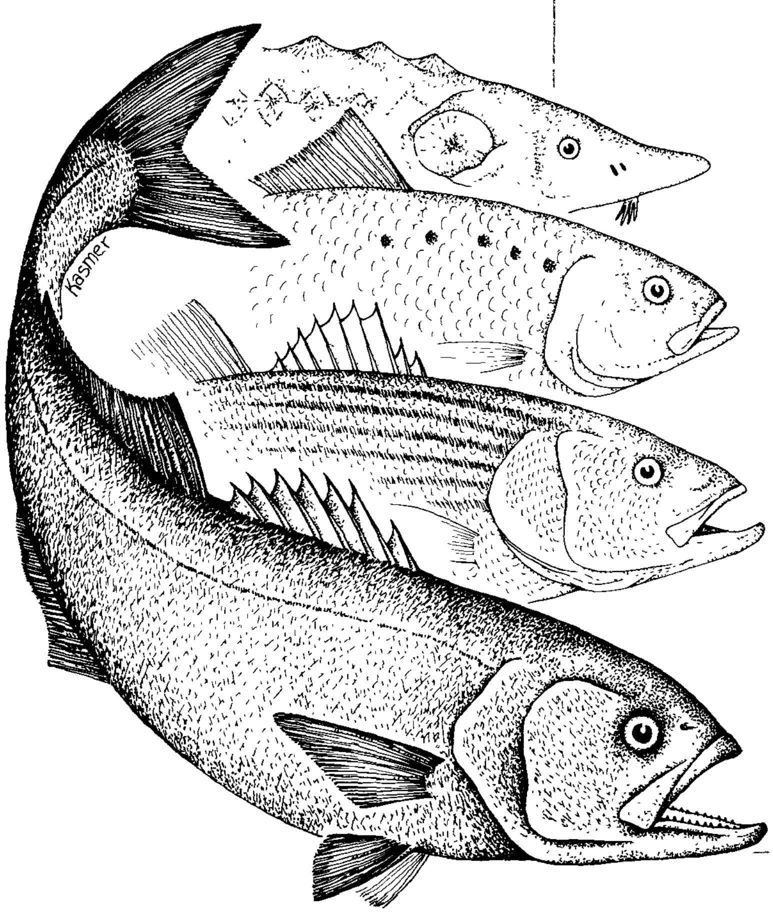
United States
Environmental Protection
Agency

Region 3
Sixth and Walnut Streets
Philadelphia, PA 19106

September 1983

CHESAPEAKE BAY: A PROFILE OF ENVIRONMENTAL CHANGE



CHESAPEAKE BAY: A PROFILE OF ENVIRONMENTAL CHANGE

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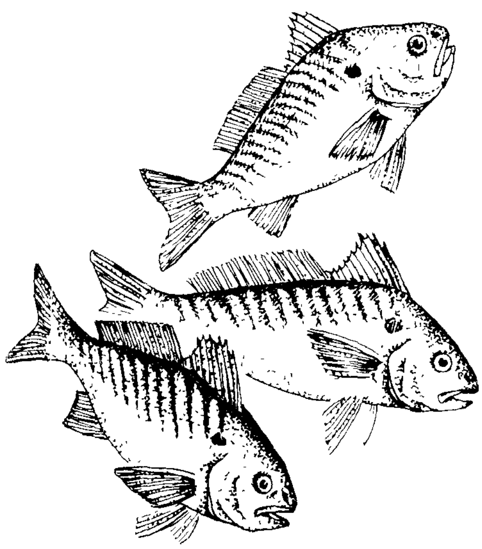

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FOREWORD

Chesapeake Bay has experienced significant changes in recent years. Concern for this national resource prompted the U.S. Congress in 1976 to direct the U.S. Environmental Protection Agency (EPA) to conduct a study of the Bay's resources and water quality, and to develop appropriate management strategies. This document, *Chesapeake Bay: A Profile of Environmental Change*, is the third of four final reports developed by the EPA's Chesapeake Bay Program (CBP). It provides a characterization of the health of the Bay and its tributaries. The project was initiated by the former director, Tudor T. Davies, and the deputy director, Thomas B. DeMoss. Their vision and encouragement resulted in the establishment of a comprehensive data base that permitted Bay-wide analysis of water quality and resource trends over time. This analysis would not have been possible without the many years of effort by Bay scientists. The CBP gratefully acknowledges their contribution to our understanding of Chesapeake Bay.

This characterization of the Bay makes a significant step forward by attempting to link water quality trends to Bay resource trends, thus, making science useful to managers and citizens. Such a retrospective approach is imperfect because large gaps in the data base and necessary assumptions limit our ability to make strong scientific causal inferences. Yet, similar approaches suffering from similar imperfections, such as the efforts in the Great Lakes and in the Thames River in England, have yielded interpretations used directly to restore the ecosystem. Without doubt, some of our water quality-ecosystem linkages are strong; others are more tenuous — none are incorrect. Nonetheless, what we now have, for the first time, is a comprehensive assessment of the state of the Bay, and it clearly suggests the need for action. The CBP hopes that the findings of this report will assist managers in developing strategies that will modify the Bay's water quality and improve the ecosystem.

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Staff support for this project was provided through a cooperative agreement (X-003264) with The University of West Florida under the supervision of Dr. Jerry Oglesby, Project Manager.

Over the years, many EPA, The University of West Florida, and GEOMET Technologies, Inc. personnel have been involved in this massive effort. We extend special thanks to Mary Barrow, William Cook, Elise Christenson, Ian Gillelan, Mike Keller, John Klein, Jan Kuzava, James Smullen, Charles Strobel, Duane Wilding, and Maryann Wohlgemuth for their assistance with this project.

In addition, the Chesapeake Bay Foundation; Citizens Program for Chesapeake Bay; District of Columbia Department of Environmental Services; Maryland Department of Health and Mental Hygiene, Office of Environmental Programs; State of Maryland Department of Natural Resources; Commonwealth of Pennsylvania Department of Environmental Resources; Susquehanna River Basin Commission; Virginia Council on the Environment; and the Virginia State Water Control Board are gratefully acknowledged for their cooperation, active support, and sustained interest in the Chesapeake Bay Program.

EXECUTIVE SUMMARY

INTRODUCTION

The Chesapeake Bay Program (CBP) characterization report, *Chesapeake Bay: A Profile of Environmental Change*, describes trends in water and sediment quality, and in the living resources of Chesapeake Bay. The water quality parameters evaluated include nutrients, dissolved oxygen, organic chemical compounds, and heavy metals. The living resources that were assessed include phytoplankton, submerged aquatic vegetation (SAV), benthic organisms (including shellfish), and finfish. Trends in water and sediment quality, and in living resources, including the interrelationships among these factors, were used to characterize the current state of the Bay. The CBP's characterization of Chesapeake Bay is based on a ranking of specific areas, or segments, of the Bay with regard to selected nutrient and toxicant variables, and to the diversity and abundance of the living components.

WATER AND SEDIMENT QUALITY

The quality of the Bay's water and sediments reflects both the natural physical and chemical characteristics, and the impact of human activities. Over 150 tributaries drain the 64,000 square mile (16.6×10^4 square kilometers) watershed. Along with fresh water, the rivers bring other materials into the Bay: nutrients, sediments, and toxic substances. Although the Bay has the ability to assimilate much of this material, most remains within the estuary. Human activities have greatly contributed to the input of nutrients, sediment, and a variety of synthetic chemicals, heavy metals, and other potential toxicants into Chesapeake Bay.

Chesapeake Bay has changed greatly since the time of the first settlers. The delivery of nutrients to the Bay has increased, reflected by increases in runoff containing suspended sediment and fertilizers, and sewage effluents. The amount of toxic materials—heavy metals and organic chemicals—has similarly increased as industrialization has progressed. Many of these changes occurred before the first scientific surveys of the Bay. For that reason, it is sometimes difficult to show strong recent trends, as the bulk of the change had already taken place before any data were collected.

Nutrient Enrichment

Nutrients, such as nitrogen and phosphorus, are essential for plant growth and, thus, for primary productivity in the estuary. However, in excess, these nutrients can cause problems, including blooms of undesirable algae, reduction in dissolved oxygen (DO), and decreased water clarity. Currently, the northern Bay and upper portions of the tributaries have relatively high nutrient concentrations; the mid-Bay, lower portions of the tributaries, and eastern embayments have moderate concentrations of nutrients; and the lower Bay (where sufficient data exist) appears to be not enriched. When data from 1950 to 1980 are analyzed, they indicate that, in most areas, water quality is degrading; that is, nutrient levels are increasing. Total nitrogen concentrations are declining in the Patapsco, lower Potomac, and upper James Rivers; total phosphorus concentrations are declining in the upper Potomac River and throughout the James River. Elsewhere, trends are increasing (or stable) for most forms of nutrients, particularly in the upper and mid-Bay main stem and larger tributaries.

Dissolved Oxygen Trends

To assess the management implications for resources of these nutrient trends, it is valuable to examine a related parameter, dissolved oxygen. As nutrient levels increase, phytoplankton (algal) growth is encouraged and more organic matter is produced. Decay of this organic matter consumes oxygen. If more oxygen is used than is supplied by reaeration or photosynthesis, as often occurs in deep water, the water becomes anoxic (no oxygen) and devoid of most forms of life except anaerobic bacteria. This process occurs naturally in some Bay areas during the summer; however, high nutrient loads can increase its severity.

Both the chlorophyll *a*, an indicator of algal biomass, and the DO trends suggest that the duration and extent of anoxia have been accelerated in the Bay in recent years. There were no anoxic waters and only limited areas of low DO in the main stem of the Bay during July of 1950. In July 1980, however, a very large area of the main stem of the Bay was experiencing anoxic conditions. It is estimated that the volume of water with DO concentrations equal to or less than 0.7 mg L^{-1} was 15 times greater in 1980 than in 1950. The duration of oxygen depletion has also increased. It was sporadic during the mid-1950's; occurred from mid-June to mid-August during the 1960's; and, in 1980, began during the first week in May and continued into September. This increase in the spatial and temporal extent of low DO levels reduces the area of the Bay that can support normal finfish and shellfish populations.

Organic Compounds in Water and Sediments

Organic compounds can occur naturally; the ones of major concern are synthetically produced. The distribution of organic compounds, such as hydrocarbons, pesticides, and herbicides, in the bottom sediments and the water column of the main Bay, and an analysis of limited tributary data, suggest that organic compounds concentrate near sources, at river mouths, and in maximum turbidity areas. The highest concentrations of organic chemicals in the sediments were found in the Patapsco and Elizabeth Rivers, exceeding 100 parts per million (ppm) at several locations. In the main Bay, highest concentrations of organic substances occur in the northern half. Most observed sediment concentrations range from 0 to 10 ppm; however, in the upper Bay, some stations had levels of total organics over 50 ppm. These general trends suggest that many of the problem organic compounds in the Bay tend to adsorb to suspended sediments, and then accumulate in areas dominated by fine-grained sediments.

Metal Contamination

Metals are chemical elements which occur naturally in the environment; however, in excess, they can become toxic to organisms. Many areas of the Bay show metal concentrations that are significantly higher than natural background levels. The Contamination Index (C_I) was developed by comparing present concentrations of cadmium (Cd), copper (Cu), chromium (Cr), nickel (Ni), lead (Pb), and zinc (Zn) in the Bay's surface sediments to predicted natural levels from the weathering of rock in the Bay watershed and measured pre-colonial levels from sediment cores. If the present concentration of a given metal exceeded these natural Chesapeake Bay background levels, it was considered to be anthropogenically enriched. The most contaminated sediments are located in the Patapsco and Elizabeth Rivers, both heavily industrialized tributaries. Metal concentrations up to 100 times greater than natural background levels were found in these areas. High levels of metal contamination ($C_I > 14$) were also found in the upper Potomac, upper James, small sections of the Rappahannock and York Rivers, and the upper mid-Bay. Moderate contamination occurs in the Susquehanna Flats and off the mouth of the Potomac River. These trends suggest that higher concentrations are found near industrial sources and in areas where fine sediments accumulate, such as in the deep shipping channel of the upper Bay. In general, there is little movement of metals out of the most contaminated areas, except when physically transported, as might occur through movement or disposal of contaminated dredge material.

Significant levels of particulate and dissolved metals occur in the water column. Concentrations of particulate Cd, Cr, Cu, Ni, and Zn are greatest in the upper Bay and near the turbidity maximum; actual values vary greatly with the tidal cycle and the amount of suspended sediment. High dissolved values, some exceeding EPA water quality criteria, have been observed, particularly for Cd, Cu, Ni, and Zn. These high values are most frequent in areas near industrial sources, and near upper reaches of the main Bay and western shore tributaries.

LIVING RESOURCES

Major changes in Bay resources can be identified, including shifts in the relative abundance of species or the types of biological communities found in certain areas. The CBP focussed on individual living resource groups (e.g., submerged aquatic vegetation, finfish), describing the documented trends, and comparing present conditions with the potential status.

Phytoplankton in Two Well-Documented Areas

The upper Bay (above the Bay Bridge) and upper Potomac River (tidal-fresh reach) have shown increased dominance by a single species of phytoplankton and increased biomass. Such changes are considered to be indicative of eutrophication and, in fact, have paralleled changes in nutrient enrichment in these areas. These two areas are those for which the best data are available; similar changes may be occurring elsewhere, or could be expected to occur if nutrient enrichment continues.

The Potomac River's tidal-fresh reach was characterized in the 1960's and 1970's by massive blue-green algal blooms, indicators of excess nutrients. Increased phosphorus control in the watershed appears to have been beneficial. In 1979, algal populations were diverse, with blue-greens composing only 25 percent of the total. Total cell counts (biomass) for 1979 and 1980 were also considerably lower than in the past.

Trends in nutrient enrichment of the upper Bay tributaries have closely paralleled those of the upper Potomac during the 1960's. Massive algal blooms have been frequently reported in the upper main Bay (above the Bay Bridge), with elevated chlorophyll levels caused by increasing numbers of blue-green algae. By comparison, observors of this area from 1965 to 1966 reported only an occasional occurrence of blue-green algae. It is estimated that cell numbers in this area have increased approximately 250-fold since 1955.

Decline of Submerged Aquatic Vegetation

Since the late 1960's, a dramatic, Bay-wide decline has occurred in the distribution and abundance of submerged aquatic vegetation. Loss has moved progressively down-estuary. Submerged aquatic vegetation now occupies a significantly more restricted habitat than at any time during the past. The role of SAV in the ecosystem has been reduced; its ability to recover from this current status is uncertain. Changes in the distribution and abundance of Bay waterfowl, which feed on SAV, have paralleled these vegetation changes.

Annual surveys of SAV conducted by the Maryland Department of Natural Resources and the U.S. Fish and Wildlife Service Migratory Bird and Habitat Research Laboratory have shown that the number of vegetated stations in Maryland dropped from 28.5 percent in 1971 to 4.5 percent in 1982. Species diversity also declined significantly. Comparison of the habitat filled in 1978 with the expected habitat shows that the areas of greatest loss (upper Bay, western shore tributaries, and upper Eastern Shore tributaries) correspond with areas of greatest nutrient enrichment.

Changes in Benthic Invertebrates

Benthic animals are considered good indicators of pollution because most are relatively immobile and cannot readily escape unfavorable conditions. Changes in benthic biomass, community structure, and diversity can indicate a variety of stressful conditions. Where sufficient data exist, spatial comparisons were made of current conditions, particularly in the main Bay and in certain tributaries. Trends in diversity and the relative abundance of pollution-tolerant annelids were investigated. In the main Bay, benthic abundance and diversity seem to be related to physical aspects of the environment (i.e., salinity and sediment type). Highest diversity occurs in the lower Bay. In some polluted tributaries, especially the Patapsco and Elizabeth Rivers, significant declines in species diversity and enhancement of pollution-tolerant annelids, relative to molluscs or crustacea, are observed. These changes are characteristic of stressed communities.

Trends in Commercial Shellfish

The density of annual oyster spat set is a measure of the success of oyster reproduction and recruitment, and is a reasonable predictor of oyster harvest. Comparison of the average oyster spat set for the past ten years with the previous ten to thirty years shows significant declines in the upper main Chesapeake Bay and the Chester, James, Nanticoke, Patuxent, Pocomoke, Potomac, Rappahannock, and Wicomico Rivers, Eastern Bay, Fishing Bay, and Pocomoke Sound. In general, 1980 was a good year for spat fall, particularly in Eastern Shore tributaries; this fact is related to high salinities during the spawning period. Spat set in the upper Chesapeake and its western tributaries was generally light even in this good year.

The harvest of oysters for Chesapeake Bay has declined since 1880, but has remained relatively stable since 1960 to 1965. This is in part due to management practices, such as shell and seed planting. The harvest for the western shore has decreased significantly during the period from 1962 to 1980; the harvest for the Eastern Shore increased significantly. This is consistent with the Eastern Shore's consistently better spat set. For the Chesapeake Bay as a whole, declines in oyster harvest have been somewhat offset by an increased harvest of blue crabs. As a result, the Bay-wide landings of shellfish have not changed greatly from 1962 to 1970 and 1970 to 1980. However, overall shellfish harvest for the western shore has decreased significantly during this period.

Shifts in Finfish Harvest

The CBP examined trends in harvest and other indicators (young-of-the-year surveys) for the major commercial species historically landed in Chesapeake Bay. These include freshwater spawners such as striped bass, white perch, yellow perch, catfish, shad, and alewife; marine spawners such as menhaden, croaker, spot, bluefish, and weakfish; and three estuarine forage fish, Bay anchovy, mummichog, and Atlantic silverside.

The Maryland juvenile index provides consistent data since 1958 for the upper Bay, and the Nanticoke, Choptank, and Potomac Rivers. Juvenile indices of most anadromous and freshwater species show declines in recent years, with the exception of the Potomac River where white perch and yellow perch have increased. Information for Virginia waters is not directly comparable, because of differences in methodology and target species (sciaenids). However, trends in marine-spawning fish were similar in both data sets. Marine spawners show general overall increases in all basins, although some species show declines in the most recent surveys. In Maryland, mummichog shows an increasing pattern similar to that of marine spawners, while the Bay anchovy and Atlantic silverside show declines. However, the anchovy has been increasing in Virginia tributaries surveyed during the same period. This may reflect differences in water quality or habitat (particularly the availability of SAV, used as shelter by this species) between the two states.

Harvests of anadromous and freshwater species have declined in Chesapeake Bay. The downward trend in American shad has been continuous since 1900, while declines in river herring and striped bass landings have been more recent. Landings of alewife, shad, and yellow perch are now at unprecedented low levels. Harvests of marine spawners, on the other hand, have increased in most areas. Menhaden landings have risen steadily since 1955; the increase in bluefish landings has been more recent. The increased yield of marine spawners and decreased yield of freshwater spawners represent a major shift in the proportion of the finfishery accounted for by each group: during 1881 to 1890 marine spawners accounted for about 75 percent of the fishery; during 1971 to 1980 they accounted for 96 percent.

Because freshwater-spawning fish and estuarine-spawning shellfish spend all or most of their sensitive life stages in the Bay, their well-being may be considered as an indication of the health of the estuary. Thus, the simultaneous declines in most of these species is reason for concern.

RELATIONSHIPS BETWEEN WATER AND SEDIMENT QUALITY, AND LIVING RESOURCES

Organisms respond directly to changes in their habitat, food supply, competitors, or predators. Major factors which affect the Bay's living resources include natural variables such as freshwater inflow, temperature, or other organisms, as well as human-induced stress such as nutrient and toxicant enrichment. Distinguishing between effects triggered by anthropogenic, as opposed to natural, causes is often difficult because of the natural variability of organism distribution and abundance. Although the CBP was unable to pinpoint causes for specific resource changes, the similarity in patterns and the overlap in the distribution of water or sediment quality and living resource trends in the Bay should be considered as more than a striking coincidence.

Submerged Aquatic Vegetation

The Chesapeake Bay Program supported a major research effort to identify the causes of the recent SAV decline. Investigators focussed on two main hypotheses: (1) the use of toxic agricultural materials, particularly herbicides, has increased in recent years. Runoff of these substances from agricultural areas may be reducing or eliminating SAV. (2) Reduction in the light available to the plants because of an increase in water column turbidity or increased growth of epiphytes (or both) may be causing the decline. Nutrient enrichment was considered a major factor affecting both turbidity and epiphyte growth. Research sponsored by the CBP implicated light limitation as the most important factor regulating the Bay-wide SAV loss. Herbicides could be important locally, or close to sources (although areas affected may represent significant habitat).

The CBP's research conclusions are supported by field observations. Comparison of a map of current SAV status to Bay nutrient conditions reveals that vegetation now occurs primarily in areas that are not enriched or only moderately enriched. Statistical analysis (rank correlation) shows a significant correlation between declines in SAV and increased nutrient concentrations in many areas. The major nutrient which appears to correlate with SAV abundance is nitrogen. A negative response to maximum chlorophyll *a* values, an analog of both nutrient loading and turbidity, was also found. These analyses support experimental results linking the recent loss of Bay vegetation to increases in nutrient loadings and ultimately to light stress caused by increasing phytoplankton biomass and epiphytic growth.

Benthic Organisms

Major anthropogenic factors which could adversely affect benthic organisms in Chesapeake Bay are toxic materials, either in bed sediments or in the overlying water column, and nutrients. Toxicants can produce either acute (elimination of susceptible species) or sublethal (accumulation in body tissues) effects. Nutrient enrichment can alter the Bay's benthic community structure by stimulating phytoplankton production. Excessive production of organic material has been linked to the increased duration and extent of low DO values in Chesapeake Bay, decreasing available benthic habitat.

Episodes of low DO have been cited as the major factor limiting benthic distribution in deeper waters of the upper and mid-main Bay. The documented increase in extent of anoxic water in the mid-Bay can be related to complete loss of benthic habitat or replacement with ephemeral assemblages. This may have secondary impacts on bottom-feeding predators such as crabs or fish, which can be stressed by food limitation and reduction of habitat. Recent changes in the mid-Bay blue crab fishery, especially the necessity to set pots in shallower water, may be a direct result of these anoxic episodes.

Changes in benthic diversity, abundance, and community structure could be related to toxic contamination of sediments only in areas recognized as "impacted" (e.g., the Patapsco River and the Elizabeth River). These areas are characterized by low benthic diversity and abundance, and dominance by pollution-tolerant annelids, in comparison to non-polluted reference areas. Elsewhere, other factors, primarily physical or biological, are apparently controlling benthic distributions. However, bioaccumulation of certain metals in tissues of shellfish could be correlated with enrichment of those metals in the bed sediments, even in the main Bay.

Oysters

Oysters (and other shellfish) are benthic organisms, but because of their commercial importance, oysters are treated separately here. Factors affecting benthic communities in general (i.e., low DO water, toxicants in sediments or water column) will impact oysters as well. In addition, oysters are potentialy vulnerable to shifts in phytoplankton species brought about by nutrient enrichment. Phytoplankton species usable as food can be replaced by undesirable or inedible forms. Comparison of EPA water quality criteria to measured and estimated concentrations of toxicants in the water column revealed a number of violations in the areas of oyster habitat; these were chiefly for heavy metals. Although the duration or extent of high toxicant concentrations is unknown, the observations may be significant. Some populations may be more vulnerable than others to diseases such as MSX or "Dermo." The impact of these protozoan parasites has increased in recent years because of higher salinities resulting from drought conditions.

In addition, oyster habitat may be adversely impacted by the increased rate of sedimentation in Chesapeake Bay. Beds may be buried, or spat set impeded, by deposition of sediment. Loss of once productive oyster bars in the upper Bay is probably due in part to sedimentation over the past 100 years.

Fishery Landings and Juvenile Index

Although total fishery landings have increased since 1920, the distribution of landings among species has changed significantly. Anadromous and other freshwater-spawning fish such as shad and striped bass have declined greatly; marine spawners have remained stable or increased. The finfish juvenile index, a measure of recruitment success, reflects these changes as well.

Several causes of this change in distribution have been suggested: (1) nutrient enrichment may lead to food web shifts, as suggested by changes in phytoplankton species, primarily affecting early life stages; (2) the level of toxicants, particularly heavy metals, pesticides, and chlorine, in major spawning areas are elevated and, in fact, have exceeded EPA criteria in some spawning and nursery areas used by anadromous fish; (3) habitat is being lost because of increased area of low DO water; (4) adverse climatic conditions (freshwater inflow, temperature, etc.) have reduced the spawning success of anadromous species; (5) overfishing is affecting stock sizes; (6) construction of dams represents a physical obstruction that impedes spawning success of shad and other anadromous species; and (7) modifications of upstream spawning and nursery habitat, such as wetlands destruction and stream channelization, further stresses fishery stocks. It is possible that all of these factors are contributing to the changes observed in Chesapeake Bay resources.

STATE OF THE BAY

The Chesapeake Bay Program developed a numerical ranking system to assess status of certain water quality and living resource variables to provide a snapshot of the state of the Bay. Nutrients, toxic materials, and submerged aquatic vegetation were used as the primary classification variables because of their importance and the completeness of data. Variables were ranked numerically by Bay segment. Nutrient and toxicant (metal contamination) ranks were summed to give water and sediment quality status Baywide. Similarly, resource ranks were summed to assess status according to living resources.

When ranks for water and sediment quality are aggregated, the following assessment can be made: the poorest water and sediment quality now occurs in the upper Bay, upper reaches of the Potomac and James Rivers, and the Patuxent River. The lower Potomac and James Rivers, the mid-Bay, and most of the York and Rappahannock Rivers, and the Eastern Shore tributaries exhibit moderate water and sediment quality. The lower Bay main stem, Mobjack Bay, Pocomoke Sound, lower Choptank River, and Eastern Bay currently have the best water and sediment quality.

In terms of resource quality, the following assessment is made: The western shore shows a general pattern of decline in certain resources. In fact, the upper Patuxent, lower James, and Patapsco and Middle Rivers showed poor ranks. Tangier Sound and the lower and mid-Bay display moderate resource quality. The Eastern Shore generally appears to be the most productive region of the Bay system. This conclusion is supported by the regional comparison; Eastern Bay, and the Choptank and Chester Rivers ranked most favorably.

When one compares the rank of water quality to the ranks of living resources, it is evident that areas with the best resource quality correspond to areas with the best water and sediment quality (eg., eastern embayments). This suggests that, to improve the quality of life in the Bay, water and sediment pollution must be reduced. Achieving this goal will require the concerted effort of everyone.

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TABLES

TECHNICAL SYMBOLS and GLOSSARY

Ag	silver
Al	aluminum
As	arsenic
ATP	adenosine triphosphate
Cd	cadmium
Co	cobalt
Cr	chromium
Cu	copper
DO	dissolved oxygen
Fe	iron
Hg	mercury
IPF	inorganic filterable fraction
JTU	Jackson turbidity unit
m	meter
mg	milligram
MGD	millions of gallons per day
ug L^{-1}	microgram per liter
ml L^{-1}	milliliter per liter
mg L^{-1}	milligram per liter
Mn	manganese
Mo	molybdenum
MPN	most probable number
MSX	*Minchinia nelsoni*, a sporozoan, also name given to disease in oysters caused by this sporozoan
ng	nanogram (one billionth of one gram)
NH_3	ammonia
Ni	nickel
NO_2	nitrite
NO_3	nitrate
PAH	polynuclear aromatic hydrocarbon
PAR	photosynthetically active radiation
Pb	lead
PCB	polychlorinated biphenyl
POTW	publicly owned treatment work
ppb	parts per billion
ppm	parts per million
SAV	submerged aquatic vegetation
Sc	scandium
Si	silicon
TKN	total Kjeldahl nitrogen
TN	total nitrogen
TP	total phosphorus
U	uranium
Zn	zinc

arenacious	having shell or test constructed of sand grains, as of foraminifera
bioturbation	the disturbance of the environment (i.e., bottom sediments) through biologic activity
calcareous	of, like, or containing calcium carbonate, calcium, or lime
depth averaged	arithmetic average of two or more samples taken at different depths
diagenesis	physical and chemical changes occurring to sediments during and after the period of decomposition up until the time of consolidation
hypoxia	containing low levels of oxygen, as in water or body tissues
isopleth	the line connecting points on a graph or map that have equal values with regard to certain variables
oxycline	region of relatively rapid change in dissolved oxygen concentration with depth, usually dividing upper well-oxygenated waters from lower poorly oxygenated waters
^{210}Pb profiles	radioactivity of the lead 210 isotope in sediments, with depth
planimetry	the measuring of the areas of plane forms
sciaenid	any of a family of mostly saltwater fishes, including drums and croakers, that make drumming or rumbling sounds

INTRODUCTION

Characterizing an environment, that is, pulling together data in a way that identifies functional relationships between various components of an ecosystem, is a management tool used by many resource managers. The concept stems from a growing knowledge that man's utilization of the environment brings about major, yet often subtle, changes in how the system works. An extensive series of coastal characterizations is presently being done by the U.S. Fish and Wildlife Service. Other groups that have used this approach with great success include the International Joint Commission on the Great Lakes and the Thames Water Authority in Great Britain. Several partial assessments have been done on Chesapeake Bay (Cronin 1983, Lippson 1979, McErlean et al. 1972, Mackiernan et al. 1982, Mihursky et al., 1981, Shea et al. 1980, U.S. Army Corps of Engineers 1973, U.S. Army Corps of Engineers 1975). Although these studies begin to describe ecological interactions within the Bay, the picture is incomplete. An important tool for expanding the existing ecological information on the Bay is an assessment of trends in water quality and of how living resources have changed over time.

Characterizing the quality of the Chesapeake Bay water and sediment, and changes in resources, began as a response to the Chesapeake Bay Program's (CBP) 1976 Congressional directive. According to the mandate, principal factors altering the Bay's quality were to be addressed. As part of this assessment, a set of analyses was used allowing a description of the Bay's water quality and living resources and how these have changed over the past 25 years to be developed.

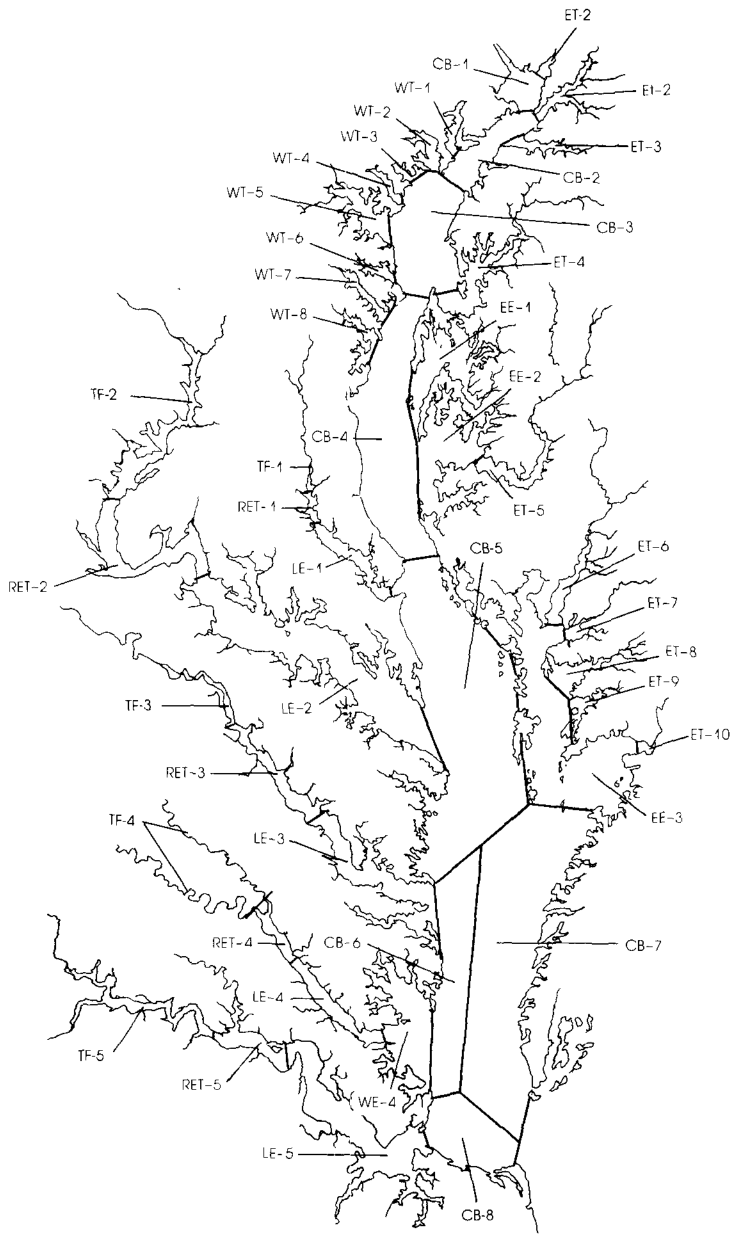
This characterization provides more than a description and synthesis of information, however; it also tries to link changes occurring in the Bay to potential causes. For example, one analysis attempts to relate nutrient concentrations in various areas to the abundance of Bay grasses living there. Making such potential linkages will help us better understand resource changes that may occur in water receiving nutrients or other pollutants. Effective strategies to control pollutants can then be developed.

Characterization has produced additional benefits for water quality managers. Segmentation, the division of the Bay into regions with similar features, allows data to be logically assembled, mapped, and classified. It allows us to assess past and present conditions and to compare trends in various regions (Appendix A). In addition, the understanding we have gained of water quality processes and their relationships to resources will help shape the program's recommended monitoring strategies. Finally, characterization has identified gaps in information and, thus, points to methods for improving future data collection.

The Program tried to approach characterization from a viewpoint that stresses the understanding of the Bay as an ecosystem. The Chesapeake ecosystem possesses many individual components linked through a variety of physical, chemical, and biological processes. Benthic communities, for example, are closely linked to the sediment environment and processes while anadromous fish depend on tidal freshwater spawning areas. Because of these interrelationships, the individual components can complement one another to maintain functioning of this ecological system. However, these links also allow effects of perturbations, such as pollution, to be proliferated through the system.

Although this ecosystem view tells us not to

Segments of Chesapeake Bay and Their Principal Characteristics



Segment	Characteristics
Tidal-fresh reaches	
Ches Bay N. (CB-1) Up Patuxent (TF-1) Up Potomac (TF-2) Up. Rapp (TF-3) Up. York (TF-4) Up James (TF-5)	• dominated by freshwater inflow of the river system • spawning areas for anadromous and semi-anadromous fish • resident habitat for freshwater fish • dominated by freshwater plankton and aquatic vegetation
Transition zones	
Up Bay (CB-2) M Patuxent (RET-1) M. Potomac (RET-2) M Rapp (RET-3) M York (RET-4) M. James (RET-5)	• slight salinity (3–9 ppt, mean) influence • zones of maximum turbidity where suspended sediment causes light limitation of phytoplankton production most of the year • areas are valuable sediment traps, concentrating material associated with sediments including absorbed toxic chemicals
Lower estuarine reaches	
Up C. Bay (CB-3) L Patuxent (LE-1) L Potomac (LE-2) L Rapp. (LE-3) L York (LE-4)	• upstream limit of deep water anoxia • moderate salinity (7–13 ppt, mean) • two-layer, estuarine circulation driven primarily by freshwater inflow
L James (LE-5) Sec W Tribu (WT-1-8) E S Tribu (ET-1-10)	• weaker estuarine circulation characterized by limited flow/flushing characteristics • water quality controlled by the density structure of the main stem of the Bay at the tributary mouth
Lower Main Bay	
Chesapeake Bay Lower Central (CB-4)	• water deeper than 30′ usually experiences oxygen depletion in summer—can be toxic to fish, crabs, shellfish and benthic animals • mean salinity of 9 to 14 ppt • rich in nutrients
Chesapeake Bay South (CB-5)	• influenced by inflow from Potomac and Patuxent and rich in nutrients • mean salinity of 10 to 17 ppt • subject to summer anoxia and contains most of the deeper Bay waters
Chesapeake Bay General West (CB-6)	• net southward flow • mean salinity of 14 to 21 ppt
Chesapeake Bay General East (CB-7)	• net northward flow • mean salinity of 19 to 24 ppt
Chesapeake Bay Mouth (CB-8)	• net southeastward flow • mean salinity of 19 to 23 ppt

Embayments	
E Bay (EE-1) L Choptank (EE-2) Tangier Sound (EE-3) Mobjack Bay (WE-4)	• have salinities similar to adjacent Bay waters • shallow enough to permit light penetration for submerged aquatic vegetation growth • influenced strongly by wind patterns

Estuaries have a capacity to assimilate waste before experiencing significant ecological damage, but, this ability can vary dramatically from one area to another To assess water quality of areas with similar characteristics the CBP divided the Bay into regions, or segments using natural processes such as circulation and salinity These 45 segments were used as a framework to map and evaluate past and present conditions of Chesapeake Bay ignore the many other components while assessing the condition of one, certain assumptions were made to approach the task practically. First, it was assumed that areas with similar physical and chemical characteristics will behave similarly. This allowed a comparison of trends and responses in similar segments. It was also assumed that information gained from simple systems, such as laboratory microcosms, can be related to the natural environments of the Bay (with proper caution). Furthermore, the limitations of available data were recognized and, consequently, so were uncertainties of some of the analyses. Sometimes, data were compared that were not collected at the same time or place, or for the same purposes. Nor were the effects of human intervention always separated from natural variability in the analyses.

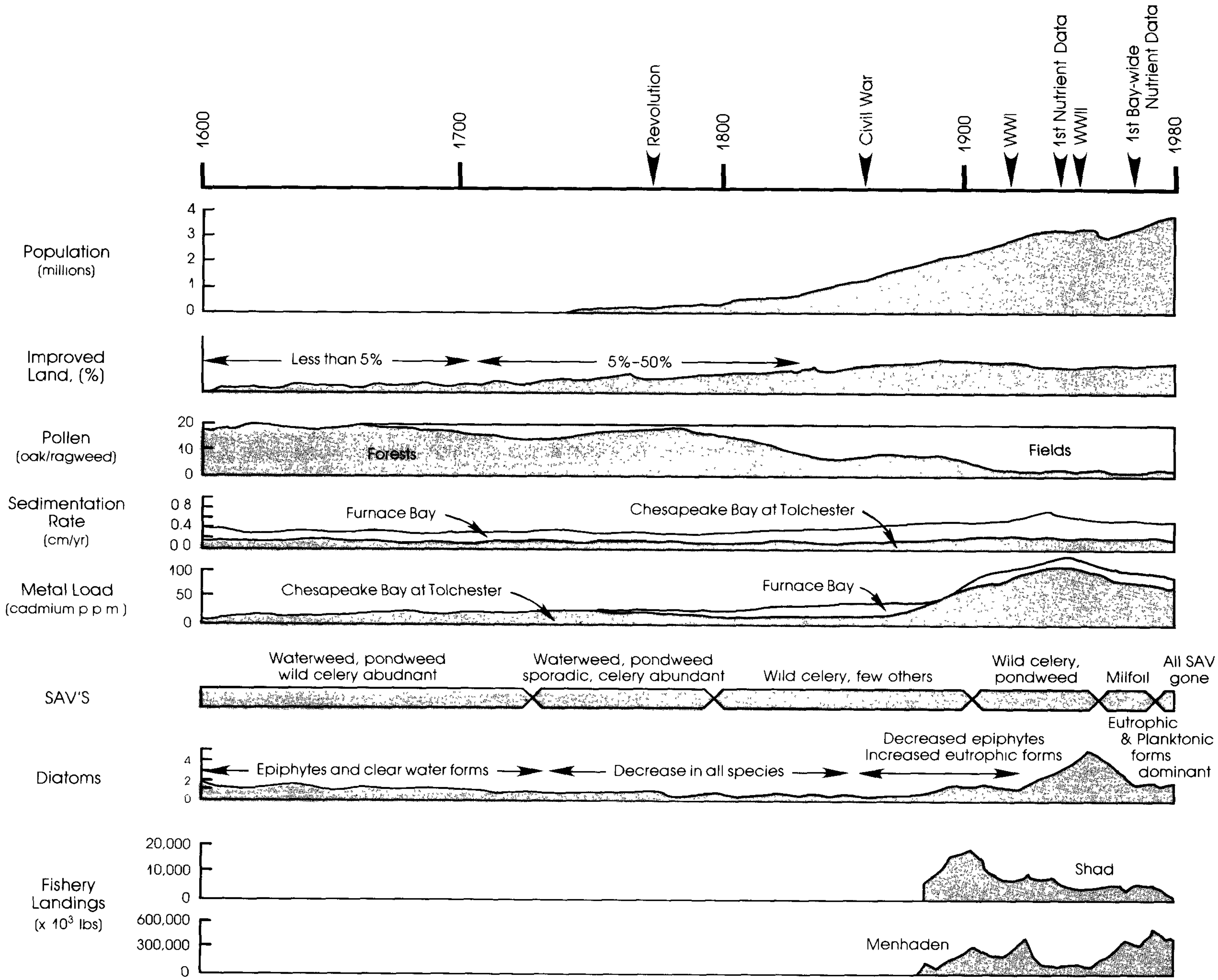
In interpreting results, only statements from a narrow historical standpoint—a slash of the Bay's history—can be made. As the chart shows, many aspects of the Bay's environment had already been altered significantly by the time scientific monitoring and research began. The metal supply to Chesapeake Bay, for example, began to increase considerably about the time of the Civil War, peaking shortly after World War II. This suggests that some benthic communities have been exposed to higher-than-natural levels of certain metals for over 50 years or more. Changes in land-use, including clearing of forest for farmland, had been significant by the mid-1800's. Sedimentation rate had increased greatly over the same time span. Increased sediment and nutrient loads caused alterations in species dominance, abundance of diatoms, and in submerged aquatic vegetation (SAV). A reduction in clean-water diatom populations in the 1800's was the first clear signal that nutrient enrichment of the upper Bay was occurring.

Recognizing the sense of the ecosystem perspective as well as the limitations imposed by practicality, the Program set out to determine present conditions and trends in about 45 areas of the Bay. The CBP tried to assess what had created those conditions. A great deal of present and historical data were collected for statistical analysis (Appendix A contains a summary of statistical analyses). From that point, current conditions and trends in each segment, or groups of segments, were looked at, and hypotheses linking the trends in water quality to those observed in resources were set forth. Where enough information was available, some cause-and-effect relationships in the Bay are explained, and others suggested. Where possible, the portion of the cause that is most likely human-related is shown.

The following report is organized into three main chapters with several appendices containing most of the support material used in the characterization process. The first chapter characterizes the Bay's water and sediment quality using nutrients, dissolved oxygen, fecal coliform levels, metals, and organic compounds. The second chapter looks at changes in abundance of the Bay's major resources, including phytoplankton, submerged aquatic vegetation, benthic invertebrates, shellfish, and finfish. The final chapter shows what linkages exist between the water quality parameters and resoures, particularly of submerged vegetation, benthic organisms, and shellfish.

In overview, this report provides a framework for the widest application of Program results. A list of all of the products and their relationship to the characterization report includes:

- Forty final reports on individual research projects with summaries of each report (listed in Appendix A).
- *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 that affect the quality of Chesapeake Bay.
- *Chesapeake Bay: A Framework for Action* identifies control alternatives for agriculture, sewage treatment plants, industry, urban runoff, and construction; estimates costs and effectiveness of different approaches to remedy environmental problems.

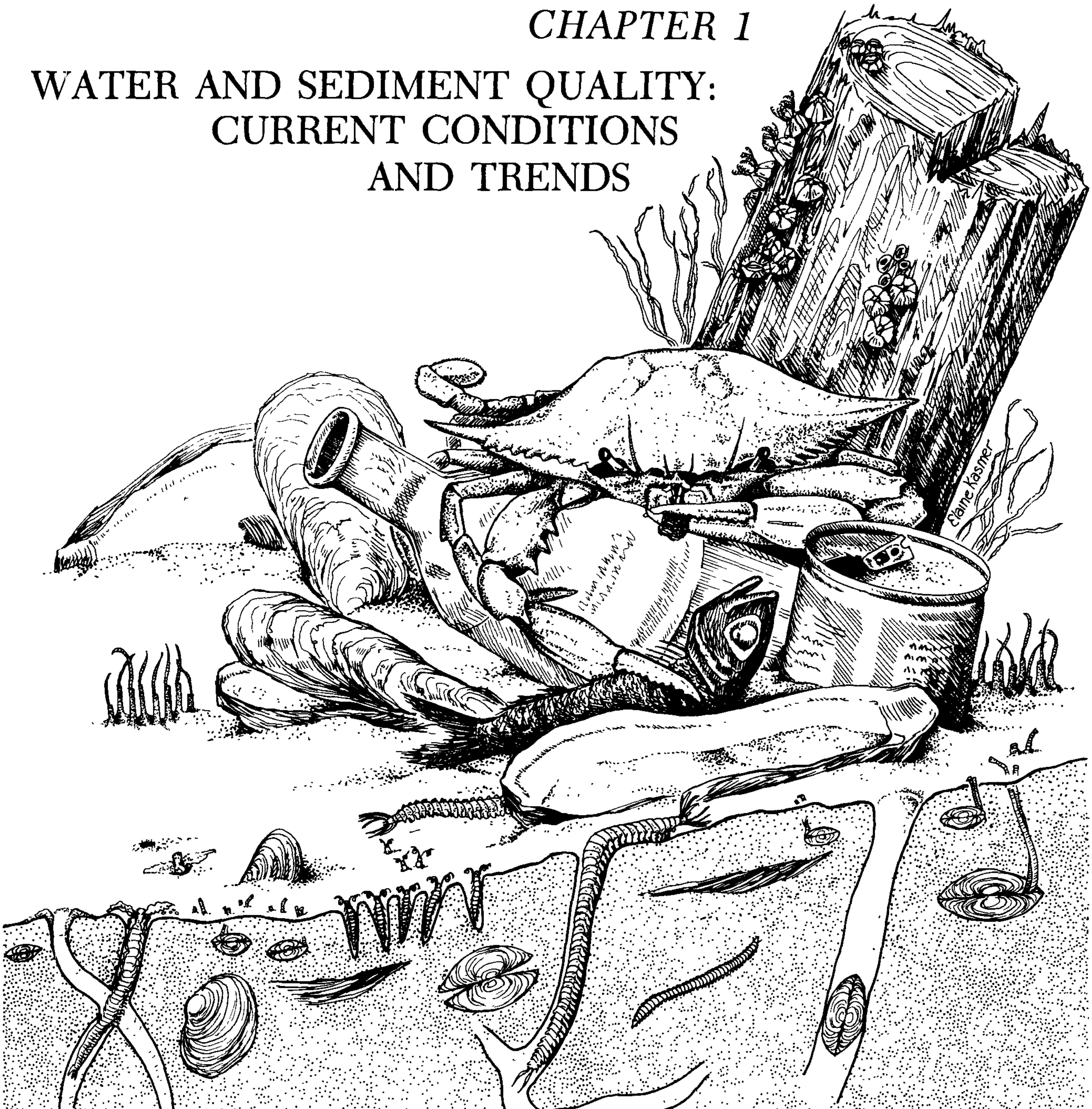


TIME HISTORY OF NORTHERN CHESAPEAKE BAY, 1600 TO 1980. An important aspect of understanding how Chesapeake Bay will respond to pollution is to examine the Bay's past. In the northern Bay, human activity, beginning at the top of the chart with population growth, has been changing water quality since the time line began (see Appendix A for further discussion).

For scientists, resource managers, and interested public, *Chesapeake Bay Program: A Profile of Environmental Change* bridges the gap between the technical, scientific results as presented in the 40 final reports and the synthesis papers, and the management report. Just as the synthesis papers provide a sound technical foundation for the CBP's characterization process, this volume provides information from which to develop management options as presented in the final report.

CHAPTER 1

WATER AND SEDIMENT QUALITY: CURRENT CONDITIONS AND TRENDS



SECTION 1

INTRODUCTION

CHARACTERIZING WATER AND SEDIMENT QUALITY

Water quality can be described as the extent to which water can support aquatic life and meet standards for various human uses. Water quality depends in part on the individual capability of a body of water to assimilate various pollutants. For an estuary such as the Chesapeake, assimilative capacity depends on the types and amounts of riverine pollutants (Table 1), and on biogeochemical processes in the Bay. When the Bay's assimilative capacity is stressed or exceeded, water and sediment quality can deteriorate. Aquatic life and man's use of the Bay can, in turn, be affected.

Rivers and streams entering the Bay carry both life-supporting materials and pollutants. They carry direct-waste discharges as well as nutrients and other chemicals draining from urban and agricultural lands of the watershed. Many of the chemicals brought into the Bay are necessary to biological processes; however, in excess they cause problems. For example, nutrients such as nitrogen and phosphorus are needed by aquatic plants, but if they occur in high concentrations, noxious algal blooms may occur. Populations of useful plankton species and submerged aquatic vegetation may be reduced as a consequence. As the algae decay, they consume oxygen in the water. If more oxygen is consumed than biologically produced by photosynthesis or made available through reaeration and mixing, the water becomes anoxic and devoid of most forms of life. In addition to nutrients, disease-carrying agents or pathogens may be associated with land runoff and sewage treatment plant discharges. Organic chemicals, heavy metals, and other toxicants from point source discharges, land runoff, and atmospheric pollution eventually enter the Bay. Once in the estuary, these substances follow a variety of fates, including entrapment in the sediments and uptake by estuarine organisms.

Two major objectives of the Chesapeake Bay Program (CBP) were to characterize the present (1980) water quality in Chesapeake Bay system and to identify long-term water quality trends in the historical data base. In the sections that follow, present conditions and trends in nutrients, dissolved oxygen, fecal coliforms, organic compounds, and metals, and how these factors may affect the quality of the Bay's waters and sediments are discussed. The Program is cautious, however, in some of the interpretations expressed, especially changes noted in many of the small tributaries and embayments where data are few, concentrated in short time intervals, or do not cover enough years to establish a firm trend. Additionally, various sorts of water quality data were rarely collected at the same time or place. Thus, present conditions and trends are based on data integrated within segment boundaries. For more than one-half of the segments, the 1980 data were inadequate so that 'present conditions' were determined using information from 1977 to 1980.

These difficulties are compounded when the historical (1950 to 1980) data are analyzed for trends. First, some analytical methods have changed over time, usually resulting in greater sensitivity and reliability in recent years. Few investigators have actually compared current and past methods to quantify the differences in results that each gives. Second, few, if any, laboratories have cross-calibrated their analytical and sample-handling procedures to determine that all obtain the same numerical results on the same samples. Third, the temporal and spatial distribution of

TABLE 1.
SOURCES AND EFFECTS OF POLLUTANTS (Adapted From Council on Environmental Quality 1981)

Type of Water	Waste Water Sources	Water Quality Measures	Effects on Water Quality+	Effects on Aquatic Life+
Suspended organic and inorganic material	Mining discharges, municipal discharges, industrial discharges, construction runoff, agricultural runoff, urban runoff, silvicultural runoff, natural sources, combined sewer overflows *shoreline erosion, *dredge spoil	Suspended solids, turbidity, biochemical oxygen demand, sulfides, *sedimentation rates.	Reduced light penetration, deposition on bottom, benthic deoxygenation	Reduced photosynthesis changed bottom organism population, reduced fish production, reduced sport fish population, increased non-sport fish population, *reduction in rooted aquatic vegetation
Nutrients—nitrogen, phosphorus and organic matter	Municipal discharges, agricultural runoff, combined sewer overflows, industrial discharges, urban runoff, natural sources, *atmospheric sources	Nitrogen, phosphorus, dissolved oxygen	Increased algal growth, dissolved oxygen reduction	Increased production, fish or shellfish kills, incr non-sport fish population reduced sport fish population *reduction in rooted vegetation, *change or elimination of benthic habitat, *loss of shellfish habitat
Disease-carrying agents—human feces, warm-blooded animal feces	Municipal discharges, watercraft discharges, urban runoff, agricultural runoff, feedlot wastes, combined sewer overflows, industrial discharges	Fecal coliform, fecal streptococcus, other microbes	Health hazard for human consumption and contact	Inedibility of shellfish for humans
Organic chemicals—e.g., detergents, household aids, pesticides, *herbicides, *industrial effluents	Industrial discharges, urban runoff, municipal discharges, combined sewer overflow, agricultural runoff, silvicultural runoff, transportation spills, mining discharges	Cyanides, phenols, toxicity bioassays, *other organics	Toxicity of natural organics, biodegradable or persistent synthetic organics, *accumulation in sediments	Fish or shellfish kills, tainted fish or shellfish, reduced reprod., *abnormal development, *bioaccumulation in food chain *elim. of susceptible species
Inorganic materials, e.g., metals, acids, other chemicals	Mining discharges, acid mine drainage, industrial discharges, municipal discharges, combined sewer overflows, urban runoff, oil fields, agricultural runoff, irrigation return flow, natural sources, cooling tower down, transportation spills, coal gasification, *acid precipitation, dredge spoil	pH, acidity, alkalinity, dissolved solids, chlorides, sulfates, sodium, specific metals, toxicity bioassays, *salinity, *total residual chlorine	Acidity, salination, toxicity of heavy metals or other chemicals, accumulation in sediments	Reduced biological productivity, fish or shellfish kills, reduced reproduction, tainted fish or shellfish, *bioaccumulation in food chain, elimination of susceptible species

+Does not show additive or synergistic effects.
*CBP additions to CEQ Table.

data are quite patchy during the 30-year period considered by the CBP. Even in years where data density is generally high, the cold months are usually poorly represented. Annual means are thus skewed toward the warmer months, and processes that allow nutrients to accumulate in the water during winter for utilization in spring cannot be clearly identified. Fourth, it is difficult and sometimes impossible to apply the most desirable statistical analyses to identify trends (e.g., time-series analysis). These shortcomings not withstanding, an attempt at trend analysis yielded some interesting and useful results for analyses done in larger areas of the Bay.

SECTION 2
NUTRIENT LEVELS AND DISSOLVED OXYGEN

INTRODUCTION

This section presents data on seasonal and spatial trends in a number of water quality variables over time. Ten water quality parameters were considered in this assessment: orthophosphate, total phosphorus, total Kjeldahl nitrogen (TKN), nitrate (NO_3), nitrite (NO_2), ammonium, total nitrogen (TN), chlorophyll *a*, (uncorrected for phaeophytin), dissolved oxygen (DO), and turbidity. The uncorrected chlorophyll *a* data were used because numerically they dominate the chlorophyll *a* data base, particularly in earlier years. Annual and seasonal trends in each variable were evaluated. Some of the nutrients, notably orthophosphate, NO_2, NO_3, and ammonium, usually exist at very low concentrations in the water column because of uptake by microbial organisms (including bacteria, phytoplankton, and small heterotrophic forms). For this reason, total phosphorus (TP) and TN concentrations (which include amounts present in phytoplankton) are often better indicators of trophic conditions than the somewhat ephemeral soluble nutrient concentrations. Exceptions exist for NO_3, especially in spring, and for the other nutrients in areas of the Bay system near the fall line or near major point sources. Total Kjeldahl nitrogen measures most, but not all, organic nitrogen and includes virtually all of the living and detrital particulate nitrogen as well as the soluble ammonium nitrogen. A general discussion of nutrient processes and effects can be found in Chesapeake Bay: Introduction to an Ecosystem (U.S. EPA 1982a).

In this section, present conditions and historical trends in nutrients and other water quality parameters are defined in terms of concentrations. Measured concentrations are actually the net result of several interacting factors, including basin topography, freshwater inflow, water circulation patterns (which influence residence time), nutrient and sediment loading rates, and many biological and chemical processes. The concentration changes measured over time are related to the dynamic balance among these interacting components. If a particular trend in nutrient concentrations suggests that the system is changing, then the processes influencing the concentrations in each segment should be evaluated when controls are considered. These processes (input rates, recycling, settling, flow) are discussed in detail in the series of synthesis papers produced by the Chesapeake Bay Program (U.S. EPA 1982b).

DATA ANALYSIS AND TREND ASSESSMENT

The present condition of the Bay was determined using an average of the annual means of the four most recent years—1977 through 1980. The following criteria were applied in calculating these means: at least three observations of each parameter were needed in each month of each year to calculate a monthly mean; a minimum of two monthly means were necessary to calculate a seasonal mean; and two seasonal means were needed to produce an annual mean. This procedure ensures at least a minimum adequacy of data before characterizing each segment of the Bay. Also, by averaging over a period of four years where data are available, the effect of year to year variability of freshwater inflow is reduced.

Trends were established by use of Pearson's correlation of each parameter over time that included all data in the CBP computer data base. The data were determined to have a significant trend if the coefficient (r) was associated with an alpha level equal to, or less than 0.05. Analysis employed the same criteria as described above for current conditions.

Only statistically significant relationships were used in the assessment of water quality trends in each segment. For this evaluation, we considered in an initial screen that if any one nutrient showed increasing concentrations over time, the overall water quality of that Bay segment should be examined further. A more thorough assessment requires a measure of total nutrient content (i.e., TN and TP). Based on present nutrient status (i.e., TN and TP), as well as potential oxygen demand, Bay segments were ranked to determine present conditions Bay-wide. The results of these assessments are summarized below.

In addition to nutrients, the problem of increased phytoplankton biomass (as reflected in chlorophyll *a* values) and decreased DO in deeper waters of the estuary is addressed. Both these parameters can be related to nutrient enrichment, and both have implications for the integrity of the Bay ecosystem.

The data concerning current conditions and historical trends for nutrients and dissolved oxygen are tabluated and discussed further in Appendix B.

Nutrients and Phytoplankton

When the amount of nutrients delivered to a water body increases, the first and most obvious response is often an increase in phytoplankton biomass. This is frequently observed in lakes; in flowing-water systems, physical dispersion may prevent build up of phytoplankton standing crop. Grazing by herbivores may also reduce algal biomass, as long as dominant phytoplankton species remain usable by consumers. In freshwater areas, proliferation of inedible blue-green algae has led to serious water quality problems. Buildup of organic material – either from algal biomass or indirectly through deposition of zooplankton fecal material – produces an oxygen demand in affected water. Areas with high photoplankton biomass (i.e., high chlorophyll *a* values) sometimes exhibit elevated amounts of diurnal DO from algal photosynthesis, followed by extremely low nighttime DO levels (Clark and Roesch 1978). Reduction in oxygen results from algal respiration and bacterial oxidation of organic material.

Growth of phytoplankton and potential biomass depends on the availability of certain chemical nutrients – primarily phosphorus and nitrogen, but sometimes silica or carbon – as well as light, temperature, and other physical variables. Nutrients are utilized in certain (atom to atom) ratios that reflect the chemical composition of the organic material produced. If one of the nutrients is missing, or is in short supply, or if light levels or temperature are at sub-optimum levels, then phytoplankton growth is limited by that factor – even if the remaining factors are abundant or favorable.

The concept that a single nutrient usually limits phytoplankton standing crop (biomass) is commonly used in water quality management. This is useful because it simplifies the application of nutrient controls. In fresh waters, phosphorus is most often the limiting nutrient (that is, the nutrient in shortest supply), so management strategies can conveniently focus on phosphorus control. However, this management concept may not apply to the Bay system as a whole. In the oceans, nitrogen is usually considered the most limiting nutrient, but as yet there is little concern over managing nutrient inputs to the ocean.

The Chesapeake Bay estuary lies between freshwater and the ocean, so we might expect it to have characteristics of both environments. Indeed, it appears that when and where riverine influence dominates, the Bay tends to be phosphorus-limited. When and where the oceanic influence dominates, the Bay is nitrogen-limited. In addition, low light and temperature may override the influence of nutrients in winter. In the upper reaches of the Bay and tributaries during the summer, turbidity can produce light-limited conditions.

Estimates of annual nitrogen and phosphorus inputs were compared with annual phytoplankton production in a variety of estuarine systems (which included data from mid-Chesapeake Bay and the Patuxent River) (Boynton et al. 1982). A reasonable relationship was identified between loading and production for nitrogen ($r^2 = 0.60$) but not for phosphorus ($r^2 = 0.08$). Boynton et al. concluded that nitrogen dynamics are more important than those of phosphorus in regulating estuarine phytoplankton production.

At an October 1982 CBP workshop on limiting factors in the Bay, scientists generally agreed on the following points:

- The best assessment is that phosphorus controls phytoplankton biomass yield in tidal-freshwater and low-salinity areas throughout the warm season; nitrogen controls phytoplankton biomass yield during the summer in higher-salinity waters. The controlling influence is probably not uniform over all higher-salinity areas of the Bay and tidal tributaries. In cold months, light and temperature are often limiting. Light is also important in turbid areas.
- Low soluble and particulate nitrogen to phosphorus (N to P) ratios suggest potential nitrogen limitation; these are typically observed during the summer in mesohaline reaches.
- In the Calvert Cliffs area, an approximate linear relationship exists between nitrogen loading from the Susquehanna River and both primary productivity and chlorophyll *a*; the same relationship does not occur with phosphorus (Boynton et al. 1982).
- Phosphorus, in particular, and probably nitrogen, play a minor role in controlling the specific growth rate of phytoplankton (as opposed to potential biomass). Important here are species-specific nutrient uptake characteristics in relation to ambient nutrient conditions or production of enzymes such as alkaline phosphatase or nitrate reductase; this has been determined from dose-response experiments and field observations.
- Physical dispersion influences biomass yield in the Bay. Where dispersion is high, biomass does not accumulate, although production of organic material may be large.
- The decrease in magnitude of the spring bloom and increase in the temporal extent of the summer bloom in the upper Bay indicate a shift from nutrient limitation to nutrient saturation of phytoplankton in summer.
- Denitrification and nitrification may be important cycling and loss mechanisms for nitrogen in the upper estuary (Kemp et al. 1982d, Seitzinger et al. 1980).

Upstream processes can affect the transport of nutrients downstream and their concentration there. Particularly in spring, when freshwater inflow is high, residence time of water in the upper Bay is short. Nutrients delivered there are not used completely, but are passed through to downstream areas. This "pulse" of spring nutrients, primarily NO_3, can be detected well into the mesohaline reaches of the estuary (D'Elia 1982, Taft et al. 1982). Results from both modeling and field studies indicate that reducing phosphorus inputs to tidal freshwater regions may allow proportionally more nitrogen to pass through the tidal-fresh regions into the saline waters. This occurs because as phosphorus is reduced, less phytoplankton biomass is produced, and less nitrogen is required for growth. So, rather than settling out as particulate organic material, the soluble nitrogen (primarily NO_3) continues downstream for potential utilization in the higher salinity reaches (Taft et al. 1978). Although reduction in riverine phosphorus loads could theoretically be offset by rapid recycling of this nutrient within the upper estuary, the short residence times would actually allow much of the nitrogen to pass downstream. In this way, the single nutrient strategy for phosphorus removal may actually allow more nitrogen delivery to downstream reaches that are usually nitrogen-limited. Increased available inorganic nitrogen will support increases in production of organic nitrogen and carbon in the mid-Bay. This in turn will result in larger biological oxygen demands in these areas.

OXYGEN DEMAND AND NUTRIENTS

One principal result of nutrient enrichment of concern to managers is the oxygen demand created by increased production of organic material. When organic matter is decomposed by bacterial action, oxygen is used. Decomposition of large amounts of organic matter can result in loss of dissolved oxygen in the water (sometimes complete anoxia occurs). Criteria describing the current condition of the Chesapeake Bay system are related to nutrient levels and corresponding potential oxygen demand. For the simplest case, planktonic organic material and its oxidation can be represented, after Richards (1965), by the following formula:

$$(CH_2O)_{106}\ (NH_3)_{16}\ H_3PO_4 + 138\ O_2 \Rightarrow$$
Organic Material

$$106\ CO_2 + 122\ H_2O + 16\ HNO_3 + H_3PO_4$$
Remineralized Nutrients

The coefficients in this relationship are multiplied by the atomic weights of the elements to obtain the masses involved. That is, for nitrogen, 16 times the atomic weight of 14 gives 224 mass units, and, for oxygen, 138 times 32 gives 4416 mass units. Converting to milligrams (mg) shows that planktonic organic material containing 0.224 mg nitrogen would require 4.42 mg oxygen for complete degradation, giving a ratio of 19.7 mg oxygen used for every 1 mg nitrogen oxidized.

This relation between organic material and the oxygen required for its complete mineralization was used to construct a table of potential oxygen concentrations and organic material concentrations expressed in terms of carbon, nitrogen, and phosphorus (Table 2). The table is entered with a nutrient concentration value: for example, 0.035 mg P L^{-1} represents a potential oxygen demand by organic material of 5 mg. Thus, a saturation DO concentration of 7 mg L^{-1} would be reduced by 5 mg to 2 mg L^{-1} if all of the organic material associated with 0.035 mg P L^{-1} were oxidized. The potential oxygen demand is expressed as mg rather than mg L^{-1} because, in some cases, the potential demand is much greater than the actual saturation oxygen concentration. Oxygen demand in excess of available dissolved oxygen will lead to the production of toxic hydrogen sulfide (H_2S).

If this approach is carried further, a total nitrogen concentration of 0.75 mg L^{-1} and a total phosphorus concentration of 0.105 mg L^{-1} would have a potential oxygen demand of 15 mg oxygen L^{-1}. However, expected oxygen saturation values at salinities and temperatures typical of upper Bay deep waters are only 12.5 mg L^{-1} oxygen in March and 9.5 mg L^{-1} in May. Thus, if total nitrogen concentrations are maintained near 0.75 mg L^{-1} in the upper Bay, the potential oxygen demand exceeds the available oxygen.

This simple approach relating nutrient levels to oxygen demand does not account for either reaeration of the water from the atmosphere or

TABLE 2.
RELATION BETWEEN ORGANIC MATERIAL, NUTRIENTS, AND POTENTIAL OXYGEN DEMAND

Organic Material Oxidized (mg L^{-1})			Potential Oxygen Demand (mg)
C	N	P	
0.28	0.05	0.007	1
1.40	0.25	0.035	5
2.80	0.50	0.070	10
4.20	0.75	0.105	15
5.60	1.00	0.140	20
7.00	1.25	0.175	25
8.40	1.50	0.210	30
9.80	1.75	0.245	35
11.20	2.00	0.280	40
12.60	2.25	0.315	45

steady accumulation of organic material in deep water (which can increase oxygen demand). Reaeration of the water column is a function of both diffusion and mixing; aeration of bottom water is reduced when strong vertical stratification exists, as occurs during warm months in much of the Bay. The net result of these interacting processes can be seen in the 1980 data set for Chesapeake Bay Institute (CBI) station 848E (off the Rhode River), where deep-water oxygen concentrations were about 10 mg L^{-1} (7.0 ml L^{-1}) in March and zero in May 1980 at all sampled depths below 11 m (Table 3). Of the total oxygen loss, a decrease of 3 mg L^{-1} was caused by temperature and salinity effects on oxygen solubility. Loss of the remaining 7 mg L^{-1} was due to biological and chemical oxidation reactions. If the above relationship is applied, this 7 mg L^{-1} oxygen loss is due to the oxidation of organic material containing 0.35 mg L^{-1} nitrogen.

This concept relating nutrients, organic material, and potential oxygen demand may be extended into a classification scheme for Chesapeake Bay and its tributaries. The classification scheme helps define a range of uses that are compatible with a given number of environmental factors. A relative ranking of the Bay segments was developed based on nutrient concentrations of TN and TP; using the above formula, the potential oxygen demand for that level of nutrients was also estimated. Table 4 assigns class 1 through 6 to total nitrogen and total phosphorus concentration ranges observed in the Chesapeake Bay Program data base. Class 1 is envisioned as a relatively pristine situation (uninfluenced by human intervention) and class 6 is the most enriched.

The issue arises as to the appropriateness of using nutrient trends to assess water quality, versus chlorophyll *a* or oxygen. In light of the relationship between nutrient loadings, plankton growth, and dissolved oxygen levels, a Bay-wide assessment using the nutrient data base was made. This relationship is reinforced by observed trends in deep-water DO concentrations at selected stations in the Bay, and in observed changes in phytoplankton biomass.

SEASONAL PATTERNS OF NUTRIENTS AND DISSOLVED OXYGEN

When data from the last four years are compared, seasonal nutrient concentrations for 1978 are generally the highest recorded; those for 1980

TABLE 3.
COMPARISON OF DO CONCENTRATIONS OVER DEPTH AT CBI STATION 848E MARCH 20, 1980 TO MAY 21, 1980. (FROM CRONIN ET AL. 1982)

March 20, 1980			May 21, 1980		
Depth (m)	DO mg L^{-1}	Oxygen saturation percent	Depth (m)	DO mg L^{-1}	Oxygen saturation percent
0.0	8.52	102.28	0.0	5.65	86.94
4.0	8.29	99.32	4.0	5.39	84.87
8.0	8.19	98.08	8.0	4.44	69.96
10.0			10.0	0.59	9.17
12.0			12.0	0.00	0.00
16.0	7.00	82.63	16.0	0.00	0.00
32.0	5.65	66.12	30.0	0.00	0.00

TABLE 4.
CLASSIFICATION OF CHESAPEAKE BAY WATER USING TOTAL NITROGEN (TN) AND TOTAL PHOSPHORUS (TP)

Class	TN $mg\ L^{-1}$	TP $mg\ L^{-1}$	Potential DO Demand mg
1	0 -0.40	0 -0.056	8
2	0.41-0.60	0.057-0.084	12
3	0.61-0.80	0.085-0.112	16
4	0.81-1.00	0.113-0.140	20
5	1.01-1.75	0.141-0.245	35
6	1.76+	0.246+	36+

are frequently the lowest (Appendix B, Section 8, Table 38). This illustrates the importance of nonpoint sources of nutrients and the impact of rainfall on nutrient loadings; 1978 was a wet year, and 1980 a dry year.

When seasonal trends in nutrient concentrations are examined, a general pattern results. In the mid to upper Bay (CB-1-3), the tidal-fresh reaches of the Potomac, Rappahannock, York, and James Rivers (TF 1-5), and the mid-Potomac (RET-2) (the segments chosen for this analysis), total phosphorus concentrations usually increase from spring to summer, then decrease through the fall and winter. Reduced nonpoint source loads, utilization of phosphorus by phytoplankton, and losses of particulate P to the sediments may account for these reductions. Increase in concentrations from spring to summer may be due to remineralization of phosphorus from bottom sediment.

Annual and seasonal nitrogen concentrations are inversely related to phosphorus concentrations in most areas: that is, increasing phosphorus concentrations are generally associated with decreasing nitrogen concentrations; decreasing phosphorus concentrations are generally associated with increasing nitrogen concentrations. For example, distinct increases in phosphorus concentrations from spring to summer are associated with decreases in nitrogen concentrations during the same period. In CB-2, CB-3, and TF-2, for example, low seasonal phosphorus concentrations observed for 1980 are associated with relatively high nitrogen concentrations for the same period.

The reasons for this phenomenon may be twofold: in tidal-fresh areas, this may reflect the reduced uptake of nitrogen by phytoplankton that occurs when the amount of the limiting nutrient available (phosphorus) is decreased. Downestuary, large nitrogen concentrations associated with a spring freshet may be reduced by phytoplankton uptake and decreased nonpoint source loadings in summer. Concentrations increase again in the fall when the photosynthetic demand for nitrogen is reduced, and the nonpoint source contribution of nitrogen increases with greater runoff. The importance of losses of particulate nitrogen to the sediments or of nitrification (and loss to the atmosphere) are not known.

The greatest range in phosphorus concentrations and the largest values generally occur in the spring. This probably reflects varying intensity of the spring freshet and the importance of nonpoint sources. Total nitrogen concentrations are at their lowest during summer. Low summer total nitrogen concentrations may be attributed to an increased demand for nitrogen coupled with a decreased supply from nonpoint sources during this time of increased primary productivity and reduced rainfall. The pattern for NO_3 and NO_2 differs from that of ammonium (NH_3) and TKN. Inorganic nitrogen (NO_2 and NO_3) declines from spring to summer, then increases through fall to a high in winter, reflecting patterns of loading and decreased phytoplankton utilization. Concentrations of NH_3 and TKN are low at all seasons and show relatively little seasonal variablility. Phytoplankton utilization probably accounts for most of the use of NH_3. The smallest range in phosphorus concentrations generally occurs in the fall. Reduced ranges in concentration during the fall indicate more stable conditions. Influence of erratic runoff from nonpoint sources in spring is reduced, but steady point source load continues in all seasons. The tidal-fresh areas of the upper tributaries are where publicly owned treatment works (POTWs) and industries are concentrated, and they account for a significant portion of phosphorus load (58 percent) to the Chesapeake Bay tidal system (U.S. EPA 1983).

The Patuxent River is a special case; in all seasons, highest nutrient concentrations are found in the tidal-fresh portion of the river. There is a direct (as opposed to inverse) relationship between phosphorus and nitrogen concentrations; both increase throughout the year and then decrease from fall to winter. This reflects the importance of POTW loadings in this river, particularly during seasons when runoff is reduced.

In general, chlorophyll concentrations increase from spring to summer, except in lower Bay areas. Values in the fall are also often high, similar to or slightly below that of summer. Not unexpectedly, chlorophyll *a* concentrations are lowest in winter.

Dissolved oxygen shows a typical pattern, reflecting effects of temperature, salinity, and biological processes. Values are highest in winter, declining through spring to summer lows, and then increasing through fall to winter.

