section. However, Lazrus et al. (1970) and Davis and Galloway (1981) have done some work on dryfall atmospheric deposition of metals. Lazrus et al. (1970) showed that the deposition of metals from the atmosphere varies by a factor of three or less between North Carolina and Northern Virginia. Thus, the atmospheric deposition over the Bay is probably fairly uniform. Based on a residence time of 4.7 days for small aerosols (particles $<$ 1 u) and a predominantly easterly air flow, Davis and Galloway (1981) revealed that atmospheric contaminants may reach the Bay from industrialized areas of the midwest. Deposits in industrialized areas, such as Baltimore, consist of heavy particulates that settle out rapidly, as well as small aerosols that rain out in the vicinity of the city. Thus, the concentration of metals in dryfall around Baltimore decreases with distance from the city (Baltimore Regional Planning Council, unpublished data), but such industrial centers constitute only a small percentage of the Bay's area.

CBP funded projects investigated atmospheric inputs to the Bay from precipitation. Two sources were used to evaluate atmospheric loads -- storm data from the Maryland Geological Survey and marsh cores. Data from the Maryland Geological Survey's sampling of six storm events from April to September 1981 were used to compute atmospheric loadings listed in Table 2. Because the areal variability of the deposition rate from each storm could not be determined at this time, we developed loading estimates that assume uniform concentrations over the entire Bay. Omitted from these estimates are dryfall loading rates and deposition that occur on the land surface in the drainage basins, eventually reaching the Bay or tributaries from surface runoff. Because of these limitations, the values presented in Table 2 are conservative estimates of total atmospheric deposition.

TABLE 2. ATMOSPHERIC INPUT OF SELECTED METALS FROM WETFALL TO CHESAPEAKE BAY AND TRIBUTARIES

Metals	Volume[1]-Weighted Concentration (ug/g)	Main Bay[2] (metric tons/year)	Main Bay and Tributaries[3] (metric tons/year)
Cd	0.23	2	3
Cu	2.20	16	28
Fe	6.85	50	87
Mn	1.77	13	22
Ni	1.95	14	25
Pb	2.66	19	34
Zn	65.20	467	825

1 Based on sampling from six storm events. Data from Maryland Geological Survey (Conkwright et al. 1982).

2 Surface area of Main Bay = 6,500 km^2.

3 Surface area of Bay & Tributaries = 11,500 km^2.

Loadings computed using average annual precipitation of 1.1 meters.

Results from these studies show that quantities of metals entering Bay waters from atmospheric deposition are significant. The concentrations of metals in the atmosphere are proportional to the total mass of the metals released into the atmosphere from fossil fuel combustion, manufacturing processes, and many other anthropogenic and natural processes. The input of Zn, as shown in Table 2, is high because of its high emissions from fossil fuel combustion and other manufacturing processes like plating and cement production (Forstner and Wittman 1979). The total load of Zn from the atmosphere is at least double the amount from point source (Table 1). This suggests that some of the remote areas of the Bay, where anthropogenic contamination is assumed to be negligible are, in fact, areas receiving heavy inputs of metals, especially Zn. Other areas receiving high amounts of metals must also absorb elevated levels from the atmosphere, thereby worsening the problem.

Marsh deposits can record the atmospheric flux of trace metals deposited over time, thus providing another estimate of atmospheric input. The surface of the high marsh, Spartina patens, which is exposed to the atmosphere 95 percent of the time, retains most all atmospheric inputs. Although marsh cores from the Bay are scarce, McCaffrey and Thornson (1980) can estimate the atmospheric flux to the Bay from another core from Farm River Marsh, in Long Island Sound, Connecticut. In the Farm River Marsh core, all of the metals are assumed to have been deposited from the atmosphere. The concentration of these metals (Cu, Pb, and Zn) from the marsh samples was divided by the concentration of ^{210}Pb present in the sample. All of the ^{210}Pb in the marsh samples is assumed to have been deposited from the atmosphere (Helz et al. 1981). The metal to ^{210}Pb ratio from the marsh core is then assumed to be similar for the Helz cores, because the deposition rate between Long Island and the Bay is probably nearly the same. Therefore, by knowing ^{210}Pb concentrations in the Chesapeake cores, and applying the ratio from the marsh core, an estimate of the atmospheric contribution of these metals can be made.

In the northern Bay, core 4 (Table 3) shows that approximately 10 percent of the Cu (Cu/^{210}Pb Cu) and five percent of Zn (Zn/^{210}Pb Zn) is supplied from the atmosphere. However, in other cores from the central Bay (not shown) about 25 percent of the Cu and 13 percent of the Zn is of atmospheric origin. Consequently, the atmosphere becomes an important source in zones distant from sources of water pollution. When atmospheric and water pollution occur concurrently, the trend of "excess" metal over the background for the marsh, representing the atmospheric flux, is similar to those of Bay sediments as shown in Figure 1. Thus, atmospheric sources contribute to the increase of excess metals with time.

The trend, observed in Figure 1 for Zn in core 4 and in the Farm River marsh core, shows that Zn appears to be decreasing from a maximum value occurring around 1930 to 1940. The recent decrease could be due to an alteration in manufacturing processes or shifts in fossil fuel consumption (burning more oil instead of coal), thereby releasing less Zn to the atmosphere.

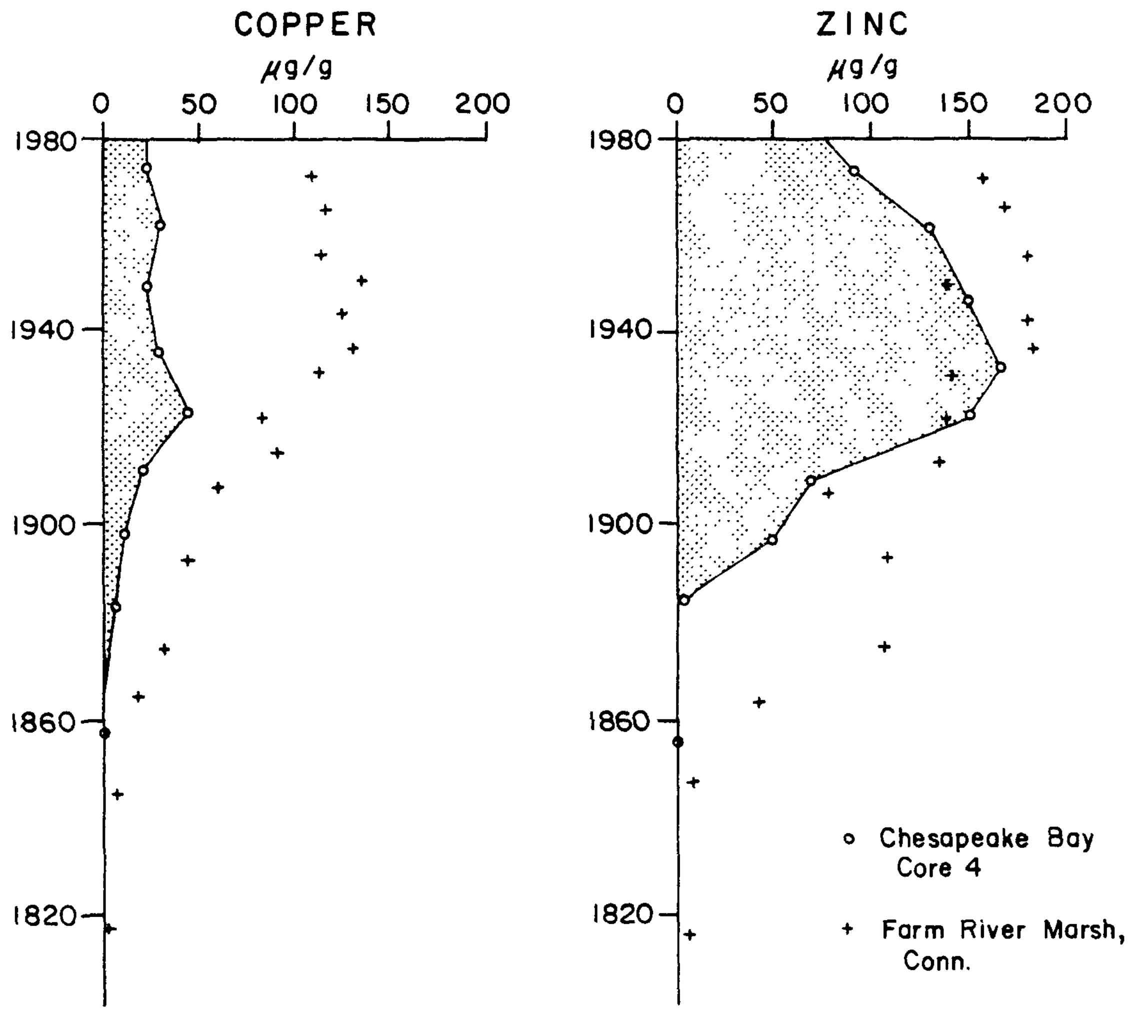

Figure 1. Graph of age versus metal content of Cu and Zn showing historical increase of "excess" metal concentration in Chesapeake Bay core 4 (Helz et al. 1981), by comparison to Farm River Marsh concentrations (McCaffrey and Thornson 1980).

TABLE 3. TOTAL EXCESS METALS IN CHESAPEAKE BAY CORES CONTRASTED WITH FARM RIVER MARSH

Core	^{210}Pb Standing Crop (dpm cm^{-2})[2]	Cu (ug cm^{-2})	Zn (ug cm^{-2})	Cu/^{210}Pb	Zn/^{210}Pb
4	7	793	3000	113	428
18	10.5	630	1500	60	142
60	10.0	644	1500	64	150
Farm River Marsh[1]				13	19

[1]From Benniger (1978).

[2]dpm cm^{-2} - decays per minute per square centimeter.

Urban Runoff

As previously discussed, the deposition of airborne pollutants to the Bay's surface may be an important transport mechanism. Another pathway by which atmospheric pollutants enter the Bay is urban runoff. Some rainwater (and dust) deposited in urban areas eventually reaches the Bay. This transport is facilitated by the high percentage of paved surface area in urban regions. Flowing over the roads and other impervious and pervious surfaces, runoff accumulates certain metals in dissolved and particulate phases, notably Pb from the combustion of leaded gasoline, Zn from the abrasion of tires, and Cu and Cr from automobile brake shoes.

Although urban runoff is usually considered a nonpoint source, on a Bay-wide scale, loadings from the three major cities in the Bay area are of sufficient magnitude to represent major localized point sources. Table 4 shows annual loadings of metals from Baltimore, Hampton Roads, and Washington, DC runoff. Loadings were computed from data supplied by Hartigan (October 21, 1981, memorandum). Concentrations of metals in runoff were derived by averaging results from runoff data collected during the Metropolitan Washington NURP study and an early 208 monitoring study in the Occoquan River and Four Mile Run basins of Northern Virginia. Surface runoff volumes were obtained by assuming that soils are sandy loam and by computing values for the various land use categories based on 1967 hourly rainfall record (rain gage at Washington National Airport).

The loading values listed in Table 4 show that urban runoff is a significant source of metals. Metals exhibiting the highest loadings are Fe, Pb, and Zn. Iron is not considered a toxic metal; loading values are included only for comparison. The high Pb and Zn values reflect local sources of these metals such as automobile exhaust, incinerators, refuse, and other urban activities that generate dust, gases, and other noxious by-products. Since rain is the major component of runoff, the concentrations of metals in rain and other forms of precipitation will also cause high metal loadings in urban runoff.

TABLE 4 URBAN RUNOFF LOADINGS FROM THREE MAJOR METROPOLITAN AREAS OF CHESAPEAKE BAY (AREA VALUES IN METRIC TONS/YEAR)

Metro Area	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Baltimore	5	3	3	291	5	6	35	19
Norfolk/ Newport News/ Hampton	1	4	2	213	3	4	26	15
Washington, DC	1	4	4	473	7	10	50	29
Total	7	11	9	977	15	20	111	63

River Sources

An estimate of annual loadings of selected metals at the fall line of three rivers, the Susquehanna, Potomac, and James, was derived from samples collected approximately bi-weekly to monthly by the U.S. Geological Survey between October 1978 and April 1981 (Lang and Grason, unpublished). Loading values were computed, using one of the methods described below.

Prediction Model--

Various mathematical models were used to fit a relationship between concentration (C) and flow (Q) or loading rate (LR) and flow. The various models used were as follows: C versus Q, ln(C) versus Q, C versus ln(Q), ln(C) versus ln(Q), C/Q versus 1/Q, ln(C/Q) versus 1/Q, C/Q versus ln(1/Q), ln(C/Q) versus 1/Q, LR versus Q, ln(LR) versus Q, LR versus ln(Q), and ln(LR) versus ln(Q). A concentration and/or loading rate was then computed for each day, using the best model and observed daily flow rates. These daily loadings were then summed for the total annual loading.

Sum of Averages--

To obtain loadings using this method, a flow weighted, mean daily concentration was first calculated as follows:

$$C_{mean} = \frac{[(C_{inst})(Q_{inst})]}{Q_{inst}}$$

This value was then multiplied by mean daily flow to obtain a daily loading. Daily loadings for each month were then averaged to give an averge daily loading for that month. These averages were then multiplied by the number of days in the month to give a monthly loading.

The monthly loadings were averaged to give an average monthly loading. Where no samples were taken in a month, the monthly average was used for these months, and the monthly loadings were summed to give a yearly loading.

Mean or Median Value from Sampling Data Applied to Long-Term Mean Annual Flow--

This method involved using the mean or median value of the various parameters as reported by the USGS (Lang and Grason 1980) and the long-term mean annual flow to compute the loadings.

The loadings and the computation method used for each metal are listed in Table 5(a). The discharge flows for these years and the long-term mean annual flows are listed in Table 5(b). The flow rates for 1979 were significantly above normal for all three rivers and, for 1980, were somewhat less than normal except for the James which was approximately ten percent higher than the long-term mean annual flow. Therefore, the computed average loading for these years is probably slightly higher than normal.

TABLE 5(a). ESTIMATED AVERAGE ANNUAL LOADINGS FOR VARIOUS METALS FROM THE MAJOR TRIBUTARIES OF CHESAPEAKE BAY FOR 1979-1980 PERIOD* (VALUES IN METRIC TONS/YEAR) (FROM LANG AND GRASON 1980)

Parameter	Susquehanna @ Conowingo Dam		Potomac @ Chain Bridge		James @ Cartersville, VA		Totals
Al-D	6,509	(2)	1,724	(2)	2,631	(2)	10,864
Al-S	156,061	(2)	36,061	(2)	30,890	(2)	223,012
Al-T	161,618	(2)	37,626	(2)	33,884	(2)	233,128
As-T	82	(2)	13	(2)	20	(1)	115
Cd-T	65	(3)	4	(2)	6	(3)	75
Co-T	59	(2)	39	(1)	48	(2)	146
Cr-T	383	(3)	105	(1)	63	(3)	551
Cu-T	390	(2)	86	(1)	41	(1)	517
Fe-D	1,844	(1)	839	(2)	567	(1)	3,250
Fe-S	192,422	(2)	76,227	(2)	27,783	(1)	296,432
Hg-T	23	(2)	-		6	(2)	29
Mg-D	232	(2)	61	(1)	31	(2)	324
Mn-D	7,552	(2)	86	(3)	104	(2)	7,742
Mn-S	7,326	(2)	1,929	(3)	2,277	(2)	11,532
Mn-T	14,469	(2)	1,933	(3)	2,327	(2)	19,229
Ni-T	229	(1)	109	(1)	64	(1)	402
Pb-T	174	(3)	102	(3)	31	(3)	307
Zn-T	837	(1)	322	(1)	285	(1)	1,444

*Values listed represent the mean of 1979 and 1980 calender year loadings.

(1) Computed using a model
(2) Computed using sum of averages method
(3) Computed using the reported mean or median value applied against the long term mean annual flow

D - Dissolved
S - Suspended
T - Total

TABLE 5(b); ANNUAL AND LONG-TERM MEAN ANNUAL FLOWS FOR THE SUSQUEHANNA, POTOMAC, AND JAMES RIVERS[1]

	1979 Calendar Year ($ft^3\ sec^{-1}$)	1980 Calendar Year ($ft^3\ sec^{-1}$)	Long Term Average ($ft^3\ sec^{-1}$)
Susquehanna	52,200 (+34%)[2]	28,400 (-27%)	38,900
Potomac	20,400 (+79%)	11,000 (- 3%)	11,400
James	12,000 (+70%)	7,790 (+10%)	7,050

[1]Data from U.S. Geological Survey, unpublished

[2]Values in parenthesis represent the percent difference from the long-term mean annual flow.

Table 5(a) lists loading values for 13 metals of which several -- Al, Fe, Mg, Mn -- are not considered toxic. Some of these metals, such as Al and Fe, are contributed primarily from natural erosion processes and cannot indicate pollution. All of the metals in this list occur in crustal material and, therefore, are naturally found in rivers. This makes it difficult to determine the natural from the anthropogenic contributions, a subject more fully discussed in Section 4. It is important to mention, however, that even though some metals are contained in naturally-occurring soil and crustal material, the rate of this sediment entering the river may be dramatically enhanced by farming and other rural and urban activities.

Of importance to note in Table 5(a) are the high loadings for Cr, Cu, and Zn. These values reflect contributions from point and nonpoint sources, erosion, and other sources. Zinc values are particularly high and may be the direct result of the observed high concentrations of Zn in the precipiation that falls on these drainage basins. Of the three rivers, the Susquehanna produces the highest loadings, primarily because of the higher flows in this river.

Concentrations of total metal content in the rivers vary with total suspended material and with river flow. As shown in Figure 2, the concentration of suspended Fe at high inflow is more than 20 times the concentration at low inflow, and Mn is more than 15 times the concentration at low inflow. Some metals, like Mn, also exhibit seasonal changes in partitioning between dissolved particulate concentrations (Figure 2). Particulate Mn is more dominant than dissolved Mn in spring, summer, and fall--a trend associated with influx of decaying organic matter in winter (Carpenter 1975). Such changes in partitioning and the varying metal concentrations with sediment loads make determination of loading estimates difficult.

A comparison of the 1980 loadings on the Susquehanna River with values computed by Carpenter in 1965-1966 is presented in Table 5(c). These data show that loading values for Cd, Cu, Fe, and Zn are very similar. Manganese shows a slight increase, but Co and Ni show moderate to high decreases. The most notable change is the Cr loading that was

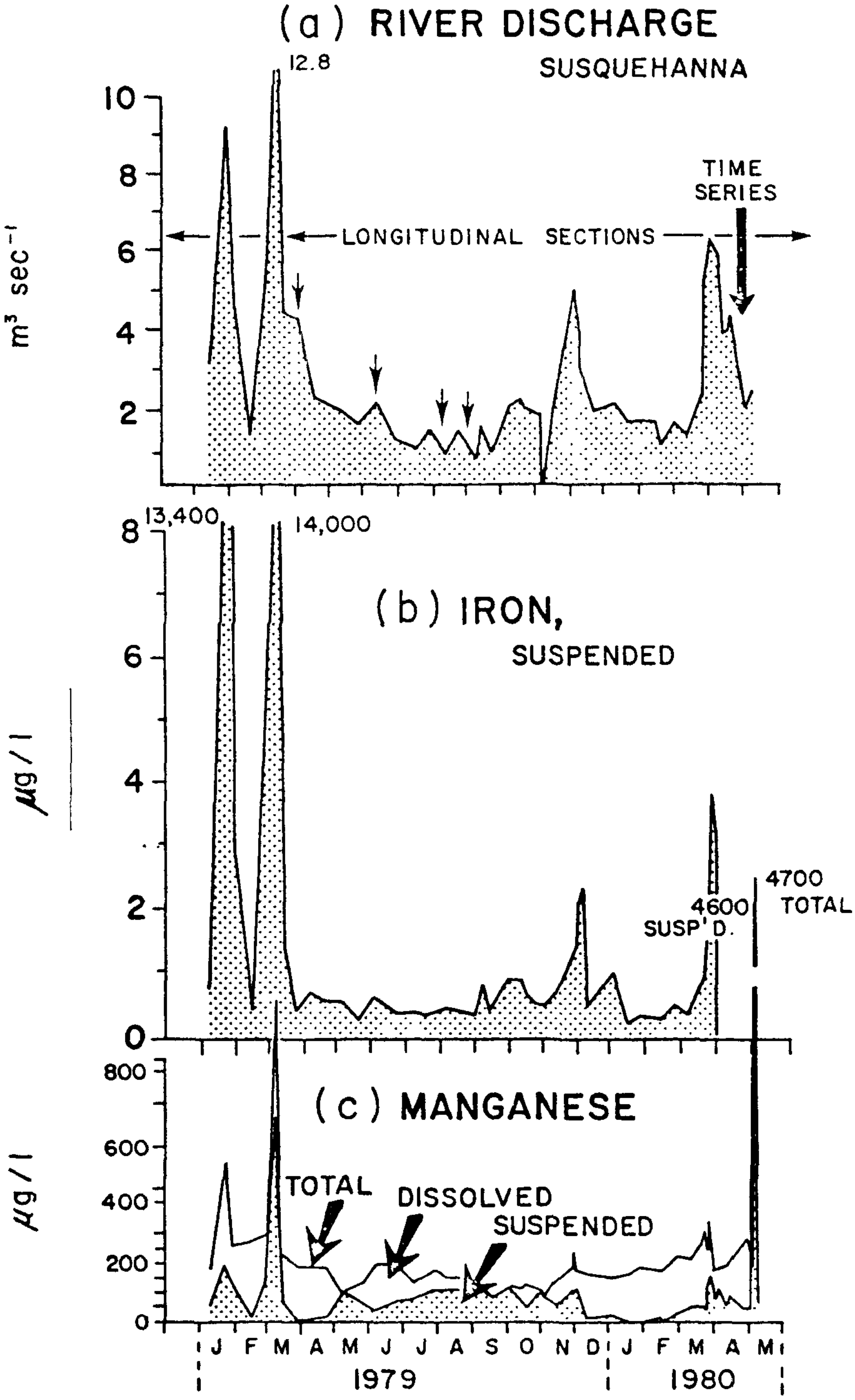

Figure 2. Temporal variations of: (a) Susquehanna River discharge at Conowingo Dam, and corresponding (b) Fe, and (c) Mn concentrations, dissolved, suspended, and total. Data from Lang and Grason (1980) b'ised)n instantaneous measurements and samples at peak inflows.

approximately 300 percent higher in the 1980 estimates than in the 1965-1966 estimates.

Comparison of the loadings from the three rivers in Table 5(a) indicates that the Susquehanna contributes a greater proportion of metals than the Potomac or James. To provide an estimate of the relative yield (or load per unit area) from these river basins, loading rate factors were computed by dividing the loadings listed in Table 4 by the area of the drainage basin above the fall line for each river system. These values are listed in Table 5(d). Generally, the Susquehanna appears to be no more enriched than the Potomac or James. Although certain metals are more enriched in one river system compared to the other two, the differences are significant for only several metals and may be largely explained by errors in sampling or loading computation.

TABLE 5(c). COMPARISON OF COMPUTED LOADINGS FOR THE SUSQUEHANNA RIVER WITH THOSE OF CARPENTER[1] (LOADINGS IN METRIC TONS/YEAR)

Metal	1980 Computed Loadings with Flow = 28,400 $ft^3 sec^{-1}$	Annual Loadings Reported by Carpenter[2] Flow = 28,012 $ft^3 sec^{-1}$	Percent Difference From Carpenter (%)
Cd	2	2	0
Co	20	90	-78
Cr	220	50	+340
Cu	106	100	+6
Fe	36,500	40,000	-9
Mn	6,100	5,000	+22
Ni	150	200	-25
Zn	570	600	-5

[1]Carpenter, J. H., W. L. Bradford, and V. Grant (1975).
[2]Sampled approximately one mile downstream from Conowingo dam every week for the period of April 1965 through August 1966.

Although rivers are a major source of metals, it is not known what proportion of these loadings enter the Bay. Monitoring on the Susquehanna generated loading values for the river just prior to discharge into the Bay, but the James, Potomac, and many other tributaries discharge into fresh water, tidal, and brackish-water reaches of substantial length.

Prior studies of eight Bay tributaries indicate that the bulk of suspended sediment is trapped within the tributaries--for example, in the Back River (Helz et al. 1975), the Chester (Palmer 1974), the Choptank (Yarbro 1981), the Patuxent (Keefe et al. 1976), the Rappahannock (Nichols 1977), and the James (Nichols 1972, O'Connor 1981). Entrapment of sediment is recorded either by direct measurements of suspended sediment transport, or by historical shoaling rates with an evaluation of these rates in relation to inputs of suspended sediment from different sources.

TABLE 5(d). METAL LOADING RATE FACTORS FOR THE SUSQUEHANNA, POTOMAC, AND JAMES RIVER DRAINAGE BASINS* (VALUES IN METRIC TONS/KM^2)

Metal	Susquehanna	Potomac	James
Al-D	240	149	420
Al-S	5,759	3,119	4,937
Al-T	5,964	5,255	5,415
As-T	3	1	3
Cd-T	2	1	1
Co-T	2	3	8
Cr-T	14	9	10
Cu-T	14	7	7
Fe-D	68	73	91
Fe-S	7,110	6,594	4,440
Hg-T	1		1
Mg-D	9	5	5
Mn-D	279	7	17
Mn-S	270	167	364
Mn-T	534	167	372
Ni-T	8	9	10
Pb-T	6	9	5
Zn-T	31	28	46
Basin Area (km^2)	27,100	11,560	6,257

* Values computed by dividing loadings listed in Table 5(a) by the area of the drainage basin above the USGS monitoring station.

The ability of these rivers to trap river-borne sediment was determined by calculating a capacity inflow ratio, using intertidal volume for capacity, and potential inflow (drainage area times annual precipitation) for inflow assuming all precipitation is runoff. As indicated in Table 6, tributary estuaries such as the Rappahannock and Choptank act as very efficient sediment traps. Therefore, if most of the sediment is trapped in the estuarine portion of these rivers, then the bulk of river-borne toxicants that are adsorbed to the sediment are also likely trapped. Despite the high efficiency of these rivers to trap sediment, some sediment will escape, especially during storms. At such times, these rivers and other similar areas should be monitored for exceptionally high levels of toxicants.

TABLE 6. DATA FOR CAPACITY/INFLOW RATIOS AND PERCENTAGE OF SUSPENDED SEDIMENT TRAPPED

System	Capacity/Inflow	Sed. Trapped	Source
Rappahannock	0.7	90%	Nichols (1977)
Choptank	2.0	92%	Yarbro (1981)
Susquehanna - Northern Chesapeake Bay	0.04	75%	Biggs (1970)

A summary of total metal influx to Chesapeake Bay and its tributaries from different natural and anthropogenic sources is presented in Table 7. The estimates are products of two quantities, average metal concentration and rate of discharge. Accuracy of the data varies with the number of measurements per unit time, seasonal variations in constituent composition, and many other factors. This table shows that the sum of industrial and municipal wastewater loadings (point sources) represents a major contribution of metals to the Bay. Rivers are the only other source that exceed the point sources. However, the loadings from rivers actually represent a combination of the other sources that discharge into these rivers above the point where loadings were estimated. That is, the river-loading estimates contain some fraction of anthropogenic and natural contributions and become a pathway for these sources. From the results shown in Table 5(d), it appears that the relative proportions of the metal sources in these river systems are fairly uniform. However, because point sources do contribute to some part of the river loadings and are also one of the major sources for the Bay, this suggests that for most metals, point sources are probably the major source to the Bay, with loadings from urban runoff and shoreline erosion significant for some metals.

The upper Bay and the upper reaches of the Potomac and James estuaries are critical areas for fish spawning and other biological activities. From our studies of metal concentrations in the Bay (discussed in Section 3 and Section 4), we know that the Northern Bay does exhibit elevated metal concentrations. Therefore, the Susquehanna River represents a major source of metals, causing the Northern Bay to have elevated concentrations.

DISTRIBUTION AND CONCENTRATION OF DISSOLVED METALS

Some of the metals, entering the Bay from any one of the sources previously discussed, will dissolve in the estuarine water. In this form, metal data are available for Cd, Ce, Co, Cr, Cu, Fe, Mn, Mo, Ni, Pb, Sc, Th, U, and Zn in surface water and bottom water for one sampling cruise during June-July 1979 (Kingston et al. 1982).

Kingston's data show that a correlation exists between metal concentration and salinity for Cr, Mo, and U (Figure 3a, Figure 3b). Uranium and Mo concentrations increase linearly with increasing salinity and approach average seawater concentrations at the upper end of the

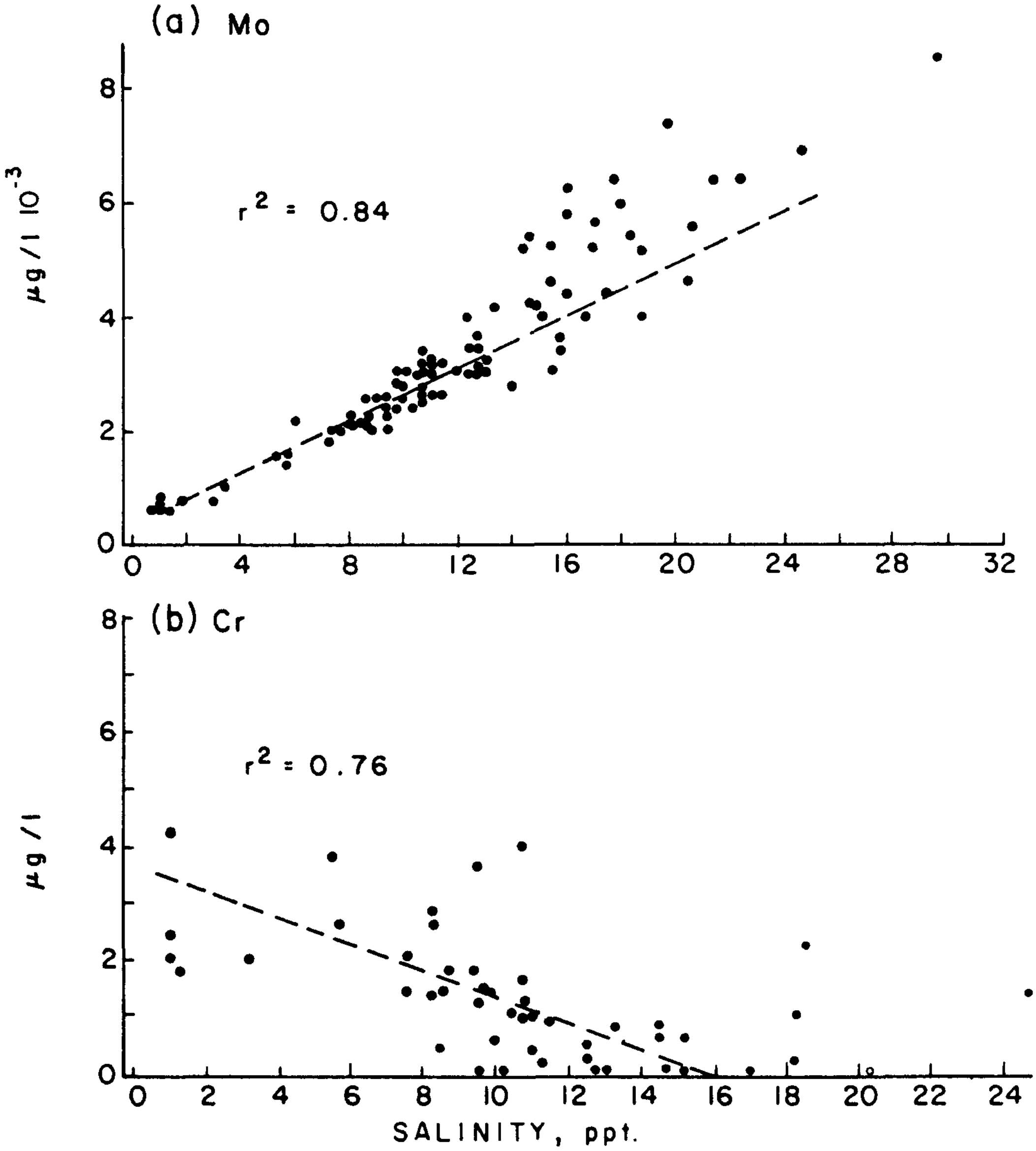

Figure 3. Plot of (a) dissolved Mo content versus salinity, and (b) dissolved Cr content versus salinity for samples from surface water along the Chesapeake Bay length, June-July, 1979. Data from Kingston (1982).

TABLE 7. LOADINGS OF METALS FROM THE MAJOR SOURCES AND PATHWAYS TO CHESAPEAKE BAY (VALUES IN METRIC TONS/YEAR)

Source	Cd	Cr	Cu	Fe	Pb	Zn
Industry	178 (66)[1]	200 (19)	190 (22)	2,006 (1)	155 (22)	167 (6)
Municipal Wastewater	6 (2)	200 (19)	99 (12)	625 (1)	68 (10)	284 (10)
Atmospheric	3 (1)	---	28 (3)	87 (1)	34 (5)	825 (29)
Urban Runoff	7 (2)	10 (1)	9 (1)	977 (1)	111 (16)	63 (2)
Rivers	75 (28)	551 (53)	517 (59)	199,682 (77)	307 (43)	1444 (50)
Shore Erosion	1 (1)	83 (8)	29 (3)	57,200 (22)	28 (4)	96 (3)

[1]Values in parenthesis represent percent of total loading

salinity range. This trend indicates that marine waters are the source of these metals, and that the concentration gradient is a result of dilution of marine water by river runoff. It also indicates that these metals are not significantly involved in chemical or biological processes in the Bay. By contrast, Cr concentrations decrease as salinity increases to a value approximating average seawater concentration at the upper end of the salinity range. This relationship indicates that river runoff is the major source of Cr, and that dilution by marine water controls dissolved Cr concentrations in the estuary. The scatter in the Cr data, however, is much greater (Figure 3b) than that for Mo, possibly indicating the influence of other processes in addition to dilution by marine waters.

All of the other dissolved metals investigated, Cd, Ce, Co, Cu, Ni, Pb, and Zn, are significantly affected by processes other than dilution. Therefore, plots of dissolved metal concentration versus salinity show little correlation. Cadmium, Cu, Ni, Sn, and Zn tend to decrease in concentration with increasing salinity, although there is much scatter in the data. Differences in metal concentrations in relation to salinity may arise from varying strength of sources (marine versus freshwater, or others), fluctuating chemical behavior (oxidizing versus reducing, salinity differences), hydrodynamic mixing patterns, and other factors.

Patterns of enrichment emerge from plots of the ratio of dissolved metal in surface water to dissolved metal in bottom water, versus salinity of the surface water (Figure 4). If surface waters are enriched (contain elevated concentrations) in a metal, the ratio is greater than one; if bottom waters are enriched, the ratio is less than one; if the surface and bottom concentrations are the same, the ratio is equal to one. For example, in Figure 4a, the dissolved-Cu-concentration-in-near-surface-water samples to dissolved-Cu-concentration-in-near-bottom-water ratios are mostly greater than one, and significantly greater than one in the 10 to 15

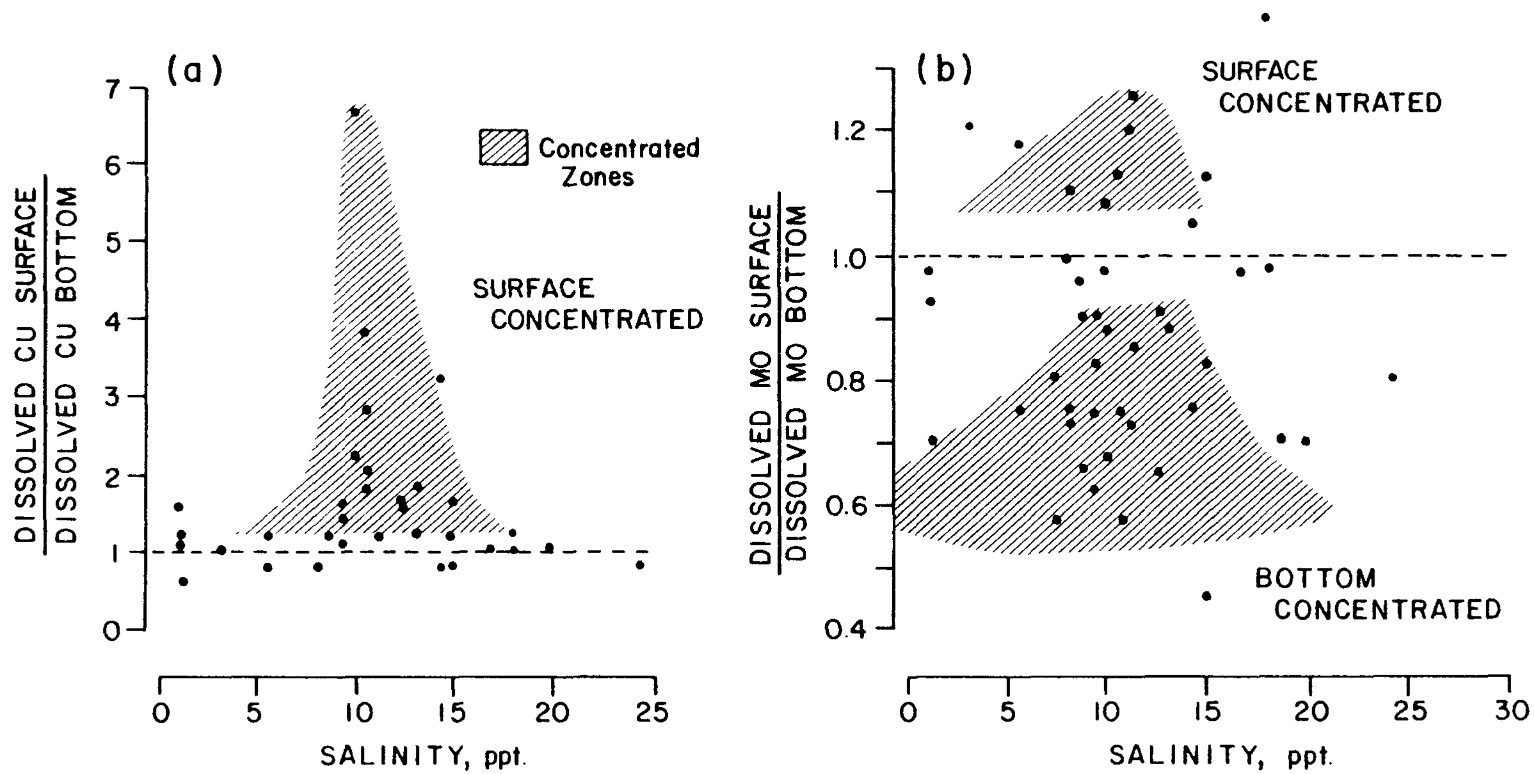

Figure 4.(a) Ratio of dissolved Cu concentration in surface water to dissolved Cu concentration in bottom water versus salinity, and (b) Ratio of dissolved Mo concentration in surface water to dissolved Mo concentration in bottom water versus salinity. Concentrated zones, shaded. Data from Kingston (1979).

ppt range of salinity values. This suggests that the mid-Bay (where salinities range from 10 to 15 ppt) has much higher Cu concentrations in the surface waters, relative to the bottom waters. Salinity indicates the relative position along the estuary where enrichment occurs. The term enrichment refers to the concentration of the metal in the surface water as a function of concentration in bottom water. This ratio does not indicate absolute concentration and cannot be used as an index of abnormal metal content.

Figure 5a compares the ratio of dissolved metal concentration in surface water to dissolved metal concentration in bottom water, with the ratio of surface water salinity to bottom-water salinity. On these plots, a salinity ratio of one indicates there is no halocline and, therefore, little or no stratification. The data displayed in Figure 5 can be divided into four quadrants. For example, in Figure 5(b), the ratios of dissolved-Mo-concentrations-in-near-surface-water samples to dissolved-Mo-concentrations-in-near-bottom-water samples, appear to fall primarily in the bottom, left-hand quadrant. This indicates that Bay waters display a tendency for Mo concentrations to be higher in salty, bottom waters than surface waters. If the ratio exceeds one, the surface water is more saline; if the ratio is less than one, the bottom water is more saline. As in the previous graphs, a metal ratio greater than one indicates surface enrichment, whereas a ratio less than one indicates bottom enrichment.

Plots like those of figure 5b show that Cu, Ni, and Zn are strongly enriched in surface waters, particularly under conditions of strong halocline development. Under the same conditions, Co, Cr, and Mo are strongly enriched in bottom waters. Similar data show that Cd is enriched in low-salinity surface water. Cobalt shows enrichment in surface waters of salinity up to approximately eight ppt, and in bottom waters over the salinity range from eight to 15 ppt. Chromium is enriched in surface waters up to 15 ppt salinity and in bottom waters from eight to 20 ppt. Copper, Ni, and Zn are strongly enriched in surface waters from five to 18 ppt. Uranium is enriched in bottom waters in the range seven to 15 ppt.

Table 8 summarizes univariate statistics for near-bottom and near-surface dissolved metal concentrations throughout Chesapeake Bay as sampled and analyzed by Kingston et al. (1982). Because of the high precision and accuracy used in these analyses,.the information in Table 8 represents data generated for the first time for several metals in Bay waters. These numbers, then, are "benchwork" values from which to compare future numbers, and can indicate potential increases or decreases in contaminated areas.

The NBS investigations (Kingston et al. 1982) analyzed particulate as well as the dissolved concentrations in the sample. This information provides better understanding of how the various metals partition between dissolved and adsorbed phases. Dissolved metal concentrations are very important, because this phase is completely biologically available. Therefore, some of the maximum values shown in Table 8 may be hazardous to aquatic life in Bay waters where these high concentrations are found.

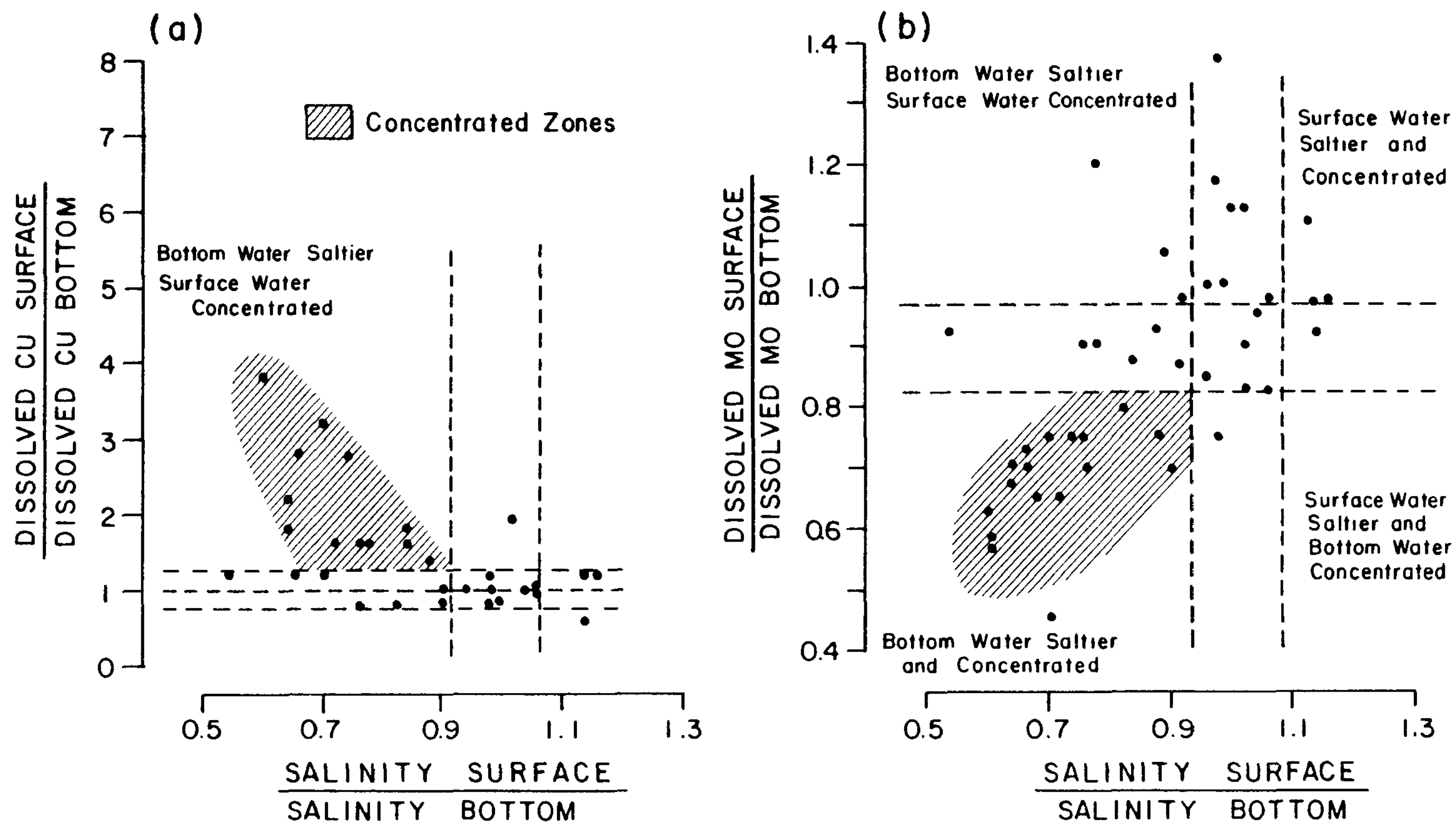

Figure 5. Plot of the ratio of (a) dissolved Cu concentration and (b) dissolved Mo concentration in surface water to bottom water versus the ratio of surface salinity to bottom salinity. Concentrated zones, shaded. Data from Kingston (1979).

TABLE 8. SUMMARY OF MEAN AND MEDIAN METAL CONCENTRATIONS AND RANGE OF BAY-WIDE VALUES (UG/L) (DATA FROM KINGSTON ET AL. 1982) CRUISE OF JUNE-JULY 1979.

		Dissolved		
	N*	Mean	Median	Range
Cd	45	0.05	0.04	0.007-0.101
Co	102	0.07	0.05	0.01-0.56
Cr	102	0.17	0.11	0-1.68
Cu	79	0.66	0.48	0.15-2.25
Fe	102	3.12	1.63	0.09-71.67
Mn	102	13.88	3.34	0-388
Mo	102	3.26	2.93	0.61-8.68
Ni	102	1.21	1.15	0.5-2.59
Pb	102	0.11	0.05	0-1.59
Sc	102	0.0006	0.0005	0.0002-0.002
Sn	9	0.86	0.86	0.31-1.61
Th	39	0.001	0.001	-
U	102	0.93	0.88	0.13-2.57
Zn	102	1.19	0.42	0-11.11

*N is number of samples treated.

DISTRIBUTION AND CONCENTRATIONS OF METALS IN SUSPENDED MATERIAL

Chesapeake Bay Program research has shown the distribution of metals in suspended material displays marked longitudinal and vertical gradients. Although concentrations were highly variable between samples and surveys, the mean metal content per gram of material exhibits distinct trends (Nichols et al. 1981). Content of the metals, As, Cd, Cu, Pb, Hg, Ni, Sn, and Zn, reached a maximum in near-surface suspended material of the central Bay, shown in Figures 6a, 6b, and 6c. Because this part of the Bay is an area of high biological activity, elevated levels of these metals could threaten biota there. The concentrations for these metals were higher than farther landward near major sources in the Susquehanna River mouth and Baltimore Harbor zone. Particularly high maxima or "hot spots" were observed for Cu and Cd (Figure 7 and Figure 8). The mean concentrations for Cu and Cd were five to 10 times greater than the Susquehanna River mouth. Secondary maxima occurred in the main Bay off Baltimore Harbor for surface concentrations of Cd, Mn, Ni, Pb, Sn, and Zn (Figure 7 and Figure 8). High levels of metals at these "hot spots" indicate areas of possible toxic impacts.

Metal concentrations were higher in surface and mid-depth suspended material than near the bottom, a trend resulting in stratified distributions. For example, Cu, Ni, Sn, and Zn concentrations were higher in surface than in near-bottom water in the same zone by a factor of two or more. Again, these results indicate where unnatural levels of metals can occur, with a consequence of increased risk of toxicity.

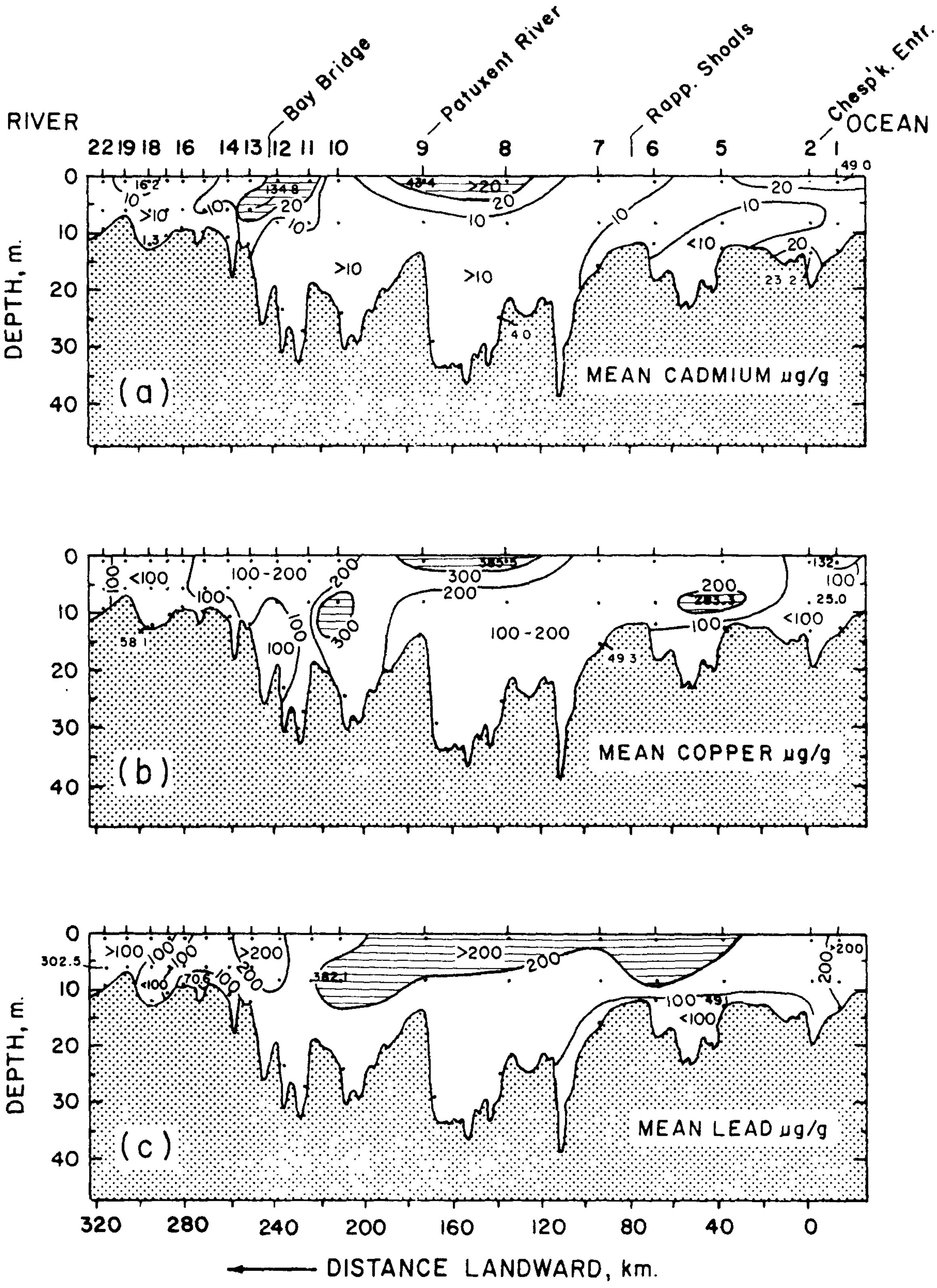

Figure 6. Longitudinal-depth distributions of mean metal concentration per gram of suspended material, along the axis of Chesapeake Bay, for (a) Cd, (b) Cu, and (c) Pb. Relatively high zones, shaded. Data from Nichols et al. (1981).

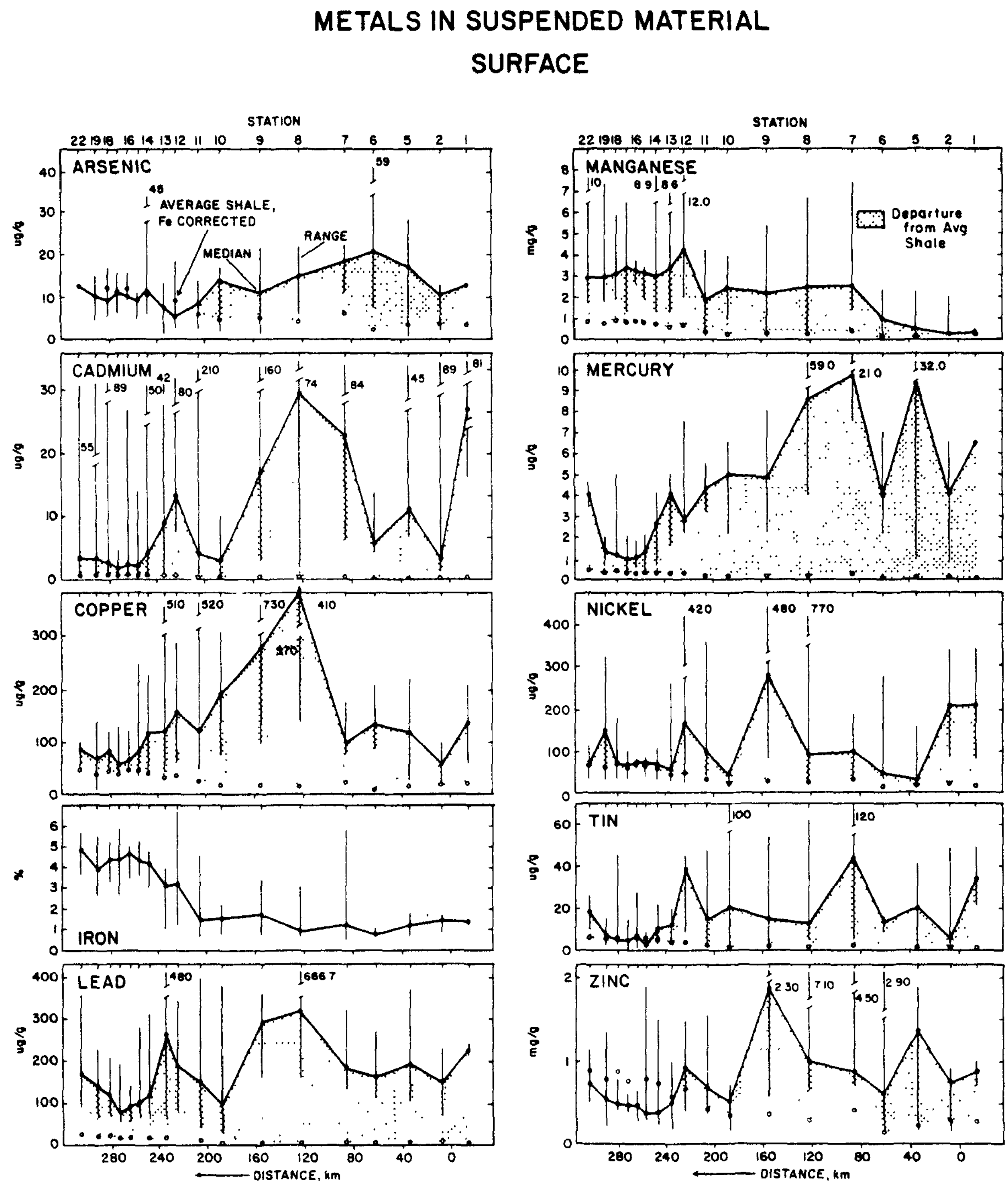

Figure 7. Distribution of metal content in surface suspended material with distance along the Bay axis. Median values and range of concentrations from all available observations of Nichols et al. (1981). Shaded zone indicates magnitude of departure between median values and mean values for Fe-corrected average shale, open circles.

METALS IN SUSPENDED MATERIAL
NEAR-BOTTOM

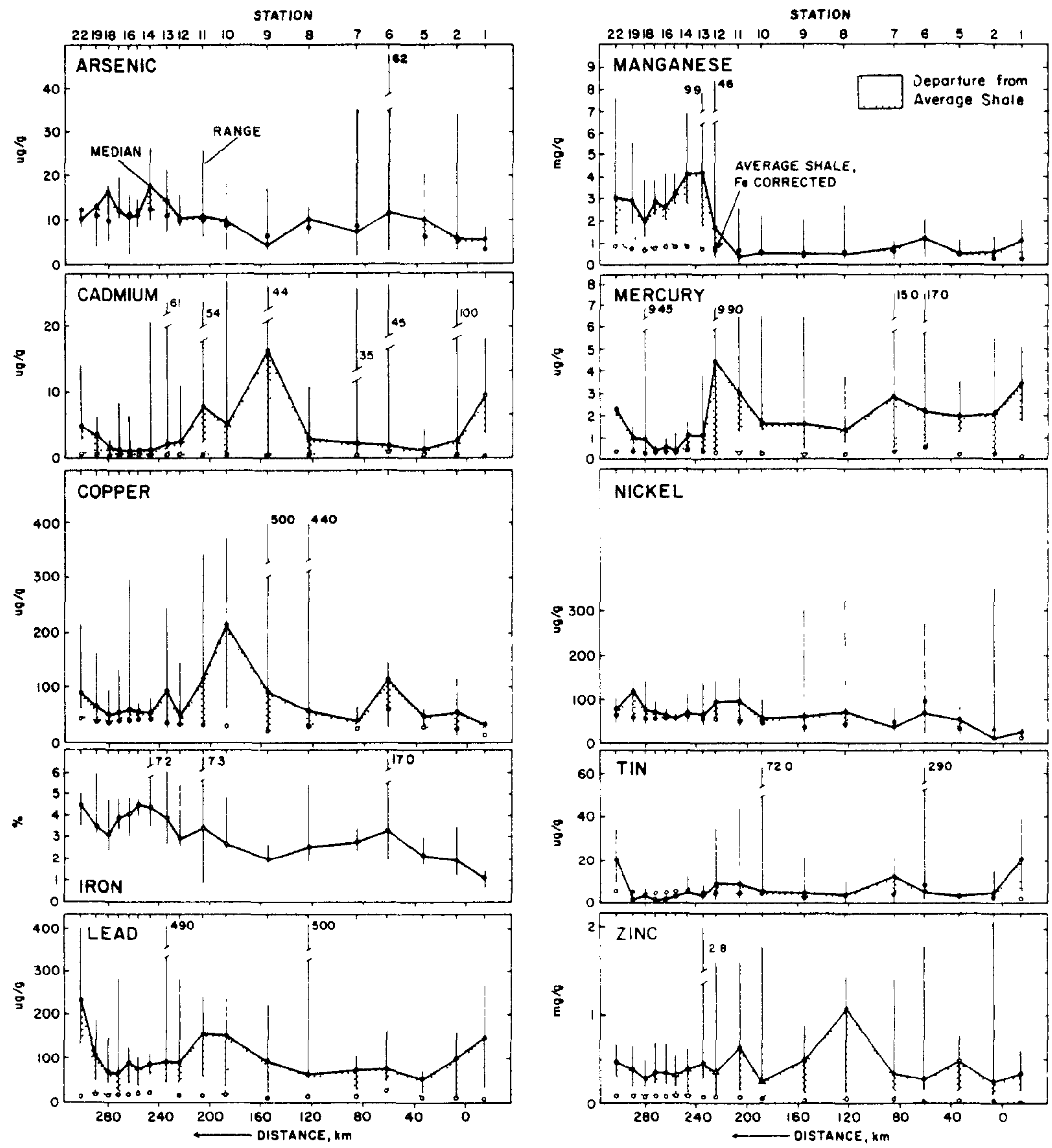

Figure 8. Distribution of metal content in near-bottom suspended material with distance along the Bay axis. Median values and range of concentrations from all available observations of Nichols et al. (1981). Shaded zone indicates magnitude of departure between median values and mean values for Fe-corrected average shale, open circles.

Concentrations of metals in suspended material changed with season. Seasonal changes were marked by a 10-fold increase in surface Cu concentrations between March to April and May to August (Nichols et al. 1981). Zinc was higher in March to April than at other times, whereas Pb was highest during June. Table 9(a) summarizes the mean metal concentrations and range of values at all sample depths throughout the Bay (Nichols et al. 1981). Table 9(b), from Kingston et al. (1982), supports these values.

TABLE 9(a). SUMMARY OF MEAN METAL CONCENTRATIONS AND RANGE OF BAY-WIDE VALUES, PER GRAM OF SUSPENDED MATERIAL, LEFT; AND WEIGHT PER VOLUME OF SUSPENDED MATERIAL, RIGHT (DATA FROM NICHOLS ET AL. 1981) FOR MORE THAN 550 SAMPLES AND 8 CRUISES ALONG THE BAY-LENGTH BETWEEN MONTHS OF MARCH AND SEPTEMBER 1979 AND 1980

Metal	Mean	Range	Metal	Mean	Range
As ug/g	13.00	0.55-100.00	As ug/L	0.32	0.006-5.00
Cd ug/g	14.16	0.12-790.00	Cd ug/L	0.14	0.003-3.80
Cu ug/g	127.96	9.90-570.00	Cu ug/L	1.84	0.068-17.00
Fe ug/g	3.11×10^7	$0.29-17 \times 10^7$	Fe mg/L	88×10^5	$1.0-1200 \times 10^5$
Hg ug/g	3.89	0.5-59.00	Hg ug/L	0.035	0.01-0.47
Mn ug/g	2880	80-46,000	Mn ug/L	65.13	0.48-1000.00
Ni ug/g	95.80	4.80-770.00	Ni ug/L	2.00	0.03-34.00
Pb ug/g	160.30	21.00-730.00	Pb ug/L	2.27	0.10-15.00
Sn ug/g	17.97	0.25-290.00	Sn ug/L	0.20	0.01-4.80
Zn mg/g	750	100-7100	Zn ug/L	11.02	0.55-94.00

TABLE 9(b). MEAN, MEDIAN, AND RANGE OF METAL CONTENT FOR ONE CRUISE ALONG THE BAY-LENGTH (JUNE-JULY 1979) (DATA FROM KINGSTON 1982)

ug/l	N*	Mean	Median	Range
Cd	51	0.018	0.008	0.001-0.11
Co	102	0.24	0.06	0.17-2.37
Cr	102	0.75	0.23	0-5.31
Cu	102	0.65	0.36	0.1-4.69
Fe	102	342.45	131.50	14-2911
Mn	102	38.16	19.20	1.2-349
Mo	12	0.08	0.03	0.01-0.25
Ni	102	0.57	0.27	0.03-5
Pb	96	0.75	0.23	0.01-7.3
Sc	102	0.11	0.04	0.003-0.93
Sn	-	-	-	-
Th	100	0.10	0.04	0.002-0.68
U	86	0.029	0.012	0.002-0.192
Zn	90	2.15	0.73	0-15.2

*N is number of samples treated.

Table 9c. · COMPARISON OF MEAN PARTICULATE, DISSOLVED AND TOTAL METAL CONTENT IN SURFACE AND BOTTOM WATER FOR STATIONS THROUGHOUT CHESAPEAKE BAY, JUNE-JULY 1979; DATA FROM KINGSTON (1982).

Metal	Surface Water				Bottom Water			
	Particulate	Dissolved	Total	D/T x 100	Particulate	Dissolved	Total	D/T x 100
Co	0.207	0.046	0.255	18	0.270	0.083	0.354	10%
Cr	0.622	0.133	0.756	17	0.877	0.199	1.077	18
Fe	264.600	1.619	266.200	1	420.300	4.634	424.900	1
Mo	0.060	2.974	2.278	99	0.097	3.551	2.648	97
Sc	0.086	0.001	0.087	0.1	0.134	0.001	0.135	0.4
Th	0.077	0.001	0.150	1.0	0.113	0.001	0.150	0.9
U	0.024	0.830	0.846	98	0.035	1.020	0.986	97
Zn	1.882	1.756	3.636	48	2.410	0.623	3.095	20
Cd	0.016	0.048	0.072	66	0.018	0.044	0.065	68
Cu	0.563	0.771	1.437	53	0.733	0.560	1.376	41
Mn	33.750	2.645	36.410	7	42.56	25.120	67.680	37
Pb	0.554	0.111	0.662	17	0.927	0.117	1.045	11
Ni	0.478	1.283	1.761	73	0.672	1.155	1.828	63

Concentrations of metals and other chemical constituents can be expressed in several ways, including concentration expressed as weight of the specific metal per unit weight of suspended material, and per unit volume of water. The expression used depends on the substance (water or sediment) being analyzed. Discussion of metal concentrations thus far has been based on concentrations expressed on a weight per weight basis. However, when metal distributions reported as weight per volume are examined, the metal concentrations are directly proportional to the concentration of total suspended material. Therefore, mean metal concentrations of As, Fe, Mn, Ni, Pb, Sn, and Zn were highest in the zone of the turbidity maximum where suspended sediment concentrations are highest (Nichols et al. 1981). Likewise, near-bottom metal concentrations of most metals were usually higher than surface concentrations, resulting in stratified distributions.

In addition to seasonal variations, metal concentrations were highly variable on short-time scales. For example, concentrations of Cu and Pb per gram of suspended material from the turbidity maximum zone of the northern Bay, varied more than two-fold over a tidal cycle. By contrast, Fe, Mn, and Zn varied within relatively narrow limits. These fluctuations are associted with large fluctuations of suspended material entering the Bay, and moderate fluctuations of particle size and organic content as tidal currents resuspended sediment from the bed. Such short-term (tidal) changes added to long-term (seasonal) variations produce wide ranges in metal content. These variations must be taken into account for planning metal samplings for monitoring and meaningful interpretation of data.

Despite the wide spatial and temporal variations of metal concentrations, many metals correlated statistically with each other, allowing the potential use of one or several as predictors. For example, from the VIMS cruise series (Nichols et al. 1981), Fe-Mn, Cu-Zn, and Ni-Zn, Ni-Fe, and Zn-Fe had $r > 0.80$. Many metals from the NBS cruise (Kingston et al. 1982) also correlated with each other: Co, Cr, Fe, Sc, Th, Zn, Cu, Mn, Pb, and Ni with $r > 0.90$. These associations reflect the affinity of metals for suspended material through adsorption or uptake, and show that many metals display similar behavior. Metals like Mo, U, and Cd did not correlate, however, because they tend to stay in solution. The similar behavior of these metals can be used to predict the occurrence of unknown concentrations when only one metal is known. Moreover, Fe was found useful as a surrogate element since it is naturally abundant. Iron also varies within relatively narrow limits throughout the Bay. Its use for normalizing enrichment factors is demonstrated in a separate section.

A comparison of the mean metal content of the dissolved fraction and the corresponding particulate fraction per volume of suspended material [Table 9(c)] reveals several significant trends. The ratio of dissolved to total metal content provides an index to the mobility of the metal, and thus its availability to biota. For example, Mo and U are dominately in dissolved form in both surface and bottom water, whereas Co, Fe, Mn, Pb, Sc, and Th are dominately in the particulate form. Note that Zn displays much higher percentages in surface water than in bottom water. Therefore, samples of surface water alone are not indicative of the dissolved Zn content in bottom water. By contrast, Mn (both particulate and dissolved) is much higher in bottom water than in surface water in summer. This trend probably reflects mobilization and release of Mn from central Bay sediments during summer anoxia. The index provides an indication of which metals organisms are exposed to in summer. Since dissolved metals generally have a shorter residence time in the Bay than particulate metals, the index further predicts that metals like Mo and U will likely escape the Bay whereas Co, Cr, Fe, Mn, and Sc are most likely retained in the estuary. The fate of other metals probably varies with natural biochemical and sedimentological processes native to the Bay.

DISTRIBUTION AND CONCENTRATION OF METALS IN BOTTOM SEDIMENTS

During the Bay Program, surface sediments were analyzed for As, Cd, Co, Cr, Fe, Hg, Mn, Ni, Pb, and Zn by Helz et al. (1981) and Nichols et al. (1981). All of these metals are more concentrated in the fine fraction ($<$ 63 um) of bottom sediments than in bulk samples and show that the Susquehanna River is a major source of most metals. Figure 9 illustrates the Cu distribution in bulk and in $<$ 63 um surface sediments of the Bay. Copper in the fine fraction decreases seaward from the Susquehanna mouth, indicating a river source. Copper also decreases eastward across the Bay, suggesting that seaward transport carries contaminated sediment seaward along the western shore. This pattern is consistent with the observed salinity pattern and net circulation of the Bay. An alternate cause of the western shore enrichment is the input from Baltimore Harbor and western shore tributaries.

Zinc distribution in bulk and fine sediments is illustrated in Figure 10. Zinc values in the silt-clay fraction are highest in the Bay off of Baltimore Harbor and decrease both landward and seaward, suggesting that Baltimore Harbor is a source of Zn to the Bay. Two mechanisms may be responsible for metal transport from the Harbor in particulate form: the estuarine circulation and dredge spoil disposal. More than 4.6 million cubic meters of dredged material have been disposed in the Bay off the Harbor (Schubel and Williams 1976). However, from the metal distributions, it is not possible to identify the magnitude of either of these mechanisms. Tidal action may be partially responsible. However, we do not feel it is a dominate factor and believe the data suggest riverine sources. The bulk Zn distribution displays relatively high concentrations in the lower Bay off the Rappahannock mouth. The high clay content of these sediments is probably responsible for the elevated concentrations observed in bulk samples.

Chromium and Pb exhibit surface sediment distribution patterns similar to Zn with maximum concentrations occurring in the fine fraction off Baltimore Harbor. The distribution of the metals Mn, Fe, Co, and Ni mirror Cu distributions, with highest values found in the northern Bay and along the western shore. Metal to Fe ratios of bottom sediment decrease with distance seaward from the Susquehanna River, indicating the river is a major source of Mn, Ni, and Zn.

METALS IN INTERSTITIAL WATER

Until recently, the massive reservoir of materials contained in the bottom sediments of the Bay has largely been ignored as a potential source

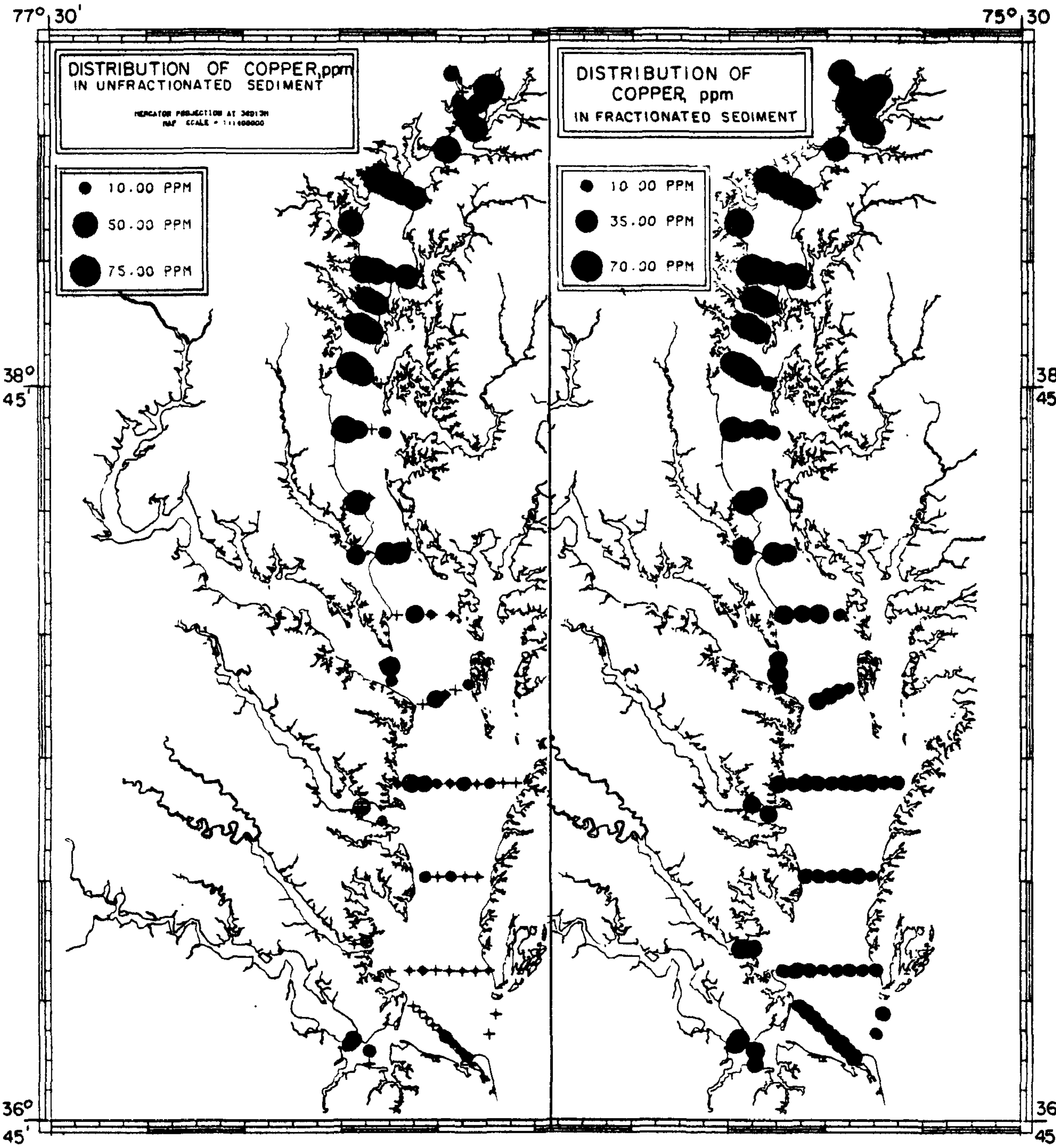

Figure 9. Distribution of Cu content in bottom sediments of (a) bulk bed sediment, unfractionated, and (b) the less than 63 u size fraction, fractionated. Data from Helz et al. (1981).

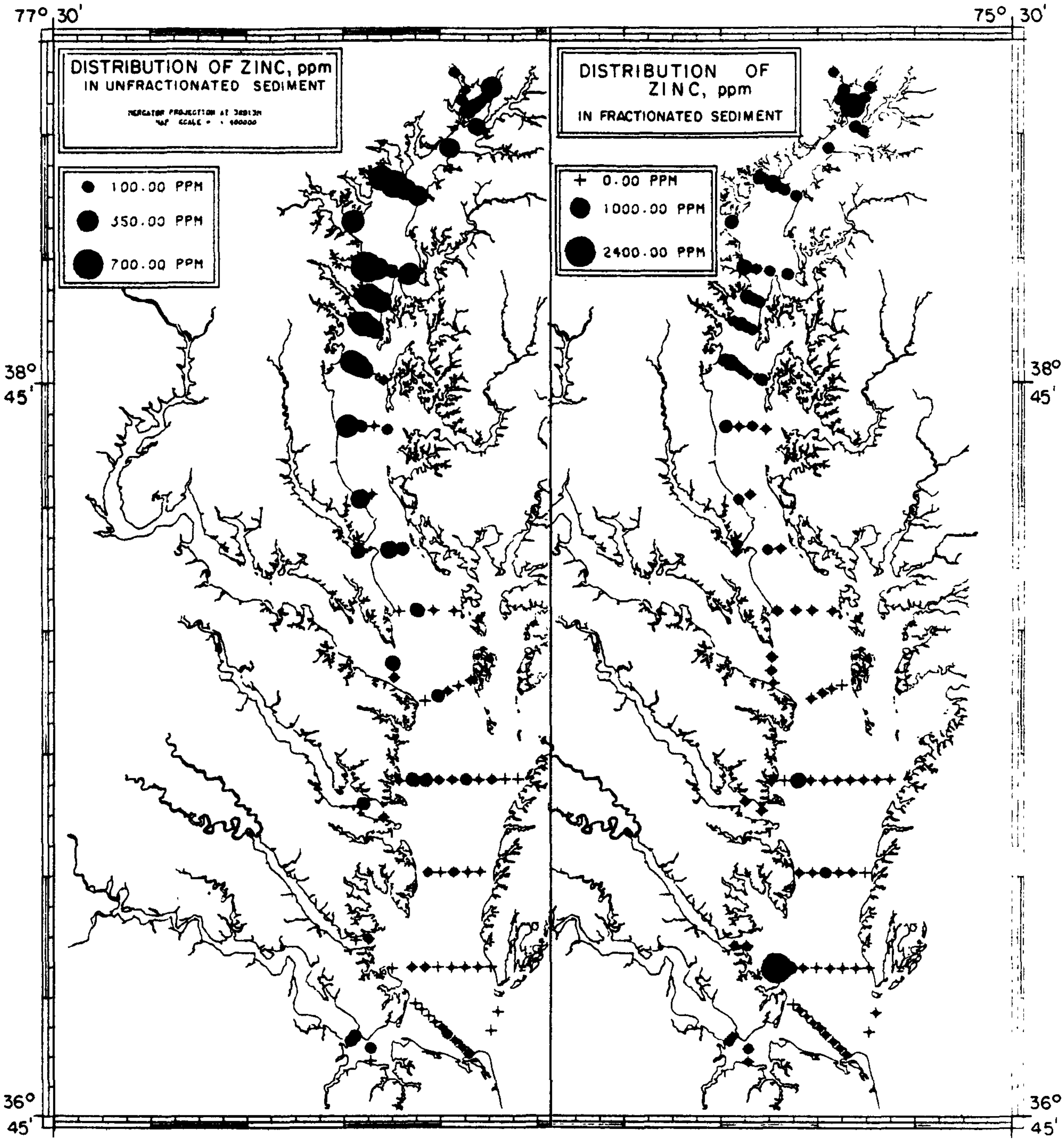

Figure 10. Distribution of Zn content in bottom sediments of (a) bulk sediment, and (b) the less than 63 u size fraction. Data from Helz et al. (1981).

of nutrients and trace elements. Previous investigations, Berner (1979) and Bricker and Troups (1975), show a substantial transfer of trace metals from the sediment to the water column. The principal vehicle for transporting this material from the sediment to the overlying water is the interstitial or pore water (water contained in the sediment). Many of the constituents of interstitial waters are derived from chemical reactions of water with the solid material of the sediment.

The constituents and parameters measured on 97 cores by Hill et al. (1981) and Tyree et al. (1981) are:

Na^+, K^+, NH_4^+, Ca^{++}, Mg^{++}, F^-, Cl^-, NO_3^-,

NO_2^-, $PO_4^=$, $SO_4^=$, $SO_3^=$, $HCO_3^=$, pH, $pS^=$, Eh,

Conductivity, Fe, Mn, and SiO_2.

A subset of these cores was analyzed for the trace metals Pb, Cd, Cu, and Zn. Figure 11 is a graphical presentation of some core data of a representative station.

The transport of dissolved constituents across the sediment-water interface proceeds in response to concentration differences. Constituents migrate from areas of high concentration to more dilute areas according to Fick's law (Lerman 1979). Generally, the concentration of nutrients (such as NH_4^+, $PO_4^=$, and HCO_3^-) and trace elements in the interstitial water exceeds the concentration in the overlying water column. Thus, the gradient predicts that these materials are transported from the sediment into the water column.

The chemical sedimentary environment controls the concentration of constituents in the interstitial water that, in turn, controls the transport of materials between the water column and sediment, and within the sediment. Three major chemical sedimentary environments have been identified for the main portion of the Bay: the northern Bay; the central Bay, including upper and lower parts; and the southern Bay, including two subsections (Figure 12). The chemical environments are classified according to a set of parameters, which influence and reflect the redox state of the sediment. These parameters are: major ionic composition of the interstitial water; organic carbon content of the sediment; reduced sulfur content of the sediment; degree of $SO_4^=$ reduction; Eh; and the concentrations of dissolved sulfide species, Fe, Mn, and NH_4^+. The three environments correspond to Berner's (1981) method of classification of sedimentary environments.

The northern Bay, as shown in Figure 12, is primarily characterized by: (1) ratios of the major ion concentrations that differ in comparison to ratios from marine-dominated environments, (2) high organic carbon content (five to six percent), (3) absence of dissolved sulfide species, (4) complete ($>$80 percent) reduction of available $SO_4^=$, and (5) the most positive Eh values in the Bay. The primary chracteristics of the central Bay environment are: (1) intermediate to high organic content (two to five percent), (2) high concentration of dissolved species, (3) variable degree of $SO_4^=$ reduction between cores, and (4) the most negative Eh in the Bay. The southern Bay characteristics are: (1) low organic carbon (zero to two percent), (2) essentially no $SO_4^=$ reduction ($\leq$20 percent),

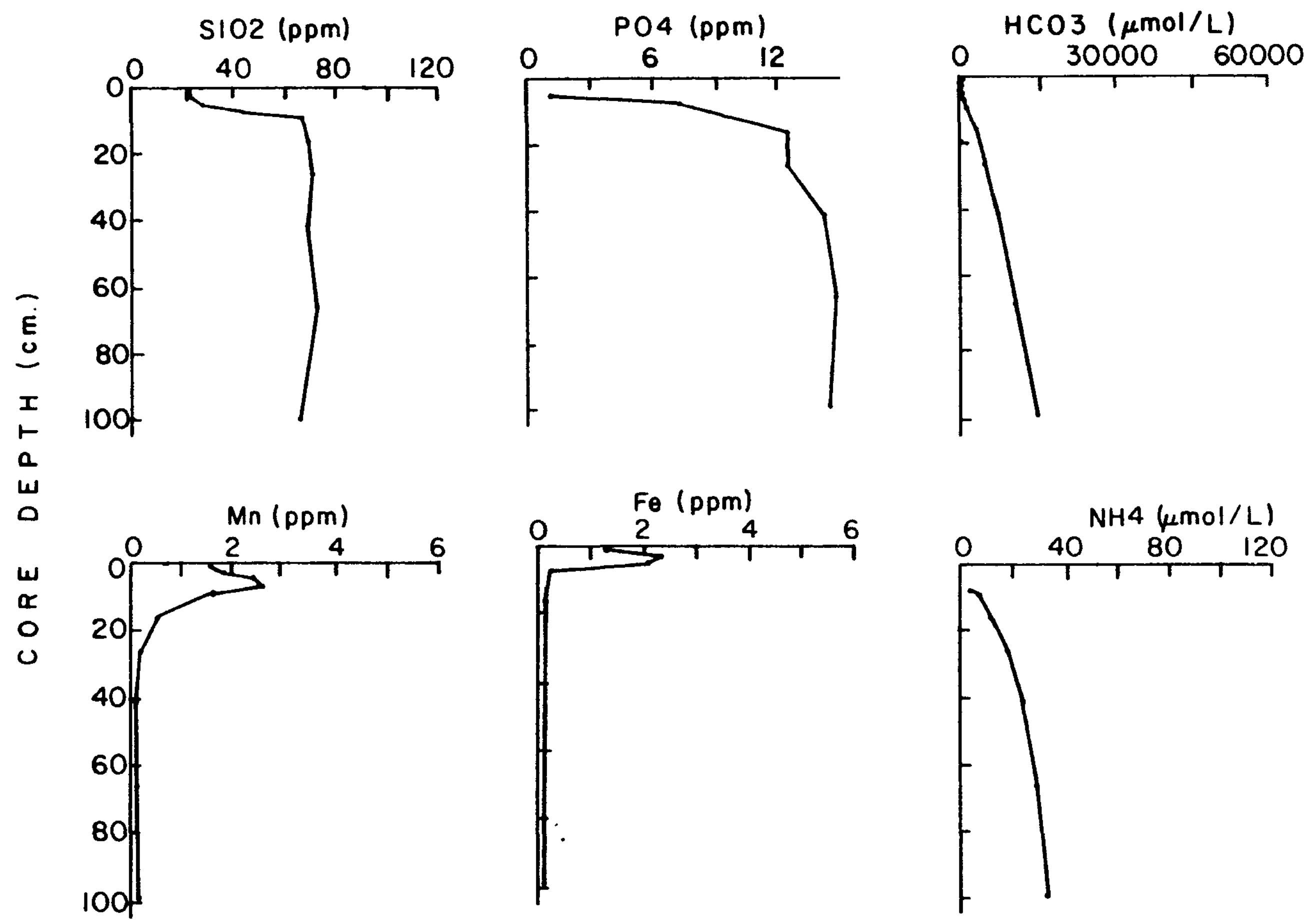

Figure 11. Vertical profiles of SiO_2, PO_4, HCD_3, Mn, Fe, and NH_4 in interstitial water composition for a station in central Chesapeake Bay, September-November, 1978. Data from Conkwight (1981) and Tyree et al. (1981).

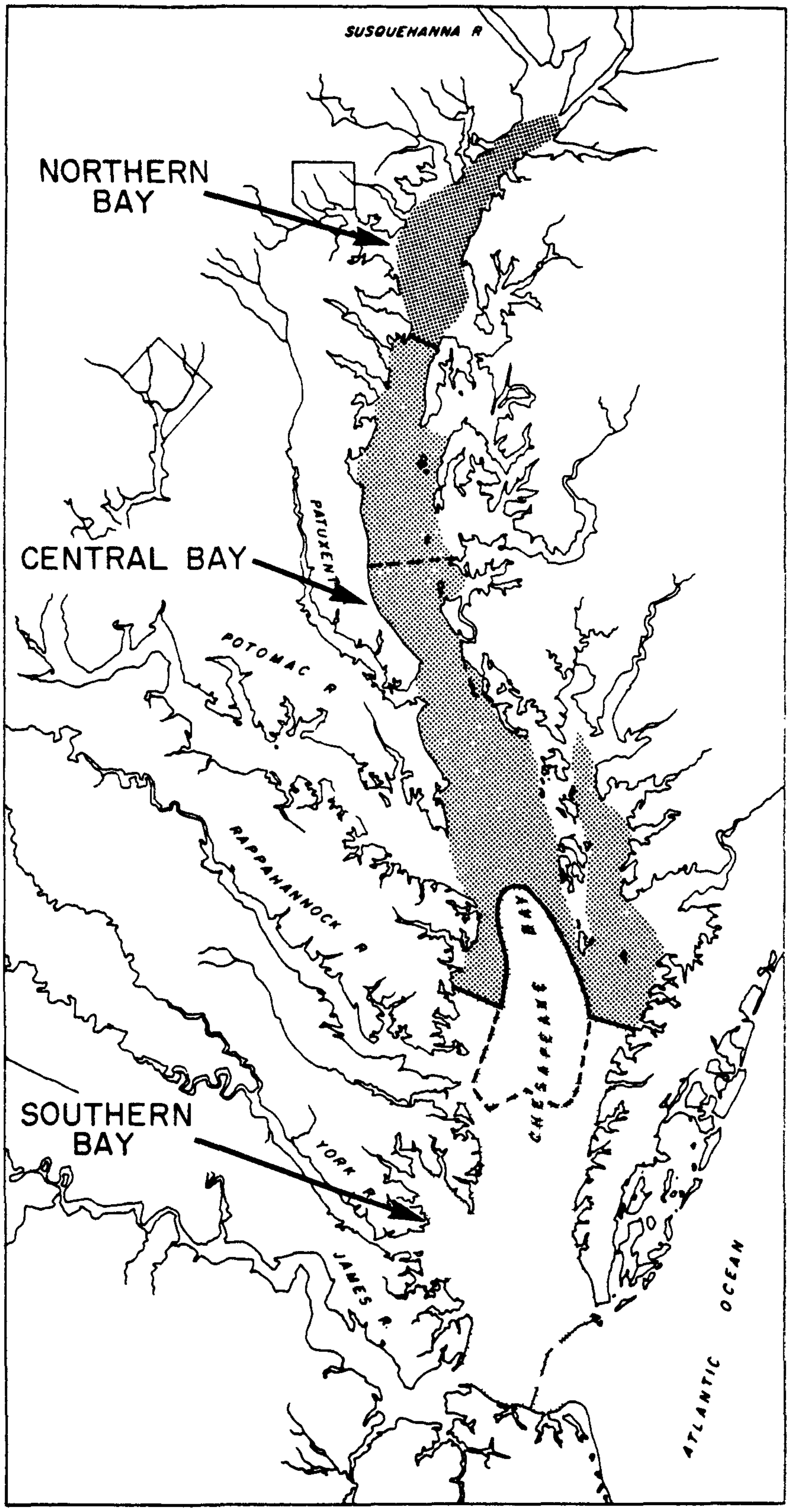

Figure 12. Distribution of chemical sedimentary environments in Chesapeake Bay, based on data of Hill and Conkwright (1981).

(3) very little detectable NH_4^+, and (4) and Eh more positive than the central Bay, but more negative than the northern Bay.

Estimates of the transport of material, with respect to the sediment-water interface, according to the three major chemical sedimentary environments, are presented in Table 10. The ranges include seasonal changes of temperature and salinity, which can markedly effect the chemical environment. The fluxes calculated from the concentration gradients generally indicate: (1) NH_4^+, HCO_3^-, and $PO_4^=$ are added to the overlying water column in the northern and central Bay; (2) Fe and Mn are transported to the overlying water column in the northern Bay, but stabilized in the sediments in the central and southern Bay; (3) sediments contribute sulfide (HS^-) to the overlying water of the central Bay, and (4) $PO_4^=$ is stabilized in the sediments of the southern Bay. The trace metal data indicate that the concentration of the metals in the interstitial water corresponds to the chemical sedimentary environments, but the concentration gradient profiles are too complicated for a simple Fick's law estimate.

TABLE 10 GENERAL ESTIMATED RANGES OF FLUXES DIVIDED ACCORDING TO CHEMICAL ENVIRONMENT, VALUES EXPRESSED AS u MOLS/M^2/DAY

	NH_4^+	Fe^{++}	Mn^{++}	HCO_3^-	$PO_4^=$	HS^-
Northern Bay	+ 50–+700	- 20–+70	-100–+60	+800–+3000	+ 30–+80	**
Central Bay	+200–+2000	-100– 0	- 60–+30	+100–+20,000	- 20–+70	+400–+30,000
Southern Bay	**	- 30–-10	-30–-10	*	-100–-20	*

** - Chemical species below detection limits in these areas
 * - Core data did not fit the simplified model used to estimate the fluxes

Note: Positive flux values reflect transport into the overlying water column; conversely negative values reflect transport into the sediment.

SECTION 3

FINDINGS FROM STUDIES ON ORGANIC COMPOUNDS

The following chapter explains the results from CBP research on the distribution and concentration of organic compounds in Chesapeake Bay. Since polynuclear aromatic compounds (PNAs) constitute the largest proportion of toxic synthetic substances entering the Bay (and are also listed on EPA's Pollutant list), much of the CBP research focused on these compounds. Other organic compounds, including dieldrin, terpenoid, DDT, and other pesticides were detected. However, extensive, quantitative analyses were performed on PNAs. In this section, sources of PNAs to the Bay are discussed, followed by results of analyses on levels of organic compounds found in bottom sediments and oysters. The remainder of the chapter interprets these results and considers important factors affecting the distribution and abundance of organic compounds.

SOURCES

The major source of most of the organic compounds (PNAs) entering the Bay is the burning of fossil fuels, coal, oil, and wood. Sources from the Patapsco River also produce compounds made up of substituted benzenes. These compounds are also released in industrial processes such as coal liquefication and gasification (Bjoreth and Dennis 1979, Cooke and Dennis 1980). Simple substituted aromatic compounds are assembled at high temperatures (combustion gases) to produce PNA compounds, with different compounds dependent primarily on the combustion temperature and secondarily on the fuel source. As indicated by PNA analysis of old sediments deposited prior to human's use of fossil fuel, very few aromatic compounds were produced by organisms. Most PNA compounds produced by combustion differ from those in oil or in the complex polymeric network of coal in that combustion products are generally not substituted.

Specific sources of PNAs in the Bay region include vehicles burning gasoline and diesel oil, coal and oil fired power plants, coal and oil fired heating industrial plants, oil and wood home heating, and forest and refuse fires. PNA compounds can be transported from the locations of the sources to the Bay by air-borne particulates containing PNA (smoke and exhaust), airborne volatile PNAs, water-borne particulates (sediment) containing land runoff and river-borne PNAs, and compounds carried in solution by rivers and land runoff. Some small amounts of PNAs are produced in the Bay by the combustion of vessel fuels.

Within the Bay, large concentrations of PNAs were found at the mouths of rivers. Some small subestuaries, like the Elizabeth River and Baltimore Harbor, with very high industrial actvity and population density, can also produce high local PNA concentrations. PNA compounds are probably continuously increasing throughout the Bay, because these many sources repeatedly produce PNA that is stable over long periods in the Bay water and sediments. A final source of PNA to the Bay is long-range atmospheric transport by Northern Hemisphere air currents. Chesapeake Bay is receiving air-borne PNA in vapor and particulates introduced in other regions of the

United States or other Northern Hemisphere countries. Contributions to PNA concentrations in the Bay from such long-range sources are probably uniform from place to place, because the Bay and its watershed area (which together serve as a PNA collection basin) are small with respect to the areal extent of single air masses.

ORGANIC COMPOUNDS IN BOTTOM SEDIMENTS

Analyses of sediment samples collected for the Bay Program during the spring and fall of 1979 (Bieri et al. 1981) revealed that over 300 organic compounds were abundant enough to either be identified or given a surrogate name by assigning a relative retention time. Only a small percentage of these 300 is not toxic in certain amounts. In some samples, the complexity and abundance of compounds present were so great that many individual species at relatively low concentrations were undoubtedly not detected. It is, therefore, probable that thousands of compounds were present. An example is presented in Figure 13, which is an actual gas chromatogram showing individual peaks. These peaks represent at least one compound superimposed on a background of peaks from numerous compounds of lower concentrations. This is commonly called an unresolved complex mixture.

The distribution of organic compounds in bottom sediments (Figure 14) is presented as bar graphs representing summed concentrations on a logarithmic scale of chromatographically resolvable compounds eluting in the "aromatic" fraction. The figures show that the highest total concentrations are encountered in the northern portion of Chesapeake Bay. Furthermore, samples from Stations 2, 4, 6, 7, 10, 11, and 12 in the lower Bay are almost devoid of these compounds. However, with the exception of the fall 1979 sample from Station 9, samples from river mouth stations, numbers 1, 3, 5, and 8, contained substantial sums--between 100 and 1000 parts per billion (Figure 14).

To demonstrate that the northern Bay and the river mouths have unnaturally high levels of organic compounds, it is necessary to account for variations in sediment character. Fine-grained sediments usually contain higher organic concentrations than coarse sediments, and this can explain some of the anomalous distributions. In general, sediment samples from the northern Bay and the major river mouths contained a higher fraction of silt and clay than elsewhere. When the samples are normalized for silt and clay content the distributions (Figure 15) change in the concentration sums in the northern Bay with the exception of Station 27, Fall 1979. In the lower Bay, only Stations 1, 3, 9, 11 and 12 have increased. Without further analyses of samples collected within the subestuaries, it is impossible to determine whether high concentrations in sediments collected near the major river mouths were due to sediment grain size or unnaturally high inputs from upstream. Normalizing the northern Bay data did not substantially change the distribution pattern. With the exception of the fall Station 19 sample, there is a trend of increasing concentrations from below the Potomac River mouth toward the Baltimore Harbor mouth. North of Baltimore, the concentration sums decrease and then increase to another maximum toward the Susquehanna mouth. Inside the Susquehanna mouth (Station 27) samples showed considerable variation between spring and fall, differences that may arise from variations of

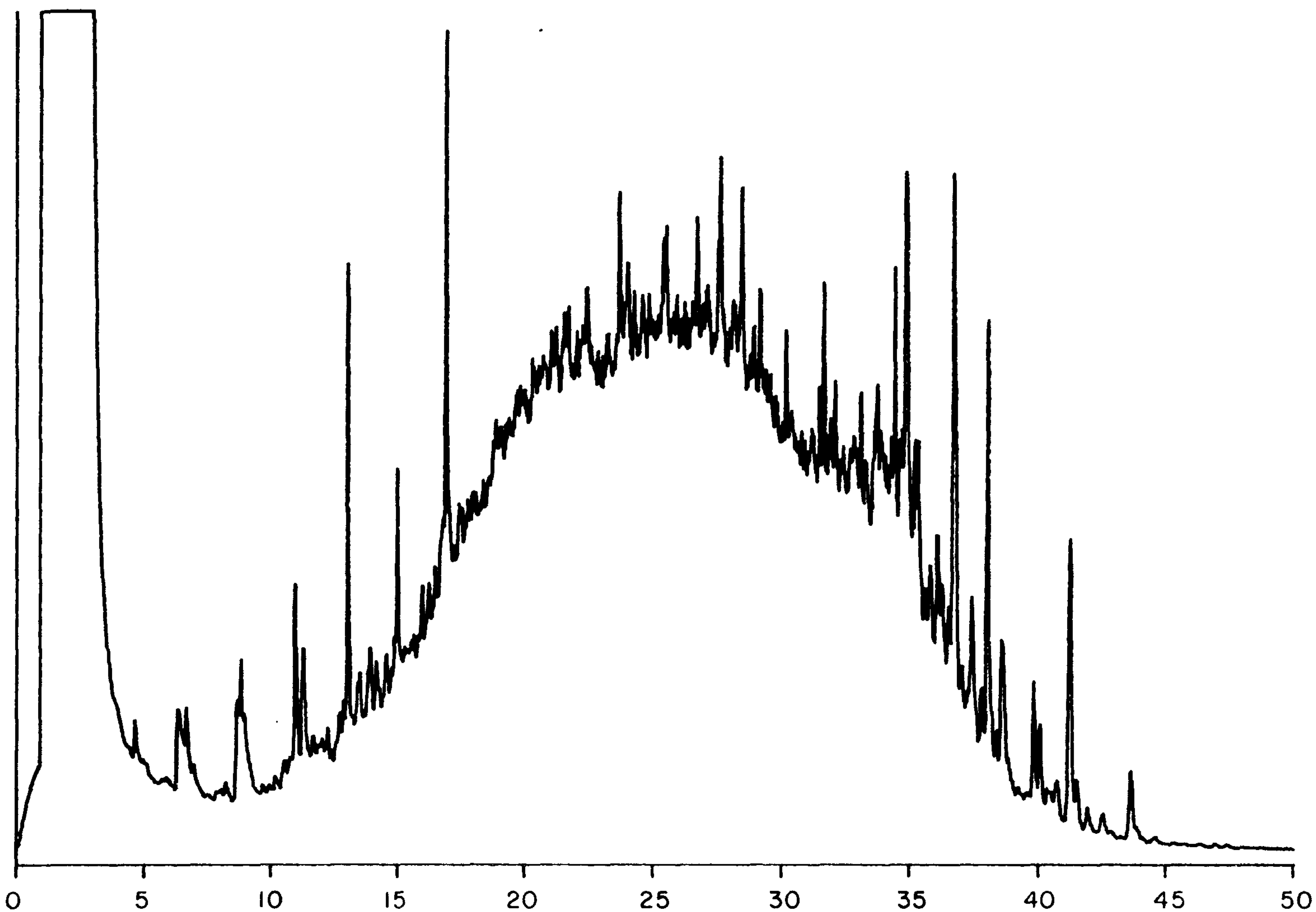

Figure 13. Typical gas chromatogram of a sediment sample. From Bieri et al. (1981).

