EPA/620/R-99/004
October 1999

EMAP-Virginian Province
Four-Year Assessment (1990-93)

John F. Paul
Atlantic Ecology Division
National Health and Environmental Effects Research Laboratory
U.S. Environmental Protection Agency
Narragansett, RI 02882

John H. Gentile
Center for Marine and Environmental Analyses
Rosenstiel School of Marine and Atmospheric Science
University of Miami
Miami, FL 33149

K. John Scott
Science Applications International Corporation
Newport, RI 02840

Steven C. Schimmel, Daniel E. Campbell,
and Richard W. Latimer
Atlantic Ecology Division
National Health and Environmental Effects Research Laboratory
U.S. Environmental Protection Agency
Narragansett, RI 02882

United States Environmental Protection Agency
National Health and Environmental Effects Research Laboratory
Atlantic Ecology Division
27 Tarzwell Drive
Narragansett, RI 02882

NOTICE

The information in this document had been funded wholly or in part by the United States Environmental Protection Agency (Environmental Monitoring and Assessment Program, Office of Research and Development) through the Atlantic Ecology Division. Support for SAIC was provided by EPA Contract #68-01-0005, M.P. Gant, Project Officer. It has been subjected to the Agency's peer and administrative review, and has received approval for publication as an EPA document. Mention of trade names or commercial products does not constitute endorsement or recommendation for use. This is contribution no. 1837 of the Atlantic Ecology Division, National Health and Environmental Effects Research Laboratory.

The suggested citation for this report is: Paul, J.F., J.H. Gentile, K.J. Scott, S.C. Schimmel, D.E. Campbell, and R.W. Latimer. 1999. EMAP-Virginian Province Four-Year Assessment Report (1990-93). EPA 600/R-99/004. U.S. Environmental Protection Agency, Atlantic Ecology Division, Narragansett, Rhode Island.

ABSTRACT

Assessments with the four years (1990-93) of ecological condition data collected by the USEPA Environmental Monitoring and Assessment Program (EMAP) in estuaries of the Virginian Biogeographic Province (Cape Henry to Cape Cod) were presented. EMAP data were used to quantify, with confidence, the condition of ecological resources within the broad-expanse of estuarine waters comprising the Virginian Province, as well as its large and small estuarine systems and five major tidal rivers. Over the four-year period, sufficient sampling sites were available to characterize the condition of ecological resources for four major watershed systems within the province (Chesapeake Bay, Delaware Bay, Hudson-Raritan system, and Long Island Sound), and three tidal rivers in Chesapeake Bay (Potomac, Rappahannock, and James Rivers). Results clearly showed that the EMAP objectives were not only reasonable but were achievable with available indicators collected with a probability-based sampling design. It was shown that the EMAP design can be used to quantify with confidence the condition of ecological resources. Reducing the uncertainties in the assessment should be approached through a systematic program of directed research.

Key Words: EMAP; Environmental Monitoring; Virginian Province; estuaries; Indicators (biology); Estuarine Assessment; Chesapeake Bay; Delaware Bay; Long Island Sound

EXECUTIVE SUMMARY

Introduction and Purpose

The scientific community and the public have become increasingly concerned with the apparent widespread extent of environmental impacts from anthropogenic pollutants. Global climate change, acidic deposition, ozone depletion, non-point source pollution, and habitat alteration threaten our ecological systems at regional, national, and global scales. Despite these concerns, the status of the nation's ecological resources has not been well documented, making it difficult to establish quantitatively whether environmental policies and programs designed to limit anthropogenic impacts on natural ecosystems are effective.

In 1988, the U.S. Environmental Protection Agency's Science Advisory Board (SAB, 1988) recommended the implementation of a long-term (10-12 years) program to monitor the status and trends of the nation's ecological resources to identify emerging environmental problems before they reach crisis proportions. The Environmental Monitoring and Assessment Program (EMAP) is the Agency's response to the Science Advisory Board's recommendation.

The Environment Monitoring and Assessment Program is a long-term, nationwide program initiated by EPA's Office of Research and Development (ORD). EMAP's goals are: (1) estimate the current status, trends, and changes in selected indicators of the condition of the nation's ecological resources on a regional basis with known confidence; (2) estimate the geographic coverage and extent of the nation's ecological resources with known confidence; (3) seek associations among selected indicators of natural and anthropogenic stress and indicators of ecological condition; and (4) provide annual statistical summaries and periodic assessments of the nation's ecological resources.

The implementation of EMAP will provide answers to several environmental questions and in so doing provide the information required to formulate environmental protection policies for the 1990s and beyond. For example,

- What are the status, areal extent, and geographical distribution of the nation's ecological resources? How much confidence do we have in such estimates?
- What proportion of these resources is declining or improving? Where? At what rate?
- What factors are contributing to declining condition of ecological resources, that is, can knowledge of exposure and habitat indicators be used to explain observed ecological condition?

- Are pollution control, reduction, mitigation, and prevention programs achieving overall improvement in ecological condition?

To answer these questions EMAP developed a flexible probability-based sampling design that can be scaled to the problem setting, has the power to determine status with known confidence, and is capable of detecting temporal and spatial patterns and trends. In addition, EMAP employed a suite of biotic and abiotic condition indicators to characterize the ecological resources, principal stressors, and habitat conditions.

The EMAP ecological condition data presented in this report were collected over a four-year period (1990-93) from the estuarine waters of the Virginian Biogeographic Province (Cape Henry to Cape Cod). The results clearly demonstrate that the EMAP data can be used to quantify, with confidence, the condition of ecological resources within the broad-expanse of estuarine waters comprising the Virginian Province, as well as its large and small estuarine systems and five major tidal rivers. Analyses of individual year data were used to estimate inter-annual variability. Although individual year data could be used to estimate ecological condition across the province, it results in larger uncertainties because of the smaller sample size. Over the four-year period, sufficient sampling sites were available to characterize the condition of ecological resources for four major watershed systems within the province (Chesapeake Bay, Delaware Bay, Hudson-Raritan system, and Long Island Sound), and three tidal rivers in Chesapeake Bay (Potomac, Rappahannock, and James Rivers). The uncertainties of single-year estimates for these systems were not always acceptable due to the reduced number of sampling sites, particularly in systems like the Hudson-Raritan and the individual tidal rivers.

Areal and Spatial Patterns of Indicators

The benthic communities in the Virginian Province were estimated to be impacted, over the four-year period, in 25$\pm$3% of the estuarine area, with annual estimates ranging from a low of 23$\pm$7% in 1991 to a high of 28$\pm$7% in 1990. Analysis of the three resource classes indicates that the large estuarine systems had the smallest percent area but largest absolute area of impacted benthos, 19$\pm$4% (3099 km^2), with greater percent areal impacts in the small estuarine systems and tidal rivers, 37$\pm$6% and 38$\pm$14%, respectively. Chesapeake Bay, which exhibited benthic impacts of 23$\pm$5%, accounted for 45% of the impacted benthic area within the Virginian Province. The Potomac and Rappahannock Rivers have identical percent impacted areas, 44$\pm$22% and 44$\pm$33%, respectively. The James River, however, exhibited a smaller percent area of benthic impact (19$\pm$23%). Delaware Bay and Long Island sound exhibited benthic impacts over 24$\pm$12% and 28$\pm$11% of their areas, respectively. The percent area of benthic impact in the Hudson-Raritan system, however, was the greatest of the major estuarine systems examined (72$\pm$8%). Together, the four major estuarine systems account for 79% of the impacted benthic area within the Virginian Province.

The percent area of bottom waters in the Virginian Province, *in toto*, with low dissolved oxygen conditions (DO ≤ 2 ppm) was 5±2%. Small estuarine systems had only 1±1%, large systems had 5±2%, and tidal rivers 14±6%. These data suggest that the tidal rivers are most at risk from low dissolved oxygen. Most of the low dissolved oxygen is focused in the main stem of Chesapeake Bay and the mouths of the Potomac and Rappahannock Rivers. Using moderate to severe hypoxia as the criterion (DO ≤ 5 ppm), impacted conditions were observed in 24±3% of the province area, 17±5% of the small systems area, 27±4% of the large systems, and 18±7% of the tidal rivers. These analyses suggest the large estuarine systems are potentially at risk from moderate reductions in dissolved oxygen. The major estuarine system analyses showed that approximately 31% of Chesapeake Bay and 48% of Long Island Sound areas have moderate to severe hypoxia.

The percent area of the Virginian Province sediments having moderate to severe sediment toxicity (survival ≤ 80%) was 9±2%. Examination of individual resource classes showed that the percent area exhibiting sediment toxicity was 4±4% of the tidal rivers, 9±3% of the large systems, and 12±6% of the small systems. These analyses indicate that small estuarine systems are at greatest risk from toxic sediments. Moderate to severe sediment toxicity was observed in 15±14% of the Hudson-Raritan system, 13±7% of Long Island Sound, 6±3% of Chesapeake Bay, and 2±2% of Delaware Bay. Severe toxicity (survival ≤ 60%) occurred in only 1% of the estuarine sediments of the province and was distributed across resource classes. Severe toxicity was observed in Delaware Bay (1±2%), Hudson-Raritan system (7±10%), and Long Island Sound (5±4%).

Sediment contamination condition in the Virginian Province was determined using multiple thresholds for comparison with observed concentration of contaminants and distribution of effects. Anthropogenic enrichment of metals was examined by determining the metals enrichment above crustal levels. Metals enrichment was observed in 49±4% of the province area. Small systems and tidal rivers had a greater enrichment than large systems, 64±7% and 69±10% compared with 42±5%, respectively. Metals enrichment ranged from 85±11% and 86±8% in the Hudson-Raritan system and Long Island Sound, respectively, to 44±5% and 39±14% in Chesapeake and Delaware Bays, respectively.

Sediment contaminant condition for potential biological effects was determined by using exceedence of Effects Range-Median (ER-M) and Effects Range-Low (ER-L) values (Long *et al.*, 1995). Exceedence of an established ER-M for selected trace metals and organic chemicals has been postulated as one method of ranking the contamination of marine sediments. The ER-M exceedence for metals was the greatest in small estuarine systems, 9±4% of the area. The Hudson-Raritan exhibited the greatest percent area with ER-M metals exceedence (27±9%), whereas the other major systems exhibited 1-5% areal exceedence.

Exceedence of at least one ER-M value for organics was found in 4±1% of the area of the Virginian Province. The percent area was relatively low in large and small estuarine systems (2%) and higher in tidal rivers (14±2%). The Delaware Bay system was the least impacted of the four major systems, less than 1% of its area contained sediments exceeding organic ER-M values. Values for Chesapeake Bay and Long Island Sound were 2-4%, while the Hudson-Raritan system exhibited 44±14% of the area containing sediments with organic concentrations that exceeded at least one ER-M value. This is consistent with the documented PCB contamination in the Hudson River.

Approximately half the entire province (50±4%) had sediment contaminant concentrations below ER-L levels. Small systems and tidal rivers exhibited lower areal extent of sediments below ER-Ls than large systems, 35±6% and 25±12% vs. 58±5%. Only 1±3% of the sediments in the Hudson-Raritan system were observed to have contaminant concentrations below ER-L levels. In other words, almost the entire Hudson-Raritan system has sediments above levels observed to potentially elicit biological responses. In contrast, half or more of Chesapeake Bay (50±5%) and Delaware Bay (62±14%) were observed to have sediment concentration levels below ER-L values. In Long Island Sound, 24±12% of the sediments were below ER-L levels.

Overall, 52±5% of the estuarine waters of the Virginian Province were in good condition, *i.e.,* these waters exhibited unimpacted benthic conditions and bottom dissolved oxygen > 5 ppm and sediment toxicity acute survival > 80% and no ER-M exceedence for sediment contaminants. Small estuarine systems had 55±14% of the area in good condition, tidal rivers had 52±13%, and large systems 51±5%.

Indicator Associations

Analyses of associations were conducted between ecological condition indicator and both stressor and habitat indicators to provide possible explanations for the observed condition of ecological resources. Analysis of benthic communities and habitat indicators show that impacted benthic communities tend to be associated with muddy (> 80% silt-clay), moderate TOC content (1-3%) sediments, and polyhaline bottom waters (> 18 o/oo). In contrast, unimpacted benthic communities tend to associate with sandy (< 20% silt-clay), low TOC content (< 1%) sediments, and with polyhaline bottom waters (> 18 o/oo).

Association analysis of benthic communities with stressors for the entire province indicates that moderate to severe hypoxia (DO ≤ 5 ppm), sediment toxicity (survival ≤ 80%), and sediment contamination (ER-M exceedence) together account for 54% of the impacted benthic area. The remaining 46% of impacted benthos is not associated with any of the three stressor indicators. Low dissolved oxygen is the principal stressor of concern in large systems and tidal rivers, being associated with 56% and 45% of the impacted benthos, respectively. Moderate to severe hypoxia occurs primarily in benthic impacted area of western Long Island Sound, while severe hypoxia (DO ≤ 2 ppm) occurs primarily in the main stem of Chesapeake Bay. The analysis suggests that resources in the large estuarine systems of the Virginian Province, which have the capacity to stratify and receive large inputs of nutrients, are, on the whole, at greater risk from low dissolved oxygen (*e.g.*, eutrophication) than from sediment contamination. In small estuarine systems, sediment toxicity and ER-M exceedence are the principal stressors of concern, accounting for approximately 26% of the observed impacted benthos. This analysis suggests that the risk to benthic resources in small estuarine systems is from sediment contamination and not from low dissolved oxygen conditions. However, there is a large portion unexplained by the three measured stressors.

Conclusions

The four-year assessment of the EMAP-Virginian Province Demonstration Project illustrates several important contributions to environmental monitoring in the areas of sampling design and indicator research. In particular, the sampling design is both systematic and probabilistic in nature; is extremely flexible; provides areal estimates, with confidence, of the condition of all indicators; is spatially explicit allowing for a variety of spatial analyses; accommodates post-stratification of data; can be scaled to the problem setting without losing its defining characteristics; and, most important, has comparability, which permits the direct comparison of results across differing spatial scales and between different categories/populations of resources (*e.g.*, large and small systems and tidal rivers).

The four-year assessment of Virginian Province data affords the opportunity to evaluate the applicability, sufficiency, and effectiveness of EMAP's indicator program. The multi-indicator approach used by EMAP has proven both practical and necessary. Traditional monitoring programs often measure only one type of indicator, either exposure or effect. EMAP, however, by including the patterns of both natural and anthropogenic stressors and habitat factors, provides information critical for forming hypotheses that explain the observed ecological observations. There are limitations, however, to the indicator program in the Virginian Province as illustrated by the uncertainties in the analyses. First, only one ecological indicator, benthic community condition, was successfully developed to assess the status of ecological condition in the Province. Second, there was no development of stressor indicators for enrichment and physical perturbation. This assessment emphasizes the need for additional research on ecological and stressor indicators to reduce uncertainties in the assessments.

The value of the EMAP design and indicator program is illustrated by its ability to identify successfully and quantify the major environmental problems in the estuarine waters of the Virginian Province. When the EMAP conclusions are compared with analyses using other environmental data from the states and federal agencies, the general conclusions are the same. The agreement between conclusions drawn from EMAP and those from existing data could be viewed as an initial validation of the EMAP concept. This is important because it provides evidence that the EMAP design and indicators can capture the major ecological problems successfully when applied to data poor environmental areas. The EMAP design supplements many other studies because it allows the quantification of the degree of uncertainty (confidence limits) of the results, and provides for quantitative comparisons across systems.

Although uncertainties remain, the results of the four-year Virginian Province assessment are encouraging. The Demonstration Project clearly showed that the EMAP objectives were not only reasonable but were achievable with available indicators collected with a probability-based sampling design. It was shown that the EMAP design can be used to quantify with confidence the status and condition of ecological resources. Reducing the uncertainties in the assessment should be approached through a systematic program of directed research.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACKNOWLEDGMENTS

The results of the program presented in this report could not have been conducted without the efforts of literally hundreds of individuals, who assisted in design, development, field sampling, laboratory analyses, total quality management, administrative support, information management, and data analysis. These are the individuals, whose dedication and perseverance in conducting the EMAP-Virginian Province activities, that made our job of producing this assessment report enjoyable. We would like to acknowledge in particular two individuals, Rick Linthurst and Jay Messer, whose leadership made EMAP a reality and permitted us the opportunity to "just do it". Special thanks to Sandi Benyi and Melissa Hughes for assistance with the data analysis. Thanks to Jerry Pesch, Rich Pruell, Charles Strobel, and Steve Weisberg for their critical reviews of this report. This is contribution no. 1837 of the Atlantic Ecology Division, National Health and Environmental Effects Research Laboratory.

EMAP has evolved as a program since the conduct of the Four-Year Virginian Province Demonstration Project. The information presented in this report concerning EMAP was current during the conduct of the Demonstration Project. It should not be implied that this information represents the present status of EMAP.

SECTION 1: INTRODUCTION

The scientific community and the public have become increasingly concerned that the impact of pollutants extends far beyond the local scale. Global climate change, acidic deposition, ozone depletion, non-point source pollution, and habitat alteration threaten our ecological systems at regional, national, and global scales. Years of scientific study have convinced us that ecosystem responses to natural and anthropogenic disturbances are often complex and difficult to characterize, even as these studies have heightened awareness of these environmental problems. The status of the nation's ecological resources has not been well documented in the past, and establishing quantitatively whether environmental policies and programs designed to limit anthropogenic impacts on natural ecosystems are effective has been difficult.

Despite the implementation of stricter environmental control programs in coastal regions, the perception of the scientific community and the informed public is that water and sediment quality and the abundance and quality of living marine resources have declined in the past 10 to 15 years. The perceived decline in estuarine and coastal environmental quality has been noted in the popular press and the scientific literature (Smart *et al.*, 1987; Morganthau, 1988; Toufexis, 1988). These problems are exemplified by the following:

1. Increases in the frequency, duration, and extent of water containing insufficient oxygen to sustain living resources (USEPA, 1984; Officer *et al*., 1984; Parker *et al.,* 1986; Rabalais *et al*., 1985; Whitledge, 1985);

2. Accumulation of contaminants in bottom sediments and in the tissues of fish and shellfish to levels that threaten human health and the sustainability of fish and shellfish populations (OTA, 1987; NRC, 1989);

3. Increased evidence that many restoration and mitigation efforts have not replaced losses of critical habitats (Sanders, 1989; The Conservation Foundation, 1988);

4. Increased incidence of pathological problems in fish and shellfish (Sinderman, 1979; O'Connor *et al*., 1987; Buhler and Williams, 1988; Capuzzo *et al*., 1988);

5. Increased frequency and persistence of algal blooms and associated decreases in water clarity (USEPA, 1984; Pearl, 1988; Smayda and Villareal, 1989);

6. Increased incidence of closures of beaches, shellfishing grounds, and fisheries because of pathogenic and chemical contamination (Smart *et al.*,1987; FDA, 1971, 1985; Hargis and Haven, 1988; Broutman and Leonard, 1988; Leonard *et al.*, 1989); and

7. Increased incidence of human health problems from consuming contaminated fish and shellfish or swimming in contaminated waters (Fein *et al.,* 1984; Malins, 1989).

In 1988, the U.S. Environmental Protection Agency's Science Advisory Board (SAB, 1988) recommended the implementation of a program to monitor ecological status and trends that would identify emerging environmental problems before they reach crisis proportions. The Environmental Monitoring and Assessment Program (EMAP) is the Agency's response to the Science Advisory Board's recommendation. Coincidentally, the National Research Council's Marine Board, in a review of marine and estuarine monitoring systems (NRC, 1990), recommended the creation of a national network of regional monitoring programs for estuarine and coastal environments. This review recognized the need for new monitoring programs that build on existing information and expand the information base landward in order to identify the factors contributing to coastal pollutant problems.

1.1 Overview of EMAP

The Environment Monitoring and Assessment Program is a nationwide program initiated by EPA's Office of Research and Development (ORD). EMAP was developed in response to the demand for information about the degree to which existing pollution control programs and policies protect the nation's ecological resources. EMAP is an integrated federal program; ORD is coordinating, planning, and implementing EMAP in conjunction with other federal agencies, including the Agricultural Research Service, the Bureau of Land Management, the U.S. Fish and Wildlife Service, the U.S. Forest Service, the U.S. Geological Survey, and the National Oceanic and Atmospheric Administration (NOAA), and with other offices within EPA (*e.g.*, Office of Water). These other agencies and offices participate in the collection and analysis of EMAP data and will use it to guide their policy decisions, as appropriate.

The goal of EMAP is to contribute to decisions on environmental protection and management by monitoring and assessing the condition of the nation's ecological resources. To accomplish this goal, EMAP has worked to accomplish four objectives: (1) estimate the current status, trends, and changes in selected indicators of the condition of the nation's ecological resources on a regional basis with known confidence; (2) estimate the geographic coverage and extent of the nation's ecological resources with known confidence; (3) seek associations among selected indicators of natural and anthropogenic stress and indicators of ecological condition; and (4) provide annual statistical summaries and periodic assessments of the nation's ecological resources.

EMAP has been designed to provide the information required to formulate environmental protection policies for the 1990s and beyond by providing answers to the following questions:

1. What are the status, extent, and geographical distribution of the nation's ecological resources?

2. What proportion of these resources is declining or improving? Where? At what rate?

3. What factors are likely to be contributing to declining condition?

4. Are pollution control, reduction, mitigation, and prevention programs achieving overall improvement in ecological condition?

Assessment of status and trends in the condition of the nation's ecological resources requires collecting data in a standardized manner, over large geographic scales, and for long periods of time. Such assessments cannot be accomplished by aggregating data from the many individual, short-term monitoring programs conducted in the past or are being conducted currently (NRC, 1990). Differences in the parameters measured, collection methods, timing of sample collection, and program objectives limit the value of historical monitoring data and existing monitoring programs for making integrated regional and national assessments.

EMAP was proposed and developed by EPA/ORD because an integrated monitoring and assessment program that samples ecological resources in proportion to their abundance (probability-based) offers considerable advantages over historical monitoring approaches. These include:

1. EMAP was designed to improve the definition of the extent and magnitude of pollution problems at regional and national scales.

2. Simultaneous, probability-based sampling of pollution exposure, environmental condition, and biological resources is important for associating environmental stressors with impacted ecological condition.

3. EMAP data are geographically referenced; therefore, the distribution and spatial patterns for impacted ecological conditions can be analyzed.

4. The EMAP sampling design is flexible in that it does not restrict the aggregation or post-stratification that can be applied to the data. The only restriction is on the number of sample points that may be available to conduct the aggregation.

5. The temporal revisits to sampling sites permits information that can be used to capture the direction of ecological change resulting from remediation programs.

In summary, EMAP can provide objective assessments of the severity and extent of environmental problems and the degree to which impacted resources are responding to efforts to protect or restore them.

1.2 Overview of Virginian Province

EMAP identified boundaries for seven estuarine regions (Figure 1-1) based upon biogeographic provinces defined by NOAA and the U.S. Fish and Wildlife Service using major climatic zones and prevailing ocean currents (Bailey, 1983; Terrell, 1979). The four years of data collected over 1990-93 in the estuarine waters of the Virginian Province, which covers approximately 23,573 km^2 and includes the wide expanse of irregular coastline from Cape Cod, Massachusetts, to the mouth of Chesapeake Bay (Cape Henry, Virginia), were to be used to evaluate the feasibility of regional sampling and to evaluate and improve the sampling design and indicators prior to the actual nation-wide implementation of EMAP in estuaries.

The Virginian Province was selected as the testing ground for the EMAP estuarine monitoring effort because there is a public perception that estuaries in this region of the country are deteriorating more rapidly than in other regions. Many estuaries in this province have been investigated intensively by scientists, and a considerable amount of information was available for use in designing the Virginian Province monitoring activities. Six EPA National Estuary Programs were in place in the Virginian Province in 1990. In addition, many management decisions were forthcoming, including development of a restoration plan for the New York Harbor complex, and development of management plans and evaluation of previous management actions for many large estuaries, including Delaware Bay, Chesapeake Bay, and Long Island Sound. Development of such plans presented an opportunity to demonstrate how EMAP monitoring data can assist in the formulation of environmental programs and policies.

Estuaries were selected as one of the first resources to be sampled by EMAP. Estuaries are among the most productive of ecological systems. Historically, more than 70% of commercial and recreational landings of fish and shellfish were taken from estuaries (NOAA, 1987). Estuaries also provide feeding, spawning, and nursery habitats, and are part of migratory routes for many commercially and recreationally important fish and wildlife, including threatened and endangered species (Lippson *et al.*, 1979; Olsen *et al.,* 1980). The public values estuarine ecosystems for recreation (*e.g.*, boating, swimming, hunting, and fishing) and aesthetic appeal. Approximately $7 billion in public funds are spent annually on outdoor marine and estuarine recreation in the 33 coastal states (NOAA, 1988). Millions of tourists visit coastal beaches annually, and coastal property is among the nation's most valuable real estate. About half the nation's population now lives in coastal areas, and by the year 2010, the population in these areas will have grown by almost 60% to 127 million people (Culliton *et al.,* 1990). The estuarine and coastal environment also provides cooling waters for energy

Figure 1-1. Biogeographic provinces used by EMAP to delineate coastal resources of the United States.

production, transportation routes for ships, and space for economic development. Most of the nation's major ports are located in estuaries.

Estuaries are complex transition zones between streams, rivers, and coastal oceans. They have physical features that concentrate and retain pollutants, and they tend to serve as repositories for the many pollutants released into the atmosphere and the nation's surface waters. The ecological condition of estuaries is influenced strongly by human activities in the watershed, particularly land use patterns and the release of pollutants to the environment. In many coastal regions, water and sediment quality and the abundance of living resources are perceived to have declined despite the implementation of pollution control programs.

1.3 Scope of Report

EMAP monitoring activities were conducted in the Virginian Province for the four-year period, 1990-93, using a probability-based sampling design. Data were collected and analyzed in a consistent manner. An assessment of results from the first year's activities was reported in Weisberg *et al.* (1993). The 1991, 1992, and 1993 monitoring activities were reported in Schimmel *et al.* (1994), Strobel *et al.* (1994), and Strobel *et al.* (1995), respectively. Strobel *et al.* (1995) also included statistical summaries for all four years of data. This report presents an assessment of all four years (1990-93) of EMAP monitoring data collected in the estuarine waters of the Virginian Province. In particular, this report provides estimates of ecological resources for both the Province and for the major estuarine systems in the Province, evaluates associations between ecological condition and other indicators, and attempts an evaluation of the effectiveness of the program in meeting its objectives. This report is EMAP's first attempt at providing a multi-year interpretive assessment of regional-scale ecological conditions. It builds upon the efforts of numerous investigators who have been involved with EMAP.

The next section (Section 2) provides background on the EMAP approach for resource class classification, sampling design, and ecological indicators. Section 3 summarizes the assessment for condition indicators and the association between stressors and ecological condition for both the entire province and the major estuarine systems in the province. Section 4 provides a discussion of results of the four-year assessment organized around a series of environmental management questions. Appendix A provides an overview of the sampling and analytical methods for the parameters used in this report. Appendix B presents the refinements to the benthic index that were developed for the four-year assessment.

SECTION 2: EMAP APPROACH

There are three distinct elements of EMAP monitoring that were the guiding forces in designing the sampling plan for the Virginian Province efforts. First, probability-based sampling sites were selected in an unbiased manner so that resources (*e.g.,* estuarine waters in the Virginian Province) were sampled in proportion to their abundance in a resource size class (discussion in next section). Probability-based sampling permits estimation of the condition of the portion of the resource that was not sampled based on knowledge of the sampled portion. Estimates of the proportion of the total area sampled that is impacted can be made with quantifiable confidence. Furthermore, the level of confidence in the estimate can be increased by increasing the number of sites sampled or through the incorporation of existing data.

Second, EMAP focuses on indicators of biological response and uses measures of exposure from stress or pollution for interpreting biological response data. Traditionally, estuarine monitoring has focused on measures of stress (*e.g.*, concentration of contaminants in sediment) and attempted to infer biological impacts based upon laboratory bioassays. The advantage of the ecologically-based approach emphasized in EMAP is that it can be applied to situations where multiple stressors exist, acting separately or in combination, and where natural processes are complex and cannot be easily described. This is certainly the case in estuarine systems, which are subject to an array of anthropogenic inputs and exhibit great biotic diversity and complex physical, chemical, and biological interactions.

Third, EMAP is conducted on regional and national scales using standardized methods. Entire regions are sampled within a defined time window (index period) to characterize the resource and to ensure comparability of data within and among sampling years.

2.1 Resource Classification

For design considerations, EMAP classifies estuaries into three classes (or strata): large estuarine systems, small estuarine systems, and large tidal rivers (Holland, 1990). Large estuarine systems are defined as systems having surface areas greater than 260 km^2 and aspect ratios (length/average width) less than 18. Applying these criteria to the Virginian Province results in the identification of twelve large estuarine systems with a total surface area of 16,097 km^2 or 68% of the province's estuarine area (Table 2-1). Large tidal rivers, defined as systems having surface areas greater than 260 km^2 and aspect ratios greater than 18, include the Hudson, Potomac, James, Delaware, and Rappahannock Rivers. These five tidal rivers have a total surface area of 2,601 km^2 or 11% of the total province area. Small estuarine systems are defined as systems having surface areas less than 260 km^2 but greater than or equal to 2.6 km^2. Applying these criteria to the Virginian Province results in the identification of one hundred forty-four

Table 2-1. Summary of the Total Areas (km^2) for Resource Classes and Major Estuarine Systems in Virginian Province.

	Large Systems	Small Systems	Tidal Rivers	Total
Virginian Province	16,097	4,875	2,601	23,574
Chesapeake Bay (including rivers listed below)	7,044	2,315	2,048	11,407
Potomac River	-	127	1,124	1251
Rappahannock River	-	26	345	371
James River	-	100	578	678
Delaware Bay	1,784	30	245	2,059
Hudson-Raritan	-	451	309	760
Long Island Sound	3,069	276	-	3,345

small estuarine systems with a total surface area of 4,875 km^2, or 21% of the province.

2.2 Sampling Design

EMAP uses a probability-based sampling design over time and space to develop a cost-effective monitoring program (Overton *et al.*, 1991). The statistical approach is similar in concept to other federal statistical programs or surveys, such as those conducted by the Census Bureau, Bureau of Labor Statistics, and National Agriculture Statistics Service. The principal distinction is that these programs focus on producing estimates of the characteristics of human populations, business establishments, or agricultural enterprises. In contrast, EMAP focuses on producing estimates of attributes of ecological resource populations, such as ecological condition of estuarine waters of the Virginian Province.

EMAP was designed to assess regional populations of ecological resources in the United States. The design permits estimates of the condition, geographic coverage (*i.e.,* spatial distribution), and extent for regional populations of ecological resources. The design permits population estimates to be provided with known statistical confidence. EMAP intended to make these estimates not only for a specific point in time (current status) but also repeated over time (trends). The design enables associations (empirical relationships) to be investigated between condition indicators and stressor indicators for the ecological resources.

The EMAP sampling design provides unbiased estimates of the status and trends in indicators of ecological condition with a known level of confidence. The value of the EMAP sampling design is that it is both systematic in areal coverage yet probabilistic relative to the sampling strategy (Overton *et al.*, 1991). This design, therefore, is capable of both determining areal extent (with confidence intervals) and the spatial patterns of ecological indicators irrespective of the characteristics of their statistical distributions. EMAP proposed to base its status assessments on data collected over a four-year baseline (Holland, 1990). This multi-year cycle was chosen to dampen the inter-annual variability resulting from natural phenomena such as extremely dry or wet years and major climatic disturbances such as hurricanes.

EMAP does not attempt to fully characterize naturally-occurring seasonal variability or to assess the status of ecological resources for all seasons. An index period (July-September) was chosen for estuarine sampling to represent that portion of the year when the measured ecological parameters are expected to show the maximum response to pollutant stress (Connell and Miller, 1984; Sprague, 1985; Mayer *et al*., 1989); dissolved oxygen concentrations are lowest (Holland *et al*., 1988; USEPA, 1984; Officer *et al*., 1984); fauna and flora are most abundant; and within-season variability is expected to be minimized. A consequence of the index approach is that short-term, episodic events may not be detected. This approach is consistent with EMAP's goals of determining the long-term status and trends of ecological resources that can then be used as the basis for intensive site-specific research to understand the reasons for the observed deviations form baseline conditions.

Sampling sites in the large estuarine class were selected using a randomly-placed systematic grid (Holland, 1990; Paul *et al.,* 1992). The distance between the systematically-spaced sampling points on the grid was approximately 18 km. The grid is an extension of the systematic grid proposed for use by EMAP (Overton *et al.,* 1991). The center points of the grids are the sample sites. A linear analog of the systematic grid was used for site selection in the large tidal rivers (Holland, 1990; Paul *et al.,* 1992). A systematic linear grid was used to define the spine of the five large tidal rivers, where the starting point of the spine was at the mouth of the river. The first transect was randomly located between river kilometer 0 and 25. Additional transects were then placed every 25 km up the river to the head of tide. The 144 small estuarine systems were randomly sampled from the entire list of small systems in the province (Holland, 1990; Paul *et al.,* 1992). They were ordered from north to south by combining adjacent estuaries into groups of four. One estuary was selected randomly from each group without replacement for each of the four years of sampling. The location of the sample within each selected small system was randomly placed.

Application of the sampling design to the three estuarine resource classes resulted in 425 sampling sites over the four-year period. The distribution of these sites is shown in Figure 2-1. Additional sampling sites were visited over the four-year period to meet

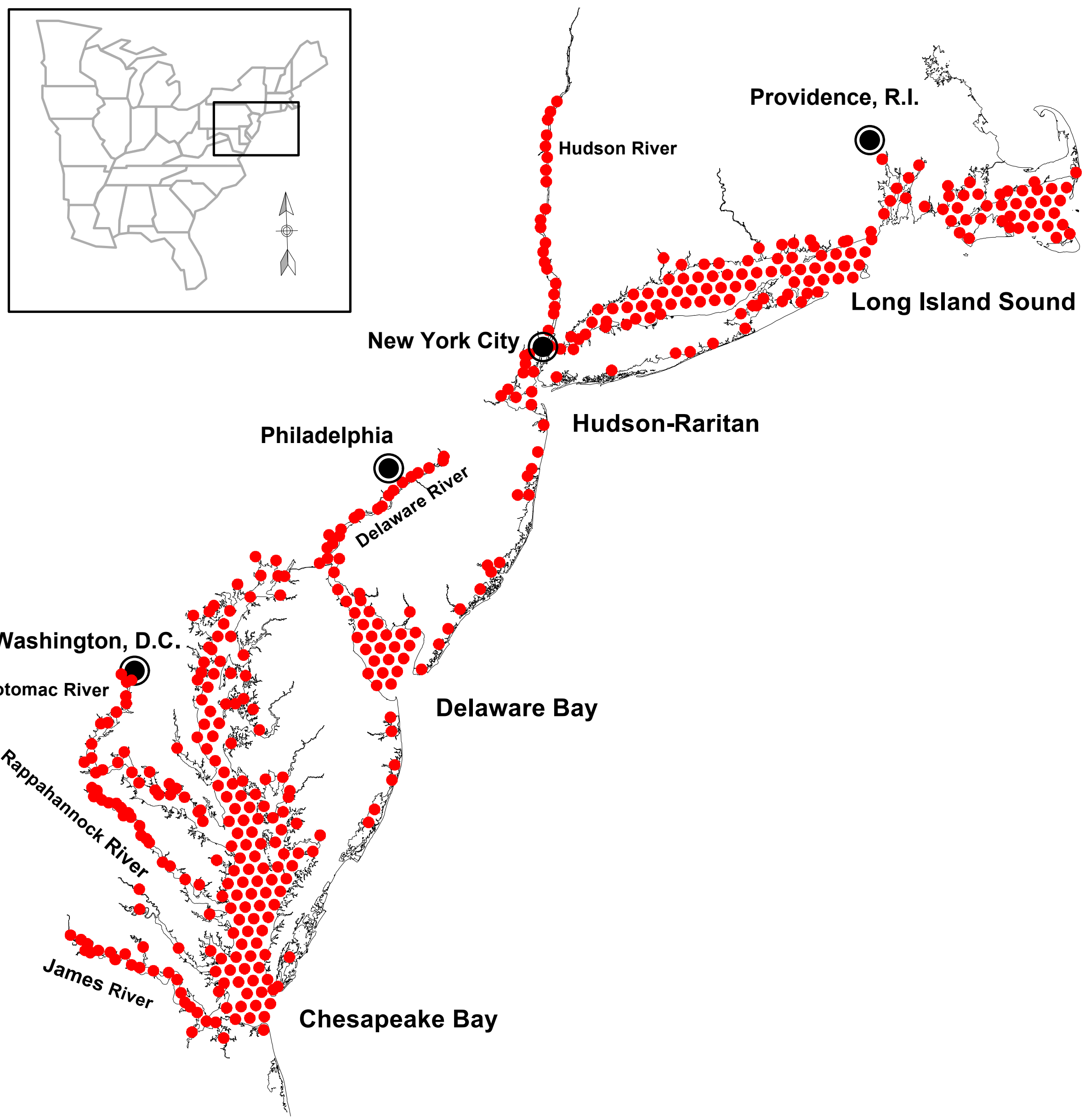

Figure 2.1. Distribution of probability-based sampling sites in the Virginian Biogeographic Province during the period 1990-1993.

specific research needs, including indicator testing and evaluation, and spatial and temporal variability estimates. Refer to Holland (1990), Weisberg *et al.* (1993), Schimmel *et al.* (1994), and Strobel *et al.* (1994, 1995) for details on these additional sampling sites.

