N62578.AR.000566
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DRAFT FINAL ALLEN HARBOR AND CALF PASTURE POINT MARINE ECOLOGICAL RISK ASSESSMENT VOLUME 1 OF 2 TECHNICAL REPORT NCBC DAVISVILLE RI

2/1/1996

EA ENGINEERING SCIENCE AND TECHNOLOGY

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BREAK: 3.6
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SDMS DocID 467844

Draft Final

Allen Harbor Landfill and Calf Pasture Point Marine Ecological Risk Assessment Report:

Volume I: Technical Report

Naval Construction Battalion Center Davisville, Rhode Island

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February 1996

29600.32.3392

Draft Final

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Volume I: Technical Report

Naval Construction Battalion Center
Davisville, Rhode Island

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February 1996

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1.0 EXECUTIVE SUMMARY

This report describes the results of a marine ecological risk assessment (ERA) conducted at the Naval Construction Battalion Center (NCBC) -Davisville, RI. The Naval facility at NCBC is located adjacent to Allen Harbor in the lower West Passage of Narragansett Bay. The site is on the National Priorities List (NPL), and a Federal Facility Agreement is in place. NCBC is being closed under Base Realignment and Closure (BRAC) procedures and must comply with requirements specified under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), the National Contingency Plan (NCP), and Rhode Island State Statutes. These regulations mandate assessment of the risk of hazardous waste disposal sites on human health and the environment and identification of appropriate cleanup levels

The ERA described in this report has been prepared following the Work/Quality Assurance Project Plan (W/QAPjP), Narragansett Bay Ecorisk and Monitoring for Navy Sites, referred to herein as the "Master Work Plan", and the site-specific W/QAPjP for Allen Harbor included as Addendum D of the Master Plan (URI/SAIC, 1995).

This risk assessment follows the organization suggested in Eco Update (U.S. EPA, 1991a) with appropriate elements from the EPA Region I Supplemental Risk Assessment Guidance for the Superfund Program (U.S. EPA, 1988) and Risk Assessment Guidance for Superfund, Volume II Environmental Evaluation Manual (U.S. EPA, 1989). These guidance documents recommend a "weight of evidence" approach to assessing potential ecological risks. The approach should be based on evaluation of constituent analytical data relative to environmental benchmarks, direct field observations, and selected field and laboratory studies from the scientific literature.

Problem Formulation

Proposed Constituents of Concern (CoCs) were identified for this investigation using a rationale which links the source (Allen Harbor Landfill and Calf Pasture Point) to potential marine receptors in Allen Harbor and Narragansett Bay through plausible exposure pathways. In this approach, Hazard Quotients (HQs) were calculated as the ratio of the concentrations in the matrices of concern (ground water, surface soil, sediment and tissues) to the appropriate benchmark (e.g., biological effects, water quality criteria).

The proposed offshore CoC list includes polyaromatic hydrocarbons (PAHs), DDE, polychlorinated biphenyls (PCBs), and some metals. Because relevant criteria, action limits, or effects levels values have not been developed for all of the compounds analyzed, only selected compounds within each of the chemical classes (PAHs) were identified during the initial screening using these benchmarks.

Sampling Plan

Sampling locations in Allen Harbor were selected to fill data gaps remaining from earlier studies; a total of 39 stations were selected to provide a spatial coverage of three primary habitats: vegetated wetlands (V), intertidal wetlands (W), and sub-tidal, offshore habitats (D). In addition to the Allen Harbor stations, similar habitats also were sampled at two reference sites. The mid-bay reference site, Coggeshall Cove (PCC) on Prudence Island, is due east-northeast of Allen Harbor. A second reference site, at Jamestown Sheffield Cove (JSC), was selected as a baseline unimpacted site. This latter site has similar hydrographic characteristics as Allen Harbor, but lacks the influences of significant industrial development.

Exposure Assessment

The distribution of sediment and tissue CoCs from the Phase III study exhibit patterns which are consistent with a transport pathway from the Allen Harbor landfill to proximal wetland/intertidal zones with subsequent redistribution and deposition of a portion of the CoCs to the immediate offshore ("deep") areas of the landfill and northern embayment (between the landfill and Calf Pasture Point). There was no evidence of significant down core (i.e., buried) contamination in these offshore areas. In contrast, the sediments and shellfish tissues at intertidal and subtidal sites adjacent to Calf Pasture Point do not generally contain concentrations above those expected from regional input sources. Surface water sample results also did not show elevated CoC concentrations. Hence, the exposure data suggest that the Allen Harbor landfill is the principal Navy-related source of CoCs to Allen Harbor. The following discussion provides a more detailed description of these general findings.

Concentrations of total PAHs (tPAH) in sediments of the vegetated landfill sites (V1-V4) at the Allen Harbor sites are elevated relative to those at the PCC and JSC reference sites by up to two orders of magnitude. However, the tPAH concentrations after normalization to TOC are comparable between Allen Harbor and reference sites. The relative abundances of individual PAH compounds are dominated by the high molecular weight (HMW) four- and five-ring compounds (e.g., benzofluoranthenes and pyrene), which are possibly indicative of hydrocarbon combustion products, urban runoff and/or waste sources from nearby urban areas.

Concentrations of PAHs in intertidal sediments are highest adjacent to the southern portion of the landfill. In particular, tPAH concentrations at site W1 are several fold higher than those at adjacent intertidal sites, and 17-40X higher than sediment tPAH concentrations at the reference intertidal stations. Concentrations of tPAH and HMW-PAHs at sites W1-W5 exceed the respective Effects Range-Low (ER-L) values, criteria which describe the lower 10th percentile of observations on sediment

toxicity for a particular compound. The concentrations of HMW-PAHs at site W6 also exceed the respective ER-L value (1700 ug/kg). The tPAH concentrations in subsurface sediments are consistently lower by several-fold than those in the corresponding surficial sediments from the offshore sites. Normalized and non-normalized concentrations of tPAH exhibit clear distribution patterns relative to southern landfill proximity. Higher molecular weight PAH compounds, suggestive of combustion, urban runoff, and/or heavier lubricating or bunker fuel sources are the primary contributors to the tPAH concentrations in these sediments.

Sediment DDE concentrations in the vegetated wetlands sites near the Landfill are several fold higher than those at Calf Pasture Point and at the two reference sites. Similarly, DDE concentrations at the intertidal sites are highest near the landfill, but relatively low around Calf Pasture Point and at the reference sites.

Spatial distributions of total PCB (tPCB) concentrations in sediments from the subtidal offshore sites are similar to distributions of tPAHs and DDE in intertidal sediments, as well as to the apparent depositional patterns within the Harbor. Additionally, an unusually high concentration of chlorobenzene is present in sediments at intertidal site W6.

Concentrations of selected metals in intertidal surface sediments indicated increased metal concentrations for station W4-W6 and W1-W2. Concentrations of zinc, copper, and lead in sediments from a number of vegetated wetlands and intertidal stations (V1-V2, W1-W2, and W4-W6) all exceed the ER-L criteria. Similarly, zinc, copper, and lead concentrations at the several subtidal stations (D5-D10, D14) also exceeded the ER-L. Copper concentrations in surface and subsurface sediments at several of the other subtidal stations (e.g., D1, D11, and AH5) exceeded the ER-L. Chromium and nickel concentrations at Station W5 were also higher than the ER-L values. Several subtidal stations in the northern portion of the harbor (D7, D8, and D10) also contained nickel concentrations that exceeded the ER-L. Thus, there is a

general pattern of apparent landfill-associated metal enrichment. In contrast, this trend is not readily apparent in subsurface offshore sediments.

Bioavailability (hence potential toxicity) of sediment metals is correlated with divalent metal activity in interstitial water. To estimate divalent metal concentrations, simultaneously extracted metals/acid volatile sulfide (SEM/AVS) measurements on sediments are conducted. SEM/AVS ratios greater than one indicate divalent metal bioavailability. Six of 32 nearshore stations (W5-W7, W13, V3, and V4), two of 14 offshore stations (D1 and D12), and both reference stations (V1, W1) had sediments with SEM/AVS ratios exceeding unity. Hence, there exists the potential for metal-related toxicity if ambient metal concentrations exceed the threshold toxic concentration.

Direct measurement of pore water metals concentration in Allen Harbor sediments reveal that the analytes Ag, Cd, Hg and Zn were near or below the Method Limit of Quantitation (MLQ) for these elements. To assess the significance of those metals with quantifiable concentrations, data were compared to Chronic Water Quality Criteria (CWQC) and reference site concentrations.

Porewater HQs at offshore, deep (D) stations indicated occasional exceedences for Pb, As, Cu, Hg and Ni. For Pb, only one station, D1, exceeded CWQC. Copper (Cu) concentrations exceeded CWQC and reference values at D2, D3, D6, D11 and D14. Mercury (Hg) exceeded CWQC and reference concentrations only at D1. Arsenic (As) exceeded CWQC at D1 while Nickel (Ni) exceeded CWQC at D8, but the reference data from Prudence Coggeshall Cove (PCCD1) indicate that high Ni and As is not uncommon to Narragansett Bay where Allen Harbor is located. Porewater HQs at offshore subsurface, deep core (DC) stations indicated occasional exceedance of CWQC and elevations relative to reference sites for Pb (DC11), Hg (DC1, DC3 and DC7-DC9) and Ni (DC-8).

Porewater HQs at wetland-intertidal (W) stations indicated exceedance of CWQC for Pb, Cu and Hg. In contrast, exceedances for Cu were observed at stations W3, W10, and W12, and HQs for W5-W8 approached unity. Ni concentrations approached the CWQC at station W5

Porewater HQs at vegetated (V) stations indicated exceedance of CWQC. Concentrations of Hg were above CWQC at all vegetated sites, but were similar to or less than the reference site HQs for this metal. In contrast, the HQ for Ni at stations V1 and V3 both exceeded CWQC and were two-fold higher than reference HQ values.

Four bivalve species, including the hard shell clam (HC), ribbed mussel (RM), soft shell clam (SC), and oyster (CV), and a fish species mixture (MF) were collected and analyzed for the same CoCs as with sediments (except VOAs). Significant variation in organics (PAHs, PCBs, pesticides) concentrations were observed among species and stations, and did not present clear spatial trends. Because organics accumulate in the lipid fraction of tissues, lipid concentration of the tissue was used to normalize residue concentrations. In general, lipid-normalized results indicate elevated and relatively uniform residue concentrations for harbor biota relative to reference site biota.

Effects Assessment

Ecological effects are quantified by evaluating the relationships between exposure patterns and resulting responses of ecological systems, as determined from measurement endpoints identified during problem formulation. This included laboratory evaluations of bulk surface sediments and pore waters were conducted using the 10-day amphipod (*Ampelisca abdita*) mortality test and the sea urchin (*Arbacia punctulata*) sperm cell toxicity test, respectively, and field-based measurements of whole organism and cell-based measurements of biota condition.

Overall conclusions from the combined amphipod and sea urchin tests indicate generally similar results: the principal stations indicating toxicity correspond to southern and northern landfill areas and Calf Pasture Point. The primary difference is that 2-4 offshore stations are associated with significant to potential toxicity based on the sea urchin test, but no offshore stations exhibit significant toxicity from the amphipod test. Studies of bivalve distribution and abundance revealed significant declines in the condition indices with proximity to the landfill. Lower condition indices were correlated with tPAH, tPCB and zinc concentrations. In addition, significantly greater incidence of the blood cell disorder, hematopoietic neoplasia (Hn), was found to occur at the southern landfill region relative to other areas in Allen Harbor and reference areas in Narragansett Bay. The following discussion provides further description of these findings.

Mean amphipod survival was significantly lower and less than or equal to 80% of the control for 4 out of the 32 stations (W5, W6, W13, and V3) and significantly lower for an additional 5 stations (W9 field duplicate, W11, and W12, V1 and V4). Thus, results from this test indicate that the primary stations with potentially significant toxicity occur mainly in intertidal and marsh areas near the southern and northern ends of the landfill and at Calf Pasture Point. No significant toxicity was observed for any offshore stations.

Sea urchin tests were conducted on surface sediment pore waters from intertidal, offshore and marsh stations. Results indicated that mean fertilization percentage was significantly less (defined as less than 50% of the control) for 3 (D1, D13, and W6) out of 32 stations. Further, for conservative reporting purposes, an additional 3 stations (D2, D10, and W8) had mean percentages that were greater than 50 but less than or equal to 70%. Duplicate analyses gave very similar results, indicating a high degree of precision. Thus, the primary stations with significant or potential toxicity occur mainly in intertidal and shallow offshore areas near the southern and northern ends of the landfill and at Calf Pasture Point.

Statistical analyses of fish length/weight data indicated that Station W5 was significantly reduced relative to other stations. However, inspection of the tissue metals data for this station did not indicate any obvious differences; an alternate hypothesis to explain the results is that the overall pattern among the stations represents natural differences in fish year classes as related to habitat preferences.

Hematopoietic neoplasia is a proliferative blood disorder of bivalve molluscs. Previous evaluations of the soft-shell clam, *Mya arenaria,* from Allen Harbor have indicated the endemic occurrence of this disease. Allen Harbor stations (W1, W3, W4, W5, W6, W8, W9, W11, W12, W13, and W14) were grouped into four designations; Landfill South - Vegetated/Wetland (LS-VW), Landfill North - Wetland (LN-W), Calf Pasture Point - Wetland (CP-W), and Calf Pasture Point - South Wetland (CP-SW), based on exposure data and geographic- and depth-related characteristics. Groups LS-VW, LN-W, CP-W, and CP-SW within Allen Harbor were compared to the "urbanized" reference area Upper Narragansett Bay, and to the "pristine" reference area Lower Narragansett Bay. Neoplasia incidence was statistically greater in the LS-VW region of Allen Harbor when compared to both Upper and Lower Narragansett Bay reference regions, suggesting adverse effects on soft shell clams inhabiting this area.

Risk Characterization

In the Risk Characterization, a weight of evidence was developed to rank various areas of Allen Harbor relative to their degree of adverse ecological impacts. The approach involved comparisons of apparent adverse impacts with conditions at reference stations, the analysis of CoC concentrations *vs* observations of adverse effects, analysis of CoC bioaccumulation, comparisons of toxicity evaluations with observed ecological effects, comparisons of exposure point concentrations with established standards and criteria for offshore media, and comparisons of exposure point concentrations with published information regarding the toxicity of CoCs.

The combined results of Exposure and Effects Assessment Phases clearly suggest significant environmental impacts to the intertidal environment in the vicinity of the Allen Harbor Landfill which extends throughout the southern edge of the landfill as well as to a limited area immediately adjacent to the north side of the landfill. In contrast, the weight of evidence did not support a finding of adverse ecological risks occurring at the Calf Pasture Pt. site, since elevated CoC concentrations did not co-occur with toxicological or field effects indicators on a station-specific basis. A review of results of the EPA Risk Assessment Pilot Study (RAPS) was conducted and is supported here; there is no indication of Navy-related ecological impacts general impacts to the central, eastern or southern portions of Allen Harbor. The RAPS study did observe toxicity in sediments collected at landfill sites, a finding corroborated by the present investigation. It is notable that the exposure-based risk for slight risk zones (LS-D, LM-D and LN-W) is higher than effects, suggesting that there exists the potential for future risk on target receptors which have yet to materialize.

Exposure indicators of risk are primarily evident based on bulk sediment chemistry including SEM/AVS ratios, and tissue chemistry, with exposure mostly associated with PAHs, PCBs, pesticides (DDE), and metals. All stations in region LS-VW have high sediment contaminant concentrations and an apparent, related trend in receptor tissues. Results providing the strongest evidence of operative causal mechanisms include highly significant, inverse correlations between bivalve condition and sediment/porewater Hazard Quotients for PCBs and PAHs; amphipod toxicity and porewater PCB HQs, increased incidence of neoplasia with PAH body residues, reduced length in fish and reduced condition in oysters with increasing body burden of metals. In each case, observations at one or more stations comprising the LS-V/W EEZ were influential in driving the response, with northern Landfill intertidal stations (LN-W) also involved in a majority of stations. In contrast, lack of co-located occurrences of significant toxicity and elevated CoC concentrations for the three Calf Pasture Point groups suggests that presently minimal impacts are occurring.

Although the study findings do provide a coherent picture of the spatial magnitude and extent of risks, elucidating the CoC drivers underlying these trends is less straight forward. While fish and oyster condition appear affected by zinc, neoplasia in clams appears related to PAH body residues. Amphipod mortality appears driven by porewater PCBs; while PAHs were discounted as a principal driver of impacts on this measurement endpoint. Causal factors underlying the sea urchin trends could not be convincingly determined. SEM/AVS measurements, as indicators of metal bioavailability, did not explain toxicity responses.

A possible factor underlying the apparent diversity of responses may be related to the spatial sampling scales. Recognizing that landfill materials were not homogeneously mixed prior to disposal, the measurement endpoints may, under such conditions, be responding to highly site-specific exposure sources and not be expected to cohesively implicate one or more CoCs. Variability among station chemistry data observed in this study does support this hypothesis. It is also possible that each measurement endpoint has differential sensitivity to the respective CoC exposure, amplifying the apparent heterogeneity of responses.

Other CoCs may be entering the Allen Harbor system through creek systems to the north and south of the site. Other system stressors, such as spills and discharges from recreational boaters, are also important factors in the overall characterization of risks to Allen Harbor, but those effects appear to be localized to the south harbor area. Hence it is concluded that the Allen Harbor landfill represents a significant, stressor to indigeneous biological communities in the immediate vicinity of the landfill, particularly the southern and northern intertidal areas. Despite this stressor, the habitat has all appearances of a viable and productive community which is quite unique for Narragansett Bay.

2.0 INTRODUCTION

This report describes the results of a marine ecological risk assessment conducted at the Naval Construction Battalion Center (NCBC) -Davisville, RI. The Naval facility at NCBC is adjacent to Allen Harbor in the lower West Passage of Narragansett Bay (Figure 2-1). The site is on the National Priorities List (NPL), and a Federal Facility Agreement is in place. NCBC is being closed under Base Reutilization and Closure (BRAC) procedures and must comply with requirements specified under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), the National Contingency Plan (NCP), and Rhode Island State Statutes. These Federal regulations mandate assessment of the risk of hazardous waste disposal sites on human health and the environment and identify appropriate cleanup levels.

2.1 Background

The Ecological Risk Assessment (ERA) described in this report has been prepared following the Work/Quality Assurance Project Plan (W/QAPjP), Narragansett Bay Ecorisk and Monitoring for Navy Sites, referred to herein as the "Master Work Plan", and the site-specific W/QAPjP for Allen Harbor included as Addendum D of the Master Plan (URI/SAIC, 1995). This assessment does not consider terrestrial, freshwater wetland, or human health risks associated with these sites; separate reports are being prepared to address these issues (EA, 1995). Rather, this assessment focuses on impacts of Navy related activities on intertidal and subtidal habitats of Allen Harbor and greater Narragansett Bay.

The Master Work Plan and the Allen Harbor Addendum collectively provide a thorough description of the approaches and methodologies utilized to conduct the ERA for Allen Harbor. The scope of this report is to present the results of the ERA and includes an overview of the sampling and analysis activities conducted in support of

the ERA. Complete descriptions of sampling and analytical methodologies are provided in the Work Plan; any deviations from the plan are noted where appropriate in this report and in the QA/QC Appendix.

This ERA follows the organization suggested in Eco Update (U.S. EPA, 1991a) with appropriate elements from the EPA Region I Supplemental Risk Assessment Guidance for the Superfund Program (U.S. EPA, 1988) and Risk Assessment Guidance for Superfund, Volume II Environmental Evaluation Manual (U.S. EPA, 1989). These guidance documents recommend a "weight of evidence" approach to assess potential ecological risks. The approach should be based on evaluation of constituent analytical data relative to environmental benchmarks, direct field observations, and selected field and laboratory studies from the scientific literature.

To guarantee that the required activities are conducted to meet these objectives, this report was developed following general guidance provided by the U.S. EPA (1989, 1992, 1994) and incorporates input provided by U.S. EPA Region I, the State of Rhode Island, and Natural Resource Trustees, representatives of which jointly constitute the Narragansett Bay Ecorisk Advisory Group (see Addenda for identification of specific group members). The scope of this ERA report includes:

1. Problem Formulation. This involves determining the nature and extent of contamination of offshore (intertidal and subtidal) media associated with Navy sources. Specifically, this activity involves identification of contaminated media, identification of constituents of concern (CoCs), evaluation of the spatial extent of contamination, identification of the ecological receptors potentially at risk from CoCs, and identification of appropriate assessment and measurement endpoints. The information generated during the Problem Formulation is integrated into a conceptual model, which identifies the possible exposure scenarios and mechanisms of ecological impact associated with the CoCs.

2. Exposure and Ecological Effects Assessments. These assessments include collection of information to quantify chemical exposures and observed or predicted ecological effects resulting from exposure. Exposure Assessment involves quantification or estimation of the concentrations of CoCs in environmental media in the exposure pathways from source to ecological receptors. Ecological Effects Assessment involves a combination of toxicological literature review, *in situ* characterizations of the status of receptor species, toxicity evaluations of exposure media, and modeling exercises to predict the occurrence of adverse ecological impact. Site-specific Exposure and Ecological Effects Assessment activities are determined based on the conceptual model developed at the end of Problem Formulation.

3. Characterization of Ecological Risks. This represents a weight of evidence approach involving comparisons of apparent adverse impacts with conditions at reference stations, analysis of CoC concentrations *versus* observations of adverse effects, analysis of CoC bioaccumulation, comparisons of toxicity evaluations with observed ecological effects, comparisons of exposure point concentrations with established standards and criteria for offshore media, and comparisons of exposure point concentrations with published information regarding the toxicity of CoCs. The results of these analyses are summarized together with information obtained during each study to characterize ecological risks associated with each Navy site in Narragansett Bay.

4. Communication of the study objectives, methods, and findings of the ERA is provided in a format which supports informed risk management decisions for each site.

Based on these guidelines, this ERA presents background information integrated with contemporary data to develop the Problem Formulation (Section 3), Exposure and Ecological Effects Assessments (Sections 4 and 5, respectively), Risk Characterization (Section 6), Summary and Conclusions (Section 7). Appendices that include raw data for Exposure and Effects Assessments (Appendices A and B, respectively), QA/QC Summary Information (Appendix C) and Geophysical Data (Appendix D) are provided.

2.2 Purpose, Scope, and Objectives

The purpose of this report is to describe the information that was collected to evaluate risks to ecological receptors in Allen Harbor and greater Narragansett Bay, from chemical wastes associated with Navy activities adjacent to Allen Harbor. The general approach taken in this investigation follows that described in the main body of the Master Work Plan (URI and SAIC, 1995).

The U.S. EPA's ERA Framework and applicable EPA Region I guidance were used to generate and interpret the data required to complete this risk assessment. The objectives are as follows:

- Assess ecological risks to the offshore environments of Allen Harbor and Narragansett Bay from chemical stressors associated with the Allen Harbor Landfill and Calf Pasture Point;

- Develop information sufficient to make informed risk management decisions regarding site-specific remedial options; and

- Support communication to the public of the nature and extent of ecological risks associated with Allen Harbor Landfill and Calf Pasture Point.

This ERA builds upon and incorporates findings of previous ERA and RI/FS studies at Allen Harbor, and specifically addresses three data gaps remaining from these earlier studies. These data gaps are as follows:

- Need to assess the risk to biological populations from exposure to chemical contaminants in surfical sediments adjacent to the landfill sites;

- Need to determine the potential migration of groundwater through bedrock to subsurface sediments of Allen Harbor and resultant ecological impacts should these sediments be resuspended through mechanical, biological, or storm-driven events; and

- Need to extend the investigation of ecological risks in Allen Harbor to include the nearshore environments of Calf Pasture Point.

The following sections present and discuss the data requirements and data products of the Allen Harbor ERA, including Problem Formulation, Exposure and Ecological Effects Assessments, and Characterization of Ecological Risks.

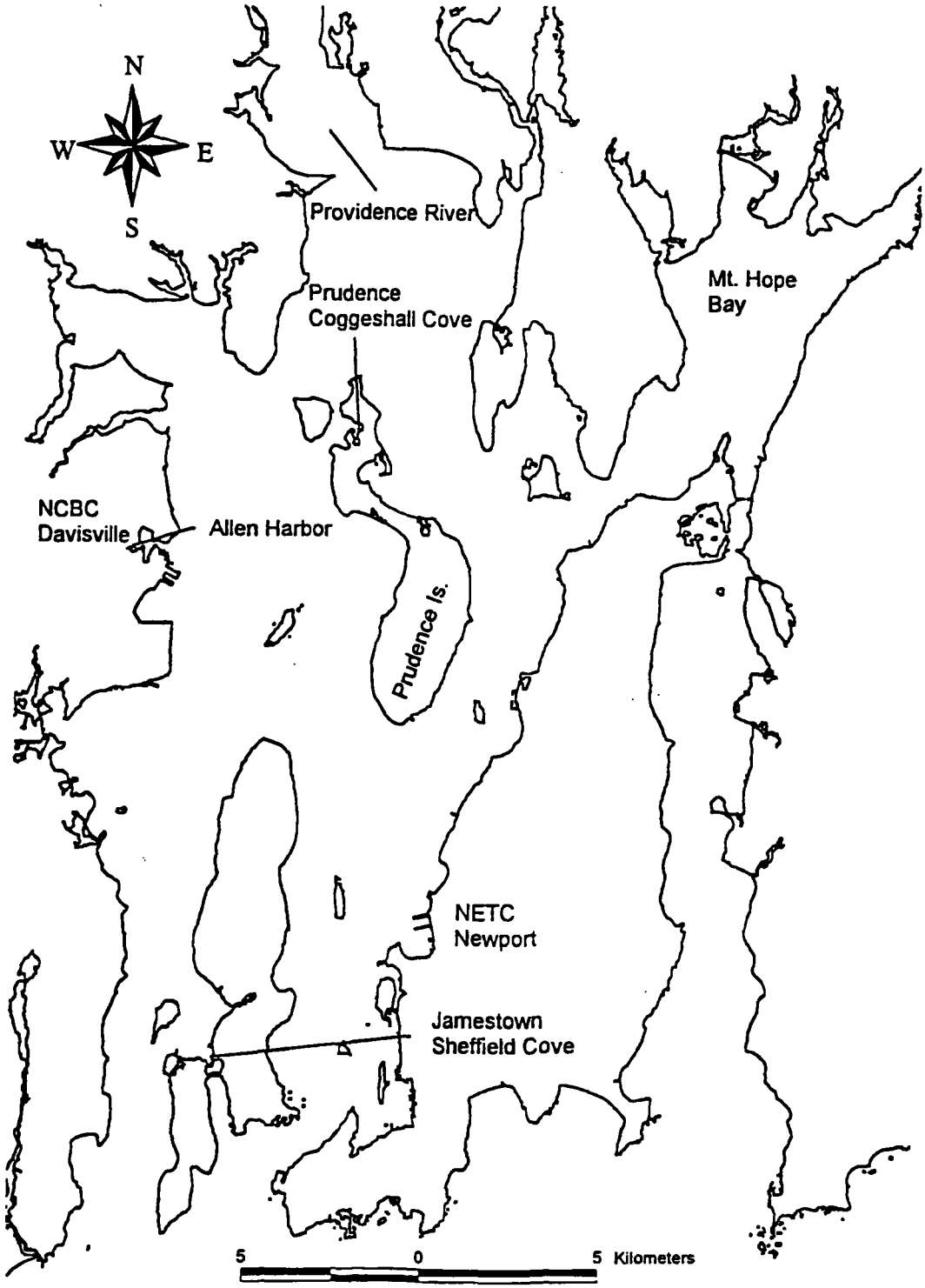


Figure 2.1-1 Allen Harbor ERA site and reference locations in Narragansett Bay.

3.0 PROBLEM FORMULATION

Five principal activities have been conducted in support of the Problem Formulation component for the Allen Harbor ERA:

- Site Description, including characterization of the nature and extent of contamination of offshore media associated with Allen Harbor Landfill and Calf Pasture Point (Section 3.1);
- Specification of assessment and measurement endpoints (Section 3.2);
- Identification of constituents of concern (CoCs, Section 3.3);
- Identification of the ecological receptors potentially at risk from site-related CoCs (Section 3.4); and
- Development of a site-specific conceptual model of ecological risks associated with the Allen Harbor Landfill and Calf Pasture Point (Section 3.5).

A summary of accomplishments from sampling and analysis activities also is provided (Section 3.6).

3.1 SITE DESCRIPTION

The primary objectives of the site characterization are to identify the types and spatial extent of marine and estuarine habitats that are present in and around Allen Harbor Landfill and Calf Pasture Point and identify the species and biological communities that may be exposed to chemical constituents present in soils,

sediments, and surface waters. Section 3.1.1 summarizes onshore site characterization information as presented in the Phase II Remedial Investigation (RI) conducted by TRC Environmental Corporation (TRC) for the U.S. Department of the Navy, Northern Division at Site 09 - Allen Harbor Landfill (TRC, 1994)..

3.1.1. NCBC Davisville Regional Geology/Hydrogeology

3.1.1.1. Geology

NCBC Davisville is located within the Narragansett geologic basin. The principal bedrock unit forming the basin is the Rhode Island formation. The formation consists of a metamorphosed grey to dark grey and greenish gneiss and shale. There are three general unconsolidated geologic units in the NCBC area: till, water-laid deposits, and wind-deposited material. The majority of the surficial geologic material in the NCBC Davisville area is water-laid deposits consisting of stratified sands and silts with variable amounts of gravel. Fine-grained glaciolacustrine deposits (lakebed deposits) underlie Calf Pasture Point. Fill materials are present at Allen Harbor landfill and at Calf Pasture Point. Wind-deposited material is not an important geologic deposit at NCBC Davisville. The glacial soils in the NCBC Davisville area contain groundwater, with the stratified sands and gravels yielding the largest quantities.

3.1.1.2. Hydrogeology

Allen Harbor is an estuarine embayment of the larger Narragansett Bay marine system and is connected to Narragansett Bay by a narrow, dredged channel. Allen Harbor landfill and Calf Pasture Point are located within the Allen Harbor Watershed. The Allen Harbor Watershed also includes salt marshes adjacent to the landfill (Site 09), upgradient salt and freshwater marshes, and the open water areas immediately north of the landfill.

Groundwater in the NCBC Davisville area is unconfined; therefore by gravity, groundwater flows from higher areas to lower areas such as local lakes, streams, swamps, or Narragansett Bay. For the NCBC Davisville area the direction of the regional groundwater flow is west to east.

The Rhode Island Department of Environmental Management (RIDEM) has classified groundwater in Rhode Island to protect and restore the quality of the state's groundwater resources for use as drinking water and other beneficial uses, and to assure protection of public health and welfare, and the environment. Groundwater in the vicinity of Site 09 has been classified as Class GB, corresponding to groundwater that may not be suitable for drinking water without treatment due to known or presumed degradation, such groundwater is primarily located in highly urbanized areas or in the vicinity of solid waste, hazardous waste, or sewage sludge disposal sites.

Surface waters surrounding Allen Harbor lie within the Hunt River drainage basin. Flow from streams and rivers at NCBC eventually discharges into Narragansett Bay. In general, the surface water quality of Narragansett Bay adjacent to NCBC Davisville is classified by RIDEM Division of Water Resources as Class SA. Class SA sea water is defined as water which supports boating and other secondary contact recreational activities, fish and wildlife habitat, industrial cooling, and which has good aesthetic value.

A wetland drainage system exists in the southern and southwestern portions of Allen Harbor and consists of reed meadow/scrub-scrub swamp and a salt marsh. The swamp is bordered by Westcott Road to the north and east and by upland wooded and deteriorating developed areas to the south and west. The marsh borders the southern shore of Allen Harbor as well as the southern edge of the landfill area, located along the western side of Allen Harbor. The swamp discharges into the salt marsh to the north via a one-meter-wide stream channel that flows under a bridge on Westcott Road. This drainage discharges to Allen Harbor through several diffuse

braided channels and hollows, as well as a sinuous central stream channel flowing through the salt marsh. The marsh also receives drainage from off-site wetlands to the west via a culverted stream channel under Sanford Road. This drainage discharges to Allen Harbor by the way of a narrow estuary.

The effect of tides associated with the proximity of Allen Harbor is most readily seen in the wetland areas at the north and south ends of the landfill. The tide flows in and out of these salt water marshes on a twice-daily basis through culverts under Sanford Road. Typical tidal fluctuations between high and low tides are on the order of 3 to 4 feet.

3.1.2. Risk Assessment Pilot Study (RAPS) results

The following site characterization for the offshore portion of Allen Harbor Landfill and Calf Pasture Point was extracted from Munns et al. (1991) for data collected at stations depicted on Fig. 3.1-1 and by TRC (1994). Although there are data compatibility issues (see Section 3.2), to the extent possible, RAPS data will be included in the evaluation of the Phase III Marine Ecological Risk Assessment Study for Allen Harbor.

3.1.2.1. Chemistry.

PCBs. Results of the RAPS Phase I survey for PCB levels in sediments, water and tissue are summarized in Table 3.1-1. Concentrations of PCBs in sediment at Allen Harbor stations ranged from 182 to 505 ppb dry wt. Highest concentrations occurred at the northern and southern ends of Allen Harbor. Intertidal stations had relatively low PCB concentrations. Concentrations in Allen Harbor sediments were 2 to 6-fold higher than the down-bay reference site at North Jamestown, but were comparable (e.g., within 2-fold) to concentrations offshore of Allen Harbor in central Narragansett Bay.

Tissue PCB concentrations followed spatial trends observed in sediments. Deployed blue mussels (*Mytilus edulis*) exhibited similar contaminant concentrations among Allen Harbor sites, indicating comparable water column exposures. Tissue concentrations in the hard shell clam, *Mercenaria mercenaria* (MERC), were similar among Allen Harbor sites, and comparable to offshore reference stations but 3- to 4-fold higher than the North Jamestown station. Concentrations of PCBs in soft shell clam (*Mya arenaria*) tissue from Allen Harbor were about 2 to 3 times higher than the offshore reference value. Oyster (OYST) tissue concentrations from three stations near the landfill base ranged 1.2 to 1.6 ppm dry weight, which were 2- to 3-fold higher than the Prudence Island reference site (PI).

Water column PCB concentrations were at or near analytical detection limits for dissolved and particulate fractions as well as in groundwater and seep water matrices.

PAHs. Results of the RAPS Phase I survey for PAH contamination in sediments, water and tissue are summarized in Table 3.1-2. In general, subtidal Allen Harbor sediments were much more contaminated than Allen Harbor intertidal sediments. Concentrations in subtidal sediments ranged from 2 to 5 times higher than the North Jamestown reference site, with Station AH-8 generally exhibiting maximum concentrations.

As with the PCB data, tissue concentrations for PAHs followed trends similar to those found in sediments. Deployed blue mussels exhibited 3- to 9-fold increases at AH5, in central Allen Harbor, over reference sites in Narragansett Bay. Tissue concentrations in the hard shell clam, were similar among Allen Harbor sites, comparable to offshore reference stations, and 2- to 4-fold higher than the Mount View (MV), Greenwich Bay (GB) and Potter Cove (PC) reference stations. Concentrations of PAHs in soft shell clam tissue were similar among sites and within a factor of 2 to 3 of the offshore reference value. Oyster tissue PAH concentrations from three stations

near the landfill ranged from non-detectable to 0.4 ppm, reflecting values that were 2- to 3-fold higher than the Prudence Island reference site (PI).

Pesticides. Results of the RAPS Phase I survey for pesticides in sediments, water and tissue are summarized in Table 3.1-3. Among the pesticide series, the DDD-DDT-DDE compounds were notable for their sediment concentrations in excess of reference values, especially at subtidal Station AH-2 and intertidal Stations AH-12 and AH-14. Similar trends were apparent in oyster tissue concentrations collected at these sites. Other species, including blue mussels, hard shell and soft shell clams, did not exhibit this pattern.

Metals. Results of the RAPS Phase I survey for metals in sediments, water and tissue are summarized in Table 3.1-4. Sediment metals concentrations in intertidal sediments at Allen Harbor Station AH14 were 100-fold higher for zinc and 50-fold higher for chromium, nickel, lead, and cadmium when compared to other Allen Harbor stations and reference stations.

Metal concentrations in bivalve tissues, with the exception of Pb and Zn, did not exhibit increases commensurate with those observed in sediments but were generally found to be similar to or less than organisms collected from reference sites. In addition, nereid worms did not show elevated body burdens at the single Allen Harbor site compared to the MV reference station.

3.1.3. TRC Phase II RI Results

Stations surveyed for contaminant concentrations in the TRC Phase II RI were selected on a watershed-by-watershed basis. Ten stations applicable to the Allen Harbor Watershed were included. Measurable concentrations of volatile organic compounds (VOAs), semivolatile organic compounds (SVOAs), heptachlor, and several metals were observed in surface waters, but the actual data were not reported

or summarized by station.

Four VOA compounds were detected in sediment samples. High PAH concentrations were measured in sediment near the base of the landfill; total PAHs were near 78,550 ppb (ug/kg) dry weight. Similarly, the highest pesticide and PCB concentrations were also detected at this station. In contrast, at another station, located further north along the landfill toe, these constituents were non-detectable.

Surveys by TRC, conducted during the ecological assessment activities for Site 09, determined that the Allen Harbor Landfill is covered with relatively sparse vegetative growth that appears to be less than ten years old, including such species as sumac, willow, *Phragmites*, red cedar, grasses, flowering cherry and autumn olive. Twenty-one bird species were observed on the landfill, while the adjacent salt marsh was characterized by several shore bird species. Debris at the face of the landfill provides a hard intertidal substrate which supports algae (e.g., *Ascophylum, Fucus*) and invertebrates (e.g., *Littorina, and Balanus*). The salt marsh benthos was reported as typical of New England salt marshes, providing a feeding area for piscivorous birds, crab-eating birds, and small mammals.

The State of Rhode Island (RIDEM, 1989) conducted an endangered species survey of East Davisville, also referred to as the Main Center. It describes the area as having fringing saline and brackish marsh, which do not provide suitable habitat for rare species, and upland areas which are slowly reverting to natural communities of shrubs.

3.1.4. Narragansett Bay Project Habitat Survey

In 1991, The Narragansett Bay Project contracted Applied Science Associates to map habitats and natural resources in and around Narragansett Bay (French et al., 1992). Aerial photo data collected in April 1988 were obtained, interpreted, and

translated into ArcInfo Geographic Information System (GIS) format. Figure 3.1-2 shows habitat classifications in the vicinity of Allen Harbor. A habitat key code and summary description is provided in Table 3.1-5. The table shows a variety of habitat types around the harbor, ranging from the landfill areas and Quonset Airport, to the fringe salt marsh and wetland salt marsh directly adjacent to Allen Harbor Landfill and Calf Pasture Point. Allen Harbor proper is classified in its entirety as dredged channel. Extensive areas of the harbor shoreline were artificially created by dredging during base construction.

3.2 Assessment and Measurement Endpoints

The Master Work Plan includes a target analyte list which was developed in response to the regulatory requirements of the RI/FS for NETC Newport and NCBC Davisville and through recognition of a number of potential chemical stressors associated with past disposal practices and other Naval operations (Table 3.2-1). The list was based upon observations of chemical constituents observed during previous offshore (*e.g.*, Quinn et al., 1994; Battelle Ocean Sciences,1994) and on-shore investigations (*e.g.*, TRC Environmental Corporation, Inc,. 1994), and includes both metals (Hg, Ni, Zn, Cu, Cd, Cr, and Pb) and organic compounds (PAHs, PCBs, butyltins, and organochlorine pesticides (OCPs)).

The list also reflects current understanding of those chemicals which are both of toxicological importance and persistent in estuarine systems. It encompasses selected potentially toxic chemicals which may serve as indicators of human activity (although for different uses) and whose discharge into the environment has been enhanced through industrialization (NOAA, 1991). In addition to the NOAA list, commonly referred to as the "Status and Trends" list, a suite of volatile organic compounds (VOCs) has been added in response to special concerns at the Allen Harbor site (e.g., inputs to Allen Harbor from leaching groundwater).

Data generated during the Risk Assessment Pilot Study (RAPS) are also included in the evaluation. These data are described in Section 3.1.2. There are a number of compatibility issues between the RAPS data and Phase III data generated for this study which greatly complicate the comparison of data between stations and areas at Allen Harbor. Data compatibility include non-identical analyte lists, and data gaps in the form of missing analytes at many stations. It should also be noted that some of the RAPS concentration data differ significantly from Phase III data for sampling locations that are within 125 meters of each other. For example, RAPS data for stations AH2 and AH3 show DDE and HCB sediment concentrations one to two orders of magnitude greater than concentrations found in the other RAPS samples, or in any of the Phase III samples. Analytes that are common to the RAPS study and the Phase III study are shown in Table (3.2-1).

Based upon the preliminary considerations of the above stressors, their potential ecological effects, and ecosystems which may be at risk, and in keeping with the requirements of the RI/FS process, a suite of assessment endpoints were then identified as being important in this assessment. As indicated in Table 3.2-2, these include the general quality of estuarine sediments and water, and the status of natural resource species.

Several measurement endpoints were employed at the Allen Harbor Landfill and Calf Pasture Point sites as indicators of the higher level ecological and societal values represented by the assessment endpoints (Table 3.2-2). The measurement endpoints have been selected based upon their relevance to:

- The assessment endpoint and receptors of concern, their relevance to expected modes of action, and effects of CoCs;

- Determination of adverse ecological effects;

- Availability of practical methods for their evaluation; and

- Their usefulness in extrapolating to other endpoints.

Most of these measurement endpoints have been used in other studies, and have proven to be informative indicators of ecological status in marine and estuarine systems with respect to the stressors identified as important in this assessment. Many serve a dual purpose in that they provide information relevant to two or more assessment endpoints.

In addition to the measurement endpoints used to evaluate the occurrence of, or potential for, adverse ecological effects, exposure point measurements were employed to evaluate exposure conditions. As shown in Table 3.2-3, these exposure point measurements include chemical measurements made in environmental media (water, sediment, pore water, and biota), as well as geochemical attributes of exposure media which may influence the availability of constituents to receptors.

3.3 Constituents of Concern

Proposed Constituents of Concern (CoCs) have been identified for this investigation using a rationale which links the source (Allen Harbor Landfill and Calf Pasture Point) to potential marine receptors in Allen Harbor and Narragansett Bay through plausible exposure pathways. In this approach, Hazard Quotients (HQs) have been calculated as the ratio of the chemical concentrations in matrices of concern (ground water, surface soil, and sediment) to the appropriate benchmarks for these media (Ambient Water Quality Criteria (AWQC), Effects Range-Low (ER-L), Proposed Action Limits (PALs) and New Jersey Environmental Clean-up Responsibility Act (NJ ECRA)). Compound-specific HQs exceeding 0.7 have been identified as constituents, which, as part of a conservative approach, are considered to pose a potential risk to ecological receptors. Identification of HQs consists of a three-step process for defining

the preliminary onshore CoCs, preliminary offshore CoCs, and proposed CoCs; each of these steps is summarized in Sections 3.3.1 through 3.3.3, respectively.

1. Using a hazard quotient (HQ) approach, chemical concentrations in onshore ground water and surface soil were compared with appropriately conservative biological benchmarks for these media to identify contaminants elevated above levels presumed to be protective of biological systems. These chemicals were designated "preliminary onshore CoCs".

2. Similarly, chemical concentrations in marine sediments, measured during the studies described above, were compared with appropriately conservative biological benchmarks for these media to identify contaminants elevated above levels presumed to be protective of biological systems. These chemicals were designated "preliminary offshore CoCs".

3. The lists of preliminary onshore CoCs and preliminary offshore CoCs were compared to identify chemicals that were common to both and those that will be designated as "proposed CoCs".

4. Preliminary CoCs not common to both lists were evaluated further for their toxicity, persistence in the marine environment, potential for bioaccumulation, and concentration relative to background levels. Those suspected of posing ecological risk based upon such characteristics were also included as proposed CoCs.

A component of this assessment was to conduct the HQ screening as described above. The final selection of CoCs for offshore exposure media will be made following completion of the Exposure Assessment.

3.3.1. Preliminary Onshore CoCs.

Using an HQ approach, chemical concentrations in onshore groundwater, seep water, and surface soil were compared with appropriately conservative water quality and soil quality criteria to identify constituents elevated above levels presumed to be protective. Chemical contaminant data are from (1) groundwater and seep samples from sites adjacent to the Landfill during the Phase I RAPS study (Munns et al., 1991), (2) from samples collected at Site 09 in the Allen Harbor Landfill during the Phase II investigation (TRC, 1994), and (3) from groundwater wells sampled during the present Phase III study. Contaminant-specific criteria used to calculate HQs are the EPA marine water quality criteria for chronic exposure for ground water and seep water matrices, and Federal Proposed Action Levels (PALs), the New Jersey Environmental Clean-up Responsibility Act (NJ ECRA).

Ratios of the maximum concentrations of several volatile organic, polycyclic aromatic, and chlorinated pesticide and PCB compounds, and trace metals in ground waters and surface soils to the corresponding criteria and/or benchmark values exceeded 0.7 (allowing for uncertainty for possible risks at HQ = 1). These chemicals were designated "preliminary Allen Harbor CoCs" as summarized in Table 3.3-1. Chemical concentrations and the associated HQs are listed in Appendix A. Similar data for Calf Pasture Point are not presently available. Organotin concentrations in ground water and surface soils from the landfill have not been measured.

- What will this be updated to incl. Site 7 -

3.3.2. Preliminary Offshore CoCs

Chemical concentrations in marine sediments (intertidal, vegetated, and offshore) measured during the studies described above were compared with nationally recognized conservative benchmarks for sediments to identify constituents elevated above levels presumed to be protective of biological systems (Table 3.3-2). For this

evaluation, the minimum of the matrix-specific criteria was used as the screening value for the particular compound.

As will be discussed in Section 5.4, sediment quality criteria presently exist for only a limited number of organic compounds. Consequently, compounds lacking national criteria were evaluated by comparing concentrations in Allen Harbor with those in the reference site sediments (Table 3.3-3). In this case, the reference values were applied on a habitat-specific basis (i.e., wetland-intertidal, vegetated wetland, deep and deep core sampling sites) using the average concentration for both sites.

The results of this screening process for the development of the preliminary offshore CoC list is summarized in Table 3.3-3. As in the previous section, only chemicals with site/reference concentration ratios exceeding 0.7 were included.

Preliminary offshore CoCs consisted of all nine trace metals, individual and summed PAHs, the chlorinated pesticides, all PCB congeners and total PCBs (sum of congeners x 2), all butyltins and all VOAs. Results from the present ERA survey were consistent with those of the RAPs study with respect to the specific classes of compounds which are elevated in marine sediments within Allen Harbor.

3.3.3. Proposed CoCs.

The lists of onshore CoCs and offshore CoCs were compared to identify chemicals in common to both (Table 3.3-4). The proposed CoC list contains all metals, PAHs, PCBs, pesticides, buttyltins and VOAs on the selected list of analytes.

3.4 Receptors of Concern

Identification of ecological systems/species/receptors of concern (hereafter termed "receptor of concern") involved evaluations of the importance of the receptor to

the ecology of Allen Harbor and Narragansett Bay, its sensitivity to stressors associated with the site, and its aesthetic and/or commercial importance as a natural resource of the harbor and bay. The site characterization for Allen Harbor identified a number of estuarine systems and habitat types (Section 3.1.4). The nature of chemical stressors originating from the NCBC sites suggests several of these to be potentially at risk, including:

- Nearshore habitats directly adjacent to landfill areas;
- Pelagic communities, including plankton and fish;
- Infaunal benthic communities in sediment depositional areas;
- Soft- and hard-bottom epibenthic communities; and
- Commercial, recreational, and/or aesthetically important natural resource species.

Added to this list are ecological systems involving critical habitats, such as eelgrass beds, seal haul-out rocks, bird rookeries, and unique spawning areas. Although French et al. (1992a, b, c) provides a bay-wide perspective of habitat types, the lack of information concerning critical habitats in immediate association with landfill sites in Allen Harbor represents a data gap which is addressed in this study.

The identification of estuarine systems and habitats potentially at risk from the Navy disposal sites in Allen Harbor provides a natural progression to the selection of target receptors of concern (Table 3.4-1). Each of these receptors meets one or more of the criteria identified above. Many of these are important resource species for Narragansett Bay, but also they can be considered surrogate receptors for larger groups of organisims. For instance, the hard shell clam, *Mercenaria mercenaria,* is an

important commercial species for Rhode Island, as well as an indicator species for infaunal bivalves in general. However, as discussed below, not all of these species occurred at all stations at each site. For example, nearshore, highly exposed habitats associated with some areas of Allen Harbor were likely to be unsuitable for soft shell clams due to the unavailability of appropriate substrate (fine-grained intertidal sediments).

Measurement endpoints directly evaluating the effects of CoCs on this avian aquatic species have not been included in this study. However, stressors introduced to the harbor may indirectly affect terrestrial and avian systems. For example, shellfish contaminated with chemicals or pathogens may be consumed by shorebirds and other animals, resulting in direct or indirect biological effects. For this reason, an avian target receptor of concern is also included in Table 3.4-1. However, the offshore ERA reported does provide information supporting the investigation of the potential impacts to this species group from ingestion of contaminated media (water, sediments, finfish and shellfish). This was accomplished by the generation of data on contaminant concentrations in media hypothesized to be part of the exposure pathways to these receptors, and also by characterizing the spatial distribution and concentration of these matrices. Hence, relevant issues for this trophic group with regard to the ERA framework are addressed from this perspective.

3.5 Conceptual Model

Conceptual models are developed to provide a framework for hypotheses concerning how a given stressor might cause ecological impacts on receptors of concern (U.S. EPA, 1992). Four models have been developed for this assessment using a tiered strategy where models in earlier tiers that are more general and inherently carry greater uncertainty progress to the more complex, "n-th tier" model which has greater complexity and certainty for the specific pathway being evaluated. In the process of further refinement of models in subsequent tiers, hypotheses are

retained or rejected based on existing knowledge of constituents and receptors of concern.

The initial three tiers describe stressor origin, transport, fate, and effects at different spatial and temporal scales: 1) the general down-stream gradient of chemical contamination, 2) initial release and transport of site-specific CoCs to the bay from Navy landfills, and 3) longer-term transport, fate, and effects of those CoCs.
The fourth and final tier models include specific receptors and stressors as identified in Section 3.2.

3.5.1. Tier 1 Model

The first tier of the conceptual model describes the general down-bay, higher-to-lower gradient in stressor concentration described earlier (Figure 3.5-1). As a result of this model evaluation, reference stations which are located in the mid- and down-bay environments have been selected to provide baseline indicators of risk under both "urbanized" and relatively pristine conditions. Although many sources contribute to this gradient, and local sources may influence specific stressor concentrations anywhere in Narragansett Bay, this model suggests that contaminant concentrations in the immediate vicinity of Navy disposal areas should be evaluated within the context of the lower Bay so that extent and significance of Navy disposal areas on the ecology of the Bay can be determined. It is assumed in this model that there are no ecologically significant stressors which are more concentrated in the south than in the north.

3.5.2. Tier II Model

The second tier of the conceptual model describes the local release of constituents from landfill sites adjacent to Allen Harbor into the harbor and greater Narragansett Bay (Figure 3.5-2.). The first hypothesis framed by this model is that CoCs are being transported from land-based sources to adjacent coves and

Narragansett Bay, predominately *via* surface water routes including stream inputs from North and West Creeks, storm drains (Spink Neck) as well as erosion of chemical pollutants bound to soil and dust particles on the landfill. A hypothesis that atmospheric deposition as well as Navy-related direct dumping (exclusive of the landfill operations) and spills are important pathways of contaminant movement into the marine environment at this site is rejected.

The geographical configuration of Allen Harbor is such that it is isolated from the main flow of tidal currents in the bay, and hence promotes sediment deposition immediately adjacent to the disposal areas. Accordingly, the restricted circulation results in two hydrographically-connected but hydrodynamically distinct spatial subsystems: the harbor and greater Narragansett Bay. The second hypothesis is that constituents released into the harbor are likely to experience a longer residence time and thus reach higher concentrations than those released directly into greater Narragansett Bay. Transport out of the harbor to lower Narragansett Bay is limited by the rate of flushing. Based on the limited hydrographic information, this transport is expected to be low, except during periods of extreme meteorological events. Thus, a localized gradient would be expected in contaminant concentrations, with the highest levels occurring in the inner harbor areas adjacent to the landfills.

The third hypothesis is that there are alternate, ecologically significant transport pathways for Navy-related stressors associated with the landfills. The decision to reject this hypothesis is based on known surface water and groundwater transport pathways, and existing data that indicate a lack of stressor- or receptor-related significant adverse effects in nearby Narragansett Bay.

3.5.3. Tier III Model

The third tier of the model describes details of the aquatic behavior of constituents hypothesized to operate in the Allen Harbor system (Figure 3.5-3). The

model arrows indicate that the short-term behavior of constituents in the water column depends on their solubility, degradation rates, and sorption to particulate matter. Individual molecules will absorb and desorb in a dynamic fashion, maintaining an apparent equilibrium relative to sorption state. Dissolved constituents are transported to other parts of the estuary by prevailing current patterns. Bound constituents will be transported horizontally in association with particles, but may also settle to the bottom in localized depositional areas.

Once on the bottom, local currents may result in bedload transport of sediment, resulting in a further redistribution of the constituents. Subsequent deposition of uncontaminated particles may bury earlier settling particles, and eventually remove them from contact with ecological systems. Partitioning dynamics similar to those in the water column will occur in the sediments and interstitial (pore) waters in response to the geochemical conditions (e.g., redox potential) of those sediments. Constituents may be available to biological systems in the water column, pore water, and surficial sediments, resulting in biological uptake and/or direct toxicological effects.

Based on this generalized conceptual model, ecosystems potentially at risk are hypothesized to include nearshore habitats, pelagic, benthic, and epibenthic communities, and natural resource species. In addition, stressor partitioning dynamics suggest that risks to receptors should be highest in nearshore/intertidal areas adjacent to the landfill sites, and that the assessment should focus on CoCs associated with depositional sediments. Stressors which conform to this model of contaminant behavior include metals, organic constituents such as PAHs, PCBs, butyltins, and pesticides.

3.5.4. Tier IV Models

The initial three tiers describe the origin, transport and fate of stressors at different spatial and temporal scales. To complete the model, receptors and stressors

specific to the Allen Harbor Landfill and Calf Pasture Point sites are added in the fourth and final tier, which describes exposure pathways (from source to receptor) hypothesized for the site.

The fourth tier describes hypothesized exposure pathways relating CoCs in the harbor to the receptors of concern identified in Table 3.4-1. These were developed for receptors by ecological habit (pelagic, epibenthic and infaunal predator), and the exposure pathways are illustrated in Figures 3.5-4 to 3.5-6, respectively. Illustrated in these figures are the routes of CoC transport from terrestrial sources, through intermediate sources (runoff, soils), to the proximal source of exposure, and to receptors. These proximal sources become the exposure points in the Exposure Assessment (Section 4.0). Also illustrated are the measurement endpoints which will be evaluated in the Ecological Effects Assessment (Section 5.0).

3.6 Sampling and Analysis Summary

3.6.1. Sediment Sampling Activities

Sampling locations in Allen Harbor are shown in Figure 3.6-1. These stations were selected to fill data gaps remaining from earlier studies; the rationale for their selection is summarized in Table 3.6-1. A total of 39 stations were used to provide a spatial coverage of three primary habitats; vegetated wetlands (V), intertidal wetlands (W), and sub-tidal, offshore habitats (D). In addition to the Allen Harbor stations, collections were also performed in each of these habitats at two reference sites. The mid-bay reference site, Coggshall Cove (PCC) on Prudence Island, is located due east northeast of Allen Harbor (Fig. 3.6-1). This site is at approximately the same distance along the north-south Narragansett Bay contamination gradient, and thus is an appropriate reference site for assessing baseline contaminant impacts in the absence of Navy activities. A second reference site, at Jamestown Sheffield Cove, was selected as a baseline unimpacted site with similar hydrographic characteristics as

Allen Harbor, but lacking the influences of significant industrial development.

A sample collection and laboratory analysis summary is shown in Table 3.6-2. Collections of surface sediments were completed at all stations from approximately mid-March to May 1, 1995. At each station, surficial sediment (top 6 cm) of an undisturbed sample was collected and composited to obtain the required volume for all analyses. A sub-sample for VOA analyses was also collected at wetland intertidal sites prior to compositing. Deep cores were collected at all subtidal stations using a standard piston corer (the biological corer). The cores were transported to the laboratory for storage at 4°C until logging and sectioning (see Section 4.2). Sectioned sediment samples were treated as described for surficial grab samples.

Composited sediment samples were subjected to the suite of measurement endpoints as discussed in Section 3.2. This included chemical analyses for total metals, simultaneously-extractable metals (SEM), acid volatile sulfides (AVS), PCBs, PAHs, pesticides, total organic carbon (TOC), pore water metals, and grain size. VOA samples were also taken but not composited.

3.6.2. Biota Sampling Activities

The biota sampling summary is also summarized in Table 3.6-2. The summary deviates somewhat from the proposed sampling plan because the actual distribution of available organisms within Allen Harbor was not known at the time the plan was developed. In all cases, biota were collected within a 10 m radius of the sediment sampling location. Bivalve sampling was attempted at all 39 stations. A mixture of ribbed mussels, oysters, soft shell and hard shell clams were obtained at the wetland intertidal sites. Ribbed mussels were the main species found at the vegetated sites, whereas hard shell clams were the only species found at the deep sites. At least one species was collected at every station. At those stations with sufficient abundance, a subset of animals were subjected to 24 hr depuration to allow voiding of gut contents.

This was possible at 14 of 39 stations. Additional samples were collected for duplicate chemical analyses.

Traps for the collection of *Fundulus* spp. were deployed at all intertidal stations and checked daily. Collection activities were somewhat impeded by non-availability of the target fish species. Our collections included a mixture of two *Fundulus* spp., *Fundulus heteroclitus* and *Fundulus majalis*. These two species, commonly called the "common mummichog" and "striped mummichog", respectively, were sufficiently abundant at five stations to permit separate analyses of tissues for metals. At other stations, either one of the two species were sufficiently abundant for analyses, while at some, pooling of the two species was necessary to provide sufficient tissue mass for chemical analyses. Potential differences in stressor exposure to the two fish species are evaluated in Section 4.3. Overall fish collections were successful at 10 of 16 proposed stations.

Assessment of hematopoietic neoplasia (Hn) in soft shell clams were completed for 9 stations in Allen Harbor, including four stations south of the Allen Harbor Landfill, three to the north of the landfill, and one station at Calf Pasture Pt. These data were compared to Hn results obtained during the RAPS study. Condition indices and size distribution data were completed at 10 stations for fish (W1-W10) and at all stations for bivalves.

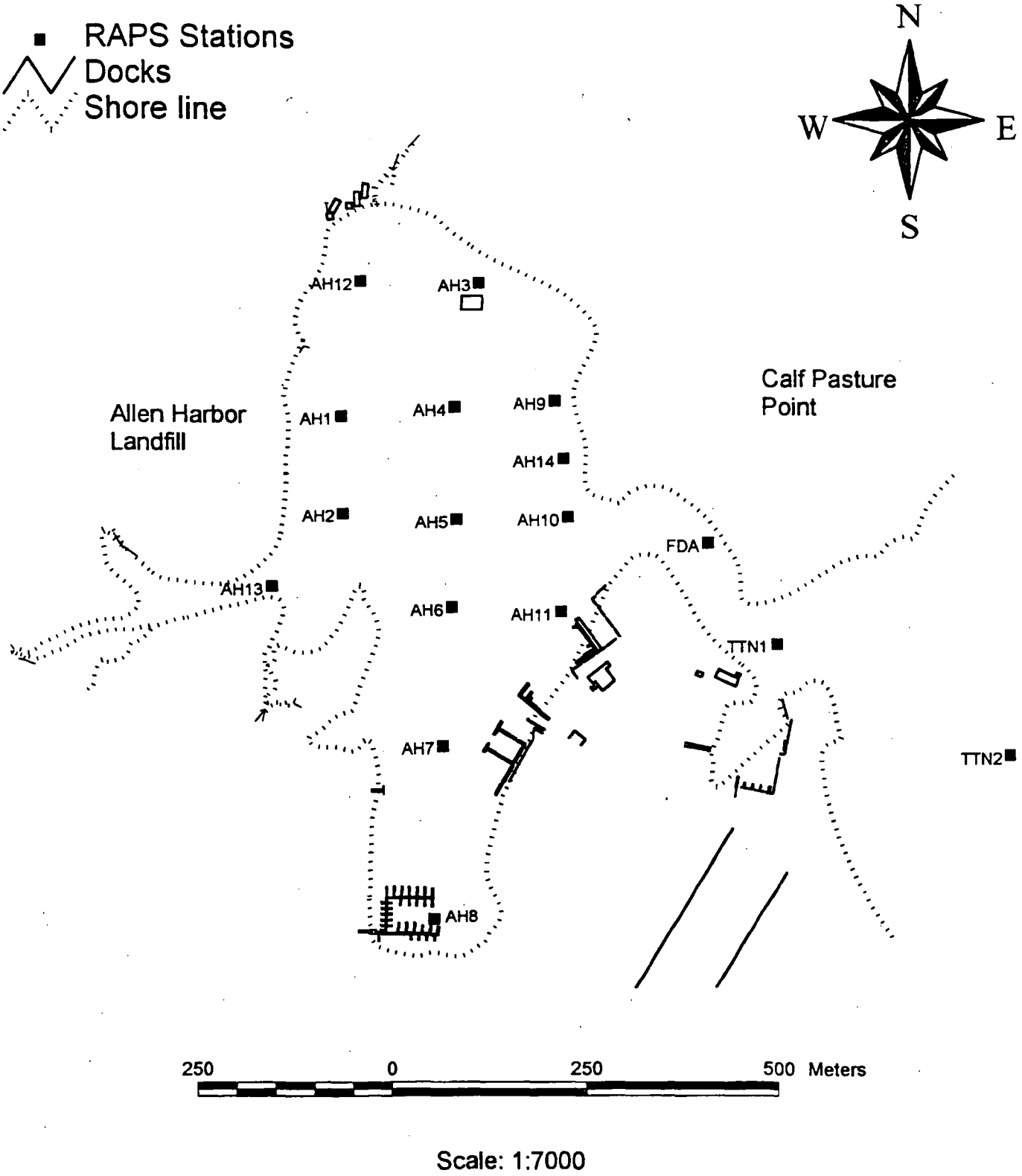


Figure 3.1-1 Locations of the intertidal and subtidal stations for the RAPS study.

Figure 3.1-2. Map of habitat types around Allen Harbor.

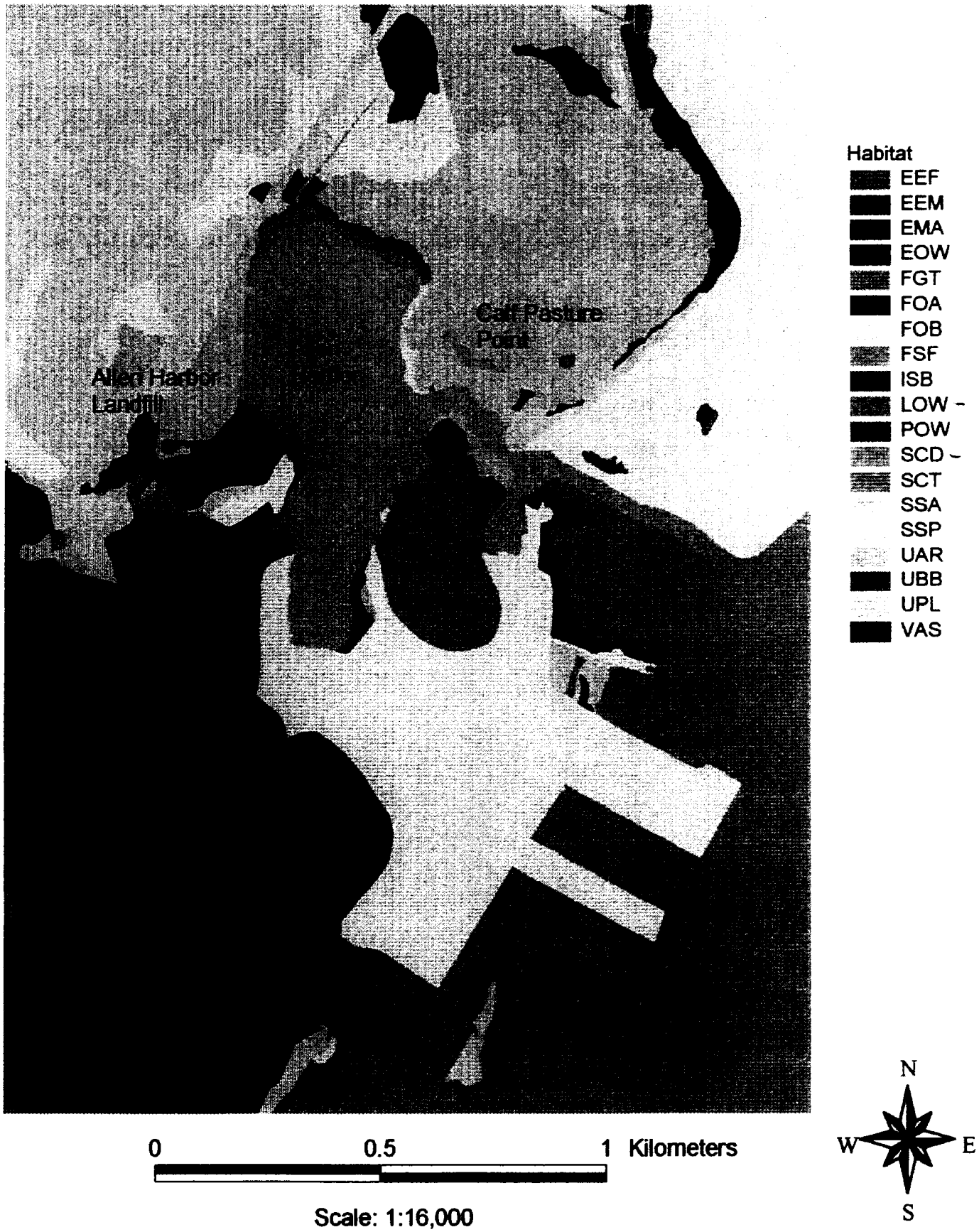


Figure 3.1-2 (continued). Habitat coding for Figure 3.1-2.

Habitat Code	Description
EEF	Estuarine emergent fringe saltmarsh
EEM	Estuarine emergent wetland (salt marsh)
EMA	Estuarine emergent wetland, marsh/wet meadow
EOW	Estuarine open water
FGT	Fringe gravel terrace
FOA	Forested conifer wetland
FOB	Forested deciduous wetland
FSF	Fringing sand flat
ISB	Intertidal sand beach
LOW	Lacustrine open water
POW	Palustrine open water
SCD	Dredged channel
SCT	Tidal channel
SSA	Scrub-shrub wetland: Shrub swamp
SSP	Subtidal sand (depositional)
UAR	Supratidal artificial –
UBB	Back beach
UPL	Upland
VAS	Macroalgal

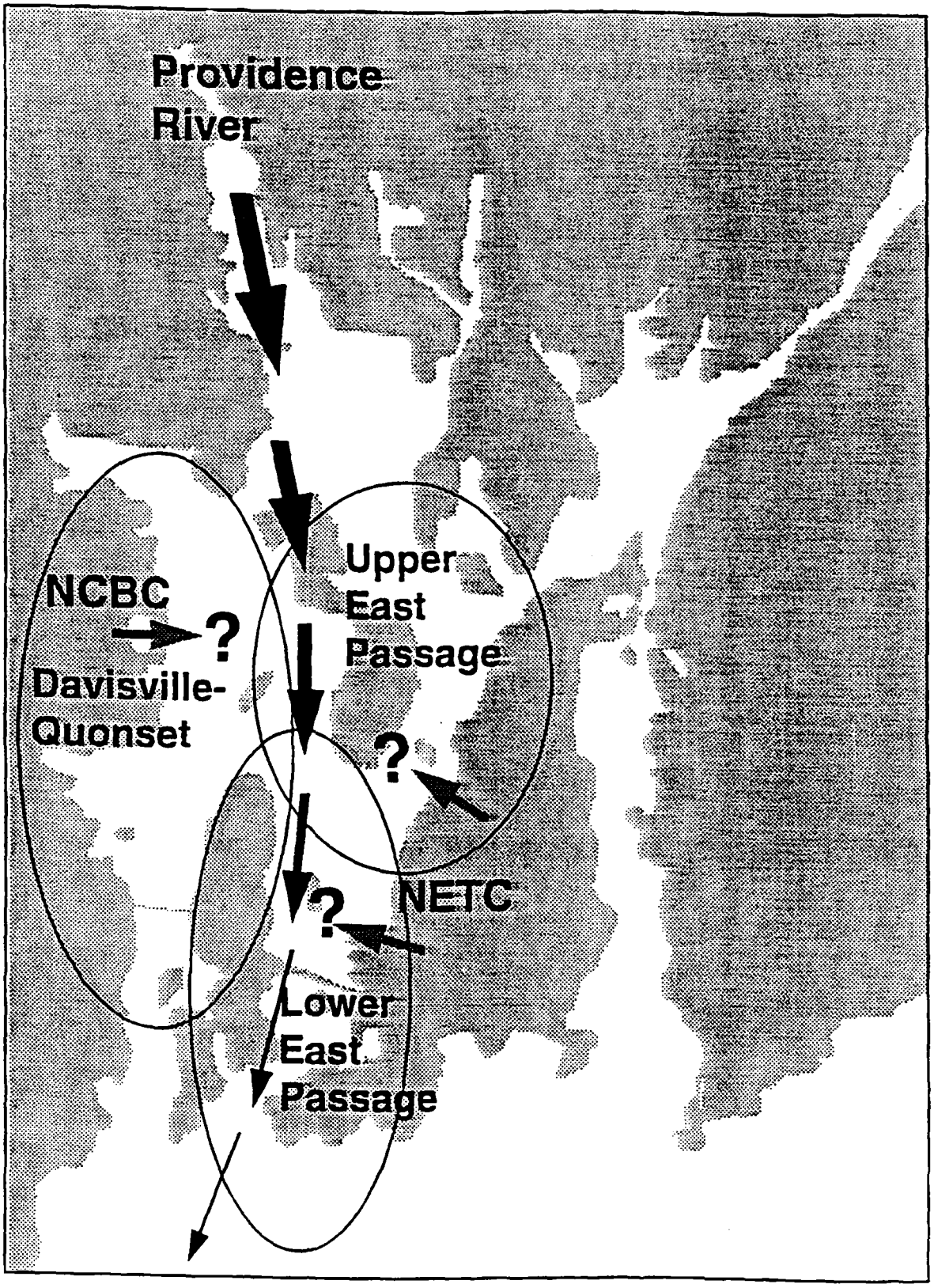


Figure 3.5-1. Relationship between Naval facilities, study areas, and the down-bay gradient of contamination in Narragansett Bay, RI.

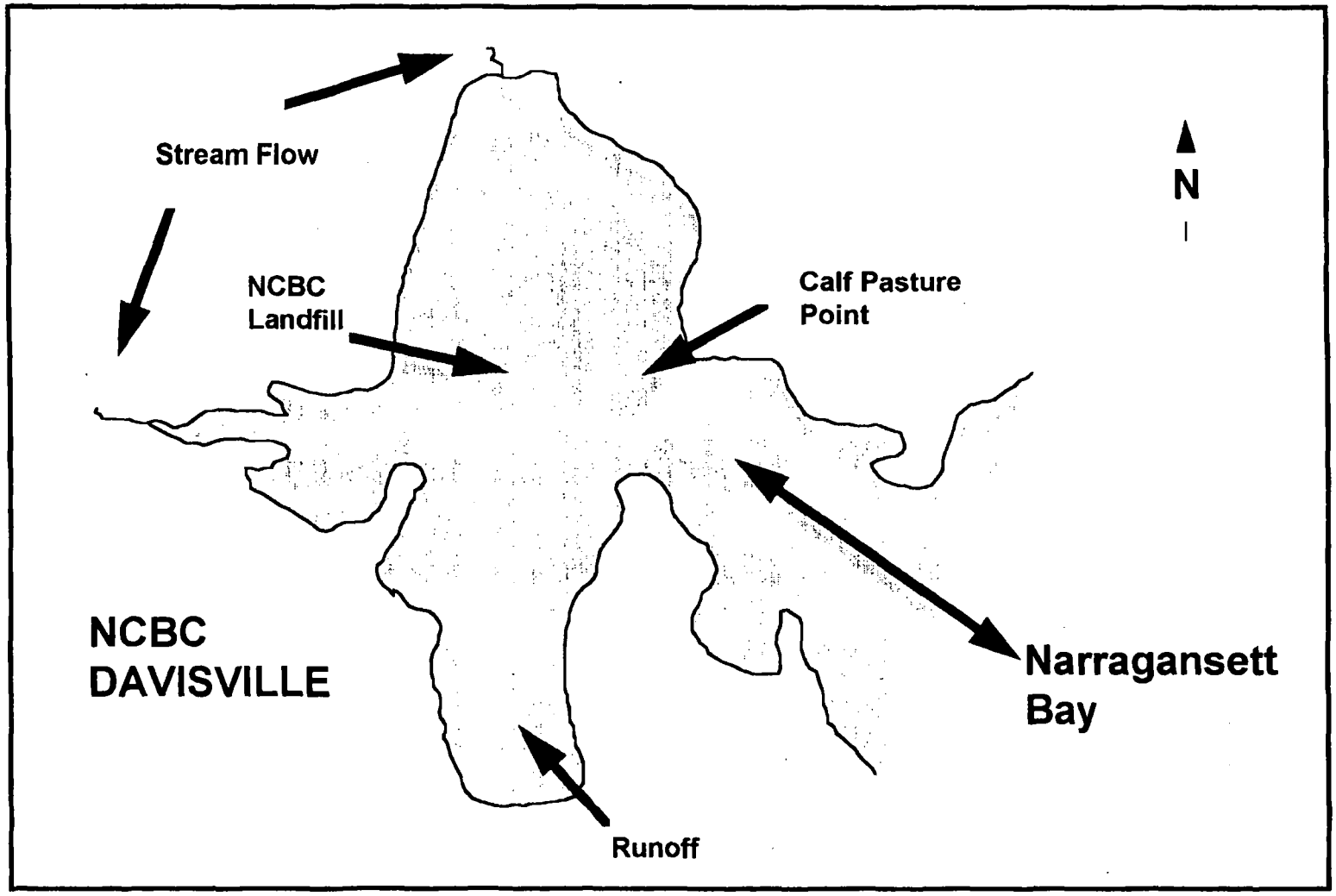


Figure 3.5-2. Second tier conceptual model of contaminant transport for the Allen Harbor Landfill and Calf Pasture Point sites.

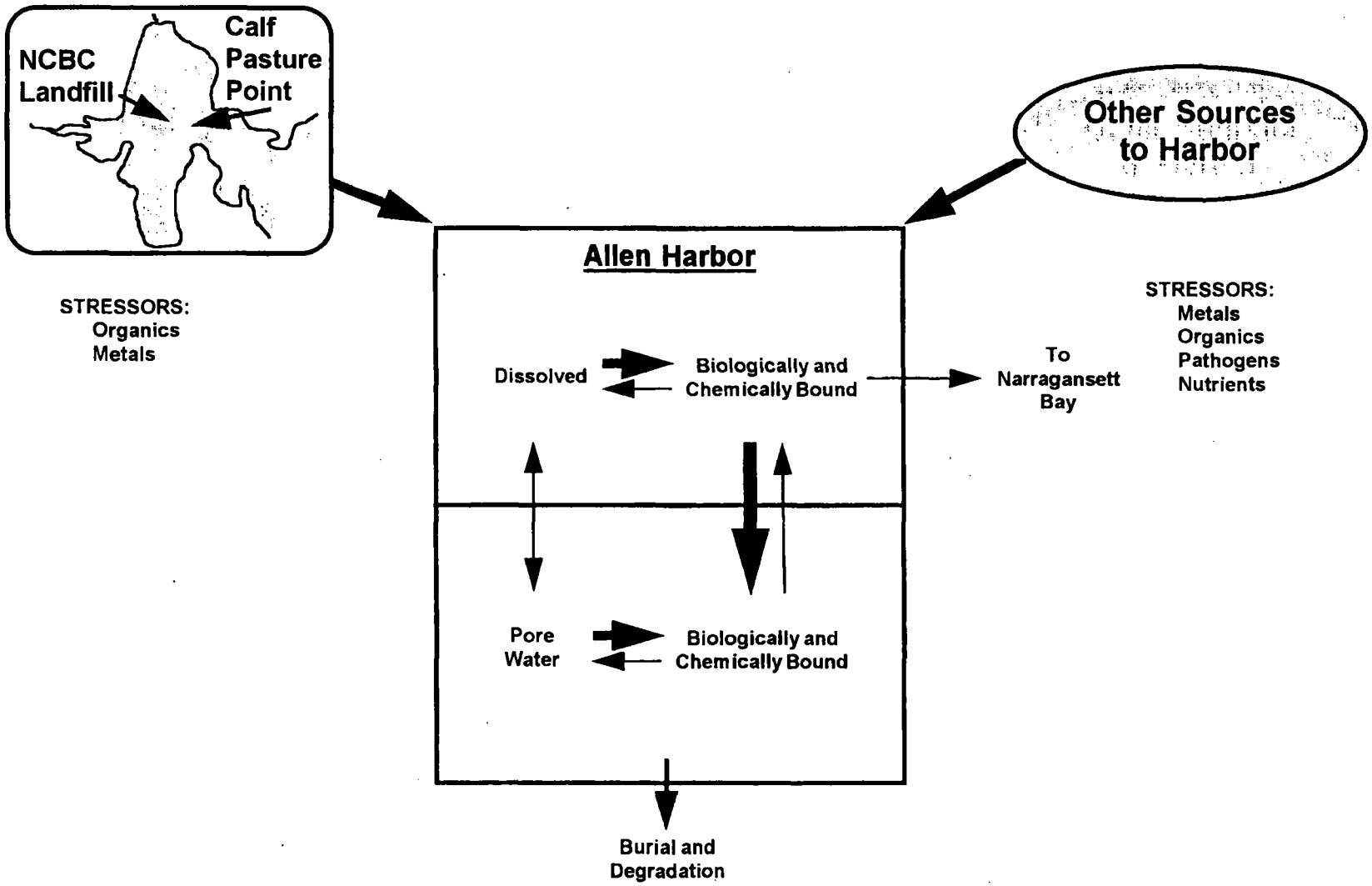


Figure 3.5-3. Third tier conceptual model of contaminant behavior for the Allen Harbor Landfill and Calf Pasture Point sites.

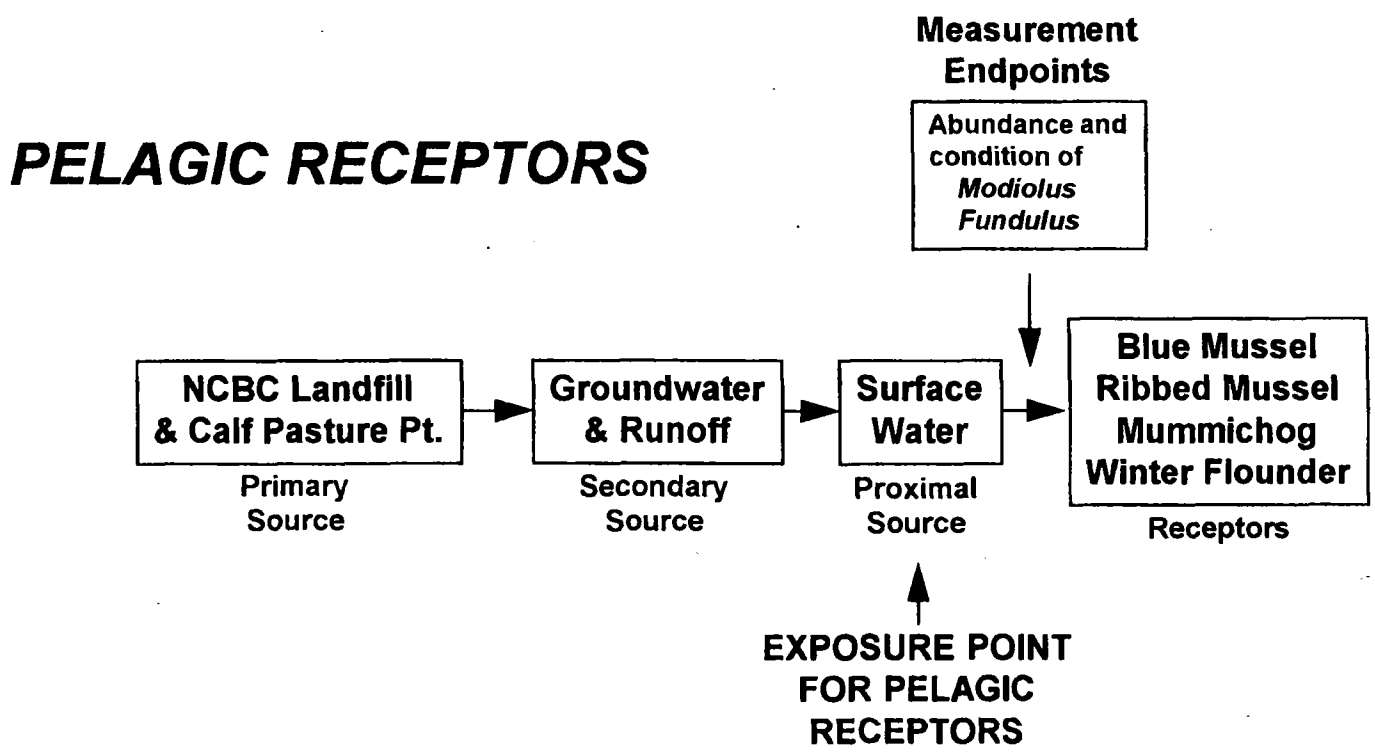


Figure 3.5-4. Fourth tier conceptual model for the Allen Harbor Landfill and Calf Pasture Point sites-exposure pathway to pelagic organisms.

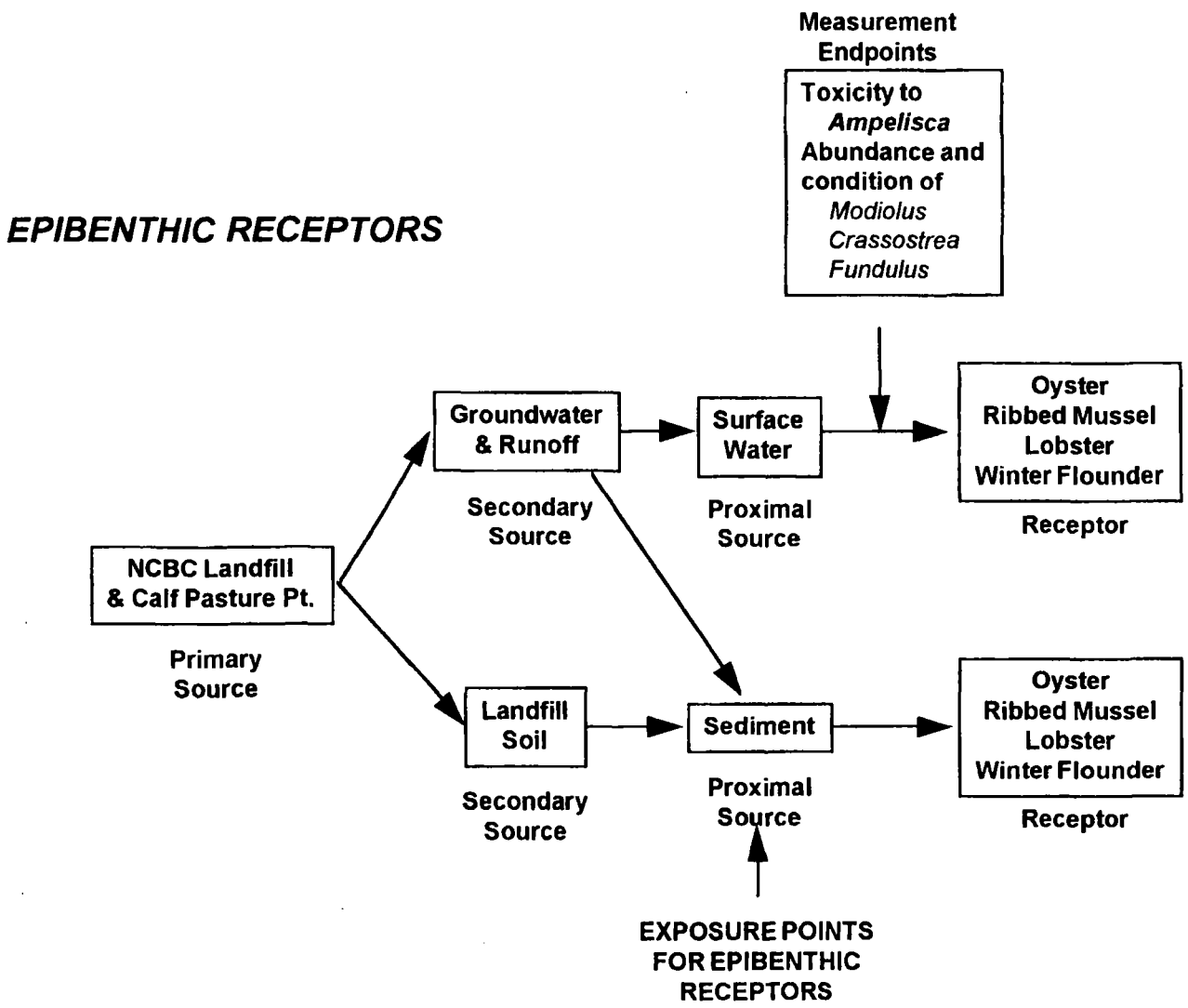


Figure 3.5-5. Fourth tier conceptual model for the Allen Harbor Landfill and Calf Pasture Point sites-exposure pathway to epibenthic organisms.

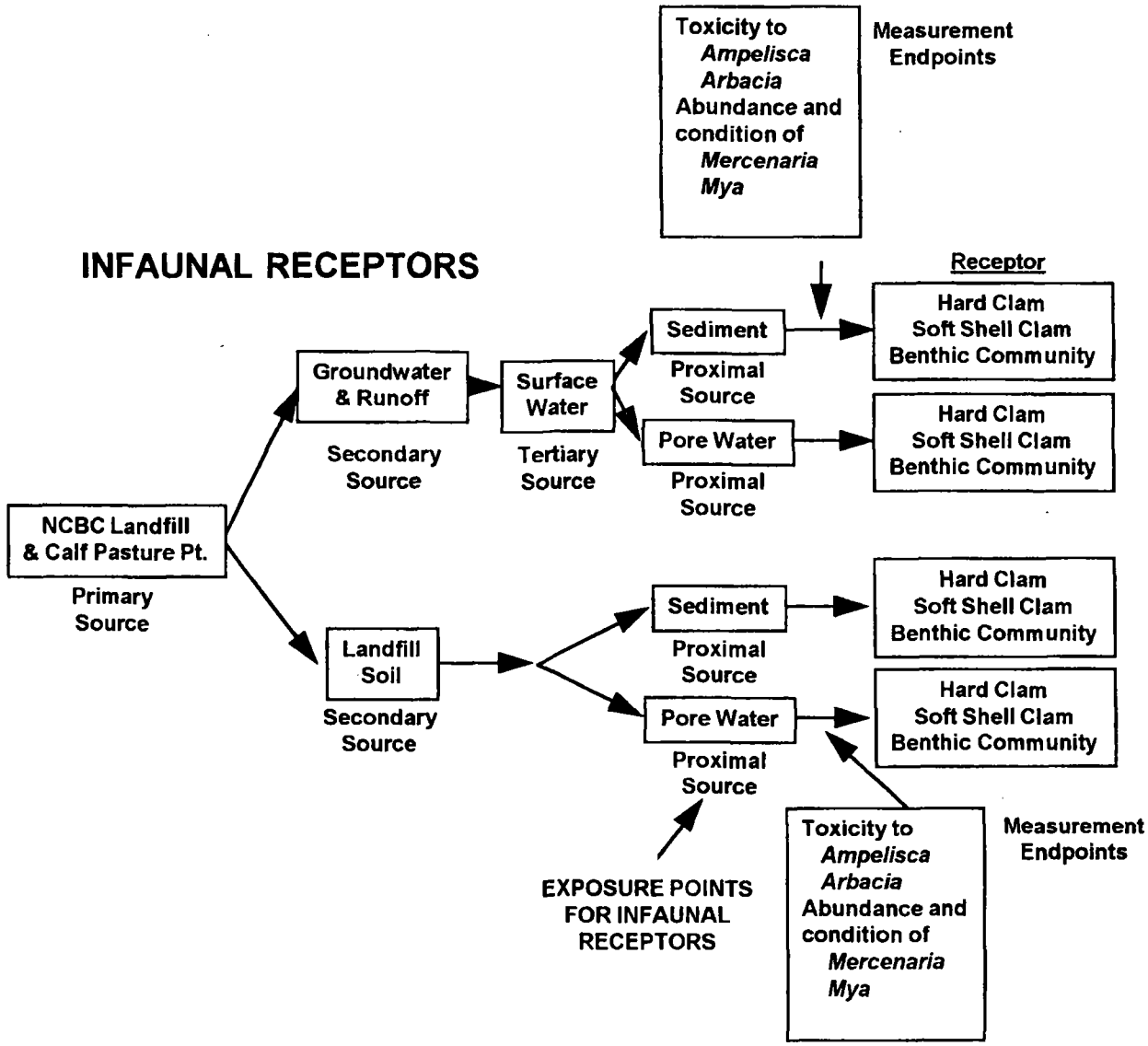
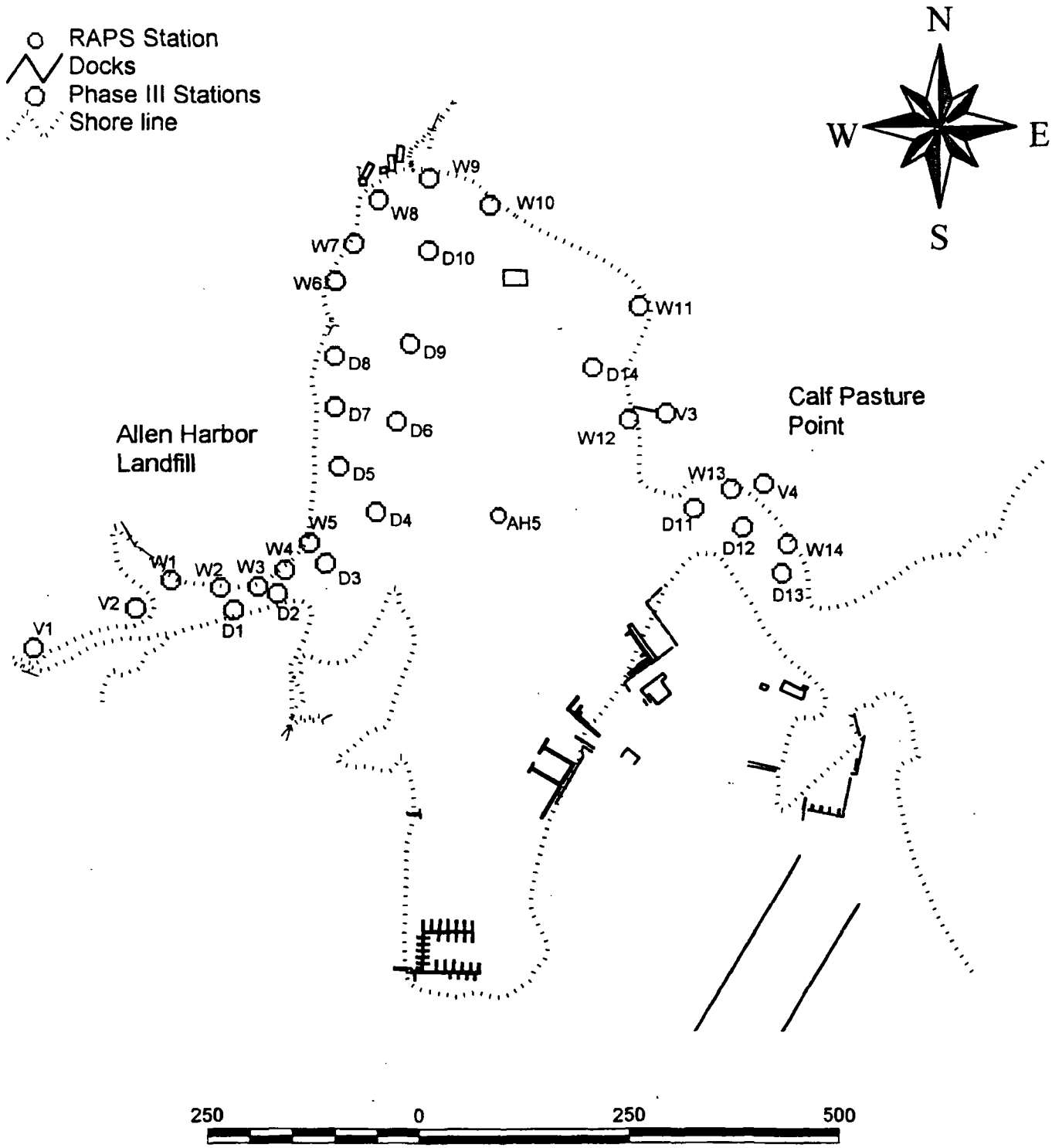


Figure 3.5-6. Fourth tier conceptual model for the Allen Harbor Landfill and Calf Pasture Point sites-exposure pathway to infaunal organisms.



Scale: 1:7000

Figure 3.6-1. Locations of vegetative (V), wetland (W) and subtidal (D) stations for the Phase III study.

Table 3.1-1. Results of the RAPS Phase I survey for PCB contamination in sediments, water and tissue (data from Munns et al., 1991). Units are ppb.