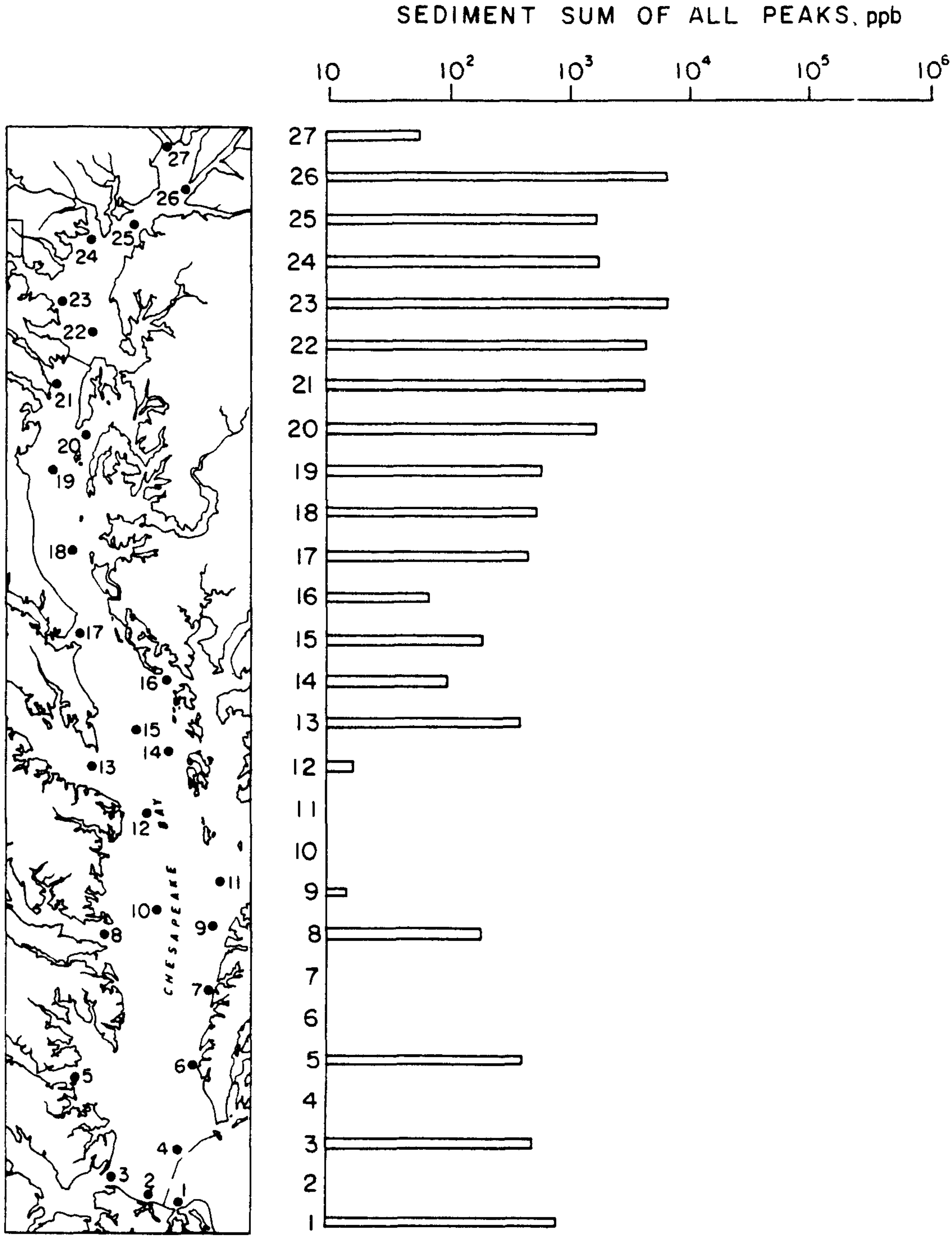

Figure 14. Chart of station locations and bar graph representing concentration sums of all resolvable peaks for organic compounds in sediments, spring samples 1979. Data from Bieri et al. (1981).

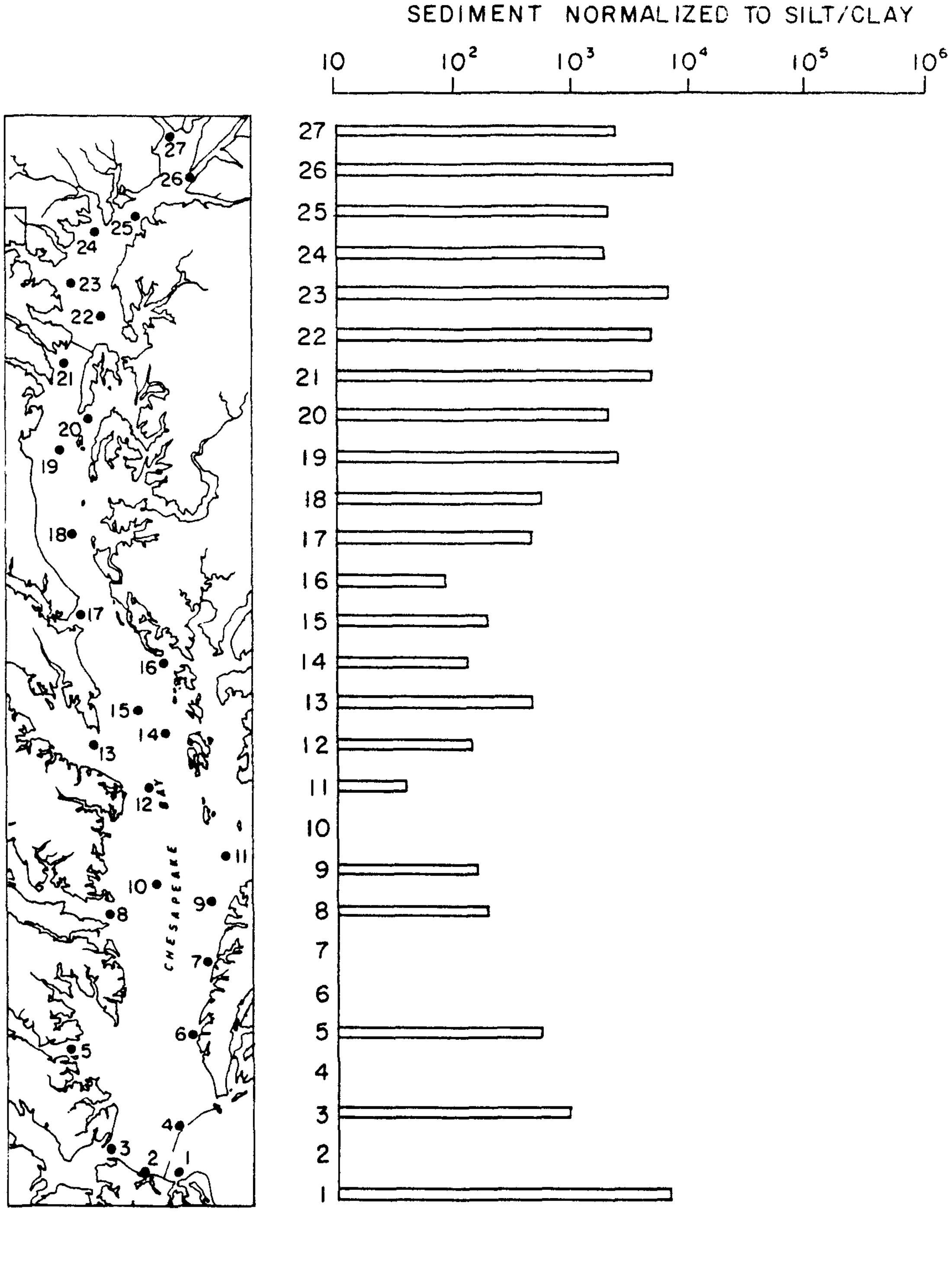

Figure 15. Chart of station locations and bar graph representing concentration sums (ppb) of all resolvable peaks for organic compounds after normalizing for silt and clay content. Spring samples, 1979. Data from Bieri et al. (1981).

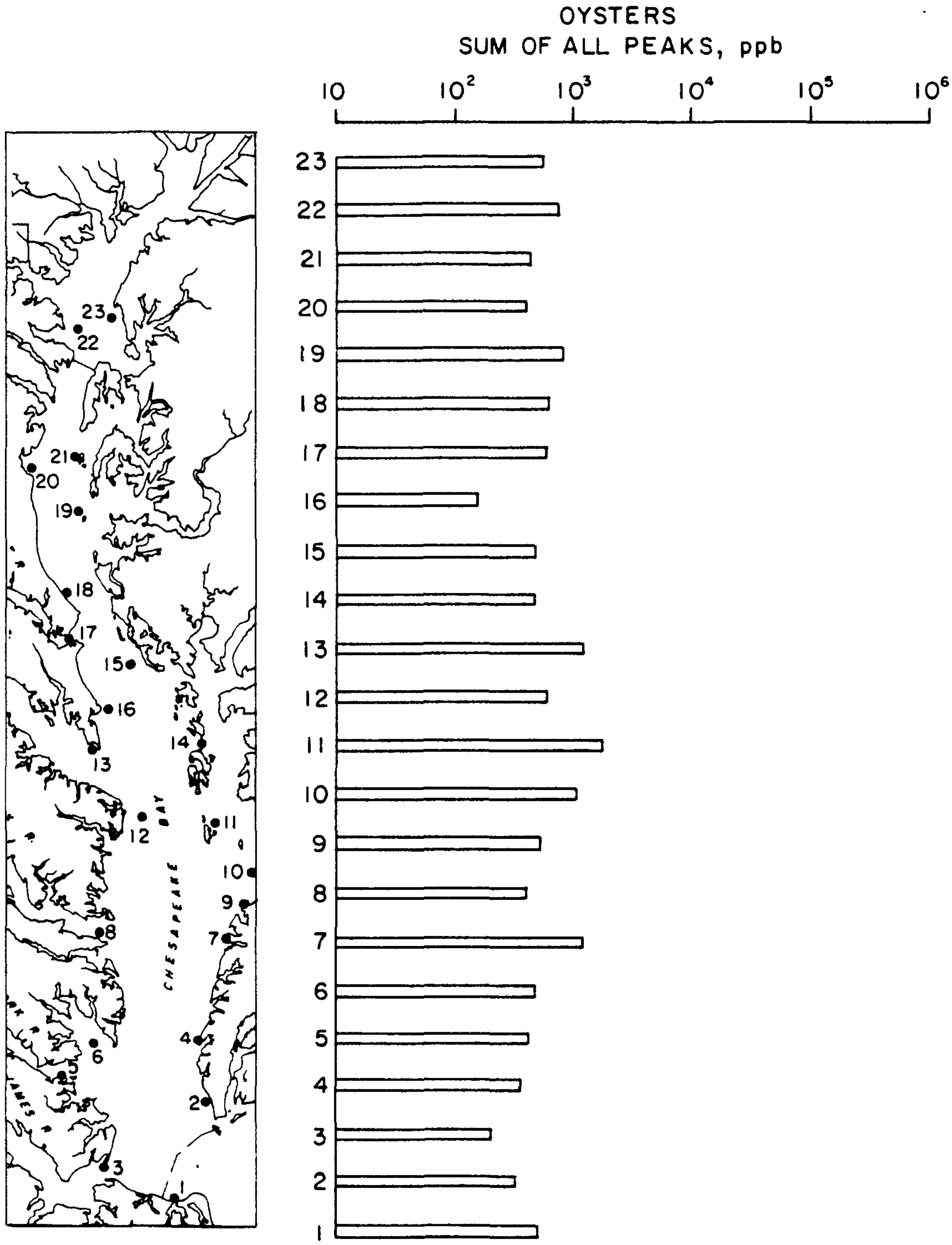

Figure 16. Chart of station locations and bar graphs representing concentration sums of all resolvable peaks for organic compounds in oysters, spring samples, 1979 (Bieri et al. 1981).

river flow that scour sediments during high spring flow and deposit sediment during low fall flow.

The trends for PNAs follow the trends for sums of all concentrations: (1) the concentrations are higher in samples from the northern Bay than in the southern Bay; (2) in the southern Bay, highest concentrations are found near river mouths; (3) concentrations tend to increase up the Bay from the Potomac River mouth toward Baltimore Harbor; and (4) the Susquehanna River mouth sediments show considerable variability, but can reach extreme concentrations. Data displayed for several individual members of the PNA family show even more clearly that a concentration maximum occurs in the northern Bay in the vicinity of Baltimore Harbor, suggesting that this area is an important source of PNA families (Bieri et al. 1981)

ORGANIC COMPOUNDS IN OYSTERS

In addition to sediments, oysters were also collected during the Bay Program and their tissue was analyzed for organic compounds. The gas chromatograms of oyster tissue extracts were much less complex than those of sediments, with the concentration of individual compounds substantially lower. The graphs for oysters (Figure 16) show no longitudinal trends like those in sediments (Bieri et al. 1981). In addition, methyl esters of fatty acids were present in most samples, as were some ketones. We hypothesize that many of these compounds have a biogenic or natural origin. Since they are often present in higher concentrations than identified pollutants, the summed concentrations may not represent a realistic pollutant content in oysters. Therefore, we examined the number of compounds detected and their distributions rather than their sums. Altogether, we identified 127 organic compounds. Oysters collected at the mouth of the James River contained 94 of these compounds. Oysters collected from Occohonnock Creek (Station 7) contained 27, and those from near Baltimore Harbor (Station 22) had 24. The oysters that contained the next highest numbers of compounds were from Holland Point (Station 20) with 23 compounds, and Onancock Inlet (Station 10) with 19 compounds. Although this analysis suggests that these areas have the highest contamination of organic compounds in oysters, there is no apparent reason why oysters from the Occohonnock Creek, Holland Point, and Onancock Inlet should compare to the James River and Baltimore Harbor, where sediment concentration of organic compounds is greatest. It is very likely that salinity or some other physical or chemical factor is influencing the levels of organic compounds in oysters.

If only the most concentrated compounds are considered, a similar pattern emerges. There were 42 compounds detected whose individual concentrations exceeded 50 ppb. The samples from the James River mouth (Station 3) contained 29 percent of these. The next highest were from Baltimore Harbor (Station 22) with 24 percent. These were followed by Station 10 with 21 percent, Station 20 with 17 percent, and Station 7 with 14 percent. In summary, the following sequence emerges from abundance of compounds: James River > Occohonnock Creek > near Baltimore Harbor > Holland Point > Onancock Inlet. For individual compound concentrations greater than 50 ppb: James River > near Baltimore Harbor > Onancock Inlet > Holland Point > Occohonnock Creek. In both cases the same five stations emerge as being the highest.

Although the presence of oysters in these locations indicates that numbers and levels of organic compounds in their tissue are probably not lethal, elevated concentrations can reach biota higher in the food web. Oysters and other invertebrates can store organic compounds in their tissue, passing on that amount to consumers. These organisms, in turn, may accumulate harmful levels.

Comparison of the compounds detected in the oysters with those found in nearby sediments showed little correlation (Bieri et al. 1981), indicating that oysters are not so useful as sediments to monitor the Bay for organic compounds. In sediment samples, the most abundant compounds were PNAs. With the exception of dibenzo-thiophene, fluoranthene, pyrene, and benzo(e)pyrene, none were detected in oysters. This could be due to the compounds not being biologically available to the oysters; or the oysters may depurate them very rapidly, or metabolize them to other compounds that were not identified.

ORGANIC COMPOUNDS IN BALTIMORE HARBOR

The CBP's sampling effort in Baltimore Harbor was identical to the work previously discussed for the main Bay. In addition, the CBP funded the Monsanto Research Corporation (MRC) to sample the major industrial and POTW dischargers in Baltimore Harbor. Together these two projects provided a mechanism by which the compounds found in Harbor sediments could be traced to possible sources in industrial and POTW effluents. Concentrations of the organic compounds in the Harbor sediments were generally much higher than those samples from the Bay. Additionally, many of the compounds found in the sediments were also detected in the point source dischargers.

Forty-one bottom sediments were collected from the Patapsco River and Baltimore Harbor during spring 1981 (Bieri et al. 1981). The PNAs dominate the aromatic compounds in the river as in samples from Chesapeake Bay proper. In some cases, the concentrations were ten to twenty times higher than the highest found in the Bay. The concentrations of the PNAs within the river also vary drastically with location. This suggests that there are either point sources of PNAs or non-uniform water circulation and sediment type that cause the organic compounds to accumulate more in specific areas. It is likely that a combination of these two factors is responsible for the distributions.

Figure 17 represents the concentrations of one of the PNAs, [benzo(a)pyrene], normalized to silt and clay content, in the channel sediments from the Patapsco River. It is obvious that there are several areas where relatively high levels exist. Point sources may be partially responsible for the anomalously high concentrations that at one location reach 5.5 ppm. The benzo(a)pyrene concentration in Bay sediments is depicted by the cylinder farthest to the right. The concentration here is about equal to that of the station next closest within the Patapsco, 260 ppb versus 290 ppb, respectively. This suggests, but does not prove, that the peak of PNAs found in the Bay near the Patapsco River mouth could be the result of transport from the Patapsco.

One sample from the Patpasco River gave a very anomalous gas chromatographic fingerprint that was dominated by an abundance of compounds with relatively low rentention times and high concentrations. The

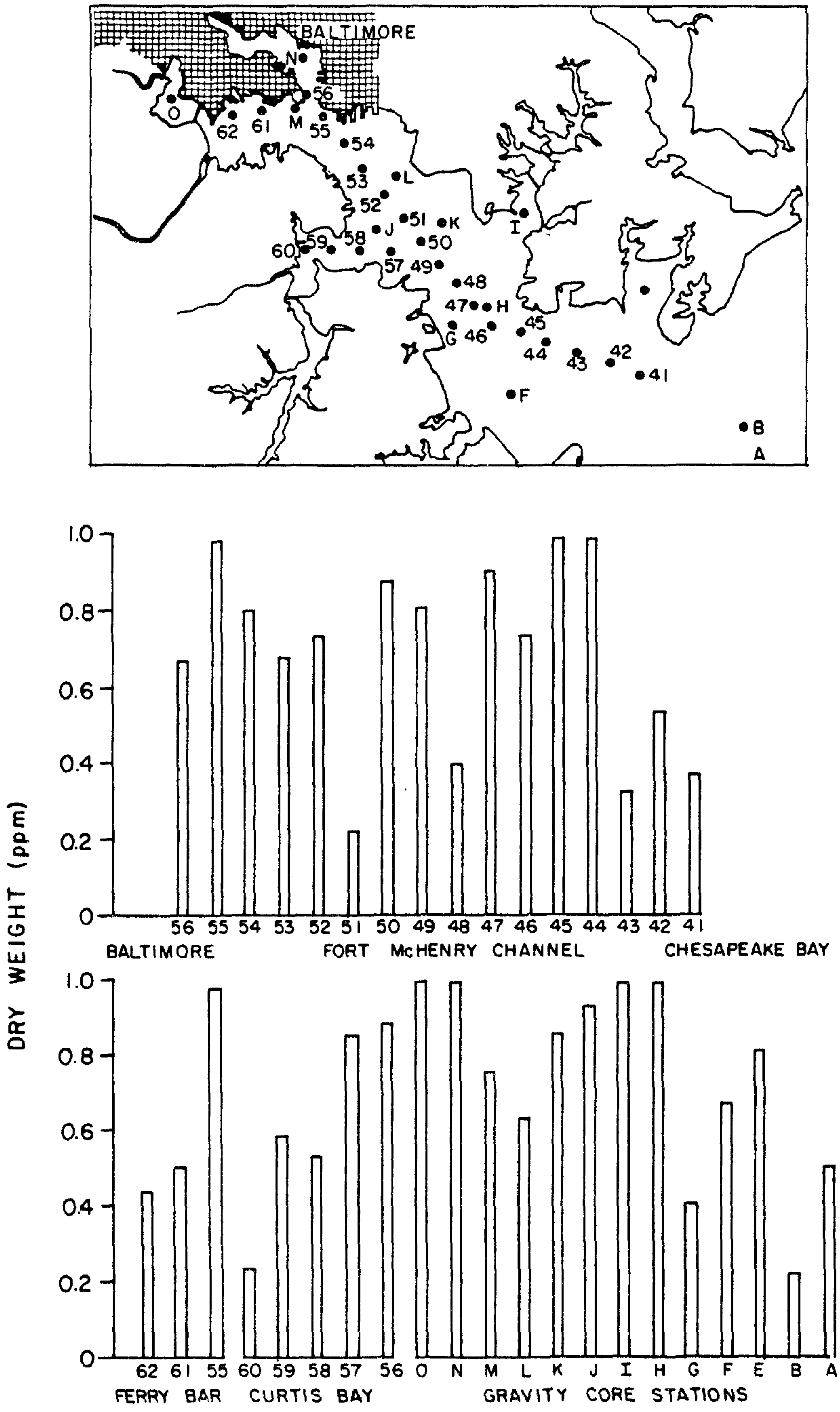

Figure 17. Distribution of PNA, benzo(a)pyrene in channel sediments from Baltimore Harbor and the Patapsco River. Relative concentration relates to height of column at each location.

compounds were not PNAs. Mass spectrometric analysis and comparison with EPA-NIH Mass Spectral Data Base showed that they were composed of substituted benzenes. The mass spectrometry data files were searched to see if these compounds were present at any other locations but had been hidden by more concentrated PNAs. The search showed that several of the substituted benzenes were either definitely present, probably present, or not present. The substituted benzene, 6-phenyldodecane, has a widespread distribution within the Patapsco River, and data indicate that sediments in the adjacent Bay also probably contain it. The sample with the highest concentration was collected landward from the river mouth.

Effluent sampling data generated by Monsanto Research Corporation (1981) showed that an effluent collected very near the sediment station contained substituted benzenes and, specifically, 6-phenylododecane. Using this compound as a tracer, we must conclude that organic compounds can enter the Patapsco River from point sources, travel throughout the river, and probably into the Bay. The fact that 6-phenyldodecane was only "probably present" in the two eastern most samples prevents stating that this is definitely the case, but it is difficult to conceive of a mechanism that would totally stop the eastward migration of the compound at the mouth of the River. It is not surprising that these two stations yield data that are less definitive than the others, because they are in the Bay where more mixing and dispersion occurs, and they are farthest from the source.

The methodology developed through the Bay Program for analyzing organic compounds within sediment of Chesapeake Bay has tremendous potential as an analytical tool for tracking known and unknown organic compounds in the system. The technique essentially generates a chromatographic "fingerprint" of the peaks found in the sample. These peaks are "tagged" by co-injecting relative retention markers and labeling each peak with a relative retention number. This becomes important when an unknown peak is found in a point source discharge and also in nearby sediment or resident fish tissue. This information allows one to "flag" potential problem compounds that may be building up or bioaccumulating in the Bay system. The technique was used in Phase II of the Monsanto Research Corporation Source Assessment Effluent Analysis and IMS sediment and oyster tissue analyses. A wealth of data on organic compounds is now available in the CBP data banks, and can be used for years, even decades to come.

In summary, the basis for our argument, stating that some of the organic compounds in the northern Bay sediments come from the Patapsco River, is that (1) PNA concentrations along the Bay rise near the Patapsco River mouth, (2) concentrations are much higher in the Patapsco River than in the Bay, and (3) the distribution of 6-phenylododecane is wide spread. Additional identification of compounds found in Harbor and Bay sediments and detected in the point source effluents has been done, but will not be discussed further in this paper.

CONCLUSIONS

Results from studies on organic compounds show that Chesapeake Bay contains many polynuclear aromatic hydrocarbons with lesser amounts in Bay oysters (Bieri et al. 1981). Because PNA compounds are fairly stable, they are transported by current flow and sediment motion to other locations in the Bay. In general, PNA compounds associate with sediment particles, partitioning in such a way that concentrations on sediment particles are much higher than in solution.

The influence of a local PNA source on PNA concentrations in the Bay will depend on the proximity of the source to the Bay, the magnitude of the source, the prevailing wind and water runoff patterns, and the characteristics of Bay sediments and current in the local region.

From this information, it can be expected that PNA concentrations in the Bay should be highest in areas of sedimentation near industrial regions, high population density areas, and power plant sites. Gradually, over a period of years, diffusion, advection, and sediment transport will spread PNA compounds over wider areas of the Bay. Although PNA transport from potential sources to sinks in the Bay can be described, quantitative measures of concentrations and transport rates are scant and inadequate.

The question which must be answered is: are the concentrations primarily the result of human activity or do they occur naturally from sources such as natural oil seeps or forest fires? The distribution and abundance of the PNAs within the Bay and the Patapsco River indicate that human activity is mainly responsible. The established origin of most unsubstituted PNAs (perylene is an exception) in high temperature reactions (Badger 1962, Schmelz and Hoffman 1976, Youngblood and Blumer 1975, Hase and Hites 1976) leaves little doubt about this fact. Since such pyrosynthesized PNAs can travel considerable distances (Lunde and Bjorseth 1977, Lunde et al. 1976), their occurrence is widespread. This may explain the presence of such PNAs in the relatively pristine areas of the Ware and Rhode Rivers, where chrysene concentrations range from 26 to 110 ppb, and benzo(a)pyrene from seven to 100 ppb. The majority of these PNAs, however, likely settle close to the source and, from there, reach the Bay by runoff and river transport.

With the increasing combustion of fossil and other carbonaceous fuels, it is likely that the PNA levels in the Bay will increase. Unfortunately, the toxicity data required to assess the resulting impact on the Bay's biota are inadequate. We do not know the toxicities of the individual components, much less the combinations, and we do not know if they are available to the biota. But the fact that many of them are carcinogenic, mutagenic, and teratogenic to mammals is enough cause for concern.

SECTION 4

PATTERNS OF TOXIC METAL ENRICHMENT

A limited, but important aspect of CBP research on metals in the Bay includes several studies on factors affecting their distribution and concentration. The dynamic nature of the Bay largely influences the behavior of metals and, consequently, their threat to the estuary. This section describes studies conducted on some of the behavioral aspects of metal inputs. It includes sections on processes affecting metal distribution; enrichment of metals above natural levels; historical trends in metal enrichment; and the important relationship between metals and sediment.

INTERPRETATION OF PROCESSES AFFECTING METAL DISTRIBUTIONS

Chemical substances like trace metals are continuously added to estuaries by inflowing tributary rivers, shoreline erosion, the coastal marine environment, the atmosphere, and the biosphere. Much of this material, dissolved and particulate, consists of the natural products of weathering, erosion processes, and of biological activity. In addition, anthropogenic products and wastes enter the estuary either directly in effluent discharges or by nonpoint source runoff. A large proportion of both the natural and anthropogenic material is intimately associated with sediments, particularly those of fine particle size and large surface area.

Suspended material is not only a reservoir for metals, but a vehicle that carries metals from their source to their depositional sink. It is an exchange medium for scavenging and removal of toxic metals from the water column. The metal distributions per liter of water show that the zone of the turbidity maximum is the most enriched (elevated above natural levels) part of the suspended material reservoir (Nichols et al. 1981). Additionally, time-series observations show that much material is resuspended from the bed, and that river-borne material is most likely trapped in the convergence of seaward-flowing river water and landward-flowing estuarine water. Enrichment is enhanced by small particle size (5-11u) of the material and by the relatively long residence time of particles in this zone.

In the central and lower Bay, metals borne on suspended material can be transported along two pathways, a hydrodynamic route, and a bioecologic route. The hydrodynamic route is revealed by dispersion patterns of metals in bottom sediments (Helz et al. 1981), whereby seaward transport from potential sources is indicated along the west side of the Bay. This route is in accord with the path of estuarine flow and the salinity regime. Landward transport through the lower Bay is indicated from metal distributions of Cr (Helz et al. 1981) that extend landward from the Bay mouth along the eastern side.

The relatively enriched metal content of central Bay surface water suggests that metals like Cd, Cu, Ni, and Pb follow a bioecological path. Because the enriched zone is generally an area of high suspended organic loads with more than 50 percent combustible organic material, it seems likely that the metals are assimilated from solution by phytoplankton or from suspension by zooplankton. Once in the food chain the metals can be further enriched (or bio-magnified) in fish or filter-feeding shellfish.

METAL ENRICHMENT

Both nonpoint and point sources contribute metals and many organic compounds to the Bay and tributaries from anthropogenic sources (Huggett et al. 1974b, Helz 1976, Brush 1974). These levels are superimposed on a background of natural concentrations. To assess the impact of human activity and control amounts reaching the Bay, it is critical to distinguish natural from anthropogenic levels.

Some organic compounds occur rarely, or not at all in nature, and their presence and concentration in sediments is direct evidence of anthropogenic input. The metals, however, occur both naturally and anthropogenically. For a given concentration of metal, there is no direct way to determine the portion that is natural and that which is anthropogenic. One method is to derive a ratio of the metal in question to a baseline metal also contained in the sample. The baseline metal should have no known anthropogenic source and should be naturally abundant so that no known pollution sources could significantly affect its concentration. The accuracy of this method can be verified by statistical tests. The precision would require comparison to known standards, which for this particular measurement, do not exist. Therefore, we cannot verify the precision and have not, at this time, determined the accuracy of this method.

Two metals, Al and Fe, were chosen to derive the ratios for determining anthropogenic levels of metals. Scandium was used by Kingston et al. (1982) in suspended sediment samples, because it is believed to have no anthropogenic sources. Aluminum and Fe were used in bottom sediments, and Fe was used in fluid mud samples. Concentrations of metals in these samples were normalized using Sc, Fe, or Al in ratios with concentrations of the metals in average crustal or shale material. For example, the ratio of Fe in average shale to Fe in Bay sediment and also to the concentration of metal in crustal material, yields an expected value for Bay sediment. The complete relation is:

$$EF = \frac{(X/Fe) \text{ sediment sample}}{(X/Fe) \text{ crust or shale}}$$

Where X/Fe is the ratio of the concentration of metal X to Fe in the sediment sample and in the crust.

The advantage of this geochemical baseline level is that it provides a standard for comparing data throughout the Bay. It assumes that the Chesapeake drainage basin is representative of average crust, and that a uniform crustal average exists throughout the region. Consequently, it does not account for local metal variations. Because the method is chemical, it is independent of sediment physical properties like particle size; it is affected, however, by compositional changes such as varying organic content within sediment.

Analyses show that enrichment factors in bed sediment for Cd, Co, Mn, Pb, and Zn are largely greater than two, and occasionally reach seven in the Baltimore-Susquehanna River area (Figure 18). For As, Cr, Cu, Hg, Ni,

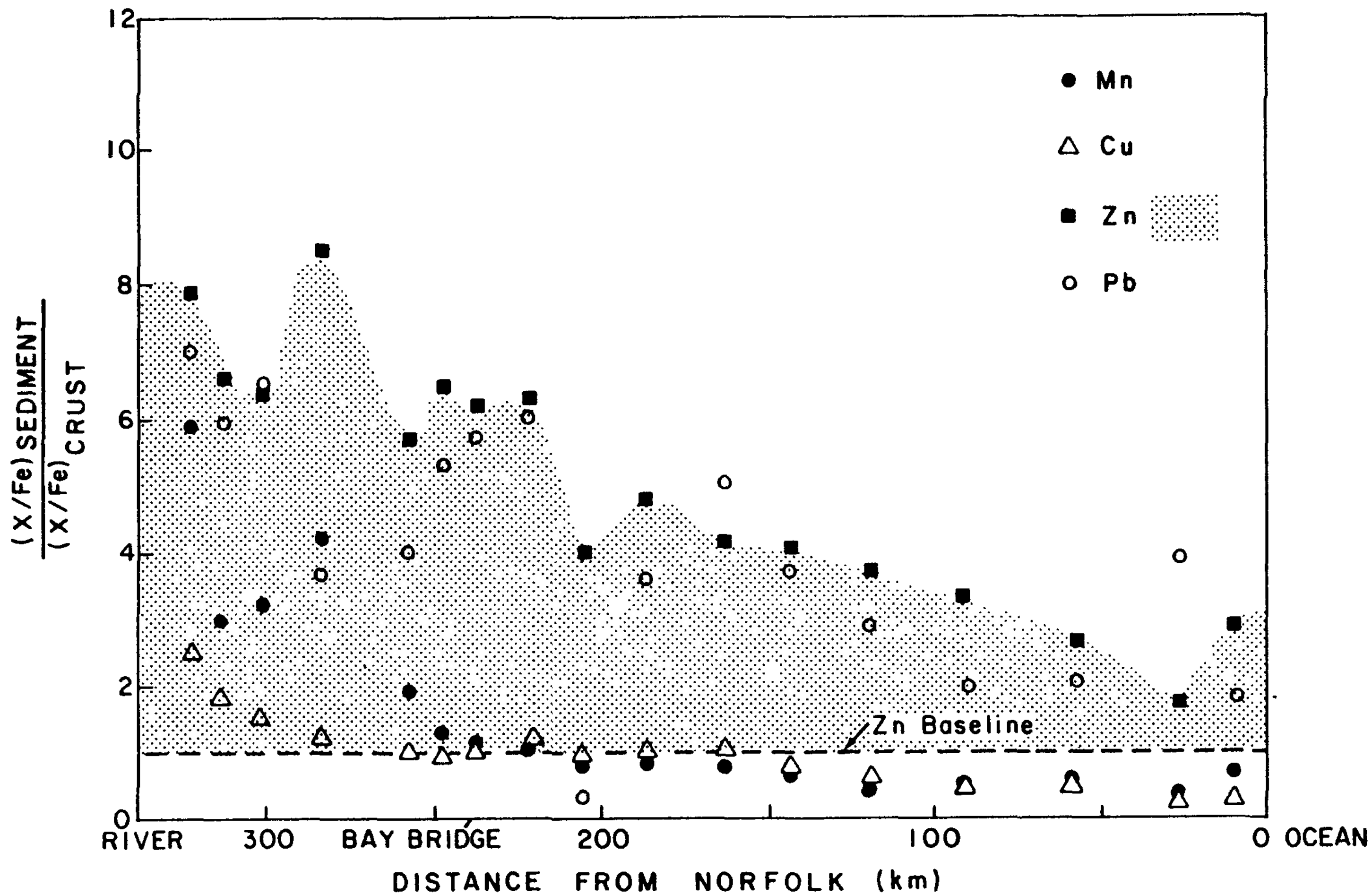

Figure 18. Longitudinal distribution of enrichment factors for Cu, Mn, Pb, and Zn in bed sediments along the length of Chesapeake Bay. Zn enrichment zones shaded. From Helz et al. 1981.

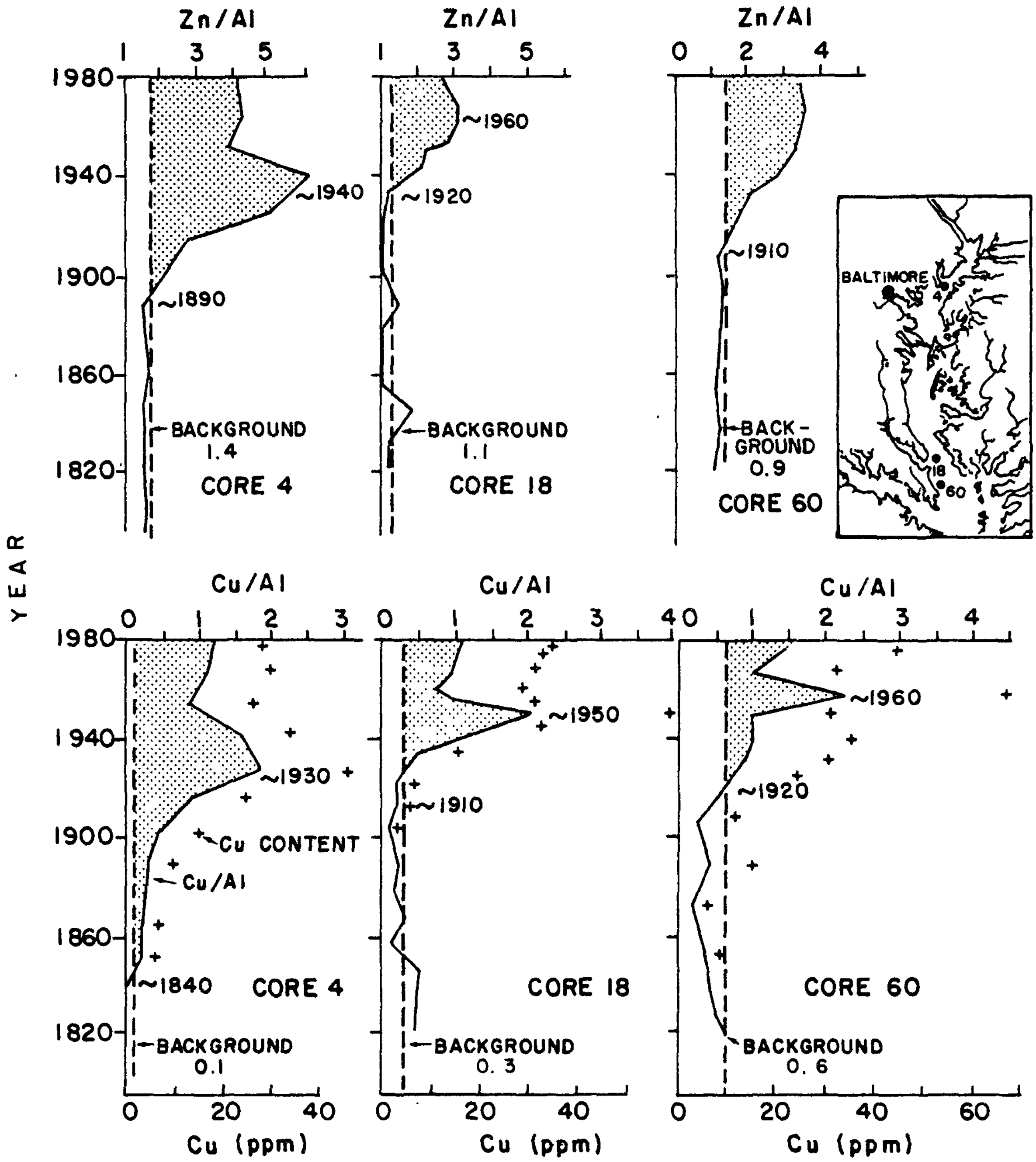

Figure 19. Metal/aluminum ratios, Zn/Al and Cu/Al, for three cores from northern and central Chesapeake Bay, cores 4, 18, and 60. Data from Helz et al. 1981. Dates in ^{210}Pb years; departure of metal/aluminum and metal/iron ratios from background in each core, shaded.

of four kilometers per year.

When interpreting concentration profiles from sediment core samples, we must be sure that the vertical concentration gradients are not a result of diagenetic processes that may alter the chemical environment within these sediments. Interstitial water data of Hill and Conkwright (1981) on oxidation-reduction (redox) potential and pH values were examined to provide an indication of the magnitude of the various chemical diagenetic processes in the sediment core samples. These data reveal no correlations between redox and pH, and metals, so we assume that the upward changes for the metal/aluminum ratios are not diagenetic; that is, there has been enrichment of trace metals with time. It is not now possible, and may never be, to assign a specific cause or source to these metal increases. However, we can speculate that human activity in the watershed and Bay has been sufficient to cause widespread perturbations. Deforestation for agriculture, mining, industrial pollution, the construction of three hydropower dams in the 1920s and 1930s, the construction of the sea-level Chesapeake and Delaware Canal, air pollution, domestic sewage, floods, and hurricanes probably all contribute to the changes observed.

Metal enrichment ratios in surface sediments vary in known geological patterns in the Baltimore-Susquehanna River zone as shown in Figure 18. The ratios increase near the surface of cores with time, matching those patterns in Figure 19. These results show that the northern Chesapeake sediments are experiencing important anthropogenic sources for Co, Cu, Ni, Pb, and Zn.

METAL - SEDIMENT RELATIONSHIPS

Analyses of metal concentrations and sediment characteristics performed during the CBP reveal a close association between metal content and certain sediment parameters. Ninety-six paired samples of surface sediments from the southern Bay metals and sediment parameters were subjected to stepwise regressions of metal content and sediment parameters. Every metal analyzed had a significant correlation with at least three independent variables (Table 11). Every metal had the highest correlation with percent silt and clay; metals in southern Bay sediments were dominantly associated with the fine particulate fraction. Over 30 years of research in other estuaries has consistently verified this finding (Forstner and Whittman 1979). Correlations with latitude represent axial variation and with longitude, lateral variation that, in turn, may reflect origins. These sources can be either up-bay or western-shore rivers, or an association with salinity that is higher seaward and along the eastern shore, than along the western shore. The regression equations are useful for predicting the metal content of bed sediments in the southern Bay when only sediment size analyses are available.

TABLE 11. RELATIONSHIP OF BULK CHEM ANALYSES OF METALS (HELZ ET AL. UNPUBLISHED) VERSUS SEDIMENT PARAMETERS (BYRNE ET AL. UNPUBLISHED) BY STEPWISE REGRESSION

Metal	R^2 [1]	Stepwise Regression Ranked Parameters[2]
Cd	.856	Silt, Clay, Latitude
Co	.763	Silt, Clay, Carbon, Latitude, Longitude
Cr	.885	Silt, Clay, Mean Size, Latitude, Longitude
Cu	.797	Silt, Clay, Mean Size, Longitude
Fe	.822	Silt, Clay, Mean Size, Carbon, Latitude, Longitude
Mn	.738	Silt, Clay, Carbon, Latitude, Longitude
Ni	.850	Silt, Clay, Carbon, Latitude, Longitude
Pb	.791	Silt, Clay, Carbon, Latitude, Longitude
Zn	.769	Silt, Clay, Mean Size, Carbon, Latitude, Longitude

[1]Significant at .0001

[2]The parameters are percent silt, percent clay, mean size, percent organic carbon, percent sulfur, percent H_2O, Latitude, Longitude. Parameters were not ranked when they did not meet a 0.15 significance level.

SECTION 5

FINDINGS ON SEDIMENTS AND BIOTA

This section describes results of CBP research on aspects of sediment and biota that influence the fate and transport of metals in the Bay. The first part discusses physical and chemical characteristics of sediment, as well as patterns of sedimentation. The second half of the section describes the character of benthic animals in the Bay and how their activities influence the availability of toxic chemicals.

CHARACTER OF BED SEDIMENTS

Because of the close association between metals and sediment, the character of bottom sediment (including its texture, water content, carbon and sulphur content), and sedimentation rates were determined in detail (Kerhin et al. unpublished, Byrne et al. 1982, Carron 1979).

Information about the surface sediments was derived from more than 4000 samples collected on a 1.0 to 1.4 Km grid. Grain size of the sand fraction was analyzed by a Rapid Sediment Analyzer, and the clay and silt fractions were analyzed by settling and pippette withdrawal and a Coulter Electronic Counter. Total carbon and sulfur were analyzed in a LECO induction furnace equipped with a gasometric carbon analyzer and an automatic titrater. Water content was determined gravimetrically by weight loss on drying.

Texture

Sediment texture is characterized by its particle size, with sand the largest and clay the smallest component. Bay sediments are differentiated into 10 classes according to the percentages of sand (0.063-2mm), silt (0.004-0.063 mm), and clay (0.0006-0.004 mm), following Shepard (1954). Of the three end members, sand covers 57.4 percent of the total Bay surface area; silt and clay less than 2.2 percent, whereas the rest of the area consists of mixtures of sand, silt, and clay. Of the total sand area (3600 Km^2), 60 percent lies in Virginia. Sand, together with mixtures of sand, silty-clay, and sandy-silt types, cover 85 percent of the total Bay area, with nearly all the silty clay in Maryland and most silty sand in Virginia.

The distribution of sediment types in the Bay is controlled by the kind of material supplied and by the processes at the site of deposition. In the northern Bay, with the exception of the Susquehanna Flats, the predominate sediment type, silty clay, accumulates in the vicinity of a potential source, the Susquehanna River. As the Bay becomes wider seaward and the relative influence of river-derived sediment decreases, sand and clay eroded from banks and shores are the most abundant sediment. Sand accumulates in more energetic zones, for example, on shoals less than about six meters, and close to its shore source. Silty clay, by contrast, resides in deep water greater than about 10 meters, a less energetic zone of inhibited wave stirring on the bed. This fine-grained sediment includes river-borne as well as marine material, shore sediment, and some skeletal material produced in the central Bay itself. The basic pattern of sand on the shoals and silty clay at greater depths is interrupted by patches of mixed sediment, silty sand, clayey sand, and sand-silt-clay. A linear zone of clay at intermediate depths along the western side, between the South River and the Potomac represents a terrace exposure of old Coastal Plain formations. Similarly, a large zone of sand on shoals along the eastern side, between Bloodsworth-Smith and Tangier Islands, is probably relic sediment.

Sediments of the southern Bay are distinctly coarser than elsewhere. Silt predominates over clay and, therefore, zones of fine sediment in deep water are clayey-silt or sandy-silt. Sand resides on shoals less than 12 meters and in channels of the Bay entrance. Locally, deep channels greater than 20 meters that are scoured by currents are floored by coarse sand.

Water Content

Sediments with high clay and silt content have a correspondingly high water content and thus, potentially high toxicant content. The mean water content of surface samples expressed as percent of wet sediment by weight, range from 16 to 83 percent for Maryland (Kerhin et al. unpublished) and from 13 to 75 percent for Virginia (Byrne et al. unpublished). The mean of all samples in Maryland is 47.4 percent and 30 percent for Virginia. A plot of water content versus mud (clay and silt) content for Virginia sediments is shown in Figure 20. This graph shows a linear trend whereby water content increases with increasing mud content. A similar trend was revealed for Maryland except for clay samples from the relic terrace zone of the upper middle Bay, an area with relatively less water content for a given clay content. The high water content of fine sediment (greater than about 64 percent dry weight or equivalent to a density of 1.30 g/cm^3) defines fluid mud that is a sub-reservoir for toxicants.

Carbon and Sulfur

Organic carbon and sulfur affect the fate of toxicants in sediments by determining the redox state of the sediments after deposition. When organic matter and sulfate of seawater is reduced, hydrogen sulfide (H_2S) is produced, and metal sulfides (as Fe_2SO_4) are formed and concentrated in the sediment. Thus, they are more available to biota.

Organic carbon in bed sediments averages 2.2 percent dry weight for Maryland and 1.0 percent for Virginia. The bulk analyses of organic carbon include organic matter of plant and animal tissues as well as skeletal parts. Isolated high values reaching 10 percent in the northern Bay are attributed in part to bituminous coal particles. The organic carbon content shows a preference for fine sediment (Byrne et al. unpublished). Regression analyses indicate strongest associations with clay fractions. Consequently, organic carbon content is higher (greater than three percent) in the deep central Bay, where fine sediment accumulates, than in the nearshore zones of sandy sediment. Inner parts of tributary embayments like Mobjack Bay and Pocomoke Sound contain more than three percent organic carbon content. The distributions of organic carbon content reveal two main sources: the Susquehanna River for the northern Bay and primary production for the central Bay. Mid-Bay organic carbon levels are the

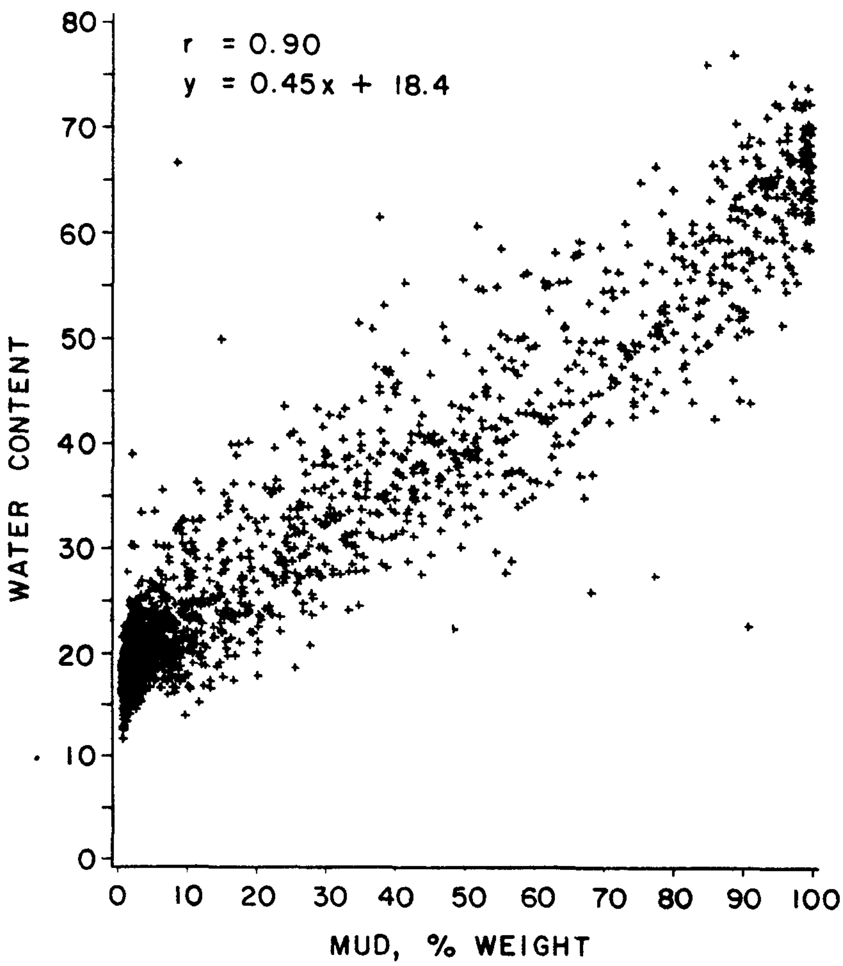

Figure 20. Relationship of percent water content to percent mud content for surface sediment samples from the Bay. Data from Byrne et al. (1982) and Kerhin et al. (1982).

result of high biological activity (primary production) in this area. The high productivity levels produce elevated carbon concentrations in the water, ultimately causing high levels of detritus (organic matter containing high organic carbon content) to be deposited and mixed with bottom sediments. The Susquehanna River contains high carbon levels from natural detrital matter (leaves, humus, fecal material, etc.), pollution sources (such as POTWs), and other natural and anthropogenic carbon sources in this drainage basin.

The bulk analyses of sulfur measure the total reduced sulfur in the form of sulfide and metal compounds, organic sulfides, and sulfate residues of interstitial waters. Sulfur content of northern Bay sediments is relatively low, less than 0.5 percent for most samples (Kerhin et al. 1982), whereas the middle Bay has one to two percent. Anomalously high values found in the main channel off the Choptank River are believed to be caused by the flux of sulfur out of nearby or underlying Miocene sediments (Kerhin et al. 1982). This is indicated by the interstitial water chemistry that exhibits positive down-core sulfide fluxes. Sulfur content of samples from Virginia is generally less than 0.5 percent except for deep zones south of the Potomac River mouth.

Sulfur in the sediments is derived from two main sources, seawater and decomposition of proteins in organic detritus. Relatively high sulfur content of middle Bay sediments is probably derived from landward-moving oceanic water as well as by deposition of phytoplankton degradation products from near-surface water. Sulfur content increases with organic carbon content in most Bay samples except the northern Bay which has relative low sulfur content and high organic carbon. This relation probaby relates to a terrigenous influence of waste from the Susquehanna River.

Patterns of Sedimentation

Changes in water depth of the Bay were established by comparing depth soundings on old charts. These changes relate to sediment deposition and erosion. Charts of Virginia provided good coverage between 1850-1860 and 1950-1960, but in Maryland some were surveyed after 1900, and the record of depth changes spans 30 years (Carron 1979, Byrne et al. 1982, Kerhin et al. 1982).

Charts of the northern Bay generally show that shoaling areas (long-term filling in greater than 0.5 meters per century) exceed deepening areas. Changes greater than 2.5 meters per century are recorded locally in the channel near Tolchester and off Kent Island, Maryland (Figure 21). As the channel deepens farther seaward, high shoaling ($>$2.5 meters per century) occurs locally on the channel floor of the upper and lower middle Bay. The deepest holes of the channel leading through the central Bay are sites of depth increase or erosion, in excess of two meters per century. Farther seaward, erosional zones are also recorded in deeper parts of the main channel near Cape Charles and in the north-entrance channels to the Bay.

Zones flanking the main channel display many variations that relate to bathymetry and geology as well as to modern sedimentary processes. For example, inner parts of marginal shoals are deepening, an indication of erosion, but outer (channel-ward) parts show no change or slight shoaling. In a general way, the change from deepening to shoaling relates to the concept of erosion of a marine shoreline as it approaches adjustment to

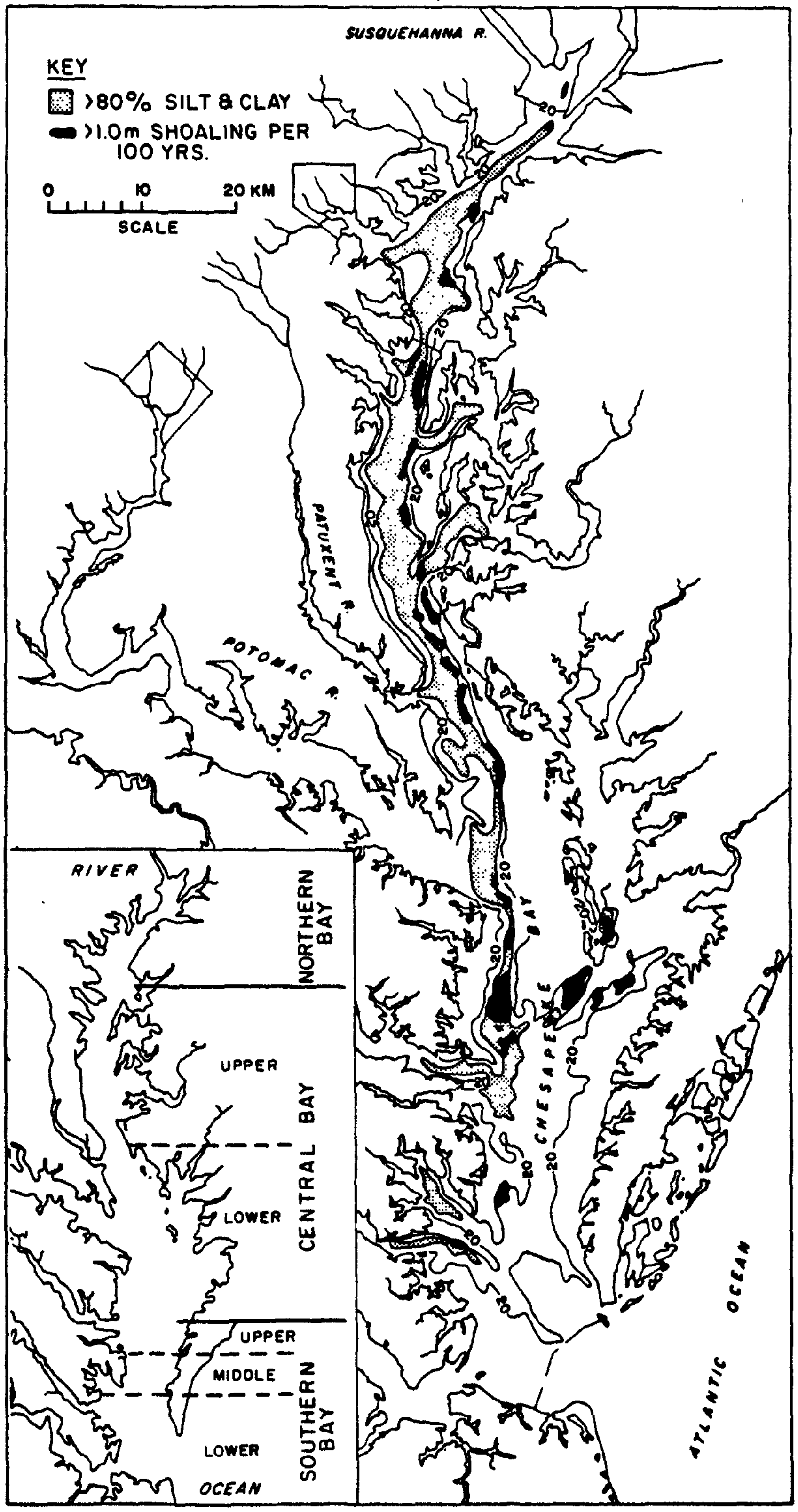

Figure 21. Sedimentation zones in areas of fine sediment, greater than 40 percent clay, with greater than 1.0 m of shoaling per 100 years, in the Bay proper. Data from Byrne et al. 1982 and Kerhin et al. (1982).

modern wave processes. The material eroded from the shore or inner shallows must be transported either laterally or channel-ward where it is deposited in deep, less energetic zones along the adjacent channel. The maximum shoaling rate in Virginia occurs in water depths of eight to 12 meters. For example, the clay terrace off Calvert County is largely erosional. It contains shoaling patches of sand along nearshore parts, suggesting offshore transport of eroded shore sand. Variable patterns on the "sand shield" around Tangier and Smith Islands, either slight deepening or shoaling in depths less than seven meters, indicate the constant reworking of sediments by wave action, local shoreline sources of sediment, migration of longshore bars, and relic sedimentary features. Other areas, like the steep eastern side of the main channel south of Core Point, have alternating patterns of shoaling and deepening that suggest slumping of the channel wall. This is confirmed by sub-bottom profiles that show slump scars at the slope break of the eastern channel wall and multiple sediment layers on the nearby channel floor.

The Chesapeake entrance and Bay floor, extending landward about 40 kilometers, is predominately shoaling (Figure 21). Most deposition occurs on elongate shoals; some occurs on flanks of the large Horseshoe Shoal, the main Chesapeake channel floor, and the lower part of old Plantation Flats. Most of the shoaling material is fine to very fine sand, probably derived from the Bay entrance on adjacent shores and inner shelf, and transported landward by the net residual bottom flow.

Toxicants may be expected to accumulate in areas of fine sediment shoaling. The rate of toxicant accumulation will vary from place to place in proportion to the shoaling rate (Figure 21). By contrast, deep channels where erosion is active, are poor places to dump waste materials because the currents would remove them. Areas in which the channel is stable or shoaling are the best sites for disposing waste materials.

BENTHIC ORGANISMS

Benthic organisms act with physical processes to either enhance or inhibit movement of toxic material. They can redistribute dissolved toxicants in interstitial water or mix contaminated sediment within the bed, as well as between the bed and overlying water. Through their feeding and burrowing activities, they can bury new surface sediment or expose older deposits. At the same time, their activity can stabilize surface sediments through binding or tube building. On the other hand, they can mobilize sediment by decreasing compaction and increasing water content. By feeding and filtering suspended sediment and by excretion, they produce fecal material and, in turn, promote sedimentation.

Character of Benthic Fauna

The distribution of benthic organisms in Chesapake Bay has been documented in a number of studies (Boesch 1977a, 1977b; Holland et al. 1977; and Loi and Wilson 1979), most of which indicate that both physical (salinity, substrate type, depth) and biological (competition and predation) factors influence the distribution and abundance of the macrobenthos. The wide range of habitats sampled in this study affords the opportunity to make generalizations concerning species distribution on a Bay-wide basis. To avoid the confounding effects of seasonality on community structure, fall 1978 and summer 1979 collections were considered separately in a numerical classification analysis (Diaz and Schaffner 1981, Reihnarz and O'Connell 1981).

Community Composition

Of the animals sampled in the Bay, polychaete annelids were the most abundant and diverse taxonomic group, consisting of 23,797 individuals and 95 species. Crustaceans were second in abundance and diversity with 10,427 individuals and 48 species, and molluscs were third with 5,088 individuals and 43 species. Miscellaneous groups were represented by 310 individuals and 17 species.

Although the number of species did not change drastically from fall 1978 to summer 1979, a great disparity existed between the number of individuals and the relative composition of fauna collected. Some of this disparity is explained by an increase in the percentage of muddy stations sampled in the summer relative to the fall. More importantly, summer collections, particularly in the lower Bay, contained large numbers of juvenile polychaetes that were presumably recruited to the sediments during the spring. Low abundances in fall collections may result from the heavy predation pressure, by blue crabs and fish, exerted on these populations throughout the summer (Virnstein 1977).

Species Diversity

Mud habitats were generally less diverse and had fewer species than sand or mixed-sediment habitats. In some cases, these results related to the fact that stations were located in deep channels or sound areas where periodic oxygen depletion resulted in a depauperate fauna (Diaz and Schaffner 1981, Reinharz and O'Connell 1981).

Vertical Distribution

The majority of macrobenthic organisms, in all salinity regimes and sediment types, were found in the upper 10 centemeters of the sediment column. Generally, mixed or sandy sediments had the greatest percentage of deep-living organisms. Most of the organisms below 10 centimeters are annelids.

Bioturbation

Evidence from both the vertical distribution studies and x-radiography suggests that nearly all of the benthic communities in the Bay have the potential to move and mix sediments, which in turn can affect the fate and distribution of sediment-bound toxicants. The modifications of physical structure in sediments by organisms (bioturbation) fall into three categories: (1) the construction of tubes as dwelling structures, (2) the abandonment and subsequent filling-in of old tubes, and (3) general sediment disturbance and mixing from locomotion. Analyses of the degree of bioturbation as estimated from x-radiography of box cores indicate that levels of bioturbation and types of biogenic structures vary depending on both salinity regime and sediment type (see Reinharz et al. 1980, Nilson et al. 1980).

Sandy habitats in the Bay are generally restricted to the head and mouth of the Bay as well as to some areas along the eastern shore. Physical structures preserved in these regions include cross-bedding patterns and ripple lamination. In shallow, high energy regions of the upper Bay, some of these structures have been completely disrupted because of wave action. Sands in the lower Bay generally have a uniform bioturbated sediment fabric, reflecting movement and mixing by communities composed of a highly mobile fauna.

Mud habitats are most abundant in the lower salinity regimes of the Bay, north of the Rappahannock River. Physical structures dominate the muddy sediments of deep channels and holes at the mouths of major rivers. Stressful fluid mud substrate and periodic summer anoxia allow only the temporary settling of opportunistic species.

Muds in shallower regions are less likely to suffer anoxic conditions and have a more diverse fauna for mixing sediments. In all areas of the Bay, biogenic structural diversity is greatest in shallow mud habitats.

Bay-wide patterns in degree of bioturbation, based on x-rays of sediment cores, are summarized in Figure 22. Sediments are highly bioturbated (90-100 percent) throughout most of the Bay. Areas where bioturbation is low include the uppermost oligohaline reaches of the Bay, deep channels, sounds, and river mouths that are presumably subjected to periodic oxygen depletion and often characterized by fluid mud substrate.

Biological Sediment Mixing and Fate of Toxicants

Evidence from both the vertical distribution studies and x-radiography suggests that nearly all of the benthic communities in the Bay have the potential to move and mix sediments and, in turn, influence the fate and distribution of sediment-bound toxicants. Several studies (Rhoads 1963, Gordon 1966) have measured particle mixing rates of common marine invertebrates of shallow-water North Atlantic habitats and have found them to exceed annual sedimentation rates. Depending on local sedimentation rates, sediment-bound toxicants may be retained in the upper sediment layers as a result of biological activities.

Areas of high sedimentation rate (generally in the oligohaline salinity regime of the upper Bay [Figure 21] and in some channel areas) were generally found to have low levels of bioturbation. Thus, the fate of sediment-bound toxicants in these areas would probably be primarily controlled by non-biological physical factors such as storms. The fate of toxic materials in the mud habitats of the central and lower Bay, where bioturbation averages greater than 90 percent, would probably be influenced by biological mixing. The probability for retention of toxicants in surface-sediment layers in these habitats seems high because of the turnover of sediments by animals.

The effect of bioturbation on the vertical distribution of heavy metals in the sediment is revealed by depth distribution of radioactive lead. This isotope, 210 Pb, is delivered uniformly to the Bay from atmospheric sources. Once in the sediments, its concentration is proportional to the rate of sedimentation and time because it radioactively decays. The deeper the sediment, the less ^{210}Pb (Helz et al. 1981). This is found to be the case in areas where there is little or no bioturbation; for example, in the deep muddy channels of the middle Bay. However, in areas of high bioturbation there is a zone of uniform ^{210}Pb concentration that corresponds to a biologically active zone where animals are mixing the sediments. Such areas were found in the upper and lower Bay where bioturbation caused mixing of sediments down to levels equivalent to 50 years of deposition. Therefore, in these areas toxicants are not likely to be buried.

SECTION 6

TOXIC SUBSTANCES AND BIOTA

An important question remaining in the CBP's investigation of toxic substances is whether or not levels found in the Bay are harmful to the many organisms living there. Although assessing the toxicity of metals and organic compounds was not part of the CBP's original scope of work, a limited evaluation of some metals and organic compounds was done. Further assessment of the problem is presented in the third CBP final report, Characterization of Chesapeake Bay (in progress). Specifically, the characterization report includes discussion of levels of organic compounds and metals in the water column and bed sediment, with a separate section on Kepone in the James River.

This section addresses toxicity studies done during the research portion of the Bay Program. It includes results from the CBP's exposure assessment, experiments on histopathology of a native bivalve, and bioassays of sediment and industrial effluent.

EXPOSURE ASSESSMENT

This discussion only addresses concentrations of toxic chemicals in the water column measured during the CBP Toxic Substances Program, and for which we have EPA criteria. The EPA Ambient Water Quality Criteria Documents (EPA 1980) for priority pollutants, lists the criteria values. These are expressed as the total recoverable concentration in the water column, including dissolved, plus the potentially biologically available fraction associated with suspended sediment. Assuming that any metal attributable to enrichment is potentially biologically available to biota, we can calculate the "available" concentration of that metal. Adding this to the concentration of dissolved metal produces a reasonable, and probably conservative, estimate of the total recoverable value.

Except for the Baltimore-Susquehanna River mouth zones, no metal exceeded the EPA criteria in the Bay proper. Above Baltimore, several stations barely exceeded the 24-hour average (chronic) criteria for Cd or Cu. The criteria violated are based on subtle chronic effects of sensitive species, the impact of which is not understood, and the calculated concentrations exceeded these criteria only marginally. These violations alone do not necessarily imply a serious ecological impact. Additionally, there is some evidence that organisms can acclimate to toxic substances, thereby lowering their sensitivity to those toxicants. On the other hand, there may be species that are more sensitive than the species tested. In addition, synergistic interactions may greatly increase the toxicity of a pollutant, thereby affecting the biota even at sub-criteria levels.

Although this assessment does not show immediate ecological impacts, the toxicity of some Bay sediment (see section on Sediment Bioassays) and the proximity of metal concentrations to EPA criteria values (recommended levels for water) indicate that north of Baltimore the Bay may border on toxic impacts. Additional loadings of toxic substances to these waters may, therefore, prove harmful to the biota.

TOXICITY STUDIES

Histopathology

Diaz et al. (1981) conducted preliminary studies on populations of the bivalve, Macoma balthica, to determine potential toxic effects. [See "Characterization of Chesapeake Bay" in progress for more complete analyses.] Macoma balthica is an infaunal species that burrows to 30 centimeters deep in soft mud. Although not a commercial species, Macoma was selected because it has varied feeding habits in both surface deposits and suspended material, and it is ubiquitous. Seven hundred and forty clams were analyzed for abnormalities from relatively contaminated sites of the Patapsco and Elizabeth Rivers and from relatively uncontaminated sites of the Rhode and Ware Rivers. Of the 740 clams examined, only 26 pathogenic cases, or 3.5 percent, were found (Table 12). No statistical relationship is evident between the pathogenic conditions and the river system in which the clams reside, indicating that the data do not reveal any adverse effects of sediment-associated contaminates.

Sediment Bioassays

Since many potential toxicants accumulate in the sediments at concentrations higher than in the water column, preliminary bioassays were performed on sediment from 70 sites throughout the Bay and selected tributaries including the Patapsco and Elizabeth Rivers. The infaunal amphipod Repoynius abronius, a species considered sensitive to sediment contamination, was collected from relatively uncontaminated sediment and water from Oregon. Repoynius abronius was placed in test sediment from the Bay, and in the relatively uncontaminated sediment for control, at the EPA Marine Science Center, Newport, Oregon. The samples were split and run in both quiet (non-stirred) and stirred, aerated, overlying water of 25 ppt salinity. The stirring action was induced to release interstitial water and obtain a common salinity in all samples. After ten days, the number of survivors were recorded from sieved samples.

The highest mortalities, greater than 90 percent, occurred in stirred and non-stirred samples from the upper reaches of the Patapsco and Elizabeth tributaries and from the northern Bay, particularly in the zone between Baltimore and the Susquehanna River mouth. As shown in sections III and IV, sediments from this zone are generally more enriched in metals and organic compounds than elsewhere. The results of these experiments conclude that toxicants may cause experimental mortality.

TABLE 12. SUMMARY OF HISTOLOGICAL ABNORMALITIES FOUND IN *MACOMA BALTHICA* CLAMS FROM UPPER AND LOWER BAY TRIBUTARIES (DATA REPRESENT NUMBER OF CLAMS WITH ABNORMALITIES; PARENTHESES INDICATE THE PERCENT OF TOTAL FROM THE RIVER)

	Total Clams Examined	Number of Pathogenic Cases Dermo	Bacteria	Glandular Cysts	Total
Upper Bay					
Patapsco River	404	7(1.73)	1(0.25)	1(0.25)	9(2.23)
Rhode River	189	2(1.06)	1(0.53)	5(2.65)	8(4.23)
Lower Bay					
Elizabeth River	83	1(1.21)	0(0.0)	1(1.21)	2(2.41)
Ware River	64	2(3.12)	0(0.0)	5(7.81)	7(10.93)
Totals	740	12	2	12	26

Effluent Toxicity Tests

Of an estimated 5000 discharges in the Chesapeake region, approximately 1000 are considered to have the potential for discharging toxic material based on criteria established by the National Enforcement Investigation Center of the U.S. Environmental Protection Agency. As part of the CBP Source Assessment Program, effluent from fifty of these dischargers was sampled and characterized in terms of major chemical species (down to 1-10 ppm) and their potential toxic effect on biota as determined by bioassay tests. The selections were based on industries with the highest potential for toxicity (not known toxicity problems). The criteria for ranking the industries were based on flow rate of effluent and expected concentration of chemicals in the effluent. The bioassays were conducted to evaluate, or indicate toxicity of the effluent. The dischargers from which effluent was sampled during the Program are shown in Appendix E. This appendix also shows the many different bioassays performed and the experimental results. Values of results are expressed as percentages of diluted effluent that caused death for various species tested. The EC_{50}, LC_{50}, [or SC_{20}, EC_{50} (Effluent Concentration)] is the percentage of effluent that would inhibit growth by 50 percent. LC_{50} (lethal concentration) is the percentage of effluent that caused a 50 percent kill of the species. SC_{20} is the percentage of the effluent that stimulated growth by 20 percent. Bioassays were performed on fish, several invertebrates, bacteria, and seagrass. Table 13 shows the kinds of tests used.

TABLE 13. TESTS USED FOR MEASURING POTENTIAL TOXICITY OF INDUSTRIAL EFFLUENT

Organism	Test
Fathead minnow	96 hr. LC50
Sheepshead minnow	96 hr. EC50
Daphnia sp.	48 hr. LC50
Mysid shrimp	96 hr. LC50
Thalassia sp.	3-week EC50
Marine bacteria	EC50 Microtox

Results: Bioassays of Fathead minnows and Sheepshead minnows were tested at minimal, low, moderate, and high toxicity values (NT-75, 50-75, 25-49, and 0-24 respectively, Appendix F). Twenty percent of the effluents sampled exhibited moderate to high toxicity, whereas 80 percent exhibited minimal to low.

Invertebrate bioassays of Daphnia and mysid shrimp were tested at minimal, low, moderate, and high toxicity values, NT-75, 50-75, 25-49, and 0-24 respectively (Appendix G). With the results of these two bioassays combined, approximately 30 percent of the effluents sampled indicated moderate to high toxicity. In addition, the mysid shrimp appeared more susceptible than the Daphnia to the toxic substances found in the effluents.

A Marine Bacteria Bioluminescence Bioassay indicates that 50 percent of the effluent samples were moderate to highly toxic. However, a bioassay on Thalassia (Sea Grass) displayed little or no effect from the effluents (Appendix H).

Mutagenic and cytotoxic effects were tested by utilizing Salmonella/microsomal (Ames Test) spot tests and plate incorporation assays (not listed in Table 13). These were performed on filtrates and extracts of 10 effluent samples. No mutagenic response was observed in the pour-plate assay with the particulate recovered from sample filtration (Appendix I). A positive mutagenic response in sample A108 Filtrate I was observed using the plate assay. The spot test of effluent sample A104 Filtrate I showed an increase in revertants over the control, but no clear positive response.

The Chinese hamster ovary (CHO) mammalian cell cytotoxicity assays showed that effluent samples from A105, A106, A110 exhibited medium level toxicity for the sample as received; A100 showed low toxicity in samples A102, A103, A104, A106, A108, and A110 (Appendix J). Acetone extracts of the particulate showed low or very low toxicity ratings for samples A100, A103, A106, A107, A108, and A110. Samples A101 and A109 showed no toxicity for any of the three types of sample.

In summary, effluent bioassays on fish, invertebrates, and bacteria indicate that 20 to 50 percent of the effluents sampled had moderate to high toxicity. A greater risk of toxicity in the Bay is generally associated with high effluent toxicity.

SECTION 7

CONCLUSIONS, INTERPRETATIONS, AND MANAGEMENT IMPLICATIONS

The following abbreviated statements are organized to review the key observational findings (underlined) followed by an interpretation and management implication(s).

METALS

1. The Bay receives metals from human and natural sources through rivers, the atmosphere, and industry. The rivers are a dominant pathway for Cr, Cu, Fe, and Zn; industry is a dominant source of Cd, and the atmosphere is a significant pathway for Pb and Zn. Metal input to the main Bay is greatest from the Susquehanna River.

Metal input from rivers is relatively high because of large contributions from geologic weathering and soil erosion of fine sediment in the drainage basins. Additionally, rivers supply metals from municipal and industrial effluents and, indirectly, from atmospheric deposition on the drainage basin. The Susquehanna River is a strong pathway because of its relatively large water and sediment discharge.

The Susquehanna is the only river that discharges directly into the Bay. Main tributaries, like the James and Potomac, discharge into estuaries that entrap sediment and sediment-borne toxicants.

2. Bay water contains the metals, Mo and U, mainly in dissolved form ($>$ 90 percent of total metal), and they positively and linearly correlate with salinity. The metals Cd, Co, Cr, Cu, Ni, Pb, and Zn occur both in dissolved and particulate form (between 10 and 90 percent are dissolved), whereas more than 90 percent of the Fe, Mn, Sc, and Th occurs in particulate form.

Relatively high concentrations of Mo and U are probably controlled by alkalinity of Bay water and by dilution of seawater with river inflow. The concentrations of metals Cd, Co, Cr, Cu, Pb, Ni, and Zn are controlled by complex interactions of chemical solubility, sediment adsorption, and bioconcentration; Fe, Mn, Sc, and Th distributions are mainly a function of sediment adsorption-precipitation reactions. Metals in dissolved form are diluted, mixed, and flushed through the Bay and, therefore, their effects are short-lived. Metals in particulate form, however, have a longer residence time in the Bay and can build up to high concentrations through bioaccumulation and sediment adsorption.

The relevant management practice is to monitor and control metals discharge while taking into consideration the different solubilities, bioavailability, and adsorption properties of the different metals. Through consideration and understanding of these properties, one can better regulate the type, amount, and location of allowed discharges. As an example, dissolved metals are readily taken up by plankton, whereas particulate metals are likely consumed by suspension feeders or benthic filter feeders. Adverse effects, however, will vary with the chemistry of the metal and the response of the organism to the metals.

3. Concentrations of As, Cd, Cu, Hg, Ni, Pb, Sn, and Zn per gram of suspended material are maximal in near-surface suspended material of the central Bay. Enrichment factors range: Cd, 10-118; Cu, 12-27; Pb, 37-51; and Zn, 16-74. The percentage of organic matter in this zone is generally higher than elsewhere.

The association of a relatively high content of metals with organic matter in the same zone suggests that biological activity is the proximal cause of accumulation. The metals can be derived from multiple sources, natural or anthropogenic.

Control of bioaccumulations can be affected by changes in water quality that will reduce productivity. These changes include lower light, increased turbidity, lower nutrient input, and reduced mixing. However, some biota, such as phytoplankton, require certain metals, like Mn for photosynthesis. Other metals such as cupric ions, with extreme reactivities, interfere with uptake of essential metals. Because metals, sediments, and nutrients are interrelated, they need to be managed together. Piecemeal management of single components cannot succeed.

Most control measures have focused on near-field discharges and immediate effects. There is a need to manage for subtle changes and "far-field" effects. Processes leading to bioaccumulation and particle concentration in the turbidity maximum need to be taken into account in any effective management plan. Moreover, water, particulates, sediments, and biota should be managed as a dynamic system in which trace metals are continually being repartitioned.

4. Secondary maxima of Cd, Mn, Ni, Pb, Sn, and Zn concentrations per gram of suspended material are found in near-surface water of the Bay off the Patapsco River.

These secondary "hot spots" suggest that metals are derived in part from the Patapsco River and Baltimore Harbor via near-surface currents or, for another part, by periodic resuspension from old dredged material on the Bay floor.

The relevant management practice is to stabilize potential sources of contaminated sediment from the Harbor either by removing future dredged material from the system or by stabilizing the natural sediment through consolidation, dewatering, or grass cover.

5. Sediments from the northernmost part of the Bay floor are enriched relative to average crustal shale in Cd, Co, Cu, Mn, Ni, Pb, and Zn by factors of two to eight. Cd, Pb, and Zn are enriched throughout the main Bay by factors of two to six relative to average shale.

The Susquehanna River is a distinctive primary source of metals in bed sediments of the northernmost Bay. This is confirmed by similar enrichment factors and similar metal-Fe ratios in the river and northern Bay. The metals are sequestered in fine sediment and associated with river-borne organic material. Since enrichment factors diminish markedly with distance seaward from Kent Island, contaminated sediment is probably not transported seaward of the Patapsco mouth in quantity. This assumes diagenetic processes are not contributing significantly to the seaward reduction of enrichment. Instead, metals mainly accumulate in the turbidity maximum zone where suspended sediment is trapped. Once deposited, the metals can be resolubilized and, thus, released from contaminated sediment and potentially available to the biota.

Because the Bay system is complex, it requires a fairly sophisticated input of technical information about the system being managed. It should be managed with a scientific data base and a knowledge of processes affecting behavior transport and fate of potential toxics. Therefore, effective management decisions should be coupled to monitoring data and scientific knowledge of processes.

The new information on distribution of enriched bed sediment provides data with which to broadly classify potential dredged material. Such a classification provides input for decisions on dredged spoil management -- its best use, disposal techniques, or dumping sites.

6. The Bay floor is a major sink for metals and organic compounds. More than 60 percent of the total input of Fe, Mn, Ni, Pb, and Zn is retained in the bed sediments.

Bed sediments in the central and northern Bay are enriched with metals, (Cu, Pb, Zn) to depths of 14 to 26 cm, representing about 60 to 90 years of deposition. Metal enrichment reaches a peak between four and 18 cm (1930 and 1960) and diminishes toward the surface.

The enriched metal peaks in the northern Bay probably represent peak metal loading from a dominant source, the Susquehanna. The influx was first felt in the northern Bay and later in the central Bay. Zones of fast sedimentation are sensitive to contamination. When metals are buried deeper than the zone of active diagenesis, they may be effectively immobilized and thus unavailable to biota.

Since sediments record long-term changes in metal loading, they can provide an indication of future trends if the depositional flux is coupled to the input flux. Whereas analyses of water samples from contaminated zones may not detect some toxic chemicals in small amounts, sediments with toxic substances that are strongly sorbed can build up to levels and thus be readily detected.

7. Major transport pathways for metals follow either a hydrodynamic route or a bioecologic route. The principal sinks for toxics are located in near-source zones where fine sediment accumulates.

The hydrodynamic route through the northern Bay follows the pattern of estuarine circulation; that is, seaward through the river and upper estuarine layer, and landward through the lower layer. This route leads to entrapment of contaminated sediment near the inner limit of salty water close to its major source the Susquehanna River. Secondary sinks of accumulation occur in less energetic zones: the central Bay axial basin and inner reaches and mouths of tributaries that promote moderate to fast sedimentation and accumulation of fine sediment.

8. More than 300 organic compounds were detected in Bay sediments. Most were PNAs having anthropogenic sources, and many compounds are among EPA's priority pollutants.

The organic compounds tend to associate with fine suspended material in the water and accumulate on the Bay floor as the suspended material settles. Because of their polarity, some organic compounds may occur in dissolved form, but they are below the detection limit of most present-day instrumentation. Significant concentrations of priority pollutants are cause for concern about sources and effects on Bay ecology.

9. Concentrations of organic compounds in bed sediment are greatest in the northern Bay. Seaward from the Patapsco River, concentrations decrease to the Potomac River mouth. In the southern Bay, concentrations near tributary mouths are greater than elsewhere.

The Susquehanna River is a source of many organic compounds. The compounds are likely supplied from pollution sources and atmospheric deposition on the drainage basin, and they accumulate in the turbidity maximum zone where fine sediment is trapped. Accumulation at tributary mouths relates either to the accumulation of fine sediment or to sources of contamination in the tributaries.

If contaminates have distinctive point sources as industrial discharges they should be controlled pursuant to Federal and state policy.

10. Concentrations of organic compounds are higher and more variable in the Patapsco River than in the main Bay.

A Patapsco River source of organic compounds is indicated by the distribution of concentrations that are high in landward parts of the river. Additionally, they vary as the location of sources varies within the river. Most PNAs, however, are widespread, mixed, and lack specific sources. Part of the contaminated sediment is trapped within Baltimore Harbor and the Patapsco River, but some escapes to the Bay. This is revealed by the occurrence of a Patapsco derived compound, 6-phenylodecane, in the main Bay. Since concentrations diminish seaward from the river mouth and down Bay, dispersion of significant quantities is probably low.

11. More than 120 organic compounds were detected in oysters from the Bay. The compounds, methyl esters, fatty acids, and ketones, were present in most oysters, but PNA's were scarce.

The organic compounds in oysters may have a biogenic or natural origin. Because the composition in oysters differs from sediments, and has fewer PNAs, oysters are of lesser importance for general monitoring of organic compounds in the Bay. The oyster, however, can be useful for monitoring specific PNA compounds as benzo(a)pyyrene which is a suggested carcinogenic compound or an oyster metabolite.

12. Bay-wide bioassays reveal that sediments from inner reaches of the Patapsco and Elizabeth Rivers and from the northern-most Bay have a higher toxicity than elsewhere.

Effluent bioassays of fish, invertebrates, and bacteria indicate that 20 to 50 percent of the effluents sampled had moderate to high toxicity.

The occurrence of relatively high toxicity and low survival rate generally relates to zones of high metal content and high organic compounds in bed sediments close to major sources. We speculate that high sediment toxicity is produced by a combination of high metal content and high loads of organic compounds. It remains to be determined what acceptable levels of sediment pollution the Bay resources can endure. Generally, a greater risk of toxicity in the Bay is associated with high effluent toxicity, unless organisms can adapt to certain concentration levels.

SECTION 8

RESEARCH NEEDS

Chesapeake Bay is a very complex estuarine system, and our knowledge of hydrodynamic, sedimentological, and bio-ecological processes is limited. The data gained in this study point to gaps in our knowledge that deserve future research.

1. Inasmuch as results show that some sediment-associated toxicants occur outside major harbors (the Patapsco River and Hampton Roads) and seaward of Kent Island, it remains to be determined how much material presently escapes the harbors and northern-most Bay. Is the contaminated sediment outside the harbors a product of disposal activities or presently escaping near-source contamination zones? Do harbor contaminates contribute to up-Bay, or up-tributary, contamination zones by landward transport?

2. Since results show maximal particulate concentrations of abnormally high Cd, Cu, Pb, and Zn in surface waters of the central Bay, a location far from major sources, it remains to be determined how they get there. The distribution of metal in various states (dissolved, colloidal, particulate; organic or inorganic) must be determined together to demonstrate how the metals are partitioned on a seasonal basis. We must learn if metals stimulate production of organic matter like plankton or, by contrast, affect the health of organisms in the central Bay. And, does bio-accumulation and turnover make the metals more or less mobile?

3. Whereas the present research deals mainly with metals and organic compounds supplied to the Bay at more or less normal conditions, episodic events may control their distribution. Floods, hurricanes, and storms can produce exceptional conditions for massive resuspension and dispersal of sediment-borne metals. Observations are needed to study the impact of short-term events with respect to the following: How much sediment and toxicant are released or mobilized by an event compared to average conditions? What are the corresponding effects on marine resources? How long does it take to recover, decontaminate, or come to a new chemical equilibrium?

4. Synthesis results reveal that atmospheric inputs of potentially toxic material can compose a significant portion of the total toxic load. It appears that atmospheric inputs are relatively important in areas far from contamination sources, especially for metals like Cd, Cu, and Pb, and the organic compound like PNAs. We must determine, in detail, the magnitude and extent of atmospheric inputs relative to water-borne inputs. With increasing use of fossil fuels, are atmospheric imputs increasing the total toxicant input to the Bay despite controls on water-borne inputs? There is a need to determine if atmospheric inputs are from distant sources and homogeneous, affecting the entire Bay. Because atmospheric dry and wetfall collects on salt marshes, and the flux can be recorded by marsh deposits, attention should focus on high marsh sediments that reflect atmospheric influence. The historical record combined with monitoring should provide an early warning of increasing anthropogenic inputs from the atmosphere.

5. To ascertain the validity of data acquired, future efforts should account for variability of field sampling through a rigorous statistical sampling plan. This study reveals that the concentrations of metals and organic compounds can vary widely with location, especially in suspended material. Verifying results are needed to account for short-term tidal variations; fortnightly, neap-spring changes; and seasonal as well as non-periodic changes of episodic events.

6. Chesapeake Bay has, at least on one occasion, been the recipient of the direct disposal of pesticides like Kepone (Huggett et al. 1980). Fortunately, the quantities were small and the assimilation capacity large enough so that no adverse effects on the biota were noted. The disposal of such compounds in this manner was, and is, illegal. This indicates that laws alone are insufficient to protect the Bay and that chemical monitoring is necessary. The chemical monitoring of effluents and sediments collected near outfalls shows that more effort of this type is needed to prevent future "Kepone episodes" (Bieri et al. 1981). Key sinks in the Bay also require monitoring. Because some dissolved toxicants are difficult to detect in near-source zones, monitoring of peripheral sediment sinks having fast deposition can provide an early warning of increased loading. (For details see separate Monitoring Recommendations, Flemmer et al., unpublished)

In this study over 300 organic compounds were analyzed, but results indicate that "thousands of other compounds are present at low concentrations." Therefore, monitoring needs to account for a wide compositional range of organic compounds having low concentrations. These data are needed to establish valid baselines as well as to detect anomalous concentrations of pollutants before they build up. To guide State water pollution control authorities, an effluent toxicity characterization program is needed to screen industrial effluents for toxic chemicals and to determine their degree of toxicity, both acute and chronic.

7. Additional toxicity data are needed to evaluate impacts on the Bay's living resources and to formulate diagnostic criteria that are generally accepted. Little is known about the toxicity of individual components, and less is known about the toxicity of populations or communities. Most bioassays have examined acute effects; little is known about long-term chronic effects. Moreover, the Bay ecosystem is complex and dynamic, involving the interactions of physio-chemical parameters and biological components with time. We need to know if the toxicants found in the Bay are biologically available. Once organisms are exposed to toxicants, can they adapt to certain concentration levels? Most bioconcentrations have been treated as static levels in tissues of organisms. Some organisms, however, accumulate toxicants quickly, whereas others that metabolize slowly can accumulate toxicants slowly but to high levels. Therefore, bioaccumulation needs to be examined as a dynamic equilbrium determined by the metabolism rate.

8. A major problem for future research is determining the relative capacities of different parts of the Bay to assimilate toxicants. Although a numerical model can predict the distribution and resulting concentration of a given input and its residence time, toxicants are subject to

transformation and building up through biological and sedimentological processes. A single concentration level value applied to the entire Bay is not a universally valid criteria for control because it does not take into account the characteristics of the receiving segment. We need to know the relationship between the contaminate concentrations and their toxic effect on the biota in each receiving segment. This requires much better data and a greater understanding than now exists. In particular, we need to overcome the difficulties of: (1) making accurate measurements of diverse and potentially toxic compounds at very low concentrations; (2) measuring the toxicity effects of chemicals on organisms; and (3) making valid interpretations by comparing laboratory results and field observations.

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APPENDIX A

INVENTORY OF PROJECT DATA DISCUSSED IN THIS REPORT

Project Name	Organization/ Principal Investigator(s)	Sample Collection Dates	Spatial Description	Sample Type	Parameters	Analytical Methods
Toxic Point Source Assessment of Industrial Discharges to the Chesapeake Bay Basin	Monsanto Research Corp./ Rawlings	April, 1978-May, 1981	54 Point sources located near Baltimore and Hampton/ Richmond	Effluent	Bioassays, NPDES Parameters, metals organics	ICAP, GC, GC/MS etc.
The Characterization of the Chesapeake Bay: Systematic Analysis of Toxic Trace Elements	National Bureau of Standards/ Kingston	June-July, 1979	51 Stations* Bay-wide	Near bottom & surface Water Column, Suspended and Dissolved	Metals	NAA, GFASS
Chesapeake Bay Sediment Trace Metals	University of Maryland/Helz	April, 1977-May, 1979	200 Surface Bed Sediment Samples Collected from 25 Traverses, Bay-wide	Dried sediment, Fine and Coarse Fractions	Metals	AAS
Chesapeake Bay Sediment Trace Metals	University of Maryland/Helz	Nov., 1978, May-June, 1979	17 Gravity Sediment Cores, Bay-wide, sampled every 2 cm.	Dried sediment	Metals	DCP
Chesapeake Bay Earth Science Study - Interstitial Water Chemistry	Maryland Geological Survey/ Bricker & Hill	Sept.-Oct., 1978, June-July, 1979, March, 1980	1 Meter Gravity* cores taken at 97 stations, Bay-wide	Pore Water samples at 10 depth intervals	Metals, pH, Eh, $pS^{=}$, conductivity, SiO_2	Electrode measurements, AAS

Project Name	Organization/ Principal Investigator(s)	Sample Collection Dates	Spatial Description	Sample Type	Parameters	Analytical Methods
Sediment Pore Water Chemistry	College of William & Mary/ Tyree	Sept.-Oct., 1978, June-July, 1979, March, 1980	1 Meter Gravity* cores taken at 97 stations, Bay-wide	Pore Water samples at 10 depth intervals	Major ions and nutrients	Ion chromatography
Investigation of Organic Pollutants in the Chesapeake Bay	Virginia Institute of Marine Science/Huggett & Beri	Spring and Fall, 1979	27 Stations, Bay wide	Bed sediment	Organic Compounds	GC, GC/MS
Investigation of Organic Pollutants in the Chesapeake Bay	Virginia Institute of Marine Science/Huggett	Spring & Fall, 1979	23 Stations Bay wide	Oyster tissue	Organic Compounds	GC, GC/MS
Chesapeake Bay Earth Science Study - Animal Sediment Relationship	Maryland Geological Survey/ Reinharz	Sept.-Nov., 1978, June-July, 1979	52 Stations* in northern Bay	Box cores, samples at 7 depth intervals	Biotic assessment, radiographs, grain size, salinity, temp.	
The Biogenic Structure of Chesapeake Bay Sediments	Virginia Institute of Marine Science/Boesch & Diaz	Sept., 1978, April, 1979, June, 1979	50 stations* in Bay below Md./Va. state line	Box cores, sampled at 7 depth intervals	Biotic assessment radiographs, grain size, salinity, temp.	
Fate, Transport & Transformation of Toxics: Significance of Suspended Sediment and Fluid Mud	Virginia Institute of Marine Science/Nichols	March-April, May, August, 1979; April May, 1980	Bay longitudinal and harbor transects	Water column and fluid mud/bed sediment	Metals, temp., salinity, D.O., pH, solids, organic carbon, particle size	AAS, GFASS, etc.
Baseline Sediment Studies to Determine Distribution, Physical Properties, Sedimentation Budgets & Rates	Virginia Institute of Marine Science/Byrne	Nov. 1978-June 1979	2,018 stations in southern Bay (1.4 km grid)	Bed sediment	Particle size, % water, carbon, sulfur, sedimentation rate	

Project Name	Organization/ Principal Investigator(s)	Sample Collection Dates	Spatial Description	Sample Type	Parameters	Analytical Methods
Chesapeake Bay Earth Science Study- Sedimentology of the Chesapeake Bay	Maryland Geological Survey/ Kerhin	April, 1976- Oct. 1981	4,200 stations in northern Bay	Bed sediment (upper 8-10 cm)	Particle size, % water, carbon, sulfur, sedimentation rate	

* Samples for several projects collected at same locations.

NAA - Neutron Activation Analysis
GFASS - Graphite Furnace Atomic Absorption Spectroscopy
ASS - Atomic Absorption Spectroscopy
DCP - Direct Current Argon Plasma Emission Spectroscopy
GC - Gas Chromatograph
GC/MS - Gas Chromatograph/Mass Spectroscopy
ICAP - Inductively Coupled Argon Plasma

APPENDIX B

SUMMARY OF DATA SOUCES FOR TRACE METALS IN THE CHESAPEAKE BAY AND TRIBUTARIES

Area	Reference	Metals	Component
James, York, & Rappahannock Rivers	Huggett et al. (1971)	Hg	Bed Sediments
Potomac River	Pheiffer (1972)	Ag,Ba,Cd,Co,Cr, Cu,Fe,Li,Mn,Ni, Pb,Sr,V,Zn	Bed Sediments
James River	Huggett & Bender (1975)	Cu,Zn	Oysters & Bed Sediments
Northern Bay (1974)	Owens et al.	B,Ba,Ce,Cr,Mn,V Zn,Zr	Bed Sediments
Patapsco River & Balto. Harbor	Villa & Johnson (1974)	Cd,Cr,Cu,Hg,Mn,Ni Pb,Zn	Bed Sediments
Northern Bay (1974)	Sommer & Pyzik	Co,Cu,Ni,Pb,V	Bed Sediments
Northern Bay	Cronin (1974)	Fe,Mn,Zn	Bed Sediments
Northern Bay & Susquehanna	Carpenter	Co,Cr,Cu,Fe,Mn,Ni, (1975) Zn,Cd,Pb	Dissolved and Suspended Sediment
Central Bay	Matisoff (1975)	Fe,Mn	Interstitial Sediment & Water
Back River	Helz et al. (1975)	Cd,Cu,Fe,Mn,Pb,Zn	Bed Sediments
Northern Bay	Helz (1975)	Cd,Co,Cr,Cu,Pb,Fe Mn,Ni,Zn	Bed Sediments
Rappahannock River	Huggett et al. (1975)	Cu,Zn	Bed Sediments

(APPENDIX B, CONTINUED)

Area	Reference	Metals	Component
Northern Bay	Matisoff et al. (1975)		
Rhode River	Frazier (1976)	Cd,Cu,Fe,Mn,Zn	Bed Sediments
Elizabeth River	Johnson & Villa (1976)	Cd,Cr,Cu,Hg,Pb,Zn	Bed Sediments
Patuxent River	Ferri (1977)	Cd,Co,Cr,Cu,Fe, Mn,Ni,Pb,Zn	Bed Sediments
Northern Bay	Schubel and Hirschberg (1977)	Cr,Cu,Ni,Pb	Bed Sediments
Patapsco River & Balto.Harbor	EPA-440/5-77-015A	As,Cd,Cr,Cu,Hg, Mn,Ni,Pb,Zn	Bed Sediments
Northern Bay	Goldberg et al. (1978)	Ag,Al,Cd,Co,Cr,Cu, Fe,Mn,Ni,Pb,Zn,V	Bed Sediments
Northern Bay	Eaton et al. (1979)	Mn	Dissolved Bed Sediments
Northern Bay	Eaton (1980)	Fe,Ti,Zn	Suspended Sediments

APPENDIX C.

SUMMARY OF DATA FOR ORGANIC CHEMICALS
IN CHESAPEAKE BAY AND TRIBUTARIES

Area	Reference	Organic Chemicals	Component
Chesapeake Bay & Selected Tribs.	Munson & Huggett (1972)	DDT compounds	Oysters
James, Rappahannock, & Potomac Rivers	Barnard (1971)	DDT compounds	Fish
Chester River	Munson (1973)	PCBs, Chloradane, DDT	Sediments Shellfish
Northern Bay	Munson (1975)	PCBs Chloradane DDT	Sediments Shellfish Zooplankton
Cape Charles, Lynnhaven Bay	Goldberg et al. (1978)	PCBs DDT compounds PNAs, DAHs	Oysters
James River	U.S. EPA (1978)	Kepone	Soil, water, Bed sediments
James River	Huggett (1980)	Kepone	Bed sediments & biota
James River	Huggett & Bender (1980)	Kepone	Biota, Bed sediments, Suspended sediments
James River	Lunsford (1980)	Kepone	Bed sediments
James River	Nichols & Gutshall (1981)	Kepone	Bed sediments

APPENDIX D

AREAL DISTRIBUTION OF SEDIMENT TYPE IN CHESAPEAKE BAY: FROM DATA OF KERHIN ET AL. (1982) AND BYRNE ET AL. (1982)

	Sand		Silt		Clay		Sand-Silt-Clay		Sandy Clay		Silty Clay		Clayey Silt		Sandy Silt		Silty Sand		Clayey Sand		Total Area
Bay Segment	km^2	%[1]	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	
Northern	218.6	6.1	-	-	3.8	2.9	78.2	26.2	-	-	405.8	34.3	15.9	5.3	-	-	7.8	1.5	8.8	7.6	738.9
Upper Middle	630.9	17.5	-	-	126.3	97.1	61.1	20.4	2.5	100	447.5	37.8	52.4	17.4	-	-	41.8	7.8	32.9	28.4	1395.4
Lower Middle MD[3]	590.7	16.4	-	-	-	-	57.3	19.2	-	-	282.0	23.8			4.5	4.7	-	-	53.4	46.0	987.9
VA	820.8	22.8	3.1	100	-	-	49.0	16.4	-	-	49.0	4.1	138.4	46.0	26.3	27.5	44.8	8.4	13.5	11.6	1144.9
Upper Southern	211.9	5.9	-	-	-	-	22.9	7.7	-	-	-	-	57.0	18.9	49.3	51.5	160.5	30.0	5.0	4.3	506.6
Central Southern	262.9	7.3	-	-	-	-	11.2	3.7	-	-	-	-	9.9	3.2	9.1	9.5	138.3	25.9	2.4	2.1	433.8
Lower Southern	864.9	24.0	-	-	-	-	19.2	6.4	-	-	-	-	27.5	9.1	6.5	6.8	140.6	26.3	-	-	1058.7
Sediment Type Totals	3600.7	57.4[2]	3.1	0.1	130.1	2.1	298.9	4.8	2.5	0.1	1184.3	18.9	301.1	4.8	95.7	1.5	533.8	8.5	116.0	1.8	6266.2

* Exclusive of Choptank River and Eastern Bay and Tangier and Pocomoke Sounds in Maryland.
Exclusive of Pocomoke and Tangier Sounds, Mobjack Bay, and lower tributary areas of Rappahannock, Piankatank, York, and James Rivers.

[1] Percent of area for given sediment type by segment.

[2] Percent of area for given sediment type for entire Bay.

[3] Dividing line between Maryland and Virginia taken as latitude 37°55' (Smith Point).