2.3 Ecological Indicators

EMAP monitors ecological indicators to assess status, changes, and trends in the condition of the nation's ecological resources (Hunsaker and Carpenter, 1990). Indicators are defined as any characteristic of the environment that can provide quantitative information on the condition of ecological resources, the magnitude of stress, the exposure of a biological component to stress, or the amount of change in the condition of the resource.

Ecological condition and response to perturbation are determined by the interaction of all the physical, chemical, and biological components of the system. Because it is impossible to measure all these components, EMAP's strategy has been to emphasize indicators of ecological structure and function that represent the condition of ecological resources, relative to societal values. EMAP has selected, developed, and evaluated indicators for the following reasons: (1) to describe the overall condition of ecological resources, (2) to permit the detection of changes and trends in condition, and (3) to provide preliminary diagnosis of possible factors that might contribute to unacceptable conditions caused by human or natural stressors.

EMAP defines two general types of ecological indicators, condition and stressor indicators. A condition indicator is any characteristic of the environment that provides quantitative estimates on the state of an ecological resource and is based on something valued by society. There are two types of condition indicators: biotic and abiotic. Condition indicators are used to estimate the status, trends, and changes in ecological condition as well as the extent of ecological resources. EMAP estimates the regional distribution of quantitative values for each of these indicators within and among resource categories. All estimates are accompanied by specified levels of confidence (95% confidence limits for this report) so the user knows the certainty of the estimates. Condition indicators discussed in this report include the benthic index (a combination of structural properties of the benthic animal assemblages), dissolved oxygen concentrations in bottom water, acute toxicity of sediments, and sediment contaminant concentrations.

Stressor indicators are characteristics of the environment that cause changes in the condition of an ecological resource. Both natural and human-induced stressors are examined. Monitoring selected stressor and condition indicators allows EMAP to seek associations between indicators of stress and observed ecological conditions. Associations provide insight or possible causality, and lead to the formulation of hypotheses regarding factors (*e.g*., land use activities, sources, *etc*.) that might be contributing to the unacceptable ecological conditions. Associations can provide direction for regulatory, management, or research programs by suggesting the possibility of causal relationships. In this report the benthic index is associated with the following indicators of stress: dissolved oxygen, sediment toxicity, sediment contaminant concentrations, salinity, sediment grain size, and sediment organic carbon content. Note that a parameter can function as both a condition indicator and an indicator of stress, depending on its context, *e.g*., low dissolved oxygen can be both an indicator of stress and an indicator of condition.

SECTION 3: INDICATOR STATUS AND ASSESSMENT

Areal and spatial analyses of EMAP data were conducted for the province and each of the three estuarine resource classes at the regional and watershed scales. EMAP's probability-based sampling design assigns an areal weight to each sampling station. This design permits the calculation of areal-based cumulative distribution functions (CDFs) for each of the indicators. The estimation procedures in Heimbuch *et al.* (1995a) were used to calculate CDFs and 95% confidence intervals (C.I.) that are the foundation for the analyses in this report. Each condition indicator is assigned a "critical value" or threshold that separates impacted from unimpacted conditions. For example, dissolved oxygen is assigned a critical value of 2.0 ppm based upon numerous examples in the scientific literature that clearly suggest that dissolved oxygen values less than 2.0 ppm cause both acute and chronic ecological effects. Estimates of the relative (percent of total resource class or estuarine system area) and absolute (km^2) areal extent of "impacted condition" were calculated for each condition indicator and resource category at both province-wide and major estuarine system scales (refer to next paragraph for specific major estuarine systems). It is important to note that analyses of both percent area and absolute area may be useful for developing different management options. An impact is not necessarily due to anthropogenic stress alone. Natural stress can also contribute to ecological condition. EMAP indicators, as currently applied, cannot unambiguously discriminate between natural and anthropogenic stress.

While estimates of resource condition on an areal basis are one of the strengths of the EMAP probabilistic design, EMAP data are also spatially explicit and, therefore, amenable to a variety of landscape analyses. For example, stations below a critical value can be spatially displayed for the province or specific estuarine systems. Spatial representation of impacted indicator values is particularly useful because they can be linked to land-use patterns that are potentially amenable to control or remediation strategies. Although EMAP's primary objective is to describe status and trends at the province level, estimates can also be generated for subpopulations or different groupings of the estuarine resource. In addition to the three resource classes defined for the sampling design, we have also aggregated and analyzed the sampling data from stations in four major estuarine systems in the Virginian Province (Chesapeake Bay, Delaware Bay, the Hudson-Raritan system [New York/New Jersey harbor area, Hudson River, and Raritan Bay], and Long Island Sound), along with three major tidal river systems within Chesapeake Bay (the James, Rappahannock, and Potomac Rivers). This type of classification integrates multiple land-based sources of anthropogenic pollutants (point and non-point) into riverine systems that discharge to the estuaries. These analyses illustrate the flexibility and power of the EMAP design and can be used to provide an assessment of relative ranking of the potential risks for each estuarine system. However, there are practical limitations imposed by application of the design due to small sample sizes for certain systems, as will be discussed in Section 4.5.

These approaches, while illustrating the flexibility of the EMAP design, provide a suite of analysis options that are scale-independent which can be applied to any problem setting. We have made no presumption which of these approaches is most useful from an assessment perspective. However, spatial analysis combined with areal estimates of condition produce information that is comprehensive and consistent with risk-based principles of assessment and management. In practice, the analysis approach selected will be dependent upon the specific management question being addressed.

Results from the analysis of each major indicator collected over the four-year period of the study are presented and discussed in the remainder of this section. The presentation for each indicator includes: (1) background on the importance of the indicator; (2) percent and absolute estuarine area impacted for the Virginian Province and the three resource classes; (3) spatial representation of the impacted estuarine areas in the province; (4) percent and absolute area and spatial presentation illustrating impacted estuarine sampling areas for the four major estuarine systems and the three tidal river systems; and (5) a summary of the status of the indicator.

3.1 Benthic Community Condition Indicator

The status of biological resources in the Virginian Province was characterized by evaluating the condition of bottom dwelling (benthic) invertebrate assemblages. The importance of the role of benthic communities in estuarine ecosystems is well established (Holland *et al*., 1987, 1988; Rhoads *et al.*, 1978; Pearson and Rosenberg, 1978; Sanders *et al*., 1980; Boesch and Rosenberg, 1981), and for the purposes of this report, the condition of benthic assemblages will be the only biological condition indicator presented. Benthic assemblages were used as an indicator because previous studies suggested that they are sensitive to pollutant exposure (Pearson and Rosenberg, 1978; Boesch and Rosenberg, 1981). They also integrate responses to exposure and disturbance over relatively long periods of time (months to years). Their sensitivity to pollutant stress is, in part, because benthic organisms live in the sediment, a medium that accumulates environmental contaminants over time (Nixon *et al.*, 1986; Schubel and Carter, 1984). Their relative immobility also restricts benthic organisms from avoiding pollutant exposure and environmental disturbances.

Three samples were collected at each site using a stainless steel, Young-modified van Veen grab that samples a surface area of 440 cm^2. They were sieved through a 0.5 mm screen and preserved for laboratory analysis. Organisms were identified and counted. Biomass of key taxa was measured. Other taxa were grouped according to taxonomic type (*e.g.*, polychaetes, amphipods, decapods) for biomass determination.

Characteristics of benthic assemblages, often expressed as indices, have been used to measure and describe ecological status and trends of marine and estuarine environments for several decades (Sanders, 1956, 1960; Boesch 1973; Pearson and Rosenberg, 1978; Holland *et al*., 1988). This literature has identified a diverse array of benthic assemblage attributes that can be used to characterize ecological status and trends, including: 1) measures of biodiversity/species richness, 2) changes in species composition, 3) changes in the relative abundance or productivity of functional groups, 4) changes in relative abundance and/or productivity of "key" species, 5) changes in biomass, and 6) relative size of biota (Weisberg *et al.*, 1993).

An index, based upon several structural properties of benthic assemblages, was used to summarize the benthic data and characterize estuarine biological condition in this report. Discriminant analysis was used to identify a combination of characteristics of benthic assemblages that distinguishes between reference sites (*i.e.,* clean sites) and sites with known pollutant exposure. The sites used to develop this index were distributed across the entire Province and encompass geographic diversity as well as a variety of habitats. The algorithm for the EMAP Virginian Province 1990-93 benthic index used for this report is:

*1.389 * (salinity normalized Gleason's D based upon infauna and epifauna - 51.5) / 28.4*
*- 0.651 * (salinity normalized tubificid abundance - 28.2) / 119.5*
*- 0.375 ** (spionid abundance - 20.0) / 45.4,

where

salinity normalized Gleason's D based upon infauna and epifauna =
*Gleason's D / (4.283 - 0.498 * bottom salinity*
*+ 0.0542 * bottom salinity^2*
*- 0.00103 * bottom salinity^3) * 100*,

Gleason's D = S/ln(N),

N = total number of individuals,

S = number of species,

salinity normalized tubificid abundance =
*tubificid abundance - 500 * exp(- 15 * bottom salinity),*

ln(...) denotes natural logarithm,

and

exp(...) denotes the exponential function.

This version of the benthic index is a refinement to that presented in Schimmel *et al.* (1994) and was developed from data collected in all four years. The index represents an attempt at reducing a complex set of biological measurements to a simple, interpretable value. Details of the rationale and development of the benthic index and the procedure used to distinguish impacted from unimpacted (reference) benthic condition are presented in Appendix B.

3.1.1 Benthic Condition

The condition of the benthic communities in the Virginian Province, as determined by the benthic index, is summarized in Table 3-1 and illustrated in Figure 3-1. Impacted benthic communities comprised 25±3% of the province area for the four-year period, with annual values ranging from a low of 23±7% in 1991 to a high of 28±7% in 1992. This means that, for the four-year period, 75±3% of the estuarine area of the Virginian Province had unimpacted benthic communities. Of the three resource classes, the large estuaries had the smallest percent area with impacted benthic communities, 19±4% for the four-year period, with annual values ranging from 13±7% for 1991 to 27±18% in 1992. The percent of area with impacted benthic resources was markedly larger in the small estuarine systems and tidal rivers, 37±6% and 38±14%, respectively, for the four-year period. This is likely a reflection of the proximity of the small estuarine systems and tidal rivers to urban areas which are the major sources of anthropogenic stress.

Table 3-1. Areal Estimates for Impacted Benthic Communities, as Determined by EMAP-Virginian Province Benthic Index, for the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	26	7	23	7	28	7	25	10	25	3
Large Systems	15	23	13	7	27	18	22	9	19	4
Small Systems	44	22	50	18	35	14	28	33	37	6
Tidal Rivers	58	27	35	34	25	12	35	4	38	14

(b)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	6,115	1,650	5,382	1,650	6,650	1,650	5,908	2,357	5,894	738
Large Systems	2,450	3,638	2,012	1,095	4,319	2,854	3,614	1,380	3,099	645
Small Systems	2,153	1,061	2,455	880	1,683	698	1,375	1,614	1,796	307
Tidal Rivers	1,514	705	914	896	648	308	918	101	998	361

Analysis of impacted benthic condition based on absolute area provides a different perspective (Table 3-1). Absolute areal extent (km^2) was calculated by multiplying the area of each resource class (see Table 2-1) by the percent area impacted for each indicator. On an area basis, the large systems contain the greatest area (3,099 km^2) of impacted benthic communities.

Two additional points are worth noting: 1) the variability of annual values was similar among the three resource classes although the temporal patterns were different, and 2) the percent area of impacted benthos was similar in the tidal river and small estuarine classes despite an almost two-fold difference in values for the absolute area (km^2) impacted. These differences result from the difference in actual areas that each resource class covers.

The spatial patterns of benthic impact, illustrated in Figure 3-2, provide additional insight into the status and condition of benthic resources in the Virginian Province. It is clear that impacted benthic communities are associated primarily with the major tidal rivers and small bays and estuarine systems. These data confirm observations from extant data that our riverine systems are often the focus for anthropogenic stress. It should be noted that this spatial display (Figure 3-2) identifies the EMAP sampling sites having impacted or unimpacted benthic communities, as measured by the benthic index, when the sample was collected. Lack of data for a particular location indicates either that no sample was taken or that an acceptable sample was not obtained.

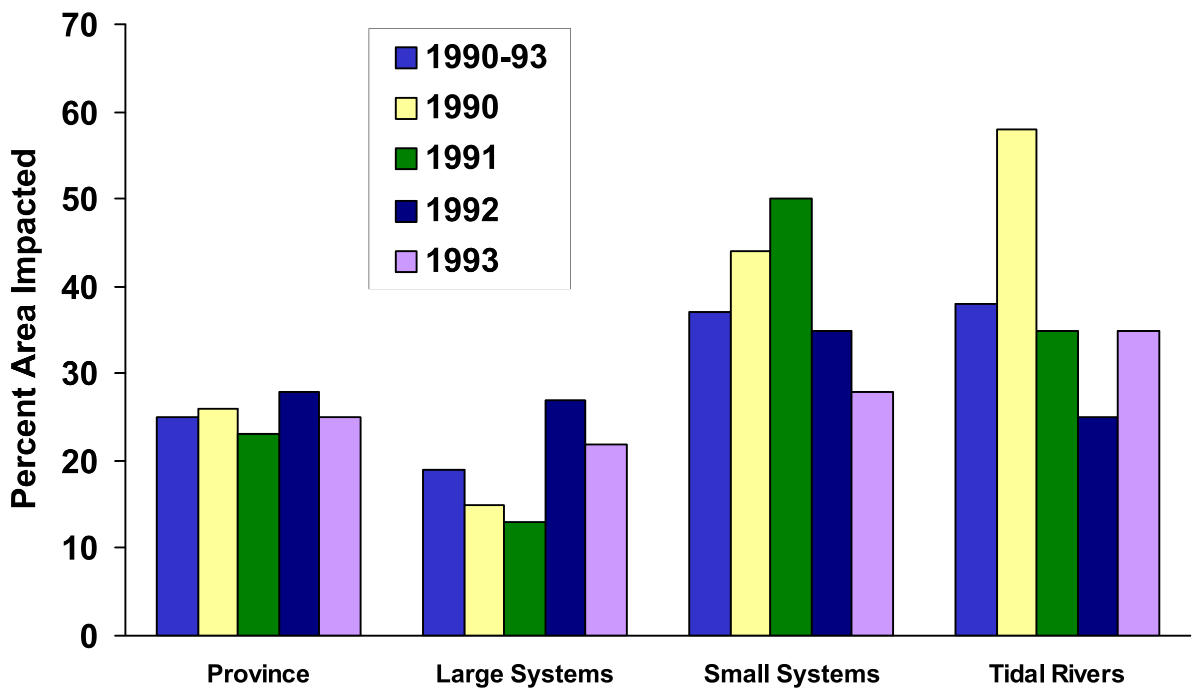

Figure 3-1. Benthic impact for Virginian Province and resource classes across the years 1990-93.

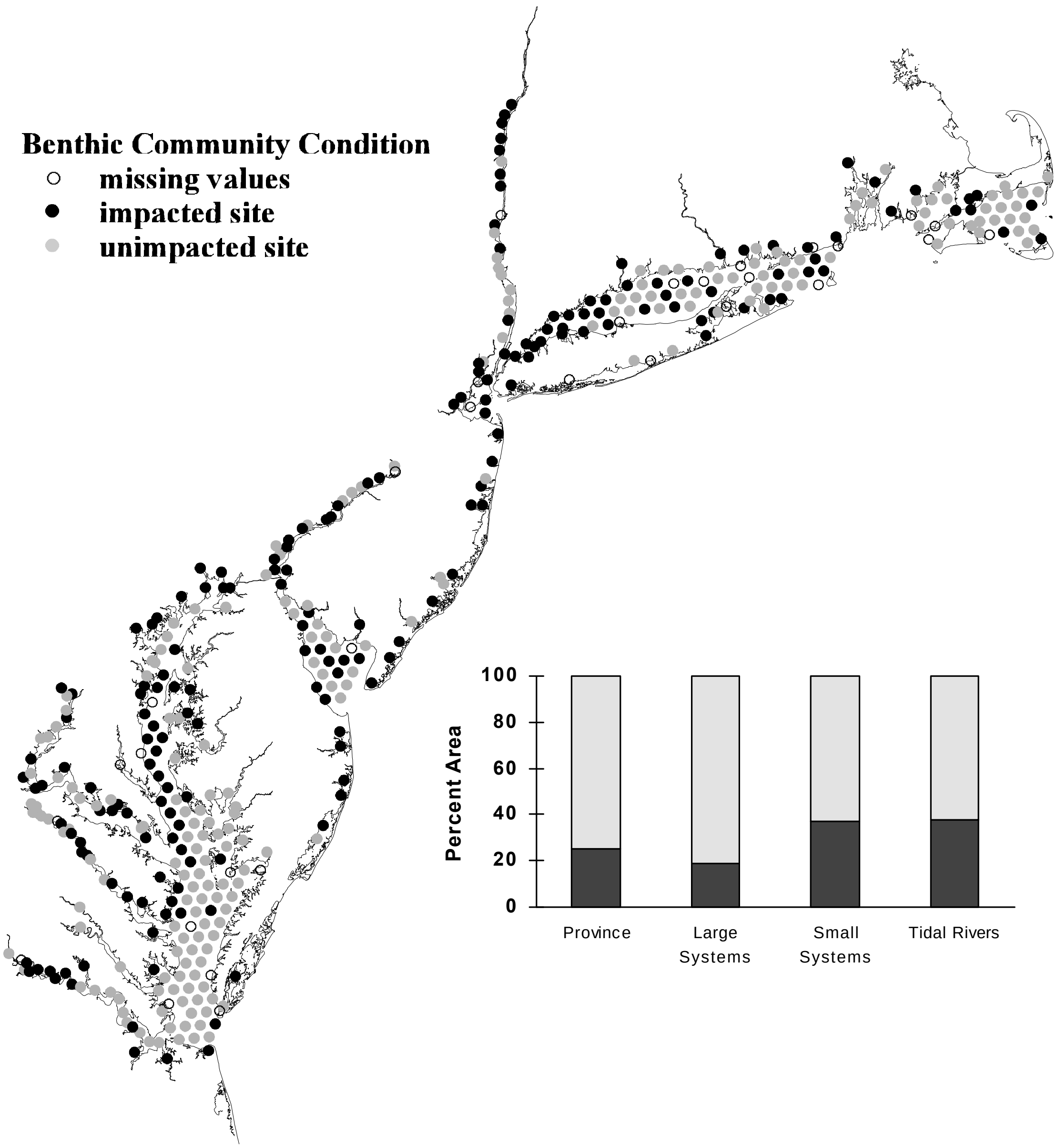

Figure 3-2. Condition of benthic communities for Virginian Province for the period 1990-93.

Although the benthic index used in this report appears to work well for distinguishing sites of differing benthic condition, it is not the only effective index for assessing the condition of estuarine benthic resources (for an example, see Ranasinghe *et al.,* 1993 or Weisberg *et al.*, 1997). An important point about the benthic index development is that it is based upon empirical data and is designed to distinguish between impacted and unimpacted conditions; conditions which were determined by independent criteria.

3.1.2 Benthic Condition: Major Estuarine Systems

An alternative approach for analyzing EMAP monitoring data is to determine the potential status and risks to benthic resources for specific geographic areas rather than for the three estuarine resource classes at the province level. The geographic areas used in the following analyses were classified based on their hydrology into major watersheds or estuarine drainage areas. Analysis of EMAP data from Chesapeake Bay, Delaware Bay, the Hudson-Raritan system, and Long Island Sound illustrate the flexibility of the EMAP sampling design (Table 3-2 and Figures 3-3 and 3-4).

3.1.2.1 Chesapeake Bay

Chesapeake Bay comprises 11,406 km^2 or approximately 48% of the Virginian Province. Analyses were conducted to determine the condition of the benthic resource for Chesapeake Bay *in toto,* large and small estuarine systems, and tidal rivers within the boundaries of the watershed. These analyses indicate that 23±5% of the benthic

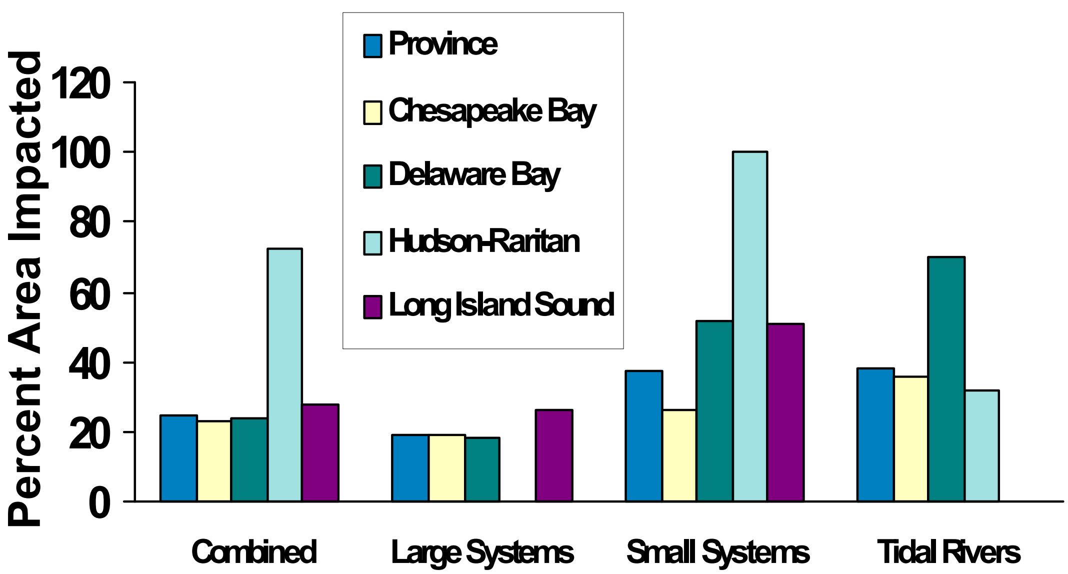

Figure 3-3. Benthic impact for major estuarine systems of Virginian Province for the period 1990-93.

Table 3-2. Areal Estimates for Impacted Benthic Communities, as Determined by EMAP Virginian Province Benthic Index, for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	33	10	20	11	22	9	27	11	23	5
Potomac River	*	*	*	*	*	*	*	*	44	22
Rappahannock River	*	*	*	*	*	*	*	*	44	33
James River	*	*	*	*	*	*	*	*	19	23
Delaware Bay	12	4	13	23	26	24	44	32	24	12
Hudson-Raritan	78	13	59	0†	83	19	70	20	72	8
Long Island Sound	30	22	39	24	39	24	12	17	28	11

(b)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	3,764	1,141	2,276	1,255	2,481	979	3,063	1,292	2,672	542
Potomac River	*	*	*	*	*	*	*	*	551	275
Rappahannock River	*	*	*	*	*	*	*	*	163	121
James River	*	*	*	*	*	*	*	*	130	159
Delaware Bay	246	91	275	476	543	493	915	664	502	240
Hudson-Raritan	589	101	451	0†	633	145	529	154	551	61
Long Island Sound	988	723	1,299	786	1,299	786	397	569	941	359

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

† Due to assumptions in the estimation procedures, if one resource class entirely exceeds the criterion and other resource classes have no exceedences, the C.I. becomes zero.

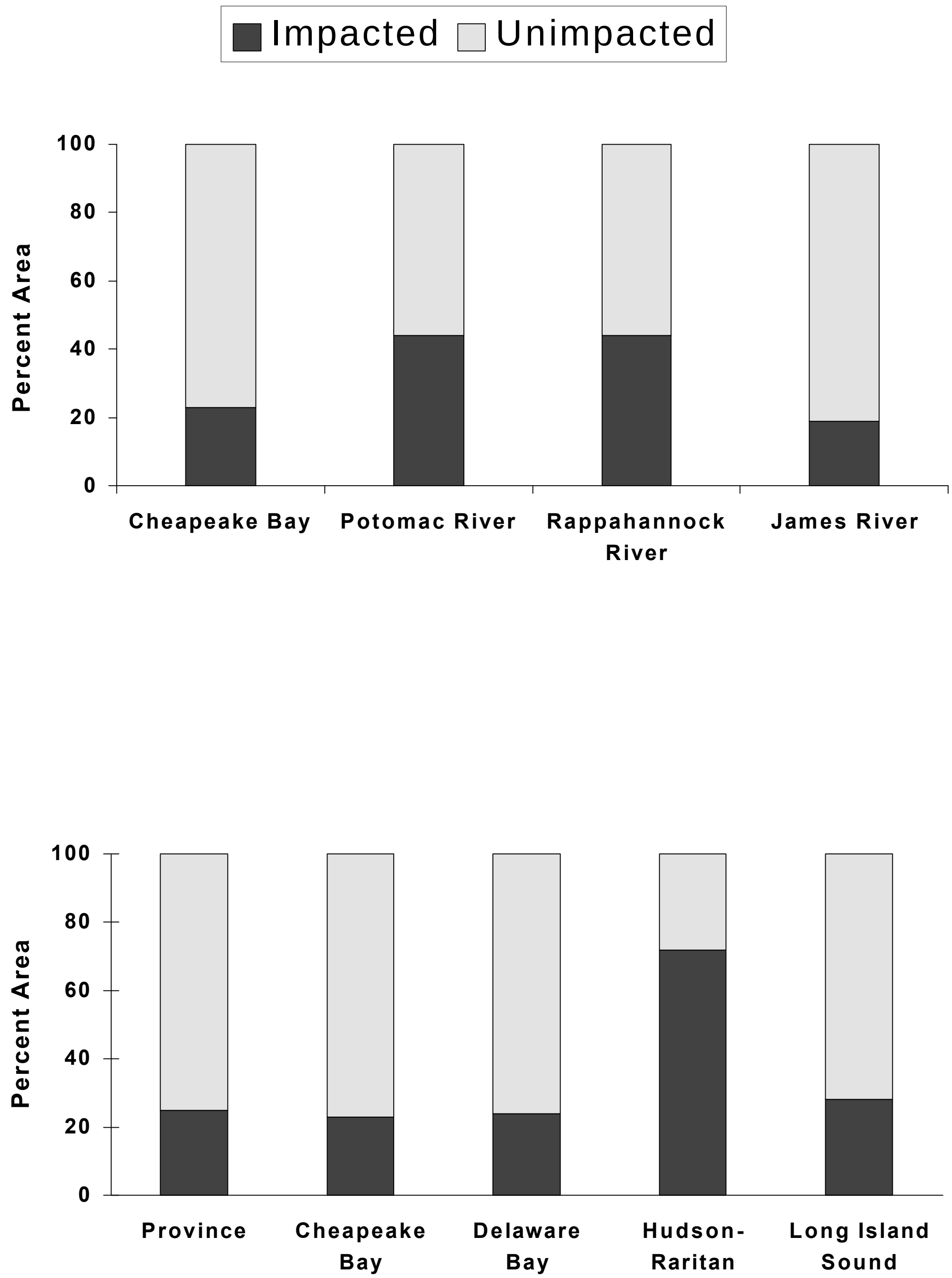

Figure 3-4. Condition of benthic communities for major estuarine systems in Virginian Province for the period 1990-93.

resources of Chesapeake Bay are impacted (Table 3-2). Distribution of impacted benthos among the resource class is as follows: 26±8% and 36±17% of the area of the small estuarine systems and tidal rivers, respectively, and 19±6% for the large estuarine class (Figure 3-4).

For the Potomac, Rappahannock, and James Rivers, 44±22%, 44±33%, and 19±23%, respectively, of the areas had impacted benthos. It should be noted that the 95% confidence intervals for these individual tidal river systems are large due to the smaller number of sample points that were available to produce the estimates.

3.1.2.2 Delaware Bay

Delaware Bay comprises an area of 2,059 km^2 or approximately 9% of the Virginian Province. Benthic resources were impacted in 24±12% of this area (Table 3-2). Examination of individual resource classes showed that 18±17% of the large estuarine class had impacted benthic resources, and the small estuarine systems and the tidal river with 52±22% and 70±2%, respectively. However, because the area of the small systems and the tidal river, 275 km^2, was only 12% of the estuarine area, the overall influence on the analysis for the Delaware system is muted.

3.1.2.3 Hudson-Raritan

The Hudson-Raritan system comprises only 760 km^2 or approximately 3% of the Virginian Province. Impacted benthic resources made up 72±8% of the total area (Table 3-2). Individual resource class analyses show that 32±19% of the tidal river and almost all of the small estuarine systems have impacted benthos. The Hudson-Raritan system exhibits the most areally impacted benthic resources of the four major estuarine systems.

3.1.2.4 Long Island Sound

Long Island Sound comprises 3,344 km^2 or approximately 14% of the Virginian Province. Benthic resources were impacted in only 28±11% of the total area (Table 3-2). Individual resource class analyses show that 26±12% of the large estuarine class and 51±12% of the small estuarine systems have impacted benthos. The absolute areal extent of benthic degradation is much greater in large estuarine class despite a factor of two larger percent area for small systems.

3.2 Dissolved Oxygen Condition Indicator

Dissolved oxygen (DO) is a fundamental requirement for the maintenance of balanced indigenous populations of fish, shellfish, and other aquatic biota. Most estuarine populations can tolerate short exposures to low dissolved oxygen concentrations. However, prolonged exposures to less that 60% oxygen saturation may result in altered behavior, reduced growth, adverse reproductive effects, and mortality (Vernberg, 1972; Reish and Barnard, 1960). Exposure to less than 30% saturation (~ 2 ppm, for seawater at summer temperatures) for 1 to 4 days causes mortality to most biota, especially during summer months, when metabolic rates are high. Stresses that can occur in conjunction with low dissolved oxygen (*e.g.*, exposure to hydrogen sulfide or ammonia) may cause as much, if not more, harm to aquatic biota than exposure to low dissolved oxygen concentration alone (Brongersma-Sanders, 1957; Theede, 1973). In addition, aquatic populations exposed to low dissolved oxygen concentration may be more susceptible to adverse effects of other stressors (*e.g.*, disease, toxic substances).

Water column profiles for water quality parameters were collected at each station using a SeaBird SBE-25 Sea Logger CTD. The unit was equipped with probes to measure salinity, temperature, depth, pH, dissolved oxygen (DO), light transmission, fluorescence, and photosynthetically active radiation (PAR).

For assessments with the 1990-93 Virginian Province data, benchmarks of 2 and 5 ppm are used for DO measured in the bottom waters. A concentration of approximately 2 ppm often is used as a threshold for oxygen concentrations thought to be extremely stressful to most estuarine biota. A threshold concentration of 5 ppm is used by many states to set water quality standards. The USEPA, as of this writing, has not established DO water quality criteria for estuarine and marine waters, but has developed draft criteria for review and comment.

3.2.1 Dissolved Oxygen Condition

The condition of dissolved oxygen in bottom waters of the Virginian Province is summarized in Table 3-3 and illustrated in Figure 3-5. The estuarine area with low dissolved oxygen (DO $\leq$ 2.0 ppm) in bottom waters was 1$\pm$1% in small estuarine systems, 5$\pm$2% in large estuarine systems; and 14$\pm$6% in tidal rivers. The four-year value for the Virginian Province as a whole was 5$\pm$2%. Annual values ranged from 2 to 8% in large systems, 4 to 7% in the province, < 1% in small estuarine systems, but was highly variable (zero to 35%) in the tidal rivers. Percent area analyses of low dissolved oxygen suggest that tidal river systems are potentially at risk while absolute areal analyses indicate that large systems are at greatest risk from low dissolved oxygen. This is due to the large areas associated with the large systems and because these systems are most likely to stratify in the summer.

The area of bottom waters in the Virginian Province impacted by moderate to severe hypoxia ($\leq$ 5.0 ppm) was 17$\pm$5% in small estuarine systems, 27$\pm$4% in large estuarine systems, and 18$\pm$7% in tidal rivers. The four-year estimate of percent area impacted by moderate to severe DO for the Virginian Province was 24$\pm$3%. Annual

Table 3-3. Areal Estimates for Dissolved Oxygen Condition of Bottom Waters in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)

dissolved oxygen < 2 ppm

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	7	3	4	3	6	4	5	4	5	2
Large Systems	4	6	2	4	8	11	8	8	5	2
Small Systems	0	0‡	1	2	0	0‡	1	3	1	1
Tidal Rivers	35	16	21	20	0	0‡	0	0‡	14	6

dissolved oxygen ≤ 5 ppm

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	20	6	17	6	30	7	27	9	24	3
Large Systems	22	8	15	7	38	10	32	10	27	4
Small Systems	7	7	21	12	12	10	23	29	17	5
Tidal Rivers	37	17	21	20	13	11	3	5	18	7

(b)

dissolved oxygen ≤ 2 ppm

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	1,544	771	964	705	1,306	903	1,268	875	1,275	408
Large Systems	631	947	349	700	1,305	1,740	1,215	1,214	876	372
Small Systems	0	0‡	69	108	0	0‡	53	158	35	65
Tidal Rivers	912	429	546	512	0	0‡	0	0‡	365	167

dissolved oxygen ≤ 5 ppm

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	4,745	1,431	4,024	1,414	7,011	1,631	6,353	2,122	5,594	740
Large Systems	3,472	1,360	2,450	1,188	6,091	1,603	5,164	1,544	4,293	718
Small Systems	321	323	1,029	575	586	490	1,125	1,392	825	263
Tidal Rivers	952	444	546	512	334	298	66	132	475	188

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

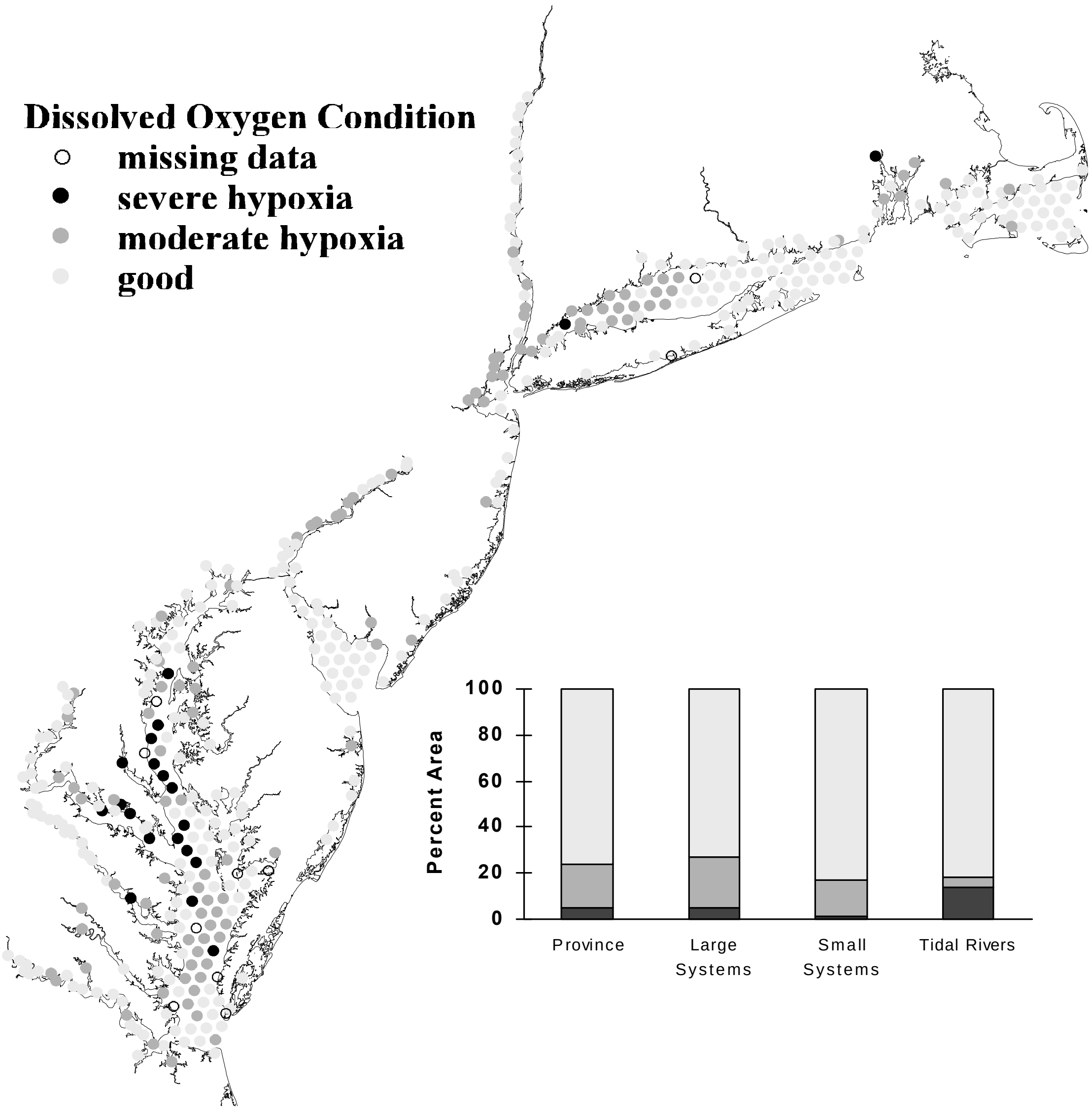

Figure 3-5. Dissolved oxygen condition of bottom waters in Virginian Province for the period 1990-93.

values were in the range of 17 to 30% in the province, 15 to 38% in large systems, 7 to 23% in small estuarine systems, and 3 to 37% in the tidal rivers. Analyses of moderate to severe dissolved oxygen suggest that large estuarine systems are potentially at risk.