PRESENT STATUS AND TRENDS OF NUTRIENTS AND DISSOLVED OXYGEN

Present Status

Depth-averaged means for TN and TP, for all data from the years 1977 to 1980, are given in Table 5 for all segments where data meet

TABLE 5.
DEPTH-AVERAGED MEANS OF ALL YEARS FROM 1977 to 1980 OF TOTAL NITROGEN (TN) AND TOTAL PHOSPHORUS (TP) FOR ALL BAY SEGMENTS MEETING DATA MINIMUM CRITERIA. NOTE: SOME MEANS REPRESENT FEWER THAN FOUR YEARS (n=YEARS REPRESENTED IN MEANS)

Segment	TN	n	TP	n	Segment	TN	n	TP	n
CB-1	1.508	2	0.094	3	TF-3	0.916	2	0.151	2
CB-2	1.240	4	0.109	4	RET-3	0.556	1	0.109	2
CB-3	1.261	4	0.095	4	LE-3	0.563	1	0.078	2
CB-4	0.923	3	0.078	4	TF-4	0.617	2	0.123	2
CB-5	0.970	3	0.065	3	RET-4	—	—	—	—
WT-2	1.456	1	0.062	1	LE-4	—	—	0.085	2
					TF-5	1.138	1	0.184	2
WT-5	1.790	3	0.110	3					
WT-6	0.570	1	0.076	1	LE-5	0.624	2	0.101	2
WT-8	0.803	1	0.100	1	ET-2	1.229	1	0.101	1
TF-1	2.188	2	0.420	2	ET-4	0.909	2	0.178	2
RET-1	1.041	1	0.147	2	ET-5	1.09	2	0.106	2
LE-1	0.959	1	0.126	1	ET-6	1.066	1	0.081	1
TF-2	0.666	4	0.140	4	ET-7	1.440	1	0.132	1
RET-2	1.208	4	0.131	4	ET-10	0.942	1	0.090	1
LE-2	0.515	1	0.062	2	EE-1	0.589	1	0.091	1
					EE-3	0.696	1	0.064	1

minimum criteria. Four-year annual and seasonal means for all water quality variables are tabulated and included in Appendix B, Section 8, Tables 35 to 38. It should be noted that some segments lacking TN or TP information have data for other forms of nitrogen or phosphorus. It was sometimes necessary to use these data in assessing nutrient trends in such areas. For comparative purposes, TN and TP concentrations in segments with incomplete data (i.e., not recent or not meeting minimum criteria to compute annual means) are included in Table 6.

When Bay segments are ranked on the basis of TN and TP, according to the system discussed previously, the following assessment can be made: upper and mid-Bay, main stem, upper western shore tributaries, tidal-fresh reaches, and some riverine-estuarine transition zones of major tributaries have total nitrogen concentrations that are characteristic of class 5 or 6 water (Table 7 and Figure 1). Total phosphorus concentrations characteristic of class 5 and 6 waters tend to occur in upper reaches of major western shore tributaries. Water quality generally improves down-estuary and down-tributary to class 3 or 4. Eastern embayments (EE segments) and small western shore tributaries south of the Patapsco River also have relatively low ambient nutrient concentrations, class 3 or 2 for both nitrogen and phosphorus (Table 7 and Figure 2a). Insufficient TN and TP data exist to rank the Virginia main Bay segments (CB-6, 7, and 8); the few observations available, however, are low.

Chlorophyll concentrations, a measure of phytoplankton biomass, have a pattern similar to that of nutrients: higher in the upper Bay, tidal-fresh, and transition areas of major tributaries, and in certain smaller western shore tributaries than in

TABLE 6.
MEAN VALUES OF TOTAL NITROGEN (TN) AND TOTAL PHOSPHORUS (TP) IN BAY SEGMENTS WHERE INSUFFICIENT RECENT DATA EXIST TO COMPUTE ANNUAL MEANS. YEAR AND MONTH(S) WHEN DATA WERE COLLECTED ARE INDICATED.

Segment	TN	TP	Date
CB-7	0.008		1979, months 2, 8
EE-2	0.57	0.064	1976, months 5, 6, 8
ET-1	1.00	0.07	1976, mean of spring and summer
ET-3	0.744	0.070	1977, months 6, 8
		0.091	1979, summer
ET-8	0.672	0.081	1976, month 7
ET-9	0.734	0.051	1976 month 7
RET-4	0.593	0.172	1979, months 3, 4, 6
	0.566	0.114	1978, months 7, 8
RET-5	1.015	0.106	1979, months 7, 9
	1.128	0.119	1978, months 7, 8
WE-4	0.60		1976, month 9
		0.06	1977, months 4, 7
		0.066	1978, month 5
WT-1	1.20	0.050	1977, summer
WT-4	5.32		1980, summer
		0.368	1976, 1977, June mean
WT-7	0.454	0.069	1977, summer
	1.347	0.077	1978, winter
LE-4	0.568	0.084	1978, months 7, 9

TABLE 7.
PRESENT RANK OF CHESAPEAKE BAY SEGMENTS ACCORDING TO NUTRIENT CONCENTRATIONS BASED ON DEPTH-AVERAGED MEANS OF 1977 TO 1980 (TABLE 4). RANKS ASSIGNED ON THE BASIS OF INCOMPLETE DATA ARE INDICATED WITH AN ASTERISK (*)

Segment	TN	TP	Segment	TN	TP
CB-1	5	3	WE-4	2*	2*
CB-2	5	3			
CB-3	5	3	TF-1	6	6
CB-4	4	2	RET-1	5	5
CB-5	4	2	LE-1	4	4
CB-6	ND	ND			
CB-7	1*	ND	TF-2	5	4
CB-8	ND	ND	RET-2	5	4
			LE-2	2	2
WT-1	5*	2*			
WT-2	5	2	TF-3	4	5
WT-3	ND	ND	RET-3	2	3
WT-4	6*	6*	LE-3	2	2
WT-5	6	3			
WT-6	2	2	TF-4	3	4
WT-7	2	2*	RET-4	2*	3
WT-8	3	3	LE-4	2*	3
EE-1	2	3	TF-5	5	5
EE-2	2*	2*	RET-5	5*	5*
EE-3	3	2	LE-5	3	3
ET-1	4*	2*	ET-6	5	2
ET-2	5	3	ET-75	4	
ET-3	3*	2*	ET-8	3*	2*
ET-4	4	5	ET-9	3*	2*
ET-5	5	3	ET-10	4	3

the more saline regions (Appendix B, Section 8). Summer chlorophyll levels are typically higher than in spring, except in CB-4 and 5, which show a pronounced spring bloom pattern. Data from 1979 show that the same cycle occurs in CB-7. Heinle et al. (1980) suggest that it is useful to consider three levels of chlorophyll *a* concentrations in the Bay: $\leq$30 ug L^{-1} (low); 30 to 60 ug L^{-1} (medium); and >60 ug L^{-1} (high). There is insufficient information in CB-6 and 8 to assess current conditions or phytoplankton biomass cycles.

Analysis of measured spring and summer Jackson turbidity unit (JTU) and Secchi disk measurements taken since 1960 in the tidal-fresh and lower estuarine portions of the major western tributaries and main Bay segments CB-1 through CB-4 does not reveal any trend in water clarity from year to year. However, there does appear to be a recent change in seasonal water clarity. For the most part, pre-1972 JTU measurements indicate that more turbid conditions exist in spring than summer. More recent JTU measurements

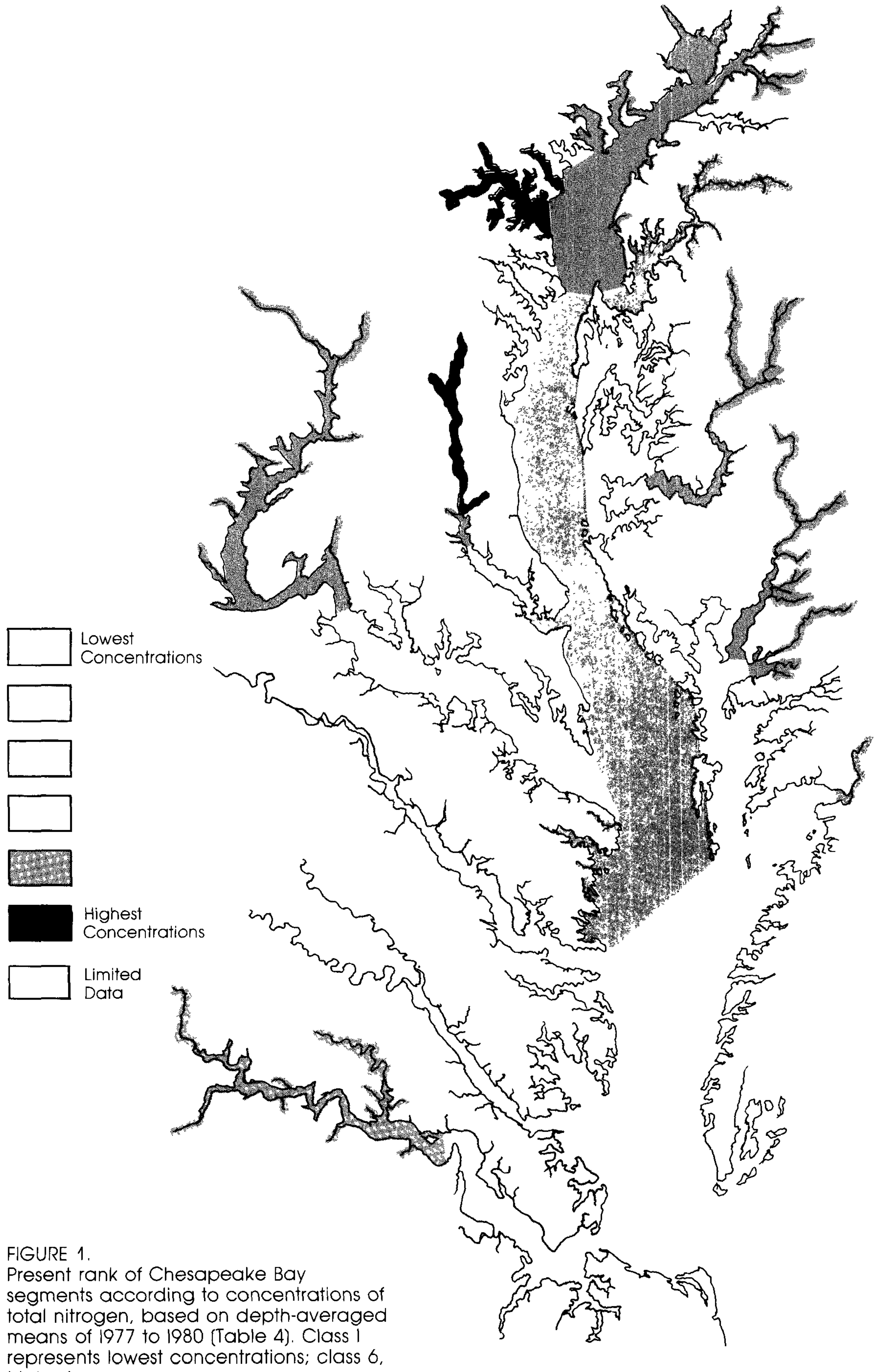


FIGURE 1.
Present rank of Chesapeake Bay segments according to concentrations of total nitrogen, based on depth-averaged means of 1977 to 1980 (Table 4). Class 1 represents lowest concentrations; class 6, highest.

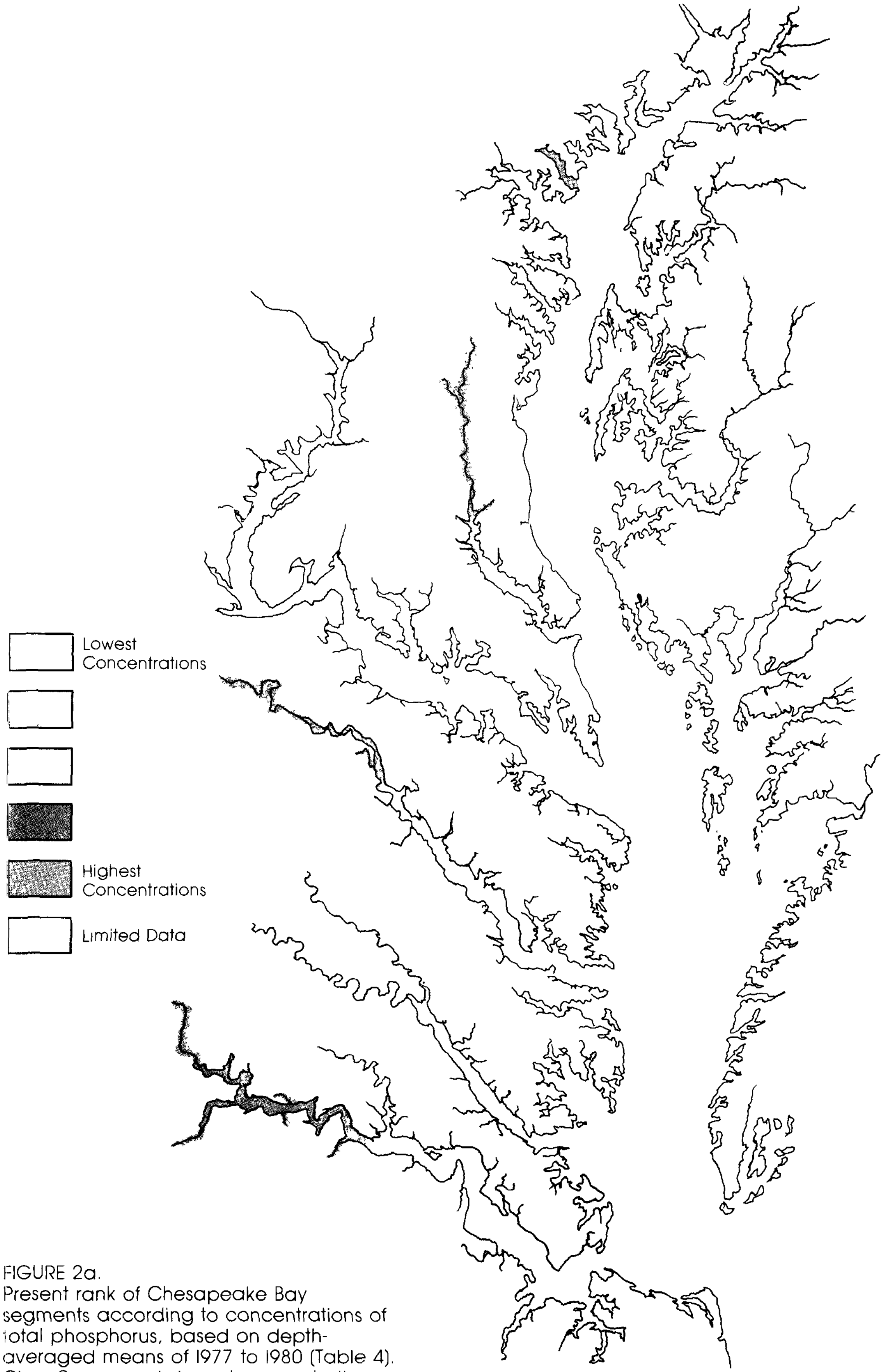


FIGURE 2a.
Present rank of Chesapeake Bay segments according to concentrations of total phosphorus, based on depth-averaged means of 1977 to 1980 (Table 4). Class 2 represents lowest concentrations of TP; class 6, highest.

and Secchi disk measurements, however, indicate more turbid conditions in summer than spring. The increased turbidity in summer waters may reflect increased algal growth reducing water clarity and light transmission.

Tidal-fresh areas are generally more turbid than lower estuary areas. The tidal-fresh Patuxent was the most turbid; the lower Potomac was the least turbid of all areas examined. There are very little data in the Rappahannock and York Rivers.

The present conditions of the Bay for nutrients and chlorophyll *a* are assessed based on partial spatial coverage within any segment. To provide a better idea of the extent of spatial coverage, as an example, a map of the depth-averaged coverage of annual means (1977 to 1980) for total phosphorus displayed by USGS 7 1/2-minute quadrangle (Figure 2b) is shown. Maps of total nitrogen (depth-averaged) and surface concentrations of chlorophyll *a* (i.e., <10 m) are shown for annual, spring, and summer means for 1977 to 1980 in Appendix B (Section 8).

Nutrient Trends

Nutrient trends were examined for increases or decreases over the period of record. (Note: this differs from current status that included data from 1977 to 1980.) Most data on nitrogen and phosphorus distribution began in 1964 with a few studies occurring in the late 1930's (Newcombe and Brust 1940, Newcombe and Lang 1939, Nash 1947). Chlorophyll *a* (uncorrected for decomposition products), an index of phytoplankton standing crop, is included in this assessment. Earliest data on chlorophyll *a* were collected in the 1950s.

Trends in nitrogen and phosphorus were evaluated to see if any form of these nutrients showed a statistically significant change over time at an alpha level equal to, or less than 0.05, using Pearson correlation. Trends are summarized in Appendix B (Tables 40 and 41). Sample sizes of less than 5 were excluded in the analysis. This simple screening approach was judged to be a useful starting point but, as discussed later, a more meaningful assessment requires an estimate of the total nutrients (i.e., TN and TP). A case can be made for the inclusion of silica, a key constituent of diatoms, in the trend assessment, but historical data are poorly represented in the data base (D'Elia et al. 1983).

Initially the data were examined for trends occurring during any season or annually. For this screening assessment, we described phosphorus by both its inorganic filterable fraction (IPF) and TP; the latter includes particulate and dissolved organic fractions as well as IPF. In theory, the IPF is immediately available to phytoplankton; the organic forms can undergo remineralization and eventually become available to phytoplankton. Concentrations are averaged throughout the water column because mixing can eventually bring phosphorus at depth in contact with phytoplankton. Several regions of the main Bay, such as CB-1, 2, 3, and 5, Tangier Sound (EE-3), and a few segments in some tributaries, showed an increase in TP or IPF during one or more seasons (Figure 3). Segments that showed only a decreasing trend are also represented. The decreasing trends in the James River estuary, the upper tidal freshwater Potomac River, and mid-Bay in Maryland (CB-4) are notable.

Similar to phosphorus, nitrogen was examined in the screening mode; nitrogen is described in terms of inorganic forms, NH_3, NO_2, NO_3, and TKN, which includes organic nitrogen and ammonia-nitrogen. The sum of these components is TN after accounting for the free ammonia in the TKN component. Examination of nitrogen trends in a screening mode showed an increasing trend in either TN or in any form of nitrogen seasonally in the following segments: CB 1-4, ET-2, ET-4, ET-6, ET-7, TF-1, LE-1, TF-2, RET-2, LE-2, and TF-4 (Figure 4). A decreasing trend, not accompanied by any increasing trends, occurred only in the Patapsco River (WT-5), the Northeast River, and the James River estuary (TF-5 and LE-5).

Trends in both nitrogen and phosphorus were considered together, with any increasing trend in any form taking precedence over a decreasing trend. With that assessment, most segments show an increasing trend in at least one nutrient type (Figure 5).

However, an assessment based on trends in individual nutrient forms (e.g., IPF or NO_3-nitrogen) is not a straight-forward procedure. The increase or decrease in IPF or NO_3 is not necessarily a rigorous index of an improving or degrading

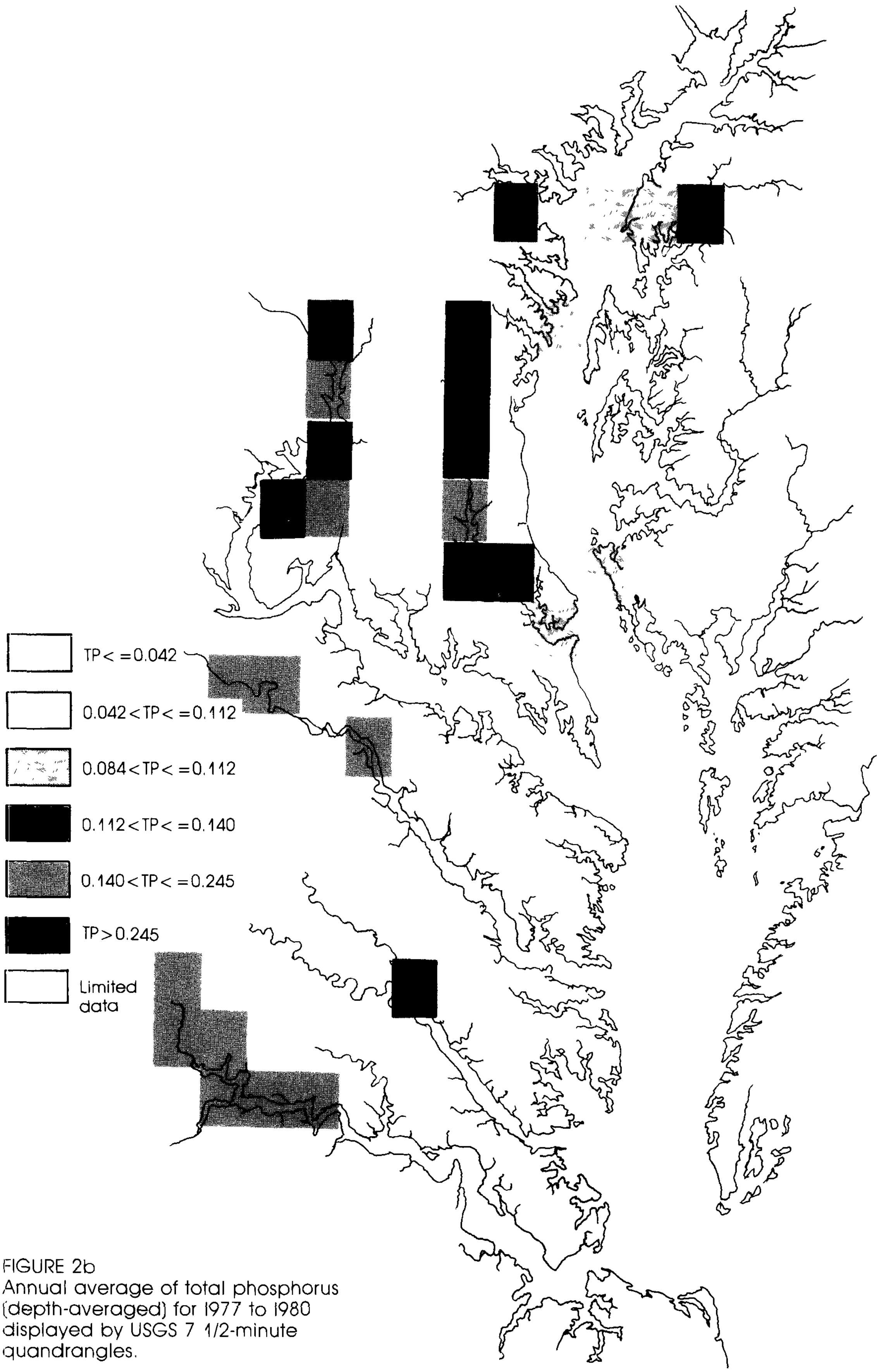


FIGURE 2b
Annual average of total phosphorus (depth-averaged) for 1977 to 1980 displayed by USGS 7 1/2-minute quandrangles.

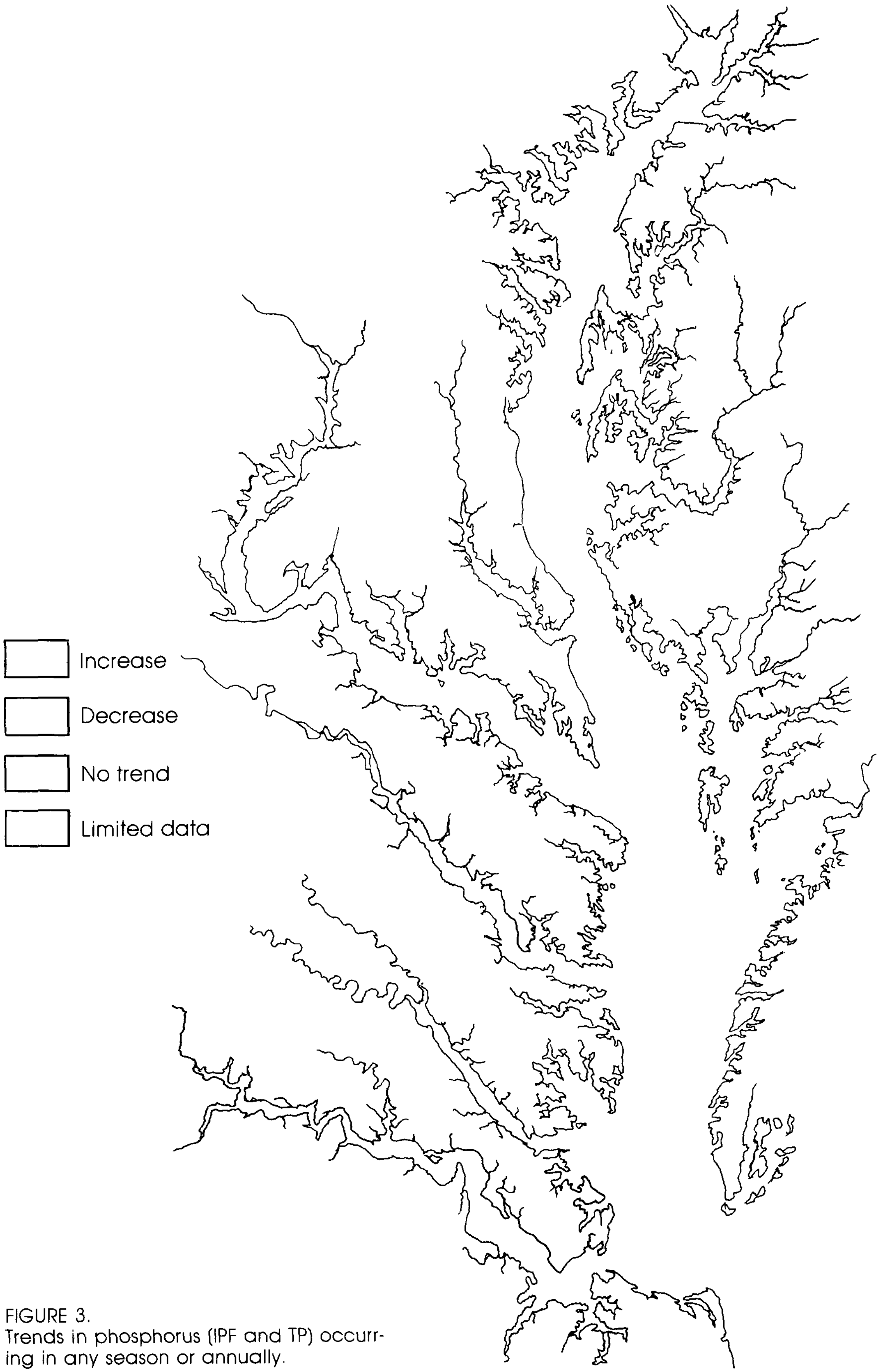


FIGURE 3.
Trends in phosphorus (IPF and TP) occurring in any season or annually.

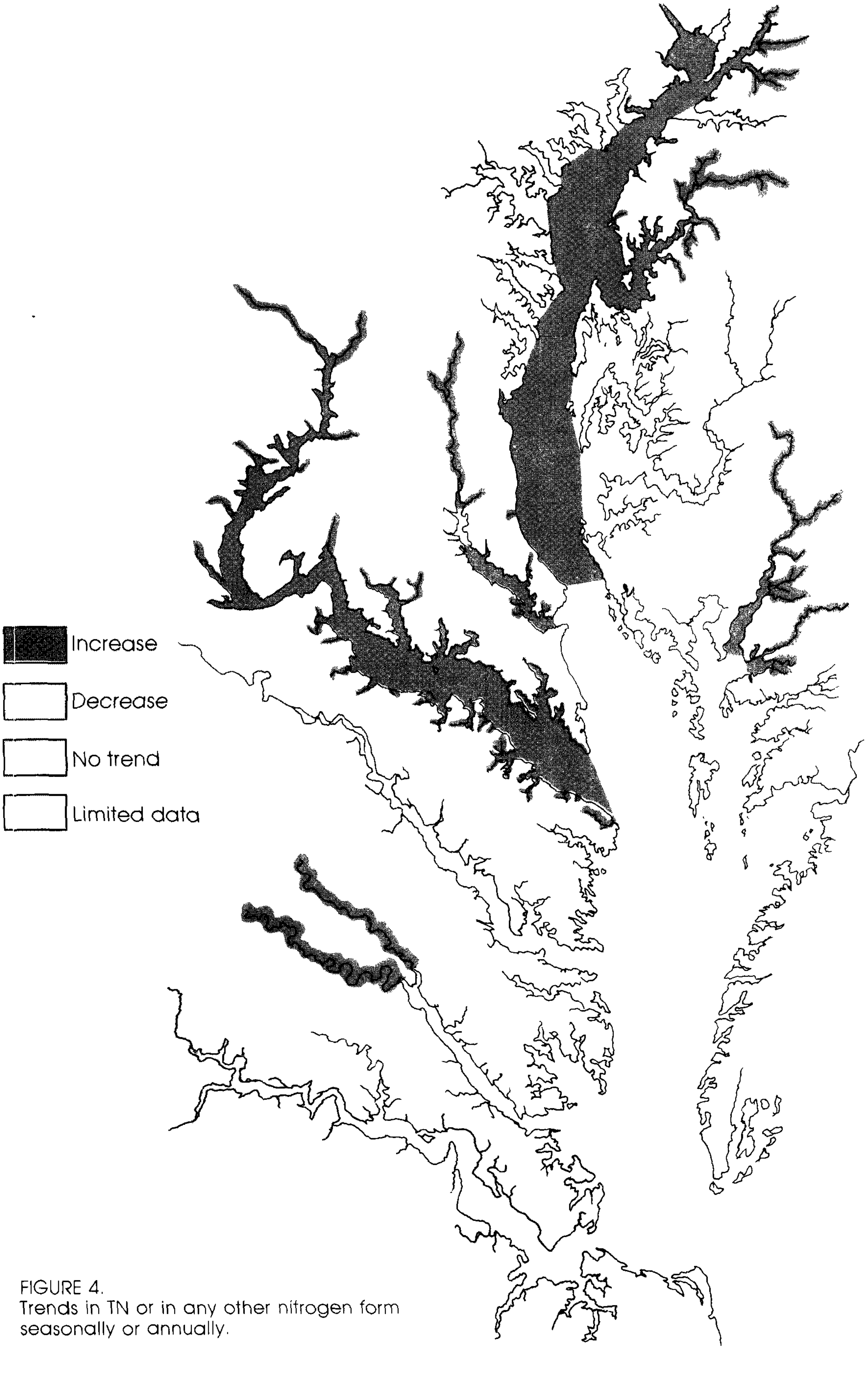


FIGURE 4.
Trends in TN or in any other nitrogen form seasonally or annually.

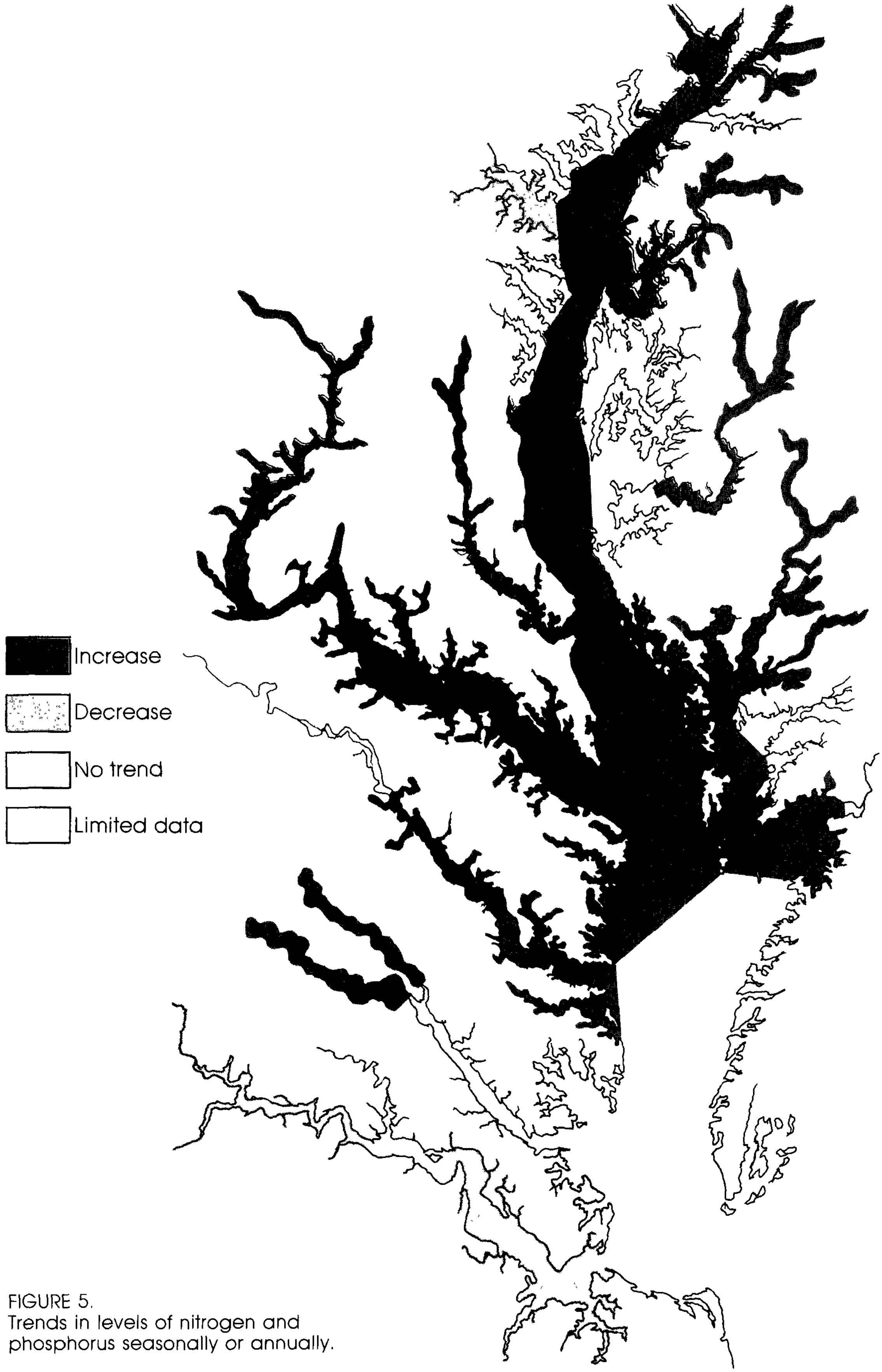


FIGURE 5.
Trends in levels of nitrogen and phosphorus seasonally or annually.

water quality situation. Nitrate can increase as a consequence of nitrification, a microbial process, while the TN may remain unchanged. Organic nitrogen can decrease in concentration through natural remineralization processes or upgraded sewage treatment processes with a subsequent increase in NO_3. However, depending on the relative change, the total nitrogen concentration may actually decrease. In a similar vein, organic phosphorus may decrease while IPF shows an increase. Thus, it is desirable to have a measure of the total nutrient content (i.e., TN, TP) for an ecological assessment. Unfortunately, the nutrient data base in the Chesapeake Bay and tidal tributaries is relatively poorly characterized in terms of these total forms. This should be addressed in future monitoring activities.

It is probably more biologically meaningful to examine trends in individual nutrient fractions because of differences in availability and preference for phytoplankton. Individual trends in total phosphorus and IPF are summarized in Figure 6a. Annual changes (representing at least two significant seasonal trends) show an increasing trend in CB-1, CB-2, and CB-5 and a decreasing trend in the tidal Potomac (TF-2). Note that relatively few segments are represented, compared to results from the less constrained screening approach. Quite a few segments showed no significant trends, although samples were sufficient for statistical analysis (Appendix B, Section 8).

Seasonal changes are shown in Figure 6b. Eight segments show an increase in IPF for one or more seasons: the lower Potomac (LE-2), mid-Patuxent (RET-2), lower Rappahannock (LE-3), mid-Rappahannock (RET-3), and several eastern shore segments. Only in three cases were significant increasing trends noted for TP and IPF in the same segments (CB-1, 2, and 5). Segment TF-2 showed a decreasing trend in both. Similar relationships between TP and IPF may occur elsewhere, but either the data were insufficient or trends were not statistically significant. The screening approach and results from the individual annual and seasonal trends compare favorably when IPF and TP are given equal weight (Figures 3 and 6). However, as pointed out earlier, TP is believed to be a more reliable indicator of phosphorus enrichment or reduction.

The lack of TN data limits a more thorough assessment of trends in nitrogen. Figure 7 shows an increasing annual trend in TN for ET-4, TF-1, and LE-3; TN decreased in LE-2, TF-5, ET-1, and ET-7. So few segments have sufficient TN data, that little annual comparison can be made among segments. Annually, NO_3 and NO_2 showed an increasing trend in CB-1, 2, and 3, ET-2, 4, 6 and 7, TF-1, 2, RET-2, and LE-2 (Figure 8). Some of these segments showed a decreasing annual trend in TKN and ammonia (Appendix B, Section 8, Table 40) and increasing trends in NO_3. This may result from nitrification processes in use at sewage treatment plants, i.e., the upper tidal Potomac (TF-2). Seasonal trends in nitrogen are spatially complex since both increasing and decreasing trends were observed for the various forms of nitrogen in some segments in different seasons (Appendix B, Section 8, Table 40).

Figure 9 shows the combination of annual and seasonal trends in chlorophyll *a*. Annual seasonal trends were mapped, and a segment showing an increasing trend took precedence over any decreasing trends. This criterion showed that most of the main Bay segments experience increasing levels of chlorophyll *a*. There is a general correlation between increasing trends in chlorophyll *a* and segments showing an increasing trend in one or more nutrients.

It has been known for many years that there is an annual cycle of oxygen concentration in many Bay waters deeper than 9 to 11 meters (Taft et al. 1980). Low DO has been recorded from Chesapeake Bay since 1917; in September 1912, bottom water in the lower estuarine portion of the Potomac River was less than 35 percent saturated with oxygen (<2.0 ml L^{-1}) (Sale and Skinner 1917). However, during the same period of time, bottom water between Annapolis and Hampton Roads was over 60 percent saturated. Off the Patuxent, bottom water at 10 meters was 100 percent saturated in late September 1912. Newcombe et al. (1939) reported low dissolved oxygen in the Patuxent River and also in the Bay main stem at two deep stations, for 1937, and for parts of 1936 and 1939. In 1936 and 1937, dissolved oxygen reached 0 ml L^{-1} in August, although it was over 1.0 ml L^{-1} for most of the year (Figure 10). When compared to the oxygen regime for 1980, a year with similar spring freshwater inflows of 1937, it becomes apparent that presently the onset and

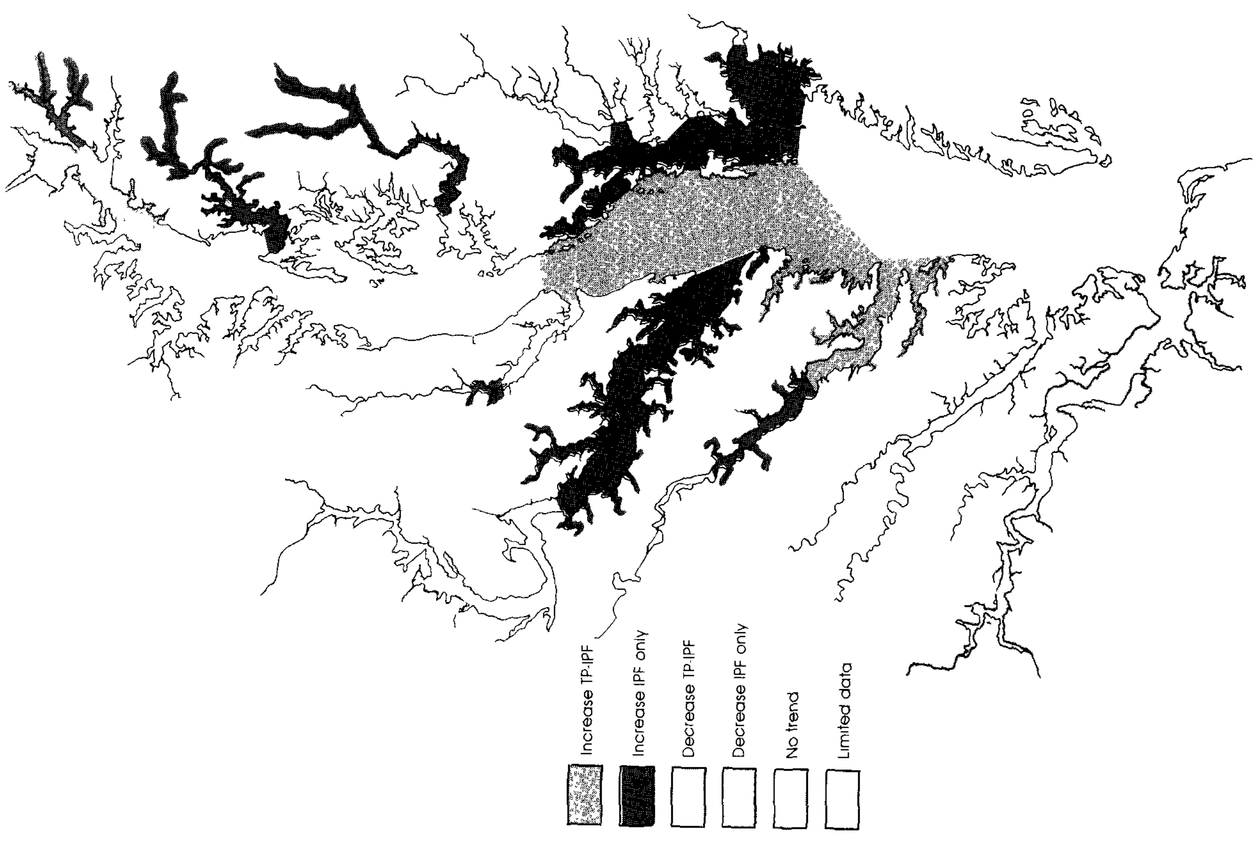


FIGURE 6a.
Annual trends in total phosphorus and its inorganic filterable fraction (IPF).

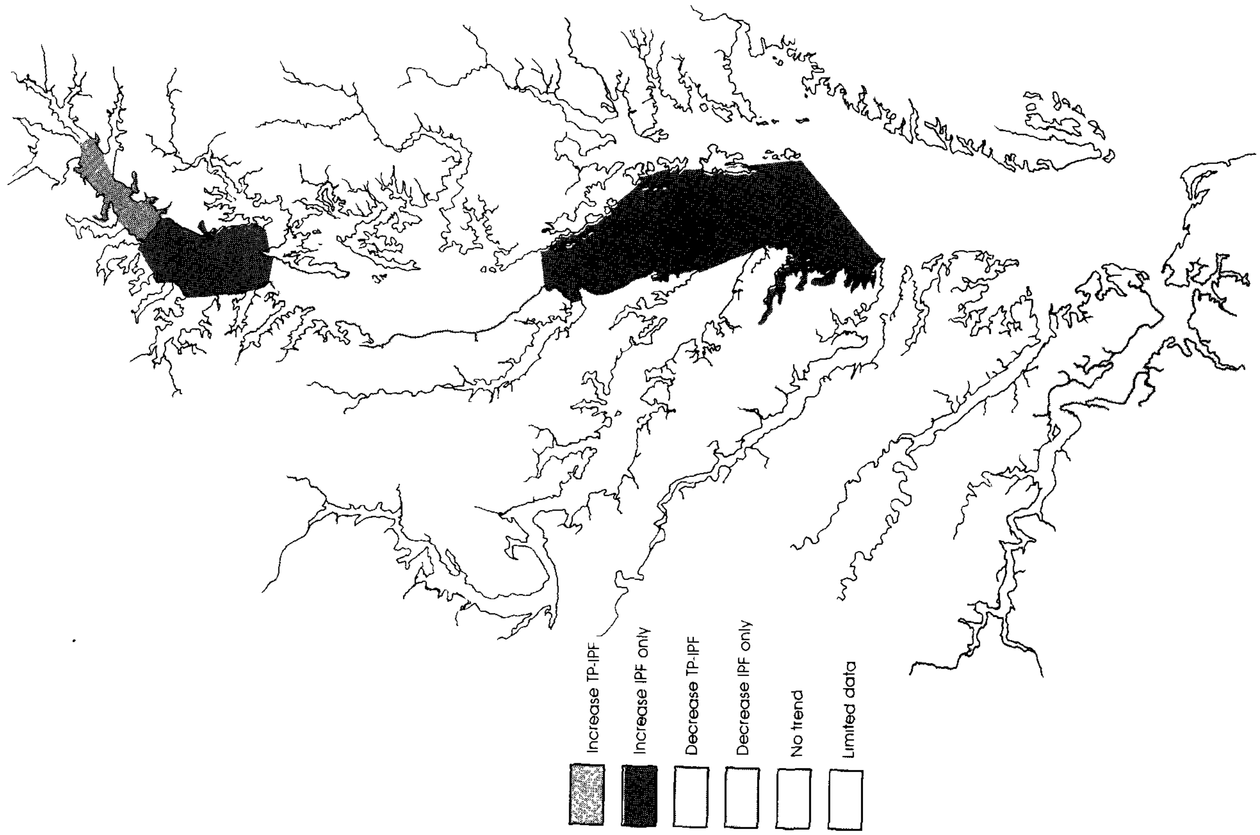


FIGURE 6b.
Seasonal trends in total phosphorus and its inorganic filterable fraction (IPF).

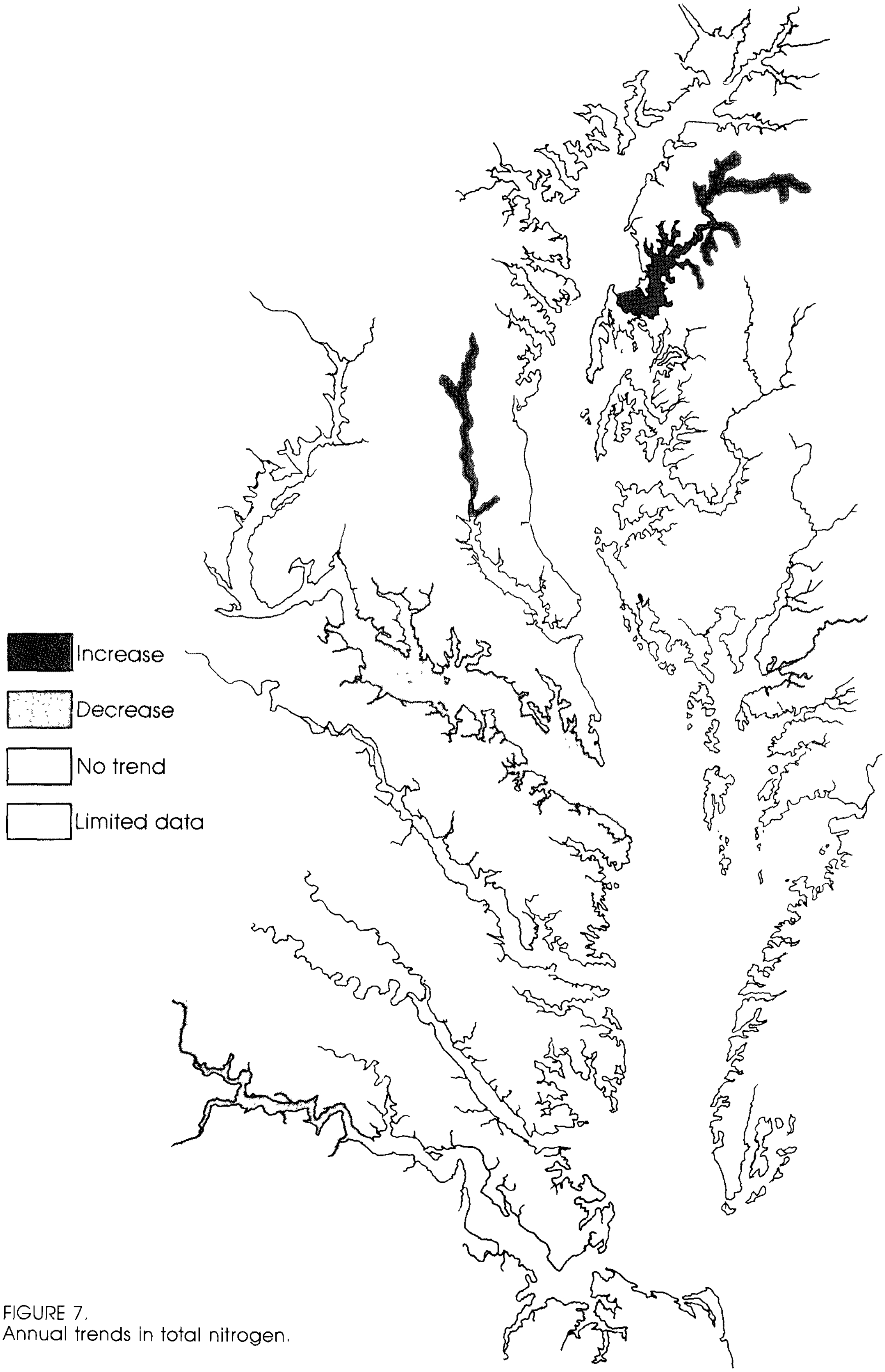


FIGURE 7.
Annual trends in total nitrogen.

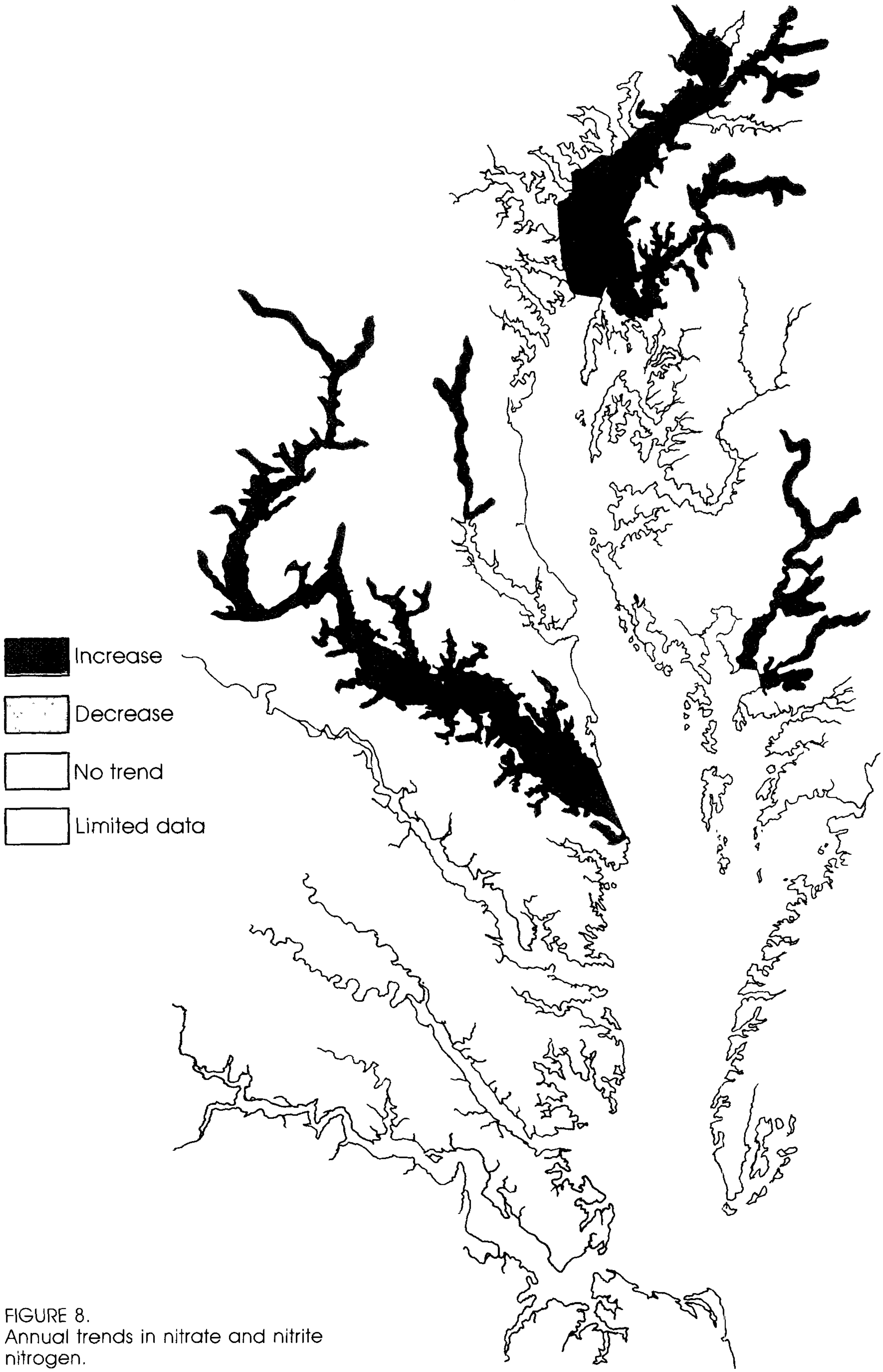


FIGURE 8.
Annual trends in nitrate and nitrite nitrogen.

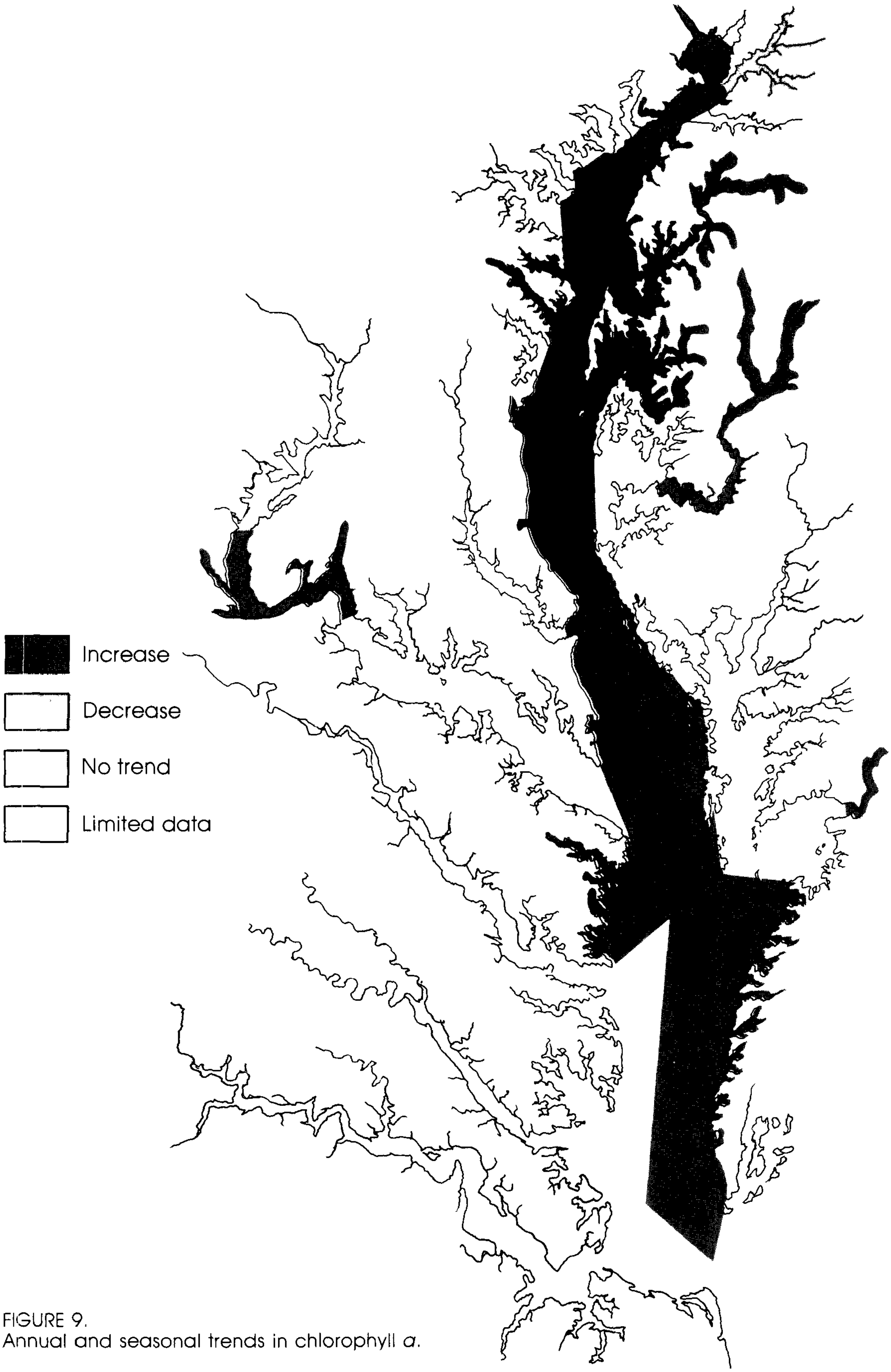


FIGURE 9.
Annual and seasonal trends in chlorophyll *a*.

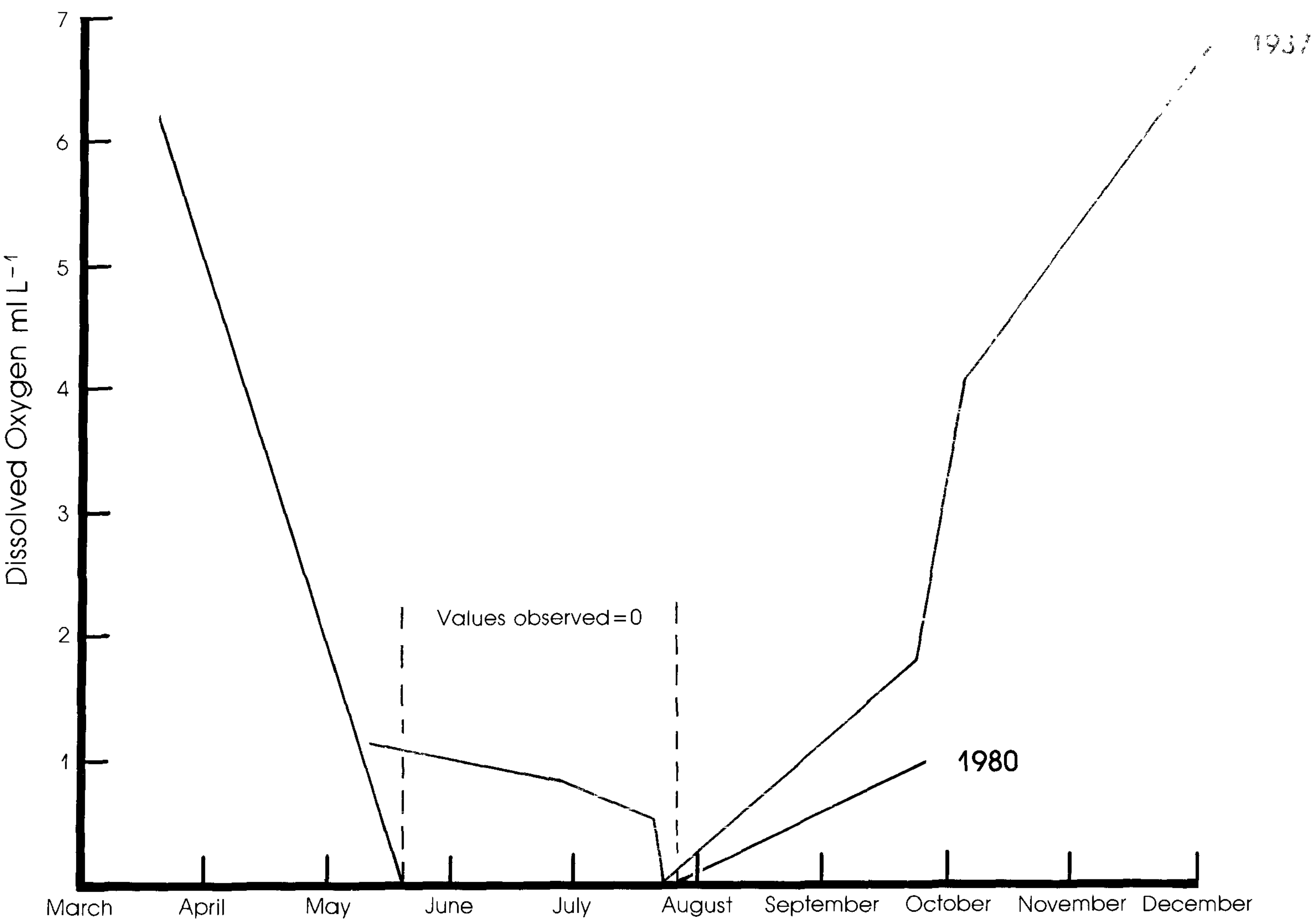


FIGURE 10. Concentrations of bottom dissolved oxygen for two years with similar spring flow. (Note: values for late May through late July 1980 equal 0.0 ml L^{-1} D.O.).

duration of anoxia are much greater. These two stations (station 1 of Newcombe, 818 P of CBI) are very close together, and it is reasonable to assume similar responses of DO at each.

During winter this lower layer is well oxygenated, but the normal reduction in water mixing rates, which occurs in spring and summer, results in oxygen being used faster in the deep water than it is replenished from the atmosphere. The result is oxygen depletion to levels that will not support typical Bay biota. There appears to be a change in this normal cyclic behavior, based on oxygen data collected by Chesapeake Bay Institute over a period of 30 years. As indicated in Figure 11, the volume of water with DO concentrations equal to or less than 0.7 mg L^{-1} (0.5 ml L^{-1}) was 15 times greater in 1980 than in 1950. On a Bay-wide basis, oxygen depletion seems to be increasing over time. In the mid-1950's, it was sporadic in intensity and extent. In the 1960's, oxygen depletion occurred from mid-June to mid-August for each year recorded with relatively complete data. During the 1970's, duration and extent of low dissolved oxygen generally continued to increase. In 1980, low DO began during the first week in May and continued into September. In addition, the presence of hydrogen sulfide, an indicator of complete anoxia, has been observed more frequently in the 1970's and 1980's than previously. This trend could result from increased organic material in the deep layer available to consume oxygen during its degradation, or from lower DO concentrations in the surface layer that influence the oxygen gradient, or from both. The DO data shown in Figure 11 were examined to see if the trend would likely result from antecedent conditions of wind and tides. The tentative conclusion is that the observed concentrations of DO and calculated volumes of water characterized by low DO levels are not related to the antecedent conditions of wind and tide. It is now inferred that the low levels of DO are related to increasing nutrient loads. A more thorough analysis of the dissolved oxygen trends is included in Appendix B, Section 5.

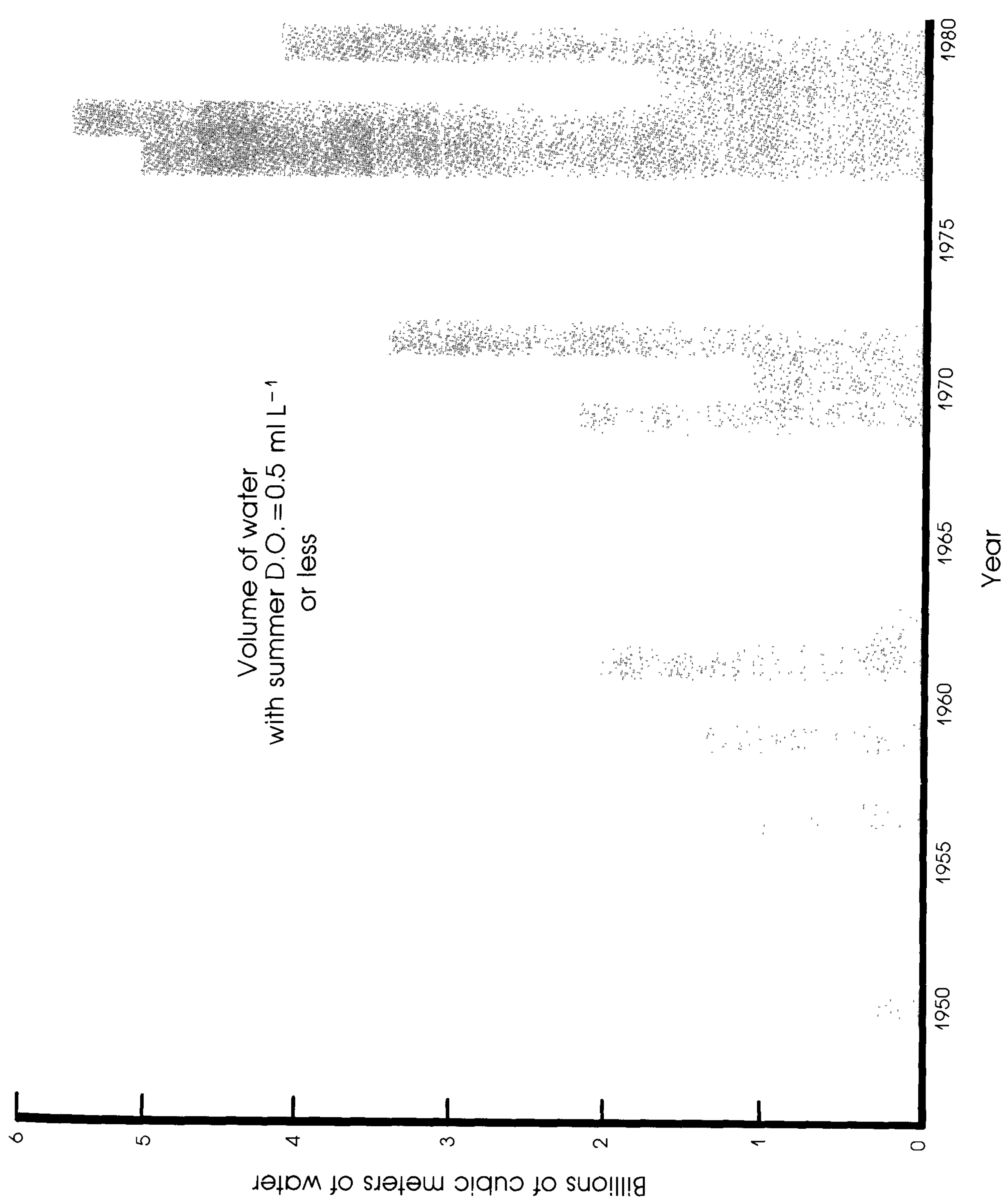


FIGURE II. Volume of water in Chesapeake Bay with low levels of dissolved oxygen, 1950 to 1980.

SECTION 3
SEDIMENT: SOURCES, TRANSPORT, AND TRAPPING

Sources and amounts of sediments to Chesapeake Bay, where they are transported, and how they are trapped, can affect the quality of the estuary. Sediments are introduced into the Bay by rivers, shore erosion, the sea and biological activity. The patterns of erosion, deposition, and sediment composition depend on circulation patterns, water depth, erosion, and local conditions. Additionally, agricultural and urban activities have increased the susceptibility of the land to erosion and contributed additional particulate matter. Physical and chemical processes of the Bay prevent much of the sediment from leaving the estuary, and many of the sub-estuaries become efficient sediment traps.

The physical and chemical characteristics of bottom sediments and the patterns of erosion and deposition have been measured and mapped by the CBP for both the Maryland and Virginia portions of the main Bay (Byrne et al. 1982, Kerhin et al. 1982). The most northern part of the Bay is composed of clayey silt and mixed sediment. The remainder of the northern Bay is dominated by sand along near-shore zones and by silty clay in the deeper axial channel. Most researchers have concluded that the prime source of this suspended sediment is the Susquehanna River, especially during storm conditions and flooding. Overall, the southern Bay contains coarser sediments. Some silty clays are contributed from the northern Bay, but the influence of the Bay mouth is significantly greater. The patterns of sand and clayey silt suggest that the Bay mouth is the major source of the larger sand and silt-sized material. Scouring of geologically older sediments and shoreline erosion are other minor sources of sandy material.

Major erosion and deposition zones have been identified throughout the main stem of the Bay. For example, there is an extensive area of Bay floor erosion from Kent Island south to the mouth of the Rappahannock River (segments CB-4, 5). Furthermore, it can be seen that throughout the Bay, at the confluence of major tributaries, the patterns of erosion and deposition are highly variable. This is due to the overall circulation patterns, mixing effects, and density differences of the waters.

Once sediment enters the Bay, its distribution is primarily influenced by the circulation pattern. Bedload material brought in by the rivers moves seaward with the freshwater discharge and is deposited where reversing tidal currents are first encountered. Thus, sediment tends to accumulate between the upstream (flood) and downstream (ebb) limit of salt intrusion (i.e., the landward-flowing lower layer). Suspended material brought into the Bay and its tributaries is carried seaward in the upper layer until it either mixes or settles downward into the lower layer that flows back up the estuary. Because of this process and the mixing that occurs between the upper and lower layers, suspended sediments tend to accumulate in an area between the tidal-fresh and lower-estuarine segments of the Bay, producing a zone of maximum turbidity. These riverine- estuarine-transition areas are characterized by large salinity ranges and high turbidities (low Secchi values) as compared to the adjoining segments (Appendix B, Section 1).

An understanding of the distribution of suspended sediment in the Bay is critical to any evaluation of the environmental quality of the Bay. High levels of suspended material in the water column inhibit the light penetration necessary for the growth of aquatic plants. Furthermore, toxic substances tend to concentrate in turbid areas for most metals, and organic compounds tend to adsorb to the sediments in the physical and chemical conditions typical of estuaries. Under varying flow conditions, this "suspended sediment front" will shift upstream or downstream. For a further discussion of the processes involved, see *Chesapeake Bay Program Technical Studies: A Synthesis* (U.S. EPA 1982b).

In light of these physical and chemical processes, it is improbable that much of the sediment introduced into the Bay escapes to the sea. Furthermore, studies have shown that several of the Bay's sub-estuaries are efficient sediment traps. Using a box model approach, Biggs (1970) determined that approximately 75 percent of the sediment introduced to northern Chesapeake Bay was trapped between the Susquehanna River and the Bay Bridge. Yarbro et al. (1981) have also used the box model approach for the Choptank and found that 89 percent of the total suspended solids remain trapped in the Choptank River. Nichols and Thompson (1973), using historical shoaling rates for the Rappahannock, concluded that over 90 percent of the suspended sediment from runoff was trapped in the upper reaches of the estuary. These findings suggest that probably most of the sub-estuaries of Chesapeake Bay trap sediment very efficiently.

To develop a Bay-wide assessment of sediment trapping efficiency, water resource engineers used a technique to predict the useful life of a reservoir before it fills with sediment. Using field data collected on various reservoirs, they have developed a curve depicting the relationship between the percent sediment trapped and the ratio of the reservoir volume to the inflow of water (Linsley and Franzini 1972). The median curve for normal reservoirs indicates a general relationship shown in Table 8 between the capacity and inflow ratio as well as the percent sediment trapped.

**TABLE 8.
RELATIONSHIP BETWEEN CAPACITY AND INFLOW RATIO, AND PERCENT SEDIMENT TRAPPED**

Capacity and Inflow Ratio	Percent Sediment Trapped
0.02	60
0.03	70
0.07	80—85
0.1	85—90
0.2	90—95
0.3	95
0.5—10	95—100

To apply this technique to Chesapeake Bay tributaries, a tidewater volume to potential inflow ratio was used for each of the tributaries (Table 9). This procedure established that the Pamunkey and Mattaponi Rivers have the lowest sediment trapping efficiency, and Eastern Bay and the Honga River have the highest sediment trapping efficiency. Based on these calculations, it is reasonable to assume that all of the sub-estuaries of Chesapeake Bay efficiently trap sediments and their associated toxic compounds.

TABLE 9.
SEDIMENT TRAPPING EFFICIENCY OF CHESAPEAKE BAY TRIBUTARIES BASED ON THEIR TIDEWATER VOLUME TO POTENTIAL INFLOW RATIO (DATA FROM SEITZ 1971 AND CRONIN 1971)

Tributary	Potential Inflow Ratio
I. Tributaries with 80 to 85 percent trapping efficiency:	
Pamunky	0.07
Mattaponi	0.07
II. Tributaries with 90 to 95 percent trapping efficiency:	
Gunpowder	0.02
Sassafras	0.02
James	0.02
Northeast River	0.03
York	0.04
III. Tributaries with 95 to 100 percent trapping efficiency:	
Elk	1 *
Bush	0.5
Potomac*	0.5*
Rhode	0.6
Rappahannock*	0.7
Patuxent*	0.8
Patapsco	0.8
Back	1.0
South	1
West	1
Wicomico	1
Piankatank	1
Pocomoke Sound	1.5
Middle	1.5
Chester*	2
Magothy	2
Severn	2
Miles	2
Choptank*	2 *
Tangier Sound	2
Mobjack	3
Lower Choptank	3
Honga	5
Eastern Bay	6

*Sediment trapping efficiency has been calculated associated with sediments.

SECTION 4
FECAL COLIFORM TREND ASSESSMENT

INTRODUCTION

Microbiological indicators such as coliform bacteria are used to estimate the potential presence of viruses and pathogenic bacteria and, thereby, the safety of water for contact recreation and shellfish harvesting. Thus, fecal coliform concentrations are considered an indication of microbiological water quality and can be used in characterizing segments of the Bay. A major use of fecal coliform data is in determining shellfish closure areas (discussed below). Some bacteria are indigenous to natural waters; others gain access from land, air, or human and animal wastes. Major nonpoint sources of coliform bacteria are decomposing organic materials and animal wastes that are deposited on land or in feedlots and barnyards. Large concentrations of waterfowl may also be an important source. Nonpoint source runoff washes large numbers of these microbes into the estuary from their natural habitats in soil and on vegetation. Marginally treated and untreated sewage and liquid wastes from industries such as dairies, canneries, frozen food processing, meat-packing, tanning, textiles, and pulp and paper plants may contribute large numbers of micro-organisms directly to receiving waters (Geldreich 1981).

