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VIRGINIAN PROVINCE DEMONSTRATION REPORT

EMAP-ESTUARIES - 1990

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ABBREVIATIONS

ACE	U.S. Army Corps of Engineers
ANOVA	analysis of variance
APHA	American Public Health Association
ARS	Agricultural Research Service
BI	benthic index
BLM	Bureau of Land Management
CDF	cumulative distribution function
CEQ	Council on Environmental Quality
CPR	cardio-pulmonary resuscitation
CTD	conductivity, temperature, depth
DO	dissolved oxygen
EMAP	Environmental Monitoring and Assessment Program
EMAP-E	EMAP Estuaries Resource Group
EPA or USEPA	U.S. Environmental Protection Agency
ER-L	effects range-low value
ER-M	effects range-median value
FDA	Food and Drug Administration
FS	U.S. Forest Service
FWS	U.S. Fish and Wildlife Service
GFAA	graphite furnace atomic adsorption
IBI	Index of Biotic Integrity

ICP-AES	inductively coupled plasma-atomic emission spectrometry
ITE	Indicator Testing and Evaluation
LTDO	Long-term Dissolved Oxygen
loran	long range navigation
MSE	mean squared error
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
ORD	Office of Research and Development
OSI	Organism-Sediment Index
OTA	Office of Technology Assessment
PAH	polycyclic aromatic hydrocarbons
PAR	photosynthetically active radiation
PCB	polychlorinated biphenyls
ppm	parts per million
ppt	parts per thousand
QA/QC	quality assurance/quality control
REMOTS®	Remote Ecological Monitoring of the Seafloor
RPD	redox potential discontinuity
SDD	Secchi disc depth
SRM	standard reference material
USGS	U.S. Geological Survey

SECTION 5
EVALUATION OF THE SAMPLING DESIGN

One of the objectives of the 1990 Demonstration Project was to evaluate whether the EMAP sampling approach is adequate to assess status and trends in estuarine resources with an acceptable degree of confidence. An additional objective was to identify which attributes of the design should be altered to improve the confidence in assessments of status. The Demonstration Project included study elements to address the following questions about the sampling design:

- What is the appropriate time frame for sampling?
- Should sampling be stratified based on salinity, substrate type, or other system features?
- What is the precision of the status estimates?
- Can the precision of status estimates be improved? If yes, how?
- Can the estimation of precision for status be improved? If yes, how?
- What is the power of the design for detecting changes in status (i.e., trends) over time?
- Should index sites be included as part of the design?

This section discusses each of these questions with regard to results of the 1990 Demonstration Project.

5.1 DETERMINATION OF THE APPROPRIATE TIME FRAME FOR SAMPLING

EMAP does not have the resources to sample adequately on all time scales of interest for understanding biological responses to stress in estuaries; therefore, sampling was confined to a portion of the year when living resources and ecological processes were expected to show the greatest response to anthropogenic stress (i.e. an index period). In the *Near Coastal Program Plan for 1990: Estuaries* (Holland 1990), summer was identified as the most appropriate index period for the Virginian Province because it is the time when 1) dissolved oxygen levels are most likely to approach critical low values; 2) contaminant exposure is likely to be greatest because of low dilution flows and peak metabolic activity; and 3) living resources are most abundant, maximizing the probability of collecting organisms required for assessments. Sampling too early in the summer, before response indicators have been exposed to the lowest dissolved oxygen concentrations, would result in underestimating the extent of degraded ecological conditions. Sampling too late in the summer, after recovery processes have begun, also would result in underestimating degraded ecological condition.

To define the boundaries of the index period for the Virginian Province more precisely, the stability of two indicators, dissolved oxygen concentration (exposure indicator) and the benthic index (response indicator), was examined across three sampling intervals (20 June to 18 July, 19 July to 31 August, 1 September to 22 September). These indicators were selected for examination because they represent 1) both biological and physical processes, and 2) two of the major categories of indicators measured in EMAP monitoring.

The stability of bottom dissolved oxygen measurements and the benthic index was evaluated using a paired t-test to compare indicator responses between sampling intervals. Comparisons between intervals 1 and 2 were conducted separately from those between intervals 2 and 3. Comparisons were limited to sites sampled in both intervals. The comparisons were made for the province as a whole and, separately, for stations north and south of the Hudson River. The latter analysis was conducted to identify any latitudinal differences in the boundaries of the index period. In addition, the semi-continuous data on dissolved oxygen concentration were used to calculate weekly mean and minimum dissolved oxygen values for 16 stations that had relatively complete data records. These results were examined for temporal patterns that might not be detected using only the point measurements taken at each station in each sampling interval.

Point measurements of bottom dissolved oxygen concentration differed significantly between intervals 1 and 2 (Table 5-1). The lowest weekly average and weekly minimum bottom dissolved oxygen values at most stations occurred in early August (Tables 5-2 and 5-3). Average bottom dissolved oxygen concentration did not differ significantly between intervals 2 and 3 (Fig. 5-1, Table 5-1). These findings are based only on point measurements, since continuous bottom dissolved oxygen measurements were not made during interval 3. The late summer continuous bottom dissolved oxygen records, however, suggest that bottom dissolved oxygen concentrations were beginning to increase at many stations, especially those north of the Hudson, toward the end of August.

The benthic index did not differ significantly between the sampling intervals (Table 5-1). Although bottom dissolved oxygen concentrations decreased from interval 1 to interval 2, the sites with low bottom dissolved oxygen and poor benthic conditions presumably had sufficient exposure to low dissolved oxygen stress by June for benthos to respond, or the degraded benthos at these sites reflected previous exposure to low dissolved oxygen stress (i.e., the previous year).

Table 5-1. T-test comparison of dissolved oxygen concentrations and the benthic index among periods. Values in the table are p-values.

	Interval 1 vs. 2	Interval 2 vs. 3
Dissolved Oxygen	0.04	0.71
Benthic Index	0.48	0.28

The stability of the benthic index over much of the summer period permits future field efforts to be scheduled to characterize the temporally variable dissolved oxygen exposure accurately. Based on the dissolved oxygen patterns, future sampling in the Virginian Province will not be initiated until late July and will be scheduled for completion by the end of August; however, given the similarity in exposure between intervals 2 and 3, samples that cannot be obtained in August can be collected through the third week of September, if necessary.

It is inappropriate to define precise boundaries for the index period in the Virginian Province based on a single year of data. Year-to-year variation in meteorological patterns may cause between-year differences in index period boundaries; therefore, deployments to measure dissolved oxygen continuously were conducted from June to September of 1991 and may be conducted in future years. In 1991, meters were placed at four locations that were determined to be representative of the Virginian Province: two in the northern half of the province and two in the southern half. These data will be used to estimate year-to-year variation in the stability of patterns of bottom dissolved oxygen concentration. In addition, the index period boundaries will be reviewed periodically to determine whether the sampling period needs to be adjusted as new indicators are incorporated into the program. The index period boundaries defined by this study are applicable only to the Virginian Province. EMAP-E will need to define province-specific index period boundaries before initiating sampling in other provinces.

5.2 STRATIFIED SAMPLING

Estuaries vary in size, shape, and ecological characteristics. Many, like the Chesapeake Bay, are large, continuously distributed resources that consist of expansive regions of relatively homogeneous habitats; others (e.g., small bays, inlets, and salt ponds) are relatively discrete resources composed predominantly of one habitat type (e.g., high salinity muds). It would not be cost-effective to sample such vastly different estuarine types with a sampling design that treated all estuaries the same. Excessive numbers of

Table 5-2. Average dissolved oxygen concentration (ppm) by week for stations with relatively complete Hydrolab records. Lowest values for each station are shaded. Blanks indicate missing data.

WEEKLY MEAN DO (mg/l)									
	Beginning of July				WEEK			End of August	
System and Station Number	1	2	3	4	5	6	7	8	9
Northern									
East Long Island Sound (22)	5.2	4.1	3.2	2.9	2.1	2.1	1.5	6.6	4.2
Middle Long Island Sound (26)		3.7	3.7	3.5	2.2	2.0			
Narragansett Bay (70)		5.2	5.2	5.8	5.2	3.8	3.3	4.3	4.2
Shrewsbury River (100)	6.7	6.4	8.0	7.7	5.0	5.2		8.2	7.9
Mystic River (106)	6.5	5.5	4.2	7.3	4.0	4.9	4.9	4.9	3.5
Barnegat Bay (256)		7.2	8.1	5.7	5.6	7.2	6.2	5.5	3.6
Southern									
Chesapeake Bay-Mainstem Mouth (54)	5.2	5.0		4.4	3.5	2.3	3.3	3.7	2.4
Chesapeake Bay-Near Mobjack Bay (61)	9.1	9.4	7.5	9.8	7.2	4.9	5.8	4.8	
Chesapeake Bay-Elizabeth River (86)	3.7	2.4	2.7	2.4	2.0	2.0	1.5	2.2	0.6
Chesapeake Bay-Tangier Sound (41)	6.5	5.9	6.0	5.4	5.4	5.5	5.1	5.4	4.5
Chesapeake Bay-Mainstem Near Choptank (65)		3.7	4.9		3.8	2.3	1.8	2.5	2.8
Potomac River-Maryland Point (1984)		4.6	4.4	3.5	1.0	4.5	7.2	5.0	5.3
Rappahannock River-TFO (192)		9.0	8.1	7.2	6.3	5.5	6.2	6.0	
Anacostia River (88)		4.1	0.7	2.7	4.0	3.8	2.3	3.5	1.2
Indian River Bay (150)	6.5	6.0	5.0	5.6	6.7	6.8	5.0	5.6	4.6
Delaware River (223)	9.0		5.8	5.7	5.9	5.4	4.1	5.4	5.5

Table 5-3. Weekly minimum dissolved oxygen concentrations (ppm) for stations with relatively complete Hydrolab records. Lowest values for each station are shaded. Blanks indicate missing data.

	WEEKLY MINIMUM DO (mg/l)								
	Beginning of July				WEEK			End of August	
System and Station Number	1	2	3	4	5	6	7	8	9
Northern									
East Long Island Sound (22)	4.7	3.3	2.7	2.2	1.4	1.4	0.7	6.3	1.2
Middle Long Island Sound (26)		3.5	3.3	3.2	2.1	0.7			
Narragansett Bay (70)		3.7	4.2	4.7	4.0	1.5	1.0	2.1	2.8
Shrewsbury River (100)	1.3	1.3	4.4	3.4	1.9	0.6		7.3	6.6
Mystic River (106)	6.2	2.4	2.6	4.9	0.5	0.4	0.1	0.6	2.0
Barnegat Bay (256)		6.6	7.2	1.9	1.5	4.6	4.1	3.6	1.7
Southern									
Chesapeake Bay-Mainstem Mouth (54)	4.0	3.5		3.2	1.3	0.4	1.4	1.0	1.2
Chesapeake Bay-Near Mobjack Bay (61)	7.6	4.9	6.2	8.4	2.9	1.2	3.0	2.6	
Chesapeake Bay-Elizabeth River (86)	3.1	0.4	0.8	0.2	0.4	0.2	0.0	0.1	0.0
Chesapeake Bay-Tangier Sound (41)	4.6	3.7	3.2	4.3	2.6	2.6	1.6	1.5	1.3
Chesapeake Bay-Mainstem Near Choptank (65)		0.1	0.1		0.0	0.0	0.0	0.3	0.4
Potomac River-Maryland Point (1984)		1.3	1.2	0.6	0.1	3.0	6.6	0.5	3.3
Rappahannock River-TFO (192)		7.6	6.8	6.8	4.7	1.8	5.6	4.8	
Anacostia River (88)		2.1	0.0	1.2	2.1	0.9	0.3	0.4	0.5
Indian River Bay (150)	4.2	4.0	2.7	3.5	5.0	5.0	2.1	3.4	1.8
Delaware River (223)	8.2		5.5	5.4	5.4	4.9	2.1	2.5	3.5

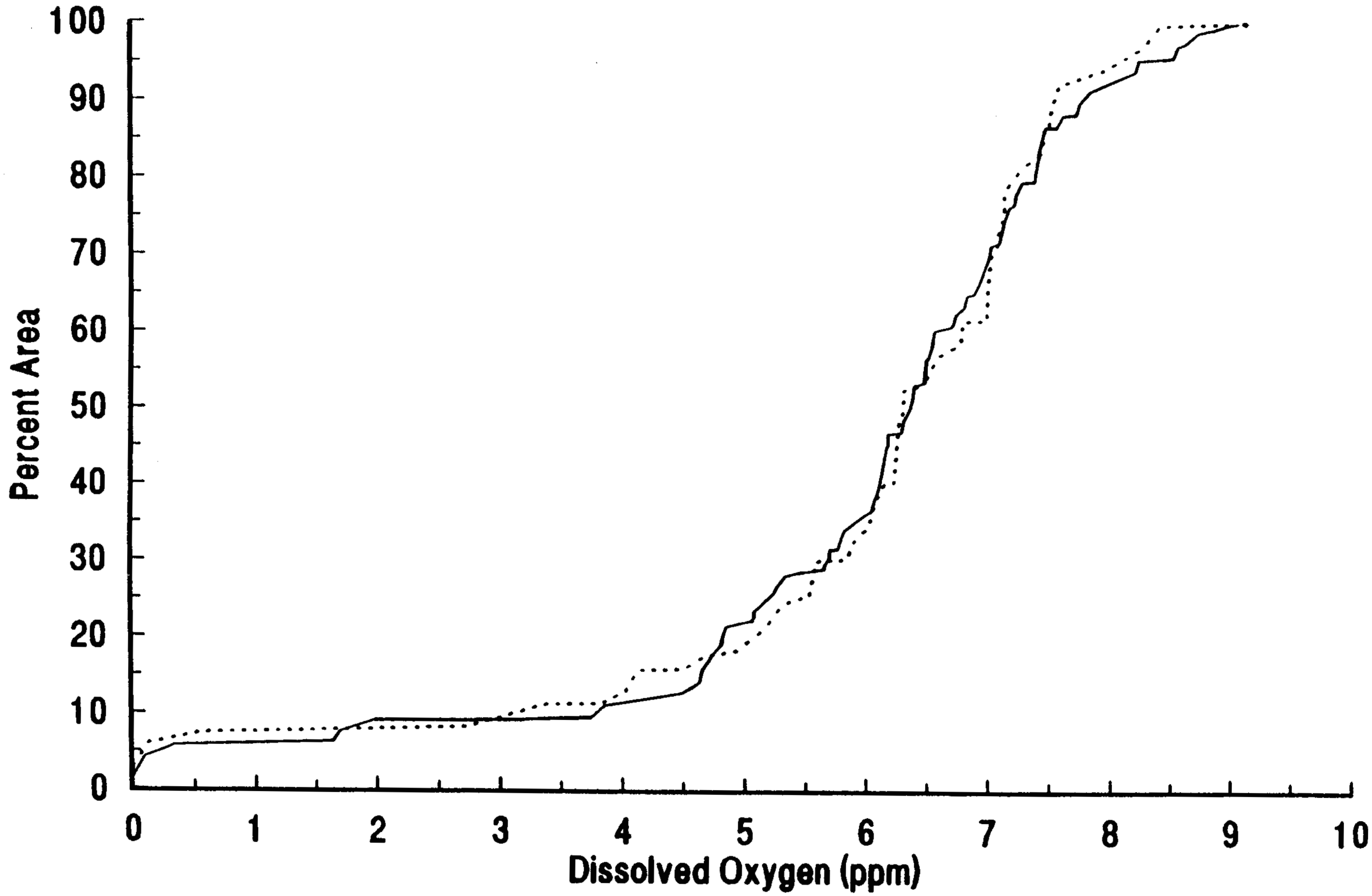


Figure 5-1. Comparison of cumulative distribution frequency for dissolved oxygen between sampling intervals 2 (solid line) and 3 (broken line). Data are limited to sites sampled in both intervals.

samples would be collected from extensive and abundant resources, and rare resources would not be represented adequately. The present EMAP sampling design for estuaries stratifies them into three size classes (i.e., large estuaries, tidal rivers, and small estuaries) that have similar physical features and are likely to respond to stress in similar ways. Stratification allowed the sampling design to be customized for the specific geographic features of each class so that sampling effort was sufficient to assess the extent of degradation for specific subpopulations with small total areas.

The preliminary assessment results presented in Section 6 show that the extent of degraded area differs substantially among the three estuarine classes. Small estuaries and tidal rivers have a higher proportion of degraded area than large estuaries. These differences are sufficient to justify allocation of a disproportionate number of samples to small estuaries and tidal rivers, because regulators and resource managers may want to target these types of systems for further research and remediation activities.

Supplemental stratification schemes considered for EMAP sampling in estuaries have included stratifying by salinity, substrate type, or specific estuarine system (e.g., Chesapeake Bay, Long Island Sound). The rationale for further stratification by substrate and salinity is that estuarine biota respond differentially to habitat parameters, as supported by the analyses presented in Section 6. In addition, pollution exposure is affected by many of these habitat parameters. For example, contaminants tend to accumulate in fine grained sediments, and low dissolved oxygen concentrations are more likely to develop in mesohaline habitats that have two-layered circulation. Stratification based on salinity and substrate characteristics would enable sample sizes in each stratum to be based on precision requirements.

The stratification process requires establishing a sampling frame prior to sampling; misclassification of sites within a class should be minimal. This was a major reason for choosing a stratification scheme based on estuary size. Size data were readily available, and the probability of misclassification was low. Stratification by substrate was considered to be difficult because detailed maps of sediment characteristics are not available for much of the Virginian Province. Even less information is likely to be available about the regional distribution of substrate characteristics in other regions of the country. *A priori* stratification by salinity also was considered problematic because the spatial extent of salinity classes can vary substantially from year to year, depending on rainfall patterns and, in some estuaries such as San Francisco Bay, water management practices.

To estimate the degree of misclassification that would be expected if strata were defined by substrate type, the NOAA database of sediment characteristics for estuaries of the East Coast, the most comprehensive database in existence on substrate distribution for the Virginian Province, was used to project the silt-clay content for each of the EMAP sampling sites visited in 1990. The estimates of silt-clay content were organized into three classes: 1) sand – less than 20% silts and clays; 2) mixed sediments – 20% to 80% silts and clays, and

3) mud – greater than 80% silts and clays. The projected data was compared to the actual substrate type identified by sampling in 1990 to estimate the classification efficiency of the historical data.

No prediction of sediment type was possible for about one-third of the EMAP sites sampled in 1990. Most of these sites were located in small estuaries that are poorly represented in the NOAA historical database. The predicted substrate type differed from actual measurements for more than 50% of the remaining sites (Table 5-4). Stratification by substrate type, therefore, would be highly impractical given existing knowledge of sediment distribution in the Virginian Province.

Table 5-4. Comparison of sediment type predicted from NOAA database with that measured at EMAP sampling sites. Numbers in the boxes represent number of sampling sites. Correct classifications occur on the diagonal.

		PREDICTED		
		Sand	Mixed	Mud
MEASURED	Sand	**24**	**8**	**2**
	Mixed	**20**	**28**	**25**
	Mud	**6**	**10**	**9**
Marginal Misclassification Rate		52%	39%	75%

Overall Misclassification Rate = 54%

There are several possible reasons for the high misclassification rate for substrate type: 1) the available substrate data for the Virginian Province are not sufficiently detailed relative to sediment patch size; 2) the extrapolation techniques are not sufficiently advanced to project substrate type in the Virginian Province adequately; or 3) the imprecision in EMAP measurements was large. To determine whether measurement error for EMAP samples was large, substrate information was compared at sites that were sampled in both intervals 2 and 3. Less than 10% of the sites changed sediment classes between the periods. These findings and QA/QC checks indicate that the measurement error in EMAP sediment grain size analysis was low; therefore, explanations 1 and 2 above are more likely to be responsible for the high misclassification rates.