APPENDIX E

SUMMARY OF CHESAPEAKE BAY TOXIC SOURCE ASSESSMENT AND BIOASSAY TESTS

Plant Name	S/C Code	Industry Type	Date of Sampl.	Fish Bioassay: Minnow 96 Hr. LC50	Fish Bioassay: Sheepshead 96 Hr. LC50	Invertebrate Bioassay: Daphnia LC50 48NC	Invertebrate Bioassay: Mysid 96 Hr. LC50	Algal Bioassay: Selenastrum EC50 4 day	Algal Bioassay: Selenastrum SC20 4 day	Algal Bioassay: SC20 4 day	Algal Bioassay: EC50 14 day	Algal Bioassay: Skeletonoma 96 Hr. EC50	Algal Bioassay: Skeletonoma 96 Hr. SC20	Marine Bacter. Microtox	Seagrass Thalassia	Cytotoxic Effect: AMES	Cytotoxic Effect: CHO
Pre-Phase I																	
1. Bethlehem Steel	3312	steel industry (021)	4/14/78				NT					38.4% 41.5%	NS				
2. Bethlehem Steel	3312	steel industry (1010)	4/14/78				100% (100% mortality)					30.5%	NS		NT		
3. Bethlehem Steel		steel industry (112B)	4/14/78				100%					NT	NS		NT		
4. E & R Creosote	2491	Wood Preserving	4/14/78				--					NT	NS		NT		
5. Va. Chemicals	2811 2819 2869	Chemical Manufacturer	4/27/78									NT	2.3%				
6. Chesapeake Corp.	2611 2621 2631	Pulp & Paper	5/3/78				NT					3.7%	NS				
7. Amoco	2911	Petroleum Refining	5/3/78				100% (100% mortality)					45%	7.5%				
8. Badische Corp.	2824 2299	Synthetic Fibers	7/7/79				50% (100% mortality)					8.2% 6.8%	NS		NT		
9. Allied Chemical	2821 2824	Synthetic Resins & Fibers	7/7/78				NT					NT	16.8%		NT		

PLANT NAME	S/C Code	INDUSTRY TYPE	Date of Sampl.	FISH BIOASSAY: MIN-NOW 96 Hr LC50	FISH BIOASSAY: SHEEPS-HEAD 96 Hr LC50	INVERTEBRATE BIOASSAY: DAPHNIA LC50 48NC	INVERTEBRATE BIOASSAY: MYSID 96 Hr LC50	ALGAL BIOASSAY: SELENASTRUM EC50 4 day	ALGAL BIOASSAY: SELENASTRUM SC20 4 day	ALGAL BIOASSAY: SC20 4 da	ALGAL BIOASSAY: EC50 14 da	ALGAL BIOASSAY: SKELETONOMA 96 Hr EC50	ALGAL BIOASSAY: SKELETONOMA 96 Hr SC20	MARINE BACTER. MICRO-TOX	SEAGRASS THALASSIA	MUTAGENIC and CYTOTOXIC EFFECTS: AMES	MUTAGENIC and CYTOTOXIC EFFECTS: CHO
10. Allied Chemical (cooling water)	2821 2824	Synthetic Resins & Fibers	8/5/18				NT					NT	NS		NT		
11. Hopewell, STP	4952	POTW, Secondary	9/28/78				50%					13.5% 7.5%	NS				
Phase I																	
1. Westvaco A100	2621 2631	Pulp & paper	11/26/78	NT		NT,NT	45.7%	62.1%	.04	NS	NT	NT	.07%			No response	Low
2. Badische Corp. A101	2824 2299	Synthetic Fibers	11/30/79	NT, NT	NT	94%, NT NT	NT 44.9%	45.6	1.96	NS,NS	11% 14%	18% 19.9%	1%			NR	NT
3. Crompton A102	227	Textile Dyeing	11/21/79	NT		NT, 66.8%	NT	NT	.01	39%	56-100%	NT	.53%			NR	Low
4. Amoco A103	2911	Petroleum Refining	12/1/79	NT	NT	45%, NT	NT,NT	NT	.05	7%	56-100%	11%, NT	.02			NR	NT
5. Aileen A104	221	Textile Dyeing	11/21/79	60%		77%, NT	46.4%	NT	.004	32-56%	NT	53.3%	.64%			NR	Low
6. Va. Fiber A105	2631	Pulp & paper	11/26/79	NT		NT, NT	NT	13.6%	NS	NS	61%	NT	.88%			NR	Low
7. Ches. Corp. A106	2611 2621 2631	Pulp & paper	11/30/79	NT, NT		NT, NT NT, NT	87.9%, NT	28.6% 43.4	.09 .02	NS, 57%	NT,NT	63.5% 42.4%	.73% .49%			NR,NR	Low,low
8. E. I. DuPont A107	2824 2299	Synthetic Resins & Fibers	11/20/79	NT		NT, NT	NT	NT	8%	8%	NT	NT	.04%			NR	NT
9. VOTAN A108	3111	Leather Tanning	11/20/79	60%		22%,44.4%		5.1	1	13%	32-56%	29.8%	1.03%			Positive	Low
10. Va. Chemicals A109	2811 2819 2869	Inorganic & Organic Chemicals	11/27/79	NT		6%,9.6%	NT	81.5	.4	NS	56%	44.3%	20%			NR	NT

PLANT NAME	S/C Code	INDUSTRY TYPE	Date of Sampl.	FISH BIOASSAY MIN-NOW 96 Hr LC50	FISH BIOASSAY SHEEPS-HEAD 96 Hr LC50	INVERTEBRATE BIOASSAY DAPHNIA LC50 48NC	INVERTEBRATE BIOASSAY MYSID 96 Hr LC50	ALGAL BIOASSAY SELENASTRUM EC50 4 day	ALGAL BIOASSAY SELENASTRUM SC20 4 day	ALGAL BIOASSAY SC20 4 da	ALGAL BIOASSAY EC50 14 da	ALGAL BIOASSAY SKELETONOMA 96 Hr EC50	ALGAL BIOASSAY SKELETONOMA 96 Hr SC20	MARINE BACTER. MICRO-TOX	SEAGRASS THALASSIA	MUTAGENIC and CYTOTOXIC EFFECTS AMES	MUTAGENIC and CYTOTOXIC EFFECTS CHO
Phase II																	
1. FMC Corp. B1295	2821 2879	Pesticides & Plasticides				1%	.69%	5.6%	.1%			3.57%	.1%				
2. Bethlehem Steel B1305	3312	Iron & Steel				NT	50.5%	NT	NS			53%	.32%				
3. Bethlehem Steel (Back Rvr. POTW) B1315	4952	Plant Intake Water				NT	27%	NT	.1%								
4. Bethlehem Steel B1365	3312	Coke Prod.				30.8%	NT	62%	.1%			49.5%	1.4%				
5. SCM Adrian Joyce B1435	2816	Inorganic Pigment				51.2%	13.3%	3.3%	NS								
Phase III																	
1. Badische A101	2824 2299	Synthetic Fibers	4/16/81	NT										NT			
2. Va. Chemicals A109	2811 2819 2869	Inorg. & Organ. Chem.	4/23/81	NT			24%							NT			
3. Atlantic Wood Preservers B112	2491	Wood Preserving	4/20/81	80%										30%			
4. Amer. Tobacco B111	2621	Tobacco Processing	4/9/81				NT										

PLANT NAME	S/C Code	INDUSTRY TYPE	Date of Sampl.	FISH BIOASSAY MIN-NOW 96 Hr LC50	FISH BIOASSAY SHEEPS-HEAD 96 Hr LC50	INVERTEBRATE BIOASSAY DAPHNIA LC50 48NC	INVERTEBRATE BIOASSAY MYSID 96 Hr LC50	ALGAL BIOASSAY SELENASTRUM EC50 4 day	SELENASTRUM SC20 4 day	SC20 4 da	EC50 14 da	SKELETONOMA 96 Hr EC50	SKELETONOMA 96 Hr SC20	MARINE BACTER. MICRO-TOX	SEAGRASS THALASSIA	MUTAGENIC and CYTOTOXIC EFFECTS AMES	MUTAGENIC and CYTOTOXIC EFFECTS CHO
5. E. I. DuPont - Spruance B113	2821 2829	Synthetic Resins & Fibers	4/7/81	NT										NT			
6. ICI Americas B117	3079	Poly-ester Film & Plastics	4/8/81	17.5%			NT							12%			
7. Phillip Morris B1240	2111	Tobacco Prods.	4/8/81				59%										
8. HRSD - Western Branch C150	4952	POTW	4/27/81	48%										5%			
9. VEPCO-Dutch Corp. C151		Elec-tric Power	4/13/81				56%										
0. Smith-Douglas Div. Borden Chem. C1530		Phos-phate Fert.	4/21/81		NT									NT			
11. Royster Co. C1540		Ammonia Fert.	4/30/81				5%										
12. HRSD - Boat Harbor 1550	4952	POTW	4/22/81				15%										
13. HRSD - Lamberts Point 1560	4952	POTW	4/28/81	24%										.7%			
14. Rich-mond STP 1580	4952	POTW	4/13/81				87%										

PLANT NAME	S/C Code	INDUSTRY TYPE	Date of Sampl.	FISH BIOASSAY MIN-NOW 96 Hr LC50	FISH BIOASSAY SHEEPS-HEAD 96 Hr LC50	INVERTEBRATE BIOASSAY DAPHNIA LC50 48NC	INVERTEBRATE BIOASSAY MYSID 96 Hr LC50	ALGAL BIOASSAY SELENASTRUM EC50 4 day	ALGAL BIOASSAY SELENASTRUM SC20 4 day	ALGAL BIOASSAY SC20 4 da	ALGAL BIOASSAY EC50 14 da	ALGAL BIOASSAY SKELETONOMA 96 Hr EC50	ALGAL BIOASSAY SKELETONOMA 96 Hr SC20	MARINE BACTER. MICRO-TOX	SEAGRASS THALASSIA	MUTAGENIC and CYTOTOXIC EFFECTS AMES	MUTAGENIC and CYTOTOXIC EFFECTS CHO
15. Smith-field Pack. C159		Meat Procesa-ing	5/4/81	42%										5%			
16. Sheller Globe C160		Fabri-cated Rubber	4/15/81	NT										NT			
17. Ports-mouth STP C161	4952	POTW	4/29/81	97%			10%										
18. Hope-well Reg-ional, STP C163		POTW	4/9/81	59%										56%			
19. HRSD, Williams-burg C164		POTW	4/14/81	7%										2%			
20. Amer. Recovery B142S	4963	Waste Neutral-ization					4%										
21. Allied Chem. C157	2821 2824		4/14/81				77%										
22. Patap-sco, STP B141S	4952	POTW					3%										
23. Tenne-cox B133	2821	Plasti-cizers					41%										
24. SCM-Adrian Joyce B1435	2816	Inorg. Chems.					22%										
25. Allied Chem. C1695	2879						12%										
26. SCM - St.Helena B126S	2816	Inorg. Oxides					21%										

PLANT NAME	S/C Code	INDUSTRY TYPE	Date of Sampl.	FISH BIOASSAY MIN-NOW 96 Hr LC50	FISH BIOASSAY SHEEPS-HEAD 96 Hr LC50	INVERTEBRATE BIOASSAY DAPHNIA LC50 48NC	INVERTEBRATE BIOASSAY MYSID 96 Hr LC50	ALGAL BIOASSAY SELENASTRUM EC50 4 day	SELENASTRUM SC20 4 day	SC20 4 da	EC50 14 da	SKELETONOMA 96 Hr EC50	SKELETONOMA 96 Hr SC20	MARINE BACTER.* MICRO-TOX	SEAGRASS THALASSIA	MUTAGENIC and CYTOTOXIC EFFECTS AMES	CHO
27. Armco B1475	3312	Stainless Steel Mfg.					54%										
28. Eastern Stainless Steel B1495		Stainless Steel Mfg.					21%										

APPENDIX F

RESULTS OF FISH BIOASSAYS FOR EFFLUENT SAMPLES BY SPECIES

Toxicity Index	Fathead Minnow	Sheepshead Minnow	Totals
Minimal	14	3	17
75-NT[2]*			
Low	3		3
Moderate	2		2
25-49			
High	3		3
0-24			
Totals	22	3	25

*[2] NT is not toxic; a 100% effluent concentration did not kill 50% of the test species.

APPENDIX G.

RESULTS OF INVERTEBRATE BIOASSAYS FOR EFFLUENT SAMPLES BY SPECIES

Toxicity Index	Daphnia (Magna)	Mysid Shrimp	Total
Minimal	9	18	27
75-NT[2]*			
Low	2	8	10
50-74			
Moderate	2	4	6
25-49			
High	2	11	13
Totals	15	41	56

*NT[2] is not toxic; a 100% effluent concentration did not kill at least 50% of the test species.

APPENDIX H.

RESULTS OF BACTERIAL AND GRASS BIOASSAYS

Toxicity Index	Microtox (Marine Bacteria)	Thalassia (Sea Grass)
Minimal 75-NT	5	6
Low 50-74	1	
Moderate 25-49	1	
High	5	—
	12	6

APPENDIX I.

RESULTS OF SALMONELLA/MICROSOMAL ASSAYS FOR MUTOGENICITY OF CHESAPEAKE BAY EFFLUENT SAMPLES

Plant Number/Sample	Spot Test	Plate Incorporation
Filtrate I*		
A101	(-) negative	(-) negative
A102	(-) negative	(-) negative
A103	(-) negative	(-) negative
A104	(-) inconclusive	(-) negative
A106	(-) negative	(-) negative
A107	(-) negative	(-) negative
A108	(-) negative	(+) positive
A109	(-) negative	(-) negative
Filtrate II**		
A100	0	-
A105	-	-
A110	-	-
Particulate *** - Acetone Extract		
A100	Not performed	negative
A101	Not performed	negative
A102	Not performed	negative
A103	Not performed	negative
A104	Not performed	negative
A105	Not performed	negative
A106	Not performed	negative
A107	Not performed	negative
A108	Not performed	negative
A109	Not performed	negative
A110	Not performed	negative

* Filtrate I - Filtrate from initial filtering through a .45 u filter.
** Filtrate II - Filtrate I passed through a 0.2 u filter.
*** Particulate - Material retained on polyester drain disc and a 5 u teflon filter.

APPENDIX J.

RESULTS OF MAMMALIAN CELL
CLONAL ACUTE CYTOTOXICITY ASSAY

	Neat Effluents Sterilized by Antibiotic Addition		Filtered Sterilized Effluents		Particulate Extract, Acetone Concentrate	
Sample Number	EC_{50},[a] pL/mL	Toxicity rating	EC_{50} pL/mL	Toxicity rating	EC_{50},[b] pL/mL	Toxicity rating
A100	150	L[c]	ND[d]		600	L
A101	ND		ND		ND	
A102	C[e]		200	L	ND	
A103	ND		200	L	700	VL[f]
A104	ND		250	L	ND	
A105	25	M[g]	ND		ND	
A106	45	M	200	L	300	L
A107	ND		ND		650	VL
A108	C		200		700	VL
A109	C		ND		ND	
A110	55	M	200		300	L

[a]Effective concentration at 50% killing

[b]Normalized to toxicity of particulate extracts recovered from 1,000 mL of neat sample.

[c]Low, 60-600 pL/mL.

[d]No toxicity found at highest concentration tested and with no contamination.

[e]Microbial contamination; toxicity not determined.

[f]Very low, 600PL/mL

[g]Moderate, 6-60PL/mL

PART IV

SUBMERGED AQUATIC VEGETATION

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INTRODUCTION

This part of the CBP's scientific synthesis summarizes and integrates almost three years of research on the occurrence of submerged aquatic vegetation (SAV) in Chesapeake Bay, the role and value of SAV in the Chesapeake Bay ecosystem, and major factors controlling SAV's past and future survival. The four chapters comstituting the SAV part draw on the findings of over a dozen separate research projects, each of which has produced a final report containing a detailed account of research design, methods, and results. These projects are listed in Appendix A. In addition to CBP-funded research, this part includes information from other research as well as from personal communications.

The CBP included SAV as a critical research area because of its ecological role and value, its precipitous decline during the 1970's, and the urgent need to discover why the grasses were disappearing. The life history of SAV and its decline in Chesapeake Bay have been fully presented in a 1978 Summary of Literature on SAV in Chesapeake Bay (Stevenson and Confer). The papers presented here seek to further clarify the SAV problems presented in the 1978 Summary and to suggest reasons for its decline.

Four features of SAV's role in the Bay -- food source, habitat, nutrient buffer, and sediment trap -- illustrate its ecological importance. As a food source, SAV had a partly documented, partly assumed role in the ecology and economy of Chesapeake Bay. SAV is eaten by ducks, geese, and some fish, and it contributes to the detritus-based food web. SAV also provides habitat for many organisms--nurseries for juvenile stages of some fish species; refuge for molting blue crabs, other invertebrates, and certain fish species; a stable habitat for infauna; a substrate for epiphytic plants and animals; and a habitat for all fauna subsisting directly on SAV and its epiphytes, or the detritus derived from them. Additionally, SAV was thought to buffer nutrients in the Bay by absorbing nutrients from the water column during spring runoff and releasing them in autumn as detritus. SAV was also considered to be a nutrient "pump," taking up nutrients from the sediment through its roots and releasing them as detritus. Other presumed functions of SAV were the baffling of water movement, causing sediment to settle to the bottom, and the binding of sediment, helping to mediate shoreline erosion.

The most compelling evidence for the decline of SAV during the 1970's is an upper-Bay annual summer submerged vegetation survey conducted by the U.S. Fish and Wildlife Service and the Maryland Department of Natural Resources and an aerial mapping survey of the lower Bay SAV conducted by the Virginia Insitute of Marine Science (VIMS). The Maryland survey shows that 28.5 percent of 640 sample stations had SAV in 1971, whereas only 21 percent had SAV in 1972, 10.5 percent in 1973, and 14.9 percent in 1974 (Kerwin et al. 1976). This survey covered the entire Maryland portion of the Bay, except the Potomac River, making it by far the most extensive survey available. The aerial survey of lower Bay SAV shows dramatic declines in the Rappahannock, Piankatank, and York Rivers between 1971 and 1974 (Orth and Gordon 1975). The 1970's decline was especially alarming because it affected all areas and all species, though not all to the same degree, something not observed in previous distribution shifts. The only other documented major perturbations were the decimation of eelgrass (Zostera marina) in the 1930's presumably caused by a disease organism and the outbreak of Eurasion milfoil (Myriophyllum spicatum) in the late 1950's and early 1960's. These changes directly affected only one species.

In our search for reasons for the Bay-wide SAV decline, we assumed that one or more fundamental properties of the Bay ecosystem were being altered. Disease was ruled out because it probably would not have affected all species. Point sources of pollution, although they may have been a contributing cause, were probably not the direct underlying cause because of their localized nature. We conjectured that herbicides from agricultural runoff were directly harming SAV, that sediment loading was increasing turbidity thereby decreasing the amount of light available to SAV, and that nutrient loading to the system was stimulating the growth of phytoplankton, which were further shading the SAV and competing for nutrients. One of the disturbing features of these working hypotheses was that they pointed to a gradual and fundamental change in the Bay, thought to be brought about largely by the increased human activity associated with a population growth of more than 100 percent in the Bay area during the last 40 years.

Following the decision to include SAV as a study area in the CBP, a Plan of Action that set forth the goals and objectives of the study was developed. The study's ultimate goal was to develop a plan for managing the Bay system to maintain SAV as a viable resource. To meet that goal, we conducted basic research on the structure and function of SAV-based ecosystems, including inventories of the biota and observations of ambient, abiotic variables in SAV beds and at nearby sites that were devoid of SAV but otherwise similar. In addition to observations of the natural ecosystem, manipulative studies were designed in the field and laboratory on system dynamics. These manipulative studies aimed at better understanding the role and value of SAV and the factors controlling its growth and survival. This latter information would elucidate causes of the recent decline in SAV, as well as the requirements for future survival. Finally, interpretation of aerial photography and analysis of SAV seeds in Bay bottom cores were to be used to investigate current and past distribution and abundance of SAV. This information would put in historical perspective the magnitude of the current decline and provide a baseline against which to measure future changes.

The following four papers are organized around fundamental questions of interest to someone charged with managing this valuable resource. The first question is: Is there a problem concerning SAV in Chesapeake Bay? To answer this, one first must show that there has been a decline in SAV that is different in character or degree from natural fluctuations. The first paper addresses this point. Second, one must show that SAV has some value and that its loss will have negative ecological and economic impacts: the subject of the second paper. If there is a problem, the next question must be: What caused it? As stated above, we explored various hypotheses about the decline. Separate papers (three and four) are devoted to herbicides and light as they were thought to be the most likely causes. A list of the detailed Management Questions and answers appears at the end of the SAV synthesis.

Technical Glossary

abiotic:	Without life, inorganic.
anoxia:	Total deprivation of oxygen.
biotic:	Of life, or caused by living organisms.
copepod:	Small, sometimes parasitic, crustacea living in either salt or fresh water.
denitrification:	Single-celled organism, mainly marine and often with a cellulose shell.
detritus:	Accumulation of disintegrated material, or debris.
dinoflagellate:	Single-celled organism, mainly marine and often with a cellulose shell.
fluvial:	Of, found in, or produced by a river.
halocline:	A level of marked change, especially increase, in the salinity of seawater at a certain depth.
nonpoint source:	Source of a nutrient or other constituent coming from diffuse areas such as pasture and forests, and atmosphere.
point source:	Source of nutrients or other constituents coming from a distinct source such as a pipe from a sewage treatment plant.
primary producitivity:	The amount of organic matter made in a given time by the autotrophic organisms in an ecosystem.
rotifer:	Microscopic invertebrate animal found mostly in fresh waters, having one or more rings of cilia at the front end to the body.
Secchi depth:	Depth at which a Secchi disk can be seen. The Secchi disk is an instrument for measuring the light attenuation of natural waters.
watershed:	The area drained by a river or river system.

TECHNICAL SYMBOLS

C	-- Celsius
cm	-- centimeter
CW	-- carapace width
d	-- day
g	-- gram
h	-- hour
ha	-- hectare
kg	-- kilogram
km	-- kilometer
L	-- liter
LAI	-- leaf area index
m	-- meter
M	-- molar
ug-at	-- microgram atom
uE	-- micro-Einstein
umoles	-- micro-moles
MLW	-- mean low water
mm	-- milimeter
$NADPH_2$	-- coenzyme reducing carbon dioxide to sugar in photosynthesis
nm	-- nanometer
pMax	-- maximum rate of photosynthesis
ppb	-- parts per billion
ppm	-- parts per million
ppt	-- parts per thousand
sec	-- second
y	-- year

DISTRIBUTION AND ABUNDANCE OF SUBMERGED AQUATIC VEGETATION IN CHESAPEAKE BAY: A SCIENTIFIC SUMMARY

by

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CONTENTS

FIGURES

TABLES

SECTION 1

INTRODUCTION

The Chesapeake Bay, with its extensive littoral zone and broad salinity regime of 0 to 25 ppt, supports many different species of submerged aquatic vegetation (SAV) (Anderson 1972, Stevenson and Confer 1978, Orth et al. 1979). Approximately ten species of submerged vascular plants are abundant in the Bay, with another ten species occurring less frequently. In many areas, more than one species is found in a particular bed of SAV because of the similarity in the physiological tolerances of some species. Between regions of the Bay, salinity appears to be the most important factor in controlling the species composition of an individual bed of SAV (Stevenson and Confer 1978), while sediment composition and light regime are important factors in controlling the distribution of SAV within regions of the Bay. All species, regardless of the salinity regime, are found in regions of the Bay's littoral zone and are located in water less than two to three meters deep (mean low water - MLW), primarily because of low levels of light that occur below these depths (Wetzel et al. 1981).

Three associations of SAV can be described in Chesapeake Bay based on their salinity tolerances as well as on their co-occurrence in mixed beds of SAV (Table 1) (Orth et al. 1979, Stevenson and Confer 1978). The first association, consisting of *Najas guadalupensis* (bushy pondweed), *Ceratophyllum demersum* (coontail), *Elodea canadensis* (waterweed), and *Vallisneria americana* (wildcelery), contains species that can tolerate fresh to slightly brackish water and are found in the upper reaches of the Bay and in the tidal freshwater areas of the Bay tributaries. The second association, including *Ruppia maritima* (widgeon grass), *Myriophyllum spicatum* (Eurasian watermilfoil), *Potamogeton pectinatus* (sago pondweed), *Potamogeton perfoliatus* (redhead grass), *Zannichellia palustris* (horned pondweed), and *Vallisneria americana* (wildcelery), is tolerant of slightly higher salinities than the first group. This group is found in the middle reaches of the Bay and its tributaries. The third group, consisting of *Zostera marina* (eelgrass) and *Ruppia maritima* (widgeon grass), is tolerant of the highest salinities in the Bay and is found in the lower sections of the Bay and its tributaries.

Since 1978 SAV has been the subject of an intensive research program funded by the U.S. Environmental Protection Agency's Chesapeake Bay Program (EPA/CBP). SAV was determined to be a high priority area of research in this program because of its high primary productivity; its important roles in the Chesapeake Bay ecosystem -- a food source for waterfowl, a habitat and nursery area for many species of commercially important fish and invertebrates, a shoreline erosion control mechanism, and a nutrient buffer. Most importantly, research was focused on SAV because of the dramatic, Bay-wide decline of these species in the late 1960's and 1970's.

Table 1. SPECIES ASSOCIATIONS OF SAV IN CHESAPEAKE BAY AND ITS TRIBUTARIES BASED ON THEIR SALINITY TOLERANCES AS WELL AS THEIR CO-OCCURRENCE WITH OTHER SPECIES (COMMON NAME OF EACH SPECIES GIVEN IN PARENTHESIS)

Group 1	Group 2	Group 3
Ceratophyllum demersum (coontail)	*Myriophyllum spicatum* (Eurasian watermilfoil)	*Ruppia maritima* (widgeon grass)
Elodea canadensis (common elodea)	*Potamogeton pectinatus* (sago pondweed)	*Zostera marina* (eelgrass)
Najas guadalupensis (southern naiad)	*Potamogeton perfoliatus* (redhead grass)	
Vallisneria americana (wildcelery)	*Ruppia maritima* (widgeon grass)	
	Vallisneria americana (wildcelery)	
	Zannichellia palustris (horned pondweed)	

One of the main elements of the SAV program was to examine the current distribution and abundance of submerged grasses in Chesapeake Bay using aerial photography to map the vegetation. In addition, the historical record of aerial photography was examined for recent evidence (less than 40 years) of alterations in SAV abundance, and a biostratigraphic analysis of sediment was performed to detect evidence of longer term (greater than 40 years) alterations in the abundance or species composition SAV beds in several locations within the Bay. A comparison was made to answer basic questions on the magnitude of the present decline of SAV as compared with documented historic declines, and to determine whether the curent decline was part of a natural cycle or a decline attributed to recent non-cyclic perturbations.

SECTION 2

METHODS

The accurate delineation of SAV communities to analyze their distribution and abundance is difficult, especially when the areas of interest may incorporate hundreds of miles of shoreline that are subject to turbid water conditions. These communities are not static, but represent dynamic elements whose distribution and abundance can vary in both space and time. Distinct differences in SAV beds can be observed in time frames of less than two months. To avoid the problems associated with labor-intensive field surveys that provide only a limited view of SAV distribution, remote sensing techniques (aerial photographs) were used to acquire a synoptic view of the existing beds of SAV.

In 1978, the entire shoreline of Chesapeake Bay and its tributaries, from the Susquehanna Flats to the mouth of the Bay, was flown with light planes equipped with mapping cameras to acquire aerial photographs of all existing beds of SAV. Beds of SAV observed on the aerial film were mapped directly onto U.S.G.S. topographic quadrangles, and the areas of each bed were determined with an electronic planimeter (see Orth et al. 1979, and Anderson and Macomber 1980 for detailed information on methodologies used for this work). Field surveys of selected sites corroborated information observed on the aerial photographs and provided species information. Aerial photography comparable to that obtained in 1978 was acquired in 1980 and 1981 for Virginia's SAV only.

Data on the past distribution and abundance of SAV in the Bay were acquired from several sources: aerial photographs of the Bay's shoreline and near-shore zone dating back to 1937; reports of field surveys conducted by state and Federal laboratories, as well as by individual scientists throughout the Bay area; studies on the biostratigraphical analysis of estuarine sediments for seeds and pollen of SAV species (Brush et al. 1980, 1981); and anecdotal information supplied by watermen, landowners, and other interested citizens who had observed changes in the abundance of SAV in numerous areas of the Bay during the last 40 years.

We have organized the discussion of SAV distribution into three zones (Figure 1). The area between the mouth of the Bay to a line stretching from the mouth of the Potomac River to just above Smith Island will be referred to as the lower Bay zone; the area between Smith Island and Chesapeake Bay Bridge at Kent Island will be referred to as the middle Bay zone; and the area between Chesapeake Bay Bridge and Susquehanna Flats will be referred to as the upper Bay zone. These zones have distinct salinity regimes that influence the type of SAV community that will grow within each area. The salinity within each zone roughly coincides with the major salinity zones of the estuaries: polyhaline (18-25 ppt), lower zone; mesohaline (5-18 ppt), middle zone; oligohaline (0.5-5.0 ppt), upper zone. Despite the fact that the major rivers (James, York, Rappahannock, Potomac, and Patuxent) as well as the smaller tributaries (for example, Choptank, Chester, and Piankatank) of the Bay have their own distinct salinity patterns, the distribution of the grasses in each river will be discussed within the zone where it connects to the Bay proper.

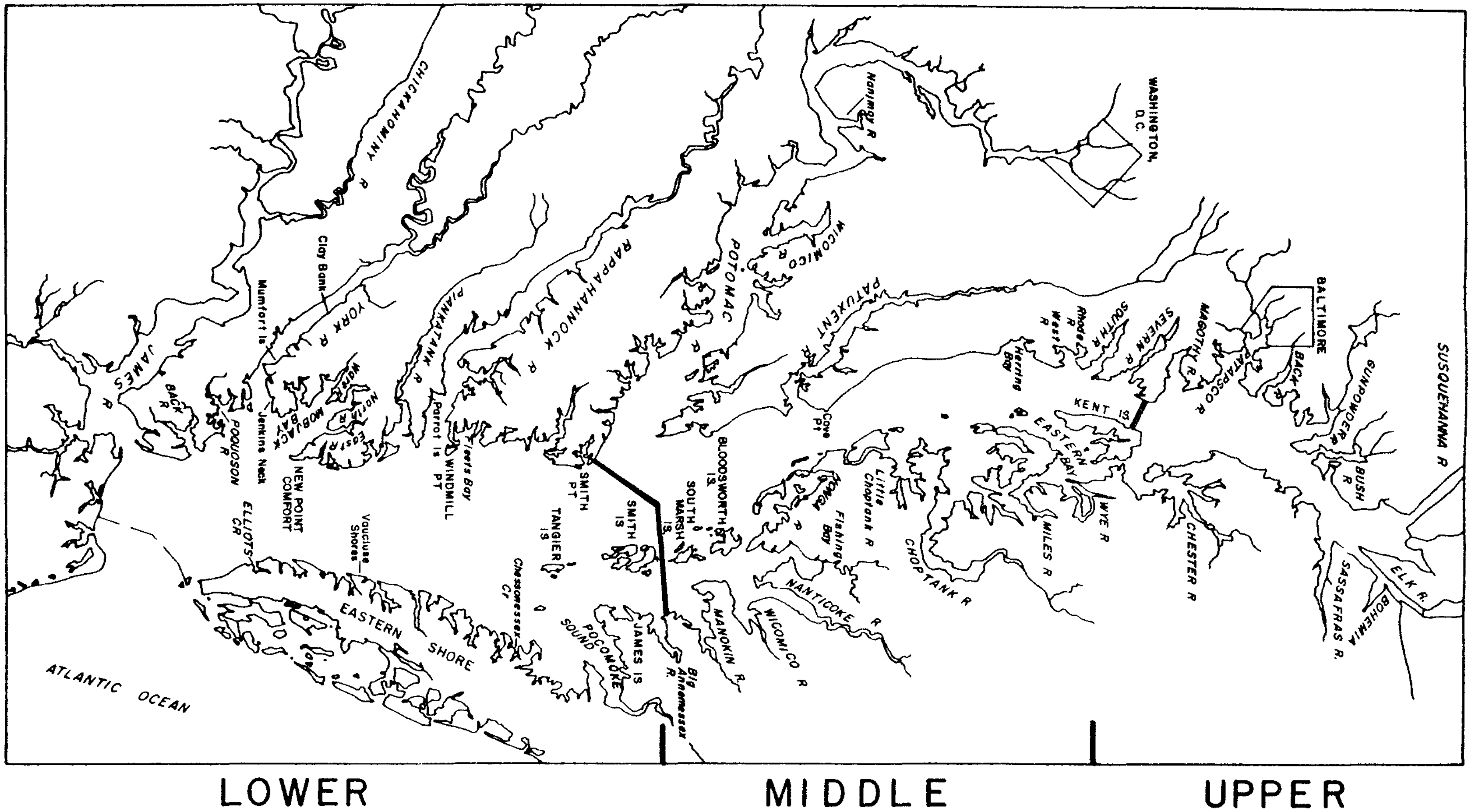

Figure 1. Map of Chesapeake Bay showing the zonation of the Bay into the lower, middle and upper zones.

SECTION 3

PRESENT DISTRIBUTION

The results of the 1978 SAV aerial survey and mapping of the entire Bay and its tributaries documented the existence of significant stands of vegetation (Orth et al. 1979, Anderson and Macomber 1980). A total of 16,044 ha (39,629 acres) of bottom was found to be vegetated. Table 2 presents area values for major sections within each zone.

In the lower Bay zone (Figure 1) where salinities range from 16-18 ppt to 25 ppt, two species predominated: eelgrass (*Z. marina*) and widgeon grass (*R. maritima*). Horned pondweed (*Z. palustris*) was present, but occurred infrequently. In 1978, there were approximately 9400 ha (23,218 acres) of bottom covered with SAV in this zone. This included 46 ha (114 acres) of SAV that were found in the Chickahominy River, a fresh to brackish water tributary of James River. These areas ranged from very dense to very sparse in SAV coverage. The largest and most dense grass flats were concentrated in several main regions: (1) along the western shore of the Bay from just north of the James River to the Rappahannock River, especially in the region of Mobjack Bay; (2) behind protective sandbars along the Bay's eastern shore; and (3) in the shoal area between Tangier Island and Smith Island. The SAV bed between Tangier and Smith Island was the single, most extensive vegetated area in the entire Bay, with a total area coverage of 2394 ha (5912 acres) or 26 percent of the total vegetated bottom in the lower zone and 15 percent of the total vegetated bottom in the entire Bay. 1980 data for the upper Bay were not available.

Updated aerial photographs taken of the lower Bay in 1980 and 1981 indicate a decrease in abundance in 1980 followed by slight rebounding in 1981 (Table 3). The pattern of change determined for one section of the Mobjack Bay area since 1974 (Figure 2) illustrates a decrease in vegetation in the outer, generally deeper portions of the beds, a common pattern in areas where the vegetation has declined. It is significant to note that in one intensively sampled site in the York River a general increase in vegetation abundance was observed from 1978 to 1981. Examination of this site revealed that this increase was a result of a large number of seedlings, many with seed coats still evident, that were growing only in the most shallow areas of this location. Subsequent rapid growth and spreading of the seedlings are indicative of the potential importance of seeds to the reestablishment of the vegetation (Orth and Moore, in press).

In the middle zone of the Bay (Figure 1), SAV was found to shift from *Zostera*-*Ruppia* dominated beds to the lower salinity *Potamogeton*, *Zannichellia*, *Vallisneria*, and *Myriophyllum* beds. This zone contained 4,546 ha (11,229 acres) of bottom covered with SAV in 1978. The greatest concentration of vegetation (77 percent of 3500 ha) was located in the Little Choptank River to Eastern Bay area of the eastern shore (Table 2). Only five percent or 227 ha (561 acres) of the vegetation occurred between the Little Choptank River and Smith Island. An equally small amount [six percent or 273 ha (674 acres)] occurred along the western shore of the Bay

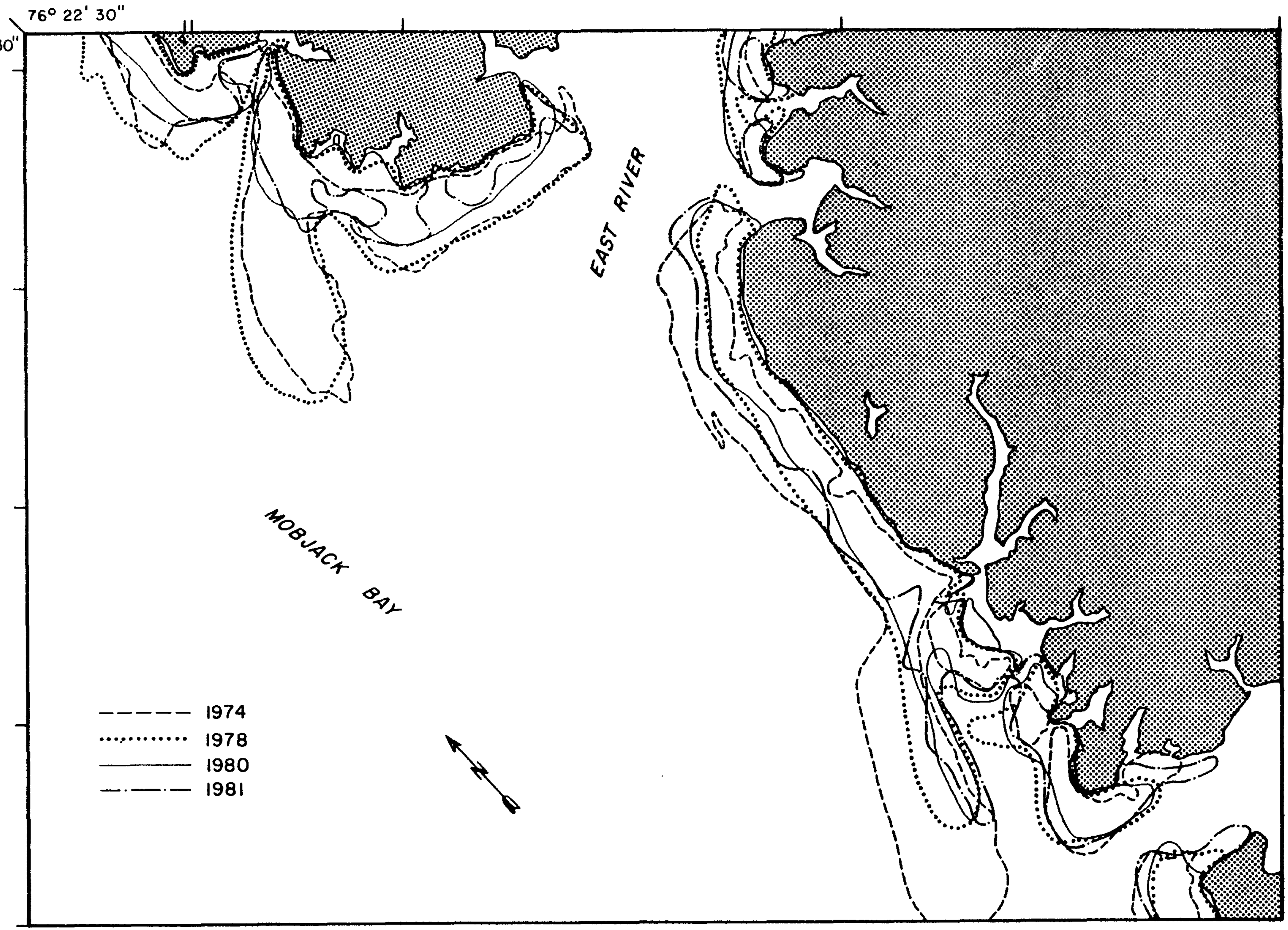

Figure 2. Map of mouth of East River and a portion of Mobjack Bay showing changes in SAV distribution from 1974 to 1981.

TABLE 2. NUMBERS OF HECTARES OF BOTTOM, COVERED WITH SUBMERGED AQUATIC VEGETATION IN 1978 FOR DIFFERENT SECTIONS WITHIN THE THREE ZONES IN THE CHESAPEAKE BAY (NUMBERS OF HECTARES ROUNDED OFF TO NEAREST WHOLE NUMBER)(DATA FROM ORTH et al. 1979, ANDERSON AND MACOMBER 1980)

Section	Hectares	Zone Totals
1. Susquehanna Flats	110	Upper
2. Upper Eastern Shore (Elk, Bohemia, and Sassafras Rivers)	29	
3. Upper Western Shore (Bush, Gunpowder, Middle, Back and Magothy Rivers, and Baltimore Harbor)	484	2098 hectares
4. Chester River	1475	
5. Central Western Shore (Severn, South, and West Rivers, and Herring Bay)	241	
6. Eastern Bay (Wye, East, and Miles Rivers)	1800	
7. Choptank River (Harris and Broad Creeks, Tred-Avon and Little Choptank Rivers, and Trippe Bay)	1740	Middle
8. Patuxent River	3	
9. Middle Western Shore (Herring Bay to mouth of Potomac River)	11	4546 hectares
10. Lower Potomac River Section (Nanjemoy Creek to mouth of Potomac)	541	
11. Middle Eastern Shore (Honga River to Smith Island and including Fishing Bay, Nanticoke, Wicomico, and Manokin Rivers)	210	
12. Tangier Island Complex (includes from Smith Island and Big Annemessex River to Chesconessex Creek)	3759	
13. Lower Eastern Shore (Chesconessex Creek to Elliots Creek)	1991	Lower
14. Reedville (includes area from Fleets Bay to Great Wicomico River)	364	
15. Rappahannock River (includes Rappahannock and Piankatank Rivers, and Milford Haven)	93	9354 hectares
16. New Point Comfort Region	271	
17. Mobjack Bay (includes East, North, Ware, and Severn Rivers)	1785	
18. York River (Clay Bank to mouth of York)	157	
19. Lower Western Shore (includes Poquoson and Back Rivers)	925	
20. James River (Hampton Roads area only)	9	

TABLE 3. NUMBERS OF HECTARES OF BOTTOM, COVERED WITH SUBMERGED AQUATIC VEGETATION IN 1971, 1974, 1978, 1980, AND 1981 FOR DIFFERENT SECTIONS IN THE LOWER BAY ZONE (NUMBERS OF HECTARES ROUNDED OFF TO NEAREST WHOLE NUMBER)(* INDICATES SECTIONS THAT WERE NOT MAPPED THAT YEAR) (DATA FROM ORTH AND GORDON 1975, ORTH et al. 1979, AND UNPUBLISHED DATA)

	Year				
Section	1971	1974	1978	1980	1981
Tangier Island Complex (Includes from MD-VA border to Chesconessex Creek)	*	*	2814	2420	2794
Lower Eastern Shore (Chesconessex Creek to Elliots Creek)	*	*	1991	1370	1691
Reedville (Includes area from Windmill Pt. to Smith Pt.)	*	*	364	31	133
Rappahannock River (Includes Rappahannock and Piankatank Rivers, and Milford Haven)	1273	68	93	3	43
New Point Comfort Region	168	233	271	182	207
Mobjack Bay (Includes East, North, Ware, and Severn Rivers)	1294	1593	1785	1317	1275
York River (Clay Bank to mouth of York)	493	141	157	135	142
Lower Western Shore (Includes Poquoson and Back Rivers)	1620	1069	925	1002	996
James River (Hampton Roads area only)	*	7	9	0	0
TOTAL FOR LOWER BAY ZONE			8,409	6,460	7281

from the mouth of the Potomac River to Chesapeake Bay Bridge, including the South, Severn, Rhode, and West Rivers. The Patuxent River had virtually no vegetation with only three ha (7.4 acres) being observed along the entire length of the river. A small amount [12 percent or 545 ha (1346 acres)] of the total vegetation in this zone was found in the Potomac River in the vicinity of Nanjemoy Creek, Port Tobacco River, Mathias Point Neck, and Mattox and Machodoc Creeks, at a distance of 50 to 100 km from the river's mouth. These beds fringe the shoreline on the lower portions of the creeks and the Potomac River proper, near U.S. 301 bridge, and are dominated by P. perfoliatus and V. americana. This was the only vegetation found along the entire length of the Potomac River, except for small pockets of SAV that existed at the heads of several small marsh creeks (Carter and Haramis 1980, Carter et al. 1980). In addition, this is the only area of comparable vegetation found along any of the Bay's major western tributaries (James, York, Rappahannock, Potomac, and Patuxent Rivers). Less intensive surveys in 1979 showed only slight decreases from the 1978 distributional patterns to those in 1979, but considerable declines in 1981 were observed throughout the middle zone of the Bay (personal information from unmapped data).

The upper zone of the Bay (Figure 1) contained 2098 ha (5182 acres) of substrate covered with SAV in 1978 (Table 2), with the species association shifting from Group 2 to Group 1 (Table 1). Susquehanna Flats had 110 ha (272 acres) of vegetation in 1978, most of which occurred in scattered beds. This was a very small area when compared to abundance of SAV in the late 1960s and early 1970s. Only two species were present on the Flats in 1978, Eurasian watermilfoil (M. spicatum) and wildcelery (V. americana); eleven species were found by researchers in 1971 (Bayley et al. 1978). Approximately 23 percent of the total bottom area covered with SAV in this zone was in the Gunpowder, Middle, Bush, and Magothy Rivers located along the western shore, whereas almost no vegetation was present in the Elk, Bohemia, and Sassafras Rivers on the eastern shore. About 70 percent [1469 ha (3628 acres)] of the total bottom area covered with vegetation was present in the Chester River and Eastern Neck area. The Chester River area contained a diverse assemblage of SAV, with seven species recorded during the 1978 survey. Less intensive surveys in 1979 show little change in distribution patterns from 1978, but surveys in 1981 indicate considerable declines in this zone.

In summary, the survey of SAV in the Bay in 1978 indicated the presence of many apparently healthy beds in various sections of the Bay. There were, however, large sections devoid of almost all vegetation where, in earlier years (1965-1970), luxuriant beds persisted (see Figure 5). Tributaries with major reductions of SAV included portions of the York, Rappahannock, Potomac, Patuxent, Choptank, Chester, and Piankatank Rivers. SAV populations in other areas along the main stem of the Bay, including Susquehanna Flats, the area between Smith Point on the Potomac River, and Windmill Point on the Rappahannock River, and an area between Smith Island and Eastern Bay, which includes many smaller rivers, have also significantly declined. More recent evidence from ground truth surveys and aerial photographs taken from 1978 to 1981 indicate that this decline has continued in certain areas. This suggests a widespread but complex pattern of recent major decline, involving the entire spectrum of SAV communities found in the Bay, from the mouth of the Bay to Susquehanna Flats at the head of the Bay.

SECTION 4

PAST DISTRIBUTION

A detailed discussion of past trends of SAV distribution and abundance is hindered by the lack of adequate data for many sites over a long period of time. A review of the available historical information indicates that SAV has generally, in the past, been very abundant throughout the Bay. In the last 50 years, however, there have been several distinct periods where SAV, in some large portions of the Bay, has undergone major fluctuations, although SAV populations have been known to undergo erratic oscillations within small areas (Stevenson and Confer 1978).

HISTORICAL TRENDS (1700-1930)

The pattern of SAV distribution and abundance in the Bay during this period was determined primarily from indirect evidence, pollen and seed analysis, and qualitative observations. Aerial photography can usually provide good evidence for the presence of SAV, but was not generally available until the late 1930s. If it can be assumed that less urbanization during this period resulted in better water quality throughout the Bay and its tributaries (Heinle et al. 1980), conditions may have been more favorable for the growth of SAV.

Biostratigraphical analysis of sediments for SAV seeds and pollen from Furnace Bay (Brush et al. 1980), a small embayment off Susquehanna Flats, indicates the continuous presence of SAV seeds from the 17th century. However, there appear to have been some changes in species of SAV (for example, declines of *Najas* spp.) corresponding to changes in land use, such as deforestation. Increased erosion and sedimentation from these practices possibly resulted in more turbid water conditions and, thus, the eventual decline of species less adapted to low light levels.

The Potomac River, the largest tidal tributary in the Bay, historically contained numerous species of SAV that were very abundant. Several species (wild celery, coontail, naiad, and elodea) were reported in the vicinity of Washington, D.C. in one of the earliest accounts (Seaman 1875). Cumming et al. (1916) provided a map of the Potomac River below Washington, DC that showed the river having a narrow channel and wide shallow margins that he reported to be extensively vegetated with curly pondweed (*P. crispus*), wildcelery (*V. americana*), and coontail (*C. demersum*). Many other pondweed species were reported at mouths of tributaries below Washington, D.C. (Hitchcock and Standley 1919), indicating the widespread presence of SAV species in the tidal portion of the Potomac River.

Eelgrass (*Z. marina*) apparently underwent some decline in Chesapeake Bay area in the late 19th century, although the magnitude of the decline was never quantified. Cottam (1934, 1935) states that a guide from the Honga River Gunning Club reported on the decline of eelgrass in Dorchester County, Maryland in 1893-1894. Cottam also reports an interview with a member of the Maryland Game Commission who commented on the decline of eelgrass in Chesapeake Bay in 1889 (at the time of the Johnstown Flood) and stated that it was 25 years before eelgrass fully recovered. Cottam documents other declines of eelgrass along the east coast of the U.S. -- one as early as 1854. From these accounts, it appears that eelgrass has undergone several fluctuations during this period (1700-1930), suggesting some irregular, though undefined, perturbations on the system.

In summary, evidence suggests that in the Bay: (1) SAV was apparently much more widespread from 1700 to 1930 than it is today; (2) SAV had been a persistent feature of shallow water habitats, although there may have been some localized shifts in species composition of the beds; and (3) abundance of eelgrass has apparently undergone changes several times.

RECENT PAST (1930-1980)

With an increased awareness of the value of SAV as a food source for waterfowl wintering in the Bay and observations of major fluctuations in the Bay and elsewhere, researchers placed more focus on the distribution and abundance of SAV during this period. This research led to the availability of more quantitative information; as a result, a much greater perspective can be obtained. During these last 50 years, there have been two distinct events in which significant changes occurred within individual species of SAV: (1) the eelgrass wasting disease in the 1930's; and (2) the watermilfoil (*M. spicatum*) problem in the late 1950's and early 1960's. Even far more dramatic are the changes in SAV populations in the Bay in the 1960's and 1970's, when, unlike the eelgrass and milfoil events, all species in almost all areas of the Bay were affected to some degree. The following three sections discuss each of these periods.

The Eelgrass Wasting Disease (1931-1932)

The most documented decline of a species in the Bay was that of eelgrass in the early 1930's. This decline was recorded not only in the Bay area, but also along the entire east coast of the U.S. and the west coast of Europe (Cottam 1934, 1935; den Hartog 1970; Rasmussen 1977). Cottam (1934) comments, based on information from his surveys of historical records and personal inquiries of fishermen, watermen, and scientists, that "in the memory of man there has been no period of scarcity at all comparable to the present one (1931-1932 compared to other past periods)." The extent of the decline in Chesapeake Bay was never quantified, but aerial photographs taken in 1937, five to six years after the height of the decline, are available for almost all of the shoreline in the lower Bay. A review of many areas in the lower Bay and subsequent mapping of six sites (Orth et al. 1979) shows areas of bottom in shallow water covered with large amounts of submerged vegetation (it was assumed to be eelgrass based on knowledge of present day patterns and anecdotal information from long-time residents of these areas). All six areas showed subsequent increases in later years up to 1972. Although quantitative information is lacking prior to the wasting disease, we assume that the vegetation present in 1937 represented partial recovery from the height of the decline in 1931-1932. Cottam (1935) confirmed our conclusions from aerial photographs when he reported that Chesapeake Bay eelgrass was showing "an encouraging change, with a few localized areas fast approaching the normal."

One indication of the magnitude and severity of the decline of eelgrass, experienced not only in Chesapeake Bay but also along the east coast of the U.S. and the west coast of Europe, was found in the coastal lagoons on Virginia's seaside. These areas contained dense beds of eelgrass that supported a large bay scallop industry. The post-veliger larvae of the scallop require eelgrass as a setting substrate (Gutsell 1930). Without eelgrass, there can be no scallops because a scallop lives, at the longest, two years, and a change or disappearance of eelgrass results in rapid shifts of the scallop population. Indeed, this is what happened (Table 4). The commercial fishery that resulted in a harvest of over 14,000 kg per year in the late 1920s and early 1930s completely declined in 1933, over a span of just two years. Eelgrass has never recovered in the seaside bays as compared with Chesapeake Bay and many other areas where it had substantially declined (Cottam and Munro 1954), nor has the scallop industry ever returned.

TABLE 4. CHANGES IN AMOUNT OF SCALLOPS (SHUCKED MEAT) HARVESTED FROM THE DELMARVA PENINSULA FROM 1928-1975 (COLLATED FROM U.S. FISHERIES DIGEST)

Year	Harvested scallops (kg shucked meat)
1928	5,050
1929	16,038
1930	25,549
1931	17,170
1932	9,220
1933	0
1934	0
1981	0

The Milfoil Problem (1959-1965)

A second major period of extensive SAV fluctuation in the Bay was the large increase in Eurasian watermilfoil (*M. spicatum*) in the late 1950's and early 1960's (Stennis 1970, Bayley et al. 1978, Stevenson and Confer 1978b). The area affected by the milfoil was restricted to the upper Bay area and a large section of the Potomac River (Figure 3). The intolerance of milfoil to high salinity water limited its downward expansion in the Bay, but reasons for its sudden expansion in abundance during this period are not well understood. Until 1955, milfoil was found only sporadically in the Bay, apparently introduced from Europe to the U.S. between 1880 and 1900 (Rawls 1978). Biostratigraphic evidence substantiated its recent arrival to Chesapeake Bay (Brush et al. 1980). Milfoil seeds were found in sections of sediment cores from Furnace Bay near Susquehanna Flats and dated only to approximately 1935, though sediments from the cores had recorded events, including the presence of other SAV species, to 1770.

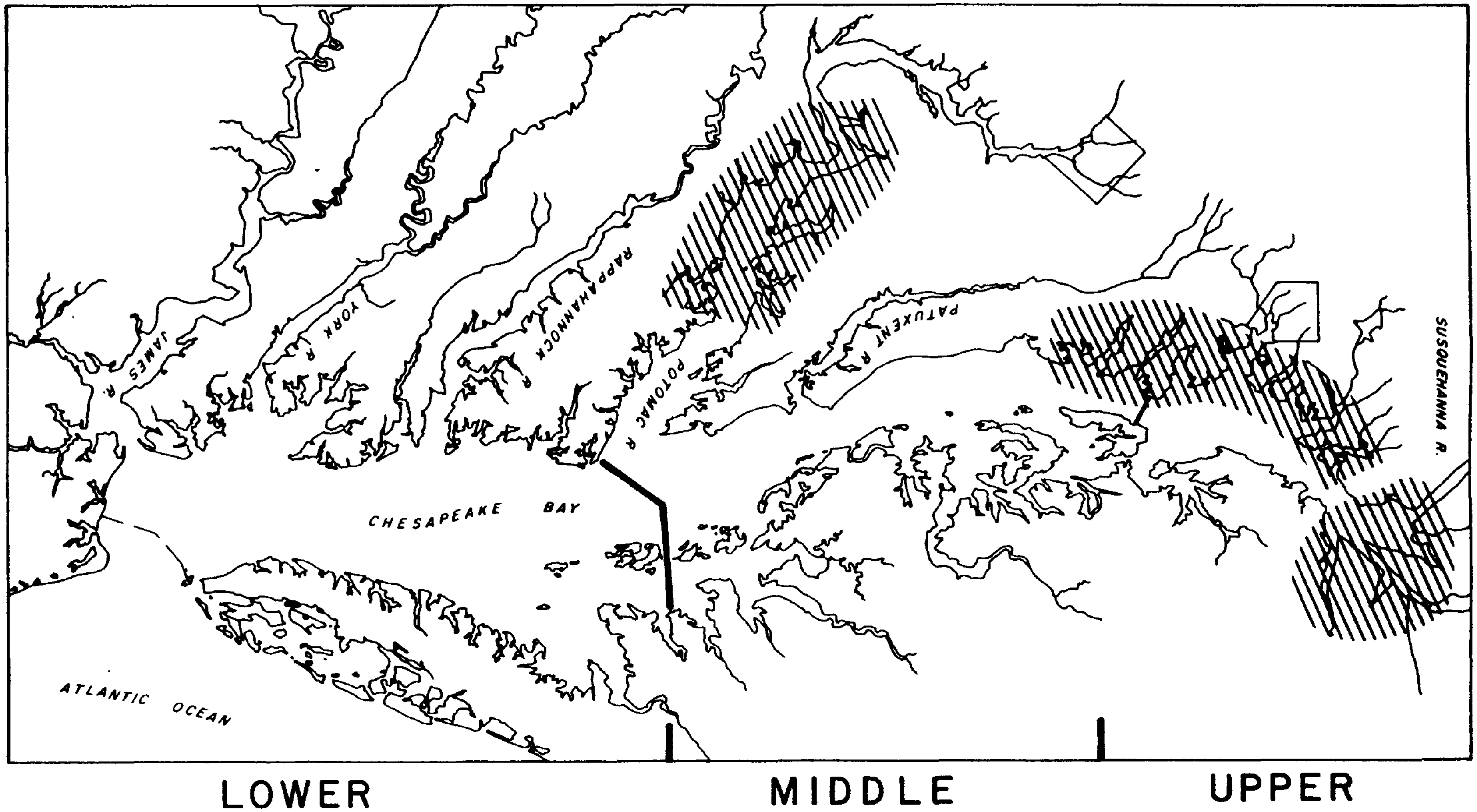

Figure 3. Location of regions (cross-hatching) in the Bay area which were considered to be severely impacted by the growth of Eurasian watermilfoil from 1959-1963.

Milfoil increased Bay-wide from 20,200 ha (49,894 acres) in 1960 to 40,500 ha (100,035 acres) in 1961 (Rawls 1978). In contrast, the 1978 baywide SAV survey found that only 16,000 ha (39,525 acres) of bottom were covered by all SAV species combined. In creeks along the Potomac River, the milfoil reached densities so high that it was considered a nuisance, and attempts to eradicate it with applications of 2-4 D were initiated (Rawls 1978).

The Susquehanna Flats area typifies the changes noted during the rapid expansion of milfoil. In 1957, a survey conducted of SAV found that milfoil did not occur at any sampling stations. Subsequently, it was found in one percent of these stations in 1958, 47 percent in 1959, 82 percent in 1960, and 89 percent in 1961 and 1962. After 1962, milfoil declined in the Flats, with slight increases in 1966 and 1967. The most serious effect associated with the rapid increase in milfoil was a decline in other native species such as common elodea (E. canadensis), naiad (N. guadalupensis), and wildcelery (V. americana). The decline of native species is shown in Figure 4. For example, this graph shows that in 1963 abundance of native plant material was below 50, while abundance of watermilfoil was over 200. Bayley et al. (1978) suggest that the decline of native species was due to competitive exclusion by milfoil. As milfoil declined, these native species returned, but were found at a lower density and covered less area than prior to the milfoil expansion (Bayley et al. 1978).

The Bay-wide Problem (1960-1980)

In the 1960's and 1970's a number of field surveys and aerial surveys were conducted to estimate the distribution and abundance of SAV in the Bay. These estimates, when considered with the results of the SAV distribution projects funded by the Bay Program, reveal dramatic results. The combined data show a pattern of vegetation decline that includes all species in all sections of the Bay and a present abundance of vegetation that may be at its lowest level in recorded history.

The results of this recent decline were first evident in changes in diving duck populations in the Bay (Perry et al. 1981). Two species, in particular, the canvasback (Aythya valisineria) and the redhead (Aythya americana), have shown significant population declines in the last 10 years in the Bay despite increases in the overall North American and Atlantic flyway populations. These two duck species have traditionally used SAV as food (Stewart 1962). The decline in their preferred food source presumably led to the decline in the total number of ducks found in the Bay. Since the SAV decline, canvasbacks have altered their feeding habits to include clams, and redheads still feed predominantly on vegetation.

To illustrate the major changes of SAV populations that have occurred in the Bay area in the last 20 years, we have delineated SAV distribution on a Bay-wide basis at five-year intervals beginning in 1965 and subsequently in 1970, 1975, and 1980 (Figures 5, 6, 7, and 12). 1965 was chosen as a starting point because of the lack of complete information for Bay-wide determination prior to 1965; the compounding problem of the explosion in the late 1950's of Eurasian watermilfoil, which declined by 1965; and the relatively abundant Bay-wide distribution of SAV during this time, apparent from archival photographs and anecdotal information. Though the scale of the map is small in relation to the generally small size of

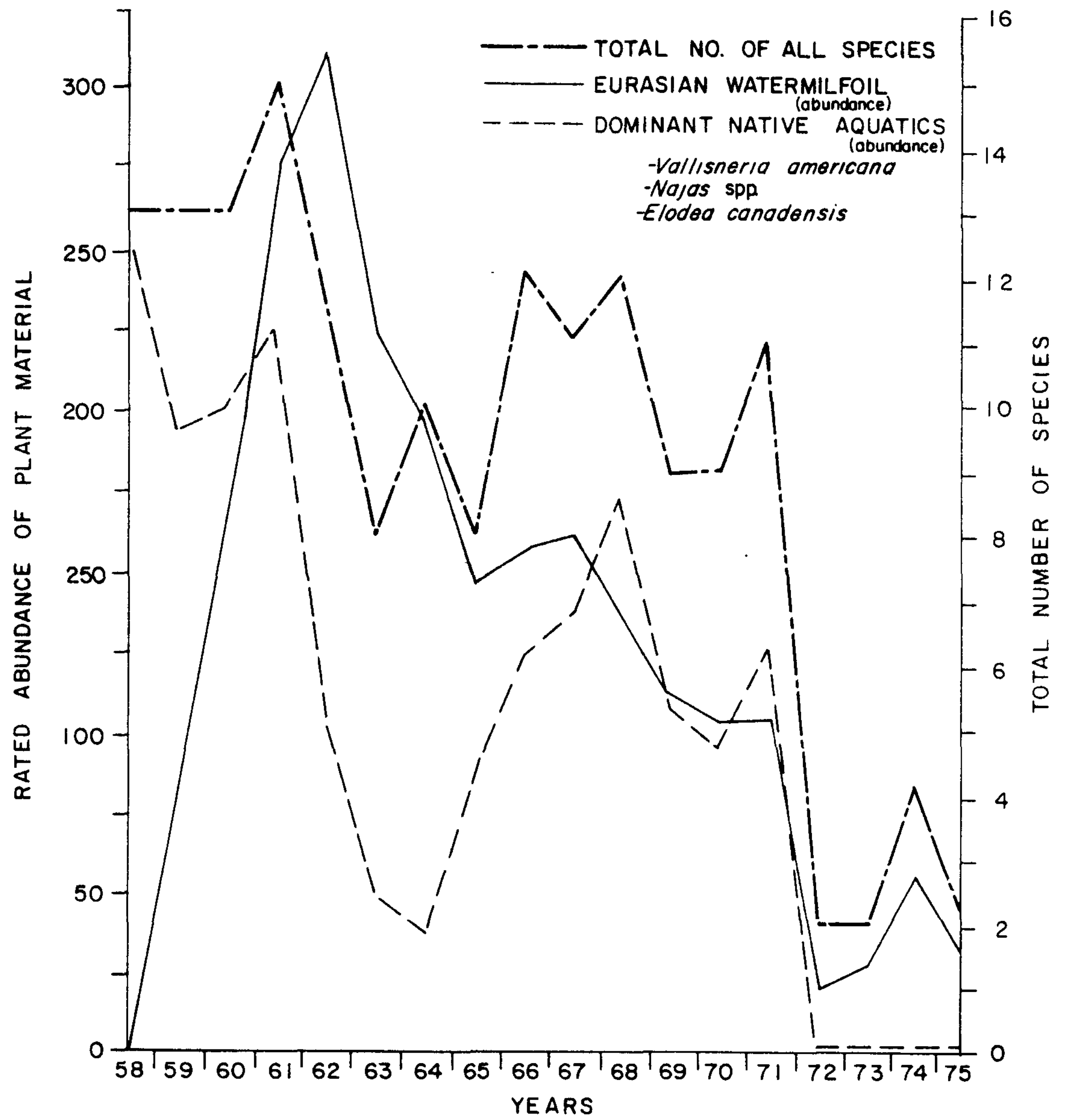

Figure 4. Population fluctuations of watermilfoil compared to the dominant native species and total number of species found on the Susquehanna Flats from 1958-1975 (figure adapted from Bayley et al. 1978).

most SAV areas, the changes that occurred in SAV distribution in each of the five-year intervals were sufficiently dramatic so as to appear quite distinct in the respective figures. Note, for example, the large changes in abundance of SAV in Susquehanna Flats area, Patuxent, and Potomac Rivers from 1965 to 1975. We are aware that the small scale is not suitable for small populations of SAV related to the size of the entire Bay, but the overall changes in SAV on a Bay-wide basis are more easily perceived on this size map. Though in some respects the following maps are qualitative, they represent the culmination of a large effort to incorporate whatever quantitative data were available with the most reliable qualitative data. These maps are the first effort to place into perspective the complex changes that have been observed in SAV populations over the last 20 years.

1965--

In 1965, SAV was quite abundant throughout the Bay and in all of the major tributaries (Figure 5) despite the compounding effects of the milfoil problem in the early 1960's (Bayley et al. 1978). One area, however, that had been reported to have abundant SAV (Cumming et al. 1916), but no longer contained any, was the freshwater tidal portion of the Potomac River (Carter and Haramis 1980, Carter et al. 1980). The SAV of this area apparently declined in the 1930s and had all but disappeared by 1939 (Martin and Uhler 1939). The lower reaches of the Potomac still contained abundant stands of vegetation in 1965 based on evidence from aerial photographs of the Coan, Yeocomico, and lower Machodoc Rivers and from personal accounts of local watermen. In addition, an intensive benthic survey for the soft shell clam, Mya arenaria, in the lower Potomac in 1961 revealed abundant stands of SAV. The lower reaches contained eelgrass, while numerous brackish water species abounded farther upstream (Pfitzenmeyer and Drobeck 1963).

1965-1970--

By 1970 there were still substantial stands of SAV throughout the Bay but evidence indicates some major losses had occurred in several areas (Figure 6). Vegetation in the entire Patuxent River had all but completely disappeared (R. Anderson, personal communication) by 1970, with declines being first noted in the mid-1960's. Anecdotal accounts indicate that populations of eelgrass adjacent to Chesapeake Biological Laboratory at the mouth of the Patuxent River were severely depressed in the late 1960's and gone by 1970. The vegetation in the lower Potomac River evidenced in aerial photographs of the 1960's was also almost completely absent. In addition, vegetation in many of the eastern shore upriver sections of the Choptank, Chester, Gunpowder, and Bush Rivers, as well as in the entire Nanticoke and Wicomico Rivers in the middle and upper Bay zones, was absent or in very reduced abundance (Boynton, personal communication).

SAV in some localized areas around the Bay, including Susquehanna Flats (Bayley et al. 1978) and the Chester River area (Anderson and Macomber 1980), had increased in coverage from 1965 to 1970, though not to previous levels. The increase in these years may have been the result of the reemergence of native SAV species in response to the decline of milfoil (Bayley et al. 1978).

One of the first significant surveys of the upper Bay during this period was that conducted by Stotts from 1967 to 1969 (Stotts 1970). Over

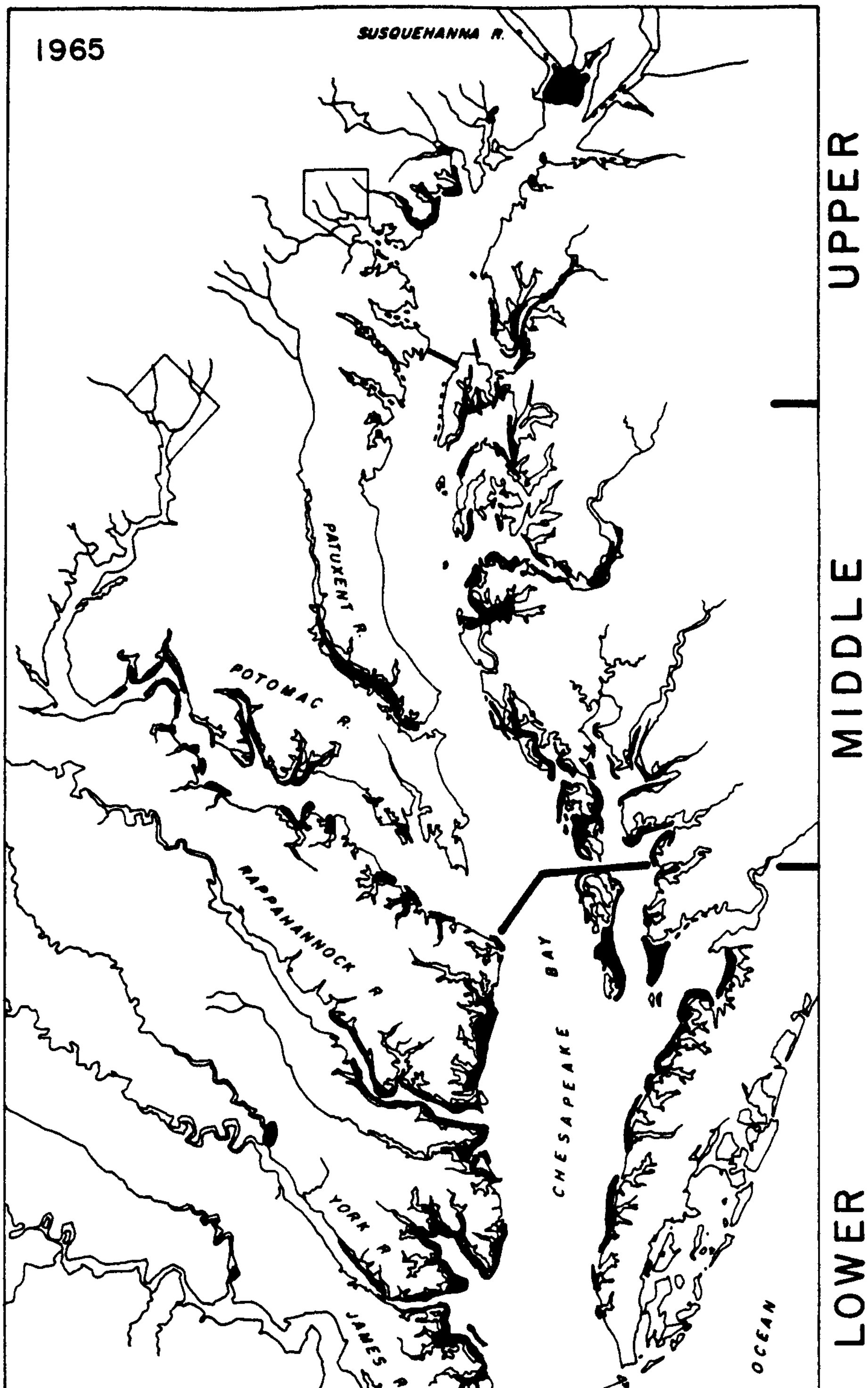

Figure 5 • Distribution of SAV in Chesapeake Bay - 1965.

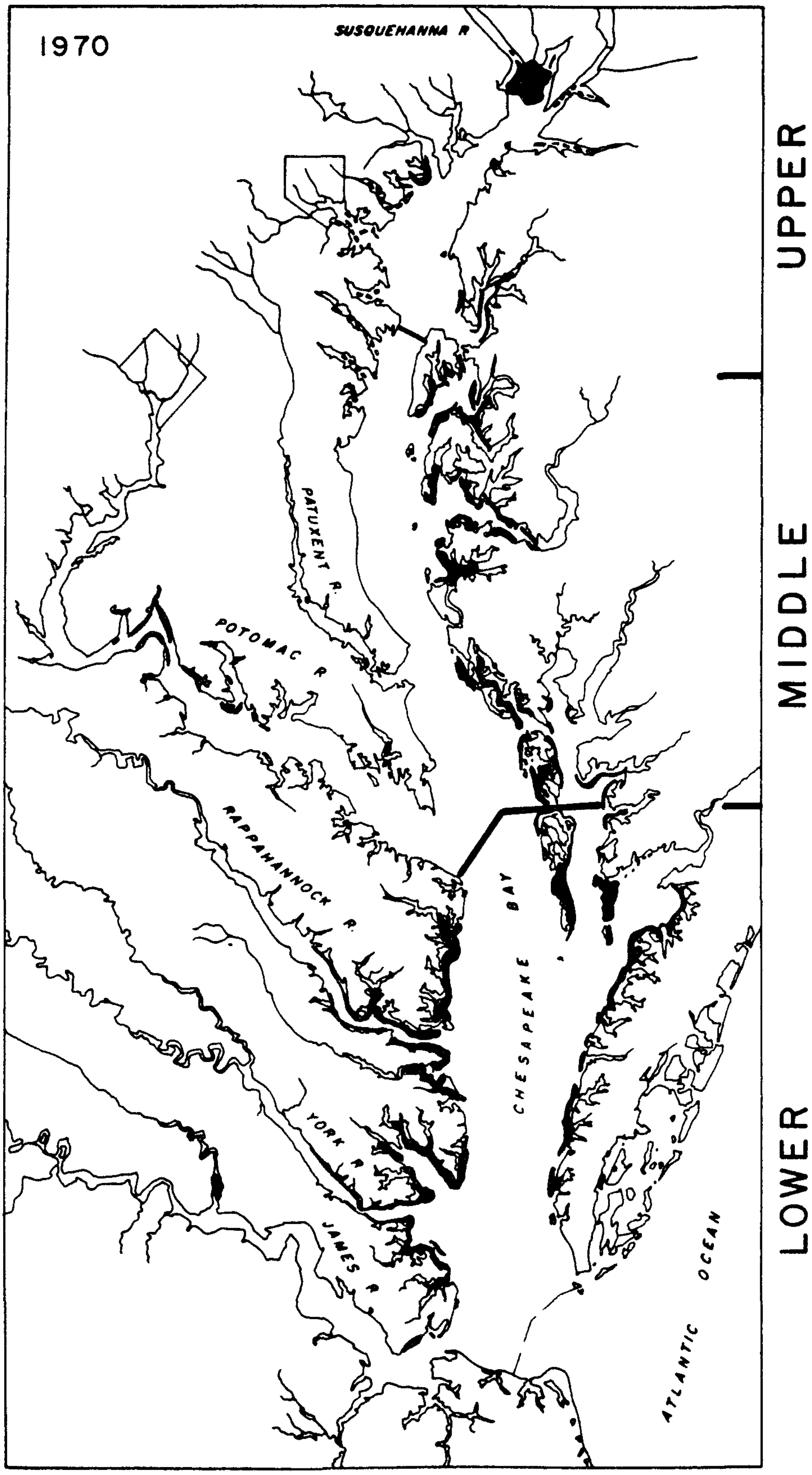

Figure 6. Distribution of SAV in the Chesapeake Bay - 1970.

1,000 transects were sampled from the Virginia - Maryland border to Susquehanna Flats. The survey findings indicate that many areas contained significant beds of vegetation, especially in the more southern locations, from the Choptank River to Smith Island. Stotts reported, however, that large declines of SAV occurred in July and August in several locations north of the Choptank and that SAV did not appear as robust as in the more southern areas, indicating that these systems were being stressed by environmental factors. Examination of aerial photographs taken in September, 1970, shows large beds of vegetation in the same areas where SAV was reported to be abundant by Stotts' survey, especially in the lower reaches of the Chester River, Eastern Bay, Little Choptank River, Honga River, and Bloodsworth Island.

In contrast to the declines evidenced during this period in the upstream, low salinity regions of the Bay and its tributaries, the higher salinity regions vegetated with eelgrass and widgeon grass showed as yet little evidence of any deterioration. Aerial photographs document that extremely dense beds characterized much of the shoreline of the lower Bay and its tributaries, and many areas showed a continued increase in coverage since the 1930's (Orth and Gordon 1975, Orth 1976, Orth et al. 1979).

1970-1975--

By 1975 the Bay-wide situation for SAV had changed dramatically along the entire length of the Bay proper (Figure 7). Indeed, the abundance of vegetation in 1975 represented what we feel was, until then, the lowest recorded abundance of vegetation in Chesapeake Bay and its tributaries as far back as records indicate. The decline of SAV that first began in the mid-1960s and continued to the early 1970's, was now observed in all sections of the Bay, with some areas affected more than others. This decline also appeared to accelerate after Tropical Storm Agnes influenced the Bay in June 1972.

Much of the information available for this period for the upper and middle Bay zones is from the 644 station survey of SAV conducted once a year in Maryland waters beginning in 1971 by the Maryland Department of Natural Resources and the U.S. Fish and Wildlife Service (Kerwin et al. 1977; unpublished files). Their data showed that SAV declined in the surveyed areas between 1971, when 28.5 percent of the stations were vegetated, and 1973, when 10.5 percent of the stations were vegetated (Table 5, Figure 8). SAV fluctuated at comparatively low levels from 1974 to 1975, decreasing to 8.7 percent in 1975. The number of major areas with no SAV increased from five in 1971 to 11 in 1975, an increase of 100 percent (Figure 1 and Table 5). This survey also shows that individual sections of the Bay had not exhibited a uniform trend, but that the head of the Bay and lower eastern shore have fared the worst, while the middle sections of the Maryland eastern and western shores fared the best.

Large reductions in vegetation were observed immediately after Agnes, in July and August, 1972, in many sections of the upper Bay zone (Figure 7), principally the Elk, Bohemia, Sassafras, Back, Middle, Magothy, and Chester Rivers, Howell and Swan Point, Susquehanna Flats, and the headwaters of the Bush and Gunpowder Rivers (Figure 7 and Table 5) (Kerwin et al. 1977). In addition, sections of the middle Bay zone, primarily those in the northern end, such as the Severn River, appeared to be rapidly denuded of grasses. The species that were most affected were the fresh and

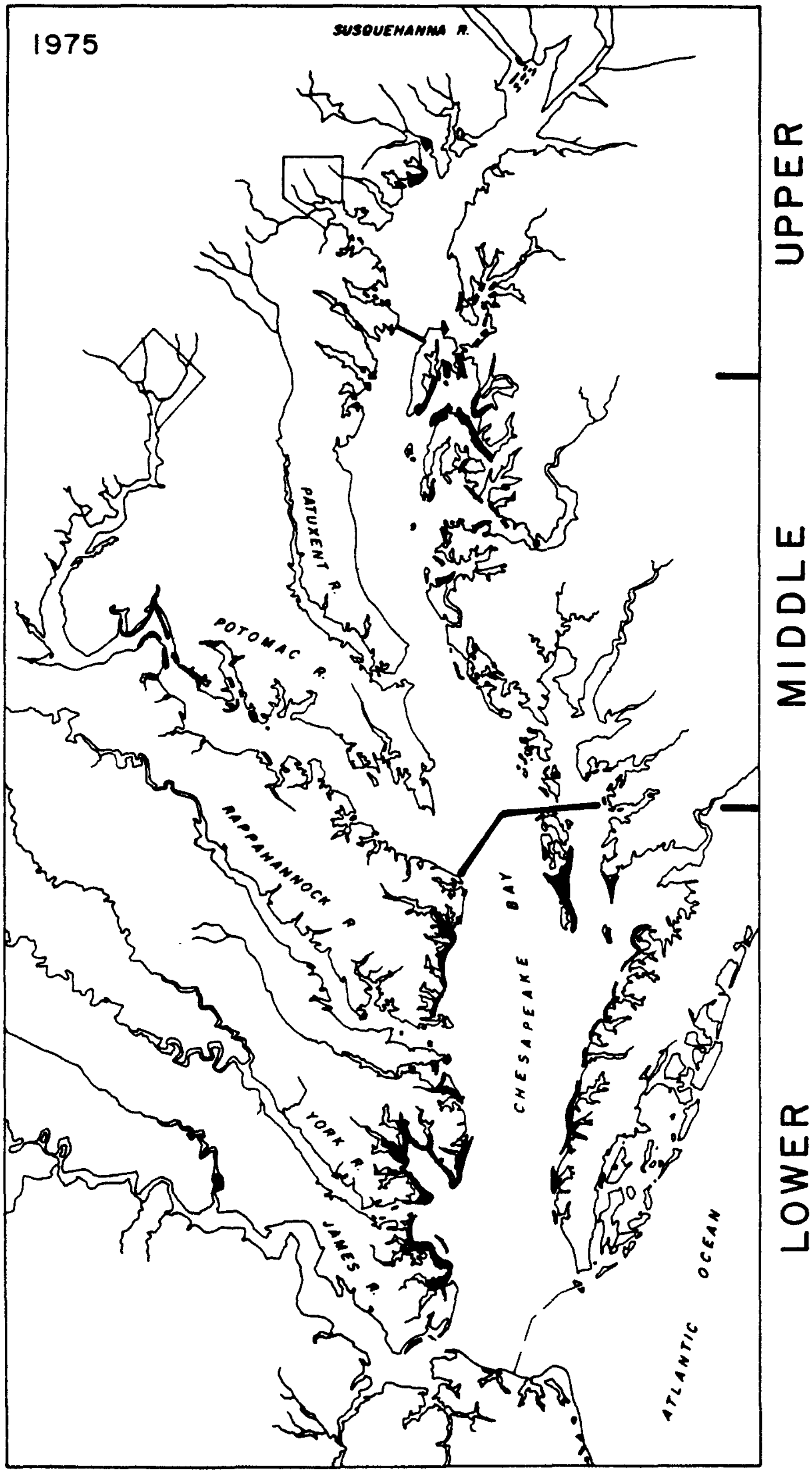

Figure 7. Distribution of SAV in the Chesapeake Bay - 1975.

Table 5. Percent of sampled stations containing submerged aquatic vegetation for various locations in the Maryland section of the Chesapeake Bay (compiled from U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory (as reported in Stevenson and Confer, 1978) and unpublished files from Maryland's Department of Natural Resources) (**no stations sampled for this location).

River System	Avg. No. of Stations Sampled	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980
Elk & Bohemia Rivers	15-16	6.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Sassafras River	10	30.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Howell & Swan Points	12	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eastern Bay	43-47	34.04	46.51	34.04	36.17	21.74	42.22	28.00	26.10	17.30	34.80
Choptank River	56-60	35.00	39.66	19.30	27.59	1.72	41.07	25.00	28.30	26.70	25.00
Little Choptank River	19	21.05	21.05	0.00	0.00	0.00	15.79	5.00	5.30	5.30	0.00
James Island & Honga River	34	44.12	35.29	2.94	5.88	5.88	8.82	3.00	0.00	0.00	0.00
Honga River	29-30	50.00	40.00	13.33	16.66	10.35	17.24	3.00	3.00	0.00	0.00
Bloodsworth Is.	40-46	37.50	22.73	10.87	11.63	6.98	2.22	4.00	0.00	0.00	2.20
Susquehanna Flats	27-37	44.44	2.70	0.00	13.51	11.11	8.57	11.00	2.70	8.10	0.00
Fishing Bay	24-25	8.00	4.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nanticoke & Wicomico Rivers	30-31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Manokin River	14-15	40.00	46.67	13.33	20.00	7.14	6.67	20.00	0.00	0.00	0.00
Patapsco River	20-21	0.00	5.00	4.76	9.25	**	9.52	14.00	9.50	9.50	0.00
Big & Little Annemessex Rivers	18-20	70.00	60.00	30.00	57.89	33.33	30.00	30.00	15.00	0.00	5.00
Gunpowder & Bush River Headwaters	7-9	11.11	0.00	0.00	0.00	**	0.00	11.00	0.00	11.10	22.20
Pocomoke Sound (Maryland)	20-22	18.18	10.00	4.76	**	15.00	9.09	10.00	4.50	0.00	0.00
Magothy River	12	33.33	0.00	16.67	16.66	**	16.67	25.00	8.30	16.70	16.70
Severn River	13-15	40.00	20.00	26.67	26.67	0.00	46.15	20.00	26.70	20.00	13.30
Back, Middle & Gunpowder Rivers	22	13.64	4.55	4.55	4.55	9.09	4.55	9.00	4.50	4.50	9.10
Curtis & Cove Points	6-21	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.30	0.00
South, West & Rhode Rivers	8-10	0.00	0.00	0.00	0.00	0.00	12.50	0.00	0.00	0.00	0.00
Chester River	34-36	61.11	36.11	26.47	23.52	25.00	25.71	38.00	44.40	33.30	38.90
Love & Kent Points	8	0.00	0.00	0.00	12.50	0.00	0.00	0.00	0.00	0.00	0.00
Smith Island (Maryland)	11-17	64.71	45.46	25.00	35.29	22.22	35.29	24.00	5.80	17.60	47.10
Patuxent River	47-50	2.00	4.26	0.00	4.00	0.00	2.04	2.00	2.00	2.00	0.00
Percent of stations vegetated		28.5	21.0	10.5	14.9	8.7	15.0	12.3	9.6	7.9	9.8
Number of stations with no SAV recorded		5	9	12	9	11	8	8	12	13	16

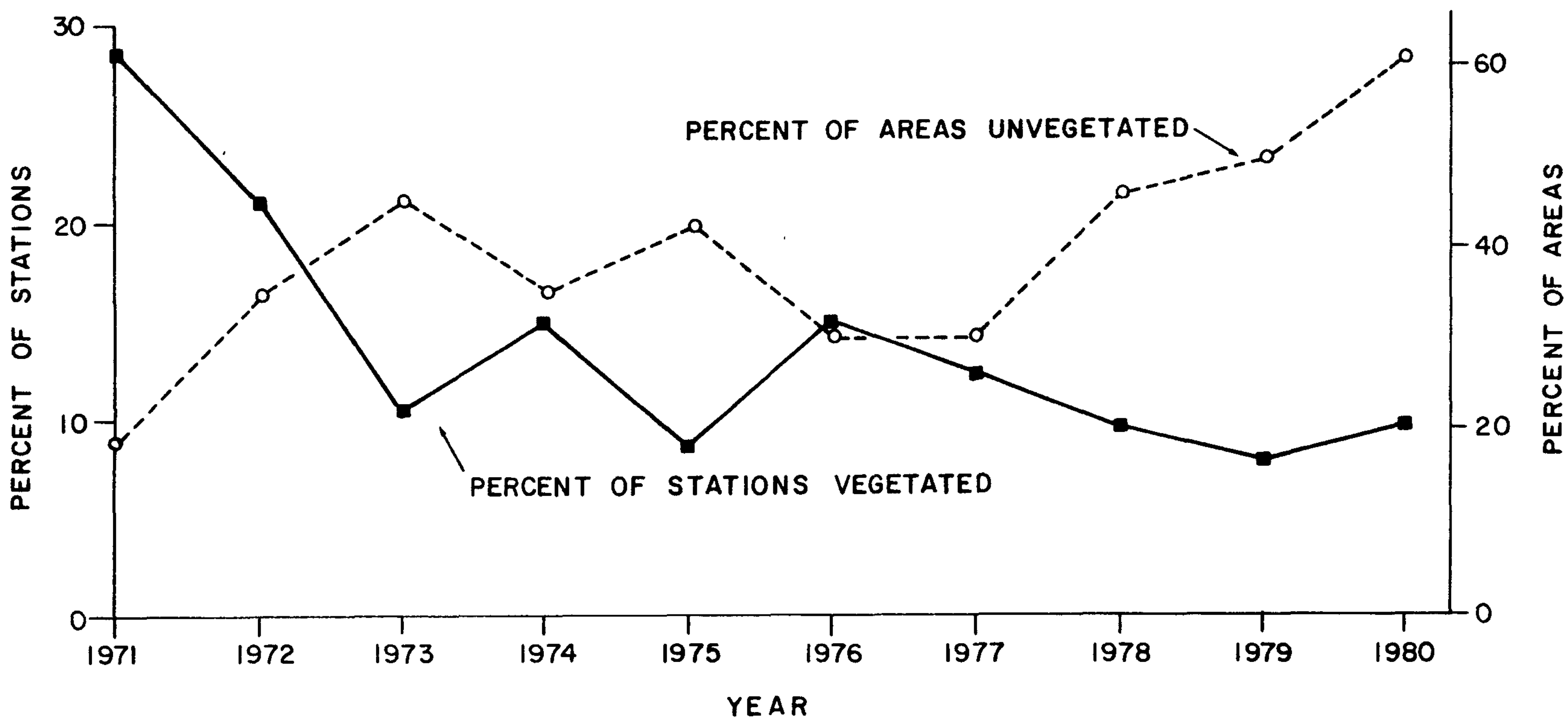

Figure 8. Trend in SAV occurrence in the Maryland portion of Chesapeake Bay. Values represent the percent of stations with SAV (n = 644 stations) and the percent of unvegetated areas (n = 26 areas) (from Kerwin et al. 1977; unpublished data from Kerwin et al. 1977; unpublished data from Maryland's Department of Natural Resources).

brackish water species: coontail (C. demersum), common elodea (E. canadensis), southern naiad (N. guadalupensis), wildcelery (V. americana), sago pondweed (P. pectinatus), and redhead grass (P. perfoliatus) (Table 1).

Vegetation in the middle and lower zones of the Bay started to decline in 1973. In the middle zone, regions affected were: the Choptank and Little Choptank Rivers, James Island, Manokin River, Big and Little Annemessex Rivers, and Bloodsworth and Smith Islands. Species affected in these areas included many of the same low salinity species that were rapidly lost from the upper Bay section in 1972 as well as the higher saline species, eelgrass and widgeon grass. The decline of SAV at some locations on the lower eastern shore where eelgrass and widgeon grass had predominated is shown in Figure 9.

In the lower zone, where data are available primarily from detailed aerial photographs (Orth and Gordon 1975, Orth et al. 1979), vegetation in the York, Rappahannock, and Piankatank Rivers, as well as in many small tributaries, was reduced substantially during this period (Figure 7). To highlight the changes that occurred with SAV communities in the lower Bay, six areas were mapped for historical changes in the distribution and abundance of SAV (Orth et al. 1979). These changes are shown in detail for one of the sites: Mumfort Island in the York River (Figure 10). SAV coverage in the lower Bay generally increased at all these sites from the 1930s to 1970; there was a marked decline beginning around 1970 (Figures 10 and 11). Our data, especially for the York River, indicated that the decline of SAV occurred in the summer of 1973, as evidenced by the presence of large beds of SAV in April 1973 that were absent in April 1974. Comparison of means indicated that there were significant differences between pre-1972 and post-1972 coverages at Parrott Island in the Rappahannock River ($p=0.001$), Mumfort Island in the York River ($p=0.002$), and East River in Mobjack Bay ($p=0.038$). At Jenkins Neck at the mouth of the York River, where the trend was more gradual, regression analysis indicates a significant decline ($p=0.02$). At Fleets Bay, just above the mouth of the Rappahannock River, regression analysis indicates the decline was significant ($p=0.019$). Only Vaucluse Shores on the eastern shore showed no significant decline ($p=0.14$).

Several distinct patterns in the decline of vegetation in the lower Bay are evidenced. First, it appears that losses of vegetation were greatest in all the areas where eelgrass formerly reached its upriver or upbay limits. For example, eelgrass beds disappeared from the Maryland portion of the eastern shore while remaining in the Virginia portion. Along the western shore of the lower Bay, SAV beds declined the most in the northern areas and least in the southern areas. Within the major tributaries, SAV disappeared, leaving only some beds at the mouths of the rivers. In nearly all the small creeks and tributaries where eelgrass beds continued to exist in 1975, the former distribution included areas further upstream. Second, in addition to the upstream-downstream movement, it appears that the vegetation declined in the deeper, offshore sections of the beds rather than in the shallower, nearshore areas (Figure 2).

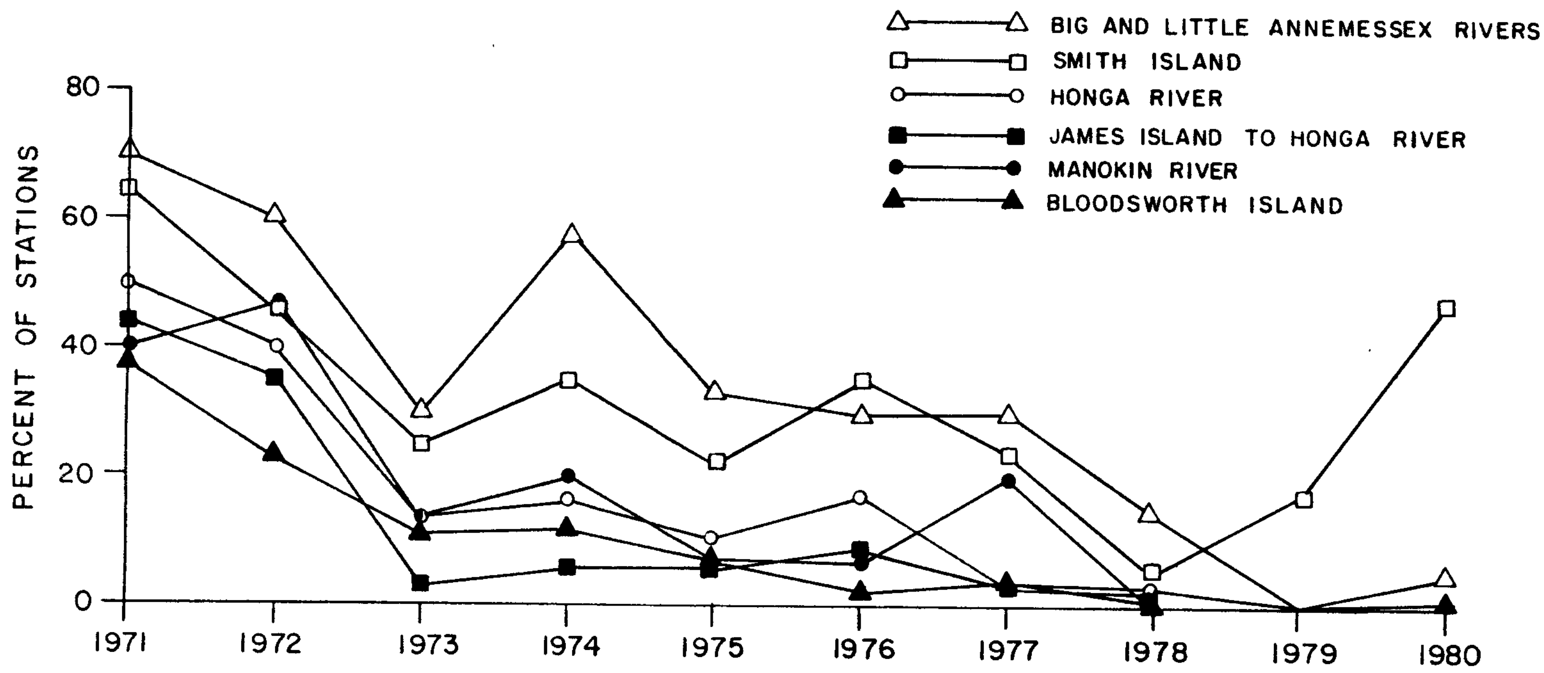

Figure 9. Trends in SAV occurrence in six areas in the middle Bay zone where SAV had markedly declined (data from Kerwin et al. 1977; unpublished data from Maryland's Department of Natural Resources).

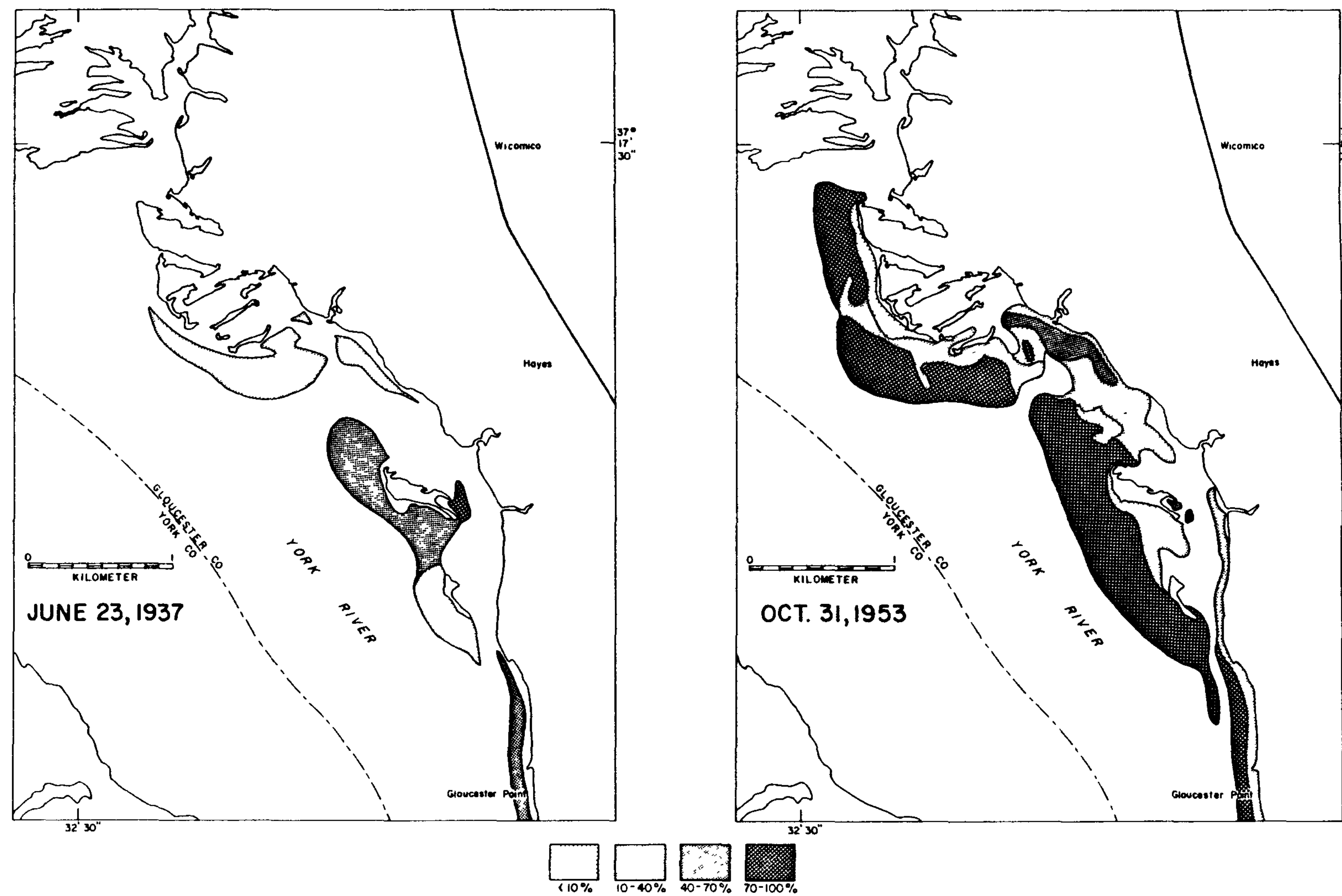

Figure 10. Changes in the distribution and abundance of SAV at the Mumfort Island area in the York River, 1937-1978. Density of SAV shown as very sparce (10% coverage), sparce (5-40%), moderate (40-70%), or dense (70-100%)(data from Orth et al. 1979).

