

Report to the
MASSACHUSETTS BAYS PROGRAM

**EXAMINING LINKAGES BETWEEN CONTAMINANT INPUTS
AND THEIR IMPACTS ON LIVING MARINE RESOURCES
OF THE MASSACHUSETTS BAY ECOSYSTEM THROUGH
APPLICATION OF THE SEDIMENT QUALITY TRIAD METHOD**

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MASSACHUSETTS BAYS PROGRAM

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FOREWORD

The roots of the Massachusetts Bays Program extend back to 1982, when the City of Quincy filed suit against the Metropolitan District Commission and the Boston Water and Sewer Commission over the chronic pollution of Boston Harbor, Quincy Bay, and adjacent waters. Outdated and poorly maintained sewage treatment plants on Deer Island and Nut Island were being overwhelmed daily by sewage from the forty-three communities in the Metropolitan Boston area. Untreated and partially treated sewage were spilling into Boston Harbor.

Litigation over the pollution of Boston Harbor culminated in 1985 when the United States Attorney filed suit on behalf of the Environmental Protection Agency against the Commonwealth of Massachusetts for violations of the Federal Clean Water Act. The settlement of this suit resulted, in 1988, in the creation of the Massachusetts Water Resources Authority, the agency currently overseeing a multi-billion dollar project to repair and upgrade Metropolitan Boston's sewage treatment system. In addition, the settlement resulted in the establishment of the Massachusetts Environmental Trust - an environmental philanthropy dedicated to improving the Commonwealth's coastal and marine resources. \$2 million in settlement proceeds were administered by the Trust to support projects dedicated to the restoration and protection of Boston Harbor and Massachusetts Bay.

The Trust provided \$1.6 million to establish the Massachusetts Bays Program, a collaborative effort of public officials, civic organizations, business leaders, and environmental groups to work towards improved coastal water quality. The funding was used to support both a program of public education and a scientific research program focussing on the sources, fate, transport and effects of contaminants in the Massachusetts and Cape Cod Bays ecosystem. To maximize the efficiency of limited research funding, the sponsored research program was developed in coordination with research funded by the MWRA, the United States Geological Survey, and the Massachusetts Institute of Technology Sea Grant Program.

In April, 1990, following a formal process of nomination, the Massachusetts Bays Program became part of the National Estuary Program. The additional funding provided as part of this joint program of the Environmental Protection Agency and the Commonwealth of Massachusetts has been used to continue a coordinated program of research in the Massachusetts Bays ecosystem, as well as supporting the development of a Comprehensive Conservation and management plan for the coastal and marine resources of Massachusetts and Cape Cod Bays. The study described in this report explores the relationships among chemical contamination, toxicity and the benthic communities of sites on the coast of Massachusetts and Cape Cod Bays.

The information in this document has been subject to Massachusetts Bays Program peer and administrative review and has been accepted for publication as a Massachusetts Bays Program document. The contents of this document do not necessarily reflect the views and policies of the Management Conference.

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Executive Summary

Study Objectives and Design

The Massachusetts Bays Program (MBP) recently released a draft Comprehensive Conservation and Management Plan (CCMP) that serves as a blueprint for coordinated action to restore and protect the water quality and diverse natural resources of the Massachusetts/Cape Cod Bays ecosystem (MBP 1994). The plan identifies a need to characterize baseline conditions of living resources and their habitats in Massachusetts and Cape Cod Bays and to develop a better understanding of how various anthropogenic activities may adversely affect them. The study was conducted in an effort to develop new information that could be used in addressing this particular need. In addition, the data are intended to help the MBP achieve one of its overall goals of providing a scientific basis for management decisions directed at protecting living resources and habitats of this valuable coastal system.

In this study, the Sediment Quality Triad (SQT) method was used to examine linkages between sediment contaminant concentrations and their potential impacts on living benthic resources of the Massachusetts/Cape Cod Bays ecosystem. This method combines sediment contaminant analyses, sediment toxicity testing, and measures of ambient biological conditions as a means of quantitatively assessing pollution-induced degradation of the benthic environment. This is an integrative approach of linking measured contaminant levels in sediments to their capacity to cause toxic effects (as indicated by laboratory toxicity tests with field sediments) and to actual adverse conditions in populations of resident organisms living in these same sediments. The method was developed originally with data derived from polluted harbors on the west coast of the United States, including Puget Sound (Long and Chapman 1985, Chapman 1986) and San Francisco Bay (Chapman et al. 1987, Long and Morgan 1990). Since then, the method has been used to assess pollutant impacts in other U.S. coastal systems, including the Galveston Bay estuary, Texas (Carr 1993); around an oil platform in the Gulf of Mexico (Chapman et al. 1991); and in the Tampa Bay estuary, Florida (SAIC 1992, USFWS 1992). Currently the method is being applied in the New York/New Jersey Harbor system. Results of these studies have demonstrated the strength of using multiple indicators as a basis for identifying areas where sediment contamination is responsible for ecosystem degradation.

Key objectives of the study were as follows:

1. To determine whether chemical contaminants in sediments at any of the 12 Massachusetts/Cape Cod Bay stations were present at concentrations known to cause adverse effects on marine organisms

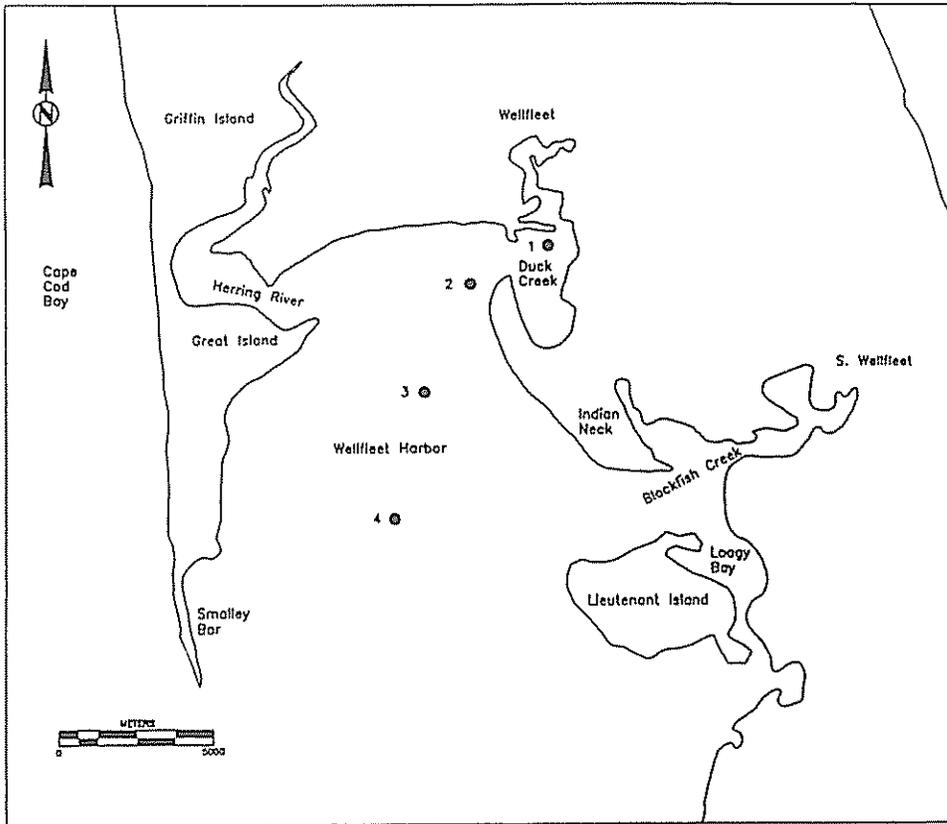
Executive Summary (continued)

2. To determine whether sediments and/or sediment porewaters collected at the 12 Massachusetts/Cape Cod Bays stations were significantly toxic to test populations of marine organisms (amphipod *Ampelisca abdita* and sea urchin *Arbacia punctulata*) based on comparisons of survival and other sublethal biological responses in negative controls
3. To examine patterns in macroinfaunal community structure among the various sites and identify any signs of pollutant-related stress in these assemblages
4. To examine relationships between the chemical, toxicological, and biological data as a means of identifying sites where sediment contamination could have been responsible for observed bioeffects (significant toxicity responses and/or altered benthic community structure)
5. To compare differences in contaminant trends and degree of biological impacts among the three harbor areas

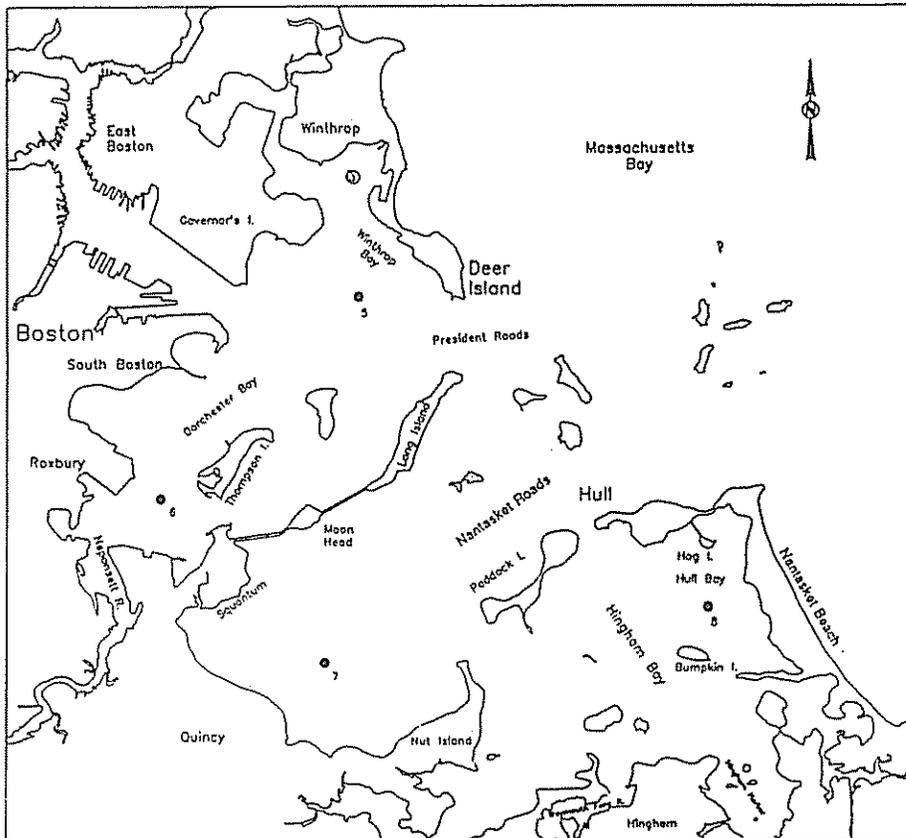
Sediment samples were collected during July 1993 at 12 stations in Massachusetts and Cape Cod Bays along the Massachusetts coastline. These stations consisted of three suspected contaminated sites and a corresponding suspected reference site in each of three harbor systems. In order to provide as broad a baywide coverage as possible with the 12 stations, the three harbor systems were selected from a northern area (Salem/Beverly Harbors), central area (Boston Harbor), and Cape Cod Bay (Wellfleet Harbor). Within each harbor, four stations were established and sampled in order to (1) provide a basis for examining differences in the measured environmental variables among stations within a particular harbor system and (2) provide a measure of spatial variability within each harbor, so that contaminant trends and associated biological impacts could be compared among the three different harbor systems. Sampling zones within a harbor were selected to include both suspected contaminated and reference areas, (Stations 4, 8, and 12), based either on background information or on consideration of the local geography and proximity to anthropogenic influences. Also, wherever possible, background sediment data were used to locate sampling zones in known depositional environments.

Synoptic measurements were made of pollutant concentrations, sediment/porewater toxicity, and benthic community structure as a basis for examining potential linkages between sediment contamination and adverse impacts on living benthic resources of the Massachusetts Bay/Cape Cod Bay nearshore ecosystem.

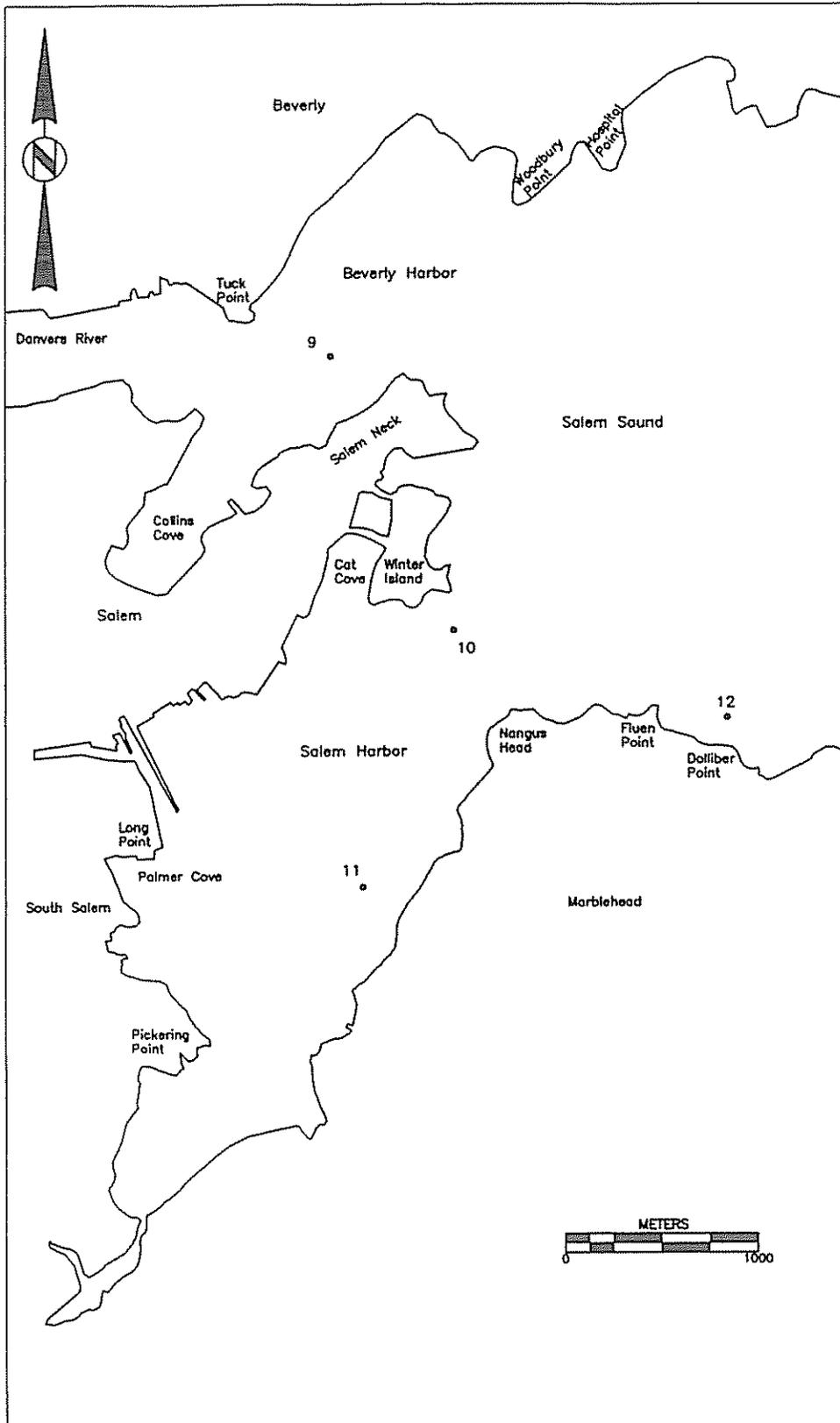
Chemical and physical variables that were measured in sediments consisted of total hydrocarbon content (THC), polynuclear aromatic hydrocarbons (PAHs), chlorinated pesticides, polychlorinated biphenyls (PCBs), metals, grain size, total organic carbon (TOC), and E_p . Dissolved oxygen (DO), temperature, pH, and salinity were measured in samples of near-bottom water. Ammonia, hydrogen sulfide (H_2S), DO,



Sampling stations in Wellfleet Harbor



Sampling stations in Boston Harbor



Sampling stations in Salem and Beverly Harbors

Executive Summary (continued)

pH, and salinity were also measured in porewater samples. Sediment toxicity was measured using two standard methods: a solid-phase acute toxicity test with the marine amphipod *Ampelisca abdita* (ASTM 1990) and an early-life-stage toxicity test with the sea urchin *Arbacia punctulata* (Weber et al. 1988) modified for exposures to sediment porewater samples. The sea urchin test incorporated both a sperm-cell fertilization test and a test for abnormal embryonic development (to the echinopluteus stage). Measures of infaunal community structure and composition (total numbers of species, Shannon-Weaner diversity and evenness, combined species abundances, individual species abundances) were based on the macrofaunal (> 0.5 mm) size component and on species-level data.

Results

The study data were analyzed using several complementary approaches. Measured concentrations of key contaminants were compared to the No-Observed-Effect Level (NOEL) and Probable Effects Level (PEL) of MacDonald (1992), and to the Effects Range-Low (ER-L) and Effects Range-Median (ER-M) values of Long and Morgan (1990), as a means of evaluating whether these contaminants were present in sediments at concentrations reported to cause adverse effects on marine organisms. The NOEL and ER-L values are similar in range (usually within a factor of two of one another) and provide estimates of the maximum concentration at which no effect is observed and the lowest concentration at which adverse biological effects on some marine organisms (presumably the most sensitive ones) are observed, respectively. The PEL and ER-M values are also similar in range (usually within a factor of two of one another) and indicate the concentration above which bioeffects are expected to occur on a wider variety of benthic organisms.

Statistical correlations between selected chemical/physical, biological, and toxicological variables, using station means as observations, were tested. The chemical contaminant variables included in the analysis were those variables that exceeded corresponding bioeffect (NOEL or ER-L) values at one or more of the stations. Contaminants without reported bioeffect values were excluded. Most of these contaminants (e.g., berillium, selenium, thallium) were below detectable concentrations at all stations. Conclusions were drawn for correlations between each key biological or toxicological variable (infaunal density, infaunal species richness, amphipod percent survival, sea urchin percent fertilization, and sea urchin percent normal embryological development) and the various chemical/physical variables, including (1) un-normalized chemical/physical variables, (2) sediment chemical variables (both organic compounds and metals) normalized to silt+clay, and (3) organic chemical variables normalized to total organic carbon (TOC) and metals normalized to aluminum. Determination of significance was based on whether the Type I error probability for the null hypothesis of no correlation ($H_0: r=0$) was \leq the Dunn-Sidak adjusted significance level (based on an unadjusted P of 0.05).

Executive Summary (continued)

An additional data comparison method was used as a means of interpreting results of the three Sediment Quality Triad components collectively. A similar approach was introduced by Chapman (1990) and adopted by Carr (1993) in other sediment quality surveys. In the present application, evidence of contaminant-induced degradation at a station is provided by the combination of concentrations exceeding one or more NOEL/ER-L for a chemical, one or more significant toxicity occurrences, and the presence of a stressed benthic community. Low species richness (≤ 50 species from combined replicates at a station) was used in this study as a somewhat subjective indicator of a stressed benthic community; other benthic community parameters were too variable to use for this purpose. A cut-off point of 50 species was based on comparison of numbers of species at reference sites relative to the other sites. All reference sites had more than 50 species.

Based on these criteria, six stations (Stations 1 and 3 in Wellfleet Harbor, Stations 6 and 7 in Boston Harbor, and Stations 10 and 11 in Salem Harbor) showed strong signs of contaminant-induced degradation of the benthic environment, i.e., a combination of sediment contaminant loading above reported bioeffects levels (either the "no observed effect level" [NOEL] of MacDonald 1992, or the "effects range-low" [ER-L] value of Long and Morgan 1990); significant sediment toxicity (one or more significant reductions in amphipod survival, sea urchin gamete fertilization, or sea urchin embryological development); and the presence of a stressed benthic community (indicated by low species richness). At least one contaminant (at Station 3) and up to 23 contaminants (at Station 10) were present at these stations at potentially toxic concentrations above the reported NOEL/ER-L values. In addition, unionized ammonia was present in porewater at three of these sites (Stations 1, 10, and 11) at concentrations that could have caused toxicity and benthic community degradation. Station 1 in the innermost area of Wellfleet Harbor also had a very high concentration of sulfide, which could have contributed to the observed bioeffects.

Elevated concentrations of chemical contaminants (above NOEL or ER-L values) and significant toxicity responses were observed at Station 4 in Wellfleet Harbor, Stations 5 and 8 in Boston Harbor, and Stations 9 and 12 in Salem/Beverly Harbors, though there was no clear indication of a stressed benthic community at these sites. Stations 8 and 12 also had high concentrations of unionized ammonia in porewater, which could have caused or contributed to the observed sediment toxicity.

Station 2 in Wellfleet Harbor contained sediments that exhibited toxicity but did not have contaminant concentrations that exceeded NOEL or ER-L values. The combined data for this site suggested that unmeasured chemicals or conditions were causing the observed bioeffects.

Station 1 in the innermost portion of Wellfleet Harbor showed the strongest evidence of a degraded benthic system. This station had the second lowest species richness and the strongest toxicity responses among the 12 sites. The remaining three

Executive Summary (continued)

Wellfleet Harbor sites also showed stronger toxicity responses than other harbor sites. In addition, most Wellfleet Harbor stations except Station 4 (the targeted reference site for this harbor) were inhabited by benthic communities that were relatively species poor (defined here as ≤ 50 species per site).

Linking Ecological Degradation with Chemical Contaminants

Contaminants that could have been responsible for the observed bioeffects at Wellfleet Harbor sites were dieldrin (Station 1), lead (Station 1), and chlordane (Stations 1, 3, and 4). All three of these contaminants were present at concentrations that exceeded corresponding NOEL or ER-L values, which serve as threshold effects levels, though none exceeded higher "probable effects level" (PEL) (MacDonald 1992) or "effects range-median" (ER-M) (Long and Morgan 1990) concentrations known to cause adverse bioeffects on a wider variety of benthic organisms. As noted above, concentrations of unionized ammonia and sulfide were also very high at Station 1 and could have caused, or contributed to, the observed bioeffects at this site.

Though the strongest bioeffects were observed in Wellfleet Harbor, the sites in this harbor system had the lowest overall sediment contamination. The most contaminated sites were in Boston and Salem/Beverly Harbors. The number of contaminants that exceeded the corresponding NOEL or ER-L values ranged from 25 to 16 among the Boston Harbor sites, and from 23 to 16 among Salem/Beverly Harbor sites, compared to only 1 to 3 among Wellfleet Harbor sites.

Stations 10 and 11 in Salem Harbor were the most and second-most contaminated sites with respect to the overall amounts by which contaminants exceeded their corresponding NOEL/ER-L values. Concentrations of pesticides and metals were especially high at these two sites. The third-most contaminated site was Station 5 off Deer Island in Boston Harbor. This station had among the highest concentrations of dieldrin and total DDT. Station 9 in Beverly Harbor was ranked as the fourth-most contaminated site, which had the highest concentrations of total and individual PAHs. PAH assemblages at all stations were dominated by the 4- to 6-ringed PAHs (e.g., fluoranthene, pyrene, benzo(b)fluoranthene), which are primarily of pyrogenic origin.