To estimate the potential misclassification rate associated with stratification by salinity, salinity data from Maryland's Benthic Monitoring Program in the Chesapeake Bay (Holland et al. 1989) were examined for the period 1984 to 1988, and the percentage of sites that switched salinity classes among years was calculated. Three salinity classes were defined for this analysis: 1) brackish (less than 5 ppt), 2) transitional (5 to 18 ppt), and 3) high salinity (greater than 18 ppt). Comparisons were limited to August data, since this is the period targeted for future sampling in the Virginian Province. More than one-third of the sites switched salinity classes over the five-year period. Since the analysis was conducted using only five years of data, it probably underrepresents the rate of misclassification that might occur over several decades (a more appropriate period for EMAP planning). If Long Island Sound data were used, the percent of sites switching salinity class presumably would be much lower because the Sound is characterized by much less year-to-year salinity variation than the Chesapeake Bay.

Stevens and Olsen (1992) investigated the impact of misclassification on the precision of an estimated CDF when the sample was allocated to optimize the estimate of the proportion of a population that is below a critical value x_c. They concluded that the potential benefit from optimal allocation is small, and even under ideal circumstances, precision is improved only in the immediate neighborhood of x_c. Under moderate misclassification (greater than 5%), the attempted optimal allocation was worse than proportional allocation.

The approach of Stevens and Olsen was applied to hypothetical distributions of the benthic index (BI) to illustrate these points. The population of benthic index values was assumed to consist of a mixture of two normally distributed subpopulations, one with mean of 4.5 and standard deviation of 0.5, the other with mean of 7 and standard deviation of 1.5. The first subpopulation was centered on the critical value between degraded and nondegraded and could be considered degraded. The second subpopulation approximated the observed distribution of the benthic index for the entire Virginian Province (Fig. 5-2). The behavior of optimal and proportional allocation was compared by calculating and plotting the width of 90%, one-sided confidence intervals over the range of the response for a sample size of 50. Allocation was optimized at the critical value x_c using the equations given by Cochran (1977). The entire population was assumed to be a 50/50 mix of the two subpopulations.

This scenario was selected for close to maximal potential benefit from optimal allocation when complete, error-free information on population distribution is available. The critical value was set at $x_c = 4.5$, the mean of population 1, so that $F_1(x_c) = 0.5$, $F_2(x_c) = 0.047$, and stratum proportions differed by a factor of 10. Optimal allocation split the sample 70/30, compared to proportional allocation of 50/50. Under this scenario, optimal allocation did give slightly more precise (smaller confidence interval) estimates of the composite population proportion below x_c, but at the price of a larger confidence interval over most of the range (Fig. 5-3a).

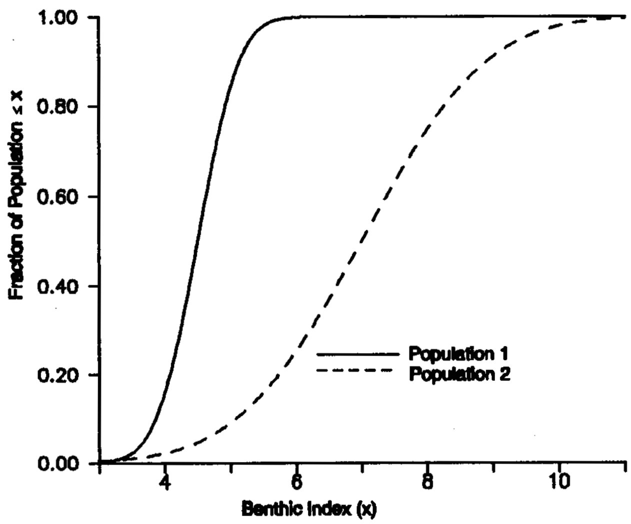


Figure 5-2. Hypothetical distribution functions used in analysis of stratum misclassification effects

When a 20% misclassification rate was assumed for both populations, however, proportional allocation became the superior approach (Fig. 5-3b). "Optimal" allocation was no longer beneficial, even at the critical value, and was substantially worse than proportional allocation over much of the range of the indicator. This degree of misclassification is to be expected in environmental applications and is less than or comparable to the observed rates using salinity or sediment type.

Resource management and environmental agencies for seven of the estuaries in the Virginian Province are in the process of developing comprehensive management and restoration plans. The current sampling design for estuaries is adequate for providing baseline estimates of status against which to evaluate the effectiveness of management plans for three of these estuaries: Chesapeake Bay, Long Island Sound, and Delaware Bay. The EMAP sampling grid can be adapted easily to increase sample density in geographical areas of interest. For example, during the 1990 Demonstration Project, the sampling density in the Delaware Estuary was intensified by a factor of four from a 280 km^2 grid-frame to a 70 km^2 grid-frame, allowing population estimates for Delaware Bay to be made with greater precision (see Section 5.6). Similar intensification could be accomplished in each of the remaining systems in the province that are developing management plans (i.e., Narragansett Bay, Buzzards Bay, New York Harbor Complex, and

(a)

(b)

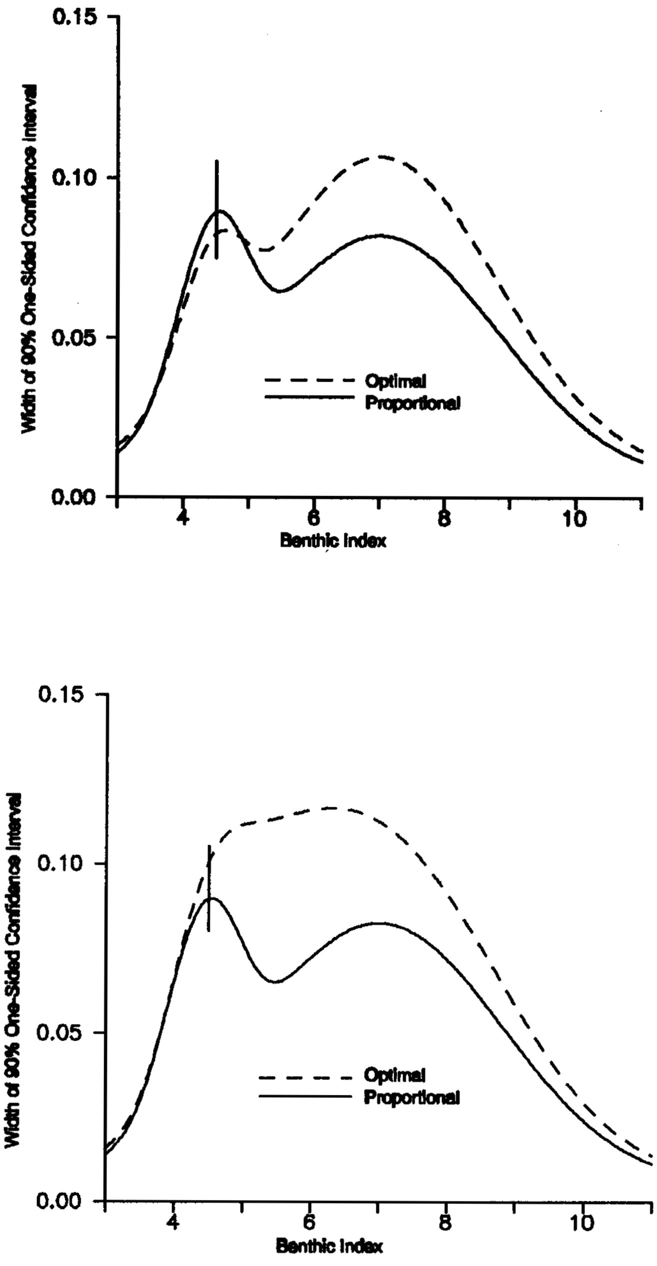


Figure 5-3. Comparison of optimal vs proportional allocation. (a) No misclassification, x_c = 4.5, 50/50 population mix. (b) 20% misclassification, x_c = 4.5, 50/50 population mix.

Delaware Inland Bays) to obtain baseline estimates of their status. The number of samples required to generate system-specific status estimates for these four systems with reasonable confidence exceeds the existing EMAP budget. EMAP is exploring opportunities to develop partnerships with one or more of these programs to demonstrate the value of information obtained by intensifying EMAP monitoring in individual systems.

5.3 PRECISION OF STATUS ESTIMATES

One of the goals of EMAP is to describe status with known and acceptable levels of confidence. Defining "acceptable" confidence levels is an iterative process in which the need for additional precision must be weighed against the cost of obtaining it. A first step in this process is to identify the precision with which the extent of degraded area can be estimated for various estuarine populations of interest (e.g., province as a whole, size classes of estuaries, specific estuaries) with the existing sampling effort.

In this section, precision estimates are presented for each of the response and exposure indicators used in the preliminary evaluation of environmental condition of Virginian Province estuaries. The discussion of precision estimates is centered around two representative indicators, the benthic index and the acute toxicity of sediments to indigenous biota. For these indicators the precision estimates are presented as confidence intervals around the entire cumulative distribution function for the province as a whole (Fig. 5-4); for each estuarine class (Fig. 5-5); and for three large estuaries, Chesapeake Bay, Delaware Estuary, and Long Island Sound (Fig. 5-6). Although precision estimates surrounding the entire CDF are interesting, the most relevant aspect is the confidence interval at the critical levels used to delineate degraded and nondegraded areas, which is how precision estimates for the remaining indicators are presented (Table 5-5). Specific methods used in calculating precision estimates were provided in Section 2.

For the benthic index, the critical value used to distinguish degraded from nondegraded areas was 3.4; for sediment toxicity, that value was 80% of control survival. Based on the current sampling effort, there is a 90% chance that the limits of 16% to 30% for the benthic index shown in Fig. 5-4 include the actual value for percent of area characterized by degraded biological resources. For sediment toxicity, the estimated value is 8%, and there is a 90% chance that the limits of 3% to 13% include the actual value for the area of the province having sediments that are toxic to the test organism, *Ampelisca abdita*.

Estimates of the percent of degraded area for the three classes of estuaries were less precise than for the province as a whole because fewer samples were taken in each class. For the benthic index, the confidence intervals surrounding the estimates of percent

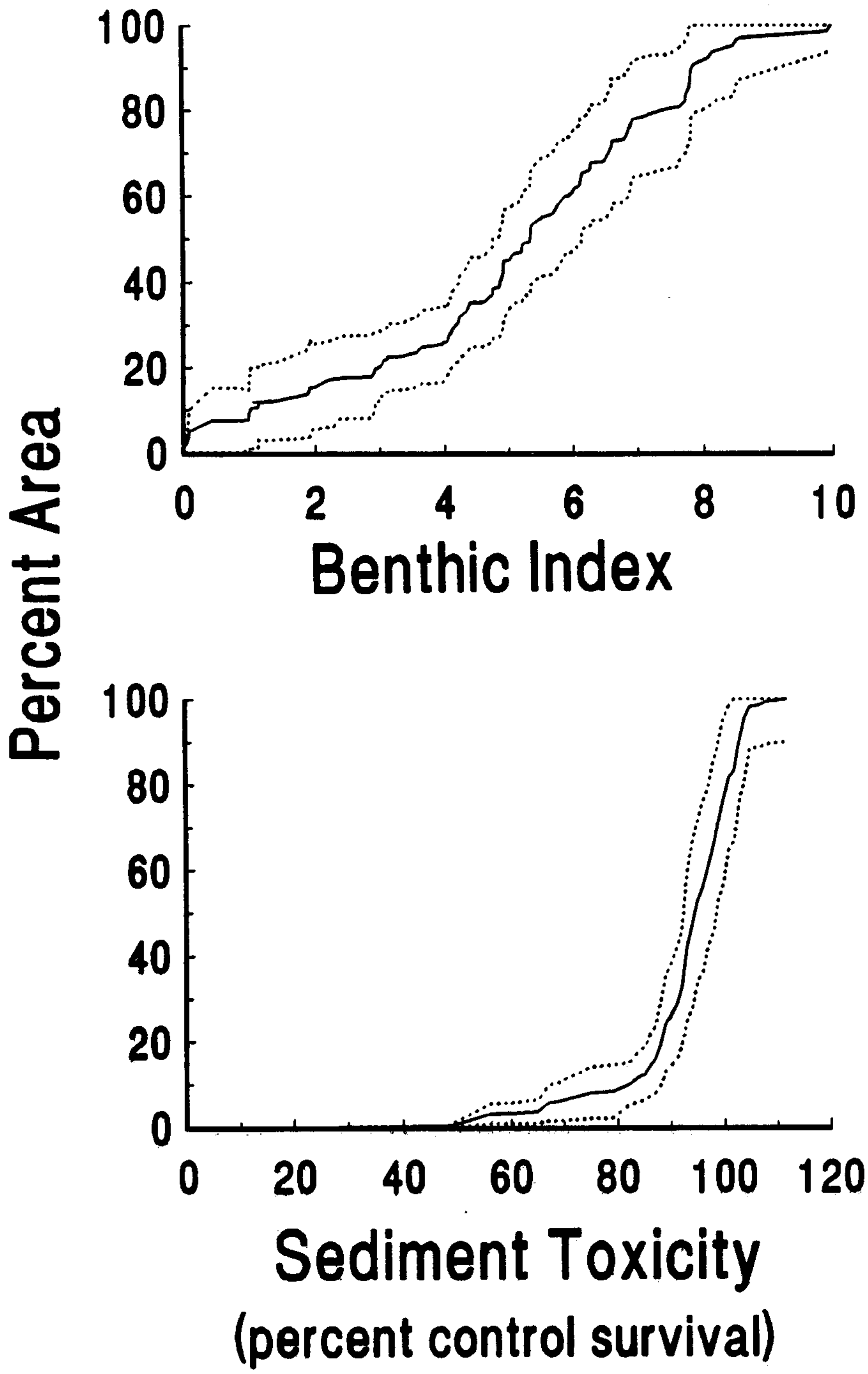


Figure 5-4. Confidence intervals (90%) for estimates of benthic index and sediment toxicity over the entire Virginian Province

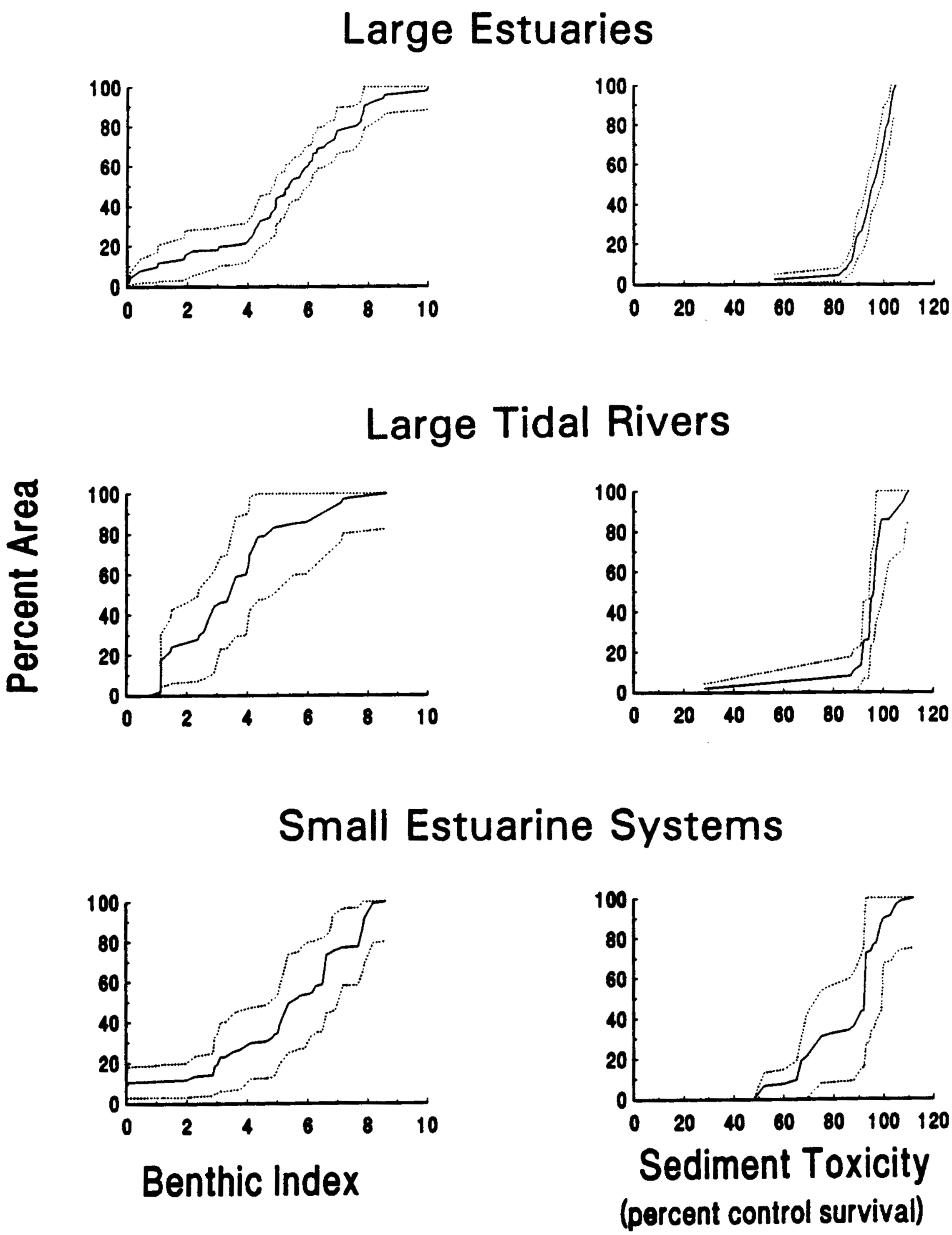


Figure 5-5. Confidence intervals (90%) for estimates of the benthic index and sediment toxicity by resource classes

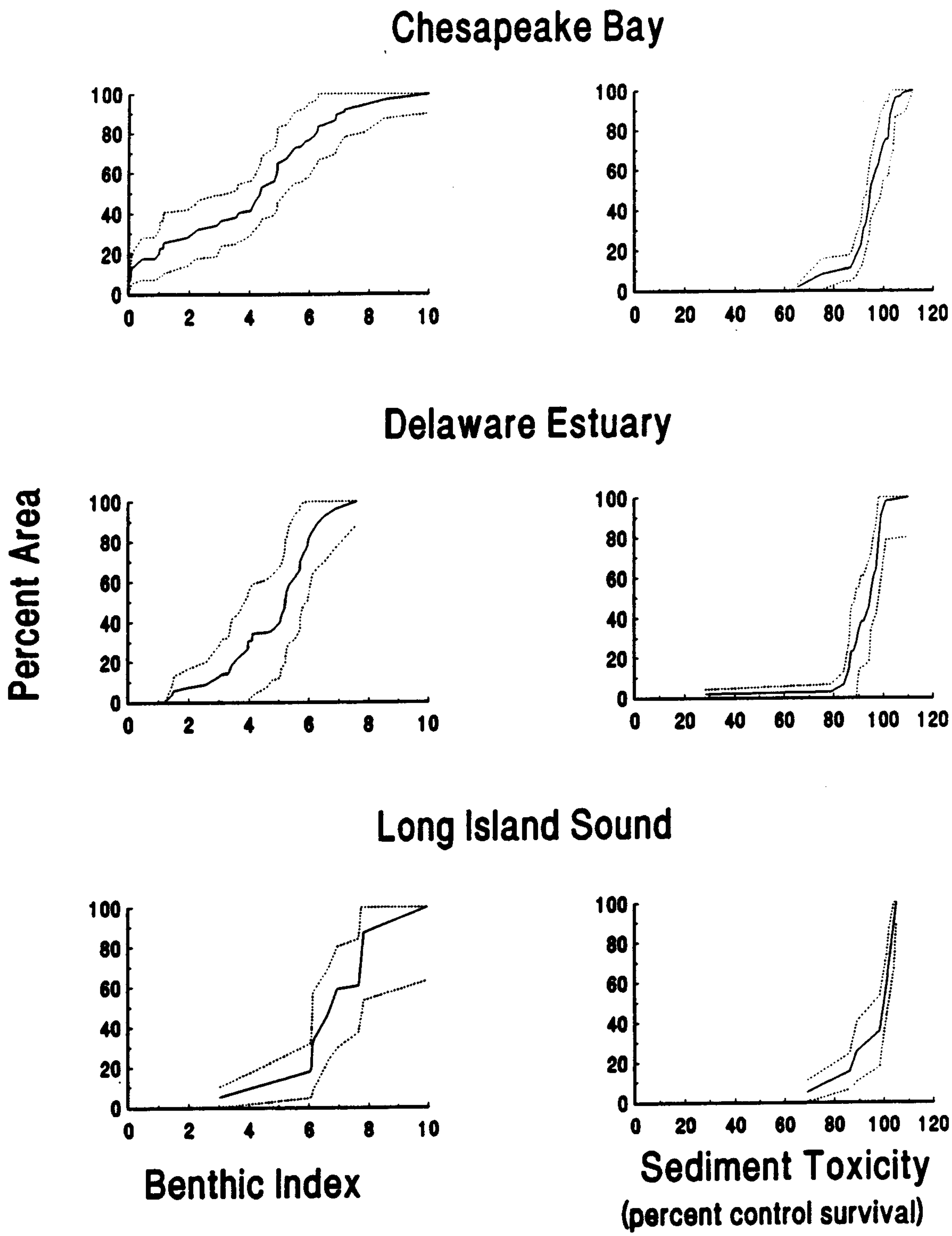


Figure 5-6. Confidence intervals (90%) for estimates of the benthic index and sediment toxicity by resource system

Table 5-5. Confidence intervals (90%) associated with the estimates around critical values of response and exposure indicators. Critical values appear in parentheses.