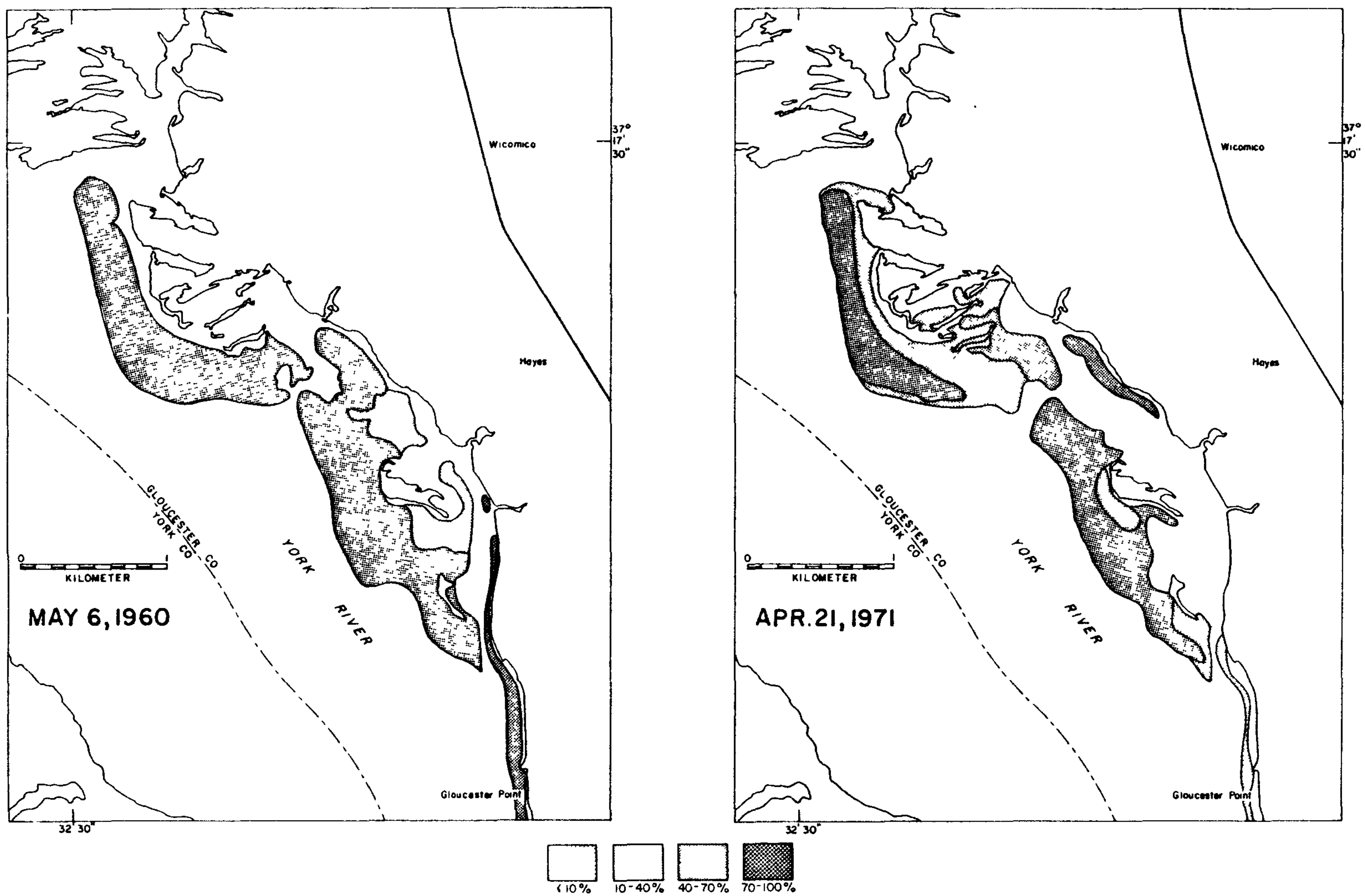

Figure 10. (continued)

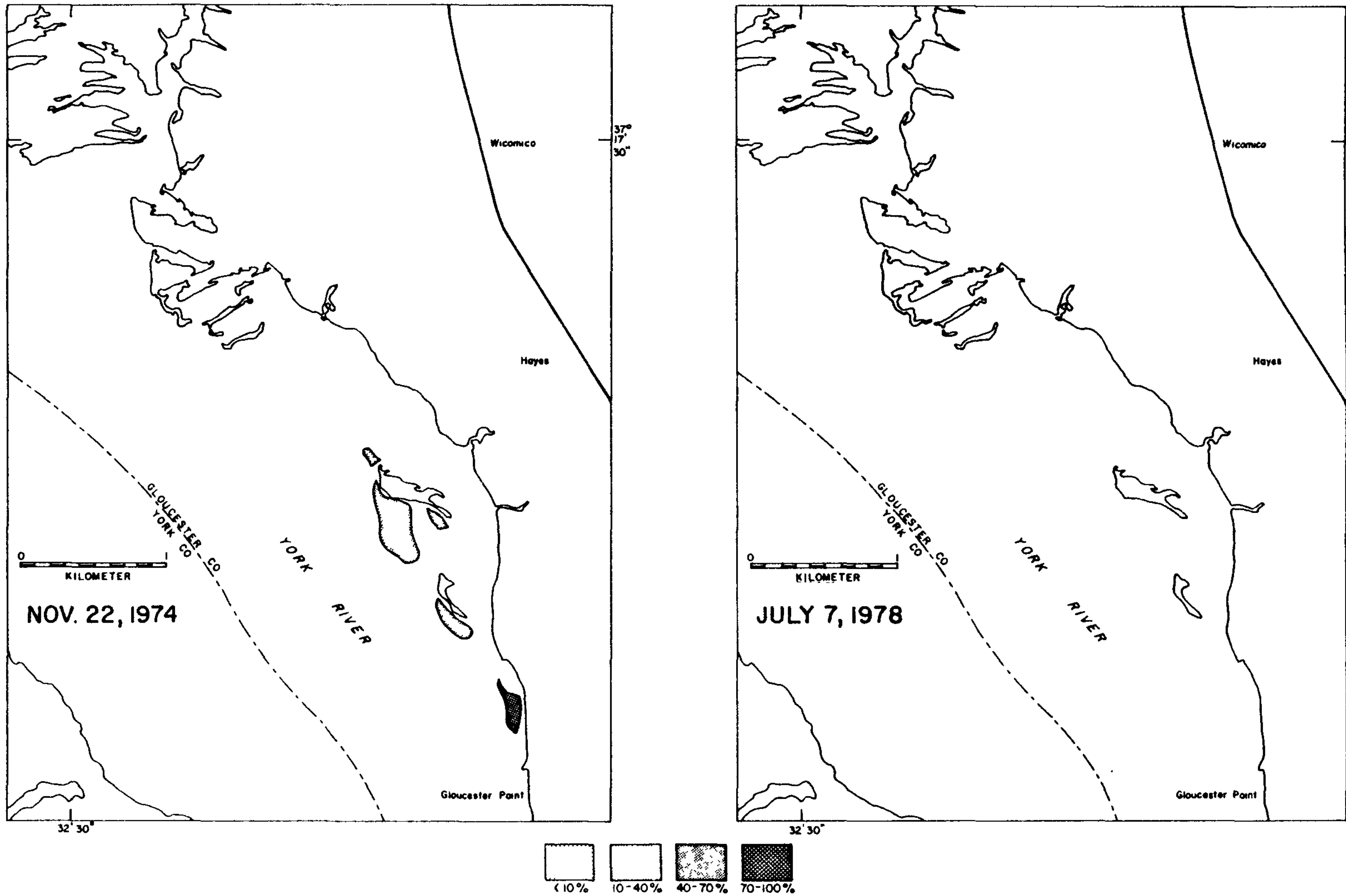

Figure 10. (continued)

1975-1980--

Between 1975 and 1980, the Bay-wide status of SAV appeared to be one of continuing decline in almost all areas of the Bay (Figure 11). The upper Bay survey by the Maryland Department of Natural Resources continued to show a small percentage of stations vegetated with SAV with a trend toward decreasing levels to 1979 (unpublished data). A small increase was observed in 1980, but this was due to a large increase in vegetated stations at the Smith Island site (Table 5 and Figure 9). All sites, where a decline in abundance in the early 1970's from the lower eastern shore was observed, except for Smith Island, continued to decline to much lower levels (Figure 9). Another significant point was the continual increase in the number of areas that contained no SAV. By 1980, 16 areas, or 62 percent of the total areas identified for this survey now contained no SAV, compared with five areas or 19 percent in 1971 (Table 5 and Figure 8).

In the lower Bay zone, the total for the mapped areas of the western shore from the Rappahannock River to the James River between 1974 and 1978 remained similar (Table 3). Although there were observed declines, losses were offset by increases in the sizes of some grassbeds, especially those in Mobjack Bay. Losses were observed in many of the smaller beds that remained in some localities after the 1973-1974 period, but had totally disappeared by 1978, particularly in Fleets Bay, where 76 percent of the vegetation mapped in 1974 declined by 1978. Between 1978 and 1980, almost all sections of the lower Bay declined. Now, in some sections (Rappahannock River and Reedville), almost no SAV remains (Table 3, and Figures 11 and 12).

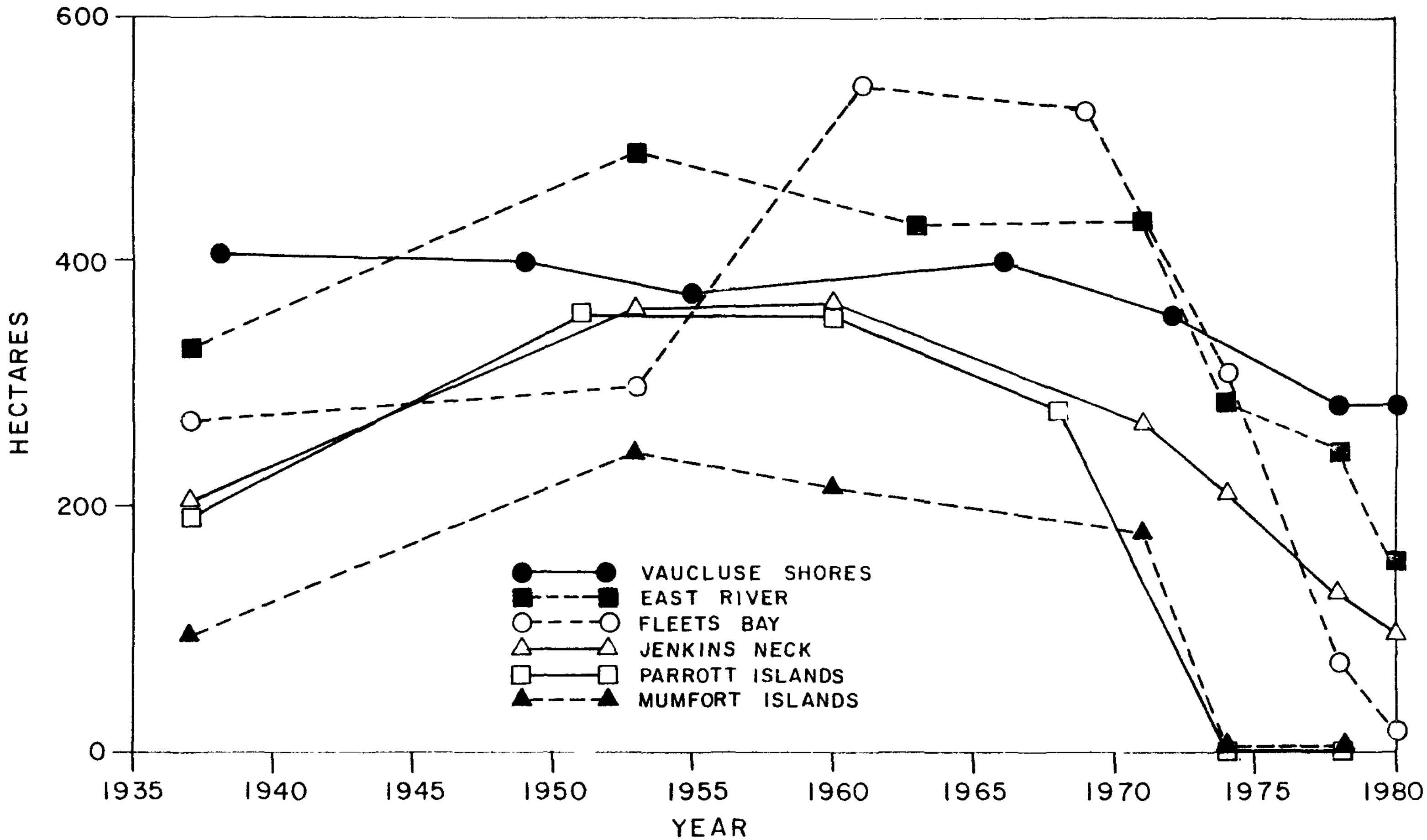

Figure 11. Trends in SAV coverage at six sites in the lower zone of Chesapeake Bay (data from Orth et al. 1979).

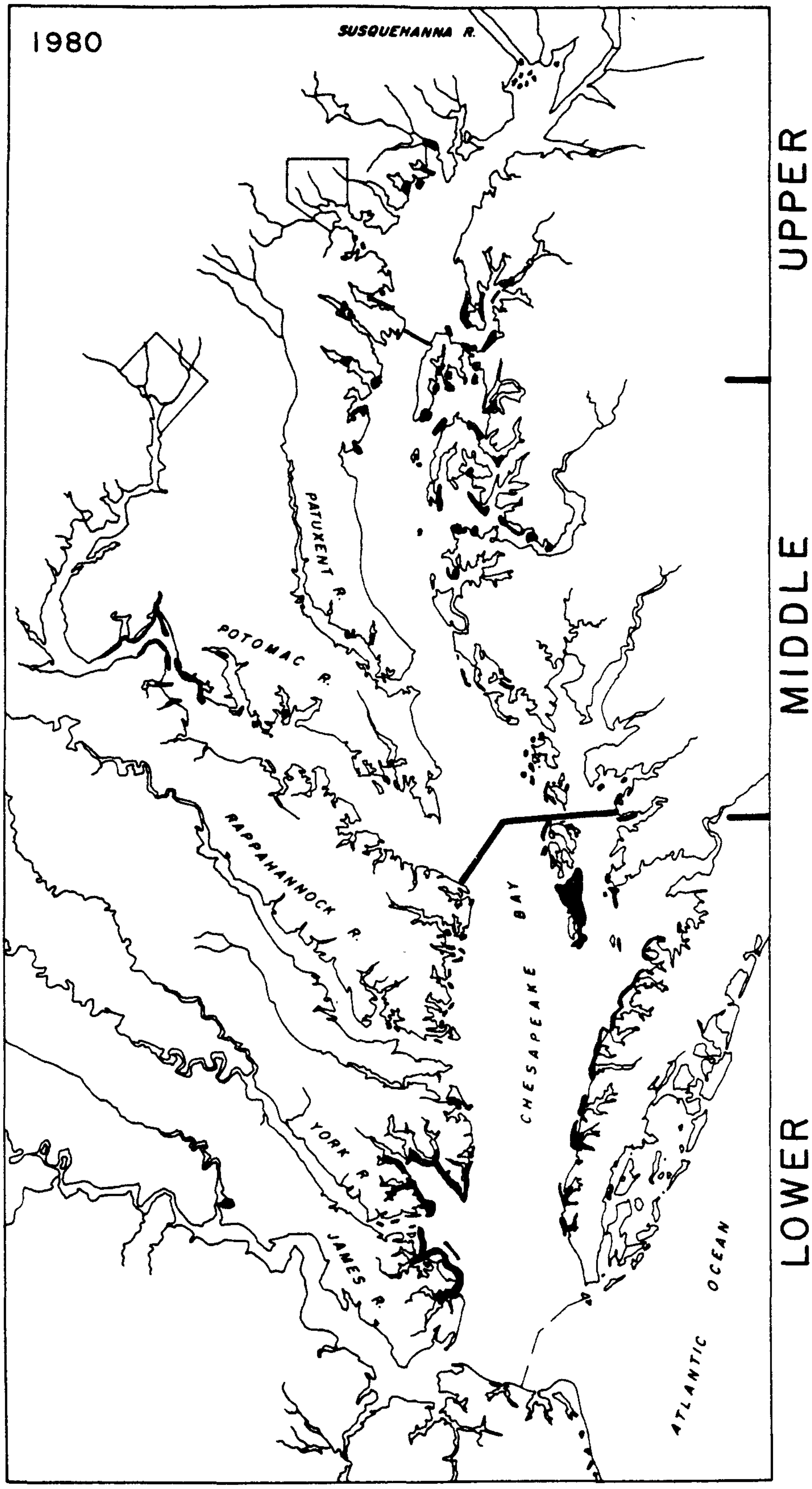

Figure 12. Distribution of SAV in Chesapeake Bay - 1980.

SECTION 5

THE ATLANTIC COAST

There is little evidence to suggest that there have been recent significant changes in SAV distribution along the east coast of the U.S. comparable to those documented for Chesapeake Bay. The uniqueness of the Chesapeake Bay estuary, with its extensive littoral areas and marked salinity gradient makes comparisons difficult. In addition, only recent interest, by the scientific community and management agencies in SAV communities, has resulted in any significant work on the historic distribution of SAV in other areas.

Eelgrass is a species distributed widely along the coastline of the eastern United States and Canada, from North Carolina to Nova Scotia. As mentioned in the previous section, eelgrass populations underwent a dramatic reduction along the east coast of the U.S. in the 1930's. This decline had dramatic effects on waterfowl populations, fisheries, and shoreline erosion. Declines in other years were noted by Cottam (1934, 1935), but recovery always followed these declines in most of the reported areas. At present, North Carolina, which has extensive beds of eelgrass located within its bays and sounds, with a few beds found along the tidal rivers, is attempting to determine the present distribution of SAV in the region. Researchers in the area report no apparent widespread changes in eelgrass distribution in the last 10 years (M. Fonseca, G. Thayer, personal communication). There have been localized changes in eelgrass beds, but these have been due to physical perturbations by man or to other localized distrubances. Davis and Brinson (1976) report on the distribution of SAV in the Pamlico River, but again report no significant, recent changes in their abundance. In South Carolina and Georgia there are, at present, no significant stands of SAV, primarily because of the very turbid conditions that exist in the estuaries found there.

North of Chesapeake Bay there appears to be no SAV in the Delaware Bay at present, and data on whether it ever occurred there are not available. In New Jersey, SAV beds dominated by eelgrass and widgeon grass are found in the sounds located to the west of the barrier islands (Good et al. 1978, Macomber and Allen 1979). There is a lack of historic data on SAV in the region but, again, there is no direct evidence of any large scale changes in the existing beds.

New York researchers indicate no reports of significant losses in eelgrass beds; on the contrary, eelgrass appears to be increasing in abundance (Churchill, personal communication).

Rhode Island SAV beds persist in many of the small tidal lagoons adjacent to Long Island Sound. These systems still contain abundant vegetation and apparently have not undergone recent significant alterations.

In Massachusetts, Maine, Canada, and Rhode Island, there have been no reports of changes in SAV communities. Accurate data are lacking, however, because there are no scientists presently involved in any extensive SAV research programs.

In summary, it appears that the declines in eelgrass or other SAV species in the Bay are not part of a widespread and synchronous loss of vegetation along the east coast of the U.S., although these conclusions are hampered by the lack of comprehensive data on the current and historical distribution of SAV in other areas. It is most likely that the water quality problems affecting the distribution of grasses in the Bay are regional in nature, involving the Bay, its tributaries, and their drainage basins.

SECTION 6

WORLDWIDE PATTERNS

As in Chesapeake Bay, many coastal and estuarine regions of the world contain varying amounts of shallow water areas that support SAV beds ranging from large, very dense areas in the Caribbean to small, sparse areas in some European countries. The grass beds around the world occur under a wide range of physical, chemical, and biological parameters. Yet despite these differences, they share a common ground in their functional roles in their respective ecosystem: a habitat and nursery area, a food source for waterfowl, a sediment stabilizer, a nutrient buffer, and a source of detritus. Recent interest in SAV systems worldwide has paralleled the increasing interest in the role and value of Bay SAV systems and an interest in their proximity to industrialized areas, causing them to become increasingly stressed by man-made perturbations. Recent examples from the Netherlands (Nienhuis and DeBree 1977, Verhoeven 1980), England, (especially some very pertinent examples from freshwater areas) (Wyer et al. 1977, Eminson 1978, Phillips et al. 1978), Wales (Wade and Edwards 1980), Scotland (Jupp and Spence 1977), Denmark (Sand-Jensen 1977, Kiorboe 1980), France (Peres and Picard 1975, Maggi 1973, Verhoeven 1980), Israel (Litav and Agami 1976), Australia (Cambridge 1975, Larkum 1976), Japan (Kikuchi 1974a, 1974b) and the Virgin Islands (Van Epoel 1971), suggest that losses in SAV communities are highly correlated with changing water quality conditions. In many of the above examples, where SAV has been described as greatly reduced or declining, this reduction has always been associated with decreasing water clarity as a result of increased eutrophication, with subsequent increases in epiphytes and phytoplankton due to sewage or agricultural inputs, or as a result of higher loads of suspended sediments due to dredging or runoff from deforested areas.

On the other hand, increases in water clarity have been shown to result in expansion of SAV. The diking of the Gravelingen estuary in the Netherlands resulted in a salt water lake with reduced currents and no tidal effects. This resulted in a reduced total suspended solid load, and, thus greater light penetration. Subsequently, eelgrass increased almost 400 percent in 10 years and was found in water depths of up to five meters, far deeper than before the diking (Nienhuis 1980).

Large reductions of SAV communities have also been associated with natural causes of diseases. The eelgrass wasting disease of the 1930s, which resulted in massive declines of eelgrass along the east coast of the U.S. and west coast of Europe was originally attributed to a disease organism, *Labyrinthula*, but later attributed to climatalogical changes in temperature (Rasmussen 1973, 1977). In Australia, decline of SAV was attributed to migrating sand waves that smothered the grasses (Kirkman 1978). However, the more recent declines cited in the literature have been associated with man-induced alterations rather than with natural ones.

There are still vast areas of SAV in many parts of the world, particularly in the Gulf of Mexico, the Caribbean, and Australia that are not presently affected by industrial or urban development [one area in southern Florida was estimated to have 500,000 ha (1,235,000 acres) of turtlegrass (*Thalassia testudinum*) (J. Zieman, personal communications)].

SECTION 7

CONCLUSIONS

The period of 1965 to 1980 represents what we feel was an unprecedented decline of SAV in Chesapeake Bay. Loss of SAV communities was first observed in the late 1960's in the upper Bay areas, and in particular, the Patuxent, lower Potomac River (SAV beds in the freshwater tidal portions had been absent since the 1930's), and the upper reaches of some of the smaller tributaries (for example, the Chester and Choptank Rivers). By 1970, almost all the vegetation in the Patuxent River and lower Potomac River was gone. The decline of SAV in the Bay accelerated in the early 1970s and continued through 1980, with the most rapid decline occurring from 1972 to 1974. Several sections in the Bay that once contained abundant SAV virtually had none by 1980 (for example, the Patuxent, Piankatank, and Rappahannock Rivers); other sections had only small stands remaining (for example, the Potomac and York Rivers, and Susquehanna Flats). In addition to this trend of SAV populations declining from "up-estuary" to "down-estuary", it appears that within individual beds the declines occurred first in the areas of greatest depth.

The present abundance of all SAV species in the Bay [16,000 ha (39,520 acres)] is probably the lowest level recorded in the Bay's history. Figure 13 shows this cumulative pattern of decline over the last 20 years, with the arrows representing the former to present limits of distribution. Figure 14 outlines these sections of the Bay where SAV has been most severely affected.

SAV in the Bay has experienced other large scale changes in the recent past, although none involving so great a spectrum of species types. In the 1930's, a decline of SAV primarily involved eelgrass except for the tidal freshwater portion of the Potomac River where all SAV species disappeared. Eelgrass gradually returned to all areas of the Bay, but there has been little regrowth of SAV in the upper Potomac. In the late 1950's and early 1960's, the sudden rapid expansion of Eurasian watermilfoil created problems by choking many waterways in sections of the Potomac River, Susquehanna Flats, and western tributaries of the upper Bay.

On a much broader latitudinal scale, the entire east coast of the United States and the west coast of Europe, eelgrass populations also declined during the 1930's. This decline was subsequently followed by a gradual return in most areas. Near Chesapeake Bay, in the shallow lagoons behind the barrier islands of the Delmarva Peninsula, the eelgrass has never recovered. This has drastically affected the scallop industry that was associated with this species of SAV. Regarding the decline of SAV in the 1960's and 1970's in Chesapeake Bay, there is little evidence yet to suggest that a simultaneous decline occurred with SAV communities in other areas along the east coast of the United States. Reports indicate that on a worldwide basis, despite their abundance in certain areas, SAV communities are becoming increasingly affected by man-induced perturbations, declining in areas where there is extensive industrial and/or urban development.

Given the current situation, a very important question can be raised as to the ability of these systems to return to their previous levels of

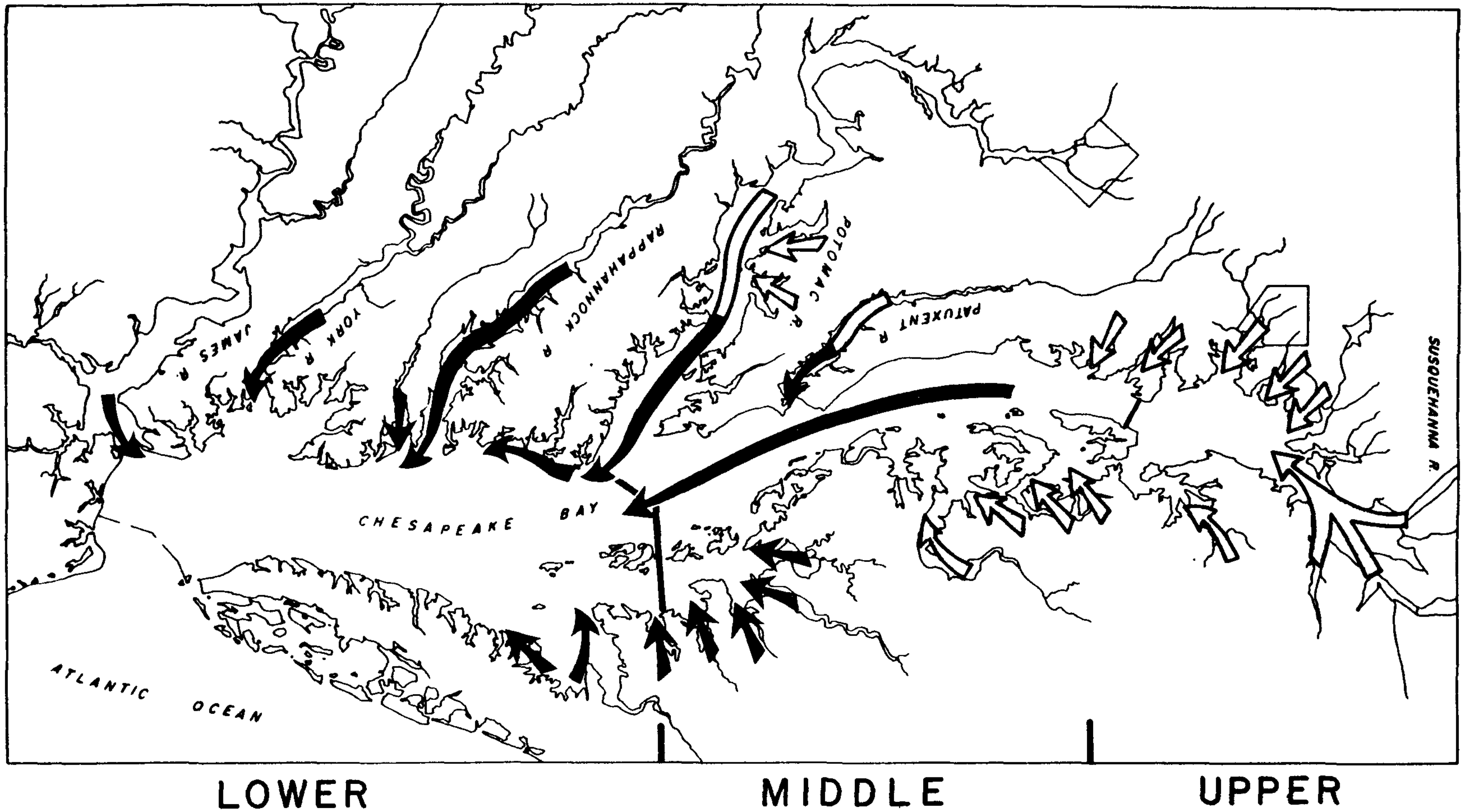

Figure 13. Pattern of recent changes in the distribution of SAV in Chesapeake Bay. Arrows indicate former to present limits. Solid arrows indicate areas where eelgrass (Zostera marina) dominated. Open arrows indicate other SAV species. (Orth et al. 1982)

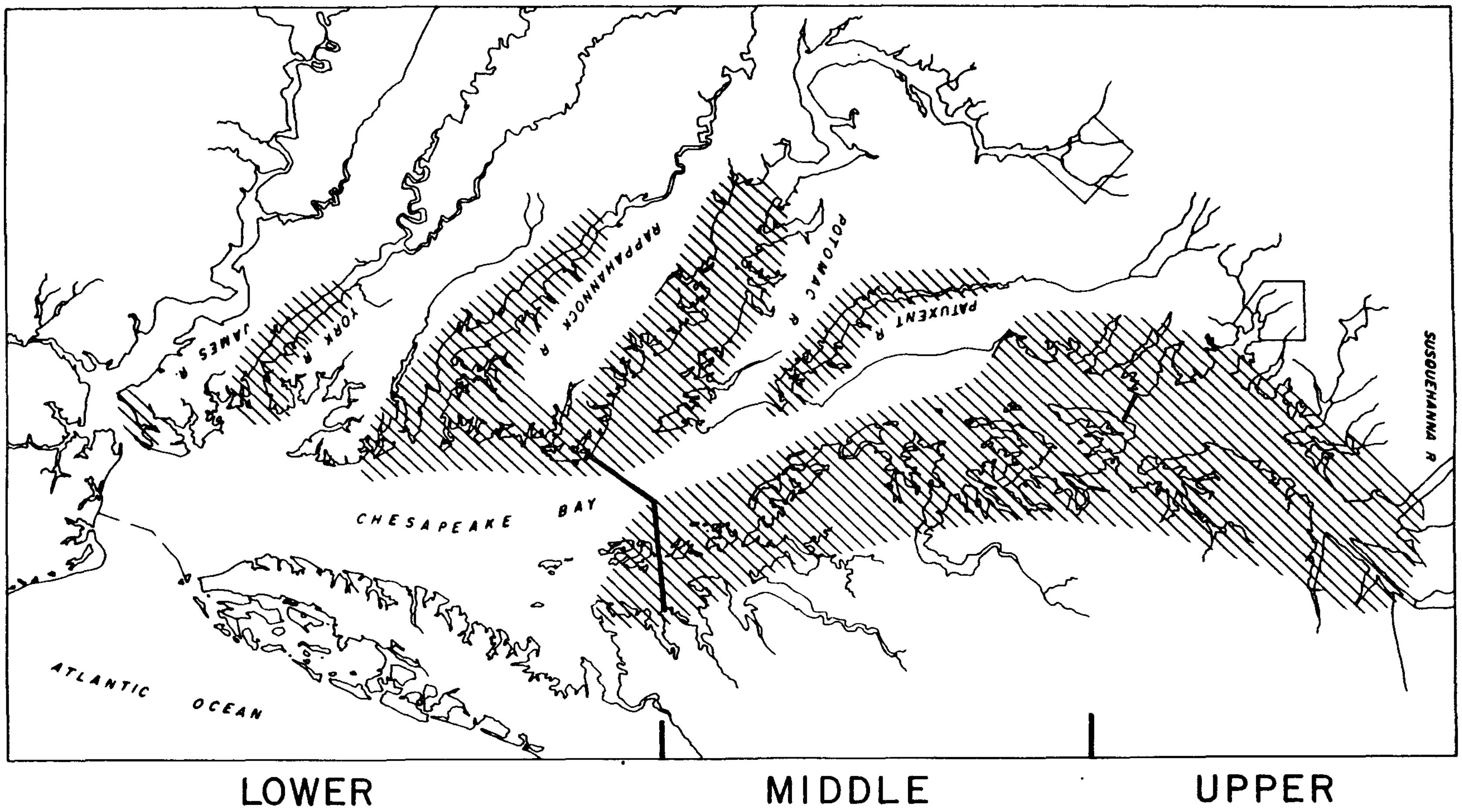

Figure 14. Location of sections of the Bay where SAV has experienced the greatest decline.

abundance in the Bay. Indeed, recovery may not occur because the current levels of SAV are so low or non-existent that natural recruitment via vegetative propagation or seed dispersal may be limited. Recent success with SAV transplantation experiments, moving whole plants into denuded areas in the Potomac River and lower Bay, indicates that these regions may now be capable of supporting SAV (Orth et al. 1981; V. Carter, personal communication). Thus, transplanting SAV may be a viable method, and in some areas the only way, for the reintroduction of these plant communities.

The future of SAV in Chesapeake Bay is one of uncertainty. We know that historically there have been several periods of SAV decline in the Bay. The vegetation has returned to some areas; others have remained barren. The pattern of continued decline of SAV in the Bay over the last 20 years suggests a chronic deterioration of water quality. Unless the complex interaction of factors leading to this deterioration can be understood and reversed, SAV communities in many areas may remain a part of the Bay's past.

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ECOLOGICAL ROLE AND VALUE OF

SUBMERGED MACROPHYTE COMMUNITIES:

A Scientific Summary

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SECTION 1

INTRODUCTION

Documentation of past distribution and abundance of submerged aquatic vegetation (SAV) within Chesapeake Bay began in the late 1800's, but information was sparse until the 1950's when surveys were initiated in the upper reaches of the Bay. Recent analyses of SAV seed distributions in sediment cores taken from various locations in the Bay (Brush et al. 1980), and reviews of old aerial photographs (Anderson and Macomber 1980, Orth 1981) confirm the concept that over historical time SAV was a diverse, abundant, and widespread feature of Chesapeake Bay. However, in the last two decades drastic changes in this component of the Bay ecosystem have occurred. The results of annual field surveys, several aerial surveys, and recent field studies all support the conclusion that SAV in the Bay has changed in species density, diversity, abundance, and distribution. This decline might be of minor concern if it involved the disappearance of only one or two species of SAV, or if the decline were part of a normal ecological cycle from which SAV would recover. Data indicate, however, that the majority of SAV species has been negatively affected, that the recent decline is not a part of a repetitive cycle, and that this phenomenon is Bay-wide. The documentation of this decline, coupled with consideration of possible ecological and commercial implications, provided the motivation to initiate intensive studies of the role and value of SAV communities in Chesapeake Bay. Locations of major study sites for the Bay Program research in Chesapeake Bay are indicated in Figure 1.

Current information concerning SAV communities indicates that they possess several important ecological features. Of these, four distinct hypotheses were examined in the Bay Program: (1) estimating the magnitude of SAV organic matter production available to food webs; (2) examining the habitat value of SAV to infaunal and juvenile nekton species; (3) estimating the role of SAV in modifying, reducing, and serving as a sink for nearshore sediments; and (4) examining the role of SAV in modifying nutrient dynamics of nearshore areas. This paper discusses these hypotheses.

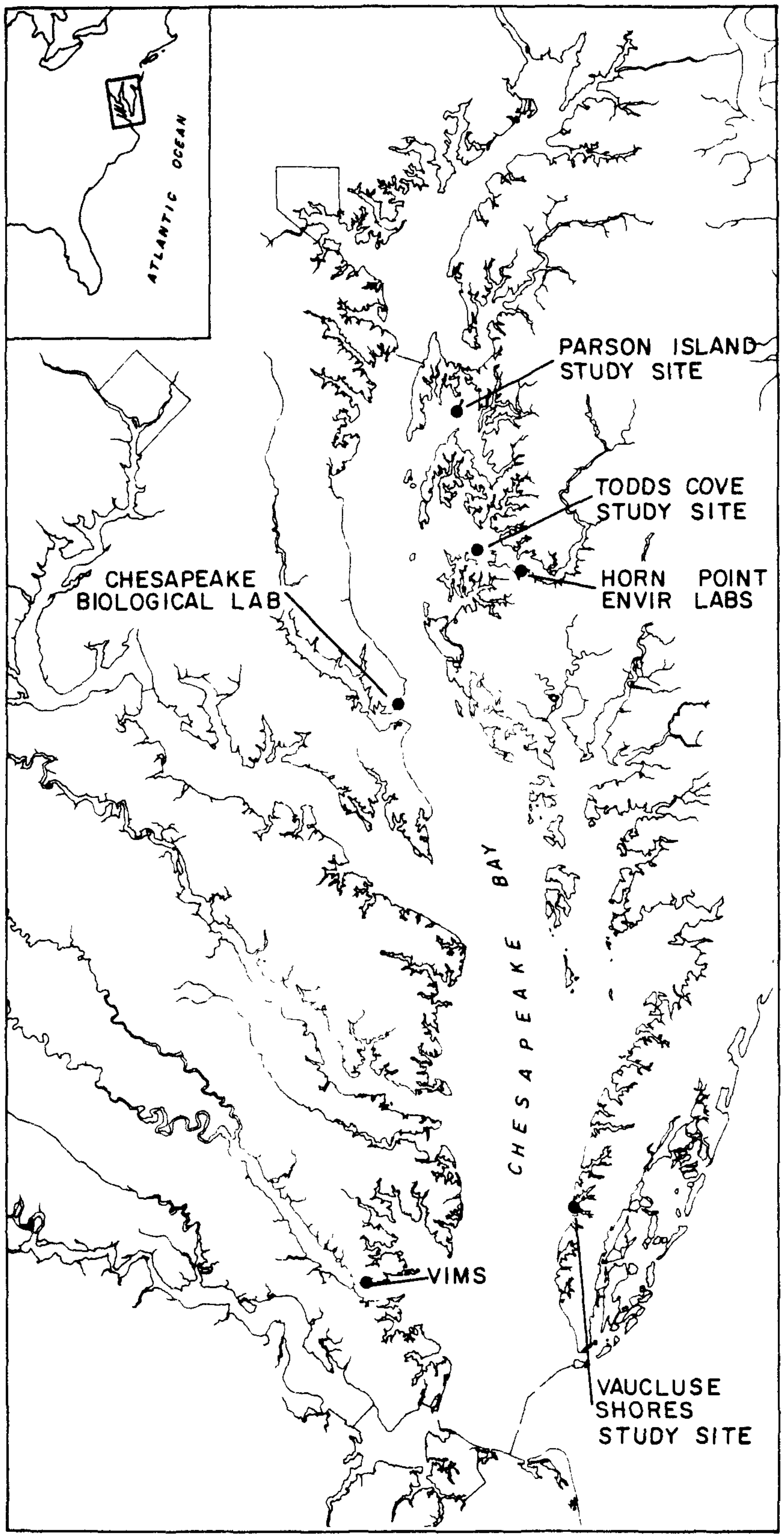

Figure 1. SAV intensive study sites for the upper and lower Bay.

SECTION 2

THE IMPORTANCE OF SAV PRODUCTION

APPROACH

In this section, the importance of SAV production (both above-and-below-ground biomass) is assessed from several points of view. First, the magnitude of SAV organic matter production in the Bay is compared with values obtained from global literature. This information sets the range of values in the Bay against major variables such as latitude and environmental gradients. Second, seasonal patterns of biomass and production of major regions of the Bay are examined. This analysis provides insight into the timing of environmental controls (such as temperature and salinity) and into the availability of SAV community organic matter to the food web. Third, the magnitude of production among various autotrophic components (such as Bay grass, attached epiphytes, benthic microflora, macroalgae, and phytoplankton) is compared, because SAV total community production results from the additive nature of these components. Fourth, the relative contribution of organic matter by major sources (riverine input, marshes, benthic algae, phytoplankton, and SAV) to the Bay system is estimated. For the upper Bay (mouth of the Potomac River to the head of the Bay), we compared the magnitude of three major sources of organic matter in 1960 (pre-decline of SAV) and 1978 (post-decline of SAV). Finally, we assessed how organic matter produced by SAV is used in Chesapeake Bay food webs.

BACKGROUND

In numerous reviews, the productivity (or rate of biomass accumulation) of submerged aquatic macrophyte communities has been characterized as among the highest recorded for aquatic systems. For instance, McRoy and McMillan (1973) state "a seagrass meadow is a highly productive and dynamic ecosystem; it ranks among the most productive in the ocean." Phillips (1974) reports that productivity for the seagrass *Thalassia testudinum* (a tropical species) ranges from 200 to 3,000 $gCm^{-2}y^{-1}$ and, for *Zostera marina* (a temperate species), values up to 600 $gCm^{-2}y^{-1}$ have commonly been recorded. (Organic carbon is assumed to approximate 50 percent of the plant material on a dry weight basis.) These rates are comparable to those reported for such productive terrestrial systems as tropical rain forests and intensive agricultural fields (Odum 1971). Compared with available measurements of phytoplankton productivity (Boynton et al. 1981a), a major source of organic matter in many aquatic food webs, SAV rates are truly indicative of highly productive ecosystems. In some cases, SAV produces so much biomass that eradication is necessary. For example, in Chesapeake Bay during the 1960's, Eurasian milfoil (*Myriophyllum spicatum*), a non-native species, showed high biomass production at nuisance levels, and research focused on control through herbicides and mechanical removal (Rawls 1965).

Reviews of SAV distribution in Chesapeake Bay conducted by Orth (1981) and Anderson and Macomber (1980), appraisal of archival aerial photography, and anecdotal comments by long-term residents of the area all indicate that

SAV was once a ubiquitous component of the Bay system (in at least a qualitative fashion); the Bay-shore was fringed in a productive, habitat-rich green wreath. Although considerable scientific research has been conducted on several species of SAV (Thalassia testudinum, the tropical turtlegrass and Zostera marina, the temperate eelgrass), many characteristics of the freshwater-brackish species in Chesapeake Bay have received little attention prior to the late 1970's (Stevenson and Confer 1978). Since then, considerable effort has been expended in documenting biomass, productivity and other faunal characteristics, habitat values, relationships of productivity to food-web utilization, and nutrient requirements of SAV. This work was done when SAV was in a period of severe decline, particularly in the upper Bay. Therefore, results summarized here may have an inherent bias because SAV is now only a small component of the Bay system; however, we believe that our results present an appropriate perspective that adjusts for potential bias.

Values of net biomass production (Pa) observed in Chesapeake Bay appear to be quite similar to those observed in other temperate and semi-tropical SAV systems distributed over large latitudinal and environmental gradients (Figure 2). Values of Pa in this global sampling ranged from about two to 20 $gO_2m^{-2}d^{-1}$, and typical values were in the range of three to seven $gO_2m^{-2}d^{-1}$. Conversion of oxygen values to organic matter (assuming a photosynthetic quotient of 1.25, which is the ratio of oxygen evolved to carbon dioxide fixed photosynthetically and the carbon equivalent of organic matter of 0.5) gave typical values from about two to four grams of organic matter $m^{-2}d^{-1}$. These values are comparable to those associated with intensive agriculture and other highly metabolic ecosystems (Penfound 1956, Odum 1971). The highest average value found in the literature was for Z. marina (in Alaska) of about 20 $gO_2m^{-2}d^{-1}$.

In sharp contrast to the comparability of Pa values between SAV systems, estimates of SAV biomass showed high variability. For example, SAV biomass ranged from just a few g m^{-2} in some Chesapeake Bay communities to over 7000 g m^{-2} in a Thalassia meadow in Puerto Rico (Figure 2). Estimates of SAV biomass within the same system (Figure 2) also exhibited a large range. For example, biomass of Zostera in Alaska and Thalassia in Florida varied by factors of three to six. McRoy and McMillan (1973) suggest that such gradients reflect differences in local environments that promote or inhibit the accumulation of large standing stocks. In general, the highest standing stocks of SAV occur in areas where the water is relatively clear (light penetrates to the bottom), deep enough to allow for substantial vertical growth, and devoid of excessive wave action.

A second observation suggested in Figure 2 is that average biomass (and even maximum biomass) estimates in Chesapeake Bay communities are low relative to those reported for other areas. Average values of Zostera and Ruppia in the lower Bay were generally below 200 g m^{-2}, and values for Potomageton pectinatus and P. perfoliatus in the upper Bay were generally below 100 g m^{-2}. A quantitative evaluation of this observation is not possible because of the nature of available data; however, several reasons can be suggested. At the present time, sufficient light to support vigorous growth of SAV does not penetrate much beyond one meter in most littoral regions of the upper Chesapeake (Boynton et al. 1981a). Thus, growth is restricted to very shallow regions where there is a limited water

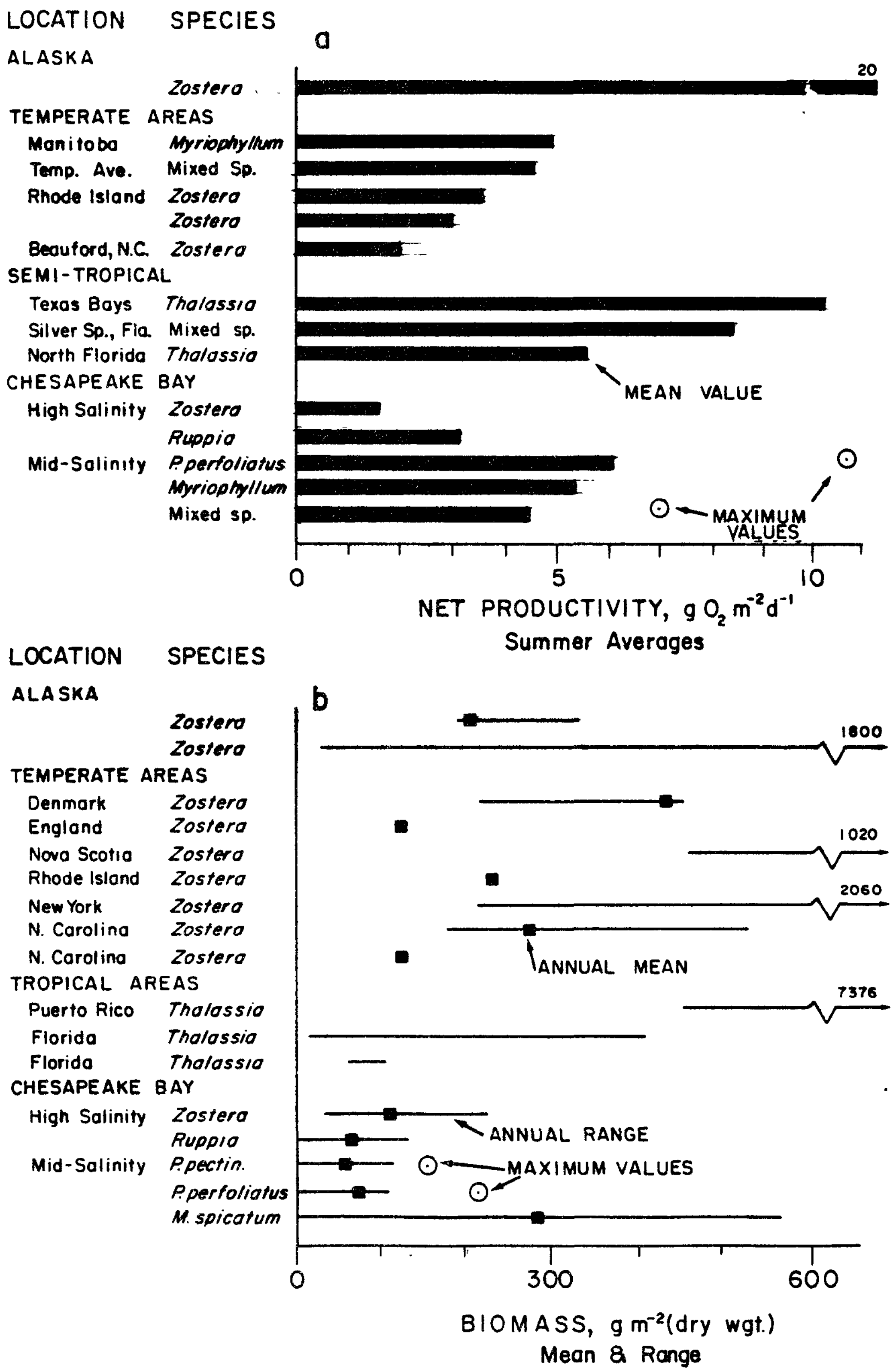

Figure 2. (a) SAV net productivity and (b) biomass in Chesapeake Bay and other SAV systems with selected values from Alaska, temperate, and tropical areas. Data from Kaumeyer et al. 1981, and McRoy and McMillan 1973.

column to support the vertical development of SAV, and the potential for wave, thermal, and waterfowl grazing stresses is maximized. In previous years (pre-1960) when light penetration was not so restricted, SAV in the upper Bay may have grown in waters of greater depth and were characterized by higher standing stocks.

SEASONAL PATTERNS OF BIOMASS AND PRODUCTION IN CHESAPEAKE BAY

Understanding seasonal patterns of primary production and standing stock is important in developing a better knowledge of SAV community dynamics. Productivity patterns provide insight to critical periods of SAV growth and to factors potentially limiting growth. Periods of high and low biomass indicate times of enhanced habitat and availability of organic matter to Bay food webs. Data concerning SAV were scarce prior to the initiation of the Chesapeake Bay Program; further work will refine the detail of the patterns reported in this section.

Estimates of SAV above-ground biomass based on a few locations (mean of three to six replicates and random quadrant of 0.10 to 0.25 m^2) for several species in the lower and upper Bay and for one introduced species are summarized in Figure 3. In a comparative sense, several things are apparent. First, peak biomass of M. spicatum, Z. marina, and R. maritima occurred in decreasing order, and biomass of R. maritima approximated that of P. pectinatus and P. perfoliatus. Second, with the exception of M. spicatum, mean biomass values were consistently higher in the lower Bay, often by a factor of two or more. Third, in the lower Bay, above-ground biomass persisted through winter months, but in the upper Bay above-ground material was present only during the warmer months. Finally, periods of peak biomass appeared to occur earlier in the year (June) in the lower Bay than in the mid-salinity zone (July to August).

Recent declines in SAV may have created changes in these biomass levels and seasonal patterns. It appears, however, that the SAV decline has been more severe in the upper Bay than in other locations (see chapter 1 of this Part). Quantitative information concerning biomass levels or seasonal persistence prior to the initiation of the decline is unavailable; however, anecdotal information suggests that general biomass values were higher in the upper Bay than they are at the present time. Data presented in Figure 3 for a mid-salinity site (Eastern Bay) support this idea.

Estimates of below-ground biomass, expressed as root:shoot ratios (RSR) differ both seasonally and geographically. They indicate that a higher proportion of the photosynthetic output of the plant is going into non-photosynthetic tissues (roots and rhizomes) that act as overwintering components (Schulthorpe 1967, Lipschultz et al. 1979) and sites of nutrient uptake (Penhale and Thayer 1980). Values of below-ground biomass were generally higher in the lower Bay than in the upper Bay, indicating that, for a unit of above-ground biomass, considerably more root-rhizome material was present in lower-Bay SAV communities (Figure 4). Furthermore, root-rhizome material was clearly present throughout the year at lower Bay sites. In the upper Bay, the situation is not so clear because of the limited sampling. At a site in the Choptank River (P. perfoliatus dominated), below-ground biomass persisted through at least part of the winter months; however, in a mixed R. maritima and P. pectinatus bed in Eastern Bay, below-ground biomass was not evident in late fall. Field

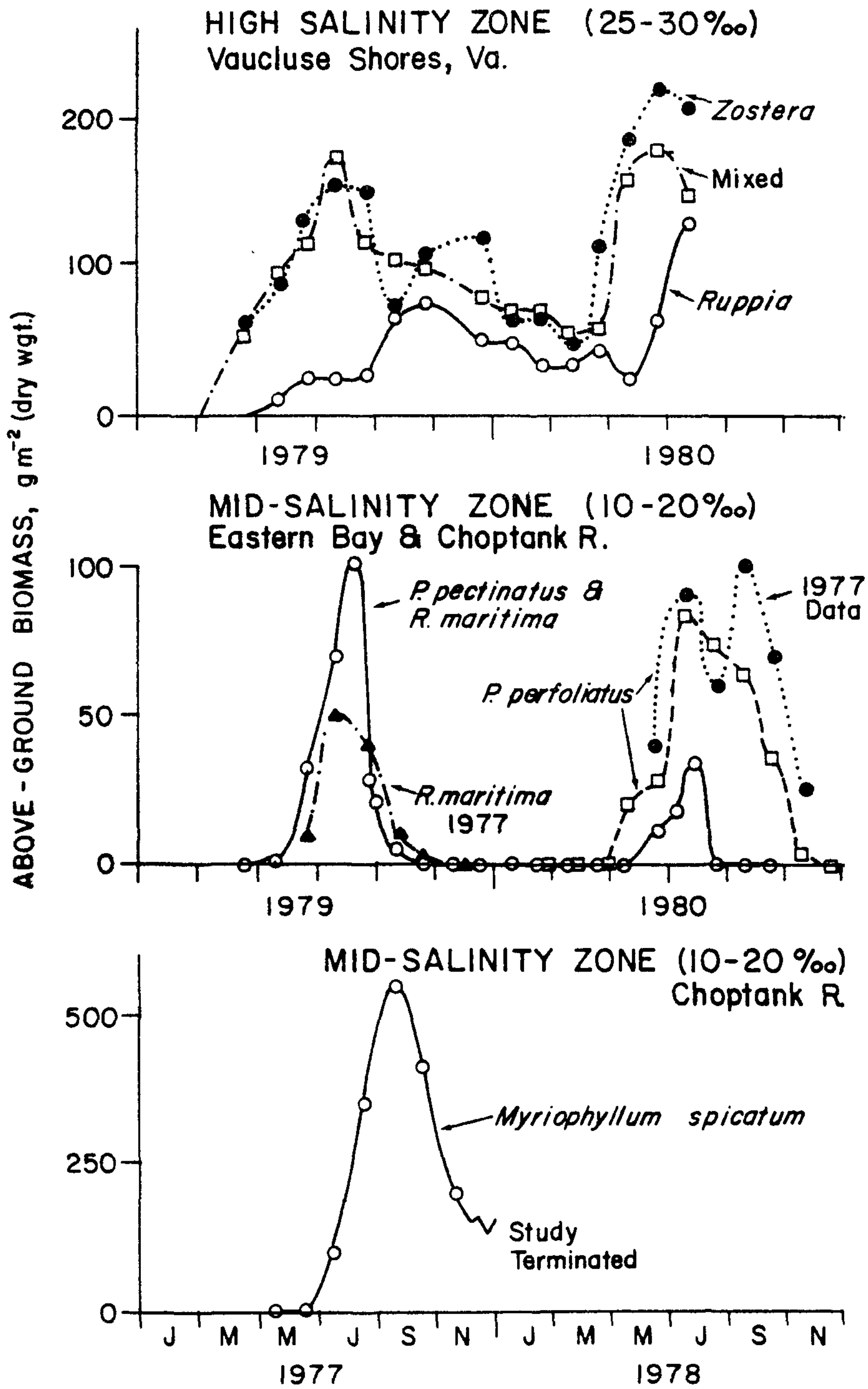

Figure 3. Estimates of above-ground biomass (gm^{-2} dry weight) in high and mid-salinity zones of Chesapeake Bay. Data from Kemp et al. 1981, and Wetzel et al. 1981.

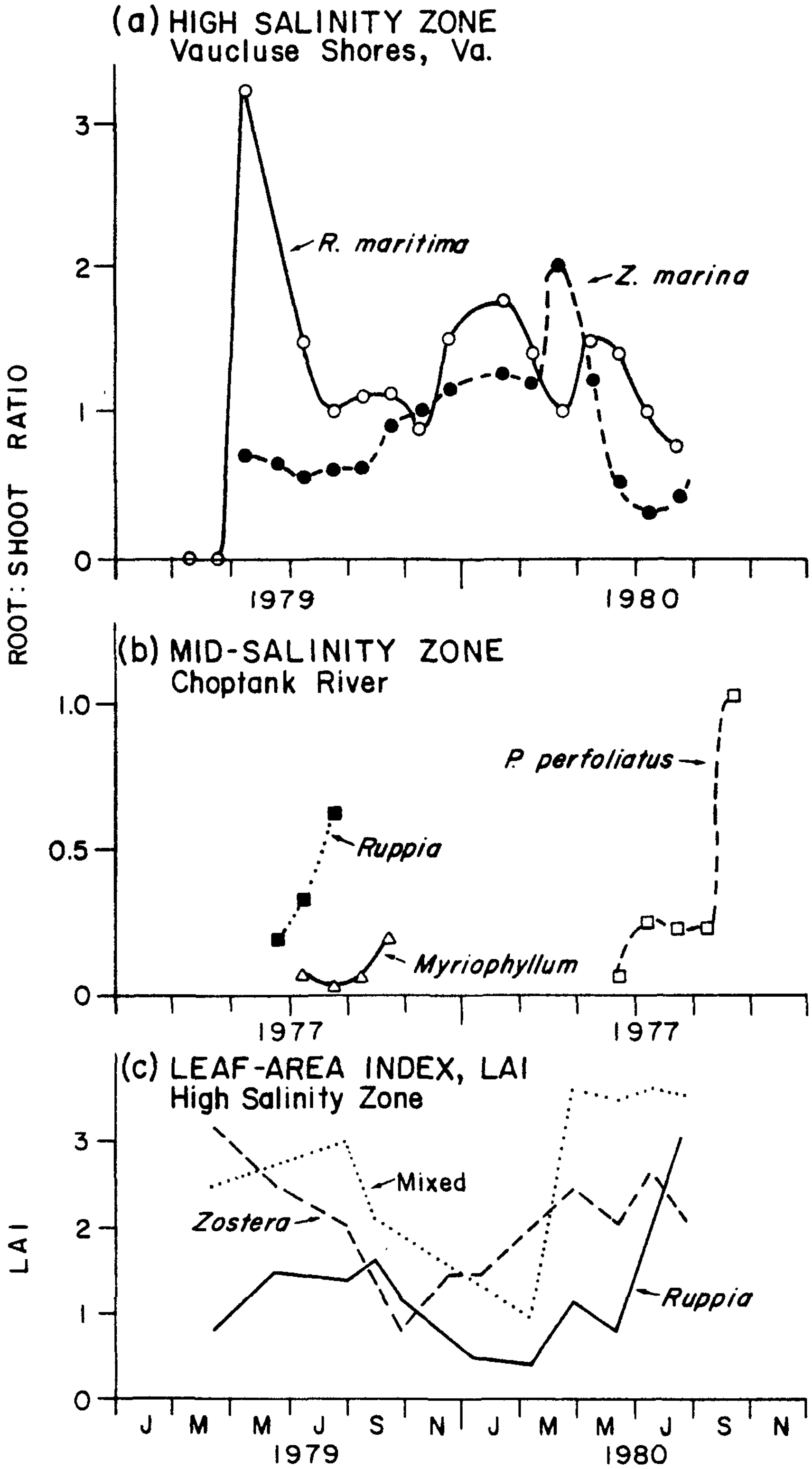

Figure 4. Seasonal patterns of root:shoot biomass ratios of selected species of SAV for (a) high salinity and (b) mid-salinity zones. (c) Shows the seasonal pattern in LAI for a high salinity area. Data from Kemp et al. 1981, and Wetzel et al. 1981.

observations confirmed that below-ground structures of SAV in the upper Bay appeared poorly developed relative to those in other areas of the Bay and very poorly developed relative to tropical Thalassia meadows. The low root to shoot ratios recently observed in the upper Bay may be indicative of stressed plants.

The leaf-area index (LAI), or the amount of photosynthetic surface per unit of biomass, is a fundamental characteristic of SAV community structure. LAI differences between SAV communities demonstrate the importance of light in regulating SAV communities and their adaptability to different light regimes. Increases in plant density can lead to potential increases in photosynthesis to a point, but can also lead to decreases in light availability through mutual shading. Data reported by Wetzel et al. (1981) exhibit differences between different SAV communities. Average LAI values are greatest in a mixed Zostera and Ruppia bed followed by successively lower values in pure stands of Zostera and Ruppia (Figure 4c). At the study site, the Ruppia bed was located in shallow water; the mixed and Zostera beds were at successively greater depths. The authors attribute the pattern in LAI to differing light regimes in these areas: the shallow Ruppia bed may have been photo-inhibited; the mixed bed near to optimal; and the deep Zostera bed intermediate because of insufficient light. Dennison (1979) reports a similar pattern for a Zostera bed and underscores both the importance of light in regulating SAV communities and the adaptability of SAV to different light regimes.

In addition to different mean LAI values, SAV communities exhibited differences in the vertical distribution of these values. In the Ruppia bed, values were greatest near the bottom of the canopy, presumably because of photo-inhibition nearer the surface. In the deeper, mixed, and Zostera communities, maximum LAI values were observed closer to the surface, probably owing to reduced light availability at greater depths. The maximum LAI values observed by Wetzel et al. (1981) were on the lower end of values reported for other seagrass communities (Jacobs 1979, Aioi 1980, Gessner 1971), suggesting that at least in these beds self-shading was not a major factor limiting light availability.

Estimates of seasonal net production rates (Pa) for SAV communities in the upper and lower Bay are summarized in Figures 5 and 6. In the mid-salinity environment, values of Pa correlate well with temperature, light, and SAV biomass. In general, rates were high during July and August when SAV biomass, light, and temperature were high and decreased sharply to lower values during the colder months. Figure 5 emphasizes the difference in community net production in vegetated and non-vegetated littoral areas. Clearly, during those periods of the year when SAV is present (May to September), the rate at which new organic matter is created is considerably higher in vegetated littoral areas.

Additional insights concerning the metabolic characteristics of SAV communities can be gained by comparing the ratio of Pa (new organic matter created during the day) to respiration (Rn: consumption of organic matter during the night). Data indicate that Pa:Rn is greater than 1.0 during the early SAV growth periods and that Pa:Rn is less than 1.0 during the late summer and fall. This observation suggests that most SAV biomass is generated in the early growing season; during the summer and fall, high daytime rates of Pa are observed, but the daily net production is consumed during the hours of darkness. Essentially, the metabolic demands of

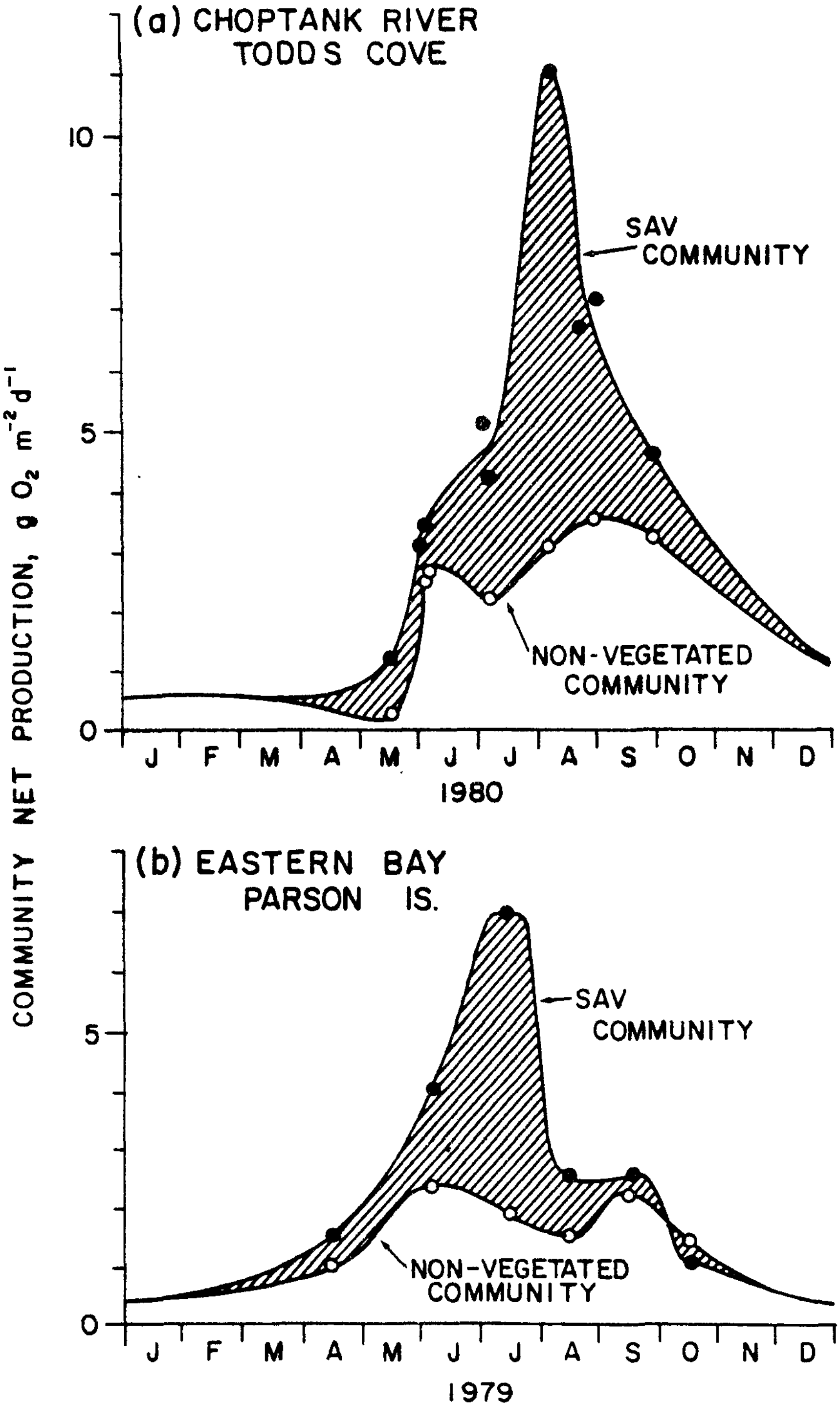

Figure 5. Net SAV community production in $gO_2m^{-2}d^{-1}$, including estimates of non-vegetated community production for (a) Todds Cove and (b) Parson Island. Data from Kemp et al. 1981.

non-photosynthetic organisms, or heterotrophs, and plant respiration matches or slightly exceeds the generation of new organic matter. Estimates of macroscopic heterotroph abundance (infauna, epifauna, and finfish for example) correlate with this pattern in that general abundances are low early in the growing season, but increase rapidly (as do metabolic demands) as the season progresses.

Seasonal patterns of Pa for a Zostera and Ruppia community in the lower Bay are given in Figure 6. Distinctive patterns emerge for each community. Rates were high in the Zostera community in the spring and fall with a summer minimum, but rates were highest in the Ruppia bed during the summer. The seasonal shifts in maximum growth may partly explain the successful coexistence of these two species. The values given in Figure 6 are in hourly units derived from measurements made prior to midday. Afternoon values were generally lower and often indicated a heterotrophic condition (Wetzel et al. 1981). The reason for this strong diel pattern in Pa is not known, but nutrient or CO_2 limitation is suspected.

ANALYSIS OF THE COMPONENTS OF SAV COMMUNITY PRODUCTION

This section places the various autotrophic components into perspective by comparing the relative contribution of organic matter produced by various autotrophic components of SAV beds, including epiflora, macroscopic algae, and benthic flora. Each component contributes a certain amount to the overall production of the community and provides a more or less desirable food source for the associated heterotrophic community.

Because of technical problems, temporal and spatial variability, and the time-consuming nature of the measurements, there appear to be only a few such studies available with which to compare results obtained in Chesapeake Bay. Estimates of production and biomass attributable to various autotrophic components of SAV communities are given in Table 1. However, from areas outside of Chesapeake Bay, available data suggest that epiphytes and macro-algae constitute a significant and, at times, a dominant feature of SAV community production and biomass.

Data from Chesapeake Bay are preliminary, but inspection suggests that epiphytic primary producers can constitute a substantial portion of the total community Pa. As we have shown earlier (Figure 5), phytoplankton production can also substantially contribute to overall SAV community production. There is little data to suggest that epiphyte or macro-algae constitute a substantial portion of community biomass.

One of the problems in interpreting these data involves the high variability associated with measurements of benthic and epiphytic production rates (Murray, pers. comm.). Apparently, short-term (day-week) changes in bottom sediments due to wave and tidal action can radically change benthic and epiphytic community structure and associated rates. Thus, estimation of seasonal or annual importance is particularly difficult. However, preliminary evaluations suggest significant, although not dominant, roles for epiphytes associated with SAV.

SAV PRODUCTION IN THE CONTEXT OF ESTUARINE ECOSYSTEMS

The importance of SAV production can also be assessed in terms of its contribution of organic matter to an estuarine system. In the shallow

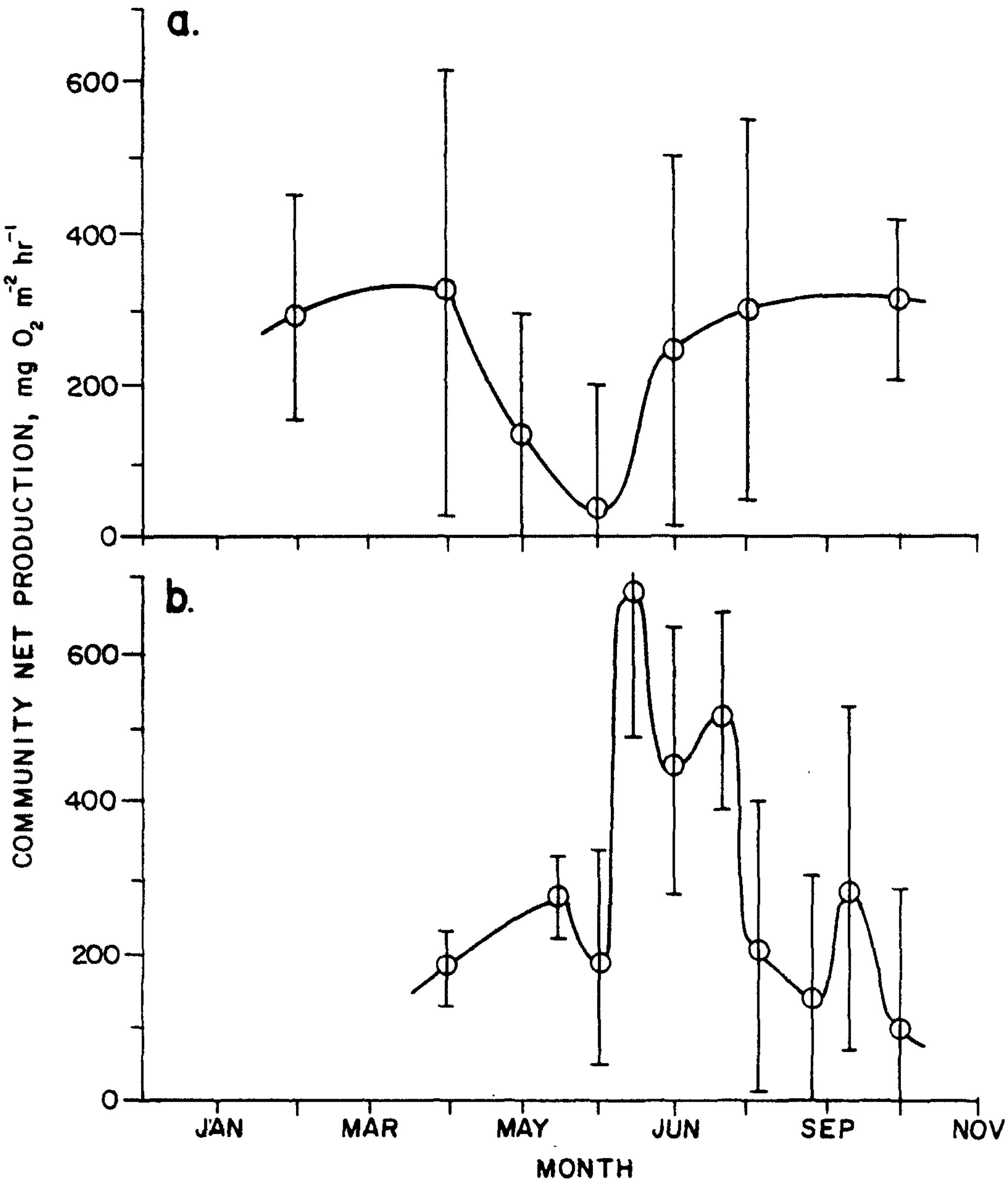

Figure 6. Net SAV community production in $mgO_2m^{-2}d^{-1}$ dominated by (a) _Zostera_ and (b) _Ruppia_ in the lower Chesapeake Bay. Dots are mean values; bars are standard deviations.

TABLE 1. DATA FROM SELECTED SOURCES INDICATING THE PARTITIONING OF (a) PRODUCTION (Pa), $gCm^{-2}y^{-1}$ AND (b) BIOMASS gm^{-2} (ORGANIC) BETWEEN VARIOUS AUTOTROPHIC COMPONENTS OF SAV COMMUNITIES

a. Location	Species	Seagrass	Epiphytes	Benthic micro-algae	Macro-algae	Phytoplankton	Reference
Florida	Thalassia	1000	200	---	---	---	Jones 1968
Mass.	Zostera	---	20	---	---	---	Marshall 1970
Calif.	Ruppia	28	---------------267 (Epiphytes, Benthic micro-algae and Macro-algae combined)			91	Wetzel 1964
N.Carolina	Zostera	330	73	---	---	---	Penhale 1977
Ches. Bay	Zostera[a]	0.48	0.17	-0.05	---	0.09	Murray (pers.comm.)
	P.pectinatus	0.5-2.2	---	---	---	0.3-1.0	Kaumeyer et al. 1981
	P.perfoliatus	1-3.0	---	---	---	0.5-1.0	Kaumeyer et al. 1981

a) Daily estimates in summer period.

b. Location	Species	Seagrass	Epiphytes	Benthic micro-algae	Macro-algae	Phytoplankton	Reference
Europe	Cymodocea	400-700	---	---	375	---	Gessner and Hammer 1960
Alaska	Zostera						
Kinzarof		1500	---	---	393		McRoy 1970
Klawak		415	---	---	29		
Others		113	---	---	2.4		
N.Carolina	Zostera	80	25	---	---	---	Penhale 1977
Ches. Bay	P.pectinatus	20-60	0.1-0.6	---	---	---	Staver et al. 1981
	P.perfoliatus	20-80	0.1-0.6	---	---	---	Staver et al. 1981

estuarine systems near Beaufort, N.C., studies were conducted on four major primary producers, with results pointing to the potential importance of seagrasses. Wetzel et al. (1981) summarize these studies and report annual productivity estimates of 66, 249, 330, and 73 $gCm^{-2}y^{-1}$ for phytoplankton, salt marshes, Zostera, and SAV epiphytes, respectively. Orth et al. (1979) report that SAV is an important autotrophic component in certain areas of the lower Chesapeake Bay. Stevenson (personal communication) estimates that some 40 percent of in situ production in Chesapeake Bay could be attributed to SAV in 1963, but only six percent could be assigned to SAV in 1975. Decreasing SAV abundance, especially in the upper Bay, make present estimates of SAV contributions to in situ productivity even smaller than the 1975 figures.

Estimates of relative seasonal contributions of sources of organic matter to the upper Bay provide insight into the seasonal stability of the food supply to food webs and form the basis for further evaluation of the nutritional quality of the various sources (Figure 7). These trends were developed from various kinds of information including the work of Flemer (1970), Biggs and Flemer (1972), Heinle et al. (1977), Kemp and Boynton (1980), Taft et al. (1980), Kemp et al. (1981), and Wetzel et al. (1981). Figure 7 suggests that because of the diverse sources, the organic matter supply to the upper Bay is relatively constant throughout the year and may, in part, explain the high productivity of the estuarine system (Nixon 1980). During the late winter and spring, it appears that upland drainage is the dominant source of most organic matter; in late spring and summer, phytoplankton production assumes a dominant role; in early fall SAV may have been an important source in the past; and in the winter the input of marsh vegetation via ice scouring and transport to the Bay may be important in some regions. Benthic microalgae are probably not significant primary producers due to the typically short euphotic zones encountered in the Bay (light limitation) and the high rates of sediment deposition and resuspension that deter community development.

COMPARISON OF SAV WITH OTHER MAJOR SOURCES OF ORGANIC MATTER TO THE BAY

A simplified organic matter budget is presented in Table 2 for the upper portion of Chesapeake Bay (upstream at the mouth of the Potomac River) for two periods (1960 and 1978). SAV was a distinctive and quantitatively important feature of the Bay during the early 1960's, and severely restricted in 1978.

This budget indicates that SAV may have been an important source of organic matter to low and mid-salinity portions of the Bay. During the 1960 period, we estimated that phytoplankton production was comparable to SAV production, and each of these was larger than riverine input. Some evidence indicates that between 1960 and 1978 both phytoplankton production and riverine input increased (Heinle et al. 1980, Boynton et al. 1982), with SAV production much lower during the late 1970's. Our estimates indicate that in the upper portion of Chesapeake Bay, SAV contributed about 30 percent of organic matter production during the 1960's when SAV was abundant, and on the order of four percent in 1978.

Because of high variability, these estimates are only guides as to the relative importance of various sources of organic matter in the low and mid-salinity portions of Chesapeake Bay. Considerable year-to-year

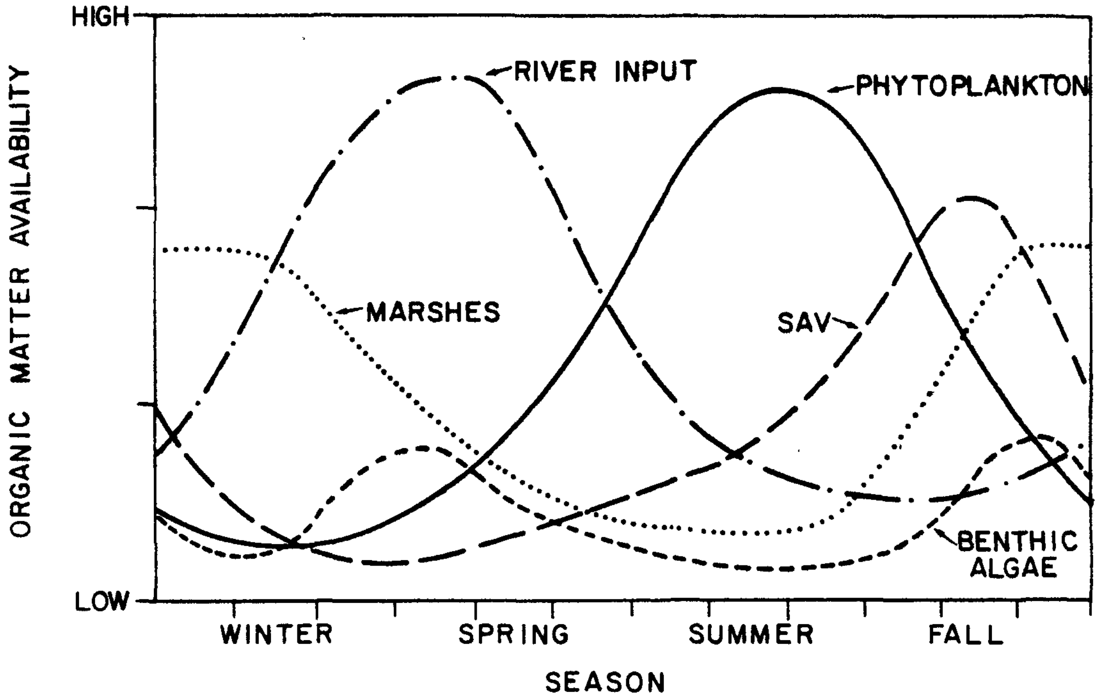

Figure 7. Hypothetical seasonal pattern and availability of organic matter to the Chesapeake Bay food web.

Table 2. ESTIMATED MAGNITUDE OF THREE SOURCES OF ORGANIC MATTER TO CHESAPEAKE BAY FOR TWO TIME PERIODS (ALL VALUES ARE IN UNITS OF $gCy^{-1}x10^{11}$)

Source[a]	Time Periods	
	1960	1978
Phytoplankton[b] production	3.8 (56)	3.8 (79)
SAV production[c]	2.2 (33)	0.2 (4)
Riverine input[d]	0.8 (11)	0.8 (17)
	6.8 (100)	4.8 (100)

[a] Includes area of Bay and tributaries above the mouth of the Potomac River ($1.5 \times 10^9 m^2$).

[b] Annual rate of production estimated at 250 g C $m^{-2}y^{-1}$ (Flemer 1970).

[c] Annual production estimated at 360 g C $m^{-2}y^{-1}$ based on rates reported by Kaumeyer et al. (1981) and Wetzel et al. (1981) for a 180-day growing season. Areal distribution of SAV estimated at $6x10^8m^2$ in 1960 (Rawls, in prep.; Stevenson, pers. comm.) and $0.7x10^8m^2$ in 1978 (Anderson and Macomber 1980).

[d] Riverine input of organic matter from Biggs and Flemer (1972).

variability in the absolute amounts delivered from riverine sources occurs and probably varies by a factor of one to two. Boynton et al. (1982) have also shown that phytoplankton productivity in the mid-salinity portion of Chesapeake Bay can vary by as much as a factor of three. The year-to-year variability in SAV productivity has not been evaluated, although observations by Orth (1981) indicate that there is some degree of fluctuation. Despite the probable errors involved in this calculation, it seems that in the early 1960's SAV was a significant autotrophic component in the upper Chesapeake Bay, but at the present time is a minor component. An important ecological consequence is the possiblity of less food for higher trophic levels such as fish.

FOOD-WEB UTILIZATION OF SAV

SAV can enter animal food webs either through direct grazing of living plants or consumption of SAV detritus at some point in decomposition processes. Several techniques have been used to establish the degree to which SAV is used as a food source but, unfortunately, each has substantial limitations. The most widely used technique is direct visual identification of material in the digestive system. This technique is relatively simple, but analyses are often time-consuming, and the degree to which food items can be identified is often limited to larger items that are resistant to digestion. A second approach involves a relatively expensive chemical technique in which the ratio of stable $C^{12}:C^{13}$ isotopes is determined for both plant food items and associated predators. The technique is based on different plant groups having characteristically different $C^{12}:C^{13}$ ratios. Animals feeding on a particular plant will, in time, approximately reflect the food source ratio. This technique is of limited value in establishing SAV food-web relationships because there are several primary producers associated with SAV communities, each of which has a distinctive $C^{12}:C^{13}$ ratio (Bunker et al. 1981b).

In spite of these limitations, several substantial results have emerged using these techniques to link SAV production to utilization in Bay food webs. Perhaps the most definitive linkage is between SAV and waterfowl. Direct grazing on SAV by waterfowl seems to be important both in Chesapeake Bay and elsewhere, and grazing in itself can impact the distribution of SAV locally. McAtee (1917) reports that generally, SAV is excellent food for waterfowl, that leaves, stems, roots, and rhizomes are all commonly used, and that P. perfoliatus is a particularly desirable species. Conversely, SAV can also be significantly affected by waterfowl grazing. For instance, Jupp and Spence (1977) reports that waterfowl grazing reduces SAV biomass by a factor of one to five in certain areas of Loch Leven, Scotland, and that overall, grazing removes about 20 percent of SAV biomass from the Loch. In Chesapeake area, Rawls (in preparation) notes that feeding by swans can transform a field of clover to "a hog wallow" overnight. Intensive grazing by swans during the 1980-1981 winter was probably partly responsible for the poor 1981 growth of P. perfoliatus at one of our intensive study sites in the Choptank River (Todds Cove site).

Studies of the dependence of Chesapeake Bay waterfowl on SAV for food have been conduced by Wilkins (1981), Rawls (in prep.), Perry et al. (1976) and Stewart (1962); results have been summarized by Stevenson and Confer (1978) and Munro and Perry (1981). Vegetable matter is an extremely important food item for waterfowl in the upper Chesapeake Bay (Table 3). Of some 2,747 birds examined by Rawls (in prep.), 78 percent of food material was vegetable. Several species of SAV (P. perfoliatus, R. maritima, M. spicatum, and N. guadalupensis) were prominant items averaging about 23 percent by volume in the diet of all waterfowl species considered. The birds analyzed in this study were collected between 1958 and 1968 during fall and winter hunting seasons. That so many birds contained significant quantities of SAV clearly indicates that SAV in the upper Bay persisted far longer into the fall and winter seasons than it does now. A shortened growing season, such as we now see in the upper Bay, is another index of stress on SAV.

Munro and Perry (1981) also developed long-term (1972 to 1980) SAV and waterfowl distribution data to test the hypothesis that variations in waterfowl populations were related to variations in SAV abundance. Though they found few statistically significant relationships between these two factors, they observed that the most important waterfowl wintering areas were also the most abundantly vegetated areas in recent years (lower Choptank and Chester Rivers and Eastern Bay). Munro and Perry further suggest that waterfowl have adapted to the SAV decline primarily by wintering elsewhere in the Atlantic Flyway and that future increases in SAV abundance will produce positive responses in waterfowl populations. In Loch Leven, Allison and Newton (1974) found a very good correlation between SAV abundance and waterfowl densities, and Hocutt and Dimmick (1971) found P. pectinatus to be the preferred food of some waterfowl species. Along similar lines, Wilkins (1981) reports that waterfowl use of SAV areas in Virginia is greater than in non-vegetated zones. Feeding studies suggest that waterfowl grazing on SAV-associated invertebrate populations is sufficiently intense (2 to 25 g m^{-2} dry material removed per year) to influence infaunal densities.

Aside from the direct grazing pathway by which SAV can enter food webs, the vast majority of studies, including those in the Chesapeake, indicate that most SAV material enters food webs through detrital pathways. den Hartog (1967), for instance, states that direct grazing is not an important feature of SAV communities; Ott and Maurer (1977) found that only a small fraction of Posidonia oceanica was consumed while live. Mann (1971), Day (1967), and Harrison and Mann (1975) all reached similar conclusions that agree with the general finding that direct grazing in most macrophyte-dominated aquatic systems is small. Mann (1972) and others indicate that SAV (and macrophytes in general) are a relatively poor source of food while alive because of low nitrogen content. Mann (1972) suggests the following scheme:

> Thus, the process of decomposition of leaf litter in coastal waters may take the following form. There is an initial period of autolysis during which soluble materials leach out. Bacteria and fungi then colonize the material and begin to render soluble by enzyme action some of the previously insoluble material. The micro-organisms absorb a proportion of the material they digest, and some escapes. Populations of predators such as ciliates and nematodes begin to build up. Macrobenthic organisms begin to tear off pieces of the plant material with its attached community of micro-organisms. They strip off the micro-organisms as the detritus passes through their guts, the feces are recolonized and the process is repeated by coprophagy. The cumulative result of this process is a steady reduction in particle size, with a consequent increase in surface-area-to-volume ratio, an increase in microbial populations and a reduction in the Carbon:Nitrogen ratio of the detritus.

In Chesapeake Bay, food web dependence seems to follow this pattern. Brooks et al. (1981), in a study of a seagrass bed in Virginia, report that seabass, pipefish, pigfish, and white perch are epibenthic feeders utilizing amphipods and shrimp that are, in turn, detrital feeders. Data further indicate that large predators (weakfish, bluefish, and sandbar sharks) entered the SAV bed with little food in their stomachs and left after feeding. Food items for these predators can generally be traced back to a detrital source, some fraction of which is probably SAV in origin. Again there was little evidence of direct grazing based on stomach

TABLE 3. FOOD HABITS OF WATERFOWL IN THE UPPER CHESAPEAKE BAY, MARYLAND [a,b] (FROM STEVENSON AND CONFER 1978)

Waterfowl species	Animal food (percent)	Vegetable food (percent)	Total (percent)	Predominant foods percent total volume
Canvasback	47.76	51.85	99.61	19.65 Baltic clam 18.42 Corn 16.32 Soft-shelled clam 14.29 Redhead grass 7.44 Widgeongrass
Redhead	23.40	76.59	99.99	29.29 Corn 15.19 Redhead grass 14.74 Widgeongrass 10.53 Soft-shelled, Baltic, and Mitchell's clams 6.73 Conrad's false mussel
Lesser Scaup	47.56	52.47	100.03	20.48 Widgeongrass 12.32 Soft-shelled clam 11.59 Corn 10.85 Redhead grass 6.89 Mussel
Bufflehead	67.42	32.59	100.01	13.52 Widgeongrass 11.85 Redhead grass 10.00 Barnacle 8.52 Fish 7.22 Mud crabs
Goldeneye	63.09	36.87	99.96	19.44 Mud crab 17.67 Corn 14.88 Soft-shelled clam 9.22 Barnacle 9.00 Bivalves (unidentified fragments)
Mallard	5.00	94.80	99.80	24.14 Corn 10.41 Redhead grass 8.17 Widgeongrass 9.13 Other submerged macrophytes 1.64 Conrad's false mussel 1.31 Soft-shelled clam

[a] Based on waterfowl gizzards collected during 1959-1968 hunting seasons
[b] Rawls (in press)

TABLE 3. (continued)

Waterfowl species	Animal food (percent)	Vegetable food (percent)	Total (percent)	Predominant foods percent total volume
Black Duck	6.44	93.54	99.98	17.52 Corn
				15.50 Redhead grass
				14.20 Widgeongrass
				8.40 Milfoil
				1.91 Conrad's false mussel
				1.76 Amphipods
Canada Goose	0.00	100.00	100.00	32.42 Grasses (Gramineae)
				29.61 Corn
				6.97 Milfoil
				5.11 White clover
				2.99 Crab grass

analyses. In an extensive study of feeding habits in upper Bay SAV communities, Bunker et al. (1981a) found little evidence of direct grazing by fish, although some SAV seeds and plant leaves were found in stomachs. Energy flow appeared to enter food webs as detritus and pass through epifaunal and infaunal invertebrates to small and large fish. Carr and Adams (1973) and Adams (1976) report similar results for fish communities in *Thalassia* and *Zostera* beds. Bunker et al. (1981a) note that many epifaunal species that are important food items are also closely associated with SAV.

Attempts were also made to quantify food sources of the invertebrate community using stable carbon isotope techniques to more closely relate SAV to foodweb production. While other investigators have reported some successes with this technique, studies in the lower Bay by van Montfrens (1981) and Bunker et al. (1981b) in the upper Bay yielded interesting but ambiguous results. The basic problem was that there were four or five available sources of organic matter (SAV, phytoplankton, epiflora, benthic microflora, and sediment detritus) each having a different ^{13}C ratio. Thus, unless an invertebrate had an extreme ^{13}C ratio (either high or low) there were an unlimited number of solution to the feeding equation. With a few exceptions, most animals had intermediate values (-13 to -18) suggesting that they were feeding on a mixture of detrital sources or a single detrital source with an intermediate ^{13}C value. In contrast to this, Fry and Parker (1979) reported that SAV detritus was an important feature of the organic matter supply in Texas seagrass systems. They found that inshore animals had less negative ^{13}C ratios (-8.3 to -14.5) than did offshore animals (-15.0 to -19.0) and that the differences corresponded to the less negative and more negative ^{13}C ratios associated with SAV (-7 to -12.2) and phytoplankton (-20 to -26), respectively. There appear to be several possible reasons for the differences in Chesapeake Bay and the Texas studies. First, SAV in Texas were a dominant component of the aquatic system and thus abundant SAV detritus was probably available through most of the year. In contrast, SAV are presently a marginal item in Chesapeake Bay. Furthermore, visual inspection of our study sites indicates that most of the SAV biomass is probably rapidly exported from littoral areas prior to becoming detrital particles of appropriate sizes. Thus, animals in Chesapeake Bay SAV beds may not have sufficient opportunity to feed on detrital SAV such that their ^{13}C ratios closely reflect SAV ratios. Secondly, phytoplankton are a dominant feature of Chesapeake Bay and we have demonstrated that SAV communities can effectively filter plankton from the water column via their baffeling effects on currents (Boynton et al. 1981b). Thus, there is an effective supply of nutrient-rich organic matter with a very negative ^{13}C ratio (-22) available to SAV food webs. In view of the above circumstances, it is not surprising that ^{13}C ratios did not clearly indicate SAV detritus to be of dominant importance in Chesapeake Bay littoral zone foodwebs.

In summary, the majority of studies suggest that SAV is available primarily as detritus and, in some localities, is very important. Because of the complexity of organic matter sources in Chesapeake Bay and the current marginal distribution of SAV, a quantitative assessment of SAV importance as a food source was not possible. However, CBP results show that SAV in the Bay is probably used by heterotrophs of one type or another, and that SAV's physical structure concentrates other foods (phytoplankton, epiphytic algae, benthic microalgae) for animal consumption.

From a Bay-wide perspective, several recent studies provide additional support for this argument. Kemp and Boynton (1981) constructed seasonal and annual carbon budgets for three meter and six meter depth zones in the mid-salinity portion of the estuary and found that on an annual basis virtually all carbon inputs were used. They conclude that "despite the considerable interactions both within the benthic community...and between the benthos and other parts of the estuarine ecosystem... photosynthesis ultimately limits...metabolism." If heterotrophic metabolism is organic matter limited, it follows that SAV would also be used if, as has already been shown, this material is a suitable food source. Boynton et al. (1981c) have also shown that, in most portions of the Bay, the amount of organic material being sequestered into deep sediments is a small fraction (three to five percent) of that being produced in overlying waters, again suggesting that if suitable organic matter is available, it will tend to be refined. Thus, loss of SAV production may well lead to loss of animal production.

SECTION 3

THE HABITAT VALUE OF SAV SPECIES IN CHESAPEAKE BAY

It is generally accepted that meadows of SAV serve as primary nursery habitats for a diverse assemblage of commercially valuable biota and forage species. Though many sampling studies have documented impressive numbers of animals in vegetated areas, the mechanisms underlying the proposed nursery role of SAV beds have only scarcely been elucidated. The two most obvious explanations for the great abundance of SAV-associated organisms are that SAV provides them with food and shelter.

These two features are supported by past research. It is clear that SAV beds are among the most productive systems known (McRoy and McMillian 1977), and it is equally clear that most of this production is not grazed by resident organisms (Ogden 1976). Instead, SAV detritus (Mann 1972, Klug 1980) and SAV epiphytes (Morgan 1980) provide most of the energy available to secondary consumers in SAV beds. Recent studies, using $^{12}C:^{13}C$ ratios to trace the source of carbon present in secondary consumers, show that SAV-derived carbon provides a significant fraction of the energy used by secondary consumers in Texas turtlegrass meadows (Fry and Parker 1979), but a rather small fraction in a newly established North Carolina eelgrass bed (Thayer et al. 1978). To date there have been no comparative studies of the growth rate of organisms living in (versus outside) SAV meadows. Thus, only indirect generalizations can be made regarding the relative importance of SAV-derived carbon for the growth and survival of associated fauna.

Growing evidence suggests, however, that SAV protects its fauna from their predators. For example, Nelson (1979) shows that eelgrass provides amphipods significant amounts of protection from predatory finfish, and Stoner (1980) shows that several kinds of benthic plants provide amphipods protection from finfish. Plant surface area affords the best estimate of a plant's protective ability. Recently, Heck and Thoman (1981) found that turtlegrass and several species of red algae provide significant amounts of protection to tethered crabs in field trials, and that both artificial and live eelgrass provide grass shrimp (*Palaeomonetes pugio*) significant amounts of protection from predatory killifish (*Fundulus heteroclitus*).

STRATEGIES AND METHODS USED IN CBP HABITAT STUDIES

Against this background of published information, a series of studies was designed to determine the extent to which SAV beds in Chesapeake Bay serve as sites of densely aggregated animal species (indicating the use of SAV for food) and as areas providing, through the physical presence of the plants themselves, important amounts of shelter from predators for a wide range of invertebrate and fish species.

Several field sampling studies were funded under the CBP to answer the first question. One study compared standing stock and secondary production of all macrofaunal ($>$ 0.5 mm) organisms that inhabited the bed with similar estimates for nearby unvegetated bottoms (lower Bay). A second study, with similar aims, was done at two upper-Bay eastern shore beds of mixed species composition (Parson Island and Todds Cove, Maryland). Both these studies used a variety of sampling techniques, including seining, trawling, and gill netting, depending on the size and mobility of the target species. A third sampling study compared the use of the upper Bay eastern shore grass bed at Parson Island by commercially important fishes and blue crabs with a lower-Bay eelgrass bed, near the mouth of the York River, and with unvegetated habitats. Trawling and gill-net sampling were used in this study.

Field and laboratory experiments were also conducted to investigate the ability of SAV to provide animals with shelter and protection from predators. Field experiments used exclusion cages to evaluate the intensity of predation by fishes and blue crabs on infaunal populations in vegetated versus unvegetated habitats. By excluding predators from certain areas, we estimated what predation-free infaunal population densities would be in both vegetated and unvegetated areas. Then, by comparing ratios of standing crop in vegetated and unvegetated areas before and after caging, we estimated the amount of protection provided by vegetation in natural conditions. In the lower Bay, caging experiments were performed at the eastern shore Vaucluse Shores site; in the upper Bay, caging experiments were carried out in the Todds Cove portion of the Choptank River.

Laboratory microcosm experiments were conducted to estimate the amount of protection provided by SAV for infaunal bivalves, shrimps, crabs, and fishes. The first set of experiments was designed to test the ability of low, medium, and high density artificial eelgrass blades and rhizome mats to provide protection for the infaunal bivalve *Mulinia lateralis* and for juvenile blue crabs (Heck and Thoman 1981). Predators were adult blue crabs. This set of experiments was conducted in wading pools (2.43 m in diameter x 0.45 m in height) with recirculating water. The second element used larger tanks (3.66 md x 0.9 mh) to examine protection for spot (*Leiostomous xanthurus*) and silversides (*Menidia menidia*) by medium and high densities of artificial eelgrass. Predators used were summer flounder (*Paralichthys dentatus*) and weakfish (*Cynoscion regalis*). Laboratory studies were also performed to evaluate the protection artificial and living eelgrass, and living widgeongrass provided grass shrimp. Experimental tanks were 1.3 md x 0.3 mh, and recirculated water was used. Predators were killifish (*Fundulus heteroclitus*). Controls in all of these experiments involved identical treatments conducted in unvegetated tanks.

Though these laboratory studies are similar, each was designed to investigate different aspects of predator-prey relations in vegetated habitats. These factors included studying the importance of prey escape behavior, predator-prey size in relation to the size of SAV patches, and differences in the amount of protection provided by different species of SAV.

IN SITU ANIMAL ABUNDANCES

Invertebrates

In both the upper and lower Chesapeake Bay, field study results indicate that infaunal abundance and diversity were higher in vegetated than unvegetated areas. In the lower Bay, polychaetes dominated eelgass and widgeongrass areas, and bivalves dominated unvegetated sites, although a large overlap in species composition existed between the two types of habitats. These differences in dominant taxa are due partly to the type of bottom. Muddy sediments deposited in seagrass beds favor deposit feeders such as polychaetes; the sandy sediments in non-vegetated areas favor suspension-feeding bivalves. Some of the other differences in species composition occur because epifaunal, grass-blade associated invertebrate species inevitably occur in sediment samples from SAV areas, even though they are not residents of the infauna. Greater abundances in SAV areas are due primarily to the large numbers of polychaetes, oligochaetes, isopods, and grass-blade organisms collected at these sites.

In the upper Bay region, polychaetes dominated both SAV and unvegetated sites, although numbers were greater in vegetated areas, as were overall abundances of oligochaetes and isopods. Bivalves were more abundant in unvegetated areas in spring, but became more abundant in SAV beds during summer (Bunker et al. 1981c, Ejdung et al. 1981). Amphipod abundances were greatest at the unvegetated site and in the lower Bay. These differences reflect the suitability of fine sediments for deposit feeders in SAV areas versus the suitability of sandy sediments for suspension feeders in unvegetated areas.

There were also some notable differences in the abundances of organisms as related to salinity. Diversity was much lower at the low salinity (7 to 11 ppt) sites in the upper Bay than at the higher salinity (14 to 22 ppt) site in the lower Bay. For example, maximum density at the high salinity eelgrass site was 90,000 individuals per m^2. The reason for the relatively low infaunal abundances at the low-salinity site is not known.

Because unvegetated habitats support a virtually non-existent epifauna, (defined as the animal assemblage growing on SAV and other emergent bottom features), only vegetated habitats were sampled for epifaunal organisms. Epifaunal density was higher at the more densely vegetated Todds Cove bed than at the Parson Island bed. However, epifaunal densities per g SAV (excluding polychaetes) were very similar at the two sites, ranging from around 50 to 200 individuals per g SAV biomass. The isopod Erichsonella attenuata was dominant in Parson Island collections, and gastropods and tanaids were dominant at Todds Cove. Amphipods, grass shrimp, and chironomids were present at both sites (Staver et al. 1981).

Epifaunal abundances at the eelgrass sites in the lower Bay were much higher than those found at the low salinity SAV sites in the upper Bay even though polychaetes were not included in the upper Bay sampling. Numbers ranged from around 20 individuals per g SAV in November to more than 9,200 individuals per g SAV in April. Dominant species included isopods, gastropods, polycahetes, and barnacles (Diaz and Fredette 1981).

Differences in salinity between the two intensively studied areas were probably responsible for the large abundance of barnacles in the lower Bay and at least partly responsible for differences in total abundance between sites.

