CHESAPEAKE BAY WATER QUALITY MONITORING PROGRAM

LONG-TERM BENTHIC MONITORING AND ASSESSMENT COMPONENT LEVEL I COMPREHENSIVE REPORT

JULY 1984—DECEMBER 1997

Prepared for

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FOREWORD

This document, *Chesapeake Bay Water Quality Monitoring Program: Long-Term Benthic Monitoring and Assessment Component, Level I Comprehensive Report (July 1984—December 1997)*, was prepared by Versar, Inc. at the request of Dr. Robert Magnien of the Maryland Department of Natural Resources under Cooperative Agreement CA-98-02/07-4-30508-3734 between Versar, Inc., and the University of Maryland Center for Environmental and Estuarine Studies. The report assesses the status of Chesapeake Bay benthic communities and evaluates their responses to changes in water quality.

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ACKNOWLEDGMENTS

We are grateful to the State of Maryland's Environmental Trust Fund which partially funded this work. The benthic studies discussed in this report were conducted from the University of Maryland's research vessels and we appreciate the efforts of their captains and crew. We thank Nancy Mountford, Tim Morris, and Suzanne Arcuri of Cove Corporation who identified benthos in many of the samples and provided current taxonomic and autecological information. We also thank those at Versar whose efforts helped to produce this report — the field crews who collected samples, Nathan Huff, Erin Klingebiel, and the other laboratory technicians for processing samples, Thuzar Myint for managing and analyzing data, Dr. Don Strebel and Allison Brindley for GIS support, and Sherian George and Gail Lucas for document production support.

We appreciate the efforts of Dr. Daniel M. Dauer and Anthony (Bud) Rodi of Old Dominion University who coordinate many activities of the Virginia Benthic Monitoring Program. Marcia Olson and other staff of U.S. EPA's Chesapeake Bay Program helped us assess baywide spatial linkages between dissolved oxygen stress and benthic community condition.

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1.0 INTRODUCTION

1.1 BACKGROUND

Monitoring is a necessary part of environmental management as it provides the means for assessing the effectiveness of previous management actions and the information necessary to focus future actions (NRC 1990). Towards these ends, the State of Maryland has maintained an ecological monitoring program for Chesapeake Bay since 1984. The goals of the program are to:

- quantify the types and extent of water quality problems (i.e., characterize the "state-of-the-bay");
- determine the response of key water quality measures to pollution abatement and resource management actions;
- identify processes and mechanisms controlling the bay's water quality; and
- define linkages between water quality and living resources.

The program includes elements to measure water quality, sediment quality, phytoplankton, zooplankton, and benthic invertebrates. The monitoring program includes assessments of biota because the condition of biological indicators integrates temporally variable environmental conditions and the effects of multiple types of environmental stress. In addition, most environmental regulations and contaminant control measures are designed to protect biological resources; therefore, information about the condition of biological resources provides a direct measure of the effectiveness of management actions.

The Maryland program uses benthic macroinvertebrates as biological indicators because they are reliable and sensitive indicators of habitat quality in aquatic environments. Most benthic organisms have limited mobility and cannot avoid changes in environmental conditions (Gray 1979). Benthos live in bottom sediments, where exposure to contaminants and oxygen stress are most frequent. Benthic assemblages include diverse taxa representing a variety of sizes, modes of reproduction, feeding guilds, life history characteristics, and physiological tolerances to environmental conditions; therefore, they respond to and integrate natural and anthropogenic changes in environmental conditions in a variety of ways (Pearson and Rosenberg 1978; Warwick 1986; Wilson and Jeffrey 1994; Dauer 1993).

Benthic organisms are also important secondary producers, providing key linkages between primary producers and higher trophic levels (Virnstein 1977; Holland et al. 1980, 1989; Baird and Ulanowicz 1989; Diaz and Schaffner 1990). Benthic invertebrates are among the most important components of estuarine ecosystems and may represent the largest standing stock of organic carbon in estuaries (Frithsen 1989). Many benthic organisms, such as oysters and clams, are economically important. Others, such as polychaete worms and shrimp-like crustaceans, contribute significantly to the diets of economically important bottom-feeding juvenile and adult fishes, such as spot and croaker (Homer et al. 1980; Homer and Boynton 1978).

The Chesapeake Bay Program's decision to adopt Benthic Community Restoration Goals (Ranasinghe et al. 1994a updated by Weisberg et al. 1997) enhanced use of benthic macroinvertebrates as a monitoring tool. Based largely on data collected as part of Maryland's monitoring effort, these goals describe the characteristics of benthic assemblages expected at sites exposed to little environmental stress. The Restoration Goals provide a quantitative benchmark against which to measure the health of sampled assemblages and ultimately the Chesapeake Bay. Submerged aquatic vegetation (Dennison et al. 1993) and benthic macroinvertebrates are the only biological communities for which such quantitative goals have been established in Chesapeake Bay.

1.2 OBJECTIVES OF THIS REPORT

This report is the fourteenth in a series of Level I Comprehensive reports produced annually by the Long-Term Benthic Monitoring and Assessment Component (LTB) of the Maryland Chesapeake Bay Water Quality Monitoring Program. Level I reports summarize data from the current sampling year, provide a limited examination of how conditions in the current year differ from conditions in previous years of the study, and how data from the present year contribute to describing trends in the bay's condition.

The report contents reflect that this is the third report following a major redesign and refocus of the program. Approaches introduced only in recent years are extended, developed, and better defined. For example, the trend analyses presented in Chapter 3 examine changes in benthic communities at LTB trend stations in greater detail than in the past; the first multiple year estimates of degraded area are provided for many sub-regions of the Bay. The presentation of estimates of Bay area meeting the Chesapeake Bay Program's Benthic Community Restoration Goals, rather than Maryland estimates only, reflects improved coordination and unification of objectives among the Maryland and Virginia benthic monitoring programs. The sampling design and methods in both states are now compatible and complementary.

One of the major differences between this report and previous Level I reports is the information presented in Appendix D. Linkages between benthos and other components of the ecosystem are documented here. Some of the components affect benthos through direct exposure while watershed factors are remote and must act indirectly. The study exemplifies recent work using monitoring data to identify consequences of human activity which, strictly, is not part of the monitoring program but is complementary to it. The manuscript has been submitted for publication in *Estuaries*, a refereed scientific journal.

1.3 ORGANIZATION OF REPORT

This report is organized into five chapters and five appendices, and is supplemented by listings of data collected during the current year. Chapter 2 presents the field, laboratory, and data analysis methods used to collect, process, and evaluate LTB samples. Chapter 3 presents an assessment of trends in benthic condition at sites sampled annually by LTB in the Maryland Chesapeake Bay. Chapter 4 presents an assessment of the area of the Bay that meets the Chesapeake Bay Benthic Community Restoration Goals. Chapter 5 lists literature cited throughout the report. Appendix A amplifies information presented in Table 3-2 by providing p-values and rates of change for the 1984-1997 fixed site community attribute trend analysis. Appendix B presents B-IBI values for fixed sites in summer 1997. Appendix C presents the same information for random sites sampled in summer 1997. Appendix D is a draft manuscript submitted to *Estuaries*, which documents associative linkages between impaired benthic condition in the tidal estuary and exposure to impaired water and sediment quality, as well as anthropogenic impacts and nutrient loadings in non-tidal watersheds. Appendix E presents the results of 1985-1997 trend analysis to facilitate comparisons with other components of DNR's program.

Listings of data collected during 1997 are presented in a separate volume containing three supplements, which is not considered to be part of this report. Supplements 1 and 2 list biological and environmental data collected at fixed sites in spring and summer, respectively, while Supplement 3 lists data collected at random locations in the Maryland Chesapeake Bay during summer.

2.0 METHODS

2.1 SAMPLING DESIGN

The LTB sampling program contains two primary elements: a fixed site monitoring effort directed at identifying trends in benthic condition and a probability-based sampling effort intended to estimate the area of the Maryland Chesapeake Bay with benthic communities meeting the Chesapeake Bay Program's Benthic Community Restoration Goals (Ranasinghe et al. 1994a, updated by Weisberg et al. 1997). The sampling design for each of these elements is described below.

2.1.1 Fixed Site Sampling

The fixed site element of the program involves sampling at 27 sites, 23 of which have been sampled since the program's inception in 1984, 2 since 1989 and the last 2 since 1995 (Figure 2-1). Sites are defined by geography (within 1 km from a fixed location), and by specific depth and substrate criteria (Table 2-1).

The 1997 fixed site sampling continues trend measurements which began with the program's initiation in 1984. In the first five years of the program, from July 1984 to June 1989, 70 fixed stations were sampled 8 to 10 times per year. On each visit, three benthic samples were collected at each site and processed. Locations of the 70 fixed sites are shown in Figure 2-2.

In the second five years of the program, from July 1989 to June 1994, fixed site sampling was continued at 29 sites and a stratified random sampling element was added. Samples were collected at random from approximately 25 km^2 small areas surrounding these sites (Figure 2-3) to assess the representativeness of the fixed locations. Stratum boundaries were delineated on the basis of environmental factors that are important in controlling benthic community distributions: salinity regime, sediment type, and bottom depth (Holland et al. 1989). In addition, four new areas were established in regions of the bay targeted for management actions to abate pollution: the Patuxent River, Choptank River, and two areas in Baltimore Harbor. Each area was visited on four to six sampling cruises each year.

From July 1994 to the present, three samples were collected in spring and summer at each of the current suite of 27 sites (Table 2-1, Fig. 2-1). This sampling regime was selected as being most cost effective after analysis of the first ten years of data jointly with the Virginia benthic monitoring program (Alden et al. 1997).

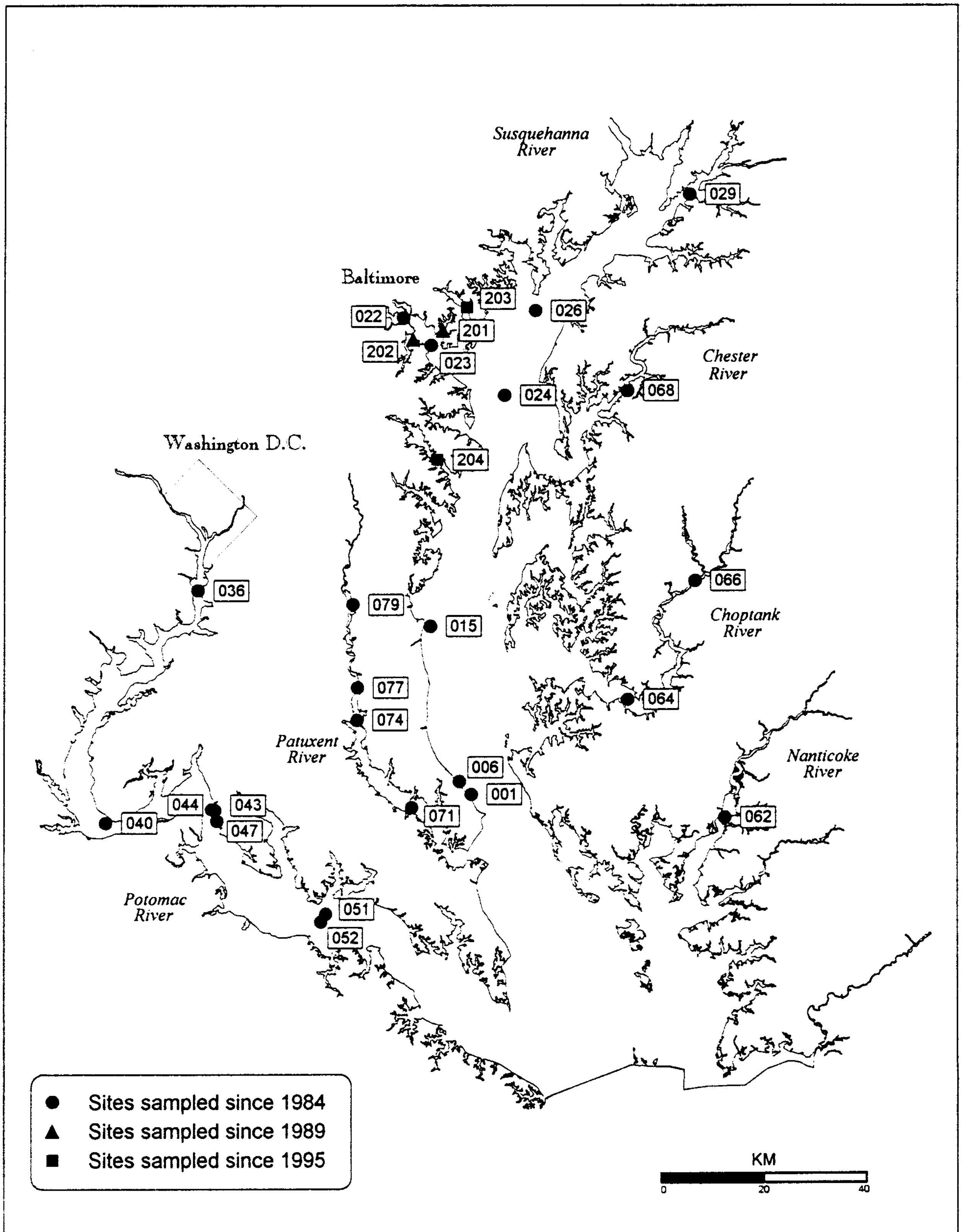

Figure 2-1. Fixed sites sampled in 1997

Table 2-1. Location, habitat, sampling gear, and habitat criteria for fixed sites sampled in and after 1995

Region	Habitat	Station	Latitude	Longitude	Sampling Gear	Habitat Criteria		
						Depth (m)	Siltclay (%)	Distance (km)
Potomac River (7)	Tidal Freshwater	036	38° 46.18'	77° 02.27'	WildCo Box Corer	<=5	>=40	1.0
	Oligohaline	040	38° 21.44'	77° 13.85'	WildCo Box Corer	6.5-10	>=80	1.0
	Shallow Low Mesohaline	043	38° 23.04'	76° 59.36'	Modified Box Corer	<=5	<=30	1.0
	Shallow Low Mesohaline	047	38° 21.90'	76° 59.10'	Modified Box Corer	<=5	<=30	0.5
	Deep Low Mesohaline	044	38° 23.13'	76° 59.76'	WildCo Box Corer	11-17	>=75	1.0
	Shallow High Mesohaline	051	38° 12.32'	76° 44.30'	Modified Box Corer	<=5	<=20	1.0
	Deep High Mesohaline	052	38° 11.53'	76° 44.88'	WildCo Box Corer	9-13	>=60	1.0
Patuxent River (4)	Tidal Freshwater	079	38° 45.02'	76° 41.36'	WildCo Box Corer	<=6	>=50	1.0
	Low Mesohaline	077	38° 36.26'	76° 40.52'	WildCo Box Corer	<=5	>=50	1.0
	Low Mesohaline	074	38° 32.83'	76° 40.51'	WildCo Box Corer	<=5	>=50	0.5
	Deep High Mesohaline	071	38° 23.70'	76° 32.95'	WildCo Box Corer	12-18	>=70	1.0

Table 2-1. Continued

Region	Habitat	Station	Latitude	Longitude	Sampling Gear	Habitat Criteria		
						Depth (m)	Siltclay (%)	Distance (km)
Baltimore Harbor and Back River (5)	Outer Harbor	023	39° 12.49'	76° 31.42'	WildCo Box Corer	4-7	>=50	1.0
	Middle Branch	022	39° 15.29'	76° 35.26'	WildCo Box Corer	2-6	>=40	1.0
	Bear Creek	201	39° 14.05'	76° 29.85'	WildCo Box Corer	2-4.5	>=70	1.0
	Curtis Bay	202	39° 13.07'	76° 33.85'	WildCo Box Corer	5-8	>=60	1.0
	Back River	203	39° 16.50'	76° 26.67'	Young-modified Van Veen	Not set	Not set	1.0
Mid-Western Tributaries (1)	Severn River	204	39° 00.40'	76° 30.30'	Young-modified Van Veen	Not set	Not set	1.0
Upper Eastern Tributaries (1)	Chester River	068	39° 07.97'	76° 04.74'	WildCo Box Corer	4-8	>=70	1.0
Choptank River (2)	Oligohaline	066	38° 48.08'	75° 55.33'	WildCo Box Corer	<=5	>=60	1.0
	Low Mesohaline	064	38° 35.42'	76° 04.18'	WildCo Box Corer	7-11	>=70	1.0
Lower Eastern Tributaries (1)	Nanticoke River	062	38° 23.03'	75° 51.02'	Petite Ponar Grab	5-8	>=75	1.0
Mainstem (6)	Oligohaline	029	39° 28.77'	75° 56.69'	WildCo Box Corer	3-7	>=40	1.0
	Low Mesohaline	026	39° 16.28'	76° 17.42'	WildCo Box Corer	2-5	>=70	1.0
	Mid-depth High Mesohaline	024	39° 07.32'	76° 21.34'	WildCo Box Corer	5-8	>=80	1.0
	Shallow High Mesohaline	015	38° 42.90'	76° 30.84'	Modified Box Corer	<=5	<=10	1.0

Table 2-1. Continued

Region	Habitat	Station	Latitude	Longitude	Sampling Gear	Habitat Criteria		
						Depth (m)	Siltclay (%)	Distance (km)
Mainstem (contd.)	Shallow Polyhaline	001	38° 25.19'	76° 25.02'	Modified Box Corer	<=5	<=20	1.0
	Shallow Polyhaline	006	38° 26.54'	76° 26.60'	Modified Box Corer	<=5	<=20	0.5

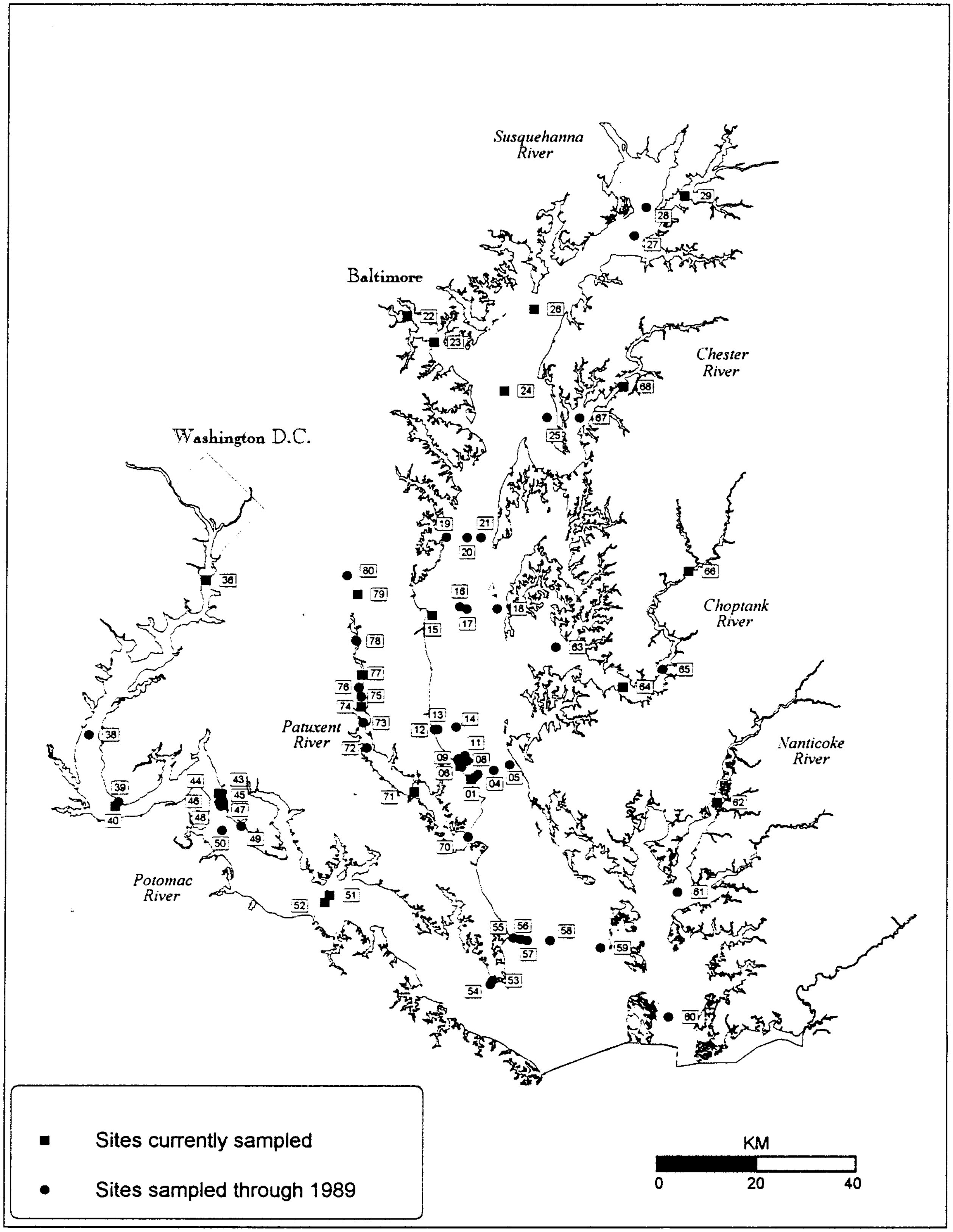

Figure 2-2. Fixed sites sampled from 1984 to 1989

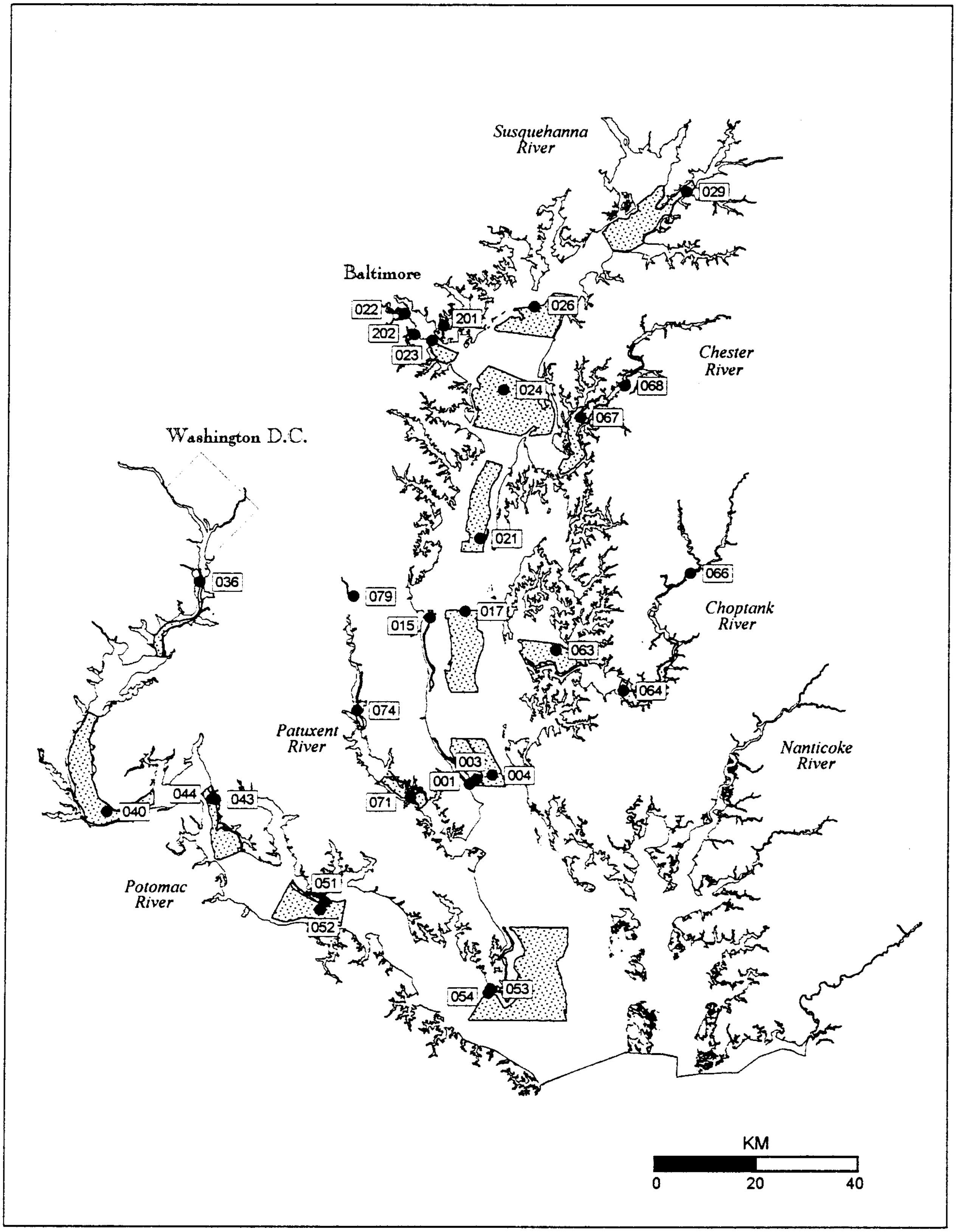

Figure 2-3. Small areas and fixed sites sampled from 1989 to 1994

2.1.2 Probability-based Sampling

The second sampling element, which was instituted in 1994, was probability-based summer sampling intended to estimate the area of the Maryland Chesapeake Bay and its tributaries that meet the Chesapeake Bay Benthic Community Restoration Goals (Ranasinghe et al. 1994a, updated by Weisberg et al. 1997). Different probability sample allocation strategies were used in 1994 than in later years. In 1994, the design was intended to estimate impaired area for the Maryland Bay and one sub-region, while in later years the design targeted five additional sub-regions.

