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Final Report

SEDIMENT QUALITY OF THE NY/NJ HARBOR SYSTEM

An Investigation under the Regional Environmental Monitoring and Assessment Program (R-EMAP)

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FOREWORD

The Environmental Monitoring and Assessment Program (EMAP) is a long-term, interagency environmental monitoring and research program overseen by EPA's Office of Research and Development (ORD). Its goal is to provide the public, scientists and Congress with information that can be used to evaluate the overall condition of the Nation's ecological resources. The program is designed to operate on a broad geographic scale.

EMAP has entered into partnerships with EPA Regional offices, other Federal agencies and States to assess environmental quality at smaller, regional or local scales. These Regional EMAP (REMAP) projects adapt the EMAP approach to assess specific areas more precisely than can be accomplished by existing data or EMAP alone. These projects also provide the opportunity to apply EMAP's statistical design and ecological indicators at localized scales. The REMAP project for the New York-New Jersey Harbor complex is one of ten REMAP efforts in the country (U.S.EPA, 1993a).

The study results presented in this report are based on a REMAP proposal which was jointly developed by U.S.EPA-Region 2 and NY-NJ Harbor Estuary Program (HEP) participants. The study was jointly funded by U.S. EPA/ORD/EMAP, the NY-NJ HEP and U.S.EPA-Region 2.

EXECUTIVE SUMMARY

A number of studies have documented high concentrations of contaminants in sediments of the New York-New Jersey Harbor and Bight Apex. Based on these findings, U.S.EPA-Region 2 and the New York-New Jersey Harbor Estuary Program (NY-NJ HEP) identified the development of a sediment management and monitoring strategy as an integral part of the Comprehensive Conservation and Management Plan (CCMP) for the Harbor and Bight Apex. An unbiased baseline of sediment quality was needed to measure progress of management actions. Existing data were insufficient for developing this baseline.

To provide the baseline data needed to evaluate progress, 168 sites in the Harbor and Bight Apex were sampled in the summers of 1993 and 1994, using a stratified random design. Fourteen sampling sites were allocated in each year to each of six sub-basins (Newark Bay, Lower Harbor, Upper Harbor, Jamaica Bay, western Long Island Sound, and the Bight Apex). Surficial sediment contaminant concentrations, two sediment toxicity tests (*Ampelisca abdita* and Microtox™), and benthic macrofaunal community structure were measured at each site.

Contamination was widespread, with most of the Harbor samples (102 of 112) having at least one chemical exceeding an ERL (Effects Range-Low) concentration, a threshold at which biological effects are possible, and 50% of the Harbor exceeding at least one ERM (Effects Range-Median) concentration, a threshold above which biological effects are more likely. A toxicological response was also observed for 45% of the Harbor. Newark Bay was the most contaminated sub-basin, with 92% of its area exceeding an ERM concentration and 49% of its area showing a toxicological response. In contrast, only 7% of the area in the Bight Apex exceeded ERM concentrations and toxicity was only observed at one Bight Apex location, which was located near an area of historical dredged material disposal.

Contamination was distributed across chemical classes. At least one pesticide, one metal and total PCBs were present at concentrations above ERM for one-third of the Harbor area. The ERL for Total PCB was exceeded at 87% of the Harbor. Mercury and chlordane were the only individual chemicals for which more than 25% of the area in the Harbor exceeded an ERM concentration. Twenty-six individual chemicals had mean concentrations for the entire Harbor that exceeded their ERL concentrations. Mercury, DDT and total PCBs were the only chemicals for which average concentrations exceeded ERM values.

The condition of benthic communities was strongly associated with chemical contamination. At the 66% of the Harbor area where impacted benthic communities were observed, there also was a toxicological response and/or at least one chemical exceeding its ERM concentration. In contrast, only 14% of the Harbor area without a toxicological response and without a chemical exceeding ERM concentration had impacted benthic macroinvertebrates.

The sampling design and methods used in this study were compatible with those of EPA's Environmental Monitoring and Assessment Program-Estuaries (EMAP-E), allowing unbiased comparison of conditions in the NY-NJ Harbor with those in the entire mid-Atlantic region. Based on comparisons with EMAP-E data collected from 1990 through 1993 from the Virginian Province (coastal area from Cape Cod to, and including Chesapeake Bay), the NY-NJ Harbor was found to have higher average sediment concentrations for 58 of the 59 chemicals measured in this study. NY-NJ Harbor sediments are responsible for more than 90% of the spatial extent of exceedances of the total PCBs ERM and 69% of the mercury ERM exceedances in the Virginian Province, even though the Harbor constitutes only 4% of the area in the Province.

An index of benthic quality specific to the Harbor was developed as a tool to evaluate the health of benthic macroinvertebrates. This Benthic Index of Biotic Integrity (B-IBI) was similar to the IBIs developed for freshwater biota. It was developed for four different salinity and grain-size habitat combinations. Five measures were ultimately used in the index; number of taxa, abundance, biomass, abundance of pollution-indicative taxa and abundance of pollution-sensitive taxa. Overall, the B-IBI was able to distinguish correctly 93% of the stressed sites from reference sites.

Sediment quality in the Harbor has undoubtedly improved due to actions taken as a result of recent environmental legislation and improved stewardship. Further major improvements cannot be expected immediately, and will probably be more subtle than improvements to date. The Harbor bottom will continue to integrate loadings of contaminants, organic materials and sediments from the watersheds and airsheds surrounding it. The most obvious “next steps” are to estimate how rapidly sediment quality and associated biological health improve under current watershed protection and pollution prevention activities. Some of these steps are included in the NY-NJ HEP CCMP. Other efforts are being undertaken as a subsequent REMAP investigation.

ACKNOWLEDGMENTS

An assessment of this scope relies heavily on the expertise that was associated with its development, implementation and interpretation. This investigation has especially benefitted from strong management support and partners involved in the planning and execution of the study.

We thank Richard Caspe, Kevin Bricke, Roland Hemmett, Richard Spear, Mario Del Vicario, Randy Braun and Seth Ausubel, U.S.EPA-Region 2 managers, for their effective support and broad coordination of this investigation.

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Contract assistance was provided by Vern Laurie and Tom Murray of ORD. An additional funding mechanism was provided and overseen by the Hudson River Foundation (HRF).

DEDICATION

This report is dedicated to the memory of Dr. Barbara Metzger, Director of the Environmental Services Division, until her untimely death on February 14, 1996. This report strives to represent her dedication to protection of the environment and her conviction that the U.S.EPA use sound scientific practices to achieve that goal.

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1.0 INTRODUCTION

1.1 BACKGROUND

The New York-New Jersey Harbor system is an important economic, recreational, and aesthetic resource supporting many kinds of habitat and species. Among the many important species of fish and shellfish in this estuarine and coastal system are striped bass, white perch, tomcod, and blue crabs in the estuarine portion; and sea bass, bluefish, menhaden, herring, sturgeon, shad, hake, winter flounder, lobster, clams and oysters in the marine portion. Historically, the Harbor supported several large commercial and recreational fisheries. Currently, there remain some isolated, small-scale commercial fisheries (e.g., clams, crabs, menhaden) and a large recreational fishery (MacKenzie, 1992). Since the Estuary is on the Atlantic flyway, it is also an important resting and feeding area for migrating birds. Many birds, both migratory and regional, utilize the Harbor environs for feeding and raising young. Birds commonly found in the region include herons, egrets, ducks, plovers, sandpipers, gulls and geese. Bald eagles and peregrine falcons, both federally-listed endangered species, are less common inhabitants.

The land uses above and surrounding the New York-New Jersey Harbor Estuary make the Harbor particularly susceptible to toxic contamination. For more than a century, it has been the recipient of pollutants generated by the human activities that exist around it. The Harbor is surrounded by a population of more than 20 million people and concentrated refining and manufacturing industries. It is also one of the most heavily utilized shipping ports on the east coast. Sources of toxicants found in the Harbor include municipal and industrial discharges, atmospheric inputs, non-point source runoff, hazardous waste sites, landfills, combined sewer overflows and accidental spills. Additionally, Harbor sediments are contaminant reservoirs which can function as secondary sources. Since the Bight Apex and Long Island Sound receive Harbor outflow, both are affected by Harbor contaminants. One dedesignated (dredged material) and several inactive (acid waste, cellar dirt and sewage sludge) dumpsites also are located in the Bight Apex.

Contaminated sediments pose a substantial threat to Harbor resources and are a management challenge. Dredging and disposal of contaminated sediments are controversial issues. Adverse changes in the biota of the system have been documented, and many of these changes have been linked to toxic contamination (Mayer, 1982; U.S.EPA, 1990a). The consequences of contamination in the NY-NJ Harbor are extensive. The states around the Harbor advise restricted consumption of striped bass, bluefish and blue claw crabs from large portions of the Estuary because the levels of PCB and/or dioxins exceed guidelines for human consumption. Areas that were once productive shellfish beds no longer exist or have reduced populations that are restricted for harvesting (MacKenzie, 1992). Bioaccumulation of contaminants in fish, shellfish and crustacea has been documented (Belton et al., 1985; NYSDEC, 1988; Hauge et al., 1990; Zongwei et al., 1994; NOAA, 1996). Benthic macroinvertebrate communities appear to be impacted by sediment contaminants (Franz and Harris, 1988; Steimle and Caracciolo-Ward, 1989). Other investigations have described elevated levels of contaminants in sediments (NOAA, 1991; Huntley et al., 1993; Bonnevie et al., 1995) and sediment toxicity (Scott et al., 1990; Schimmel et al., 1994; Long et al., 1995b).

Most studies of toxic contamination in the NY-NJ Harbor system have focused on measuring the concentrations of contaminants in the sediment. Characterizing sediment condition is a logical way to describe toxic contamination in an estuarine system because the sediment is both a sink for contaminants that adsorb to fine particles, and a source for toxic contaminants that are rereleased to the water column when sediments are disturbed by natural events (e.g., seasonal turnover, bioturbation, violent storms) or human activity (e.g., dredging, vessel traffic). In addition, the food chains for many estuarine species begin in the sediment; therefore, contaminants in the sediment can be propagated widely throughout an estuarine ecosystem.

Existing studies have been useful in establishing concern about contaminants in the NY/NJ Harbor system. In a review of historical data on toxic contamination, Squibb et al. (1991) identified 12 metals and 43 organic chemicals that are present in the Harbor water, biota or sediments at concentrations that may affect the integrity of the system. NOAA’s National Status and Trends program, which has conducted sampling in the NY-NJ Harbor, identified it as having some of the highest metals concentrations found nationwide. Based on these data and the integral relationship between contaminated sediments and the health of the Estuary, the Harbor Estuary Program, U.S.EPA-Region 2, states and local governments have made addressing the biological effects of contaminated sediments a high priority (U.S.EPA-Region 2, 1996).

While these existing data are sufficient for the purpose they were designed for and to raise concerns about sediment contamination in the Harbor, they are insufficient for developing an effective contaminant strategy for the NY-NJ Harbor complex for several reasons:

- Much of the existing data is limited, outdated, or unreliable, causing Squibb et al. (1991) to recommend characterizing the problem further before acting to correct it.

- Data from historic studies are insufficient for evaluating the areal extent of toxic contamination throughout the NY/NJ Harbor and in each of its sub-basins because most studies of sediment contamination in the system were initiated to resolve site-specific problems rather than to support regional management decisions. Sampling has been limited to specific “hot-spots” around known or assumed contaminant sources. These data cannot be extrapolated to unsampled areas, which would be necessary to reliably characterize the condition of the entire system and specific sub-basins. Although Squibb et al. (1991) were able to identify a large number of contaminants in the Harbor, they were unable to evaluate pervasiveness of the contamination.

- There is little opportunity to assess from existing data the biological effect of contaminants that were measured in the sediment, or whether the effects differ in different portions of the estuary complex. This is because sediment contaminant data were collected independently of biological data.

These shortcomings are major impediments to developing a management strategy. Information about the distribution of the contamination problems among sub-basins is necessary for determining whether the management emphasis should be focused on a regional, watershed, or a local site-specific scale. Developing management strategies on the basis of “hot spot” information alone may result in misdirection of management efforts, particularly if the problem is more widespread than limited “hot spot” data would suggest. Also, information on biological effects of contaminants is critical to identifying the scope of the problem and determining the resources appropriate to remediate it. The ecological significance of contaminant levels documented from purely chemical surveys is unknown in the absence of information on direct toxicity of those contaminants and/or data documenting the relative status of biological communities, such as the benthos, exposed to these materials. Areas where contaminant levels are high but biological availability and toxicity are low may be addressed best with management strategies different from those appropriate for areas where significant impacts to biota are evident.

1.2 OBJECTIVES

This project was designed to support resource management decisions related to pollution control and remediation throughout the NY-NJ Harbor and Bight Apex and to assist the Harbor Estuary Program (HEP) in developing a contaminant monitoring strategy to be followed as part of the Comprehensive Conservation and Management Plan (CCMP) for the NY-NJ Harbor system. This investigation was designed around several objectives:

Objective 1. Estimate with known confidence the percent of area in each of six major sub-basins of the NY-NJ Harbor system in which the benthic environment is "degraded", "not degraded", or "not evidently degraded" with respect to benthic macroinvertebrate assemblages, sediment toxicity, and concentrations of sediment contaminants, and,

Objective 2. Identify statistical associations among particular chemical contaminants and degraded benthos or toxic sediments.

A third objective was identified because the HEP and the Region recognized that a tool to represent benthic quality was needed:

Objective 3. Develop and validate a managerially useful index of benthic quality for the NY/NJ Harbor system, based on the condition of benthic macroinvertebrate assemblages.

1.3 RELATIONSHIP TO THE CCMP

The New York-New Jersey Harbor Estuary Program (HEP) has prepared a Comprehensive Conservation and Management Plan (New York-New Jersey Harbor Estuary Program, 1996). The CCMP included a section on management of toxic contamination. The goals of the HEP plan for toxics are:

- To establish and maintain a healthy and productive Harbor/Bight ecosystem, with no adverse ecological effects due to toxics.
- To ensure that fish, crustacea and shellfish caught in the Harbor/Bight are safe for unrestricted human consumption.
- To ensure that dredged sediments in the Harbor are safe for unrestricted ocean disposal.

In order to take steps toward attainment of these goals, the HEP plan includes actions to reduce continuing inputs of toxic chemicals to the Harbor and Bight from sources such as municipal discharges, industrial discharges, combined sewer overflows, storm water discharges, and non-point sources.

The data from this investigation will be used to support the HEP goals. For example, the benthic index and toxicity tests will be interpreted in relation to a "no adverse effects" level. Also, where available, numeric criteria and tests used in regulatory decision-making will be used to interpret the data. This investigation's surficial sediment sampling represents recently deposited sediments and contaminants. Therefore, interpretation of the *A. abdita* toxicity test results, sediment chemistry and benthic macroinvertebrate structure information will help managers assess the potential future distribution of dredged material unsuitable for ocean disposal. This information, combined with an evaluation of the causes of toxicity, also will help focus strategies to control continuing sources of contamination.

1.4 ORGANIZATION OF THE REPORT

The purpose of this report is to present summarized data and interpretation to address the three objectives that were defined at the start of the project.

The report has nine chapters. Chapter 2 defines the indicators that were used and how they were measured. Chapters 3, 4, 5 and 6 report results from each of the indicator classes, both in terms of mean condition and percent of area above or below specified threshold values, and relates these to previous studies in the Harbor. Chapter 7 analyzes the associations between the various indicators and Chapter 8 provides discussion of the results in terms of management implications. Chapter 9 contains all references cited in the report. Several appendices are included: A - sampling station locations and maps, B - analytical detection limits, C - benthic index development, D - aluminum-normalization procedure, E - tables of means and % of area exceedances of ERMs for all chemicals measured in the study, F - dioxin bioaccumulation calculations, G - mean abundances of all benthic species, H - *Clostridium perfringens* results, and I - benthic index values for individual stations. Appendix J contains explanatory information for the data disk that is included inside the back cover. The disk, in Excel format, contains unmanipulated data from this investigation.

2.0 METHODS

2.1 DESIGNATION OF STUDY AREA

Based on hydrogeography and similar source characteristics, the study area was divided into six sub-basins (Figure 2-1): Upper Harbor, Newark Bay, Lower Harbor (includes Raritan and Sandy Hook Bays), Jamaica Bay, western Long Island Sound and the New York Bight Apex. The New York-New Jersey Harbor, for purposes of this investigation, includes the lower portions of the Hudson, Passaic, Harlem, Hackensack and Raritan Rivers, upstream to a near-bottom salinity of 15 ppt, the East River to Long Island Sound, and Lower Harbor to the Atlantic Ocean. The New York Bight Apex is defined as the area of ocean bounded on the northwest by the transect from Sandy Hook, NJ to Rockaway Point, NY, the east by 73° 30' W longitude, and the south by 40 10' N latitude. The eastern boundary of the western Long Island Sound sub-basin is 73° 24' W longitude (from Eaton's Neck Point, NY to Norwalk, CT). The area of each sub-basin was determined using Geographic Information System (GIS) ARCInfo software (Table 2-1).

Table 2-1
Sub-basin Areas and Percent of Study Areas

Sub-basin	**Area (km^2)**	**% of Study Area**
Lower Harbor	318	11.1
Upper Harbor	104	3.7
Jamaica Bay	47	1.7
Newark Bay	32	1.1
W. Long Island Sound	476	16.6
Bight Apex	1883	65.8
Harbor Total*	501	17.6
Study Area Total	2861	100.0

*The Harbor Total includes Lower and Upper Harbors, Jamaica Bay and Newark Bay.

2.2 STUDY DESIGN

There are two different strategies for sampling to estimate characteristics of the field. Often sampling sites are selected by their anticipated ability to reflect regional characteristics. Samples are presumed *a priori* to be "representative" of their surrounding areas. This is termed judgmental or purposive sampling. The alternative strategy, termed probabilistic, ensures that

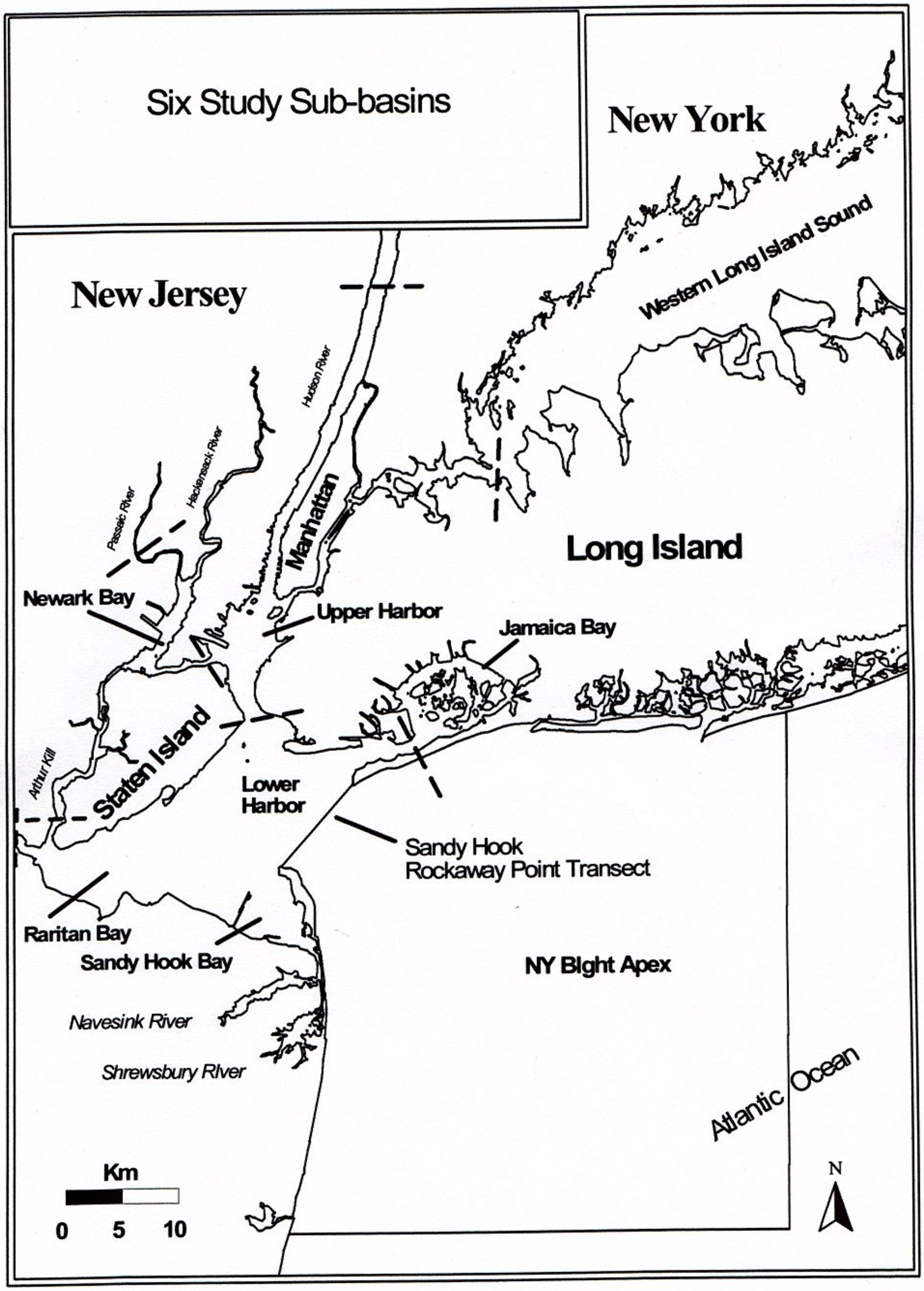


Figure 2-1. Map of the six study sub-basins: Upper Harbor, Newark Bay, Lower Harbor (includes Raritan and Sandy Hook Bays), Jamaica Bay, western Long Island Sound and the New York Bight Apex.

every element in the population sampled has some chance of actually being sampled. For instance, in the present investigation, every potential benthic grab sample in the study area could have been sampled (i.e., potential grabs not sampled were excluded only by chance, not by judgement). While this probability sampling is not the most appropriate in all situations, it does have important advantages over purposive sampling. Probability samples can provide unbiased estimates of population characteristics with specified confidence limits. These confidence limits become smaller as sample size is increased.

Sampling stations for the present investigation were selected probabilistically using a stratified random approach. The strata corresponded to each of six sub-basins where independent estimates of condition were needed. Fourteen stations were assigned to each sub-basin in each of the two years of sampling, for a total of 28 stations in each sub-basin (Appendix A). Each year, sites were selected by randomly placing a grid structure over the study area, selecting 14 grid cells at random from each stratum, and selecting a random location from within the selected cells. Cells were of equal area within strata, except for the Newark Bay stratum, where grid cell size was altered to ensure sampling in the Arthur Kill, Passaic River, and the Hackensack River.

Sampling was conducted between late July and late September of 1993 and 1994. A summer index period was chosen for several reasons. This time period has been identified as most appropriate for this area (Holland, 1990). Pollution stress is expected to be at its highest because dissolved oxygen values are low and contaminant exposure is at its maximum due to high temperatures and low dilution flows. Benthic organisms are usually more abundant, which increases the success of sampling. While some indicators vary between July and September, most of the measures that this investigation focused on, such as benthos, toxicity and chemistry, are stable during that time period. The U.S.EPA Environmental Monitoring and Assessment Program (EMAP) evaluated benthic response and found it did not vary unacceptably between late July and September (Weisberg et al., 1993). A summer index period also ensures compatibility with EMAP, which is useful since it allows comparison to those results and referencing to EMAP benthic macroinvertebrate data for the development of a benthic index.

2.3 SAMPLING PROCEDURES

The U.S.EPA vessels, R/V CLEAN WATERS and OSV PETER W. ANDERSON, were used for sample collection. Sampling stations were located using LORAN-C and a Global Positioning System (GPS) or Differential-GPS (D-GPS). Depth of the water column was determined using sonar. Field procedures followed Reifsteck et al. (1993).

2.3.1 Water Column

A SeaBird model SBE 25 “Sealogger” CTD unit was used to obtain a vertical profile of depth, dissolved oxygen, pH, temperature, and salinity at each station. Measurements were made from within a meter of the water surface to approximately a meter above the sediment/water interface.

Water clarity was measured using a 20-cm Secchi disk. Dissolved oxygen, temperature and salinity at the surface were measured using a Winkler titration, NBS thermometer and a refractometer, respectively, and compared with the CTD results.

2.3.2 Sediment

A 0.04-m^2 or 0.1-m^2, stainless steel, Young-modified van Veen grab was used to collect surficial sediment for chemical analysis and toxicity testing. Multiple grabs were required to collect enough volume for analysis. Overlying water was carefully drained by allowing suspended floc to settle for approximately one minute and then carefully suctioning off the overlying water with a clean section of Tygon® tubing. For Acid Volatile Sulfide (AVS)/Simultaneously Extracted Metals (SEM) analysis, aliquots of the top 2 cm were taken from the undisturbed surface of multiple individual grabs using a 60-cc syringe which had the narrow end removed to create a mini-corer. AVS samples were not homogenized. When a sample container was filled to the top, it was sealed with Teflon® tape and immediately frozen. The remaining top 2 cm of sediment from each grab were removed using clean stainless steel spoons. A composite of all grabs was homogenized in a clean glass mixing bowl for 10 minutes. Subsamples were removed for metals, organics, grain size, TOC and toxicity tests, and transferred to clean sample containers that were stored on ice. The van Veen grab was rinsed with ambient seawater between grabs at a station and thoroughly cleaned with detergent and water between stations.

2.3.3 Benthos

Three benthic macroinvertebrate grabs per sampling station were collected using the 0.04-m^2 Young-modified van Veen grab. Benthic grabs were alternated with sediment chemistry/toxicity grabs. Benthic samples were gently washed through a 0.5 mm mesh sieve. The material that remained was preserved in a 10% buffered formaldehyde-rose bengal solution.

2.4 PHYSICAL/CHEMICAL/BACTERIOLOGICAL LABORATORY METHODS

Methods used for chemical analyses are summarized in Table 2-2. Individual chemical parameters are listed in Table 2-3 and detection limits are in Appendix B. PAHs, TOC, grain size, and total recoverable metals were analyzed at the U.S.EPA-Region 2 Laboratory in Edison, NJ. PCB, pesticides and butyltins were analyzed, under contract to the Hudson River Foundation (HRF), by the Geochemical and Environmental Research Group (GERG) of Texas A&M University, College Station, TX. Acid volatile sulfide (AVS), simultaneously extracted metals (SEM) and total metals were analyzed by the Trace Element Research Laboratory (TERL) of Texas A&M. Selected samples for dioxins and furans were analyzed by Battelle Labs, Columbus, OH. The GERG and TERL laboratories both participated in the NOAA Status and Trends Interlaboratory Comparison exercise.

Table 2-2
Summary of Physical/Chemical Analytical Methods

Parameter	Method	Reference
PAHs	Methylene chloride extraction; determination by GC/MS	TSB SOP C-48 (U.S.EPA-Region 2, 1994a)
PCB/Pesticides	Methylene chloride extraction; determination by HRGC/ECD	GERG SOPs-ST02, ST04
Major and Trace Elements	Total metals: HNO_3 and HF acid digestion: Hg-CVAAS;Cu, Ni, Pb, Cr, Sb, Sn, As, Se, Ag, Cd-GFAAS; Al, Fe, Mn, Si, Zn-FAAS	GERG SOPs-ST08, ST09, ST10, ST11
Major and Trace Elements	Total recoverable metals: HNO_3/H_2O_2 or microwave digestion: Hg-CVAF;Cu, Ni, Cr, Ag, Al, Fe, Mn, Sb (1993); Zn-ICP; Pb, Cd, As (1993), Se-GFAAS; As (1994), Sb (1994)-HYDAAS	TSB SOPs C-5, C-8, C-72, C-73, C-74 (U.S.EPA-Region 2, 1994b-f)
Hexavalent Chromium	Chelation with APDC, extraction with MIBK; determination by FAAS	MCAWW 218.4 (U.S.EPA, 1983)
Dioxins and Furans	Extraction with toluene; determination by HRGC/HRMS; second column confirmation for 2,3,7,8-TCDD	Method 1613 - Rev. A (U.S.EPA, 1990b)
AVS/SEM	AVS-selective generation of H_2S, gravimetric, colorimetric or titrametric determination; SEM-filtration of AVS digestate, determination by FAAS, ICPAES or CVAAS	GERG SOPs-9130, ST11, ST09, ST10
Butyltins	Tropolone extraction; determination by HRGC/FPD or HRGC/MS	GERG SOP-9013
TOC	Acidification with H_3PO_4; determination using a CO_2 analyzer	MCAWW 415.1 (U.S.EPA, 1983)
Grain size	Sieving and pipette analysis	U.S.EPA, 1993b

All analyses employed appropriate quality assurance samples. Quality assurance goals were developed and followed for each analysis (Adams and Hunt, 1993). Except in isolated instances, all quality assurance goals were met or exceeded. Data were entered into two separate databases and then compared electronically to ensure accuracy in data entry.

Table 2-3

Analytical Measurements for Sediment Samples

Polyaromatic Hydrocarbons (PAHs)		
Acenaphthene	Biphenyl	1-Methylnaphthalene
Acenaphthylene	Chrysene	1-Methylphenanthrene
Anthracene	Dibenz(a,b)anthracene	Naphthalene
Benz(a)anthracene	2,6-Dimethylnaphthalene	Perylene
Benzo(b,k)fluoranthene	Fluoranthene	Phenanthrene
Benzo(g,h,i)perylene	Fluorene	Pyrene
Benzo(a)pyrene	Ideno(1,2,3-c,d)pyrene	2,3,5-Trimethylnaphthalene
Benzo(e)pyrene	2-Methylnaphthalene	

DDT and its Metabolites		**Chlorinated Pesticides other than DDT**	
o,p’-DDD	p,p’-DDE	Aldrin	Heptachlor
p,p’-DDD	o,p’-DDT	Alpha-Chlordane	Heptachlor epoxide
o,p’-DDE	p,p’-DDT	Trans-Nonachlor	Hexachlorobenzene
		Dieldrin	Lindane ((-BHC)
		Endrin	Mirex

Major Elements	**Trace Elements**		
Aluminum	Antimony	Copper	Selenium
Iron	Arsenic	Lead	Silver
Manganese	Cadmium	Mercury	Tin
Silicon	Chromium	Nickel	Zinc

PCB Congeners (20)

No.	Congener Name	No.	Congener Name
8	2,4'-dichlorobiphenyl	118	2,3',4,4',5-pentachlorobiphenyl
18	2,2',5-trichlorobiphenyl	126	3,3',4,4',5-pentachlorobiphenyl
28	2,4,4'-trichlorobiphenyl	128	2,2',3,3',4,4'-hexachlorobiphenyl
44	2,2',3,5-tetrachlorobiphenyl	138	2,2',3,4,4',5'-hexachlorobiphenyl
52	2,2',5,5'-tetrachlorobiphenyl	153	2,2',3,4,4',5'-hexachlorobiphenyl
66	2,3',4,4'-tetrachlorobiphenyl	170	2,2',4,4',5,5'-hexachlorobiphenyl
101	2,2',4,5,5'-pentachlorobiphenyl	180	2,2',3,3',4,4',5-heptachlorobiphenyl
105	2,3,3',4,4'-pentachlorobiphenyl	187	2,2',3,4,4',5,5'-heptachlorobiphenyl
110/77	2,3,3',4',6-pentachlorobiphenyl/	195	2,2',3,3',4,4',5,6-octachlorobiphenyl
	3,3',4,4'-trichlorotetrabiphenyl	206	2,2',3,3',4,4',5,5',6-nonachlorobiphenyl
		209	2,2',3,3',4,4',5,5',6,6'-decachlorobiphenyl

Dioxin and Furan Congeners*			
2,3,7,8-TCDD	1,2,3,4,6,7,8-HpCDD	2,3,7,8-TCDF	2,3,4,6,7,8-HxCDF
1,2,3,7,8-PeCDD	OCDD	1,2,3,7,8-PeCDF	1,2,3,4,6,7,8-HpCDF
1,2,3,4,7,8-HxCDD		2,3,4,7,8-PeCDF	1,2,3,4,7,8,9-HpCDF
1,2,3,6,7,8-HxCDD		1,2,3,4,7,8-HxCDF	OCDF
1,2,3,7,8,9-HxCDD		1,2,3,7,8,9-HxCDF	

Other Measurements				
AVS/SEM	Grain Size	*Clostridium*	TOC	Butyltins

*only analyzed on Upper Harbor, Jamaica Bay, & Lower Harbor samples

2.4.1 Major and Trace Elements

Sediment samples were prepared for bulk metals analyses using two procedures: 1) digestion with nitric and hydrofluoric acids (total metals) and 2) digestion with nitric acid (total recoverable metals). Subsequent data analyses are based on total metals results. Mercury was analyzed by cold vapor atomic absorption (CVAA). Copper, nickel, lead, chromium, hexavalent chromium, antimony, tin, arsenic, selenium, silver and cadmium were analyzed by graphite furnace atomic absorption spectroscopy (GFAAS). Other metals (aluminum, iron, manganese, silicon and zinc) were determined by flame atomic absorption spectroscopy (FAAS). Metal concentrations are reported on a dry weight basis. The sediment SRM used was National Research Council of Canada (NRCC) MESS2.