Finfish

Finfish sampling in the protected Todds Cove bed and the exposed Parson Island site (Figure 8) found greater abundances and species richness in vegetated than unvegetated bottoms, with greatest numbers occurring in the

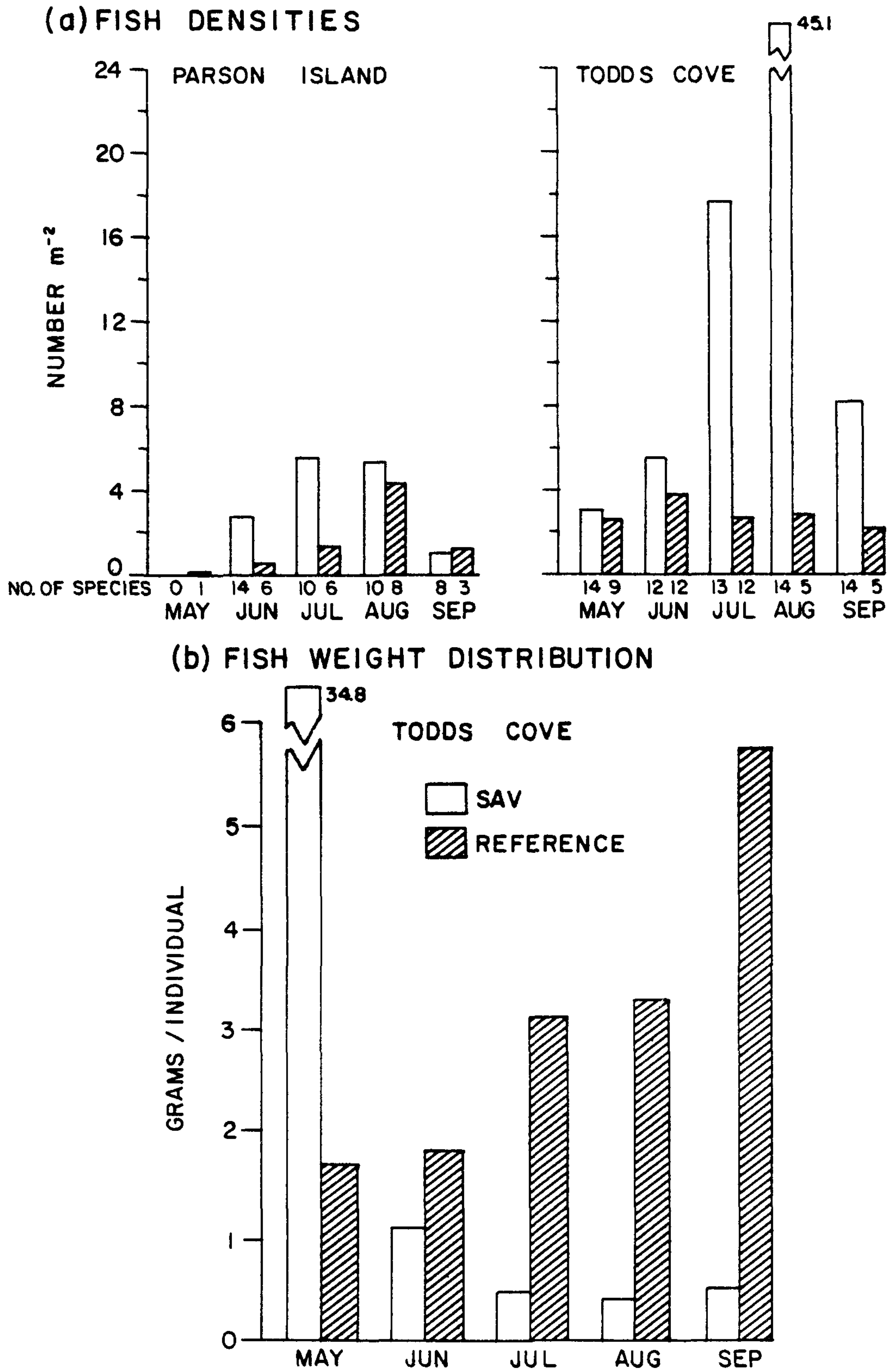

Figure 8. Average monthly (a) fish densities at Parson Island and Todds Cove sites for vegetated and non-vegetated (reference) areas and (b) fish weight distribution at Todds Cove for vegetated and non-vegetated areas. Data from Lubbers et al. 1981.

protected SAV area. Fish densities at Todds Cove are among the highest yet reported in the literature (Lubbers et al. 1981).

In addition, a plot of average weight per individual (Figure 7) during the summer period suggests that the SAV bed at Todds Cove was continually used as a nursery area for small fish while larger sized individuals predominated in unvegetated reference areas. However, Heck (personal communication) often found fish abundance (as indicated from otter trawl samples) to be as large on sandy bottoms as in SAV at the Parson Island site. This difference occurred because of the chance catch of schools of spot (Leiostomous xanthurus) over sandy areas, and probably because SAV abundance dropped precipitously during the course of the study. Large fish predators such as bluefish and cownosed rays were found in both vegetated and unvegetated habitats and, in both studies, more fish were taken at night than during the day. There was little indication that the low salinity SAV beds serve as nursery areas for commercially valuable finfish, although any conclusion concerning such values might be biased due to the severely depressed distribution of SAV in the upper Bay.

Fish sampling programs in lower Bay eelgrass meadows on the York River and at Vaucluse Shores found much greater abundances and species richness in these higher salinity SAV beds than on nearby unvegetated bottoms, and much greater night than day catches (Brooks et al. 1981). Some large fish predators, such as weakfishes and sandbar sharks, seem to forage most often over vegetated bottoms while others, such as bluefish, appear to forage indiscriminately over both vegetated and unvegetated areas.

The main conclusion of these studies on fish, that fish communities are richer in vegetated than unvegetated areas, was expected, because similar results have been found previously in other SAV habitats in North Carolina (Adams 1976), Florida (Livingston 1975), and Texas (Hoese and Jones 1963). What was not expected was the finding that few commercially important finfish use the SAV beds as significant nursery habitats. This result is surprising, because many juveniles of commercial species such as sea bass, snappers, and groupers use SAV as nursery areas in latitudes south of Chesapeake Bay (Adams 1976, Livingston 1975, Weinstein and Heck 1978). The role of SAV for commercial fishes in the Chesapeake Bay system seems to be largely that of a rich foraging place for adults although, once again, it is important to emphasize that the current restricted distribution of SAV may bias these conclusions. For instance, major spawning and juvenile habitats for striped bass once existed in the upper Bay (Susquehanna Flats), an area that was densely populated with SAV. More representative patterns of commercial fish use of SAV habitat might best be evaluated through historical correlations of SAV and juvenile fish distributions; this is being done in the CBP's environmental characterization.

Blue Crabs

Information on blue crab abundances was collected at the same time fishes were sampled at intensive study sites. Investigators found low numbers of juvenile blue crabs in the upper Bay, but extremely large numbers of blue crabs in eelgrass meadows of the lower-Bay. Up to 10,000 times as many blue crabs were found at the lower Bay than upper-Bay SAV sites. Studies by Heck (personal communication), which used identical sampling techniques in both low and high salinity SAV beds and adjacent unvegetated areas, found lower Bay crab numbers ranging from a few to a thousand times more abundant than the upper Bay beds during the spring and summer months. In addition, most of the crabs in the high salinity eelgrass beds were juvenile females ($<$ 100 cm CW) that constituted the breeding stock of future generations. Blue crab densities in unvegetated areas were found to be as large or larger than those recorded from upper Bay SAV beds. In contrast, far fewer crabs were taken on sand than in SAV at the lower-Bay site. This difference in sand versus SAV crab abundances between sites is probably due to the presence of many juveniles at the eelgrass site, most of which require SAV for protection from predators. The adult crabs in upper Bay SAV beds apparently do not require vegetation for protection, except when molting, and occur on both vegetated and unvegetated bottoms. The sampling gear used to collect crabs in both locations was not efficient for collecting molting crabs. Thus, the role of upper Bay SAV in providing protection to molting crabs may have been greatly underestimated.

The conclusion drawn from these studies is that there seems to be only a very limited blue crab nursery role played by upper Bay SAV beds. Lower Bay eelgrass beds, however, serve as primary blue crab nursery habitats and support very large numbers of juvenile blue crabs throughout the year.

STUDIES ON SAV AS PROTECTION

Results of soft-bottom predator exclusion experiments are often difficult to interpret because of several commonly encountered problems, including an inability to completely exclude predators from caged areas and accurately estimate the effects that the presence of the cage itself produces on the physical environment (Virnstein 1978, Dayton and Oliver 1980, Peterson 1979). Caging studies conducted in Chesapeake Bay encountered these problems, and the results of these studies, therefore, must be interpreted with caution and circumspection.

Caging experiments in the lower Bay show that infaunal densities increased in caged areas, and that this increase was most pronounced on unvegetated bottoms (Orth 1981). There was little evidence that cages altered the physical environment by changing sedimentation patterns, although predators did periodically invade caged areas. Epifaunal densities were higher in caged than uncaged areas shortly after the installation of cages in eelgrass, but shading by cages subsequently reduced eelgrass biomass and also led to declining epifaunal numbers.

Caging studies conducted in the upper Bay were less conclusive than those conducted in the lower Bay. Some evidence suggests that predation may be important in reducing infaunal densities; however, technical difficulties, such as small fish passing through the cage walls, weaken the results.

The first element of experimental predation studies shows that, in the presence of artificial vegetation and rhizomes, juvenile blue crabs received a significant amount of protection from predation. The infaunal bivalve *Mulinia lateralis*, however, received very little protection from either the artificial leaves or rhizome mat. The second element shows that the predators *Paralicthys dentatus* (summer flounder) and *Cynoscion regalis* (weakfish) captured progressively fewer spot (*Leiostomous xanthurus*), and silversides (Menidia menidia) as vegative cover increased from 0 to 22 percent (Orth 1981).

The second study (Heck and Thoman 1981) found that dense amounts of artificial and live eelgrass, and live widgeongrass, provided grass shrimp with significant amounts of protection from fish predators. Low and medium densities of SAV did not provide much shelter. Furthermore, widgeongrass provided greater protection per unit of surface area than either living or artificial eelgrass.

The seemingly disparate results of these microcosm experiments can be understood within the following framework. Mobile epibenthic animals, such as fish and blue crabs in the former study, and grass shrimp in the latter study, do derive protection from predators in the presence of SAV. However, the amount of protection received is probably a function of plant surface area. Thus, SAV species with finely branched leaves and high surface areas should provide better protection for prey taxa than plants with simple leaves, all other factors being equal. Infaunal species, such as burrowing bivalves, should generally receive less protection from predators than epifaunal species of SAV habitats. For shallow burrowing species like Mulinia lateralis, SAV beds provide little or no protection from predators. However, for species that burrow below the SAV rhizome mats, there should be reduced predator success in vegetated habitats compared with that in unvegetated areas.

This hypothesis explains the results of the microcosm experiments and is amenable to further testing and verification. It is likely, however, that a number of other unstudied variables, such as size of predator and prey in relation to SAV dimensions and the foraging strategy of the predators, also influence predator-prey relations in SAV beds.

SECTION 4

INFLUENCE OF SAV ON SEDIMENT DYNAMICS

Sediment processes in estuarine and coastal systems have been the focus of numerous studies in the past several decades. Results of such studies indicate that sediment processes strongly influence light attenuation in the water column, produce shoaling or scouring, affect the composition of benthic invertebrate communities, and influence the exchange of materials between sediments and overlying waters. The sediment processes which produce such effects can be characterized as a cycle that includes the following components: (1) yield of "new" sediments from land erosion, runoff, and shoreline erosion; (2) deposition of suspended sediments; (3) resuspension of deposited sediments due to tidal and wave action and; (4) transport of resuspended sediments to different locations.

The resuspension-deposition [(3) - (2)] portion of the cycle dominates littoral zone sediment dynamics and can affect the health of SAV. A higher cycling rate increases seston levels and reduces light availability to SAV. This cycle is shown diagramatically in Figure 9 and suggests the probable magnitude of different portions of the cycle in deep and littoral estuarine areas. The diagram shows that wave action is the major source of energy to resuspend unconsolidated sediments in the littoral zone, and tidal energy provides the major force to resuspend and transport sediments in deep water. The relative contribution of major new sources of sediment to the Bay include material washed in from the watershed and shoreline erosion. Most shoreline material enters the deposition and resuspension cycle from the margins of the Bay, whereas fluvial sources follow the deep water transport path.

REVIEW OF SEDIMENT PROCESSES

Aspects of sediment dynamics and turbidity patterns have received considerable attention in Chesapeake Bay. Net sedimentation rates have been repeatedly estimated for various portions of the open Bay (Biggs 1970, Schubel and Hirschberg 1977, Brush et al. 1981) and for some tributaries (Roberts and Pierce 1976, Yarbro et al. 1981). Increases in turbidity have been documented for both the open Bay (Heinle et al. 1980) and for some tributaries (Kemp 1980). The increases are apparently due to increased algal stocks and seston levels. In a crude fashion, the decline of SAV communities in northern Chesapeake Bay parallels increasing trends in turbidity and nutrient loading. However, most of this work has been done in deep areas of the Bay region. In this summary, we have focused on vegetated and non-vegetated littoral areas less than two meters in depth. In addition, previous measurements have largely been devoted to estimating net sedimentation rates, and as Oviatt and Nixon (1975) have pointed out, only a small fraction of total "sediment activity" is measured when such estimates are made.

Study Area	Sediment Cycling Rate, g $m^{-2}y^{-1}$	Depth,m	Reference
Naragansett Bay	$7 - 18 \times 10^3$	7	Oviatt & Nixon, 1975
Departure Bay, B.C.	3×10^3	32	Stephens et al. 1967
Surf Zone, California	$7 - 330 \times 10^6$	2	Shepard, 1963
Buzzards Bay, Mass.	6×10^4	15	Rhodes & Young, 1970
York River, Va.	8×10^3	-	Haven & Morales-Alamo, 1972
Upper Patuxent River	2×10^5	4 - 6	Boynton et al. 1981b
Lower Patuxent River	$3 - 6 \times 10^5$	10-12	Boynton et al. 1981b
Littoral Zone, Non-SAV	$3 - 4 \times 10^5$	1 - 2	Boynton et al. 1981b
Littoral Zone, SAV	$0 - 3 \times 10^4$	1 - 2	Boynton et al. 1981b

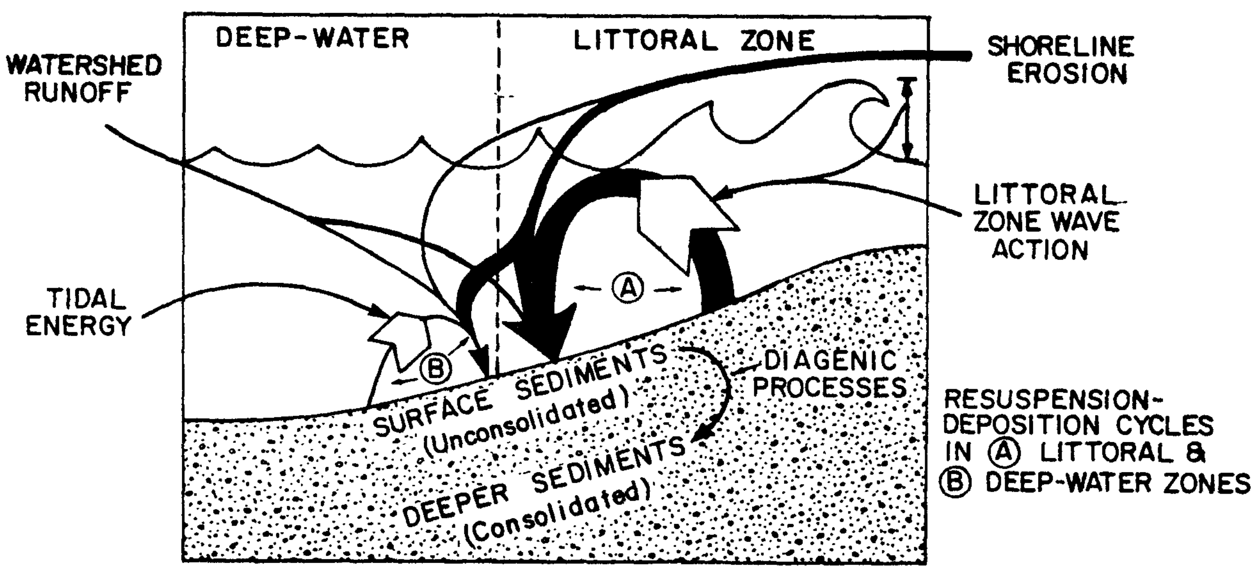

Figure 9. Major physical sediment processes in Chesapeake Bay showing sources and energy for sediment transport.

TABLE 4. SEDIMENTATION RATE IN mm y^{-1} AT SEVERAL LOCATIONS IN CHESAPEAKE BAY AND OTHER SELECTED ESTUARIES

Study Area	Net Sedimentation Rate, mm y^{-1}	Technique	Reference
Narragansett Bay	0.3 - 0.4	Mass Balance	Farrington 1971
Delaware Bay	1.5	Not available	Oostdam & Jordan 1972
Patuxent Estuary			
Upper	37.0	Mass Balance	Roberts & Pierce 1976
Upper	4.0 - 7.0	Pollen Dating	Brush et al. 1981
Lower	4.0	Pollen Dating	Brush et al. 1981
Lower	5.0 - 10.0	Sediment Traps	Boynton et al. 1981
Chesapeake Bay			
Upper	4.5 - 9.0	Pb^{210}	Hirschberg & Schubel 1979
Upper	6.0 - 10.0	Pollen Dating	Brush et al. 1980
Mid	1.5	Pollen Dating	Brush et al. 1980
Mid	1.1	Mass Balance	Biggs 1970
Mid	0.9 - 1.2	Pb^{210}	Hirshberg & Schubel 1977

A considerable number of measurements of net sedimentation rates and sediment cycling rates (summation of resuspension-deposition) were made in estuarine environments using a variety of techniques. We summarize some of these measurements in Table 4 with special emphasis on Chesapeake Bay. Net sedimentation estimates for areas in the tidal Bay system ranged from 0.3 to 37 mm y^{-1}. This broad range is not surprising in view of the strong gradients in seston concentration and sediment input rates encountered in estuarine systems. In the turbid upper section of Chesapeake Bay, for example, estimates ranged from 4.5 to 10 mm y^{-1}; in the mid-salinity region, rates ranged from 0.9 to 1.5 mm y^{-1}. A similar pattern was evident in the Patuxent River. To compare the magnitude of net sedimentation with sediment cycling rates (i.e. deposition-resuspension-deposition), accumulation rates (in mm) were converted to a weight basis. On this basis, net sedimentation in Chesapeake Bay ranged from about 600 to 6,000 g $m^{-2}y^{-1}$ of dry sediments.

In sharp contrast to these values, sediment cycling rates were far higher, especially in shallow water environments, and indicate that cycling dominates sediment processes. Significantly, an estimate from an SAV community (Choptank River) was among the lowest we encountered in estuarine systems and illustrates the importance of these communities in stabilizing sediments at the surface.

ROLE OF SAV IN SEDIMENT PROCESSES

Several previous investigations have led to an understanding of the mechanisms by which SAV can modify sediment substrates (Ginsburg 1956, Ginsburg and Lowenstam 1958, Wanless 1981). Specifically, the rhizome-root complex can stabilize sediments, and the physical structure of seagrass blades and epiphytes can slow currents, allowing sediments to settle. This complex can also substantially reduce current-and-wave-induced resuspension. Considerable evidence suggests that SAV can play an important role through those mechanisms in nearshore sediment dynamics. Scoffin (1970) found that dense beds of Thalassia protected bottom sediments from current speeds up to 70 cm sec^{-1}; extensive bottom-sediment erosion did not begin until current speeds reached 150 cm sec^{-1}. In Florida, Ball et al. (1967) found that bottom erosion was minimal in seagrass covered areas following the passage of a hurricane, but exposed sand areas were extensively modified. Also in Florida, Wanless (1981) found sedimentary sequences that probably resulted from trapping and consolidation of suspended particles by SAV. He states that the vertical sediment record indicates increased trapping of storm-generated sediments and decreased bedload transport as SAV became established. In a Zostera bed in Denmark, Christiansen et al. (1981) infer, from inspection of sediment cores and historical SAV distributions, that the Zostera die-back in the 1930's resulted in disturbance and mobilization of nearshore sediments and a movement of sediments into a local harbor. Moreover, Christiansen determined that coastal morphology was stable during periods when eelgrass was present, but significant changes occurred when it was absent.

In the Chesapeake area, Orth (1977) reports that sediment particle diameter decreased, and organic matter content and infaunal densities increased in bottom sediments in areas with SAV compared with those that did not have such coverage. Based on these findings and observations that showed less sediment disruption during storms in vegetated zones and less dispersion of dyed sand patches, Orth concludes that SAV is effective at trapping and consolidating suspended sediments. It appears that substantial beds of SAV can effectively modify littoral zone sediment dynamics through sediment trapping and consolidation of sediments at the surface. Because sediment processes may be most active in littoral zones, sediment processes in deeper areas may also be affected by lateral transport and deposition (Webster et al. 1975).

CHESAPEAKE BAY PROGRAM STUDIES

We hypothesized that SAV communities can play a significant role in modifying littoral zone light regimes by baffling of wave and tidal currents, thus reducing sediment resuspension. Conversely, we hypothesized that high turbidities in some areas of the Bay have contributed to the decline of SAV communities. Chesapeake Bay Program studies were designed to (1) document patterns of light attenuation on several time scales (seasonal, diel, tidal cycle) in littoral communities having SAV and in those not having SAV; (2) relate observed light attentuation patterns to concentrations of materials in the water column to identify the relative importance of light attenuating factors; and (3) examine the potential of SAV communities as natural sediment traps.

Data on suspended sediments and light attenuation from intensive study sites are presented in Figure 10. This figure shows differences between vegetated and non-vegetated areas plotted against tidal stage for Todds Cove in the Choptank River and Parson Island sites in Eastern Bay. These plots indicate that as turbid offshore waters enter SAV beds on rising tides, sediments are effectively removed, thus increasing light transparency. It

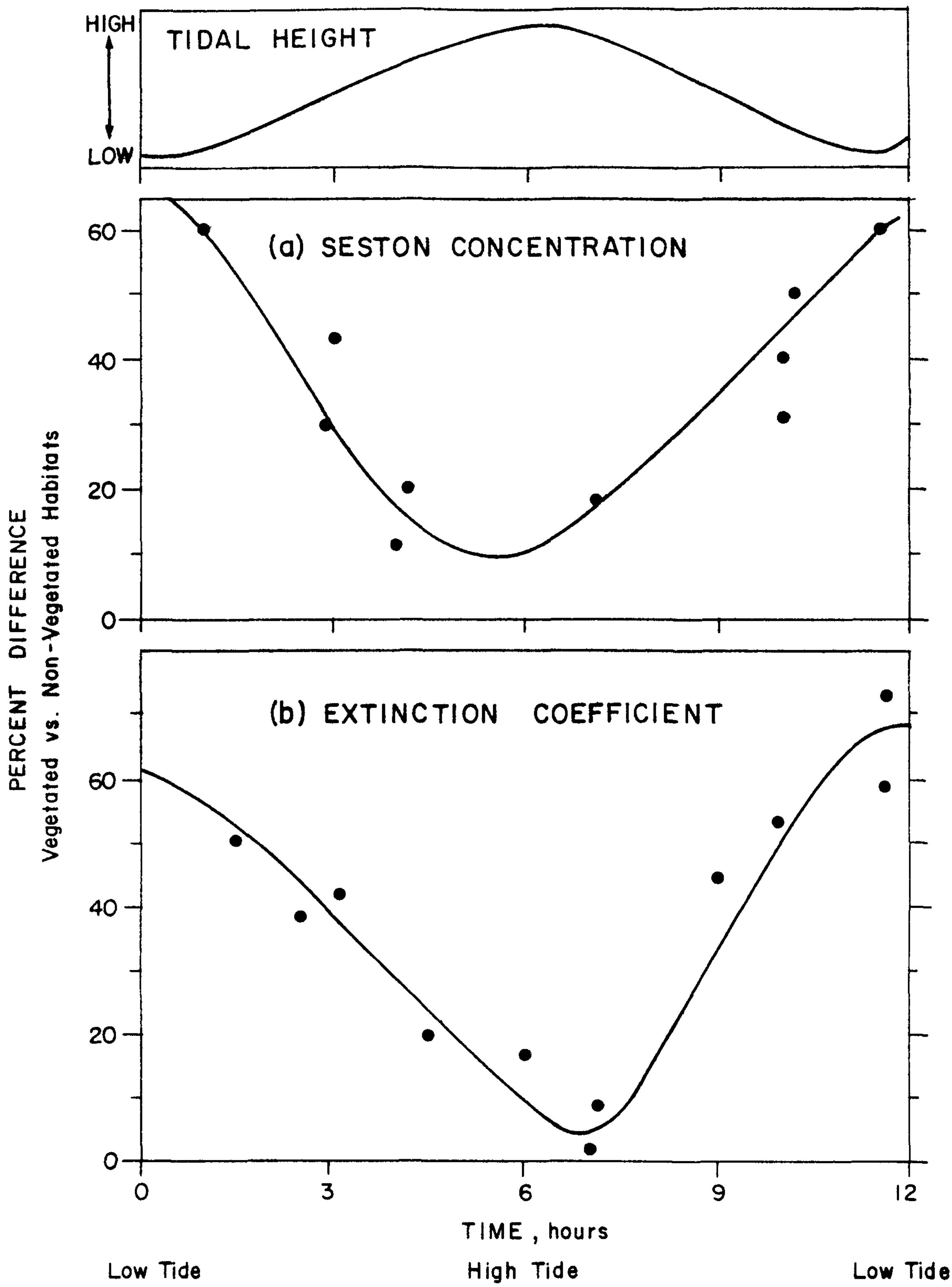

Figure 10. Percent difference between vegetated and non-vegetated habitats for (a) suspended sediment and (b) attenuation coefficient during a tidal cycle.

appears that by high tide, turbid inflowing waters have exceeded the filtering capacity of the bed. As tidal height decreases, the bed apparently effectively filters sediments because the gradient between vegetated and non-vegetated areas again increases to a maximum.

Other qualitative observations show that SAV varies in its ability to decrease turbidity. Boynton et al. (1981b) found that areas with SAV, dominated by P. perfoliatus (a highly branched species), were more successful at decreasing turbidity than at a site where P. pectinatus (a thin-bladed single leaf species) dominated. Even under conditions when SAV biomass was comparable between the two areas, it appeared that P. perfoliatus was more effective in clarifying surrounding waters. On several occasions, turbid water was observed entering a P. pectinatus bed; turbidity increased rapidly in the P. pectinatus sections, but remained considerably lower in the P. perfoliatus areas. Visually, the P. perfoliatus beds appeared as clear areas against a turbid background.

Other qualitative data show that water clarity is affected by the size of an SAV bed. On several occasions, Boynton et al. (1981b) noted turbidity gradients within an SAV bed. Turbidity was greatest at the edge of the bed and decreased with distance into the bed. It seems that for a given off-shore turbidity regime there is a "critical bed size" above which SAV can effectively modify the local environment in a fashion favorable for continued growth (reduce seston levels; increase light penetration). Small SAV beds may not be able to so modify local light regimes and would thus be disadvantaged if light is limiting growth.

Interpretation of data concerning sediment cycling in littoral zones is quite difficult. Boynton et al. (1981b) hypothesized that there would be substantial differences in the amount of material collected in both surface and bottom cups of sediment traps deployed in SAV beds and non-vegetated reference areas. They anticipated that the structure of SAV would effectively reduce resuspension and hence values from the bed would be markedly lower. In fact, while values from the SAV bed were lower, dramatic differences were not consistently evident. It is possible that most resuspension - deposition occurred during storm events and that during these events wave energies were high enough to overcome the baffeling effect of SAV in these marginal communities, leading to substantial deposition in all areas. Further inspection of climatic data may clarify this possibility. Another possibility is that material collected in cups in the SAV area was a mixture of resuspended materials and true sedimentation, while resuspended material made up the bulk of the collection in the reference area. Substantial reduction in seston-based turbidities (Figure 9) support this suggestion.

In spite of the lack of large differences betwen SAV and references area collection, there was a reasonably consistent pattern evident with respect to collection rate and SAV biomass, particularly for the bottom collection cups (Figure 10). Both surface and bottom cups had small collection rates when biomass was above 150 g m^{-2} and rates five to 10 times higher when biomass was bewlow 50 g m^{-2}. When viewed in this fashion, it appears that resuspension is clearly reduced in proportion to SAV biomass.

Given the dynamic nature of the sediment-water interface in littoral environments, estimates of net sediment retention/compaction are exceedingly difficult to obtain and clearly beyond anything that can be inferred from sediment traps. A crude estimate can, however, be obtained utilizing the seston data presented earlier. If we attribute the tidally related changes in seston concentration to dposition, then seasonal estimates can be made. If we take 90 mg L^{-1} as an estimate of mean seston concentration in littoral reference areas (Boynton et al. 1981b) and, as suggested in Figure 9, assume that approximately 50 percent of the suspended material is deposited on each tide, then daily deposition can be estimated. Taking six months as the period when substantial SAV biomass is present allows expansion of diel estimates to seasonal estimates. This procedure yields daily deposition rates of about 63 g $m^{-2}d^{-1}$ and seasonal estimates (180 days) of 1200 g m^{-2}. Assuming that there is about equivalent to 0.2 cm per growing season. The potential errors associated with such a calculation are obvious, but it is interesting that such a reasonable value emerges. Little information is currently available to suggest whether or not this material is sufficiently consolidated to be considered as lost to the sediment deposition-resuspension-deposition cycles. For example, we do not know if material deposited during the summer period when SAV are present is subsequently lost when SAV die-off in the early fall. Considering the important role of roots and rhizomes in this process and the low below-ground biomass observed in Chesapeake Bay SAV communities, it seems doubtful if this material is permanently consolidated at present, although it may have been in the past.

Some evidence suggests that SAV can cause sediment to compact, thus preventing resuspension of sediments. Net sediment retention-compaction can be crudely estimated by using the suspended sediment data presented earlier. If we attribute the tidally related changes in seston concentration to deposition, then seasonal estimates can be made. If we take 90 mg L^{-1} as an estimate of mean suspended sediment concentration in littoral reference areas (Boynton et al. 1981b) and, as suggested in Figure 10, assume that approximately 50 percent of the suspended material is deposited on each tide, then daily deposition can be estimated. Taking six months as the period when substantial SAV biomass is present allows for expansion of diel to seasonal estimates. This procedure yields daily deposition rates of about 63 g $m^{-2}d^{-1}$ and seasonal estimates (180 days) of 1,200 g m^{-2}. Assuming that there is about 0.6 g cm^{-3} of inorganic material in consolidated sediments, this deposition is equivalent to 0.2 cm per growing season.

The potential errors associated with such a calculation are obvious, but it is interesting that such a reasonable value emerges. Currently, little information is available to suggest whether or not this material is sufficiently consolidated to be lost to the sediment deposition-resuspension-deposition cycles. Because of the important role of roots and rhizomes in this process and the low below-ground biomass observed in Chesapeake Bay SAV communities, it seems doubtful that this material is permanently consolidated at present. However, SAV biomass levels characteristic of the early 1960's may have been high enough to prevent significant resuspension of bottom sediments. Likewise, root-rhizome structure of these times may have more effectively consolidated bottom sediments.

COMPARISON OF SEDIMENT SOURCES WITH DEPOSITION IN SAV BEDS IN CHESAPEAKE BAY.

To place the sediment-trapping characteristics of SAV in the context of larger-scale sediment processes in Chesapeake Bay, we have developed a series of calculations that compare the magnitude of two major sediment sources to the deposition rate observed in SAV communities (Table 5).

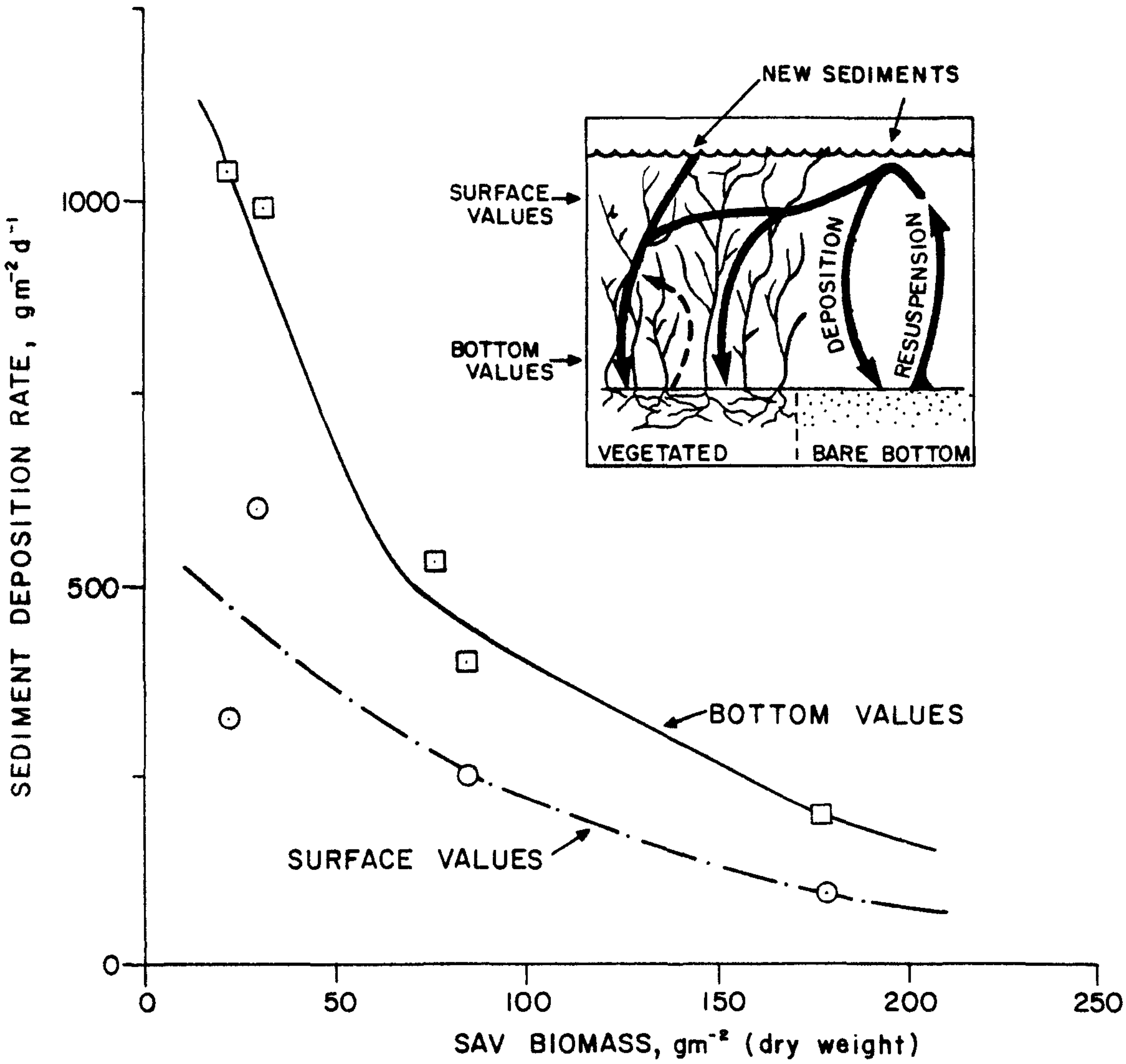

Figure 11. Relationship between SAV biomass and sediment deposition (adapted from Boynton et al. 1981b).

TABLE 5. ESTIMATED ANNUAL SEDIMENT DEPOSITION IN SAV COMMUNITIES RELATIVE TO SEVERAL SEDIMENT SOURCES IN CHESAPEAKE BAY FOR 1960 AND 1978. (ALL VALUES IN METRIC TONS PER YEAR x 10^6)

Sources	Time Periods 1960	1978
Riverine Input[a]	0.491	0.491
Shoreline Erosion[b]	0.375	0.375
Total	0.866	0.866
Deposition in SAV Communities[c]	0.72	0.08

[a]Includes Bay and tributaries above the mouth of the Potomac River ($1.5 \times 10^9 m^2$).

[b]Annual estimates of riverine and erosional sediment inputs from Biggs (1970). Assumed that inputs were relatively constant between time periods.

[c]Deposition in SAV communities estimated to be 1200 g $m^{-2}y^{-1}$ (Boynton et al. 1981; Ward, pers. comm.)

Major sediment sources include riverine input and shoreline erosion to the portion of Chesapeake Bay above the mouth of the Potomac River. We assume that estimates developed by Biggs (1970) are representative of both the early 1960's and late 1970's periods. The amount of sediment deposited during the SAV growing season was calculated from data of Boynton et al. (1981b), who estimate that some 1,200 grams of sediment may have been deposited per square meter of SAV community over an estimated 180-day growing season. Table 5 indicates that a large percentage of sediment may have been deposited in SAV communities during the 1960 period. However, in the late 1970's, when SAV distributions were severely reduced, the amount of deposition was less than 10 percent of the input. Although this calculation is preliminary, it suggests that SAV in the past may have played an important role in sequestering sediments in Chesapeake Bay, and that the amount of sediment presently deposited in SAV communities is small relative to estimates of sediment input.

LIGHT LIMITATION OF PHOTOSYNTHESIS

Although there appear to be emerging patterns concerning the role of SAV in modifying littoral zone turbidity and sediment cycling processes, it is still necessary to establish relationships between ambient light intensities and functions of SAV growth. Kemp et al. (1981) conducted a number of experiments to establish SAV responses (photosynthetic rate) to a range of light intensities. The two species investigated were P. perfoliatus and M. spicatum. They found that light saturated photosynthesis occurred at about 500-600 uEinsteins for both species, and that about 150 uEinsteins provided enough light to reach 50 percent of the maximum rate of photosynthesis (1/2 Pmax is similar to the Michaelis-Menten half saturation constant).

To extend these data to broad geographic regions of northern Chesapeake Bay, we examined attentuation coefficients characteristic of several locations during growing seasons (May to September). In Figure 12, a typical summer light intensity (just below the water surface) of 1,000 uEinsteins was attenuated using a range of attentuation coefficients (1.0, 2.0, 3.0) so as to display the light energy reaching various depths. Also plotted on the diagram (dashed horizontal lines) are the light intensities at Pmax, 1/2 Pmax, and 1/4 Pmax for *P. perfoliatus*. Thus, if an attentuation coefficient of 1.0 was observed, sufficient light to maintain light saturated photosynthesis reached a depth of 0.6 meters. The depth at which sufficient light penetrates to maintain photosynthetic rates at 1/2 Pmax is also given for various locations (Table 6).

These data suggest that in most locations light saturated photosynthesis does not occur in water depths greater than 0.25 to 0.5 meters. Moreover, sufficient light does not penetrate beyond 1.0 meter to maintain photosynthetic rates at 1/2 Pmax. Thus, it appears that only in the most shallow or most clear environments is light not limiting to SAV photosynthesis. These calculations may underestimate the limiting role of light because they are based on a subsurface light intensity of 1,000 uEinsteins, a value reached mainly during the middle of a typical summer day. On overcast days and in the early morning and late afternoon, values are considerably lower, and the depths of Pmax would be more shallow. Additional work, now in progress, may allow the development of better relationships between photosynthesis and light, as well as between photosynthesis and biomass.

TABLE 6. LITTORAL ZONE LIGHT EXTINCTION COEFFICIENTS DURING THE SUMMER IN CHESAPEAKE BAY DEPTHS AT WHICH 1/2 Pmax OCCURS ARE SHOWN DATA FROM TWILLEY 1981

	Location	Extinction Coefficient	Depth 1/2 P_{Max}
(1)	Upper Bay	2.2	0.8
(2)	Lower Bay	2.4	0.7
(3)	Tributaries	2.4	0.7
(4)	Upper Patuxent	3.5	0.5
(5)	Lower Patuxent	1.7	1.1
(6)	Eastern Bay	1.3	1.4
(7)	Lower Choptank	1.9	0.9

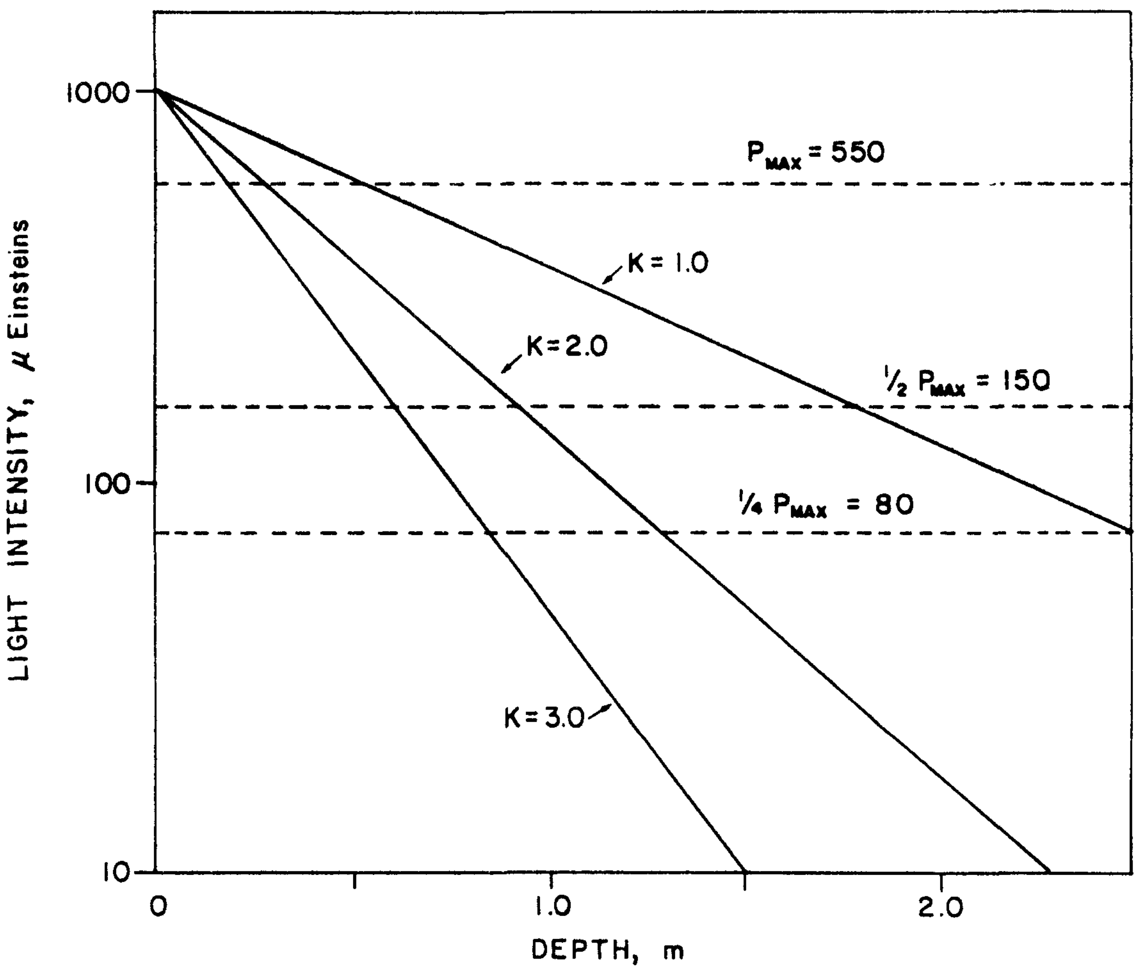

Figure 12. Relationship between surface light intensity and light attenuation of the water column, expressed at attenuation coefficients (K). Data from Boynton et al. 1981.

SECTION 5

NUTRIENT PROCESSES IN SAV COMMUNITIES

This section summarizes current knowledge of the effects of SAV communities on littoral zone nutrient regimes. Submerged vascular plants have two potential sources of nutrients available for uptake and incorporation into new biomass. Dissolved nutrients in the water can be taken up by leaves and stems, with some species using sediment nutrient reservoirs. SAV can also modify chemical conditions in sediments so that oxidized and reduced conditions prevail and can lead to several transformations of nitrogen and phosphorus.

This section addresses four categories of nutrient processes including: (1) nutrient concentration and fluxes in SAV communities; (2) nutrient regulation of SAV growth; (3) nitrogen transformations, including fixation, nitrification, and denitrification; and (4) nutrient releases associated with decomposition processes.

NUTRIENT CONCENTRATIONS AND FLUXES

Recent studies in Chesapeake Bay and global literature support the notion that SAV communities buffer nutrients by removing them from the water column, thus reducing concentrations. Pertinent examples from studies in Chesapeake Bay include the work of Twilley et al. (1981) and Kaumeyer et al. (1981). Twilley conducted a series of water quality measurements in the Choptank River estuary, an eastern shore tributary. Measurements were taken along the longitudinal axis of the estuary on a monthly basis from April through September. At one point adjacent to an intensively studied SAV community, water quality measurements were taken in an SAV bed in waters of moderate depth (4 m), and along the longitudinal axis of the estuary (deep water). Throughout this period, nutrient concentrations were consistently and dramatically lower in littoral, as opposed to deeper, sections along that sampling transect. Specificially, ammonium concentrations were one to 10 times lower, nitrate two to 10 times lower, and orthophosphate generally two to four times lower in the SAV community than in deeper offshore waters. Similar results were obtained at the Parson Island site, where nutrient concentrations appeared to be lower in an SAV community than in adjacent offshore waters (Kemp et al. 1979). An important ecological implication of these findings is that SAV may compete with phytoplankton for nutrients, thus reducing potential excessive algal blooms.

To elucidate mechanisms causing nutrient concentrations to be lower in the littoral zone, Kaumeyer et al. (1981) initiated a series of studies using a variety of sampling chambers in an SAV bed in the Choptank River estuary. The chambers were spiked with different levels of ammonium, nitrate, and phosphate, and concentrations of these nutrients (as well as dissolved oxygen) were measured hourly over six to 12 hour periods during both day and night. A typical set of results is given in Figure 13. Nutrient concentrations rapidly decreased from initial-spiked

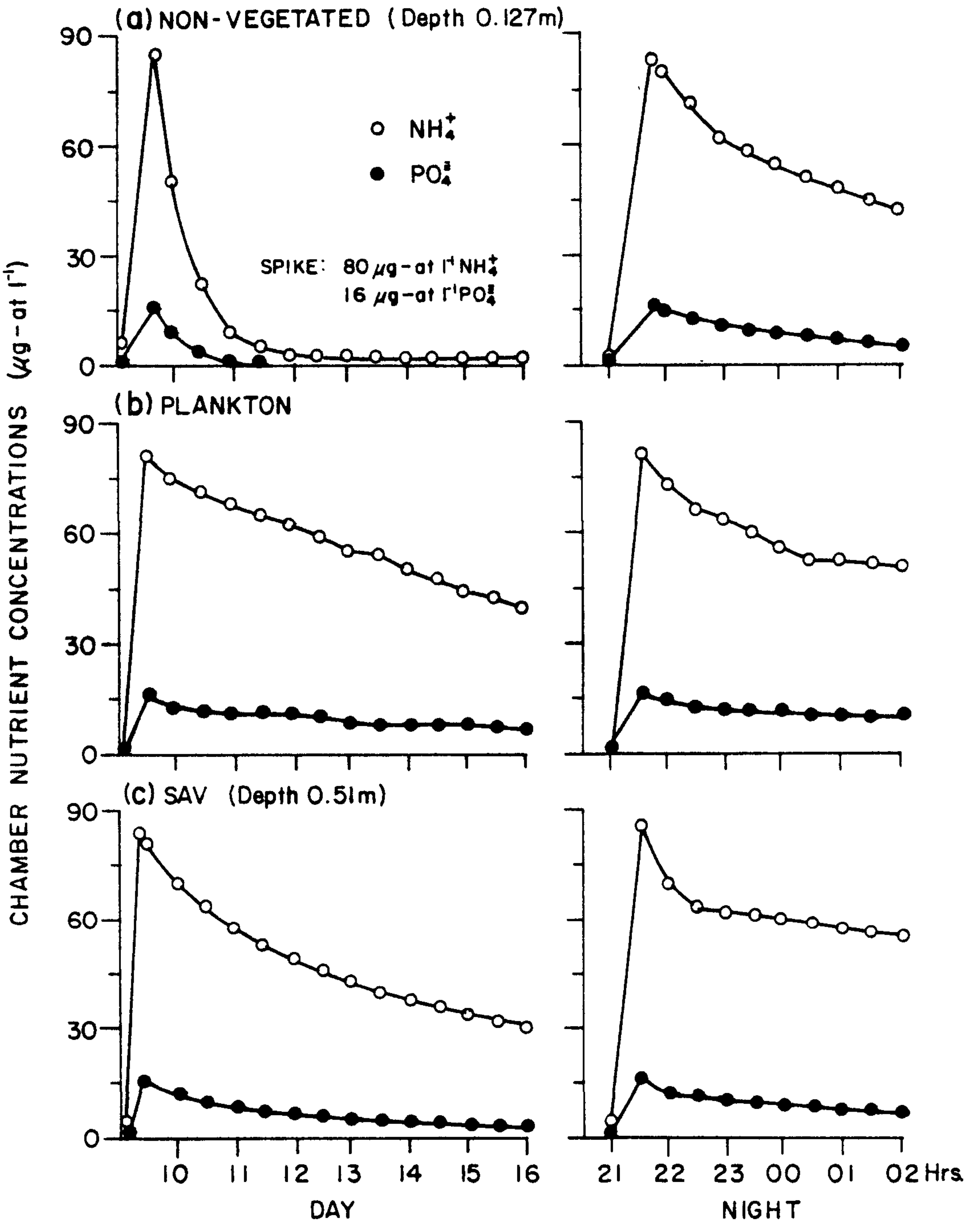

Figure 13. Chamber nutrient flux at Todds Cove, Choptank River, July 1980 for ammonia-nitrogen and dissolved inorganic phosphate. Day and night nutrient concentrations in experimental chambers all given for (a) non-vegetated sites, (b) plankton, and (c) vegetated sites.

concentrations to lower levels, and generally returned to near-ambient in less than 24 hours. Uptake rates, in most cases, depended on nutrient concentrations in the chambers. In addition, Kaumeyer et al. found where littoral communities were exposed to both nitrate and ammonium, both forms of nitrogen were taken up, although ammonium was generally taken up somewhat faster. These investigators did not attempt to partition uptake between plankton, benthos, and SAV, but apparently uptake rates were fast in all littoral zone communities investigated. These results suggest that there are mechansisms, not restricted to SAV, through which nutrients, entering the littoral zone, can be rapidly removed. Howard-Williams (1981) reports similar results from dosing a dense bed of P. pectinatus. He found that this community could rapidly reduce nutrient concentrations, and that filamentous algae associated with SAV were responsible for most of the phosphorus uptake.

In addition to these studies, substantial observational and experimental evidence indicates that SAV removes dissolved nutrients from the water column at a high rate. Mickle and Wetzel (1978) investigated SAV-nutrient exchanges in laboratory systems containing Scirpus and Myriophyllum. In these flow-through systems, nitrate and ammonium were introduced, and output concentrations monitored. Ammonium and nitrate concentrations decreased substantially after passing through SAV, particularly in the Myriophyllum beds. We conclude that littoral SAV systems are effective in damping higher concentrations entering the littoral zone following rainfall events. McCord and Loyacano (1978) further found that Chinese water chestnut (Eleocharis deucis) in freshwater ponds is effective in removing nitrate and ammonium from the water column. In their studies, ponds with water chestnut had lower concentrations of both nutrients and phytoplankton. Net nitrogen removal rates were estimated to be in the range of 4 mg $m^{-2}d^{-1}$. Twilley et al. (1981) found that nutrients (ammonium, nitrate, and dissolved inorganic phosphate) are removed at substantial rates from brackish-water ponds dominated by P. perfoliatus and R. maritima.

Although it appears that SAV reduces nutrient concentrations, existing evidence suggests that when loading rates and concentrations of nutrients reach certain levels, SAV is no longer effective. In fact, SAV can be stressed at these elevated levels of nutrients through several mechanisms. For example, Jupp and Spence (1977) found that in Loch Leven, Scotland, the diversity of SAV was reduced from about 23 species in 1910 to about 12 in 1975, and that this pattern of decreased diversity and abundance generally paralleled the increase in cultural eutrophication. They found that when phosphorus levels approached 2 ug-at I^{-1}, algal stocks increased (particularly blue-greens), while SAV distribution, species diversity, and abundance decreased to very low biomass levels (0 to 20 g m^{-2}). They further suggest that algal blooms may decrease SAV vigor through attenuation of light and increases in pH. Chlorophyll levels in Loch Leven were reported to exceed 200 ug I^{-1}, a concentration far in excess of those normally found in Chesapeake Bay at this time. They found that SAV tended to recover when chlorophyll levels were decreased to the vicinity of 20 to 40 ug I^{-1}.

In summary, Jupp and Spence (1977) constructed the following story concerning the effects of cultural eutrophication on SAV distribution. These events are similar to the sequence of decline in Chesapeake Bay SAV.

Increased loading of nutrients to Loch Leven, in particular phosphorus, increased algal stocks that led to a decrease in available light, both through/attenuation in the water column and through fouling on SAV by epiphytic species. The light restriction led to a restricted depth zone in which SAV species could flourish, and this, of course, was in shallower water. This restricted zone of growth was in an area where SAV was subjected to increased stresses by both wave action, and decreased light due to resuspension of littoral sediments and intensive grazing by waterfowl.

In other studies, similar results were found. Mulligan et al. (1976) found that SAV subjected to very high levels of nitrogen and phosphorus fertilization (4,000 g-at L^{-1} of nitrogen, 25 ug-at L^{-1} of phosphorus) were eliminated in pond ecosystems. They conclude that high loading rates of nitrogen and phosphorus favor phytoplankton stocks. Similar conclusions were reached by Sand-Jensen (1977) and Phillips et al. (1978). In the Chesapeake Bay area, experimental work by Twilley et al. (1981) suggests that loading rates, resulting in initial nitrogen (N) and phosphorus (P) concentrations of 60 and 6 ug-at L^{-1}, respectively tend to favor the development of algal stocks and the elimination of SAV.

Thus, it appears that the role of SAV in buffering nutrient concentrations in the nearshore zone has at least two aspects. If loading rates are moderate, SAV (and other littoral zone components) can rapidly decrease these concentrations to low levels. If loading rates and resulting concentrations are sufficiently high, SAV is disadvantaged and, in some cases, lost from the system and replaced by a phytoplankton component.

NUTRIENT REGULATION OF SAV GROWTH

Over the past fifty years, considerable (though sporadic) research has been directed toward understanding sources from which SAV obtains nutrients. Various studies indicate that root uptake is the major mechanism through which nutrient demands are met (McRoy and Barsdate 1970, Cole and Toetz 1975, Nichols and Keeney 1976, Twilley et al. 1977). In contrast to this, other evidence suggests that foliar uptake, under some conditions, is the predominant pathway (Nichols and Keeney 1976, Cole and Toetz 1975). We suggest that nutrient uptake is facultative in that if nutrient concentrations in the water column are very low, and adequate nutrient reserves exist in the sediment, then root uptake will dominante. Alternatively, if adequate nutrients are present in the water column, then foliar uptake will predominate.

To define ammonium uptake kinetics (Marbury et al. 1981), experiments were conducted in the upper Bay using _P. perfoliatus_. Foliar uptake matched classical Michaelis-Menten kinetics for both day and night conditions, with root uptake also partially described by these kinetics. In several experiments, Marbury found that K_m (K_m is the substrate concentration with the rate of nutrient uptake's one-half of the maximum uptake) approximates 15 u moles and corresponds to an uptake rate of approximately 0.19 to 0.31 mg N g of plant^{-1}hr^{-1}. These rates are comparable with those reported by other authors for several different species of SAV (Nichols and Keeney 1975, McRoy and Alexander 1975, Cole and Toetz 1975). During periods of rapid growth (May-July), total concentration of inorganic N in the water column often approaches or exceeds values of K_m and the interstitial concentrations of ammonium are in the range of one to three m moles. Because of this, it is doubtful that nitrogen limits submerged macrophytes in the mid-salinity and brackish water portions of Chesapeake Bay. (Marbury, personal communication). To the contrary, both observational and experimental evidence indicate that nutrient loading, particularly of N, may be sufficiently high to favor the replacement of SAV communities by phytoplankton (Phillips et al. 1978, Twilley 1981).

In the lower Bay, SAV growth may be somewhat more regulated by nutrient availability. Orth (1977) added large amounts of commercial N and P fertilizer to the sediment surface in Zostera beds and found significant increases in length, biomass, and number of stems. Sediments were sandy and may have had low concentrations of interstitial nutrients as has previously been reported for such sediments. This, coupled with characteristically low-nutrient concentrations in the water column, may produce nutrient-limited growth.

NITROGEN FIXATION, NITRIFICATION, AND DENITIRIFICATION

This section discusses three important processes in the nitrogen cycle: nitrogen fixation, nitrification, and denitrification. SAV's ability to convert dissolved nitrogen gas into an organic form ("fixing") is important during times of inorganic nitrogen impoverishment. Nitrification is the bacterial-mediated oxidation of ammonia to nitrate in the presence of free oxygen; denitrification is the reverse process of bacterial-mediated reduction in the absence of free oxygen. These last two processes provide energy to certain bacteria depending on whether the environment is aerobic or anaerobic.

These three processes are of ecological as well as of water quality significance, because they represent sources and sinks of nitrogen and may reflect the potential for regulating phytoplankton growth in many areas of Chesapeake Bay. A substantial range in N-fixation rates has been observed in seagrass communities. It appears that in nutrient-poor waters (low ambient concentrations of N in both the water column and in sediments), SAV growth can be N-limited (Patriquin 1972). Much of the nitrogen used in SAV growth may be supplied by N-fixation (e.g., Capone et al. 1979). Patriquin (1972) reports high rates of N-fixation in Thalassia beds, and Patriquin and Knowles (1972) conclude, based on studies in a variety of Thalassia beds in the Caribbean, that most of the N requirements are supplied by fixation. Fixation rates in these studies range between two to 10 mg-at $m^{-1}d^{-1}$, rates that are capable of supporting most, if not all, of the calculated N demand.

In contrast to these results, Lipschultz et al. (1979) report low rates of N-fixation in seagrass meadows in the Choptank River estuary. In the areas investigated by Lipschultz, nitrogen was abundant in the water column, and sediment reserves were substantial. Thus, it appears that N-fixation is facultative in the sense that if severe N-limitation exists in environments otherwise amenable to seagrass growth, N-fixation becomes a prominent feature. Conversely, in those systems, such as the mid-salinity and brackish zones of Chesapeake Bay, where abundant reserves of ammonium are contained in interstitial waters, N-fixation is simply not required.

Unfortunately, less is known about the rates of nitrification and denitrification in seagrass ecosystems. To our knowledge, the only published information available at this time is the work of Iizumi et al. (1980). In this study, a Zostera bed was investigated using N^{15} techniques. The authors report that rates of denitrification ranged from 0.5 to 1.2 x 10^{-9} g-at $g^{-1}h^{-1}$, and that nitrification rates were quite similar. When these values are converted to an areal basis, denitrification and nitrification are important aspects of the sediment-water nutrient cycle. Iizumi et al. (1980) report that high nitrification rates are directly coupled to denitrification, as expected, because the entry product (nitrate) to the denitrification pathway is the end product of nitrification. Moreover, they found that nitrification in anoxic sediments was made possible by the transport of oxygen from the foliar portion of SAV to the root zone. Thus, there were small microzones of oxidized sediment in which nitrification could proceed. After nitrate was produced, it diffused into the anoxic zone, where denitrifying bacteria rapidly transformed nitrate to nitrogen gas.

In studies conducted in an SAV community in the Choptank River and in brackish-water experimental ponds, Twilley et al. (1981) found that denitrification rates in both areas ranged from 50 to 100 uM $m^{-2}d^{-1}$; however, rates tended to be lower in SAV than in non-vegetated littoral zones (although such differences were not statistically significant). Jenkins (personal communication) found much higher rates of denitrification (about 200 to 300 ug-at N $m^{-2}d^{-1}$) in deeper portions of Chesapeake Bay waters in the spring when nitrate was abundant in overlying waters. Rates were low or undetectable at other times of the year when nitrate was not present in the water column. Evidence that nitrification rates are substantial in SAV communities is accumulating from studies of SAV beds in the upper Bay, and these rates appear to be substantially higher than nitrification rates in soft-bottom communities lacking SAV.

What then are the mechanisms responsible for these observations? At this point, it seems that oxygen produced in the foliar portions of SAV is translocated to the roots and from the roots into the interstitial waters, supplying the oxygen needed to support nitrification. Although the nitrate produced could be used in denitrifiction, evidence at this point indicates that other processes may out-compete denitrification for this nitrate. Recent studies by Terlizzi (personal communication) of diel nitrogen cycling in P. perfoliatus-dominated microcosms (700 liter with natural estuarine sediments and water) showed that nitrate concentrations increased in the roots during the daylight hours and decreased at night with a concomitant appearance of nitrite. They suggest that the oxygen produced in this reaction was used in support of root respiration at night, yielding nitrite. The eventual fate of the nitrite produced in these roots is not currently known, although some of it leaked from the roots into the interstitial and overlying waters; the nitrite had nearly vanished by the return of daylight. Whether or not nitrite was oxidized to nitrate or reduced to nitrous oxide or nitrogen gas is presently not known. Thus, in contrast to the studies of Iizumi et al. (1980), these results suggest that denitrification is important in deep waters when nitrate is abundant in the water column. In SAV communities, measurements of denitrification have, by and large, indicated that rates are small. Nitrification, on the other hand, appears to be enhanced by the translocation of oxygen to the root zone, but all of the nitrate produced does not appear to enter the denitrification pathway.

NUTRIENT RELEASE AND OXYGEN DEMAND ASSOCIATED WITH SAV DECOMPOSITION

A great deal of evidence points to the importance of submerged and emergent macrophytes as a source of detritus available to coastal and estuarine heterotrophs. Paralleling this, there is a considerable amount of scientific literature concerning the decomposition and release of nutrients for some higher plants, and in particular, decomposition characteristics of Spartina. Surprisingly, less is known about decomposition characteristics of submerged macrophytic vegetation. Several studies are available, however, that are pertinent to a discussion of the decomposition process.

In addition to the role of SAV as a detrital food-source, the relative impact of decomposing plants has been investigated in terms of oxygen utilization. We hypothesized that submerged aquatic vegetation serves as a temporary nutrient sink in that during the growth of SAV, N and P are taken up from either the water or sediment, depending on local conditions, and incorporated in SAV biomass. However, SAV decomposes; during this process oxygen demand is exerted, and nutrients are presumably released back to the water column.

Data from studies comparing SAV with phytoplankton and a macrophytic alga suggest that SAV decomposition exerts a small oxygen demand, tending to retain nutrients to a greater extent than other plants. Some experiments investigated the extent of the oxygen demand exerted during the decomposition process and the rapidity with which nutrients are released to the water column. Results are summarized in Figure 14 (Twilley, personal communication). In these experiments, a variety of primary producers, characteristic of the Chesapeake Bay system including Ulva and Spartina, were placed in small laboratory microcosms and allowed to decompose over a 90-day period. At frequent intervals, oxygen concentration, rate of oxygen concentration change, and ammonium and orthophsophate concentrations were monitored. As indicated in Figure 14, the dry-weight loss expressed as a percent per day was highest in phytoplankton and Ulva, somewhat less in three SAV species, and lowest in Spartina. The mean dry-weight loss per day developed in these experiments was only slightly lower than those observed in field studies. Spartina and phytoplankton species had the highest rates of oxygen utilization; rates for the three SAV species (Milfoil, Potomageton, and Ruppia) were the lowest. These results suggest that SAV exerts only a small oxygen demand on a daily basis over the decomposition period. This observation is important because in some parts of Chesapeake Bay, bottom waters become anoxic during the summer because of excessive deposition of labile organic material (primarily of phytoplankton origin).

Nutrient releases from SAV species and Spartina were low relative to the release observed for phytoplankton cultures and Ulva. After 70 days of incubation in microcosms, the ammonium concentrations in experimental systems of Milfoil, Potomageton, Ruppia, and Spartina were on the order of one to two ug-at L^{-1}, while phytoplankton and Ulva decomposition resulted in concentrations in excess of 10 to 14 ug-at L^{-1}. A similar, although

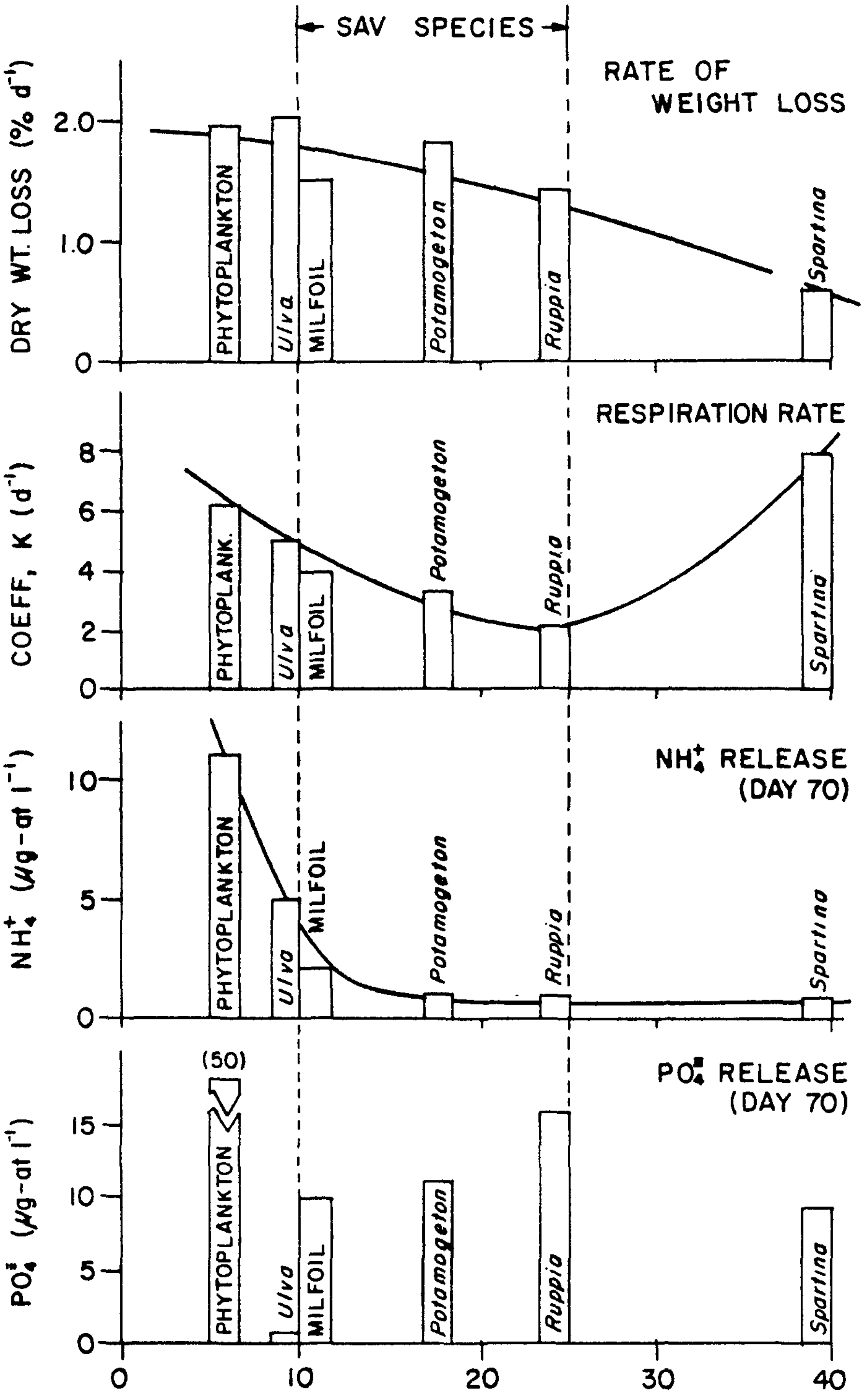

Figure 14. Comparisons of weight loss, respiration rate, ammonia-nitrogen release and dissolved inorganic phosphorus release for representative species of SAV, algae (Ulva), and marsh grass (Spartina alterniflora).

not quite so radical, difference was also noted for orthophosphate. After 70 days of incubation, phosphate concentrations in the phytoplankton tanks were about 50 ug-at L^{-1}, while in the SAV and Spartina microcosms concentrations ranged from about 9 to 15 ug-at L^{-1}.

Harrison and Mann (1975) conducted decomposition experiments in the laboratory using Zostera blades exposed to 20°C (68°F) temperatures. They observed that Zostera lost up to 35 percent of its dry weight in 100 days in decomposition. The decomposition rates for whole leaves and particles less than one millimeter were approximately 0.5 percent and one percent a day. Leaching of organic matter was responsible for a large fraction of organic matter loss. In terms of the nutrient content of detrital material, the addition of bacteria markedly increased the nitrogen content of organic matter, but did not substantially change the decay rate of detrital particles. The addition of protozoa with the bacteria increased both the nitrogen content and the decay rate of detritus; C:N ratios changed from about 20:1 in living blades to a minimum of 11:1 in detrital particles subjected to bacterial and protozoan treatments. Harrison and Mann further found that total organic matter, dissolved organic C, particulate organic C, and N were also highest in new Zostera leaves and decreased rapidly after death.

In studies using the same species, Thayer et al. (1977) found that during senescence, N content decreased and subsequently increased as blades became detrital. They attributed this action to microbial growth and further speculated that most of the nitrogen increase was due to microbial immobilization of N from surrounding waters. If bacterial immobilization of dissolved N is a general feature of the decomposition process, then it represents yet another mechanism by which SAV can reduce ambient nutrient concentrations in the water column.

In studies conducted in Chesapeake Bay, Staver (personal communication) placed above-ground portions of living P. perfoliatus in three-millimeter and one-millimeter mesh nylon bags and suspended these in the field. Bags were retrieved at different times, and the amount of SAV material remaining was measured. Results of these studies indicate that, at the temperatures commonly encountered [25 to 30°C (77 to 86°F)], decomposition in these bags was rapid, averaging about two percent a day (Figure 15). Although C:N ratios of this material are not available, we expect that over time the N content of remaining material would increase. Although data concerning decomposition and the nutritive status of decomposing material are far from complete, evidence from other areas indicates that as submerged macrophytic material dies, there is an initial loss in many components, including N, followed by an increase in N content, probably mediated by bacterial incorporation of N from the surrounding medium. This material probably serves as an adequate food source for many heterotrophs that ingest detrital particles, metabolize the microorganisms, and excrete the detrital fragment.

COMPARISON OF NUTRIENT BUFFERING CAPACITY OF SAV WITH IMPORTANT SOURCES

To evaluate the potential nutrient buffering role of SAV in the context of Bay-wide nutrient sources, we have developed a crude budget for which the magnitude of nitrogen sources to the upper Chesapeake Bay are compared with the amount of nitrogen incorporated into SAV biomass during a normal growing season.

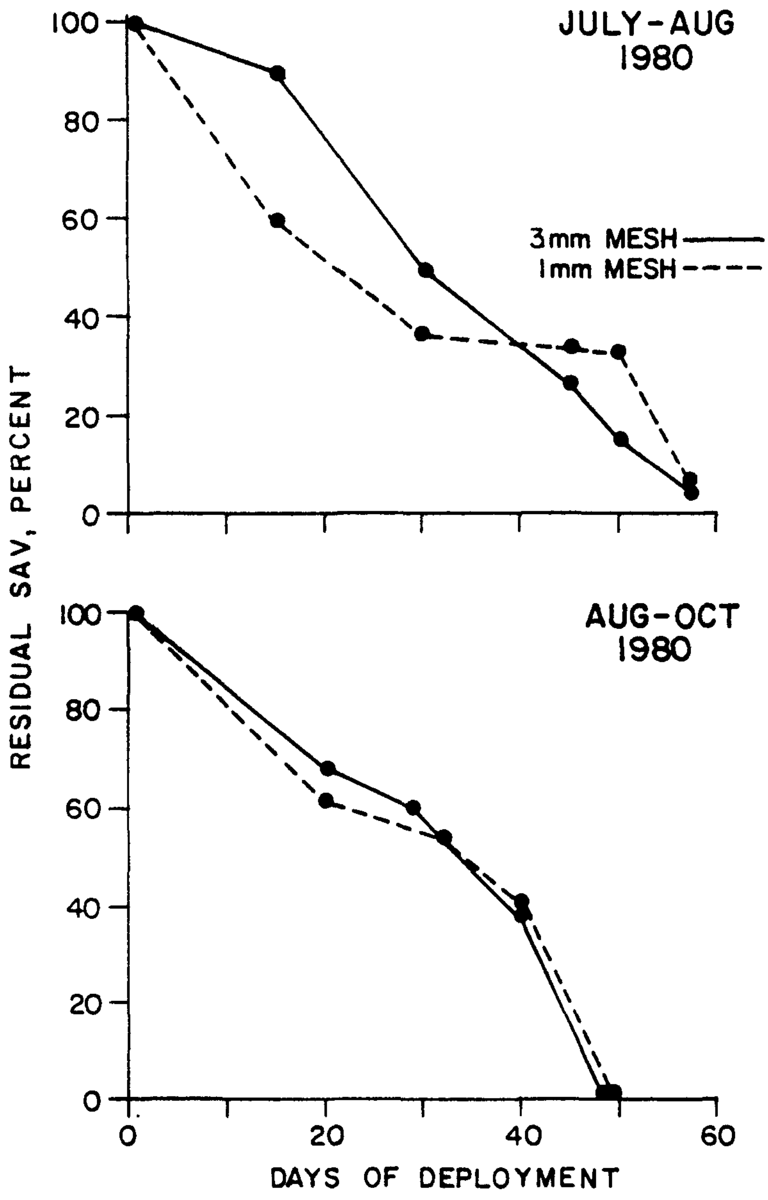

Figure 15. Decomposition rates of _P. perfoliatus_ estimated using _in situ_ litter bags. Data were collected in the vicinity of the Todds Cove study site in the Choptank River (Staver, personal communication).

As indicated in Table 7, something on the order of five percent of the total nitrogen input to the upper Chesapeake Bay could have been immobilized by incorporation into SAV biomass during the 1960's. An extremely small percentage of total nitrogen input may be immobilized by SAV uptake at the present time (0.5 percent). Estimates of sewerage input during the 1960's were not available, and we were not able to contrast SAV uptake relative to sewerage input. However, it is interesting to note that SAV uptake in the 1960's could account for approximately 50 percent of the present sewerage input. Uptake represents only one of several possible mechanisms used by SAV to buffer the nutrient regime in estuarine waters. As indicated earlier, denitrification may represent a substantial sink, although at this point the exact magnitude of this process remains unclear. It should be pointed out that Table 7 provides no estimate of atmospheric input; however, it probably approximates 10 to 12 percent of the total value for 1978 (Smullen et al. 1982).

TABLE 7. ESTIMATED INPUTS OF NITROGEN TO THE UPPER CHESAPEAKE BAY FROM RIVERINE AND SEWAGE SOURCES, AND UPTAKE OF NITROGEN BY SAV (ALL VALUES ARE IN UNITS OF $KgNy^{-1}x10^6$) (MACOMBER 1980; STEVENSON, PERSONAL COMMUNICATION)

Sources	Time Periods 1960	1978
Riverine Input[a]	50	50
Sewage Inputs[b]	d	5.3
Total	≈ 50	55.3
SAV Uptake[c] (During growing season)	2.4	0.3

[a]Riverine source of nitrogen calculated using regression relationships between Susquehanna River flow and nutrient concentrations (Guide and Villa 1972).

[b]Sewage input data from Smullen (personal communication)

[c]Uptake calculated using N content of SAV of 2% and SAV Standing crop of 200 gM^{-2} for both time periods. Areas of SAV coverage were estimated as $600x10^6m^2$ and $66x10^6m^2$ in 1960 and 1978, respectively (Rawls, in prep.; Anderson and Macomber 1980; Stevenson, personal communication).

[d]Not available.

SECTION 6

SUMMARY

Four distinct processes related to the ecological role and value of SAV were examined in the Chesapeake Bay Program: (1) estimating the magnitude of SAV organic matter production and availability to local food webs; (2) examining habitat value of SAV to infaunal and juvenile nekton species; (3) estimating the role of SAV in modifying, reducing, and serving as a sink for nearshore sediments; and (4) examining the role of SAV in modifying nutrient dynamics of nearshore regions.

As we have shown in previous sections of this report, it appears that SAV influences each of these processes. However, the importance of the SAV component in the Bay community at the present time is probably small because of the restricted distribution of this vegetation. At one time, SAV probably played a substantial role in organic matter production, habitat maintenance, and sediment and nutrient dynamics. The purpose of this section is to highlight findings concerning the role of SAV in the above processes and to place processes associated with SAV in the context of large portions of Chesapeake Bay.

ORGANIC MATTER PRODUCTION AND UTILIZATION

The productivity rates of SAV communities in Chesapeake Bay are comparable to with those observed in other SAV systems distributed over large latitudinal ranges and environmental gradients. Net productivity values associated with several types of SAV in Chesapeake Bay were as high as those reported for other species in other areas. In sharp contrast to the comparability of production values between SAV systems, estimates of SAV biomass exhibited a large over all range, and substantial differences were evident within the same type of system. In general, higher standing stock values of SAV occurred in areas where the water is relatively clear, deep (enough to allow for substantial vertical growth of SAV), and devoid of extensive wave action. Moreover, average biomass (and even maximum biomass) estimates in Chesapeake Bay were low relative to those reported for other areas. For instance, average values of *Zostera* and *Ruppia* in the lower Bay were generally below 200 g m^{-2}; values for *Potomageton pectinatus* and *P. perfoliatus* in the upper Bay were generally below 100 g m^{-2}. At the present time, sufficient light to support vigorous growth of SAV does not penetrate much beyond one meter in most littoral regions of the upper Chesapeake Bay. Thus, growth is restricted in very shallow regions where there is a limited water column to support the vertical development of SAV, and the potential for wave, thermal, and waterfowl grazing stresses is maximized. In earlier years (pre-1970) when light penetration was not so restricted, SAV in the upper Bay may have grown in waters of greater depth and been characterized by higher standing stocks.

Comparison of SAV biomass for several species in the lower and upper Chesapeake Bay has made several differences apparent: (1) peak biomass of *M. spicatum* was greater than *Z. marina*, and the biomass of this species was greater than *R. maritima*; (2) the peak biomass of *R. maritima*, *P.*

pectinatus and P. perfoliatus approximated each other; (3) with the exception of M. spicatum, mean biomass values were consistently higher in the lower Bay, often by a factor of two or more; (4) in the lower Bay, above-ground biomass persisted through winter months, but in the upper Bay, above-ground material was present only during the warmer months; and (5) periods of peak biomass occurred earlier in the year (June) in the lower Bay than in the mid-salinity zone (July to August). We do not have quantitative information concerning biomass levels or seasonal persistence prior to the initiation of the decline. However, ancedotal information suggests that biomass values were higher in the upper Bay than they are at the present time and persisted through the fall months.

Submerged aquatic vegetation can enter heterotrophic food webs either by direct grazing of living plants or by consumption of SAV detritus. The majority of studies conducted suggest that SAV is an adequate food item, that it is primarily available as detritus and, in some localities, it may be a dominant food source. Several substantial results link SAV production to use in Bay food webs, with perhaps the most definitive connection between SAV and waterfowl. Numerous authors found that vegetable matter was an extremely important food item for waterfowl in the upper Chesapeake Bay. Furthermore, the most important waterfowl wintering areas are also those most abundantly vegetated. It appears that direct grazing on SAV by waterfowl is important both in the Chesapeake and elsewhere, and that grazing in itself can locally impact the distribution of SAV.

Aside from this direct grazing pathway, the vast majority of studies, including those in Chesapeake Bay, indicate that most SAV material enters food webs through detrital pathways. For example, in the lower Bay, sea bass, pipefish, pigfish, and white perch are epibenthic feeders, using amphipods and shrimp that are, in turn, detrital feeders. Data further indicate that large predators enter SAV beds with little food in their stomachs and leave after feeding. Food items for these feeders can generally be traced back to detrital sources, some fraction of which is probably SAV in origin. In an extensive study of feeding habits in the upper Bay, little evidence was found for direct grazing by fish on SAV, although some SAV seeds and plant parts were found in stomachs. Energy flow appears to enter food webs as detritus and to pass through epifaunal and infaunal invertebrates to small and large fish. Numerous epifaunal species, which are important food items for many consumers, were also closely associated with SAV.

Because of the complexity of organic matter sources in Chesapeake Bay and the current marginal distribution of SAV, a quantitative assessment of SAV's importance as a food source is not possible. However, it is reasonable to argue that the available SAV is used by heterotrophs, and that SAV's physical structure concentrates other foods (phytoplankton, epiphytic algae, and benthic macroalgae) for animal consumption. Studies conducted concurrently with SAV research indicate that on an annual basis virtually all carbon inputs in Chesapeake Bay were utilized by heterotrophs of one sort or another. Thus, if heterotrophic metabolism is organic-matter limited, it follows that SAV would also be used if, as has already been shown, this material is a suitable food source. Furthermore, loss of SAV production may well lead to loss of fishery production, especially if production by phytoplankton fails to compensate for the loss of food to higher trophic levels.

HABITAT VALUE OF SAV

Studies in the upper and lower Chesapeake Bay indicated that infaunal abundance and diversity is higher in vegetated than in unvegetated areas. Because unvegetated habitats support a virtually non-existent epifauna, epifaunal densities were naturally higher at the vegetated sites and were important food items in Chesapeake Bay food webs.

Finfish sampling at sites in the upper Bay indicate greater abundances and species richness in vegetated than in unvegetated bottoms; fish densities were among the highest yet reported in the literature. In addition, average weight per individual during the summer period was low in SAV communities as compared with unvegetated areas, suggesting that SAV communities are continually used as nursery areas for small fish, and larger-sized animals predominate in unvegetated areas. Fish sampling programs in the lower Bay also found greater abundances and species richness in eelgrass meadows than in nearby unvegetated bottoms. Some large fish predators, such as weakfish and the sandbar shark, foraged most often over vegetated bottom, whereas others, such as bluefish, appeared to forage indiscriminately over both vegetated and unvegetated areas.

The main conclusion of these field studies is that fish communities are richer in vegetated than unvegetated areas. However, few commercially important finfish were found to use SAV beds as significant nursery habitats. The role of SAV for commercial fishes in the Chesapeake system seems to be largely that of a rich foraging place for adults, and not that of a nursery habitat although, once again, it is important to emphasize that the current restricted distribution of SAV may bias these conclusions. (For instance, major spawning and juvenile habitats for striped bass once existed in the upper Bay in an area that was densely populated with SAV.) More representative patterns of commercial fish use of SAV habitat might best be evaluated through historical correlations of SAV and juvenile fish distributions.

Information concerning blue crab abundance was collected at the same time fish were sampled at sites in the upper and lower Chesapeake Bay. During comparable months, up to 10,000 times as many blue crabs were found at the lower Bay site. In addition, most of the crabs in the high-salinity eelgrass beds were juvenile females that constituted the breeding stock for future generations. The conclusion drawn from these studies is that SAV in the upper Bay serves as a very limited blue crab nursery. Lower Bay eelgrass beds, however, serve as primary blue crab nurseries, supporting very large numbers of juvenile blue crabs throughout the year. It should be noted, however, that upper Bay SAV beds may well provide a protective habitat for molting adult blue crabs.

Experimental studies involving exclusion of predators from certain areas of SAV beds indicate that predation rates on some infaunal taxa were lower in vegetated than unvegetated areas, and predation rates on epifaunal species seemed to be lower in SAV habitats than elsewhere. Laboratory microcosm experiments supported the notion that SAV provides less protection for infauna than it provides for epifauna of SAV beds. It also seems that artificial eelgrass provides protection roughly equivalent to that of live eelgrass, and that SAV species with finely divided leaves provide (other factors being equal) more protection than do SAV with simple, unbranched leaves. It is clear that SAV-associated animals do feed in the beds, and that the food supply is considerably greater in SAV communities than in other available habitats.

SEDIMENT PROCESSES

Results of recent studies indicate that SAV can substantially influence sediment dynamics in littoral zones. Specifically, SAV stabilizes sediments; slows currents, allowing sediments to settle (which increases light penetration into the water column); and substantially reduces current-and-wave-induced resuspension. In Chesapeake Bay, Orth (1977) reports that sediment particle diameter decreased, and organic matter content and infaunal densities increased in sediments in areas with SAV, as compared with those that did not have such coverage. Other findings and observations showed less sediment disruption during storms in vegetated zones, and less dispersion of dyed sand patches. Based on this information it was concluded that SAV is effective at trapping and consolidating suspended sediments.

Data developed at intensive study sites in the upper Chesapeake Bay indicated that as turbid water entered SAV beds on rising tides, sediments were effectively removed, increasing light transparency. It appears that by high tide, turbid inflowing waters normally exceed the filtering capacity of SAV beds. As tidal height decreases, the bed effectively filters sediments, and the turbidity gradient between vegetated and non-vegetated areas again increases to a maximum. In addition, resuspension was reduced in SAV communities with the reduction proportional to SAV biomass.

Because of the dynamic nature of the sediment-water interface in littoral environments, estimates of net sedimentation are exceedingly difficult to obtain. A crude estimate was made from observations based on differences in seston concentrations inside and outside SAV beds. These calculations indicated that daily deposition rates of sediment were about 63 g $m^{-2}d^{-1}$, yielding seasonal estimates on the order of 1,200 g m^{-2}. If we assume that there is about 0.6 g cm^3 of inorganic material in consolidated sediments, this deposition is equivalent to about two millimeters per growing season. At the present time, we do not know if material deposited when SAV was present is subsequently lost when SAV dies in the fall. Considering the important role of roots and rhizomes in the process of sediment consolidation and the low below-ground biomass observed in Chesapeake Bay SAV communities, we doubt that this material is permanently consolidated at present, although it may have been in the past.

To place the sediment-trapping characteristics of SAV in the context of larger-scale sediment processes in Chesapeake Bay, we have developed a series of calculations that compare the magnitude of two major sediment sources with the deposition rate observed in SAV communities. Major sediment sources include riverine input and shoreline erosion to the portion of Chesapeake Bay above the mouth of the Potomac River. The amount of sediment deposited during the SAV growing season was based on the data of Boynton et al. (1981b). They estimated that some 1,200 grams of sediment were deposited per square meter of SAV community over an estimated 180-day growing season. A large percentage of sediment input could have been deposited in SAV communities during the 1960 period. However, in the late 1970's, when SAV distributions were severely reduced, the amount that could have been deposited was reduced to something less than 10 percent of the input. Although this calculation should be considered preliminary, it suggests that SAV in the past may have played an important role in sequestering sediments in Chesapeake Bay, and that the amount of sediment deposited in SAV communities at the present time is small relative to estimates of sediment input. Presumably, the difference in sediment trapped in 1960 versus 1978 is spread over the bottom of the Bay, with some part available to resuspension.

NUTRIENT PROCESSES IN SAV COMMUNITIES

In an earlier section of this report, we argued that SAV communities are capable of buffering nutrients between littoral and pelagic zones of the estuary. Recent studies in Chesapeake Bay tend to support this notion. Measurement of nutrient concentrations in offshore areas and in SAV communities indicates that nutrient concentrations are consistently lower in the SAV communities. In addition, experimental studies involving the addition of nutrients to SAV communities indicate that nutrients are rapidly removed from the water column; ambient nutrient levels are reestablished 12 to 24 hours after additions. It has not been determined, however, which autotrophic component is most responsible for the uptake of these nutrients.

Several experimental studies were also conducted to examine rates of nitrification and denitrification in SAV communities. These experiments attempted to quantify the potential of SAV as a nutrient sink. In studies conducted in an SAV community in the Choptank River and in brackish water experimental ponds, we found that both areas exhibited substantial denitrification rates (50 to 100 ug-at $m^{-2}d^{-1}$) and rates tended to be lower in SAV than in nonvegetated littoral zones. Studies in the upper Bay also showed that nitrification rates are substantial in SAV communities. These rates were higher in SAV areas than in soft-bottom communities not having SAV. Although nitrate produced from this reaction could be used in denitrification, evidence at this point indicates that other processes may out-compete denitrification for this nitrate.

In addition to the role of SAV as a detrital food source, the relative impact of decomposing plants on oxygen utilization and nutrient release rates was investigated. We found that *Spartina alterniflora* and phytoplankton species had the highest rates of oxygen utilization, and three SAV species had the lowest rates. These results suggest that SAV exerts only a small oxygen demand on a daily basis over decomposition periods. This observation is important in that bottom waters in some parts of Chesapeake Bay become anoxic during the summer because of excessive deposition of labile organic material.

Nutrient release rates from SAV species and *Spartina* were low relative to release rates observed for phytoplankton cultures and *Ulva*. After 70 days of incubation, ammonium concentration in experimental systems of SAV was one to two ug-at L^{-1}, while phytoplankton and *Ulva* decomposition resulted in concentrations in excess of 10 to 15 ug-at L^{-1}, respectively. These data suggest that SAV exerts a small oxygen demand while decomposing and tends to retain nutrients relative to phytoplankton and to the one macrophytic alga tested.

To evaluate the potential nutrient buffering role of SAV in the context of Bay-wide nutrient sources, we developed a crude budget in which the magnitude of nitrogen sources to the upper Chesapeake Bay were compared with the amount of nitrogen incorporated into SAV biomass during a normal growing season. About five percent of the total nitrogen input to the upper Chesapeake Bay could be immobilized via incorporation into SAV biomass during the 1960's. In contrast, an extremely small percentage of total nitrogen input (0.5 percent) could be immobilized via uptake by SAV at the present time.

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HERBICIDES IN CHESAPEAKE BAY AND THEIR EFFECTS ON SUBMERGED AQUATIC VEGETATION:

A Synthesis of Research Supported by U.S. EPA Chesapeake Bay Program

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SECTION 1

INTRODUCTION

The widespread use of herbicides for weed control in the last several decades has contributed substantially to expanding agricultural production in North America. Annual applications of herbicides in the United States currently amount to some 175,000 metric tons active ingredients. The dramatic increase in herbicide use since 1950 has followed a general pattern of exponential growth as seen in Figure 1. Also depicted in this figure is the ever increasing importance of the *s*-triazine herbicides (and in particular atrazine) between 1960 and 1975.

Inevitably, a fraction of the herbicides applied to agricultural fields is transported to nearby watercourses by runoff and subsurface interflow. Significant concentrations of these compounds have been observed in streams, lakes, and estuaries throughout North America (Richard et al. 1975, Truhlar and Reed 1976, Newby et al. 1978, Frank and Sirons 1979, Hormann et al. 1979). Since many of these compounds are also registered for aquatic weed control (individually or as part of a formulation), there appears to be considerable potential for inadvertent damage to non-target plant species in the hydrosphere.

Submerged aquatic vegetation (SAV) in Chesapeake Bay has undergone a marked decline throughout the estuary since the mid-1960s (Stevenson and Confer 1978). Both the piedmont, and coastal-plain portions of the Bay's watershed are actively farmed, and herbicide use in this region has generally followed trends in the rest of the United States. The general coincidence in timing of events (that is, introduction of *s*-triazines versus the initial decline in SAV) led to a serious concern among scientists, resource managers, and other citizens of the region as to the potential role that these herbicides may have played in the loss of SAV in Chesapeake Bay.

The U.S. Environmental Protection Agency's (EPA) Chesapeake Bay Program (CBP) established SAV as one of three major themes of a multi-year research effort. Causes of the SAV decline, with considerable emphasis placed on investigating the interactions between herbicides and SAV in the estuary, were among the issues addressed in this program. Numerous aspects of herbicide fate, transport, and effects were examined in this research. The interrelationships among various processes and the potential linkage between herbicide application and effects on SAV are depicted in Figure 2. Some of the herbicides placed on agricultural fields percolate into subsurface waters where they reach a sorption equilibrium with soil particles and are taken up by weeds. The herbicide compound usually kills the weeds. A portion of the compound degrades to various metabolites, and a portion enters the estuary through runoff, leaching, and streamflow. Some of the herbicide may be volatilized and/or transported with dust, thereupon entering the estuary through fallout. The herbicide may then be taken up by SAV, causing them phytotoxic stress. In the estuary, the herbicide partitions to sediment and water in response to the physical factors of salinity, pH, and temperature as well as to the specific chemistry of the sediment and herbicide. Again, some of the herbicide is lost to degradation.

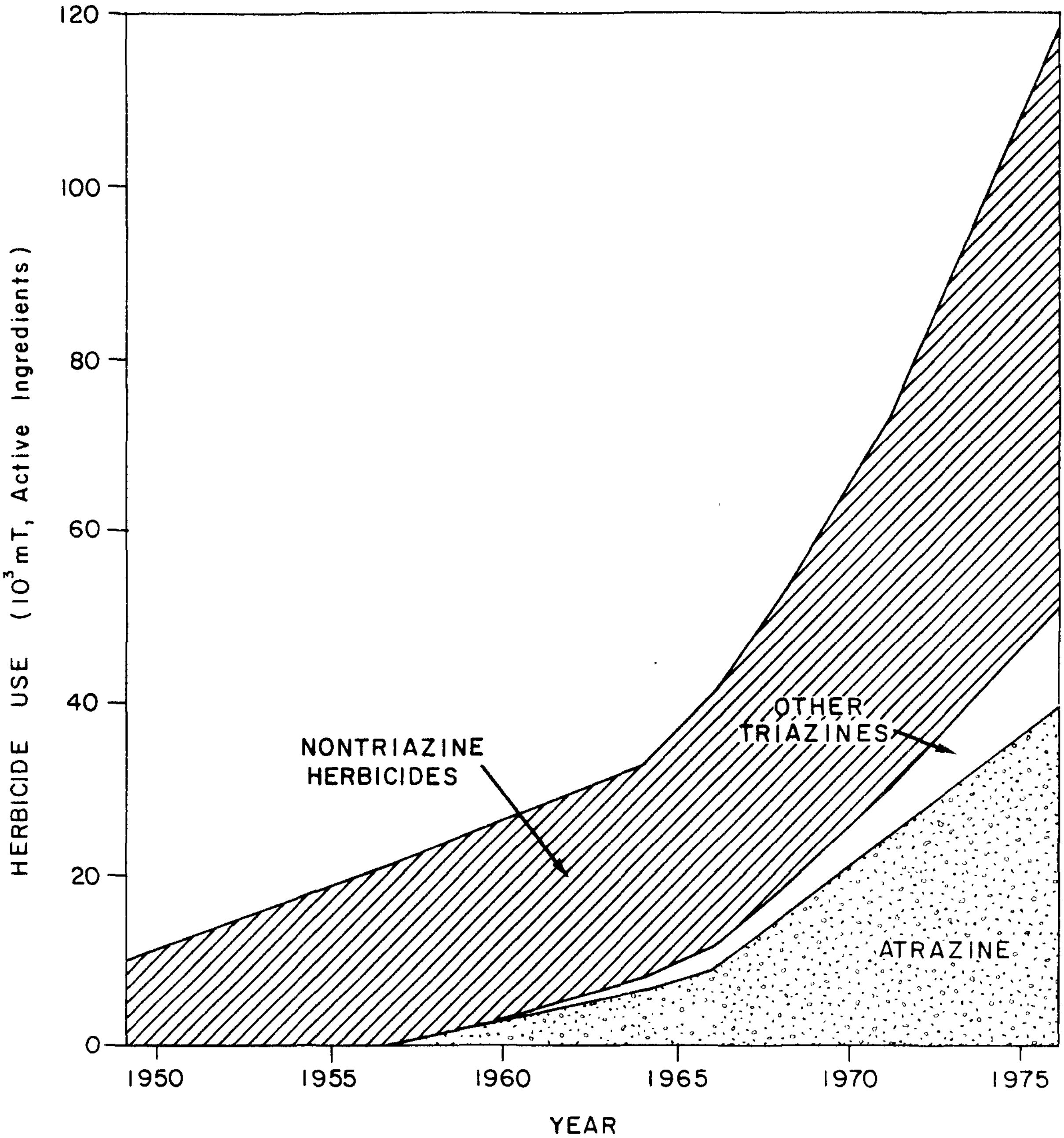


Figure 1. Herbicide use in the United States. (Data are from Eichers et al. 1978 as adapted in Stevenson et al. 1981.)

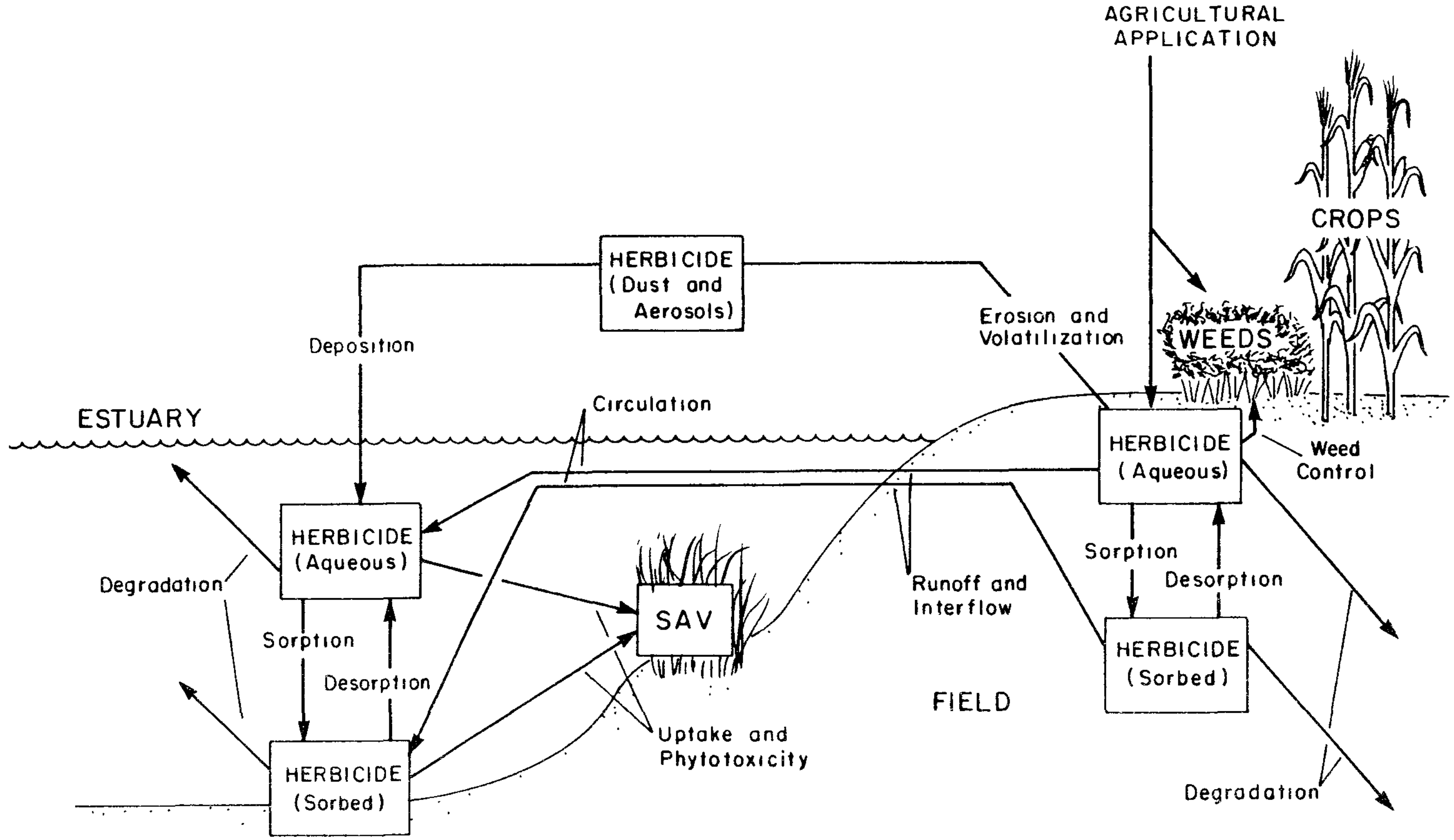


Figure 2. Herbicides in the Chesapeake Bay region.

In the following pages, the results of herbicide-related research from the CBP are synthesized. First the nature of these herbicides and the rationale for selection of two compounds for intensive study is discussed, then these results are examined in the context of the conceptual framework of Figure 2 and in relation to pertinent research done elsewhere. Finally, the overall implications of these research findings are evaluated in terms of the role of herbicides in the SAV decline.

SECTION 2

RATIONALE FOR SELECTION OF COMPOUNDS STUDIED IN THE CHESAPEAKE BAY PROGRAM

A wide assortment of herbicides is used within the Chesapeake Bay watershed, and it would be impossible to study all of them in detail. It was decided that initial research should, therefore, focus on the two major compounds. All pertinent criteria, in terms of potential impact on SAV, were considered, and atrazine and linuron were chosen. In this section we discuss the general chemistry of the important herbicides in the Chesapeake Bay region, as well as patterns of their use. We then present the rationale for selecting these two particular compounds for intensive study.

HERBICIDE CHEMISTRY AND USE

There are over 140 herbicidal compounds listed in the current Herbicide Handbook (Weed Science Society of America, 1980), which probably represents the majority of those weed-control substances registered with EPA. Eight compounds from six chemical groups were chosen for discussion here, based on amounts of each used in the Bay region. The annual use-rates for major herbicides in 1975 are summarized in Figure 3(a) for Maryland and Virginia, and Figure 3(b) for the Choptank River watershed. Clearly, the four most heavily used compounds are atrazine, alachlor, linuron, and simazine. Application rates are also shown for six additional compounds, of which four have been chosen for further discussion.

