NAVAL AIR STATION NORTH ISLAND
SAN DIEGO, CALIFORNIA
Remedial Investigation/RCRA Facility Investigation
Site 1 – Shoreline Sediments

Final Report Volume I

November, 1999

Prepared For:	**Points of Contact:**
U.S. Navy Southwest Division Naval Facilities Engineering Command	Mark Bonsavage Site 1 Remedial Program Manager
Prepared By:	
Space and Naval Warfare Systems Center (SPAWAR) San Diego, California	Kenneth Richter, Ph.D.
With:	
Arthur D. Little, Inc. Encinitas, California	Cynda Maxon
And	
MEC Analytical Systems, Inc. Carlsbad, California	Cindy Collins, Ph.D.

TABLE OF CONTENTS

VOLUME I

ACRONYMS AND ABBREVIATIONS vi

EXECUTIVE SUMMARY ES-1

Background ES-1
Approach ES-1
Results and Recommendations ES-2

Section 1 INTRODUCTION 1-1

1.1 Purpose of Report 1-1
1.2 Report Organization 1-2
1.3 Site Background 1-3
1.3.1 Site Location 1-3
1.3.2 Site History 1-4
1.3.3 Aerial Photograph Review 1-6
1.3.4 Demography and Land Use 1-9
1.3.5 Bay Bathymetry and Hydrodynamics 1-10
1.4 Evolution of the Study Design 1-12
1.5 Previous Investigations 1-13
1.5.1 Outfalls 1 and 2 1-14
1.5.2 Outfall 3 1-14
1.5.3 Outfall 4 1-15
1.5.4 Outfall 5 1-16
1.5.5 General Studies 1-16

Section 2 REMEDIAL INVESTIGATION 2-1

2.1 Study Design 2-1
2.1.1 In-bay Surface Sediments 2-1
2.1.2 Subsurface Sediments 2-5
2.2 Field Methods 2-8
2.3 Analytical Program 2-9
2.3.1 Physical Tests 2-9
2.3.2 Chemical Tests 2-9
2.4 Data Analysis 2-18
2.4.1 Descriptive Statistics 2-18
2.4.2 Comparative Statistics 2-19
2.4.3 Gradient Analyses 2-21
2.5 Quality Assurance/Quality Control Methods 2-21
2.5.1 Field Program 2-21
2.5.2 Data Management 2-22
2.5.3 Physical and Chemical Analyses 2-22
2.5.4 Bioassays 2-22
2.5.5 Data Management 2-22

Section 3 PHYSICAL CHARACTERISTICS 3-1

3.1 Surface Sediment Grain Size 3-1
3.2 Total Organic Carbon 3-3
3.3 Designation of Sediment Groups 3-3

Section 4 NATURE AND EXTENT OF CONTAMINATION 4-1

4.1 Surface Sediments 4-1
4.1.1 Polycyclic Aromatic Hydrocarbons (PAHs) 4-1
4.1.2 Organochlorine Pesticides 4-5
4.1.3 Polychlorinated Biphenyls (PCBs) 4-6
4.1.4 Metals 4-11
4.1.5 Butyltins 4-12
4.1.6 Cyanide and Sulfides 4-12
4.1.7 Chemicals Not Statistically Compared 4-12
4.2 Subsurface Sediments 4-14
4.2.1 In-Bay Outfalls 3-8 4-16
4.2.2 Ocean Outfall 1,2 4-16
4.2.3 Ocean Outfall 16 4-18
4.3 Chemicals of Potential Concern 4-20

Section 5 NATURE AND EXTENT OF TOXICITY AND BIOACCUMULATION 5-1

5.1 10-day Amphipod Toxicity Test 5-1
5.2 4-day Mysid Shrimp Toxicity Test 5-5
5.3 20-day Polychaete Toxicity Test 5-5
5.4 48-Hour Bivalve Toxicity Test 5-6
5.5 28-Day Bioaccumulation in the Bentnose Clam 5-8
5.5.1 Polycyclic Aromatic Hydrocarbons (PAHs) 5-8
5.5.2 Pesticides and PCBs 5-9
5.5.3 Metals 5-10
5.5.4 Chemicals with Insufficient Detections for Statistical Comparison 5-11
5.5.5 Comparison of Individual Stations 5-12

Section 6 RISK ASSESSMENT SCREENING ECOLOGICAL AND HUMAN HEALTH 6-1

6.1 Ecological Risk Assessment Screening 6-1
6.1.1 Objectives and Approach 6-1
6.1.2 Conceptual Site Model for Ecological Risk 6-2
6.1.3 Review of Results 6-2
6.1.4 Data Trends 6-4
6.1.5 Ecological Risk Screening Conclusions 6-5
6.2 Human Health Risk Assessment Screening 6-6
6.2.1 Objectives and Approach 6-6
6.2.2 Chemicals Screened 6-6
6.2.3 Exposure Setting 6-8
6.2.4 Exposure Pathways and Conceptual Site Model 6-11
6.2.5 Cancer Risk and Non-Cancer Hazard 6-11
6.2.6 Cancer Risk for Ocean Outfalls 6-15
6.2.7 Human Health Risk Screening Conclusions 6-16

Section 7 CONCLUSIONS AND RECOMMENDATIONS 7-1

7.1 Ecological Risk Screening Conclusions 7-1
7.2 Human Health Risk Screening Conclusions 7-1

Section 8 REFERENCES 8-1

APPENDICES

APPENDIX A RCRA Facility Investigation Requirements
APPENDIX B Volatile Organic Compounds
APPENDIX C Radiological Isotopes
APPENDIX D Biological Test Results
APPENDIX E Physical and Chemical Test Results
APPENDIX F Comments and Responses

VOLUME II

Statistical Analyses
External Validation

VOLUME III

Field and Analytical Methods
Quality Assurance/Quality Control Methods
Quality Control Results
Data Validation Reports

LIST OF FIGURES

Figure 1-1. Locations of Naval Air Station North Island and the outfalls investigated 1-4
Figure 1-2. Aerial photograph of NAS North Island, 1953. 1-7
Figure 1-3. Aerial photograph of NAS North Island, 1975. 1-8
Figure 1-4. Aerial photograph of outfall 16 vicinity, NAS North Island, 1977. 1-9
Figure 1-5. Bathymetry along NAS North Island. 1-11
Figure 2-1. Locations of the 9 Site 1 outfalls around the perimeter of NAS North Island and the 10 reference stations in San Diego Bay. 2-2
Figure 2-2. NAS North Island sampling stations identified by outfall. Semicircle sampling arcs are shown for 30, 100, and 150 meter stations. 2-3
Figure 2-3. Fine-grained sediment distributions around NAS North Island. 2-4
Figure 2-4. Total organic carbon distribution around NAS North Island. 2-4
Figure 2-5. Selection and location of reference stations. 2-7
Figure 3-1. Regression of percent total organic carbon with percent fine sediment. 3-4
Figure 3-2. Percent total organic carbon with percent fine sediment by station. 3-5
Figure 3-3. Locations of coarse-grained and fine-grained reference stations in San Diego Bay. 3-7
Figure 3-4. NAS North Island sampling stations identified by outfall and sediment type group. 3-8
Figure 4-1. Relative loadings for principal components 1 and 2. 4-4
Figure 4-2. Station scores for principal components 1 and 2. 4-4
Figure 4-3. Significant results for t-tests on contaminant concentrations in surface sediments. 4-6
Figure 4-4. Organic chemical concentrations (ng/g) for outfall surface sediments above the 99% predictive limit of corresponding reference stations. 4-7

Figure 4-5. Metal concentrations (mg/g) for outfall surface sediments above the 99% predictive limit of corresponding reference stations. 4-8
Figure 4-6. Ratios of in-bay surface concentrations to 99% reference predictive limit. 4-9
Figure 4-7. Ratios of subsurface contaminant concentrations to 99% predictive limit for ocean channel and in-bay outfalls. 4-14
Figure 5-1. T-test results for toxicity endpoints for fine and coarse sediment groups. 5-2
Figure 5-2. Toxicity results below the 95% predictive limit for corresponding sediment group reference stations. 5-3
Figure 5-3. Chemical concentrations in tissues exposed to outfall sediments above the 99% predictive limit for corresponding reference stations. 5-13
Figure 5-4. A: Silver concentration in clam tissue vs. sediment for outfall stations exceeding the coarse reference station mean tissue concentration. B: Same plot as A, except sediment silver concentration is normalized by sediment percent fines. 5-14
Figure 6-1. Conceptual site model for ecological receptors - NAS North Island. 6-3
Figure 6-2. Current land use at NAS North Island (from U.S. Navy 1991). 6-9
Figure 6-3. Potential future land use at NAS North Island (from U.S. Navy 1991). 6-10
Figure 6-4. Conceptual site model for human health screening. 6-13
Figure 6-5. Residential cancer risk in surface sediments based on 1998 PRGs. 6-17
Figure 6-6. Residential cancer risk at intertidal outfall stations based on 1998 PRGs. 6-18
Figure 6-7. Residential non-cancerous hazard in surface sediments, based on 1998 PRGs. 6-19
Figure 6-8. Summed hazard index by organ system, and station 6-20

LIST OF TABLES

Table 1-1. Summary of historical discharges of industrial waste (1938-45 to 1972) from outfalls 1 through 8, and 16. 1-5
Table 1-2. Aerial photographs reviewed (March 1953 - August 1978). 1-6
Table 1-3. Water depth and habitat type of locations sampled at in-bay outfalls 3-8. 1-12
Table 1-4. Summary of previous investigations of sediment quality at NAS North Island Site 1 outfalls. ... 1-14
Table 2-1. Station identification, distance from outfall, and location. 2-6
Table 2-2. Station identifications for BPTCP stations used as Site 1 reference stations. 2-8
Table 2-3. Physical and chemical tests performed for sediments and tissues. 2-9
Table 2-4. Semivolatile organic analytes. Analyzed by USEPA Method 8270 modified. 2-10
Table 2-5. Pesticides and PCBs. Analyzed by U.S. EPA Method 8082 modified. 2-12
Table 2-6. Metal and butyltin analytes, corresponding methods, and dry weight detection limits. 2-13
Table 2-7. Cyanide and sulfides. 2-14
Table 2-8. Test description, species used, measured effects, and relevance of selected laboratory sediment bioassays. 2-15
Table 3-1. Size class for sediment grain size. 3-1
Table 3-2. Percent gravel, sand, silt, and clay, mean grain size, and total organic carbon (TOC) for each station. 3-2
Table 3-3. Designation of outfall and reference stations by percent fines. 3-6
Table 4-1. Mean concentrations (± std error) of summed low and high molecular weight PAHs in surface sediments. 4-2
Table 4-2. Mean concentrations (± std error) of select chlorinated pesticides and PCBs in surface outfall and reference stations. Concentrations are ng/g dry weight. 4-5
Table 4-3. Mean metal concentrations (± std error) in surface sediments at outfall and reference stations. 4-10
Table 4-4. Mean concentrations of butyltins in surface sediments at outfall and reference stations. 4-11
Table 4-5. Summary of analytes that were detected in outfall sediments but had less than 3 detectable values in the corresponding reference group. 4-13

Table 4-6. In-bay subsurface analytes with concentrations greater than the corresponding reference 99% predictive limit (ratios >1). 4-15
Table 4-7. Ocean outfall 1,2 subsurface analytes with concentrations greater than the corresponding reference 99% predictive limit (ratios >1). 4-17
Table 4-8. Ocean outfall 16 subsurface analytes with concentrations greater than the corresponding reference 99% predictive limit (ratios >1). 4-18
Table 4-9. Analytes not considered initial chemicals of concern. 4-20
Table 4-10. Chemicals of concern for ecological risk screening assessment. 4-21
Table 5-1. Summary of negative control, outfall, and reference station toxicity test results for *Rhepoxynius abronius*. 5-1
Table 5-2. Summary of negative control, outfall, and reference station toxicity results for *Holmesimysis costata*. 5-5
Table 5-3. Summary of negative control, outfall, and reference station toxicity test results for *Neanthes arenaceodentata*. 5-6
Table 5-4. Summary of negative control, outfall, and reference station toxicity test results with *Crassostrea gigas*. 5-7
Table 5-5. Mean concentrations (± std error) of summed low and high molecular weight PAHs and PAH_{16} in clam tissues. 5-8
Table 5-6. Mean total concentrations (± std error) of select chlorinated pesticides and PCBs in clam tissues. 5-9
Table 5-7. Mean concentrations (± std error) of metals in clam tissues. 5-10
Table 5-8. Summary of analytes that were detected in outfall tissues but had less than three detectable values in the corresponding reference group. 5-11
Table 6-1. Decision matrix used to determine ecological risk (adapted from Bechtel 1995). 6-1
Table 6-2. Elevated chemicals in sediment and clam tissue at stations with significant toxicity. 6-5
Table 6-3. Ecological evaluation (applied decision matrix) for Site 1 in-bay surface sediments. 6-5
Table 6-4. Chemicals and corresponding 1998 residential soil preliminary remediation goals (PRGs) 6-7
Table 6-5. Residential cancer risk and hazard values based on 1998 PRGs. 6-14
Table 6-6. Organ system-specific hazard index for important metal contaminants. 6-16