Matrix	Samptype	Station	Arochlors 1242	Arochlors 1254	Total PCB
Sediment	Intertidal	AH13	0	7.9	7.9
		FDA	0	22.1	22.1
		MP	0	8.6	8.6
Sediment	Subtidal	AH02	0	182	182
		AH03	1.2	497	498
		AH05	0	204	204
		AH08	0	505	505
		AH10	0	202	202
		GB	0	48.7	48.7
		MV	0	208.2	208.2
		NJ	0	71.9	71.9
		PC	0	348	348
Tissue	Blue Mussel	AH05	55	1071.5	1125
		LAB	34.5	983.5	1015.5
		MV	18.9	905	922.5
		TTN2	34.4	909.5	942
	Hard Shell Clam	AH02	0	149	149
		AH03	0	204	204
		AH05	0	376	376
		AH08	0	249	249
		AH10	0	235	235
		GB	0	142	142
		MV	0	158.3	158.3
		NJ	0	87.3	87.3
		PC	0	233	233
	Soft Shell Clam	AH13	0	150	150
		FDA	0	248	248
		MP	6.3	135.5	142
	Oyster	AHLM	47.8	1600	1650
		AHLN	30.2	1260	1290
		AHLW	60	1420	1480
		PI	17.3	595	606
Water	Dissolved	AH02	nd	nd	nd
		MV	nd	nd	nd
	Ground	LND	0	0	0
	Particulate	AH02	nd	0	0
		MV	nd	0	0
	Seep	LANDS	0	0.9	0.9

Table 3.1-2. Results of the RAPS Phase I survey for PAH contamination in sediments (ppb), water and tissue (data from Munns et al., 1991)

	Subtidal Sediment								
Compound	AH8	AH3	AH2	AH5	AH10	NJ	GB	MV	PC
Fluorene	50.3	18.2	24.7	9.4	13.0	15.0	1.0	10.0	13.9
Phenanthrene	811.0	231.0	198.0	137.0	165.0	152.2	7.6	82.8	115.0
Anthracene	75.8	68.7	260.0	21.6	33.9	43.4	2.1	18.1	25.6
C1-Phen+Anth	477.0	179.0	180.0	90.4	102.0	83.3	5.7	64.8	81.2
C2-Phen+Anth	357.0	134.0	120.0	72.3	82.5	59.0	5.2	56.9	71.7
C3-Phen+Anth	215.0	77.5	64.1	53.0	59.5	36.0	3.9	40.8	52.6
C4-Phen+Anth	102.0	34.1	39.1	20.3	26.5	17.6	2.3	23.3	29.8
Fluoranthene	1510.0	508.0	604.0	351.0	427.0	303.6	20.4	209.0	293.8
Pyrene	1670.0	487.0	462.0	348.0	414.0	271.4	19.1	207.0	299.2
Benz[a]anthracene	447.0	202.0	253.0	146.0	147.0	124.3	9.0	86.9	131.6
Chrysene	818.0	312.0	398.0	172.0	250.0	126.4	14.0	113.7	172.4
Benzofluorathenes	1710.0	640.0	637.0	458.0	499.0	279.6	32.1	322.0	443.0
Benzo[e]pyrene	651.0	271.0	259.0	167.0	172.0	99.3	12.1	132.3	172.2
Benzo[a]pyrene	631.0	267.0	269.0	140.0	165.0	138.7	10.5	132.2	181.6
Perylene	165.0	104.0	89.6	51.5	52.3	41.9	3.6	44.2	56.2
Indeno[1,2,3-cd]pyrene	600.0	233.0	271.0	152.0	146.0	124.8	13.0	196.4	257.4
Benzo[ghi]perylene	564.0	193.0	230.0	142.0	128.0	111.8	11.8	179.7	242.6
PAHs-276	1620.0	586.0	670.0	429.0	385.0	355.7	34.3	525.8	687.2
Dibenz[a,h]anthracene	187.0	68.3	75.3	48.6	47.5	36.4	3.4	52.1	82.2
PAHs-278	642.0	224.0	267.0	156.0	135.0	141.8	11.9	188.5	251.8
Coronene	159.0	32.1	58.6	49.0	28.8	41.8	4.6	101.2	170.6
PAHs-302	1010.0	139.0	245.0	312.0	275.0	296.1	27.3	610.9	1066.8
10-Benzotriazole	414.0	711.0	725.0	273.0	285.0	85.4	50.6	1254.8	2396.0
Chloro-Benzotriazole	84.9	84.0	88.1	43.4	47.9	19.1	9.7	205.2	427.8

Table 3.1-2, con't.

	Intertidal Sediment						Oyster			
Compound	AH12	AH13	AH14	FDA	CC	MP	AHLN	AHLW	AHLM	PI-Mean
Fluorene	5.4	0.1	5.1	2.0	39.0	73.8	11.2	10.9	11.0	5.7
Phenanthrene	58.9	1.1	58.0	21.3	569.0	554.9	37.2	35.7	34.5	18.6
Anthracene	8.6	0.3	9.4	9.3	119.0	234.3	6.3	7.4	5.9	2.1
C1-Phen+Anth	28.3	0.8	46.4	17.3	281.0	296.6	61.9	46.9	38.8	16.4
C2-Phen+Anth	23.7	1.1	49.1	18.8	201.0	100.1	132.0	82.6	85.3	36.4
C3-Phen+Anth	19.2	1.3	34.2	17.9	105.0	66.3	68.7	39.4	50.8	25.9
C4-Phen+Anth	11.7	0.9	14.8	9.7	42.1	20.4	19.4	12.1	14.7	11.4
Fluoranthene	139.0	3.8	245.0	51.9	1630.0	407.1	435.0	370.0	336.0	114.3
Pyrene	123.0	3.3	194.0	47.4	1450.0	322.8	216.0	180.0	158.0	58.7
Benz[a]anthracene	46.9	1.3	48.3	17.3	601.0	166.5	45.4	32.9	60.1	17.3
Chrysene	71.7	2.6	99.7	31.5	578.0	154.9	89.0	82.0	70.5	36.8
Benzofluorathenes	161.0	6.1	150.0	61.5	1210.0	212.5	21.2	17.3	25.0	22.3
Benzo[e]pyrene	60.4	2.1	56.2	24.3	437.0	48.6	16.8	11.6	14.3	11.6
Benzo[a]pyrene	61.6	1.5	47.2	21.0	703.0	112.2	1.1	1.7	1.3	1.3
Perylene	36.2	0.5	25.1	8.2	192.0	23.4	1.8	1.0	1.6	0.5
Indeno[1,2,3-cd]pyrene	63.5	1.5	45.6	21.4	554.0	54.9	0.6	0.4	0.2	1.0
Benzo[ghi]perylene	59.0	1.4	43.2	21.0	503.0	45.6	1.7	1.1	0.8	2.4
PAHs-276	178.0	4.3	131.0	62.9	1630.0	175.6	4.6	4.6	2.7	6.1
Dibenz[a,h]anthracene	22.7	0.6	16.8	6.8	168.0	30.4	0.2	0.3	0.2	0.8
PAHs-278	71.2	1.6	55.5	23.3	542.0	101.6	1.8	2.4	2.1	4.7
Coronene	20.4	0.5	14.6	7.5	173.0	13.0	0.5	0.4	0.2	0.8
PAHs-302	149.0	1.5	108.0	42.5	1160.0	138.3	2.6	2.7	2.0	2.5
10-Benzotriazole	49.6	3.1	12.6	21.9	59.0	22.0	14.9	13.7	5.9	22.9
Chloro-Benzotriazole	12.5	0.6	3.1	3.7	12.7	3.2	5.4	5.3	4.7	5.2

Table 3.1-2, con't.

	Blue Mussel				Soft Shell Clam					
Compound	AH5	MV	TTN	LAB	AH13	AH14	AH12	MP	CC	FDA
Fluorene	9.8	1.9	11.1	9.9	4.2	11.6	4.8	9.0	3.8	12.2
Phenanthrene	24.0	6.3	12.7	19.1	18.7	57.6	43.6	41.2	15.9	71.1
Anthracene	7.9	2.0	4.1	2.5	4.8	8.4	6.8	4.8	1.5	12.2
C1-Phen+Anth	40.1	8.9	17.2	15.0	27.6	54.7	47.3	59.1	21.2	178.0
C2-Phen+Anth	69.6	15.0	28.6	18.4	50.5	80.5	73.5	102.7	37.8	397.0
C3-Phen+Anth	53.3	17.6	31.1	17.2	45.0	50.8	53.1	66.5	30.4	343.0
C4-Phen+Anth	22.3	10.1	13.2	8.1	16.6	20.8	17.9	20.5	10.6	113.0
Fluoranthene	417.5	42.1	161.0	58.2	124.0	265.0	286.0	146.3	97.9	277.0
Pyrene	344.0	42.1	129.0	36.5	103.0	218.0	199.0	105.6	75.9	219.0
Benz[a]anthracene	31.4	9.3	18.3	7.4	19.5	62.5	44.8	27.2	17.3	56.9
Chrysene	62.1	11.1	28.7	13.2	32.8	65.5	72.9	40.0	26.2	79.8
Benzofluorathenes	44.0	25.4	40.0	20.9	38.3	48.9	59.8	32.4	26.5	70.0
Benzo[e]pyrene	42.8	20.7	37.7	17.5	31.9	39.1	47.4	24.1	20.4	53.6
Benzo[a]pyrene	3.7	4.9	6.9	4.2	12.9	6.8	13.9	6.5	4.9	11.7
Perylene	4.6	2.4	3.6	2.3	7.7	3.6	13.3	4.1	5.5	6.9
Indeno[1,2,3-cd]pyrene	6.0	7.3	10.1	5.5	8.3	8.2	12.7	6.2	6.9	9.8
Benzo[ghi]perylene	10.6	10.7	16.9	10.2	16.3	15.5	22.0	11.3	11.0	28.1
PAHs-276	24.2	23.0	37.9	21.5	33.0	29.6	44.0	30.5	26.2	59.6
Dibenz[a,h]anthracene	1.7	2.4	3.3	1.8	2.4	10.3	4.7	2.3	1.8	4.1
PAHs-278	9.2	8.7	12.8	8.6	12.5	0.0	15.3	14.7	7.7	21.4
Coronene	2.4	2.6	3.6	2.6	2.9	1.8	2.5	1.1	1.7	3.7
PAHs-302	9.3	10.3	14.9	10.3	2.6	7.1	14.9	4.9	9.0	9.1
10-Benzotriazole	310.0	333.9	607.5	343.5	156.0	193.0	62.7	448.0	188.0	233.0
Chloro-Benzotriazole	44.9	47.5	92.3	49.5	21.0	29.2	13.1	44.4	27.7	22.0

Table 3.1-2, con't.

	Hard Shell Clam							
Compound	AH2	AH3	AH5	AH8	AH10	GB	MV	PC
Fluorene	4.7	5.2	3.8	4.6	3.1	5.3	2.3	1.4
Phenanthrene	15.6	13.2	10.4	27.2	17.9	48.3	6.5	5.1
Anthracene	5.9	8.2	3.4	5.1	4.7	11.1	2.5	1.3
C1-Phen+Anth	31.3	33.4	24.6	27.9	37.7	27.0	18.8	12.7
C2-Phen+Anth	64.0	84.8	61.5	53.4	91.5	27.7	52.1	24.9
C3-Phen+Anth	36.2	80.2	37.3	38.2	62.5	15.7	34.4	19.3
C4-Phen+Anth	13.8	25.3	11.0	11.1	21.0	5.7	13.1	9.3
Fluoranthene	214.0	250.0	226.0	307.0	276.0	122.2	117.2	60.9
Pyrene	153.0	174.0	162.0	264.0	207.0	85.0	88.5	59.7
Benz[a]anthracene	27.6	35.0	15.1	40.0	25.1	37.5	12.0	7.5
Chrysene	31.9	39.7	29.0	48.5	40.3	27.7	19.7	14.1
Benzofluorathenes	37.1	37.7	16.7	48.4	28.3	54.0	20.4	13.5
Benzo[e]pyrene	10.0	8.5	11.7	24.0	19.0	22.0	10.4	12.1
Benzo[a]pyrene	3.5	1.8	3.1	8.4	4.7	21.2	3.4	2.7
Perylene	2.4	4.0	2.7	3.1	4.2	7.0	1.8	2.2
Indeno[1,2,3-cd]pyrene	5.1	3.5	2.1	7.8	4.0	15.5	3.3	2.6
Benzo[ghi]perylene	5.9	4.8	3.8	9.2	6.0	14.4	5.0	4.8
PAHs-276	21.8	16.2	7.3	24.5	13.2	52.0	13.3	10.2
Dibenz[a,h]anthracene	1.8	1.8	0.6	2.1	1.0	4.9	0.9	1.2
PAHs-278	7.4	13.6	3.9	10.4	4.7	21.6	4.6	3.5
Coronene	1.9	3.2	1.0	2.1	1.6	4.0	1.4	0.9
PAHs-302	12.4	8.6	3.3	9.6	5.3	34.2	4.2	3.8
10-Benzotriazole	143.0	144.0	87.1	51.6	74.1	72.2	180.8	189.6
Chloro-Benzotriazole	19.5	16.5	14.5	13.1	17.0	14.1	29.9	37.1

Table 3.1-3. Results of the RAPS Phase I survey for pesticide contamination in sediments, water, and tissue (data from Munns et al., 1991). Units are ppb.

Matrix	Sample	Station	alpha-BHC	g-BHC	a-Chlordane	g-Chlordane	DDD	DDT	DDE	HCB
Sediment	Intertidal	AH12	0.00	0.00	0.00	0.32	22.70	0.21	1.34	0.50
		AH14	0.00	0.00	0.00	0.00	26.40	0.23	1.06	0.25
		CC	0.06	0.00	6.00	0.09	4.99	0.24	0.21	0.03
		MP	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
	Subtidal	AH02	0.00	0.00	7.36	5.89	23.30	47.40	31.73	5.16
		AH03	0.00	0.00	1.70	1.05	6.38	8.29	37.63	3.03
		AH05	0.00	0.00	0.48	0.49	1.78	0.94	1.57	0.14
		AH08	0.00	0.00	1.32	1.08	4.03	0.68	3.27	0.23
		AH10	0.00	0.00	0.52	0.50	1.73	0.62	2.00	5.43
		GB	0.00	0.00	0.00	0.03	0.13	0.03	0.25	0.01
		MV	0.28	1.41	15.41	3.13	4.91	9.02	7.32	0.16
		NJ	0.00	0.03	0.04	0.12	0.40	0.31	0.59	0.02
		PC	0.00	0.02	2.83	0.88	2.87	0.36	5.46	0.33
Tissue	Blue Mussel	AH05	1.33	0.63	4.28	10.06	16.65	3.57	14.70	0.79
		LAB	0.83	0.46	6.24	6.87	9.33	3.03	10.29	0.50
		MV	0.49	0.63	8.63	9.06	12.45	1.39	12.45	0.38
		TTN	0.91	0.55	7.72	8.02	11.33	2.06	11.30	0.45
	Hard Clam	AH02	0.00	0.00	1.75	2.85	3.79	0.00	5.03	0.67
		AH03	0.00	0.00	2.56	3.29	2.14	1.53	5.86	0.79
		AH05	0.00	0.00	0.00	0.00	0.00	0.00	3.16	0.66
		AH08	0.00	0.00	0.00	1.77	0.00	0.00	0.00	0.00
		AH10	0.00	0.00	0.00	1.49	0.00	0.00	0.00	0.00
		GB	0.00	0.00	1.41	2.39	2.35	0.00	0.00	0.00
		MV	0.37	0.00	1.72	2.05	0.88	0.00	4.55	0.72
		NJ	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		PC	0.00	0.00	1.48	2.34	0.00	0.00	2.66	0.29
	Oyster	AHLM	0.93	0.66	12.40	13.20	1.56	29.40	27.20	0.00
		AHLN	0.67	0.47	8.70	9.39	8.68	17.23	19.40	0.35
		AHLW	1.00	0.75	9.74	9.76	5.02	32.80	37.00	0.00
		PI	1.01	0.56	5.04	5.74	0.00	12.90	5.60	0.00
	Soft Shell	AH13	0.58	0.30	1.00	2.10	56.20	1.49	2.40	0.47
		AH14	0.52	0.32	0.00	1.79	38.50	0.00	4.14	0.57
		CC	0.64	0.63	0.00	2.57	29.80	0.00	2.93	0.69
		MP	0.66	0.63	4.00	2.81	46.80	0.00	1.27	0.34
Water	Seep	LANDS	0.00	0.00	0.01	0.00	0.01	0.22	0.24	0.00
	Dissolved	AH02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		MV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Ground	LND	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Particulate	AH02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		MV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 3.1-4. Results of the RAPS Phase I survey for metals contamination (ppb dry) in sediments and tissue (data from Munns et al., 1991)

Matrix	Sample	Station	Cu	Zn	Cr	Pb	Ni	Cd	Mn	Fe	Ag	Hg	As
Sediment	Intertidal	AH12	24.3	79.3	13.4	19.02	9.98	0.41	96	11536	0.26	0	2.42
		AH13	8.4	43.3	5.88	5.63	5.26	0	57	84342	0	0.02	1.4
		AH14	519.2	2828	57.07	275.5	101.9	0.49	494	56658	0.25	0	10.64
		FDA	12.1	38.1	9.09	9	6.1	0.04	73	8577	0.13	0.03	2.22
		CC	12.5	28.3	10.43	7.73	6.39	0.14	67	5961	0.09	0	1.23
		MP	6	20.8	5.87	5.64	4.04	0.05	48	14178	0.07	0.02	1.14
	Subtidal	AH02	82.9	142.9	58.05	54.16	15.73	0.79	231	27015	1.16	0.27	4.11
		AH03	96.6	176.9	60.06	64.9	20.45	1.26	229	28593	1.43	0.49	3.97
		AH05	76.5	133.5	41.04	41.71	19.12	0.67	180	22058	0.74	0	7.22
		AH08	127.3	193.6	52.59	68.42	24.44	1.02	201	26015	1.03	0	9.98
		AH10	80.5	127.7	41.37	42.71	18.28	0.57	182	21480	0.89	0	6.87
		GB	12	26.2	9.99	17.59	2.67	0.04	55	3232	0.21	0	1.26
		MV	96.8	143.7	73.88	57.19	18.57	0.49	232	23376	1.32	0.27	4.19
		NJ	22.7	60.8	24.68	19.67	10.2	0.05	130	12478	0.34	0	3.32
		PC	193	237	94.39	108.96	28.71	1.66	228	23736	2.64	0	7.36
Tissue	Blue Mussel	AH05	10.6	108.6	1.56	3.07	3.27	1.15	18	313	0.14	0	2.75
		TTN	10.4	138.7	2.76	4.52	5.1	1.28	19	449	0.09	0	2.74
		LAB	7.7	95.4	0.85	3.44	2.02	1	10	283	0.1	0	3.55
		MV	11.4	171	2.36	6.8	4.75	2.12	16	547	0.14	0	3.22
	Hard Clam	AH02	16.9	70	0.13	2.24	4.19	0.58	62	220	0.94	0.06	1.94
		AH03	20.4	123.5	0.1	2.34	5.65	0.89	46	239	1.19	0.06	1.77
		AH05	26.3	130.3	0.29	0.87	11.03	0.87	31	159	1	0	5.64
		AH08	29.4	127.7	0.66	1.3	11.55	1	43	289	0.79	0	6.03
		AH10	18.6	132.1	0.19	1.5	6.82	0.61	58	165	1.02	0	4.67
		GB	16.8	137.3	0.78	0.86	12.51	0.58	82	69	0.73	0	8.18
		MV	16.1	124.6	0.66	1.55	12.57	0.46	49	148	0.73	0.02	6.68
		NJ	13.8	120.7	1.04	1.1	13.66	0.77	78	140	1.55	0	11.05
		PC	22	106.4	0.53	2.4	10.74	0.66	35	106	0.97	0	5.7
	Soft Shell Clam	AH12	50	173	2.93	12.54	5.81	0.5	27	4859	1.1	0	5.38
		AH13	13.7	80.8	0.38	0.8	2	0.58	5	365	1.95	0.09	5.05
		AH14	23.3	100.9	2.05	2.68	3.18	0.37	14	2421	1.58	0	4.11
		FDA	15.7	86	0.7	0.62	2.26	0.43	8	318	1.37	0	4.53
		CC	20.6	85.1	2.09	3.4	3.58	0.25	16	1007	1.72	0	3.44
		MP	25.1	80.9	0.89	1.49	2.12	0.31	12	598	2.37	0.04	3.55
	Oyster	AHLM	368.2	4348	0.29	0.9	3.55	2.89	6	838	0.4	nm	3.16
		AHLN	704.4	3060	0.35	1.78	1.64	4.32	9	491	4.97	nm	1.71
		AHLW	793.1	4059	0.4	1.38	1.65	4.85	9	853	0.7	nm	2.69
		PI	139.8	3191.3	0.4	0.77	3.18	1.83	8	107	0.72	nm	3.78
	Nereid Worm	AH02	28	182.7	0.36	6.86	2.99	0.5	8	807	nm	nm	nm
		MV	26.5	215.4	0.29	9.77	4.89	0.87	12	982	nm	nm	nm

nm = not measured

Table 3.1-5. Geology and resulting habitat coding for Bay coastline for Figure 3.1-2. Coverage is a composite of geologic, wetlands and nearshore data for Narragansett Bay and have been combined to define wildlife habitats. Data Source: Vertical aerial photographs, 4800 scale, taken 4-13-88,4-26-88.

Habitat Code	Description
EEF	Estuarine emergent fringe saltmarsh
EEM	Estuarine emergent wetland (salt marsh)
EMA	Estuarine emergent wetland, marsh/wet meadow
EOW	Estuarine open water
FGT	Fringe gravel terrace
FOA	Forested conifer wetland
FOB	Forested deciduous wetland
FSF	Fringing sand flat
ISB	Intertidal sand beach
LOW	Lacustrine open water
POW	Palustrine open water
SCD	Dredged channel
SCT	Tidal channel
SSA	Scrub-shrub wetland: Shrub swamp
SSP	Subtidal sand (depositional)
UAR	Supratidal artificial
UBB	Back beach
UPL	Upland
VAS	Macroalgal

TABLE 3.2-1. Target analytes for chemical characterization.

Analyte		Sample matrix	Target method detection limits[a]
Polycyclic Aromatic Hydrocarbons (PAHs)			
		sediment	5 ng/g
		biota	10 ng/g
naphthalene	fluoranthene		
2-methylnaphthalene	pyrene		
1-methylnaphthalene	benz[a] anthracene		
biphenyl	chrysene		
2,6-dimethylnaphthalene	benzo [b] fluoranthene		
acenaphthylene	benzo [k] fluoranthene		
acenaphthene	benzo [e] pyrene		
1,6,7-trimethylnaphthalene	benzo [a] pyrene		
fluorene	perylene		
phenanthrene	indeno [1,2,3-cd] pyrene		
anthracene	dibenz [a,h] anthracene		
1-methylphenanthrene	benzo [ghi] perylene		
Organo-Chlorine Pesticides (OCPs)		sediment	1 ng/g
		biota	2 ng/g
Aldrin			
hexachlorobenzene			
Mirex			
o.p' - DDE			
p.p' - DDE			

TABLE 3.2-1. (Continued)

Analyte	Sample matrix	Target method detection limits[a]
Polychlorinated Biphenyl (PCB) Congeners		
	sediment	1 ng/g
	biota	2 ng/g
8 (2 4')[b]		
18 (2 2'5)		
28 (2 4 4')		
29 (2 4 5)		
44 (2 2'3 5')		
50 (2 2'4 6)		
52 (2 2'5 5')		
66 (2 3'4 4')		
77 (3 3'4 4')		
87 (2 2'3 4 5')		
101 (2 2'3 5 5')		
104 (2 2'4 6 6')		
105 (2 3 3'4 4')		
118 (2 3'4 4'5)		
126 (3 3'4 4'5)		
128 (2 2'3 3'4 4')		
138 (2 2'3 4 4'5)		
153 (2 2'4 4'5 5')		
154 (2 2'4 4'5 6')		
170 (2 2'3 3'4 4'5)		
180 (2 2'3 4 4'5 5')		
187 (2 2'3 4'5 5'6)		
188 (2 2'3 4'5 6 6')		
195 (2 2'3 3'4 4'5 6)		
200 (2 2' 3 3' 4 5 6 6')		
206 (2 2'3 3'4 4'5 5'6)		
209 (2 2'3 3'4 4'5 5'6 6')		
Major elements		
aluminum	sediment	0.18 µg/g
	water	75.0 µg/L
	biota	0.18 µg/g
iron	sediment	0.5 µg/g
	water	20.0 µg/L
	biota	0.5 µg/g
manganese	sediment	0.01 µg/g
	water	0.50 µg/L
	biota	0.01 µg/g

TABLE 3.2-1. (Continued)

Analyte	Sample matrix	Target method detection limits[a]
Trace elements		
copper	sediment	0.01-0.7 µg/g
nickel	water	0.5-3.0 µg/L
chromium	biota	0.01-0.7 µg/g
lead		
silver		
cadmium	sediment	0.05 µg/g
	water	0.20 µg/L
	biota	0.005 µg/g
zinc	sediment	0.003 µg/g
	water	0.10 µg/L
	biota	0.003 µg/g
arsenic	sediment	0.08 µg/g
	water	3.0 µg/L
	biota	0.08 µg/g
mercury	sediment	0.125 µg/g
	water	0.10 µg/L
	biota	0.125 µg/g
Butyltins	sediment	1.0 ng Sn/g
	biota	1.0 ng Sn/g
monobutyltin		
dibutyltin		
tributyltin		

TABLE 3.2-1. (Continued)

Analyte	Sample matrix	Target method detection limits[a]
Volatile Organic Compounds	sediment	1.0 ng/g dry
1,1-dichloroethane		
dichloroethene		
1,1,1-trichloroethane		
2-butanone		
1,2-dichloroethane		
1,2-dichloropropane		
2-hexanone		
1,1,2-trichloroethane		
acetone		
benzene		
bromodichloromethane		
bromomethane		
bromoform		
cis 1,3-dichloropropane		
carbon disulfide		
carbon tetrachloride		
chloroform		
chlorobenzene		
chloroethane		
chloromethane		
dibromochloromethane		
ethyl benzene		
methylene chloride		
4-methyl-2-pentanone		
styrene		
1,2-dichloroethene		
trans 1,3-dichloropropane		
trichloroethene		
toluene		
tetrachloroethane		
tetrachloroethene		
total xylenes		
vinyl chloride		

[a] Sediments and tissues measured on a dry weight basis.
[b] congener number (position of chlorines)

Table 3.2-2. Assessment and measurement endpoints for the Allen Harbor Landfill and Calf Pasture Point sites.

Assessment Endpoint/Receptor	Measurement Endpoint
Vitality of Pelagic Community/ Mummichog Fish Winter Flounder	 Mummichog condition Mummichog Tissue Residues
Vitality of Epibenthic Community/ Oyster Ribbed Mussel	 Oyster condition Oyster Tissue Residues Ribbed Mussel condition Ribbed Mussel Tissue Residues
Vitality of Infaunal Community/ Soft Shell Clams Hard Clams	 Soft Shell Clam Condition Soft Shell Clam Tissue Residues Hard Shell Clam Condition Hard Shell Clam Tissue Resiudes Soft Shell Clam Neoplasia Porewater toxicity to sea urchin gametes Bulk Sediment Toxicity to amphipods

Table 3.2-3. Exposure point measurements for the Allen Harbor Landfill and Calf Pasture Point sites.

Exposure Medium/ Receptor	Exposure Point Measurement
Sediment	o Bulk sediment and pore water chemistry o Bulk sediment and pore water toxicity o Geotechnical characteristics (grain size), o Ammonia o Organic carbon o SEM/AVS
Water	o Water column chemistry
Biota	o Tissue chemistry

TABLE 3.3-1. Preliminary List of Onshore Constituents of Concern for the Allen Harbor ERA

CONSTITUENT	PROPOSED FEDERAL ACTION LIMIT[1]	NEW JERSEY ECRA GUIDELINE[2]	WATER QUALITY CRITERIA- CHRONIC VALUE[3]	PHASE II SOILS[4]	PHASE 1 SEEP	PHASE I GW	PHASE II GW	PHASE III GW
PESTICIDES								
ALPHA-CHLORDANE			0.004		X			
DDT			0.001		X			
TOTAL PCBs (congeners)	90	1000	0.03		X			
POLYAROMATIC HYDROCARBONS (PAHs)								
NAPHTHALENE								
PHENANTHRENE			4.6				X	
TOTAL PAH		10,000		X				
VOLATILE ORGANIC ANALYTES (VOAs)								
1,1,2,2-TETRACHLOROETHANE			9020			X		X
TETRACHLOROETHENE			450				X	
TOTAL VOA		1000						
METALS								
ARSENIC	80	20	36	X				
CADMIUM			9.3				X	
CHROMIUM			50				X	
COPPER		170	2.9	X			X	
LEAD			8.5	X			X	
MERCURY	20	1	0.025				X	
NICKEL			8.3				X	
SILVER			0.92				X	
ZINC		350	86				X	

1- Federal Register (55 FR 30865, 27 July 1990)
2- New Jersey Environmental Clean-up Responsibility Act (ECRA) Guidance Values, (ng/g - organics, ug/g metals)
3- U.S. EPA Ambient Water Quality Criteria Summary (May, 1991)
4- PHASE I, II Data from TRC Environmental Corporation (1994)

Table 3.3-2. Published sediment and water quality screening criteria for target analytes and derived screening values (SV).

		Sediment Critera (ug/g)						Water Quality Criteria (ug/L)				
Group	Analyte	AET	AL	ER-L	ER-M	SQC	SV	FA	FC	SA	SC	SV
Metals	Arsenic	57		8.2	70		8.20	360	190	69	36	36
	Cadmium	5.1		1.2	9.6		1.20	3.9	1.1	43	9.3	1.10
	Chromium	260		81	370		81.00	1700	210	1100	50	50
	Copper	390		34	270		34.00	18	12	2.9	2.9	2.90
	Lead	450		46.7	218		46.70	82	3.2	240	8.5	3.20
	Mercury	0.41		0.15	0.71		0.15	2.4	0.012	2.1	0.025	0.01
	Nickel	140		20.9	51.6		20.90	1400	160	75	8.3	8.30
	Silver	6.1		1	3.7		1.00	4.1	0.12	7.2	0.92	0.12
	Zinc	410		150	410		150.00	120	110	95	86	86
PAHs	2-Methylnaphthalene			0.07	0.67		0.07					
	Acenaphthene	0.5	1.3	0.016	0.5	1.3	0.02		23	0.97	0.71	0.71
	Acenaphthylene	1.3	71	0.044	0.64		0.04	1688	665			665
	Anthracene	0.96	0.58	0.0853	1.1		0.09	2.78	2.2			2.20
	Benzo(a)anthracene	1.3	4	0.261	1.6		0.26	10	1			1.00
	Benzo(a)pyrene	1.6	73	0.43	1.6		0.43	5	0.08			0.08
	Benzo(b)fluoranthene	3.2					3.20					
	Benzo(k)fluoranthene	3.2	3.8				3.20					
	Chrysene		409	0.384	2.8		0.38	1020	102	1.4		1.40
	Dibenz(a,h)anthracene			0.0634	0.26		0.06					
	Fluoranthene	1.7	6.2	0.6	5.1	6.2	0.60	3980	6.16	40	16	6.16
	Fluorene	0.54	2	0.019	0.54		0.02	212	8			8.00
	High Molecular Weight PAHs			1.7	9.6		1.70					
	Low Molecular Weight PAHs			0.552	3.16		0.55					
	Naphthalene	2.1	11	0.16	2.1		0.16	1600	620			620
	Phenanthrene	1.5	1.8	0.24	1.5	1.8	0.24	30	6.3	7.7	4.6	4.60
	Pyrene	2.6	97	0.665	2.6		0.67	1010	101			101
	Total PAHs			4.00	44.79		4.00			300		300
PCBs	PCB Sum of Congeners x 2			0.0227			0.02			0.03		0.03
Pesticides	Aldrin		0.002				0.0020	3	0.22			0.22
	Hexachlorobenzene	0.022	6				0.0220	6	3.68			3.68
	Mirex							100	0.001			0.00
	p,p'-DDE			0.0022			0.0022					
VOCs	1,1,1-Trichloroethane		3.7				3.70	42300	1300	31200		1300
	1,1,2,2-Tetrachloroethane		16				16.00	20300	6900			6900
	1,1,2-Trichloroethane		12				12.00	43000	13000	9400		9400
	1,1-Dichloroethane									113000		113000
	1,1-Dichloroethene		10				10.00	108000	8614			8614
	1,2-Dichloropropane		5.6				5.60	127000	6000	10300	3040	3040
	Benzene		0.61				0.61	5300	530	5100	700	530
	Bromodichloromethane		18				18.00	43200	15215			15215
	Carbon Disulfide		0.31				0.31	2100	210			210
	Carbon Tetrachloride									50000		50000
	Chlorobenzene							2370	2100			2100
	Dibromochloromethane		21				21.00					
	Ethylbenzene	0.01	51				0.01	9090	4600	430		430
	Styrene		48				48.00	4020	6800			4020
	Tetrachloroethene	0.057	18				0.06	4990	840			840
	Toluene		4.5				4.50	5500	1000	6300	5000	1000
	Trichloroethene		0.23				0.23	40700	100			100
	Xylene (total)	0.04	19				0.04	13400	1340			1340
	trans-1,3-Dichloropropene									790	244	244

AET = Apparent Effects Threshold
AL = Equilibrium Partitioning- Aquatic Life (1 % TOC)
ER-L = Effects Range- Low
ER-M = Effects Range- Medium
SQC = Sediment Quality Criteria

WQC-FA = Water Quality Criteria = Freshwater Acute Value
WQC-FC = Water Quality Criteria = Freshwater Chronic Value
WQC-SA = Water Quality Criteria = Saltwater Acute Value
WQC-SC = Water Quality Criteria = Saltwater Chronic Value
Screening Value = Minima of matrix-specific criteria

Table 3.3-3. Preliminary offshore Constituents of Concern (CoC) for the Allen Harbor ERA.

			Deep Stations			Deep Core Stations			Vegetated Stations			Wetland Stations			CoC
Group	Analyte	Code	Max. Loc. Value	Benchmark	H.Q.*	Max. Loc. Value	Benchmark	H.Q.*	Max. Loc. Value	Benchmark	H.Q.*	Max. Loc. Value	Benchmark	H.Q.*	Screen (HQ>0.77)
Metals	Arsenic	1	12.38	8.20	1.51	16.69	8.20	2.03	28.17	8.20	3.44	27.34	8.20	3.33	CoC
(ug/g)	Cadmium	1	1.26	1.20	1.05	0.63	1.20	0.52	0.63	1.20	0.52	3.00	1.20	2.50	CoC
	Chromium	1	76.00	81.00	0.94	98.00	81.00	1.21	66.00	81.00	0.81	128.00	81.00	1.58	CoC
	Copper	1	127.30	34.00	3.74	55.00	34.00	1.62	114.00	34.00	3.35	519.20	34.00	15.27	CoC
	Lead	1	85.00	46.70	1.82	49.00	46.70	1.05	97.00	46.70	2.08	465.00	46.70	9.96	CoC
	Mercury	1	0.49	0.15	3.27	0.60	0.15	4.00	0.40	0.15	2.67	0.40	0.15	2.67	CoC
	Nickel	1	33.25	20.90	1.59	27.25	20.90	1.30	34.00	20.90	1.63	101.90	20.90	4.88	CoC
	Silver	1	3.56	1.00	3.56	0.56	1.00	0.56	1.79	1.00	1.79	3.38	1.00	3.38	CoC
	Zinc	1	229.25	150.00	1.53	142.00	150.00	0.95	221.50	150.00	1.48	2828.00	150.00	18.85	CoC
PAHs	1,6,7-Trimethylnaphthalene	2	7.51	0.13	57.77	8.30	0.08	110.67	7.78	0.54	14.41	23.76	3.73	6.38	CoC
(ng/g)	1-Methylnaphthalene	2	14.06	0.26	54.08	6.68	0.29	23.03	13.53	0.72	18.92	45.50	3.03	15.02	CoC
	1-Methylphenanthrene	2	35.00	0.19	184.21	19.92	0.13	159.36	44.09	1.64	26.97	172.65	13.77	12.54	CoC
	2,6-Dimethylnaphthalene	2	39.06	0.25	159.43	31.68	0.32	100.57	21.14	0.57	37.09	38.50	3.96	9.72	CoC
	2-Methylnaphthalene	1	23.95	70.00	0.34	13.32	70.00	0.19	20.85	70.00	0.30	49.26	70.00	0.70	CoC
	Acenaphthene	1	18.11	16.00	1.13	16.35	16.00	1.02	34.21	16.00	2.14	295.21	16.00	18.45	CoC
	Acenaphthylene	1	43.94	44.00	1.00	11.67	44.00	0.27	41.66	44.00	0.95	38.01	44.00	0.86	CoC
	Anthracene	1	260.00	85.30	3.05	42.92	85.30	0.50	90.04	85.30	1.06	564.37	85.30	6.62	CoC
	Benzo(a)anthracene	1	447.00	261.00	1.71	87.21	261.00	0.33	324.51	261.00	1.24	1509.14	261.00	5.78	CoC
	Benzo(a)pyrene	1	631.00	430.00	1.47	90.81	430.00	0.21	368.13	430.00	0.86	1518.02	430.00	3.53	CoC
	Benzo(b)fluoranthene	2	425.17	2.19	194.59	192.42	0.95	202.55	604.31	18.46	32.74	2200.78	105.65	20.83	CoC
	Benzo(e)pyrene	2	651.00	1.12	583.86	87.64	0.53	165.36	294.41	9.77	30.13	1075.58	54.65	19.68	CoC
	Benzo(g,h,i)perylene	2	564.00	1.17	484.12	65.88	0.54	122.00	238.84	8.70	27.45	1021.11	44.03	23.19	CoC
	Benzo(k)fluoranthene	2	241.93	0.51	474.37	53.68	0.24	223.67	179.53	4.56	39.37	740.96	33.72	21.97	CoC
	Biphenyl	2	8.72	0.99	8.85	7.87	0.29	27.61	7.84	0.62	12.75	18.87	1.74	10.84	CoC
	Chrysene	1	818.00	384.00	2.13	146.60	384.00	0.38	466.96	384.00	1.22	1436.22	384.00	3.74	CoC
	Dibenz(a,h)anthracene	1	187.00	63.40	2.95	18.28	63.40	0.29	65.19	63.40	1.03	284.74	63.40	4.49	CoC
	Fluoranthene	1	1510.00	600.00	2.52	187.87	600.00	0.31	519.40	600.00	0.87	3238.45	600.00	5.40	CoC
	Fluorene	1	50.30	19.00	2.65	26.19	19.00	1.38	42.82	19.00	2.25	350.66	19.00	18.46	CoC
	High Molecular Weight PAHs	1	3412.59	1700.00	2.01	1422.55	1700.00	0.84	4225.28	1700.00	2.49	19392.48	1700.00	11.41	CoC
	Indeno(1,2,3-cd)pyrene	2	600.00	1.13	530.97	68.22	0.47	145.15	256.94	8.18	31.41	1081.49	44.50	24.31	CoC
	Low Molecular Weight PAHs	1	321.16	552.00	0.58	188.68	552.00	0.34	339.78	552.00	0.62	1682.07	552.00	3.05	CoC
	Naphthalene	1	41.29	160.00	0.26	25.37	160.00	0.16	29.74	160.00	0.19	92.05	160.00	0.58	0
	Perylene	2	165.00	0.40	412.50	213.18	0.54	398.47	89.03	2.30	38.71	279.43	10.82	25.84	CoC
	Phenanthrene	1	811.00	240.00	3.38	101.40	240.00	0.42	393.44	240.00	1.64	2518.08	240.00	10.49	CoC
	Pyrene	1	1670.00	665.00	2.51	290.33	665.00	0.44	476.93	665.00	0.72	2488.48	665.00	3.74	CoC
	Total PAHs	1	3733.75	4022.00	0.93	1567.92	4022.00	0.39	4565.06	4022.00	1.14	21074.55	4022.00	5.24	CoC
PCBs	101 (2 2'3 5 5')	2	7.12	0.04	199.60	2.05	0.04	47.51	9.40	0.04	223.01	40.70	0.34	119.85	CoC
(ng/g)	105 (2 3 3'4 4')	2	2.37	0.09	27.00	0.58	0.08	7.43	2.10	0.02	90.71	4.40	0.05	88.18	CoC
	118 (2 3'4 4'5)	2	4.13	0.03	126.83	1.94	0.15	13.11	5.90	0.02	258.21	16.50	0.27	61.41	CoC
	128 (2 2'3 3'4 4')	2	2.26	0.13	17.41	0.69	0.20	3.40	2.40	0.07	35.35	9.90	0.16	63.85	CoC
	138 (2 2'3 4 4'5)	2	16.21	0.06	277.39	3.15	0.02	166.91	28.10	0.05	528.69	149.20	0.24	623.22	CoC
	153 (2 2'4 4'5 5')	2	23.50	0.06	382.11	3.39	0.07	48.02	38.30	0.03	1322.97	191.70	0.23	850.68	CoC
	170 (2 2'3 3'4 4'5)	2	13.91	0.49	28.34	4.07	0.15	27.76	32.30	0.99	32.76	112.60	2.35	47.90	CoC
	18 (2 2'5)	2	1.45	0.59	2.45	8.36	0.06	129.94	2.10	0.04	58.41	1.80	0.04	42.50	CoC
	180 (2 2'3 4 4'5 5')	2	14.25	0.03	439.77	1.69	0.04	39.05	18.80	0.02	839.29	138.60	0.14	1000.72	CoC
	187 (2 2'3 4'5 5'6)	2	7.47	0.02	347.40	0.97	0.10	9.64	11.00	0.05	216.32	76.00	0.05	1510.93	CoC
	195 (2 2'3 3'4 4'5 6)	2	1.64	0.01	127.14	1.00	0.10	9.88	1.70	0.07	24.30	10.10	0.10	96.74	CoC
	206 (2 2'3 3'4 4'5 5'6)	2	1.17	0.02	51.97	1.58	0.02	92.72	1.80	0.02	75.47	3.60	0.10	36.13	CoC
	209 (2 2'3 3'4 4'5 5'6 6')	2	2.11	0.49	4.27	3.72	0.01	609.93	3.00	0.11	26.49	6.70	0.16	41.15	CoC
	28 (2 4 4')	2	2.64	0.12	21.92	1.87	0.06	33.87	9.00	0.33	27.21	4.70	0.61	7.75	CoC
	44 (2 2'3 5')	2	0.86	0.05	16.47	0.51	0.04	12.37	1.20	0.02	74.07	5.80	0.06	94.39	CoC
	52 (2 2'5 5)	2	2.51	0.03	79.36	1.18	0.06	18.33	3.50	0.03	128.13	10.30	0.28	36.90	CoC
	66 (2 3'4 4')	2	5.37	0.05	109.75	1.30	0.09	14.15	2.20	0.05	47.57	14.70	0.16	92.60	CoC
	8 (2 4)	2	0.69	0.13	5.15	0.26	0.13	1.99	0.40	0.14	2.84	1.40	0.25	5.68	CoC
	Sum PCB Congeners x 2	1	505.00	22.70	22.25	53.04	22.70	2.34	295.20	22.70	13.00	1345.20	22.70	59.26	CoC

Table 3.3-3. continued.

			Deep Stations			Deep Core Stations			Vegetated Stations			Wetland Stations			CoC
Group	Analyte	Code	Max. Loc. Value	Benchmark	H.Q.*	Max. Loc. Value	Benchmark	H.Q.*	Max. Loc. Value	Benchmark	H.Q.*	Max. Loc. Value	Benchmark	H.Q.*	Screen (HQ > 0.77)
Pesticides	Aldrin	2	2.09	0.04	54.54	0.06	0.04	1.62	0.10	0.04	2.48	0.30	0.19	1.62	CoC
(ng/g)	Hexachlorobenzene	2	12.82	0.01	1842.12	0.11	0.00	48.53	0.40	0.10	3.94	2.10	0.02	84.34	CoC
	Mirex	2	0.11	0.02	5.71	0.11	0.04	2.93	0.50	0.03	17.86	0.10	0.04	2.24	CoC
	o,p'-DDE	1	0.19	2.20	0.09	0.39	2.20	0.18	0.60	2.20	0.27	1.10	2.20	0.50	0
	p,p'-DDE	1	37.63	2.20	17.10	2.82	2.20	1.28	21.60	2.20	9.82	23.00	2.20	10.45	CoC
Butyltins	Dibutyltin	2	4.35	1.24	3.50	1.52	0.83	1.83	2.17	0.52	4.19	13.82	5.42	2.55	CoC
(ng/g)	Monobutyltin	2	1.50	0.55	2.74	1.00	0.79	1.27	0.50	0.77	0.65	1.27	0.77	1.66	CoC
	Tributyltin	2	8.10	1.24	6.51	4.86	0.10	49.50	3.15	1.25	2.52	7.76	2.21	3.50	CoC
VOAs	1,1,1-Trichloroethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
(ng/g)	1,1,2,2-Tetrachloroethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	1,1,2-Trichloroethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	1,1-Dichloroethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	1,1-Dichloroethene	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	1,2-Dichloroethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	1,2-Dichloroethene (total)	2				10	2.45	4.08	6.3	3.60	1.75	5	2.45	2.04	CoC
	1,2-Dichloropropane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	2-Butanone	2				20	4.90	4.08	13	7.20	1.81	14.4	4.90	2.94	CoC
	2-Hexanone	2				10	2.45	4.08	6.3	2.15	2.93	22	2.45	8.98	CoC
	4-Methyl-2-Pentanone	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Acetone	2				50	12.50	4.00	32	18.50	1.73	680	12.50	54.40	CoC
	Benzene	1				6.4	2.45	2.61	6.3	3.60	1.75	7.2	1.22	5.90	CoC
	Bromodichloromethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Bromoform	2				10	3.10	3.23	6.3	3.60	1.75	7.2	2.20	3.27	CoC
	Bromomethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Carbon Disulfide	2				16	2.45	6.53	2.3	3.60	0.64	34	2.45	13.88	CoC
	Carbon Tetrachloride	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Chlorobenzene	2				10	2.45	4.08	9.9	3.60	2.75	673	2.45	274.69	CoC
	Chloroethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Chloroform	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Chloromethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Dibromochloromethane	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Ethylbenzene	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Methylene Chloride	2				69	23.50	2.94	65	16.50	3.94	147	11.45	12.84	CoC
	Styrene	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Tetrachloroethene	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Toluene	2				10	2.45	4.08	6.3	3.60	1.75	12	1.30	9.23	CoC
	Trichloroethene	2				10	2.45	4.08	6.3	2.95	2.14	3	2.45	1.22	CoC
	Vinyl Chloride	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	Xylene (total)	2				10	1.23	8.16	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	cis-1,3-Dichloropropene	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC
	trans-1,3-Dichloropropene	2				10	2.45	4.08	6.3	3.60	1.75	7.2	2.45	2.94	CoC

Codes:
1 - Minimum published benchmark value is used as sediment screening criteria.
2 - No published benchmarks are available, mean reference location value is used as sediment screening criteria.
* - H.Q. = Maximum Location Value/Benchmark.

Table 3.3-4. Proposed Constituents of Concern (CoC) for the Allen Harbor Landfill and Calf Pasture Point Phase III ERA.

Group	Analyte	ONSHORE		OFFSHORE	CoC
		GROUNDWATER	SOIL	RAPS/PHASE III	Selection
Metals	Arsenic			X	CoC
(ug/g)	Cadmium	X	X	X	CoC
	Chromium	X		X	CoC
	Copper	X	X	X	CoC
	Lead	X	X	X	CoC
	Mercury	X		X	CoC
	Nickel	X		X	CoC
	Silver	X		X	CoC
	Zinc	X		X	CoC
PAHs	1,6,7-Trimethylnaphthalene			X	CoC
(ng/g)	1-Methylnaphthalene			X	CoC
	1-Methylphenanthrene			X	CoC
	2,6-Dimethylnaphthalene			X	CoC
	2-Methylnaphthalene			X	CoC
	Acenaphthene			X	CoC
	Acenaphthylene			X	CoC
	Anthracene			X	CoC
	Benzo(a)anthracene			X	CoC
	Benzo(a)pyrene			X	CoC
	Benzo(b)fluoranthene			X	CoC
	Benzo(e)pyrene			X	CoC
	Benzo(g,h,i)perylene			X	CoC
	Benzo(k)fluoranthene			X	CoC
	Biphenyl			X	CoC
	Chrysene			X	CoC
	Dibenz(a,h)anthracene			X	CoC
	Fluoranthene			X	CoC
	Fluorene			X	CoC
	High Molecular Weight PAHs			X	CoC
	Indeno(1,2,3-cd)pyrene			X	CoC
	Low Molecular Weight PAHs			X	CoC
	Naphthalene	X			CoC
	Perylene			X	CoC
	Phenanthrene			X	CoC
	Pyrene			X	CoC
	Total PAHs	X		X	CoC
PCBs	101 (2 2'3 5 5')			X	CoC
(ng/g)	105 (2 3 3'4 4')			X	CoC
	118 (2 3'4 4'5)			X	CoC
	128 (2 2'3 3'4 4')			X	CoC
	138 (2 2'3 4 4'5)			X	CoC
	153 (2 2'4 4'5 5')			X	CoC
	170 (2 2'3 3'4 4'5)			X	CoC
	18 (2 2'5)			X	CoC
	180 (2 2'3 4 4'5 5')			X	CoC
	187 (2 2'3 4'5 5'6)			X	CoC
	195 (2 2'3 3'4 4'5 6)			X	CoC
	206 (2 2'3 3'4 4'5 5'6)			X	CoC
	209 (2 2'3 3'4 4'5 5'6 6')			X	CoC
	28 (2 4 4')			X	CoC
	44 (2 2'3 5')			X	CoC
	52 (2 2'5 5)			X	CoC
	66 (2 3'4 4')			X	CoC
	8 (2 4)			X	CoC
	Sum PCB Congeners x 2	X		X	CoC

Table 3.3-4. Proposed Constituents of Concern (CoC) for the Allen Harbor Landfill and Calf Pasture Point Phase III ERA.

DDT?

Group	Analyte	ONSHORE GROUNDWATER	ONSHORE SOIL	OFFSHORE RAPS/PHASE III	CoC Selection
Pesticides (ng/g)	Aldrin			X	CoC
	Hexachlorobenzene			X	CoC
	Mirex			X	CoC
	o,p'-DDE				
	p,p'-DDE			X	CoC
Butyltins (ng/g)	Dibutyltin			X	CoC
	Monobutyltin			X	CoC
	Tributyltin			X	CoC
VOAs (ng/g)	1,1,1-Trichloroethane			X	CoC
	1,1,2,2-Tetrachloroethane	X		X	CoC
	1,1,2-Trichloroethane			X	CoC
	1,1-Dichloroethane			X	CoC
	1,1-Dichloroethene			X	CoC
	1,2-Dichloroethane			X	CoC
	1,2-Dichloroethene (total)			X	CoC
	1,2-Dichloropropane			X	CoC
	2-Butanone			X	CoC
	2-Hexanone			X	CoC
	4-Methyl-2-Pentanone			X	CoC
	Acetone			X	CoC
	Benzene			X	CoC
	Bromodichloromethane			X	CoC
	Bromoform			X	CoC
	Bromomethane			X	CoC
	Carbon Disulfide			X	CoC
	Carbon Tetrachloride			X	CoC
	Chlorobenzene			X	CoC
	Chloroethane			X	CoC
	Chloroform			X	CoC
	Chloromethane			X	CoC
	Dibromochloromethane			X	CoC
	Ethylbenzene			X	CoC
	Methylene Chloride			X	CoC
	Styrene			X	CoC
	Tetrachloroethene	X		X	CoC
	Toluene			X	CoC
	Trichloroethene			X	CoC
	Vinyl Chloride			X	CoC
	Xylene (total)			X	CoC
	cis-1,3-Dichloropropene			X	CoC
	trans-1,3-Dichloropropene			X	CoC

X = HQ > 0.7.

CoC = Constituent of Concern

Table 3.4-1. Target ecological systems/species/receptors of concern for the Allen Harbor Landfill and Calf Pasture Point sites.

Habitat	Ecological System/Species/Receptor of Concern
Pelagic	blue mussel (*Mytilus edulis*)[1,2] mummichog (*Fundulus* spp.) winter flounder (*Pseudopleuronectes americanus*)[2]
Epibenthic	ribbed mussel (*Modiolus demissus*), oyster (*Crassostrea gigas*)
Benthic	hard shell clam (*Mercenaria mercenaria*) soft shell clam (*Mya arenaria*)
Aquatic	osprey (*Pandion haliaetus*)[4]

[1]surrogate for pelagic species when collected from mid-upper water column (e.g. mooring floats)
[2]not proposed for sampling
[3]present abundances of this species do not permit their collection for this study.

Table 3.6-1. Proposed sediment collection stations for Allen Harbor and rationale for selection. Refer to Figure 3.6-1 for station locations.

Rationale for Selection	Station Numbers
A. VEGETATED/WETLAND INTERTIDAL SAMPLES	
1. Allen Harbor Landfill - wetland intertidal data gap	W1 - W10
2. Allen Harbor Landfill - vegetated wetland data gap	V1 - V2
3. Calf Pasture Point - wetland intertidal data gap	W11 - W14
4. Calf Pasture Point - vegetated wetland data gap	V3 - V4
5. Define environmental gradient, support data comparability assessment	AH-5
6. Reference stations (Prudence Coggshall Cove, PCC) Jamestown Sheffield Cove (JSC)	PCC-V1, PCC-W1, JSC-V1, JSC-W1
B. OFFSHORE, SUBTIDAL SAMPLES	
1. Allen Harbor - surface sediment data gap	D1 - D10
2. Calf Pasture Point - surface sediment data gap	D11 - D14
3. Allen Harbor Landfill - subtidal, deep sediment data gap	DC1 - DC10
4. Calf Pasture Point - subtidal, deep sediment data gap	DC11 - DC14
5. Define environmental gradient, support data comparability assessment	D-AH5, DC-AH5
6. Reference stations; Prudence Coggshall Cove (PCC), Jamestown Sheffield Cove (JSC)	PCC-D1, PCC-DC1 JSC-D1, JSC-DC1

Table 3.6-2. Allen Harbor Ecological Risk Assessment Sample Collection/Chemical Analysis Summary.

Sample ID			Sediment						Bivalves													Fish						Geotechnical			Bioassay					
	STA	STA	Metals, Organics				VOAs		MUSSEL				SOFT SHELL			OYSTER			HARD CLAM			Metals			Organics											
SITE	TYPE	NUM	SUR	DUP	BOT	DUP	SUR	BOT	ND	DEP		DUP	ND	DEP	DUP	ND	DEP	DUP	ND	DEP	DUP	CM	ST	MX	CM	ST	MX	LO	GS	TOC	HN	AMP	BOT	CI	ARB	BOT
AH	W	1	1				1		1A			1A												1			1		1	1	1	1		1	1	
AH	W	2	1				1		1A							1A										1			1	1		1		1	1	
AH	W	3	1				1		1A	1C						1A		1A				1	1		1				1	1	1	1		1	1	
AH	W	4	1				1		1A	1A						1A	1A					1	1						1	1	1	1		1	1	
AH	W	5	1				1		1A	1A			1A	1A								1	1		1				1	1	1	1		1	1	
AH	W	6	1				1		1A							1A	1A					1	1		1				1	1	1	1		1	1	
AH	W	7	1				1						1A	1A	1A				1A	1A	1A			1			1		1	1	0	1		1	1	
AH	W	8	1				1		1A							1A,D	1A							1	1				1	1	1	1		1	1	
AH	W	9	1	1			1									1A		1A	1A	1A		1	1						1	1	1	1		1	1	
AH	W	10	1				1									1A	1A		1A,D		1A	1							1	1	0	1		1	1	
AH	W	11	1				1		1A,D	1A,D			1A			1A						1		1					1	1	0	1		1	1	
AH	W	12	1	1			1		1A				1A	1A															1	1	1	1		1	1	
AH	W	13	1				1						1A,D			1A								1			1		1	1	0	1		1	1	
AH	W	14	1				1		1A	1A			1A										1						1	1	0	1		1	1	
PCC	W	1	1				1									1A			1A			1		1		1			1	1		1		1	1	
JSC	W	1	1				1		1A							1A								1			1		1	1	0	1		1	1	
AH	V	1	1				1		1A																				1	1		1		1	1	
AH	V	2	1				1		1A																				1	1		1		1	1	
AH	V	3	1				1		1A	1A																			1	1		1		1	1	
AH	V	4	1A				1		1A																				1	1		1		1	1	
PCC	V	1	1				1		1A							1A													1	1		1		1	1	
JSC	V	1	1				1		1A																				1	1		1		1	1	
AH	D	1	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	2	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	3	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	4	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	5	1	1	1			1											1A									1	2	2		1	1	1	1	1
AH	D	6	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	7	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	8	1	1	1			1											1A									1	2	2		1	1	1	1	1
AH	D	9	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	10	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	11	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	12	1	1	1			1											1A									1	2	2		1	1	1	1	1
AH	D	13	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	14	1		1			1											1A									1	2	2		1	1	1	1	1
AH	D	AH-5	1		1			1											1A									1	2	2		1	1	1	1	1
PCC	D	1	1		1			1											1A									1	2	2		1	1	1	1	1
JSC	D	1	1		1			1											1A									1	2	2		1	1	1	1	1
TOTAL:			39	5	17		22	17	17	6		1	6	3	1	12	4	2	21	2	2	8	6	7	4	2	4	17	39	39	14	39	17	39	39	17
Group Totals:				61				39		24				10			18			25			21			10										

Sample ID
AH = Allen Harbor landfill
PCC= Prudence Coggshall Cove
JSC = Jamestown Sheffield Cove
W = Wetland Intertidal
D = Deep offshore
V = Vegetated Marsh
SUR = top 0-6cm
BOT = Deep core sample

SEDIMENT Chemistry Parameters:
Organics = PCBs, PAHs, Pesticides, Butyltins
Metals = Total, Pore Water, SEM/AVS
VOC = Volatile Organics
A = Does Not Include SEM/AVS

Mussel = *Modiolus demissus*
Hard Clam = *Mercenaria mercenaria*
Soft Shell = *Mya arenaria*
Fish: CM = *Fundulus heteroclitus*, ST =*Fundulus majalis*, MX = *mixture*

Bivalve, Fish Chemistry Parameters:
Organics (PCB/PAH/PEST)
Total Metals
Butyltins
Tissue Lipids
A = Metals And Organics
B = Metals Only
C = Organics Only
D = Split Samples (Metals Only)

GeoTechnical
MS = Sediment Core Magnetic Susceptability
GS = Sediment Grain Size
TOC = Sediment Total Organic Carbon

Bioassay:
AMP = Amphipod Test
ARB = Arbacia Test
HN = hematopoietic neoplasia
CI = Bivalve Condition Index

4.0 EXPOSURE ASSESSMENT

In the Exposure Assessment phase, activities have primarily included the collection of information needed to quantify or estimate the concentrations of CoCs in the relevant environmental media. These results are described in four sections: (1) and examination of sources and exposure pathways of CoCs (Section 4.1), (2) fate and transport analyses for CoCs (Section 4.2), (3) estimates of exposure point concentrations from sediment and biota residue data (Section 4.3), and (4) analysis of uncertainty relating to these results (Section 4.4).

4.1 SOURCES AND EXPOSURE PATHWAYS OF COCS

Several exposure pathways are likely from contaminant sources associated with historical activities at the Allen Harbor Landfill and at Calf Pasture Point. Characterization of landfill constituents by studies conducted for the Phase I and Phase II Risk Assessment Pilot Studies (Munns et al. 1991;1992) and the Phase I and Phase II Remedial Investigation (TRC 1991; TRC 1994) concluded that PAHs, PCBs, numerous metals, and a chlorinated pesticide were present in concentrations that represented potential for ecological risks. This was confirmed by analyses of soil and ground water samples. In general, sources and exposure pathways for constituents from the landfill and from Calf Pasture Point to the marine environment and associated biota are addressed as part of the series of conceptual models presented in Section 3.5. Initial exposure pathways as defined by the second tier model are concluded to occur primarily via surface water flows and erosion from the landfill and Calf Pasture Point. The third tier model describes the behavior of dissolved and particle-bound constituents in the aquatic environment, including transport by and/or association with surface water, sediments, and pore water. Finally, the fourth tier model identifies sources and exposure pathways for biological receptors including (1) surface water exposures of pelagic organisms such as fish and filter feeding infauna and epifauna,

and (2) soil (particle), sediment, and pore water exposures to bottom-dwelling fish, infauna, and epifauna.

Contaminant exposure routes for aquatic biota can include accumulation from water, sediments, and pore water via partitioning across cell membranes, incidental contact or feeding mode ingestion of sediments (e.g., by deposit feeding invertebrates), and consumption of contaminated prey. Thus, it is important to identify the behavior and potential effects of CoCs as a key part of the risk assessment. Based on the general models described above a more detailed evaluation of exposure pathways can be derived for specific classes of CoCs as related to their chemical behavior, solubilities, and toxicity (summarized additionally in Section 5.1).

Some organic constituents, including DDE and PCBs, identified in source samples are characterized by relatively low solubilities in water and high solubilities in lipid phases (e.g., many animal tissues). These low water solubilities tend to result in a net transfer of such compounds from aqueous to particulate phases, with subsequent accumulation in sediments and to a lesser degree, pore water (via partitioning; Clayton et al. 1977). Transfer of this type of CoC to organisms living on or in the sediments can occur through direct uptake (e.g., dermal contact or sediment ingestion) or through partitioning from interstitial pore water. However, because of the tendency for these compounds to remain absorbed to sediments, there should be relatively low dissolved-phase concentrations above the sediments, thereby minimizing exposures to pelagic organisms.

It is notable that respiratory surfaces of water-breathing organisms such as fish and invertebrates provide an effective transfer mechanism for these constituents between lipid-rich tissues and the aqueous environment. Thus, the concentrations of highly lipid-soluble organics in these organisms may be somewhat controlled by these partitioning mechanisms. Consequently, contaminant concentrations in these species may be more dependent on the lipid content as related, for example, to reproductive

condition, than on magnification of the chemical within a food chain (Clayton et al. 1977). In contrast to water-breathing organisms, air-breathing organisms associated with aquatic environments (e.g., water fowl or birds) do not have external surfaces that readily facilitate the transfer of lipid-soluble chemicals between internal lipid and external water phases. Consequently, these species are more susceptible to biomagnification (i.e. uptake from food) of such chemicals. As noted in Clayton et al. (1977), concentrations of constituents such as PCBs in water breathing biota from different trophic levels (e.g., zooplankton, herring, and salmon) can be very similar when the values are lipid-normalized. In contrast, concentrations in air-breathing aquatic biota (e.g., seals) can be considerably higher.

Other source organic constituents, particularly PAHs, also tend to have low water solubilities (solubility decreases with increasing molecular weight) and primarily are found associated with particles and sediments (Pruell, 1988). Thus, the principal risk from PAHs would be to bottom-dwelling fish and invertebrates, including filter feeders that ingest PAH-laden particles. However, in contrast to chlorinated compounds such as PCBs there appears to be a reduced association of PAHs with lipid-rich tissues (Tracey and Hansen, in press). The fact that PAHs can be extremely toxic to invertebrates is perhaps consistent with observations of reduced bioaccumulation. The primary effects from PAHs are as carcinogens, particularly at the point of contact, as influenced by the formation of metabolic intermediates.

Metals, such as, silver, lead, zinc, arsenic, manganese, mercury, and chromium +3 all are relatively insoluble in aqueous media and tend to be associated with particles and sediments. Thus, organism exposure pathways are expected to be similar to those noted for the organic constituents. In contrast, nickel, copper, cadmium, and chromium +6 are relatively soluble and characteristically are associated with dissolved phases. However, for copper, various complex reactions ultimately result in deposition in bottom sediments, and for arsenic, methylation can result in releases back into the water column. Exposure pathways for dissolved phases

primarily would result in water column effects on pelagic organisms and filter feeders. Toxicity responses are highest for copper, mercury, silver, chromium +6, and to a lesser extent zinc, manganese, and arsenic. It is notable that most of the chromium in aquatic environments occurs as chromium +3, therefore substantially reducing its potential toxicity. Copper toxicity is greatest in fish and invertebrates, but is moderated substantially in higher animals due to homeostatic mechanisms that limit adsorption. Mercury is of substantial concern because of high potentials for bioconcentration and magnification of methyl mercury within food chains. Lead also is known to bioaccumulate in some bivalves, but not in fish, and manganese may be associated with bioaccumulation in fish but not in higher animals (also due to homeostatic compensating mechanisms). Finally, carcinogenic responses have been documented for several metals including chromium +6, arsenic, nickel and, potentially cadmium and lead.

4.2 Fate and Transport Analysis

This section evaluates the spatial distribution of constituents in bottom sediments and biological tissues to describe the possible fate and transport of constituents from the landfill and Calf Pasture Point to receptors of concern. Comparative information is available mainly from data collected during the Phase III and Phase I RAPS studies.

4.2.1. Sediments

Sediment samples were collected from a series of vegetated wetlands (V), intertidal wetlands (W), and subtidal offshore (D) locations within Allen Harbor and at two reference sites--the mid-bay reference site in Coggshall Cove at Prudence Island (PCC) and the baseline (i.e., unimpacted) site at Jamestown Sheffield Cove (JSC). Surficial sediment (upper 6 cm) samples collected at these stations represent recently deposited materials. Subsurface sediments (from >50 cm depths in cores) were also

collected at the subtidal offshore stations (DC) to enable evaluations of potential migration and accumulation of ground water-associated constituents in subsurface sediment layers.

Grain Size/TOC: The grain size characteristics of sediments from the vegetated wetlands, intertidal, and subtidal sites are shown in Figure 4.2-1. Sediments within the vegetated wetlands sites near the landfill and Calf Pasture Point are characterized as primarily silts (>85% silt-sized materials) and silty-sand (approximately 50-70% silts), respectively. The intertidal sediments adjacent to the southern portion of the landfill (Stations W1, W2 and ,W4) generally contain relatively higher proportions of fine-grained particles than do intertidal sediments within the northern and eastern portions of the Harbor. Subtidal sediments in most areas of the harbor comprise fine-grained particles having >90% fines, with the exception of Stations D1-D3 and D11-D13 which have relatively higher percentages of sand-sized particles. Sediments within the vegetated wetlands and intertidal regions of the PCC and JSC reference sites consist primarily (>85%) of sand-sized particles with low percentages of silt/clay-sized particles.

Sediment TOC concentrations (Figure 4.2-2) vary considerably throughout Allen Harbor and the two reference sites. This variability is consistent with the spatial variability in grain size characteristics. In general, TOC concentrations are higher in the vegetated wetlands sites than those in sediments from adjacent intertidal and subtidal areas, with the exception that TOC levels in sediments from the wetlands and adjacent intertidal areas at the reference sites are comparable. For the intertidal areas of Allen Harbor, sediment TOC concentrations are relatively higher along the southern and western portions of the landfill (Stations W1-W6) than at the northern and eastern shoreline areas. Sediment TOC concentrations are relatively uniform throughout subtidal areas in the northern portion of Allen Harbor (D5-D10, D14, and AH5) and consistently higher than those in subtidal portions of the western and eastern areas of the harbor. Grain size and TOC concentrations in subsurface layers at the subtidal

stations do not exhibit any obvious spatial patterns either horizontally or vertically among stations.

PAHs: Concentrations of total PAHs (tPAH) represents summed concentrations for 24 compounds including perylene in sediments are presented in Figure 4.2-3. Concentrations for individual PAH compounds are listed in Table A1-1.2. The relative abundances of individual PAH compounds are dominated by the higher molecular weight (HMW), four- and five-ring compounds (e.g., benzofluoranthenes and pyrene), which are possibly indicative of combustion, street runoff, and/or waste crankcase oil sources from the landfill or nearby urban areas. The highest concentrations of total PAHs were observed in intertidal wetland sites, and the lowest concentrations in deep core subtidal sediments. Vegetated wetland and subtidal sites had comparable sediment PAH concentrations. Allen Harbor sediments contained PAHs in concentrations that were more than 10x greater than at associated reference sites. However, these differences were diminished somewhat when concentrations were TOC normalized.

Sediments from each of the vegetated landfill sites (V1-V4) contain elevated concentrations of PAHs relative to those at the PCC and JSC reference sites. Concentrations at the Allen Harbor sites were up to two orders of magnitude higher than those at the PCC reference site. However, the tPAH concentrations (normalized to TOC) for Allen Harbor sites V1, V3, and V4 were comparable to TOC-normalized concentrations at the JSC reference site.

Concentrations of PAHs are highest in intertidal sediments adjacent to the southern portion of the Landfill. In particular, tPAH concentrations at site W1 are several fold higher than those at adjacent intertidal sites, and more than one order of magnitude higher than sediment tPAH concentrations at the reference intertidal stations. Total PAH concentrations in sediments from the other intertidal sites along the southern Landfill also are elevated relative to levels in the northern and eastern

portions of the Harbor. Normalized and non-normalized concentrations of tPAH (Figure 4.2-3) exhibit clear distribution patterns relative to distance from the southern landfill. Higher molecular weight PAH compounds, suggestive of combustion, runoff, and/or heavier lubricating or bunker fuel sources (e.g., Pruell and Quinn, 1988; Quinn et al. 1994), are the primary contributors to tPAH concentrations in these sediments.

Concentrations of tPAH in subtidal sediments generally follow trends in grain size and TOC, with relatively higher and uniform levels in western and northern portions of the harbor (Stations D5-D10, D14, and AH5), whereas tPAH concentrations are lower at sites within the southern and eastern portions of the Harbor (Stations D1-D3 and D11-D13). Total PAH concentrations in subsurface sediments from the offshore stations typically are lower than the respective surficial sediments. However, the spatial patterns in tPAH concentrations in subsurface sediments do not correspond well with those in the surficial sediments or with proximity to the landfill or Calf Pasture Point. The absence of better correspondence between tPAH concentrations in surface and subsurface sediments may be due to differences within the harbor in the extent of physical and/or biological mixing of the sediments and subsequent potential for remobilization of constituents to the overlying bottom waters. The PAHs did not exhibit any noticeable trend of increased contaminant concentration with sediment depth.

Chlorinated pesticides. Of the chlorinated pesticides measured during the Phase III study, p,p-DDE was the only compound present at levels which consistently exceeded the analytical detection limits. Concentrations of p,p'-DDE, a primary breakdown product of the chlorinated pesticide DDT, in sediments are presented in Figure 4.2-4. Concentrations of the other chlorinated organics are listed in Table A1-1.1 and A1-1.3. Chlorinated pesticides/PCBs did not exhibit any noticeable trend of increased contaminant concentration with depth.

Sediment DDE concentrations at the vegetated wetlands sites near the landfill were several fold higher than those at Calf Pasture Point and at the two reference

sites (Figure 4.2-4). Similarly, DDE concentrations at the intertidal sites were highest near the landfill, but relatively low around Calf Pasture Point and at the reference sites. Spatial patterns in DDE concentrations at the subtidal offshore sites were similar to those observed for tPAH, and generally were higher within the northern portion of the harbor than in the western and eastern areas. Deep core subtidal sediments contained the lowest pesticide concentrations.

PCBs. Total PCB concentrations (i.e., 2x the summed concentrations of 18 PCB congeners) are plotted for each station in Figure 4.2-5. Similar to DDE concentrations, tPCB levels at the vegetated wetlands sites near the landfill were several fold higher than tPCB concentrations at vegetated wetlands stations near Calf Pasture Point which, in turn, contained concentrations that were relatively higher than those measured at the reference sites. The highest tPCB concentrations for intertidal sites also occurred adjacent to the landfill (Stations W1 and W6). The other intertidal sites near the landfill (W2-W9) also contained elevated tPCB concentrations relative to those adjacent to Calf Pasture Point and at the reference sites. Spatial distributions of tPCB concentrations in sediments from the subtidal offshore sites were similar to distributions for tPAHs and DDE in subtidal sediments and depositional patterns within the harbor.

Organotins: Spatial trends in concentrations of butyltins (BT) in Allen Harbor sediments are presented in Figure 4.2-6. In general, the within station dibutyltin (DIBT) and tributyltin (TRIBT) compounds. concentrations were similar while the monobutyltins (MONOBT) were of reduced or nondetectable concentrations. Sediments at the vegetated wetland Stations V2-V4 contained approximately two-fold higher concentrations of BTs than those at the two reference sites while concentrations at Station V1 were less than the reference sites. Relative to the vegetated wetland stations, the intertidal stations adjacent to the landfill contained slightly higher TBT concentrations, although the highest concentrations observed at these sites are comparable to the TBT concentration at the PCC reference site. TBT

concentrations are generally uniform throughout the subtidal areas along the eastern edge of the landfill (stations D4-D10) and are also slightly higher than concentrations in subtidal sediments south of the landfill (D1-D3) and adjacent to Calf Pasture Point (D11-D14). Concentrations of TBT at the RAPS subtidal stations (AH-2 to AH-8) are at least tenfold higher than concentrations at Phase III subtidal stations. Concentrations of TBT in the subsurface sediments from most of the subtidal offshore stations, with the exception of site DC13 and, to a lesser extent DC3, DC14 and reference sites, are at or near the analytical detection limits.

Total Sediment Metals. Spatial patterns Ag, As, Ni and Hg in Allen Harbor sediments generally followed trends observed for concentrations of sediment organics (Figure 4.2-7). Relatively higher levels of several metals occurred within the vegetated wetlands Stations V1 and V2, intertidal Stations W1-W2 and W4-W8, and subtidal stations D4-D10, D14, and AH5. These stations are located along the southern landfill, whereas the subtidal sites are primarily within the eastern and northern portions of the Harbor. Elevated metal concentrations were not as evident at stations adjacent to Calf Pasture Point.

As expected, patterns in elevated sediment metal concentrations also coincided with distributions of fine-grained sediments. However, some differences were apparent between the spatial patterns for sediment metals and those for organic constituents. For example, the highest metal concentrations occurred at intertidal Station W5, adjacent to the eastern portion of the landfill, whereas the highest tPAH and tPCB concentrations occur at Stations W1 and W4, respectively. Concentrations of zinc, lead, nickel, and copper tended to covary over all sites, whereas patterns in sediment chromium concentrations were not as consistent. The reasons for these differences are not clear, although they may reflect a nonhomogeneous composition of waste materials which are migrating from the landfill.

Further comparison of total metal distributions across Allen Harbor habitats revealed comparable mean concentrations in subtidal habitats and vegetated wetlands, which were two times greater than mean concentrations in intertidal wetland habitats (Figure 4.2-7). The highest maximum total metal concentrations were also observed in intertidal wetlands, and result from high zinc concentrations at the AH14 station that was sampled during the RAPS study. Deep core subtidal total metal concentrations were slightly lower than in other habitats, but metal concentrations in all Allen Harbor habitats exceeded reference site concentrations by approximately one order of magnitude.

Subtidal surficial cores contained the highest mean concentrations of silver, and copper (Figure 4.2-7). Vegetated wetlands had the highest concentrations of arsenic, and intertidal wetlands contained the highest mean concentrations of copper, lead, and zinc. Mean cadmium, mercury, and nickel sediment concentrations were comparable across all habitats. Cadmium and mercury concentrations were below detection limits in the majority of samples in all habitats. It is notable that chromium and arsenic concentrations in deep core subtidal habitat sediment did not differ greatly from concentrations in surficial subtidal sediments.

Volatile Organic Compounds: The spatial distribution of selected VOAs in sediments by habitat are summarized in Figure 4.2-8. Six compounds are included which were above method detection limits. VOAs were measured in wetland/intertidal and in the subsurface offshore sediments (deep core). Methylene chloride concentrations were elevated relative to reference at stations DC1, DC2 and DC5 as well as in peat material of station DC6. Concentrations were also somewhat higher at stations DC10-DC12. This compound was also elevated in sediments from the southern landfill vegetated and wetland sites. Acetone followed the distribution of methylene choride and was notable for relatively high concentrations in the wetlands stations W1, W2 and W11.

An unusually high concentration of chlorobenzene was present in sediments from intertidal station W6. This compound was also measured in groundwater samples from the landfill during the Phase I and Phase III studies. Volatile organic compound concentrations in deep core (DC) samples were measured as an indicator of migration of groundwater through the offshore sediments and to provide information for evaluating this potential transport pathway. Aside from compounds that likely are related to laboratory contaminant sources, only low levels of 1,1,1-trichloroethane and benzene were present in the subsurface sediments at the subtidal Allen Harbor sites. At the two reference sites, benzene was present at comparable levels to sites in Allen Harbor, whereas 1,1,1-trichloroethane was not detected at reference sites. Thus, there is no detectable accumulation of VOAs in subsurface sediments at the subtidal sites in Allen Harbor. Based on these data, there is no evidence of groundwater migration and transport of VOA through the offshore sediments. Similarly, comparisons between VOA concentrations in DC samples and those in "peat samples" from the sediment cores do not indicate any substantial accumulation of VOAs in pockets of organic materials.

Porewater metals. Sediment pore water chemistry results are summarized in Table A1-3.1. For concentrations less than the Method Limit of Quantitation (MLQ), results are presented as 1/2 the MLQ value so as to permit further data manipulations. The analytes Ag, Cd, Hg and Zn were near or below the Method Limit of Quantitation for these elements. (One exceptional value for Cd, 8.4 ug/L, was found at the Prudence Coggeshall Cove (PCC) Reference site). The metal As exhibited a range of 2.5 - 65 ppm (ug/L) with highest concentrations found in deep core sediments. One notable exception was As at station D1 at 44.6 ug/L. Chromium was typically low (< 2 ug/L) with exception of vegetated station V1 (71.6 ug/L) and wetland/intertidal station W7 (10.4 ug/L). Copper (Cu) was above the MLQ (2.5 ug/L) at deep stations D2, D3, and D6, and at wetland/intertidal stations W3, W5, W6 and W8, and at vegetated stations V3 and V4. Nickel was found to be occasionally higher than the MLQ (8 ug/L)

at D9, DC8 and DC13. For Pb, station D1 (11.2 ug/L), DC12 (12.2 ug/L) and stations W4 and W5 (7.8 and 9.8 ug/L, respectively) were notable.

SEM Metals: Toxicity of sediment metals is correlated with divalent metal activity in interstitial water. SEM/AVS (Simultaneously Extractable Metals/Acid Volatile Sulfide) analyses were performed to address the bioavailability of divalent metals in sediments. In general terms, metal availability is controlled by the concentration of insoluble metal sulfides that act to bind divalent metals such as cadmium, copper, mercury, nickel, lead, and zinc (DiToro et al. 1990). Weak acid treatment of the sediment volatilizes these reactive sulfides with bound metal as a gas (hence the metals are simultaneously extracted with the sulfide). Conceptually, divalent metals bound to AVS are not be toxic to sediment-associated biota. Therefore, a comparison of the amount of AVS and SEM in the sediment allows a determination of metal bioavailability and potential toxicity. For example, if the SEM/AVS ratio is greater than one (or SEM-AVS > 0) then there is more metal available than sulfide and the metal is assumed to be bioavailable.

For the present evaluation, samples for SEM/AVS analysis were collected at 14 intertidal (W1-W14; surface sediments only), 14 offshore (D1-D14; surface and subsurface), 2 reference (PCC and JSC; surface and subsurface), and 4 marsh stations (V1-V4; surface only). Duplicate samples were collected at Stations W9 and W12 and showed a high degree of precision (Table 4.2-1). Results indicated that SEM/AVS ratios greater than one (i.e., indicating metal bioavailability) were associated with surface samples from 6 (W5-W7, W13, V3, and V4) of 32 stations, offshore subsurface samples from 2 (D1 and D12) of 14 offshore stations, and the surface and subsurface sample from the JSC reference and the surface sample from the PCC reference (Table 4.2-1). Thus, the stations associated with the highest potential for metals bioavailability corresponded to the areas of the northern and southern landfill and to Calf Pasture Point. However, of these, the high SEM/AVS values (approximately 8-44) for 5 stations (V3, V4, PCC-D1, D12, and JSC-D1) are influenced

by a numerical artifact (division by low values below the detection limit of 0.1 μmole/gram). Therefore, these data actually represent conditions of little or no AVS. Hence it is preferable to consider the SEM-AVS result as a better indicator of metal bioavailability. Finally, it is notable that data from both reference stations also indicate bioavailability of metals.

Inspection of individual metal values from stations having SEM/AVS ratios greater than one indicated high concentrations in particular for Stations W5 (copper, zinc, lead, and cadmium), V3 and V4 (lead), and W7 (lead) (Table 4.2-1). Station W6 also had elevated lead concentrations, as well as SEM/AVS ratio near 1 (0.95).

4.2.2. Tissues

Tissues of four bivalve species, the hard clam (HC), ribbed mussel (RM), soft shell clam (SC), and oyster (CV), and one fish species mix (MF) were collected and the tissues analyzed for concentrations of PAHs, selected chlorinated pesticides and PCBs, organotins, and metals. Analyses of chemical constituents in tissues are used to further evaluate spatial patterns in the environmental concentrations and bioavailability of CoCs within Allen Harbor. A selection of bivalve samples were depurated (DEP) to remove sediment from the gut. Concentrations of organic compounds are normalized to percentages of tissue lipids to account for differences between species and sites in the lipid content, which is known to affect the partitioning and accumulation of organic constituents.

Lipids. Concentrations of lipids in tissues of ribbed mussels, hard clams, soft shell clams, oysters and fish at various stations are shown in Figure 4.2-9. Concentrations were lowest in hard shell clams (2-3%), comparable in soft shell clams, oysters ribbed mussels and fish (4-7%). Significant station-to-station variation in tissue lipid content was observed, although no discernable spatial trends were apparent.

PAHs: Concentrations of tPAHs in fish and shellfish tissues from the vegetated wetlands, intertidal, and subtidal sites are shown in Figure 4.2-10. Large differences in concentrations of tissue tPAHs among species are apparent. Overall, concentrations of tPAH in the ribbed mussel are several times higher than those in other species. Higher concentrations of lipid-normalized and non-normalized tPAH in ribbed mussel tissues were attributable primarily to the relatively higher concentrations of the lower molecular weight PAHs, such as naphthalene and the alkyl homologs (Table A1-2.2D). Residue patterns for two fish species, being either the common mummichog (MF-COM) or stripped killifish (MF-STP), were remarkably similar both spatially and magnitude for lipid normalized and non-normalized values. Fish residues were generally lower than shellfish residues by two- to five-fold, mostly likely because of either reduced exposure or enhanced PAH metabolism relative to bivalves.

Tissue tPAH concentrations in shellfish from the two reference sites were approximately half that of shellfish obtained from within Allen Harbor (Figure 4.2-10). Differences in PAHs among fish samples were less pronounced, but were slightly higher on average for harbor stations vs. reference locations. Other than differences between Allen Harbor and the reference sites, no consistent spatial patterns in tissue tPAH concentrations are apparent. Specifically, tPAH levels in shellfish from sites closest to the landfill and Calf Pasture Point were not noticeably elevated compared to those in shellfish from the remaining Allen Harbor sites.

Chlorinated pesticides: Concentrations of DDE in tissues of the four shellfish species and fish are shown in Figure 4.2-11. Normalized and non-normalized DDE concentrations in tissues of the hard clam are uniformly low throughout Allen Harbor and the reference sites, with the exception of the slightly elevated levels in the sample from Station D2 located offshore from the southern landfill. In contrast, DDE concentrations in ribbed mussel exhibit an apparent site-specific response with the highest concentrations occurring at locations near the southern landfill and lower concentrations elsewhere. Oyster and soft shell clam tissue DDE concentrations were

slightly higher at stations adjacent to the landfill, although the magnitude of differences were relatively small compared to that observed for the ribbed mussel. Lipid-normalized DDE concentrations in both species were relatively uniform across stations. Pesticide residues for the fish species were three- to eight-fold higher than in bivalves yet were fairly constant across stations, and levels were approximately twice that of residues found in fish from reference locations.

PCBs. Total PCB concentrations in shellfish tissues are presented in Figure 4.2-12. The spatial distribution of tPCB in ribbed mussels was almost identical to patterns observed for DDE in this species, with highest concentrations occuring in samples from the southern landfill intertidal areas (Stations W1-W5), followed by northern landfill and western Calf Pasture Point (Stations W6-8 and W11-12, respectively). Hard clam tissue tPCB concentrations were uniform across subtidal sites (with the exception of elevated levels at Station D2) while intertidal Station W7 contained slightly higher tPCB concentrations than other intertidal sites. Relatively high tPCB levels at DAH5 reflect the station average of RAPS data and present study data (205 vs 64 ng/g, respectively; Table A1-1.1A)), perhaps reflecting a downward temporal trend for CoCs in Allen Harbor.

Total PCB concentrations in oysters from the landfill intertidal sites (W2-W10) were approximately twice that found for organisms collected at the reference intertidal sites. Residue levels in hard clams and soft shell clams were about two-fold lower than ribbed mussel and oyster tissue concentrations. Soft shell clams at Station W7 contained relatively higher tPCB concentrations than at the adjacent intertidal sites following the pattern found for hard clams. PCB residues for the fish species exhibited much greater spatial variability than was found for PAHs; levels in the common mummichog (MF-COM) varied almost three-fold across closely situated stations (i.e. W3 vs W5; W6 vs. W8) possibly suggesting an exposure to some localized contaminant source. Lipid-normalized concentrations for stripped killifish (MF-STP) and common mummichogs at adjacent stations W2 and W3, respectively, were

remarkably similar suggesting that species differences had minimal effect on the interpretation of PCB bioavailability for this trophic group.

Organotins. Concentrations of tributyltin (TRIBT) in Allen Harbor biota are summarized in Figure 4.2-13. No consistent trends in TRIBT tissue residues were observed which related spatially to the landfill or Calf Pasture Point as potential sources, nor were harbor biota, with the possible exception of fish, enriched with Sn relative to reference station organisms. The mummichog fish (MF) tissue residues at harbor stations did appear somewhat elevated relative to reference sites.

Total metals: Results of tissue metal residues in bivalves and fish are summarized in Figures 4.2-14-4.2-18. Spatial patterns among the shellfish and fish were not apparent. However, tissue metal concentrations for individual species did not appear to follow spatial trends in sediment metal concentrations; coefficients for regression of tissue vs. sediment metal concentrations were less than 0.1 for most species and metal combinations. Among the five species, only soft shell clams had metal concentrations (including zinc, chromium, lead, and copper) which appeared to covary among different sites. Animals from Station W5 contained elevated concentrations of zinc, chromium, lead, nickel, and copper compared to values in clams from other intertidal sites. Soft shell clams from Stations W12 and W13 also contained somewhat elevated levels of nickel and chromium, respectively.

Relative to concentrations in reference site organisms, oysters from Stations W4 and W11 contained elevated zinc, chromium, lead, and nickel levels, although copper concentrations were not appreciably higher. Elevated chromium and nickel concentrations also occurred in oysters from Stations W2 and W9. Additionally, highly elevated (> 2500 mg/kg) zinc concentrations occurred in tissues of oysters from all of the intertidal sites except W13. Ribbed mussels from Stations W1, W3, W4, W8, and W11 contained concentrations of nickel and chromium that were elevated relative to concentrations in reference site organisms. Elevated chromium and nickel

concentrations also occurred in ribbed mussels from the vegetated wetland Stations V3 and V4.

Zinc concentrations in tissues of hard clams from all of the subtidal sites except D1 were higher than those at the reference site. Relatively high copper concentrations also occurred in hard clams at Station D6 and at the intertidal Stations W7, W9, and W10. Elevated nickel concentrations occurred in fish tissues from Station W4, and slightly higher concentrations relative to reference site levels of lead occurred in fish from Stations W1 and W6. Zinc, chromium, and copper concentrations in fish from Allen Harbor were comparable to those in fish from the reference site.

4.2.3. Water Sampling Results

Water sampling for PCBs, PAHs, pesticides and metals was conducted in Allen Harbor on 15 June, 1995. In Allen Harbor, sampling was conducted approximately 1-2 hr after slack low tide, such that the greatest potential contribution of non-tidal source waters entering Allen Harbor could be measured. Stations included D5 and D6 along the Allen Harbor landfill face, and at D12 adjacent to Calf Pasture Point.

Concentrations of total PCBs as Aroclors ranged from 8-15 ng/L at reference sites to 10-32 ng/L at sites in Allen Harbor (Table A1-4.1). Total PAH concentrations (sum dissolved + particulate after blank correction of 44 ng/L) ranged from 5-10 ng/L at reference sites to 50 -110 ng/L at sites in Allen Harbor. Pesticide concentrations (as a sum of 5 pesticides) were non-detectable at Allen Harbor sites.

Concentrations of Ag, As and Ni were non-detectable in both dissolved and particulate fractions of water samples collected in conjunction with organics samples discussed above. Detectable concentrations of Cd, Cr, Cu, Pb and Zn were found in water samples as shown in Figure 4.2-19. With the possible exception of Zn,

dissolved, particulate and total metals were bracketed by concentrations found at the reference sites, ranging from 2-14 µg/L.

Water concentrations in the present study were higher than concentrations reported in the previous RAPS Phase I investigation (Table 3.1-1); measurements at RAPS station AH2 was 38 ng/L for tPAHs, and non-detectable for PCBs and pesticides (metals data were not available). This difference could be due to locational and or temporal sampling differences, as the present sampling strategy was specifically designed to capture maximum landfill-related concentrations but further north than station AH2.

4.2.4. Overall Horizontal Spatial Patterns

Sediment and tissue contaminant data from the Phase III study exhibited fairly consistent distribution patterns which suggested transport of contaminants from the landfill to Allen Harbor, with subsequent redistribution and deposition of some portion of the constituents within the offshore areas of the northern embayment between the landfill and Calf Pasture Point. In particular, concentrations of several classes of organic constituents in sediments and tissues were elevated within the intertidal areas adjacent to the southern portion of the landfill, thereby suggesting active transport via water runoff or soil erosion from the landfill. In general, the magnitude of the contamination decreases within the intertidal areas to the north of the southern extension of the landfill, with the exception of occasional spikes in some constituents along the eastern shoreline (i.e., sites W5 and W6). The results also suggest localized inputs of constituents from adjacent portions of the landfill. By comparison, contaminant concentrations at the subtidal sites directly offshore from the southern portion of the landfill typically are comparable to, or lower than, the corresponding concentrations at subtidal sites within the northern portion of the harbor. The comparatively higher contributions of sand-sized particles and corresponding lower organic carbon levels at sites offshore the southern landfill area, suggest that

accumulation of fine-grained sediments, containing proportionately higher contaminant loading, is lower than that which occurs within the northern embayment of the harbor.

Compared with the landfill area, contaminant migration from Calf Pasture Point appears to be minimal. Although sediments within the vegetated wetlands sites at Calf Pasture Point contain elevated concentrations of some of the CoCs, the sediments and shellfish tissues at intertidal and subtidal sites adjacent to this area do not contain concentrations above those observed at reference sites.

These results suggest that the landfill is an active source of CoCs to Allen Harbor. Some redistribution of particle-associated constituents likely occurs due to sediment resuspension via physical and biological mixing and transport by tidal currents within the harbor. The relative proportion of the total contaminant mass emission represented by migration of CoCs from the landfill can not be determined without a more detailed accounting of mass emissions from other regional sources (e.g., surface runoff, atmospheric deposition, and exchange with Narragansett Bay). Similarly, the relative proportions of the CoCs deposited within the harbor versus CoCs removed via biological uptake and/or transported into Narragansett Bay is presently unknown.

4.2.5. Vertical Spatial Patterns

As discussed in Section 3.0, a significant data gap addressed in this ERA is the vertical distribution of sediment contamination in relation to potential ground water infiltration. This transport pathway has important implications for ecological risk from resuspension of this material. An important part of this investigation was the need to interpret results in such a way that vertical heterogeneity could be distinguished from horizontal heterogeneity, thereby detecting, for example, "hot spots" of contamination which may be related to Allen Harbor Landfill and/or Calf Pasture Pt. sources. Hence, an objective of the sampling program was to identify a sedimentary layer across the

harbor from which sediments of comparable depositional history could be sampled for chemical and toxicological analyses. Once this layer was identified, greater confidence in discerning spatial trends in contaminant distribution was possible.

A technique called low field magnetic susceptibility (MS) was employed for this purpose. This method which involves the measurement of the concentration of magnetic minerals in field collected cores following the methodology of King et al., (1983). Logs of magnetic susceptibility down-core and between-cores were used to assess the spatial (vertical and horizontal) pattern of sediment deposition. Comparisons of the individual core logs allows a rapid evaluation of the patchiness and, where multiple sediment cores from the same study sites are available, it is also time to use magnetic susceptibility logs and lithologic descriptions to correlate between core sites (Thompson and Oldfield, 1986). In addition, low-field susceptibility is a useful proxy indicator for the rate of soil erosion because of the high concentration of magnetic minerals found in the "A" horizon of temperate upland soils (Thompson and Oldfield, 1986). For example, soil erosion is accelerated by forest clearance and subsequent grazing and cultivation activity that accompanied European settlement of North America. Furthermore, both sewage and the byproducts of industrial activity and fossil fuel combustion tend to have enhanced magnetic susceptibility values.

For the present study of the vertical and horizontal distribution of sediment constituents in Allen Harbor, four replicate cores were collected at all Deep (D) stations, with samples generated at depth described as Deep Core (DC). After completion of the magnetic susceptibility logging, deep sediments from cores were subsampled and analyzed as follows: (1) organic constituents; (2) trace metals; (3) VOAs; (4) grain size; (5) organic carbon; and (6) toxicity studies.

MS logs revealed a marked increase in susceptibility values in the upper levels of many of the Allen Harbor sediment cores. A representative lithostratigraphic column and susceptibility stratigraphy for a site adjacent to the Allen Harbor landfill are shown

in Figure 4.2-20. The uppermost layer is characterized by higher susceptibility values that are at maximum values lower in the layer and decrease toward the sediment-water interface. The decreasing upwards trend is due in part to the higher water content of the surface sediments as well as diminished soil erosion rates as the intensity of land-use has declined during the 20th century. Hence, this upper layer is interpreted as being deposited after European settlement in the area (~1670 AD) and sediments in this upper unit usually contained higher contaminant concentrations.

Below the upper layer is a layer of very dense silty-clay that appears to be continuous in the core sites surrounding the landfill. This clay layer is noteworthy because it will act to either strongly inhibit or prevent the vertical migration of constituents in pore waters. The unit below the clay layer is a clayey-silt with abundant plant debris. Cores from several sites terminated in a dense basal *Spartina* peat. This peat was sampled for evidence of the migration of VOA into this porous lithology.

A general representation of the susceptibility record obtained from cores at Stations D1-D10, located adjacent to the Allen Harbor Landfill, is shown in Figure 4.2-21. The susceptibility logs and sediment intervals composited from each core for chemical analysis are contained in Appendix D-4. It was possible to identify a common stratigraphic layer for Stations D1-D10 adjacent to the landfill in Allen Harbor. However, the stratigraphy of sites from the east side and from central Allen Harbor were not readily correlatable to core sites adjacent to the landfill. The lithologic description from one core from each site is shown on the figures. The core data also reveal that linear sedimentation rates were found to vary by 4X within the study area. The range in sedimentation rates is estimated to be ~0.06 - 0.25 cm/year. This estimate is obtained by assuming that the upper unit and the clay layer were deposited during the last 315 years.