	Province	Large Estuaries	Large Tidal Rivers	Small Estuarine Systems	Chesapeake Bay	Delaware Bay	Long Island Sound
Benthic index (3.4)	23 ± 7	20 ± 8	46 ± 32	23 ± 14	36 ± 12	15 ± 9	5 ± 2
Sediment toxicity (<80% survival)	8 ± 5	2 ± 3	2 ± 3	32 ± 18	8 ± 4	3 ± 3	5 ± 2
Dissolved oxygen (<5 ppm)	21 ± 7	26 ± 9	19 ± 26	7 ± 5	30 ± 11	4 ± 4	51 ± 36
Clostridium (> 250 cells/1,000 gm)	9 ±4	4 ± 2	18 ± 12	21 ± 13	10 ± 4	1 ± 1	15 ± 16
Sediment metals (> ER-L values)	36 ± 9	28 ± 10	74 ± 31	52 ± 18	37 ± 10	17 ± 12	58 ± 36
Sediment Pesticides (> ER-L values)	12 ± 4	5 ± 4	36 ± 24	27 ± 14	15 ± 5	34 ± 17	16 ± 19
Gross pathology of demersal fish (# per 1,000 fish)	17 ± 8	9 ± 7	3 ± 3	36 ± 20	26 ± 6	8 ± 8	0 ± 0
Marine debris (presence)	14 ± 5	10 ± 6	32 ± 17	23 ± 16	16 ± 7	13 ± 10	23 ± 24
Water clarity (Secchi depth <1 m)	13 ± 4	7 ± 5	64 ± 30	17 ± 11	15 ± 5	35 ± 18	0 ± 0

degraded area in the small estuarine system and large tidal river classes exceed 50% of the mean (Table 5-5). The EMAP Design and Statistics Team is developing appropriate statistical methods for comparing estimates of degraded area among classes, but as a rough approximation, non-overlapping confidence intervals are required for differences to be significant. Using this criteria and 90% confidence intervals, the differences in the benthic index between the large tidal river and small estuarine system classes were not significant based on the 1990 data, even though the estimate of percent degraded area for large tidal rivers was twice that for small estuarine systems (Table 5-5). For sediment toxicity, the percent of area in small systems that exhibited unacceptable toxicity to test organisms could be distinguished statistically from that in large estuaries and tidal rivers, but only because these differences were 20-fold. Estimated CDFs for two large estuaries, the Delaware Bay system and Chesapeake Bay system, appeared quite different. With only one year of data, however, the confidence limits are too wide to consider the differences statistically significant (Fig. 5-6), even though the sampling density in the large estuary and large tidal river portions of the Delaware system was enhanced to approximately three times (four times normal in the Bay; two times normal in the river) the number of sites that would be sampled in a typical year.

These confidence intervals indicate a high degree of uncertainty associated with estimates based on a single year of data; however, EMAP's goal was never to make single-year estimates with great confidence. Instead, the intention was to derive status estimates on the basis of four-year running averages (Stevens et al. 1991; Holland 1990). The four-year running average increases sample size and expands the spatial scale of the systematic grid. It also allows for averaging over interannual variability so that status estimates are not based on a single year of potentially anomalous conditions.

Assuming that variance in 1990 was representative of that in other years, and that interannual variability is small compared to within-year variability, estimates of status for the Virginian Province using a full, four-year base period (i.e., a fourfold increase in sample size) will be approximately twice as precise as those presented in Table 5-5. Based on this assumption, the existing sampling design should be able to detect differences in the extent of degraded area as small as twofold between major subpopulations after four years. The magnitude of interannual variability and the specific procedure for combining four years of data into a single estimate of status will affect precision estimates, but these factors are presently unknown or undecided.

5.3.1 Improving the Precision of Estimates

Levels of precision varied among the three estuarine classes and among indicators (Table 5-5). Estimates for large tidal rivers generally had wider confidence intervals than large estuaries and small estuarine systems, particularly for the integrated indices. The higher degree of uncertainty for tidal rivers compared to other classes suggests that ways to increase precision within the tidal river class need to be investigated. Precision in the tidal river class could be increased simply by allocating a greater number of samples to the sampling stratum; however, unless the total amount of sampling effort for the program is increased, such reallocation would require reduced precision in another class.

Another means for increasing precision is to reconfigure the allocation scheme so that the areas of river segments are more equal in size. The size of the confidence limits produced by the Yates-Grundy estimator of variance is strongly influenced by the inclusion probabilities for tidal river segments and the indicator response values. Large differences in segment areas result in large differences in inclusion probabilities and, generally, in a large estimate of variance, regardless of the variability in indicator values. Under the present configuration, the tidal rivers are partitioned into 25 km segments. Defining segments of equal length facilitates making status estimates on the basis of river length; however, the primary basis for making status estimates is area because area represents the only practical way to integrate findings across strata of different sizes. At present, there is more than an order of magnitude difference in area among the river segments within the tidal river class. This results, in part, from inherent size differences among the river systems, but mainly from the disparity in width between the upper and lower portions of these tidal rivers.

Reconfiguring the allocation scheme within the tidal river class so that all inclusion probabilities (i.e., areas) are of similar magnitude would result in higher precision for status estimates. There are several alternatives for accomplishing this within the context of the existing design. One is to divide the river into segments of equal area, rather than equal length, and then sample each segment in each year using the existing procedure. Another alternative is to intensify the systematic grid used for large estuaries until a sufficient sample size is achieved in tidal rivers. These possibilities and other alternative allocation strategies for tidal rivers currently are being evaluated by the EMAP Estuaries Resource Group and the EMAP Statistics and Design Team.

5.3.2 Using Replicates to Improve Precision

In small estuarine systems, the precision of the estimated CDFs is a function of the number of systems sampled and the number of replicate samples obtained in each system. The precision of the estimates can be increased by allocating sampling effort optimally between systems and replicates. For example, if the largest source of variation is between systems, effort is most appropriately targeted at sampling more systems to increase precision. Conversely, if the largest source of variation occurs within systems, effort should be targeted at obtaining replicate samples within systems. An analysis was conducted to assess potential improvements in precision from reallocating sampling effort for estimating the percentage of small systems that have a benthic index value below the critical threshold (i.e., less than 3.4). In particular, the analysis addressed whether the precision associated with the benthic index response could be increased more effectively by increasing the number of sampled systems, or by increasing the number of replicates within systems.

During the Demonstration Project, benthic samples were collected from only a single location in 26 small estuarine systems, and at least two randomly selected locations were sampled in the remaining six small systems (Back River, Mystic River, Mullica River, Mattaponi River, Elizabeth River, and Indian River Bay). An estimate of the variation in the value of the benthic index between small systems was obtained from the data for all 32 sampled systems; an estimate of the average within-system variation was obtained from the six systems with replicate samples.

The approximate mean squared error (assuming that the within system variation in the benthic index response is constant across all systems) of the CDF estimate (Cochran 1977; eq. 11.27) was calculated as:

$$MSE(m,n) = \left[\frac{N^2}{n}\,(1-f_1)\,\frac{S_1^2}{A^2}\right] + \left[\frac{NS_2^2}{nA^2}\sum_{i=1}^{N}\frac{A_i^2}{m}\right]$$

where

n = number of small systems sampled

N = number of small systems in the Virginia Province (137)

f_1 = n/N

S_1^2 = $\dfrac{\sum_{i=1}^{N} A_i^2(\bar{y}_1 - \bar{\bar{y}})}{N-1}$ = between system variation

A_i = area of system i

$\bar{y}_i$ = probability of the benthic index being less than 3.4 in system i

$\bar{\bar{y}}$ = $\dfrac{\sum_{i=1}^{N} A_i\bar{y}_i}{A}$

A = $\sum_{i=1}^{N} A_i$ = total area of small systems in the Virginia Province (4875.2 km^2)

S_2^2 = within system variation

m = number of replicate samples obtained in each system

Unbiased estimates of the between system variation ($\hat{S}_1^2$) and the average within-system variation ($\hat{S}_2^2$) were obtained from the 1990 small system data. Estimates of the approximate mean squared error for various values of the number of systems sampled (n) and the number of replicates per system (m) were calculated by inserting these values and the variance estimates into the above formula. Assuming a baseline sampling strategy of 32 systems and one sample per system, the relative efficiency of adding systems and/or replicates was calculated as:

$$RE(m,n) = \sqrt{\frac{MSE(m,n)}{MSE(32,1)}}$$

This analysis indicated that the precision of the estimates was affected more by the number of systems sampled than by the number of replicates per system. If replicates were added at randomly selected locations in each of the 32 small systems sampled in 1990 (i.e., 64 total samples), the confidence intervals would shrink by only 14%. In comparison, if the 32 additional samples were allocated to different small systems (i.e., 64 systems sampled without replication), the confidence intervals would be reduced by 43%. Only eight samples in new systems would be required to achieve the 14% improvement in precision obtained by adding replicates in all 32 small systems. Sampling additional systems provides a greater increase in precision than obtaining additional replicates within systems because the variation across systems was much greater than the variation within systems. This finding is consistent with a comparison between the benthic indices at index and random sites within systems in which no significant differences were detected (see Section 5.5); however, the difference in benthic indices among small systems was large.

Although replicate samples do not appear to be the most efficient allocation of effort, discontinuing them at this time may not be warranted. Replicates provide estimates of within-system variation that are necessary for estimating the mean squared error of CDFs. Although these could be estimated from data collected during the 1990 Demonstration Project, replicate samples were obtained at only 6 of the 32 sampled stations, and only in a single year. The representativeness of conclusions drawn from these data needs to be evaluated after gathering the same type of information from additional systems and in additional years; moreover, these data need to be gathered in other provinces because the patterns observed in small estuarine systems of the mid-Atlantic may not be repeated in other areas of the country. Finally, randomly-located replicate samples were collected only in small systems during the 1990 Demonstration Project, further limiting the ability to generalize conclusions based on these data. Additional analyses are needed to determine the validity of these conclusions for large tidal rivers and large estuaries. As discussed above, replicate samples within grid cells of the large estuaries collected in 1991 can be used to conduct such analyses.

5.3.3 Improving Estimation of Precision

EMAP-E and the EMAP Statistics and Design Team are developing data analysis methods and evaluating alternative sampling designs to improve the program's ability to characterize the precision of estimates. Three issues identified during the 1990 Demonstration Project are being addressed. The first is that current estimates of variance assume a constant sample size and do not incorporate the variance due to random sample size. Using the randomized grid to select sampling sites in the large estuaries, and possibly tidal rivers in the future, results in random sample sizes; therefore, assuming constant sample sizes may result in underestimating variance. This shortcoming is being addressed by developing variance estimators that account for random sample sizes. Systematic spatial separation of samples in large estuaries is the second important issue in estimating variance. The separation is expected to reduce variance because only small-scale spatial variability will be included in the variance of estimates. The current method of variance estimation, however, represents large-scale spatial variability (i.e., differences between sites that are no closer than the spacing between adjacent points on the systematic grid). If spatial autocorrelation is substantial, the small-scale variability will be lower than the large-scale variability. Accordingly, the current method of variance estimation will produce overestimates of variance. The third issue concerns the presence of joint inclusion probabilities that are zero and the use of approximations for the joint inclusion probabilities in the Horvitz-Thompson estimator for variance. The Horvitz-Thompson variance estimator is unbiased if all joint inclusion probabilities are non-zero, and the joint inclusion probabilities for the pairs of stations actually sampled are known. The consequences of not satisfying these assumptions are not known.

To address these issues, two changes were made in the sampling conducted in the Virginian Province in 1991. A study that sampled along transects between sampling sites in the systematic grid was conducted to provide information on the magnitude of small scale sampling variability. These data will be used to evaluate the reduction in variance gained by spatial separation of samples. In addition, large estuarine systems in the Louisianian Province were sampled by randomizing the position of the overlaid grid, and then randomly selecting a sample point within the 280 km^2 area around each sampling site. Sampling in this manner will result in unbiased variance estimation (Robson et al. 1991). The two alternatives being tested in the Virginian and Louisianian provinces will provide data for determining which is most robust for long-term use in EMAP.

5.4 EVALUATION OF POWER FOR TREND DETECTION

Two classes of sampling design have been suggested to be most appropriate for long-term ecological monitoring: rotating panel designs (Duncan and Kalton 1987) and serially-alternating designs (Overton et al. 1990), like the one used in EMAP-E. The rotating panel design prescribes that a set of sites will be visited for several consecutive years, and then eliminated from future consideration. As a set is eliminated from the monitoring program, it is replaced with a new set of equal size, which then remains in the survey for a number of years. In a rotating panel design the total sample is divided evenly into a number of sets equal to the number of years a set remains in the sample. For example, in a four-year rotation, one-fourth of the sample is replaced every year. The serially-alternating design also splits the total sample into several equal-sized sets, but only one set is visited each year. A set is not revisited until all other sets have been visited, and serial revisitation is continued indefinitely. The basic serially-alternating design does not prescribe any replacement or annual revisits. Both designs can be augmented by adding a set of sites that are visited annually for the duration of the monitoring program.

The serially-alternating design was selected for use in EMAP-E based on a general linear model that compared the relative efficiency of these two designs (Urquhart et al. 1991). The linear model allowed Urquhart et al. to consider the estimation of both status and trends under various levels of population variation, measurement error, and interannual variation, and to incorporate some correlation between years and between sites measured at different times. The serially-alternating design was almost always more efficient than the rotating panel and was never less than 99% as efficient, despite the wide range of possibilities investigated. Urquhart et al. (1991) concluded that the augmented serially-alternating design offers a substantial advantage in ability to make estimates for subpopulations because more sites are visited sooner than with the rotating panel design.

Their exploration of the two designs also included an investigation of the optimal fraction of resources devoted to augmentation (annual sampling). They found that the fraction of resources devoted to annual sampling in the augmented serially-alternating design depended primarily on the two correlations and the number of years of sampling. The dependence on the variance components was substantially less. The optimal fraction increased as the site and year correlations increased and declined substantially after the first cycle of visits.

The correlation of sites over years models the identity of sites over time. If the characteristics of a site change very slowly, then the site correlation would be close to 1. If, however, the sites exhibit little identity over time, the site correlation would be closer to 0. One might expect terrestrial sites to have site correlations near 1 and estuarine sites to have somewhat lower correlations. The correlation between years reflects persistence of an effect and persistence of the source of the effect. For example, a drought might persist for several years, but once normal rainfall is reestablished, the effect of the drought may disappear quickly. Table 5-6 gives the optimal fraction after four and eight years of sampling for various values of annual and site correlation for a four-year, augmented, serially-alternating design. The optimal fractions are never more than 25% and are that large only for correlations that are probably unrealistically high for EMAP-E. Realistic considerations suggest that 10% to 15% of the total sample size allocated to annual sampling should be nearly optimal. Based on these analyses, EMAP-E initially will revisit 10% of its sites annually and will re-evaluate this proportion after several years of data have been collected from which to assess the site correlation.

Table 5-6. Optimal percent of sample size devoted to annual sampling (adapted from Urquhart et al. 1991)

Site Correlation	**After 4 Years**					**After 8 Years**				
	Correlation Between Year Effects									
	0.00	**0.05**	**0.10**	**0.25**	**0.50**	**0.00**	**0.05**	**0.10**	**0.25**	**0.50**
0.50	0	0	0	0	0	0	0	0	0	0
0.75	0	0	0	2	10	0	0	0	0	0
0.90	8	10	10	15	23	0	0	0	0	2
0.95	13	13	15	17	25	2	2	2	2	4
1.00	13	13	13	15	23	2	2	2	4	6

EMAP's goal is to detect changes of 2% per year in the amount of degraded area over a ten year period, using selected response and exposure indicators. A model was developed, and a power analysis was performed to ascertain the effectiveness of the present program for accomplishing this goal and to provide a tool for determining optimal allocation of sampling effort among resource classes. The model was based upon the assumption of a linear trend in the proportion of degraded area over three EMAP sampling cycles and was applied to the benthic index data obtained during the 1990 Demonstration Project. The benthic index was selected for the first application of the model because of the importance the program places on assessment of environmental condition based upon biological response indicators.

5.4.1 Formulation of the Model

The model used to test for trend detection was a two-factor analysis of variance (ANOVA) model with an additional term for time trend. The two factors, sites and years, involve random effects, since the specific sites selected can be viewed as a sample from the population of all possible sites, and the years observed can be viewed the same way. Interaction effects between sites and years also were included in the model. Site effects, year effects, and site-by-year interaction effects were assumed to be three independent random vectors, each multivariate-normally distributed with expectation zero and a suitable variance-covariance matrix.