The 1994 sample allocation scheme was designed to produce estimates for the Maryland Bay and the Potomac River. The Bay was divided into three strata with samples allocated unequally among them (Table 2-2); sampling intensity in the Potomac was increased to permit estimation of degraded area with adequate confidence, while mainstem and other tributary and embayment samples were allocated in proportion to their area.

Table 2-2. Allocation of probability-based baywide samples, 1994

Stratum	Area		Number of Samples
	km^2	%	
Maryland Mainstem (including Tangier and Pocomoke Sounds)	3611	55.5	27
Potomac River	1850	28.4	28
Other tributaries and embayments	1050	16.1	11

In subsequent years, the stratification scheme was designed to produce an annual estimate for the Maryland Bay and estimates for six subdivisions. Samples were allocated equally among strata (Figure 2-4, Table 2-3). Figure 2-5 shows the locations of the probability-based Maryland sampling sites for 1997. Regions of the Maryland mainstem deeper than 12m were not included in sampling strata because these areas are subjected to summer anoxia and have consistently been found to be abiotic.

A similar stratification scheme was used by the Commonwealth of Virginia in 1996 and 1997, permitting annual estimates for the extent of area meeting the Benthic Restoration Goals for the entire Chesapeake Bay (Table 2-3, Figure 2-6). These samples were collected and processed, and these data were analyzed by the Virginia benthic monitoring program.

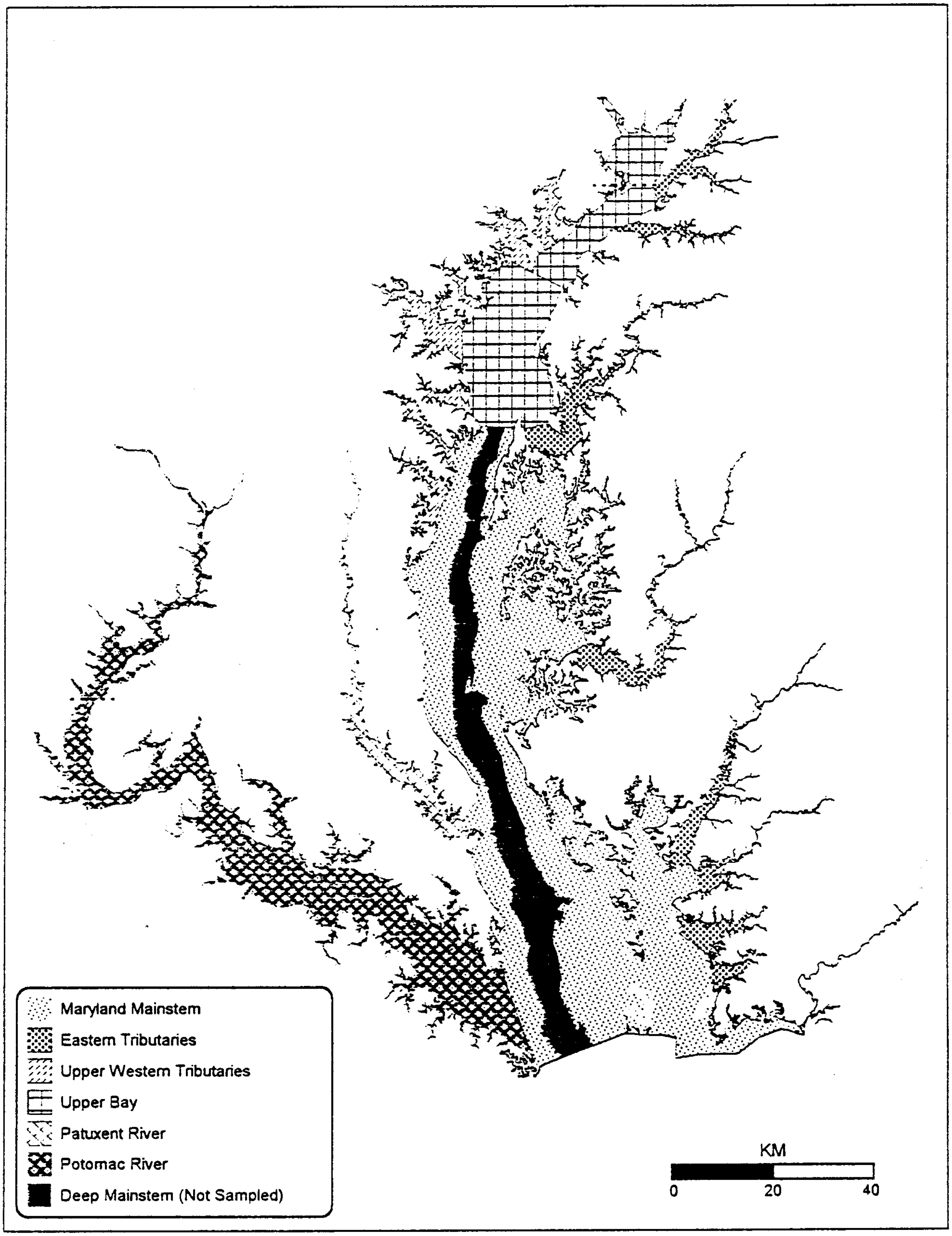

Figure 2-4. Maryland baywide sampling strata in and after 1995

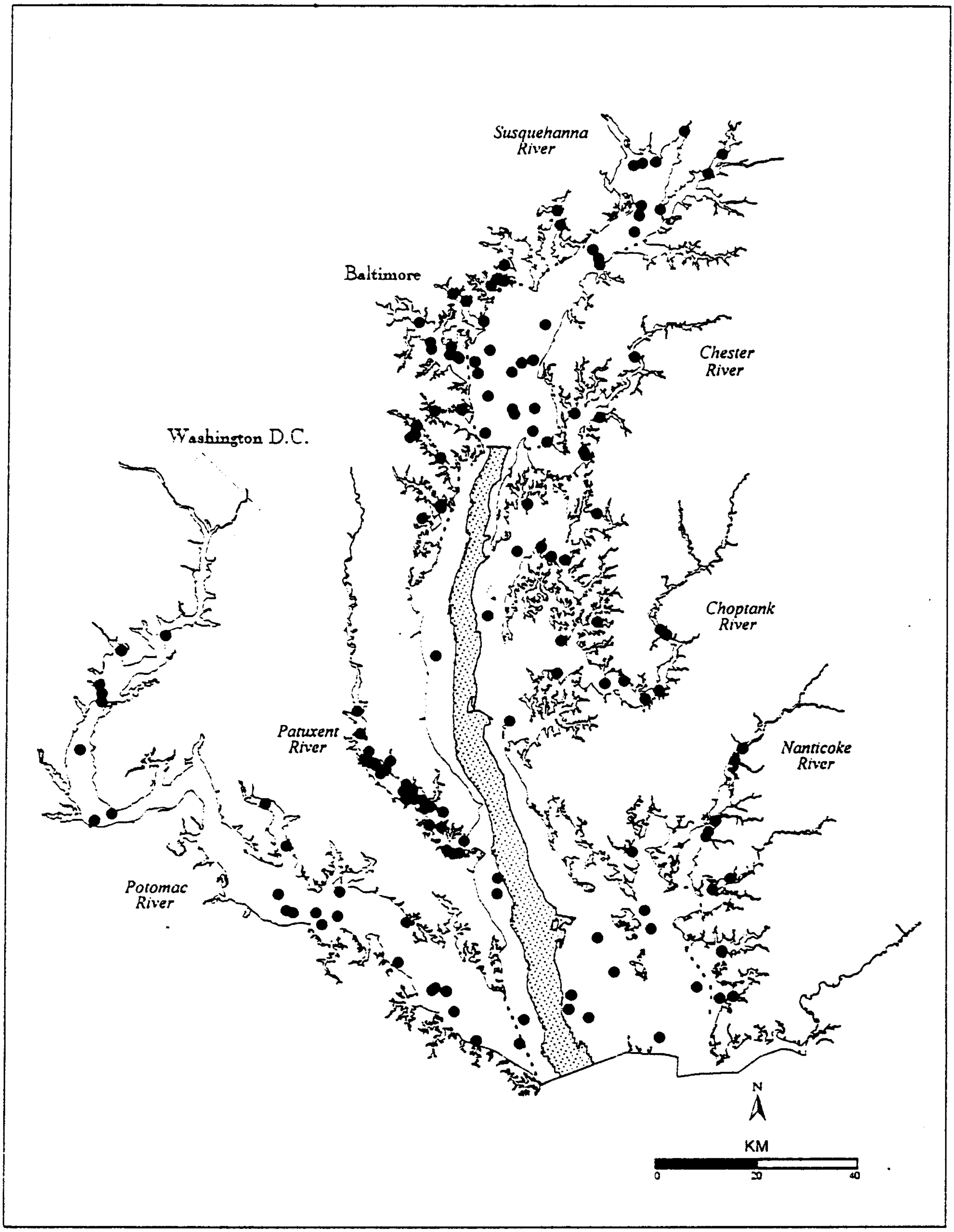

Figure 2-5. Maryland probability-based sampling sites for 1997

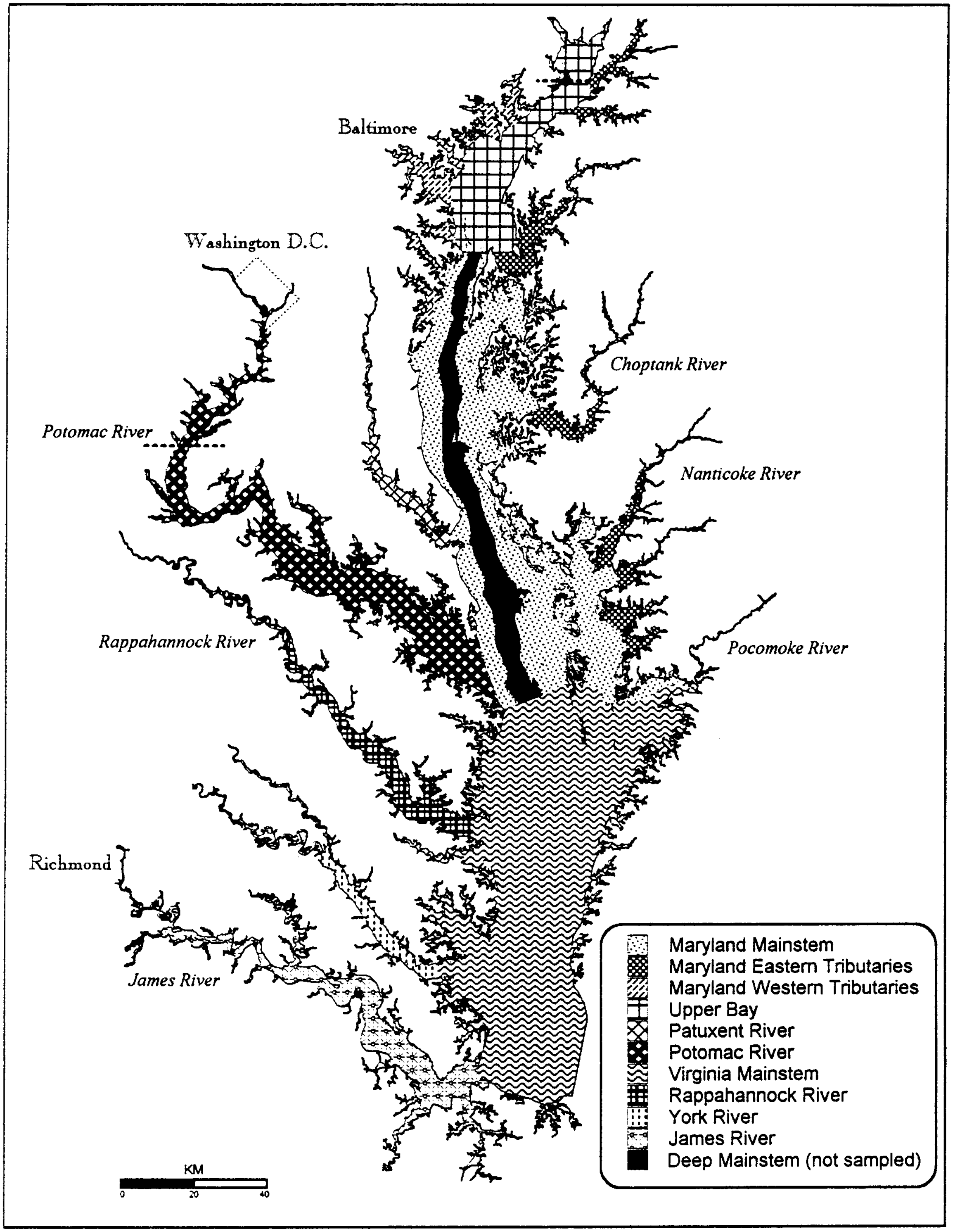

Figure 2-6. Chesapeake Bay-wide stratification scheme

Table 2-3. Allocation of probability-based baywide samples, in and after 1995. Maryland areas exclude 676 km^2 of mainstem habitat deeper than 12 m and 291 km^2 of tidal freshwater habitat. Virginia strata were sampled by the Virginia Chesapeake Bay benthic monitoring program commencing in 1996.

State	Stratum	Area			Number of Samples
		km^2	State %	Bay %	
Maryland	Mainstem	2,552	48.4	24.0	25
	Eastern Tributaries	534	10.1	5.0	25
	Western Tributaries	292	5.5	2.8	25
	Upper Bay	658	12.5	6.2	25
	Patuxent River	128	2.4	1.2	25
	Potomac River	1,112	21.1	10.5	25
	TOTAL	5,277		49.6	150
Virginia	Mainstem	4,120	76.8	38.7	25
	Rappahannock River	372	6.9	3.5	25
	York River	187	3.5	1.8	25
	James River	684	12.8	6.4	25
	TOTAL	5,363		50.4	100

2.2 SAMPLE COLLECTION

2.2.1 Station Location

From July 1984 to June 1996, stations were located using Loran-C. After June 1996 stations were located using a differential Global Positioning System. The NAD83 coordinate system was used throughout.

2.2.2 Water Column Measurements

Water column vertical profiles of temperature, conductivity, salinity, dissolved oxygen concentration (DO), oxidation reduction potential (ORP), and pH were measured at each fixed site. The profiles consisted of water quality measurements at 1 m intervals from surface to bottom at sites 7 m deep or less, and at 3 m intervals, with additional measurements at 1.5 m intervals in the vicinity of the pycnocline, at sites deeper than 7 m. Surface and bottom measurements were made at all other sampling sites. Table 2-4 lists the measurement methods used.

2.2.3 Benthic Samples

Samples were collected using four kinds of gear depending on the program element and habitat type. For the fixed site element (Table 2-1), a hand-operated box corer ("post-hole digger") which samples a 250 cm^2 area to a depth of 25 cm was used in the nearshore shallow habitats of the mainstem bay and tributaries. A Wildco box corer, which samples an area of 225 cm^2 to a depth of 23 cm, was used in deep-water (more than 4 m) habitats in the mainstem bay and tributaries. A petite ponar grab, which samples 250 cm^2 to a depth of 7 cm, was used at the fixed site in the Nanticoke River to be consistent with previous sampling in the 1980s. At the two fixed sites first sampled in 1995 and at all probability-based sampling sites, a Young grab, which samples an area of 440 cm^2 to a depth of 10 cm was used.

Sample volume and penetration depth were measured for all samples; Wildco and hand-operated box cores penetrating less than 15 cm, and Young and Petite Ponar grabs penetrating less than 7 cm into the sediment were rejected and the site was re-sampled.

In the field, samples were sieved through a 0.5-mm screen using an elutriative process. Organisms and detritus retained on the screen were transferred into labeled jars and preserved in a 10% formaldehyde solution stained with rose bengal (a vital stain that aids in separating organisms from sediments and detritus).

Two surface-sediment sub-samples of approximately 120 ml each were collected for grain-size and carbon analysis from an additional grab sample at each site. They were frozen until processed in the laboratory.

2.3 LABORATORY PROCESSING

Organisms were sorted from detritus under dissecting microscopes, identified to the lowest practical taxonomic level, and counted. Oligochaetes and chironomids were mounted on slides and examined under a compound microscope for genus and species identification.

Table 2-4. Methods used to measure water quality parameters

Parameter	Period	Method
Temperature	July 1984 to November 1984	Thermistor attached to Beckman Model RS5-3 salinometer
	December 1984 to December 1995	Thermistor attached to Hydrolab Surveyor II
	January 1996 to present	Thermistor attached to Hydrolab Datasonde 3 or Hydrolab H2O
Salinity and Conductivity	July to November 1984	Beckman Model RS5-3 salinometer toroidal conductivity cell with thermistor temperature compensation
	December 1984 to December 1995	Hydrolab Surveyor II nickel six-pin electrode-salt water cell block combination with automatic temperature compensation
	January 1996 to present	Hydrolab Datasonde 3 or Hydrolab H2O nickel six-pin electrode-salt water cell block combination with automatic temperature compensation
Dissolved Oxygen	July to November 1984	YSI Model 57 or Model 58 Oxygen Meter with automatic temperature and manual salinity compensation
	December 1984 to December 1995	Hydrolab Surveyor II membrane design probe with automatic temperature and salinity compensation
	January 1996 to present	Hydrolab Datasonde 3 or Hydrolab H2O membrane design probe with automatic temperature and salinity compensation
pH	July to November 1984	Orion analog pH meter with Ross glass combination electrode manually compensated for temperature
	December 1984 to December 1995	Hydrolab Surveyor II glass pH electrode and Lazaran reference electrode automatically compensated for temperature
	January 1996 to present	Hydrolab Datasonde 3 or Hydrolab H2O glass pH electrode and standard reference (STDREF) electrode automatically compensated for temperature
ORP	December 1984 to December 1995	Hydrolab Surveyor II platinum banded glass ORP electrode

Ash-free dry weight biomass was determined by comparable techniques during the sampling period. For samples collected from July 1984 to June 1985, biomass was directly measured using an analytical balance for major organism groups (e.g., polychaetes, molluscs, and crustaceans). Ash-free dry weight biomass was determined by drying the organisms to a constant weight at 60°C and ashing in a muffle furnace at 500°C for four hours. For samples collected between July 1985 and August 1993, a regression relationship between ash-free dry weight biomass and size of morphometric characters was defined for 22 species (Ranasinghe et al. 1993). The biomass of the 22 selected species was estimated from these regression relationships. These taxa (Table 2-5) were selected because they accounted for more than 85% of the abundance (Holland et al. 1988). After August 1993, ash-free dry weight biomass was measured directly for each species by drying the organisms to a constant weight at 60°C and ashing in a muffle furnace at 500°C for four hours.

Table 2-5. Taxa for which biomass was estimated for samples collected between 1985 and 1993

Polychaeta	**Mollusca**
Eteone heteropoda	*Acteocina canaliculata*
Glycinde solitaria	*Corbicula fluminea*
Heteromastus filiformis	*Gemma gemma*
Marenzelleria viridis	*Haminoe solitaria*
Neanthes succinea	*Macoma balthica*
Paraprionospio pinnata	*Macoma mitchelli*
Streblospio benedicti	*Mulinia lateralis*
	Mya arenaria
	Rangia cuneata
	Tagelus plebeius
Crustacea	
Cyathura polita	
Gammarus spp.	
Leptocheirus plumulosus	
Miscellaneous	
Carinoma tremaphoros	
Micrura leidyi	

Silt-clay composition and carbon content were determined for one of the two sediment sub-samples collected at each sampling site. The other sample was archived for quality assurance purposes (Scott et al. 1988). Sand and silt-clay particles were separated by wet-sieving through a 63-μm, stainless steel sieve and weighed using the procedures described by Plumb (1981) and Buchanan (1984). Carbon content of dried sediments was determined using an elemental analyzer; sediment carbon content was measured with a Perkin-Elmer Model 240B analyzer from 1984 to 1988, and an Exeter Analytical Inc., Model CE440 analyzer in and after 1995. The results from both instruments are comparable.

2.4 DATA ANALYSIS

Analyses for the fixed site and probability-based elements of LTB were both performed in the context of the Chesapeake Bay Program's Benthic Community Restoration Goals and the Benthic Index of Biotic Integrity (B-IBI) by which goal attainment is measured. The B-IBI, the Chesapeake Bay Benthic Community Restoration Goals, and statistical analysis methods for the two LTB elements are described below.