4.2.6. Contaminant Source Characterization

Quantitation of additional ("extended") PAH and pesticide analytes in tissues and sediments from selected stations were conducted to provide additional data for chemical fate and transport and intra-study data comparability assessments. The list of PAH analytes includes alkylated homologues which are useful in differentiating oil-related from combustion-related sources, as well as several pesticides which provide more extensive screening for this contaminant group (Table 4.2-2). A total of 35 pesticide and 24 PAH samples were quantified for sediment concentrations of the extended analytes (Appendix Tables A1-~~1.11~~ 1.10 and A1-~~1.12~~ 1.11, respectively), while a total of 24 PAH and 23 pesticide samples were quantified for tissue concentrations of the extended analytes (Tables A1-2.7 and A1-2.8, respectively). PCB congener data from sediment and tissue samples were also included in this analysis (Appendix Tables A1-1.1 and A1-2.1, respectively).

section 4.3.2 not in this section

Principal Components Analyses (PCA) were conducted on the data in attempt to better characterize differences in the source and pathways of CoC transport to nearshore environments of Allen Harbor which would assist in explaining differences in patterns of bioaccumulation and toxicity to be discussed in Section 5. The approach taken in this analysis is to use the pattern of chemical mixtures at individual stations to identify station differences as to both the strength (concentration) and uniqueness of chemical constituents.

A principal component is a numerical value which can be viewed as an aggregate variable expressing the degree of similarity for a particular feature of the data set. By design, every principal component is independent of one another, and hence expresses unique properties of the data set. Because differences in chemical concentration are one of the most unique features of the data set, the first principal component is primarily an indicator of CoC concentration. Subsequent principal components express additional aspects reflecting nature of chemical mixtures. For

stations which exhibit unique chemical patterns, the underlying compound-specific chemical data (in PCA units, called "loadings") are examined to determine those chemicals which drive unique chemical distributions at the station and thus the CoCs which should be considered among the potential CoC risk drivers for the risk analysis.

Principal components analyses were performed using a conventional software package (Systat, 1995). In the present analysis, class-specific data sets (PAHs, PCBs and pesticides) were prepared as input data files to the software package as raw concentration data (z-score normalization was evaluated but rejected as the technique did not improve component score separation). The number of principal components generated by the package was limited to three, which was found to cumulatively express > 90% of the variation in the data.

Results of a principal component analyses (PCA) of sediment PAH, PCB, and pesticide data are shown in bivariate plots of the primary components for individual compounds and stations (Figures 4.2-22 to 4.2-24). Component loadings for most stations are highest on Axis 1, with the exception of the reference stations JSCDC1 and PCCD1, which have the highest loadings on Axis 2, and the station AHDC1 that has the highest loading on Axis 4. Axis 1 for the bivariate plot of station scores explains 80.3% of the total variance, and primarily separates the intertidal site AHW1 and, to a lesser extent, AHW2 through AHW5 from all other vegetated wetlands, intertidal, and subtidal sites. Based on factor scores for these two axes, the remaining sites, including the PCC and JSC reference sites, are tightly clustered and do not exhibit any other obvious spatial patterns relative to the landfill. Relatively greater separation of stations is apparent along Axis 3, although this component explains only 3.7% of the total variance. Some of the subtidal stations in the central and northern portions of Allen Harbor appear somewhat distinct along Axis 3, although clear spatial patterns relative to the landfill are not indicated. Surficial and subsurface sediments from the same station exhibit differences along Axis 3 that are comparable to

differences between station pairs corresponding to sites adjacent to and removed from the landfill.

Factors 1 and 2 combined for individual PAHs explain 89% of the total variance. All but six of the PAH compounds have maximum loadings on Axis 1, which represents 78.4% of the total variance. Compounds with the highest positive factor scores correspond to higher molecular weight PAHs (e.g., flouranthene, chrysene, benzoflouranthenes, and benzopyrenes) which likely originated from combustion and/or urban runoff sources (Figure 4.2-22). In contrast, the majority of lower molecular weight PAHs (parent and alkylated naphthalenes, flourenes, and dibenzothiophenes) have negative scores on Axis 1. These latter suites of PAHs are more characteristic of petroleum sources. The exceptions of the above trends are the parent and alkylated phenanthrenes, which have positive scores, and alkylated chrysenes with low negative scores. These compounds can be present in both petroleum and combustion sources. Alkylated chrysenes also may be related to biogenic sources or interferences in analyses of biogenic materials. Plots of Axis 1 versus Axis 3 do not provide any further distinctions between suites of PAH compounds; whereas, crossplots of Axes 2 and 3 indicate a distinct grouping of alkylated naphthalenes, dibenzothiophenes, and flourenes in the lower right hand quadrant.

Combining the results from PCA on stations and compounds suggests that sediments at intertidal sites adjacent to the Landfill (AHW1 through AHW6) contain relatively elevated abundances of higher molecular weight PAHs which are characteristic of combustion products. These materials could have originated from combusted petroleum and/or wood products, waste oils (e.g.., used crank case oils), or other weathered petroleums which had leached or eroded from the Landfill and been transported and deposited in the adjacent intertidal sediments. Sites AHW1 and AHW6 are obviously different from most other sites, and they also are different from each other. Site AHW1 represents the extreme end in a gradient corresponding to

relatively greater abundances of higher versus lower molecular weight PAHs; whereas, the relatively high scores for site AHW6 on Axis 2 represent relatively elevated abundances of the parent, C3-, and C4-substituted naphthalenes, substituted (C2- and C3-substituted) flourenes, and substituted (C2- through C4-) phenanthrenes. Separation of this site along Axis 2 suggest that other or additional sources contribute petrogenic PAHs to these sediments. The absence of appreciable overlap of the surface and subsurface samples from the subtidal sites suggest that sources of PAHs to the sediment horizons are somewhat distinct. Therefore, the PAH composition of sediments deposited at these sites has changed over time, and it is unlikely that migration of PAHs from the Landfill has had a substantial effect on the subsurface sediments.

The results of PCA of sediment PCB data are essentially identical to those for the PAH data. Axis 1, which explains 99% of the total variance of the sediment PCB data, separates the intertidal stations W1 through W5 from all other stations except W6, which primarily is distinguished along Axis 2 (Figure 4.2-23). Station W1 represents the maxima in the gradient relative to total PCBs and primary PCB components (i.e., congeners 138, 153, and 180). Station W6 has a distinct PCB signal that suggests a separate source. Axis 3 distinguishes some of the subtidal sites, but there is no strong relationship with distance from the Landfill (Figure 4.2-23B).

Results of PCA of the sediment organochlorine data are consistent with those for the PAH and PCB data, except that vegetated wetland stations V4, V3 and to a lesser extent, V1 are distinguished from all other sites but intertidal Stations W1 through W6 (Figure 4.2-24). Separation of these intertidal sites from the other Allen Harbor sites is related to abundances of p,p'-DDE and p,p'-DDD, which are primary degradation products of DDT. The vegetated wetland Stations V4 and V3 are distinguished from other sites by enriched DDT. These differences potentially reflect contributions from a relatively fresh (i.e., unweathered) source of DDT at Calf Pasture

Point. All other stations, including the subtidal sites and vegetated wetlands Station V2, cluster with the reference sites.

In total, intertidal stations W1 through W6 and, to a lesser extent, the vegetated wetlands Stations V1, V3 and V4 appear geochemically distinct from all other Allen Harbor and reference stations. Further, sediment conditions at Stations W1 and W6 are appreciably different from each other, which suggests multiple, distinct sources of organic contaminants from the Landfill to the intertidal regions of Allen Harbor. The vegetated wetland sites are distinct only with respect to chlorinated pesticides, but not PAHs or PCBs. Finally, the subtidal sites do not appear appreciably different from the reference site, and no significant horizontal or vertical contaminant gradients or trends are apparent within these regions.

4.3 Estimate of Exposure Point Concentrations

The previous section described the general spatial distributions of CoCs within the vegetated wetlands, intertidal, and offshore subtidal sediments and within shellfish tissues in Allen Harbor and at the two references sites. This section summarizes the magnitude of the CoC concentrations in sediments and tissues. These concentrations are important for evaluating potentials for contaminant exposures and risk, as well as environmental conditions relative to contaminant-specific criteria or benchmarks.

4.3.1. Sediments

Spatial patterns of tPAHs, DDE, PCBs, TBT, total metals and VOAs in sediments from the Allen Harbor and reference sites were discussed in Section 4.2.1 (Figs. 4.2-3 through 4.2-8). Concentrations of these CoCs are discussed below.

tPAHs: Concentrations of total PAHs (tPAH) in the vegetated wetlands sediments within Allen Harbor range from 920 to 4,600 ug/kg compared with tPAH

concentrations of 26 ug/kg and 261 ug/kg in sediments from vegetated wetlands sites at the PCC and JSC reference sites, respectively (Table 4.3-1A). Intertidal sediments within Allen Harbor contain tPAH concentrations ranging from 20 - 21,000 ug/kg, as compared to 1313 ug/kg and 584 ug/kg for intertidal sediments at PCC and JSC respectively. The highest tPAH concentration within the Allen Harbor intertidal region occurred at Station W1 near the southern portion of the landfill. Concentrations of tPAHs in surficial sediments from the subtidal deep sites within Allen Harbor ranged from approximately 690 - 8100 ug/kg compared to tPAH concentrations of 15 to 18 ug/kg in reference site subtidal sediments.

TOC normalized tPAH concentrations in vegetated wetlands sediments range from 172 to 1,100 ug/kg, and from 13 to 368 ug/kg at associated reference sites (Table 4.3-2A). TOC normalized tPAH concentrations in intertidal wetlands sediments range from 58 to 4023 ug/kg, and from 850 to 886 ug/kg at associated reference sites (Table 4.3-2A). Concentrations of tPAHs in surficial sediments from the subtidal deep sites within Allen Harbor ranged from approximately 395 - 2182 ug/kg compared to tPAH concentrations of 27 to 40 ug/kg in reference site subtidal sediments. For each habitat, approximately 80 to 90% of the tPAH concentration in sediments corresponds to higher molecular weight (HMW) compounds, whereas the lower molecular weight (LMW) compounds (e.g., napthalenes and phenanthrenes) represent a considerably smaller component. Despite the large differences between sites in tPAH concentrations, the relative abundances of higher to lower molecular weight PAHs are remarkably consistent. This could indicate that a localized source of combustion or pyrolytic PAHs predominates over petrogenic sources. However, the relative proportions of HMW to LMW PAHs in sediments within the intertidal regions of the reference sites are consistent with those within Allen Harbor. Quinn et al. (1994) also reported similarly high proportions of HMW-PAHs in sediments from Coddington Cove. Thus, it is more likely that the relative abundances of individual PAH compounds in Allen Harbor sediments reflect regional input sources compounded with those associated with contaminant migration from the landfill.

Pesticides. Concentrations of chlorinated pesticides other than p,p'-DDE typically were at or below the respective analytical detection limits (approximately 1 ug/kg) in sediments within Allen Harbor and the reference sites (Table A1-1.20). Consequently, the following discussion of chlorinated pesticides in sediments focuses primarily on p',p-DDE (DDE). Within the Allen Harbor vegetated wetlands sites, sediment DDE concentrations ranged from 3.3 - 22.5 ug/kg; levels within the vegetated wetlands at the reference sites ranged from 0.2 - 0.4 ug/kg (Table 4.3-1B). Similarly, DDE concentrations in sediments from intertidal areas of Allen Harbor ranged from 0.1 - 24.0 ug/kg, whereas levels at the corresponding reference site locations ranged from 0.3 - 0.9 ug/kg. Subtidal, deep and deep core sediments contained DDE concentrations ranging from 1.4 - 40 and 0.2 - 3.1 ug/kg, respectively vs. 0.2 ug/kg at the reference sites.

TOC normalized sediment DDE concentrations within vegetated wetlands ranged from 0.7 - 3.2 ug/kg and 0.1 - 0.5 ug/kg at reference sites, whereas concentrations in sediments from intertidal wetland areas ranged from 0.24 - 8.13 ug/kg (Table 4.3-2B). Concentrations at corresponding reference sites were relatively low (0.1 - 0.7 ug/kg. TOC normalized DDE concentrations in subtidal, deep habitats ranged from 0.64 - 12.1 ug/kg, and were comparable to concentrations in other Allen Harbor habitats. Subtidal reference sites had concentrations of 0.1 - 0.5 ug/kg. TOC normalized DDE concentrations in subsurface sediments within Allen Harbor were relatively low, ranging from 0.6 to 12 ug/kg.

Sediments in subtidal offshore habitats contained greater concentrations of aldrin, HCB, and p,p'-DDE than other Allen Harbor habitats. Concentrations of o,p'-DDE and TOC normalized p,p'-DDE were also elevated in intertidal habitats. Sediment concentrations of mirex were comparable to concentrations at reference sites for all habitats. It should be noted that the p,p'-DDE summary values for the subtidal offshore habitats are skewed by sampling results of two RAPS sites that are 2 to 3 orders of magnitude greater than Phase III sample results.

PCBs. Concentrations of tPCBs in sediments of intertidal wetland sites ranged from 3.2 - 1345.2 ug/kg, (Table 4.3-1C) which was by far the widest range found in all habitats examined. tPCB concentrations at associated reference sites ranged from 8.82 - 13.50 ug/kg. Mean tPCB concentrations in vegetated wetland habitats and subtidal habitats were comparable at 159.95 ug/kg and 142.33 ug/kg respectively. tPCB concentrations ranged from 56.8 - 295.2 ug/kg in vegetated wetland stations and 2.1 - 6.3 ug/kg at associated reference sites. In subtidal stations, tPCB concentrations ranged from 16 - 505 ug/kg and 2.4- 7.5 ug/kg at reference sites. Subsurface (deep core) sediments at subtidal stations contained relatively low tPCB concentrations, ranging from 6.6 - 53 ug/kg. Associated reference sites tPCB concentrations ranged from 2.1 to 3.60 ug/kg.

TOC normalized sediment tPCB concentrations at different Allen Harbor habitats are summarized in Table 4.3-2C. Intertidal wetland habitat TOC normalized tPCB concentrations ranged from 7.8 - 528 ug/kg vs. 9.9 - 13.5 ug/kg at reference sites. Vegetated wetlands habitats had a range of 11 - 77 ug/kg while tPCB concentrations at comparable reference sites ranged from 1.3 - 9.5 ug/kg. TOC normalized tPCB concentrations in subtidal habitats ranged from 9.2 - 143 ug/kg, compared to 5.4 - 19 ug/kg at reference sites. Deep core subtidal sediments contained the lowest TOC normalized tPCB concentrations ranging from 2.5 - 40 ug/kg.

TBT: Concentrations of TBT (reported as the concentration of Sn in the sample) are uniformly low throughout the portions of Allen Harbor sampled during the Phase III study, and approximately one order of magnitude lower than in the RAPS study results. Sediments from the vegetated wetlands, intertidal, and subtidal sites contained TBT concentrations ranging from approximately 0.4 - 4.7, 0.1 - 53 ug/kg, and 0.7 - 112 ug/kg, respectively (Table 4.3-1D). The corresponding TBT concentrations in the vegetated wetlands areas of the reference sites were not detectable, and concentrations within the intertidal areas of the reference sites was

approximately 3 ug/kg. Phase III results showed maximum TBT concentrations of 6.9 ug/kg in subtidal sites, and 7.7 ug/kg in intertidal wetlands, which are two orders of magnitude less than RAPS results. For comparison, Quinn et al. (1994) measured TBT concentrations in Coddington Cove ranging from 32 - 372 ug Sn/kg and Wade et al. (1990) reported concentrations from <5 - 187 ug/kg in marine sediments. Consequently, sediment TBT concentrations do not appear to be elevated within Allen Harbor.

Metals. The highest total metal concentrations were found at intertidal wetland stations, and ranged from 604 - 3793 ug/kg compared to a range of 34.6 - 44.4 ug/kg at associated reference stations (Table 4.3-1E). Elevated concentrations were due in large part to the elevated concentrations of zinc, copper, and lead at intertidal wetland sites. Concentrations of these metals ranged from 1.13 - 2828 ug/kg for zinc, 3.13 - 519.19 ug/kg for copper, and 3.00 - 465 ug/kg for lead (Table A1-1.6A). Total metal concentrations at sudtidal and vegetated wetland stations were comparable with respective ranges of 106.3 - 534 ug/kg, and 128 - 559 ug/kg compared to 26.1 - 35.0 ug/kg and 34 - 48 ug/kg at reference sites. Elevated metal concentrations in the subtidal and vegetated wetland habitats were dominated by elevated concentrations of zinc (31.2 - 214 ug/kg and 19.7 - 221.5 ug/kg), lead (14- 77.3 and 30 - 97 ug/kg), copper (22 - 127 and 24- 114 ug/kg), and chromium (29 - 74 and 27 - 66 ug/kg). It should be noted however that deep core and surficial subtidal sediments contained comparable chromium concentrations, while other metals were generally found in greater concentrations in surficial sediments.

VOAs: As discussed in Section 4.2, concentrations of VOAs in Allen Harbor sediments typically are low and within the range of <1 to several ug/kg (Table 4.3-1F). The exceptions are spuriously high values which are probably attributable to analytical artifacts and/or laboratory contamination, and an unusually high concentration of chlorobenzene at intertidal Station W6. Accumulations of high VOA concentrations in marine sediments are unlikely (Section 4.2), except possibly in areas with high

sediment deposition rates and/or large mass emissions, due to the high solubility and low partitioning behavior of this compound class.

4.3.2. Tissues

Spatial distributions of tPAHs, DDE, tPCBs, TBT and metals in shellfish tissues were discussed in Section 4.2 (Figures 4.2-9 through 4.2-13). The magnitude of these concentrations relative to levels in each of the four species measured at the reference sites are discussed below.

tPAHs: In general, tPAH concentrations are considerably higher in ribbed mussels than in the other three shellfish species (Table 4.3-3). Ribbed mussels collected from the vegetated and wetlands sites within Allen Harbor contained tPAH concentrations ranging from 2,425 - 5870 ug/kg compared to concentrations of 2,180-2,356 at the reference sites. Soft shell and hard shell clams and oysters from intertidal areas of Allen Harbor contained 647-977 ug/kg, 200-355 ug/kg, and 254-499 ug/kg tPAH, respectively. Mummichog fish contained 100-200 ug/kg, while at the reference sites the fish contained 135-144 ug/kg tPAH. Average concentrations of tPAH in hard clams and oysters from the reference areas contained 133 ug/kg and 175 ug/kg tPAH, respectively; no data for soft shell clams from the reference site are available for comparisons.

Pesticides. Similar to the sediments, concentrations of chlorinated pesticides, other than p,p'-DDE, were typically are at or near the detection limits in shellfish tissues from Allen Harbor and from the reference sites. Consequently, the following discussion focuses on DDE concentrations in shellfish tissues. Relative to tPAH concentrations in shellfish tissues, DDE concentrations are more consistent among the four shellfish species (Table 4.3-3). Ribbed mussels within the vegetated wetlands and intertidal areas of Allen Harbor contained a DDE concentration of 13.3-52.0 ug/kg; the corresponding tissue DDE concentrations at the reference sites ranged from 9.3-12

ug/kg. Tissues of soft shell clams contained 6.9 - 14.4 ug/kg DDE and oyster tissues contained 23-47 ug/kg DDE. For comparison, the average DDE concentration in oyster tissues from the intertidal areas of the reference site was 22 ug/kg. The tissues of hard shell clams from the intertidal and subtidal areas of Allen Harbor contained 3.4-21 ug/kg while the corresponding reference site areas was 3.2 ug/kg. Mummichog fish collected from intertidal areas of Allen Harbor and reference locations ranged from 59.4-116.4 and 30.8-44.8 ug/kg, respectively.

PCBs. The tPCB concentrations in ribbed mussels from the vegetated wetlands and intertidal sites ranged from 188-1267 ug/kg (Table 4.3-3). In comparison, reference site concentrations for this species ranged from 144-274 ug/kg. Soft shell clam and oyster tissues from intertidal areas of Allen Harbor contained 178-840 ug/kg and 302-1202 ug/kg, respectively, while oyster tissues at the reference site contained 192-289 ug/kg tPCB. Finally, tPCB concentrations in hard clams from the intertidal and subtidal areas of Allen Harbor ranged from 57-545 ug/kg compared to concentrations of 51-120 ug/kg at the reference sites.

Organotins. Spatial distributions butyltins (BT) in Allen Harbor tissues were summarized in Section 4.2 (Figure 4.2-13). In general, the total BT concentration was dominated by tributyltin (TBT) compounds. In general, tissue concentrations of TBT were 2-45 ng Sn/g dry weight for each of the species evaluated (Table 4.3-3). Concentrations of TBT in shellfish tissues exceeding this range included a single, non-depurated *Mya* sample from station W5 (MYA-ND); however this value was nearly 10-fold higher than concentrations found in the same collection but depurated (MYA-DEP). Given that ribbed mussel (both depurated and nondepurated samples) as well as sediment Sn concentrations at this station were unremarkable (Figure 4.2.6), this result would appear to be an artifact of the laboratory analysis. Only mummichog fish (MF) tissue residues at harbor stations (2-16 ng Sn/g) appeared elevated relative to reference sites (0.3-1.5 ng Sn/g).

Metals: Maximum concentrations of chromium and in tissues of ribbed mussels from intertidal stations (27 and 66 ug/g, respectively) were about an order of magnitude higher than corresponding tissue concentrations from the reference site (2.8 and 6.6 ug/g, respectively) and about five-fold higher based on mean concentrations (Table 4.3-4). In contrast, other metals showed generally comparable or lower concentrations at Allen Harbor stations vs. reference locations

Residue concentrations of metals in soft shell clams were less than 10 ug/g except for zinc which ranged from 61-230 ug/g. No soft shell clams could be collected at the reference site for comparison. Hard shell clams from subtidal areas of Allen Harbor contained 34-115 ug/g zinc which was slightly higher than in clams from the reference site (74-89 ug/g). Concentrations of other metals in clams from both intertidal and subtidal sites occasionally exceeded the respective concentrations in the reference site clams; however, maximum coincentrations were less than 13 ug/g in all cases.

Metal residues in oysters were less than 10 ug/g at Allen Harbor sites except for copper, mercury and zinc (Table 4.3-4). Although copper residues at reference locations (8.6-323 ug/g) were comparable to Allen Harbor oysters (41-367 ug/g), observed mercury (0.08-49 ug/g) and zinc (529-5157 ug/g) levels were far higher than observed at reference sites (0.01-0.10 and 94-450 ug/g, respectively). Except for zinc, metal concentrations in the tissues of *Fundulus* spp.from Allen Harbor were less than 10 ug/g and were comparable to reference site values. Zinc residue values were 150-250 ug/g in Allen Harbor vs. 180-220 ug/g at reference locations.

4.3.3. Summary of Exposure Point Concentrations

Considerable variation was observed in the concentration of metals in sediments and tissues of biota from Allen Harbor. At some locations within Allen Harbor and for selected contaminant classes, exposure point concentrations are up to

one to two orders of magnitude higher than those represented by the reference levels. Similarly, elevated concentrations of selected contaminants in tissues of shellfish from Allen Harbor indicate higher exposure and/or bioavailability than for areas representative of the background exposure conditions.

Some metals are present naturally in tissues of marine organisms (e.g. copper and zinc), and are required for physiological functions. Background metal concentrations vary between species depending on physiological requirements, feeding habits, gender, age, and sexual maturity. However, excessive concentrations can become toxic, and shellfish tissues from within Allen Harbor contain contaminant concentrations that are elevated compared to levels associated with regional input sources, as represented by the magnitude of corresponding contaminants at the reference sites.

4.4 UNCERTAINTY

Contaminant sources from the landfill and Calf Pasture Point areas have been well characterized based on the Phase I and II RAPS and RI studies. However, the exposure pathways as reflected by the first through fourth tier models (Section 3 and Section 4.1) are necessarily conceptual and cannot account for all the complexities, including proximal and distal sources and receptors, of a natural ecosystem. These uncertainties also are driven by incomplete knowledge of the chemical behavior of the CoCs, even though considerable information is available on solubility, partitioning, and toxicity for several analytes. Nonetheless, existing information on the chemical constituents and a reasonably thorough understanding of the Allen Harbor ecosystem have allowed sufficient and relevant data to be targeted, collected, and interpreted for the risk assessment.

Fate and transport evaluations for the exposure assessment focused on spatial (horizontal) and vertical (sediment layering) patterns as well as data comparability among the study phases (i.e., temporal consistency). The placement of sampling

stations was largely based on providing "visually complete" (essentially "gridded") coverage of the various habitats (e.g., marsh, intertidal, and offshore) within the potential impact and reference zones of the study region. Station placement was guided additionally by results from prior phases; however, visual coverage was still the principal method applied. The uncertainty associated with any sampling station is whether it is truly representative of the habitat and impact/reference zone being evaluated. Collection of station replicates is one method that allows assessment of within-station variability (i.e., the representativeness of a sample), although only single samples were collected per station for this study. Further, various methods of statistical power analysis can used to determine, for example, the number of randomly placed samples that are necessary to characterize, with a desired level of confidence, a particular sampling regime (habitat and zone).

For the present study, comparison of the data variability among stations within a habitat zone is the primary method used to assess adequacy and representativeness of the sampling grid. Conceptually, this is fundamental to ongoing debates about the uncertainties of extrapolations (and assumptions) from point measurements to broader spatial areas. Quantitative approaches using Geographic Information Systems (GIS) technology have been recently reported (Clifford et al., 1995) which appear to provide an effective, unbiased method for estimating spatial extent of risk, with minimum uncertainty and maximum data usage. These techniques will undoubtedly be more prevalent in future studies where large databases are available for quantitation.

An area of uncertainty for the assessment is the temporal comparability of data among the phases. The general study design assumes that there have not been substantial changes in environmental conditions and chemical contaminant concentrations at individual sampling sites, as representative of particular habitat and sampling zones, over the various phases. However, in practice interannual and seasonal variations occur in every environment, thereby changing to some degree the conditions that influence contaminant sources, exposure pathways, and receptors.

Further, the navigational methods used to locate stations varied somewhat among the phases, with the present study using more accurate methods (± 3m) than earlier phases lacking direct measurement of position. Nonetheless, the validity of the assumptions concerning insignificant temporal changes was tested by comparing the relative percent difference (RPD) between concentrations of representative CoCs for sets of two stations at AH5 which were sampled during different phases differed by approximately a factor of two (see Section 4.2). It is not readily discernable whether this difference is related to temporal trends or to laboratory and/or micro-scale variation in sediment composition.

Different laboratories were involved in the analysis of samples which could cause differences in the results, lending a degree of uncertainty. The University of Rhode Island (URI) and Texas A&M were services laboratories responsible for sediment and tissue metals and organics analysis, respectively. Tissue metals samples were sent to the University of Mississippi (UMISS) for analysis, while sediment samples which were to be analyzed by URI were sent to another laboratory (CEIMIC Corporation) for analysis.

For trace metal analyses sediment samples were prepared according to the microwave total digestion method used by URI. Reasonable results were obtained for most analytes with the exceptions of Ag, As, and Ni. Either the recoveries of these elements were low, or the data seemed irregular. These analytes were rerun by URI, and URI values are used in the final data compilation. In addition, Zn was rerun by URI as a check on the comparability of the CEIMIC and URI data, with the results showing that the data are comparable. The URI Zn data are used in the final data compilation.

A comparison of trace metal data obtained from split samples (four organism tissue samples from the same four sites in Allen Harbor) were analyzed by URI and the University of Mississippi. Results are shown in Table 4.4-1. The major observation

that can be made about individual analyte comparisons is that the comparability is variable depending on the specific analyte. Major potential sources of variability include: 1) the samples represent a random division of the organisms obtained at each site and may not be that comparable due to variations in the age/size distribution of organisms sent to each lab, and 2) the trace metal digestion SOPs used by the two labs were different. Mississippi used a total digestion with the HF method, whereas URI used a partial digestion with nitric acid method. In general, Mississippi obtained higher values for Al, Fe, Cr, and Ag; metals that generally give significantly higher recoveries when using a total digestion method. Comparable ranges of values were obtained for Zn, Pb, Ni, Cd, and As; metals that generally give similar recoveries in tissues using either a partial or a total digestion method. It is noteworthy that very high Zn concentrations were observed in oyster samples by URI, but not by Mississippi.

In general, the results obtained indicate good comparability (< 2-fold mean variation) given the differences in the samples and in digestion method. The mean RPD by analyte ranged from a low of 24% (Cd) to a high of 150% (Cr), and the overall RPD for metals data between studies was 86%.

A comparison of the trace metal results obtained from two studies of sediments from Allen Harbor station AH-5 are shown in Table 4.4-2. The results from ERLN, 1991 are very comparable to those obtained in this study particularly when the differences in trace metal digestion method are considered; an overall RPD of 46.8% was observed. The largest RPD was 200% for Hg, which perhaps reflects better measurement and/or instrumentation for the 1995 study (Leeman Labs Hg Analyzer Model P200) vs. the 1991 study (Perkin Elmer 403 AA with MHS-20 integrator), but also difficulties due to concentrations at or near detection limits. Thus, the metal concentrations obtained in this study (1995) are either comparable, or are somewhat higher than those obtained in 1991. Given that new data were obtained using a total digestion method, the higher values for several of the analytes are reasonable.

A comparison of % recoveries of trace metals obtained by analysis of standard reference materials (SRM) for the ERL-N 1991 study and the URI (1995) study was also performed (Table 4.4-3). Comparable results were obtained for most analytes. URI (1995) recoveries for Hg and Pb are low in comparison to ERL-N (1991) for tissues. Low lead recovery may be due to the value for the SRM being near the detection limit.

Finally, the exposure point estimates are based on representative chemical analytes that, due to practicality, are a subset of the total possible compounds that could be analyzed. However, the analytes have been carefully selected as a result of extensive screening and analyses during the present and previous phases and are onsidered to be appropriately conservative and representative of source constituents.

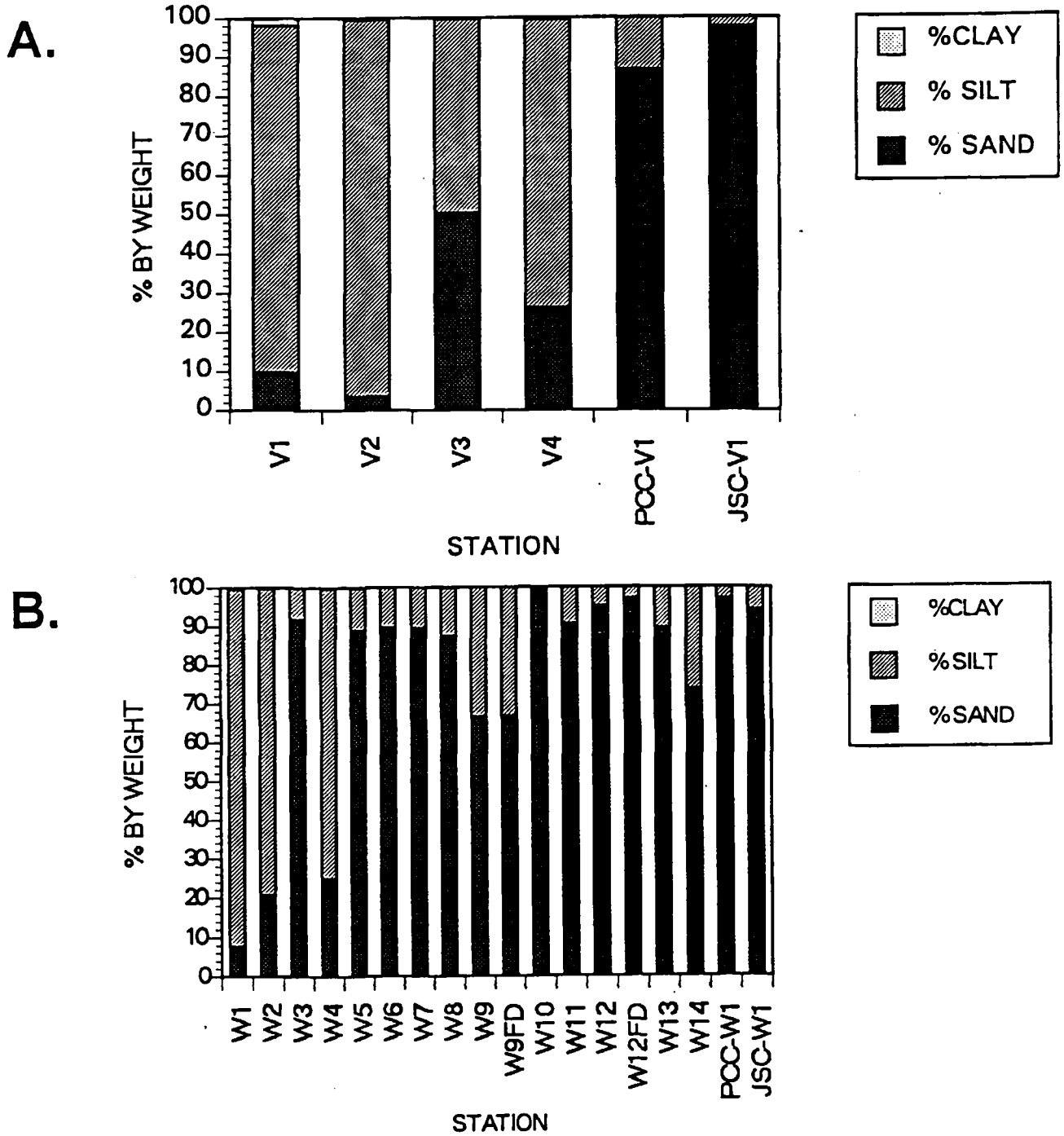
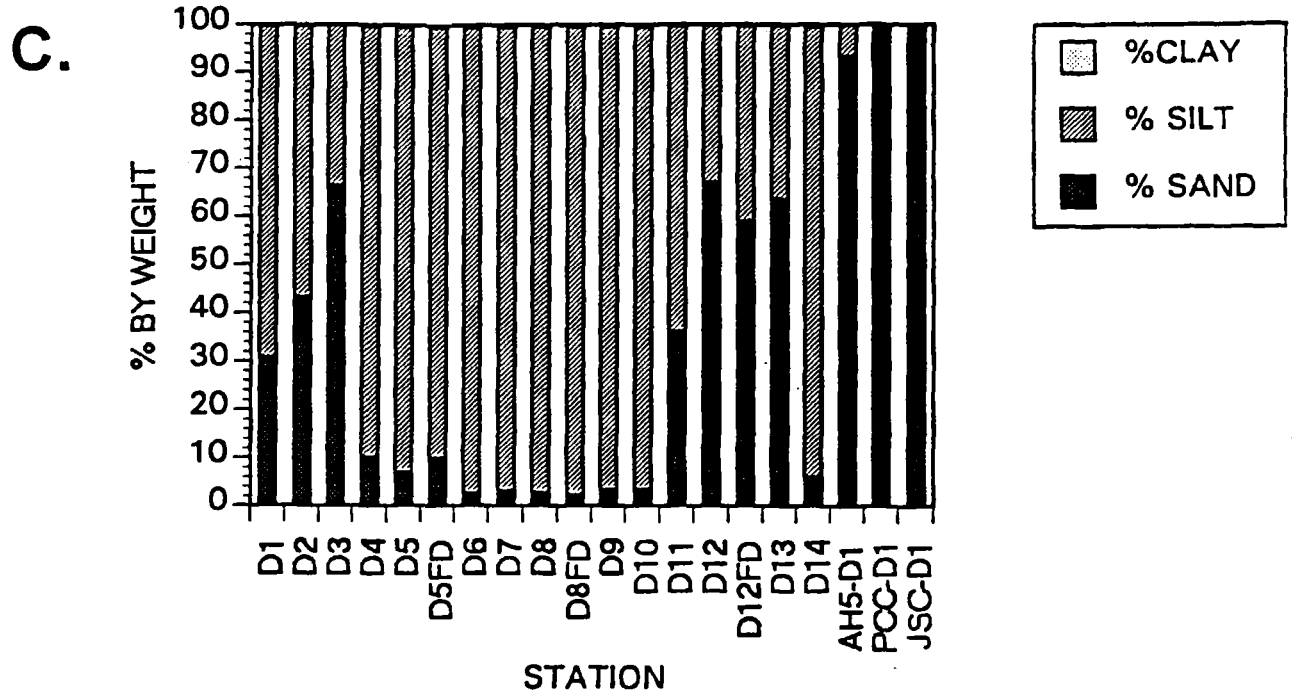


Figure 4.2-1. Grain size characteristics for: A) vegetative, B) wetland, C) subtidal-deep and D) subtidal-deep core sites.



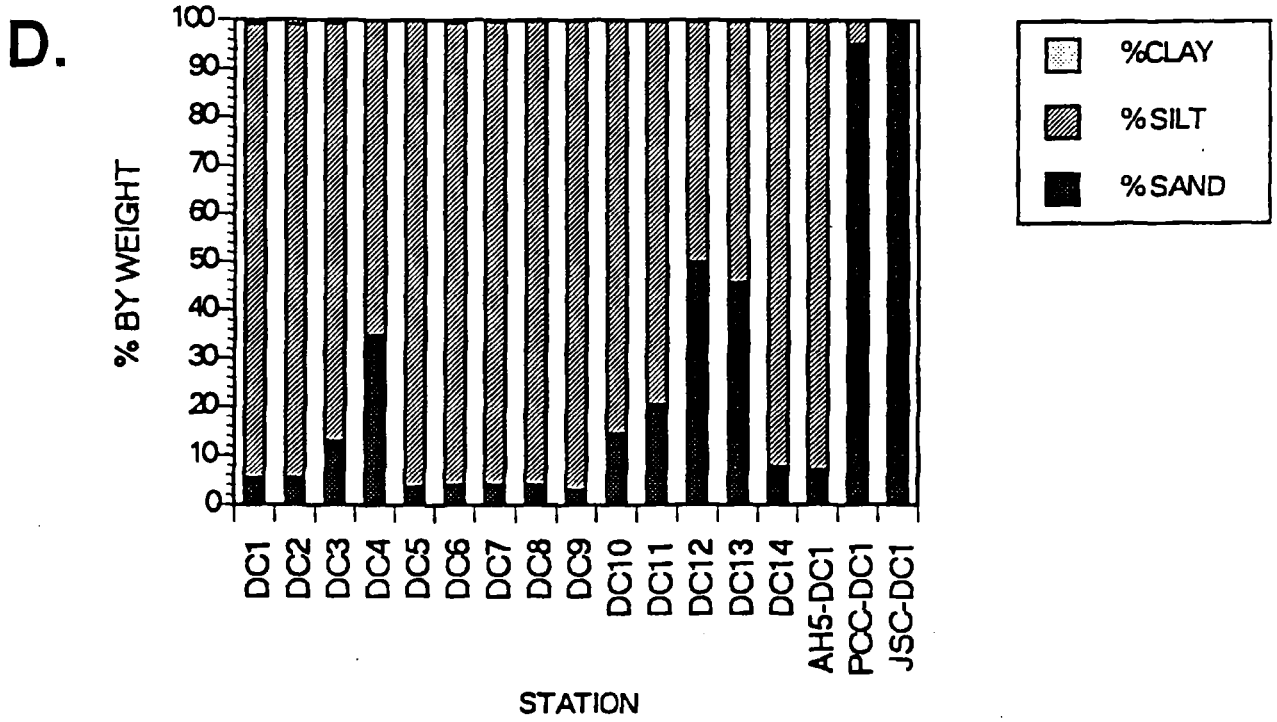


Figure 4.2-1. (con't.)

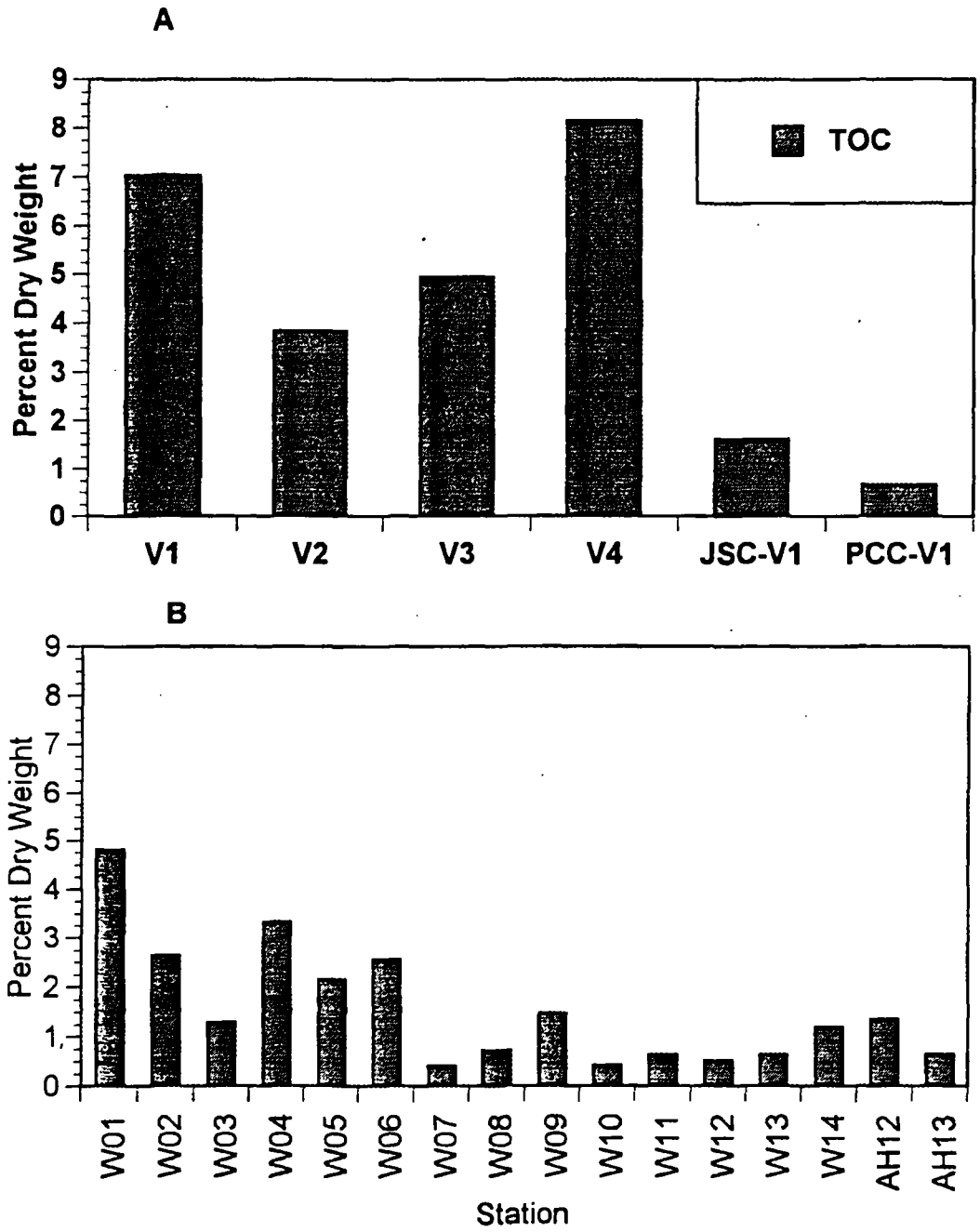
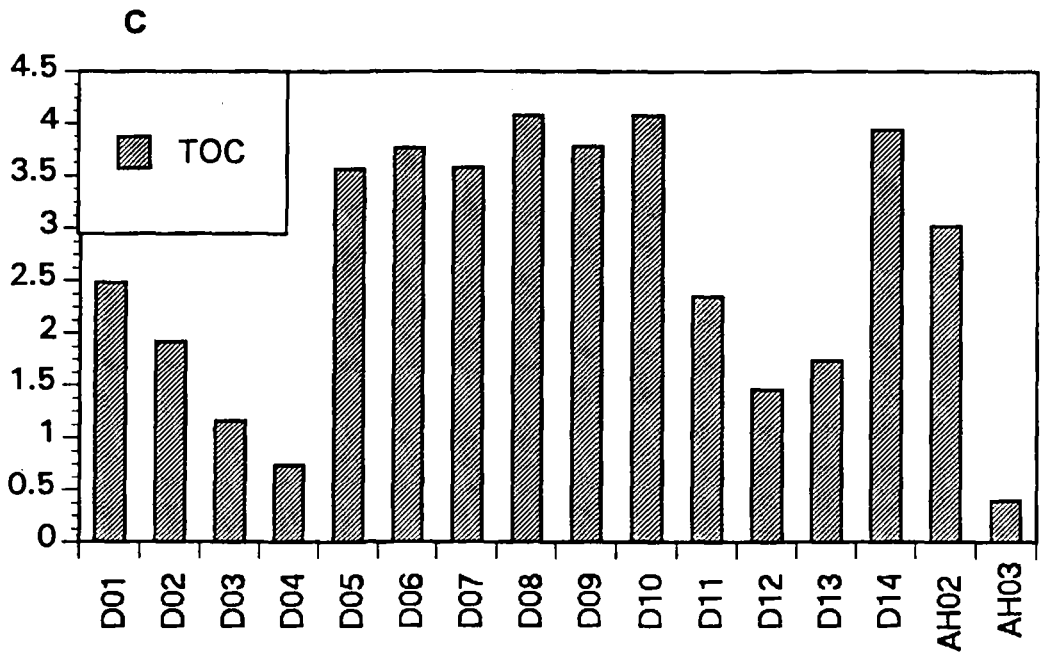


Figure 4.2-2. Total organic carbon concentrations (% dry wt.) of sediments from: A) vegetative, B) wetland, C) subtidal-deep, and D) subtidal-deep core sites.



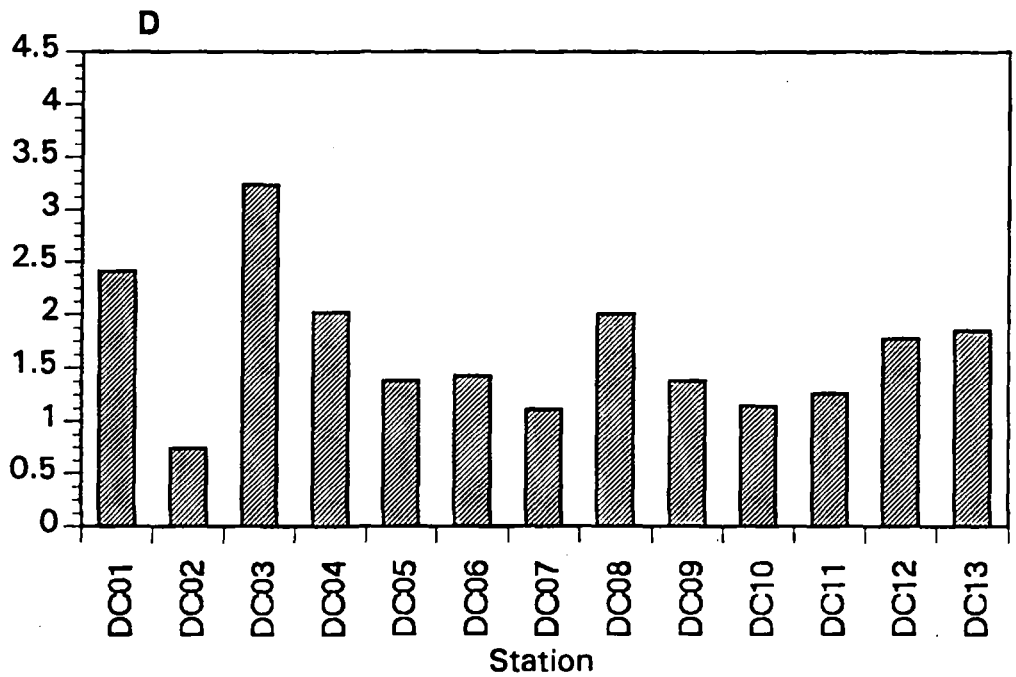


Figure 4.2-2. (con't.)

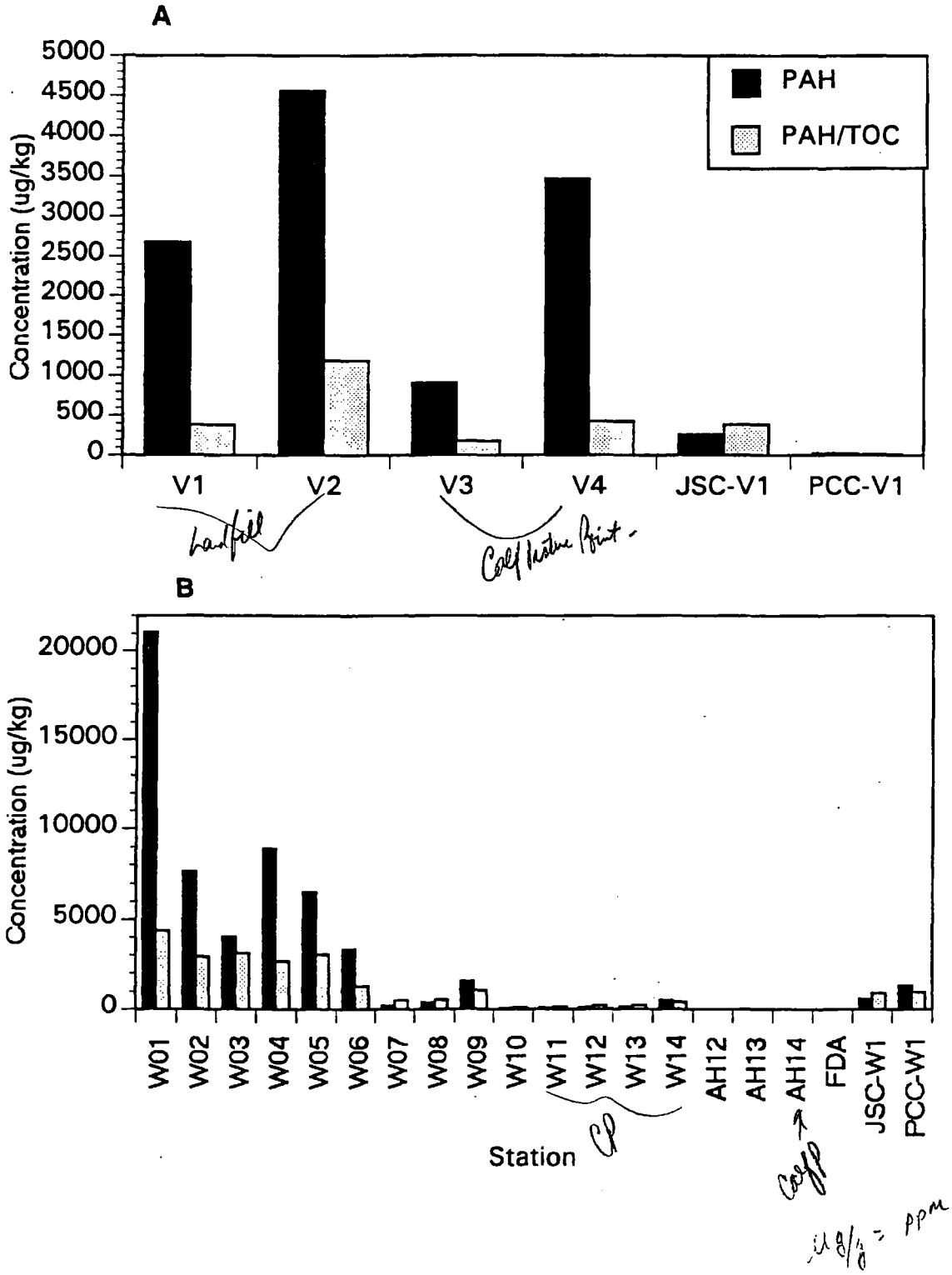
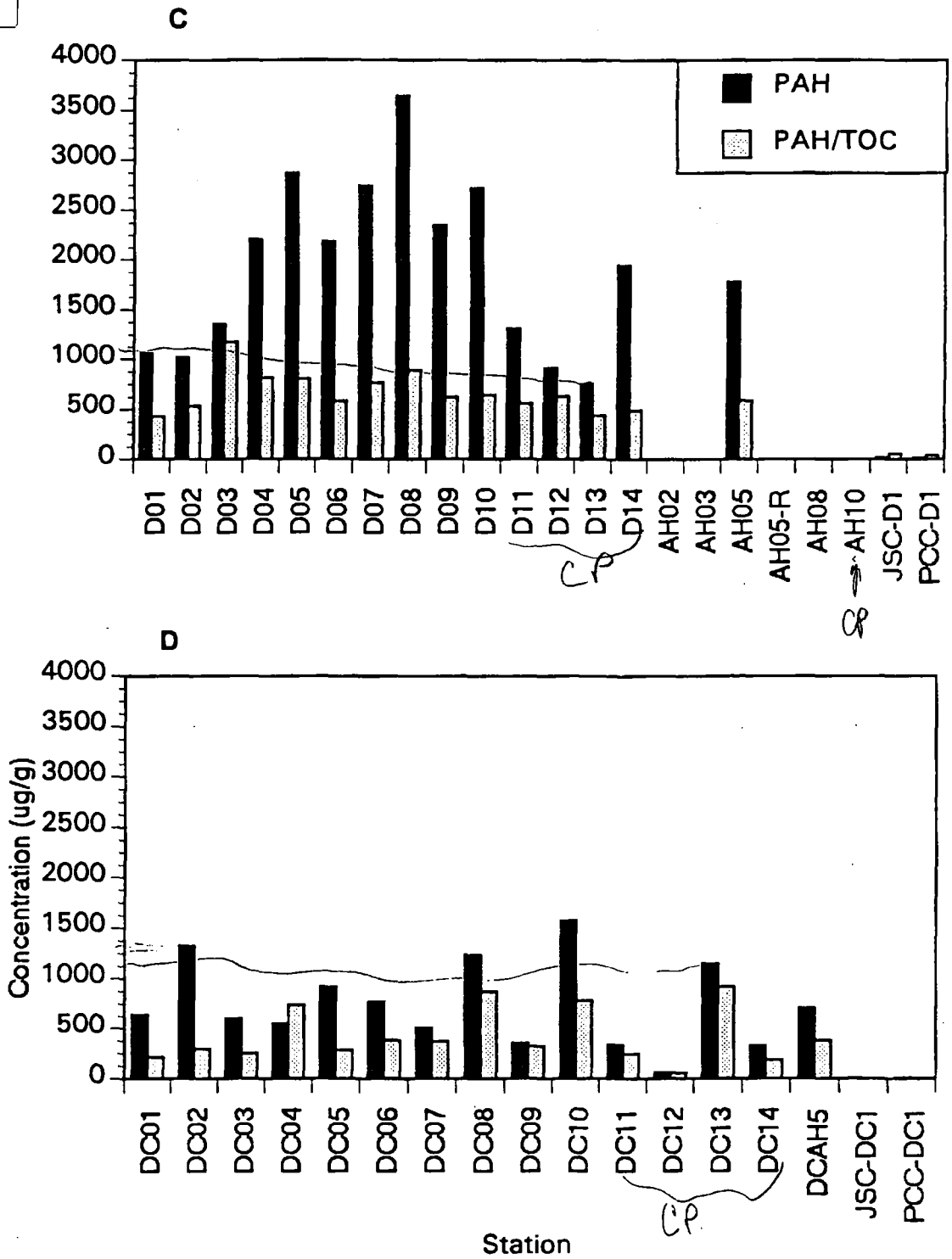


Figure 4.2-3. Total PAH concentrations (µg/kg dry wt.), normalized and non-normalized to TOC, of sediments from: A) vegetative, B) wetland, C) subtidal-deep, and D) subtidal-deep core sites.



DC02 Deeper is 1300ppb

DC13 Same as D13

Figure 4.2-3. **(con't.)**

D002 shallow is 1800 ppb

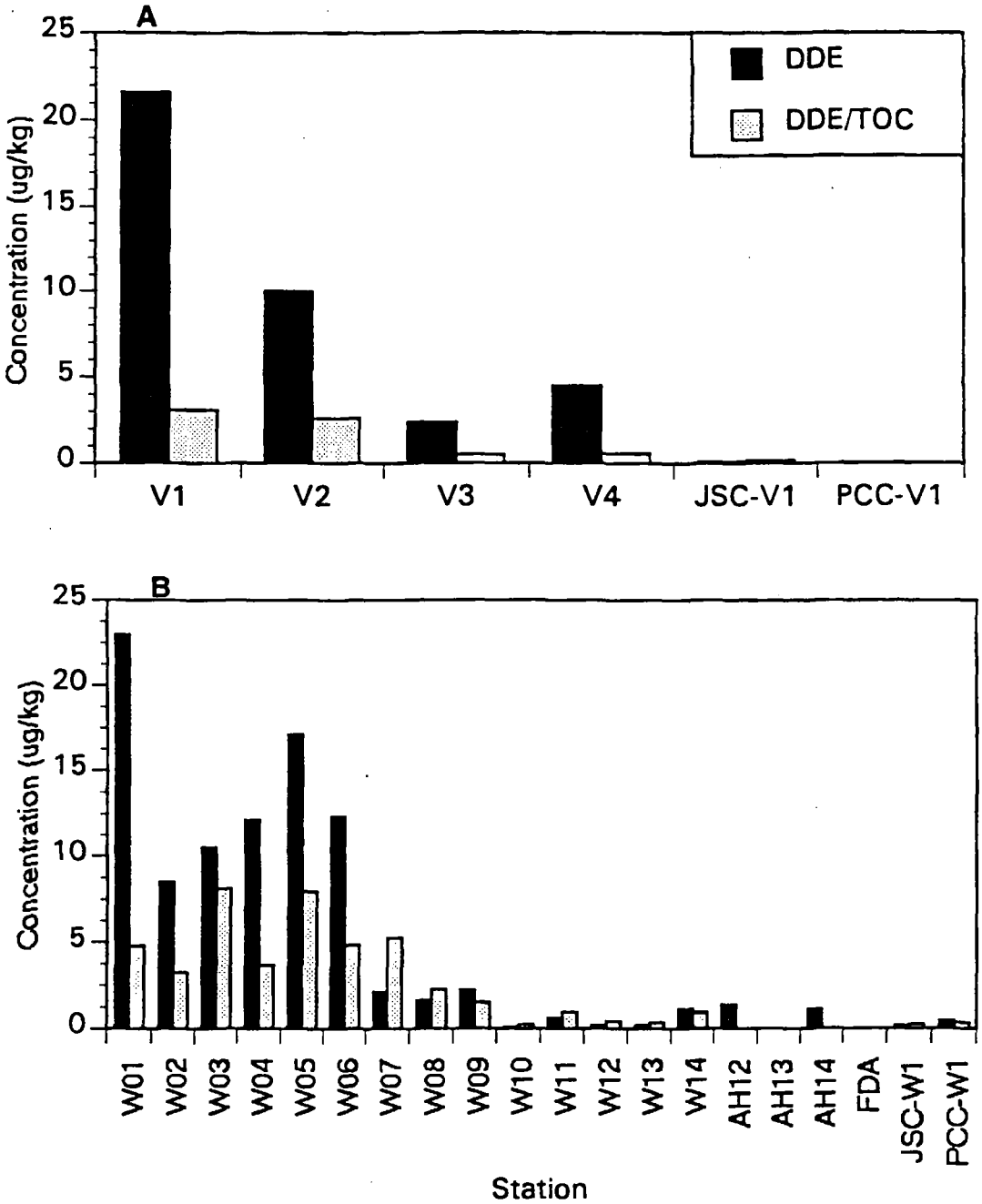


Figure 4.2-4. DDE concentrations (µg/kg dry wt.), normalized and non-normalized to TOC, of sediments from: A) vegetative, B) wetland, C) subtidal-deep, and D) subtidal-deep core sites.

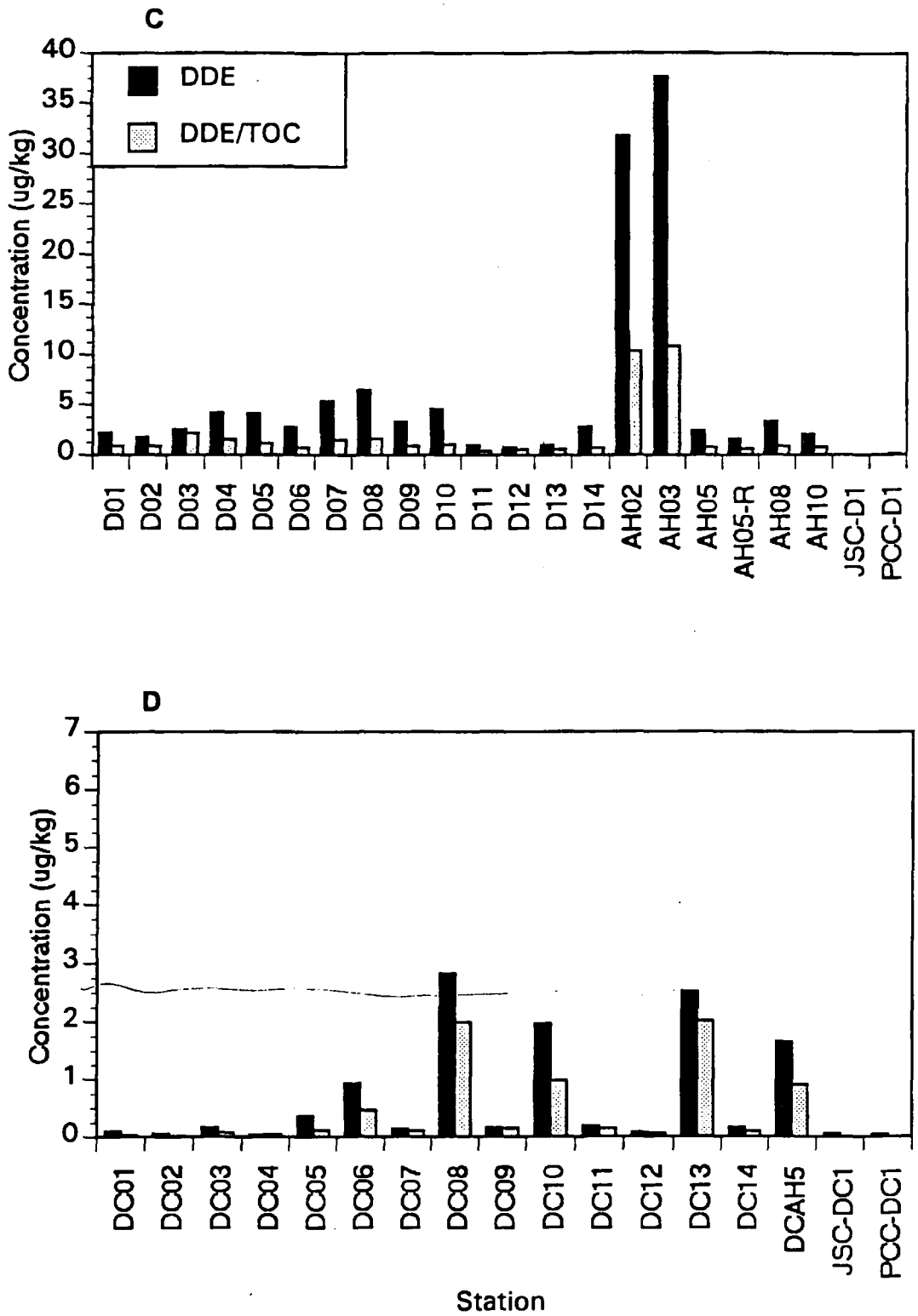


Figure 4.2-4. (con't.)

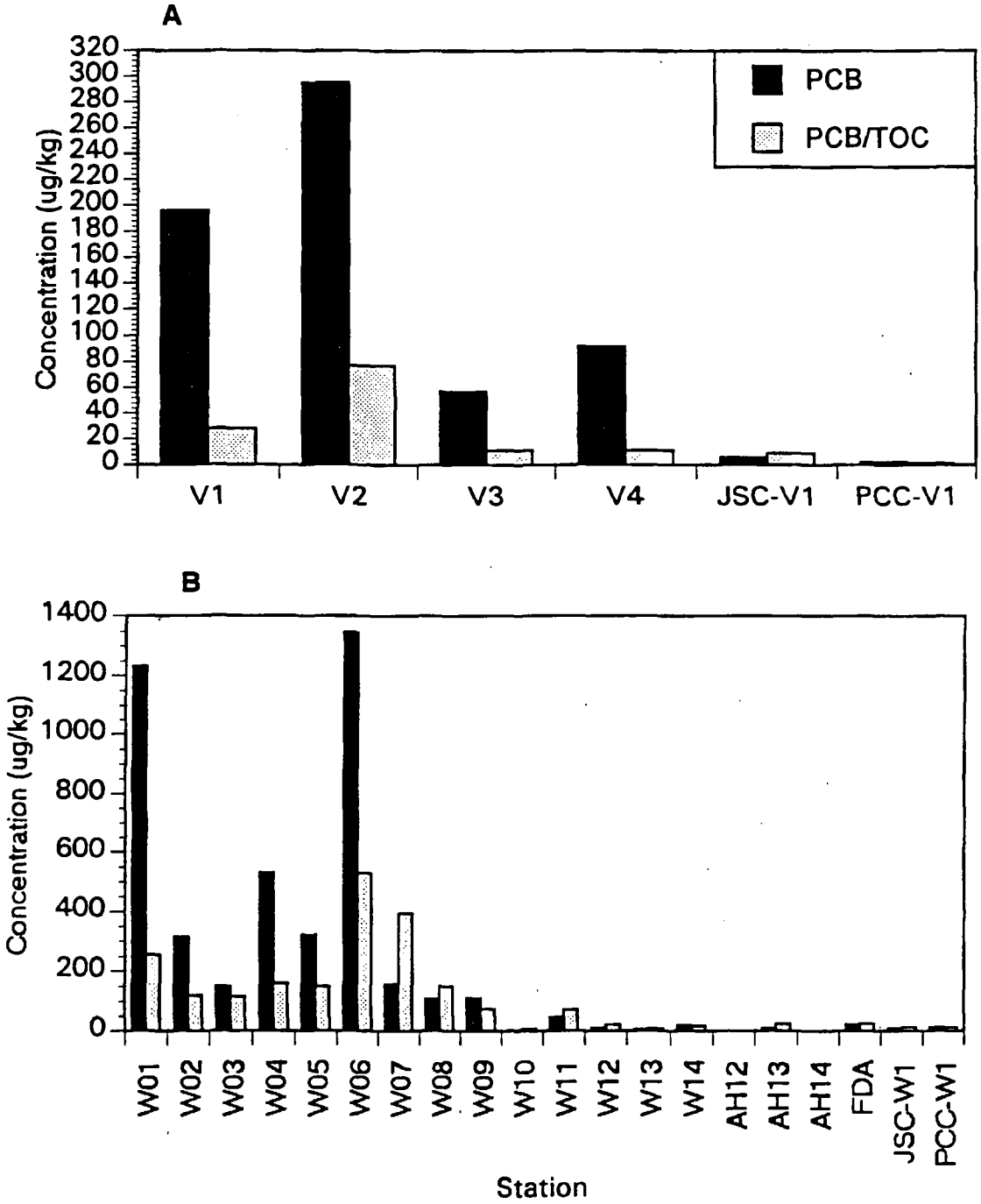
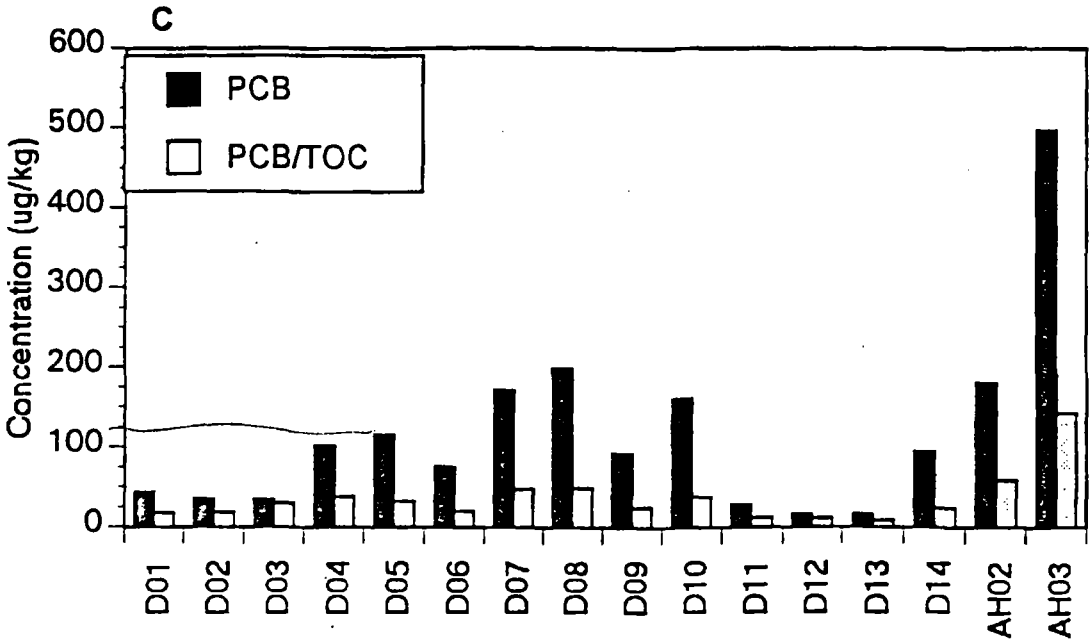


Figure 4.2-5. Total PCB concentrations (µg/kg dry wt.), normalized and non-normalized to TOC, of sediments from: A) vegetative, B) wetland, C) subtidal-deep, and D) subtidal-deep core sites.



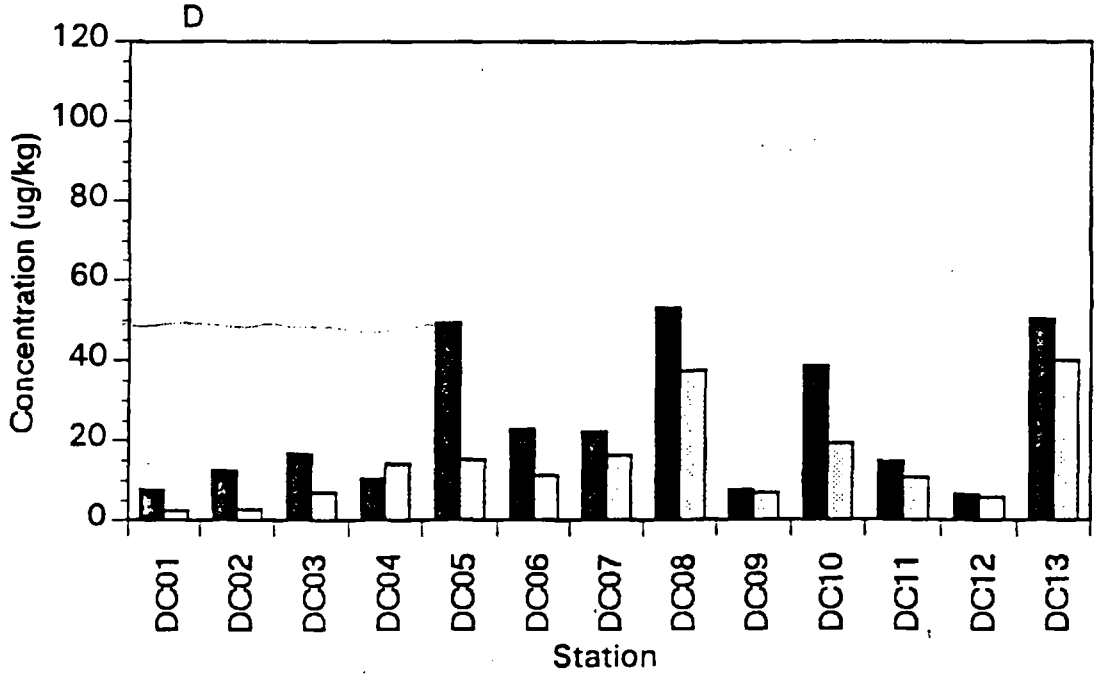


Figure 4.2-5. (con't)

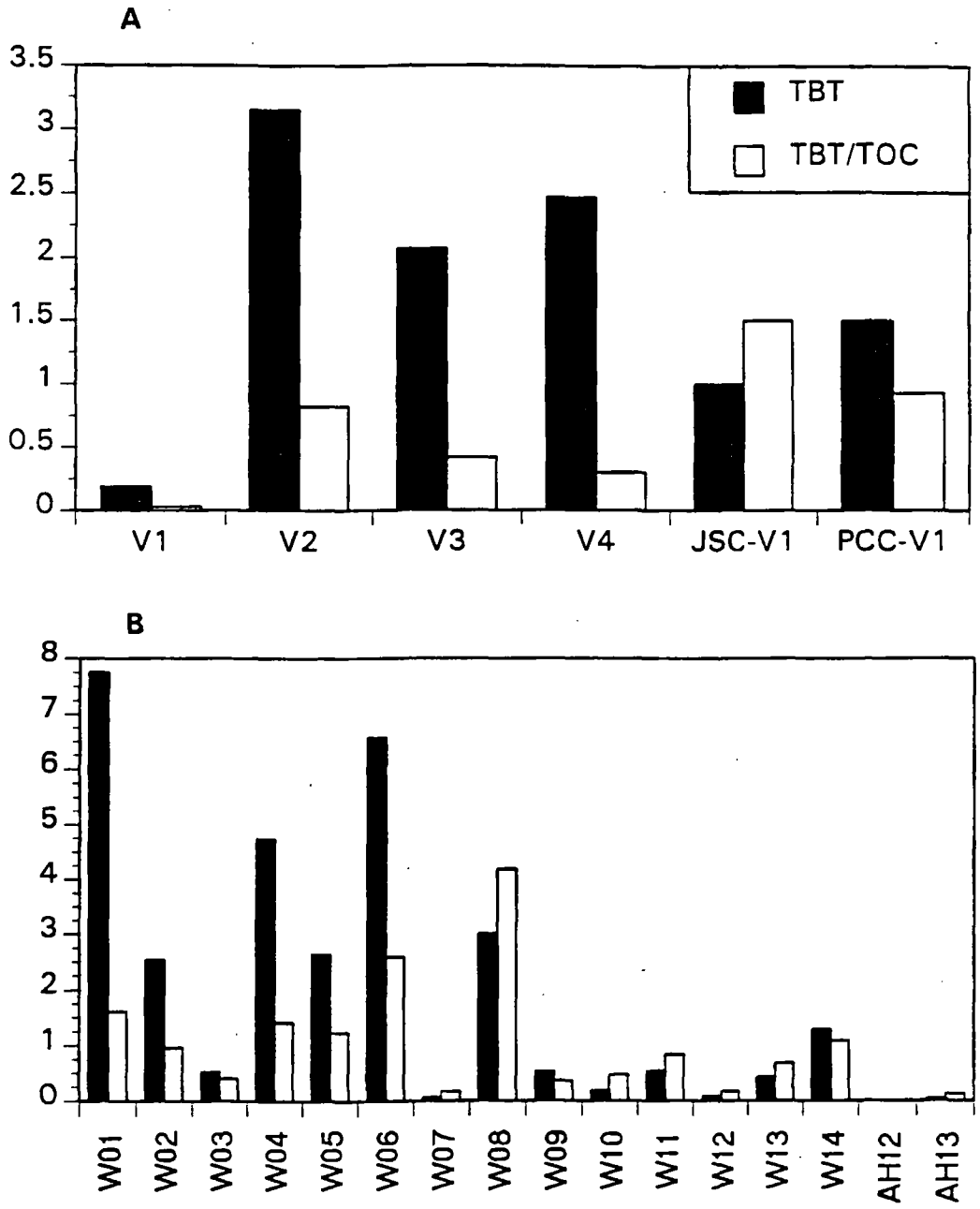
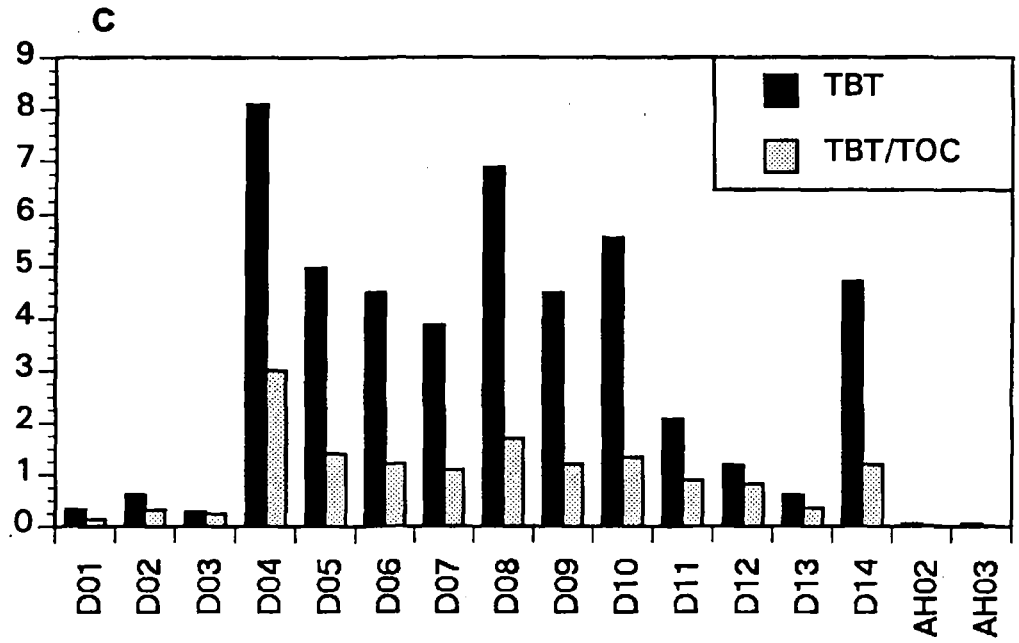


Figure 4.2-6. Butyltin concentrations (µg Sn/kg dry wt.) of sediments from: A) vegetative, B) wetland, C) subtidal-deep, and D) subtidal-deep core sites.



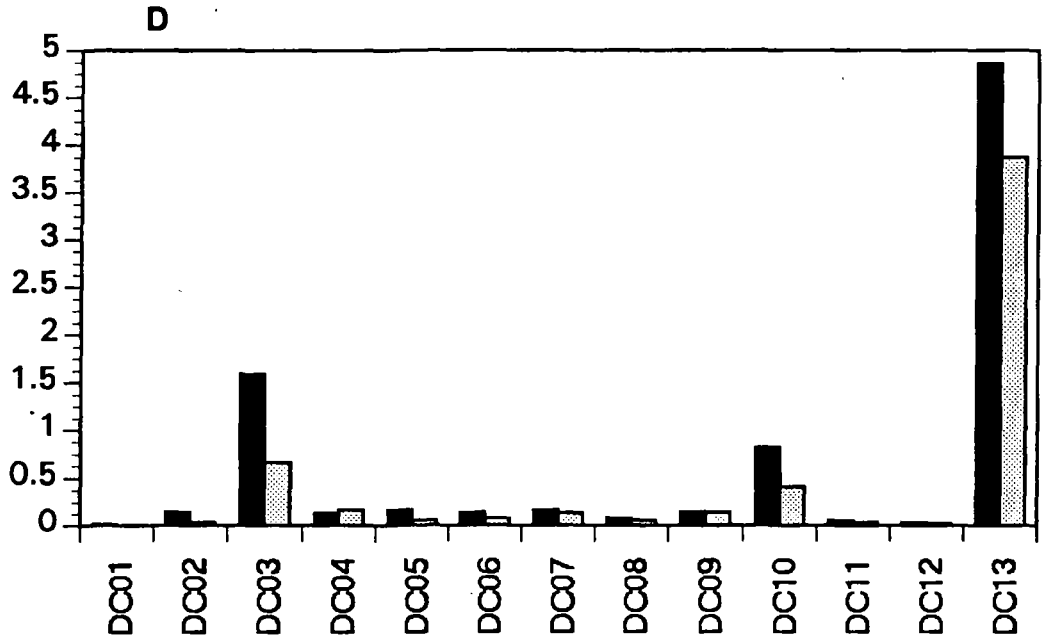


Figure 4.2-6. (con't.)

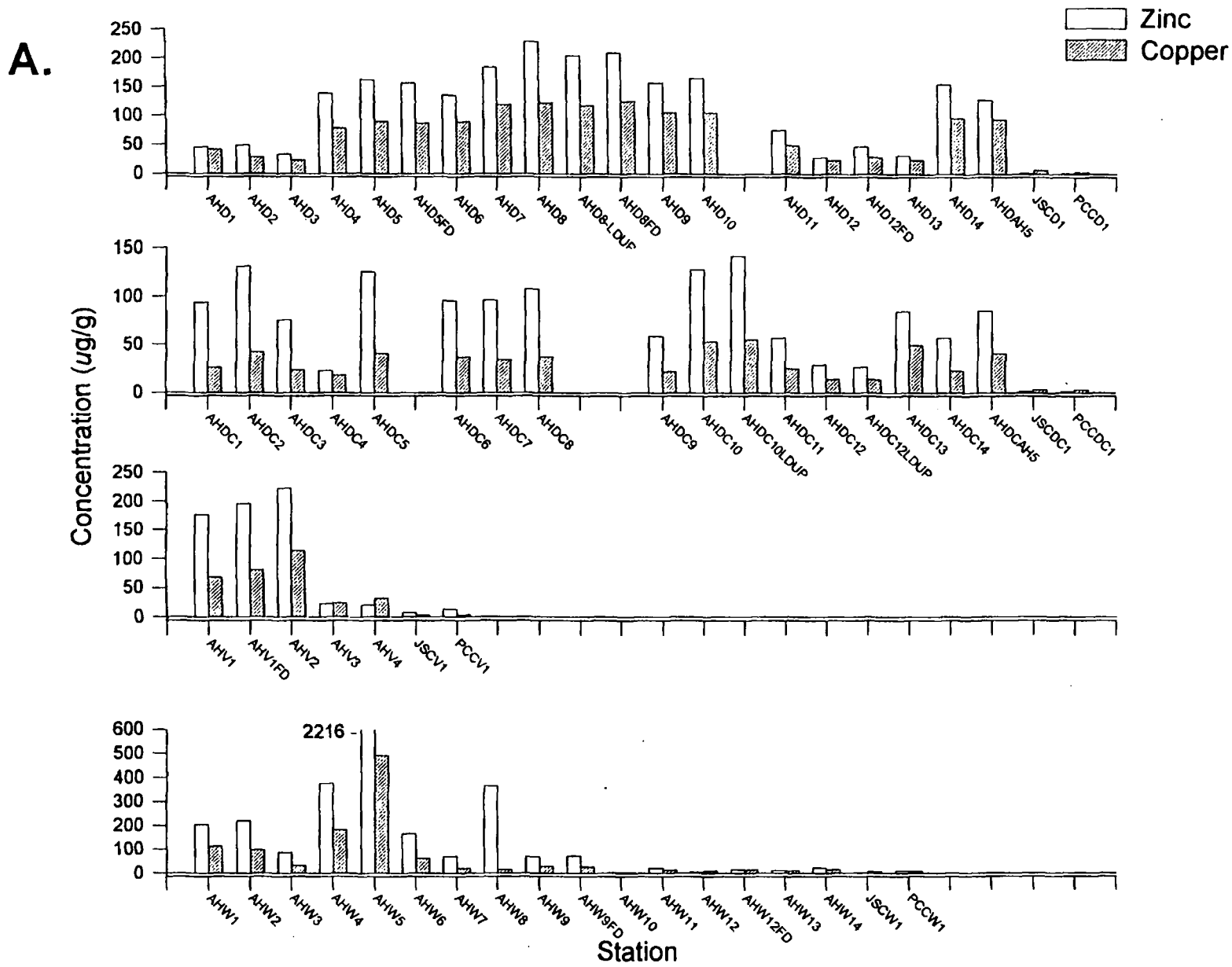


Figure 4.2-7. Concentrations of metals (mg/kg) in sediments from Allen Harbor: A) Zinc and Copper, B) Chromium and Lead, C) Nickel and Arsenic and D) Silver, Cadmium and Mercury.

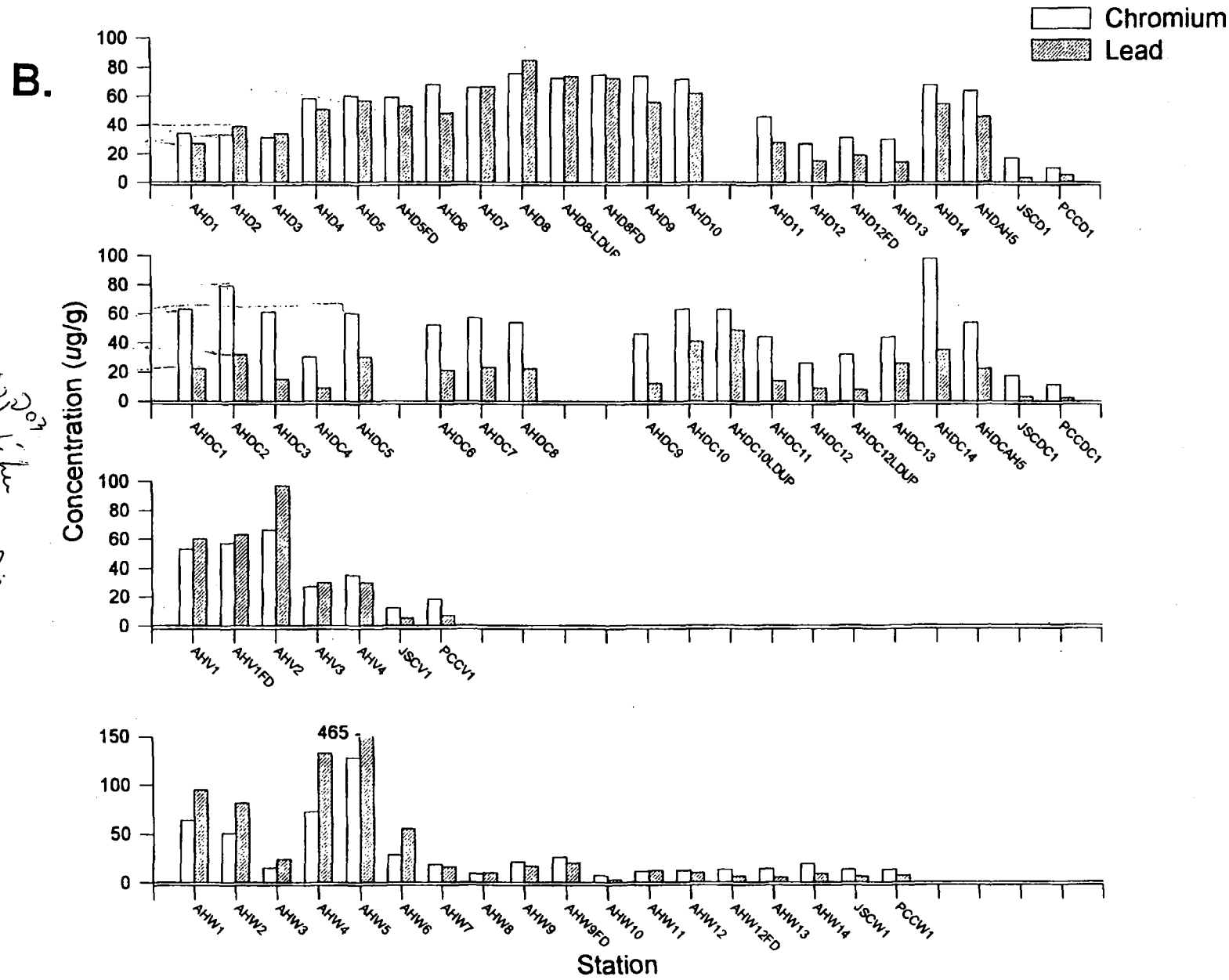
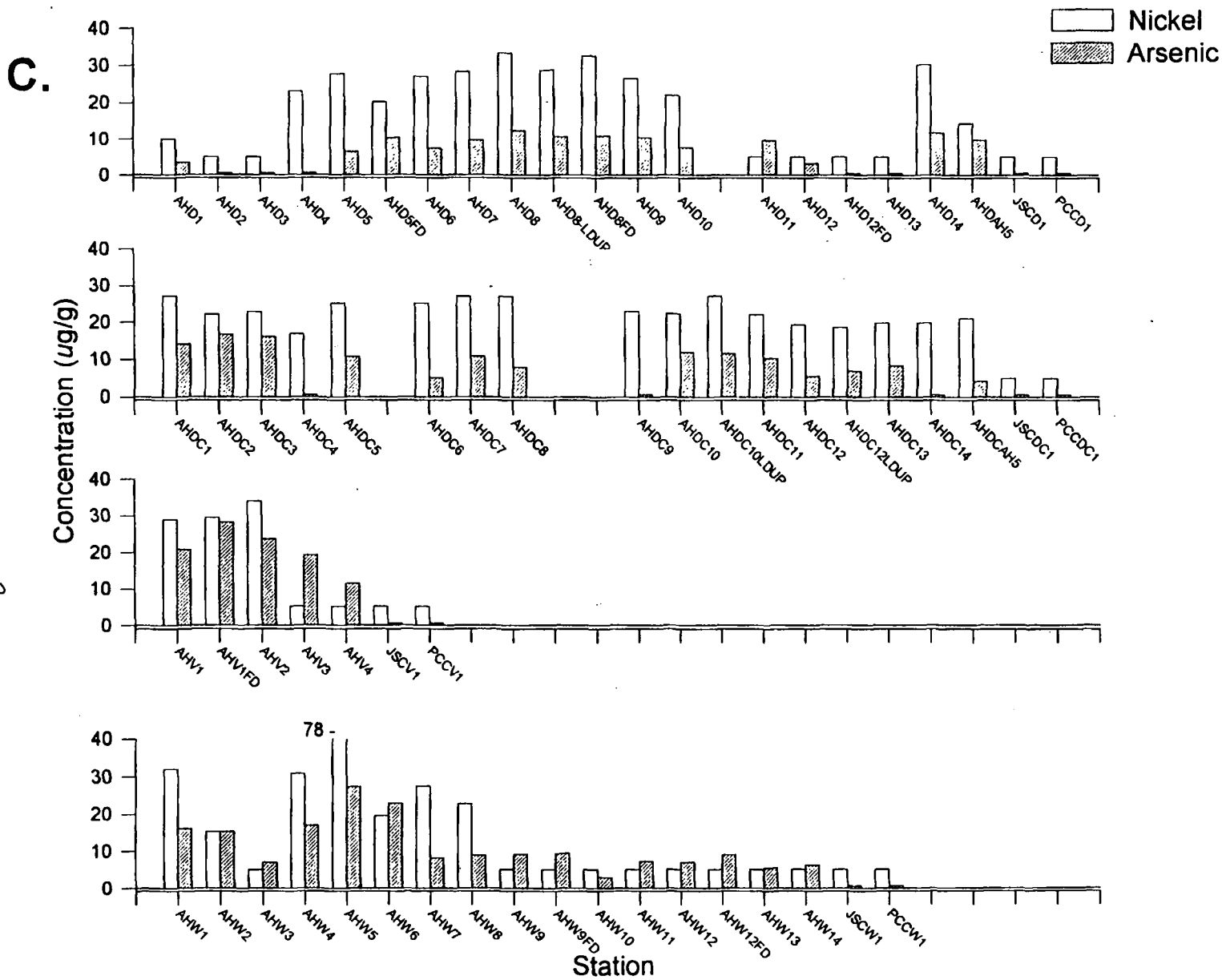


Figure 4.2-7 (cont.)



Both Ni & Arsenic higher in DC2 & DC3

Figure 4.2-7 (cont.)

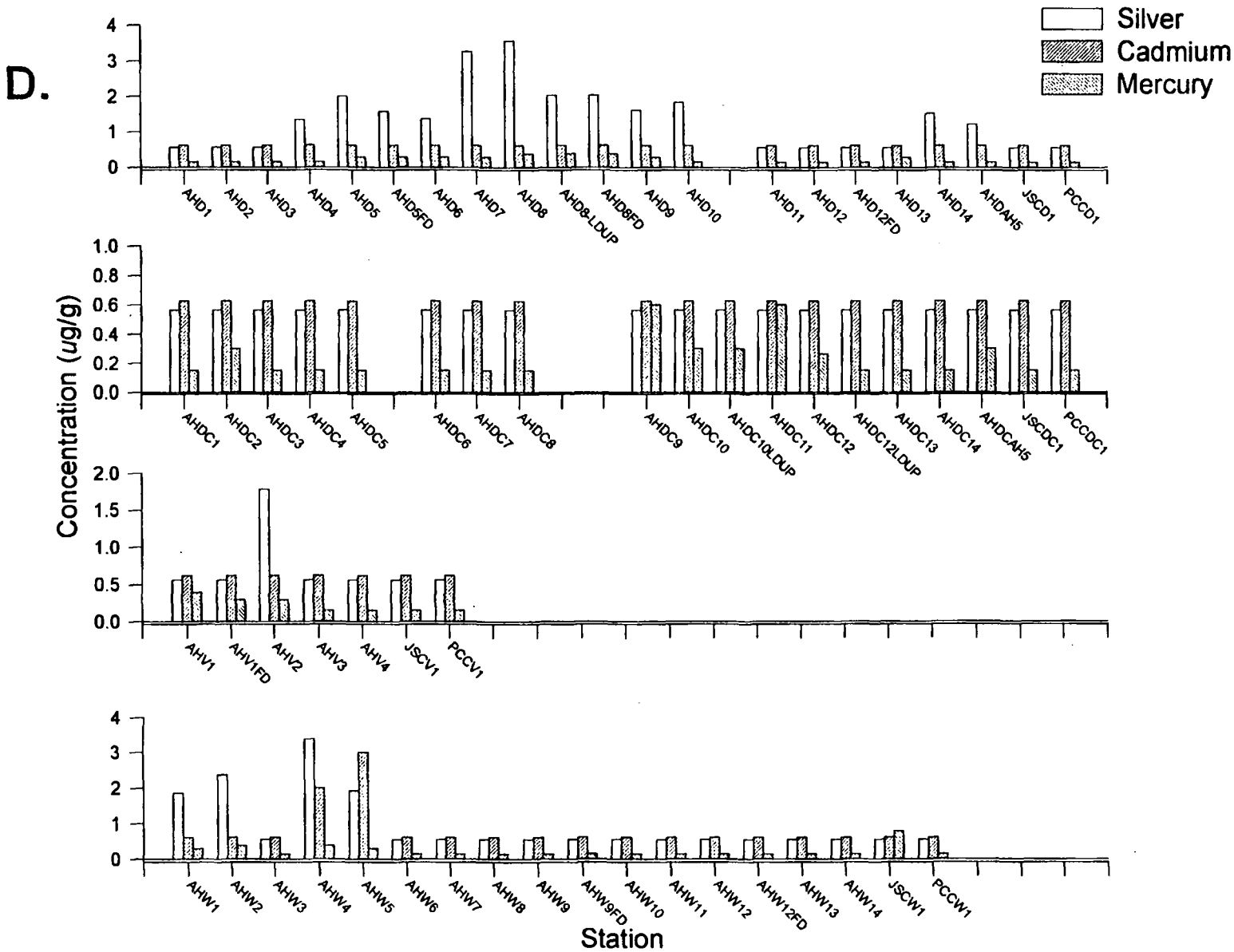


Figure 4.2-7 (cont.)