Typically, PCBs, dieldrin, total DDT, silver, copper, and zinc were present at the highest concentrations among Boston Harbor sites. In comparison, phenanthrene, benz(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, pyrene, total PAHs, chlordane, total DDE, total DDD, DDT+DDD+DDE, arsenic, chromium, lead, and mercury were usually present at the highest concentrations among Salem/Beverly Harbor sites. All stations in Boston and Salem/Beverly Harbors had one or more of these contaminants present at concentrations that exceeded NOEL or ER-L values and could have caused the observed bioeffects. These effects consisted of significant sediment (or porewater) toxicity at all sites and

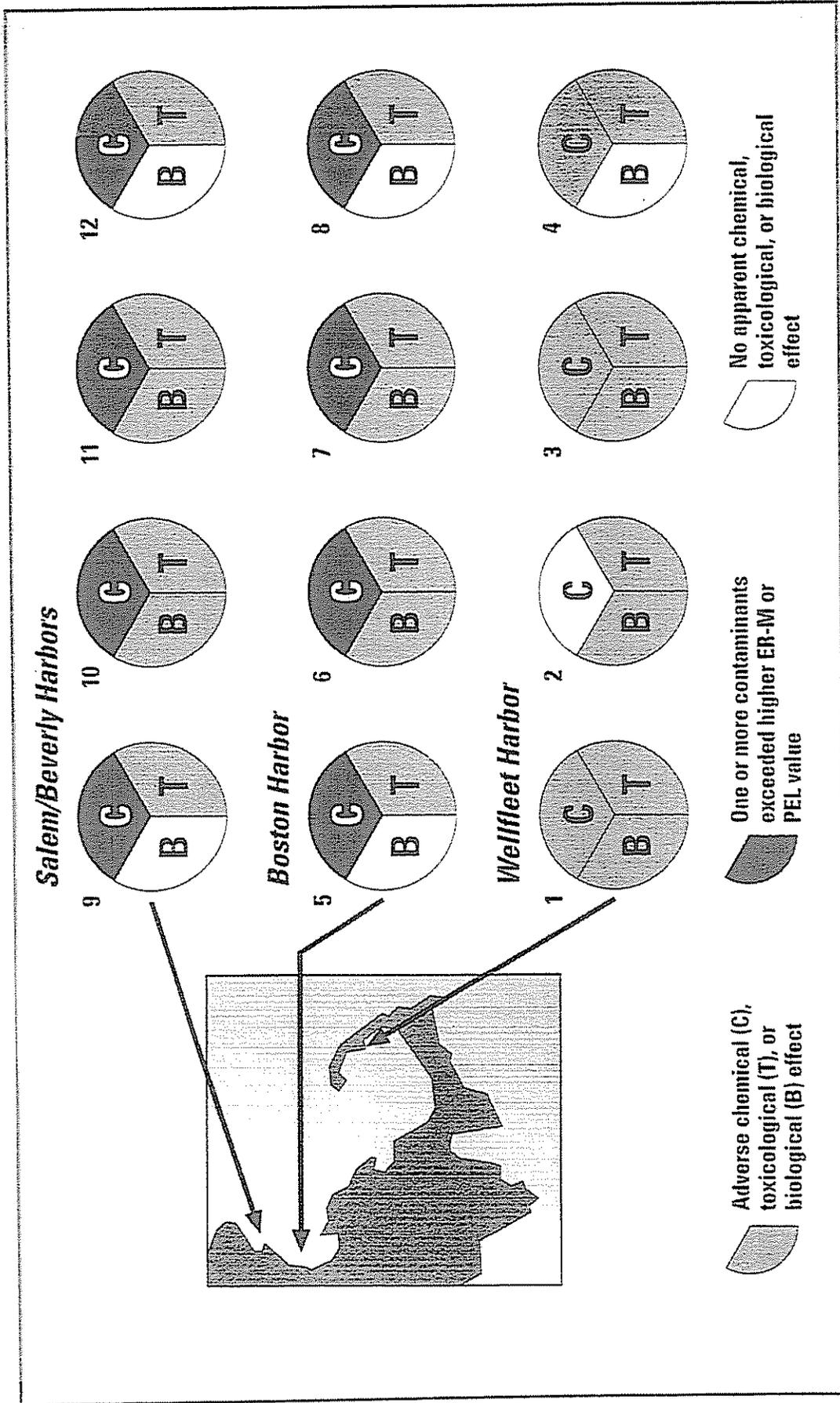
Executive Summary (continued)

the presence of a species-poor benthos at Stations 6 and 7 in Boston Harbor and Stations 10 and 11 in Salem Harbor.

Contaminants that may be causing the most ecological harm in these latter two harbor systems are silver, chlordane, and DDT in Boston Harbor and chromium, lead, chlordane, and DDD in Salem/Beverly Harbors. These contaminants were present at one or more sites at concentrations that exceeded the higher PEL and/or ER-M bioeffect values. Silver exceeded PEL/ER-M values at all four sites in Boston Harbor. In comparison, chromium exceeded PEL/ER-M values at all sites in Salem/Beverly Harbors. Remaining PEL/ER-M exceedances were as follows: lead (Station 11), chlordane (Stations 6, 10, and 11), DDT (Station 5), and DDD (Stations 9, 10, 11).

Stations targeted as reference sites within each harbor system appeared to be the least degraded based on the combined sediment contaminant, toxicity, and benthic community data. These sites seem reasonable to use as reference sites in any future sediment quality monitoring in the region. However, as this study has shown, it is very difficult to find any nearshore depositional environment in the region that is completely free of chemical contaminant inputs or some level of ecosystem degradation.

This study also demonstrates that factors other than chemical contaminant loading must be considered as possible causes of the biologically adverse condition of sediments in some of these coastal harbor systems. High organic loading and associated increases in the ammonia and hydrogen sulfide content of sediment porewater may be important factors contributing to the high toxicity of Wellfleet Harbor sediments, which appear to have experienced far less chemical contamination than the more urbanized Boston and Salem/Beverly Harbor systems.



Summary of Sediment Quality Triad Results

1.0 Introduction

1.1 Background and Scope of Study

The Massachusetts Bays Program (MBP) recently released a draft Comprehensive Conservation and Management Plan (CCMP) that serves as a blueprint for coordinated action to restore and protect the water quality and diverse natural resources of the Massachusetts/Cape Cod Bays ecosystem (MBP 1994). The plan identified a need to establish baseline conditions of the health of living resources and their habitats in Massachusetts and Cape Cod Bays and to develop a better understanding of how various anthropogenic activities may adversely affect them. The present study was conducted in an effort to develop new information that could be used in addressing this particular need. In addition, the data are intended to help the MBP achieve one of its overall goals of providing a scientific basis for management decisions directed at protecting living resources and habitats of this valuable coastal system.

In this study, the Sediment Quality Triad (SQT) method was used to examine linkages between sediment contaminant concentrations and their potential impacts on living benthic resources of the Massachusetts/Cape Cod Bays ecosystem. This method combines sediment contaminant analyses, sediment toxicity testing, and measures of ambient biological conditions as a means of quantitatively assessing pollution-induced degradation of the benthic environment. This is an integrative approach of linking measured contaminant levels in sediments to their capacity to cause toxic effects (as indicated by laboratory toxicity tests with field sediments) and to actual adverse conditions in populations of resident organisms living in these same sediments. The method was developed originally with data derived from polluted harbors on the west coast of the United States, including Puget Sound (Long and Chapman 1985, Chapman 1986) and San Francisco Bay (Chapman et al. 1987, Long and Morgan 1990). Since then, the method has been used to assess pollutant impacts in other U.S. coastal systems, including the Galveston Bay estuary, Texas (Carr 1993); around an oil platform in the Gulf of Mexico (Chapman et al. 1991); and in the Tampa Bay estuary, Florida (SAIC 1992, USFWS 1992). Currently the method is being applied in the New York/New Jersey Harbor system (D. Suszkowski, Hudson River Foundation, personal communication). Results of these studies have demonstrated the strength of using multiple indicators as a basis for identifying areas where sediment contamination is responsible for ecosystem degradation.

Sediment samples were collected simultaneously for chemical and physical analyses, toxicity testing, and an assessment of benthic community structure at 12 stations in Massachusetts and Cape Cod Bays (Figure 1.1). The stations consisted of four sites from shallow subtidal depths (0.3 to 5.1 m Mean Lower Low Water [MLLW]) in each of three harbors: Wellfleet Harbor, Boston Harbor, and Salem/Beverly Harbors. Stations were selected within these protected harbor areas to maximize the chance of sampling in nearshore depositional environments where sediment-sorbed contaminants would be more likely to accumulate and persist. Substrates at most stations are represented by fine-grained sediments (< 75 percent sand), although stations in Wellfleet Harbor exhibited higher percentages of sand.

1.0 Introduction (continued)

From each of the 12 stations, the following samples were collected and analyzed: three replicate grab samples of sediment for analysis of benthic community structure (replicates analyzed separately in the laboratory); one composite sediment sample, pooled from multiple random grabs, for sediment and porewater toxicity testing (composite sample divided into five replicates in the laboratory and the replicates tested separately in comparison to negative controls); one composite sediment sample, pooled from multiple random grabs, for analysis of chemical contaminants (one composite sample analyzed in the laboratory); and other supportive measurements of physical/chemical properties of sediment and near-bottom water.

Chemical and physical variables that were measured in sediments consisted of total hydrocarbon content (THC), polynuclear aromatic hydrocarbons (PAHs), chlorinated pesticides, polychlorinated biphenyls (PCBs), metals, grain size, total organic carbon (TOC), and E_n . Dissolved oxygen (DO), temperature, pH, and salinity were measured in samples of near-bottom water. Ammonia, hydrogen sulfide (H_2S), DO, pH, and salinity were also measured in porewater samples. Sediment toxicity was measured using two standard methods: a solid-phase acute toxicity test with the marine amphipod *Ampelisca abdita* (ASTM 1990) and an early-life-stage toxicity test with the sea urchin *Arbacia punctulata* (Weber et al. 1988) modified for exposures to sediment porewater samples. The sea urchin test incorporated both a sperm-cell fertilization test and a test for abnormal embryonic development (to the echinopluteus stage). Measures of infaunal community structure and composition (total numbers of species, Shannon-Weaner diversity and evenness, combined species abundances, individual species abundances) were based on the macrofaunal (> 0.5 mm) size component and on species-level data.

1.2 Objectives

The overall purpose of this study was to provide a quantitative assessment of potential linkages between sediment contamination and adverse impacts on living benthic resources of the Massachusetts Bay/Cape Cod Bay ecosystem, based on the above multiple indicators of ecological condition in samples from the three different harbor systems. In an effort to fulfill this overall goal, data were examined with respect to several key objectives:

1. To determine whether contaminants in sediments at any of the 12 Massachusetts/Cape Cod Bay stations were present at elevated concentrations reported to cause adverse effects on marine organisms
2. To determine whether sediments and/or porewater from sediment collected at the 12 Massachusetts/Cape Cod Bays stations were significantly toxic to test populations of marine organisms (amphipod *Ampelisca abdita* and sea urchin *Arbacia punctulata*) based on comparisons of survival and other sublethal biological responses in negative controls
3. To examine patterns in macroinfaunal community structure among the various sites and identify any signs of pollutant-related stress in these assemblages

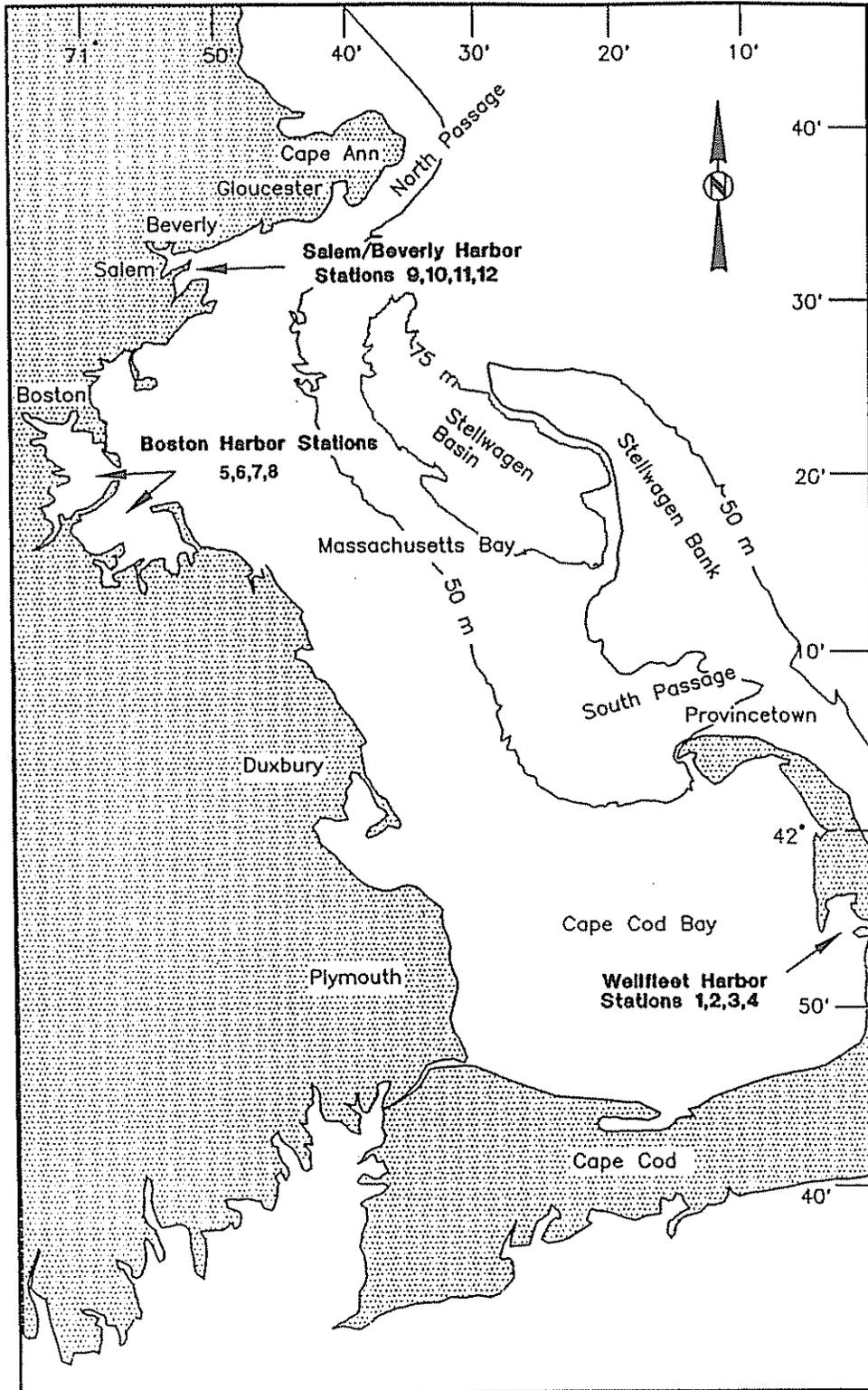


Figure 1.1 Study area and sediment sampling locations

1.0 Introduction (continued)

4. To examine relationships between the chemical, toxicological, and biological data as a means of identifying sites where sediment contamination could be responsible for observed bioeffects (significant toxicity occurrences and/or altered benthic community structure)
5. To compare differences in contaminant trends and degree of biological impacts among the three harbor areas

These objectives were examined by a combination of statistical tests and other data-comparison methods. Wherever appropriate these objectives were framed as null hypotheses and tested statistically.

Two additional objectives that we hope to have accomplished through this research are:

1. To establish a series of triad sites that can be used in the future for monitoring trends on how pollution-induced impacts in the bays are changing with time
2. To provide data of potential use in ongoing efforts (by EPA and various states) to develop sediment quality criteria for the protection of living benthic resources

1.3 The Study Area

Massachusetts and Cape Cod Bays form an interconnected estuarine-inner shelf system off the Massachusetts coast between Cape Ann and Cape Cod (Figure 1.1). Massachusetts Bay is bounded on the north by Cape Ann, on the west by the Massachusetts coast, on the south by Cape Cod Bay, and on the east by Stellwagen Bank, a topographic high defined by the 50-m depth contour. Cape Cod Bay is a shallower semi-enclosed embayment bounded largely by the Massachusetts coast south of Duxbury and the Cape Cod peninsula. There are no prominent submarine barriers between Cape Cod and Massachusetts Bay. Surface water circulation within both bays is dominated by a counter-clockwise flow that enters Massachusetts Bay from the Gulf of Maine through North Passage off Cape Ann, continues south and east into Cape Cod Bay, and exits back into the Gulf of Maine through South Passage off Provincetown (Bigelow 1927, Bumpus 1974, Geyer et al. 1992).

Pollutant distributions within the two interconnected bays are believed to be influenced by this circulation pattern in combination with contaminant inputs from urban centers such as Boston (Shea et al. 1991). MacDonald (1991) described an overall gradient of decreasing sediment contamination extending from Boston Harbor into Massachusetts Bay. Townsend et al. (1991) described a similar enrichment gradient with respect to nutrients. Sources of contamination in Boston Harbor have included sewage and industrial effluents, sewage sludge (although no longer discharged since 1991), combined sewer overflows, and storm drains. Such inputs have resulted in high sediment concentrations of metals, pesticides, PCBs, and PAHs (MacDonald 1991, Cahill and Imbalzano 1991). These same contaminants have been found at high concentrations in the Salem/Beverly harbor system as well (MacDonald

1.0 Introduction (continued)

1991, Camp, Dresser, & McKee, Inc. 1991). Shea et al. (1991) reported generally low contaminant levels and overall high sediment quality for the majority of Massachusetts and Cape Cod Bays, with the exception of Boston Harbor, Salem/Beverly Harbors, and the Massachusetts Bay Disposal Site located further offshore in Stellwagen Basin.

Glaciation and sea-level fluctuations have created an irregular seafloor topography across much of the region which, in combination with oceanographic processes, has resulted in an extremely patchy distribution of modern sedimentary environments, consisting of sites of erosion and nondeposition, sediment reworking, and deposition (Knebel 1993). The estuarine parts of the system serve as effective traps for finer-grained materials because of their protected nature (Knebel 1993). However, diverse sediment types can occur within these areas. For example, inside Boston Harbor depositional environments (containing finely grained muddy sands and muds) are found over extensive subtidal flats and within sheltered depressions, however sites of erosion and sediment reworking occur as well (Knebel 1993). Sediments along the majority of the Cape Cod Bay coastline consist of high percentages of sand (Young and Rhoads 1971, Schlee et al. 1973).

The biological resources of Massachusetts and Cape Cod Bays are rich and abundant. Gilbert et al. (1976) found high infaunal densities, species diversity, and species richness in several parts of the bay system. The region supports a variety of commercially and recreationally important species of fishes and invertebrates. Six species of endangered cetaceans and five species of endangered or threatened sea turtles also inhabit these bays (EPA 1993).

2.0 Methods

Summary descriptions of methods used in various field and laboratory components of the study are provided in this section. References to unpublished Standard Operating Procedures (SOPs) used by participating laboratories are made for several of the methods (in addition to published standard methods wherever appropriate). These SOPs, which provide further details of the specific procedures, are not included in the present report but are listed with full titles in Appendix A.

2.1 Site Selection

Twelve stations were established in Massachusetts and Cape Cod Bays (Figure 1.1). These stations consisted of three suspected contaminated sites and a corresponding suspected reference site in each of three harbor systems. In order to provide as broad a baywide coverage as possible with the 12 stations, the three harbor systems were selected from a northern area (Salem/Beverly Harbors), central area (Boston Harbor), and Cape Cod Bay (Wellfleet Harbor). Within each harbor, four stations were established and sampled in order to (1) provide a basis for examining differences in the measured environmental variables among stations within a particular harbor system and (2) provide a measure of spatial variability within each harbor, so that contaminant trends and associated biological impacts could be compared among the three different harbor systems.

Locations of stations within each of the three harbor systems are illustrated in Figures 2.1 through 2.3. All stations but one (Station 12 in the Salem/Beverly Harbor area, see below) were selected based on the following or comparable procedure. Each harbor was divided into a series of subtidal sampling zones demarcated on digitized maps and each zone then further subdivided with a grid. Sampling zones within a harbor were selected to include both suspected contaminated and reference areas, (Stations 4, 8, and 12), based either on background information or on consideration of the local geography and proximity to anthropogenic influences. Also, wherever possible, background sediment data were used to locate sampling zones in known depositional environments.

All grid cells within a zone were numbered and one cell from each zone was selected as a station using a random number table. Thus, within a zone, stations were selected randomly. Target coordinates for each station were determined as the center of each randomly selected grid cell. If unsuitable substrates (e.g., sites in the middle of a navigation channel or with excessive amounts of cobble, gravel, and shell hash) were encountered in the field at the target coordinates for a particular station, then an alternate site was sought first by moving the vessel to slightly different positions within the same grid cell, or next by moving to another randomly selected cell. Choices were targeted to a depth range of about 1 to 10 m MLLW so that sampling would occur consistently within the nearshore subtidal depth zone. Also, wherever possible, substrates consisting of less than about 75 percent sand were sampled. Station 12, which was intended to serve as a reference site for Salem/Beverly Harbor, was established at the location shown in Figure 2.3 because this was the only site in the vicinity of Salem and Beverly Harbors where nautical charts indicated the

2.0 Methods (continued)

presence of a mud bottom and where high sediment contamination had not been documented in other independent studies.

Coordinates (latitude and longitude) and depths of the final station locations are provided in Table 2-1. A brief description of the three harbor sampling areas is given below.

Wellfleet Harbor -- Wellfleet Harbor sampling locations (Figure 2.1) consisted of Station 1 in the inner harbor area, Stations 2 and 3 in the middle harbor area, and Station 4 in the outer part of the harbor. All four of these stations have high percentages of sand ranging from 84 to 97 percent. The outermost Station 4 was intended to serve as a reference area relative to the other stations, which are located closer to municipal and recreational activities in the inner harbor area. There are no major industrial or urban centers in the area comparable to those influencing Boston Harbor and Salem/Beverly Harbors. Consistent with this point, the authors are not aware of any prior reports of high contaminant loading in this harbor system. Wellfleet Harbor also is a study area for the "Mini-Bays" program.

Boston Harbor -- Boston Harbor sampling locations (Figure 2.2) consisted of Station 5 off Deer Island, in a depositional environment between the mouth of Winthrop Bay and President Roads; Station 6 in southwest Dorchester Bay, in a depositional environment between Roxbury, Squantum, and Thompson Island; Station 7 in Quincy Bay, in a depositional environment between Squantum and Nut Island; and Station 8 in Hull Bay, in a depositional environment between Bumkin Island and Hog Island. The presence of depositional environments in these particular areas is indicated in Knebel et al. (1990, Fig. 9) and in Rendigs and Oldale (1990, Map MF-2124). Kelly and Kropp (1992, Table 7) also reported high percentages of silt and clay at their stations R7 (southwest Deer Island), T4 (southwest Dorchester Bay), T7 (Quincy Bay), and T8 (Hull Bay).

Among 23 National Status and Trends (NS&T) Program sites in New England, sites off southwest Deer Island, in Dorchester Bay, and in Quincy Bay were ranked as the first, second, and fourth-most contaminated, respectively, with respect to overall sediment contamination from a variety of metals, DDT, PCBs, and PAHs (MacDonald 1991). Cahill and Imbalzano (1991) evaluated a number of other historical records of organic and metal contamination in Boston Harbor sediments in addition to the NS&T database and found concentrations in excess of upper-limit sediment quality criteria (concentrations above which effects are usually seen for most benthic species) in several areas of the harbor including Deer Island and Dorchester Bay. For example, these investigators reported concentrations of copper in Dorchester Bay in excess of 390 ppm (the Apparent Effects Threshold value); of benzo[a]pyrene off Deer Island in excess of 1,063 $\mu\text{g/g-OC}$ (EPA interim sediment criteria value based on the Equilibrium Partitioning approach); and of phenanthrene in excess of 102 $\mu\text{g/g-OC}$ (EPA interim criteria value) for Deer Island and Dorchester Bay.

Table 2-1 Station depths and coordinates.

Station	Location	Depth (meters at mean low)	LAT	LONG
1	Inner Wellfleet Harbor	0.30	41 55'47" N	070 01'43" W
2	Mid Wellfleet Harbor	0.49	41 55'29" N	070 02'07" W
3	Mid Wellfleet Harbor	2.3	41 54'42" N	070 02'23" W
4	Outer Wellfleet Harbor	2.8	41 54'10" N	070 02'51" W
5	Deer Island	4.7	42 20'40" N	070 58'44" W
6	Dorchester Bay	2.8	42 18'33" N	070 01'36" W
7	Quincy Bay	3.1	42 16'49" N	070 59'16" W
8	Hull Bay	2.4	42 17'21" N	070 53'47" W
9	Beverly Harbor	3.3	42 32'16" N	070 52'29" W
10	Outer Salem Harbor	5.1	42 31'30" N	070 52'01" W
11	Inner Salem Harbor	3.6	42 30'46" N	070 52'23" W
12	Dolliber Point	4.3	42 31'14" N	070 51'00" W

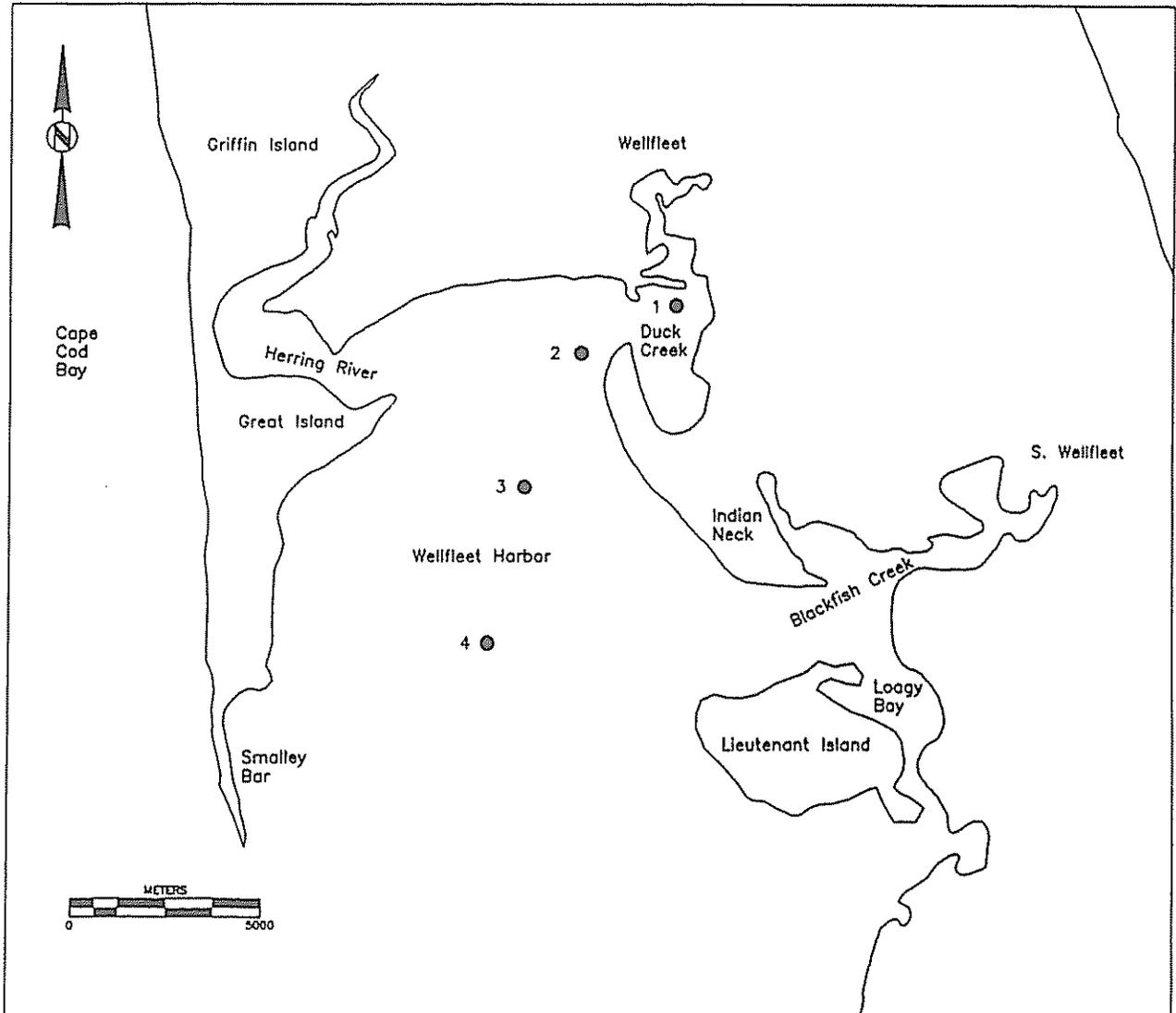


Figure 2.1 Sampling Stations In Wellfleet Harbor.