ACRONYMS AND ABBREVIATIONS

ARAR	applicable or relevant and appropriate requirements
ASTM	American Society for Testing and Materials
ATG	Allied Technology Group
BESAP	Bioassay and Environmental Sampling and Analysis Plan
BHC	hexachlorocyclohexane
BPTCP	Bay Protection and Toxics Cleanup Program
CDFG	California Department of Fish and Game
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CTO	Contract Task Order
CVAAS	cold vapor atomic absorption spectrophotometry
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichlorethane
DGPS	Differential Global Positioning System
DNR	do not report
DTSC	(Cal-EPA) Department of Toxic Substances Control
EC50	Effects Concentration for 50% of test organisms
ER-L	effects range low
ER-M	effects range median
ESI/RFI	Extended Site Investigation/RCRA Facility Investigation
EPA	United States Environmental Protection Agency
FAAS	flame atomic absorption spectrophotometry
GC/ECD	gas chromatography/electron capture detection
GC/FPD	gas chromatography/flame photometric detector
GC/MS	gas chromatography/mass spectrometry
GFAAS	graphite furnace atomic absorption spectrophotometry
HCB	hexachlorobenzene
HLA	Harding Lawson Associates
HMW	high molecular weight
HPLC	high pressure liquid chromatograph
ICP/MS	inductively coupled plasma/mass spectrophotometry
IUPAC	International Union of Pure and Applied Chemistry
IWTP	Industrial Waste Treatment Plant
LC50	Lethal Concentration for 50% of test organisms
LMW	low molecular weight
MDL	method detection limit
MEC	MEC Analytical Systems, Inc.
MLLW	mean lower low water
NAS	Naval Air Station

NOAA	National Oceanic and Atmospheric Administration
NOEC	No Observable Effects Concentration
NRaD	Naval Research and Development
OSWER	Office of Solid Waste and Emergency Response
PAH	polynuclear aromatic hydrocarbon
PCB	polychlorinated biphenyl
PIV	post-injection volume
PQL	Practical Quantitation Limits
PRG	preliminary remediation goal
QA	quality assurance
QAPP	Quality Assurance Project Plan
QC	quality control
RCRA	Resource Conservation and Recovery Act
RDT&E	Research, Development, Technology, and Evaluation Division
RFI	RCRA Facility Investigation
RI	Remedial Investigation
RPD	relative percent difference
RSD	relative standard deviation
RWQCB	California Regional Water Quality Control Board
SANDAG	San Diego Association of Governments
SDG	sample delivery group
SI/FS	site investigation/feasibility study
SOP	standard operating procedure
SPAWAR	Space and Naval Warfare Systems Center
SRM	standard reference material
STLC	soluble threshold limiting concentration
SVOC	semivolatile organic compound
SWDIV	Southwest Division Naval Facilities Engineering Command
SWMU	solid waste management unit
SWRCB	State Water Resources Control Board
TOC	total organic carbon
TTLC	total threshold limiting concentration
USACOE	United Stated Army Corps of Engineers
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
VOC	volatile organic compound
ZGFAAS	Zeeman graphite furnace atomic absorption spectrophotometry

EXECUTIVE SUMMARY

This report presents the background, approach, results and recommendations of a Remedial Investigation/RCRA Facility Investigation (RI/RFI) conducted for Installation Restoration Site 1 - Shoreline Sediments at NAS North Island.

Background

Installation Restoration Site 1 constitutes the San Diego Bay marine sediments and fresh water channel sediments that were potentially impacted by historical discharges of industrial waste and sewage from 16 NAS North Island storm water outfalls. Nine of these outfalls were investigated in this RI/RFI: outfalls 1 through 8 and 16. Outfalls 9 through 15 were part of a time-critical removal action and were not included in this investigation. Outfalls 1, 2 and 16 drain into two separate freshwater channels located on the ocean side of North Island. Outfalls 1 and 2 discharge side by side into one of these channels and were treated together in this investigation. While normally blocked by beach sand, water in the channels occasionally flowed into the ocean during heavy winter rains. Outfalls 3 through 8 empty into San Diego Bay and are located at nearly regular intervals around the bay perimeter of North Island, beginning with outfall 3 at the mouth of the bay.

From the 1930s to the 1970s these storm water outfalls discharged raw sewage, metal waste, organic solvents, spilled fuel, and pesticide residue which potentially impacted surrounding shoreline sediments. The purpose of this study was to investigate whether these sediments pose unacceptible risk to human health or bay ecology, and if so, require future actions (e.g., more studies, clean-up).

Approach

Together with the regulators, environmental groups, and the Restoration Advisory Board, the Navy developed a unique approach to assess ecological risk posed by Site 1 marine sediments, since no absolute sediment quality standards exist. This approach compared the results of chemistry, toxicity, and bioaccumulation tests for outfall sediments with results from relatively clean, non-toxic reference sediments from San Diego Bay. Only bay sediments near outfalls 3 through 8 were evaluated for marine ecological risk, since sediments in the ocean channels associated with outfalls 1, 2 and 16 were often dry and influenced by fresh water discharge only.

A key feature of this study was choosing reference sediments to compare Site 1 sediments with for assessing ecological risk and future actions. Two selection criteria had to be met: 1) reference sediments had to have the same physical characteristics as the outfall sediments, and 2) reference sediments had to represent clean and non-toxic sediments in San Diego Bay. The first criterion was important because physical characteristics, in particular grain size and total organic carbon, determine both a sediment's capacity to adsorb (bind) chemical contaminants, and the bioavailability (accessability) and toxicity of those chemicals. Fine-grained sediments with a high percentage of organic carbon tend to adsorb and accumulate chemicals but are often not toxic because the chemicals are not available to marine organisms. Coarse-grained sediments in contrast often will not adsorb contaminants well, but those present are usually bioavailable and hence toxic. Previous studies of Site 1 sediments found both fine- and coarse-grained sediments near the outfalls, as well as a range of organic

carbon content. Therefore, if adsorbed chemical concentrations, toxicity, and bioaccumulation were going to be tested, it was important that outfall sediments be compared to reference sediments of similar grain size and organic carbon. In order to apply the second criterion, the Navy reviewed sediment chemistry and toxicity data from a bay-wide study conducted jointly by the National Atmospheric and Oceanic Administration (NOAA) and the State Water Quality Control Board. Of 102 locations sampled in San Diego Bay for that study, 15 of the least toxic and cleanest were chosen as potential reference sites for the Site 1 investigation. Of these fifteen, ten were selected as reference sites based on their similarity to the range of grain size and organic carbon of Site 1 sediments. In order to put relative differences between outfall and reference sediments in a broader perspective, chemical concentrations that were higher in outfall sediments were compared to guidelines developed using chemistry and toxicity data reported for sediments collected throughout the United States (Long et al. 1995). These guidelines, known broadly as "effects range-low" and "effects range-median" (ER-L/ER-M) define sediment chemical concentration ranges that are rarely or occasionally associated with adverse biological effects in marine and estuarine sediments. These guidelines are based on hundreds of laboratory and field studies conducted throughout the United States.

To assess human health risk, surface sediments from all Site 1 outfalls were screened using sediment chemical concentrations against the U.S. EPA's residential Preliminary Remediation Goals (PRGs). Preliminary Remediation Goals are defined as soil chemical concentrations that pose little or no cancer risk or non-cancerous hazard to hypothetical residents living in close physical contact of the soil for 70 years. Four areas including subtidal and intertidal sediments, and sediments from each of the two ocean channels were evaluated separately. The maximum concentration of each chemical in surface sediments from each of the four areas was compared to the corresponding PRG concentration. The resulting cancer risk and non-cancerous hazard indices were summed for all individual chemicals for each area. Each area was then assessed in terms of the total potential threat to human health.

RESULTS AND RECOMMENDATIONS

Comparison of Site 1 outfall sediments to reference sediments of appropriate grain size, suggested that there were no "hot-spots" of chemical contamination, toxicity or bioaccumulation. Several chemicals measured in the bay sediments were elevated compared with reference sediments, including total DDT, PCBs, and cadmium. However, these outfall sediments were at or below ER-L guideline concentrations. Toxicity results showed no significant differences between Site 1 and reference sediments. Bioaccumulation results indicated that silver was the only chemical significantly higher in clams exposed to Site 1 sediments compared to those exposed to reference sediments. However, silver tissue concentrations in clams exposed to Site 1 sediments were near background concentrations measured in similar west coast estuarine animals, and were derived from outfall sediment silver concentrations below the ER-L guideline concentration. Therefore, based on chemistry, toxicity, and bioaccumulation results, no further action was recommended for Site 1 sediments in San Diego Bay from an ecological perspective.

In addition to the assessments performed on surface sediments, subsurface sediment cores were evaluated to look for vertical trends in chemical concentrations only. Sediment cores were taken at each of the San Diego Bay outfalls and at the two ocean channels. In-bay sediments were examined down to 6 ft; ocean channel sediments were examined to 10 ft to sample above and below the water table which resided at approximately

8 ft. In general, most contaminants decreased with depth in both the in-bay and ocean channel sediments. However, DDT, PCBs, cadmium, and lead were elevated compared with surface reference sediments in cores collected closest to the ocean outfalls 1, 2 and 16. Although elevated, none of the contaminants exceeded corresponding ER-M values.

The results of the conservative human health risk screening, which included cancer risk and non-cancerous hazard revealed that only channel sediments associated with outfall 16 warranted further investigation due to the hazard posed by cadmium, lead and other metals found in the surface sediments. An analysis of remedial alternatives for outfall 16 is recommended.

Section 1 INTRODUCTION

This report presents the results, conclusions, and recommendations of the Remedial Investigation (RI)/Resource Conservation and Recovery Act (RCRA) Facility Investigation (RFI) conducted April 1 through December 31, 1996, at Navy Installation Restoration (IR) Site 1 - Shoreline Sediments, Naval Air Station (NAS) North Island in San Diego, California. The investigation was performed by Space and Naval Warfare Systems Center (SPAWAR) on behalf of the United States Navy, Southwest Division Naval Facilities Engineering Command (SWDIV).

1.1 Purpose of Report

The purpose of this report is to document the results, findings and conclusions of the RI/RFI conducted to assess potential contamination and associated risks in shoreline sediments caused by historical stormwater discharges containing hazardous waste. Shoreline sediments adjacent to the historical storm drain system of 16 outfalls comprise IR Site 1 and therefore, warrant remedial investigation. The conclusions presented in this report will be used to determine if removal or remedial actions are required to achieve No Further Action and ultimately site closure. Outfalls 9 through 15 were not included in this RI/RFI as these shoreline sediments are the subject of a time-critical removal action. A separate RI/RFI will be conducted for these outfalls at the conclusion of the cleanup.

The Site 1 RI/RFI was conducted to fulfill the regulatory requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Resource Conservation and Recovery Act (RCRA). The Navy routinely manages all of its past hazardous waste sites under the CERCLA regulatory framework through the Department of Defense Installation Restoration program. The Installation Restoration program at NAS North Island initiated in 1983 included an assessment study. Site 1 - Shoreline Sediments was one of 12 Installation Restoration sites originally identified. Around that time, the NAS North Island Industrial Waste Treatment Plant (Site 11) became the subject of RCRA requirements to treat and store hazardous wastes. In 1989 a RCRA Hazardous Waste Permit (USEPA No. CA7170090016) was issued for the Industrial Waste Treatment Plant which required corrective actions that included conducting RFIs for solid waste management units (e.g., No. 1, IR Site 1).

While the Navy intends to manage environmental actions at Installation Restoration Site 1 under CERCLA, this investigation also is intended to address the requirements of RCRA corrective action managed by the California EPA Department of Toxic Substances Control (DTSC), formerly California Department of Health Services. A checklist of RI/RFI requirements addressed in this report is presented in Appendix A. The RI/RFI work plan (Bechtel 1995; MEC 1996) and this report are based on the following primary guidance documents: Interim Final RCRA Facility Investigation Guidance, Office of Solid Waste and Emergency Response Directive 9502.00-6 and USEPA 530/SW89.031, May 1989; and Guidance for Conducting Remedial Investigations and Feasibility Studies under Comprehensive Environmental Response, Compensation, and Liability Act, Office of Solid Waste and Emergency Response Directive 9355.3-01 USEPA, October 1988.

In addition, the following regulatory agencies provided input during team meetings contributing to the development and implementation of this investigation:

- California DTSC
- California Regional Water Quality Control Board, Region 9 (RWQCB)
- California Department of Fish and Game
- U.S. Fish and Wildlife Services
- National Oceanic and Atmospheric Administration (NOAA)
- U.S. Environmental Protection Agency (USEPA), Region IX

Overall objectives of this RI/RFI are:

- To determine the nature and extent of contamination in Site 1 sediments;
- To determine the nature and extent of toxicity in Site 1 sediments;
- To determine whether Site 1 sediments pose unacceptable risk to ecological or human health; and
- To determine the need for further studies, remedial action, or no further action for Site 1 sediments.