The model was formulated by letting Y_{ij} denote the observed value of the response or exposure indicator at site i in year j. The basic idea underlying the variance-covariance structure is that the degree of correlation between a pair of observations, Y_{ij} and $Y_{i'j'}$, increases as the geographic distance between sites i and i', $d(i,i')$, decreases and as the time difference between years j and j', $|j-j'|$, decreases.

In addition to the site and year terms, the model included a vector of time trend terms for the years observed. These terms are not linear themselves, but were constrained to correspond to a linear trend across years in the proportion of area that is classified as degraded. This is because the main concern is not Y_{ij}, but the binary indicator variable, X_{ij}, each value of which tells whether or not the corresponding Y_{ij} is degraded.

Temporal autocorrelation in both year and site effects was incorporated into the model, following and extending the approach of Urquhart et al. (1991). Additionally, because of the extensive character of estuarine resources, spatial autocorrelation was incorporated as a power function of distance between sample sites. The model for the response indicator data Y_{ij} (e.g., benthic index) is given by the equation:

$$Y_{ij} = \mu + \theta_i + \tau_j + (\theta\tau)_{ij} + \beta(j) + \epsilon_{ij} \quad \textbf{for site} \quad i=1,\ldots,n, \ \textbf{year} \ j=1,\ldots,t$$

where

Y_{ij} = observed value of the indicator at site i in year j

μ = population mean (over all sites and years)

θ_i = random effect of site i

τ_j = random effect of year j

$(\theta\tau)_{ij}$ = random interaction effect of site i and year j

$\beta(j)$ = time trend term for year j

ϵ_{ij} = random error term for site i in year j

The covariance structure for the random effects assumed in this model was:

$$\underline{\theta} \sim MVN(0,\Sigma_\theta) \quad where\ \Sigma_\theta(n \times n) = [\rho_\theta^{d(i,i')}\sigma_\theta^2]$$

$$\underline{\tau} \sim MVN(0,\Sigma_\tau) \quad where\ \Sigma_\tau(t \times t) = [\rho_\tau^{|j-j'|}\sigma_\tau^2]$$

$$\underline{\theta\tau} \sim MVN(0,\Sigma_{\theta\tau}) \quad where\ \Sigma_{\theta\tau}(nt \times nt) = [\rho_{\tau\epsilon}^{|j-j'|}\rho_{\theta\epsilon}^{d(i,i')}\sigma_{\theta\tau}^2]$$

$\underline{\theta}, \underline{\tau},$ and $\underline{\theta\tau}$ are uncorrelated with each other

The parameters of the model are: μ, $\underline{\beta}$, σ_θ^2, σ_τ^2, $\sigma_{\theta\tau}^2$, σ_ϵ^2, ρ_θ, ρ_τ, $\rho_{\theta\epsilon}$, $\rho_{\tau\epsilon}$; the terms $\beta(j)$, j=1,...,t, are defined to correspond to a linear trend in the proportion of degraded area. The random effects θ_i (i=1,...,n), τ_j (j=1,...,t), and $(\theta\tau)_{ij}$ are random variables.

For the purpose of power calculations, multivariate normality, hence statistical independence among the three vectors of random effects, was assumed. Note that the model of Urquhart et al. (1991) can be obtained from the above model as a special case by dropping $\underline{\theta}$ and $\underline{\beta}$, and assuming no spatial covariance.

5.4.2 The Trend Detection Procedure

Investigating the question of whether a time trend is present was begun by estimating the parameters of the model from the data by maximum likelihood, then testing for linear trends using hypotheses H_0: $\beta = 0$ *vs* H_1: $\beta \neq 0$. Supplementing this analysis informally by plotting $\overline{Y_{.j}}$ against year j (for j=1,...,t) can be informative. The rest of this subsection provides a more detailed discussion of the procedure for testing for trends.

The problem was transformed into degraded/nondegraded terms because the quantity of primary interest was not Y_{ij}, but the binary indicator variable X_{ij} corresponding to Y_{ij}, which is defined by

$$X_{ij} \equiv \text{Indicator } \{Y_{ij} \leq y_0\} \equiv \text{Indicator } \{Y_{ij} \text{ is degraded}\} \text{ for all } ij$$

$$= \begin{cases} 1 \text{ if } Y_{ij} \leq y_0, \\ 0 \text{ otherwise} \end{cases}$$

For example, for the benthic index (BI) with y_0 = 3.4,

$$X_{ij} = \begin{cases} 1 \text{ if } BI_{ij} \leq 3.4, \\ 0 \text{ otherwise} \end{cases}$$

The proportion of area in year j that is degraded is $p_j = E_{i|j}(X_{ij})$; furthermore, by definition,

$$p_j = F_{i|j}(y_0) = Pr[\,Y_{ij} \text{ randomly chosen at time } j \text{ satisifies: } Y_{ij} \le y_0]$$

In this analysis, the issue of deconvolution was ignored, thus retaining extraneous variation (e.g., measurement error) in the distributions.

The values of X_{ij} were used to estimate the cumulative distribution function of the benthic index for the province at time j, $F_{i|j}(y_0)$, via

$$p_j = \sum_{i=1}^{n} w_i X_{ij} \text{ for } j = 1,\ldots,t$$

where w_i is the known sampling weight (see Sec. 2.4.2) given by

$$w_i = a_i / \Sigma_{i'}^{a_{i'}}$$

The trend in the true population value p_j over time, as the year j varies from 1 to t, was modeled by assuming that it is a linear function of j:

$$p_j = \nu + \psi j \qquad \text{for } j=1,\ldots t$$

The test procedure was to fit the generalized least squares (GLS) regression model

$$\hat{p}_j = \nu + \psi j + \delta_j \qquad \text{for } j=1,\ldots,t$$

to the data, where the departure from linearity in the trend is modeled as

$$\underline{\delta} \sim MVN(0, V) \qquad \text{with } V(t \times t) = Var(\underline{\delta}),$$

and then to test H_0: $\psi = 0$ *vs* H_1: $\psi \ne 0$.

For purposes of power calculation, the form of the cumulative distribution function F_j was assumed to be normal for each value of $j=1,\ldots,t$:

$$F_j(y_0) = \Phi\left(\frac{y_0 - (\mu + \beta(j))}{\sigma}\right)$$

so that

$$(\mu+\beta(j)) = y_0 - \sigma\phi^{-1}(\nu+\psi j)$$

where σ is the standard deviation of the full model for Y_{ij}:

$$\sigma^2 = \sigma_\theta^2 + \sigma_\tau^2 + \sigma_{\theta\tau}^2 + \sigma_\epsilon^2$$

To implement this procedure, the sequence of steps was:

(1) Obtain maximum likelihood estimates of parameters $\mu,\ \underline{\beta},\ \sigma_\theta^2,\ \sigma_\tau^2,\ \sigma_{\theta\tau}^2,\ \sigma_\epsilon^2,\ \rho_\theta,\ \rho_\tau,\ \rho_{\theta\epsilon},\ \rho_{\tau\epsilon}$

(2) Use these to obtain $\hat{V} = \hat{Var}(\underline{\hat{\delta}})$, the estimated variance-covariance matrix of $\hat{p}_j$

(3) Obtain the estimate $\hat{\psi}$ and its standard error, $\hat{\sigma}_{\hat{\psi}}$

The resulting test of H_0: $\psi=0$ vs H_1: $\psi\neq 0$ is: if $|(\hat{\psi}-0)/\hat{\sigma}_{\hat{\psi}}| > c$ (critical value), then reject H_0. The calculations needed for this test include the following:

(i) $E(X_{ij}) = Pr[Y_{ij} \leq y_0]$, which comes right from the cumulative distribution function

(ii) $E(X_{ij}X_{ij'}) = Pr[Y_{ij} \leq y_0$ and $Y_{ij'} \leq y_0]$

(iii) These expectations give $Var(X_{ij})$, $Cov(X_{ij}, X_{ij'})$, et. These, in turn, give the variance-covariance matrix of the values of $\hat{p}_j$

$$Var(\hat{p}_1,\ldots,\hat{p}_j,\ldots,\hat{p}_J) = Var(\underline{\hat{\delta}}) = V$$

(iv) Solve for $\hat{\psi},\ \hat{\nu},\ \hat{\sigma}_{\hat{\psi}}^2,\ \hat{\sigma}_{\hat{\nu}}^2,\ \hat{Cov}(\hat{\psi},\hat{\nu})$

(v) The test is then: if $|(\hat{\psi}-0)/\hat{\sigma}_{\hat{\psi}}| > c$ (critical value), then reject H_0

The power of the test is given approximately by:

$$1 - Pr\{|\hat{\psi}/\hat{\sigma}_{\hat{\psi}}| < c\} = 1 - Pr\{-c < (\hat{\psi}/\hat{\sigma}_{\hat{\psi}}) < c\}$$

$$\approx 1 - Pr\{-c-(\psi/\hat{\sigma}_{\hat{\psi}}) < (\hat{\psi}-\psi)/\hat{\sigma}_{\hat{\psi}} < c-(\psi/\hat{\sigma}_{\hat{\psi}})\}$$

$$= 1 - \phi(c-(\psi/\hat{\sigma}_{\hat{\psi}})) + \phi(-c-(\psi/\hat{\sigma}_{\hat{\psi}}))$$

where $\hat{\sigma}^2_{\hat{\Psi}}$ is taken from the variance-covariance matrix of the parameter estimates $\hat{\theta}$ and $\hat{\Psi}$:

$$(X'X)^{-1}X'VX(X'X)^{-1}$$

and X is the design matrix for a test for difference between the mean value of P from the first four years of study and the last four years of study (assuming a four year sampling cycle).

5.4.3 Component Parameter Estimates

Variance and covariance components (i.e., σ_θ, σ_τ, $\sigma_{\theta\tau}$, σ_ϵ, ρ_θ, ρ_τ, $\rho_{\theta\epsilon}$, $\rho_{\tau\epsilon}$) of the model were estimated using an ANOVA model with random effects applied to two data sets. The first data set was that collected during the Demonstration Project, supplemented by data from 10 sites that were revisited in 1991 to estimate interannual variability. The second data set was that of Dauer et al. (1989), which contains five years (1985-1989) of benthic monitoring data collected from the Virginia portion of the Chesapeake Bay. The EMAP-E samples from 1990 provided estimates of the spatial variance component (Table 5-7) based on the multiple stations that were sampled within each resource class (large estuary, large tidal river, and small estuarine systems). The stations that were revisited in 1991 provided the basis for estimates of the temporal variance component (interannual variability). Because estimates of interannual variability based on only two years of data are likely to be imprecise, a second set of interannual variability estimates was computed for comparison using the data from Dauer et al. (1989). Neither of these data sets provided sufficient information to partition interaction variance from residual variance; therefore, these two variance components were treated together as residual variance in the power analyses.

For large systems, the temporal variance component in the EMAP-E data, which was based on five stations, was not significantly different from zero; however, a significant temporal variance component was found in the Dauer et al. data (Table 5-7), which is also based on five stations. Because of the relatively small number of years contained in these data sets, neither estimate is likely to be very reliable. Unusually high temporal variability was expected in the Dauer et al. data due to the location of the stations. For these reasons, both the estimate based on the Dauer et al. data and the zero estimate from the EMAP-E data were used in the power analysis; using variability estimates from both data sets allowed examination of the upper and lower extremes of interannual variability. The EMAP-E data were used to estimate the residual variance component for large systems (Table 5-7) because this component represents variability specific to the program's sampling protocols.

Table 5-7. Estimates of variance components used in the power analysis based on EMAP data. Value in parentheses represents temporal variance estimate from the Dauer et al. (1989) Chesapeake Bay data set.

Variance Component	System Class		
	Large Estuaries	Large Tidal Rivers	Small Estuarine Systems
Spatial	2.694	3.564	2.850
Temporal	0 (1.869)	0	0
Residual	1.491	2.602	1.228

Only two stations in tidal rivers were sampled in both 1990 and 1991, and these stations were in two different rivers; therefore, the ANOVA could not be applied to this data set, and the residual variance component for tidal rivers was estimated from the Dauer et al. data set of seven stations in two rivers. Analysis of the Dauer et al. data set produced a non-significant temporal variance component; therefore, a variance estimate of zero (Table 5-7) was used in the power analysis.

For small estuarine systems, the temporal variance component in the EMAP-E data, which was based on three stations, was not significantly different from zero. The temporal variance component in the Dauer et al. data for four stations within the small estuarine system class also was not significantly different from zero. Based on these two results, the temporal variance component was set to zero for power analyses for small estuarine systems. As for large systems, the residual variance component was estimated from the EMAP-E data (Table 5-7).

Analyses of the EMAP-E and Dauer et al. data revealed no evidence of spatial autocorrelation. Visual inspection of plots of estimated covariances between responses obtained at fixed distances produced no discernable pattern with respect to distance. Temporal and interaction autocorrelations could not be estimated uniquely from the available data. Accordingly, all autocorrelation values were set to zero in the power analyses.

Predicted values of the mean benthic index value ($\mu + \beta(j)$) in any year, j, (e.g., 1990 to 2000) under the hypothesis of a 2% change in area per year were based on estimates of the proportion of degraded area in 1990. An estimate of the proportion of degraded area in each of the three resource classes in 1990 was computed as described in Section 2.4.2. Each of these estimates then was used to predict the mean benthic index value in year j as a function of the variance components:

$$\mu + \beta(j) = y_0 - Z_j\sigma$$

where Z_j is defined such that $\phi(Z_j) = P_j$

$$P_0 = \text{estimated proportion of degraded area in 1990}$$

$$P_j = P_0 + 0.02j$$

and

$$\sigma = [\sigma_\tau^2 + \sigma_\phi^2 + \sigma_{\phi\tau}^2 + \sigma_\epsilon^2]^{\frac{1}{2}}$$

5.4.4 Results from the Trend Detection Model

Based upon the parameter estimates developed from the EMAP data sets, the power analysis model indicates a high probability that the present EMAP sampling design and level of sampling effort are sufficient to achieve the goal of detecting 2% change per year in percent of the province with degraded benthic assemblages (Table 5-8). At the province level, the probability of detecting a 2% change exceeded 99%, and even the probability of detecting a 1% change per year was 95%. The power for trend detection was less at the class level than for the province but was still acceptable; the probability of detecting a 2% trend equalled or exceeded 88% for each of the classes. The power for detecting a 1% change per year was less; the probability exceeded 80% only for the large estuary class.

Power analysis based upon parameter estimates from the Dauer et al. (1989) data set provided a less optimistic picture of the ability of the existing sampling program to meet its goals for trend detection (Table 5-8). At the province level, the power for detecting a 2% change per year was estimated to be only 29%; that probability rose to 73% for detecting a 3% change per year. Use of the Dauer et al. data set did not affect power estimates for the tidal river and small estuarine system classes, but it limited the power of detecting even a 3% change per year in the large estuary class to 52%.

Table 5-8. Power for trend detection in the benthic index over three EMAP sampling cycles. Values in parentheses represent power based on temporal variance estimates from the Virginia Chesapeake Bay Monitoring Program (Dauer et al. 1989).

	Trend (% Per Year)			
	1%	**2%**	**3%**	**-1%**
Province	.95 (.22)	.99 (.29)	.99 (.73)	.99 (.36)
Large Estuaries Class	.83 (.17)	.99 (.32)	.99 (.52)	.95 (.25)
Large Tidal Rivers Class	.41	.88	.99	.44
Small Estuarine System Class	.60	.98	.99	.68

Neither the EMAP nor the Dauer et al. data sets are ideal for developing provincewide estimates of interannual variability. The EMAP data set has the appropriate spatial resolution but is based on only two years of data. The Dauer et al. data set has better temporal resolution but is limited spatially and may not represent the whole province. The area sampled in the Virginia program is characterized by unusually large interannual variation in dissolved oxygen conditions, which would exaggerate interannual variability in the quality of the benthic assemblage. The interannual variability estimated using the EMAP data set is undoubtedly an underestimate, and the Virginia value is probably an overestimate. The true number probably falls between the two.

The model development work conducted to date indicates that the present design and allocation of effort could be sufficient to detect trends at the desired level under a select set of model assumptions and parameter estimates; however, additional effort is needed to document that these assumptions are valid. Four activities are planned to improve upon the existing analysis. First, EMAP will continue to gather additional years of data to improve the parameter estimates for the trend detection model. This will be done in both the Virginian and Louisianian provinces. Parameter estimates will be supplemented by additional data sets from other areas of the country, but obtaining such data sets is difficult because so few long-term data sets exist, most do not include all the parameters (e.g. biomass) that are necessary to calculate the benthic index, and some use methods that are significantly different from those used by EMAP. Second, the power analysis will be extended to other indicators, such as sediment chemistry and dissolved oxygen, for which there are several available data sets that would be appropriate for developing estimates of interannual variability. Third, efforts will be made to improve the sensitivity of the trend detection model. The present model was based on a single type of test, a test for differences. Other tests, such as one based on a regression model, might prove to be more powerful. Fourth, the model will be applied to other types of trends. The present test examines linear trends in percent of degraded area; future model runs might examine power for trend detection in parameters such as average condition and will consider the effect of nonlinear trends.

5.5 INDEX SITES

During the 1990 Demonstration Project, nonrandom index sites were sampled in large tidal river segments and in small estuarine systems. Index sites are the locations most likely to be exposed to pollution insults (e.g., low dissolved oxygen stress or the effects of contaminated sediments), if such conditions occur. Index sites for the 1990 Demonstration Project were located in depositional, muddy environments, where fine-grained sediments accumulate (Holland 1990). Whereas randomly sampled base sites are designed to provide unbiased estimates of the areal extent of pollution effects (i.e., to provide data for estimating the extent of degraded condition), index sites are designed to determine the number of river segments or small systems that have degraded condition in habitats that are particularly vulnerable to pollution, without having to conduct intensive surveys. Together, information from base and index sites can be used to distinguish between degradation occurring in a small number of systems but a large proportion of the area of those systems from degradation manifested in a large number of systems but a small proportion of each.

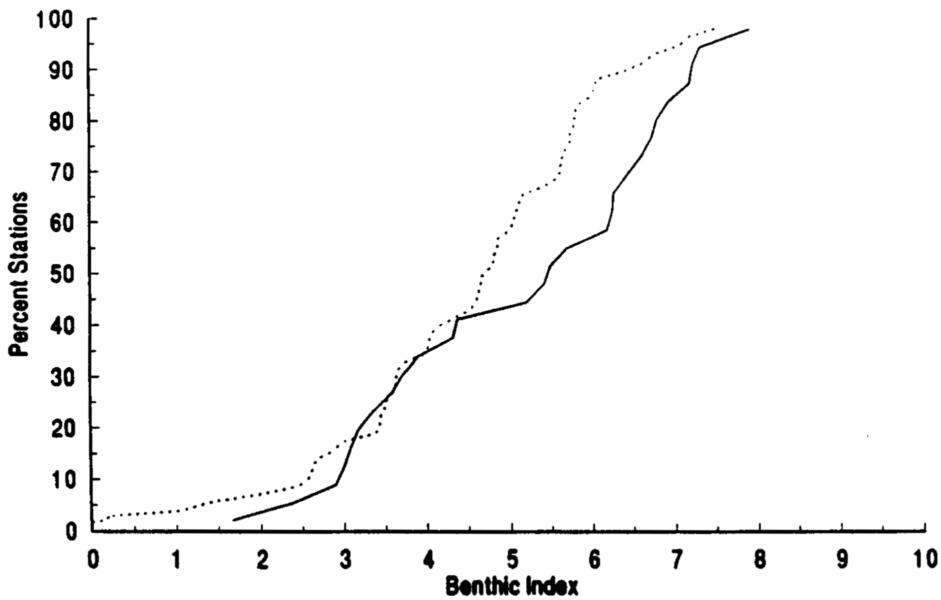
One of the goals of the 1990 Demonstration Project was to assess the value of sampling at index sites. For this assessment, benthic index values were compared between index and random sites. No statistical difference (paired t-test, $p < 0.05$) was found in either the large tidal rivers or the small estuarine systems (Fig. 5-7). The similarity of the benthic index between index and random sites indicates that there is no value added by sampling at these sites. This could be because

- degradation in small estuarine systems and large tidal rivers is widespread, affecting most of the small systems or tidal river segments; or

- depositional, muddy environments are the appropriate location to use as index sites, but identify depositional areas were not identified accurately in the 1990 sampling program (i.e., we sampled in the wrong places).