2.4.1 The B-IBI and the Chesapeake Bay Benthic Community Restoration Goals

The B-IBI is a multiple-attribute index developed to identify the degree to which a benthic assemblage meets the Chesapeake Bay Program's Benthic Community Restoration Goals (Ranasinghe et al. 1994b, updated by Weisberg et al. 1997). The B-IBI provides a means for comparing relative condition of benthic invertebrate assemblages across habitat types. It also provides a validated mechanism for integrating several benthic community attributes indicative of habitat "health" into a single number that measures overall benthic community condition.

The B-IBI is scaled from 1 to 5, and sites with values of 3 or more are considered to meet the Restoration Goals. The index is calculated by scoring each of several attributes as either 5, 3, or 1 depending on whether the value of the attribute at a site approximates, deviates slightly from, or deviates strongly from values found at the best reference sites in similar habitats, and then averaging these scores across attributes. The criteria for assigning these scores are numeric and depend on habitat. The index has not yet been developed for tidal freshwater habitats and application is presently limited to summer samples; data from habitats and time periods for which the B-IBI has not yet been developed were not used for B-IBI based assessment.

Benthic community condition was classified into four levels based on the B-IBI. Values less than or equal to 2 were classified as severely degraded; values from 2 to 2.6 were classified as degraded; values greater than 2.6 but less than 3.0 were classified as marginal; and values of 3.0 or more were classified as meeting the goal. Values in the marginal category do not meet the Restoration Goals, but they differ from the goals within the range of measurement error typically recorded between replicate samples.

2.4.2 Fixed site trend analysis

Trends in condition at the fixed sites were identified using the nonparametric technique of van Belle and Hughes (1984). This procedure is based on the Mann-Kendall statistic and consists of a sign test comparing each value with all values measured in subsequent periods. The ratio of the Mann-Kendall statistic to its variance provides a normal deviate that is tested for significance. Alpha was set to 0.1 for these tests because of the low power for trend detection for biological data. An estimate of the magnitude of each significant trend was obtained using Sen's (1968) procedure which is closely related to the Mann-Kendall test. Sen's procedure identifies the median slope among all slopes between each value and all values measured in subsequent periods.

The van Belle and Hughes procedure extends the Mann-Kendall test for use in testing for trends across multiple seasons and/or multiple strata (Gilbert 1987). Multiple-strata or multiple season tests address more global issues, such as testing for trends in the whole Potomac River, rather than a single site within the Potomac. Examining trends across multiple sites increases the power for trend detection by increasing the effective sample size. The test using combinations of sites (and/or seasons) was conducted in two parts. The first part tested for homogeneity of response across the groups to be combined. Combination is inappropriate if individual trends are significantly heterogenous (similar to the lack of validity of a two-way analysis of variance when there is a significant inter-effect interaction). In the second part, a chi-square test based on the normal deviates was used to determine the significance of the "global trend." The magnitude of the global trend was estimated by extending Sen's (1968) procedure to determine the median slope for all slopes for the multiple strata being tested (Gilbert 1987).

2.4.3 Probability-Based Estimation

The Maryland Bay was divided into three strata (Bay Mainstem, Potomac River, other tributaries and embayments) in 1994 (Table 2-2). It was divided into six strata in and after 1995 (Figure 2-4, Table 2-3). The Virginia Bay was divided into four strata in 1996 (Figure 2-6, Table 2-3).

To estimate the amount of area in the entire Bay that failed to meet the Chesapeake Bay Benthic Restoration Goals (P), we defined for every site *i* in stratum *h* a variable y_{hi} that had a value of 1 if the benthic community met the goals, and 0 otherwise. For each stratum, the estimated proportion of area meeting the goals, p_h, and its variance were calculated as the mean of the y_{hi}'s and its variance, as follows:

$$p_h = \bar{y}_h = \sum_{i=1}^{n_h} \frac{y_{hi}}{n_h}, \tag{1}$$

and

$$\mathrm{var}\,(p_h) = s_h^2 = \sum_{i=1}^{n_h} \frac{(y_{hi} - \bar{y}_h)^2}{n_h - 1}. \tag{2}$$

Estimates for strata were combined to achieve a statewide estimate as:

$$\hat{P}_{ps} = \bar{y}_{ps} = \sum_{h=1}^{6} W_h \bar{y}_h, \tag{3}$$

were the weighting factors, W_h, = A_h/A and A_h were the total area of the *h*th stratum. The variance of (3) was estimated as:

$$\mathrm{var}\ (\hat{P}_{ps}) = V(\bar{y}_{ps}) = \sum_{h=1}^{6} W_h s_h^2/n_h. \tag{4}$$

For combined strata, the 95% confidence intervals were estimated as the proportion plus or minus twice the standard error. For individual strata, (e.g., each of the 10 strata in 1996) the exact confidence interval was determined from tables.

3.0 TRENDS IN FIXED SITE BENTHIC CONDITION

Twenty-seven sites in areas targeted for pollution abatement and other management actions are monitored annually by the LTB program to assess whether benthic community condition is changing. This chapter presents B-IBI trend analysis results for 23 of these 27 sites. Trend analysis was not performed for two sites (Stations 203 and 204) because they were not sampled until 1995 and two other sites (Stations 36 and 79) because they are in tidal freshwater habitats, for which a B-IBI has yet to be developed. Fourteen-year (1984-1997) results are presented for 21 sites while only 9-year (1989-1997) results are presented for the two sites in Baltimore Harbor (Stations 201 and 202), which were first sampled in 1989. Figure 2-1 is a map of trend site locations. Our methods are described in Chapter 2.

The results presented in this chapter identify trends in benthic condition from 1984 to 1997, the duration of the Maryland benthic monitoring component. However, the Virginia benthic monitoring program and several other components of the Maryland Program did not start sampling until 1985. In order to facilitate comparisons of results across all programs, we also present results of trend analysis from 1985 to 1997 in Appendix E. Data collected in the first year of our program were not used for these analyses.

The B-IBI (Weisberg et al. 1997) is the primary measure used in trend analysis because it integrates several benthic attributes into a measure of overall condition. It provides context for interpretation of observed trends because it has been calibrated to reference conditions. For example, significant trends which result in a change of status (sites that previously met the Chesapeake Bay Restoration Goals which now fail, or vice versa) are of greater management interest than trends with rates of change that will not result in a status change soon. While we choose to emphasize trend analysis on the B-IBI because of interpretability in terms of bottom habitat condition, trends for individual attributes that comprise the B-IBI were also analyzed and are also presented here.

3.1 RESULTS

Statistically significant ($p<0.1$) B-IBI trends were detected at 13 of the 23 sites (Table 3-1). Benthic community condition declined at five of these sites (significant decreasing B-IBI trend) and improved at eight sites. Trends in community attributes that are components of the B-IBI are presented in Table 3-2 and Appendix B.

3.1.1 Declining Trends

The five sites that declined in benthic condition were all in relatively low salinity habitats. Four were located in riverine low mesohaline habitats (Stations 43, 47, 74, and 77) and the other was in the oligohaline upper Bay (Station 29). Two of the four riverine sites

Table 3-1. Trends in benthic community condition, 1984-1997. Trends in B-IBI values were identified using the Van Belle and Hughes (1984) procedure. Current conditions are based on 1995-97, and initial condition on 1984-86 mean B-IBI values (shown in parentheses). ns: not significant; NA: not applicable; [a]: 1989-1997 trends.

Station	Trend Significance	Median Slope (B-IBI units/yr)	1984-1986 (Initial Condition)	1995-1997 (Current Condition)
Potomac River				
040	ns	0.00	Meets Goal (3.23)	Marginal (2.91)
043	$p < 0.01$	-0.00	Meets Goal (3.80)	Meets Goal (3.67)
044	$p < 0.01$	0.04	Degraded (2.52)	Degraded (2.56)
047	$p < 0.001$	-0.04	Meets Goal (4.05)	Meets Goal (3.67)
051	ns	0.00	Marginal (2.81)	Marginal (2.93)
052	ns	0.00	Severely Degraded (1.33)	Severely Degraded (1.26)
Patuxent River				
071	ns	0.00	Degraded (2.16)	Degraded (2.22)
074	$p < 0.05$	-0.04	Meets Goal (3.72)	Meets Goal (3.44)
077	$p < 0.01$	-0.10	Meets Goal (3.49)	Degraded (2.20)
Choptank River				
064	$p < 0.001$	0.13	Degraded (2.52)	Meets Goal (3.93)
066	ns	0.00	Degraded (2.57)	Degraded (2.20)
Maryland Mainstem				
026	$p < 0.01$	0.04	Meets Goal (3.13)	Meets Goal (3.53)
024	$p < 0.01$	0.10	Marginal (2.77)	Meets Goal (3.74)
015	ns	0.00	Marginal (2.65)	Marginal (2.81)
006	$p < 0.01$	0.08	Degraded (2.54)	Meets Goal (3.48)
001	$p < 0.1$	0.03	Marginal (2.98)	Meets Goal (3.44)
Maryland Western Shore Tributaries				
022	$p < 0.01$	0.09	Degraded (2.29)	Meets Goal (3.27)
023	$p < 0.05$	0.07	Degraded (2.36)	Meets Goal (3.00)
201	ns[a]	0.00	NA	Severely Degraded (1.31)
202	ns[a]	0.00	NA	Severely Degraded (1.13)
Maryland Eastern Shore Tributaries				
029	$p < 0.1$	-0.04	Meets Goal (3.02)	Degraded (2.47)
062	ns	0.00	Meets Goal (3.40)	Meets Goal (3.53)
068	ns	0.00	Meets Goal (3.36)	Meets Goal (3.67)

Table 3-2. Summer temporal trends in benthic community attributes 1984-97. Monotonic trends were identified using the Van Belle and Hughes (1984) procedure. ⇑: Increasing trend; ⇓: Decreasing trend. +: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; shaded trend cells indicate increasing degradation; unshaded trend cells indicate improving conditions. [a]: trends based on 1989-1997 data; [b]: attribute trend based on 1990-1997 data. Additional detail is provided in Appendix A.

Station	B-IBI	Abundance	Biomass	Shannon Diversity	Pollution Indicative Taxon Abundance	Pollution Sensitive Taxon Abundance	Pollution Indicative Taxon Biomass[b]	Pollution Sensitive Taxon Biomass[b]
Potomac River								
036	NA	⇓ +	⇓ **	⇑ **	⇑ **		⇑ +	
040					⇑ **			
043	⇓ **		⇑ *		⇑ **			
044	⇑ +		⇑ *	⇑ **	⇑ **	⇑ *		
047	⇓ **		⇑ **		⇑ **			
051			⇓ **	⇑ +				
052								
Patuxent River								
079	NA		⇑ **	⇑ **	⇑ **		⇓ **	
077	⇓ **	⇑ **			⇑ **	⇓ *		
074	⇓ *	⇑ **			⇑ **		⇑ +	
071				⇑ *	⇓ **		⇓ *	
Choptank River								
066					⇑ **			
064	⇑ **	⇑ **	⇑ **	⇑ +		⇑ **		
Maryland Mainstem								
026	⇑ **	⇑ +	⇑ **			⇑ **		
024	⇑ **	⇑ **						
015		⇑ +						
006	⇑ **	⇑ +				⇑ **		⇑ *
001	⇑ +	⇑ **				⇑ *		
Maryland Western Shore Tributaries								
022	⇑ **	⇑ **	⇑ **	⇑ *		⇑ **	⇓ *	⇑ +
023	⇑ *	⇓ *		⇑ **		⇑ **		
201 [a]					⇓ +			
202 [a]								
Maryland Eastern Shore Tributaries								
029	⇓ +	⇑ *			⇑ **	⇑ **	⇑ *	⇓ **
068		⇑ +	⇑ **	⇑ *		⇑ *		
062		⇑ **	⇑ +		⇓ *			

(Stations 43 and 47) were located in the Potomac River near Morgantown, while the other two (Stations 74 and 77) were located in the Patuxent River near Chalk Point. Stations 47 and 74 are within thermal impact areas of the Morgantown and Chalk Point Steam Electric Facilities, respectively.

The declining sites varied in benthic condition. Two sites met the Restoration Goal and all samples collected here have done so continuously since the program began. A third currently meets the Restoration Goal, but the number of samples failing has increased over time. The other two sites previously met the Restoration Goal but now support degraded benthic communities.

Overall, Stations 43 and 47 in the Potomac River met the Benthic Community Restoration Goals (Table 3-1) although some individual attributes failed (Table 3-3). B-IBI values, though declining, are well above the Goal and the rate of decline is small (Table 3-1). Not a single sample from either site has ever failed the B-IBI (Table 3-3).

On the average, Station 74 in the Patuxent River also currently meets the Restoration Goal (Table 3-1) but the number of samples failing the Goal has increased (Table 3-3). The proportion of failing samples increased steadily from none before 1987, to 20% from 1988-92, to almost 40% from 1993-97 (Table 3-3). In our last report (Ranasinghe et al. 1997), this station was identified as being at risk of failing the Restoration Goal based on marginal status and strong significantly declining trend. However, in 1997 the B-IBI score was exceptionally high (4.2), reducing the median rate of decline from 0.07 to 0.04 B-IBI units per year. Therefore, the risk of imminent failure has been reduced.

Station 29 in the Upper Bay and Station 77 in the Patuxent River initially met the Restoration Goal, but currently support degraded benthic communities (Table 3-1). The proportion of samples failing the Goal increased from 12% to 89% at Station 77 and from 52% to 67% at Station 29 (Table 3-3).

At all five sites, declines were due to increases in excess abundance or excess biomass (Table 3-2), both of which are symptoms of eutrophic conditions. At all these sites, increasing rates of abundance and biomass failure were associated with the declining trend (Table 3-3) and, in every case, failure was due to excess, rather than insufficient, abundance or biomass (Table 3-2, Appendix B). Increases in abundance and biomass above reference conditions are often associated with organic enrichment (e.g., Pearson and Rosenberg 1978, Weisberg et al. 1997). Increasing trends in biomass were detected at Stations 43 and 47, and increasing trends in abundance were detected at Stations 29, 74 and 77 (Table 3-2). Increasing trends in abundance of pollution indicative taxa were detected at all five sites (Table 3-2). The improvement at Station 74 between 1996 and 1997 referred to above was mainly due to a substantial decrease in total abundance from "excessive" to "acceptable" levels. None of these five sites are located in areas subject to summer hypoxia or anoxia. The observed declines are probably due to eutrophication.

Table 3-3. Percentages of samples failing the Restoration Goal over three time periods at sites with declining benthic community condition.

Attribute	Samples Failing Restoration Goal (%)		
	1984-1987	1988-1992	1993-1997
Station 29			
n	21	10	12
Total abundance	14.3	10.0	33.3
Total biomass	47.6	100.0	75.0
Shannon-Wiener Index	66.7	60.0	91.7
Abundance of pollution indicative taxa	0.0	0.0	33.3
Abundance of pollution sensitive taxa	19.1	0.0	25.0
B-IBI	52.4	60.0	66.7
Station 43			
n	24	9	16
Total abundance	8.3	0.0	25.0
Total biomass	33.3	77.8	50.0
Shannon-Wiener Index	0.0	0.0	0.0
Abundance of pollution indicative taxa	0.0	11.1	0.0
B-IBI	0.0	0.0	0.0
Station 47			
n	24	6	9
Total abundance	8.3	0.0	44.4
Total biomass	16.7	66.7	66.7
Shannon-Wiener Index	0.0	0.0	0.0
Abundance of pollution indicative taxa	0.0	0.0	0.0
B-IBI	0.0	0.0	0.0
Station 74			
n	24	10	13
Total abundance	0.0	10.0	38.5
Total biomass	33.3	70.0	46.2
Shannon-Wiener Index	8.3	10.0	7.7
Abundance of pollution indicative taxa	8.3	20.0	38.5
B-IBI	0.0	20.0	38.5
Station 77			
n	24	7	9
Total abundance	8.3	14.3	0.0
Total biomass	37.5	85.7	100.0
Shannon-Wiener Index	12.5	85.7	33.3
Abundance of pollution indicative taxa	0.0	42.9	100.0
B-IBI	12.5	42.9	88.9

3.1.2 Improving Trends

The eight improving sites were distributed throughout the Maryland Bay (Table 3-1) and generally were in more saline habitats than the declining sites. They were located in the upper Bay (Stations 24 and 26), in mainstem nearshore sand habitats near Calvert Cliffs (Stations 1 and 6), in Baltimore Harbor (Stations 22 and 23), in the deep portion of the Potomac River near Morgantown (Station 44), and in the Choptank River (Station 64). Four of the eight sites (Stations 22, 23, 26, and 44) were located in low mesohaline habitats and the other four (Stations 1, 6, 24 and 64) were in high mesohaline habitats.

Five of the eight sites (Stations 22, 23, 24, 44, and 64) are located in areas historically exposed to summer hypoxia and anoxia. Their improving trends may be due to reduced exposure to oxygen stress which could be due to decreases in the extent, intensity, duration, or frequency of low dissolved oxygen episodes in the Bay which are not measurable by our single point in time summer dissolved oxygen measurements. Stations 22 and 23 are in Baltimore Harbor, a region containing many areas with high concentrations of chemical contaminants in bottom sediments (Ranasinghe et al. 1994b) as well being subject annually to summer oxygen stress.

Seven of the improving sites currently meet the Benthic Community Restoration Goals (Table 3-1). Station 26 in the upper Bay met the Goal throughout, while Stations 1, 6, 22, 23, 24, and 64 improved over time from initially degraded conditions (Table 3-1). The eighth, Potomac River Station 44, on average does not meet the Restoration Goal (Table 3-1), but the number of samples failing decreased from 50% from 1984-87 to about 30% from 1993-97 (Table 3-4).

The sharp improvements observed at most of these sites are consistent with reduced exposure to low oxygen conditions. Dramatic decreases in the proportions of samples failing the Restoration Goals occurred at Stations 22, 23, 24, and 64 (Table 3-4). For example, at Station 24, all 1993-97 samples met the Restoration Goal although half the 1984-87 samples failed. All 1993-97 samples met the Goal at Station 64, too, although about 70% of the samples failed in 1984-87 and 1988-92. At Stations 22 and 23 in Baltimore Harbor, more than 80% of the samples failed from 1984-87 while less than 50% of the samples failed from 1993-97 (Table 3-4).

The changes in proportions of B-IBI attributes meeting the Restoration Goal at these stations is also consistent with reduced low oxygen stress. The increases in abundance, biomass, and diversity (Tables 3-2 and 3-4) from low to acceptable levels were observed at most stations. Abundance and/or biomass increased from depauperate to reference levels at Stations 22, 23, 24, 44 and 64 (Tables 3-2, 3-4). The Shannon-Wiener index increased to acceptable levels at Stations 22, 23, 26, 44, and 64. Abundances of species sensitive to pollution also increased at all stations except Station 24 (Table 3-2). At Station 22, in the Middle Branch of Baltimore Harbor, we observed improving trends in four of five attributes contributing to the B-IBI (Table 3-2).

Insufficient oxygen time series data are available to support or refute our hypothesis than oxygen is the cause of these improvements. The benthos may be a more sensitive indicator of these conditions because they are constantly and continuously *in situ*, whereas instrument measurements are made infrequently and at single points in time.

Table 3-4. Percentages of samples failing the Restoration Goal over three time periods at sites with improving benthic community condition.

Attribute	Samples Failing Restoration Goal (%)		
	1984-1987	1988-1992	1993-1997
Station 1			
n	21	14	13
Total abundance	42.9	0.0	0.0
Total biomass	28.6	50.0	23.1
Shannon-Wiener Index	23.8	28.6	23.1
Abundance of pollution indicative taxa	19.1	21.4	23.1
Abundance of pollution sensitive taxa	42.9	42.9	15.4
B-IBI	23.8	35.7	23.1
Station 6			
n	21	7	9
Total abundance	90.5	42.9	33.3
Total biomass	52.4	100.0	44.4
Shannon-Wiener Index	42.9	71.4	22.2
Abundance of pollution indicative taxa	28.6	85.7	0.0
Abundance of pollution sensitive taxa	42.9	14.3	0.0
B-IBI	66.7	100.0	11.1
Station 22			
n	21	15	17
Total abundance	9.5	26.7	23.5
Total biomass	100.0	66.7	35.3
Shannon-Wiener Index	42.9	26.7	23.5
Abundance of pollution indicative taxa	61.9	85.7	88.2
B-IBI	81.0	60.0	47.1
Station 23			
n	18	10	16
Total abundance	0.0	20.0	6.3
Total biomass	33.3	60.0	31.3
Shannon-Wiener Index	44.4	20.0	18.8
Abundance of pollution indicative taxa	61.1	50.0	37.5
B-IBI	88.9	60.0	43.8
Station 24			
n	21	10	14
Total abundance	47.6	20.0	0.0
Total biomass	50.0	50.0	0.0
Shannon-Wiener Index	23.8	30.0	7.1
Abundance of pollution indicative taxa	0.0	0.0	0.0
B-IBI	47.6	30.0	0.0

Table 3-4. (Continued)

Attribute	Samples Failing Restoration Goal (%) 1984-1987	1988-1992	1993-1997
Station 26			
n	21	10	12
Total abundance	9.5	0.0	0.0
Total biomass	66.7	90.0	75.0
Shannon-Wiener Index	52.4	10.0	25.0
Abundance of pollution indicative taxa	0.0	10.0	0.0
B-IBI	19.1	0.0	0.0
Station 44			
n	24	9	13
Total abundance	25.0	33.3	15.4
Total biomass	50.0	22.2	23.1
Shannon-Wiener Index	70.8	55.6	23.1
Abundance of pollution indicative taxa	33.3	67.7	53.9
B-IBI	50.0	33.3	30.8
Station 64			
n	18	11	14
Total abundance	33.3	45.5	21.4
Total biomass	16.7	9.1	0.0
Shannon-Wiener Index	77.8	63.6	35.5
Abundance of pollution indicative taxa	50.0	71.4	
B-IBI	66.7	72.7	0.0

4.0 BAYWIDE BOTTOM COMMUNITY CONDITION

4.1 INTRODUCTION

The fixed site monitoring presented in Chapter 3 provides useful information about trends in the condition of benthic biological resources at 23 locations in the Maryland bay but it does not provide an integrated assessment of the Bay's overall condition. The fixed sites were selected for trend monitoring because they are located in areas subject to management action and, therefore, are likely to undergo change. Because these sites were selected subjectively, there is no objective way of weighting them to obtain an unbiased estimate of Maryland Bay-wide status.

An alternative approach for quantifying status of the bay, which was first adopted in the 1994 sampling program, is to use probability-based sampling to estimate the bottom area populated by benthos meeting the Chesapeake Bay Benthic Community Restoration Goals. Where the fixed site approach emphasizes quantifying change at selected locations, the probability sampling approach emphasizes quantifying the spatial extent of problems. While both approaches are valuable, developing and assessing the effectiveness of a Chesapeake Bay management strategy requires understanding the extent and distribution of problems throughout the Bay, instead of assessing only site-specific problems. Our probability-based sampling element is intended to provide that information, as well as a more widespread baseline data set for assessing the effects of unanticipated future contamination (e.g., oil or hazardous waste spills).