Until the mid-1970's, the total coliform measurement was universally used to express the sanitary quality of water and waste effluents. Fecal coliforms (found in feces of warm-blooded animals) compose a portion of the total coliform group; they may be distinguished from other coliform bacteria by various microbiological tests (American Public Health Association 1976). Present standards in Chesapeake Bay are based upon fecal, rather than total, coliform levels because they are more likely to assess the possible presence of pathogens. It is recognized that the validity of fecal coliform levels as indicators of water quality has been questioned, particularly in estuarine and marine waters because levels are influenced by dilution, tributary flow, light, temperature, salinity, predators, and tidal variation. However, we have not yet found a more appropriate indicator. Fecal coliforms are included in this assessment as an indicator of environmental trends, and the results are not meant to imply public health risk.

DATA BASES

The data base used for characterizing the Maryland portion of the Bay was obtained from the Maryland State Department of Environmental Health and Mental Hygiene and was reported in median values of fecal coliforms for each station. The Division of Technical Analysis in Maryland analyzes fecal coliform data collected at 1402 sampling stations within the Chesapeake Bay area. Samples are collected once a month in most areas and twice in selected areas. Fecal coliforms are measured as a Most Probable Number (MPN) per 100 milliliters. The fecal coliform data analyzed cover the 1975 to 1981 time period.

Data used for characterization of the Virginia portion of the Bay were provided by the Virginia State Department of Health, Bureau of Shellfish Sanitation. The data are reported as median values of fecal coliforms for each of 3 to 50 sampling stations in 98 shellfish growing areas. For this analysis, data from 1975 to 1981 were used. It is important to recognize when reviewing these data that there is a significant difference between the

Virginia and Maryland data bases in the representation of the areas restricted for shellfish harvesting. The difference is that the State of Maryland samples shellfish areas with the same frequency throughout the year whereas the State of Virginia will decrease the number of samples taken within an area that is closed to shellfish harvesting. For the statistical analysis, median values were weighted in an effort to compensate for this difference.

ANALYSIS OF COLIFORM LEVELS

Mean fecal coliform levels from selected tributaries of Chesapeake Bay were converted to natural logs and were compared using analysis of variance procedures followed by a Student-Newman-Keuls test. Differences were considered significant at the $p \leq 0.05$ level. The natural logs of the MPN means rather than the MPN means were used because the distribution of MPN values is logarithmically normal.

The first set of comparisons was made between the James, Potomac, York, and Rappahannock Rivers. The means were significantly different between groups and not significantly different within groups. The following significantly different groups of means were distinguished (listed in descending order): Group A—James River, Group B—York River, and Group C—Potomac and Rappahannock Rivers (Table 10). It should be noted that data from several Maryland tributaries were not included when the Potomac River mean was calculated. The higher fecal coliform levels in the James River are attributable to concentrations in the upstream portion of rivers and creeks and reflect land-use patterns in these areas.

In Maryland, three western shore tributaries were compared to three eastern shore tributaries to determine if water quality in the more urbanized western shore rivers was significantly different from water quality in the eastern shore rivers. Their means fell into three significantly different groups (Table 11) with the Tred Avon River (Eastern Shore) having the highest values, and Harris Creek (Eastern Shore) and the Severn River (western shore) having the lowest values.

A third set of comparisons was made between seven Maryland eastern shore tributaries. The results showed five significantly different groups that are listed in Table 12. The Great Wicomico River had the highest values and was significantly different from the other six rivers, while Fishing Bay had the lowest values.

DETERMINATION OF SHELLFISH CLOSURE AREAS

Shellfish are defined as all edible molluscan species of oysters, clams, and mussels. The oyster, *Crassostrea virginica*, is considered to be the most economically important of these species in the Chesapeake Bay area. The oyster's filter-feeding process enables it to obtain nutrients from microorganisms, small detrital particles, and phytoplankton. However, this type of feeding may also concentrate pollutants from the water column in

TABLE 10.
GROUPS OF MAJOR WESTERN SHORE TRIBUTARIES BASED ON STUDENT- NEUMAN-KEULS TEST OF Ln FECAL COLIFORM LEVELS (LnMPN). MEANS ARE SIGNIFICANTLY DIFFERENT BETWEEN GROUPS BUT NOT SIGNIFICANTLY DIFFERENT WITHIN GROUPS, AT $p \leq 0.05$

Basin	Ln(MPN)	n	Group
James	2.8374	837	A
York	2.1723	3084	B
Potomac	1.9689	2347	C
Rappahannock	1.9191	319	C

TABLE 11.
GROUPS OF SMALLER TRIBUTARIES FROM EASTERN AND WESTERN SHORE BASED ON STUDENT-NEUMAN-KEULS TEST OF Ln FECAL COLIFORM LEVELS (LnMPN). MEANS ARE SIGNIFICANTLY DIFFERENT BETWEEN GROUPS BUT NOT SIGNIFICANTLY DIFFERENT WITHIN GROUPS, AT $p \leq 0.05$

River	Ln(MPN)	n	Group
Tred Avon	2.2229	236	A
West and Rhode	1.9439	217	B
Broad Creek	1.7734	147	B & C
Harris Creek	1.6357	141	C
Severn	1.6092	235	C

TABLE 12.
GROUPS OF EASTERN SHORE TRIBUTARIES BASED ON STUDENT-NEUMAN-KEULS TEST OF Ln FECAL COLIFORM LEVELS (LnMPN). MEANS ARE SIGNIFICANTLY DIFFERENT BETWEEN GROUPS BUT NOT SIGNIFICANTLY DIFFERENT WITHIN GROUPS, AT $p \leq 0.05$

River	Ln(MPN)	n	Group
Grt. Wicomico	2.8527	105	A
Pocomoke Sound	2.0948	112	B
Choptank	1.9783	398	B
Nanticoke	1.8275	153	B & C
Miles	1.6710	159	C & D
Chester	1.4742	305	D & E
Fishing Bay	1.3071	121	E

its tissues. Intensive monitoring of shellfish growing areas is necessary to ensure safety in the consumption of these products. Fecal coliform bacterial levels are monitored by Maryland and Virginia in all waters capable of propagating shellfish (including public and leased beds). If the bacteriological standard is violated, the area will generally be closed to shellfish harvesting. Harvesting can be resumed when the Virginia Department of Health or the Maryland Department of Health and Mental Hygiene determine that the water quality has met the established standards.

Other factors besides the bacteriological standards are considered in the closure of shellfish harvesting waters. Areas in the Choptank River have been closed to harvesting for the protection and propagation of prime seed area. The closures are instituted through the Maryland Department of Natural Resources. Waters have been closed based upon the possible threat of pollution from point sources such as sewage treatment plants and industries, upon the results of sanitary surveys on dwelling units, and upon the consideration that certain locations were sources of boat pollution and animal waste pollution. Buffer zones are closure areas established around actual and potential sources of pollution. The source determines the size of the buffer zone. The most common source associated with buffer zones is wastewater treatment plants. The effectiveness and reliability of wastewater treatment, distances of pollutants from shellfish areas, the effects of winds, runoff, stream flow, and tidal currents are important items to be considered when establishing buffer zones. Figure 12 shows shellfish closure areas (1981 for Virginia and 1982 for Maryland) and the location of Publicly Owned Treatment Works.

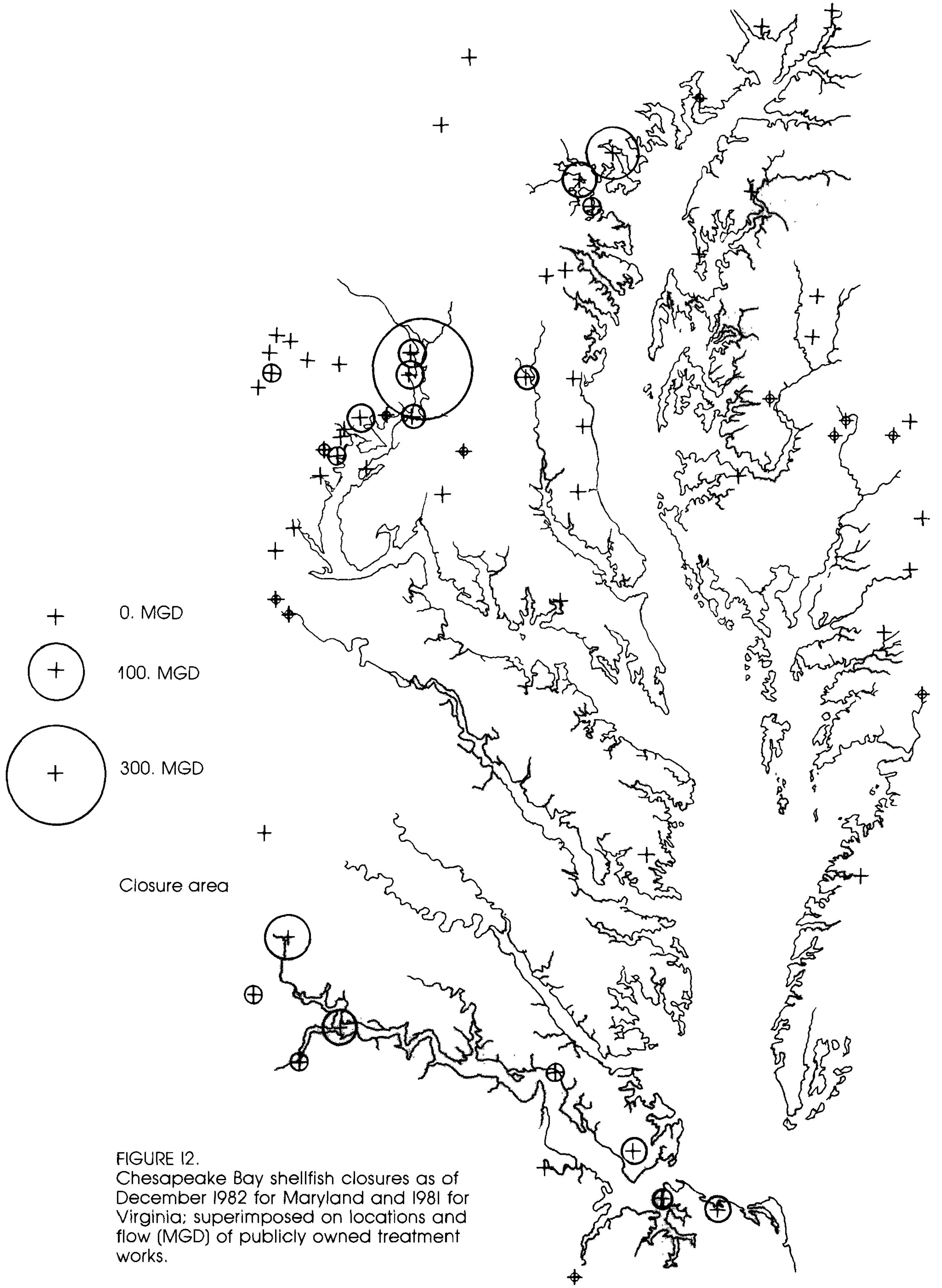


FIGURE 12.
Chesapeake Bay shellfish closures as of December 1982 for Maryland and 1981 for Virginia; superimposed on locations and flow (MGD) of publicly owned treatment works.

SECTION 5

ORGANIC COMPOUNDS IN THE WATER AND SEDIMENTS

INTRODUCTION

Organic compounds are a broad class of chemicals that include synthetic organic compounds as well as naturally occurring compounds. The groups of organic compounds most frequently studied include pesticides, herbicides, polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), phenols, substituted benzenes, and halogenated aliphatics. Many have been or are presently being discharged into the waters of Chesapeake Bay and its tributaries. Once introduced to Bay waters, organic compounds can adsorb to sediment, dissolve in the water column, and concentrate in the biota; or they can be removed from and/or be destroyed in the estuarine environment. Removal and destruction processes can include photolysis, hydrolysis, volatilization, and other chemical reactions.

Some of the most frequently investigated organic compounds are those contained in the U.S. Environmental Protection Agency Priority Pollutant List (Appendix B). The dominant fates of the priority pollutants are adsorption to sediment, biological uptake, and chemical and physical destruction. In general, the structural activity of each compound usually determines its fate. Compounds that are hydrophilic (attracted to water) will exist primarily in the water column and can eventually leave the Bay through its exchange with the ocean. However, hydrophobic (repelled by water) compounds will adsorb rapidly to sediment particles and, through sedimentation processes, be trapped in the Bay environment. The chemical properties of these compounds and the physical and chemical conditions, such as salinity and pH, determine the toxicity of the compounds to biota and their bioavailability.

DATA ANALYSIS

The analytical capabilities for analysis of organic compounds in an estuarine system have been developed only recently. For many compounds, the methodologies and detection capabilities are still not sensitive enough to routinely measure environmental concentrations at which toxicity to biota may occur. In addition to the low sensitivity, most analyses are restricted to searching for a limited number of compounds; thus, many organic compounds go undetected. These problems have made it difficult to fully "characterize" the organic compounds in the Bay and its tributaries. Nonetheless, in segments where sufficient data exist, an effort was made to evaluate bottom sediment, water column, point source discharge, and oyster tissue data collected by CBP researchers and the States of Virginia and Maryland. The data base that was created was re-examined for quality and accuracy as described in Section 6.

To develop a graphic assessment of the current condition of the main Bay, we relied on CBP research by Bieri et al. (1982b). Their analysis by gas chromatography of sediment samples collected in the spring and fall of 1979 revealed over 300 organic compounds in sufficient abundance to be recognizable. The concentration sums of all identifiable compounds at given station locations are shown in Figure 13. This figure shows that organic compounds are most abundant in the upper Bay sediments, decreasing seaward, and increasing

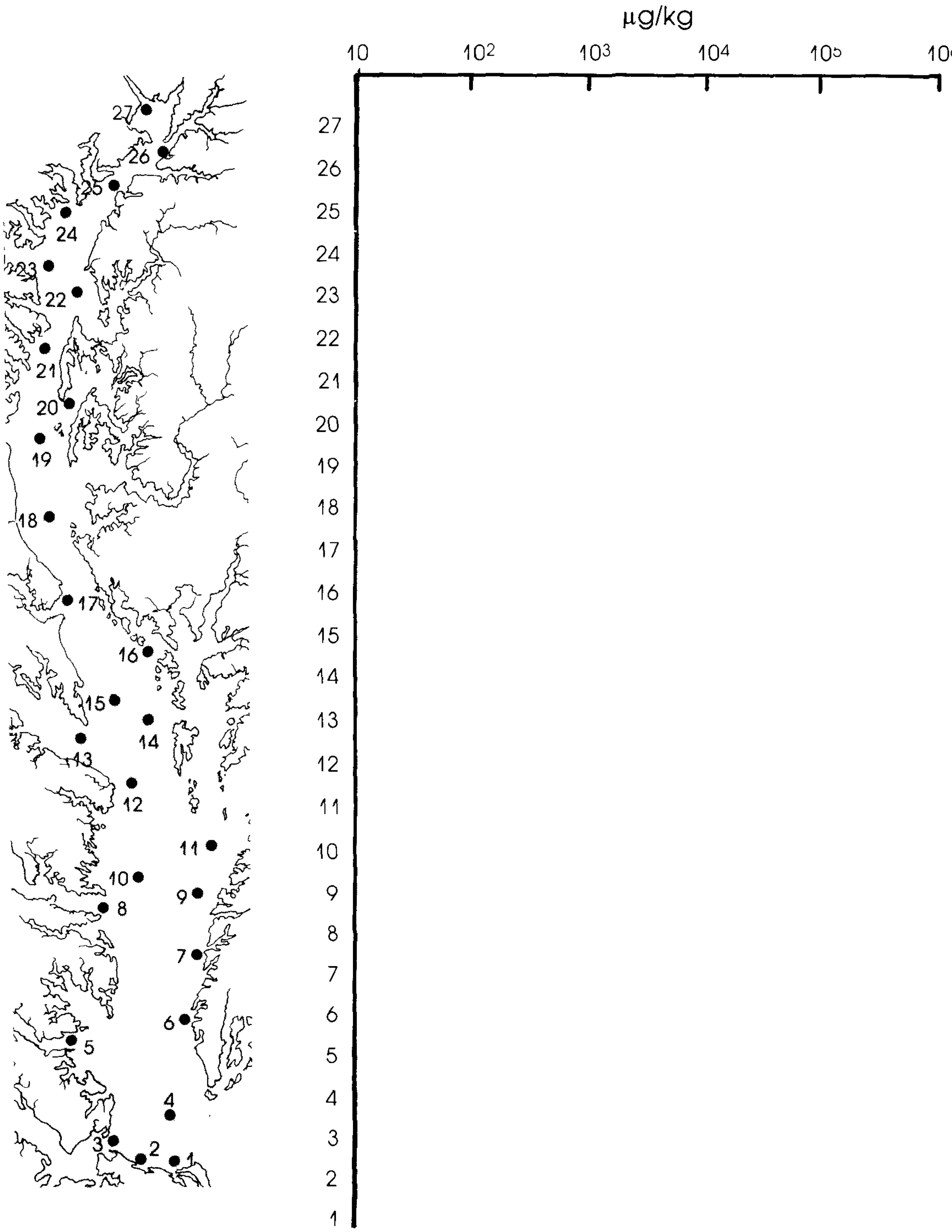


FIGURE 13. Station locations and bar graphs representing concentration sums (ppb) of all recognizable peaks for organic compounds after normalizing for silt and clay content. (From Bieri et al. 1982c).

again near mouths of lower Bay tributaries (Note: concentrations are on a log scale). Additional sediment data retrieved from the CBP-compiled data base revealed levels above EPA criteria in five western tributaries and Pocomoke Sound.

Data on levels of toxic organic compounds in the waters of the Bay and its tributaries are largely limited to pesticides, PCBs, and to very spotty data on volatile halogenated hydrocarbons and phenols. There are relatively few observations compared to the number for heavy metals in the water column (Appendix B). To provide a framework for assessing the water column data, observed concentrations were compared to established EPA water quality criteria levels for priority pollutants. Aldrin and dieldrin criteria levels were exceeded in Pocomoke Sound and portions of the Potomac, Rappahannock, York, and James Rivers. Five observations exceeded chronic levels of dieldrin in the tidal-fresh portion of the James River (TF-5), and two observations of endrin in Pocomoke Sound (EE-3) exceeded acute values. Chapter 3 discusses implications of these concentration levels for the living resources of Chesapeake Bay.

Measured concentrations of total residual chlorine were elevated according to criteria from a 1982 draft EPA document.[2] Most of the 358 observations were in tidal freshwater, and 67 percent exceeded the draft criteria. On the basis of this data alone, it is difficult to assess the impact of chlorination on Bay waters. It is likely that analytical methods used in the past have not been sensitive enough to accurately measure low ambient concentrations.

BAY-WIDE SUMMARY

Organic Compounds in the Sediments

The distribution of organic compounds in the bottom sediments of the main Bay (Figure 13) and an analysis of the limited tributary data generally suggest that organic compounds concentrate at river mouths and in maximum turbidity areas. This is apparently due to the tendency of organic compounds to adsorb to suspended material accumulating in these areas. It is interesting to note that the higher concentrations also appear to be associated with sediments of increasing fineness. The highest concentration of organics in the bed sediments were found in the tidal-fresh portions of the Patapsco and Elizabeth Rivers. Overall, 480 compounds were identified in surface samples from the Patapsco estuary, and 310 compounds were identified in samples taken in the Elizabeth estuary (Bieri et al. 1982c). In both areas the sum of all organic compounds detected exceeded 100 ppm at several locations. Concentrations of PAHs ranged from 1 to 90 ppm in Patapsco River sediments and from 1 to over 100 ppm in Elizabeth River sediments. In the Patapsco River, levels of PCBs were as high as 8,000 ppb. It is apparent that these unusually high levels are due to industrial activities in these harbors. This is further substantiated by the fact that core analyses show that variation in both phthalate esters and polynuclear aromatics correlate well with historical rates of coal production and use in the United States (Peterson 1982).

Organic Compounds in the Waters

Organic compounds in the waters of the Bay and its tributaries appear most often associated with anthropogenic point and nonpoint sources located in industrialized areas (Bieri et al. 1982c). Toxicants associated with suspended material do not appear to migrate from these highly contaminated areas such as the Patapsco River (Palmer 1974). This would be expected in light of our understanding of the sediment trapping efficiency of the sub-tributaries of the Bay.

On a Bay-wide level, the primary organic groups of concern are the pesticides and herbicides because they are extensively used on agricultural lands that comprise a large portion of the watershed. Chesapeake Bay Program research on two major herbicides, atrazine and linuron, indicates that ambient levels did not exceed 5.5 ppb in the main stem of the Bay or 3.5 ppb in the western tributaries between 1976 and 1980 (Kemp et al. 1982a). These levels are below the concentrations that would begin to significantly inhibit the growth of submerged aquatic plants (U.S. EPA 1982b). However, lethal levels of atrazine as high as 140 ppb have been found in the Rhode River following post-spraying rainfall events (Wu 1980). Thus, in smaller tributaries near sources of herbicides, biological effects may occur. Nonetheless, pesticide levels throughout the Bay are generally in the range of 1 ppb (generally considered safe), except at riverine headwaters following application and rainfall events. In such situations, the observed levels of pesticides exceed EPA's water quality criteria. Because of the way data were collected, duration or extent of these high values is not well known.

KEPONE IN THE JAMES RIVER—AN EXAMPLE

The organic pesticide Kepone was discharged into the James River at Hopewell, Virginia, until 1975. This synthetic organic substance can be easily bioconcentrated by organisms, from phytoplankton to fish. In fact, Nichols and Cutshall (1981) found that the highest concentrations of Kepone occurred in zooplankton, averaging 2.9 ppb, whereas the total (bulk) suspended material averaged 0.09 ppb. Laboratory studies by Bahner et al. (1977) have further established that shellfish and finfish are capable of bioconcentrating Kepone to concentrations near or above the Food and Drug Administration Action Level of 0.30 ppm, when exposed to water with total Kepone residues as low as 0.022 ppb. Kepone was detected at or above 0.022 ppb during 1976 to 1978 within all reaches of the James estuary. Because of the high levels of Kepone, much of the James River was closed to fishing in 1975. Thus, the Kepone incident became one of the major pollution events in the Bay.

Kepone has been extensively studied as a result of public concern. Through these efforts much has been learned about the processes that affect the movement of synthetic organics through the system. Concentrations of dissolved Kepone have ranged from as high as 45 ppb near the source (Saleh and Lee 1978) to 1.0 to 9.8 ppb 75 km^3 seaward from the source (Slone and Bender 1980). The distribution generally indicates downstream dilution and mixing away from the source. However, the bulk of Kepone in the water is associated with particulate material, and Kepone is difficult to remove from sediment particles over the range of pH and salinity values occurring in the James River (Huggett et al. 1980). It is, therefore, valuable to examine the concentrations in the suspended sediment.

The Kepone content of suspended material varies with distance from the source depending on hydrologic conditions. At relatively low river inflow, August 1977, concentrations ranged from 258 ppb near the source to less than 10 ppb at 118 km downstream. Additionally, the concentrations are higher in surface water than near the bottom. The relatively high surface values result from locally high concentrations of organic matter as indicated by relatively high concentrations of ATP and a high percent organic carbon of the suspended material.

At relatively high river inflow, April 29 to 30, 1978, when the inner limit of saline water was restricted to the lower estuary and there was substantial sediment influx, the Kepone content of suspended material varied widely with time and location. However, highest concentrations generally were found near the source and near the turbidity maximum. In fact, on April 25, 1978, concentrations reached 420 ppb in the turbidity maximum area, 80 to 100 km downstream from the source. This zone contained high loads of organic material, including up to 35 percent of the total suspended material. Again, these high Kepone concentrations appear to be related to bioaccumulation of plankton. This has been confirmed by direct examination of filtered suspended material as well as by corresponding analysis of zooplankton populations (Jordan et al. 1979).

Temporal variations in the concentrations of Kepone also appear to be related to the organic matter in the water column. Over a tidal cycle, there is generally a tendency for concentrations to peak near slack water when organic matter makes up relatively high percentages of the suspended load (Nichols and Trotman 1979). Seasonal trends are marked by peak Kepone content in summer, July to September, 1977 and 1978 (Table 13). During the summer, high Kepone content in the water column is most likely caused by bioaccumulation of plankton that proliferate more in summer than in winter, making up a significant portion of the suspended load (Lunsford 1981). This trend suggests that organisms are more exposed during the summer than at other times.

An examination of vertical sediment core profiles of Kepone concentrations with depth shows

TABLE 13.
SEASONAL KEPONE WATER-COLUMN CONCENTRATIONS IN JAMES RIVER ESTUARY, 1976 TO 1978 (LUNSFORD ET AL. 1980)

	1976			
Season	Sample Size	Median	Mean	Range
Winter	131	0.03	0.05	ND[1] – 0.95
Spring	76	0.03	0.04	ND – 1.20
Summer	70	0.03	0.05	ND[2] – 0.97
Fall	64	0.00	0.01	ND – 0.16

	1977			
Season	Sample Size	Median	Mean	Range
Winter	9	0.00	0.01	ND – 0.04
Spring	84	0.00	0.00	ND – 0.05
Summer	64	0.04	0.04	ND – 0.19
Fall	60	0.00	0.01	ND – 0.07

	1978			
Season	Sample Size	Median	Mean	Range
Winter	54	0.00	0.00	ND[3] – 0.14
Spring	33	0.03	0.04	ND – 0.15
Summer	52	0.04	0.05	ND – 0.29
Fall	51	0.02	0.03	ND – 0.30

[1]Non-detectable; 0.01ppb
[2]Non-detectable; 0.02 ppb
[3]Non-detectable; 0.02 ppb or 0.05 ppb.

the trends over the life span of Kepone contamination, 1966 to 1976 (Figure 14). Contamination of sediments began shortly after production started in 1966 and gradually increased to 1973. When production increased markedly in 1974, contamination also increased. Diminished Kepone concentrations after 1976 relate to the halt of production. It is evident that general annual trends recorded in the sediments vary with source input and that the 1975 high levels of Kepone in the sediments are gradually being removed from the system by flushing or burial processes.

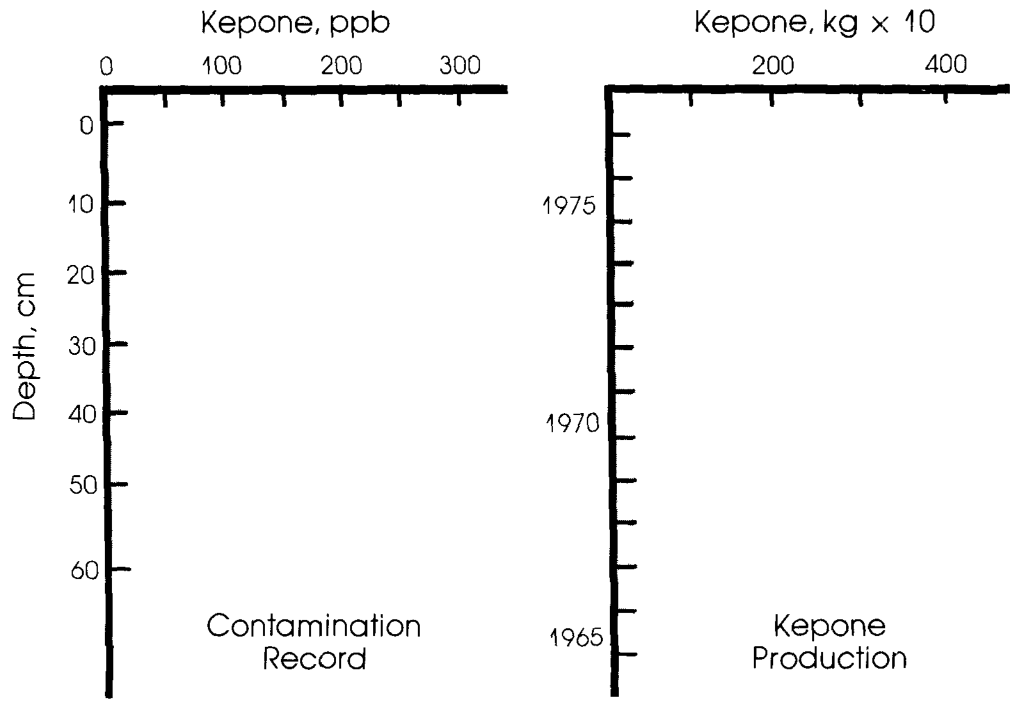


FIGURE 14. Annual trends of Kepone concentations in bed sediments compared to the Kepone production record (Nichols and Cutshall 1981).

SECTION 6
METALS IN THE WATER AND SEDIMENTS

INTRODUCTION

Metals are naturally-occurring chemical elements, but can be anthropogenically enriched. Several of the metals, such as copper (Cu) and zinc (Zn), are essential trace elements for most living organisms. However, if organisms are exposed to high concentrations of metals, they may suffer chronic or lethal effects. High concentrations of metals are generally found in industrialized areas, coming largely from industrial plant discharges, sewage treatment plants, runoff from urbanized areas, and atmospheric pollution. Once in the Bay environment, physical, chemical, and biological processes affect the ultimate fate of the metals. (Maps showing station locations for metals in the water and sediments are in Appendix B, Section 2.) A more complete discussion of these processes can be found in *Chesapeake Bay Program Technical Studies: A Synthesis* (U.S. EPA 1982b).

The metals found in Chesapeake Bay occur in many forms. Some are chemically stable with limited biological availability; other forms are very unstable and available to biota. Metals can be bound to organic or inorganic components, in either dissolved or particulate phases. Some data on metals in the water column are expressed as total metals, and other data include the specific dissolved or particulate components of the various metals. From some of the investigations of water column concentrations, particularly Nichols et al. (1981), it is evident that the range of metal concentrations at any given station can vary depending on various physical and chemical factors. For some metals, the variability of particulate water-column concentrations within one tidal cycle at a particular station can be as great as the variability from Bay-wide sampling (Nichols, et al. 1982). For this reason and because more data are available in sediments, the bottom-sediment data base was used as the primary source to characterize the Bay and tributary segments for levels of metal contamination.

DATA BASES AND ANALYSIS

Data from the water column and bottom sediments were assembled to provide as comprehensive a data base as possible. Temporal and spatial trends were identified as well as the relationships and the interactions among toxic chemical parameters. Chesapeake Bay Program project data were the primary source for the main Bay analysis. Dissolved and particulate metal data from Kingston et al. (1982) and Nichols et al. (1982) provided main Bay coverage for 1979 to 1980. The data cover 14 metals in solution and 11 to 14 metals in particulate form from surface and near bottom waters. Metals data from sediment cores and surface samples (Helz 1981) were used to characterize the main Bay sediments. Additional data covering a larger temporal scale (approximately 1960 to present) and, also an expanded geographical coverage into the tributary segments, were obtained from STORET and from Maryland and Virginia water quality data bases. Additional data on bottom sediments in the tributaries were collected and summarized by the staff of the Maryland Department of Health and Mental Hygiene. The Virginia State Water Control Board staff also assisted in gathering and entering additional data. All laboratory analyses followed EPA standards and included participation in the EPA water quality assurance program. The data were assumed to be log-normally distributed and were subjected to univariate statistics. Tables summarizing the data are included in Appendix B.

The data were analyzed to determine if the metal concentrations were above natural background levels. In the bottom sediments, the six most frequently sampled metals [cadmium (Cd), Cu, chromium (Cr), nickel (Ni), lead (Pb), Zn] were analyzed by comparing pre-colonial, natural pristine levels as determined from cores, with current levels. A full discussion of the methodology used to determine this "Metal Contamination Index" (C_I) and the values calculated for the different CBP segments are included in Appendix B. For the water column analysis, the concentrations of the various metals were normalized (proportioned) to a reference metal [usually iron (Fe), aluminum (Al), or scandium (Sc)] that occurs ubiquitously in the Bay, or compared with their abundance in either shale or average crustal material. A more complete discussion of this methodology is included in *Chesapeake Bay Program Technical Studies: A Synthesis* (U.S. EPA 1982b), and Kingston et al. (1982). These assessments of metal enrichment above natural background levels give us some indication of the impact of human activities. High levels of metal enrichment warrant some concern; however, Bay organisms will only be affected if the actual metal concentrations exceed the species' levels of tolerance (Chapter 3 and Appendix D). To determine if metal concentrations in Bay waters ever reached levels that could threaten Bay organisms, data on water column metals were screened against the U.S. EPA's Water Quality Criteria. These criteria and a discussion of the analysis are included in Appendix B.

To assess the transport and fate of metals in the Bay, water column data were also evaluated along salinity gradients. The Bay is a site where river water and seawater mix. If metals transported in solution by rivers undergo no biological or chemical reactions, then the trends of concentration are due solely to physical mixing processes and dilution. This relationship, known as conservative mixing, is a linear function of salinity whose slope will be negative for a metal more concentrated in river water than in seawater. In theoretical conservative mixing, data lie close to a straight line joining river and ocean end members; concentrations would be highest (or lowest) in the river water, lowest (or highest) in the ocean water. Uranium (U) and molybdenum (Mo) behave conservatively. On the other hand, if the metal concentrations are affected by processes other than dilution, then they behave nonconservatively; they deviate from a straight line and lie above, indicating addition to the Bay waters, or below, indicating biological or chemical removal from Bay waters. All of the remaining trace metals behave nonconservatively in the Bay.

When the average dissolved metal concentration of the Susquehanna River, or alternately, the average from world rivers, is compared with the average metal content of the Bay (Kingston et al. 1982) and the average for seawater, it is evident that the metal content of river water exceeds that of Bay water and seawater for all metals except Ni (Table 14). This trend indicates substantial mixing and dilution of river-borne metals in the Bay, or substantial removal of metals in the Bay prior to their discharge to the ocean. For example, data indicate that some metals such as Cd, Cu, Pb are taken up by phytoplankton and zooplankton and, thus, become incorporated into organic matter. Depositing this organic material is one path in which metals from the water column are deposited in the sediments.

SUMMARY OF METALS IN BED SEDIMENTS

The CBP approach to assessing metal trends in the bed sediment was to use cores that documented changes over time. Sediment cores, analyzed for trace metals and with an established geochronology, can estimate trace metal inputs, assuming no diagenetic migration of metals through the length of the core. The cores were carefully selected, because burrowing activities of benthic organisms in aerobic environments could disturb the sedimentary record, create an "artificial" ^{210}Pb distribution and, thus, influence trace metal patterns. Three cores with ^{210}Pb profiles that showed a clear chronology and no evidence of bioturbation were used for the analysis. The change in sediment composition and the associated metal concentrations over time was taken into account by using a ratio of silicon (in-

TABLE 14.
COMPARISON OF MEAN DISSOLVED METAL CONTENT IN SUSQUEHANNA RIVER WATER, BAY WATER, AND OCEAN WATER

Metal	River water[1] ug L^{-1}	Bay Water[2] ug L^{-1}	Ocean Water[3] ug L^{-1}
Cadmium (Cd)	0.8	0.02	0.1
Cobalt (Co)	1.0	0.1	0.1
Copper (Cu)	1.80 (7)[3]	0.3	2.0
Chromium (Cr)	10.0	0.2	0.2
Lead (Pb)	1.0	0.1	0.03
Nickel (Ni)	ND (0.3)[3]	1.2	2.0
Zinc (Zn)	26.0	0.9	2.0

[1]Based on data of Lang and Grason (1980).
[2]From data of Kingston et al. (1982) for summertime only.
[3]From data for world rivers and natural concentrations of seawater. Turekian (1971)
ND is non-detectable.

dicative of sandy material) to aluminum (indicative of clay material). A model was developed to separate the sediments into classes based on their metal content and their silicon to aluminum (Si/Al) ratios (Appendix B, Section 6). By using a statistical relation between the Si/Al ratio and log-metal content of old pre-pollution sediments in the Bay, a natural Chesapeake background concentration for the metal can be estimated. Then, the predicted natural Chesapeake values can be compared to the actual observed values and to average levels world-wide. Details of this analysis are in Appendix B, Section 6.

Metals analyzed by Helz (1981) include Cr, Zn, Ni, Cu, Fe, Pb, Mn, Co, Al, and Cd. The observed values for most of the metals are generally higher than estimated natural Chesapeake values, with the exception of Cr. Figure 15 illustrates the changes in observed metal concentrations for Zn and Cr over time compared to natural background levels. Zinc shows significant increases in the late 1800's coincident with peak land clearance from timbering and agriculture, as well as from coal mining in the Susquehanna drainage basin. Chromium shows no historic enrichment in the cores. Using the model, we can compute a contamination factor (C_f) for each sample and each metal with an observed Si/Al ratio, compared to pre-pollution samples from deep in Bay sediments.

$$\text{Contamination Factor } (C_f) = C_o - C_p$$

where C_o = surface metal concentration
and C_p = predicted metal concentration

Thus, if the C_f equals 1.0, the sample contains concentrations of that particular metal that exceed natural Chesapeake Bay sediment by 100 percent. If the C_f equals 0, then the sample contains concentrations of that particular metal equal to natural concentrations. If the C_f is negative, then the sample contains concentrations of the metal below expected natural concentrations. Taking the sum of the individual contamination factors for available metals, we can compute a Contamination Index (C_I) for each sediment sample:

$$\text{Contamination Index } (C_I) = \sum_{n=1}^{n=6} C_f$$

Figure 16 shows the level of metal contamination in the Bay based on Cf's for Cd, Cr, Cu, Pb, Ni, and Zn.

The Contamination Index is a useful indicator of potential problem areas in the Bay. It is ap-

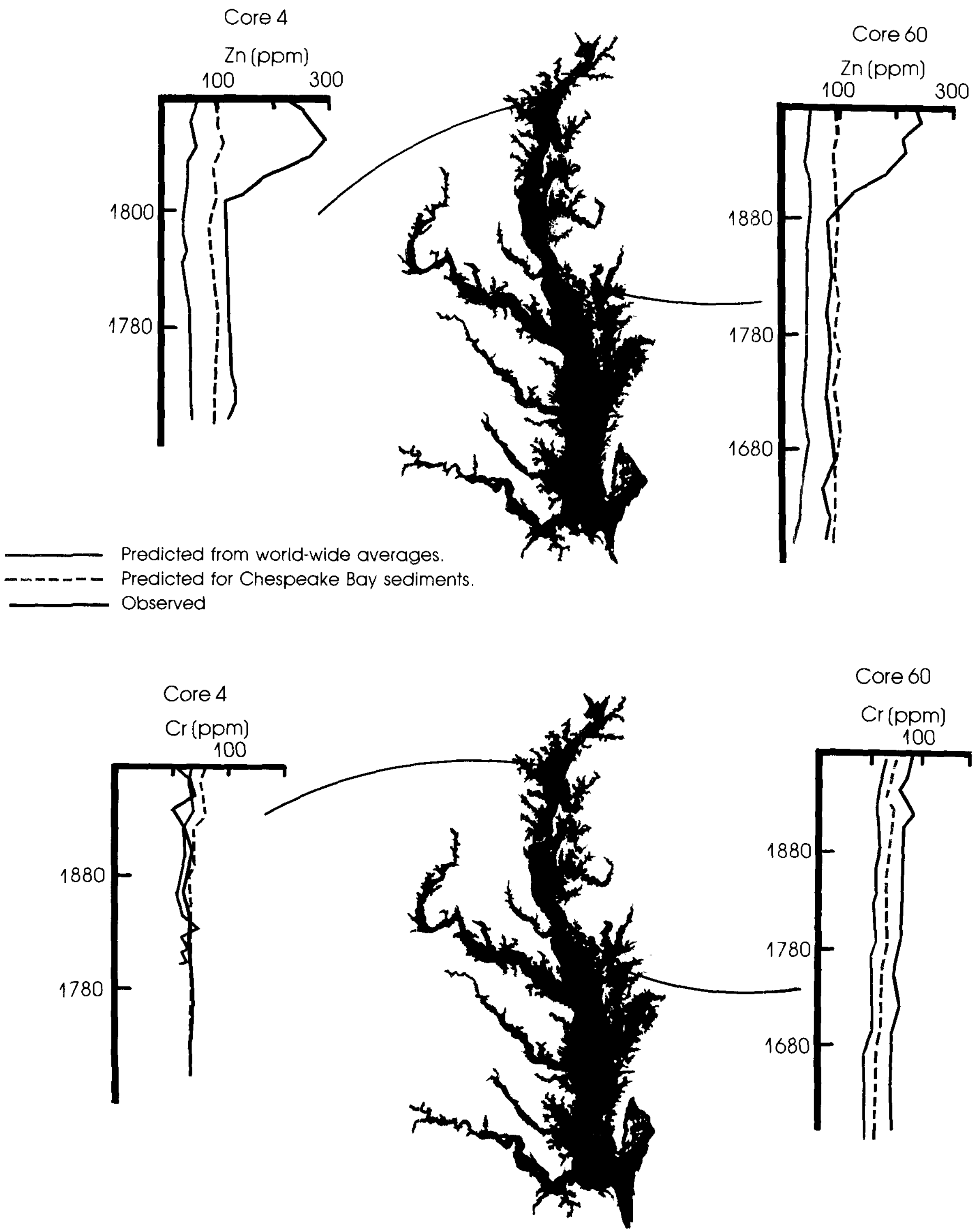


FIGURE 15. Zinc and chromium concentrations down-core compared to natural background levels.

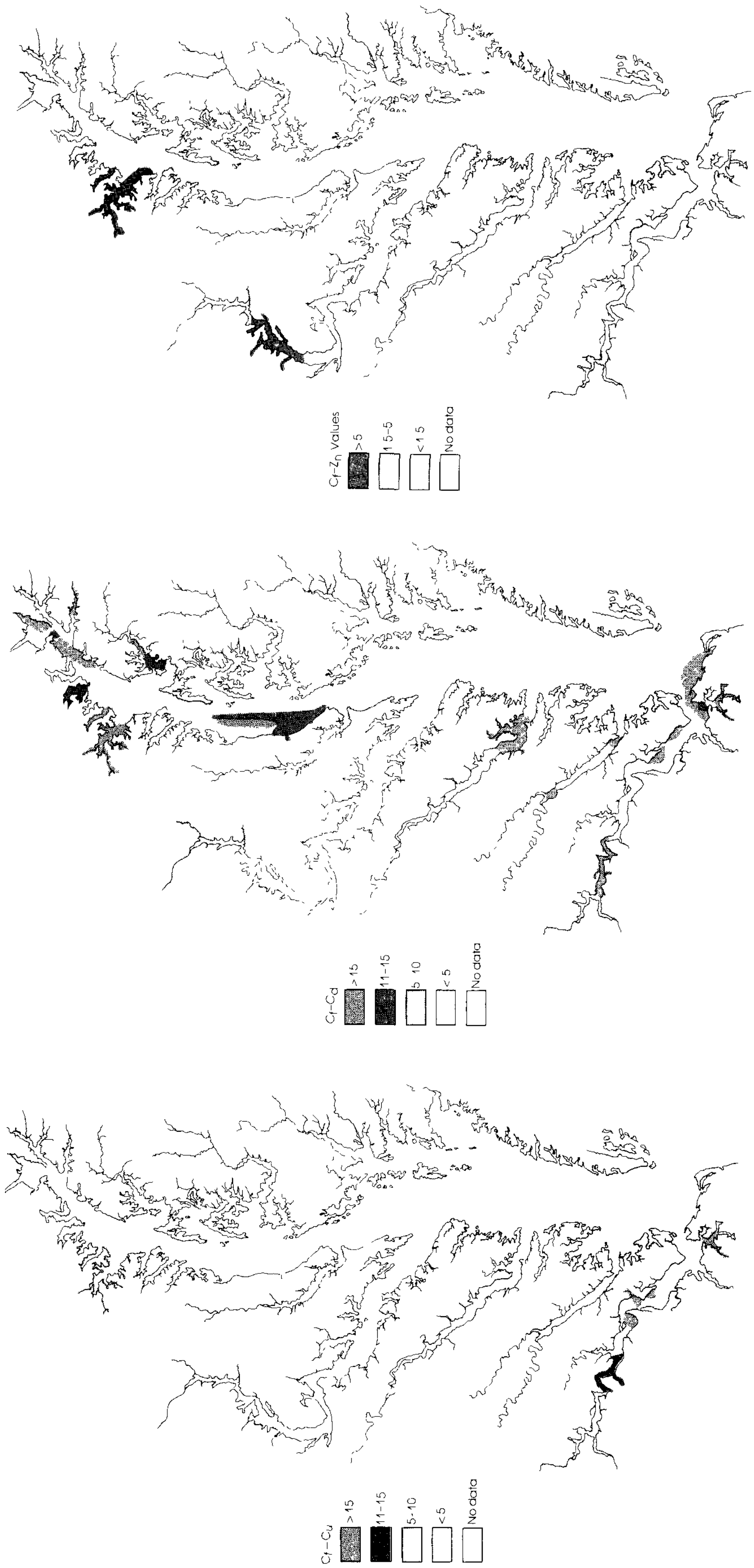


FIGURE 16. Copper, cadmium, and zinc enrichment in the Bay based on the contamination factor (C_F).

parent from Figure 17 that the most contaminated areas are near industrialized parts of the Bay such as Baltimore and Norfolk. For example, the Patapsco River contamination factors are Cd (64), Cu (27), Pb (19), Zn (6), Cr (5), and Ni (0.1). The upper Potomac and upper James Rivers, upper-mid Bay, and small sections of the Rappahannock and York are moderately contaminated. Minor contamination occurs in the Susquehanna Flats and the lower Potomac River. Although these trends are suggestive, it is important to evaluate the metal levels in terms of their toxicity to benthic organisms in the Bay. This will be discussed in Chapter 3.

Actual concentrations of metals in the sediments are given in Appendix B, Tables 5, 8, and 12. Figures are in ug/g (or ppm). It is interesting to note that mean concentrations of Cu, for example, often exceed the level at which burrowing activity of a west coast clam, *Protothaca staminea*, is greatly reduced — 33 ppm (Phelps, in prep.). However, failure of Chelex-sorbed Cu to affect burrowing rates indicates that tightly bound, less bioavailable forms of the metal have a much lower potential toxicity. Dissociation of metals from the sediments and availability to biota will determine potential biological impacts, more than bulk concentrations *per se*.

SUMMARY OF METALS IN THE WATER COLUMN

An evaluation of the dissolved metal concentrations at specific locations reveals significantly measurable levels throughout the Bay (Kingston 1982). The dissolved content of mean Cd, Cr, Ni, and Zn is higher in the upper Bay segments CB 1 to 3 than elsewhere. Concentrations decrease seaward in more salty water, segments CB 6–8. Cadmium exhibits the greatest seaward change, whereas Zn displays the least change. The Bay-wide decrease of Cr correlates with salinity, suggesting that Cr comes from the Susquehanna River. Its concentration partly results from dilution and conservative mixing of river water with ocean water (Figure 18). The other metals, Cd, Cu, Ni, and Zn, are affected by sedimentological and biological processes, in addition to dilution and mixing, that tend to remove them from solution. The Susquehanna River is a significant source of metals to the Bay and follows the general trend that rivers carry higher metal loads than does seawater.

Particulate metal concentrations in the water column are also quite variable. The concentrations (ug L^{-1}) tend to vary with changes in total suspended material as it resuspends and settles to the bed with changes in tidal-current strength. However, general trends are apparent in the average mean concentrations for the different Chesapeake Bay segments. Figure 18 shows the average particulate concentrations from five cruises between March through September 1979 to 1980 (Nichols et al. 1982).

Mean concentrations of particulate Co, Cr, Cu, Ni, Pb, and Zn per liter of water are greatest in the upper Bay (CB 1–3), a zone near the Susquehanna River that includes the turbidity maximum. Localized high concentrations occur in the bottom water of CB-2, particularly where the concentrations of total suspended material are high. Particulate metal concentrations vary widely in CB-2 with time, ranging more than 8-fold within three hours as a result of sediment resuspension from the bed. An analysis of metal enrichment based on Cd to Fe ratios indicates that Cd concentrations in mid-depth and surface waters of the central (CB-5) and lower Bay segments (CB 6–8) are 30 times greater than the background levels. It seems likely that phytoplankton uptake of Cd in these areas could be responsible for these elevated levels (Nichols et al. 1982).

The metals data in the tributaries of the Bay are primarily total metal concentrations (Maryland and Virginia "106" data). An assessment of this data suggests that the western shore tributaries of the lower Bay, such as the James River, have elevated metal content of total Cu, Pb, and Zn in their upper (TF) and lower (LE) segments, and a lesser content in central segments (RET). Other tributaries have localized concentrations of various metals in either upper or lower segments, except for the Rappahannock River which has higher Pb and Zn per liter of water in the central segment. Most of these trends reflect nearness to industrial or wastewater pollution sources; however, concentrations of Pb and Zn per liter of water in the central Rappahannock could be

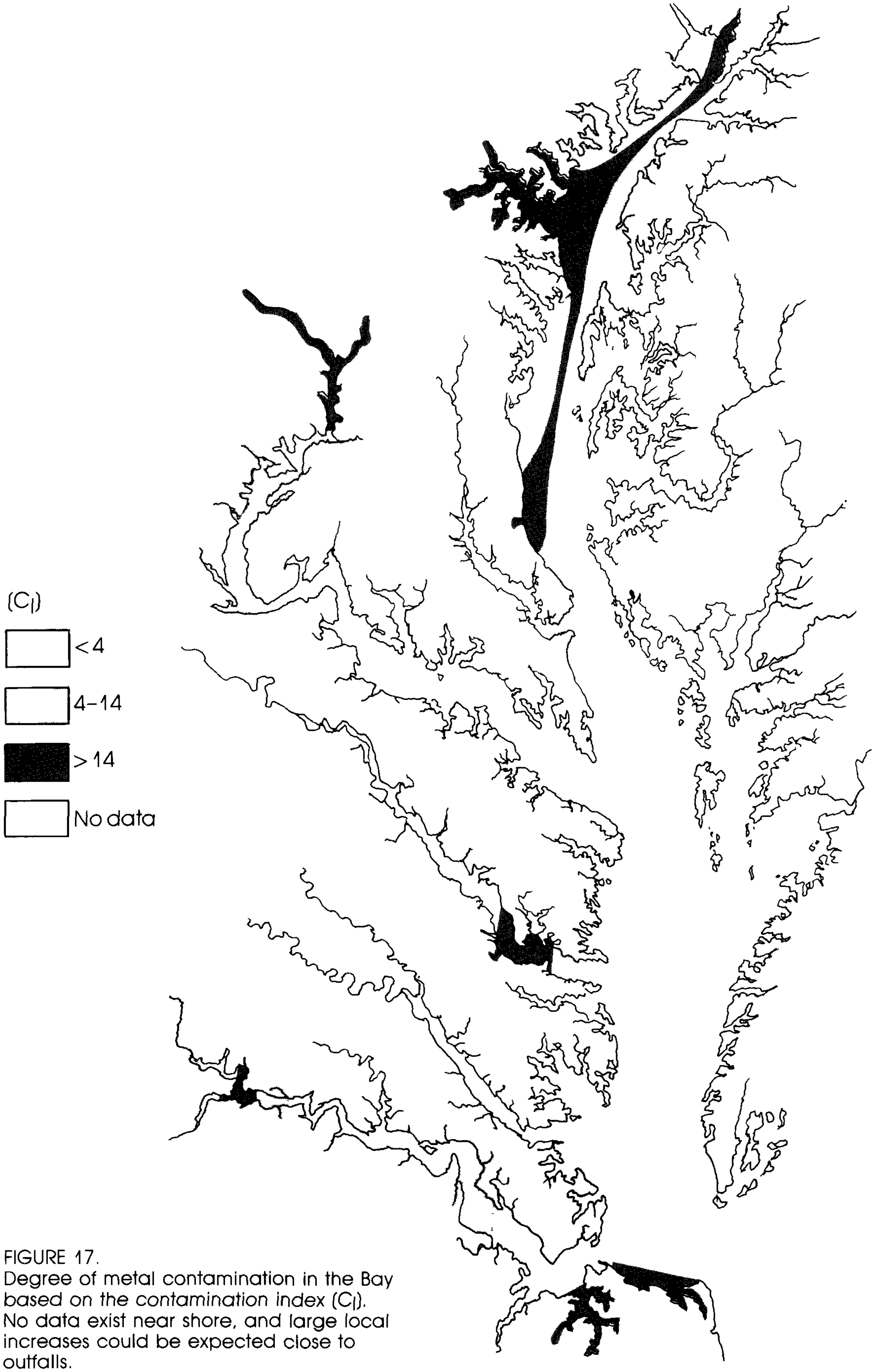


FIGURE 17.
Degree of metal contamination in the Bay *based on the contamination index* (C_I). No data exist near shore, and large local increases could be expected close to outfalls.

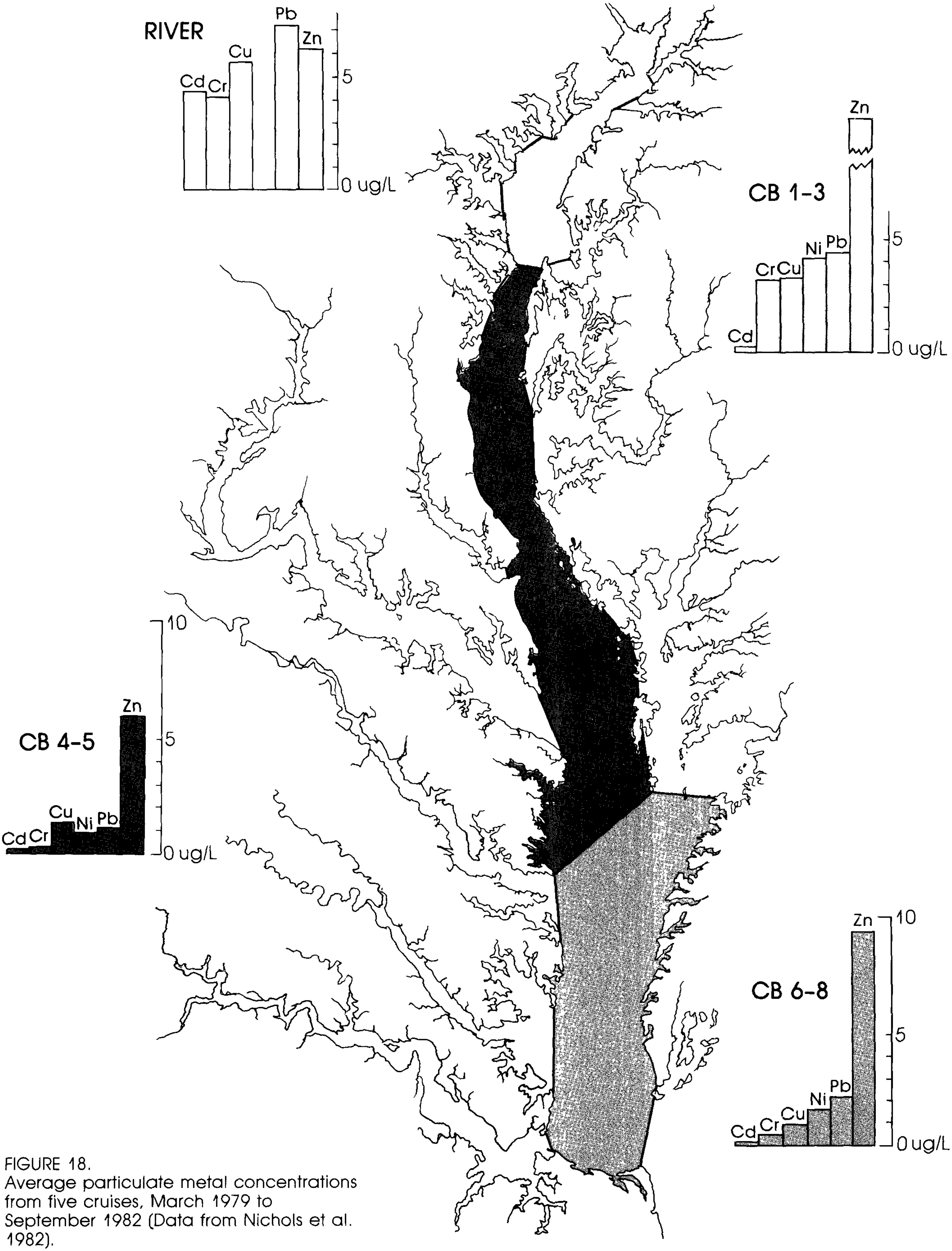


FIGURE 18.
Average particulate metal concentrations from five cruises, March 1979 to September 1982 (Data from Nichols et al. 1982).

enhanced by a high content of suspended material associated with the turbidity maximum.

In summary, the concentration of dissolved, particulate, and total metals does not warrant drastic action throughout the entire Bay basin. However, the quantity of samples (taken both before and after 1975) exceeding the EPA Water Quality Criteria raises general concern (Figures 19a and 19b) (see Appendix, B Section 3).

It can be seen that these criteria "violations" in the main Bay are concentrated in small tributaries and along the shore. It is reasonable to assume that the sampling distribution reflects specific monitoring or research needs, not a general coverage of the region. It should also be noted that sampling does not give a good indication of duration of instances of high metals concentrations. Such information is needed to assess potential biological impacts.

For the main Bay and lower estuarine zones of the western tributaries, estimated dissolved Cu and Ni exceed chronic and acute saltwater criteria. The chronic criterion for Zn is exceeded in Baltimore Harbor and the tidal-fresh portion of the Potomac River. On the lower eastern shore, the Pocomoke Sound region has significant amounts of Cu above the chronic criterion.

The broad distribution of "dissolved" observations above criteria should increase the awareness of the possible environmental effects of trace metals. However, it should be viewed in the context of the ecology of the Bay, the distribution of sampling, and the general lack of information on the form of the metal. This will be further discussed in the chapter on relationships between water and sediment quality and living resources (Chapter 3).

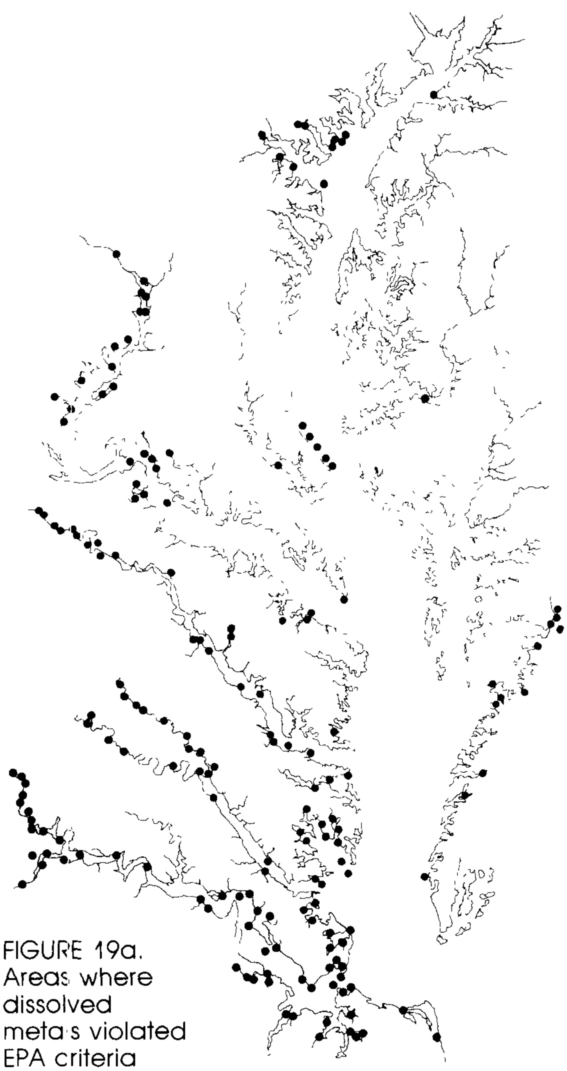


FIGURE 19a. Areas where dissolved metals violated EPA criteria before 1971 to 1975.

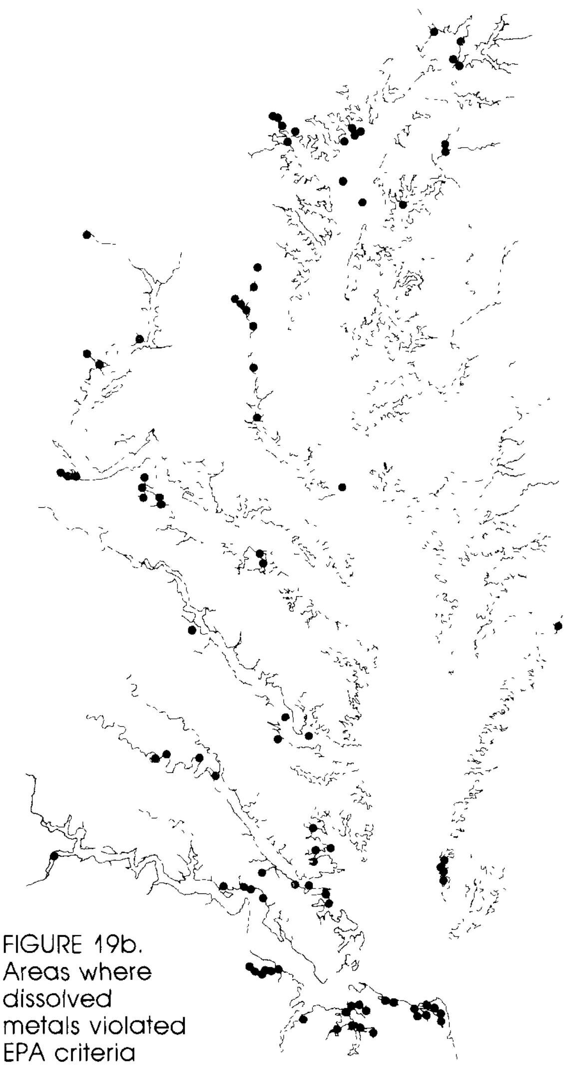


FIGURE 19b. Areas where dissolved metals violated EPA criteria after 1975.

SECTION 7

METALS AND PESTICIDES IN OYSTERS

INTRODUCTION

When sediments and water are enriched by trace metals or pesticides, shellfish tend to bioaccumulate those substances in their tissue. The process of concentrating toxicants in shellfish tissue is known as magnification or biomagnification. An analysis of metals and pesticides in shellfish and finfish tissues provides information on their suitability for human consumption. Because the organisms act as biomonitors, an assessment also enables detection of excessive metal and pesticide loadings in the environment and helps to characterize the water quality of the Bay. It is well known that certain metals and pesticides can cause severe human health effects, such as neurological disorders, muscle and bone deterioration, etc. To protect public health, the Food and Drug Administration has established standards for some toxic substances in the edible portions of finfish and shellfish. These standards are called "FDA Action Levels" and are summarized in Table 15. Unfortunately, action levels have not yet been developed for many of the metals and organic compounds that are frequently found at elevated levels in the Bay. Concentrations of toxic chemicals in finfish and shellfish that appear to be above background levels (i.e., two standard deviations above the mean) should be investigated. Furthermore, an evaluation of toxic chemical concentrations in organisms allows a better understanding of the water and sediment quality of the Bay. A more detailed discussion of the ecological implications of these concentrations is included in Chapter 3.

DATA ANALYSIS

The data on metal and pesticide levels in oysters were compiled and analyzed by the Maryland Department of Health and Mental Hygiene (Eisenberg and Topping 1981) and the Virginia State Water Control Board (Gilinsky and Roland 1983). Data for the State of Maryland cover the period 1976 to 1980. The data set for Virginia includes data from the Virginia Bureau of Shellfish Sanitation, the Virginia Institute of Marine Science (VIMS), the Virginia State Water Control Board, and other studies. It covers a time span from 1967 to 1980 for metals and from 1964 to 1980 for pesticides. In both state-wide data bases, tissue samples were analyzed for several heavy metals including Cu, mercury (Hg), Zn, Pb, Cd, Cr, and arsenic (As). The pesticide and polychlorinated biphenyl concentrations included in the data bases are DDT, DDD, DDE, chlordane, dieldrin, and polychlorinated biphenyls (PCBs). The sampling station locations are shown in Appendix B.

To characterize the current conditions and trends, oyster data were analyzed by CBP segments and regions. Univariate statistics for each year were calculated by river basin and/or segments. The mean levels of pesticides, PCBs, and metals in oysters are summarized in Appendix B (Tables 15 to 21).

In Virginia waters, oyster tissue metal concentrations in the York and Rappahannock Rivers are highest in the riverine-estuarine transition segment (Appendix B, Section 7, Tables 15 through 18), possibly reflecting predominately nonpoint source fall line metal transport. In addition, there were historical mining operations in the York River basin. In the James River basin, metal tissue concentrations are highest in the Elizabeth River and lower estuary segment (Appendix B, Section 7, Tables 15 through 18). Pesticide concentrations in oyster tissues are highest in the Elizabeth River,

TABLE 15.
***FDA ACTION LEVELS FOR SHELLFISH AND FINFISH**

	Shellfish	Finfish
Chlordane	0.3 ppm	0.3 ppm
DDT, DDE, DDD	5.0 ppm	5.0 ppm
Dieldrin	0.3 ppm	0.3 ppm
Heptachlor	0.3 ppm	0.3 ppm
Heptachlorepoxide	0.3 ppm	0.3 ppm
Kepone	0.3 ppm	0.3 ppm
Mercury	1.0 ppm	1.0 ppm
Mirex		0.1 ppm
Polychlorinated Biphenyls	5.0 ppm	5.0 ppm
Toxaphene		5.0 ppm

*United States Department of Health of Human Services Public Health Service, Food and Drug Administration. 1980. Action Levels for Poisonous or Deleterious Substances in Human Food and Animal Feed.

with the lower estuarine portions of the James and York Rivers having similar tissue concentrations (Gilinsky and Roland 1983).

In the Maryland tributaries, the metal concentrations in oysters were highest in the Patuxent River and the upper Bay. High levels of chlordane and PCBs in oyster tissues were found in the east Chesapeake (Kent Island) and the west Chesapeake area from Baltimore Harbor to the Rhode River (Eisenberg and Topping 1981).

There were not enough data for a detailed analysis of metals and pesticides in fish tissues in Maryland or Virginia. However, mean tissue DDE and PCB concentrations were calculated for several fish species having greater than twenty data points for Virginia waters (Gilinsky and Roland 1983). It appears that bluefish and channel catfish have higher DDE concentrations than the other fish species, and that PCB concentrations are highest in channel and white catfish. A similar trend was noted for bluefish in Maryland waters (Eisenberg and Topping 1981).

BAY-WIDE SUMMARY

Cluster Analysis of Metal Concentration by Segment

Cluster analysis was used to delineate groups of CBP segments with similar mean metal concentrations in oyster tissue. This provides a more objective method for ranking segments based on their relative level of metal contamination. Also, it allows identification of segments where processes are leading to high metal availability to organisms. The means were based on from one to 411 tissue samples per segment. The analysis was carried out separately for each of the following metals and chemicals: Cu, Cr, As, Pb, Hg, and Zn. Clustering was carried out via the agglomerative average linkage method in SAS (SAS User's Guide, *Statistics*, 1982 Edition. SAS Institute, Inc.). Clustering was halted at five groups. These groups were then used to define the class intervals shown in Figures 20a through g, and in Table 16 (LC50 values are given in Appendix B). The lower limit for each class was calculated as the mid-point between the minimum mean concentration (i.e., data value) in each group and the maximum mean concentration in the group with the next lowest average concentration. The upper limit for each class was calculated as the midpoint between the maximum mean concentration in each group and the minimum mean concentration in the group with the next highest average concentration.

The metal levels in oyster tissues are different throughout the Bay. Levels of metals in oyster tissues vary among the segments depending on the

TABLE 16.
CLASSIFICATION OF MEAN METAL CONCENTRATIONS IN OYSTER TISSUES IN CHESAPEAKE BAY BASED ON CLUSTER ANALYSIS

Class	Zinc 200	Copper 8.7	Mercury 0.015	Arsenic 0.023	Lead 0.05	Cadmium 0.32	Chromium 0.40
I	RET-5	RET-5	EE-3	ET-6	ET-7	WT-1	WT-7
	WT-1	WT-1	EE-1	ET-8	ET-8	RET-5	ET-7
			WT-8	WT-8	ET-5		WT-8
			ET-4	WT-7	EE-2		ET-6
			RET-1	CB-3	ET-6		CB-5
			CB-3		WT-7		EE-1
			CB-5		CB-3		EE-2
			WT-7		WT-8		CB-4
			ET-7		LE-2		ET-4
					RET-1		LE-2
					LE-1		RET-1
					ET-4		LE-1
							CB-3
							ET-8

Concentrations, mg/kg (ppm)

Class	Zinc 200-747	Copper 8.7-51.7	Mercury 0.016-0.03	Arsenic 0.023-0.06	Lead 0.05-0.115	Cadmium 0.32-0.86	Chromium 0.40-1.56
II	LE-2	WT-5	EE-2	EE-1	CB-4	LE-2	ET-5
	LE-4	ET-5	ET-5	ET-5	EE-3	RET-3	EE-3
	EE-1	WT-8	LE-2			EE-3	
	RET-3	LE-4	LE-1			WE-4	
	LE-3	CB-8	CB-4			LE-3	
	CB-8	LE-3				WT-5	
	WE-4	WE-2				ET-8	
	EE-3	WE-4					
		CB-5					
		RET-3					
		EE-2					
		ET-7					
		ET-6					
		ET-8					
		LE-1					
		EE-3					
		ET-4					
		CB-4					

(continued)

Table 16 (continued).

Concentrations, mg/kg (ppm)

Class	Zinc 748-1381	Copper 51.8-84.7	Mercury 0.031-0.06	Arsenic 0.061-0.105	Lead 0.116-0.24	Cadmium 0.087-1.34	Chromium 1.57-2.98
III	WT-5	RET-1	WT-5	LE-2	EE-1	ET-4	CB-8
	EE-2	RET-4	ET-6	EE-3		EE-2	
	LE-1	WT-7		CB-4		EE-1	
	LE-5	CB-3		ET-7		ET-7	
	RET-1			ET-4		ET-5	
	ET-6					CB-8	
	WT-8					ET-6	
	RET-4					CB-5	
	ET-5						
	ET-7						
	ET-4						
	CB-5						
	CB-3						

Concentrations, mg/kg (ppm)

Class	Zinc 1382-2784	Copper 84.8-104.2	Mercury 0.061-0.165	Arsenic 0.106-0.175	Lead 0.241-0.36	Cadmium 1.35-2.01	Chromium 2.99-4.0
IV	WT-5	Eliz.R.	ET-8	LE-1	CB-5	WT-8	LE-4
	ET-8			EE-2		LE-5	RET-4
				RET-1		RET-4	Eliz.R.
						CB-4	
						LE-4	
						Eliz. R.	

Concentrations, mg/kg (ppm)

Class	Zinc 2764	Copper 104.2	Mercury 0.165	Arsenic 0.175	Lead 0.36	Cadmium 2.01	Chromium 4.00
V	Eliz.R.	LE-5	WT-1	CB-5	WT-5	RET-1	LE-5
				WT-5		LE-1	RET-3
						CB-3	
						WT-7	

metal (Figures 20a to g). Highest levels of Cu (Class 5) were found in the James River (LE-5) and highest levels of Cr were in the James River (LE-5) and the Rappahannock Rivers (RET-3). Highest levels of Zn occur in oyster tissues in the Elizabeth River. The Patuxent River (RET-1 and LE-1), the Severn River (WT-6), and a main Bay segment (CB-3) had the highest levels of Cd in oyster tissues. Highest levels of As were found in oysters in a main Bay segment (CB-5); highest levels of Pb were in the Patapsco River (WT-5); and highest levels of Hg were in the Bush River (WT-1). In contrast, lowest concentrations (Class 1) tend to occur in smaller tributaries, but there are many exceptions (Table 16).

Mean Cu and Zn levels in Bay oyster tissue correlate with the corresponding contamination factors (C_f) in surface sediments indicating some relationship between metal sediment enrichment and metal concentration by oysters (Chapter 3). Copper, Cd, and Zn levels in oyster tissue tend to be higher near the more urbanized centers such as the Patuxent River, the upper Bay, the lower James River, and the Elizabeth River. They are lower in the Potomac River, the tidal-fresh segment of the Rappahannock River, and Mobjack Bay. However, trace metal levels in tissues are influenced by a number of environmental factors which are discussed in Chapter 3. Furthermore, it should be noted that concentrations of these metals in the Chesapeake Bay are less than the averages reported by a survey of corresponding metal concentrations in oysters from Maine through North Carolina (Pringle and Shuster 1967). Throughout the period of record, there was little problem with shellfish or fish tissue metal contamination that violated FDA action levels.

The levels of pesticides in oyster tissue do not exceed FDA action levels and are similar to concentrations along the entire Atlantic Coast. Pesticide concentrations are highest in the Elizabeth River oyster tissues (Gilinsky and Roland 1983). Levels of oyster contamination by DDT in both states have decreased over time because it has presumably not been introduced into the system for several years and has broken down to DDE and DDD. The by-products of DDT are less toxic, and their levels are detectable, but not known to be harmful at these concentrations. Garreis and Pittman (1981) found levels of some pesticides in Choptank River oysters directly related to rainfall. This was in contrast to trace metal concentrations, and probably represents the nonpoint source of these pesticides.