An evaluation of the sediment data from index and random sites indicated that 50% of the random sites for both small estuarine systems and large tidal rivers had finer-grained sediments than their corresponding index sites, suggesting that the criteria used in 1990 to select index sites did not identify depositional, muddy environments accurately. With the

Small Estuarine Systems



Large Tidal Rivers

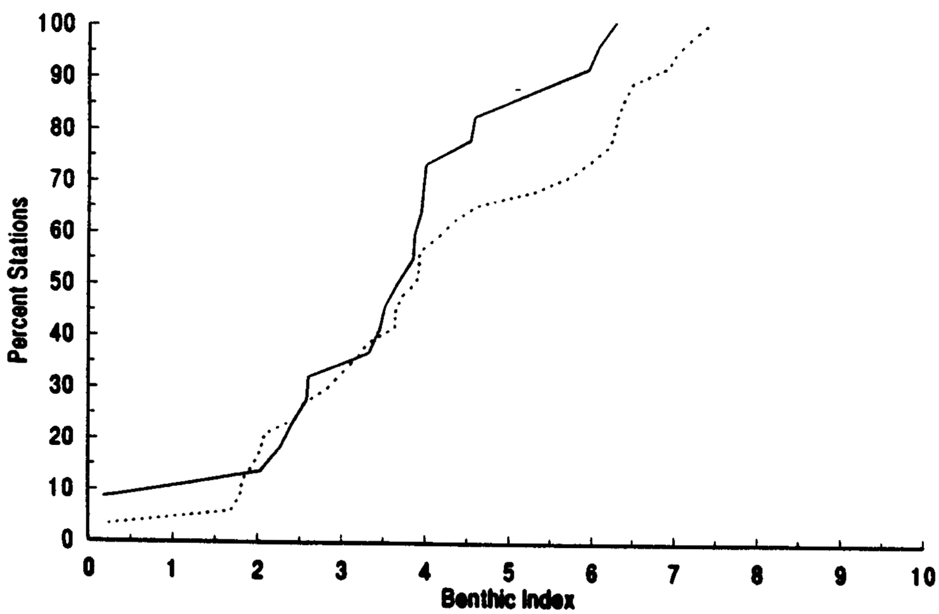


Figure 5-7. Comparisons of CDFs for the benthic index between index (solid line) and random sites (broken lines) in small estuarine systems and large tidal rivers

available data, therefore, we cannot determine if any value would be added by sampling in "true" depositional habitats. As a result, EMAP-E used available sediment maps and the scientific knowledge of local experts to define index sites in small estuarine systems for the 1991 sampling program (index sites were discontinued in large tidal rivers in 1991). The physical characteristics of sediments, sediment contaminant concentrations, sediment toxicity to indigenous biota, and the benthic index will be measured at these sites. If an evaluation of the 1991 data continues to reveal difficulty in identifying index sites that are at least as degraded as random sites, sampling at index sites will be discontinued in 1992.

5.6 DETERMINATION OF THE APPROPRIATE SPATIAL SAMPLING SCALE

The EMAP-E sampling design in the large estuary and large tidal river classes contains both systematic and random elements. One of the questions investigated in the 1990 Demonstration Project was whether the spatial scale (i.e., grid dimensions) of the systematic element was appropriate to the spatial patterns of the resources being sampled. To address this question, supplemental samples were taken in the Delaware Estuary to augment the spatial scale in that system, and the response at the higher sampling density was compared to that obtained by the base monitoring effort. The Delaware Estuary was selected for this study because it is composed primarily of the large tidal river and large estuary classes (ie., it contains proportionately little area in its tidal tributaries). In the Delaware Bay, the supplemental samples represented a fourfold increase in spatial coverage, intensifying the scale from grid cells of 280 km^2 to cells of 70 km^2. In the Delaware River, the increase was twofold, and the supplemental sites were placed half way between the base sites. The basic design provided 10 samples in the Delaware Estuary. Spatial supplements resulted in 24 additional sites.

Status estimates for four parameters (depth, substrate type, sediment toxicity, and the benthic index) calculated from the base samples were compared to those obtained when the supplemental samples were added. For all of these parameters, the CDFs produced by the base stations alone were similar to those produced when the supplemental stations were included (Fig. 5-8). The similarity of the response was greatest for the benthic index, for which the percent of area classified as degraded (i.e., an index value less than 3.4) differed by less than 10% between the two data sets. The similarity for the depth CDF was also striking, although the CDF did differ at the upper end of the range because two supplemental sites occurred at depths several meters greater than any of the base sites. For sediment toxicity, there was no difference in the range of values observed, and the difference at the threshold value used in the preliminary assessment (80% mortality) was small. For substrate type, differences in the CDFs between the two data sets were most pronounced (71% vs. 92%) at the boundary used for identifying mud sediment (less than 20% silt/clay content). Both data sets identified the system to contain a very high percentage of mud, and the difference between the two curves was well within the confidence limits of the estimates. Together, these analyses indicate that

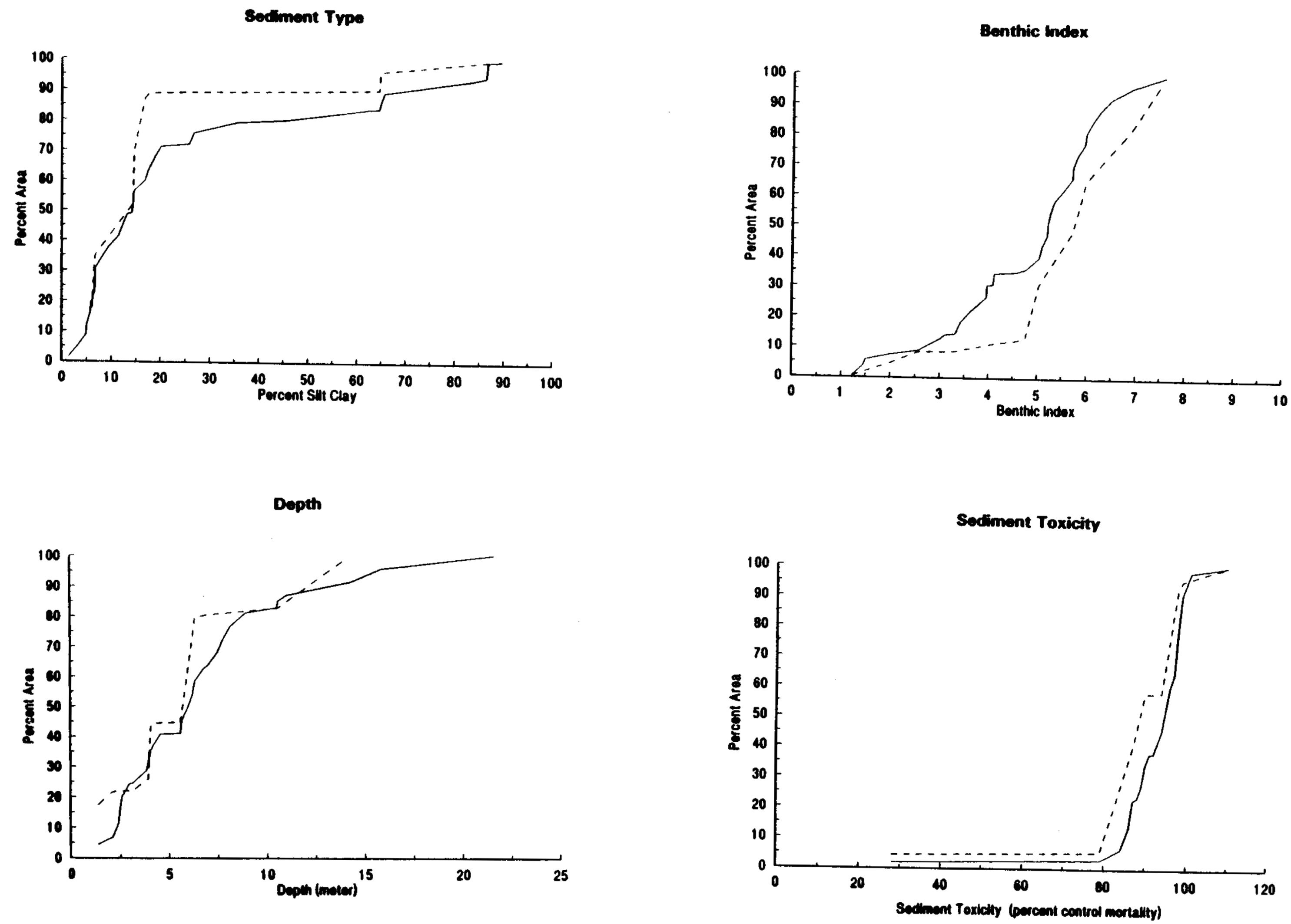


Figure 5-8. Comparison of CDFs for sediment type, benthic index, depth, and sediment toxicity, between the base (dashed lines) and intensified sampling efforts (solid lines) in the Delaware Estuary

the spatial scale of sampling in estuarine systems such as the Delaware is appropriate to the pattern of the resources being described and that, if among year variability is not extensive, the patterns observed after one year of sampling are the same as can be expected after the four-year assessment cycle is completed.

Section 6
Preliminary Estimate of Ecological Status of Virginian Province Estuaries

Introduction

The EMAP Demonstration Project in the estuaries of the Virginian Province was conducted during the summer of 1990 (June through September). A probability-based sampling design was used so that estuarine resources and characteristics were sampled in proportion to their areal distribution (Overton et al. 1990; Stevens et al. 1991). This sampling design makes it possible to estimate, with known confidence, the proportion or amount of area having defined environmental characteristics.

Five hundred sampling visits were completed at 217 sites in estuaries between Cape Cod and the mouth of the Chesapeake Bay (Fig. 6-1). A series of indicators that are representative of the overall health of estuarine resources was measured at each site. These indicators were designed to address three major attributes of concern to estuarine scientists, environmental managers, and the public: 1) biotic integrity, or the existence of healthy, diverse, and sustainable biological communities; 2) pollutant exposure, or the condition of the physico-chemical environment in which biota live; and 3) societal values, or indicators related to public use of estuarine resources. The specific methods used to measure these indicators and the calculations used to produce preliminary estimates of condition based on these measurements are presented in Section 3.

This section presents a preliminary evaluation of the condition of estuaries in the Virginian Province based upon the data collected during the 1990 Demonstration Project. Status statements of the ecological condition of the estuaries in the Virginian Province that will meet all program requirements are intended to be based on four consecutive years of monitoring information. However, preliminary status estimates can be calculated based on a single year of monitoring information.

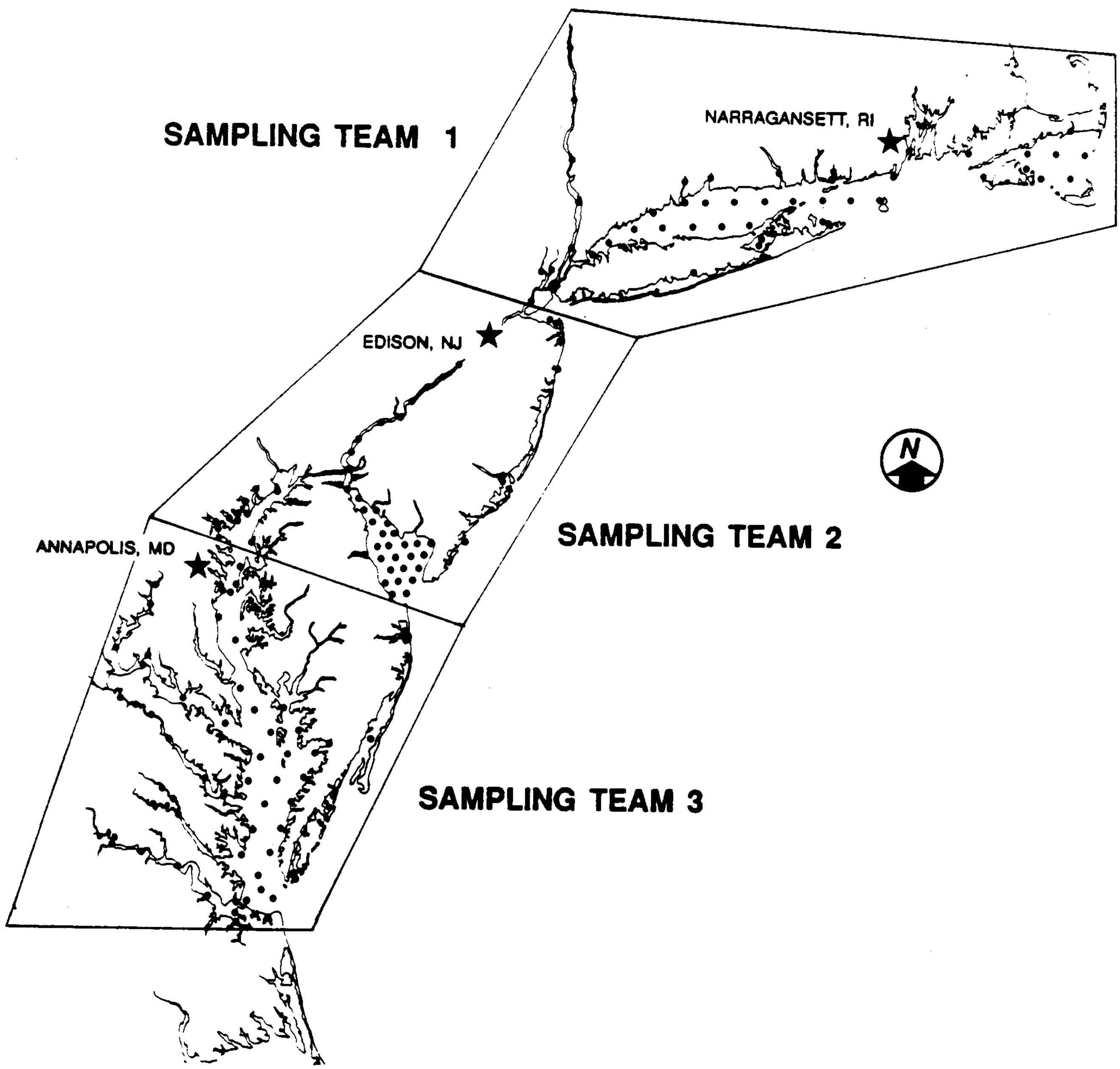


Figure 6-1. Location of the EMAP 1990 Demonstration Project sample sites in the estuaries of the Virginian Province.

This single year estimate is representative of only that particular year and will have more uncertainty associated with it than the 4-year composite estimate. The 1990 ecological status estimate represents a first attempt at presenting information from a rich and unique data set to a varied audience. The EMAP audience includes specialists with intimate knowledge of estuaries and the specific resources of the Virginian Province, and others with limited knowledge of estuarine ecology and the environmental perturbations affecting estuarine resources. Development of the approach used in this evaluation was aided by completing an example assessment report (Frithsen et al. 1991) and by a series of workshops with potential users of EMAP data. This report continues the process of identifying the questions of greatest interest to users of EMAP data and of experimenting with meaningful ways to present data and information.

Biotic Integrity

Preliminary evaluations of biotic integrity are based on two indicators: assemblages of bottom dwelling animals and the health of individual fish.

The condition of biological resources in the Virginian Province was evaluated using two indicators: one that measured the condition of bottom dwelling (benthic) animal assemblages, and one that measured the health of fish. The benthic indicator, which is discussed first, uses measures of species composition, abundance, and biomass to evaluate the condition of benthic assemblages. The fish indicator is based upon measures of visible pathological abnormalities (e.g., lesions and tumors) and reflects the response of

Benthic Index

A benthic index based upon several structural and functional properties of benthic assemblages was used as part of the preliminary evaluation. This index represents a first attempt at reducing a complex set of measurements to a simple, interpretable value. It is consistent with the EPA directive to integrate biological criteria into assessments of ecological condition. The index was developed by using discriminant analysis to identify a combination of characteristics of benthic assemblages that distinguishes reliably between regional reference sites and sites with known pollution exposure. The index has been partially validated, but several additional years of data will be required for complete validation; therefore, assessments of condition based on the index should be considered preliminary. Details on the methodology used to develop the benthic index and the validation steps completed to date are given in Section 4.

individual fish to pollutants and contaminants. Both of these indicators provide integrated measures of environmental condition in estuaries.

Benthic assemblages were used as an indicator because previous studies suggested that they are sensitive to pollutant exposure (Pearson and Rosenberg 1978; Boesch and Rosenberg 1981). They also integrate responses to exposure over relatively long periods of time (months to years). One reason for their sensitivity to pollutant exposure is that benthic organisms live in the sediment, a medium that accumulates environmental contaminants over time (Nixon et al. 1986; Schubel and Carter 1984). Their relative immobility also restricts benthic organisms from avoiding pollutant exposure and environmental disturbances.

The estuarine area in the Virginian Province having benthic resources with poor community structure is estimated to be between 16-30%.

Preliminary estimates based on the 1990 Demonstration Project indicate that 23% (± 7%) of the estuarine area in the Virginian Province had benthic resources characterized by low species richness, low abundance of selected indicator species, and low mean weight for selected indicator species. This benthic community type is referred to herein as "degraded"; however, "degraded" does not imply anthropogenic causes of the observed condition. The "degraded" ecological condition may be due to natural environmental conditions. Of the approximately 23,574 km^2 (9,102 mi^2) of estuaries in the Virginian Province, about 5,422 km^2 were characterized by benthic communities with low species numbers and low abundance of indicator species. The total degraded area was about one and a half times the size of Rhode Island.

"Degraded" Conditions

For the benthic index, "degraded" conditions were defined relative to regional reference sites, as described in Section 4. This definition is relative to present conditions and may be conservative in estimating the extent of degradation relative to historical conditions. At the minimum, however, the definition provides a benchmark against which future conditions can be measured.

Although EMAP's primary objective is to describe status and trends at the province level, estimates can also be generated for subpopulations. The EMAP sampling design defined three classes of estuarine resources according to surface area and shape: large estuaries, large tidal rivers, and small estuarine systems. These classes were defined because estuaries of different sizes may respond differently to anthropogenic impacts. Large estuaries like Chesapeake Bay, Delaware Estuary, and Long Island Sound typically have large, complex watersheds and are affected by multiple environmental stresses and contaminant sources. Large tidal rivers, such as the Delaware and Potomac rivers, also drain complex watersheds, but their geometry (length-to-width ratio) enhances interactions with bordering terrestrial systems, including urbanized areas. In addition, flushing rates in large tidal rivers are typically faster than in large estuaries. Small estuarine systems such as New Bedford Harbor and Barnegat Bay typically have small watersheds and less diverse contaminant sources than large estuaries.