Probability-based sampling has been employed previously by LTB, but the sampled area included only 16% of the Maryland Bay (Ranasinghe et al. 1994a) which is insufficient to characterize the entire Bay. Probability-based sampling was also used in the Maryland Bay by the U.S. EPA's Environmental Monitoring and Assessment Program (EMAP), but at a sampling density too low to develop precise condition estimates for the Maryland Bay. The 1994-1997 sampling represents the first efforts to develop area-based Maryland Bay-wide bottom condition statements.

This year, for the second time, estimates of tidal bottom area meeting the Benthic Restoration Goals are included for the entire Chesapeake Bay. The estimates are possible because the Virginia benthic monitoring program also included a probability-based sampling element, which started in 1996. The Virginia sampling is compatible and complementary to the Maryland effort and is part of a joint effort by the two programs to assess the extent of "healthy" tidal bottom Baywide.

This chapter presents the results of the 1997 Maryland and Virginia tidal Chesapeake Bay-wide probability-based sampling and adds a fourth year of results to LTB's tidal Maryland-wide time series. The analytical methods for estimating the areal extent of Bay bottom meeting the Restoration Goals were presented in Chapter 2.

4.2 RESULTS

Of the 150 Maryland samples collected with the probability-based design in 1997, 84 met and 59 failed the Chesapeake Bay Benthic Community Restoration Goals (Figure 4-1). Seven samples were not evaluated because they were collected in tidal freshwater, for which Restoration Goals have not yet been established. Of the 250 probability samples collected in Chesapeake Bay in 1997, 129 met and 111 failed the Restoration Goal; 10 could not be evaluated. The Virginia sampling results are presented in Figure 4-2.

Consistent improvements in Maryland baywide condition, though within the uncertainty margins of the estimates, were observed each year (Fig. 4-3). Results from the individual sites were weighted based on the stratified sampling design to estimate tidal Maryland area failing the Restoration Goal. In 1997, 56% (±11%) of the Maryland Bay was estimated to fail the Restoration Goal, compared with 59% (±12%) in 1996, 60% (±11%) and 68% (±13%) in 1994. Expressed as areas, 3,354 ±655, 3,500 ±679, 3,567 ±667, and 4,445 ±768 km^2 of the tidal Maryland Chesapeake Bay remained to be restored in 1997, 1996, 1995, and 1994, respectively.

The Potomac River and the mid-Bay mainstem were in poorest condition of the six Maryland strata, while the eastern tributaries, upper Bay, and the Patuxent River were in the best condition (Figure 4-4). From 1994-1997, more than 60% of the Potomac River (688-822 km^2) failed the Restoration Goals each year and more than two-thirds of that area (about half the Potomac River bottom) was severely degraded (Table 4-1). The Maryland mainstem had the largest amount of degraded area (>2,000 km^2) and more than two-thirds of that area was severely degraded. In contrast, at least 52% of the area in the eastern shore tributaries met the Restoration Goals each year and less than 20% of the eastern tributary bottom area was severely degraded.

Although the Baywide estimate of area failing the Restoration Goal increased from 1996 to 1997, the increase was within the margin of uncertainty of the estimate (Fig. 4-5). An estimate of 50% (±10%) or 5,673 ±1,084 km^2 of the tidal Chesapeake Bay failing the Restoration Goal in 1997 was calculated by weighting results from the 250 probability sites in Maryland and Virginia for 1997. Comparable values for 1996 were 47% (±9%) and 5,331 ±1,066 km^2.

Baywide, the Maryland mid-Bay mainstem and the Potomac, York, and Rappahannock Rivers were in the worst condition both for 1997 (Figure 4-4) and over the 1994-1997 study period (Figure 4-6, Table 4-1). Over 60% of each failed the Restoration Goals each year. On average, the York River had over 70% degraded bottom, but only a relatively small area (24-26%) was severely degraded. The upper Bay, Patuxent River, and the Maryland eastern tributaries were in the best condition in 1997 (Figure 4-4), while the James River, lower mainstem, upper Bay, and Maryland eastern tributaries were in the best condition overall, averaging less than 40% degraded area over the study period (Figure 4-6, Table 4-1). Only one severely degraded sample was collected from the lower mainstem each year.

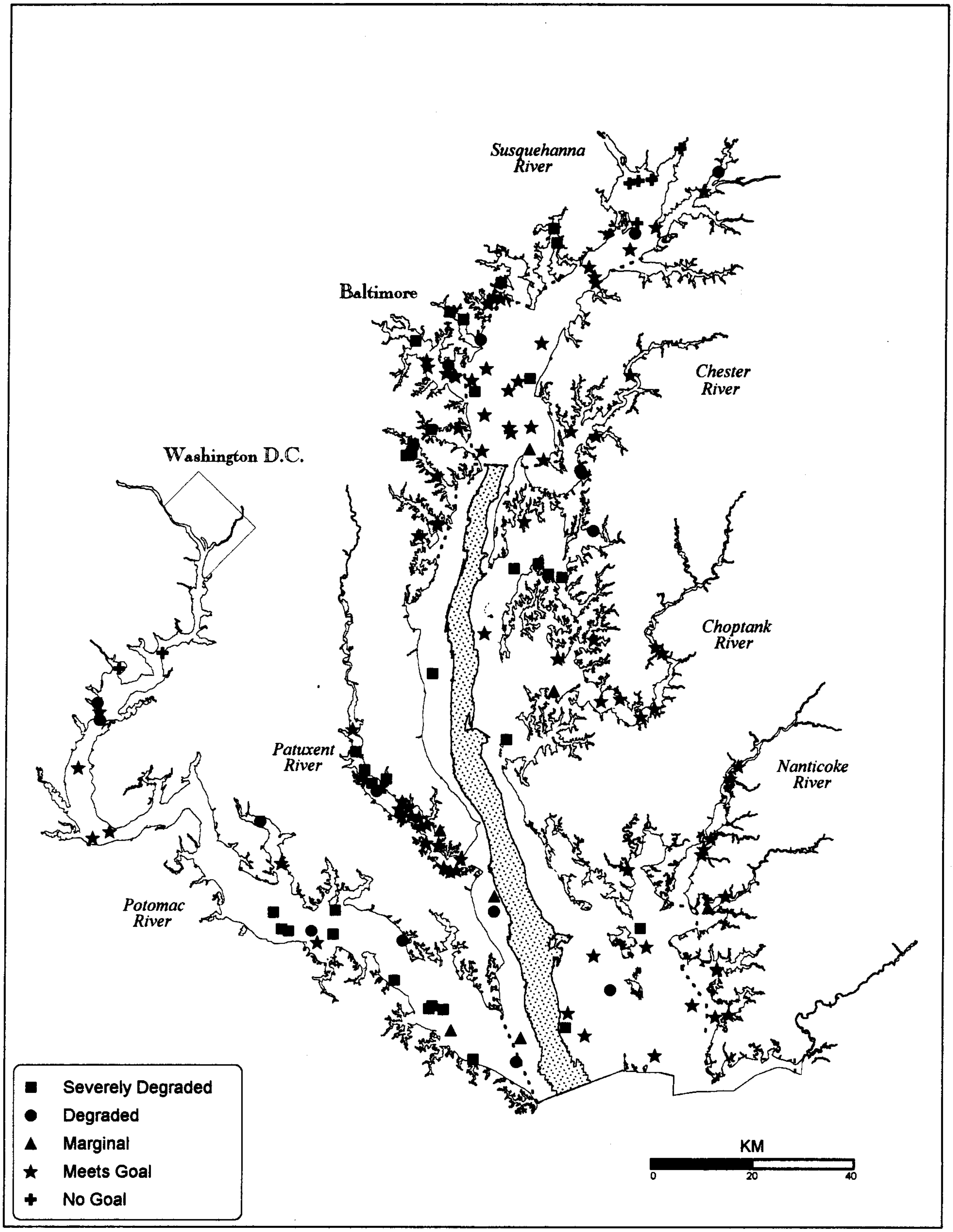

Figure 4-1. Results of probability-based benthic sampling of the Maryland Chesapeake Bay and its tidal tributaries in 1997. Each sample was evaluated in context of the Chesapeake Bay Benthic Community Restoration Goals.

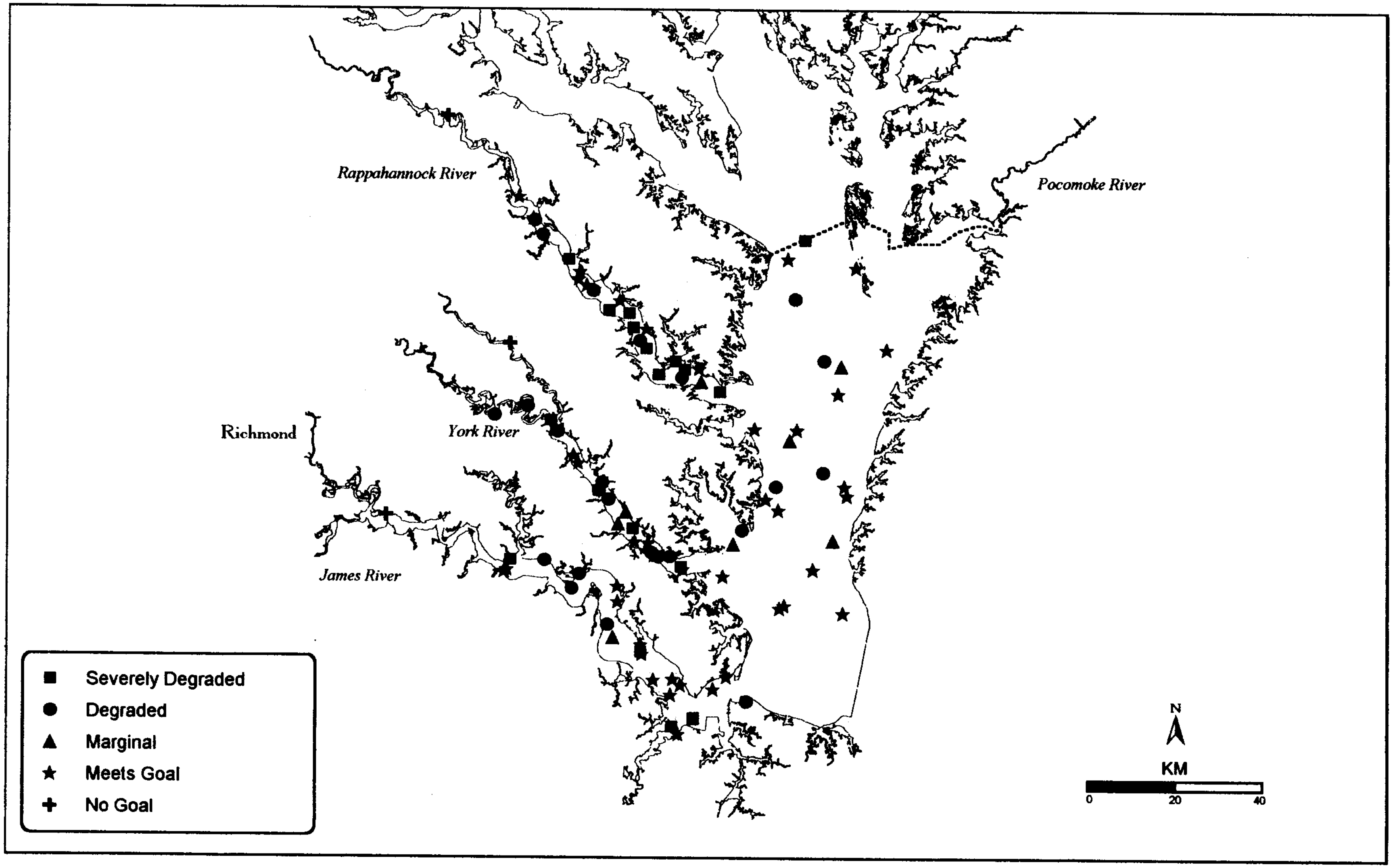

Figure 4-2. Results of probability-based benthic sampling of the Virginia Chesapeake Bay and its tidal tributaries in 1997. Each sample was evaluated in context of the Chesapeake Bay Benthic Community Restoration Goals.

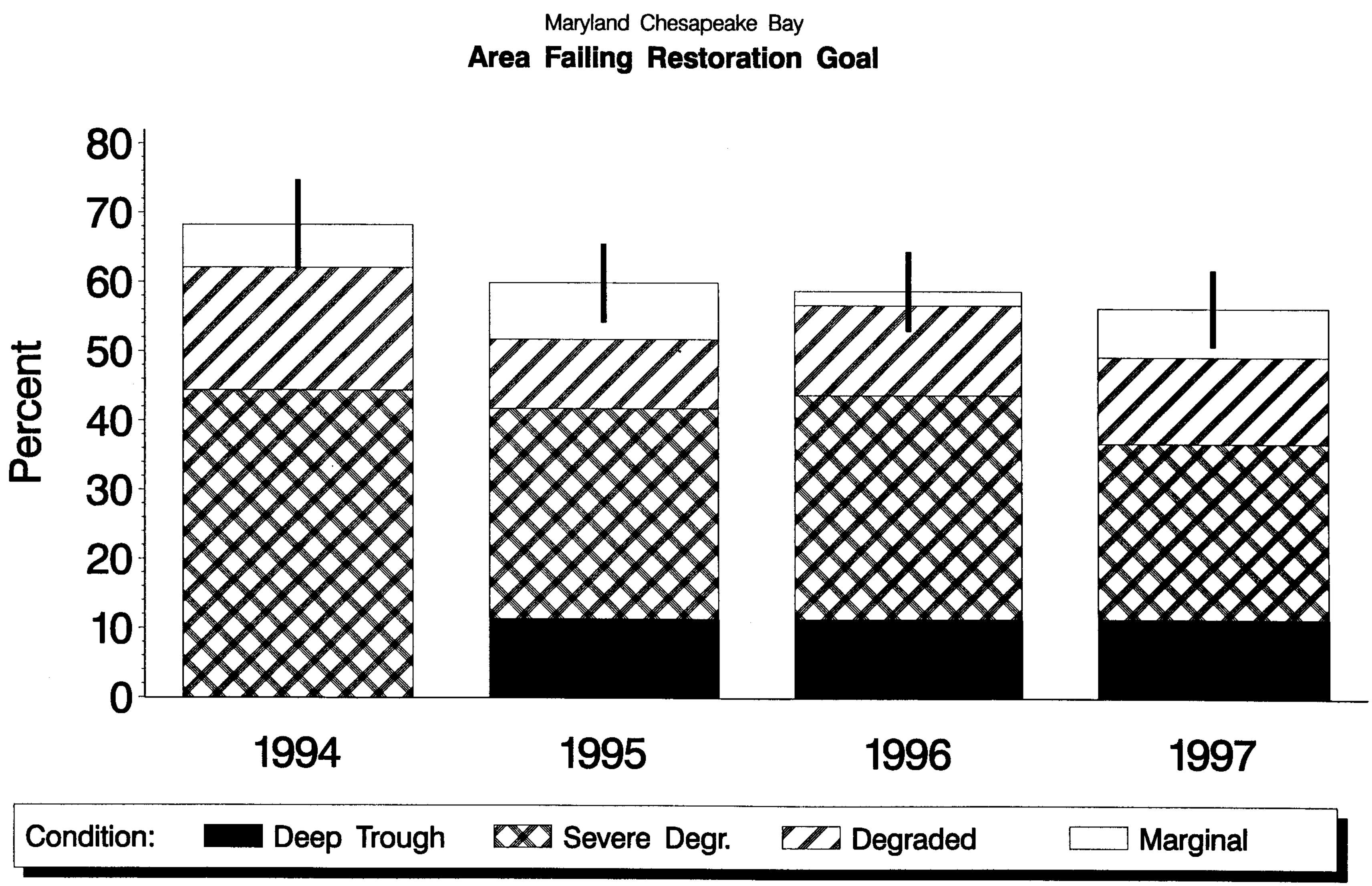

Figure 4-3. Proportion of the Maryland Bay failing the Chesapeake Bay Benthic Community Restoration Goals from 1994 to 1997. The error bars indicate ±1 standard error.

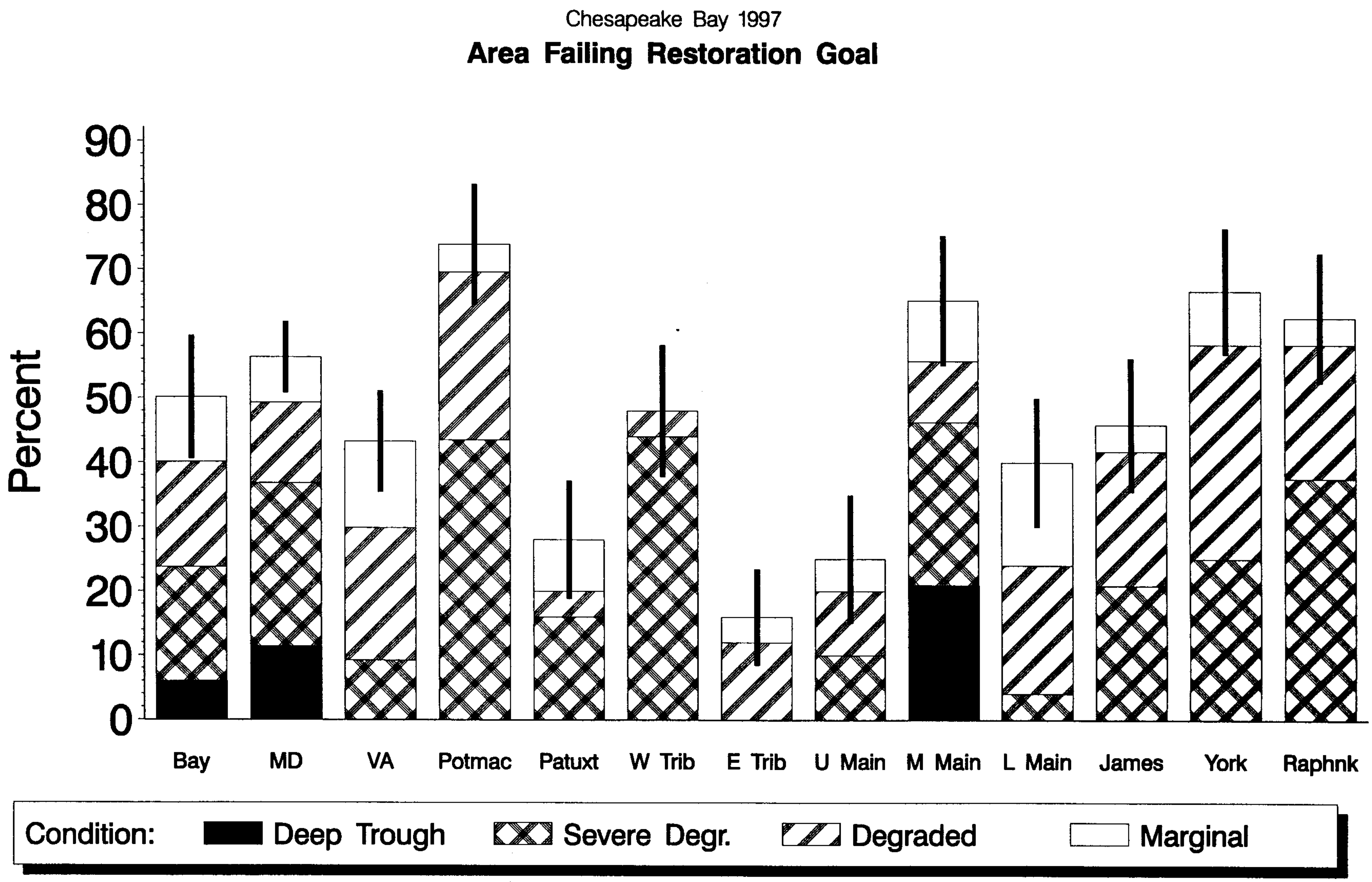

Figure 4-4. Proportion of the Chesapeake Bay, tidal Maryland and Virginia Bay, and sampling strata failing the Chesapeake Bay Benthic Community Restoration Goals in 1997. The error bars indicate ±1 standard error.

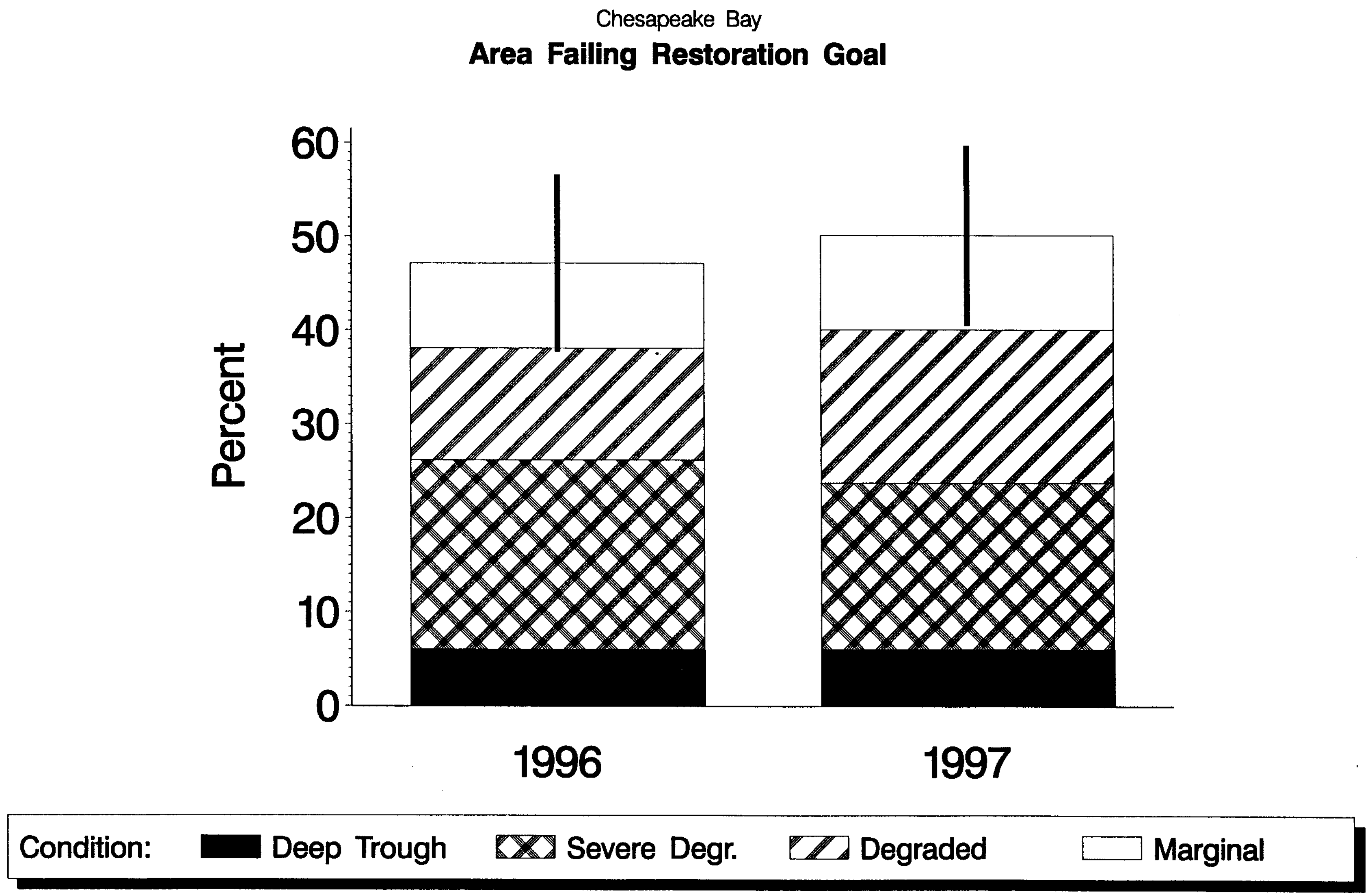

Figure 4-5. Proportion of the Chesapeake Bay failing the Chesapeake Bay Benthic Community Restoration Goals in 1996 and 1997. The error bars indicate ±1 standard error.