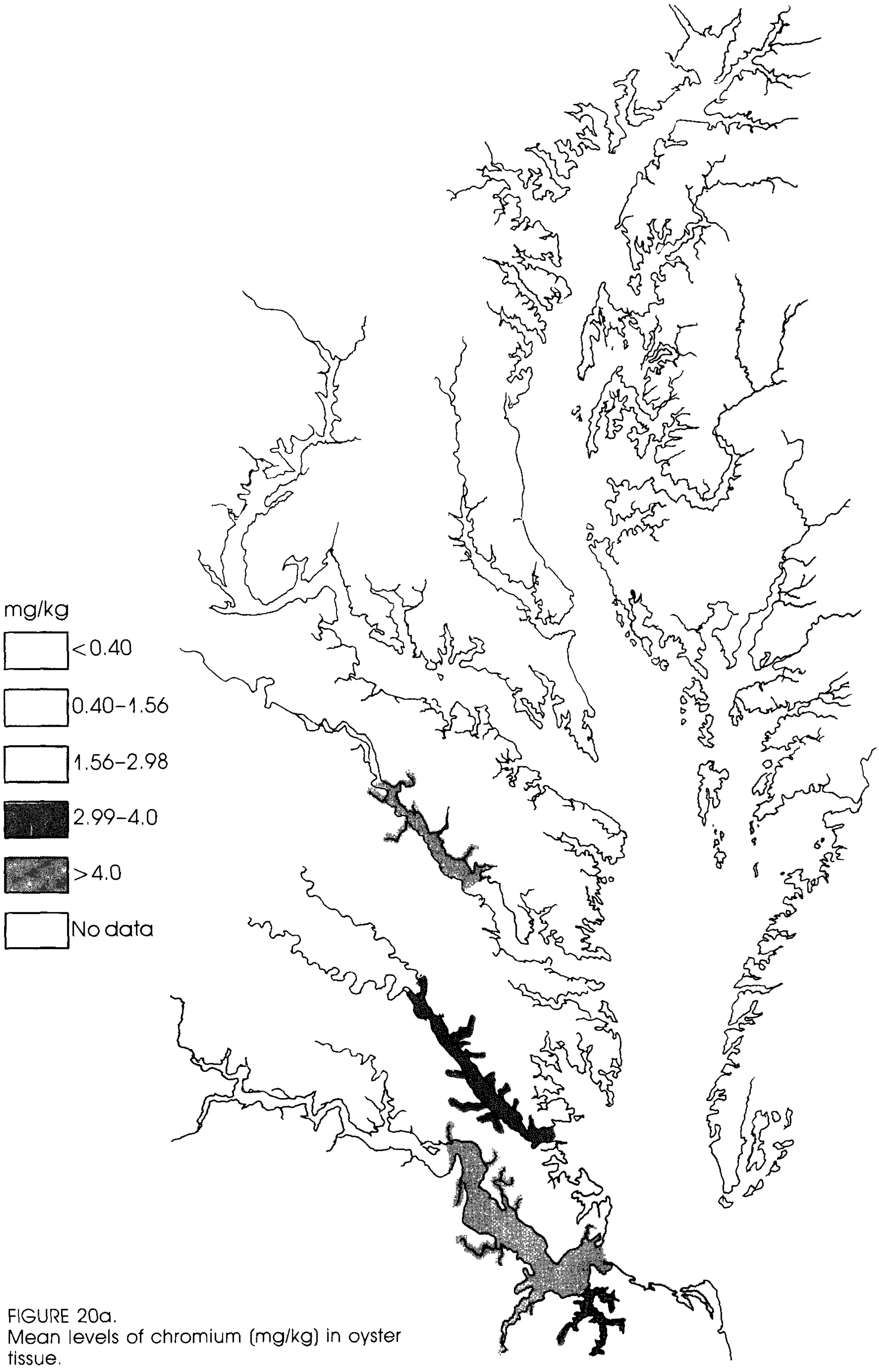


FIGURE 20a.
Mean levels of chromium (mg/kg) in oyster tissue.

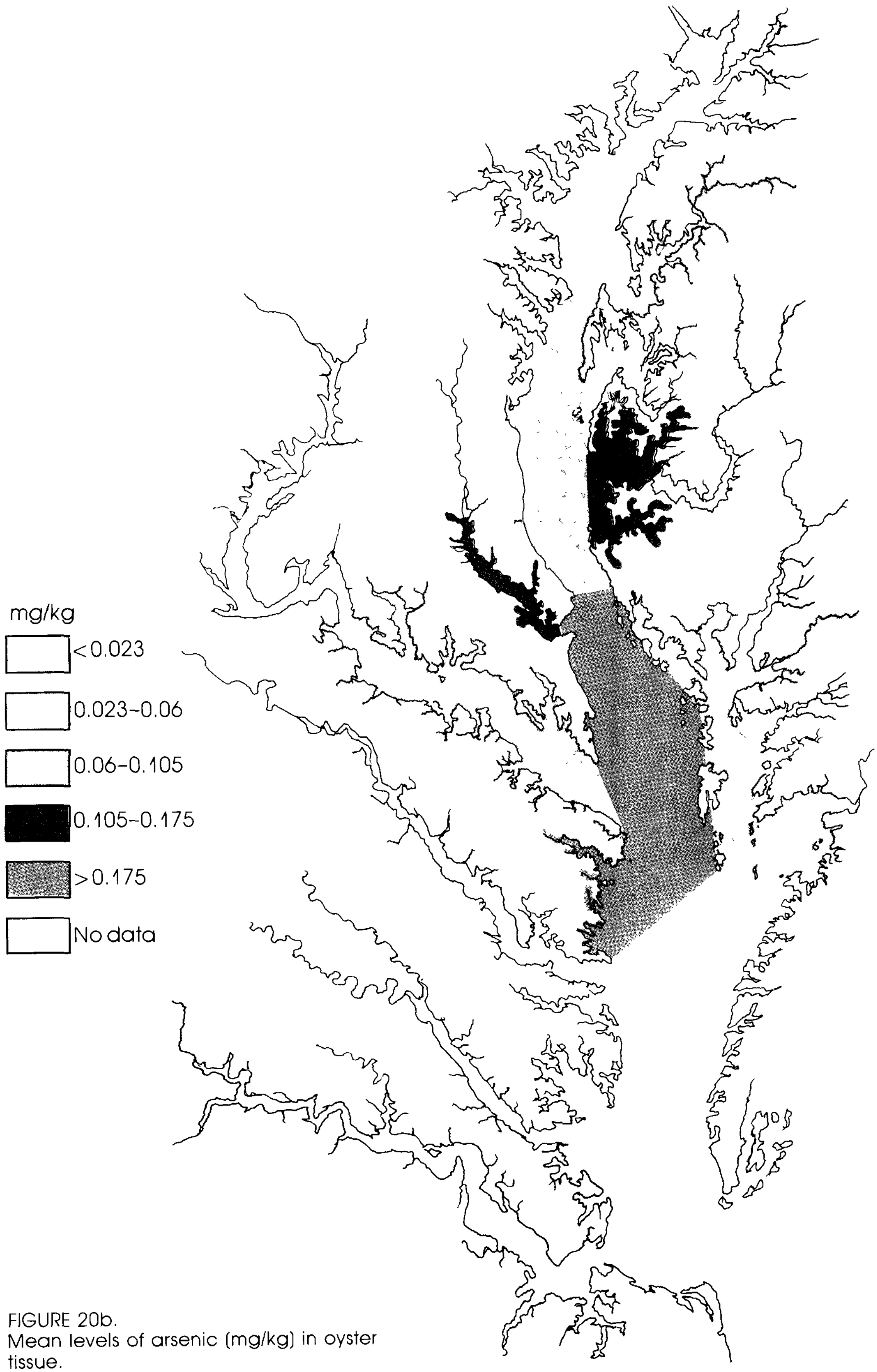


FIGURE 20b.
Mean levels of arsenic (mg/kg) in oyster tissue.

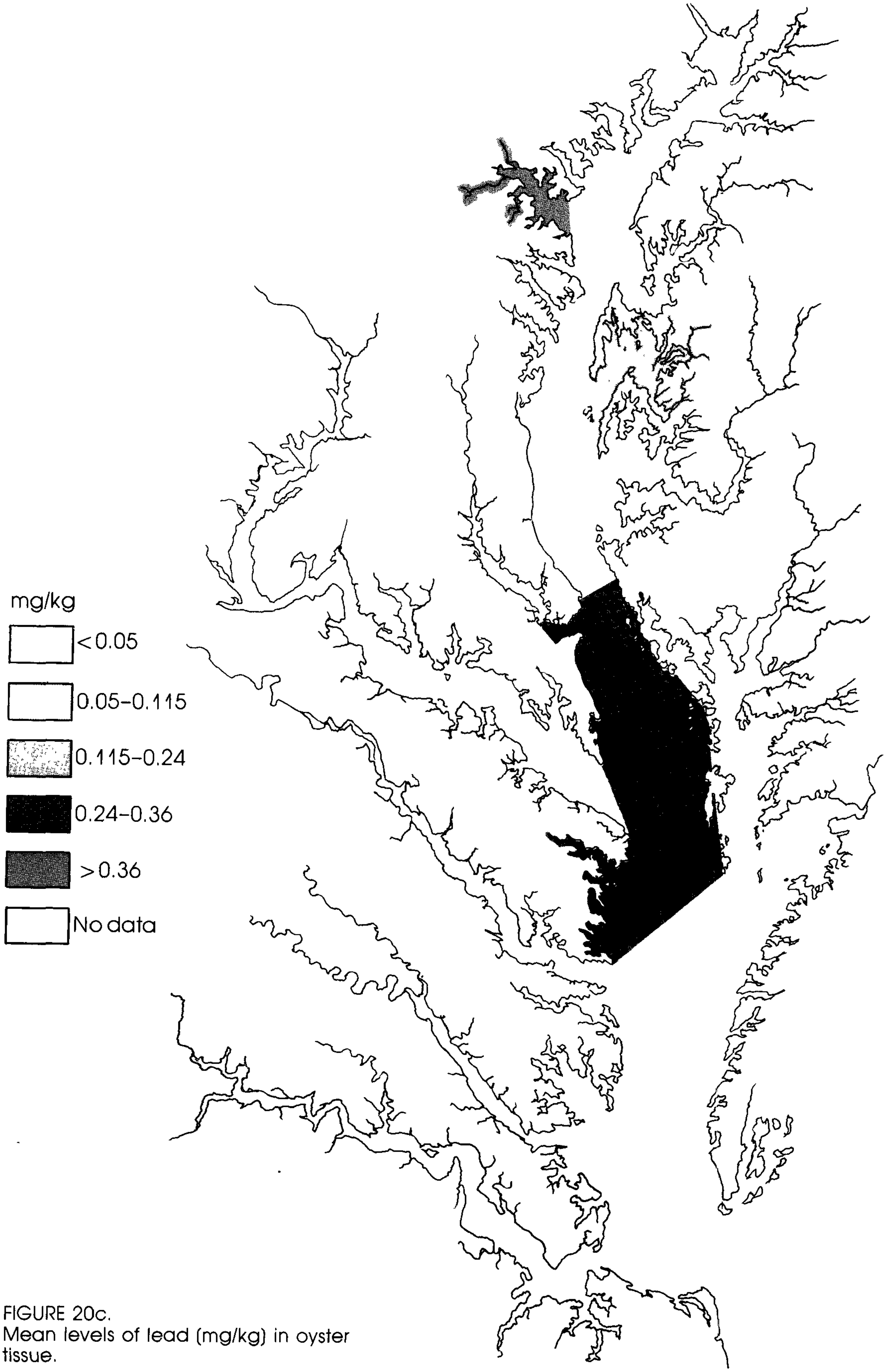


FIGURE 20c.
Mean levels of lead (mg/kg) in oyster tissue.

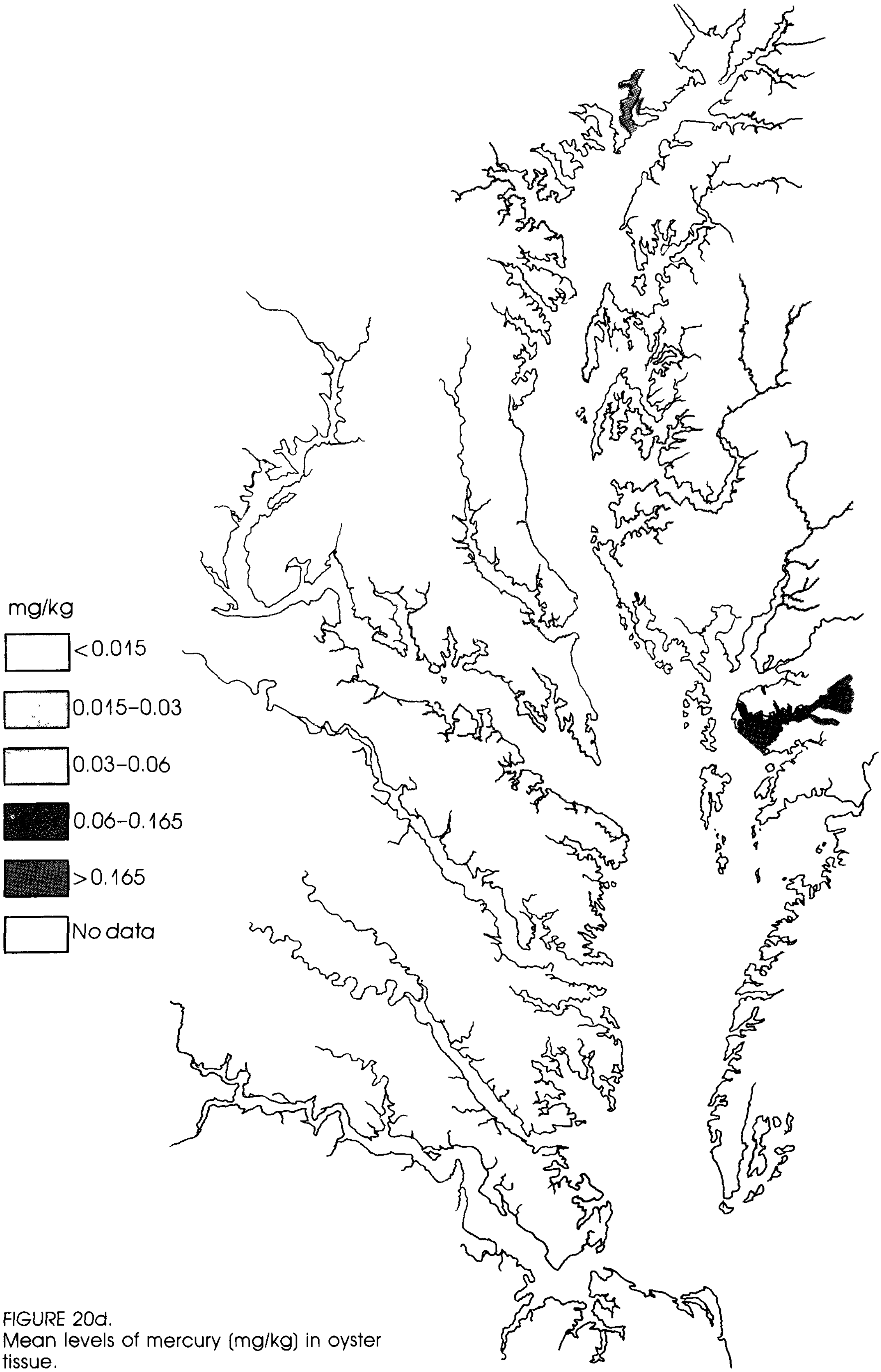


FIGURE 20d.
Mean levels of mercury (mg/kg) in oyster tissue.

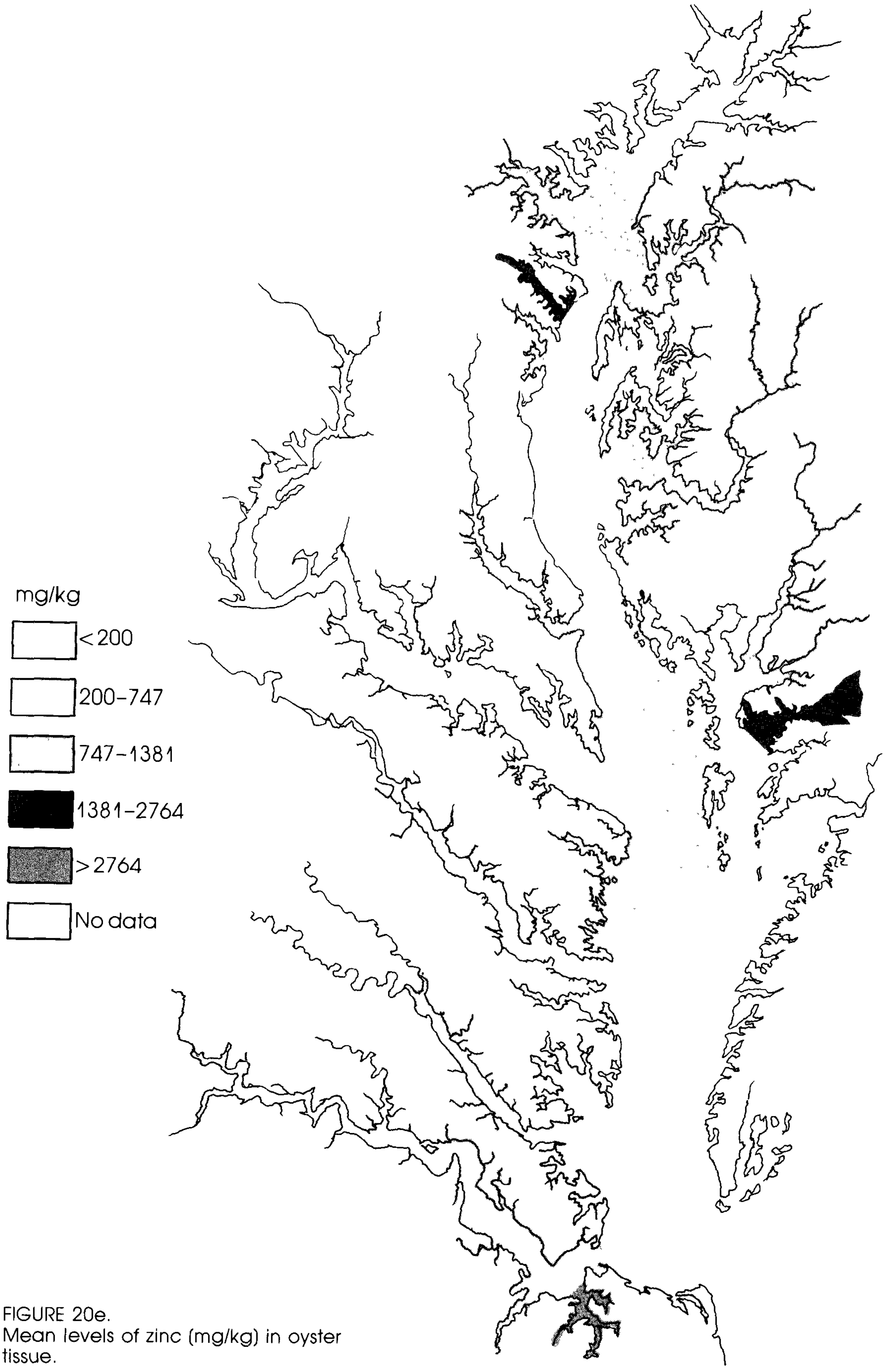


FIGURE 20e.
Mean levels of zinc (mg/kg) in oyster tissue.

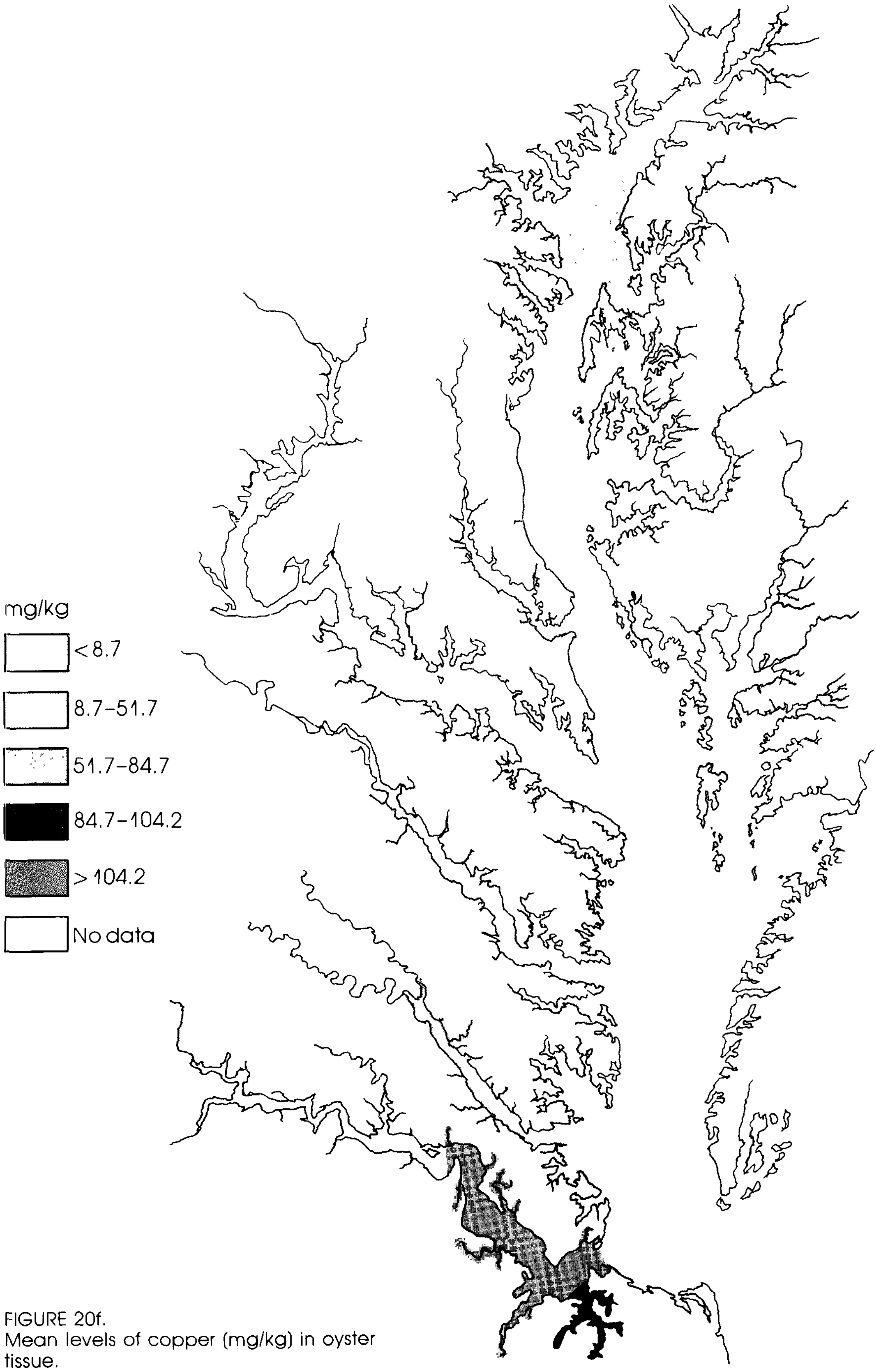


FIGURE 20f.
Mean levels of copper (mg/kg) in oyster tissue.

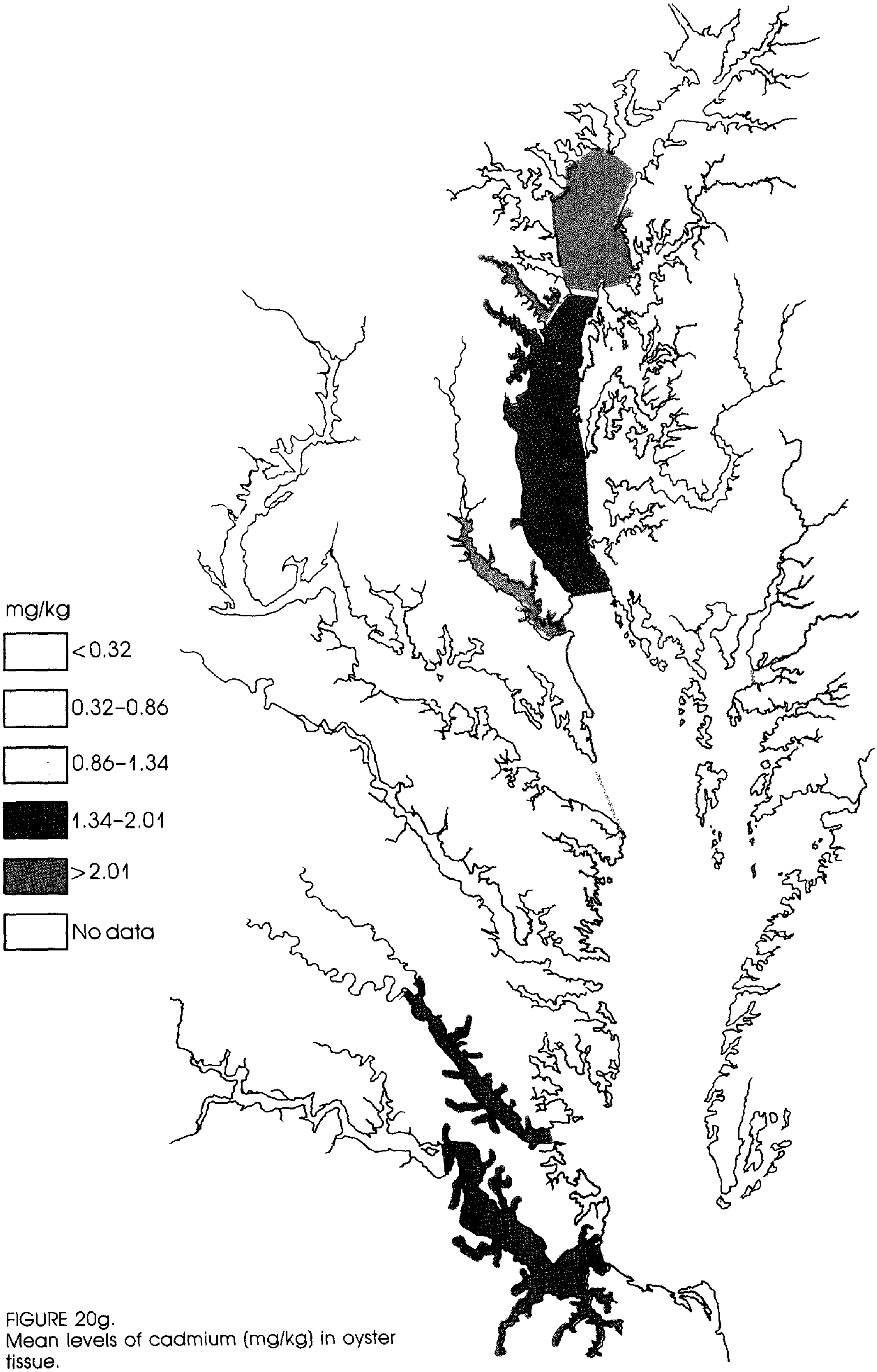


FIGURE 20g.
Mean levels of cadmium (mg/kg) in oyster tissue.

SECTION 8
SUMMARY BY GEOGRAPHIC AREAS

This section summarizes the current conditions and trends by geographic area: main bay, Eastern Shore, and western shore. Present status and current trends are both described. These are not synonomous: an area may currently have high levels of nutrients, but nevertheless show a significant decline over time, or vice-versa. Data tables giving information on the basis for the following assessment are in Appendix B, Section 8.

MAIN BAY (CB 1-8)

The main Bay is primarily influenced by the Susquehanna River. The Susquehanna accounts for approximately 50 percent of the freshwater inflow to the Bay (70 percent above the Potomac). As a consequence, this river is the most significant single source of nutrients and toxic chemicals to the main Bay (U.S EPA 1982b). For this reason, the upper Bay is enriched with nutrients (class 5) (Figure 1) and metals in the water column (Figure 18). Moving down the Bay, concentrations begin to decrease. Nutrient levels in the water column are still high in the mid-Bay, but decrease to background levels in the lower Bay. The metals in the water column are relatively high in the upper Bay (CB 1-3), drop to intermediate levels off the Patuxent and Potomac (CB-5), and then decrease to background levels in most parts of the lower Bay.

The concentration of organic compounds and the degree of metal contamination in the sediment follow the same general pattern as the water-column nutrients and metals. Because metals and organic compounds tend to preferentially adsorb to fine grain sediments, an effort was made to "normalize" the organic compound and metal values, so that they reflect comparative enrichments. In Figure 13, it is apparent that the upper Bay is enriched with organic compounds. Levels decrease down-Bay, then increase at stations off Hampton Roads. The degree of metal contamination (C_I) in the sediment follows a slightly different pattern (Figure 17). It is moderate in the Susquehanna Flats, then increases to high levels off Baltimore, probably from long-term input from polluted Patapsco River sediments in this area. The C_I then decreases down the Bay to levels of minimal contamination, except that very high values are found in the Elizabeth River, Hampton Roads, and Lynnhaven areas.

In terms of trends, one sees an increase in at least one form of nitrogen or phosphorus in most of the upper and mid-Bay (Figure 5). As was discussed in Section 2 of this chapter, this screening assessment actually obscures some individual nutrient trends. Some form of nitrogen is increasing in CB 1-4; similarily, segments CB-1, 2, 3, and 5 show increases in some form of phosphorus. Nitrate or nitrite-nitrogen show increases in CB-1, 2, and 3, while TKN and NH3 generally show no trends or declines. Chlorophyll is increasing annually or seasonally in every main Bay segment where data exist: CB-1, 2, 3, 4, 5, and 7. These trends may be contributing to the temporal and spatial increases in summertime anoxic bottom water in the upper and mid-Bay.

EASTERN SHORE (ET 1-10, EE 1-3)

The Eastern Shore tributaries and embayments are shallow, highly productive waters. Nutrient inputs from land runoff have long contributed to this productivity. However, assessment of present status and trends shows a change in nutrient enrichment in many of the eastern tributaries, and in Tangier and Pocomoke Sounds. All tributaries except the Sassafras, Manakin, and Annamessex Rivers are class 4 or 5 waters, based on nitrogen. We see an increase in at least one form of nitrogen or phosphorus in all but the above-mentioned rivers; the lower Choptank River and Eastern Bay also show no nutrient trends. Individual nutrient species trends are less consistent. Again, when NO_3 or NO_2 increase, these trends are generally accompanied by decreases in ammonia or TKN. Chlorophyll shows annual or seasonal increases in ET-5, ET-10, and EE-1. Data are relatively incomplete in much of the Eastern Shore for chlorophyll; it is possible that more complete information would reveal trends in other tributaries.

Data are also few for toxicants in Eastern Shore areas. An assessment of existing information indicates that there are relatively few problems with toxic substances. Although the data are very sporadic, they indicate no metal contamination in the Eastern Shore sediments (Figure 17). There have been relatively few recorded observations of water column metals and pesticides that exceed EPA water quality criteria. Toxic chemicals are apparently not a problem in the Eastern Shore tributaries and embayments at this time except for local situations. Agricultural chemicals (pesticides and herbicides) might represent the major potential problem.

WESTERN SHORE (WT 1-8, TF 1-5, RET 1-5, LE 1-5)

The western shore of Chesapeake Bay is characterized by a series of small tributaries (WT 1-8) and major tributaries (Patuxent, Potomac, Rappahannock, York, and James Rivers). Most of the Bay's major population centers are located along these tributaries. It is, therefore, not surprising that many sections of the western shore are stressed with nutrients and toxic chemicals.

Western Shore Tributaries (WT 1-8)

Upper western shore tributaries are presently class 5 or 6 waters when ranked on the basis of TN or TP: these include segments WT-1, 2, 4, and 5. Segments WT 6-8 are class 2 or 3. Although there is a paucity of long-term data, it appears that some segments are experiencing degrading water quality. In fact, Back River (WT-4) in Maryland has reached a state in which its DO levels are decreasing from the presence of excessive organic material in the water (chl *a* maximum is 200 ug L^{-1}). Municipal sewage treatment plant discharges, failing septic systems, and storm water runoff all contribute to the problem.

Toxic chemicals are a problem primarily in the Baltimore area. There have been many recorded violations of organic compound and metal criteria in the water column in Baltimore Harbor (WT-5) and in the Back River (WT-4) areas. Also, the degree of metal contamination is higher in the Baltimore area than in any other area in Maryland (Figure 17). On the basis of this analysis, it is apparent that the Patapsco River is severely polluted. Fortunately, much of this pollution appears to stay in the harbor under average conditions due to the circulation pattern. Thus, the Patapsco River generally serves as a sink rather than a source of pollution except when humans transport its polluted sediments into the main Bay during dredging activities.

Patuxent River (TF-1, RET-1, LE-1)

The Patuxent River contains high levels of nutrients; it is class 6 for both TN and TP in the tidal-fresh reaches, improving to class 4 in the lower estuary. Its chlorophyll *a* maximum levels range from 200 ug L^{-1} in the tidal-fresh segment to 70 ug L^{-1} in the lower estuary. These values are among the highest observed in the Bay in similar segments. It is, therefore, not unusual in the summer to see minimum DO values approaching 2 mg L^{-1} in the tidal-fresh segment and anoxic conditions (0 mg L^{-1}) in the deep water of the lower estuary. An analysis of the long-term trend data indicates continuing water quality degradation, although a decline in TP in summer should be noted. In terms of toxic chemicals, there have been a few recorded dissolved metal criteria violations in the river, most often in the tidal-fresh areas. Unfortunately, there are insufficient data to assess the degree of organic chemical or metal contamination in the sediments.

Potomac River (TF-2, RET-2, LE-2)

The Potomac estuary supports major development and urbanization, particularly in its tidal-fresh reaches. Significant degradation of water quality had occurred by the mid-1900's. Because of public pressure in the late 1960's, an effort was made to significantly reduce the sewage treatment plant nutrient loadings to the river and to improve water quality. This policy appears to have had some positive effect on the Potomac River. We see a decrease in TP in the upper segment and a decrease in TN in the lower segment (Figures 3 and 4): seasonal TN declines are observed in TF-2 in summer. This general trend is further corroborated by Secchi depth measurements that have increased in the 1970's, indicating better light penetration due to lower levels of organic and inorganic material in the water.

In addition to the nutrient enrichment problems in the upper Potomac estuary, there are some problems with toxic chemicals. There have been a number of recorded violations of total residual chlorine and dissolved metal criteria in the upper Potomac (Figures 19a and b). Also, the degree of metal contamination is high (>14) in the upper reaches (Figure 17). This toxic chemical problem is not so evident in the lower portion of the river. The urbanized Washington, DC area is an important source of toxic substances to the river, although the major source of some metals is from above the fall line

Rappahannock and York Rivers (TF 3-4, RET 3-4, LE 3-4)

These rivers are considered to be the least impacted areas of the western shore. Nonetheless, they contain moderate levels of nutrients (class 4 or 5), particularly in tidal-fresh reaches. Trend analysis indicates that phosphorus is increasing in the lower and mid-Rappahanock River; nitrogen is increasing in the upper York River. No trends in chlorophyll were identified.

Toxic chemical problems are evident in several sections of the Râppahannock and York Rivers. Violations of EPA water quality criteria occur along both rivers (Figures 19a and b). The degree of metal contamination in the sediment is high (>14) in the lower Rappahannock and at the juncture of the Mattaponi and York Rivers.

The principal factor responsible for the high degree of contamination is Cd. It is possible that the high concentrations of Cd are natural: the result of weathering of the Fairhaven diatonaceous member of the Calvert Formation, known to be high in Cd and known to be exposed in the lower Rappahannock and along the Calvert Cliffs.

Unusually high metal concentrations in the water column are also found on the York at the juncture of the Mattaponi and Pamunkey Rivers (Figure 17). These findings justify continued monitoring of the Rappahannock and York Rivers for possible toxicant contamination.

James River (TF-5, RET-5, LE-5)

The James River contains high levels of nutrients (class 5 in TF and RET segments). Point sources are most important in this river, as the James River supports several major urban centers. In the upper James, TN and TP concentrations have reached a maximum of 2 mg L^{-1}l and 0.5 mg L^{-1}l and DO levels have dipped to 3 mg L^{-1}l during the period of record. However, both nitrogen and phosphorus are declining through most of the estuary. No trends in chlorophyll were identified.

Toxic metals and organic compounds occur in high concentrations in several areas of the James River. Recorded toxic violations occur all along the river but appear to be more concentrated in the uppermost reaches of the James River and in the Elizabeth River (Figures 19a and b). The degree of metal contamination in the sediment is very high in the Elizabeth River and the upper James and relatively low in the lower sections of the James River (Figure 17).

Although there are insufficient data to characterize the general level of organic compounds in the James River, it is valuable to look at the one compound that has been extensively studied— Kepone (Section 4). Discharged in tidal-fresh waters, relatively high levels were detected as far as 120 km from the source. Because organic compounds tend to adsorb to sediment, it is not surprising that high concentrations of Kepone are found in turbid waters of the James River and in the bed sediments of the turbidity maximum area. The Kepone incident has resulted in intensified efforts to improve water quality in the river.

CONCLUSIONS

From information presented in this chapter, several conclusions can be drawn:

- Upper and mid-Bay main stem, tidal-fresh, and transition reaches of major tributaries, and many smaller tributaries contain high levels of nutrients. Nutrient concentrations tend to decline down estuary.
- The pattern of toxic substances in water or sediments — metals and organic compounds — is generally similar. However, toxic contamination is also found near urban areas in the lower Bay.
- Chlorophyll *a*, an indicator of increased phytoplankton biomass and, thus, of nutrient enrichment, is showing an upward trend throughout the Bay main stem and in several tributaries.
- Nitrogen concentrations (including TN, NO_2, NO_3, TKN, and ammonia) are increasing in the upper half of Chesapeake Bay, and in several tributaries, including the Chester, Patuxent, Potomac, York, Nanticoke, and Wicomico Rivers. Declining trends occur in the Patapsco and James Rivers.
- Phosphorus concentrations (including TP and IPF) are increasing in the upper and mid-Bay, except for CB-4 where declining trends were observed. Increased phosphorus trends also occur in the Chester, Choptank, mid-Patuxent, lower Potomac, and Rappahannock Rivers, and in Tangier and Pocomoke Sound. Declines occurred in the upper Potomac and throughout the James.
- There is an increase in the duration and extent of low DO in deep water of Chesapeake Bay in summer. This trend is most pronounced in the upper and mid-Bay; there is no apparent anoxia in lower Bay deep waters, probably due to rapid mixing and exchange in this region. The DO trend is best explained by increased production of organic material due to elevated levels of nutrients in the upper half of the Chesapeake.
- Levels of trace metals and organic chemicals in the tissues of shellfish seem to parallel (in a general way) the observed patterns of toxic contamination in the Bay.

Although the ability to quantify this "pollution" is imperfect, we believe these conditions and trends warrant attention. Although the Eastern Shore contains moderate to high levels of nutrients, waters of this area show very little toxic chemical contamination. The lower Bay is relatively unimpacted except locally. With additional effort in reducing input, significant improvements in several areas of the Bay could be achieved. For example, increased pollution control in the Potomac and James Rivers has resulted in improved water quality at least with regard to nutrients.

SECTION 9

RANKING CBP SEGMENTS ACCORDING TO CURRENT WATER QUALITY CONDITIONS

INTRODUCTION

Ranking segments can help identify areas of the Bay experiencing degraded environmental quality due to high levels of nutrients and toxicants. A ranking system has been established to characterize the water and sediment quality of the 45 Bay segments (Table 17). Water nutrient concentrations, scaled linearly 1 to 6, and sediment toxic metal concentrations, scaled 1 to 5, are added for each segment to give an overall water quality value. A high value reflects high enrichment.

Nutrient Ranking

The overall nutrient ranking used here was developed from data compiled by the CBP of annual mean nutrient concentrations. Two variables, total phosphorus and total nitrogen, were each grouped into six gradations based upon their concentrations in the estuary. Ranges of TN and TP used to develop these ranks are given in Table 4 of this chapter. The overall rank was based on the nutrient in greatest abundance. Data were available for 43 segments (Figure 21).

Toxic Metal Ranking

The toxicant enrichment ranking shows the relative distribution of metals in bottom sediments of the Bay segments. It is based on the Contamination Index that sums the anthropogenic enrichment above natural levels for the six most frequently sampled metals in the surface sediments. After being mapped and contoured according to three divisions ($C_I \leq 4$, 4 to 14, >14, indicating less than 400 percent; 400 to 1400, and over 1400 percent enrichment)(Figure 17), the metal contamination map was over-laid by the CBP segmentation map of the Bay. Values ranking from 1 to 5 to represent the three categories and two gradations between them values were given for the 26 segments for which sufficient data existed.

As discussed earlier in this chapter, concentrations of organic substances in the sediments are too poorly characterized spatially to be of use in this assessment.

TABLE 17.
RANK OF CBP SEGMENTS ACCORDING TO SELECTED PRESENT WATER QUALITY CONDITIONS

Segment	TN	TP	Nutrient Overall	C_I	Water Quality* Overall
CB-1	5	3	5	3	8
CB-2	5	3	5	4	9
CB-3	5	3	5	4	9
CB-4	4	2	4	3	7
CB-5	4	2	4	2	6
CB-6	ND	ND	—	1	1+
CB-7	1*	ND	1	1	1
CB-8	ND	ND	—	1	1+
WT-1	5*	2*	5	ND	5+
WT-2	5	2	5	3	8
WT-3	3	2	3	3	6
WT-4	6*	6*	6	5	11
WT-5	6	3	6	5	11
WT-6	2	2*	2	ND	2+
WT-7	2	2	2	ND	2+
WT-8	3	3	3	ND	3+
EE-1	2	3	3	ND	3+
EE-2	2*	2*	2	ND	2+
EE-3	3	2	3	1	4
WE-4	2	2	2	1	3
TF-1	6	6	6	ND	6+
RET-1	5	5	5	ND	5+
LE-1	4	4	4	3	7
TF-2	5	4	5	4	9
RET-2	5	4	5	2	7
LE-2	2	2	2	2	4
TF-3	4	5	5	ND	5+
RET-3	2	3	3	ND	3+
LE-3	2	2	2	4	6
TF-4	3	4	4	ND	4+
RET-4	2*	3	3	ND	3+
LE-4	2*	3	3	ND	3+
TF-5	5	5	5	3	8
RET-5	5*	3*	5	1	6
LE-5	3	3	3	3	6

(continued)

Table 17 (continued).

Segment	TN	TP	Nutrient Overall	C_I	Water Quality* Overall
ET-1	4*	2*	4	5	9
ET-2	5	3	5	3	8
ET-3	3*	2*	3	ND	3+
ET-4	4	5	5	1	6
ET-5	5	3	5	1	6+
ET-6	5	2	5	ND	5+
ET-7	5	4	5	1	6
ET-8	3*	2*	3	ND	3+
ET-9	3*	2*	3	ND	3+
ET-10	4	3	4	ND	4+

* = Based on limited data (Table 4)
ND = Limited data
\+ = Ranking based on nutrients or C_I only

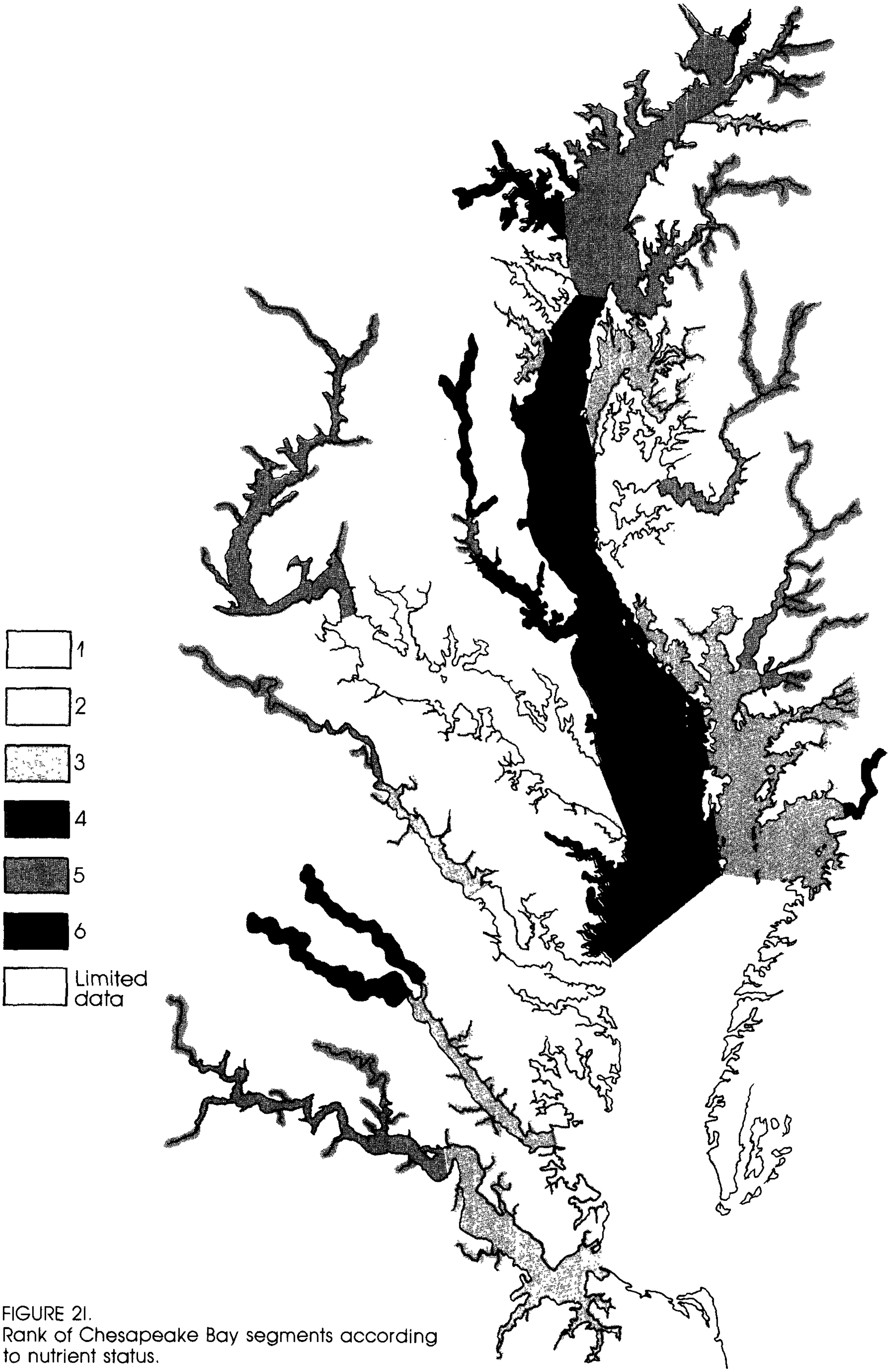
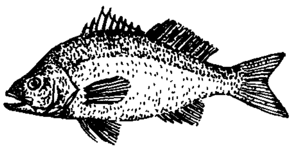
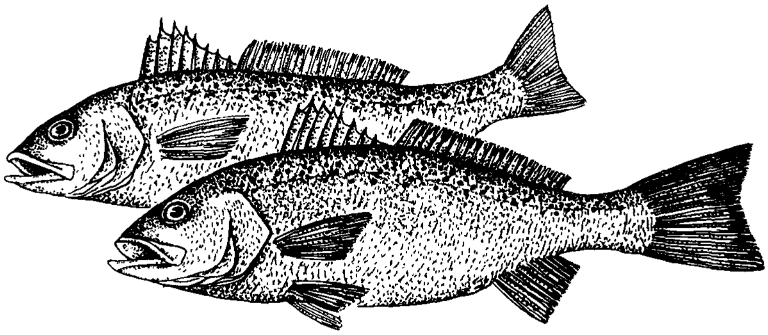
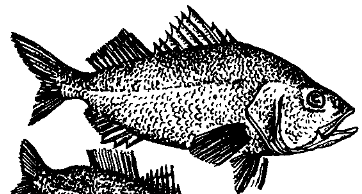
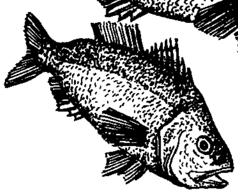


FIGURE 21.
Rank of Chesapeake Bay segments according to nutrient status.



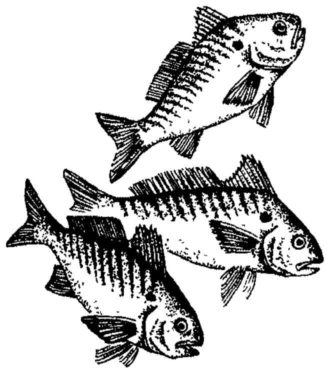


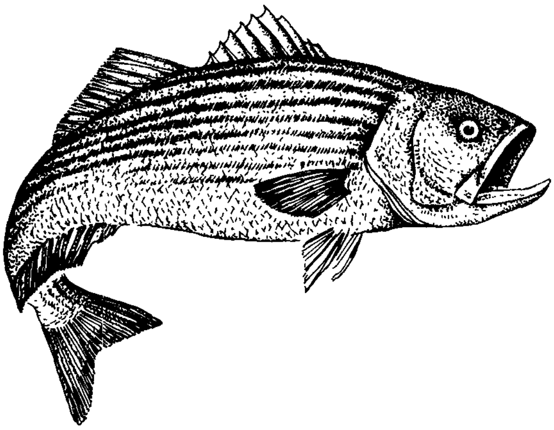


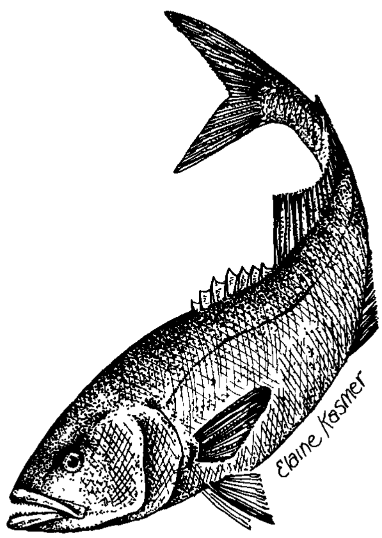


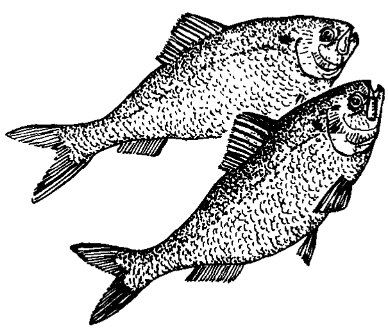
CHAPTER 2

LIVING RESOURCES: A HISTORY OF BIOLOGICAL CHANGE









CHAPTER 2

LIVING RESOURCES: A HISTORY OF BIOLOGICAL CHANGE

SECTION 1
INTRODUCTION

HOW DO WE ASSESS IMPORTANT BIOLOGICAL CHANGES?

Chesapeake Bay supports a diversity of life, from microscopic plants to fish, birds, and mammals. The well-being of the Bay's biota depends on the physical and chemical processes that take place within it (Chapter 3). The objective of this chapter is to describe the health of Chesapeake Bay and to identify biological changes that may have resulted from water quality changes.

A temperate, natural system is normally fairly diverse, containing several trophic levels with many species at each level. Major change may be reflected when species decline dramatically or become overly abundant. A biological continuum may exist, ranging from a system in which many species are present in moderate abundance, to a degraded system in which a few pollution-tolerant species persist. Biological changes also occur in response to factors other than water quality (Chapter 3).

In this analysis, biological change is assessed through historical analysis of trends in species distribution, abundance, and harvest. The present condition in the Bay system was spatially compared, assuming that physically and chemically similar areas can be expected to be biologically similar. Where major differences exist, it is inferred that biological change has occurred

This chapter discusses phytoplankton, submerged aquatic vegetation (SAV), benthic organisms, shellfish, and commercial finfish because the historical and/or spatial data were sufficient to attempt an assessment. In addition, these resources are of ecological and/or economic importance. In an effort to define regional and local differences, results are discussed in terms of two geographical divisions: basins (Figure 22) and segments.

INDICATORS OF CHANGE

Phytoplankton form the major base of the estuarine food web. Changes in abundance, diversity, and species composition of phytoplankton may cause turbidity, toxicity, odor, and unaesthetic floating mats. Other direct effects include shading of submerged grasses and loss of food (through shifts to unpalatable phytoplankton species) for zooplankton through fish. Excessive phytoplankton biomass can lead to loss of dissolved oxygen from bottom waters. Historical changes in phytoplankton species composition, diversity, and biomass, as documented for upper Chesapeake Bay (above the Bay Bridge) and upper Potomac River (tidal-fresh reach), are discussed.

Submerged aquatic vegetation is a major component of the detrital food web. These Bay grasses provide habitat for many crabs, fish, and their food. This chapter focuses on historical changes in the distribution and abundance of SAV and compares the present condition of beds across geographical areas.

Benthic invertebrates may indicate environmental change because they may be exposed to low dissolved oxygen and toxic materials. A spatial comparison of the present species composition and diversity in Baltimore Harbor is presented here.

The shellfishery is a major economic and cultural element of the Chesapeake Bay region. Historical changes in commercial landings of

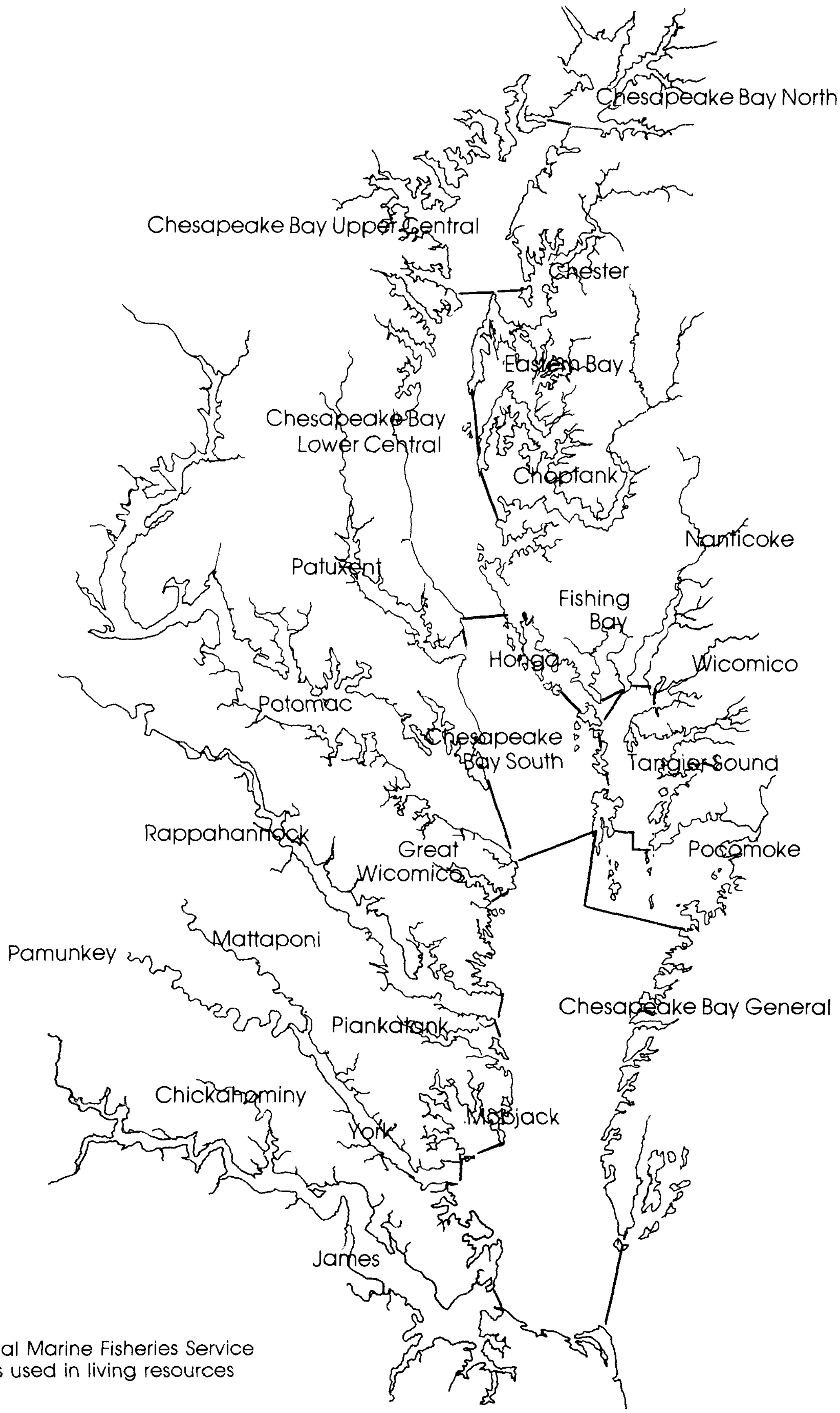


FIGURE 22.
NOAA National Marine Fisheries Service (NMFS) basins used in living resources analysis.

oysters and crabs, as well as oyster spat set (the abundance of juvenile oysters), are described. As immobile bottom-dwellers, oysters are good indicators of some forms of environmental stress.

Commercial finfish are of tremendous economic importance; landings are the most spatially and temporally complete biological data available. Landings are not an unequivocal indicator of stocks; however, the conclusions drawn from the analysis of landings data are supported with basic biological data, the finfish juvenile index. Attempts to adjust landings data for fishing effort would be of little value because of the unavailability of acceptable effort data (Rothschild et al. 1981).

MAJOR BIOLOGICAL COMPONENTS AND THEIR INTERACTIONS

To interpret biological changes, the species involved should be understood as well as possible. Important information includes environmental tolerances, life cycles, and interactions among species.

In the analysis of finfisheries, data existed for 13 species taken from 18 major basins. Thus the data were aggregated. Because its life cycle helps determine the likelihood that a species will be exposed to environmental stress, the CBP considered grouping species on the basis of their spawning habits. This suggestion was supported by similar trends in harvest and juvenile index data within such aggregations (discussed in Section 6). Thus, finfish are considered in two groups: marine spawners and freshwater spawners.

During the spring, freshwater spawners (alewife, catfish, shad, striped bass, white perch, and yellow perch) release their eggs in the fluvial or tidal-fresh reaches of the Bay system. Such reaches tend to be the first exposed to toxic chemicals and suspended sediments from the rivers feeding into the Bay. Some of these areas have also been subjected to extensive physical changes such as construction of dams and deposition of sediment.

Larvae and juveniles of some marine spawners enter the Bay during the late spring and summer to reach their upstream nursery areas; others enter in late fall through the winter (e.g., spot and croaker). Anoxia of bottom waters during this period can restrict their habitat and food availability, particularly for spot and croaker, which tend to feed on the bottom. Oysters and crabs are also potentially subject to hypoxic waters. The oyster habitat is detailed in Appendix C, Section 2.

One of the aspects that determines a species' ability to survive and reproduce in its environment is its environmental tolerances, or "survival envelope." When these tolerances are exceeded by natural or man-induced stress, the species will be reduced or eliminated.

When organisms are exposed to non-optimal conditions for some factors, they may be more vulnerable to other stress. For example, the common sand shrimp (*Crangon* sp.) has a broader tolerance for salinity and temperature at saturated dissolved oxygen levels than at low (2 to 3 ppm) dissolved oxygen levels (Haefner 1970). Thus, variations in water quality can affect an organism's sensitivity to other forms of stress.

One life stage may be more sensitive to stress than another. For most species, the period of reproduction (spawning, hatching, and early larval development) is most critical. For example, the salinity tolerance of striped bass eggs ranges from 0 ppt to about 10 ppt, but adults can tolerate salinity gradients from freshwater to ocean water (0 to 35 ppt) (Setzler et al. 1980). In general, larvae are considered to be more sensitive than other life stages of fishes (Sindermann et al. 1982).

Once its environmental tolerances are understood, a species' life history must be known so that the likelihood that sensitive stages will be exposed to stressful conditions can be assessed. Life cycles of major species are considered in detail in Appendix C, Section 1.

Water quality may affect organisms directly through physiological effects, or indirectly through species interactions. The well-being of one species can affect that of another through feeding relationships, competition for food or habitat, and in other ways. The feeding relationships that support fish in Chesapeake Bay are shown in Figure 23; they are described in greater detail in Appendix C, Section 1. Figure 23 graphically shows that the larger phytoplankton (net phytoplankton) directly provide food for several adult fish species and for zooplankton. The direction of the arrows shows the pathways of food in

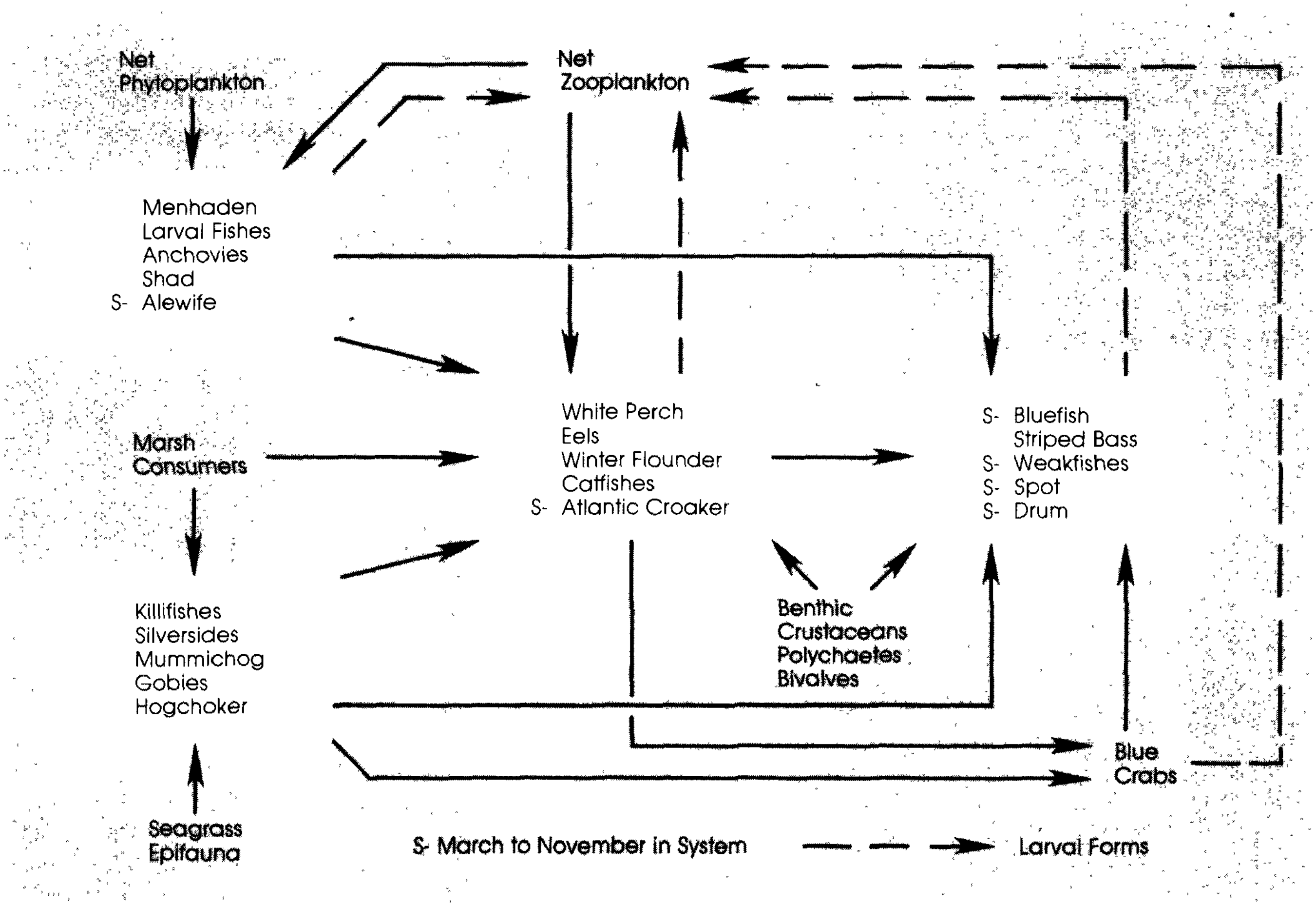


FIGURE 23. Feeding relationships of fish in Chesapeake Bay (after Green 1978). [Seasons(s) as specified in this chart are incorrect for spot which occur year-round. March to November also applies to shad.]

the food web for the other components.

Of basic importance to the estuarine food web are the primary producers. These are the plant forms, able to use light as an energy source. All other organisms in the Bay depend directly or indirectly on plants for food. In Chesapeake Bay, the most important primary producers are phytoplankton, followed by Bay grasses. Marshes and other wetlands also contribute to estuarine food webs, mostly through detrital inputs.

Many benthic species (oyster, clam, etc.) feed on living and detrital phytoplankton. The species of phytoplankton is important, as this determines whether it can be used for food. Ryther (1954) describes the classic example of Moriches Bay, New York, in which changes in phytoplankton species composition led to the demise of a once-prosperous oyster industry. The change in phytoplankton species resulted from increased nutrient inputs by duck farms.

Submerged aquatic vegetation provides a direct and indirect food source as well as habitat for many estuarine animals. It may supply much of the diet of sheepshead (*Archosargus probatocephalus*) as well as turtles; as a major source of detritus, SAV provides food for a variety of filter-feeders (Stevenson and Confer 1978). Submerged aquatic vegetation affords substrate for organisms like mussels, barnacles, and molluscs (Green 1978). Fish, including Atlantic silversides

(*Menidia menidia*), feed on these epifauna. The blue crab depends on SAV beds to furnish shelter from predators (Darnell 1959).

With the exception of the Atlantic menhaden that feeds on phytoplankton as an adult, most of the fishes treated in this study, as well as crabs, tend to be carnivorous and are opportunistic feeders. Changes in one food source may not affect them significantly because they can switch to another source. However, some species may be more limited; sciaenids, such as spot and croaker, depending heavily on benthic organisms for much of their food.

Understanding predator-prey relationships suggests a second major species interaction: competition for food. It could be hypothesized that species with similar food habits are potential competitors; if food becomes limiting, one species may decline relative to the other. Also, one species may depress populations of the other species (May et al. 1979).

SECTION 2
CHANGES IN BIOMASS AND SPECIES COMPOSITION OF PHYTOPLANKTON IN TWO WELL-DOCUMENTED AREAS

INDICATIONS OF ENRICHMENT

Because most phytoplankton use inorganic nutrient elements directly, changes in abundance and species composition of these primary producers may be the first indications of excessive nutrient enrichment in a system. In this section, two areas, the upper Bay (above the Bay Bridge) and upper Potomac River (tidal-fresh reach), in which increased dominance by single species and increased biomass have paralleled increased nutrient enrichment, will be discussed. These are the areas for which the most comprehensive data are available; similar changes may be occurring elsewhere, or could occur if nutrient enrichment increases.

Excessive nutrient enrichment, a corollary of eutrophication, frequently produces increased biomass of algae as its first observable effect (Heinle et al. 1979). Excess nutrients may lead to blue-green algal blooms in the tidal-fresh portions of estuaries (Darnell and Soniat 1981). In the early stages, this may be beneficial to harvestable resources. The increased algal biomass (indicated by high cell numbers and high chlorophyll *a* concentrations), however, may not be used by grazers, either because the phytoplankton growth rate is too rapid or because the forms produced are inedible (Darnell and Soniat 1981). As a result, excess organic matter accumulates in the water column. Subsequent decay of this accumulated organic matter depletes dissolved oxygen from the water column.

Excellent relationships have been demonstrated between phosphorus loads and chlorophyll *a* concentrations in lakes (Schindler 1981). Estuaries are more complex and less well-studied, but Lee and Jones (1981) have shown a relationship between nutrient loads and chlorophyll *a* concentrations in the Potomac River.

In addition to increased algal biomass, shifts in species may occur in response to nutrient enrichment. Blue-green algae are particularly successful under enriched conditions because some species can fix atmospheric nitrogen, or may produce toxins detrimental to other algae (Lane and Levins 1977). The tidal-fresh portion of the Potomac estuary is the best known example of blue-green algal blooms causing water quality problems in the Chesapeake Bay system (Clark et al. 1980). Enrichment of higher salinity reaches may result in blooms of dinoflagellates (Darnell and Soniat 1981).

ENRICHMENT IN CHESAPEAKE BAY

The Potomac River has demonstrated changes in rooted vegetation and phytoplankton that parallel increases in nutrient enrichment. Prior to 1920 the river was subject to no major plant nuisances (Pheiffer et al. 1972, Champ et al. 1981, Haramis and Carter 1983). In the 1920's, the river experienced an invasion of water chestnut (a rooted aquatic plant), which has been linked to overenrichment from wastewater discharges (Champ et al. 1981, Haramis and Carter 1983). Another rooted plant, Eurasian water milfoil, began to replace water chestnut in the late 1940's and became a major plant problem. In the 1960's, milfoil was in turn supplanted by massive blue-green algae blooms, causing widespread nuisance conditions (Haramis and Carter 1983). These algal blooms persisted into the 1970's though the amount of organic carbon from wastewater discharges had been reduced by almost 50 percent (Pheiffer et al. 1972). In September 1978, under maximum bloom conditions, the blue-greens amounted to 80 percent of the phytoplankton population with total cell counts ranging from 60 to 80 million cells per liter (Clark et al. 1980) (chlorophyll *a* values during this period approached 140 ug L^{-1}).

Some researchers believe that water quality in the upper Potomac River has improved because of recent phosphorus reductions by extensive removal from wastewater discharges starting in 1970 (Champ et al. 1981). The 1979 algal populations were mixed. Phytoplankton composition under maximum bloom conditions shifted from dominance by large numbers of filamentous blue-green algae to smaller numbers of single-celled blue greens, which made up only about 25 percent of the summer population. Phytoplankton counts for 1979 and 1980 were lower than in previous years,[4] perhaps caused partly by high-flow conditions that have characterized the river in the past few years. Such conditions tend to flush out phytoplankton.

The Potomac River is surpassed only by the Susquehanna River in its contribution of nutrient load to the Bay (Smullen et al. 1982). Trends in nutrient enrichment of the upper Bay tributaries have closely paralleled those of the Potomac, and a similar process is probably underway in the upper main Bay. Visual observations of massive algal blooms are frequently noted. The elevated chlorophyll levels recorded in this area in the early 1970's were due in part to increasing occurrences of blue-green algae (Clark et al. 1973). Earlier observations of water from this area by Flemer (unpublished) revealed only occassional occurrences of blue-green algae.

In 1979, a study by the Maryland Power Plant Siting Program showed cell counts at stations located in segment CB-2, ranging from 10 million to 27 million cells per liter during the month of June (Grant et al. 1979). In 1980, phytoplankton sampling was conducted by the State of Maryland, Water Resources Administration (Allison 1980). Total cell counts in the upper Bay ranged from 18.5 to 21.2 million cells per liter in June. Chlorophyll *a* values in 1980 approached 114 ug L^{-1} in CB-2 (Chapter 1).

From October 1955 to October 1956, a phytoplankton study (Whaley and Taylor 1968) was conducted on the Bay by continual underway sampling. The June cell counts in CB-2 ranged from 2200 to 12,900 cells per liter. However, Whaley and Taylor, who used net samples, probably missed smaller species caught in Allison's whole water samples. Assuming that nannoplankton compose between 75 and 93 percent of the phytoplankton population (McCarthy et al. 1974, VanValkenburg and Flemer 1974), Whaley and Taylor's numbers could be elevated to as much as 86,000 cells per liter. Allison's 21.2 million cells per liter represents an increase of 247 fold over the 86,000 cells per liter. This increase is probably attributable to an increase in nutrient supply in the upper Bay. With increasing nutrient enrichment in the segments of the upper Bay, we can expect to see the total cell counts elevated with a movement toward blooms of nuisance algal forms like those documented in the Potomac River.

Much of the information on the upper Bay cited above is limited and should be supported by a more concerted effort. It is recommended that intense monitoring of the upper Bay for nutrient inputs and actual phytoplankton species identification and enumeration be done. This would provide concrete evidence of the species shifts believed to be taking place.

SECTION 3
THE DECLINE OF SUBMERGED AQUATIC VEGETATION

The Chesapeake Bay's other major primary producer, submerged aquatic vegetation, is an important ecological resource. About 10 species of submerged vascular plants are collectively called submerged aquatic vegetation (Table 18). Submerged aquatic vegetation harbors food, provides habitat to major fish species, and is a source of food for waterfowl and aquatic mammals. The grasses have undergone a precipitous decline in the past 10 to 15 years. In this section, the time course of that decline and its geographic nature is discussed. Finally, a geographic assessment of the present use of its habitat by SAV with respect to its potential is shown. Data contributing to this section are given in Appendix C, Section 5.

Submerged aquatic vegetation was the subject of a major Chesapeake Bay Program research effort (Orth and Moore 1982). Since the late 1960's a dramatic, Bay-wide decline has occurred in the abundance of submerged aquatic vegetation, and its distribution has changed. Although the rates and patterns of decline have not been uniform throughout the estuary, some patterns do appear. Loss has moved progressively down-estuary, paralleling nutrient enrichment trends. Submerged aquatic vegetation now occupies a significantly more restricted area than previously. As a consequence, the scope of its role in Bay ecosystem processes has also been reduced, and its ability to recover from its current status without an improvement in ambient water quality is questionable.