1.2 Report Organization

This report is organized into three volumes. Volume I contains the main report body and Appendices A through F. Volume II contains computer programs and statistical outputs from analyses of data. Volume III contains detailed descriptions of field and analytical methods, quality control methods and results, and external data validation reports for sediment and tissue chemistry results. Volume I is organized into eight sections as follows.

Section 1- Introduction presents the purpose of the report, citing the regulatory requirements and background for the investigation. This section also details the site background, including location, history, aerial photographs, as well as the site setting. A history of how the study design evolved based on previous investigations and team meetings is also presented.

Section 2 - Remedial Investigation is a study design and methods section. Specific objectives of this investigation are presented along with the study design, rationale for chemical analyses, and bioassay organism selection as well as abbreviated methods used to locate field stations and collect and analyze sediments and porewater. This section also provides methods for analysis of data.

Sections 3, 4 and 5 are results and discussion sections for physical, chemical and bioassay data, respectively. Results for these sections are organized following their corresponding method descriptions in Section 2. Section 3 provides sediment results for total organic carbon and grain size, and sets the stage for the importance of these characteristics in separating Site 1 sediments into two groups that are maintained in subsequent analyses of chemical and biological data. Section 4 presents an overview of chemical results, comparisons of outfall stations to reference stations and interpretation of contamination. Identification of anthropogenic versus naturally occurring metals is discussed, as well as chemical fingerprinting of polycyclic aromatic hydrocarbons (PAHs) to identify petroleum sources. Bioassay results are presented and discussed in Section 5, which is the biological equivalent of Section 4. Quality control results for all tests are presented in Volume III.

Section 6 - Ecological and Human Health Risk Assessment Screening. Section 6.1 integrates chemical and bioassay data from Sections 4 and 5 to assess potential impacts to the benthic ecology and the need for any action at Site 1, such as additional studies and/or cleanup. The decision matrix from the work plan is used as a guide in this assessment. The human health screening assessment is presented in Section 6.2 and is based on chemical concentrations measured in the sediments using preliminary remediation goals for soil. This assessment follows standard guidance from USEPA and California DTSC.

Section 7 presents report conclusions and recommendations;

Section 8 lists all references cited.

Volume I also includes six appendices, as follows:

Appendix A provides a summary table of RCRA requirements for Site 1 and the corresponding report section where they are addressed.

Appendices B - C are methods and laboratory results for sediment VOC and radiological analyses, respectively, performed to support other investigations.

Appendices D - E are summary tables of all biological, physical, and chemical data collected in support of this RI/RFI.

Appendix F presents comments to the draft version of this report furnished by the Restoration Advisory Board (RAB), along with corresponding comments provided by the primary authors of this report and SWDIV.

1.3 Site Background

The following sections provide a brief overview of site location and site history. An aerial photograph review, discussion on land use and summary of water currents in San Diego Bay also are included.

1.3.1 Site Location

NAS North Island is a 10.2-sq km (2,520-acre) facility located in the cities of San Diego and Coronado, California. The Station, operated by the U.S. Navy, is located at the northern end of a peninsula separating San Diego Bay from the Pacific Ocean, and is bounded to the north and west by San Diego Bay, to the south by the Pacific Ocean, and to the east by the City of Coronado. Site 1 consists of the shoreline sediments that received historical discharges from 16 surface stormwater outfalls: 13 that discharged directly into San Diego Bay and three that discharged into drainage ditches that intermittently flowed to the Pacific Ocean. This investigation evaluates sediment quality in areas adjacent to nine of these outfalls, identified as outfalls 1 through 8 and 16 (Figure 1-1). Outfalls 9 through 15 are not included in this investigation as they are the subject of a time-critical removal action.

Outfalls 1 and 2 are on the southern portion of the Station, adjacent to the Building 39 Run-off Catchment Area (IR Site 7) and outfall 16 is next to the Golf Course Garbage Disposal Area (IR Site 5). These outfalls empty into channels that end at the beach, where they typically are blocked by natural sand berms. However, the channels periodically drain to the Pacific Ocean, usually during storm events. Outfalls 3 through 8 are located along the western and northern shorelines of NAS North Island and discharge into San Diego Bay. Outfall 3 is on the western shoreline at Pier Bravo immediately west of the Chemical Waste Disposal Area (IR Site 9);

outfall 4 is northeast from outfall 3 and is adjacent to the Defense Reutilization Marketing Office (IR Site 10); and outfalls 5 through 8 are along the northern shoreline. The shoreline and channel sediments associated with these nine outfalls are collectively referred to as Site 1 in this RI/RFI.

Figure 1-1. Locations of Naval Air Station North Island and the outfalls investigated (designated by number).

1.3.2 Site History

NAS North Island has been used by the U.S. Navy as an air station and maintenance facility since 1917. The Station consists of an airfield and several building complexes that house maintenance and cleaning operations. Large areas (approximately 1.5 sq km) of the western and northern shorelines, including portions of the existing

air field, were built on fill materials dredged from San Diego Bay during 1936 and 1940 (SWDIV 1992). Industrial operations at NAS North Island began in 1920, although significant quantities of aircraft maintenance and repair wastes were not generated until the 1940s. By 1972, an estimated 700,000 gallons of industrial wastes per year were generated by facility operations. Solid and liquid industrial and municipal wastes were disposed at a number of sites on the facility, and liquid wastes also were discharged through the stormwater system into San Diego Bay and the Pacific Ocean (Brown and Caldwell 1983).

The original stormwater system consisted of ten outfalls which were used from 1917 to the early 1930s for discharge of sewage and stormwater from industrial and residential areas of the northern and eastern areas of the facility (Jacobs Engineering 1991). Outfalls 1 through 16 were constructed after fill operations had been completed, and these were used until 1963 to discharge sewage and until 1972 to discharge industrial wastes and stormwater runoff directly to San Diego Bay and the Pacific Ocean. In 1963, NAS North Island was connected to the sewage system of the City of San Diego, and sewage was conveyed to the municipal wastewater treatment plant. In 1972, all industrial waste sources were connected to the industrial waste sewer leading to the industrial waste treatment plant on base (Jacobs Engineering 1991). Descriptions of historical discharges, which began in 1938-45, from outfalls 1, 2, 3 through 8, and 16 are summarized in Table 1-1. Presently, these outfalls discharge only stormwater runoff from NAS North Island.

The bulk chemical characteristics of the historical outfall effluents have not been evaluated. However, constituents of the industrial wastes generated by the Navy included organic solvents, caustics, acids, plating solutions, cyanide wastes, metals, paint and paint removal sludge, lubricating oils and other refined petroleum products. Wastes may have contained some persistent and potentially harmful chemicals (these are discussed further in Section 4.3, Chemicals of Potential Concern). For example, industrial wastes disposed over a 50-year period at the facility contained approximately 70 tons of metals, of which an estimated 80% was discharged from outfalls 5 through 11 into San Diego Bay (Brown and Caldwell 1983; Jacobs Engineering 1991).

Table 1-1. Summary of historical discharges of industrial waste (1938-45 to 1972) from outfalls 1 through 8, and 16.

Outfall Number	Description of Historical Waste Discharges
1, 2	800 gal/day: wastes from washing and stripping of ground support equipment, including caustic soda, paint flakes, and Turco cleaner rinse
3	1,000 gal/day: sanitary sewage and wastes from wash racks, including detergents
4	10,000 gal/day: heavily chlorinated salt water from fire lines
5	360,000 gal/day: wash down from test cell floor, wastes from plating shop, parts degreaser, and aircraft paint and strip complex
6	4,500 gal/day: wash waters and oil and grease wastes from aircraft washing and vehicle maintenance
7	300,000 gal/day: wash waters and wastes from aircraft washing, drone maintenance, chemical cleaning and paint stripping, and food processing
8	1,000 gal/day: steam condensate
16	20,000 gal/day: wash waters from aircraft and vehicle washing

source: Jacobs Engineering 1991

1.3.3 Aerial Photograph Review

Historic aerial photographs of NAS North Island were reviewed to document changes in shoreline position and channel migration. This information was used to position sampling locations to provide relevant data regarding historical discharges. Some of the discharge pathways between outfalls and receiving waters have changed over the years, particularly for the ocean outfalls 1, 2 and 16.

Eighteen aerial photographs dated March 1953 to August 1978 were obtained from the San Diego County Department of Public Works, United States Department of Agriculture (USDA), and Whittier College (Table 1-2). Three of the photographs were shot at oblique angles with the remaining shot vertically. The 1953 to 1972 time span covers the period of highest waste discharge through the Site 1 outfalls.

Table 1-2. Aerial photographs reviewed (March 1953 - August 1978).

Photograph Date	Photograph Description	Flight/ID No.
03/31/53	Site 1, Point Loma, San Diego Bay	AXN-4M-96
03/31/53	Site 1, Point Loma, Pacific Ocean	AXN-4M-97
11/11/53	Site 1 (northern portion)	19500 / T-4-SD-1-1
03/04/58	Site 1, Coronado Island, Pacific Ocean, San Diego Bay	C-23023 / XI-SD-12-82
04/07/64	Site 1 and Point Loma	AXN-1DD-168
04/07/64	Site 1 (northern portion) and Point Loma	AXN-1DD-169
11/10/65	Site 1, Coronado Island, Pacific Ocean, San Diego Bay	2595
12/10/69	Site 1 (northeast portion) and San Diego Bay	57-16
12/10/69	Site 1, Coronado Island and San Diego Bay	57-17
03/06/70	Site 1, Point Loma, Pacific Ocean, San Diego Bay	30067
11/25/73	Site 1 (northern portion) and Point Loma	30-3
11/25/73	Site 1 (northern portion) and Point Loma	30-4
06/17/74	Site 1 (western portion) and Pacific Ocean	29-2
06/17/74	Site 1 (western and northern portions)	29-3
06/17/74	Site 1 (northeastern portion)	29-4
07/31/75	Site 1, Point Loma, Pacific Ocean, San Diego Bay	5939
01/31/77	Site 1 (western portion) and Point Loma	8416
08/17/78	Site 1 (western and northern portions)	210-19C-36 / SDCO

The northern and western bay shorelines of NAS North Island, where outfalls 3-8 are located, remained relatively stable between 1953 and 1978 (Figures 1-2 and 1-3, respectively). The shoreline follows a relatively smooth curve starting at the San Diego Bay entrance and winding northeast. Although there are small “bumps” in the shoreline, these features remained fixed between 1953 and 1978. These features are probably due to sediment accumulation associated with the pier structures, most of which were built prior to 1953.

The southern (ocean) shoreline of NAS North Island, where outfalls 1, 2 and 16 are located, remained relatively stable in shape during the time period investigated, although it appears that the shoreline position advanced (accreted). It was difficult to quantify this accretion because the photographs were not registered to fixed points. Typical season variation in beach width or daily variation due to the tides could not be separated from net beach accretion using these photographs.

Figure 1-2. Aerial photograph of NAS North Island, 1953. Source: Aerial Fotobank, Inc.

The photographs also were reviewed to track changes in the position and completed pathway of the small drainage channels associated with outfalls 1, 2, and 16. The channel associated with outfalls 1 and 2 was not visible in the photographic record until April 1964; then it was visible in December 1969, March 1970, June 1974, July 1975, January 1977, and August 1978. The channel did not migrate between April 1964 and August 1978, although it may have increased in width and length. The apparent variation in width may have been due to the presence of flood flows, and the increase in length may have been due to shoreline accretion as discussed.