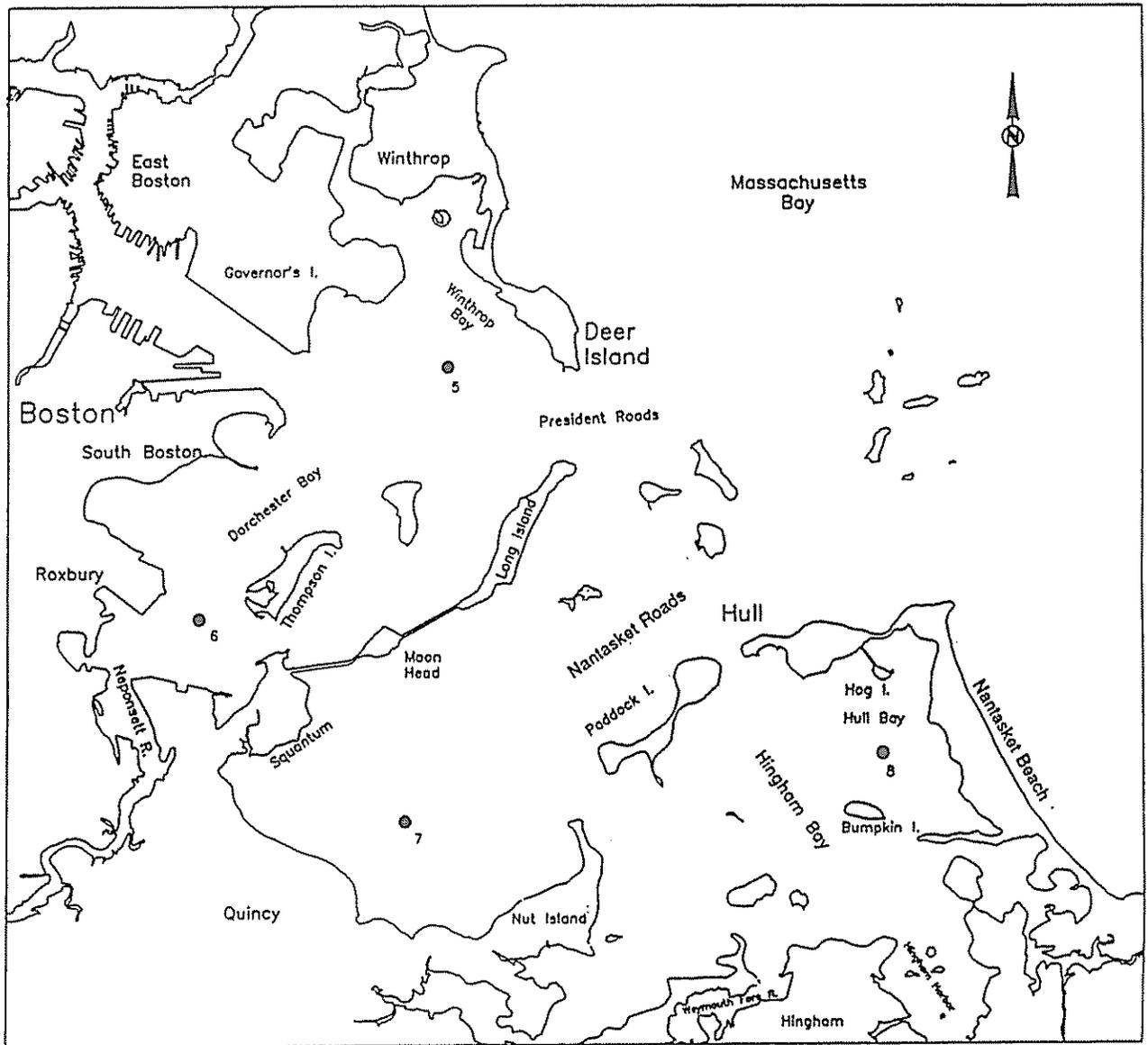


Figure 2.2 Sampling stations in Boston Harbor

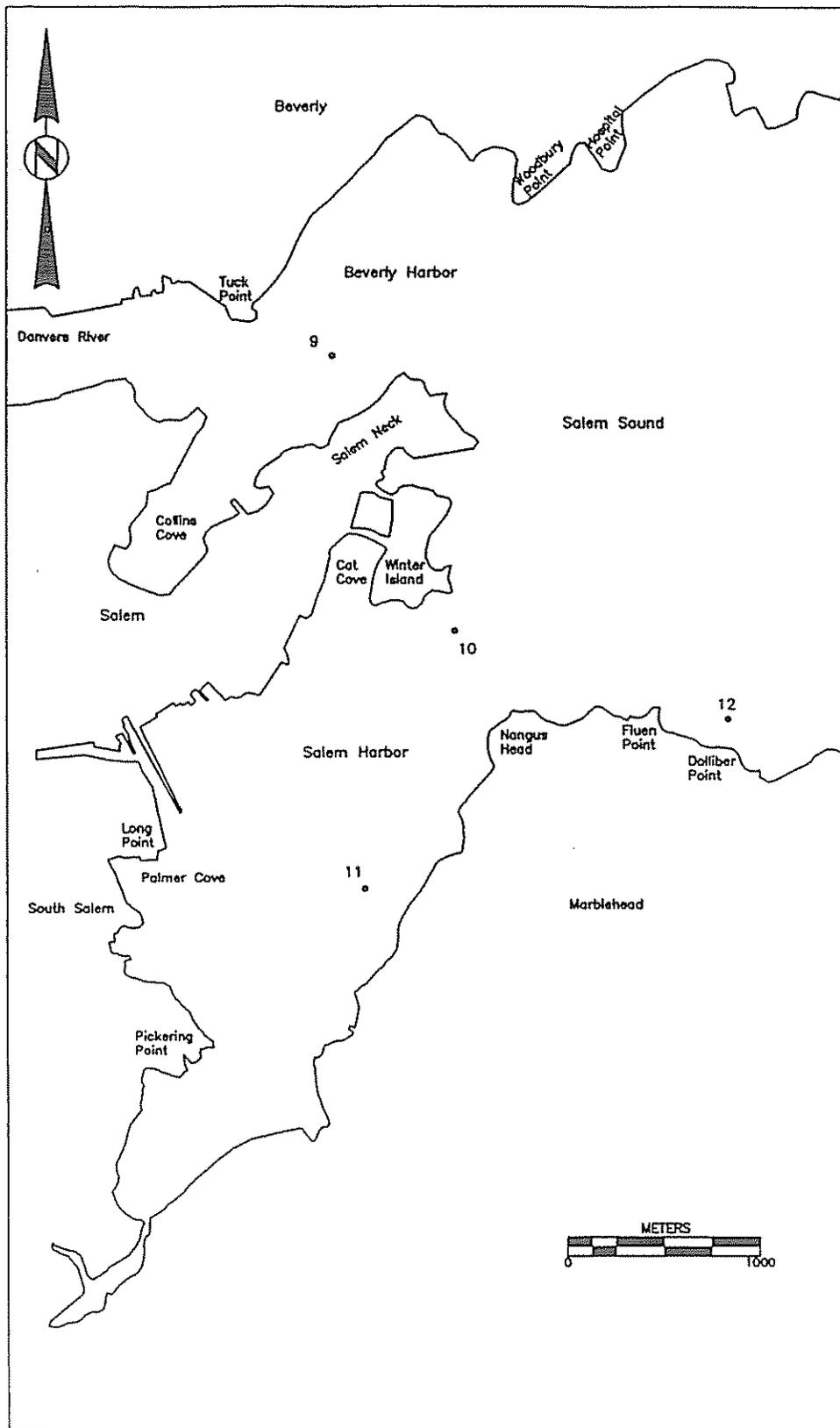


Figure 2.3 Sampling stations in Salem and Beverly Harbors

2.0 Methods (continued)

Station 8 in Hull Bay was intended to serve as a reference site for Boston Harbor. MacDonald (1991) reported that the lowest levels of sediment contamination in Boston Harbor generally are found in the southeastern part of the harbor in addition to the mouth of the harbor. Kelly and Kropp (1992) found highest species richness out of eight stations in Boston Harbor at their station T8 off Bumkin Island in Hull Bay.

Salem/Beverly Harbors -- Salem/Beverly Harbor sampling locations (Figure 2.3) consisted of Station 9 in Beverly Harbor, Station 10 in outer Salem Harbor, Station 11 in inner Salem Harbor, and Station 12 located within a protected coastal indentation near Dolliber Point off Marblehead. Station 12, removed from the immediate industrial and municipal influences inside the harbors, was intended to serve as a reference area for the other three stations. These stations appeared to be in depositional environments containing sediments with relatively high percentages of silt and clay, though the reference site had a higher percentage of sand than the other sites inside the harbors.

Similar to Boston Harbor, there is a historical record of sediment contamination in Salem and Beverly Harbors. Among the 23 NS&T Program sites in New England, a site in Salem Harbor was ranked as the third-most contaminated site overall with respect to nine metals, DDT, PCBs, and PAHs. The Salem Harbor site was ranked as the most or second-most contaminated site with respect to Cd, Cr, Pb, Zn, and DDT. Camp, Dresser, & McKee, Inc. (1991) also reported high levels of As, Cr, and PAHs in sediments from Salem Harbor in excess of Massachusetts Class III sediment concentrations for dredging and disposal operations.

2.2 Field Sampling

At each of the 12 stations, three replicate grabs for infaunal analyses were collected with a 0.1-m² modified van-Veen grab sampler. After subsampling for E_h , contents of the grabs were live-sieved in the field with a 0.5-mm mesh screen. Material retained on the screen was placed in polyethylene jars with 10 percent buffered formalin and transferred to the laboratory at Ruff Systematics, Inc. (Solana Beach, California) for subsequent infaunal analysis.

Prior to sieving, E_h was measured in each of the three undisturbed grabs at 2-cm intervals from the surface to the bottom of the grabs to identify the depth of the redox potential discontinuity (RPD) using an Orion 9678BN redox probe. In addition, pH, DO, temperature, and salinity were measured at each station in samples of near-bottom water collected with a Go-Flo™ bottle. These water quality parameters were measured with a Horiba (Model U10) probe system. Water depth was recorded with the vessel's fathometer.

At each of the 12 stations the grab sampler was used to collect approximately 6 L of surficial sediment (upper 2 cm) for chemical analyses and the two toxicity tests. These samples were collected after the infaunal samples were collected to minimize disturbance of the seafloor prior to sampling the infauna. As these samples were

2.0 Methods (continued)

collected, surface water was decanted and the upper 2 cm of sediment removed from the grabs. Once approximately 6 L of surficial sediment were obtained, the sediment was combined in a large bowl and homogenized by stirring until no color or textural differences could be visually detected. Two liters of the homogenized sediment were placed in polyethylene jars with lids, stored on ice, and transferred to T.R. Wilbury Laboratories, Inc. (Marblehead, Massachusetts) to be used in the solid-phase sediment toxicity test with *Ampelisca abdita*. A minimum of 1 L of sediment from each site was saved as backup material until all testing was completed.

Three liters of the homogenized sediment from each site was placed in polyethylene jars with lids, stored on ice, and shipped to Corpus Christi State University to be used in the porewater toxicity tests with sea urchins. The extraction of porewater from these sediments was performed using a pressurized squeeze extraction device (Corpus Christi SOP No. F10.9). Salinity, DO, pH, sulfide, and ammonia (total ammonia nitrogen and unionized ammonia nitrogen) were measured in each of the sediment porewater samples.

The remaining 1 L of homogenized sediment from each site was used for analysis of TOC, grain size, hydrocarbons, trace metals, chlorinated hydrocarbon pesticides, and PCBs. Approximately 100 mL of sediment were aliquoted separately for grain size and TOC analysis. These sediment samples were placed in plastic Ziplock bags and transported on dry ice to subcontract laboratories for analysis. In addition, 400 mL of the homogenized sediment were aliquoted separately for organic compounds and metals analyses. These samples were placed in pre-cleaned 500-mL glass jars, shipped on dry ice to Arthur D. Little, and kept frozen (-20°C) in the laboratory for analysts.

In addition to field sediment samples, quality control deck samples (i.e., field blanks and equipment blanks) were collected and held for possible analysis in the event that any anomalies were noted in field-sample data. One field blank was collected at each station and one equipment blank was collected for each day of field operation. The field blank served as a measure of any potential atmospheric contamination during the sediment aliquoting procedure and of any potential contamination associated with the glassware. A pre-cleaned 500-mL glass jar was carried into the field, opened during the aliquoting process when the sample jar was open, and returned to the laboratory with the field samples. This blank was stored with the field samples at -20°C. The equipment blank served as a measure of any potential contamination that may have been associated with the grab sampler. After the sediment samples were collected, the grab was decontaminated by a series of acetone/methylene chloride/acetone/deionized water rinses. The grab was then rinsed a final time with high-purity deionized water and the rinsate collected directly into a pre-cleaned 2-L glass bottle. The rinsate was stored refrigerated at 4°C (held in coolers with blue ice on board the vessel and during transport to the lab) and then extracted with methylene chloride within approximately 14 days.

Sampling was conducted from a 25-ft Privateer (owned and operated by TG&B Marine Services of Falmouth, Massachusetts) equipped with a davit and motorized

2.0 Methods (continued)

capstan for operating the grab sampler. Vessel/station positioning was accomplished using a combination of the vessel's Loran and a Magellan Nav Plus 1000 satellite receiver (GPS). Sampling was conducted during three separate cruise legs: Wellfleet Harbor was sampled July 2-3, 1993; Salem/Beverly Harbors were sampled July 8-9, 1993; Boston Harbor was sampled July 12-13, 1993.

Test populations of the amphipod *Ampelisca abdita* used in the sediment toxicity tests were purchased from a commercial supplier (East Coast Amphipod, Kingston, Rhode Island) in Fishing Cove, Wickford, Rhode Island. Additional surficial sediment also was obtained from the site where amphipods were collected as a negative control for comparison against field sediments. These sediments were processed and transferred to the laboratory following the same procedures described above for the field sediments. Gametes for the sea urchin tests were obtained from adult *Arbacia punctulata* collected from jetties in Port Aransas, Texas. Control porewater (i.e., negative controls for the sea urchin tests) were obtained from sediments collected at a nearby reference site in Red Fish Bay, Texas.

2.3 Chemical and Physical Characterizations

Chemical contaminants that were measured in sediment samples consisted of total hydrocarbons, PAHs, chlorinated pesticides, PCBs, and metals (see Tables 2-2, 2-3 and 2-4). These lists include all of the analytes typically measured on EPA EMAP and NOAA NS&T programs (including the concurrent NOAA survey of sediment contamination and toxicity in Boston Harbor). Other supporting chemical and physical properties in various matrices were measured as well. At each site, sediment grain size and TOC were measured from composited subsamples of the three replicate infaunal grab samples. E_n was measured prior to sieving in each of the three undisturbed infaunal grabs at 2-cm intervals from the surface to a depth of 10 cm. Salinity, temperature, DO, and pH were also measured in samples of near-bottom water at each station. In addition, salinity, temperature, DO, pH, sulfide, and ammonia were measured in porewater samples at each station. Brief descriptions of the procedures used to analyze these various parameters are provided below.

2.3.1 Organic compound analyses. The frozen sediment samples were thawed and homogenized. A 30-g subsample was taken for organic chemical compound analysis and a 5-g subsample was taken for moisture content analysis (to provide information for reporting analyte concentrations on a per-dry-weight-sediment basis). Remaining portions of the samples were archived. An overall summary of laboratory methods for the extraction and analysis of organics is provided in Table 2-5.

PAHs, PCBs, and chlorinated pesticides were co-extracted from sediments simultaneously (Arthur D. Little SOP No. ADL-2819). The sediment to be extracted was placed in a clean glass or Teflon[®] container, mixed with sodium sulfate, and spiked with the appropriate amount of surrogate compounds. For PCBs and pesticides, 0.2 µg each of 4,4'-dibromooctafluorobiphenyl (DBOBF) and PCB

Table 2-2 List of measured PAH and alkyl PAH target compounds.

Compound	Surrogate Reference	Compound	Surrogate Reference
Naphthalene	1	Fluoranthene	2
C1-Naphthalenes	1	Pyrene	2
1-Methylnaphthalene	1	C1-Fluoranthenes/Pyrenes	2
2-Methylnaphthalene	1	C2-Fluoranthenes/Pyrenes	2
C2-Naphthalenes	1	C3-Fluoranthenes/Pyrenes	2
2,6-Dimethylnaphthalene	1		
C3-Naphthalenes	1	Chrysene	3
C4-Naphthalenes	1	C ₁ -Chrysenes	3
Acenaphthylene	1	C ₂ -Chrysenes	3
Acenaphthene	1	C ₃ -Chrysenes	3
		C ₄ -Chrysenes	3
Biphenyl	1		
		Benzo[a]anthracene	3
Fluorene	1	Benzo[b]fluoranthene	3
C ₁ -Fluorenes	1	Benzo[k]fluoranthene	
C ₂ -Fluorenes	1		3
C ₃ -Fluorenes	1	Benzo[a]pyrene	3
		Benzo[e]pyrene	3
Dibenzothiophene	2		3
C ₁ -Dibenzothiophenes	2	Perylene	3
C ₂ -Dibenzothiophenes	2		3
C ₃ -Dibenzothiophenes	2	Indeno[1,2,3-c,d]pyrene	3
Phenanthrene	2	Dibenzo[a,h]anthracene	3
1-Methylphenanthrene	2		
Anthracene	2	Benzo[g,h,i]perylene	3
C ₁ -Phenanthrenes/Anthracenes	2		
C ₂ -Phenanthrenes/Anthracenes	2		
C ₃ -Phenanthrenes/Anthracenes	2		
C ₄ -Phenanthrenes/Anthracenes	2		
<u>Surrogate Compounds</u>			
Acenaphthene-d ₁₀	1,A		
Phenanthrene-d ₁₀	2,A		
Benzo(a)pyrene-d ₁₂	3,B		
Naphthalene-d ₈	A		
<u>Internal Standards</u>			
Fluorene-d ₁₀	A		
Chrysene-d ₁₂	B		

Table 2-3 List of measured PCB congeners and pesticides.

PCB No.	Compound Name
8	2,4'-dichloro biphenyl (BP)
18	2,2',5-trichloro BP
28	2,4,4'-trichloro BP
44	2,2',3,5'-tetrachloro BP
52	2,2',5,5'-tetrachloro BP
66	2,3',4,4'-tetrachloro BP
77	3,3',4,4'-tetrachloro BP
87	2,2',3,4,5'-pentachloro
101	2,2',4,5,5'-pentachloro
105	2,3,3',4,4'-pentachloro
118	2,3',4,4',5-pentachloro
126	3,3',4,4',5-pentachloro
128	2,2',3,3',4,4'-hexachloro BP
138	2,2',3,4,4',5'-hexachloro BP
153	2,2',4,4',5,5'-hexachloro BP
154	2,2',4,4',5,6'-hexachloro BP
170	2,2',3,3',4,4',5-heptachloro BP
180	2,2',3,4,4',5,5'-heptachloro BP
187	2,2',3,4',5,5',6-heptachloro BP
195	2,2',3,3',4,4',5,6-octachloro BP
200	2,2',3,3',4,5',6,6'-octachloro
206	2,2',3,3',4,4',5,5',6-nonachloro BP
209	decachloro BP
Total PCBs	Sum of all PCBs
<u>Pesticides</u>	
	hexachlorobenzene
	gamma-BHC (Lindane)
	heptachlor
	aldrin
	heptachlor epoxide
	cis-chlordane
	trans-nonachlor
	dieldrin
	mirex
	2,4'-DDE
	4,4'-DDE
	2,4'-DDD
	4,4'-DDD
	2,4'-DDT
	4,4'-DDT

Table 2-4 Summary of trace and major metals analyzed from sediment samples and corresponding methods. (All concentrations reported on a dry weight basis in $\mu\text{g/g}$, unless designated otherwise.)

Element	Technique	EPA SW-846 Sample Preparation Method ^a	EPA SW-846 Instrument Method ^a	EPA-CLP Instrument Method ^b
Ag	GFAA	3050	NA	272.2
As	GFAA	3050	7060	206.2
Se	GFAA	3050	7740	270.2
Pb	GFAA	3050	7421	239.2
Tl	GFAA	3050	7841	279.2
Al ^c	ICAP	3050	6010	200.7
Be	ICAP	3050	6010	200.7
Cd	ICAP/GFAA	3050	6010/7131	200.7/213.2
Cr	ICAP	3050	6010	200.7
Cu	ICAP	3050	6010	200.7
Ni	ICAP	3050	6010	200.7
Sb	ICAP/GFAA	3050	6010/7041	20.7/204.2
Sn	ICAP/GFAA	3050	6010/NA	200.7/NA
V	ICAP	3050	6010	200.7
Zn	ICAP	3050	6010	200.7
Fe ^c	ICAP	3050	6010	200.7
Mn	ICAP	3050	6010	200.7
Hg	CVAA	7471	7471	245.5

^aEPA (1986)

^bEPA (1991)

^cReported in % dry weight

GFAA = Graphite Furnace Atomic Absorption Spectrophotometry

ICAP = Inductively Coupled Plasma Atomic Absorption Spectrophotometry

CVAA = Cold Vapor Atomic Absorption Spectrophotometry

Table 2-5 Summary of laboratory methods for the analysis of organic compounds in sediment samples. (All concentrations were reported on a dry weight basis in ng/g.)

Analyte Group	Extraction Methods and Reference	Cleanup Method and Reference	Fractionation Method and Reference	Instrumental Method and Reference	Surrogate Compounds	Internal Standards
PAHs*	Sequential solvent extraction with sonication; followed by drying over NO_2SO_4 and concentration by Kuderna-Danish evaporation (SOP ADL-2819)	Alumina-preparative chromatography column (SOP ADL-2821)	High Performance Liquid Chromatography (SOP ADL-2821)	GC/MS, Selected Ion Monitoring Mode (SOP ADL-2827)	Acenaphthene- d_{10} Phenanthrene- d_{10} Benzo(a)pyrene- d_{12} Naphthalene- d_4	Fluorene- d_{10} Chrysene- d_{12}
PCBs, Chlorinated Pesticides*	Same	Same	Same	Single-column gas chromatography-electron capture detection (GC-ECD)	4,4'-dibromoocta-fluorobiphenyl (DBOFB) PCB congeners 81,103,198	Tetrachloro-m-xylene (TCMX)

*Refer to Tables 2-2 and 2-3 for lists of individual target PAHs and PCBs/Pesticides, respectively.

2.0 Methods (continued)

congeners 81, 103, and 198 were spiked as surrogates. For PAHs, 0.4 µg each of naphthalene-d₈, acenaphthene-d₁₀, phenanthrene-d₁₀, and benzo(a)pyrene-d₁₂ were spiked as surrogates.

Each sample was shaken vigorously for approximately 30 sec. and then subjected to sequential solvent extraction with sonication to promote solvent/matrix contact and phase partitioning. Resultant extracts were combined for each sample following centrifugation. Each extract was then dried over Na₂SO₄, concentrated by Kuderna-Danish evaporation to less than 10 mL, transferred to a 25-mL vial, and further concentrated by nitrogen evaporation to approximately 1 mL. The total extract weight of each sample was determined at this step and recorded as the Total Extractable Content (TEC).

Next, each extract was loaded onto an alumina preparative chromatography column for cleanup (Arthur D. Little SOP No. ADL-2821). This cleanup procedure is applicable to all of the targeted organic analyte classes. Following the alumina column cleanup, a second extract weight was taken and reported as Total Hydrocarbons (THC). THC is the total extractable content minus polar-organic compounds and provides a measure of total saturated and aromatic hydrocarbon fractions reflecting hydrocarbons of combined petrogenic, pyrogenic, and diagenic origins. Though THC is not a direct indicator of specific hydrocarbon sources, it is still a useful measure to consider because of its wide use in past studies as an indicator of overall hydrocarbon inputs from combined sources.