Large tidal rivers had a greater estimated proportion of "degraded" benthic resources than other types of estuaries; however, the absolute area of "degradation" was greatest in large estuaries.

The prevalence of "degraded" benthic resources was dissimilar among the three classes of estuaries sampled during the 1990 Demonstration Project. Proportionately, large tidal rivers were the most "degraded"; 46% (± 32%) of the area of large tidal rivers in the Virginian Province was estimated to have "degraded" benthic resources (Fig. 6-2). The proportion of

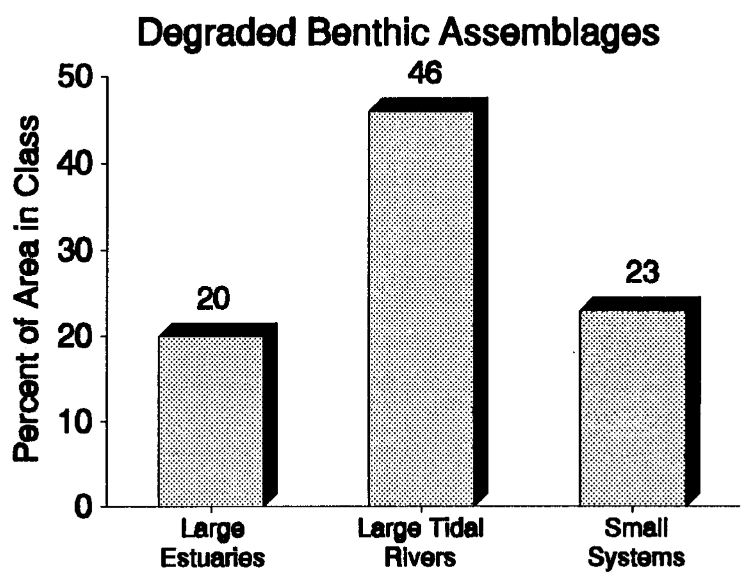


Figure 6-2. Estimated percent of estuarine area in the three classes of estuarine resources having degraded benthic communities (see Table 5-5 for levels of uncertainty)

area within large estuaries containing degraded benthic resources was less than half that of tidal rivers; however, owing to the larger size of this class, the actual area of degradation was greatest in large estuaries. In large estuaries, the total area with degraded benthic resources was about 3,365 km^2, compared to only 839 km^2 in large tidal rivers and 1,119 km^2 in small estuarine systems.

Less than 0.1% percent of the fish examined that are commercially or recreationally harvested had visible pathological disorders.

The second indicator used to assess the biological integrity of estuaries was the occurrence of visible pathological problems, such as tumors and lesions, in fish. Although several factors may contribute to the occurrence of such disorders, they most frequently occur in response to high contaminant concentrations in the environment. Studies have shown that conditions such as fin erosion, skin tumors, and ulcers are most prevalent in polluted habitats (O'Connor et al. 1987; Buhler and Williams 1988). Unlike benthic organisms, fish are highly mobile and can move out of contaminated estuarine areas; therefore, ascribing an areal extent to estimates of pathological abnormalities is difficult. Instead, pathological disorders are expressed in terms of prevalence within a region, water body, or class of water bodies.

Fish examined that are closely associated with bottom sediment had a greater prevalence of pathological disorders.

Less than 1% (4 fish in 1,000) of the fish examined during the 1990 Demonstration Project had visible pathological disorders (Fig. 6-3) establishing an estimated background level for gross pathologies at about 0.2%-0.7%. The prevalence of abnormalities was even lower (less than 1 in 1,000) among commercially and recreationally harvested species. Pathological disorders were more prevalent in fish that feed on or are associated with sediments (demersal fish). The prevalence of pathological disorders in demersal fish was 17 (± 8) fish per 1,000.

Visible pathological disorders in fish examined were most prevalent in small estuarine systems.

The prevalence of visible pathological disorders differed substantially among the three classes of estuarine resources. Thirty-six (± 20) of every 1,000 demersal fish in small estuarine systems had pathological disorders; this rate was significantly greater than in large estuaries or large tidal rivers (Fig. 6-4).

Precision of Estimates
One of the goals of EMAP is to provide estimates of status with a known and acceptable degree of confidence. Confidence intervals for the description of estuarine status presented here appear in Section 5. Data collected during the 1990 Demonstration Project are being used to evaluate the allocation and level of sampling intensity necessary to meet the precision needs of the program.

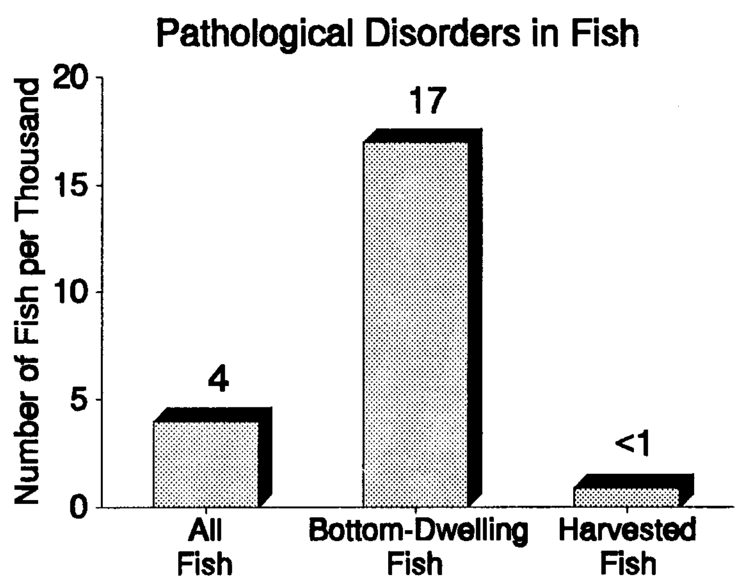


Figure 6-3. Prevalence of fish with pathological disorders in the Virginian Province. Presented for all fish, bottom dwelling, and commercially or recreationally harvested fish

Pollutant exposure was measured with three indicators: dissolved oxygen, sediment contaminants, and sediment toxicity.

Pollutant Exposure

Although EMAP's major objective is to describe the status of estuaries using indicators of ecological condition, environmental managers are also interested in descriptions of the extent and magnitude of pollutant exposure. Measures of pollutant exposure historically have been the mainstay of

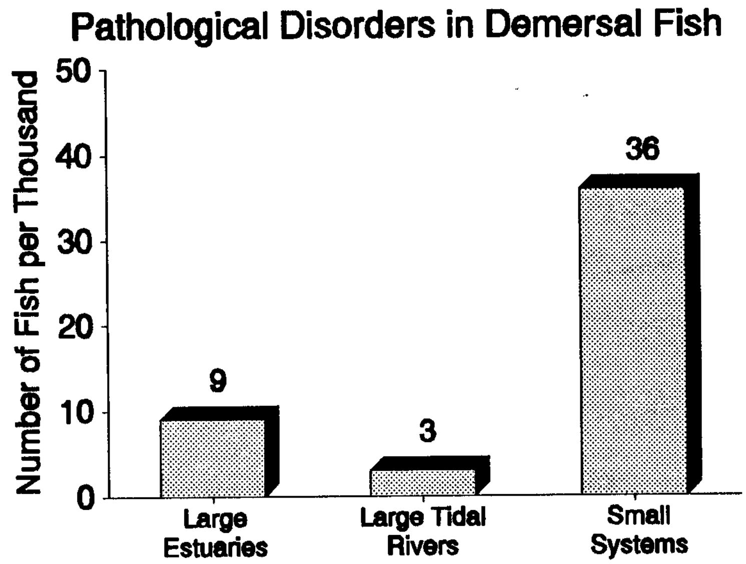


Figure 6-4. Prevalence of pathological disorders in demersal fish in the three classes of estuarine resources.

environmental monitoring programs. Indicators of pollutant exposure measured during the 1990 Demonstration Project were water column dissolved oxygen concentrations, sediment toxicity, and the concentration of contaminants in sediments.

Bottom dissolved oxygen concentration below 5 ppm, the water quality standard for many states in the province, was estimated to occur in 14-28% of the province.

Two to sixteen percent of the estuarine area in the province was estimated to have oxygen concentrations below 2 ppm, which is considered extremely stressful to biota.

Dissolved oxygen is a fundamental requirement for estuarine organisms. A threshold concentration of 5 ppm is used by many states to set water quality standards. Bottom waters in 21% (± 7%) of the estuarine area of the Virginian Province had dissolved oxygen concentrations that failed to meet this criterion (Fig. 6-5). A concentration of approximately 2 ppm often is used as a threshold for oxygen concentrations thought to be extremely stressful to most estuarine biota. Results from the 1990 Demonstration Project

> **Oxygen Measurements**
> Two types of dissolved oxygen measurements were made during the Demonstration Project: point measurements and continuously-recorded measurements. Estimates of dissolved oxygen conditions in this report are based on the point measurements. The continuously- recorded data were collected at a subset of sites to determine whether EMAP can cost-effectively gather additional information for estimating measures that cannot be estimated from a single point measurement, such as percent of time below a critical value. Analyses of the continuously-recorded data are presented in Section 4.

indicate that bottom water dissolved oxygen concentrations below this threshold were found in 9% (± 7%) of the Virginian Province (Fig. 6-5).

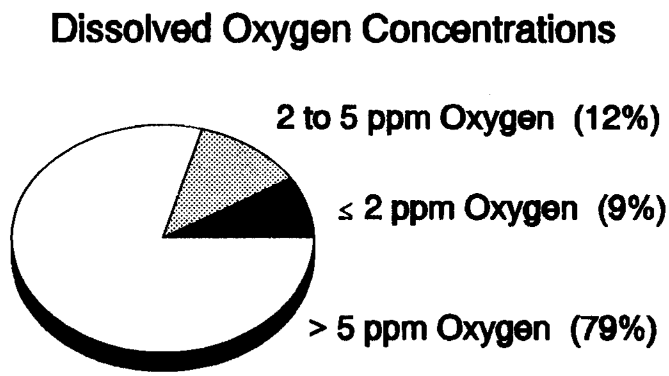


Figure 6-5. Estimated percent of estuarine area in the Virginian Province with bottom dissolved oxygen concentrations below 2 ppm and 5 ppm.

Previous studies have shown that the probability of finding low dissolved oxygen concentrations is greater in areas where there is density stratification of the water column. This occurs because stratification reduces exchange between bottom waters and productive surface waters, which are generally more oxygen-rich due to phytoplankton production and diffusion from the atmosphere. Results from the 1990 Demonstration Project are consistent with these findings. More than half of the area having oxygen concentrations below 2 ppm was strongly stratified.

Natural vs. Anthropogenic Degradation

No distinction is made in this report between degraded conditions brought on by anthropogenic activities and those due to natural causes. For indicators such as dissolved oxygen, identifying degraded environmental conditions due solely to anthropogenic inputs implies a degree of understanding of causes and effects that is not currently available. Low oxygen concentrations usually occur as a result of organic enrichment fueled by excess nutrients (eutrophication). Low oxygen concentrations may also occur simply as a result of prolonged water column stratification and minimal advective flow, as in many meromictic lakes.

Two indicators of the potential effects of contaminants on biota were measured during the Demonstration Project. The first was a bioassay for acute toxicity in which estuarine biota were exposed to the sediments under controlled laboratory conditions. The second was direct measurement of contaminant concentrations in the sediment.

Between three and thirteen percent of the area in the Virginian Province were estimated to have sediments that were toxic to estuarine organisms.

Sediment bioassays are the most direct measure of contaminant-induced effects on biological communities (Chapman 1988). Mortality in these laboratory exposure tests provides evidence of toxic contamination without requiring interpretation of how complex mixtures might interact to affect biota. Based upon results of bioassays, 8% (± 5%) of the Virginian Province were estimated to contain sediment that was toxic to estuarine organisms.

Direct measurement of contaminants complements the bioassay by identifying the compounds most likely to have produced the toxicity. The most important contributors to acute toxicity were three metals, lead, mercury, and zinc. The concentrations of these metals exceeded the threshold concentration for biological effects (ER-L) defined by Long and Morgan (1990) at 90% of the sites with acute toxicity. In contrast, only 17% of the Demonstration Project sites where there was mortality in bioassays had concentrations of an organic contaminant that exceeded the ER-L value.

Toxicity Tests

Ten-day acute toxicity tests were conducted using the amphipod *Ampelisca abdita*. This was the most abundant benthic species identified in the EMAP samples. It cannot tolerate exposure to salinities less than 15 ppm for prolonged periods; therefore, all toxicity tests were run at a standard salinity of 30 ppt. To evaluate the sensitivity of this test to contaminants in brackish water, selected sediments were also tested using the freshwater amphipod *Hyallela azteca*. These tests confirmed *Ampelisca's* sensitivity to contaminants in brackish water. These results are presented in Section 4.

Sediments in 30-48% of the province were estimated to have contaminants at concentrations that could potentially cause sublethal effects in biota.

Direct measurement of sediment contaminant concentrations also provides an early warning of pollutant exposure. Whereas sediment bioassays measure acute toxicity, direct measurement of contaminant concentrations can be used to evaluate the potential for sublethal biological effects. Currently, however, contaminant concentrations that are likely to produce sublethal effects are not well-defined. For the Demonstration Project data, the potential for sublethal effects from sediment contamination was evaluated by comparing contaminant concentrations from all sites to ER-L values (Fig. 6-6). ER-L values represent concentrations at which any

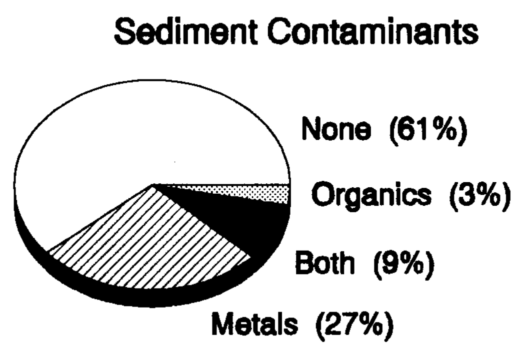


Figure 6-6. Estimated percent of area in the Virginian Province with sediment contaminant concentrations greater than Long and Morgan (1990) ER-L values.

type of biological effect (sublethal and some lethal) was noted in at least 10% of contaminant exposure studies. Based on this approach, 39% (± 9%) of the Virginian Province had concentrations of contaminants that have the potential to cause at least sublethal effects in biota. The appearance of no contaminants in 61% of the sediments of the Virginian Province (Fig. 6-6) refers to the absence of measurable quantities of the contaminants listed in Table 2-7. Other unmeasured contaminants may exist in these areas.

Metals were the most prevalent contaminants at concentrations potentially leading to biological effects. Thirty-six percent of the area contained elevated concentrations primarily of lead, nickel, and zinc. Organic contaminants at concentrations of potential biological concern were found in 12% (± 4%) of the province. Of the organic contaminants, chlorinated pesticides were the most prevalent; polyaromatic hydrocarbons (PAH) and polychlorinated biphenyls (PCB) were found at biologically relevant levels in less than 1% of the province. Estimates of areal extent of the individual contaminants measured in the program are provided in Section 4.

Metals vs Organic Contaminants

The relative prevalence of different classes of contaminants in the environment is an important ecological question since many of these classes come from different sources and require different management strategies. Although the data here suggest that metals are more prevalent at biologically-relevant concentrations than organic contaminants, this conclusion should be considered preliminary. Biologically critical values for many of the organic contaminants are in the part per billion range and, in many cases, are lower than the laboratory detection limit achieved in the first year of EMAP (see section 3); thus, these are minimum estimates of organic contamination. Section 4 provides a thorough discussion of the relative prevalence of organic and inorganic contaminants based on alternative critical values that are less sensitive to detection limits.

Estimated exposure to low dissolved oxygen was greatest in large tidal rivers; toxic sediments were most prevalent in small estuarine systems.

The types of pollutant exposure differed substantially among the three classes of estuarine resources. Dissolved oxygen stress was most prevalent in large estuaries and tidal rivers (Fig. 6-7), whereas toxic sediments were most prevalent in small estuarine systems. Toxic sediments were estimated to be found in 32% (± 18%) of the area of small estuaries. In comparison, 2% (± 3%) of the sediments in large estuaries and tidal rivers were estimated to be toxic. Contaminants at concentrations that potentially result in chronic toxicity were most prevalent in tidal rivers and small estuarine systems (Fig. 6-7). Highest concentrations of contaminants, however, were most prevalent in small estuarine systems, consistent with the high prevalence of acute toxicity found in this resource class (see Section 4).

Specific Estuarine Systems

EMAP was designed primarily to develop regional estimates of condition, and the level of sampling effort is commensurate

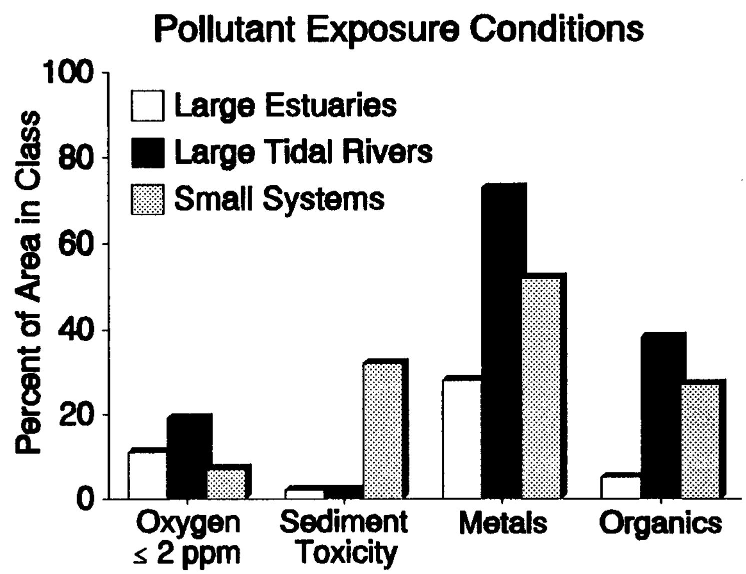


Figure 6-7. Pollutant exposure conditions for the three classes of estuarine resources

with that goal. Based upon present sample allocation, EMAP will also have a sufficient number of samples to evaluate the condition of three individual estuarine systems in the Virginian Province – Chesapeake Bay, Delaware Estuary, and Long Island Sound – after completing a four-year sampling cycle. To meet the needs of other regional and state programs, the EMAP sampling design may be enhanced to evaluate estuarine resources of any size. EMAP is currently developing partnerships with regional and state

> **EMAP Sampling Cycle**
> As currently envisioned, EMAP will make assessments of environmental status using a four-year cycle of monitoring data. Multiple years of data afford a greater number of sample points and minimize the effects of natural interannual variation due to climate and other influences. The number of sample points after four years will result in greater confidence in the estimates than is possible after a single year. This is particularly relevant when only a portion of the sampling frame is used for estimation, such as in the examination of individual estuaries. Complete assessments will use results from the 1990 Demonstration Project and the three years that follow to establish baseline conditions for the estuaries of the Virginian Province.

resource managers to design and implement enhanced sampling in specific estuaries.