2.4.2 Organic Compounds

For analysis of pesticides and PCBs, aliquots of sediment were dried using sodium sulfate and soxhlet extracted using methylene chloride for six hours. The extract was concentrated using a Kuderna-Danish technique and the methylene chloride replaced with hexane. Extracts were cleaned up with a silica gel/alumina column eluting with a 50:50 mixture of pentane and methylene chloride. This fraction, which was primarily the aromatic and chlorinated hydrocarbons, was again concentrated using a Kuderna-Danish technique and the mixed solvent was replaced with hexane. The chlorinated pesticides and PCBs were quantified using high resolution capillary gas chromatography with electron capture detection (GC/ECD). The GC column used was a 30 m, 0.25 mm I.D. fused silica column with a DB-5 bonded phase. The data are reported in ng/g dry weight. The sediment SRM used with these samples was National Institute of Technology (NIST) 1941a.

Twenty-two polycyclic aromatic hydrocarbons (PAHs) were measured (U.S.EPA-Region 2, 1994a). A 10-g aliquot of sediment was dried with anhydrous sodium sulfate and soxhlet extracted with methylene chloride for 16 hours. The extract was dried by using a sodium sulfate drying column and concentrated using a Kuderna-Danish apparatus to 1 ml. A GC/MS with a 30 m, 0.25 mm I.D. DB-5 fused silica capillary column was used for analysis. A mass range of 33 to 450 amu was used. Results are reported as ug/kg, dry weight. The SRMs used were NIST 1941a and 2260.

Butyltin analysis included mono-, di-, tri- and tetrabutyltin. Samples were freeze-dried and extracted using 0.2% tropolone in methylene chloride on a roller table for three hours. The extract was concentrated using a Kuderna-Danish apparatus and treated with Grignard reagent to hexylate the butyltins. Extracts were neutralized and cleaned up with a silica gel/alumina column. The fraction was again concentrated using Kuderna-Danish techniques and the mixed solvent was replaced with hexane. Final volume of the extract was 1.0 ml. Butyltin quantification was done on a high resolution capillary gas chromatograph with either flame photometric detection (HRGC/FPD), equipped with a tin selective 610 nm filter or a mass spectrometer (HRGC/MS). The GC column used was a 30 m, 0.32 mm I.D. fused silica capillary column with DB-5 or DB-5MS bonded phase. A mass of 121 m/z was monitored for quantification with a secondary ion of 191 m/z monitored for confirmation. The standard reference material, NRCC PACS-1, also was analyzed.

The analytical method for AVS analysis employed selective generation of hydrogen sulfide and gravimetric, colorimetric or titrametric determination (depending on the expected concentration of sulfide). Following AVS analysis and digestate filtration, SEM analysis was performed for cadmium, copper, lead, mercury, nickel and zinc using FAAS, ICPAES or CVAAS. Results are reported as umol/g (dry wt.).

Analysis of selected sediments for seventeen dioxin and furan congeners was done according to Method 1613-Revision A (U.S.EPA, 1990b). Frozen sediment samples were thawed and centrifuged to remove excess water. Approximately 10 g of sediment was used for determination of percent solids. Another 10 g was combined with quartz sand for extraction. All samples were spiked with isotopically labeled analogs of 15 of the 17 2,3,7,8-substituted PCDDs/PCDFs prior to extraction. The samples then were extracted for 20 hours using toluene in a Soxhlet/Dean Stark apparatus. Extracts were spiked with $^{37}CL_4$-2,3,7,8-TCDD cleanup standard, partitioned against base and acid solutions, and processed through acid/base silica, basic alumina, and carbon AX-21/Celite cleanup columns. The carbon AX-21 Celite columns were back eluted with 30 mL toluene rather than the method-specified 20 mL as the laboratory has found that the extra toluene has increased the recovery of OCDD/F in the past. Extracts were spiked with 1,2,3,4-TCDD-$^{13}C_{12}$/1,2,3,7,8,9-HxCDD-$^{13}C_{12}$ recovery standard and concentrated to a final volume of 20 uL. These extracts were analyzed by high resolution gas chromatography/high resolution mass spectrometry (HRGC/HRMS) in the selected ion monitoring mode on a DB-5 capillary column at an instrument resolution of approximately 10,000 (10% valley). Most samples were diluted to reduce chromatographic interference problems. Because 2,3,7,8-TCDF is not completely resolved from other tetrachlorinated isomers on the DB-5 column, second column confirmation of 2,3,7,8-TCDF levels above 1 ng/kg dry wt. was performed on a DB-Dioxin column. The standard reference material, EDF-2513 (Cambridge Isotope Laboratories), was processed with each batch of samples.

2.4.3 Sediment Physical Parameters

Grain size analysis was performed according to U.S.EPA (1993b), except samples were not digested with hydrogen peroxide. Samples were treated with sodium hexametaphosphate as a dispersant. Sand was defined as the fraction that was retained on a 63-u sieve. Percent silt and percent clay were determined using pipette analysis of the filtrate. Percent moisture was obtained by accurately weighing 10 g of sediment, drying overnight at 105°C and reweighing. The total organic carbon (TOC) method was based on the U.S.EPA method MCAWW 415.1 (U.S.EPA, 1983), modified for sediment using a boat sampling module.

2.4.4 Bacteriological Analysis

Concentrations of *Clostridium perfringens* spores have been used as an indicator of sewage contamination (Hill et al., 1993; O'Reilly et al., 1995). *C. perfringens* is a obligate anaerobe bacterium found in fecal material. It can survive extreme environmental conditions. This study evaluated the concentrations of the spores in Harbor sediments. The membrane filter method of Emerson and Cabelli (1982) was used. Mean concentrations of *C. perfringens* spores are expressed as confirmed counts per gram (wet weight) of sediment

2.5 TOXICITY METHODS

2.5.1 Amphipod Sediment Toxicity Tests

Batches of a tube-dwelling amphipod, *Ampelisca abdita*, were supplied by East Coast Amphipod of Kingston, Rhode Island. The amphipods and control sediment were collected from the Narrow River, Rhode Island and the U.S. Army Corps of Engineers Long Island Sound (LIS) reference station. Control sediment was press-sieved through a 0.5 mm mesh stainless steel sieve to remove resident amphipods and debris. Test sediment was press-sieved through a 2.0 mm stainless steel sieve to remove large debris and predaceous organisms. If amphipods were present, the test sediments were press-sieved through a 1.0 mm stainless steel sieve. Organisms were acclimated at 20°C and 30 ppt salinity prior to testing. Temperature and salinity did not change by more than 3°C and 3 ppt, respectively, during any 24 consecutive hours of acclimation. Amphipods were fed the marine alga, *Phaeodactylum tricornutum,* during acclimation. Ten-day acute, static, non-renewal sediment toxicity tests were conducted according to ASTM (1991, 1992) and U.S.EPA (1993b) test protocols. For each toxicity test, 200 ml of composited, press-sieved sample were placed in 1 L glass test chambers and covered with 600 ml of seawater. Five replicate test chambers were used for each sample. Each replicate contained 20 organisms.

Post-test enumeration of amphipods was performed without knowledge of sample identity to prevent bias. If less than 20 amphipods were found, the test sediment was stored in the dark for up to 48 hours to encourage emergence of any remaining amphipods. Final organism counts were confirmed by a second scientist. Minimum control survival for satisfying test performance criteria was 90%. Sodium dodecyl sulfate (SDS) was used as a reference toxicant to evaluate the sensitivity of each batch of amphipods. Reference toxicant results were all within the acceptable range for this species. *A. abdita* assays were conducted by the U.S.EPA-Region 2 Bioassay Laboratory in Edison, NJ and SAIC, Narragansett, RI. These two laboratories participated in an interlaboratory comparison which showed that the laboratories produced comparable results.

2.5.2 Microtox™ Assays

The Microtox™ assay for marine sediments is considered to be a rapid screening alternative to standard acute toxicity testing with fish or invertebrates (Giesy and Hoke, 1990) and has been recommended as a first stage assay in a tiered testing arrangement (Sloof, 1985). In this investigation it was used to supplement the *Ampelisca abdita* acute amphipod assay. The Microtox™ assay is based on the inhibition of light emission by the luminescent bacterium *Photobacterium phosphoreum* in the presence of toxicants. Freeze-dried luminescent bacteria are reconstituted in control and test solutions and incubated, then luminescence is measured on serial dilutions after 5 to 15 minute exposures. The percent inhibition of light transmission, converted to an EC_{50} value, is the measure of toxicity. Microtox™ assays were conducted by ToxScan, Watsonville, CA.

The solvent extraction method adapted from Long and Markel (1992) was employed. Before extraction, excess water from the top of the samples was decanted and discarded. The sediment was homogenized and a 3.3 g wet weight sample was weighed into a 50-ml Pyrex centrifuge tube with a Teflon-lined screw cap. Samples were dried by and extracted with dichloromethane (DCM). Solvent exchange and concentration were performed using a Kuderna-Danish flask attached to a Snyder column. Extracts were tested in duplicate following Micobics Corp. recommended procedures (1992). The sediment extracts were diluted 1:100 with Microtox™ diluent. Serial dilutions of 100, 50, 25, 12.5, 6.25, 3.13 and 0 percent of this stock solution were made using Microtox™ diluent. The 0% dilution is a reagent blank used to measure spontaneous decay in bacterial luminescence of any treatment. Percent decrease in luminescence relative to the reagent blank was calculated and these data were used to obtain the 50% inhibition concentration (i.e., EC_{50}). Results were converted to mg dry wt./ml.

Control sediment from the U.S. Army Corps of Engineers Long Island Sound (LIS) reference station was tested along with the Harbor samples. Ethanol reagent blanks with no sediment and extraction blanks were prepared and tested. Reference toxicant testing using phenol was conducted with each set of sediment assays and results were acceptable according to the test protocols.

2.6 BENTHIC MACROINVERTEBRATE ASSEMBLAGES

Three replicate grabs for benthic macroinvertebrate community structure were obtained at each station. The grabs were processed by being washed through a 0.5 mm screen on-board the sampling vessel. Invertebrates from two of the replicates were sorted and identified, the third replicate was archived. Procedures for sorting, identifying, and measuring the biomass of benthic macroinvertebrates followed EMAP-E procedures (Klemm et al., 1993; Frithsen et al., 1994). The macrobenthos were identified to the lowest practical taxonomic category. Rare or previously undocumented specimens from the Harbor were put aside in a reference collection. Ten percent of all samples were reprocessed and subjected to a second QA evaluation.

Taxonomic identifications were verified using reference organisms obtained from EMAP's reference collection. Sample processing was conducted by Versar, Inc. (Columbia, MD) and Cove Corporation (Lusby, MD). Species identifications and enumerations were done by Cove Corporation and biomass measurements were done by Versar, Inc.

Organisms were grouped by taxa for biomass determination. To standardize the biomass measurements, all samples were preserved in a 10% solution of buffered formaldehyde for at least two months before the biomass measurement. Hard-bodied organisms (bivalves <2cm and gastropods) were acidified in 10% HCL until all visible traces of shell material were removed.

Bivalves larger than 2 cm were shucked before determination of biomass. Biomass was determined as dry wt. after drying for at least 48 hours at 60°C.

2.7 DATA ANALYSIS

2.7.1 Chemical Data

For several classes of compounds, data analyses were performed on summed results. Total PCBs were the sum of the concentrations of the 20 congeners in Table 2-3 multiplied by 2.0 (NOAA, 1989). Total PAHs were the sum of the concentrations of the 23 individual PAHs. Total chlordane was the sum of the concentrations of heptachlor, heptachlor-epoxide, oxychlordane, gamma-chlordane, alpha-chlordane, trans-nonachlor and cis-nonachlor. Non-detects were not included in the calculation of total concentrations.

Data analyses for metals were based on total metals results.

2.7.2 Toxicity Data

Amphipod survival data were not transformed, since an examination of a large historical data set from SAIC has shown that *A. abdita* percentage survival data meet the requirement of normality (Thursby et al., 1997).

For Microtox™ analyses, the concentration and response data were log-transformed before using analysis of covariance (ANCOVA) to conduct a pair-wise comparison to determine significant differences between samples from each station and control sediment.

2.7.3 Benthic Macroinvertebrate Data

Nine individual measures (Table 2-4) and one composite index (benthic index of biotic integrity or B-IBI) were used to evaluate the condition of benthic assemblages in the study area. Diversity was evaluated by using species richness (number of species) and the Shannon-Wiener diversity index (Shannon and Weaver, 1949).

A multi-metric benthic index of biotic integrity (B-IBI) was developed for the NY/NJ Harbor (Appendix C). The EMAP-E 1990-1993 Virginian Province data, excluding Chesapeake Bay and site with salinities less than 15 ppt, were used to develop the index. The B-IBI incorporated five of the benthic macroinvertebrate metrics in Table 2-4 into a single value that described the condition of the benthos. These five metrics were those which most effectively distinguished normal sites from all others. The metrics were evaluated for four different salinity and grain size habitats (Table 2-5) and threshold values were defined for each.

Table 2-4
Individual Benthic Macroinvertebrate Measures Assessed

Species Diversity	**Species Composition**
Number of taxa (#)*	Abundance of pollution-indicative taxa (%)*
Shannon-Wiener Diversity (H')	Abundance of pollution-sensitive taxa (%)*
Abundance and Biomass	**Trophic Composition**
Abundance (#/m^2)*	Abundance of deposit feeding taxa (%)
Biomass (g/m^2)*	Abundance of suspension feeding taxa (%)
	Abundance of carnivores/omnivores (%)

* Measures used in B-IBI.

Table 2-5
B-IBI Habitat Categories

Habitat	
Salinity Class	Sediment Type
Polyhaline (15-28 ppt)	Mud (>40% silt+clay)
	Sand (<40% silt+clay)
Euhaline (28-35 ppt)	Mud (>40% silt+clay)
	Sand (<40% silt+clay)

The index was calculated by scoring each selected metric as 5, 3, or 1 depending on whether its value at a site approximated, deviated slightly from, or deviated greatly from conditions at the best reference sites. The B-IBI value for each station is calculated as the mean score of the five metrics. A mean score of 5 indicated that the site was approximately equivalent to the best reference sites. A score of 3 or 1 indicated that the site slightly deviated or greatly deviated from conditions at the best reference sites and would be considered to have impacted benthos. The overall validation efficiency of the B-IBI was 93%. The average difference between replicates was 0.32. Ninety-one percent of the replicates at the same site scored similarly. At most of the sites where the replicates scored differently, the replicates had similar numerical values, but were on either side of the index threshold of 3.

2.7.4 Condition Estimates

Two types of characterizations were done for this investigation. Individual sub-basins were separately characterized for each parameter, resulting in six characterizations. The “Harbor” characterization includes four of the six sub-basins that are commonly known as the Harbor proper; Jamaica Bay, Newark Bay, Lower Harbor and Upper Harbor. The watersheds, sources, physical and hydrological characteristics of western Long Island Sound and the Bight Apex were significantly different from the Harbor proper.

The condition of each stratum and the Harbor as a whole was assessed in two ways: 1) mean condition and, 2) percent of area exceeding threshold (or critical) values for selected parameters. The spatial distribution of degraded and non-degraded stations was also evaluated using GIS (Geographic Information System) display of individual station results.

This investigation used specific terminology to distinguish different bases for determining “sediment quality.” Sediments with unusually high chemical concentrations were considered “contaminated.” Significant results of sediment toxicity tests indicated “toxic” sediments. Measurable departures from normal benthic macroinvertebrate assemblages indicated “impacted” or “abnormal” benthic assemblages. Only when two or three of these sediment quality indications were abnormal, were the sediments described as “degraded.”

2.7.4.1 Mean Condition

Since the sampling stations within each stratum or sub-basin (except Newark Bay) were selected with equal inclusion probabilities, the mean parameter values for a stratum, *h*, and its variance were calculated as:

$$\overline{Y}_h = \sum_{i=1}^{n_h} \frac{Y_{ih}}{n_h} \qquad \textbf{(1)}$$

$$s_h^2 = \sum_{i=1}^{n_h} \frac{(y_{ih} - \bar{y}_h)^2}{n_h - 1} \qquad \textbf{(2)}$$

where

y_{ih} was the variable of interest (e.g., concentration of mercury), and

n_h was the number of samples collected from stratum h.

The weighted mean value for L strata with combined area A is given by

$$\bar{y}_{st} = \sum_{h=1}^{L} W_h \bar{y}_h \qquad \textbf{(3)}$$

where the weighting factors, $W_h = A_h/A$, ensure that each stratum h is weighted by its fraction of the combined area for all L strata. An estimator for the variance of the stratified mean (3) is

$$V(\bar{y}_{st}) = \sum_{h=1}^{L} W_h^2 Var(y_h) \qquad \textbf{(4)}$$

Strata were combined to develop estimates for the study area as a whole and for the New York/New Jersey Harbor, which includes all strata except western Long Island Sound and the Bight Apex, following Holt and Smith (1979). Confidence intervals were calculated as 1.64 times the standard error, where the standard error was the square root of the variance.

The samples from Newark Bay were treated as a cluster sample, in which the cells formed clusters (areas) of unequal size. Mean parameter values were calculated as area-weighted means:

$$\bar{q} = \sum_{i=1}^{n_h} c_i y_{ih} / C \qquad \textbf{(5)}$$

where

c_i was the area of sampling cell *i*,

C was the combined area of all the cells sampled,

y_{ih} was the variable of interest (e.g., concentration of mercury), and

n was the number of cells sampled.

The standard error was calculated using the jackknife estimator (Cochran 1977; Efron and Gong 1983):

$$F_j = \{[(n\text{-}1)/n]E(:_{(i)} - :_{(@)})^2\}^{1/2}$$

where

$$\mu_{(i)} = \sum_{i \neq j} c_i \overline{y} \;/\; (C - c_j)$$

was the weighted mean value deleting the *j*th cell and

$$\mu_{(\cdot)} = \sum \mu_{(i)} / n$$

was the jackknife estimate of the mean *y* for the *n* cells.

2.7.4.2 Mass Estimates

Total mass of contaminants in surficial sediments were estimated from bulk density and volume of sediment, and contaminant concentration. Wet sediment bulk density was calculated as:

$D = (1\text{-}p)r_s + pr_w$

where: p = porosity (mean of 0.4 assumed)

r_s = density of sediment (quartz, etc.) particles (2.65 g/cm^3)
r_w = density of water (1 g/cm^3).

So, D Ñ (1-0.4)A2.65+0.4 = 2 g/cm^3, or 4 g/cm^2 (= 40@10^6 kg/m^2) in the top two centimeters of sediment. The concentration of mercury, for example, averaged over surficial sediments in the Harbor was 0.74 ppm, dry weight (Table E-1). So, mean mercury concentration Ñ 0.74@(1-p) = 0.74@0.6 = 0.44 ppm, wet weight. Therefore, the mass of mercury in surficial sediments of the Harbor was approximately:

mercury concentration @ unit mass of sediment @ Harbor surface area

= 0.44 mg/kg @ 40@10^6 kg/km^2 @ 501 km^2
= 8,800 kg Hg.

2.7.4.3 Percent of Area Estimates

Estimates of percent of area exceeding selected thresholds (e.g., mercury concentration greater than ERM) were calculated as $p = B/n$ (except in Newark Bay), where B was number of samples exceeding the threshold and n was the total number of samples in the stratum. For strata with equal inclusion probability, the exact confidence intervals for p were calculated from the binomial distribution using the formula of Hollander and Wolfe (1973). Below detection limit values were included as zero for percent of area estimates.

The confidence interval for combined strata was calculated using the normal approximation to the binomial, with the 90% confidence interval of stratified estimates of proportions, p_{st}, estimated as:

$$p_{st} \pm 1.64[\mathrm{Var}(p_{st})]^{1/2},$$

where

$$p_{st} = \sum_{h=1}^{L} W_h \, p_h$$

$$Var(p_{st}) = \sum_{h=1}^{L} W_h^2 Var(p_h)$$

The formulas for estimating means and variances for Newark Bay also were used to estimate the percentage of area in Newark Bay with *y* values that fell into some defined class. An indicator variable, $I_{i,}$ was assigned the value 1 if the value of y_i fell in a specified class, and 0 otherwise. The sample mean and variance of I_i was an estimate of the proportion of area in Newark Bay that had *y* values within the specified class.

2.8 SELECTION OF THRESHOLD VALUES

To conduct the data analyses needed to produce percent of area estimates, threshold values or “levels of concern” were required. The threshold values used were either proposed (proposed SQC), established by regulation or Agency guidance (e.g., *Ampelisca abdita* toxicity), or were screening guidelines (e.g., contaminant ERLs and ERMs).

2.8.1 Physical Data Thresholds

For grain size, a value of 40% silt-clay was used to distinguish between sand (<40% silt-clay) and mud (>40% silt-clay) substrate. This cut-off was established using cluster analysis on Environmental Monitoring and Assessment Program (EMAP) data from 525 randomly selected sites, sampled between 1990 and 1993 in the Virginian Province.

2.8.2 Chemical Data Thresholds

For chemical contaminants, three conventions were evaluated: 1) the "Effects Range-Low (ERL)” and “Effects Range-Median (ERM)" values of Long and Morgan (1991) and Long et al. (1995a); 2) two conventions which incorporate equilibrium partitioning theory (U.S.EPA, 1994): Proposed Sediment Quality Criteria (SQC) and Acid Volatile Sulfides (AVS); and, 3) aluminum normalization for metals (Appendix D).

For determination of potential biological effects, this study’s chemical data, except dioxins and furans, were evaluated using the effects-based guidelines of Long and Morgan (1991) and Long et al. (1995a). This approach utilizes data from laboratory spiked bioassays, equilibrium partitioning models and synoptic chemical and biological data from field surveys. Ranges of chemical concentrations are determined that are usually associated with biological effects (Effects Range-Median or ERM), and at which biological effects begin to be seen (Effects Range-Low or ERL). New York State has adopted some of these ERLs and ERMs for Sediment Guidance Criteria (NYSDEC, 1994 and 1996). The Long and Morgan (1991) and Long et al. (1995a) values were used because they include thresholds for most of the chemicals that were measured, allowing this study to provide an integrated contaminant response. Alternative thresholds and evaluation methods, such as proposed sediment quality criteria (U.S.EPA, 1994), SEM-AVS (DiToro et al., 1990; NOAA, 1995) and aluminum normalization (Appendix D) also were applied.

Concentrations of seventeen dioxin and furan congeners also were measured in sediments of three sub-basins: Jamaica Bay, Lower Harbor and Upper Harbor. Sediments that are contaminated with dioxins and furans contain a complex mixture of congeners. Individual congeners differ greatly in their toxicity and carcinogenicity and although specific individual congeners may not be present in concentrations of concern, the combined effect of existing concentrations may be toxicity. A “toxicity equivalency factor (TEF)” was applied to each congener, then summed across all dioxin and furan congeners to give “toxicity equivalents (TEQ)”. This permitted estimation of total dioxin/furan toxicity (U.S.EPA, 1989; Cura et al., 1995). The TEQs calculated were for human health application. TEFs for aquatic organisms are still in the development stage and do not address all congeners. Therefore, comparison to interim guidelines was made for 2,3,7,8-TCDD risk to aquatic life and associated wildlife. A level of 100 pg/g 2,3,7,8,-TCDD has been suggested as interim guidance for high risk to sensitive fish species (U.S.EPA, 1993c).

2.8.3 Sediment Toxicity Thresholds

Significant toxicity for the amphipod, *A. abdita*, was defined as survival less than or equal to 80% of the mean control survival and statistically different ($p<0.05$) from controls (U.S.EPA/U.S.ACE, 1991). For Microtox™, a significant effect was defined as an EC_{50} statistically less ($p<.05$) than the performance control and 70% or less of the control EC_{50}. This 70% criterion is used by the Puget Sound Dredge Disposal Analysis (PSDDA, 1989).

2.8.4 Benthic Index Thresholds

Threshold values for each measure (metric) in the NY/NJ Harbor Benthic Index of Biotic Integrity (B-IBI) were established based on the distribution of its values at reference sites. Similar to the Index of Biotic Integrity (IBI) approach (Kerans and Karr, 1994), each measure was scored as 5, 3, or 1 based on whether its value at a site approximated, deviated slightly from, or deviated greatly from conditions at the best reference sites. Threshold values were established at the 5th and 50th (median) values for reference sites in each habitat. Metric values below the 5th percentile compared to the reference sites were scored as a 1; values between the 5th and 50th percentile were scored as a 3; and values above the 50th percentile were scored as a 5. An index value for a location was calculated by taking the mean of the scores for the individual measures at a location. If the mean of all the benthic index metrics at a location was less than or equal to 3, the location was considered to have impacted benthos.

3.0 PHYSICAL PARAMETERS

3.1 BACKGROUND

Many factors potentially influence chemical and biological measurements. The measurement of physical parameters provides information necessary to interpret chemical and biological data accurately. Sediment grain size and total organic carbon content can determine the magnitude and distribution of contaminants (Burton, 1995). Fine-grained sediments generally retain more contamination than sands because of the greater surface area to volume ratio of fine particles and surface electric charges that can render them more chemically and biologically reactive (Plumb, 1981; Power and Chapman, 1995). Physical characteristics, such as salinity, sediment type and depth, are important parameters because they can influence the distribution and abundance of benthic assemblages (Snelgrove and Butman, 1994; Holland et al., 1989).

At each site where sediment was collected, water column depth, temperature, salinity, and dissolved oxygen were measured. The water column measurements consisted of a single CTD profile at each station. Physical characteristics of the sediments included grain size (as % silt-clay) and total organic carbon (TOC) content.

3.2 CHARACTERIZATION OF THE HARBOR

3.2.1 Depth

All Harbor sub-basins, except the Upper Harbor, had similar mean depths (Table 3-1). The Upper Harbor mean at 10 m, was 3-4 m deeper than other sub-basins in the Harbor. The mean

Table 3-1

Area-Weighted Means of Depth and Sediment Physical Parameters

(± 90% confidence interval)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W.LI. Sound	Bight Apex
Depth (m)	6.9 ±0.9	6.4 ±1.1	6.7 ±1.5	5.9 ±1.2	10.1 ±1.8	16.6 ±2.4	22.2 ±2.8
% Silt-Clay	34.8 ±6.1	30.3 ±9.7	68.1 ±8.6	26.8 ±8.8	51.0 ±10.1	63.2 ±10.5	7.7 ±3.4
% TOC	1.9 ±0.3	1.9 ±0.7	2.3 ±0.6	1.7 ±0.4	2.5 ±0.5	2.3 ±0.7	1.2 ±0.4

depth for the entire Harbor was 7 m. Portions of the Harbor are dredged to maintain shipping channels.

3.2.2 Percent Silt-Clay

The mean percent silt-clay varied greatly among sub-basins (Figure 3-1). Average percent silt-clay in sediments of the entire Harbor was 35%. Newark Bay was the muddiest sub-basin with 68% silt-clay and Lower Harbor was the sandiest with only 26% silt-clay. These same patterns were also apparent when results are expressed as areal extent. In terms of spatial extent, 39% of the Harbor is predominantly mud (>40% silt-clay). Eighty-five percent of Newark Bay was dominated by mud compared to 29% of Jamaica Bay and Lower Harbor.

3.2.3 Total Organic Carbon (TOC)

The average total organic carbon (TOC) in Harbor sub-basins ranged from 1.7 to 2.5%, with the sub-basins not significantly different from one another (Table 3-1). When TOC was examined on an areal basis, the sub-basins were not as similar, with Upper Harbor, Newark Bay and Lower Harbor having a considerable percent of area with TOC exceeding 1.5% (Figure 3-2). Sixty-two percent of the sediments in the Harbor contained between 0.5 and 3.4% TOC. There also were no sites in Newark Bay where TOC was less than 0.5%, whereas TOC less than 0.5% occurred over at least 10% of the area in every other sub-basin.

3.2.4 Water Column Profile

The water column results are all based on a single measurement at each station during the study period.

All sub-basins were similar to one another with regard to mean bottom water temperature during the sampling timeframe (Table 3-2). Means ranged from 20.8°C in Jamaica Bay to 23.4°C in Newark Bay. Mean bottom water temperature for the entire Harbor was 22.0°C.

Mean bottom salinity for the entire Harbor was 26.2 ppt. Newark Bay had an average salinity of 22.4 ppt, which was significantly lower ($p<0.01$) than any of the other systems. The lowest salinity value measured during the study was 1.3 ppt in the Passaic River; all other values exceeded 12 ppt.