After the cleanup step, the extracts were fractionated using high performance liquid chromatography (HPLC) as described in Arthur D. Little SOP ADL-2821. This fractionation method produces one fraction that contains PAHs, PCBs, and pesticides. The resulting fractionated extracts were concentrated to approximately 1 mL and spiked with an appropriate amount of recovery standards. The recovery standard used for PCB/pesticide quantification was tetrachloro-*m*-xylene (TCMX), which was spiked into the extracts at approximately 0.1 µg. The PAH recovery standards, fluorene-d₁₀ and chrysene-d₁₂, were spiked at approximately 0.2 µg.

Gas chromatography/mass spectrometry (GC/MS) with selected ion monitoring (Arthur D. Little SOP No. ADL-2827) was used to analyze an aliquot of the extract fraction for each of the target PAHs listed in Table 2-2. An initial calibration was performed using standards at concentrations of approximately 25, 75, 250, 1,250, and 5,000 ng/mL. Results of instrument calibrations and all other quality assurance/quality control (QA/QC) analyses were evaluated with respect to the data quality objectives summarized in Table 2-6. Mean relative response factors for individual target analytes were used to quantify analytes in sample extracts. However, concentrations of target analytes were not adjusted for surrogate recoveries. This reporting convention constitutes a deviation from the above ADL SOP but is the reporting convention requested by EPA/MBP.

Concentrations were reported for THC, total PAHs, and each of the individual PAH analytes (parent compounds and alkylated homologues) listed in Table 2-2.

Table 2-6 Data quality objectives and criteria for sediment chemistry measurements.

QC Sample or Parameter	Frequency	Data Quality Objective/ Acceptance Criteria
<u>Polynuclear Aromatic Hydrocarbons</u>		
Initial Calibration	Prior to every batch sequence	5-point curve. RRF ^b ≤ 30% RSD ^c
Continuing Calibration (Using mid-level calibration standard)	Every 10 field samples or 16 hours, whichever is more frequent	Δ RRF ^d ≤ 30%
Sediment SRM 1941	One per batch	Values must be within ± 30% of true value on average for all analytes; not to exceed ± 35% of true value for individual analytes.
Procedural Blank	One per batch	No more than 2 analytes to exceed 5x target MDL unless analyte not detected in associated sample(s) or analyte concentration > 10x blank value.
Duplicate Sample	One per batch	RPD ^e ≤ 30% for 70% of all analytes
Matrix Spike	One per batch	%R = 60-125%
Surrogate (Internal) Standards	Every Sample	%R = 60-125%
Target MDLs ^a	--	1-5 ng/g (dry weight)
<u>Polychlorinated Biphenyls and Pesticides</u>		
Initial Calibration	Prior to every batch sequence	5-point curve. Standard curve correlation coefficient (r) > 0.9950 for all analytes
Continuing Calibration (Using mid-level calibration standard)	Every 10 field samples or 16 hours, whichever is more frequent	Δ RRF ^d ≤ 25%
Sediment SRM 1941	One per batch	Values must be within ± 30% of true value on average for all analytes; not to exceed ± 35% of true value for individual analytes.
Procedural Blank	One per batch	No more than 2 analytes to exceed 5x target MDL unless analyte not detected in associated sample(s) or analyte concentration > 10x blank value.
Duplicate Sample	One per batch	RPD ^e ≤ 30% for 70% of all analytes
Matrix Spike	One per batch	%R = 60-125%
Surrogate (Internal) Standards	Every sample	%R = 60-125%
Target MDLs ^a	Sediment	0.2 ng/g (dry weight)

Table 2-6. Data quality objectives and criteria for sediment chemistry measurements. (Continued)

QC Sample or Parameter	Frequency	Data Quality Objective/ Acceptance Criteria
Metals		
Initial Calibration	Prior to every batch of samples	5-point curve to assess linearity. RSD ^c for initial and subsequent calibrations must agree to within 2%.
Continuing Calibration	Every 10 samples	See initial calibration
Procedural Blank	One per batch	Not to exceed 3x MDL unless analyte not detected in associated sample(s) or analyte concentration > 10x blank value.
Matrix Spike	One per batch	Method of additions used to assess signal suppression/enhancement
Sample Replicate Analysis	One triplicate analysis per batch	RSD ^c ≤ 25%
Sediment SRM 1646	One per batch	Values must be within ±30% of true value on average for all analytes; not to exceed ±35% of true value for individual analytes.
Target MDLs ^a	--	Concentrations in ppm (µg/g dry weight) by element: Al (8 ppm); Fe (2.5 ppm); Mn (1.0 ppm); As (0.3 ppm); Cd (0.03 ppm by GFAA or 0.3 ppm by ICAP); CR (0.5 ppm); Cu (0.5 ppm); Pb (0.3 ppm); Hg (0.02 ppm); Ni (1.0 ppm); Se (0.3 ppm); Ag (0.02 ppm); Sn (0.8 ppm by GFAA or 8 ppm by ICAP); Zn (2.0 ppm); Sb (0.5 ppm by GFAA or 5.0 ppm by ICAP); Tl (0.5 ppm); Be (0.05 ppm); V (0.5 ppm).

^aMethod Detection Limits

^bRelative Response Factor (RRF) = response of a given target analyte relative to the response of its associated internal standard.

^cRelative Standard Deviation (RSD) = $S/\text{Mean} \times 100\%$

^dChange in RRF (ΔRRF) = $[(\text{RRF of continuing calibration check} - \text{avg. RRF from initial calibration}) \div \text{avg. RRF from initial calibration}] \times 100$.

^eRelative Percent Difference (RPD) = $[\text{Concentration 1} - \text{Concentration 2}] \div \text{Mean Concentration}_{1,2} \times 100$.

2.0 Methods (continued)

Dual-column gas chromatography-electron capture detection (GC-ECD) (Arthur D. Little SOP No. ADL-2818) was used to analyze an aliquot of each extract for the PCB congeners and chlorinated pesticides listed in Table 2-3. An RTX-5 column was used as the primary GC column for quantification; a DB-17 column was used as a secondary column to confirm the identification of analytes. An initial calibration was performed with standards at concentrations of 5, 10, 50, 100, 200, and 400 ng/mL. As with PAHs, results of instrument calibrations and all other QA/QC analyses were evaluated with respect to the data quality objectives summarized in Table 2-6. Concentrations of target analytes were not adjusted for surrogate recoveries.

2.3.2 Metals analyses. A sediment sample from each of the 12 stations was analyzed for the metals listed in Table 2-4. Initially, each sediment sample was thawed and carefully homogenized with a Teflon[®] or plastic mixing rod. One aliquot was removed for Hg analysis; another aliquot was removed for the analysis of remaining metals. An additional aliquot was removed for determination of moisture content to provide information for reporting metal concentrations on a per-dry-weight-sediment basis.

Sample preparation and analysis was based on standard procedures provided in EPA SW-846 (EPA 1986) and the EPA-Contract Laboratory Program (CLP) Statement of Work (EPA 1991) (see Table 2-4). Sample preparation followed Method 7471 (EPA 1986) for Hg and Method 3050 (EPA 1986) for all remaining metals. Both sample preparation methods basically consist of preparing an extract of the sediment by digesting the sample with concentrated acid under heat and then bringing the extract up to a specified volume with deionized water.

Labware used in the digestion process was acid-washed and rinsed with double deionized water. Procedural blanks and replicate samples were prepared with each set of samples. Standard Reference Material #1646 also was prepared and analyzed. Potential matrix interferences were monitored carefully for all elements using the method of standard additions. Results of all QA/QC analyses were evaluated to ensure that they met the data quality objectives summarized in Table 2-6.

Metals in the extracts were analyzed by a combination of instrumentation techniques depending on the individual metals, their concentrations, and potential matrix interferences. Inductively coupled argon plasma (ICAP) atomic absorption spectrophotometry was used to analyze Al, Be, Cd, Cr, Cu, Ni, Sb, Sn, V, Zn, Fe, and Mn. ICAP analysis of these elements followed Method 6010 of the EPA SW-846 series. Graphite furnace atomic absorption (GFAA) spectrophotometry, which has lower detection limits, was used to analyze Ag, As, Se, Pb, and Tl. GFAA analysis of these elements was based on Method 7060 (EPA 1986) for As, Method 7740 (EPA 1986) for Se, Method 7421 (EPA 1986) for Pb, Method 7841 (EPA 1986) for Tl, Method 7131 (EPA 1986) for Cd, and Method 7041 (EPA 1986) for Sb. Analysis of As followed Method 272.2 from the EPA-CLP Statement of Work (EPA 1991). Hg was analyzed by cold-vapor atomic absorption (CVAA) spectrophotometry following Method 7471 from the EPA SW-846 series.

2.0 Methods (continued)

2.3.3 Other supporting chemical and physical measurements. A variety of other chemical and physical properties of sediments, sediment porewaters, and near-bottom water were measured in samples from each of the 12 stations to provide supporting information for the interpretation of the chemical, toxicological, and biological data. A summary of these measurements along with the corresponding methods, performing laboratory, and rationale for their inclusion are provided in Table 2-7.

2.4 Sediment Toxicity Studies

2.4.1 Solid-phase test with amphipods. The purpose of this test was to determine the toxicity of sediment collected from the 12 Massachusetts/Cape Cod Bays sites to amphipods exposed for 10 days under static conditions. Overall test performance was based on the ASTM (1990) protocol for conducting solid-phase toxicity tests with the marine amphipod *Ampelisca abdita*. Tests were conducted with subsamples of the same sediment on which chemical analyses were performed. These tests were performed by T.R. Wilbury Laboratories in Marblehead, Massachusetts.

Test sediments were stored at approximately 4°C in the original shipping container until used for testing. Sediments were not allowed to freeze at any time during storage or transport; daily temperature logs were maintained to verify laboratory storage temperatures. Samples were tested in two separate batches within 12 days of collection. Samples from Stations 1 through 4 (Wellfleet Harbor) and Stations 9 through 12 (Salem/Beverly Harbors) were tested as one batch, and samples from Stations 5 through 8 (Boston Harbor) were tested as a separate batch. An additional negative control (i.e., sediment from the amphipod collection site) was tested along with each batch of Massachusetts/Cape Cod Bay field samples.

Amphipods were acclimated to test conditions for 96 hours before the start of a test. The animals were maintained in 100 percent dilution water (natural seawater filtered at 20 microns) under flow-through conditions in a 20-L glass aquarium that contained a layer of sediment from the collection site. The animals were fed once daily with live marine algae *Skeletonema costatum*. During this acclimation period, the temperature range was 21.2° to 22.4°C, the dissolved oxygen concentration was at least 6.8 mg/L, the salinity range was 32 to 33 ppt, and the pH range was 8.2 to 8.5. No amphipod mortality was noted during the 96-h acclimation period.

One day prior to initiation of the toxicity test, each sample of sediment was thoroughly homogenized, examined to remove ambient organisms (by press sieving on 0.5-mm sieve without the addition of water, or by direct removal if sediments were too coarse for sieving), and then added to test vessels to achieve a depth of approximately 4 cm. A polyethylene sheet was placed on the sediment and the vessel was gently filled to approximately 90 percent of capacity with dilution water. The polyethylene was removed carefully and the vessel was placed in a water bath and aerated overnight.

Table 2-7 Summary of supporting physical/chemical measurements and corresponding methods.

Parameter	Matrix	Units	Methodology	Reference	Performing Laboratory	Assessment Use
Moisture Content	Sediment	Weight %	Dry & weigh	-	ADL	Report Contaminant concentrations on a dry weight basis.
Total Organic Carbon (TOC)	Sediment	Weight %	Acid treatment to remove inorganic carbon followed by combustion at 900°C in Leco carbon analyzer	SOP ADL-2829	Global Geochemistry ^b	1. Measure of organic carbon loading in sediments. 2. Used to normalize organic chemical parameters.
Eh ^a	Sediment	mV	Orion model 9678 BN redox probe	-	ADL	Assess extent of anoxia in subsurface sediment layers.
Grain size	Sediment	Weight % gravel, sand, silt, clay	Sieve and pipette	Folk (1974)	GEO/PLAN ^c	1. Provides information on particle size distribution of sediments and nature of sedimentary environment. 2. Used as a normalization factor for organic compounds and metals. 3. Influence on infaunal distributions.
Salinity ^a	Near-bottom water	ppt	Horiba model U10 water-quality probe system	-	ADL	Influence on biological/toxicological responses.
Temperature ^a	Near-bottom water	°C	Horiba model U10 water-quality probe system	-	ADL	Influence on biological/toxicological responses.

^aIn-situ field measurement

^bGlobal Geochemistry, Canoga Park, CA

^cGEO/PLAN Associates, Hingham, MA

Table 2-7 Summary of supporting physical/chemical measurements and corresponding methods. (continued)

Parameter	Matrix	Units	Methodology	Reference	Performing Laboratory	Assessment Use
pH*	Near-bottom water	-	Horiba model U10 water-quality probe system	-	ADL	Influence on biological/toxicological responses.
Dissolved oxygen(DO)*	Near-bottom water	mg/L (ppm)	Horiba model U10 water-quality probe system	-	ADL	Influence on biological/toxicological responses.
Water Depth*	-	m	Fathometer	-	ADL	Influence on biotic and abiotic variables.
Temperature	Pore water	°C	YSI model 59 DO/ temperature meter	-	Corpus Christi State Univ.	Influence on biological/toxicological responses.
Salinity	Pore water	ppt	Reichert temperature-compensated refractometer	-	Corpus Christi State Univ.	Influence on biological/toxicological responses.
pH	Pore water	pH units	Orion model 290A pH meter	-	Corpus Christi State Univ.	Influence on biological/toxicological responses.
DO	Pore water	mg/L and % saturation	YSI model 59 DO meter	-	Corpus Christi State Univ.	Influence on biological/toxicological responses.
Sulfide	Pore water	mg/L	Orion (model 94-12) ion-selective electrode in conjunction with Orion (model 290A) meter	-	Corpus Christi State Univ.	Influence on biological/toxicological responses.
Ammonia (total ammonia and unionized)	Pore water	mg/L	Orion (model 95-12) ion-selective electrode in conjunction with Orion (model 290A) meter	-	Corpus Christi State Univ.	Influence on biological/toxicological responses.

*In-situ field measurement

2.0 Methods (continued)

Active, apparently healthy juvenile amphipods, *Ampelisca abdita*, were used to initiate the test. The test specimens were from a single lot and of approximately the same size (the mean wet weight of control organisms at the end of the test was 0.8 mg). Identification of the test animals was verified using an appropriate taxonomic key (Bausfield 1973).

The relative sensitivity of these amphipods was evaluated in a static acute toxicity test with the reference toxicant CuCl_2 in seawater without sediment. The 48-h median lethal concentration (LC50) for this toxicant was 0.05 mg/L (95 percent confidence interval = 0.04 to 0.07 mg/L). This value is within the lower end of the range of values reported in the literature for the acute toxicity of inorganic copper to marine crustaceans (EPA 1985).

Treatments for the solid-phase tests consisted of a single concentration of each sample (100 percent sediment) and the negative control. The tests were conducted under static conditions at a target temperature of $20 \pm 1^\circ\text{C}$. Twenty amphipods were indiscriminately distributed to each of five replicates per each treatment. Amphipods were not fed during the test.

Test vessels consisted of 1-L glass jars containing 4 cm of sediment and filled to approximately 90 percent capacity with dilution water (water depth was 10 cm). Test vessels were incubated in a water bath to control temperature. Gentle aeration was supplied to all test vessels except during daily observation periods. Photoperiod was automatically controlled and adjusted to 24 h light and 0 h dark to encourage burrowing. The photoperiod was maintained with cool-white fluorescent lights that provided a light intensity of $3 \mu\text{E}^{-1}\text{m}^{-2}$.

Obvious mortality and sublethal effects (emergence from sediment and inability to burrow when prodded, and obvious changes in appearance or behavior) were recorded at 24-h intervals throughout the tests. Dead animals were removed every 24 h or when first observed. At the conclusion of a test the sediment from each vessel was sieved to remove amphipods. The number of live organisms, the time required for excavated amphipods to burrow into control sediment, and any other abnormal sublethal effects were recorded in each vessel. The tests were conducted within the following acceptability criteria requirements: 10 percent mean control mortality overall and 20 percent control mortality in any vessel with an acceptable temperature range. Mean control survival during the tests was 92 to 95 percent and at least 85 percent survival occurred in each of the control exposures.

Dissolved oxygen concentration was measured in each replicate vessel every day with a YSI (Model 57) meter. Salinity (measured with a Atago S10 refractometer), pH (measured with a Beckman Model pH 12 meter), and temperature (measured with a Beckman Model pH 12 meter) were determined from one replicate of each treatment every day. Temperature also was continuously recorded in a beaker of water incubated among the test vessels. Water quality was stable throughout the 10-day exposure period for all treatments (see Appendix B, Tables B-7 through B-10).

2.0 Methods (continued)

2.4.2 Porewater tests with sea urchins. Two kinds of sediment porewater tests were performed, both using the sea urchin *Arbacia punctulata*. The first was a sperm-cell toxicity test used to determine whether porewater samples from the 12 bay sites caused statistically significant reductions in the fertilization of exposed gametes relative to a control. The second was a morphological development test used to determine whether porewater samples from the 12 sites caused statistically significant developmental effects on exposed embryos relative to a control. Both tests, performed at Corpus Christi State University in Port Aransas, Texas, are based on the EPA standard method given in Weber et al. (1988) but incorporated modifications developed at the Corpus Christi testing facility, as discussed below.

Surficial sediment samples from the 12 sites in Massachusetts/Cape Cod Bays were received in Port Aransas, Texas three to six days after collection. It was noted that the blue ice included with samples from sites 9 through 12 had thawed during shipment and these samples were not chilled upon receipt. All shipments were accompanied by sample tracking sheets, and samples were logged into laboratory sample tracking systems. The samples were either refrigerated (4°C) or processed immediately upon receipt. All porewater samples were extracted 0 to 1 days after receipt.

Porewater was extracted from the sediments using a pressurized squeeze-extraction device. This extractor is made of polyvinyl chloride (PVC) and uses an 8- μ m polyester filter. It is the same device used in previous sediment quality assessment surveys in Tampa Bay, Florida (SAIC 1992, USFWS 1992) and in Galveston Bay, Texas (Carr 1993). The apparatus and extraction procedures are detailed in Corpus Christi SOP F10.9.

Porewater was extracted from eight of the sediment samples the day they were received, and four were held refrigerated (4°C) until processed the following day. After extraction, the porewater samples were centrifuged in polycarbonate bottles at 4,200 g for 15 minutes to remove any suspended particulate material and were then frozen. Two days before the start of a toxicity test the samples were moved from the freezer to a refrigerator (4°C), and one day prior to testing were thawed at room temperature (20°C) or in a tepid water bath. Temperature of the samples was maintained at 20 \pm 1°C. Sample salinity was measured and adjusted to 30 \pm 1 ppt, if necessary, using ultrapure sterile water or concentrated brine (Corpus Christi SOP F10.12). Only three porewater samples from test sites (Stations 10, 11, and 12) with salinities at 32 ppt and the control porewater sample (35 ppt) needed to be diluted to satisfy the test salinity requirement. Other water-quality measurements (DO, pH, sulfide, and ammonia) also were made as described below. Unionized ammonia concentration, expressed as nitrogen (UAN), was calculated for each sample using the respective salinity, temperature, pH, and total ammonia (TAN) values. Any samples containing less than 80 percent DO saturation were gently aerated. Following the water-quality measurements and adjustments the samples were centrifuged, as was done before freezing, and were stored overnight at 4°C but returned to 20 \pm 1°C before the start of the toxicity tests.

2.0 Methods (continued)

The sperm-cell fertilization test was conducted following the procedures outlined in Corpus Christi SOP F10.6, which as mentioned above is a modification of the EPA standard method given in Weber et al. (1988). This is a one-hour test conducted with replication under static conditions at $20\pm 1^{\circ}\text{C}$, using porewater samples from the study sites and additional negative control site. Each of the 12 porewater samples was tested in a dilution series design at 100, 50, 25, and 12.5 percent of the water-quality adjusted sample with 5 replicates per treatment. Dilutions were made with 0.45- μm filtered seawater.

In the EPA procedure (Weber et al. 1988), the sperm density is determined spectrophotometrically. Tests conducted at the Corpus Christi testing facility have shown that if the sperm is collected "dry," which is not done in the EPA method, the sperm remains viable for at least 8 hours when kept on ice. This variation in the EPA method allows a pretest to be conducted with various combinations of eggs and sperm at different dilutions thus optimizing the sensitivity of the test. The best egg/sperm combination is then used in the toxicity test rather than four different arbitrary egg/sperm combinations as recommended in the EPA method.

The release of gametes from adult sea urchins to use in the fertilization tests was facilitated by electrical stimulation in seawater with a 12V transformer. Sperm and eggs were exposed to the replicate treatments in disposable glass scintillation vials, each containing 5 mL of porewater from a test or control site. After a 1-h incubation period, 100 eggs were examined per sample and the percentage of fertilized eggs, defined by the presence of a fertilization membrane surrounding the egg, was recorded as the standard biological endpoint.

A reference porewater sample collected from Redfish Bay, Texas, which had been handled identically to the test samples, was included with each fertilization toxicity test as a negative control. This site is far removed from any known sources of contamination and has been used previously as a reference site (Carr and Chapman 1992, Carr 1993). In addition, a dilution-series test with sodium dodecyl sulfate (SDS) was included as a positive control. This test resulted in an LC50 of 2.72 mg SDS/L (95 percent confidence interval of 2.51 to 2.94 mg/L). This value is within the range of acceptable values that have been established previously in the Corpus Christi laboratory for the sea urchin fertilization test with SDS.

The sea urchin embryological development test followed the procedures outlined in Corpus Christi SOP F10.7, which also is based on the EPA standard method (Weber et al. 1988). Each of the 12 porewater samples was tested in a dilution-series design with 5 replicates per treatment. In a preliminary trial, all 100 percent and 50 percent dilutions of the water-quality adjusted samples were toxic, so the test was run again with dilutions of 50, 25, 12.5, and 6.25 percent. Procedures for obtaining sperm and eggs were identical to those used in the fertilization tests. Reference porewater (negative control) and a positive control test with the reference toxicant SDS also were included as described for the fertilization test.

2.0 Methods (continued)

The tests were conducted for 48 h under static conditions at $20\pm 1^{\circ}\text{C}$. Sperm and eggs were exposed to the replicate treatments in disposable glass scintillation vials, each containing 5 mL of porewater from a test or control site. After the 48-h incubation period, 100 eggs/embryos were removed per sample and the following measurements recorded: number of normally developed echinoplutei, number of echinoplutei with abnormalities, number of embryos arrested in earlier developmental stages, and number of unfertilized eggs. The percentage of normal echinoplutei relative to the total number of eggs/embryos were recorded for each sample as the biological endpoint.

The SDS positive control resulted in an LC50 of 6.74 mg SDS/L (95 percent confidence interval of 6.42 to 7.07 mg/L). This value is within the range of acceptable values that had been established previously in the Corpus Christi laboratory for the sea urchin embryological development test with SDS.

Water-quality measurements were obtained in each treatment at the beginning of the fertilization and embryological development tests. Salinity was recorded to the nearest 1 ppt with a temperature-compensated Reichert refractometer; pH was recorded to the nearest 0.01 pH unit with an Orion (Model 290A) pH meter; DO concentration and percent saturation were recorded to the nearest 0.1 mg/L and 1 percent, respectively, with a YSI (Model 59) DO meter; ammonia was recorded to the nearest 0.01 mg/L with an Orion ion-selective electrode (Model 95-12) in conjunction with an Orion (Model 290A) meter; sulfide was recorded to the nearest 0.01 mg/L with an Orion ion-selective electrode (Model 94-12) in conjunction with an Orion (Model 290A) meter; and temperature was recorded to the nearest 0.1°C with a YSI (Model 59) DO/temperature meter.

2.5 Infaunal Community Analysis

Macroinfauna (animals > 0.5 mm) were identified and enumerated from each of the three replicate, 0.1-m^2 samples collected at each of the 12 Massachusetts/Cape Cod Bay stations. These data were used to characterize the infaunal assemblages at each of the stations and to provide a basis for identifying sites where high sediment contamination and/or toxicity are linked potentially to the presence of degraded benthic communities. All benthic samples were processed at Ruff Systematics in Solana Beach, California.

Once samples were received in the laboratory, they were immediately transferred from 10 percent buffered formalin to 70 percent ethanol. Animals were separated from debris in each of the samples under a dissecting microscope. A biological stain (rose bengal) was added to facilitate sorting. Sorted specimens were identified to the lowest possible taxon, i.e., the species level wherever possible. As species were identified, and the number of individuals per each species recorded, they were placed back in 70 percent ethanol and archived permanently by species. A list of the numbers of individuals per each species was recorded for each replicate sample (Appendix B, Table B-13).