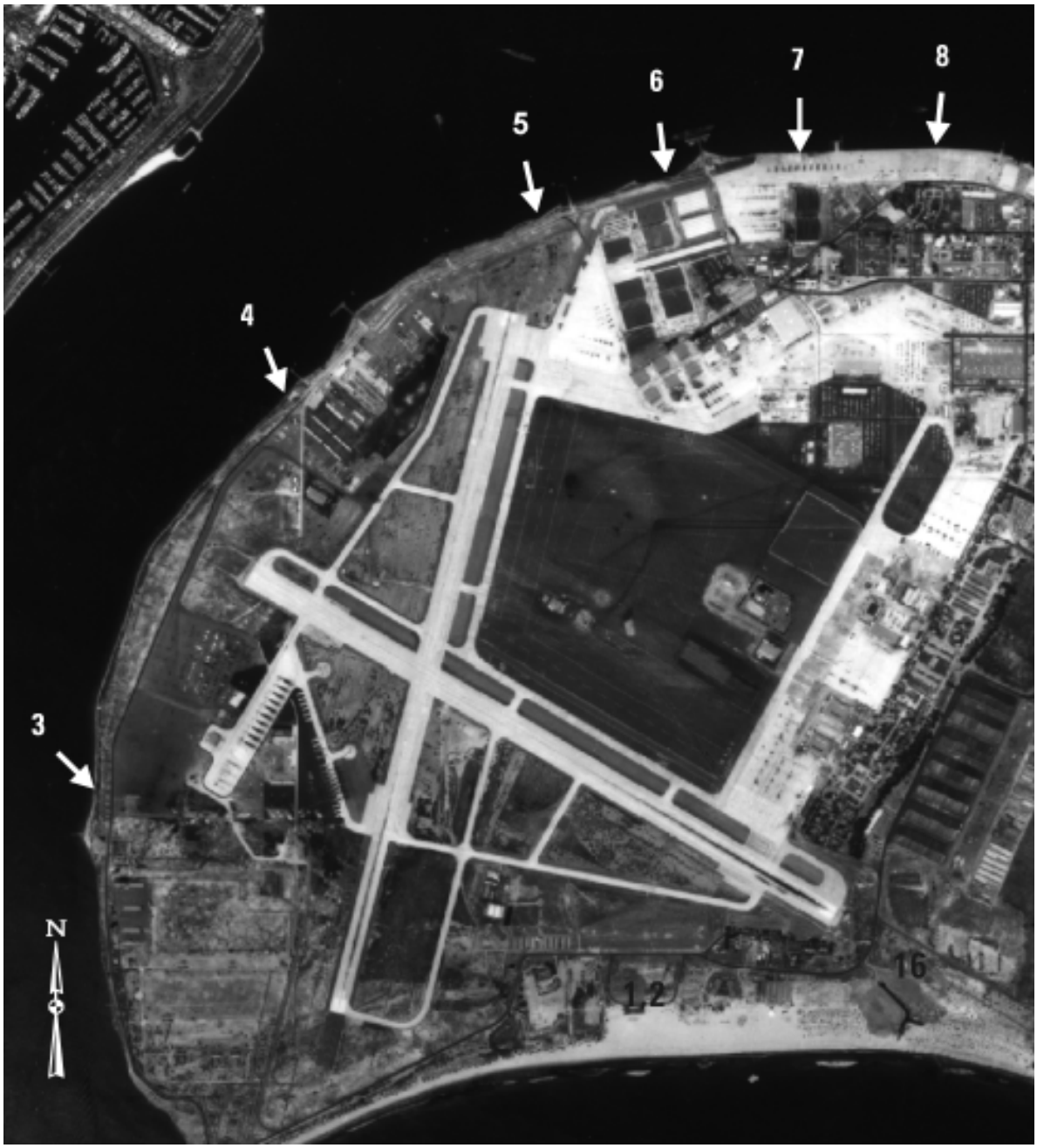


Figure 1-3. Aerial photograph of NAS North Island, 1975. Source: Aerial Fotobank, Inc.

The channel associated with outfall 16 was visible throughout the photographic record. It did not appear as though the channel migrated between 1953 and 1978, although there was a change worth noting. Between 1953 and 1975, the channel consisted of a single line through the beach. In some cases (e.g., storm flows) the channel was connected to the ocean, while at other times it was not; however, there was always a single channel. In July 1975, it appears that sediment was placed on the beach in the location of the outfall 16 drainage channel and that the channel broke into two separate channels. The main channel followed the prior alignment; however, another smaller channel ran parallel to the shoreline for approximately 50 m and was also present in January 1977 (Figure 1-4). There was no flow observed across the beach and into the ocean in either of these photographs. As discussed below in Section 2.2, only the main channel was sampled in this study.

Figure 1-4. Aerial photograph of outfall 16 vicinity, NAS North Island, 1977. Source: Aerial Fotobank, Inc.

1.3.4 Demography and Land Use

In general, NAS North Island land uses include naval aircraft and ship support installations, industrial activities, commercial facilities, and recreational areas. A summary of the demographics and land use for the entire NAS North Island area is presented in Section 6.2, where land use is evaluated in relationship to human health. More detailed information regarding land use and demography can be found in the NAS North Island Master Plan (ASLA and Associates 1991).

Marine habitat in the bay and recreational beaches on the ocean are potential sensitive areas for contaminants discharged at Site 1. Marine habitat includes open water, subtidal and intertidal soft-bottom and hard-bottom (e.g., rocky) areas, and supratidal (e.g., beach dunes) areas. Navy personnel occasionally utilize the ocean shoreline, including Site 1 areas associated with outfalls 1, 2 and 16, for recreational beach activities, such as sun bathing and swimming. Potential ecological and human receptors exposed to contaminants in these areas are discussed in Section 6.

1.3.5 Bay Bathymetry and Hydrodynamics

San Diego Bay is a small, crescent-shaped bay, 43 sq km in area at mean lower low water, with an average depth of 6.5 m measured from mean sea level. The bathymetry of the bay is quite varied. A deep shipping channel covers much of the northern part of the bay, while the southern region is very shallow. The channel is the result of ongoing maintenance dredging operations and scour resulting from the high water velocities characteristic of the entrance area. The shallow area was formed by fluvial sedimentation from the rivers that empty into the bay and by historic dredge and fill activities. A contour map of the bathymetry along NAS North Island encompassing Site 1 is provided in Figure 1-5.

The bathymetry along NAS North Island is characterized by a smooth bottom profile extending from the entrance to the east side of North Island, with the southwest portion bounded by the Pacific Ocean at Imperial Beach. There are no historic data available for the bathymetry in the vicinity of the outfalls; however, water depth measurements were taken during the field effort associated with the present study. Water depth measurements at each sampling location (identified in Section 2) were corrected for tidal variations using predicted tides within San Diego Harbor (Tidelines 1996). The corrected water depths as well as the habitat (i.e., subtidal or intertidal) of locations sampled at each outfall are shown in Table 1-3.

Generally, San Diego Bay can be treated as a vertically well-mixed estuary. Freshwater inflow to the bay is low and occurs only during infrequent winter storms. Inflow enters primarily in south San Diego Bay from the Otay and Sweetwater Rivers, which are regulated by storage reservoirs. Stormwater also enters through Schweizer, Chollas, and Paleta Creeks (Figure 1-1) on the central eastern shore as well as a variety of stormdrains around the bay. Vertical stratification may persist for several days after a strong storm in the south bay but is detectable from a day to only a few hours elsewhere, particularly in the more energetic mouth of the bay.

Currents in San Diego Bay are produced predominately by semidiurnal tides. Maximum tidal range in spring tides exceeds 2 m, while mean tidal range is about 0.85 m. Typical tidal current speeds range from 30 to 50 cm/s near the inlet (though they may approach 100 cm/s during spring tides) and from 10 to 20 cm/s in the southern region of the bay. Typical westerly afternoon winds are about 5 m/s (10 knots) and result in weak wind-driven circulation only in the south bay.

Currents around the bay perimeter of North Island tend to follow the main navigational channel. At the entrance to the bay off Ballast Point, both incoming and outgoing tides produce fairly uniform velocities across the width of the bay (Wang et al. 1998). Further into the bay, currents of 50 cm/s are commonly measured in the channel adjacent to Shelter Island, Harbor Island, and at the bay constriction beneath the Coronado Bridge. Currents adjacent to North Island monotonically decrease in speed from the vicinity of outfall 3 to outfall 8. In effect, the currents hug the outside, northern and eastern shores of the bay on incoming and outgoing tides. Sediment grain size in the vicinity of these outfalls reflects this current regime (Section 3). Sediments near the energetic mouth tend to be fine to medium sand; sediments off outfalls 6 through 8 generally tend to be dominated by silts and clays. In addition to currents, wave action from ocean swell and ship wakes tend to be more pronounced near the mouth of the bay and tend to erode finer sediments along the shore. This effect is seen at all bay outfalls resulting in coarser, nearshore sediments at the outfall discharge than those in deeper water away from the outfalls.

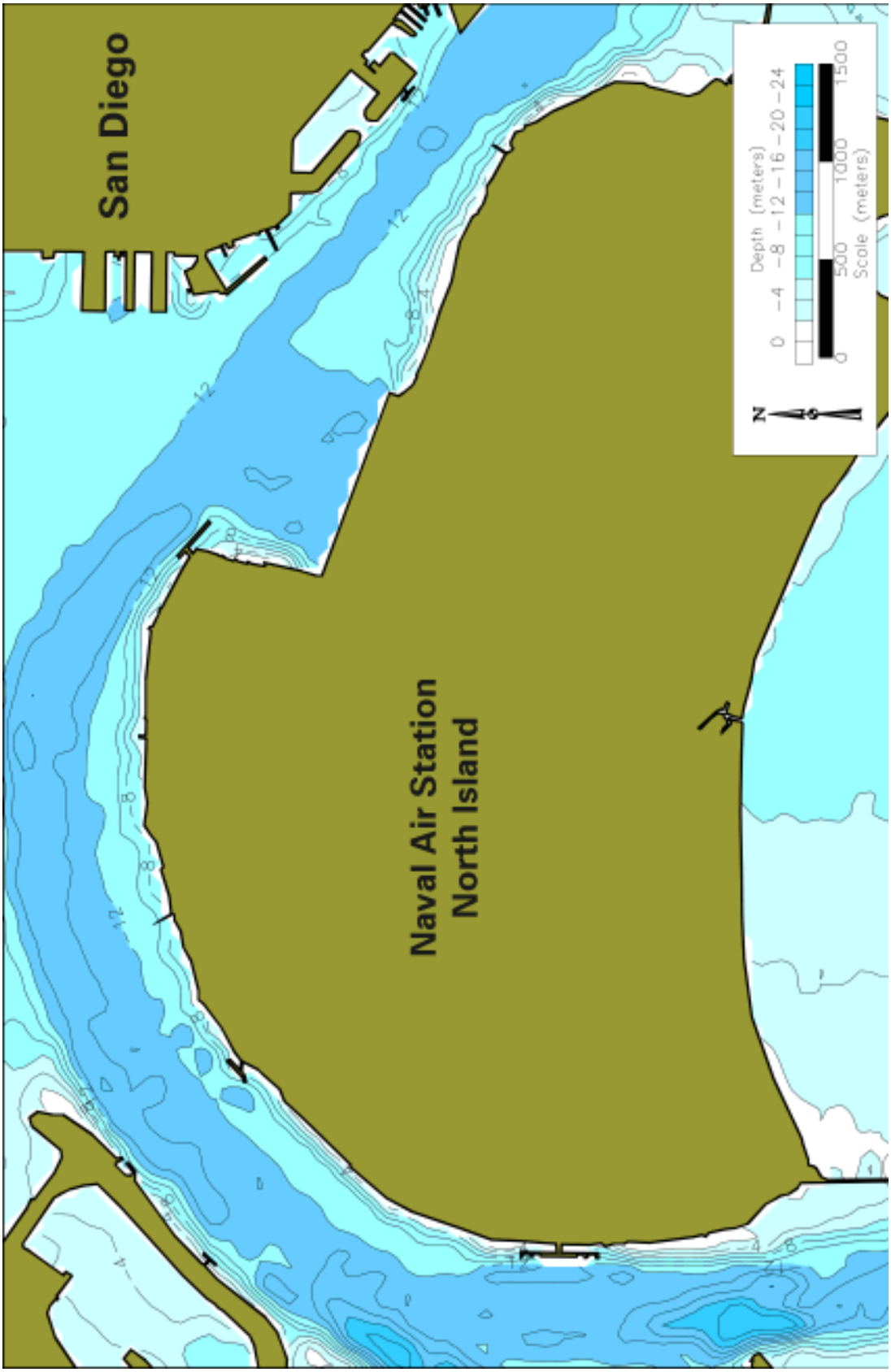


Figure 1-5. Bathymetry along NAS North Island.

Table 1-3. Water depth and habitat type of locations sampled at in-bay outfalls 3-8.

Outfall	Distance from Outfall (m)	Sampling Date	Water Depth (m, MLLW)	Habitat
3	3	6/26/96	-3.5	Subtidal
	10	6/26/96	-4.0	Subtidal
	30	6/26/96	-6.6	Subtidal
	100	6/26/96	-10.5	Subtidal
	150	6/26/96	-12.7	Subtidal
4	3	7/02/96	0.0	Intertidal
	10	6/26/96	0.0	Intertidal
	30	6/26/96	0.0	Intertidal
	100	6/26/96	-10.4	Subtidal
	150	6/26/96	-11.5	Subtidal
5	3	7/02/96	-0.2	Subtidal
	10	7/02/96	-0.3	Subtidal
	30	7/02/96	-0.5	Subtidal
	100	6/26/96	-10.1	Subtidal
	150	6/27/96	-10.8	Subtidal
6	3	7/02/96	0.0	Intertidal
	10	7/02/96	-0.6	Subtidal
	30	6/26/96	-9.1	Subtidal
	100	6/26/96	-9.6	Subtidal
	150	6/26/96	-10.8	Subtidal
7	3	7/02/96	-0.4	Subtidal
	10	6/26/96	-0.2	Subtidal
	30	6/26/96	-9.0	Subtidal
	100	6/26/96	-6.8	Subtidal
	150	6/28/96	NA	Subtidal
8	3	7/02/96	0.0	Subtidal
	10	6/26/96	-0.6	Subtidal
	30	6/26/96	-3.6	Subtidal
	100	6/26/96	-8.3	Subtidal
	150	6/26/96	-10.9	Subtidal

NA = Not available

1.4 Evolution of the Study Design

As discussed in Section 1.1, the Site 1 RI/RFI evolved from 13 years of environmental management required under CERCLA and RCRA regulatory framework. Several key documents developed during that time, along with input from the regulatory agencies identified in Section 1.1, contributed to the investigative approach used for Site 1. These documents and the key meeting in which regulatory input was received are described.