In general, dissolved oxygen concentrations are extremely variable temporally and spatially. This study obtained a single measurement of dissolved oxygen at each station. New York City has a more complete dissolved oxygen data set (Brosnan and O'Shea, 1994; 1995) which was

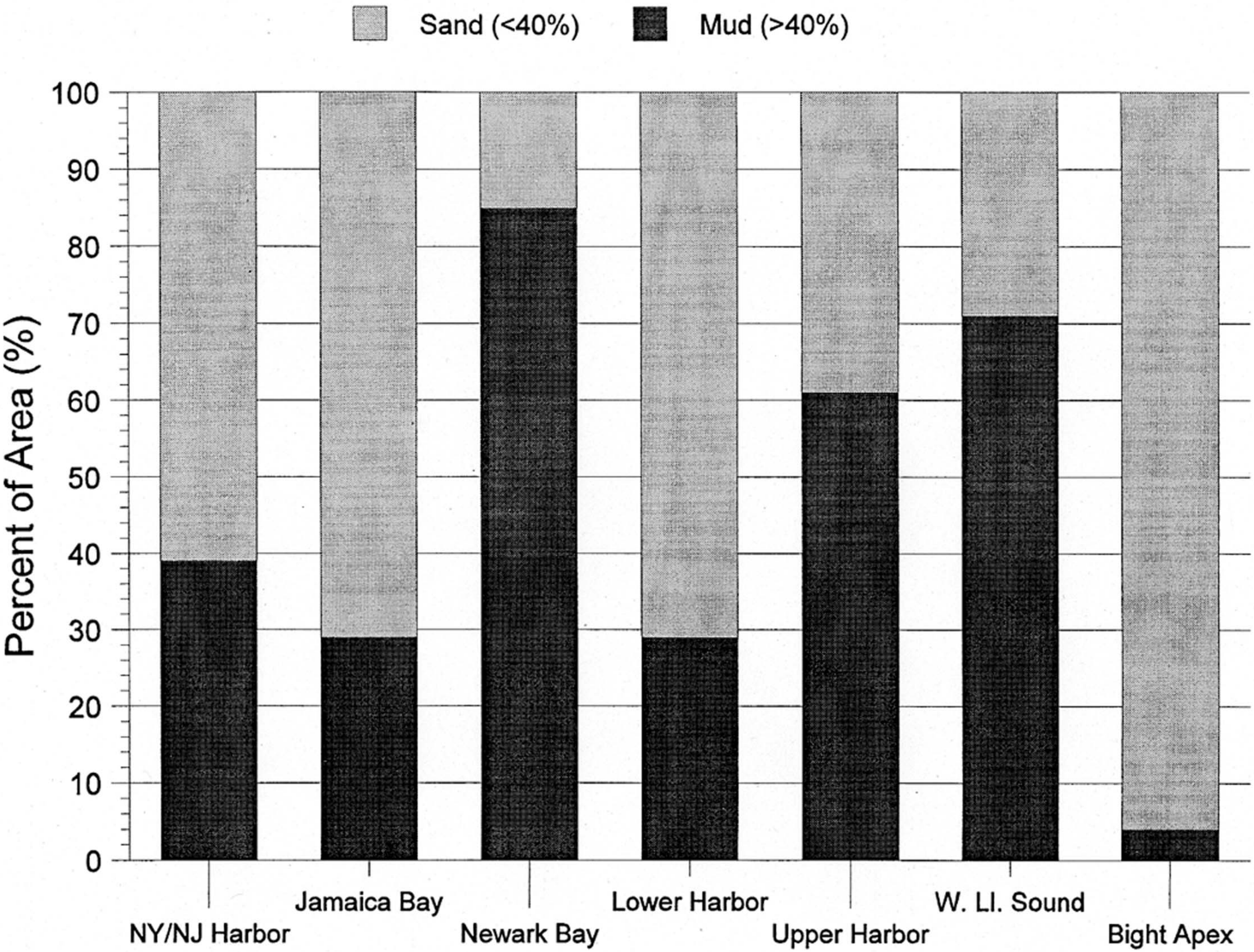


Figure 3-1. Percent of area distribution of substrate type (as % silt/clay).

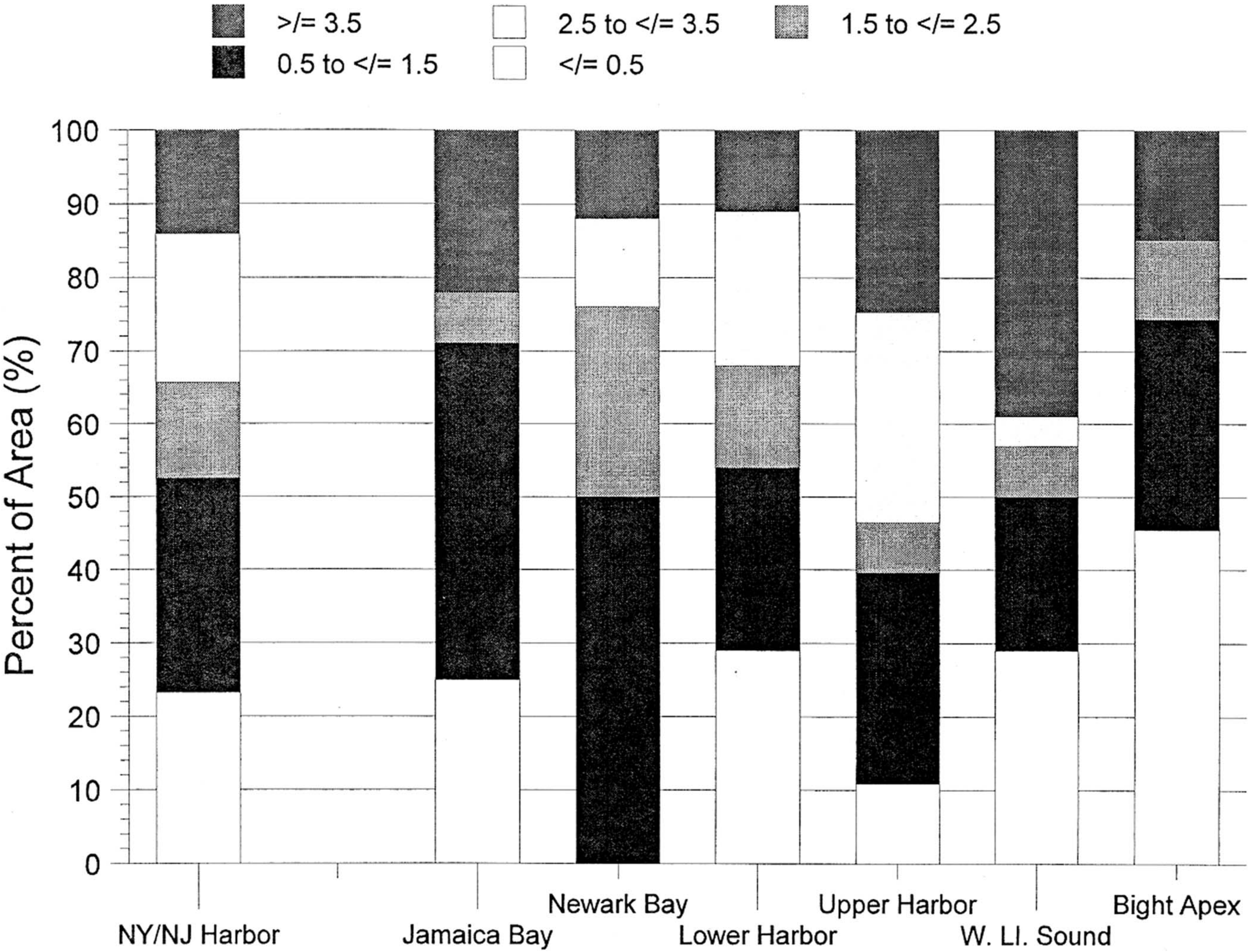


Figure 3-2. Percent of area distribution of total organic carbon levels (%).

used for developing the benthic index. Therefore, dissolved oxygen results from this investigation were not reported here.

Table 3-2
Area-Weighted Means of Water Column Physical Parameters
(± 90% confidence interval)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W.LI. Sound	Bight Apex
Bottom Temp. (°C)	22.0 ±0.5	20.8 ±0.5	23.4 ±0.8	22.3 ±0.7	21.3 ±0.9	17.5 ±0.6	11.1 ±1.2
Bottom Salinity (ppt)	26.2 ±0.4	27.1 ±0.5	22.4 ±0.8	26.9 ±0.5	24.8 ±1.5	28.7 ±0.6	33.6 ±0.9

3.3 CHARACTERIZATION OF WESTERN LONG ISLAND SOUND AND THE BIGHT APEX

3.3.1 Depth

Western Long Island Sound and the Bight Apex were both, on average, about 20 m deeper than the Harbor (Table 3-1).

3.3.2 Percent Silt-Clay

The Bight Apex had a low mean % silt-clay compared to the Harbor (8% versus 35%) and was the sandiest sub-basin in the study (Figure 3-1). The mean % silt-clay in western Long Island Sound (63%) was comparable to Newark Bay (68%) and approximately double that of the Harbor. Fifty percent of the area of western Long Island Sound was mud compared to 4% of the Bight Apex (Figure 3-1).

3.3.3 Total Organic Carbon (TOC)

TOC levels in the Bight Apex were significantly less than in the Harbor (Table 3-1). The Bight Apex had a mean of 1.2 % TOC. The mean in western Long Island Sound was comparable to Newark Bay and Upper Harbor. Forty-six percent of the area in the Bight Apex had less than 0.5% TOC, compared to 29% in western Long Island Sound (Figure 3-2).

3.3.4 Water Column Profile

The average bottom water temperature in the Bight Apex, at 11.1°C, was 11°C less than the Harbor mean and the lowest of all the sub-basins (Table 3-2). The average for western Long Island Sound was 17.5°C.

The Bight Apex had an average salinity of 34.6 ppt, and western Long Island Sound, 28.7 ppt (Table 3-2). Both were higher than the Harbor mean or other sub-basins.

Because of the spatial and temporal variability of dissolved oxygen levels and the fact that this investigation obtained a single dissolved oxygen measurement at each station, those results are not included here.

3.4 COMPARISON TO PREVIOUS STUDIES

Several other investigations and monitoring programs have produced physical data for the Harbor and/or Bight Apex. TOC and grain size in the Harbor have been measured as part of other contaminant investigations (Long et al., 1995b; Strobel et al., 1995) and as study objectives (Suszkowski, 1978; Jones et al., 1979; Coch, 1986). The most spatially and temporally extensive database for dissolved oxygen, temperature and salinity is that of the New York City Department of Environmental Protection (NYCDEP) which sampled 52 stations at least bi-monthly year-round (weekly in the summer) in the Harbor in 1993 and 1994 (Brosnan and O'Shea, 1994; 1995). U.S.EPA conducted monitoring of dissolved oxygen and temperature in the Bight Apex (U.S.EPA-Region 2, 1994g; 1995). The data from the present study were collected to aid in interpretation of other study parameters and were not intended to represent comprehensive temporal coverage.

Physical data collected during this investigation was similar to historical data. The Harbor mean TOC for the present study was 1.9%. The Long et al. (1995b) investigation (using similar sub-basin boundaries to the present investigation) produced a range of 0.07% to 5.0% and a mean of 2.6%. Long et al. found a range of 0.0% to 76.7% silt-clay in the Harbor, with a mean of 39.3% silt-clay as compared to a mean of 34.8% in the present investigation. An earlier study by Coch (1978), incorporating data from Suszkowski (1978), showed that Newark Bay was 66% silt-clay; this study produced an estimate of 68%. The area that is approximately Upper Harbor in the present study had 51% of its area predominantly mud, compared to 34% in the Coch investigation. Other basin boundaries in the Coch investigation were significantly different to preclude direct comparison with the present investigation. Additionally, grain size and TOC distributions in the Harbor may have substantially changed.

Salinity and temperature measurements were similar between this and other investigations.

4.0 SEDIMENT CHEMISTRY

4.1 BACKGROUND

Chemically contaminated sediments, directly and indirectly, pose a significant threat to Harbor resources. Striped bass, bluefish and blue claw crabs from large portions of the estuary should not be consumed because the levels of PCBs and/or dioxins exceed guidelines (NY-NJ HEP, 1996). Areas that were once productive shellfish beds no longer exist or have reduced populations that are restricted for harvesting. Bioaccumulation of contaminants and effects on benthic macroinvertebrate communities also have been observed. Dredging and disposal of contaminated sediments is a major management issue because of the potential adverse biological effects that could result in disposal areas.

For determination of potential biological effects, chemical data, except that for dioxins and furans, were evaluated using the aquatic effects-based guidelines of Long and Morgan (1991) and Long et al. (1995a). This approach utilizes data from laboratory spiked bioassays, equilibrium partitioning models and synoptic chemical and biological data from field surveys. Two concentrations are determined for each chemical that are associated with incidence of biological effects in the dataset that was used for development (Table 4-1). The Effects Range-Low (ERL) value is the concentration at which adverse biological effects begin to be seen, and the Effects Range-Median (ERM) concentration is that usually associated with adverse biological effects. New York State has adopted some of the ERLs and ERMs for Sediment Guidance Criteria (NYSDEC, 1994 and 1996). The Long and Morgan (1991) and Long et al. (1995a) values were used because they include thresholds for most of the chemicals that were measured, allowing this study to provide an integrated contaminant response. Alternative thresholds and evaluation methods, such as proposed sediment quality criteria (U.S.EPA, 1994), SEM-AVS (DiToro et al., 1990; NOAA, 1995) and aluminum normalization also were applied.

Concentrations of seventeen dioxin and furan congeners were measured in sediments of three sub-basins: Jamaica Bay, Lower Harbor and Upper Harbor. Sediments that are contaminated with dioxins and furans contain a complex mixture of congeners. Individual congeners differ greatly in their toxicity and although individual congeners may not be present in concentrations of concern, their combined concentrations may be toxic. A “toxicity equivalency factor (TEF)” was applied to each congener, then summed across all dioxin and furan congeners to give “toxicity equivalents (TEQ)”. This permits estimation of total dioxin/furan toxicity, expressed as an equivalent amount of 2,3,7,8-TCDD (U.S.EPA, 1989; Cura et al., 1995). TEFs for aquatic organisms are still in the development stage and do not address all congeners. A level of 100 pg/g 2,3,7,8-TCDD has been suggested as interim guidance for high risk to sensitive fish species (U.S.EPA, 1993c).

Table 4-1

ERL and ERM Concentrations for Sediment Trace Metals and Organic Compounds

(Long and Morgan, 1991; Long et al., 1995a)

Chemical Analyte	ERL Concentration	ERM Concentration
Trace Elements (ppm)		
Antimony	2	25
Arsenic	8.2	70
Cadmium	1.2	9.6
Chromium	81	370
Copper	34	270
Lead	46.7	218
Mercury	0.15	0.71
Nickel	20.9	51.6
Silver	1	3.7
Zinc	150	410
Polychlorinated Biphenyls (ppb)		
Total PCBs	22.7	180
DDT and Metabolites (ppb)		
DDT	1	7
DDD	2	20
p,p’-DDE	2.2	27
DDE	2	15
Total DDT	1.58	46.1
Other Pesticides (ppb)		
Chlordane	0.5	6
Dieldrin	0.02	8
Endrin	0.02	45
Polynuclear Aromatic Hydrocarbons (ppb)		
Acenaphthene	16	500
Acenaphthylene	44	640
Anthracene	85.3	1100
Benzo(a)anthracene	261	1600
Benzo(a)pyrene	430	1600
Chrysene	384	2800
Dibenz(a,h)anthracene	63.4	260
Fluoranthene	600	5100
Fluorene	19	540
2-Methylnaphthalene	70	670
Naphthalene	160	2100
Phenanthrene	240	1500
Low molecular weight PAHs	552	3160
High molecular weight PAHs	1700	9600
Pyrene	665	2600
Total PAH	4022	44792

4.2 CHARACTERIZATION OF THE HARBOR

4.2.1 Mean Condition

Chemical contamination was found to be pervasive in the Harbor. The mean values for every contaminant for which ERL and ERM thresholds exist, except cadmium, were above ERL levels (Appendix E). The Harbor means for mercury, parent DDT and total PCBs exceeded ERM values.

Of the Harbor sub-basins, Newark Bay had the highest average concentration of all the metals measured, except for manganese and silicon (Appendix E).

The Upper Harbor had the highest mean concentrations of individual and total PAHs and endrin, but for all other organic contaminants, Newark Bay had the highest mean concentration. For chemicals in the DDT family (e.g., parent DDT, DDD, DDE and total DDT), Newark Bay had a mean concentration that was 10 or more times higher than the next highest sub-basin.

The mean concentration of tributyltins in the Harbor was 30.1 ppb. Mean concentrations in Jamaica Bay and Upper Harbor were similar (38.6 and 32.5 ppb). Newark Bay's average concentration was about twice as high (69.3 ppb). Tributyltin threshold concentrations for biological effects have not been defined.

4.2.2 Areal Extent

Chemical contamination was present throughout the Harbor. When expressed on an area basis, 87% of the Harbor exceeded an ERL concentration for at least one contaminant, and 50% of the Harbor exceeded an ERM concentration for at least one contaminant (Figure 4-1).

Within the Harbor, Newark Bay and the Upper Harbor had the most pervasive contaminant problem, with 92% and 79% of their areas, respectively, exceeding an ERM value for at least one chemical. These two sub-basins, at 98% and 100%, also had the highest percent of area exceeding at least five ERLs. The entire Harbor exceeded five or more ERLs at 57% of its area.

Estimates of the percent of area in the Harbor that exceeded an ERL and/or ERM for any metal, pesticide, PAH and total PCBs showed that all contaminant groups appeared to contribute to Harbor contamination (Figure 4-1). No single contaminant group predominated. Metals, pesticides and total PCBs contaminated approximately the same percentages of the Harbor at ERM levels.

Examination of the individual chemicals showed that mercury, chlordane and total PCBs were the most pervasive at levels above ERMs. The most ubiquitous metal was mercury, with 75% of the area of the Harbor exceeding the ERL and 34% exceeding the ERM (Figure 4-2). All other

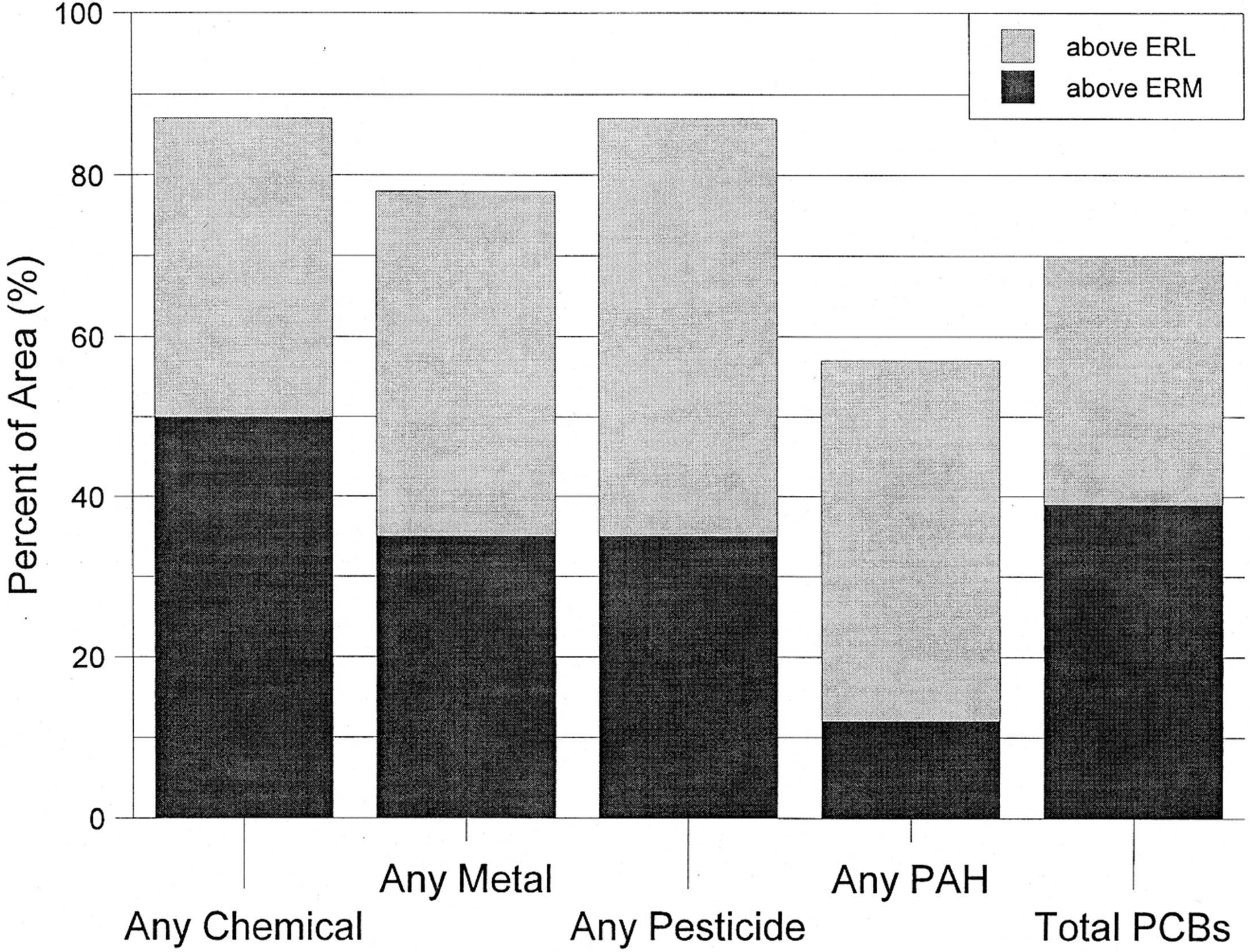


Figure 4-1. Percent of area (%) with any chemical or chemicals within groups exceeding an ERL or ERM (Long & Morgan, 1991; Long et al., 1995a) in the Harbor.

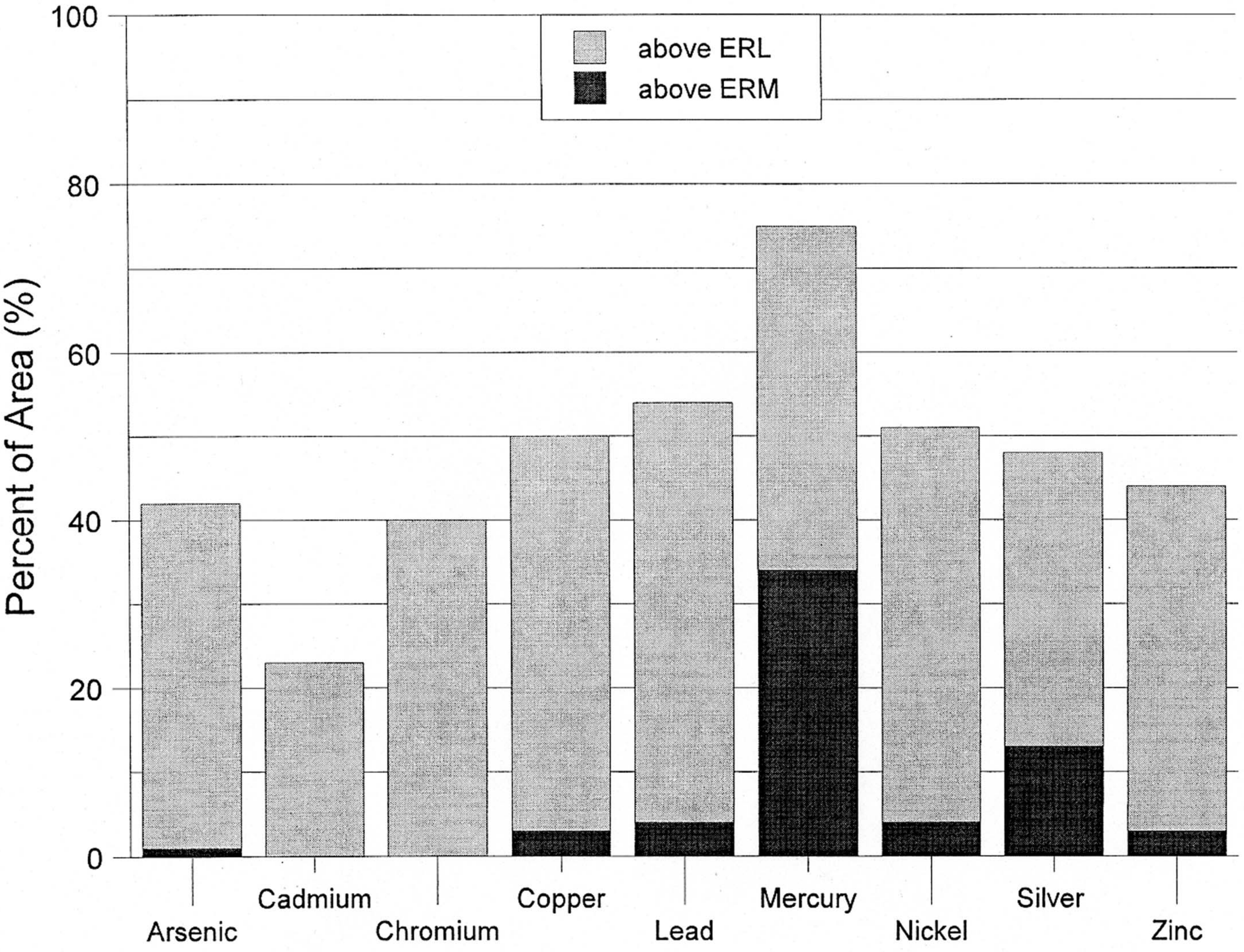


Figure 4-2. Percent of area (%) greater than ERL and ERM values (Long & Morgan, 1991; Long et al., 1995a) for individual metals in the Harbor.

metals also caused significant amounts of area, ranging from 23 to 54%, to exceed an ERL value. Of the pesticides, chlordane resulted in the greatest percent area (32%) above an ERM (Figure 4-3). Organic contaminants above ERL values affected from 56% to 83% of the Harbor area.

The distribution of individual chemicals was not uniform across sub-basins (Appendix E). Many of the individual chemicals were sub-basin specific. Several, such as mercury and chlordane, were ubiquitous. Mercury had the highest percent area of all the metals exceeding an ERM. Focusing on mercury in each of the sub-basins showed that 91% of the area in Newark Bay and 46% of the area in the Upper Harbor exceeded the ERM concentration (Figure 4-4). Chlordane, the most prevalent pesticide Harbor-wide at a level that has probable biological effects, exceeded the ERM in 91% of Newark Bay. All sub-basins had some area above the ERM concentration for chlordane (Figure 4-5).

It was possible to distinguish some general patterns of chemical distribution in sediments. The pattern of mercury distribution in the Harbor indicated that a possible source or sources exist in or above Newark Bay (Figure 4-6). Concentrations were elevated down the Arthur Kill across Raritan Bay to Sandy Hook Bay, following the circulation pattern for this part of the Harbor. Total PCBs (Figure 4-7) and total PAHs (Figure 4-8) exhibited similar patterns.

The actual area above and below specified threshold levels was calculated, in addition to percent of area. Approximately 436 of the Harbor’s 501 km^2 were above the ERL for at least one contaminant and approximately 250 km^2 were above the ERM for a least one contaminant. The total area for specific contaminants also was estimated. Mercury and total PCBs concentrations above ERMs affected approximately the same total area of the Harbor (Figure 4-9). Although Newark Bay had pervasive, elevated levels of these contaminants, because of its small relative size, it did not contribute as much as other sub-basins to the total contaminated area.

Total mass of contaminants in the surficial sediments was also calculated. As an example, the estimated mass of mercury in the Harbor was more than three times that of total PCBs (Figure 4-10). Comparing sub-basins, a higher quantity of mercury and total PCBs was estimated for the Bight Apex, despite the low concentration of these chemicals in that sub-basin.

4.2.3 Dioxins and Furans

Concentrations of seventeen congeners of dioxins and furans were measured at each station in Jamaica Bay, Lower Harbor and Upper Harbor. The biotic effects of dioxins and furans are roughly additive, although congeners differ greatly in their toxicity and carcinogenicity. These features were of concern because most sediments, if contaminated with dioxins and furans, have them present as complex mixtures and although individual congeners may not be present in concentrations of concern, their combined concentrations may be toxic. A "toxicity equivalency factor" has been quantified for each congener, allowing estimation of total dioxin/furan toxicity, expressed as "toxicity equivalents" (U.S.EPA, 1989; Cura et al., 1995).

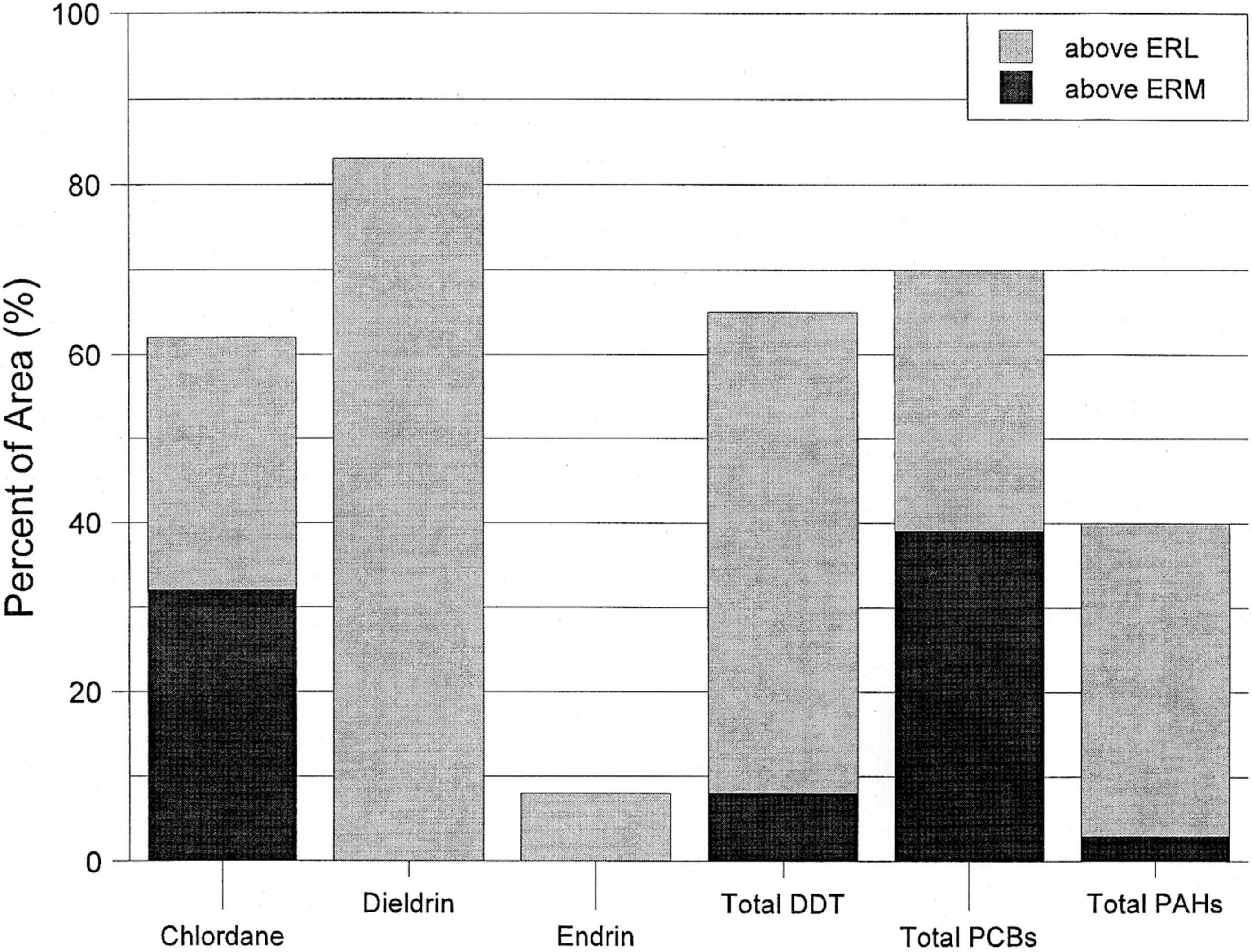


Figure 4-3. Percent of area (%) greater than ERL and ERM values (Long & Morgan, 1991; Long et al., 1995a) for individual or classes of organics in the Harbor.