Many of the important herbicides are produced by chlorination of aromatic compounds, including 2,4-D and dicamba; other compounds include chlorinated aliphatic acids, heterocyclic derivatives, and organometals (Mrak 1974). In Table 1, some chemical properties of herbicides, grouped in terms of their ionic and acidic nature, are summarized. Water solubility, molecular weight, and vapor pressure are presented. In addition, octanol-water partition coefficients (K_{ow}) are listed to provide a relative index of the compound's hydrophobicity. The K_{ow} is highly correlated with the ability of an herbicide to bioaccumulate, or be biologically incorporated across a membrane lipid bilayer. The uses of these compounds depend largely on their chemical characteristics. Several key aspects of herbicide use in the Chesapeake Bay region are provided in Table 2. Various information is compiled here, including the year that the herbicide was introduced for public use, the main crops (in Maryland and Virginia) with which it is used, and the associated planting and tillage practices, as well as the timing and rate of application.

The cationic and acidic herbicides are generally more water soluble than the others (Table 2). Paraquat, as a salt, is highly soluble in water, but virtually insoluble in organic solvents; 2,4-D is more generally soluble. The _s_-triazine compounds (atrazine and simazine) are among the least water soluble with moderate organic solubility; trifluralin dissolves readily in octanol, but not so readily in water. Compounds with high vapor pressure, such as dicamba, are more likely to volatilize under wet conditions and enter the hydrosphere with precipitation; the _s_-triazines, with their low vapor pressure, are less likely to follow that route of transport.

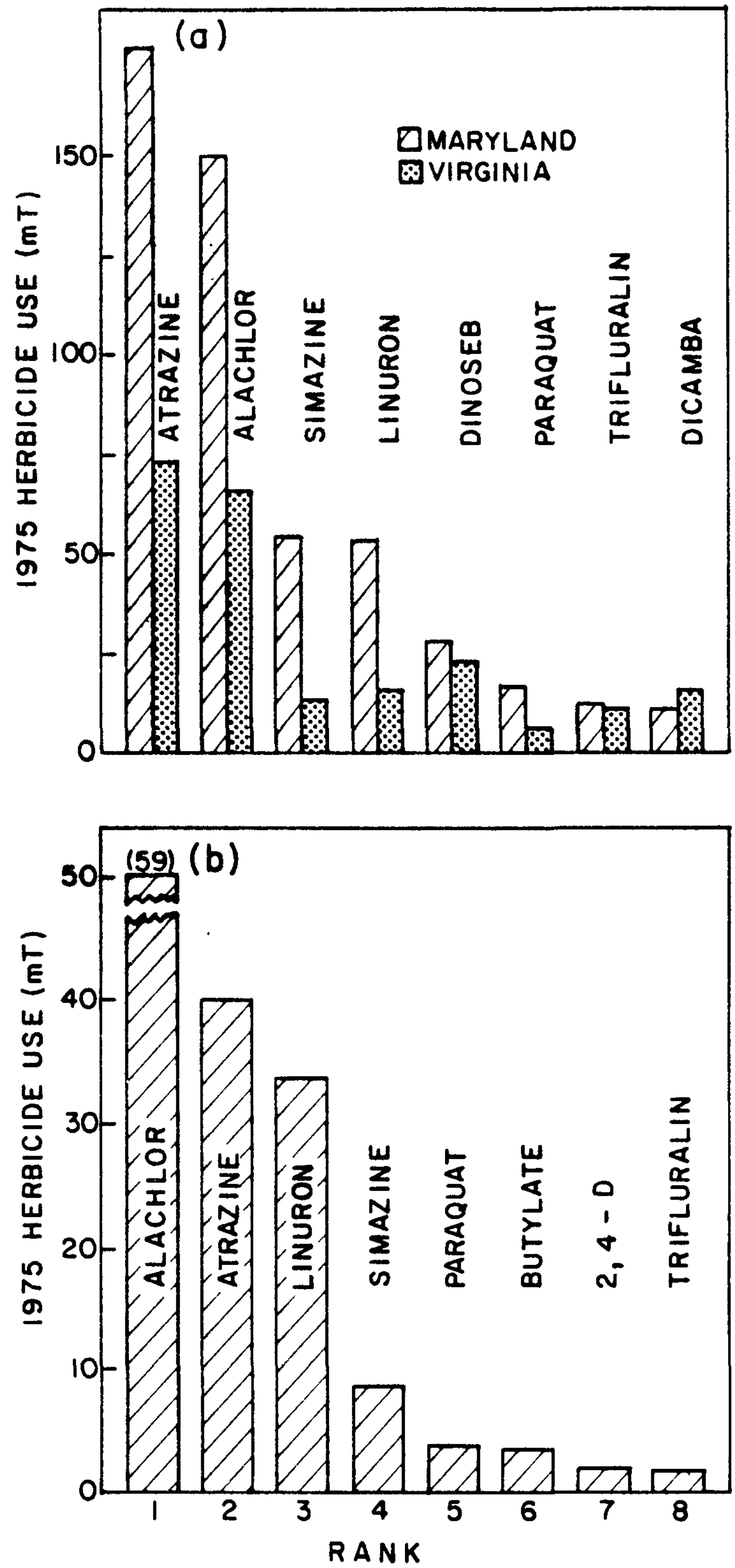


Figure 3. Estimated herbicide use in the Chesapeake Bay region for 1975: (a) Maryland and Virginia and (b) Choptank watershed (Stevenson and Confer 1978).

TABLE 1. CHEMICAL PROPERTIES OF MAJOR HERBICIDES IN CHESAPEAKE BAY REGION

CHEMICAL CLASSIFICATION

Common Name (Trade Name)	Manufacturer[b]	Chemical Name[b]	Water Solubility[c] (ppm)	Vapor Pressure[a] (mm Hg $\times 10^{-6}$)	Molecular Weight[c]	Solvent Partitioning[d] K_{ow}
IONIC HERBICIDES						
I. Cationic						
Paraquat (Orthoparaquat)	Chevron	1,1-dimethyl-4,4-bipyridinium-dichloride	Low	Low	186 (257, Salt)	1
II. Basic						
Atrazine (Aatrex)	Ciba-Geigy	2-chloro-4-(ethylamino)-6(isopropylamino)-s-triazine	33	0.3	216	226
Simazine (Princep)	Ciba-Geigy	2-chloro-4-bis)ethyl-amino)-s-triazine	5	0.006 -0.036	202	88
III. Acidic						
2,4-D	Dow	(2,4-dichlorophenoxy) acetic acid	650	0.4	221	443
Dicamba	Veliscol	3,6-dichloro-O-anisic acid	4500	3570	221	-
NON-IONIC HERBICIDES						
IV. Substituted Anilines						
Trifluralin (Treflan)	Elanco	a-a-a-trifluro-2,6-di-nitro-N,N-dipropyl-p-toluidine	0.05 -24	114	335	1150

(Continued)

TABLE 1. (Continued)

CHEMICAL CLASSIFICATION Common Name (Trade Name)	Manufacturer[b]	Chemical Name[b]	Water Solubility[c] (ppm)	Vapor Pressure[a] (mm Hg $\times 10^{-6}$)	Molecular Weight[c]	Solvent Partitioning[d] K_{ow}
V. Phenylureas						
Linuron (Lorox)	Dupont	3-(3,4-dichlorophenyl-1-methoxy-1-methylurea	75	15	249	-
VI. Substituted Anilides						
Alachlor (Lasso)	Monsanto	2-chloro-2,6-diethyl-N-(methyloxymethyl) acetanilide	148 -242	22	270	434

[a]Vapor pressure at 20-25°C (WSSA 1980, Mrak et al. 1974)

[b]WSSA (1980), Wauchope (1978)

[c]Water solubility at 23-27°C (WSSA 1980, Stevenson and Confer 1978)

[d]Octanol-water partition coefficient, K_{ow} (Rao et al. 1981)

TABLE 2. USES OF MAJOR HERBICIDES IN CHESAPEAKE BAY REGION

CHEMICAL CLASSIFICATION

Common Name	Year Introduced[a]	Associated[b,c,d] Crops	Tillage[c,d,e]	Timing of Application[c,d]	Rate of Application[b,c,f] (Kg a'i'/ha)
IONIC HERBICIDES					
I. Cationic					
Paraquat	1966	Corn, Soy	D,N	Pre-plant	0.3 - 0.6
II. Basic					
Atrazine	1959	Corn, Sorgum	C,N	Pre-plant Pre-emerge	1.1 - 2.2
Simazine	1958	Corn	C,N	Pre-plant	1.1 - 1.4
III. Acidic					
2,4-D	1958	Corn, Wheat, Soy Sorgum	C	Post-emerge	0.4
Dicamba	1962	Corn, Wheat	C	Post-emerge	0.2
NON-IONIC HERBICIDES					
IV. Substituted Anilines					
Trifluralin	1963	Soybeans	C	Pre-plant	0.8
V. Phenylureas					
Linuron	1962	Soy, Corn, Sheat	C,N,D	Pre-emerge Post-emerge	0.8 - 1.7
VI. Substituted Anilides					
Alachlor	-	Corn, Soy, Wheat	C,N,D	Pre-emerge	1.7 - 2.2

Sources: [a]Stewart et al (1975), [b]Bingham et al (1971), [c]Weaver et al (1975), [d]Stevens et al. (1981).
[e]Abbreviations: C = conventional tillage, N = no-till or minimum-till, D = double-cropping
[f]Recommended rates for Eastern Shore (Stevenson and Confer 1978)

Paraquat is highly sorbed, tending to adhere essentially irreversibly to surfaces of soil particles. Hence, it is used as a "contact" herbicide, sprayed directly on the weed foliage. It is used at low application rates before planting of the crop, particularly with no-till farming and double cropping (Table 2). The postemergent herbicides, such as 2,4-D and dicamba, must be highly specific to broad-leaf weeds, and are, thus, used primarily with corn and small grains at low application rates. The s-triazines are also effective in control of broad-leaf weeds and are very versatile, being applied at relatively high rates both pre- and post-emergence of corn, under either conventional or no-till conditions. Linuron and alachlor are also versatile compounds with a wide range of uses, although linuron is associated most closely with soybeans.

RATIONALE FOR SELECTION

At the outset of this research program in the spring of 1978, six criteria were used for selecting the two herbicidal compounds for focus during the CBP. These criteria are related to the fate-and effects-pathways described in Figure 2. Starting with total application to agricultural lands in the Bay region, we considered how long the herbicide persists on the field (that is, available for runoff to the estuary). The solubility and actual percentage of each compound transported into surrounding waterways suggest something about its relative mobility. In 1978 there was a distinct paucity of information concerning either the actual concentrations of these compounds occurring in the Bay, or their toxicity to SAV, but we used what scant data were available. Necessarily, the weighting of these factors was relatively subjective, representing our perception of importance and reliability of information. A ranking among the six most important herbicides led to the following: atrazine, alachlor, linuron, paraquat, trifluralin, and 2,4-D. It might be noted that though its current use is substantially reduced, 2,4-D was included here because it was one of the most common compounds used in the 1960's. Simazine was not considered in this ranking because of its close similarity to atrazine.

By these criteria, atrazine ranked clearly as the major compound with trifluralin falling to the bottom of the list. The relative importance among the other four herbicides was virtually indistinguishable, and each probably deserves further scrutiny in its own right. Nevertheless, we selected linuron as the other substance for CBP focus, primarily because of its relative longevity reported for agricultural soils, and because it is associated so closely with soybean production. Over recent years, corn and soybeans have become the most important crops in the region, and the two selected herbicides, atrazine and linuron, are, respectively, most significant for those two crops.

SECTION 3

DISTRIBUTION OF HERBICIDES IN THE BAY

This section summarizes observed concentrations of atrazine and linuron in Chesapeake Bay and its tributaries, and attempts to relate these concentrations to runoff rates. We will initially examine concentrations along the axis of the main Bay and then move into successively higher-order tributaries toward the source-waters that drain agricultural lands.

OPEN-BAY CONCENTRATIONS

The maximum concentrations of atrazine and linuron reported in either the open waters of the main-stem Bay or a first-order tributary, such as the Choptank and Patuxent Estuaries, between 1976 and 1980 were about 3.5 ppb (surface water). In Figures 4a and 4b, we present data for the main Bay for June and July of 1977 and 1980 (from Austin et al. 1978, Newby et al. 1978, Means et al. 1981b). Concentrations of atrazine and linuron never exceeded about 1.3 ppb, and were generally highest at lower salinities. Patterns of concentration-versus-salinity exhibited nonconservative behaviors, probably reflecting either non-steady-state input conditions or significant sources other than the Susquehanna River (Stevenson et al. 1981). General trends for the two years were quite similar.

TRIBUTARY CONCENTRATIONS

Herbicides were also monitored in two major estuarine tributaries of the Bay. Mixing diagrams of herbicide concentration-versus-salinity are also provided for 1980 data from the Choptank River (Figure 4c, 4d, 4e). The absence of a relationship in the June data was probably owed to the meager runoff that occurred in late May through June of that year, and the small runoff experienced during July generated a weak relationship for that month. Linuron concentrations were relatively high at the head of the estuary as well as at about 13 ppt salinity, suggesting runoff sources both up-river and down-estuary. Linuron concentrations in June and July were virtually undetectable, and the higher August values correspond to the July planting of double-cropped soybeans. Zahnow and Riggleman (1980) reported no detectable aqueous concentrations of linuron in the Choptank and other tributaries in 1977 to 1978, although some herbicide was found in up-river sediments. Atrazine was measured at numerous stations throughout Virginia's Bay waters, and two longitudinal profiles along the Rappahannock River are presented in Figure 5a for June and August of 1979. Highest values were 3.5 ppb in the freshwater reaches; estuarine concentrations never exceeded 1.0 ppb (Hershner et al. 1981).

Samples were obtained for analysis of estuarine sediments and suspended particulate matter at most stations in the Bay and tributary surveys. Atrazine was detected periodically in estuarine sediments at low concentrations (about 5.0 ppb) in Maryland waters (Means et al. 1981b). Similarly, sediment concentrations were rarely detectable in Virginia, although one value in excess of 30 ppb was reported for a sample from the

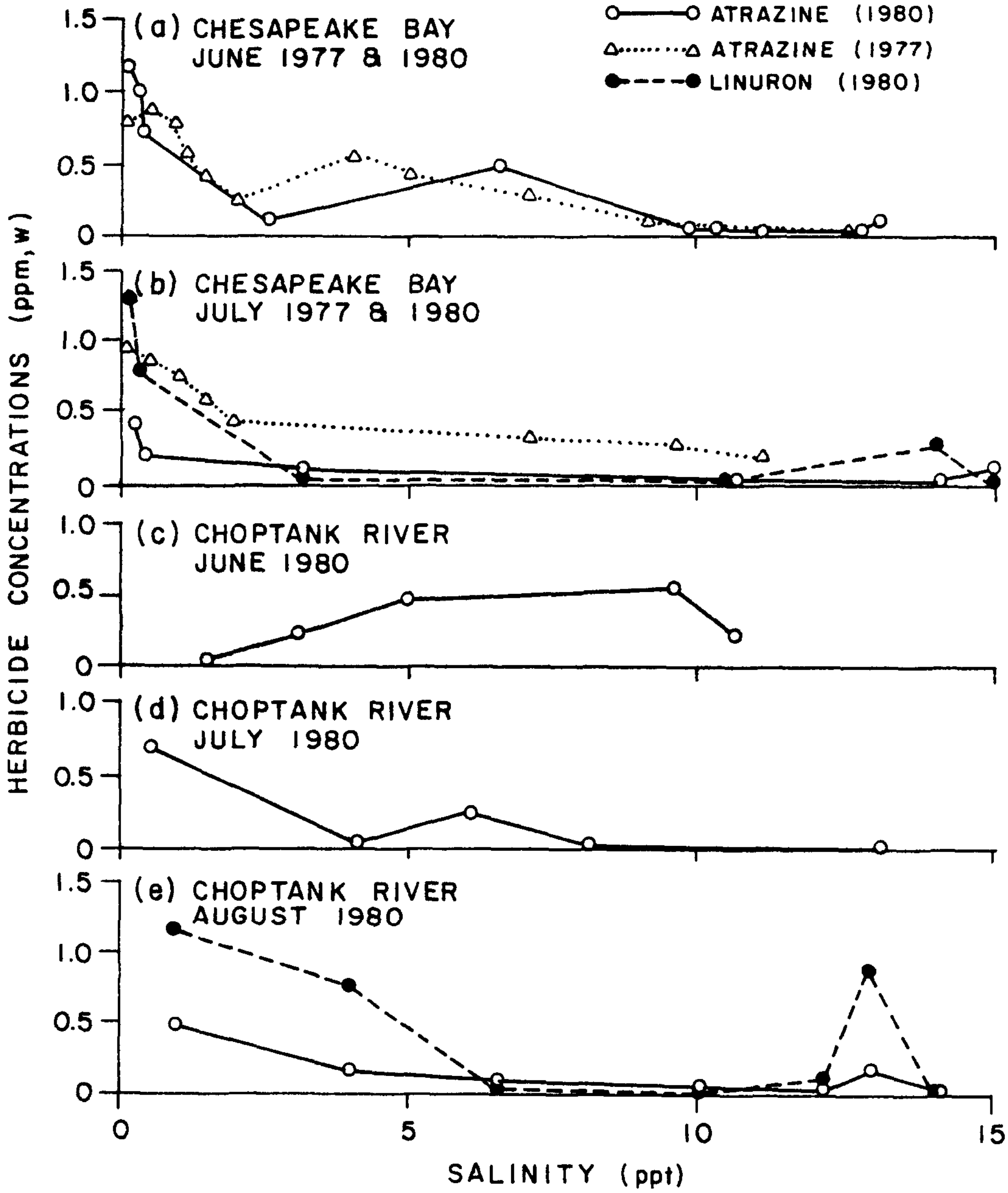


Figure 4. Concentrations of atrazine and linuron in Chesapeake Bay for June and July of 1977 and 1980 and in the Choptank River for June through August of 1980. (Data for 1980 are from Means et al. 1981b; and for 1977 from Austin 1978, and Newby et al. 1978.)

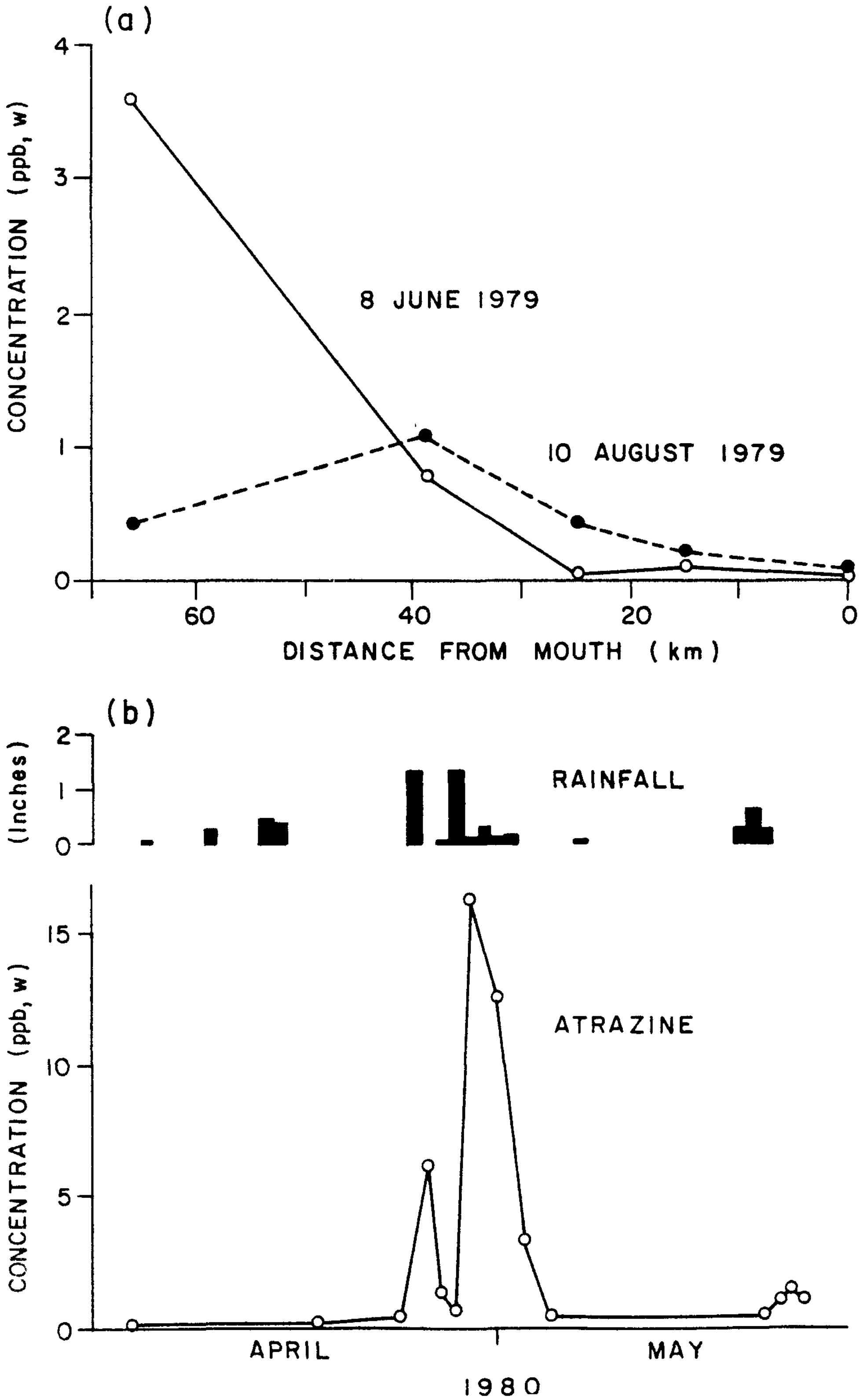


Figure 5. Concentration of atrazine (a) in the Rappahannock River, Virginia; and (b) in runoff from the Severn River, Virginia (Hershner et al. 1981). Inches of rainfall for April and May are also shown in this figure.

head of the tidal creek portion of Severn River (Figure 5b) (Hershner et al. 1981). Atrazine concentrations were never detected in suspended estuarine solids sampled in the field during 1980 (Means et al. 1981b).

It seems surprising, at first, that herbicide concentrations in the Choptank were no greater than those observed in the main Bay during 1980 surveys, in view of the far shorter transit time between field and estuary in the tributary. The spring of 1980, however, was a period of extraordinarily low runoff, particularly in the eastern shore region; total rainfall for May and June was 8.8 cm (3.5 in). Precipitation in the Choptank watershed for May and June of 1981 was 26.2 cm (10.3 in), much greater than the previous year and even slightly greater than normal, with a 20.4 cm (8 in) average for 1971-1980. Thus, 1981 represents a year when relatively high herbicide concentrations would be expected in the estuary.

RUNOFF CONCENTRATIONS

Means et al. (1981b) monitored atrazine and linuron concentrations during base flow and after all runoff events in spring and summer 1980-1981 at the creek and small embayment draining a 94 ha (232 acres) experimental watershed at Horn Point Environmental Laboratories (HPEL). These data (for 1981) are summarized in Figure 6a. Herbicide concentrations were also measured in the Choptank River headwaters and estuary after a major storm in mid-May 1981 (Figure 6b). Concentrations of atrazine in the river reached 9.0 ppb and exceeded 2.0 ppb well into the estuary. Linuron concentrations of 2.0 to 3.0 ppb were found in both fresh and brackish waters, with no apparent relation to salinity. Such high values of linuron were unexpected, since this event preceded soybean planting, and they probably represent localized runoff from treated fields of small grains. Atrazine concentrations in 1981 runoff from the HPEL watershed (draining primarily corn fields) reached peak levels of about 20, 45, 10, and 13 ppb during the four spring runoff events described in Figure 6a. Concentrations at the drainage creek during the same period in 1980 exceeded 3.0 ppb for only one short event (May 1), when peak values were 18.3 ppb. The flow in the Choptank headwaters at Beaver Dam exhibited a marked maximum (9.0 ppb) only during the first two closely spaced events in 1981. Concentrations as high as 20 ppb were observed once in the small estuarine embayment (Lakes Cove) receiving direct runoff from the HPEL watershed.

Rainfall during the spring of 1980 was considerably greater on the western shore of Virginia, where almost 10 cm (3.9 in) of rain fell during the eight-day period April 25 to May 2. Hershner et al. (1981) monitored for atrazine in the headwaters of the Severn River (draining extensive agricultural land), and reported maximum concentrations of about 16 ppb during the runoff generated by two successive downpours of 3.0 cm (1.2 in). These data are provided in Figure 5b. Somewhat higher concentrations were reported by Hershner et al. (1981) for 1979, with four values above 10 ppb measured at a small tidal creek in the upper Severn River during an April runoff event. One extreme value (108 ppb) was observed during this episode in a drainage creek. The wet spring of 1981 was not studied in Virginia, but it appears that general spatial and temporal distributions of herbicides are similar in upper and lower Bay regions, both being highly responsive to hydrologic conditions.

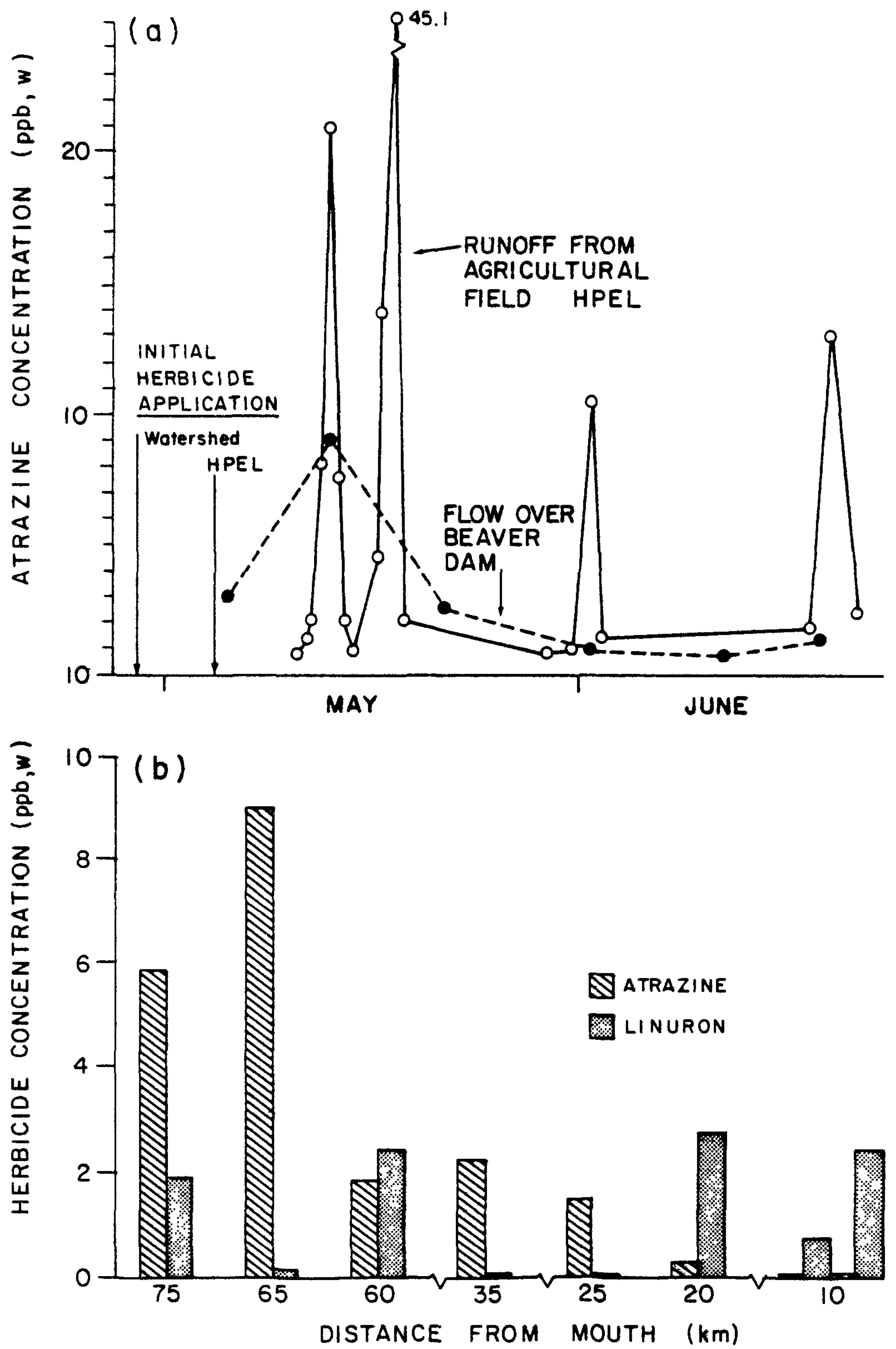


Figure 6. (a) Temporal patterns of atrazine concentrations in runoff from Choptank watershed (spring), and (b) spatial distribution of herbicides in Choptank River and estuary (May 10-13, 1981) (Means et al. 1981b).

OTHER RUNOFF STUDIES IN BAY REGIONS

Herbicide concentrations in the Rhode River on Maryland's western shore have been intensively studied. Correll et al. (1978), Wu et al. (1977), and Wu (1980) reported 1976 concentrations and runoff rates for atrazine and alachlor in the Rhode River basin and estuary. In general, their results are consistent with those of Means et al. (1981b) and Hershner et al. (1981), where peak runoff concentrations of dissolved herbicide were about 35 ppb and 3.0 ppb for atrazine and alachlor, respectively. They also reported concentrations of herbicides sorbed to suspended solids, which were periodically on par with the dissolved form but were in excess of values observed by Means et al. (1981b). Herbicides found in Rhode Estuary never exceeded 1.0 ppb (dissolved) for either compound. Atrazine has also been measured by the U.S. Geological Survey in the Susquehanna River (at Harrisburg and Conowingo) and several small tributary creeks for 1978-1980 (Ward 1980 quoted in Stevenson et al. 1981). Concentrations were generally in the range of 1.0 to 5.0 ppb, though one exceptionally high value (68 ppb) was found at Goods Run in May 1980.

Wu (1980) estimated that about one percent of the atrazine and 0.2 percent of the alachlor applied to agricultural fields in the Rhode River basin entered the watercourse. These runoff rates are within the range, but on the low side, of values reported in the literature (Wauchope 1978). Of almost 50 estimates of atrazine runoff compiled by Schueler (1979) from various North American fields, we calculate a mean of 2.6 percent from a range of 0 to 17 percent. Much less information is available for alachlor and linuron; however, reported values range from 0.02 to 14 percent and appear to be near (slightly less) atrazine values. Data from the HPEL flume have not yet been analyzed in terms of percent losses, but these forthcoming values may add some insights to this issue.

MAJOR FACTORS AFFECTING RUNOFF

Numerous factors influence the rate and concentration of herbicide runoff and should be considered when interpreting results from the Chesapeake Bay region. Among these factors are: chemical nature of the compound; slope of the land; rainfall intensity, duration, and timing; soil type; plant cover; and drainage density. Slope and precipitation are particularly important factors that can profoundly influence runoff. We have plotted overall percent loss of atrazine applied to the field, versus the topographic slope of the field, for six different sites in the Eastern and Central United States in Figure 7a. There is considerable scatter in these data, because variables other than slope are also operative. Nonetheless, there appears to be a positive relationship that follows a hyperbolic, or logistic shape, with greatest effects found in the region of five to 10 percent slope. Although comparison of data from different watersheds must be viewed with caution, one might infer that runoff data from the coastal-plain portions of the eastern and western shores of the Bay (generally less than about six percent slope) may be comparable to one another.

Another important consideration is the time interval between application of herbicide to the field and a given rainfall-runoff event. Herbicide concentrations are highest in the first runoff and generally

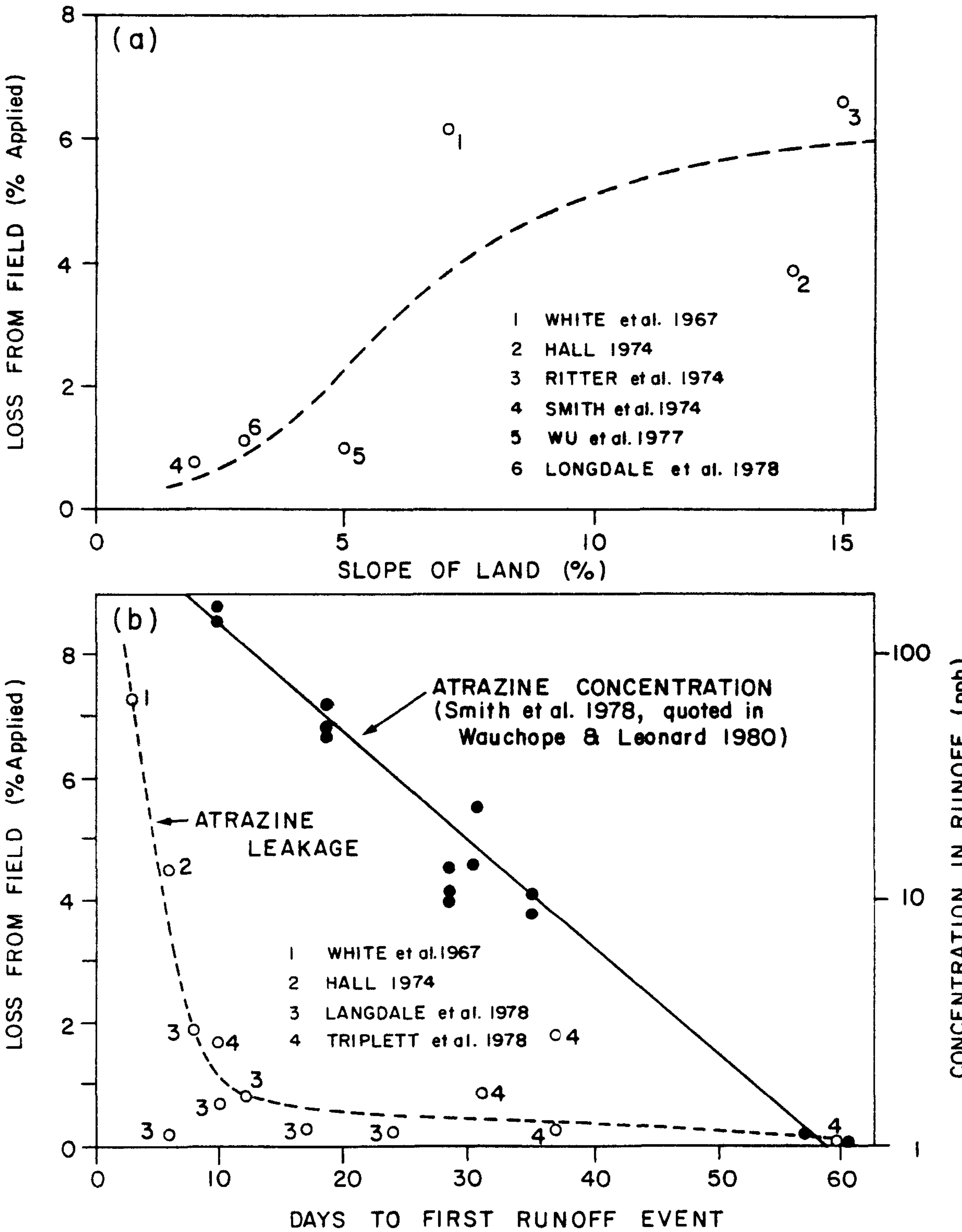


Figure 7. (a) Effect of basin slope on loss of atrazine, and (b) effects of time interval to first runoff event on atrazine loss from agricultural fields.

decrease exponentially in subsequent events, as indicated for atrazine by the solid line in Figure 7b. This effect is the result of several factors, including degradation, plant uptake, leaching, and depletion of initial mass; it emphasizes the fact that highest herbicide concentrations occur in the period shortly after field application. Wauchope and Leonard (1980) have used literature data to develop an empirical function that generalizes this relation:

$$C_t = AR(1 + 0.44t)^{-1.6} \qquad (1)$$

where C_t is the runoff concentration at event time t, R is the application rate, and A is the availability index (a function of the chemistry of the particular compound). Moreover, the total amount of herbicide transported into surrounding watercourses over the whole season is also a function of the timing of the first runoff event after application (Figure 7b, dashed line). A similar, first-order decay function generally describes this relation. The data compiled in this figure suggest that if no runoff occurs within the first 10 days after application, total atrazine loss to the watercourse will probably be less than one percent of that applied, while runoff within three days following application can lead to seven percent loss. Thus, both the concentration and total amount of herbicide entering the estuary may depend largely on the time interval between herbicide application and rainfall-runoff events.

SECTION 4

ENVIRONMENTAL BEHAVIOR OF HERBICIDES

Two key processes that determine the fate of herbicides in the environment are adsorption and degradation. As suggested in Figure 2, these processes occur in both terrestrial and aquatic (estuarine) environments. Before the initiation of CBP research in 1978, very little was known about the nature of these processes as they occur in estuaries. Hence, parallel experiments were designed to examine adsorption and degradation in simulated estuarine and soil systems, representative of typical conditions in the Bay and its watershed. Emphasis was placed on atrazine, although experiments were also performed with linuron and other compounds. Atrazine degradation appears to proceed more rapidly in systems with sediments and/or soil, than with water alone, and Jones et al. (1981b) have postulated that most of the degradation may be preceeded by sorption, followed by desorption. Hence, the two processes are intimately coupled.

SORPTION REACTIONS

The adsorption of dissolved herbicides to solid surfaces proceeds as a function of aqueous concentration (C) until an equilibrium is achieved between C and the adsorbed concentration [x/m, where x is the weight of herbicide adsorbed to solid (micrograms), and m is the weight of solids (grams)]. The equilibrium relation is often described using the Freundlich equation,

$$\frac{x}{m} = (K_d)(C)^{1/n} \qquad (2)$$

where n is a constant describing the shape of the equilibrium relation, and K_d is the sorption coefficient (for example: Giles et al. 1960, Bailey and White 1970, Kempson-Jones and Hance 1979, Travis and Etnier 1981). High K_d values indicate a stronger tendency for adsorption. Several studies suggest that organic matter in the substrate tends to be the controlling factor for adsorption of many non-polar organic compounds such as herbicides (Bailey and White 1964; Karickhoff et al. 1979; Means et al. 1979, 1980). Therefore, it is convenient to normalize K_d values to the organic matter of the sorbant,

$$K_{oc} = K_d/(\text{decimal fraction organic carbon}) \qquad (3)$$

Values of K_{oc} for atrazine and linuron have been reported for a wide variety of soils, ranging from 47 to 394 (atrazine) and 124 to 2678 (linuron), with typical values being 170 and 670, respectively (Rao et al. 1981).

Sorption isotherms for atrazine with agricultural soils, estuarine sediments, and estuarine colloids, and for linuron with estuarine sediments and colloids, were determined from the Bay region (Means et al. 1981a; Means and Wijayaratne 1981). The atrazine data are summarized in Figure 8a. All isotherms were linear over the range of concentrations tested (n =

1.0). The K_{oc} were about 10 times greater for colloids (2,000-14,000) than those for sediments (200-400), which were in turn generally greater than those obtained for soils (100-200). Values of K_{oc} for linuron with sediment and colloidal matter from the Patuxent River were 3.4 times greater than for atrazine with the same substrates. This relationship is almost identical to the relationship between the two compounds for soils, where linuron K_{oc}s were 3.9 times greater. These values of K_{oc} for colloids appear to be consistent with the findings of Wu et al. (1980), who reported that estuarine surface microlayers at Rhode River were typically enriched with atrazine by a factor of about 10 to 30 over bulk water concentrations. If it is assumed that this enrichment is due to sorption by the hydrophobic colloidal matter concentrated at this air-water boundary with associated organic carbon of 5.0 ppm, then the K_{oc} values would be about 2,000 for the samples of Wu et al. (1980).

Correll and Wu (1981) have reported K_d values ranging from 5.0 to 260 (depending on C) for atrazine dissolved in distilled water and adsorbed to Rhode River estuarine sediment. These values are two to 100 times greater than those of Means et al. (1981a) and other investigators. The highest K_d's of Correll and Wu would correspond to K_{oc}s of Means et al., only if the Rhode River sediment were 50 percent organic carbon. They do report extremely high organic carbon percentages; however, even these are too low (five to 27 percent) to explain the differences (Correll et al. 1978). Moreover, their data imply that Freundlich isotherms would be non-linear in the same general concentration range as that given by Means et al. (1980) and others and, using their data, we calculate n = 2.3 with K_d = 126 (Equation 2). It is difficult at this point to resolve these discrepancies.

The adsorption of atrazine and linuron has been extensively studied on a wide variety of soils (Talbert and Fletchall 1965, McGlamery and Slife 1966, Green and Obien 1969, Harris and Warren 1967, Weber et al. 1969, Bailey and White 1970, Grover and Hance 1970, Hurle and Freed 1972, Colbert et al. 1975, Hiltbold and Buchanan 1977, Dao and Lavy 1978). A number of factors have been identified that influence the adsorption of these compounds to soils, including pH, temperature, moisture, electrolytes, and organic matter. Of these factors pH and salinity were examined to determine their potential effects on herbicide adsorption under estuarine conditions. Salinity exerted a small (five percent) negative effect on adsorption with sediments between 5.0 and 15 ppt, but the overall effect from 0 to 15 ppt was erratic and probably nonsignificant (Figure 8b). For colloids, on the other hand, salinity between nine and 19 ppt appeared to have a substantial negative effect (29 percent). This pattern was consistent between experiments (where salinity was manipulated) and field observations (where salinity varied along the estuarine axis), but was opposite to that which would be predicted as a result of "salting-out" of the hydrophobic herbicide. This finding suggests that salinity affects more the nature of the colloidal material than the solubility of the herbicide (Means and Wijayaratne 1981). It was found that pH also influenced K_{oc} for colloids with atrazine and linuron. Both herbicides exhibited maximum K_{oc} at a pH approximating that of the estuarine environment from which water and colloids were taken. Increasing or decreasing pH by one unit (that is, between pH 7.0 and 9.0) caused a 2.0 to 20 percent decrease in K_{oc} for the two herbicides, although at pH 5.0 to 6.0 K_{oc} dropped by 25 to 35 percent.

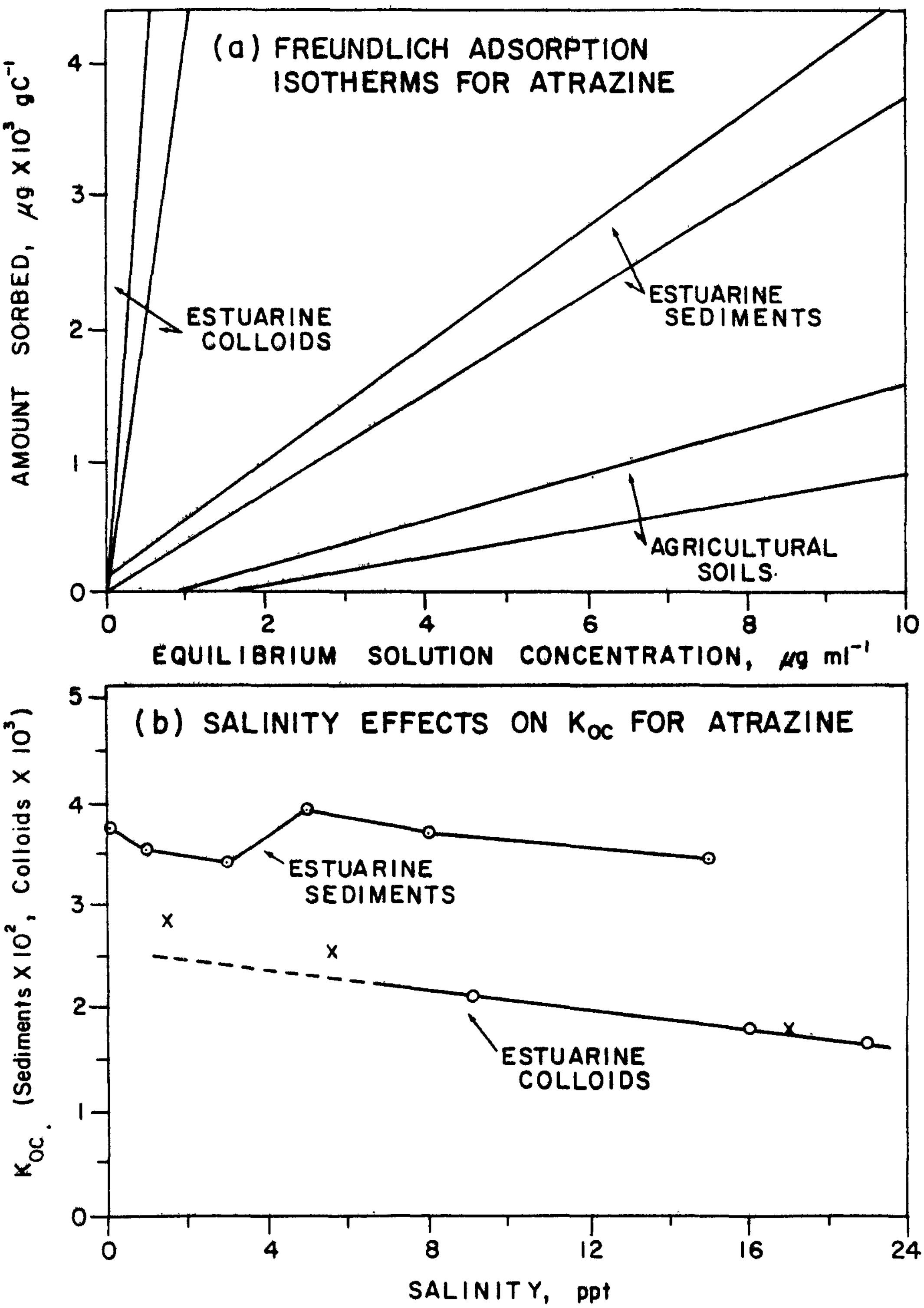


Figure 8. (a) Freundlich adsorption isotherms for atrazine and (b) effects of salinity on K_{oc} for atrazine, from laboratory experiments (o) and field observations (x) normalized to laboratory conditions. (Means et al. 1980, Means and Wijayaratne 1981.)

The implications of these sorption data are that atrazine and linuron are readily susceptible to runoff and leaching from agricultural fields, even without particulate soil erosion. Control of soil erosion alone will not control atrazine transport to the estuary. Means et al. (1981) suggest that once in the estuary, dissolved atrazine will adsorb readily to suspended sediments and colloidal material. Sediment-sorbed atrazine will move only through resuspension and sediment transport; colloidal-bound herbicide may travel great distances and concentrate in organic film at the air-water interface, typical of coastal waters, particularly during the fall. The expected fate of linuron would be analogous, except less would leave the field in dissolved form, but more would adsorb to estuarine particles.

HERBICIDE DEGRADATION

An important factor contributing to the potential toxicity of herbicides, such as atrazine and linuron, is the longevity of these substances in the field or estuary. Numerous studies have described the kinetics of atrazine degradation in various soil environments, and a wide range of physical factors (such as pH, temperature, moisture, clay, and organic content of soils) has been shown to affect this process (for example, Swanson and Dutt 1973; Best and Weber 1974; Hiltbold and Buchanan 1977; Hance 1979; Kempson-Jones and Hance 1979; Kells et al. 1980). Direct experiments were conducted for atrazine degradation in flasks with estuarine water and sediments maintained in natural light under field temperatures (Jones et al. 1981b). We also monitored atrazine and linuron concentrations in laboratory microcosms [25 L (6.6 gal) and 700 L (184.9 gal)] over eight weeks and thus, indirect estimates of degradation were obtained (Cunningham et al. 1981a, 1981b).

Much of the information on herbicide longevity developed by agronomists and soil scientists refers to persistence in agricultural soils (that is, the time required for 90 percent or more of the compound to disappear from the site of application). Reported values of "field persistence" result both from degradation and mobility of the compound. Nonetheless, to the extent that mobility may not vary excessively among the compounds, persistence provides a rough estimate of relative degradation rates in the field. A summary of persistence data (defined as above) for nine herbicides important in the Chesapeake Bay region is presented in Figure 9 (Stewart et al. 1975). Except for paraquat (the persistence of which is largely related to its highly sorptive nature), the s-triazines, atrazine and simazine, are the most persistent of these compounds. Thus, it appears that atrazine is among the more persistent herbicides in common use, and understanding its degradation should provide a conservative perspective on herbicides in general.

Herbicide degradation is often described as a first-order decay process

$$C_t/C_o = e^{-kt} \qquad (4)$$

where c_t is the amount of the compound remaining at time t; C_o is the initial amount and; k is a decay rate coefficient. Others have suggested that higher order processes better describe the herbicide degradation (for

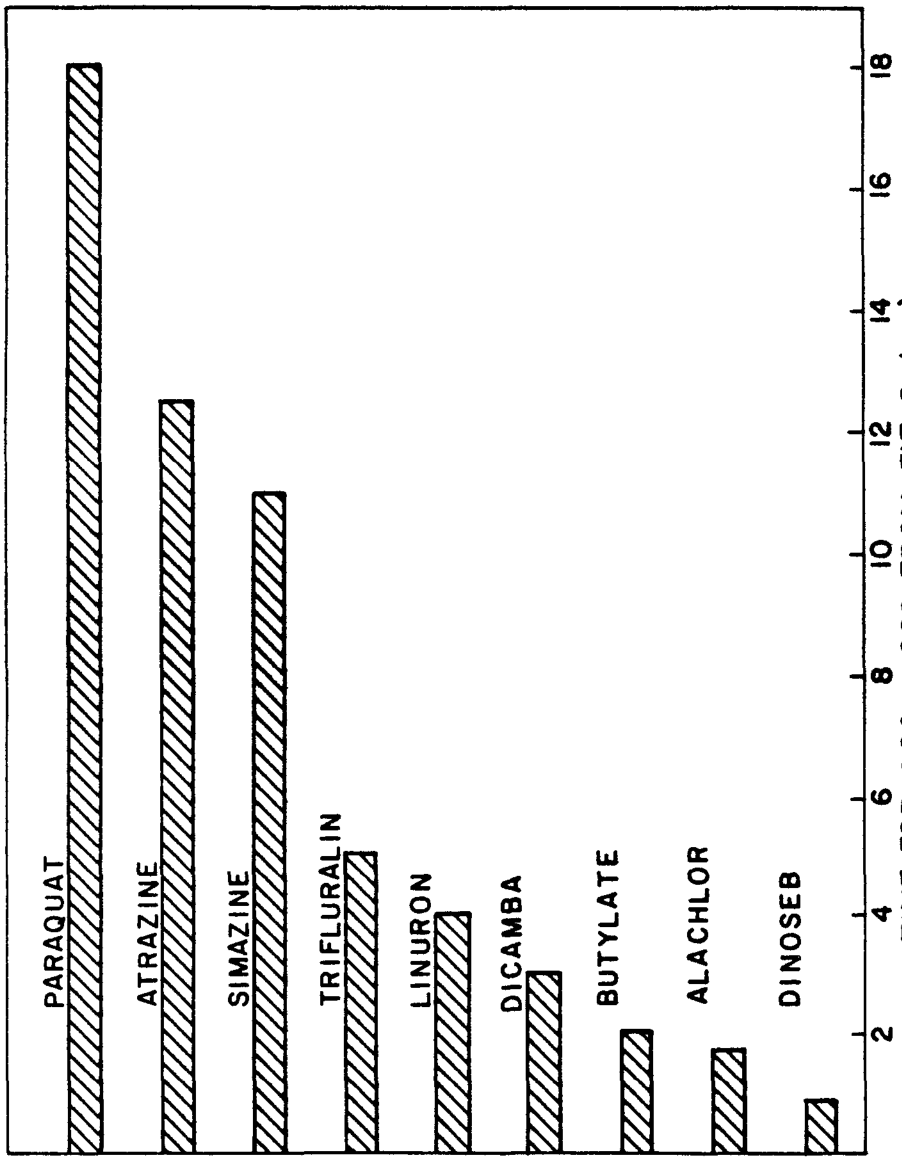


Figure 9. Approximate persistence of nine herbicides in soil (Stewart et al. 1975).

example, Hamaker 1972; Kempson-Jones and Hance 1979) so that the more general relation applies

$$C_t = [C_o^{(1-n)} + (n-1)kt]^{(1/(1-n)} \quad (5)$$

where n is the apparent order of reaction. The overall rate of reaction is often described by the half-life ($T_{1/2}$), or time required for disappearance of 50 percent of the original substance. This $T_{1/2}$ is equal to (0.693/k) for first-order reactions.

Atrazine can degrade through chemical and biological processes into metabolites, some of which may be toxic. The degradation of atrazine to its metabolites can occur through: chemical hydrolysis to hydroxyatrazine; dealkylation to either the de-ethylated, deisopropylated, or deaminated atrazine forms followed by hydrolysis; or conjugation. The dealkylations and ring cleavage are generally considered to be biologically (enzymatically) mediated, but the hydrolysis to hydroxyatrazine is controlled by physical parameters, most notably pH (Armstrong et al. 1967). Ring cleavage of atrazine is a very slow process, typically causing losses of only a few percentage points over several years; however, prior hydrolysis to hydroxyatrazine does increase the rate of ring cleavage (Armstrong et al. 1967). The biological degradation is performed by soil fungi and bacteria, with the organisms using mainly the side-chains as carbon sources, nitrogen sources, or both (Kaufman and Blake 1970).

The degradation of ^{14}C ring-labeled atrazine in two estuarine water-sediment microcosms (from Choptank and Tangier) and two soil systems (well-drained Sassafras and poorly drained Mattapex) was compared over an 80-day period under high- and low-oxygen tensions (Jones et al. 1981b). In the estuarine systems, total residues moved from water to sediments over the course of the experiment, and the relative percentage of parent and daughter compounds changed rapidly during the first several weeks (Figure 10). The initial degradation products generated in the estuarine systems, as revealed by thin-layer chromatography and autoradiography, appeared to be the same as for the soil systems, with hydroxyatrazine being the major short-term metabolite. By the 21st day of the experiment, the percentage of total extracted residues corresponding to atrazine, monodealkyalted atrazine, and hydroxyatrazine were 65, 10, and 25 for the Choptank, and 15, eight, and 77 for the Tangier (the estuarine systems); 66, 5.0, and 29 for the Sassafras, and 93, 2.0, and 5.0 for the Mattapex (the soil systems).

Atrazine degradation was far more rapid in the estuarine systems than in the soils. Half-lives for the herbicide ranged from three to nine days in overlying estuarine water, and 15 to 20 days for estuarine sediments, as compared with 330-385 days for agricultural soils. A portion of the residues adsorbed to sediments and soils was nonextractable, and these half-life estimates employed the conservative assumption that extractable and nonextractable residues were similarly distributed among the three major metabolites. Oxygen tension appeared to have negligible effect on atrazine degradation; however, the low oxygen systems were not completely anoxic. The relative degradation rates of atrazine in the three environments, as observed in this experiment and other studies from the literature, are summarized in Table 3, where mean half-lives are 14, 45, and 180 days, respectively, for estuarine water, aquatic sediments, and agricultural soils. Thus, atrazine is less likely to be a problem in the

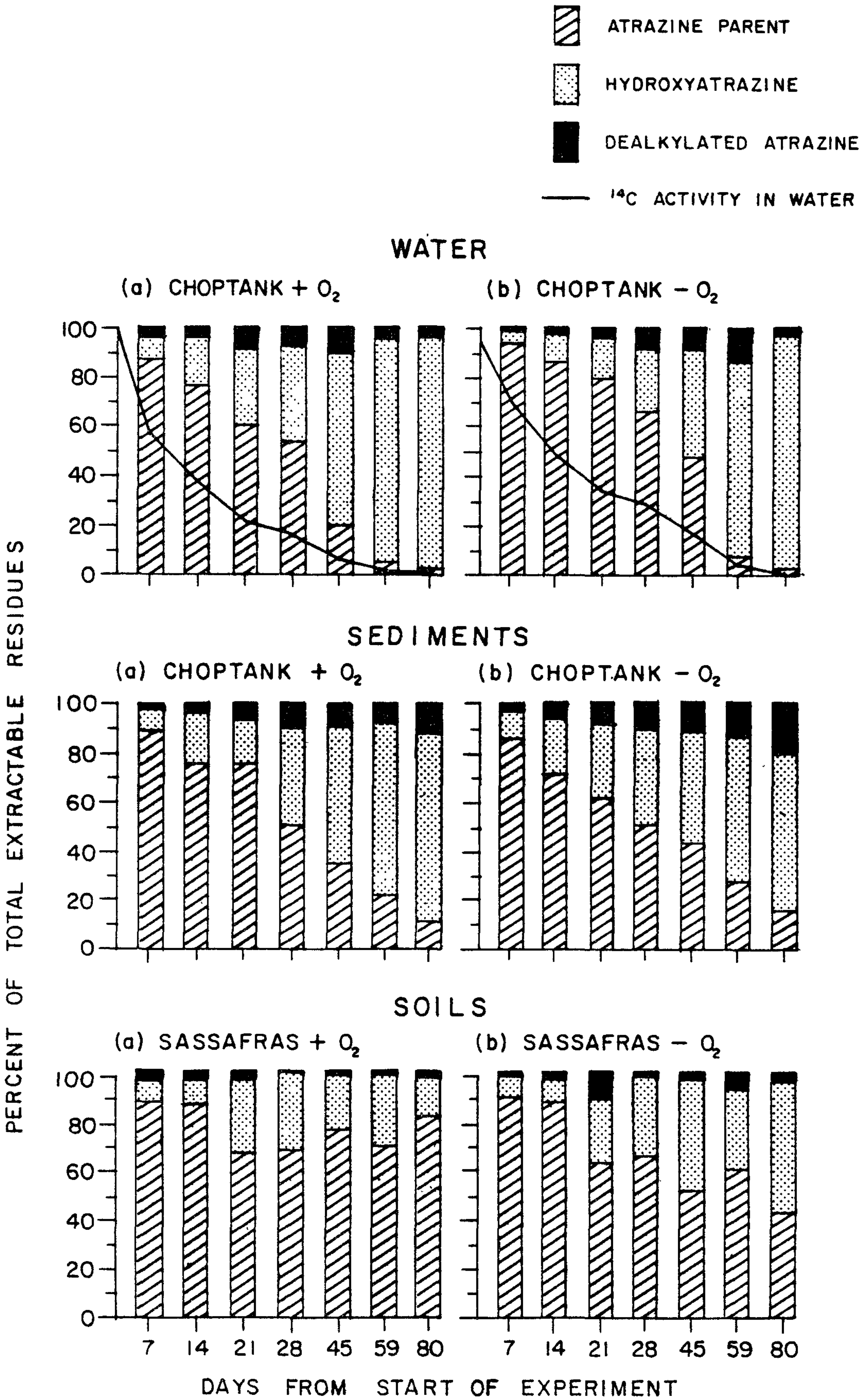


Figure 10. Loss of ^{14}C-ring labeled atrazine from experimental systems and percent of total residuals as parent compound and as two major metabolites (adapted from Jones et al. 1981).

estuary, because it rapidly degrades; however, because the herbicide's half-life on farmland soils is longer than in the estuary, it remains potentially available for runoff long after application.

TABLE 3. SUMMARY OF ATRAZINE DEGRADATION RATES IN AGRICULTURAL AND ESTUARINE ENVIRONMENTS[a]

Environment	Half-life (days)[b]	
	Mean	Range
Agricultural Soils	180	10-1200
Aquatic Sediments	45	6-145
Estuarine Water	14	3-30

[a]Summarized after Jones et al. (1981).

[b]Time required for 50 percent degradation of original compound.

Herbicide data from our estuarine microcosm phytotoxicity experiments (Cunningham et al. 1981a, 1981b) indicated that overall disappearance of atrazine from the water-sediment environment occurred somewhat more slowly than in the estuarine flask systems of Jones et al. (1981b), but still more rapidly than for soils (Figure 11). Half-lives of atrazine were on the order of 60-80 days in the microcosms. The slower decomposition in these systems may be a function of the fact that the experiments were performed in artificial lights, which would contribute less to photodecomposition (Jordan et al. 1964), or perhaps higher pH and/or reduced organic substrate. A value of $t_{1/2}$ from similar microcosm data of Correll and Wu (1981) was calculated to be about 30 days, which is closer to the combined sediment-water $T_{1/2}$ of 10 to 20 days from Jones et al. (1981b). Microcosm experiments involving linuron indicated much faster degradation of this herbicide, with $T_{1/2}$ = 10 days (Cunningham et al. 1981b), a result consistent with the relative persistence of the two compounds in soil (Figure 9).

It appears that atrazine is a relatively good agricultural weed-control compound, in that it persists in soils (where it can perform its designed function) 10 times longer than in the estuary (where nontarget species might be exposed to its toxic effects). Apparently, atrazine is one of the more persistent herbicides in use, and its half-life in the estuary is at least six times greater than linuron. This 6:1 relationship is similar to the 3.5:1 relationship reported for the persistence (time for 90 percent disappearance from field application) of these two compounds in the field. Therefore, the field persistence data (such as Figure 9) may, in some cases, provide a crude index of potential estuarine longevity.

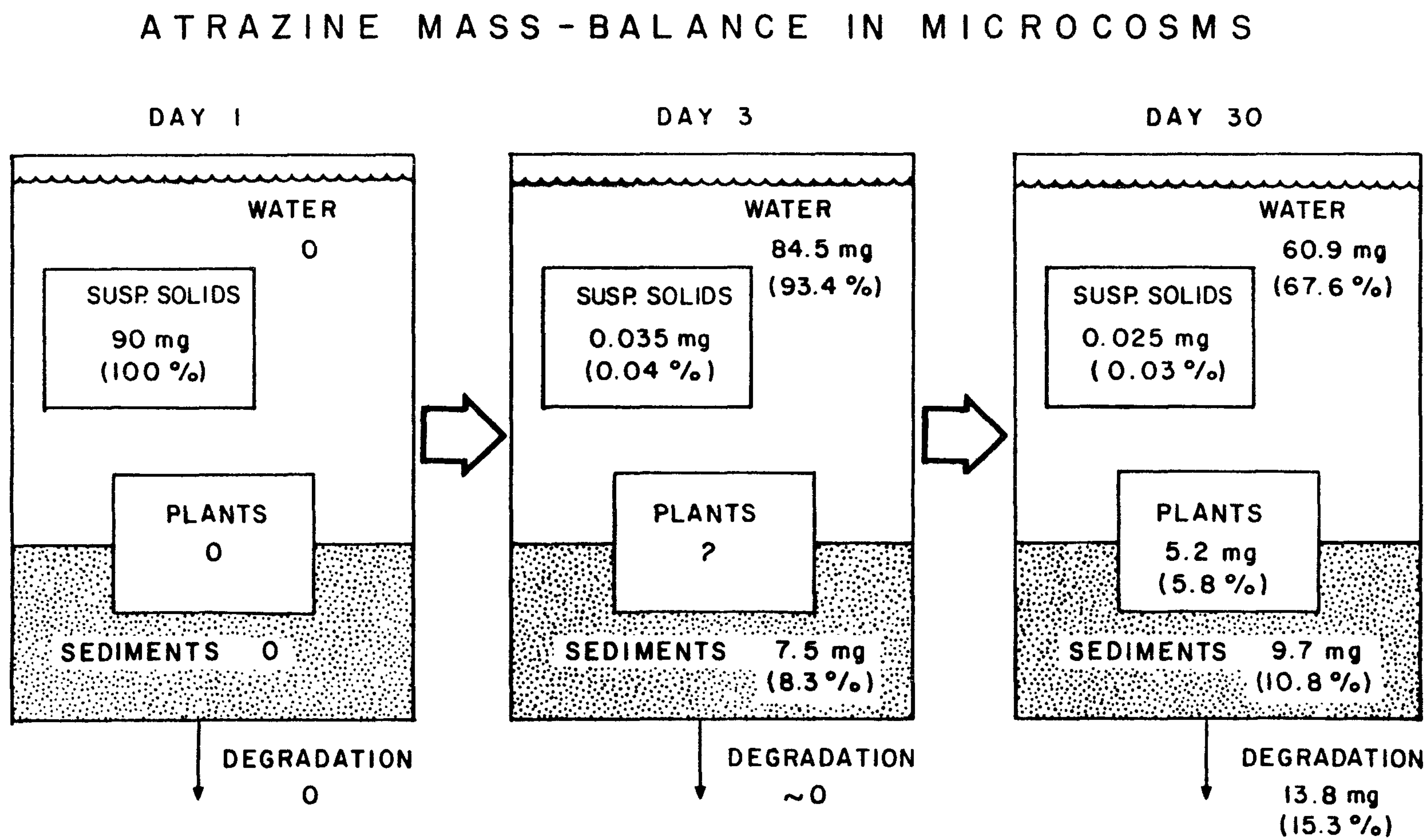


Figure 11. Atrazine mass-balance after one, three, and 30 days in experimental microcosms containing estuarine water and sediments along with SAV (Cunningham et al. 1981a).

SECTION 5

TOXICITY OF HERBICIDES IN THE ESTUARY

In this section the toxicity of herbicides used in the Chesapeake Bay region is considered. The major concern is herbicide phytotoxicity to SAV, particularly in Chesapeake Bay. We also review other aspects of herbicide toxicity for animals, algae, and emergent aquatic plants, as well as mutagenic action of these compounds. We begin with a general review of the known mechanisms of toxicity, again emphasizing atrazine and linuron.

TOXIC MECHANISMS

Herbicides can kill plants by interfering with photosynthesis, respiration, and other aspects of plant metabolism. The major herbicides in use today can be categorized as to their site of action. Four sites are recognized: the chloroplast, the mitochondria, protein synthesis, and membrane permeability. Of these, the chloroplast-related group of herbicides are in the widest use in the Chesapeake Bay watershed. Two herbicides in this group are atrazine and linuron. Both of these compounds appear to inhibit the Hill reaction of photosynthesis at the same location within the chloroplast, stopping electron transport leading to the production of the reduced cofactor ($NADPH_2$) for the fixation of CO_2. There is disagreement among investigators as to the exact location of this attack (Ebert and Dumford 1976), but most concede that it is between the initial electron acceptor Q in photosystem II and plastoquinone (Gysin and Knusli 1960, Moreland and Hilton 1976). More specifically, recent information indicates that both atrazine and linuron compete for the same protein-binding site on the thylakoid membrane, possibly causing a conformation change, blocking electron transport (Brewer et al. 1979).

The fact that the site of inhibition is within the chloroplast itself can dictate the relative toxicity of a compound or its organic solubility. The lipoid-rich membrane environment of the chloroplast makes penetration of more polar compounds difficult, thus reducing their access to the binding site. In the case of atrazine, the daughter products show a decreasing phytotoxicity in the order of de-ethylated atrazine > de-isopropylated atrazine > hydroxyatrazine. This toxicity inversely correlates with their relative polarities (Lamoureux et al. 1970) and the order relates to how soluble herbicides are. Therefore, solubility data can provide some insight into relative toxicity.

Resistance to photosynthetic inhibitors in plants is manifested either in the ability to degrade the parent compound to nontoxic metabolite(s), to complex the compound through conjugation, or to acquire altered binding sites on the chloroplast membrane through genetic selection. Degradation of the parent compound may be enzymatically or nonenzymatically controlled. Corn contains the compound benzoxazinone that nonenzymatically hydrolyses atrazine to hydroxyatrazine. Corn also contains enzymes that degrade atrazine to its dealkylated products. Sorghum can conjugate atrazine by a glutathione s-transferase enzyme, which removes the chlorine from the molecule, allowing a bond to form between the triazine ring and the sulfur of glutathione (Shimabukuro 1968). Pfister et al. (1979)

suggest that plant species may develop resistance to herbicides by the evolutionary selection of altered binding sites on the chloroplast membrane. Theoretically, SAV might develop resistance to herbicides through similar genetic mechanisms which provide a means of increasing degradation of the herbicide within the plant cells, or tying it up.

TOXICITY TO ANIMALS

Since the toxic mechanisms for many of the important herbicides act directly on the chloroplast, it would be anticipated that effects of these compounds on heterotrophic organisms would be substantially less (Stevenson et al. 1981). Toxicity data for various estuarine animals tend to support this hypothesis. For example, the fiddler crab, Uca pugnax, was observed to withstand concentrations up to 100 ppm atrazine with no demonstrable effects in bioassays (Davis et al. 1979). Only at 1,000 ppm was the escape-response ability of fiddler crabs damaged, so that normal activities in the saltmarsh ecosystem were impaired. Even when fed cordgrass (Spartina alterniflora) containing atrazine, box crabs showed little behavioral response (Pillai et al. 1979). Similarly high levels of atrazine resistance have been reported for mud crabs, Neopanope texana (Newby et al. 1978). Shrimp and oysters have been shown to be somewhat more sensitive to atrazine in bioassay experiments, with shrimp exhibiting 30 percent mortality in 96 hours at 1.0 ppm atrazine, and oysters showing no effects at this concentration (Butler 1965).

MUTAGENCITY

An increasing concern in recent years has been the discovery of the mutagencity of pesticides and/or pesticide metabolites. The issue has been complicated further by the instances in which a nonmutagenic parent compound can be activated by either plant or animal metabolism into a mutagenic substance. Herbicides, like most pesticidal compounds, often contain chlorine or bromine substituents, aromatic rings, and amine groups. These functional moieties have often been associated with mutagenic activity in organic molecules. It should be noted that, in some cases, mutagenicity has been associated with a trace byproduct from commercial production of the pesticide (that is, dioxins in 2,4-D and nitrosamines in trifluralin). A summary of available information on the mutagencity of the major herbicides used in the Chesapeake Bay region is presented in Table 4. The issue of genetic toxicity of the herbicides was not addressed in any of the research funded by the CBP-SAV program. It remains, however, an important question that needs to be considered in the future.

TABLE 4. MUTAGENICITY OF MAJOR HERBICIDES IN CHESAPEAKE BAY REGION[a]

Compound	Relative Mutagenicity	Comments
Sodium Azide	++	plant activated
Atrazine	+	plant activated
Simazine	+	
Cyanazine	+	plant activated
Diquat	NT	
Paraquat	NT	
2,4-D	-	dioxin contaminant, by-product +++
Dicamba	-	
Trifluralin	-	nitrosoamine, by-product ++
Linuron	-	
Alachlor	+	plant activated
Propachlor	+	plant activated, synergistic enhancement with triazines

[a]Source .

[b]Symbols: +++ extremely large effect; ++ large effect; + some effect; - no effect; NT not tested

PHYTOTOXICITY FOR ALGAE AND EMERGENT PLANTS

Phytoplankton vary widely in their susceptibilities to atrazine, with toxic concentrations ranging from 20 to 1,000 ppb (Stevenson et al. 1981). Davis et al. (1979) reported that 100 ppb atrazine caused some effects on mixed algal assemblages from coastal waters. For the diatoms, Thallassiosira and Nitzschia, the LD_{50} was 1,000 ppb. Reductions in cell density of 10, 90, and 100 percent were obtained at concentrations of 20, 200, and 500 ppb, respectively, for the chlorophyte, Chlamydamonas spp. (Loeppky and Tweedy 1979, Hess 1980). Pruss and Higgins (1974) reported no lasting effects on algal populations in a lake treated with 100 ppb simazine. Chlorella pyrenoidosa possesses a high degree of resistance to atrazine, where 1,000 ppb were required for 50 percent reduction in chlorophyll-a (Kratky and Warren 1971). Metz et al. (1979) found similar resistance for Chlorella strains, which was attributed to the ability of this species to exist heterotrophically. Bryfogle and McDiffett (1979) showed that productivity in algal cultures treated with 400 ppb simazine was actually enhanced, although species diversity was reduced with resistant Chlorella strains dominating the experimental systems.

Atrazine effects on Spartina alterniflora have been studied extensively by Davis et al. (1979). Cordgrass biomass was unaffected at 10 ppb; at 100 ppb, a 34 percent reduction in biomass was observed; at 1,000 ppb, approximately 46 percent of biomass was lost. Apparently, Spartina has some detoxification capability for atrazine, by a mechanism similar to that of sorghum (Pillai et al. 1977). Thus, laboratory studies suggest that neither algal nor marsh grass populations would be seriously damaged at atrazine concentrations in the range observed in Chesapeake Bay; however, herbicide treatment could conceivably contribute to phytoplankton species shifts that allow monospecific bloom conditions.

SAV PHYTOTOXICITY

The crucial relationship in this discussion is the potential phytotoxic effect of herbicides on SAV. Two herbicides, atrazine and linuron, were tested against SAV species Potamogeton perfoliatus (a dominant native) and Myriophyllum spicatum (an exotic that was extremely abundant just before initial SAV decline in 1964). Though historically important in Chesapeake Bay, both of these species are of freshwater origin. The effects of atrazine on the marine and estuarine seagrasses, Zostera marina and Ruppia maritima, have also been tested (Hershner et al. 1981). Experimental exposures ranged from incubations of six to 24 hours both in situ and in vitro, to five weeks in laboratory microcosms of three sizes and designs. Similar microcosm studies were performed by Correll and his colleagues, using atrazine with three additional Bay species: Potamogeton pectinatus, Zannichellia palustris, and Vallisneria americana. Thus, a broad data base now exists on this topic, with seven species, two herbicides, and six experimental designs.

Effects on Photosynthesis and Respiration

The general response of P. perfoliatus to atrazine treatment in laboratory microcosms is shown in Figure 12 (Cunningham et al. 1981b), where mean values for apparent photosynthesis, P_a (net O_2 production during the day), are given for microcosms under control, and under six herbicide dosages over a nine-week experimental period. The shaded portion of each graph represents the departure of actual metabolic rates from expected values (based on both pretreatment and control data). Similar data have also been reported for atrazine effects on M. spicatum, and for linuron toxicity to both SAV species (Cunningham et al. 1981b). In addition, P. perfoliatus response to low concentrations (1.0 to 25 ppb) of the atrazine was retested.

In general, the response patterns of SAV to herbicides were similar to that shown in Figure 12, where marked decreases in photosynthesis were observed at concentrations greater than 50 ppb, with some less pronounced effects at lower concentrations. Myriophyllum, however, exhibited slightly greater resistance to atrazine, but virtually identical response to linuron as compared with P. perfoliatus. At 5.0 ppb atrazine, P_a for M. spicatum was actually enhanced over controls. Recent short-term experiments indicated that the P_a response of R. maritima to atrazine was similar to that of P. perfoliatus (Jones, unpublished data). Simple two-way analysis

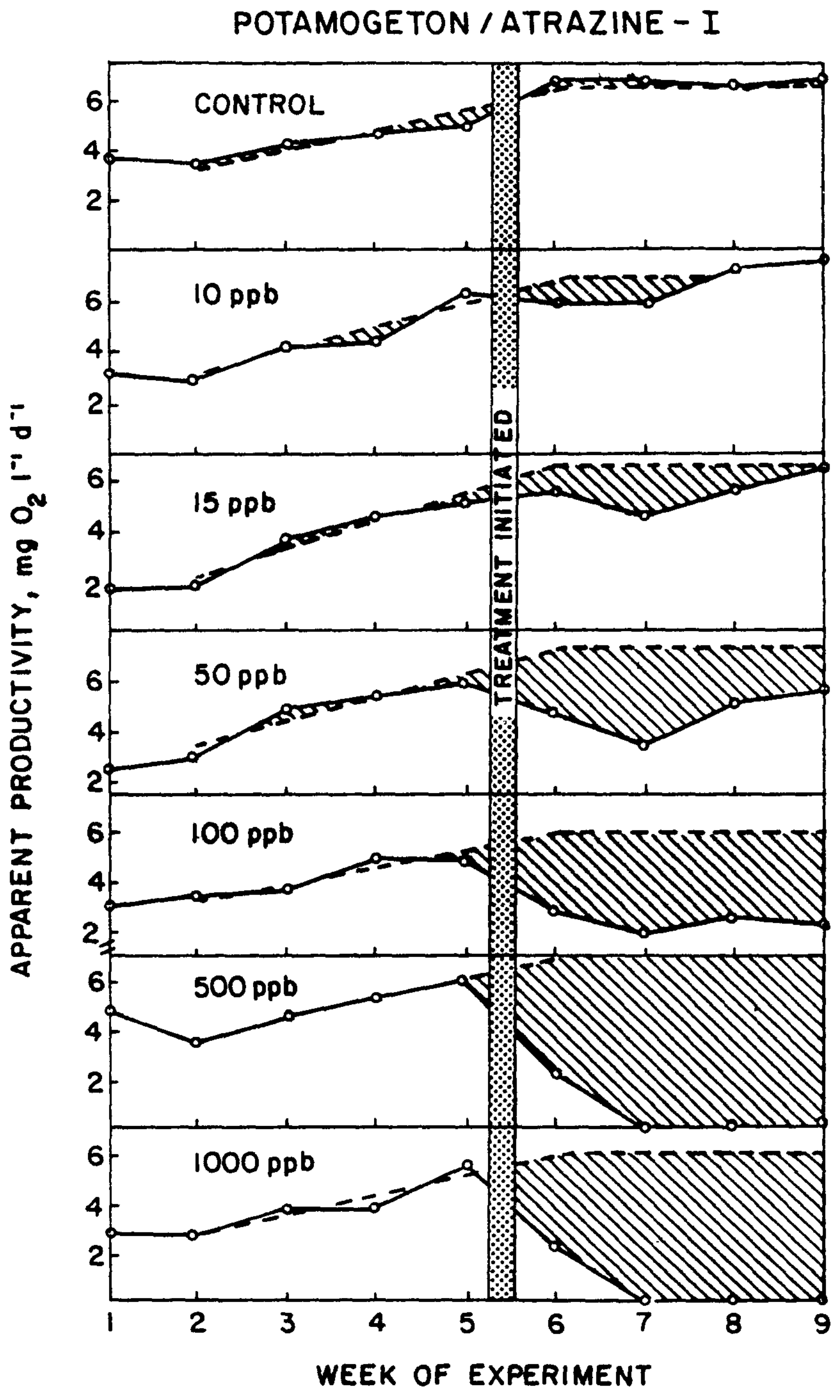


Figure 12. Summary of measurements of apparent photosynthesis for microcosms containing Potamogeton perfoliatus treated with atrazine (0 to 1 ppm). Data points are means of duplicate measurements on duplicate systems (n = 4) (Cunningham et al. 1981b).

of variance suggests that effects were always significant ($p \leq 0.05$) for concentrations greater than 50 ppb, and sometimes significant at 5.0 ppb; however, further statistical analysis is still in progress.

At all concentrations less than 50 ppb, *Potamogeton* P_a exhibited a trend of recovery toward control levels after the second post-treatment week. The same pattern was found for *M. spicatum* with atrazine, and for both species with linuron. For both species, however, incipient recovery from linuron treatment occurred after the first post-treatment week. Rates of recovery were similar in all cases, being about 0.5-1.5 mg $O_2L^{-1}d^{-1}wk^{-1}$.

The ratio of apparent photosynthesis to dark respiration (P_a:R) provides a measure of the energy balance for plants and has been used as an index of stress. The point where P_a just equals R is termed the "compensation point"; $P_a:R > 1$ indicates net growth, and $P_a:R < 1$ suggests net loss of plant material. It was found that P_a:R offered a useful reference to monitor SAV stress from herbicides; an example of such data is given in Figure 13 from Cunningham et al. (1981a). Here, P_a:R showed a small decrease in growth immediately following low-dose treatment, but recovery by the second post-treatment week. Yet, P_a:R for controls steadily increased, and P_a:R at high dosage dropped to near zero where it remained. Similar patterns were obtained for linuron and for other SAV species. Even though atrazine and linuron do not directly affect respiration, P_a and R are often closely coupled (because of the labile nature of early photosynthate); hence, decreased P_a often leads to some reduction in R. Therefore, P_a:R appears to provide a more integrated index of stress than P_a alone, and it represents a reasonable predictor of long-term changes in plant biomass.

The phytotoxicity of atrazine was also tested for the seagrass, *Zostera marina*, using *in situ* 24-hour incubations under clear plexiglass domes (Hershner et al. 1981). The nature of the responses observed for atrazine-treated systems varied considerably at low concentrations, but concentrations of 1,000 ppb consistently caused 100 percent loss of P_a (Figure 14). Some experiments, for example, showed regressive decrease in P_a with increased concentrations of the herbicide (Figure 14a); others indicated no effect at 1.0 ppb (Figure 14b), and still others suggested no significant impact of 1.0 or 10.0 ppb on P_a (Figure 14c). Summarizing the results of these experiments, a significant linear relationship was obtained between the log of herbicide dosage (X) and reduction in P_a, Y (compared to vehicle controls), despite the large degree of variability at 1.0 and 10.0 ppb, where

$$Y = 31.3X + 0.04 \qquad (r^2 = 0.76) \qquad (6)$$

for 12 experiments. Omitting three outliers at 10 ppb improved the fit ($r^2 = 0.92$), without significantly changing the slope (33.8).

The overall impact of herbicides on SAV apparent photosynthesis was remarkably similar for both herbicides, and for all three species that were studied intensively (Figure 15). The following regression was obtained for 34 varied experiments, using the same model indicated in Equation 6 (percent effect on Pa versus log herbicide concentration):

$$Y = 33.6X - 14.4 \qquad (r^2 = 0.89) \qquad (7)$$

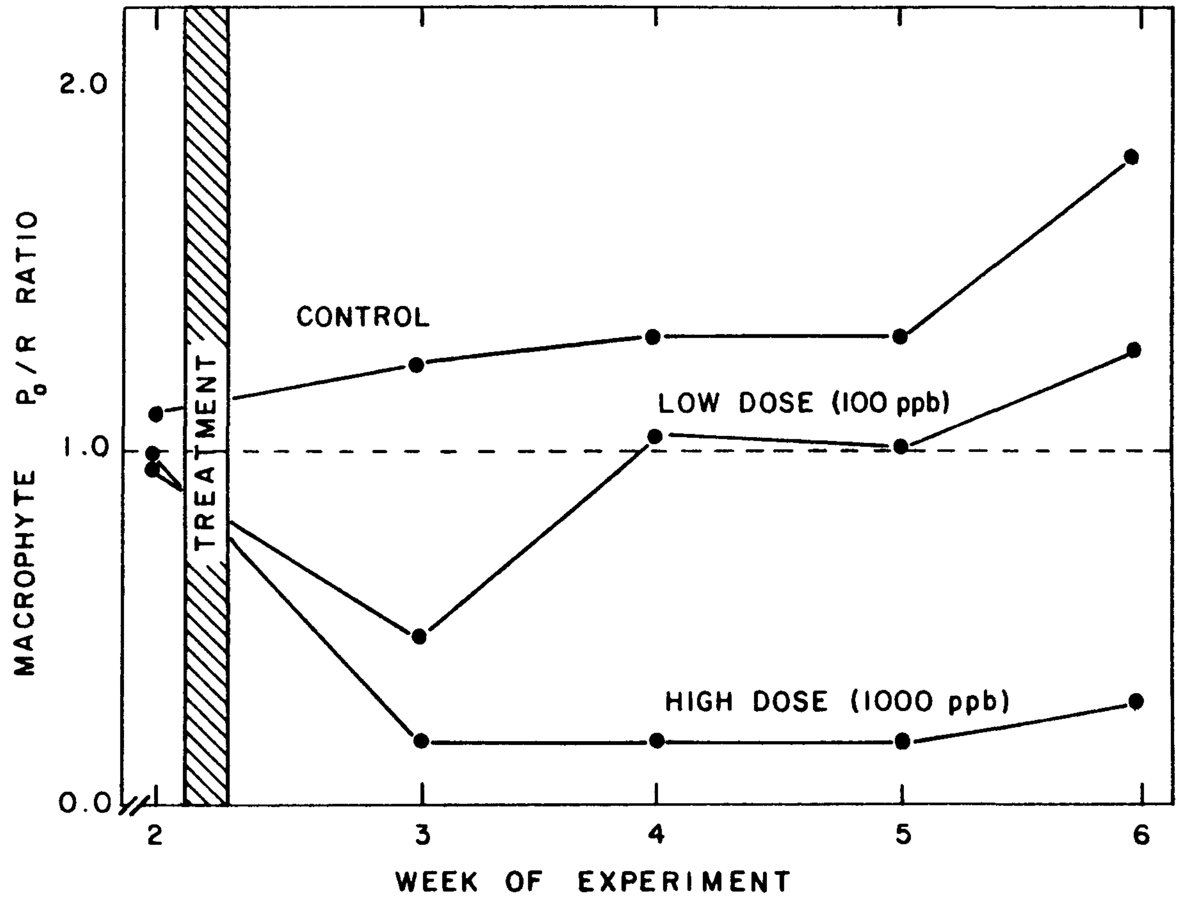


Figure 13. Ratio of apparent photosynthesis to night respiration (Pa/R) treated with two levels of atrazine (Cunningham et al. 1981a).

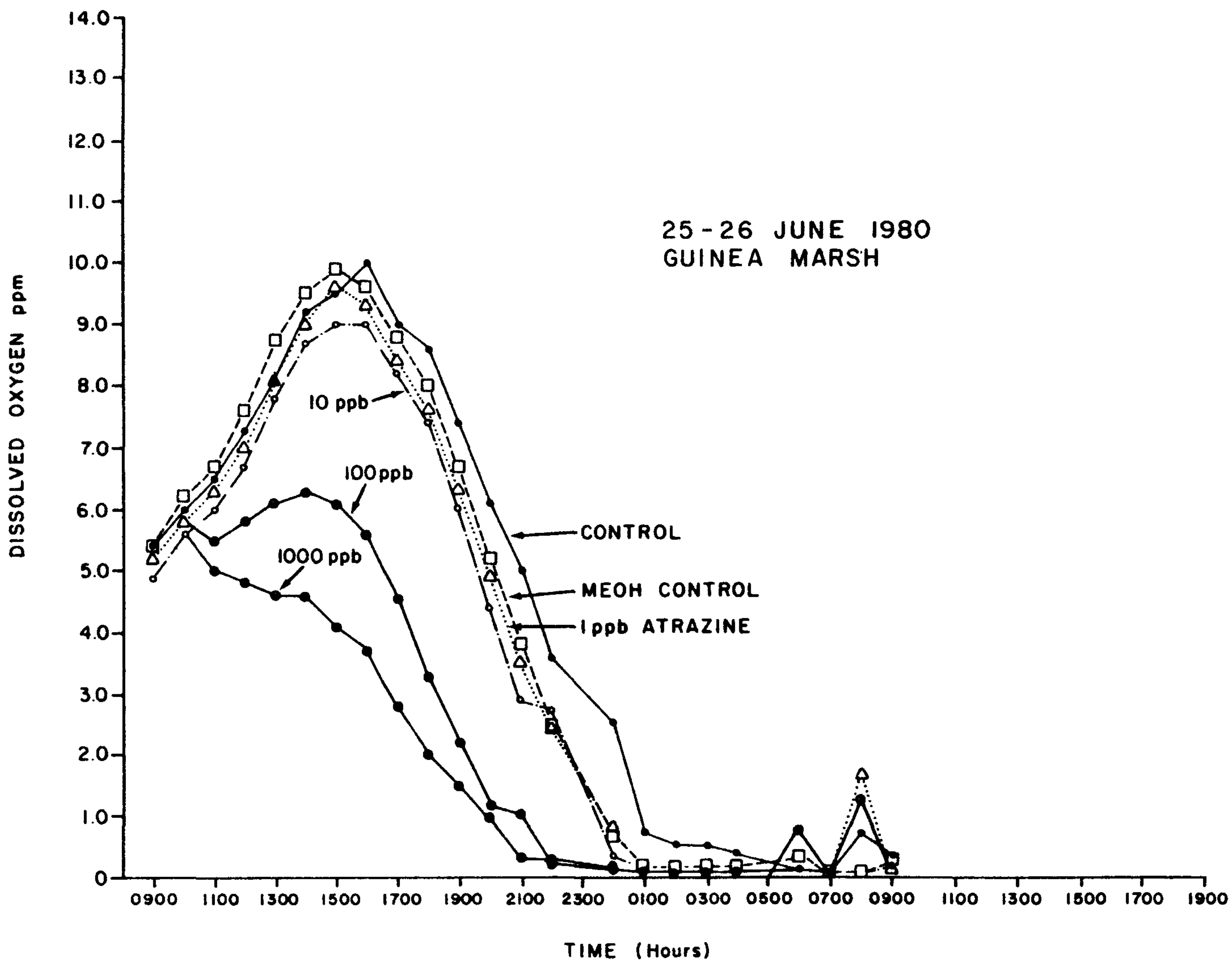


Figure 14. Typical patterns of diel O_2 under <u>in situ</u> domes covering Zostera marina communities tricated with atrazine and shading in Guinea Marsh, VA (Hershner et al. 1981).

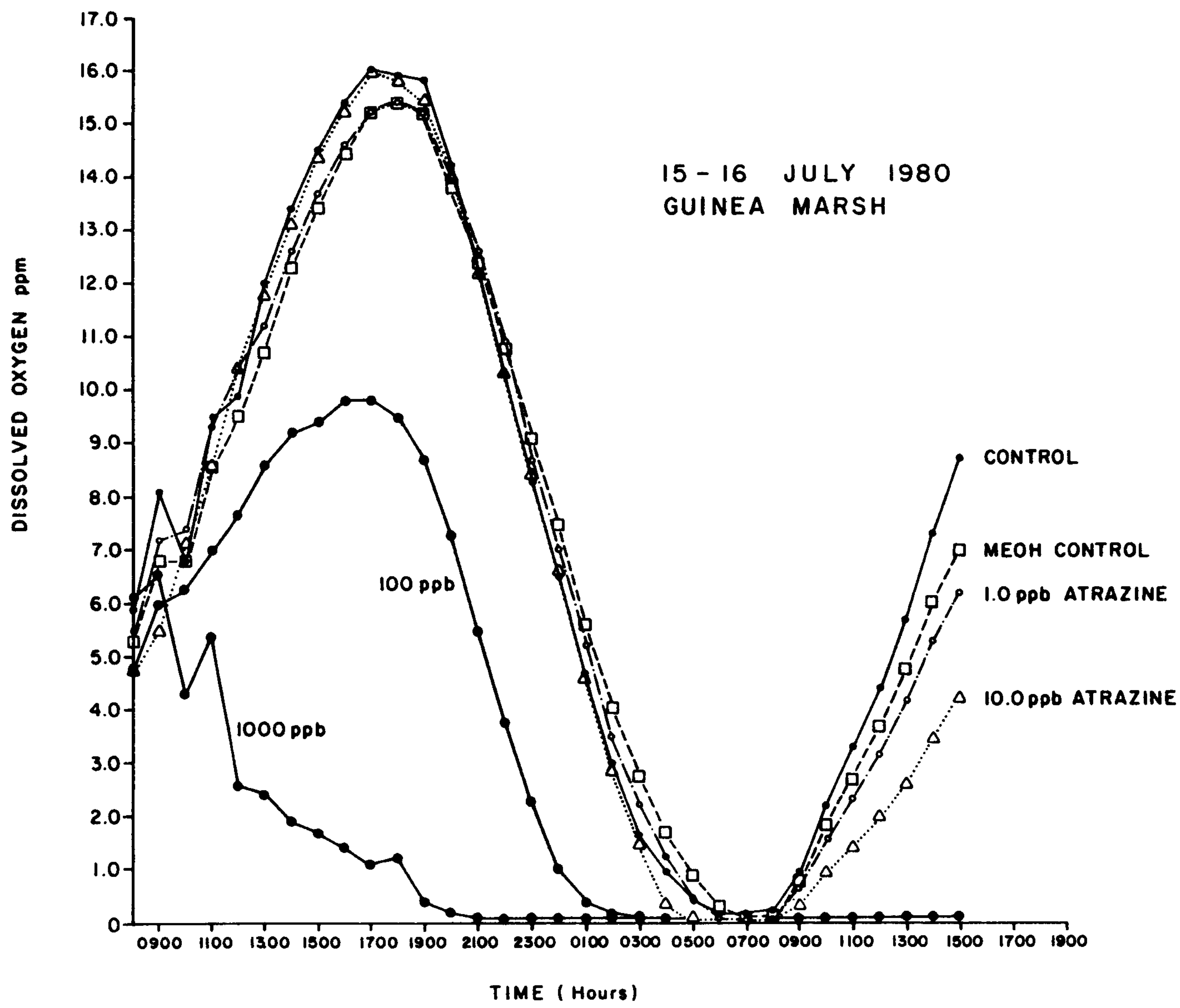
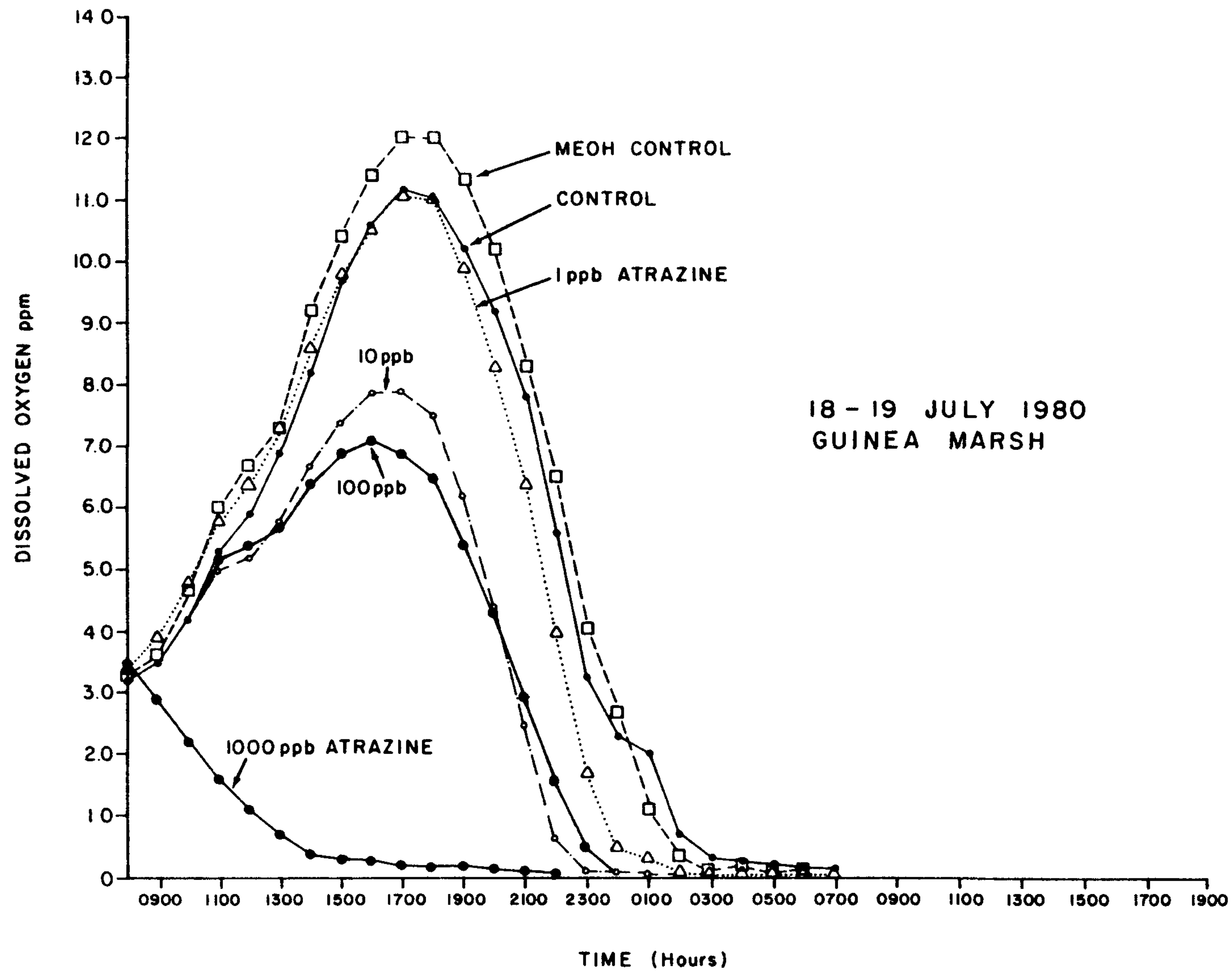


Figure 14. Continued.



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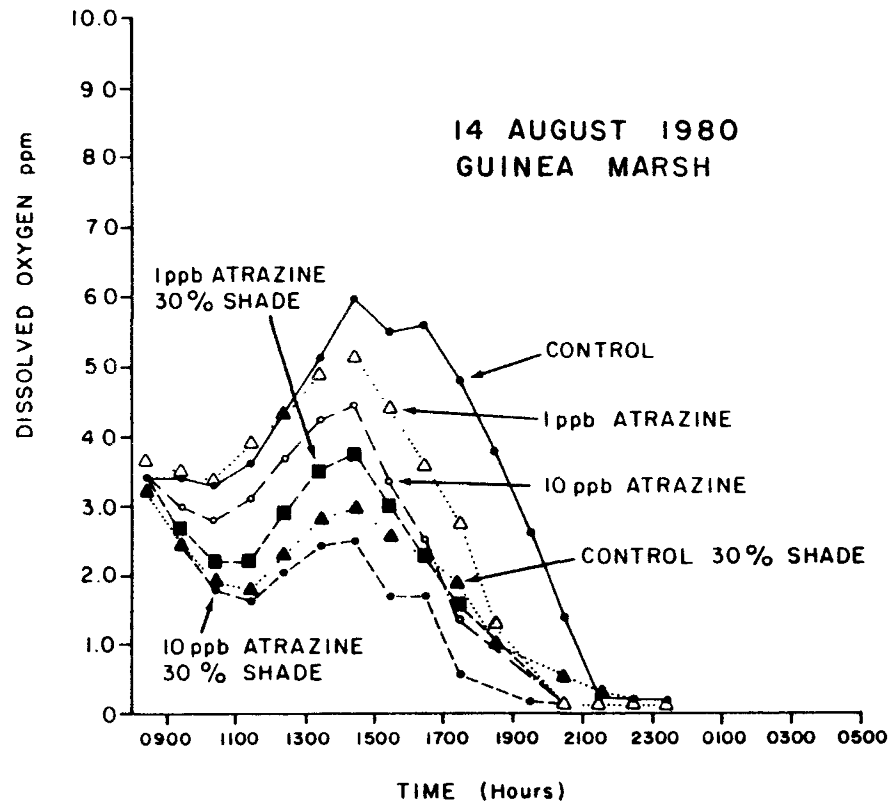

Figure 14. Continued.

for 34 varied experiments. Some differences were found between atrazine and linuron effects, and among the three SAV species. Of all species, Zostera exhibited the greatest effect at low herbicide levels, with an apparent threshold concentration (intercept of x-axis) of about 1.0 ppb; M. spicatum was the most resistant with a threshold of about 6 ppb. This model predicts that at 10 ppb atrazine, the resulting reductions of SAV P_a for Myriophyllum, Potamogeton, and Zostera would be approximately 0, 17, and 33 percent, respectively. Thus, in the lower-Bay waters small concentrations may have greater effect on SAV (that is, Zostera) photosynthesis.

Correll and his colleagues (Correll et al. 1978, 1978; Correll and Wu 1981) have reported other herbicide-SAV experiments for Chesapeake Bay plants. They have investigated atrazine effects on a second species of Potamogeton (P. pectinatus) and on Z. marina, as well as on two additional freshwater genera (Zannichellia palustris and Vallisneria americana). They have also reported some results of linuron effects on Z. palustris. Time-course experiments (21-48 days) have been performed for a range of herbicide concentrations. In general, the patterns of responses reported are similar to those in Figure 15, where, for example, linuron effects seem to be greater than those of atrazine. Correll's results, however, appear to suggest considerably greater resistance to atrazine for all test species. For example, the maximum effect found for any species at 75 ppb was for Z. palustris, which exhibited about a 40 percent reduction in P_a, and the minimum effect reported by Cunningham et al. (1981b) was 42 percent for M. spicatum at 100 ppb atrazine. Moreover, two of four species tested exhibited significant enhancement of P_a by 75 ppb (Correll and Wu 1981). One of those enhanced species was Z. marina, the same plant that Hershner et al. (1981) reported never exhibited less than 47 percent reduction in P_a at 100 ppb for five experiments.