Submerged aquatic vegetation is restricted to shallow areas of Chesapeake Bay because of its requirements for light. Reduction in light availability has been implicated as a significant source of stress to SAV. For example, increased nutrients leading to increased phytoplankton growth and greater production of leaf-surface epiphytes have been shown in pond studies to decrease SAV bio-

TABLE 18.
MAJOR SPECIES OF SUBMERGED VASCULAR PLANTS (SAV) IN CHESAPEAKE BAY

Ceratophyllum demersum	Coontail
Elodea canadensis	Elodea
Myriophyllum spicatum	Eurasian Watermilfoil
Najas guadalupensis	Water Nymph
Potamogeton pectinatus	Sago Pondweed
Potamogeton perfoliatus	Redheadgrass
Ruppia maritima	Widgeongrass
Vallisneria americana	Wildcelery
Zannichellia palustris	Horned Pondweed
Zostera marina	Eelgrass

mass (Kemp et al. 1982b). Kemp et al. (1982b) have reviewed other microcosm, pond, and field data that implicate nutrients as a major cause of SAV decline. Microcosm studies have shown that turbidity (total suspended solids) causes decreased photosynthesis (Kemp et al. 1982c). Herbicides can inhibit SAV photosynthesis but, in most parts of Chesapeake Bay, plants are not likely to be exposed to concentrations from which they could not recover (Kemp et al. 1982b).

Work on submerged aquatic vegetation in the Potomac River, by the U.S. Geological Survey (USGS), also concluded that excessive nutrients are the primary cause of SAV declines in the tidal river (storms and other extreme events are also important factors) (Haramis and Carter 1983). However, evidence from transplant and caging experiments, in which grazers are kept in with the plants (results still in preparation), indicates that grazing may have significant effects on the ability of plants to recolonize former habitat. The survival of SAV populations in the estuarine-riverine transition zone may be linked to adverse effects of this unstable environment on potential biological competitors or predators (e.g., periphyton, phytoplankton, and grazers) (Haramis and Carter 1983).

HISTORICAL CHANGES

Historically, SAV has been abundant throughout the estuary. Biostratigraphic analysis of selected sites shows continuous presence of the grasses over several centuries. However, changes in species composition were apparent from 1930 to 1965 (Orth and Moore 1982). From 1958 to 1962, there was a dramatic proliferation of the exotic species watermilfoil (*Myriophyllum spicatum*) in the upper Bay (Bayley et al. 1978). A similar pattern occurred in Lake Mendota, Wisconsin, where *Myriophyllum* invasion occurred in response to eutrophication (Lind and Cottam 1969). In Chesapeake Bay, the milfoil populations declined after 1962, probably because of pathogens, and the native vegetation returned (Bayley et al. 1978). However, plants were present in less abundance than before the milfoil invasion, and species composition was different. This pattern may have resulted from increased turbidity and nutrients in the upper Bay (Bayley et al. 1978).

Progressive and general decline in SAV abundance has characterized the period from 1965 to the present (Orth and Moore 1982, Figure 24). This decline was first observed in the upper Bay and in fresher reaches of tributaries. The lower Bay first showed effects in the early 1970's.

Statistically significant declines in SAV coverage have been demonstrated in a survey of SAV at approximately 650 stations conducted annually by the Maryland Department of Natural Resources and the United States Fish and Wildlife Service (USFWS) Migratory Bird and Habitat Research Laboratory. At least 50 stations occurred in each Chesapeake Bay Program segment analyzed (Appendix C, Section 5). In 1971, 28.5 percent of the stations were vegetated; coverage declined to about 10.5 percent by 1973 and dropped to 5 percent in 1981 (Figure 25). In 1971, 10 Bay segments reported no vegetated stations, but in 1981, 14 segments had none. Statistically significant declines in SAV coverage occurred since 1971 within seven of the vegetated Maryland segments (CB-5, EE-1, EE-3, ET-5, ET-8, ET-9, WT-7), as well as in the total surveyed area (Appendix C, Section 5). Other segments showed no clear trends, or had so little remaining vegetation in 1971 that significant declines after that date could not be demonstrated.

Preliminary results from the 1982 summer survey indicate a continued downward trend (James 1982, Macknovitz 1982). In 1982, only 4.5 percent of the 646 stations visited had rooted vegetation. On the Eastern Shore, Smith Island had the highest percentage of SAV (41.2 percent vegetated stations). Vegetation in the Chester River (ET-4) had dropped to 0.0 percent and to 4.3 percent in Eastern Bay. Vegetation increased slightly to 6.7 percent in the Choptank River. On the western shore, the Susquehanna Flats showed a strong increase (to 13.5 percent; however, milfoil was the only species recorded). Submerged aquatic vegetation in the Magothy and Severn Rivers declined to 0.0 percent in 1982. In general, SAV trends continue downward, although gains occurred in the Susquehanna Flats and Smith Island areas.

Species diversity also declined significantly in eight segments (CB-5, EE-1, EE-2, EE-3, ET-5,

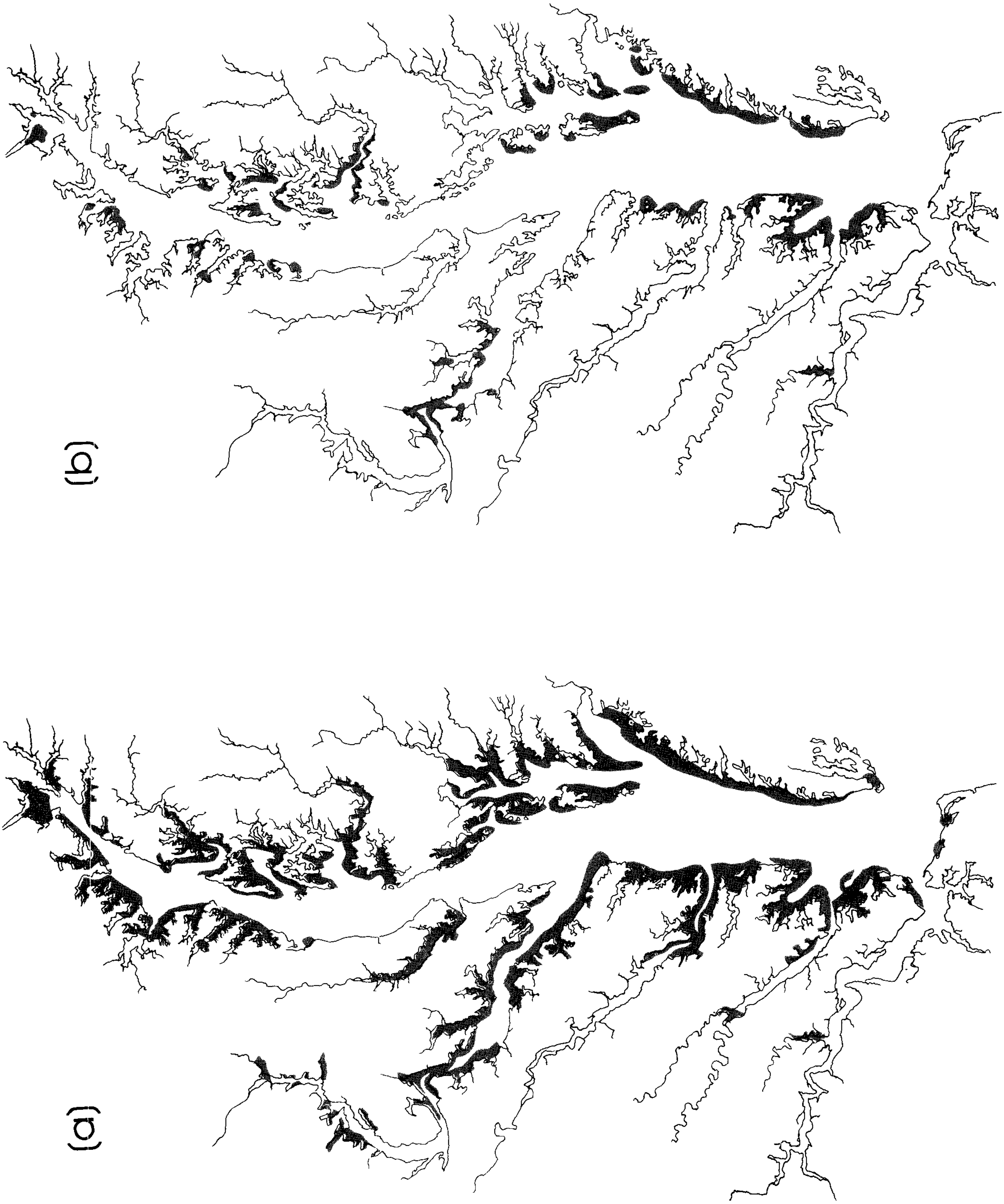


FIGURE 24. General area of SAV distribution in (a) 1965 and (b) 1980.

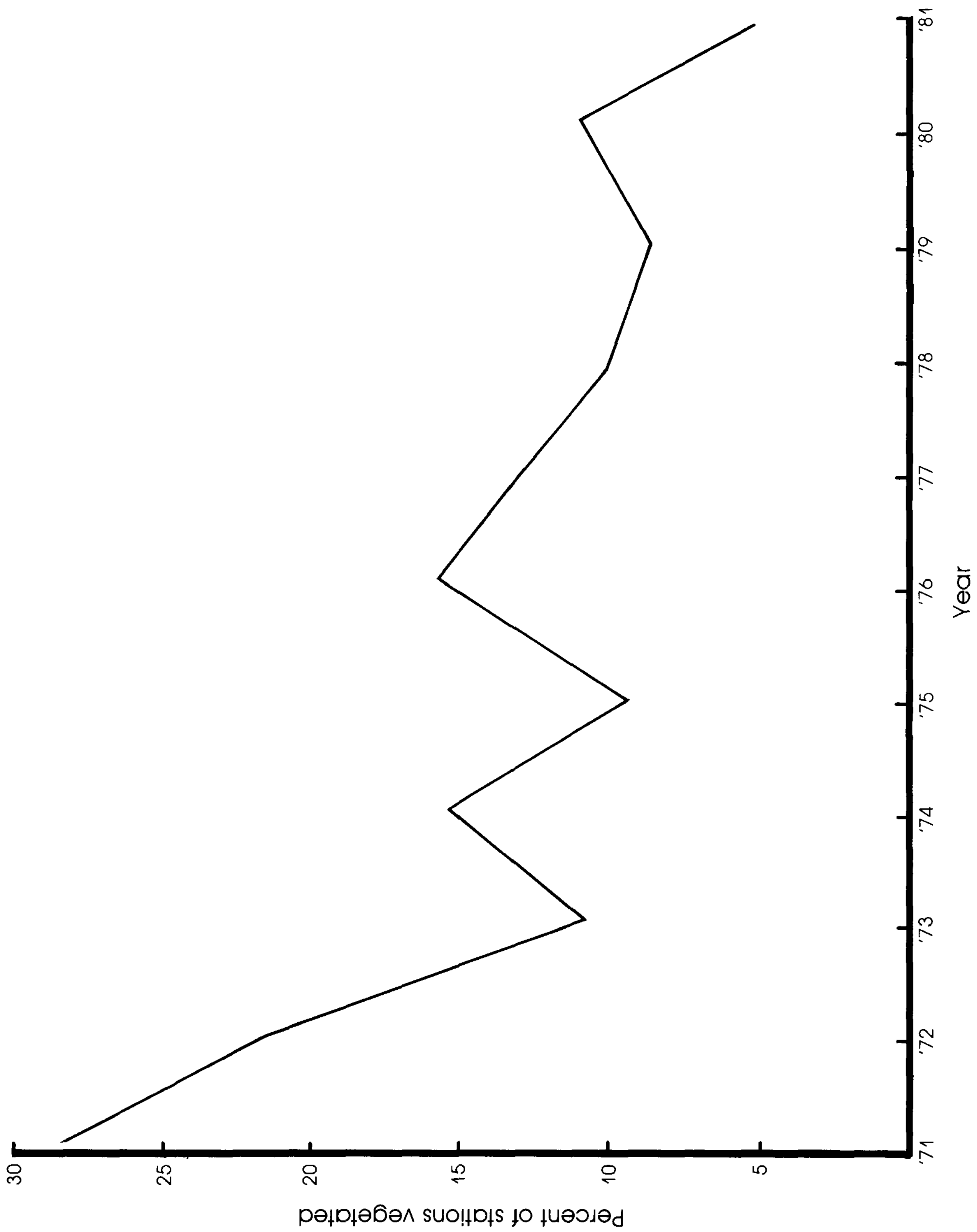


FIGURE 25. Percent of submerged aquatic vegetation sampling stations in Maryland with vegetation, 1971 to 1981 (data from the U.S. Fish and Wildlife Service and Maryland Department of Natural Resources).

ET-9, WT-6, WT-7) and in the total surveyed area (Appendix C, Section 5). Remaining SAV populations are primarily characteristic of the higher salinity forms. Relatively little SAV characteristic of tidal-fresh and oligohaline waters remain; these populations are chiefly confined to small tributary creeks and the headwaters of some major rivers.

ASSESSMENT OF PRESENT CONDITION IN CHESAPEAKE BAY SEGMENTS

To estimate the present ability of Chesapeake Bay segments to support submerged aquatic vegetation, the 1978 distribution (the most recent year for which Bay-wide data are available) was compared with the maximum distribution to be expected. The expected distribution is based on the potential habitat (Shea et al. 1980), the area for which substrate, salinity and depth meet the requirements for various species of SAV. Because of the requirement for light, submerged aquatic vegetation is limited in its distribution to a depth of 2 to 3 meters in Chesapeake Bay (Stevenson and Confer 1978). To be conservative, two meters was selected as the boundary of the SAV potential habitat for this Bay-wide analysis. The area of potential habitat was determined by planimetering the two-meter contour area for each sampling area. The resulting potential habitats are listed in Appendix C, Section 5.

Because factors other than depth affect the distribution of SAV, the potential habitat will rarely be entirely occupied. To estimate the proportion of potential habitat that could be expected to be occupied by SAV, two approaches were taken: case studies and ground truth surveys.

Aerial photographs of the Poplar Island and Tilghman Island areas in 1971 were obtained from the United States Fish and Wildlife Service. SAV had not begun to decline in these areas in 1971. The two-meter contour and SAV distribution were planimetered based on the aerial photographs; SAV was found to occupy 25 to 30 percent of the potential habitat.

The USFWS – Migratory Bird and Habitat Research Laboratory (MBHRL) has also surveyed areas in Maryland that fall within the potential habitat range since 1971. The percent of sites vegetated ranged from 7 to 83 (Appendix C, Section 5).

As a result of these approaches, 50 percent was chosen as the proportion of potential habitat that could be expected to be vegetated to reflect the theoretical range of 25 to 83 percent. The purpose of establishing this expected habitat is not to quantify SAV decline, but to formulate a theoretical baseline for geographically comparing Chesapeake Bay segments. The area of expected habitat is shown for each sampling area in Appendix C, Section 5.

The percentage of expected habitat filled in 1978 was determined for each sampling area; the areas were then ranked from 6 to 1 (Appendix C, Section 5). The ranking scheme (6 = 0 to 2.5 percent; 5 = 2.6 to 6.3 percent; 4 = 6.4 to 15.8 percent; 3 = 15.9 to 39.8 percent; 2 = 39.9 to 75 percent; 1 = 76 to 100 percent) was chosen to provide greater resolution at the lower end of the range and reflects the logarithmic frequency distribution of values. The range of 6 to 1 was chosen to be consistent with the nutrients scheme (Chapter 1), in which a 1 rank indicates a better condition than presently exists.

The ranks of Chesapeake Bay segments are shown in Appendix C, Section 5, and Figure 26. To corroborate the ranking of segments using the expected habitat approach, the USFWS/MBHRL data for Maryland were assessed (Appendix C, Section 5). The maximum of sites vegetated between 1971 and 1981 was used as a baseline; the 1978 percent of vegetated sites was divided by the maximum and ranked according to the scheme discussed previously. There is good agreement for the Maryland segments.

According to this analysis, SAV now occupies about 24 percent of its expected habitat for the Bay as a whole; 19 percent in Maryland and 37 percent in Virginia.

WETLANDS

Submerged aquatic vegetation is one of many types of wetland found in the Chesapeake Bay area (McCormick and Somes 1982). Other types, also of ecological importance, include fresh and brackish marshes, saline high and low marsh, wooded and shrub swamps, and transition zones

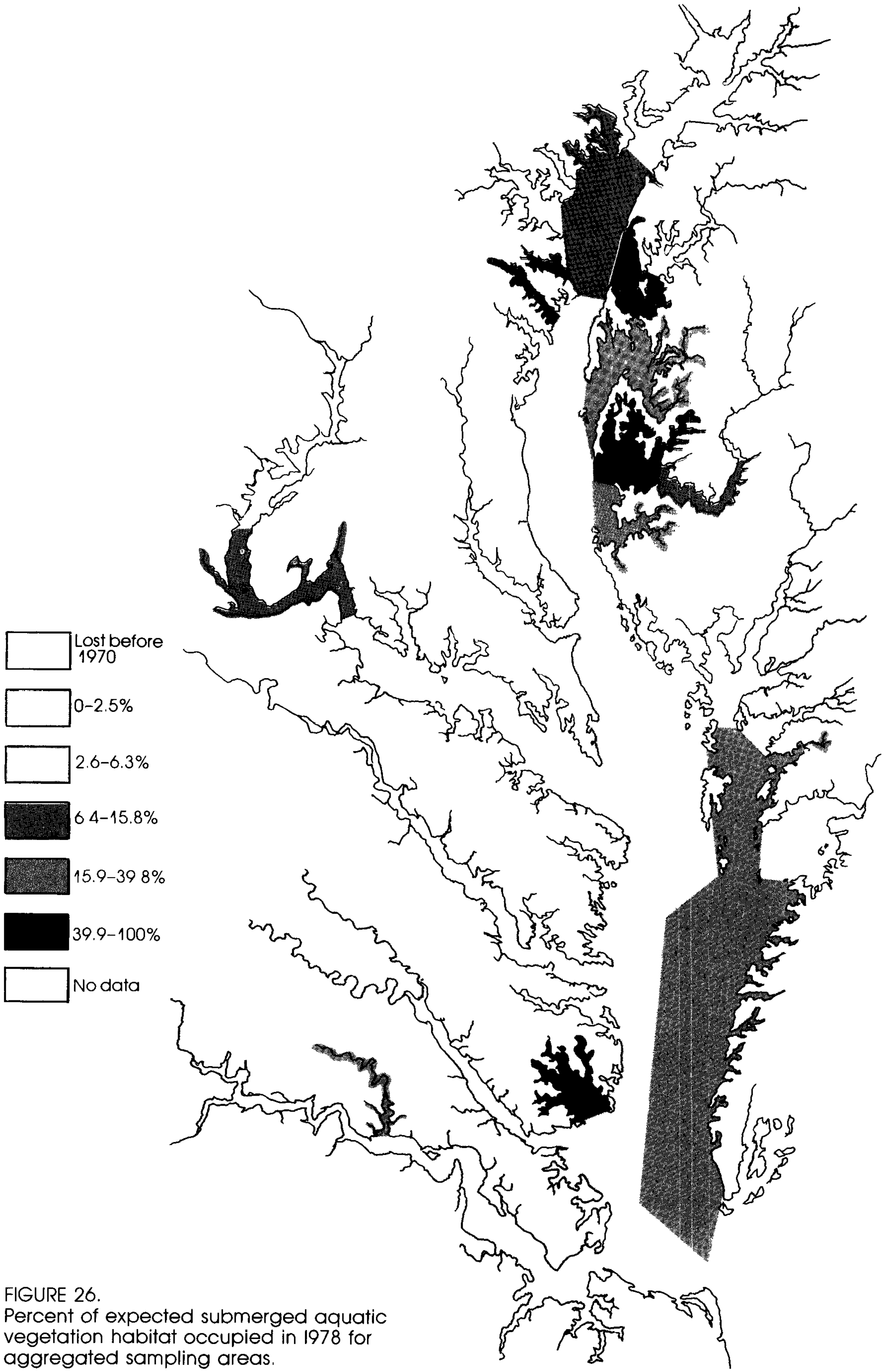


FIGURE 26.
Percent of expected submerged aquatic vegetation habitat occupied in 1978 for aggregated sampling areas.

between these. Wetlands, both headwater and tidal, are important to the Chesapeake Bay ecosystem for a variety of reasons. They are shelter, feeding, and breeding areas for mammals, waterfowl, finfish, and shellfish. In fact, Daiber et al. (1976) estimate that 80 to 90 percent of the total Chesapeake Bay seafood harvest depends at some stage on wetlands. One major function is the contribution of detrital material into food webs that support major finfish larval feeding patterns: an example is the relationship between inputs of ice-scoured detritus from freshwater marshes into zooplankton food webs and the spawning success of striped bass (Heinle et al. 1976). Marshes and other wetlands thus serve as transformers or processors of nutrients and as seasonal sinks, releasing in winter nutrients incorporated during the growing season.

Vegetated wetlands are also important as buffers to the aquatic environment. Nutrients from runoff or other sources may be removed by the wetland through denitrification, precipitation, sorption on organic matter, and vegetative assimilation (Van der Volk et al. 1979). Differences among the nutrient removal efficiencies of wetlands appear primarily related to differences in hydrology ((Van der Volk et al. 1979, Nichols 1983). Wetlands may often intercept considerable quantities of sediment from runoff, as well as protect vulnerable shoreline areas from erosion (Allen 1979, Novitzki 1979). Wetlands can absorb and hold considerable quantities of water; it is estimated by Kusler (1977) that one acre of flooded marsh may absorb over 300,000 gallons of water. In this manner, wetlands contribute to flood control through water storage and slowing, as well as to groundwater recharge (Davis 1978). Pollutants such as heavy metals, pesticides, and fecal contaminants may also be absorped and trapped in wetlands. Local conditions within the wetland largely determine the level of removal of these pollutants (Gallagher and Kibby 1980).

Despite the ecological importance of marshes and other wetlands, losses of these areas have occurred throughout the Chesapeake Bay watershed. It was estimated by Metzgar (1973) that some 23,700 acres of wetlands were lost in the state of Maryland between 1942 and 1969, about 7 percent of the total wetlands in the state at the beginning of the period. In 1976, all coastal (tidal) wetlands in Maryland were mapped; at that time 261,000 acres were identified (McCormick and Somes 1982). In Virginia, approximately 4000 acres were lost between 1955 and 1969, out of a total area of 220,000 acres of tidal marsh (of which 90,000 acres are salt marsh) (Settle 1969). In Maryland, most losses (52 percent) were due to agricultural drainage, 13 percent to housing development, six percent to industrial development, four percent to marinas, and five percent to dredging and spoil activities (Metzgar 1973). Erosion, natural succession, and drainage for mosquito control accounted for much of the remainder. Overall losses are probably much greater. The Report of the Maryland Conservation Commission for 1908 to 1909 states that (at that time) wetlands within Maryland totaled some 500,000 acres, of which 204,400 acres were salt marsh and the remainder fresh marsh and swamp. This represented one-twelfth of the total area of the state. Losses in Virginia were estimated by Settle (1969) to be due to channelization (purposes not specified) (47 percent), residential development (27 percent), and 17 percent to industrial projects. The magnitude of loss of non-tidal wetlands has not been quantified for either state; an ongoing National Wetlands Inventory may allow estimates of this acreage.[5]

A growing appreciation of the role and value of wetlands led to passage of wetlands protective legislation in Maryland (1970) and Virginia (1972). These acts were designed to protect tidal wetlands; non-tidal wetlands are still vulnerable. In recent years, the rate of wetlands loss has slowed to approximately 20 to 25 acres per year per state of vegetated tidal wetlands.[6]

SECTION 4
SPATIAL TRENDS IN BENTHIC ORGANISMS

Benthic animals are useful indicators of pollution because most are relatively immobile and cannot readily escape unfavorable conditions (Boesch 1973, Pfitzenmeyer 1975). Changes in benthic biomass, community structure, and diversity can indicate a variety of stressful conditions (Boesch 1977a). Stresses may be natural (e.g., salinity, temperature, and predation) or anthropogenic (e.g., nutrient enrichment, toxicant contamination). For example, increases in nutrient loading may alter sediment organic content to favor dominance by detritivores (Bascom 1982). Toxic pollutants may eliminate sensitive species, producing benthic communities dominated by resistant, opportunistic forms (Grassle and Grassle 1974, Dauer et al. 1979). Reduced dissolved oxygen in overlying waters can severely limit benthic fauna; where anoxic episodes are frequent, benthic communities may become ephemeral or be eliminated completely (Holland et al. 1979, Mountford et al. 1977). Toxic materials may be concentrated in tissues of benthic organisms, potentially affecting the organism's physiological processes (Frazier 1976) or even the suitability of shellfish for human consumption.

Historical data on benthic organisms, that is, data showing trends over some period of time, are not available for many areas of Chesapeake Bay. Although most major regions of the Bay have had benthic surveys performed within the last two decades, time history information exists only for such regions as Calvert Cliffs, and the Patuxent, Potomac, lower York and lower James Rivers. Current conditions are better documented. Under auspices of the EPA's Chesapeake Bay Program, present conditions were investigated for the Maryland portion of the Bay mainstem (Reinharz and O'Connell 1981), Virginia mainstem (Nilsen et al. 1981), Patapsco River (Reinharz 1981), and Elizabeth River (Schaffner and Diaz 1982). These data were examined for trends in species diversity and community structure. The ratio of annelids to crustacea and annelids to molluscs was calculated for each station; previous studies have shown dominance by polychaetes and oligochaetes in stressed environments (e.g., Pfitzenmeyer 1975, Dauer et al. 1979) (Figure 27). Species diversity (the Shannon diversity index) was also compared for evidence of spatial trends.

MAIN BAY

Diversity, annelid:crustacean, and annelid:mollusc ratios were compared over the Bay mainstem. Stations having less than 25 percent sand were used to minimize the effect of substrate. (Sand substrates support benthic communities very different from those on fine grained sediments. Also, sand does not accumulate toxicants to any great extent.) The range in values was large, reflecting the natural variability in benthic distribution (Reinharz and O'Connell 1981). Species diversity was higher in more saline stations, reflecting the general distribution of benthic organisms along a salinity gradient (Boesch 1977a). Annelid:mollusc ratios were more variable in the upper Bay (mean = 40.0, range 1 to 242) than in the Virginia stations (mean = 6.3, range 0.8 to 58) (Table 19). The pattern for the annelid:crustacean ratio is similar: upper Bay mean = 73.7, range 0.01 to 598; lower Bay mean = 35.5, range 0.3 to 242. These trends are

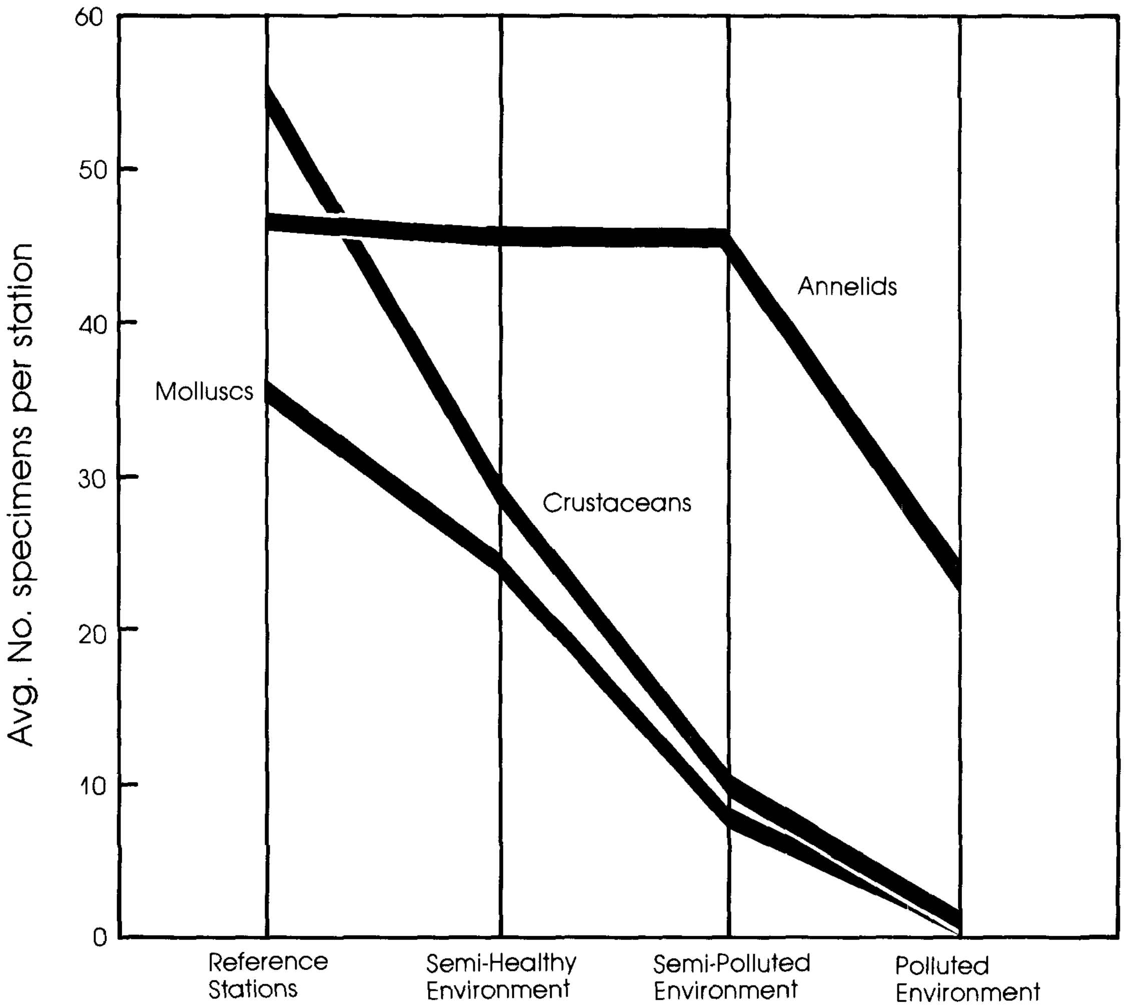


FIGURE 27. Number of specimens of benthos in Chester River reference stations and in three environmental zones of Baltimore Harbor (station represents 0.2m² area) (Pfitzenmeyer 1975).

TABLE 19.
RANGE OF ANNELID:MOLLUSC VALUES FOR CHESAPEAKE BAY SEGMENTS

Segment	Range of Values	Number of Stations
CB-1, 2	34	1
CB-3	4.5– 66.0	2
CB-4	1.6– 58	6
CB-5	1.0–242.0	7
CB-6	0.8– 8.9	3
CB-7	1.4– 16.0	4
EE-3	1.6– 58	5
LE-3	3.8	1
WE-4	6.1– 57.0	2

not significant, although they do reflect a greater tendency for dominance of upper Bay communities by polychaetes.

TRIBUTARIES

The Patapsco and Elizabeth Rivers are heavily affected by human activities (Reinharz 1981, Schaffner and Diaz 1982). The Rhode and Ware Rivers, relatively "unpolluted" reference estuaries, were selected for comparison. In addition, Patapsco River benthos was surveyed by Pfitzenmeyer (1975) and compared to that of the Chester River. Water and sediment quality and resources in the Elizabeth River were characterized by the Virginia State Water Control Board in 1982.

In the Patapsco and Elizabeth Rivers, the effects of pollution are reflected in benthic community changes, including loss of species richness and diversity, and enhancement of resistant, opportunistic forms such as annelids, relative to crustacea and molluscs. Highly impacted stations in the Elizabeth River were characterized by young individuals of a few resistant species, indicating low survival of recruits (Schaffner and Diaz 1982). These stations also had the lowest species richness (number of species) and diversity. In the Patapsco River (which showed the most distinct trends), diversity declines up estuary, generally along the gradient of increasing contamination of metals and organic chemicals (Bieri et al. 1982a) (Table 20, Figure 28). Similarly, dominance by annelids increased greatly at impacted stations, when compared to reference areas (Table 20). This situation will be discussed further in Chapter 3.

TABLE 20.
SPECIES DIVERSITY (H), REDUNDANCY (r), ANNELID:MOLLUSC, AND ANNELID:CRUSTACEAN RATIOS (AS NUMBERS OF INDIVIDUALS) ALONG A GRADIENT OF POLLUTION IN THE PATAPSCO RIVER (REINHARZ 1981)

Station*	H	r	Annelid: Mollusc	Annelid: Crustacean
P_0	0.330	0.864	23	—
P_1	0.561	0.831	15	—
P_3	0.343	0.906	51	253
P_4	0.590	0.783	11	1276
P_5	0.246	0.893	37	—
P_2	0.838	0.491	—	—
P_9	0.678	0.731	29	350
P_8	1.173	0.630	5	62
P_{10}	1.296	0.634	33	115
P_{11}	1.193	0.676	30	115
P_6	1.615	0.523	2	203
P_7	1.416	0.603	3	47
P_{13}	1.400	0.549	14	138
P_{12}	2.879	0.307	3	11
P_{14}	2.715	0.312	4	0.9

*See Figure 28 for station locations.

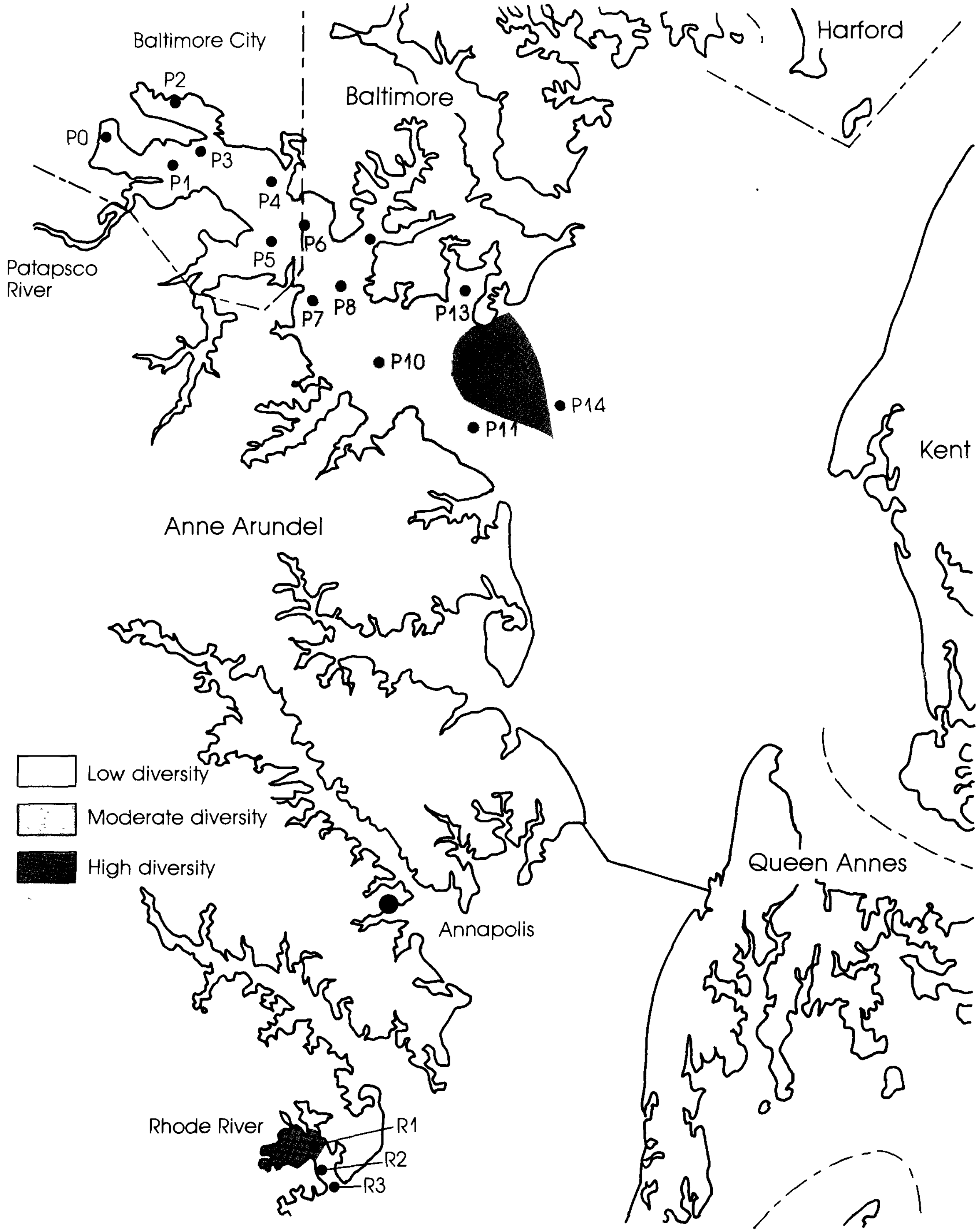


FIGURE 28. Diversity index of benthic communities in the Patapsco and Rhode Rivers (Reinharz 1981).

SECTION 5

THE SHELLFISHERY: CHANGES IN OYSTERS AND CRABS

The Chesapeake Bay shellfishery is an integral part of the economic and social qualities of the region. The American oyster, *Crassostrea virginica*, and blue crab, *Callinectes sapidus*, sustained harvests of 21,958,100 and 58,956,500 pounds, respectively, in 1980. (Other shellfish species, like the soft clam and hard clam, comprise much smaller harvests and will not be discussed here.) This section focuses first on oysters, describing their environmental requirements, harvest, spat set, and condition index. Then, the harvest of blue crabs will be summarized.

ENVIRONMENTAL REQUIREMENTS OF OYSTERS

Because they are sedentary organisms, oysters cannot avoid long-term environmental stresses in their surroundings (Appendix C, Section 1). Salinity and temperature affect their feeding rate and reproductive success. Oysters are physiologically restricted to areas ranging in salinity from 5 to 35 ppt. However, the diseases caused by *Minchinia (Haplosporidium) nelsoni* (MSX) and *Perkinsus marinus* (Dermo), as well as predators, reduce their abundance in areas of over 15 ppt salinity. Adult oysters can tolerate temperatures from 1 to 35C;[7] peak spawning occurs between 24 and 28C.

Another factor affecting oyster survival is suspended sediment which, upon settling, can smother adult oysters and prevent setting of spat (Galtsoff 1964). Excessive sedimentation in upper reaches of rivers has shifted the upstream limit of oysters downstream several miles (Alford 1968).

Low oxygen concentrations may also affect oyster populations; oyster larval growth ceases when oxygen levels drop below 2.4 mg L^{-1} (1.7 ml L^{-1}). Adults can survive up to five days in water with oxygen levels less than 1.0 mg L-1 (0.7 ml L-1)(Galtsoff 1964). Haven et al. (1978) believe that there may be some connection between the decline in spat fall and the oxygen deficiency observed in the deeper regions in the lower portions of the Rappahannock River.

Oysters are particularly sensitive to heavy metals. Calabrese et al. (1973) found that embryos are most sensitive to Hg, silver (Ag), Cu, and Zn, with LC_{50} concentrations of 0.0056, 0.0058, 0.103, and 0.31 ppm, respectively (Kennedy and Breisch 1981). Shuster and Pringle (1969) reported a strong sensitivity to Cd among adult oysters with 100 percent mortality occurring after a 20 week exposure to 0.2 ppm.

OYSTER HARVEST

The oyster industry in Chesapeake Bay is heavily managed, and harvest practices vary in Maryland and Virginia. Observed declines or apparant stability in oyster harvests may be partly related to economic factors and management practices. Oyster landings include those from public and privately leased areas.

Oyster harvest in Maryland comes largely from public oyster grounds; very few private leases exist. Maryland carries out extensive shell planting which may enhance spat set and, ultimately, harvests (Ulanowicz et al. 1980). In Virginia, oyster harvests come from public grounds and large areas of leased grounds. The leased grounds, which must be planted if they are to be productive, yield about 50 percent of the total production.[8]

Harvest data were obtained and analyzed as described in Appendix C, Section 2. Oyster landings data were obtained in a much different manner in the 1800's and early 1900's than at present.[9] As discussed in Appendix C, Section 3, methods changed in about 1930; as a result, data taken before 1930 are more suspect than recent data.

Harvest of oysters in Chesapeake Bay is shown in Figure 29. Annual yields of about 30,000,000 pounds of shucked oyster meat were sustained between 1930 and 1960; since 1960, annual yields have been about 20,000,000 pounds. Landings appear to have decreased dramatically between 1880 and 1930, but data from this period may be unreliable.

Harvest data for individual basins are available from 1962 to the present. To assess historical changes in harvest in Chesapeake Bay basins, we divided the data into two periods of approximately equal length, 1962 to 1970 and 1971 to 1980. Oyster harvest was recorded as pounds per acre of natural and privately leased bars (Appendix C, Section 2).

Annual mean oyster harvests for the two periods considered are shown in Table 21. Signifi-

TABLE 21.
OYSTER HARVEST FOR 1962 TO 1970 AND 1971 TO 1980, ANNUAL MEAN (POUNDS PER ACRE OF NATURAL OYSTER HABITAT)†

	1962-1970	1971-1980
Western Shore		
Patuxent River	67.6	84.5
Potomac River	98.0	44.5*
Rappahannock River	40.3	30.8*
York River	23.9	7.6*
James River	46.3	23.4*
Main Bay		
Chesapeake Bay		
North	—	—
Upper Central	35.9	13.4
Lower Central	87.0	125.7
South	8.9	13.7
General	22.2	6.8*
Eastern Shore		
Chester River	116.6	104.6
Eastern Bay	37.3	155.0*
Choptank River	963.2	1553.1*
Honga River	12.6	43.8
Fishing Bay	19.4	50.2*
Nanticoke River	489.8	384.9
Wicomico River	67.0	59.4
Tangier Sound	11.1	48.0*
Pocomoke Sound	4.5	14.6

†Table does not separate natural recruitment from seed planting efforts.
*Significantly different from 1962 to 1970 at 0.05 level.

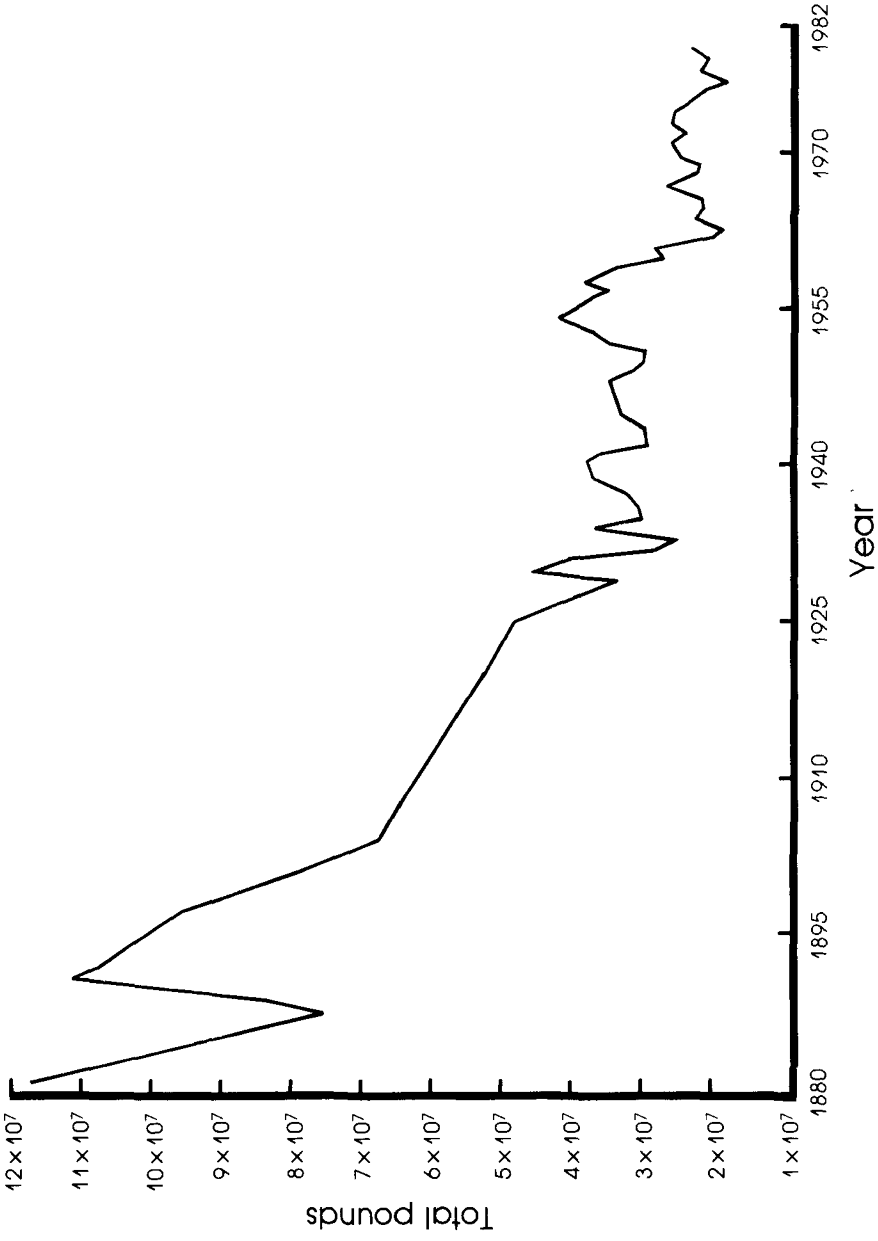


FIGURE 29. Historical pounds of shucked oyster meat for Chesapeake Bay, 1880 to 1981.

cant decreases occurred in four of the five western shore basins; significant increases occurred in four eastern shore basins. This shift from the western to eastern shore suggests a change that should be examined by more extensive analysis of biological, social, and economic factors.

OYSTER SPAT SET

The density (spat per bushel of cultch or suitable substrate) of oyster spat set indicates the success of natural oyster reproduction and recruitment. Oyster spat set is a reasonable predictor of oyster harvest (Davis et al. 1981). The Maryland Department of Natural Resources has collected oyster spat set data in the Maryland portion of Chesapeake Bay since 1939 (Meritt 1977; Davis et al. 1981); the Virginia Institute of Marine Sciences has collected similar information since 1946 (Haven et al. 1978). The methodology of oyster spat set data collection is described in more detail by Davis et al. (1981).

Comparison of the average oyster spat set for the past ten years with the previous ten to thirty years indicates significant declines in a number of locations in the Bay (Davis et al. 1981 and Table 23). Recent declines are most evident in upper, central Chesapeake Bay, certain rivers—Chester, James, Nanticoke, Patuxent, Pocomoke, Potomac, Rappahannock, and Wicomico—and Eastern Bay, Fishing Bay, and Pocomoke Sound (Table 22). The year 1980 was a good one for spat fall in the Choptank River, Eastern Bay, and Tangier Sound, areas that have shown increased oyster harvest. Spat set from 1980 was also high in the Honga River and at the mouth of the Potomac River, while spat set in the upper Chesapeake and its western tributaries was generally light (Figure 30). The high spat set of 1980 is most readily explained by high salinities in the Bay during the 1980 to 1981 spawning season that favored high survival and set of larval oysters. However, the York and James Rivers did not experience this high spat set, indicating that factors other than salinity were involved. Predators and disease have more effect on spat in Virginia than does salinity.

Trends in spat set have been documented in detail for the Potomac River (Krantz and Carpenter 1981, Figure 31). Although spat set in the lower Potomac varies in response to salinity, set in the middle and upper Potomac has been suppressed since the late 1960's and has shown no increase in 1980. This result suggests that some river-borne factor may be responsible for decreasing spat sets.

TABLE 22.
TRENDS IN MEANS OF OYSTER SPAT SET FOR THE PERIOD 1938-1980, COMPARING POST-1970 TO PRE-1970

Basin	Trend	Basin	Trend
CB South	N	Nanticoke River	—
CB Low Central	N	Patuxent River	—
CB Upper Central	—	Pocomoke River	—
Chester River	—	Pocomoke Sound	—
Choptank River	N	Potomac River	—
Eastern Bay	—	Rappahannock	—
Fishing Bay	—	Tangier Sound	N
Honga River	N	Wicomico River	—
James River	—	York River	N

+ = Positive Trend at >0.05
– = Negative Trend at >0.05
N = Not significantly different

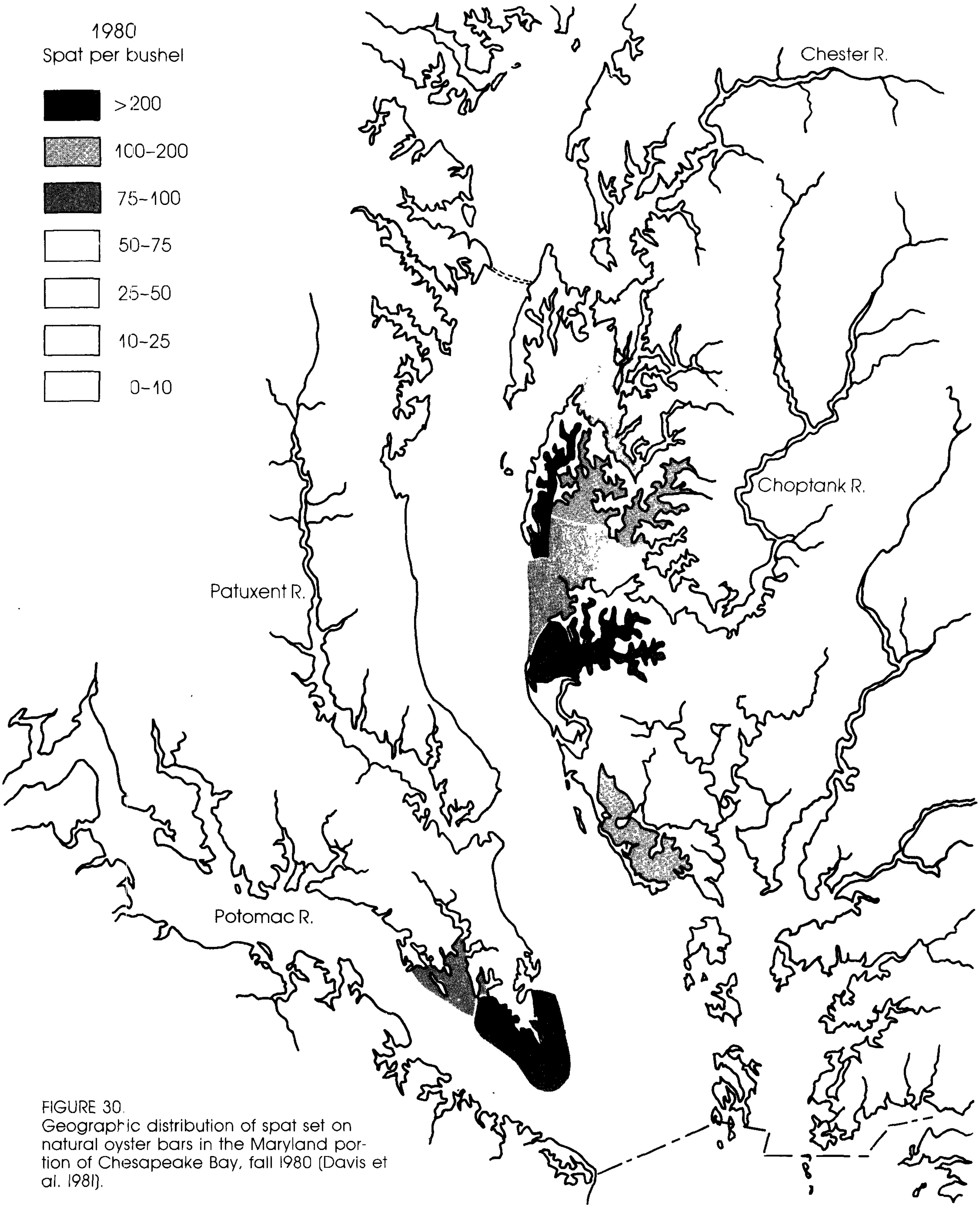


FIGURE 30.
Geographic distribution of spat set on natural oyster bars in the Maryland portion of Chesapeake Bay, fall 1980 (Davis et al. 1981).

TABLE 23.
CRAB HARVEST* FOR 1962 TO 1970 AND 1971 TO 1980, ANNUAL MEAN (POUNDS PER ACRE OF BASIN)

	1962-1970	1971-1980
Western Shore		
James River	15.08	4.66[b]
Patuxent River	1.81	3.79[b]
Potomac River	12.83	9.30
Rappahannock River	52.57	27.89[b]
York River	69.73	50.26
Main Bay		
North	0.18	0.52[b]
Upper Central	4.41	6.21
Lower Central	44.58	38.52
South	3.86	19.69[b]
General	41.23	33.91
Eastern Shore		
Chester River	4.78	5.32
Choptank River	0.22	0.71[b]
Eastern Bay	7.79	4.58
Fishing Bay	ND	ND
Honga River	ND	ND
Nanticoke River	ND	ND
Pocomoke Sound	ND	ND
Tangier Sound	97.69	40.70[b]
Wicomico River	ND	ND

*Harvest figures do not take catch per unit effort (CPUE) into account. CPUE data are not available.
[b]Significantly different from 1962 to 1970 at 0.05 level

OYSTER CONDITION INDEX

Data used for determination of the oyster condition in Maryland were obtained from the Maryland Department of Natural Resources, Tidewater Administration, Oxford, Maryland. Evaluation of these data showed no clear spatial or temporal patterns. Analysis relating these data with stress index and spat set data did, however, show some interesting correlations in the Chester and Patuxent Rivers (Chapter 3).

Oyster condition data for Virginia were taken from Haven et al. (1978) and from Dexter Haven's files at the Virginia Institute of Marine Sciences. Evaluation of these data revealed no clear temporal trend; however, the Rappahannock River showed consistently higher values. Haven (1965) suggests that this may be related to the nutritional content of waters. Natural sources of organic detritus or algal species composition may be factors. The condition index in Virginia does not appear to be related to spat set (Haven et al. 1978).

HARVEST OF CRABS

Many biological and ecological differences exist between blue crabs and oysters. For example, blue crabs are mobile and able to leave unfavorable conditions, while oysters are sessile. Blue crabs spawn in the mouth of the Bay; oysters remain in the estuary.

Landings data for blue crabs were obtained in the same way as that for oysters and are subject to the same caveats. Landings for Chesapeake Bay, 1880 to 1981, are shown in Figure 32. Harvests of crabs have generally increased since the 1930's, in contrast to the decrease exhibited by oysters. On the average, crabs have been declining Bay-wide since 1970. There have, however, been some shifts in areas where crabs are captured (Chapter 3).

Landings for individual basins are shown in Table 23. Significant increases occurred in the Patuxent and Choptank Rivers, and Chesapeake Bay, north and south. Significant decreases occurred in the Rappahannock and James Rivers, and Tangier Sound.

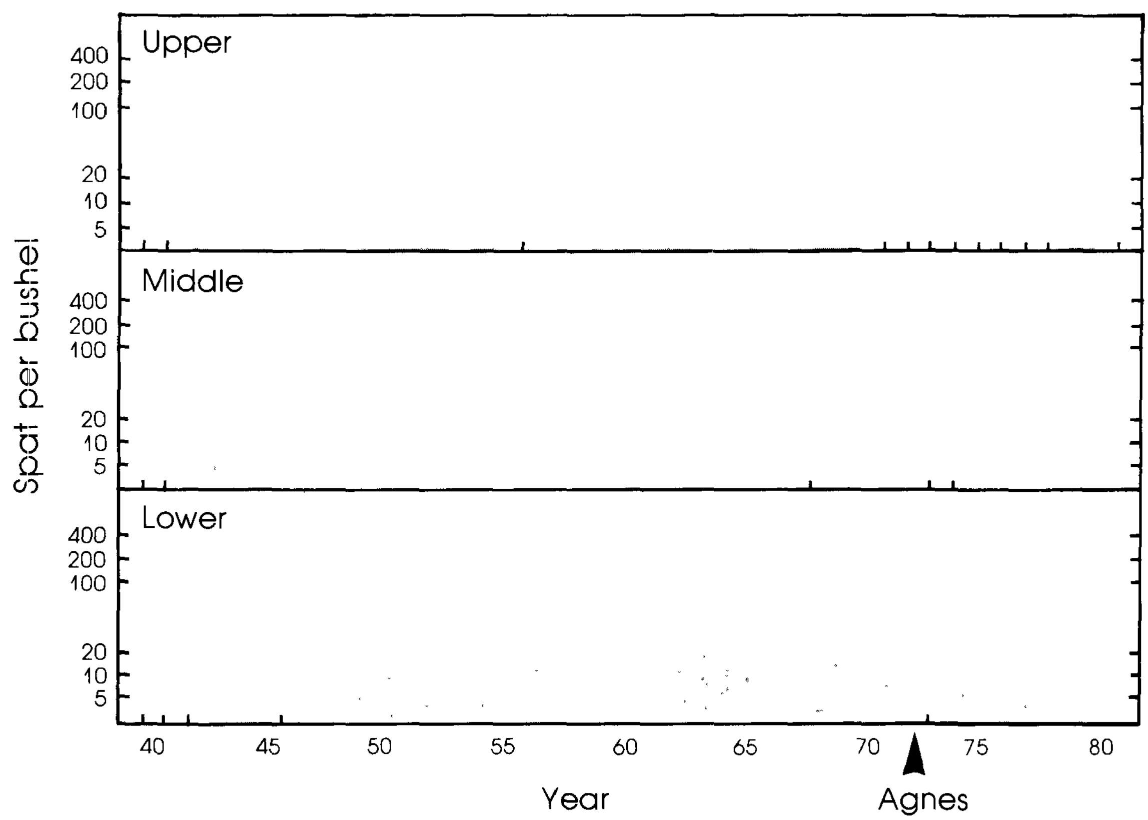


FIGURE 31. Oyster spat settlement on natural cultch in the Potomac River, 1939 to 1980 (Krantz and Carpenter 1981).

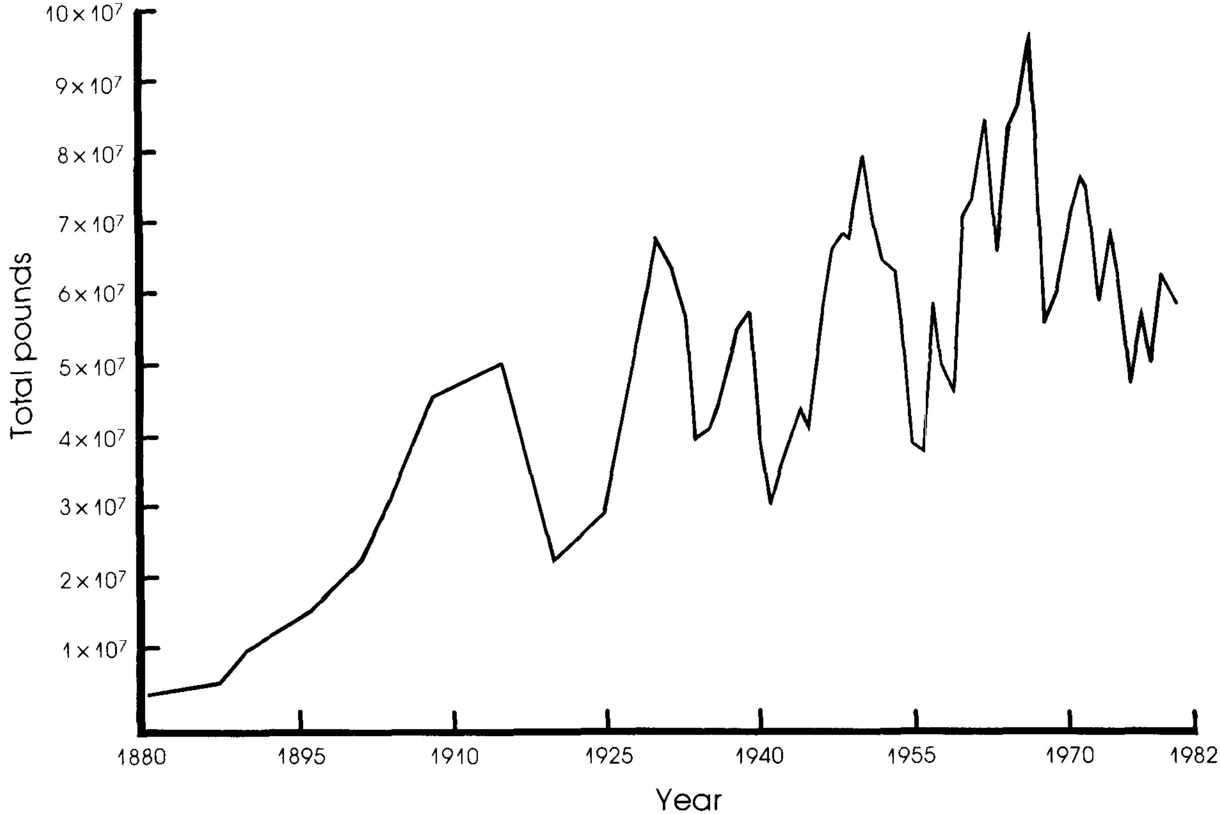


FIGURE 32. Historical landings of blue crabs for Chesapeake Bay, 1880 to 1981.

SECTION 6
CHANGES IN THE FINFISHERY

Another economically important biological resource in Chesapeake Bay is its finfishery. The species listed in Table 24 were assessed because of their economic importance. In this section, the environmental tolerances of these organisms are discussed first to understand some of the factors that affect them. This information is used in Chapter 3 to compare species sensitivities with ambient water quality conditions. More detailed information on environmental tolerances is included in Appendix C, Section 1, and in Kaumeyer and Setzler-Hamilton (1982).

Next, commercial landings of the species listed in Table 24 are examined. A discussion of the historical procedures for collecting landing statistics and caveats concerning their use is found in Appendix C, Sections 3 and 4. Landings are a reflection of stock sizes, but are affected as well by fishing effort. Because of the unavailability of acceptable historical effort data for most species

TABLE 24.
PRINCIPAL COMMERCIAL FINFISH SPECIES IN CHESAPEAKE BAY

Common Name	Scientific Name	Total 1980 Landings (lbs X 1000)
Alewife	*Alosa pseudoharengus*	(1369.1[a])
Blueback herring	*Alosa aestivalis*	(1369.1[a])
Bluefish	*Pomatomus saltatrix*	2791.2
Catfish	*Ictalurus sp.*	2265.7[b]
Croaker	*Micropogonias undulatus*	622.1
Menhaden	*Brevoortia tyrannus*	443,977.6
Shad	*Alosa sapidissima*	903.3
Spot	*Leiostomus xanthurus*	1755.3
Spotted sea trout	*Cynoscion regalis*	(5113.6[c])
Striped bass	*Morone saxatilus*	2563.3
Weakfish	*Cynoscion nebulosus*	(5113.6[c])
White perch	*Morone americana*	1101.9
Yellow perch	*Perca flavescens*	28.0
		462491.1

[a]Combined in landing statistics as Alewife; 1369,100 lbs for both species combined.
[b]Represents three species (*I. catus, I. nebulosus,* and *I. punctatus)*
[c]Combined in landing statistics as sea trout; 5113,600 lbs for both species combined.

(Rothschild et al. 1981), an attempt to adjust landings statistics for fishing effort was not done. Insufficient data were available to fully evaluate recreational catch, but Williams et al. (1982) have shown that recreational catch in the Maryland portion of the Chesapeake Bay may represent from about 40 percent of the combined recreational and commercial landings of striped bass to 90 percent of the total landings of spot. Therefore, recreational fishing may have a substantial impact on game fish species.

Commercial landings are the most spatially and temporally complete fishery data available. However, these data are affected by economic factors like fishing effort, as well as stock abundance. Thus, trends in commercial landings will be substantiated with data from surveys of young fish, expressed as the juvenile index, for four basins in Maryland and two basins in Virginia.

Juvenile finfish seining survey data have been collected by the Maryland Department of Natural Resources annually since 1954. Replicate hauls were first collected from a total of 22 permanent sampling locations in four major spawning areas using a 30.5 m beach seine in 1961.

Methodology was standardized in 1966 when samples were first taken at each station once in July or the first week of August, once during the last three weeks of August, and once in September or the first week in October. The Maryland Department of Natural Resources annual striped bass juvenile index for each river system is the average catch/haul in river over the sampling year. The expanded index for Chesapeake Bay in Maryland for a given year is a weighted average catch/haul for all stations sampled in that year.

The Virginia Institute of Marine Science has been collecting data on abundance and distribution of juvenile fishes in Virginia waters with bottom trawls since February 1955. Efforts have been focused primarily on the sciaenids (croaker, spot, and weakfish), hence the choice of bottom trawls. Analysis of these data through March 1982 are presented by species and river and include annual estimates of relative abundance, five point moving averages, and Virginia commercial landings (Wojcik and Austin 1982). These surveys are a reasonable indicator of stock sizes for young fish, particularly bluefish, striped bass, and white perch in Maryland, and croaker, spot, and weakfish in Virginia.

The thirteen species in Table 24 will be considered in two groups, based on the similarity of trends among species in each group. The groups, freshwater spawners (alewife, catfish, shad, striped bass, white perch, and yellow perch) and marine spawners (bluefish, croaker, menhaden, spot, spotted sea trout, and weakfish), are also significantly different in the proximity of their spawning grounds to riverine sources of anthropogenic materials.

ENVIRONMENTAL REQUIREMENTS AND TOLERANCES OF ESTUARINE FISH

The principal water quality parameters available to gauge possible impacts on finfishes include concentrations of dissolved oxygen, heavy metals, and halogenated hydrocarbons. Unfortunately, data are completely lacking for at least one half of the species examined. Interpretation and comparison of marine and estuarine toxicity data are difficult tasks because of the large number of variables affecting an organism's response to a particular pollutant. However, to give the reader some indication of the level of sensitivity to the key water quality parameters currently known, the following summary is provided. More detail is available in Appendix C, Section 1 and Kaumeyer and Setzler-Hamilton (1982).

Dissolved Oxygen

Thornton (1975) compared the DO tolerances of some 24 estuarine and marine fishes experimentally and from the literature, concluding "that a conservative estimate for maintaining diversity for marsh [estuarine] fish would require minimum permissible dissolved oxygen levels of 2 to 3 ml L^{-1} [2.8 to 4.2 mg L^{-1}] at 20 to 25C." He also ranked these 24 species in order of their sensitivity to low oxygen levels. Listed in order from most to least sensitive are those of special concern in this analysis (Thornton 1975):

Blueback herring	*Alosa aestivalis*
Alewife	*Alosa pseudoharengus*
Atlantic menhaden	*Brevoortia tyrannus*

Atlantic silversides	*Menidia menidia*
Bluefish	*Pomatomus saltatrix*
Weakfish	*Cynoscion sp.*
Spot	*Leiostomus xanthurus*
Striped Bass and White perch	*Morone sp.*
Mummichog	*Fundulus sp.*

Adults of *Alosa aestivalis* will die at oxygen levels of 2.0 to 3.0 mg L^{-1} if held up to 16 hours. For striped bass, *Morone saxatilis*, oxygen concentrations of 3.3 mg L^{-1} at 18C are lethal for yolk sac larvae (Rogers and Westin 1981); concentrations of 5 to 6 mg L^{-1} seem acceptable for larval development (Bogdanov et al. 1967).

Toxicity Studies for Selected Species

Spot

Middaugh et al. (1975) examined the effect of cadmium on larvae and the effect of DO reduction on Cd toxicity. With exposure concentrations ranging from 0.09 to 0.80 ppm Cd, they noted 17 to 42 percent less survival at 1.6 ppm than at 6.5 ppm DO. This result is most likely related to increased stress on the larvae at lower oxygen concentrations.

Six pesticides (dieldrin, DDE, DDT, endosulfan, endrin, and toxaphene) were tested for their effect on juvenile spot by Lowe (1964, 1966, undated). All were very toxic: for most of these compounds, exposure to 0.001 ppm resulted in at least 50 percent mortality.

Hansen et al. (1971) examined the chronic toxicity of PCBs (Arochlor® 1254) to juvenile spot. They determined an LC_{50} value of 0.005 ppm under several temperature and salinity combinations with exposure for 18 and 26 days.

White Perch

The major toxicity data for white perch are in a study by Rechwoldt et al. (1977), who reported a 96 hr LC_{50} value of 0.42 ppm for larval white perch using aldrin (pesticide), and 40.0 ppm using 2,4-D (herbicide).

Striped Bass

Of all species examined, striped bass had the largest toxicity data base. O'Rear (1972) compared the toxicity of Cu and Zn to the embryos. Copper was the most toxic with a 48 hr LC_{50} value of 0.74 ppm. Hughes (1973) has tested the tolerance of larval striped bass to Cd, Cu, and Zn. She determined that Cd was by far the most toxic. With a 96 hr exposure, 0.001 ppm cadmium chloride was required for 50 percent mortality. Juvenile striped bass also exhibited strong sensitivity to Cd with a 96 hr LC_{50} of 0.002 ppm. Of all the metals tested, Zn and Cr were the least toxic to juvenile striped bass, with LC_{50} values ranging from 10 to 18 ppm in 96 hr tests.

Hughes (1973) also examined the toxicity of aldrin and dieldrin to striped bass larvae. She determined that the larvae were more sensitive to dieldrin. Several investigations have tested the toxicity of nine pesticides to juveniles. Endrin was the most toxic, with a 96 hr LC_{50} value of 0.1 ppb (Korn and Ernest 1974).

The herbicide data were somewhat conflicting. Hughes (1973) reported a 96 hr 2,4-D LC_{50} of 3.0 ppm, while Rechwoldt et al. (1977) listed an LC_{50} of 70.1 ppm with similar test conditions. It was not possible to determine a possible explanation for the reported differences in toxicity to 2,4-D.

Benville and Korn (1977) tested ethyl benzene, benzene, and toluene (monocyclic aromatic hydrocarbons). The juveniles tested were relatively tolerant to all three compounds; 96 hr LC_{50} values ranged from 4.3 to 7.3 ppm. Ethyl benzene was the most toxic.

The current toxicity data are useful in that the relative intolerance of several species to particular pollutants is apparent. For example, striped bass are extremely sensitive to Cd. The striped bass tolerance will be compared with current monitoring data for Cd in Chapter 3 to see if survival is possibly being limited by this metal.

COMMERCIAL FINFISH LANDINGS

Landings data were obtained and analyzed as described in Appendix C, Section 3 for the species listed in Table 24. These species were selected because of their economic importance. NOAA and/or Maryland fishery codes were initially grouped into basins (Figure 22). These 32 basins were then grouped into thirteen major tributary basins, which were finally combined to produce total landings for the entire Chesapeake Bay (Appendix C, Section 3). This approach provided information for local, regional, and Bay-wide comparisons.

Because of a lack of local and regional information for the entire period of record (1880 to 1980), the long-term analysis of fisheries trends is restricted to the entire Chesapeake Bay. Analysis by decade (Table 25) indicates that landings of marine spawners have increased from 38.08 pounds per acre (1881 to 1890) to 143.88 pounds per acre (1971 to 1980). This increase has occurred in two waves: an early peak occurred from 1911 to 1920 (129.30 pounds per acre), followed by an abrupt decline, and an increase to present levels. Most of this increase is accounted for by menhaden; bluefish have shown a dramatic increase during the 1970's; croaker and sea trout, on the other hand, show sporadic recent increases that do not approach historical levels.

As a group, freshwater spawners have declined from a maximum of 20.44 pounds per acre (1901 to 1910) to 5.64 pounds per acre (1971 to 1980). The largest yields throughout the period have been of alewife (herring); landings of these species, as well as of shad and yellow perch, are now at unprecedented low levels. Striped bass landings were maximal from 1961 to 1973, but have declined abruptly in the present decade. White perch also show a recent decline; catfish show no clear pattern.

The increased yield of marine spawners and decreased yield of freshwater spawners have resulted in a major shift in the proportion of the finfishery accounted for by each group: from 1881 to 1890, the proportion of freshwater to marine spawners was 25:75; from 1971 to 1980, this proportion was 4:96.

RELATIONSHIP BETWEEN LANDINGS AND JUVENILE INDEX

For some species, fisheries landings tend to track the occurrence of successful year classes if fishing pressure is stable or landings are adjusted for catch per unit effort. Examples of such concurrence may be seen in Atlantic menhaden (Cronan 1981), as well as in striped bass (Boone 1980) and white perch (Richkus and Summers 1981). Without doubt, in the absence of several consecutive successful year classes, fishery landings do decline, often dramatically.

To show relationships between juvenile finfish surveys (Boone 1980) and landings, and to attempt to validate landings trends for the principal species in the Chesapeake, the Maryland juvenile index for each species was aggregated for the four collection areas of the Bay (Nanticoke River, Choptank River, Potomac River, and head of the Bay). Moving averages of these data were plotted against total Chesapeake Bay landings for each species. For example, a moving average of three years aggregates three annual means of juvenile index to form a single mean (i.e., mean for 1960, 1961, and 1962) for part of a longer period of record and repeats the process for all possible sequential combinations of three years (1961, 1962, and 1963, etc.) for the entire period of record. Moving averages that bracketed the typical time for a year class to enter the fishery were selected. For example, striped bass are thought to enter the fishery at two to three years of age, but the slowest-growing fish of the species treated here, white perch, is thought to enter the fishery between six to eight years of age. Therefore, two to eight year moving averages were calculated for species where data were available. The moving averages have the effect of lagging the data by the number of years covered by the moving mean; that is, a 3-year moving mean of the juvenile index should provide the best fit with the landings curve, if the landings are mainly comprised of 3 year-old fish.

Table 26 indicates the expected age to enter the fishery and the "best fit" moving average for each species treated. Regression analysis was also conducted in which the juvenile index was lagged against landings, with less success.