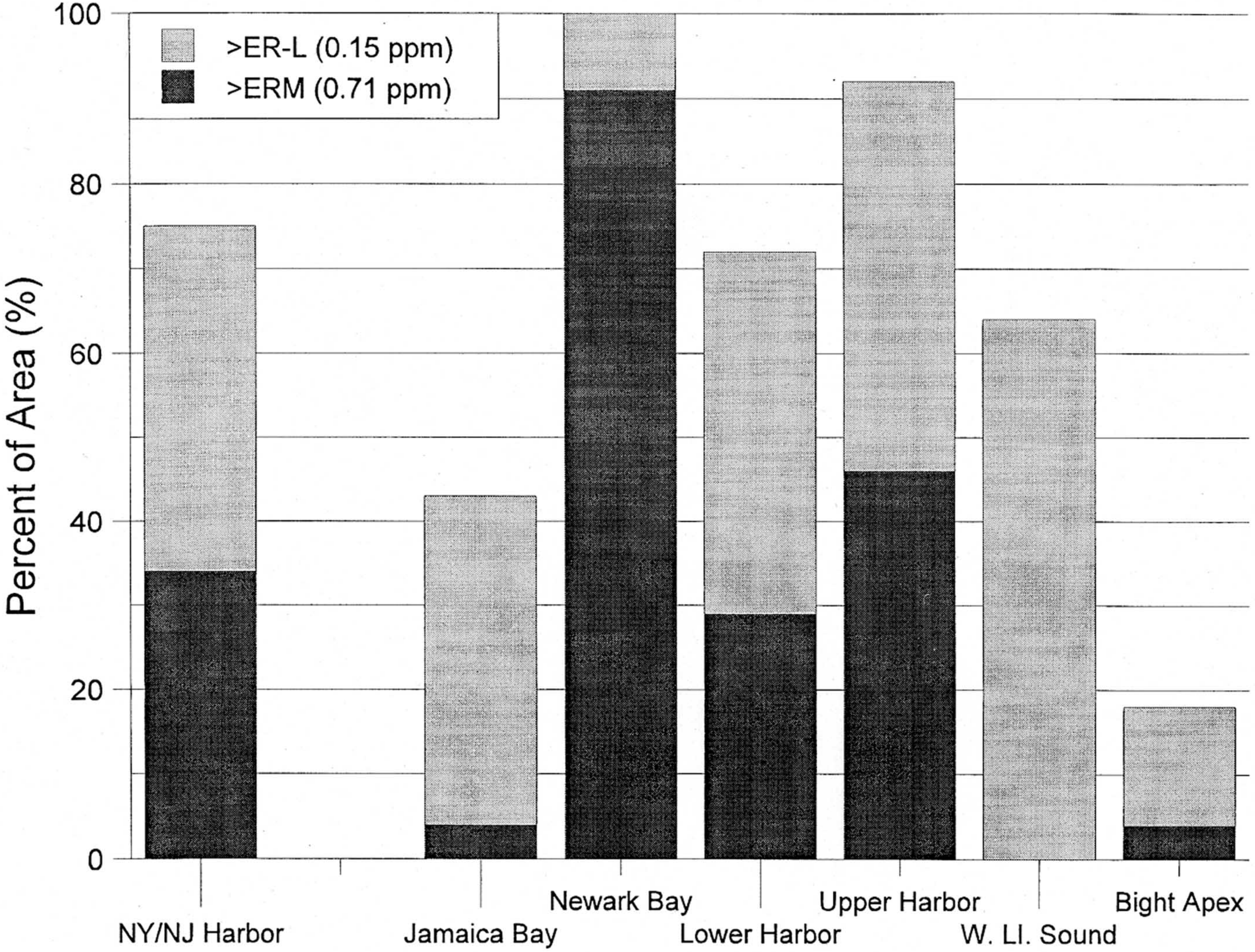


Figure 4-4. Percent of area (%) exceeding the mercury ERL and ERM (Long & Morgan, 1991; Long et al., 1995a).

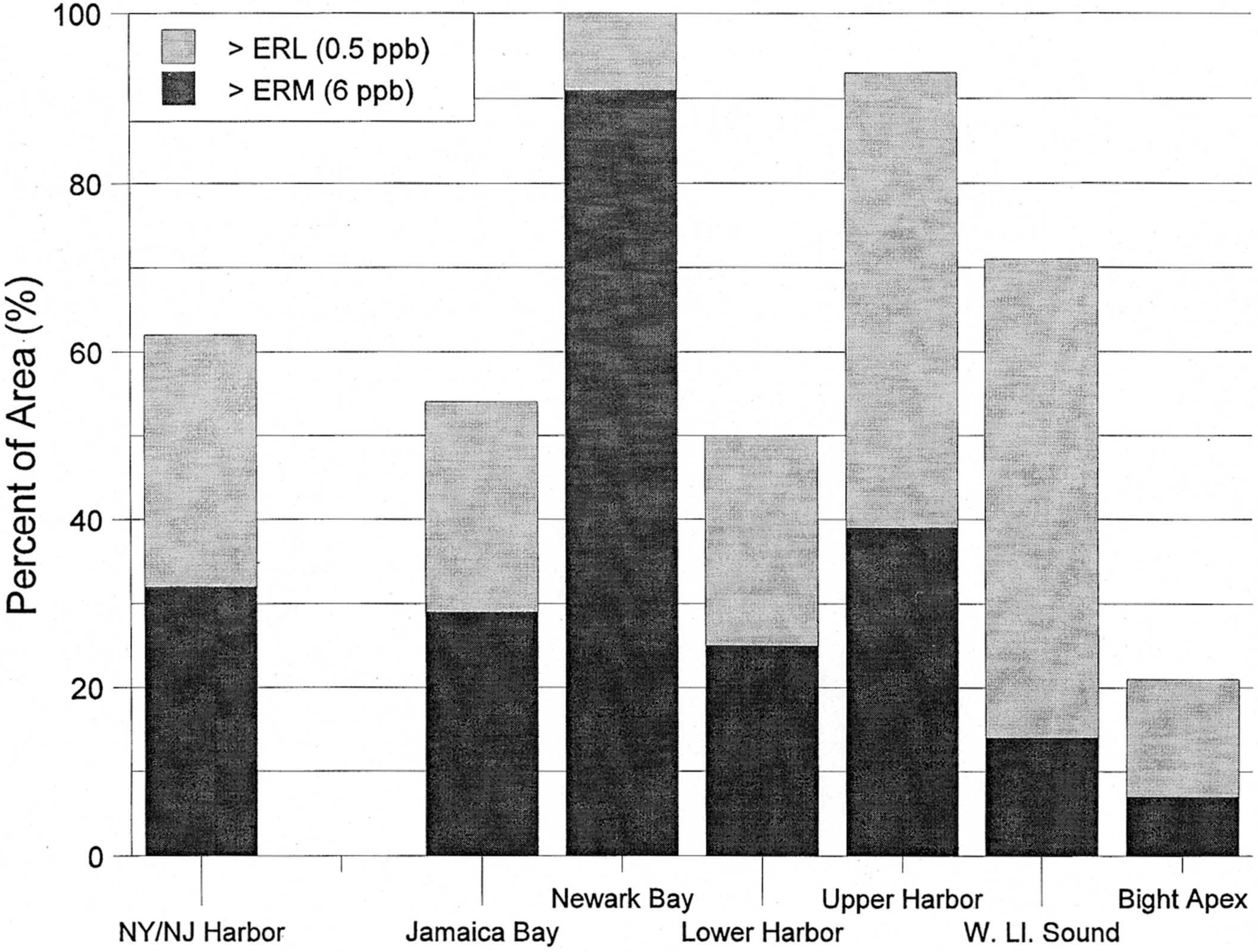


Figure 4-5. Percent of area (%) exceeding the chlordane ERL and ERM (Long & Morgan, 1991; Long et al., 1995a).

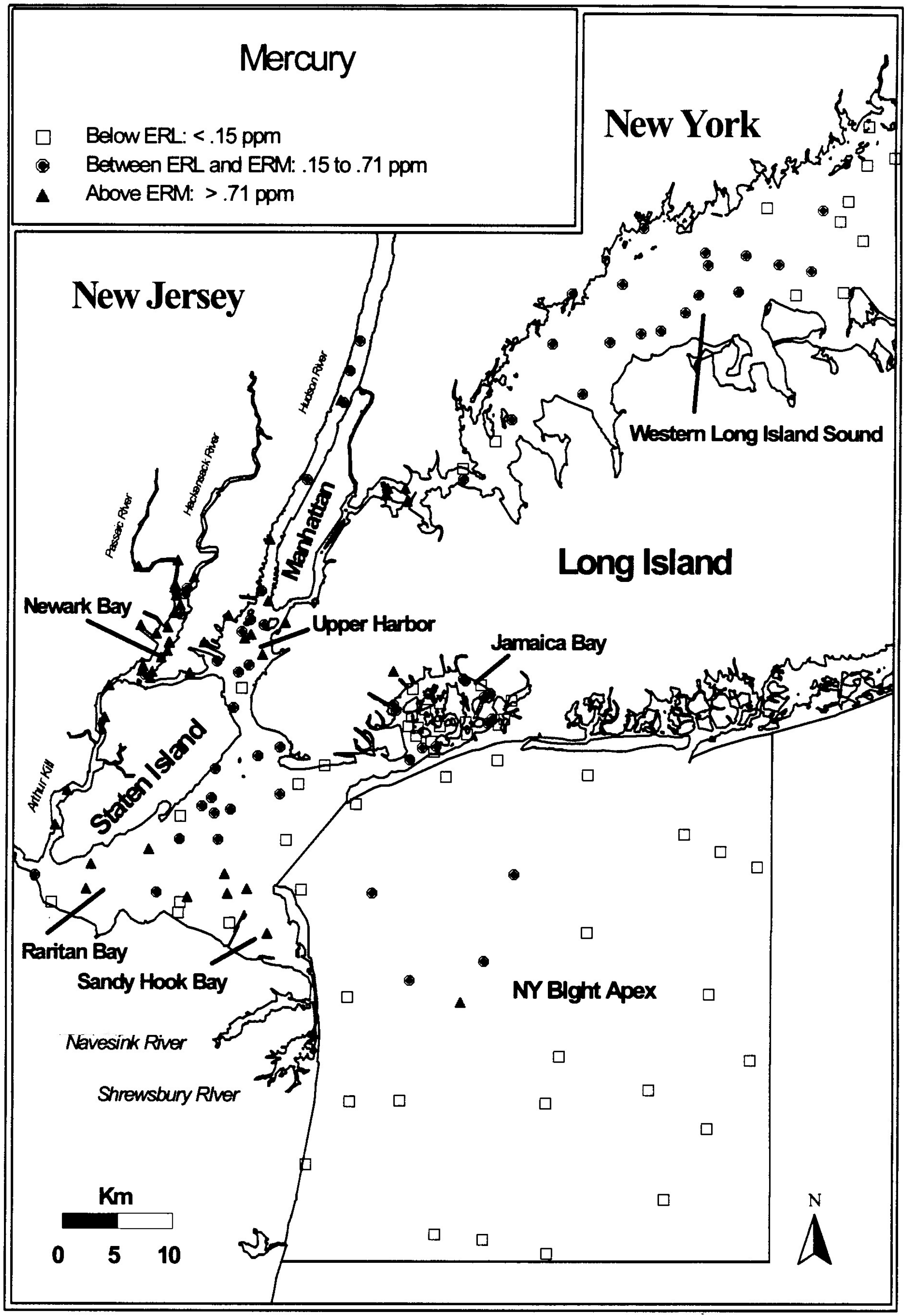


Figure 4-6. Distribution of sediment mercury concentrations by station. ERL and ERM values (Long et al., 1995a) are equivalent to NY State Sediment Guidance Criteria lowest and severe effects levels (NYSDEC, 1994 and 1996).

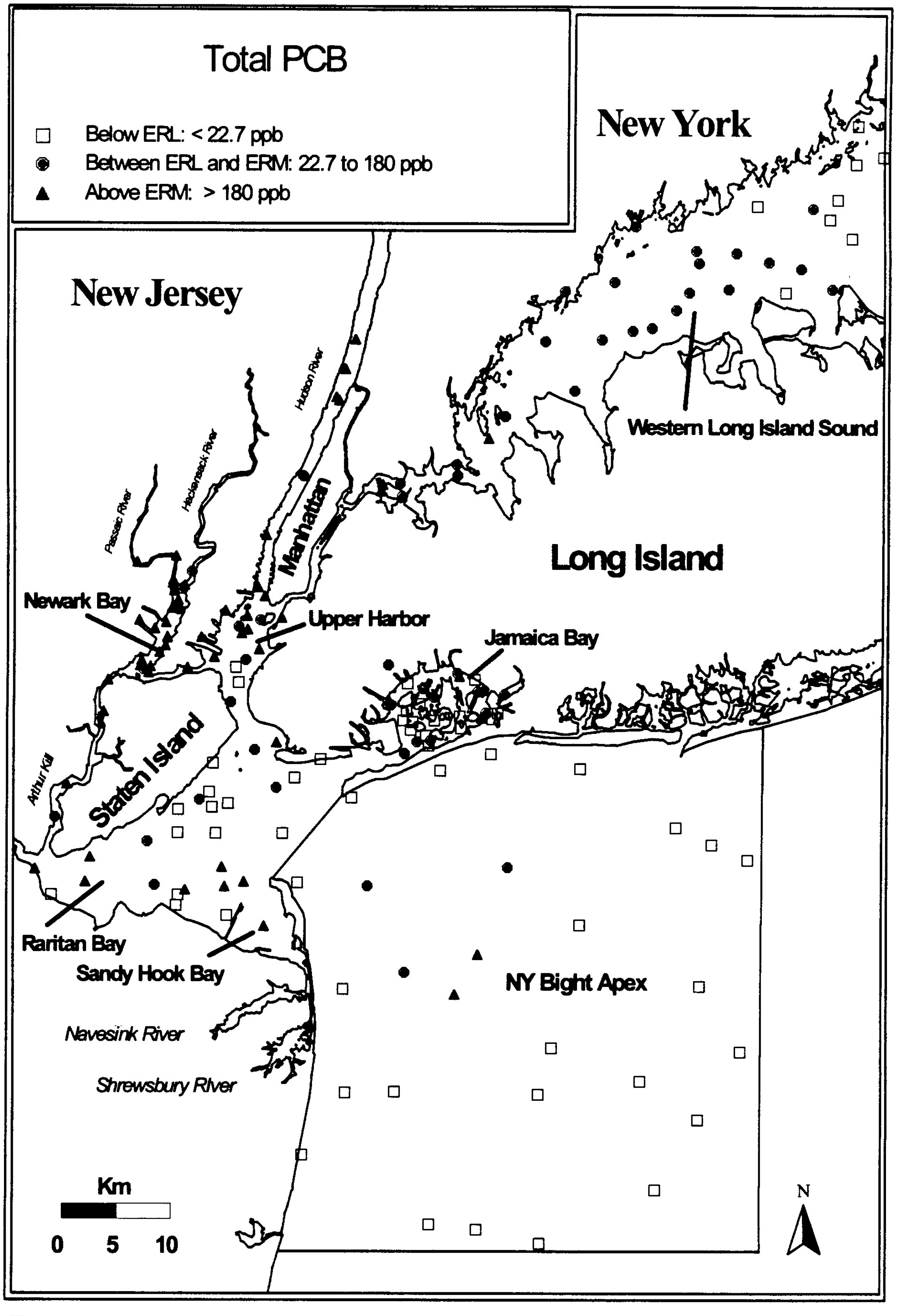


Figure 4-7. Distribution of sediment Total PCB concentrations by station. TPCB is the product of the sum of the 20 congeners in Table 2-3 and 2.0 (NOAA, 1989). ERL and ERM values are according to Long et. al. (1995a).

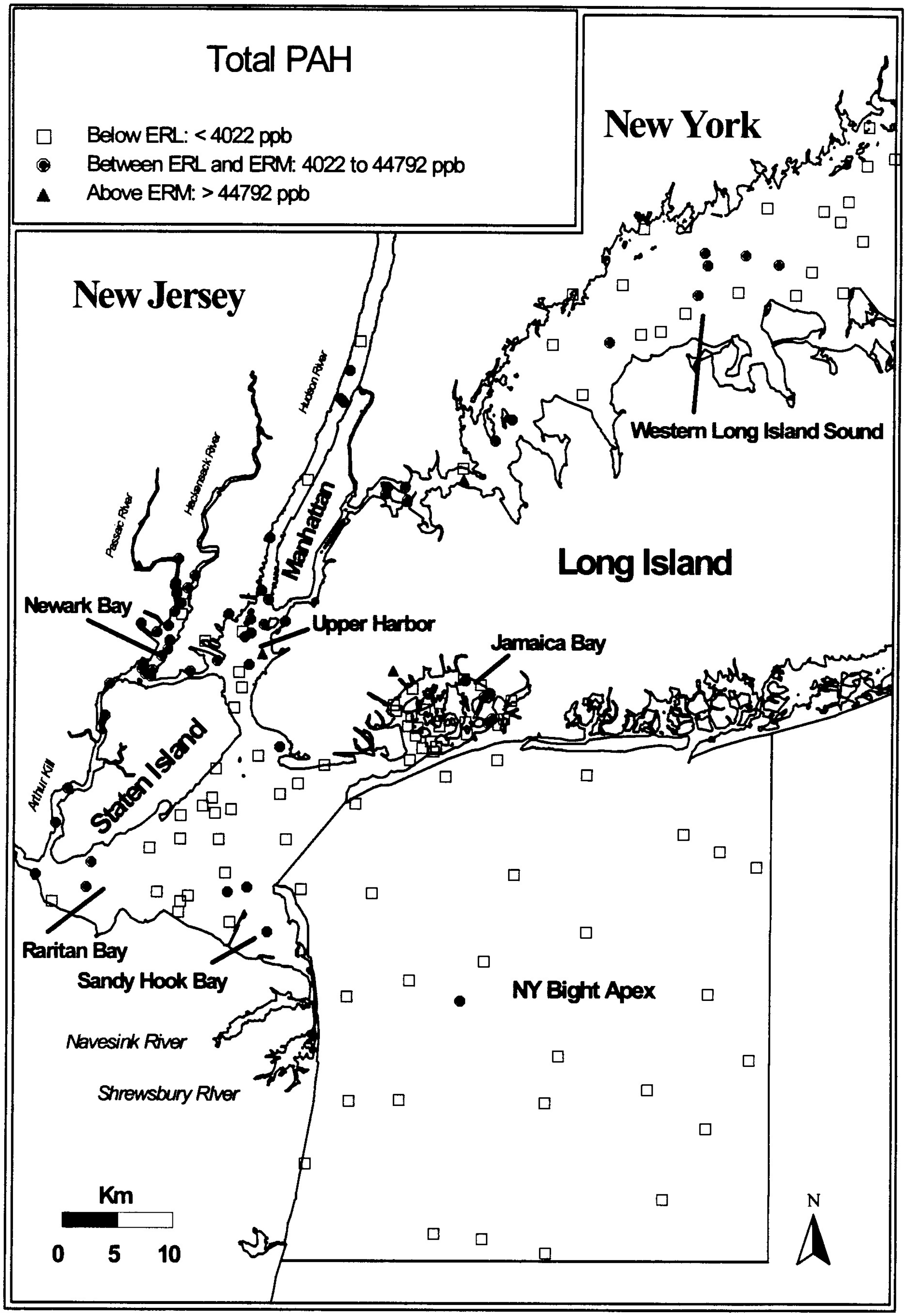


Figure 4-8. Distribution of sediment Total PAH concentrations by station. TPAH is the sum of the 23 individual PAHs in Table 2-3. ERL and ERM values are according to Long et. al. (1995a).

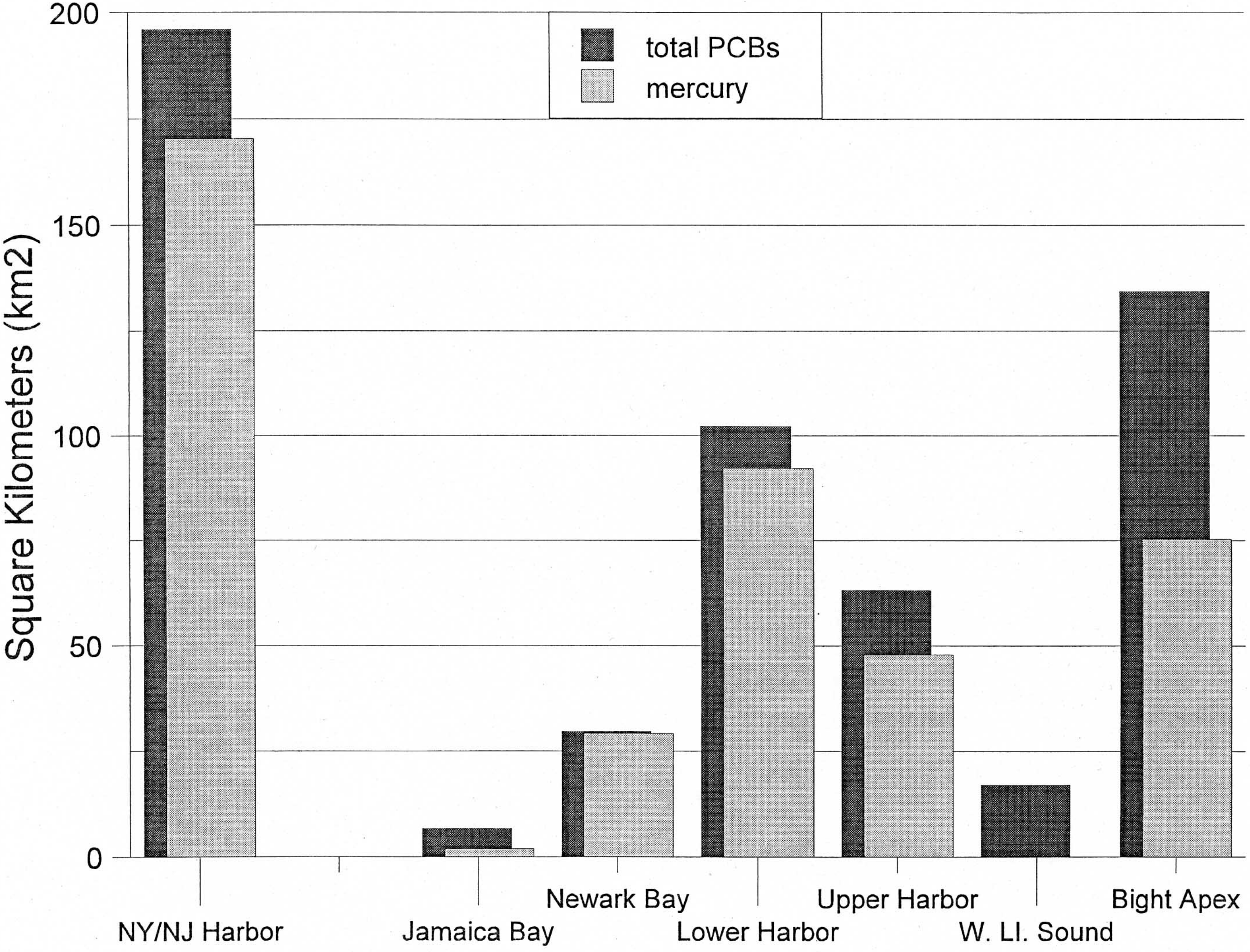


Figure 4-9. Total area (km^2) above the mercury and total PCB ERMs (Long et al., 1995a). The area of the Harbor is 501 km^2.

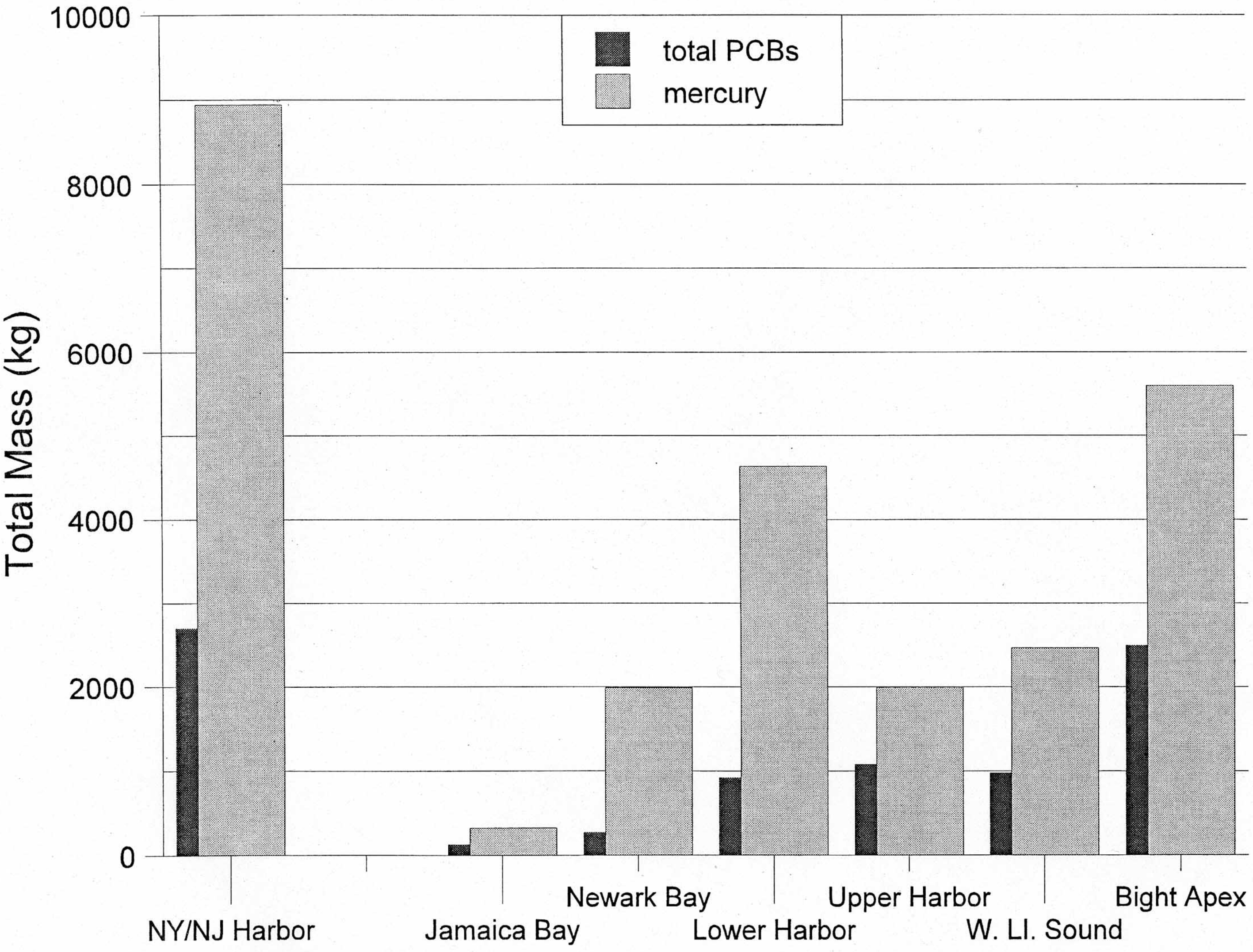


Figure 4-10. Total mass (kg) of mercury and total PCBs in surficial sediments of the Harbor (based on area-weighted mean concentrations).

Among the three sub-basins where dioxins were measured, mean concentrations of the most toxic dioxin congener, 2,3,7,8-TCDD, were highest in Lower Harbor (Table 4-2). Similarly, incorporating all congeners into the calculation of TEQs resulted in the Upper Harbor having a significantly higher amount of 2,3,7,8-TCDD equivalents than the other two sub-basins.

Table 4-2

Mean Concentrations Of 2,3,7,8-TCDD In Sediments Of Three Sub-Basins

(± 90% confidence limits)

	2,3,7,8-TCDD (ng/kg, dry wt.)	Human Health Toxicity Equivalents
Jamaica Bay	4.0 ±2.6	16.4±9.1
Lower Harbor	7.5 ±3.4	17.0±7.1
Upper Harbor	5.5 ±1.8	22.2±5.7

Using the Biota to Sediment Accumulation Factor (BSAF) approach, EPA has derived interim guidance (U.S.EPA, 1993c) for assessing risk of 2,3,7,8-TCDD only, to aquatic life and associated wildlife (Table 4-3). Comparison of the values presented in the interim guidance to

Table 4-3

Interim Environmental Concentrations Associated With TCDD Risk To Aquatic Life And Associated Wildlife (table adapted from U.S.EPA, 1993c)

Organism	**Sediment Concentration (pg/g dry wt.)**
Low Risk	
Fish	60
Mammalian Wildlife	2.5
Avian Wildlife	21
High Risk to Sensitive Species	
Fish	100
Mammalian Wildlife	25
Avian Wildlife	210

Fish lipid of 8% and sediment organic carbon of 3% assumed where needed.
For risk to fish, BSAF of 0.3 used; for risk to wildlife, BSAF of 0.1 used.
Low risk concentrations are derived from no-effects thresholds for reproductive effects (mortality in embryos and young) in sensitive species.
High risk concentrations are derived from TCDD doses expected to cause 50 to 100% mortality in embryos and young of sensitive species.

the 2,3,7,8-TCDD sediment concentrations found in this study indicated less than low risk to fish and avian wildlife, with low risk to mammalian wildlife.

Using a Theoretical Bioaccumulation Approach (TBP) (U.S.EPA, 1993c), this study estimated that concentrations of 3 pptr and 21 pptr (as toxicity equivalents), would be representative of low and high risk, respectively, to mammalian wildlife consuming food contaminated with dioxins and furans (Appendix F). Applying these values to the data from the three sub-basins indicated that none of the mean concentrations found would be considered “high risk” to mammalian or avian wildlife.

4.3 CHARACTERIZATION OF WESTERN LONG ISLAND SOUND AND THE BIGHT APEX

4.3.1 Mean Condition

Overall, the Bight Apex was relatively uncontaminated when compared to the Harbor (Appendix E). Western Long Island Sound had the lowest mean for total chlordane. For all other contaminants measured, the Bight Apex had the lowest mean concentrations.

4.3.2 Areal Extent

Exceedances of at least one ERM were not as common in western Long Island Sound and the Bight Apex (21% and 7%), but western Long Island Sound exceeded at least one ERL in 100% of its area. The Bight Apex, which exceeded more than five ERLs in 18% of its area, did not have as pervasive a pattern of ERL exceedances as the Harbor.

The percent of area above specific ERL and ERM values in the Bight Apex and western Long Island Sound also was estimated. These two sub-basins had 4% and 0% of area exceeding the mercury ERM value, but 18% and 64%, respectively, exceeded the ERL. The Bight Apex had 21% of its area above the ERL concentration for total chlordane and 7% above the ERM.

4.4 ALTERNATIVE THRESHOLDS

The results presented in the last two sections have been based largely on interpreting the chemical concentrations relative to the thresholds suggested by Long et al. (1995a). The Long and Morgan (1991) and Long et al. (1995a) values are emphasized because they include thresholds for most of the chemicals that were measured, allowing this study to provide an integrated contaminant response. Other thresholds and approaches for interpreting sediment chemistry data for a more limited set of chemicals have been suggested. This section interprets the data in the context of some of those alternatives.