Few other comparable data are available in the literature. Walker (1964) reported that Potamogeton sp. was effectively controlled in fish ponds (that is, removed from ponds for at least two months) with treatments of 0.5-1.0 ppm simazine. Herbicide bioassay experiments with the submerged vascular plants, Myriophyllum brasiliense and Elodea canadensis, showed that oxygen evolution was suppressed (25 and 40 percent less O_2, respectively) by simazine at concentrations of 120 ppb (Sutton et al. 1969). Fowler (1977) has shown, more recently, that effects of a related s-triazine herbicide (DPX 3674) on Myriophyllum verticillatum and P. pectinatus could be detected at 125 ppb. Stevenson et al. (1981) have reported some unpublished data of J. Forney for E. canadensis, where a one percent growth inhibition was found at 3.2 ppb atrazine, and a 50 percent inhibition occurred at 80 ppb.

Effects on SAV Population, Biomass, and Physiomorphology

Total SAV biomass responded to herbicide treatment in a manner analogous to plant photosynthesis. Typical biomass data from microcosm experiments are provided in Figure 16, where P. perfoliatus exhibited significant reduction in plant matter at concentrations greater than 50 ppb linuron, and M. spicatum followed a secular decrease in biomass when exposed to linuron concentrations from 5.0 to 1,000 ppb, although the loss was significant (compared with control) only at concentrations greater than

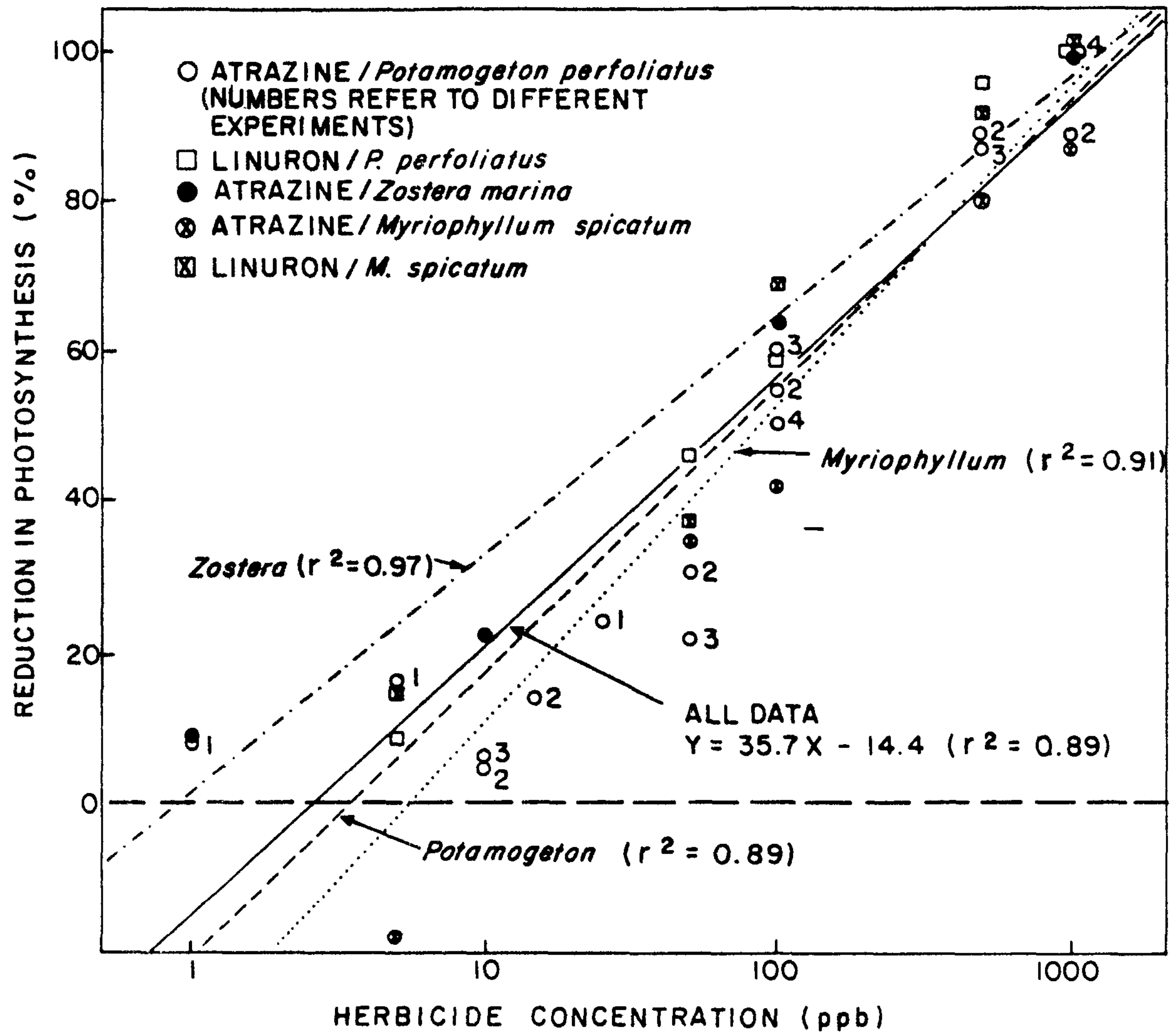

Figure 15. Effects of herbicides on SAV photosynthesis. Numbers adjacent to points for atrazine/P. perfoliatus indicate different experiments. (Data are compiled from Cunningham et al. 1981a, 1981b; Jones et al. 1981a; and Hershner et al. 1981.)

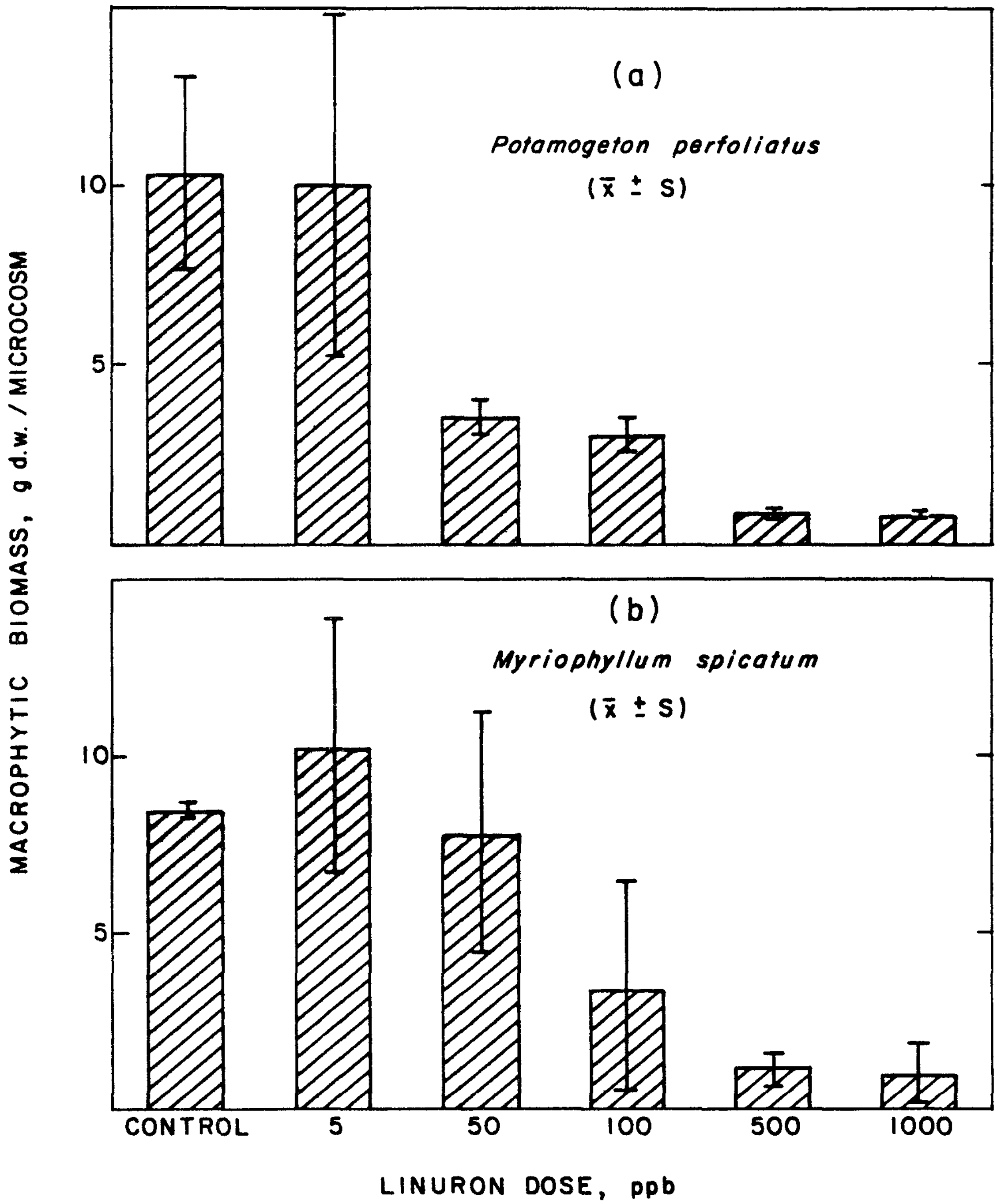

Figure 16. Summary of measurements of plant biomass in microcosms containing (a) Potamogeton perfoliatus, and (b) Myriophyllum spicatum, treated with linuron (0 to 1 ppm) (Cunningham et al. 1981b).

100 ppb. A slight, but insignificant increase in biomass was found at 5.0 ppb for M. spicatum. These data are from the final week of the experiment, and time-course effects of herbicide dosage on biomass lagged metabolic responses by one to two weeks. Ratios of above:below-ground biomass were generally unaffected by herbicide treatment, but shoot density was enhanced at moderate dosage (100 ppb) and sharply reduced at high doses (Table 5). Although above:below-ground biomass ratios can be an index of stress, none was found in this case. Hershner et al. (1981) reported only small, insignificant effects on Zostera shoot height, density, leaves-per- shoot, and mortality at atrazine concentrations less than 100 ppb after 27 days, whereas marked effects were apparent at 1,000 ppb (Figure 17). However, P. perfoliatus exhibited significant etiolation of stems and increases in chlorophyll a content of leaves at 100 ppb atrazine (Table 5), both of which are typical responses to light stress.

TABLE 5. SUMMARY OF SELECTED STRUCTURAL CHARACTERISTICS OF POTAMOGETON PERFOLIATUS POPULATIONS IN MICROCOSM COMMUNITIES TREATED WITH THE HERBICIDE, ATRAZINE (CUNNINGHAM ET AL. 1981a)[a]

	Treatment		
Structural[b] Characteristic	Control	Low (0.1 ppm)	High (1.0 ppm)
Chlorophyll-a ($mg\ m^{-2}$)	28 $\pm$ 8	158 $\pm$ 16	114 $\pm$ 5
Foliar Biomass (B_a) ($g\ d.w.\ m^{-2}$)	44.3 $\pm$ 17.1	24.3 $\pm$ 8.7	0
Rhizobial Biomass (B_b) ($g\ d.w.\ m^{-2}$)	40.0 $\pm$ 12.9	20.0 $\pm$ 8.6	0
Ratio, B_b:B_a	0.93 $\pm$ 0.22	0.94 $\pm$ 0.54	-
Unit Length of shoots ($cm\ g^{-1}$)	24	53	63
Shoot density ($no.\ m^{-2}$)	468	495	134

[a]Data are from samples taken in the final (6th) week of the experiment.
[b]Given are mean values $\pm$ standard deviation, where n = 12 for chlorophyll and n = 6 for biomass. Values for shoot length and shoot density are measurements from harvest of entire plant population for duplicate microcosms at each treatment.

Correll and Wu (1981) examined the effects of atrazine on V. americana in some detail, after initial screening experiments indicated that it was the most sensitive of the four species they examined. They monitored mortality, vegetative reproduction, and leaf growth as indices of herbicide stress; they observed about a 35 to 40 percent increase in mortality above control at 12 ppb atrazine, with similar losses of reproduction and growth. No significant effects were seen at 3.2 ppb atrazine. This effect

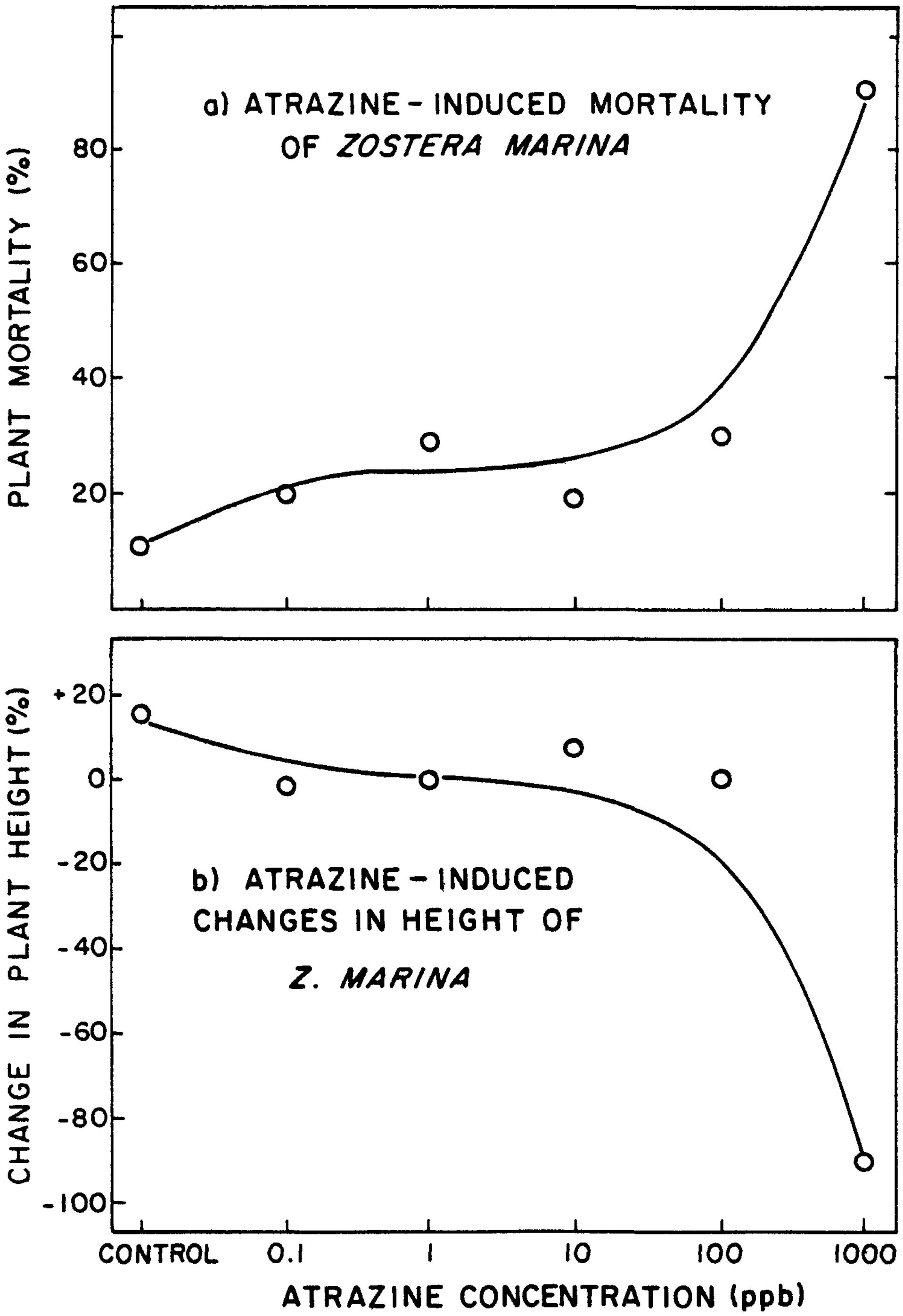

Figure 17. Effects of atrazine on (a) percent mortality of Zostera and (b) height of Zostera (Hershner et al. 1981).

of 12 ppb is reasonably consistent with the decreases in biomass that were found for P. perfoliatus at 50 ppb (either linuron or atrazine), where up to 60 percent reduction (for example, Figure 16) occurred (Cunningham et al. 1981b). Correll and Wu (1981), however, found almost a 40 percent increase in mortality of Vallisneria at 12 ppb atrazine, but only a 20 percent decrease in P_a at 75 ppb. It would seem that effects on P_a should be greater than those on survivorship, since reduction in photosynthesis does not necessarily lead to death, whereas the inverse is true. We have no explanation for this apparent inconsistency.

Other Factors Affecting Phytotoxicity

To place these bioassay experimental results into proper perspective, it is necessary to consider several factors that influence the ultimate effect of herbicides on SAV.

Acute Versus Chronic Exposures--

Somewhat surprisingly, the relative toxicities of herbicides to SAV appeared to be independent of exposure-time. Experiments involving incubations of six to 24 hours (Jones et al. 1981b, Hershner et al. 1981) yielded results virtually identical to those obtained from exposures of four to five weeks (Cunningham et al. 1981a, 1981b). In the linuron experiments, dose-response patterns closely followed those for atrazine, even though only 10 percent of the parent linuron remained after a four-week period. On the other hand, P_a of Potamogeton treated with 50 ppb atrazine dropped to 35 percent of controls two weeks after treatment, but recovered to 70 percent of control levels two weeks later, even though herbicide concentration remained at 75 percent of initial levels (Figure 12). Thus, it appears that the initial short-term exposure to herbicides, at a given concentration, largely determines the subsequent pattern of stress and recovery.

Based on recently-conducted, time-series measurements of ^{14}C-labeled atrazine uptake by P. perfoliatus, it appears that most uptake occurs within one to two hours, and no additional incorporation can be measured after two days of exposure to constant concentrations (T. Jones, unpublished data). This finding suggests that, in nature, even ephemeral exposure of SAV to herbicides, such as that following a runoff event, can induce the same general phytotoxic response and recovery as observed in the batch microcosms experiments (Figure 12). The rate at which P. perfoliatus loses previously-incorporated herbicide is currently being examined at University of Maryland Center for Environmental and Estuarine Studies (UMCEES) with continuous, subsequent exposure to herbicide-free water. Apparently, the metabolic recovery shown in Figure 12 involves some sort of enzymatic detoxification rather than depuration (active or passive removal via excretion or some other means).

Mode of Uptake--

Jones et al. (1981b) reported that P. perfoliatus uptake of ^{14}C-labeled atrazine could occur either through shoot or root pathway, although shoot uptake appeared to dominate. This finding is generally consistent with findings of previous investigators. Aldrich and Otto (1959), for example, reported that P. pectinatus was equally capable of either root or shoot incorporation of 2,4-D-1-C, and Funderburk and Lawrence (1963) found that simazine was also taken up through both routes by the SAV, Heteranthera dubia, but that other herbicides showed no root-to-shoot translocation through the stem. Frank and Hodgson (1964) also reported uptake of fenac by both roots and shoots of P. pectinatus, but little or no translocation in either direction. Hence, while some herbicide uptake by SAV roots is possible, limited ability to translocate up the stem reduces the importance of this mechanism for chloroplast-active compounds such as atrazine and linuron. In addition, the relatively high adsorptive tendency of these herbicides to sediment surfaces may reduce the herbicide exposure of roots.

Atrazine uptake by P. perfoliatus occurs hyperbolically, as a function of external concentration (C_e). At C_e less than about 450 ppb internal plant concentrations of atrazine, C_i, were less than C_e, while C_i approaches C_e at about 500 ppb. Moreover, as C_i approaches C_e, P_a approaches zero. Thus, atrazine incorporation does not follow a strict Fickian diffusion at low herbicide concentration, although the first-order process is approximated at $C_e < 500$ ppb (Jones et al. 1981b).

It had been postulated that herbicides bound to suspended sediments and/or colloids, which subsequently settled on SAV leaf surfaces, represented a potential mechanism for magnifying the concentrations to which plants are exposed. However, recent experiments have indicated that P. perfoliatus shows little or no uptake of atrazine from herbicide-bound sediments placed on SAV leaves (T. Jones, unpublished data). In addition, Correll and Wu (1981) found that atrazine concentrated in surface microlayer films exhibited no greater phytotoxicity than the same quantity mixed in a large water-volume bathing SAV in microcosm experiments.

Combined Stresses--

Although it is important to understand the individual effects of herbicide stress on SAV, many environmental factors act simultaneously on the plants in nature. Therefore, the significance of combined effects of herbicide-herbicide, herbicide-light, and herbicide-nutrient interactions must be considered.

Some preliminary investigations of atrazine-linuron combinations with P. perfoliatus suggest that 25 ppb of each herbicide reduced P_a more than 50 ppb of either herbicide combination for the last two post-treatment weeks. However, there was no difference estimated by the Colby (1967) formulation for the first two post-treatment weeks (Cunningham et al. 1981b). Additional experiments with herbicide combinations are in progress, but these results are not yet available. The agricultural weed-control literature contains some information on combined herbicide effects, but these reports are also inconclusive. For example, Horowitz and Herzlinger (1973) found that only one out of seven combination experiments with diruon, simazine, trifluralin, and fluometurn at 0.1 and 0.5 ppm exhibited significant synergism. Akobundu et al. (1975) observed synergistic action between atrazine and alachlor, but this nonlinear effect was small. Appleby and Somabhi (1978) investigated the reported antagonisms between atrazine (or simazine) and glyphosate and found that the interaction was due more to the physical binding in spray solution than to any biochemical mechanism. At this point, the importance of herbicide-herbicide synergisms for SAV toxicity in the Chesapeake Bay region is unclear; however, the effect is probably small.

A consistent pattern of interaction between light and herbicides has been observed for SAV and other plants. When subjected to 30 percent shading, the relative response of Z. marina to atrazine treatment (Figure 14d) was markedly reduced as compared to 100 percent light (Hershner et al. 1981). When the effects of atrazine (50 ppb) on P. perfoliatus photosynthesis were tested over a full range of light intensities, it was found that the relative toxicity (reduction in P_a) is greater at less-than-saturating light intensities (T. Jones et al., unpublished data). Hodgson and Otto (1963) showed that two species of Potamogeton were more sensitive to contact herbicides at high, rather than low light intensities, and similar results have been reported for algae (McFarlane et al. 1972) and weeds (for example, Hammerton 1967). This relationship probably exists, because the more active chloroplasts operating at high light levels are more susceptible to herbicidal damage. In a related experiment, we observed that epiphytic sediments significantly reduced atrazine uptake by P. perfoliatus leaves (T. Jones et al., unpublished data). This effect may be attributable to a combination of physical buffering and sorption by epiphytic sediments, as well as the light-herbicide relationship mentioned above.

Under conditions of nutrient sufficiency, which occur for SAV throughout most of the upper and middle Bay, nutrient additions would be expected to show little effect on herbicide phytotoxicity. This conclusion was reached after a series of recent experiments. However, SAV grown for months in microcosms can experience nutrient (or CO_2) limitation (Kemp et al. 1980) and thus provide a simple way of addressing the question of nutrient limitation. In Figure 18, the results of four different atrazine-Potamogeton experiments are summarized in a manner that may reveal such a relationship. A negative trend was found between herbicide toxicity at 25 ppb and maximum P_a at full incubator light intensity (150 uE $m^{-2}s^{-1}$). It might be inferred that nutrient limitation (or some other environmental stress reducing peak P_a) increases herbicidal action on SAV. The potential effect of nutrient-induced epifloral growth attached to SAV on herbicide stress has not been examined, but it might be expected to act in much the same fashion as did epiphytic sediments.

Metabolites of Herbicides--

One herbicide-SAV issue that has yet to be addressed in this research is the potential toxicity of herbicide metabolites, or degradation products. In a previous Section, it was shown that the dealkylated daughter products of atrazine degradation occur at persistently low levels ($<$ 10 percent of original atrazine) in soils, sediments, and water. Various investigators have shown that the dealkylated degradation products of atrazine retain some toxicity (although considerably less than the parent compound) to terrestrial plants (Shimabukuro 1968, Kaufman and Blake 1970). The carry-over toxicity, which has been reported for atrazine-treated fields from one year to the next, has been attributed by some to the persistence of the N-de-ethylated metabolites. Both Sirons et al. (1973) and Dao et al. (1979), for example, have reported carry-over toxicity after atrazine application to croplands for as long as two years. At present, the toxicity to SAV from metabolites of atrazine and other herbicides is not known, nor are the levels of these compounds in the

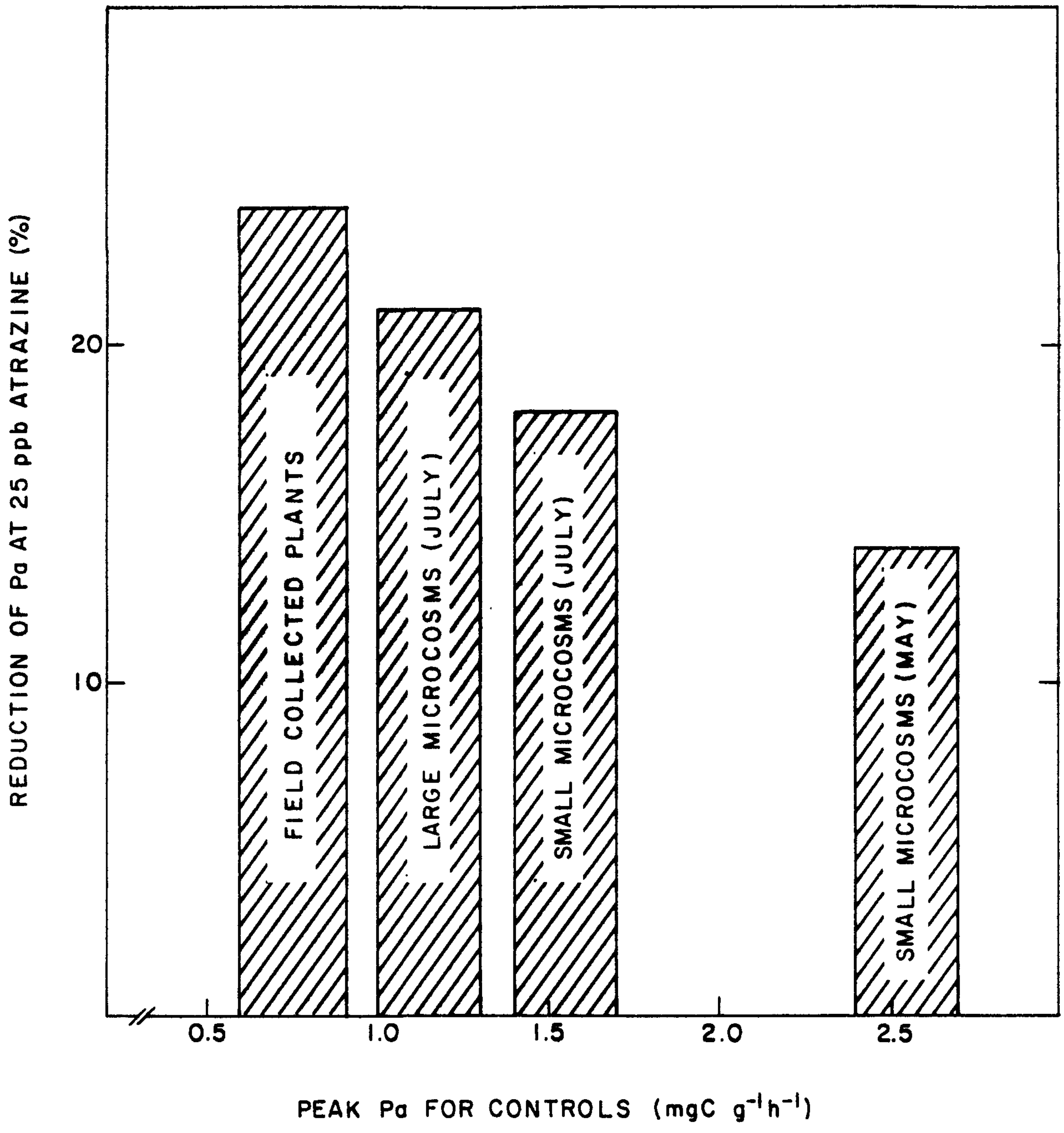

Figure 18. Effects of plant vigor on atrazine response for *P. perfoliatus* (Cunningham et al. 1981b).

environment. Although the potential for metabolite build-up in estuarine sediments may appear remote, the issue remains the one major gap in our understanding of the overall herbicide-SAV issue.

SECTION 6

SUMMARY AND IMPLICATIONS

SUMMARY OF RESEARCH FINDINGS

In this paper we have highlighted the results of extensive research supported by EPA-CBP to investigate the behavior of agricultural herbicides in an estuarine environment, particularly in relation to Chesapeake Bay's dwindling communities of submerged aquatic vegetation. The relative, potential importance of various herbicidal compounds in relation to SAV was considered, and atrazine and linuron were selected for primary focus. The watercourses in this coastal region have been systematically sampled for herbicide concentration from the mainstem Chesapeake Bay, to primary tributaries, to secondary bays and coves, to creeks that drain agricultural fields. Maximum observed concentrations of these two major herbicides in the four levels of tributaries were about four ppb, 7.0 ppb, 20 ppb, and 100 ppb, respectively. High herbicide concentrations of about 10 to 20 ppb were observed to occur in estuarine waters for ephemeral periods of two to eight hours. The length of time between herbicide application to the cropland and the first rainfall-runoff event, and the extent and intensity of rainfall, are key factors governing the transport of herbicides from the field to the estuary.

The degradation of atrazine in estuarine environments appears to occur far more rapidly than in agricultural soils, with half-lives of two to 26 weeks, respectively. The longevity of linuron is less than that of atrazine and, in fact the latter compound appears to be one of the most persistent herbicides used in the watershed. Atrazine exhibits moderate tendency for adsorption to soils and estuarine sediments. Most of the herbicide running off from field to watercourse does so in the dissolved form, rather than bound to soil particles, and most of the atrazine in the estuary is similarly found in the dissolved state. Estuarine colloids have about 10 times greater ability to bind atrazine than do sediments and soils. Salinity and circum-neutral pH appear to exert little influence on herbicide-sediment sorption; however, increased salinity does tend to decrease the proportion of herbicide bound to colloidal matter.

Atrazine brings about a dramatic stress response for several species of SAV at concentrations of 50 to 100 ppb. At these concentrations, reductions of photosynthesis are always significant, and full recovery of photosynthetic rates may not be attained. The relation between percent loss of photosynthesis and herbicide concentration generally follows a semilogarithmic function for all species and both compounds tested. This model predicts threshold toxicities at herbicide concentrations ranging from 1.0 to 7.0 ppb. Combining all experimental data for three species and two herbicides yielded a highly significant regression ($r^2 = 0.89$), which predicts about 10 to 20 percent loss of SAV photosynthesis at 5.0 to 10.0 ppb herbicide. Similar herbicidal effects were observed for plant structural characteristics.

Experiments with P. perfoliatus and atrazine indicate that herbicide uptake, which is a function of external concentration, proceeds to equilibrium within one hour, and that depuration (loss of herbicide) upon exposure to clean water occurs very slowly, with toxic effects still apparent after days of cleasing. Reduced light level (above compensation light) and/or presence of epiphytic sediments appear to decrease the relative stress effect of herbicide on SAV, but nutrient deficiency and plant senescence may tend to increase herbicide effects. There are currently few data to support the hypothesis that combinations of two or more herbicides act in any other than an additive fashion. The potential toxicity of herbicide metabolites (degradation products) to SAV is a matter about which very little is known.

DID HERBICIDES CAUSE THE SAV DECLINE?

From the evidence that has been compiled here, the answer to this question is most likely no. Herbicide concentrations in excess of 20 ppb were not found in estuarine waters in various surveys since 1977, under a range of situations, including those which approach worst-case runoff conditions. Under such extreme conditions, concentrations of 10 to 20 ppb were observed, but rarely lasted more than four to eight hours. Although exposures to 20 ppb (of one hour or more) will cause significant loss of productivity, full metabolic recovery would be expected within one to four weeks following initial contact. Moreover, herbicides degrade rapidly in the estuarine environment, with half-lives measured in days and weeks; residual concentrations do not appear to build up in sediments. The hypothesized mechanisms of increasing SAV exposure to herbicides via concentration of the compounds in epiphytic sediments or surface-layer films do not appear to represent significant factors. One of the caveates that remains unresolved is the fact that very little is known about estuarine concentrations and SAV toxicities of major herbicide metabolites. The de-ethylated daughter products of atrazine degradation do tend to persist for months under estuarine conditions, and weed-control literature attributes carryover toxicity (after atrazine application) to this metabolite.

ARE HERBICIDES A PROBLEM?

Ephemeral herbicide concentrations in excess of 5.0 ppb do occur periodically in some estuarine water that once contained extensive SAV beds. In general, such concentrations appear to cause losses in SAV productivity of 10 to 20 percent, even when exposures are brief (about an hour), and recovery may take days to weeks. The effects of repeated, brief exposures to such concentrations are not known. A reasonable assumption, however, would be that if the time interval between runoff events (which might yield such deleterious concentrations) is greater than SAV recovery time, then the partial loss of photosynthesis may persist. Such reductions in SAV productivity will definitely add to the generally stressed conditions that these plants currently experience in the estuary. The sources of most of these stresses include such factors as salinity extremes, waterfowl grazing, uprooting by cownose rays, and turbulent waters or violent wave action caused by major storm events, as well as water-column turbidity and the accumulation of epiphytic materials. Herbicide-induced loss of productivity (though minor) could act in concert with many of these stressors to create intolerable conditions for SAV existence.

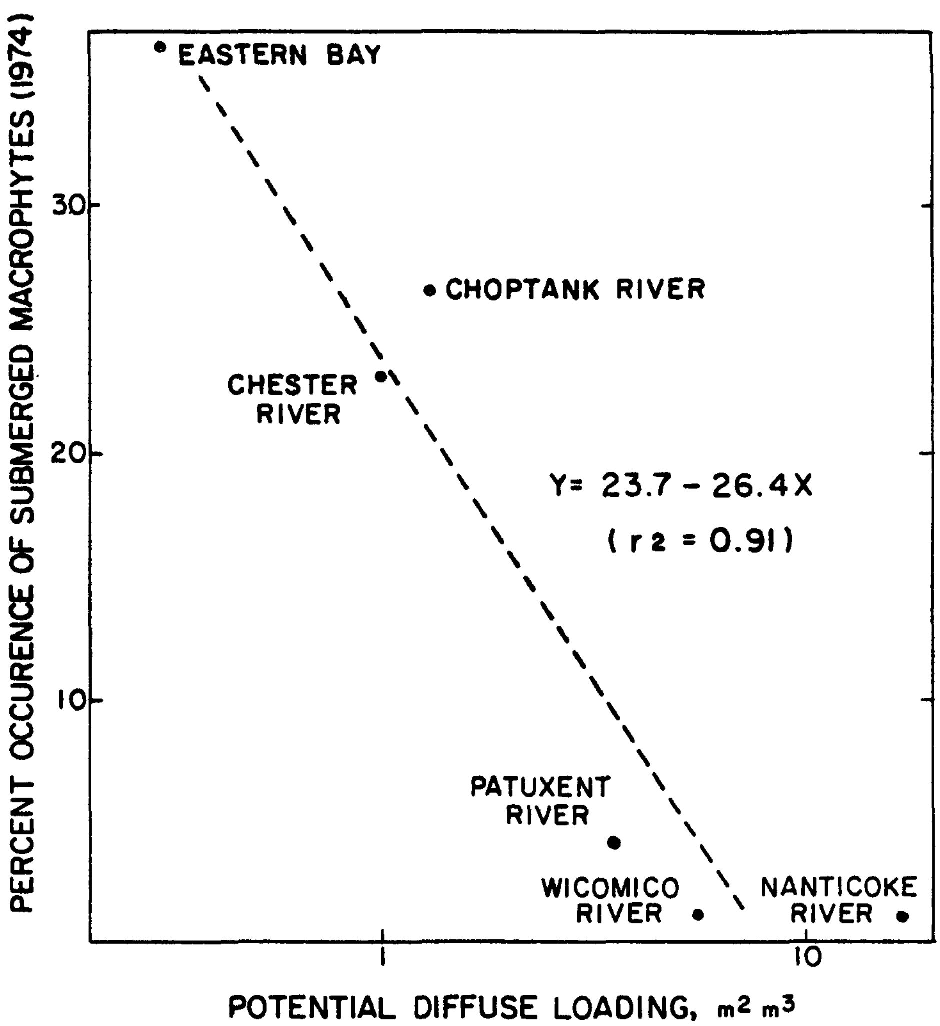

Figure 19. Correlation of potential diffuse loadings and percent occurrence of SAV in 1974 (Stevenson and Confer 1978).

The source of herbicidal compounds to Chesapeake Bay is agricultural runoff. There appears to be a relationship between potential loadings (that is, watershed area divided by estuarine volume) of nonpoint, or diffuse-source materials (including herbicides, nutrients, and sediments), and SAV abundance in six major tributaries (Figure 19). This correlation suggests that the greater the loadings from runoff, the more extensive the decline has been.

Although the development of recommended farming practices is well beyond the scope of this research endeavor, it is hoped that these research results and their environmental implications will be considered by the agricultural community in the evolution of improved farming approaches. The importance of agriculture in the socio-economic milieu of the Chesapeake region is unquestionable. Our recommendation is simply that the estuarine resource values be considered in concert with the land-based values to develop balanced patterns of human enterprise.

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LIGHT AND SUBMERGED MACROPHYTE COMMUNITIES IN CHESAPEAKE BAY: A SCIENTIFIC SUMMARY

by

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SECTION I

INTRODUCTION

The initial focus of submerged aquatic vegetation (SAV) research in the U.S. Environmental Protection Agency (EPA), Chesapeake Bay Program (CBP) was evaluation of the structural and functional ecology of these communities. In the upper Bay, *Myriophyllum spicatum* and *Potamogeton perfoliatus* are the dominant species; the dominant species in the lower Bay are *Zostera marina* and *Ruppia maritima*. Studies centered on various aspects of productivity (both primary and secondary), trophic structure, and resource utilization by both ecologically and economically important species. Much of the initial research was descriptively oriented because of a general lack of information on Chesapeake Bay submerged plant communities. These investigations created the data base necessary for the development of ecologically realistic simulation models of the ecosystem. Following these initial studies, the research programs in both Maryland and Virginia evolved toward more detailed analyses of specific factors that potentially limit or control plant growth and productivity. Previous results indicated certain environmental parameters and biological processes that possibly limited and controlled SAV distribution and abundance. Specifically, these included light, nutrients, herbicides and fouling (epibiotic growth). Laboratory and field studies were devoted in the later phases of the CBP-SAV program toward investigating these interactions. This work is among the first studies in North America to investigate light quality as a major environmental factor affecting the survival of sea grasses.

The overall objectives of this later work were to evaluate more precisely environmental and biological factors in relation to submerged aquatic plant community structure and function. Both the published literature and the results of CBP-SAV program studies indicate that the interaction of these environmental parameters, together with other physical and biological characteristics of the ecosystem, determine the longer term success or failure of SAV communities (den Hartog 1970, den Hartog and Polderman 1975, Williams 1977, Wetzel et al. 1982).

BACKGROUND

A major goal of CBP-SAV research was to investigate the response of Bay grasses to various environmental variables. Studies centered on the four dominant submerged aquatics in the Bay. Understanding the relationship between environmental factors and the productivity and growth of SAV was determined to be the first step necessary in attaining the overall goals of the management program. Natural and man-made changes in environmental quality may favor one species or another, or result in alteration of the entire community. The basic responses of the grasses, as well as the entire community, must be determined before environmental change can be evaluated in terms of specific management criteria.

Studies in the various CBP-SAV research programs that address environmental regulation and control of SAV communities focused on nutrient regulation [primarily nitrogen as ammonium (NH_4^+) and nitrate

(NO_3^-)], light and photosynthesis, and other biological and physical-chemical factors influencing light energy distribution.

The results of studies in the lower Bay communities suggest a net positive response to short-term nutrient additions and support the observation by others that these communities are nutrient limited (Orth 1977). The most consistent positive response is associated with *Ruppia* dominated communities, and the most variable is associated with the deeper *Zostera* community (Wetzel et al. 1979). In contrast, Kemp et al. (1981b) observed that upper-Bay SAV communities did not appear nutrient limited, but were perhaps limited by suboptimal light conditions. These results, together with community metabolism studies, suggest that light and the environmental factors controlling available light are key factors governing plant community growth and productivity. Light-temperature-turbidity regimes and their interaction may explain, in large part, observed variability in distribution and abundance. Changes in these parameters, governed by either natural or man-induced events and, perhaps, determined over longer time scales, influence variation in distribution and abundance in Chesapeake Bay ecosystem as a whole.

Throughout Chesapeake Bay, submerged aquatic plant communities exhibit a distinct zonation pattern from the shallower inshore high-light area to the deeper, low-light area of the beds. These characteristic distribution patterns also suggest different physiological responses to and control by local environmental conditions, principally light.

Studies were initiated in August, 1979, on lower-Bay *Ruppia*-*Zostera* communities and continued for an annual cycle to investigate the effects of light and temperature on specific rates of seagrass photosynthesis. The experiments were ^{14}C uptake studies in which plants were removed from the sediment, placed in a set of screened jars, and incubated in a running seawater system using ambient sunlight. The plants were exposed to 100, 50, 30, 15, 5, and 1 percent of ambient light to determine the effect of light quantity on photosynthesis. Experimental designs comparable with these were also conducted for upper-Bay species. Results are discussed later in this paper in Section 3.

In conjunction with these studies, measures of leaf area index (LAI) were also conducted. Physiologically, the photosynthesis-light relationship determines the light levels at which SAV can grow and reproduce, that is, succeed. A greater leaf area exposed to light results in greater productivity; however, light reaching the plants is not only determined by physical factors controlling light penetration through the water column, but by plant self-shading. Maximum plant biomass can in part be related to leaf area. The leaf area index (plant area per sediment surface area) estimates maximum leaf density and thus potential area available to intercept light (Evans 1972, cited in McRoy and McMillan 1979).

Leaf surface area also provides a substrate for epiphytic growth. Leaf area samples were collected to characterize the three main vegetation zones typical of lower-Bay communities. These data were used to provide a more accurate description of light penetration through the plant canopy as well as to evaluate potential morphological adaptation of the plants to various light environments. To complement these specific ^{14}C studies and LAI measures, field studies were completed to determine the effect of *in situ* light reduction through artificial shading. Light reductions of 70 to 20 percent of ambient were used. The results of these studies support the hypothesis that total community metabolism is governed by, and is very sensitive to, available light. During the course of these investigations, light data collected in the field for various environmental (climatic) conditions indicated that natural light reductions of these magnitudes were common. To determine the overall effects of light reduction, specific factors were investigated more thoroughly using both laboratory and in situ experimental approaches for light-photosynthesis relationships, as well as studies that determined those environmental variables controlling light energy distribution and availability to the plant communities.

Studies initiated during the later phases of the CBP-SAV research program investigated the effects of epiphytic growth and metabolism, and the interactive effects of light and acute exposure to the herbicide atrazine. Studies on epiphyte colonization were along two lines: the epiphytic community as a primary producer and food source, and as a competitor with the vascular plant community for available light. Experiments completed suggest that the epiphyte community at times dominates metabolism of the community and limits light available for vascular plant photosynthesis. What remained to be determined was what environmental conditions favor colonization, and at what point does the resulting colonization stress the vascular plant.

These various research activities provide a data and information base that serve management needs and identify specific research areas where additional information is required for integration and synthesis. The work proposed in the later part of the CBP-SAV program centered on filling what were considered major gaps in information and the data base. The synthesis report that follows is directed to our current state of understanding of light energy properties and distribution in Chesapeake Bay and to the relation of this information to past and current knowledge about SAV community growth and survival.

THE RESEARCH PROGRAM ON LIGHT AND SAV: AN OVERVIEW

It has been the working hypothesis of the Chesapeake Bay Program-SAV group that changes in such water quality variables as suspended particulates (both living and non-living), dissolved substances, and nutrients alter, directly or indirectly, underwater light regimes in such a way as to limit benthic macrophyte primary production. Plants absorb light energy for the process of photosynthesis, converting water and carbon dioxide into organic compounds. White light (visible sunlight) is composed of a spectrum of colors that are used selectively by green leaves based on the plant's specific pigment complexes. Chlorophyll requires mainly red and blue light for photosynthesis; these wavelengths are absorbed, and the green and yellow bands are reflected. The accessory pigments also absorb in the blue region.

As light penetrates the water column, the energy content and spectral quality are changed by absorption and scattering. Water itself, dissolved substances, and particulate materials are responsible for both the absorption (conversion into heat energy) and the scattering of light. Selective absorption and scattering by these factors result in attenuation of specific light wavelengths causing a "color shift" (Kalle 1966, Jerlov 1976). Scattering, the change in direction of light propagation, returns some of the incident radiation toward the surface and thus further reduces the total light energy available to support photosynthesis. Phytoplankton act as both scattering and selectively absorptive and reflective particles and are in direct competition with other primary producers for the same wavelengths of light--the red and blue bands.

The temporal and spatial distribution of particulate materials and dissolved substances are largely determined by climatic variables and biological processes. Wind velocity and direction, tidal amplitude and frequency, current velocity, rain, and land runoff all interact to induce variations in water quality parameters and subsequently the spectral composition of light in the water column (Dubinsky and Berman 1979, Kranck 1980, Anderson 1980, Thompson et al. 1979, Scott 1978, Riaux and Douville 1980).

Based on these general premises, the light research program encompassed four basic facets: (1) description of the submarine light environment together with measures of various water quality parameters; (2) description of climatic and oceanic forcing functions; (3) detailed studies of photosynthesis-light relations by individual species and for entire SAV communities; and (4) analysis of the relationships and correlations among the above data and other available information. The measurement and collection of light, water quality parameters, climatic and oceanic forcing functions were made simultaneously with the light-photosynthesis investigations. Studies on both shores of the upper and lower Chesapeake Bay in vegetated and non-vegetated regions were undertaken.

Characterization of the light environment was accomplished using a Biospherical Instruments Model MER-1000 Spectroradiometer (Booth and Dunstan 1979). Specific attenuation in 12 biologically important wavelengths and integrated photosynthetically active radiation (PAR) values were calculated from these data. The spectral irradiance measurements were made in quantum units as suggested for biological studies by the Special Committee on Oceanographic Research (SCOR) of the International Association of Physical Oceanographers (IAPO).

There is a paucity of data on spectral irradiance in marine environments (Jerlov 1976). There are even fewer studies reporting data for estuarine waters, Chesapeake Bay being no exception. Burt (1953, 1955a, b), using a shipboard spectrophotometer, analyzed filtered seawater samples from Chesapeake Bay and concluded that the primary factor in light extinction was the filterable, particulate matter. Seliger and Loftus (1974) studied the spectral distribution of light in shallow water in a subestuary in the upper Bay in July and found a marked reduction of light in the 400-500 nm region of the spectrum. Champ et al. (1980) report an observed "orange-shift" for measurements made in the upper Bay during August, 1977, using a submersible solar illuminance meter equipped with optical filters. They suggest that there is a continuum of spectral shifts toward the penetration of longer wavelengths from oceanic to coastal to estuarine waters. This corroborates and extends Kalle's "yellow shift" theory (Kalle 1966). Kalle contends that the shift to longer wavelengths is more pronounced as the concentrations of suspended particles increases. These investigations make up, in large part, the only complementary data base and, to our knowledge, no data exists in and around SAV habitats.

Broad band (PAR) transmittance was determined with a Montedoro-Whitney *in situ* combination beam transmissometer and nephelometer. The transmittance data were used to calculate the attenuation coefficient "defined as the absorption coefficient plus the total scattering coefficient" (Jerlov 1976, Kiefer and Austin 1974). van Tine (1981) found significant correlations between absence of submerged aquatic vegetation and low transmittance values in an estuary in the Gulf of Mexico.

Total particulate matter (TPM), particulate organic matter (POM), particulate-ATP, particulate chlorophyll a, particulate inorganic matter (PIM) were monitored in light spectral studies. These various measures were used to estimate phytoplankton, zooplankton, detritus, and inorganic fractions of the TPM.

Wind velocity and direction, water current velocity, tidal stage and depth were determined concurrently with the other measures. Kiley (1980) suggests a close relationship between wind and current for the York River. In an effort to explain turbidity values, Williams (1980) calculated significant positive correlations between wind and turbidity for upper-Bay subestuaries. Ginsburg and Lowenstam (1957) and Scoffin (1970) showed a baffling effect of SAV on currents that caused particulate matter to settle out, generally improving the local light environment. Collection and analyses of these data formed the basis for characterization of the natural light environment and of the factors that are principal controls.

Various lines of evidence, as discussed earlier, suggest light in general as a major factor controlling the distribution and productivity of seagrasses. Preliminary studies demonstrated both potential nutrient and light quantity effects on plant community metabolism. In the later phases of CBP-SAV research, both field and laboratory studies were designed and carried out in a more quantitative sense on photosynthesis-light relations in Chesapeake Bay SAV communities.

For the field approaches, the entire SAV community and its interactions were included in experimental designs. Short-term shading experiments reflected the community response to daily variations in light quantity due to such natural phenomena as cloud cover, tidal stage, and storm events. Long-term shading studies reflected community response to possible situations where water quality deteriorates to the point where light penetration is reduced. The purpose of these studies is to estimate at what point, relative to light quantity, the SAV communities would die out. For the latter effort, sets of neutral density mesh canopies were placed in selected SAV areas for long term studies. Shaded and control areas were studied at regular intervals over the course of these experiments (1-2 months). With this design, community metabolism and various plant community parameters (e.g., leaf area index, chlorophyll a and b, biomass, and other plant meristic characters) were measured. Studies were carried out in spring, summer, and early fall, 1981, to include the major growth and die-back periods.

Past research programs in the CBP-SAV program resulted in several hypotheses that might explain both the short and longer term survival of Bay grasses. Among these, the potential for light, including those variables influencing light, or more specifically light-energy distribution, as a major environmental variable controlling SAV distribution, growth, and survival was postulated. The intent of the remaining sections of this report is to provide the general characteristics of light in natural aquatic systems with emphasis on Chesapeake Bay; to summarize the research results throughout the Bay relative to light and Bay grasses; and to discuss the potential for light or light-related casualty of Bay grass declines.

SECTION 2

LIGHT IN CHESAPEAKE BAY

GENERAL CHARACTERISTICS OF ESTUARINE OPTICAL PROPERTIES

The study of the interaction of solar energy with estuarine waters necessitates not only an understanding of the properties of light and water, but also of the myriad living and non-living entities, both dissolved and suspended, which affect the propagation of light in aquatic environments.

The sun emits electromagnetic radiation in discrete packs or quanta (Q) of energy called photons. The energy content ($\mathcal{E}$) of each quantum is directly proportional to the frequency (ν),

$$\mathcal{E} = h\nu$$

and indirectly proportional to the wavelength (λ),

$$\mathcal{E} = \frac{hc}{\lambda}$$

where h is Planck's universal constant, and c is the speed of light in a vacuum. This means that quanta of shorter wavelengths contain more energy than quanta of longer wavelengths.

The complete spectrum of downward irradiance for incoming solar radiation at the top of the atmosphere, at sea level, and at several water depths is illustrated in Figure 1a. Most of the energy reaching the earth's surface is contained within the shorter wavelengths (0.4 to 1 u or 400 to 1,000 nm[1]). Not surprisingly, this region includes the wavelengths of greatest biological importance, that is, 400 to 700 nm, the photosynthetically active region of the spectrum termed PAR or PHAR. There is almost no energy outside the PAR region at a depth of 1 m. Most of the "missing" energy has been converted to heat by absorption. Only four to 11 percent of incident irradiance between 300-700 nm is reflected from the surface or backscattered out of the water column (called albedo) (Clark and Ewing 1974).

The properties and concepts in optical oceanography are usually divided into two mutually exclusive classes, inherent and apparent. Inherent properties, such as absorption and scattering, are independent of changes in insolation (incoming light), whereas apparent properties, such as underwater irradiance, vary with changing solar and atmospheric conditions.

As light passes through the water column, its energy content and spectral quality are changed by absorption and scattering due to water itself, dissolved substances, and suspended particles. The combined effect of these processes is termed attenuation. The spectral distribution of the total attenuation coefficient (α), measured with the beam transmissometer, generally shows high attenuance at both ends of the PAR. Since α is an aggregated coefficient, it is informative to consider the component parameters that cause the observed attenuance.

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[1] 1 nm = 10^{-3} um = 10^{-9} m

Scattering is the change in direction of light propagation caused by diffraction, refraction, and reflection due to particles, water molecules, and dissolved substances. Scattering is wavelength dependent, but in an irregular and complex manner. Absorption is a thermodynamically irreversible process wherein protons are converted to thermal, kinetic, or chemical energy; photosynthesis is an example. Much of the attenuance in the long wavelengths is due to the water itself, as shown by James and Birge (1938) for pure water and by Clarke and James (1939) for filtered seawater (see Figure 1). The effect of sea salts on attenuance is insignificant. Pure water or pure seawater show a constant light attenuation. Of course, natural water bodies (particularly estuaries) are not pure, but contain constantly varying particulate and dissolved substances. Burt (1958), using uncontaminated filtered seawater samples, was able to determine the attenuance due to dissolved substances. By subtracting this from the total attenuation coefficient of non-filtered seawater, he was able to calculate the light attenuance due to particulate matter. The energy of blue and red wavelengths is selectively absorbed by particles, as shown in the example given by Prieur and Sathyendranath (1981) (Figure 1b). The shorter wavelengths are also attenuated by yellow substance or Gelbstoff (see Figure 1b), the collective name given to a complex mixture of organic compounds by Kalle (1966). Gelbstoff is formed from carbohydrates resulting from organic matter decomposition. Sources are both allocthonous (swamps, marshes, land runoff) and autocthonous (planktonic and benthic organisms). Flocculation of fine suspended and colloidal materials in estuaries probably promotes the reaction, as does the presence of amino acids (Kalle 1966).

The apparent optical properties of a body of water result from the measurement of natural light fields underwater, that is, the measurement of _in situ_ radiant flux. Irradiance (E) (the flux of light per unit area) is usually collected with a flat circular opal glass (or plastic) diffuser (2π collector). The diffuser is designed so that light received from all angles is transmitted to the sensor according to Lambert's cosine law. In other words, the irradiance transmitted is proportional to the incident radiant intensity multiplied by the cosine of the angle of incidence. Jerlov (1976) reports that the ratio of cosine collection of downwelling irradiance (E_d) to equal hemispherical collection (E_o) is generally in the range of 0.75 to 0.85 downwelling. 2π irradiance is the apparent property of water bodies most commonly measured for biological purposes, and was the measure used in CBP-SAV research. Of course, irradiance can be expressed as either energy or quanta and measured in broad spectral regions, such as the PAR, or at discrete wavelengths (spectral irradiance). A family of downwelling spectral irradiance curves, in quanta, is shown in Figure 2 for a _Zostera marina_ bed on the eastern shore of Chesapeake Bay. This figure shows that both total light energy and that of specific wavelengths are lost with depth. At 0.1 meter, for example, a lot of surface insolation, particularly in the photosynthetically important 400-500 range, has been lost.

Primary producers or autotrophs contain light-capturing pigments to carry out photosynthesis. Most phytoplankton possess a pigment complex similar to that of seagrasses and other higher plants. These pigment systems absorb strongly in the blue and red regions (chlorophyllous pigments). Figure 1b illustrates how combinations of water column

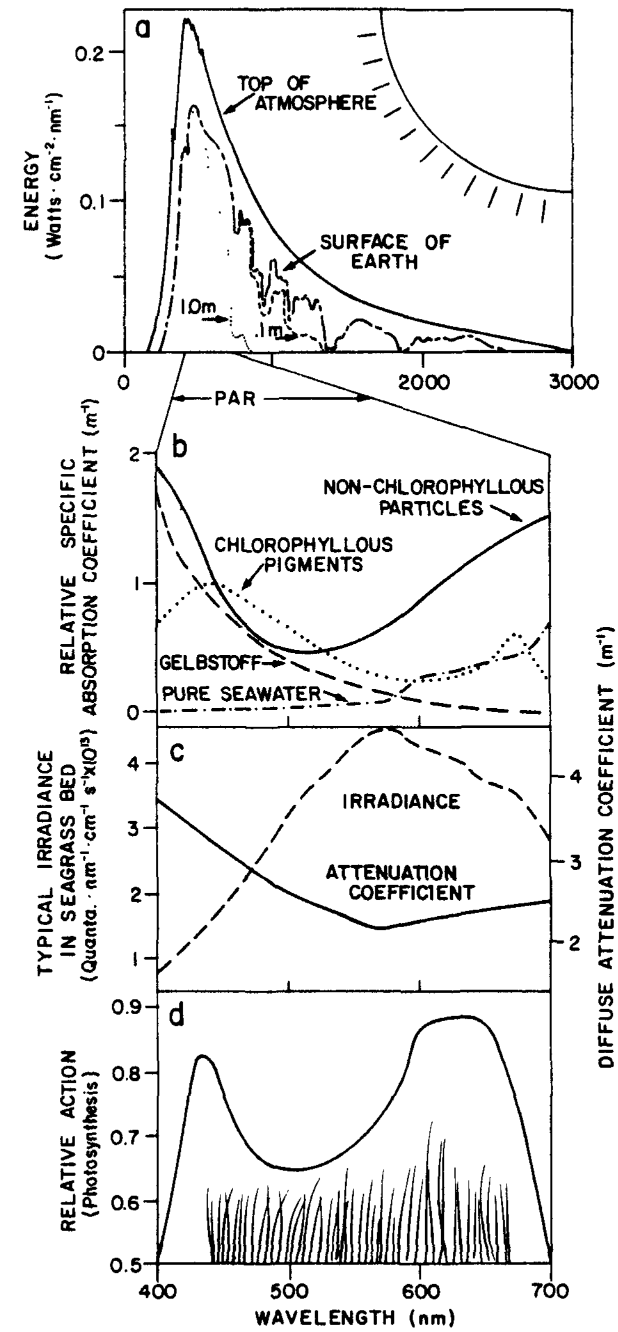

Figure 1. Theoretical path of light from top of atmosphere to benthic estuarine macrophytes. (a) Spectral energy distribution of light at top of atmosphere, at the surface of the earth, and at two depths in the ocean on a clear day (redrawn from Jerlov 1976 and Gates 1971). (b) Relative spectral absorption of various constituents of estuarine waters (redrawn from Prieur and Sathyendranath 1981). (c) Typical spectral irradiance and attenuation in a Chesapeake Bay seagrass bed (Wetzel et al. 1981). (d) Mean quantum action spectrum for higher plants. 1.0 represents the highest photosynthetic response observed by Inada in an individual species (redrawn from Inada 1976).

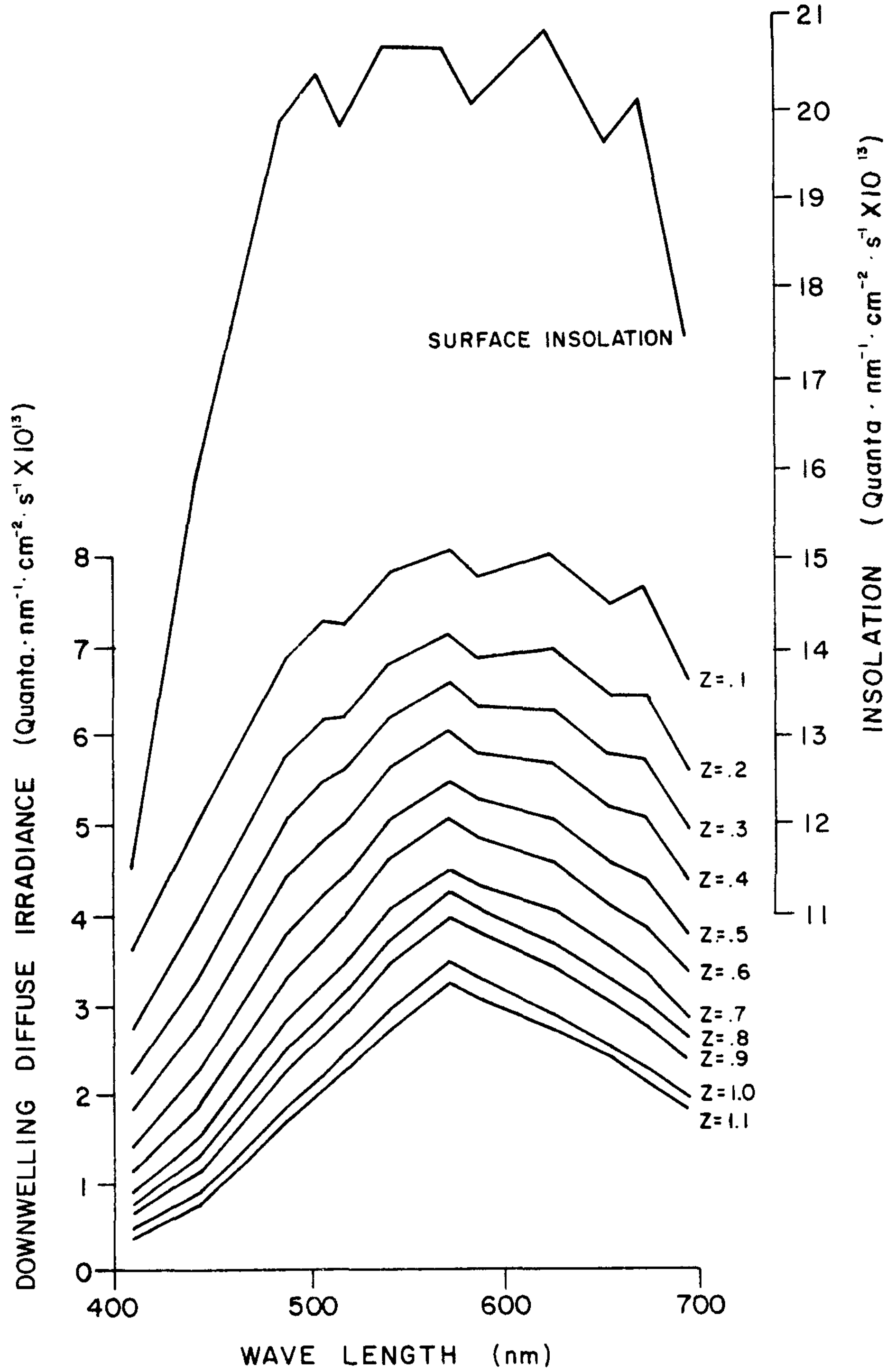

Figure 2. Downwelling spectral quanta irradiance at the surface and at several depths (Z) above the canopy of a Zostera marina bed on the eastern shore of lower Chesapeake Bay (Vaucluse Shores) at 1230 E.S.T. on a cloudy April day. The scale for the insolation is on the right (Wetzel et al. 1982).

constituents cause specific spectral attenuation patterns. As these constituents change both temporally and spatially, the resultant spectral absorption pattern changes. Prieur and Sathyendranath (1981) have attempted to classify water bodies based on combinations of these factors.

The diffuse downwelling (or vertical) attenuation coefficient[2] (K_d) expresses the decay of irradiance as an exponential function,

$$k_d = \frac{-\ln \frac{E_2}{E_1}}{(z_2 - z_1)}$$

where E_2 is the irradiance at depth Z_2; E_1 is the irradiance at depth Z_1; and $(Z_2 - Z_1)$ is the distance between the two measurement depths in meters. The units of K_d are m^{-1}.

If $(Z_2 - Z_1)$ brackets the air-water interface, it will include the effects of reflection and inflate the estimate of K_d. K_d calculated between depths measures the effects of inherent properties of the layer of water on the propagation of light through that distance. Because this distinction is not always specified in the literature, it is sometimes difficult to compare attenuation values. The well-defined spectral attenuation coefficient (K_d or λ) is a particularly useful parameter for comparing underwater irradiance between water bodies, seasons, and wavelengths. Because K_d varies with depth in shallow water (10 m), comparisons should be made at the same depths. Figure 1c shows a typical spectral distribution of both E_d and K_d over the PAR in a Chesapeake Bay grass-bed. The distribution is a result of the additive effects of the attenuations and scattering of seawater, dissolved substances, non-chlorophyllous particles, and phytoplankton (see Figure 1b). Pierce et al. (1981) determined, by step-wise multiple linear regression, that chlorophylls a and c and inorganic particles explain most of the observed variation in spectral attenuation in the Rhode River Estuary (upper Chesapeake Bay).

The diffuse attenuation coefficient (K_d) and the total attenuation coefficient (α) derived from the beam transmissometer measure two different properties with no simple relation. Calculation of α is based on a spectrally-defined and emission-controlled collimated light source that is designed to eliminate diffuse (scattered) light. K_d, however, is based on the natural diffuse submarine light field. Secchi disk readings (D_s) are actually attempts to measure K_d. According to Idso and Gilbert (1974), the relationship

$$k = \frac{1.7}{D_s}$$

is valid for depths between 1.9 and 35.0 meters.

The light energy reaching the benthic plants of an estuary is usually reduced in both the blue and red portions of the spectrum, exactly those

[2] Often incorrectly termed extinction coefficient.

portions to which higher plants such as seagrasses respond the most efficiently. The mean quantum action spectrum for 50 species of higher plants is presented in Figure 1d (Inada 1976). A photosynthetic action spectrum is produced by exposing a plant to controlled amounts of energy (or quanta) at discrete wavelengths and by measuring its photosynthetic response. The action spectrum in this figure is normalized to the highest observed photosynthetic rates for red light. The curve presented here is an approximation of the likely action spectrum for seagrasses. A major peak falls in the 400-500 nm (blue) range, a region in estuarine waters where very little light is available because of absorption by inorganic particles, phytoplankton, and Gelbstoff.

Temporal variations in light distribution, both in the atmosphere and underwater, are due directly and indirectly to the relative motions of the earth, moon, and sun. The distance between the earth and sun and between the earth and moon determines not only the amount of energy received by the earth, but also the depth of water through which it must travel to reach the seagrasses. The seasonal distribution of nutrients and the resultant plankton blooms and runoff (with particulate and dissolved loads and changed salinity regimes) also cause temporal variations in estuarine underwater optical properties. Storms and wind increase land runoff, currents, and waves. In shallow areas, this action increases resuspension. Scott (1978) found that it took 11 days for the submarine irradiance to return to pre-storm levels in an estuary in Australia. In littoral regions, average submarine light conditions may be partly controlled by the interaction of the local coastal morphology with prevailing wind patterns.

Diurnal variations have two components: solar elevation and tidal variation (amplitude and frequency). Since the interface between water and air is a boundary between media of different optical densities, an electromagnetic wave striking it splits into a reflected and a refracted wave. Reflection of combined sun and skylight from a horizontal, flat surface varies asymptotically with solar elevation between three to six percent at angles greater than 30° from the horizon. Below 30°, the reflectance increases dramatically up to 40 percent at 5°. Reflection below 30° is wavelength dependent. The longer waves are reflected more because the changing quantity of diffuse atmospheric light at low sun angles (Sauberer and Ruttner 1941). Wave action, on the other hand, reduces reflection at low angles.

Tidal cycles in estuaries not only change water bodies and their associated seston and dissolved components, but also cause resuspension of sediments and differences in depth. These are, of course, highly idiosyncratic for specific systems (Burt 1955b, Scott 1978).

LIGHT ATTENUATION IN CHESAPEAKE BAY

A comparison of diffuse downwelling spectral attenuation coefficients reported for Chesapeake Bay and its tributaries is presented in Figure 3 along with Jerlov's (1976) most turbid coastal water classification curve (Type 9). For Chesapeake Bay, the earliest measurements of $k_d(\lambda)$ were made by Hurlburt (1945) (Figure 3a). His values fall in the lower range of more recent _in situ_ measurements. The shaded areas in Figure 3a represent the range of values measured by Wetzel et al. (1982) from March through

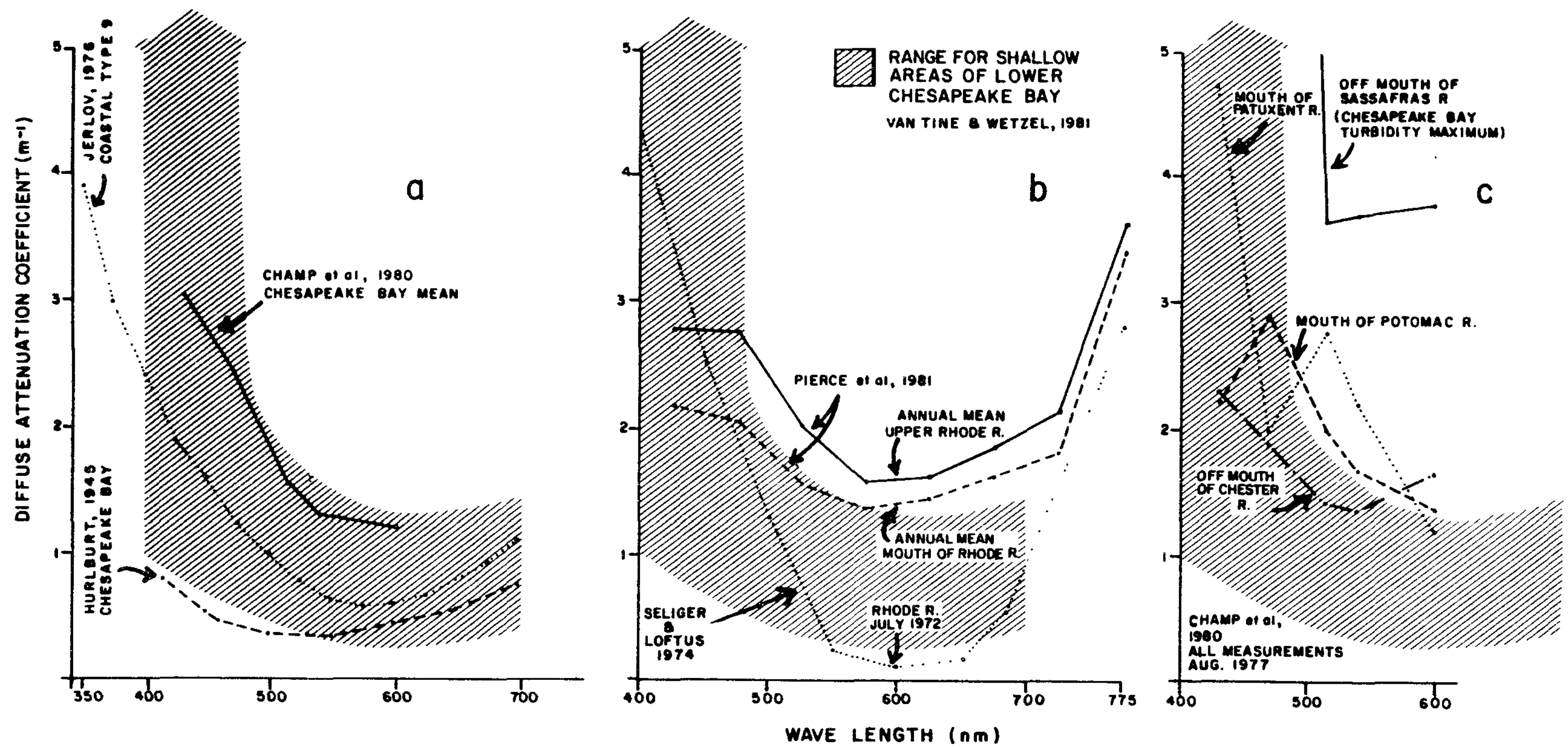

Figure 3. Comparison of diffuse downwelling spectral attenuation coefficients for Chesapeake Bay with Jerlov's most turbid coastal classification. (a) Plotted from tables in Jerlov 1976; Hurlburt 1945; Champ et al. 1980 (b) Plotted from table in Pierce et al., 1981 and calculated from 4π irradiance curves in Seliger and Loftus 1974 (c) Plotted from tables in Champ et al., 1980.

July, 1981, in shallow regions of the lower Bay (<3 m). Jerlov's curve falls in these observed ranges, showing that the data fall within the range of the most turbid coastal waters. Champ et al. (1980) conducted a light characterization survey of Chesapeake Bay during August, 1977. Their mean values are shown in Figure 3a along with their specific site measurements in and near the mouths of the Sassafrass, Patuxent, Potomac and Chester Rivers in Figure 3c. Their mean values fall within the upper ranges measured in the lower Bay (Wetzel et al. 1982).

Pierce et al. (1981) intensively monitored the Rhode River during 1980 and 1981. Their annual mean attenuation values for an upriver station and one at the mouth are plotted in Figure 3b. The upriver station was found to be consistently more turbid, presumably because of its proximity to autocthonous sources. Attenuation at both stations was higher for green, yellow, and red wavelengths than observed in the lower Bay; however, attenuations in the shorter wavelengths were in the same range. Maximum penetration was at 575 nm and minima at 775 and 425 nm. Lower Bay maxima were similar, and minimum measured was at 410 (775 was not measured). Seliger and Loftus (1974) derived curves from 4π irradiance measurements in the Rhode River that generally agree with the measurements of Pierce et al. (1981), except in region 500 to 700 nm. Their measures fall within the observations made for the lower Bay (Wetzel et al. 1982). The differences noted in the 500 to 700 nm range may be due to upwelling irradiance measured by the spherical collector.

Results of the August, 1977, survey by Champ et al. (1980) are shown in Figure 3c. Their attenuation measurements in the turbidity maximum zone at the mouth of the Sassafras River are the highest reported for the Bay. As noted, there is nearly no available light below 500 to 600 nm. Wetzel et al. (1982) observed similar, very high attenuations in the blue region (400 to 500 nm) at lower-Bay sites during a spring runoff event following a major rain storm. The attenuation of green wavelengths (~500 to 550 nm) in the summer was much higher at the mouths of the Patuxent and Potomac Rivers (upper Bay) than at the mouths of the York, Severn, and Ware Rivers (lower Bay). Figure 4 illustrates the lower Bay sampling stations.

A summary of the recent Chesapeake Bay data on diffuse downwelling 2π irradiance attenuation coefficients indicates a severe attenuation of light energy in the photosynthetically important (400 to 500 nm blue, and 700 to 775 nm near infrared) regions of the spectrum. Attenuation in the short wavelengths was particularly marked in the turbidity maximum region of the Bay at the mouth of the Sassafras River, and at the mouth of the Patuxent River during August (Champ et al. 1980) and at lower-Bay sites during spring runoffs (Figure 5). The mean Bay attenuation coefficients calculated by Champ et al. (1980) are about 1.0 m^{-1} higher than Jerlov's (1976) most turbid coastal water classification.

Comparison of Light Attenuation in Vegetated and Unvegetated Sites of the Bay

An analysis of the spectral attenuation coefficients at shallow sites in the lower Chesapeake was undertaken to determine if correlations existed between the presence or absence of benthic macrophytes (*Zostera marina* and *Ruppia maritima*) and specific spectral patterns (Wetzel et al. 1982). The specific question, what are the light quality differences between vegetated

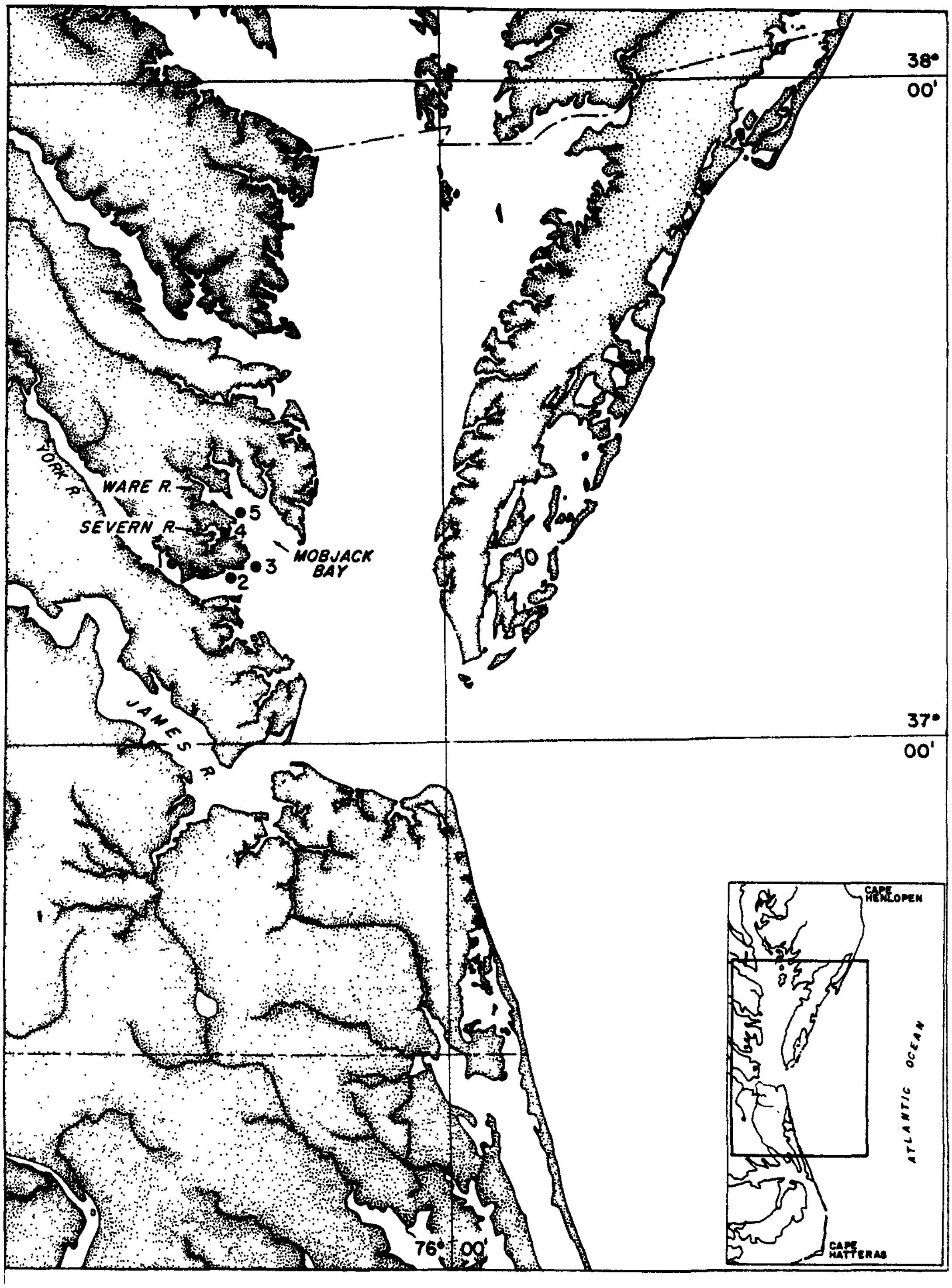

Figure 4. Locations of lower Bay stations (Wetzel et al. 1982).
(1) Mumfort Is., York R. (2) Allen's Is., York R.
(3) Guinea Marshes (4) Mouth of Severn R., Mobjack Bay
(5) Four Point Marsh, Ware R. Mobjack Bay.

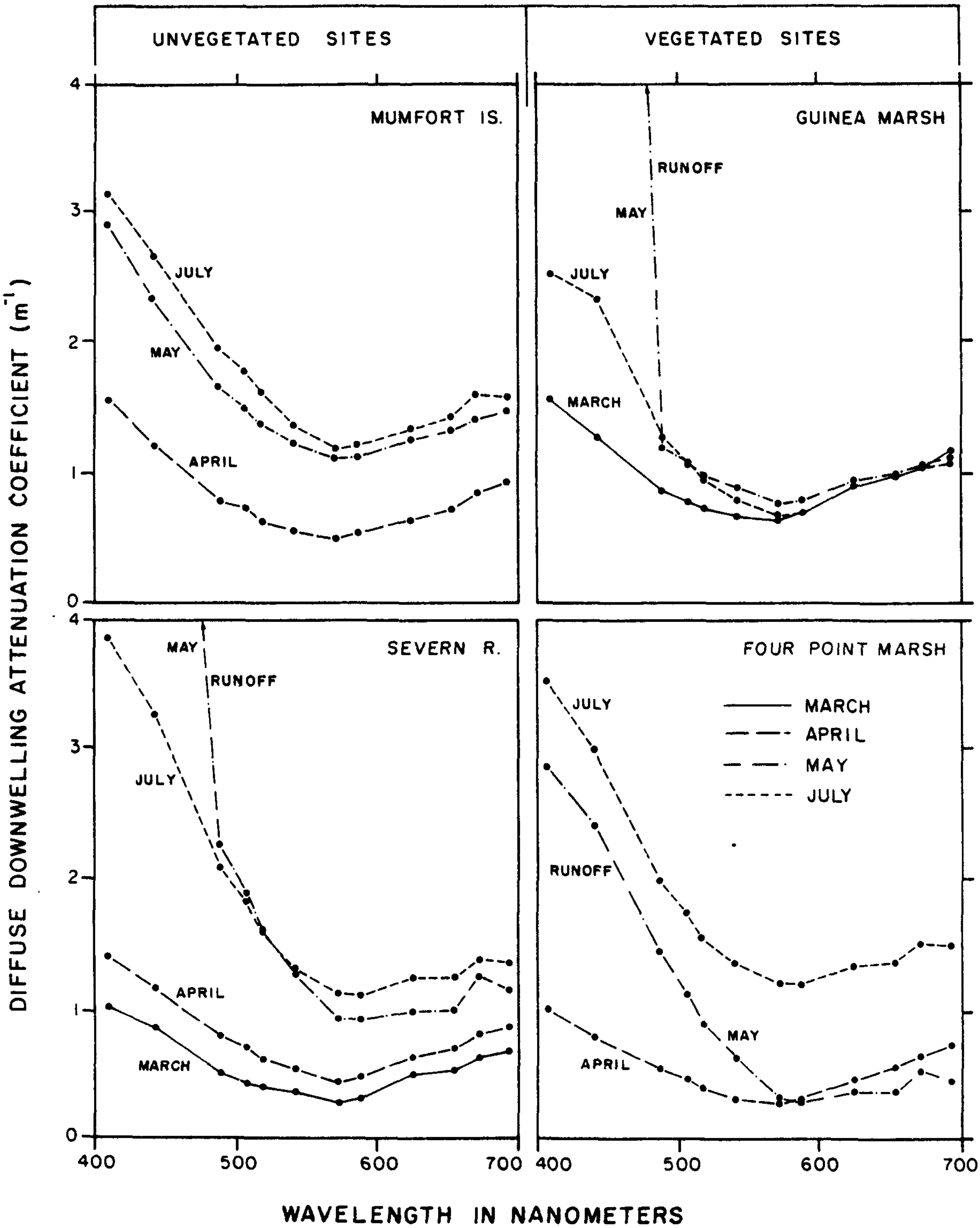

Figure 5. Mean monthly diffuse downwelling spectral attenuation coefficients for vegetated and unvegetated sites in the lower Chesapeake Bay. All coefficients calculated for the depth interval 0.1 to 0.5 m. Mumfort Island (York River) and Severn River sites: unvegetated. Guinea Marsh and Four Point Marsh (Ware River) sites: vegetated (from Wetzel et al. 1982).

and unvegetated sites, was addressed. The sites (Figure 4) were chosen because of their varied vegetational histories (Orth et al. 1981). The Mumfort Island (York River: Station 1) and Severn River (Station 4) sites are presently unvegetated. The Guinea Marsh (Station 3) and Four Point Marsh (Ware River: Station 5) sites have seagrass beds. Both the Severn River and Four Point Marsh sites are affected by agricultural runoff (C. Hershner, personal communication). The Allen's Island site, Station 2, is presently unvegetated, but has recently been replanted by Orth and associates. Twelve wavelengths (410, 441, 488, 507, 520, 540, 570, 589, 625, 656, 671, 694 nm $\pm$ 5 nm) and total PAR were analyzed at depths of 0.1 and 0.5 m. Downwelling irradiance (E_d) was measured as Quanta nm^{-1} cm^{-2} sec^{-1}, each reading representing the mean of 250 scans. Diffuse downwelling spectral attenuation was calculated between 0.1 and 0.5 m.

The mean spectral attenuation values ranged from about 0.2 to 9.0 m^{-1}. Integrated PAR attenuation varied from about 0.5 to 1.6 m^{-1} (Figure 6). A clear seasonal pattern of extreme attenuation of blue wavelengths was evident at all sites beginning in May. This was probably due to a combination of increased particulates associated with runoff events and seasonal plankton blooms.

Mean PAR attenuation coefficients were found to be significantly lower (mean difference of 0.47 m^{-1}) in vegetated than in unvegetated sites during May, 1981 (Figure 6). This was due to a lower attenuation in the 500 to 700 nm region of the spectrum at vegetated sites (Figure 5), despite the effects of high blue attenuation due to runoff. A significant difference among sites based on PAR attenuation coefficients was also observed in July; however, one vegetated site (Four Point Marsh) was grouped with the unvegetated sites having higher attenuation (Figure 6). This was due to the increased attenuation of wavelengths above 500 nm at the Four Point Marsh site during July. The only general light quality differences between vegetated and unvegetated sites that was evident from these analyses were the reduced attenuation in the 500 to 700 nm region at vegetated sites during May.[3]

Kaumeyer et al. (1981) measured a significant difference in PAR attenuation coefficient inside and outside SAV beds at Todds Cove, Md. during July, August, and September, 1980. The vegetated areas were from 0.4 m^{-1} to approximately 2.0 m^{-1} lower. Significant differences were not found in attenuation inside and outside grassbeds at the Parson Island study site. Table 1 summarizes the results of their studies.

Historical Data Bases and Optical Properties of Chesapeake Bay Waters

Most of the historical light data for Chesapeake Bay has been collected by Secchi disc. This method is not ideal, but can be used to indicate trends. Heinle et al. (1980) reviewed Secchi disc light data for both mid-Bay and the Patuxent River, which was chosen because of the extensive data base (Figure 7). Transparency has decreased since the 1930's,

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[3] Subsequent measurements and analyses extend and corroborate this conclusion. Not only is the mean violet and blue attenuation lower in vegetated sites but the variation is also less (see Wetzel et al 1982).

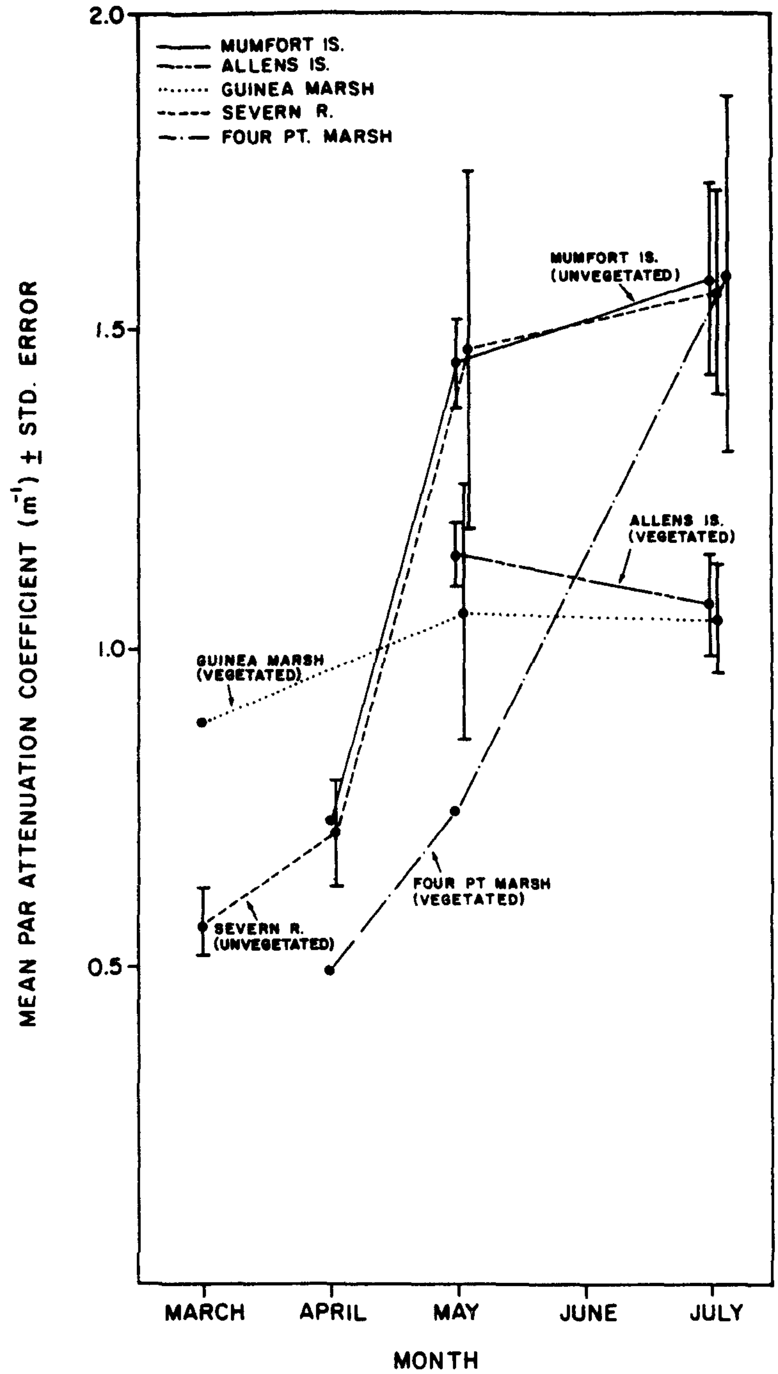


Figure 6. Mean monthly downwelling PAR attenuation coefficient ± 1 standard error of the mean for vegetated and unvegetated sites in the lower Chesapeake Bay (from Wetzel et al. 1982).

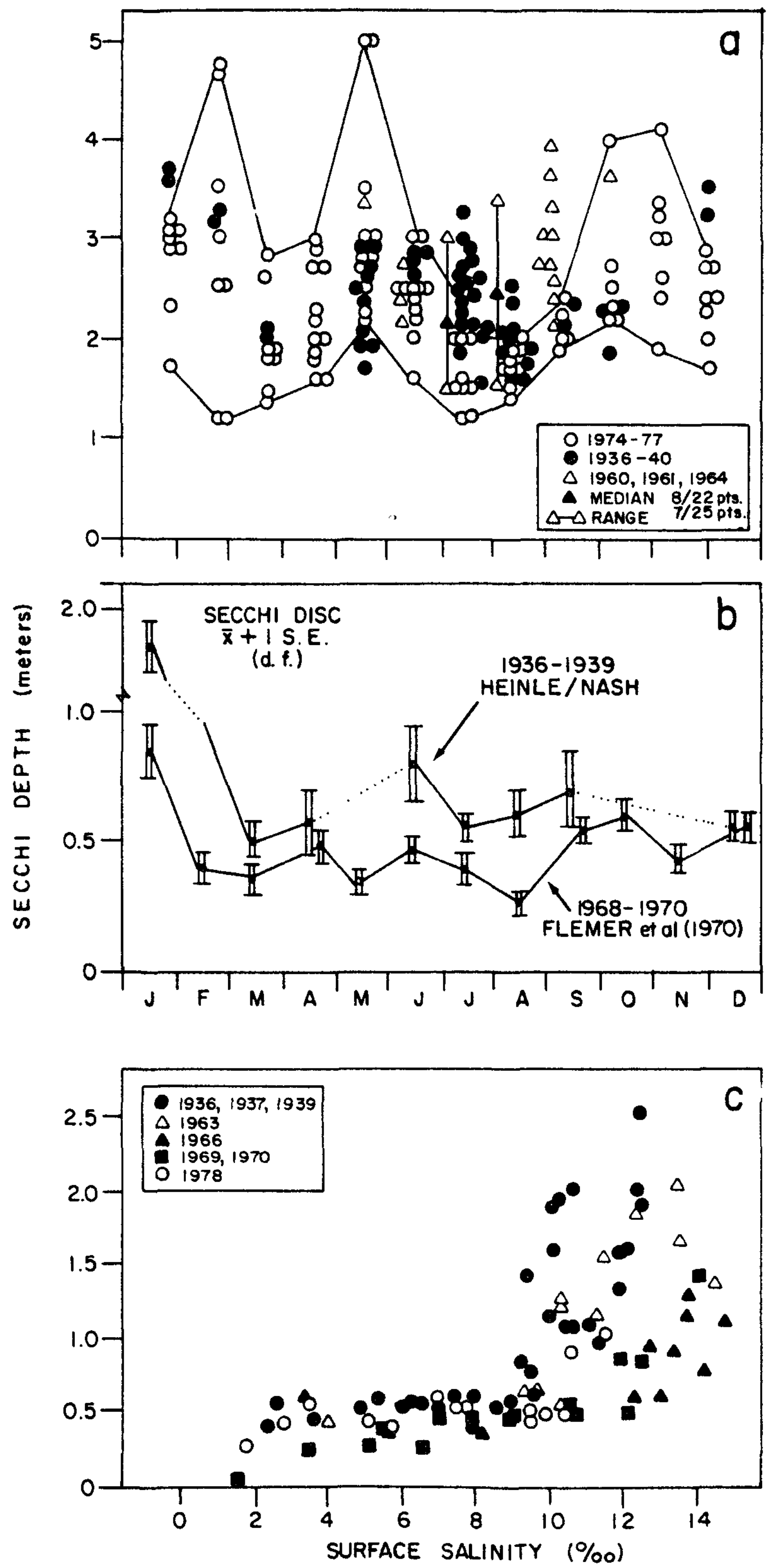


Figure 7. Historical Chesapeake Bay Secchi disc values (from Heinle et al. 1980, and references therein). (a) monthly mid-Bay means. (b) monthly means Patuxent River estuary (from Mihursky and Boynton 1978). (c) Patuxent River Secchi depth versus salinity, July.

Table 1. COMPARISON OF MEAN PAR ATTENUATION COEFFICIENTS INSIDE AND OUTSIDE OF VEGETATED AREAS AT TODDS COVE, MD., 1980 (KAUMEYER ET AL. 1981)

Month	Location	$K_{PAR}(m^{-1})$
June	SAV	2.6 ± 0.20
	Reference Site	2.5 ± 0.75
July	SAV	2.5 ± 0.30
	Reference Site	2.9 ± 0.70
August	SAV	1.8 ± 0.56
	Reference Site	3.1 ± 0.33
September	SAV	1.9 ± 0.34
	Reference Site	3.8 ± 0.96

especially during the winter in the mid-Bay region (Figure 7a). An increase in turbidity, as estimated by Secchi disc measures, has been quite dramatic in the Patuxent (Figures 7b, 7c). Mid-1970's Secchi disc data for rivers in the upper Chesapeake Bay are reported in Table 2 from Stevenson and Confer (1978). The values are generally low ($\leq$1.0 m) and are similar to those reported for the Patuxent during the 1960's and 1970's (Figures 7b, 7c).

Increases in chlorophyllous pigments, due to phytoplankton blooms caused by increased nutrients, can have a severe effect on light attenuation in the photosynthetically critical blue and red spectral regions (Figures 1b, 1d). Historical chlorophyll data for Chesapeake Bay and Patuxent River are summarized in Figures 8 and 9. Chlorophyll concentrations have increased dramatically in the upper and mid-Bay since the early 1950's. Concentrations as high as 100 to 200 ug L^{-1} were not unusual. In contrast, lower-Bay concentrations have not significantly changed (Figure 8b). Concentrations in the Patuxent River have increased

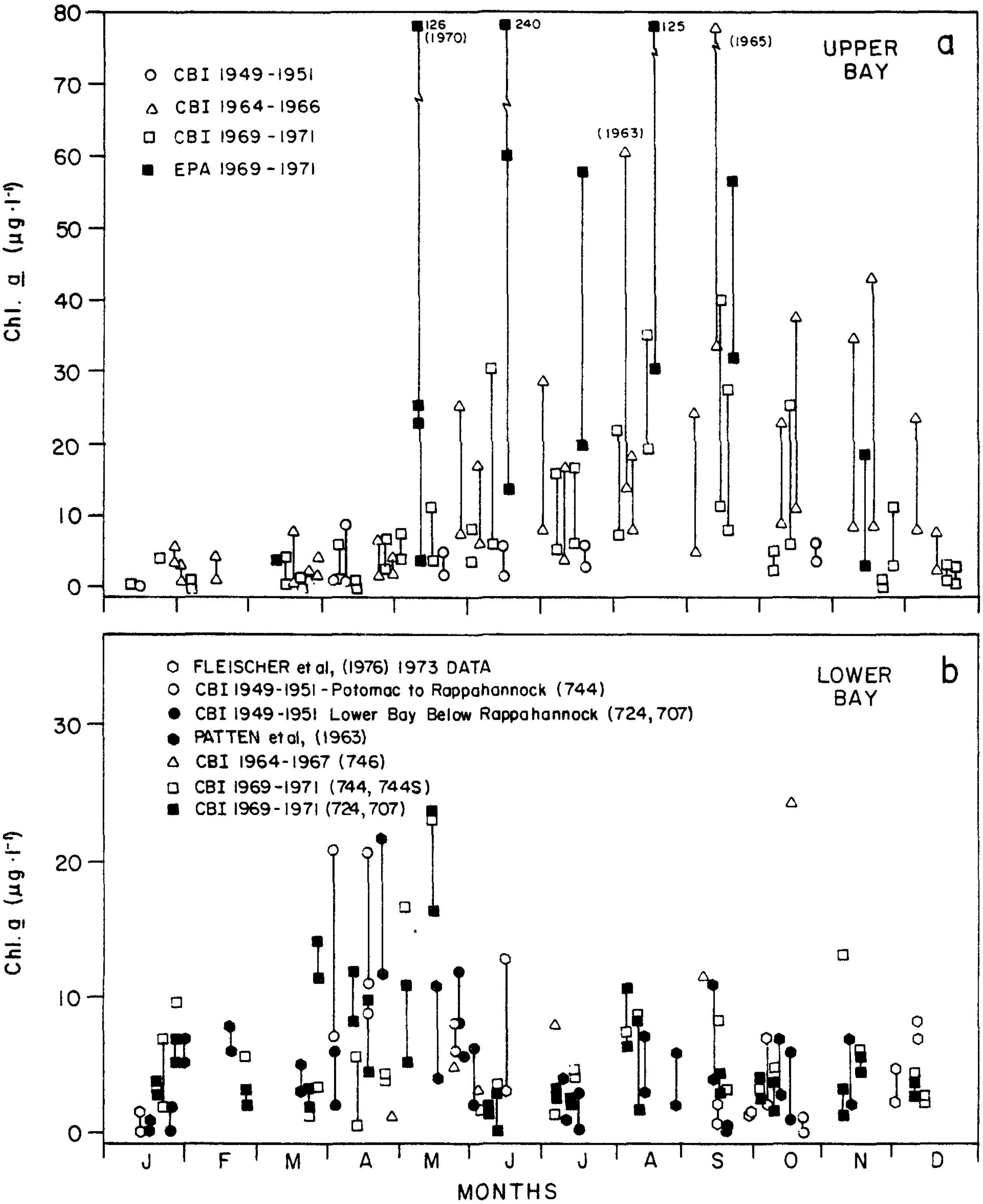


Figure 8. Summary of historical chlorophyll a data for the Chesapeake Bay. (a) upper Bay. (b) lower Bay (Redrawn from Heinle et al. 1980).