Through the use of this method, bluefish provide the best apparent correspondence among marine spawners; menhaden provide a fair to poor fit. This result indicates that although Maryland's juvenile index is not designed explicitly to sample marine spawners, it does provide some indication of future harvest for bluefish and menhaden. It also appears that increased harvests of bluefish

TABLE 25.
UNIT AREA FINFISH LANDINGS FOR CHESAPEAKE BAY, 1881 TO 1980, ANNUAL MEAN BY DECADE (POUNDS PER ACRE)

	1881-1890	1891-1900	1901-1910	1911-1920	1921-1930	1931-1940	1941-1950	1951-1960	1961-1970	1971-1980
Freshwater spawners										
Alewife	7.51	10.48	15.43	9.24	7.99	6.63	7.58	9.78	10.28	2.66
Catfish		0.34	0.42	0.46	0.32	0.33	0.66	0.89	0.64	0.62
Shad	4.02	5.50	3.61	2.71	2.89	1.66	1.53	1.60	1.29	0.61
Striped bass	0.64	0.57	0.43	0.50	0.64	0.63	1.44	1.43	2.18	1.37
White perch	0.41	0.46	0.39	0.34	0.26	0.22	0.49	0.54	0.69	0.37
Yellow perch	0.44	0.36	0.16	0.15	0.07	0.07	0.07	0.04	0.03	0.01
Total	13.02	17.71	20.44	13.40	12.17	9.54	11.77	14.28	15.11	5.64
% of all finfish species	25.5	22.9	18.3	8.6	17.9	13.2	15.7	13.7	12.4	3.8
Marine spawners										
Bluefish	0.74	0.71	0.21	0.09	0.17	0.21	0.09	0.09	0.14	1.00
Croaker	0.49	1.00	1.56	6.67	7.52	10.26	11.44	3.00	0.30	1.13
Menhaden	35.90	55.75	87.14	129.30	43.00	47.71	47.13	85.76	105.47	140.32
Sea Trout	0.95	2.08	2.57	5.36	5.24	4.75	4.60	0.80	0.53	1.43
Total	38.08	59.54	91.48	141.42	55.93	62.93	63.26	89.65	106.44	143.88
% of all finfish species	74.5	77.1	81.7	91.3	82.1	86.8	84.3	86.3	87.6	96.2
% of all finfish accounted for by menhaden	70.2	72.2	77.9	83.5	63.1	65.8	62.8	82.5	86.8	93.8
All finfish species total	51.10	77.25	111.92	154.82	68.10	72.47	75.03	103.93	121.55	149.52

TABLE 26.
CORRESPONDENCE OF MARYLAND FINFISH JUVENILE INDEX AND FINFISH LANDINGS FOR CHESAPEAKE BAY (1958 TO 1980)

Species	Approx. Age Enters Fishery (yrs.)	Visual Correspondence	R Value (>0.05)	Best Fit Moving Mean (yrs.)
Alewife	2 +	Poor	—	None
Bluefish	1-3	Good	0.97	4
Catfish	3 +	Poor	0.53	8
Croaker	2 +		(Insufficient Data)	
Menhaden	1 +	Fair—Poor	0.38	3
Shad	4-6		(Insufficient Data)	
Spot	3-5	Poor	—	None
Striped bass	2-3	Good	0.71	4
Weakfish	2-4		(Insufficient Data)	
White perch	6 +	Fair	0.54 + 0.50	3 + 6
Yellow perch	4 +	Poor	—	None

and menhaden are not strictly a function of increased effort, but are supported by increased availability of fish.

Juvenile indices of other marine spawners have behaved similarly during the historical record. Pearson correlation analysis indicates that menhaden correlated (0.05 level) with bluefish ($r = 0.70$) and with spot ($r = 0.87$). Spot correlated with croaker ($r = 0.55$) and bluefish ($r = 0.50$). Thus, to some extent, the annual variation in the juvenile index of these species seems to be controlled by similar factors.

A good relationship has been shown between landings and juvenile index for striped bass (Boone 1980) and white perch (Richkus and Summers 1981). Chesapeake Bay Program data, unadjusted for fishing effort, also showed a good relationship with striped bass and a fair relationship with white perch (Table 26). Thus, the juvenile index does provide some support for using landings as an indicator of trends for these two species. The decline in striped bass and white perch availability suggested by decreased landings is supported by the juvenile index data.

Juvenile success of some freshwater spawners may be affected by similar factors. For example, Pearson correlation analysis indicates that juvenile indices of white perch correlated with those of alewife ($r = 0.69$) and striped bass ($r = 0.47$) at the 0.05 level. Thus, these species may be responding to similar environmental factors.

LANDINGS AND JUVENILE INDICES OF MARINE SPAWNERS

As previously discussed, landings of menhaden have increased during the historical period from 1880 to 1980. This increase is shown graphically in Figure 33, which illustrates the early upward trend, culminating in approximately 1920, followed by a second upward trend (data were acquired less frequently before 1930, accounting for the lower apparent variability during the early period).

Historical landings of bluefish are shown in Figure 34, indicating the recent major peak in 1978 with a very slight decline since then. Landings of croaker and weakfish, showing declines since the 1940's but with sporadic recent increases, are shown in Figures 35 and 36.

Commercial landings data for individual basins are available since 1962. Mean annual commercial landings for marine spawners for two periods, 1960 to 1970 and 1971 to 1980, are shown in Table 27. These two periods were selected

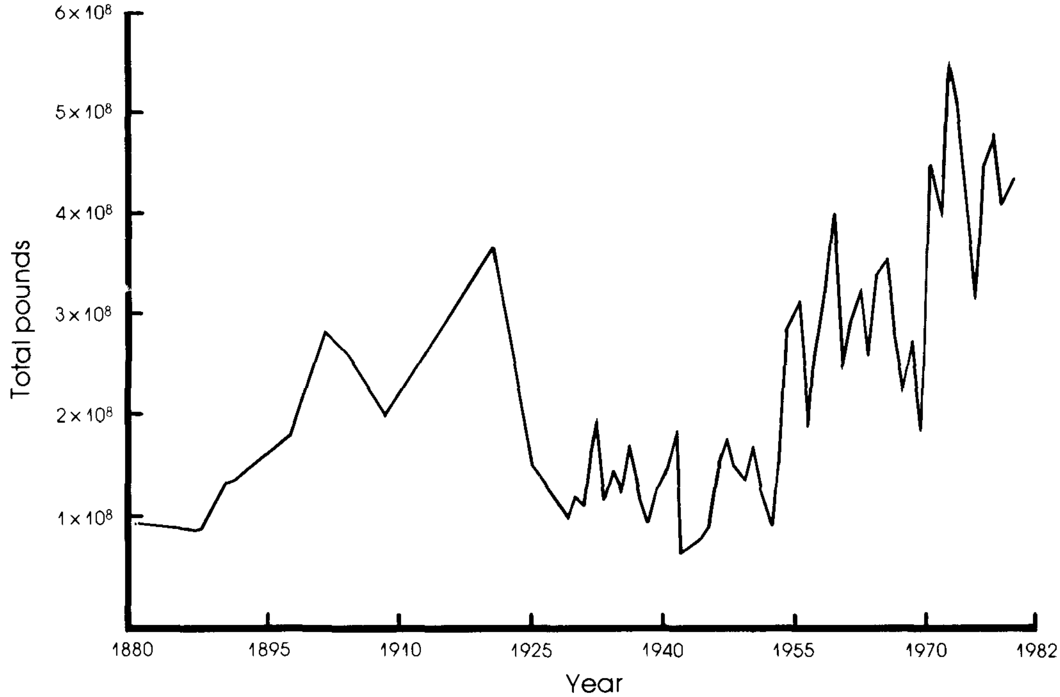


FIGURE 33. Historical landings of menhaden for Chesapeake Bay, 1880 to 1981.

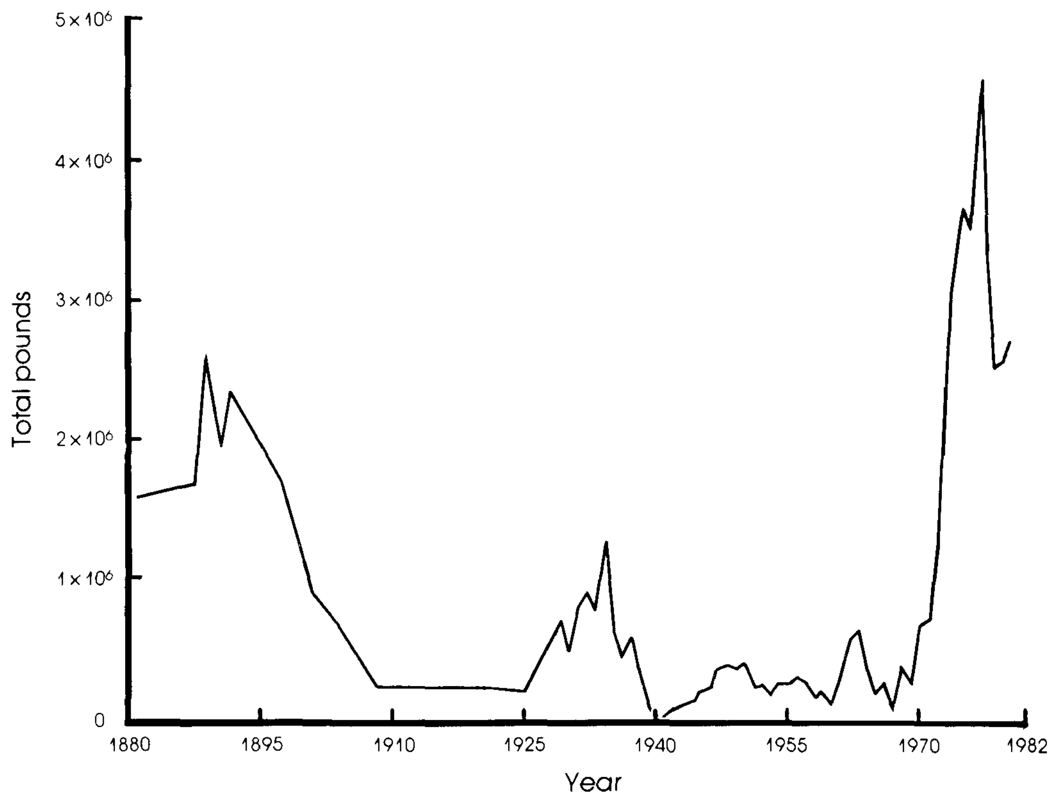


FIGURE 34. Historical landings of bluefish for Chesapeake Bay, 1880 to 1981.

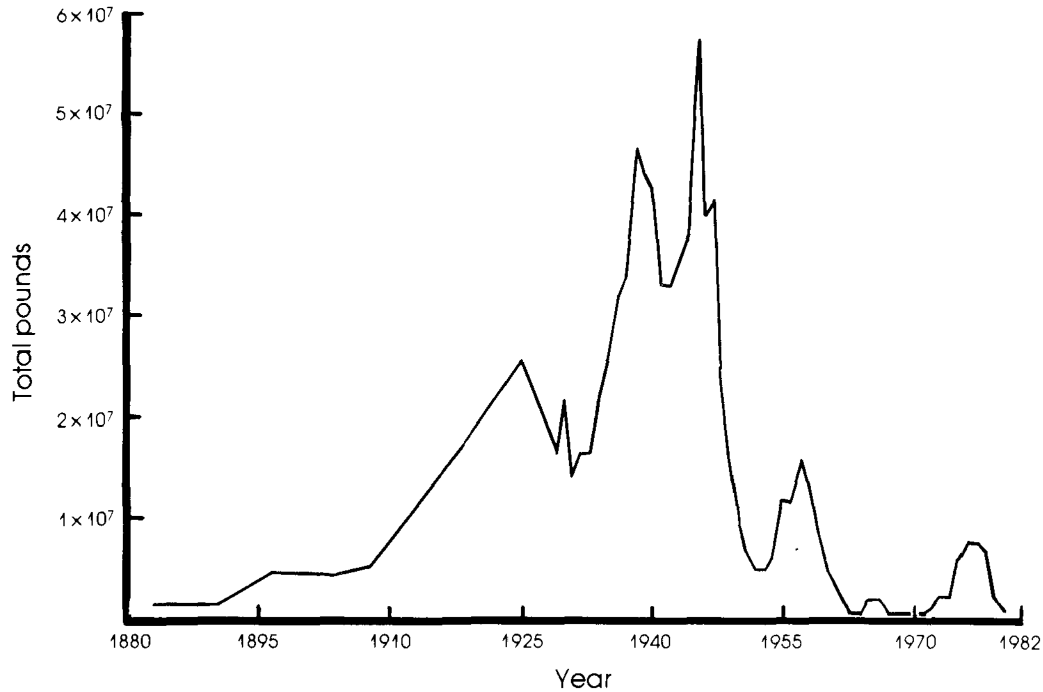


FIGURE 35. Historical landings of croaker for Chesapeake Bay, 1880 to 1981.

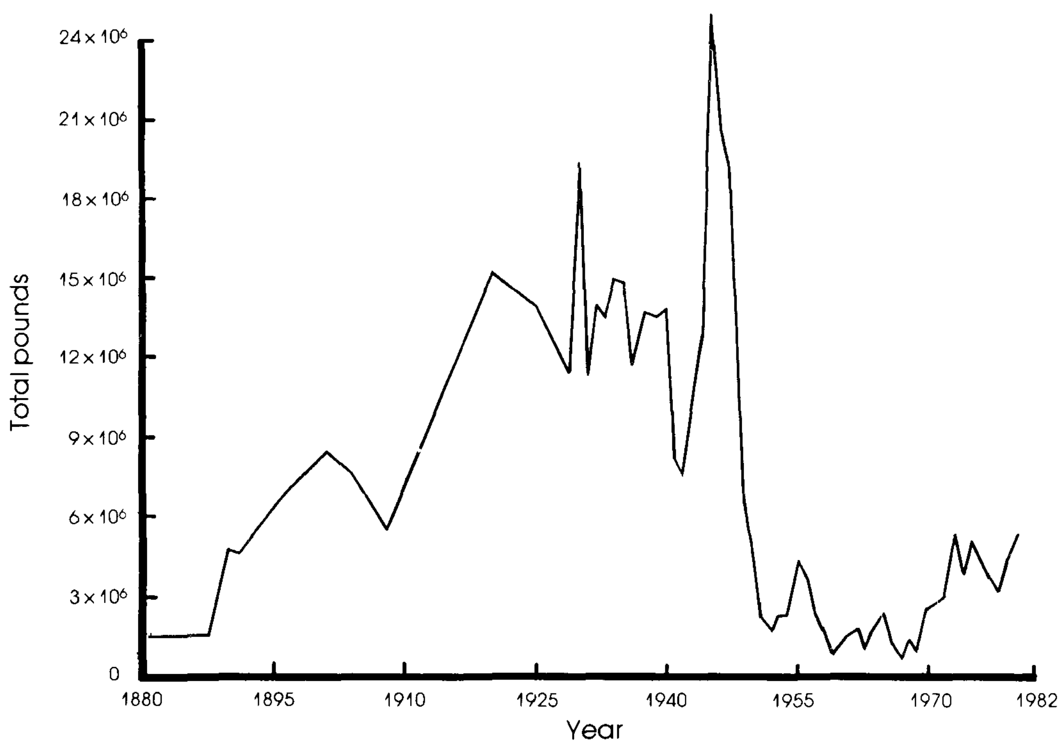


FIGURE 36. Historical landings of weakfish for Chesapeake Bay, 1880 to 1981.

TABLE 27.
LANDINGS FOR MARINE-SPAWNING FINFISH, ANNUAL AVERAGE FOR 1962 to 1970 AND 1971 to 1980 (POUNDS/ACRE)**

	1962-1970	1971-1980
Western Shore		
James River	13.21	0.47*
Patuxent River	0.12	0.31*
Potomac River	20.73	45.14
Rappahannock River	38.34	67.43
York River	47.44	38.68
Main Bay		
North	1.37	0.51
Upper Central	0.12	1.88*
Lower Central	4.28	4.84
South	1.99	5.21*
General	314.23	454.19*
Eastern Shore		
Chester River	0.08	0.88*
Choptank River	0.24	0.69
Eastern Bay	0.02	0.27
Fishing Bay	4.68	14.15*
Honga River	0.93	16.64*
Nanticoke River	1.78	3.18
Pocomoke Sound	0.09	0.14
Tangier Sound	0.54	0.35
Wicomico River	0.66	0.25

*Significantly different from 1962 to 1970 at 0.05 level
**One pound/acre = 0.112 g/m^2

because trend analysis showed shifts in the trends of both freshwater and marine spawning fishes occurring around 1969 to 1970. Basins in which significant change occurred showed increased landings of marine spawners, with the exception of the James River (whose finfishery was closed for part of the 1971 to 1980 period).

Trends in finfish juvenile survey data were determined by inspection of moving means (three-year in Maryland; five-year in Virginia) for each basin (head of Bay, Potomac, Choptank, Nanticoke, Rappahannock, and York). The Maryland survey is designed to assess the abundance of young-of-the-year (age 0) alewife, bluefish, striped bass, shad, white perch, and yellow perch (Boone 1980). It also provides evidence of the abundance of multiple year classes of Atlantic menhaden, Atlantic silversides, Bay anchovy, catfish, spot, and mummichog. The Virginia survey, on the other hand, is designed to assess the abundance of young croaker, spot, and weakfish and is not exclusively selective for age 0 fish (Wojcik and Austin 1982).

Trends in Maryland tributaries were verified by comparing pre- and post-1970 means for the period of record, 1958 to 1980 (Table 28). The marine spawners menhaden, bluefish, and spot show statistically significant increases in all four

TABLE 28.
COMPARISON OF 1971 TO 1981 ANNUAL MEAN JUVENILE INDICES WITH 1958 TO 1970 ANNUAL MEANS

Species	CB North	Choptank	Nanticoke	Potomac
Alewife[1]	N	–	N	N(+)
American shad[1]	–	–	–	N
Atlantic menhaden[2]	+	+	+	+
Atlantic silversides[2]	–	–	–	–
Bay anchovy[2]	–	–	–	N(–)
Bluefish[1]	+	+	+	+
Catfish[2]	+	N(–)	N	+
Mummichog[2]	+	+	+	N
Spot[2]	+	+	+	+
Striped bass[1]	N	N	N	–
Weakfish[2]	N	N	N	N
White perch[1]	N	–	N	+
Yellow perch[1]	N	–	N	+

+ = Statistically significant increase (0.05 level)
– = Statistically significant decrease (0.05 level)
N = Not significantly different
( ) = Increase or decrease at 0.10 level
[1]Based on 0 age class fish
[2]Based on mixture of age classes

areas. These statistical increases in annual means are manifestations of several strong year classes during the recent period; Figure 37 shows a typical example.

The Virginia small fish trawl (Wojcik and Austin 1982) showed the trends in the York and Rappahannock Rivers to be similar to those determined by the CBP analysis of the Maryland Department of Natural Resources juvenile finfish survey (Boone 1980) for marine spawners.

LANDINGS AND JUVENILE INDICES OF FRESHWATER SPAWNERS

Landings of alewife, shad, striped bass, white perch, and yellow perch show substantial declines over previous levels in Chesapeake Bay. Landings of shad and yellow perch have declined steadily over the period (Figures 38 and 39); declines in alewife, white perch, and striped bass landings have been more recent (Figures 40, 41, 42).

Mean annual commercial landings for freshwater spawners are shown in Table 29. Significant decreases occurred in several basins; these decreases were fairly evenly distributed among the three regions.

Comparison of the Maryland juvenile indices of freshwater spawners for the period 1971 to 1981 with the previous period (1958 to 1970) (Table 28) indicates that alewife declined in the Choptank River; American shad declined in all basins except the Potomac River; striped bass showed a statistically significant decline only in the Potomac River; and white perch and yellow perch showed increases in the Potomac River. That striped bass appeared to decline in only one basin may seem surprising because of the public perception of a dramatic Bay-wide decline. By inspection, one sees declining trends in all four basins studied. The

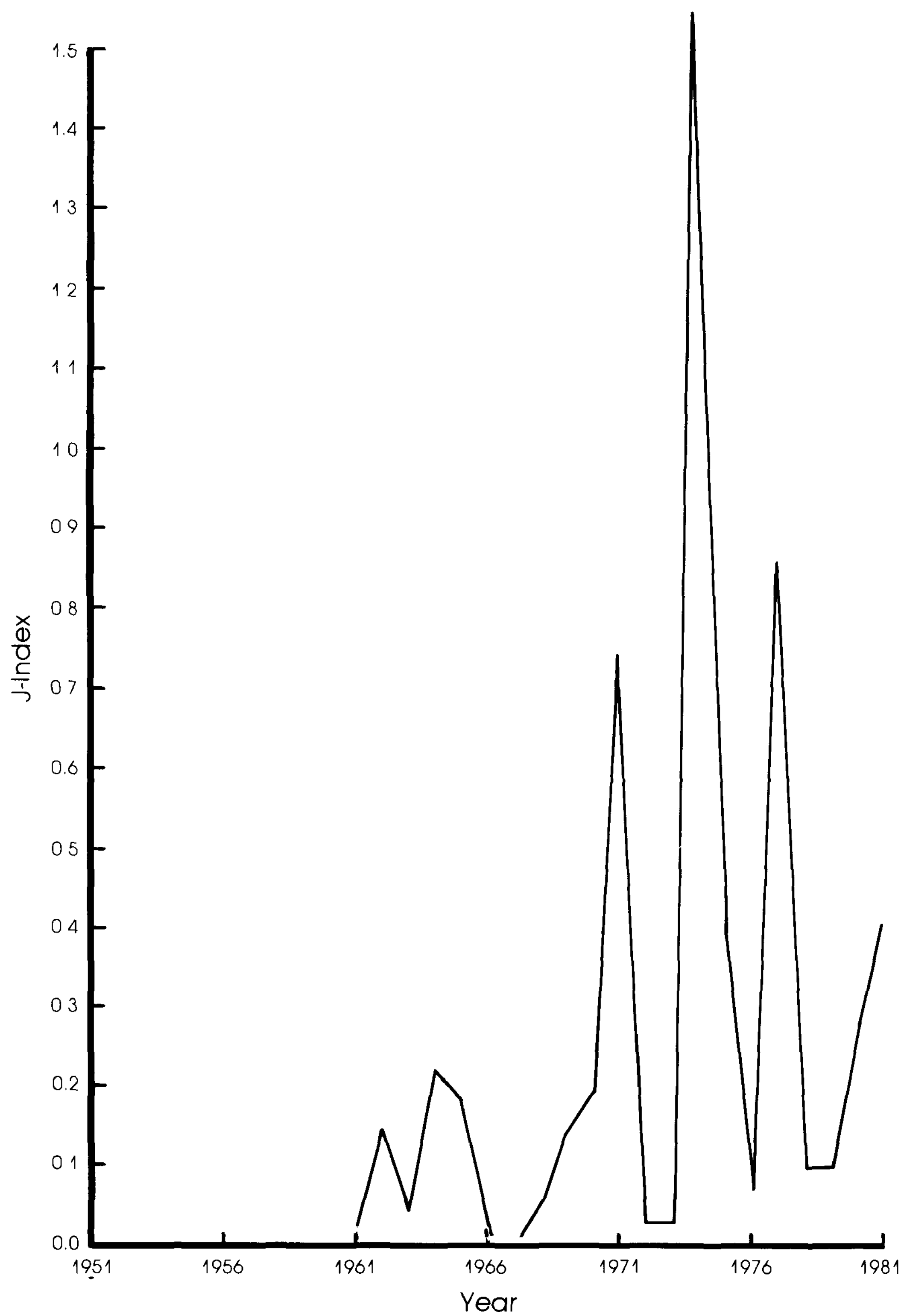


FIGURE 37. Historical changes in the juvenile index (weighted average catch per seine sample for all stations sampled that year) for bluefish in Chesapeake Bay.

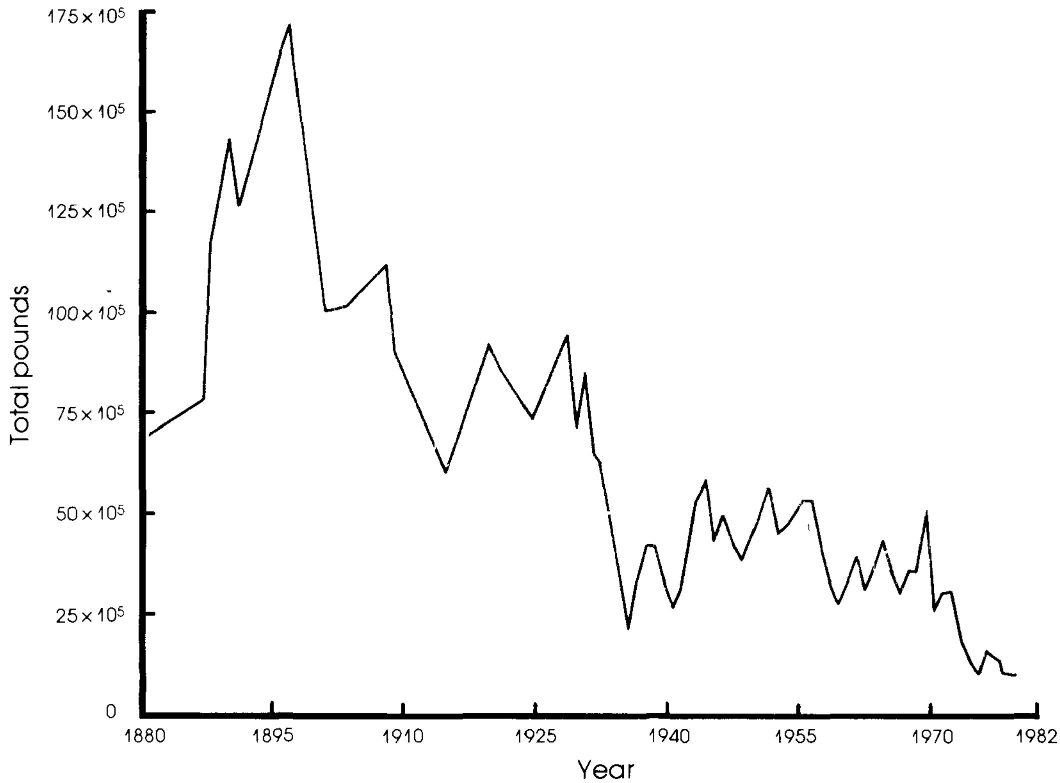


FIGURE 38. Historical landings of American shad for Chesapeake Bay, 1880 to 1981.

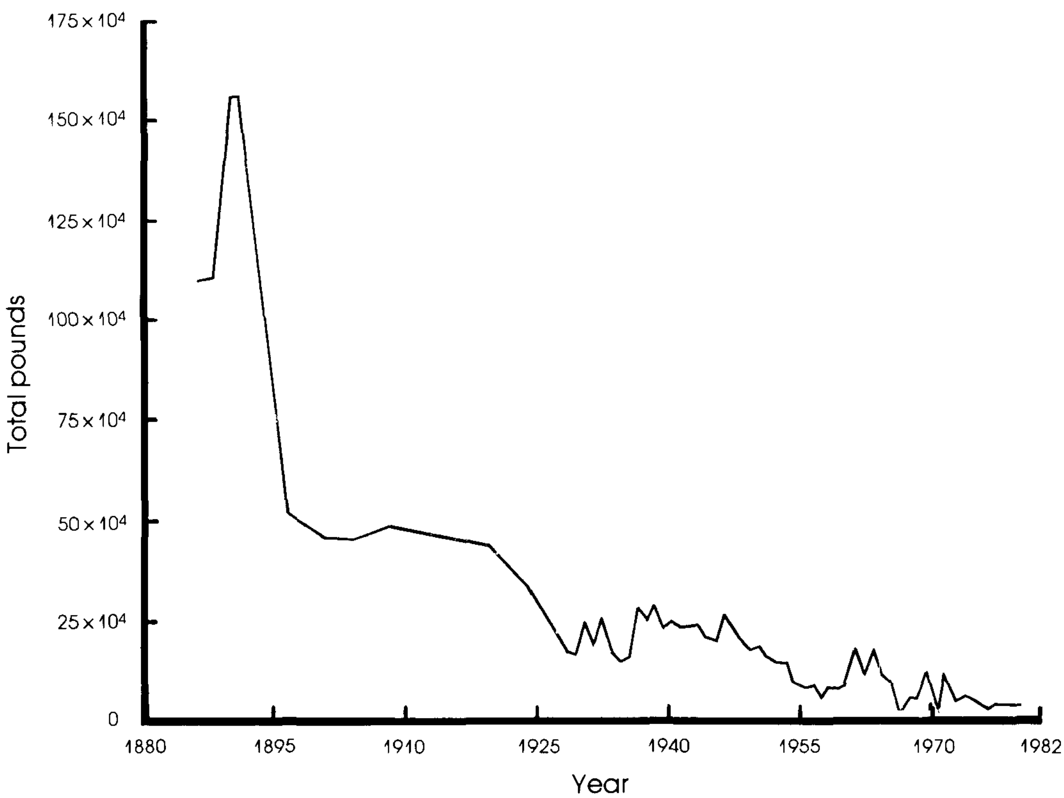


FIGURE 39. Historical landings of yellow perch for Chesapeake Bay, 1880 to 1981.

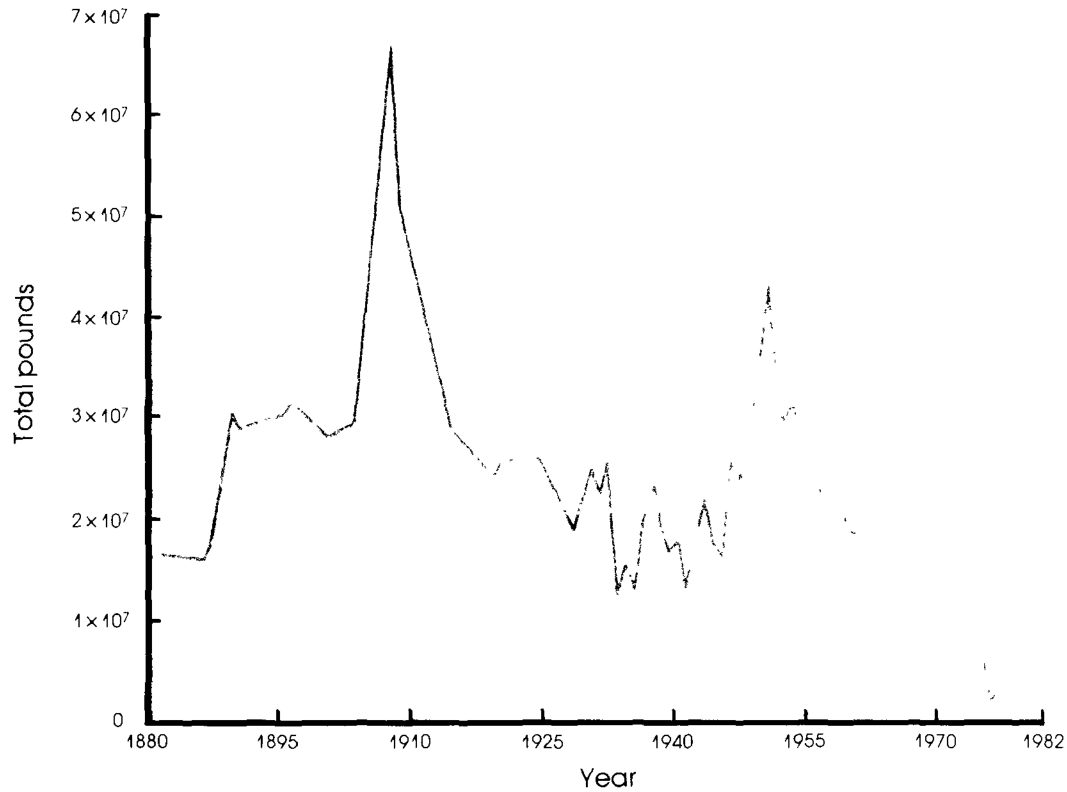


FIGURE 40. Historical landings of alewife for Chesapeake Bay, 1880 to 1981.

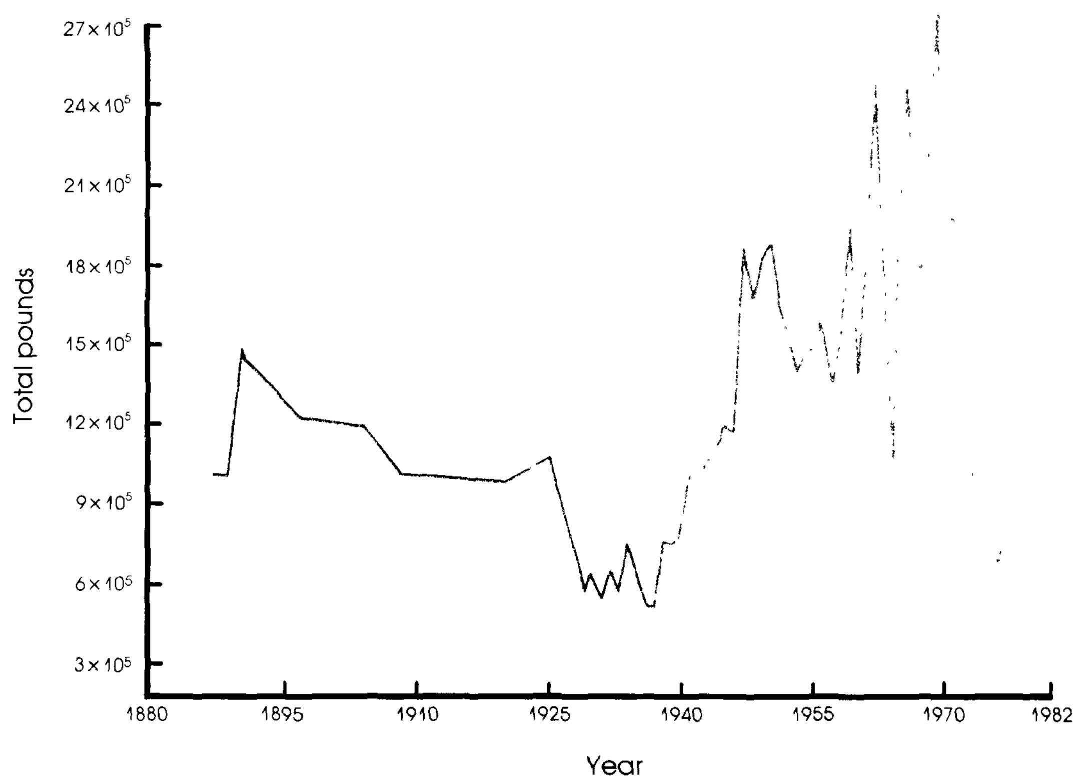


FIGURE 41 Historical landings of white perch for Chesapeake Bay, 1880 to 1981.

FIGURE 42. Historical landings of striped bass for Chesapeake Bay, 1880 to 1981.

statistical result occurred because there was a very strong year class early in the recent period, followed by a decline. For example, in the Nanticoke River (Figure 43) there was a very strong year class in 1972, followed by a decade of no strong year class. Declining trends may be contributed to by climate (as it affects the success of year classes), overfishing of spawning stock, and anthropogenic effects that destroy spawning areas (such as the installation of Conowingo Dam on the Susquehanna River which has restricted shad spawning), and, finally, by the effects of toxic materials on the various life stages of fishes.

The Maryland Department of Natural Resources juvenile finfish survey for 1982 has been completed. Indications are that striped bass recruitment is average in the Nanticoke River, better than average in the Choptank River, average in the Potomac River, and relatively poor in the head of the Bay. Reproduction of marine spawners continues to be successful.[10] These trends support the conclusions drawn by the CBP based on the period of record for the juvenile finfish survey of 1958 to 1981.

Because the Virginia survey (Wojcik and Austin 1982) is not designed to capture most of the freshwater spawners that are taken in the Maryland Department of Natural Resources survey, results in this group of species cannot be compared. Recent modifications in collecting procedures should improve fisheries comparisons in the future.

ESTUARINE FORAGE FISH

Juvenile indices for three estuarine species, mummichog (*Fundulus heteroclitus*), Atlantic

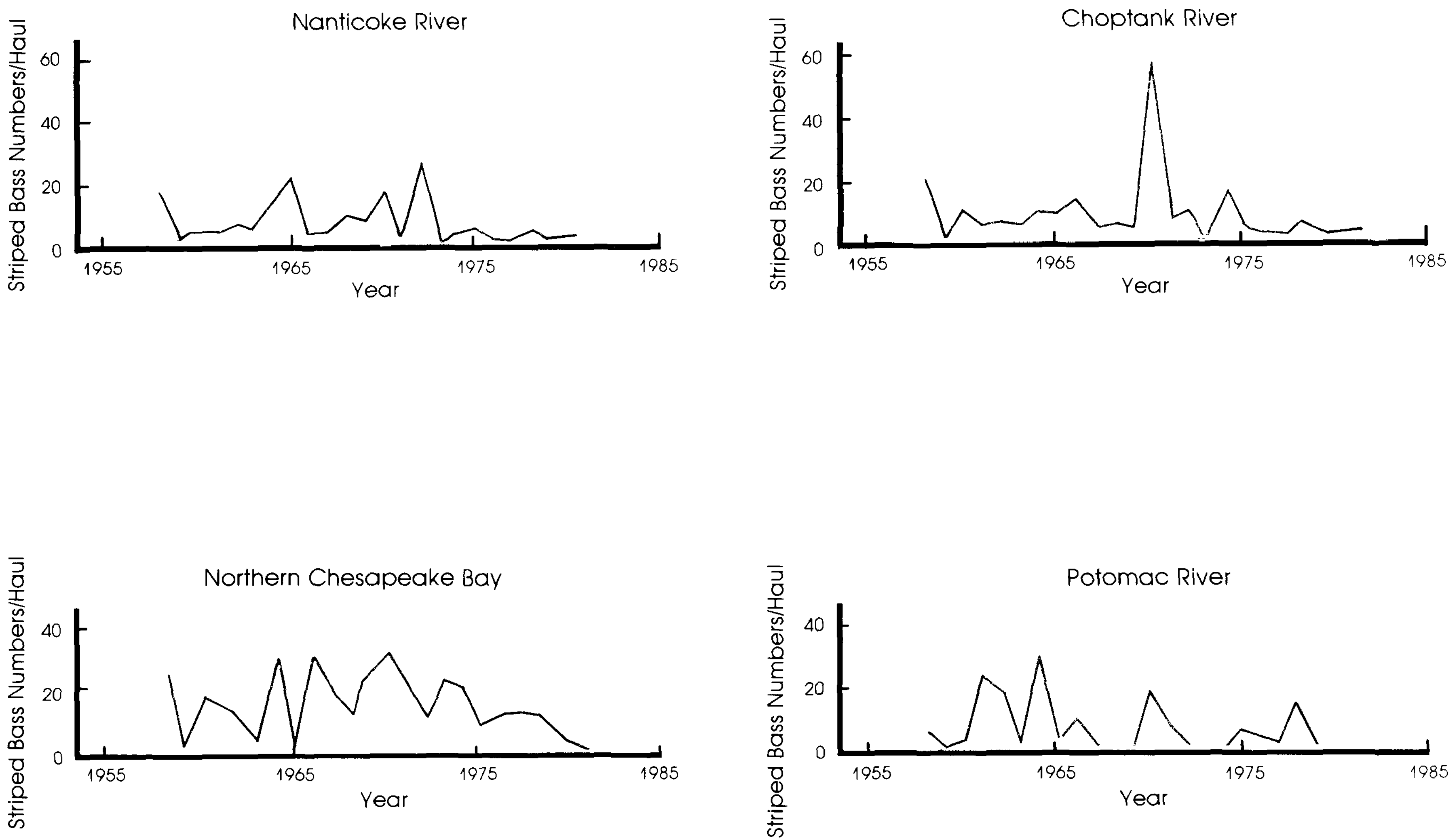


FIGURE 43. Historical changes in juvenile index (weighted average catch per seine sample for all stations sampled that year) for striped bass in northern Chesapeake Bay, and the Potomac, Choptank, and Nanticoke Rivers.

TABLE 29.
LANDINGS FOR FRESHWATER-SPAWNING FINFISH, ANNUAL AVERAGE FOR 1962 TO 1970 AND 1971 TO 1980 (POUNDS/ACRE)**

	1962-1970	1971-1980
Western Shore		
James River	26.85	9.39*
Patuxent River	4.74	4.13
Potomac River	34.56	13.90*
Rappahannock River	52.61	20.13*
York River	51.76	8.31
Main Bay		
North	20.86	12.49*
Upper Central	10.93	7.13*
Lower Central	3.51	1.66*
South	1.61	1.07
General	14.63	3.73*
Eastern Shore		
Chester River	9.27	4.73*
Choptank River	12.53	4.53*
Eastern Bay	1.73	1.04*
Fishing Bay	14.50	4.74*
Honga River	1.02	0.80
Nanticoke River	26.89	12.57*
Pocomoke Sound	0.50	0.32
Tangier Sound	1.15	0.50*
Wicomico River	8.23	6.77

*Significantly different from 1962 to 1970 at 0.05 level
**One pound/acre = 0.112 g/m^2

silverside (*Menidia menidia*), and Bay anchovy (*Anchoa mitchilli*) were analyzed to determine trends in these noncommercial species. Results for Maryland are shown in Table 28. Mummichog increased in all basins. Bay anchovy and Atlantic silversides, which use SAV as habitat, declined in all basins. The Virginia survey (Wojcik and Austin 1982) differs in that while the Bay anchovy has been declining during the past decade in the Maryland tributaries surveyed (Table 29), the Bay anchovy has been shown to be increasing in the Virginia tributaries surveyed during the same period.

SECTION 7

SYNTHESIS: A GEOGRAPHIC ASSESSMENT

Changes in several biological resources have been discussed, from the ecologically important phytoplankton to the economically important commercial fisheries.

Historical changes have been seen in phytoplankton abundance and species composition in the two areas (upper Bay, upper Potomac) for which data are available. These changes are usually associated with nutrient enrichment.

The decline of SAV begun in the late 1960's has continued to the present, following a down-estuary direction. Thus, at present, the most abundant SAV is found in the lower Bay and eastern shore.

Benthic animal species composition in the Patapsco River, Elizabeth River, and main Bay was assessed. The Patapsco and Elizabeth Rivers demonstrated a decrease in diversity and an increase in pollution-tolerant species in more impacted areas. The upper Bay (segments CB 1-5) had very high annelid:mollusc ratios when compared to those of the lower Bay segments (CB 6, 7).

During the past decade, oyster spat levels have decreased from pre-1970 averages in most regions of the Bay. The years 1980 and 1981 were good ones for many of the eastern shore tributaries; this was probably due to high salinities.

Oyster landings have decreased since 1880, but have remained relatively stable since 1962. However, there has been a basic change in the spatial distribution of the oyster fishery. Harvest in the western shore has decreased substantially; in the Eastern Shore, it has increased. This may reflect a shift in fishing effort from the western to the eastern side of the Bay.

RANKING OF CHESAPEAKE BAY SEGMENTS ACCORDING TO CURRENT CONDITONS

To summarize the present condition of some important resource variables, and to identify areas of lower resource productivity, a numerical ranking system was developed. Ranking of individual segments with respect to their potential according to their biological resources is useful because it allows one to compare patterns of resource variables with patterns of water quality to identify possible relationships.

In this characterization, segments will be ranked according to their 1978 distribution of SAV, their 1971 to 1980 anadromous and estuarine fishery landings, and according to their 1980 to 1981 oyster spat set. These variables were selected as representing the condition of resources because basic biological components (SAV and spat set) provided two of the variables, while a biological-economic component provided the third (fishery landings). These variables provided appropriate spatial distribution.

Submerged Aquatic Vegetation

Segments are ranked on the SAV scale according to the extent to which their SAV expected habitat is filled. The expected habitat is defined as 50 percent of the potential habitat, the area delineated by the 2 m bathymetric contour. Two meters is the maximum depth at which SAV are likely to be found; test studies showed that, on the average, half of this area could be expected to contain SAV under the best conditions.

Spat Set from 1980 and 1981

Spat set from 1980 and 1981 is used to rank segments because those were recent years of especially successful spat set partly because salinities were optimal for spat. Thus, basins should have been allowed to express their maximum potential unless other factors, such as water quality, prevented their doing so.

Because spat fall is in part a function of salinity, before segments could be compared with each other, it was desirable to normalize against salinity. This was attempted by ascertaining the maximum spat fall observed in each segment and dividing the present spat fall by this maximum.

Anadromous and Estuarine Fishery Landings

On a Bay-wide basis, the decade 1971 to 1980 represents a major shift: while landings of marine spawners continued to increase, landings of anadromous species decreased. The anadromous and estuarine landings provide an indication of the extent to which any tributary contributes to this recent trend.

Because fishery data are not available by segment, basin data are applied to all segments in the basin.

Table 30 summarizes the segment rankings and the total, where at least two variables were available. Where one variable was missing, the mean of the other two was added to provide equivalent values (shown in parentheses).

Trends, as determined by three- and five-year running means, indicated that juvenile indices of marine spawners increased, and those of freshwater spawners decreased. The Potomac River is a notable exception, in which indices of some freshwater spawners increased.

The past ten years indicate a basic change in the pattern of Chesapeake Bay fisheries harvest over the previous period. While freshwater spawners have entered an unprecedented decline, marine spawners, coincidentally, have continued to increase.

REGIONAL ASSESSMENT OF CURRENT CONDITIONS AND TRENDS

Western Shore

The western shore shows a general pattern of decline in certain biological resources. Detrimental changes in species composition and increased biomass of phytoplankton have been shown in the upper Potomac River and may occur in other western shore tributaries if conditions deteriorate. The Potomac River itself has improved in recent years because of advanced waste treatment there (Champ et al. 1981). A result of this improvement has been a more diverse algal population with a decrease in the numbers of filamentous blue-green algae.[11]

Submerged grasses were lost from the maximum-turbidity reaches of all western shore tributaries except the Potomac River before 1970, as well as from the tidal-fresh reaches of the Patuxent, Potomac, and James Rivers. At present, no tidal-fresh reaches sustain substantial SAV. The maximum-turbidity reach of the Potomac River still contains SAV; most lower-estuarine SAV is confined to the York and Potomac Rivers. Submerged aquatic vegetation still does occur in some small western shore tributaries that are less enriched, such as the Magothy and Severn Rivers and Seneca Creek.

Oyster harvest has declined significantly in the Potomac, York, and James Rivers. (The latter is apparently not related to the closure of the river because of Kepone contamination in 1975; the ban on shellfish was very brief.)[12] The fishery appears to have remained constant in the Patuxent River. Spat set has declined significantly (over pre-1970 values) in all but the York River where no significant trend appeared.

Juvenile indices of freshwater spawners have not declined in the Potomac River as extensively as in the other areas surveyed, possibly also a reflection of improved water quality there. Landings of freshwater spawners decreased in all areas of the western shore.

**TABLE 30.
SUMMARY OF SEGMENT RANKS**

	1978 SAV	1971-1980 Landings	1980-1981 Oyster Spat Set	Total
Western Shore				
TF-1	6	5	— (5)	16
RET-1	6	5	— (5)	16
LE-1	6	5	5	16
TF-2	6	5	5	16
RET-2	4	5	— (4)	13
LE-2	6	5	3	14
TF-3	6	2	— (4)	12
RET-3	6	2	6	14
LE-3	6	2	4	12
TF-4	6	2	— (4)	12
RET-4	6	2	6	14
LE-4	5	2	6	13
TF-5	6	3	— (5)	14
RET-5	6	3	— (5)	14
LE-5	6	3	4	13
WT-1	6	6	— (6)	18
WT-2	6	6	— (6)	18
WT-3	6	6	— (6)	18
WT-4	4	6	— (5)	15
WT-5	6	6	— (6)	18
WT-6	3	6	— (4)	13
WT-7	3	3	— (3)	9
WT-8	5	3	6	14
WE-4	4	4	— (4)	12
Main Bay				
CB-1	6	6	— (6)	18
CB-2	6	6	— (6)	18
CB-3	4	6	— (6)	16
CB-4	5	3	3	11
CB-5	5	5	2	12
CB-6	5	4	— (4)	13
CB-7	3	4	— (3)	10
CB-8	—	4	—	—
Eastern Shore				
EE-1	3	2	3	8
EE-2	3	4	2	9
EE-3	4	4	2	10
ET-1	6	6	— (6)	18
ET-2	6	6	— (6)	18
ET-3	6	6	— (6)	18
ET-4	3	4	6	13
ET-5	5	4	— (4)	13
ET-6	6	4	— (5)	15
ET-7	6	4	4	14
ET-8	6	3	5	14
ET-9	3	3	— (3)	9

Main Bay

The upper-main Bay (above the Bay Bridge) shows changes in phytoplankton species composition and biomass that are a reflection of increases in nutrient enrichment. This region no longer supports SAV, has a depressed juvenile finfish index, and is not a source of high fish yields. It shows high annelid:mollusc ratios when compared to those ratios from the lower Bay (segments CB 5-7). These differences, however, are not statistically significant.

Eastern Shore

The Eastern Shore sustains relatively healthy SAV growth in Eastern Bay, the mouths of the Chester and Choptank Rivers, and Tangier Sound. Greater loss has occurred in Pocomoke Sound, Wicomico River, Nanticoke River, Fishing Bay, and Honga River.

Harvests of oysters have increased in the Eastern Shore region, an indication of its relatively good environmental quality. The high 1980 to 1981 spat set was largely confined to the Eastern Shore, particularly Eastern Bay, Tangier Sound, and the Choptank River. With respect to finfish, landings of marine spawners have increased, while those of freshwater spawners declined in all basins except Tangier Sound.

CONCLUSIONS

The two major primary producers of the Bay system, Bay grasses and phytoplankton, have undergone substantial changes. Phytoplankton biomass and species composition, difficult to document because of their ephemeral nature, have shown eutrophication-related changes in the upper Potomac and main Bay (above the Bay Bridge). The abundance of Bay grasses has declined dramatically, particularly in fresher reaches of the estuary.

Species diversity and composition of benthic fauna show pollution-related changes in the Patapsco and Elizabeth Rivers. The upper main Bay has a higher proportion of pollution-tolerant annelids than the lower Bay, but the effect is not strong enough to be clearly distinguishable from response to natural gradients. It appears that the benthos of the main Bay is not stressed enough to show clear pollution-related effects, while the Patapsco and Elizabeth Rivers are sufficiently stressed.

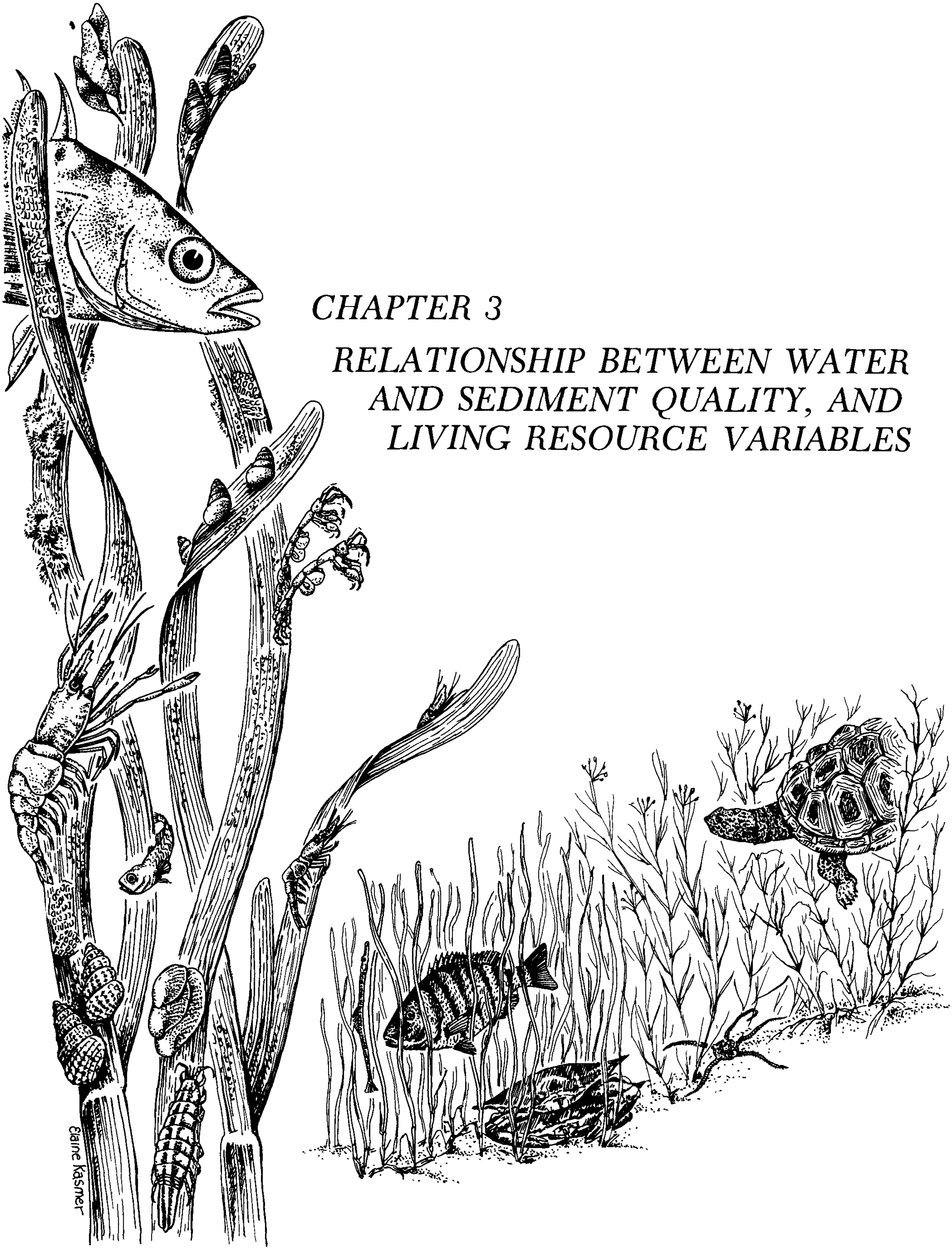
Harvests of oysters have increased on the Eastern Shore, but decreased on the western shore. Spat set shows a similar pattern: while densities have declined everywhere, the recent high spat set (1980 to 1981) was sustained primarily by Eastern Shore areas.

Juvenile indices of freshwater-spawning finfish show recent absence of strong year classes, which is also reflected by declines in harvests. The fishery for marine-spawning finfish continues to be strong. We do not know what the role of natural climatic factors is in determining the health of the fishery. However, the unprecedented nature of the recent decline in freshwater spawners suggests that other factors in addition to climate may be affecting these organisms. Their spawning areas are in close proximity to riverine sources of anthropogenic materials, implying that they are much more likely than other fish to be exposed to such materials at a sensitive stage. It would not be surprising that effects of anthropogenic materials should be expressed first by freshwater-spawning fish, though, in reality, there is little information to compare the relative sensitivity of juvenile marine spawners to anthropogenic materials in nursery areas.

The Eastern Shore generally appears to be the most productive region of the Bay system. This conclusion is supported by the regional comparison; Eastern Bay, and the Choptank and Chester Rivers ranked most favorably. The upper Patuxent, lower James, and Patapsco and Middle Rivers showed poor ranks.

CHAPTER 3

RELATIONSHIP BETWEEN WATER AND SEDIMENT QUALITY, AND LIVING RESOURCE VARIABLES



CHAPTER 3

RELATIONSHIP BETWEEN WATER AND SEDIMENT QUALITY, AND LIVING RESOURCE VARIABLES

SECTION 1
INTRODUCTION

Chapters 1 and 2 summarized information on trends in water and sediment quality, and living resources. This chapter examines the relationships between these observed trends by determining if there is a reasonable potential linkage and if such a relationship can be demonstrated analytically. After a discussion of how various environmental parameters can affect the Bay's ecosystem, and ultimately, its living resources, more detailed information will be presented on finfish, oysters, other benthic organisms, and submerged aquatic vegetation.

ANALYTICAL PROCEDURE

It is important to mention that relatively few data sets were collected at the same time and place. Major climatic variables and physical factors, (e.g., temperature, salinity, and freshwater flow) typically provided the most complete data for time-series analyses. Unfortunately, most historical water quality data and biological data (e.g., finfish juvenile index, spat set, and fishery landings) were not obtained in the same spatial and temporal scales. This created difficulties with many statistical approaches. In some cases, therefore, analyses were made by visual inspection of tabular or graphical material. Such conditions have forced a more conservative interpretation of trends than in cases where statistical requirements are met.

Briefly, for the analytical strategy for this chapter, reasonable potential effects of observed water and sediment quality trends on Bay organisms were postulated, and these possible responses were documented by the use of published material. With these hypotheses as a framework, available data were analyzed to identify where these effects appear to be demonstrated. Incompleteness of much data necessitated more descriptive analyses in some cases. The CBP tried to work with the data in spite of many shortcomings, and we hope that we have worked within acceptable grounds of credibility.

THE ECOSYSTEM PERSPECTIVE

The Trophic Web

A better understanding of how environmental perturbations affect living resources can be achieved if Chesapeake Bay is examined from an ecosystem perspective. The system as a whole acts to sustain the productivity of the estuary, but can also transfer effects of environmental changes to many levels.

As discussed in Chapter 1, Chesapeake Bay, organisms inhabiting it, and the processes that link them constitute the Bay ecosystem. Material and energy are transferred from one portion of the system to another through a trophic web (an important concept to understand in evaluating the effect of environmental influences on resources). These relationships can be detailed conceptually by the use of a trophic diagram, showing major components and flows. Figure 44 is one such diagram, representing a simplified plankton-based food web leading to finfish. Theoretically, similar diagrams could be constructed for all components of the Bay ecosystem (Green 1978, Mackiernan et al. 1982). The actual species represented by the boxes, the direction and relative importance of the pathways, and the magnitude of flows vary with

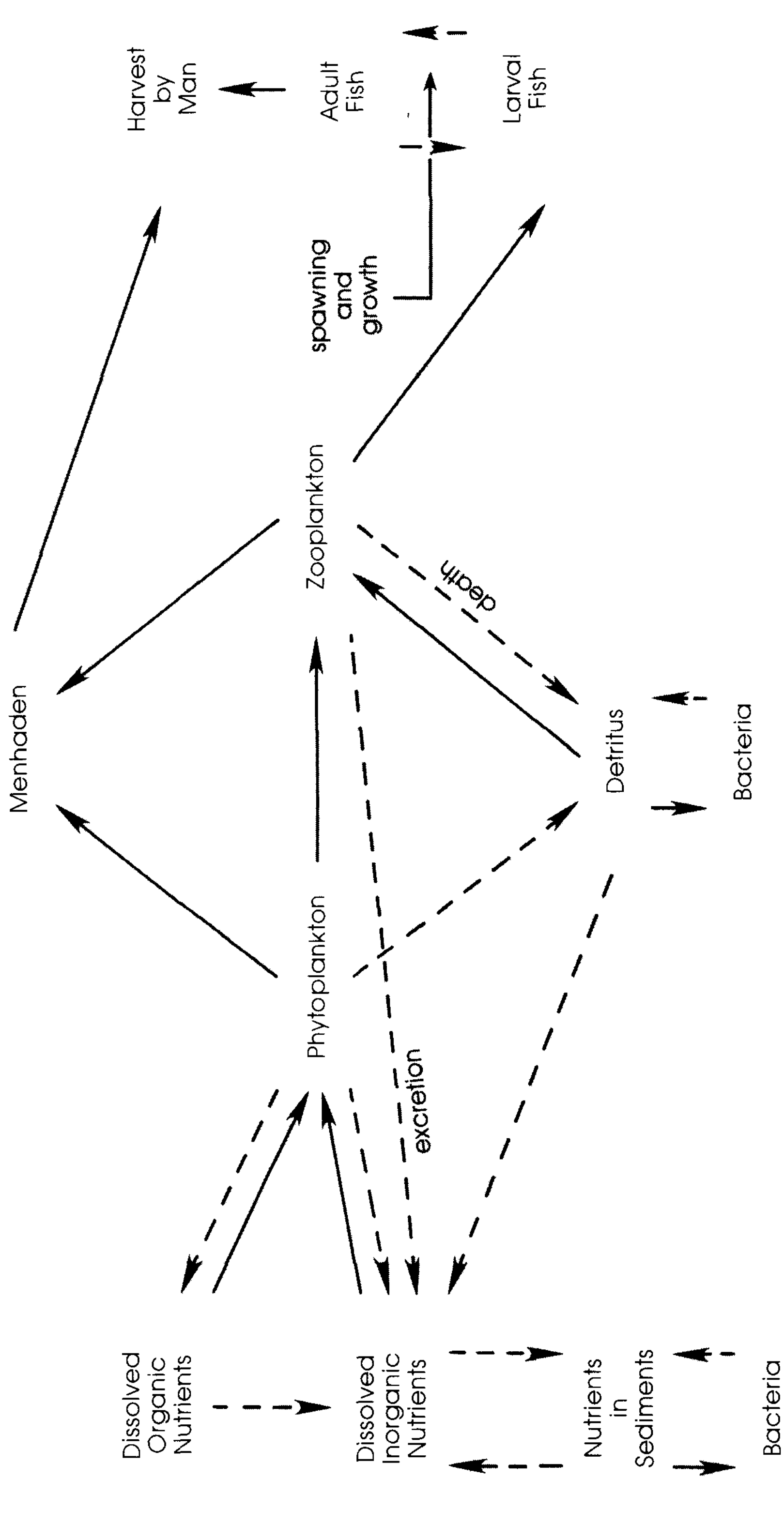


FIGURE 44. Simplified trophic diagram showing a pathway from phytoplankton to finfish.

season, location, and physical aspects of the environment. The system, in a sense, represents a series of switches that can shunt material or energy flows in various directions. A change (in species makeup, for example) at one level can impact other portions of the ecosystem by altering direction or size of these flows.

Because of such interrelationships, organisms may be affected by factors acting on other trophic levels. This is a key point — many hypothetical impacts of water quality discussed in this chapter do not act on the resource species directly, but on some ecosystem component upon which it depends. For example, even though an environmental perturbation does not directly impact a resource, that resource could decline if survival tolerances of a key food organism were exceeded. Similarly, changes in population of a predator, parasite, or competitor could affect the abundance of a resource species.

RELATIONSHIPS BETWEEN WATER QUALITY AND LIVING RESOURCES

Anthropogenic Impacts on Resources

In general, organisms can be considered as responding to a number of interacting environmental variables. Some of these are natural, usually related to climate, hydrography, or biological interactions; others may be primarily anthropogenic, coming from many human activities. Estuaries, particularly those of the temperate zone, are inherently variable environments; organisms inhabiting them are subject to a wide annual range of air and water temperatures, freshwater inflow, wind, light, and circulation. It has been demonstrated that much of the annual variability in spawning success of fish and shellfish, for example, is influenced by natural variation of flow, temperature, and salinity (Setzler et al. 1980, Ulanowicz et al. 1980). Although these may be altered by man's actions (e.g., changes in timing and extent of freshwater inflow due to dams), most are within the range of tolerance of estuarine organisms.

Anthropogenic impacts may increase natural environmental variability, or may introduce non-natural stresses outside the adaptive repertoire of estuarine organisms (e.g., toxic chemicals). Trends in water quality identified in Chapter 1 with a major anthropogenic component, are:

1. Nutrient enrichment, particularly of the upper Bay and western shore tributaries. Increased eutrophication is reflected in increased particulate carbon and turbidity. In conjunction with summer stratification of the water column, this can lead to three more trends in water quality with an anthropogenic component;
2. Extended periods of low dissolved oxygen or even complete anoxia in deeper water of the upper and mid-Chesapeake;
3. Bay sediments that are anthropogenically-enriched with large amounts of heavy metals, organic chemicals, and similar toxic materials; and
4. Levels of toxic materials in the water column that sometimes exceed the published EPA ambient water quality criteria, particularly in the upper Bay and western tributaries.

DETECTING EFFECTS OF ANTHROPOGENIC STRESS ON LIVING RESOURCES

Separating effects from natural, as opposed to anthropogenic, causes, or even showing any actual cause and effect relationship, is no trivial task. A major problem is that the complexities of ecosystem function, coupled with the natural variability of organism distribution and abundance, make it difficult to identify changes and to ascribe particular causes to the observed effects (Wolfe et al. 1982). Sindermann (1980) describes in some detail the difficulty of isolating and quantifying pollution effects on resource species (as distinct from effects of natural environmental variations). Paraphrasing Sindermann (1980), chemical pollutants cause stress and death in individual marine animals; this can be easily demonstrated, and has been done repeatedly. Descriptions of lethal and sublethal effects of heavy metals, petroleum compounds, and halogenated hydrocarbons abound in the experimental literature. Whether or not stress from chemical pollutants can have significant quantifiable effects on resource species abundance (apart from localized effects in severely contaminated coastal and estuarine zones) is much more difficult to demonstrate and has not been documented satisfactorily.

A recent review of ecological stress in the New York Bight (Mayer 1982) supports this view, for even in an area as heavily impacted as the New York Bight, it is difficult to demonstrate pollution-induced changes in the growth and distribution of populations of plankton (Lee et al. 1982), invertebrate communities (Wolfe et al. 1982), fishes (Sindermann et al. 1982), and species or groups (Boesch 1977a, 1982; Franz 1982), except in the most heavily-impacted areas.

Changes to some populations have been demonstrated in certain areas including: elimination of sensitive species or groups (Boesch 1977a); changes in species diversity of the community (Jacobs 1975); increase in pathologies (tumors, lesions), or incidence of disease (Sindermann et al. 1982). Laboratory studies, or use of microcosm systems, where one variable at a time can be manipulated, can help link observed cause and perceived effect. Examples of the latter are the Controlled Ecosystems Pollution Study (CEPEX) conducted on the west coast, and the Marine Ecosystems Research Laboratory (MERL) studies currently underway in Rhode Island. Coupled with improved monitoring and field experimental procedures, designed to answer specific questions, such studies can strengthen the understanding of cause and effect relationships. Nevertheless, complete understanding is probably not possible (or feasible). In that light, Sindermann (1980) argues that "to insist on demonstration of easily discernible effects on overall species abundance is to establish too harsh a criterion of pollution damage. A much more acceptable concept is that the effects of pollution, clearly demonstrated on even a single individual or a local population, must be considered a cause for management action to protect the total population—just as is the case with humans."

SECTION 2
POTENTIAL IMPACTS OF WATER AND SEDIMENT QUALITY ON LIVING RESOURCES

In this section reasonable potential impacts (called hypothetical impacts here) of documented trends in water and sediment quality upon the Bay's living resources will be formulated. The impacts of natural variables, such as freshwater inflow and temperature, insofar as they appear to relate to observed trends in living resources, will also be discussed. Other factors will be addressed where appropriate (e.g., fishing pressure, habitat loss due to dams), although these were not part of the data analysis. These hypothetical impacts will be tested by various analytical procedures, depending on strength and availability of the data. Finally, those relationships which seem supported by available data and analyses, as well as those where relationships could not be demonstrated, will be summarized.

It is anticipated that in many cases strong relationships cannot be supported by the data; the signal is lost in the noise of natural variability. However, even the framing of reasonable hypotheses is a useful exercise, because this allows researchers and managers to focus on areas where effects could be expected, and where monitoring or research efforts may yield the most useful information.

NUTRIENT ENRICHMENT

Increases in levels of, primarily, forms of phosphorus and nitrogen may increase standing crops of phytoplankton (Heinle et al. 1980) (Chapter 1). Provided the phytoplankton species composition matches nutritional requirements of larvae of commercial species (or zooplankton which feed these species), this early stage of eutrophication may actually benefit productivity in portions of the system (Ukeles 1971; Sharp et al. 1982).

a. As nutrient enrichment progresses, however, chlorophyll concentrations increase, as do particulate carbon and turbidity. Nutrient enrichment during warmer temperatures results in greater fluctuation in DO concentrations in surface water, and to extended periods of near anoxia in bottom water (also discussed below) (Heinle et al. 1980; Taft et al. 1980).

   Hypothetical Impact

   Effects of these changes could include: shading of SAV by increased phytoplankton or epiphyte growth, leading to decline in SAV abundance (Twilley et al. 1981); and/or organic enrichment of sediments leading to changes in benthic communities such as increased dominance by detrivores (Bascom 1982).

b. Shifts in phytoplankton species composition may occur, possibly resulting in blooms of undesirable species (Ryther 1954).

   Hypothetical Impact

   Effects of these changes could include: increased dominance of blue-green algae in tidal-fresh areas, or flagellates in saline regions (Thomas 1972); and/or alteration of biomass and composition of zooplankton or filter-feeding communities, with ultimate impacts on larval fish (Greve and Parsons 1977) or shellfish (Ryther 1954).

c. The extent and duration of low DO in deeper water has apparently increased in recent decades (Taft et al. 1980) (Appendix B).

Hypothetical Impact

Impacts of this change could include: elimination of natural benthic community assemblages in deeper water (or replacement with ephemeral opportunistic assemblages) (Mountford et al. 1977); and/or decreased habitat for oysters and other commercially important shellfish (Haven et al. 1978). Increased incidence of shellfish mortality at the boundaries of the impacted area might be expected.[13] Finally, habitat for fish, particularly demersal species such as sciaenids, is reduced; this could be reflected in declines in abundance or condition.[14]

CONTAMINATION WITH TOXICANTS

a. The sediments are enriched with metals and contain hundreds of organic compounds. High levels of heavy metals have been measured in Chesapeake Bay sediments; this anthropogenic enrichment may reach ten times or more background (natural) values. Similarly, over 300 anthropogenic organic chemicals have been isolated from Bay sediments; most are toxic (Bieri et al. 1982a).

Hypothetical Impact

Impact of these toxicants could include: changes in benthic community structure or abundance; elimination of sensitive species and replacement by a resistant fauna (Boesch 1973); bioaccumulation of toxic materials in tissues of shellfish, other benthic species, or submerged aquatic vegetation (O'Connor and Rachlin 1982); and/or change in physiology, reproductive success, or behavior of benthic dwelling species (Schaffner and Diaz 1982; Phelps et al., in prep.). There is also the possibility of further accumulation of these materials in the tissues of benthic-feeding fish and waterfowl.

b. The water column may be contaminated with toxic materials. Levels of heavy metals, certain organic materials (chiefly pesticides), and total residual chlorine have been measured at levels which exceed the EPA ambient water quality criteria. As these criteria have been developed to protect the integrity of aquatic ecosystems, measured concentrations above these levels may represent reason for concern if exposures are of sufficient duration.

Hypothetical Impact

Impacts of these toxicants could include: mortality or stress to exposed organisms, particularly sensitive larval stages (e.g., oysters, larval fish). This might be reflected in reduced recruitment to impacted populations, lowered resistance to disease or other stresses, or even gross tissue abnormalities (lesions, etc.) (Calabrese et al. 1982). Bioaccumulation of material in tissues of fish, filter-feeding benthic species, or even plankton or SAV may result (O'Connor and Rachlin 1982). This could have implications for higher trophic levels as well.

IMPACTS OF NATURAL VARIABLES

It is useful to discuss the effects of major natural variables to gain a perspective on the possible impacts of anthropogenic factors. It is not unreasonable that variability due to one, or a combination of man-induced stresses, may represent only a small fraction of that caused by natural perturbations. However, even an incremental increase in the mortality of a larval fish, for example, or a small percentage decline in photosynthetically available light to SAV may result in an eventual severe impact to the resource.

Although estuarine organisms are to a great extent tolerant of an estuary's naturally-variable environment, much of the year to year differences in distribution and abundance can be accounted for by variations in freshwater inflow, salinity, air and water temperature, wind, and other parameters. The influences of stochastic processes, particularly of extreme events such as storms or droughts, can be considerable (Boesch et al. 1976). Finally, biological interactions such as predation, competition, or disease influence the distribution of many species (Virnstein 1977).

An example of the effects of natural variables on a resource is the striped bass. Success of striped bass year classes has been positively correlated with colder than normal winters and high spring runoff (Setzler et al. 1980). Cold winters are thought to contribute increased detrital loads in spawning areas because of ice scouring marshes and shorelines (Heinle et al. 1976). The detritus provides additional food for zooplankton, which in turn are fed upon by the larval fish. Increased runoff carries dissolved nutrients that support the phytoplankton and larval fish food chain; it also physically extends the area of the spawning and hatching grounds.

In contrast, blue crabs spawn near the mouth of Chesapeake Bay, while menhaden spawn 16 to 25 km[15] from shore over the continental shelf. Larvae of both species drift with shelf currents and are highly dependent on these currents to return them at the appropriate time to the mouths of suitable estuaries. When shelf circulation—driven by climate and wind patterns—is toward the west or northwest during this critical period, there is a high probability that these and other marine-spawned species will be returned to the estuary, resulting in a successful year class (Nelson 1979).[16] Within the Bay, freshwater outflow drives upstream flow of saline water at depth, assisting these species in reaching low salinity nursery areas.

SECTION 3
SUBMERGED AQUATIC VEGETATION AND WATER QUALITY

INTRODUCTION

Submerged aquatic vegetation (SAV) was one of three major problem areas addressed by CBP research. For this reason, an integrated picture of factors impacting SAV exists; these factors can be reasonably implicated in recent vegetation declines. The two major anthropogenic factors that were hypothesized in Section 2 as affecting SAV were:

1. Impacts of toxic materials, particularly herbicides, the use of which has increased greatly during the period of maximum SAV decline;
2. Reduction in available light to the plants, either by an increase in turbidity of the water column or an increased growth of epiphytes on the plant leaves (or a combination of these). Turbidity changes were hypothesized to be due to increased suspended material and/or phytoplankton growth, enhanced by nutrient enrichment.

In this section, results of CBP-supported research in these areas are discussed, and an attempt to relate field observations to research results is made.