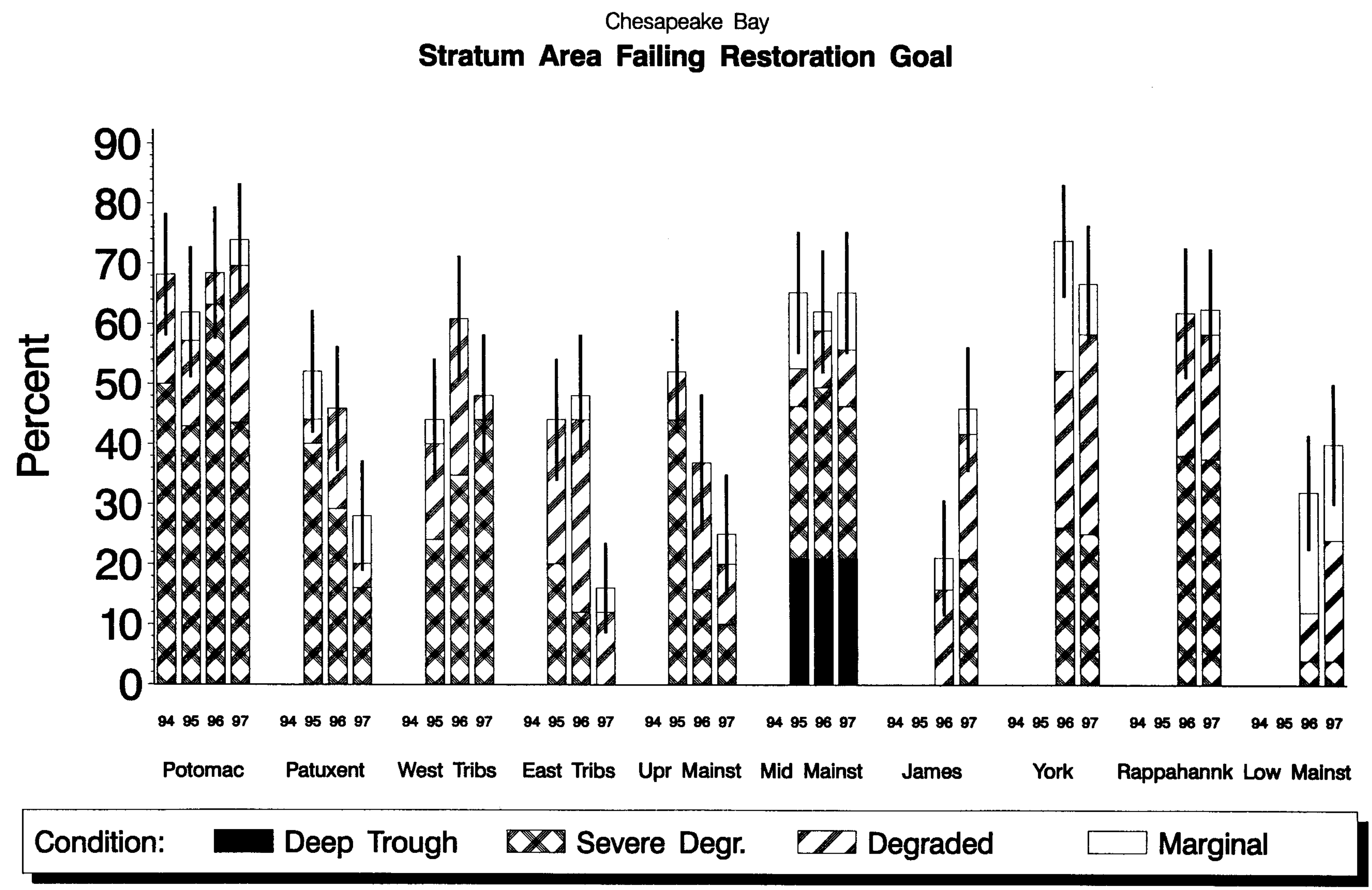

Figure 4-6. Proportion of the sampling strata failing the Chesapeake Bay Benthic Community Restoration Goals. The error bars indicate ±1 standard error.

Table 4-1. Estimated tidal area (km^2) failing to meet the Chesapeake Bay Benthic Community Restoration Goals in the Chesapeake Bay, Maryland, Virginia, and each of the ten sampling strata.

Region	Year	Severely Degraded	Degraded	Marginal
Chesapeake Bay	1996	2,960	1,347	1,024
	1997	2,685	1,849	1,138
Maryland	1994	2,892	1,147	406
	1995	2,487	596	483
	1996	2,604	772	123
	1997	2,191	743	419
Virginia	1996	355	575	901
	1997	494	1,106	719
Potomac River	1994	925	336	0
	1995	477	159	53
	1996	702	59	0
	1997	484	290	48
Patuxent River	1995	51	5	10
	1996	37	21	0
	1997	20	5	10
Upper Western (MD) Tributaries	1995	70	47	12
	1996	102	76	0
	1997	129	12	0
Upper Eastern (MD) Tributaries	1995	107	128	0
	1996	64	171	21
	1997	0	64	21
Upper Mainstem Bay	1995	290	53	0
	1996	104	139	0
	1997	66	66	33

Table 4-1. Cont'd

Region	Year	Severely Degraded	Degraded	Marginal
Middle Mainstem Bay	1995	1,493	204	408
	1996	1,595	306	102
	1997	1,493	306	306
Lower Mainstem (VA)	1996	165	330	824
	1997	165	824	659
Rappahannock River	1996	142	89	142
	1997	140	78	16
York River	1996	49	49	41
	1997	47	62	16
James River	1996	0	108	36
	1997	142	142	28

4.3 DISCUSSION

Although about half the Chesapeake Bay and half the Maryland Bay failed the Chesapeake Bay Benthic Community Restoration Goals, much of this area had B-IBI values greater than two, indicating only mild degradation that should respond quickly to moderate improvements in water quality. Nearly half the degraded Chesapeake Bay bottom (2,371 km^2 in 1996 and 2,987 km^2 in 1997) and about a third of the degraded Maryland Bay bottom (895-1,160 km^2 from 1994-1997) were only slightly impaired. Of the additional 2,500 km^2 supporting severely degraded benthic communities, only 680 km^2 were located in the deep (> 12m) waters that are perennially anoxic and probably beyond the scope of present mitigation efforts. Future LTB efforts will involve coordination with the Chesapeake Bay Program oxygen mapping efforts to assess which of the remaining 1870 km^2 of severely degraded benthos were located in areas of periodic hypoxia, and which are located in areas that the Chesapeake Bay modeling efforts predict are likely to improve in response to nutrient reduction efforts.

The estimates of degraded area for regions measured in multiple years were generally similar between years, with most estimates included within the confidence interval of other years (Figure 4-6). The spatial patterns of degradation were also similar between years (Figures 4-1 and 4-2, Ranasinghe et al. 1996, 1997). While between-year differences were small, they were in the direction expected from the abnormally strong 1994 spring freshet and the cooler, milder summers experienced in 1996 and 1997. High spring flows have been theorized to cause earlier and spatially more extensive stratification within the Bay, leading to more extensive hypoxia. More severe and widespread low dissolved oxygen conditions were to more extensive hypoxia. More severe and widespread low dissolved oxygen conditions were observed in 1994 relative to previous years, and the larger amount of area failing the Benthic Community Restoration Goal was most likely a result of these oxygen conditions. Conversely, smaller temperature gradients existing when summer surface water temperatures are lower weaken stratification and reduce hypoxia, potentially accounting for the smaller amount of area failing the Restoration Goals in 1996 and 1997.

The probability-based Chesapeake Bay-wide estimates developed in this chapter are the result of reviews conducted jointly by the Maryland and Virginia Chesapeake Bay benthic monitoring programs. The multi-year review examined program objectives, analysis techniques, and power of the programs to detect trends. One objective that emerged from the program review process was a goal of producing a Bay-wide area estimate of degraded benthic communities with known and acceptable uncertainty. That goal has now been accomplished in two consecutive years.

Although these Bay-wide estimates significantly improve the living resource information available to managers, current information should be considered preliminary because the thresholds on which they are based are undergoing necessary, further development. The estimates are dependent on a fully validated threshold for assessing the condition of the benthic community in each sample collected. Although the Chesapeake Bay Benthic Community Restoration Goals (Ranasinghe et al. 1994b, updated by Weisberg et al. 1997) provide most of the necessary thresholds, there were some gaps which are being filled.

Thresholds are presently being developed for tidal freshwater areas, which constitute about 7% of the Bay and 4-10% of the Maryland Bay, depending on river flow. Thresholds are not currently available for these areas because insufficient data were available when Goals were initially established. The effort was based on preexisting data that were not collected for the purpose of establishing thresholds. The effort currently underway includes a sampling effort to collect necessary data. Once tidal freshwater thresholds are established, the estimates will encompass the entire Bay. Because of their location close to human activity, and the limited potential for dilution due to their small size, tidal freshwater areas are important for Bay management.

The Goals can also be improved by refinement of aspects not examined in detail during initial Goal development. Examples are: identifying how many and which attributes are minimally necessary for calculating Goal attainment if all community attributes are not measured; ascertaining whether different thresholds should be applied to assess sites where single as opposed to multiple samples are collected; and identifying the magnitude of deviation from the 3.0 nominal threshold value that constitutes statistically and ecologically significant deviation from the Restoration Goal. With Chesapeake Bay Program assistance, we hope to improve the Goals by addressing these technical issues.

These efforts will result in evolution of the Goals leading to changes in estimates of the area of the Bay meeting Restoration Goals. These revisions are expected to be small, and should amount to fine-tuning rather than order of magnitude changes. One strength of the probability-based sampling element is that the amount of area meeting revised Goals can be recalculated in future years so that trends in the area meeting the Goals can be compared in a consistent and rigorous fashion.

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APPENDIX A

FIXED SITE COMMUNITY ATTRIBUTE 1984-1997 TREND ANALYSIS RESULTS

Summer temporal trends in benthic community attributes 1984-1997. Monotonic trends were identified using the Van Belle and Hughes (1984) procedure. $^{+}$: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; shaded trend cells indicate increasing degradation; unshaded trend cells indicate improving conditions. [a]: trends based on 1989-1997 data; [b]: attribute trend based on 1990-1997 data.

Station	B-IBI	Abundance (#/m^2)	Biomass (g/m^2)	Shannon Diversity	Abundance of Pollution Indicative Taxa (%)	Abundance of Pollution Sensitive Taxa (%)	Biomass of Pollution Indicative Taxa[b] (%)	Biomass of Pollution Sensitive Taxa[b] (%)
Potomac River								
036	NA	-173.44^{+}	-7.95 **	0.10 **	4.76 **	0.00	0.01^{+}	-0.00
040	0.00	21.70	-0.02	0.01	1.31 **	0.86	-0.00	0.00
043	-0.00 **	57.14	3.58 *	0.01	0.51 **	0.32	0.00	-0.10
044	0.04^{+}	-10.18	0.19 *	0.09 **	2.53 **	1.28 *	-0.09	0.43
047	-0.04 **	6.67	6.51 **	0.00	0.43 **	0.49	-0.00	-0.10
051	0.00	33.33	-0.42 **	0.02^{+}	-0.74	0.03	-0.13	-1.11
052	0.00	15.04	0.00	0.00	0.00	0.00	0.00	0.00
Patuxent River								
079	NA	-32.73	0.05 **	0.13 **	3.29 **	0.00	-7.14 **	0.00
077	-0.09 **	128.64 **	-0.01	-0.02	5.22 **	-1.22 *	9.77	-3.93
074	-0.07 *	160.00 **	0.08	0.02	0.76 **	0.32	0.02^{+}	-0.00
071	0.00	17.48	-0.00	0.07 *	-2.86 **	0.00	-4.44 *	0.00
Choptank River								
066	0.00	50.13	0.00	0.02	2.00 **	-0.25	0.07	-2.94
064	0.13 **	114.55 **	0.48 **	0.03^{+}	-0.45	1.72 **	0.01	0.95
Maryland Mainstem								
026	0.04 **	82.55^{+}	3.98 **	0.03	0.15	4.96 **	0.00	0.00
024	0.10 **	159.58 **	0.11	0.02	0.20	0.24	-0.01	1.74
015	0.00	54.77^{+}	0.02	-0.02	-0.34	-0.43	-0.10	-0.36
006	0.08 **	56.33^{+}	-0.02	0.02	-0.34	2.18 **	-0.11	34.13 *
001	0.03^{+}	136.00 **	0.06	-0.00	-0.42	0.82 *	-0.17	1.36

Station	B-IBI	Abundance (#/m²)	Biomass (g/m²)	Shannon Diversity	Abundance of Pollution Indicative Taxa (%)	Abundance of Pollution Sensitive Taxa (%)	Biomass of Pollution Indicative Taxa[b] (%)	Biomass of Pollution Sensitive Taxa[b] (%)
Maryland Western Shore Tributaries								
022	0.09 **	151.52 **	0.32 **	0.06 *	0.84	0.68**	-0.83 *	4.63+
023	0.07 *	-72.07 *	0.09	0.05 **	-0.52	0.34 **	-0.22	0.00
201 [a]	0.00	33.28	-0.00	0.09	-3.03 +	0.00	8.11	0.00
202 [a]	0.00	13.26	0.00	0.00	0.00	0.00	13.33	0.00
Maryland Eastern Shore Tributaries								
029	-0.04+	280.00 *	0.57	0.00	4.21 **	3.60 **	0.01 *	-0.24 **
068	0.00	193.43+	1.87 *	0.04 **	0.22	2.36 *	-0.03	0.13
062	0.00	228.33 **	0.08+	-0.01	-0.26 *	0.63	0.00	5.34

APPENDIX B

FIXED SITE B-IBI VALUES, 1997

Appendix B. Fixed site B-IBI values, 1997.					
Station	Date	Latitude (Decimal Degrees)	Longitude (Decimal Degrees)	B_IBI value	Status
001	02 Sept 97	38.41837	76.41823	3.33	Meets Goal
006	02 Sept 97	38.44090	76.44335	4.33	Meets Goal
015	02 Sept 97	38.71418	76.51310	3.33	Meets Goal
022	03 Sept 97	39.25372	76.58673	3.40	Meets Goal
023	03 Sept 97	39.20667	76.52328	3.80	Meets Goal
024	03 Sept 97	39.12202	76.35610	4.00	Meets Goal
026	27 Aug 97	39.27118	76.29022	3.00	Meets Goal
029	26 Aug 97	39.47942	75.94448	3.40	Meets Goal
036	22 Sept 97	38.76940	77.03682	-	None
040	22 Sept 97	38.35725	77.23090	3.80	Meets Goal
043	16 Sept 97	38.38297	76.98850	3.40	Meets Goal
044	16 Sept 97	38.38493	76.99405	3.00	Meets Goal
047	16 Sept 97	38.36567	76.98432	3.80	Meets Goal
051	16 Sept 97	38.20485	76.73907	3.67	Meets Goal
052	16 Sept 97	38.19140	76.74863	1.33	Severely Degraded
062	08 Sept 97	38.38388	75.84993	3.80	Meets Goal
064	02 Sept 97	38.59040	76.06968	4.33	Meets Goal
066	26 Sept 97	38.80118	75.92237	2.60	Degraded
068	19 Sept 97	39.13260	76.07800	4.20	Meets Goal
071	05 Sept 97	38.39462	76.54842	2.33	Degraded
074	05 Sept 97	38.54737	76.67558	3.80	Meets Goal
077	05 Sept 97	38.60453	76.67518	1.80	Severely Degraded
079	12 Sept 97	38.75033	76.68933	-	None
201	03 Sept 97	39.23413	76.49747	1.00	Severely Degraded
202	03 Sept 97	39.21793	76.56417	1.40	Severely Degraded
203	27 Aug 97	39.27492	76.44468	1.40	Severely Degraded
204	25 Aug 97	39.00662	76.50502	2.67	Marginal

APPENDIX C

RANDOM SITE B-IBI VALUES, 1997

Appendix C. Random site B-IBI values, 1997.					
Site	**Date**	**Latitude (Decimal degrees)**	**Longitude (Decimal degrees)**	**B-IBI value**	**Status**
Potomac River					
PMR-04101	15 Sept 97	37.96027	76.29092	2.33	Degraded
PMR-04102	15 Sept 97	37.96418	76.38843	2.00	Severely Degraded
PMR-04103	15 Sept 97	38.01535	76.43902	2.67	Marginal
PMR-04104	15 Sept 97	38.05152	76.45708	1.00	Severely Degraded
PMR-04105	15 Sept 97	38.05293	76.49020	1.67	Severely Degraded
PMR-04106	15 Sept 97	38.05762	76.48277	1.67	Severely Degraded
PMR-04108	15 Sept 97	38.10177	76.56967	1.67	Severely Degraded
PMR-04109	16 Sept 97	38.16603	76.74527	3.00	Meets Goal
PMR-04110	16 Sept 97	38.17290	76.55243	2.33	Degraded
PMR-04111	16 Sept 97	38.18207	76.70962	1.33	Severely Degraded
PMR-04112	16 Sept 97	38.18595	76.81120	2.00	Severely Degraded
PMR-04113	16 Sept 97	38.18687	76.75872	2.33	Degraded
PMR-04114	16 Sept 97	38.18995	76.82668	2.00	Severely Degraded
PMR-04115	16 Sept 97	38.21872	76.84580	2.00	Severely Degraded
PMR-04116	26 Sept 97	38.30548	76.83047	3.00	Meets Goal
PMR-04117	22 Sept 97	38.34243	77.25937	3.40	Meets Goal
PMR-04118	22 Sept 97	38.35478	77.22107	3.40	Meets Goal
PMR-04119	26 Sept 97	38.37927	76.88047	2.60	Degraded
PMR-04120	22 Sept 97	38.46672	77.29623	3.00	Meets Goal
PMR-04121	22 Sept 97	38.55338	77.25083	2.60	Degraded
PMR-04122	22 Sept 97	38.56762	77.25133	3.00	Meets Goal
PMR-04123	22 Sept 97	38.58353	77.25708	2.60	Degraded
PMR-04124	22 Sept 97	38.64550	77.21032	-	None
PMR-04125	22 Sept 97	38.67458	77.11298	-	None
PMR-04126	16 Sept 97	38.22465	76.70643	1.33	Severely Degraded

Appendix C. Random site B-IBI values, 1997.

Site	Date	Latitude (Decimal degrees)	Longitude (Decimal degrees)	B-IBI value	Status
Patuxent River					
PXR-04201	04 Sept 97	38.29648	76.44745	3.33	Meets Goal
PXR-04202	04 Sept 97	38.29853	76.43412	3.33	Meets Goal
PXR-04203	04 Sept 97	38.30333	76.46217	4.00	Meets Goal
PXR-04205	04 Sept 97	38.31910	76.42395	3.67	Meets Goal
PXR-04207	05 Sept 97	38.34287	76.47592	4.00	Meets Goal
PXR-04208	05 Sept 97	38.34680	76.50363	3.00	Meets Goal
PXR-04209	05 Sept 97	38.37038	76.47323	2.67	Marginal
PXR-04210	05 Sept 97	38.39083	76.52045	3.00	Meets Goal
PXR-04211	05 Sept 97	38.39245	76.56053	3.33	Meets Goal
PXR-04212	05 Sept 97	38.39355	76.53882	3.00	Meets Goal
PXR-04213	05 Sept 97	38.40393	76.56377	2.67	Marginal
PXR-04214	05 Sept 97	38.40628	76.56680	3.00	Meets Goal
PXR-04215	05 Sept 97	38.40883	76.54443	3.33	Meets Goal
PXR-04216	05 Sept 97	38.43695	76.61918	2.33	Degraded
PXR-04217	05 Sept 97	38.44500	76.60670	3.00	Meets Goal
PXR-04218	05 Sept 97	38.45142	76.63048	1.67	Severely Degraded
PXR-04219	05 Sept 97	38.45658	76.64425	4.00	Meets Goal
PXR-04220	05 Sept 97	38.45955	76.59742	1.00	Severely Degraded
PXR-04221	12 Sept 97	38.46153	76.65542	3.33	Meets Goal
PXR-04222	12 Sept 97	38.47557	76.64692	2.00	Severely Degraded
PXR-04223	12 Sept 97	38.50722	76.66823	1.33	Severely Degraded
PXR-04224	05 Sept 97	38.54675	76.67578	3.80	Meets Goal
PXR-04226	05 Sept 97	38.37598	76.51623	3.33	Meets Goal
PXR-04227	05 Sept 97	38.41852	76.55958	3.67	Meets Goal
PXR-04228	05 Sept 97	38.37927	76.50363	3.33	Meets Goal

Appendix C. Random site B-IBI values, 1997.					
Site	Date	Latitude (Decimal degrees)	Longitude (Decimal degrees)	B-IBI value	Status
Maryland Western Tributaries					
MWT-04301	28 Aug 97	38.89282	76.53705	3.00	Meets Goal
MWT-04302	28 Aug 97	38.91222	76.49298	4.00	Meets Goal
MWT-04303	28 Aug 97	38.91700	76.49422	4.33	Meets Goal
MWT-04304	25 Aug 97	39.00012	76.49658	3.40	Meets Goal
MWT-04305	25 Aug 97	39.03573	76.56998	1.80	Severely Degraded
MWT-04306	25 Aug 97	39.04092	76.55645	1.00	Severely Degraded
MWT-04307	25 Aug 97	39.05528	76.55403	1.00	Severely Degraded
MWT-04308	25 Aug 97	39.05693	76.55403	1.00	Severely Degraded
MWT-04309	28 Aug 97	39.08328	76.51348	1.80	Severely Degraded
MWT-04310	28 Aug 97	39.08620	76.45075	3.33	Meets Goal
MWT-04311	03 Sept 97	39.17792	76.45962	3.40	Meets Goal
MWT-04312	03 Sept 97	39.18010	76.46412	3.40	Meets Goal
MWT-04313	03 Sept 97	39.18387	76.48173	4.20	Meets Goal
MWT-04314	03 Sept 97	39.19392	76.52392	4.20	Meets Goal
MWT-04315	03 Sept 97	39.19827	76.47808	2.60	Degraded
MWT-04316	03 Sept 97	39.20523	76.52697	3.40	Meets Goal
MWT-04317	03 Sept 97	39.24063	76.55352	1.00	Severely Degraded
MWT-04318	27 Aug 97	39.28022	76.44500	1.00	Severely Degraded
MWT-04319	27 Aug 97	39.29243	76.47732	1.00	Severely Degraded
MWT-04320	27 Aug 97	39.30803	76.39077	3.40	Meets Goal
MWT-04321	27 Aug 97	39.31723	76.36135	3.40	Meets Goal
MWT-04322	27 Aug 97	39.32118	76.37692	3.40	Meets Goal
MWT-04323	29 Aug 97	39.34637	76.36037	1.80	Severely Degraded
MWT-04324	29 Aug 97	39.41908	76.23427	1.80	Severely Degraded
MWT-04325	29 Aug 97	39.44400	76.24155	1.40	Severely Degraded