- *Initial Assessment Study of Naval Air Station, North Island, California. (Brown and Caldwell 1983).* This study first identified potential contaminant disposals areas at NAS North Island, setting the stage for future environmental assessments and establishing the IR program. Geographical areas for IR Sites 1 through 12 were identified in this report along with contaminant disposal types, practices, and potential impacts.
- *NAS North Island San Diego California Sites 1, 5, 6, 9, and 10 Remedial Investigation/Feasibility Study Work Plan. (Jacobs Engineering 1991a).* This was the initial work plan developed for Site 1. This plan was agency approved; however, the plan presented only a conceptual approach to biological testing and over-emphasized sediment chemistry sampling.
- (*NAS North Island San Diego California) Sites 1, 5, 6, 9, and 10 Remedial Investigation/Feasibility Study Sampling and Analysis Plan. (Jacobs Engineering 1991b).* This was the sampling and analysis plan developed in support of the initial work plan cited above. This plan also was agency approved; however, it was superceded by the development of a revised workplan by Bechtel (see Bechtel 1995).
- *Bioassay and Environmental Sampling and Analysis Work Plan - Shoreline Sediments, Naval Air Station, North Island, California (BESAP) (Bechtel 1995).* This combined study plan, quality assurance project plan, and health and safety plan replaced the corresponding documents previously prepared by Jacobs Engineering. These plans also were agency approved, with some open issues. The document was slightly modified by SPAWAR, when tasked to implement the BESAP and conduct the RI/RFI.
- *Meeting Minutes. SPAWAR, San Diego. April 1, 1996. NAS North Island IR Site 1 Remedial Investigation Kick-off Meeting (SPAWAR 1996).* The RI/RFI kick-off meeting was held on April 1, 1996 to resolve open issues in the BESAP. These meeting minutes summarized the proposed changes to the study design (numbers and locations of sampling stations), selection of reference stations, assumptions in statistical inference tests, and the decision matrix. These changes received regulatory approval by California DTSC in a letter dated July 10, 1996.
- *Work Plan and Quality Assurance Project Plan for NAS North Island - Site 1 - Shoreline Sediments (MEC 1996).* This combined study plan and quality assurance project plan was produced to augment the corresponding plans previously produced by Bechtel. This document also details program changes presented by SPAWAR and accepted by the attending regulatory representatives at the April 1, 1996 kick-off meeting. This document includes detailed standard operating procedures and analytical method descriptions that were not provided in the BESAP.

1.5 Previous Investigations

Previous investigations of sediment quality have been conducted at only five of the nine Site 1 outfalls investigated in this study, and these were limited in scope and duration (Table 1-4). Dredging has not occurred in the immediate vicinity of these outfalls, except near outfall 3, where dredging between the main shipping channel and Pier Bravo occurred in 1964 and 1966, and adjacent to outfall 4, where land-based excavation occurred in 1995 to remove two slag deposits. Thus, historical dredging has not substantially altered sediment quality conditions near the outfalls. Brief descriptions of previous investigations are summarized in Table 1-4 and presented below.

Table 1-4. Summary of previous investigations of sediment quality at NAS North Island Site 1 outfalls.

Outfall	Date	Purpose of Investigation	Reference
1, 2	1993	Sediment analyses for metals, volatile and semivolatile organic compounds, and polychlorinated biphenyls	SWDIV 1993; Jacobs Engineering 1993
3	1995	Sediment core analyses for volatile organic compounds in pore water	Jacobs Engineering 1996
	1994	Sediment testing (bulk chemical and bioassay) for dredged material disposal	CAS 1994
	1988	Sediment collections in the vicinity of outfall 3 and the Chemical Waste Disposal Area	HLA 1989 (cited in SWDIV 1992)
	1985	Verification study of sediments near outfall pipe. Shallow sediment samples analyzed for metals, total organic halides, and total organic carbon.	HLA 1985a
4	1995	Sediment radioactivity testing at the slag deposit site, pre- and post-removal action.	AGT 1996
	1989	Characterization study of sediments near outfall pipe. Shallow vibracore and one deep vibracore samples analyzed for metals, VOCs, SVOCs, organochlorine pesticides, and petroleum hydrocarbons.	HLA 1989
	1985	Verification study of sediments near outfall pipe. Shallow sediment samples analyzed for metals, total organic halides, and total organic carbon.	HLA 1985a
5	1989-90	Two-phase sediment sampling in the vicinity of outfall 5 for metals, polychlorinated biphenyls, pesticides, semivolatile organic compounds, petroleum hydrocarbons, radioactivity, and total organic carbon.	SANDAG 1991
	1988	Remedial investigation of sediments near outfall 5 for metals, volatile and semivolatile organic compounds, chlorinated pesticides, and polychlorinated biphenyls.	HLA 1989

1.5.1 Outfalls 1 and 2

One surface sediment sample was collected near ocean outfalls 1 and 2 and analyzed for metals, volatile and semivolatile organic compounds (VOCs and SVOCs), and polychlorinated biphenyls (PCBs) as part of the Extended Site Investigation/RCRA Facility Investigation (ESI/RFI) for NAS North Island Sites 3, 4, 7, and 12. Sediment from this location contained elevated concentrations of some metals (i.e., cadmium, copper, lead, and zinc) and detectable concentrations of some polycyclic aromatic hydrocarbons (PAHs), (i.e., fluoranthene, phenanthrene, and pyrene, [Jacobs Engineering 1993]).

1.5.2 Outfall 3

Harding Lawson Associates (HLA 1985a) collected 11 shallow water samples (< 2.0 ft depth) from sediments adjacent to the outfall pipe. Two samples were collected from four locations (0-1.5 ft range, 1.5-2 ft range) and one sample was collected from three locations (< 1.0 ft depth). Each sample was analyzed for metals, total organic halides, and total organic carbon. Metal concentrations were generally less than the STLC criteria except for slight exceedances of cadmium and nickel. Total organic halides were detected in several samples (12-67 μg/L).

HLA collected and analyzed sediments in 1988 from the vicinity of outfall 3 and soil from the Chemical Waste Disposal Area in 1988 which this outfall drains; results are summarized in SWDIV (1992). Sediments near outfall 3 contained moderate petroleum hydrocarbon concentrations, but phthalates, phenols, pesticides, and PCBs were not detected, and metal concentrations were comparable to background levels. In contrast, soil samples from the Chemical Waste Disposal Area contained elevated concentrations of petroleum hydrocarbons, PCBs, organic halide and VOCs, phenols, phthalates, and some metals (i.e., cadmium, chromium, lead, and nickel).

Sediment quality near outfall 3, adjacent to Pier Bravo, was evaluated by Columbia Aquatic Sciences in 1994 to determine whether materials considered for maintenance dredging were suitable for disposal at an ocean dredged material disposal site. Sediment cores were collected at seven locations, and the composited subsamples from each core were analyzed for bulk chemical contaminants and toxicity. The majority of sediments were considered uncontaminated and non-toxic and therefore suitable for disposal at an ocean dredged material disposal site (LA-5). The exception was sediment from the side of the pier facing the bay in which significant bioaccumulation of metals and PAHs occurred in laboratory exposures of test organisms.

Jacobs Engineering (1996) also collected sediment cores at six locations offshore from the Chemical Waste Disposal Area and analyzed whole sediments for suites of VOCs associated with sediment pore waters. The compounds cis-1,2-dichloroethene, 1,1-dichloroethene, and vinyl chloride were detected in several samples. Potential VOC contamination from IR Site 9 groundwater to sediments adjacent to Pier Bravo is currently being investigated by SPAWAR.

1.5.3 Outfall 4

HLA (1985a) collected 16 shallow samples (< 2.0 ft depth) from sediments adjacent to the outfall pipe. Two samples were collected from each of the eight locations (0-1.5 ft range, 1.5-2 ft range) and analyzed for metals, total organic halides, total organic carbon, and PCBs. No PCBs were detected, but selected metals exceeded Soluble Threshold Limiting Concentration (STLC) criterion and organic halides were detected. No metals exceeded their respective Total Threshold Limiting Concentration (TTLC) criterion.

HLA (1989) collected 15 shallow (up to 5 ft depth) vibracore and 1 deep (9 ft depth) vibracore samples from sediments located just offshore of outfall 4 as part of a characterization study using an approved work plan dated May 6, 1985 (HLA 1985b). Samples were analyzed for metals, VOCs, SVOCs, organochlorine pesticides, and petroleum hydrocarbons. No aromatic volatile or halide organic compounds and no PCBs were detected. Selected metals only slightly exceeded STLC criteria; no metals exceeded their respective TTLC criteria. SVOCs were detected at low concentrations (< 0.26 μg/L).

Recent investigations of radioactivity levels in slag deposit sediments were conducted near outfall 4 (ATG 1996). The purpose of these investigations was to evaluate the magnitude and spatial extent of excess radioactivity from radium-226, in the vicinity of the slag deposit associated with the Defense Reutilization and Marketing Office operations (IR Site 10). A radiological survey conducted in July 1995 indicated low levels of contamination that generally were localized within two, 20-ft by 40-ft outcrops along the shoreline. The slag piles were removed subsequently in July-August, 1995. A confirmation study conducted in August 1995 indicated that radium levels were within background, but concentrations of several metals (e.g., cadmium, copper, lead, nickel, and

zinc) exceeded nominal background levels. The shoreline slag area of IR Site 10 is the subject of an extended RI/RFI currently being conducted by Oakridge National Laboratory and SPAWAR.

1.5.4 Outfall 5

HLA (1989) collected sediment cores from 17 locations near outfall 5 and analyzed the samples for grain size, metals, VOCs, SVOCs, chlorinated pesticides, and PCBs as part of a remedial investigation. No VOCs, pesticides, or PCBs were detected; however, measurable quantities of total petroleum hydrocarbons, selected PAHs, and two phthalate esters (potentially from laboratory contamination) were present in the sediments. Additionally, concentrations of some metals (e.g., cadmium, copper, and lead) were elevated in one or more samples.

As part of the two-phase San Diego Bay Cleanup Project conducted by San Diego Association of Governments (SANDAG 1991), surface and subsurface sediment samples were collected during 1989 and 1990 at 78 sites within San Diego Bay, including one location near outfall 5. The purpose of the study was to characterize water and sediment quality within the bay. Sediments collected in 1989 were analyzed for metals only; whereas, sediments collected in 1990 were analyzed for metals, as well as PCBs, pesticides, SVOCs, petroleum hydrocarbons, radioactivity (gross alpha and beta emission), and total organic carbon (TOC). Because the study was intended to characterize sediment quality within the entire Bay, specific conclusions regarding sediment quality near outfall 5 were not addressed.

1.5.5 General Studies

In addition to these sediment outfall-specific studies, other general monitoring programs have provided information on environmental quality and chemical contamination in the vicinity of the NAS North Island outfalls. The State Water Resources Control Board (SWRCB), in conjunction with National Oceanic and Atmospheric Administration (NOAA), California Department of Fish and Game (CDFG), and U.S. Environmental Protection Agency (USEPA), conducted a study of sediment quality within San Diego Bay as part of the state-wide Bay Protection and Toxic Cleanup Program (BPTCP 1996). The BPTCP study in San Diego Bay included collection and analysis of sediments for grain size, TOC, metals, PAHs, pesticides, and PCBs, as well as determinations of pore water and sediment toxicity. These data were collected in 1993-94 and are available in draft form. The BPTCP study collected samples from 102 sites within the bay, and although none of the sampling locations corresponded to the NAS North Island outfall areas, several of the sites were used as reference areas for the present study as discussed in Section 2.

The study design and abbreviated methods of data collection and analyses to support project objectives are described in this section. Overall objectives of the RI/RFI listed in Section 1.1 are restated as specific objectives of the sediment investigation as follows: 1) to determine whether Site 1 shoreline sediments near select outfalls are more contaminated than designated reference sediments located within San Diego Bay; 2) to determine whether Site 1 sediments are more toxic than reference sediments; 3) to determine whether clams exposed to Site 1 sediments bioaccumulate contaminants to higher concentrations than those exposed to reference sediments; and, from these data 4) to determine whether Site 1 sediments pose unacceptable risk to ecological or human health. Detailed descriptions of field and laboratory methods, data analysis, and quality control methods are included in Volume III.1 and III.2. Quality control results are included in Volume III.3 and III.4.