EVIDENCE FROM RESEARCH

Herbicides

The major toxic materials considered in assessing anthropogenic factors on SAV decline were agricultural herbicides. The use of these chemicals has increased significantly in the recent decade, and it was hypothesized that runoff from agricultural fields might deliver enough of these toxicants to impact SAV in Bay waters (Stevenson and Confer 1978).

Details of the research projects and results supported by the CBP are contained in the 1982 EPA report, *Chesapeake Bay Program Technical Studies: A Synthesis*. In general, projects focused on the following areas: fate and transport of herbicides within the estuary, action of the herbicide on SAV, and potential community effects of herbicide exposure. The herbicides atrazine and linuron — most widely used — were the primary types examined.

In general, maximum herbicide concentrations observed did not exceed 1 ppb in the estuarine portions of the lower Bay and 5 ppb in the upper Bay (Hershner et al. 1982, Boynton et al. 1983). Ephemeral concentrations of up to 20 ppb were observed in some estuarine areas; these persisted for two to eight hours at most. Much higher levels (up to 100 ppb) were observed after rainstorm events in small tidal tributaries adjacent to fields.

Plants exposed to atrazine and linuron show a rapid reduction in photosynthesis, followed (in the case of low herbicide concentrations) by gradual recovery. The higher the initial exposure, the longer the recovery time. Plants exposed to 10 to 15 ppb atrazine took two to five weeks to regain a photosynthetic rate comparable to controls (Cunningham et al. 1982). Plants treated with 50 ppb atrazine and above showed severe and persistent loss of productivity.

There are four conclusions from the herbicide research:

- Because herbicide concentrations did not exceed 20 ppb in estuarine waters and rarely exceeded 5 ppb, it is likely that herbicides are not the primary cause of SAV decline in most areas of the Bay.

- Herbicides may be a major impact in small tributaries directly adjacent to herbicide-treated fields.
- Herbicides degrade relatively rapidly and do not appear to build up in sediments. However, relatively little is known about phytotoxicity of major herbicide degradation products (Jones et al. 1982).
- Very low (ambient) concentrations of herbicide (less than 10 ppb) are predicted to cause a reduction of 10 to 20 percent of SAV photosynthesis. If exposure intervals are less than SAV recovery time, this could lead to increases in the stressed condition of the plants. In particular, light limitation could act in concert to adversely impact plant populations (Kemp et al. 1982c).

Light Limitation

Research on light limitation of SAV focussed on several major areas: characteristics of light in Chesapeake Bay; factors contributing to light attenuation; and response of SAV communities to changes in light regime. Many details are contained within the above-cited Synthesis Report.

In general, light falling on the SAV bed can be attenuated and scattered by suspended particulate material (both living and dead), as well as by dissolved and colloidal materials of organic origin collectively termed Gelbstoff (Wetzel et al. 1982). In addition, epibiota and sediment on the SAV leaves can further reduce the amount of photosynthetically active radiation (PAR) reaching the plants. Sources of suspended material may include sediments carried in river runoff, material from local shore erosion resuspended from the bottom, or of organic origin (phytoplankton and detritus, in particular) (Biggs 1970).

There is evidence that the quantity of light has been reduced in recent years, and that quality (that is, the amount in the critical blue and red spectral regions) has also declined at a lower Bay study site (Wetzel et al. 1982). Spectral measurements for the upper Bay are available only for channel areas; however, the general pattern is for selective light attenuation in the blue region of the spectrum, with a shift to orange as the most penetrating wavelength (Champ et al. 1980). Increases in chlorophyll pigments, due to greater phytoplankton biomass, have also occurred in the upper and mid-Bay, as well as in most major tributaries. This is linked to increases in nutrient enrichment in the corresponding areas of the Chesapeake (Heinle et al. 1980). As discussed in Chapter 1, there has been an increase in summer turbidity in recent years for many areas of the Bay.

In addition, nutrient enrichment has been hypothesized as leading to increased fouling of SAV leaves with epibiota (epiphytes and certain particle-filtering animals such as bryozoans). Epiphytes can significantly reduce the amount of PAR reaching the leaf surface: less than 10 percent of incoming radiation was transmitted through dense epiphytic cover on older *Zostera* blades (Borum and Wuim-Andersen 1980).

These conclusions from the light-limitation research support the hypotheses proposed in Section 2 on nutrient enrichment:

- Studies indicate a reduction in the quantity for the upper and lower Bay and in the quality of available light. For the growing season, there appears to be a progressive increase in light attenuation in the PAR part of the spectrum. There is a seasonal component to atteñuation, although it is highly variable. Vertical PAR attenuation increases in the spring, affecting the initiation of growth for most SAV species.
- *In situ* studies indicate that Bay plant communities are generally operating under conditions of light limitation. No apparent light saturation is reached in the upper Bay, nor did *Zostera* communities in the lower Bay exhibit light saturation.
- Decline of SAV over the past several decades has in general followed a "down-estuary" pattern, corresponding to a pattern of nutrient enrichment.
- Microcosm and mesocosm (pond) experiments indicate that nutrient loading leads to a progressive enhancement of phytoplankton biomass, seston, and epiphytes (Twilley et al. 1981).
- The combination of water column attenuation and epiphytic shading was sufficient to markedly reduce SAV photosynthesis.

Loss of SAV occurred in ponds exposed to highest nutrient loads.

- Preliminary evidence suggests that the inability of light-stressed plants to compensate for respiration may lead to reduced energy storage for over-wintering and spring regrowth (Twilley et al. 1982).
- Reduction of grazers by various natural or anthropogenic perturbations may allow excessive fouling of SAV leaves by periphyton (Orth et al. 1983).

Ability to Recolonize

Transplantation experiments have indicated that survival of SAV is possible in some areas now denuded of plants[17] (Orth et al. 1982). These observations emphasize the importance of availability of seeds or propagules in determining whether SAV can recolonize former habitat. If no nearby source of vegetation exists, and if sediments no longer contain viable seeds, tubers, or rhizomes, then physical transplanting may be necessary to restore some beds. Impacts of grazers or "users" of SAV and present suitability of former habitat must be considered in these cases (Orth et al. 1982)

EVIDENCE FROM FIELD OBSERVATIONS

Research indicates that light limitation, possibly related to nutrient enrichment, is the major cause of SAV declines in most areas of the Bay. One might expect to see correlations between SAV abundance and such water quality parameters as nutrient concentrations, chlorophyll *a*, turbidity, and DO (through their relationship to phytoplankton biomass). Observations made in the field suggest that some of these correlations exist.

Biostratigraphic evidence from sediment cores shows that the decline of SAV, the reduced abundance of epiphytic diatoms, and the decrease in diatom diversity, along with the simultaneous increase of sedimentation rate and the number of planktonic diatoms, are related to anthropogenic impacts (Brush and Davis 1982). Clearing of land for agriculture in the 1800's and increased sewer loadings were apparently major contributing factors. For example, since the 1600's, Furnace Bay (off the Susquehanna Flats) has changed from a clear, SAV-dominated embayment to a turbid phytoplankton-dominated water body.

Comparison of maps of current SAV distribution, and of areas of greatest historic decline (Figures 45a and 45b) with a map of Chesapeake Bay nutrient status (Figure 46) reveals striking correspondence. Submerged aquatic vegetation now occurs primarily in those areas characterized with low or moderate amounts of nutrients: Eastern Bay, lower Choptank, Tangier Sound, and the lower Bay main stem, as well as a few small western shore tributaries (Magothy and Severn Rivers). Greatest loss has occurred in areas with the highest levels of nutrients: Susquehanna Flats, upper and mid-Bay, major western shore tributaries, and upper reaches of Eastern Shore tributaries (Orth and Moore 1981).

Trend Comparison

Comparison of graphs of SAV decline with similar water quality trend information reveals a number of apparent correlations. Such comparisons suggest, but do not confirm, relationships between SAV and changes in water quality variables. For example, in CB-1 a negative correlation is apparent with SAV against both nitrate (Figure 47a) and annual chlorophyll *a* concentrations. In EE-1, Eastern Bay, SAV decline appears to be related to increases in total chlorophyll *a* levels (Figure 47b).

Regression Analysis

Because SAV declines are hypothesized to be related to some water quality factors, certain variables were tested (by correlation analysis) against vegetation abundance in those Chesapeake Bay segments where sufficient data existed. A parametric test (Pearson's correlation coefficient) and a non-parametric test (Spearman's rank correlation coefficient) were used. The 11-year data set from the Maryland DNR and the USFWS on SAV abundance was used as an estimator of vegetation abundance. Among the water quality variables screened were: TN, nitrate, TP, dissolved inorganic phosphorus, chlorophyll *a*, turbidity, Secchi depth, DO, salinity, temperature, and pH.

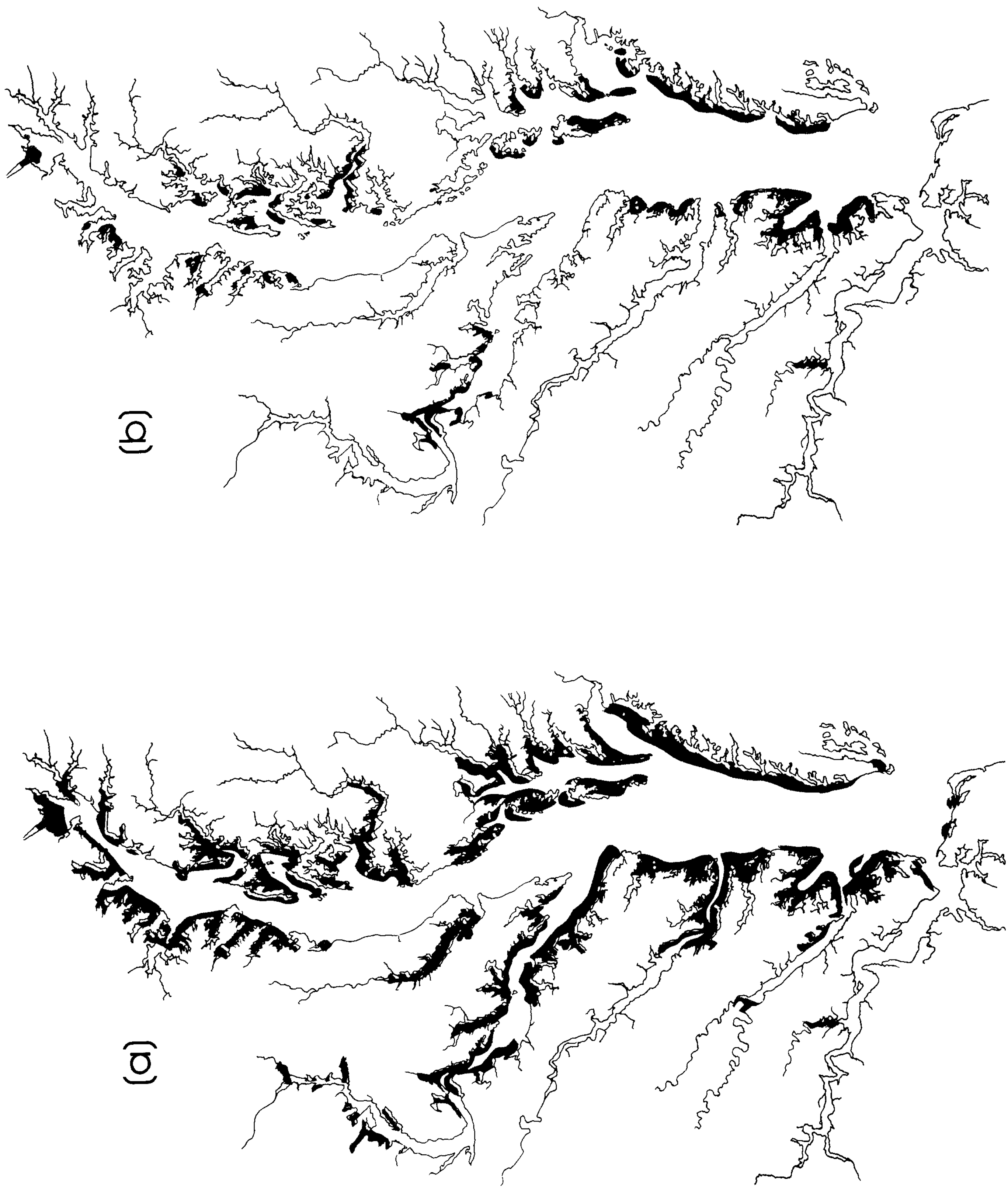


FIGURE 45. General area of SAV distribution in (a) 1965 and (b) 1980.

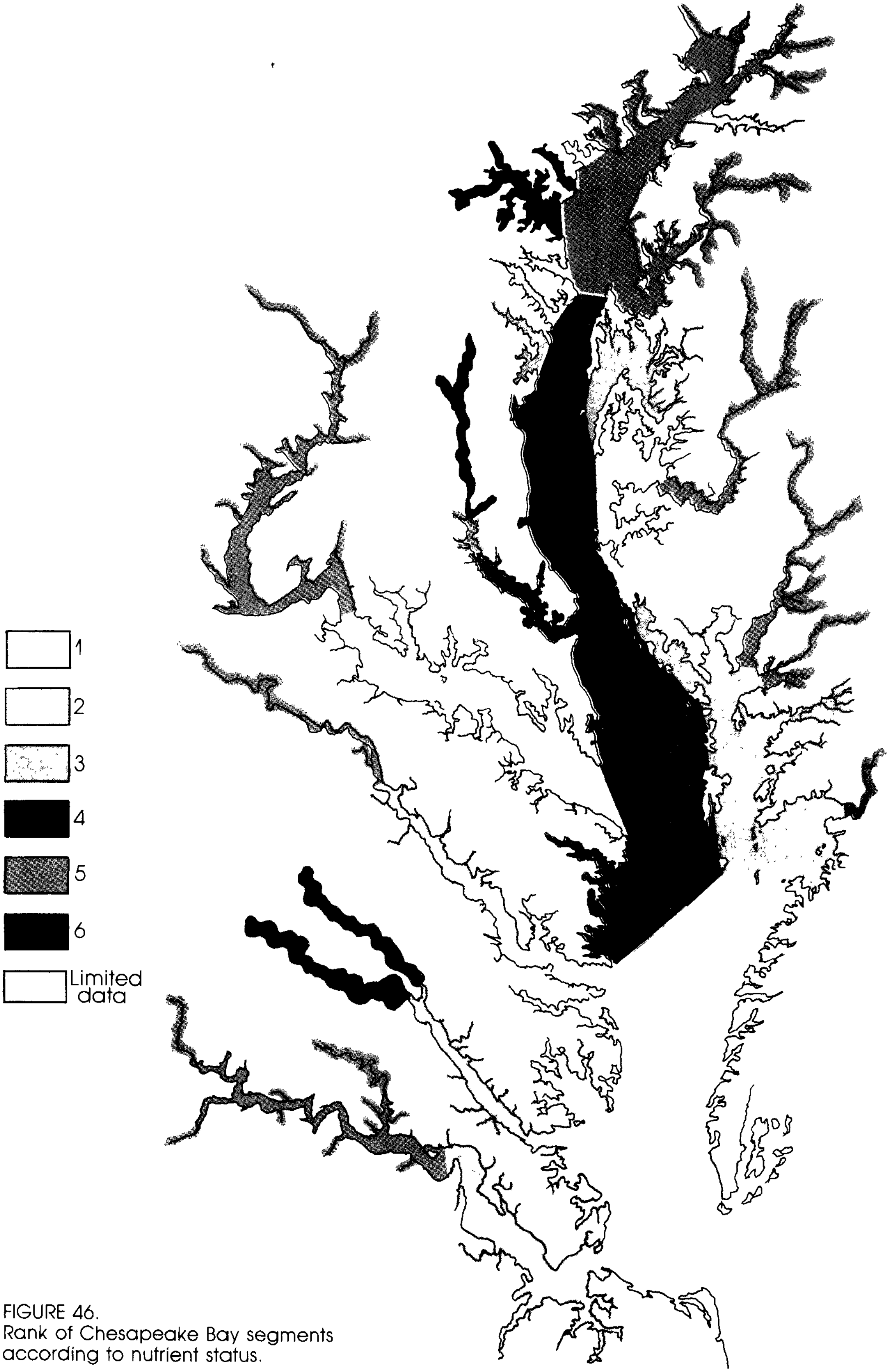


FIGURE 46.
Rank of Chesapeake Bay segments according to nutrient status.

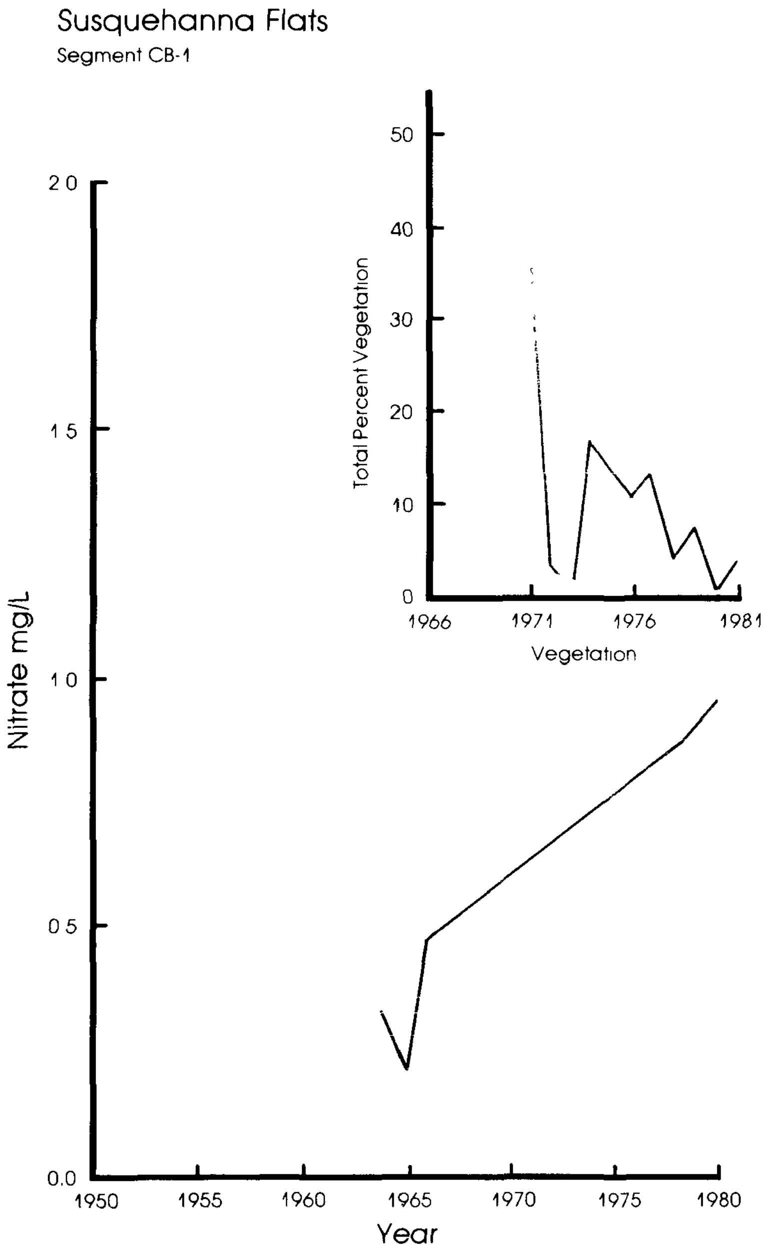


FIGURE 47a. Percent total vegetation compared with mean total nitrate in the Susquehanna Flats, CB-1 (SAV data from 1971 to 1982).

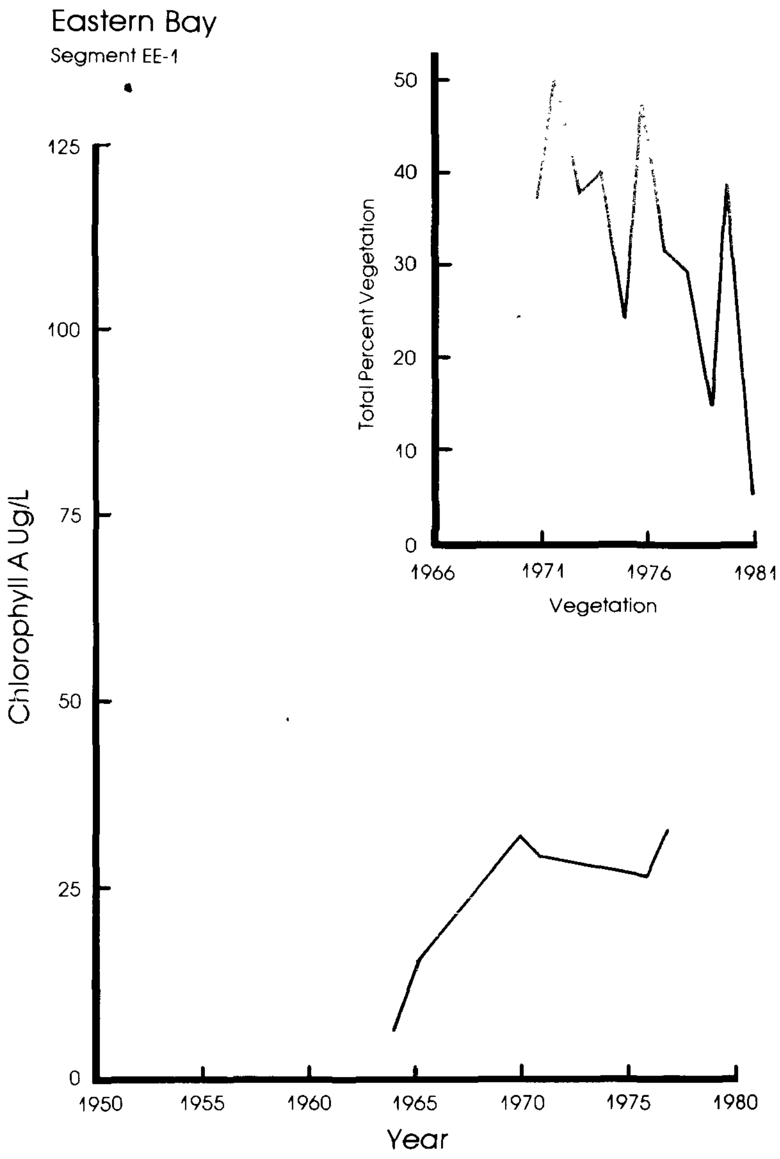


FIGURE 47b. Percent total vegatation compared with mean chlorophyll a of previous year in EE-1, 1971 to 1981.

TABLE 31.
SUMMARY OF CORRELATION ANALYSIS OF WATER QUALITY VARIABLES AGAINST SAV. TABLE OF STATISTICS IS LOCATED IN APPENDIX D, SECTION 2
+ = POSITIVE CORRELATIONS; − = NEGATIVE CORRELATIONS; DIGIT INDICATES NUMBER OF CORRELATIONS WITH EACH VARIABLE IN EACH SEGMENT

SEGMENT	VARIABLE																					
	TN	95-TN	NO_3	95-NO_3	TP	95-TP	IPF	95-IPF	Turbid.	95-Turbid.	Secchi	95-Secchi	Chl *a*	95-Chl *a*	DO	95-DO	Salinity	95-Salinity	Temp.	95-Temp.	pH	95-pH
CB-1	1−	4−													1−		1+	1+	2−		2+	
CB-2	1−																					
CB-3		4−	2−				2−															
CB-4	1+								1−													4−
CB-5	4−	10−			1+	2+			3−		1+				1+	2+		1+	2−	2−		
EE-1		2−			1−											1−	2−				1+	
EE-2				1−														1−			1+	
EE-3			3−						6−	1−				3+	4+		2+		1−		6+	
ET-4	1−		3−	4−	1+	4+					2−				3−	6−		3−	4+ 1−	4+	1+	
ET-5	2+	2−			1+	1+	2+ 1−	1−	2+ 1−	2+			3−		1+ 2−		1−	2−	2+		2+	
LE-1					2+		1+						5+		1−				1−			
WT-2							1−	3−		2+			1−		2−							2−
WT-3		1−							1−	1−			1−									
WT-5	2−			1−							2+	1+		3−	6+		1+				1+	
Subtotals by geographic areas.																						
Main Bay	6−	14− 1+	6−		1+	2+	2−		4−		1+				1− 1+	2+	2+	2+	4−	2−	2+	4−
Eastern Shore	1−	2−	9−	4−	1− 1+	5+	2− 2+	1−	7− 2+	1− 2+	2−		3− 1+	3+	5− 5+	7−	3− 2+	6−	2− 5+	4+	10+	
Western Shore	2−	1−		1−	2+		1− 1+	3−	1−	2+	2+	1+	2− 5+	3−	3− 6+		2− 1+		1−		1+	2−
Grand Total	9− 3+	17−	15−	5−	1− 5+	7+	5− 3+	4−	12− 2+	1− 4+	2− 3+	1+	5− 6+	3− 3+	9− 12+	7− 2+	5− 5+	6− 2+	7− 5+	2− 4+	13+	6−

These were compared to total percent vegetation using annual, spring, summer means, and 95th percentile values for each variable in each segment. Data were tested using direct comparison of a particular year's SAV data against water quality variables of that year (e.g., 1971 to 1971, 1972 to 1972). In addition, under the hypothesis that the growing conditions of a previous year might have a significant effect on SAV success the next growing season, vegetation data were tested against water quality variables for the preceding year (e.g., 1971 SAV against 1970 variables).

Summary results of the correlations are presented in Table 31. Complete results of the analyses are given in Appendix D, Section 2. There are differences as to which variables correlate with SAV in the various segments; also, a number of areas show few significant relationships. However, some generalities can be made. The greatest number of significant correlations occur between SAV and nutrients, particularly nitrogen and IPF. Correlations are, with few exceptions, negative between SAV and TN, NO_3 and IPF, and positive between SAV and TP. Correlations with turbidity are usually negative; those with Secchi depth are generally positive. There is little consistency in correlations with DO. There are negative correlations with chlorophyll *a* levels of the preceding year in most segments; however, positive correlations exist when chlorophyll levels of the current year are assessed. This may be due to the peak of chlorophyll *a* coming after initial growth of SAV within a particular year. Salinity showed positive correlations with SAV in main Bay segments, but generally negative in Eastern Shore areas. Temperature, however, demonstrates negative correlations with SAV in main Bay segments and positive values in Eastern Shore areas. The water quality variable, pH, always correlates positively with SAV, but the 95th percentile of pH values (i.e., very high pH values) shows negative correlations.

It should be emphasized that this analysis only identifies correlations (i.e., correspondence of trends) in SAV and water quality variables. It does not show cause and effect. However, the majority of correlations identified are consistent with hypotheses presented in the preceding sections.

Multiple Regression Analysis

To achieve better insight into the contribution of water quality variables to SAV abundance, step wise least squares multiple regression analysis was used to identify factors that best explained observed vegetation trends.

For the first trials, all of the previously listed water quality variables were included. However, a low number of observations of certain variables (i.e., $n < 10$) in some segments necessitated their elimination before regression equations could be successfully derived.

Complete results of the preliminary analyses are given in Appendix D. There is relatively little consistency from segment to segment or from season to season among the major independent variables in the equations. It is not unexpected that SAV responses should differ from area to area because different SAV species are involved; also areal trends in water quality vary. In addition, the selection of variables affects the outcome of the analysis.

As these analyses were, by necessity, limited by the 11-year SAV data base from the MD DNR and the USFWS, they are, at best, suggestive rather than predictive. With small data sets, it is unlikely that an independent variable beyond the first or second have predictive capability.[18] Therefore, these results should be viewed with some caution and should be considered preliminary. In addition to the above caveats, it is admittedly difficult to identify or eliminate spurious correlations, or those where a variable represents a surrogate or analog of the actual (but not tested) predictor. Also, in some segments, paucity of water quality leads to low degrees of freedom, weakening the statistical validity of the resulting equation.

In CB 1-3, most SAV variability can be explained by a negative correlation with annual or spring NO_3 concentrations. In CB-4, a negative correlation exists with summer NO_3, but a positive one exists with spring TP. Annual TN was the major (negative) predictor in CB-5.

Results in eastern embayments and tributaries were less consistent, possibly due to smaller data sets. Major predictors are TP (negative, EE-1; positive, ET-4), summer TN and turbidity (negative, EE-3), and DO (positive, ET-5).

In western tributaries, major predictors are turbidity (negative, WT-2), Secchi depth (positive, WT-5 and WT-6), TN (negative, WT-5; positive, WT-2 and WT-6), and NO_3 (negative, WT-2, WT-3, and WT-5).

In general, SAV seems to respond negatively to nutrients, particularly to TN and NO_3 concentrations. This is, however, not exclusively true. The multivariable equations are only suggestive, not conclusive. It should be emphasized that none of these relationships are causative; SAV could be responding to a non-tested variable that co-occurs with the tested predictors.

Bay-wide Comparison of Segments

The preceding linear and multiple regression analyses serve to identify water quality factors that may be affecting SAV abundance within each segment. To determine if any factor or factors could be acting consistently on all segments, a non-parametric test, Spearman's rank correlation coefficient, was used. Total percent vegetation within each segment was compared to a number of water quality variables, including TN, NO_3, NH_3, TP,

TABLE 32.
SPEARMAN RANK CORRELATION COEFFICIENT RESULTS FOR SUBMERGED AQUATIC VEGETATION AGAINST WATER QUALITY VARIABLES. r_s=CORRELATION COEFFICIENT, ALPHA = LEVEL OF PROBABILITY THAT r_s IS NOT EQUAL TO ZERO.

x	y	r_s	Alpha
% SAV	− $\bar{x}$ annual TN	0.70	0.001
% SAV	− $\bar{x}$ annual TN of preceding year	0.70	0.001
5 yr $\bar{x}$ % SAV	− 5 yr $\bar{x}$ TN	0.41	0.05
% SAV	− $\bar{x}$ annual NO_3	0.08	N.S.
% SAV	− $\bar{x}$ annual NO_3 of preceding year	0.43	0.025
% SAV	− $\bar{x}$ summer TN	0.11	N.S.
% SAV	− $\bar{x}$ maximum summer TN	−0.09	N.S.
5 yr $\bar{x}$ % SAV	− 5 yr $\bar{x}$ summer TN	−0.11	N.S.
% SAV	+ $\bar{x}$ annual TP	0.10	N.S.
% SAV	− $\bar{x}$ annual TP	0.08	N.S.
% SAV	+ $\bar{x}$ annual TP of preceding year	0.03	N.S
% SAV	+ maximum annual TP	0.08	N.S.
% SAV	+ maximum annual TP of preceding year	0.06	N.S.
% SAV	− $\bar{x}$ annual chl *a*	0.30	0.10
% SAV	+ $\bar{x}$ annual chl *a*	0.16	N.S.
% SAV	− $\bar{x}$ annual chl *a* of preceding year	0.19	N.S.
% SAV	+ $\bar{x}$ annual chl *a* of preceding year	0.13	N.S.
% SAV	− annual maximum chl *a*	0.37	0.05
% SAV	− annual max. chl *a* of preceding year	0.25	N.S.
% SAV	+ annual maximum chl *a*	0.20	N.S.
% SAV	+ annual dissolved oxygen	0.37	N.S.

DO, and chlorophyll *a*. Annual means, five-year means, and maximums of various parameters were tested. The Maryland DNR and USFWS SAV data from 22 Maryland Bay segments were used. Results are given in Table 32.

Percent SAV was compared for possible positive or negative relationships with nutrients, chlorophyll *a*, and DO. Significant negative relationships were identified between percent SAV and mean annual TN of both the current and preceeding year ($p \leq 0.001$). In addition, if five-year means of SAV are compared to five-year means of TN, they are significant at the 95 percent level. There was no apparent relationship between SAV and annual NO_3, but a significant negative correlation was observed between SAV and NO_3 of the preceeding year (alpha = 0.025). No significant correlations were found between SAV and total phosphate. When chlorophyll *a* levels (an indication of possible nutrient enrichment) are compared to SAV levels, a significant correlation occurs with maximum chlorophyll *a* of the preceding year. In addition, the relationship of SAV to mean annual chlorophyll *a* (of current year) is significant at the 90 percent level.

In general, on a comparative segment basis, SAV appears to be responding negatively to increased TN of both the current and the preceding year. This, as well as the negative relationship with NO_3 of the preceding year, seems to support results of previous correlation analysis. Negative response to maximum chlorophyll *a*, an analog of both nutrient loading and turbidity, is also consistent with the SAV/nutrient enrichment hypothesis.

SUMMARY

When using a variety of statistical approaches with field observations, SAV abundance generally shows a negative relationship with nutrient nitrogenous concentrations, as well as with turbidity and chlorophyll *a* levels. Total nitrogen, NO_3, and IPF levels seem to most consistently correlate with SAV trends. These results are consistent with experimental conclusions that link the recent loss of Bay vegetation to an increase in nutrient enrichment and, ultimately, to light stress due to increased phytoplankton and epiphyte growth. The large number of correlations with nitrogen are interesting, and perhaps significant. A plot of percent vegetation in each segment against annual mean TN of the previous year (using the most recent year for which both SAV and TN data are available) is revealing (Figure 47c). Physical variables, such as salinity, temperature, and pH also appear to exert influence on SAV.

These observed correlations between SAV abundance and several parameters associated with nutrient enrichment are supported by strong experimental evidence. Such an integrated approach is the most satisfactory means of identifying causative relationships. In particular, microcosm and mesocosm experiments allow testing of various hypotheses under controlled but reasonably "natural" conditions. Similar methodologies may be indicated to strengthen understanding of other potential relationships between water quality and living organisms.

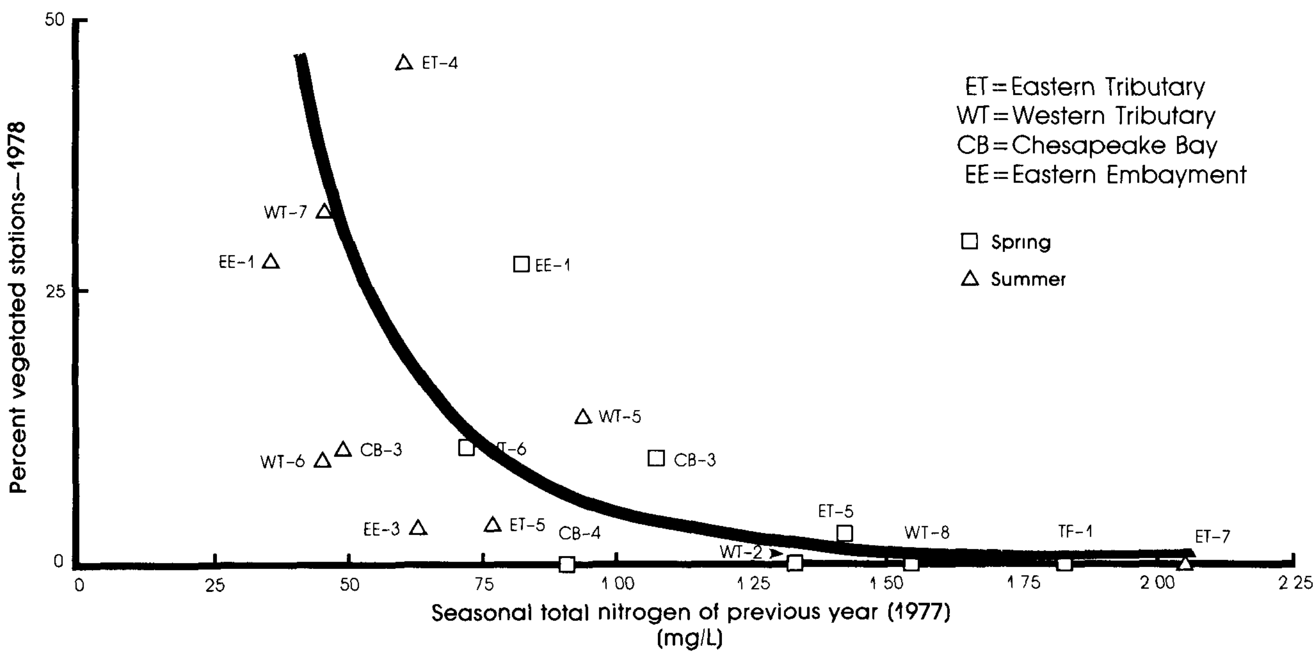


FIGURE 47C. Correlation between percent vegetated stations and annual total nitrogen of previous year.

SECTION 4
BENTHIC ORGANISMS AND WATER AND SEDIMENT QUALITY

INTRODUCTION

Increased stress on benthic communities can result in changes in the abundance and structure of these assemblages (Boesch 1977a). Sensitive species may be eliminated, for example, and resistant forms enhanced. In particular, populations of opportunistic polychaete or oligochaete worms may come to dominate in stressed areas, while relatively sensitive molluscs and crustaceans are reduced in abundance (e.g., Pfitzenmeyer 1975). In addition to effects on sensitive species or groups, overall species diversity may be reduced (Jacobs 1975). For example, heavily impacted areas of the New York Bight and Lower Bay Complex area showed low faunal abundance and low diversity as well as an absence of sensitive species (Mearns and Word 1982, Wolfe et al. 1982).

Such changes may be caused by natural or anthropogenic perturbations. Toxic contamination of bed sediments and overlying water with heavy metals or organic chemicals may be reflected in the accumulation of these materials in animal body tissues (O'Connor and Rachlin 1982), alteration of physiological processes such as respiration, shell deposition (Calabrese et al. 1982), or disruption of reproduction or development (Epifanio 1979, Calabrese et al. 1982). Nutrient enrichment may result in increased organic loads to the sediments, changing their structure and thus altering the type of benthic organisms found within them. In particular, increased numbers of infaunal detritivores have been associated with organic loading from sewage treatment outfalls (Bascom 1982). Oxygen depletion, due both to bacterial decomposition of organic matter and night-time respiration of algal blooms, is a major factor influencing benthic communities (Pearce et al. 1976, Mearns and Word 1982). In mid-Chesapeake Bay, summer hypoxia in water below 10 meters depth severely limits distribution and survival of benthic fauna (Holland et al. 1977, Mountford et al. 1977).

There is a certain difficulty in assessing the effects of anthropogenic stress on benthic communities. Natural variability in organism distribution and the dominant controlling mechanisms of sediment type, salinity, and predation, complicate identification of distinct cause and effect relationships (O'Connor 1972; Boesch 1973, 1977a; Virnstein 1977, 1979; Wolfe et al. 1982). Experiments with simplified systems (i.e., microcosms or mesocosms), where single factors may be manipulated, can give insight into community effects. An examination of the responses of benthic organisms (on communities) along a gradient of stress may also be useful.

EVIDENCE FROM RESEARCH

A series of experiments conducted at the University of Rhode Island's Marine Ecosystems Research Laboratory (MERL) used microcosms to investigate impacts of nutrient and toxicant stress on coupled benthic and pelagic communities. For example, nutrient additions designed to simulate sewage loading produced a marked response in microcosm benthic communities (Nixon et al. 1983). At intermediate nutrient concentrations (comparable to the Potomac River or Delaware estuaries), benthic abundance and biomass were greatly elevated, with a particular increase in bivalves. Secondary production was effectively shunted from zooplankton to the benthos. At high nutrient levels, comparable to the Hudson estuary, the alteration of the ecosystem structure was large; normally dominant bivalves were replaced by an almost exclusively polychaete community.

Similarly, additions of low concentrations of petroleum (No. 2 fuel oil) to the microcosms led to a dramatic reduction in benthic biomass and diversity (Olsen et al. 1982). As concentrations of oil rose in the surface sediments, sensitive species (such as the amphipod *Ampelisca)* were eliminated. Recovery did not occur within one year after oil additions ceased. Several interesting side effects were observed: phytoplankton biomass and productivity increased in oiled tanks, apparently because of reduction in herbivorous benthic species. Biomass of fish and wall-attached molluscs also increased. Water column nutrients declined, probably because of reduced rates of recycling by benthic organisms.

Application of microcosm results to field conditions is not always easy. However, comparison of the behavior of MERL microcosms to Narragansett Bay conditions shows remarkable similarity in annual cycles, in diversity, and in the abundance of organisms (Olsen et al. 1982).

EVIDENCE FROM FIELD OBSERVATIONS

Field experiments can sometimes help corroborate results observed in the laboratory. Blue mussels transplanted to sites in Narragansett Bay along a pollution gradient of (primarily) hydrocarbons and heavy metals showed significant declines in condition and physiological performance as stress increased (Widdows et al. 1981). A before-and-after study of benthic assemblages close to an area of oil pollution showed a reduction in species richness and total biomass as the concentration of petroleum in the sediments increased over time (Addy et al. 1978). Similarly, benthic species richness and biomass increased significantly as controls over oil pollution were implemented (Reish 1971, Leppakowski and Lindstrom 1978).

The distribution and abundance of benthic macrofauna were studied by several CBP projects as part of an investigation on toxic substances in Chesapeake Bay. Benthic forms, particularly infauna, can influence the movement of toxic materials within the sediments, or the movement of dissolved materials out of the sediments to overlying waters. The Bay main stem and several tributaries were studied (Nilson et al. 1981, Reinharz 1981, Reinharz and O'Connell 1981, Schaffner and Diaz 1982). These data give an excellent idea of spatial distribution of benthic macrofauna within the Bay; however, except for a limited number of sites, temporal information is incomplete.

Major anthropogenic factors identifiable in Chesapeake Bay and hypothesized previously as affecting benthic organisms are:

1. Impacts of toxic materials in bed sediments or in the overlying water column; effects might be acute (elimination of susceptible species) or sublethal (accumulation in body tissues, etc.).
2. Impacts of nutrient enrichment, primarily through the link to increased duration and extent of low DO, and resulting loss of habitat.

Toxic Materials

Direct comparison of several benthic community parameters to contamination of bed sediments was made. Unfortunately, data on organic chemicals were too few to allow meaningful comparisons. However, it should be emphasized that organic chemicals can still be considered a major potential threat to benthic species. One conclusion from a CBP-sponsored workshop on toxicant and organism relationships was the consensus that more information is needed on the distribution and biological effects of synthetic organic compounds in Bay sediments. Ongoing microcosm studies on potential effects of contaminated dredge spoil from the Elizabeth River show toxicity to be related to the suspended solid phase and associated with the organic fraction.[19] In addition, motile species were observed to actively leave contaminated sediment when clean substrate was available (Alden et al. 1981). When observing community structure, highly significant differences were apparent in control, "dump," and "adjacent" sites (Alden et al. 1981). Much of this was due to loss of motile species. MERL microcosm experiments with petroleum (No. 2 fuel oil) showed low concentrations to have a much more drastic effect than metals on benthic infauna[20] (Olsen et al. 1982).

Data on heavy metals in sediments were available from many areas of the Bay (Chapter 1); these were used in several analyses that allowed the CBP to look at the effects of heavy metals on diversity, dominance, and abundance. These analyses and results are detailed in Section 2 of Appendix D. Contamination of sediments with metals (C_I, the Contamination Index) and the potential toxicity of the sediments (T_I, the Toxicity Index[21]) were compared to benthic diversity and community structure. The Shannon diversity index ($\bar{H}$) and the annelid: crustacean and annelid:mollusc ratio were the community parameters tested (discussed in Chapter 2).

In the Bay mainstem, temporal and spatial variability in diversity appeared more related to the estuarine salinity gradient and sediment type than to C_I. Although there was a slight difference in annelid:mollusc and annelid:crustacean ratios between upper and lower Chesapeake Bay, no significant relationships could be identified using the Spearman rank correlation test. One difficulty is that benthic organisms and toxic materials were not sampled at exactly the same places. Innate variability of organisms' distribution would tend to obscure any relationships in such cases.

The Patapsco River has been subject to significant contamination with toxicants and could be expected to show more effect than does the main Bay. Within the Patapsco, diversity of benthic organisms generally declines along a gradient of increasing contamination with metals and organic chemicals (Reinharz 1981) (Figures 48 and 49, Table 33). Only stations near the mouth retained diversity comparable to the Rhode River reference area. Species found in the most contaminated areas are ephemeral opportunists, generally annelids, inhabiting only the upper layers of sediment. Anthropods and molluscs become more important in less polluted regions. For example, the tube-dwelling amphipod *Leptochierius plumulosus* is an important species in the Rhode River. However, in the Patapsco, it is found only at the two least contaminated stations (Figure 50). This is similar to the observation of Wolfe et al. (1982) that the tube-dwelling amphipod *Ampelisca* was absent from impacted areas of the New York Bight.

Statistically significant relationships were identified between contamination of sediments, diversity, and community composition. In general, the weighted Toxicity Index (described in Appendix D) appeared a better measure of potential impacts than did the Contamination Index alone. In the Elizabeth River, another severely contaminated tributary, trends were less distinct. This is probably because there are smaller differences in contamination from site to site within the river (Virginia State Water Control Board 1982). However, Schaffner and Diaz (1982) identified a group of stations characterized by shallow dwelling, young populations of relatively low diversity; these stations were considered impacted by high levels of toxicants in the bed sediments.

The effect of sediment contamination on benthic organisms was further explored using bioassay techniques. Using Elizabeth and Patapsco River sediments, bioassays were performed to determine the effect of sediments on the survival rate of a burrowing amphipod (*Rhepoxynius abronius*) (Swartz and DeBen, in prep.). Statistical analysis indicated that survivorship strongly correlates with the degree of contamination (C_I), as well as with the C_f for Ni and Zn, and approximates an exponential response to dose (Figure 51). An estimated LC_{50} would be $C_I = 15$. However, it should be emphasized that this association does not necessarily imply causation. Unmeasured metals or organic materials co-associated with the measured parameters may be contributing to, or actually causing, the observed mortality.

This view is supported by the observation that Spearman rank correlation of the annelid:mollusc and annelid:crustacean ratios with the contamination factor (C_f) for both Zn and Ni in the Patapsco showed no significant relationship. Thus, the relation between C_I and percent survival cannot be used to identify specific anthropogenic substances whose control can result in improved survival. However, it does indicate the probable presence of one or more toxic materials in the tested sediments.

In general, these correlations support the finding that a relationship exists between benthic community diversity, species composition, and con-

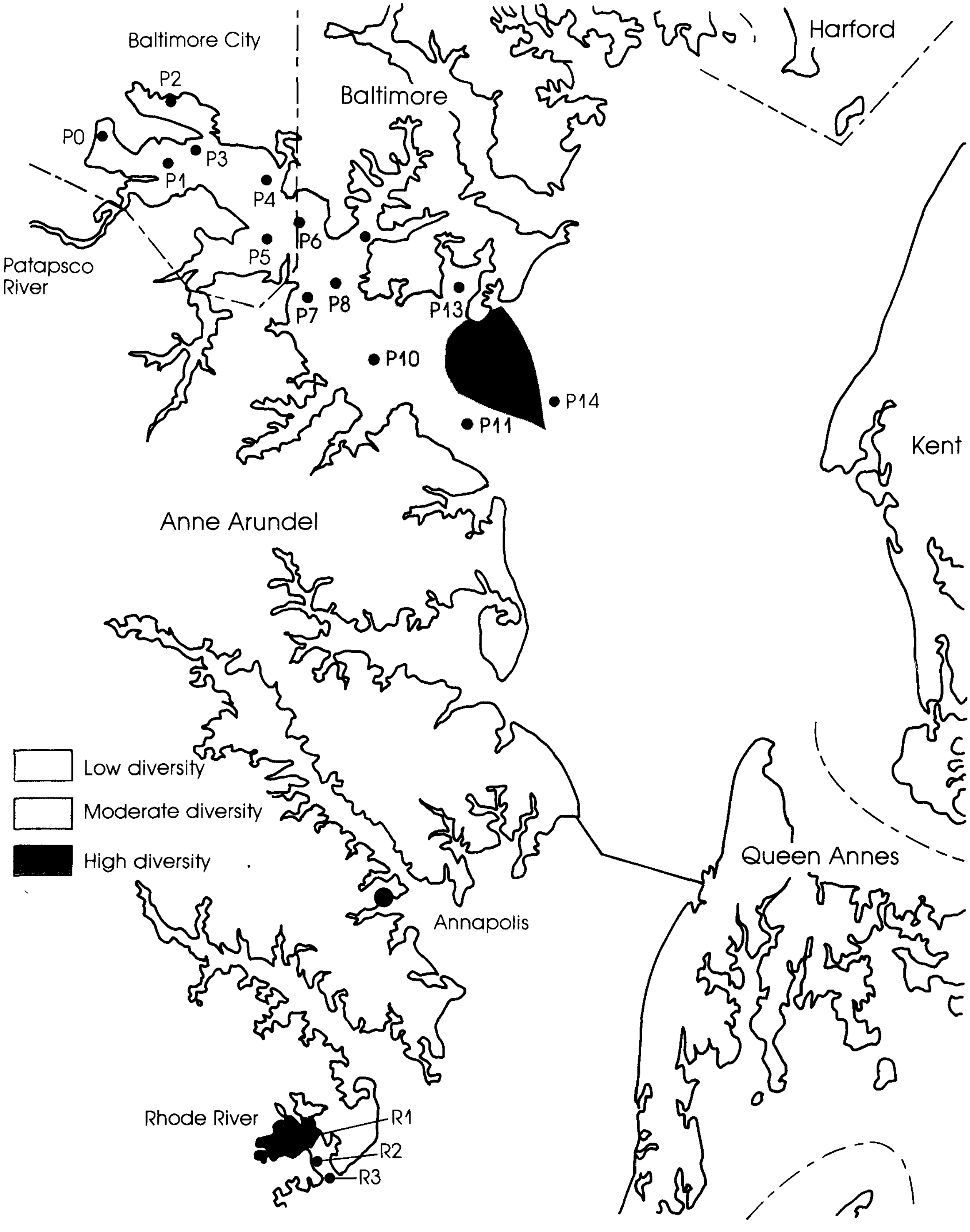
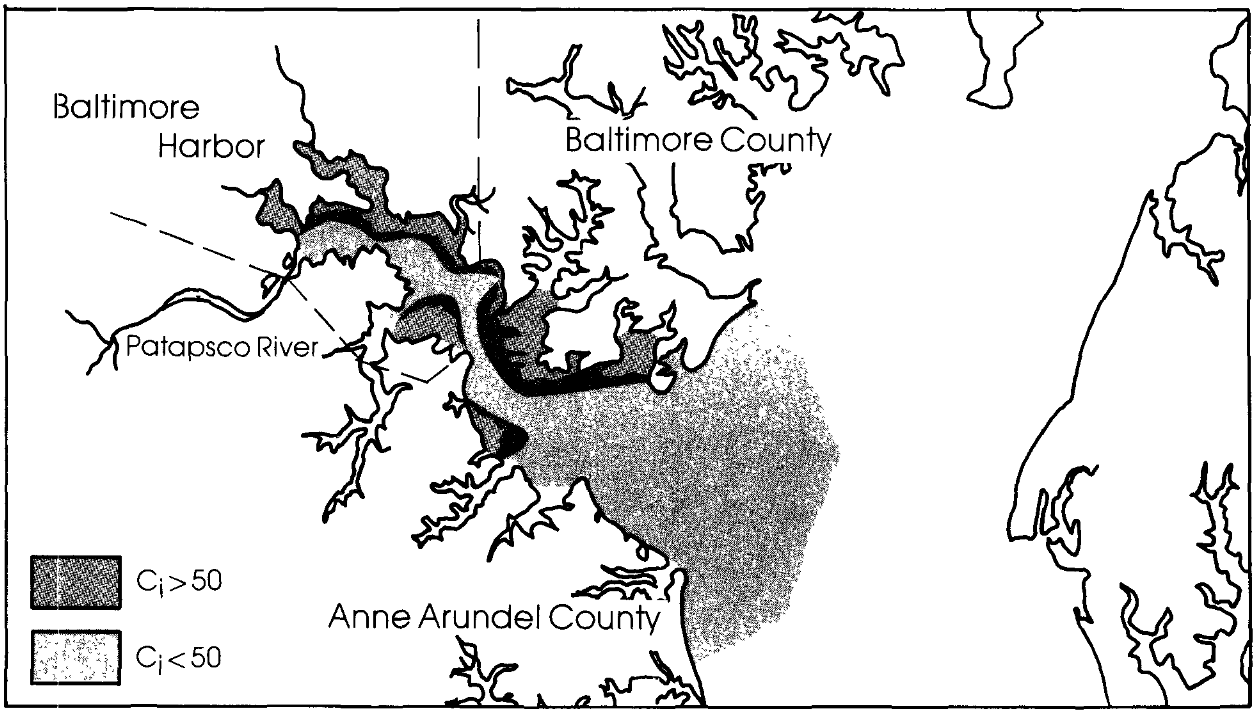
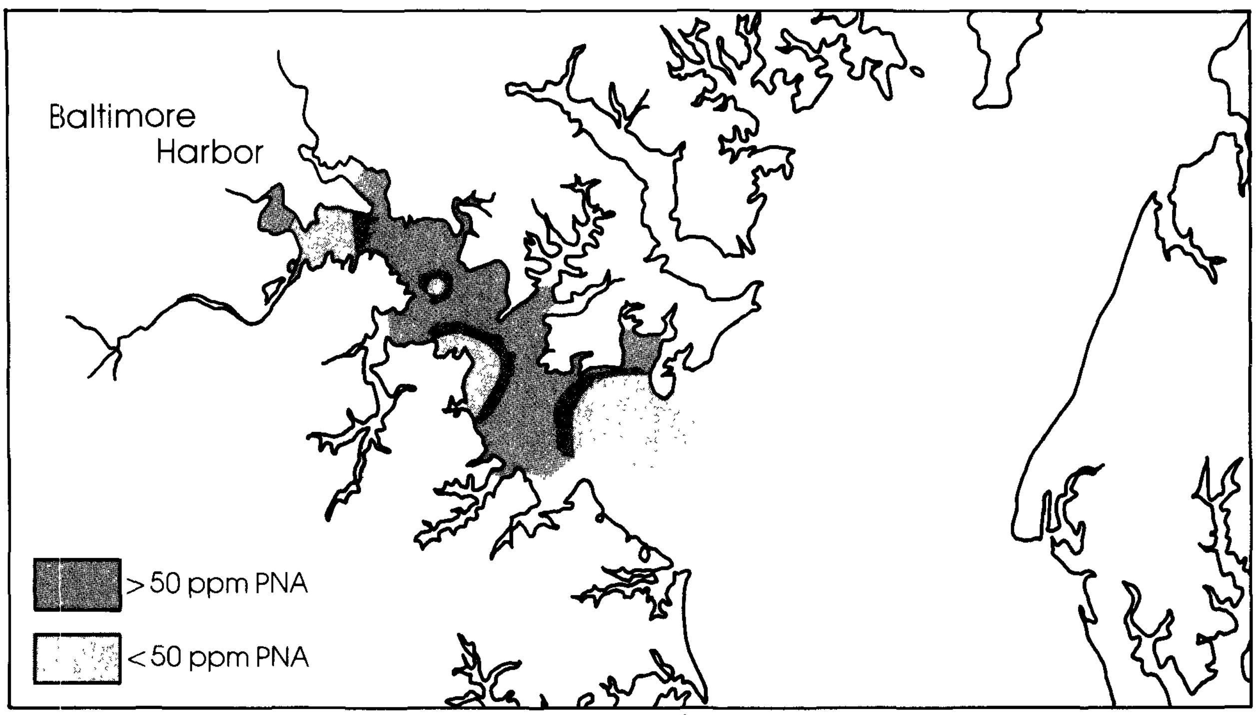


FIGURE 48. Diversity index (d) of benthic communmities in the Patapsco and Rhode Rivers (Reinharz 1981).



Metal Contamination (C_i) of the Patapsco River (Data from Biggs, 1982)



Distribution of PNA, Benzo (a) Pyrene in channel sediments from Baltimore Harbor and the Patapsco River. (Data from Huggett, 1982)

FIGURE 49. Contamination of Patapsco River and Baltimore Harbor sediments with heavy metals and organic chemicals (Bieri et al. 1982b).

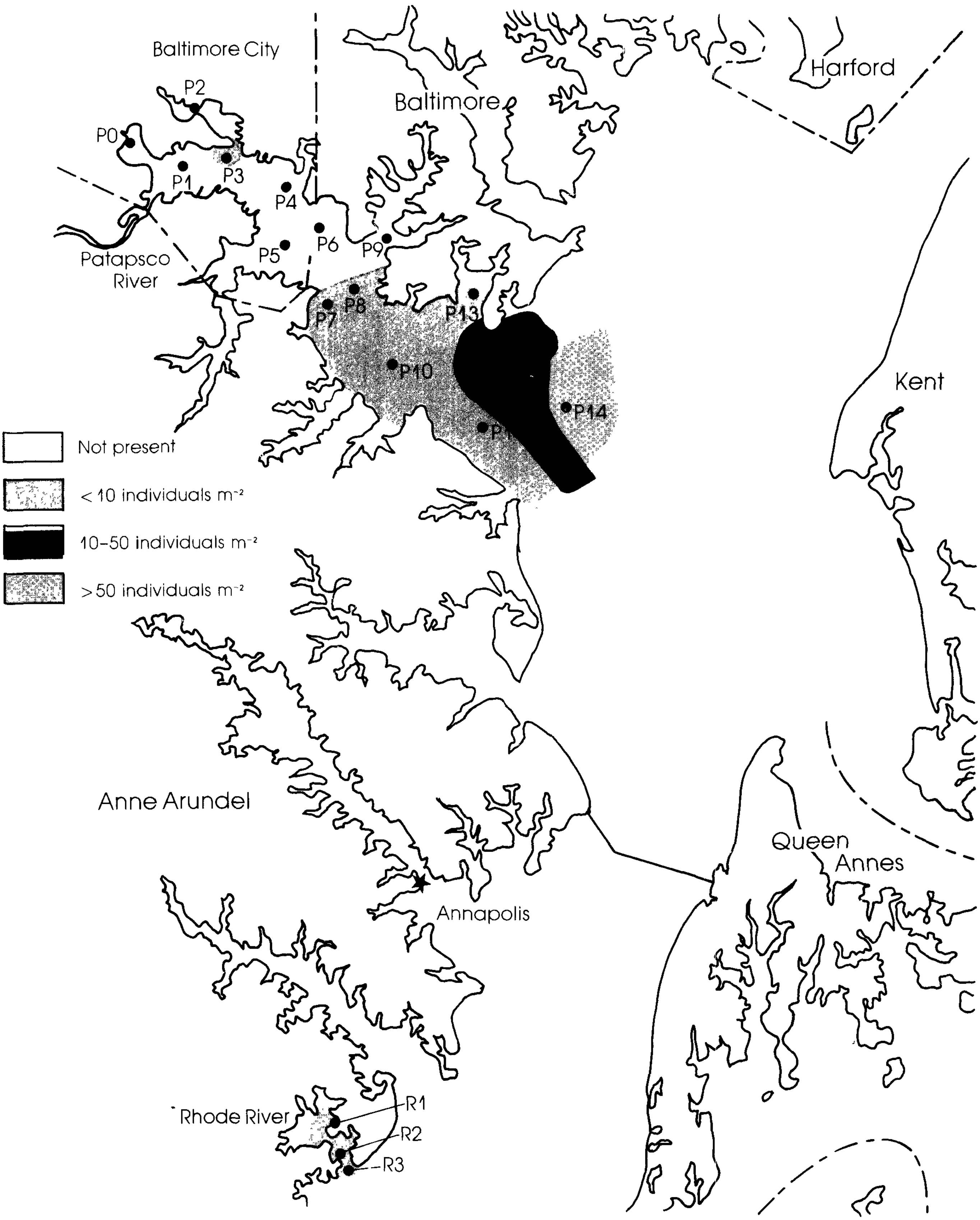


FIGURE 50. Density of *Leptochierus plumulosus* in the Patapsco and Rhode Rivers (Reinharz 1981).

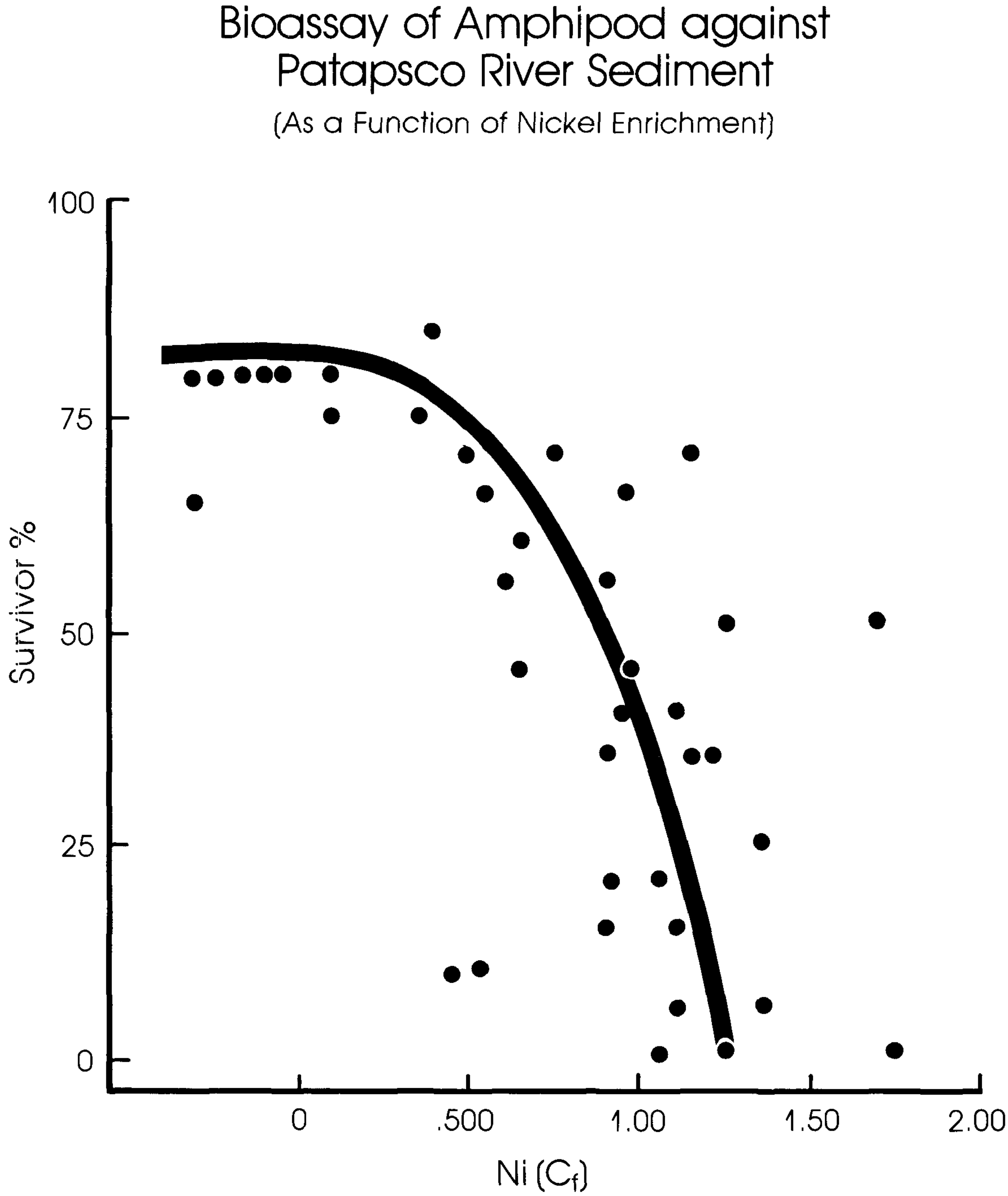


FIGURE 51. Percent survivorship of the amphipod *Rhepoxynium abronius* compared with the contamination factor for nickel.

TABLE 33.
DIVERSITY, REDUNDANCY, AND SPECIES NUMBER FOR PATAPSCO AND RHODE RIVER STATIONS. GROUPS ARE ALL SIGNIFICANTLY DIFFERENT FROM ONE ANOTHER

Station	$\bar{H}$	r	N
P_0	0.330	0.864	1
P_1	0.561	0.831	8
P_3	0.343	0.906	8
P_4	0.590	0.783	6
P_5	0.246	0.893	4
P_2	0.838	0.491	3
P_9	0.678	0.731	5
P_8	1.173	0.630	8
P_{10}	1.296	0.634	10
P_{11}	1.193	0.676	11
P_6	1.615	0.523	9
P_7	1.416	0.603	10
P_{13}	1.400	0.549	8
P_{12}	2.879	0.307	16
P_{14}	2.715	0.312	14
Reference			
R_1	2.286	0.420	15
R_2	2.348	0.369	13
R_3	2.501	0.366	15

P= Patapsco River stations
R= Rhode River stations
$\bar{H}$= diversity
r= redundancy
N= number of species present

tamination of the sediments, at least in the more impacted areas. Only the most general relationships could be demonstrated for the main Bay, indicating that other aspects of the environment tend to mediate benthic abundance and distribution. Only when levels of toxic materials become high—perhaps exceed some threshold—does the sediment contamination begin to play a major role in benthic community ecology.

Foraminifera in Baltimore Harbor and the Elizabeth River

Historical perspective on changes in sediment metal loads or changes in associate microflora or fauna can be obtained by examination of sediment cores. Foraminifera, which are microscopic animals, have proven to be useful indicators of environmental conditions. For example, several studies have shown changes in foraminifera abundance and shifts in species composition near sewage outfalls (Bandy et al. 1965, Bates and Spencer 1979). To better assess man's impact on the Bay's ecology, the CBP funded studies of foraminifera from Baltimore Harbor and from the Elizabeth River (Ellison and Broome 1982, Ogilvie and Ellison 1983).

In the preliminary study, three cores were taken from the harbor and one from the river. Ten representative core sections were sampled. Each sample was weighed, disaggregated in water, screened, and washed. Foraminifera were separated from the sediment, mounted, and identified using standard procedures. The diversity and abundance in the foraminifera were compared to metal concentrations in the sediment.

On the basis of both investigations, the researchers concluded that the change in benthic communities paralleled salinity and increased enrichment of the bed sediments with metals. Six species were identified in the studies: one agglutinated or arenacious species, *Ammobaculites crassis*, and five calcareous species: *Ammonion beccarii*, *Buliminella* sp., *Cibicides lobatulus*, *Elphidium clavatum* and *Elphidium subarcticum*. In general, large populations and non-diverse communities were found at the higher salinity sites: the mouth of Baltimore Harbor and the Elizabeth River. The somewhat fresher upstream sites in Baltimore Harbor, which are highly enriched with trace metals and synthetic organic compounds, had only one species, *A. crassus*, in limited numbers. In all four cases, there was an increase in *A. crassus* in the surface samples relative to the samples taken at depth. This paralleled an increase in concentrations of Mn. In the higher salinity cores, the populations of *Elphidium clavatum* decreased up-core, paralleling the general increase in metal concentrations (Ellison and Broome 1982).

Subsequently, six additional cores were taken – two in the Elizabeth River and four in Baltimore Harbor.

The diversity of foraminiferal communities declines up the estuary of Baltimore Harbor. Although this is typical of other estuaries and can be related to lowered salinity, it is unclear why the number of forams (especially *A. crassus*) also declines upstream. Possibly, salinity is not the primary, or only factor (Ogilvie and Ellison 1983). For example, an unknown factor potentially influencing these relationships is the low level of DO.

In the Elizabeth River, two cores with similar trace metal distributions had different densities and diversities of foraminifera. Salinities are similar at these sites so these differences may also be caused by another factor (Ogilvie and Ellison 1983).

In some deeper and less contaminated Baltimore Harbor sediments, *A. crassus* is found in the greatest number – comparable to other less contaminated estuaries. This suggests that, in the more contaminated areas of the harbor, metals have played a role in preventing this hardy species from forming larger populations (Ogilvie and Ellison 1983).

Toxic Materials in Tissues of Shellfish

Contamination of sediments or overlying water with toxic materials may result in an accumulation of these materials in the tissues of benthic organisms (O'Connor and Rachlin 1982). Tissue levels are not simply related to the bulk concentration of toxicants in sediments, particularly in the case of trace metals, but rather to their bioavailability. Toxic materials may be bound in forms that are normally unavailable biologically or chemically, but which may be liberated by rigorous laboratory analyses (Bricker 1975). Levels of toxicants in animal tissues can, however, be considered an indicator of the presence of processes which render these contaminants available to organisms.

Marine organisms employ a variety of mechanisms for sequestering and detoxifying metals, including the production of metalloproteins (O'Connor and Rachlin 1982). For that reason, apparent body burdens of some contaminants may not result in measurable physiological impacts, at least to some threshold (e.g., Frazier 1976). However, mussels transplanted along a gradient of pollution show an accumulation of Ni in body tissues, corresponding to the concentration in the overlying water column; physiological stress and other indicators showed a similar gradient (Widdows et al. 1981).

Accumulation from contaminated sediments varies with the species of organism, as well as with the nature of the sediment; bioavailability was generally related to EDTA extraction values for the metals rather than to bulk concentration (Ray et al. 1981). Ayling (1974) found correspondence between oyster (*Crassostrea gigas*) tissue levels and sediment metal concentrations for Zn and Cd; Cu and Cr appeared more related to the size of the animal. Similarly, Frazier (1976) found a relationship between sediment metal levels and oyster tissue concentrations; the relationship, however, was not linear. Oysters in areas with high environmental metal concentration had significantly thinner shells ($\sim$16 percent).

The processes governing availability of sediment and water column toxicants to benthic organisms are not well understood. Evidence exists suggesting that tissue concentrations reflect these processes (environmental concentrations also reflect them to a varying extent). However, considerable work is needed before these processes are identified, and before ecological effects of tissue contamination are understood (O'Connor and Rachlin 1982). Biomagnification of toxic materials through the food chain has been documented in numerous instances and is one potential serious impact of the accumulation of toxicants in tissues of major forage organisms (Wolfe et al. 1982).

Trends of metals and pesticides in Chesapeake Bay shellfish were discussed in Chapter 1. Because the majority of these data concern oysters, the relationship of tissue contamination to sediment concentration will be discussed later in the subsection on oysters and water and sediment quality.

Nutrient Enrichment, Low Dissolved Oxygen

Bascom (1982) and Mearns and Word (1982) relate changes in species composition of benthic communities subject to high organic loads from municipal wastewater discharges. An Infaunal Index was developed that used feeding characteristics and the number of individuals in each trophic group to identify degraded areas. In general, such areas were characterized by increased biomass, reduced species richness, and shifts from dominance by suspension feeders to surface and subsurface deposit feeders, primarily polychaetes. This index was calculated for a number of mud (<25 percent sand) stations from the Bay mainstem (Reinharz and O'Connell 1981, Nilson et al. 1981) and examined for possible spatial differences relating to nutrient enrichment patterns in Chesapeake Bay. Although there was no significant spatial trend, in general, polychaetes dominated to a greater extent up-estuary. Suspension feeders such as *Ampelisca* were more abundant in the lower Bay. However, it is probable that salinity and physical variability of the environment are most important in mediating this organism distribution. The Infaunal Index would be useful, however, in examining data sets from a before-and-after situation, to identify effects of nutrient or wastewater controls, etc. Similarly, it could be employed in localized areas where spatial variability of the environment is minimal.

A major factor affecting benthic populations is occurrence of low DO concentrations in the overlying water column. Instances of low DO values in the New York Bight area have been associated with faunal changes, including extensive loss of oyster beds (Carriker et al. 1982, Franz 1982). These hypoxic events have been caused by increased carbon loading from sewage and runoff, and from decay of phytoplankton blooms stimulated by excess nutrient enrichment (Mearns and Word 1982). It was suggested by Mearns and Word that a large-scale reduction in nitrogen input, especially in the summer, might alleviate some of these problems.

Episodes of low DO have been described for the Chesapeake Bay area since 1917; in September 1912, bottom water in the lower estuarine portion of the Potomac River was less than 35 percent saturated with oxygen (<2.0 ml L^{-1}) (Sale and Skinner 1917). At the same period of time, bottom water between Annapolis to off Hampton Roads was over 60 percent saturated. Strong evidence exists indicating that the areal extent and duration of such episodes of low DO has increased over time in Chesapeake Bay, particularly in the past 20 years (Chapter 1). At the present time, for example, almost 19 percent of the bottom area of Bay segments CB-2 through 5, are impacted by DO values of less than 0.5 ml L^{-1} (Table 34). Furthermore, these changes appear related to increases in nutrient loadings into the upper Bay (Chapter 1).

Numerous Chesapeake Bay investigators have linked changes in benthic fauna to episodes of low DO. Near-total faunal depletion occurred at mid-Bay sites >9 m deep caused by episodes of hypoxia (Holland et al. 1977); this single factor was the major influence limiting benthic biomass below 8 m depth (Holland et al. 1979). Low DO generally restricts oyster beds to depths less than 10 m (Haven et al. 1981); however, where circulation is good, oysters can exist at much greater depths (Merrill and Boss 1966).

In areas subject to low DO, benthic communities may be completely eliminated, or replaced by ephemeral, opportunistic assemblages recruited after cessation of hypoxic conditions (Holland et al. 1977). Most benthic species are stressed when oxygen concentrations fall below 1.5 to 2.0 ml L^{-1} for extended periods and are killed by exposures to 0.5 ml L^{-1} or less (Table 34). As the area impacted by these DO levels has increased, summer benthic communities become restricted to the shallower Bay margins. Reduction in available forage could have impacts on bottom-feeding predators, particularly sciaenid fishes or crabs. Predation pressure on benthic communities is very intense during the summer months (Homer and Boynton 1978, Holland et al. 1979, Virnstein 1979); food limitation could affect abundance, growth, and condition of predators.