The spatial patterns of low dissolved oxygen ($\leq$ 2.0 ppm) and moderate hypoxia (2 ppm < DO $\leq$ 5 ppm) are illustrated in Figure 3-5. Most of the low dissolved oxygen is found in the main stem of Chesapeake Bay and the mouths of the Potomac and Rappahannock Rivers. Only two records of low dissolved oxygen were found outside Chesapeake Bay. Moderate hypoxia, however, was more equally distributed throughout the province including western Long Island Sound, the Hudson-Raritan system, the upper Delaware River, and the central to lower portions of Chesapeake Bay. Approximately one-half (48$\pm$12%) of Long Island Sound exhibits moderate to severe hypoxia. The results of these estuarine system analyses are summarized in Tables 3-4 and 3-5 and illustrated in Figure 3-6.

The results for dissolved oxygen conditions were based upon single point-in-time values recorded during the visit to each station. The usefulness of the point-in-time results depends upon the stability of the areal extent of low dissolved oxygen through the index period. This was tested with province-wide data acquired during 1990 for two successive sampling intervals (19 July to 31 August and 1 September to 23 September) (Weisberg *et al*., 1993). Comparisons of the cumulative distribution functions for bottom dissolved oxygen concentrations for the two intervals demonstrated that the distribution of dissolved oxygen was stable between the two sampling periods. Therefore, the point-in-time measurements of dissolved oxygen were adequate for estimates of status during the summer index period.

3.3 Sediment Toxicity Condition Indicator

Sediment toxicity tests are the most direct measure available for determining the toxicity of contaminants in sediments. These tests provide information that is independent of chemical characterizations and ecological surveys (Chapman, 1988). They improve upon the direct measure of contaminants in sediments because many contaminants are tightly bound to sediment particles or are chemically complexed and are not biologically available (USEPA, 1989). However, sediment toxicity cannot be used instead of the direct measurement of sediment contaminant concentrations, since the latter are an important part of interpreting observed mortality in toxicity tests (USEPA, 1994a).

Sediment toxicity testing has had many applications in both marine and freshwater environments (Swartz, 1987; Chapman, 1988) and has become an integral part of many benthic assessment programs (Swartz, 1989). A particularly important application of sediment toxicity testing is in programs seeking to establish contaminant-specific effects.

Table 3-4. Areal Estimates for Dissolved Oxygen Condition (DO $\leq$ 2 ppm) of Bottom Waters for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	13	7	5	4	12	8	10	7	10	3
Potomac River	*	*	*	*	*	*	*	*	24	12
Rappahannock River	*	*	*	*	*	*	*	*	15	11
James River	*	*	*	*	*	*	*	*	0	0‡
Delaware Bay	0	0‡	0	0‡	0	0‡	0	0‡	0	0‡
Hudson-Raritan	0	0‡	0	0‡	0	0‡	0	0‡	0	0‡
Long Island Sound	0	0‡	10	16	0	0‡	0	0‡	3	4

(b)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	1,533	760	540	439	1,313	859	1,168	819	1,136	368
Potomac River	*	*	*	*	*	*	*	*	306	145
Rappahannock River	*	*	*	*	*	*	*	*	57	41
James River	*	*	*	*	*	*	*	*	0	0‡
Delaware Bay	0	0‡	0	0‡	0	0‡	0	0‡	0	0‡
Hudson-Raritan	0	0‡	0	0‡	0	0‡	0	0‡	0	0‡
Long Island Sound	0	0‡	341	524	0	0‡	0	0‡	85	131

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

Table 3-5. Areal Estimates for Dissolved Oxygen Condition (DO $\leq$ 5 ppm) of Bottom Waters for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	28	9	19	9	37	10	38	16	31	5
Potomac River	*	*	*	*	*	*	*	*	25	12
Rappahannock River	*	*	*	*	*	*	*	*	39	20
James River	*	*	*	*	*	*	*	*	4	<1
Delaware Bay	3	6	0	0‡	2	7	4	8	3	4
Hudson-Raritan	6	9	7	8	47	50	12	37	17	17
Long Island Sound	51	25	44	25	51	25	42	25	48	12

(b)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	3,193	1,067	2,115	1,027	4,231	1,103	4,353	1,825	3,550	524
Potomac River	*	*	*	*	*	*	*	*	309	145
Rappahannock River	*	*	*	*	*	*	*	*	145	76
James River	*	*	*	*	*	*	*	*	26	<7
Delaware Bay	66	129	0	0‡	45	153	80	160	68	85
Hudson-Raritan	44	68	49	61	360	380	93	281	130	129
Long Island Sound	1,705	829	1,477	829	1,705	829	1,410	829	1,592	414

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero

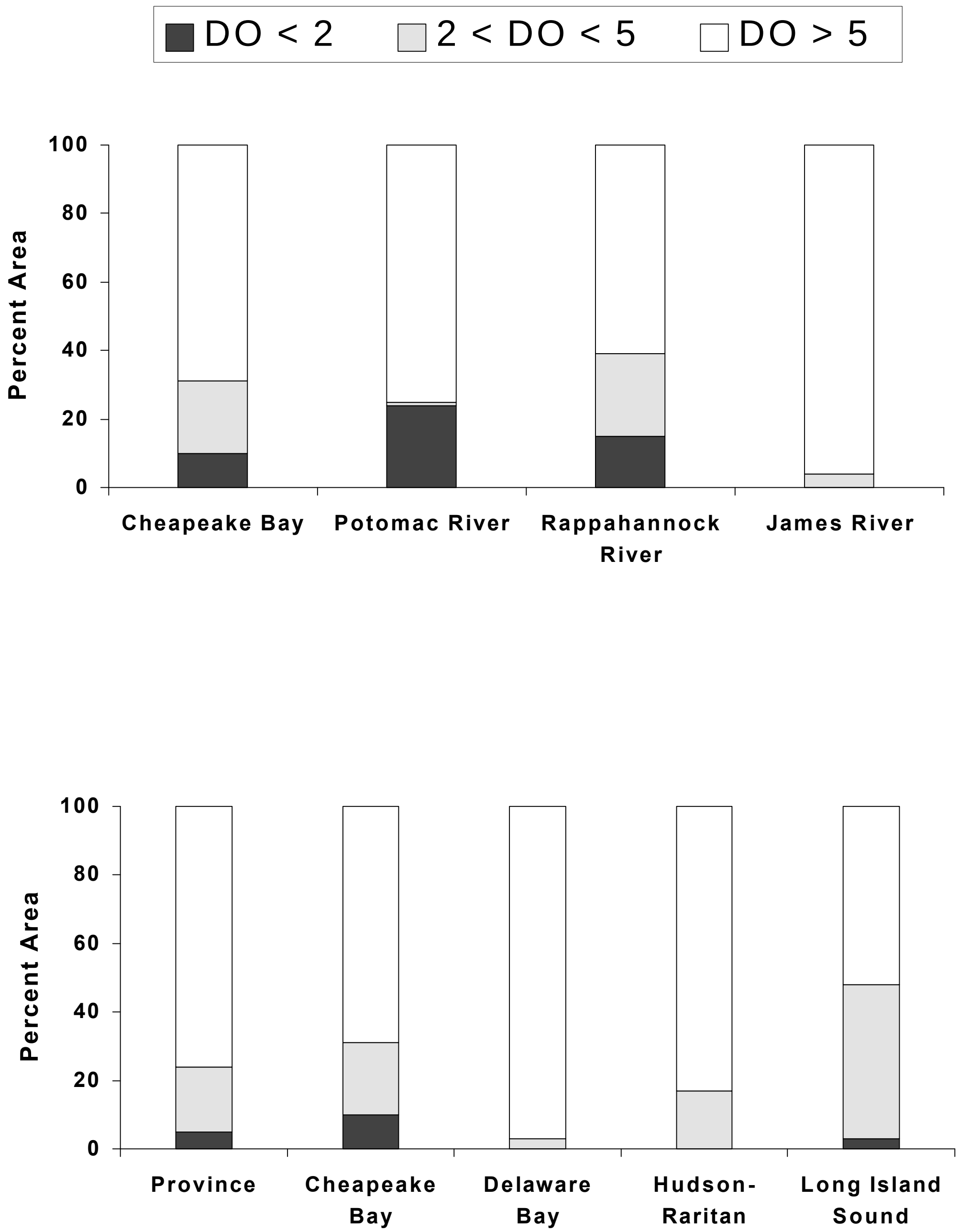

Figure 3-6. Dissolved oxygen condition of bottom waters for major estuarine systems in Virginian Province for the period 1990-93.

Data collected over the four years by EMAP in the Virginian Province measured the acute toxicity of surficial sediments, *i.e.,* top 2 cm. The sediments used for the toxicity test were a subsample of the same composite from which sediment contaminant concentrations and sediment physical/chemical properties were determined.

Sediment toxicity tests were performed on composite sediment samples collected from each station using the standard 10-day acute test method (Swartz *et al.,* 1985; U.S. EPA, 1995b, taken from U.S. EPA, 1994a) and the tube-dwelling amphipod *Ampelisca abdita.* Amphipods were exposed to sediment from the site for 10 days under static conditions. *Ampelisca abdita* has been shown to be both acutely and chronically sensitive to contaminated sediments (Breteler *et al.*, 1989; Scott and Redmond, 1989). Because it is a tube dweller, *Ampelisca* is tolerant of a wider range of sediment types than *Rhepoxynius*, the genus of amphipod that is commonly used in sediment toxicity evaluations (Long and Buchman, 1989). Less than 80% survival relative to control survival in the tested sediments was used as a benchmark for determining toxic sediments, and less than 60% survival was used to define severely toxic sediments.

3.3.1 Sediment Toxicity Condition

The toxic condition of bottom sediments in the Virginian Province is summarized in Table 3-6 and illustrated in Figure 3-7. For the four-year period, sediment toxicity ($\leq$ 80% survival) was observed in 9$\pm$2% of the bottom sediments of the Virginian Province. Toxicity was observed in 4$\pm$4% of the tidal rivers, 9$\pm$3% of the large estuaries, and 12$\pm$6% of small estuarine system area. The range of annual values for sediment toxicity in small estuarine systems was high (1-31%) and low in tidal rivers (0-8%). Analysis of sediment toxicity data on percent area affected suggests that the small estuarine systems are at greatest relative risk from toxic sediments (in the mean but not statistically), while analysis of absolute area indicates that large estuaries exhibit over twice the area of toxic sediments as do small estuarine systems. Toxicity appears to be spread over the entire province (Figure 3-7).

Severe toxicity ($\leq$ 60% survival) was generally low, occurring in only 1% of the estuarine sediments of the Virginian Province, and primarily confined to the northern extent of the province (Figure 3-7). Small estuarine systems exhibited the largest percent area with severe toxicity (3$\pm$4%), although this distinction may not be significant.

Distribution of sediment toxicity in the major estuarine systems for the Virginian Province are illustrated in Figure 3-8 and summarized in Tables 3-7 and 3-8. Toxicity ($\leq$ 80% survival) was exhibited in all of the major estuarine systems, ranging from a low of 2$\pm$2% in Delaware Bay to 15$\pm$14% in the Hudson-Raritan system. The toxicity was observed across all of the classes of these estuarine systems except for Delaware Bay,

Table 3-6. Areal Estimates for Bottom Sediment Toxicity Condition, as Determined from Acute Amphipod Bioassay, in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)

amphipod survival < 60%

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	3	3	1	1	<1	<1	2	3	1	1
Large Systems	2	5	0	0‡	0	0‡	2	4	1	2
Small Systems	6	11	4	6	1	2	4	8	3	4
Tidal Rivers	3	5	0	0‡	0	0‡	0	0‡	<1	<1

amphipod survival < 80%

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	8	5	21	7	6	4	3	3	9	2
Large Systems	2	5	24	9	8	10	2	4	9	3
Small Systems	31	23	17	12	1	2	6	9	12	6
Tidal Rivers	3	5	8	7	3	14	0	0‡	4	4

(b)

amphipod survival < 60%

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	764	707	212	236	73	73	540	707	330	170
Large Systems	375	749	0	0‡	0	0‡	335	671	177	288
Small Systems	316	518	213	287	73	97	205	394	152	197
Tidal Rivers	72	129	0	0‡	0	0‡	0	0‡	<26	<26

amphipod survival < 80%

	1990		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	1966	1179	4875	1650	1346	945	620	707	2105	462
Large Systems	375	749	3833	1408	1207	1610	335	671	1437	447
Small Systems	1519	1137	833	581	73	97	285	452	577	270
Tidal Rivers	72	129	209	195	66	366	0	0‡	91	110

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

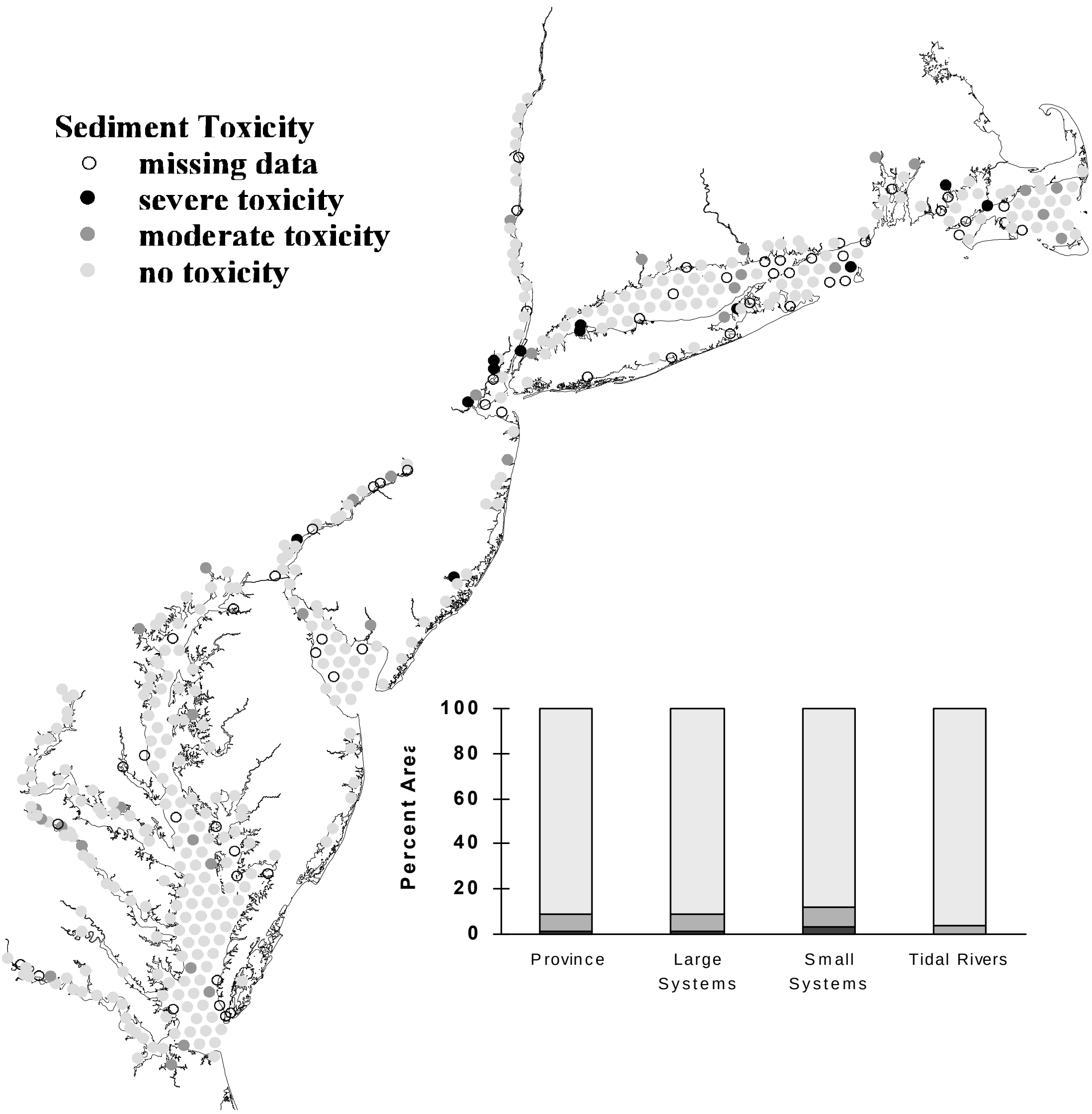

Figure 3-7. Toxicity condition of bottom sediments, as determined from acute amphipod bioassays, in Virginian Province for the period 1990-93.

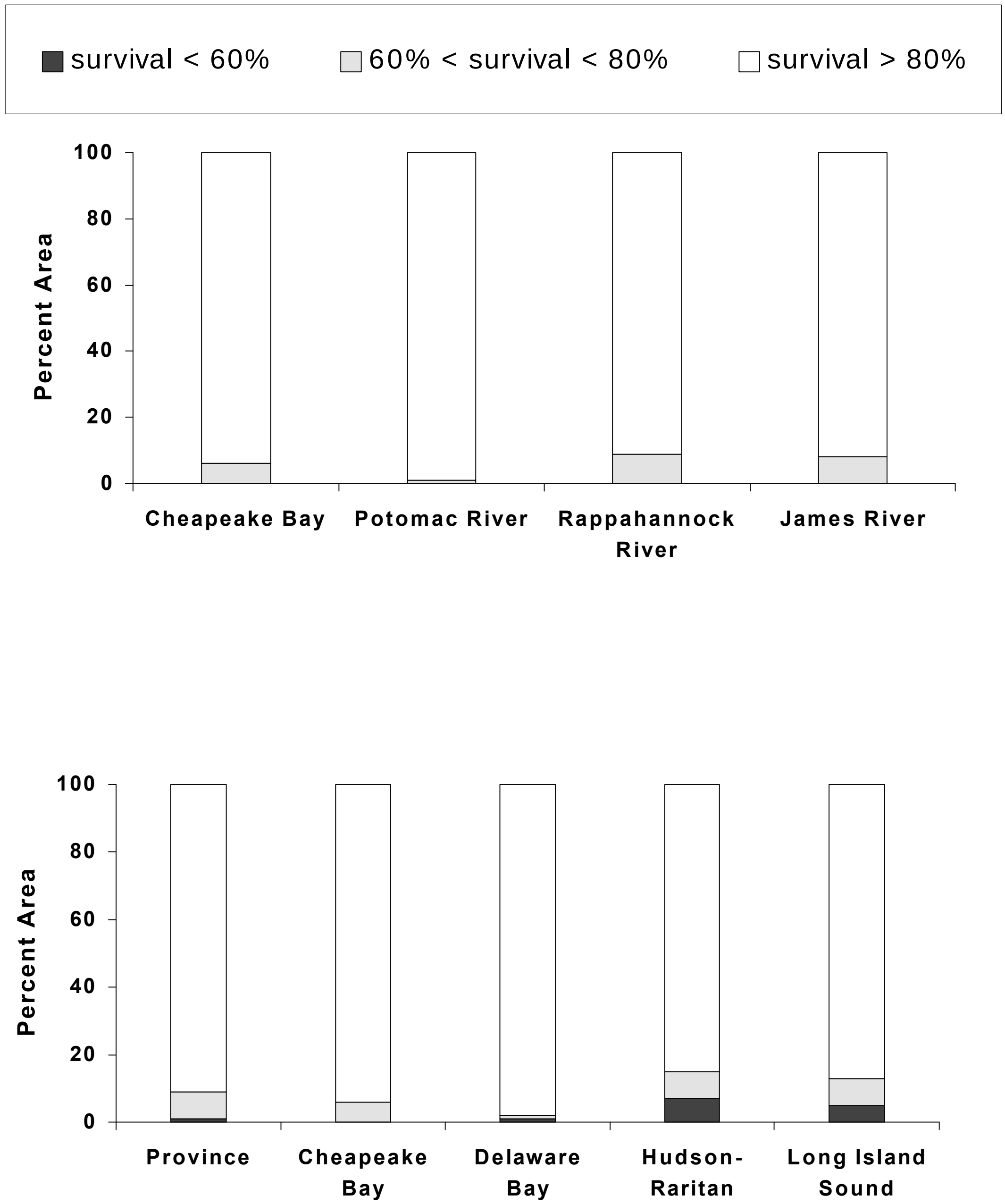

Figure 3-8. Toxicity condition of bottom sediments, as determined from acute amphipod bioassays, for major estuarine systems in Virginian Province for the period 1990-93, amphipod survival < 80%.

Table 3-7. Areal Estimates for Bottom Sediment Toxicity Condition (Survival ≤ 80%) for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	8	6	19	9	1	3	0	0‡	6	3
Potomac River	*	*	*	*	*	*	*	*	1	<1
Rappahannock River	*	*	*	*	*	*	*	*	9	11
James River	*	*	*	*	*	*	*	*	8	11
Delaware Bay	4	8	3	5	0	0‡	0	0‡	2	2
Hudson-Raritan	11	19	24	36	16	29	12	37	15	14
Long Island Sound	0	0	23	19	10	16	13	17	13	7

(b)	**1990**		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	909	684	2,199	972	63	349	0	0‡	695	342
Potomac River	*	*	*	*	*	*	*	*	13	<12
Rappahannock River	*	*	*	*	*	*	*	*	33	39
James River	*	*	*	*	*	*	*	*	54	77
Delaware Bay	90	162	61	112	0	0‡	0	0‡	38	49
Hudson-Raritan	87	143	181	274	122	220	93	281	117	106
Long Island Sound	0	0‡	777	622	341	524	430	552	423	246

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

Table 3-8. Areal Estimates for Bottom Sediment Toxicity Condition (Survival $\leq$ 60%) for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	0	0‡	0	0‡	0	0‡	0	0‡	0	0‡
Potomac River	*	*	*	*	*	*	*	*	0	0‡
Rappahannock River	*	*	*	*	*	*	*	*	0	0‡
James River	*	*	*	*	*	*	*	*	0	0‡
Delaware Bay	4	8	0	0‡	0	0‡	0	0‡	1	2
Hudson-Raritan	0	0‡	5	11	16	29	11	35	7	10
Long Island Sound	0	0‡	3	6	0	0‡	13	17	5	4

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	0	0‡	0	0‡	0	0‡	0	0‡	0	0‡
Potomac River	*	*	*	*	*	*	*	*	0	0‡
Rappahannock River	*	*	*	*	*	*	*	*	0	0‡
James River	*	*	*	*	*	*	*	*	0	0‡
Delaware Bay	90	162	0	0‡	0	0‡	0	0‡	23	41
Hudson-Raritan	0	0‡	40	84	122	220	84	266	54	76
Long Island Sound	0	0‡	102	201	0	0‡	430	552	151	138

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

where toxicity was only observed in the tidal river. Severe toxicity (≤ 60% survival) was observed only in Delaware Bay (1±2%), the Hudson-Raritan system (7±10%), and Long Island Sound (5±4%). This toxicity was found primarily in small systems in the Hudson-Raritan and Long Island Sound, but only in the tidal river portion of Delaware Bay. In summary, sediment toxicity is often associated with the portions of rivers, bays, and estuaries that are near areas of intense urbanization where a variety of anthropogenic stressors from land-based activities impact estuarine ecosystems.

Sediment toxicity tests have repeatedly demonstrated a dose response relationship with single and multiple contaminants under spiked sediment laboratory conditions (*e.g.*, Di Toro *et al*., 1990; Swartz *et al*., 1994). Field studies also have established correlations between sediment toxicity and measures of sediment contamination (Long and Morgan 1990; Long *et al*. 1995). However, concerns have been expressed that the effects of factors other than presence of contaminants may cause toxicity resulting in false positive results (Spies, 1989). The most often cited non-contaminant factors that affect toxicity are particle size and pore water ammonia. The amphipod used in the sediment toxicity test, *Ampelisca abdita*, inhabits fine-grained sediment, and is one of the more abundant and widespread species found in benthic communities of the Virginian Province, in water with salinity greater than 15 o/oo. The potential effects of particle size would thus be operative in the coarser fraction of the size range. The silt-clay content of the sediments tested over the four years ranged from < 1 to 100%, yet there was no relationship, statistical or otherwise, between amphipod survival and silt-clay content.

Estimates of pore water and overlying water concentrations of total and un-ionized ammonia that are toxic to *Ampelisca abdita* were established in U.S. EPA (1994a). No effect concentrations were estimated to be 30 µg/l and 0.4 µg/l for total and un-ionized ammonia, respectively, at pH 7.7. Concentrations exceeding these levels indicate the potential for some toxicity due to ammonia. Ammonia concentrations were measured in pore water of sediment collected in 1993 prior to sediment toxicity testing. There was no conclusive evidence that ammonia was causing toxicity; sediment with elevated ammonia concentrations were as likely to be non-toxic as toxic.

A sediment toxicity test that is predictive of benthic community responses to contaminants should have endpoints that correlate with community parameters of interest. Comparison of amphipod survival in sediment toxicity test with abundance of ampeliscid and non-ampeliscid amphipods is presented in Figure 3-9. Ampeliscid abundance did not exceed $1140/m^2$ when survival in the toxicity test was less than 50%. At 80% survival, only six sediments contained ampeliscids at densities > $1140/m^2$. The remaining high densities of these ampeliscids were found at non-toxic sites. The presence of ampeliscids at moderate to low abundances ($0\text{-}1140/m^2$) in sediments determined to be toxic is not unusual. When sediments are disturbed, as they are in the process of collection and sample processing prior to toxicity testing (*i.e.*,

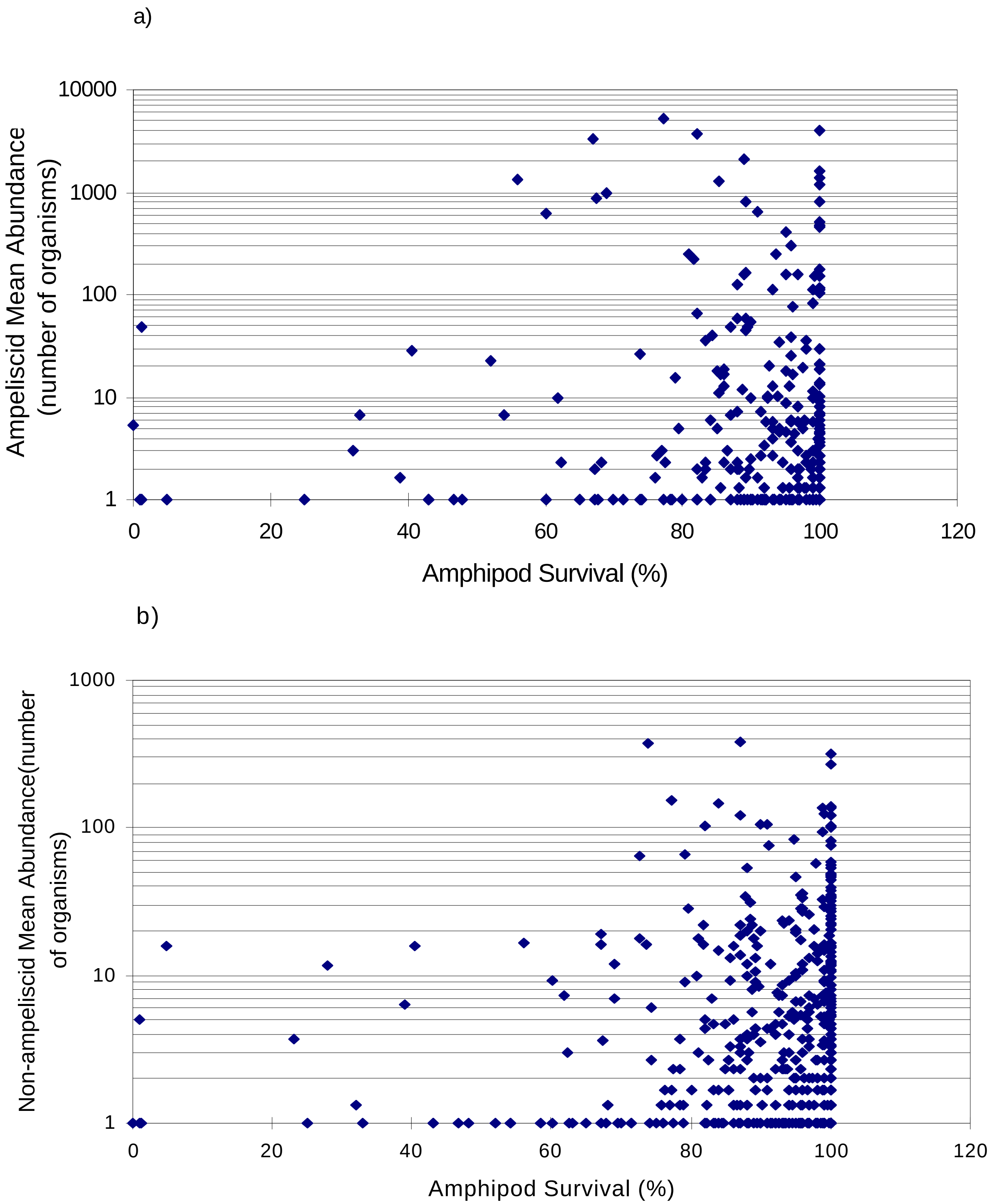

Figure 3-9. Relationships between benthic organism abundance and sediment toxicity bioassay. a) Ampeliscid abundance vs. amphipod survival. b) Non-ampeliscid abundance vs. amphipod survival.

press sieved and homogenized), chemical equilibrium can be disrupted and contaminants made available to cause toxicity. This effect on contaminant release has been described for non-polar organic compounds (Word *et al*., 1994), and may be responsible for the toxicity observed in the six sediments with high ampeliscid abundance and survival $\leq$ 80%. It should be noted that these sediments exhibited concentrations of at least one metal at or above ER-L values in conjunction with elevated PAHs at some stations.

3.4 Sediment Contamination Condition Indicators

Metals, organic chemicals, and fine-grained sediments enter estuaries from freshwater inflows, point sources of pollution and various nonpoint sources, including atmospheric deposition. Contaminants generally are retained within estuaries and accumulate within the sediments (Turekian, 1977; Forstner and Wittman, 1981; Nixon *et al.,* 1986; Hinga, 1988; Schubel and Carter, 1984) because most have an affinity for adsorption onto particles (Hinga, 1988; Honeyman and Santschi, 1988). Chemical and microbial contaminants generally adsorb to fine-grained materials in the water and are deposited on the bottom, accumulating at deposition sites, including regions of low current velocity, deep basins, and the zone of maximum turbidity. The concentrations of contaminants in sediments is dependent upon interactions between natural (*e.g.*, physical sediment characteristics) and anthropogenic factors (*e.g*., type and volume of contaminant loadings) (Sharp *et al*., 1984).

Composite, surficial sediment samples were collected using the same procedures over the four-year period and analyzed for the NOAA National Status and Trends suite of contaminants (NOAA, 1992) using subsamples from the homogenized sediment samples. The NOAA suite includes chlorinated pesticides, polychlorinated biphenyls (PCBs), polycyclic hydrocarbons (PAHs), major elements, and metals (Table 3-9).

Benchmarks of sediment contamination currently are expressed in a number of ways and can be grouped into two ways that describe one's understanding of what sediment contamination represents. The first, and most common, form of expression simply presents contaminant concentrations as a cumulative distribution and a criterion for contamination is selected based on the nature of the distribution. Exceedence of high percentile concentrations, that are sometimes normalized to the proportion of silt-clay, are then denoted as contaminated. Examples of this approach are found in O'Connor (1990) and Daskalakis and O'Connor (1994). The second way to establish whether a site is contaminated is to compare the contaminant concentration against those considered to be naturally occurring or at background levels. The mere presence of synthetic organics causes a sediment to be considered contaminated. Regression relationships of metal concentrations to those of conservative crustal elements, such as aluminum, are used to determine the degree of enrichment for these compounds

Table 3-9. Chemical Measurements Conducted for Sediments of the Virginian Province.

Polycyclic Aromatic Hydrocarbons (PAHs)

Acenaphthathene
Acenaphthlylene
Anthracene
Benz(a)anthracene
Benz(a)pyrene
Benzo(a)pyrene
Benzo(k)fluoranthene
Benzo(g,h,i)perylene
Benzo(e)pyrene
Benzo(b)fluoranthene
Biphenyl
Chrysene
Dibenz(a,b)anthracene
2,6-dimethylnaphthalene
Fluoranthene
Fluorene
1-methylnaphthalene
2-methylnaphthalene
2-methylphenanthrene
Naphthalene
Perylene
Phenanthrene
Pyrene

Major elements

Al	Aluminum
Fe	Iron
Mn	Manganese
Si	Silicon

Trace elements

Sb	Antimony
As	Arsenic
Cd	Cadmium
Cr	Chromium
Cu	Copper
Pb	Lead
Hg	Mercury
Ni	Nickel
Se	Selenium
Ag	Silver
Sn	Tin
Zn	Zinc

DDT and its metabolites

o,p'-DDD
p,p'-DDD
o,p'-DDE
p,p'-DDE
o,p'-DDT
p,p'-DDT

Other chlorinated pesticides

Aldrin
Alpha-chlordane
Trans-Nonachlor
Dieldrin
Heptachlor
Heptachlor epoxide
Hexachlorrobenzene
Lindane (gamma-BHC)
Mirex

PCB congeners

Congener Number	Location of Cl's
8	2 4'
18	2 2' 5
28	2 4 4'
44	2 2' 3 5'
52	2 2' 5 5'
66	2 3' 4 4'
101	2 2' 4 5 5'
105	2 3 3' 4 4'
118	2 3' 4 4' 5
128	2 2' 3 3' 4 4'
138	2 2' 3 4 4' 5'
153	2 2' 4 4' 5 5'
170	2 2' 3 3' 4 4' 5
180	2 2' 3 4 4' 5 5'
187	2 2' 3 4' 5 5' 6
195	2 2' 3 3' 4 4' 5 6
206	2 2' 3 3' 4 4' 5 5' 6
209	2 2' 3 3' 4 4' 5 5' 6 6'

Other measurements

Tributyltin
Acid volatile sulfides
Total organic carbon

(Windom *et al*., 1989; see Strobel *et al.*, 1995, for the specific regressions used). These two approaches are useful in cases where a knowledge of land-based or atmospheric source inputs is important, and there is less regard for the relationship between the chemical concentration and biological or ecological effects in the environment.

There are two classes of approaches that establish criteria for contamination that relate the observed concentration to some biological effect. In the first approach, bulk chemical concentrations are compared to concentrations known to either: (1) cause biological effects in spiked-sediment or spiked-water laboratory experiments; or (2) are associated with biological effects in field studies. Examples of these approaches are the Puget Sound apparent effects thresholds (AETs), State of Washington screening level concentrations (SLCs), and effects range median (ER-M) and effects range low (ER-L) concentrations of Long and Morgan (1990), as updated in Long *et al*. (1995). All of these approaches benefit from the weight of evidence afforded by large data sets associating bulk concentration with biological effect, but suffer from a failure to incorporate the effects of multiple chemicals in complex mixtures.

The second approach to effects-based criteria relies on equilibrium partitioning theory which predicts the concentration of chemical that elicits a biological effect (bioavailable concentration) from a bulk sediment concentration. This approach incorporates the role of sediment binding factors, such as organic carbon (OC) for organic compounds, in defining chemical availability (Di Toro *et al*., 1991). The EPA sediment quality criteria for acenaphthene (USEPA, 1993a), phenanthrene (USEPA, 1993b), fluoranthene (USEPA, 1993c), and dieldrin (USEPA, 1993d) are based upon this approach. A total PAH concentration of 200 µg/g-OC approximates the sediment quality criteria values for the three PAHs in this list. An equilibrium partitioning-based concentration of total DDTs of 100 µg/g-OC has been described by Swartz *et al*. (1994) to cause biological effects.