2.0 Methods (continued)

Re-checks of sorted benthic material were performed on 10 percent of the infaunal samples (i.e., 4 of the 36 samples originally sorted) as a QA/QC measure. Per each batch of eight samples processed, one sample was re-checked for this purpose. If in this sample the sorter failed to remove at least 95 percent of the animals originally present, a second sample from this same batch of eight samples was re-checked. If this second sample also failed the QC, then all remaining samples in this same batch were re-checked as well. This same process was repeated for each batch of eight samples. Because a senior-level taxonomist performed the original sorts, the sorting accuracy was within the acceptable range in all cases. Taxonomic identifications were verified by comparing samples against archived voucher specimens available at Ruff Systematics.

2.6 Data Analysis

Concentrations and compositional patterns of hydrocarbons, chlorinated pesticides, PCBs, and metals in each of the sediment samples (one sample per station) were examined graphically and by straightforward data comparisons in an effort to identify potential contaminant sources and to evaluate whether concentrations at any of the sites were at levels known to cause adverse effects on marine organisms. The latter reported bioeffects data, used for comparison with the present sediment chemistry data, consisted of the Effects Range-Low (ER-L) and Effects Range-Median (ER-M) values of Long and Morgan (1990) and the No-Observed Effect Level (NOEL) and Probable Effects Level (PEL) values of MacDonald (1992). The strength and direction of association between the various chemical variables and toxicological and biological endpoints were also examined by correlation analysis.

Among-site differences in replicated toxicological and biological variables were examined statistically by ANOVA and post-hoc pair-wise comparison tests. Prior to running these analyses, the data were first tested for violations of assumptions of normality and homogeneity of variances. The Shapiro-Wilks test and Bartlett's test were used to determine conditions of normality and homogeneity of variances, respectively. Data transformations were used wherever necessary to help establish normality and homogeneity of variances: arcsines or arcsine square-roots were used for % survival and sublethal responses for amphipods and sea urchins; and \log_{10} , or $\log_{10}(X + 1)$, transformations were used for benthic infaunal variables.

For normal data, mean differences were tested with the parametric ANOVA F test. A-posteriori identification of where significant mean differences existed was determined by the Tukey-Kramer Honestly Significant Difference (HSD) test (Tukey 1953, Sokal and Rohlf 1981). This is a post-hoc, pair-wise comparison test recommended for normally distributed data with equal variances (Day and Quinn 1989). The Games-Howell test (Games and Howell 1976, Sokal and Rohlf 1981) has been recommended as a pair-wise comparison test for the case of normal data with unequal variances (Day and Quinn 1989), however none of the data in the present study displayed this condition (at least once appropriate transformations were made). For the case of nonnormal data, the Kruskal-Wallis nonparametric ANOVA (Sokal and Rohlf 1981) and Scheffé's pair-wise comparison test (Scheffé 1953) were used.

2.0 Methods (continued)

Scheffé's test is robust to both nonnormality and unequal variances and is not limited to sample size.

Toxicological response variables were analyzed to determine whether any of the sediment and porewater samples from the 12 study sites were toxic to populations of amphipods and sea urchins in comparison to the corresponding negative controls. As an initial step, statistical differences among treatments (12 test sediments plus control) were determined using one-way ANOVA on percent survival (for amphipods) and other sublethal variables (for sea urchins) transformed usually to arcsine square-roots, if necessary, to establish conditions of normality and homogeneity of variances (an arcsine transformation was required in one case). Following the ANOVA, a Dunnett's one-tailed test (Dunnett 1955, 1985; Zarr 1974; Miller 1981) was used to identify which test samples were different in comparison to the corresponding controls. Prior to running these analyses, the data were tested for normality and homogeneity of variances using the procedures described above. The relationship between sediment toxicity and other measured environmental variables was examined by correlation analysis.

The trimmed Spearman-Kärber method with Abbott's correction (Hamilton et al. 1977) was used to calculate EC_{50} and LC_{50} values for the dilution-series tests with sea urchin gametes and embryos. Data points defined as outliers ($>\pm 4$ SD from the mean of the other replicates within the treatment) were excluded from the data sets prior to statistical analyses.

Response variables examined as part of the benthic community assessment consisted of numbers of species and individuals (all species combined); the Shannon information function, H' (Shannon and Weaver 1949); the associated evenness component, J' (Pielou 1966); and densities of individual dominant species. Base 2 logarithms were used to calculate H' . Relative differences in percent taxonomic composition (on the basis of both abundance and numbers of species) among the various stations were compared as well.

Among-site differences in selected benthic response variables were tested statistically as a two-stage procedure consisting of a one-way ANOVA F test, or the distribution-free Kruskal-Wallis test in the nonparametric case, followed by the appropriate pair-wise comparison test as described above (either Tukey's HSD or Scheffé's test). These tests were run with replication ($n = 3$) and with site as a main effect.

Numerical classification, or "cluster analysis," also was used to help identify patterns of faunal similarity among the different stations. Normal (Q mode) numerical classification (Boesch 1977) was performed on $\log_{10}(X + 1)$ transformed data. Group-average sorting (= unweighted pair-group method; Sneath and Sokal 1973) was used as the clustering method and Bray-Curtis similarity (Bray and Curtis 1957) was used as the resemblance measure. Results were expressed in the form of a dendrogram in which samples were ordered into groups of increasingly greater similarity based on resemblances of component-species abundances. Thus samples clustered closely together display greater similarities than samples spaced further

2.0 Methods (continued)

apart and the degree of separation can be used to depict spatial differences due to some environmental factor or combination of factors.

Correlation analysis was used to examine the strength and direction of association between the various chemical, toxicological, and biological variables. Correlation analysis was based on Pearson's product-moment correlation coefficient, r . The significance of the correlations, i.e., the null hypothesis that any two variables are not correlated ($H_0: r = 0$), was tested as a t -test, with $n-2$ df , and with the significance level adjusted to correct for experimentwise error rate using the Dunn-Sidak method (Ury 1976, Sokal and Rohlf 1981).

A final data analysis approach was the use of straightforward data comparisons coupled with simple graphics to help interpret one set of data in light of those from the other study components. Similarly, information on bioeffect concentrations of contaminants from past studies was considered, as mentioned above, to help in interpreting the relationships between chemical, toxicological, and biological data in the present study.

3.0 Results and Discussion

3.1 Supportive Chemical and Physical Properties of Sediments, Porewater, and Near-Bottom Water

Water quality and other supporting physical measurements made on sediment and near-bottom water samples are summarized in Table 3-1. Depths at MLLW ranged from 0.30 to 5.1 m. DO, temperature, salinity, and pH in near-bottom water ranged from 6.79 to 11.57 mg/L, 16.3° to 24.5°C, 29.6 to 30.9 ppt, and 7.44 to 7.90 respectively. None of the DO values were below the Massachusetts standard of 6 mg/L (Code of Massachusetts Regulations, Title 314). Sediment temperatures (upper 8 cm) were typically 1° to 7°C cooler than near-bottom water temperatures at corresponding stations, with the exception of the shallower Wellfleet Harbor stations, which had slightly higher temperatures in both sediments and near-bottom water. Sediment TOC ranged from 0.23 to 4.29 percent and showed wide variations within two of the three harbors (e.g., Wellfleet Harbor Station had the lowest and the second highest TOC concentrations; Salem/Beverly Harbor had the highest and second lowest TOC concentrations). Wellfleet Harbor sites had coarser sediment than the other two harbor systems. Percentages of sand ranged from 84 to 97 percent in Wellfleet Harbor and from 11 to 72 percent in Boston and Salem/Beverly Harbors.

Sediment redox data are presented in Appendix B, Table B-14. Though these data displayed a large degree of variability, some trends were evident. Most stations, with the exception of Stations 8, 10, and 12, showed a preponderance of negative mV values among the various replicates and depth intervals, reflecting the presence of anaerobic/reducing environments. Several stations showed logical patterns of decreasing (more negative) mV values with increasing sediment depth. Station 1 in inner Wellfleet Harbor had the most anaerobic sediments (lowest mV values) and Station 12 off Marblehead had the most aerobic sediments (highest mV values).

Water-quality measurements for porewater samples are summarized in Table 3-2. Initial salinity of porewater from the 12 test sites ranged from 29 to 32 ppt and the reference porewater salinity was 35 ppt. Porewater dissolved oxygen concentrations were above 88 percent saturation for all samples. Values for pH ranged from 7.26 to 7.76 in the test samples and pH was 8.20 in the control sample. Porewater concentrations of sulfide and ammonia were high at some sites. The concentration of total sulfide (H_2S , HS^- , and S^{2-}), for example, was at or above the detection limit of 0.01 mg/L in samples from all Wellfleet Harbor sites and was especially high (0.42 mg/L) at the innermost Station 1. Past studies with the sea-urchin toxicity tests generally have shown significant impacts at sites where total sulfide concentrations in porewater are ≥ 0.01 mg/L (S. Carr, personal communication). Given a total sulfide concentration at Station 1 of 0.42 mg/L, the concentration of toxic unionized H_2S is estimated to be 130 $\mu g/L$, which greatly exceeds the EPA Water Quality Criteria value of 2.0 $\mu g/L$ (EPA 1976). Porewater concentrations of unionized H_2S exceed the EPA criterion for all Wellfleet Harbor sites (Table 3-2). Because of the volatility of H_2S , this is likely to be a highly conservative estimate of the actual unionized H_2S concentration in ambient porewater. Measurements in the present study were made in laboratory extracted samples.

Table 3-1 Supporting water quality and sediment data by station (NB = near-bottom water; SED = sediment).

Station	Depth ¹ (m)	DO-NB (mg/L)	T-NB (°C)	T-SED ² (°C)	Salinity -NB (ppt)	pH-NB	SED TOC (% Wt)	% Gravel	% Sand	% Silt	% Clay
1	0.30	6.79	24.5	23.5	29.9	7.63	2.93	6.5	85.8	4.5	3.2
2	0.49	7.23	21.9	22.0	30.9	7.44	0.23	0.4	96.7	1.4	1.5
3	2.3	7.52	22.0	22.1	30.9	7.84	1.47	1.4	84.1	8.8	5.7
4	2.8	7.06	22.2	22.3	30.8	7.84	0.87	1.6	90.9	4.5	3.0
5	4.7	9.27	18.8	14.4	30.3	7.75	2.12	0	29.6	44.6	25.8
6	2.8	8.48	20.3	17.2	29.6	7.49	2.48	0	36.6	46.7	16.7
7	3.1	8.31	21.1	17.8	30.6	7.70	1.96	0.4	36.0	49.4	14.1
8	2.4	8.67	19.2	17.1	30.8	7.53	1.69	0.2	58.1	27.2	14.4
9	3.3	10.60	16.3	12.2	30.4	7.90	2.07	10.8	67.0	15.1	7.1
10	5.1	10.60	18.8	11.9	29.9	7.50	0.87	0.4	51.4	34.8	13.4
11	3.6	9.27	20.9	13.8	29.9	7.80	4.29	0	11.0	84.0	4.9
12	4.3	11.57	16.4	12.8	29.9	7.80	1.81	0	72.3	19.4	8.2

¹Depth in meters at MLLW

²Temperature in upper 8 cm of sediment; average of 4 measurements at 2-, 4-, 6-, 8-cm depth intervals

Table 3-2 Water quality parameters after salinity adjustment and original salinity of sediment porewater samples.

Site	Salinity ¹ (ppt)	DO ² (mg/L)	% DO	pH	TAN ³ (mg/L)	UAN ⁴ (µg/L)	Total Sulfide (mg/L) ⁵	Unionized H ₂ S (µg/L) ⁶	% OUS ⁷
1	29	7.5	97	7.36	54.6	402	0.42	130	100
2	31	7.5	97	7.26	11.3	66	0.01	3.6	100
3	31	7.4	96	7.46	14.7	136	0.01	2.6	100
4	30	6.9	90	7.32	10.0	67	0.01	3.3	100
5	31	7.8	100	7.62	12.0	160	<0.01	ND	100
6	31	6.8	88	7.72	6.24	104	<0.01	ND	100
7	31	7.1	92	7.66	4.76	69	<0.01	ND	100
8	31	7.2	93	7.69	12.0	187	<0.01	ND	100
9	31	7.0	91	7.44	6.93	61	<0.01	ND	100
10	32	7.7	100	7.70	31.2	498	<0.01	ND	94.0
11	32	7.6	97	7.76	33.3	609	<0.01	ND	94.0
12	32	7.6	98	7.71	13.8	225	<0.01	ND	94.0
REF ⁸	35	7.2	93	8.20	0.61	30	<0.01	ND	85.7

¹Salinity of sample prior to adjustment. Samples adjusted to 30 ± 1‰.

²Dissolved oxygen.

³Total ammonia expressed as nitrogen.

⁴Unionized ammonia expressed as nitrogen (approximate values calculated from ammonia equilibrium equation).

⁵Total sulfide: $H_2S = HS^- + S^{-2}$

⁶Unionized H₂S, (approximate values calculated from sulfide equilibrium equation).

⁷Percent of original unadjusted sample after salinity adjustment.

⁸Reference porewater extracted from sediment collected near Lydia Ann Channel in Redfish Bay, Texas.

3.0 Results and Discussion (continued)

Total ammonia (dissolved NH_3 and NH_4^+) in control porewater was 0.61 mg/L and in the test samples ranged from 4.76 to 54.6 mg/L. Unionized ammonia (UAN, the dissolved NH_3 portion) in the control was 30 $\mu\text{g/L}$ and ranged from 61 to 609 $\mu\text{g/L}$ in the test samples. The UAN LOEC determined in the Corpus Christi toxicity testing facility is 800 $\mu\text{g/L}$ for the fertilization test and 90 $\mu\text{g/L}$ for the embryological development test. These LOECs were not exceeded in any of the 12 samples for the fertilization test but were exceeded in 5 of the 12 samples (Stations 1, 8, 10, 11, and 12) for the embryological development test in the 50 percent dilution treatment. Some caution must be exercised, however, in comparing porewater concentrations to LOEC derived from water-column toxicity tests, which usually are conducted, as required by most testing protocols, at TOC levels that are considerably lower than levels commonly found in sediment porewater. The higher TOC levels in porewater indicate the presence of organic compounds that could reduce the bioavailability of any chemical contaminants that are present.

3.2 Sediment Contaminants

Raw data on concentrations of sediment hydrocarbons, metals, PCBs, and chlorinated pesticides are presented in Appendix B, Tables B-1, B-2, B-3, and B-4 respectively. Bar graphs of the concentrations of these contaminants plotted by station are given in Appendix C. Plots of individual PAH distributions in each sediment sample are given in Appendix D.

Measured concentrations of key contaminants were compared to the No-Observed-Effect Level (NOEL) and Probable Effects Level (PEL) of MacDonald (1992), and to the Effects Range-Low (ER-L) and Effects Range-Median (ER-M) values of Long and Morgan (1990), as a means of evaluating whether these contaminants were present in sediments at concentrations known to cause adverse effects on marine organisms (Table 3-3). The NOEL and ER-L values are similar in range (usually within a factor of two of one another) and provide estimates of the maximum concentration at which no effect is observed and the lowest concentration at which adverse biological effects on some marine organisms (presumably the most sensitive ones) are observed, respectively. The PEL and ER-M values are also similar in range (usually within a factor of two of one another) and indicate the concentration above which bioeffects are expected to occur on a wider variety of benthic organisms. Only contaminants that have reported NOEL, PEL, ER-L, or ER-M values are included in Table 3-3.

All contaminants (or summed contaminant parameters) except acenaphthylene, naphthalene, 2-methylnaphthalene, Cd, Sb, and Ni exceeded the corresponding NOEL or ER-L value (lower of the two) at one or more stations (Table 3-3). The magnitude of these exceedances is summarized by contaminant and station in Table 3-4. Stations 1 through 4 in Wellfleet Harbor were the cleanest sites. There were no exceedances at Station 2, and only one contaminant (chlordane) exceeded the NOEL/ER-L value at each of Stations 3 and 4. In both cases, these exceedances were small (within a factor of two of the NOEL/ER-L value). The most contaminated of the four sites in Wellfleet Harbor was the innermost harbor

Table 3-3 No-observed-effect level and probable effects level (NOEL and PEL, respectively; MacDonald 1992) and effects range-low and effects range-median (ER-L and ER-M, respectively; Long and Morgan 1990) for key contaminants. Exceedances of the lower NOEL or ER-L values by station are also shown.

Contaminant	NOEL	PEL	ER-L	ER-M	Stations exceeding NOEL or ER-L ¹
PCBs (ng/g)					
Total PCBs	24	260	50	400	5(63), 6(92), 7(105), 8(55), 9(58), 10(47), 11(mean of 52)
Chlorinated Pesticides (ng/g)					
Total Chlordane ²	-	-	0.5 ³	6.0 ³	1(1.3), 3(0.66), 4(0.58), 5(3.4), 6(6.6), 7(4.2), 8(4.8), 9(4.3), 10(6.2), 11(8.8), 12(2.5)
Dieldrin	-	-	0.02	8	1(0.08), 5(0.63), 8(0.19), 10(0.65)
Total DDT	4 to 5	270	1.0	7.0	5(11.1), 7(2.5), 8(3.8), 10(2.0), 11(mean of 2.3), 12(2.2)
Total DDE	1.7	130	2.0	15.0	5(2.7), 6(3.4), 7(4.5), 8(8.9), 9(2.9), 10(6.4), 11(mean of 6.1), 12(4.0)
Total DDD	-	-	2.0	20.0	5(12), 6(14), 7(13), 8(9.1), 9(20), 10(71), 11(mean of 27), 12(14)
Total DDT, DDE, DDD	-	-	3.0	350	5(26), 6(17), 7(20), 8(22), 9(24), 10(79), 11(mean of 35), 12(20)
Polynuclear Aromatic Hydrocarbons (ng/g)					
Acenaphthene	22	450	150	650	5(23), 9(35)
Anthracene	85	740	960	960	5(115), 9(340), 11(mean of 93)
Fluorene	18	460	35	640	5(42), 6(24), 8(19), 9(107), 10(29), 11(mean of 39), 12(21)
Naphthalene	130	1100	340	2100	None

¹Measured concentration in parentheses. Units for contaminants: pesticides, PCBs, and PAHs (ng/g); metals (µg/g).

²Includes heptachlor, heptachlor epoxide, cis-chlordane, and transnonachlor.

³Component compounds not identified.

Table 3-3 No-observed-effect level and probable effects level (NOEL and PEL, respectively; MacDonald 1992) and effects range-low and effects range-median (ER-L and ER-M, respectively; Long and Morgan 1990) for key contaminants. Exceedances of the lower NOEL or ER-L values by station are also shown. (continued)

Contaminant	NOEL	PERL	ER-L	ER-M	Stations exceeding NOEL or ER-L ¹
Polynuclear Aromatic Hydrocarbons (ng/g) (continued)					
Phenanthrene	140	1200	225	1280	5(338), 6(216), 7(169), 8(170), 9(906), 10(251), 11(mean of 369)
Benz(a)anthracene	160	1300	230	1600	5(353), 6(282), 7(185), 8(170), 9(793), 10(277), 11(mean of 399), 12(205)
Benzo(a)pyrene	230	1700	400	2500	5(389), 6(329), 9(865), 10(330), 11(mean of 461), 12(230)
Chrysene	220	1700	400	2800	5(346), 6(273), 9(762), 10(304), 11(mean of 437)
Dibenzo(a,h)-anthracene	31	320	60	260	5(68), 6(61), 7(45), 8(34), 9(144), 10(62), 11(mean of 86), 12(39)
Fluoranthene	380	3200	600	3600	5(641), 6(465), 9(1648), 10(541), 11(mean of 828), 12(418)
Pyrene	290	1900	350	2200	5(533), 6(428), 7(316), 9(1339), 10(449), 11(mean of 647), 12(344)
Σ PAHs	-	-	4000	35000	5(6855), 6(5353), 9(15511), 10(5671), 11(mean of 8009)
Metals (µg/g)					
As	8	64	33	85	5(13), 6(14), 10(13), 11(16)
Cd	1	7.5	5	9	None
Ag	0.5	2.5	1	2.2	5(24), 6(3.4), 7(3.3), 8(mean of 2.9), 9(0.58), 10(1.2), 12(0.57)
Sb	-	-	2	25	None
Cr	33	240	80	145	5(118), 6(117), 7(81), 8(mean of 74), 9(363), 10(705), 11(718), 12(345)
Cu	28	170	70	390	5(52), 6(96), 7(54), 8(mean of 43), 10(39), 11(57)

Table 3-3 No-observed-effect level and probable effects level (NOEL and PEL, respectively; MacDonald 1992) and effects range-low and effects range-median (ER-L and ER-M, respectively; Long and Morgan 1990) for key contaminants. Exceedances of the lower NOEL or ER-L values by station are also shown. (continued)

Contaminant	NOEL	PERL	ER-L	ER-M	Stations exceeding NOEL or ER-L ¹
Metals (µg/g) (continued)					
Pb	21	160	35	110	1(57), 5(59), 6(94), 7(67), 8(mean of 54), 9(58), 10(80), 11(145), 12(57)
Hg	0.1	1.4	0.15	1.3	5(0.41), 6(0.67), 7(0.65), 8(mean of 0.30), 9(0.44), 10(0.53), 11(0.80), 12(0.40)
Ni	-	-	30	50	None
Zn	68	300	120	270	5(110), 6(136), 7(94), 8(mean of 84), 9(77), 10(108), 11(133), 12(81)
Be	-	-	-	-	None
Mn	-	-	-	-	None
Se	-	-	-	-	None
Tl	-	-	-	-	None

3.0 Results and Discussion (continued)

Station 1. Three contaminants (dieldrin, chlordane, and Pb) exceeded the corresponding NOEL/ER-L values at this site by factors of 4.0, 2.6, and 2.7 respectively.

The most contaminated sites were in Boston and Salem/Beverly Harbors (Table 3-4). The number of contaminants that exceeded the corresponding NOEL or ER-L value ranged from 25 to 16 in Boston Harbor and from 23 to 16 in Salem/Beverly Harbors. The magnitude of these exceedances was larger at sites in Salem/Beverly Harbors. For example, overall the amounts by which contaminants exceeded their corresponding NOEL/ER-L values were usually the largest at Stations 10 and 11 in Salem Harbor, due mostly to high concentrations of pesticides and metals. Station 9 in Beverly Harbor had the highest concentrations of total and individual PAHs. The most contaminated site in Boston Harbor was Station 5 in the vicinity of Deer Island.

The chemicals of possible concern in Wellfleet Harbor are dieldrin, chlordane, and Pb. However, none of these contaminants exceeded the higher PEL or ER-M values at these sites. Among Boston Harbor and Salem/Beverly Harbor sites, all contaminants except acenaphthylene, naphthalene, 2-methylnaphthalene, Cd, Sb, and Ni were present at concentrations that could pose a biological threat. However, the higher PEL and/or ER-M values were exceeded only by Ag (Stations 5, 6, 7, 8) Cr (Stations 9, 10, 11, 12), Pb (Station 11), chlordane (Stations 6, 10, and 11), DDT (Station 5), and DDD (Stations 9, 10, 11). PCBs, dieldrin, total DDT, Ag, Cu, and Zn were present typically at higher concentrations among Boston Harbor sites than among Salem/Beverly sites. In comparison, acenaphthene, anthracene, fluorene, phenanthrene, benz(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, pyrene, total PAHs, chlordane, total DDE, total DDD, Σ DDT+DDE+DDD, As, Cr, Pb, and Hg were usually present at higher concentrations among Salem/Beverly Harbor sites.

The toxicity imparted by PAHs is influenced by the composition as well as the concentration of the PAH assemblage. In a recent similar investigation of sediment toxicity in the Delaware Estuary, a significant positive correlation was found between sediment toxicity (10-day solid-phase toxicity test with *Ampelisca abdita*) and PAH concentrations (Costa and Sauer 1994). Significant sediment toxicity was primarily found along a portion of the Delaware River where the sediment PAH assemblages were dominated by petroleum-derived PAHs (presumably related to the operation of the numerous oil refineries along that portion of the Delaware River, commonly referred to as "Refinery Row"). The PAH assemblages at all 12 stations sampled in this Massachusetts Bays Program study are dominated by the 4- to 6-ringed PAHs (refer to PAH distributions presented in Appendix D). These PAH distributions indicate that the PAH loading at the study sites is primarily of pyrogenic origin. Combustion-derived PAHs (e.g., fluoranthene, pyrene, chrysene) are widely found as background contaminants in estuarine sediments near urban, industrial areas. Thus, exceedances of NOEL/ER-L values (refer to Table 3-4) by 2- and 3-ringed PAHs that are primarily derived from petroleum sources (i.e., acenaphthene, fluorene) should be viewed cautiously. Given the presence of the more abundant pyrogenic PAHs (including phenanthrene, anthracene, and their alkylated homologues, which derive

Table 3-4. Magnitude of NOEL/ER-L exceedances by contaminant and station. Zeros indicate that NOEL/ER-L value was not exceeded.