4.4.1 Proposed Sediment Quality Criteria

The U.S.EPA has proposed Sediment Quality Criteria (SQC) for five chemicals (U.S.EPA, 1994). This approach was based on equilibrium partitioning theory to establish individual chemical concentrations in interstitial water that do not exceed water quality criteria (WQC) (DiToro et al., 1991). SQC are normalized to the TOC content of the sediment. The approach assumes that water quality criteria are protective of infaunal organisms, chemical concentrations in the interstitial water are in equilibrium with that adsorbed to the sediment particles, and porewater is the primary route of organism exposure. The calculation incorporates an organic carbon normalization step.

Exceedances of SQC's were rare, with none of the chemicals exceeding the SQC threshold for more than 3% of the area in the Harbor (Figure 4-11). For dieldrin and endrin, no samples exceeded SQC's. In addition, none of the samples from the Bight Apex or Western Long Island Sound exceeded SQC for any chemical.

The interpretation based on SQC's was very similar to that based on the ERM thresholds for these chemicals. For three of the five chemicals, there was complete agreement between the two approaches (Figure 4-11). For fluoranthene and phenanthrene, the estimates for percent of the Harbor with exceedances between the two approaches differed by less than 3% and were not significantly different. If comparing SQC to ERL thresholds, there was no agreement, as four of the five chemicals exceed nearly 100% of the Harbor area at the ERL concentrations.

4.4.2 Acid Volatile Sulfides

Equilibrium partitioning theory also has been applied with regard to acid volatile sulfides in sediments. AVS in combination with simultaneously extracted metals (SEM) is used to indicate when several divalent metals (Cd, Cu, Ni, Pb, Zn) would not be bioavailable (DiToro, 1990). If the difference between the molar concentrations of SEM and AVS (SEM - AVS) is <0, the theory states that the sulfides should be binding all of the metal and none should be available to cause toxicity (NOAA, 1995). When SEM is in excess (SEM - AVS > 0), the sediments are described as potentially toxic. This theory does not take into account other contaminants that could be causing an effect.

Thirty-six percent of the Harbor was found to have SEM in excess of AVS (Figure 4-12). Within the Harbor, excess SEM was most prevalent in Lower Harbor, where it occurred over 54% of the area. Excess SEM occurred in only 7% of western Long Island Sound, but occurred for more than half of the Bight Apex.

The SEM-AVS results were highly inconsistent with the ERM or ERL based metals results (Figure 4-12). Based on ERL/ERM, it appeared that metal toxicity should have been highest (or metal non-toxicity lowest) in Newark Bay, where ERM metals concentrations were exceeded for

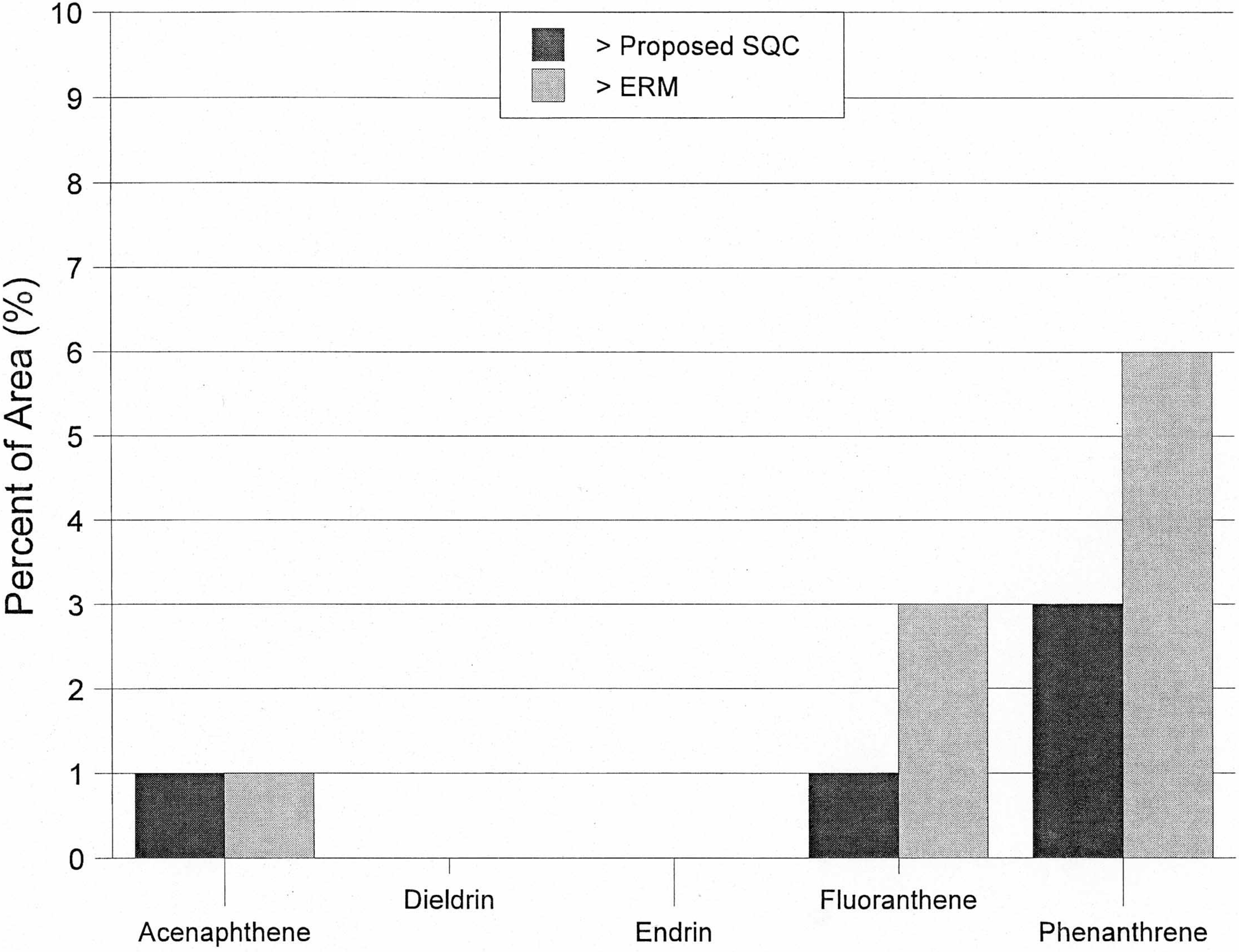


Figure 4-11. Comparison of percent of area (%) in the Harbor exceeding a proposed Sediment Quality Criterion (U.S. EPA, 1994) or an ERM value (Long and Morgan, 1991; Long et al., 1995a). Note: the y axis is 0-10 %.

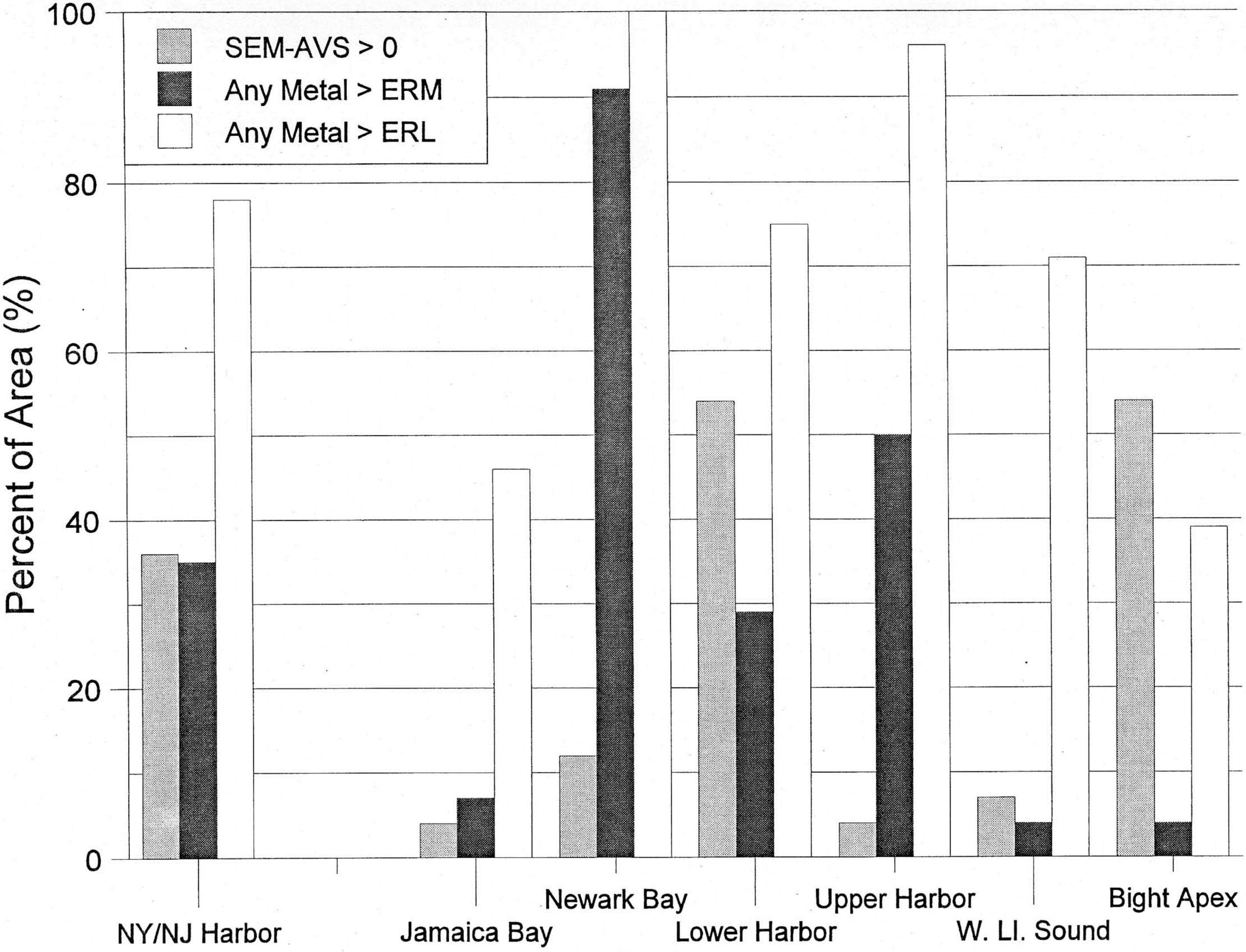


Figure 4-12. Comparison of percent of area (%) with SEM in excess of AVS (SEM-AVS>0) and percent of area with any metal greater than ERM (Long & Morgan, 1991; Long et al., 1995a).

91% of the area. But SEM in excess of AVS occurred at only 13% of Newark Bay's area. In other words, SEM-AVS theory would predict that 87% of Newark Bay should have been non-toxic. However, metal ERLs and ERMs, indicated that 0% and 25%, respectively, was non-toxic. Similarly, in the Upper Harbor, 50% of the area exceeded ERM, whereas only 4% had excess SEM. In contrast, only a single station in the Bight Apex exceeded an ERM for metals, whereas 54% of the area in the Bight Apex had excess SEM.

4.4.3 Aluminum-Normalization

Analyses so far have focused on identifying amount of area where contaminants are at concentrations of biological concern, but another relevant question was what percent of the Harbor has been subjected to anthropogenic enrichment for each chemical compound. Most organic contaminants are of anthropogenic origin, so detection and enrichment are synonymous. However, a portion of the metals in sediments results from natural weathering of crustal rocks, with naturally higher concentrations of metals occurring in fine-grained, depositional sediments. One challenge in accurately assessing the spatial extent of contamination is separating the anthropogenic contribution to observed concentrations of metals from concentrations attributable to natural mineral weathering.

Several techniques have been developed to address this concern (Luoma, 1990, Schropp et al. 1990), the most popular of which is aluminum-normalization (Daskalakis and O'Connor, 1995; Hanson et al., 1993; Loring, 1991; Schropp et al., 1990). Using this approach, aluminum is treated as a conservative tracer of crustal decomposition, since anthropogenic contributions of aluminum are small relative to natural pools in sediment. A set of non-contaminated sites are identified and statistical relationships between each metal and aluminum are established for those sites. Significant deviation from those relationships indicate anthropogenic enrichment. This investigation used the relationships derived by Weisberg et al. (in prep.) for identifying sites with anthropogenic metal enrichment (Appendix D).

Most of the Harbor was found to be enriched in at least one metal (Table 4-4). Nine of the 12 metals measured were enriched over more than 50% of the area of the Harbor. Zinc (80%), mercury (75%), lead and silver (both 70%), were the metals enriching the most Harbor area.

Newark Bay had the highest number of metals enriching greater than 50% of its area (11 out of the 12). The only metal for which Newark Bay did not have the highest percent of enriched area was silver. Upper Harbor had the most enriched area (96%) for this metal. Upper Harbor also was comparable to Newark Bay for enrichment by copper, tin and mercury.

Compared to the Harbor, the Bight Apex had an almost equivalent percent of area enriched with antimony and arsenic. All other values were substantially below the Harbor values. Only one metal, arsenic at 54%, was enriched in more than 50% of the Bight Apex's area. Western Long Island Sound was more similar to the Harbor, with eight of the 12 metals enriching greater than

50% of its area. Copper (93%) and zinc (86%) were the most pervasive in western Long Island Sound, with the extent of copper enrichment being comparable to that found in Newark Bay.

Table 4-4
Percent of Area With Anthropogenically Enriched Levels of Metals
(parentheses represent 90% confidence intervals)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W. LI Sound	Bight Apex
Antimony	54 (32-53)	32 (20-46)	85 (72-97)	54 (40-67)	54 (40-67)	14 (6-27)	46 (33-60)
Arsenic	52 (41-62)	25 (14-38)	88 (79-97)	50 (36-64)	57 (43-70)	18 (9-31)	54 (40-67)
Cadmium	46 (36-56)	36 (24-50)	94 (88-100)	36 (24-50)	68 (54-80)	46 (33-60	7 (2-18)
Chromium	66 (56-76)	43 (30-57)	89 (80-98)	64 (50-76)	75 (62-86)	57 (43-70)	18 (9-31)
Copper	66 (56-77)	50 (36-64)	97 (93-101)	57 (43-70)	93 (82-98)	93 (82-98)	14 (6-27)
Lead	70 (60-80)	50 (36-64)	95 (90-100)	68 (54-80)	79 (65-88)	68 (54-80)	25 (14-38)
Mercury	75 (66-84)	43 (30-57)	98 (94-101)	71 (58-83)	93 (82-98)	54 (40-67)	18 (9-31)
Nickel	7 (3-11)	4 (0-13)	48 (26-70)	4 (0-13)	7 (2-18)	0 (0-8)	4 (0-13)
Silver	70 (60-80)	54 (40-67)	94 (89-100)	61 (47-73)	96 (87-100)	71 (58-83)	18 (9-31)
Selenium	63 (52-73)	54 (40-67)	80 (65-95)	61 (47-73)	68 (54-80)	54 (40-67)	21 (12-35)
Tin	38 (28-49)	29 (17-42)	51 (30-73)	36 (24-50)	46 (33-60)	32 (20-46)	14 (6-27)
Zinc	80 (72-88)	50 (36-64)	96 (91-101)	82 (69-91)	82 (69-91)	86 (73-84)	25 (14-38)

4.5 RELATIONSHIP BETWEEN CHEMISTRY AND GRAIN SIZE

Grain size is a controlling factor in the adsorption of contaminants onto sediments. Fine sediments typically accumulate higher levels of contaminants than coarse sediments, due to a higher surface area to volume ratio and surface charges that cause these particles to be more chemically and biologically reactive than coarser particles (Power and Chapman, 1995). Depositional areas, which accumulate fine particles, frequently have higher levels of contaminants than coarse sediment zones.

The 39% of the Harbor that was predominantly mud (>40% silt/clay) had 95% exceedance of at least one ERM (Figure 4-13). This can be compared to the sand portion of the Harbor where only 16% of the area exceeded a contaminant ERM.

4.6 COMPARISON TO PREVIOUS STUDIES

No other investigations have sampled the Harbor for sediment contaminants using a probabilistic sampling approach. Therefore, while the results of other investigations can be compared to the present investigation to confirm general magnitude and variety of contaminants, other investigations cannot be used to compare the areal extent of contaminants from the present investigation. However, the ranges of concentrations and specific contaminants determined for the present investigation generally agree with those obtained from other investigations in the Harbor.

Another sediment quality investigation (Long et al, 1995), took place during approximately the same time period as this investigation but sampled in the winter season. The primary purpose of the Long et al. investigation was to evaluate sediment toxicity and the investigation was conducted in two phases. The first phase sampled Harbor-wide and selected samples for chemical analysis after toxicity test results were examined. The second phase focused on Newark Bay and selected stations prior to sampling for chemical analysis to represent a gradient of contamination.

Mercury in the Long et al. investigation of the entire Harbor, generally ranged from 1.0 to 5.0 ppm, with a few samples from the East River at around 5.0 ppm, and one sample from the Arthur Kill at 15 ppm. The present investigation had mercury concentrations ranging from non-detected to 6.7 ppm with a mean of 1.0 ppm. The highest mercury values (5.4 and 6.7 ppm) were found in the Arthur Kill.

In the Long et al. investigation, total PCBs (sum of 20 congeners) generally ranged from 100 to 200 ppb. Several stations in the Arthur Kill and East River were above 450 ppb. The East River had a high value of 1973 ppb. This investigation had a range of .03 to 2482 ppb PCBs (sum of 20 congeners) with a mean of 205 ppb. In the Upper Harbor sub-basin, three stations in the East River had PCBs concentrations ranging from 373 to 430 ppb. A single station east of Governor's

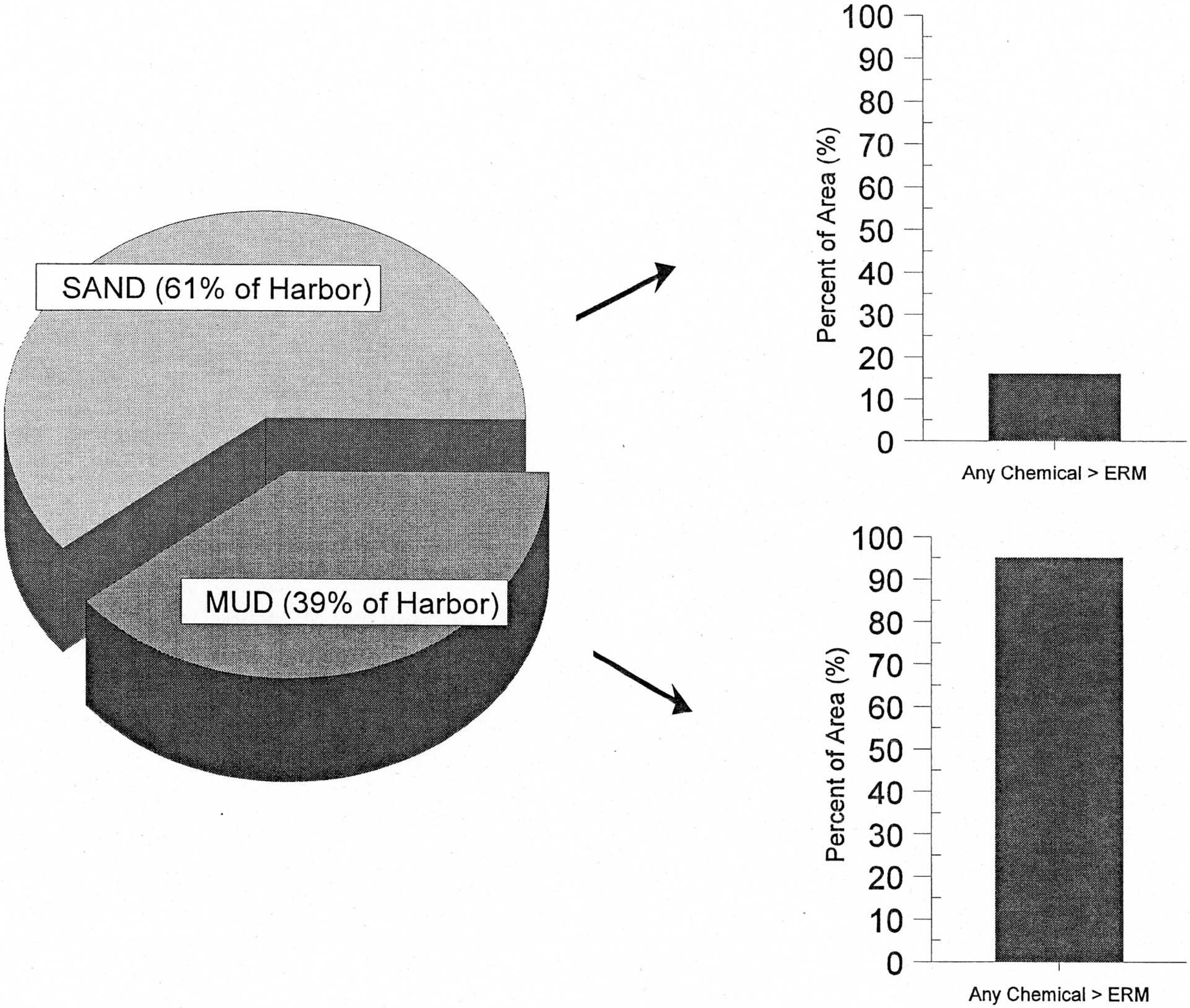


Figure 4-13. Relationship between substrate type (sand ≤ 40% silt/clay, mud ≥ 40%) and percent of area in the Harbor with at least one chemical greater than ERM (Long & Morgan, 1991; Long et al., 1995a).

Island had 425 ppb. One station in the Lower Hudson River was measured at 403 ppb and an Upper Hudson River station had a concentration of 947 ppb. In the Newark Bay sub-basin, a station in the Elizabeth Ship Channel had a value of 1435 ppb. Three stations in the Passaic River were above 650 ppb, with one having the investigation's high concentration of 2481 ppb.

In the Long et al. investigation, total PAHs (sum of 24 congeners) generally ranged from 4,000 to 20,000 ppb. Five sites from the East River and one from the Kill van Kull exceeded 20,000 ppb. The highest concentration of total PAHs was 1,123,355 ppb in the East River.

Concentrations of 2,3,7,8 TCDD (as TEQs) ranged from 13 pg/g at a single reference station in Upper Harbor to 874 pg/g in the lower Passaic River.

Earlier investigations are summarized in a review by Squibb et al. (1991). They concluded that many portions of the Harbor exceeded the ERM values that existed at that time.

4.7 COMPARISON TO A LARGER GEOGRAPHIC AREA

The probabilistic design, sampling methods and laboratory procedures used for the present investigation were the same as those used by the U.S.EPA Environmental Monitoring and Assessment Program (EMAP). This compatibility allowed direct comparison of the data obtained under EMAP and this investigation. The EMAP Virginian Province effort encompassed the coastal zone of the east coast from Cape Cod south to the mouth of Chesapeake Bay. It included the NY-NJ Harbor Estuary.

Comparison to the Virginian Province indicated that the NY-NJ Harbor is heavily and extensively contaminated. The NY-NJ Harbor had a statistically higher ($p< 0.10$) mean sediment contaminant concentration than the Virginian Province for 50 of the 59 chemicals measured (Table 4-5). In addition, for several chemicals, specifically mercury and total PCBs, the Harbor had a large portion (69% and 100%, respectively) of the areal extent of ERM exceedances in the Virginian Province, even though the Harbor constitutes only 4% of the area in the Province (Figure 4-14).

East Coast tributyltin concentrations from purposive sediment sampling ranged from <10 to 770 ppb (Krone, Stein and Varanasi 1996). The levels measured by the present investigation were comparable to the low end of the East Coast range. EMAP, using a probabilistic approach, obtained a similar range for the Virginian Province (12 to 764 ng/g).

Table 4-5

Comparison of Mean Sediment Contaminant Concentrations between the Virginian Province (1990-1993) and the NY/NJ Harbor (1993-1994)

(± represents 90% confidence intervals for NY/NJ Harbor data; ± for Virginian Province data represents the standard error)

	NY/NJ Harbor	Virginian Province	Harbor Larger
METALS (ppm)			
Aluminum	43456 ±4229	35697 ±1238	T
Antimony	1.49 ±0.48	0.54 ±0.024	T
Arsenic	10.33 ±2.05	6.60 ±0.30	T
Cadmium	0.79 ±0.13	0.21 ±0.01	T
Chromium	78.09 ±10.11	37.82 ±1.73	T
Copper	72.53 ±17.40	19.57 ±1.55	T
Iron	23483.6 ±2897.0	19664 ±729	T
Lead	78.84 ±12.83	38.29 ±1.31	T
Manganese	495.26 ±44.14	556.45 ±46.56	
Mercury	0.74 ±0.14	0.09 ±0.01	T
Nickel	24.07 ±2.90	14.39 ±0.87	T
Selenium	3.82 ±1.02	0.27 ±0.02	T

Silver	1.59 ±0.30	0.24 ±0.03	**T**
Tin	4.96 ±1.54	2.34 ±0.14	**T**
Zinc	170.06 ±25.56	79.65 ±4.61	**T**
ORGANICS (ppb)			
Total PCBs = (3congeners) x 2	224.35 ±42.25	17.57 ±3.72	**T**
Parent DDT	9.57 ±9.38	0.58 ±0.07	**T**
DDD	14.16 ±5.98	0.99 ±0.21	**T**
DDE	8.53 ±2.54	1.31 ±0.24	**T**
Total DDT	31.59 ±16.64	2.62 ±0.45	**T**
Aldrin	0.50 ±0.05	0.02 ±0.01	**T**
Alpha Chlordane	1.15 ±0.22	0.29 ±0.09	**T**
Chlordane	5.11 ±1.01	0.47 ±0.15	**T**
Dieldrin	0.80 ±0.12	0.31 ±0.08	**T**
Heptachlor	0.45 ±0.06	0.06 ±0.02	**T**
Heptachlor epoxide	0.39 ±0.05	0.08 ±0.03	**T**
Hexachlorobenzene	0.46 ±0.15	0.03 ±0.01	**T**

Lindane	0.43 ±0.07	0.06 ±0.02	T
Mirex	0.56 ±0.17	0.02 ±0.01	T
Trans-Nonachlor	0.71 ±0.14	0.12 ±0.056	T
Acenaphthene	82.78 ±65.43	23.80 ±13.83	T
Acenaphthylene	122.93 ±41.89	6.96 ±1.69	T
Anthracene	365.05 ±220.76	51.86 ±29.99	T
Benzo(a)anthracene	486.83 ±129.35	99.59 ±46.19	T
Benzo(a)pyrene	433.96 ±116.40	87.25 ±28.38	T
Benzo(e)pyrene	302.69 ±72.98	66.62 ±18.71	T
Benzo(k)fluoranthene	781.78 ±177.51	173.74 ±52.91	T
Benzo(g,h,i)perylene	303.05 ±83.12	65.65 ±17.93	T
Biphenyl	32.16 ±11.74	11.19 ±2.28	T
Chrysene	544.76 ±145.85	110.17 ±45.25	T
Dibenz(a,h)anthracene	79.42 ±31.10	9.99 ±1.49	T
2,6-Dimethylnaphthalene	198.15 ±57.34	22.50 ±4.24	T
Fluoranthene	743.25 ±278.61	217.00 ±105.53	T

Fluorene	176.41 ±182.11	33.82 ±14.93	T
Ideno(1,2,3-c,d)pyrene	291.62 ±90.08	71.22 ±19.49	T
2-Methylnaphthalene	89.91 ±42.02	40.11 ±8.28	T
1-Methylnaphthalene	46.37 ±24.30	18.75 ±4.25	T
1-Methylphenanthrene	156.10 ±88.28	24.46 ±9.84	T
Naphthalene	163.96 ±100.34	47.25 ±10.23	T
Perylene	333.54 ±113.69	85.44 ±11.57	T
Phenanthrene	628.06 ±520.48	194.25 ±117.15	T
Pyrene	767.60 ±269.73	232.29 ±113.38	T
2,3,5-Trimethylnaphthalene	47.00 ±29.87	10.39 ±2.81	T
Total PAHs	7177.4 ±2607.9	1704.30 ±653.64	T
Monobutyltin	5.32 ±1.37	2.23 ±0.33	T
Dibutyltin	16.33 ±6.04	5.11 ±0.76	T
Tributyltin	30.08 ±8.52	8.29 ±0.95	T
Total Butyltin	55.90 ±15.35	15.62 ±1.44	T

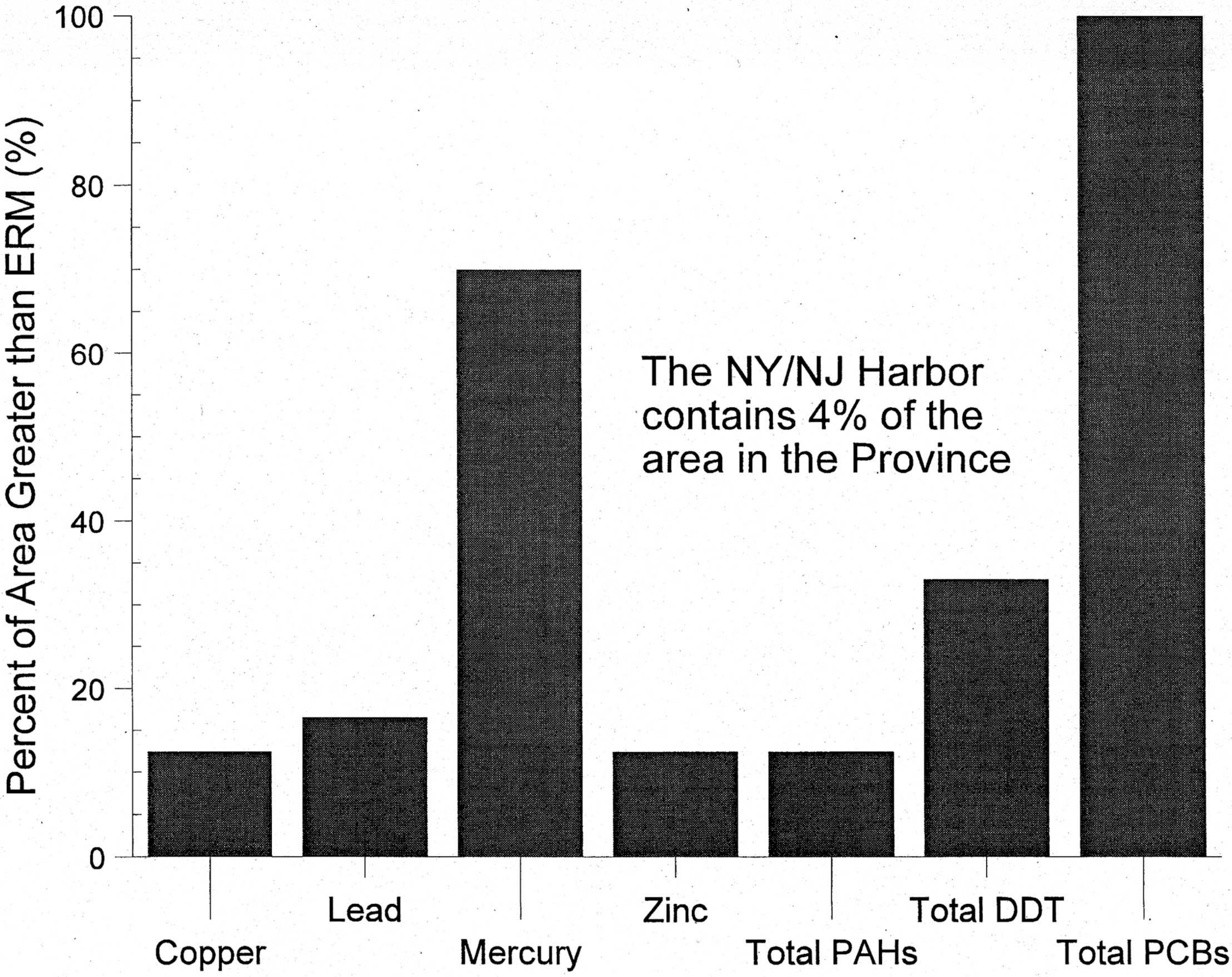


Figure 4-14. Contribution of the NY/NJ Harbor to percent of area in the Virginian Province with ERM exceedances.