Increasing instances of oyster mortality (measured as percent empty shells or "boxes") have been reported for upper Bay sites in recent years.[22] One hypothesis is that intrusions of low oxygen water onto shelf areas occur with greater frequency now because the area impacted by hypoxia has increased. Because of the low gradient of this shelf, slight decreases in depth of the oxycline can greatly increase the area under stress. For example, if the 0.5 ml L^{-1} DO isopleth occurs at 12 m, 26 percent of CB-4 bottom is impacted. If depth of the isopleth decreases to 10 m, the area impacted more than doubles, to 58 percent (Table 34). Internal waves at the oxycline may cause variations of 1 m or so in depth within minutes (Flemer and Biggs 1971). Additionally, wind events can shift Bay surface waters, allowing deep low oxygen water to intrude onto the shelf. "Crab wars" or "jubilees" may occur when northwest winds persist for hours or days; crabs and other mobile species congregate in the shallows or even move onto the beach, to avoid anoxic conditions (Carpenter and Cargo 1957).

Recent interviews with watermen support this hypothesis and give further insight into possible impacts of DO and other water quality parameters on crab and oyster harvests. Some of the major changes in crab and oyster harvest with depth were summarized by Mr. Pete Sweitzer of Tilghman Island, Maryland. Prior to the 1960's, blue crabs were potted in deep water, often at depths greater than 15.3 m, from about April 15 to mid-May, and from September 15 into early fall. This fishery captured the male crabs as they emerged from winter hibernation in the deep channel. Now, there is little potting either in the early spring or late fall, because too few crabs are taken to warrant the effort. No crabs are presently taken in oyster dredges in deep water of the mid-Bay; winter males are now observed in relatively shallow, sandy areas at the channel shelf "break". Crabs potted in the summer are taken at a maximum of 3.7 m. The condition of crabs caught in August seems to be weaker than in previous years, with many crabs dying before they are landed. Watermen report that changes in oyster catches with depth have not been as dramatic as with crabs, but that there is a tendency now to take fewer oysters in deeper water (>9.2 m) than in the mid-1960's.

The watermen also cite changes in the fouling communities on crab pots and in water clarity. In the late 1940's, little fouling occurred over

TABLE 34.
AREA OF CHESAPEAKE BAY BOTTOM AFFECTED BY LOW DO WATERS IN SUMMER; % = PERCENT OF BAY SEGMENTS CB 3, 4, AND 5 IMPACTED

DO Level	July 1950		July 1969		July 1980	
ml L^{-1}	$m^2 \times 10^6$	%	$m^2 \times 10^6$	%	$m^2 \times 10^6$	%
0.5	62.3	2.1	344.0	11.3	603	19.9
1.0	228.0	7.5	535.0	17.6	789	26.0
2.0	824.0	27.2	629.0	20.7	1196	39.4
3.0	1191.0	39.3	889.0	29.3	1417	46.7
4.0	1545.0	50.9	1455.0	48.0	2022	66.7

a three to five day period. Beginning in the late 1950's, a green growth covered the pots, replaced by a grey mud in the late 1960's. By the late 1970's, this material became dominated by a brown "grass-like growth." Visibility seems to be less now than in earlier years. The water, they say, grows more cloudy earlier in the spring; this condition lasts longer into the fall.

SUMMARY

Nutrient enrichment affects benthos through the impact of increasing low summer DO in deeper water. Complete loss of benthic habitat in some areas or replacement with ephemeral assemblages, as well as possible mortalities of commercially important species such as clams, crabs, and oysters are effects that have been observed in the field. Though the relationship between hypoxic events and increased mortalities of oysters, etc. is at this point speculative, it is worth further investigation.

Effects of toxic substances are difficult to separate from natural variability in the field, at least in those areas where contamination is minimal or moderate. In heavily impacted areas, however, benthic community structure is changed: sensitive species are eliminated and opportunistic, resistant forms are relatively enhanced; generally, a loss of species diversity occurs. The accumulation of toxicants in body tissues of benthic species occurs. These may have direct physiological impacts on the organisms or may be transmitted and biomagnified in the food chain.

SECTION 5
OYSTERS AND WATER AND SEDIMENT QUALITY

INTRODUCTION

Oysters are essentially immobile and, for that reason, are considered excellent indicators of environmental conditions. Because of their commercial importance, fairly good data exist (both spatially and historically) on distribution, general abundance, harvest, spat set, and condition. Their ecology and physiology are also well-known. However, fundamental questions still perplex scientists and managers. For example, what is the role of the natural cycles of environmental variables versus the anthropogenic factors in the distribution and abundance of oysters in Chesapeake Bay? Are the spatial and temporal scales of these controlling factors reasonably constant over time? What are the effects of harvest and management (i.e., seed or clutch plantings)? For example, Ulan[illegible]z et al. (1980) showed that variations in spat density and seed plantings explain 56 percent of the variability in the annual harvest.

Though such studies increase the understanding of the environmental regulation of oysters, they fail to explain the long-term decline in oyster harvest since the late 1800's (Figure 52). Conventional wisdom indicates that sedimentation, poor water and sediment quality, and over-harvesting (economically related) are probably key factors; however, their importance over the past century is still a matter of conjecture. The constancy in oyster harvest since about 1960 is probably related to shell planting, especially in Maryland.

The purpose of this section is to assess the nature of available environmental information concerning the oyster in the Bay and provide a synthesis of this information to gain a better understanding of the factors operating as potential environmental controls.

FACTORS POTENTIALLY IMPACTING OYSTERS

Numerous natural and anthropogenic factors can impact oyster distribution, harvest, and recruitment. Among these are:

- Impacts of nutrient enrichment and resulting oxygen depletion. Excessive nutrients could result in shifts of phytoplankton species to forms less usable by oysters (e.g., Ryther 1954), or in the fouling of shells with epibiota; the latter would interfere with the settling of young oysters. Episodes of low DO could kill oysters outright, as well as restrict productive beds to shallow areas.
- Impacts of sedimentation and increased suspended solids loads. Sediment deposited on shells prevents the setting of spat and, if excessive, can "smother" beds or interfere with feeding.
- Impacts of toxic materials in bed sediment or water column. This could result in the reduction of spat set or oyster survival, or in the accumulation of toxicants in oyster tissues.
- Impacts of disease, parasites, or predators. The effects of disease may be exacerbated by other stresses (Farley et al. 1972).
- Impacts of natural variables, particularly freshwater discharge and salinity, or management practices (shell or seed planting) on distribution and abundance. Oyster larvae require salinities above 10 to 12 ppt to successfully develop and set (Shea et al. 1980); for that reason, recruitment at lower salinity bars is often sporadic.

It is probable that all these factors act at one time or another to affect oysters. In the follow-

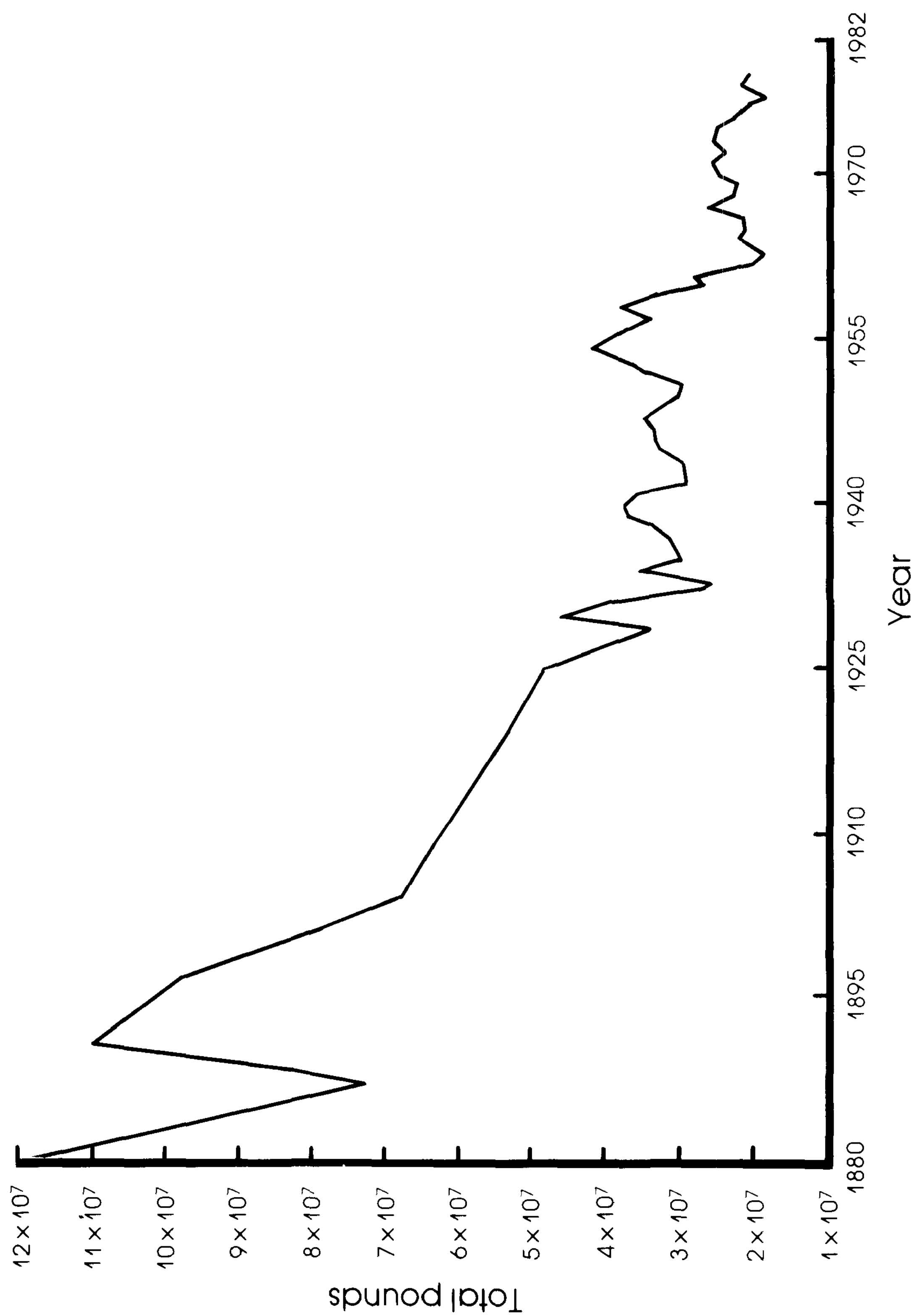


FIGURE 52. Historical pounds of shucked oyster meat in Chesapeake Bay.

ing section, each will be discussed in greater detail.

Nutrient Enrichment and Oxygen Depletion

Nutrient enrichment may cause shifts in phytoplankton species to forms less suitable for oyster food. Ryther (1954) reported a case where runoff from duck farms produced excessive nutrients in Moriches Bay, Long Island, New York; a predominantly diatom flora was replaced by a small non-motile green algae. These were not usable by oysters for food, so the local oyster fishery declined precipitously. It is well-documented that all algae are not equally suitable for oyster food (Walne 1963, 1970). Eutrophication often results in blooms of types considered less desirable: bluegreens, non-motile greens, and dinoflagellates (Ryther and Officer 1981). Bluegreens are not generally a problem in saline areas where oysters are found. However, in some coastal regions, blooms of certain dinoflagellate species are the cause of episodes of paralytic shellfish poisoning (PSP); the dinoflagellate blooms themselves have been related to the nutrient enrichment of coastal waters, at least in some areas (Prakash 1975).

In Chesapeake Bay, there is little evidence that significant changes in the phytoplankton community structure have taken place in oyster-growing areas (Chapter 2). Although blooms of dinoflagellates occur frequently, these are not species generally known to be toxic (Mackiernan 1968). However, continued monitoring of phytoplankton communities and comparison to historical data are recommended.

As with benthic organisms in general, the strongest link to nutrient enrichment is probably through DO. Oyster larval growth ceases at 1.7 ml L^{-1} DO, and adult oysters close up when levels reach 0.7 ml L^{-1} or less. During this time, they undergo anaerobic metabolism, which is energetically costly. In warm summer months, oysters can survive about five days in this manner; if anaerobic conditions persist much longer, they will die (Kaumeyer and Setzler-Hamilton 1982). In many areas of the Bay, oysters are restricted to depths less than 10 m by low oxygen (Haven et al. 1980). Increased mortalities of oysters observed in recent years may be due to intrusion of hypoxic water into shelf areas, or the combined stress of low DO and disease such as MSX.[23]

Landings of all commercial finfish and shellfish from 1980 were compared to nutrient concentrations for the most recent year of record. A significant inverse relationship ($p \leq 0.01$) was found between shellfish from 1980 and mean annual TN (i.e., high nitrogen values were correlated with low harvest). This relationship was apparently due primarily to oysters: a significant inverse relationship ($p \leq 0.05$) existed between 1980 oyster harvest and TN. The Chester River was omitted in this comparison because of its recent unexplained oyster mortality. Also, this relationship was not statistically significant in other years.) Such a relationship could be due to low DO impacts, food web shifts, or both.

Sedimentation

The removal of substrate (by harvest) has lowered the profile of many oyster bars in Chesapeake Bay; that is, they are closer to the sediment surface (Marshall 1954). This makes them more vulnerable to sedimentation. As discussed in Chapter 1, sediment transport and deposition into Chesapeake Bay has increased dramatically in the last 150 years. The existence of extensive buried oyster beds where substrates are now unsuitable is an indication of sedimentation effects (Alford 1968). Sediment also affects recruitment; a few millimeters of sediment on a shell may prevent the setting of spat (Galtstoff 1964). For that reason, spat set is usually better on freshly planted shell (Davis et al 1981). In addition, Ulanowicz et al. (1980) have shown that spat set success negatively correlates with the previous year's harvest; this may result from removal of substrate (or possibly brood stock).

Toxic Materials

The comparison of the EPA water quality criteria to measured and estimated concentrations of toxicants in the water column reveals a number of instances where these criteria are exceeded (Chapter 1, and Appendices B and D). Oyster larvae are sensitive to Cu and Hg, while adults are sensitive to Cd (Kaumeyer and Setzler Hamilton 1982). Thus, although direct cause-and-effect cannot be demonstrated, the occurrence of comparatively high levels of toxicants in the water column near oyster beds would be cause for concern.

The accumulation of trace metals by oysters was described by Ayling (1974). He found Cr and Cu to be adsorbed to a weight proportional to, and limited by, the size of the oyster and apparently independent of sediment concentration. Cadmium and Zn were accumulated in proportion to their concentration in the sediment. Lead was concentrated randomly at sites containing high sediment Pb concentrations.

Mean levels of Cd, Cu, and Zn in oyster tissues from Chesapeake Bay were compared to the sediment contamination factors for each metal, using Spearman rank correlation coefficient. Levels of Zn in shellfish tissue correlated at $p \leq 0.01$ with Zn in bed sediment, in agreement with Ayling's findings. Copper levels in oyster tissue also positively correlated with sediment concentrations ($p \leq 0.01$). Tissue concentrations could not be related to the size of the organisms because tissue analysis had been performed on composite samples of various sized oysters. No statistical relationship could be demonstrated between Cd in oysters and that in sediments, although levels of both tend to be low in Mobjack Bay, the Potomac River, and the upper main Bay.

Disease

The impacts of disease and predators can be significant. In the early 1960's, an epidemic of MSX, the protozoan parasite *Minchinia (Haplosporidium) nelsoni*, decimated the Virginia oyster population. From 50 to 90 percent of susceptible individuals succumbed, and harvest fell by two thirds (Fincham 1983). The disease MSX is restricted to waters generally greater than 15 ppt salinity, and the mid-1960's drought years hastened its spread. Lower salinities since 1968 and during the 1970's, as well as some acquisition of genetic resistance by the oysters, had reduced impacts in the last decade. However, there is strong evidence that drought conditions in the 1980's has again allowed MSX to spread. This time, areas of Maryland as far north as the Choptank River and Eastern Bay have been affected, and the parasite has moved into regions with previously unexposed populations (Fincham 1983). Present increases in oyster mortality in these areas are probably related to the disease, although losses in deeper regions may also be caused by DO stress. If salinities decline in the upper Bay, oysters may recover from the parasitic infection.

Other diseases and predators are more prevalent at higher salinities; these include "Dermo" disease (*Perkinsus marinus*), the oyster drills *Urosalpinx* and *Eupleura*, the mud blister worm *Polydora*, the boring sponge *Clionia*, and the spat predator *Stylochus*, a flatworm. Many of these weaken the oyster and may render it more vulnerable to other stresses.

Natural Variables

The success of biological resources is also affected by natural variables. Ulanowicz et al. (1980) have shown that success in spat set correlates positively with high salinity and negatively with the previous year's harvest. Krantz and Carpenter (1981) report that the impact of Tropical Storm Agnes (1972) on the Potomac River oyster populations was largely responsible for a decrease in production and a shift in harvest from upstream of Cobb Island to the lower Potomac below Ragged Point-Piney Point. In addition, there has been no significant oyster recruitment in the upstream estuarine portion of the Potomac River since 1965; this may be due in part to increased freshwater flows during the last decade. However, failure of spat set in the upper Potomac in response to increased salinities during 1980 to 1981 (low-flow years) points to a decline in water quality as an additional stress (Figure 31). Biggs (1981) notes that buried oyster reefs occur in Chesapeake's upper main stem above the current distribution of oysters. These (and similar buried reefs in the Potomac) indicate that oysters once inhabited the reaches of Chesapeake tributaries no longer suitable for them. Deforestation of the watershed has created a more "flashy" pattern of runoff where peak flows may be 30 percent greater today, shifting the riverine-estuarine transition zone downstream, and compressing oyster habitat. Finally, Davis et al. (1981) have reported very high oyster spat sets for the years 1980 to 1981 in the Maryland portion of

Chesapeake Bay. These years were characterized by above average salinities during the oyster spawning season (May through July). Spat set was larger, on the average, during 1980 in areas of the middle and upper Bay than had been observed in many years.

However, this greater-than-average spat set did not occur in all reaches of the Bay, suggesting that other factors are operating there that affect oyster recruitment. For example, the Chester River exhibited spat set between about 10 to 25 spat per bushel during the 1940's and 1950's, and since 1971, has averaged zero spat set. This may result, in part, from a loss of natural spawning stock because the present production depends on seed planting. If spat set conditions are assumed to be adequate now, then some recruitment from the nearby Bay would be expected. Thus, a water and sediment quality factor could be limiting spat set in the Chester River and similar areas.

SUMMARY

It is apparent that a number of factors — both natural and anthropogenic — affect oyster distribution and abundance. Physical variables, such as salinity, disease, predators, and the socioeconomic aspects of harvest and management tend to obscure most relationships. However, sufficient evidence exists to give cause for concern. Available habitat and harvest are related negatively to increased nutrients and areas of low oxygen water, and tissue contamination can be linked to materials in the bed sediments. The loss of harvestable areas and the failure of natural recruitment in upstream areas potentially most impacted by man are strong arguments that anthropogenic trends in water or sediment quality affect oysters in Chesapeake Bay. The close monitoring of future trends, particularly in possible high impact areas, is recommended.

SECTION 6
FINFISH AND WATER QUALITY

INTRODUCTION

Finfish may respond to a number of natural or anthropogenic variables. Although mobile and thus able to avoid certain stressful conditions, fish may nevertheless be impacted by factors operating at the individual, population, or community level. A number of natural environmental factors have been shown to affect spawning and juvenile survival (Ulanowicz et al. 1981); changes in recruitment may then be reflected in catch statistics (Richkus and Summers 1981). Similarly, toxic pollution may result in outright mortality of sensitive individuals or species, in an increase in incidence of disease or pathologies, in an accumulation of toxicants in fish tissues, or in more subtle effects on spawning success (Sindermann et al. 1982). It is, however, often difficult to separate out the more moderate impacts from responses to natural climatic cycles. This section addresses these considerations to make the most reasonable assessment of effects of water quality on finfish.

Five major hypotheses have been advanced to explain observed trends in finfish (Chapter 2):

1. Increased toxic material contamination in areas where anadromous fishes spawn and larval development occur is exceeding the tolerances of these vulnerable early life stages. In addition, accumulation of toxic chemicals in fish body tissues may be affecting spawning success (e.g., viability of eggs) or resistance to disease.
2. Nutrient enrichment of areas of Chesapeake Bay is impacting spawning success of anadromous fish, either by shifting phytoplankton communities (altering the type and the amount of zooplankton upon which these fish larvae feed) or by increasing areas impacted by low DO (reducing available habitat for many species). Marine and estuarine spawning species that utilize low-salinity nursery areas, or that inhabit bottom waters (e.g., sciaenids) could also be adversely impacted by nutrient enrichment. Finally, the loss of SAV, an important habitat and forage area for juvenile fish and their prey, has been linked to increases in nutrient loads to the Bay.
3. Freshwater spawning fish are being reduced in number by sub-optimal climatic conditions that affect the entire Bay watershed, and that are expressed as variations in temperature, precipitation, and river flow. This hypothesis also embodies the concept that moderately increased freshwater flows may improve hatching and larval success in some species by carrying more nutrients into the nursery area (supporting the phytoplankton and zooplankton larval fish food chain), as well as by physically extending the area of the spawning and hatching grounds (Herrgesell et al. 1981, Mihursky et al. 1981). Additionally, climatic conditions (e.g., wind and flow) may enhance marine spawners principally by increasing recruitment into the estuary.
4. Some species (particularly anadromous fish) are being reduced in number by overfishing. This hypothesis embodies the concept that sport and commercial fishing are harvesting so many fish of these species that

remaining breeding stock is not sufficient to sustain recruitment.

5. Modification of habitat, particularly in areas of the watershed used by spawning anadromous fish and in those used as nursery areas, may be having significant impacts on the ability of some tributaries to support recruitment. Among these modifications are the construction of dams, the draining and loss of wetlands, stream channelization, and the urbanization of watersheds.

IMPACTS OF TOXIC MATERIALS

Studies of the impact of toxic pollutants, including heavy metals, organic chemicals, and petroleum on fishes of the New York Bight identified a number of responses: heavy metals resulted in the increased mortalities of (particularly) early life stages; the sublethal exposure of adults to toxic pollutants drains energy reserves and may affect their response to other environmental stresses (Calabrese et al. 1982); and the abnormalities in mackerel eggs were associated with elevated toxicant levels (Longwell and Hughes 1982). However, it was difficult to show many changes in abundance directly attributable to pollution, because of the simultaneous effects of natural environmental variation and over-fishing (Sindermann et al. 1982).

Research sponsored by the Emergency Striped Bass Research Study (USFWS, NMFS) is revealing some effects due, in part, to toxic pollutants. Whipple et al. (1982) studied striped bass in the San Francisco Bay area, correlating pollutants (metals, and chlorinated and petroleum hydrocarbons) with parasite burdens, body and liver condition, and with egg and gonad condition. It was estimated that there was a 45 percent reduction in the number of viable eggs for 1978 females before spawning; pollutants most implicated were Zn and *monocyclic aromatic hydrocarbons* (MAH). Closer to home, work in progress by Ludke et al. (1982) reveals that combinations of contaminants (Table 35) at concentrations similar to those found in spawning habitats of Chesapeake Bay significantly decreased the survival of striped bass larvae, particularly in freshwater. These contaminants include pesticides, other organic chemicals, and heavy metals. Preliminary results indicate that larvae exposed to the highest concentrations in freshwater experienced 90 percent mortality within 15 days, as compared to 10 percent in controls. Sublethal effects included changes in swimming, feeding, and predator avoidance.

TABLE 35.
MIXTURE OF CONTAMINANTS CAUSING LETHAL AND SUBLETHAL EFFECTS IN LARVAL STRIPED BASS (LUDKE ET AL. 1982)

Compound	Concentration L^{-1}	Compound	Concentration L^{-1}
Arochlor 1248	10 ng	Fluoranthrene	40 ng
Arochlor 1254	10 ng	Pyrene	40 ng
Arochlor 1260	10 ng	Benzoanthrene	40 ng
DDE	3 ng	Chrysene	40 ng
Toxaphene	3 ng	Atrazine	1 ug
Chlordane	5 ng	Simazine	1 ug
Kepone	15 ng	Arsenic	1 ug
Perylene	40 ng	Selenium	2 ug
Fluorene	40 ng	Lead	1 ug
Phenanthrene	40 ng	Cadmium	3 ug
Anthrene	40 ng	Copper	1 ug

Analyses performed by the CBP (discussed in Chapters 1 and 3, and Appendices B and C) show that measured and estimated levels of heavy metals and chlorine often exceed the EPA water quality criteria in areas where anadromous fish spawn and develop. In particular, Cd, Cu, and Ni criteria are exceeded. As the larvae of many fish are extremely sensitive to these metals (Kaumeyer and Setzler-Hamilton 1982), high ambient concentrations would be potentially serious. Although the data do not allow an estimate of actual extent and duration of exposure, a prudent approach would regard these findings with concern. It appears as though the potential for significant acute or sublethal impacts due to toxicants may exist in many areas.

NUTRIENT ENRICHMENT AND DISSOLVED OXYGEN

Impacts of nutrient enrichment might be reflected in food web shifts of finfish, particularly those of larval forms, or in direct effects due to low DO.

The correlation of abundance and survival of fresh water spawning species such as striped bass with the density of zooplankton food organisms (Setzler et al. 1980, Martin et al. 1982) indicates the importance of plankton food webs to finfish. Hypothetically, the shifts in the phytoplankton community to less desirable forms might result in lower zooplankton abundance (Officer and Ryther 1980). Eutrophication with associated algal and water quality problems may be one reason for striped bass failing to use previous spawning areas near Washington, DC (Lippson et al. 1979). However, as discussed in the section on oysters, there is little evidence that major changes have occurred in phytoplankton communities in most other Bay areas. This could be, admittedly, due to the paucity of data.

However, in the Chowan estuary in North Carolina, current work is building a case for a nutrient enrichment-food web shift effect on fish.[24] Increased nutrient loading from nonpoint sources has caused proliferation in blue-green algae and in other small forms. A decline in the size and abundance of herbivorous zooplankton accompanied this trend. A reduction in the growth rate of larval and juvenile blueback herring, which are spawned in the area, has been observed. Gut analysis has shown a shift toward benthic feeding in these normally planktivorous fish. These results are preliminary, but observed effects are consistent with the food chain impact hypothesis.[25]

One major direct impact on finfish, both adults and juveniles, could be observed increases in the extent and duration of hypoxic events. Tolerances of fishes vary with the species, the stage of life cycle, the level of stress from other factors, the previous history of exposure (acclimation time), the temperature of the water, and other intrinsic and extrinsic factors (Anon. 1976). However, a sufficient body of information has been developed to set criteria levels objectively and to suggest the general effects expected at various concentrations of DO.

To quote from the Environmental Protection Agency's Quality Criteria for Water (Anon. 1976):

> Dissolved oxygen historically has been a major constituent of interest in water quality investigations. It generally has been considered as significant in the protection of aesthetic qualities of water as well as for the maintenance of fish and other aquatic life. Traditionally, the design of waste treatment requirements was based on the removal of oxygen demanding materials so as to maintain the DO concentration in receiving waters at prescribed levels. Dissolved oxygen concentrations are an important gauge of existing water quality and the ability of a water body to support a well-balanced aquatic fauna. A minimum concentration of DO to maintain good fish populations is 5.0 mg L^{-1} (3.6 ml L^{-1}).

The reader is referred to a documented discussion of the rationale for the selection of 5.0 mg L^{-1} (3.6 ml L^{-1}) in the EPA Quality Criteria for Water (Anon. 1976, pp. 123 to 127). It is of interest to note that the EPA DO criterion is based principally on freshwater fish.

Davis (1975) established minimal DO requirements for fish and other aquatic life in Canada using an extensive review of available literature. The oxygen criteria for fish are based on oxygen response thresholds for freshwater, marine, and salmonid fishes. The data were treated in a fashion to normalize the different experimental conditions, fish tested, and factors such as temperature.

Three levels of protection were devised by Davis (1975) to provide flexibility in interpretation:

"Level A is one standard deviation above the mean response level for the group and represents an oxygen level that assures a high degree of safety for very important fish stocks in prime areas.

Level B represents the oxygen level where the average member of a species in a fish community begins to exhibit symptoms of oxygen stress. Some degree of risk to a portion of fish populations exist[s] at this level if the oxygen minimum period is prolonged beyond a few hours.

Level C At this level a large portion of a given fish population or fish community may be affected by low oxygen. The values are one standard deviation below the B level or class average for the group. The deleterious effect may be severe, especially if the oxygen minimum is prolonged beyond a few hours."

The pertinent criteria for water temperatures of 0 to 25C adapted from Davis (1975) are given in Table 36.

Thornton (1975) compared the DO tolerances of some 24 mid-Atlantic estuarine and marine fishes experimentally and from the literature and concluded "that a conservative estimate for maintaining diversity for marsh (estuarine) fish would require minimum permissible DO levels of 2 to 3 ml L^{-1} (2.8 to 4.2 mg L^{-1}) at 20 to 25C." He also ranked 24 species in order of their sensitivity to low oxygen levels. Kaumeyer and Setzler-Hamilton (1982) summarized oxygen tolerances where available for the species of interest in this study, and Carpenter and Cargo (1957) reported on blue crab oxygen tolerances.

Table 37 is a composite listing of oxygen tolerances from the above-mentioned sources.

Doudoroff (1957) and Mackenthun (1969) describe the following probable effects of DO deficiency on fishes based on a review of the literature:

3.6 ml L^{-1} Probably the limiting or critical level below which normally varied fish populations do not persist if subjected to such conditions for long periods of time.

2.1 ml L^{-1} Rapidly fatal at fairly high temperatures to susceptible fishes and to resistant species under otherwise unfavorable conditions of water quality. Known to affect physiology and reproductive processes in a variety of fishes.

1.4 ml L^{-1} Often critical to sensitive forms.

0.7 ml L^{-1} Typically, tolerated only by the most resistant species, although susceptible species may be able to tolerate these levels if they have sufficient time to acclimate to low DO concentrations.

In July 1976, substantial mortalities of surf

TABLE 36.
SUGGESTED LEVELS OF PROTECTION FOR SELECTED GROUPS OF FINFISH

Group	Protection Level	ml O_2L^{-1}	Range of % Saturation (0-25°C)
Freshwater mixed	A	3.85	60-66
fish population	B	2.80	47-48
(no salmonids)	C	1.75	35-36
Marine, non-anadromous	A	6.13	88-100
	B	4.73	69-98
	C	3.15	50-65
Anadromous marine	A	6.30	100
species including	B	4.55	79-94
salmonids	C	2.80	57-58

clams, ocean quahogs, and other benthic animals, including fishes, were recorded in an 8,600 square kilometer[26] area of the New Jersey continental shelf. At the height of the event, DO values were measured at 0 to 2 ml L^{-1} within the area (Swanson and Sindermann 1979). An intensive investigation by government and academic scientists revealed that, although there were a few recorded finfish mortalities, adult finfish, in most cases, were able to avoid the low DO area barring some form of entrapment. Although adult fishes either avoided the large area of anoxic water or perished, the area is a spawning ground for a host of fishes. One may conclude that spawning was disrupted off New Jersey during the summer of 1976 (Azarovitz et al. 1979). It should also be obvious that commercially or recreationally important species are typically no longer available to fishermen in areas experiencing such stresses.

It is apparent from Table 37 and from the criteria established by Davis (1975) that most of the species listed demonstrate behavioral stress, and probably some form of metabolic stress, when oxygen concentrations fall below 3 ml L^{-1}; the majority of the species will die if subjected to oxygen concentrations of less than 0.5 ml L^{-1} for a day or more.

In another portion of this report (Chapter 1), the history of anoxic water intrusion in Chesapeake Bay is described, and evidence of the increase in the anoxic condition during the summer (July) since 1950 in the area between Gibson Island above the Bay Bridge to a line approximately between Tangier Island and the mouth of the Great Wicomico River is provided (Figure 53). These data are summarized below for July 1950 and July 1980 in Table 38.

By comparing the tolerances of representative species (Table 37 and Doudoroff 1957, Mackenthun 1969) with the amount of potential habitat excluded to them by DO levels below their tolerances (Table 38) one may offer the following conclusions about current (1980) conditions in Chesapeake Bay:

1. Nearly 100 percent of the 23,000 x 10^6 cubic meters of mid-Bay water during Ju-

TABLE 37.
SUMMARY OF OXYGEN TOLERANCE DATA INVOLVING PRINCIPAL CHESAPEAKE BAY SPECIES

Species		LD_{50} (ml L^{-1}) Death occurs	EC_{50}(ml L^{-1}) Stress behavior
Alosa sapidissma	American shad	2.0–3.6	1.7–2.1
Alosa aestivalis	Blueback herring	1.4–2.1	—
Pomatomus saltatrix	Bluefish	0.9	2.3
Crassostrea virginica	American oyster	0.7	1.7
Menidia menidia	Atlantic silverside	0.6	1.1
Ictalurus punctalus	Channel catfish	0.5–0.6	—
Ictalurus nebulosus	Brown bullhead		4.8–4.9
Morone saxatilus	Striped bass		
	Larvae, Juveniles	0.5	2.9
	Adults	—	3.0*
Morone americana	White perch	0.4–0.7	2.0
Callinectes sapidus	Blue crab	0.5	1.8
Brevoortia tyrannus	American menhaden	0.4	1.0
Leiostomus xanthurus	Spot	0.4	0.9–2.7
Fundulus heteroclitus	Mummichog	0.04	0.9–3.2**

Note *—adults avoid areas with less than 44 % oxygen saturation.
**—reduced hatching of eggs below this level.

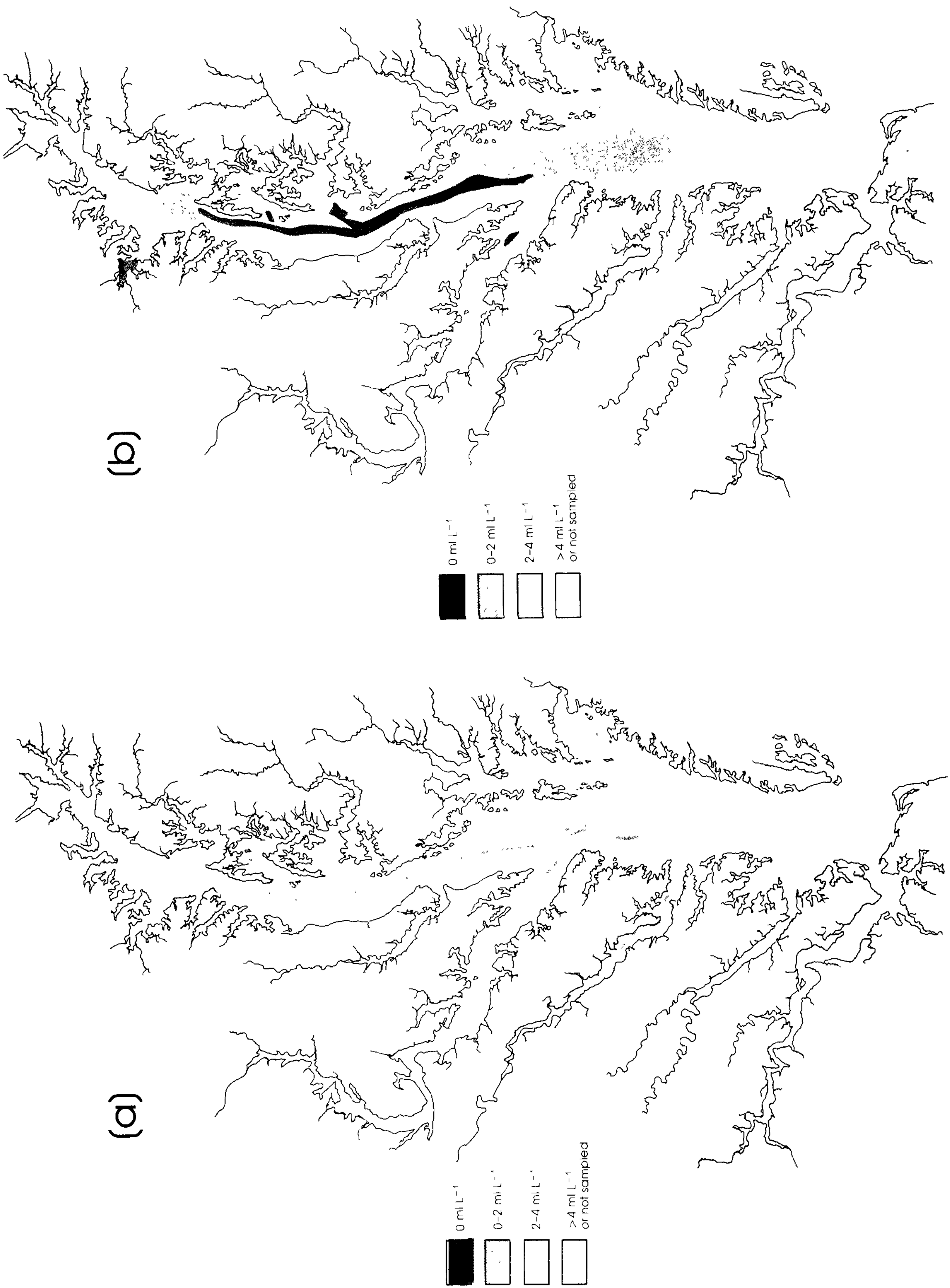


FIGURE 53. Extent of anoxic bottom water in the main stem of Chesapeake Bay in (a) 1950 and (b) 1980.

TABLE 38.
VOLUME OF WATER AFFECTED BY REDUCED OXYGEN CONCENTRATIONS EXPRESSED AS MILLIONS OF CUBIC METERS AND AS THE PERCENT OF THE ESTIMATED TOTAL VOLUME OF AQUATIC HABITAT AVAILABLE (23000×10^6 CUBIC METERS). ERROR OF ESTIMATE IS APPROXIMATELY ± 5 PERCENT

	July 1950		July 1980		
Oxygen Concen. ml L^{-1}	$m^3 \times 10^6$	Percent Affected	$m^3 x 10^6$	Percent Affected	Oxygen Isopleth depth, below surface meters
0.5	294	1	4294	19	13
1.0	1366	6	6444	28	12
2.0	6445	28	10680	46	10
3.0	10702	45	13488	59	7
4.0	15396	67	23000	100	4

ly is estimated to be below or barely above the EPA criterion (3.6 ml L^{-1}) for maintaining healthy fish populations. This proportion of habitat loss has increased from 67 to 100 percent since 1950 (Table 38).

2. Nearly 60 percent of the total available habitat is estimated to have DO concentrations below the level (3.0 ml L^{-1}) thought to produce respiratory, metabolic, and behavioral stresses on most fish species. This proportion of habitat loss has increased from 45 to 59 percent since 1950.
3. Approximately 20 percent of the total available habitat is estimated to have DO concentrations below the level (0.5 ml L^{-1}) necessary to sustain life in all but a few hardy fish species. This proportion of lethal anoxic habitat has increased from 1 to 19 percent since 1950.

In summary, during those years in the summer to early fall, when natural hydrographic conditions and anthropogenic inputs facilitate anoxic conditions, the majority of fish species would be excluded by potential lethal conditions from about 20 percent of the deeper waters of the mid-Bay area; very likely, they would be excluded by behavioral avoidance from about 60 percent of the mid-Bay waters, leaving only shallow (2 to 3 meters) surface waters available for feeding and spawning.

The fishes and shellfish most likely to be affected by the mid-Bay anoxic conditions are those that are primarily benthic feeders (e.g., spot, croaker, and channel catfish), Bay spawners that spawn in mid-late summer (e.g., Bay anchovy, Atlantic silversides, and weakfish), and benthic dwellers (e.g., Blue crab, soft clam, and the oyster) in waters below approximately 7.0 meters in depth. Of the species mentioned as examples, all have shown declines in the last decade except for blue crabs and weakfish. Benthic species were discussed more fully in Section 5.

Table 39 [adapted from Lippson et al. (1979) and Shea et al. (1980)] shows that a variety of species are thought to frequent the deep trough area of Chesapeake Bay. Those species that would typically enter the area to feed or spawn during the period from May through October would most likely be excluded from the area by hypoxic conditions. Those species potentially affected would include striped bass (Flemer et al., draft manuscript), summer flounder, spot, croaker, silver perch, black and red drum, Bay anchovy, blue crabs, oyster, and menhaden, as well as many other organisms not listed.

The exclusion of a variety of species from the

TABLE 39.
PERIOD OF UTILIZATION OF DEEP WATER BY ECOLOGICALLY AND COMMERCIALLY IMPORTANT SPECIES OF FINFISH AND BLUE CRABS

Adult winter flounder	November—May
Adult summer flounder	April—November
Adult and juvenile striped bass	October—March
Adult and juvenile white perch	November—March
Gizzard shad	November—April
Spot	April—October
juvenile	March—May
Atlantic croaker	March—November
juvenile	September—April
Silver perch	June—October
Black and red drum	June—October
Bay anchovy, adult and juvenile	November—April
Atlantic silversides	only during coldest weather
Male blue crabs (upper, mid-Bay)	October—April
Female blue crabs (lower Bay)	October—April
Menhaden, post larvae	January—March
pre-juvenile and juveniles	March—October

deep trough from May through October has been substantiated by a trawl survey conducted by the Maryland Department of Natural Resources (Gucinski and Shaughnessy 1983). They collected 19 species from the area in November 1982 when oxygen values for most of the area were above 7 ppm (5.0 ml L^{-1}). In contrast, from May through October in 1982, only five or fewer species were collected from the area; these typically were from shallow, more highly oxygenated stations. During this period, there was a substantial volume of completely anoxic water in deeper channels; values in the majority of the area were less than 5 ppm (3.6 ml L^{-1}) (Gucinski and Shaughnessy 1983). During the August and October 1982 sampling efforts, no fish or crabs were collected at the six deepest stations when DO levels were determined to be less than 1 ppm at these depths.

Although it is not possible to quantify precisely the effects of habitat exclusion on individual fisheries, it is fair to say that for the majority of species treated in this analysis, the reduction of available habitat for spawning and feeding must have a depressing effect on local fisheries.

IMPACT OF NATURAL VARIABLES

There is no doubt that natural variables—in particular those related to climate—have major impacts on finfish. Precipitation, runoff, wind speed and direction, and temperature can directly and indirectly affect the success of various species in different ways and at different times of their life cycles. For example, Ulanowicz et al. (1981) and Heinle et al. (1980) report that the success of alewife and shad is related to high winter temperatures; the success of blue crab, menhaden, soft clam, striped bass, and white perch is related to colder winter temperatures. Croaker and spot have been shown by Norcross and Austin (1981) and Wojcik and Austin (1982) to suffer severe winter kills when winter temperatures are unusually low. Menhaden may be a special case; although its success is related to colder inland winter temperatures, Sutcliffe et al. (1977) have shown that its success is also related to high winter sea-surface temperatures.

The effects of increased precipitation on fisheries manifest themselves principally in the reduction of salinity in the estuary, in the increases in dissolved nutrients and suspended solids, and in the increases in the physical effects of flow through the estuary. The latter effects, when extreme, can reduce the spawning habitat of freshwater spawning species by the disruption of spawning sites. At the same time, flow can enhance transport of marine-spawned species into the estuary by stratifying the vertical water column and facilitating the upstream flow of the bottom "salt layer" (Tyler and Seliger 1978). Herrgesell et al. (1981) and Anon. (1981b) report that the improved success of alewife, blue crab, oyster, shad, soft clam, striped bass, and weakfish is related to increased freshwater flow, particularly at the time of spawning and early larval life. The positive effects of increased runoff are thought to include: (1) increased transport of plant nutrients and detritus to the nursery area that increases phytoplankton and zooplankton populations on which shellfish and fish larvae feed; (2) a physical expansion of the optimal nursery area, particularly for fresh to slightly brackish spawners; and (3) increased upstream transport of estuarine and marine-spawned larvae.

Colder winter temperatures can affect the timing and volume of spring runoff through ice and snow formation. During colder winters, spring runoff (with its associated detritus and dissolved nutrient load) and increases in water temperatures are delayed. Theoretically, in such cases, the positive effects of runoff may be more in phase with the spawning times of species like the striped bass (Mihursky et al. 1981).

The striped bass requirement for a flowing water nursery (Setzler et al. 1980) undoubtedly makes this species unique. The bass not only requires higher freshwater flows for survival and hatching of its eggs, but probably tolerates higher flows than other freshwater spawning species. These special requirements also appear to have caused the striped bass to shift its principal ancestral spawning area from the Susquehanna Flats to the Chesapeake and Delaware Canal because of the optimal current velocities for suspension and survival of the eggs in the latter location (Dovel and Edmunds 1971).

Most of the work to date relating climate and other factors to finfish utilizes the statistical regression approach. It should be emphasized that these correlations are suggestive, but not a definitive explanation of cause and effect. Information on the role of climatic variables in explaining natural variability is a necessary condition to the assessment of anthropogenic effects.

JUVENILE INDEX

Young-of-the-year juvenile finfish collected in four representative tributary areas of the Bay (Head of Bay, Potomac River, Choptank River, and Nanticoke River) were used to assess the impact of various natural environmental variables on finfish. This juvenile index is believed to be a better index of stock abundance than the direct use of landings because it is influenced less by fishing pressure and other factors. Though not immune to uncertainty as an index of stock abundance (Polgar 1982), the juvenile index was correlated with environmental variables to elucidate possible factors that affect the recruitment of young fish into the harvestable population. Details of these analyses can be found in Section 4 of Appendix D.

FINFISH JUVENILE INDEX REGRESSION ANALYSIS

The relationship of the finfish juvenile index to natural and water quality variables was addressed through regression analysis (details of the analysis are in Appendix D, Section 4). Linear regression and multiple regression analysis were performed.

Linear Regression Analysis

By linear regression analysis, the juvenile index was compared with freshwater inflow and air temperature in the four tributaries. Results are summarized in Table 40. In general, species responded positively to increases in flow and air temperature. In the Northern Bay, alewife responded negatively to February and March flows, which may be related to water temperature. The same may be true for anchovy and silverside. In both the Potomac and the Nanticoke, striped bass responded negatively to increased April air temperatures.

Although Table 41 indicates some subtle dif-

TABLE 40.
RESULT OF LINEAR REGRESSION ANALYSIS OF JUVENILE INDEX AGAINST AIR TEMPERATURE

Species	Basin	Time	Correlation Coefficient	P ≤ .05
Alewife	Choptank	Feb. & March	−.46	.0281
Spot	Choptank	Feb. & March	.43	.0381
Spot	Choptank	Feb., March & April	.44	.0351
Atlantic Menhaden	Potomac	Feb. & March	.49	.0165
Bluefish	Potomac	Feb. & March	.66	.0007
Catfish	Potomac	Feb. & March	.45	.0312
Spot	Potomac	Feb. & March	.48	.0209
Atlantic Menhaden	Potomac	Feb., March & April	.58	.0037
Bluefish	Potomac	Feb., March & April	.73	.0001
Catfish	Potomac	Feb., March & April	.52	.0109
Spot	Potomac	Feb., March & April	.49	.0170
Atlantic Menhaden	Potomac	March	.54	.0078
Bluefish	Potomac	March	.56	.0051
Spot	Potomac	March	.48	.0210
Striped Bass Age 0	Potomac	April	−.49	.0178
Atlantic Menhaden	Upper Bay	March	.51	.0136
Yellow Perch Age 0	Upper Bay	March	−.46	.0286
Weakfish	Upper Bay	April	−.42	.0447
Munnichog	Choptank	February	−.48	.0216
Yellow Perch Age 0	Nanticoke	February	−.52	.0101
Spot	Nanticoke	March	.42	.0475
Striped Bass Age 0	Nanticoke	April	−.44	.0360
Spot	Choptank	Spring	.52	.0103

ferences among species and among river basins as they relate to flow, the most believable results are those represented by the combined basins (aggregated flows and aggregated juvenile indexes). This approach shows that striped bass responds positively to strong spring flows, results that agree with Mihursky et al. (1981). The marine spawners, bluefish, menhaden, and spot are responding positively to strong fall, winter (which are combined as "late"), late and annual flows. This argues for the estuarine transport of the larval and juvenile forms of these species by the upstream migration of the bottom waters (Tyler and Seliger 1978).

Multiple Regression Analysis — Analytical Methodology

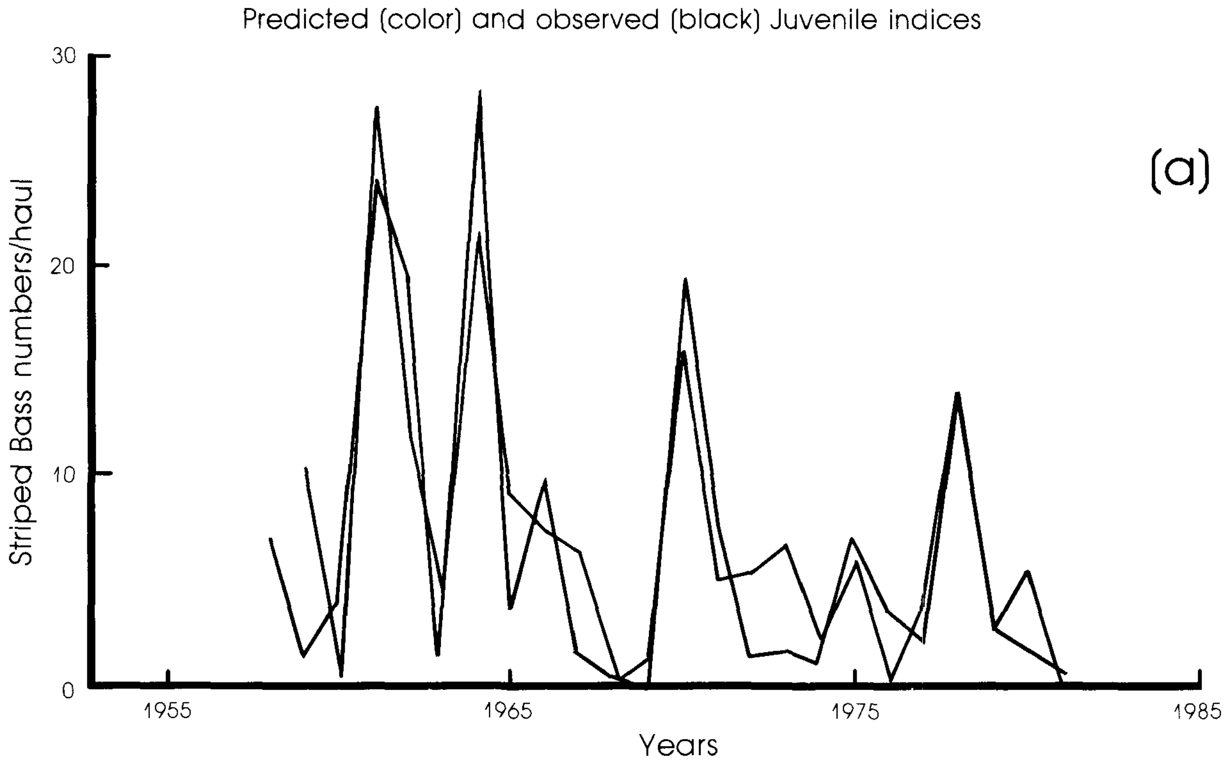
A multivariate regression analysis was used to

TABLE 41.
RELATIONSHIP AS REPRESENTED BY R VALUES AND DETERMINED BY CORRELATION ANALYSIS ($P \leq 0.05$) FOR FINFISH JUVENILE INDEX VERSUS FLOW ($N = 24$)

Species	Annual Flow	Winter Flow	Spring Flow	Summer Flow	Fall Flow	Early Flow	Late Flow
Choptank River							
Alewife				0.40			
White perch		−0.42					
Menhaden	0.48	0.50				0.56	
Mummichog		0.51				0.46	
Nanticoke River							
Anchovy	−0.49	−0.44	−0.43			−0.49	
Potomac River							
Striped bass			0.38				
Bluefish		0.43					
Silversides	−0.46	−0.53				−0.46	
Upper Bay							
Spot					0.59		0.60
Striped bass			0.47				
Bluefish		0.51					
Silversides	−0.54	−0.49	−0.41			−0.53	−0.42
Combined Basins							
Striped bass			0.45				
Bluefish	0.42	0.52			0.43		0.52
Menhaden		0.60			0.46		0.41
Spot	0.45	0.42			0.67		0.65
Silversides	−0.60	−0.49			−0.43	−0.54	−0.51

identify the freshwater variables that best explain observed trends in the juvenile index. Maximum and minimum freshwater inflows, as "windows" of flow in periods from 7 to 28 days, and air temperature were used. Emphasis in the analysis was placed on freshwater spawners and selected forage fish because these species spawn within the Bay system, including fluvial streams; they were hypothesized to have sensitive young life stages exposed to higher concentrations of natural and anthropogenic factors than marine spawners.

In lieu of the non-continuous record of the water quality data, the initial analyses included only the juvenile indices, air temperature (surrogate of water temperature), and stream flow. For all months, the juvenile indices were regressed in a step-wise fashion using a maximum r^2 improvement against streamflow and air temperature. Predictive models for each juvenile index species in each basin were obtained, and equations using air temperature and streamflow were derived (examples shown in Figure 54). Through the use of the residuals from each statistically significant equation, other water quality variables were tested. Because of infrequent data in the Choptank and Nanticoke Rivers, the water quality



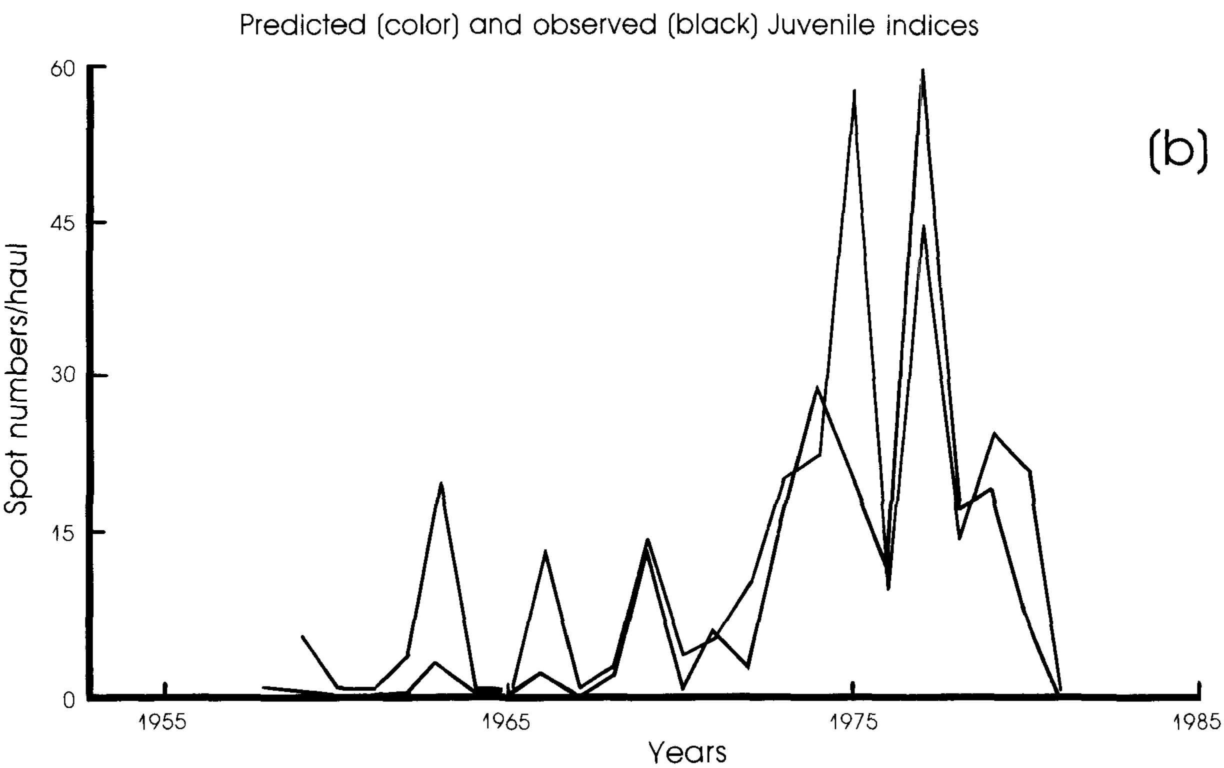


FIGURE 54. Observed and predicted juvenile indices for (a) striped bass in the Potomac River and (b) spot in the Choptank River based on models derived from multiple regression analysis.

tests in these Rivers were excluded. Monthly MAX R^2 stepwise regression of water quality variables including salinity, TN, TP, DO, and chlorophyll was performed against the residuals from the physical models to see if improvement can be made on the models. Because of the infrequent number of years available, these results may be considered suggestive only.

Summary of Results from Juvenile Index versus Water Quality—Multiple Regression Models

Although up to four significant variables were allowed to enter multiple regression models describing relationships between the success of juvenile finfish indexes and various water quality parameters, the first variable to enter was accepted as having the best biological credence. Based on moving 7, 14, 21, and 28 day windows, almost all relationships with minimum and maximum flows were established in April and May, reflecting possible influences on spawning times for freshwater spawners and larval recruitment times for marine spawners. However, there was almost no consistency within a species, among spawning groups (fresh or marine) or basin locations. Based on monthly means for temperature significant positive and inverse relationships were established in March, April, and May possibly relating to spawning and recruiting; the relationships again lacked consistency within species or among spawning groups and basins.

The two relationships, showing consistency with the literature, include the positive relationships among winter temperatures (November to December) and the juvenile index for bluefish, catfish, menhaden, and spot (this is consistent with the findings of Dow 1975) and an inverse relationship in the Potomac basin among colder than normal winters and the success of striped bass year classes or the juvenile index (this agrees with Mihursky et al. 1981).

IMPACTS OF OVERFISHING

Fishing stocks may be reduced by growth over-fishing and recruitment over-fishing (Cushing 1975). In the former, recruitment is not affected by over-fishing, but fishing mortality is so high that young fish are not given the chance to grow, perhaps not even to reach reproductive maturity. A local example was seen in the Atlantic menhaden fishery during the 20th century where the fishery was initially established for approximately ten-year-old fish and has progressively reduced the availability of older fish until now. At present, one- to two-year old fish constitute the great majority of the catch (Henry 1965, Price 1973, and Cronan 1981).

The other form of over-fishing is recruitment over-fishing where death by fishing is great enough to reduce recruitment. Herring (Cushing and Bridger 1966) and whales are examples of fisheries that have suffered from recruitment over-fishing (Cushing 1975).

Another way of stating the issue is that recruitment is dependent upon stock density in recruitment over-fishing and independent of stock density in growth over-fishing. Typical of both forms is that the catch per unit of effort is reduced as over-fishing progresses (Cushing 1975).

Typically, however, with the exception of a few carefully studied fisheries in Chesapeake Bay (e.g., the Atlantic menhaden), it is virtually impossible to predict levels of over-fishing. Catch statistics (landings) cannot be related to species abundance because the effects of fishing effort, reporting error, and market demand are virtually unknown and cannot be adjusted for using currently collected statistics (Rothschild et al. 1981). Bortone (1982) attempted to develop predictive models for Chesapeake Bay fisheries relating landings (catch) to quasi-effort statistics such as numbers of boats, numbers of fishermen, numbers of nets, etc. and was unsuccessful in establishing meaningful relationships among these parameters.

Therefore, although there has been a long-standing concern that over-fishing by both commercial and sports interests has caused declines in local fisheries (McHugh 1981 and Williams et al. 1982), there is little statistical evidence to support the claim or to interpret the level of effect for the majority of Chesapeake Bay fisheries. It is interesting to note that one important conclusion of the Emergency Striped Bass Research Study (Anon. 1981a) is that "the level of fishing mortality affects the magnitude of the contaminant mortality that the stock can withstand and vice versa." That is to say, over-fishing can reduce the resiliency of the stock to rebound from toxic material stresses and vice versa.

These two forces may well be reducing the stocks of anadromous (fresh water spawners) species as evidenced by their measurable decline during the past ten years (Chapter 2).

HABITAT MODIFICATION

Habitat modification — construction of dams or similar structures, draining of wetlands, or modification of stream channel through dredging or channelization — can have significant impacts on the spawning success of anadromous and semi-anadromous fish, or on the suitability of the area for a finfish nursery.

Dams

Large dams and impoundments can alter the volume and timing of freshwater inflow to the estuary, which in turn may affect downstream salinity, circulation, transport, and flushing. This in turn can affect habitat for numerous estuarine species, including finfish (Shea et al. 1980, Mackiernan et al. 1982). Water quality downstream is often affected: impounded water warms more slowly in spring and remains warmer longer in autumn; water released from depth may be deficient in DO or significantly colder than ambient river water; released surface waters typically contain more algae and less silt than river water (Ridley and Steel 1975).

One major effect of dams and other in-stream structures is the blockage of spawning migrations of anadromous or semi-anadromous species. The impact of the barrier depends on its location and physical structure, as well as on the species of fish involved. For example, alewife and salmonids may pass barriers that halt shad, sturgeon, or striped bass (Fefer and Shettig 1980). Barriers may be man-made or natural (waterfalls, beaver dams, logjams). Some relatively small barriers, such as poorly-designed road culverts, may block passage of white or yellow perch (O'Dell et al. 1980). The extent of the potential impact is documented by a survey of anadromous fish spawning areas done from 1975 to 1979 by the Maryland Department of Natural Resources (O'Dell et al. 1980). For example, within the Chester River study area, 153 constructed barriers were documented on 120 streams; eight streams had natural barriers. Of the 120, thirteen streams had completly blocked migration of such species as white and yellow perch, alewife, and blueback herring. The reduction of shad due to dam construction on the Susquehanna has been well documented (Weinrich et al. 1980).

Wetlands Loss

Loss of wetlands adjoining Chesapeake Bay and its tributaries can occur through natural or anthropogenic processes: the latter include filling, draining, dredging, ditching, spoils disposal, and other development. It is estimated that approximately 23,700 acres (approximately 7 percent) of tidal wetlands were lost in Maryland between 1942 and 1967; in Virginia, 4000 acres were lost between 1955 and 1969 (Metzgar 1973, Settle 1969). The amount of non-tidal wetlands lost is unknown, but probably significant. Daiber et al. (1976) estimate that 80 to 90 percent of Chesapeake Bay's total seafood harvest is dependent on wetlands at some point in its life cycle. This dependence may be direct, through habitat or grazing, or through the detrital food-web pathway. Commercially or recreationally important animals such as furbearers or waterfowl also heavily use wetlands heavily for food or habitat. Wetlands also act as buffers for habitat, through sediment trapping, erosion control, floodwater absorption, and nutrient trapping and transformation (Metzgar 1973, Davis 1978, Nichols 1983). Marshes may also absorb and trap other pollutants such as metals and pesticides (Gallagher and Kibby 1980).

Because of their value as sources of detritus and as habitat buffers, the loss of wetlands in areas impacting anadromous fish spawning and nursery grounds could represent a significant problem.

Other Modifications

Stream channelization, bulkheading, and watershed modification can all have significant impacts on water quality or physical suitability of a habitat for finfish. The clearing of streamside vegetation often results in warming of the stream; the removal of marginal wetlands eliminates the previously discussed "buffering" capacity and allow a more rapid transport of water from the surrounding areas downstream (Spier et al. 1976). In an investigation of the effects of channelization on small coastal plain streams in Maryland, Spier et al. (1976) found substantial amounts of sediment generated during the active construction phase, the period of initial stabilization (which is greater than 3 years in these streams), and through induced erosion downstream. Biological impacts noted were the reduction in the diversity and type of benthic communities, compared with unmodified and older modified streams, although biomass levels were not significantly different (Spier et al. 1976). Fish populations in these small, relatively unstable streams, were not so significantly impacted as were the larger coastal plain streams studied by Tarplee et al. (1971). However, impacts on anadromous species in these and impacted downstream areas were not evaluated in Spier et al. (1976). Streams appeared to shift from allochothonous detrital-based food webs to autochothonous production because of (in stabilized streams) aquatic and streamside vegetation. In North Carolina, ditching and drainage practices in coastal plain streams were shown to create unstable salinity and food supply conditions in downstream nursery areas, making them less suitable for shrimp, spot, croaker, flounder, and blue crab (Pate and Jones 1981). As a result of this study, it was recommended by the authors that (among other things) consideration be given to the cumulative impacts of extensive alteration of small estuaries, and that wooded swamps and marshes be utilized as natural filters for upland drainage.

The modification of watersheds due to urbanization and other development, particularly the increase in area of impervious surfaces, has been related to declines in water quality in small streams (Klein 1979). Nutrients and toxic substances (particularly heavy metals, oil, and salt) are elevated and fish diversity reduced in direct relation to the amount of imperviousness in the watershed.

SUMMARY

A variety of factors impact finfish, and natural variables represent major influences on fish distribution, abundance, and recruitment. However, evidence suggests that anthropogenic stresses also have the capacity to affect fish populations; such effects may be occurring in Chesapeake Bay. In particular, nutrient and toxicant enrichment of low-salinity spawning and nursery areas may be related to recent declines in anadromous species. The loss of potential habitat in large areas of the upper and mid-Bay due to summer low DO is well-documented. Demonstrating clear effects on finfish populations will not be a simple a task, however. Though the effects of harvest, particularly on already declining populations, may hypothetically be significant, sufficient data do not exist to demonstrate such impacts. Finally, habitat modification, especially construction of dams, has the potential to seriously impact important freshwater-spawning finfish.

SECTION 7
SUMMARY

In general, a number of the hypotheses formulated in the preceeding chapter appear supported by available information. Other relationships may exist in Chesapeake Bay, but our data are insufficient to demonstrate clear linkages. However, comparison with information from other regions, as well as from microcosm experiments, lend credence to the hypotheses. Finally, certain hypotheses cannot be supported by the data now available; further monitoring or research is suggested.

SUBMERGED AQUATIC VEGETATION

Both the field data and the laboratory analyses support the hypothesis that increases in nutrients represent the major impact on SAV in Chesapeake Bay. Submerged aquatic vegetation appear to be responding primarily to forms of nitrogen and to IPF on a Bay-wide basis. Whether the mechanism is operating through increases in phytoplankton biomass or epibiota on SAV leaves, or both, is not certain. However, negative correlations of SAV abundance with chlorophyll *a* concentrations lend support to the phytoplankton hypothesis. There appears to be little field or laboratory evidence to support the hypothesis that herbicides are a primary cause of SAV declines. Localized effects, particularly in small tributaries or on populations already light-limited, may occur.

BENTHIC ORGANISMS (INCLUDING OYSTERS)

Benthic organisms, including commercial shellfish such as oysters, are possibly affected by a number of factors. Where sediment contamination by toxicants is relatively severe, significant changes are observed in benthic community abundance and structure. However, when sediment toxicant burdens are slight, then other factors including natural variables and biological interactions such as predation appear to control benthic organisms. The proposed loss of benthic habitat due to low DO — linked to nutrient enrichment — is supported by field observations. This phenomenon may also be related to observed mortalities of shellfish in affected areas. Finally, the accumulation of toxicants in tissues of benthic organisms may have physiological impacts, or may be magnified in the Bay food web.

FINFISH

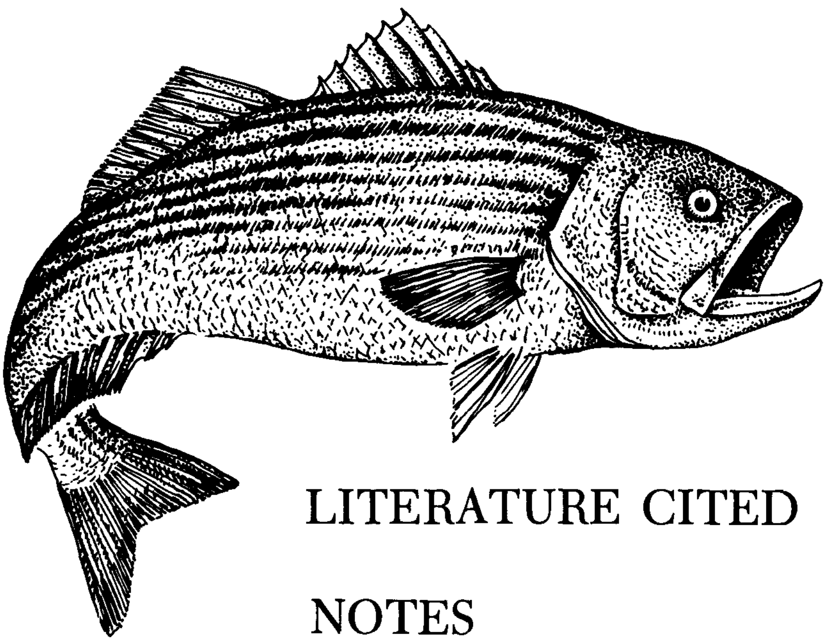
As with benthic organisms, finfish appear to respond to a number of factors, both natural and anthropogenic. There is evidence that the levels of toxicants observed in some areas of the Bay are within the range to affect survival or health of larval fishes. However, no clear cause and effect can be demonstrated at this point; the relationship remains hypothetical. There is little support from Chesapeake Bay for the hypothesis that food-web shifts have been impacting anadromous species, except in a few areas. Again, this is primarily due to lack of extensive data on spatial and temporal trends in phytoplankton and zooplankton populations. However, the loss of habitat for fishes in the Bay main-stem due to summer low DO is supported by field observations. Whether this loss of habitat is producing major impacts on finfish populations is as yet unknown. The hypothesis that finfish populations are responding strongly to natural, climatic variables is supported by a number of correlative techniques. In fact, the difficulty in assessing anthropogenic impacts is primarily due to the strong influence of natural factors. However, it should be kept in mind that even a relatively small additional impact on some populations could eventually result in significant effects. Finally, there are insufficient data to address the subject of possible over-fishing of some species, although ongoing studies by other programs (particularly the Striped Bass Study group) may shed light on this subject.

CONCLUSION

As anticipated in the introduction to this chapter, there is considerable difficulty in demonstrating unequivocal "cause and effect" between changes in environmental quality and trends in living resources. At best, we can show significant correlations, which support reasonable hypotheses as to potential effects. When these correlations are further strengthened by the results of research or monitoring performed under relatively controlled conditions, the cases become stronger and more compelling.

We stated earlier that "the correspondence of the patterns infers that the nutrient and toxicant enrichment reflect . . . human intervention in the Chesapeake Bay ecosystem, and that at least some trends in SAV, shellfish, and anadromous fish reflect the integrated response to that intervention." Based, in part, on this assessment, Bay managers will decide what actions are appropriate.

The Bay ecosystem is not amenable to exact solutions because uncertainty always will surround the causes of problems. The CBP anticipates that future research and monitoring will continue to reduce uncertainty. However, prudent management will not wait for science to provide a precise level of assurance or run the risk of the "ultimate experiment." It is contemplated that a coherent approach to problem identification and probable cause will yield dividends in spite of our acknowledged uncertainties.



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NOTES

1. 1 m = 3.28 ft
2. Personal communication: "Total Residual Chlorine Criteria," Bill Brungs, U.S. EPA Naragansett, 1983.
3. 1 km = 0.62 mi.
4. Personal Communication: "Phytoplankton Cell Numbers in the Potomac River," Ron Cohen, USGS, 1982.
5. Personal Communications: "Wetlands Inventory," Anonymous, Maryland Department of Natural Resources, 1983.
6. Personal Communication: "Wetland Loss," G. Silberhorn, Virginia Institute of Marine Science, 1983.
7. [1°C = 5/9(°F − 32)]
8. Personal communication: "Oyster Harvest in the James River," Dexter Haven, VIMS, 1982.
9. Personal communication: "Oyster Harvest in Maryland," Harold Davis, Maryland Department of Natural Resources, 1982.
9. Personal communication: "Juvenile Finfish Survey," Joe Boone, Maryland Department of Natural Resources, 1982.
10. Personal communication: "Phytoplankton Cell Numbers in the Potomac," Ron Cohen, U.S.G.S., 1982.
11. Personal communication: "Phytoplankton Cell Numbers in the Potomac," Ron Cohen, U.S.G.S., 1982
12. Personal communication: "Oyster Biology," Dexter Haven, VIMS, 1982.
13. Personal Communication: "Oyster Mortality," G. Krantz, Horn Point Environmental Laboratory, 1982.
14. Personal Communication: "Finfish Habitat," H. Austin, VIMS, 1982.
15. 1 km = 5/8 mile
16. Personal Communication: "Blue Crab Recruitment," C. Epifanio, University of Delaware, 1982.
17. Personal Communication: "SAV Transplant Experiments," V. Carter, USGS, 1982.
18. Personal Communication: "Multiple Regression Interpretation," R. Ulanowicz, Chesapeake Biological Laboratory, 1982.
19. Personal Communication: "Results of Microcosm Experiments with Benthos," R. Alden, Old Dominion University, 1983.
20. Personal Communication: "Effects of Organic Materials on Benthos in MERL Tanks," C. Oviatt, U.R.I. 1982.
21. The Toxicity Index was developed in an attempt to modify the Contamination Index to better predict potential biological impact of contaminated sediments. It is based on weighting the contamination factors for the six metals by their relative toxicities from bioassay information. The analysis is described in detail in Appendix D, Section 1.
22. Personal Communication: "Oyster Mortality," G. Krantz, H.P.E.L., 1982.
23. Personal Communication: "Oyster Mortality," G. Krantz, H.P.E.L., 1982.
24. Personal Communication: "Preliminary Results of Food Web and Finfish Study," S. Mozley, N.C. State University, 1983.
25. Personal Communication: "Preliminary Results of Food Web and Finfish Study," S. Mozley, N.C. State University, 1983.
26. 1 km^2 = 0.386 mi^2