Appendix C. Random site B-IBI values, 1997.					
Site	Date	Latitude (Decimal degrees)	Longitude (Decimal degrees)	B-IBI value	Status
Maryland Eastern Tributaries					
MET-04401	17 Sept 97	38.04487	75.84405	3.00	Meets Goal
MET-04402	17 Sept 97	38.04860	75.81337	3.33	Meets Goal
MET-04403	17 Sept 97	38.12792	75.84122	3.33	Meets Goal
MET-04404	17 Sept 97	38.13007	75.84138	4.00	Meets Goal
MET-04405	17 Sept 97	38.23882	75.86377	2.67	Marginal
MET-04406	17 Sept 97	38.25888	75.82407	4.67	Meets Goal
MET-04407	08 Sept 97	38.33322	75.88028	3.00	Meets Goal
MET-04408	08 Sept 97	38.34235	75.87458	3.00	Meets Goal
MET-04409	08 Sept 97	38.36102	75.85957	4.20	Meets Goal
MET-04410	08 Sept 97	38.46850	75.81790	3.40	Meets Goal
MET-04411	08 Sept 97	38.49043	75.79995	3.40	Meets Goal
MET-04412	02 Sept 97	38.57722	76.02252	3.80	Meets Goal
MET-04413	02 Sept 97	38.59190	75.99163	3.80	Meets Goal
MET-04414	02 Sept 97	38.60360	76.11325	4.60	Meets Goal
MET-04415	02 Sept 97	38.60820	76.07052	4.20	Meets Goal
MET-04416	26 Sept 97	38.69140	75.97605	3.00	Meets Goal
MET-04417	26 Sept 97	38.70083	75.99127	3.80	Meets Goal
MET-04419	19 Sept 97	39.00872	76.16692	2.33	Degraded
MET-04420	19 Sept 97	39.01480	76.17057	2.33	Degraded
MET-04421	19 Sept 97	39.07623	76.13555	4.60	Meets Goal
MET-04422	19 Sept 97	39.08353	76.19283	4.20	Meets Goal
MET-04423	26 Aug 97	39.44857	76.00758	3.40	Meets Goal
MET-04424	26 Aug 97	39.51420	75.89883	3.40	Meets Goal
MET-04425	26 Aug 97	39.54843	75.86668	2.20	Degraded
MET-04426	19 Sept 97	39.18525	76.06007	3.40	Meets Goal

Appendix C. Random site B-IBI values, 1997.					
Site	Date	Latitude (Decimal degrees)	Longitude (Decimal degrees)	B-IBI value	Status
Upper Bay					
UPB-04601	03 Sept 97	39.03165	76.25487	3.80	Meets Goal
UPB-04602	03 Sept 97	39.04595	76.39598	3.33	Meets Goal
UPB-04604	03 Sept 97	39.05047	76.28782	2.67	Marginal
UPB-04605	03 Sept 97	39.07997	76.33085	3.33	Meets Goal
UPB-04606	03 Sept 97	39.08883	76.33578	3.33	Meets Goal
UPB-04607	03 Sept 97	39.09073	76.28428	3.33	Meets Goal
UPB-04608	03 Sept 97	39.11103	76.39228	4.33	Meets Goal
UPB-04609	03 Sept 97	39.15212	76.41610	1.80	Severely Degraded
UPB-04610	03 Sept 97	39.15493	76.33860	4.00	Meets Goal
UPB-04611	03 Sept 97	39.17215	76.31662	3.67	Meets Goal
UPB-04612	03 Sept 97	39.17218	76.42253	3.80	Meets Goal
UPB-04613	03 Sept 97	39.17722	76.28983	2.00	Severely Degraded
UPB-04614	03 Sept 97	39.19343	76.39047	4.60	Meets Goal
UPB-04615	27 Aug 97	39.24462	76.40492	2.60	Degraded
UPB-04616	27 Aug 97	39.34887	76.14298	3.40	Meets Goal
UPB-04617	27 Aug 97	39.35983	76.14607	3.00	Meets Goal
UPB-04618	27 Aug 97	39.37595	76.15873	3.40	Meets Goal
UPB-04619	26 Aug 97	39.40828	76.06578	3.40	Meets Goal
UPB-04620	26 Aug 97	39.43727	76.05533	2.20	Degraded
UPB-04621	26 Aug 97	39.45525	76.05057	-	None
UPB-04622	26 Aug 97	39.52635	76.07035	-	None
UPB-04623	26 Aug 97	39.53080	76.05053	-	None
UPB-04624	26 Aug 97	39.53315	76.01970	-	None
UPB-04625	26 Aug 97	39.58895	75.95508	-	None
UPB-04626	27 Aug 97	39.24052	76.26407	3.80	Meets Goal

Appendix C. Random site B-IBI values, 1997.					
Site	Date	Latitude (Decimal degrees)	Longitude (Decimal degrees)	B-IBI value	Status
Mid-Bay					
MMS-04501	17 Sept 97	37.97458	75.97842	3.00	Meets Goal
MMS-04502	15 Sept 97	38.00303	76.28187	2.67	Marginal
MMS-04503	17 Sept 97	38.00803	76.13692	3.67	Meets Goal
MMS-04504	17 Sept 97	38.02243	76.18070	2.00	Severely Degraded
MMS-04505	17 Sept 97	38.04825	76.17580	3.67	Meets Goal
MMS-04506	17 Sept 97	38.06478	75.89567	4.00	Meets Goal
MMS-04508	17 Sept 97	38.09025	76.08187	2.33	Degraded
MMS-04509	17 Sept 97	38.15088	76.12058	3.00	Meets Goal
MMS-04510	17 Sept 97	38.16757	76.00122	3.00	Meets Goal
MMS-04511	17 Sept 97	38.20088	76.01652	2.00	Severely Degraded
MMS-04512	16 Sept 97	38.22682	76.34745	2.33	Degraded
MMS-04513	08 Sept 97	38.30503	76.04568	4.67	Meets Goal
MMS-04514	02 Sept 97	38.53408	76.32705	2.00	Severely Degraded
MMS-04515	02 Sept 97	38.64900	76.49773	1.67	Severely Degraded
MMS-04516	02 Sept 97	38.67842	76.21418	3.00	Meets Goal
MMS-04517	02 Sept 97	38.71307	76.13290	3.40	Meets Goal
MMS-04518	02 Sept 97	38.72167	76.38153	3.33	Meets Goal
MMS-04519	04 Sept 97	38.82305	76.20768	1.67	Severely Degraded
MMS-04520	04 Sept 97	38.82847	76.23908	1.33	Severely Degraded
MMS-04521	04 Sept 97	38.83690	76.31737	1.00	Severely Degraded
MMS-04522	04 Sept 97	38.84585	76.26268	1.67	Severely Degraded
MMS-04524	04 Sept 97	38.90608	76.13862	2.60	Degraded
MMS-04525	04 Sept 97	38.92097	76.29598	3.00	Meets Goal
MMS-04527	02 Sept 97	38.62112	76.22153	2.67	Marginal
MMS-04528	16 Sept 97	38.25440	76.34733	2.67	Marginal

APPENDIX D

Benthic Community Condition in Relation to Water Quality, Sediment Quality and Watershed Stressors in Chesapeake Bay. Manuscript Submitted to Estuaries.

Benthic Community Condition in Relation to Water Quality, Sediment Quality and Watershed Stressors in Chesapeake Bay

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Abstract

Associations between macobenthic community condition, measures of water column and sediment exposure, and measures of anthropogenic stress throughout the watershed were examined for the Chesapeake Bay, USA. Correlation analysis was used to examine associations between sites with poor benthic condition and measures of pollution exposure in the water column and sediment. The extent of low dissolved oxygen events was strongly correlated with benthic community condition, explaining 42% of the variation in benthic response. Sediment contaminants explained about 10% of the variation in benthic response, but most of this effect resulted from responses at a few locations including Baltimore Harbor and the Southern Branch of the Elizabeth River. After removing the effects of low dissolved oxygen events, the residual variation in benthic community condition was weakly correlated with surrogates for eutrophication - concentrations of total nitrogen, total phosphorus and chlorophyll *a*. Associations between benthic condition and anthropogenic inputs and activities in the watershed were also studied by correlation analysis. Benthic condition was negatively correlated with measures of urbanization (i.e., population density, point-source loadings, and total nitrogen loadings) and positively correlated with watershed forestation. Significant correlations were observed with population density and nitrogen loading below the fall-line, but not above it, suggesting that near-field activities have a greater effect on benthic condition than activities in the upper watershed. Based upon the associations identified by this study, efforts to improve the biotic integrity of the Chesapeake Bay should include (1) reducing nonpoint source loadings of nitrogen, (2) reducing point source loadings of nitrogen, (3) reducing loadings of contaminants from all sources, (4) increasing the spatial extent of vegetated riparian buffer zones, and (5) implementing best management practices. The last two activities may, to some extent, ameliorate some of the effects of reducing forests and increasing urban and agricultural land use. Our results are consistent with the assumption that nutrient and contaminant reduction will eventually result in measurable and predictable improvements in the condition of Chesapeake Bay estuarine biota.

Introduction

Coastal seas, bays, lagoons and estuaries have become increasingly degraded due to anthropogenic stresses (Nixon 1995). Although relationships between watershed stressors, land use, tributary levels of nutrients and contaminants, and the integrity of the biotic communities of receiving waters have been previously well studied in freshwater ecosystems (Allan et al. 1997) few studies have addressed these relationships in estuarine ecosystems (Comeleo et al. 1996; Valiela et al. 1997).

At watershed spatial scales in freshwater ecosystems, land use patterns affect nutrient, contaminant, and sediment loadings as well as aquatic habitat diversity (Correl et al. 1992; Roth et al. 1996; Jordon et al 1997c). All of these factors interact to affect the condition of biotic communities. The most diverse and trophically complex aquatic communities are associated with high proportions of forested land use; biotic communities with compromised integrity are associated with agricultural land use, and the most impacted biotic communities are associated with high levels of urban development (Mangun 1989; Lenat and Crawford 1994; Grubaugh and Wallace 1995; Lamberti and Berg 1995; Allan et al. 1997).

Changes from forested to agricultural and urban land use reduce the physical diversity of aquatic habitats (Richards et al. 1996). The wood debris and detritus associated with forested watersheds support diverse fish and macroinvertebrate communities. In agricultural and urban watersheds, increased sedimentation and reductions in debris decreases habitat heterogeneity (Richards et al. 1996).

At smaller spatial scales, riparian forests and wetlands may ameliorate the effects of agricultural and urban land use (Johnston et al 1990; Correll et al. 1992; Osborne and Kovacic 1993). The size and composition of riparian zones affect hydraulically-driven variables including sediment loads, nutrient loads, contaminant loads and temperature, and increase habitat diversity by acting as a source of woody debris and detritus (Correll 1997).

The proportion of forested, agricultural, and urban land use in a watershed influences the delivery of nutrients, sediments and contaminants into receiving waters through surface flow (Osborne and Wiley 1988; Correll 1983), groundwater flow (Correll et al. 1992; Jordan et al. 1997c) and atmospheric deposition (Hinga et al. 1991; Correll et al. 1987; Lajtha et al. 1995). Increased nutrient loads are associated with high levels of agricultural and urban land use in freshwater systems (Ostry 1982; Benzie et al. 1991; Hall et al. 1994; Hall et al. 1996; Allan et al. 1997) and there is a strong relationship between land use and mean total nitrogen and phosphorus loads to coastal watersheds (Duda 1982; Correll et al. 1992; Jaworski et al. 1992; Balls 1994; Hopkinson and Vallino 1995). Agricultural land use is often associated with increased nutrient loads from fertilization of crops, fertilization of pasture lands and livestock wastes (Ustach et al. 1986; Fisher and Oppenheimer 1991; Turner and Rabalais 1991; Jaworski et al. 1992; Correl et al. 1997). Urban land use increases nutrient loads from a variety of sources including sewage treatment plant effluents (Valiela et al. 1997), septic field drainage (Valiela and Costa 1988;

Weiskel and Howes 1992), and surface water runoff from large areas of impervious surfaces (Klein 1979; Novotny et al. 1985). Numerous studies have demonstrated that forested watersheds, or those with significant vegetated riparian zones, reduce nutrient loadings in both surface (Correll 1997; Verchot et al 1997a) and subsurface groundwater (Lowrance 1992; Nelson et al. 1995; Hill 1996; Correll 1997; Verchot et al. 1997b; Gold et al. 1998).

Increased nutrient loads alter the trophic structure of aquatic ecosystems. Many first- and second-order freshwater streams are strongly heterotrophic with energy flow dominated by allocthonous sources of organic matter (Fisher and Likens 1973; Conners and Naiman 1984; Howarth et al. 1991; Correll 1997). The balance between autotrophs and heterotrophs changes as rates and sources of primary production shift from detrital and macroalgal sources to microalgal blooms fueled by increased nutrient loadings. Increased nutrient loads in coastal ecosystems result in increased algal blooms (Boynton et al. 1982; Malone et al. 1986, 1988; Fisher et al. 1992), increased low dissolved oxygen events (Taft et al. 1980; Officer et al. 1984; Malone et al. 1996), alterations in the food web (Malone 1992) and declines in living resources (Kemp et al. 1983; USEPA 1983).

Sediment and contaminant loads are increased in agricultural and urban dominated watersheds (Hoffman et al. 1983; Beasley and Granillo 1988; Howarth et al. 1991; Lenat and Crawford 1994), mainly due to storm-water runoff (Medeiros et al. 1983; Schmidt and Spencer 1986; Corbett et al. 1997). The type of tillage in agricultural areas and the amount of impervious surface in urban areas are both important factors influencing these loads. The contaminants include a variety of pesticides, herbicides, heavy metals and organic chemicals (Wilber and Hunter 1979; Lenat and Crawford 1994; Vernberg et al. 1992). Soil erosion increases nutrient loads of particle-bound phosphorus as well as sediment loads (Jordan et al. 1991a).

Our study concerns the Chesapeake Bay, which is the largest estuary in the United States and of considerable economic importance. The Chesapeake Bay has a drainage basin of approximately 165,760 km^2 (USEPA, 1983) and its watershed comprises over 150 rivers, streams and creeks covering portions of six states and the District of Columbia. Six major tributaries, the Susquehanna, Potomac, Rappahannock, James, York and Patuxent Rivers, contribute almost 90% of the total freshwater input to the Bay (USEPA, 1983). Its economic importance is at least threefold: historically, it supported some of the most productive commercial fisheries in the world; it is a center of recreational and tourism activity; it includes Hampton Roads and Baltimore, two of the nation's largest port complexes.

Environmental conditions in the Chesapeake Bay and its tributaries have deteriorated significantly over the past 50 years, resulting in declines in a variety of living resources including submerged aquatic vegetation, finfish and shellfish. These declines have been attributed primarily to increased eutrophication and toxic substances (EPA 1983; Dauer et al. 1998). More than 50 research projects funded by the EPA, beginning in 1976, related observed declines in living resources to trends of increasing concentrations of nutrients and toxics in the water and sediments.

Environmental conditions in the Chesapeake Bay have been monitored since 1984 in response to the observed declines. Data are available for abundance, biomass, and species diversity of various living resources; concentrations of dissolved oxygen and nutrients in the water column; concentrations of chemical contaminants in the sediment; and measures of human activity in the watershed, such as population density, land use, and loadings of nutrients and toxics. Monitoring was one aspect of the Chesapeake Bay Agreements implemented between the EPA, the State of Maryland, the Commonwealths of Pennsylvania and Virginia, and the District of Columbia in 1983 and 1987 to share responsibilities for a comprehensive, long-term program to restore and revitalize the bay.

Few studies of estuaries like Chesapeake Bay have attempted to explore relationships between living resources, their exposure to water-column and sediment pollution, and human activity throughout the watershed on regional scales (Comeleo et al. 1996) due to complexity of the environment and the biota. Compared to freshwater systems, estuaries pose additional challenges due to the complexity and variability of physical and chemical factors (Hopkinson and Vallino 1995); tidal mixing, salinity gradients, and salinity-driven water chemistry differences complicate efforts to associate causes and effects in estuarine systems.

The habitat-specificity of biotic communities also hampers estuarine studies at large spatial scales. The numbers and kinds of organisms vary with salinity zone and sediment type, and confound efforts to assess relative condition and to associate causes and effects across habitat boundaries.

The recent development of a Benthic Index of Biotic Integrity (B-IBI) for Chesapeake Bay provides a framework for conducting habitat-independent assessments of benthic community condition (Ranasinghe et al. 1994a; Weisberg et al. 1997). Benthic communities are living estuarine resources which provide important ecological linkages with other trophic levels. Strong relationships have been found between freshwater benthic invertebrate community condition and land use patterns (Mangun 1989; Lenat and Crawford 1994; Grubaugh and Wallace 1995; Lamberti and Berg 1995), although few studies have examined estuarine benthic community condition in relation to watershed stressors (Lerberg 1997).

Our objective is to relate the condition of Chesapeake Bay benthic communities, their exposure to water column and sediment-borne pollutants, and watershed activities that may change their pollution exposure. We attempt to achieve the full potential of the monitoring program, obtaining a quantitative understanding, on bay-wide scales, of the relationship between the B-IBI, a habitat-independent living resource condition measure, and water quality and watershed measures. Our approach is correlative and our intent is to identify associations between benthic community condition and measures of exposure and stress throughout the Chesapeake Bay watershed.

Methods

Response, Exposure, and Stressor Variables

Three kinds of indicators were identified from available Chesapeake Bay monitoring programs: a biological response variable, exposure variables, and stressor variables. The biological response variable was the Benthic Index of Biotic Integrity (B-IBI) developed for macrobenthic communities of the Chesapeake Bay (Ranasinghe et al. 1994a; Weisberg et al. 1997). The B-IBI integrates a variety of metrics associated with healthy benthic community structure (including abundance, biomass, species diversity, feeding guilds, and pollution-sensitive and pollution-indicative species). Validation of the B-IBI with independent data from the Chesapeake Bay indicates that the index correctly distinguishes stressed sites from reference sites 93% of the time (Weisberg et al. 1997). Benthos are good indicators of local environmental conditions because most benthic species are relatively immobile and do not migrate. Therefore, individuals cannot avoid stressful environmental conditions and the condition of the benthic community reflects the quality of the immediate environment. Since many species of benthic invertebrates are relatively long-lived, changes in the characteristics of macrobenthic communities represent the integrated result of environmental changes over time.

Exposure variables, the second kind of indicator, are measures of the occurrence or magnitude of physical, chemical, or biological stress. Five exposure variables were selected: bottom water dissolved oxygen concentrations; sediment contaminant concentrations; and water-column concentrations of total nitrogen, total phosphorous, and chlorophyll *a* (Table 1). Dissolved oxygen was selected because of the documented effects of hypoxia on benthic communities of the Chesapeake Bay (Holland et al. 1977, 1987; Pihl et al. 1991; Dauer et al. 1992; Dauer 1993; Dauer et al. 1993; Diaz and Rosenberg 1995). Hypoxia is most frequently associated with deeper areas (below the pycnocline) but wind-driven seiching can bring oxygen-depleted bottom waters over adjacent shallow areas (Tuttle et al. 1987; Breitburg 1990). Contaminants were selected because they present diverse stresses to benthic communities, are a serious threat to both living resources and human health (Baker 1980a, 1980b; National Research Council 1989), and have been previously shown to affect benthic community structure in Chesapeake Bay (Dauer 1993; Dauer et al. 1993). Nitrogen, phosphorous, and chlorophyll *a* concentrations were selected as surrogates for eutrophication. Levels of chlorophyll *a* can be related to patterns of nutrients in Chesapeake Bay (Harding and Perry 1997) and chlorophyll levels are considered the best indicator of nitrogen and phosphorus enrichment in Chesapeake Bay (Harding 1994). Benthos may benefit from higher productivity at low levels of eutrophication but suffer reductions in diversity and function at higher levels of enrichment (Pearson and Rosenberg 1978; Diaz and Rosenberg 1995).

Stressor variables, the third category of indicators, are measures of human activity or natural phenomena that potentially affect living resources indirectly by causing changes in exposure variables. The selected stressor variables included land use patterns, human population density, and point- and nonpoint source loadings of nitrogen and phosphorus (Table 1).

We used data from the Chesapeake Bay from 1984 to 1991. Data were identified by reviewing annual reports for each component of the Chesapeake Bay Water Quality Monitoring Program, interviewing principal investigators of the Water Quality Monitoring Program, and reviewing the Chesapeake Bay Basin Monitoring Program Atlas compiled by the U.S. EPA Chesapeake Bay Program Office (Heasly et al. 1989). In addition, two computerized databases - the CHESSIE database maintained by the U.S. EPA Chesapeake Bay Program Office, and the Nonpoint Source Electronic Bulletin Board maintained by the U.S. EPA Nonpoint Source Information Exchange - were searched for information about ongoing projects. Data sets were acquired if they included measures relevant to our objectives and if the data were collected between 1984 and 1991. Data sets that included annual observations over the entire period were preferred, but data sets consisting of observations for only a few years were included if data with long-term annual time series were not available. Greater detail concerning the process of identifying, acquiring and standardizing data for this study can be found in Chaillou et al. (1992) and Ranasinghe et al. (1994b).

Segmentation of Chesapeake Bay and its tributaries

Values of the B-IBI and exposure variables were calculated for segments of the Chesapeake Bay and its tributaries (Fig. 1). We used a slightly modified version of the Chesapeake Bay Program (CBP) segmentation scheme (Heasly et al. 1989) to aggregate each kind of data. The first modification of the CBP segmentation scheme was to divide segments into "deep" (below pycnocline) and "shallow" (above pycnocline) subdivisions if salinity, dissolved oxygen, or temperature concentration differed considerably between the surface and bottom during summer (data collected between June and September). This subdivision of segments was done to preserve within-segment, depth-related differences in exposure to hypoxia. Segments were subdivided if the difference between surface and bottom mean summer values was greater than 2 ppt for salinity, 2 ppm for dissolved oxygen concentration, or 4° C for temperature. The boundary between the shallow and deep subdivisions of a segment was defined as the average of the pycnocline and thermocline depths. The following segments were divided into deep and shallow subdivisions: CB-3, CB-4, CB-5, CB-6, CB-7, CB-8 (Mainstem Bay); WT-5 (Patapsco River), ET-4 (Chester River), ET-5 (Choptank River), RET-1 (Middle Patuxent River), LE-1 (Lower Patuxent River), RET-2 (Middle Potomac River), LE-3 (Lower Rappahannock River), LE-4 (Lower York River), and WE-4 (Mobjack Bay). For details of the actual depth separating deep and shallow classes see Ranasinghe et al. (1994b). The second modification was to create a separate segment for the Southern Branch of the Elizabeth River, which was originally part of segment LE-5 (the Lower James River). Although most CBP segments are relatively homogenous with respect to geographic features and salinity regimes, the sediments of the Southern Branch of the Elizabeth River contain higher concentrations of contaminants than other lower James River sediments (Alden et al. 1988; Dauer 1993; Dauer et al. 1993). These modifications resulted in the data being divided into 62 segments for analysis.