5.0 SEDIMENT TOXICITY

5.1 BACKGROUND

The analysis and interpretation of chemical data to determine potential biological response must assume that there is a particular association between chemical contamination and biological effects. Toxicity tests can provide this information directly and can control confounding factors, such as temperature, salinity and dissolved oxygen. They also integrate the effects of complex mixtures of chemicals in sediment, including chemicals that are not measured. However, toxicity tests should not be used alone because individual species of test organisms vary in their sensitivity to chemicals and the relevance of toxicity test results to field conditions is difficult to establish (Chapman, 1995). The value of toxicity tests is best realized when they are interpreted in conjunction with chemistry and in situ biological response (e.g., benthic macroinvertebrate community structure).

This investigation used two measures of toxicity: (1) survival of the amphipod, *Ampelisca abdita*, as a percentage of control survival, and (2) inhibition of light emission by the bacterium, *Photobacterium phosphoreum*, when exposed to organic extracts of test sediment relative to control sediment. Grain size analysis was used to examine the association between toxicity and substrate type.

Sediments at a station were considered toxic using the *Ampelisca abdita* toxicity test if percent survival was less than 80% compared to control. These criteria are similar to U.S.EPA/U.S.ACE (1991). Sediments were considered “highly toxic” if *A. abdita* survival was less than 60% compared to survival in control sediments.

Sediments were considered toxic using the Microtox™ assay if the EC_{50} was 70% or less and significantly different ($p<.05$) from the control EC_{50} (PSDDA, 1989). The degree of Microtox™ toxicity is measured as the dry weight of sediment that provides enough organic extract to inhibit normal bacterial luminescence by 50%, i.e., EC_{50}, (Long and Markel, 1992). Measured Microtox™ toxicity is expressed as a percentage of control EC_{50} (0.12 and 0.22 mg dry wt/ml in this study). Therefore, EC_{50} values and percentages of control values are inversely proportional to sediment toxicity.

5.2 CHARACTERIZATION OF THE HARBOR

5.2.1 Mean Condition

Mean percent survival of *Ampelisca abdita* (as percent of control survival) was comparable within each sub-basin of the Harbor except Newark Bay (Table 5-1). Mean survival within

Newark Bay was significantly less (p<.10) than the Harbor as a whole. Lower Harbor exhibited the highest mean survival.

Mean Microtox™ values varied substantially among Harbor sub-basins, but Newark Bay also exhibited the greatest toxicity relative to the entire Harbor (p<.05). Jamaica Bay and Upper Harbor were similar to one another. Lower Harbor had the least percent of area toxic in the Microtox™ assay.

Table 5-1

Mean % Survival for *Ampelisca abdita* and
Mean % Microtox™ Bioluminescence Inhibition
(± 90% confidence intervals)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W. Long Is.Sound	Bight Apex
*Ampelisca abdita**	87.9 ±4.1	84.9 ±7.7	66.5 ±15.1	91.0 ±5.9	86.6 ±6.3	97.0 ±1.4	94.9 ±1.9
Microtox™ *	365 ±86	257 ±100	122 ±56	452 ±132	224 ±96	237 ±113	765 ±119

* Adjusted for control survival or control bioluminescence inhibition.

5.2.2 Areal Extent

Out of a total area of 501 km^2, an estimated 75 km^2 (15%) of the Harbor proper was toxic to *A. abdita* and 40 km^2 (8% of the total area) was highly toxic (Figure 5-1). Newark Bay and Jamaica Bay have more widespread toxic sediments (46 and 25%, respectively) than the rest of the Harbor. However, only Newark Bay has a larger percent area of highly toxic sediments than other Harbor sub-basins (p<.10). Although relatively large percentages of Newark and Jamaica Bay sediments were toxic, these were the smallest Harbor sub-basins. The total toxic area of these sub-basins (26 km^2) was approximately 1/3 of the acreage of toxic sediments in the entire Harbor. Individual stations toxic to *A. abdita* were concentrated in the Kills (Newark Bay sub-basin) and the mouth of Jamaica Bay (Figure 5-2). Highly toxic stations exhibited a similar pattern, with the addition of several stations in the back bay portion of Jamaica Bay.

Based on the Microtox™ assay, 38% (190 km^2) of the Harbor area was found to be toxic (Figure 5-3). Sub-basins in the Harbor were similar with regard to percent of area toxic in the Microtox™ assay. The estimated percentages of sub-basins considered toxic using Microtox™ ranged from 39% in Upper Harbor to 50% in Jamaica Bay. Over the Harbor proper, the Microtox™ assay characterized 2.5 times more area as toxic than the *A. abdita* assay (Table 5-2).

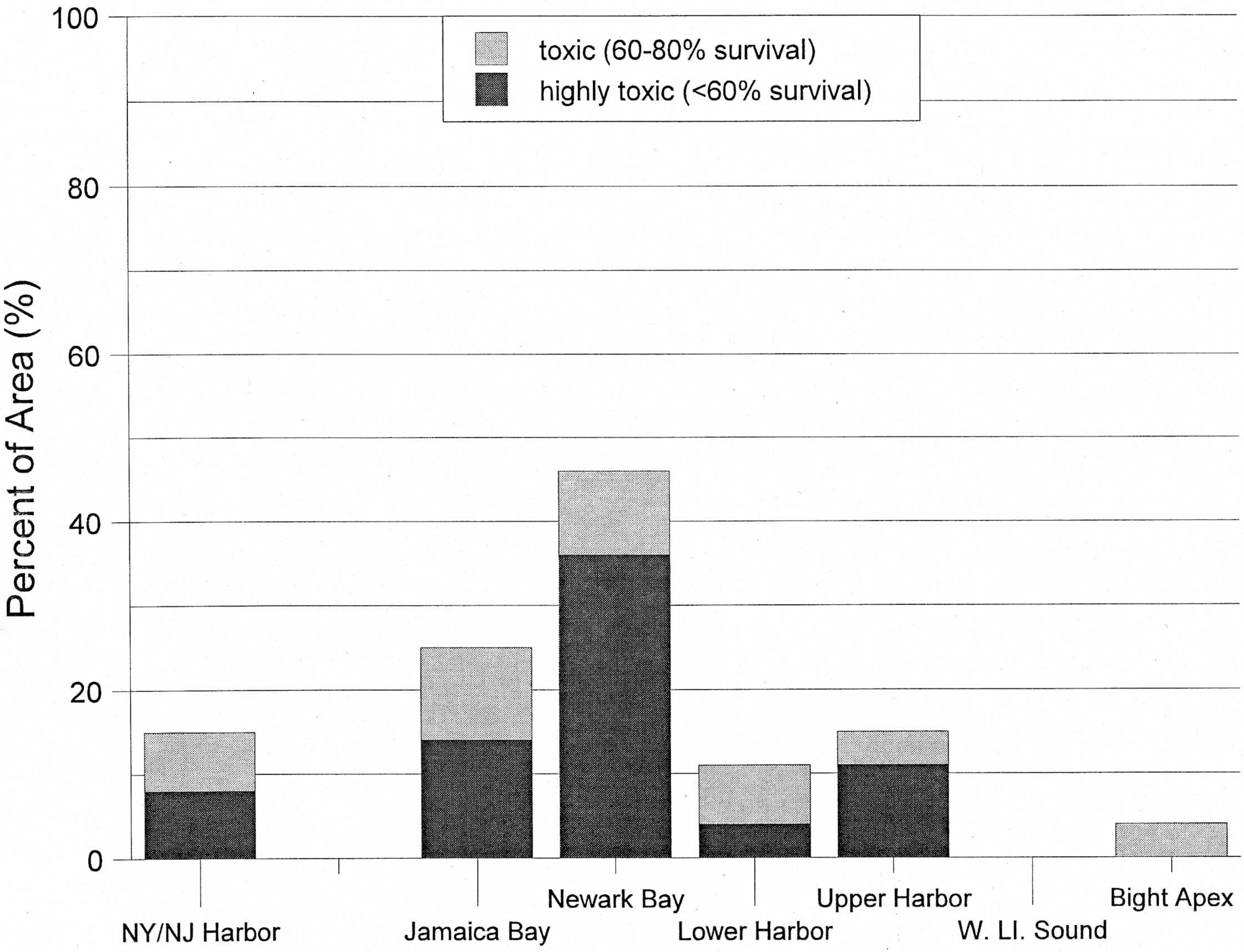


Figure 5-1. Percent of area exhibiting *Ampelisca abdita* toxicity.

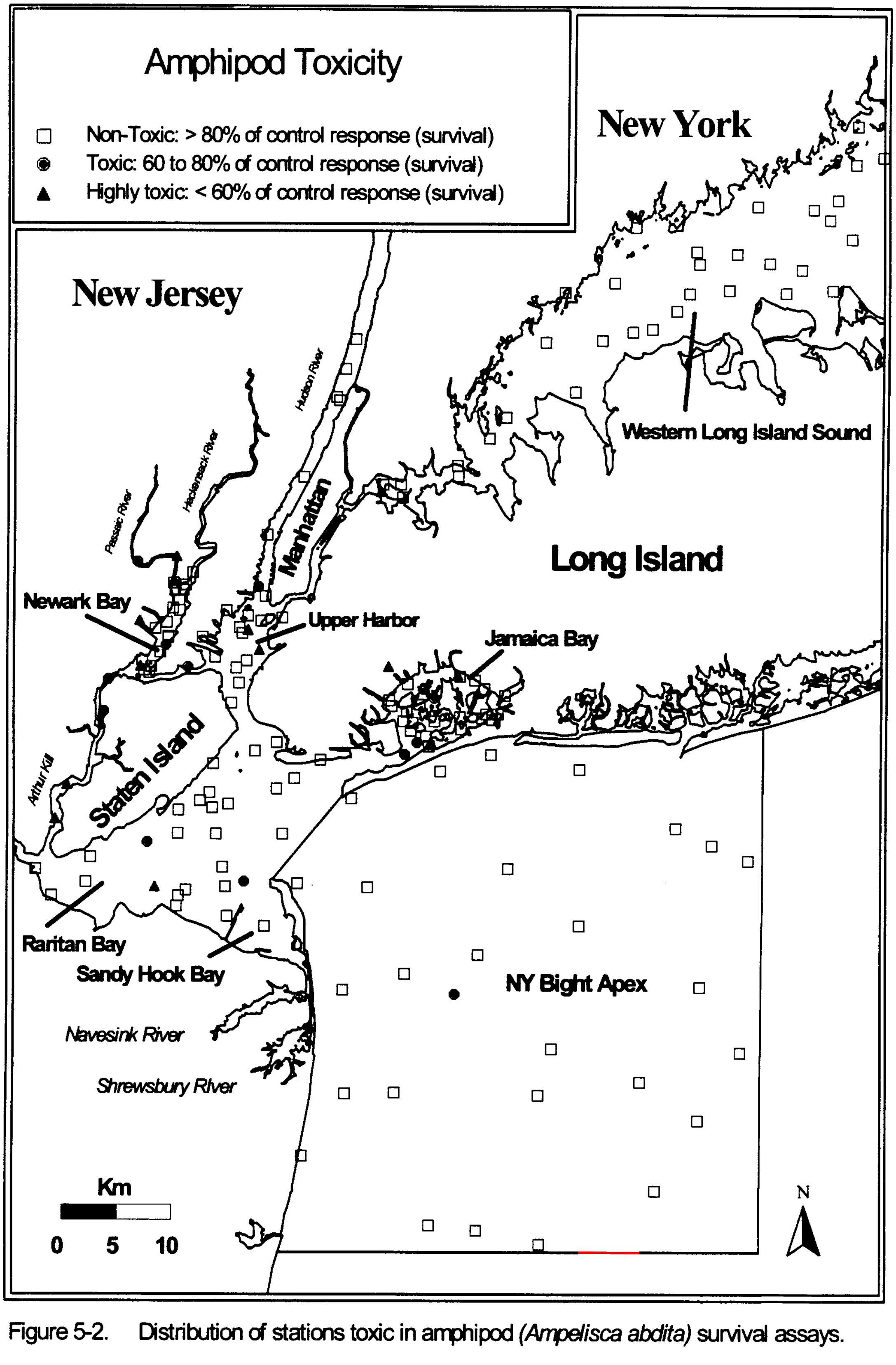


Figure 5-2. Distribution of stations toxic in amphipod *(Ampelisca abdita)* survival assays.

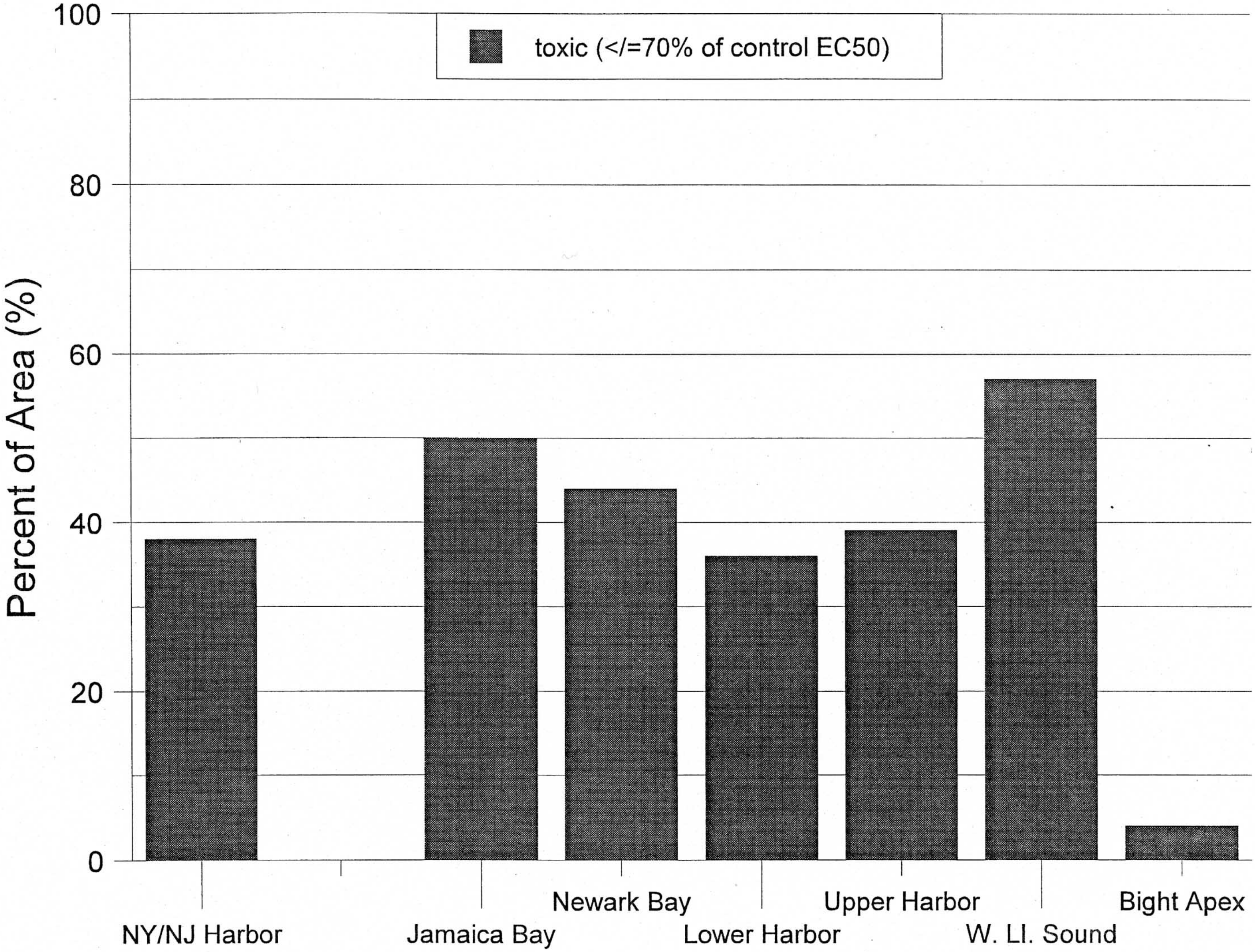


Figure 5-3. Percent of area toxic in the Microtox™ assay.

The majority of the toxic stations for Microtox™ were clustered on the south shore of Lower Harbor, the perimeter of Jamaica Bay, the Kills and the Passaic River (Figure 5-4).

Using a positive result in either assay as an indication of toxic conditions, resulted in 45% of the Harbor considered toxic (Table 5-2).

Table 5-2
Percent of Area Toxic in the *A. abdita* and Microtox™ Assays
(± 90% confidence intervals)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W. LI Sound	Bight Apex
Ampelisca abdita*	15 (8-22)	28 (15-39)	46 (23-68)	11 (0-18)	15 (2-23)	0 (0-8)	4 (0-8)
Microtox™**	38 (28-49)	50 (36-64)	44 (22-67)	36 (24-50)	39 (27-53)	57 (43-70)	4 (0-13)
Ampelisca abdita or Microtox™	45 (34-55)	50 (36-64)	49 (28-71)	43 (30-57)	46 (33-60)	57 (43-70)	4 (0-13)

* Significant toxicity is percent survival # 80% mean control survival (U.S.EPA, 1991).
** Significant toxicity is an EC_{50} statistically less than the control and #70% of the control EC_{50} (PSDDA, 1989).

5.3 CHARACTERIZATION OF WESTERN LONG ISLAND SOUND AND THE BIGHT APEX

5.3.1 Mean Condition

Mean *A. abdita* survival was higher in Bight Apex and western Long Island Sound sediments than in Harbor sediments. Mean survival in these two areas also was higher than any of the individual Harbor sub-basins (Table 5-1).

The least toxic sediments using mean Microtox™ results were in the Bight Apex. Mean western Long Island Sound toxicity was above that for the Harbor as a whole, but was comparable to Jamaica Bay, Upper Harbor and Newark Bay.

5.3.2 Areal Extent

Based upon *A. abdita* assays, the Harbor as a whole and each Harbor sub-basin had proportionately more toxic area than either western Long Island Sound or the Bight Apex (Figure

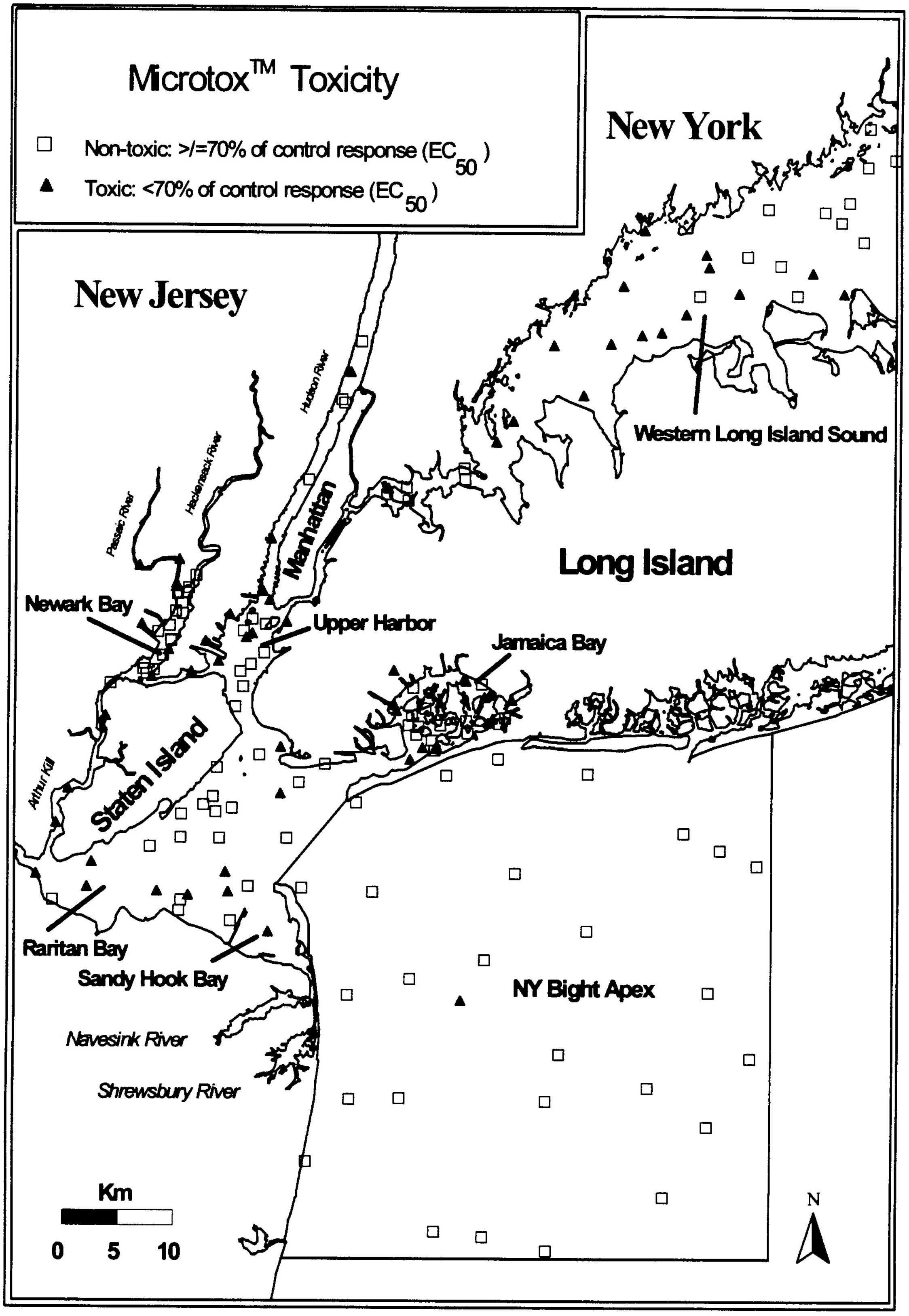


Figure 5-4. Distribution of stations inhibiting Microtox™ bioluminescence.

5-1). There was no significant difference between the percent of toxic sediments estimated in western Long Island Sound and the Bight Apex. However, the areal estimate of toxic sediments in the Bight Apex was 67 km^2, nearly as much toxic area as estimated for the Harbor. The single toxic site in the Bight Apex was located in an area of historical dredged material disposal. Western Long Island Sound had no sites that exhibited *A. abdita* toxicity (Figure 5-2).

More area in western Long Island Sound was characterized as toxic in the Microtox™ assay than in the Harbor as a whole or any other sub-basin. Only 4% of the Bight Apex was classified as toxic. The Microtox™ assay indicates the same extent of toxicity in the Bight Apex as is indicated by the *A. abdita* assay. Individual toxic sites were clustered in the portion of western Long Island Sound closest to the Harbor and at a single location in the Bight Apex (Figure 5-4). The Bight Apex site was also toxic to *A. abdita.* Microtox™ and *A. abdita* results for western Long Island Sound substantially disagree, as 17 sites (57% of the area) were toxic in the Microtox™ assay but none were toxic to *A. abdita*.

5.4 RELATIONSHIP BETWEEN TOXICITY AND GRAIN SIZE

Generally, sediment toxicity is expected to be greater and more prevalent in finer grained substrates (Power and Chapman, 1995). The percent of area in the Harbor with *A. abdita* toxicity was examined by substrate category (Figure 5-5). Of the 39% of the Harbor sediments that were mud (>40% silt-clay), approximately 26% was toxic to *A. abdita* and 76% toxic to Microtox™. Conversely, the sand portion (<40% silt-clay) of the Harbor (61%) exhibited toxicity to *A. abdita* and Microtox™ at 7% and 14% of its area, respectively. Overall, *A. abdita* toxicity was slightly more predominant in mud than sand, but Microtox™ toxicity was significantly higher in mud than sand. Regression analyses of toxicity *vs.* grain size at individual stations showed that *A. abdita* toxicity was not related to the fraction of silt-clay in sediments (P = .05), but Microtox™ toxicity was significantly greater as the fraction of silt-clay increased.

5.5 COMPARISON TO PREVIOUS STUDIES

Previous studies of sediment toxicity in the Harbor generally have used non-random sampling strategies. Although useful for other purposes, these non-probabilistic approaches prevent reliable characterization of the Harbor, or even portions of the Harbor. This is true no matter how reliably the sediment toxicity tests assayed the sampled sediment. However, the intensive sampling of Newark Bay in the Long et al. (1995b) investigation did identify “hot spots” more fully than the present investigation.

This study’s sediment toxicity results were broadly similar to those of Long et al. (1995b). That investigation and the present investigation both conclude that: (1) less than 50% of the entire Harbor was toxic to *Ampelisca* survival or to Microtox™ luminescence, and (2) the Newark Bay

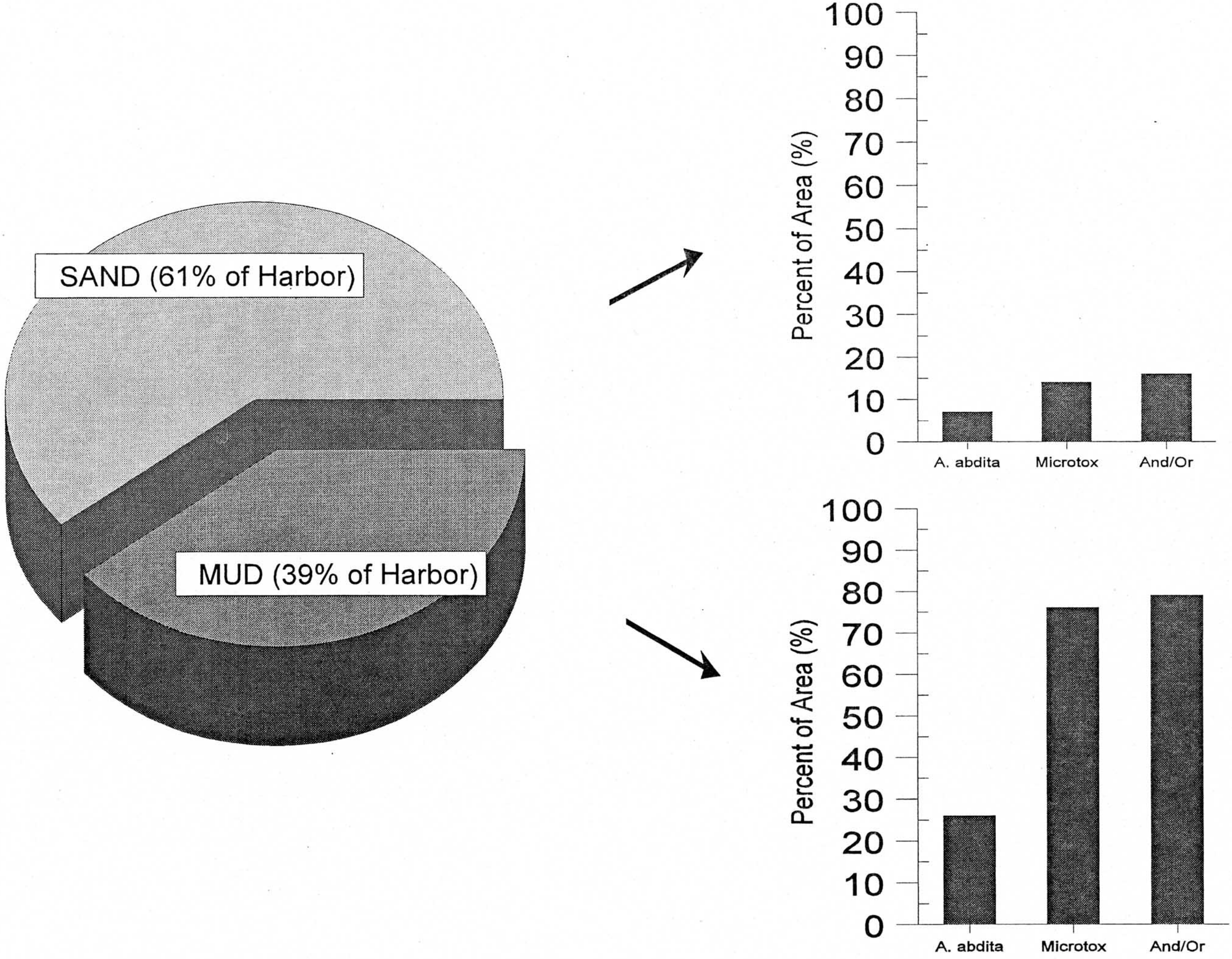


Figure 5-5. Relationship between substrate type (sand ≤40% silt/clay, mud ≥40%) and percent of area in the Harbor with toxicity to *A. abdita*, Microtox™, or *A. abdita* and /or Microtox™.

sub-basin is the most toxic region of the Harbor. Both investigations had similar study areas; the Long et al. (1995b) investigation did not include Jamaica Bay.

However, the Long et al. investigation and this study estimated somewhat different areas of sediment toxicity in the individual sub-basins. Based upon *Ampelisca* survival tests, this study estimated that 46% (23-68%) of the Newark Bay sub-basin was toxic (Figure 5-1), whereas Long et al. (1995b) estimated that 85% of this sub-basin was toxic.

Approximately 40% of the Long et al. study area was toxic to Microtox™ luminescence. The present investigation estimated that less than 20% of the Long et al.'s study area was toxic in the Microtox™ assay, although 38% (28-49%) of the Harbor exhibited Microtox™ toxicity (Figure 5-1).

6.0 BENTHIC MACROINVERTEBRATES

6.1 BACKGROUND

Although sediment chemistry and toxicity assays provide useful insights into sediment quality, they provide only limited understanding of ecological damage (Keeler and McLemore, 1996). For most management purposes, a principal goal is protection and remediation of biological resources. This goal requires a reliable understanding of biological effects of contaminants.

Because of several attributes, bottom-dwelling invertebrates (benthos) provide useful indications of biological response to environmental conditions. Since the ultimate disposition of many contaminants is into sediments where benthic macroinvertebrates live and feed, they are directly exposed to contaminant effects. Because they are relatively sedentary and cannot avoid exposure, benthos can provide an accurate indication of local environmental conditions. Bottom dwelling organisms (benthos) are also relatively long-lived and, as an essential component of the food web, are an important link between primary producers and higher trophic levels (Diaz, 1995). Additionally, benthos significantly affect oxygen, nutrient, and carbon cycles (Blackburn and Henriksen, 1983). They exhibit a broad diversity of sizes, feeding modes and life history characteristics, with a range of responses to environmental stress, making them especially suitable as integrators of contaminant effects (Frithsen & Holland, 1992).