Chemical Parameter	1	2	3	4	5	6	7	8	9	10	11	12	Σ Sta 1-4	Σ Sta 5-8	Σ Sta 9-12
PCBs	0	0	0	0	2.6	3.8	4.4	2.3	2.4	2.0	2.2	0	0	13.1	6.6
Acenaphthene	0	0	0	0	1.1	0	0	0	1.6	0	0	0	0	1.1	1.6
Acenaphthylene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anthracene	0	0	0	0	1.4	0	0	0	4.0	0	1.1	0	0	1.4	5.1
Fluorene	0	0	0	0	2.3	1.3	0	1.1	5.9	1.6	2.2	1.7	0	4.7	11
Naphthalene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2-Methyl naphthalene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Phenanthrene	0	0	0	0	2.4	1.5	1.2	1.2	6.5	1.8	2.6	0	0	6.3	11
Benz(a)anthracene	0	0	0	0	2.2	1.8	1.2	1.1	5.0	1.7	2.5	1.3	0	6.3	11
Benzo(a)pyrene	0	0	0	0	1.7	1.4	0	0	3.8	1.4	2.0	1.0	0	3.1	8.2
Chrysene	0	0	0	0	1.6	1.2	0	0	3.5	1.4	2.0	0	0	2.8	6.9
Dibenzo(a,h)anthracene	0	0	0	0	2.2	2.0	1.5	1.1	4.6	2.0	2.8	1.3	0	6.8	11
Fluoranthene	0	0	0	0	1.7	1.2	0	0	4.3	1.4	2.2	1.1	0	2.9	9.0
Pyrene	0	0	0	0	1.8	1.5	1.1	0	4.6	1.5	2.2	1.2	0	4.4	9.5
Σ PAHs	0	0	0	0	1.7	1.3	0	0	3.9	1.4	2.0	0	0	3.0	7.3
Dieldrin	4.0	0	0	0	32	0	0	9.5	0	33	0	0	4	42	33
Total Chlordane	2.6	0	1.32	1.16	6.8	13	8.4	9.6	8.6	12	18	5.0	5.1	38	44
Total DDT	0	0	0	0	11	0	2.5	3.8	0	2.0	2.3	2.2	0	17.3	7.5
Total DDE	0	0	0	0	1.6	2.0	2.6	5.2	1.7	3.8	3.6	2.4	0	11	12
Total DDD	0	0	0	0	6.0	7.0	6.5	4.6	10	36	14	12	0	24	72
Total DDT, DDE, DDD	0	0	0	0	8.7	5.7	6.7	7.3	8.0	26	12	6.7	0	28	53
As	0	0	0	0	1.6	1.8	0	0	0	1.6	2.0	0	0	3.4	3.6
Cd	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ag	0	0	0	0	2.4	3.4	3.3	2.9	1.2	1.2	0	1.14	0	12	1.2
Sb	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cr	0	0	0	0	3.6	3.5	2.5	2.2	11	21	22	10	0	12	64
Cu	0	0	0	0	1.9	3.4	1.9	1.5	0	1.4	2.0	0	0	8.7	3.4
Pb	2.7	0	0	0	2.8	4.5	3.2	2.6	2.8	3.8	6.9	2.7	2.7	13	16
Hg	0	0	0	0	4.1	6.7	6.5	3.0	4.4	5.3	8.0	4.0	0	20	22
Ni	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Zn	0	0	0	0	1.6	2.0	1.4	1.2	1.1	1.6	2.0	1.2	0	6.2	5.9
Number of contaminants that exceeded corresponding NOEL or ER-L value	3	0	1	1	25	21	16	17	21	23	22	16			
Σ Exceedance Factors	9.3	0	1.3	1.2	107	70	55	60	98	165	117	54			

3.0 Results and Discussion (continued)

from both petrogenic and pyrogenic sources), acenaphthene, fluorene (in particular) and other petrogenic-sourced PAHs in general are not likely causes of the observed bioeffects. The naphthalene fraction, which is often associated with adverse effects on the benthos, was at concentrations below NOEL/ER-L values at all sites.

Within each harbor system, the stations targeted as reference sites were either the least or second least contaminated site with respect to overall contaminations (Table 3-4). For example, in Wellfleet Harbor, the outermost Station 4 had only one small NOEL/ER-L exceedance (chlordan). In Boston Harbor, Station 8 in Hull Bay had the second lowest frequency of exceedances (17 out of 31 contaminants). The overall magnitude of these exceedances were also the second lowest at this site. Among Salem/Beverly Harbor sites, Station 12 outside of the harbor had the lowest frequency and magnitude of exceedances.

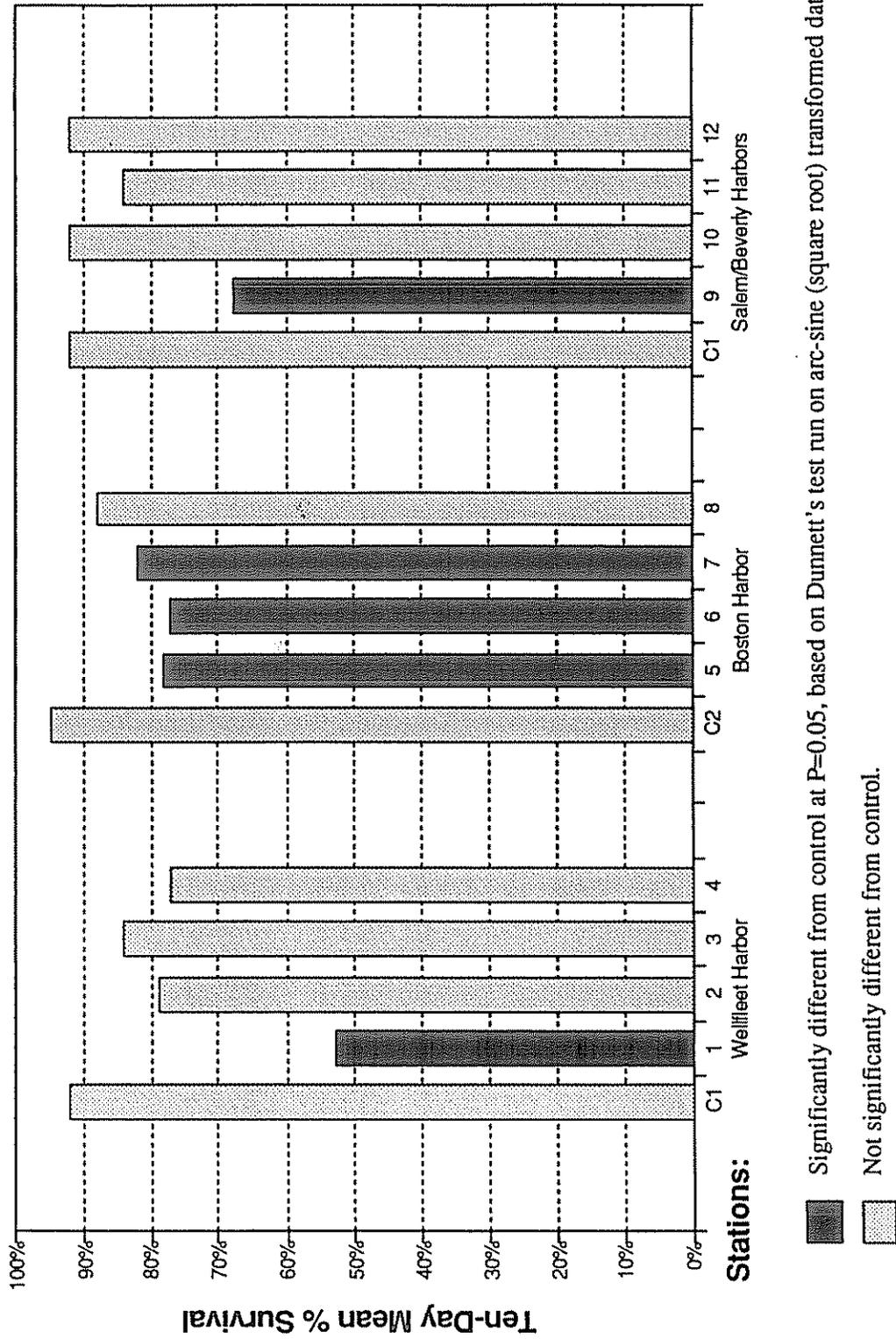
3.3 Sediment Toxicity

3.3.1 Adverse effects of sediment on survival of the amphipod *Ampelisca abdita*. Raw data from analytical toxicity tests are given in Appendix B, Table B-5 (for overall 10-day exposure results) and Table B-6 (daily survival). Daily records of DO, salinity, temperature, and pH are presented in Appendix B in Tables B-7, B-8, B-9 and B-10, respectively.

Results of ANOVA revealed a significant difference (at $P = 0.05$) in mean percent survival of amphipods among the various sediment treatments, consisting of the 12 Massachusetts/Cape Cod Bays stations and negative controls [For C1 versus Stations 1 through 4 and 9 through 12: calculated F ratio = 5.889; critical F value at 8,36 $df = 2.27$. For C2 versus Stations 5 through 8: calculated F ratio = 4.306; critical F value at 4,20 $df = 2.87$]. Dunnett's test (Figure 3.1) showed that sediments from five stations--Station 1 in Wellfleet Harbor, Stations 5 through 7 in Boston Harbor, and Station 9 in Beverly Harbor--had significantly higher mortality of amphipods (at $P = 0.05$) in comparison to corresponding controls. Mean survival of amphipods after 10 days in sediments from these five sites ranged from 53 percent to 82 percent while mean control survival ranged from 92 percent to 95 percent. Lowest survival occurred in sediment from Station 1 in inner Wellfleet Harbor. No sublethal effects (such as emergences from sediment, inability to burrow when prodded, or other obvious changes in appearances or behavior) were noted in any of the sediments.

3.3.2 Adverse effects of sediment porewater on gamete fertilization and embryological development of the sea urchin *Arbacia punctulata*. The sea urchin fertilization and embryological development tests were both performed using the same test and control porewater samples. Water-quality measurements made on these samples are discussed above in Section 3.1.

Raw data and means from the fertilization and embryological development tests are given in Appendix B, Tables B-11 and B-12 respectively. Results of the Dunnett's comparison performed on the data are presented in Table 3-5 and 3-6, and this information is summarized by station in Table 3-7. EC_{50} s, where calculable, are



Significantly different from control at P=0.05, based on Dunnett's test run on arc-sine (square root) transformed data.

Not significantly different from control.

Figure 3.1 Comparison of mean percent survival of amphipods in sediments from various stations in Massachusetts and Cape Cod Bays and negative controls.

Table 3-5 Mean sea urchin fertilization test data (In percent) for sediment porewater samples. Asterisks denote significant differences between site and reference porewater (Dunnett's *t*-test; * *P* ≤ 0.05, ** *P* ≤ 0.001).

100% WQAS ¹		50% WQAS ²		25% WQAS ³		12.5% WQAS ⁴	
Site	Mean ± SD	Site	Mean ± SD	Site	Mean ± SD	Site	Mean ± SD
REF ⁵	95.8 ± 1.9	REF ⁵	95.8 ± 1.9	REF ⁵	95.8 ± 1.9	REF ⁵	95.8 ± 1.9
9	97.0 ± 1.9	12	97.0 ± 2.3	9	97.5 ± 2.4	11	94.2 ± 2.2
12	96.2 ± 1.5	10	96.0 ± 1.9	10	96.8 ± 2.6	10	93.8 ± 2.2
6	94.2 ± 1.3	9	94.8 ± 3.1	11	95.8 ± 2.9	12	92.5 ± 3.0
10	92.0 ± 1.6	7	94.6 ± 2.2	5	95.0 ± 3.7	7	92.3 ± 5.0
7	91.8 ± 4.4	5	94.0 ± 2.9	12	93.8 ± 1.6	6	90.0 ± 5.0
5	90.6 ± 3.4	11	93.4 ± 4.3	8	93.8 ± 1.5	5	88.4 ± 3.7
11	90.0 ± 2.9*	6	93.4 ± 3.0	7	89.0 ± 4.4*	2	87.0 ± 5.5
8	66.6 ± 12.5**	8	92.8 ± 4.7	6	87.2 ± 8.5*	8	86.2 ± 9.7
4	5.4 ± 3.1**	4	15.4 ± 4.8**	2	41.2 ± 3.9**	9	85.6 ± 11.6
3	2.6 ± 2.4**	3	14.4 ± 7.3**	4	35.0 ± 7.2**	4	76.0 ± 6.1**
2	2.0 ± 1.6**	2	9.4 ± 2.9**	3	24.4 ± 5.2**	3	28.6 ± 14.5**
1	0.6 ± 0.5**	1	2.8 ± 2.0**	1	2.6 ± 1.5**	1	3.6 ± 2.2**

¹100% of water quality adjusted sample.

²50% of water quality adjusted sample.

³25% of water quality adjusted sample.

⁴12.5% of water quality adjusted sample.

⁵ Reference porewater extracted from sediment collected in Redfish Bay, Texas. The 100% water quality adjusted sample was used as the control for each dilution series.

Table 3-6 Mean sea urchin embryological development test data (in percent) for sediment porewater samples. Asterisks denote significant differences between site and reference porewater (Dunnett's t-test; * $P \leq 0.05$, ** $P \leq 0.001$).

50% WQAS ¹		25% WQAS ²		12.5% WQAS ³		6.25% WQAS ⁴	
Site	Mean \pm SD	Site	Mean \pm SD	Site	Mean \pm SD	Site	Mean \pm SD
REF ⁵	98.6 \pm 1.1	REF ⁵	98.6 \pm 1.1	REF ⁵	98.6 \pm 1.1	REF ⁵	98.6 \pm 1.1
7	97.0 \pm 0	7	98.4 \pm 1.5	9	98.2 \pm 0.8	5	98.4 \pm 1.9
6	65.6 \pm 30.1**	9	98.0 \pm 1.2	6	98.2 \pm 0.4	7	98.2 \pm 1.9
9	37.6 \pm 16.2**	8	97.4 \pm 1.5	5	97.8 \pm 1.3	11	98.0 \pm 0.7
4	17.2 \pm 15.5**	4	96.5 \pm 1.9	7	97.8 \pm 0.8	4	97.8 \pm 1.3
2	11.0 \pm 4.5**	6	96.2 \pm 2.0	12	97.6 \pm 0.9	8	97.6 \pm 2.2
3	0 \pm 0**	2	93.6 \pm 3.4*	2	97.0 \pm 3.3	2	97.6 \pm 1.1
8	0 \pm 0**	5	91.2 \pm 1.3**	4	97.0 \pm 2.4	12	97.6 \pm 1.1
1	0 \pm 0**	12	75.6 \pm 15.0**	8	96.2 \pm 1.1	6	97.0 \pm 1.6
10	0 \pm 0**	3	21.0 \pm 10.7**	10	95.2 \pm 7.6	9	96.6 \pm 2.3
11	0 \pm 0**	11	0 \pm 0**	3	85.8 \pm 12.5*	10	96.6 \pm 2.3
12	0 \pm 0**	10	0 \pm 0**	11	35.2 \pm 31.8**	3	87.2 \pm 11.6**
5	0 \pm 0**	1	0 \pm 0**	1	0 \pm 0**	1	67.2 \pm 7.2**

¹50% of water quality adjusted sample.

²25% of water quality adjusted sample.

³12.5% of water quality adjusted sample.

⁴6.25% of water quality adjusted sample.

⁵ Reference porewater extracted from sediment collected in Redfish Bay, Texas. The 100% water quality adjusted sample was used as the control for each dilution series.

Table 3-7 Summary of Dunnett's *t*-test results for the sea urchin fertilization and embryological development tests with sediment porewater.

Site	Mean % Fertilization at the % WQAS ¹ Tested				Mean % Normal Development at the % WQAS ¹ Tested			
	100%	50%	25%	12.5%	50%	25%	12.5%	6.25%
1	0.6**	2.8**	2.6**	3.6**	0**	0**	0**	67.2**
2	2.0**	9.4**	41.2**	87.0	11.0**	93.6*	97.0	97.6
3	2.6**	14.4**	24.4**	28.6**	0**	21.0**	85.8*	87.2**
4	5.4**	15.4**	35.0**	76.0**	17.2**	96.5	97.0	97.8
5	90.6	94.0	95.0	88.4	0**	91.2**	97.8	98.4
6	94.2	93.4	87.2*	90.0	65.6**	96.2	98.2	97.0
7	91.8	94.6	89.0*	92.3*	97.0	98.4	97.8	98.2
8	66.6**	92.8	93.8	86.2	0**	97.4	96.2	97.6
9	97.0	94.8	97.5	85.6	37.6**	98.0	98.2	96.6
10	92.0	96.0	96.8	93.8	0**	0**	95.2	96.6
11	90.0*	93.8	95.8	94.2	0**	0**	35.2**	98.0
12	96.2	97.0	93.8	92.5	0**	75.6**	97.6	97.6

¹Water quality adjusted porewater sample.

Note: Significant differences from respective controls (* $P \leq 0.05$, ** $P \leq 0.01$).

3.0 Results and Discussion (continued)

given in Table 3-8. The results of the Dunnett's test indicated that fertilization was significantly ($P \leq 0.05$) reduced in 6 of the 12 sediment porewater samples (Stations 1 through 4, 8, and 11) in comparison to control porewater. Porewater samples from Stations 1 through 4 (all in Wellfleet Harbor) were highly toxic, with mean percent fertilization in 100-percent samples ranging from 0.6 to 5.4 percent in comparison to 95.8 percent for the control. The 50 percent and 25 percent dilutions from these 4 stations in addition to Stations 6 and 7 were all significantly toxic as well, and the 12.5 percent dilutions of samples from Stations 1, 3, and 4 also had significantly reduced fertilization (Table 3-7).

In a preliminary assay (run in a dilution series of 100, 50, 25, and 12.5 percent of the original samples), all of the 50 percent dilutions were significantly toxic to sea urchin embryos. In the present assay, in which a series of dilutions from 50 to 6.25 percent were tested, results of Dunnett's test indicated that normal development was significantly reduced in 92 percent (11 of 12) of the 50-percent sediment porewater samples in comparison to control porewater. Fifty percent (6 of 12) of the 25 percent dilutions and 17 percent (2 of 12) of the 12.5 percent and 6.25 percent dilutions had significantly reduced normal development (Table 3-7).

EC_{50} values for the fertilization test (Table 3-8) were lowest for Stations 1 through 4 in Wellfleet Harbor, with values ranging from <12.5 percent to 23.6 percent. EC_{50} values for the embryological development test were lowest for Stations 1 (7.5 percent) and 3 (18.6 percent) in Wellfleet Harbor, and Stations 10 (17.3 percent) and 11 (11.2 percent) in Salem Harbor. Station 1 in Wellfleet Harbor had the lowest EC_{50} values for both percent fertilization and percent normal embryological development.

3.4 Benthic Community Structure

3.4.1 Diversity and other community characteristics. A breakdown of the relative percent composition of major taxonomic groups for each of the 12 stations is presented in Table 3-9. Percentages were calculated with respect to both abundance and numbers of species. Annelids (predominantly polychaetes) dominated the samples by percent abundance (52 to 99 percent) and percent species (51 to 66 percent). Crustaceans and molluscs were either second or third in rank for these parameters depending on the station. Together, annelids, crustacea and molluscs represented about 95 percent or more of the total fauna. A quantitative list of species by station and replicate is given in Appendix B, Table B-13.

Table 3-10 summarizes diversity and other community-level parameters by station. Total faunal densities (m^{-2}) and average numbers of species per sample ($0.1 m^2$) are also plotted by station in Figure 3.2. Densities ranged from 2213/ m^2 at Station 1 in Wellfleet Harbor to 105,843/ m^2 at Station 8 in Boston Harbor. Average numbers of species per sample ranged from 21 at Station 1 to 53 at Station 5 in Boston Harbor, and total numbers of species (over all replicates within a station) ranged from 31 at Station 7 in Boston Harbor to 69 at Station 12 outside of Salem Harbor. H' ranged

Table 3-8 EC₅₀ values for sediment pore water samples.

Site	Fertilization Test		Embryological Development Test	
	EC ₅₀ ¹	95% Confidence Limits	EC ₅₀ ¹	95% Confidence Limits
1	<12.5	-	7.5	NA ²
2	23.6	21.5-25.8	36.2	35.1-37.4
3	<12.5	-	18.6	16.9-20.4
4	20.7	18.3-23.4	37.6	36.1-39.1
5	73.3	71.2-75.5	33.5	32.2-34.9
6	>100	-	>50	-
7	>100	-	>50	-
8	>100	-	68.3	65.1-71.6
9	>100	-	43.6	39.9-47.7
10	>100	-	17.3	16.8-17.8
11	>100	-	11.2	10.5-12.0
12	>100	-	30.2	28.4-32.0

¹Percent of water quality adjusted porewater sample.

²95% confidence limits not reliable.

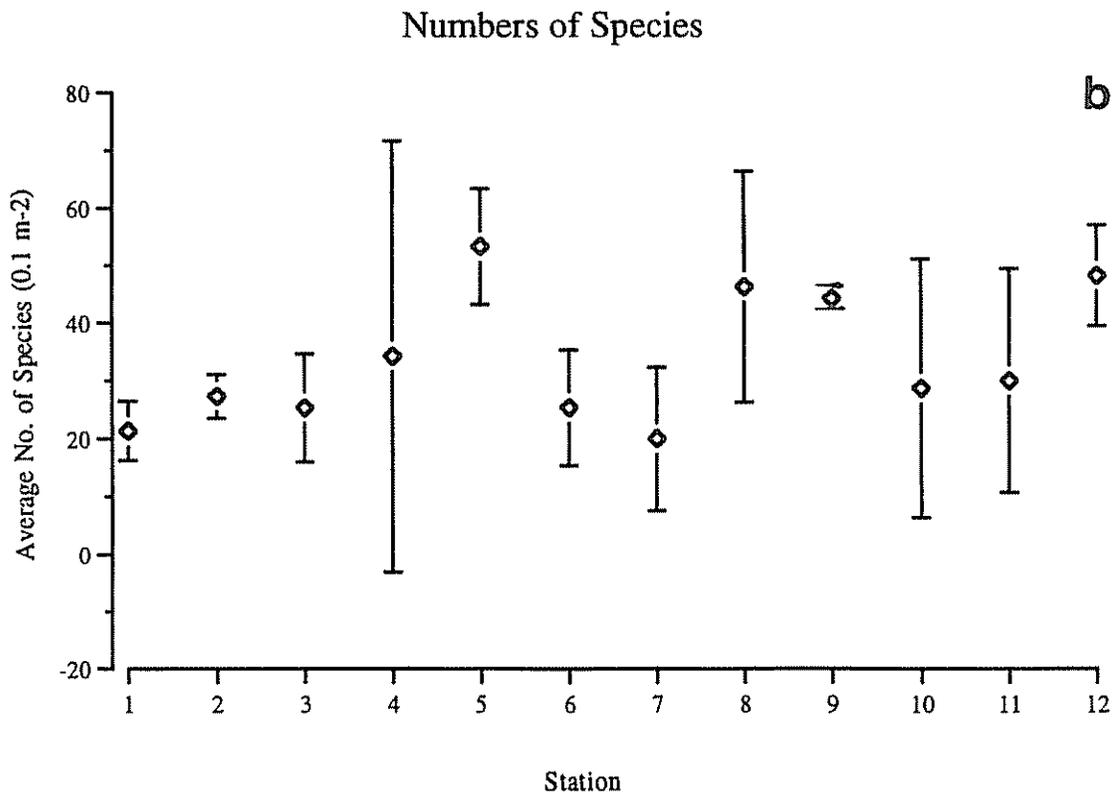
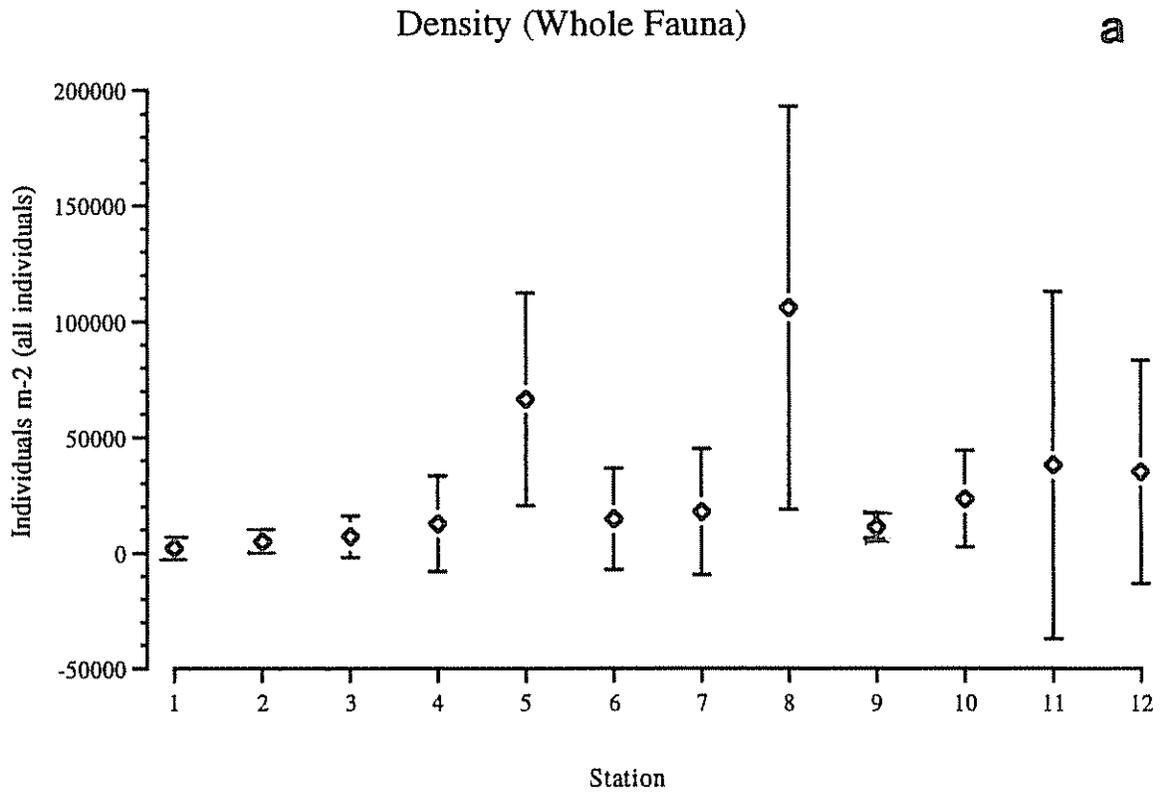


Figure 3.2 Graphs of (a) total faunal densities (m⁻²) and (b) numbers of species per sample (0.1m²) by station with 95% confidence Intervals.