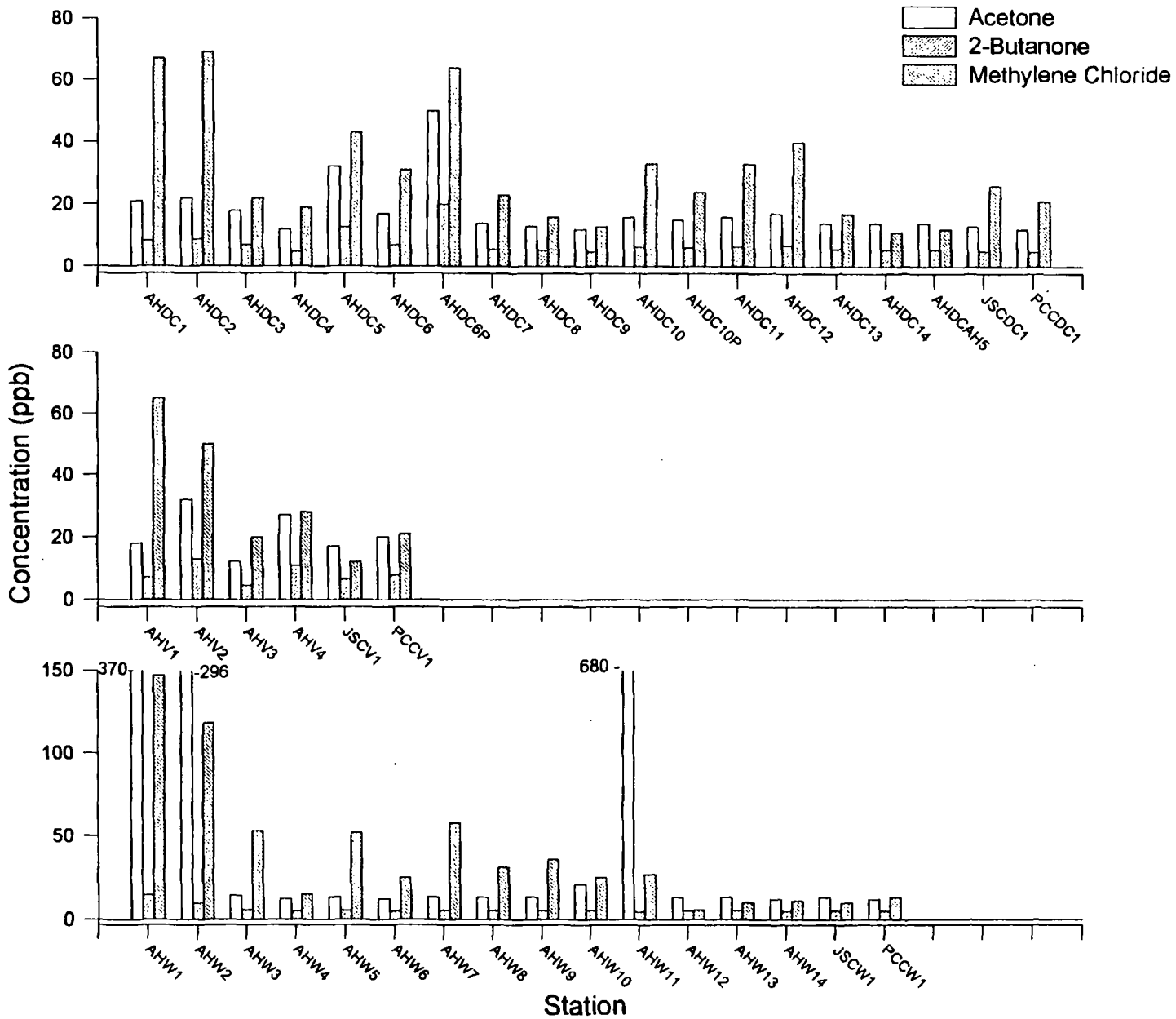


Figure 4.2-8. VOA concentrations (µg/kg) in sediments from Allen Harbor.

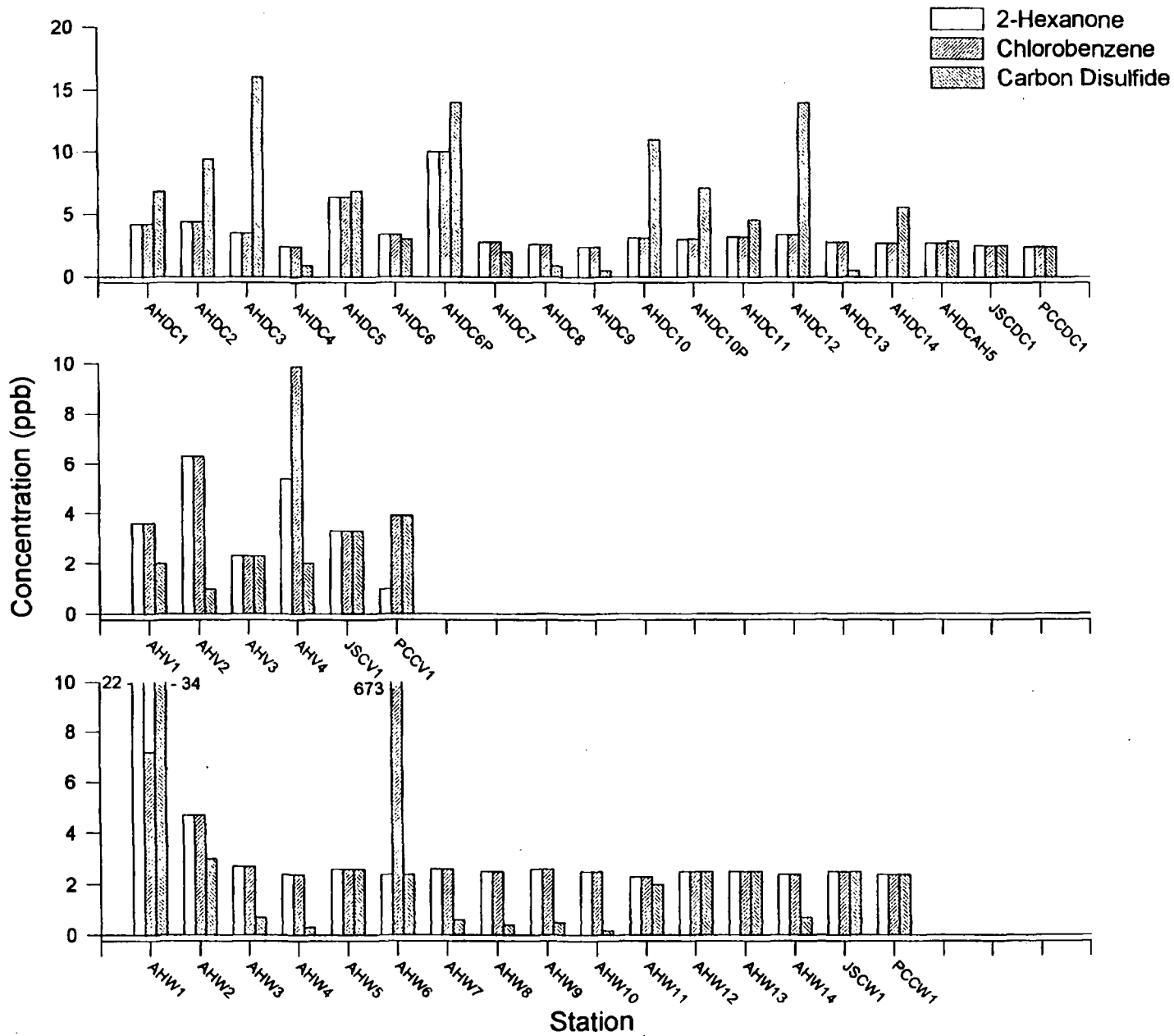


Figure 4.2-8. (con't.)

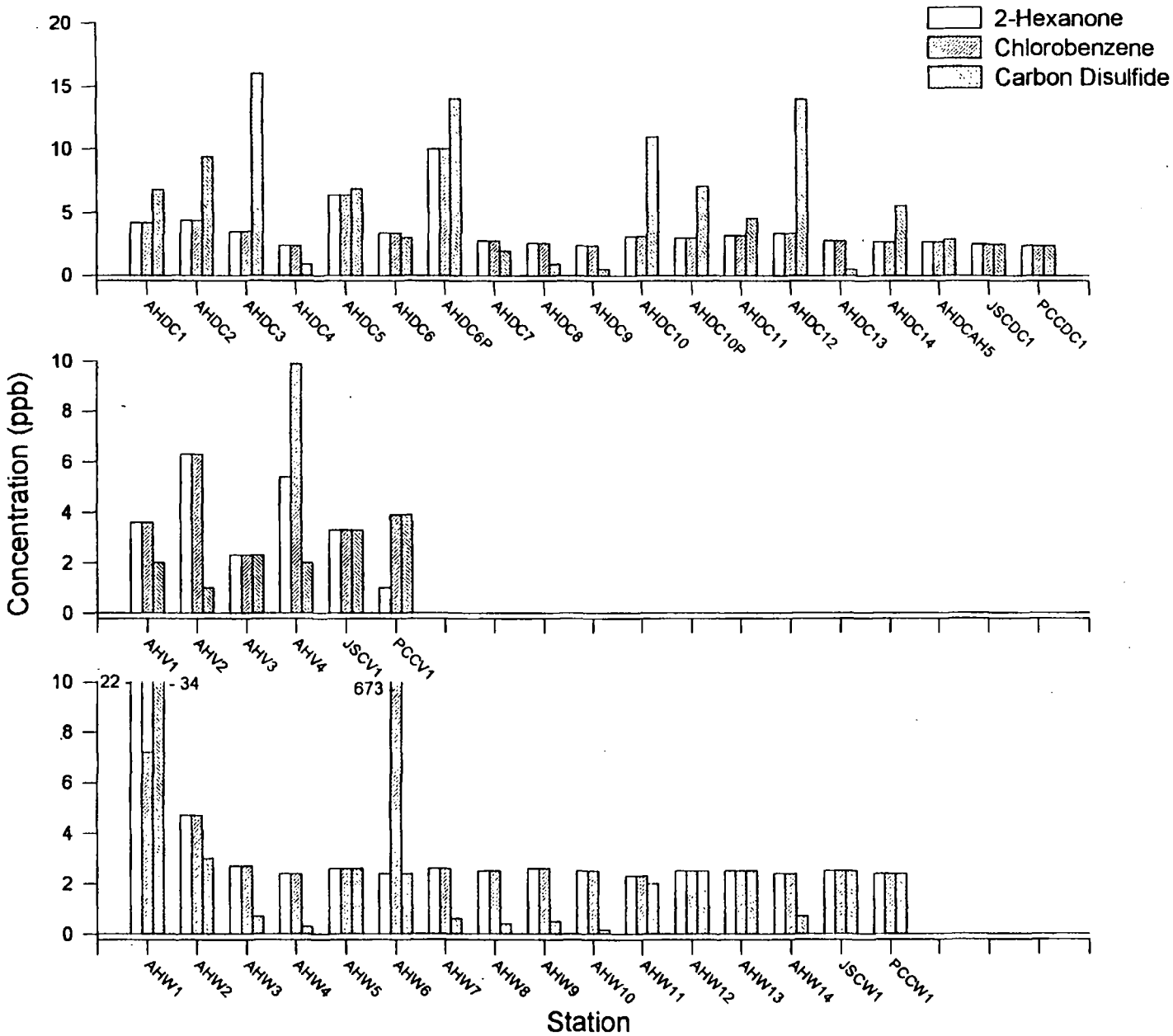


Figure 4.2-8. (con't.)

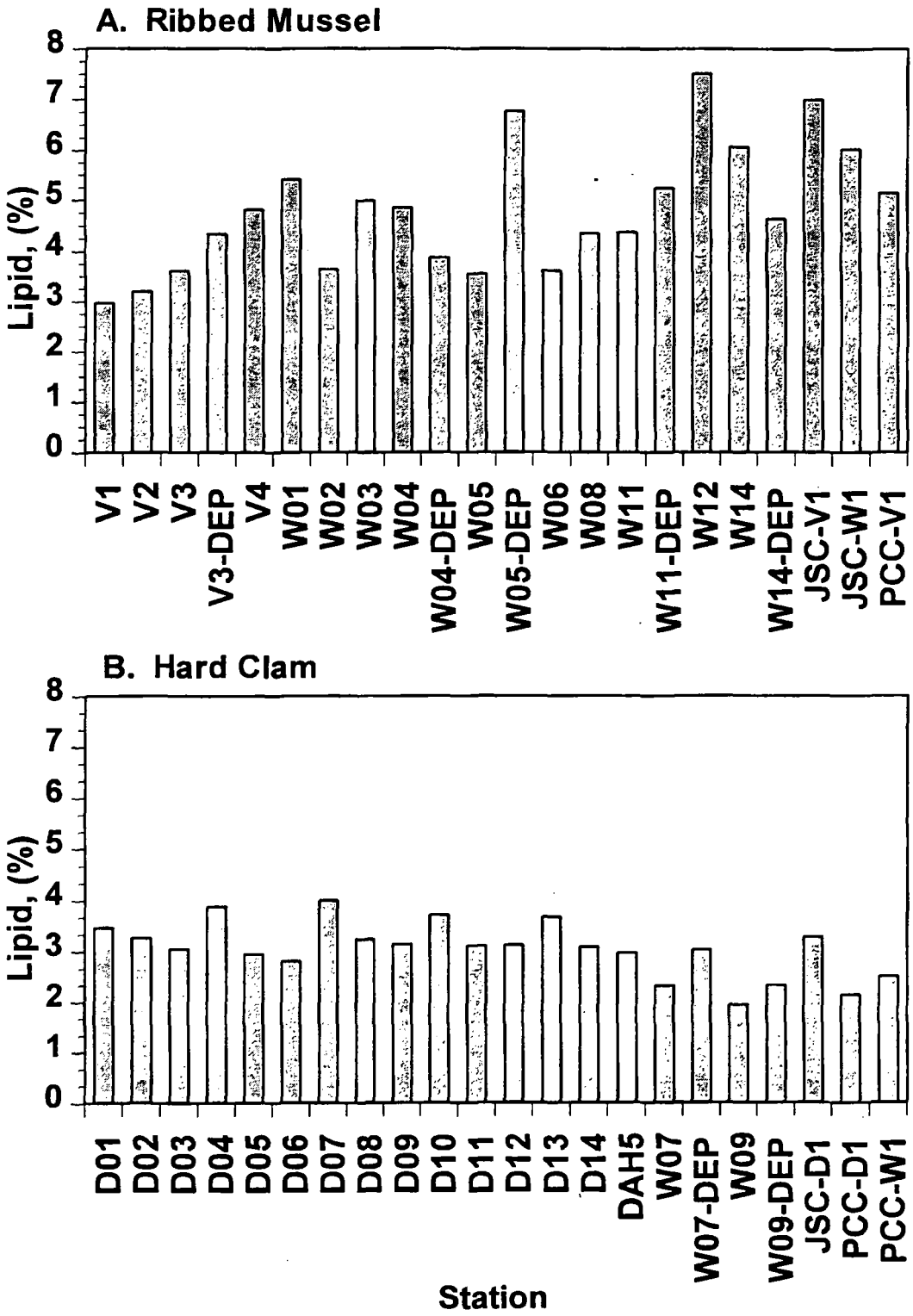
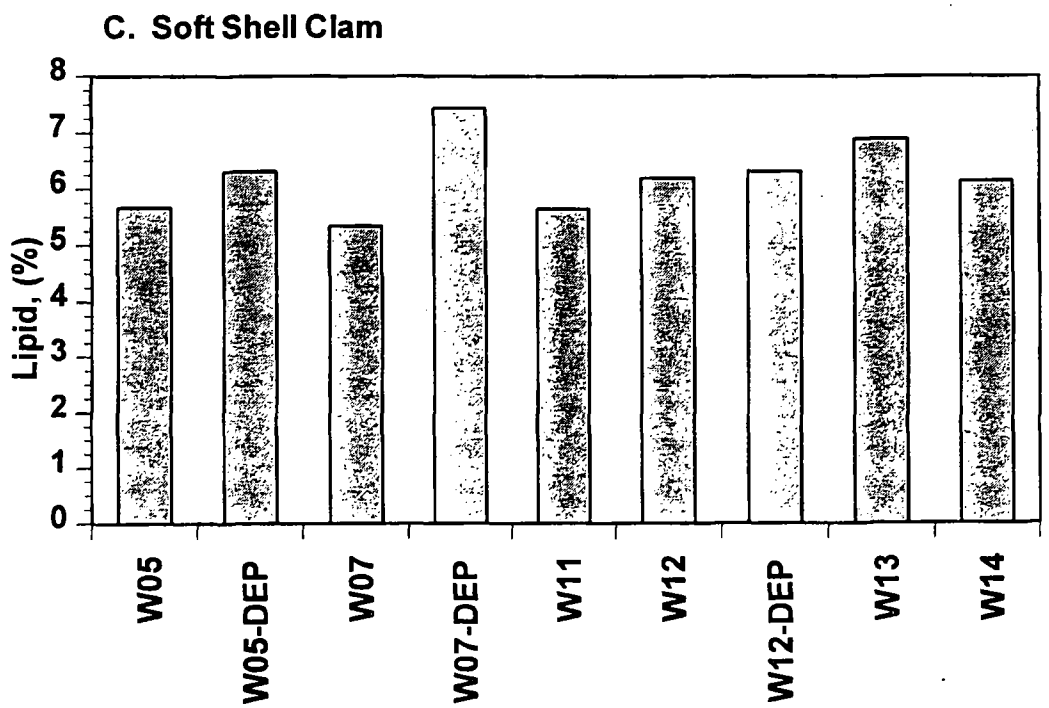


Figure 4.2-9. Lipid concentrations (% dry wt.),used for tissue normalizations in: A) ribbed mussels, B) hard clams, C) soft shell clams, D) oysters, and E) mummichog fish.



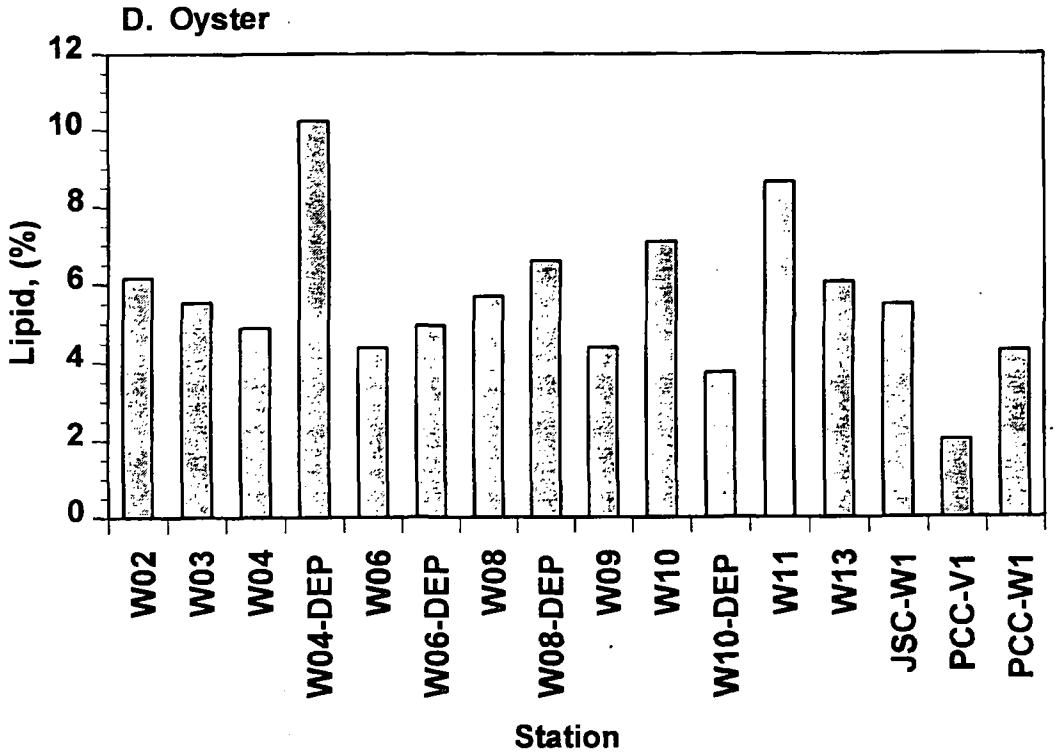


Figure 4.2-9. continued

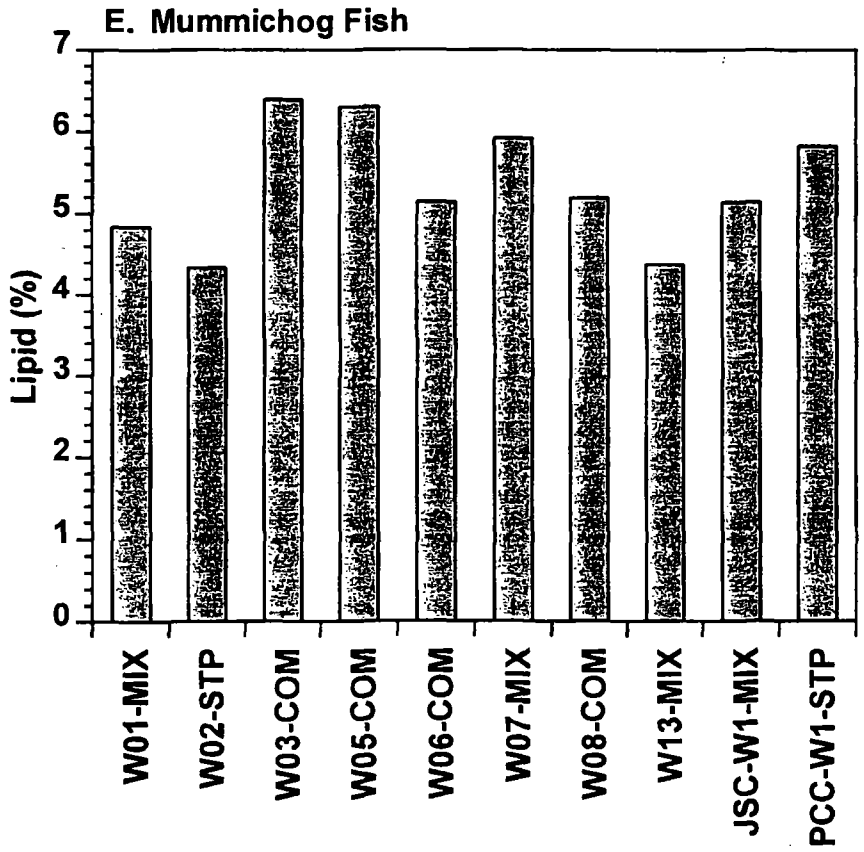


Figure 4.2-9. continued

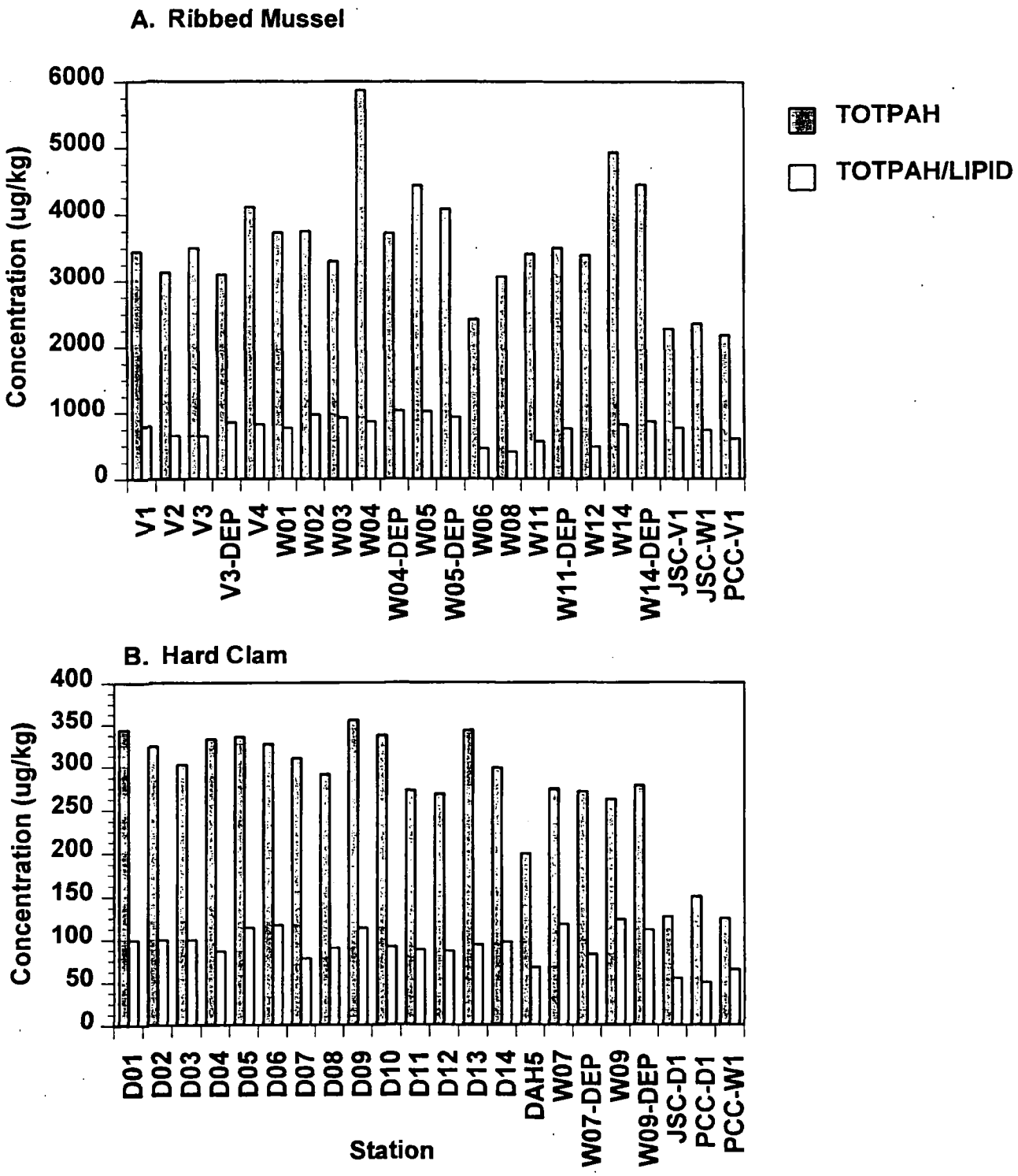
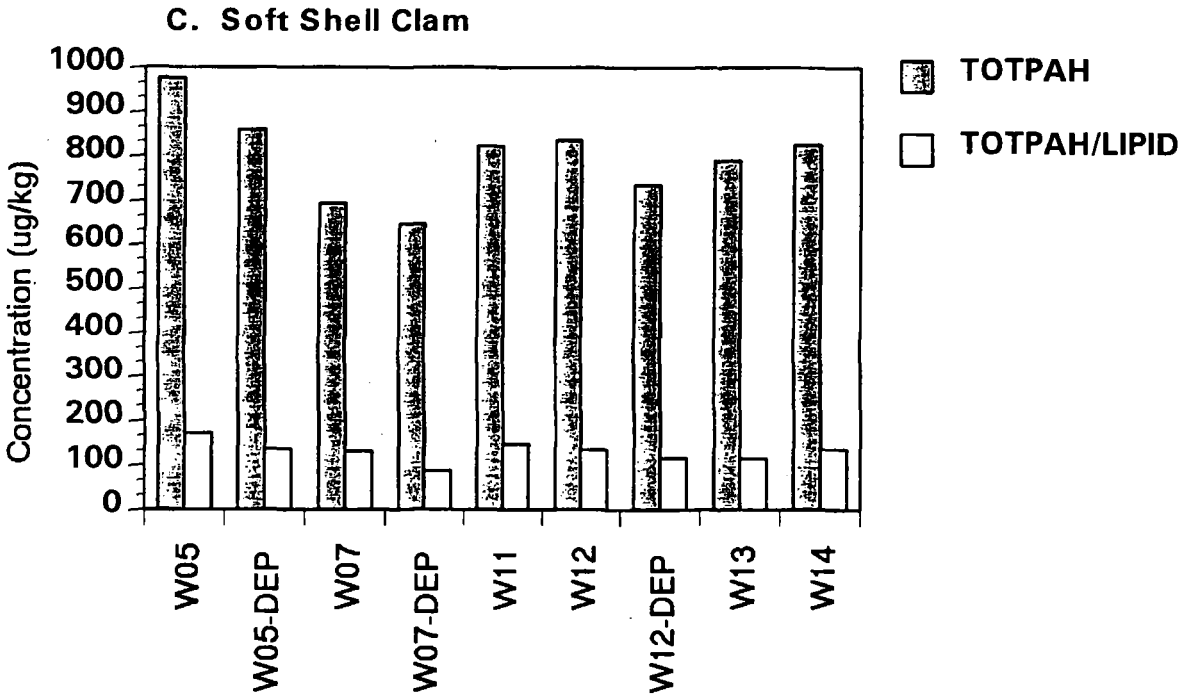


Figure 4.2-10. Total PAH concentrations (μg/kg dry wt.), normalized and non-normalized to lipids in: A) soft shell clams and oysters, B) ribbed mussels and hard shell clams, and C) mummichog fish.



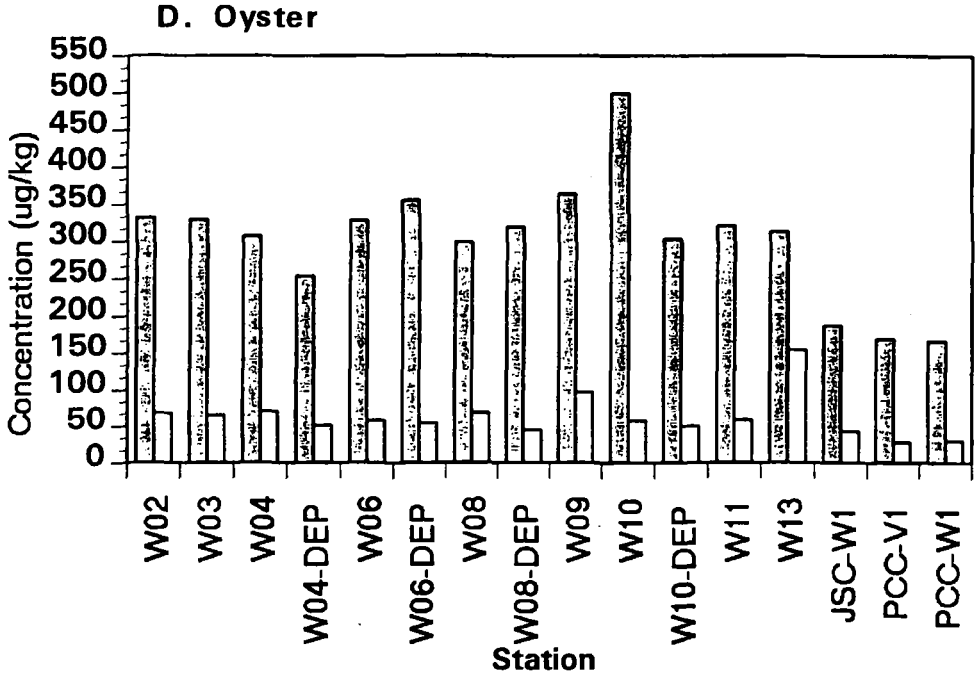


Figure 4.2-10. continued

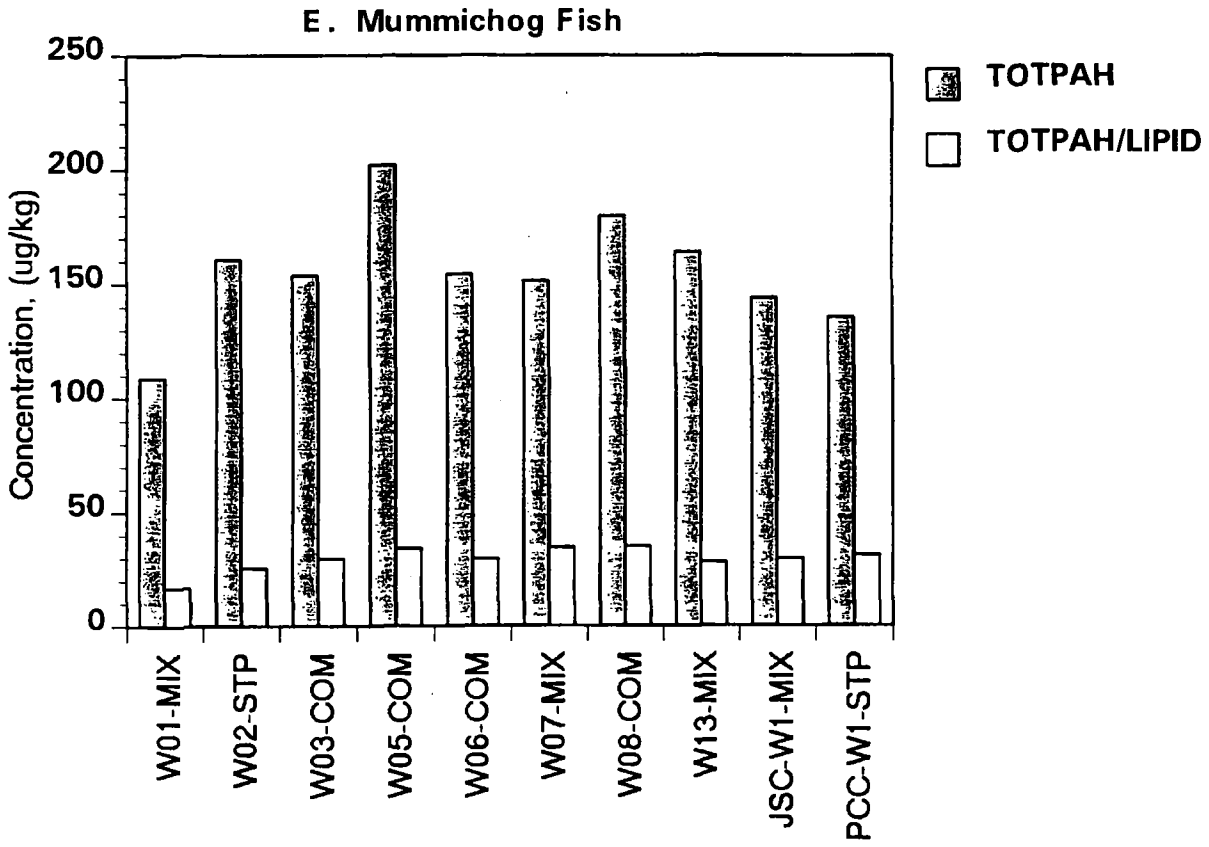


Figure 4.2-10. continued

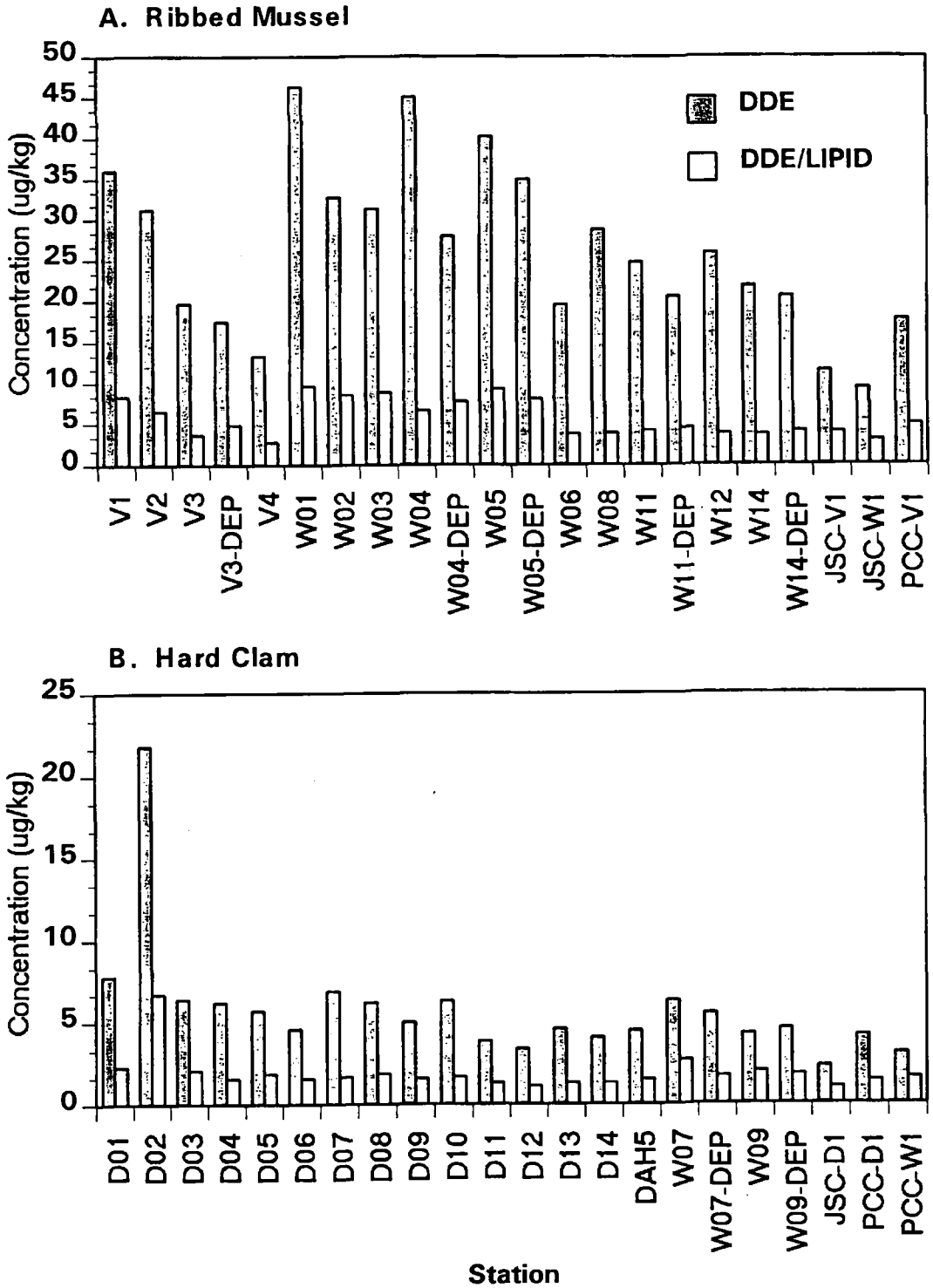
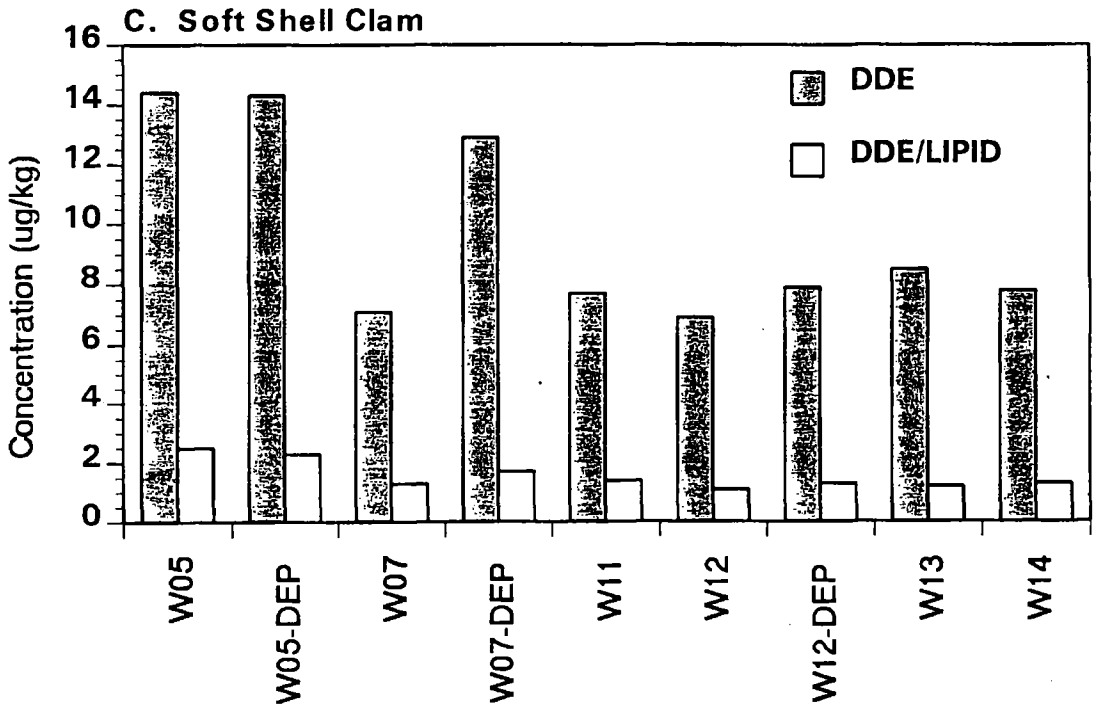


Figure 4.2-11. DDE concentrations (µg/kg dry wt.), normalized and non-normalized to lipids in: A) ribbed mussels, B) hard clams, C) soft shell clams, D) oysters, and E) mummichog fish.



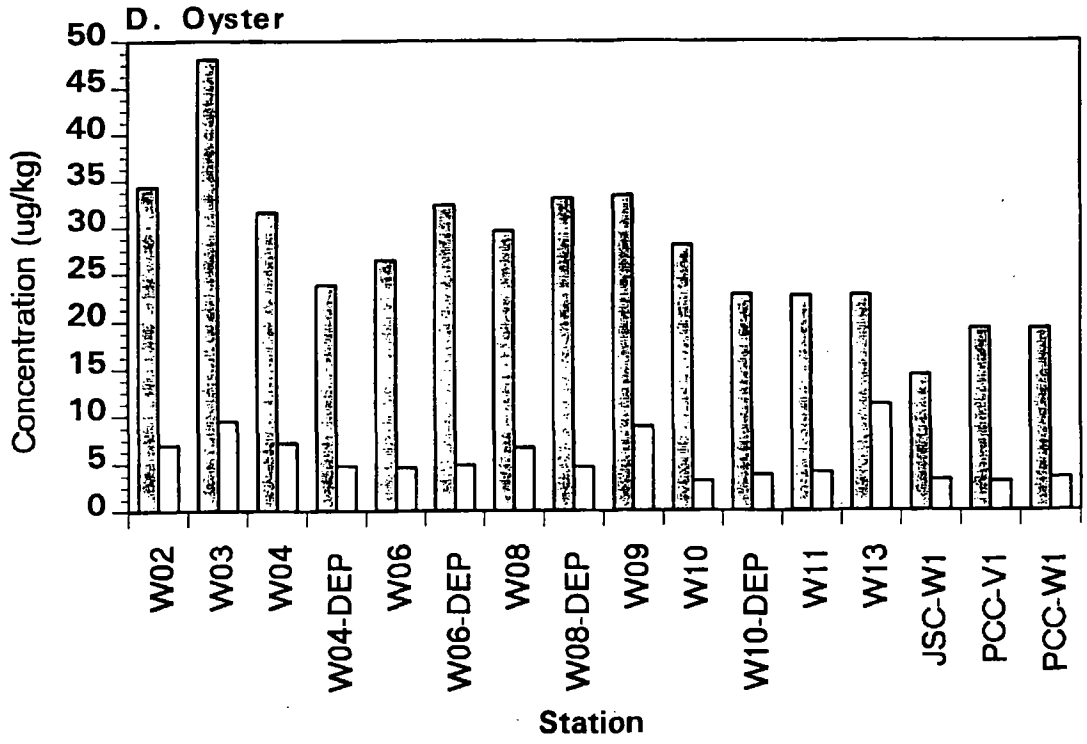


Figure 4.2-11. continued

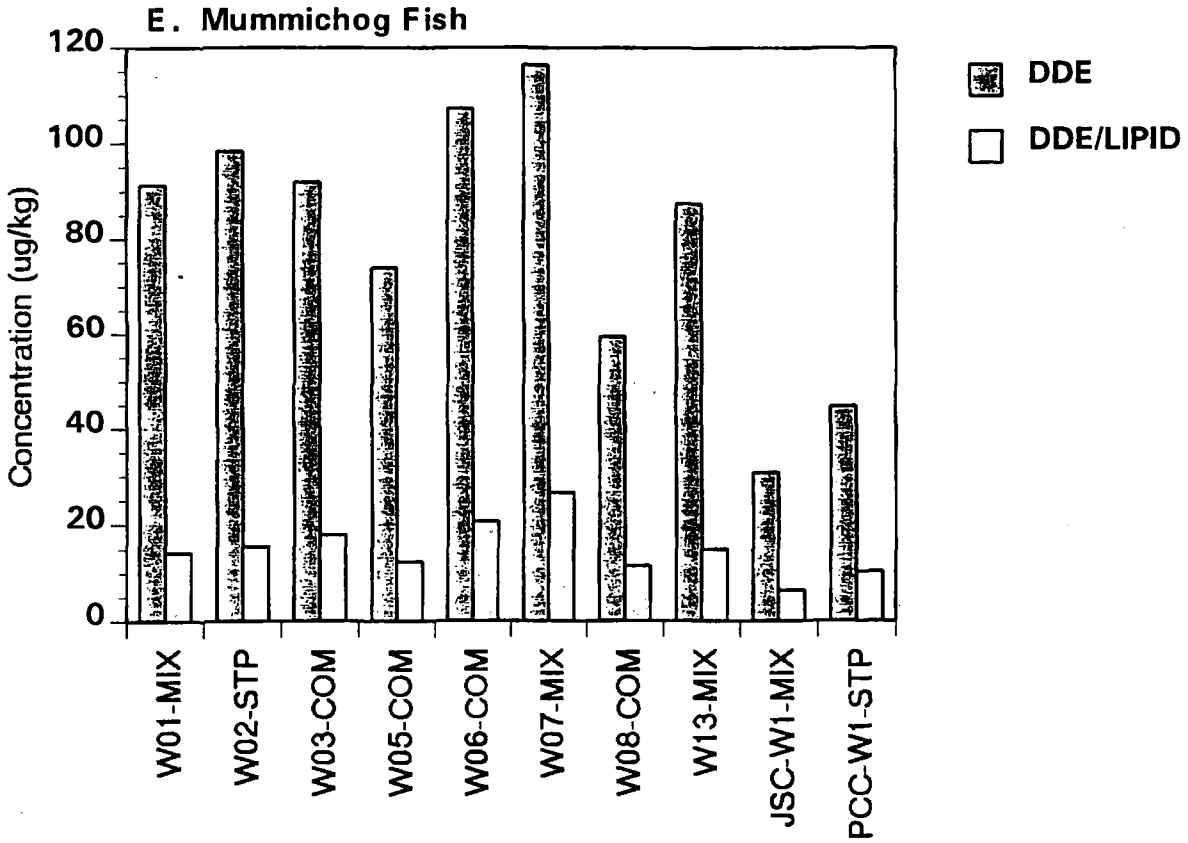


Figure 4.2-11. continued

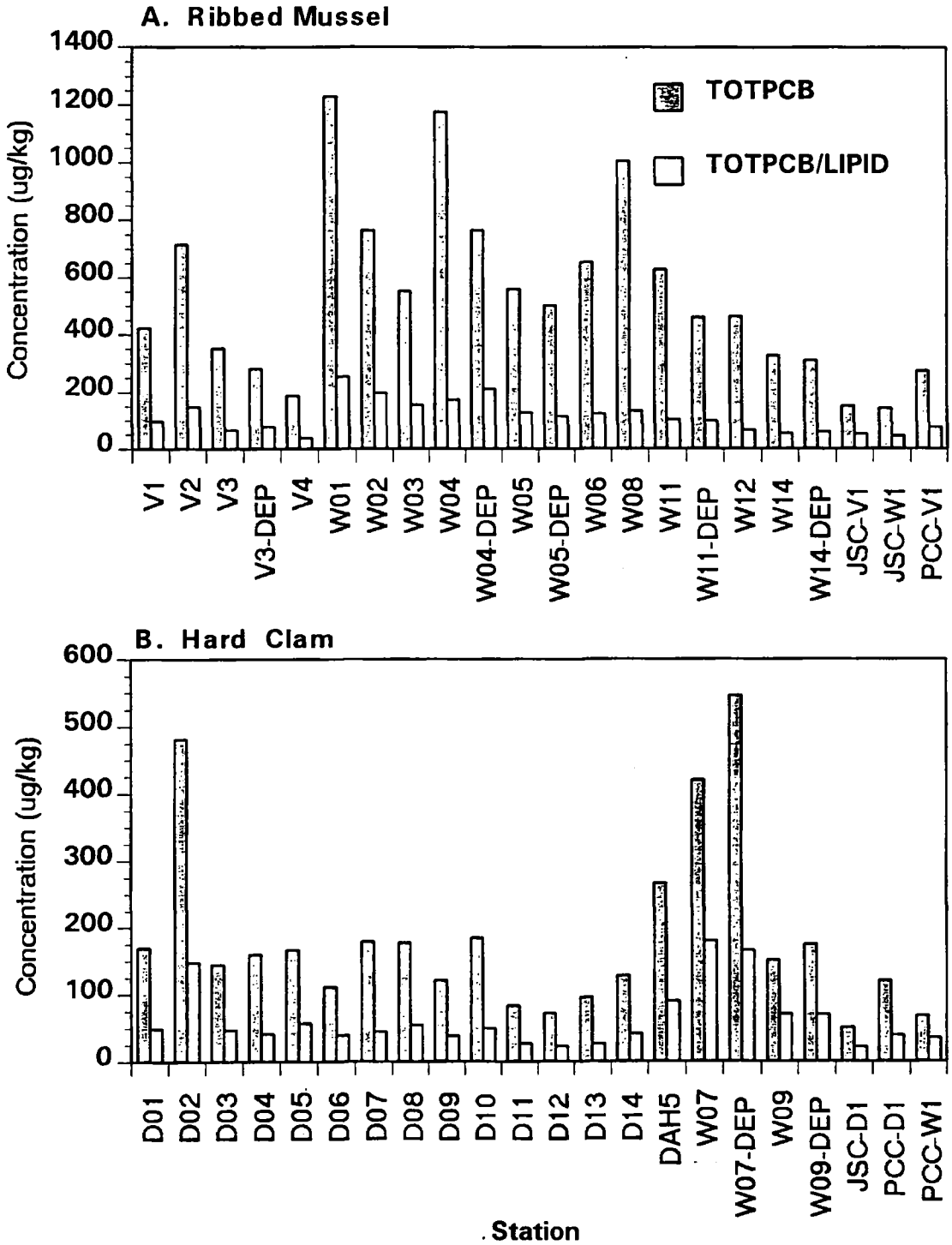
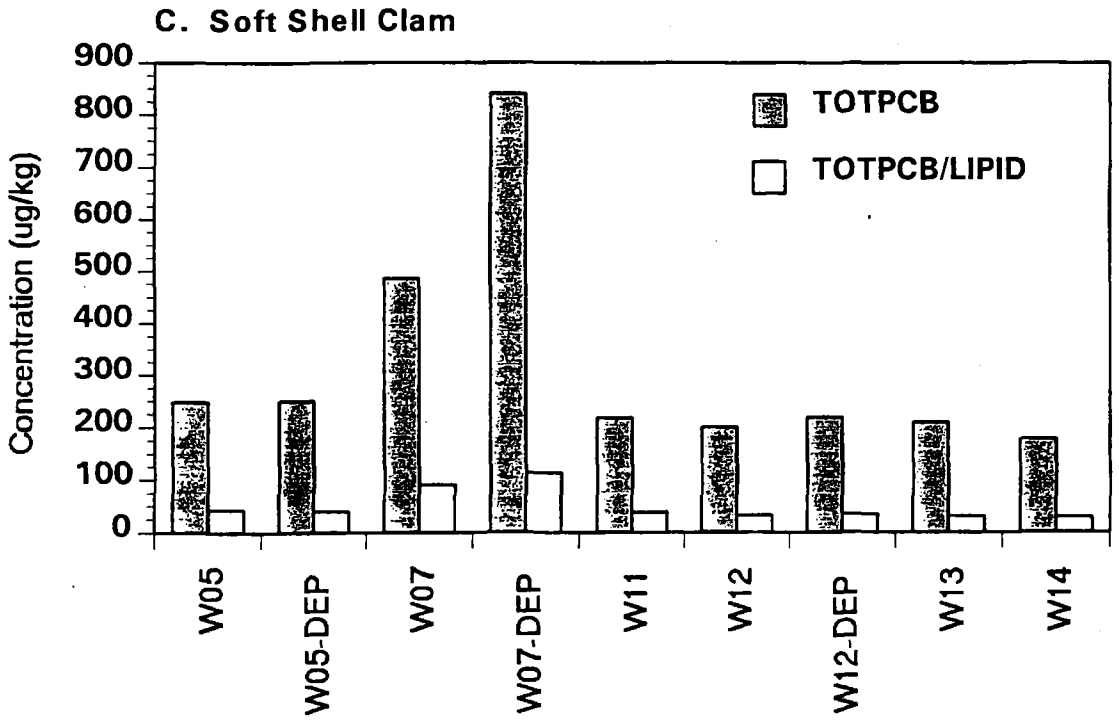


Figure 4.2-12. Total PCB concentrations (μg/kg dry wt.), normalized and non-normalized to lipids in: A) ribbed mussels, B) hard clams, C) soft shell clams, D) oysters, and E) mummichog fish.



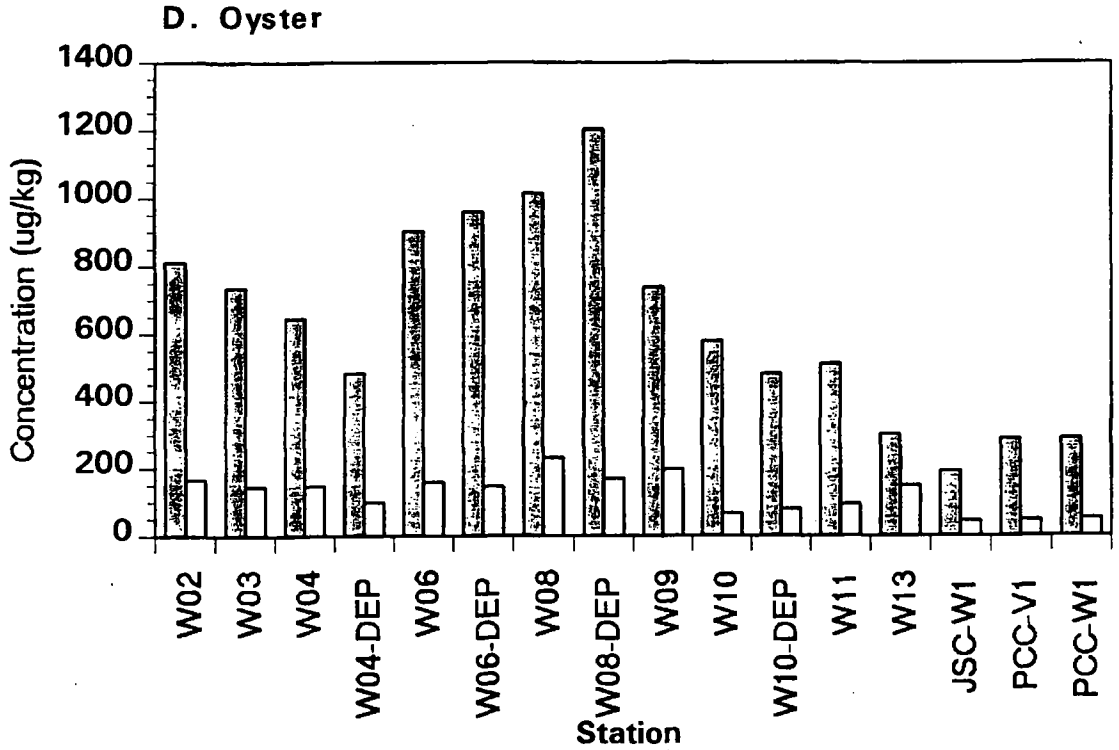


Figure 4.2-12. continued

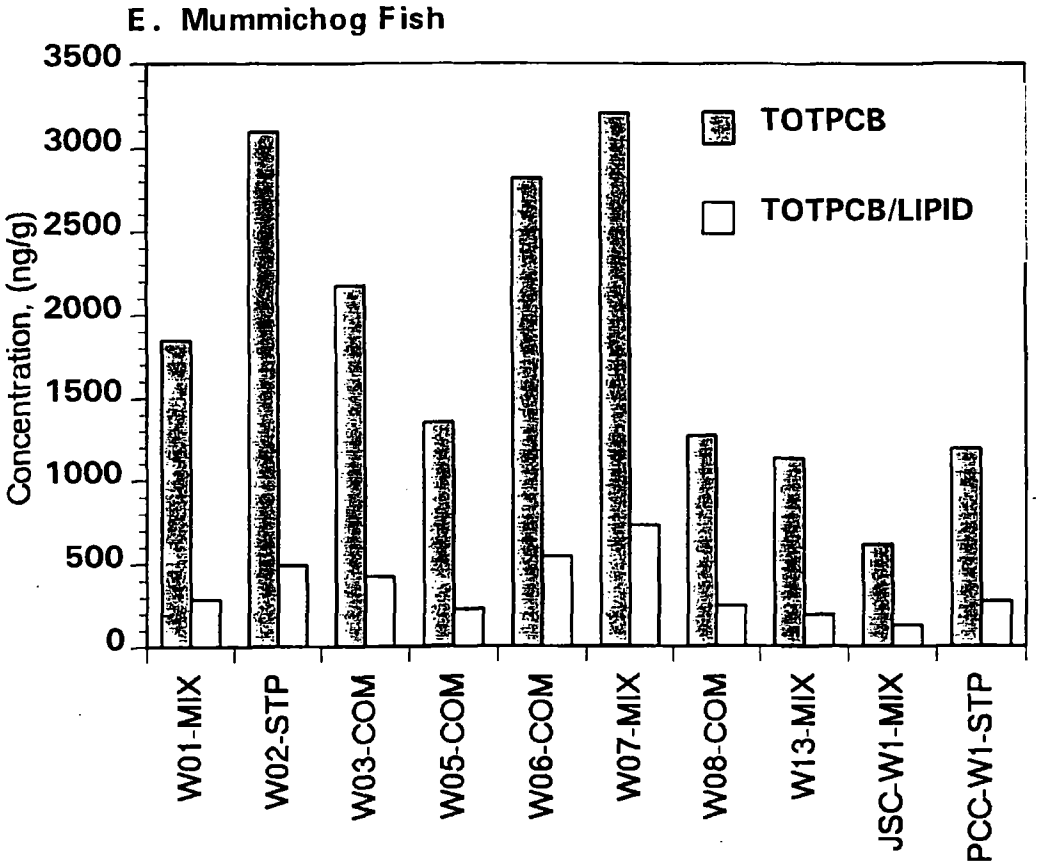


Figure 4.2-12. continued

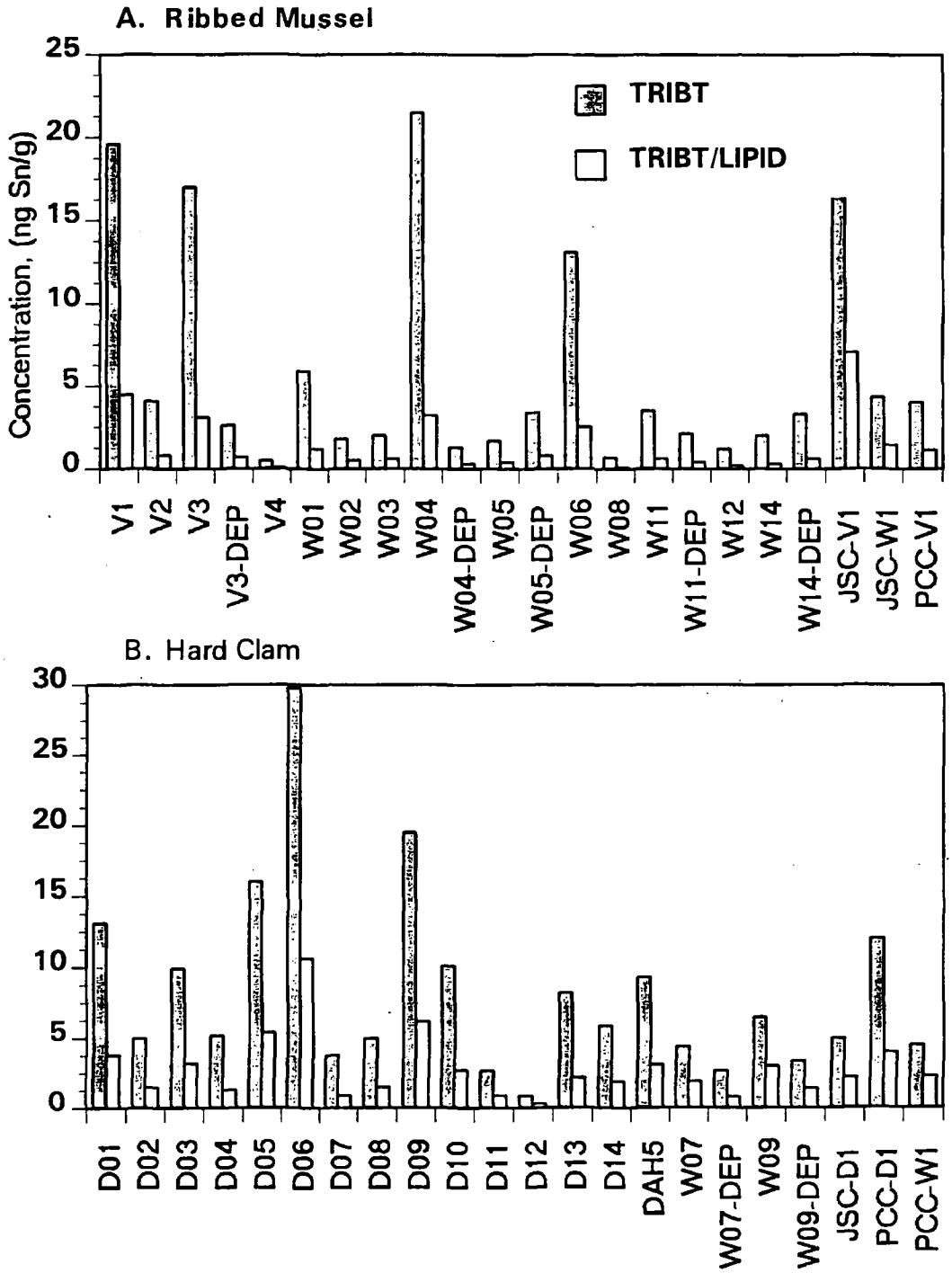
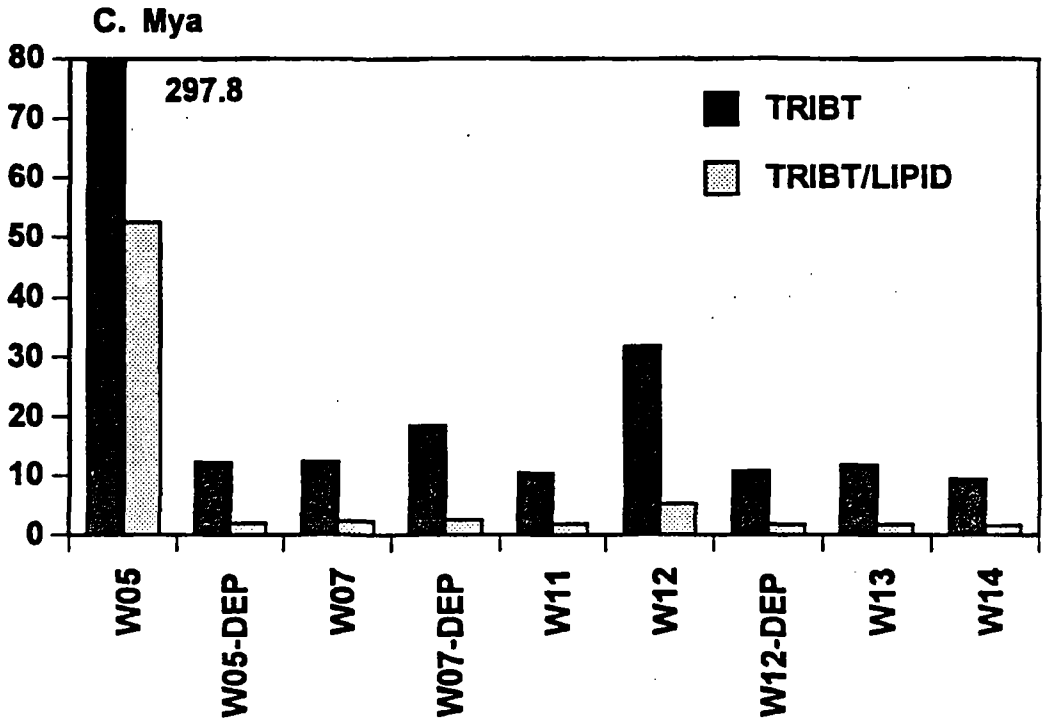


Figure 4.2-13. Butyltin concentrations (μg Sn/kg dry wt.) in: A) ribbed mussels, B) hard clams, C) soft shell clams, D) oysters, and E) mummichog fish



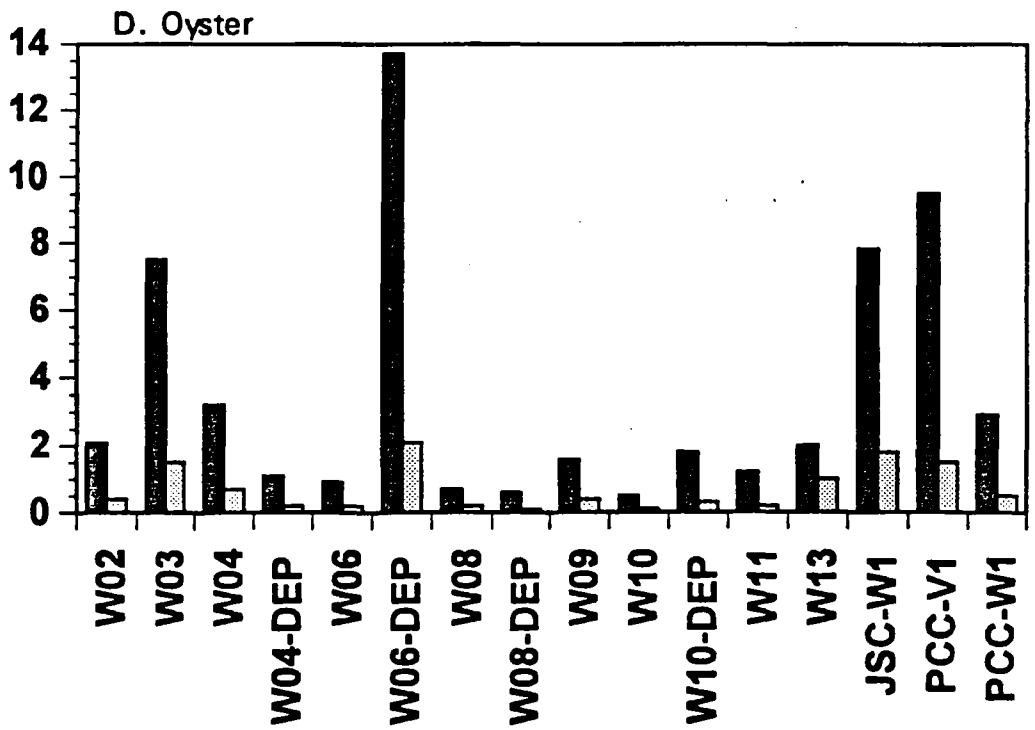


Figure 4.2-13. continued

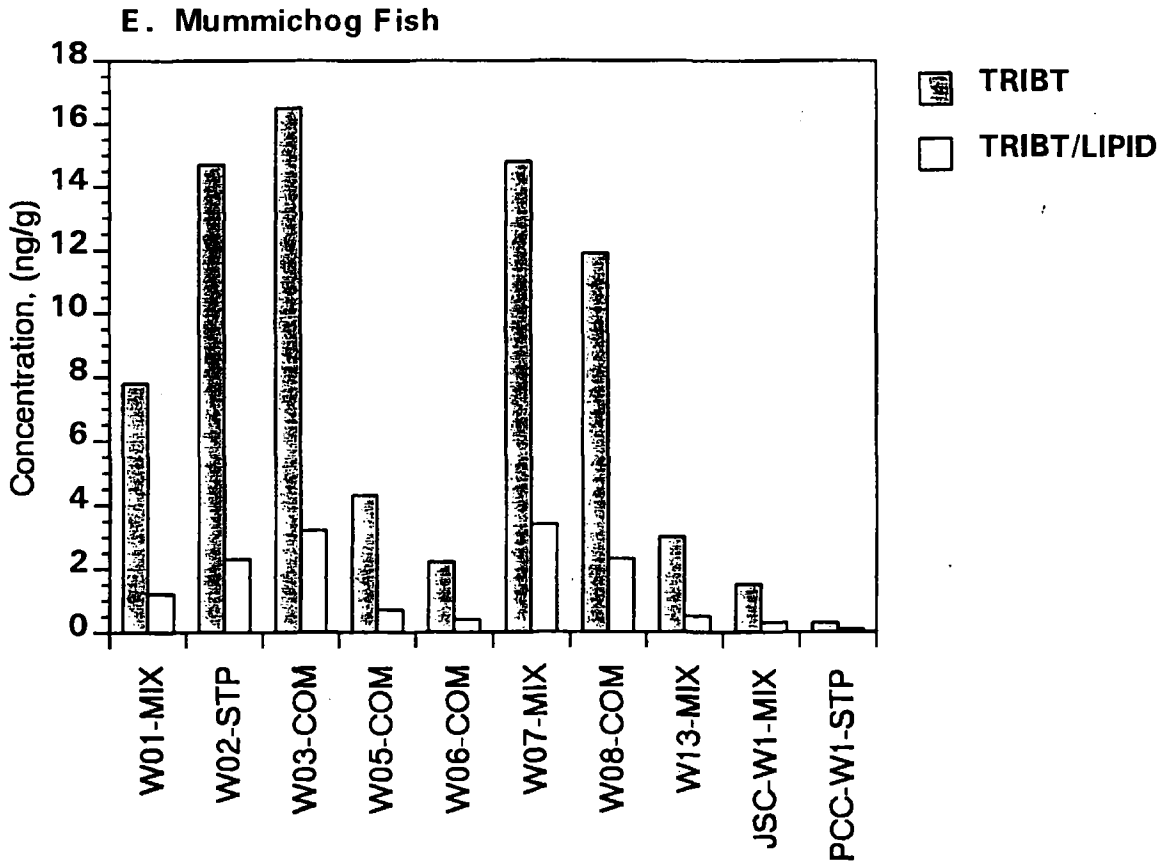


Figure 4.2-13. continued

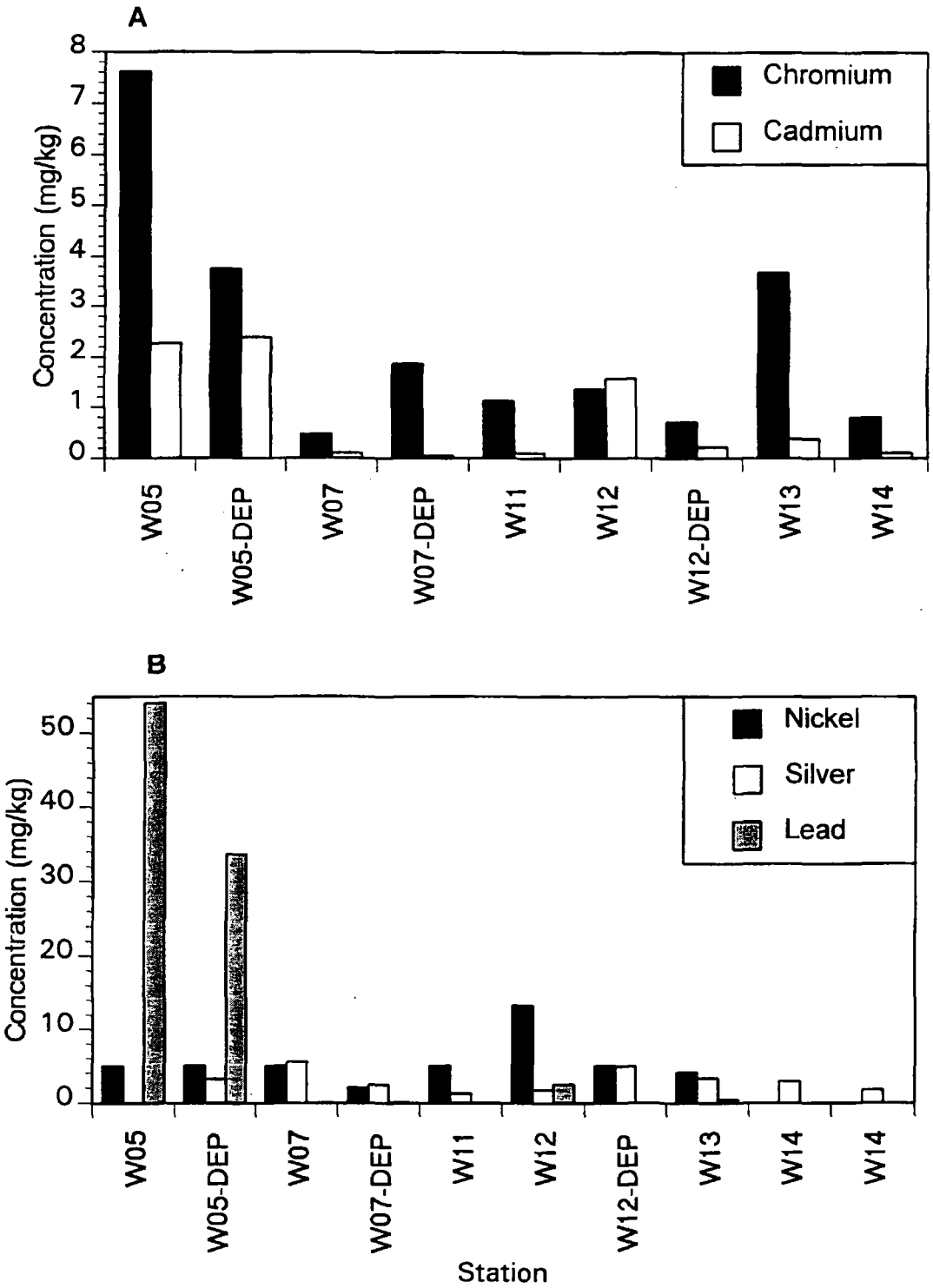
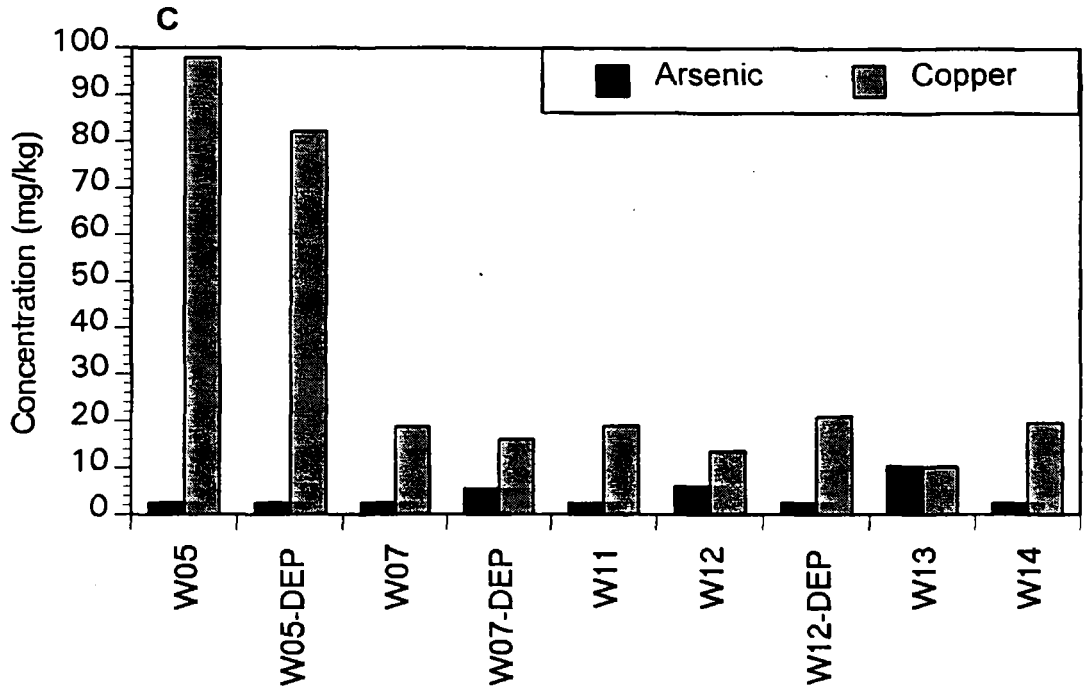


Figure 4.2-14. Concentrations of metals (mg/kg dry wt.) in soft shell clams from Allen Harbor: A) Chromium and Cadmium, B) Nickel, Silver and Lead, and C) Arsenic and Copper, and D) Zinc.



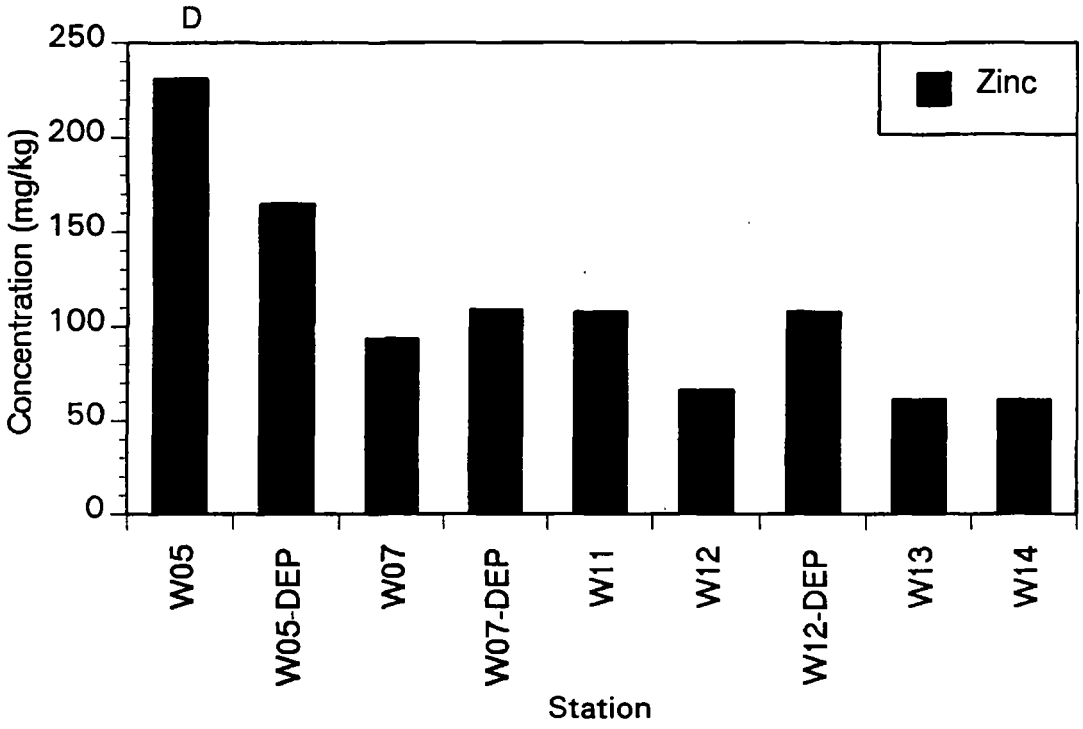


Figure 4.2-14. continued

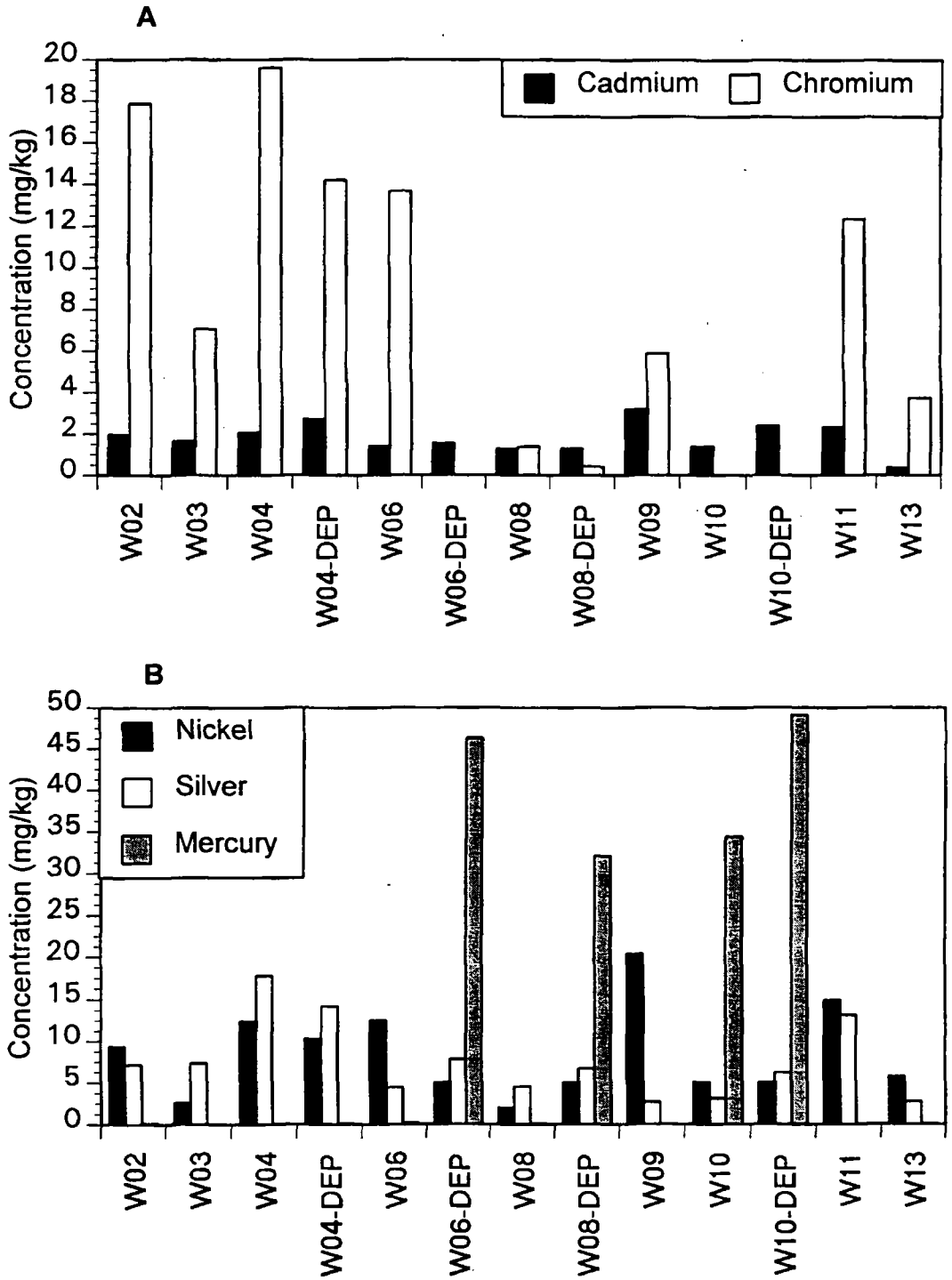
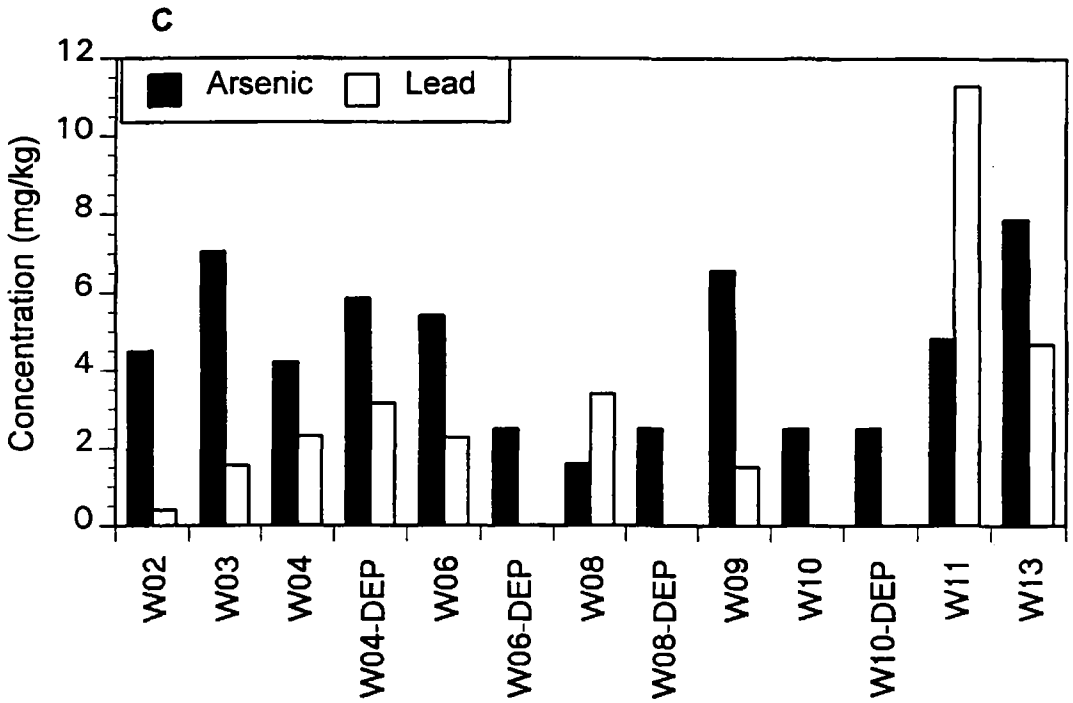


Figure 4.2-15. Concentrations of metals (mg/kg dry wt.) in oysters from Allen Harbor: A) Cadmium and Chromium, B) Nickel, Silver and Mercury, C) Arsenic and Lead, and D) Copper and Zinc.



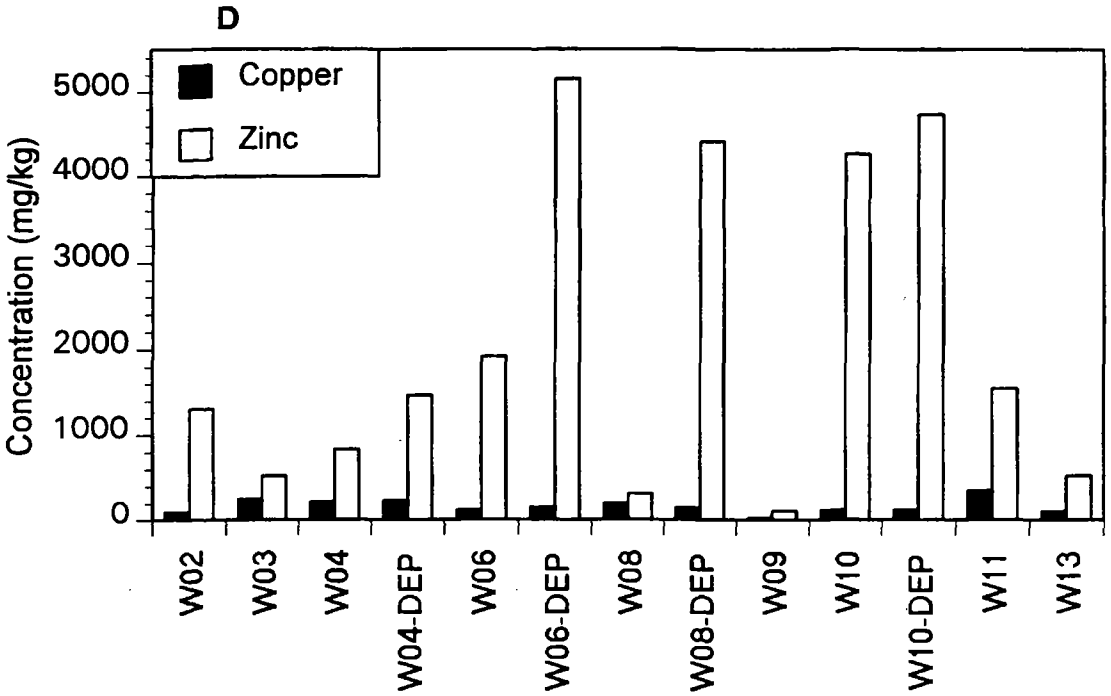


Figure 4.2-15. continued

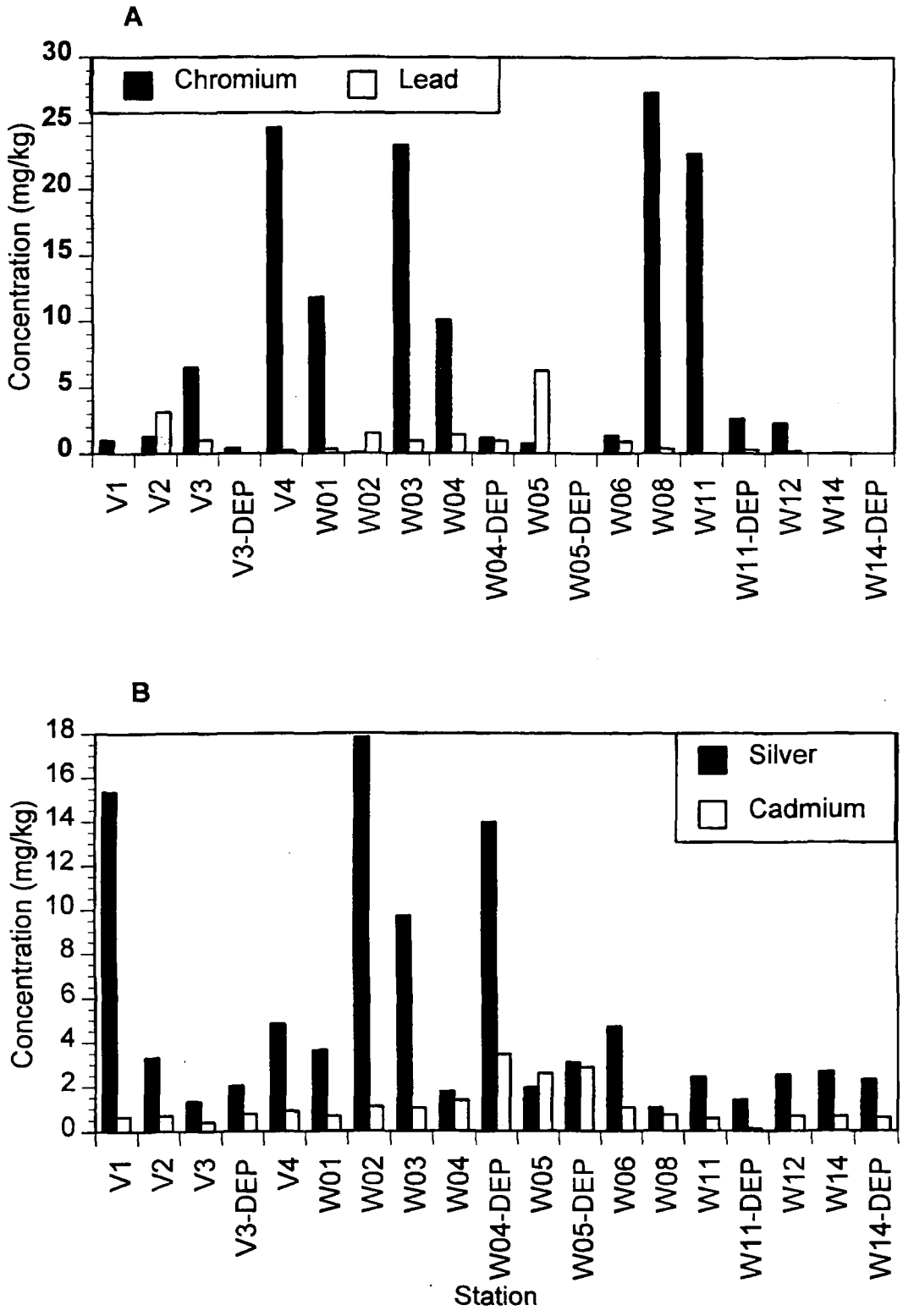


Figure 4.2-16. Concentrations of metals (mg/kg dry wt.) in ribbed mussels from Allen Harbor: A) Chromium and Lead, B) Silver and Cadmium, C) Arsenic, Copper and Mercury, and D) Nickel and Zinc.