Many measures have been suggested for describing benthic communities. This study assessed several individual structural measures to quantify the status of benthic macroinvertebrate assemblages (Table 2-4). Species diversity, a measure of community structure, is indicative of the species utilizing the available habitat. It is expressed here as number of species (species richness) and as the Shannon-Wiener composite index (Shannon and Weaver, 1949). Evenness (distribution among species) is a basic characteristic of benthic community structure. Biomass also is an integral component of community structure, since it is the basis for energy flow and has been shown to be responsive to pollution stress (Warwick, 1986; Dauer and Connor, 1980; Luckenbach et al., 1990; Pearson and Rosenberg, 1978). Total abundance is also used as an indicator for contaminant effects (Becker et al., 1990) and, along with biomass, is a measure of total biological activity at a site. The use of benthic species that are pollution-tolerant or pollution-sensitive has also been used to determine the ecological health of a location (Grassle and Grassle, 1974 and 1976).

However, more than one measure or indicator, combined into an index of benthic invertebrate structure, can distinguish more effectively than individual measures between normal and abnormal benthic assemblages (Pearson and Rosenberg, 1978; Gray, 1995). A multi-metric benthic index of biotic integrity (B-IBI), similar to the fresh water Index of Biotic Integrity (IBI) (Karr, 1991; Kerans and Karr, 1994) was developed for the NY/NJ Harbor (Appendix C). Five metrics which most effectively distinguished normal sites from all others were selected for the B-IBI (Table 2-4). These metrics were evaluated for four different salinity and grain size habitats

(Table 2-5). The index was calculated by scoring each selected metric as 5, 3, or 1 depending on whether its value at a site approximated, deviated slightly from, or deviated greatly from conditions at the best reference sites. The B-IBI value for each station is calculated as the mean score of the five metrics. A mean score of 5 indicated that the site was approximately equivalent to the best reference sites. A score of 3 or 1 indicated that the site slightly deviated or greatly deviated from conditions at the best reference sites and would be considered to have impacted benthos. These scoring criteria defined normal and abnormal benthic assemblages.

The overall validation efficiency of the B-IBI was 93%. The average difference between replicates was 0.32. Ninety-one percent of the replicates at the same site scored similarly. At most of the sites where the replicates scored differently, the replicates had similar numerical values, but were on either side of the index threshold of 3.

6.2 CHARACTERIZATION OF THE HARBOR

6.2.1 Diversity and Taxonomic Composition

A total of 239 infaunal species were represented in the Harbor (Table 6-1). The mean number of species per sample in the entire Harbor was 19.2 (Table 6-2). Mean species diversity (Shannon-Wiener) in the Harbor was 2.3 (Table 6-2). Shannon-Wiener diversity was similar in all sub-basins, but taxonomic composition varied greatly among sub-basins.

Table 6-1

Species Richness (Total Number of Species)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W. LI. Sound	Bight Apex
Number of Species	239	137	91	166	152	180	231

One difference among sub-basins of the Harbor was the relatively few species present in Newark Bay (Figure 6-1). The mean numbers of species per sample (species richness) was not significantly less ($p>0.1$) in Newark Bay than in Jamaica Bay (the next least species-rich sub-basin). However, Newark Bay species richness was significantly less ($p<.05$) than any of the other three Harbor sub-basins. Nearly half the total number of species was polychaetes, consistently in each sub-basin (Figure 6-1). Molluscs and arthropods were represented by roughly equal numbers of species in each sub-basin. Depending upon the sub-basin, amphipod species constituted from 10 to 18% of all species identified. Three taxa (Amphipoda, Mollusca, and Polychaeta) include about 85% of all taxa identified (Figure 6-1). Area-weighted mean abundances for all benthic macroinvertebrate species identified in the study also were calculated (Appendix G).

Table 6-2

Means of Benthic Variables

(± 90% confidence interval)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W. LI. Sound	Bight Apex
Abundance (# organisms/m^2)	40,000 ±14,000	39,000 ±15,000	11,000 ±4,700	52,000 ±22,000	12,000 ±3,600	19,000 ±7,400	32,000 ±8,200
Species Richness (as # species/sample)	19.2 ±1.7	17.7 ±2.7	14.1 ±2.6	20.6 ±2.6	17.1 ±2.3	20.6 ±2.8	28.9 ±3.6
Pollution-Sensitive Species (%)	13 ±5.6	3.6 ±2.0	0.3 ±0.3	18 ±8.6	6.8 ±5.6	8.1 ±4.7	50 ±9.2
Pollution-Indicative Species (%)	31 ±3.5	46 ±8.4	65 ±7.1	20 ±5.0	49 ±6.3	28 ±8.3	3.0 ±1.0
Biomass (g/m^2)	31 ±11	10 ±5.1	5.4 ±2.0	50 ±31	56 ±35	22 ±7.4	29 ±15
Species Diversity (Shannon-Wiener)	2.3 ±0.17	2.1 ±0.20	2.1 ±0.3	2.4 ±0.26	2.5 ±0.15	2.4 ±0.29	2.3 ±0.23

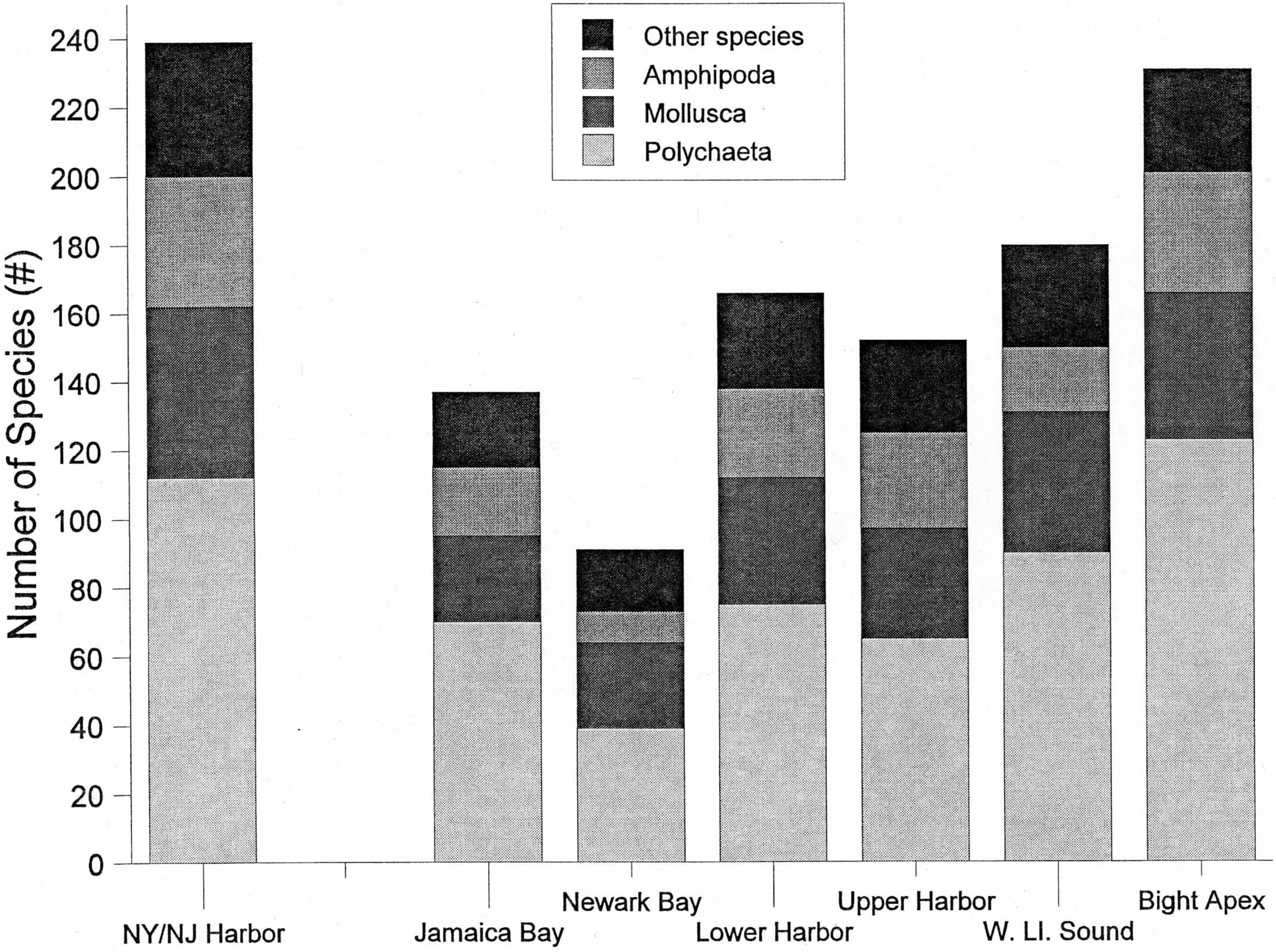


Figure 6-1. Numbers of benthic macrofaunal species, by major taxon.

Lists of pollution-sensitive and pollution-indicative species (Table 6-3) were developed by comparing relative abundance of taxa between reference sites and degraded sites in the EMAP-E Virginian Province data. Pollution-indicative taxa were those for which average abundance, average percent of abundance, and frequency of occurrence were all higher at degraded versus reference sites. Pollution-sensitive taxa were those for which average abundance, average percent of abundance, and frequency of occurrence were all higher at reference than degraded sites, and for which percent of abundance at reference sites averaged at least 0.2%.

Table 6-3

Pollution-Sensitive and Pollution-Indicative Taxa

Pollution-Sensitive Taxa

- Mollusca
  - *Acteocina canaliculata*
  - *Tellina agilis*
  - *Spisula solidissima*
- Arthropoda
  - *Ampelisca agassizi*
  - *Ampelisca verrilli*
  - *Byblis serrata*
  - *Rheopoxynius hudsoni*
- Polychaeta
  - *Ampharete arctica*
  - *Aricidea catherinae*
  - *Caulleriella spp.*
  - *Clymenella torquata*
  - *Glycinde solitaria*
  - *Levinsenia gracilis*
  - *Macroclymene zonalis*
  - *Nephtys picta*
  - *Ninoe nigripes*
  - *Polygordius spp.*
  - *Sabaco elongatus*
  - *Scalibregma inflatum*
  - *Spiophanes bombyx*

Pollution-Indicative Taxa

- Mollusca
  - *Mulinia lateralis*
- Oligochaeta
  - Oligochaetes
- Polychaeta
  - *Capitella spp.*
  - *Polydora cornuta*
  - *Streblospio benedicti*

6.2.2 Abundance and Biomass

The mean abundance and biomass for the Harbor was 40,000 organisms/m^2 and 31 g/m^2, respectively (Table 6-2).

The mean abundance of benthos was significantly lower (p<.01) in both Newark Bay and Upper Harbor than in any other Harbor sub-basin (Table 6-2). Pollution-sensitive species were significantly less abundant (p<.05) in Newark Bay than elsewhere in the Harbor, and significantly more abundant in Lower Harbor (p<.05) than elsewhere in the Harbor. Pollution-indicative species were generally distributed inversely to pollution-sensitive species: i.e., pollution-indicative species were least abundant in Lower Harbor and most abundant in Newark Bay (p<.05). Biomass of the benthos was significantly lower (p<.05) in Jamaica and Newark Bays than in Lower and Upper Harbors.

6.2.3 Benthic Index

A multi-metric benthic index of biotic integrity (B-IBI), similar to the fresh water Index of Biotic Integrity (Karr, 1991; Kerans and Karr, 1994), was developed for the NY/NJ Harbor (Appendix C). Values of the benthic index (B-IBI) at a sampling site can range from one (impacted assemblage) to five (normal assemblage). Benthic structure in about half (53%) of the entire Harbor area exhibited measurable departure from the structure at reference sites (Table 6-4). Most of this area (47%) was in a category indicative of intermediate impact (B-IBI values of 2 to 3).

Measurable benthic impacts (B-IBI<3) were most widespread in Newark Bay, Upper Harbor and Jamaica Bay (Figure 6-2). Estimates of impacted benthic area ranged from 39% for Lower Harbor to 98% for Newark Bay (Table 6-4). The distribution of individual stations with impacted benthos (Figure 6-3) shows the most highly impacted sites were located in the Newark Bay sub-basin and in the back bay portion of Jamaica Bay. Newark Bay had only one station of 28 that was comparable to reference conditions.

Table 6-4

Percent of Area within B-IBI categories

(90% confidence intervals are in parentheses)

	Harbor	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W. LI Sound	Bight Apex
1 to <2 impacted	6 (3-9)	18 (9-31)	18 (0-38)	0 (0-8)	14 (6-27)	7 (2-18)	0 (0-8)
2 to 3 moderately impacted	47 (37-57)	46 (33-60)	80 (60-100)	39 (27-53)	61 (47-73)	46 (33-60)	0 (0-8)
$3-5 unimpacted	47 (37-58)	36 (24-50)	2 (0-6)	61 (47-73)	25 (14-38)	46 (33-60)	100 (92-100)

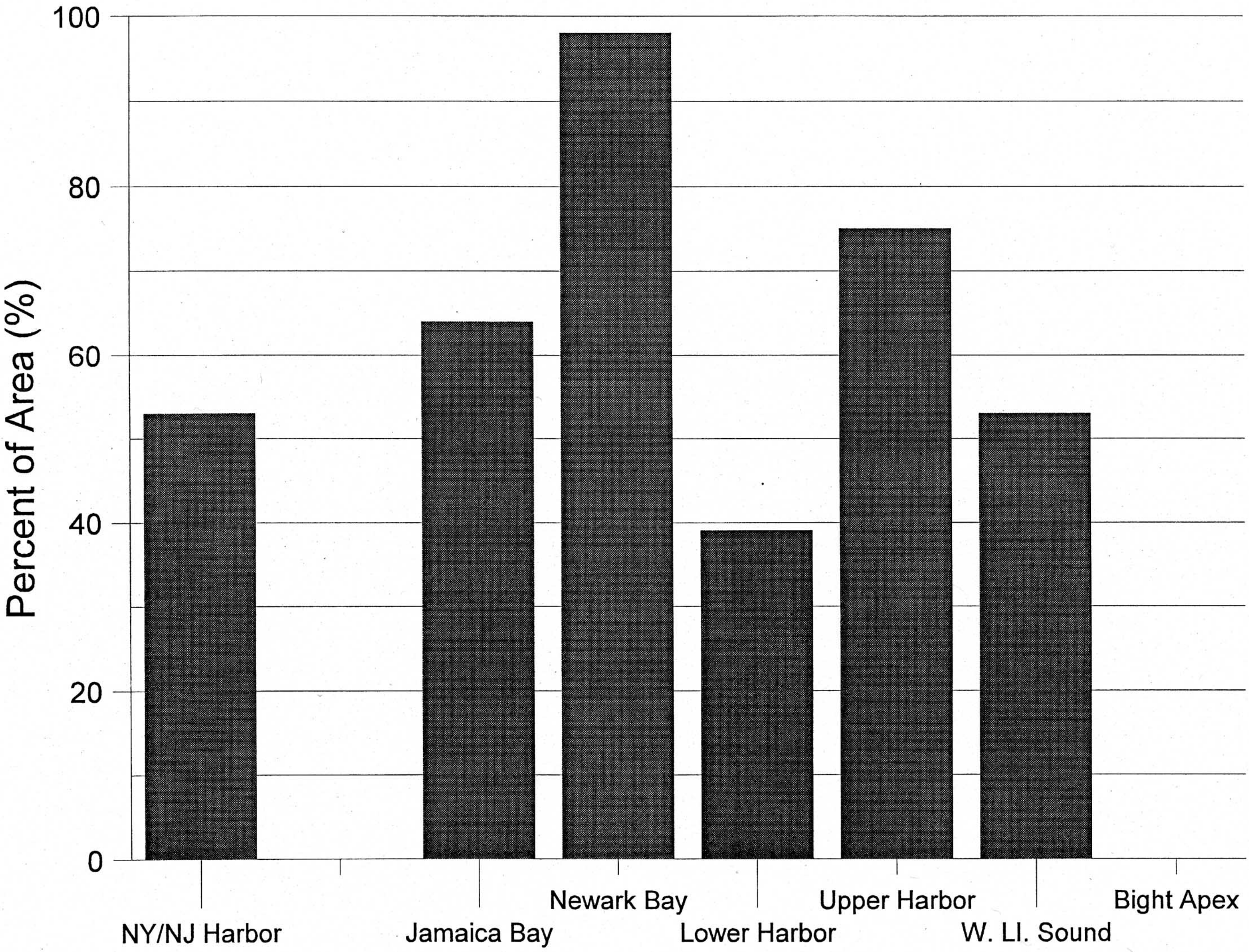


Figure 6-2. Percent of area with impacted benthos (B-IBI<3).

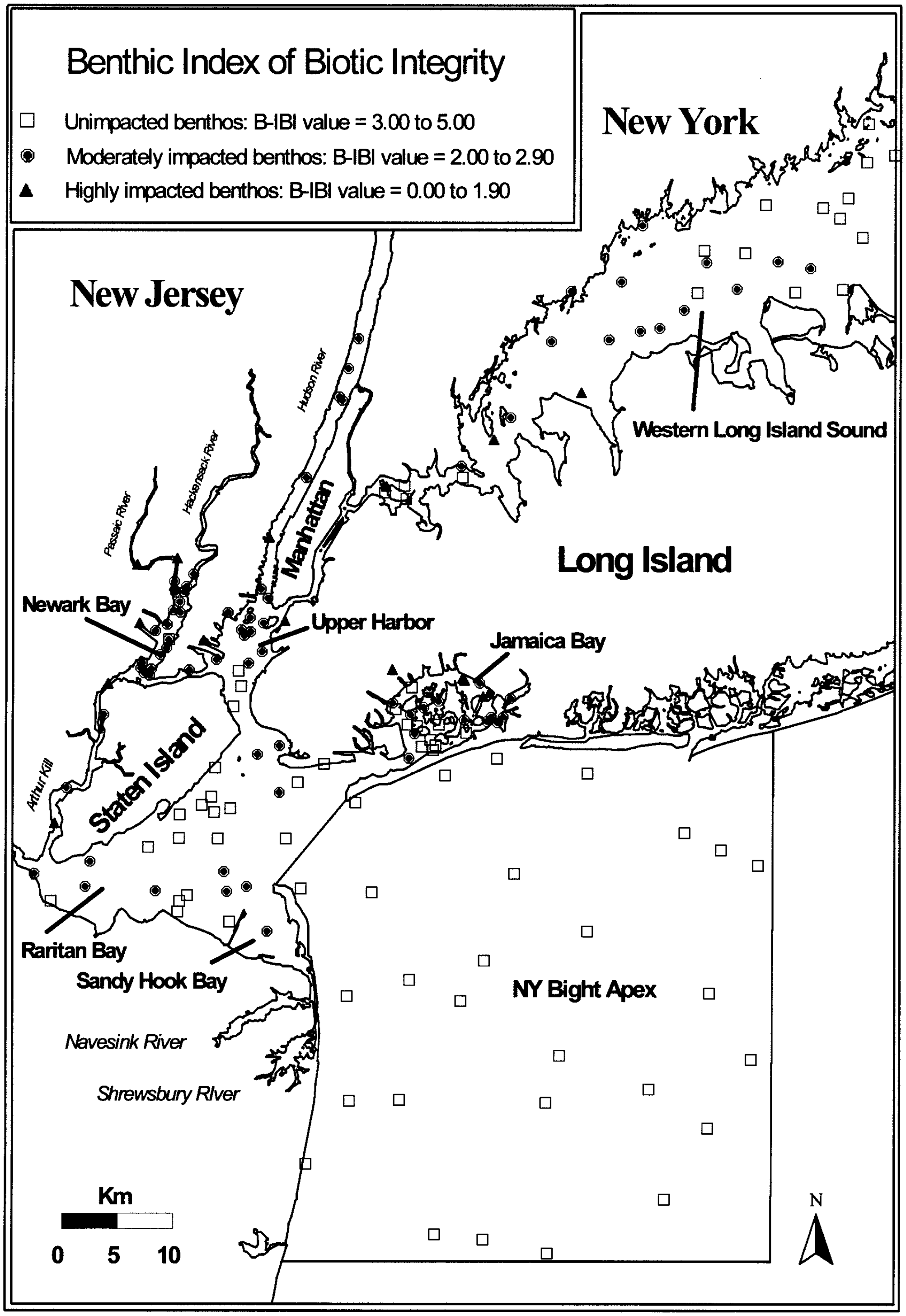


Figure 6-3. Distribution of stations by Benthic Index of Biotic Integrity (B-IBI) values.

6.3 CHARACTERIZATION OF WESTERN LONG ISLAND SOUND AND THE BIGHT APEX

6.3.1 Diversity and Taxonomic Composition

More species were identified in western Long Island Sound and in the Bight Apex, than in any of the Harbor sub-basins. The relative abundance of amphipods, molluscs and polychaetes was similar in coastal waters and the Harbor (Figure 6-1). The mean number of benthic species per station in the Bight Apex (28.9) was greater than in western Long Island Sound or within any sub-basin of the Harbor ($p<.05$).

6.3.2 Abundance and Biomass

The mean abundance of benthos did not differ significantly ($p=.05$) between western Long Island Sound, the Bight Apex and the Harbor. However, pollutant-indicative species in the Apex were significantly less abundant than in any other sub-basin ($p<.05$). Conversely, pollution-sensitive species were more abundant in the Apex than in any other sub-basin ($p<.01$). Benthic biomass in both western Long Island Sound and the Apex was intermediate between sub-basins of the Harbor (Table 6-2).

6.3.3 Benthic Index

The percent of area in each category of the benthic index was similar between the Harbor and western Long Island Sound (Table 6-4). Nearly 50% of the area in each would be considered to have impacted benthos, but most of the 50% was in the intermediate category of the B-IBI. The Bight Apex had approximately 100% of its area in the highest category (similar to reference conditions) of the B-IBI.

6.4 COMPARISON TO PREVIOUS STUDIES

Previous studies of benthic invertebrate structure in the Harbor primarily have used non-random sampling strategies. Although useful for other purposes, these non-probabilistic data prevent reliable generalizations beyond the specific locations sampled. However, some broad comparisons have been made to the current investigation.

Comparisons between this investigation and other studies regarding benthic structure must consider both natural and sampling variability, and differences in techniques. Uncertainty due to natural and sampling variability tends to decrease as less specific structural features are compared. For example, comparisons of abundance for a species, e.g., the amphipod *Ampelisca abdita*, are less certain (and less ecologically significant) than comparative abundance of all amphipods, or of several amphipods which function similarly. Unfortunately, differences in techniques cause the greatest problems in comparisons among studies. They preclude most quantitative comparisons of even reliably estimable parameters. For instance, differences in methods among studies within the Harbor would make most biomass comparisons less than useful, although regional comparisons can be made.

This study's estimates of mean macrofaunal benthic abundance (# organisms/m^2) were also not quantitatively comparable with other studies in the Harbor. This investigation's estimate of abundance for Lower Harbor was substantially higher (52,000/m^2) than that (660/m^2) of another study in Lower Harbor (Steimle and Caracciolo-Ward 1989). Two probable reasons for this disparity are: (1) entirely summer sampling by this investigation versus primarily winter collections, when benthic densities are minimal, by Steimle and Caracciolo-Ward, and (2) use of 0.5 mm mesh benthic sieves in the present investigation versus 1.0 mm mesh sieves used by Steimle and Caracciolo-Ward. Additional methodological contributions to this disparity are possible. Similar sieve-size differences also probably contributed to this study's higher estimates of mean abundance in Newark Bay (11,000/m^2) versus 2,300/m^2 in August samples from the southern portion of Newark Bay (Cerrato 1986). The same sieve-size difference, plus differences in grab size, precluded useful comparisons of species richness (species per benthic grab) between Cerrato (1986) and the present investigation.

Within the Bight Apex, mean macrofaunal abundance as estimated by this investigation (32,000/m^2 ±8,200) was similar to an October 1994 estimate of 26,000/m^2 within a 79 km^2 rectangle surrounding the existing dredged material dumpsite (Hunt, 1996).

Several authors have postulated that benthic structural quality in the Harbor or parts of it has degraded, or improved, since the late 1950s. After correcting and enlarging the available benthic macrofaunal data set from 1957-60 and 1973-74 surveys, Steimle and Caracciolo-Ward (1989) questioned the significance of all benthic structural differences presumably indicative of negative trends in Lower Bay. An even more extensive benthic survey of Lower Harbor in 1986-87 (Cerrato, Bokuniewicz and Wiggins, 1988) did not indicate substantial changes in benthic structure from the 1957-60 or 1973-74 surveys interpreted by Steimle and Caracciolo-Ward (1989). Indeed, the mean, 1986-87 structural parameters estimated by Cerrato et al. reflected normal conditions or deviated only slightly from them, as defined by this study's benthic index. These observations were consistent with apparent improvements in summer minima of dissolved oxygen concentrations. Although dissolved oxygen monitoring is limited to New York waters of Lower Bay, improvements in summer bottom oxygen concentrations of Lower Harbor became evident (depending upon the site) from 1945 to the late 1970s. Once improvements became evident, they continued improving to 1995 (T.M. Brosnan, 1995, personal communication).

A number of benthic faunal surveys were conducted since 1972 in the Newark Bay area. These data sets included less than 30 stations per survey and were confounded by differences in sampling and analysis techniques, sampling locations and seasons, interannual variability, etc. Hypothesized hypoxic impacts on the Newark Bay benthos, particularly in deeper areas, do not appear to have been evaluated during periods of extreme hypoxia (Cerrato, 1986). Despite the limitations of these data, analyses indicated that total abundance and species diversity were unusually low, at least until the May and August 1985 sampling of Cerrato (1986). Cerrato (1986) concluded that data from two seasonal cruises may or may not indicate real, temporary or long-term, differences from prior benthic structure which is itself inferred from limited, purposeful sampling.

Cautious but interesting observations on the benthic infauna of Jamaica Bay were based upon sampling during 1981 and 1982 (Franz and Harris 1988). The authors emphasized within-Bay associations. The only strong, evident pollutant influence on benthic structure were total organic carbon content of sediments. *A. abdita* was also sampled at three sites in Jamaica Bay during spring and summer of 1988-89 (Franz and Tanacredi, 1992). This work documents the existence of two productive cohorts in the Bay. The authors indicated that productivity of *A. abdita*, at least at the sites sampled, was comparable to the total macrobenthic production for several North Atlantic estuaries. These authors also suggested that the large amounts of particulate organic carbon in Bay sediments stimulated this high *Ampelisca* productivity.

It seems probable that the principal departures from historically "normal" benthic structure in the Harbor area had already taken place before the late 1950s, perhaps much earlier.

7.0 ASSOCIATIONS

7.1 BACKGROUND

Along with areal extent of contamination, an additional goal of this investigation was to determine if the contaminants in the NY/NJ Harbor and Bight Apex system were associated with biological effects. This was done most effectively by an integrated assessment. This type of assessment provides more information concerning the ecological significance of contamination than any of the measures can supply individually. An integrated assessment can include two or more of the following components: sediment toxicity tests, sediment chemical analyses, tissue chemical analyses, pathological studies, and community structure analyses (Chapman et al., 1995). This investigation used three of the five components: sediment toxicity tests, sediment chemical analyses and community structure analyses. This type of integrated assessment is commonly known as the Sediment Quality Triad approach (Chapman, 1990).

The Sediment Quality Triad has had multiple estuarine and marine applications (Long and Chapman, 1985; Chapman et al., 1987; Chapman et al., 1991). It offers several advantages that are not realized when using a one or two component approach. In a complex sediment mixture, such as is found in the NY/NJ Harbor, the triad approach incorporates interactions among contaminants (such as additivity, antagonism and synergism) and the effects of any unidentified chemicals. It is more comprehensive than individual measures, but does not assess non-sedimentary ecosystem components such as fish and mammals or human health. Associations are assessed here using a weight-of-evidence approach.

7.2 ASSOCIATION BETWEEN CHEMISTRY AND BENTHIC CONDITION

Contamination by chemical constituents appears to be a prominent factor affecting the health of benthic macroinvertebrate assemblages in the NY-NJ Harbor. The percent of area with impacted benthos was closely related to the level of contaminants in sediments of the Harbor. In the 53% of the Harbor that had abnormal benthic assemblages, 79% of this area exceeded an ERM for at least one contaminant (Figure 7-1). Only 16% of the area with normal benthic assemblages exceeded an ERM for any toxicant.

Three individual chemicals or classes exceeded their ERM values in more than 50% of the impacted benthic area (Table 7-1). Because these toxicants (mercury, chlordane and total PCBs) exceeded their ERM values at relatively few sites without evidence of benthic degradation (16% of the Harbor area), they were strongly associated with impacted benthos. No other chemicals measured were half as widespread in association with impacted benthos at concentrations likely to cause biotic effects.