The three systems for which estimates can be made differ substantially in their physical characteristics. Long Island Sound is the deepest; over 50% of its area was estimated to be 20 m or deeper compared to estimates of less than 5% of the Chesapeake and Delaware systems. The Delaware Estuary had the greatest proportion of estimated sand substrate and the least proportion of mud habitat. Less than 10% of the Delaware was estimated to contain mud substrate, whereas mud accounted for 32% of Long Island Sound sediments (Fig. 6-8). The Delaware was also the most turbid. Whereas water with visibility of greater than 1 m constituted an estimated 85% (± 5%) of the area in Chesapeake Bay and nearly 100% of Long Island Sound, it occurred in less than 65% (± 18%) of the Delaware.

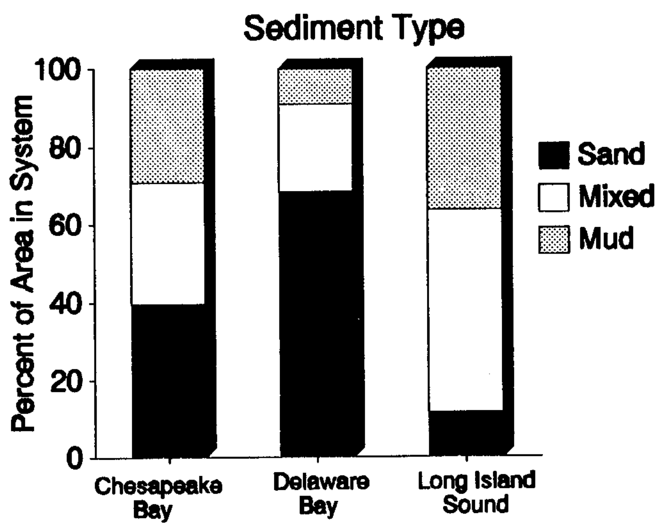


Figure 6-8. Sediment types in the three major estuarine systems in the Virginian Province

Long Island Sound was characterized by the relative absence of brackish and transitional salinity waters; an estimated 95% of its area contained marine waters (Fig. 6-9). Brackish and transitional waters each accounted for at least 10% of the area in the other two estuaries; transitional waters constituted more than 50% of the area in Chesapeake Bay. The degree of water column stratification was determined from delta sigma-t, a measure used in physical oceanography to describe density differences between surface and bottom waters. Long Island Sound was the least stratified of the three estuaries. One-fifth of the area in the Chesapeake and Delaware systems had a delta sigma-t of four or greater, indicating strong density stratification. Delta sigma-t was not

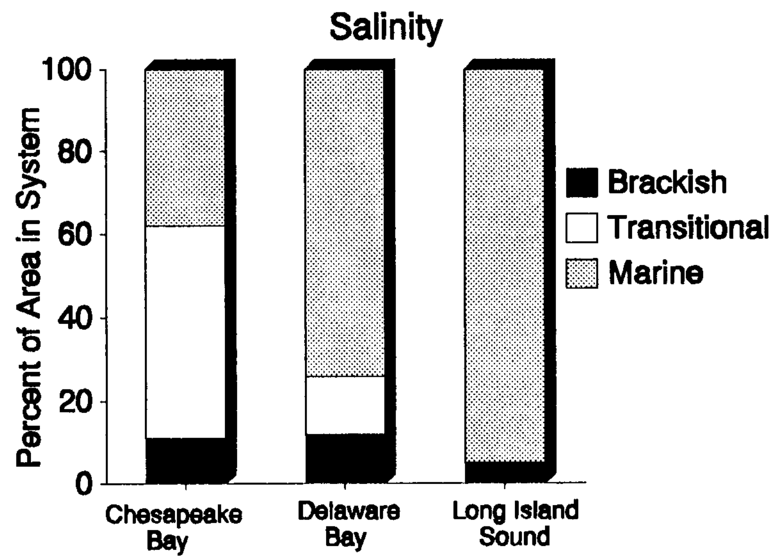


Figure 6-9. Salinity habitats in the three major estuarine systems in the Virginian Province: brackish (0 to 5 ppt), transitional (5 to 18 ppt), and marine (> 18 ppt).

greater than three anywhere in the Long Island Sound. Presumably, the absence of a large transitional salinity zone and the lesser degree of water column stratification reflect fewer major tributaries providing freshwater input to Long Island Sound than in the other two major systems in the Virginian Province.

Of the largest estuarine systems in the Virginian Province, Long Island Sound had the highest estimated proportion of area with oxygen concentrations less than 5 ppm; Chesapeake Bay had the highest proportion below 2 ppm.

Long Island Sound had the largest estimated percent of area (51% ± 36%) with bottom dissolved oxygen concentration less than 5 ppm (Fig. 6-10). These data support previous findings of degraded conditions in the western basin of this estuary (Parker and O'Reilly 1991; Welsh and Eller 1991). Although the Chesapeake Bay had a smaller estimated percentage of area with bottom DO concentrations less than 5 ppm, 19% (± 11%) of its area had concentrations less than 2 ppm compared to less than 1% (± 7%) for the rest of the province. The lower concentrations in the Chesapeake Bay are due, in part, to the greater degree of water column stratification especially in deeper, central portions.

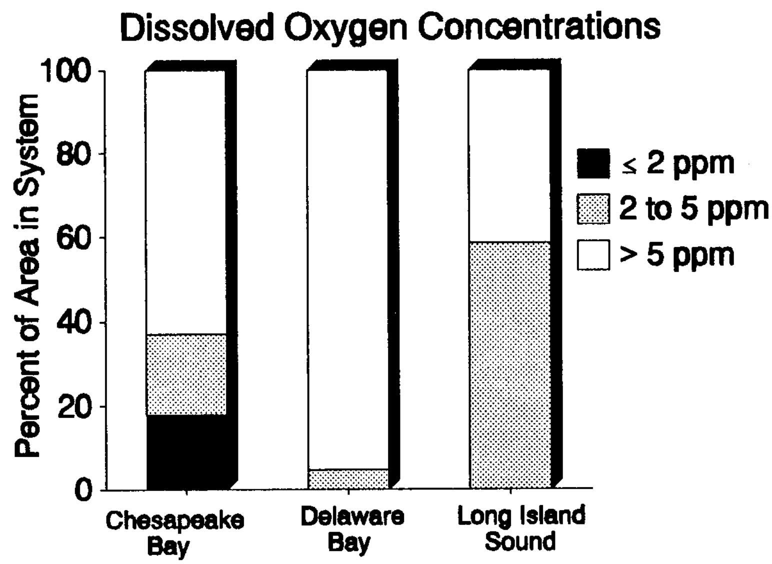


Figure 6-10. Bottom dissolved oxygen conditions in the three major estuarine systems in the Virginian Province.

> Critical Values
>
> Dissolved oxygen conditions were described by comparison with two critical values, 2 ppm and 5 ppm. The first value was selected because of its perceived biological importance, as determined in laboratory exposure studies, whereas the second was selected because it is used as a water quality standard in many states. The biologically important value is species-dependent, and the regulatory value varies among states in the province. One of the strengths of EMAP is that an unbiased estimate for the amount of area below any critical value can be generated with known confidence. EMAP will continue to interact with its clients to identify the most appropriate critical values for evaluations of ecological condition.

Of the largest systems, sediments that are estimated to be toxic to biota were most prevalent in Chesapeake Bay; contaminants at concentrations likely to cause sublethal effects were estimated to be most prevalent in Long Island Sound.

Sediment toxicity was most widespread in Chesapeake Bay. Sediment from an estimated 8% (± 5%) of its area was toxic to amphipods in bioassays. This was about double the percent of area with toxic sediments found in the Delaware Estuary and in Long Island Sound.

Sediment Contaminants
Sediment quality criteria for contaminant concentrations are being developed by EPA but are not yet available. The ER-L values of Long and Morgan (1990), which are concentrations that cause biological effects in 10% of exposure tests, were used as criteria for defining high concentrations of contaminants. The effect of selecting alternative critical values to estimate the percent of area with high contaminant concentrations, as well as a discussion of the interpretation of these alternative critical values, is presented in Section 4.

Sediment contaminants at concentrations that potentially cause at least sublethal biological effects (i.e., above ER-L values) were most widespread in Long Island Sound, covering an estimated 58% (± 36%) of the area. In contrast, only 39% (± 19%) of the Delaware estuary and 37% (± 10%) of Chesapeake Bay had similarly elevated sediment

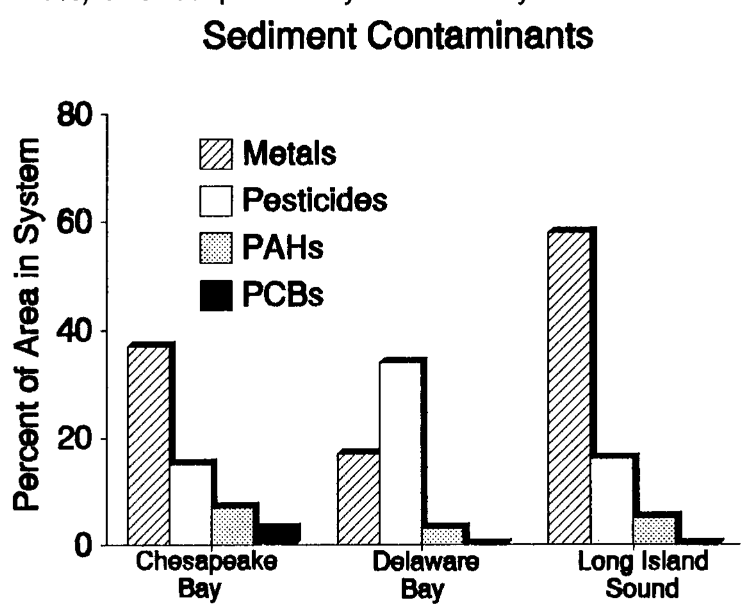


Figure 6-11. Prevalence of four classes of contaminants in the three largest estuarine systems in the Virginian Province.

contaminant levels. In Long Island Sound and Chesapeake Bay, heavy metals were more prevalent than organic contaminants (Fig. 6-11). In the Delaware, however, concentrations of chlorinated pesticides, notably dieldrin, were above ER-L values over an estimated 34% (± 19%) of the area and were more prevalent than high concentrations of metals.

Between one-quarter and one half of the area in Chesapeake Bay had benthic resources characterized by low number of species and low abundance of indicator species.

Thirty-six percent (± 12%) of the area of Chesapeake Bay had "degraded" benthic resources characterized by low number of species and low number of indicator species (Fig. 6-12). Only 15% (± 9%) of the area in the Delaware and 5% (± 2%) in Long Island Sound had similar "degradation". Much of the area in Chesapeake Bay with poor benthic assemblages occurred in locations where dissolved oxygen stress was most severe. Several authors have suggested that some of this degradation may be natural rather than anthropogenic, resulting from dissolved oxygen depletion induced by stratification (Officer et al. 1984).

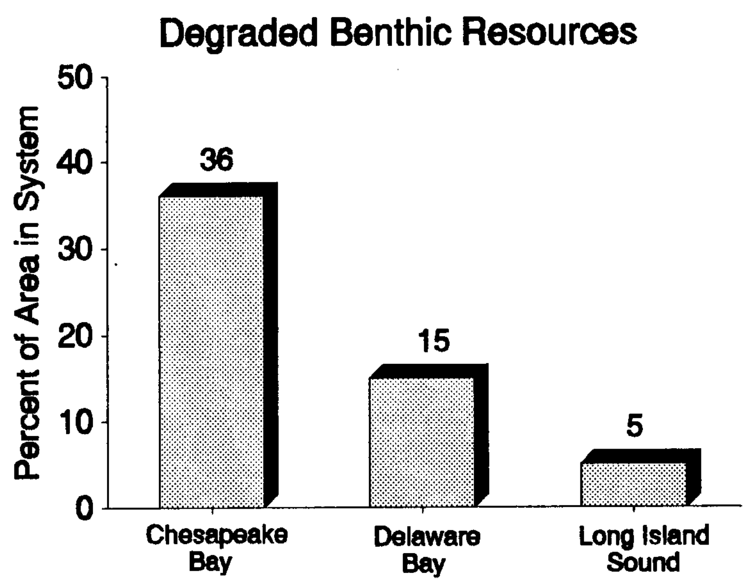


Figure 6-12. "Degraded" benthic resources in the three largest estuarine systems in the Virginian Province. (See Table 5-5 for uncertainty).

Specialized Habitats

EMAP's sampling design makes it possible to post-classify monitoring data to make unbiased estimates of environmental condition for specific estuarine areas or habitats of interest to estuarine resource managers and scientists. Examples of specific subpopulations of interest may include those defined by salinity or substrate characteristics. The ability to post-classify is limited only by the number of samples available in each subpopulation.

Muddy sediments represent a specialized habitat of interest. Previous studies have shown that both inorganic and organic contaminants have a greater affinity for binding to fine-grained sediments (NOAA 1988, 1991). Because of their tendency to accumulate contaminants, these sediments generally have the greatest potential to be toxic to estuarine organisms. Fine-grained sediments (those with a silt-clay content greater than 20%) constituted an estimated 58% of the estuarine area in the Virginian Province.

Toxicity and elevated contaminant concentrations were most prevalent in fine-grained sediments.

Demonstration Project results confirm that elevated sediment contaminants and sediment toxicity are associated with fine-grained sediments. Sixteen percent of the area with fine-grained sediments was toxic, compared to less than 1% of the area with sandy sediments. Ninety-nine percent of all toxic sediments and 91% of sediments having elevated concentrations of contaminants were fine-grained.

Degraded benthic assemblages also were more prevalent in fine-grained sediment habitats, possibly because of the greater potential for increased contaminant concentrations in this type of habitat. Seventy-four percent of the estuarine area with degraded benthic assemblages had fine-grained sediments. Only 14% of all habitats having coarse sediments (those with a silt-clay content less than 20%) contained degraded benthic communities.

Salinity describes another specialized habitat of interest because it is an important factor controlling ecological processes and the distribution of organisms (Remane and Schlieper 1971). Sixty-seven percent of the estuarine area in

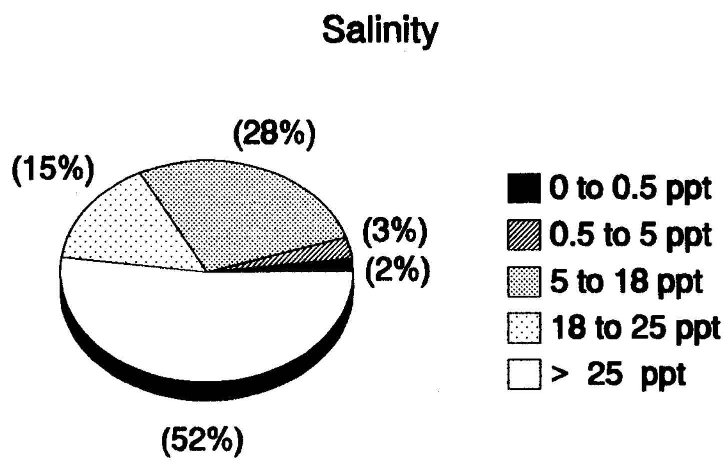


Figure 6-13. Salinity habitats in the Virginian Province as a percent of total estuarine area.

the province had bottom salinities typical of marine environments (greater than 18 ppt) (Fig. 6-13). Twenty-eight percent of the province had salinities characteristic of transitional regions between fresh and marine waters (5 to 18 ppt salt), and 5% of the estuarine area in the province had brackish water (0 to 5 ppt salt).

Tidal freshwater habitats, which occur in only 2% of the province, were proportionately more degraded than other salinity habitats.

Tidal freshwater is a particularly important habitat because it is the spawning grounds for anadromous fishes such as striped bass. Although tidal freshwater composed a small percentage of the area of the province, the benthic index suggested that it was proportionately more degraded than other salinity habitats. Forty-one percent of the tidal freshwaters in the province contained degraded benthic assemblages compared to less than 2% of the remaining area.

Ninety percent of the transitional salinity area in the province, which is critical habitat for oyster production, occurred in Chesapeake Bay.

Management decisions can be influenced not only by the proportion of a specific habitat type that is degraded, but also by the distribution of that type of habitat. Approximately 28% of the area in the province had transitional salinity (5-18 ppt). Of this transitional area, approximately 90% is in the Chesapeake Bay system. Transitional habitats support spawning and nursery activities of many important estuarine species, such as blue crab and oysters.

Associations

Associations

In this report, associations are identified between pollution exposure and biological response indicators. In the future, EMAP intends to examine associations between environmental condition and external stresses to the environment. Measures of external stress include parameters such as human population density, land use, industrial activity, and pollution discharge, many of which are presently monitored as part of other programs. EMAP, in partnership with other agencies, will compile this information for use in identifying associations and make it available in an integrated data base to interested users.

One strength of the EMAP sampling design is the co-location of many types of measurements at each site. Co-location of data facilitates identification of associations between biological responses and measures of pollutant exposure. Although these associations do not define cause and effect, they can be used to gauge how biological response indicators reflect measures of pollutant exposure and to formulate hypotheses concerning causal relationships.

Fifty percent of the area estimated to have "degraded" benthic resources was associated with low dissolved oxygen, whereas only 12% was associated with toxic sediments.

Based on association, low dissolved oxygen concentration appears to contribute more to the degradation of benthic resources than does sediment toxicity. Fifty percent of the area with degraded benthic assemblages had concentrations of dissolved oxygen less than 5 ppm in bottom waters (Fig. 6-14). In contrast, toxic sediments were found in only 11% of the area with degraded benthos.

Thirty-nine percent of the area with degraded biological resources could not be associated with either low dissolved oxygen or toxic sediments. Some of this area may have had low dissolved oxygen exposure, but not at the time samples were collected. About half of this area contained sediment contaminant concentrations with the potential to cause sublethal biological effects. Degradation may have resulted from pollutant exposure from a contaminant that was not measured during the 1990 Demonstration Project. As EMAP evolves, these areas will be the subject of special studies to identify any new or emerging types of environmental exposure.

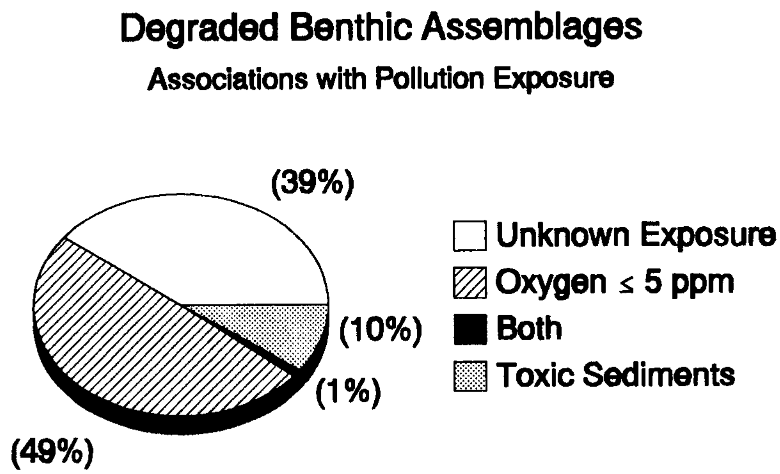


Figure 6-14. Association between degraded benthic assemblages and the dissolved oxygen and sediment toxicity pollutant exposure indicators.