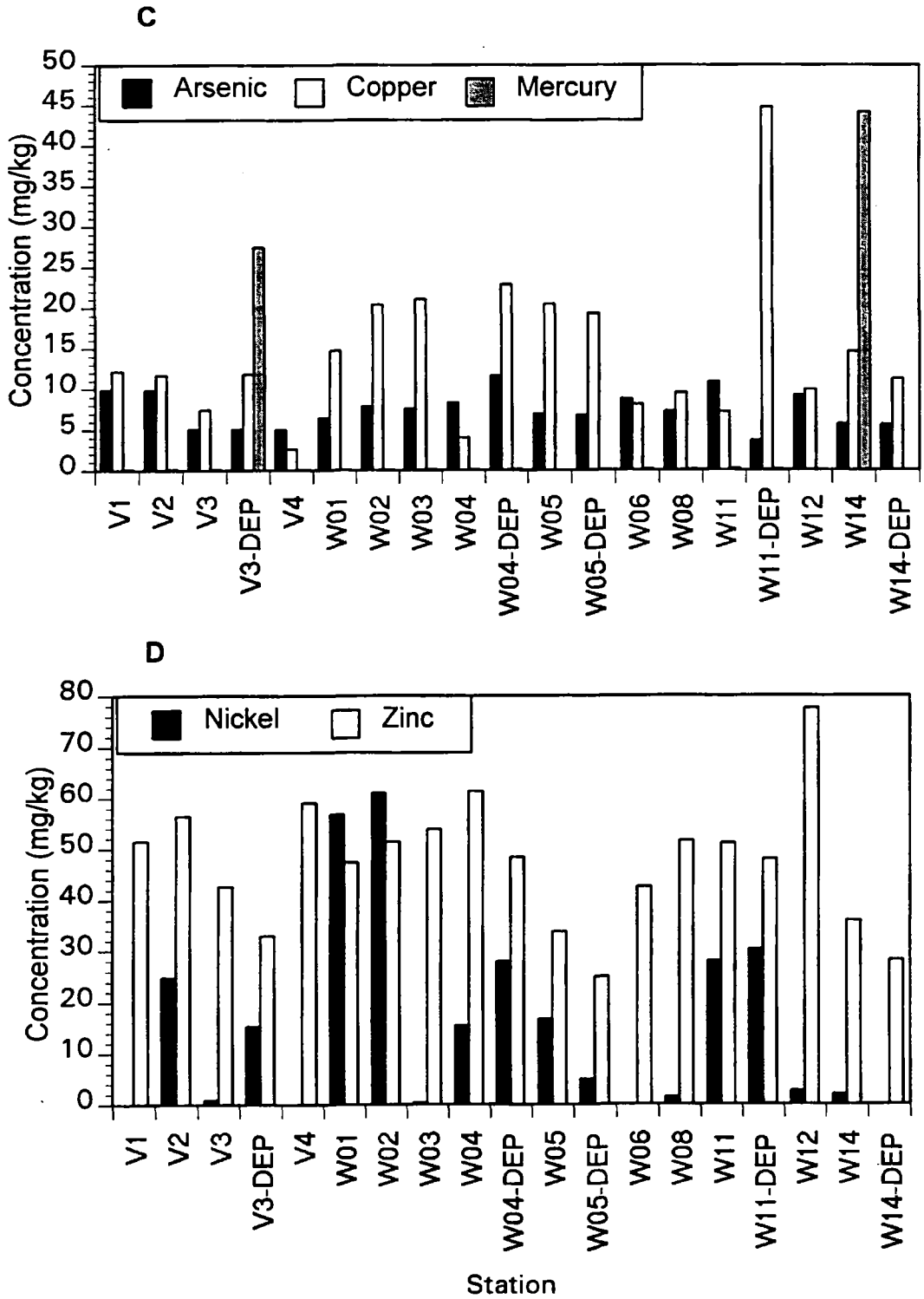


Figure 4.2-16. continued

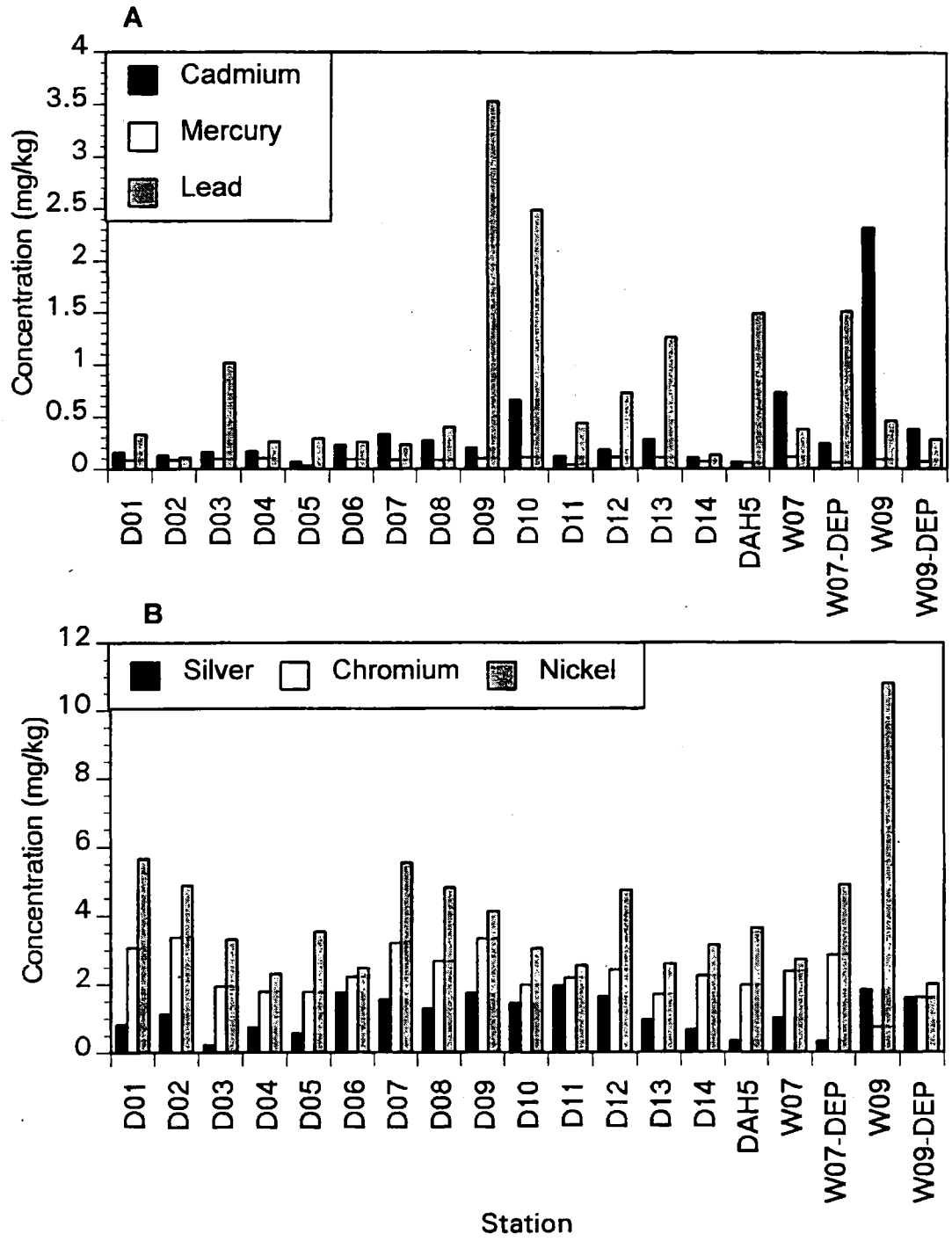


Figure 4.2-17. Concentrations of metals (mg/kg dry wt.) in hard clams from Allen Harbor: A) Cadmium, Mercury and Lead, B) Silver, Chromium and Nickel, C) Arsenic and Copper, and D) Zinc.

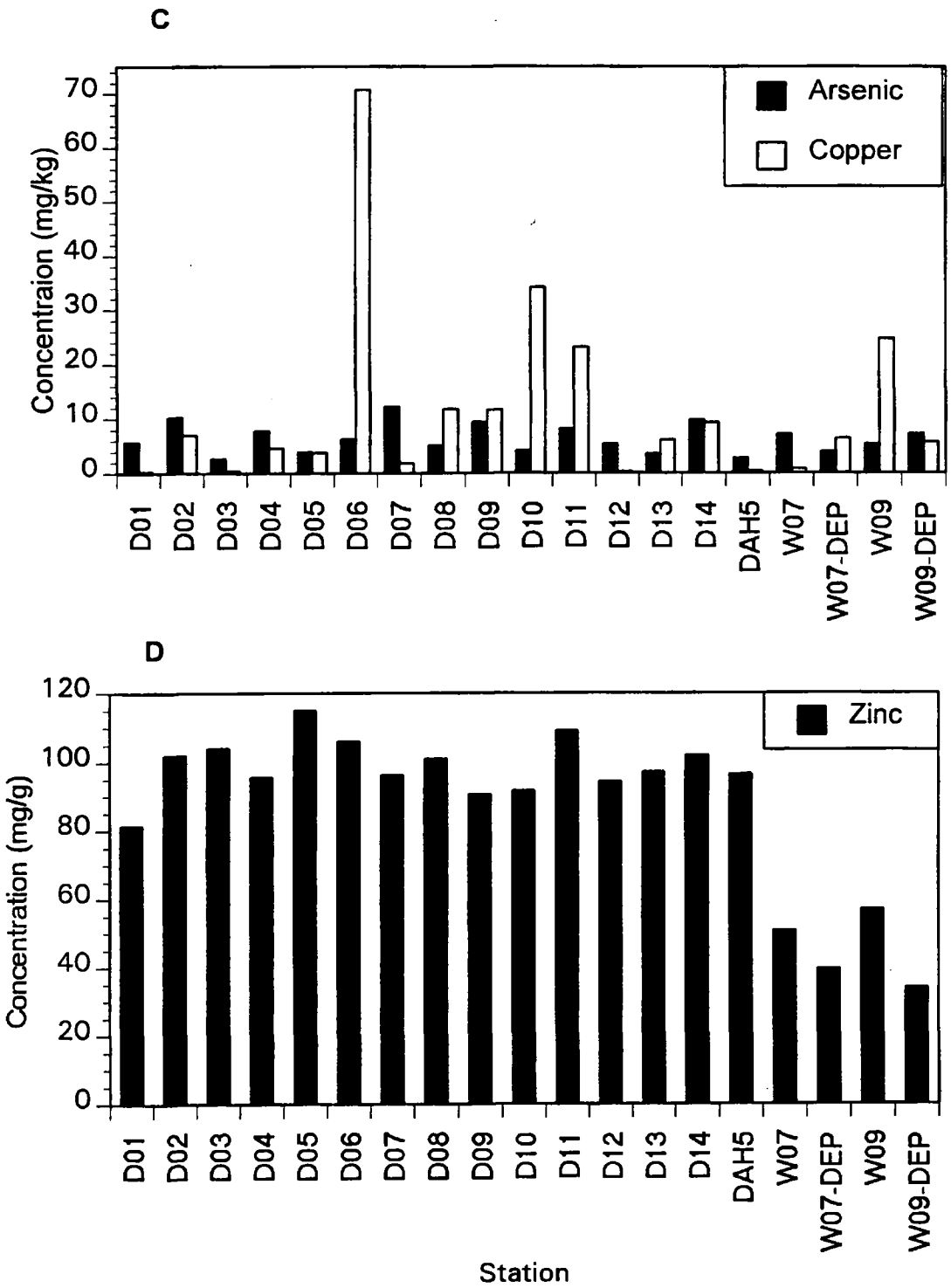


Figure 4.2-17. continued

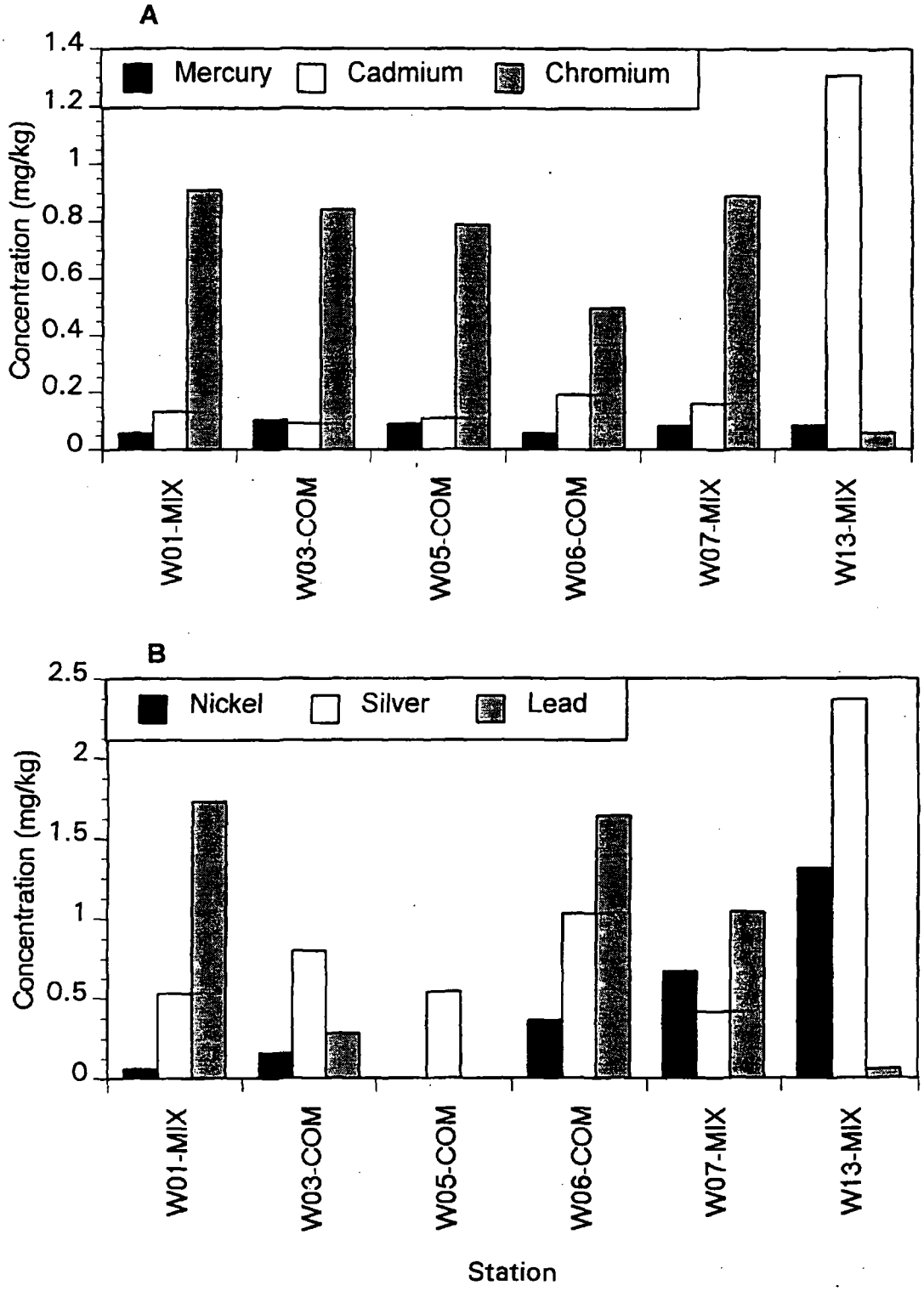


Figure 4.2-18. Concentrations of metals (mg/kg dry wt.) in mummichog fish from Allen Harbor: A) Mercury, Cadmium, and Chromium, B) Nickel, Silver and Lead, C) Copper and Arsenic, and D) Zinc.

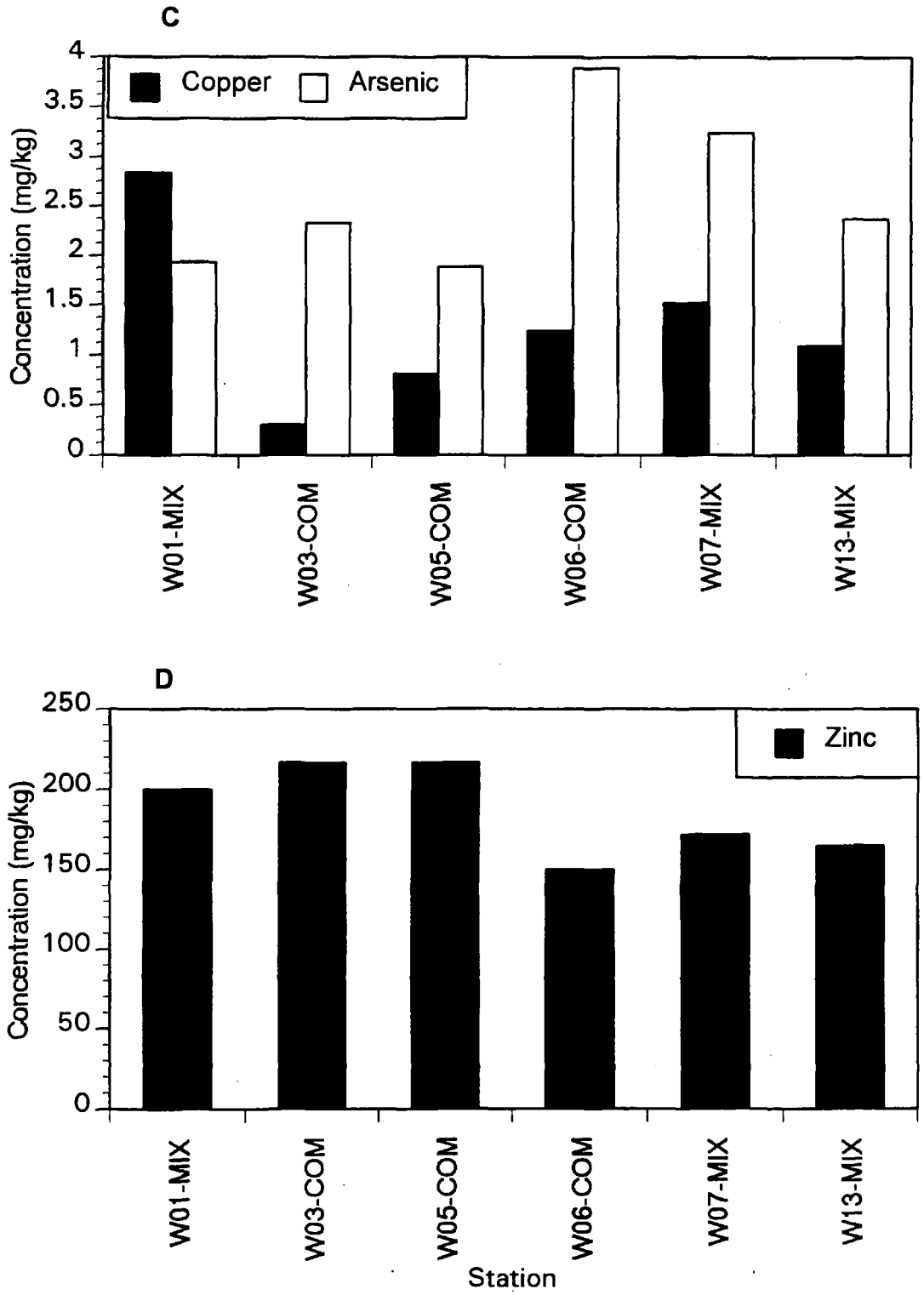


Figure 4.2-18. continued

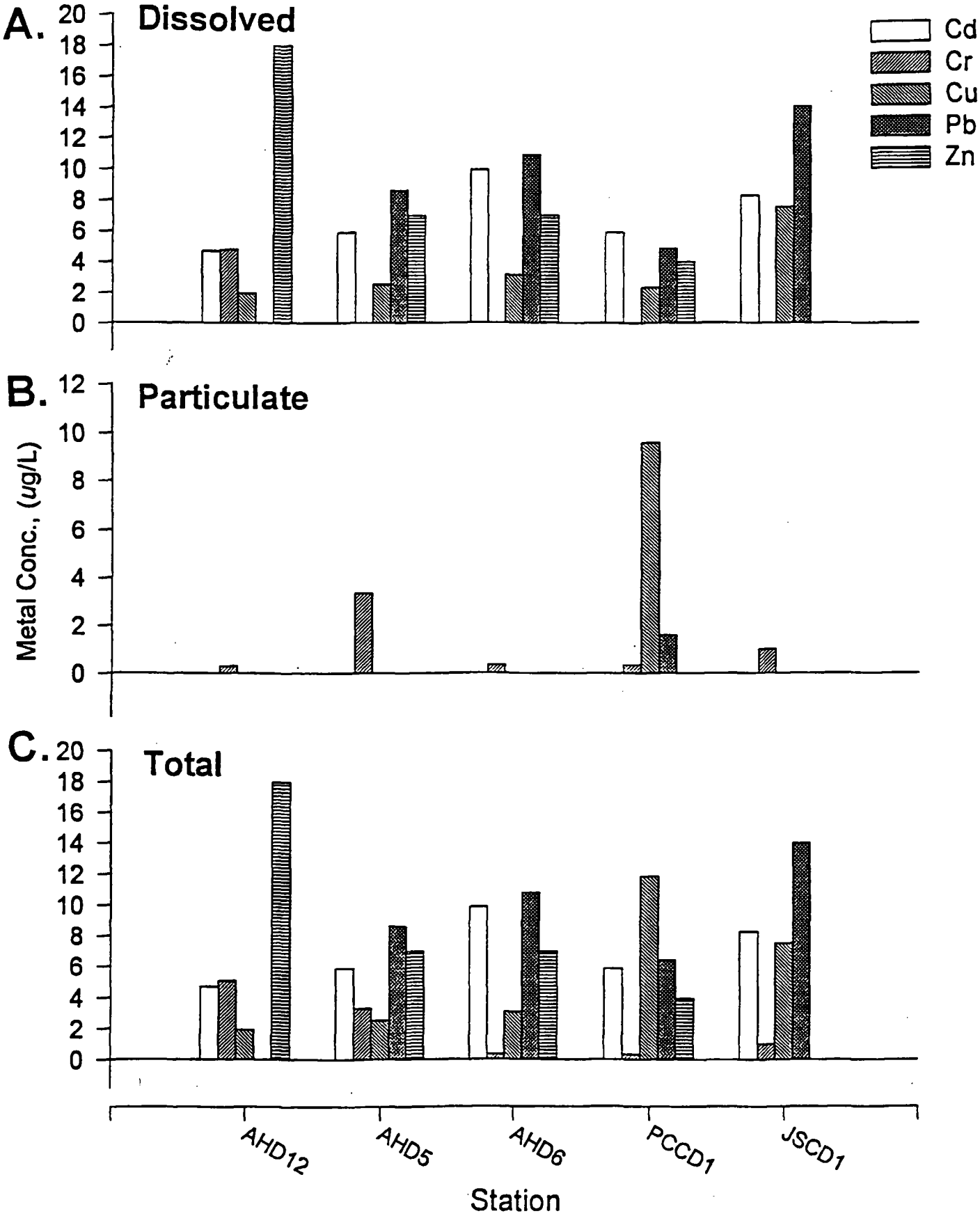


Figure 4.2-19. Concentrations of metals (µg/L) in water samples from Allen Harbor: A) Dissolved, B) Particulate, and C) Total.

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WHOLE CORE SUSCEPTIBILITY

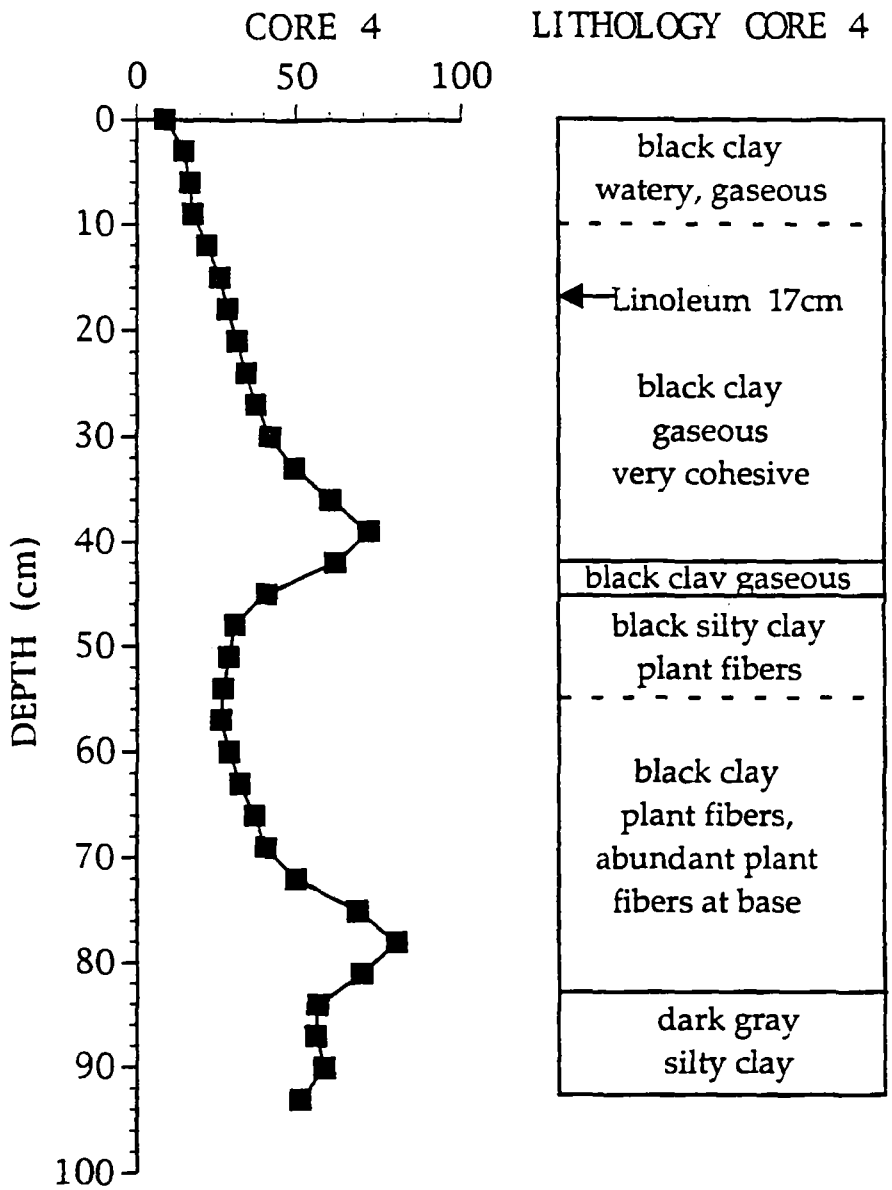


Figure 4.2-20. Whole core magnetic susceptibility profile and lithology description for a deep sediment core in Allen Harbor.

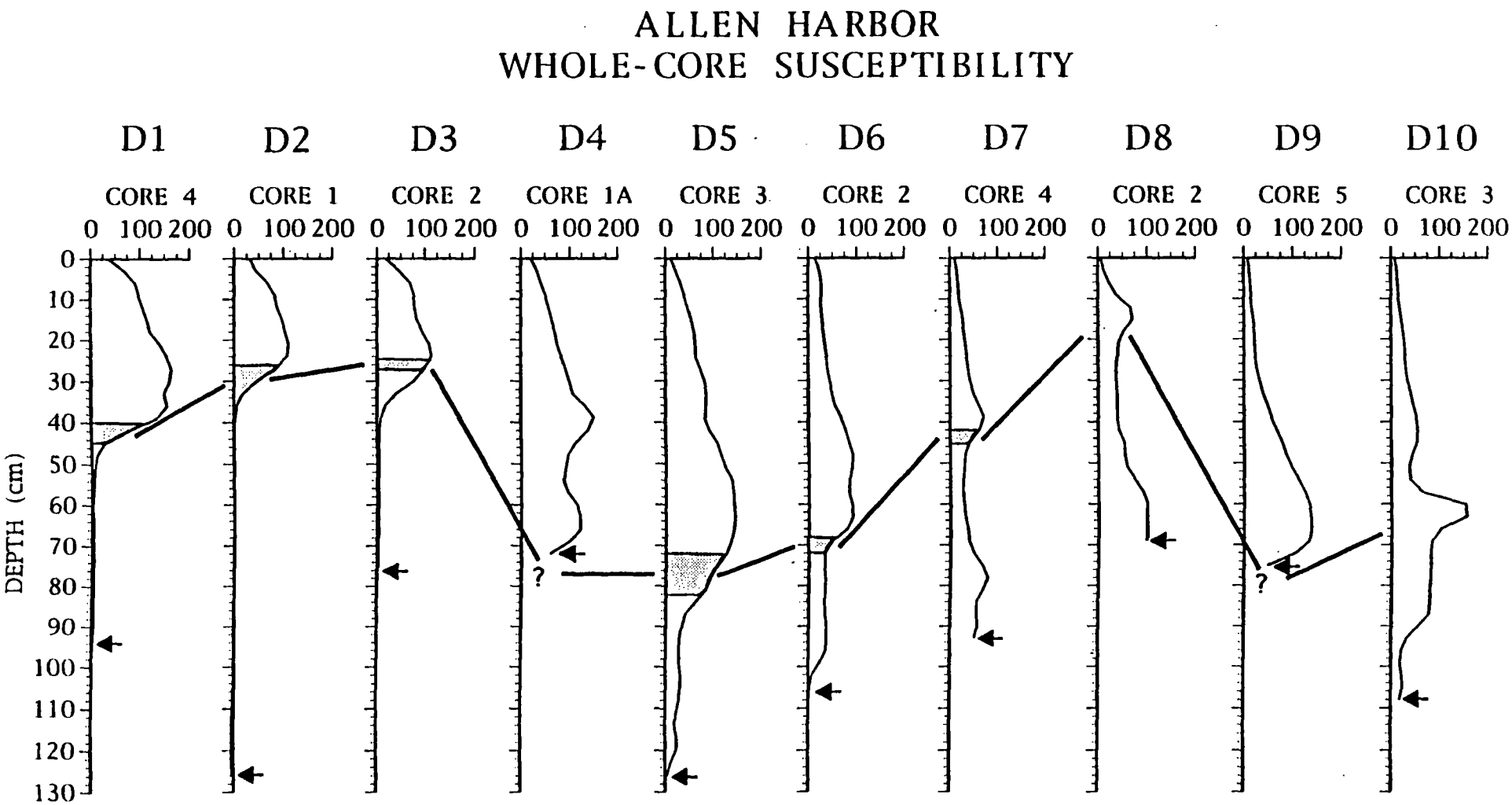


Figure 4.2-21. Cross core magnetic susceptibility correlations and sampling depths for deep cores collected in Allen Harbor. Arrow indicates core depth.

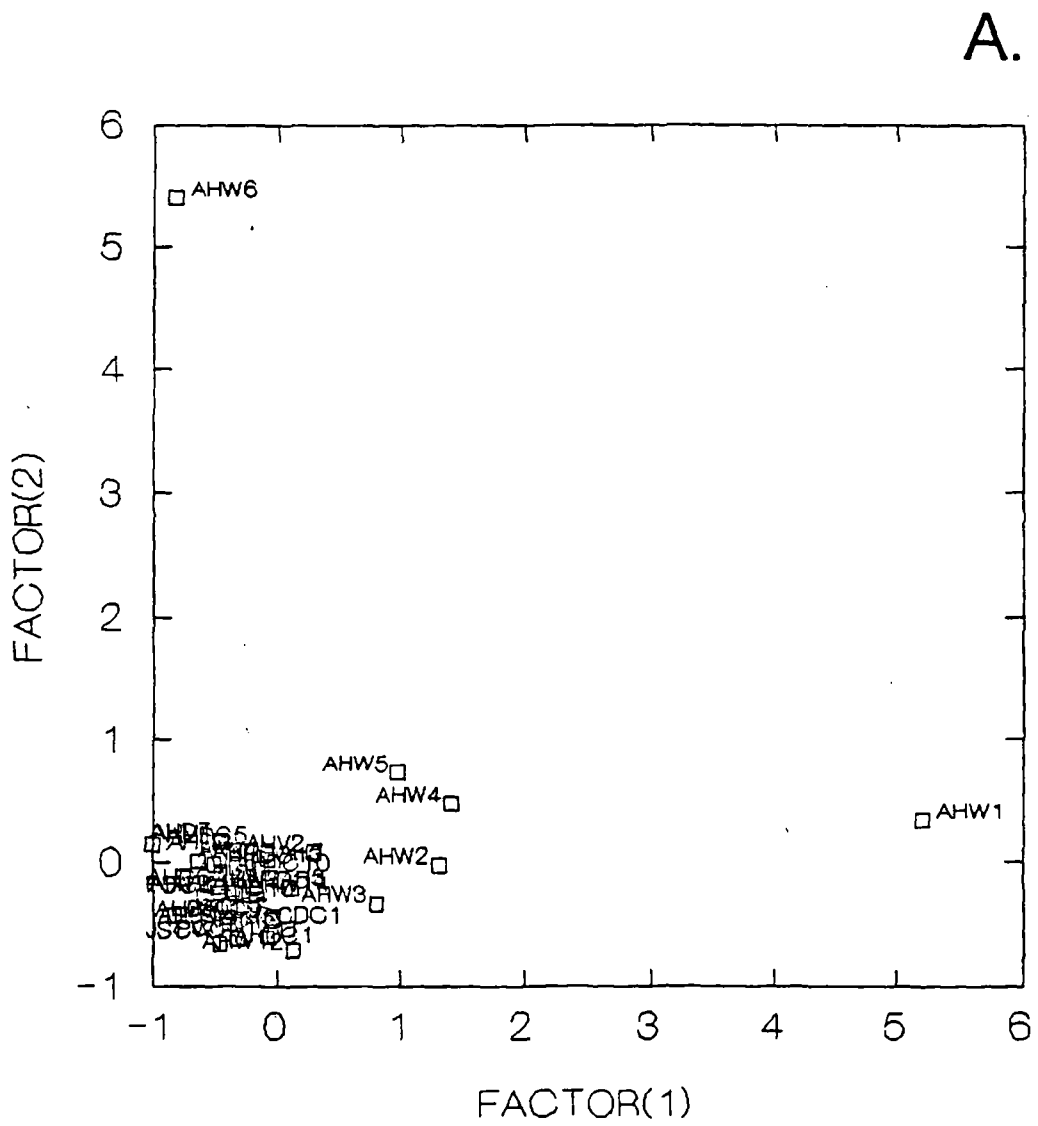


Figure 4.2-22. Principal Component Analyses (PCA) of extended PAH Analytes in Allen Harbor Sediments: A) PC2 vs. PC1, B). PC3 vs. PC1 and C) PC3 vs. PC2.

B.

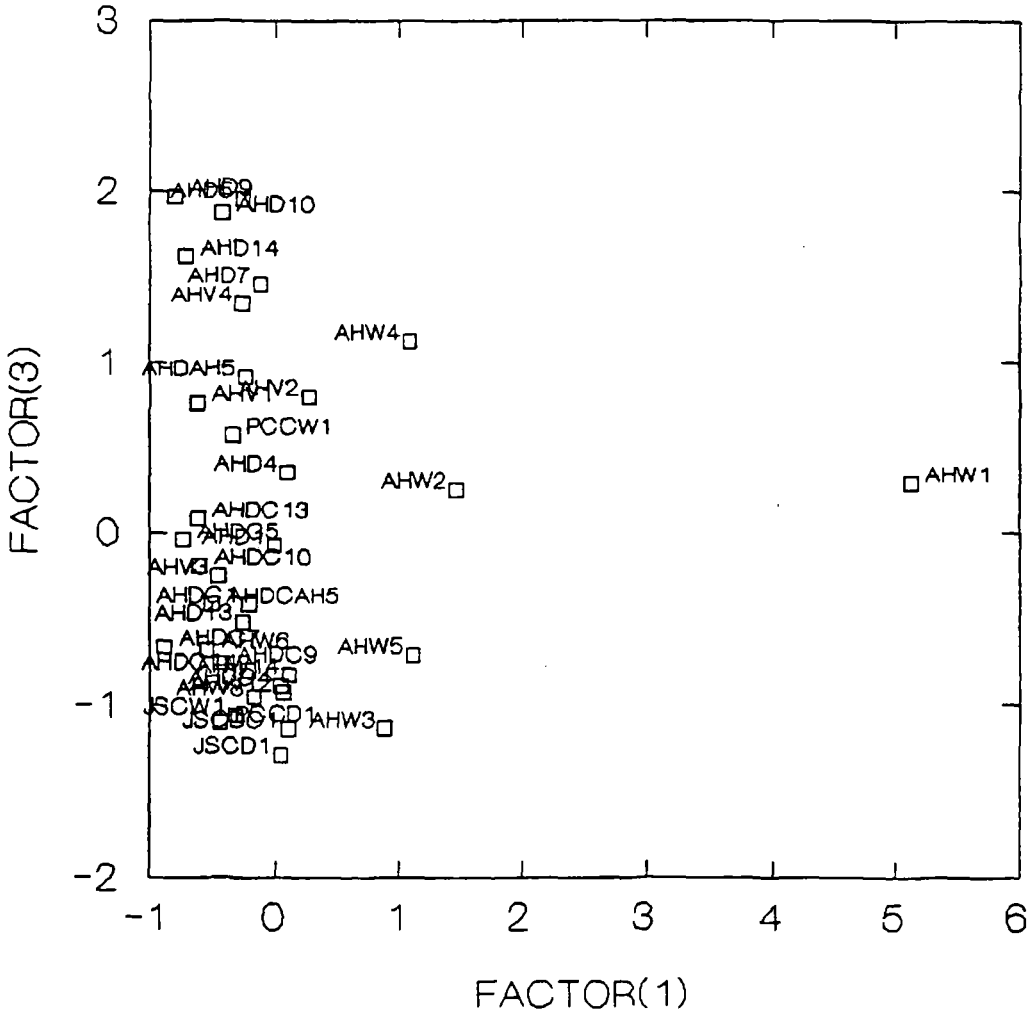


Figure 4.2-22. (con't.)

C.

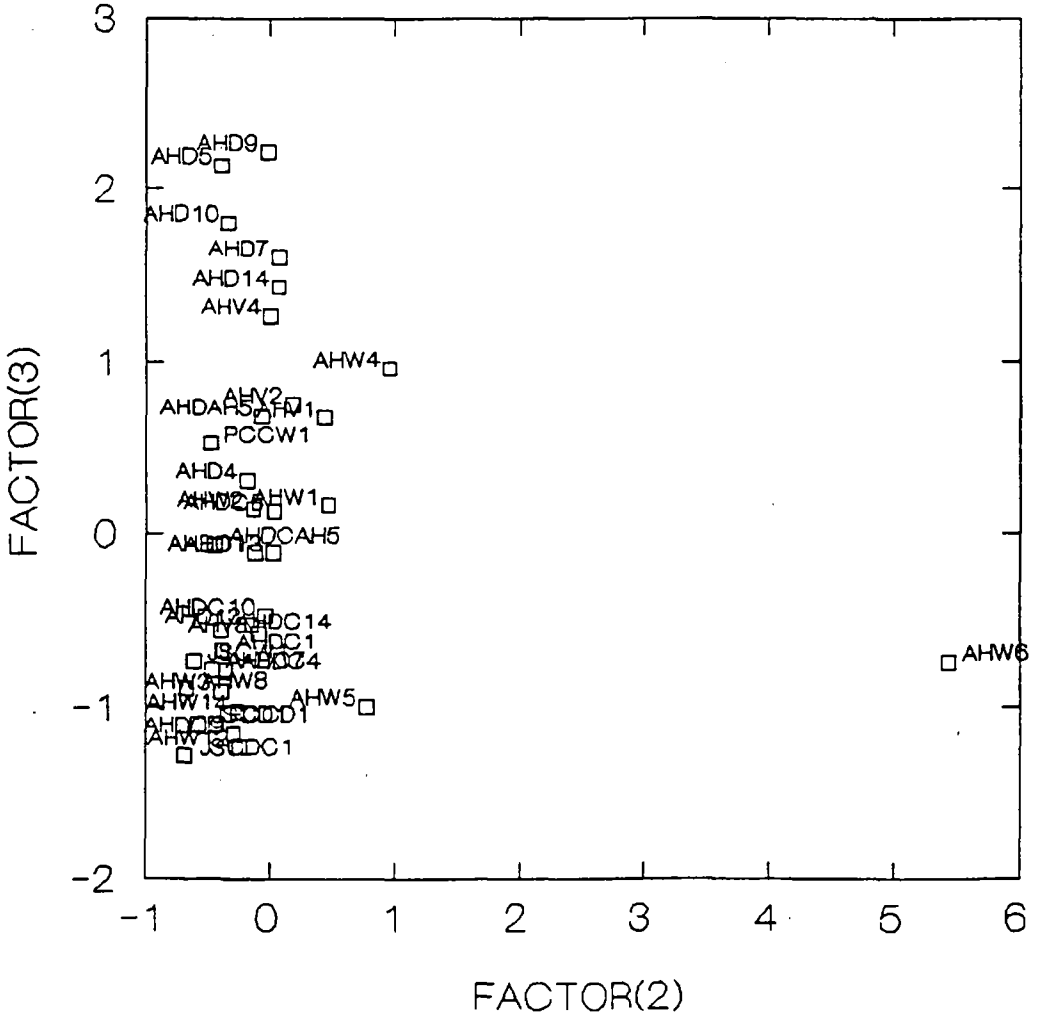


Figure 4.2-22. (con't.)

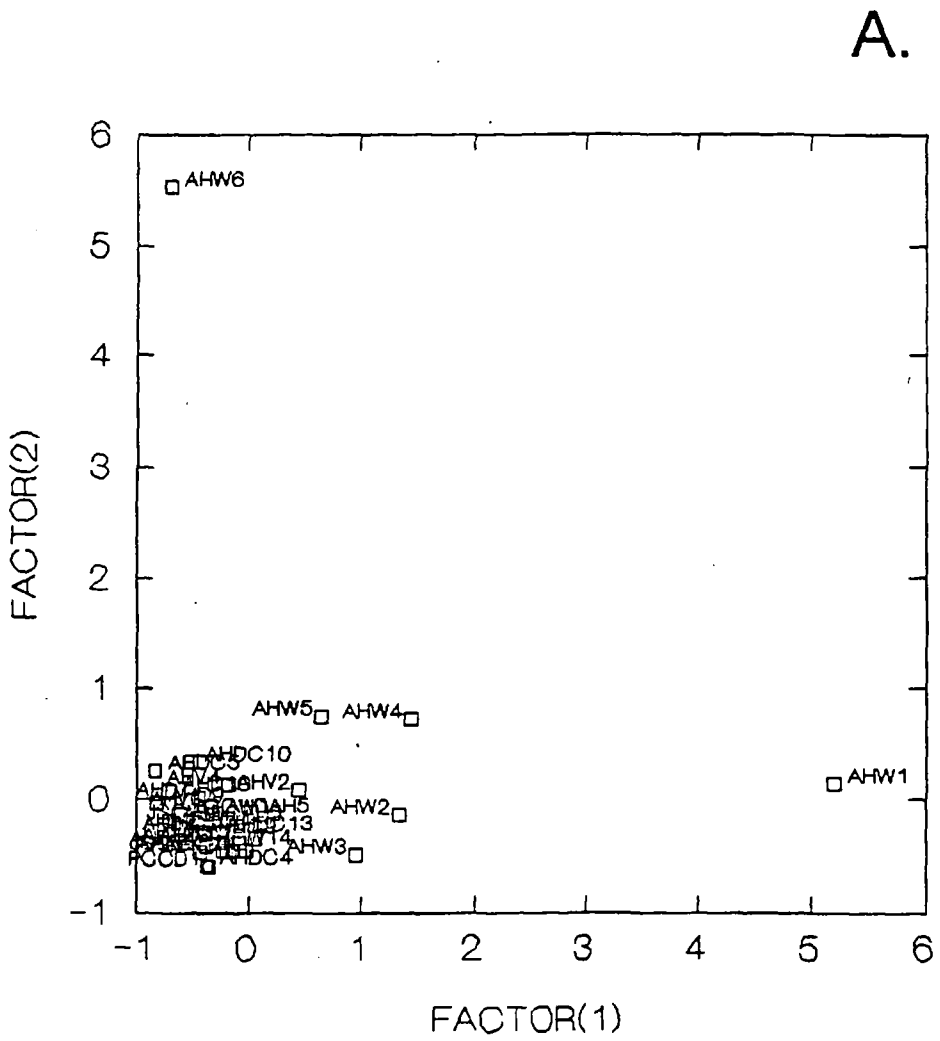


Figure 4.2-23. Principal Component Analyses (PCA) of PCB Analytes in Allen Harbor Sediments: A) PC2 vs. PC1, B). PC3 vs. PC1 and C) PC3 vs. PC2.

B.

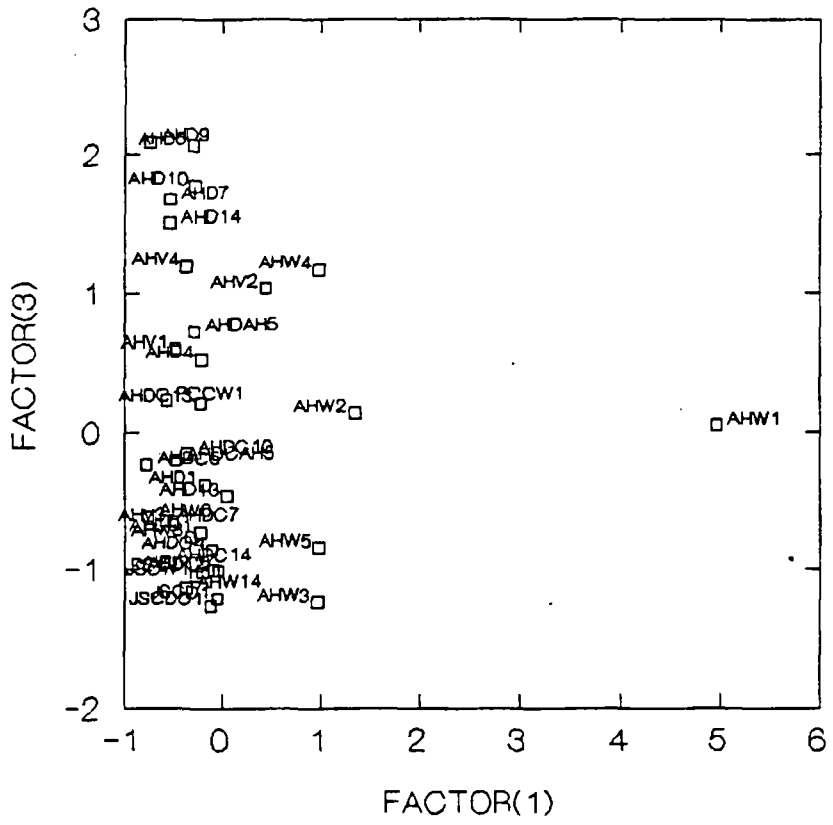


Figure 4.2-23. (con't)

C.

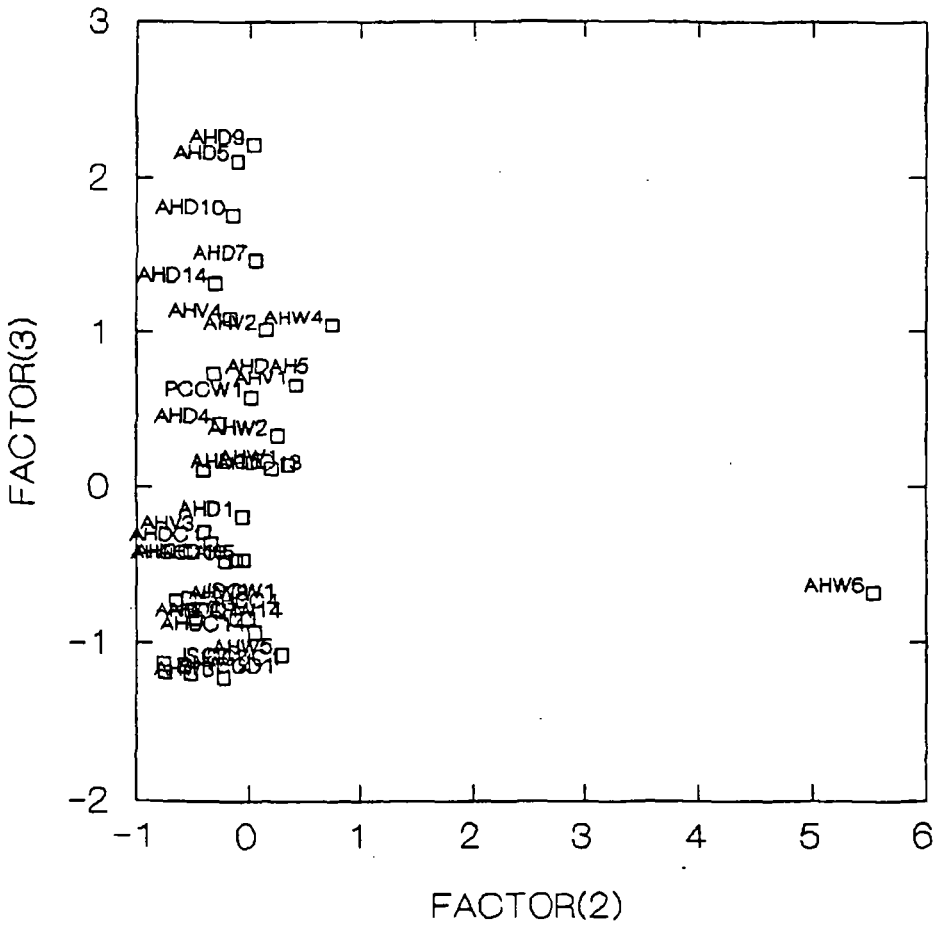


Figure 4.2-23. (con't)

A.

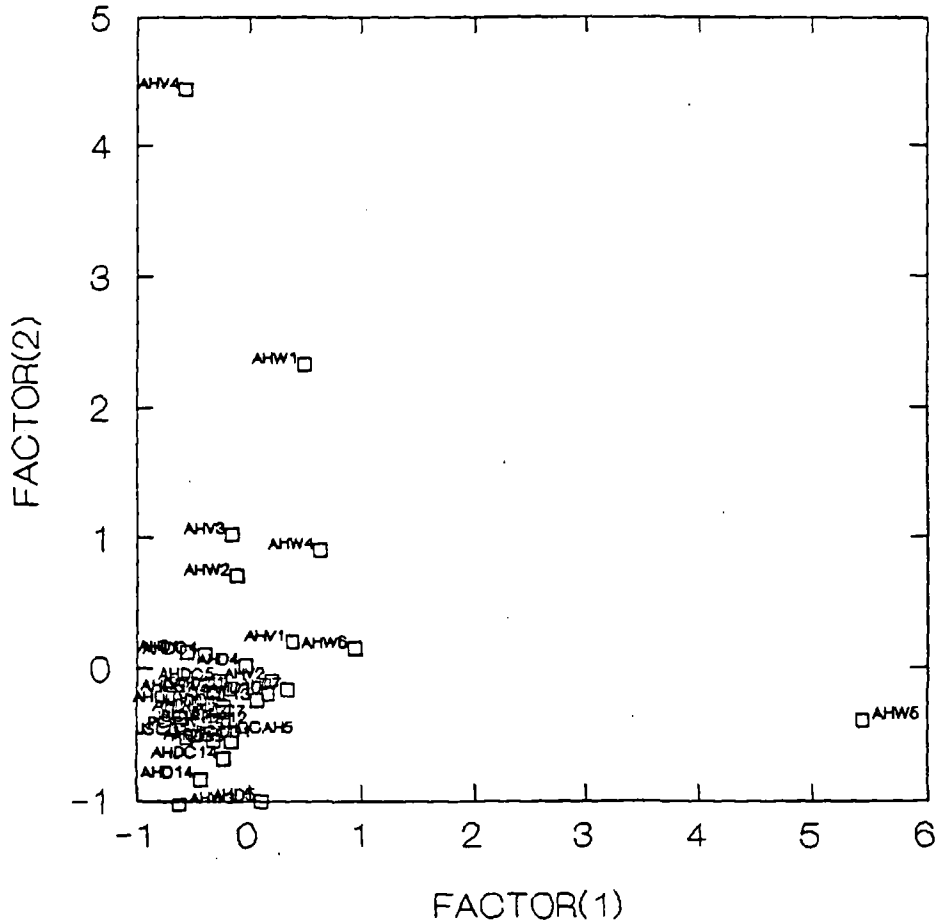


Figure 4.2-24. Principal Component Analyses (PCA) of extended pesticide Analytes in Allen Harbor Sediments: A) PC2 vs. PC1, B). PC3 vs. PC1 and C) PC3 vs. PC2.

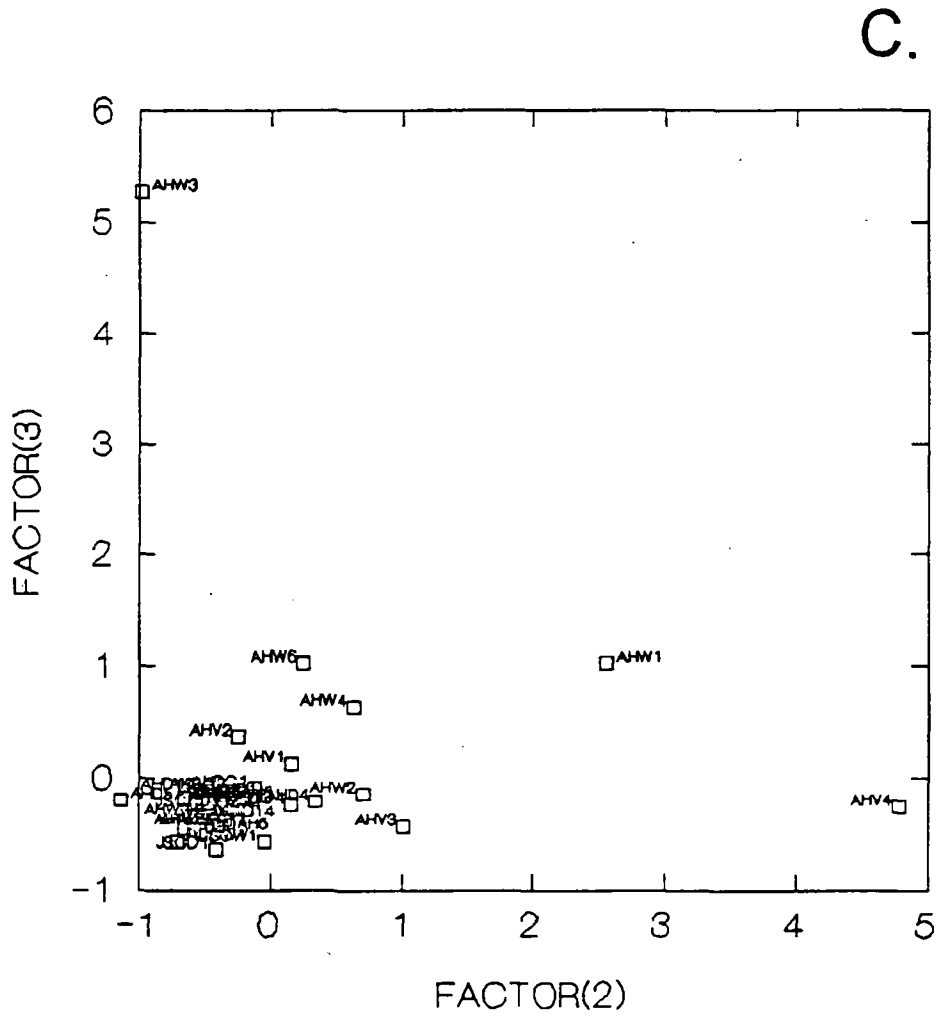


Figure 4.2-24. (con't.)

B.

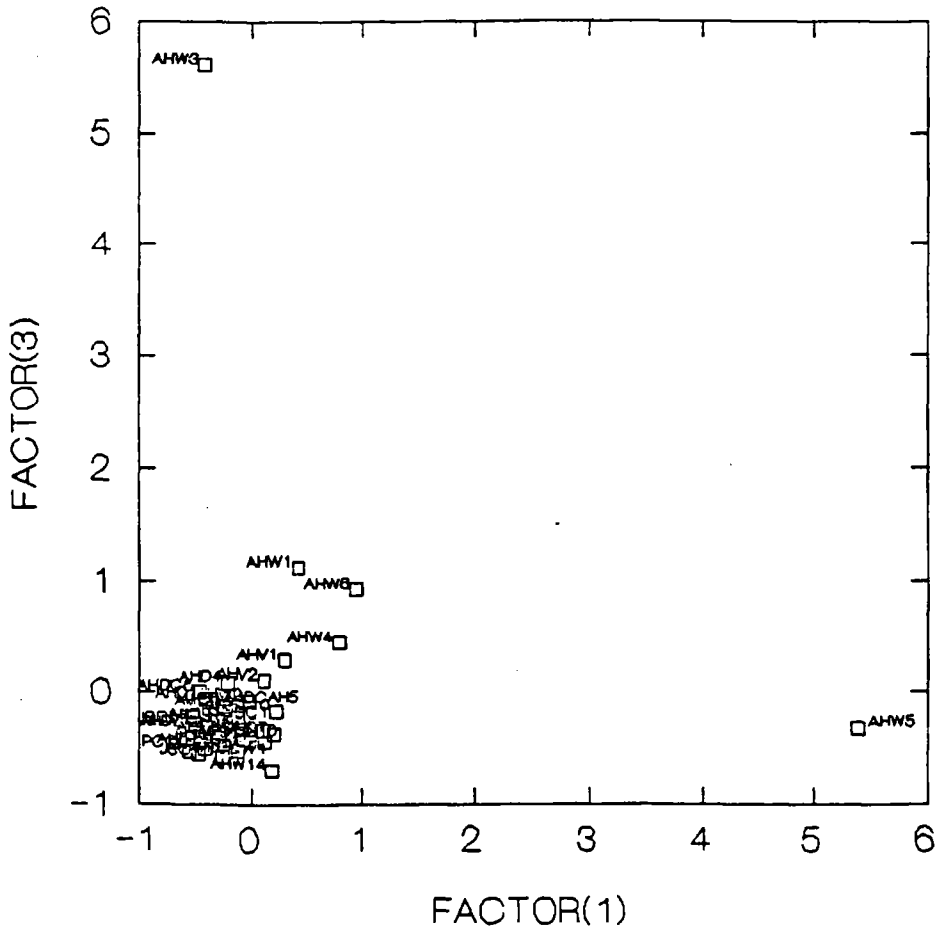


Figure 4.2-24. (con't.)

Table 4.2-1. Acid volatile sulfide (AVS) and simultaneously extracted metals (SEM) of Allen Harbor sediments.

FIELDID	AVS Conc., (uMol/g dry)	SEM Metals (uMol/ g dry wt) CU	ZN	PB	NI	CD	HG	SEM Conc., (uMol/g dry)	SEM/AVS	SEM-AVS
AH-D1	30.20	0.32	0.47	0.24	0.24	0.21	0.20	1.68	0.06	-28.52
PCC-D1	0.01	0.08	0.09	0.06	0.07	0.06	0.06	0.43	9999.00	0.42
JSC-D1	0.28	0.07	0.08	0.06	0.07	0.06	0.06	0.39	1.25	0.11
AH-D10	27.24	0.51	0.81	0.50	0.48	0.46	0.46	3.24	0.12	-24.00
AH-D11	16.21	0.27	0.31	0.18	0.22	0.20	0.20	1.38	0.10	-14.83
AH-D12	15.81	0.16	0.22	0.14	0.14	0.13	0.13	0.93	0.06	-14.88
AH-D13	27.87	0.20	0.28	0.18	0.19	0.18	0.18	1.22	0.04	-26.65
AH-D14	18.55	0.43	0.70	0.41	0.44	0.39	0.39	2.75	0.15	-15.79
AH-D2	21.90	0.29	0.42	0.27	0.28	0.26	0.26	1.77	0.08	-20.12
AH-D3	6.39	0.32	0.41	0.28	0.30	0.27	0.27	1.86	0.27	-4.53
AH-D4	42.93	0.38	0.62	0.38	0.37	0.35	0.35	2.45	0.06	-40.48
AH-D5	64.65	0.44	0.76	0.46	0.46	0.43	0.43	2.99	0.05	-61.66
AH-D6	36.30	0.39	0.65	0.39	0.41	0.36	0.36	2.56	0.07	-33.73
AH-D7	30.61	0.62	0.97	0.58	0.59	0.52	0.52	3.81	0.12	-26.80
AH-D8	65.28	0.53	0.83	0.44	0.41	0.37	0.37	2.95	0.05	-62.33
AH-D9	23.45	0.42	0.67	0.43	0.43	0.40	0.40	2.75	0.12	-20.69
AH-DAH5	17.70	0.65	0.87	0.63	0.66	0.60	0.60	4.01	0.20	-13.69
AH-DC1	1.16	0.16	0.27	0.17	0.19	0.18	0.18	1.15	0.94	-0.01
PCC-DC1	0.01	0.10	0.17	0.10	0.10	0.10	0.10	0.67	9999.00	0.66
JSC-DC1	0.01	0.07	0.08	0.05	0.06	0.05	0.05	0.36	9999.00	0.35
AH-DC10	26.65	0.36	0.54	0.34	0.33	0.29	0.29	2.13	0.08	-24.52
AH-DC11	9.62	0.14	0.19	0.13	0.15	0.13	0.13	0.88	0.09	-8.73
AH-DC12	0.01	0.11	0.15	0.09	0.11	0.08	0.08	0.63	9999.00	0.62
AH-DC13	22.91	0.18	0.33	0.20	0.21	0.19	0.19	1.29	0.06	-21.62
AH-DC14	2.03	0.19	0.25	0.16	0.16	0.15	0.15	1.06	0.49	-0.97
AH-DC2	2.49	0.23	0.21	0.13	0.15	0.13	0.13	0.99	0.44	-1.50
AH-DC3	1.34	0.13	0.21	0.11	0.12	0.10	0.10	0.77	0.61	-0.56
AH-DC4	2.08	0.13	0.15	0.10	0.11	0.09	0.09	0.67	0.32	-1.41
AH-DC5	13.88	0.27	0.44	0.26	0.27	0.24	0.24	1.71	0.13	-12.17
AH-DC6	13.79	0.19	0.37	0.34	0.37	0.36	0.36	1.99	0.11	-11.80
AH-DC7	19.88	0.18	0.28	0.19	0.19	0.17	0.17	1.20	0.06	-18.68
AH-DC8	24.42	0.25	0.40	0.23	0.23	0.22	0.21	1.54	0.06	-22.88
AH-DC9	7.51	0.21	0.24	0.19	0.19	0.17	0.15	1.15	0.15	-6.36
AH-DC-AH5	9.16	0.71	0.88	0.65	0.68	0.64	0.64	4.18	0.38	-4.98
AH-V1	5.83	0.55	0.73	0.48	0.49	0.45	0.45	3.15	0.54	-2.68
PCC-V1	0.15	0.22	0.24	0.14	0.16	0.12	0.12	1.00	7.15	0.86
JSC-V1	0.42	0.24	0.28	0.18	0.19	0.17	0.17	1.23	2.53	0.82
AH-V2	21.04	0.38	0.66	0.42	0.42	0.38	0.38	2.62	0.12	-18.42
AH-V3	0.01	0.22	0.25	0.20	0.19	0.17	0.17	1.20	9999.00	1.19
AH-V4	0.01	0.20	0.18	0.15	0.16	0.13	0.13	0.94	9999.00	0.93
AH-W1	20.24	0.45	0.75	0.47	0.48	0.39	0.39	2.94	0.15	-17.30
PCC-W1	5.42	0.22	0.29	0.20	0.22	0.20	0.20	1.32	0.22	-4.10
JSC-W1	0.31	0.54	0.57	0.39	0.39	0.36	0.36	2.61	6.92	2.30
AH-W10	2.22	0.11	0.16	0.11	0.11	0.10	0.10	0.69	0.31	-1.53
AH-W11	1.07	0.28	0.37	0.23	0.25	0.23	0.23	1.59	1.48	0.52
AH-W12	3.47	0.13	0.16	0.09	0.11	0.09	0.09	0.66	0.20	-2.81
AH-W12FD	4.48	0.14	0.17	0.11	0.13	0.10	0.10	0.74	0.17	-3.74
AH-W13	0.68	0.20	0.22	0.16	0.18	0.13	0.13	1.02	1.58	0.34
AH-W14	1.91	0.09	0.14	0.09	0.10	0.08	0.08	0.58	0.31	-1.33
AH-W2	30.25	0.48	0.74	0.48	0.46	0.44	0.43	3.03	0.10	-27.22
AH-W3	7.34	0.34	0.45	0.33	0.35	0.32	0.32	2.13	0.29	-5.21
AH-W4	47.86	0.90	1.31	0.84	0.81	0.77	0.77	5.40	0.11	-42.46
AH-W5	3.29	2.83	4.39	2.71	2.59	2.53	2.53	17.58	5.34	14.29
AH-W6	1.94	0.36	0.46	0.31	0.32	0.28	0.28	2.02	1.07	0.08
AH-W7	0.80	0.21	0.24	0.13	0.15	0.12	0.12	0.96	1.29	0.16
AH-W8	2.92	0.25	0.36	0.23	0.24	0.22	0.22	1.52	0.52	-1.39
AH-W9	21.45	0.19	0.33	0.24	0.25	0.23	0.23	1.45	0.06	-20.00
AH-W9FD	29.58	0.26	0.49	0.33	0.35	0.32	0.32	2.07	0.07	-27.51

Table 4.2-2. Extended PAH and pesticides quantified for source characterization assessments.

Analyte Abbreviation	Compound
Polyaromatic Hydrocarbons	
C1CHRYS	C1-Chrysene
C1DIBEN	C1-Dibenzothiophene
C1FLPY	C1-Fluoranthene/Pyrenes
C1FLUOR	C1-Fluorenes
C1NAPH	C1-Naphthalene
C1PHAN	C1-Phenanthrene & Anthracene
C2CHRYS	C2-Chrysene
C2DIBEN	C2-Dibenzothiophene
C2FLUOR	C2-Fluorenes
C2NAPH	C2-Naphthalene
C2PHAN	C2-Phenanthrene & Anthracene
C3CHRYS	C3-Chrysene
C3DIBEN	C3-Dibenzothiophene
C3FLUOR	C3-Fluorenes
C3NAPH	C3-Naphthalene
C3PHAN	C3-Phenanthrene & Anthracene
C4CHRYS	C4-Chrysene
C4NAPH	C4-Naphthalene
C4PHAN	C4-Phenanthrene & Anthracene
Organo-chlorine Pesticides	
A_CHLOR	Alpha Chlordane
DDD_OP	2,4' DDD
DDD_PP	4,4' DDD
DDT_OP	2,4' DDT
DDT_PP	4,4' DDT
DIELDR	Dieldrin
G_BHC	Gamma BHC
HEPCHLO	Hepta Chlor
HEPEPOX	Hepta Epoxide
T_NANCH	Trans-Nonachlor

Table 4.3-1. Summary statistics for CoC concentrations in Allen Harbor sediments by habitat. Units are ng/g.

		Site-Related				Reference			
Group	Habitat	Mean	Minimum	Maximum	No. Stations	Mean	Minimum	Maximum	No. Stations
A. High Molecular Wt. PAHs	Subtidal, deep	2217.66	689.49	8054.00	20	14.41	10.80	18.02	2
	Subtidal, deep core	652.63	48.29	1422.55	15	7.70	6.17	9.23	2
	Vegetated Wetlands	2697.77	852.03	4225.28	4	133.44	21.31	245.57	2
	Intertidal Wetlands	2909.83	19.88	19392.48	18	881.21	554.65	1207.76	2
Low Molecular Wt. PAHs	Subtidal, deep	155.91	30.98	310.45	20	4.76	4.42	5.09	2
	Subtidal, deep core	85.14	14.40	188.68	15	3.70	3.59	3.81	2
	Vegetated Wetlands	215.63	66.36	339.78	4	10.32	4.85	15.78	2
	Intertidal Wetlands	227.38	0.47	1682.07	18	67.91	29.82	105.99	2
Total PAHs	Subtidal, deep	2373.56	756.06	8180.10	20	19.17	15.22	23.11	2
	Subtidal, deep core	737.77	62.69	1567.92	15	11.40	9.98	12.82	2
	Vegetated Wetlands	2913.41	918.39	4565.06	4	143.76	26.16	261.35	2
	Intertidal Wetlands	3137.21	20.35	21074.55	18	949.11	584.47	1313.75	2
B. Total Pesticides	Subtidal, deep	7.62	1.47	40.65	20	0.12	0.06	0.18	2
	Subtidal, deep core	1.01	0.16	3.12	15	0.16	0.14	0.17	2
	Vegetated Wetlands	10.65	3.30	22.50	4	0.27	0.18	0.35	2
	Intertidal Wetlands	6.56	0.10	24.20	16	0.62	0.30	0.94	2
C. Total PCBs	Subtidal, deep	142.33	15.90	505.00	20	4.92	2.39	7.45	2
	Subtidal, deep core	23.30	6.62	53.04	15	2.84	2.08	3.60	2
	Vegetated Wetlands	159.95	56.80	295.20	4	4.20	2.11	6.30	2
	Intertidal Wetlands	273.68	3.20	1345.20	16	11.16	8.82	13.50	2
D. Total Butyltins	Subtidal, deep	20.23	0.70	112.56	19	3.03	2.98	3.09	2
	Subtidal, deep core	1.19	0.10	6.47	15	1.72	1.65	1.79	2
	Vegetated Wetlands	3.58	0.37	4.68	4	2.54	2.03	3.04	2
	Intertidal Wetlands	9.50	0.16	53.15	16	8.41	1.13	15.68	2
E. Total Metals	Subtidal, deep	328.41	106.30	533.97	20	30.59	26.15	35.03	2
	Subtidal, deep core	226.50	98.65	334.93	15	27.65	24.15	31.15	2
	Vegetated Wetlands	313.26	128.37	558.90	4	41.28	34.28	48.28	2
	Intertidal Wetlands	604.41	23.58	3793.07	18	39.54	34.68	44.40	2
F. Total VOAs	Subtidal, deep	nd	nd	nd	nd	nd	nd	nd	nd
	Subtidal, deep core	161.81	99.90	290.65	15	113.83	110.50	117.15	2
	Vegetated Wetlands	202.45	105.60	278.70	4	148.10	133.30	162.90	2
	Intertidal Wetlands	295.51	93.65	787.40	14	99.72	97.64	101.80	2

nd = no data

Table 4.3-2. Summary statistics for CoC concentrations in Allen Harbor sediments by habitat (TOC normalized). Units are ng/%TOC.

		Site-Related				Reference			
Group	Habitat	Mean	Minimum	Maximum	No. Stations	Mean	Minimum	Maximum	No. Stations
A. High Molecular Wt. PAHs	Subtidal, deep	729.73	395.23	2182.66	20	33.94	27.47	40.41	2
	Subtidal, deep core	374.07	42.51	818.61	15	nd	nd	nd	nd
	Vegetated Wetlands	505.20	172.44	1100.33	4	191.04	13.28	368.81	2
	Intertidal Wetlands	1210.47	58.47	4023.47	16	868.36	850.59	886.13	2
Low Molecular Wt. PAHs	Subtidal, deep	50.62	11.60	93.04	20	11.33	11.24	11.41	2
	Subtidal, deep core	45.53	12.68	99.95	15	nd	nd	nd	nd
	Vegetated Wetlands	40.56	13.43	88.48	4	13.36	3.02	23.70	2
	Intertidal Wetlands	95.82	1.38	348.99	16	61.75	45.73	77.76	2
Total PAHs	Subtidal, deep	780.35	428.32	2216.83	20	45.27	38.71	51.82	2
	Subtidal, deep core	419.60	55.19	918.56	15	nd	nd	nd	nd
	Vegetated Wetlands	545.75	185.87	1188.82	4	204.41	16.30	392.51	2
	Intertidal Wetlands	1306.30	59.85	4372.46	16	930.11	896.32	963.89	2
B. Total Pesticides	Subtidal, deep	2.57	0.64	12.06	20	0.29	0.14	0.45	2
	Subtidal, deep core	0.62	0.13	2.48	15	nd	nd	nd	nd
	Vegetated Wetlands	1.85	0.67	3.20	4	0.32	0.11	0.53	2
	Intertidal Wetlands	3.50	0.24	9.45	14	0.58	0.46	0.69	2
C. Total PCBs	Subtidal, deep	44.31	9.22	143.52	20	12.15	5.37	18.94	2
	Subtidal, deep core	13.96	2.55	40.10	15	nd	nd	nd	0
	Vegetated Wetlands	31.86	11.25	76.88	4	5.39	1.31	9.46	2
	Intertidal Wetlands	132.21	7.80	528.58	16	11.71	9.91	13.52	2

nd = no data

Table 4.3-3. Summary statistics for PAHs, pesticides, PCBs, and TBTs in tissue samples from Allen Harbor. Units are ng/g dry weight, except TBTs, which are ng Sn/g dry weight.

			Statistical Summary							
			Site Related				Reference			
Chemical Group	Analyte	Species	Mean	Minimum	Maximum	n	Mean	Minimum	Maximum	n
PAHs	LMW PAHs	ribbed mussel	2352.88	1307.60	4303.88	19	1550.51	1452.85	1700.31	3
	HMW PAHs		1419.07	961.18	1810.97		721.76	481.25	858.04	
	Total PAHs		3771.96	2425.31	5870.75		2272.26	2181.56	2356.40	
	LMW PAHs	soft shell clam	166.13	114.70	330.34	9				0
	HMW PAHs		636.46	491.99	843.27					
	Total PAHs		802.60	647.42	976.59					
	LMW PAHs	hard clam	73.26	56.79	91.76	20	52.00	49.37	54.36	3
	HMW PAHs		228.50	142.70	285.38		81.69	70.22	100.51	
	Total PAHs		301.76	199.49	355.61		133.69	124.58	149.88	
	LMW PAHs	oyster	109.07	86.96	134.16	13	71.67	67.88	73.96	3
	HMW PAHs		223.60	166.59	375.67		103.34	94.23	113.95	
	Total PAHs		332.67	253.55	499.13		175.01	167.40	187.91	
	LMW PAHs	mummichog fish	111.55	67.99	152.09	8	87.51	85.14	89.88	2
	HMW PAHs		47.82	40.92	56.59		52.28	45.75	58.80	
	Total PAHs		159.37	108.90	202.14		139.79	135.63	143.94	
Pesticides	p,p'-DDE	ribbed mussel	28.59	13.30	52.30	19	12.80	9.31	17.68	3
	p,p'-DDE	soft shell clam	9.82	6.93	14.41	9				0
	p,p'-DDE	hard clam	6.22	3.43	21.76	20	3.23	2.34	4.22	3
	p,p'-DDE	oyster	29.80	22.70	47.06	13	22.13	14.35	19.35	3
	p,p'-DDE	mummichog fish	90.75	59.40	116.44	8	37.84	30.82	44.85	2
PCBs	Total PCBs	ribbed mussel	600.00	187.84	1267.00	19	190.27	144.35	273.93	3
	Total PCBs	soft shell clam	322.87	178.49	840.09	9				0
	Total PCBs	hard clam	193.01	57.07	545.44	20	79.71	50.96	120.07	3
	Total PCBs	oyster	715.90	302.51	1202.32	13	256.64	192.58	289.35	3
	Total PCBs	mummichog fish	2111.45	1131.15	3204.73	8	903.59	616.46	1190.72	2
TBTs	Tributyltin	ribbed mussel	5.56	0.49	21.45	19	9.74	4.00	20.90	3
	Tributyltin	soft shell clam	45.48	6.46	297.77	9				0
	Tributyltin	hard clam	8.42	0.90	29.79	20	7.17	4.50	11.98	3
	Tributyltin	oyster	2.65	0.51	13.74	13	6.73	2.92	9.45	3
	Tributyltin	mummichog fish	9.40	2.16	16.47	8	0.91	0.33	1.49	2

Note - n refers to the number of samples taken for a given chemical group and species combination.

Table 4.3-4. Summary statistics for metals in tissue samples from Allen Harbor. Units are ug/g dry weight.

	Statistical Summary					
	Site Related			Reference		
Analyte	Mean	Minimum	Maximum	Mean	Minimum	Maximum
Ribbed Mussel (site related: n = 19; reference: n = 3)						
Arsenic	7.58	4.90	11.64	8.43	5.86	12.27
Cadmium	1.13	0.33	3.45	2.01	1.06	3.32
Chromium	7.31	0.09	27.31	1.77	0.95	2.80
Copper	13.45	2.58	27.50	22.43	3.67	44.70
Lead	1.25	0.17	6.25	1.89	0.25	3.86
Mercury	6.76	0.04	44.20	0.13	0.06	0.22
Nickel	16.37	0.40	66.53	3.45	1.26	6.60
Silver	5.01	1.04	17.84	12.92	5.64	17.36
Zinc	46.74	24.96	77.64	68.45	49.48	87.09
Total Metals	105.59	69.03	157.47	121.48	87.74	168.54
Soft Shell Clam (site related: n = 10; reference n = 0)						
Arsenic	3.66	2.50	6.44			
Cadmium	0.78	0.04	2.39			
Chromium	2.24	0.56	7.62			
Copper	33.77	13.54	98.01			
Lead	10.51	0.10	54.06			
Mercury	21.18	0.08	47.65			
Nickel	5.55	2.07	13.29			
Silver	2.95	1.27	5.53			
Zinc	112.05	61.29	230.92			
Total Metals	192.69	109.53	451.21			
Hard Clam (site related: n = 20; reference: n = 3)						
Arsenic	6.54	2.69	12.22	11.34	5.91	17.66
Cadmium	0.37	0.07	2.32	0.29	0.20	0.37
Chromium	2.27	0.75	3.37	3.18	2.65	4.06
Copper	12.84	0.31	70.80	14.31	3.75	33.84
Lead	0.86	0.11	3.53	0.76	0.59	0.91
Mercury	0.41	0.03	6.63	0.04	0.01	0.10
Nickel	4.13	2.01	10.81	4.45	2.90	5.26
Silver	1.29	0.23	3.78	2.54	1.69	4.15
Zinc	86.64	34.09	115.47	83.28	74.10	89.64
Total Metals	115.35	52.98	190.02	120.19	95.97	145.10
Oyster (site related: n = 13; reference: n = 3)						
Arsenic	4.50	2.05	9.33	7.00	5.09	9.00
Cadmium	1.85	0.37	2.92	1.70	1.07	2.81
Chromium	7.77	0.21	19.60	4.76	2.24	8.41
Copper	174.45	41.80	367.23	148.87	8.62	323.28
Lead	2.54	0.39	11.30	1.56	0.55	2.85
Mercury	16.24	0.08	49.15	0.11	0.09	0.14
Nickel	8.14	3.48	14.89	4.12	1.64	6.50
Silver	8.05	2.73	17.76	6.17	1.75	12.94
Zinc	2531.01	529.67	5157.03	214.89	94.34	452.67
Total Metals	2754.56	660.27	5378.13	389.17	137.79	796.64
Mummichog Fish (site related: n = 19; reference: n = 3)						
Arsenic	3.14	1.05	6.37	3.65	1.25	6.83
Cadmium	0.20	0.09	1.31	0.13	0.10	0.19
Chromium	0.69	0.06	1.05	1.27	0.69	1.76
Copper	1.55	0.30	2.84	2.77	1.94	3.67
Lead	0.76	0.06	1.73	0.37	0.14	0.70
Mercury	0.06	0.01	0.12	0.06	0.01	0.10
Nickel	1.00	0.06	10.11	0.47	0.41	0.50
Silver	0.82	0.41	2.37	0.81	0.69	0.92
Zinc	181.65	129.24	217.39	170.67	159.67	185.84
Total Metals	211.41	154.33	252.32	197.84	181.38	223.86

Table 4.4-1. Data comparability assessment for metals analyses of split tissue samples by two laboratories.

A. Results

Split	LAB	TA/LOC	SPP	Zn	Cr	Pb	Ni	Mn	Cd	Cu	Ag	As	g (ng/g)
				Metal									
1	URI	W8	OYS	5583.0	0.3	0.3	3.3	4.2	1.4	170.9	2.0	3.0	50.7
	MISS	"	"	316.0	1.4	3.4	2.0	4.9	1.3	197.0	4.6	1.6	87.0
2	URI	W11	RM	20.2	0.3	0.0	0.1	0.0	0.5	10.2	1.3	6.1	11.0
	MISS	"	"	51.1	22.7	0.0	30.3	7.2	0.6	7.2	2.4	10.9	157.0
3	URI	W10	HC	104.0	0.3	0.4	7.2	33.5	0.5	10.9	0.6	4.6	19.9
	MISS	"	"	65.4	4.2	1.4	4.3	13.0	0.4	33.2	6.1	15.5	3.0
4	URI	W13	MYA	74.8	1.1	0.5	2.1	28.5	0.2	21.3	2.1	3.4	17.5
	MISS	"	"	61.4	3.7	0.4	4.1	11.7	0.4	10.2	2.9	10.4	97.0

B. Relative Percent Difference

1	W8	OYS	178.6	129.4	171.6	0.0	16.6	6.0	14.2	77.9	60.9	52.7
2	W11	RM	86.7	195.3	0.0	198.7	198.9	12.4	34.5	60.2	56.5	173.8
3	W10	HC	45.6	171.7	116.4	50.7	88.1	29.8	101.4	164.4	108.5	147.6
4	W13	MYA	19.7	106.4	22.2	64.1	83.6	50.0	70.6	32.7	101.4	138.8
		Mean RPD:	82.6	150.7	77.5	78.4	96.8	24.5	55.2	83.8	81.8	128.2
		Overall RPD:	86.0									

Table 4.4-2. Data comparability assessment for metals analyses of station AH-5 sediment by two studies.

		Metal (ug/g dry)									
Study	Digestion Method	Cu	Zn	Cr	Pb	Ni	Cd	Mn	Ag	Hg	As
RAPS Phase I, 1991	2N HNO3	77	134	41	42	19	1.67	180	1.7	0.0	7.2
SAIC, 1995	HCl, HNO3, and HF	94	128	64	46	15	0.5	316	1.2	0.26	9.8
	RPD:	19.9	4.6	43.8	9.1	23.5	29.1	54.8	52.6	200.0	30.6
	Overall RPD:	46.8									

Table 4.4-3. Comparison of tissue and sediment SRM recoveries by two studies.

	SRM Recoveries (%)			
	RAPS Phase I (1991)		SAIC (1995)	
Analyte	Tissue	Sediment	Tissue	Sediment
Cu	80-116	97-102	95	83-114
Zn	71-99	84-86	100	96-112
Cr	28-89	34-41	88	84-121
Pb	88-119	74-86	83	85.7
Ni	75-109	65-81	143	100
Cd	79-103	92-111	94	113
Mn	81-110	53-62	81	74-102
Fe	84-101	67-72	88	96-103
As	63-101	58-73	66	109
Ag	105-141	nm	131	nm
Hg	88	71	36	92
Al	nm	nm	48	69-102

5.0 ECOLOGICAL EFFECTS ASSESSMENT

Ecological effects are quantified from the relationships between exposure patterns and resulting responses of ecological systems, as determined from measurement endpoints identified during problem formulation (Chapter 3). These assessments include literature-reported evaluations of the known effects of CoCs to receptors of concern; direct measurement of the toxicity of exposure media, in this case, sediments and pore water, to appropriately sensitive marine species (the amphipod *Ampelisca* and the sea urchin *Arbacia*, respectively); site-specific investigations of the abundance and condition of receptors of concern; and collation of toxicity-based criteria and standards for exposure media identified in exposure pathways. These assessments include discussion of the uncertainty associated with the analyses.

5.1 KNOWN EFFECTS OF COCS

Constituents of concern as identified in Chapter 3 consist primarily of PAHs (e.g., anthracene, benzo(a)pyrene, flouranthene, and phenanthrene), the chlorinated pesticide DDE, PCBs, the metals Ag, As, Cd, Cr, Cu, Hg, Mn, Ni, Pb, and Zn, volatile organics (e.g., 1,2-dichlorethene, trichloroethene, and chlorobenzene) and tributyltin (TBT).

Potential effects of the CoCs on biological receptors are influenced strongly by their chemical behavior, solubility, and toxicity. For example, Ni, Cu, Cd, and Cr 6+ have relatively high solubility and thus, higher dissolved phase concentrations than many organic constituents such as PAHs and relatively insoluble metals, such as Ag, Pb, Zn, and Cr +3. Subsequently, dissolved constituents may be transported throughout the water column by current and tidal flows, while constituents associated with particles tend to be transported horizontally, commonly settling to the bottom in sediment depositional areas. Once on the bottom, the sediment particles can be

transported as bedload or resuspended, resulting in redistribution of the constituents. Dissolved or particle/sediment-bound constituents may be available to biological receptors in the water column, pore waters and sediments, potentially resulting in biological uptake and/or direct toxicological effects. Impacts to organisms can then be strongly influenced, for example, by the affinity of various constituents for tissue lipids and the type of cellular or subcellular effects associated with particular compounds and elements.

The following describes the chemical behavior and known effects of key constituents of concern.

Arsenic - Arsenic in surface water can undergo complex patterns of transformation including oxidation-reduction reactions, biotransformation, precipitation, and adsorption. This complexity results in extremely mobile behavior in aquatic systems, with sorption onto clays, iron oxides, manganese compounds, and organic material a typical fate. Sediments can serve as reservoirs for much of this arsenic, and sediment-bound material (arsenate/arsenite) that has been methylated by aerobic and anaerobic bacteria that may be released back into the water column (Eisler 1987a). Bioconcentration of arsenic occurs in aquatic organisms, primarily in algae and lower invertebrates. Biomagnification in aquatic food chains does not appear to be significant, although some fish and invertebrates contain high levels of arsenic compounds that are relatively inert toxicologically (Eisler 1987a). Arsenic in seafood occurs primarily as complex methylated or organic chemical species which are less toxic and more readily excreted than inorganic arsenic. The Effects Range-Low (ER-L) and Effects Range-Medium (ER-M) benchmarks, defined by Long et al. (1995) as the lower 10th and 50th percentiles of all concentrations of a contaminant observed to cause a biological effect, over a range of studies and species, are 8.2 and 70 mg/kg, respectively (also see Section 5.4). There is good evidence that arsenic is carcinogenic in humans, although evidence of arsenic-induced carcinogenicity in animals is mostly negative. In contrast, very high oral doses of sodium arsenite may

be teratogenic and fetotoxic. Arsenic is a weak inducer of chromosomal aberrations, and is a known teratogen in vertebrates (Eisler 1988). Arsenic exposure may produce behavioral impairment, and leads to death at high concentrations. In aquatic invertebrates, arsenic exposure may lead to decreased growth, reproductive impairment, and death. Pre-exposure to sublethal levels of arsenic may result in increased tolerance to this element upon re-exposure (Eisler 1988). It is generally agreed that inorganic arsenic is more toxic than organic arsenic, and that trivalent forms are more toxic than are pentavalent forms.

Cadmium - Cadmium is relatively mobile in the aquatic environment based on relatively high solubility compared to most heavy metals. It does not form volatile compounds in aqueous settings nor does biological methylation occur (Eisler, 1985). There is sufficient animal evidence to support an association of cadmium inhalation with carcinogenicity, although there is no support for similar effects from oral exposure. The ecological effects on fish, wildlife, and invertebrates have been summarized by Eisler (1985, p. iii):

> "...cadmium is a known teratogen and carcinogen, a probable mutagen, and had been implicated as the cause of severe deleterious effects on fish and wildlife. The freshwater biota is the most sensitive group; concentrations of 0.8 to 9.9 µg Cd/L (ppb) in water were lethal to several species of aquatic insects, crustaceans, and teleosts, and concentrations of 0.7 to 5.0 ppb were associated with sublethal effects such as decreased growth, inhibited reproduction, and population alterations. ...Marine organisms were more resistant than freshwater biota. Decapod crustaceans, the most sensitive saltwater group, died at concentrations of cadmium in seawater ranging from 14.8 to 420 ppb. Sublethal effects to marine animals recorded at Cd concentrations of 0.5 to 10 ppb included decreased growth, respiratory disruption, altered enzyme levels, and abnormal muscular contractions; effects were usually most obvious at relatively low salinities and high temperatures. ...birds are comparatively

resistant to the biocidal properties of cadmium. ...Sublethal effects of cadmium in birds, which were similar to those in other animals, included growth retardation, anemia, and testicular damage; however, these effects were observed at higher concentrations than in aquatic biota."

The ER-L and ER-M values for cadmium in sediments are 1.2 and 9.6 mg/kg, respectively (Long et al. 1995).

Chromium - Chromium +6 occurs only rarely in nature, except from anthropogenic contamination, because it is readily reduced to chromium +3 in the presence of oxidizable organic matter. However, chromate and dichromate (chromium +6) compounds are stable in many natural waters because of the low concentration of reducing material, and thus may undergo intermedia transport. In contrast, chromium +3 compounds, the form most commonly observed in biological systems, are generally insoluble in water. The effects of chromium on wildlife, fish, and invertebrates have been summarized by Eisler (1986, p. 29):

"...chromium is mutagenic, carcinogenic, and teratogenic to a wide variety of organisms and Cr^{+6} [hexavalent Cr] has the greatest biological activity. However, information is lacking on the biological activities of water soluble Cr^{+3} [trivalent Cr] compounds, organochromium compounds, and their ionic states. Aquatic plants and marine polychaete worms appear to be the most sensitive groups tested. In exposures of Cr^{+6}, growth of algae was inhibited at 10.0 ppb, and reproduction of worms at 12.5 ppb. At higher concentrations, Cr^{+6} is associated with abnormal enzyme activities, altered blood chemistry, lowered resistance to pathogenic organisms, behavioral modifications, disrupted feeding, histopathology, osmoregulatory upset, alterations in population structure and species diversity indices, and inhibition of photosynthesis. Not all sublethal effects observed were permanent, but the potential foe acclimatization of organisms to Cr is not well documented."

Cr exposure at high concentrations also produces death. Sensitivity to Cr varies widely among species, even among those which are closely related (Eisler 1986). Chromium +6 is classified as a human carcinogen, but chromium +3 still is being evaluated for its carcinogenic potential. Consequently, most of the chromium in aquatic environments eventually is expected to precipitate in sediments. The ER-L and ER-M values for chromium in sediments are 81 and 370 mg/kg, respectively (Long et al. 1995).

Copper - The two processes that primarily influence the fate of copper in the aquatic environment are sorption and chemical speciation. Speciation is determined by the oxidation-reduction potential of the copper compound and the media pH. In contaminated settings, copper may form complexes with organic material in the water; however, copper ultimately settles out of the water column and is deposited in sediments. Various processes including sorption onto clay minerals, hydrous iron, manganese oxides, and organic material reduce the level of copper compounds in aquatic media. In organically rich sediments, the sorbed and precipitated copper may become redissolved through complexation and can persist in the water for long periods.

Copper is an essential element for most organisms, although the distinction between deficiency and toxicity in some organisms, including algae and some invertebrates, is small if there is limited ability to control absorption. Fish are sensitive to copper, and it is thought that their gills do not provide an effective barrier to absorption (Hammond and Beliles 1980). Copper is toxic to aquatic plants and animals at relatively low levels. In addition to affecting survival, Cu exposure has been associated with development of histopathological lesions in molluscs and fish (Gardner and LaRoche 1973; Sunila, 1984), inhibition of egg hatching in fish (Gardner and LaRoche 1973), impairment of fertilization and larval development in polychaetes and echinoderms (Reisch 1964; Young and Nelson 1974; Bougis 1965), and retardation of growth (Karbe 1972), Cu is particularly active in disruption of enzymatic systems (Albergoni

and Piccinni 1983). Copper is not strongly bioaccumulated and does not appear to transfer significantly through aquatic (or terrestrial) food chains. Bioconcentration factors are in the range of 10 to 100, although in some molluscs it can reach 30,000 (U.S. EPA 1984). This may be because copper proteins in the blood of many shellfish act as oxygen carriers. For example, American oysters have been documented to have tissue concentrations of 1,500 mg/kg (Hammond and Beliles 1980). Acute toxic effects on lower marine biota have been demonstrated at concentrations ranging from 5.8 to 600 ug/l (U.S. EPA 1986; Anderson et al. 1991). In contrast, effects on marine mammals and birds have not been demonstrated conclusively; this probably is because of the more effective homeostatic mechanisms that limit adsorption in these higher organisms. There are no human carcinogenicity data available for exposure to copper, and animal data are incomplete. The ER-L and ER-M values for copper in sediments are 34 and 270 mg/kg, respectively (Long et al. 1995).

Lead - The chemistry of lead in aqueous solutions is highly complex because of its occurrence in many forms, although it has a tendency to form compounds of low solubility. The divalent form (Pb +2) is the stable ionic species of lead. Hydroxide, carbonate, sulfide and, more rarely, sulfate may act as solubility controls. Lead may occur either as adsorbed ions or surface coatings on sediment mineral particles, or it may be carried as a part of suspended living or nonliving organic matter in the water (Eisler 1988b). The ER-L and ER-M values for lead in sediments are 46.7 and 218 mg/kg, respectively (Long et al. 1995). Lead can bioaccumulate in some shellfish, such as mussels, but does not appear to bioaccumulate in fish. The effects of lead on fish, invertebrates, and wildlife are summarized by Eisler (1988b). In vertebrates, Pb is known to modify the structure and function of the kidney, bone, central nervous system, and the hematopoietic system, and produces adverse biochemical, histopathological, neuropsychological, ferotoxic, teratogenic, and reproductive effects. Inhibition of blood delta aminolevulnic acid dehydratase (ALAD), an enzyme critical in heme formation, has been observed as a result of exposure to lead in a variety of marine fish, in invertebrates, and in birds. At sufficiently high concentrations, Pb

effects are manifested in estuarine organisms as reduced growth, fecundity, and survivorship. Lead is classified as a probable human carcinogen, based on animal (primarily rat) studies.

Mercury - Mercury forms a wide variety of complexes with organic ligands, the compounds of which (e.g., methylmercury) are toxicologically and environmentally significant (Eisler 1987a). Mercury is very persistent when released into the environment, with the major removal mechanism occurring by adsorption onto particles and subsequent settlement to sediments. Mercury can become methylated to a highly toxic form, methylmercury, by biological and chemical processes (EPA 1980). Methylation occurs most readily under anaerobic conditions. Mercury has no known essential status or function in organisms, and is a mutagen and teratogen (Eisler 1987). Bioaccumulation and toxic effects of mercury in aquatic systems are highly complex and are influenced by water temperature, salinity, hardness, pH, age of an organism, prior exposure, reproductive state (related to lipid content), trophic level, and metabolism. Mercury is considered to be one of the most toxic of the heavy metals (Nriagu 1979). At higher concentrations, mercury is toxic to a wide range of marine invertebrates and fish, and its acute toxicity varies among species. For instance, Hg is acutely toxic to the mysid shrimp, *Mysidopsis bahia*, at concentrations as low as 3.5 µg/L, whereas the acute value for winter flounder is 1,678 µg/L (U.S. EPA 1985). In general, fish tend to be less sensitive to Hg than are crustaceans and molluscs. In addition to mortality, Hg exposure can result in impairment to reproduction, development, and growth in estuarine plants and animals. For example, productivity and time to first reproduction in *Mysidopsis* was affected in 28-d life cycle tests at concentrations of 1.6 µg/L. Methylmercury can be bioconcentrated and biomagnified through aquatic food chains, with higher concentrations generally observed at the higher trophic levels (e.g., carnivorous fish and piscivorous birds; Eisler 1987). Concentrations of mercury in ocean sediments have been shown to be reflected in the tissues of epifauna (Klein and Goldberg 1970). Bioconcentration factors range upwards to almost 200,000 for marine zooplankton (Hirota et al. 1983), and transfer

rates in piscivorous fish and birds have been documented up to 36,000 (Eisler 1981). Adverse effects on reproduction in birds have been demonstrated at concentrations as low as 5 mg/kg. Mercury is not known to be a human carcinogen (U.S. EPA 1992). The ER-L and ER-M values for mercury in sediments are 0.15 and 0.71 mg/kg, respectively (Long et al. 1995).

Nickel - Very little information on the fate of nickel in the environment could be found in the literature. It is generally characterized as moderately soluble in water. U.S. EPA (1986) provides the following insights into the adverse effects of nickel:

> "Mechanisms of nickel toxicity are varied and complex, and, as with other heavy metals, significant effects occur at cell membranes and membranous tissues, such as gills, In fish, hematological effects such as hyperglycemia, lymphopenia, and erythocytosis have been reported in association with nickel intoxication..."

Nickel exposure has resulted in reduced photosynthesis in aquatic plants (plankton and macrophytes), inhibition of enzyme systems in a variety of organisms, stunted growth and development, reproductive impairment, and at sufficiently high levels, death. Exposure levels associated with these effects are summarized in U.S. EPA (1986). Nickel is classified as a human carcinogen (U.S. EPA 1992). The ER-L and ER-M values for nickel in sediments are 20.9 and 51.6 mg/kg, respectively (Long et al. 1995).

Silver - The toxicity of silver to aquatic life is probably dependent on water hardness; the harder the water, the higher the silver concentration that is needed to be toxic. Silver and its compounds have high chronic toxicity to aquatic life. As with all of the CoCs discussed in this section, the adverse effects of silver include impairments to survival, growth, development, and reproduction in estuarine organisms. Quoting from U.S. EPA (1987, p. 2):

"Symptoms of silver intoxication in aquatic organisms appear to be similar to those caused by other heavy metals. Separation and disruption of the gill epithelium is frequently observed, resulting in esphisia. Damage may be the result of silver ions reacting directly at the gill membrane, or as an indirect result of hematological osmotic imbalances".

Such effects on gill structure often manifest as impairments to respiration, an effect particularly noted on molluscs (see U.S. EPA 1987). Other effects noted in laboratory exposures (summarized in U.S. EPA 1987) include reductions in chlorophyll *a* in phytoplankton populations, ionic imbalance in polychaete coelomic fluid, histopathological changes, impairment of fertilization success and abnormal larval development, and disruption of enzymatic systems. There is no conclusive evidence that silver is carcinogenic to humans. The ER-L and ER-M values for silver in sediments are 1.0 and 3.7 mg/kg, respectively (Long et al. 1995).