Table 3-9 Relative percent composition of major taxonomic groups by abundance and species numbers for each station.

Station	% Abundance				% Species			
	% Annelida	% Crustacea	% Mollusca	% Other	% Annelida	% Crustacea	% Mollusca	% Other
1	86.1	6.3	7.4	0.2	54.1	21.6	21.6	2.7
2	85.2	3.1	11.4	0.3	59.0	10.3	28.2	2.6
3	51.5	42.3	6.2	0.0	56.4	20.5	23.1	0.0
4	76.0	20.7	3.3	0.0	57.4	20.4	20.4	1.9
5	87.3	8.1	4.4	0.1	55.9	26.5	13.2	4.4
6	94.7	5.0	0.3	0.0	59.0	17.9	17.9	5.1
7	98.0	0.9	1.1	0.0	61.3	12.9	22.6	3.2
8	94.7	5.0	0.2	13.3	64.1	17.2	15.6	3.1
9	86.5	1.9	11.4	0.2	58.5	18.5	20.0	3.1
10	96.8	1.0	2.1	0.1	60.9	10.9	26.1	2.2
11	99.2	0.5	0.3	0.0	66.0	22.0	10.0	2.0
12	90.5	8.1	1.3	0.2	50.7	24.6	20.3	4.3

Table 3-10 Diversity and other community-level parameters by station.

Station	Ind/m ²	avg No. Sps ^a	Total S ^b	H' ^b	J' ^b
1	2213	21	37	1.963757	0.5438
2	4947	27	39	2.31428	0.6317
3	6930	25	39	2.243341	0.6123
4	12437	34	54	2.343514	0.5875
5	66243	53	68	2.095865	0.4967
6	14557	25	39	1.153274	0.3148
7	17677	20	31	1.133395	0.3301
8	105843	46	64	0.952034	0.2289
9	11100	44	65	2.671838	0.6401
10	23200	29	46	1.055915	0.2758
11	37863	30	50	0.912633	0.2333
12	34967	48	69	2.103694	0.4968

^aPer replicate (0.1m²)

^bBased on 3 combined replicates (0.1m² each)

3.0 Results and Discussion (continued)

from 0.95 at Station 8 to 2.34 at Station 4. J' ranged from 0.23 at Station 11 to 0.64 at Station 9.

On the average, densities and numbers of species were lower at Wellfleet Harbor stations than in the other two harbor systems (Figure 3.2). Also, these variables were either the highest or second highest at the designated reference sites within each harbor system (i.e., Station 4 in Wellfleet Harbor, Station 8 in Boston Harbor, Station 12 in Salem/Beverly Harbor). H' diversity typically was highest or second highest for reference sites with the exception of Station 8 in Boston Harbor. The low diversity at this site was due largely to the extremely high density of the polychaete *Polydora cornuta* (86,530/m²), which reduced the evenness component.

3.4.2 Dominant taxa. Dominant (10 most abundant) species and estimates of their abundances by station are presented in Table 3-11. These species, which accounted for 83 percent or more of the cumulative percent abundance of all species at a given station, largely characterizes the benthic communities of these shallow subtidal harbor systems. In all but two cases (Stations 4 and 9), the two highest-ranked dominants represented more than 50 percent of the cumulative percent abundance of all species.

The most persistent and strongly ranked dominant among the Wellfleet Harbor stations was the polychaete *Streblospio benedicti*. The polychaete *Polydora cornuta* was the most persistent and strongly ranked dominant among Boston Harbor and Salem/Beverly Harbor stations. Densities of these two species are plotted by station in Figures 3.3a (*Streblospio*) and 3.3b (*Polydora*). Both species are known to be tolerant of a wide range of polluted and organically enriched sediments (Pearson and Rosenberg 1978). The high densities of these species at some stations--e.g., at Stations 6, 7 and 8 in Boston Harbor and Stations 10 and 11 in Salem Harbor, where proportions of these species represented more than half of the total faunal abundance--may be in response to polluted conditions at these sites.

3.4.3 Numerical classification of station groups. Numerical classification of samples combined over the three replicates within a station defines two major cluster groups (Figure 3.4). These groups consist of Cluster Group A containing all but one station (Station 7) in Boston Harbor and Salem/Beverly Harbors, and Cluster Group B containing Station 7 in Boston Harbor and all stations in Wellfleet Harbor. The separation of these two cluster groups reflects the generally lower total densities and numbers of species, and higher-ranked dominance of *Streblospio benedicti* (in contrast to *Polydora cornuta*), at Station 7 and at Stations 1 through 4 in comparison to the other sites.

3.4.4 Statistical analysis of among-station differences. Results of ANOVA, run on \log_{10} -transformed data, revealed a highly significant difference in mean total density among the 12 stations [$P > F=0.0001$; calculated F ratio = 11.154; $df = 11$

Table 3-11 Dominant macroinfaunal species (A=Amphipoda, B=Bivalvia, C=Cumacea, D=Decapoda, G=Gastropoda, O=Oligochaeta, P=Polychaeta).

Wellfleet Harbor				Wellfleet Harbor			
Station 1				Station 2			
Species	Ind.m ²	CUM%	Species	Ind.m ²	CUM%	Species	CUM%
<i>Sireblospio benedicti</i> (P)	890	40.2	<i>Sireblospio benedicti</i> (P)	2030			41.0
<i>Microphthalamus szcelkowi</i> (P)	396	58.1	<i>Leitoscoloplos robustus</i> (P)	510			51.3
<i>Heteromastus filiformis</i> (P)	170	65.8	<i>Tubificoides nr. pseudogaster</i> (O)	333			58.1
<i>Tubificoides nr. pseudogaster</i> (O)	163	73.2	<i>Capitella capitata species complex</i> (P)	293			64.0
<i>Crassostrea virginica</i> (B)	87	77.1	<i>Heteromastus filiformis</i> (P)	290			69.9
<i>Polydora cornuta</i> (P)	67	80.1	<i>Mercenaria mercenaria</i> (B)	277			75.5
<i>Pagurus longicarpus</i> (D)	63	83.0	<i>Spio setosa</i> (P)	197			79.4
<i>Capitella capitata species complex</i> (P)	63	85.8	<i>Acteocina canaliculata</i> (G)	180			83.1
<i>Polydora spp. juv.</i> (P)	57	88.4	<i>Scoletepis squamata</i> (P)	157			86.3
<i>Tellina agilis</i> (B)	43	90.4	<i>Polydora cornuta</i> (P)	150			89.3
All fauna (37 species)*	2213	100.0	All fauna (39 species)*	4947			100.0
Station 3				Station 4			
<i>Sireblospio benedicti</i> (P)	2100	30.3	<i>Heteromastus filiformis</i> (P)	2750			22.1
<i>Ampelisca abdita</i> (A)	1397	50.5	<i>Sireblospio benedicti</i> (P)	2623			43.2
<i>Ampelisca spp. juv.</i> (A)	973	64.5	<i>Ampelisca abdita</i> (A)	1543			55.6
<i>Heteromastus filiformis</i> (P)	550	72.4	<i>Leitoscoloplos robustus</i> (P)	1517			67.8
<i>Leucon sp. 1</i> (C)	317	77.0	<i>Polydora cornuta</i> (P)	987			75.7
<i>Capitella capitata species complex</i> (P)	250	80.6	<i>Ampelisca spp. juv.</i> (A)	863			82.7
<i>Microprotopus ranyei</i> (A)	210	83.6	<i>Tubificoides nr. pseudogaster</i> (O)	503			86.7
<i>Asabellides oculata</i> (P)	187	86.3	<i>Tharyx acutus</i> (P)	410			90.0
<i>Tharyx acutus</i> (P)	173	88.8	<i>Capitella capitata species complex</i> (P)	237			91.9
<i>Tubificoides nr. pseudogaster</i> (O)	167	91.2	<i>Tellina agilis</i> (B)	183			93.4
All fauna (39 species)*	6930	100.0	All fauna (54 species)*	12437			100.0

*Total of 3 replicate samples (0.1m² each)

Table 3-11 Dominant macroinfaunal species (A=Amphipoda, B=Bivalvia, C=Cumacea, D=Decapoda, G=Gastropoda, O=Oligochaeta, P=Polychaeta). (continued)

Boston Harbor				Boston Harbor			
Station 5				Station 6			
Species	Ind.m ²	CUM%	Species	Ind.m ²	CUM%	Species	CUM%
<i>Polydora cornuta</i> (P)	27403	41.4	<i>Polydora cornuta</i> (P)	10673	73.3		
<i>Eteone longa</i> (P)	12697	60.5	<i>Sireblospio benedicti</i> (P)	1527	83.8		
<i>Phyllodoce mucosa</i> (P)	7020	71.1	<i>Ampelisca abdita</i> (A)	577	87.8		
<i>Ampelisca abdita</i> (A)	3617	76.6	<i>Tharyx acutus</i> (P)	513	91.3		
<i>Streblospio benedicti</i> (P)	3083	81.2	<i>Tubificoides nr. pseudogaster</i> (O)	270	93.2		
<i>Spio thuitini</i> (P)	2403	84.9	<i>Asabellides oculata</i> (P)	167	94.3		
<i>Mya arenaria</i> (B)	2020	87.9	<i>Eteone longa</i> (P)	150	95.3		
<i>Neanthes virens</i> (P)	1107	89.6	<i>Capitella capitata species complex</i> (P)	93	96.0		
<i>Tharyx acutus</i> (P)	777	90.8	<i>Ampelisca spp. juv.</i> (A)	87	96.6		
<i>Asabellides oculata</i> (P)	683	91.8	<i>Nephtys neotena</i> (P)	77	97.1		
All fauna (68 species)*	66243	100.0	All fauna (39 species)*	14557	100.0		
Station 7				Station 8			
<i>Streblospio benedicti</i> (P)	9567	54.1	<i>Polydora cornuta</i> (P)	86530	81.8		
<i>Polydora cornuta</i> (P)	6237	89.4	<i>Ampelisca abdita</i> (A)	3683	85.2		
<i>Tubificoides nr. pseudogaster</i> (O)	1017	95.2	<i>Phyllodoce mucosa</i> (P)	2800	87.9		
<i>Asabellides oculata</i> (P)	177	96.2	<i>Neanthes virens</i> (P)	2733	90.5		
<i>Ampelisca abdita</i> (A)	123	96.9	<i>Aricidea catherinae</i> (P)	1827	92.2		
<i>Mya arenaria</i> (B)	103	97.4	<i>Tubificoides apectinatus</i> (O)	1393	93.5		
<i>Polydora spp. juv.</i> (P)	103	98.0	<i>Ampelisca spp. juv.</i> (A)	1353	94.8		
<i>Capitella capitata species complex</i> (P)	70	98.4	<i>Tubificoides nr. pseudogaster</i> (O)	1350	96.1		
<i>Tharyx acutus</i> (P)	63	98.8	<i>Spio thuitini</i> (P)	1120	97.1		
<i>Nassaruis trivittatus</i> (G)	47	99.0	<i>Polydora spp. juv.</i> (P)	390	97.5		
All fauna (31 species)*	17677	100.0	All fauna (64 species)*	105843	100.0		

*Total of 3 replicate samples (0.1m² each)

Table 3-11 Dominant macroinfaunal species (A=Amphipoda, B=Bivalvia, C=Cumacea, D=Decapoda, G=Gastropoda, O=Oligochaeta, P=Polychaeta).(continued)

Salem/Beverly Harbors			
Station 9		Station 10	
Species	Ind.m ⁻²	CUM%	Species
<i>Polydora cornuta</i> (P)	2823	25.4	<i>Polydora cornuta</i> (P)
<i>Tubificoides apectinatus</i> (O)	1467	38.6	<i>Tharyx acutus</i> (P)
<i>Scoletoma hebes</i> (P)	1293	50.3	<i>Nephtys neotena</i> (P)
<i>Aricidea catherinae</i> (P)	1053	59.8	<i>Phyllodoce mucosa</i> (P)
<i>Mytilidae spp. juv.</i> (B)	637	65.5	<i>Nephtys incisa</i> (P)
<i>Tubificoides nr. pseudogaster</i> (O)	620	71.1	<i>Tubificoides apectinatus</i> (O)
<i>Tharyx acutus</i> (P)	477	75.4	<i>Aricidea catherinae</i> (P)
<i>Phyllodoce mucosa</i> (P)	380	78.8	<i>Mya arenaria</i> (B)
<i>Lumbrineridae spp. juv.</i> (P)	310	81.6	<i>Mytilidae spp. juv.</i> (B)
<i>Tellina agilis</i> (B)	193	83.4	<i>Microphthalmus szelkowitzii</i> (P)
All fauna (65 species)*	11100	100.0	All fauna (46 species)*
Station 11			
Species	Ind.m ⁻²	CUM%	Species
<i>Polydora cornuta</i> (P)	30197	78.7	<i>Polydora cornuta</i> (P)
<i>Tharyx acutus</i> (P)	3487	87.7	<i>Phyllodoce mucosa</i> (P)
<i>Phyllodoce mucosa</i> (P)	973	90.3	<i>Tharyx acutus</i> (P)
<i>Capitella capitata species complex</i> (P)	777	92.3	<i>Tubificoides apectinatus</i> (O)
<i>Tubificoides nr. pseudogaster</i> (O)	690	94.1	<i>Photis pollex</i> (A)
<i>Neanthes virens</i> (P)	417	95.2	<i>Aricidea catherinae</i> (P)
<i>Tubificoides apectinatus</i> (O)	400	96.2	<i>Pygospio elegans</i> (P)
<i>Streblospio benedicti</i> (P)	107	96.6	<i>Parougia caeca</i> (P)
<i>Microphthalmus szelkowitzii</i> (P)	90	96.9	<i>Capitella capitata species complex</i> (P)
<i>Ampelisca abdita</i> (A)	83	97.1	<i>Nephtys ciliata</i> (P)
All fauna (50 species)*	37863	100.0	All fauna (69 species)*
Station 12			
Species	Ind.m ⁻²	CUM%	Species
<i>Polydora cornuta</i> (P)	14160	40.5	<i>Polydora cornuta</i> (P)
<i>Tharyx acutus</i> (P)	5557	56.4	<i>Phyllodoce mucosa</i> (P)
<i>Phyllodoce mucosa</i> (P)	3833	67.3	<i>Tharyx acutus</i> (P)
<i>Capitella capitata species complex</i> (P)	2893	75.6	<i>Tubificoides apectinatus</i> (O)
<i>Tubificoides nr. pseudogaster</i> (O)	2250	82.1	<i>Photis pollex</i> (A)
<i>Neanthes virens</i> (P)	1790	87.2	<i>Aricidea catherinae</i> (P)
<i>Tubificoides apectinatus</i> (O)	767	89.4	<i>Pygospio elegans</i> (P)
<i>Streblospio benedicti</i> (P)	553	91.0	<i>Parougia caeca</i> (P)
<i>Microphthalmus szelkowitzii</i> (P)	540	92.5	<i>Capitella capitata species complex</i> (P)
<i>Ampelisca abdita</i> (A)	227	93.1	<i>Nephtys ciliata</i> (P)
All fauna (50 species)*	34967	100.0	All fauna (69 species)*

*Total of 3 replicate samples (0.1m² each)

3.0 Results and Discussion (continued)

versus 24]. Tukey's HSD test revealed significant differences (at $P \leq 0.05$) for the following statistical pairs: 1 versus 2, 3, 4, 8, 9, 10, 11; 3 versus 5; 4 versus 5; 5 versus 8 and 11; 6 versus 8 and 11; 7 versus 8 and 11; 8 versus 12; 9 versus 11; 10 versus 11; and 11 versus 12. Table 3-12 summarizes these results on a per-harbor basis (also refer to Figure 3.2 for plots of station means). Among Wellfleet Harbor sites, Station 1 (in the innermost harbor area) had a significantly lower density than the other three stations, which were similar to one another. Among Boston Harbor sites, Station 8 in Hull Bay had a significantly higher density than the other three stations, which were similar to one another. Among Salem/Beverly Harbor sites, Station 11 (in the innermost harbor area) had a significantly higher density than the other three sites, which were similar to one another.

Results of the Kruskal-Wallis nonparametric test revealed a highly significant difference in mean numbers of species among the 12 sites [$P > F=0.0001$; calculated F ratio = 8.943; $df = 11$ versus 24]. Scheffe's test revealed significant differences (at $P \leq 0.05$) for the following station pairs: 1 versus 12 and 5; 3 versus 5; 6 versus 5; and 7 versus 12 and 5. There were no significant differences in this variable among stations within Wellfleet Harbor or Salem/Beverly Harbor (Table 3-12, also refer to Figure 3.2 for plots of station means). Among Boston Harbor sites, Station 5 off Deer Island had significantly more species than Station 6 in Dorchester Bay and Station 7 in Quincy Bay.

3.5 Relationships Between Sediment Chemistry, Sediment Toxicity, and Condition of Benthic Communities.

Statistical conclusions from correlation analysis between selected chemical/physical, biological, and toxicological variables, using station means as observations, are summarized in Table 3-13. The chemical contaminant variables included in the analysis were those variables that exceeded corresponding bioeffect (NOEL or ER-L) values at one or more of the stations (Section 3.2). Contaminants without reported bioeffect values were excluded. Most of these contaminants (e.g., Be, Se, Tl) were below detectable concentrations at all stations. Conclusions are given for correlations between each key biological or toxicological variable (infaunal density, infaunal species richness, amphipod percent survival, sea urchin percent fertilization, and sea urchin percent normal embryological development) and the various chemical/physical variables, including (1) un-normalized chemical/physical variables, (2) sediment chemical variables (both organic compounds and metals) normalized to silt+clay, and (3) organic chemical variables normalized to TOC and metals normalized to aluminum. Determination of significance was based on whether the Type I error probability for the null hypothesis of no correlation ($H_0: r=0$) was \leq the Dunn-Sidak adjusted significance level (based on an unadjusted P of 0.05). Appendix E, Tables E-1 through E-15, presents individual correlation coefficients (r values) and associated Type I error probabilities for each combination of variables.

Table 3-12. Summary of results of Tukey's HSD test (A) and Scheffe's test (B) for pair-wise comparisons of station differences in densities and numbers of species, respectively, grouped by harbors. Stations connected by bars are not significantly different at $P = 0.05$.

A. Density				
Wellfleet Harbor:	1	2	3	4

Boston Harbor:	5	6	7	8

Salem/Beverly Harbors:	12	9	10	11

B. Numbers of Species				
Wellfleet Harbor:	1	2	3	4

Boston Harbor:	5	8	7	6

Salem/Beverly Harbors:	9	10	11	12

Table 3-13. Statistical conclusions of Pearson's product moment correlations between selected chemical/physical, biological, and toxicological variables. UN = correlation with un-normalized chemical/physical variable; S&C = correlation with sediment chemical variable (organic and inorganic) normalized to silt & clay; TOC/AI = correlation with organic chemical variable normalized to TOC and inorganic chemical variable normalized to AI. S = significant correlation at Dunn-Sidak adjusted significance level (to control for experiment-wise error rate), based on unadjusted *P* of 0.05; NS = not significant. See Appendix E, Tables E-1 through E-15 for details on correlation coefficients (*r* values) and associated Type I error probabilities.

Chemical/Physical Variable	Infaunal Density ^a			Infaunal Species Richness ^a			Amphipod % Survival ^f			Sea Urchin % Fertilization ^c			Sea Urchin % Normal Development ^c		
	UN	S&C	TOC/AI	UN	S&C	TOC/AI	UN	S&C	TOC/AI	UN	S&C	TOC/AI	UN	S&C	TOC/AI
DO ^a	NS	-	-	NS	-	-	-	-	-	-	-	-	-	-	-
Sed Temp ^a	NS	-	-	NS	-	-	-	-	-	-	-	-	-	-	-
TOC ^a	NS	-	-	NS	-	-	NS	-	-	NS	-	-	NS	-	-
% Sand ^b	NS	-	-	NS	-	-	NS	-	-	NS	-	-	NS	-	-
% Silt and Clay ^b	NS	-	-	NS	-	-	NS	-	-	NS	-	-	NS	-	-
TAN ^a	NS	-	-	NS	-	-	NS	-	-	NS	-	-	S	-	-
UAN ^a	NS	-	-	NS	-	-	NS	-	-	NS	-	-	S	-	-
PCBs ^a	NS	NS	NS	NS	NS	NS	NS	-	-	NS	NS	NS	NS	NS	NS
Acenaphthene ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Anthracene ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Fluorene ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Phenanthrene ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Benz(a)anthracene ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Fluoranthene ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Pyrene ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Σ PAHs ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
THC ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Dieldrin ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Chlordane ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
DDT ^a	S	S	S	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
DDE ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	NS	NS	NS	NS
DDD ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Σ DDT ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
As ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	NS	NS
Ag ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Cr ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	NS	NS	NS	NS
Cu ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Pb ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Hg ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	S	NS	S	NS	NS	NS
Zn ^a	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS

^alog₁₀ transformed

^barcsine transformed

^carcsine square-root transformed

Streblospio benedicti

a

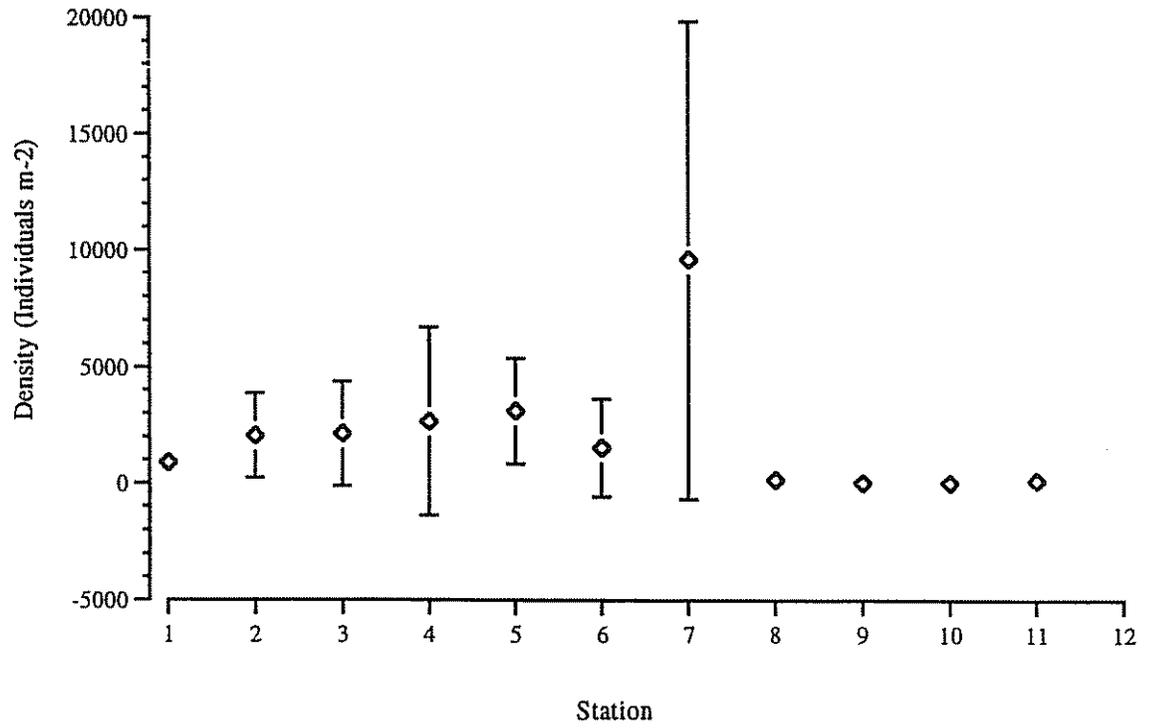


Figure 3.3a Densities, with 95% confidence intervals, of *Streblospio benedicti* (a) and *Polydora cornuta* (b) by station.