Since the sediment contaminant data collected in the Virginian Province are used primarily to interpret condition of the benthic community, the major emphasis will be on contaminant criteria related to biological effects. This section presents comparisons of chemical concentrations with: crustal levels (metals), ER-M and ER-L values (Long *et al*., 1995), EPA sediment quality criteria, a criterion for organic carbon normalized total PAHs, and the Swartz *et al*. (1994) organic carbon normalized total DDT criterion.

3.4.1 Sediment Contamination Condition: Areal Patterns

3.4.1.1 Metals enrichment

Sediment metals enrichment above crustal levels for the province is summarized in Table 3-10. The metals analyzed for enrichment include As, Cr, Fe, Hg, Mn, Ni, Sb,

Table 3-10. Areal Estimates for Enriched Metal Concentrations in Bottom Sediments in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	35	7	38	10	57	7	64	7	49	4
Large Systems	29	9	31	10	54	10	52	10	42	5
Small Systems	49	22	43	18	67	15	83	23	64	7
Tidal Rivers	46	32	70	18	59	20	100	0‡	69	10

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	8,251	1,721	8,958	2,240	13,461	1,721	15,017	1,650	11,622	849
Large Systems	4,700	1,497	5,038	1,529	8,676	1,642	8,387	1,658	6,696	789
Small Systems	2,369	1,092	2,116	853	3,271	741	4,022	1,116	3,135	317
Tidal Rivers	1,192	820	1,808	458	1,525	520	2,602	0‡	1,782	268

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

and Zn. An area is enriched if at least one of these metals exceeds the concentration expected based on crustal weathering. For the entire province during the four-year sampling interval, 49±4% of the area has at least one enriched metal. Areal extent of enrichment ranged from 35±7% in 1990 to 64±7% in 1993. Sediments in large estuarine systems exhibited enrichment in 42±5% of the area, with annual values ranging from 29±9% in 1990 to 54±10% in 1992. Small systems and tidal rivers exhibited larger percent area with metals enrichment, 64±7% and 69±10%, respectively. These two classes also exhibited higher ranges in annual values in the estimates of percent area of enrichment: 43±18% in 1991 to 83±23% in 1993 for small systems and 46±32% in 1990 to 100% in 1993 for tidal rivers. These percent area estimates indicate that small systems and tidal rivers are more enriched with metals than the large systems.

Although sediments enriched with metals are observed in significant portions of the sediments in the Virginian Province, no implication should be drawn concerning potential ecological impact. For the metals levels to be of biological consequence, the metals must be biologically available and above a level found to be toxic. The sediment metals enrichment indicator only identifies those sediments that have concentrations above expected crustal, or background, levels, and are potentially more useful in trends rather than status assessments.

Table 3-11. Areal Estimates for Bottom Sediment Contaminant Condition for Metals in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)

	Any ER-M Metal Exceedence									
	1990		**1991**		**1992**		**1993**		**1990-1993**	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	5	4	5	3	7	4	1	1	5	1
Large Systems	2	4	2	4	8	10	0	0‡	3	3
Small Systems	18	16	11	8	6	5	3	4	9	4
Tidal Rivers	3	5	11	14	5	4	8	8	7	4

(b)

	Any ER-M Metal Exceedence									
	1990		**1991**		**1992**		**1993**		**1990-1993**	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	1,294	943	1,148	592	1,638	879	349	198	1,073	297
Large Systems	335	671	335	671	1,207	1,610	0	0‡	470	404
Small Systems	868	786	532	406	313	234	140	216	429	172
Tidal Rivers	90	138	282	353	118	117	209	197	175	111

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

3.4.1.2 Metal ER-M exceedence

Sediment contaminant condition relative to ER-M exceedence for metals is summarized in Table 3-11 and illustrated in Figure 3-10. Exceedence of the ER-M for at least one metal for which ER-Ms exist was found in 5±1% of the area of the province during the four-year sampling interval. The percent area of exceedence ranged from 1% in 1993 to 7% in 1992. Sediments in large estuarine systems exhibited exceedence in 3±3% of the area with annual values ranging from zero in 1993 to 8% in 1992. Seven percent of the area (±4%) in tidal rivers contained sediments with ER-M exceedence; the percent area of exceedence was low in 1990 and 1992 (3-5%), and higher in 1991 and 1993 (11% and 8%, respectively), but these differences may not be significant. Small estuarine systems exhibited the highest proportion of area, 9±4%, with ER-M exceedence. Most of those high concentrations were found in 1990 and 1991 (18% and 11%, respectively). These ER-M metals exceedence suggest that small estuarine systems are at greater risk than are large or tidal river systems. It is worth noting that the annual variability for metal exceedence parallels that observed with sediment toxicity, which was also greatest in small systems in 1990 and in tidal rivers in 1991. These annual patterns in condition support the rationale of using the entire four-year data set to minimize the uncertainty in the description of estuarine condition.

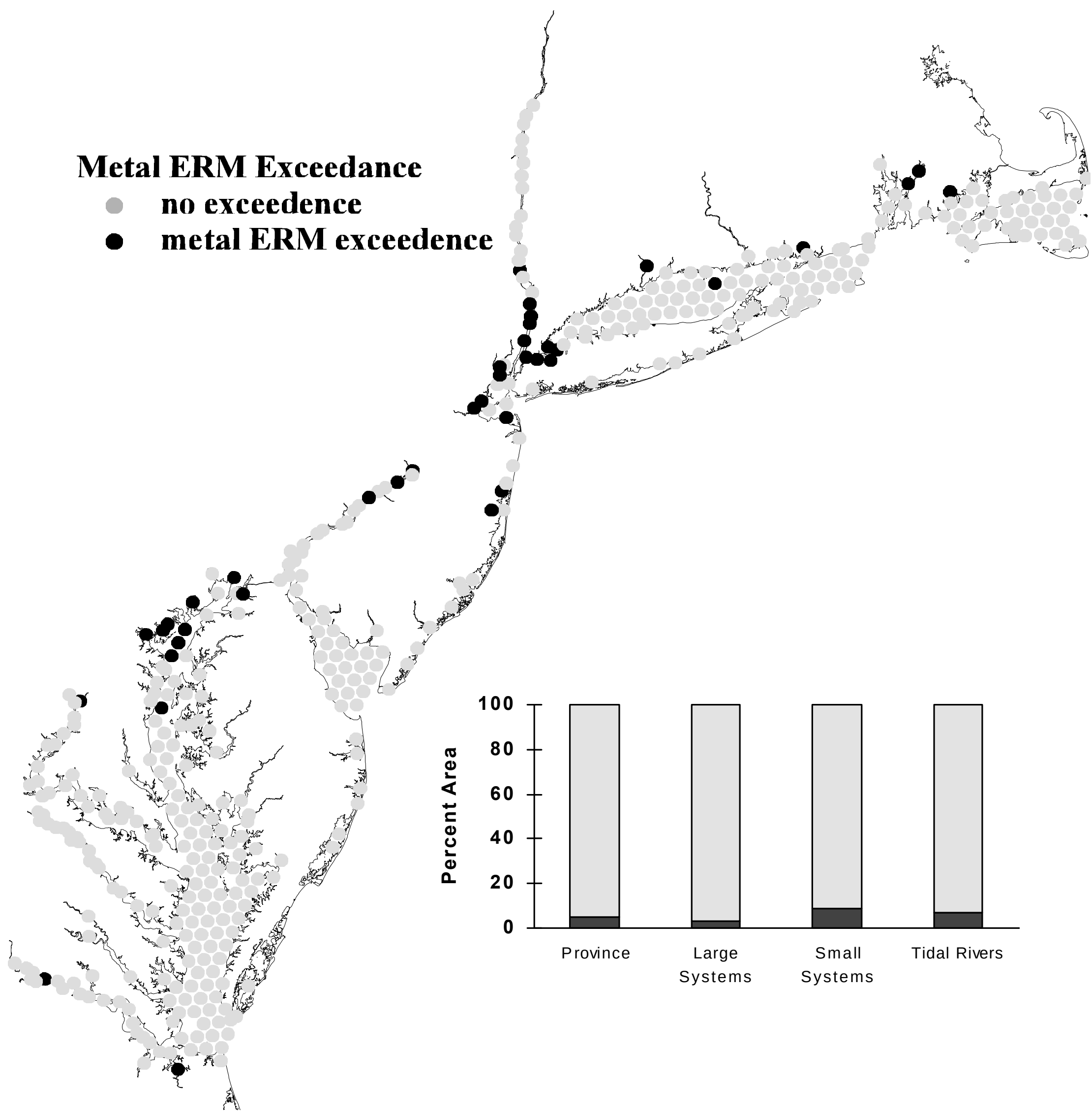

Figure 3-10. Sediment contaminant condition of bottom sediments in Virginian Province for the period 1990-93, ER-M metals exceedence.

The approach used for this indicator was developed by attempts to identify concentrations of contaminants that are rarely associated with adverse biological effects (ER-L, discussed in a later section) and those usually associated with effects (ER-M), utilizing data from an extensive search of the literature (Long *et al.*, 1995). The underlying assumption of this approach was that, if enough data are accumulated, a pattern of increasing incidence of biological effects should emerge with increasing contaminant concentrations. It has been argued that sediment quality criteria are not defensible if they do not account for factors that control bioavailability, such as acid volatile sulfides for metals and TOC for organic compounds (Di Toro *et al.,* 1991). However, Long *et al.* (1995) argue that the approach is accurate for most chemicals and that their results agree reasonably with other guidelines. They then conclude that their guidelines are likely to be reliable tools in sediment quality assessments. The debate does continue on the use of this approach as an indicator of sediment contamination.

3.4.1.3 Organic ER-M exceedence

The percent area of ER-M exceedence for organics is found in Table 3-12 and Figure 3-11. Note that because of Quality Assurance (QA) problems associated with organic chemistry results from 1990 (Strobel and Valente, 1995), the organic chemistry analyses for 1990 are not presented and are not used to produce the four-year estimates. Exceedence of at least one ER-M value was found in 3±1% of the area of the Virginian Province for the four-year period. The proportional extent of organic contamination was low in large and small estuarine systems (2%) and higher in tidal rivers (14±2%). Much of this is due to the well-documented PCB contamination in the Hudson River (Feng *et al.,* 1998). The areal extent of organic ER-M exceedence in 1991, 1992, and 1993 was 11%, 19%, and 12%, respectively. Tidal rivers appear to be at the greatest risk to organic contamination. Based on the area affected, the extent of contamination in tidal rivers is less than that in the large estuarine class.

Similar comments on the utility of this indicator, as discussed with the ER-M metals results in section 3.4.1.2), apply here.

3.4.1.4 Organic carbon normalized total PAHs

The results for potentially toxic PAH sediment contamination based on organic carbon normalization is shown in Table 3-12. Note that because of QA problems associated with organic chemistry results from 1990 (Strobel and Valente, 1995), the organic chemistry analyses for 1990 are not presented and are not used to produce the four-year estimates. The extent of estuarine area with PAH contamination, as represented by concentrations > 200 µg/g-OC, was low (5±1%) in the province for the years 1991-1993. Variability amongst years also was low. The affected area in large systems was lowest (2±2%), and that in the tidal river systems was highest (14±6%);

Table 3-12. Areal Estimates for Bottom Sediment Contaminant Condition for Organics in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)

Any ER-M Organic Exceedence

	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	‡‡	‡‡	4	2	4	2	3	2	4	1
Large Systems	‡‡	‡‡	2	4	3	5	2	4	2	3
Small Systems	‡‡	‡‡	5	6	1	2	2	4	3	2
Tidal Rivers	‡‡	‡‡	11	6	19	8	12	6	14	4

OC Normalized PAHs > 200 mg/g-OC

	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	‡‡	‡‡	6	3	5	3	5	4	5	2
Large Systems	‡‡	‡‡	2	4	3	5	0	0‡	2	2
Small Systems	‡‡	‡‡	11	8	11	7	13	18	11	7
Tidal Rivers	‡‡	‡‡	10	5	13	8	19	17	14	6

(b)

Any ER-M Organic Exceedence

	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	‡‡	‡‡	849	500	959	556	743	497	850	299
Large Systems	‡‡	‡‡	335	671	402	805	335	671	357	415
Small Systems	‡‡	‡‡	222	268	73	97	104	203	133	117
Tidal Rivers	‡‡	‡‡	291	164	484	208	303	158	360	103

OC Normalized PAHs > 200 mg/g-OC

	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	‡‡	‡‡	1,414	707	1,247	707	1,132	943	1,264	458
Large Systems	‡‡	‡‡	335	671	402	805	0	0‡	246	349
Small Systems	‡‡	‡‡	514	401	518	343	641	897	558	347
Tidal Rivers	‡‡	‡‡	272	131	327	208	490	441	363	168

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

‡‡ Due to QA problems with the organic chemistry analyses for 1990 samples (Strobel and Valente, 1995), results for 1990 organic chemistry analyses are not presented.

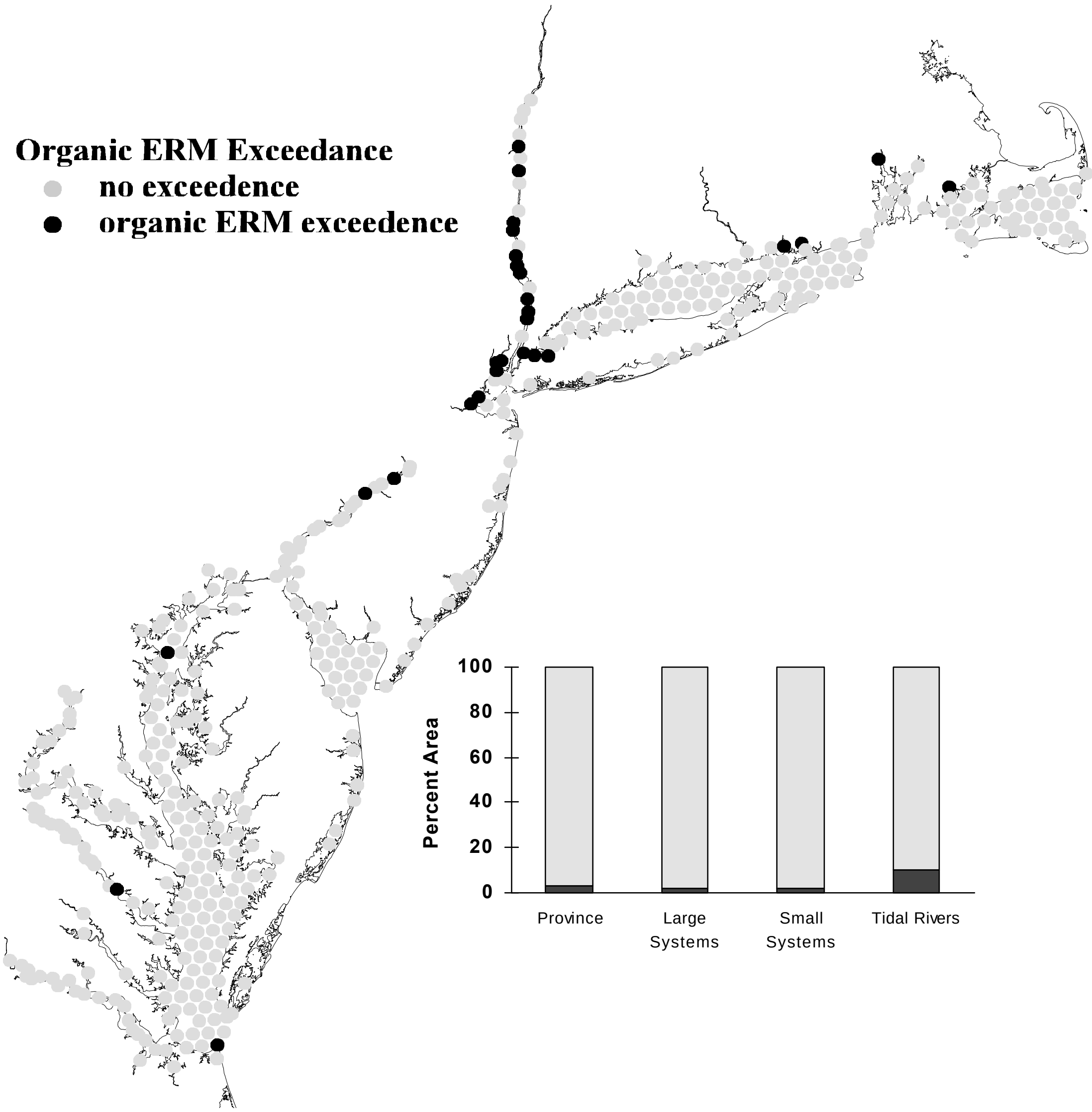

Figure 3-11. Sediment contaminant condition of bottom sediments in Virginian Province for the period 1990-93, ER-M organics exceedence.

the small estuarine systems exhibited an intermediate level of impact (11±4%). The range for annual values in large and small estuarine systems was moderate; in tidal river systems, the extent of degradation in 1991-1993 ranged from 10 to 19%. These data suggest that the risk due to bioavailable PAHs on the province scale is low, and more likely to occur in the small systems and tidal rivers.

3.4.1.5 Organic carbon normalized total DDT

There were no sample sites exceeding the criterion of total DDT concentrations > 100 µg/g-OC in the Virginian Province in 1991-1993.

3.4.1.6 Organic sediment quality criteria

The estuarine area of the Province that contained sediments exceeding any one of the sediment quality criteria for the four criteria chemicals (acenaphthene - 230 µg/g-OC; phenanthrene - 240 µg/g-OC, fluoranthene - 650 µg/g-OC; dieldrin - 20 µg/g-OC; USEPA, 1993a-d) was < 1%. Those exceedence were restricted to small estuarine systems, which exhibited < 1% of the area exceeding the criteria.

3.4.1.7 ER-L exceedence

The results for no ER-L exceedence (metals and organics) are summarized in Table 3-13. **Note that these results are depicted as areas for which there were no ER-L exceedence observed**. In contrast to the results presented for ER-M exceedence which indicated potential for biological effects, the ER-L results are presented indicating areas for which observed sediment contaminant concentrations are below levels for which biological effects may be of concern. Half of the entire province (50±4%) had sediment contaminant concentrations below ER-L levels. Annual values ranged from 44±8% in 1993 to 53±7% in 1992. The large estuarine system class exhibited the largest areal extent of sediments below ER-L values, 58±5%, with small systems and tidal rivers exhibiting 35±6% and 25±12%, respectively. The range in annual values was smallest for tidal rivers (from 21±30% in 1993 to 29±17% in 1991) and largest for small systems (from 23±17% in 1990 to 51±39% in 1993). These results indicate that sediments of the small systems and tidal rivers are more at risk to sediment contaminant levels that have been observed to potentially elicit biological responses.

3.4.2. Sediment Contamination Condition: Estuarine System Analysis

3.4.2.1 Metals enrichment

Sediment metals enrichment results for the major estuarine systems are summarized in Table 3-14. Chesapeake and Delaware Bays exhibited the least

Table 3-13. Areal Estimates for Bottom Sediment Contaminant Condition (No ER-L Exceedence) in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Province	49	7	50	7	53	7	44	8	50	4
Large Systems	60	10	60	10	65	10	46	10	58	5
Small Systems	23	17	29	18	28	13	51	39	35	6
Tidal Rivers	25	24	29	17	24	26	21	30	25	12

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Province	11,516	1,730	11,884	1,676	12,464	1,714	10,431	1,825	11,768	865
Large Systems	9,726	1,611	9,726	1,616	10,463	1,578	7,377	1,648	9,407	805
Small Systems	1,129	821	1,393	881	1,384	618	2,503	1,896	1,712	270
Tidal Rivers	660	620	766	441	618	668	550	782	648	320

enrichment of the major systems, 44±5% and 39±14%, respectively. However, within these individual systems, the small systems and tidal river classes exhibited much higher enrichment than that exhibited for the large system class, consistent with the pattern shown for the entire province. The metals enrichment for the Hudson-Raritan system and Long Island Sound were almost identical, 85±11% and 86±8%, respectively. In contrast to the pattern seen across the province, Long Island Sound large system class exhibited more enrichment (89±9%) than the small system class (59±17%). These results for the major estuarine systems indicate that Hudson-Raritan and Long Island Sound are almost entirely enriched with metals, while less than half of Chesapeake and Delaware Bays exhibit metals enrichment. However, the small systems and tidal rivers within all of these major estuarine systems exhibited significant metals enrichment. The observed patterns of enrichment appear to be consistent with the patterns of population distribution and point source discharges across the province, *i.e.*, higher levels of both in the New York City area.

3.4.2.2 Metal ER-M exceedence

The percent of estuarine area with exceedence of ER-M metals in geographic areas are provided in Table 3-15 and Figure 3-12. The Hudson-Raritan Estuary is the most contaminated major system with respect to this indicator; this system exhibited 27±9% of the area with sediments exceeding at least one ER-M value. Most of this contamination was due to mercury, lead, and silver in the tidal portion of the Hudson River (30% of the Hudson-Raritan area). The least contaminated system was Delaware

Table 3-14. Areal Estimates for Enriched Metal Concentrations in Bottom Sediments for the Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	30	9	32	9	54	11	59	9	44	5
Potomac River	*	*	*	*	*	*	*	*	53	14
Rappahannock River	*	*	*	*	*	*	*	*	81	19
James River	*	*	*	*	*	*	*	*	85	14
Delaware Bay	8	6	38	29	44	34	64	32	39	14
Hudson-Raritan	69	25	68	35	100	0‡	100	0‡	85	11
Long Island Sound	88	16	75	21	100	0‡	85	17	86	8

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	3,365	1,004	3,650	992	6,205	1,198	6,673	1,038	5,042	525
Potomac River	*	*	*	*	*	*	*	*	663	176
Rappahannock River	*	*	*	*	*	*	*	*	301	69
James River	*	*	*	*	*	*	*	*	575	94
Delaware Bay	156	115	784	597	906	696	1,316	649	793	282
Hudson-Raritan	527	190	519	266	760	0‡	760	0‡	644	84
Long Island Sound	2,950	525	2,509	692	3,345	0‡	2,843	552	2,880	258

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

Table 3-15. Areal Estimates for Bottom Sediment Contaminant Condition (Any ER-M Metal Exceedence) for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	14	6	4	5	9	6	1	2	5	3
Potomac River	*	*	*	*	*	*	*	*	<1	<1
Rappahannock River	*	*	*	*	*	*	*	*	0	0‡
James River	*	*	*	*	*	*	*	*	8	10
Delaware Bay	<1	6	1	5	0	0‡	2	7	1	3
Hudson-Raritan	34	28	29	18	30	32	26	23	27	9
Long Island Sound	7	1	3	6	12	16	<1	1	4	4

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	1,592	684	449	573	1,020	708	68	228	594	342
Potomac River	*	*	*	*	*	*	*	*	3	<12
Rappahannock River	*	*	*	*	*	*	*	*	0	0‡
James River	*	*	*	*	*	*	*	*	52	68
Delaware Bay	5	115	20	112	0	0‡	44	148	17	55
Hudson-Raritan	258	213	222	137	226	243	196	175	209	68
Long Island Sound	221	33	113	201	404	524	13	33	146	131

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

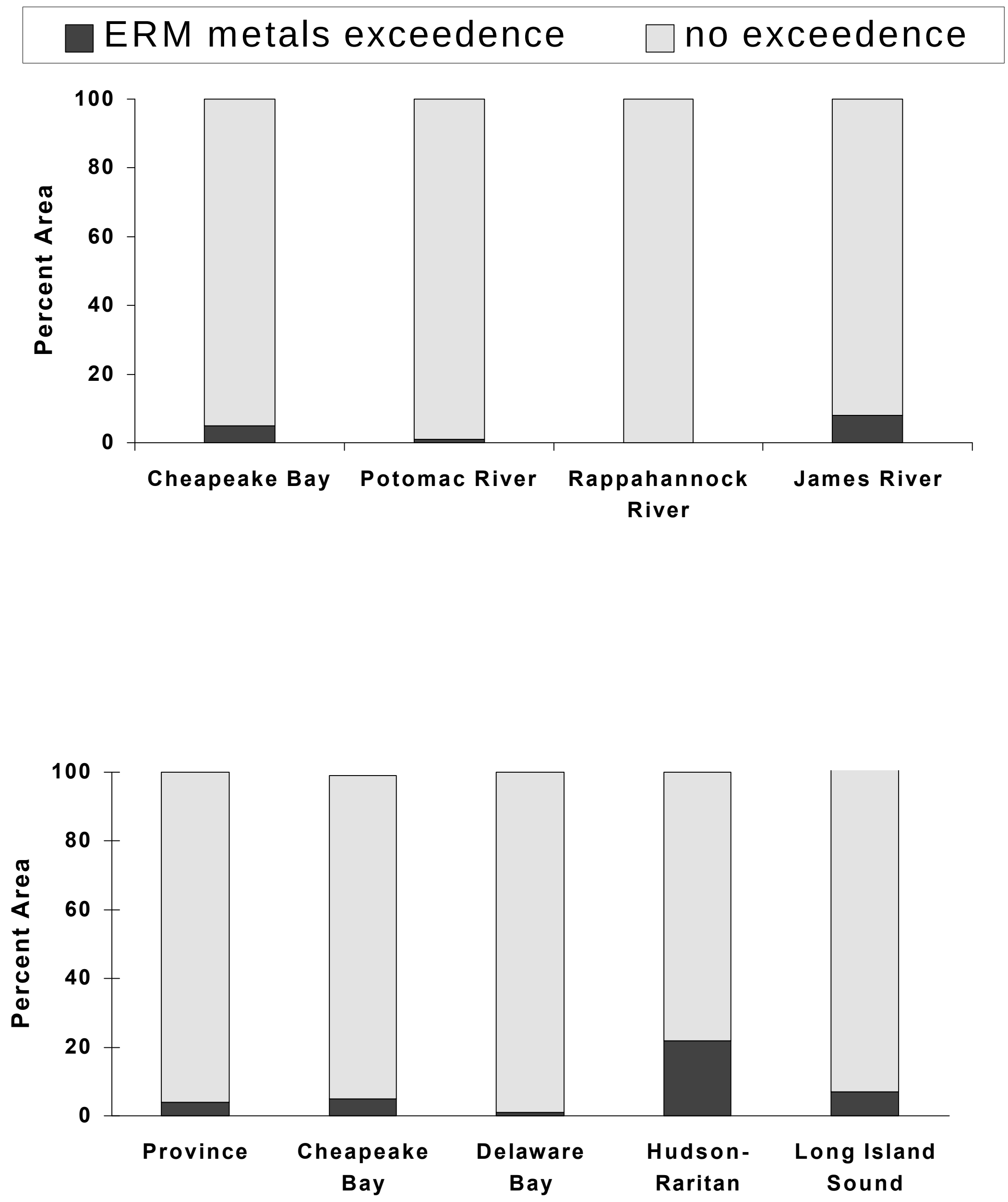

Figure 3-12. Sediment contaminant condition of bottom sediments for major estuarine systems in Virginian Province for the period 1990-93, ER-M metals exceedence.

Bay where only 1±3% of the area contained sediments exceeding ER-M values. These exceedence were restricted to elevations of lead, nickel, and zinc in the Delaware River. Chesapeake Bay and Long Island Sound were intermediate in level of contamination with percent area of exceedence of 5±3% and 4±4%, respectively. Elevations of nickel and zinc in the open waters of upper Chesapeake Bay and bordering small estuarine systems accounted for exceedence in this system. Long Island Sound contained the widest range of metal contamination with concentrations of mercury, lead, silver, copper, and nickel exceeding ER-M values. These exceedence were predominately found in small systems bordering the open waters of the western Sound.

3.4.2.3 Organic ER-M exceedence

Table 3-16 and Figure 3-13 describe the percent area of ER-M organic contaminant concentration exceedence. The Delaware Bay system was the least contaminated of the four major systems. Less than 1% of its area contained sediments exceeding organic ER-M values, and those sediments were restricted to the Delaware River. Elevated concentrations of the chlorinated pesticides pp-DDE and total DDTs accounted for these exceedence. A similarly low level of contamination was found in Chesapeake Bay and Long Island Sound; 2-4% of their areas contained sediments with organics exceeding the ER-M values. Elevations in concentration of a number of PAHs resulted in 5±6% of the area of the open waters of Chesapeake Bay and 22±23% of the Rappahannock River being contaminated relative to this indicator. High concentrations of high and low molecular weight PAHs, total DDTs, and total PCBs in 20±27% of the area in small estuarine systems bordering open waters of Long Island Sound resulted in exceedence of the ER-M values. The greatest level of organic contamination was found in the Hudson-Raritan system: 44±14% of the area of this system contained sediments with organic concentrations that exceeded at least one ER-M value. Eighty-seven percent (±23%) of the area in the tidal portion of the Hudson River, and 14±18% of the area in the small systems exhibited ER-M exceedence. Small estuarine systems were contaminated primarily with PAHs. The Hudson River was contaminated with PCBs and, to a lesser extent, DDTs.

3.4.2.4 Organic carbon normalized total PAHs

The extent of estuarine area exceeding 200 µg/g-OC total PAH is detailed in Table 3-17. The Hudson-Raritan Estuary was, by far, the most contaminated system with respect to total PAH concentrations exceeding 200 µg/g-OC. Sixty-three percent (±18%) of the area of this system contained sediments exceeding this criterion. The affected area was proportionately greater in the Hudson River (74±17%) than it was in the small systems (56±28%). The extent of OC-normalized PAH contamination was uniformly low in Chesapeake Bay, Long Island Sound, and Delaware Bay (2 to 5%). The greatest degree of contamination, other than found in the Hudson-Raritan, was in

Table 3-16. Areal Estimates for Bottom Sediment Contaminant Condition (Any ER-M Organics Exceedence) for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	‡‡	‡‡	3	4	5	5	3	4	4	2
Potomac River	*	*	*	*	*	*	*	*	0	0‡
Rappahannock River	*	*	*	*	*	*	*	*	22	23
James River	*	*	*	*	*	*	*	*	0	0‡
Delaware Bay	‡‡	‡‡	1	5	0	0‡	0	0‡	<1	2
Hudson-Raritan	‡‡	‡‡	36	18	53	32	42	23	44	14
Long Island Sound	‡‡	‡‡	3	6	0	0‡	2	3	2	2

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	‡‡	‡‡	350	479	574	543	292	439	405	282
Potomac River	*	*	*	*	*	*	*	*	0	0‡
Rappahannock River	*	*	*	*	*	*	*	*	83	84
James River	*	*	*	*	*	*	*	*	0	0‡
Delaware Bay	‡‡	‡‡	20	112	0	0‡	0	0‡	7	37
Hudson-Raritan	‡‡	‡‡	276	137	400	243	322	175	333	110
Long Island Sound	‡‡	‡‡	113	201	0	0‡	53	100	55	75

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

‡‡ Due to QA problems with the organic chemistry analyses for 1990 samples (Strobel and Valente, 1995), results for 1990 organic chemistry analyses are not presented.

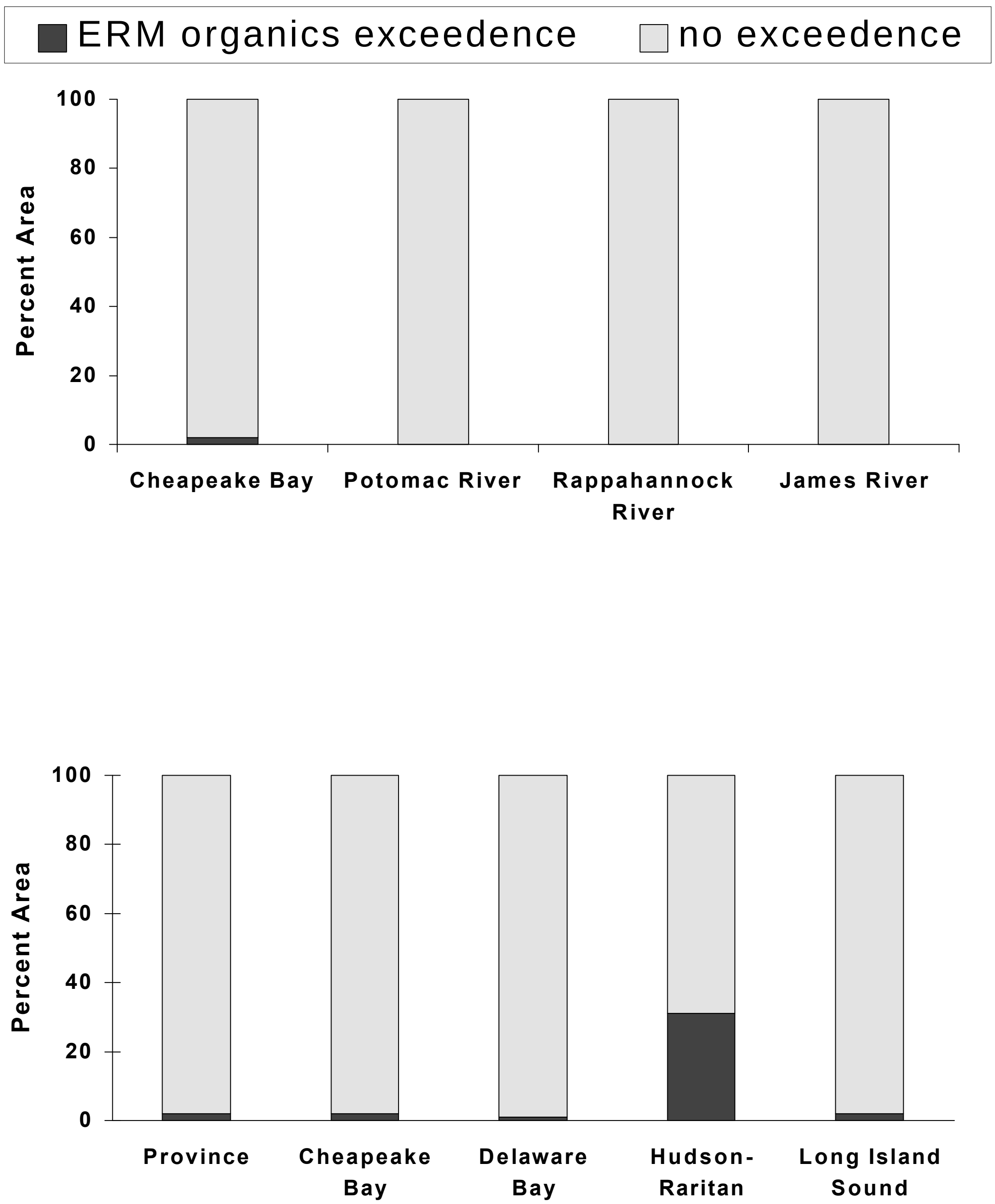

Figure 3-13. Sediment contaminant condition of bottom sediments for major estuarine systems in Virginian Province for the period 1990-93, ER-M organics exceedence.

Table 3-17. Areal Estimates for Sediment Contamination Condition (Organic Carbon-Normalized PAHs > 200 Mg/g-OC) for Major Estuarine Systems in Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	‡‡	‡‡	3	4	7	5	3	8	5	3
Potomac River	*	*	*	*	*	*	*	*	0	0‡
Rappahannock River	*	*	*	*	*	*	*	*	22	23
James River	*	*	*	*	*	*	*	*	2	17
Delaware Bay	‡‡	‡‡	5	9	0	0‡	8	10	4	5
Hudson-Raritan	‡‡	‡‡	59	38	78	14	53	37	63	18
Long Island Sound	‡‡	‡‡	4	6	0	0‡	3	5	2	3

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	‡‡	‡‡	396	480	820	543	348	913	521	388
Potomac River	*	*	*	*	*	*	*	*	0	0‡
Rappahannock River	*	*	*	*	*	*	*	*	83	84
James River	*	*	*	*	*	*	*	*	16	112
Delaware Bay	‡‡	‡‡	104	194	0	0‡	160	209	88	95
Hudson-Raritan	‡‡	‡‡	447	289	589	103	402	281	479	139
Long Island Sound	‡‡	‡‡	123	201	0	0‡	110	167	78	87

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

‡‡ Due to QA problems with the organic chemistry analyses for 1990 samples (Strobel and Valente, 1995), results for 1990 organic chemistry analyses are not presented.

the small systems bordering Long Island Sound (28±32% of the area), and in the tidal portion of Delaware River (35±39% of the area). However, because of small sample sizes for some of these estimates, uncertainties are large.

3.4.2.5 ER-L exceedence

The major estuarine system results for no ER-L exceedence (metals and organics) are summarized in Table 3-18. Remember that these results are for areas where no ER-L exceedences are observed. Almost none of the sediments in the Hudson-Raritan system (1±3%) were observed to have contaminant concentrations below ER-L levels. In other words, almost the entire Hudson-Raritan system has sediments above levels that have been observed to potentially elicit biological responses. In contrast, half or more of Chesapeake Bay (50±5%) and Delaware Bay (62±14%) were observed to have sediment concentration levels below ER-Ls. Long Island Sound exhibited 24±12% of the sediments to be below ER-L levels. Within Chesapeake and Delaware Bays, the large system class exhibited more sediment area below ER-L levels compared with the small systems and tidal rivers. In contrast, Long Island Sound results indicated more percent area with sediments below ER-L levels in small systems than in the large system class. These results for distribution of sediment contamination across the classes within the major estuarine systems are consistent with those observed for metals enrichment presented in an earlier section. The major estuarine system results indicate that the Hudson-Raritan and Long Island Sound systems are more at risk due to sediment contamination above potential biological effect levels compared to Chesapeake and Delaware Bays.