Zinc - Sorption onto sediments is probably the most common fate of zinc in the aquatic environment (Eisler, 1993). Small amounts may be partitioned into the dissolved phase through speciation into soluble zinc compounds. Formation of complexes with organic and inorganic ligands may increase the mobility of zinc in aquatic media, but these complexes also have a tendency to be absorbed more strongly onto sediments. The ER-L and ER-M values for zinc in sediments are 150 and 410 mg/kg, respectively (Long et al. 1995). Zinc is an essential element in maintaining many physiological processes, and zinc deficiency can result in severe adverse effects on growth, reproduction, and survival in plants and animals. However, exposure to excess concentrations off zinc can result in a range of adverse ecological effects. According to Eisler (1993, p. 2):

"The most sensitive aquatic species were adversely affected at nominal water concentrations between 10 and 25 µg Zn/L, including representative species of plants, protozoans, sponges, molluscs, crustaceans, echinoderms, fish, and

amphibians. Acute LC_{50} (96 h) values were between 32 and 40,930 µg/L for freshwater invertebrates, 66 and 40,900 µg/L for freshwater teleosts, 195 and >320,000 µg/L for marine invertebrates, and 191 and 38,000 µg/L for marine teleosts. Acute toxicity...was markedly affected by the age and nutrient status of the organism... Pancreatic degeneration occurred in ducks fed diets containing 2,500 mg Zn/kg ration. Ducks died when fed diets containing 3,000 mg Zn/kg feed...:

Eisler (1993) summarizes the huge body of literature pertaining to zinc in the environment. The adverse effects referred to at the beginning of the quote above include decreased growth, survival, and reproduction. Some noncarcinogenic effects of zinc to humans and animals are evident, but information on carcinogenic effects could not be located in the literature.

Tributlytin - TBT leached from antifouling paints inhibits the attachment of fouling organisms (sessile invertebrates) and has been shown to be toxic even at very low concentrations (Wade et al. 1990). TBT concentrations in aquatic sediments likely reflect partitioning between butyltins and suspended particles in the water column, although up to 99% of the TBT may reside in the sediments. Nonetheless, TBT contaminated sediments can represent a substantial source of organotin to aquatic waters (Huggett et al. 1986, as reported in Wade et al. 1990). The average ratio of TBT concentrations in bivalves compared to sediments collected nearby is 18 (range 6.8-57) in coastal waters of the U.S., suggesting a moderate bioaccumulation potential (Wade et al. 1990).

Polycyclic Aromatic Hydrocarbons (PAHs) - High molecular weight (HMW) PAHs (e.g., chrysene, benzo(a)anthracene, benzo(b)flouranthene, benzo(k)flouranthene, benzo(a)pyrene, and dibenzo(a,h)anthracene) typically have low solubility in water, high partition coefficients (i.e., higher affinity for organic matter, such as in soil and sediments, than water), and slow degradation. Based on this slow dissolution and

degradation, significant leaching of HMW PAHs into groundwater is not expected. Solubility of PAHs generally decreases with increasing molecular weight; the less soluble the PAH compound, the more likely it will adsorb to sediment particles. The primary removal mechanisms for PAHs in the aquatic environments are by volatilization, photochemical reactions, and microbial degradation (Eisler, 1987b). In general, PAHs pose little risk of acute or chronic noncarcinogenic human health effects, although in some animals there can be reproductive, epithelial, and immune system effects. In contrast, many PAHs are carcinogenic to humans and animals, primarily at the point of contact. This appears to be caused by metabolic intermediates which are produced rapidly from the PAHs at the contact point. ER-L and ER-M sediment values in ug/kg for various PAH CoCs include 16 and 500 for acenaphthene, 44 and 640 for acenapthhylene, 85.3 and 640 for anthracene, 261 and 1600 for benzo(a)anthracene, 430 and 1600 for benzo(a)pyrene, 384 and 2800 for chrysene, 63.4 and 260 for dibenz(a,h)anthracene, 19 and 540 for flourene, 600 and 5100 for flouranthene, 240 and 1500 for phenanthrene, and 665 and 2600 for pyrene, respectively (Long et al. 1995). PAHs as a group contain a number of individual organic compounds which vary in toxicity and ecological effects. According to Eisler (1987):

> "A wide variety of PAH-caused adverse biological effects have been reported in numerous species of organisms under laboratory conditions, including effects on survival, growth, metabolism, and especially tumor formation. Inter- and intraspecies responses to carcinogenic PAHs were quite variable, and were significantly modified by many chemicals including other PAHs that are weakly carcinogenic or noncarcinogenic. Until these interaction effects are clarified, the results of single substance laboratory tests may be extremely difficult to apply to field situations of suspected PAH contaminants."

In addition to the interactions alluded to by Eisler (1987), an understanding of the potential effects of PAHs is confounded by the fact that many aquatic vertebrate

(primarily fish) and invertebrate (polychaetes, crustaceans, and molluscs) species possess enzymatic systems which support metabolism of PAHs (National Research Council of Canada 1983). Such enzymatic systems have also been observed in bacteria, fungi, and algae. With respect to PAH activation and *catinogenesis*, the National Research Council of Canada (1989) states:

> "Structure-activity relationships for mutagenic and carcinogenic activity seem to favor 4-, 5- and 6-ring PAHs rather than smaller or larger compounds. It is believed that PAHs require metabolic activation to exert their carcinogenic effects...with carcinogenesis being initiated by the binding of electrophilic metabolites to critical cellular constituents. Enzymes other than mixed function oxidase (MFO), which may influence the rate of production or destruction of reactive metabolites, are found in aquatic animals and may play an important role in toxicity."

DDE - DDE, a metabolite of DDT, is very persistent in the environment. Few data are available regarding the environmental fate of DDE, although both DDT and DDD in water are subject to sedimentation, volatilization, photodegradation, and food chain uptake. Bioconcentration factors for DDE are from 10^3 to 10^5 . The ER-L and ER-M values in sediments for DDE are 2.2 and 27 ug/kg, respectively (Long et al. 1995). DDT is absorbed by humans in direct proportion to dietary exposure. Human epidemiological data are not available for DDE, although based in its structural similarity to DDT it is classified as a probable human carcinogen.

PCBs - PCBs, including various Aroclors, vary substantially in their chemical, physical, and biological properties based on their degree of chlorination. The less chlorinated Aroclors will sorb less strongly onto sediments than the highly chlorinated components. Sediment and suspended particulate transport is the dominant mode of PCBs in aqueous solutions. The ER-L and ER-M values for total PCBs in sediments are 22.7 and 180 ug/kg, respectively (Long et al. 1995). PCBs as a group contain a number of

individual congeners which vary with respect to toxicity. Exposure to PCBs in various combinations has resulted in effects on growth of phytoplankton through impairment of photosynthesis and cell division, and has been shown to influence competitive interactions between phytoplankton species (Mosser *et al.* 1972; Fisher *et al.* 1974). PCBs also affect reproduction in fish (Hansen,1974), growth in bivalves (Parrish *et al.* 1972), molting physiology of crustaceans (Fingerman and Fingerman 1977), and may adversely affect population dynamics in fish (Munns *et al.* in prep.). Hansen, (1974) demonstrated the adverse influence of PCB exposure (as Aroclor 1254) on recruitment and development of benthic and epibenthic estuarine communities in laboratory exposure systems. At high enough concentrations, PCBs cause death in a number of estuarine organism.

In summary, the CoCs identified in Section 3.3 can be grouped, at an upper screening level, based on their tendency to be associated with dissolved or particulate/sediment fractions, assuming that other, non-contaminant related factors (e.g. TOC, AVS) are similar in concentration:

- Dissolved fraction components - nickel, copper, cadmium, and chromium^{+6} have a tendency to be more prevalent in the dissolved phase than other metals when conditions permit;

- Particulate/sedimentary fraction components, including PAHs, PCBs, DDE, silver, lead, zinc, arsenic, manganese, mercury, and chromium^{+3} have a greater tendency to be particle-associated than the group above, again assuming comparable geochemical conditions.

This information will be useful in better understanding the spatial distribution of contamination and the patterns of constituent bioaccumulation and toxicity.

5.2 Toxicity Evaluations

Site-specific evaluations of bulk surface sediments and pore waters were conducted using the 10-day amphipod (*Ampelisca abdita*) mortality test and the sea urchin (*Arbacia punctulata*) sperm cell toxicity test, respectively. Both tests are directed tools to evaluate the bioavailability of constituents in the respective media. Comparison of these results to Phase I toxicity data provides a more detailed spatial evaluation of potential impacts to aquatic biota.

5.2.1. Amphipod Test Results

Amphipod tests (5 replicates each) were conducted on surface sediments from 14 intertidal (W1-W14), 14 subtidal (D1-14), and 4 vegetated wetland stations (V1-V4), with an additional field duplicate analysis performed for Station W9 (Table 5.2-1). Results indicated that mean amphipod survival was significantly lower and less than or equal to 80% of the control for 4 out of the 32 stations (W5, W6, and W13, and V3) and significantly lower for an additional 5 stations (W9 field duplicate, W11, W12, V1 and V4). The field duplicate analyses gave very similar results (87 as compared to 95% of the control).

Thus, the primary stations indicating significant toxicity occur mainly in intertidal and marsh areas near the southern and northern ends of the landfill and at Calf Pasture Point. No significant toxicity was observed for any offshore stations.

5.2.2. Sea Urchin Test Results

Sea urchin tests (2 replicates each) were conducted on surface sediment pore waters from 14 intertidal (W1-W14), 14 subtidal (D1-D14), and 4 vegetated wetland stations (V1-V4), with additional laboratory replicate analyses performed for Stations D2 and D3 (Table 5.2-1). Results indicated that mean fertilization percentage was

significantly lower (defined as less than 50% of the control) for 3 (D1, D13, and W6) out of 32 stations. Further, for conservative reporting purposes, an additional 3 stations (D2, D10, and W8) had mean fertilization percentages that were greater than 50 but less than or equal to 70%. Duplicate analyses gave very similar results, indicating a high degree of precision (Table 5.2-1). Thus, the primary stations indicating significant or potential toxicity occur mainly in intertidal and shallow offshore areas near the southern and northern ends of the landfill and at Calf Pasture Point.

Overall conclusions from the combined amphipod and sea urchin tests indicated generally similar results: the principal stations indicating toxicity corresponded to southern and northern landfill areas and Calf Pasture Point. The primary difference is that 2-4 offshore stations were associated with significant to potential toxicity based on the sea urchin test, but no offshore stations exhibited significant toxicity from the amphipod test.

For offshore sediments, the results indicating non-significant (amphipod test) to minimal (sea urchin test) toxicity are consistent with amphipod (*Ampelisca*) test results from the Phase I study (Munns et al. 1991). Further, Phase I results demonstrating significant mortality associated with the landfill also are supportive of the present conclusions that there was significant toxicity from intertidal and shallow offshore sediments near the ends of the landfill. These data further define the extent of contaminant effects by filling in sampling gaps between landfill and offshore stations studied during Phase I.

5.3 Biological Field Investigations

5.3.1. Fish Distribution and Abundance

Fish samples (*Fundulus* spp.) were collected at 10 stations (W1-W10) using traps and seines for evaluation of any significant condition (length/weight ratio;

Table 5.3-1). Two species of *Fundulus*, the mummichog (*F. heteroclitus*) and the striped killifish (*F. majalis*), and potentially some hybrid individuals, were represented in the samples. For the length/weight study, it was not practical to make taxonomic distinctions among the hundreds of individuals used for the analyses; thus, these data represent a composite of *Fundulus* spp. In contrast, the many fewer specimens required for the chemical analyses were differentiated into the three taxa and processed accordingly.

Mean lengths and weights of *Fundulus* varied between 35.8-52.7 mm and 0.9-3.9 grams, respectively, with the highest and lowest values for both parameters corresponding to Stations W6 and W5 (Table 5.3-1). The length/weight relationships are consistent with general population trends (primarily based on *F. heteroclitus*) documented by other studies (Kneib and Stiven 1978). Results of a one-way analysis of variance (ANOVA) on ranks (using Dunn's method to account for unequal sample sizes and non-normal distribution of the length/weight data) indicated that Station W5 was significantly different from all the other stations. However, careful inspection of Table 5.3-1 suggests that even though this station had the lowest overall mean values for both length and weight, the ANOVA probably was influenced by the relatively high variance (SD) for these data, thus, potentially biasing the results.

It is notable that other measurement endpoints for Station W5, such as the toxicity tests presented in Section 5.2, also suggest that this site is impacted by chemical constituents. The ANOVA results for the *Fundulus* length/weight data would be consistent with this hypothesis; however, inspection of the tissue metals data for these stations do not indicate increased residue concentrations for organics (Figure 4.2- 10 to Figure 4.2-13)) or in general for metals at Station W5. A possible exception may be a somewhat elevated zinc residue (218 ug/g) from this station. Relationships between CoC residues and impacts on biota will be addressed further in Section 6.6.

An alternate hypothesis to explain the ANOVA results is that the overall pattern among the stations represents natural differences in fish year classes as influenced by habitat preferences. Specifically, the data in Table 5.3-1 also can be divided into two groups, Stations W1-W5 and W10 (mean size range from 35.8-40.7 mm) and Stations W6-W9 (mean size range from 46.3-52.7 mm) that may correspond to these natural trends. Kneib and Stiven (1978) noted that mummichogs grow to average lengths of approximately 35 mm during their first year and 50 mm during their second year. Further, it is known that smaller individuals tend to prefer more protected, shallow habitats associated with vegetation, thereby reducing predation potentials (Abraham 1985). It is notable that Stations W1-W5, representing the smaller mean size group, are located near the primary salt marsh area within the study region. In contrast, Stations W6-W9, representing the larger mean size group, are situated in an area of fringing sand flat. The relationship of Station W10 to these potential habitat differences is not evident. Nonetheless, if it is assumed that the relatively smaller fish from Stations W1-W5 represent year one individuals and that Stations W6-W9 are associated with year two fish, at least part of the differences between these groups may be influenced by natural habitat preferences.

Uncertainties associated with this hypothesis are related to the incorporation of three *Fundulus* taxa into the length/weight analyses. These taxa are known to have somewhat different maximum sizes (Abraham 1985), although field notes from the present study suggest that, if anything, there were more *F. majalis* (typically representing larger, maximum-sized individuals) at the stations associated with smaller mean lengths. This is a pattern opposite to that which would be expected if disproportionate numbers of *F. majalis* were biasing the present results.

5.3.2. Hematopoietic Neoplasia Assessment

Hematopoietic neoplasia has been reported world-wide in field populations of 15 marine and estuarine bivalve mollusc species (Peters 1988). These include several

commercially important species, such as the soft shell clam (*Mya arenaria*) eastern oyster (*Crassostrea virginica*) and blue mussel (*Mytilus edulis*) (Harshberger *et al.* 1979, Brousseau 1987, Elston *et al.* 1988). The disease is a proliferative blood disorder that is progressive, fatal, and transmissible (Elston *et al.* 1988). It can be diagnosed by histopathological examinations or using histocytological techniques (Cooper *et al.* 1982). Several studies indicate that this disease may be caused by contact with viruses, exposure to environmental constituents or stressors, or from the synergistic effects of several etiologic agents (Peters 1988).

Previous evaluations of soft shell clams from Allen Harbor have indicated the endemic occurrence of this disease (Cooper *et al.* 1982, Walker *et al.* 1981, Munns *et al.*, 1991). In 1992, percentages of neoplasia greater than 5% were found at 8 out of 11 stations less than 5 km from the landfill and disposal site and the highest percentages were noted from stations less than 10 km from Allen Harbor (Munns *et al.* 1992).

Assessment of hematopoietic neoplasia in soft shell clams were completed for 9 stations in Allen Harbor, including four stations south of the Allen Harbor Landfill, three to the north of the landfill, and 1 station at Calf Pasture Pt. For the present study, clams were collected from quadrants at each station The quadrants served as replicates for statistical analyses. Clams were processed and slides of tissues were prepared according to procedures described in Morrison *et al.* (1993) and in the standard operating procedure in the Master Work Plan.

One thousand cells per animal were examined microscopically. Three cell types were observed: normal cells with numerous pseudopodia, round regular nuclei, and abundant cytoplasm; abnormal but non-neoplastic rounded cells with pycnotic-like nuclei; and the enlarged, rounded neoplastic cells with large, hyperchromatic nuclei surrounded by little cytoplasm.

An analysis of the relationship between the % mean incidence and the variance of neoplasia in soft-shell clams collected in Allen Harbor was used to evaluate the underlying probability distribution. Statistical analyses were performed between stations to determine differences within Allen Harbor and to determine the difference between clams collected outside of Allen Harbor in upper and lower Narragansett Bay, RI and clams collected from within the harbor. Single-factor analysis of variance (ANOVA) was used to compare data between stations within the harbor. A one-way, unpaired students's t-test (alpha=0.05) assuming unequal variance was used to compare data from upper and lower Narragansett Bay with data from Allen Harbor. Data for Narragansett Bay were obtained from a previous study (Munns *et al.* 1992). Narragansett Bay (NB) stations were grouped into lower (LNB) and upper (UNB) bay designations and were used as reference sites for statistical calculations. Allen Harbor stations W1, W3, W4, W5, W6, W8, W9, W11, W12, W13, and W14 were grouped into four designations (LS-VW, LN-W, CP-W, and CP-SW) based on geographic- and depth-related characteristics. Groups LS-VW, LN-W, CP-W, and CP-SW within Allen Harbor were compared separately to the UNB and LNB reference areas.

Raw data are presented in Appendix D.4. Summary data and statistical analyses are presented in Table 5.3-2 and 5.3-3. In total, 187 clams from 11 stations in Allen Harbor were evaluated for neoplasia. Neoplasia was found at all but two of these stations (W9 and W13). Mean incidence of neoplasia at each station ranged from 11 to 100%. The highest incidence of neoplasia was found at Station W11, although only six animals from this station were examined. Moderate to low incidence were found in soft-shell clams examined from the remaining eight stations.

ANOVA calculations indicated significant differences between the stations. When data from eight stations from upper Narragansett Bay, RI from a previous study were grouped (UNB) and compared to data from grouped Allen Harbor stations (LS-VW, LN-W, CP-W, and CP-SW), statistical significance was observed between LNB and LS-VW (Table 5.2). In addition, grouped data from five stations in lower

Narragansett Bay, RI (UNB) obtained from this same study compared to data from grouped Allen Harbor stations (LS-VW, LN-W, CP-W, and CP-SW), also demonstrated statistical significance between LNB and LS-VW (Table 5.3-3) .

In summary, the results of this investigation support the conclusion of enhanced neoplasia incidence in the Landfill South Vegetated/Wetland Region (LS-VW) region of Allen Harbor which has been found to be statistically greater when compared to both Upper and Lower Narragansett Bay regions.

5.3.3. Bivalve Condition Indices

The health of bivalves under varying environmental conditions has frequently been assessed through measures of growth rate, condition index and survival rate (Brown and Hartwick, 1988). Condition indices based on allometric relationships were developed primarily for detection of the ecophysiological status of animals in an aquaculture setting (e.g. Lucas and Benninger, 1985), but has received expanded use in water quality monitoring programs (Lawrence and Scott, 1982). Mann (1978) and Lucas and Benninger (1985) recommended the use of dry tissue weight to dry shell weight ratio as the static index of choice. Low index values are interpreted as reflecting energy deficits resulting from environmental stress or loss of gametes. Another condition index, dry tissue weight to shell volume (calculated from length) ratio has also been proposed, since the proportion of internal shell body occupied by tissue should reflect the status (related to fitness) of bivalve metabolic reserves (Brown and Hartwick, 1988). In addition, the ratio of shell weight to shell length is useful as an indicator of shell thickness. Enhanced shell thickness is interpreted as an indication of stunted shell growth due to crowding or other environmental influences.

Three bivalve condition indices (CI), including Tissue Weight/Shell Length (CI-TW/SL), Shell Weight/Shell Length (CI-SW/SL), and Tissue Weight/Shell Weight (CI-TW/SW) were calculated from samples collected during March - April, 1995. The

data includes four bivalve species for inshore (wetland and vegetative) stations and one species (hard shell clam) at the offshore, deep stations. The sampling strategy included a randomized design, whereby four 1-m^2 quadrants were sampled quantitatively for the occurrence of the target species. Due to the natural variability in bivalve distributions, it was common for the same species to be found at only 2-3 of the 4 replicates from each station. Sample analyses for condition indices involved the selection of 10 individuals from each replicate such that the full size range was adequately characterized. The mean of CI values derived from each measurement was calculated and used as the CI index for the station. The Standard Error about the mean were calculated for each station. In addition, the overall median across stations is also presented as a point of reference for interpretation of station-specific responses.

The condition indices for each bivalve species are presented in Figures 5.3-1 to Figure 5.3-4. In Figure 5.3-1, each of the condition indices for soft shell clams indicate a potential adverse effect at Station W1 as evaluated relative to other harbor stations (soft shell clams at reference sites were not available).

In Figure 5.3-2, the CI-TW/SL and CI-SW/SL indices for ribbed mussels indicate potential adverse affects at Stations W8-W11, whereas the the TW/SW response suggest impacts at Station W5. In Figure 5.3-3, the CI-TW/SL and CI-SW/SL data suggests possible adverse effects to oysters at Station W6.
In Figure 5.3-4, potential trends in hard shell clam CI-SW/SL data are apparent at Stations W6, W7 and W13, a trend shared by CI-TW/SL data with additional suggestion of impacts at Stations W2, W3 and W5. Condition indices for hard clams at subtidal sites show reduction in SW/SL and TW/SL values for station D6. CI values observed for the PCC reference station are much higher for SW/SL and TW/SL , but much lower for TW/SW, suggesting environmental conditions for bivalve growth at this station are not representative of Allen Harbor conditions. To summarize, generally good agreement between the CI-TW/SL and CI-SW/SL methods were observed and

most often suggest impacts at Stations W1, W5 and W6 and to a lesser extent, W11 and W13.

Interpretation of adverse impact based on CI data must be considered carefully in context with constituent exposure concentrations, as well as other potentially confounding factors such as temperature, food supply, and substrate type. For example, it is difficult to discern differences based on reference site data, because of potential differences in water temperature and food supply. However, these factors are perhaps less important given that such water quality gradients (exclusive of anthropogenic factors) on these small scales (20-50 m) are unlikely to exist. Substrate type can be an important factor; Stations W1, W2 and W4 had low sand content relative to other Allen Harbor stations or the reference sites (Figure 4.2-1). However, this substrate effect would not explain differences among these three stations, such that other unmeasured factors, or possibly contaminant exposure, might be causative. The results must therefore be interpreted in conjunction with other exposure and effects indicators to determine if these data support or contradict the prevailing weight of evidence.

5.4 Existing Toxicity-Based Criteria and Standards

Toxicity-based criteria and standards provide the basis for comparing expected or actual environmental concentrations of constituents to toxicological benchmark concentrations, thereby allowing an estimation or quantification of risk. For the present risk assessment the primary benchmarks utilized are the ER-L values presented in Long et al. (1995); EPA Chronic Water Quality Criteria for marine waters, as adopted by the State of Rhode Island for regulatory purposes and compared in this study to the chemical concentrations in groundwater from the landfill; and EPA Marine Chronic Water Quality Criteria (WQC), as presently available, or as estimated from marine acute or freshwater values (discussed in Section 6.2).

The ER-L concentrations correspond to the lower 10th percentile of all concentrations of a contaminant observed to cause a biological effect, over a range of studies and species (Long and Morgan 1990; Long et al. 1995). Conceptually, ER-Ls are similar to LOELs (lowest observed effect levels), representing the lowest toxicant concentration observed in bioassays to cause biological effects. Another type of benchmark, AET (Apparent Effects Threshold; PTI 1988) was developed to address individual constituents in field sediments, and represents the level of individual chemicals above which statistically significant biological effects are always expected to occur. However, as discussed in the Phase I Risk Assessment (Munns et al. 1991), ER-L values are more conservative (i.e., correspond to lower benchmark levels) than AET values, typically representing concentrations that are an order of magnitude lower. Consequently, ER-Ls are utilized primarily for this assessment.

The EPA water quality criteria were used to calculate screening benchmarks for landfill groundwater samples collected during Phase I RAPS (Munns et al., 1991) and Phase II RI (TRC, 1994). For this evaluation, porewater concentrations of organics were estimated using equilibrium partitioning to allow application of water quality criteria and provide a different method of estimating CoC exposure to target receptors (discussed in Section 6.2).

5.5 UNCERTAINTY

Numerous assumptions concerning the applicability of 1) toxicity evaluations, 2) biological field investigations, and 3) particular benchmarks as criteria and standards to evaluate impacts to biota are made that bear upon the certainty of risk derived from these effects based measures. For the ecological effects assessment the following uncertainties should be noted:

Toxicity evaluations. The evaluation of ecological effects of contaminated sediments using toxicity tests is essential because chemical concentrations alone are

not accurate predictors of biological effects. The principal advantage of the sediment toxicity testing approach is that they are performed in a manner comparable to WQC derivation exercises, (e.g. mortality or sublethal effects are observed), hence the data are directly comparable to these criteria. Uncertainties associated with toxicity testing conducted in the present study is that the responses are not chemical-specific nor do the responses observed represent chronic effects. In addition, there is uncertainty in the comparability between the sediment test species and the water test species upon which the WQC are derived (EPA, 1989b). With regard to porewater tests, there is uncertainty as to the extent that chemical bioavailability is altered by the porewater extraction method.

Biological investigations. Field survey approaches, such as the soft shell clam neoplasia and condition endpoints measured in this study, have the advantage of providing assessments of *in situ* effects without significant sampling artifact. However, a large amount of field data is required, including both seasonal and spatial coverage, such that benthic impacts can be discerned.

Benchmarks. As summarized in Section 5.4, the derivation of ER-Ls is based on very conservative assumptions concerning use of the lower 10th percentile of all concentrations of a contaminant that have been observed to cause biological effects. ER-L values are lower by one order of magnitude for most parameters as compared to AET values, representing the only other effects-based benchmark that is commonly applied. The concern is the level of conservatism that is appropriate to assess ecological risk. ER-Ls are used in this assessment to provide a protective evaluation; however, this may not account for factors that can mitigate (buffer) the responses of ecological systems to particular constituents. However, the incorporation of toxicity data on effects levels to species from different communities and trophic levels is an attempt to add another measure of realism to the final assessment. A further issue with the use of ER-Ls (or AETs) is the relatively limited list of chemicals for which values are available compared to the overall list of constituents from the study.

Use of Surrogate/Indicator Species. Several species, including soft and hard shell clams, oysters, mussels, *Fundulus* spp., and two bioassay organisms (*Ampelisca* and *Arbacia)* are used as indicators of the assumed general response of the various communities within the study region. These species represent a variety of biological endpoints which have been shown to be sensitive to contaminant inputs and whose relationship to a particular habitat and community is well established. It also was important to maintain consistency in the use of the same species, for purposes of data comparability, among the various phases of the risk assessment. Nonetheless, the use of surrogate or indicator species is associated with some undefined level of uncertainty since single or a few species cannot duplicate precisely the response of numerous to hundreds of species that comprise the various communities of the region. However, the use of multiple independent measures (species) will provide appropriate results and insight to develop a complete and protective risk assessment.

Use of Mercury in the SEM Calculations. Calculations of SEM for use in comparisons with AVS values utilize sediment data on copper, zinc, lead, nickel, cadmium, and mercury. Each of these metals, except mercury, is commonly accepted as reacting appropriately in the presence of sulfides to fulfill the assumptions of the AVS paradigm. However, there is ongoing debate about the need to include mercury in the calculations. This is because mercury can react in a manner that is more similar to an organic compound than a divalent metal. For this assessment, mercury has been included in the analysis; although, inspection of the data indicate that, based on generally low concentrations, its incorporation has little effect on the final SEM/AVS ratios. Further, the inclusion of an additional metal primarily serves to increase the ratio value, thereby representing a more conservative effects measure.

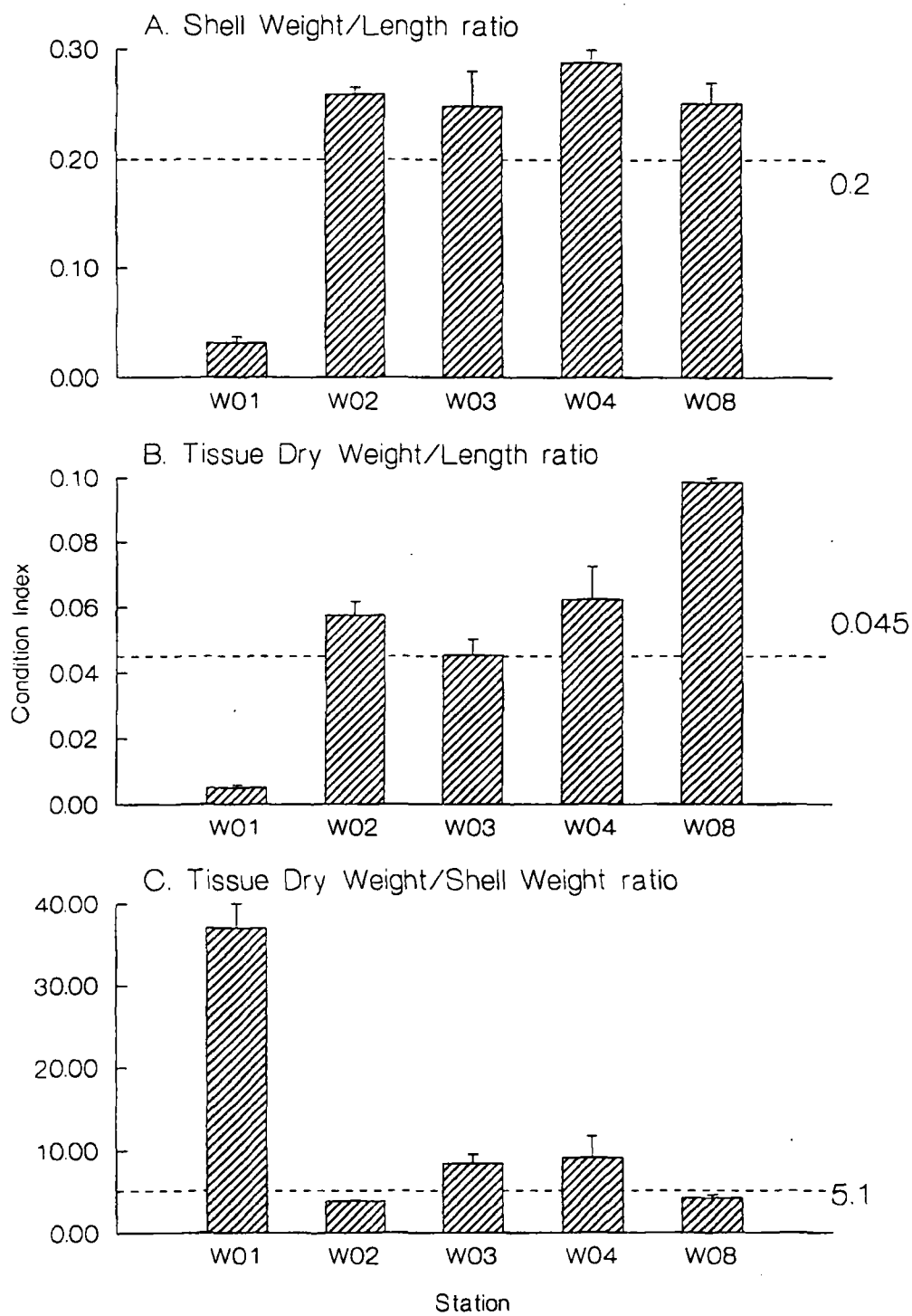


Figure 5.3-1. Condition indices for soft shell clams in Allen Harbor. A) shell weight/length ratios, B) tissue dry weight/length and C) dry tissue weight/shell weight ratios. Dashed line indicates median across all stations. Values are mean ± the Standard Error of the mean.

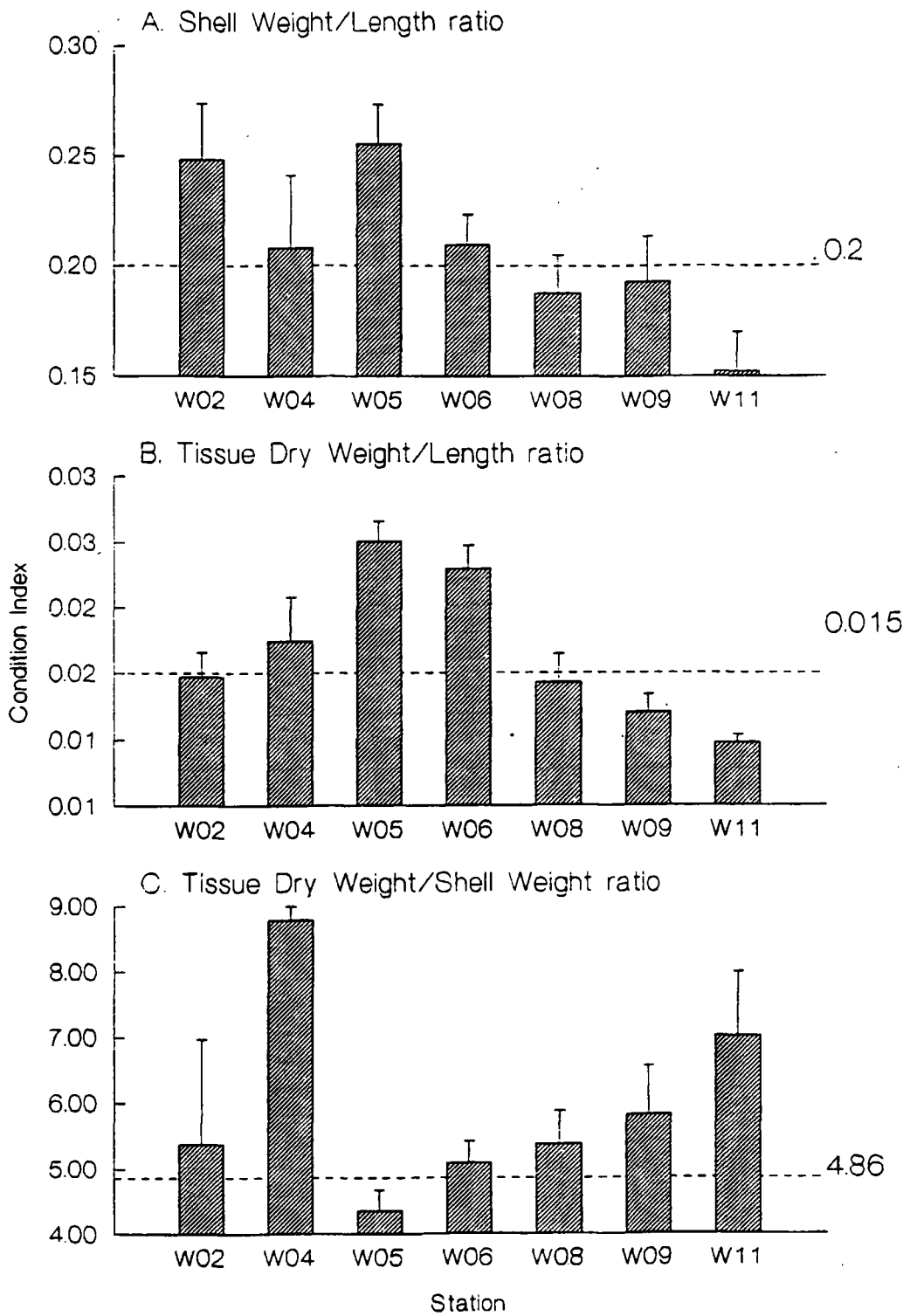


Figure 5.3-2. Condition indices for ribbed mussels in Allen Harbor. A) shell weight/length ratios, B) tissue dry weight/length and C) dry tissue weight/shell weight ratios. Dashed line indicates median across all stations. Values are mean ± the Standard Error of the mean.

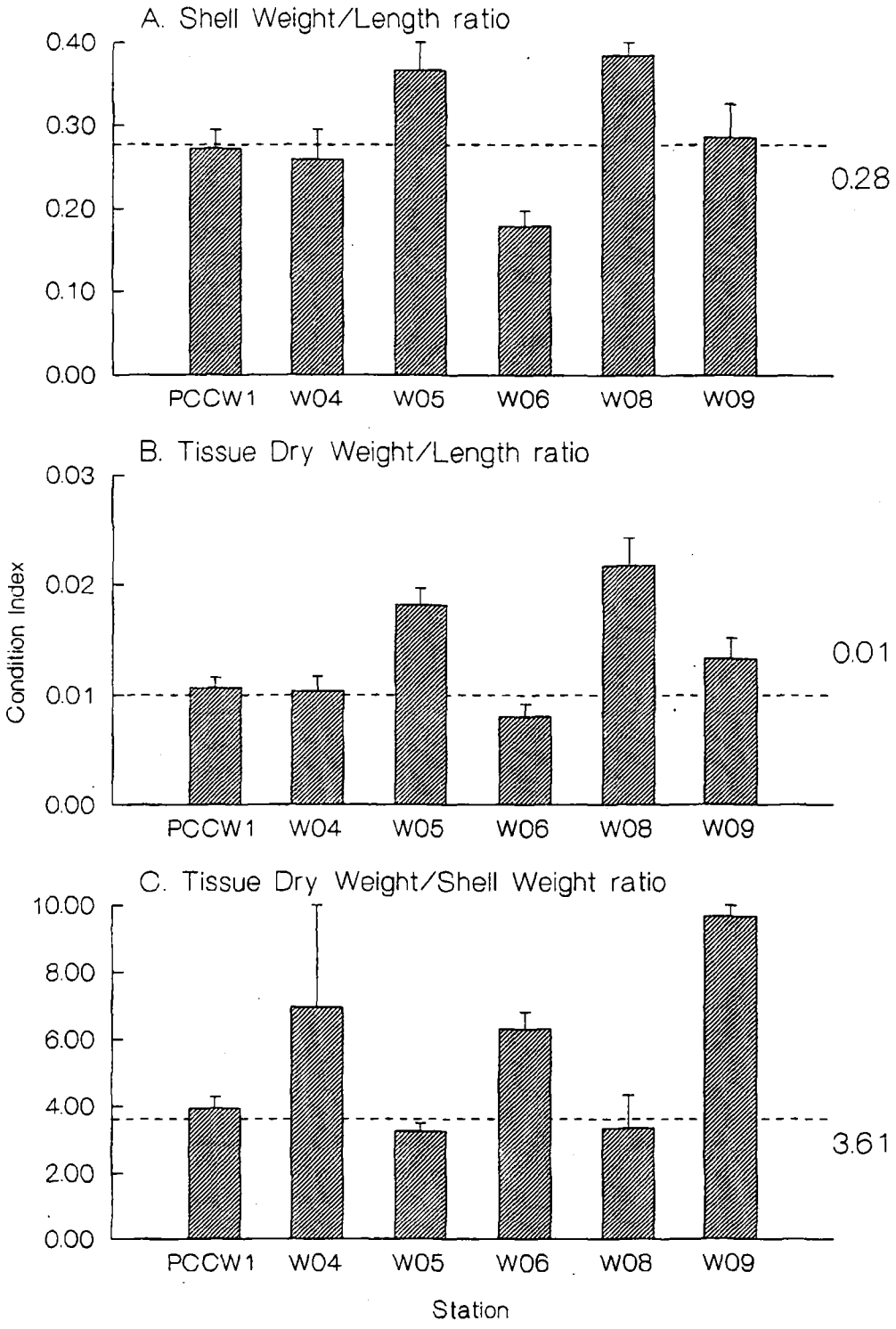


Figure 5.3-3. Condition indices for oysters in Allen Harbor. A) shell weight/length ratios, B) tissue dry weight/length and C) dry tissue weight/shell weight ratios. Dashed line indicates median across all stations. Values are mean ± the Standard Error of the mean.

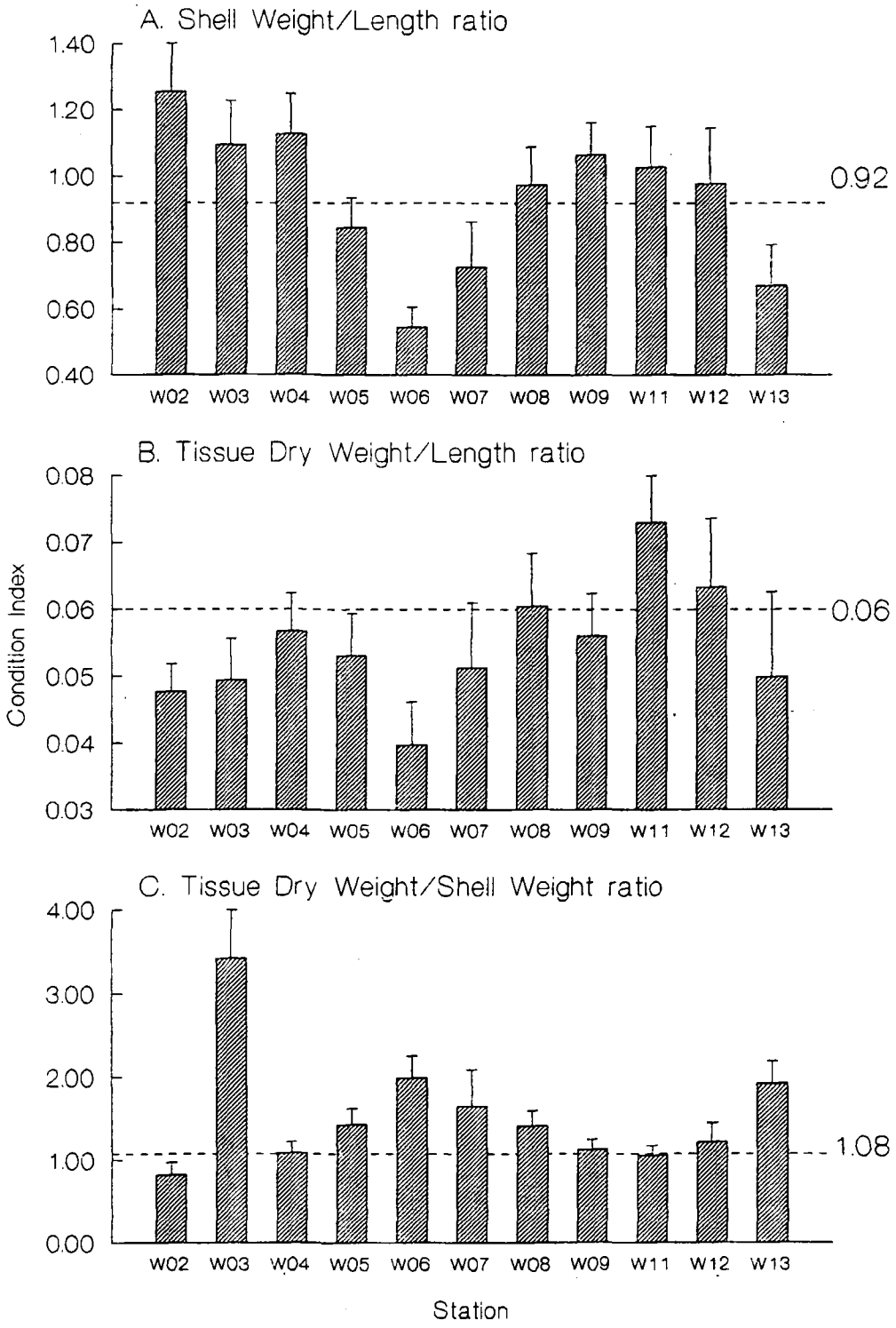


Figure 5.3-4. Condition indices for hard clams in Allen Harbor. A) shell weight/length ratios, B) tissue dry weight/length and C) dry tissue weight/shell weight ratios. Dashed line indicates median across all stations. Values are mean ± the Standard Error of the mean.

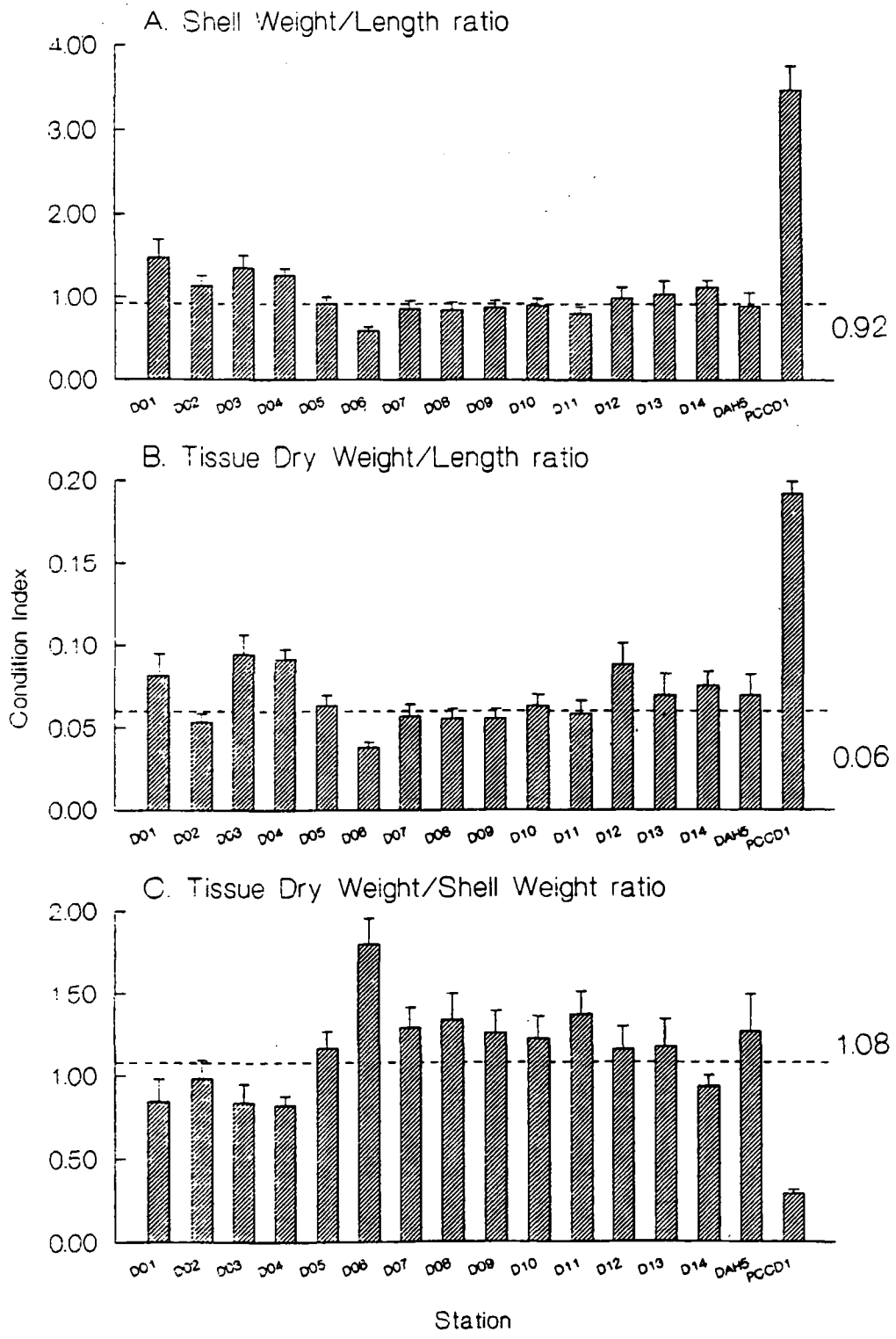


Figure 5.3-4. continued

5.2-1. Toxicity results using the amphipod (Ampelisca abdita) sediment survival and the sea urchin (Arbacia punctulata) porewater fertilization test.

Site	Location	FIELD ID	Amphipod Survival, (%)		Sea Urchin Fertilization, (%)	
AH	D	D01[1]	78.0	*	0.0	**
	D	D02	94.0		71.3	
	D	D02REP			70.0	
	D	D03	96.0		85.7	
	D	D03REP			90.3	
	D	D04	98.0		97.0	
	D	D05	96.0		96.9	
	D	D06	102.0		95.0	
	D	D07	97.0		88.8	
	D	D08	96.0		97.4	
	D	D09	98.0		95.0	
	D	D10	97.0		67.0	*
	D	D11	97.0		96.4	
	D	D12	99.0		91.3	
	D	D13	95.0		91.3	
	D	D14	97.0		94.9	
	D	DAH5	98.0		93.9	
REF	D	JSC-D1	99.0		98.5	
	D	PCC-D1	81.0		96.9	
AH	DC	DC01	100.0		99.0	
	DC	DC02	101.0		96.4	
	DC	DC03	100.0		94.4	
	DC	DC04	105.0		92.3	
	DC	DC05	98.0		92.8	
	DC	DC06	97.0		91.8	
	DC	DC07	102.0		95.9	
	DC	DC08	96.0		96.4	
	DC	DC09	100.0		95.4	
	DC	DC10	105.0		96.4	
	DC	DC11	102.0		96.9	
	DC	DC12	192.0		94.4	
	DC	DC13	93.0		0.0	**
	DC	DC14	98.0		97.9	
	DC	DCAH5	102.0		71.3	
REF	DC	JSC-DC1	98.0		100.0	
	DC	PCC-DC1	102.0		97.4	
AH	V	V1	86.0		97.9	
	V	V2	101.0		95.3	
	V	V3	60.0	*	97.4	
	V	V4	86.0		95.4	
REF	V	JSC-V1	95.0		98.5	
	V	PCC-V1	85.0		96.4	
AH	W	W01	89.0		99.0	
	W	W02	89.0		95.9	
	W	W03	99.0		96.4	
	W	W04	97.0		96.9	
	W	W05	63.0	*	98.4	
	W	W06	65.0	*	9.0	**
	W	W07	92.0		99.0	
	W	W08	93.0		68.0	*
	W	W09	87.0		95.4	
	W	W09FD	95.0		98.5	
	W	W10	98.0		95.0	
	W	W11	95.0		89.1	
	W	W12	89.0		97.0	
	W	W13	80.0		96.0	
	W	W14	96.0		93.0	
REF	W	JSC-W1	103.0		91.8	
	W	PCC-W1	94.0		98.5	

** Significant toxicity; defined as < 50% fertilization, < 60% amphipod survival.

* Slight toxicity; defined as between 50-70% fertilization, 60-80% amphipod survival

CODES: D = Deep, DC = Deep Core; V = Vegetated; W = Wetland

AH = Allen Harbor; REF = Reference Site

[1] Amphipod result not statistically ndifferent than the control

Table 5.2-2. Sediment porewater toxicity results using the sea urchin (Arbacia punctulata) fertilization index.

Site	Station	No. Replicates	% of Control Fertilized
AH	V1	2	98
AH	V2	2	95
AH	V3	2	97
AH	V4	2	95
AH	W1	2	99
AH	W2	2	96
AH	W3	2	96
AH	W4	2	97
AH	W5	2	98
AH	W6	2	9 **
AH	W7	2	99
AH	W8	2	68 *
AH	W9	2	95
AH	W9FD	2	98
AH	W10	2	95
AH	W11	2	89
AH	W12	2	97
AH	W13	2	96
AH	W14	2	93
AH	AH5	2	94
AH	D1	2	0 **
AH	D2	2	71
AH	D2 REP	2	70 *
AH	D3	2	86
AH	D3 REP	2	90
AH	D4	2	97
AH	D5	2	97
AH	D6	2	95
AH	D7	2	89
AH	D8	2	97
AH	D9	2	95
AH	D10	2	67 *
AH	D11	2	96
AH	D12	2	91
AH	D13	2	91
AH	D14	2	95
AH	DCAH5	2	71
AH	DC1	2	99
AH	DC2	2	96
AH	DC3	2	94
AH	DC4	2	92
AH	DC5	2	93
AH	DC6	2	92
AH	DC7	2	95
AH	DC8	2	96
AH	DC9	2	95
AH	DC10	2	96
AH	DC11	2	97
AH	DC12	2	94
AH	DC13	2	0 **
AH	DC14	2	98
JSC	V1	2	98
JSC	W1	2	92
JSC	D1	2	98
JSC	DC1	2	100
PCC	V1	2	96
PCC	W1	2	98
PCC	D1	2	97
PCC	DC1	2	97

** Significant toxicity; defined as < 50% fertilization

* Potential toxicity; defined as between 50-70% fertilization.

Table 5.3-1. Fish (*Fundulus* spp.) Length and Weight Data Collected from Allen Harbor.

Station	Mean Length	SD	Mean Weight	SD	Slope	Y-Intercept	R^2
W-1	40.7	7.9	1.9	1.4	0.173	- 5.17	0.91
W-2	39.5	7.4	1.6	1.3	0.162	- 4.76	0.80
W-3	37.7	6.8	1.3	0.8	0.110	- 2.85	0.83
W-4	38.2	6.6	1.5	0.9	0.129	- 3.46	0.83
W-5	35.8	4.9	0.9	0.7	0.134	- 3.74	0.76
W-6	52.7	15.0	3.9	2.8	0.178	- 5.45	0.89
W-7	46.3	11.5	2.8	2.1	0.179	- 5.48	0.96
W-8	48.9	13.8	3.2	2.7	0.191	- 6.11	0.94
W-9	51.3	12.6	3.6	2.3	0.178	- 5.57	0.92
W-10	39.4	7.0	1.3	0.7	0.090	- 2.28	0.90

Table 5.3-2. Summary results of Hematopoietic neoplasia (Hn) assessment in soft shell clams (Mya arenaria) collected from Allen Harbor and lower Narragansett Bay (LNB). Data for stations in LNB were obtained from Munns et al. (1992).

Group	Station	N	# of Affected Clams	% of Affected Clams	Mean %	SD	p value	Comment
LNB	MC	68	3	4.4	1.5	2.11		
	DC	36	0	0.0				
	BI	32	1	3.1				
	SC	24	0	0.0				
	PR	17	0	0.0				
LS-VW	W1	3	2	50.0	40.4	23.72	0.023	*
	W3	41	22	66.1				
	W4	18	2	10.3				
	W5	20	8	35.0				
LN-W	W6	16	8	63.7	31.8	31.85	0.121	
	W8	22	7	31.7				
	W9	4	0	0.0				
CP-W	W11	6	6	100.0	57.5	60.10	0.207	
	W12	20	3	15.0				
CP-SW	W13	19	0	0.0	13.3	18.86	0.269	
	W14	15	4	26.7				

*Statistically lower (alpha < .05) than the LNB treatment.

Table 5.3-3. Summary results of Hematopoietic neoplasia (Hn) assessment in soft shell clams (Mya arenaria) collected from Allen Harbor and upper Narragansett Bay (UNB). Data for stations in UNB were obtained from Munns et al. (1992).

	Station	N	# of Affected Clams	% of Affected Clams	Mean %	SD	p value	Comment
UNB	GP	6	0	0.0	8.5	9.22		
	OMC	20	1	5.0				
	BW	36	0	0.0				
	AC	36	0	0.0				
	MP	31	8	25.8				
	CC	31	5	16.1				
	TC	52	5	9.6				
	QD	18	2	11.1				
LS-VW	W1	3	2	50.0	40.4	23.72	0.035	*
	W3	41	22	66.1				
	W4	18	2	10.3				
	W5	20	8	35.0				
LN-W	W6	16	8	63.7	31.8	31.85	0.166	
	W8	22	7	31.7				
	W9	4	0	0.0				
CP-W	W11	6	6	100.0	57.5	60.10	0.227	
	W12	20	3	15.0				
CP-SW	W13	19	0	0.0	13.3	18.86	0.389	
	W14	15	4	26.7				

*Statistically lower (alpha < .05) than the LNB treatment.

6.0 RISK CHARACTERIZATION

This risk assessment follows the organization suggested in Eco Update (U.S. EPA, 1991) with appropriate elements from the EPA Region I Supplemental Risk Assessment Guidance for the Superfund Program (U.S. EPA, 1988) and Risk Assessment Guidance for Superfund, Volume II Environmental Evaluation Manual (U.S. EPA, 1989). These guidance documents recommend a "weight-of-evidence" approach to assessing potential ecological risks. The approach should be based on evaluation of constituent analytical data relative to environmental benchmarks, direct field and bioassay observations, and selected field and laboratory studies from the scientific literature.

In the following sections, site-specific risk indicators for Allen Harbor are presented in order to build the weight of evidence summary and conclusion of risk presented in Section 7. Each of the weights of evidence are discussed with regard to whether the observed ecological effect is expected to result from elevated exposure to the CoC. The lines of evidence which will be evaluated in drawing conclusions concerning risk are as follows (identified by Section):

Section 6.1. *Observed adverse effects - comparison with reference stations.* Comparisons of CoC chemical concentrations observed in Allen Harbor *versus* reference areas is presented to provide a site-specific evaluation of which areas in Allen Harbor have CoC concentrations that are elevated above background conditions;

Section 6.2. *Analysis of CoC concentration versus observed adverse effects.* Analyses are conducted to evaluate the relationships observed between measured CoC concentration and the quantified response of the measurement endpoint. For instance, if a particular CoC is causative in ecological impacts to a particular

receptor, then a change in the response of measurement endpoints associated with that receptor should be observed with increasing CoC exposure. Interpretation of these patterns are discussed with regard to whether the observed ecological effect is expected to result from elevated exposure to the CoC.

Section 6.3. *Analysis of bioaccumulation.* Elevated tissue residues in receptor species presented in Section 4.2 and 4.3 are interpreted as an indication that many CoCs are bioavailable and can potentially be transferred in other receptors through trophic interaction. This section provides analysis of observed bioaccumulation patterns in order to determine the degree to which specific habitat and or feeding preferences of target species are important in determining tissue concentrations, and thus the potential trophic transfer of CoCs to higher level food chain species (e.g., winter flounder and avian predators). Bioaccumulation factors are derived from the data to be used in food chain transfer estimates. This information is compared to the literature as a check on the validity of exposure pathway conclusions.

Section 6.4. *Analysis of toxicity versus observed ecological effects.* Results of toxicity tests conducted on sediments and porewater collected at field sites are compared with chemical concentrations measured at those sites. SEM/AVS ratios for divalent metals, and pore water equilibrium partitioning for non-ionic organic contaminants will be employed to assess availability of CoCs to ecological receptors. These data are used to elucidate those CoCs which appear responsible for observed toxicity.

Section 6.5 *Comparison of exposure point concentration with toxicity-based criteria and standards.* This analysis will involve calculation of exposure media-specific HQs and HIs using NOAA ER-Ls for sediments, and ambient water quality criteria for pore water and surface water. These data are also used to elucidate those CoCs which appear responsible for observed field effects and toxicity.

Section 6.6. *Residue-based Effects Assessment.* Tissue residue concentrations in target receptors will be evaluated against field measures of biota condition and suspected threshold concentrations for effects.

Section 6.7. *Uncertainty.* The uncertainties associated with the risk characterization components, and therefore with the entire site-specific ecological risk assessment are discussed and include identification of assumptions, remaining data gaps, and the limitations of the assessment.

It should be noted that not all lines of evidence need point to one or more CoCs as causative agents for risk to be presumed in association with that CoC. In this weight-of-evidence approach, it will only be necessary to have the preponderance of evidence suggest a causal relation in CoC-receptor pairings for risk to be concluded.

In order to organize the presentation of risk and provide a more useful framework for risk management, two zonation strategies are used to evaluate risk associated with COCs in Allen Harbor: Habitats and Ecological Exposure Zones (EEZs). Habitats define broad zonation categories (i.e., subtidal offshore habitats, vegetated wetlands, and intertidal wetlands) that are useful for determining risk to more mobile (especially terrestrial) species that might utilize all of Allen Harbor

habitats. EEZs represent a more finely divided zonation that is based on an understanding of the variability in composition and spatial distribution of chemicals found in sediments and tissues, as well as the general hydrographical and bathymetric characteristics of the harbor. EEZs will be useful in providing a better spatial definition of risks, especially to target receptor species with limited mobility. Figure (6.0-1) shows the nine EEZs that have been identified in Allen Harbor. These EEZs are:

- Landfill South - Vegetated and Wetland (LS-V/W); includes stations W1-W5, and vegetated stations V1 and V2.

- Landfill South - Deep stations (LS-D); includes the intertidal mudflat habitat to the southwest of the landfill which is influenced by the creek drainage from the South and includes stations D1-D3 and RAPS station AH13;

- Landfill Middle - Deep (LM-D); includes the deeper north central harbor area including sites D4-D10, RAPS stations AH1-AH5;

- Marina (MAR); includes the south and southeastern Allen Harbor area including the docks and piers as well as significant numbers of moored vessels;

- Calf Pasture - West (CP-W); - includes the western portion of Allen Harbor adjacent to Calf Pasture Point in which stations V3, W11-W12, and D14 are located in addition to RAPS stations AH10 and AH14 are located;

- Calf Pasture -South Wetland (CP-SW) - includes the intertidal environment south of Calf Pasture Point, containing stations W13, V4 and W14;

- Calf Pasture - South Deep (CP-SD) - includes the subtidal zone immediately adjacent to channel entrance area, characterized by sand bottom and swift currents. This zone contained stations D11, D12, D13 and RAPS station FDA;

- Reference - Includes reference sites at Jamestown Sheffield Cove (JSC) and Prudence Coggeshall Cove (PCC).

In each of the sections to follow, a semi-quantitative ranking scheme was developed in order to facilitate the utilization of results of each weight of evidence analysis in the overall risk assessment summary. The ranking strategy is intended to characterize relative degree of risk as indicated by the weight of evidence observation, such as the extent to which CoC concentrations exceed benchmark criteria (HQ>1 = slight risk (+), HQ>10 = moderate risk (++), HQ>100 = substantial risk (+++).

This type of ranking scheme is intended only as qualitative tool. The approach follows that taken by Munns et al. (1991) in the Phase I Allen Harbor ERA. In each case, the logic behind the ranking is presented and is specific to the particular weight of evidence discussed. Clearly, the ranking approach is (necessarily) somewhat arbitrary and is based on best professional judgement, since the "true" ecological risk of, for example, benchmark exceedence or observed toxicity, is not presently known. Hence, the risk manager is encouraged to evaluate the merits of the approach and test the effect of alternate ranking schemes on the general outcome of the risk evaluation.

6.1 Comparisons of Site and Reference Sediment CoCs

This section summarizes comparisons of CoC concentrations in Allen Harbor sediments with reference site sediments in order to characterize various areas of Allen Harbor with regard to extent of CoC contamination relative to background (e.g. reference) conditions. The analysis is not intended to characterize risk, *per se,* although in a qualitative sense, those areas which have grossly elevated chemical contamination relative to reference sites should be deemed high priority areas for detailed risk analysis. Conversely, areas which have levels of contamination below reference location conditions should be given lower priority.

Evidence of enrichment is based on concentration ratios (CRs) that compared sediment concentrations at Allen Harbor with those at the two reference sites. CRs were calculated for the various chemical groups, including PCBs, PAHs, VOAs, butyltins, and metals. The station-specific concentration ratio is defined as the ratio of the site-related concentration to the mean reference site concentration for the matching habitat type. PCBs were treated as a single analyte for this calculation, while for other chemical groups, CR values were individually calculated for each analyte. Station-specific concentration ratio data are reported in Appendix Tables A2-1.1 to A2-1.6. Detailed evaluation of data underlying the following CR discussion are presented in Sections 4.2 and 4.3.

For purposes of evaluating organic CoC enrichment compared to reference sites, total organic carbon-normalized data are considered more important than are the bulk concentrations because bioavailability to organisms as well as the affinity of sediments to sorb CoCs is related directly to the organic content of the sediments. Sediment concentrations for metals and butyltins were not normalized to TOC because the role of TOC in bioavailability of these CoCs is not well established. In order to facilitate the utilization of these results in the risk assessment summary, a qualitative ranking system was devised as follows:

Individual Rating Notation:

+ = Mean CR > 2;

++ = Mean CR > 2 and Max. CR > 10;

+++ = Mean CR > 10.

This ranking scheme is intended only as qualitative tool. The logic behind the ranking is that a mean two-fold elevation above reference is a enrichment factor of some concern; several stations which produce an average of two-fold enrichment and have a maximum CR > 10 is of moderate concern; and finally, a suite of stations which have pervasive, high contamination (mean CR > 10) is indicative or gross

contamination and is of substantial concern. In this analysis, CR values based on the MAR EEZ data are presented as the mean zonal concentration, but this zone is represented by one station only.

PCBs. Evaluation of CRs for total PCBs by EEZ suggest that PCBs may be of substantial concern in LS-V/W and LN-W zones, of moderate concern in the LM-D and MAR EEZs (Table 6.1-1). In contrast, there would appear to be relatively little concern in the LS-D EEZ or at Calf Pasture Point with respect to PCB contamination.

Pesticides. Pesticide CR values were highest at the CP-SD and LM-D EEZs, and are approximately twice the CR values observed at in the CS-W/V, CP-W, and MAR EEZs (Table 6.1-1). Elevated pesticide CR values in these "high" groups are driven by concentration data for two analytes, HCB and p,p'-DDE.

PAHs. PAH CR values indicate that high molecular weight (HMW) PAHs are of significant concern as LS-D, LM-D and CP-SD EEZs (Table 6.1-1). The Marina EEZ, although based on a single station, is also notable for HMW PAH contamination. CR values for the CP-W EEZ also suggests some concern for HMW PAHs. CR values for low molecular weight (LMW) PAHs follow the above trends but exhibit reduced enrichment. One notable exception is the maximum value observed for the LS-V/W EEZ, which is higher than the respective HMW PAH CR value. Total PAHs (tPAH), when considered collectively, still suggest significant enrichment at LS-M and LM-D, and to a lesser extent, LS-V/W EEZs. It is notable that little apparent enrichment of PAHs is observed at the LN-W and CP-SW EEZs.

VOAs. Concentration ratios for selected VOAs in sediments are summarized in Table 6.1-1. Some concern for VOA enrichment is apparent only at the LN-W and CP-W EEZs. The max CR value was observed in the LN-W EEZ was driven by a high chlorobenzene concentration observed at station W6.

Metals. Evaluation of metal CR values by EEZ showed a large degree of polarization between zones (Table 6.1-1). All EEZs had CR values suggesting at least some enrichment. However, the LS-V/W, LM-D and CP-W CR values suggest substantial metal contamination has occurred. These values were approximately three- to four-fold greater than the values observed in other zones, and are attributed to elevated CR values for arsenic, copper, lead, and zinc (Table A2-1.6).

Organotins. Evaluation of organotin CR values indicate that the LS-D, LM-D, CP-SD and MAR EEZ can be considered to exhibit slight enrichment (Table 6.1-1). In contrast, all other EEZs can be concluded to be unenriched with respect to organotin contamination.

Table 6.1-2 presents a summary table of the above results. A cumulative rating scheme was devised so as to facilitate description of the findings and carry forward the results to the weight of evidence evaluation in Section 7. Scoring of each individual ranking was based on '+' =1 pt., '++' = 3 pts. and '+++' = 5 pts, following the logic that additional weight should be given to environmental conditions (i.e., EEZ - chemical contamination characteristics) of significant and/or substantial CoC enrichment. Following this scheme, scores were summed across chemical classes to evaluate general enrichment trends. Results were then qualified following the scheme: 4-10 pts. = slight risk (+), 11-20 pts. = moderate risk (++),and > 20 pts. = substantial risk (+++).

The overall summary statistics tentatively suggest that the LM-D zone is most impacted, followed by the LS-V/W, LS-D, LN-W and CP-W EEZs which can be considered moderately contaminated. In contrast, the CP-SW, CP-SD and MAR EEZs would appear to have relatively little enrichment. Having stated these tentative conclusions, it is important to note that these findings are not self-supporting, but rather must viewed in context of other corroborative risk characterizations presented in the following sections.

6.2 Analysis of Sediment CoC Concentration versus Effects Measurements

This section evaluates the relationships between sediment CoC concentrations and specific effects measurement endpoints. The analysis focuses on potential exposure-response types of relationships, for example, whether increased levels of contamination in sediments are associated with increased effects to biota. For this assessment, measurement endpoints that are evaluated relative to sediment and porewater constituents include the following:

- Condition (dry tissue weight to length) of the bottom-dwelling fish, *Fundulus* spp.;
- Condition (dry tissue weight to shell length and weight ratio) of shellfish including ribbed mussels, oysters, hard shell clams, and soft shell clams;
- Incidence of hematopoietic neoplasia in soft shell clams;
- Correspondence of these measures with that of previous studies in Allen Harbor and in the primary literature.

In the sections below, results of Pearson correlation analyses (and associated probabilities of regression statistical significance) of relationships between the above effects-based measurements and 1) CoC concentrations in sediments and porewater and 2) Hazard Quotients derived from CoC concentration normalization to NOAA ER-L and Water Quality Criteria - Salt Water Chronic are presented to systematically evaluate potential CoC-based risk drivers. The comparison of organic CoCs in porewater against appropriate effects-based benchmarks (e.g. Water Quality Criteria-Saltwater Chronic (WQC-SC) values) is limited by a lack of available criteria; there presently exists only three PAH and four VOC benchmarks (see Table 3.3-2). To provide a more comprehensive screening using this approach, water-based CoCs are

proposed for calculation following the decision tree presented in Figure 6.2-1. This approach allows for calculation of "WQC-SC equivalent" benchmarks from Table 3.3-2, and assigns a data qualifier to identify the benchmark source for derivation of the Hazard Quotient.

The data qualifier (DQ) "A" is applied to HQs derived from porewater concentrations normalized to existing WQC-SC values. For CoCs possessing saltwater acute values, an 8:1 acute:chronic ratio is applied to derive the equivalent SC value (DQ = "B"). The conversion factor was derived from the mean overall acute:chronic ratio for paired chemical data contained in the EPA AQUIRE database (C. Stephan, EPA ERLD, pers. comm, Shepard, 1995). Freshwater chronic data are used directly as screening values, with assigned data qualifier "C". As with SA values, freshwater acute (FA) values were converted using a 8:1 acute:chronic ratio, and assigned DQ = "D". Finally, sediment criteria (NOAA ER-L criteria values were used as calculated from EqP assuming 1% TOC for all stations (DQ = E).

Predicted pore water concentrations (C_p, µg/L) for PCBs, PAHs, pesticides and VOAs are calculated from the corresponding sediment concentration (C_s; µg/kg) according to the equation:

$$1) \qquad C_p = C_s/(f_{oc} * K_{oc});$$

where f_{oc} is the fraction of organic carbon in the sediment (e.g. %TOC/100) * and K_{oc} = the organic carbon/water partitioning coefficient for the chemical of concern. Values for K_{oc} are determined from the relationship determined by Karickhoff (1989):

$$2) \qquad \log_{10}K_{oc} = 0.00028 + 0.983\log_{10}K_{ow};$$

where K_{ow} = the octanol/water partition coefficient. A summary of $\log_{10}K_{ow}$, $\log_{10}K_{oc}$, and corresponding K_{oc} values are summarized in Appendix Table A1-3. Station-specific

TOC and sediment CoC concentrations are reported in Appendix A1. Resultant porewater concentrations are reported in Tables A1-3.2 to A1-3.5.

6.2.1. Exposure - Biota Condition Relationships

Results of Pearson correlation analyses (r) of biota condition vs. sediment CoC concentrations are presented in Appendix A.3. For the present discussion, relationships were considered of potential ecological relevance only if three conditions were met; 1) non-zero correlations were obtained ($P>F \leq 0.05$); 2) observed correlations were negative (e.g., increasing CoC concentration was inversely related to biotic condition), and 3) the correlation explained at least 50% of the variation in the relationship ($r < -0.70$; $r^2 > 0.5$). In addition, comparisons of CoC Hazard Quotients (HQs) vs. observed effects were performed but limited to cases where CoC concentrations exceeded benchmark (e.g., HQ > 1). Finally, Hazard Indices were calculated as the sum of HQs for each compound class (metals, PCBs, PAHs, pesticides, volatile organics and organotins) and correlated against biotic condition.

6.2.1.1. Sediment Exposure - Biota Condition Relationships

Metals. No indications of potential adverse ecological effects from sediment metals were indicated from observed correlations of biota condition and sediment total metal concentration (Appendix Table A.3-1.1A). Comparison of CoC-specific Hazard Quotients with biota condition did suggest possible Hg effects on hard clam condition (tissue dry weight/shell wt), however Hg values are at or near detection such that the relationship appears spurious (Table A.3-1.1B). A second relationship was observed between arsenic HQs and soft shell clam (Mya) condition (tissue dry weight/shell wt). Here, the relationship is driven primarily by low condition and high arsenic at Stations W1, W2 and W4.

No indications of adverse ecological effects were observed from relationships of biota condition and Simultaneously Extractable Metal (SEM) (Table A.3-1.1C), or excess SEM metals as determined by SEM/AVS or SEM-AVS (Table A.3-1.1D). One indication of potential exposure-response was observed for *Fundulus* spp; station W5 was associated with substantially higher SEM than the rest of the stations, and had somewhat smaller-sized individuals (mean of approximately 2.5 mm less than any other station (Figure 6.2-2). This may represent an effect from the higher SEM, although, as noted in Section 5.3, the size pattern also could be related to natural variability and habitat preference.

PAHs. Among the five target species evaluated in this study, only one species, *M. arenaria* (MYA), exhibited potential exposure-response relationships between concentrations of PAH analytes and biotic condition (tissue dry weight/length) of potential ecological relevance (Table 6.2-1; Table A.3-1.2A). Here, several analytes exhibited strong correlations (r ~> 0.90) including acenaphthene, anthracene, fluorene, and phenanthrene, as well as low molecular weight PAHs. Each relationship was substantially influenced by high CoC values found at Station W1.

Potential exposure-response relationships were observed between hard shell clam condition and HQs for acenaphthene (r = -0.82) fluorene (-0.93), and phenanthrene (-0.85) (Table 6.2-1; Table A.3-1.2B), with the fluorene relationship shown in Figure 6.2-3. Potential exposure-response relationships were also observed between soft shell clam condition and HQs for acenaphthene (r = -0.96) and chrysene (-0.99), based on four and three data points, respectively. The maximum HQ was about 4 for chrysene, but near 20 for acenaphthene, the latter of which would appear more plausible CoC risk driver given the higher exposure quotients and spatial distribution.

PCBs. A single potential exposure-response relationship was observed between the PCB congener PCB-128 and soft shell clam dry weight/length ratio (Table

6.2-1, Table A.3-1.3B). As with the PAH relationships discussed above, this relationship was strongly driven by the Station W1 CoC concentration. PCB HQ relationships with biota condition revealed a potential exposure-response interaction between oyster shell weight/length ratio and total PCBs (Figure 6.2-4). The fact that lowest condition was observed at stations having total PCBs exceeding ER-L values by 55-60 fold (Stations W1 and W6, respectively) provides strong evidence that total PCBs may be a key risk driver for trends of reduced condition in this species.

Pesticides. Potential exposure-response relationships between pesticide concentrations and biota condition (tissue dry weight:length) were observed for aldrin with oysters (r = -0.77; Table 6.2-1, Table A.3-1.4A). The aldrin relationship was driven primarily by a single datum at W6,and was only slightly above the detection limit for this analyte (0.1 ng/g). Correlations for respective HQ - biota condition relationships failed to identify any significant interactions (Table A.3-1.4B).

VOAs. As with PAHs, the soft shell clam was the only species exhibiting does-response relationships with concentrations of selected VOAs (Table A.3-1.5A). With the exception of chloromethane (Table 6.2-1), all regressions were singularly influenced by the CoC concentration at station W1. The measured concentrations of the VOAs at this station were low (e.g. < 1 ppm). HQ correlations were not evaluated for this group due to a lack of ER-L benchmarks. However, porewater-based HQs for volatiles were evaluated and are discussed in Section 6.2.1.2, below.

6.2.1.2. Porewater Exposure Biota Condition Relationships

Metals. For porewater metals, the single relationship of potential ecological relevance occurred between Pb and tissue dry weight/shell weight for soft shell clams (Table 6.2-2, Table A.3-1.6). This relationship, unlike those observed previously for soft shell clams, was strongly influenced by a low Pb concentration at station W8 (1.5 ug/L); other stations exhibited similar Pb concentrations at 6-8 ug/L. No significant

porewater HQ- condition relationships were observed (Table A.3-1.6B). Hence, with the possible exception of Pb, porewater metals do not appear to be significant risk drivers for biota condition. No significant relationships between porewater-based HQs for metals measured in porewater and biota condition were observed (Table A.3-1.11).

PAHs. Unlike other target species, soft shell clams exhibited several strong potential exposure-response relationships between calculated porewater PAH analyte concentrations and tissue weight:shell weight ratio (Table 6.2-2, Table A.3-1.7). As had been observed for sediment concentration data, significant correlations were observed for both low and high molecular weight PAHs (r = -0.92 and -0.96, respectively) and dibenz(a,h)anthracene (r = -0.98). Identical correlations were observed for soft shell clam HQ - biota condition relationships since these CoCs were above benchmark criteria (Table A.3-1.11). The resultant HQ values for these analytes appear large relative to other WQC-based values principally because WQC-specific benchmarks are lacking for these analytes and hence were calculated from the ER-L value using the EqP model. However, despite uncertainty with regard to the degree of benchmark exceedence, the data provide clear evidence that these analytes are key CoC drivers for biota condition in soft shell clams. As with the raw concentration results, significant relationships between porewater-based HQs for PAHs and biota condition were observed for soft shell clams only , including dibenz(a,h)anthracene (r = -0.98), low molecular weight PAHs (r= -0.92) and high molecular weight PAHs (r = -0.96) (Table 6.2-2 and A.3-1.11). The relationship for high molecular weight PAHs is shown in Figure 6.2-5.

PCBs. Several calculated porewater PCB constituents were strongly correlated with reduced condition of soft shell clams (Table 6.2-2, Table A.3-1.8A). With the exception of PCB195, the analytes which tended to exhibit potential exposure-response were mid-molecular weight congeners (PCB101, PCB118, PCB128, and PCB138). Total PCBs were also inversely correlated with biota condition. The

toxicological significance of these results will be further evaluated in Section 6.5. In contrast to soft shell clams, no other target species exhibited potential exposure response relationships between calculated porewater PCBs and condition. In addition, no significant relationships between porewater-based HQs for total PCB and biota condition were observed (Table A.3-1.11).

Pesticides. Statistically significant correlations between porewater PCB concentrations and condition of hard clams, oysters, ribbed mussels and fish were not observed (Table 6.2-2, Table A.3-1.9). A statistically significant correlation was observed for o,p'-DDE, however, the input data were at detection limit in four of five cases, hence true exposure-response relationship for this analyte is unlikely. No significant relationships between porewater-based HQs and biota condition were observed (Table A.3-1.11).

Volatiles. Significant correlations were observed between calculated porewater volatile concentrations of benzene, chloromethane and toluene and biota condition indices for ribbed mussels (Table 6.2-2, Table A.3-1.10). However, none of these VOAs were found to occur at concentrations exceeding Water Quality Criteria (Table A.3-1.11).

6.2.2. Exposure - Neoplasia Incidence Relationships.

Incidence of hematopoetic neoplasia (Hn) in soft shell clams was regressed against CoC constituents in order to elucidate key drivers of this biological response. Results of Pearson correlations and associated probabilities of Hn response vs. CoC concentration for sediment and porewater. As with the biota condition data, relationships were considered of potential ecological relevance only if three conditions were met; 1) non-zero correlations were obtained ($P>F \leq 0.05$); 2) observed correlations were positive (e.g., increasing CoC concentration was proportional related to neoplasia incidence), and 3) the correlation explained at least 50% of the variation

in the relationship ($r < -0.70$; $r^2 > 0.5$). In addition, comparisons of CoC Hazard Quotients (HQs) vs. observed effects were performed but limited to cases where CoC concentrations exceeded benchmark (e.g., HQ > 1). Finally, Hazard Indices were calculated as the sum of HQs for each compound class (metals, PCBs, PAHs, pesticides, volatile organics and organotins) and correlated against biotic condition.

Results almost universally show a lack of significant correlations among all sediment and porewater constituents for raw data and Hazard Quotients, respectively (Appendix Tables A.3-2.1 and A.3-2.2). One exception was acetone; correlations in sediment and as calculated for porewater were 0.82 and 0.72, respectively. The measured sediment concentrations of this VOA at Station W1 and W11 are relatively high (370 and 680 ng/g, respectively), and drive the correlation; incidence of Hn was 67% and 100% at these stations. Whether this relationship represent true exposure-response is uncertain as the corresponding sediment values are qualified as being only estimates of the true value (Table A1-1.4). Summaries of correlations and associated probabilities of Hn response vs. HQ values for sediment and porewater are contained in Appendix Tables A4-2.1 to A4-2.4. No correlations of note were observed.

6.2.3. Exposure - Response Relationships from Prior Studies.

In the Risk Assessment Pilot Study (RAPS), Munns et al., (1991) reported that proximity to the landfill had no discernable effect on hard shell clam length or condition (tissue wet weight/shell length ratio); reduced condition and length was noted at Allen Harbor Station AH8 where high sediment PCB concentrations were observed, while highest condition indices were found at stations AH1 and AH13 near the landfill (see Figure 3.3-1). RAPS Stations AH1 and AH13 most closely correspond to Stations D6-D7 and D2-D3 of the present investigation, respectively (see Figure 3.6-1). It is difficult to cross-correlate the results across studies because of uncertainties in

sampling locations; particularly because the identified locations represent zones of greatest variation in station-to-station condition indices for this species (see Figure 5.4-5). However, the present results do agree with those previously presented to the extent that Allen Harbor hard shell clams do have reduced condition relative to reference locations.

Munns et al. (1991), also report no apparent impacts on soft shell clam condition in relation to the proximity of the landfill; the stations sampled (AH12, AH13 and AH14) however were distal to the landfill and widely separated spatially. In the present study, apparent reductions in soft shell clam condition were apparent only in the intertidal zone immediately adjacent to the southern bank of the Allen Harbor Landfill. Hence it is not unexpected that more distant stations do not reflect landfill-associated effects. Considerable seasonality was observed in the RAPS study as would be anticipated given reproductive effects on body weight due to gamete development. It is unclear whether trends observed in the present study would persist through seasonal reproductive events.

Incidence of hematopoetic neoplasia in soft shell clams was studied in the RAPS study in Allen Harbor, and included stations FDA, AH12, AH13 and AH14. These stations most closely correlate spatially with stations W12-W13, W8, W3-W4 and W12 of the present study. Relatively good agreement between the studies was observed; our Station W12-W13 had 0-15% Hn incidence vs. 12-27% at FDA, our W8 had 32% vs. 22% AH12, and our W3-W4 had 11-54% vs. 0-23% at AH13, and finally, our W12 had 15% vs. 11-14% at AH14. These results suggest there are persistent patterns of neoplasia throughout Allen Harbor. Subsequent evaluations of Hn incidence in Allen Harbor did not consider CoC specific relationships with disease prevalence, nor were VOA concentrations in sediments measured during any phase of the study.

6.2.4. Exposure - Response Summary.

A qualitative summary of observed biota condition effects discussed above are presented in Table 6.2-3. Measurement endpoint scoring was based on the extent of the change observed (including magnitude of change and spatial distribution) and the statistical significance of the observed exposure-response relationship supporting a hypothesis of cause and effect. Endpoint responses eliciting either an apparent response without exposure-response evidence are scored as "+" for slight risk, while endpoint responses which are supported by strong statistical evidence of exposure - response are scored as "++" (moderate risk). Hard clam and soft shell clam Condition Index (CI) responses, shown in Figures 6.2-3 and 6.2-5 show strong exposure-response relationships and generally indicate that populations living in the Landfill South Vegetated/Wetland are at risk. Oyster (Figure 6.2-4), ribbed mussel, fish (Figure 6.2-2) and soft shell clam (Figure 6.2-6) neoplasia exposure - response relationships were less apparent, being driven by a single datum or were not in spatial agreement.

Based on the above findings, bivalve communities occupying the LS-V/W EEZ, appear to be at moderate risk, whereas a slight risk is presumed for LN-W and CP-W EEZs. This conclusion is based primarily on the magnitude of the responses and degree of agreement among measurement endpoints. The results will be carried forward to the risk assessment summary in Section 7.

6.3 Analysis of Bioaccumulation

This section addresses the relationships between contaminant exposure and tissue residue concentration. The relationships are presented in a framework intended to define the essential operative transport and fate mechanisms for CoCs from the source to the target receptors (outlined in Section 3). In this assessment, the goal is to identify potential differences in chemical exposure to target species which might be caused by differences in habitat or feeding type. For organics, exposure pathway

differences were evaluated through the use of Biota-Sediment Accumulation Factors (BSAFs), while for metals, Biota Accumulation Factors (BAFs) were developed.

BSAF and BAF analyses discussed below include non-depurated data only in order to more fully characterize the food chain transfer to consumers of target receptors which would occur by whole body consumption. The importance of sediment gut content on the accumulation factor is addressed in a separate section in order to more accurately determine the true equilibrium condition between sediment and tissue, and hence better compare transport pathways among target receptors without interference of sediment associated with gut contents.

6.3.1. Organic CoC Bioaccumulation

Preliminary analyses of bioaccumulation involved the comparison of tissue CoCs for target species with co-located sediment CoC concentrations by performing linear regressions and determining correlation coefficients (r^2) for representative organic constituents (total PCBs, total PAHs, p',p-DDE). Several combinations of species and type of CoC were evaluated using this approach; (Figure 6.3-1). Correlations implying the strongest relationships (i.e., represented by the highest r^2 values) were for soft shell clams and DDE (r^2= 0.93; Figure 6.3-1A) and PAH (r^2= 0.65; Figure 6.3-1B), hard shell clams and PCB (r^2 = 0.60; Figure 6.3-1C); and ribbed mussel and DDE (r^2 = 0.50; Figure 6.3-1D) and, based on log conversions because of the large range of data, PAH (R^2 = 0.85; Figure 6.3-1E). The remaining correlations, including all the oyster data, were very weak (r^2 = 0.05-0.30), suggesting no obvious potential exposure-response relationships. Some of the strong correlations are influenced by discrete "clusters" of data near the low and high ends of the concentration range, and may represent a statistical bias rather than a potential exposure-response trend.

The above results demonstrated that the correlation analyses approach was not effective in elucidating exposure uptake relationships. This failing was not unexpected as it inherently lacks consideration of the operative factors controlling bioaccumulation for organic contaminants. EPA's Equilibrium Partitioning theory (EqP) for non-ionic organic contaminants hypothesizes that the chemical bioavailability from sediments to organisms is controlled by the chemical activity between the carbon fraction of the sediment and the lipid fraction of the organism (Di Toro, et al. 1991). For the present investigation, the EqP model was applied in order to evaluate potential differences in chemical exposure for the target receptors which might arise through differences in habitat or feeding type. The metric which was evaluated is called the Biota-Sediment Accumulation Factor (BSAF). BSAFs are calculated using the concentrations of chemicals in an organism (µg/g lipid) divided by the concentrations of the same chemicals in sediment (µg/g_{oc}).

In the evaluation of the data, it was hypothesized that if species of different habitats have similar transport pathways for contaminant exposures, then BSAF values should not be different. In other words, a finding of similar BSAFs among species would lead to the conclusion that the mode of contaminant uptake for organics is functionally the same and best represented by the EqP model.

BSAFs from the present assessment were inter-compared for similarity of central tendency as grouped by chemical class (PCBs, PAHs, and pesticides). Since the purpose of the model is to determine contaminant uptake behavior at concentrations of potential ecological concern, data values from sediment or tissue measurements which were below detection or less than the method limit of detection were not used in the analysis because use of these data would only increase uncertainty in the BSAF estimate, particularly in cases where both numerator and denominator chemical values contained flagged results. For similar reasons, data from reference sites were excluded from the analysis.

Plots of BSAFs for each species by compound class are presented in Figure 6.3-2; results for individual species/analyte combinations are presented in Table 6.3-1. The box plots present the median value ± 25% (box top and bottom) as well as the outside range or "whiskers" (approximately = 95% confidence limits). The dashed line represents the mean of species-specific median values for each class. As shown in the Figure, the median BSAF values for PAHs, PCBs and pesticides was 0.12, 1.96 and 1.83, respectively. Collectively, the data distributions indicate that there is considerable overlap in central tendency about the median BSAF value for all species.

These results are remarkably similar to BSAF values calculated from literature values for infaunal deposit feeders, scavengers, filter feeders and benthically-coupled fish (Tracey and Hansen, in press). In that study, BSAFs for PAHs were also uniformly lower (mean 0.34) than PCB (1.03) or pesticide (1.36) classes. Because BSAF values are calculated for various species both within and among habitat groups (e.g., fish vs. bivalves), the results indicate that the organic constituent exposure pathways from the sediment to the species is also similar. For this reason, a single exposure pathway model appears most appropriate to predict ultimate fate of organic contaminants (e.g., tissues) for the target receptors of concern.

6.3.2. Metals Bioaccumulation

Trace elements as a contaminant class tend to sorb to particulates and sediment and are well known to bioaccumulate in aquatic organisms. However, models for metals bioaccumulation in fish and shellfish species do not currently exist to the extent developed for the organic contaminants as discussed above. This is in part due to the fact that many trace elements are important in plant and animal nutrition, where they play an essential role as micronutrients for tissue metabolism and growth. Among those metals detected in sediment, tissue and water at Allen Harbor, copper and zinc are considered essential trace elements, although excess bioaccumulation may result in adverse impacts on the species. Hence, some metals

are preferentially retained and bioconcentrated such that tissue residue concentrations are not expected to be found in proportion to exposure concentration.

In the following discussion, exposure pathways for metals bioaccumulation is investigated through analyses of relationships between sediment and tissue concentrations via tissue/sediment ratios. As was found for organic CoC correlations, variations in tissue CoCs for target species when compared to sediment CoC concentrations by linear regression did not effectively elucidate potential exposure pathway differences among target receptors; correlation coefficients (r^2) for representative inorganic constituents (Pb, Ni, Cu, Zn, Cr) were generally poor ($r^2 <$ .20).

An alternate approach was taken involving the use of tissue/sediment ratios, referred to here as Biota Accumulation Factors (also called Bioaccumulation Factors). The BAF model differs significantly from the BSAF model in that it only assumes that tissue residue concentrations are driven by sediment exposure concentrations and does not include media-specific partitioning factors as does the BSAF model (e.g., tissue lipid and sediment organic carbon). Hence BAF data are presented on an analyte-specific basis for each species.

Biota accumulation factors (BAFs) for the five species investigated in this study were calculated for each of the nine metals included as CoCs. As with BSAF plots, the BAF box plots represent several measures of central tendency. The median of each species metal pairing was used to calculate the overall BAF for the metal. Qualified data (e.g. non-detects) were excluded from the analysis as had been done for BSAF analyses discussed above.

The overall data show highly metal specific BAF values which ranging two orders of magnitude from 0.02 (Pb) to 1.75 (Ag) (Figure 6.3-3). Results for individual species/analyte combinations are presented in Table 6.3-1. Some general similarities

among metals are observed; Pb was least available, Cr and Ni were relatively less available (0.08 - 0.12) ; Hg, Cu, As and Cd were moderately bioavailable (0.30 - 0.76); while Ag and Zn were the most bioavailable (1.68 - 1.75). For a given metal, the data generally indicate overall similarity in BAFs among species There are perhaps two notable exceptions; enhanced bioavailability of copper and zinc in oysters. Oysters are well known for their ability to bioconcentrate these metals, thus this effect is likely related to physiological mechanisms rather than differences in exposure. Although several other metals would appear to show species-specific differences, e.g., reduced Cd and Ag in fish or high Hg in soft shell clams, limited number of observations prevent generalizations. Finally, the narrow range of variation in metals BAF for fish suggests that the two species represented in the data set (*F. heteroclitus* and *F. majalis)*, do not have significantly different exposure pathways to metals.

6.3.3. Gut Depuration Studies.

Sediment gut contents have been demonstrated in previous studies to introduce a potentially serious source of error in monitoring programs where the objective is to determine the true levels of CoCs incorporated into the organisms' tissue (Flegal and Martin, 1977; Lobel et al., 1991). Hence, depuration studies with bivalves were conducted in order to better assess the importance of sediment gut contents on the apparent exposure relationship discerned from BSAF and BAF analyses discussed above. Depuration consisted of holding the organisms for a 24-36 hr period in clean flowing seawater containing natural particulates so as to allow the organism to void sediment in the gut during normal ventilation.

Plots of BSAFs for depurated species are presented in Figure 6.3-4 by compound class. Here, the median BSAF values were 0.21, 1.02 and 1.02, respectively, for PAHs, PCBs and pesticides. These values are similar for PAHs but lower for PCBs and pesticides than the non-depurated group. This difference most likely reflects the sediment contribution within the gut of the animal, which adds

significantly to the whole body concentration that might be consumed by a predator of benthically coupled organisms (e.g. winter flounder). Interestingly, the BSAF values for PCB and pesticides suggest 1:1 correspondence for CoCs in lipid and the TOC fraction of sediment, hence equilibrium.

Additional analyses of BAF data were conducted in order to evaluate effects of sediment-associated metals on trophic transfer to predators of bivalve target species. While the prior analysis included tissue lipid and TOC normalization (hence evaluating equilibrium partitioning), the present analysis is more pertinent to predator consumption (since the predator consumes its prey whole, irregardless of lipd content). The data were calculated as the ratio of depurated to non-depurated BAFs obtained for each compound class and species (Figure 6.3-5). BAF ratio analyses were completed for both metal and organic CoC classes.

Results of the BAF ratio analysis indicated that the median depurated BAFs for organics are slightly lower than non-depurated BAFs (BAF ratios less than 1), whereas the metal BAF ratios indicate that approximately 25% loss of metal has occurred during the depuration process (Figure 6.3-5). This finding would suggest that the sediment-associated contribution of metal may be as much as one-quarter of the whole body burden (metals) which would be ingested by a predator. In contrast, the sediment-associated organic contaminants contribute relatively little (< 8%) to the CoC loading to the predator.

These results have significance with respect to food chain transfer dynamics for winter flounder, being a target receptor species for the present assessment. Flounder may consume both juvenile filter feeders or deposit feeders such as polychaete worms. In the study by Tracey and Hansen (in press), the polychaete worm, *Nereis virens* was found to have BSAF ratios of 1.37 (n = 101) and 2.58 (n = 27) for PCBs and pesticides respectively, and another polychaete, *Lumbriculus variegatus*, had BSAFs of 0.07 (n = 26) and 1.19 (n = 28) for PAHs and pesticides, respectively.

Hence, the preferred food of winter flounder displays approximately the same equilibrium condition for organics as found for the bivalves and fish in the present study. In addition, BSAF data for winter flounder (BSAF = 0.66, n = 231) indicate that PCBs are accumulated to approximately the same extent as other sediment-associated fish species (summer flounder- 2.89, n = 46; channel catfish- 2.22, n = 24). Thus it would be expected that the BSAF and BAF relationships measured for fish and bivalves in the present study represent a reasonable approximation of expected bioaccumulation in winter flounder. The potential impacts of bioaccumulated contaminants on this species will be evaluated in Section 6.6.