Polydora cornuta

b

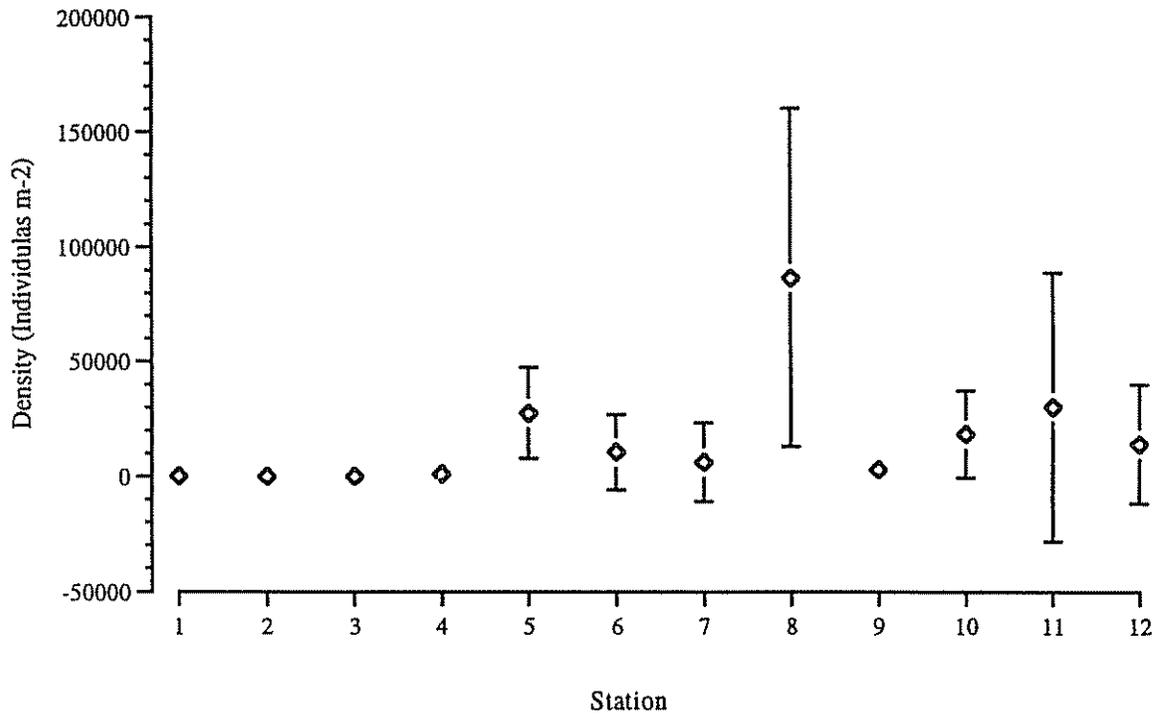


Figure 3.3b Densities, with 95% confidence intervals, of *Streblospio benedicti* (a) and *Polydora cornuta* (b) by station.

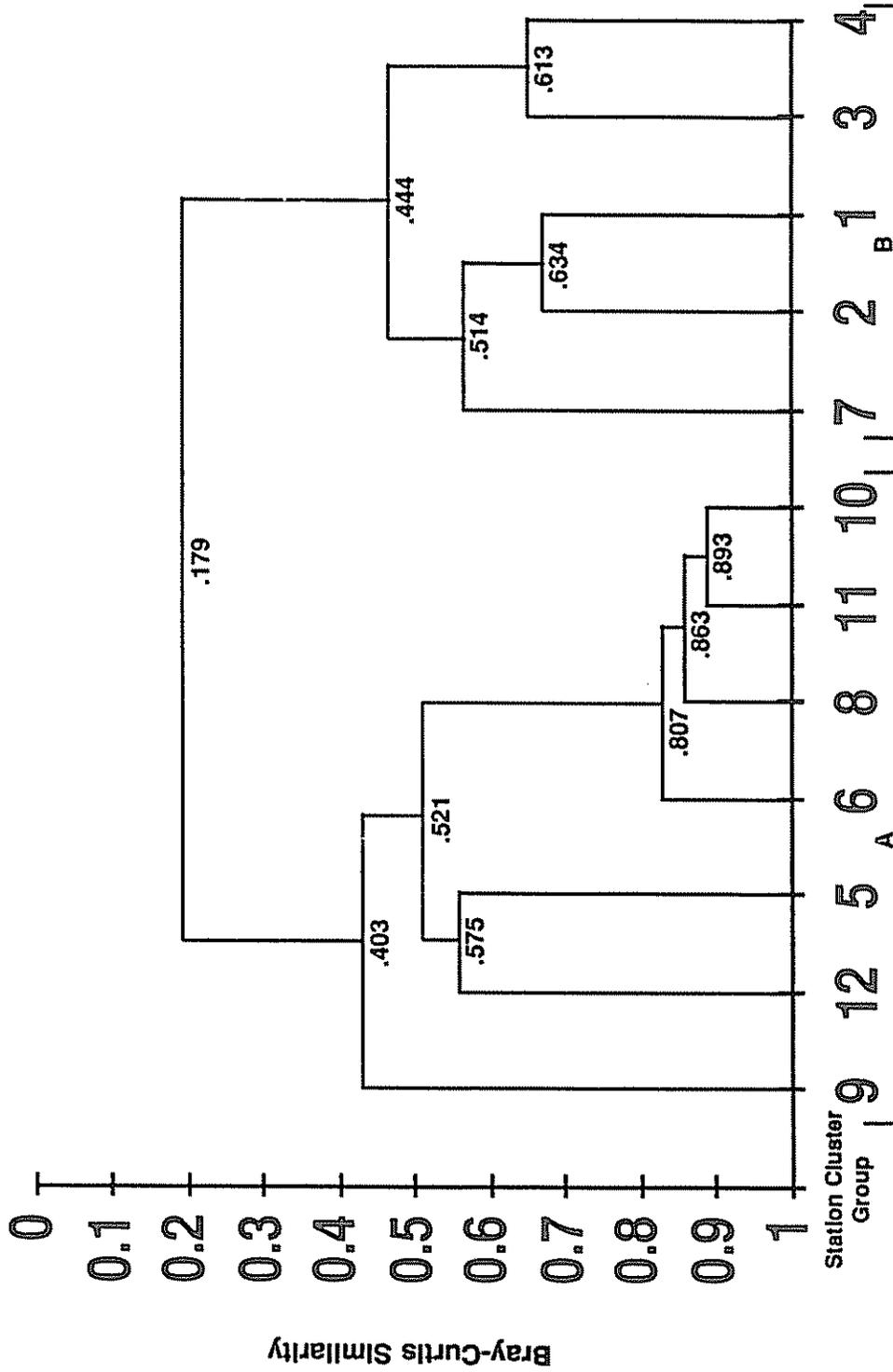


Figure 3.4 Dendrogram resulting from clustering of stations, using group-average sorting and Bray-Curtis similarity. Each station is represented by 3 combined replicates.

3.0 Results and Discussion (continued)

Infaunal density was not significantly correlated (at the Dunn-Sidak adjusted significance level) with any of the un-normalized chemical/physical variables except DDT ($r=0.860$, $P=0.000$, Table E-1). Significant positive correlations also occurred between infaunal density and DDT normalized to either silt+clay or TOC (silt+clay: $r=0.798$, $P=0.002$, Table E-6; TOC: $r=0.798$, $P=0.002$, Table E-11). The strong positive correlations between these variables are due largely to the occurrence at very high densities of the opportunistic and pollution-tolerant polychaete *Polydora cornuta* at stations (e.g., Station 8, 11, and 5) with some of the highest DDT loadings.

There were fairly strong positive correlations between infaunal density and most other organic and inorganic contaminant parameters (all except PCBs, dieldrin, and Pb; Table E-1). Though not significant at the more conservative Dunn-Sidak adjusted significance level, these correlations had Type I error probabilities under the unadjusted significance level of $P=0.05$. These correlations were due largely to the strong positive correlation between infaunal density and the silt+clay fraction $r=0.627$, $P=0.029$, Table E-1) and the tendency for many of these contaminants to be positively correlated with this same sediment parameter. It is interesting to note that when the chemical contaminant parameters are normalized to silt+clay, the strength of their associations with infaunal density diminish notably (Table E-6).

Infaunal species richness was not significantly correlated (at the Dunn-Sidak adjusted significance level) with any of the un-normalized or normalized chemical and physical variables (Table 3-13). Type I error probabilities for most comparisons were very large and exceeded even the less conservative, unadjusted significance level of $P=0.05$ (Tables E-2, E-7, and E-12 for un-normalized, silt+clay-normalized, and TOC/Al-normalized chemical data, respectively). Amphipod percent survival showed similar insignificant correlations with all un-normalized and normalized chemical/physical variables (Table 3-13 summary; Tables E-3, E-8, and E-13 for details of correlations with un-normalized, silt+clay-normalized, and TOC/Al-normalized chemical data, respectively).

Sea urchin percent fertilization was significantly correlated with most un-normalized sediment contaminant variables, consisting of acenaphthene, anthracene, fluorene, phenanthrene, benz(a)anthracene, fluoranthene, pyrene, Σ PAHs, total chlordane, DDE, DDD, Σ DDT, Cr, and Hg (Table 3-13). Correlations with these chemical variables were all positive and highly significant with Type I error probabilities ranging from near zero to 0.001 (Table E-4). In addition, there were marginally significant positive correlations with THC ($r=0.757$, $P=0.004$), As ($r=0.768$, $P=0.004$), and Cu ($r=0.783$, $P=0.003$). The positive direction of these correlations did not suggest a pattern of increasing sediment toxicity with increasing sediment contamination among the various stations. However, when the chemical variables were normalized to silt+clay, a marginally significant inverse correlation with THC resulted ($r=0.757$, $P=0.004$, Table E-9). Also, a significant inverse correlation with As ($r=-0.787$, $P=0.002$) resulted when this contaminant was normalized to Al (Table E-14).

3.0 Results and Discussion (continued)

Sea urchin percent normal development was not significantly correlated with any of the un-normalized or normalized sediment contaminant variables (Table 3-13). Type I error probabilities for all comparisons were large and exceeded even the less conservative, unadjusted significance level of $P=0.05$ (Tables E-5, E-10, and E-15 for un-normalized, silt+clay-normalized, and TOC/Al-normalized chemical data, respectively). However, there were significant inverse correlations between sea urchin percent normal development and concentrations in porewater of both total ammonia (TAN, $r=-0.972$, $P=0.000$) and unionized ammonia (UAN, $r=-0.804$, $P=0.002$) (Table E-5).

Overall, the correlation analysis did not reveal clear patterns of increasing sediment toxicity or benthic community degradation with increasing sediment contamination among the 12 sampling stations for most contaminant variables. The clearest patterns, as discussed above, were depicted only for associations between the following variables: sea urchin percent fertilization and THC normalized to silt+clay, sea urchin percent fertilization and As normalized to Al, sea urchin percent normal development and UAN, and infaunal density and DDT (un-normalized or normalized to either silt+clay or TOC).

A second, more straightforward data comparison method was used as a means of interpreting results of the three SQT components collectively (Table 3-14). A similar approach was introduced by Chapman (1990) and adopted by Carr (1993) in other sediment quality surveys. In the present application, evidence of contaminant-induced degradation at a station is provided by the combination of one or more NOEL/ER-L exceedances, one or more significant toxicity occurrences, and the presence of a stressed benthic community. Low species richness (≤ 50 species from combined replicates at a station) was used in this study as a somewhat subjective indicator of a stressed benthic community; other benthic community parameters were too variable to use for this purpose. A cut-off point of 50 species was based on comparison of numbers of species at reference sites relative to the other sites. All reference sites had more than 50 species.

Based on these criteria, there are six stations (Stations 1 and 3 in Wellfleet Harbor, Stations 6 and 7 in Boston Harbor, and Stations 10 and 11 in Salem/Beverly Harbors) with strong evidence of contaminant-induced degradation of the benthic environment. At least one contaminant (at Station 3) and up to 23 contaminants (Station 10) were present at these sites at potentially toxic concentrations (i.e., at concentrations that exceeded reported NOEL/ER-L values). It should be noted, however, that unionized ammonia (UAN) was present in porewater at three of these sites (Stations 1, 10, and 11) at concentrations that could have caused toxicity and benthic community degradation. Station 1 also had a very high concentration of sulfide, which could have contributed to the observed bioeffects.

Table 3-14 Summary of sediment quality triad results (adapted from Chapman et al. 1990, and Carr 1993).

Chemistry ^a	Toxicity ^b	Benthos ^c	Stations	Possible Conclusions
+	+	+	1,3,6,7,10,11	Evidence of environmental impacts caused by measured contaminants or other measured adverse conditions (such as high ammonia or sulfide levels).
-	-	-	none	No evidence of environmental degradation.
+	-	-	none	Measured contaminants are present at potentially toxic levels but not bioavailable.
-	+	-	none	Unmeasured contaminants or adverse conditions exist with the potential to cause degradation, but no clear benthic response.
-	-	+	none	Degraded benthos not due to presence of measured contaminants or conditions.
+	+	-	4,5,8,9,12	Measured contaminants or conditions have the potential to cause degradation, but no clear benthic response.
-	+	+	2	Unmeasured contaminants or conditions are causing degradation.
+	-	+	none	Contaminants are not bioavailable or benthic response not due to chemistry.

(+) Indicates one or more chemical contaminants exceeded NOEL or ER-L values (high ammonia or sulfide concentrations may have occurred as well).

(-) Indicates no exceedance for any measured contaminant.

^b (+) Indicates one or more toxicity endpoints were significantly different than control.

(-) Indicates no apparent toxicity effect.

^c (+) Indicates low species richness (total number of species from combined replicates within a station ≤ 50).

(-) Indicates no apparent adverse benthic effect.

3.0 Results and Discussion (continued)

Elevated chemical contaminants (above NOEL or ER-L values) and significant toxicity occurrences (one or more significant reductions in amphipod survival, sea urchin fertilization, or sea urchin embryological development) were observed at Stations 4, 5, 8, 9, and 12, though there was no clear indication of a stressed benthic community at these sites. Stations 8 and 12 had high concentrations of unionized ammonia (UAN) in porewater, which could have caused or contributed to the observed toxicity. All 12 stations showed some indication of environmental degradation (either high contaminant loading, presence of toxic sediments, or a species-poor benthos). However, at one station (Station 2), the combined SQT data suggested that unmeasured chemicals or conditions were causing the observed biological impacts.

These results are summarized in Figure 3.5. This summary also provides an indication of the degree of chemical contamination at a site by distinguishing between sites at which the threshold-level NOEL/ER-L values were exceeded and those at which the higher-level PEL/ER-M values were exceeded.

Sites with PEL/ER-M exceedances indicate the presence of contaminants at relatively high concentrations expected to cause adverse effects on a wide range of benthic organisms. Independent field studies have shown that ER-M exceedances correctly predict toxicity in a series of two to three tests in about 90 percent of the cases (comment from Massachusetts Bays Program reviewer). Contaminants at concentrations that are above the NOEL/ER-L values but less than the PEL/ER-M values could be responsible for some of the observed toxicity, but have a lower probability of being the cause relative to those contaminants that are present at concentrations above the PEL/ER-M values. All sites in Boston and Salem/Beverly Harbors had one or more chemicals that exceeded the higher PEL/ER-M values. None of the sites in Wellfleet Harbor had chemical contaminants present at concentrations that exceeded PEL/ER-M values.

3.6 Relative Status of Pollution-Induced Impacts Among the Different Harbor Systems

Stations were ranked from most to least degraded with respect to each of the three SQT components (Table 3-15). A rank sum was calculated for each station and these values were used in an overall relative ranking of stations.

Station 1 in Wellfleet harbor, an area removed from major urban and industrial centers such as Boston and Salem/Beverly Harbors, showed the strongest evidence of a degraded benthic system. This station had the second lowest species richness and the strongest toxicity responses among the 12 sites. Station 1 was ranked No. 1 consistently for all three toxicity tests and was the only site where all three toxicity endpoints were significantly reduced.

All stations in Wellfleet Harbor showed one or more significant toxicity test results. These sites received the four lowest average toxicity rankings. In addition, most

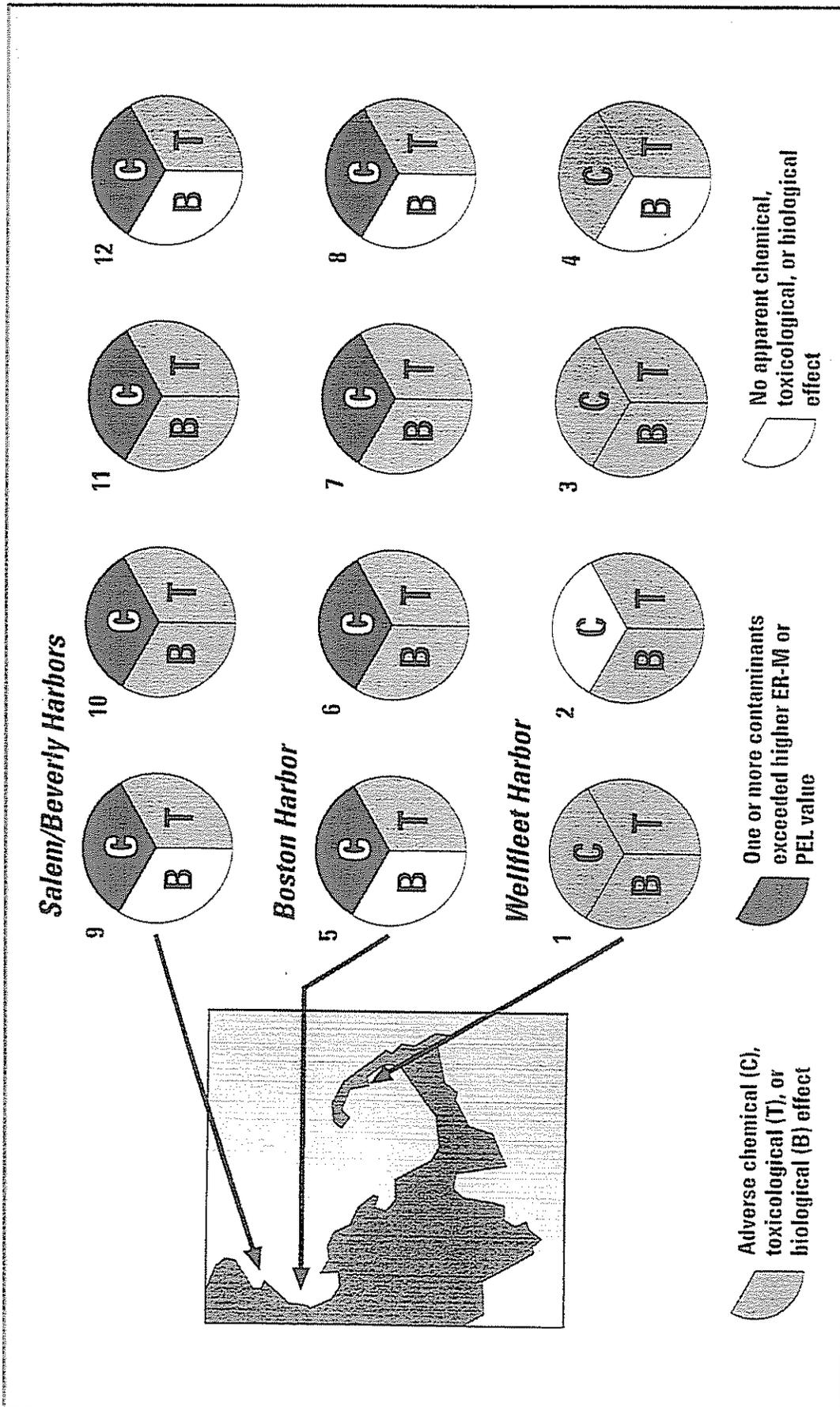


Figure 3.5 Summary of Sediment Quality Triad Results

Table 3-15 Ranking of stations with respect to sediment chemistry, toxicity, and benthic community data. The higher the ranking, the less degraded the station relative to the other stations.

Station	Ranks											Relative Ranking
	Chemistry ^a	Amphipod % Survival ^b	Sea Urchin % Fertilization ^c	Sea Urchin % Development ^d	Ave. Toxicity Rank	Infaunal Species Richness ^e	Rank Sum ^f					
1	9	1	1	1	1	2	12	1				
2	12	6	2	7	5	5	22	9				
3	10	8.5	3	4	5.2	3.5	18.7	6				
4	11	3.5	4	8	5.2	8	24.2	10				
5	3	5	7	6	6	12	21	8				
6	5	3.5	10	11.5	8.3	3.5	16.8	5				
7	7	7	8	11.5	8.3	1	16.3	4				
8	6	10	5	10	8.3	10	24.3	11				
9	4	2	12	9	7.7	9	20.7	7				
10	1	11.5	9	3	7.8	6	14.8	3				
11	2	8.5	6	2	5.5	7	14.5	2				
12	8	11.5	11	5	9.2	11	28.2	12				

^aBased on the overall magnitude of NOEL/ER-L exceedences for all chemicals (Table 3-4).

^bBased on % survival data (Figure 3.1).

^cBased on mean % fertilization data at 100% WQAS (Table 3-5).

^dBased on EC50 data from the embryological development test (Table 3-8).

^eBased on average no. species/sample (Table 3-10).

^fΣ chemistry rank and average toxicity rank and infaunal species richness rank.

3.0 Results and Discussion (continued)

stations except Station 4 (the targeted reference site) had benthic communities that were relatively species poor (defined here as ≤ 50 species per site for comparative purposes; see Table 3-14).

Contaminants that could have been responsible for the observed bioeffects at Wellfleet Harbor sites are dieldrin (Station 1), Pb (Station 1), and chlordane (Stations 1, 3, and 4). All three of these contaminants were present at concentrations that exceeded corresponding NOEL or ER-L values, which serve as threshold effects levels, though none exceeded the higher PEL or ER-M concentrations above which adverse bioeffects on a wide variety of benthic organisms are expected. Factors other than chemical contaminant loading must be considered as causes of the adverse conditions of these sediments. Concentrations of unionized ammonia and sulfide, for example, were very high at Station 1 and could have caused, or contributed to, the observed bioeffects at this site. Sulfide, in fact, was higher at all four Wellfleet Harbor sites than in any of the other harbor sites. A possible source of the high ammonia and sulfide content of these sediments, particularly in the inner harbor, is the natural decomposition of organic matter in an area of high organic loading (due to inputs from adjacent marshes) and elevated summer water temperatures (due to the shallowness of the harbor). In recent years, local residents of the Wellfleet Harbor area have observed a growing problem with siltation and stagnation in the inner harbor.

Though the strongest bioeffects were observed in Wellfleet Harbor (particularly Station 1), the sites in this harbor system had the lowest overall sediment contamination (highest rank scores for chemistry column in Table 3-15). The most contaminated sites were in Boston and Salem/Beverly Harbors. The number of contaminants that exceeded the corresponding NOEL or ER-L values ranged from 25 to 16 among Boston Harbor sites, and from 23 to 16 among Salem/Beverly Harbor sites, compared to only 1 to 3 among Wellfleet Harbor sites.

Stations 10 and 11 in Salem Harbor were the most and second-most contaminated sites with respect to the overall amounts by which contaminants exceeded their corresponding NOEL/ER-L values. Concentrations of pesticides and metals were especially high at these two sites. The third-most contaminated site was Station 5 off Deer Island in Boston Harbor. This station had among the highest concentrations of dieldrin and total DDT. Station 9 in Beverly Harbor was ranked as the fourth most contaminated site, which had the highest concentrations of total and individual PAHs.

Typically, PCBs, dieldrin, total DDT, Ag, Cu, and Zn were present at the highest concentrations among Boston Harbor sites. In comparison, acenaphthene, anthracene, fluorene, phenanthrene, benz(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, pyrene, Σ PAHs, chlordane, total DDE, total DDD, Σ DDT+DDD+DDE, As, Cr, Pb, and Hg were usually present at the highest concentrations among Salem/Beverly Harbor sites. All stations in Boston and Salem/Beverly Harbors had one or more of these contaminants present at concentrations that exceeded NOEL or ER-L values and thus, could have contributed to the observed bioeffects. These effects consisted of significant sediment (or

3.0 Results and Discussion (continued)

porewater) toxicity at all sites and the presence of a species-poor benthos at Stations 6 and 7 in Boston Harbor and Stations 10 and 11 in Salem Harbor.

Contaminants that are more likely to be causing the most ecological harm in these latter two harbor systems are Ag, chlordane, and DDT in Boston Harbor and Cr, Pb, chlordane, and DDD in Salem/Beverly Harbors. These contaminants were present at one or more sites at concentrations that exceeded the higher PEL and/or ER-M bioeffect guidelines (see Table 3-3). Ag exceeded PEL/ER-M values at all four sites in Boston Harbor. In comparison, Cr exceeded PEL/ER-M values at all sites in Salem/Beverly Harbors. Remaining PEL/ER-M exceedances were as follows: Pb (Station 11), chlordane (Stations 6, 10 and 11), DDT (Station 5), and DDD (Stations 9, 10, 11).

Stations targeted as reference sites within each harbor system received the highest overall rank scores (Table 3-15). Station 4 (Wellfleet Harbor) was ranked 10th, Station 8 (Boston Harbor) was ranked 11th, and Station 12 (Salem/Beverly Harbor) was ranked 12th based on the combined SQT data. These sites seem reasonable to use as reference sites in any future sediment quality monitoring in the region. However, as this study has shown, it is very difficult to find any nearshore depositional environment in the region that is completely free of chemical contaminant inputs or some level of ecosystem degradation.

This study also demonstrates that factors other than chemical contaminant loading must be considered as possible causes of the biologically adverse condition of sediments in some of these coastal harbor systems. High organic loading and associated increases in the ammonia and hydrogen sulfide content of sediment porewater may be important factors contributing to the high toxicity of Wellfleet Harbor sediments, which appear to have experienced far less chemical contamination than the more urbanized Boston and Salem/Beverly Harbor systems.

4.0 Acknowledgments

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