2.1 Study Design

The RI/RFI consists of physical, chemical, and biological measurements of surface and/or subsurface sediments sampled near nine outfall pipes located around the perimeter of NAS North Island (Site 1) and at 10 distinct reference locations within San Diego Bay (Figures 2-1 and 2-2). Six of the outfalls are located around the in-bay perimeter and three outfalls are located in two freshwater channels on the ocean side of North Island (outfalls 1, 2 and 16). The west-most ocean channel drains outfalls 1 and 2, which discharge to the same channel. These outfalls were treated as a single site, designated as outfall 1,2. Surface sediments were collected at in-bay outfalls 3-8 and reference locations only. Subsurface sediments were collected at the two ocean channels and at two locations for each of the six in-bay outfalls. Study design and rationale for collection of surface and subsurface sediments follow.

2.1.1 In-bay Surface Sediments

Sampling locations were based on the premise that contaminants adsorbed to particles were preferentially deposited over time close to the end of each outfall pipe. Recent hydrodynamic data (Cheng et al. 1997) along with total organic carbon (TOC) and grain size measurements made by the U.S. Navy were used to identify likely areas of contaminant deposition for in-bay outfalls. These data indicate that deposition should have occurred on the northeast perimeter of North Island, localized near outfalls 7 and 8. This region has slower tidal currents, finer-grained sediment (> 40% silt and clays), and higher TOC (> 1.5%) as compared to the other in-bay outfall regions. This is affirmed by contoured maps displaying grain size and TOC distributions based on 20 previous measurements around North Island as shown in Figures 2-3 and 2-4, respectively. Grain size and TOC influence contaminant concentrations in bed sediments by their adsorptive capacities, usually resulting in higher concentrations of chemicals occurring in finer-grained materials with higher TOC (Hassett et al. 1980; Karickhoff 1984; Förstner 1990). Furthermore, assuming that soluble chemicals sorb to bed sediments following first order kinetics (i.e., concentrations approach equilibrium asymptotically) and that equilibrium partitioning is reached rapidly (e.g., DiToro et al. 1992), sediments closest to the contaminant source should carry the highest chemical burdens, and concentration gradients should be steepest near the outfalls. The Site 1 study was based on these premises and to determine whether historical discharge from North Island Naval Air Station storm drains produced sediment contaminant concentrations that posed risk to human health or the environment. The

study also was designed so that Site 1 sediments could be compared with background conditions in San Diego Bay that resulted from bay-wide use, rather than pristine conditions perhaps found outside of the bay. The rationale for this comparison was that potential remediation efforts required would not be carried out to the extent of restoring Site 1 sediments beyond background conditions. The field sampling design follows.

Figure 2-1. Locations of the 9 Site 1 outfalls around the perimeter of NAS North Island and the 10 reference stations in San Diego Bay.

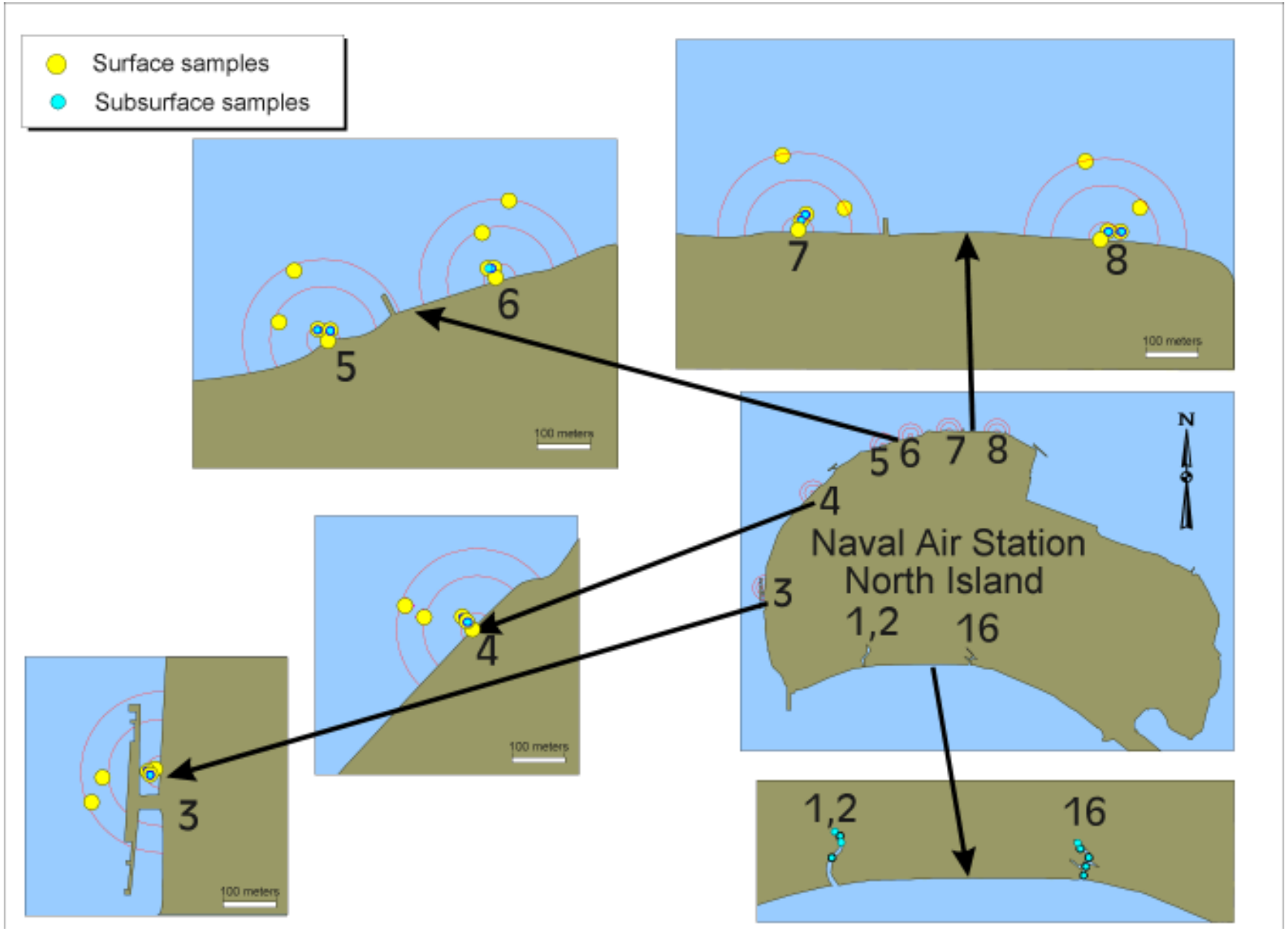


Figure 2-2. NAS North Island sampling stations identified by outfall. Semicircle sampling arcs are shown for 30, 100, and 150 meter stations.

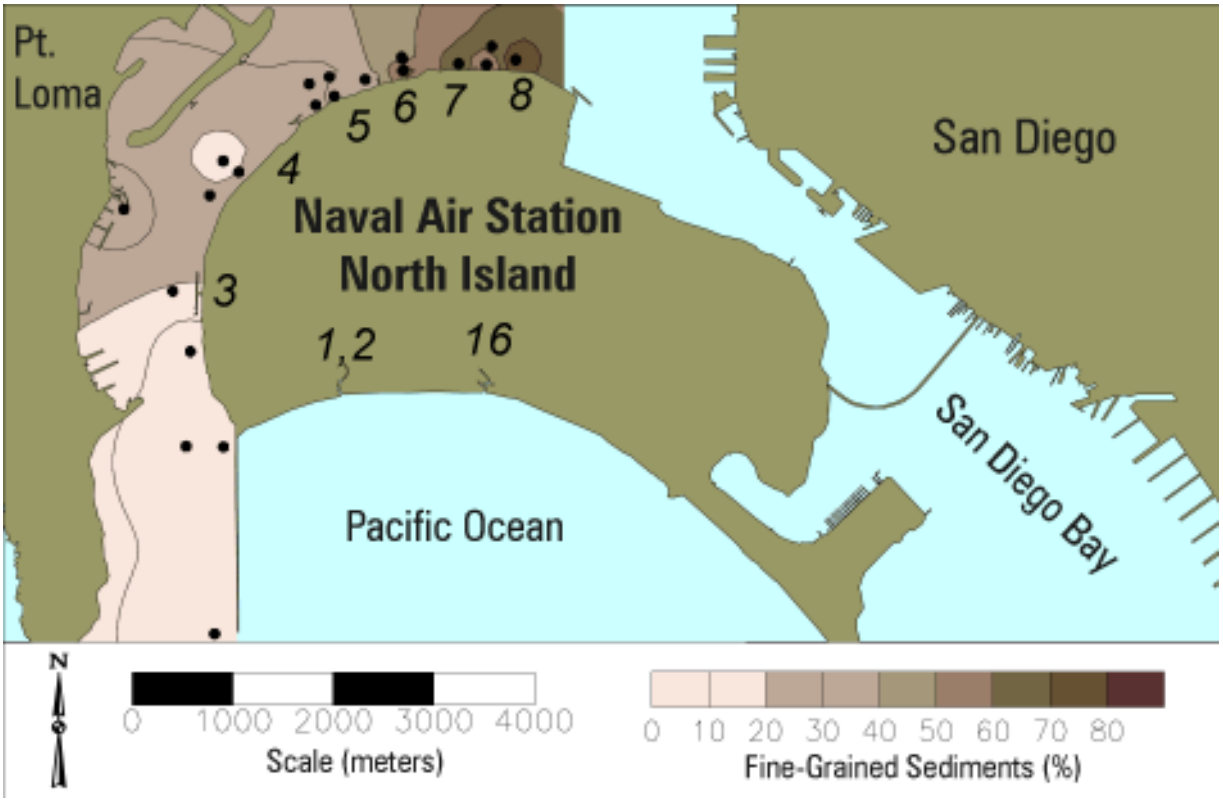


Figure 2-3. Fine-grained sediment distributions around NAS North Island.

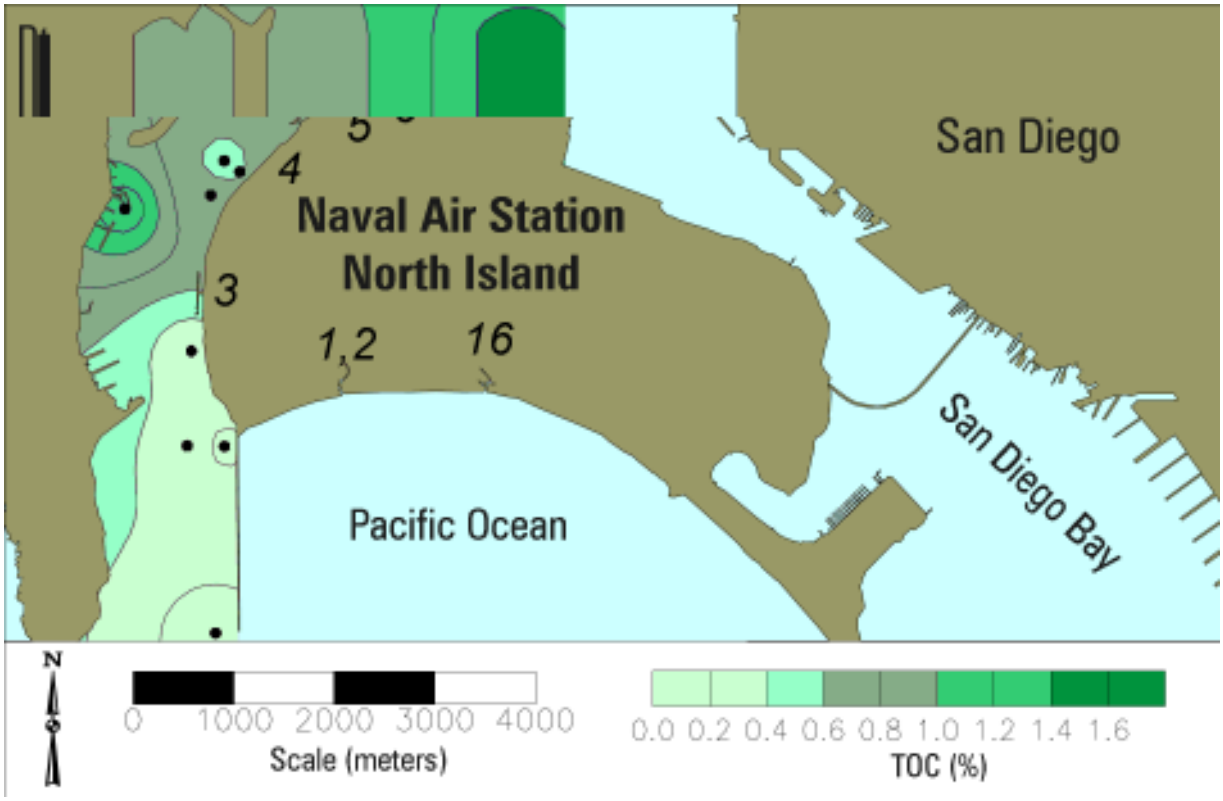


Figure 2-4. Total organic carbon distribution around NAS North Island.