For the B-IBI, nutrients, and chlorophyll, segment condition was characterized by the mean of all values for measurements at all sites within each segment-depth subdivision. For dissolved oxygen, segment condition was summarized as the percentage of summer (July 15-September 30) measurements with concentrations below 2 ppm. Benthic organisms are known to respond negatively to concentrations below this level (Diaz and Rosenberg 1995). Sediment contaminants were expressed as the number of chemicals for which average concentrations exceeded the effects range-low (ER-L) and effects range-median (ER-L) values of Long et al. (1995). ER-L values represent chemical concentrations at which biological responses are first seen and ER-M values represent concentrations at which biological responses are expected to occur. These values were selected because they represent the best available summary of chemical concentrations in terms of expected biological effects. To ensure validity of comparison among bay segments, contaminant data sets were included in the analysis only if they contained measurements for a wide array of both metals and organic chemicals.

Association Between Benthic Response and Exposure Measures

Associations between benthic response (the B-IBI) and exposure measures were evaluated using correlation analysis to test for linear relationships. Correlation analyses were conducted using each of the segment-depth subdivisions as an aggregation level for pairing benthic response and exposure variables.

Association Between Response, Exposure, and Stressor Measures

Associations between benthic response measures and stressor variables were evaluated similarly to their association with exposure variables, using correlation analysis; however, the spatial scale of aggregation was different. Instead of using bay segments, stressor variables were summarized for each of ten tributaries for which watershed level data were available (the James, York, Rappahannock, Potomac, Patuxent, Patapsco, Susquehanna, Chester, Choptank, and Nanticoke Rivers). Stressor data were aggregated for tributaries because stressors (Table 1) are measures of activities and phenomena taking place throughout tributary watersheds and because stressor data were not available on scales fine enough to quantify particular kinds of stress within segments of tributaries.

Benthic response and exposure measures had to be recalculated at the tributary level of aggregation to identify correlations with stressor variables. This was accomplished by calculating an area-weighted mean for all segment-depth subdivisions within each tributary. The tributary mean was calculated as the average of the values used to characterize each subdivision and weighted by the area of the subdivision's bottom. Areas used as weighting factors for unsubdivided segments were obtained from Cronin (1971). Areas for the deep portions of subdivided segments were calculated from Cronin and Pritchard (1975) as the 1-m thick layer of water at the deep-shallow boundary depth (see Ranasinghe et al. 1994b); areas for shallow subdivisions were obtained by subtracting the deep subdivision area from the total segment area.

Several procedures were performed to aggregate stressor data to the tributary scale. Different approaches were necessary for land use and nonpoint source loadings, point source loadings, and population estimates due to the forms in which those data were available. Estimates of tributary watershed area, land use, and nonpoint source nutrient loadings for 1985 were based on results of the Chesapeake Bay Program's Watershed Model (Linker and Allegre 1992). Results for model segments corresponding to United States Geological Survey (USGS) hydrologic units were summed for areas above and below the fall-line in each tributary. The nonpoint source loadings used for this study were sums of watershed model results calculated from animal units and land use.

Point source loadings for 1985 from municipal dischargers in each tributary were obtained from the Atlas85 database (Chesapeake Bay Program 1988). Average annual total phosphorus and total nitrogen discharge concentrations were converted to loadings (in kg/dy) for each facility located in each of the ten tributaries. Facility totals were summed for model segments above and below the fall-line.

Population estimates for model segments above and below the fall-line in each tributary were calculated assuming that populations were distributed evenly across counties and cities. County and city population estimates for 1985 were obtained from NPA Data Services, Inc. (1991), normalized for the proportion of county or city area within each model segment, and summed for above- and below-fall-line areas. The areas of each county and city contained within model segments were obtained from the Chesapeake Bay Watershed Model (Linker and Allegre 1992).

For each of the stressor variables, above fall-line, below fall-line, and total-watershed values were calculated for each tributary, except for the Susquehanna River, which has no below fall-line land area comparable to the below fall-line areas defined for the other tributaries in this study; and for the Patapsco, Chester, Choptank and Nanticoke rivers, which have small above fall-line watersheds and for which separate above fall-line estimates were unavailable. Values were expressed per acre of watershed land area.

Results and Discussion

Associations Between Benthic Community Condition and Exposure Variables

Benthic community condition was negatively correlated with exposure to low dissolved oxygen (Table 2). Exposure to low dissolved oxygen, as estimated by the percentage of measurements less than 2 ppm, accounted for 42.6% of the variation in mean B-IBI values for segment-depth subdivisions for the entire Chesapeake Bay watershed. This follows from the widespread collocation of severely degraded benthic communities and very poor oxygen conditions in segments throughout the deep mesohaline regions of the mainstem of the bay and the Patapsco, Potomac, and Rappahannock rivers; and the very good oxygen conditions in the polyhaline regions, where the benthos were healthiest (Holland et al. 1977, 1987; Pihl et al. 1991; Dauer et al. 1992; Dauer 1993; Dauer et al. 1993; Diaz and Rosenberg 1995). Mesohaline regions of temperate partially stratified estuaries may be naturally hypoxic due to circulation patterns that trap organic matter in bottom waters (Malone et al. 1988; Malone 1992). In Chesapeake Bay, the spring phytoplankton bloom is characterized by (1) large sized taxa (diatoms and dinoflagellates), (2) low zooplankton grazing rates, and (3) high rates of deposition of particulate organic matter to the benthos (Kemp and Boynton 1992; Malone 1992). With the advection upstream of additional phytoplankton biomass in subpycnocline bottom waters, mesohaline bottom waters may become rapidly depleted of oxygen when water temperatures rise and stratification of the water column prevents reaeration (Malone et al. 1988; Boicourt 1992). Presumably the oligohaline areas are flushed well enough to carry the biochemical oxygen demand created by the high nutrients and chlorophyll downstream and trap it in the mesohaline mixing zones. Oxygen stress in polyhaline areas of the lower bay is presumably ameliorated by exchange with well-oxygenated ocean waters. Clearly, benthic communities below the pycnocline in mesohaline waters are the most degraded in the bay (Holland et al. 1987; Dauer et al. 1992; Ranasinghe et al. 1994b).

Benthic community condition was also negatively correlated with exposure to sediment contaminants (Table 2), but the correlation explained only about 10% of the variation and was barely significant. In the absence of severe dissolved oxygen stress, contaminants were associated with poor benthic condition in only a few segments, specifically the Southern Branch of the Elizabeth River (SB) and the Back River (WT4). The correlation with contaminants was low, presumably because the number of segments in which contaminants were found was small relative to the number in which benthic condition was affected by dissolved oxygen stress (Ranasinghe et al. 1994b). These results and the spatial analysis of Ranasinghe et al. (1994b) suggest that, although contaminants are present throughout large areas of the bay, their effects on benthic communities are spatially limited. Only a few bay segments, for example the segments containing the Elizabeth River, the Back River and the Patapsco River, have exceptionally high concentrations of contaminants and severely degraded benthic communities (Alden et al. 1988; Dauer 1993; Dauer et al. 1993; Dauer et al. 1998). The deep regions of the Patapsco River have both high levels of contaminants and frequent low dissolved oxygen events (Ranasinghe et al. 1994b). The remainder of the bay showed little relationship between the presence of contaminants and the condition of the benthic communities. Comeleo et al. (1996) found that levels of metals and organic contaminants in estuarine sediments of Chesapeake Bay were best predicted by small-scale land use (less than 10 km from the sampling location) than by land use patterns at the level of entire watersheds. From a management perspective, these findings suggest that problems caused by contaminants are localized and do not constitute a widespread threat to benthic invertebrate communities. However, the ecological effects of chronic levels of multiple

Chester, Choptank, and Nanticoke Rivers had the highest proportion of agricultural land use. The Susquehanna, Rappahannock, York and James rivers had the least developed watersheds in terms of highest proportion of forested area. The Susquehanna and the York rivers had the best benthic condition and the Patapsco River had the poorest benthic condition (Table 3).

Measures of urban stress and agricultural stress were each correlated with a subset of the exposure variables (Table 4). Frequency of hypoxia and sediment contamination were both correlated with urban variables. Hypoxia was correlated with population density and percent urban area in the watershed. Sediment contamination was strongly correlated with population density and point source nitrogen and phosphorus loadings and moderately correlated with urban land use (Table 4). Sources of estuarine sediment contamination have historically been urban point sources (e.g., sewage and industrial outfalls) and, to a lesser degree, urban runoff and atmospheric deposition (Vernberg et al. 1992). The correlations of contaminants with proportion of urban area and population density per unit area were driven by the responses observed in the Patapsco and Patuxent rivers, which are highly urbanized and have known historical sources of contamination. The relationship between hypoxia and urbanization is less clear-cut because of the widespread hypoxia throughout the central portion of the bay. Although urban areas undoubtedly contribute to hypoxia through the biological oxygen demand of urban runoff and point-source nutrients, the eight tributaries that experienced hypoxia (Table 3) were all potentially exposed to hypoxic subpycnocline waters from the central mainstem of the bay (Boicourt 1992).

Mean annual concentrations of total nitrogen and chlorophyll *a* were both correlated with agricultural land use (Table 4). Total nitrogen concentration was also correlated with nonpoint source nutrient loads. These correlations are attributable to the three Eastern Shore tributaries, the Chester, Choptank and Nanticoke Rivers, which have the highest agricultural land use, nonpoint source nutrient loadings, nitrogen concentrations, and chlorophyll *a* concentrations (Table 3). These results are consistent with of those from both Coastal Plain (Jordan et al. 1997a, 1997c) and Piedmont watersheds (Jordan et al. 1997b) of the Chesapeake Bay where nitrogen concentrations were strongly and positively correlated with increasing proportion of croplands and there was no correlation between phosphorus concentrations and land use. Jordan et al. (1997a, 1997b) found that total nitrogen loads had the highest correlation with percentage of cropland, nitrate-nitrogen a lower correlation, and total organic nitrogen and total ammonium-nitrogen were weakly (Jordan et al. 1997a) or not correlated (Jordan et al. 1997b) with percentage of croplands. All three studies (Jordan et al. 1997a,b,c) found that total phosphorus concentrations were correlated with concentrations of suspended particulates.

Associations Between Watershed Stressors and Benthic Condition

At the total watershed level, benthic community condition was negatively correlated with indicators of urbanization (i.e., population density, point source loadings, and total nitrogen loadings; Table 5). There was also a marginally significant positive correlation with forested land use. Macroinvertebrate community condition in freshwater streams is positively related to forested land use due to decreased sedimentation rates, better water quality (Correll and Weller 1997), and high levels of detritus and woody debris that increase food quality and habitat diversity (Richards and Host 1994; Richards et al. 1997). Significant population density and nitrogen loading correlations appeared below the fall-line but not above the fall-line, suggesting that near-field stressors (below fall-line) have a greater effect than far-field stressors. The below fall-line negative relationship of benthic community condition with nitrogen loadings and absence of a relationship with phosphorus loadings is consistent with the conclusion that nitrogen concentrations are more important in determining phytoplankton biomass and rates of primary production in estuarine and coastal ecosystems (Ryther and Dunstan 1971; Boynton et al. 1982; D'Elia et al. 1986; Howarth 1988; Malone et al. 1988, 1996; Jordan et al 1991b; Correll et al. 1992; Oviatt et al. 1995; Seitzinger and Sanders 1997; Valiela et al. 1997). The correlation of benthic response with point source loadings of nutrients is probably related to sediment contaminant effects as indicated by the strong correlation between sediment contaminant levels and point source loads of both nitrogen and phosphorus (Table 4).

Data Limitations

One limitation of our analyses is that associations between benthic responses and exposure measures were evaluated on the basis of average values for bay segments, rather than by individual site values. This was done because each of the Chesapeake Bay monitoring program elements is conducted independently, and the temporal and spatial distribution of sampling differs considerably among them. Using stratum averages reduces the statistical power available to identify associations in several ways. First, it limits the data in the analysis to the number of segments rather than the number of samples collected. Second, important small-scale spatial information is lost by averaging values over large areas, such as the CBP segments. Pollution insults such as sediment contamination are often patchy, and areas with high concentrations of contaminants and poor biotic condition may occur within 100 meters of places with low contaminant concentrations and acceptable biotic condition. In our analysis, spatially heterogeneous segments were treated as having average contaminant exposure and average biotic condition, potentially obscuring higher degrees of association observable at smaller spatial scales.

Although the lack of spatial and temporal coordination among the different elements of the Chesapeake Bay Monitoring Program potentially reduces the precision of our findings, it does not appear to have biased them. One program, the USEPA's Environmental Monitoring and Assessment Program (EMAP), measured exposure data and benthic response simultaneously, allowing associations to be conducted on a site-by-site basis (Weisberg et al. 1993). When EMAP Chesapeake Bay data were examined in this way, only 9% of the area where benthos were degraded also had concentrations of any contaminant above ER-M. The consistency between this result and the analysis of our entire data set suggests that our conclusions about the spatial limitation of contaminant effects on benthos in Chesapeake Bay is robust to the limitations of the larger data set.

The largest impediment to our analysis was that available data are spatially limited. Monitoring within Chesapeake Bay is presently concentrated at a small number of fixed sites in the mid-bay mainstem and in the large tidal rivers; very few samples are collected in small systems and embayments. This design is adequate for examining trends at a selected set of sites (the original and primary purpose of the monitoring program) but does not lend itself to mapping or segment by segment correlation analysis. For some types of data, values were unavailable for many of the bay segments, or estimates of means were generated on the basis of very few samples. Based upon the results presented in Alden et al. 1997, the benthic biological monitoring programs of Maryland and Virginia added a probability-based sampling design in 1994 and 1996, respectively. Therefore, future analyses relating benthic condition to exposure variables and watershed level stressors will have much greater spatial resolution, at least for the benthos. Understanding processes operating at different spatial scales will be important to improving the effectiveness of restoration efforts (Imhof et al. 1996). For example, land use patterns at large spatial scales may be better predictors of ecological integrity if upstream processes overwhelm local land use effects (Roth et al. 1996) while sediment contaminant effects act at much smaller spatial scales (Comeleo et al. 1996).

As Table 1 indicates, the numbers of observations of exposure variables were large, except for sediment contaminants. Of the 46 strata (not partitioned by depth), no contaminant data were available for 13 segments; for another 12, chemical concentrations were characterized by fewer than three samples. The segments for which no or little data were available were mostly smaller bay segments (e.g. most of the segments, labelled either ET or WT on Fig. 1); therefore, our conclusions about the relative importance of contaminants to biotic response are probably valid bay-wide. However, contaminants were found to be important mostly in embayments and the absence of data in some of these systems leaves open the possibility that a number of other small systems have contaminant problems that are unknown or poorly described.

Conclusions

From a resource management perspective, knowledge of associations between watershed-level stressors and biological responses can be more useful than knowledge of associations with water column or sediment exposure variables because stressors represent human activities over which the resource manager has direct control. When associations with particular watershed stressors are identified, the actions required to reduce stress are more readily apparent. Although it seems intuitively obvious that human activities may affect the ecological condition of adjacent waters, demonstrating direct relationships between estuarine condition indicators and watershed stressors has been difficult (Comeleo et al. 1996). However, restoration efforts of coastal ecosystems must recognize that water quality, habitat quality and biotic integrity are affected by both local (Steedman 1988; Comeleo et al. 1996) and regional patterns of land use (Grubaugh and Wallace 1995; Roth et al. 1996; Allan et al. 1997).

Based upon the relationships indicated by this study, efforts to improve the biotic integrity of the Chesapeake Bay should include: (1) Reducing nonpoint source loads of nitrogen as indicated by the negative relationship of benthic community condition with total nitrogen concentrations (Table 2) and the positive relationship of nitrogen concentrations to nonpoint loadings (Table 4). (2) Reducing point source loads of nitrogen as indicated by the negative relationship of benthic community condition to point source loadings of nitrogen (Table 5). (3) Reducing contaminant loadings, particularly in urban areas, as indicated by the negative relationship of benthic community condition to contaminant levels (Table 2) and indicators of urbanization (population density and point-source loads of nutrients; Table 5) while contaminant levels were associated with indicators of urbanization (population density, percent urban land use and point-source loads of nutrients; Table 4). (4) Increasing the spatial extent of riparian vegetated buffer zones as indicated by the positive relationship between benthic community condition and forested land use (Table 5). And, (5) implementation of best management practices and riparian vegetated buffer zones in agricultural areas as indicated by the positive relationships between agricultural land use and both total nitrogen concentration and chlorophyll *a* concentration (Table 4).

These results emphasize the necessity of addressing the difficult task of reducing nonpoint source nutrient loads, particularly of nitrogen. Nonpoint pollution is considered the leading cause of the remaining water quality problems in the United States (Novotny and Chesters 1989; Fulton et al. 1993) and controlling nitrogen sources is essential to restoring and maintaining estuarine water quality (Valiela et al. 1997). Nonpoint source discharges account for approximately two-thirds of the nitrogen, one quarter of the phosphorus and all of the silicate inputs into the Chesapeake Bay watershed (Correll 1987). Sources of nonpoint nutrient loads are diverse, including agricultural activities, silvicultural activities, and urban runoff (Fulton et al. 1993). Management actions to reduce nonpoint sources loads must take into account the geology and hydrology of different watersheds (Staver et al. 1996). For example, Coastal Plain watersheds are more effective at storing nitrogen and converting nitrogen to gaseous forms than are Piedmont watersheds (Jordan et al. 1997b). The more poorly drained riparian forests of Coastal Plain watersheds may retain as much as 70-90% of the of the total nitrogen inputs from adjacent croplands (Jordan et al. 1993; Jordan et al. 1997a) primarily by promoting denitrification (Lowrance et al. 1984; Peterjohn and Correll 1984; Jacobs and Gilliam 1985). Better drained sandy coastal soils have a lower, 40-62%, retention efficiency (Lajtha et al. 1995). Efforts to reduce atmospheric nitrogen loads, which account for an estimated 40% of the total nitrogen load to the Chesapeake Bay (Fisher and Oppenheimer 1991) will present major interstate management challenges. Nonpoint source phosphorus loads are highly episodic, primarily associated with overland transport during storm events, and highly correlated with the concentration of suspended particles (Staver et al. 1996; Jordan et al. 1997a,b,c). Management actions to reduce erosion from upper soil horizons will be necessary to control nonpoint source phosphorus loads.

Implementing best management practices, increasing vegetated riparian buffer zones, and maintaining existing forest cover are realistic management options to improve the biotic integrity of coastal watersheds because changing land uses alter patterns of nutrient loads into watersheds and large scale changes in the amount of urban and agricultural land use in a watershed is not a realistic management option. Implementation of best management practices in agriculture dominated watersheds has resulted in improved benthic community integrity in freshwater streams (Sallenave and Day 1991). Secondary production of benthic macroinvertebrates was enhanced in watersheds with conservation tillage compared to conventional agricultural land management and was attributed to differences in concentrations of nutrients, herbicides and pesticides (Sallenave and Day 1991). However, best management practices will be less effective in coastal areas that are well-drained (Ritter 1992; Staver et al. 1996). Maintaining present levels of forest cover and increasing vegetated riparian zones (forested and wetlands) is a practical management option for reducing nutrient loads into coastal watersheds (Schlosser and Karr 1981; Valiela et al. 1997). Although the major source of nitrogen into estuarine watersheds is through atmospheric deposition (Correll 1997), retention by natural vegetation greatly reduces the amount of nitrogen that reaches the estuary (Correll and Weller 1997; Valiela et al. 1997). Riparian buffer zones effectively reduce groundwater levels of nitrogen while removing suspended particles (a major source of phosphorous loads), inorganic toxins, and pesticides from overland flow (Correll 1997). The effectiveness of riparian buffer zones is highly dependent upon hydrology, the flow rate of both ground and surface waters, and the depth of groundwater flow relative to the root zone of the riparian vegetation (Correll 1997). These hydrological factors interact with other physico-chemical processes such as physical trapping of particulates, assimilation of surface nutrients and toxic materials as affected by amounts of surface litter, and denitrification, which is affected by soil organic matter and Eh, to determine how effectively nutrients and other materials are retained.

Efforts to reduce sediment loads in the watershed are also important. Increased sediment loads affect nutrient concentrations and the quality and diversity of benthic habitats (Richards et al. 1996). Although sediment loads were not analyzed in our study, the effect of sediment loads on patterns of primary productivity is another important management consideration in restoration efforts. Along the estuarine gradient, light- and nutrient-limitation interactions are complex (Fisher et al. 1988). The general pattern is light-limitation in the upper reaches with low chlorophyll levels, high nutrient levels and high suspended solids; a chlorophyll maximum region seaward of the turbidity maximum in the mesohaline regions; and in the lower reaches nutrient limitation of phosphorus in the spring followed by nitrogen limitation in the summer (Harding et al. 1986; Pennock and Sharp 1994). Even if total nutrient loads are reduced in the upper watershed, light-limitation in the upper reaches of tributaries, e.g. in tidal freshwater regions (Harding et al. 1986; Dauer et al. 1998), may allow large loads of nutrients to reach the lower polyhaline regions of the tributaries. In addition high sediment loads can reduce the restoration of submerged aquatic vegetation through light-limitation (Stevenson et al. 1993).

Previously, most of the relationships indicated by this study were simply assumed by environmental managers of the Chesapeake Bay. Our results provide managers with additional rationale for their actions and are consistent with the assumption that nutrient and contaminant reductions will eventually show measurable and predictable improvements in the integrity of the estuarine biotic communities.