3.5 Indicator Co-occurrence

One strength of the EMAP sampling design and indicator program is the co-location of many types of measurements at each site. The synoptic nature of the data facilitates identification of associations between condition indicators and habitat and/or stressor indicators. Although these associations do not define cause and effect, they can be used both to gauge how ecological condition indicators reflect habitat condition or measures of stress and to formulate hypotheses concerning causal relationships.

3.5.1 Statistical Patterns for Benthic Impact and Habitat Condition

These analyses were conducted to determine which types of habitat (environmental) conditions (*e.g.*, silt-clay content, total organic carbon (TOC) content of the sediments, and bottom water salinity) are more likely to be associated with impacted benthic conditions. The distributions of each of the three habitat indicators are first presented as they were observed across all of the sampling sites, then as they are associated with impacted and unimpacted benthic areas.

Table 3-18. Areal Estimates for Bottom Sediment Contaminant Condition (No ER-L Exceedence) for Major Estuarine Systems in the Virginian Province. Values Are Mean Estimate and 95% Confidence Interval (C.I.). (a) Relative Area (percent). (b) Absolute Area (km^2).

(a)	1990		1991		1992		1993		1990-1993	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Chesapeake Bay	49	10	48	10	54	10	44	12	50	5
Potomac River	*	*	*	*	*	*	*	*	36	20
Rappahannock River	*	*	*	*	*	*	*	*	0	0‡
James River	*	*	*	*	*	*	*	*	23	20
Delaware Bay	92	6	50	32	53	31	52	31	62	14
Hudson-Raritan	5	13	0	0‡	0	0‡	0	0‡	1	3
Long Island Sound	32	24	45	25	31	24	28	22	24	12

(b)	1990		1991		1992		1993		1990-1993	
	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.	Area	C.I.
Chesapeake Bay	5,624	1,159	5,435	1,181	6,142	1,129	5,045	1,339	5,700	602
Potomac River	*	*	*	*	*	*	*	*	453	253
Rappahannock River	*	*	*	*	*	*	*	*	0	0‡
James River	*	*	*	*	*	*	*	*	153	138
Delaware Bay	1,887	114	1,020	654	1,100	648	1,070	648	1,269	283
Hudson-Raritan	40	101	0	0‡	0	0‡	0	0‡	10	25
Long Island Sound	1,070	803	1,505	836	1,037	803	937	736	803	401

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

Table 3-19. Areal Estimates for Association of Silt-clay Content of Sediments with Benthic Condition for the Virginian Province.

	silt-clay (%)	**entire province** (% Area)	**impacted benthic communities** (% Area)	**unimpacted benthic communities** (% Area)
Province	≤20	46	34	49
	20-80	29	24	31
	>80	26	42	20
Large Systems	≤20	52	36	56
	20-80	30	25	31
	>80	18	39	13
Small Systems	≤20	33	31	30
	20-80	28	24	36
	>80	39	44	34
Tidal Rivers	≤20	31	33	33
	20-80	19	18	22
	>80	50	49	44

3.5.1.1 Silt-clay content of sediments

The distributions of silt-clay content in the sediments of the Virginian Province are presented in Table 3-19. Muds are sediments with silt-clay content > 80%, and sands have silt-clay content ≤ 20%. Almost half of the Virginian Province (46±4%) has sandy sediments. The large systems are primarily sandy environments (52±5%), the small systems have almost equal distribution across the three silt-clay categories, and tidal rivers are mostly mud (50±16%).

The associations for silt-clay content with impacted and unimpacted benthos across the Virginian Province are presented in Table 3-19. Areal analyses indicate that impacted benthic communities tend to be associated with muddy sediments, irrespective of resource class (large, 39%; small, 44%; and tidal, 49%). Unimpacted benthic communities are associated more closely with sandy environments in large systems, while in large tidal rivers and small systems unimpacted benthos are distributed evenly across silt-clay categories.

3.5.1.2 Total organic carbon content of sediments

The distributions of TOC content in the sediments across the Virginian Province

Table 3-20. Areal Estimates for Association of Total Organic Carbon (TOC) Content of Sediments with Benthic Condition for the Virginian Province.

	TOC (%)	entire province (% Area)	impacted benthic communities (% Area)	unimpacted benthic communities (% Area)
Province	≤1	54	29	51
	1 to 3	39	51	46
	> 3	7	20	4
Large Systems	≤1	63	27	60
	1 to 3	31	50	37
	> 3	6	23	3
Small Systems	≤1	37	32	36
	1 to 3	54	52	58
	> 3	10	16	6
Tidal Rivers	≤1	34	33	17
	1 to 3	58	51	74
	> 3	8	16	9

are presented in Table 3-20. Fifty-four percent (±4%) of the Virginian Province sediments contain low TOC (TOC ≤ 1%). The large estuarine systems are primarily low TOC environments (63±5%). Small estuarine systems and tidal rivers are moderate TOC environments; 54±4% and 58±5% of the area in these systems are in the 1-3% TOC range, respectively.

Associations for TOC content with impacted and unimpacted benthos across the Virginian Province are presented in Table 3-20. For the province as a whole and all of its component resource classes, impacted benthic communities tend to be associated with the moderate TOC content sediments (1-3% range). This contrasts with unimpacted benthos across the entire province and for large systems where the association is with low organic carbon environments, and for small and tidal river systems where the association is with moderate organic environments. This pattern for unimpacted benthic communities mimics the TOC distribution across the province (Table 3-20).

3.5.1.3 Bottom water salinity

The areal estimates of bottom water salinity for the Virginian Province is

Table 3-21. Areal Estimates for Association of Bottom Water Salinity with Benthic Condition for the Virginian Province.

	salinity (ppt)	**entire province** (% Area)	**impacted benthic communities** (% Area)	**unimpacted benthic communities** (% Area)
Province	≤5	6	11	5
	5 to 18	28	28	28
	> 18	66	61	67
Large Systems	≤5	1	0	1
	5 to 18	20	24	22
	> 18	79	76	77
Small Systems	≤5	9	16	7
	5 to 18	38	21	36
	> 18	54	63	57
Tidal Rivers	≤5	37	42	38
	5 to 18	56	58	56
	> 18	7	0	6

presented in Table 3-21. Almost two-thirds of the Virginian Province bottom waters (66±3%) are polyhaline (salinity > 18 o/oo). The bottom waters of the large systems are primarily polyhaline environments (79±14%). The areal distribution of bottom water salinity in small systems closely follow the distribution of bottom water salinity represented by the entire province. The tidal rivers have mostly low salinity bottom water (37±10% and 56±4% for oligohaline (≤ 5 o/oo) and mesohaline (5-18 o/oo), respectively).

The association of bottom water salinity with impacted and unimpacted benthos across the Virginian Province are presented in Table 3-21. Impacted benthic communities tend to associate with polyhaline bottom waters in large and small systems in proportion to salinity distribution in these systems. Similarly, impacted benthos in tidal rivers are proportionately associated with low and moderate salinity environments (< 5 and 5-18 o/oo, respectively). Unimpacted benthos exhibit similar distribution in proportion to salinity occurrence. The association analysis indicates that there is no difference in associations for unimpacted and for impacted benthic communities. This confirms that the benthic index as used for this analysis is independent of salinity; in other words, the salinity normalization for the benthic index metrics was effective (see Appendix B).

3.5.1.4 Summary for habitat associations

The Virginian Province estuarine habitat is predominantly polyhaline and is composed of mostly sandy, low TOC content sediments. Impacted benthic communities are most frequently associated with muddy, moderate TOC content sediments and polyhaline bottom waters. The exception is that impacted tidal rivers are associated with the lower salinity (oligo- and mesohaline) bottom waters.

3.5.2 Statistical Pattern for Benthic Impact and Stressors

Three types of analyses were conducted to link impacted benthic communities with stressors. The first examines associations among the individual stressors that co-occur with the impacted benthos. The individual stressors include dissolved oxygen, sediment toxicity, and sediment contaminants exceeding ER-M values. Metals enrichment was not included in the analysis; there was no clear association of this stressor with impacted benthic condition as compared to unimpacted benthic condition, as indicated in Table 3-22. The second type of analysis examines associations with each stressor separately, with combination of any two stressors, and finally associations between impacted benthic communities and all three stressors. This second approach provides a description of the potential severity of stressor associations based upon the number of stressors that co-occur. The third approach is a step-wise linear regression analysis of the benthic index (condition of benthic communities) with the major habitat and stressor variables.

Table 3-22. Association of Benthic Condition with Enriched Metal Concentrations in Sediments for the Virginian Province.*

	percent area with any enriched metal concentrations		
	entire province	**impacted benthic communities**	**unimpacted benthic communities**
Province	49	56*	46
Large Systems	42	57	36
Small Systems	64	56	66
Tidal Rivers	69	56	76

* Table is read as follows: fifty-six percent of the area of the Virginian Province with impacted benthic communities also experienced enriched metal concentrations. The remaining 44% of the area with impacted benthic communities did not have enriched metal concentrations.

3.5.2.1 Association with low dissolved oxygen

Association between benthic communities and bottom water dissolved oxygen conditions are presented in Table 3-23, using both 2 and 5 ppm as criteria for the association. Twenty-two percent of the impacted benthic communities in the province

Table 3-23. Association of Benthic Condition with Bottom Dissolved Oxygen for the Virginian Province.*

	percent area with dissolved oxygen ≤ 2 ppm		
	entire province	**impacted benthic communities**	**unimpacted benthic communities**
Province	5	22*	0
Large Systems	5	30	0
Small Systems	1	3	0
Tidal Rivers	14	33	0

	percent area with dissolved oxygen ≤ 5 ppm		
	entire province	**impacted benthic communities**	**unimpacted benthic communities**
Province	24	44	19
Large Systems	27	58	22
Small Systems	17	19	14
Tidal Rivers	18	49	0

* Table is read as follows: twenty-two percent of the area of the Virginian Province with impacted benthic communities also experienced bottom dissolved oxygen ≤ 2 ppm. The remaining 78% of the area with impacted benthic communities had bottom dissolved oxygen > 2 ppm.

are associated with DO ≤ 2 ppm. In the large and tidal river systems, however, 30-33% of impacted areas are associated with DO ≤ 2 ppm. Association with low DO in the small estuarine systems was low, 3%. None of the unimpacted benthic communities are associated with DO ≤ 2 ppm in any resource class.

When 5 ppm is used as the criterion for the association, almost half of the impacted benthic communities for the province (44%) are associated with DO ≤ 5 ppm. Fifty-eight percent of the impacted benthic communities in the large system are associated with DO ≤ 5 ppm, 19% of the small systems, and 49% of the tidal rivers. Approximately one-fifth of the unimpacted benthic communities in the province and large estuarine systems are associated with DO ≤ 5 ppm. Only 14% of the unimpacted benthic communities in small estuarine systems are associated with DO ≤ 5 ppm. None of the unimpacted tidal river systems are associated with DO ≤ 5 ppm. These patterns for unimpacted benthos are consistent with the distribution of bottom DO ≤ 5 ppm across the entire province, with the exception of the large tidal rivers.

3.5.2.2 Association with sediment toxicity

Associations between benthic communities and sediment toxicity are presented in Table 3-24, using two criteria: survival ≤ 60% and ≤ 80%. Associations using survival

Table 3-24. Areal Estimates for Association of Benthic Condition with Sediment Toxicity for the Virginian Province.*

	percent area with amphipod survival ≤ 60%		
	entire province	**impacted benthic communities**	**unimpacted benthic communities**
Province	1	3*	1
Large Systems	1	3	1
Small Systems	3	4	3
Tidal Rivers	0	2	0

	percent area with amphipod survival ≤ 80%		
	entire province	**impacted benthic communities**	**unimpacted benthic communities**
Province	9	16	7
Large Systems	9	12	8
Small Systems	12	25	6
Tidal Rivers	4	7	2

* Table is read as follows: three percent of the area of the Virginian Province with impacted benthic communities also experienced amphipod survival ≤ 60%. The remaining 97% of the area with impacted benthic communities had amphipod survival > 60%.

≤ 60% as the criterion are minimal, with 3% for the province and ranging from 2% in tidal river systems to 4% in small systems. There are no clear distinctions between the associations for impacted and unimpacted benthic communities using survival ≤ 60% as the criterion for sediment toxicity.

Using the criterion of survival ≤ 80%, 16% of the impacted benthic communities in the province is associated with sediment toxicity, 25% in small systems, 7% in tidal rivers, and 12% in large systems. Sediment toxicity (≤ 80% survival) co-occurs with unimpacted benthos for 7% of the province, 6-8%of small and large estuarine systems, and 2% in tidal rivers.

3.5.2.3 Association with ER-M exceedence

Association between benthic community condition and sediment contaminant concentrations (ER-M exceedence for any metal or any organic compound) are presented in Table 3-25. For ER-M metal exceedence, one-fifth of the impacted benthic communities in small systems (20%) are associated with any sediment concentration exceeding a metal ER-M value. The remainder of associations with ER-M exceedence for metals are small. Values for large systems and tidal rivers are similar across the entire province and for the unimpacted benthic communities.

Table 3-25. Areal Estimates for Association of Benthic Condition with ER-M Exceedence in Sediments for the Virginian Province.*

	percent area with any ER-M metal exceedence		
	entire province	**impacted benthic communities**	**unimpacted benthic communities**
Province	5	7*	3
Large Systems	3	0	3
Small Systems	9	20	2
Tidal Rivers	7	3	6

	percent area with any ER-M organics exceedence		
	entire province	**impacted benthic communities**	**unimpacted benthic communities**
Province	3	5	3
Large Systems	2	0	3
Small Systems	2	5	1
Tidal Rivers	10	14	9

* Table is read as follows: seven percent of the area of the Virginian Province with impacted benthic communities also experienced ER-M metal exceedence. The remaining 93% of the area with impacted benthic communities did not have any ER-M metal exceedence

There is little association between the condition of benthic communities and ER-M exceedence for organic compounds. While 14% of the tidal river area for impacted benthic communities are associated with ER-M organics exceedence, this is not much different from the values for the entire province and for the unimpacted benthos.

3.5.2.4 Association with multiple stressors

Analyses of associations between multiple stressors and impacted benthic communities are summarized in Table 3-26 for the entire province. Note that because of the restriction that valid data had to exist for all of the stressors at the sampling site for associations to be conducted, slight differences exist between tables in this section and Tables 3-23 to 3-25. Twenty-five percent of the Virginian Province has impacted benthos, of which 39% co-occurs with low dissolved oxygen only (DO $\leq$ 5 ppm), 6% co-occurs with sediment toxicity only, 3% co-occurs with any ER-M exceedence only, 4% with any two stressor indicators, 1% with all three, and 46% is not associated with any of the stressors. These data indicate that low dissolved oxygen, sediment toxicity, and ER-M exceedence together may account for 54% of the impacted benthic area. The remaining 46% of impacted benthos is not associated with any of the three stressor indicators. Similar analyses using dissolved oxygen $\leq$ 2 ppm indicates that 60% of the impacted benthic area is not associated with any of the three stressors (Table 3-26).

Table 3-26. Co-occurrence of Stressors with Province-wide Impacted Benthic Communities (percent of impacted area). Entries are for only one stressor, any two stressors, or all three stressors. (a) Dissolved Oxygen Criterion of 5 ppm. (b) Dissolved Oxygen Criterion of 2 ppm.

(a)

only **do**	only **tox**	only **erm**	**do+tox**	**do+erm**	**tox+erm**	**do+tox+erm**	**other**
39	6	3	3	< 1	< 1	1	46

do: DO $\leq$ 5 ppm
tox: survival $\leq$ 80%
erm: exceed any ERM

(b)

only **do**	only **tox**	only **erm**	**do+tox**	**do+erm**	**tox+erm**	**do+tox+erm**	**other**
25	9	4	< 1	0	2	0	60

do: DO $\leq$ 2 ppm
tox: survival $\leq$ 80%
erm: exceed any ERM

Nineteen percent of the total large estuarine area has impacted benthos, 56% of which co-occurs with low dissolved oxygen only (DO $\leq$ 5 ppm for the criteria), 7% co-occurs with sediment toxicity only, no co-occurrence with any ER-M exceedence only, 4% co-occurs with two of the stressors, no co-occurrence with all three of the stressors, and 33% is associated with none of the three stressors (Table 3-27). These data suggest that low dissolved oxygen is the principal stressor of concern in large estuarine systems and that sediment toxicity and ER-M exceedence are of less importance.

Small estuarine systems present a somewhat different picture with 37% of their total area exhibiting impacted benthos, of which 6% co-occurs with low dissolved oxygen (using DO $\leq$ 5 ppm for the criteria), 5% co-occurs with sediment toxicity, 11% with any ER-M exceedence, 5% co-occurs with any two, 4% with all three of the stressors, and 69% is not associated with any of the three stressors (Table 3-27). These data suggest that sediment toxicity and ER-M exceedence are the principal stressors of concern in small estuarine systems, but there is a large unexplained portion.

Tidal rivers have the highest percent area (38%) of impacted benthos of the three resource classes of which 45% co-occurs with low dissolved oxygen and 6% co-occurs with sediment toxicity. Unlike the small estuarine systems, none of the impacted benthic area co-occurs with any ER-M exceedence, 2% co-occurs with any two of the

Table 3-27. Co-occurrence of Stressors with Resource Class Impacted Benthic Areas (percent of impacted area). Dissolved Oxygen Criterion of 5 ppm.

	only **do**	only **tox**	only **erm**	**do+tox**	**do+erm**	**tox+erm**	**do+tox+erm**	**other**
Large Systems	56	7	0	4	0	0	0	33
Small Systems	6	5	11	4	< 1	< 1	4	69
Tidal Rivers	45	6	0	0	2	0	< 1	46

do: DO ≤ 5 ppm
tox: survival ≤ 80%
erm: exceed any ERM

stressors, and < 1% co-occurs with all three (Table 3-27). The remaining 46% of the impacted benthic area in the tidal rivers is not associated with any of the three stressor indicators. Low dissolved oxygen is of major concern in the tidal river systems; sediment toxicity and ER-M exceedence are less important.

Similar associations occur for the three resource classes using the dissolved oxygen criteria of ≤ 2 ppm (Table 3-28). The major exception is for small systems with no co-occurrence of benthic impacts and DO ≤ 2 ppm. This implies that very low DO is of limited concern in small systems, and that a larger portion of the benthic impact is associated with toxic contaminants. However, there is a large portion unexplained by the three measured stressors.

3.5.2.5 Regression of the benthic index with individual variables

A step-wise linear regression analysis was conducted for the benthic index against the major habitat and stressor variables for the all of Virginian Province sampling sites (Table 3-29). These results indicate that 32% of the variance in the benthic index can be explained with the following variables: TOC content of sediments,

Table 3-28. Co-occurrance of Stressors with Resource Class Impacted Benthic Areas (percent of impacted area). Dissolved Oxygen Criterion of 2 ppm.

	only **do**	only **tox**	only **erm**	**do+tox**	**do+erm**	**tox+erm**	**do+tox+erm**	**other**
Large Systems	37	11	0	0	0	0	0	52
Small Systems	0	7	12	1	0	5	0	75
Tidal Rivers	32	6	2	0	0	< 1	0	59

do: DO ≤ 2 ppm
tox: survival ≤ 80%
erm: exceed any ERM

Table 3-29. Results of Stepwise Regression for Benthic Index Against Habitat and Stressor Indicators.

Grouping	Significant Variables	Model r^2
Province	TOC, DOSATB, BDO, NAPH, PHENANTH, TPCB, TDDT, STRATI, BSAL, CR, CD	0.321
Large Systems	CR, NAPH, TOC	0.256
Small Systems	AS, CLARITY, TDDT, CD	0.401
Tidal Rivers	DOSATB, BDO, TPCB	0.269

dissolved oxygen saturation at the bottom, bottom dissolved oxygen, naphthalene, phenanthene, total PCBs, total DDT, stratification, bottom water salinity, chromium, and cadmium. Similar variance could be accounted for in large systems, small systems, and tidal rivers (25%, 40%, and 27%, respectively); however, these variances were explained with three or four variables each. These variables were chromium, naphthalene, and TOC content of sediments in large systems, arsenic, water clarity, total DDT, and cadmium in small systems, and dissolved oxygen saturation at the bottom, bottom dissolved oxygen, and total PCBs in tidal rivers. Conducting a principal component analysis on individual sediment contaminant concentrations, and then using the results to represent sediment chemistry in the regression, did not explain additional variance.

SECTION 4: DISCUSSION

The prior section presented results from assessments of the ecological condition indicators and of associations between stressors and benthic condition. These results are discussed here in the context of answers to a series of environmental management questions. The format is designed to aid managers in the interpretation of the results presented in the prior section. The general questions were developed from intensive interactions that have transpired between the EMAP-Estuaries staff and environmental managers across the Virginian Province. These interactions included workshops (USEPA 1991; Cochran 1991; Queen *et al.* 1992) and individual presentations for environmental management groups, including each of the individual EPA National Estuary Programs across the Virginian Province. These questions are:

1. Is there a problem in the estuarine waters of the Virginian Province?

2. If there is a problem in the estuarine waters of the Virginian Province, what is the magnitude, extent, and distribution of the problem?

3. What factors are associated with the observed problems in the estuarine waters of the Virginian Province?

4. Are the observed problems in the estuarine waters of the Virginian Province consistent with existing knowledge?

5. What are the uncertainties associated with the conclusions?

6. How effective was the EMAP Virginian Province Demonstration Project in meeting the program objectives?

4.1 Is there a problem?

The information presented in the prior chapter indicates that the estuarine waters of the Virginian Province are having problems. Because no index of overall estuarine quality exists, the suite of indicators reported identify problems with different components of the estuarine systems. A summary of these problems follows.

A benthic index was developed and refined to determine the condition of the benthic communities in the estuarine waters of the Virginian Province. Impacted benthic communities were observed in 25±3% of the province area for the four-year period (1990-93). Of the three resource classes, the large estuarine systems had the smallest percent area with impacted benthic communities, 19±4% for the four-year period. The percent area with impacted benthic resources was markedly larger in the small estuarine systems and tidal rivers, 37±6% and 38±14%, respectively, for the four-year period.

The percent area of bottom waters in the Virginian Province with low dissolved oxygen conditions (DO ≤ 2 ppm) was 5±2% for the four-year period (1990-93). Small estuarine systems had only 1±1%, large systems had 5±2%, and tidal rivers 14±6%. These data suggest that the tidal rivers are most at risk from low dissolved oxygen. Using moderate hypoxia as the criterion (DO ≤ 5 ppm), impacted conditions were observed in 24±3% of the province area, 17±5% of the small systems, 27±4% of the large systems, and 18±7% of the tidal rivers. These analyses suggest that large estuarine systems are potentially at risk from moderate reductions in dissolved oxygen.

The percent area of the Virginian Province sediments having moderate sediment toxicity (survival ≤ 80%) was 9±2% for the four-year period (1990-93). Examination of individual resource classes showed that the percent area exhibiting sediment toxicity was 4±4% of the tidal rivers, 9±3% of the large systems, and 12±6% of the small systems. These analyses indicate that small estuarine systems are at greatest risk from toxic sediments. Severe toxicity (survival ≤ 60%) occurred in only 1% of the estuarine sediments of the province and was distributed across resource classes.

Using ER-M exceedence as a measure of sediment contamination, 5±1% of the area of the province during the four year sampling interval had observed exceedence in at least one sediment concentration for metals for which ER-Ms exist. Sediments in large estuarine systems exhibited exceedence in 3±3% of the area. Seven percent of the area (±4%) in tidal rivers contained sediments with ER-M exceedence. Small estuarine systems exhibited the highest proportion of area, 9±4%, with ER-M exceedence. These ER-M metal exceedence suggest that small estuarine systems are at greater risk than are large or tidal river systems.

Sediment concentrations for organic contaminants indicated that 3±1% of the area of the Virginian Province exhibited exceedence of at least one ER-M value. The proportional extent of organic contamination was low in large and small estuarine systems (2%) and higher in tidal rivers (10±2%). Tidal rivers appear to be at the greatest risk due to organic contamination.

Using ER-L values for determining which observed sediment contaminant concentrations are below levels for which biological effects may be of concern, half of the entire province (50±4%) had sediment contaminant concentrations below ER-L levels. The large estuarine system class exhibited the largest areal extent of sediments below ER-Ls, 58±5%, with small systems and tidal rivers exhibiting 35±6% and 25±12%, respectively. These results indicate that sediments of the small systems and tidal rivers are more at risk to sediment contaminant levels that have been observed to potentially elicit biological responses.

Overall, 52±5% of the estuarine waters of the Virginian Province were in good condition, *i.e.,* these waters exhibited unimpacted benthic conditions and bottom dissolved oxygen > 5 ppm and sediment toxicity acute survival > 80% and no ER-M exceedence for sediment contaminants. Small estuarine systems had 55±14% of the area in good condition, tidal rivers had 52±13%, and large systems 51±5%.

4.2 What is the magnitude, extent, and distribution of the problem?

Chesapeake Bay, which exhibited benthic community impacts in 23±5% of the area, accounted for 45% of the impacted benthic area within the Virginian Province. The Potomac and Rappahannock Rivers have identical percent impacted areas, 44±22% and 44±33%, respectively, with the James River having a slightly smaller percent area of benthic impact (19±23%). Delaware Bay and Long Island Sound exhibited benthic impacts over 24±12% and 28±11% of their areas, respectively. The percent areal extent of benthic impact in the Hudson-Raritan system, however, was the greatest of the major estuarine systems examined (72±8%). Together, these four estuarine systems account for 79% of the impacted benthic area within the Virginian Province.

Of the four major estuarine systems, Chesapeake Bay was the most impacted (10±3%) from low dissolved oxygen (DO ≤ 2 ppm), with the Potomac and Rappahannock Rivers particularly impacted (24±12% and 15±11%, respectively). Using moderate hypoxia as the criterion (DO ≤ 5 ppm), approximately 31% of Chesapeake Bay and 48% of Long Island Sound areas have moderate hypoxia. The moderate hypoxia in Long Island Sound was in the open water area of the western Sound.

Moderate sediment toxicity (survival ≤ 80%) was observed in 15±14% of the Hudson-Raritan system, 13±7% of Long Island Sound, 6±3% of Chesapeake Bay, and 2±2% of Delaware Bay. Moderate toxicity was observed across all of the estuarine classes of these estuarine systems, except Delaware Bay, where toxicity was only observed in the Delaware River. Severe toxicity (survival ≤ 60%) was observed in Delaware Bay (1±2%), Hudson-Raritan system (7±10%), and Long Island Sound (5±4%). This toxicity was found primarily in small systems in the Hudson-Raritan system and Long Island Sound, and only in the tidal river portion of Delaware Bay.

Using ER-M exceedence as a measure of sediment contamination, the Hudson-Raritan Estuary is the most contaminated major system, exhibiting 44±14% of the area with sediments exceeding at least one ER-M value. Most of this contamination was due to mercury, lead, and silver in the tidal portion of the Hudson River (30% of the area). The least contaminated system was Delaware Bay where only 1±3% of the area contained sediments exceeding ER-M values. Chesapeake Bay and Long Island Sound were intermediate in level of contamination with percent area of exceedence of 5±2%

and 4±4%, respectively. The exceedences in Long Island Sound were predominately found in small systems bordering the open waters of the Sound.

Sediment concentrations for organic contaminants indicated that Delaware Bay system was the least contaminated of the four major systems. Less than 1% of its area contained sediments exceeding organic ER-M values, and those sediments were restricted to the Delaware River. A similarly low level of contamination was found in Chesapeake Bay and Long Island Sound; 3-4% of their areas contained sediments with organics exceeding the ER-M values. The greatest level of organic contamination was found in the Hudson-Raritan (mostly PCBs): 33±7% of the area of this system contained sediments with organic concentrations that exceeded at least one ER-M value. Sixty-six percent (±18%) of the area in the tidal portion of the Hudson River, and 11±5% of the area in the small systems exhibited ER-M exceedence.

Almost none of the sediments in the Hudson-Raritan system (1±3%) were observed to have contaminant concentrations below ER-L levels. In other words, almost the entire Hudson-Raritan system has sediments above levels that have been observed to potentially elicit biological responses. In contrast, half or more of Chesapeake Bay (50±5%) and Delaware Bay (62±14%) were observed to have sediment concentration levels below ER-Ls. Long Island Sound exhibited 24±12% of the sediments to be below ER-L levels. Within Chesapeake and Delaware Bays, the large system class exhibited more sediment area below ER-L levels compared with the small systems and tidal rivers. In contrast, Long Island Sound results indicated more areal extent of sediments below ER-L levels in small systems than the large system class.

4.3 What factors are associated with the observed problems?

Analysis of associations was conducted between the benthic community index and both stressor and habitat indicators to provide possible explanations for the observed condition of benthic resources. Analysis of benthic communities and habitat indicators show that impacted benthic communities tend to be associated with muddy (> 80% silt-clay), moderate TOC content (1-3%) sediments, and polyhaline bottom waters (> 18 o/oo). In contrast, unimpacted benthic communities tend to associate with sandy (< 20% silt-clay), low TOC content (< 1%) sediments, and with polyhaline bottom waters (> 18 o/oo).

Association analysis of benthic communities with stressors for the entire province indicates that moderate hypoxia (DO ≤ 5 ppm), sediment toxicity, and sediment contamination (ER-M exceedence) together account for 54% of the impacted benthic area. The remaining 46% of impacted benthos is not associated with any of the three stressor indicators. Low dissolved oxygen is the principal stressor of concern in large systems and tidal rivers, being associated with 58% and 49% of the impacted benthos, respectively. However, in small estuarine systems, sediment toxicity and ER-M exceedence are the principal stressors of concern, accounting for approximately 26% of the observed impacted benthos. Similar analysis with low dissolved oxygen (< 2.0 ppm) indicates that 60% of the entire province was not associated with this suite of stressors.

4.4 Are the observed problems consistent with existing knowledge?

The only province-wide consistent data set that could be used for comparison with the EMAP Virginian Province results is the National Oceanographic and Atmospheric Administration (NOAA) National Status and Trends (NS&T) Program for Marine Environmental Quality (O'Connor, 1990; NOAA, 1992). NS&T collected sediment samples at least one time at the 32 mussel watch sites in the Virginian Province, and analyzed for the same suite of contaminants as was done for the EMAP Virginian Province (Table 3-9). Restrictions on comparing EMAP data with the NS&T data include (1) most of the sediments were collected prior to 1990 (but since 1986) and (2) sampling sites were restricted to habitats suitable for oysters and mussels (shallow water, salinity greater than 10 o/oo). Daskalakis and O'Connor (1994) report on an inventory of coastal U.S. sediment contamination, compiled from various electronic information systems, including NS&T and EMAP (1990-91 data from Virginian Province and 1991-92 data from Louisianian Province). They observe that the EMAP probability-based design produces a data set that contains the lowest proportion of sites with "High" concentration for any chemical (defined as exceeding one standard deviation above the mean of the NS&T sites), compared with the other data sets inventoried. They reported that the frequency of "Highs" for the NS&T sites exceed those for the EMAP sites because there is an urban bias in the location of NS&T sites (Cantillo and O'Connor, 1992). Daskalakis and O'Connor (1994) further observe that all data sets other than EMAP contain sites preselected (biased) for their likelihood to have elevated chemical concentrations. They conclude that extremely high sites were located near large cities, suggesting anthropogenic sources for the contaminants, and that smaller water bodies with high human activity and high mean residence times bear the majority of the pollution. These conclusions are consistent with the EMAP results that the concentrations of ER-M exceedence occur around population centers (Figure 3-10 and 3-11) and that small systems and tidal rivers are at higher risk from chemical contamination (Section 3.4.1).

Benthic community condition is available for comparison in Chesapeake Bay system (USEPA, 1995a; Ranasinghe *et al.*, 1994), where the benthic information was analyzed through use of habitat restoration goals (Ranasinghe *et al.*, 1993; Weisberg *et al.*, 1997). This approach specifies habitat-specific measures that describe characteristics of benthic assemblages expected to occur at sites with little evidence of environmental stress or disturbance. It has a similar goal to the EMAP benthic index, which is to aggregate the diversity of benthic information into a single quantifiable value.

Figure 4-1 is reproduced from the 1995 State of Chesapeake Bay Report (USEPA, 1995a) and is compared with Chesapeake Bay portion of Figure 3-2. The two approaches for determining benthic condition depict the same general pattern for benthic community impact. More detailed comparison of these approaches is currently being undertaken.

The distribution of sediment contamination is available for three of the major estuarine systems examined in this report. Figure 4-2 presents results of the analysis on toxic stress for Chesapeake Bay (USEPA, 1995a), which could be compared with Chesapeake Bay portion of Figures 3-10 and 3-11. The two programs show a similar distribution of potential impact due to pollutants. Results for Delaware Bay are available from the Comprehensive Conservation and Management Plan (USEPA, 1996). This report indicates that the highest concentrations of toxic substances occur in the urbanized area along the Delaware River. Elevated metals in bottom sediments are associated with fine, organic-rich particles, particularly near municipalities and in the central area of the Bay. Lead, zinc, cadmium, pesticides and some of the PAHs exceed the Long and Morgan ER-M values. Acute sediment toxicity was concentrated along industrialized portions of the Delaware River. These observations are consistent with the results for the Delaware Bay discussed in Section 3.4.2 and depicted in Figures 3-8, 3-10, and 3-11. The Long Island Sound Study reported that "most of the Sound's sediments do not exhibit contamination levels of concern, problems have been documented in some areas of the western Sound and in several, mostly urbanized, harbors, rivers, and embayments" (USEPA, 1994b). Sediment contamination in Long Island Sound (LIS) as observed by NOAA NS&T was summarized by Robertson *et al.* (1991). They indicate that the NS&T sites in LIS (located around the perimeter of the Sound) tend to have relatively high concentrations for both metals and organic contaminants in sediments when compared to the reference set of NS&T sites. However, in a national comparison, two areas (near New York City and Boston) clearly exhibited highly ranked contaminant concentrations more frequently than did LIS. These reported observations on sediment contamination in LIS are consistent with the EMAP results (Section 3.4.2) indicating potential toxic impact localized in the western end of the Sound and in the highly urbanized small systems around the Sound, and that LIS ranks below the Hudson-Raritan system in contaminant impact.

Information is available for dissolved oxygen (DO) conditions in Chesapeake Bay (Ranasinghe *et al.*, 1994) and Long Island Sound (USEPA, 1994b). Results from 1995 State of Chesapeake Bay Report (USEPA, 1995a) are presented in Figure 4-3, and are compared with Chesapeake Bay portion of Figure 3-5. The results from the two programs indicate similar patterns for DO stress in the Bay. Chesapeake Bay Program DO data were collected from routine cruises conducted twenty times per year (monthly from November to February and twice monthly from March to October) at approximately 140 stations (Heasly *et al.*, 1989), while the EMAP data resulted from 150 probability-based sampling sites, each sampled once during the 1990-93 summer index period.

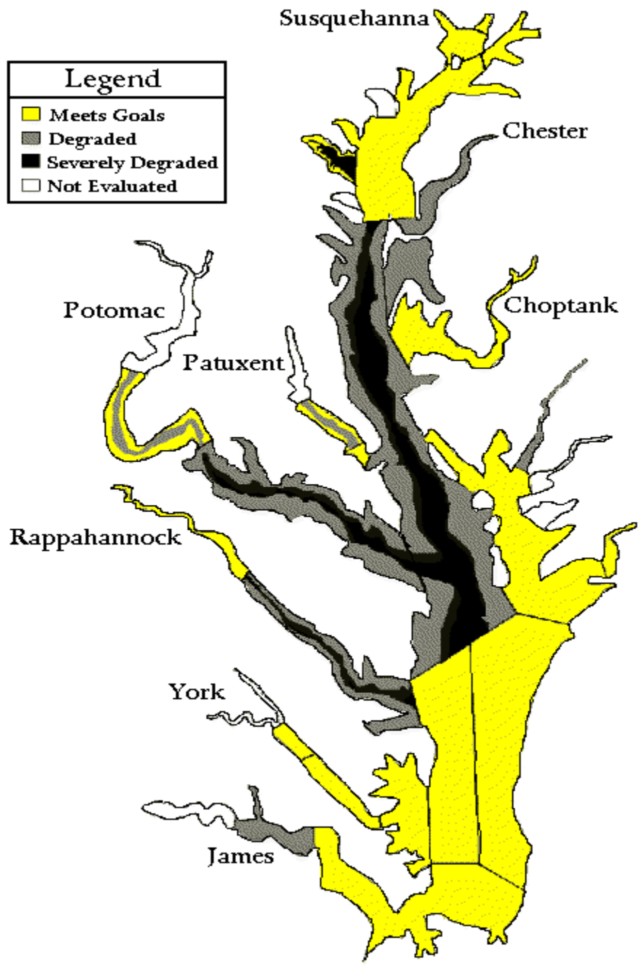

Figure 4-1. Benthic community condition in Chesapeake Bay as reported by Chesapeake Bay Program (from USEPA, 1995a). Ratings are based on average “Restoration Goals Index” scores for summer benthic community sample taken between 1984 and 1990 (refer to USEPA (1995a) for details).