Five surface sediment stations were sampled for each of the six in-bay Site 1 outfall areas (outfalls 3-8). Station identifications, distance from outfalls, and latitude/longitudes are shown in Table 2-1. Stations were located randomly on semi-circle arcs, at 3, 10, 30, 100, and 150 m distances from the end of each outfall (Figure 2-2). Random locations on these arcs, hence random directions from the outfall, were selected for two reasons. First, since water depth and distance from the outfall are decoupled, these two factors can be tested independently for separate effects on sediment contamination, toxicity, or bioaccumulation. Second, since stormwater discharge from the outfalls most likely occurred randomly through the tidal cycle (tidal currents presumably advecting contaminated discharge), preferential direction of deposition from the outfall did not occur, and random selection of station location is appropriate. Surface sediments, defined as the top 5 cm, from each of these stations were sampled for several physical measurements, bulk sediment chemical concentrations, and to conduct four solid-phase bioassays and one pore water bioassay. Additionally, 10 reference stations were sampled for the same parameters. Ten reference station locations were selected from 102 sites used in the 1993-1994 San Diego Bay Protection and Toxic Cleanup Program (BPTCP) study jointly administered by California State Water Resources Control Board (SWRCB), NOAA, and USEPA (BPTCP 1996). A discussion of criteria used to select these reference stations for use in the current study follows, since critical decisions regarding Site 1 sediments rely on comparison with these sediments.

The 102 stations in San Diego Bay sampled under the BPTCP were screened using sediment bioassay and chemistry data obtained in that study. Criteria were chosen to select the least impacted areas in San Diego Bay, based on sediment toxicity and chemical concentrations. The criteria were: (1) survival of amphipods (*Rhepoxynius abronius*) and polychaetes (*Neanthes arenaceodentata*) equal to or greater than 75%; and (2) sediment chemical concentrations that were less than NOAA "effects range median" (ER-M) (Long et al. 1995). These criteria produced 15 possible locations for use as reference stations in the investigation (Figure 2-5). Ten reference station locations were selected from these 15 to approximate anticipated ranges in grain size and TOC of the outfall sediments. Grain size and TOC results for outfall and reference stations sampled in this investigation are discussed in Section 3. Corresponding BPTCP station information for each Site 1 reference station is shown in Table 2-2.

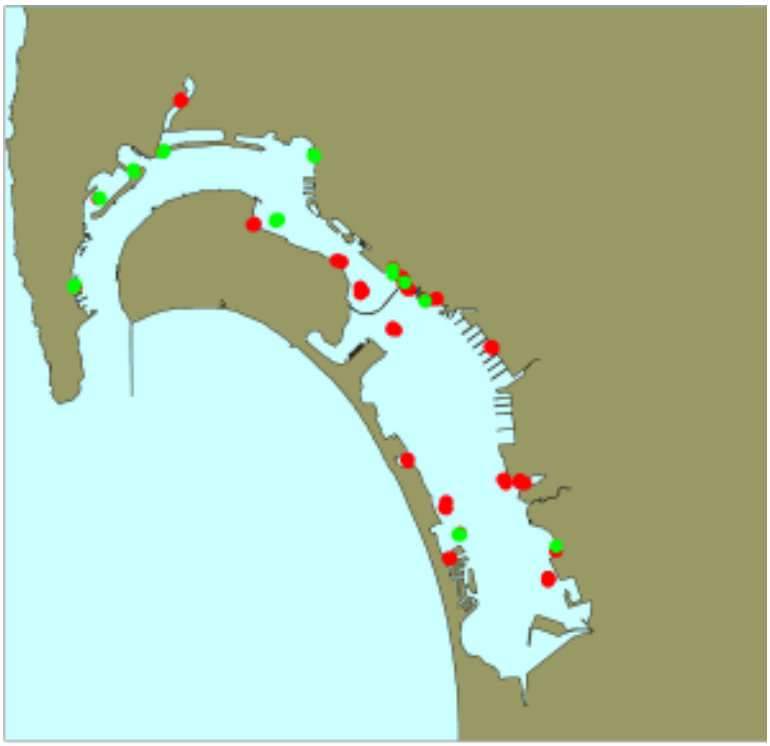
2.1.2 Subsurface Sediments

Subsurface sediment core sections were analyzed to determine the vertical extent of surface contamination. Subsurface sediment cores were collected at the same location as the surface samples collected at 10 and 30 m distance from each of the six in-bay outfalls. These subsurface sediments were sampled at 1-ft intervals to a nominal depth of 6 ft. Actual depths obtained for each core are discussed in Section 4.2. Four and five subsurface sediment cores each were sampled in the two ocean channels receiving run-off from outfalls 1,2 and 16, respectively. These cores were sampled at 1-ft-depth intervals from 0 to 6 ft and at 2-ft intervals from 6 to 10 ft. Cores were taken to 10 ft to collect sediments above and below the water table, residing at approximately 8 ft. Locations of the cores were approximately mid-channel at distances ranging from 10-30 m apart. Core locations are shown in Table 2-1 and Figure 2-2. Bulk chemistry only was measured on all subsurface cores.

Table 2-1. Station identification, distance from outfall, and location.

Outfall	Station	Distance from Outfall (m)	Latitude*	Longitude*
3	3-1	3	32°41’12.78"	117°13’10.40"
	3-2	10	32°41’12.73"	117°13’10.55"
	3-3	30	32°41’12.73"	117°13’10.57"
	3-4	100	32°41’12.62"	117°13’11.53"
	3-5	150	32°41’12.17"	117°13’11.77"
4	4-1	3	32°42’7.30"	117°13’3.92"
	4-2	10	32°42’7.37"	117°13’3.98"
	4-3	30	32°42’7.45"	117°13’4.10"
	4-4	100	32°42’7.45"	117°13’4.93"
	4-5	150	32°42’7.67"	117°13’5.30"
5	5-1	3	32°42’12.88"	117°12’10.70"
	5-2	10	32°42’13.07"	117°12’10.63"
	5-3	30	32°42’13.08"	117°12’10.92"
	5-4	100	32°42’13.17"	117°12’11.65"
	5-5	150	32°42’14.10"	117°12’11.45"
6	6-1	3	32°42’14.12"	117°12’7.07"
	6-2	10	32°42’14.17"	117°12’7.12"
	6-3	30	32°42’14.35"	117°12’7.30"
	6-4	100	32°42’14.78"	117°12’7.45"
	6-5	150	32°42’15.47"	117°12’6.90"
7	7-1	3	32°42’14.73"	117°12’1.67"
	7-2	10	32°42’14.90"	117°12'1.60"
	7-3	30	32°42’15.10"	117°12’1.50"
	7-4	100	32°42’15.08"	117°12’0.67"
	7-5	150	32°42’16.07"	117°12’2.05"
8	8-1	3	32°42’14.58"	117°11 11.80"
	8-2	10	32°42’14.75"	117°11 11.62"
	8-3	30	32°42’14.75"	117°11 11.33"
	8-4	100	32°42’15.18"	117°11 10.95"
	8-5	150	32°42’15.97"	117°11 12.02"
1,2	1-1	3	32°41’6.22"	117°12’12.83"
	1-2	10	32°41’6.02"	117°12’12.55"
	1-3	30	32°41’5.68"	117°12’12.47"
	1-4	60	32°41’4.93"	117°12’13.00"
16	16-1	3	32°41’5.78"	117°11’15.47"
	16-2	10	32°41’5.50"	117°11’15.28"
	16-3	30	32°41’5.07"	117°11’14.73"
	16-4	50	32°41’4.57"	117°11’14.63"
	16-5	70	32°41’4.20"	117°11’14.72"
Reference	R-1	NA	32°37’15.13"	117°06’8.87"
	R-2	NA	32°41’8.77"	117°14’5.33"
	R-3	NA	32°41’8.52"	117°14’5.65"
	R-4	NA	32°38’1.72"	117°08’2.03"
	R-5	NA	32°38’2.35"	117°08’2.20"
	R-6	NA	32°43’6.17"	117°10’7.77"
	R-7	NA	32°43’6.40"	117°10’7.45"
	R-8	NA	32°43’6.03"	117°10’7.37"
	R-14	NA	32°43’6.95"	117°12’14.72"
	R-15	NA	32°43’7.10"	117°12’14.68"

NA = Not applicable
* NAD 83 datum in degrees, minutes, decimal seconds



Of the 102 BPTCP stations in San Diego Bay where amphipod and polychaete bioassays were perfomed, chemistry measurements were made at 78. Of these 78 stations, 21 had survival in both bioassays greater than 75%, and are shown in green; 57 had less than 75% survival for at least one bioassay and are shown in red.

Of the 21 BPTCP stations with at least 75% survival, 15 had contaminant concentrations below the ERM; numbered and shown in green. Contaminant concentrations above ERM levels were found at 6; shown in red.
In this study, stations 1-8, 14 and 15 were selected as reference stations.

Figure 2-5. Selection and location of reference stations.

Table 2-2. Station identifications for BPTCP stations used as Site 1 reference stations.

Site 1 Station ID.	BPTCP Station ID.	BPTCP Station Name
R-1	90036	Stormdrain EA (ROHR CH. Rep 2)
R-2	90029	NSB-R1-Rep 1
R-3	90029	NSB-R1-Rep 2
R-4	93159	South Bay GG3 (X1)-Rep 2
R-5	93159	South Bay GG3 (X1)-Rep 3
R-6	90037	Stormdrain EM (Grape St.)-Rep 1
R-7	90037	Stormdrain EM (Grape St.)-Rep 2
R-8	90037	Stormdrain EM (Grape St.)-Rep 3
R-14	90104	West Basin entrance (71C)-Rep 2
R-15	90104	West Basin entrance (71C)-Rep 3

see BPTCP 1996

2.2 FIELD METHODS

Five surface samples were collected at each of the six in-bay outfalls (3-8) and at the 10 reference stations on June 26-28; July 2 and 9; and September 24, 26, and 27, 1996 for a total of 40 surface sediment samples. Sediment samples collected on July 9 and in September were for reanalyses due to testing problems. Sediment was collected on July 9 to re-run two pore water bivalve tests due to insufficient pore water volumes collected during the initial sampling effort; surface samples collected in September were used to re-run the 28-day clam bioaccumulation test for all 40 stations due to low control survival in the initial tests. Quality control issues associated with these reanalyses are addressed in Volume III.3.

Subtidal surface sediments were collected using the U.S. Navy research vessel *ECOS*; intertidal surface sediments were collected using hand cores. All stations were located using a Trimble RL 4000 II Differential Global Positioning System (DGPS) navigation unit, accurate to ± 1 m. Surface sediments, defined as the upper 5 cm, were collected using either a Teflon-coated Van Veen sampler, or a hand-corer at stations that were either too shallow to sample by boat or impenetrable due to heavy eel grass cover.

Subsurface stations were sampled from each of the six in-bay outfalls (3-8) and in the two freshwater channels (outfalls 1,2 and 16) on the ocean side of North Island. In-bay subsurface stations were sampled on June 26 and 28, 1996 at one ft intervals from 1-6 ft or until the point of refusal. Channel stations were sampled on July 16-18, 1996. All in-bay subsurface stations were located using a Trimble Nav Tract DGPS navigation unit, accurate to ± 1 m. Stations corresponded to surface stations located at 10 and 30 m from each outfall, as discussed in Section 2.1.2. One subsurface core was collected at each station.

Outfall stations in the two channels were located using a Magellan Nav 5000D DGPS navigation unit, accurate to ± 1 m. Five subsurface cores were collected in the channel of outfall 16, located from 3-70 m from the outfall terminus. A total of four cores were taken in the channel draining outfall 1,2, three in the northern portion, and one in the southern portion, where the channel emerges after passing under South Moffett Road. Core locations ranged from 3-60 m from the ends of the outfall pipes. Sediments were sampled at 1 ft intervals from 0-6 ft and at 2 ft intervals from 6-10 ft. In-bay subsurface cores were collected using a butyrate-lined vibracore sampler; ocean channel subsurface cores were collected using a butyrate-lined stratocore sampler. Detailed descriptions of surface and subsurface field methods, including sediment collection, handling, and transport, and equipment decontamination procedures are included in Volume III.1.

2.3 Analytical Program

Summaries of analytical methods for physical, chemical, and bioassay tests are presented in the following subsections. Rationale for selection of chemical analytes and ecological relevance of bioassay test organisms used also are included. Detailed descriptions of analytical and quality control methods are included in Volume III.1-2. Laboratory standard operating procedures (SOPs) are included in the Site 1 Work Plan and Quality Assurance Project Plan (QAPP) (MEC 1996) for all physical tests and bioassays and select field and analytical chemistry methods. All SOPs used in support of this RI/RFI are available for limited distribution.