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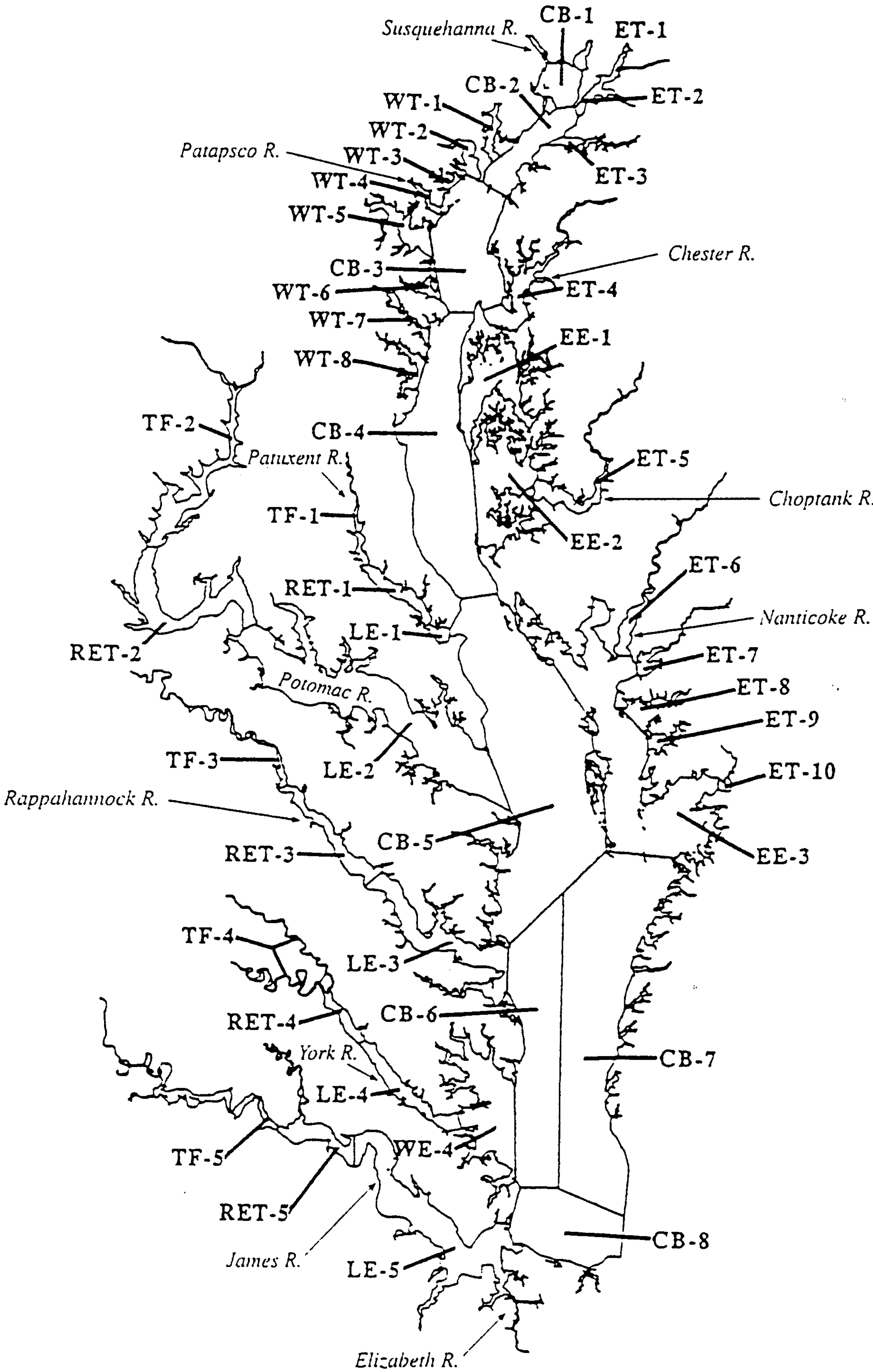

Figure 1. Mape of Chesapeake Bay showing CBP segments.

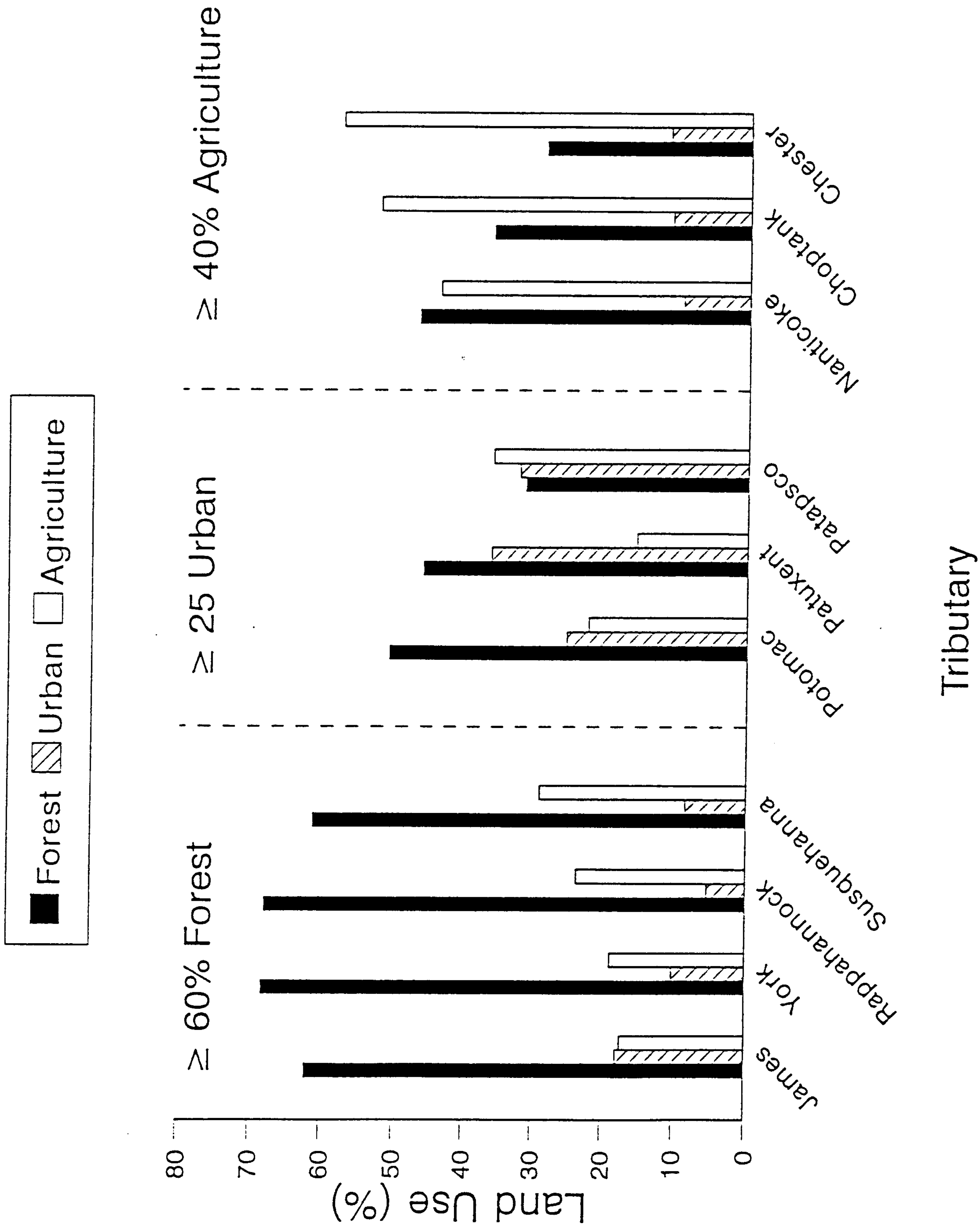

Figure 2. Proportion of land use below the fall-line (forested, urban and agricultural) for 10 tributaries of Chesapeake Bay.

Tables:

Table 1. Response, exposure and stressor variables.

Category	Variable	Period	Number of observations	Sources
Response	B-IBI	1984 - 1991	28,872	1,2,3
Exposure	Bottom water DO concentration	1984-1991	6,011	1,2,3,4,5
	Sediment contaminants	1984-1991	257	1,2,3,6
	Total nitrogen	1984-1991	Total observations for TN, TP and chlorophyll a = 80,286	4,5
	Total phosphorus	1984-1991		4,5
	Chlorophyll *a*	1984-1991		4,5
Stressors	Watershed Land Use (Forested, Agricultural, Urban)	1985	20	7
	Watershed population	1980, 1985, 1990	349	8
	Municipal point-source total nitrogen and total phosphorus loadings	1985	521	9
	Municipal nonpoint-source total nitrogen and total phosphorus loadings	1985	20	7

Sources: 1- Maryland Benthic Monitoring Program (J.A. Ranasinghe, contact), 2- Virginia Benthic Monitoring Program (D.M. Dauer, contact), 3 - EPA Environmental Monitoring and Assessment Program (R. Latimer, contact), 4 - Maryland Department of the Environment (D. Austin, contact), 5- Chesapeake Bay Program Computer Center, 6- Maryland Department of the Environment (R. Eskin, contact), 7 - Linker and Allegre (1992), 8 - NPA Data Services, Inc, (1991), 9 - Chesapeake Bay Program Atlas85 data base.

Table 2. Results of correlation analysis between B-IBI values and measures of pollution exposure. r - Pearson correlation coefficient. p - probabilty level. n - number of replicates. Analyses were performed with and without low dissolved oxygen effect. Correlations in bold print with $p \leq 0.05$.

Exposure Variable	Correlation using entire study means			Correlations with residuals after removing low DO effects		
	r	p	n	r	p	n
Percentage of summer bottom-water DO measurements < 2 ppm	**-0.652**	0.0001	49	*	*	*
No. of chemicals for which the segment mean concentration was > ER-M	**-0.314**	0.049	40	**-0.430**	0.006	40
No. of chemicals for which the segment mean concentration was > ER-L	-0.268	0.095	40	-0.265	0.099	40
Mean water-column concentration of total nitrogen	-0.131	0.402	43	**-0.390**	0.010	43
Mean water-column concentration of total phosphorus	-0.060	0.704	43	**-0.381**	0.012	43
Mean water-column concentration of chlorophyll *a*	-0.086	0.585	43	**-0.302**	0.005	43

Table 3. Area-weighted values of response and exposure variables for tributaries. Data for the Susquehanna River are drawn from Bay segment CB-1.

Tributary	Variable					
	Mean B-IBI value	Bottom DO obs < 2 ppm (%)	No. of contaminants > ER-L	Mean TN (mg/l)	Mean TP (mg/l)	Mean chlorophyll *a* (µg/l)
James	3.35	0.74	0.8	0.71	0.077	13.67
York	3.83	0.30	ND	0.79	0.105	14.65
Rappahannock	2.76	6.82	1.0	0.68	0.049	13.59
Potomac	2.85	20.81	1.6	0.54	0.033	4.52
Patuxent	2.98	16.07	3.0	0.37	0.031	5.30
Patapsco	1.77	18.49	10.0	0.51	0.024	8.61
Susquehanna	3.84	0.00	0.0	1.60	0.048	8.60
Chester	2.87	5.30	0.0	2.19	0.131	37.05
Choptank	3.26	5.82	0.0	1.66	0.089	15.62
Nanticoke	2.63	0.00	0.0	2.18	0.068	17.27

Table 4. Correlation matrix for area-weighted exposure variables with watershed values for stressor variables. Pearson correlation coefficients, p, and n values are presented. Associations in bold print were considered significant with $p < 0.05$.

Stressor Variables	% bottom DO obs < 2 ppm	No. of contaminants > ER-L	Mean TN	Mean TP	Mean active chlorophyll *a*
Population density per unit land area	**0.679** **0.031** **10**	**0.977** **0.001** **9**	-0.473 0.167 10	-0.593 0.071 10	-0.381 0.278 10
% Area under agriculture	-0.059 0.871 10	-0.219 0.572 9	**0.757** **0.011** **10**	0.446 0.196 10	**0.686** **0.029** **10**
% Forested area	-0.440 0.204 10	-0.385 0.307 9	-0.328 0.355 10	-0.023 0.949 10	-0.341 0.366 10
% Urban area	**0.685** **0.029** **10**	**0.742** **0.022** **9**	-0.424 0.222 10	-0.517 0.126 10	-0.353 0.317 10
Total nitrogen loadings per unit land area	0.127 0.726 10	0.479 0.192 9	0.302 0.397 10	-0.278 0.437 10	-0.025 0.946 10
Point-source nitrogen loadings per unit land area	0.529 0.116 10	**0.970** **0.001** **9**	-0.360 0.307 10	-0.466 0.174 10	-0.255 0.477 10
Nonpoint- source nitrogen loadings per unit land area	-0.240 0.451 10	-0.212 0.583 9	**0.663** **0.037** **10**	0.032 0.929 10	0.187 0.603 10
Total phosphorus loadings per unit land area	-0.133 0.713 10	0.052 0.894 9	0.546 0.103 10	-0.076 0.835 10	0.116 0.750 10
Point-source phosphorus loadings per unit land area	0.488 0.152 10	**0.963** **0.001** **9**	-0.343 0.331 10	-0.460 0.181 10	-0.254 0.479 10
Nonpoint- source phosphorus loadings per unit land area	-0.223 0.535 10	-0.216 0.577 9	**0.650** **0.042** **10**	0.049 0.892 10	0.322 0.365 10

Table 5. Correlation of area-weighted B-IBI values with total watershed, above-fall line, and below-fall line values for stressor variables. Pearson correlation coefficients, p, and n values are presented. Correlations discussed in text are in bold print.

Stressor Variable	Total watershed	Above fall-line	Below fall-line
Population density per unit area	**-0.701** **0.024** **10**	-0.283 0.587 6	**-0.664** **0.051** **9**
% Area under Agriculture	-0.330 0.351 10	-0.691 0.128 6	-0.271 0.480 9
% Forested area	**0.623** **0.054** **10**	0.566 0.242 6	0.549 .125 9
% Urban area	-0.514 0.129 10	-0.241 0.646 6	-0.407 0.277 9
Total point- and nonpoint-source nitrogen loadings per unit area	**-0.678** **0.03** **10**	-0.197 0.708 6	**-0.693** **0.038** **9**
Point-source nitrogen loadings per unit area	**-0.720** **0.019** **10**	-0.252 0.630 6	**-0.678** **0.045** **9**
Nonpoint-source nitrogen loadings per unit area	-0.258 0.472 10	-0.086 0.871 6	-0.287 0.454 9
Total point- and nonpoint-source phosphorus loadings per unit area	-0.450 0.192 10	-0.232 0.658 6	-0.425 0.254 9
Point-source phosphorus loadings per unit area	**-0.703** **0.023** **10**	-0.221 0.674 6	-0.535 0.138 9
Nonpoint-source phosphorus loadings per unit area	-0.265 0.460 10	-0.186 0.725 6	-0.270 0.482 9

APPENDIX E

1985-1997 FIXED SITE TREND ANALYSIS RESULTS

This appendix presents the results of trend analysis from 1985 to 1997 and is supplementary to Chapter 3 which presents results for trends in benthic condition from 1984 to 1997, the duration of the Maryland benthic monitoring component. The Virginia benthic monitoring program and several components of the Maryland Program did not start sampling until 1985 and these results are presented to facilitate comparisons across all programs. Data collected in the first year of our program were not used in these analyses.

The 1985 to 1997 trend analysis results are presented in Tables E-1 through E-3 which are similar to Tables 3-1, 3-2, and Appendix A. Trend analysis methods are described in Chapter 2.

Table E-1. Trends in benthic community condition, 1985-1997. Trends in B-IBI values were identified using the Van Belle and Hughes (1984) procedure. Current conditions are based on 1995-97, and initial condition on 1985-87 mean B-IBI values (shown in parentheses). ns: not significant; NA: not applicable; [a]: 1989-1997 trends.

Station	Trend Significance	Median Slope (B-IBI units/yr)	1985-1987 (Initial Condition)	1995-1997 (Current Condition)
Potomac River				
040	$p < 0.1$	-0.04	Meets Goal (3.60)	Marginal (2.91)
043	ns	0.00	Meets Goal (3.71)	Meets Goal (3.67)
044	ns	0.00	Marginal (2.80)	Degraded (2.56)
047	$p < 0.1$	-0.00	Meets Goal (3.89)	Meets Goal (3.67)
051	$p < 0.05$	0.05	Degraded (2.43)	Marginal (2.93)
052	ns	0.00	Severely Degraded (1.37)	Severely Degraded (1.26)
Patuxent River				
071	$p < 0.05$	-0.02	Degraded (2.59)	Degraded (2.22)
074	$p < 0.1$	-0.04	Meets Goal (3.78)	Meets Goal (3.44)
077	$p < 0.001$	-0.17	Meets Goal (3.76)	Degraded (2.20)
Choptank River				
064	$p < 0.01$	0.11	Degraded (2.65)	Meets Goal (3.93)
066	ns	0.00	Degraded (2.53)	Degraded (2.20)
Maryland Mainstem				
026	$p < 0.1$	0.00	Meets Goal (3.16)	Meets Goal (3.53)
024	$p < 0.05$	0.07	Meets Goal (3.04)	Meets Goal (3.74)
015	$p < 0.1$	0.05	Degraded (2.22)	Marginal (2.81)
006	$p < 0.1$	0.05	Degraded (2.56)	Meets Goal (3.48)
001	ns	0.03	Marginal (2.93)	Meets Goal (3.44)
Maryland Western Shore Tributaries				
022	$p < 0.1$	0.07	Degraded (2.08)	Meets Goal (3.27)
023	$p < 0.1$	0.05	Degraded (2.50)	Meets Goal (3.00)
201	ns[a]	0.00	NA	Severely Degraded (1.31)
202	ns[a]	0.00	NA	Severely Degraded (1.13)
Maryland Eastern Shore Tributaries				
029	ns	-0.04	Marginal (2.89)	Degraded (2.47)
062	ns	0.00	Meets Goal (3.42)	Meets Goal (3.53)
068	$p < 0.1$	0.00	Meets Goal (3.51)	Meets Goal (3.67)

Table E-2. Summer temporal trends in benthic community attributes 1985-97. Monotonic trends were identified using the Van Belle and Hughes (1984) procedure. ↑: Increasing trend; ↓: Decreasing trend. +: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; shaded trend cells indicate increasing degradation; unshaded trend cells indicate improving conditions. [a]: trends based on 1989-1997 data; [b]: attribute trend based on 1990-1997 data. Additional detail is provided in Table E-3.

Station	B-IBI	Abundance	Biomass	Shannon Diversity	Pollution Indicative Taxon Abundance	Pollution Sensitive Taxon Abundance	Pollution Indicative Taxon Biomass[b]	Pollution Sensitive Taxon Biomass[b]
Potomac River								
036	NA		↓ **	↑ **			↑ +	
040	↓ +		↓ *		↑ *			
043		↑ +			↑ *			
044				↑ *				
047	↓ +		↑ **	↑ *	↑ *	↓ +		
051	↑ *	↑ *	↓ *	↑ **	↓ **	↑ **		
052						↓ *		
Patuxent River								
079	NA			↑ **	↓ *		↓ **	
077	↓ **	↑ +	↓ **	↓ +	↑ **	↓ *		
074	↓ +	↑ **			↑ *		↑ +	
071	↓ *	↓ +	↓ **		↓ **	↓ **	↓ *	
Choptank River								
066		↑ +			↑ *			
064	↑ **	↑ **	↑ **			↑ **		
Maryland Mainstem								
026	↑ +	↑ **				↑ **		
024	↑ *	↑ *						
015	↑ +	↑ **			↓ *			
006	↑ +	↑ *						↑ *
001		↑ **	↑ *					
Maryland Western Shore Tributaries								
022	↑ +	↑ *	↑ +			↑ **	↓ *	↑ +
023	↑ +	↓ **		↑ +		↑ **		
201 [a]					↓ +			
202 [a]								
Maryland Eastern Shore Tributaries								
029				↓ +		↑ +	↑ *	↓ **
068	↑ +		↑ **	↑ **		↑ **		
062					↓ **	↑ +		

Table E-3. Summer temporal trends in benthic community attributes 1985-1997. Monotonic trends were identified using the Van Belle and Hughes (1984) procedure. +: $p < 0.1$; *: $p < 0.05$; **: $p < 0.01$; shaded trend cells indicate increasing degradation; unshaded trend cells indicate improving conditions. [a]: trends based on 1989-1997 data; [b]: attribute trend based on 1990-1997 data.

Station	B-IBI	Abundance (#/m²)	Biomass (g/m²)	Shannon Diversity	Abundance of Pollution Indicative Taxa (%)	Abundance of Pollution Sensitive Taxa (%)	Biomass of Pollution Indicative Taxa[b] (%)	Biomass of Pollution Sensitive Taxa[b] (%)
Potomac River								
036	NA	-85.95	-7.83 **	0.06 **	0.88	0.00	0.01 +	-0.00
040	-0.04 +	-28.45	-0.11 *	0.01	0.91 *	0.20	-0.00	0.00
043	0.00	108.57 +	0.11	0.01	0.38 *	0.05	0.00	-0.10
044	0.00	-34.58	0.01	0.05 *	0.44	0.82	-0.09	0.43
047	-0.00 +	71.11	6.26 **	0.06 *	0.38 *	-1.56 +	-0.00	-0.10
051	0.05 +	100.00 *	-0.26 *	0.03 **	-1.81 **	0.75 **	-0.13	-1.11
052	0.00	0.00	0.00	0.00	0.00	-0.00 *	0.00	0.00
Patuxent River								
079	NA	82.22	-0.07	0.09 **	0.44 *	0.00	-7.14 **	0.00
077	-0.17 **	163.43 +	-0.53 **	-0.05 +	5.55 **	-1.68 *	9.77	-3.93
074	-0.04 +	262.63 **	-0.77	-0.00	0.07 *	-0.75	0.02 +	-0.00
071	-0.06 *	-68.12 +	-0.20 **	-0.01	-3.22 **	-0.75 **	-4.44 *	0.00
Choptank River								
066	0.00	79.55 +	-0.01	0.06	2.00 *	-0.09	0.07	-2.94
064	0.11 **	104.305**	0.51 **	0.03	-0.60	1.80 **	0.01	0.95
Maryland Mainstem								
026	0.00 +	162.02 **	2.63	-0.03	-0.02	3.93 **	0.00	0.00
024	0.07 *	99.39 *	0.04	-0.01	-0.41	0.00	-0.01	1.74
015	0.06 +	105.33 **	0.08	0.00	-1.51 *	0.18	-0.10	-0.36
006	0.05 +	85.56 *	-0.00	0.02	0.00	1.12	-0.11	34.13 *
001	0.03	146.06 **	0.12 *	0.01	-0.21	0.19	-0.17	1.36

Table E-3. (Continued).

Station	B-IBI	Abundance ($\#/m^2$)	Biomass (g/m^2)	Shannon Diversity	Abundance of Pollution Indicative Taxa (%)	Abundance of Pollution Sensitive Taxa (%)	Biomass of Pollution Indicative Taxa[b] (%)	Biomass of Pollution Sensitive Taxa[b] (%)
Maryland Western Shore Tributaries								
022	0.07 +	151.52 *	0.33 +	0.02	0.38	0.88**	-0.83 *	4.63+
023	0.05 +	-96.70 **	0.01	0.03 +	-0.73	0.36 **	-0.22	0.00
201 [a]	0.00	33.28	-0.00	0.09	-3.03 +	0.00	8.11	0.00
202 [a]	0.00	13.26	0.00	0.00	0.00	0.00	13.33	0.00
Maryland Eastern Shore Tributaries								
029	-0.04	120.91	-0.04	-0.04 +	2.31	2.98 +	0.01 **	-0.24 **
068	0.00 +	25.91	3.65 **	0.05 **	0.21	4.23 **	-0.03	0.13
062	0.00	143.09	0.01	-0.04	-0.47 **	1.14 +	0.00	5.34