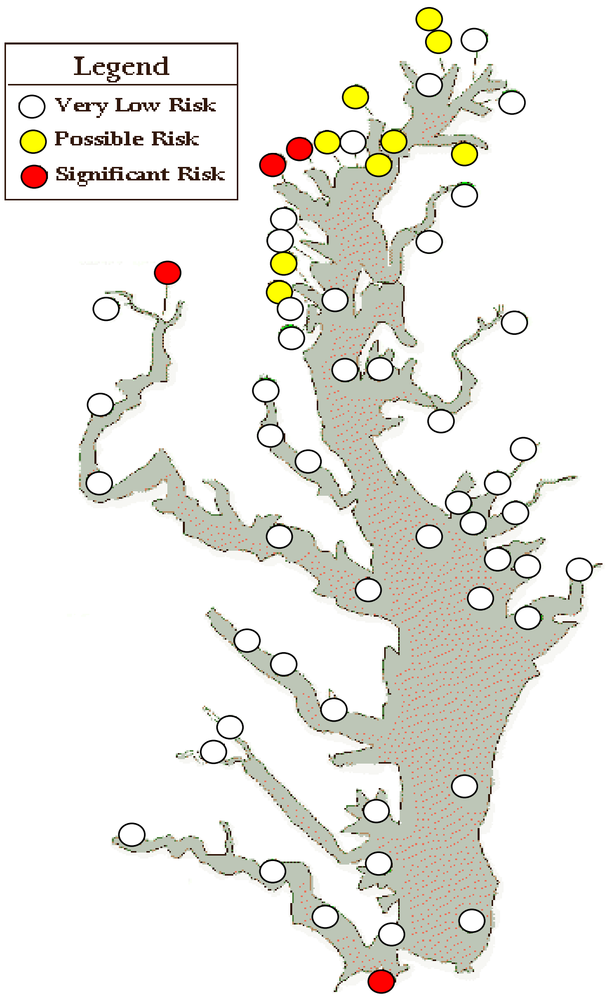

Figure 4-2. Sediment contamination and risk to aquatic life in Chesapeake Bay as reported by Chesapeake Bay Program (from USEPA, 1995a). Ranking of risk to aquatic biota from sediment contamination was based on comparisons of the average sediment concentrations of seven trace metals and eight PAHs to sediment quality guidelines (refer to USEPA (1995a) for details).

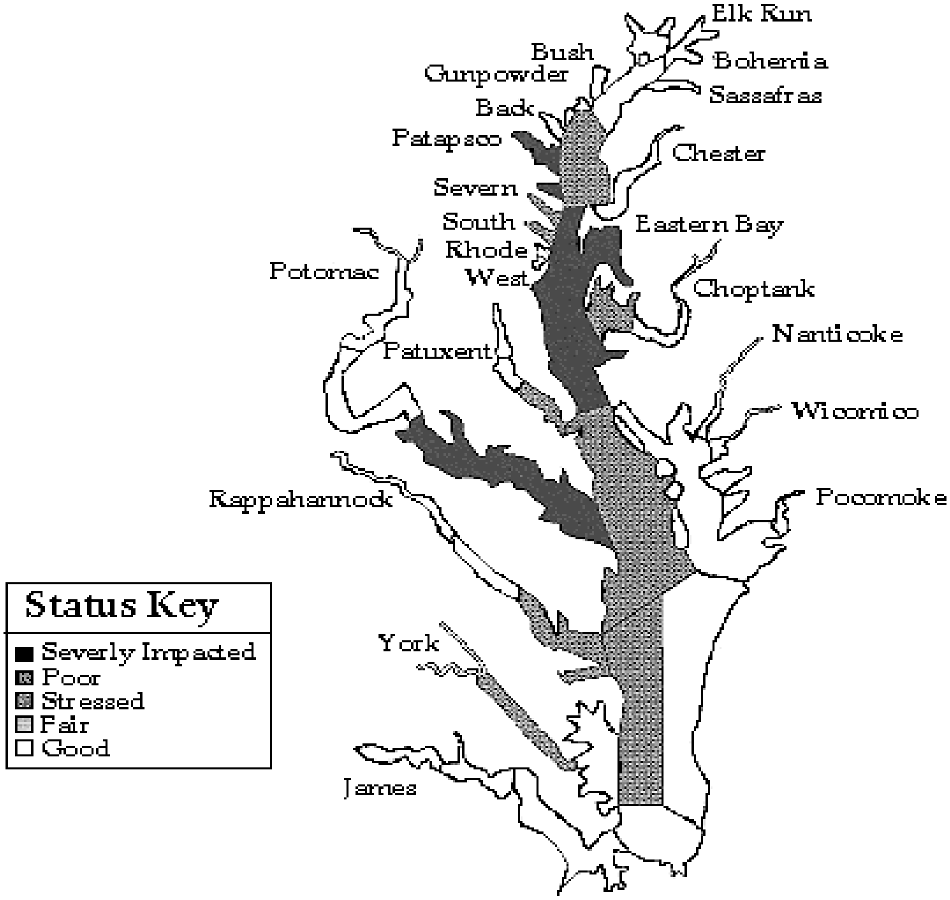

Figure 4-3. Bottom dissolved oxygen in Chesapeake Bay as reported by Chesapeake Bay Program (from USEPA, 1995a). Dissolved oxygen levels in bottom waters were evaluated relative to target concentrations required to support the growth, reproduction, survival of the Bay's fish, shellfish, and bottom dwelling organisms. A three level scale was used with "Good" areas having among the highest bottom dissolved oxygen concentrations and "Severely Impacted" areas having among the lowest dissolved oxygen concentrations (refer to USEPA (1995a) for details).

The Long Island Sound Study (LISS) reported that moderate DO stress ($\leq$ 5 ppm) was observed during the summers of 1987-93 in about one-half to two-thirds of the Sound. The distribution of low DO area encompassed the western portion of the Sound to approximately New Haven (USEPA, 1994b). This is consistent with the EMAP results for exposure to moderate DO stress areal extent (48$\pm$12% of the area during 1990-93) and distribution (Figure 3-5). The LISS reported numerous occurrences of low DO stress ($\leq$ 2 ppm) compared with only one observation at one station by EMAP. This difference can be explained by the small observed percent area with DO $\leq$ 2 ppm in the extreme western end of the Sound (approximately 100 km^2 or 3% of the Sound), and the spatial density of EMAP probability-based sampling sites. Spatial enhancement of the probability-based sampling sites in the western Sound would be necessary for EMAP to better resolve the extent of low DO stress in LIS.

These comparisons of results from the EMAP Virginian Province Four-Year Demonstration Project with published results from more intensively sampled programs indicate that the EMAP approach can provide comparable accuracy for description of the extent and distribution of estuarine impact. The EMAP approach was able to quantify the magnitude of impacted conditions with estimates of uncertainty, which is something other monitoring programs were not always capable of accomplishing.

4.5 What are the uncertainties?

All of the percent area values reported here have included an estimate of the uncertainty of the value. In Section 3, reported values for individual years and the entire four-year period were presented for the entire Province, individual estuarine classes, and the four major estuarine system. Values for the three major tidal river systems within Chesapeake Bay were only presented for the entire four-year period. Values for the individual classes within the major estuarine systems and for the individual years in the tidal rivers were not reported because of the small sample sizes (number of sampling sites available to make estimates). A small sample size results in large uncertainty about the mean value and can result in large year-to-year variability in the mean value. This is illustrated in Table 4-1, which expands upon the entries provided in Table 3-2 for impacted benthic communities. For example, uncertainty estimates for estuarine classes within Chesapeake Bay for the entire four-year period are reasonably small (less than $\pm$10%, except for tidal rivers), but uncertainty for an individual year can be much larger. The individual year mean estimates for the large system class within Chesapeake Bay range from 5 to 33%, and the four-year uncertainty estimates for the estuarine classes in Delaware Bay and Hudson-Raritan system are large (greater than $\pm$16%).

This discussion on Table 4-1 points out the discretion that needs to be employed for estimates of condition when sample sizes are small. This is not a problem with the sampling design employed; rather it illustrates the practical limit imposed by trying to

Table 4-1. Expansion of Table 3-2 to Include Estuarine Classes for Impacted Benthic Communities to Illustrate Effect of Small Sample Size on the Mean and Uncertainty Estimates. Values Are Mean Estimate and 95% Confidence Interval (C.I.). N is Number of Sample Sites Available for Four-year Estimates; Sample Sites for Individual Year Is Approximately N/4.

	1990		1991		1992		1993		1990-1993		
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	N
Chesapeake Bay	33	10	20	11	22	9	27	11	23	5	195
Large Systems	14	27	5	10	24	24	33	19	19	6	85
Small Systems	69	19	43	28	25	14	3	8	26	8	52
Tidal RIvers	57	33	44	44	12	15	32	38	36	17	58
Potomac River	85	45	63	58	*	*	37	48	44	22	20
Rappahannock River	61	45	30	60	67	68	17	83	44	33	18
James River	8	45	10	57	24	32	17	50	19	23	20
Delaware Bay	12	4	13	23	26	24	44	32	24	12	47
Large Systems	*	*	14	29	17	33	40	40	18	17	24
Small Systems	100	‡	*	*	*	*	*	*	52	22	6
Tidal Rivers	88	37	8	46	100	‡	82	60	70	21	17
Hudson-Raritan	78	13	59	†	83	19	70	20	72	8	31
Small Systems	100	‡	100	‡	100	‡	100	‡	100	‡	11
Tidal RIvers	45	33	*	*	59	47	25	50	32	19	20
Long Island Sound	30	22	39	24	39	25	12	17	28	12	53
Large Systems	25	38	33	33	33	33	13	25	26	12	38
Small Systems	80	16	100	‡	100	‡	5	13	51	12	15

* There can be a large uncertainty due to the small number of sampling sites for an individual year in a specific tidal river. Therefore, estimates for the individual years are not presented.

† Due to assumptions in the estimation procedures, if one resource class entirely exceeds the criterion and other resource classes have no exceedences, the C.I. becomes zero.

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero

make percent area estimates for which the sampling density (number of samples) is not adequate for the question posed. If the sampling design were intensified for the areas where the uncertainty estimates were unacceptable, then reduced uncertainties would result. This is illustrated in Table 4-1 by the reduced uncertainty for the four-year estimates compared with the individual year estimates.

There are other sources of uncertainty in the conduct of assessments which can not be mathematically expressed. These other sources of uncertainty are threefold: the inherent randomness of events (stochasticity), imperfect or incomplete knowledge of things that could be known (ignorance), and mistakes in the execution of assessment activities (error). Sources of uncertainty discussed are those associated with the sampling design and the indicators.

Advantages of the EMAP sampling design are: 1) it is systematic and probability based; 2) it quantifies the areal extent associated with an indicator value; 3) it is scalable to regions, watersheds, and local sites; 4) it permits estimation of uncertainties for indicator values; and 5) it provides spatially explicit patterns and distributions of ecological resources and associated habitat and stressor indicators. There are several design issues, however, that contribute to the uncertainties in the assessments. First, sampling is limited to a temporal index period (*e.g.*, once per year) which does not address intra-annual, seasonal, and episodic events that may have a long-term impact on resources. For example, one possible contribution to the unexplained portion of an association is the occurrence of a significant stress to the benthos that occurred outside the index period, and therefore went undetected. Second, the techniques of incorporating non-probabilistic extant data with the EMAP probabilistic sampling design remain to be developed and tested. Overton *et al.* (1993) have proposed one approach for this, but this has not been actually tested. Although this limits our ability to combine extant with EMAP data, it does not prevent the use of extant data to support or supplement EMAP data, such as was done in the prior section addressing how EMAP results compare with existing knowledge. The ability to supplement EMAP indicator data, as well as increase the total sample size of analyses, may contribute to significantly reducing overall uncertainty in the analyses. Finally, there is uncertainty associated with the sample density at different scales (number of sample points) and the associated uncertainty related to the absolute number of samples, as has been noted in Section 3 when confidence intervals are discussed.

Several contributions were made by EMAP to indicators for monitoring: (1) the use of suites of stressor and habitat indicators that are simultaneously measured with the condition indicators, (2) clear and unequivocal links between endpoints and their metrics, and (3) habitat indicators that are directly related to and facilitate the interpretation of both the condition and stressor indicator information. There are, however, several indicator issues that contribute to the uncertainty in these analyses, as illustrated by the large unexplained portion of the associations. First, there is the assumption that the suite of stressor indicator used in these analyses are directly related to and sufficient to discriminate between anthropogenic and natural stresses. The absence of stressor information from indicators specific to nutrient or carbon enrichment and physical stress can be postulated as a potential source of uncertainty.

This is particularly applicable in small estuarine systems and tidal rivers that are likely to experience both high nutrient and carbon loads as well are repeated physical perturbation of the benthos through dredging and shipping.

There are also uncertainties associated with existing indicators. The benthic index metric requires further testing and validation relative to specific habitat indicators (*e.g.,* salinity, grain size, *etc.*). Also, the only biotic condition indicator is for benthos. The condition of the habitats such as the pelagic zone, marshes, and submerged aquatic vegetation has not been addressed. There are a series of implicit assumptions relative to the stressor indicators that need to be tested. First, sediment toxicity and sediment chemistry should be re-examined from the perspective of bioavailability. Second, the assumption that toxicity tests are an accurate surrogate for community exposure, and that exposure is accurately coupled with relevant measures of contaminant availability needs to be rigorously evaluated. The interpretation of sediment contamination data and the combination of sediment toxicity, habitat indicators, and sediment contaminant data into a complex exposure index must be addressed. Currently there is uncertainty regarding the functional relationship between condition and stressor indicators that may limit their predictive value.

Finally, uncertainties are associated with the indicator for dissolved oxygen. Sampling uncertainty is related to the use of a single point estimate taken during the index period as a representative measure of benthic exposure when the tidal and intra-annual variability are not considered. Analyses of the 1990 data did evaluate dissolved oxygen results for different sampling windows within the index period (Weisberg *et al*., 1993). It was observed that the province-wide cumulative distribution function for bottom dissolved oxygen was stable across sampling windows. However, the capability to characterize the DO condition for a site from a single point-in-time measurement was limited. This becomes important when conducting associations. The ability to discriminate natural from anthropogenic sources of low dissolved oxygen is another source of uncertainty. This problem is important in the Virginian Province because most of the hypoxic conditions occur in Chesapeake Bay where there is considerable debate over the causes and sources of the problem (Officer *et al.*, 1984).

One overall concern with the existing suite of indicators is the large unexplained portion of impacted benthic conditions when associations are made with stressors. This large unexplained portion may be due to not including relevant indicators of important stress on the benthic community. It may also be due to how and when we actually measured the indicators; *e.g.,* significant stress may have occurred outside the index period and went undetected with the existing indicators. These are topics for future research.

Although the interpretation of data on single indicators can be improved through further refinement, taken together the suite of indicators used in the EMAP Virginian

Province Demonstration Project does provide a qualitative weight-of-evidence approach to understanding the status and condition of estuarine resources.

4.6 How effective was the Four-Year EMAP Virginian Province Demonstration Project in meeting the program objectives?

As stated in Section 1.1, the four objectives of the original EMAP were: (1) estimate the current status, trends, and changes in selected indicators of the condition of the nation's ecological resources on a regional basis with known confidence; (2) estimate the geographic coverage and extent of the nation's ecological resources with known confidence; (3) seek associations among selected indicators of natural and anthropogenic stress and indicators of ecological condition; and (4) provide annual statistical summaries and periodic assessments of the nation's ecological resources.

The prior discussion in this section, answering the environmental management questions, clearly indicates that the four-year Virginian Province Demonstration Project met the first and third objectives. The second objective was clearly met for the Virginian Province Demonstration Project, since NOAA maps were used for delineation of the estuarine boundaries (Holland, 1990). The success in producing the annual reports (Weisberg *et al.*,1993; Schimmel *et al.,* 1994; Strobel *et al.,* 1994; Strobel *et al.,* 1995) demonstrates that the fourth objective has been achieved.

The development of program data quality objectives (DQOs) (Olsen, 1992) has been proposed as a quantitative measure of evaluating success in achieving the first objective of EMAP. The EMAP Quality Assurance Management Plan (Kirkland, 1992) defines DQOs as "statements identifying the anticipated use of environmental data leading to actions to be taken by EMAP and defining the level of uncertainty a decision maker is willing to accept in the data supporting the decision (and action), expressed in quantitative, statistical terms." The target DQO proposed for status estimates with condition indicators was:

> For each indicator of condition and resource class, on a regional scale, estimate the proportion of the resource in degraded [or impacted] condition within 10% (absolute) of the value with 90% confidence based on four years of sampling.

Olsen (1992) also proposed a target DQO for indicators of condition for trend detection:

> Over a decade, for each indicator of condition and resource class, on a regional scale, detect a linear trend of 2% (absolute) per year, *i.e.,* a 20% change for a decade, in the percent of the resource class in degraded [or impacted] condition. The test for trend will have a maximum significance level of " = 0.2 and a minimum power of 0.7 ($ = 0.3).

Addressing the trends DQO is beyond the intent of this report. The application of the trends DQO to the Four-Year Virginian Province data is addressed in a separate document (Heimbuch *et al.*, 1995b). Results suggest that the design as used in the Virginian Province is capable of detecting a 2% change per year over a 12-year period with a power greater than 0.80 for dissolved oxygen and benthic condition.

The information for evaluating the ability of the Four-Year Virginian Province Demonstration Project to meet the status DQO is presented in Table 4-2, which summarizes information previously presented in this report. Note that this table, in addition to providing province-wide (regional scale) status estimates, includes the estuarine systems that status estimates were made for. Clearly, the EMAP target DQO for status has been met for almost all of the indicators for the entire province and the three resource classes. The major exceptions were for ER-L exceedence and benthic index in the tidal river class. These two indicators marginally exceed the target. This higher uncertainty in the tidal river class estimates was noted in analysis of the 1990 Virginian Province data (Weisberg *et al.*, 1993).

Even though the program DQOs were developed for the province scale, it is encouraging that the target DQO for status could also be met for most indicators in the major estuarine system analyses. The consistent exceptions were the tidal rivers in Chesapeake Bay (Potomac, James, and Rappahannock Rivers). The larger uncertainties in these estimates were primarily due to the small number of samples that were taken in these systems.

4.7 Summary and Conclusions

The four-year assessment of the EMAP-Estuaries program illustrates several important contributions to environmental monitoring in the areas of sampling design and indicator research. In particular, the sampling design is both systematic and probabilistic in nature; is extremely flexible; provides areal estimates, with confidence, of the condition of all indicators; is spatially explicit allowing for a variety of spatial analyses; accommodates post-stratification of data; can be scaled to the problem setting without losing its defining characteristics; and, most importantly, has comparability, which permits the direct comparison of results across differing spatial scales as well as between different categories/populations of resources *(e.g.*, large and small systems and tidal rivers).

The four-year assessment of Virginian Province data affords the opportunity to evaluate the applicability, sufficiency, and effectiveness of EMAP's indicator program. The multi-indicator approach used by EMAP has proven both practical and necessary. Traditional monitoring programs often measure only one type of indicator, either exposure or ecological. EMAP, however, by including the patterns of both natural and

Table 4-2. Means and 95% Confidence Intervals for Four-year Virginian Province Estuarine Condition Estimates for Entire Province, Estuarine Classes, and Major Estuarine Systems. N Is Number of Probability-based Sampling Sites.

	Province (N=425)		Large Estuarine Systems (N=197)		Small Estuarine Systems (N=133)		Large Tidal Rivers (N=95)		Chesapeake Bay (N=195)		James River (N=20)		Rappahannock River (N=18)		Potomac River (N=20)		Delaware Bay (N=47)		Hudson-Raritan System (N=31)		Long Island Sound (N=53)	
	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.	% Area	C.I.
Impacted Benthos (benthic index ≤ 0)	25	3	19	4	37	6	38	14	23	5	19	23	44	33	44	22	24	12	72	8	28	11
DO ≤ 2 ppm	5	2	5	2	1	1	14	6	10	3	0	0	15	11	24	12	0	0‡	0	0‡	3	4
DO ≤ 5 ppm	24	3	27	4	17	5	18	7	31	5	4	0	39	20	25	12	3	4	17	17	48	12
Sediment Toxicity (survival ≤ 60%)	1	1	1	2	3	4	0	0‡	0	0‡	0	0‡	0	0‡	0	0‡	1	2	7	10	5	4
Sediment Toxicity (survival ≤ 80%)	9	2	9	3	12	6	4	4	6	3	8	11	9	11	1	<1	2	2	15	14	13	7
Enriched Metals (for any metal)	49	4	42	5	64	7	69	10	44	5	85	14	81	19	53	14	39	14	85	11	86	8
ER-M Metals (exceedence for any)	5	1	3	3	9	4	7	4	5	3	8	10	0	0‡	0	0‡	1	3	27	9	4	4
ER-M Organics (exceedence for any)	4	1	2	3	3	2	14	4	4	2	0	0‡	22	23	0	0‡	<1	2	44	14	2	2
PAH Normalized for TOC (> 200 µg/g-OC)	5	2	2	2	11	7	14	6	5	3	2	17	22	23	0	0‡	4	5	63	18	2	3
Below All ER-L (metals and organics)	50	4	58	5	35	6	25	12	50	6	23	20	0	0‡	36	20	62	14	1	3	35	12

‡ Due to assumptions in the estimation procedures, if the percent area of exceedence of a criterion is either zero or 100%, then the C.I. is zero.

anthropogenic stressors and habitat factors provides information critical for forming hypotheses that explain the observed ecological observations. There are limitations, however, to the indicator program in the Virginian Province as is illustrated by the discussion on uncertainties. First, only one ecological indicator, benthic community condition, was available to be used to assess the status of ecological condition in the Province. Second, there were no explicit stressor indicators for enrichment or for physical perturbation. This assessment re-iterates the need for additional research on ecological and stressor indicators to reduce uncertainties in the assessments.

The value of the EMAP design and indicator program is illustrated by its ability to successfully identify and quantify the major environmental problems in the estuarine waters of the Virginian Province. When the EMAP characterization of the Virginian Province was compared with analyses from other existing monitoring and environmental data from the states and federal agencies, the general conclusions were the same. The agreement between conclusions drawn from EMAP and those from existing data could be viewed as an initial validation of the EMAP concept. This is very important because it provides confidence that when the EMAP design and indicators are applied to data-poor environmental areas, they are likely to capture successfully the major ecological problems. The added value of the EMAP design over many other studies, of course, is the quantification with degree of uncertainty (confidence limits) that is provided with the results.

Although uncertainties certainly remain, the results of the four-year Virginian Province assessment are encouraging. The Demonstration Project clearly indicated that the EMAP objectives were not only reasonable but were achievable with available indicators collected with a probability-based sampling design. It was demonstrated that the EMAP design can be used to quantify with confidence the status and condition of ecological resources. Reducing the uncertainties in the assessment should be approached through a systematic program of directed research.

SECTION 5: REFERENCES

Bailey, R.G. 1983. Delineation of ecosystem regions. *Environmental Management*, 7:365-373.

Boesch, D.F. 1973. Classification and community structure of macrobenthos in the Hampton Roads area, Virginia. *Marine Biology*, 21:226-244.

Boesch, D.F. and R. Rosenberg. 1981. Response to stress in marine benthic communities. In: *Stress Effects on Natural Ecosystems,* G.W. Barret and R. Rosenberg, eds., John Wiley and Sons, New York, pp. 179-200.

Breteler, R.J., K.J. Scott and S.P. Shepurd. 1989. Application of a new sediment toxicity test using the marine amphipod Ampelisca abdita to San Francisco Bay sediments. In: *Aquatic Toxicology and Hazard Assessment*, Volume 12. U.M. Cowgill, and L.R. Williams, eds., ASTM.

Brongersma-Sanders, M. 1957. Mass mortality in the sea. Pages 941-1010. In: *Treatise on Marine Ecology and Paleoecology*, J.W. Hedgpeth, ed., Memoirs, Geological Society of America, 67.

Broutman, M.A. and D.L. Leonard. 1988. National estuarine inventory: The quality of the shellfish growing waters in the Gulf of Mexico. National Oceanic and Atmospheric Administration, Strategic Assessment Branch, Rockville, MD, 43 pp.

Buhler, D.R. and D.E. Williams. 1988. The role of biotransformation in toxicity. *Fish. Aquatic Toxicology,* 11:303-311.

Cantillo, A.Y. and T.P. O'Connor. 1992. Trace element concentrations in sediments from the NOAA National Status and Trends Programme compared to data from throughout the world. *Chemistry and Ecology*, 7:31-50.

Capuzzo, J.M., M.N. Moore, and J. Widdows. 1988. Effects of toxic chemicals in the marine environment: Predictions of impacts from laboratory studies. *Aquatic Toxicology,* 11:19-28.

Chapman, P.M. 1988. Marine sediment toxicity tests. Pages 391-402. In: *Chemical and Biological Characterization of Sludges, Sediments, Dredge Spoils, and Drilling Muds*, J.J. Lichtenberg, F.A. Winter, C.I. Weber and L. Fredkin, eds., STP 976. ASTM, Philadelphia, PA.

Cochran, J.K. 1991. Forging a partnership between federal monitoring programs and the academic marine community. Summary of a workshop hosted by the Marine

Sciences Research Center, State University of New York at Stony Brook, 29-30 April 1991. Special Report 93 of the Marine Sciences Research Center, State University of New York at Stony Brook.

Connell, D.W. and G.J. Miller. 1984. *Chemistry and Ecotoxicology of Pollution*. John Wiley and Sons, New York.

Cooley, W.W. and P.R. Lohnes. 1971. *Multivariate Data Analysis*, John Wiley & Sons, Inc., New York.

Culliton, T.J., M.A. Warren, T.R. Goodspeed, D.G. Remer, C.M. Blackwell, and J.J. McDonough. 1990. 50 years of population change along the nation's coasts, 1960-2010. Second Report of Coastal Trends Service. U.S. Department of Commerce, NOAA, Strategic Assessment Branch, Rockville, MD, 41 pp.

Daskalakis, K.D. and T.P. O'Connor. 1994. Inventory of chemical concentrations in coastal and estuarine sediments. NOAA Technical Memorandum NOS ORCA 76. National Oceanic and Atmospheric Administration, National Ocean Service, Silver Spring, MD, 66 pp.

Di Toro, D.M., J.D. Mahony, D.J. Hansen, K.J. Scott, M.B. Hicks, S.M. Mayer, and M.S. Redmond. 1990. Toxicity of cadmium in sediments: the role of acid volatile sulfide. *Environmental Toxicology and Chemistry*, 9:1487-1502.

Di Toro, D.M., C.S. Zarba, D.J. Hansen, W.J. Berry, R.C. Swartz, C.E. Cowan, S.P. Pavlou, H.E. Allen, N.A. Thomas, and P.R. Paquin. 1991. Technical basis for establishing sediment quality criteria for nonionic organic chemicals using equilibrium partitioning. *Environmental Toxicology and Chemistry*, 10:1541-1582.

FDA (Food and Drug Administration). 1971. National shellfish register of classified estuarine waters. Food and Drug Administration, Shellfish Sanitation Branch, Northeast Technical Services Unit, Davisville, RI.

FDA. 1985. National shellfish register of classified estuarine waters. National Oceanic and Atmospheric Administration, Strategic Assessment Branch, Rockville, MD.

Fein, G.G., J.L. Jacobson, S.W. Jacobson, P.W. Schwartz, and J.K. Dowler. 1984. Prenatal exposure to polychlorinated biphenyls: Effects on birth size and gestational age. *Pediatrics,* 105:315-320.

Feng, H., J.K. Cochran, H. Lwiza, B.J. Brownawell, and D. J. Hirschberg. 1998. Distribution of heavy metal and PCB contaminants in the sediments of an urban estuary: the Hudson River. *Marine Environmental Research,* 45(1): 69-88.

Forstner, U. and G.T.W. Wittmann. 1981. *Metal Pollution in the Aquatic Environment*. 2nd Revised Edition. Springer-Verlag, New York.

Hargis, W.J., Jr. and D.S. Haven. 1988. Rehabilitation of the troubled oyster industry of the lower Chesapeake Bay. *Journal of Shellfish Research*, 7:271-279.

Heasly, P., S. Pultz, and R. Batiuk. 1989. Chesapeake Bay basin monitoring program atlas, Volume I, Water quality and other physicochemical monitoring programs. CBP/TRS 34/89. U.S. Environmental Protection Agency, Chesapeake Bay Program, Annapolis, Maryland.

Heimbuch, D.G., H.T. Wilson, D.S. Robson, and J.K. Summers. 1995a. Design-based estimation of the proportion of area degraded and associated variance estimators for EMAP-Estuaries sampling in the Louisianian and Virginian Provinces: 1990-1994. Draft report prepared for U.S. Environmental Protection Agency by Coastal Environmental Services, Linthicum, Maryland.

Heimbuch, D.G., H.T. Wilson, S.J. Schwager, and D.S. Robson. 1995b. Power analysis for tests for long term trends in environmental conditions of estuaries. Draft report prepared for U.S. Environmental Protection Agency by Coastal Environmental Services, Linthicum, Maryland.

Hinga, K.R. 1988. Seasonal predictions for pollutant scavenging in two coastal environments using a model calibration based upon thorium scavenging. *Marine Environmental Research,* 26:97-112.

Holland, A.F. 1990. Near Coastal Program Plan for 1990: Estuaries. EPA/600/4-90-033. U.S. Environmental Protection Agency, Environmental Research Laboratory, Office of Research and Development, Narragansett, RI.

Holland, A.F., A.T. Shaughnessy, L.C. Scott, V.A. Dickens, J.A. Ranasinghe, and J.K. Summers. 1988. Progress report: Long-term benthic monitoring and assessment program for the Maryland portion of Chesapeake Bay (July 1986-October 1987). PPRP-LTB/EST-88-1. Prepared for Maryland Power Plant Research Program and Maryland Department of the Environment, Office of Environmental Programs. Versar, Inc., ESM Operations, Columbia, MD.

Holland, A.F., A. Shaughnessy and M.H. Hiegel. 1987. Long-term variation in mesohaline Chesapeake Bay macrobenthos: Spatial and temporal patterns. *Estuaries*, 10:277-245.

Honeyman, B.D. and P.H. Santschi. 1988. Metals in aquatic systems: Predicting their scavenging residence times from laboratory data remains a challenge. *Environmental Science and Technology,* 22:862-871.

Hunsaker, C.T. and D.E. Carpenter, eds. 1990. Environmental Monitoring and Assessment Program: ecological indicators. EPA/600/3-90/060 U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Kirkland, L.L. 1992. EMAP quality assurance management plan. Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC.

Klecka, W.R. 1980. *Discriminant Analysis*. Sage University Paper Series on Quantitative Applications in the Social Sciences, Series No. 07-019, Beverly Sage Publications.

Leonard, D.L., M.A. Broutman, and K.E. Harkness. 1989. The quality of shellfish growing areas on the east coast of the United States. National Oceanic and Atmospheric Administration, Ocean Assessments Division, Rockville, MD.

Lippson, A. J., M.S. Haire, A.F. Holland, F. Jacobs, J. Jensen, R.L. Moran-Johnson, T.T. Polgar, and W.R. Richkus. 1979. Environmental Atlas of the Potomac Estuary. Prepared for Maryland Department of Natural Resources, Power Plant Siting Program. Martin Marietta Corporation, Baltimore, MD.

Long, E.R., and M.F. Buchman. 1989. An evaluation of candidate measures of biological effects for the National Status and Trends Program. National Oceanic and Atmospheric Administration Technical Memorandum NOS OMA 45. Seattle, WA, 105 pp..

Long, E.R. and L.G. Morgan. 1990. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. National Oceanic and Atmospheric Administration Technical Memorandum NOS OMA 52. Seattle, WA, 175 pp.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management*, 19(1):81-97.

Malins, D.C. 1989. Sediment contamination and marine ecosystems: Potential risks to humans. In: *Contaminated Marine Sediments -- Assessment and Remediation*. National Academy of Sciences. National Academy Press, Washington, DC, pp 155-164.

Mayer, F.L., Jr., L.L. Marking, L.E. Pedigo, and J.A. Brecken. 1989. Physiochemical factors affecting toxicity: pH, salinity, and temperature, Part 1. Literature review. U.S. Environmental Protection Agency, Gulf Breeze, FL.

Morganthau, T. 1988. Don't go near the water: Is it too late to save our dying coasts? *Newsweek*, August 1, 42-47.

Nixon, S.W., C.D. Hunt and B.L. Nowicki. 1986. The retention of nutrients (C,N,P), heavy metals (Mn, Cd, Pb, Cu), and petroleum hydrocarbons in Narragansett Bay. Pages 99-122. In: *Biogeochemical Processes at the Land-sea Boundary*, P. Lasserre and J.M. Martin, eds., Elsevier, NY.

National Oceanic and Atmospheric Administration (NOAA). 1987. NOAA estuarine and coastal ocean science framework. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Estuarine Program Office, Washington, DC.

NOAA. 1988. Federal plan for ocean pollution research development and monitoring: Fiscal years 1988-1991. Prepared by the National Ocean Pollution Program Office for the National Ocean Pollution Policy Board, Rockville, MD.

NOAA. 1992. The National Status and Trends Program, Marine Environmental Quality. National Oceanic and Atmospheric Administration, National Ocean Service, Rockville, Maryland.

National Research Council (NRC). 1989. Contaminated marine sediments -- assessment and remediation. National Academy Press, Washington, D.C. 493 pp.

NRC. 1990. Managing troubled waters: the role of marine environmental monitoring. National Academy Press, Washington, DC.

O'Connor, J.S., J.J. Ziskowski, and R.A. Murchelano. 1987. Index of pollutant-induced fish and shellfish disease. Special Report. National Oceanic and Atmospheric Administration, NOS, Rockville, MD.

O’Connor, T.P. 1990. Coastal Environmental Quality in the United States, 1990, Chemical Contamination in Sediments and Tissues. NOAA 20th Anniversary Report, Rockville, MD, 34 pp.

Officer, C.B., R.B. Biggs, J.L. Taft, L.E. Cornin, M.A. Tyler, and W.R. Boynton. 1984. Chesapeake Bay anoxia: Origin, development, and significance. *Science,* 223:22-27.

Olsen, A.R. 1992. Program data quality objectives for EMAP status and trends. Draft report. Environmental Monitoring and Assessment Program, U.S. Environmental Protection Agency, Washington, DC.

Olsen, S., D.D. Robadue, and V. Lee. 1980. An interpretive atlas of Narragansett Bay. Coastal Resources Center, University of Rhode Island, Marine Bulletin 40.

OTA (Office of Technology Assessment). 1987. Wastes in Marine Environments. Washington, DC.

Overton, J. McC., T.C. Young, and W.S. Overton. 1993. Using found data to augment a probability sample: procedure and case study. *Environmental Monitoring and Assessment,* 26:65-83.

Overton, W.S., D. White and D.L. Stevens. 1991. Design report of the Environmental Monitoring and Assessment Program. EPA/600/3-91/053. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Parker, C.A., J.E. O'Reilly, and R.B. Gerzoff. 1986. Oxygen depletion in Long Island Sound. Historical Trends Assessment Program, National Oceanic and Atmospheric Administration, Ocean Assessment Division, Rockville, MD.

Paul, J.F., K.J. Scott, A.F. Holland, S.B. Weisberg, J.K. Summers, and A. Robertson. 1992. The estuarine component of the US E.P.A.'s Environmental Monitoring and Assessment Program. *Chemistry and Ecology*, 7:93-116.

Pearl, H. W. 1988. Nuisance phytoplankton blooms in coastal, estuarine, and inland waters. *Limnology and Oceanography,* 33:823-847.

Pearson, T.H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology Annual Review*, 16:229-311.

Queen, W.H., B.J. Copeland, and J.R. Schubel. 1992. Forging a partnership between federal monitoring programs and the academic marine community. A report of the Board on Oceans and Atmosphere, National Association of State Universities and Land-Grant Colleges, Washington, DC.

Rabalais, N.N., M.J. Dagg, and D.F. Boesch. 1985. Nationwide review of oxygen depletion and eutrophication in estuarine and coastal waters: Gulf of Mexico. Louisiana University's Marine Consortium.

Ranasinghe, J.A., S.B. Weisberg, D.M. Dauer, L.C. Schaffner, R.J. Diaz, and J.B. Frithsen. 1993. Chesapeake Bay benthic community restoration goals. Prepared for the Chesapeake Bay Program Office and the Governor's Council on Chesapeake Bay by Versar, Inc. Columbia, Maryland.