6.3.4. Tissue Residue Enrichment Factors

This section summarizes comparisons of CoC concentrations in Allen Harbor biota with reference site organisms in order to characterize various areas of Allen Harbor with regard to extent of CoC tissue enrichment relative to background (e.g. reference) conditions. As with the sediment-based site/reference comparisons, the analysis is not intended to characterize risk, *per se,* although in a qualitative sense, those areas which have grossly elevated tissue contamination relative to reference sites should be deemed high priority areas for detailed risk analysis. Conversely, areas with organisms having have levels of contamination below reference location conditions should be given lower priority.

Table 6.3-2 presents a summary table of the tissue enrichment factors by species as discussed above. A cumulative rating scheme was devised so as to facilitate description of the findings and carry forward the results to the weight of evidence evaluation in Section 7. As with the sediment-based concentration ratios, a ranking of observed enrichment was based on:

+ = 1 spp. with a mean CR > 2;

++ = 1 spp. with a mean CR > 10 or 2 spp. with mean CR > 2,

following the logic that additional weight should be given to environmental conditions (i.e., EEZ - chemical contamination characteristics) of pervasive (e.g. multiple species) and/or substantial (high CR ratios) CoC tissue enrichment.

For hard clams, few of the ecological exposure zones (EEZs) from which the animal was collected were found to have animals with tissue residues exceeding the reference site concentration (Table 6.3-2A). In contrast, tissue concentration ratios for oysters from several EEZ exhibited significant enrichment, including most notably, metals enrichment at LN-W (Table 6.3-6B).

Ribbed mussels generally followed the trends observed for hard clams, exhibiting less than two-fold enrichment across the various EEZ and chemical groups (Table 6.3-2C). Notable exceptions included high metals enrichment at the CP-SW EEZ.. This finding was driven by high nickel and chromium at Station V4 and mercury at Station W14.

Concentration ratios for soft shell clams could not be evaluated since reference site collections for this species were not possible. Mummichog fish samples were successfully collected in Allen Harbor and reference sites, and concentration ratios suggest that enrichment was greater than two-fold at CP-SW and LN-W EEZs (Table 6.3-2D).

Results of the above discussion of tissue-based concentration ratios are summarized in Table 6.3-3. The data are pooled by chemical class and across species to develop a generalized picture of CoC enrichment patterns throughout the various ecological exposure zones of Allen Harbor. The most pervasive and extensive CoC enrichment was observed for the LS-V/W and LN-W EEZs, with metals, PCBs and pesticides being the primary exposure drivers. The remaining EEZs ranked approximately equal, and exhibited diverse patterns of CoC enrichment. The results will be carried forward to the risk assessment summary in Section 7.

6.4 ANALYSIS OF CoC CONCENTRATIONS *VERSUS* APPARENT ADVERSE IMPACTS

This section evaluates the relationship between CoC concentrations and apparent adverse impacts suggested by toxicity responses of two bioassay species. As described in Section 6.2, the analysis focuses on "potential exposure-response" types of relationships. For this assessment the measurement endpoints evaluated included the following:

- Toxicity of bulk sediments to amphipods (10-day mortality)
- Toxicity of pore water to sea urchin gametes (sperm cell test)

Following the interpretative framework discussed in Section 6.2, results of Pearson correlation analyses (and associated probabilities of regression statistical significance) of relationships between the above toxicity measurements and 1) CoC concentrations in sediments and porewater and 2) Hazard Quotients derived from CoC concentration normalization to NOAA ER-L and Water Quality Criteria are presented to systematically evaluate potential CoC-based risk drivers. The comparison of organic CoCs in porewater includes calculation of "WQC-SC equivalent" benchmarks from Table 3.3-2, and assigns a data qualifier to identify the benchmark source for derivation of the Hazard Quotient.

The concentration of Simultaneously Extractable Metal (SEM) measured in excess of Acid Volatile Sulfide (AVS) is considered biologically available and hence of potential toxicological significance (Di Toro, et al., 1990). The extent of SEM excess is commonly expressed as the SEM/AVS ratio. Another, perhaps better expression of excess SEM is the difference of SEM - AVS, which is less sensitive to conditions where AVS is near detection limits (e.g., resulting in very high SEM/AVS ratio). In the present study, SEM/AVS, SEM-AVS and SEM concentrations are presented to

characterize the potential toxicological significance of sediment metals on amphipod survival and sea urchin fertilization.

6.4.1. Amphipod Survival Results

For amphipods, results from the SEM comparison indicated no exposure-response relationship (correlation) with survival (Table A.3-3.1). However, four stations (W5, W6, W13, and V3) exhibited significant toxicity (< 80% survival) and had SEM/AVS values exceeding 1 (Figure 6.4-1A). Of these, Station W5 had 10-fold higher total excess (14.3 µM/L; Figure 6.4-1B), and total SEM values (17.6 µM/L; Figure 6.4-1C) than other stations with SEM/AVS > 1. For vegetated station V3, SEM concentrations are low and less likely to be of toxic. At station W5, the SEM fraction included primarily copper, zinc, lead and cadmium (Table 4.2-1). Thus, among the four stations with SEM/AVS ratios exceeding unity, only station W5 would appear to have metals concentrations of possible toxicological significance to amphipods.

In general, no significant correlations were observed between amphipod survival results and sediment or porewater HQs for any CoCs (Table A.3-3.1). In addition, many of the stations characterized by high organics concentrations were not associated with low amphipod survival values (Figure 6.4-2A-C). Examples of this pattern are depicted for Stations W1 and W4 for DDE, tPCB, and tPAH, and Station V1 for DDE. However, the principal stations that do combine significant toxicity and elevated chemical contamination are Stations W5 and W6. For PCBs, a highly significant inverse correlation was observed between amphipod survival and the corresponding HQ for porewater (Figure 6.4-3), with Stations W5 and W6 driving the regression.

A toxicity model for amphipods has been recently proposed in order to evaluate the potential importance of PAH contamination on amphipod mortality (Schwartz et al., 1995). The model involves the summation of analyte-specific

interstitial water toxic units (IWTU; porewater concentration/LC_{50}) and provides statistical estimates of probability of toxicity as a function of IWTU values. The model was applied to the present investigation to assess the potential role of PAHs on observed amphipod survival. A plot of observed mortality (100 - survival (%)) vs IWTU is shown in Figure 6.4-4 along with zones delineating areas of statistical probability for toxicity. The sumPAH model predicts with 95% confidence that amphipod mortality, due solely to PAH exposure, should be less than 24% where IWTU values are 0.185. From the observed data scatter, it can be concluded that toxicity observed at Stations D1, V3 and W6 is not due to PAH exposure. In addition, only about 1/3 of the toxicity at Station W5 should be PAH related (13/37%).

6.4.2. Sea Urchin Fertilization Results

As was generally found for the amphipod test, there were no significant correlations observed between fertilization success and sediment or porewater concentrations or associated HQs for any CoCs (Table A.3-3.1). The SEM-toxicity analyses indicate that only Station W6 exhibited both reduced fertilization (9%) and an SEM/AVS ratio >1 (Figure 6.4-5A). Relative to other stations where toxicity was not observed, the small total excess SEM (<0.1 µMol/g; Figure 6.4-5B) and low total SEM concentrations (<4 µMol/g; Figure 6.4-5C) would suggest that metals do not appear to be the primary cause of reduced fertilization success at Station W6 and consequently sea urchin tests in general.

Despite a lack of exposure-response correlations, the combination of significant toxicity and elevated PCB contamination at Station W6 (Figure 6.4-6B), but not DDE or PAH contamination at this station (Figure 6.4-6A and C) is consistent with the previous findings for amphipods where PCBs emerged as a key driver of toxicity. The precise toxicological significance of SEM metal concentrations are not well known. Because sea urchin toxicity was not observed at Station W5 where SEM metals were in excess of AVS would suggest that biologically available concentrations were not

sufficiently high to be toxic. Hence, the data would suggest that metal contamination at W5 may play a relatively minor role in comparison to PCBs as the cause of apparent toxicity to sea urchins and amphipods. Sea urchin toxicity was also not observed at Station W5 where porewater PCBs appeared to cause reduced amphipod survival. This is perhaps partial attributable to the short-term nature of the fertilization test vs. amphipod test (60 min. vs. 10 days, respectively), whereby total exposure (e.g. duration x intensity) is sufficiently reduced at W5 to account for this difference.

Other stations exhibiting significantly reduced sea urchin fertilization included Stations D1 (0%) and DC-13 (0%). These two stations were not found to exhibit significant toxicity in amphipods (Table 5.3-1), and generally had CoC concentrations below levels found at other stations where no toxicity was observed. For example, Station D1 was notable for relatively high porewater Pb concentration, being 1.3 times the ambient Water Quality Criteria Chronic Value (see Section 6.1), but another Station DC-10 was as high, but did not cause toxicity. Sub-surface sediments at the same location (Station DC-1) provided no evidence of toxicity (99% of control fertilization success), further suggesting that either the response does not reflect a real exposure , or some unmeasured CoC is responsible. Similar uncertainties accompany explanations of reduced fertilization success at Calf Pasture Point Station DC13. However, in this instance the data do suggest that the response may have been caused in part due to ammonia toxicity; given that measured concentrations were more than twice that known to cause 50% mortality (i.e., > LC_{50} of 0.6 mg/L; NOAA, 1994).

6.4.3. Comparisons of Findings with Previous Studies.

The Risk Assessment Pilot Study (RAPS) incorporated a risk assessment approach whereby potential impacts of site-related contamination on aquatic organisms in Allen Harbor were investigated in three phases. Each of the phases included laboratory bioassays conducted on Allen Harbor Landfill soil, sediment and

leachate. During Phase 1 investigations, no observations of toxicity were observed in the amphipod, *Ampelisca abdita*, when exposed to intertidal or subtidal sediments from Allen Harbor (Munns et al., 1991). However, soil samples collected from the northern and middle high intertidal of the Allen Harbor Landfill were found to exhibit significant reduction in survival (18.9 and 12.3% survival, respectively) while southern landfill material exhibited marginal (77.3% survival) toxicity. No analyses of sediment CoC concentrations were conducted in conjunction with the toxicity evaluation at the landfill stations. However, analysis of leachate samples collected from the south landfill face did reveal high PCB concentrations (sum of Aroclors = 950 ng/L), while PAH concentrations were low (< 10 ng/L). The landfill south station also contained concentrations of 1,2-dichloroethane (up to 170 ug/L), but since this analyte was not detected in the present study, some temporal or small-scale spatial differences may be operative.

The above landfill source characterization stations were resampled for chemistry and toxicity during Phase II RAPS investigations (Munns, et al., 1992). Amphipod survival, measured during spring, summer and fall, was generally not found to be significantly reduced at landfill north and south stations (< 80% survival), whereas the landfill middle station continued to exhibit variable toxicity (12-67% survival). Additional stations sampled to characterize runoff included intertidal and subtidal samples at stream inputs to the north (NC) and south (WC) of Allen Harbor landfill as well as Spink Neck creek located in south Allen Harbor. The locations of NC and WC stations cannot be precisely discerned from their report, but would appear to be approximately located near Stations W1 and W9 of the present investigation. While the NC station sediment was not toxic to amphipods (mean survival = 86%), the WC station was only marginally toxic (mean survival = 75%). These results are consistent with the present study findings, where sediments from both locations were found to be non-toxic to amphipods. Finally, a single sediment sample from the landfill middle location was tested during RAPS Phase III and found to be non-toxic to amphipods. Overall, the temporal pattern of toxicity to amphipods at the landfill middle

station is one of decreasing toxicity with time, and presently exhibits a non-toxic state; the RAPS station appears most closely located in proximity to Station D5 of the present investigation where survival was 96%.

Sea urchin fertilization tests conducted at the landfill and runoff stations during RAPS Phase II were also found to be non-toxic (fertilization success > 50%) although marginal toxicity (fertilization success = 68%) was found at one occasion at the landfill middle (LANDM) location. These results are corroborated by findings of the present investigation, where both stations were non-toxic to sea urchin gametes. Also during the Phase III RAPS, a suite of additional bioassays were conducted with leachate collected from "the middle of the face of the landfill" (Munns et al., 1993), and found decreased fertilization and increased abnormal larval development of sea urchins, reduced reproduction in red algae (*Champia parvula*), and abnormal development of coot clam larvae (*Mulinia lateralis*). The latter results, however, are only indirectly pertinent to the present risk characterization, since the sample was not collected from the habitat of target species and thus cannot be assumed to approximate true exposure conditions to those species.

6.4.4. Summary of sediment toxicity results.

Summary statistics for sediment and porewater toxicity discussed above are presented in Table 6.4-1. Following the interpretative method discussed for biota condition results presented in Section 6.2. Sites with slight toxicity (<80% survival in amphipods or < 70% sea urchins) were rated "+" for slight risk, while pronounced toxicity (<60% survival in amphipods or < 50% sea urchins) were rated "++" for moderate risk. For amphipod survival, slight toxicity was observed in 5 of 8 EEZs, whereas for sea urchin fertilization success, 2 EEZs exhibited pronounced toxicity (LS-D and LN-W) and one EEZ exhibited slight toxicity (LM-D).

A qualitative summary of risks based on toxicity results is presented in Table 6.4-2. The EEZ scoring was based on the extent of toxicity observed (including magnitude of change, co-occurrence of toxicity among bioassays and spatial distribution) and whether statistically significant exposure-response relationships were found which support a hypothesis of cause and effect. Toxicity endpoints eliciting a response without an apparent exposure-response relationship are scored as "+" for slight risk, while endpoint responses which are supported by strong statistical evidence of exposure - response are scored as "++" (moderate risk).

Exposure-response relationships supported SEM metals at Station W5 (Figure 6.4-1), porewater PCBs at Station W5 and W6 (Figure 6.4-3) and PAHs at Station W5 (Figure 6.4-4) as causal factors in observed toxicity in amphipods. Hence, apparent risk at EEZs LS-V/W and LN-W are considered moderate ("++"). Although similar exposure response relationships were not observed for sea urchin results, the co-occurrence of toxicity in both amphipods and urchins at LS-D suggests slight risk ("+"). The remaining EEZs, for which slight toxicity was observed (including LS-D, LM-D, CP-W and CP-SW) are considered not at risk ("-") since exposure-response relationships were not evident. The results will be carried forward to the risk assessment summary in Section 7.

6.5 Comparison of CoC Concentrations with Criteria and Standards

6.5.1. Sediment Criteria Comparisons

Sediment criteria have been developed for only a few of the Allen Harbor CoCs. Consequently, this section compares concentrations of these constituents of concern with national criteria, placed in a spatial distribution context. The results of this analysis will be used provide data by which various zones of Allen Harbor can be grouped and compared based on apparent risk relative to available criteria and standards.

Station Comparisons. The magnitude of concentrations relative to the ER-L, the most stringent of the screening criteria shown in Table 3.3-2, are indicated in Table 6.5-1 using symbols "+" and "++" for station CoC concentrations exceeding the ER-L (e.g. HQ > 1) and ten times the ER-L (HQ>10) respectively. Results are grouped according to Ecological Exposure Zones identified in Section 6.0.

Sediments from stations within the Landfill South Vegetated/Wetland zone contain CoC concentrations which exceed ER-Ls for most metals, PAHs, total PCBs and p,p'-DDE (Table 6.5-1). Station W5 is notably high for metals and PCBs whereas Station W1 is high for PAHs, PCBs and p,p'-DDE. Station data for the Landfill Middle-Deep zone (LM-D) show that metal, high molecular weight PAH, PCB and p,p'-DDE contamination and associated risk is apparent but "slight", given that CoC concentrations are less than ten times ER-L. The Landfill South-Deep (LS-D) and Calf Pasture Point EEZs (CP-W, CP-SW and CP-SD) EEZs appear to be at negligible risk for most CoCs except perhaps Station D14 which resembles the other Allen Harbor deep stations (D4-D10, DAH5). Similarly low risk is apparent for the Landfill North-Wetland EEZ, with the exception of station W6 which is high for PCBs.

Habitat-wide Comparisons. Habitat-based HQ statistics are summarized in Table 6.5-2. Elevated mean HQs for total metals were observed in the subtidal, vegetated wetland, and intertidal wetland habitats. The total metal HQ for the deep core subtidal habitat was below unity, as were mean HQs for reference sites. HQ values for VOAs were below unity in all habitats. Total PCBs were elevated (i.e., HQ $\geq$ 1) in subtidal habitats, while high HQ values for pesticides were observed for wetland and vegetated habitats. The high tPCBs HQs in the offshore habitats can be attributed to highly elevated HQs (i.e., HQ >10) at two RAPS study stations (AH3 and AH8).

The highest PAH HQs were observed in the intertidal and vegetated wetland habitats. However, mean HQ values for tPAHs were all below unity, and maximum

HQ values ranged from 0.44 to 2.45, indicating that concentrations of individual PAHs generally were below their ER-L reference value.

Ecological Exposure Zone Comparisons. EEZ-based HQ statistics are presented in Table 6.5-3. In order to facilitate the discussion of the results and their utilization in the risk assessment summary, a qualitative ranking system (analogous to the strategy presented in Section 6.1) was devised as follows:

Individual Ranking Notation:

+ = Mean HQ > 1;

++ = Mean HQ > 1 and Max HQ > 10;

+++ = Mean HQ > 10 and Max. HQ > 10

This ranking scheme is intended only as qualitative tool. The logic behind the ranking is that environmental conditions represented by a HQ > 1 represent a risk factor of some concern ("slight" risk); while an EEZ with mean HQ >1 and individual stations with HQ > 10 indicates more pervasive contamination and therefore is of "moderate" risk; and finally, a suite of stations which have pervasive contamination resulting in mean HQ > 10 is indicative or gross contamination and is at "substantial" risk.

Total PCBs posed at least some risk in landfill-associated EEZs and only slight risk in EEZs adjacent to Calf Pasture Point (CP-W and CP-SW). Substantial risk was indicated from HQs observed in the LS-V/W (19.1) and LN-W (15.1) zones, respectively. Pesticide HQ values indicated only of slight risk concern for the LS-V/W and LM-D zones. Total metal HQs indicated slight risk in the LS-V/W, LM-D and CP-W zones. Slight risk for PAHs was found only for the LS-V/W zone. Hazard Quotients for VOAs suggest little risk is posed by this chemical group throughout Allen Harbor.

Summary. Table 6.5-4 presents a summary table of the above findings. As was presented in Section 6.1, a cumulative rating scheme was devised so as to facilitate description of the findings and carry forward these results to the weight of evidence evaluation in Section 7. Scoring of each individual ranking was based on '+' =1 pt., '++' = 3 pts. and '+++' = 5 pts, following the logic that additional weight should be given to environmental conditions (i.e., EEZ - chemical contamination characteristics) of significant and/or substantial CoC enrichment. Following this scheme, scores were summed across chemical classes to evaluate general enrichment trends. Results were then qualified following the scheme: 3-5 pts. = slight risk (+), 6-9 pts. = moderate risk (++), and > 9 pts. = substantial risk (+++). The rationale for this scheme was that 1) a high HQ (++) for any CoC class would indicate, at the minimum, slight risk for the EEZ, 2) the cumulative risk of multiple CoC classes would be additive, 3) two or more classes of CoCs, each of moderate risk (3 pt), would together present "substantial risk" when simultaneously present in the environment.

Sediment PCB concentrations exceeding national criteria were concentrated in seven of eight EEZs; only CP-SD was not at some risk due to PCB exposure as indicated by this metric. The corresponding EEZ scores indicate that the LS-V/W and LN-W zones are at "moderate" risk, due primarily to PCBs, while the LS-D and LM-D zones were at slight risk due to mixtures of several CoC classes. These results will be carried forward to the risk assessment summary in Section 7.

6.5.2. Porewater Criteria Comparisons.

Habitat-wide comparisons. Table 6.5-5 presents results of porewater Hazard Quotients for the five main chemical classes and for habitats of Allen Harbor. Whereas metals concentrations were measured directly, organics concentrations were calculated using equilibrium partitioning (see Section 6.2).

Concentrations of pore water metals were generally below their reference concentrations, as shown by low mean HQ values in all Allen Harbor habitats (Table 6.5-5A). Nevertheless, some stations had elevated HQ values for certain metals (Table A3-3.6). In subtidal habitat stations, HQ values were elevated for arsenic at Station D1, mercury at all stations except D6, and copper at Stations D2, D3, D6, D11, and D14. Deep core subtidal sediments had elevated pore water HQ values for mercury in all stations, elevated HQ values for nickel at Station DC6 and DC13, and elevated lead HQ values at Station DC11. Mercury HQ values were also elevated in all vegetated wetland stations except Station V3, and all intertidal wetland stations. Elevated HQ values for chromium, nickel, and copper were observed at different vegetated wetland stations. Elevated HQ values for lead and copper, were observed at some intertidal wetland stations. It must be noted that most reference sites also had elevated HQ values for mercury. Evaluation of pore water metal HQs by EEZ revealed elevated values for mercury in all LS-VW, LS-D, LN-W, CP-SW, CP-SD, and Reference EEZ stations.

HQ values for pore water tPCB were most elevated in deep core subtidal sediments, and lowest in intertidal wetland sediments (Table 6.5-5B). HQ values were comparable in subtidal surficial sediments, and in vegetated wetlands. It is also noteworthy that pore water tPCB HQ values were more elevated at reference sites for vegetated and intertidal wetlands than in the Allen Harbor habitats.

The magnitude of apparent risks on a habitat wide basis are misleading in that there exists a great degree of heterogeneity in the distribution of contamination across the site, e.g., high nearest Allen Harbor Landfill and lower near Calf pasture Point. Hence the numerical combination of both areas tend to reduce the apparent risk.

Ecological Exposure Zone Comparisons. Further inspection of porewater HQs by Ecological Exposure Zone was performed to provide better resolution on possible CoC related risks to target receptors from porewater exposure. Summary results by

chemical group and EEZ are presented in Table 6.5-6 along with a qualitative risk ranking as described for sediment HQs above.

Evidence of slight risk (mean HQ > 1) was found due to total PCB exposure in porewater in most Allen Harbor zones excluding the Landfill South -Vegetated/wetland area. High molecular weight PAHs also appear to pose a slight risk at the LS-V/W station. No other CoC groups were prominent with regard to elevated porewater concentrations of concern. A summary of the above porewater information is provided in Table 6.5-7. Although absolute HQ values were low, a slight risk for porewater PCBs is presumed given the exposure response relationship demonstrated in Figure 6.4-3. These results will be carried forward to the risk assessment summary in Section 7.

6.5.3. Surface Water Criteria Comparisons.

Dissolved phase (field blank-adjusted) organics and metals data for Allen Harbor surface waters were compared against Chronic Water Quality Criteria Values to discern potential risks to aquatic organisms. PAH concentration values were 300X lower than criteria at all reference sites (WQC-total PAH = 300,000 ng/L). PCB concentrations approached the criteria value of 28 ng/L (PCBs = 30 ng/L) at station D5. Dissolved phase metals concentrations of Pb exceeded the CWQC at D5 and D6 (Figure 6.5-1). In addition, Cd and Cu also exceeded criteria at station D6. However, observed concentrations were bracketed by reference site concentrations. Of particular note was a high Cu HQ observed at the JSC reference site which is located in the general vicinity of marina activities, and is analogous to the spatial relationship between Allen Harbor landfill and the South Allen Harbor marinas. This peak is possibly related to recent introduction of recreational water craft having newly painted bottoms comprised of Cu-containing anti-foulants.

Results of water sampling were higher than concentrations reported in the previous RAPS Phase I investigation (draft report table 3.1-2); measurements at RAPS station AH2 was 38 ng/L for tPAHs, and non-detectable for PCBs and pesticides (metals data were not available). This difference could be due to locational and or temporal sampling differences, as the present sampling strategy was specifically design to capture maximum landfill-related concentrations but further north than Station AH2.

6.6 Residue-based Exposure-Effects Relationships

In this section, tissue residue concentrations in target receptors are evaluated against field measures of biota condition and suspected threshold concentrations for effects. Two separate types of analyses are presented for this evaluation. In Section 6.6.1, tissue residue effects are considered in the conceptual framework known as Critical Body Residue (CBR) analysis (McCarty et al., 1992). Section 6.6.2 presents the results of a different approach involving comparisons of tissue residue concentrations against the Tissue Screening Concentrations (TSC; Shepard, 1995).

6.6.1. Critical Body Residue Assessments

Critical Body Residues can functionally be described as the tissue residue which exists in the species at the incipient effect concentration. The utility of CBR analysis for this ERA is enhanced by the fact that the selected CoCs generally fall into a class of compounds which can contribute to a non-specific, "narcotic" mode of toxic action referred to as narcosis (McCarty et al, 1992). The mode of toxic action of narcotic chemicals is described as non-specific and physiologically reversible (McCarty et al, 1992). Thus, when CoCs in target tissues are below the effects threshold for specific toxic action (e.g., central nervous system, respiratory, digestive disruptions), the combined narcotic impact of the chemical mixture can predicted from the sum molar concentration of the mixture (McCarty et al, 1992).

greater than the incidence rate found at Upper Narragansett Bay reference locations (Table 5.3-3).

6.6-2. Tissue Screening Concentration Assessments.

Water Quality Criteria developed by the USEPA are designed to be protective of 95% of aquatic species. The fundamental basis for Tissue Screening Concentration (TSC) Assessments is the assumption that the product of the criteria value for a given CoC and the bioconcentration Factor (BCF) should provide an estimate of the tissue concentration which is also protective to the species. A recent study by Shepard (1995) involving a literature survey of paired residue/effects data for 100+ chemicals demonstrated that only 19% of the derived TSCs were higher than tissue residues found to be associated with toxic effects (e.g., the derived TSC was protective of aquatic life 81% of the time). CoCs for which the TSC approach was not protective included primarily the PAHs; It was hypothesized that these compounds exhibited enhanced toxicity through activity of metabolic breakdown products which are not measured (Shepard, 1996). Exclusion of these compounds from the analysis improved the protectiveness of TSCs to a level equivalent to water quality criteria (93%).

Given this first demonstration that TSCs can provide a level of protection equivalent to WQC, TSCs were adopted for this ERA to assess potential effects of tissue residue concentrations in target receptors. A Hazard Quotient approach was taken in this analysis, in which the measured tissue concentration and the TSC effect concentration are used as follows:

$$\text{Tissue Residue Hazard Quotient (TRHQ)} = \frac{\text{Tissue Residue Concentration}}{\text{Tissue Screening Concentration}}$$

The predicted acutely toxic CBR concentration is predicted to be approximately 1-10 µMol/g wet weight of tissue (6.6-67 mMol/kg dry wt., assuming 85% water content). Tissue lipid concentration appears to have some effect on the CBR; the CBR increases approximately 14% per % lipid over 3-8% range in lipid concentration (McCarty et al., 1992). The CBR is believed to be non-species specific (McCarty et al, 1993). Sub-lethal (i.e., chronic) critical body residue concentrations can be estimated from the acutely toxic CBR by applying the acute:chronic ratio of the respective water or sediment concentration toxicity relationship. For example, Arnold and Biddinger (1995) have reported acute and chronic CBR values of 6.4 and 0.53 mMol kg^{-1} dry weight (converted from wet weight) using a 1:12 acute:chronic ratio, respectively, for a group of selected PAHs similar to the CoCs identified in the present study.

In the present ERA, molar tissue concentrations (nMol/g dry wt) of organic contaminants were calculated from the mass concentration of chemical measured in the tissue (e.g. ng/g dry weight*CoC molecular weight, nMol/ng) , summed by chemical class (PAHs, PCBs, pesticides and TBT), and compared against representative chronic CBR benchmark values found in the literature (Table 6.6-1). The calculations show that in all cases, the measured body residue was one to two orders of magnitude below the Critical Body Residue reference benchmark.

Given the possibility that the CBR benchmarks were unprotective for target species evaluated in this ERA, Pearson correlations of body residues vs. biota condition indices were also performed to assess potential risk drivers (Table A4-1.11). These analyses failed to identify any correlations meeting the test for ecological significance ($r < -0.7$, $P<F \leq 0.05$) established for this study. However, a significant relationship ($r = -0.96$, $P>F = 0.04$) was observed between incidence of hematopoietic neoplasia in soft shell clams and the molar concentration of PAHs in the tissues of the species (Table A4-2.3; Figure 6.6-1). This relationship suggests that incidence of neoplasia at station W5 and at Calf Pasture Point may be caused by PAH exposure, although the extent of neoplasia for the latter group was not found to be significantly

Values for TSCs used in this evaluation were taken from Shepard (1996) and are presented in Table 6.6-2. The list includes all nine CoC metals, 16 of 18 PAHs, total PCBs, all five pesticides and one of three organotins (tributyltin) selected as CoCs for this ERA. Evaluation of tissue residues was performed by constructing Pearson correlation matrices for relationships between biota condition and TRHQs exceeding one.

A summary of mean and maximum observed TRHQs by species and analyte are presented in Table 6.6-3. The results show numerous CoC-receptor pairs as having TRHQs exceeding one for metals and TBT and values approaching one for PCBs. In contrast, PAHs and pesticides were at least 10-fold lower than presumed effect concentrations (note that some CoC-receptor pairs were not evaluated because of non detectable tissue residue concentrations).

Because of such low TRHQ values and the possibility that the CBRs were underprotective, especially for PAHs, TRHQs > .01 were included for subsequent correlation analyses. Full results of correlations are presented in Appendix Table A4.4-12; summary statistics for significant correlations ($r < -0.7$; $P \leq 0.05$) are summarized in Table 6.6-3. Three CoC-receptor pairings were found to have significant relationships. All significant relationships were for metals, and included hard clams and cadmium TRHQ, and oyster and fish with zinc TRHQs. The hard clam relationship was driven by a single high TRHQ at station W9, and hence is somewhat suspect as evidence of cause and effect. In contrast, highly significant and plausible exposure-response relationships were observed for zinc TRHQs (Figure 6.6-2). Correlations of hematopoietic neoplasia in soft shell clams with TSC-based HQs were not successful in identifying potential CoC risk drivers (Table A4-2.4).

6.6.3. Summary

A summary of apparent residue effects relationships is presented in Table 6.6-4. The data show that residue based effects on oysters and mummichog fish are the most likely observed effects. The observation that two independent measures of biota condition (fish length, oyster shell weight /length ratio) each are inversely related to the concentration of zinc in tissue provides substantive evidence that zinc may act as an important driver of ecological risks in Allen Harbor. In addition, the correlation of neoplasia incidence with total PAH body residues is also supportive of a potential exposure-effect relationship. These findings will be incorporated in the weight of evidence summary presented in Section 7.

6.7 UNCERTAINTY

The presumption of ecological risk in this assessment is based upon the reliability of data and interpretation presented as weight of evidence. The weight of evidence is in turn dependent upon the underlying analyses of exposure, effects and risk characterization findings. Uncertainties discussed in the exposure phase of this assessment (Section 4) included:

- Adequacy of CoC selection and behavioral characterization;
- Adequacy of fate and transport evaluations, including station selection, spatial (horizontal) and vertical (sediment layering) patterns, sample representativeness;
- Adequacy of characterization of temporal/spatial variability in CoC distribution;

- Reliability of exposure point estimation methods, including SEM/AVS calculations; porewater extraction techniques, etc..

- Adequacy of chemistry data, including comparability among past and present studies;

Uncertainties discussed in the effects assessment phase (Section 5.5) included:

- Adequacy of toxicity data, including comparability among test species and methods;

- Adequacy of biological investigations, including the appropriateness of the benthic condition endpoints measured, data analysis techniques, data availability limitations;

- Adequacy and availability of national criteria as benchmarks; and

- Appropriateness of the selected bioassay species as surrogates for the indigeneous community.

These exposure and effects uncertainties compound one another as exposure and effects data are integrated in the risk characterization. In addition to these uncertainties, there are additional uncertainties which are unique to the risk characterization, including:

- Limited toxicological data for target receptor species;

- Incomplete knowledge of community ecology including natural history (e.g., size of feeding range and site use) of many species, species

sensitivities to contaminants and trophic level transfers, and natural changes and variability in biological/ecological systems; and

- Adequacy of bioaccumulation and toxicological models;

- Lack of chemical concentration benchmarks for tissue residues;

Tissue residues were used in this ERA as an indication of exposure, however, their importance in ecological risk assessments is currently limited since evidence linking ecological effects directly with contaminant concentrations in tissue is generally lacking. The general adequacy of Tissue Screening Values applied in Section 6.6 is not known. In addition, more complete understanding of bioaccumulation and trophic transfer is required to evaluate the role of tissue residues in the status of natural resources, and to provide data for evaluating risks to human health associated with seafood consumption.

In addition, the seasonal and temporal sensitivity to pollutants has not been assessed in the present ERA, and leads to uncertainty given that, for example, seasonal rainfall may affect landfill leachate generation, or various life stages present at different times may have differential chemical sensitivity. There are also seasonal changes in redox potential as well as the concentrations of sediment organic carbon and acid volatile sulfides which may alter CoC bioavailability. Samples collected during the present assessment were not collected seasonally nor at a time period when these parameters are at seasonal minimums. Each of these factors has the potential to effect both the toxicity and bioaccumulation of the CoCs.

The application of organic (BSAF) and inorganic (BAF) bioaccumulation models have several uncertainties. The BSAF model relies on an empirical assumption that porewater concentrations are in equilibrium with sediment concentrations. This may not be the case, especially at sites such as Allen Harbor Landfill where CoC releases

could (and likely are) episodic and variable tidally, daily, seasonally and over the life of the landfill. Uncertainty with BAF models, e.g. species-specific bioaccumulation patterns for various metals, is highly site-specific and may vary among species. To complete the analyses of tissue-based Hazard Quotients in this study, it was necessary to apply these models to generate the necessary reference benchmark concentrations, and thus the resultant risk characterization carries uncertainty in this regard.

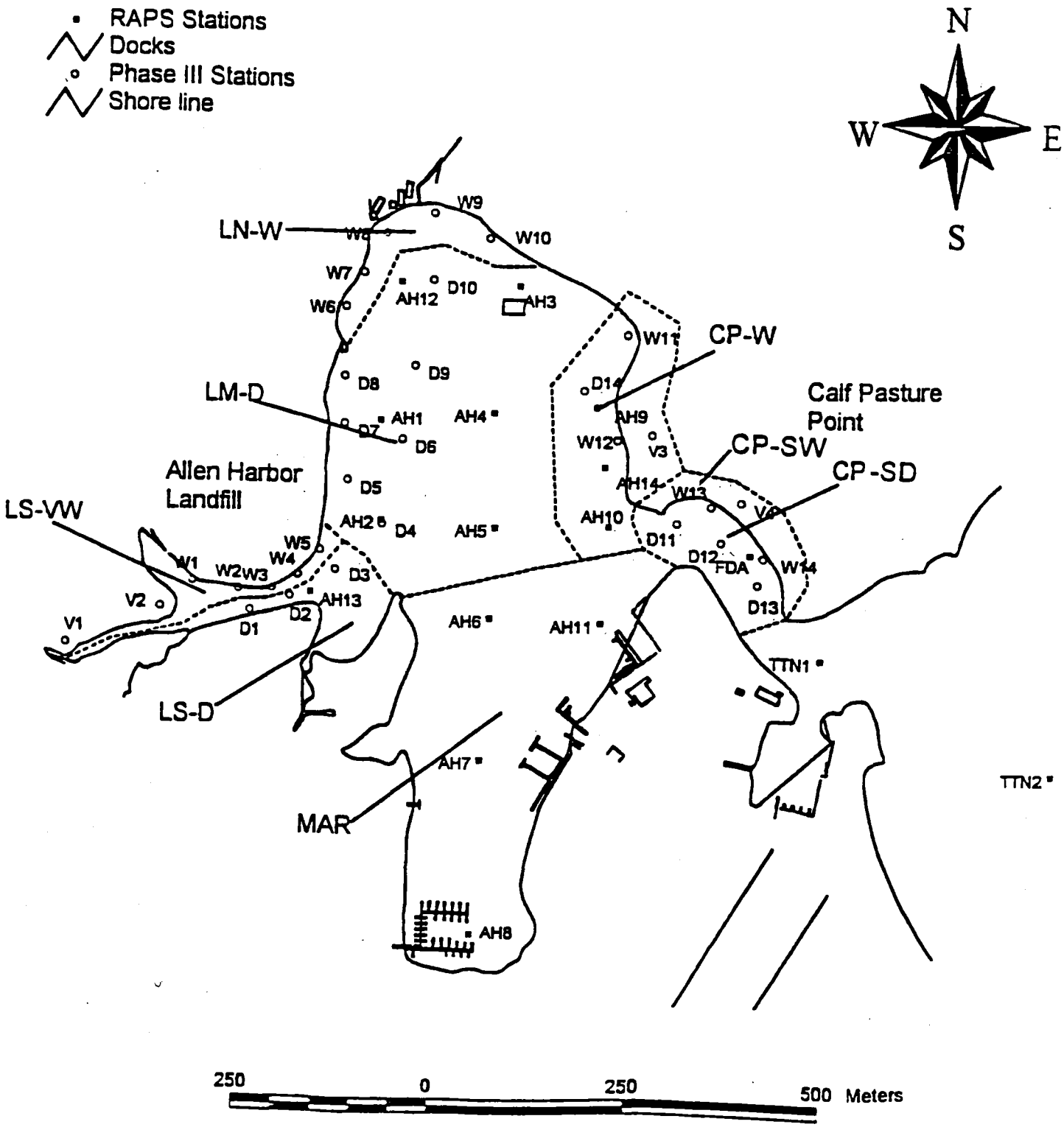
Uncertainties associated with the calculated Hazard Quotients and Hazard Indices exist because they are not necessarily reflective of all chemicals or activities of chemical mixtures. An additive approach to HQs was taken to integrate multiple contaminant effects, since information is very limited on the toxicity of simultaneous exposure to mixtures of contaminants. However, this estimation does not incorporate potential synergistic interactions among chemicals resulting in toxicity which greater than the sum of toxicities of individual chemicals, nor does it encompass risks from chemicals which were not measured.

Given that Risk Characterization is a synthesis of findings from the Exposure and Effects Characterizations, it follows that uncertainties associated with these components of the Risk Assessment can be nullifying, additive or even compounded. A prime example is in the application of Hazard Quotients and derived Indices, where the numerator and denominator each represent point concentrations with an unknown departure from the "true" concentration. Exposure - response relationships suffer the same uncertainty; separate error in estimates of survival and exposure concentration, for example, can compound to obscure true dose-response relationships or falsely suggest others which are misleading or unfounded.

The weight of evidence approach to characterization of risk is effective in reducing uncertainty because the probability that multiple exposure and effects indicators could spuriously suggest risk decreases as the number of indicators in

agreement increases. However, this approach in fact only reduces uncertainty with respect to the location and magnitude of risk; it does not specifically address the *ultimate* source of this risk, e.g. in the present study, the landfill vs. other CoC sources. This uncertainty has been addressed in the present study through analysis of CoC site/reference concentration ratios, exposure pathways, and other endpoints (e.g. magnetic susceptibility logging and principal components analysis of extended analytes) which did not suggest alternative CoC sources.

Figure 6.0-1. Location of Ecological Exposure Zones for the Allen Harbor ERA.



Scale: 1:7000

Figure 6.2-1. Pore Water Screening Criteria Selection Process and Data Qualifiers

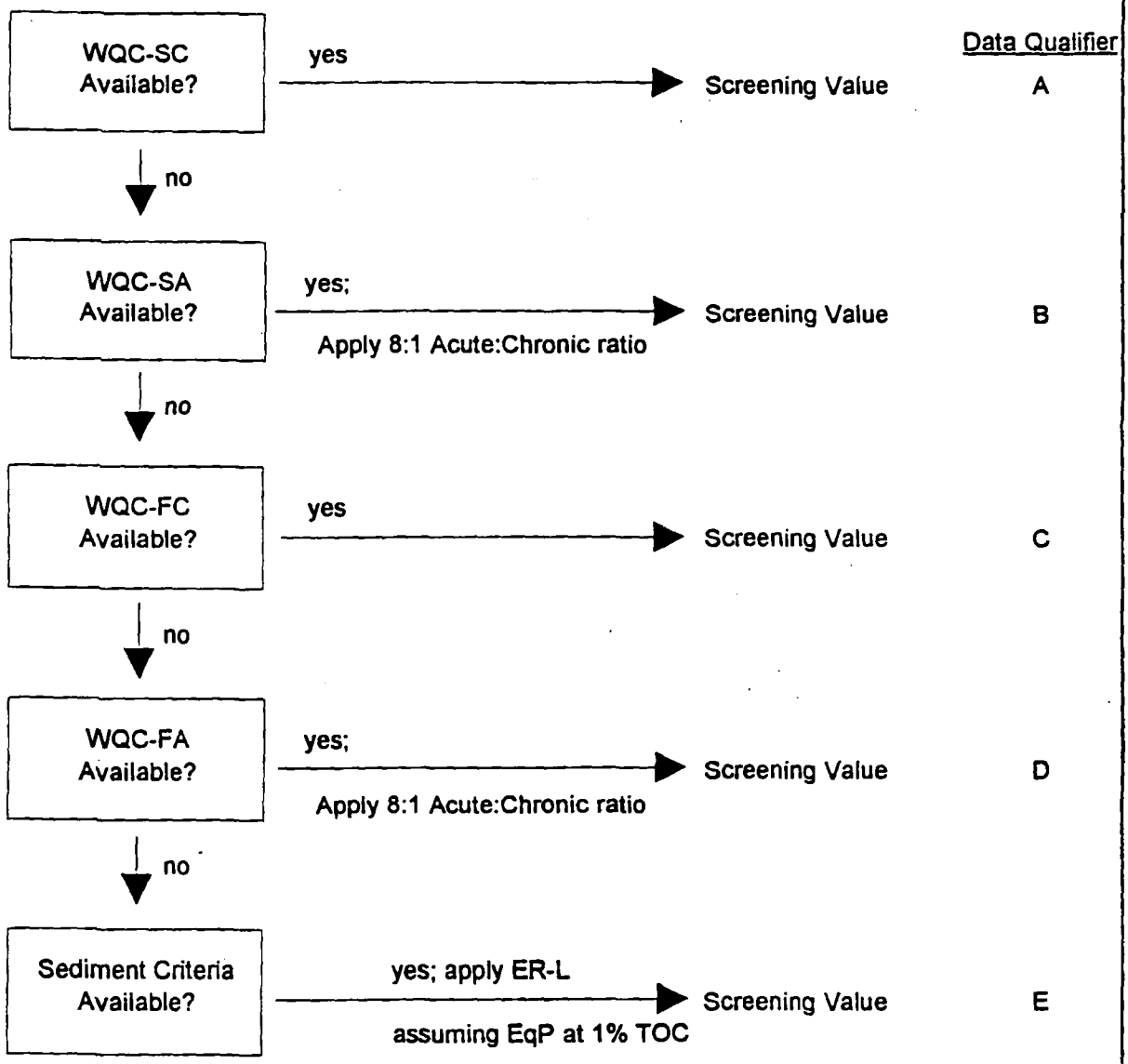


Figure 6.2-2. Plot of fish (*Fundulus* spp.) length versus simultaneously extracted metals for Allen Harbor. Regression line and upper/lower 95% confidence limits fitted to the data.

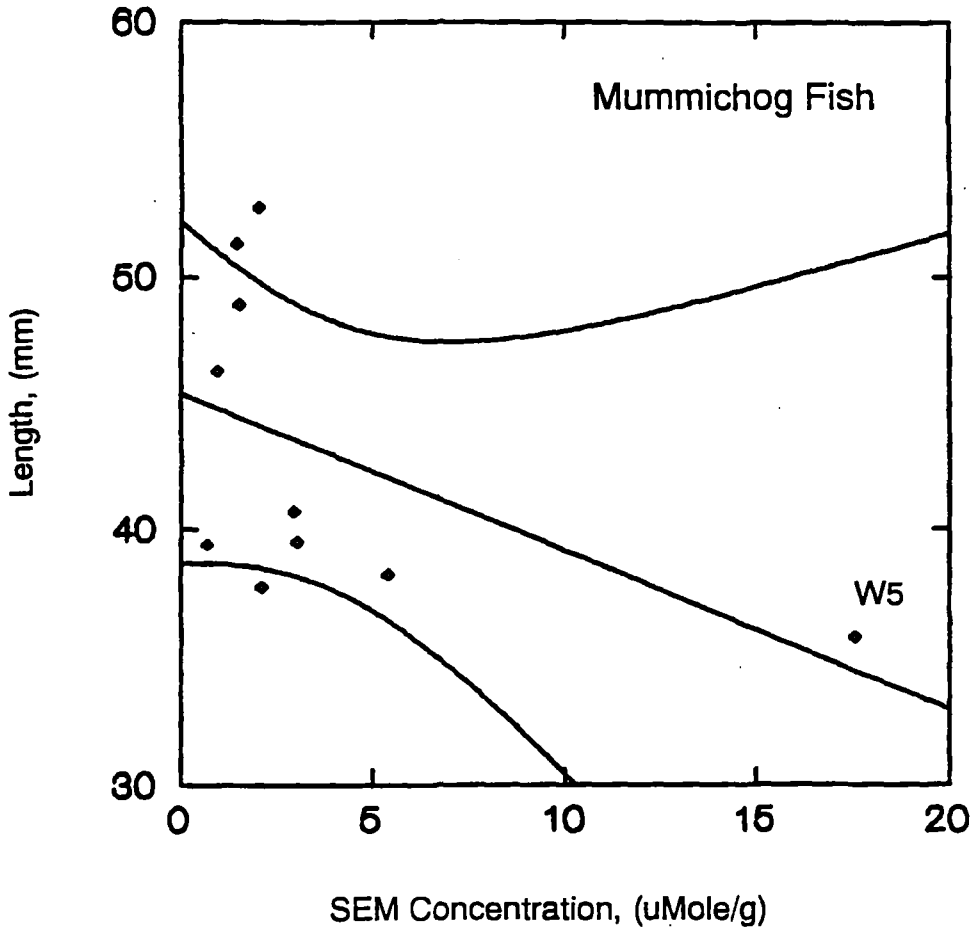


Figure 6.2-3. Plot of hard shell clam condition versus fluorene sediment Hazard Quotients for Allen Harbor. Regression line and upper/lower 95% confidence limits fitted to the data.

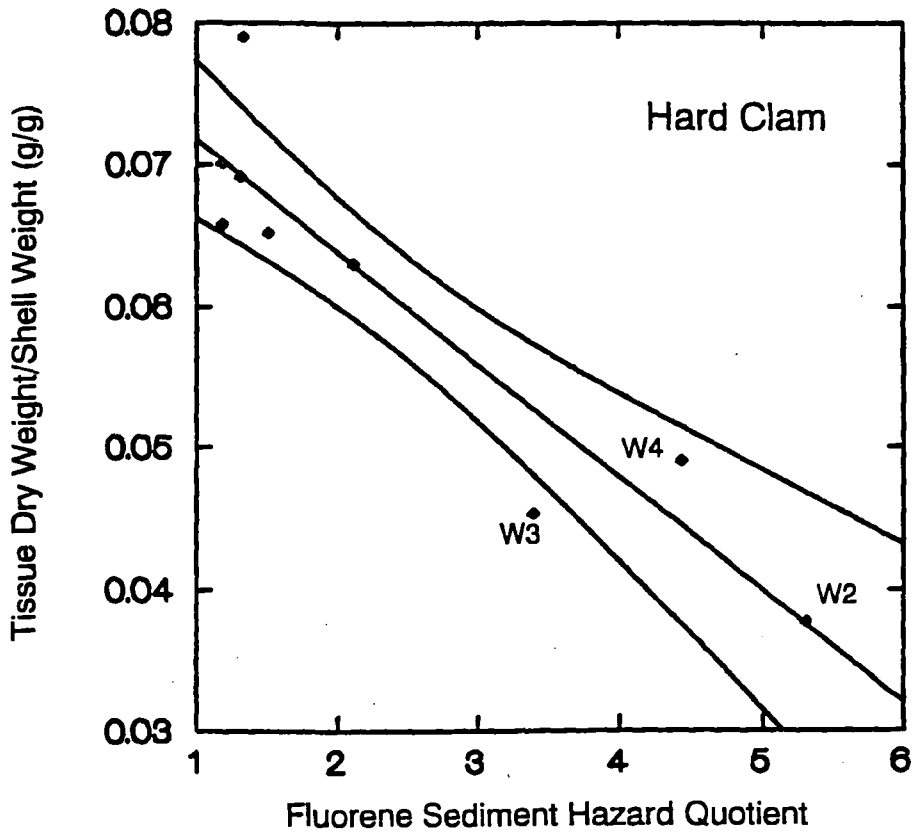


Figure 6.2-4. Plot of oyster condition versus total PCB sediment Hazard Quotients for Allen Harbor. Regression line and upper/lower 95% confidence limits fitted to the data.

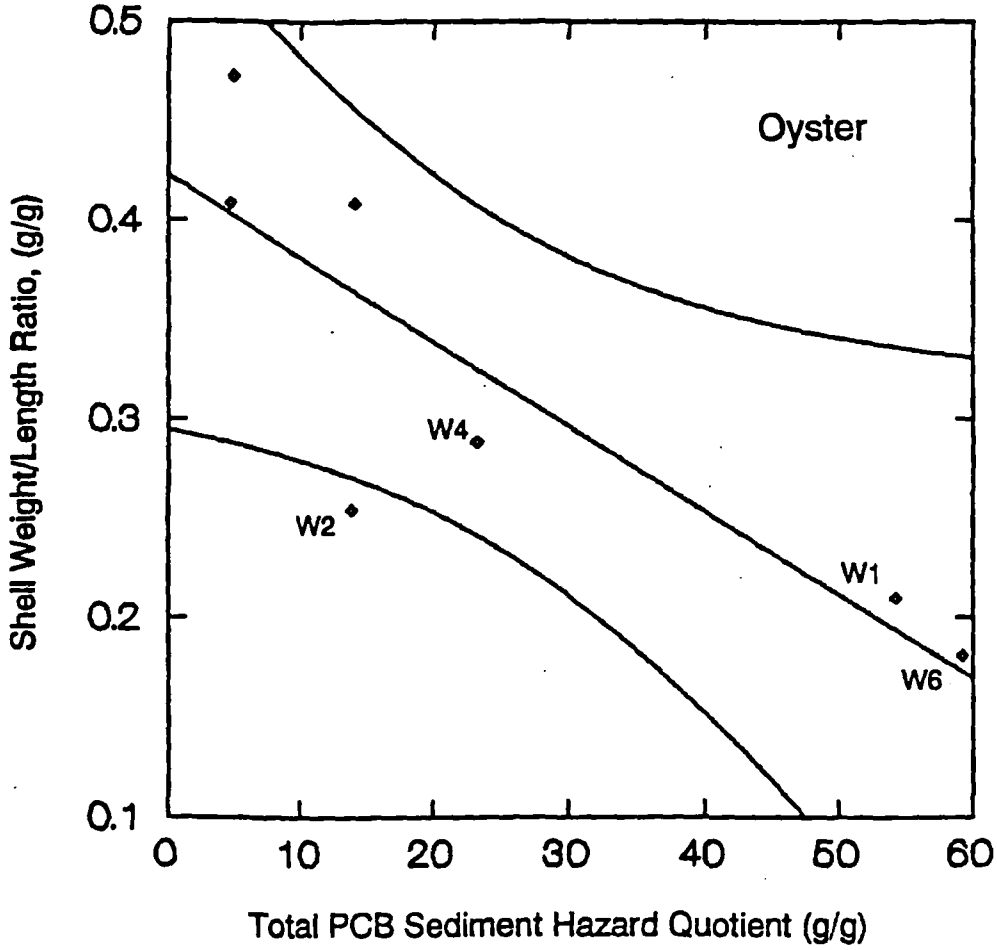


Figure 6.2-5. Plot of soft shell clam condition versus total high molecular weight PAH pore water Hazard Quotients for Allen Harbor. Regression line and upper/lower 95% confidence limits fitted to the data.

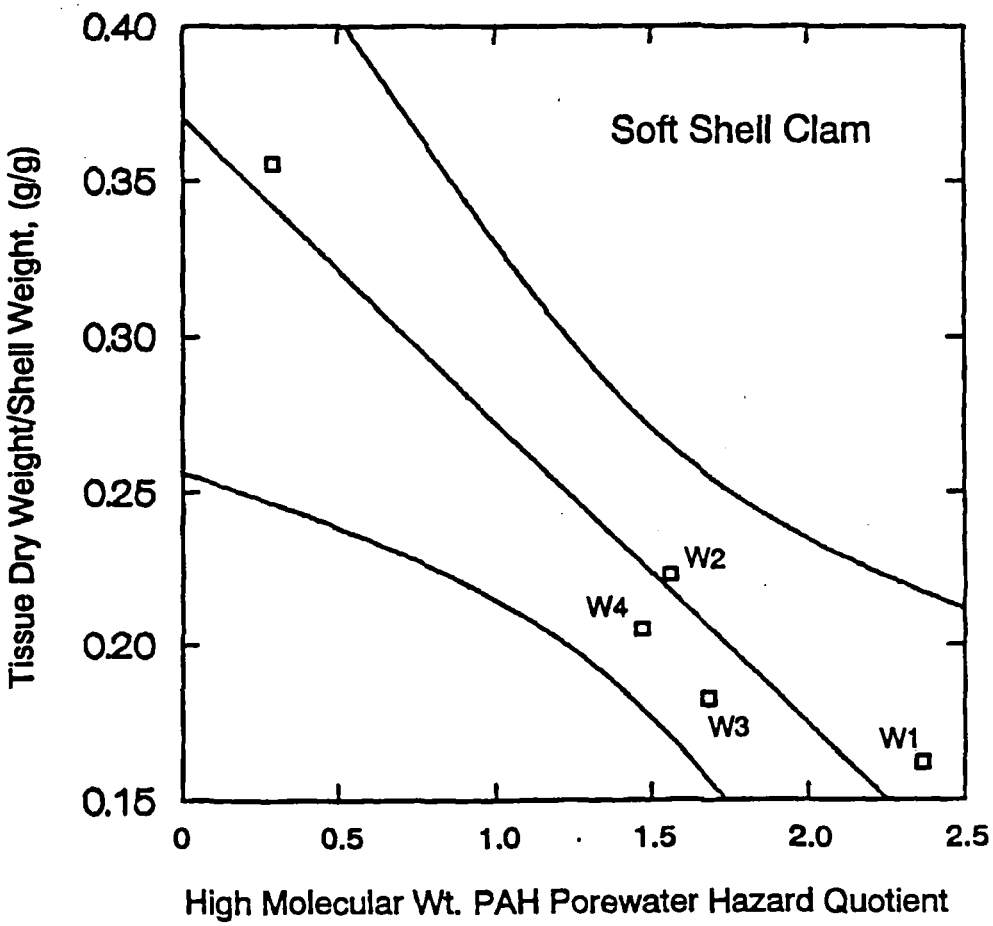


Figure 6.3-1 Tissue/sediment concentration relationships in bivalves A) soft shell clam - DDE, B) soft shell clam - PAH, C) hard shell clam - PCB, D) ribbed mussel - DDE, and E) ribbed mussel - PAH (log normalized).

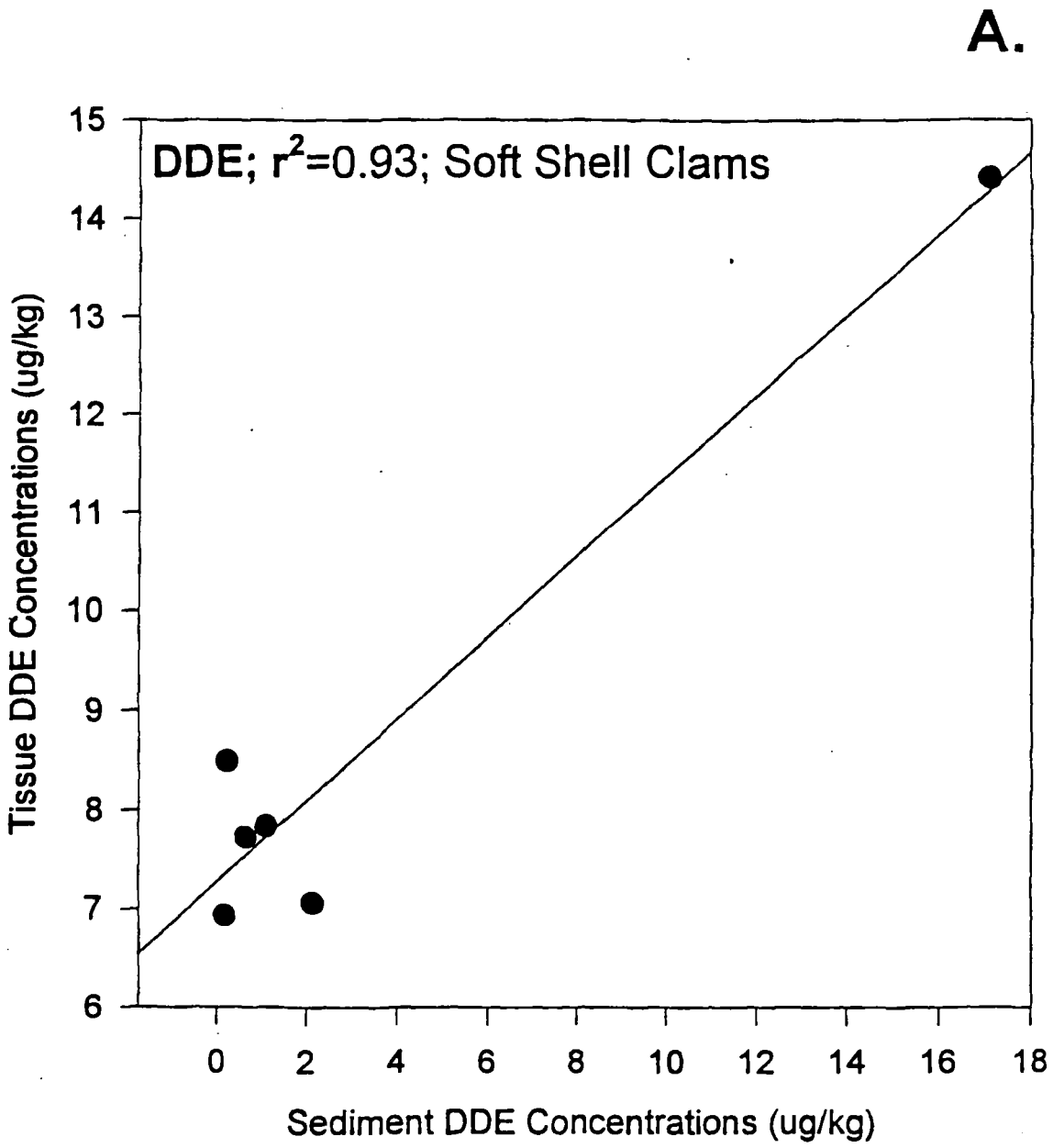


Figure 6.3-1. (con't)

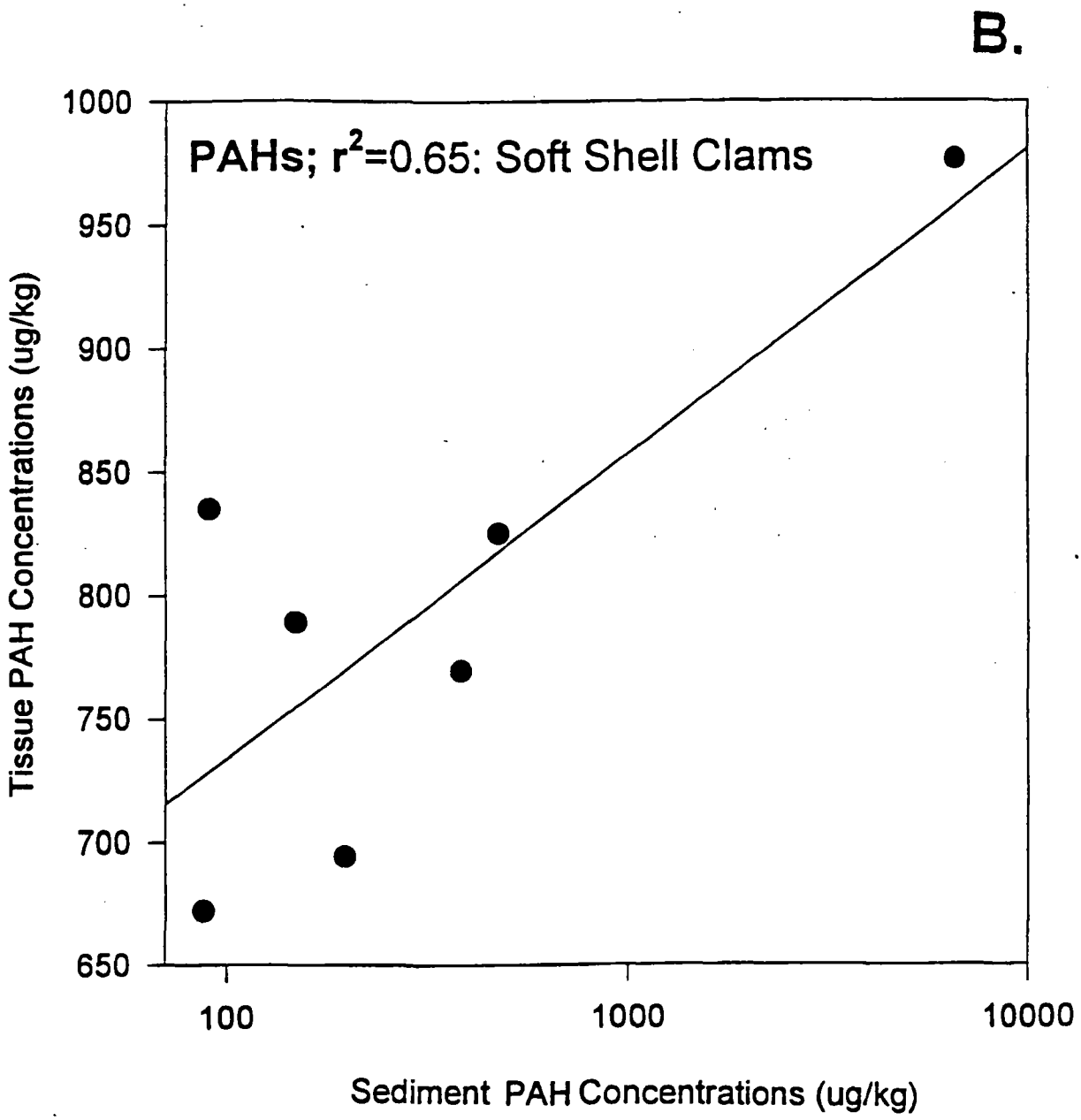


Figure 6.3-1. (con't)

C.

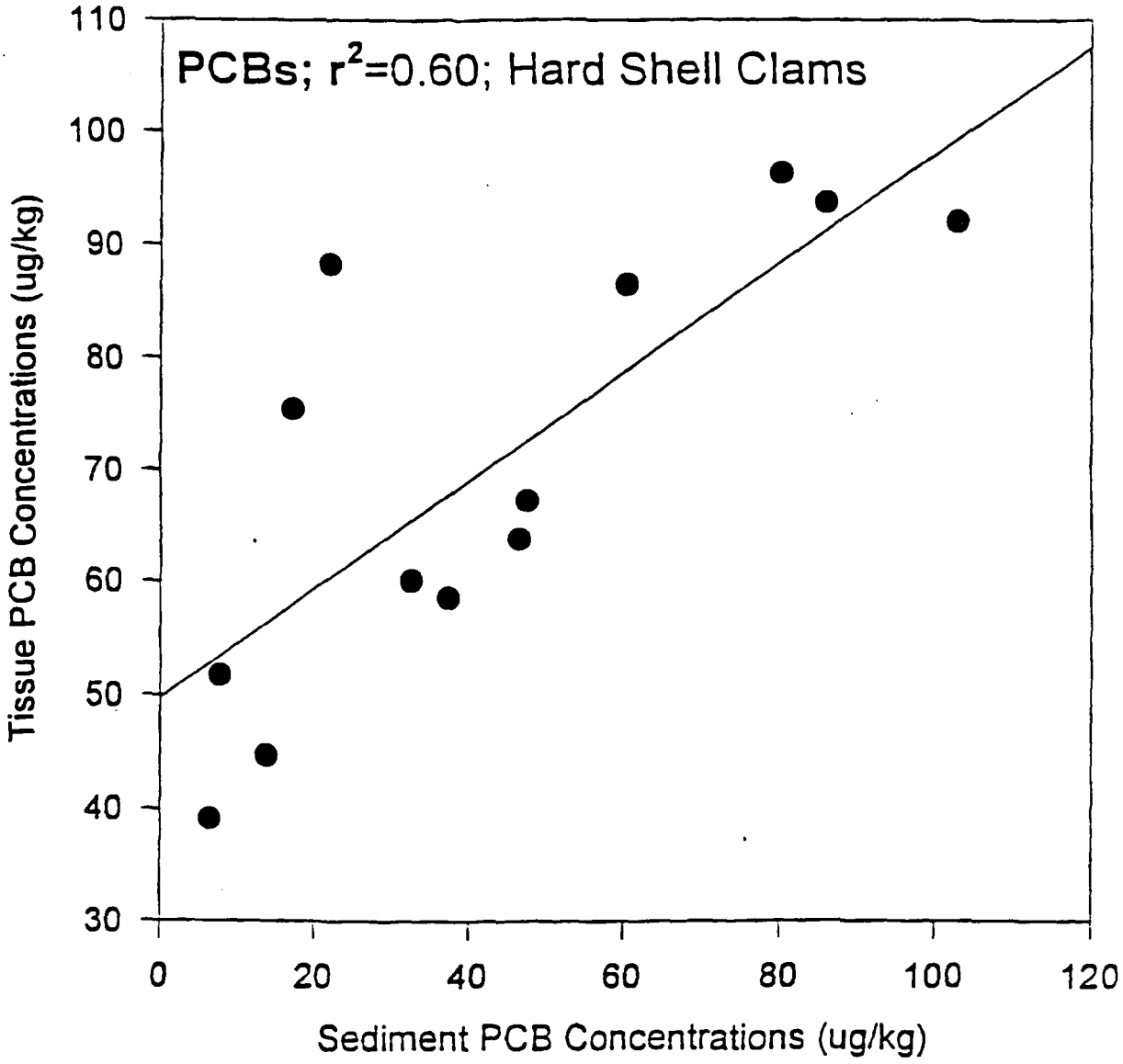


Figure 6.3-1. (con't)

D.

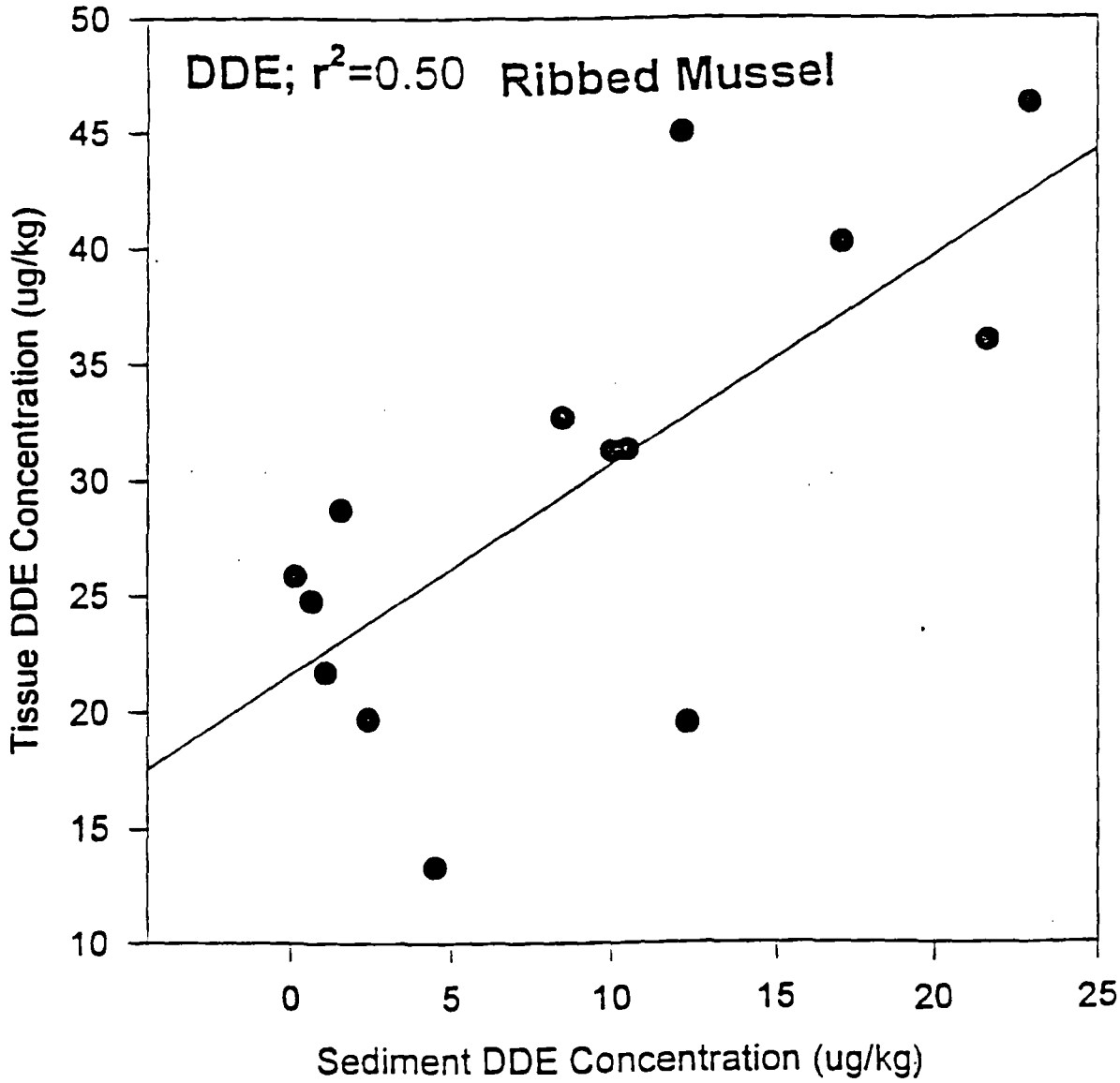


Figure 6.3-1. (con't)

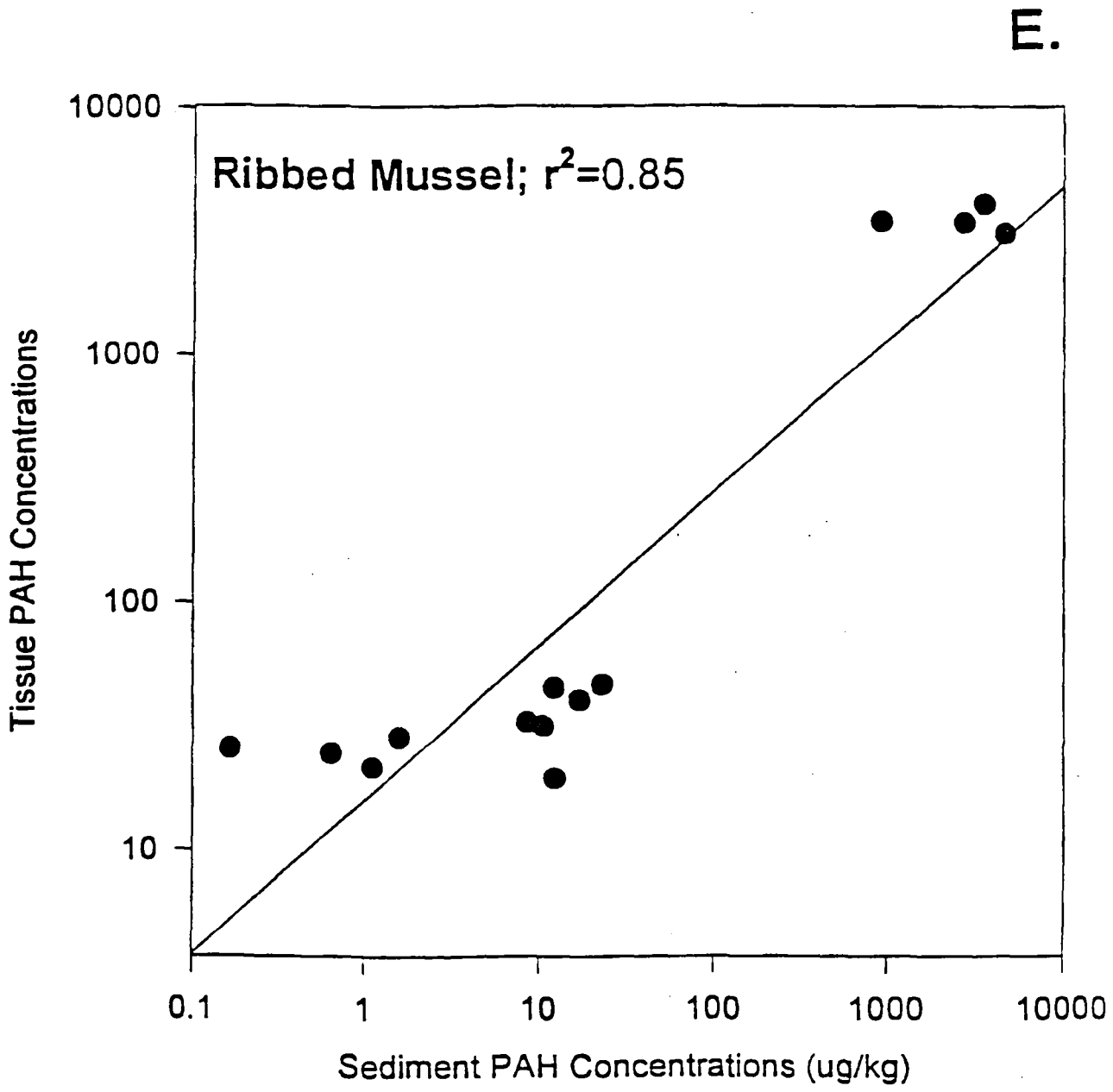


Figure 6.3-2. Biota-Sediment Accumulation Factors (µg lipid/µg organic carbon) for depurated bivalves exposed to A) total PAHs (tPAHs), B) total PCBs (tPCBs), and C) the pesticide p'p'-DDE (DDE).

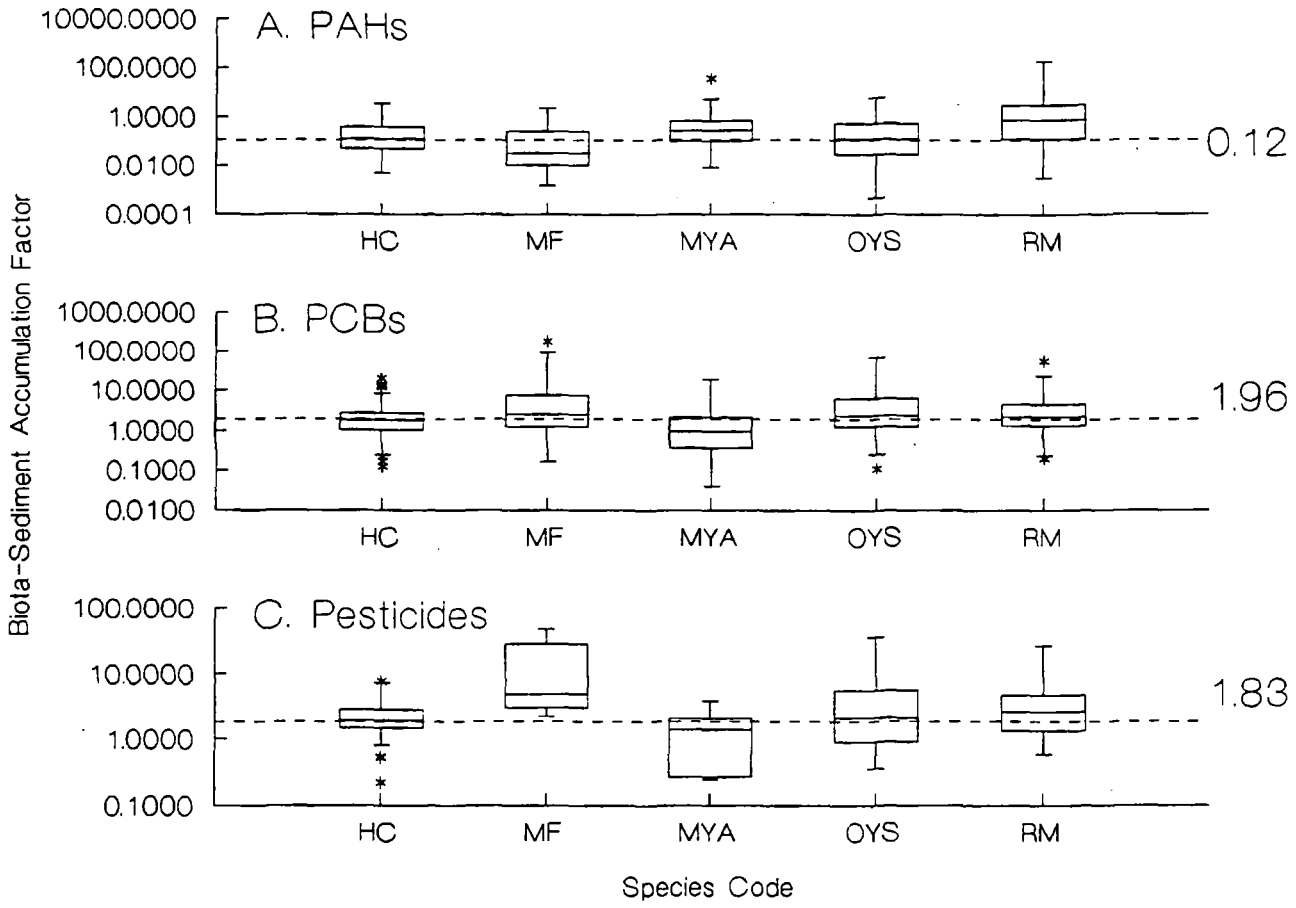


Figure 6.3-3. Biota Accumulation Factors for metals in bivalves and fish species of Allen harbor.

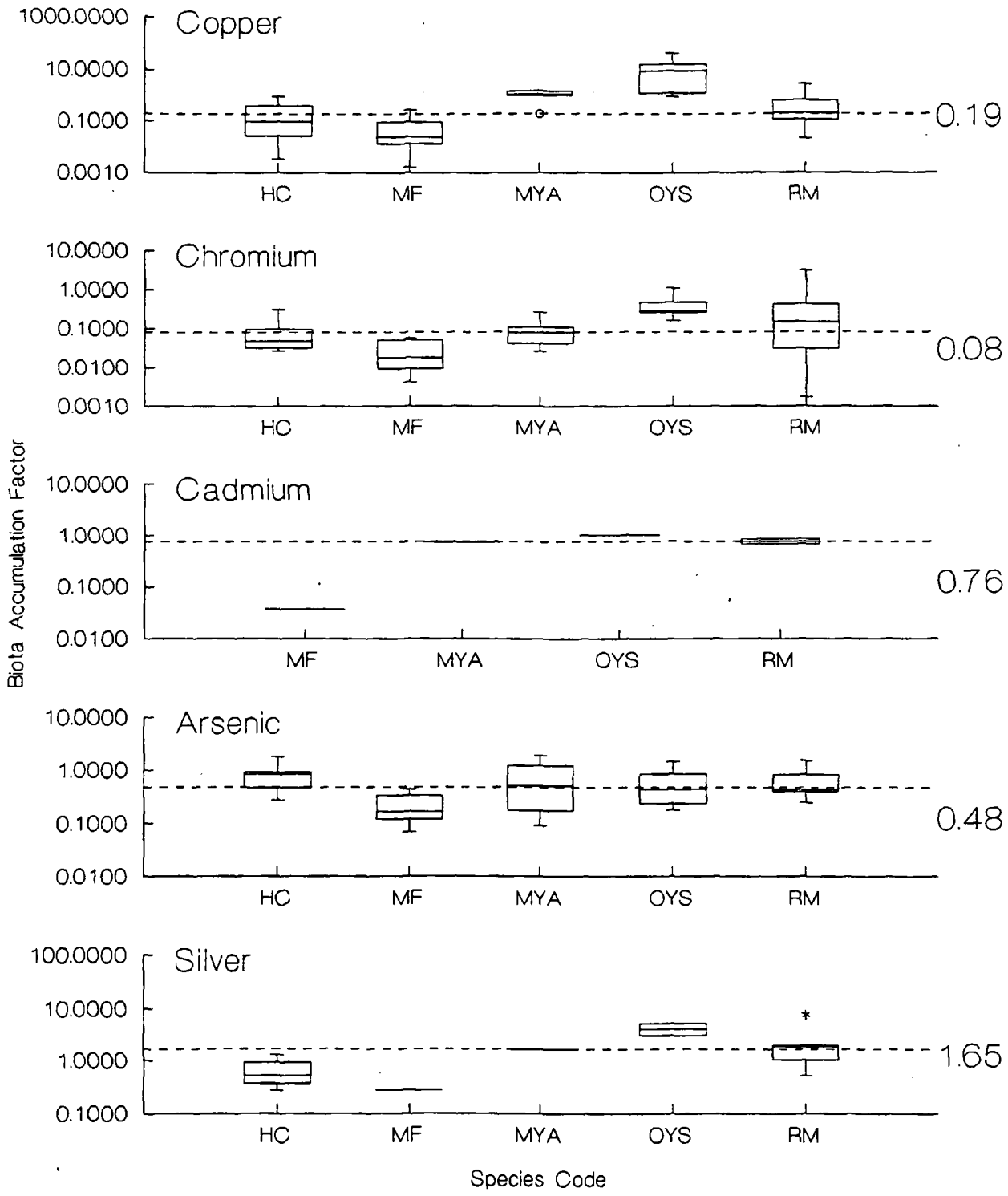


Figure 6.3-3 (con't)

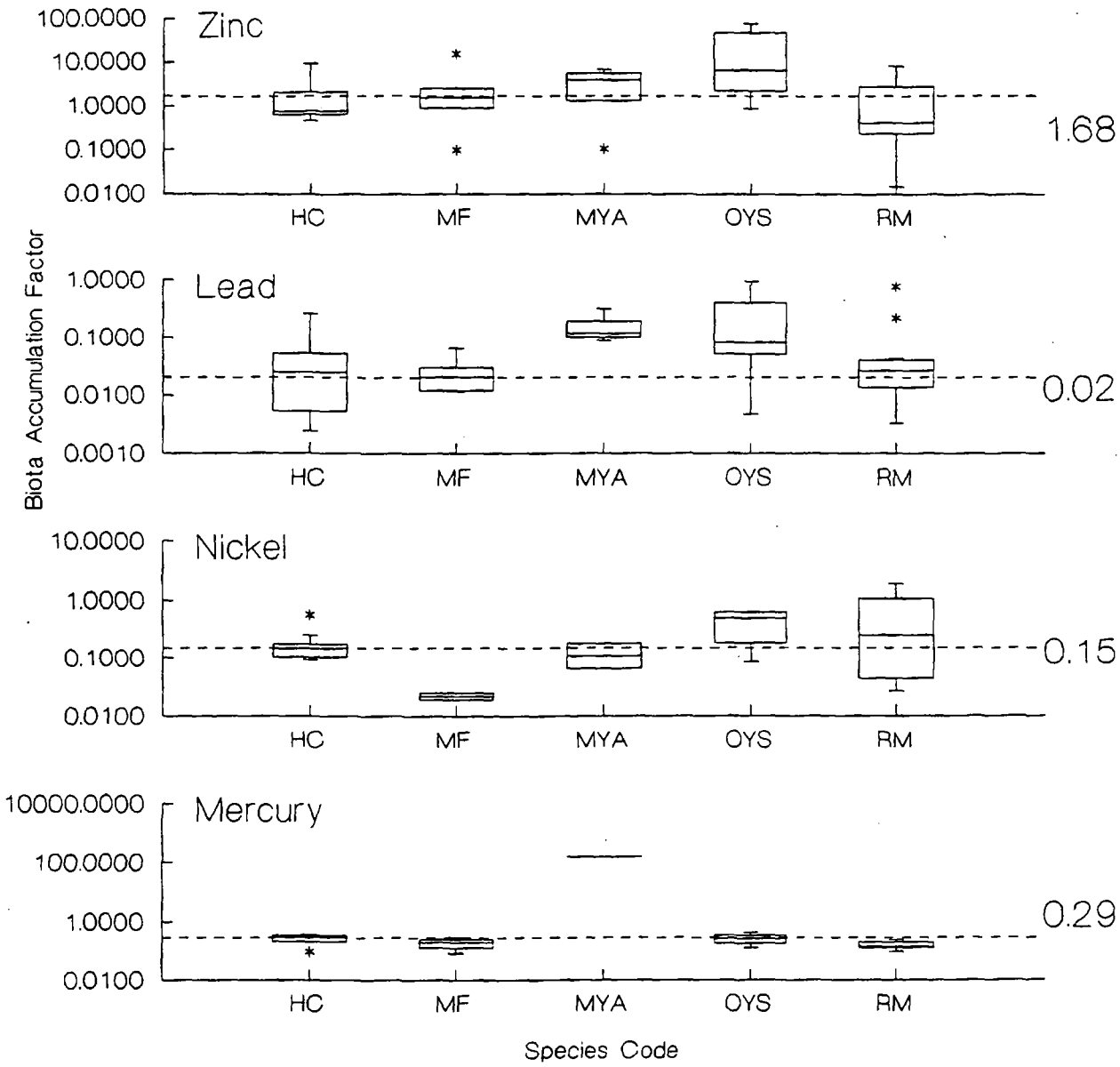


Figure 6.3-4. Biota-Sediment Accumulation Factors for depurated bivalves exposed to sediment organics.

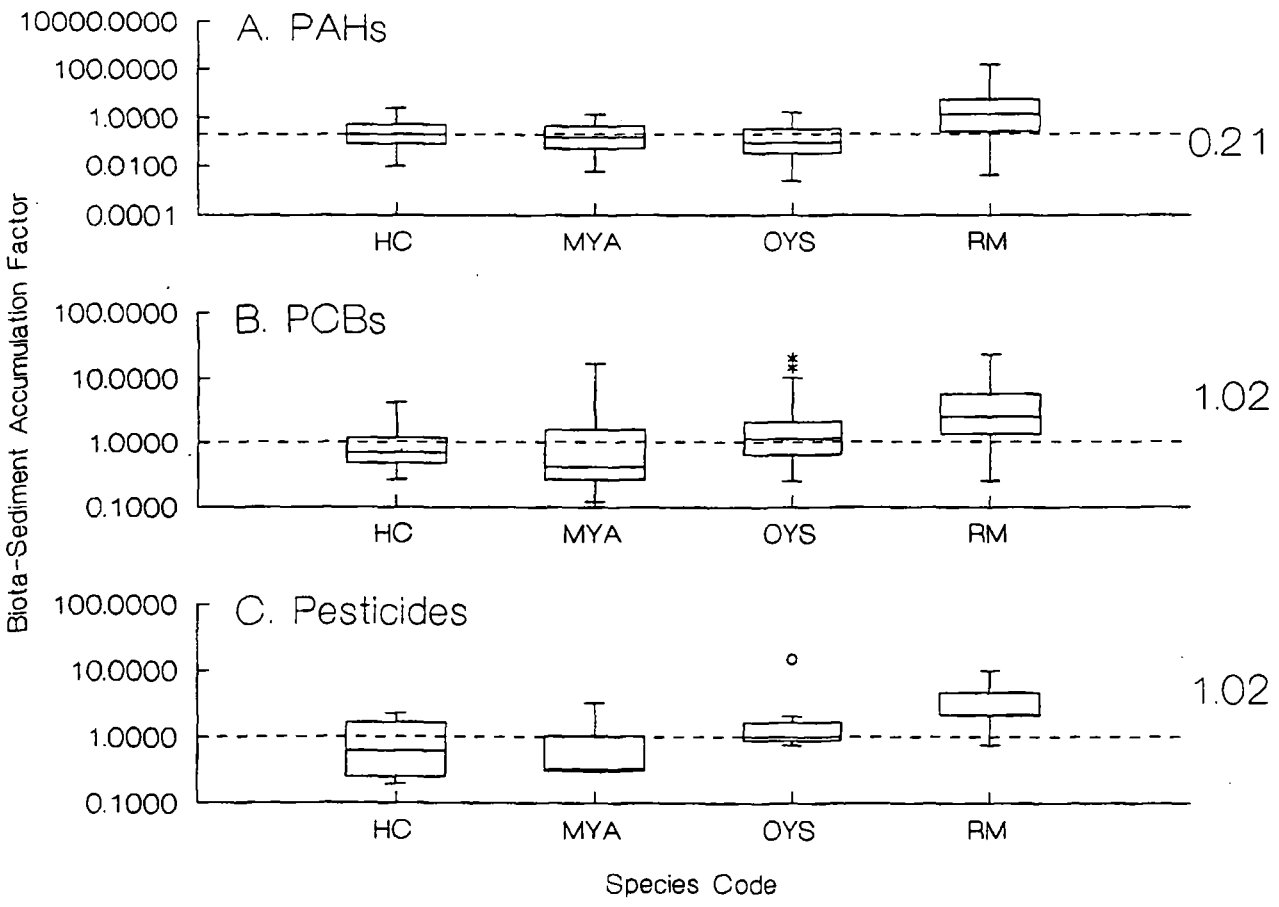


Figure 6.3-5. Ratio of depurated to non-depurated tissue concentration for bivalves.

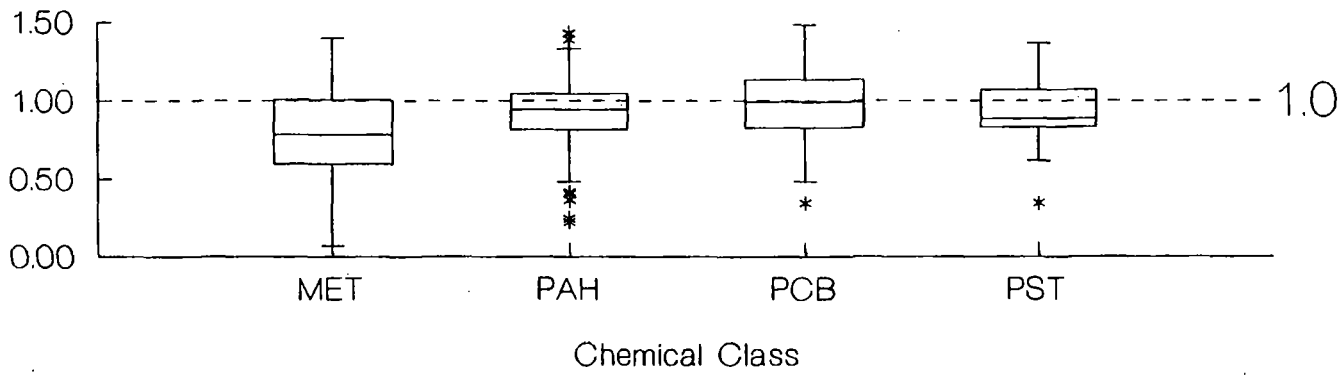


Figure 6.4-1. Plot of amphipod (*Ampelisca*) survival vs. A) SEM/AVS, B) SEM-AVS, and C) SEM sediment concentration.

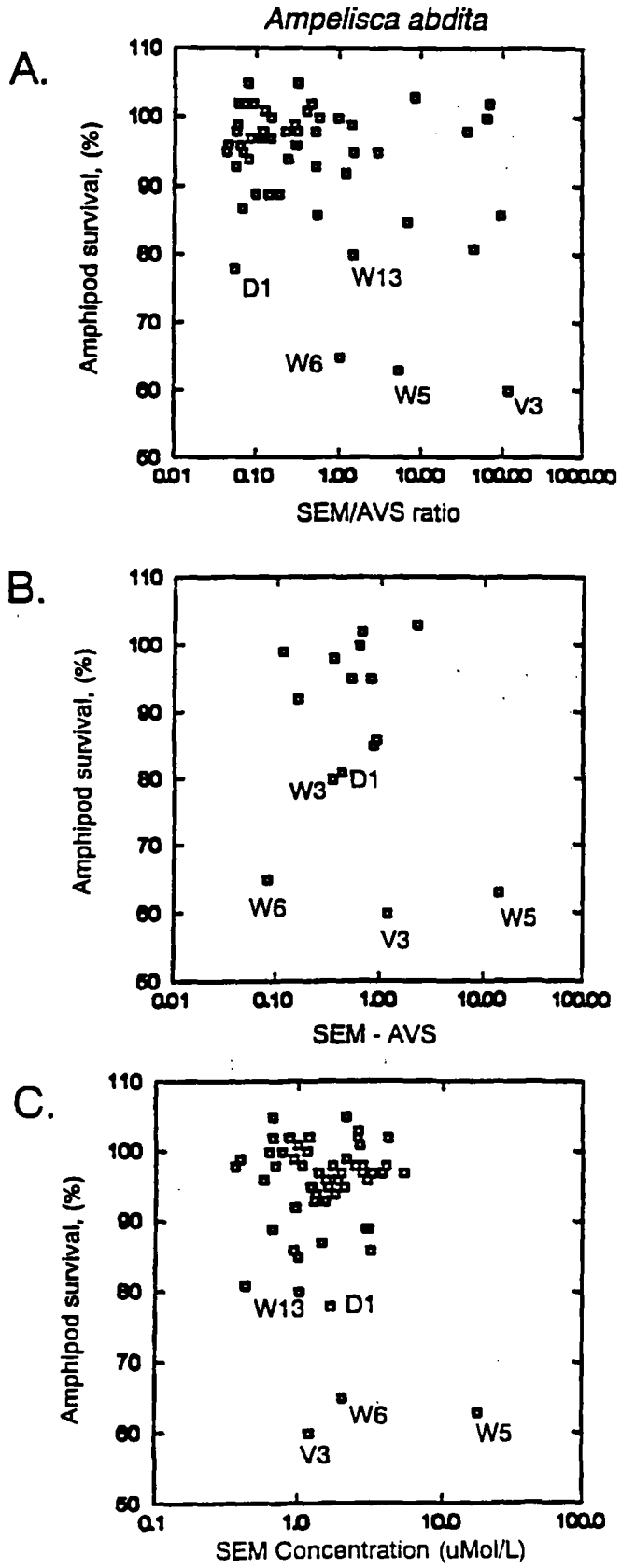


Figure 6.4-2. Plot of amphipod *(Ampelisca)* survival vs. A) DDE and B) PCB and C) PAH concentrations in test sediments. Stations are listed that have relatively high toxicity and SEM/AVS values or trends.