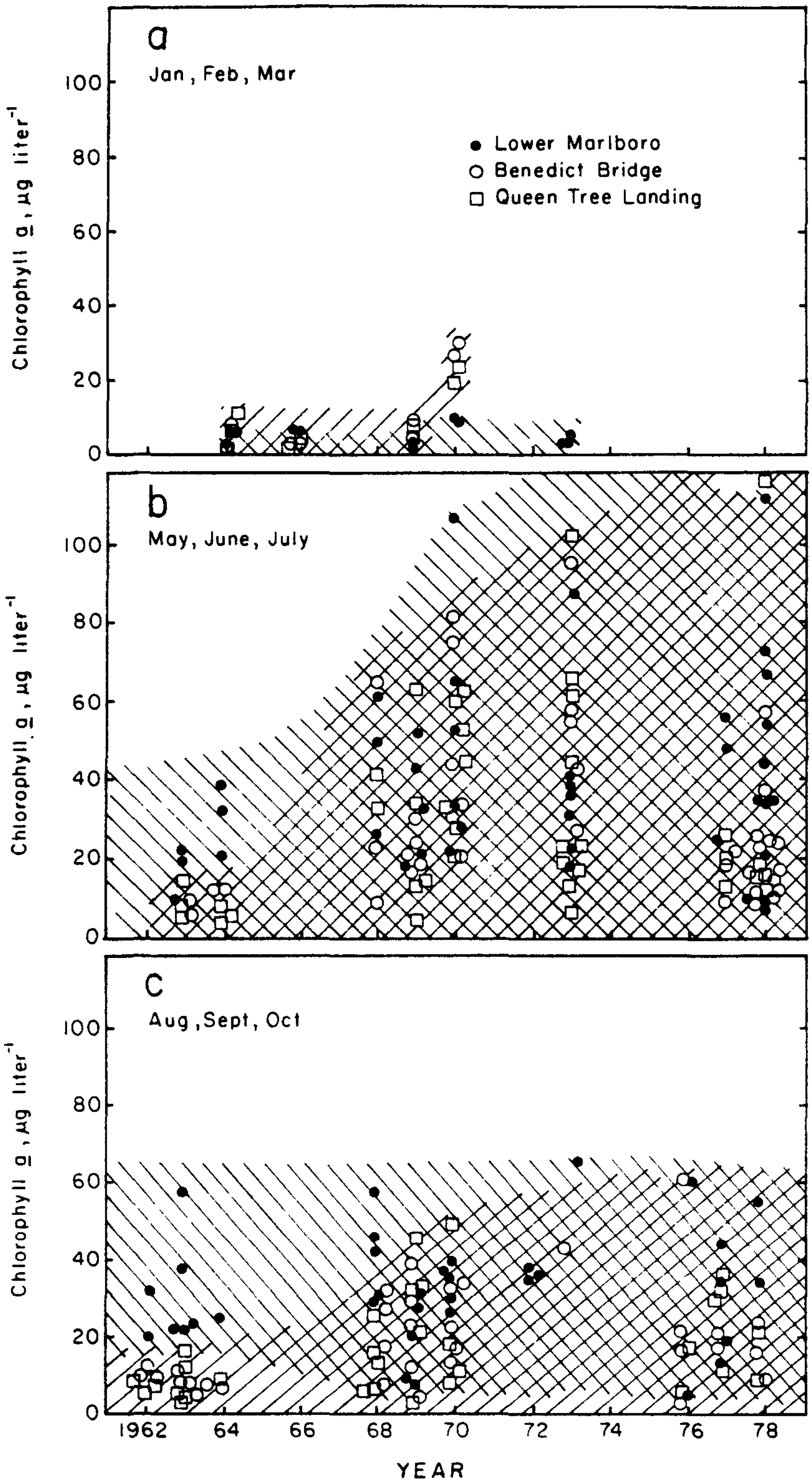


Figure 9. Summary of historical chlorophyll a data for three regions of the surface waters of the Patuxent R., Md. (a) January-March (b) May-July (c) August-October. (Cross hatching is to clarify general trends for each site.) (Redrawn from Heinle et al. 1980).

Table 2. AVERAGE SECCHI DISC DATA (cm) BY RIVER SYSTEM, MARYLAND CHESAPEAKE BAY, 1972-1976a (AS REPORTED IN STEVENSON & CONFER 1978)

River System	1972	1973	1974	1975	1976
Elk and Bohemia Rivers	33.0	35.1	-	25.7	36.3
Sassafras River	34.3	52.3	-	29.2	51.1
Howell and Swan Points	33.8	75.4	-	61.2	57.7
Eastern Bay	67.3	62.5	76.5	54.6	75.9
Choptank River	60.7	62.5	84.3	61.5	64.3
Little Choptank River	64.5	59.4	66.8	63.8	78.5
James Island and Honga River	70.1	64.0	74.2	67.1	73.4
Honga River	78.2	67.3	72.6	68.8	67.8
Bloodsworth Island	73.7	87.6	94.7	177.0	83.3
Susquehanna Flats	64.5	65.5	82.6	33.8	76.5
Fishing Bay	49.5	77.0	85.6	75.7	54.1
Nanticoke and Wicomico Rivers	55.4	58.9	65.8	61.0	58.9
Manokin River	94.2	94.7	101.3	107.4	81.0
Patapsco River	73.7	80.0	67.8	-	70.1
Big and Little Annemessex Rivers	109.7	92.7	96.3	88.1	85.1
Gunpowder and Bush River Headwaters	42.9	38.3	46.7	-	53.8
Pocomoke Sound, Maryland	101.6	82.0	-	96.8	85.9
Magothy River	83.8	97.3	73.4	-	74.4
Severn River	97.3	70.4	79.5	-	86.4
Patuxent River	80.3	80.8	61.5	66.8	62.7

Continued

Table 2. AVERAGE SECCHI DISC DATA (cm) BY RIVER SYSTEM, MARYLAND CHESAPEAKE BAY, 1972-1976a (AS REPORTED IN STEVENSON & CONFER 1978) (CONTINUED)

River System	1972	1973	1974	1975	1976
Back, Middle and Gunpowder Rivers	79.5	75.7	73.2	75.4	61.2
Curtis and Cove Point	45.2	77.0	81.8	58.9	73.7
South, West and Rhode Rivers	74.7	66.0	61.2	48.5	67.1
Chester River	76.2	73.4	100.1	87.9	85.1
Love and Kent Points	89.7	74.7	117.6	72.1	89.9
Smith Island, Maryland	78.5	76.2	89.7	139.4	87.6
Average	70.1	71.1	79.5	76.2	71.4

significantly in both the upper and lower portions (Figure 9), especially during late spring and early summer (Figure 9b). Levels in excess of 100 μgL^{-1} were common in the summer throughout the 1970's -- this is twice the concentration measured during the previous decade.

In addition to the thoroughly documented increased chlorophyll a concentration in the Patuxent, there have also been increases in most of the other tributaries of the Bay. Chlorophyll a concentrations in the Choptank, Chester, and Miles Rivers of the middle eastern shore are 1.5 to

Table 3. RANGES OF CONCENTRATIONS OF CHLOROPHYLL a (ug l^{-1}) AT SURFACE AND BOTTOM DEPTHS IN THE LOWER POTOMAC RIVER DURING 1949-1951, AND 1965-1966 (HEINLE ET AL. 1980)

Month	1949-1951		1965-1966	
	Surface	Bottom	Surface	Bottom
January	1-2	1-2	3.2-4.6	3.1-5.0
March-April	10-21	12-27+	1.1-20.0	1.1-9.5
May	3-6	9-24+	5.8-13.2	4.3-9.8
July	3-5	1-2+	9.0-13.8	1.0-1.8
October-November	1-9+	1-7	9.3-24.0	3.6-11.0

2.0 times higher presently than earliest data show. There have been upstream increases in the Magothy, Severn (Md.), and South Rivers. Concentrations up to 100 uL^{-1} were measured in the upper Potomac in the mid-1960's. Concentrations in the lower Potomac were generally higher in the 1960's than 1950, except in March and April (Heinle et al. 1980). Increased chlorophyll a concentrations have also been measured in the Rappahannock and York Rivers during the last few years. The upper James has had high concentrations similar to the upper Potomac since the

Table 4. ANNUAL MEAN FRESHWATER FLOWS AND OCCURRENCE OF HURRICANES TO ALL OF CHESAPEAKE BAY (CUBIC FEET PER SECOND) FOR 1951-1979 (HEINLE ET AL. 1980).

Year	Bay Annual Average	5-Year Average
1951	82,100	
1952	94.300	
1953	72,800	
1954 Hurricane	58,700	
1955 (2) Hurricanes	73,400	76,260
1956	76,000	
1957	64,400	
1958	81,400	
1959	66,400	
1960	77,300	73,100
1961	78,000	
1962	64,800	
1963	52,400	
1964	61,900	
1965	49,000	61,220
1966	53,300	
1967	77,200	
1968	60,100	
1969	54,900	
1970	77,200	64,540
1971	79,000	
1972 Hurricane	131,800	
1973	95,200	
1974	76,900	
1975	103,100	97,180
1976	84,400	
1977	80,100	
1978	91,300	
1979 Hurricane	113,800	92,400

mid-1960's, but the lower River still does not. Dense algal blooms have been noted in the Elizabeth, Back, and Poquoson Rivers of the lower Bay.

Heinle et al. (1980) summarized the state of the Bay graphically in terms of enrichment that they defined as deviations in concentrations of chlorophyll a from historic, natural periods of stability or steady state concentrations. Figure 10 shows the regions of the Bay that are categorized as moderately or heavily enriched. Many of these areas have experienced declines in Bay grasses on a time scale overlapping the enrichment.

Changes in dissolved organic materials, inorganic particulate matter, and allochthonous organic particulate matter in the Bay are mainly determined by inputs (runoff) of freshwater to the tributaries and by additional input due to storm events. Table 4 summarizes annual mean freshwater flow to the entire Bay and major storms during the period 1951-1979. In addition to adding large amounts of sediment to the water column, major storm events increase nutrient loads that favor phytoplankton blooms.

Suspended sediment transport and discharge of the Susquehanna River, the major source of freshwater to the Bay, are given in Table 5.

Table 5. SUSPENDED SEDIMENT TRANSPORT AND DISCHARGES OF SUSQUEHANNA RIVER (GROSS ET AL. 1978)

	Annual suspended sediment discharged (millions of metric tons per year)	
Calendar Year	Above Dam	Below Dam
1966	1.5	0.7 (60%)*
1967	1.7	0.3**
1968	1.7**	nd
1969	nd	0.32 (60%)*
1970	2.0	1.1**
1971	1.4**	1.0
1972	11.3	33
Agnes, 24-30 June 1972	7.6	30
1973	3.2	1.2 (54%)*
1974	1.7	0.8 (53%)*
1975	3.8	11
Eloise, 26-30 Sept. 1975	1.6	9.9
1976	nd	1.2

nd = no data
* Percent discharged during annual spring flood
** Records incomplete for the year

Gross et al. (1978) suggest that one-half to two-thirds of the suspended sediment discharge of the Susquehanna is deposited behind the dams or in the lower reaches of the river during years of low flow and no major flooding. During major floods, however, these deposits are eroded and transported into the Bay. Thus, dams effectively increase the amount and variability of sediment discharged under flood conditions.

It is evident that major storms, such as hurricanes, significantly increase freshwater input, but there is also an apparent wet-year, dry-year cycle imposed on the data. The five-year-flow averages (Table 4) suggest a mid-1960's depression followed by an increase through the 1970's. Although these data have not been rigorously analyzed, it is apparent that long-term changes and/or cycles in climatic conditions (rainfall, temperature, and

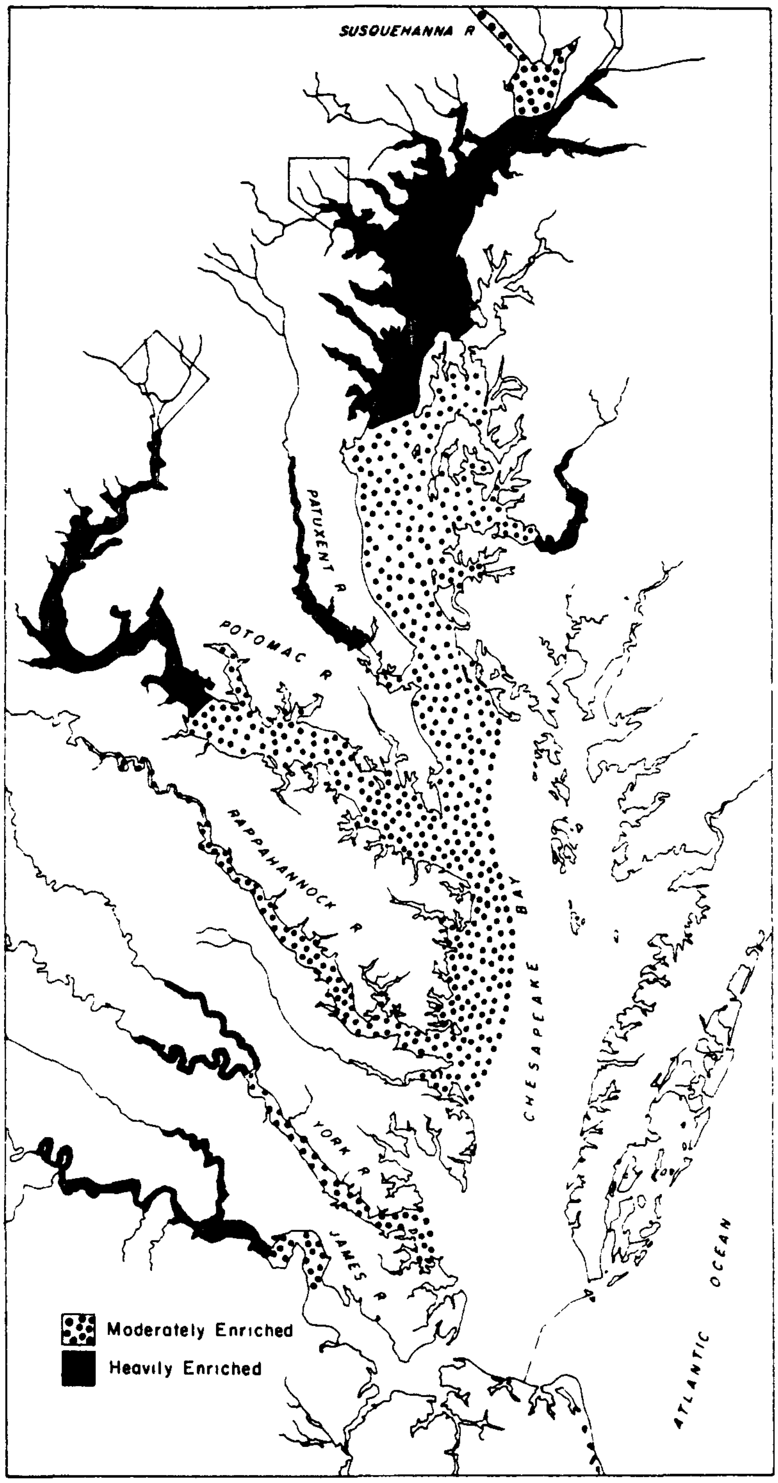


Figure 10. Portions of the Chesapeake Bay considered enriched by Heinle et al. 1980. Enrichment is defined as increase in chlorophyll a levels from historic, natural periods of stability.

major storms) influence water quality and optical properties of Bay waters. However, cause and effect relations are still poorly understood and resultant optical properties of Bay water are determined and controlled by multiple influences: runoff; nutrients; suspended particulates (both living and dead); and, as the principal driving forces, the general climatic regime.

SECTION 3

LIGHT AND PHOTOSYNTHESIS IN CHESAPEAKE BAY SAV COMMUNITIES

GENERAL REVIEW OF PHOTOSYNTHESIS

Photosynthesis is the process in which light is used as the energy source for the synthesis of organic compounds. Three basic steps are involved in the process: (1) absorption of light energy by photosynthetic pigments; (2) processing the captured light energy to produce the compounds ATP and NADPH; and (3) the reduction of CO_2 using ATP and NADPH and the production of carbohydrates. The first two steps are light-dependent and are collectively referred to as the "light reaction". The third step is light-independent and termed the "dark reaction".

The photosynthetic pigments have characteristic light energy absorption spectra in the photosynthetically active region, 400 to 720 nm. Chlorophyll *a* absorbs light more effectively at higher wavelengths (>600 nm); accessory pigments such as chlorophyll *b*, carotenoids, and others are more effective at shorter wavelengths (<600 nm). Chlorophyll *a* and the accessory pigments absorb and transfer light energy at varying efficiencies to specialized chlorophyll *a* molecules (P700) where they are used directly for biochemical reactions.

The photochemical reactions are driven by units of light energy called photons (quantum energy). The quantum energy is a function of wavelength; quanta of shorter wavelengths contain more energy than quanta of longer wavelengths. Light energy transferred to P700 is most efficient as it is used directly in the photosynthetic system; light energy transfer by chlorophyll *a* and accessory pigments is less efficient. The quantum yield, the moles of O_2 produced or CO_2 fixed per photon of light absorbed, is used to estimate the transfer efficiency.

The light utilization spectra of a particular species is called the action spectra, a characteristic curve obtained by combining the light absorption spectra and the quantum yield of intact plant cells. The action spectra is an important feature because it reflects the ability of a species to adapt to various light spectral regimes (Figure 1d). This is of particular importance when considering photosynthesis of submerged plants. In aquatic environments, spectral shifts in light energy result from the water itself, suspended organic and inorganic material, dissolved organic compounds, and other water column constituents (discussed in Section 2).

A general approach to the investigation of photosynthesis is to construct light saturation curves for various species (Figure 11a). An examination of photosynthesis-light curves (P-I curves) shows that photosynthesis (P) increases with increasing light to a point of optimal irradiance (I_{opt}) where, over a range of irradiance, the photosynthetic system is saturated and maximum photosynthesis (P_{max}) occurs. At higher irradiance, there may be a depression in the photosynthetic rate, termed photoinhibition. The initial slope of the curve ($\Delta P/\Delta I$ or α) and P_{max} are the two major parameters used in describing P-I curves (Jassby and Platt 1976). Alpha (α) is a function of the light reaction of photosynthesis and is an estimator of the quantum yield. P_{max} is a function of the dark reaction and is influenced by environmental factors or the physiological state of the plants (Parsons et al. 1977). The term I_k, proposed by Talling (1957), is the irradiance at which a linear extension of the initial slope intercepts P_{max}. I_k is regarded as indicative of the plant's adaptation to its light regime (Steeman-Nielsen 1975). I_k is irradiance where $P = 0.5\ P_{max}$ and is similar to the Michalis-Menten half-saturation constant. I_c is the irradiance at the compensation point, where photosynthesis equals respiration ($P = R$).

Characteristic P-I curves are shown in Figure 11b. Plants adapted to high and low light environments, termed sun and shade species, exhibit different P-I curves. Sun species (curve 3) generally exhibit higher P_{max} values than shade species, which exhibit greater and lower I_c values (curves 1 and 2). In the aquatic environment, with reduced availability of light, species exhibiting shade-type photosynthesis (greater photosynthetic rates at low light intensities) are at an advantage.

PHOTOSYNTHESIS OF SUBMERGED VASCULAR PLANTS IN RELATION TO LIGHT AND TEMPERATURE

In situ studies of submerged angiosperms point to the important role of light in seagrass production and distribution (Jacobs 1979, Mukai et al. 1980). In a study of *Zostera* in Denmark, Sand-Jensen (1977) showed a positive correlation between leaf production and insolation over a nine month period. Biomass and photosynthesis rates of *Posidonia* declined with depth near Malta (Drew and Jupp 1976); this was probably due to decreased light penetration with depth. In before and after studies of an estuary that was closed to the sea, Neinhuis and DeBree (1977) report that the *Zostera* population increases in density and extends to a greater depth; they suggest that this is probably due to an increase in water transparency.

In situ light manipulation experiments provided evidence of the importance of light to seagrass production. For example, at the end of a nine-month study during which ambient light was reduced by 63 percent, *in situ* *Zostera* densities were only five percent of that of the control (Backman and Barilotti 1976). In similar studies, Congdon and McComb (1979) report that lower than ambient light levels result in lower *Ruppia* biomass; as shading duration increases, higher light levels are required to sustain a high biomass.

Studies involving the epiphytic community, those organisms directly attached to submerged angiosperm blades, suggest that epiphytes have a detrimental effect because they shade the macrophytes. Both Kiorbe (1980) and Phillips et al. (1978) provide data to indicate that epiphytic development suppresses macrophyte growth. Sand-Jensen (1977) reports that *Zostera* photosynthesis is reduced by up to 31 percent due to a decreased penetration of light and inorganic carbon through the epiphytic community to the seagrass blades. Johnstone (1979) hypothesizes that the rapid linear growth of *Enhalus* leaves (up to two cm day^{-1}) is related to a shading effect from epiphytes. In contrast, the data of Penhale and Smith (1977) suggest that an epiphytic community may be beneficial in certain environments. For *Zostera* exposed at low tide, epiphytes prevent desiccation damage by trapping a film of water, and probably reduce the photoinhibitory effect of high light.

In addition to light, temperature also influences submerged macrophyte distribution and productivity rates (Biebl and McRoy 1971, Drew 1978). The

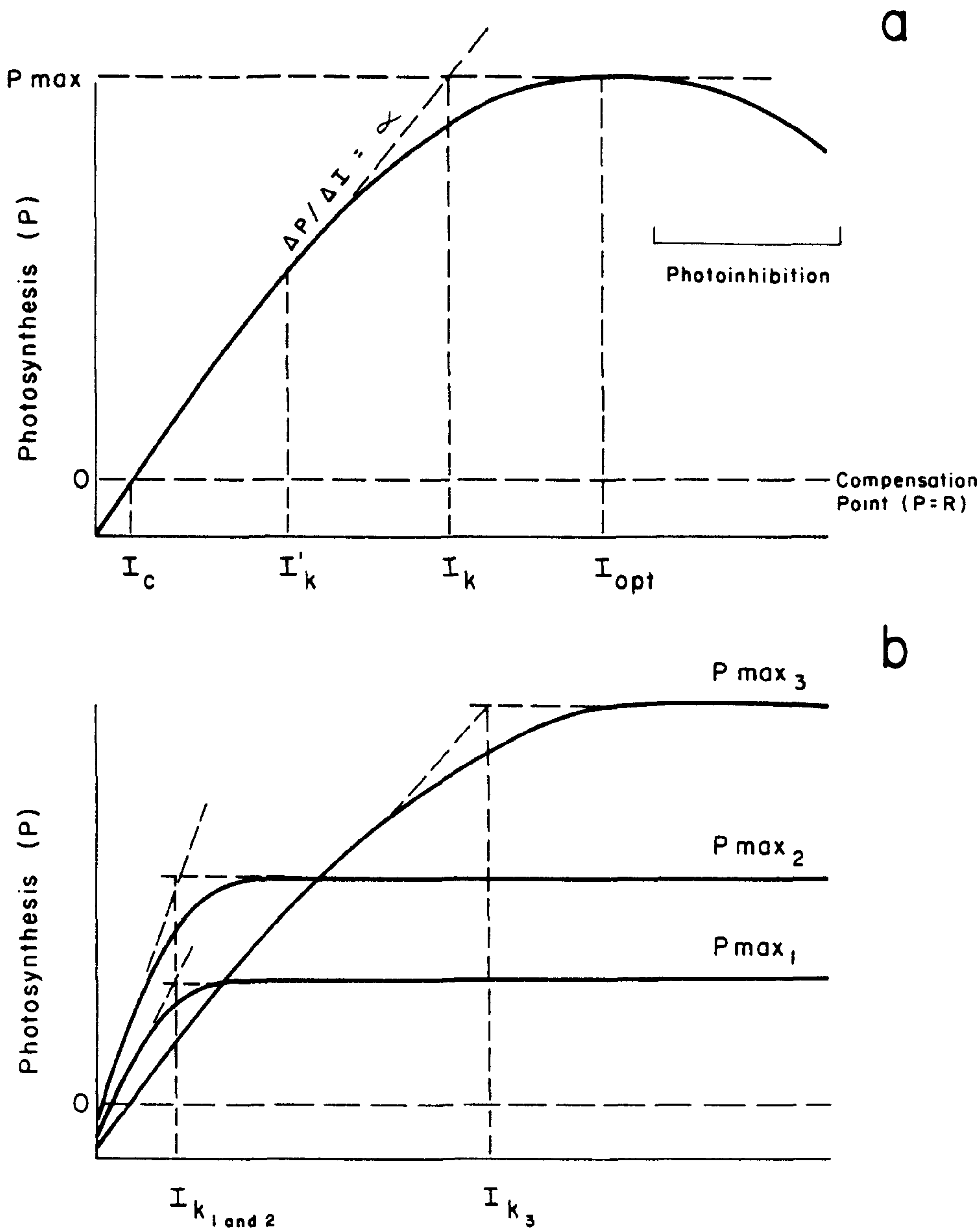


Figure 11. Diagramatic photosynthesis-light relationships. See text for description of parameters.

biogeography of marine and brackish water plants points to a temperature effect on worldwide distribution; for example, genera such as Zostera, Ruppia, Phyllospadix, and Posidonia occur mainly in temperate zones while genera such as Thalassia, Syringodium, and Halophila occur mainly in subtropical and tropical zones. Drew (1979) reports that the P_{max} of four seagrass species collected near Malta increases in direct proportion to temperature, up to temperatures [30 to 35°C (86 to 95°F)] where tissue damage occurred; decreases are not observed at environmental temperatures. In contrast, Penhale (1977) observed a decline in P_{max} from 22 to 29°C (71.6 to 84.4°F) for Zostera in North Carolina where environmental temperatures reach 34°C (93.2°F). The co-existence of species such as Ruppia and Zostera in the lower Chesapeake Bay may be a result of differential responses to both temperature and light, as apparently is the case in a Myriophyllum-Vallisneria association described by Titus and Adams (1979). They report that a greater for temperature tolerance Vallisneria, in conjunction with the temperature dependence of photosynthesis, results in a temporal partitioning of production. Vallisneria apparently favored in midsummer conditions; Myriophyllum spring and fall conditions.

Sun and shade species have been described for submerged macrophytes (Spence and Crystal 1970a, 1970b; Titus and Adams 1979). Sun species generally exhibit higher P_{max} values than shade species that exhibit lower I_c values, and lower dark respiration rates. Certain species can adapt to a wide range of light conditions. Bowes et al. (1977) cultured Hydrilla under high and low irradiances; subjecting the plants to high light increased the I_{opt} value four-fold. Plants grown under low light achieved I_c and I_k at lower intensities.

In seagrass systems, pigment relationships generally vary with light quantity or with position within the leaf canopy. The adaptive capability of seagrass pigment systems to the light environment has been shown in various studies. For example, Wiginton and McMillan (1979) report that the total chlorophyll content is inversely related to light for several Caribbean seagrasses collected at various depths. For seagrasses cultured at several light levels, the total chlorophyll content increased with decreasing quantum flux (McMillan and Phillips 1979, Wiginton and McMillan 1979). Within individual meter-long Zostera leaves, the chlorophyll *a* to chlorophyll *b* ratio varied significantly, with the lowest ratio at the basal portion of the plant (Stirban 1968). In a detailed study of chlorophyll relationships in a Zostera system, Dennison (1979) observed no substantial variation in total chlorophyll content within the leaves as a function of depth of the leaf canopy in integrated samples along a depth gradient within the bed. The chlorophyll *a* to chlorophyll *b* ratio, however, decreased from the apical to basal portion of the leaves.

Although the physiological photosynthesis-light relationship ultimately determines the light levels at which plants grow, the morphology of individual plants and the community canopy structure may play an important role in production and species distribution. In a study of Myriophyllum and Vallisneria, Titus and Adams (1979) observed that the former had 68 percent of its foliage within 30 cm (11.7 inches) of the surface, and the latter had 62 percent of its foliage within 30 cm of the bottom. Myriophyllum, an introduced species, has often displaced the native Vallisneria; a contributing factor is probably the ability of Myriophyllum to shade Vallisneria. In a detailed community structure analysis of a monospecific *Zostera* community across a depth gradient, Dennison (1979) concludes that changing leaf area is a major adaptive mechanism to decreasing light regimes.

PHOTOSYNTHESIS-LIGHT STUDIES IN CHESAPEAKE BAY

Investigations of photosynthesis-light relationships carried out through the Chesapeake Bay Program can be categorized into three general experimental designs. In the first, P-I curves were constructed for the four dominant species in Chesapeake Bay system: *Myriophyllum spicatum* and *Potamogeton perfoliatus* in the upper Bay, and *Zostera marina* and *Ruppia maritima* in the lower Bay. These experiments used whole plants or leaves subjected to various light intensities (created through the use of neutral density screens) and various temperatures.

The second approach used microcosms in which the effects of various concentrations of phytoplankton and suspended solids on light penetration and on *Potamogeton* photosynthesis were determined.

The third experimental design involved *in situ* community metabolism measurements under a wide range of natural light regimes. In certain experiments, neutral density screens were used to shade the community on a short-term basis. The experimental design and methods for each of these studies are detailed in Kemp et al. (1981b) and Wetzel et al. (1982).

P-I Relationship of Major Species

P-I curves were constructed for whole plants of *M. spicatum* and *P. perfoliatus* at 21°C (69.8°F) (Kemp et al. 1981b) (Figure 12). Both species exhibited the characteristic photosynthetic response to light with light saturation occurring between 600 and 800 uE m^{-2} sec^{-1}. *Myriophyllum* exhibited a greater P_{max} and a greater I_k than *Potamogeton*; however, the two species exhibited similar α. Although these species occur in the same general locale, they do not form dense, mixed bed stands where they would be in direct competition for light.

The photosynthetic response to light and temperature was determined for isolated *Z. marina* and *R. maritima* leaves (Wetzel et al. 1982). Since these species co-exist in the lower Chesapeake Bay, an evaluation of photosynthetic parameters of each species might suggest competitive strategies. Experiments carried out at six temperatures and under natural light indicate that light saturation of *Zostera* occurs about 300 uE m^{-2} sec^{-1} while that of *Ruppia* occurs about 700 uE m^{-2} sec^{-1}. Differences in P_{max} between *Zostera* and *Ruppia* were observed and appear related to temperature. At warmer temperatures, *Ruppia* exhibits a higher P_{max} than *Zostera*; the situation is reversed at colder temperatures (Figure 13). A summary of the data shows that *Ruppia* exhibits the greater P_{max} at temperatures greater than 8°C (46.4°F) (Table 6). A comparison between the two species shows that *Zostera* generally exhibits a greater α; this suggests a competitive advantage for *Zostera* at lower light levels.

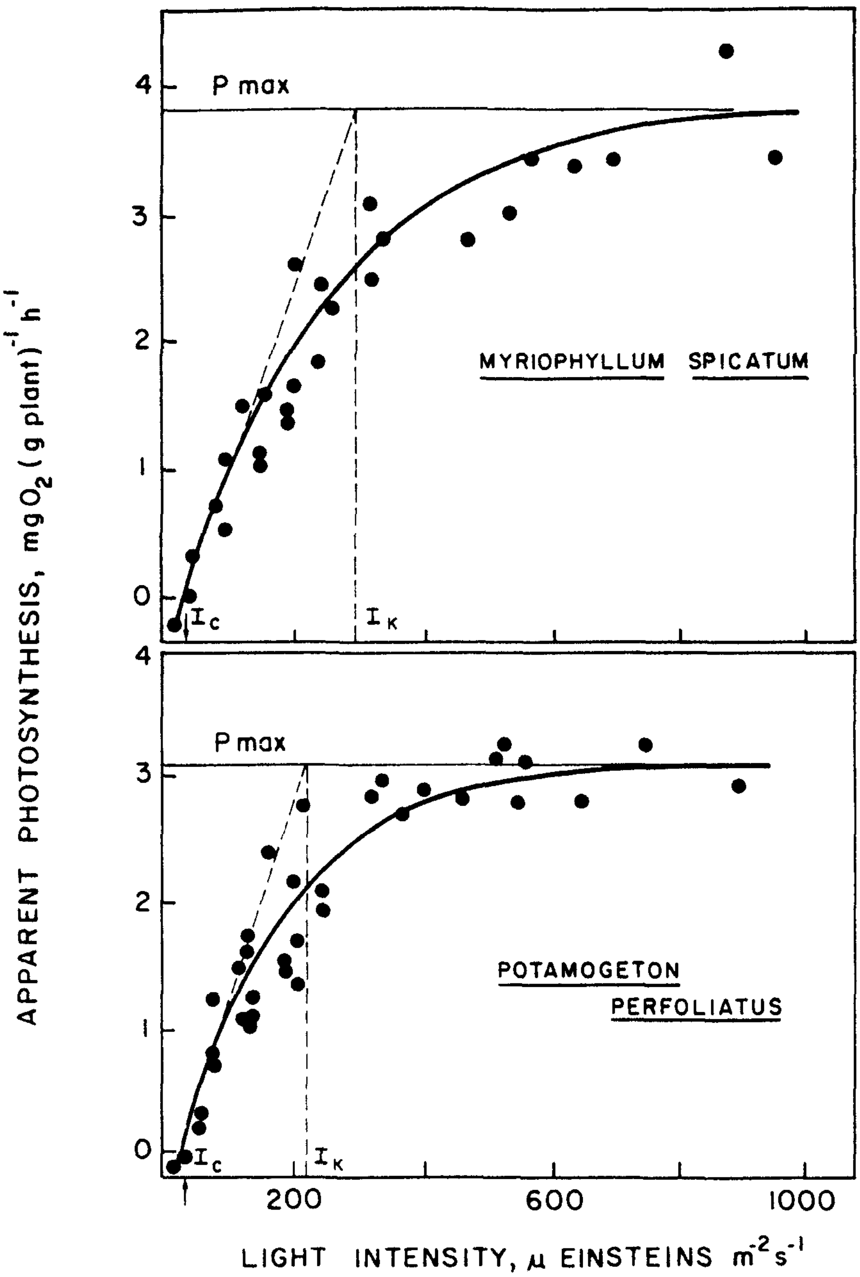
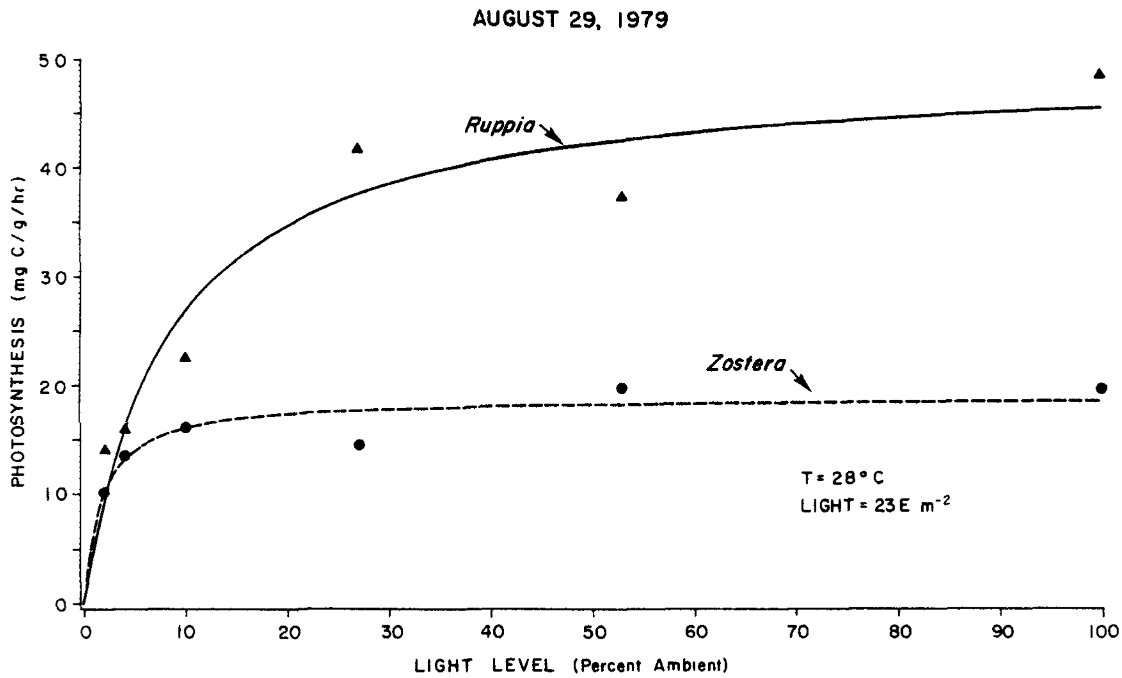


Figure 12. Photosynthesis-light curves for two species of upper Chesapeake Bay submerged vascular plants (from Kemp et al. 1981c).



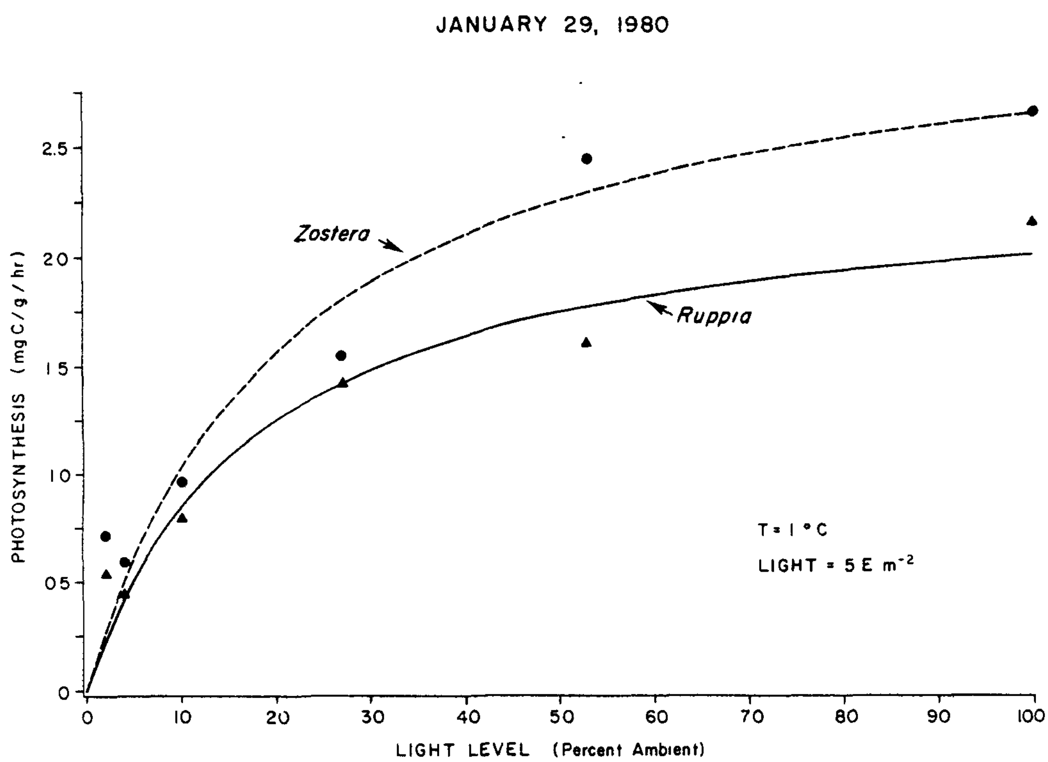


Figure 13. Photosynthesis-light curves for Ruppia and Zostera from a mixed bed site on the Eastern Shore, Virginia. Light is total light flux during 4 h ^{14}C incubations (from Wetzel et al. 1982).

Table 6. PHOTOSYNTHETIC PARAMETERS FOR RUPPIA MARITIMA AND ZOSTERA MARINA LEAVES AT VARIOUS TEMPERATURES. THE LIGHT IS THE TOTAL LIGHT FLUX DURING THE 4h ^{14}C INCUBATIONS (FROM WETZEL ET AL. 1982)

TEMP °C	LIGHT E m^{-2}	P_{max} ($mg\ C\ g^{-1}\ h^{-1}$) Ruppia	Zostera	INITIAL SLOPE Ruppia	Zostera
1	5.0	2.15	2.66	0.18	0.70
8	22.1	3.12	3.25	0.41	1.41
12	15.1	3.91	2.15	0.16	0.55
18	21.8	2.60	2.15	0.35	0.34
21	14.5	3.82	3.55	0.27	0.27
28	12.0	2.39	1.31	0.52	0.69

The data from these experiments relate to how plants capture light and process it, and suggest mechanisms for the species distribution of *Ruppia* and *Zostera* in the lower Chesapeake Bay. The results also show that temperature largely influences the distribution of these plants. *Ruppia* forms single species stands in shallow intertidal to shallow subtidal areas where high light and high temperatures are prevalent during the summer. *Ruppia* is generally more efficient at the higher light and temperature regimes in these habitats. *Zostera*, which has the greater depth range, is adapted to much lower light conditions as indicated by the lower light saturation point and greater α. In the mixed bed areas, *Ruppia* is always shaded by the longer leaved *Zostera*. During winter periods of greater water clarity, *Ruppia* receives sufficient light to survive. During summer periods, its higher P_{max} probably contributes to its survival capability during the period of greatest light attenuation.

Kemp et al. (1981c) compared values of photosynthetic parameters taken from the literature on submerged angiosperms (Table 7). Despite the fact that these parameters were obtained under a wide range of experimental conditions and over a wide range of biogeographical areas, the values are rather similar. P_{max}, which is a function of the dark reaction under optimal environmental conditions or a function of the inhibitor under supoptimal conditions, ranged from 0.9 to 3.7 mg C g^{-1} hr^{-1}. I'_k ranged from 110 to 225 uE m^{-2} sec^{-1} and I_k from 70 to 350 uE m^{-2} sec^{-1}.

Table 7. SUMMARY OF PHOTOSYNTHESIS-LIGHT EXPERIMENTS FOR SELECTED SUBMERGED AQUATIC ANGIOSPERMS[a] (FROM KEMP ET AL. 1981c)

Plant Species	P_{max}[b]	Light Parameters[c] I'_K	I_K	I_C[d]	Reference
Zostera marina	1.5	140	230	28	Drew 1979
" "	2.2	170	220	--	Penhale 1977
" "	1.2	167	280	--	McRoy 1974
" "	1.3	184	345	--	Sand-Jensen 1977
Thalassia testudenum	1.7	225	320	145	Buesa 1975
" "	2.5	170	210	--	Capone et al. 1979
Cymodocca nodosa	2.6	140	220	50	Beer and Waisel 1979
" "	1.5	130	175	40	Drew 1978
Halodule uninervis	1.6	140	220	50	Beer and Waisel 1979
Syringodium filiforme	3.7	225	290	120	Buesa 1975
Ruppia maritima	1.9	123	236	30	Nixon and Oviatt 1973
Vallisneria americana	2.2	130	100	--	Titus and Adams 1979
Ceratophyllum demersum	3.2	135	80	30	Van et al. 1976
" "	2.2	130	230	--	Guilizzoni 1977
Ranunculus pseudofluitas	3.3	115	150	20	Westlake 1967
Myriophyllum spicatum	2.8	215	180	--	Titus and Adams 1979
" "	1.9	110	70	25	Van et al. 1976
" "	1.3	200	290	30	Kemp et al. 1981c
Potamogeton pectinatus	0.9	195	350	60	Westlake 1967
P. perfoliatus	1.1	140	230	25	Kemp et al. 1981c

a Most of these data were interpolated from graphical relations provided by respective authors.

b P_{max} is light-saturated photosynthetic rate in mg C g^{-1} h^{-1}, where O_2 production data were converted to C assuming PQ = 1.2.

c Light variables: I'_K = half-saturation constant; I_K = intersection of initial slope and P_{max}; I_C = light compensation point where apparent production approaches zero. Light data converted to PAR units (uE m^{-2} sec^{-1}) assuming 1 mW cm^{-2} = 2360 Lux = 0.86 cal cm^{-2} h^{-1} = 46 uE m^{-2} sec^{-1}.

d Values for I_C are not available for experiments using the ^{14}C method which cannot measure negative net photosynthesis.

That submerged angiosperms have similar photosynthetic patterns is useful from the management point of view where decisions often must be based on information from only one or two species. However, to answer detailed questions concerning species competition or species adaptations, it is necessary to determine the interrelationship of photosynthetic patterns, pigment complement, plant morphology, and community canopy structure.

Thus, features in addition to photosynthetic parameters help determine plant community photosynthesis. Canopy structure and chlorophyll content were determined for a Ruppia-Zostera bed in the lower Chesapeake Bay (Wetzel et al. 1982). Both Ruppia and Zostera showed a concentration of leaf area (surface available for light absorption) at the lower portion of the canopy where less light penetrates (Figure 14). The wider the bar, the more concentrated the leaf material. This probably allows for a greater overall net community photosynthesis than if there were a uniform vertical distribution of leaf area. Highly significant differences were observed between the vertical stratification of leaf area of Ruppia and Zostera. Ruppia exhibits much greater leaf area than Zostera at the lower canopy (0 to 10 cm above substrate); this probably contributes to its success in the mixed bed areas where it is shaded by Zostera.

Preliminary estimates of pigment content of Ruppia and Zostera suggest differences between species (Figure 15). The highest concentrations of chlorophyll are at mid-canopy for Zostera and at top-canopy for Ruppia (Wetzel et al. 1982). Ruppia also showed a higher total chlorophyll concentration than Zostera. This higher chlorophyll concentration in combination with its canopy structure are adaptations that contribute to Ruppia's success in mixed bed areas. These estimates give us information on how changes in light quantity (from water quality changes) will affect the success of mixed SAV beds.

Microcosm Studies

The microcosm studies of Kemp et al. (1981b) show a negative effect of suspended sediments on Potamogeton photosynthesis (Figure 16). Two concentrations of fine sediment particles (<64 m in diameter, representative of particle size in nature), kept in suspension with recirculating pumps, reduced light availability in the two treatments and resulted in significantly lower photosynthesis of Potamogeton compared with a control. Kemp et al. attributed about half the decrease in productivity of treated systems to the accumulation of epiphytic solids on the plant leaves. Further consideration of the microcosm data involved calculating regressions between chlorophyll *a* or filterable solids and light attenuation coefficients. From these, it was concluded that in the northern Bay, the effect of light attenuation by phytoplankton would be small, however, the effect of suspended sediments on photosynthesis would be significant.

In situ Studies of Community Response to Light

The effect of light on plant community metabolism was investigated in upper and lower Chesapeake Bay grassbeds. In both areas, community metabolism was estimated as oxygen production in large, transparent incubation chambers. During these experiments, detailed measurements of light energy (PAR) reaching the plants were made. In some experiments, neutral density screens similar in design to the ^{14}C studies on individual species were used to decrease available light.

A summary of the upper Bay Potamogeton community response to light is presented in Figure 17, which includes estimates from both early (May) and

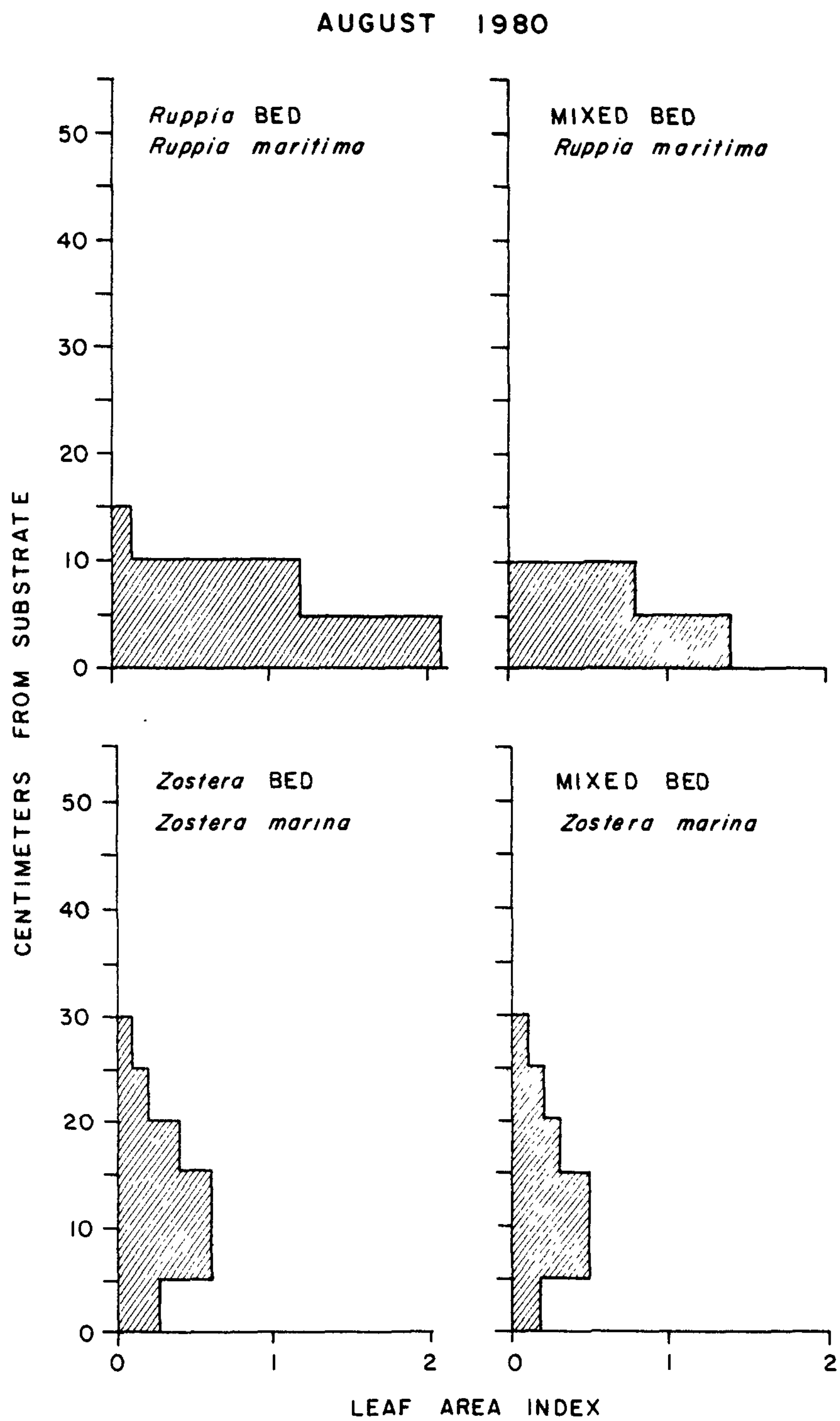


Figure 14. Vertical distribution of one-sided leaf area index (m^2 plant m^{-2} substrate) for Ruppia and Zostera at three vegetated sites on the Eastern Shore, Virginia (from Wetzel et al. 1982).

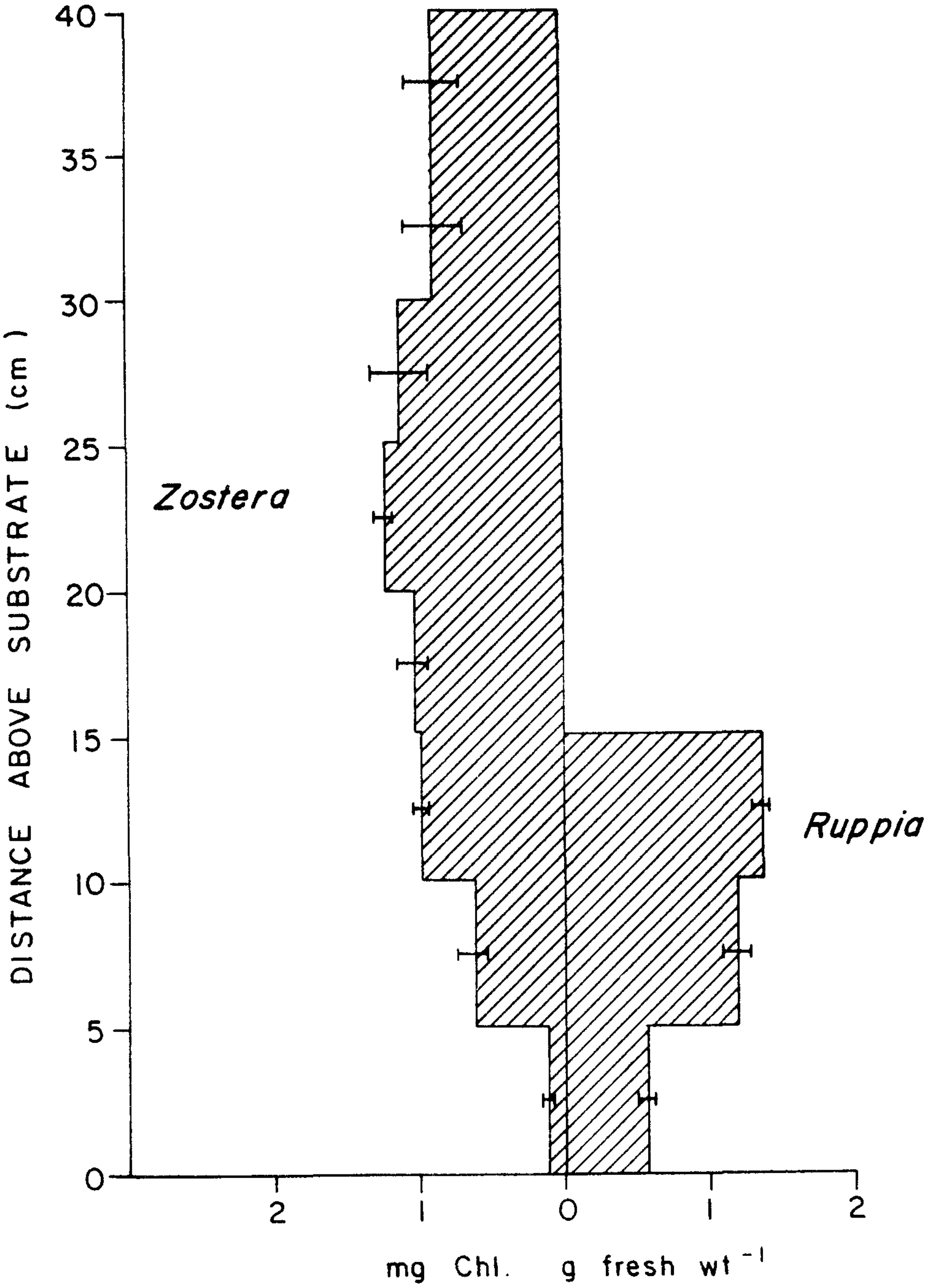


Figure 15. Vertical distribution of total chlorophyll for Ruppia and Zostera from a mixed bed area on the Eastern Shore, Virginia. Values ± standard error, n = 3 (from Wetzel et al. 1982).

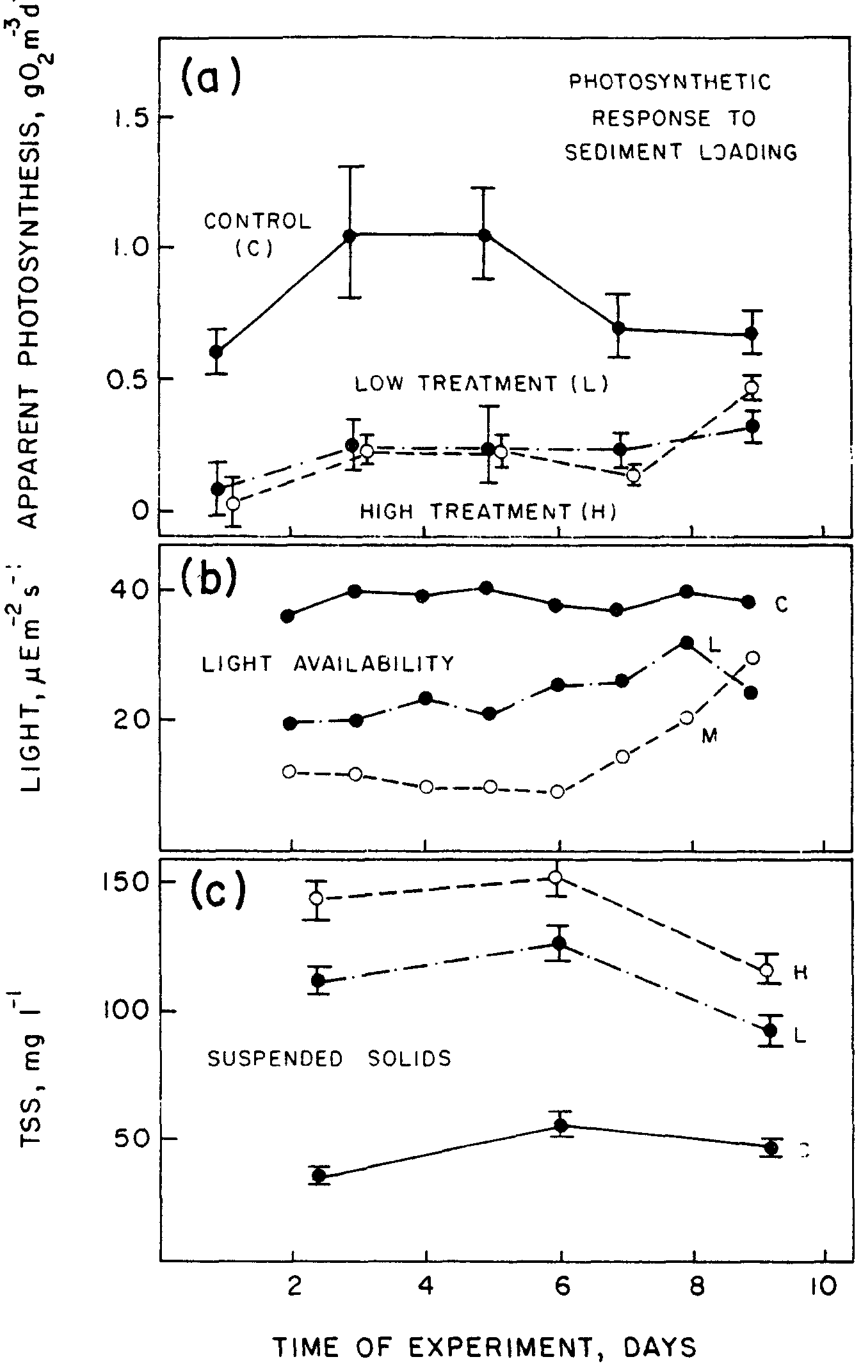


Figure 16. Effect of (c) total suspended solids (TSS) on (b) light availability and (a) rate of photosynthesis of Potamogeton perfoliatus (from Kemp et al. 1981).

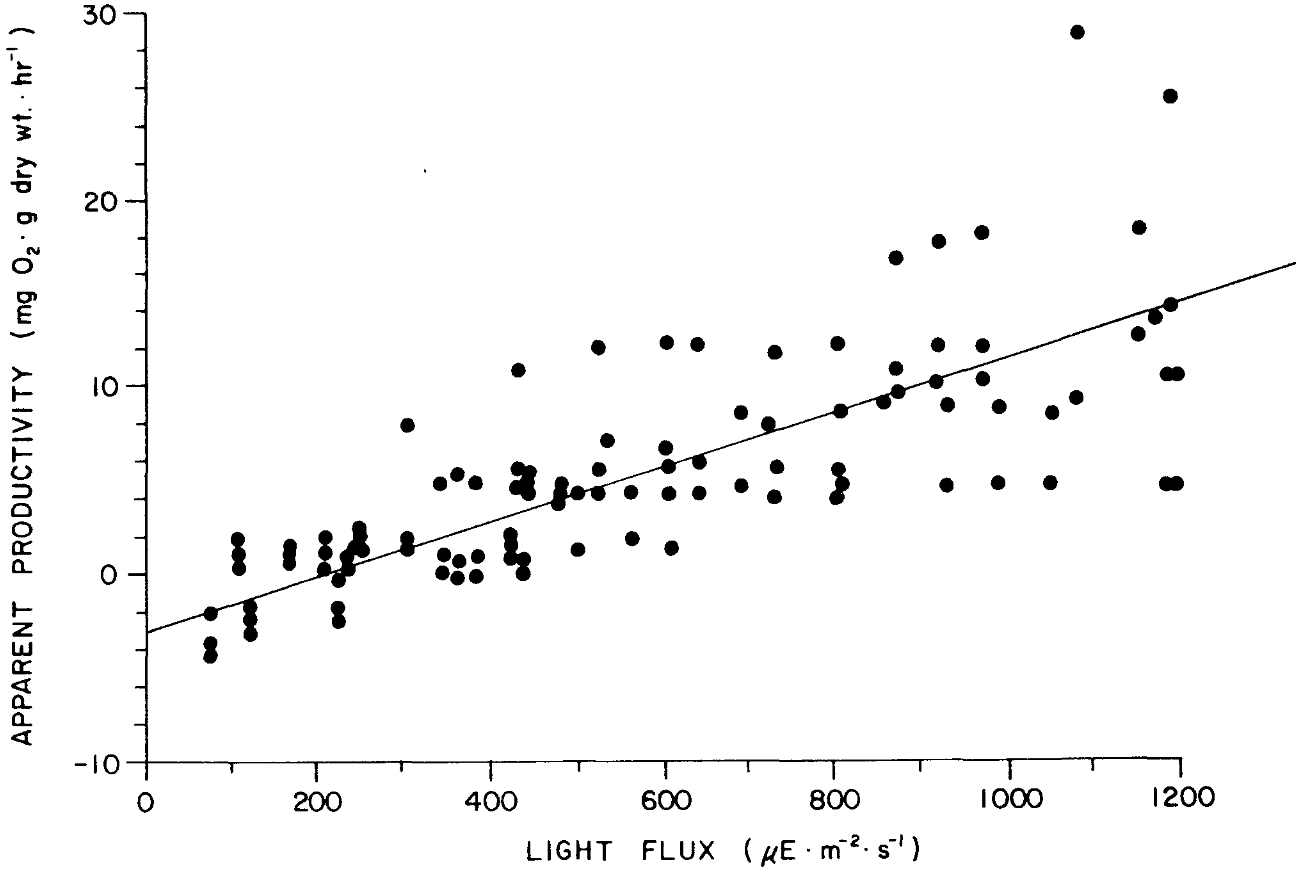


Figure 17. Response of upper Chesapeake Bay submerged vascular plants to light flux (from Boynton, unpublished data).

late (August) periods in the growing season (Boynton, unpublished data). The I_c of the plant community occurs at about 200 uE m^{-2} sec^{-1}, and data suggest that the community is not light-saturated in the ranges of measured *in situ* light flux. If the community were light-saturated, the rate of change would approach zero (P_{max}) with the line in Figure 17 leveling off. An analysis of the seasonal trends suggests no differences in the regression of light and community metabolism between seasons.

Based on these and other studies, Kemp et al. (1981b) conclude that grass communities in the upper Bay are often light limited. For example, actual subsurface light data and three theoretical light extinction coefficients were used to calculate light penetration to a depth of 0.5 m above the substrate; a depth below which *Potamogeton* grows (Figures 18a, 18b). Photosynthetic parameters, I_c, I'_k, and P_{max} were calculated from a P-I curve (Figure 18c). These parameters are identified for each light penetration curve and suggest that for much of the daylight period, the plant community is light-limited or undersaturated, as it is not operating at P_{max}. At early morning and dusk periods of the day, the community is apparently heterotrophic (i.e., no net production).

In the lower Bay, community metabolism studies were carried out in three areas: *Ruppia*-dominated, *Zostera*-dominated, and a mixed *Ruppia*-*Zostera* area (Wetzel et al. 1982). These studies were conducted under a wide range of *in situ* light regimes and under artificial shading conditions. The shallow *Ruppia* areas exhibited higher light and temperature regimes than the deeper *Zostera* areas; the mixed bed was intermediate between the two.

Short-term shading experiments resulted in a general decrease in community metabolism for both *Ruppia* and *Zostera* communities. For the *Ruppia* site, apparent productivity increased with increasing light to a midday peak and decreased during the early afternoon (Figure 19). Based on P-I curves, *Ruppia* was light-saturated during much of the day and was not photoinhibited. The unexplained afternoon depression that occurred while light was increasing may be due to increased community respiration rates under these high summer temperatures. A similar pattern was observed for the *Zostera* site where shading also resulted in decreased apparent productivity (Figure 20). In contrast, the afternoon depression in productivity rates of the *Zostera* bed was not so dramatic as in the *Ruppia* bed. This trend in *Zostera* seemed to follow the decreasing light availability unlike the response in *Ruppia*. These results are similar to those found throughout the study and support previous conclusions that the two communities are physiologically (i.e., temperature and light response) quite different.

Plots of apparent productivity versus light flux at the top of the canopy were used to compare all three habitats (Figure 21). Differences among the three sites were characteristically observed for these summer experiments. Both the *Ruppia* and the mixed bed areas showed decreases in apparent productivity at the highest light fluxes. The *Zostera* site, which did not receive the high light that other sites received, showed no decrease in rates. P-I curves for the seagrass species showed no photoinhibition, even at high summer temperatures, and suggested that the P_{max} of *Ruppia* should be greater than *Zostera* at this time of the year. As evidenced by its high apparent productivity rates, *Zostera* appears adapted to lower light levels. The erratic pattern of data points and the greater number of negative rates for *Ruppia* strongly suggest different community behavior. At the community

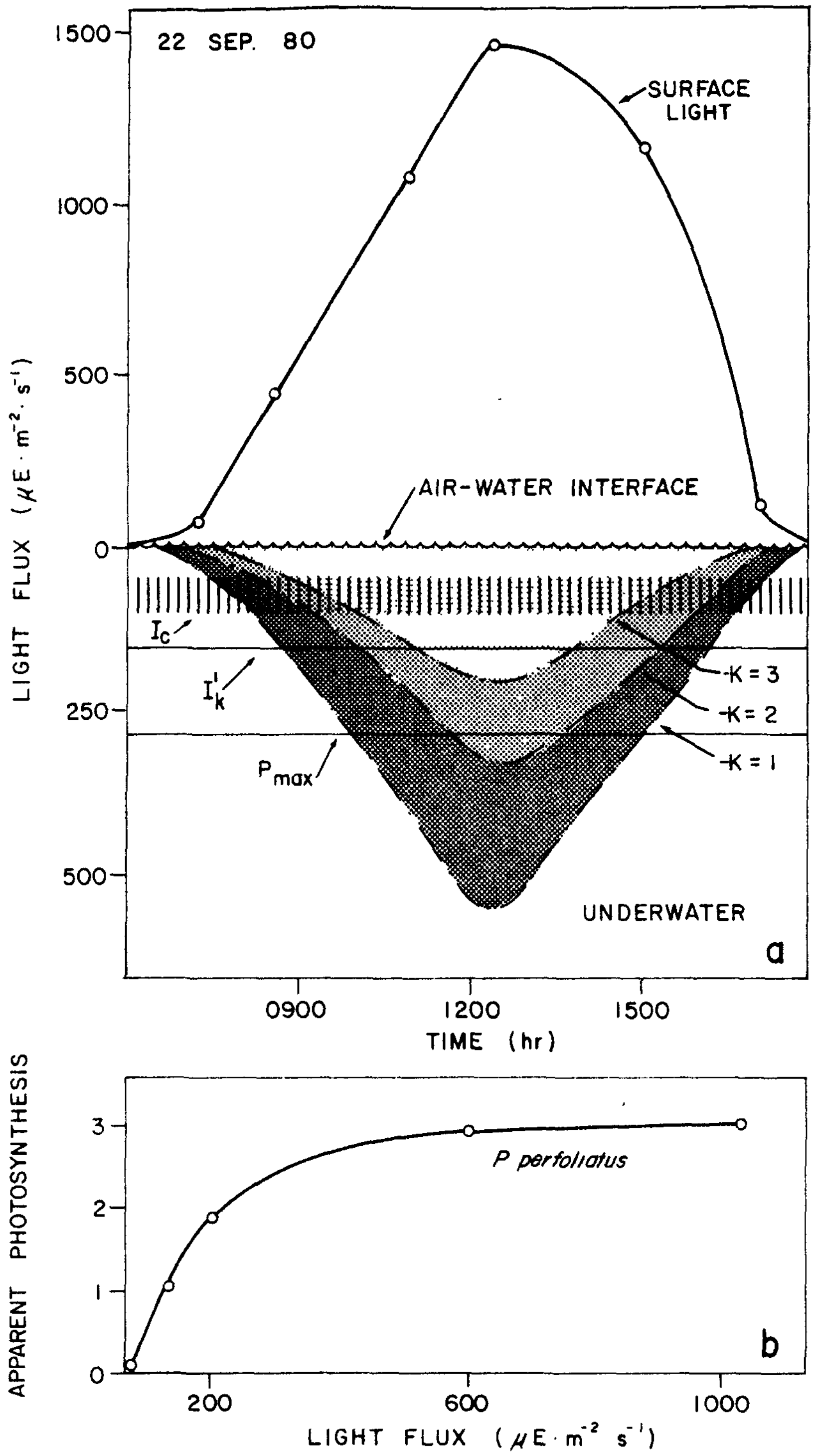


Figure 18. Diagramatic representation of (a) surface and underwater light flux at Todds Cove, upper Chesapeake Bay calculated for three light extinction (K) coefficients. (b) I_c, I_k and P_{max} calculated from P-I curve of Potamogeton perfoliatus (from Kemp et al. 1981c).

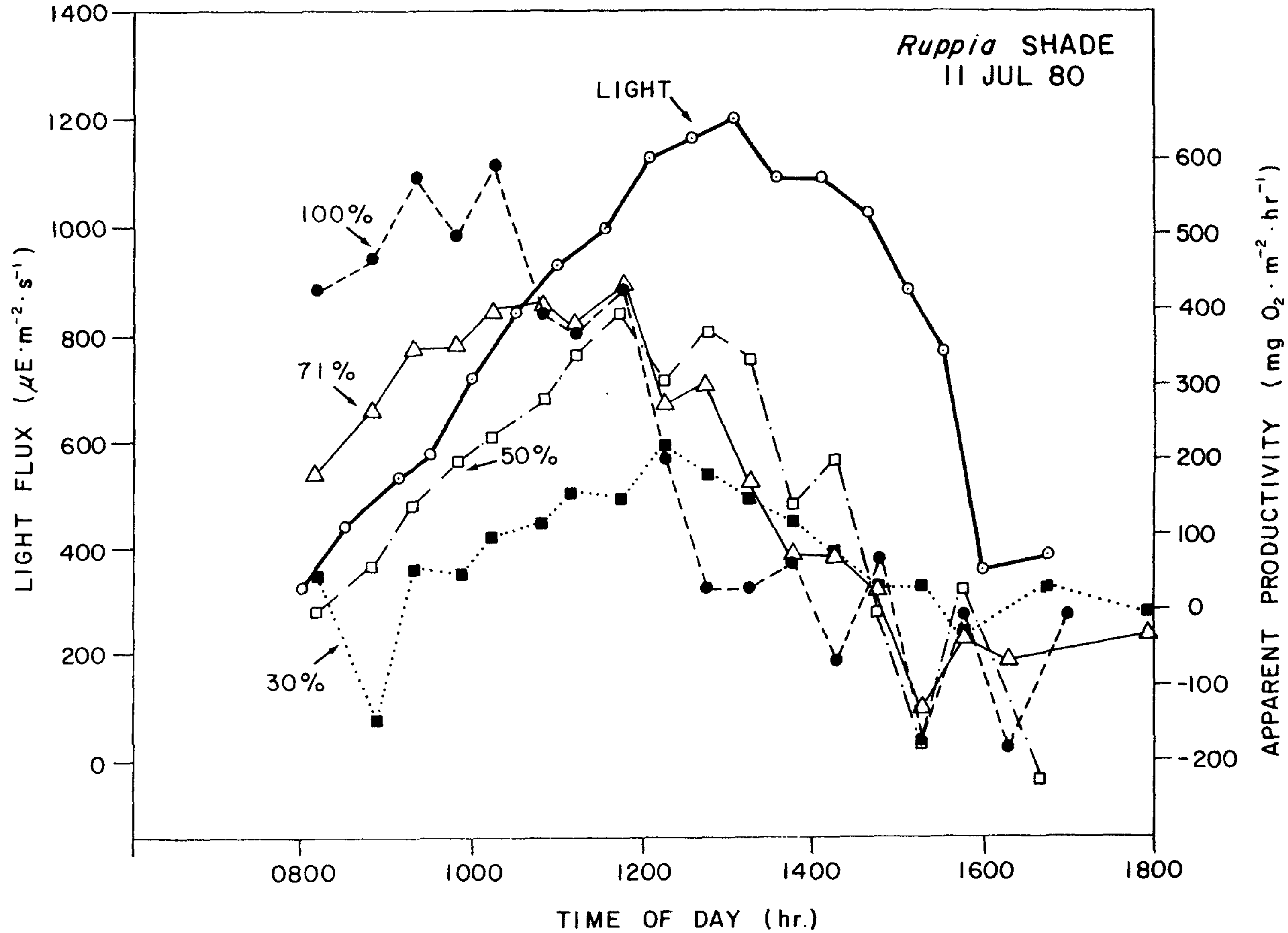


Figure 19. Apparent productivity and light flux at the canopy top vs. time of day fro Ruppia experiments at 100, 71, 50, and 30% of ambient light at the canopy top (from Wetzel et al. 1982).

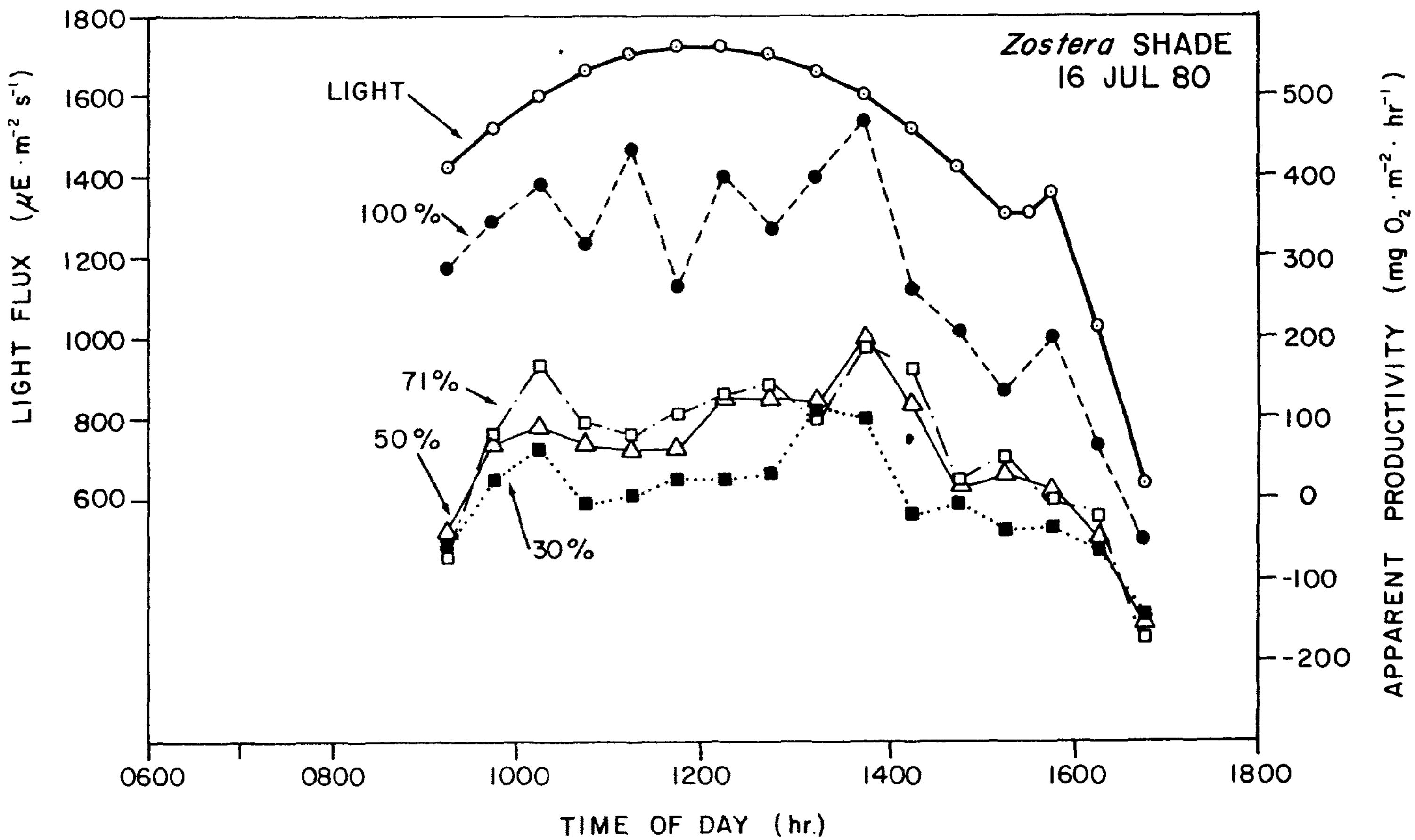


Figure 20. Apparent productivity and light flux at the canopy top vs. time of day for Zostera experiments at 100, 71, 50, and 30% of ambient light at the canopy top (from Wetzel et al. 1982).

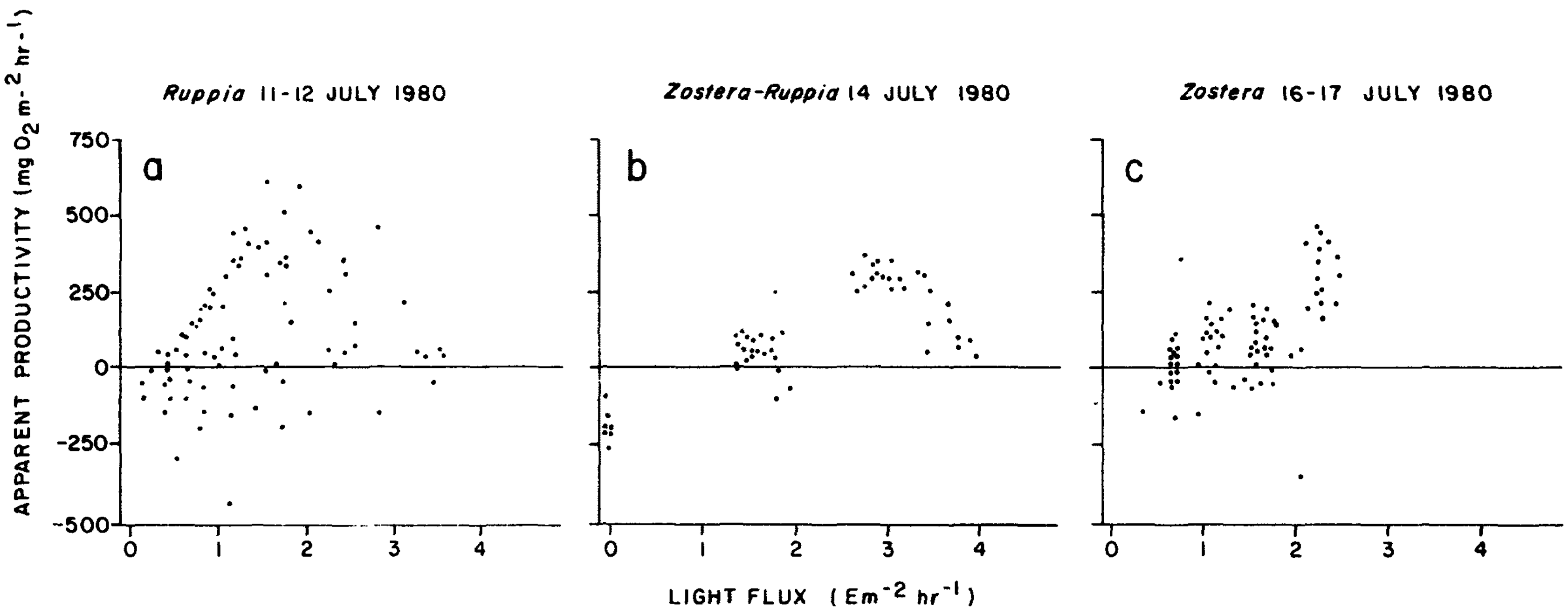


Figure 21. Apparent productivity versus light flux for (a) Ruppia, (b) mixed Ruppia-Zostera, and (c) Zostera areas, Eastern Shore, Virginia (from Wetzel et al. 1982).

level, the differences may be due to differences in community respiration rates, plant species photorespiration rates, or the photosynthetic pattern of other primary producers such as macro- and microalgae. The mixed bed site shows an intermediate pattern, suggesting an interactive effect of the presence of both species of seagrass. Under the influence of changes in water quality, these data show that mixed beds would probably survive better than a bed containing a single species.

A summary of linear regression analyses of apparent productivity versus light flux at the top of the canopy for the three areas is presented in Table 8. At the community level, the correlation coefficient, r, is strongly influenced by season, with the lower values generally observed for the winter months. These are the times of year of clearest water, and the specific rate of O_2 productivity asymptotically approaches P_{max}. Therefore the linear relationship does not adequately describe the

Table 8. APPARENT O_2 PRODUCTIVITY AND LIGHT: LINEAR REGRESSION ANALYSIS FOR LOWER BAY STUDIES (FROM WETZEL ET AL. 1982)

[mg O_2 m^{-2} h^{-1} vs. uE m^{-2} h^{-1} (AT CANOPY TOP)]

DATE	AREA	N	m	b	r	I_c uE m^{-2} h^{-1}	I_c uE $m^{-2}sec^{-1}$
14 Feb 80	Zostera	33	68.1	86.5	0.372	-	-
21 Feb 80	"	36	78.0	157	0.360	-	-
19 Mar 80	"	31	65.4	105	0.210	-	-
29 Apr 80	"	20	280	-183	0.778	0.650	181
2 May 80	"	11	582	-267	0.823	0.459	127
2 Jun 80	"	20	307	-472	0.681	1.54	427
5 Jun 80	"	30	286	-309	0.765	1.08	300
9 Jul 80	"	57	96.5	-147	0.425	1.52	423
16 Jul 80	"	76	124	- 67.1	0.542	0.541	150
19 Aug 80	"	16	89.2	- 84.5	0.793	0.947	203
23 Sep 80	"	27	108.1	-159.8	0.435	1.48	411
7 May 80	Ruppia	10	363	-357	0.980	0.983	273
11 Jul 80	"	83	52.5	- 47.2	0.215	0.899	250
21 Aug 80	"	26	385	-434	0.770	1.13	313
25 Sep 80	"	10	242.5	- 79.1	0.806	0.326	90.6
26 Sep 80	"	16	323.2	-194.5	0.532	0.602	167.2

Table 8. (CONTINUED)

[mg O_2 m^{-2} h^{-1} vs. uE m^{-2} h^{-1} (AT CANOPY TOP)]

DATE	AREA	N	m	b	r	I_c uE m^{-2} h^{-1}	I_c uE $m^{-2}sec^{-1}$
5 May 80	Mixed	28	89.7	-189	0.607	2.11	585
14 Jul 80	"	50	77.9	- 48.9	0.553	0.627	174

1
N = number of observations
m = slope
b = y-intercept
r = correlation coefficient
I_c = estimated light compensation point (x-intercept)

photosynthetic response. This is true for all measures taken at or near P_{max}.

In the Zostera community, maximum rates occur in the spring and early summer. Over this period, the estimated community light compensation point progressively increases, because of increased respiration, to the point that daily community production is negative. This corresponds to the characteristic midsummer die off of Zostera in these areas (Wetzel et al. 1981). Except for the studies carried out in winter and early spring (February and March), the community as a whole is light-limited.

The Ruppia community dominates the higher light and temperature areas of the bed. Maximum rates of apparent photosynthesis occur during the summer, and they corroborate the earlier conclusions that Ruppia has both higher P_{max} and I_c characteristics. Some data suggest that community respiration increases in early afternoon during high light and temperature conditions. These conditions are prevalent at midday low tides during July and August. Overall, Ruppia-dominated communities in the lower Bay appear adapted to increased light and temperature regimes and do not appear light-limited in the Vaucluse Shores study area.

For Chesapeake Bay system as a whole, these data and similar studies completed in upper-Bay communities suggest the extreme sensitivity of Bay grasses to available light. These data also agree very well with information on other geographical areas and species. The general conclusion is that light and factors governing light energy availability to submerged aquatic vascular plants are principal controlling forces for growth and survival.

SECTION 4

SUMMARY

The apparent optical properties of estuarine water create, in general, a light-limited environment for the process of photosynthesis. Water in itself, suspended particles, and dissolved compounds all interact to both attenuate total photosynthetically active radiation as well as to spectrally shift (selectively absorb) wavelengths most important for autotrophic production. Plant pigment systems, in general, are adapted for efficient light-energy capture in relatively narrow bands. In many cases, it is precisely these wavelengths that are most rapidly attenuated in the estuarine water column.

However, data on spectral characteristics and specific waveband attenuation in estuarine and coastal environments are lacking. Our summary of available data, Section 2, indicates that few studies have been completed that characterize these optical properties of estuarine waters and even fewer that can evaluate the data in terms of potential control on rates of photosynthesis. It is difficult, therefore, if not impossible at the present time, to speculate as to the importance or generality of specific waveband attenuation relative to photosynthesis and to autotrophic production in Chesapeake Bay as well as in other estuaries. It has only been within the past few years that submarine spectral irradiance studies have become technologically feasible, and this is reflected in the general paucity of information.

Studies in Chesapeake Bay indicate reductions in both light quality and quantity at selected study sites and during various periods of the growing season for submerged aquatic plants. Recent measures of diffuse downwelling attenuation coefficients (Section 2) in lower Bay communities indicate a severe attenuation of light energy in the photosynthetically important violet blue (400 to 500 nm) region and in the near infrared (700 to 775 nm) region of the spectrum. Also for the March through July period of study, there appears to be a progressive increase in attenuation in these spectral regions.

Comparison of vegetated and non-vegetated areas in Chesapeake Bay with regard to light quality and quantity suggests some improvement (lower attenuation) in the vegetated areas, although the data are quite variable. In the upper Bay, Kaumeyer et al. (1981) report significant differences for one site and not for another. In the lower Bay, comparison of four sites (two vegetated and two non-vegetated) indicates some differences in light quality. There are at these lower Bay sites, some improvements in attenuation in the 400 to 500 nm region in spring months (see recent report by Wetzel et al. 1982 for an updated analysis of this and additional data). The only definitive light quality differences between the sites was reduced attenuation in the 500 to 700 nm region in vegetated areas during spring, an important period in the growth of Zostera dominated communities. Diffuse downwelling attenuation in some photosynthetically sensitive spectral regions is severe. This, coupled with the general increase in attenuation during the growing season and at higher temperatures, indicates the plant communities are undoubtedly light stressed.

There is a much larger data base on plant response to total available light energy (PAR) for Chesapeake Bay as well as for other bodies of water. The dominant plant species in the Bay show the classical, hyperbolic photosynthetic response to increasing PAR. Specific plant response studies suggest physiological differences among species. The dominant upper Bay species, Myriophyllum spicatum and Potamogeton perfoliatus, light-saturate between 600 and 800 uE m^{-2} sec^{-1}, but differ in P_{max} and I_k. M. spicatum appears adapted to higher light conditions than P. perfoliatus. In a similar manner, the dominant lower Bay species, Ruppia maritima and Zostera marina, appear physiologically different with regard to light response. R. maritima is adapted to high light and temperature; Z. marina is adapted to lower light regimes and is stressed at higher, summer temperatures.

In situ studies of entire plant communities in both Maryland and Virginia indicate that the communities are, in general, operating under sub-optimal light conditions. There was no apparent light saturation reached for upper-Bay communities; that is, net apparent community productivity did not asymptotically approach a maximum value. Studies in lower-Bay communities suggest that Z. marina is light-limited the majority of its growing seasons and only in more shallow R. maritima areas did the community photosynthetic response become light-saturated. These results indicate that, at least in terms of total PAR energy and probably because of the extreme attenuation in the 400 to 500 nm region noted earlier, submerged plant communities in Chesapeake Bay as a whole are light-stressed.

Historical data relative to light (turbidity and indirectly, nutrients) and to past distribution and abundance on submerged aquatics indicate progressive Bay-wide changes in systems structure and function. Heinle et al. (1980) and Orth et al. (1971) discuss these in detail. In terms of Bay grasses and the light environment, two overall conclusions of these reports are particularily important. Heinle et al. (1980) note and document the generalized increase in nutrients (and loadings) and chlorophyll concentrations in major tributaries of Chesapeake Bay over the past several decades. Orth et al. (1981) conclude, for roughly the same time scale, that the general pattern of disappearance of submerged plant communities follows a "down-river" pattern. It also appears that upper-Bay and western shore lower-Bay communities have been the most severely impacted. These conclusions, together with our studies on the light environment and photosynthesis-light relations in SAV ecosystems, suggest that total PAR and factors increasing diffuse downwelling attenuation in the 400-500 nm region are principal driving functions controlling plant growth and survival. The specific factors at present that appear to have the greatest impact are suspended particles, both organic and inorganic, which are controlled, in large part, by climatic conditions (runoff and nutrient loading) and indirectly by associated changes in physical-chemical regimes (salinity and temperature).

In summary, it appears that Bay grasses are living in a marginal light environment, and that progressive changes in water quality as discussed by Heinle et al. (1980) will further stress plant communities. To conclude that light has been singularily responsible for recent declines in the vegetation goes beyond the data available. The data do indicate, however, the extreme sensitivity of vegetation to both qualitative and quantitative reductions of available light, and that over the past several decades water quality throughout the Bay, particularily in the tributaries, has progressively declined. Further changes in these parameters can only affect Bay grasses in an adverse way. Results show that SAV can adapt to changes in the availability of light. Long-term shading experiments (in progress) will address this question further.

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Appendix A

CBP Submerged Aquatic Vegetation Projects

Distribution of Submerged Vascular Plants in the Chesapeake Bay	Richard R. Anderson	American University
Distribution and Abundance of SAV in the Lower Chesapeake Bay, 1978, 1979	Robert J. Orth	Virginia Institute of Marine Science
Distribution of Submerged Aquatic Vegetation in Chesapeake Bay, Maryland	Robert J. Macomer	Chesapeake Bay Foundation
Biostratigraphy of the Chesapeake Bay and its Tributaries	Grace Brush	John Hopkins University
Zostera Marina; Biology Propagation and Impact Herbicides	Robert J. Orth	Virginia Institute of Marine Science
Submerged Aquatic Vegetation in the Chesapeake Bay: Its Role in the Bay Ecosystem and Factors Leading to its Decline	J. Court Stevenson W.M. Kemp W.R. Boynton	U. of MD, Center for Environmental and Estuarine Studies
The Functional Ecology of Submerged Aquatic Vegetation in the Lower Chesapeake Bay	R.L. Wetzel R.J. Orth J.V. Merriner	Virginia Institute of Marine Science
Value of Vegetated Habitats and Their Roles as Nursery Areas and Shelter from Predation	K.L. Heck, Jr.	Academy of Natural Sciences of Philadelphia
Assessment of Potential Impact of Industrial Effluents on Submerged Aquatic Vegetation	G.E. Walsh	U.S. E.P.A. Gulf Breeze Environmental Research Labs.
Effects of Recreational Boating on Turbidity and Sedimentation Rates in Relationship to Submerged Aquatic Vegetation	Jerome Williams Herman Gucinski	U.S. Naval Academy
Factors Affecting, and Importance of Submerged Aquatic Vegetation in Chesapeake Bay	W. Valentine	U.S. Fish and Wildlife Service

Submerged Aquatic Vegetation: Distribution and Abundance in the Lower Chesapeake Bay and Interactive Studies of Light, Epiphytes, and Grazers	Robert J. Orth	Virginia Institute of Marine Science
Environmental Regulation of *Zostera marina* and *Ruppia maritima*: Growth and Metabolism	R. L. Wetzel	Virginia Institute of Marine Science
Synthesis of Ecological Research from U.S. EPA's Chesapeake Bay Program: A Continuing Effort 1981-1982.	W.M. Kemp W.R. Boynton J.D. Stevenson J.C. Means	U. of MD, Center for Environmental and Esturarine Studies

SUMMARY AND CONCLUSIONS

As previously indicated, the rationale for conducting an intensive study of SAV was founded in the perceived fact that the distribution and abundance of the Bay grasses had significantly declined during the early 1970s and the intuitive feeling that the Chesapeake Bay ecosystem was healthier when the grasses were more abundant. The general feeling was that an overall degradation of the quality of the Bay's estuarine and riverine waters was, in some way, involved in the decline. Overall, the SAV research was based on a series of questions explained in the introduction to this part. This summary highlights the findings and conclusions from the CBP-SAV research as synthesized in the previous chapter, and attempts to answer these questions.

Although there is no scientific way to measure the exact distribution of SAV throughout the Bay some 50 or 100 years ago for use as a baseline against which to compare current populations, selected areas were studied using archival aerial photography and biostratigraphic analysis of bottom cores. Essentially, this work revealed that an unprecedented decline in SAV populations occurred during the period of 1965 to 1980. The decline was not species-specific, and therefore, was not felt to be the result of disease or some similar natural perturbation.

Overall, the pattern of the decline appears to have been "down river", (from up river, down to the lower estuarine portion, and from up-estuary to down-estuary). The significance of this pattern is that these up-estuary regions have, over time, been the areas subjected to the most rapid urbanization and development.

Additionally, personal communications of Dr. Robert Orth, who conducted the majority of the SAV distribution studies, suggests little evidence that a simultaneous decline has occurred in other areas along the east coast of the United States. Still, there does appear to be growing indications that throughout the world, SAV communities are becoming increasingly stressed in areas where there is extensive industrial and/or urban development.

Having documented that there has been a decline in SAV distribution and abundance in the Chesapeake Bay, the next critical question is "are the grasses a valuable component of the ecosystem?" The Chesapeake Bay Program sponsored research that investigated the role and value of SAV in the context of Bay grasses (1) contribution of organic matter to local food webs, (2) habitat to infaunal and juvenile nekton species, (3) role as a sink for sediments, and (4) role in nearshore nutrient dynamics.

The contribution of SAV to heterotrophic food webs is by either direct grazing of living plants, or by consumption of detritus. It is known that SAV serves as a food source for several waterfowl species. With the decline in SAV populations, those waterfowl have switched to another food source or occur in reduced numbers in the Bay.

Studies indicate that most SAV material enters the food webs through detrital pathways. Data indicate that large predator fish feed in SAV beds, and that their food items, e.g., amphipods, shrimp, are detrital feeders whose food source probably includes some fraction of SAV. Many epifaunal species in the estuary, which are important food items for many consumers, are closely associated with SAV.

weeks following initial contact. Moreover, herbicides degrade rapidly in the estuarine environment, with half-lives measured in days and weeks, and residual concentrations do not appear to build up in sediments. The hypothesized mechanisms of increasing SAV exposure to herbicides through concentration of the compounds in epiphytic sediments or surface-layer films, do not appear to represent significant factors. The one caveate which remains unresolved is the fact that very little is known about estuarine concentrations and SAV toxicities of major herbicide metabolites. The de-ethylated daughter products of atrazine degradation do tend to persist for months under estuarine conditions, and the weed-control literature attributes "carry-over" toxicity (after atrazine application) to this metabolite.

Ephemeral herbicide concentration in excess of five ppb do occur periodically in some estuarine water that once contained extensive SAV beds. In general, such concentrations cause losses in SAV productivity of 10 to 25 percent, even when exposures are brief (about an hour); recovery may take days to weeks even without ambient herbicides. The effects of repeated, brief exposures to such concentrations are not known. However, if the time interval between runoff events (which might yield such concentrations) is greater than SAV recovery time, then partial loss of photosynthesis may persist. Such reductions in SAV productivity will definitely add to the generally-stressed conditions that these plants currently experience in the estuary. Herbicide-induced loss of productivity could act in concert with many of these stressors to create intolerable conditions for SAV existence.

Being plants, SAV require light to grow and survive, and the apparent optical properties of estuarine water create, in general, a light-limited environment for photosynthesis. Chesapeake Bay studies indicate reductions in both light quality and quantity during SAV growing season. Diffuse downwelling attenuation coefficients in lower Bay communities indicate a severe attenuation of light energy in the photosynthetically-important blue (400 to 500 nm) region, and in the near infrared (700 to 775 nm) region of the spectrum.

Historical data relative to light (turbidity, and indirectly, nutrients) and past distribution and abundance on SAV, indicate progressive Bay-wide changes in systems structure and function. In terms of Bay grasses and the light environment, two overall conclusions are particularily important. It has been noted and documented that a generalized increase in nutrients and chlorophyll *a* concentrations in major tributaries of the Chesapeake Bay has occurred over the past several decades. It has also been concluded that for roughly the same time scale, the general pattern of disappearance of submerged plant communities follows a "down-river" pattern. It also appears that upper-Bay and western-shore-lower-Bay communities have been affeced most severely.

These conclusions, together with our studies on the light environment and photosynthesis-light relations in SAV ecosystems, suggest that total PAR and factors increasing diffuse downwelling attenuation in the 400-500 nm region are principal driving functions controlling plant growth and survival. The specific factors that, at present, appear to have the greatest impact are suspended particles, both organic and inorganic, that are largely controlled by climatic conditions (runoff and nutrient loading), and indirectly by associated changes in physical-chemical regimes (i.e. salinity and temperature).

In summary, it appears that Bay grasses are living in a marginal light environment, and that progressive changes in water quality will further stress the plant communities. To conclude that light has been singularly responsible for recent declines in the vegetation goes beyond the data available. The data do indicate, however, the extreme sensitivity of the vegetation to both qualitative and quantitative measures of available light. The data further imply that over the past several decades water quality throughout the Bay, and particularily in the tributaries, has progressively declined. More changes in these parameters can only affect Bay grasses in an adverse way.

Following three years of research, we conclude that SAV exists in a stressed environment. The sources of those stresses include such natural factors as salinity extremes, waterfowl grazing, uprooting by cownose rays and major storm events, as well as man-induced stresses such as water-column turbidity, accumulation of epiphytic materials resulting from nutrient enrichment and exposure to agricultural chemicals. The natural stresses do not appear to be responsible for the presently reduced populations of Bay grasses, because SAV has always been subjected to these pressures and the historic record, as we have been able to reconstruct it, does not reveal previous declines of such magnitude.

The issue as far as light is concerned is not simply of suspended material, both inorganic and organic, in the water. Recent observations and studies indicate that the growing nutrient enrichment of the Bay's waters is stimulating the growth of epiphytic material. Combined, the increased epiphytes and suspended materials may be the most significant cause of the reduced SAV populations. At this time the results of investigations into this issue are being analyzed.

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and Sn, factors are largely less than two or close to baseline factors throughout the Bay proper. Seaward of the Bay Bridge (Annapolis) factors generally diminish, but Cd, Pb, and Zn are greater than two. The longitudinal distribution of values does not display a maximum in the Bay near Baltimore, an expected increase if metals were emanating from Baltimore. Instead, the values mainly decrease from the Susquehanna River mouth, suggesting a river source (Helz et al. 1981). If the Susquehanna watershed is not naturally enriched compared to average crust, then the enrichment is affected by direct contamination from industrial and municipal sources or from acid mine drainage.

Bed sediments within the Patapsco River, Baltimore Harbor, are markedly enriched in Co, Cr and Zn (Sinex et al. 1981). Longitudinal distributions of enrichment factors, show that Cr increases with distance landward, and Zn is enriched throughout the Harbor. The Elizabeth River, Hampton Roads, is notably enriched in Zn with Zn/Al ratios of six to 25 (Sinex et al. 1981).

Enrichment factors for Cd, Cu, Pb, and Zn in surface suspended material of the central Bay are much greater than in bed sediments of the northern Bay. Metal/Fe ratios range from 10-118 for Cd, 12-27 for Cu, 37-51 for Pb, and 16-74 for Zn. The high enrichment factors in the central Bay are associated with high percentages of organic matter, probably produced by plankton metabolism. Additionally, the metal content of central Bay suspended material exceeds the content of oceanic phytoplankton more than nine times for Cd and Zn, and more than 19 times for Cu, Ni, and Pb.

Historic Metal Input Recorded in Sediments

Some sediments in the Bay reveal trends in metal enrichment. In sediments deposited in anoxic waters, no benthic macrofauna are present. Therefore, the sediments remain relatively undisturbed and may record the history and rate of change of metal influx. When a core of such sediments is analyzed for trace metals and dated by ^{210}Pb chronology, the vertical changes reveal variations in metal input. This approach assumes no diagenetic migration of metals through the length of the core. In oxic environments, however, burrowing activities of benthic organisms can disturb the record of sedimentary sequences, create an "artificial" ^{210}Pb distribution, and influence vertical trace metal distributions.

The vertical distribution of ^{210}Pb and metal concentrations (Helz et al. 1981) and the degree of bioturbation have been carefully examined for selected sediments of the Bay. Cores 4, 18, and 60 (Figure 19) exhibit exponential ^{210}Pb profiles, low ^{210}Pb depth-integrated concentrations, and low or moderate bioturbation. They also show no metal peaks and display a relatively uniform rock structure. In addition, core 4 has ^{137}Cs data that verify the ^{210}Pb sedimentation rate. Metal/aluminum ratios for the three cores, and ^{210}Pb chronology are presented in Figure 19. All three cores show Zn enrichment in the Zn/Al ratios near the core surface, with maximum enrichment occurring at about 1940 in core 40 and about 1960 in cores 18 and 60. The first appearance of excess concentrations is also temporally displaced down the Bay from 1890 in core 4, to 1920 in cores 18 and 60. If the source of this excess Zn is fluvial (or anthropogenic) and up-Bay, then it takes about 20 years for the metals to be transported 80 kilometers between core 4 and core 18, a nominal rate