Ranasinghe, J.A., S.B. Weisberg, J. Gerritsen, and D.M. Dauer. 1994. Assessment of Chesapeake Bay benthic macrobenthic resource condition in relation to water quality and watershed stressors. Prepared for the Governor's Council on Chesapeake Bay and the Chesapeake Bay Program by Versar, Inc. Columbia, Maryland.

Reifsteck, D.M., C.J. Strobel, and D.J. Keith. 1993. EMAP-Estuaries 1993 Virginian Province Field Operations and Safety Manual. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Narragansett, RI.

Reish, D.J., and J.L. Barnard. 1960. Field toxicity tests in marine water utilizing the polychaetous annelid *Capitella capitata* (Fabricius). *Pacific Naturalist,* 1:1-8.

Rhoads, D.C., P.L. McCall and J.Y. Yingst. 1978. Disturbance and production on the estuarine sea floor. *American Scientist*, 66:577-586.

Robertson, A., B.W. Gottholm, D.D. Turgeon, and D.A. Wolfe. 1991. A comparative study of contaminant levels in Long Island Sound. *Estuaries*, 14(3):290-298.

Sanders, H.L. 1956. Oceanography of Long Island Sound, 1952-1954. X: The biology of marine bottom communities. *Bulletin of the Bingham Oceanographic Collection,* 15:345-414.

Sanders, H.L., 1960. Benthic studies in Buzzards Bay. III. The structure of the soft bottom community. *Limnology and Oceanography,* 5:138-153.

Sanders, H.L., J.F. Grassle, G.R. Hampson, L.S. Morse, S. Garner-Price and C.C. Jones. 1980. Anatomy of an oil spill: Long-term effects from the grounding of the barge *Florida* off West Falmouth, Massachusetts. *Journal of Marine Research,* 38:265-380.

Sanders, J.E. 1989. PCB pollution in the upper Hudson River. In: *Contaminated Marine Sediments -- Assessment and Remediation*. National Academy Press, Washington, DC, pp 365-400.

SAS. 1990a. *SAS/STAT User Guide, Version 6, 4th Edition, Volume 1*. SAS Institute, Cary, North Carolina.

SAS. 1990b. *SAS/STAT User Guide, Version 6, 4th Edition, Volume 2*. SAS Institute, Cary, North Carolina.

Schubel, J.R., and H.H. Carter. 1984. The estuary as a filter for fine-grained suspended sediment. Pages 81-104. In: *The Estuary as a Filter*, S. Kennedy, ed. Academic Press, Orlando, FL.

Schimmel, S.C., B.D. Melzian, D.E. Campbell, S.J Benyi, J.S. Rosen, and H.W. Buffum. 1994. Statistical Summary: EMAP-Estuaries Virginian Province - 1991. EPA/620/R-94/005. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory,, Narragansett, Rhode Island.

Science Advisory Board (SAB). 1988. Future risk: Research strategies for the 1990s. The report of the Research Strategies Committee. Science Advisory Board. Washington, DC.

Scott, K.J. and Redmond, M.S. 1989. The effects of a contaminated dredged material on laboratory populations of the tubificious amphipod, *Ampelisca abdita*. pp. 289-303. In: *Aquatic Toxicology and Hazard Assessment: 12th Volume,* U.M. Cowgill and L.R. Williams, eds., ASTM STP 1027. Philadelphia, Pennsylvania.

Sharp, J.H., J.R. Pennock, T.M. Church, T.M. Tramontano, and L.A. Cifuentes. 1984. The estuarine interaction of nutrients, organics, and metals: A case study in the Delaware Estuary. Pages 241-258. In: *The Estuary as a Filter*, V.S. Kennedy, ed., Academic Press, Orlando, FL.

Sinderman, C.J. 1979. Pollution-associated diseases and abnormalities of fish and shellfish: A review. *Fish. Bull.* 76:717-741.

Smart, T., E. Smith, T. Vogel, C. Brown, and K. Wolman. 1987. Troubled Waters. *Business Week*, October 12, 88-104.

Smayda, T.J., and T.A. Villareal. 1989. The 1985 "brown-tide" and the open phytoplankton niche in Narragansett Bay during summer. Pages 159-187. In: *Novel Phytoplankton Blooms*, E.M. Cosper, V.M. Bricelj, and E.J. Carpenter, eds., Coastal and Estuarine Studies, Vol. 35. Springer-Verlag, NY.

Spies, R.B. 1989. Sediment bioassays, chemical contaminants and benthic ecology: New insights or just muddy water? *Marine Environmental Research*, 27:73-75.

Sprague, J.B. 1985. Factors that modify toxicity. In: *Fundamentals of Aquatic Toxicology,* G.M. Rand and S.R. Petrocelli, eds., Hemisphere Publication Corp, New York.

Strobel, C.J., H.W. Buffum, S.J. Benyi, E.A. Petrocelli, D.R. Reifsteck, and D.J. Keith. 1994. Statistical Summary: EMAP-Estuaries Virginian Province - 1992. EPA/620/R-94/019. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Narragansett, RI.

Strobel, C.J., H.W. Buffum, S.J. Benyi, E.A. Petrocelli, D.R. Reifsteck, and D.J. Keith. 1995. Statistical Summary: EMAP-Estuaries Virginian Province - 1990-1993. EPA/620/R-94/026. U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, Narragansett, RI.

Strobel, C.J. and R.M. Valente. 1995. Quality Assurance Report: EMAP-Virginian Province, 1990-1993. EPA/620/R-95/007. U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, Narragansett, RI.

Swartz, R.C. 1987. Toxicological methods for determining the effects of contaminated sediment on marine organisms. Pages 183-198. In: *Fate and Effects of Sediment Bound Chemicals in Aquatic Systems*, K.L. Dickson, A.W. Maki, and W.A. Brungs, eds., Pergamon Press, New York.

Swartz, R.C. 1989. Marine sediment toxicity tests.. In: *Contaminated sediments - assessment and remediation*. National Academy Press, Washington, DC. pp 115-129.

Swartz, R.C., W.A. DeBen, J.K. Jones, J.O. Lamberson, and F.A. Cole. 1985. Phoxocephalid amphipod bioassay for marine sediment toxicity. In: *Aquatic Toxicology and Hazard Assessment: Seventh Symposium*, R.D. Cardwell, R. Purdy, and R.C. Bahner, eds., American Society for Testing and Materials, Philadelphia, PA. pp 284-307.

Swartz, R.C., F.A. Cole, J.O. Lamberson, S.P. Ferraro, D.W. Schults, W.A. DeBen, H. Lee II, and R.J. Ozretich. 1994. Sediment toxicity, contamination and amphipod abundance at a DDT- and dieldrin-contaminated site in San Francisco Bay. *Environmental Toxicology and Chemistry*, 13:949-962.

Terrell, T.T. 1979. Physical regionalization of coastal ecosystems of the United States and its territories. FWS/OBS-79/80. U.S. Fish and Wildlife Service, Office of Biological Services.

The Conservation Foundation. 1988. Protecting America's Wetlands: An Action Agenda. The Conservation Foundation, Washington, DC.

Theede, H. 1973. Comparative studies on the resistance of marine bottom invertebrates to oxygen deficiency and hydrogen sulphide. *Marine Biology* 2:325-337.

Toufexis, A. 1988. Our filthy seas: The world's oceans face a growing threat from manmade pollution. Time, August 1.

Turekian, K.K. 1977. The fate of metals in the oceans. *Geochimica et Cosmochimica Acta*, 41:1139-1144.

U.S. Environmental Protection Agency (USEPA). 1984. Chesapeake Bay: A framework for action. Prepared for the U.S. Congress by the Environmental Protection Agency, Region 3, Philadelphia, PA.

USEPA. 1989. Briefing Report to the EPA Science Advisory Board on the equilibrium partitioning approach to generating sediment quality criteria. EPA/440/5-89-002. USEPA, Criteria and Standards Division, Washington, DC.

USEPA. 1991. Summary, Environmental Monitoring and Assessment Program - Near Coastal, Virginian Province User Network Exchange. A workshop held on 3-5 April 1991, Ocean City, Maryland.

USEPA. 1993a. Proposed sediment quality criteria for the protection of benthic organisms: Acenaphthene. U.S. Environmental Protection Agency, Office of Science and Technology Washington. In review.

USEPA. 1993b. Proposed sediment quality criteria for the protection of benthic organisms: Phenanthrene. U.S. Environmental Protection Agency, Office of Science and Technology Washington. In review.

USEPA. 1993c. Proposed sediment quality criteria for the protection of benthic organisms: Fluoranthene. U.S. Environmental Protection Agency, Office of Science and Technology Washington. In review.

USEPA. 1993d. Proposed sediment quality criteria for the protection of benthic organisms: Dieldrin. U.S. Environmental Protection Agency, Office of Science and Technology Washington. In review.

USEPA. 1994a. Methods for assessing the toxicity of sediment-associated contaminants with estuarine and marine amphipods. EPA 600/R-94/025. U.S. Environmental Protection Agency, Office of Research and Development, Narragansett, Rhode Island.

USEPA. 1994b. Summary of the Comprehensive Conservation and Management Plan. EPA 8-42-S-94-001, Long Island Sound Office, U.S. Environmental Protection Agency, Stamford, Connecticut.

USEPA. 1995a. The State of the Chesapeake Bay 1995. Chesapeake Bay Program, U.S. Environmental Protection Agency, Annapolis, Maryland.

USEPA. 1995b. EMAP-Estuaries Laboratory Methods Manual, Volume I - Biological and Physical Analyses. EPA/620/R-95/008. U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, Narragansett, RI.

USEPA. 1996. The Delaware Estuary. Discover its Secrets. A Management Plan for the Delaware Estuary. The Comprehensive Conservation and Management Plan for the Delaware Estuary. U.S. Environmental Protection Agency, Delaware Estuary Program, Philadelphia, Pennsylvania.

Valente, R. and J. Schoenherr. 1991. EMAP-Estuaries Virginian Province Quality Assurance Project Plan. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Narragansett, RI.

Valente, R. and C.J. Strobel. 1993. Environmental Monitoring and Assessment Program-Estuaries: 1993 Virginian Province Quality Assurance Project Plan. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Narragansett, RI.

Valente, R., C.J. Strobel, J.E. Pollard, KM Peres, T.C. Chiang, and J. Rosen. 1990. Quality Assurance Project Plan for EMAP Near Coastal: 1990 Demonstration Project. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Narragansett, RI.

Valente, R., C.J. Strobel, and S.C. Schimmel. 1992. EMAP-Estuaries Virginian Province 1992 Quality Assurance Project Plan. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Narragansett, RI.

Valette-Silver, N. 1992. Elemental Analyses in Marine Sediments and Biological Tissues. Quality Assurance Program: Summary 1985-1990. NOAA Technical Memorandum NOS ORCA 66. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean Resources Conservation and Assessment, Rockville, Maryland.

Vernberg, F.S. 1972. Dissolved gases. pp. 1491-1526.In: *Marine Ecology, Vol. I, Part 3,* O. Kinne, ed., Wiley-Interscience, New York.

Washington, H.G. 1984. Diversity, biotic and similarity indices, a review with special relevance to aquatic ecosystems. *Water Resources*, 19(6): 653-694.

Weisberg, S.B., J.B. Frithsen, A.F. Holland, J.F. Paul, K.J. Scott, J.K. Summers, H.T. Wilson, D.G. Heimbuch, J. Gerritsen, S.C. Schimmel, and R.W. Latimer. 1993. Virginian Province Demonstration Project Report, EMAP-Estuaries: 1990. EPA/620/R-93/006. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Weisberg, S.B., J.A. Ranasinghe, D.M. Dauer, L.C. Schaffner, R.J. Diaz, and J.B. Frithsen. 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. *Estuaries,* 20(1):149-158.

Whitledge, T.E. 1985. Nationwide review of oxygen depletion and eutrophication in estuarine and coastal waters: Executive Summary. Brookhaven National Laboratory, Upton, NY.

Windom, H.L., S.J. Schropp, F.D. Calder, J.D. Ryan, R.G. Smith, L.C. Bumey, F.G. Lewis, and C.H. Ralinson. 1989. Natural trace metal concentrations in estuarine and coastal marine sediments of the southeastern United States. *Environmental Science and Technology*. 23: 314-320.

Word, J.W., N.P. Kohn, E.S. Barrows, and J.A. Ward. 1994. The influence of sediment stability/disturbance on toxicity of non-polar organic compounds to marine amphipods. Presentation at Workshop on Utility of AVS/EqP in Hazardous Waste Site Evaluation, Woods Hole, Massachusetts.

APPENDIX A: OVERVIEW OF SAMPLING AND ANALYTICAL METHODS

An important aspect of the EMAP Virginian Province Four-Year Demonstration Project was that the data were collected and processed with consistent methods. The field methods are documented in the Field Operations and Safety Manual (Reifsteck *et al.*, 1993). Laboratory methods are documented in the EMAP-Estuaries Laboratory Methods Manual (U.S. EPA, 1995b). A performance-based approach was used for chemical analyses, consistent with the approach used by the NOAA National Status & Trends (NS&T) Program (Valette-Silver, 1992). A strict quality assurance program was used from the initiation of field activities. All activities were conducted in accordance with strict criteria described in a Quality Assurance Project Plan, which was updated annually as needed (Valente and Schoenherr, 1991; Valente and Strobel, 1993; Valente *et al.*, 1990, 1992). An accounting of the results of the EMAP Virginian Province QA Program is found in the 1990-1993 Quality Assurance Report (Strobel and Valente, 1995). An overview of the methods is provided in this appendix, summarized from Strobel *et al*. (1995). All sampling was conducted from small (24') vessels, except for fish from deep-water stations (> 25m), which were collected from larger vessels.

Water column profiles for water quality parameters were collected at each station using a SeaBird SBE-25 Sea Logger CTD. The unit was equipped with probes to measure salinity, temperature, depth, pH, dissolved oxygen (DO), light transmission, fluorescence, and photosynthetically active radiation (PAR). Water quality measurements were collected upon arrival at a sampling station; no effort was made to standardize for the time of day or stage of tide. The CTD was equilibrated at the sea surface, and then lowered through the water column until reaching a depth of one meter above the bottom. There the CTD was allowed to equilibrate again. The unit was then returned to the surface, where data were downloaded to an on-board computer for review and storage. If the CTD cast appeared unusual or failed quality control criteria, the cast was repeated. Beginning in 1991, a bottom water sample was collected, and the dissolved oxygen concentration determined with a YSI Model 58 DO meter. This measurement served as a check on the CTD probe as well as a back-up in case the CTD failed.

Benthic samples for evaluation of invertebrate composition, abundance, and biomass were collected at all sampling sites where a sample could be collected. Three samples were collected at each site using a stainless steel, Young-modified van Veen grab that samples a surface area of 440 cm^2. A small core (2 cm diameter) was collected from each grab for sediment characterization (grain size). The remaining sample was sieved through a 0.5 mm screen using a backwash technique that minimized damage to soft-bodied animals. Samples were preserved in 10% formalin-rose bengal solution and stored for at least 30 days prior to processing to assure proper fixation. In the laboratory, macrobenthic community samples were transferred from formalin to an ethanol solution and sorted. Biomass was measured for key taxa and all other taxa were grouped according to taxonomic type (*e.g.*, polychaetes, amphipods, decapods).

An additional 6-10 sediment grabs at each station were obtained for sediment chemistry and toxicity analyses. The top 2 cm of sediment was removed from each grab using a stainless steel spoon and thoroughly homogenized in a stainless steel pot. Sediment for chemistry analyses was placed in clean glass jars with Teflon liners or polypropylene containers (for organic and metals analyses, respectively), shipped on ice, and stored frozen in the laboratory prior to analysis for contaminants. Sediments were analyzed for 24 polycyclic aromatic hydrocarbons (PAHs), 18 polychlorinated biphenyl (PCB) congeners, DDTs, 11 chlorinated pesticides, tributyl tins, and 15 metals (Table 3-9). The chemical analyte list is the same as used in the NOAA NS&T Program. An additional aliquot was placed in a small polyethylene bag and refrigerated for grain size analysis. The remainder of the composite sample ($\geq$ 3,000 cm^3) was placed in a clean one gallon plastic jar for sediment toxicity testing.

Analysis of sediments for major and trace elements involved a total digestion *(i.e.*, complete dissolution) of the sediment matrix. For the metals Ag, Al, Cr, Cu, Fe, Mn, Ni, Pb, and Zn, the total digestion was accomplished using HF/HNO_3 in an open beaker on a hot plate, followed by instrumental analyses using inductively-coupled plasma-atomic emission spectrometry (ICP-AES). For metals As, Cd, Sb, Se, and Sn, a microwave digestion using HNO_3/HCL in a closed Teflon-lined pressure vessel was followed by analysis using Zeeman-corrected, stabilized temperature graphite furnace atomic absorption (GFAA). Mercury (Hg) was analyzed by cold vapor atomic absorption spectrometry.

The analysis of organic contaminants in the sediment involved sample extraction and cleanup followed by instrumental analysis. This included Soxhlet extraction, extract drying using sodium sulfate, extract concentration using Kuderna-Danish apparatus, removal of elemental sulfur with activated copper, and removal of organic interferents with gel permeation chromatography (GPC) and/or alumina. Following extraction and cleanup, PAH compounds were analyzed using gas chromatography/mass spectrometry (GC/MS). The pesticides and PCB congeners were analyzed using gas chromatography/electron capture detection (GC/ECD) with second column confirmation.

Toxicity tests were performed on the composite sediment samples from each station using the standard 10-day acute test method (U.S. EPA, 1995b, taken from U.S. EPA, 1994a) and the tube-dwelling amphipod *Ampelisca abdita*. Amphipods were exposed to sediment from the site for 10 days under static conditions in 1-L glass test chambers. Five replicates per station were tested with 20 amphipods per replicate. A performance control (*i.e.*, treatment with uncontaminated sediment) was run with each test, as was a water-only test using a reference toxicant (Cu or sodium dodecyl sulfate) to evaluate the condition of the test organisms. Eighty-five percent survival in the sediment control was required for a test to be valid. To normalize for test conditions and amphipod health, survival among treatments is expressed as percent of control survival.

APPENDIX B: REFINEMENTS TO BENTHIC INDEX

Measurements of the ecological condition of benthic communities are critical components of the overall assessment of condition of estuarine waters; benthic organisms are ideal integrators of water and sediment quality. Traditional analyses of benthic communities yield a rich data set on species composition and abundance, and functional diversity and structure. The challenge of these analyses is to relate patterns in taxonomic and functional structure of the benthos to stressor characteristics of interest to assessments that also account for the range of habitat conditions encountered in sampling estuarine waters. For the Virginian Province, important stressors are related to toxic contaminants and hypoxia; affects of these stressors on benthic organisms must be demonstrated across habitat conditions that range from marine to fresh waters and silt to sand.

An index of estuarine benthic community condition was originally developed using data from the 1990 Virginian Province Demonstration Project. The procedures used to develop this benthic index were documented in the EMAP-Estuaries Virginian Province 1990 Demonstration Project Report (Weisberg *et al*., 1993). The index was updated using data collected during 1990-91 in the Virginian Province in Schimmel *et al*. (1994). The current index, discussed in this appendix and used to report on the condition of benthic resources in this report, was developed from a subset of the data collected over the entire four years of sampling in the Virginian Province (1990-93). This effort represents EMAP's continued attempt to discover, among many individual metrics, a single metric or combination of metrics that has a high level of discriminatory power between good and poor ecological conditions. The index developed with the 1990-93 data is an improvement upon and a revision to the prior benthic indices; it has always been the intent to continually revise the index as more data became available (Weisberg *et al*., 1993).

The basic approach to develop the index is to determine the combination of individual benthic metrics that best discriminates between good and poor benthic conditions encountered during sampling. Discriminant analysis (Cooley and Lohnes, 1971; Klecka, 1980) is the analytical tool used to accomplish this. The actual process for developing the benthic index involves several discrete steps:

1. Identification of a set of candidate benthic metrics that include components of faunal and functional diversity and structure.

2. Development of a test data set that contains relatively pristine sites and those exhibiting toxic contamination, hypoxia, or both.

3. Identification of combinations of candidate metrics that discriminate between impacted and unimpacted sites.

4. Validation of the selected combination of metrics.

The test data set from 1990 sampling in the Virginian Province consisted of 19 impacted sites (4 with salinity < 5 ppt) based on high sediment contaminant concentrations (combined with toxicity) or low near-bottom dissolved oxygen levels, and 14 sites (6 with salinity < 5 ppt) classified as unimpacted, or reference, based on low sediment contaminant levels and absence of toxicity or low dissolved oxygen conditions. The 1990 benthic index correctly classified 89% of the impacted sites and 86% of the unimpacted sites. Metrics in the 1990 index included percent expected number of species (salinity normalized), number of amphipods, percent of total abundance as bivalves, mean weight per individual polychaete, and number of capitellids.

The process of validation for the 1990 benthic index involved an independent data set to ensure that the multivariate solution was not specific to the original test data set used in 1990. Using the same criteria applied in 1990 to define impacted and unimpacted (*i.e.*, reference) sites, an independent data set was established from the 1991 database (Schimmel *et al.*,1994). This validation data set consisted of 13 sites classified as impacted and 46 sites classified as unimpacted.

Of the 46 sites from 1991 classified as unimpacted, the 1990 benthic index correctly classified 39 (85%). Of the 13 sites from 1991 classified as impacted, the 1990 benthic index correctly classified 7 (54%). The relatively high overall rate of misclassification, particularly for impacted sites, was deemed unacceptable, and a decision was made to reconstruct a new benthic index using the combined 1990-91 data set.

The test data set from 1990-91 sampling in Virginian Province consisted of 31 impacted sites (5 with salinity < 5 ppt) and 51 unimpacted sites (9 with salinity < 5 ppt). The 1990-91 benthic index correctly classified 84% of the impacted sites and 85% of the unimpacted sites. Metrics in the index included mean abundance of opportunistic species, biomass/abundance ratio for all species, and mean number of infaunal species. Of particular note was that habitat normalization (*e.g.*, salinity) was not included in this index; habitat normalized metrics were considered for the index development, but were eliminated during the analysis. (See Schimmel *et al.* (1994) for details.) The benthic index developed using the 1990-91 data set suffered from poor representation of impacted and reference conditions in low salinity (< 5 ppt) in the test data set. This benthic index applied to the entire 1990-93 data set indicated a strong relationship with salinity (Figure B-1). It also appeared to misclassify good sites in the oligohaline and impacted sites in the meso- and polyhaline (see Figure B-2) with 90-91 BI applied to the 90-91 calibration data set.

Concerns with the 1990-91 benthic index were that no habitat normalization was

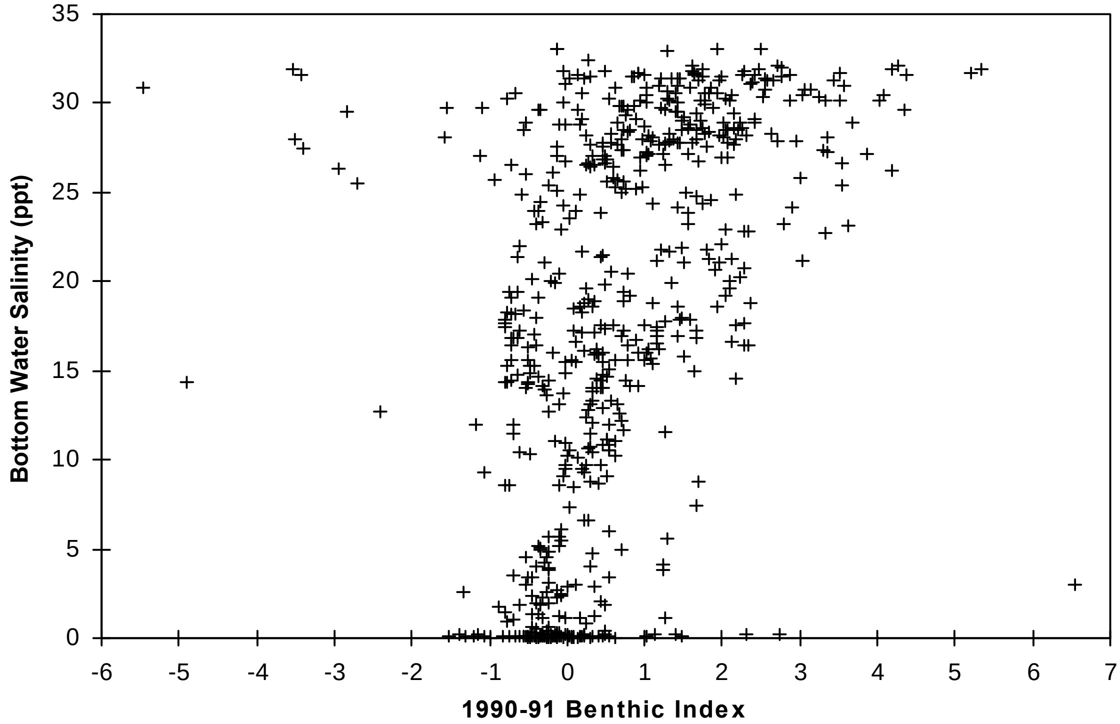

Figure B-1. The 1990-91 benthic index versus bottom water salinity for 1990-93 EMAP Virginian Province data set. Negative benthic index values indicate impacted conditions.

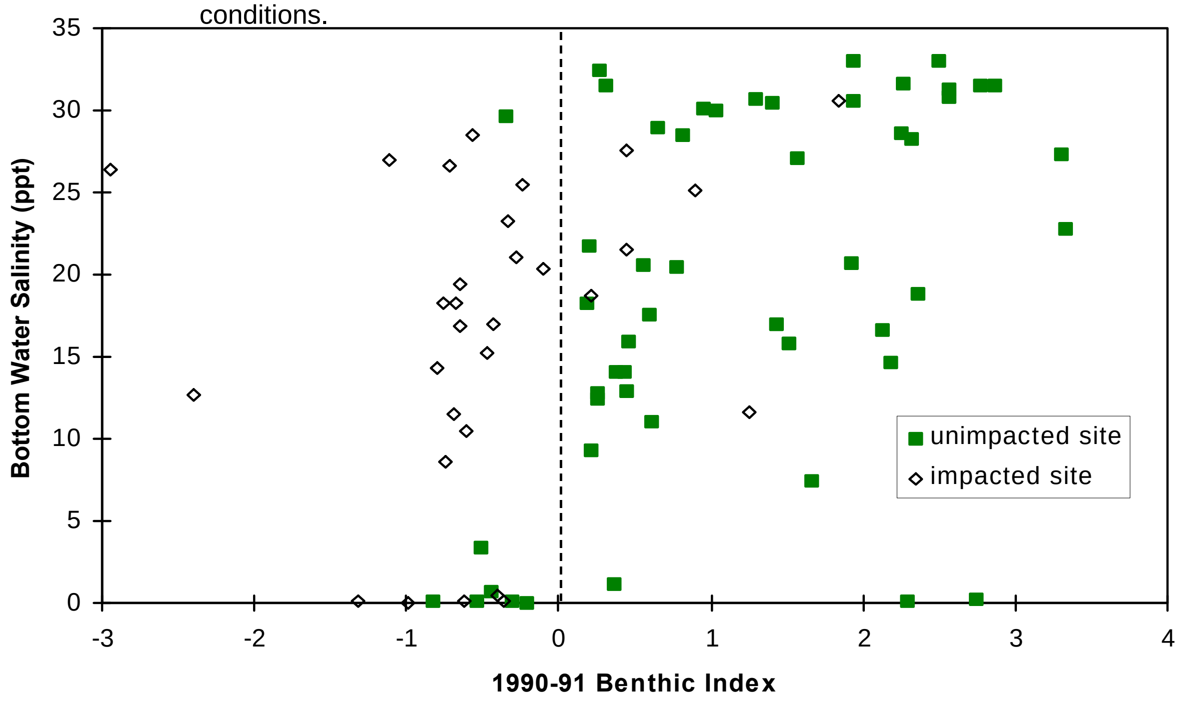

Figure B-2. The 1990-91 benthic index versus bottom water salinity for 1990-91 calibration data set (Schimmel *et al*., 1994). Negative benthic index values indicate impacted conditions.

included in any of the individual metrics and that there was an apparent bias with salinity. For these reasons, the benthic index was reconstructed using data from 1990-93 addressing the following points:

1. Importance of habitat normalization of individual benthic metrics.

2. Inclusion of a wider selection of benthic metrics as candidates for the discriminant analysis.

3. Balance of sites in the test data set for impacted and unimpacted categories and across salinity zones.

RECONSTRUCTION OF THE BENTHIC INDEX USING 1990-93 DATA

Reconstruction of the benthic index using the combined 1990-93 data followed the same basic steps described in the 1990 Demonstration Project Report (Weisberg *et. al*. 1993) and the 1991 Statistical Summary (Schimmel *et al.*, 1994). Results and discussion are presented in the following sections.

Step 1: Identify candidate benthic measures

As in 1990 and 1991, benthic abundance, biomass, and species composition data were used to define descriptors of the major ecological attributes of the benthic assemblages occurring at each sample site (Table B-1). Additional benthic diversity metrics evaluated included Shannon's H, Simpson's D, Gleason's D, and Pielou's evenness (Washington, 1984) based upon infauna, epifauna, and both infauna and epifauna. These biodiversity measures are defined as:

Shannon's H: $$H = -\sum_{i=1}^{S} \frac{n_i}{N} \log_{10} \frac{n_i}{N},$$

Simpson's D: $$D = \sum_{i=1}^{S} \frac{n_i (n_i - 1)}{N (N - 1)},$$

Gleason's D: $$D = \frac{S}{\ln N},$$

Pielou's evenness: $$E = \frac{\text{Shannon's } H}{\log_{10} S},$$

where n_i = number of individuals for species i, N = total number of individuals, and S = number of species.

Table B-1. Candidate Benthic Measures Used to Formulate the Benthic Index. T-tests Were Used to Test Equality of the Means for Each Metric for Impacted Versus Unimpacted Sites in the Test Data Set.

Candidate benthic metrics	t-test (p-value)	Direction (+ = greater mean value at unimpacted sites)
Measures of Biodiversity		
Shannon's H based on total infauna	< 0.001	+
Shannon's H based on total epifauna	0.01	+
Shannon's H based on total infauna and epifauna	< 0.001	+
Simpson's D based on total infauna	0.003	-
Simpson's D based on total epifauna	0.02	-
Simpson's D based on total infauna and epifauna	0.006	-
Gleason's D based on total infauna	< 0.001	+
Gleason's D based on total epifauna	0.007	+
Gleason's D based on total infauna and epifauna	< 0.001	+
Pielou's evenness based on total infauna	0.004	+
Pielou's evenness based on total epifauna	0.11	+
Pielou's evenness based on total infauna and epifauna	0.007	+
Measures of Community Condition		
Total benthic biomass per site	0.76	+
Mean biomass per grab	0.76	+
Mean infaunal abundance per grab	0.11	-
Mean epifaunal abundance per grab	0.07	+
Total number of infaunal species/site	< 0.001	+
Total number of epifaunal species/site	0.006	+
Mean number of infaunal species/grab	< 0.001	+
Mean number of epifaunal species/grab	0.007	+
Measures of Individual Health		
Biomass/abundance ratio	0.31	+
Mean weight of individual bivalves	0.49	-
Mean weight of individual molluscs	0.46	-
Mean weight of individual polychaetes	0.39	-
Mean weight of all individual organism except molluscs	0.36	+
Measures of Functional Groups		
Mean abundance of dominant species	0.08	-
Mean abundance of opportunistic species	0.03	-
Mean abundance of opportunistic species minus amphipods	0.01	-
Mean abundance of equilibrium species	0.1	+
Mean abundance of suspension feeding species	0.48	-
Mean abundance of deposit feeding species	0.04	-
Mean abundance of omnivore/carnivore species	0.003	+
Measures of Taxonomic Composition		
Mean abundance of amphipods	0.05	+
Mean abundance of bivalves	0.3	-
Mean abundance of gastropods	0.03	+
Mean abundance of molluscs	0.44	-
Mean abundance of polychaetes	0.68	+
Mean abundance of capitellid polychaetes	0.17	+
Mean abundance of spionid polychaetes	0.14	-
Mean abundance of tubificid oligochaetes	0.06	-
Ratio of linholl and strebene to all infauna	0.58	-

Estuaries are characterized by large natural variations in certain physicochemical conditions (*e.g.*, salinity, sediment grain size) known to be major factors controlling the diversity and abundance of resident biota. Such factors need to be identified and controlled for before the responses of candidate benthic measures to environmental stress can be characterized accurately. Pearson correlation coefficients were calculated to determine relationships between the individual metrics listed in Table B-1 and various physical habitat variables such as sediment silt-clay content, bottom water salinity, water depth, and latitude. Sediment total organic carbon (TOC) concentration was not treated as an habitat variable since it can be viewed as a potential environmental stress similar to low dissolved oxygen and sediment contamination.

Many of the candidate benthic measures were significantly ($p < 0.05$) correlated with at least one of the habitat factors measured (Table B-2). However, only seven of the correlations accounted for a significant proportion of the total variation, defined here as more than 25% ($r^2 \$ .25$). Four of these seven were measures of species richness (total number of infaunal/epifaunal species and mean number of infaunal/epifaunal species), which were positively correlated with bottom salinity. The other three were diversity measures (Shannon’s H based upon epifauna and Gleason’s D based upon infauna and both epifauna and infauna), which were positively correlated with bottom salinity. Relationships between the rest of the candidate measures and the other habitat factors (*i.e.*, latitude, silt-clay content of sediments, bottom water temperature, and water depth) occurred less frequently and did not account for as much of the total variation as relationships with salinity (Table B-2).

Table B-2. Summary of Correlations Between Habitat Indicators and the Candidate Benthic Measures for the Entire 1990-93 Data Set, Using Pearson Correlation Coefficients (significance for $p < 0.05$).

Habitat Indicator	Number of Significant Correlations	Number of Correlations with $r^2 \$ 0.10$	Number of Correlations with $r^2 \$ 0.25$
Bottom water salinity	37	11	7
Latitude	30	0	0
Bottom water temperature	32	8	0
Silt-clay content of sediments	27	9	0
Water depth	24	1	0

Polynomial regressions for these seven benthic metrics against salinity were developed according to the procedure discussed in the Weisberg *et al.* (1993). These regressions were established by fitting the 90th percentile of a 3 part per thousand (salinity) moving average of the individual benthic metric versus salinity. Refer to Table B-3 for the regression coefficients and correlation coefficients for the regressions.

Table B-3. Regression Coefficients and Correlation Coefficients (r^2) for Bottom Water Salinity Normalization Functions for Benthic Measures.

benthic metric	**polynomial for salinity normalization**	r^2
total number of infaunal species	$21.795 - 2.704\ sal + .292\ sal^2 - .0054\ sal^3$	.97
total number of epifaunal species	$8.354 - 1.641\ sal + .1496\ sal^2 - 0.0026\ sal^3$	.77
mean number of infaunal species	$13.015 - 1.353\ sal + .1700\ sal^2 - .0032\ sal^3$	.97
mean number of epifaunal species	$7.146 - 1.230\ sal + .0917\ sal^2 - .0014\ sal^3$	.63
Shannon's H based upon epifauna	$.594 - .0191\ sal + .00243\ sal^2 - .000044\ sal^3$	.74
Gleason's D based upon infauna	$3.394 - .366\ sal + .0433\ sal^2 - .000871\ sal^3$	.95
Gleason's D based upon infauna and epifauna	$4.283 - .4980\ sal + .0542\ sal^2 - .00103\ sal^3$	.95

The salinity normalized metrics were added to the candidate list of measures. The new metrics were rechecked for possible correlations with the habitat variables. No significant correlations (r^2 $ .25) were observed. The complete candidate list of measures for testing (48 total) included those identified in Table B-1 and the seven salinity normalized metrics in Table B-3.

Step 2: Develop a test data set

Sites from the 1990-93 data set that had valid benthic assemblage, bottom dissolved oxygen, bottom salinity, sediment toxicity, and sediment contaminant data (539 sites) were used to select sites for reconstructing the benthic index. Note that these sites include probability-based sampling sites as well as judgmental sites. Continuous bottom dissolved oxygen measurements were only available for 1990-91, so the point-in-time measurements for dissolved oxygen and salinity were used to incorporate data from all four years. The criteria that were applied to the 539 sites were modified from those used in Weisberg *et al*. (1993) and Schimmel *et al*. (1994). The criteria used here were more restrictive than those previously used (*i.e.*, "better" good sites and "worse" poor sites).