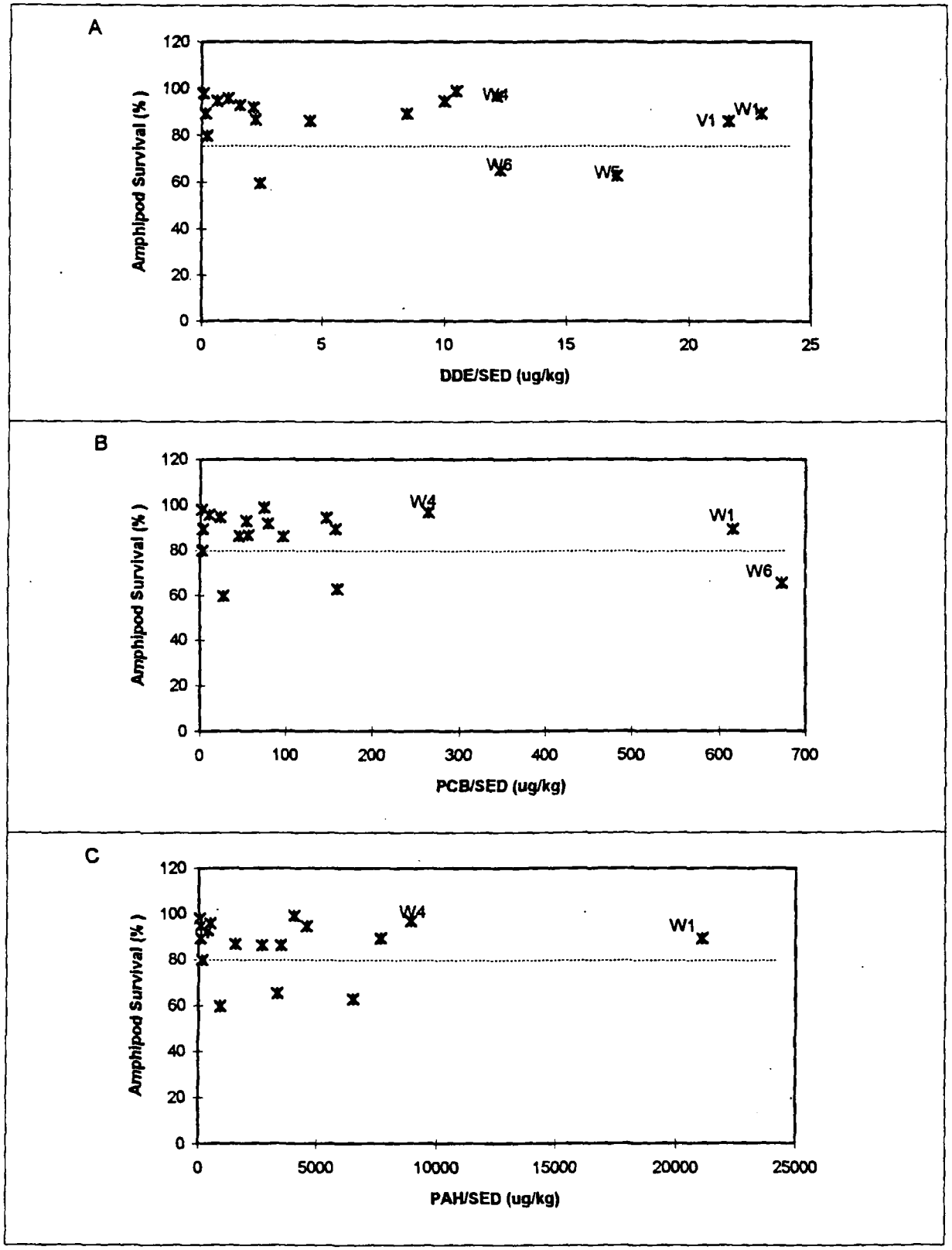


Figure 6.4-3. Plot of amphipod (*Ampelisca*) survival (%) vs. pore water PCB Hazard Quotients.

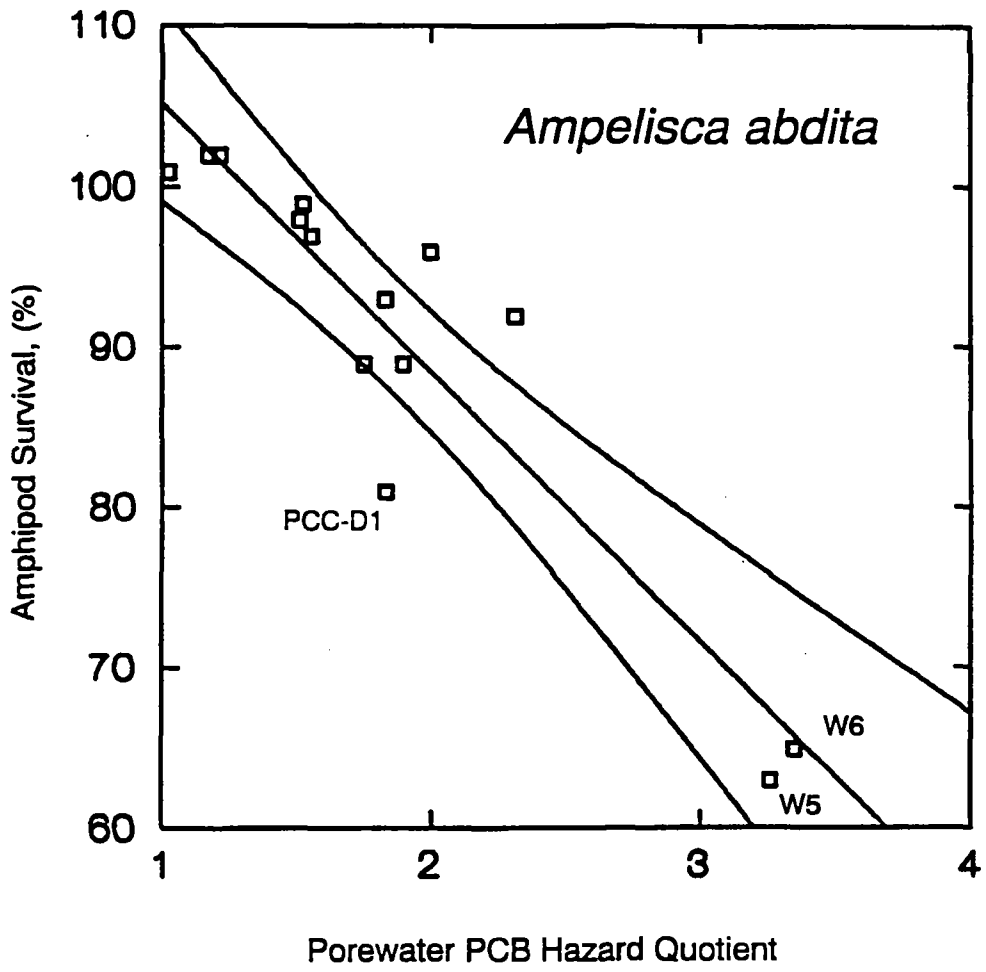


Figure 6.4-4. Plot of amphipod (*Ampelisca*) mortality (%) vs. interstitial water toxic units for PAHs.

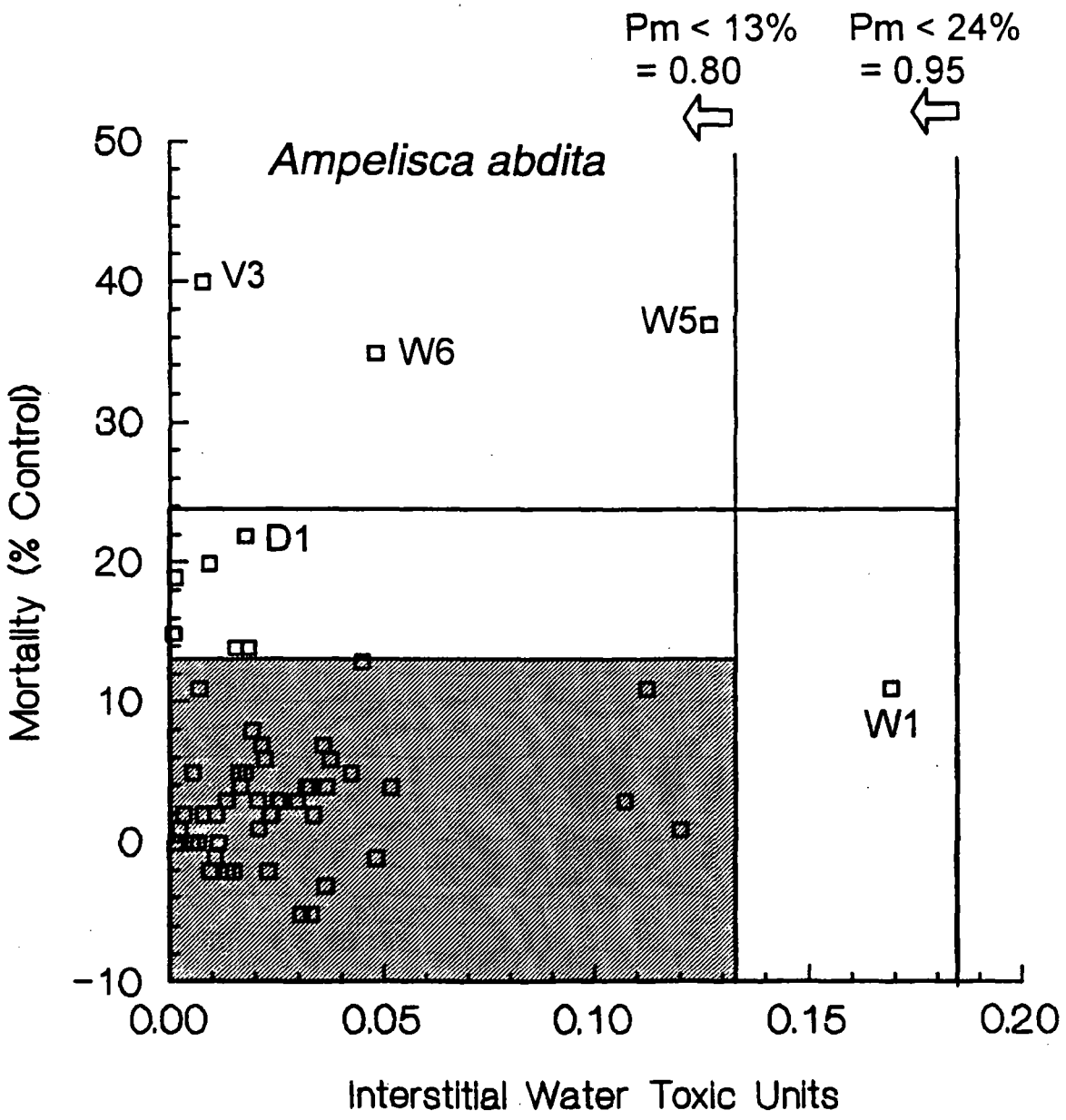


Figure 6.4-5. Plot of sea urchin (*Arbacia*) fertilization success vs. A) SEM/AVS, B) SEM-AVS, and C) SEM sediment concentration.

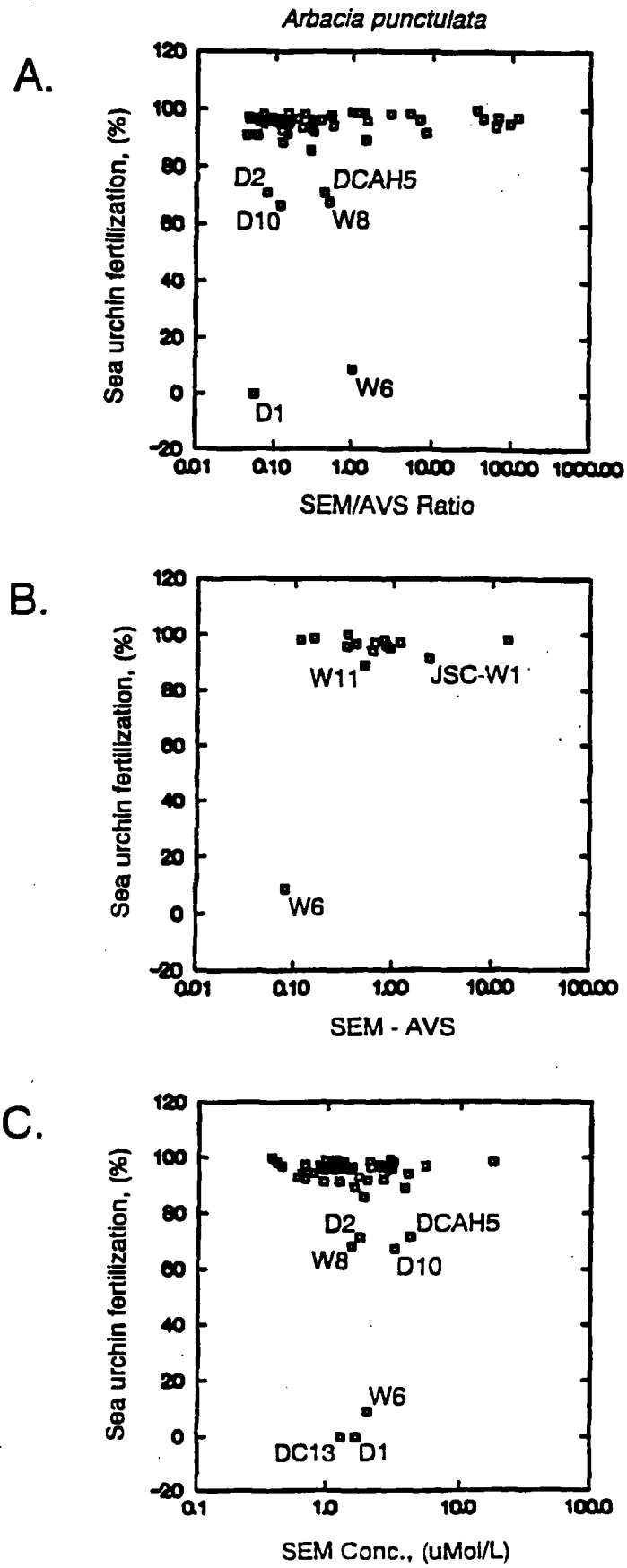
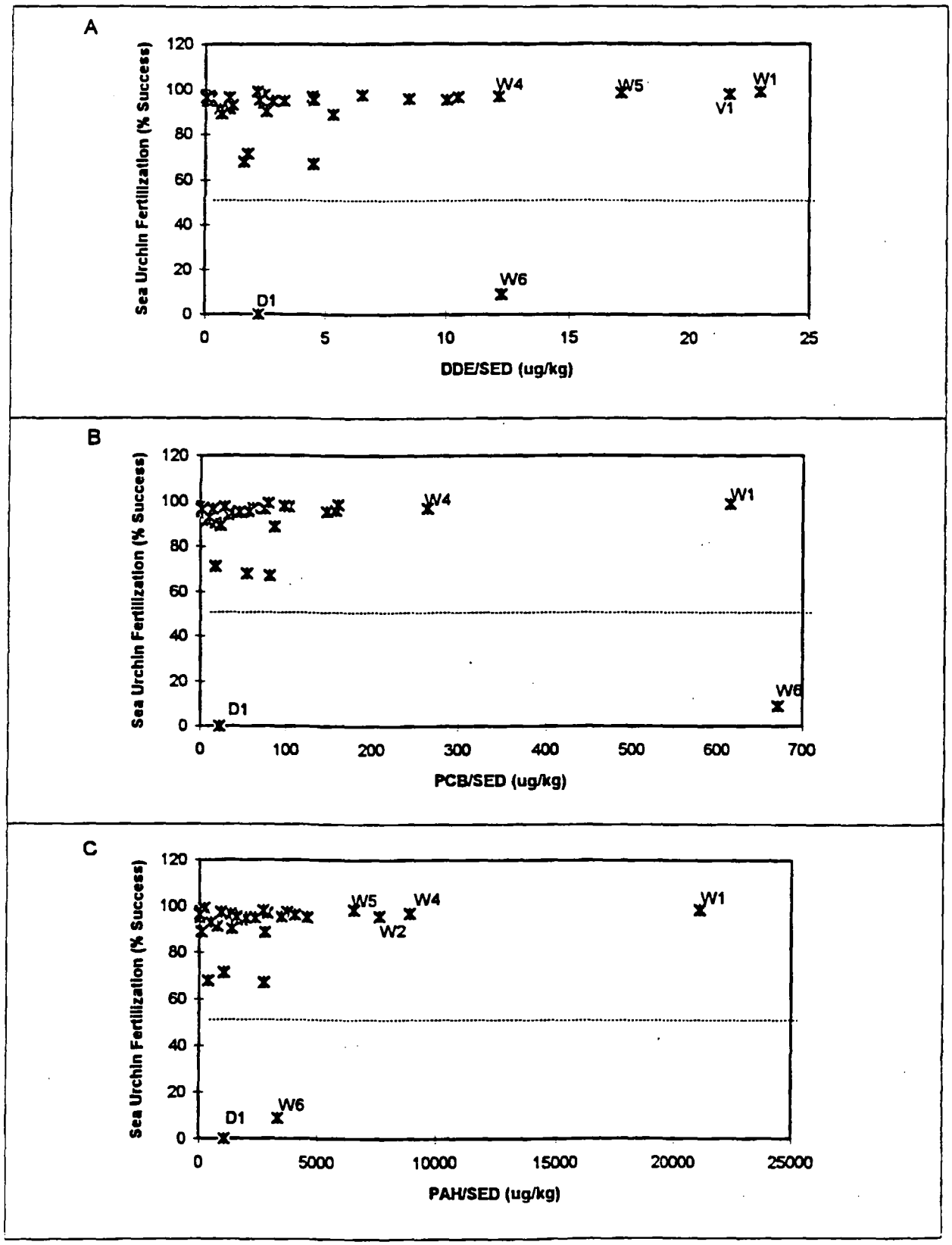


Figure 6.4-6. Plot of sea urchin *(Arbacia)* fertilization success (%) vs. A) DDE and B) PCB and C) PAH concentrations in test sediments. Stations are listed that have relatively high toxicity and SEM/AVS values or trends.



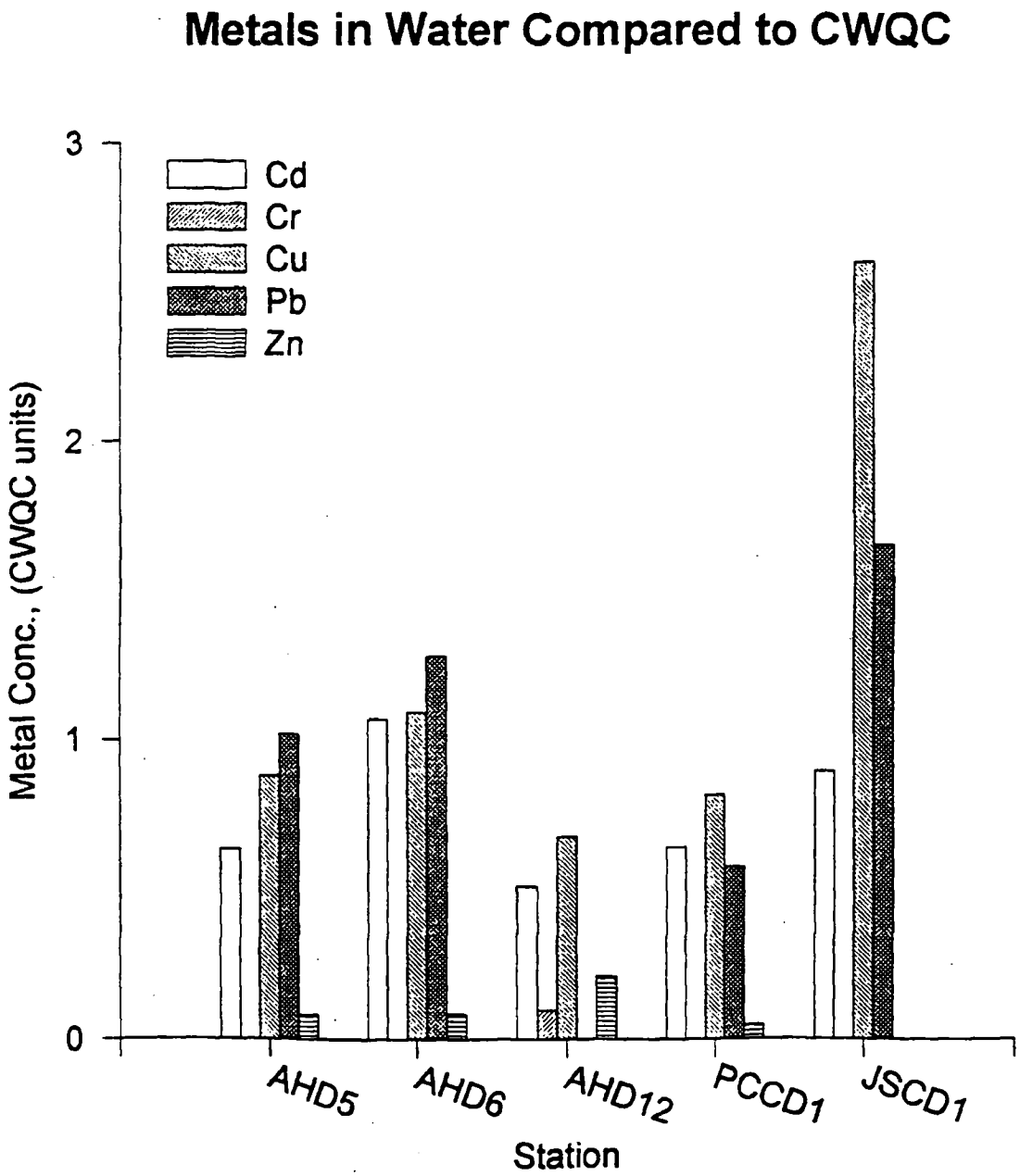


Figure 6.5-1. Hazard quotients for metals in Allen Harbor surface waters compared to the Water Quality Criteria-Chronic Values.

Figure 6.6-1. Plot of hematopoietic neoplasia (% incidence) in soft shell clams vs. total PAH Body residue concentration. Lines shown are the linear fit and upper/lower 95% confidence limits of the data.

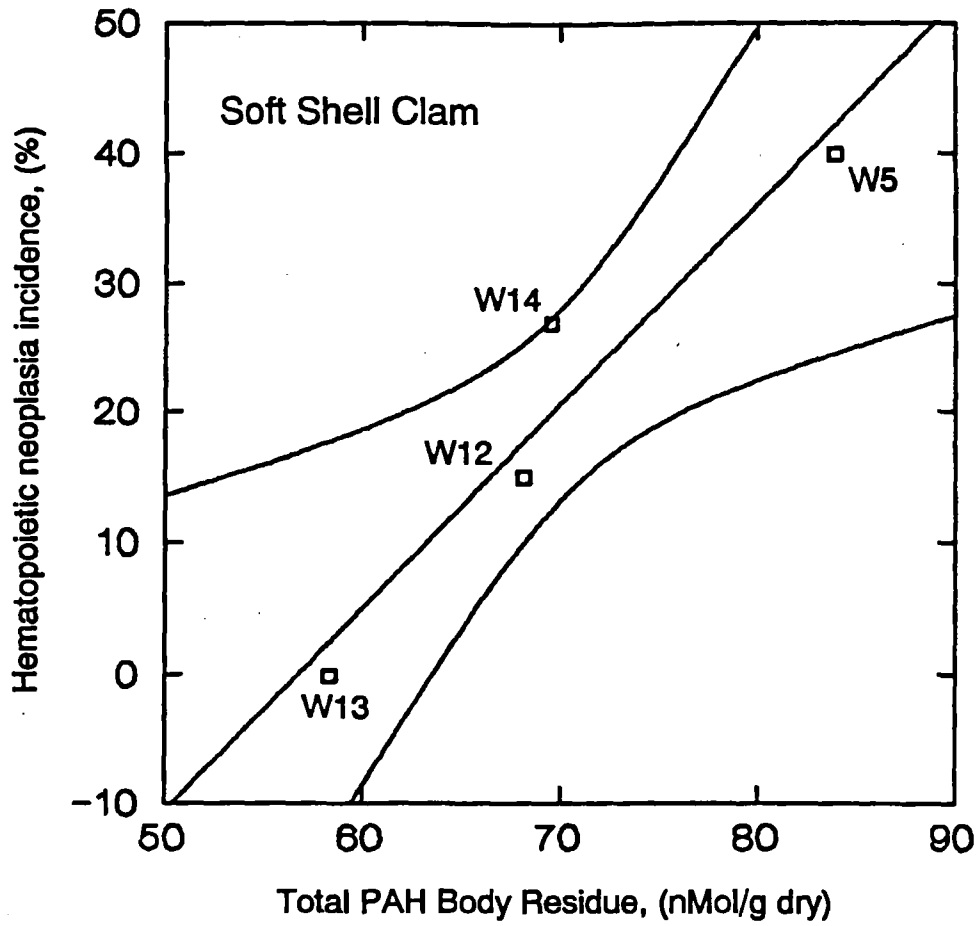


Figure 6.6-2. Plot of biota condition vs the Tissue residue Hazard Quotient for zinc: A) Shell weight to length ratios in oysters; B) mummichog fish length. Lines shown are the linear fit and upper/lower 95% confidence limits of the data.

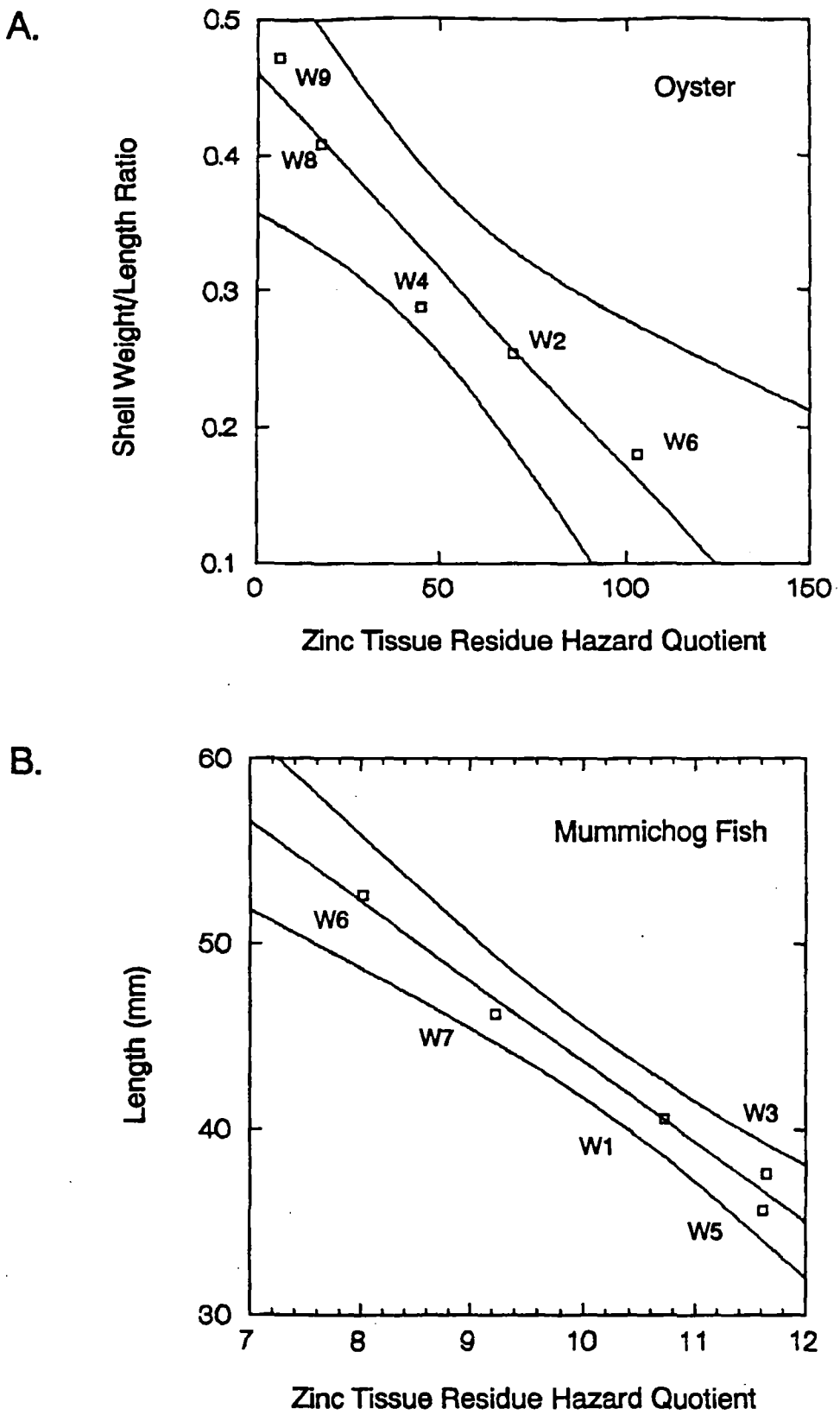


Table 6.1-1. Site/reference concentration ratio statistics for CoCs in Allen Harbor sediments summarized by ecological exposure zones (EEZ).

Group[2]	EEZ	Mean CR	Min. CR	Max. CR	Ranking[1]
A. Total PCBs	LS-V/W	12.5	5.2	21.8	+++
	LS-D	1.9	1.5	2.5	-
	LM-D	4.1	1.6	11.8	++
	LN-W	19.6	0.7	45.1	+++
	CP-W	3.7	1.7	6.4	+
	CP-SW	1.6	0.8	2.1	-
	CP-SD	0.8	0.8	1.0	-
	MAR	11.3			+
B. Pesticide Analytes	LS-V/W	7.4	0.2	23.7	++
	LS-D	5.3	0.3	19.7	++
	LM-D	12.1	0.3	42.0	+++
	LN-W	3.7	0.0	9.8	+
	CP-W	6.6	0.4	24.7	++
	CP-SW	2.2	0.3	5.8	+
	CP-SD	19.3	0.1	88.3	+++
	MAR	8.4			+
C. PAH Analytes - High Molecular Weight	LS-V/W	3.8	2.9	5.9	+
	LS-D	15.7	8.1	29.0	+++
	LM-D	22.0	15.9	31.3	+++
	LN-W	0.8	0.6	1.2	-
	CP-W	8.3	1.4	13.3	++
	CP-SW	0.9	0.6	1.5	-
	CP-SD	11.6	6.1	28.4	+++
	MAR	76.0			+
PAH Analytes - Low Molecular Weight	LS-V/W	4.9	1.4	20.7	++
	LS-D	5.6	0.4	11.4	++
	LM-D	9.4	0.8	20.7	++
	LN-W	1.0	0.4	2.7	-
	CP-W	2.4	0.5	5.0	+
	CP-SW	1.0	0.6	1.9	-
	CP-SD	6.1	0.5	13.4	++
	MAR	23.3			+
PAH Analytes - Total	LS-V/W	4.3	1.4	20.7	++
	LS-D	11.1	0.4	29.0	+++
	LM-D	16.3	0.8	31.3	+++
	LN-W	0.9	0.4	2.7	-
	CP-W	5.6	0.5	13.3	++
	CP-SW	1.0	0.6	1.9	-
	CP-SD	9.1	0.5	28.4	++
	MAR	67.9			+
D. VOA Analytes	LS-V/W	2.0	1.0	8.4	+
	LS-D				-
	LM-D				-
	LN-W	2.8	0.3	55.8	++
	CP-W	1.5	0.8	18.7	-
	CP-SW	1.2	0.6	1.8	-
	CP-SD				-
	MAR				-
E. Metal Analytes	LS-V/W	20.8	1.1	89.1	+++
	LS-D	5.5	0.8	30.0	++
	LM-D	22.4	1.1	140.3	+++
	LN-W	5.8	0.3	23.5	++
	CP-W	21.5	0.4	132.0	+++
	CP-SW	2.8	0.4	11.2	++
	CP-SD	7.5	1.0	42.4	++
	MAR	27.1			+
F. Butyltin analytes	LS-V/W	0.8	0.1	1.6	-
	LS-D	4.4	1.4	9.9	+
	LM-D	7.8	5.8	9.2	+
	LN-W	0.6	0.4	0.9	-
	CP-W	0.9	0.1	1.4	-
	CP-SW	0.8	0.3	1.3	-
	CP-SD	3.4	1.0	6.6	+
	MAR	42.7			+

[1]Ranking Notation: + = Mean CR > 2; '++ = Mean CR > 2 and Max. CR > 10; +++ = Mean CR > 10

[2]Organic contaminants are TOC-normalized.

nd =not determined

Table 6.1-2. Qualitative Summary of Sediment CoC concentration ratios (CR) by Ecological Exposure Zone.

	Sediment CoC Enrichment Factor							
	Ecological Exposure Zone[1,3]							
Measurement Endpoint	LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR
Total PCB CRs	+++	-	++	+++	+	-	-	+
Pesticide CRs	++	++	+++	+	++	+	+++	+
PAH CRs	++	+++	+++	-	++	-	++	+
VOA CRs	+	-	-	++	-	-	-	-
Metal CRs	+++	++	+++	++	+++	++	++	+
Butyltin CRs	-	+	+	-	-	-	+	+
EEZ Score[1]	++	++	+++	++	++	+	+	+

[1]Ranking Notation: + = Mean CR > 2; '++ = Mean CR > 2 and Max. CR > 10; +++ = Mean CR > 10

[2]EEZ Scoring (based on + = 1 pt, ++ = 3 pt; +++ = 5 pt): 4-10 pt = slight risk (+), 11-20 pt = moderate risk (++); > 20 pt = substantial risk

[3]Consult Section 6.0 and Figure 6.0-1 for description and location of EEZs.

na = not applicable

TABLE 6.2-1. Summary of Significant Pearson Correlation (r) and Probability (P>F_≤0.05) Values for Biota Condition (Shell weight: Length (SHELLWTL), Tissue weight: Shell length (TISDRYLE) and Tissue weight: Shell weight (TISDRYSH) ratios) vs. Sediment Concentrations.

COMPGRP	COMPARISON	SSP[1]	ANALYTE	SHELLWTL			TISDRYLE			TISDRYSH		
				r	P	n	r	P	n	r	P	n
MET	HQ	HC	Mercury							-0.720	0.029	9
		MYA	Silver							-0.954	0.046	4
PAH	RAW	MYA	Acenaphthene				-0.943	0.016	5			
			Acenaphthylene				-0.911	0.031	5			
			Fluorene				-0.931	0.022	5			
			Low Molecular Weight PAH				-0.904	0.035	5			
	HQ	HC	Acenaphthene							-0.815	0.025	7
			Fluorene							-0.929	0.000	9
			Phenanthrene							-0.846	0.034	6
		MYA	Acenaphthene									
			Chrysene							-0.999	0.027	3
PCB	RAW	MYA	PCB128				-0.879	0.050	5			
	HQ	OYS	Total PCBs	-0.853	0.015							
VOA	RAW	MYA	Benzene	-0.948	0.014	5	-0.874	0.052	5			
			Chloromethane	-0.916	0.029	5	-0.920	0.027	5			
			2-Hexanone	-0.902	0.036	5	-0.907	0.033	5			
			Methylene Chloride	-0.838	0.076	5	-0.883	0.047	5			

[1]HC = Hard Shell Clam
MYA = Soft Shell Clam
OYS = Oyster
RM = Ribbed Mussel
MF = Mummichog Fish

TABLE 6.2-2. Summary of Significant Pearson Correlation (r) and Probability (P>F ≤0.05) Values for Biota Condition (Shell weight: Length (SHELLWTL), Tissue weight: Shell length (TISDRYLE) and Tissue weight:Shell weight (TISDRYSH) ratios) vs. Pore Water Concentrations.

				Condition Index								
COMPGRP	COMPARISON	Spp[1]	ANALYTE	SHELLWTL			TISDRYLE			TISDRYSH		
				r	P	n	r	P	n	r	P	n
Metals	RAW	MYA	Lead							-0.937	0.019	5
PAHs	RAW	MYA	Acenaphthene				-0.904	0.035	5			
			Anthracene							-0.945	0.015	5
			Benzo(a)anthracene							-0.967	0.007	5
			Benzo(a)pyrene							-0.964	0.008	5
			Benzo(b)fluoranthene							-0.956	0.011	5
			Benzo(e)pyrene							-0.968	0.007	5
			Benzo(k)fluoranthene							-0.971	0.006	5
			Benzo(g,h,i)perylene							-0.945	0.015	5
			Chrysene							-0.982	0.003	5
			2,6-Dimethylnaphthalene							-0.894	0.041	5
			Dibenz(a,h)anthracene							-0.978	0.004	5
			Fluoranthene				-0.895	0.040	5	-0.948	0.014	5
			Fluorene				-0.957	0.011	5			
			High Molecular Weight PAHs							-0.962	0.009	5
			Indeno(1,2,3-cd)pyrene							-0.938	0.018	5
			Low Molecular Weight PAHs				-0.903	0.036	5	-0.924	0.025	5
			1-Methylphenanthrene				-0.882	0.048	5	-0.952	0.012	5
			Naphthalene							-0.883	0.047	5
			Perylene							-0.972	0.006	5
			Phenanthrene				-0.908	0.033	5	-0.931	0.022	5
			Pyrene				-0.882	0.048	5	-0.951	0.013	5
			Total PAHs							-0.960	0.009	5
PCBs	RAW	MYA	Silver							-0.954	0.046	4
Pesticides	RAW	MYA	o'p'-DDE							-0.954	0.012	5
VOAs	RAW	RM	Benzene				-0.720	0.029	9			
			Chloromethane				-0.700	0.036	9			
			Toluene				-0.708	0.033	9			

[1]HC = Hard Shell Clam
MYA = Soft Shell Clam
OYS = Oyster
RM = Ribbed Mussel
MF = Mummichog Fish

Table 6.2-3. Qualitative Summary for Biota Condition Effects by Ecological Exposure Zone.

	Biota Condition								
	Ecological Exposure Zone[1,3]								
Measurement Endpoint	LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR	REF
Hard Clam CI	++							na	
Soft Shell Clam CI	++							na	
Oyster CI	+			+				na	
Ribbed Mussel CI	+			+				na	
Mummichog Fish CI	+							na	
Soft Shell Neoplasia	+				+			na	
EEZ Score[2]	++	-	-	+	+	-	-	na	-

[1]Measurement Endpoint Scoring: + = observed response; ++ = observed response supported by exposure-response relationship

[2]EEZ Scoring based on + = slight risk, ++ = moderate risk; +++ = substantial risk

[3]Consult Section 6.0 and Figure 6.0-1 for description and location of EEZs.

Table 6.3-1. Summary of Biota Accumulation Factors (BAFs) and Biota-Sediment Accumulation Factors (BSAFS) for Target Receptors.

Chemical Group	Compound	Accumulation Factor: Hard Clam	Mummichog Fish	Soft Shell Clam	Oyster	Ribbed Mussel	All Species
Metals (BAF)	Arsenic	0.82	0.224	0.55	0.48	0.44	0.48
	Cadmium		0.037	0.75	1.03	0.77	0.76
	Chromium	0.05	0.016	0.08	0.31	0.18	0.08
	Copper	0.10	0.043	1.07	4.91	0.19	0.19
	Lead	0.02	0.020	0.12	0.07	0.02	0.02
	Mercury	0.29	0.250	158.84	0.34	0.15	0.29
	Nickel	0.15	0.016	0.12	0.51	0.46	0.15
	Silver	0.53	0.282	1.65	4.15	1.85	1.65
	Zinc	0.76	1.681	4.05	6.01	0.27	1.68
	Median	0.22	0.043	0.75	0.51	0.27	0.27
PAH (BSAF)	1,6,7-Trimethylnaphthalene	1.36		1.17	0.85	2.42	1.26
	1-Methylnaphthalene	1.25	0.72	1.06	0.57	18.72	1.06
	1-Methylphenanthrene	0.41		0.84	0.84	0.86	0.84
	2,6-Dimethylnaphthalene	1.05		1.71	0.59	1.56	1.30
	2-Methylnaphthalene	1.21	0.82	1.86	0.71	14.12	1.21
	Acenaphthene	0.54	0.20	0.70	0.07	3.52	0.54
	Acenaphthylene	0.11		0.23	0.22	2.19	0.23
	Anthracene	0.12		0.23	0.12	0.40	0.18
	Benzo(a)anthracene	0.04		0.28	0.02	0.10	0.07
	Benzo(a)pyrene	0.02		0.06	0.00	0.02	0.02
	Benzo(b)fluoranthene	0.03	0.003	0.13	0.01	0.05	0.03
	Benzo(e)pyrene	0.07	0.004	0.30	0.01	0.11	0.07
	Benzo(g,h,i)perylene	0.04		0.11	0.03	0.03	0.04
	Benzo(k)fluoranthene	0.02		0.13	0.05	0.02	0.03
	Biphenyl				0.38	1.26	0.82
	Chrysene	0.10		0.55	0.08	0.11	0.11
	Dibenz(a,h)anthracene	0.06		0.04			0.05
	Fluoranthene	0.37	0.003	0.80	0.15	0.68	0.37
	Fluorene	0.38	0.110	1.18	0.25	2.26	0.38
	High Molecular Weight PAHs	0.12	0.006	0.35	0.06	0.30	0.12
	Indeno(1,2,3-cd)pyrene	0.03		0.04		0.04	0.04
	Low Molecular Weight PAHs	0.38	0.185	1.18	0.59	5.05	0.59
	Perylene	0.08		0.08		0.06	0.08
	Phenanthrene	0.15	0.048	0.44	0.18	0.88	0.18
	Pyrene	0.31		0.43	0.05	1.03	0.37
	Total PAHs	0.14	0.017	0.41	0.09	0.66	0.14
	Median	0.12	0.048	0.41	0.12	0.68	0.12
PCB (BSAF)	101 (2 2'3 5 5')	2.14	6.22	1.03	3.43	3.40	3.40
	105 (2 3 3'4 4')	0.88	3.06	0.47	2.03	1.67	1.67
	118 (2 3'4 4'5)	1.96	7.36	1.18	3.23	3.52	3.23
	128 (2 2'3 3'4 4')	0.70	2.18	0.73	1.05	1.13	1.05
	138 (2 2'3 4 4'5)	1.81	28.92	1.92	2.15	3.71	2.15
	153 (2 2'4 4'5 5')	2.24	62.01	2.13	10.31	6.10	6.10
	170 (2 2'3 3'4 4'5)	0.17					0.17
	18 (2 2'5)	2.10		1.95	0.91	1.53	1.74
	187 (2 2'3 4'5 5'6)	3.05	3.32	0.63	3.30	2.62	3.05
	195 (2 2'3 3'4 4'5 6)	1.79	1.45	0.10	0.11		0.78
	206 (2 2'3 3'4 4'5 5'6)	1.57	1.06	0.04			1.06
	209 (2 2'3 3'4 4'5 5'6 6')	1.58	0.53				1.05
	28 (2 4 4')	0.44	1.39	0.68	1.02	0.51	0.68
	44 (2 2'3 5')	4.36	1.90	0.83	1.28	1.56	1.56
	52 (2 2'5 5)	2.02	3.66	10.70	4.16	1.72	3.66
	66 (2 3'4 4')	1.09	4.69	2.83	1.95	2.21	2.21
	8 (2 4)				2.07		2.07
	Median	1.80	3.19	0.93	2.05	1.96	1.96
Pesticides (BSAF)	Aldrin				0.38	0.64	0.51
	Hexachlorobenzene	1.76		0.19			0.97
	o,p'-DDE	2.69	2.24	0.24	1.79	2.14	2.14
	p,p'-DDE	1.83	4.84	1.45	3.02	2.62	2.62
	Median	1.83	3.54	0.24	1.79	2.14	1.83

Table 6.3-2. Site/reference concentration ratio statistics for CoCs in Allen Harbor biota summarized by ecological exposure zones (EEZ). A) Hard shell clams, B.) Oysters; C) Ribbed mussels; and D) Mummichog Fish.

A. Hard Shell Clams

Group	EEZ	Mean CR	Min. CR	Max. CR	Ranking[1]
A. Total PCBs	LS-V/W				-
	LS-D	1.96	0.76	4.74	-
	LM-D	1.19	0.68	2.16	-
	LN-W	2.63	0.47	5.87	+
	CP-W	1.11	0.60	2.12	-
	CP-SW				-
	CP-SD	0.75	0.42	1.13	-
	MAR				-
B. Pesticide Analytes	LS-V/W				-
	LS-D	2.39	1.03	3.08	+
	LM-D	1.60	1.39	1.99	-
	LN-W	1.03	0.60	1.46	-
	CP-W	1.44	1.12	1.86	-
	CP-SW				-
	CP-SD	1.22	1.00	1.55	-
	MAR				-
C. HMW PAH Analytes -	LS-V/W				-
	LS-D	1.86	0.98	2.81	-
	LM-D	1.96	0.88	3.22	-
	LN-W	1.88	0.73	3.34	-
	CP-W	1.85	0.90	2.96	-
	CP-SW				-
	CP-SD	1.65	1.02	2.66	-
	MAR				-
C.1 LMW PAH Analytes -	LS-V/W				-
	LS-D	1.73	1.36	2.13	-
	LM-D	1.51	1.02	2.22	-
	LN-W	1.47	1.33	1.62	-
	CP-W	1.45	1.15	1.68	-
	CP-SW				-
	CP-SD	1.67	1.18	2.16	-
	MAR				-
C.2 Total PAH Analytes	LS-V/W				-
	LS-D	1.83	0.98	2.81	-
	LM-D	1.84	0.88	3.22	-
	LN-W	1.78	0.73	3.34	-
	CP-W	1.75	0.90	2.96	-
	CP-SW				-
	CP-SD	1.66	1.02	2.66	-
	MAR				-
D. Metals Analytes	LS-V/W				-
	LS-D	0.73	0.13	1.17	-
	LM-D	0.91	0.40	1.31	-
	LN-W	1.47	0.35	4.12	-
	CP-W	0.67	0.16	1.25	-
	CP-SW				-
	CP-SD	0.79	0.50	1.22	-
	MAR				-
E. TBT Analytes	LS-V/W				-
	LS-D				-
	LM-D	1.82	1.82	1.82	-
	LN-W				-
	CP-W				-
	CP-SW				-
	CP-SD				-
	MAR				-

[1]Ranking Notation: + = Mean CR > 2; '++ = Mean CR > 2 and Max. CR > 10

Table 6.3-2, con't.

B. Oysters

Group	EEZ	Mean CR	Min. CR	Max. CR	Ranking[1]
A. Total PCBs	LS-V/W	2.52	0.92	4.57	+
	LS-D				-
	LM-D				-
	LN-W	2.82	0.30	5.74	+
	CP-W	1.65	0.59	2.95	-
	CP-SW	3.12	1.93	4.53	+
	CP-SD				-
	MAR				-
B. Pesticide Analytes	LS-V/W	3.43	2.07	4.80	+
	LS-D				-
	LM-D				-
	LN-W	2.04	1.73	2.35	+
	CP-W	1.25	1.22	1.28	-
	CP-SW				-
	CP-SD				-
	MAR				-
C. PAH Analytes - High Molecular Weight	LS-V/W	1.97	1.44	2.64	-
	LS-D				-
	LM-D				-
	LN-W	2.16	1.70	2.89	+
	CP-W	1.59	0.81	2.22	-
	CP-SW	4.57	3.41	5.70	+
	CP-SD				-
	MAR				-
PAH Analytes - Low Molecular Weight	LS-V/W	2.13	1.52	2.70	+
	LS-D				-
	LM-D				-
	LN-W	2.19	1.51	2.67	+
	CP-W	2.09	1.35	2.79	+
	CP-SW	5.82	3.67	7.95	+
	CP-SD				-
	MAR				-
PAH Analytes - Total	LS-V/W	2.00	1.44	2.70	-
	LS-D				-
	LM-D				-
	LN-W	2.15	1.51	2.89	+
	CP-W	173.00	0.81	2.79	+
	CP-SW	4.88	3.41	7.95	+
	CP-SD				-
	MAR				-
D. Metals Analytes	LS-V/W	2.12	0.66	5.05	+
	LS-D				-
	LM-D				-
	LN-W	9.66	0.51	70.75	++
	CP-W	3.33	0.60	6.65	+
	CP-SW	1.21	0.32	2.74	-
	CP-SD				-
	MAR				-
E. TBT Analytes	LS-V/W				-
	LS-D				-
	LM-D				-
	LN-W				-
	CP-W				-
	CP-SW				-
	CP-SD				-
	MAR				-

[1]Ranking Notation: + = Mean CR > 2; '++ = Mean CR > 2 and Max. CR > 10

Table 6.3-2, con't.

C. Ribbed Mussels

Group	EEZ	Mean CR	Min. CR	Max. CR	Ranking[1]
A. Total PCBs	LS-V/W	2.86	0.30	5.10	+
	LS-D				-
	LM-D				-
	LN-W	2.51	1.15	4.68	+
	CP-W	1.34	0.18	2.28	-
	CP-SW	1.01	0.15	1.75	-
	CP-SD				-
	MAR				-
B. Pesticide Analytes	LS-V/W	2.37	2.16	2.57	+
	LS-D				-
	LM-D				-
	LN-W	1.02	0.76	1.29	-
	CP-W	1.07	0.98	1.17	-
	CP-SW	0.78	0.65	0.92	-
	CP-SD				-
	MAR				-
C. PAH Analytes - High Molecular Weight	LS-V/W	1.07	0.65	1.49	-
	LS-D				-
	LM-D				-
	LN-W	0.75	0.35	1.24	-
	CP-W	0.97	0.42	1.45	-
	CP-SW	1.19	0.60	1.77	-
	CP-SD				-
	MAR				-
PAH Analytes - Low Molecular Weight	LS-V/W	1.33	0.48	3.13	-
	LS-D				-
	LM-D				-
	LN-W	0.71	0.28	1.74	-
	CP-W	0.89	0.35	1.80	-
	CP-SW	1.27	0.60	2.57	-
	CP-SD				-
	MAR				-
PAH Analytes - Total	LS-V/W	1.19	0.48	3.13	-
	LS-D				-
	LM-D				-
	LN-W	0.73	0.28	1.74	-
	CP-W	0.92	0.35	1.80	-
	CP-SW	1.22	0.60	2.57	-
	CP-SD				-
	MAR				-
D. Metals Analytes	LS-V/W	2.63	0.45	7.10	+
	LS-D				-
	LM-D				-
	LN-W	2.72	0.47	15.14	++
	CP-W	2.10	0.32	9.78	+
	CP-SW	28.55	0.08	212.66	++
	CP-SD				-
	MAR				-
E. TBT Analytes	LS-V/W				-
	LS-D				-
	LM-D				-
	LN-W				-
	CP-W				-
	CP-SW				-
	CP-SD				-
	MAR				-

[1]Ranking Notation: + = Mean CR > 2; '++ = Mean CR > 2 and Max. CR > 10

Table 6.3-2, con't.

D. Mummichog Fish

Group	EEZ	Mean CR	Min. CR	Max. CR	Ranking[1]
A. Total PCBs	LS-V/W	1.55	0.32	3.73	-
	LS-D				-
	LM-D				-
	LN-W	2.07	0.55	5.73	+
	CP-W				-
	CP-SW	0.84	0.27	1.27	-
	CP-SD				-
	MAR				-
B. Pesticide Analytes	LS-V/W	1.93	0.98	3.02	-
	LS-D				-
	LM-D				-
	LN-W	2.05	1.21	2.60	+
	CP-W				-
	CP-SW	1.11	0.70	1.80	-
	CP-SD				-
	MAR				-
C. PAH Analytes - High Molecular Weight	LS-V/W	0.78	0.74	0.84	-
	LS-D				-
	LM-D				-
	LN-W	0.85	0.85	0.85	-
	CP-W				-
	CP-SW	0.67	0.67	0.67	-
	CP-SD				-
	MAR				-
PAH Analytes - Low Molecular Weight	LS-V/W	0.95	0.95	0.95	-
	LS-D				-
	LM-D				-
	LN-W	1.22	1.22	1.22	-
	CP-W				-
	CP-SW	1.08	1.08	1.08	-
	CP-SD				-
	MAR				-
PAH Analytes - Total	LS-V/W	0.82	0.74	0.95	-
	LS-D				-
	LM-D				-
	LN-W	1.05	0.85	1.22	-
	CP-W				-
	CP-SW	0.89	0.67	1.08	-
	CP-SD				-
	MAR				-
D. Metals Analytes	LS-V/W	1.79	0.68	7.40	-
	LS-D				-
	LM-D				-
	LN-W	2.16	0.71	9.85	+
	CP-W				-
	CP-SW	2.37	0.08	11.91	++
	CP-SD				-
	MAR				-
E. TBT Analytes	LS-V/W				-
	LS-D				-
	LM-D				-
	LN-W				-
	CP-W				-
	CP-SW				-
	CP-SD				-
	MAR				-

[1]Ranking Notation: + = Mean CR > 2; '++ = Mean CR > 2 and Max. CR > 10

Table 6.3-3. Qualitative Summary for Tissue Residue Enrichment Factors by Ecological Exposure Zone.

	Tissue CoC Enrichment Factor							
	Ecological Exposure Zone[1,2]							
	LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR
Total PCBs	++	-	-	++	-	+	-	nd
Pesticide Analytes	++	+	-	++	-	-	-	nd
Total PAH Analytes	-	-	-	+	+	+	-	nd
Metals Analytes	++	-	-	++	++	++	-	nd
TBT Analytes	-	-	-	-	-	-	-	nd
EEZ Score[2]	++	-	-	++	+	+	-	nd

[1] Ranking: + = 1 species exhibiting > two-fold CoC enrichment;

++ = 1 spp. > 10-fold enrichment or 2 spp. exhibiting > two-fold enrichment

[2] EEZ Scoring based on + = slight risk, ++ = moderate risk; +++ = substantial risk

nd = not determined

Table 6.4-1. Amphipod survival (%) and sea urchin fertilization success statistics summarized by Ecological Exposure Zone (EEZ).

		Toxicity			
Bioaasay	EEZ	Mean	Minimum	Maximum	Ranking[1]
Amphipod Survival (%)	LS-V/W	89.1	63.0	101.0	+
	LS-D	89.3	78.0	96.0	+
	LM-D	97.8	96.0	102.0	-
	LN-W	87.0	65.0	98.0	+
	CP-W	85.3	60.0	97.0	+
	CP-SW	87.3	80.0	96.0	+
	CP-SD	97.0	95.0	99.0	-
	REF	92.8	81.0	103.0	-
Arbacia Fertilization Success, (%)	LS-V/W	97.1	95.3	99.0	-
	LS-D	52.3	0.0	85.7	++
	LM-D	91.4	67.0	97.4	+
	LN-W	73.3	9.0	99.0	++
	CP-W	94.6	89.1	97.4	-
	CP-SW	94.8	93.0	96.0	-
	CP-SD	93.0	91.3	96.4	-
	REF	96.8	91.8	98.5	-

[1]Measurement Endpoint Scoring:
Amphipods: + = slight toxicity (61-80% survival), one or more stations;
++ = pronounced toxicity, (< 60%) survival, 1 or more stations.
Sea urchins: + = slight toxicity (51-70% survival), one or more stations;
++ = pronounced toxicity, (< 50%) survival, 1 or more stations.

Table 6.4-2. Qualitative Summary of sediment toxicity by Ecological Exposure Zone.

	Toxicity								
	Ecological Exposure Zone[1,3]								
Measurement Endpoint	LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR[4]	REF
Amphipod Survival	+	+	-	+	+	+	-	-	-
Sea Urchin Fertilization	-	++	+	++	-	-	-	-	-
EEZ Score[2]	++	+	-	++	-	-	-	-	-

[1]Measurement Endpoint Scoring:

Amphipods: + = slight toxicity (61-80% survival), 1 or more stations; ++ = pronounced toxicity, (< 60%) survival, 1 or more stations.

Sea urchins: + = slight toxicity (51-70% survival), 1 or more stations; ++ = pronounced toxicity, (< 50%) survival, 1 or more stations.

[2]EEZ Scoring based on: + = slight risk (based on slight toxicity in both species or high toxicity in one species);

++ = moderate risk (based on toxicity in both species or toxicity supported by exposure-response relationships in one or both species).

[3]Consult Section 6.0 and Figure 6.0-1 for description and location of EEZs.

[4]RAPS Phase 1 results

Table 6.5-1. Comparisons of Exposure Point Concentrations in Sediments with Benchmark Values.

Chemical Class	Analyte	LS-VW							LS-D			LM-D							
		V1	V2	W1	W2	W3	W4	W5	D1	D2	D3	D10	D4	D5	D6	D7	D8	D9	DAH5
MET	Arsenic	+	+	+	+	-	+	+	-	-	-	-	-	-	-	+	+	+	+
MET	Cadmium	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	+
MET	Chromium	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	+
MET	Copper	+	+	+	+	-	+	++	+	-	-	+	+	+	+	+	+	+	+
MET	Lead	+	+	+	+	-	+	+	-	-	-	+	+	+	+	+	+	+	+
MET	Mercury	+	+	+	+	-	+	+	-	-	-	-	-	+	+	+	+	+	-
MET	Nickel	+	+	+	-	-	+	+	-	-	-	+	+	+	+	+	+	+	+
MET	Silver	-	+	+	+	-	+	+	-	-	-	+	+	+	+	+	+	+	+
MET	Zinc	+	+	+	+	-	+	++	-	-	-	+	-	+	-	+	+	+	+
PAH	2-Methylnaphthalene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Acenaphthene	-	+	++	+	+	+	+	-	-	-	-	-	+	-	-	+	-	-
PAH	Acenaphthylene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Anthracene	-	+	+	+	+	+	+	-	-	-	-	-	-	-	-	-	-	-
PAH	Benzo(a)anthracene	-	+	+	+	+	+	+	-	-	-	-	-	-	-	-	+	-	-
PAH	Benzo(a)pyrene	-	-	+	+	-	+	+	-	-	-	-	-	-	-	-	-	-	-
PAH	Chrysene	-	+	+	+	-	+	+	-	-	-	-	-	-	-	-	+	-	-
PAH	Dibenz(a,h)anthracene	-	+	+	+	-	+	+	-	-	-	-	-	-	-	-	+	-	+
PAH	Fluoranthene	-	-	+	+	+	+	+	-	-	-	-	-	-	-	-	-	-	-
PAH	Fluorene	+	+	++	+	+	+	+	-	-	-	+	-	+	-	+	+	-	+
PAH	Naphthalene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Phenanthrene	-	+	++	+	+	+	+	-	-	-	-	-	-	-	-	+	-	+
PAH	Pyrene	-	-	+	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-
PAH	High Molecular Weight PAHs	+	+	++	+	+	+	+	-	-	-	+	+	+	+	+	+	+	-
PAH	Low Molecular Weight PAHs	-	-	+	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Total PAHs	-	+	+	+	+	+	+	-	-	-	-	-	-	-	-	-	-	-
PCB	PCB Sum of Congeners x 2	+	++	++	++	+	++	++	+	+	+	+	+	+	+	+	+	+	+
PST	o,p'-DDE	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PST	p,p'-DDE	+	+	++	+	+	+	+	+	-	+	+	+	+	+	+	+	+	+

Chemical Class	Analyte	LN-W					CP-W				CP-SW			CP-SD			REF		
		W10	W6	W7	W8	W9	D14	V3	W1	W12	V4	W1	W14	D11	D12	D13	D1	V1	W1
MET	Arsenic	-	+	-	+	+	+	+	-	-	+	-	-	+	-	-	-	-	-
MET	Cadmium	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	+	+
MET	Chromium	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
MET	Copper	-	+	-	-	-	+	-	-	-	-	-	-	+	-	-	-	-	-
MET	Lead	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-
MET	Mercury	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	+	+	+
MET	Nickel	-	-	+	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-
MET	Silver	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	+	+	+
MET	Zinc	-	+	-	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-
PAH	2-Methylnaphthalene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Acenaphthene	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Acenaphthylene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Anthracene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Benzo(a)anthracene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Benzo(a)pyrene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Chrysene	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-
PAH	Dibenz(a,h)anthracene	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Fluoranthene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Fluorene	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Naphthalene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Phenanthrene	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Pyrene	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	High Molecular Weight PAHs	-	+	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	+
PAH	Low Molecular Weight PAHs	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PAH	Total PAHs	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PCB	PCB Sum of Congeners x 2	-	++	+	+	+	+	+	+	-	+	-	-	+	-	-	-	-	-
PST	o,p'-DDE	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
PST	p,p'-DDE	-	+	-	-	-	+	+	-	-	+	-	-	-	-	-	-	-	-

Ranking: - = HQ <1; + = HQ > 1; '++ = HQ > 10; +++ = HQ > 100

Table 6.5-2. Hazard Quotient (HQ) statistics for CoCs in Allen Harbor sediment summarized by habitats[1].

		Site-Related				Reference			
Group	Habitat	Hazard Index	Mean HQ	Min. HQ	Max. HQ	Hazard Index	Mean HQ	Min. HQ	Max. HQ
A. Metal Analytes	Subtidal, deep	9.6	1.1	0.57	2.3	2.8	0.31	0.01	1.00
	Subtidal, deep core	7.5	0.83	0.48	1.6	2.7	0.30	0.01	1.00
	Vegetated Wetlands	10.5	1.2	0.52	2.4	2.9	0.32	0.07	1.00
	Intertidal Wetlands	11.9	1.3	0.38	2.7	5.1	0.57	0.04	3.17
B. Total PCBs[2]	Subtidal, deep		6.3	0.70	22.2		0.22	0.11	0.33
	Subtidal, deep core		1.0	0.29	2.3		0.13	0.09	0.16
	Vegetated Wetlands		7.0	2.5	13.0		0.19	0.09	0.28
	Intertidal Wetlands		12.1	0.14	59.3		0.49	0.39	0.59
C. VOA Analytes	Subtidal, deep	nd	nd	nd	nd	nd	nd	nd	nd
	Subtidal, deep core	0.55	0.04	<0.01	0.36	0.34	0.02	<0.01	0.25
	Vegetated Wetlands	0.66	0.04	<0.01	0.44	0.54	0.04	<0.01	0.36
	Intertidal Wetlands	0.45	0.03	<0.01	0.30	0.37	0.02	<0.01	0.25
D. High Molecular Wt. PAHs[2]	Subtidal, deep	4.2	0.47	0.03	0.75	0.02	<0.01	<0.01	0.00
	Subtidal, deep core	1.0	0.12	0.01	0.18	0.01	<0.01	<0.01	0.00
	Vegetated Wetlands	4.7	0.52	0.04	0.89	0.23	0.03	<0.01	0.04
	Intertidal Wetlands	5.7	0.63	0.04	1.3	1.6	0.18	0.01	0.33
Low Molecular Wt. PAHs[2]	Subtidal, deep	2.7	0.45	0.12	0.83	0.04	0.01	<0.01	0.01
	Subtidal, deep core	1.2	0.20	0.07	0.44	0.03	0.01	<0.01	0.01
	Vegetated Wetlands	3.5	0.59	0.14	1.2	0.11	0.02	0.01	0.03
	Intertidal Wetlands	5.6	0.93	0.14	2.3	0.93	0.15	0.04	0.34
Total PAHs	Subtidal, deep	6.9	0.46	0.03	0.83	0.07	<0.01	<0.01	0.01
	Subtidal, deep core	2.2	0.15	0.01	0.44	0.05	<0.01	<0.01	0.01
	Vegetated Wetlands	8.2	0.55	0.04	1.2	0.34	0.02	<0.01	0.04
	Intertidal Wetlands	11.2	0.75	0.04	2.3	2.56	0.17	0.01	0.34
E. Pesticide Analytes	Subtidal, deep	3.0	1.0	0.05	2.7	0.03	0.01	<0.01	0.02
	Subtidal, deep core	0.37	0.12	<0.01	0.3	0.04	0.01	<0.01	0.03
	Vegetated Wetlands	4.4	1.5	0.01	4.4	0.05	0.02	<0.01	0.03
	Intertidal Wetlands	2.7	0.90	0.02	2.7	0.23	0.08	<0.01	0.14

[1]Non-metal CoCs calculated using equilibrium partitioning
[2]Group treated as single analyte; HQ = HI
nd = no data

TABLE 6.5-3. Hazard Quotient (HQ) and Hazard Index (HI) statistics for CoCs in Allen Harbor sediments summarized by ecological exposure zones.

Group	EEZ	HI	Mean HQ	Min. HQ	Max. HQ	Ranking[1]
A. Total PCBs[2]	LS-V/W		19.11	6.55	54.25	+++
	LS-D		1.35	0.35	1.94	+
	LM-D		7.49	2.89	21.94	++
	LN-W		15.17	0.14	59.26	+++
	CP-W		3.62	0.48	8.90	+
	CP-SW		1.54	0.26	4.04	+
	CP-SD		0.88	0.70	1.23	-
	MAR		22.25			+
	REF		0.30	0.09	0.59	-
B. Pest. Analytes	LS-V/W	6.73	2.24	0.02	6.68	+
	LS-D	1.16	0.39	0.01	0.98	-
	LM-D	4.58	1.53	0.04	4.29	+
	LN-W	1.53	0.51	0.02	1.49	-
	CP-W	0.83	0.28	0.05	0.68	-
	CP-SW	0.92	0.31	0.01	0.88	-
	CP-SD	0.56	0.19	0.03	0.42	-
	MAR		1.49			+
	REF	0.11	0.04	<0.01	0.06	-
C. Metal analytes	LS-V/W	20.36	2.26	0.80	4.64	+
	LS-D	3.97	0.44	0.18	0.78	-
	LM-D	11.81	1.31	0.58	2.82	+
	LN-W	5.88	0.65	0.20	1.11	-
	CP-W	13.62	1.51	0.45	3.65	+
	CP-SW	3.84	0.43	0.15	0.79	-
	CP-SD	5.29	0.59	0.24	1.33	-
	MAR		11.42			+
	REF	3.59	0.40	0.04	1.72	-
D. PAH Analytes - High Molecular Weight	LS-V/W	14.14	1.57	0.09	3.40	+
	LS-D	1.43	0.16	0.02	0.25	-
	LM-D	4.57	0.51	0.03	0.79	-
	LN-W	1.81	0.20	0.01	0.31	-
	CP-W	1.85	0.21	0.02	0.30	-
	CP-SW	1.76	0.20	0.02	0.36	-
	CP-SD	1.56	0.17	0.02	0.29	-
	MAR		16.84			+
	REF	0.63	0.07	<0.01	0.12	-
PAH Analytes - Low Molecular Weight	LS-V/W	12.92	2.15	0.25	5.09	+
	LS-D	0.95	0.16	0.04	0.25	-
	LM-D	3.51	0.59	0.18	0.95	-
	LN-W	1.28	0.21	0.07	0.43	-
	CP-W	1.02	0.17	0.06	0.32	-
	CP-SW	1.24	0.21	0.06	0.39	-
	CP-SD	1.32	0.22	0.06	0.45	-
	MAR		3.36			+
	REF	0.36	0.06	0.02	0.13	-
PAH Analytes - Total	LS-V/W	27.06	1.80	0.09	5.09	+
	LS-D	2.38	0.16	0.02	0.25	-
	LM-D	8.08	0.54	0.03	0.95	-
	LN-W	3.09	0.21	0.01	0.43	-
	CP-W	2.87	0.19	0.02	0.32	-
	CP-SW	3.00	0.20	0.02	0.39	-
	CP-SD	2.88	0.19	0.02	0.45	-
	MAR				20.20	-
	REF	0.99	0.07	<0.01	0.13	-
E. VOA Analytes	LS-V/W	0.64	0.04	<0.01	0.42	-
	LS-D					-
	LM-D					-
	LN-W	0.38	0.03	<0.01	0.25	-
	CP-W	0.36	0.02	<0.01	0.24	-
	CP-SW	0.51	0.03	<0.01	0.34	-
	CP-SD					-
	MAR					-
	REF	0.46	0.03	<0.01	0.30	-

[1]Ranking Notation: + = Mean HQ > 1; ++ = Max HQ > 10; +++ = Mean and Max. HQ > 10
[2]Group treated as single analyte; HQ = HI
nd = no data

Table 6.5-4. Qualitative Summary for CoC Sediment Hazards by Ecological Exposure Zones

		Sediment Hazard								
AV/MAX		Ecological Exposure Zone[1,3]								
Measurement Endpoint		LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR	REF
Total PCB HQs	15, 54	+++	+	++	+++	+	+	-	+	-
Pesticide HQs	5, 10	+	+	-	+	-	-	-	-	+
PAH HQs	2, 5	+	+	+	-	+	-	-	-	-
VOA HQs	.64, 2	-	-	-	+	-	+	-	-	-
Metal HQs	2 4	+	+	-	-	-	-	-	-	-
Butyltin HQs		nd	nd	nd	nd	nd	nd	nd	nd	nd
EEZ Score[2]		++	+	+	++	-	-	-	-	-

[1]Ranking Notation: + = Mean HQ > 1; ++ = Max HQ > 10; +++ = Mean and Max. HQ > 10

[2]EEZ Scoring (based on + = 1 pt, ++ = 3 pt; +++ = 5 pt): 3-5 pt = slight risk (+), 6-9 pt = moderate risk (++); > 9 pt = substantial risk (+++).

[3]Consult Section 6.0 and Figure 6.0-1 for description and location of EEZs.

Table 6.5-5. Hazard Quotient (HQ) statistics for CoCs in Allen Harbor sediment porewater summarized by habitats[1].

		Site-Related				Reference			
Group	Habitat	Hazard Index	Mean HQ	Min. HQ	Max. HQ	Hazard Index	Mean HQ	Min. HQ	Max. HQ
A. Metal Analytes	Subtidal, deep	4.7	0.5	<0.01	2.3	5.2	0.6	<0.01	2.3
	Subtidal, deep core	6.1	0.7	<0.01	3.8	5.6	0.6	<0.01	2.7
	Vegetated Wetlands	5.0	0.6	<0.01	1.8	4.3	0.5	<0.01	2.3
	Intertidal Wetlands	4.8	0.5	<0.01	2.3	3.8	0.4	<0.01	1.4
B. Total PCBs[2]	Subtidal, deep		2.2	1.3	3.7		1.7	0.5	2.8
	Subtidal, deep core		3.6	0.5	11.2		nd	nd	nd
	Vegetated Wetlands		2.1	1.0	3.2		6.4	2.8	10.0
	Intertidal Wetlands		1.4	0.3	3.4		3.2	2.1	4.3
C. VOA Analytes	Subtidal, deep	nd	nd	nd	nd	nd	nd	nd	nd
	Subtidal, deep core	31595	1975	<0.01	14820	nd	nd	nd	nd
	Vegetated Wetlands	62073	3880	<0.01	40547	12230	764.4	<0.01	7989
	Intertidal Wetlands	21040	1315	<0.01	13743	16528	1033	<0.01	10796
D. High Molecular Wt. PAHs[2]	Subtidal, deep		355.4	232.5	658.5		20.0	16.2	23.8
	Subtidal, deep core		220.0	25.0	481.5		nd	nd	nd
	Vegetated Wetlands		297.2	101.4	647.3		112.4	7.8	216.9
	Intertidal Wetlands		798.9	41.7	2367		510.8	500.3	521.3
Low Molecular Wt. PAHs[2]	Subtidal, deep		101.1	59.9	142.8		20.5	20.4	20.7
	Subtidal, deep core		82.5	23.0	181.1		nd	nd	nd
	Vegetated Wetlands		73.5	24.3	160.3		24.2	5.5	42.9
	Intertidal Wetlands		196.6	17.5	632.2		111.9	82.8	140.9
Total PAHs	Subtidal, deep	9.0	1.3	<0.01	8.9	193.0	27.6	<0.01	193.0
	Subtidal, deep core	36.9	5.3	<0.01	36.9	14.3	2.0	<0.01	14.2
	Vegetated Wetlands	14.3	2.0	<0.01	14.2	231.8	33.1	<0.01	231.8
	Intertidal Wetlands	18.5	2.6	<0.01	18.2	6.9	1.0	<0.01	6.7
E. Pesticide Analytes	Subtidal, deep	478.1	95.6	<0.01	465.4	61.3	12.3	<0.01	30.8
	Subtidal, deep core	245.5	49.1	<0.01	216.4	802.2	160.4	<0.01	762.4
	Vegetated Wetlands	802.2	160.4	<0.01	762.4	55.6	11.1	<0.01	37.5
	Intertidal Wetlands	1500	300.0	<0.01	1439	162.2	32.4	<0.01	129.7

[1]Non-metal CoCs calculated using equilibrium partitioning

[2]Group treated as single analyte; HQ = HI

n = no data

TABLE 6.5-6. Hazard Quotient (HQ) and Hazard Index (HI) statistics for CoCs in Allen Harbor pore water summarized by ecological exposure zones.

Group	EEZ	HI	Mean HQ	Min. HQ	Max. HQ	Ranking[1]
A. Total PCBs[2]	LS-V/W		1.65	0.53	3.25	+
	LS-D		0.54	0.40	0.69	-
	LM-D		0.51	0.40	0.75	-
	LN-W		1.52	0.37	3.36	+
	CP-W		0.45	0.32	0.56	-
	CP-SW		0.37	0.29	0.45	-
	CP-SD		0.33	0.27	0.43	-
	MAR					-
	REF		0.6	0.11	1.8	-
B. Pest. Analytes	LS-V/W	2.28	0.46	<0.01	2.17	-
	LS-D	0.61	0.12	<0.01	0.60	-
	LM-D	0.54	0.11	<0.01	0.53	-
	LN-W	1.31	0.26	<0.01	1.28	-
	CP-W	0.30	0.06	<0.01	0.29	-
	CP-SW	0.28	0.06	<0.01	0.27	-
	CP-SD	0.23	0.05	<0.01	0.21	-
	MAR					-
	REF	0.09	0.02	<0.01	0.07	-
C. Metal analytes	LS-V/W	5.31	0.59	<0.01	2.33	-
	LS-D	6.01	0.67	<0.01	2.85	-
	LM-D	4.40	0.49	<0.01	2.13	-
	LN-W	4.93	0.55	<0.01	2.30	-
	CP-W	4.47	0.50	<0.01	1.91	-
	CP-SW	4.63	0.51	<0.01	2.32	-
	CP-SD	4.00	0.44	<0.01	2.17	-
	MAR					-
	REF	4.45	0.49	<0.01	1.98	-
D. PAH Analytes - High Molecular Weight	LS-V/W		1.38	0.21	2.37	+
	LS-D		0.39	0.23	0.66	-
	LM-D		0.38	0.31	0.49	-
	LN-W		0.38	0.04	0.71	-
	CP-W		0.13	0.07	0.26	-
	CP-SW		0.19	0.13	0.23	-
	CP-SD		0.27	0.24	0.30	-
	MAR					-
	REF		0.21	0.01	0.52	-
PAH Analytes - Low Molecular Weight	LS-V/W		0.34	0.06	0.63	-
	LS-D		0.08	0.06	0.10	-
	LM-D		0.12	0.10	0.14	-
	LN-W		0.09	0.02	0.18	-
	CP-W		0.04	0.02	0.09	-
	CP-SW		0.04	0.03	0.05	-
	CP-SD		0.08	0.07	0.09	-
	MAR					-
	REF		0.05	0.01	0.14	-
PAH Analytes - Total	LS-V/W	1.48	0.19	0.00	0.57	-
	LS-D	0.39	0.05	0.00	0.15	-
	LM-D	0.42	0.05	0.00	0.17	-
	LN-W	0.40	0.05	0.00	0.17	-
	CP-W	0.13	0.02	0.00	0.05	-
	CP-SW	0.18	0.02	0.00	0.07	-
	CP-SD	0.27	0.03	0.00	0.11	-
	MAR					-
	REF	0.21	0.03	0.00	0.08	-
E. VOA Analytes	LS-V/W	<0.1	<0.1	<0.1	<0.1	-
	LS-D	<0.1	<0.1	<0.1	<0.1	-
	LM-D	<0.1	<0.1	<0.1	<0.1	-
	LN-W	<0.1	<0.1	<0.1	<0.1	-
	CP-W	<0.1	<0.1	<0.1	<0.1	-
	CP-SW	<0.1	<0.1	<0.1	<0.1	-
	CP-SD	<0.1	<0.1	<0.1	<0.1	-
	MAR	<0.1	<0.1	<0.1	<0.1	-
	REF	<0.1	<0.1	<0.1	<0.1	-

[1]Ranking Notation: + = Mean HQ > 1; ++ = Max HQ > 10; +++ = Mean and Max. HQ > 10

[2]Group treated as single analyte; HQ = HI

nd = no data

Table 6.5-7. Qualitative Summary for CoC Porewater Hazards by Ecological Exposure Zone.

	Pore Water Hazard								
	Ecological Exposure Zone[1,3]								
Measurement Endpoint	LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR	REF
Total PCB HQs	+	-	-	+	-	-	-	-	-
Pesticide HQs	-	-	-	-	-	-	-	-	-
PAH HQs	-	-	-	-	-	-	-	-	-
VOA HQs	-	-	-	-	-	-	-	-	-
Metal HQs	-	-	-	-	-	-	-	-	-
Butyltin HQs	nd	nd	nd	nd	nd	nd	nd	nd	nd
EEZ Score[2]	+	-	-	+	-	-	-	-	-

[1]Ranking Notation: + = Mean > 1; ++ = Mean HQ > 1 and Max. HQ > 10

[2]EEZ Scoring (based on + = 1 pt, ++ = 3 pt; +++ = 5 pt): 1-3 pt = slight risk (+), 4-6 pt = moderate risk (++); > 6 pt = substantial risk (+++).

[3]Consult Section 6.0 and Figure 6.0-1 for description and location of EEZs.

Table 6.6-1. Tissue residue concentrations and chronic Critical Body Residue values for target species.

		Metals nMol/g dry	PAHs nMol/g dry	PCBs nMol/g dry	Pesticides nMol/g dry	Butyltins nMol/g dry
Critical Body Residue		333.3	533.3	533.3	213.3	6.7
Reference		1	2	1	4	3
A. Hard Shell Clams						
		Metals nMol/g dry	PAHs nMol/g dry	PCBs nMol/g dry	Pesticides nMol/g dry	Butyltins nMol/g dry
Site-related						
	Mean	1.84	1.61	0.27	0.02	0.06
	Maximum	2.92	1.83	0.68	0.07	0.13
Reference						
	Mean	1.82	0.74	0.11	0.01	0.04
	Maximum	2.19	0.79	0.17	0.02	0.08
B. Mummichog Fish						
		Metals nMol/g dry	PAHs nMol/g dry	PCBs nMol/g dry	Pesticides nMol/g dry	Butyltins nMol/g dry
Site-related						
	Mean	2.95	0.96	2.93	0.29	0.05
	Maximum	3.39	1.25	4.41	0.37	0.11
Reference						
	Mean	2.65	0.80	1.27	0.12	0.01
	Maximum	2.65	0.81	1.67	0.15	0.01
C. Soft Shell Clams						
		Metals nMol/g dry	PAHs nMol/g dry	PCBs nMol/g dry	Pesticides nMol/g dry	Butyltins nMol/g dry
Site-related						
	Mean	2.43	4.23	0.38	0.03	0.23
	Maximum	5.89	4.76	0.75	0.05	1.03
Reference						
	Mean	(not available)				
	Maximum					
D. Oysters						
		Metals nMol/g dry	PAHs nMol/g dry	PCBs nMol/g dry	Pesticides nMol/g dry	Butyltins nMol/g dry
Site-related						
	Mean	26.80	1.87	0.98	0.10	0.02
	Maximum	67.37	2.62	1.43	0.16	0.06
Reference						
	Mean	5.96	0.98	0.38	0.06	0.07
	Maximum	12.31	1.04	0.43	0.06	0.10
E. Ribbed Mussels						
		Metals nMol/g dry	PAHs nMol/g dry	PCBs nMol/g dry	Pesticides nMol/g dry	Butyltins nMol/g dry
Site-related						
	Mean	1.68	23.94	0.92	0.10	0.06
	Maximum	2.53	39.09	1.78	0.18	0.16
Reference						
	Mean	1.75	14.83	0.28	0.05	0.13
	Maximum	2.46	15.12	0.41	0.06	0.22

References:
1) McCarty et al., 1992
2) Arnold and Biddinger, 1995
3) Page and Widdows, 1991
4) Hoke et al., 1994

Table 6.6-2. Tissue Screening Concentrations (TSC) for evaluation of CoC impacts on target species.

Chemical Class	Analyte	Tissue[1] Screening Conc., (ug/g dry)
Metals	Arsenic	10.7
	Cadmium	0.3
	Chromium	1.2
	Copper	1.1
	Lead	0.4
	Mercury	0.8
	Nickel	2.2
	Silver	2.5
	Zinc	18.7
PAHs	Acenaphthene	840
	Acenaphthylene	30
	Anthracene	120
	Benzo(a)anthracene	1,153
	Benzo(a)pyrene	2,773
	Benzo(b)fluoranthene	2,793
	Benzo(g,h,i)perylene	6,727
	Benzo(k)fluoranthene	2,773
	Chrysene	1,153
	Dibenz(a,h)anthracene	1,113
	Fluoranthene	120
	Fluorene	73
	Indeno(1,2,3-cd)pyrene	6,727
	Naphthalene	43
	Phenanthrene	80
	Pyrene	280
PCBs	Sum PCB Congeners x 2	2.93
Pesticides	Aldrin	4.73
	Hexachlorobenzene	213.33
	Mirex	0.12
	o,p'-DDE	0.36
	p,p'-DDE	0.36
TBT	Tributyltin	0.03

[1]Shepard, (1995)

Table 6.6-3. Summary of Tissue Screening Concentration Hazard Quotients for Target Receptors.

Chemical		Hard Clam		Mummichog Fish		Soft Shell Clam		Oyster		Ribbed Mussel	
Class	Analyte	Mean	Max.	Mean	Max.	Mean	Max.	Mean	Max.	Mean	Max.
Metals	Arsenic	0.7	1.7	0.2	0.4	0.4	1.0	0.5	0.8	0.7	1.2
	Cadmium	1.3	8.3	1.1	4.7	2.7	8.1	6.2	11.4	4.0	11.9
	Chromium	2.0	3.4	0.6	0.8	2.1	6.3	7.3	16.3	7.2	22.8
	Copper	11.2	62.5	1.2	2.5	26.4	86.5	141.8	309.5	12.0	39.4
	Lead	1.9	8.3	1.9	4.1	44.6	126.7	6.8	26.5	3.7	14.6
	Mercury	0.1	0.1	0.1	0.1	19.9	59.6	3.7	42.9	3.5	55.3
	Nickel	1.9	4.9	0.2	0.6	2.9	6.0	3.7	9.3	7.9	27.8
	Silver	0.6	1.7	0.4	1.0	1.2	2.0	2.8	7.2	2.7	7.2
	Zinc	4.9	6.2	9.9	11.6	5.6	12.4	53.5	228.2	2.9	4.7
PAHs	Acenaphthene	< 0.001	< 0.001	< 0.001	< 0.001	0.000	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Acenaphthylene	< 0.001	< 0.001			< 0.001	< 0.001	< 0.001	< 0.001	0.001	0.002
	Anthracene	< 0.001	< 0.001			< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Benzo(a)anthracene	< 0.001	< 0.001			< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Benzo(a)pyrene	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Benzo(b)fluoranthene	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Benzo(g,h,i)perylene	< 0.001	< 0.001			< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Benzo(k)fluoranthene	< 0.001	< 0.001			< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Chrysene	< 0.001	< 0.001			< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001
	Dibenz(a,h)anthracene	< 0.001	< 0.001			< 0.001	< 0.001				
	Fluoranthene	< 0.001	< 0.001	< 0.001	< 0.001	0.002	0.002	< 0.001	0.001	0.003	0.005
	Fluorene	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	0.003	< 0.001	< 0.001	0.002	0.002
	Indeno(1,2,3-cd)pyrene	< 0.001	< 0.001			< 0.001	< 0.001			< 0.001	< 0.001
	Naphthalene									0.015	0.016
	Phenanthrene	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	0.004	0.007
	Pyrene	< 0.001	< 0.001			< 0.001	< 0.001	< 0.001	< 0.001	0.001	0.002
PCBs	Sum PCB Congeners x 2	0.057	0.164	0.637	1.093	0.088	0.166	0.199	0.346	0.193	0.419
Pesticides	Aldrin							< 0.001	< 0.001	< 0.001	< 0.001
	Hexachlorobenzene	< 0.001	< 0.001			0.000	0.000				
	Mirex			0.004	0.006			0.002	0.002	0.003	0.004
	o,p'-DDE	0.001	0.004	0.003	0.006	0.002	0.002	0.003	0.009	0.005	0.010
	p,p'-DDE	0.016	0.060	0.223	0.323	0.024	0.040	0.074	0.101	0.074	0.129
TBT	Tributyltin	0.7	1.1			10.9	10.9				

Table 6.6-4. Qualitative Summary for Tissue Residue Hazard Quotients by Ecological Exposure Zone.

	Tissue Residue Effects							
	Ecological Exposure Zone[1,3]							
Measurement Endpoint	LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR
Hard Clam HQ								na
Soft Shell Clam HQ								na
Oyster HQ	+			+				na
Ribbed Mussel HQ								na
Mummichog Fish HQ	+			+				na
EEZ Score[2]	+	-	-	+	-	-	-	na

[2]EEZ Scoring based on + = slight risk, ++ = moderate risk; +++ = substantial risk

[3]Consult Section 6.0 and Figure 6.0-1 for description and location of EEZs.

7.0 SUMMARY AND CONCLUSIONS

This risk assessment follows the organization suggested in Eco Update (U.S. EPA, 1991) with appropriate elements from the EPA Region I Supplemental Risk Assessment Guidance for the Superfund Program (U.S. EPA, 1988) and Risk Assessment Guidance for Superfund, Volume II Environmental Evaluation Manual (U.S. EPA, 1989). These guidance documents recommend a "weight of evidence" approach to assessing potential ecological risks. The approach should be based on evaluation of constituent analytical data relative to environmental benchmarks, direct field observations, and selected field and laboratory studies from the scientific literature. The following discussion provides a synthesis of the risk characterization findings.

7.1 INTERPRETATION OF SEVERITY AND SIGNIFICANCE OF RISKS

A summary of environmental risk for the Allen Harbor study area is presented in Table 7.1-1. The information is presented by Ecological Exposure Zones (EEZs) corresponding to various geographic and depth-related characteristics of the region (Figure 6.0-1). Weights of evidence results are derived from risk analyses of exposure and effects characterization data presented in Section 6. In condensing this information into a single table, the reader must remain aware that the quantitative basis for each weight of evidence is numerically rooted in fundamentally different responses. As discussed in Section 6, each Weight of Evidence (WoE) ranking was based on a qualitative professional judgement in which substantial risk was indicated by triple plus (+++), moderate risk as a double plus ($^{++}$), slight risk as a single plus ($^{+}$), and no apparent risk by a minus sign (-). No attempt was made to adjust rankings of individual weights to provide comparability of rankings across individual weights of evidence.

EEZs with two or more individual WoE rankings of slight risk were considered adequate evidence of "slight" risk overall. Similarly, EEZs with two or more individual WoE rankings of moderate risk were considered adequate evidence of moderate risk overall. The risk manager should, however, review the ranking strategy and assess the effects of alternate weighting of the magnitude and spatial distribution of risks as discussed below.

Risks are evident primarily in EEZs proximal to the landfill and the western portion of Calf Pasture Point bordering on Allen Harbor (Table 7.1-1). It is notable that prospective risks are not apparent in intertidal or subtidal environments of Calf Pasture Point.

The relevance of these findings on the selected assessment endpoints/target receptors identified in Table 3.3-2 is that the vitality of pelagic, epibenthic and infaunal communities located in habitats proximal to the Allen Harbor landfill to the north (LN-W) and south (LS-V/W) may be at moderate risk. All of the selected target species were found in the locations.

Indicators of risk including measured toxicity and biota condition are in generally good agreement with predicted risks (sediment, porewater and tissue hazards) for the Landfill South Vegetated/Wetland (LS-V/W) region, and the Landfill North - Wetland (LN-W) EEZs, and reduces uncertainty with regard to the overall conclusion. In contrast, the lack of agreement among weights of evidence for LS-D, LM-D and CP-W EEZs may be in part due to the conservative nature of points of comparison and or benchmarks, such as the use of reference site chemistry and ER-Ls as indicators of risk.

Taken as a whole, risks presented in Table 7.1-1 indicate the spatial distribution of risks is restricted only to the narrow intertidal zone at the toe of the landfill. Hence, the implication of these findings with respect to selected assessment endpoints is that

the vitality of pelagic, epibenthic and infaunal communities do not generally appear to be at risk in Allen Harbor. It is apparent that even within the impacted zones, dense assemblages of target species, particularly bivalves, were found. Thus, the results of this assessment suggests that there does exist important chemical stressors, but this stress is presently not greatly impacting the viability of populations present there.

Although the study findings do provide a coherent picture of the spatial magnitude and extent of risks, elucidating the CoC drivers underlying these trends was less straight forward. While fish and oyster condition appear affected by zinc, neoplasia in clams appears related to PAH body residues. The data suggest that amphipod survival could be affected by porewater PCBs, although the HQs were relatively low (<4) relative to other HQ values observed. The PAHs were also discounted as a principal driver of impacts on this measurement endpoint.

Causal factors underlying the sea urchin trends could not be convincingly determined. SEM/AVS measurements, as indicators of metal bioavailability, did not explain toxicity responses. Statistically significant correlations between 1) bivalve condition and sediment/porewater Hazard Quotients for PCBs and PAHs; 2) amphipod toxicity and porewater PCB HQs, 3) increased incidence of neoplasia with PAH body residues, 4) reduced length in fish and reduced condition in oysters with increasing body burden of metals were observed which would be expected if true exposure response mechanisms were operative. In each of these cases, observations at one or more stations comprising the LS-V/W or LN-W EEZ were involved.

A possible factor underlying the apparent diversity of responses may be related to the spatial sampling scales. This investigation was tightly focused in the nearshore environment of the landfill, many stations were often only a few meters away from the landfill face. Recognizing that landfill materials were not homogeneously mixed prior to disposal, the measurement endpoints may, under such conditions, be responding to highly site-specific exposure sources and not be expected to cohesively implicate one

or more CoCs. Variability among station chemistry data observed in this study does support this hypothesis. It is also possible that each measurement endpoint has differential sensitivity to the respective CoC exposure, amplifying the apparent heterogeneity of responses.

7.2 Other Potential Sources of Stress and CoCs

Other sources of contamination may enter the Allen Harbor system through creeks to the north and south of the site, although this report does not directly address these sources. This assessment found no evidence to suggest that VOA transport through groundwater to Allen Harbor offshore habitats is occurring, nor were elevated CoC concentrations detected in substrate sediment layers depth which otherwise might cause concern though resuspension.

Other system stressors, such as contaminant releases associated with recreational boating, may be important factors in the overall characterization of risks to Allen Harbor, but those affects appear to be localized to the marina area (Figure 6.0-1). In addition, the weight of evidence suggests present conditions at Calf Pasture Point do not appear to provide additional environmental impact on Allen Harbor. Hence it is concluded that indigeneous biological communities in the immediate vicinity of the southern portion of the landfill are at risk primarily due to landfill-related stressors.

7.3 Limitations of the Assessment

The conclusions drawn in this assessment are based on an extensive database of sediment chemistry, biological indicators, and toxicity evaluations, with broad spatial and temporal coverage. The data are internally consistent and supportive, and generally are of very high quality, meeting and exceeding, for example, detection limits as specified by the NOAA Status and Trends Program. Therefore, the values can be

interpreted with confidence for comparisons to commonly accepted guidelines such as ER-Ls (Long, et al., 1995).

The assessment of ecological risk is a process of minimizing uncertainty with regard to characterization of exposure and effects and the integration of these data as cause-effect relationships. The risk conclusions reached in this study are based on weight of evidence; those areas exhibiting more numerous lines of evidence for or against adverse impact are associated with less uncertainty in the conclusion. The present study provides extensive weight of evidence for impacts in the south landfill zone of Allen Harbor; hence there would appear a high probability of risk occurring in this area. Other areas appear to pose little probability of risk, given a general lack of both exposure and effects. Greater uncertainty accompanies the degree of risk presently occurring at the north landfill site and at Calf Pasture Point. Each site has indicators of both exposure and effects on a station-specific basis, but the indicators are not pervasive, or well correlated.

With regard to the adequacy of the conceptual models for exposure pathways, the apparent similarity among species in chemical bioaccumulation as predicted by equilibrium partitioning suggests that this pathway is well understood. Similar models for metals bioaccumulation are not presently available, and accordingly there exists greater uncertainty with regard to the extrapolation of data from the target species to other species in the ecosystem that are of concern (e.g., winter flounder). This uncertainty is reflected in the greater variance observed among metals BAF values relative to that observed for organic chemicals. Yet, the variance is constrained to approximately 2-5 fold for metals, and two-fold for organics, hence this uncertainty may at first approximation be applicable to other unstudied species.

The present study in conjunction with prior investigations yields an extensive data set comprising both spatial and temporal coverage. Although the present investigation was constrained to a narrow sampling interval, there was an extensive

overlap of station coverage with the prior RAPS study where seasonal effects were considered. As no prior findings of ecological risk were apparent in the prior study in concurrence with findings for those areas overlapping with this study, seasonality would not appear to introduce great uncertainty into this assessment.

The present study was conducted under a comprehensive Work/Quality Assurance Plan, data validation has been performed and found to meet the study requirements. Potential errors in the study design and protocols were minimized through thorough peer review and evaluation. Data collection activities were reasonably complete, with perhaps less than desirable abundances of fish samples, particularly at reference sites. This short coming required occasional mixing of multiple species to obtain sufficient biomass for analyses and limited the extent of biological measurements to be made on this target receptor. However, the available tissue residue data does suggest comparability among the fish species used in the mixture, reducing uncertainty in characterization of exposure pathways.

Table 7.1-1 Summary of Risks by Ecological Exposure Zone for Allen Harbor Landfill and Calf Pasture Point.

Report Section	Weight of Evidence	Ecological Exposure Zone[1,3]							
		LS-V/W	LS-D	LM-D	LN-W	CP-W	CP-SW	CP-SD	MAR
6.1	Sediment CoC Enrichment Factor	++	+	+	-	-	-	+	+
6.3	Tissue CoC Enrichment Factor	++	-	-	++	+	+	-	N/A
6.4	Toxicity	++	+	-	++	-	-	-	-
6.2	Biota Condition	++	-	-	+	+	-	-	N/A
6.5.1	Sediment Hazard	++	+	+	++	-	-	-	-
6.5.2	Pore Water Hazard	+	-	-	+	-	-	-	-
6.6	Tissue Residue Effects	+	-	-	+	-	-	-	N/A
	EEZ Ranking[2]	++	+	+	++	+	-	-	+

[2]**EEZ Ranking:** **+** = slight risk; **++** = moderate risk; **+++** = substantial risk; **-** = no indication of impact; **N/A** = no data to evaluate impact.

[1]**LS-V/W** = V1, V2, W1-W5; **LS-D** = D1-D3, AH13; **LM-D** = D4-D10, AH2-AH5; **LN-W** = W6-W10,AH12;

CP-W = V3, W11, W12, D14, AH9, AH10, AH14; **CP-SW** = V4, W13, W14, FDA; **CP-SD** = D11-D13; **MAR** = AH6-AH8, AH11.

LS-V/W = Landfill South- Vegetated/Wetland **LM-D** = Landfill Middle-Deep, **LN-W** = Landfill North-Wetland, **LS-D** = Landfill South-Deep

CP-W, SW, SD = Calf Pasture Point- West, South Wetland, and South Deep, respectively; **MAR** = RAPS Phase I Marina Influence.

[3]Consult Figure 6.0-1 for location of EEZs

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