Data from the 1990 Demonstration Project also were used to identify associations between nondegraded biological resources and pollutant exposure. Ninety percent of sites having nondegraded benthos were sites where dissolved oxygen measurements were above 5 ppm, and sediments were not toxic to test biota. Only 5% of the area with nondegraded benthos corresponded to area with low dissolved oxygen conditions, another 5% corresponded to area with toxic sediments.

Societal Values

Although a major objective of EMAP is to describe the status of estuarine resources using indicators of ecological condition, certain characteristics of estuaries that are valued by society may not be reflected by these indicators. The EMAP Estuaries Resource Group intends to provide data and information to address questions commonly asked by the general public, including: Are estuaries aesthetically acceptable, with relatively clear waters and little floating algal scum and trash? Is the water safe for swimming? Are the fish safe to eat?

Data collected during the 1990 Demonstration Project allowed description of some of these attributes. Aesthetics was addressed by estimating the areal extent of trash and turbid waters in estuaries of the Virginian Province. The question of swimmability was not addressed directly, but the abundance of *Clostridium perfringens*, a bacterium indicative of sewage pollution, was measured to approach the question. Fish tissue samples were collected for contaminant analysis but were not processed (see Section 3). Tissue samples will be processed in future years of the program.

Anthropogenic marine debris (trash) was estimated to be present in 9-19% of the estuarine area of the province.

Observations concerning marine litter and debris are important because debris has multiple deleterious effects on animals (entanglement and ingestion), impacts fisheries (decreased market potential for fish and damaged vessels and gear), can economically affect tourist areas (loss of tourists, beach clean-up costs), and contributes to public perception of the general environmental condition of estuaries (Ross et al. 1991). It is estimated that trash was present over 14% (± 5%) of the estuarine area in the Virginian Province (Fig. 6-15). Paper and plastic wastes were found most frequently, followed by cans and glassware. No trash that could be specifically identified as medical or hospital waste was found.

Marine debris was most prevalent in tidal rivers and small estuaries.

Trash was most prevalent in tidal rivers and small estuarine systems. Trash was found in 32% (± 17%) of the area in tidal rivers and 23% (± 16%) of the area in small estuarine systems (Fig. 6-15). In contrast, trash was found in only 10% (± 6%) of the area in large estuaries. Presumably, proximity to the shore and urban areas influences the distribution of trash in estuaries.

Clear waters are valued by society and contribute to the maintenance of healthy, productive biological communities. Water clarity in estuaries is influenced by biological processes (phytoplankton blooms, for example), as well as by inputs of sediment and detritus from streams, rivers, and nonpoint source runoff. Although the geomorphology of some portions of estuaries causes natural turbidity, data collected during the Demonstration Project allow establishment of a baseline against which future changes in water clarity can be assessed. Less than 1% (± 1%) of the estuarine area in the province had waters with visibility less than 0.3 m; however, 13% (± 4%) of the province had water with visibility less than 1 m, the depth of one's feet when wading in waist-deep water.

High concentrations of* Clostridium*, a bacterial tracer of sewage pollution, were estimated to be in 5-13% of the province sediments.

Clostridium perfringens is an obligate-anaerobic bacterium that is present in the feces of warm-blooded animals. Its spores survive longer in the sediments than other indicators of

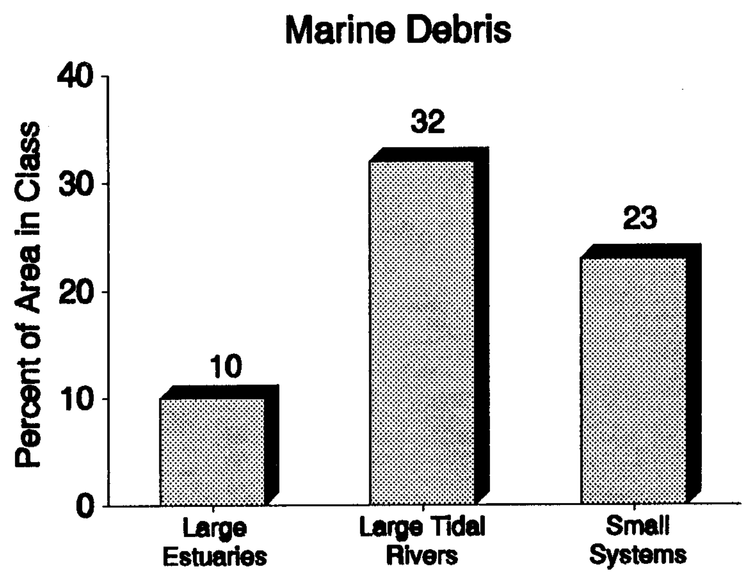


Figure 6-15. Percent of area having anthropogenic marine debris (trash) in the three classes of estuarine resources.

fecal matter (such as coliform bacteria) and, thus, provide a conservative tracer of sewage pollution (Bisson and Cabelli 1980; Duncanson et al. 1986). Spores of this bacterium were above background levels in 9% (± 4%) of the province. Similar to the pattern observed for contaminants, *Clostridium* was most prevalent in small estuarine systems and tidal rivers. Twenty-one percent of the area in small estuaries (± 13%) had high levels of *Clostridium* compared to 4% (± 2%) in large estuaries (Fig. 6-16).

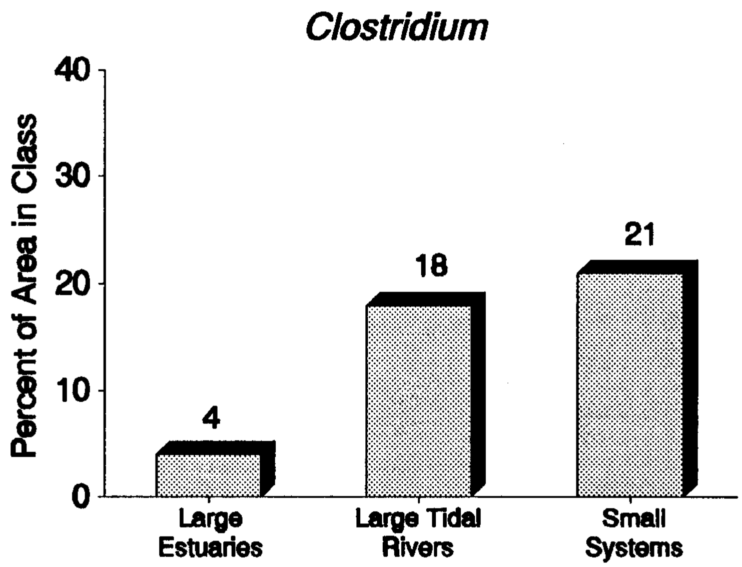


Figure 6-16. Percent of area in the three classes of estuarine resources where high sediment concentrations of *Clostridium perfringens* spores were found.

Integration of Estuarine Conditions

As EMAP reaches full implementation, a single index probably will be developed to summarize the overall condition of estuaries in the Virginian Province. That index may incorporate measures of fishability, swimmability, and aesthetics, combined with measures of biotic integrity based on benthic and fish assemblages (Fig. 6-17). Data to evaluate fishability are not yet available, but it is possible to begin constructing an overall index based upon several of the indicators measured in the Demonstration Project. The methods for combining these indicators are presented in Section 2.

About one-third of the estuarine area in the Virginian Province was estimated to exhibit some form of anthropogenic or natural environmental "degradation".

Indicators relating to biotic integrity and societal values were used to estimate overall environmental conditions in the estuaries of the Virginian Province. Thirty-six percent (± 7%) of the estuarine area in the province showed evidence of degraded biological resources or was impaired with respect to its ability to support activities valued by society (Fig. 6-18). Expressed on an areal basis, an estimated 8,436 km^2 of the 23,574 km^2 total area in estuaries of the Virginian Province were potentially degraded.

The locations of degraded biological resources were different from those impaired with respect to societal values. Both sets of conditions were found in an estimated 7% of the estuarine area, whereas degraded biological conditions alone were found in an estimated 17% of the province, and loss of societal value was associated with an estimated 12% (Fig. 6-18). This suggests that the visual symptoms of environmental degradation (trash, water clarity) or factors that might limit human contact with the water do not necessarily indicate biological degradation.

Large tidal rivers exhibited the most environmental "degradation".

An estimated 64% of the area in large tidal rivers showed evidence of either degraded biological resources or impaired societal value. In contrast, only an estimated 30% of small estuarine systems and 35% of large estuaries showed such evidence. Although tidal rivers were proportionately more degraded than large estuaries or small estuarine systems, the largest area of degraded resources occurred in the large estuaries. Approximately 70% of all degraded estuarine resources, representing 5,835 km^2, were found in large estuaries. In contrast, 1,437 km^2 of degraded resources were found in small estuarine systems and 1,164 km^2 in tidal rivers.

Program Direction

Using a probability-based sampling design to develop regional estimates of ecological status is a new approach to monitoring estuarine health. The preliminary evaluation of environmental status presented in this section was the first attempt by EMAP-E to convey the types of information that can be

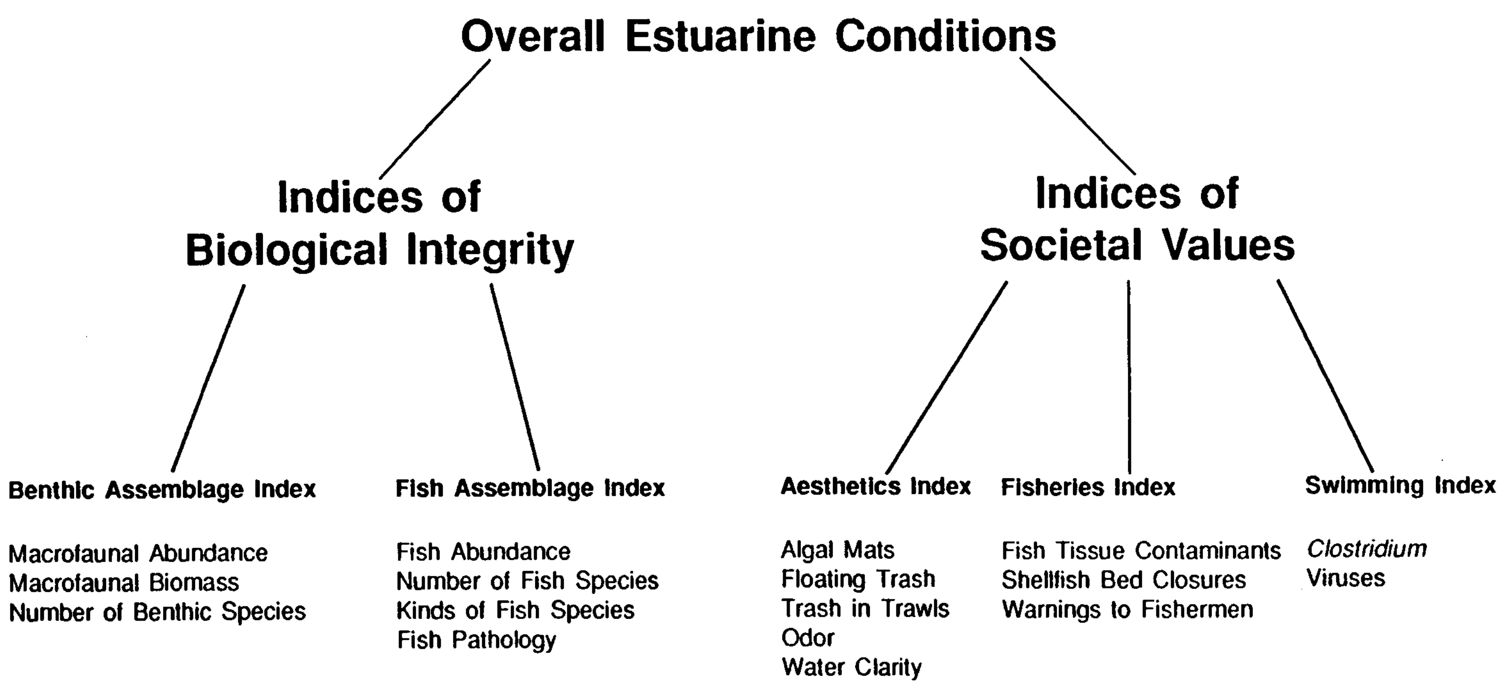


Figure 6-17. Conceptual scheme for combining indicators that might be measured by EMAP into an integrated statement about the environmental condition of estuaries.

Integrated Conditions

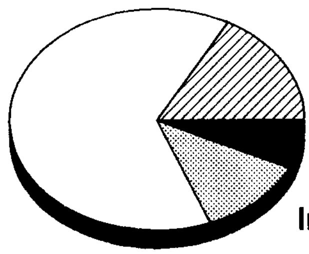


Figure 6-18. Summary of environmental conditions in the estuaries of the Virginian Province.

> **Preliminary Nature of Estimates**
> This evaluation of estuarine status is preliminary and subject to future revision. Validation of several indicators has not been completed and will require monitoring data from at least two years (see Section 4). The estimates are based upon a single year of data. EMAP assessments typically will be conducted using a four-year running average to reduce the effects of natural interannual variation in climate and other influences. Estimates based upon one year of data are intended primarily to provide the reader with examples of the kinds of assessments EMAP is planning to make after four years and to obtain responses regarding the type of assessments desired by clients.

generated using this approach. The topics addressed were defined in partnership with potential users of the program's data. This partnership is crucial to ensure that EMAP monitoring and assessment activities produce information that is relevant to users' needs.

EMAP-E will continue to expand the list of questions towards which future assessment activities are directed. That expansion will be based, in part, on continuing interactions between the Estuaries DResource Group and its data users. The list will also expand as EMAP obtains additional data. During 1991, sampling continued in the Virginian Province, allowing examination of interannual variability in ecological condition. A demonstration project was conducted in the Gulf of Mexico (Louisianian Province) in 1991, which will permit comparison of relative status of estuaries in divergent portions of the country. Similar comparisons and aggregation will be used to complete a national assessment of estuarine condition after EMAP-Estuaries is fully implemented. Finally, EMAP is also intended as a trend detection program; trends questions will be emphasized after EMAP has completed two full sampling cycles.

Throughout all EMAP monitoring and reporting activities, there is continued commitment to building partnerships with potential users and to sharing data and information to meet the needs of decision makers. Comments concerning this report, monitoring data, and the activities of the estuarine component of EMAP should be directed to:

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SECTION 7
SUMMARY AND CONCLUSIONS

The 1990 EMAP-Estuaries Demonstration Project in the Virginian Province collected data and information to address six objectives (Holland 1990). The degree to which these objectives were met is evaluated in this report. These evaluations will be used to refine the program in future years. Although the results of the Demonstration Project are now of most use to those within the program, results are also relevant to users of its products. The ultimate users of the products of EMAP provided comments during the design, implementation, and assessment phases of the 1990 Demonstration Project and are important partners in the future progress of the program. The utility of the results of the Demonstration Project to the users of EMAP data and information is summarized below for each of the six objectives.

Objective 1: Demonstrate the value of regional monitoring using an unbiased sampling approach as a basis for assessing the condition of estuarine resources.

A preliminary evaluation of the environmental condition of estuaries in the Virginian Province was presented in Section 6. The full value of this type of assessment will not be realized until users of EMAP products become familiar with and begin to use the results presented in the report; nonetheless, the utility of the results is already becoming evident. The assessment provided information about estuaries in the mid-Atlantic region that, prior to EMAP, was difficult to obtain in all but the most well-studied of estuaries. This information is unique in its ability to define regional estimates of the areal extent of specific physical habitats, biological conditions, and pollutant exposure with known confidence. The assessment results identified specific habitats and classes of estuaries that should be of greater interest because of the extent or magnitude of environmental impact identified within them. Although based upon only one year of EMAP monitoring data, these results are already of interest to environmental resource managers establishing estuarine research directions and management priorities.

Objective 2: Evaluate the ability of a suite of indicators of environmental quality to discriminate between polluted and unpolluted sites on a regional scale.

The development and testing of estuarine indicators of biological response, pollutant exposure, and habitat was described in Section 4. Based on the information presented in that section, a suite of indicators that will form the core of future EMAP monitoring activities in estuaries was identified. One of the most important accomplishments was the development and application of a framework for calibrating and validating biological indicators. This is particularly noteworthy because development of biological criteria is an Agency priority, and biological indicators that are applicable over a range of latitudes and habitat types have not been identified previously. Several years of additional data will be required to fully validate the biological indicators developed here, and the framework identified for the 1990 Demonstration Project provides a basis for conducting that validation.

Objective 3: Establish standardized methods for monitoring indicators of ecological status and trends in estuaries.

The collection and sample processing methods used during the 1990 Demonstration Project (Strobel 1990; USEPA 1991) were summarized in Section 2. These methods have been tested and evaluated and are now ready to be used directly by other estuarine monitoring programs. Development and use of standardized methods is important if EMAP is to be able to assess ecological conditions over a wide range of systems. Standardized methods will form the basis for comparisons across provinces in the future. EMAP is promoting these methods and the associated QA protocols to EPA Regions, states, and local institutions responsible for monitoring to facilitate achieving comparability and integration of information from multiple monitoring programs and estuaries. Presently, such integration is limited due to the variety of methods being used (NRC 1990).

Objective 4: Obtain data on regional-scale variability in ecological parameters to evaluate and refine the sampling design.

Results from the Demonstration Project were used to evaluate the EMAP estuarine sampling design for its ability to define environmental status and trends (Section 5). This is the beginning of a process that requires multiple years of EMAP monitoring data as well as environmental monitoring data from other, long-term monitoring programs. The process depends upon the involvement of potential clients to help define the power for detecting changes (i.e., trends) required by EPA programs. Results thus far suggest that no major modifications of the EMAP sampling design are required. The present design produces unbiased estimates with an acceptable degree of precision; however, minor modifications may improve the precision of estimates of environmental condition for particular subclasses of estuaries or habitats of interest, and these modifications need to be evaluated further.

Objective 5: Develop analytical procedures for using regional monitoring data to assess the ecological status of estuaries, and apply the procedures to establish baseline conditions in the Virginian Province

The assessment of estuarine condition presented in Section 6 begins to identify baseline conditions for the estuaries of the Virginian Province and provides examples of the kind of environmental baseline that can be defined using monitoring data collected in EMAP. This baseline condition is relative to regional environmental reference conditions defined using Demonstration Project data. Greater precision for this estimate will be possible when a full cycle (four years) of monitoring data becomes available. The baseline will then be used to evaluate changes in environmental condition over time (i.e., trends). The developers of the program realize that the influence of previous anthropogenic stress and disturbance may already be expressed at the regional reference sites selected for the 1990 Demonstration Project in subtle ways that are not evident in the measures of pollution used for the project. For that reason, EMAP is working with regional clients to identify historical data that can be used to evaluate present reference conditions.

Objective 6: Identify and resolve logistical problems associated with conducting a regional monitoring program in estuaries.

The logistical considerations involved in conducting a large-scale monitoring program such as EMAP were evaluated in Section 3. The lessons learned from completing the 1990 Demonstration Project are relevant to EMAP data users and to others involved in designing and implementing large monitoring programs. These lessons include the importance of intensive field-crew training, standardization of sampling equipment and procedures, well-defined field and laboratory quality assurance protocols, state-of-the-art communications and information management systems, and advanced preparation of laboratories for sample processing.

Although each of the objectives of the Demonstration Project was addressed in this report, the evaluations conducted for each objective represent only an initial step. Evaluations are iterative and depend upon the availability of monitoring data from subsequent years, comparisons with historical and existing monitoring programs, and continued participation by users of products of the program.

SECTION 8
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