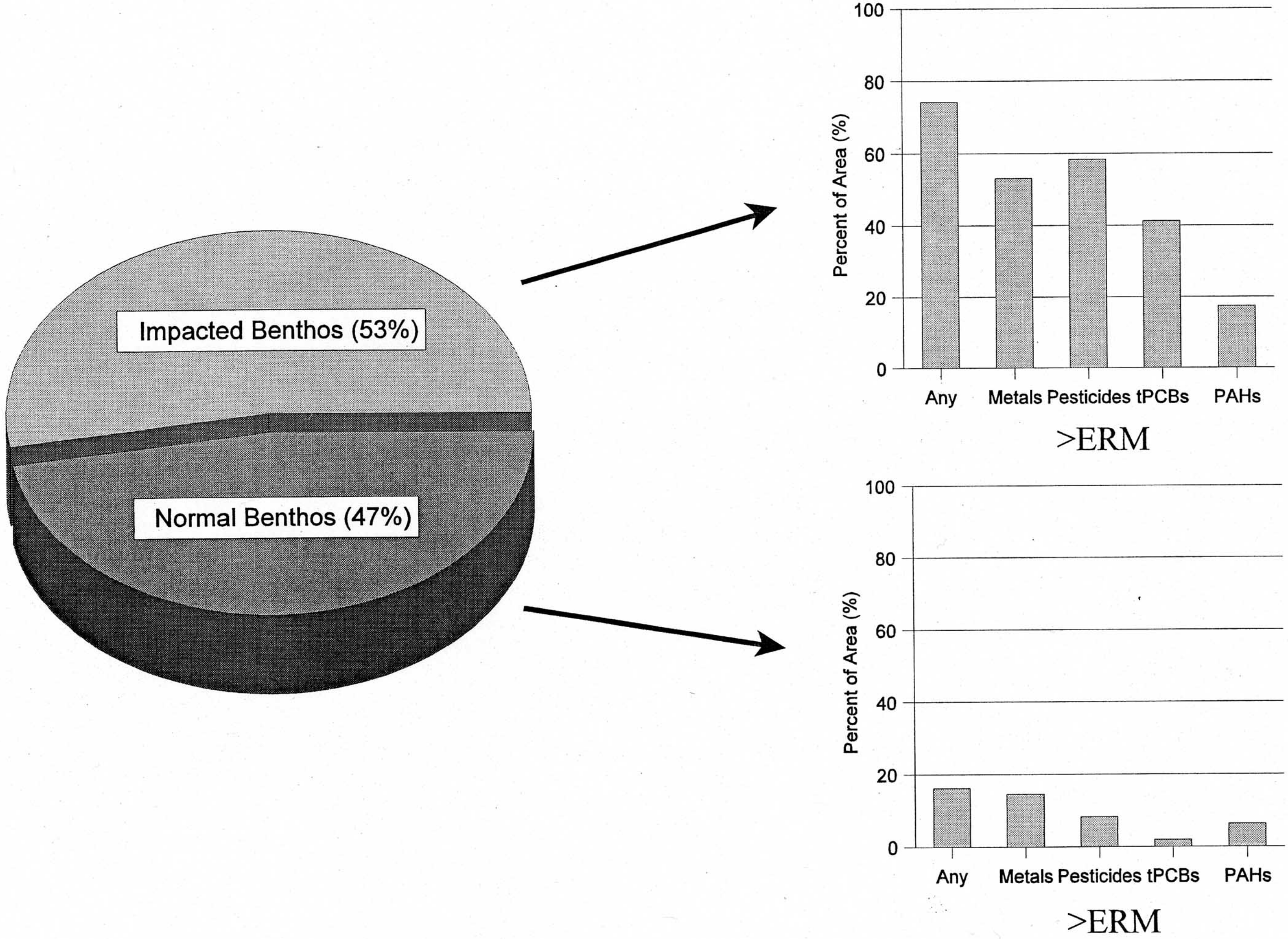


Figure 7-1. Association between benthos and percent of area with one or more contaminants greater than ERM (Long & Morgan, 1991; Long et al., 1995a)

Table 7-1

Individual Chemicals Associated with Impacted Benthos

	% of *Impacted Area* at which a Contaminant > ERM	**% of *Non-Impacted Area* at which a Contaminant > ERM**	**% of *Impacted Area* at which a Contaminant > ERL**	**% of *Non-Impacted Area* at which a Contaminant > ERL**
Metals				
Mercury	51.1	14.6	92.8	55.1
Silver	23.6	1.6	78.8	14.5
Nickel	8.3	0	82.0	16.2
Lead	6.3	1.6	82.0	22.4
Zinc	4.9	0	67.5	17.9
Copper	4.5	1.6	78.5	17.6
Antimony	1.5	0	33.4	6.7
Arsenic	1.5	0	69.0	12.2
Cadmium	0.2	0	41.5	1.9
Chromium	0	0	66.2	11.5
Organics				
Total PCBs	69.6	6.7	95.2	26.4
Total Chlordane	55.6	6.7	95.5	25.4
Total DDE	24.0	0.3	83.4	13.8
Total DDD	19.9	3.5	90.5	16.9
High Molec. Wt. PAHs	17.4	6.3	76.9	20.1
Total DDT	13.3	1.9	95.5	31.1
Low Molec. Wt. PAHs	11.0	6.3	81.6	15.1
Benzo(a)anthracene	7.4	6.3	63.9	6.6
Anthracene	6.3	4.7	71.0	12.1
Phenanthrene	6.3	4.7	57.0	8.2
Dibenz(a,h)anthracene	6.0	3.2	55.6	11.4
p,p'-DDE	5.8	0.3	77.1	13.1
Pyrene	5.8	6.3	34.4	6.6
Benzo(a)pyrene	5.6	6.3	41.6	6.6
Total parent DDT	5.2	0.3	51.1	38.3
Chrysene	4.9	3.2	59.4	6.6
Total PAHs	3.5	1.6	69.0	6.6
Fluoranthene	3.5	1.6	40.0	6.6
Acenaphthylene	2.9	3.2	79.9	14.4
Fluorene	2.1	1.6	86.9	18.3
Acenaphthene	2.1	0	81.2	13.7
2-Methylnaphthalene	2.1	0	48.9	6.6
Naphthalene	1.4	0	31.0	6.6
Endrin	0	0	100.0	99.3
Dieldrin	0	0	99.4	85.6

7.3 ASSOCIATION BETWEEN CHEMISTRY AND TOXICITY

Similar to the association between chemistry and benthic community structure, contaminants also were associated with sediment toxicity. Within the 15% of the Harbor where *A. abdita* toxicity was observed, 92% exceeded at least one ERM (Figure 7-2). Conversely, an ERM was exceeded in 42% of the non-toxic area. Metals were most often associated with toxicity, and total PAHs the least often associated.

The same three individual contaminants or classes (mercury, total chlordane, and total PCBs) that were most strongly associated with benthic impacts also appeared to be associated with toxicity (Table 7-2). However, all three also appeared to be, although less frequently, found in non-toxic areas at concentrations above ERMs.

7.4 ASSOCIATION AMONG CHEMISTRY, TOXICITY AND BENTHIC COMMUNITY STRUCTURE

Incorporating all three components of the Sediment Quality Triad strengthened the association between contaminants and biological effects, and demonstrated that a high degree of consistency existed among the components. This association was examined by partitioning the Harbor areas with impacted benthic structure into percentages of these areas with and without one or more toxicant concentrations exceeding ERM values, and percentages with or without evidence of sediment toxicity based upon *A. abdita* or Microtox™ assays (Figure 7-3). Most of the area with impacted benthic structure also had evidence of both sediment toxicity and toxicant concentrations likely to cause biological effects (66%). Approximately 86% of the area with impacted benthic structure had evidence of sediment toxicity or toxicants likely to impact benthic structure. Conversely, only 18% of those areas with normal benthos exhibited evidence of sediment toxicity or any sediment toxicant exceeding its ERM value. Consequently, it is apparent that most departures from normal benthic structure were associated with sediment toxicity or unusually high toxicant concentrations in sediments.

At only 14 of the 168 Harbor stations were abnormal benthic assemblages observed without evidence of: (1) potential biological effects (at least one chemical concentration greater than its ERM value), or (2) actual sediment toxicity (Microtox™ or *A. abdita*). Environmental stresses other than, or coincident with, toxicants could have been responsible for these abnormal benthic index values. To assess the possibility of low dissolved oxygen induced benthic impacts, existing dissolved oxygen data from the New York City Department of Environmental Protection (Brosnan and O’Shea, 1994; 1995) were evaluated. Severe hypoxic stress (< 2 mg/l D.O.) was probable at only three of the 14 stations. Several of the remaining 11 stations were exposed to exceptionally great scouring by strong currents, a stressor known to impact benthic structure.

The estimated prevalence of high toxicant concentrations (i.e., sediments with one or more toxicant concentrations exceeding ERM values) was consistent with the estimates of areas with

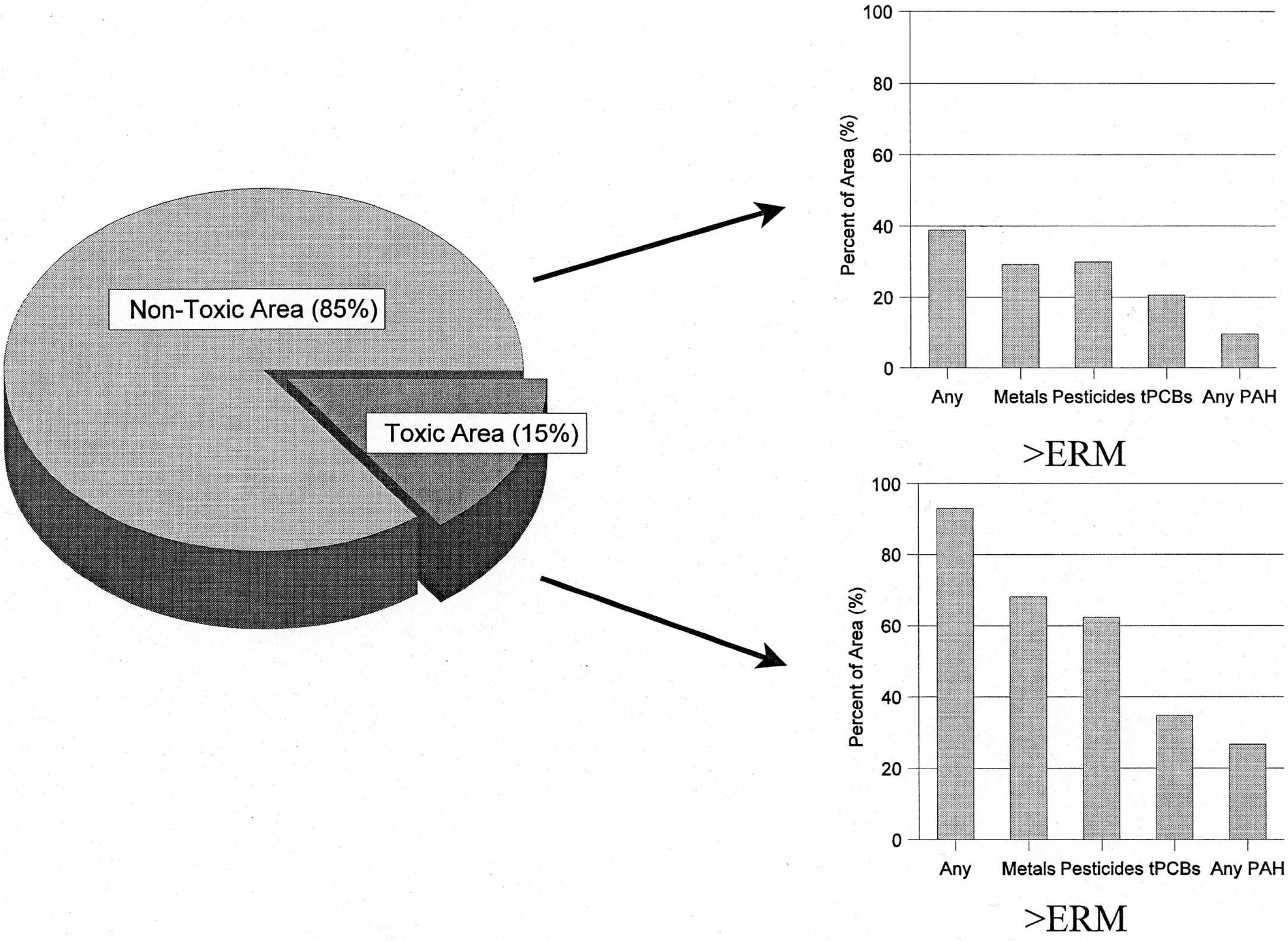
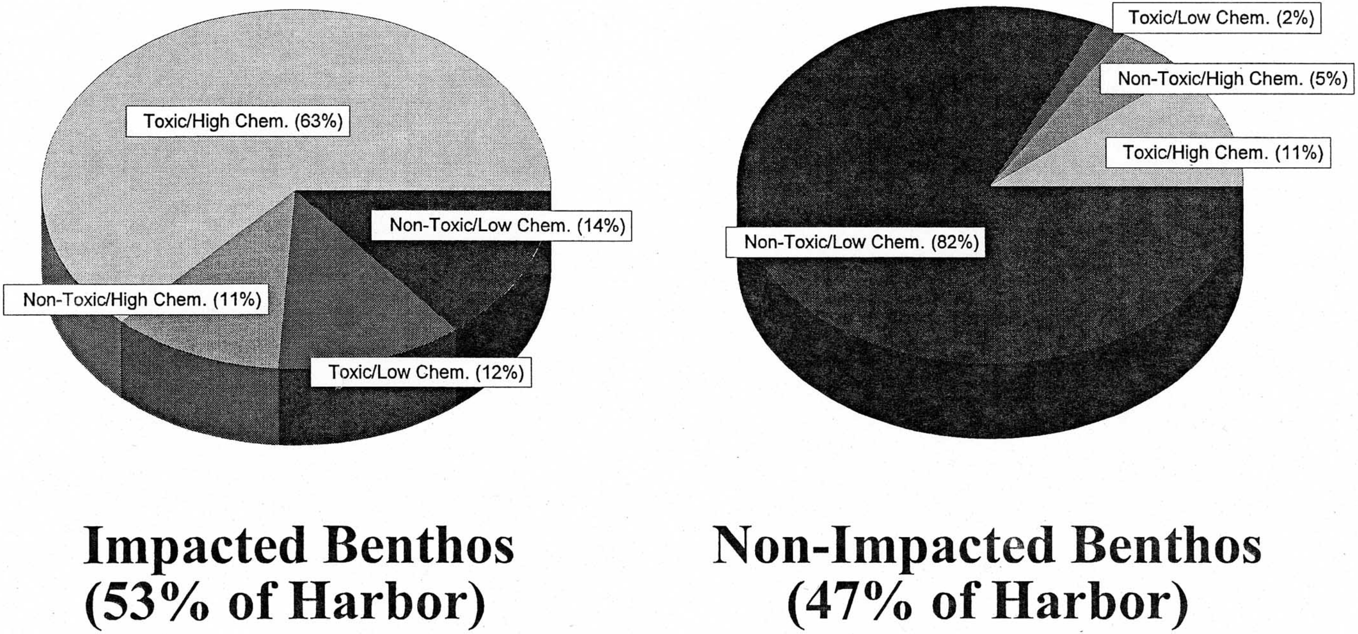


Figure 7-2. Association between *Ampelisca abdita* toxicity and one or more contaminants greater than ERM (Long & Morgan, 1991; Long et al., 1995a).

Table 7-2

Individual Chemicals Associated with Sediment Toxicity *(Ampelisca abdita)*

	% of *Toxic Area* at which a Contaminant > ERM	**% of *Non-Toxic Area* at which a Contaminant > ERM**	**% of *Toxic Area* at which a Contaminant > ERL**	**% of *Non-Toxic Area* at which a Contaminant > ERL**
Metals				
Mercury	68.8	28.0	100.0	70.7
Silver	21.2	11.8	94.6	40.5
Lead	17.3	1.8	95.4	46.7
Zinc	16.8	0.1	90.8	36.0
Copper	16.1	0.9	95.4	41.9
Nickel	15.2	2.5	75.4	46.7
Antimony	5.3	0	33.2	18.6
Arsenic	5.3	0	70.4	37.3
Cadmium	0.7	0	52.2	17.8
Chromium	0	0	73.1	34.7
Organics				
Total Chlordane	57.3	28.2	100.0	56.0
Total PCBs	49.5	37.3	100.0	65.5
High Molec. Wt. PAHs	26.7	9.7	82.3	44.5
Total DDD	24.7	10.0	95.4	49.0
Anthracene	17.5	3.5	82.3	36.4
Total DDE	16.9	12.1	95.4	42.8
Total DDT	14.6	6.8	100.0	59.1
Low Molec. Wt. PAHs	12.9	8.0	84.6	44.3
Benzo(a)pyrene	12.9	4.7	37.3	23.0
p,p'-DDE	12.8	1.6	90.8	39.3
Benzo(a)anthracene	12.4	6.0	57.8	33.2
Pyrene	12.4	4.9	29.8	19.8
Phenanthrene	12.4	4.4	56.0	30.2
Total parent DDT	12.0	1.3	58.0	42.8
Dibenz(a,h)anthracene	11.1	3.5	39.4	34.0
Chrysene	7.4	3.5	57.8	30.5
Total PAHs	7.4	1.8	60.1	36.0
Fluoranthene	7.4	1.7	42.7	21.1
Acenaphthylene	5.1	2.7	77.7	44.1
Fluorene	2.3	1.8	84.6	49.3
Acenaphthene	2.3	0.9	82.3	43.7
2-Methylnaphthalene	2.3	0.9	57.8	24.0
Naphthalene	0	0.9	40.7	15.8
Endrin	0	0	100.0	99.6
Dieldrin	0	0	100.0	91.6



Impacted Benthos (53% of Harbor)

Non-Impacted Benthos (47% of Harbor)

Figure 7-3. Association between sediment toxicity (*A. Abdita* or Microtox™), benthos (using the B-IBI) and sediment chemistry as one or more contaminants greater than ERM (Long & Morgan, 1991; Long et al., 1995a).

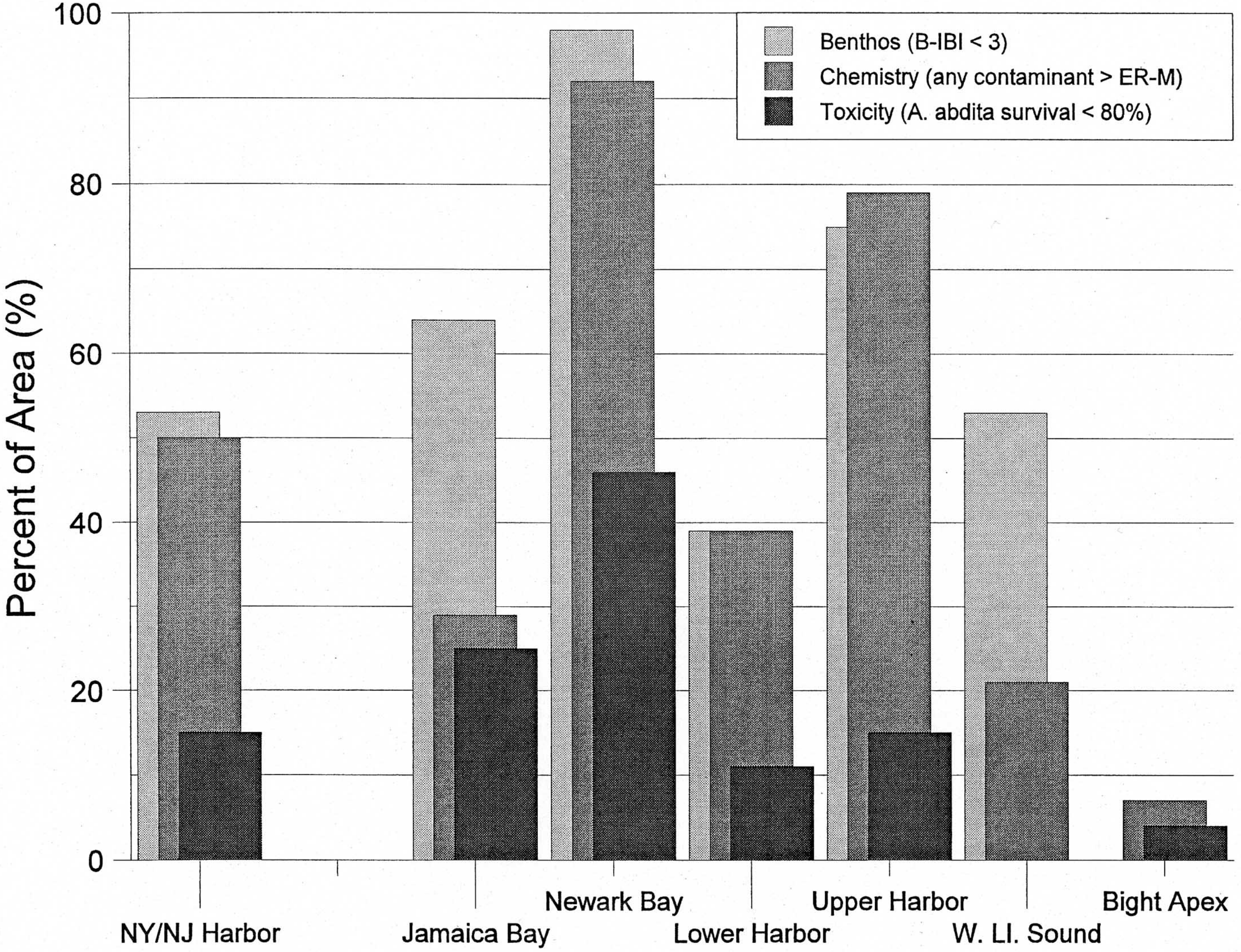


Figure 7-4. Percent of area considered degraded because of impacted benthos, sediment toxicity and sediment chemical contamination.

impacted benthos. Both indicators estimated that about 50% of the Harbor sediments were affected on average, and provided similar results even within sub-basins (Figure 7-4). However, sediments from a relatively small area of the Harbor (15%) reduced laboratory survival of *A. abdita*. The *A. abdita* test detected sediment toxicity in only about 25% of the Harbor area with ecologically significant chemical contamination and/or measured degradation of benthic assemblages. This indicated that the *A. abdita* acute sediment toxicity test was a less sensitive indicator of sediment quality than the B-IBI or ERM sediment chemistry concentrations. This difference was significant for the Harbor as a whole ($P<0.01$) and was consistent within each Harbor sub-basin and western Long Island Sound ($P<0.05$). The only exception was the lack of a significant difference between the estimated areas with ecologically significant chemical contamination and *A. abdita* toxicity within Jamaica Bay (Figure 7-4). These findings indicated that benthic structure was measurably impacted, and was predictable by chemical contamination, before acute toxicity of *A. abdita* became evident.

8.0 DISCUSSION

Previous studies have documented that sediment chemical contamination is intensive within selected portions of the NY/NJ Harbor (NOAA, 1995). This study expands on these findings by documenting the pervasiveness of the contamination. Nearly every sample collected in the Harbor, as part of this investigation, had at least one chemical exceeding an ERL concentration and one-half of the area in the Harbor had at least one chemical exceeding an ERM concentration. Contamination was pervasive across chemical groups. More than one-third of the Harbor had chemical concentrations exceeding ERM concentrations for each of the metals, pesticides and PCBs chemical groups; there were 14 individual chemicals which exceeded their ERL concentration over more than 25% of the Harbor area.

Examining a simple ranking of the sub-basins by areal extent of biologically significant levels of contaminants, toxicity and abnormal benthic communities (Table 8-1) shows that Newark Bay is consistently the most degraded sub-basin in the Harbor and Lower Harbor the least degraded. The Bight Apex and western Long Island Sound appear relatively unaffected.

Table 8-1

Relative Ranking of Sub-basins by % of Area

(1 is most degraded, 5 is least degraded)

	Jamaica Bay	Newark Bay	Lower Harbor	Upper Harbor	W. LI Sound	Bight Apex
Chemistry[1]	4	1	3	2	5	6
Toxicity[2]	2	1	4	3	6	5
Benthos[3]	3	1	5	2	4	6
Mean Ranking	3	1	4	2	5	6

[1] One or more contaminants > ERM.
[2] Significant toxicity to A. abdita (% survival </= 80% and statistically different from controls).
[3] Benthic index value <3.

Biological effects were found to be associated with chemical contamination. At 66% of the area where impacted benthic communities were observed, there also was a toxicological response and/or at least one chemical exceeding its ERM concentration. In contrast, only 14% of the area without a toxicological response and without a chemical exceeding an ERM concentration had impacted benthic communities.

The conclusions regarding the strong relationship between high chemical concentrations and biological response are based largely on an index that integrates multiple measures of the benthic community into a single value. Some authors have raised concern about analyses based solely on integrative indices (Elliott, 1994). The index approach was chosen for this investigation because the integration provides a threshold for identifying degraded assemblages and allows analysis based on areal extent of a problem. This investigation's conclusion regarding coupling of chemistry and benthic response, though, appears to be independent of the index. There were relationships between average values for several of the individual benthic metrics used in the index and presence of high chemical concentrations (Table 8-2).

Table 8-2

Association between Harbor Means of Benthic Metrics and Number of Chemicals > ERM

(± represent 90% confidence interval)

	1 or more chemicals > ERM*	2 - 5 chemicals > ERM*	6 or more chemicals > ERM*
Abundance (# organisms/m^2)	98,497 ±64,798	76,976 ±27,915	21,477 ±3108
Species Richness (as # species/sample)	24.33 ±4.63	21.24 ±1.97	18.15 ±2.43
Pollution-Indicative Species (%)	18.37 ±8.07	34.08 ±6.58	58.31 ±4.99

* May also have 1 or more contaminants >ERL.

This study's conclusions are also based on interpreting chemical concentrations relative to the thresholds suggested by Long and Morgan (1991) and Long et al. (1995a). Some authors have suggested that the likelihood of contaminant-related biological effects is more appropriately assessed using equilibrium partitioning for organic chemicals (DiToro, 1991; U.S.EPA, 1994) and acid-volatile sulfides for metals (DiToro et al., 1990, 1992), although other authors have questioned these approaches (Iannuzzi et al., 1995). The Long et al. (1995a) values were used for this investigation because they included thresholds for most of the chemicals that were measured, allowing an integrated contaminant response to be provided. The other approaches have been developed for a relatively small number of chemicals, and rely almost entirely on theoretical considerations without field assessment. When the two approaches were compared for chemicals for which thresholds are available from both approaches, the Long et al. approach consistently predicted a greater extent of contamination occurring at biologically relevant concentrations (Table 8-3). The coincidence between chemistry, toxicity and biological response was also greater for the Long et al. approach, suggesting that the partitioning approaches may underestimate the availability of contaminants, but this may be partially a function of the lesser number of chemicals for which thresholds have been developed. Re-examination of which chemical groups are leading to biological and toxicological responses may be advisable as the equilibrium partitioning approach is applied to a larger group of chemicals. The use of cumulative distribution functions, inherent with a probability-based sampling design, provides flexibility in applying different thresholds to the same data set without recollecting samples or recalculating data.

Table 8-3

Percent of Harbor Area Which Exceeded Selected Thresholds

	Percent of Area Exceeding Threshold (%)
ERM (Long & Morgan, 1991; Long et al., 1995a)	50
Proposed SQC (U.S.EPA, 1994)*	3
AVS (DiToro, 1990, 1992)**	12

* Proposed SQC exist for 3 aromatic hydrocarbons and 2 pesticides. Normalized for TOC.
**Applies to 5 divalent metals.

The most prevalent contaminants at levels of biological concern appeared to be mercury, chlordane and total PCBs, which were consistent with findings of previous studies (Long et al., 1995a) and data syntheses (Squibb et al., 1991) for NY/NJ Harbor. However, this study's conclusions were based on associations which do not necessarily result from cause/effect relationships. Correlations with other chemicals, or with mixes of other chemicals, can confound the patterns that were observed. Associations identifying which chemicals are not causing effects are more robust to confounding than are associations that imply cause. Still, mercury, chlordane and total PCBs each were found at concentrations exceeding ERM levels at more than half of the sites in the Harbor where impacted benthic assemblages were observed, but were not observed at most sites which contained healthy assemblages, which is a compelling spatial coincidence.

The chemistry problems in the NY/NJ Harbor present a difficult management challenge as each of the major chemicals of concern appears to originate from a combination of point- and non-point sources. For instance, most of the mercury input to Newark Bay has been estimated to come from point sources on the Hackensack and Passaic Rivers (Olsen et al., 1984), which is consistent with the spatial patterns this investigation observed for mercury in the Harbor. Mercury was mostly concentrated in Newark Bay and the mercury exceedances in places like Raritan Bay followed a spatial pattern suggesting flow from Newark Bay as a source. In contrast, HydroQual (1991) has estimated that 50% of the inputs for total PCBs enters from tributaries and most of the chlordane (Bopp et al., 1982) is non-point in origin. The distribution of total PCBs and chlordane was pervasive throughout the Upper Harbor, Newark Bay and Jamaica Bay, each of which has a distinct watershed.

While this study assessed which chemicals were site- or basin-specific problems within the Harbor complex, it did not address which chemicals had sources that were ubiquitous at scales beyond the boundaries of the Harbor complex. The field methods, laboratory methods and QA protocols used in the Harbor were modeled after those of EPA's Environmental Monitoring and

Assessment Program (EMAP), facilitating comparisons between the NY/NJ Harbor and the remainder of the mid-Atlantic coastal estuaries sampled by EMAP (Strobel et al., 1995), and allowing assessment of which chemicals are NY/NJ specific problems and which are issues on a wider geographic scale. Conducting the comparison with EMAP, this study found that the Harbor had higher average concentrations for 58 of the 59 chemicals measured (Table 4-4). For several chemicals, specifically mercury and total PCBs, the Harbor also had a large portion (69% and 100%, respectively) of the areal extent of ERM exceedances in the Virginian Province, even though the Harbor constitutes only 4% of the area in the Province (Figure 4-14). These findings suggest that for these chemicals the spatial scale of management action should be focused on the Harbor.

This study was focused on evaluating the relationship between contaminants and benthic community condition, but findings from other studies suggested that there also may be contaminant related food chain effects in the Harbor. Benthic macroinvertebrates are important food for fish and birds. Some of the same contaminants this study found prevalent in the sediments were also present in fish, shellfish and crustacea (Belton et al., 1985; NYSDEC, 1988; Hauge et al., 1990; Zongwei et al., 1994) and bird feathers (Burger and Gochfeld, 1993) within and near the Harbor and Bight Apex.

While this study's data were sufficient to indicate a contaminant problem in the Harbor, this study did not distinguish historical from current inputs. Crawford et al. (1995) has suggested that inputs to systems like Newark Bay have decreased by 90% over the last decade. This investigation sampled only the top two centimeters and it has been suggested that average deposition rates in the Harbor are as high as 0.3 cm per year (Olsen et al., 1984). Based on this estimate, most of the material this investigation sampled would have been deposited in the last seven years, suggesting that inputs to the system are still a problem even after substantial reductions. Better estimation of deposition rates and sediment transport within each sub-basin is a necessary precursor for determining the most appropriate management strategy to address the contaminant problem, whether it is pollution prevention, remediation, no action or a combination of strategies.

This study assessed the quality of surficial sediments in 1993-94, but these qualities will persist in potential dredged materials of the future. This investigation was principally concerned with fine-grained particles (silts and clays, <63µ in diameter) because most toxicants are strongly attached to them -- so strongly that the fines carry the toxicants with them when transported (Olsen et al. 1982). Although the dynamics of fine particles and their associated toxicants are complex functions of several processes, the locations of maximal net sedimentation (the fastest deposition) are predictable. Deposition is fastest wherever bottom currents are slow and little wave energy reaches the bottom: in coves and channels, around piers, and near the ends of salinity intrusions (Olsen et al. 1978, 1984; Abood et al. 1992).

Human intervention to improve sediment quality in the Harbor presumes at least broad understanding of where sediments and contaminants come from, and their movements within the

Harbor. Despite large uncertainties, usefully precise estimates of fine sediment fluxes exist. On average, 1.5 ±0.5 x 10^6 metric tons of fine particles accumulate in the Harbor annually. About 3/4 of this material is riverborne, 1/4 comes from marine sources, and very small contributions are from sewage solids, water column productivity and shore erosion (Olsen et al. 1984). These estimates are comparable to those of Ellsworth (1986) and Bokuniewicz and Ellsworth (1986), but a resulting fine-sediment budget did not balance. Bokuniewicz and Ellsworth suspect that existing measurements underestimate oceanic fluxes into the Harbor, and up rivers and into Jamaica Bay (by factors of about 2.5 to 4). Most of the newly introduced fines tend to mix with fines already in the Harbor, and accumulate in dredged areas of Upper Bay, Newark Bay, and Raritan Bay, at 10 to 100 times the accumulation rates elsewhere (Olsen et al., 1984; Abood et al., 1992). Dredging is the principal mechanism for removing fine-grained sediment from the Harbor. Lateral fine sediment mixing appears to be rapid throughout the Harbor (Bokuniewicz and Ellsworth, 1986). These extensive fluxes rapidly scavenge toxicants from the water column and tend to homogenize their concentrations in fine sediments. Sediments of shallow, wide areas of the Harbor are generally in equilibrium with sea level rise, and have net sedimentation rates of only 1-3mm/yr. These areas have little or no net accumulation of fine particles. However, these fines and their associated contaminants are continually resuspended, facilitating their lateral transport throughout the Harbor (Olsen et al., 1984).

These dynamics explain why today's fine surficial sediments scavenge toxicants from Harbor waters and store them in deep and protected areas such as channels and coves. As a consequence most chemical properties of the surficial sediments measured by this investigation in 1993-94 will persist in channels until these sediments are dredged. Similarly, the qualities of future dredged materials will reflect toxicant concentrations of fine sediments accumulated in previous years. Consequently, significant improvements in dredged material quality will require significant reductions in total toxicant loadings on the Harbor, wherever they come from.

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