Any of the following criteria needed to be met for a site to be classified as impacted:

1. Sediment toxicity with survival **#** 80% and significantly different from controls,

   or

2. Sediment contamination with at least one exceedence of Long *et al.* (1995) ER-M values or more than 10 exceedences of ER-L values,

   or

3. Bottom dissolved oxygen concentrations **#** 2 mg/l.

All of the following criteria had to be met for the site to be classified as an unimpacted site:

1. No sediment contaminant exceeded an ER-M value and no more than three sediment contaminants exceeded ER-L values,

   and

2. No sediment toxicity was observed (*i.e.*, survival greater than 80% and not significantly different from controls),

   and

3. Bottom dissolved oxygen concentration **$** 7 mg/l.

The selected test data set from the 1990-93 data used for calibration contained 60 sites; 30 were categorized as impacted and 30 as unimpacted (Tables B-4 and B-5). For each of these two categories, 10 sites were in each of the three salinity zones (< 5, 5-18, and > 18 ppt). These salinity zone categories are the same as used in Weisberg *et al.* (1993) and Schimmel *et al.* (1994). Because we wanted an equal number of sites in all 6 categories and the minimum number in any category was 10, each category was trimmed to 10 sites. For those categories with more than 10 sites meeting the criteria, the sites were selected that represented the best (for unimpacted sites) or worst (for impacted sites) sites that maintained balance across sediment grain sizes. The remaining sites (52) that were not used for calibration were set aside for use in the validation step.

Table B-4. Impacted Sites from 1990-93 Used to Calibrate the Benthic Index. Information for Each Site Includes Estuarine System, Date of Sampling, EMAP Station Number, and Geographic Coordinates.

Habitat Class	Impacted Sites for Calibration		
Low Salinity (< 5 ppt)	Delaware River 91-08-11 VA91-365 40E6'4" 74E50'11"	Anacostia River 90-08-26 VA90-088 38E52'11" 76E59' 51"	Hudson River 91-08-28 VA91-411 41E30'58" 73E59'30"
	Delaware River 90-08-20 VA90-228 39E58'5" 75E6'0"	Hudson River 93-07-28 VA93-726 41E56'8" 73E56'54"	Delaware River 90-07-29 VA90-223 39E45'0" 75E29'0"
	Housatonic River 90-08-21 VA90-169 41E17'12" 73E4'19"	Susquehanna River 91-07-30 VA91-351 39E34'41" 76E5'29"	Middle River 91-08-01 VA91-136 39E18'18" 76E24'36"
	Hudson River 92-08-23 VA92-570 41E48'4" 73E56'51"		
Brackish (5-18 ppt)	Chesapeake Bay 91-07-18 VA91-437 39E7'11" 76E17'39"	St Clements Bay 91-07-24 VA91-304 38E13'19" 76E43'48"	Chesapeake Bay 91-08-15 VA91-325 38E37'29" 76E27'52"
	Passaic River 90-07-31 VA90-103 40E45'0" 74E9'54"	Toms River 93-08-20 VA93-676 39E56'40" 74E11'0"	Chesapeake Bay 93-08-05 VA93-647 38E29'4" 76E26'37"
	Hackensack River 90-08-01 VA90-102 40E45'0" 74E5'12"	Potomac River 90-08-16 VA90-180 38E4'13" 76E27'53"	Chesapeake Bay 93-08-04 VA93-630 38E4'18" 76E16'36"
	Breton Bay 91-07-28 VA91-312 38E15'22" 76E39'39"		
Estuarine (> 18 ppt)	Chesapeake Bay 90-08-24 VA90-062 38E59'12" 76E21'29"	Arthur Kill 93-08-07 VA93-735 40E38'50" 74E10'48"	Blackrock Harbor 90-08-20 VA90-098 41E9'35" 73E12'37"
	Chesapeake Bay 92-08-14 VA92-497 38E25'7" 76E22'50"	Raritan River 91-08-04 VA91-369 40E30'40" 74E18'0"	Long Island Sound 91-08-05 VA91-379 40E55'32" 73E37'7"
	Chesapeake Bay 93-09-02 VA93-644 38E21' 9" 76E19' 5"	Flushing Bay 91-08-07 VA91-377 40E47'31" 73E51'41"	Providence River 93-08-15 VA93-725 41E48'41" 71E23'53"
	Passaic River 93-08-07 VA93-684 40E44'23" 74E7'6"		

Table B-5. Unimpacted Sites from 1990-93 Used to Calibrate the Benthic Index. Information for Each Site Includes Estuarine System, Date of Sampling, EMAP Station Number, and Geographic Coordinates.

Habitat Class	**Unimpacted Sites for Calibration**		
Low Salinity (< 5 ppt)	Rappahannock River 90-08-05 VA90-200 38E12'1" 77E15'6"	Potomac River 92-07-28 VA92-499 38E37'3" 77E12'45"	Rappahannock River 93-08-19 VA93-731 38E4'51" 76E57'22"
	Rappahannock River 93-08-21 VA93-639 38E14'56" 77E15'51"	James River 93-08-16 VA93-610 37E18'23" 77E15'1"	Salem River 90-08-26 VA90-252 39E34'50" 75E29'43"
	James River 90-07-23 VA90-210 37E20'0" 77E16'22"	Susquehanna Flats 93-08-01 VA93-670 39E28'49" 76E4'13"	Middle River 92-08-04 VA92-136 39E18'18" 76E24'36"
	Hudson River 91-08-29 VA91-424 42E8'0" 73E54'26"		
Brackish (5-18 ppt)	Chesapeake Bay 92-08-03 VA92-058 39E7'45" 76E16'53"	St Marys River 92-08-28 VA92-486 38E8'3" 76E28'58"	Chesapeake Bay 91-08-11 VA91-291 37E55' 3" 76E15'22"
	Nanticoke River 93-08-10 VA93-642 38E19'41" 75E53'41"	Tangier Sound 93-08-07 VA93-045 38E9'38" 76E1'34"	Manokin River 93-09-01 VA93-633 38E7'45" 75E53'25"
	Potomac River 92-08-28 VA92-485 38E7' 25" 76E29'52"	Tangier Sound 91-07-16 VA91-433 38E9'2" 76E0'55"	Chesapeake Bay 90-07-25 VA90-057 37E26'4" 76E14'7"
	Chesapeake Bay 90-08-03 VA90-060 37E42'55" 76E16'37"		
Estuarine (> 18 ppt)	Pocomoke Sound 90-08-11 VA90-039 37E53'43" 75E46'39"	Great Bay 91-07-24 VA91-348 39E29'46" 74E22'56"	Napeague Bay 90-09-11 VA90-159 41E3'42" 72 0'6"
	Mobjack Bay 92-08-21 VA92-465 37E19'46" 76E22'51"	Long Island Sound 93-08-04 VA93-697 41E6'45" 72E32'7"	Narragansett Bay 90-09-24 VA90-071 41E29'54" 71E24' 8"
	Back River 91-08-17 VA91-267 37E6'40" 76E17'47"	Lake Montauk 92-08-28 VA92-545 41E3'58" 71E55'15"	Nantucket Sound 90-08-16 VA90-037 41E31' 8" 70E17'38"
	Delaware Bay 90-08-29 VA90-004 39E8'43" 75E8'7"		

Steps 3 and 4: Identify combinations of candidate benthic measures that discriminate between unimpacted and impacted sites, and validate the combination of metrics

A series of discriminant analyses were run in succession to identify the benthic measures from Tables B-1 and B-3 which best discriminated between the impacted and unimpacted sites in the test data set and validated with the validation data set (Table B-6 and B-7). The SAS procedure for stepwise discriminant analysis, STEPDISC (SAS, 1990b), was used to select the combination of metrics for further analysis. The SAS procedure DISCRIM (SAS, 1990a) was then used to determine the discriminant function from the parameters selected by STEPDISC. Targets of 90% correct classification for calibration (using the calibration data set, Tables B-4 and B-5) and 80% for validation (using the validation data set, Tables B-6 and B-7) were set as goals for selection of metrics in the benthic index. The cross-validation classification feature of DISCRIM was used to determine the sensitivity of the index to individual sites in the calibration data set.

The results of the various discriminant analyses are summarized in Table B-8. Six variables were included in the model generated by the first stepwise discriminant analysis (Index 1): 1) salinity normalized Gleason's D based upon infauna, 2) tubificid abundance, 3) epifaunal abundance, 4) polychaete abundance, 5) Pielou's evenness based upon infauna, and 6) Shannon's H based upon infauna. This combination of metrics correctly classified 93% of the impacted sites and 83% of the unimpacted sites (Table B-8). The cross-validation feature of the discriminant analysis procedure showed that this index had correct classification of 90% for impacted and 77% for unimpacted sites (misclassification mostly for low salinity sites). The canonical r^2, which approximates the total variance explained by the analysis, was 0.65. This index missed the target goal for correct classification, had problem with correctly classifying unimpacted sites for cross-validation, and met the validation target goals. Pielou's evenness based upon infauna entered the discriminant function with negative coefficient, in contrast to how benthic communities are expected to respond with this diversity metric. Exploratory analysis indicated that salinity normalization for the tubificid abundance might be required to correct for misclassification of low salinity unimpacted sites.

The second index was developed incorporating a salinity normalized tubificid abundance metric. The salinity normalization for tubificid abundance required a different procedure than that used to normalize the other benthic metrics. Tubificids are only observed for low salinity water, with some occurrence being normal for unimpacted sites. Impacted sites would be characterized by large tubificid abundances. The salinity normalization selected was:

Table B-6. Impacted Sites from 1990-93 Used to Validate the Benthic Index. Information for Each Site Includes Estuarine System, Date of Sampling, EMAP Station Number, and Geographic Coordinates.

Habitat Class	Impacted Sites for Validation		
Low Salinity (< 5 ppt)	James River 91-08-04 VA91-273 37E14'26" 76E57'18"	Potomac River 92-08-26 VA92-188 38E44'12" 77E 2' 0"	Delaware River 92-08-20 VA92-522 39E38'33" 75E35'1"
	Hudson River 93-07-29 VA93-727 42E11'30" 73E51'27"		
Brackish (5-18 ppt)	Potomac River 90-08-06 VA90-182 38E13'6" 76E47'8"	Potomac River 91-07-28 VA91-302 38E12'26" 76E35'56"	Chesapeake Bay 90-08-19 VA90-056 38E8'40" 76E14'5"
	Rappahannock River 90-08-15 VA90-190 37E44'12" 76E35'4"	Chesapeake Bay 91-07-11 VA91-431 38E0'1" 76E7'35"	
Estuarine (> 18 ppt)	Chesapeake Bay 92-08-30 VA92-500 38E41'57" 76E25'20"	Chesapeake Bay 92-08-30 VA92-482 38E0'20" 76E12'51"	Upper NJ/NY Bay 93-08-07 VA93-173 40E38'48" 74E3'30"
	Newark Bay 92-07-28 VA92-530 40E41'36" 74E7'6"	East River 91-08-06 VA91-378 40E47'31" 73E55'54"	Taunton River 91-08-10 VA91-419 41E42'38" 71E9'49"
	Newark Bay 90-09-20 VA90-260 40E42'17" 74E6'59"	Arthur Kill 93-08-08 VA93-682 40E32'59" 74E14'52"	Kill Van Kull 91-08-03 VA91-373 40E38'52" 74E4'29"
	Arthur Kill 90-07-30 VA90-094 40E37'18" 74E12'12"	Flushing Bay 91-08-07 VA91-375 40E46'37" 73E51'19"	Taunton River 91-08-10 VA91-421 41E46'0" 71E7'23"
	Chesapeake Bay 93-09-03 VA93-626 37E56'22" 76E9'7"	Chesapeake Bay 90-08-04 VA90-046 37E27'2" 76E1'43"	New Bedford Harbor 90-08-15 VA90-099 41E38'33" 70E54'42
	Elizabeth River 90-08-01 VA90-086 36E49'55" 76E17'38"	Long Island Sound 90-07-31 VA90-096 40E55'13" 73E38'42"	

Table B-7. Unimpacted Sites from 1990-93 Used to Validate the Benthic Index. Information for Each Site Includes Estuarine System, Date of Sampling, EMAP Station Number, and Geographic Coordinates.

Habitat Class	**Unimpacted Sites for Validation**		
Low Salinity (< 5 ppt)	no sites		
Brackish (5-18 ppt)	Rappahannock River 93-08-20 VA93-621 37E56'35" 76E50'47"	Chesapeake Bay 92-08-30 VA92-058 39E7'45" 76E16'53"	Tangier Sound 91-07-15 VA91-045 38E9'38" 76E1'34"
	Chesapeake Bay 90-08-17 VA90-058 39E7'45" 76E16'53"	Chesapeake Bay 93-08-09 VA93-631 38E5'16" 76E4'5"	Tangier Sound 91-07-16 VA91-434 38E8'30" 76E0'26"
	Chesapeake Bay 93-08-04 VA93-058 39E7'45" 76E16'53"	Monie Bay 92-08-15 VA92-488 38E12'55" 75E51'5"	Tangier Sound 92-08-27 VA92-045 38E9'38" 76E1'34"
Estuarine (> 18 ppt)	Barnegat Bay 90-08-10 VA90-256 39E56'36" 74E6'7"	Indian River Bay 93-09-02 VA93-150 38E35'36" 75E6'42"	Nantucket Sound 93-08-22 VA93-724 41E35'50" 70E1'9"
	Delaware Bay 91-07-22 VA91-341 39E8'41" 75E9'0"	Reed/Absecon Bay 93-08-22 VA93-669 39E24'25" 74E27'53"	Nantucket Sound 92-07-28 VA92-557 41E22'59" 70E17'14"
	Delaware Bay 90-08-29 VA90-019 39E4'59" 74E59'8"	Fishers Sound 93-08-13 VA93-709 41E19'37" 71E56'23"	Block Island Sound 93-09-16 VA93-706 41E16'43" 71E46'31"
	Chesapeake Bay 92-08-08 VA92-454 37E0'52" 76E10'22"	Sinepuxent Bay 93-08-16 VA93-641 38E19'0" 75E6'34"	Great Sound 92-08-09 VA92-509 39E5'31" 74E46'2"
	Chesapeake Bay 93-07-26 VA93-604 37E5'49" 76E1'44"	Block Island Sound 93-09-17 VA93-699 41E8'6" 71E52'32"	Block Island Sound 93-09-17 VA93-707 41E17'6" 71E33'17"
	Delaware Bay 90-08-28 VA90-017 39E0'19" 75E8'0"	Nantucket Sound 92-07-28 VA92-558 41E23'11" 70E3'57"	

Table B-8. Results of Discriminant Analyses Conducted to Combine Candidate Benthic Measures into an Index of Benthic Condition for the 1990-93 EMAP Virginian Province Data Set.

Analysis	Selected Measures	coefficients	Canonical r^2	calibration		cross-validation		validation	
				classify unimpacted sites	classify impacted sites	classify unimpacted sites	classify impacted sites	classify unimpacted sites	classify impacted sites
Index 1	• salinity normalized Gleason's D based upon infauna • tubificid abundance • epifauna abundance • polychaete abundance • Pielou's evenness based upon Infauna • Shannon's H based upon infauna	1.214 -.698 .313 -.487 -.759 .902	0.65	83%	93%	77%	90%	85%	81%
Index 2	• salinity normalized Gleason's D based upon infauna • salinity normalized tubificid abundance • epifauna abundance • Pielou's evenness based upon Infauna • polychaete abundance • Shannon's H based upon infauna and epifauna	1.267 -.775 0.446 -.762 -.483 .823	0.67	90%	97%	87%	93%	81%	81%
Index 3	• salinity normalized Gleason's D based upon infauna • salinity normalized tubificid abundance • epifauna abundance	1.404 -.641 .354	0.61	90%	93%	90%	93%	81%	65%
Index 4	• salinity normalized Gleason's D based upon infauna and epifauna • salinity normalized tubificid abundance • spionid polychaete abundance	1.389 -.651 -.375	0.60	87%	90%	87%	87%	88%	81%
Index 5	• salinity normalized Gleason's D based upon infauna and epifauna • salinity normalized tubificid abundance • spionid polychaete abundance • Shannon's H based upon infauna	1.292 -.645 -.367 .114	0.60	90%	90%	87%	87%	88%	81%
Index 6	• salinity normalized total number of infaunal species • salinity normalized tubificid abundance • Gleason's D based upon infauna • polychaete abundance • epifauna abundance	1.068 -.835 .623 -.576 .345	0.64	90%	97%	83%	93%	81%	81%

salinity normalized tubificid abundance =
*tubificid abundance - 500 * exp(- 15 * bottom salinity)*,

where exp(...) denotes the exponential function. Note that salinity normalized tubificid abundance is negative for unimpacted sites and positive for impacted sites.

The six variables for Index 2 were (Table B-8): 1) salinity normalized Gleason's D based upon infauna, 2) salinity normalized tubificid abundance, 3) epifaunal abundance, 4) Pielou's evenness based upon infauna, 5) polychaete abundance, and 6) Shannon's H based upon infauna and epifauna. This combination of metrics correctly classified 97% of the impacted sites and 90% of the unimpacted sites. The cross-validation feature of the discriminant analysis procedure showed that this index had correct classification of 93% for impacted and 87% for unimpacted sites, correcting for the problem with Index 1. The canonical r^2 was 0.67. This index met the target goal for calibration, had no problem with cross-validation, and met the validation target goal. However, Pielou's evenness based upon infauna and polychaete abundance both entered the discriminant function with negative coefficients, in contrast to how benthic communities are expected to respond with these metrics.

Index 3 was developed similarly to Index 2, but only selecting the first three metrics identified by the stepwise discriminant analysis. This eliminated the variables that contributed to the discriminant function in an unreasonable ecological way. The three variables for this index were (Table B-8): 1) salinity normalized Gleason's D based upon infauna, 2) salinity normalized tubificid abundance, and 3) epifaunal abundance. This combination of metrics correctly classified 93% of the impacted sites and 90% of the unimpacted sites. Cross-validation showed that this index had correct classification of 93% for impacted and 90% for unimpacted. The canonical r^2 was 0.61. This index met the target goal for correct classification, had no problem cross-validation, but missed the validation target goal for the impacted sites.

Index 4 was developed by removing salinity normalized Gleason's D based upon infauna (the first metric to enter for Index 3) from the stepwise discriminant analysis and keeping the first three metrics identified. The three variables for this index were (Table B-8): 1) salinity normalized Gleason's D based upon infauna and epifauna, 2) salinity normalized tubificid abundance, and 3) spionid abundance. This combination of metrics correctly classified 87% of the impacted sites and 90% of the unimpacted sites. Cross-validation showed that this index had correct classification of 87% for impacted and 87% for unimpacted. The canonical r^2 was 0.60. This index just missed the target goal for calibration, had good cross-validation, and met the validation target goal.

Index 5 was similar to Index 4 but kept all of the metrics identified by the stepwise discriminant analysis. The four variables for this index were (Table B-8):

1) salinity normalized Gleason's D based upon infauna and epifauna, 2) salinity normalized tubificid abundance, 3) spionid abundance, and 4) Shannon's H based upon infauna. This combination of metrics correctly classified 90% of the impacted sites and unimpacted sites. Cross-validation showed that this index had correct classification of 90% for impacted and 87% for unimpacted. The canonical r^2 was 0.60. This index met the target goal for calibration, had good cross-validation, and met the validation target goal. This index provided marginal improvement over Index 4 by including a metric with a low coefficient in the discriminant function

Index 6 was developed similarly to Index 4, but by removing the salinity normalized Gleason's D based upon infauna and epifauna (the first metric to enter for Index 4) from the stepwise discriminant analysis. The five variables for this index were (Table B-8): 1) salinity normalized total number of infaunal species, 2) salinity normalized tubificid abundance, 3) Gleason's D based upon infauna, 4) polychaete abundance, and 5) epifaunal abundance. This combination of metrics correctly classified 90% of the impacted sites and 97% of the unimpacted sites (Table B-8). Cross-validation showed that this index had correct classification of 83% for impacted and 93% for unimpacted. The canonical r^2 was 0.64. This index met the target goal for correct classification, had a slight problem with cross-validation, and met the validation target goal. Polychaete abundance entered the discriminant function with negative coefficient, in contrast to how benthic communities are expected to respond with this metric.

Selection of Index for Use in Four-Year Assessments

Index 4 was selected as the benthic index to use for the four-year assessments in this report. This index just missed the target for calibration for unimpacted sites, but correct classification of one additional site would have allowed this index to meet the target goal. Marginal improvement of calibration classification was achieved with Index 5, but it was decided that inclusion of a metric with a low coefficient in the discriminant function was not significant enough to warrant use. A spatial pattern analysis using all of the 1990-93 data was conducted with both of these indices to determine if Index 5 provided any spatial difference in identification of impacted areas. No directly observable difference in patterns could be discerned between use of Index 4 and Index 5.

Cross plots of Index 4 against salinity and grain size for the calibration and validation data sets are show in Figures B-3 to B-6. No discernible bias with respect to these habitat variables are apparent in these figures.

Since a balance in number of sites between impacted and unimpacted categories for the test data set was used, the demarcation in the discriminant function score between impacted and unimpacted sites was zero. No scaling of the benthic

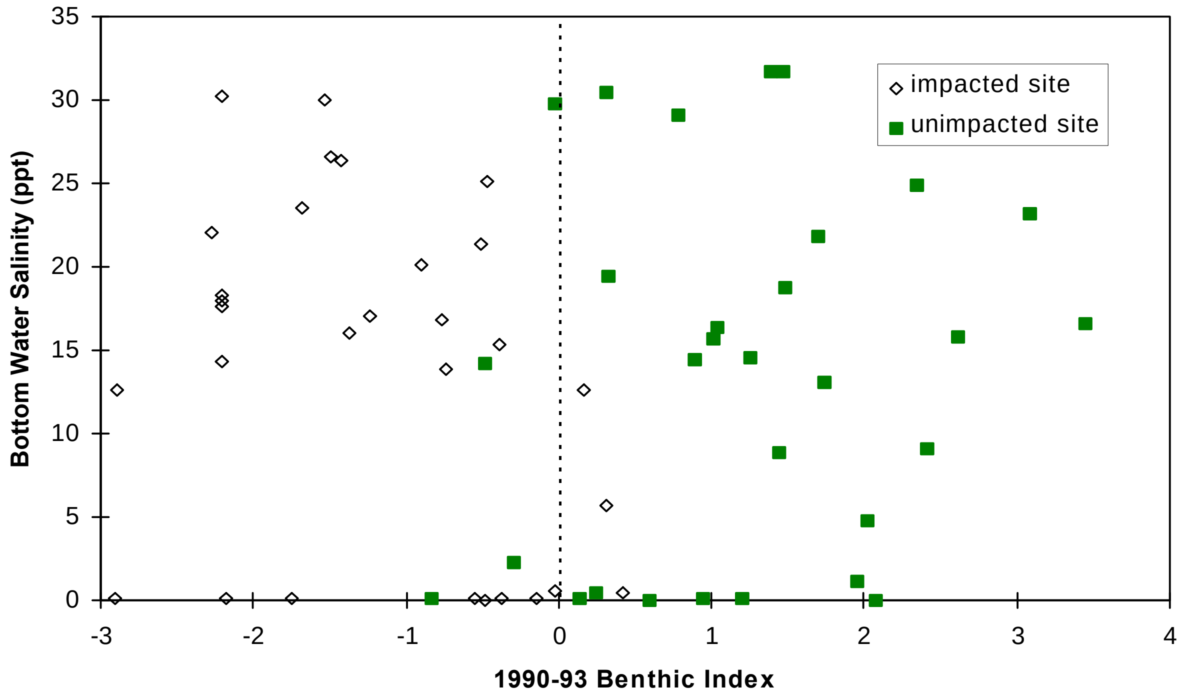

Figure B-3. The 1990-93 benthic index versus bottom water salinity for 1990-93 calibration data set. Negative benthic index values indicate impacted conditions.

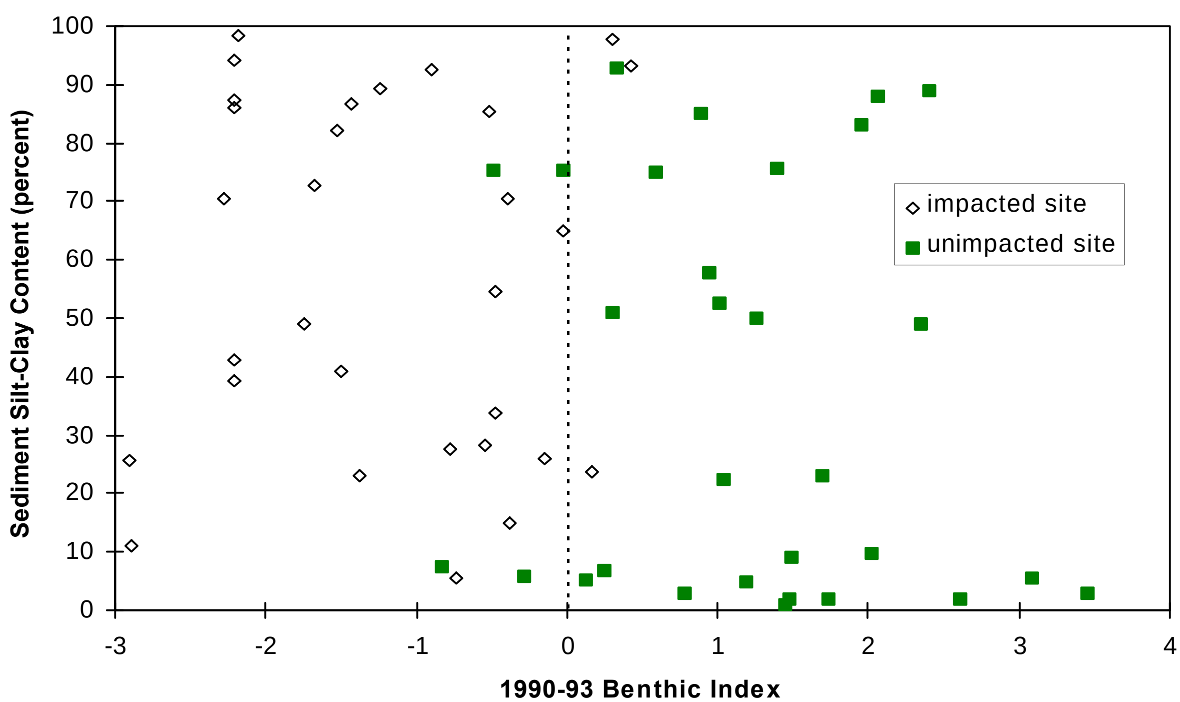

Figure B-4. The 1990-93 benthic index versus sediment silt-clay content for 1990-93 calibration data set. Negative benthic index value indicate impacted conditions.

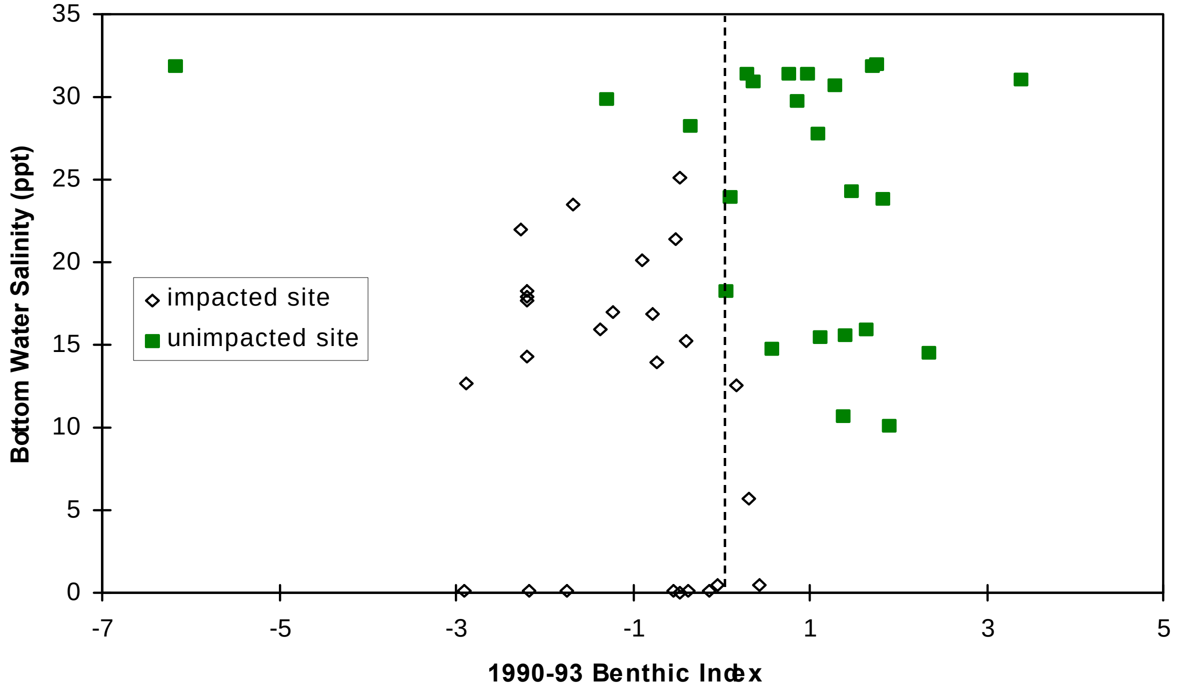

Figure B-5. The 1990-93 benthic index versus bottom water salinity for 1990-93 validation data set. Negative benthic index values indicate impacted conditions.

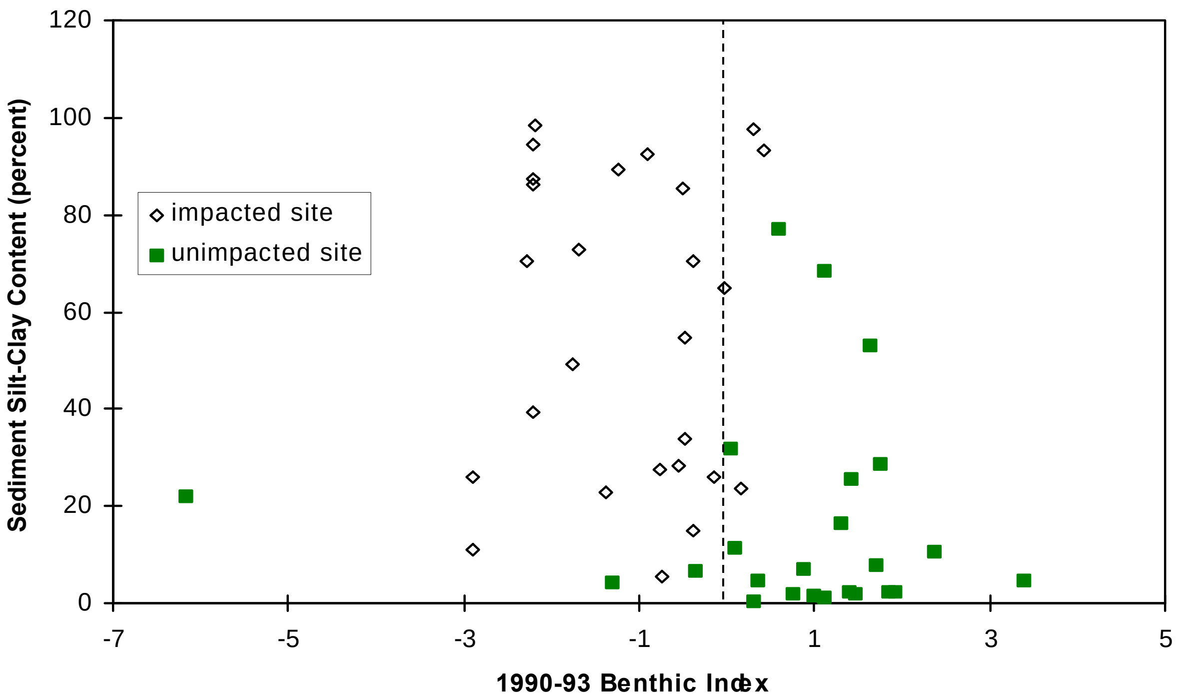

Figure B-6. The 1990-93 benthic index versus sediment silt-clay content 1990-93 validation data set. Negative benthic index values indicate impacted conditions.

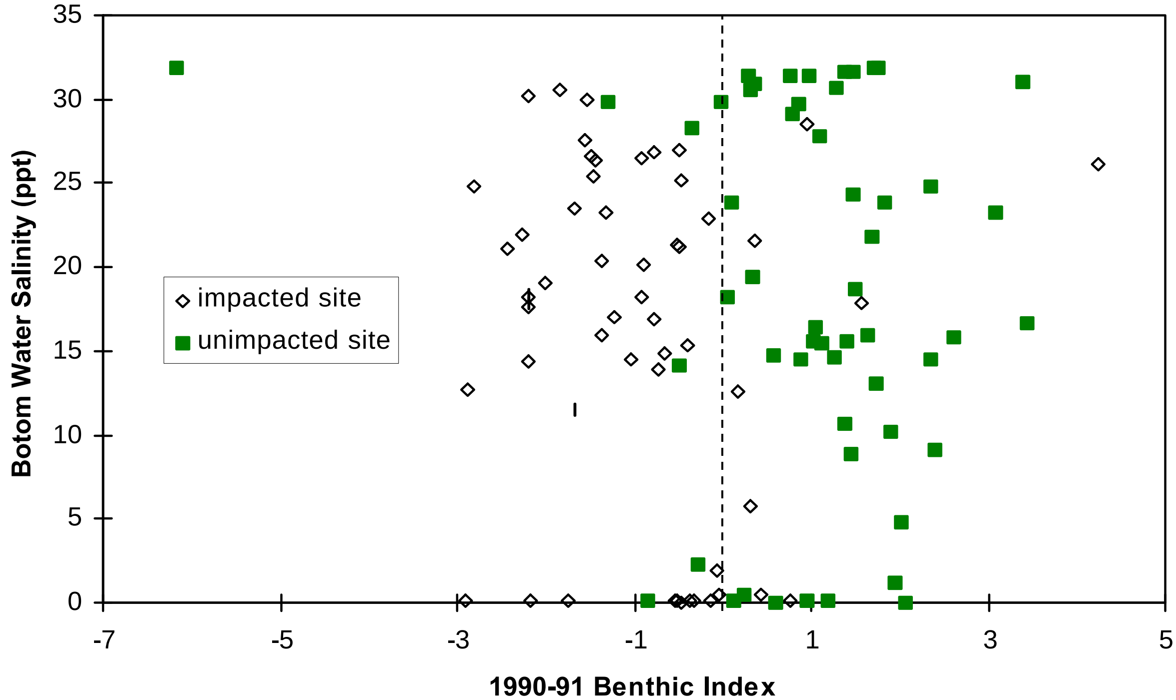

Figure B-7. The 1990-91 benthic index versus bottom salinity for combined four-year calibration and validation data sets. Negative benthic index values indicate impacted conditions.

index was required as was necessary in prior versions of the benthic index (Weisberg *et al.*, 1993; Schimmel *et al.*, 1994).

As a check to see if there was indeed improvement with the 1990-93 benthic index over the 1990-91 index, the latter index was applied to the four-year calibration and validation data sets. For the calibration data set, the 1990-91 index correctly classified 77% of the unimpacted sites and 90% of the impacted sites. For the for the validation data set, the 1990-91 index correctly classified 92% of the unimpacted sites and 65% of the impacted sites. The overall efficiency of the 1990-91 index was 85% for unimpacted sites and 78% for impacted sites. The results are displayed in Figure B-7. A salinity bias exists for the unimpacted sites. The overall efficiency of the 1990-93 index combined data sets was 86% for unimpacted sites and 86% for impacted sites.

Calculating the 1990-93 Benthic Index

The three benthic metrics in the index are: salinity normalized Gleason's D based upon infauna and epifauna, salinity normalized tubificid abundance, and abundance of spionids. The diversity measure is associated with unimpacted conditions (positive contribution). The latter two measures are associated with impacted conditions (negative contribution), with tubificid abundance important in low salinity waters and spionid abundance important in higher salinity waters. The discriminant function calculation normalizes the individual metrics using the mean and standard deviation for the metric in the test data set used for calibration.

The formula for the EMAP Virginian Province 1990-93 benthic index is:

$$
\begin{aligned}
&1.389 * (\text{salinity normalized Gleason's D based upon infauna and epifauna} - 51.5) / 28.4 \\
&\quad - 0.651 * (\text{salinity normalized tubificid abundance} - 28.2) / 119.5 \\
&\quad - 0.375 * (\text{spionid abundance} - 20.0) / 45.4,
\end{aligned}
$$

where

$$
\begin{aligned}
&\text{salinity normalized Gleason's D based upon infauna and epifauna} = \\
&\quad \text{Gleason's D} / (4.283 - 0.498 * \text{bottom salinity} \\
&\qquad + 0.0542 * \text{bottom salinity}^2 \\
&\qquad - 0.00103 * \text{bottom salinity}^3) * 100
\end{aligned}
$$

and

$$
\begin{aligned}
&\text{salinity normalized tubificid abundance} = \\
&\quad \text{tubificid abundance} - 500 * \exp(-15 * \text{bottom salinity}),
\end{aligned}
$$

and

exp(...) denotes the exponential function.