2.3.1 Physical Tests

Physical tests consisted of analysis of total organic carbon and sediment grain size (Table 2-3). These parameters were measured in surface sediments only (outfalls 3-8 and reference stations). Both analyses were performed by MEC Analytical Systems, Inc. at their Carlsbad, CA laboratory. Total organic carbon analysis was performed following American Society for Testing and Materials (ASTM) Method D2574 modified for sediments. Grain size analysis followed the method of Plumb (1981).

2.3.2 Chemical Tests

Table 2-3 lists the methods used to analyze sediment and tissue chemical parameters. A full suite of chemical analytes was measured in surface and subsurface sediments. A subset of this suite was measured in clam tissues from the bioaccumulation study described in Section 2.3.3 (Table 2-3). Arthur D. Little, Inc. (Cambridge, MA) performed analyses of PAHs, phenols, phthalate, pesticides, PCBs, and butyltins. Metal analyses were performed by Florida Institute of Technology (Melbourne, FL). Cyanide and sulfide analyses were performed by Inchcape Testing Services (Clochester, VT). Target analytes, detection limits, and corresponding analytical methods are discussed for each analytical group in the following subsections. Brief descriptions of analytical methods and rationale for analyte selection and historical relevance of contaminants to Site 1 follow.

Table 2-3. Physical and chemical tests performed for sediments and tissues.

Parameter	Analytical Method
Physical Tests	
Total organic carbon (TOC)*	ASTM D2574 modified
Sediment Grain Size*	Plumb (1981)
Chemical Tests	
Semivolatiles (PAHs, phthalate*, phenol*)	U.S. EPA 8270 modified using SIM
Pesticides and PCBs	U.S. EPA 8082 modified
Butyltins*	Uhler et al. 1991
Metals	U.S. EPA 6020 and 7000 series
Cyanide*	U.S. EPA 9010
Sulfides*	U.S. EPA 9030

*Not analyzed in tissues

2.3.2.1 PAHs, Phenol, and Phthalate

Semivolatile organic compounds consisting of polycyclic aromatic hydrocarbons (PAHs), phenol, and bis[ethylhexyl]phthalate (phthalate) were analyzed using USEPA Method 8270, gas chromatography with mass detector (GC/MS), modified for selective ion monitoring to achieve lower detection limits (i.e., 2 ng/g) (Table 2-4). Compounds were extracted using a modification of USEPA Method 3050. Sediment samples were extracted three times using sonification followed by 12 hours of shaking using methylene chloride and acetone as the extraction solvent. Tissues were extracted three times with a Tissumizer, using methylene chloride as the extraction solvent. Additional cleanup procedures were used for sediments and tissues consisting of high pressure liquid chromatography fractionation followed by silica gel and alumina column cleanup. Cleanup procedures were performed to remove interfering non-target polar compounds.

Table 2-4. Semivolatile organic analytes. Analyzed by USEPA Method 8270 modified. Target detection limit was 2 ng/g (ppb, dry weight) for all analytes.

Phenol[2]	C3-Dibenzothiophenes (D3)
Naphthalene[1] (N)	Fluoranthene[1] (FL)
C1-Naphthalenes (N1)	Pyrene[1] (PY)
C2-Naphthalenes (N2)	C1-Fluoranthenes/Pyrenes (FP1)
C3-Naphthalenes (N3)	C2-Fluoranthenes/Pyrenes (FP2)
C4-Naphthalenes (N4)	C3-Fluoranthenes/Pyrenes (FP3)
Acenaphthylene[1] (AC)	Benzo[a]anthracene[1] (BA)
Acenaphthene[1] (AE)	Chrysene[1] (C)
Biphenyl (BI)	C1-Chrysenes (C1)
Fluorene[1] (F)	C2-Chrysenes (C2)
C1-Fluorenes (F1)	C3-Chrysenes (C3)
C2-Fluorenes (F2)	C4-Chrysenes (C4)
C3-Fluorenes (F3)	Bis(ethylhexyl)phthalate
Anthracene[1] (A)	Benzo[b]fluoranthene[1] (BB)
Phenanthrene[1] (P)	Benzo[k]fluoranthene[1] (BK)
C1-Phenanthrenes/Anthracenes (P1)	Benzo[e]pyrene (BE)
C2-Phenanthrenes/Anthracenes (P2)	Benzo[a]pyrene[1] (BAP)
C3-Phenanthrenes/Anthracenes (P3)	Perylene (PER)
C4-Phenanthrenes/Anthracenes (P4)	Indeno[1,2,3,-c,d]pyrene[1] (IP)
Dibenzothiophene (D)	Dibenzo[a,h]anthracene[1] (DA)
C1-Dibenzothiophenes (D1)	Benzo[g,h,i]perylene[1] (BG)
C2-Dibenzothiophenes (D2)	

(1) EPA Priority Pollutant (EPA_{16}); (2) Not analyzed in tissues

PAHs consist of carbon and hydrogen arranged in two or more fused or linked benzene rings. PAHs tend to concentrate in sediments owing to their relative insolubility in water and high affinity for particulate matter. PAHs are found in all petroleum-based mixtures including, crude oils, refined fuels, fuel combustion products, preservatives such as creosote, and lubricating oils. Natural sources are almost always secondary to anthropogenic inputs and occur from natural oil seeps, forest fires, and direct biogenesis by microbes and plants (Kennish 1992). The most likely sources of PAHs to Site 1 sediments are sewage and industrial effluents and petroleum spills that were historically discharged from Site 1 outfalls, and combustion (pyrogenic) inputs from atmospheric dust and coal tar. PAHs are relevant to the ecological and human health of Site 1 due to their potential carcinogenic, mutagenic, and toxic effects if bioavailable above threshold concentrations.

Surface sediments from outfall and reference sites were analyzed for an expanded list of PAHs that included 20 parent (unalkylated) compounds and 21 alkylated homologues. The EPA 625/8270 list of PAH analytes (USEPA 1986) was expanded from 16 to 41 to include the alkylated homologues, dibenzothiophenes (sulfur containing compounds), and several additional high molecular weight PAHs. These compounds, particularly the dibenzothiophenes, are often prevalent after weathering, combustion, or biodegradation of petroleum, and can provide insight into the PAH source. The alkylated compounds also persist longer in the environment than their associated parent PAHs, and therefore, provide a more reliable fingerprint, even after extensive environmental degradation (Douglas et al. 1992; Sauer and Boehm 1991; Page et al. 1995).

Bis[ethylhexyl]phthalate (phthalate) and phenol were two other semivolatile organic compounds analyzed in surface sediments. Phthalate esters are ubiquitous environmental contaminants, found in most plastics. Phthalate is also a common laboratory-derived contaminant. Phenol is a fairly soluble polar compound, and is not usually detected in sediments. Like creosote, it is found in wood preservatives and therefore, is sometimes detected in sediments close to pier pilings.

2.3.2.2 Chlorinated Pesticides and PCBs

Chlorinated pesticides and PCB congeners were analyzed using gas chromatography with electron capture detector (GC/ECD) (a modification of USEPA Method 8082). Twenty-one USEPA priority pollutant pesticides and 18 PCB congeners were analyzed at less than 1 ng/g detection limits (Table 2-5).

Chlorinated pesticides are synthetic organic compounds that do not occur naturally in the marine environment. However, due to historic waste disposal practices throughout southern California, natural dispersion, and their persistence in the marine environment, concentrations of chlorinated pesticides exceed analytical detection limits near most urbanized and remote areas of the Southern California Bight (SCCWRP 1992; USEPA 1994a). Even though most of these compounds are no longer manufactured, they persist in the environment due to their high stability and affinity for fine grained particles and organic matter.

Chlorinated pesticides were largely non-detected in previous studies conducted in the vicinity of Site 1 (Section 1.4). However, analytical detection limits in previous investigations were generally one to three orders of magnitude higher than those achieved in this program. Since these compounds persist at trace levels in San Diego Bay (SANDAG 1991; SWRCB 1992), and are potentially toxic to marine organisms, they were included in this investigation.

PCBs also are a synthetic group of chlorinated hydrocarbons and have been linked to a number of health concerns, such as cancer in man and fin-erosion and lesions in fish (Kennish 1992). They were widely used in manufacturing of adhesives, caulking compounds, as additives to hydraulic fluids, paints, plastics, and most commonly as insulators in electrical transformers and capacitors. These chemicals primarily were manufactured as Aroclors7, which are mixtures of various PCB congeners. Even though PCBs persist in the environment, relative concentrations of individual congeners change as the mixture degrades, making it difficult to detect Aroclors at low levels, using standard analytical methods (i.e., USEPA method 8082). For this reason, PCBs were analyzed as 18 individual congeners, similar to those analyzed in the National Status and Trends Program (BPTCP 1996). These congeners have varying degrees of chlorination of the biphenyl molecule. Mono- through deca-chlorinated biphenyl isomers were analyzed, covering broad chlorination of congeners.

Table 2-5. Pesticides and PCBs. Analyzed by U.S. EPA Method 8082 modified. Target detection limit was 0.4 ng/g (ppb, dry weight) for all analytes.

Pesticide Analyte	PCB Analyte
Hexachlorobenzene	8* - 2,4'-Dichlorobiphenyl
alpha-BHC	18 - 2,2',5-Trichlorobiphenyl
beta-BHC	28 - 2,4,4'-Trichlorobiphenyl
delta-BHC	44 - 2,2',3,5'-Tetrachlorobiphenyl
gamma-BHC	52 - 2,2',5,5'-Tetrachlorobiphenyl
cis-Chlordane	66 - 2,3',4,4'-Tetrachlorobiphenyl
gamma-Chlordane	101 - 2,2',4,5,5'-Pentachlorobiphenyl
2,4'-DDE	105 - 2,3,3',4,4'-Pentachlorobiphenyl
4,4'-DDE	118 - 2,3',4,4',5-Pentachlorobiphenyl
2,4'-DDD	128 - 2,2',3,3',4,4'-Hexachlorobiphenyl
4,4'-DDD	138 - 2,2',3,4,4',5'-Hexachlorobiphenyl
2,4'-DDT	153 - 2,2',4,4',5,5'-Hexachlorobiphenyl
4,4'-DDT	170 - 2,2',3,3',4,4',5-Heptachlorobiphenyl
Aldrin	180 - 2,2',3,4,4',5,5'-Heptachlorobiphenyl
Dieldrin	187 - 2,2',3,4',5,5',6-Heptachlorobiphenyl
Endrin	195 - 2,2',3,3',4,4',5,6-Octachlorobiphenyl
Endrin Ketone	206 - 2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl
Endrin Aldehyde	209 - Decachlorobiphenyl
Endosulfan I	
Endosulfan II	
Endosulfan Sulfate	

* International Union of Pure and Applied Chemistry (IUPAC) identification number

2.3.2.3 Metals

Metals were analyzed using either flame or graphite furnace atomic absorption spectrophotometry, or inductively coupled plasma spectrometry with mass detector after digestion with hydrofluoric acid-nitric acid-hydrochloric acid (Table 2-6). The methods differ from standard USEPA methods, such as those used in the BPTCP study, since total instead of partial digestion is performed prior to analysis. Total digestion produces slightly higher concentrations for all metal anlytes compared to partial digestion; however, total digestion results are more reliable for source identification of metals and in discerning differences between areas, which were objectives of this study. Detection limits ranged from 0.1 μg/g for mercury to 6 μg/g for aluminum.

Although industrialization and development in the coastal zone can result in elevated concentrations of heavy metals, nearly all metals occur naturally in estuarine sediments. Therefore, determination of anthropogenic contribution of a given metal requires establishing the natural metal concentration for a particular sediment. Natural sources of metals to San Diego Bay sediments are fine-grained clay particles and aluminosilicate minerals. These high-metal content sediments are naturally mixed with low-metal content quartz sands and carbonate shell material. Relative mixtures of these inputs, as well as inputs from contaminant sources, can be discerned by normalizing metal concentrations to aluminum or iron or another determinant that is not distorted by anthropogenic inputs (Bruland et al. 1974; Trefry and Presley 1976). These metals, which occur naturally in high concentrations (i.e., > 40,000 μg/g for aluminum and > 80,000 μg/g for iron) were included in the Site 1 investigation to provide source identification information and are not considered chemicals of concern. Iron was found to be the best proxy for the metal-bearing phases of Site 1 sediments due to its strong relationships

with cobalt and percent fine (silt + clay) sediment as discussed in Section 4. It is important to determine relative inputs of metal sources, so that naturally elevated metal concentrations are not misinterpreted.

2.3.2.4 Butyltins

Sediments and tissues analyzed for butyltin compounds (Table 2-6) were extracted with hydrobromic acid, derivatized and cleaned using florisil prior to analysis using gas chromatography with flame photometric detector (GC/FPD). The butyltin method followed that of Uhler et al. (1991), as no final USEPA or ASTM method exists.

Tributyltin is a synthetic compound used primarily as an antifouling toxicant in boat paints and, to a lesser degree, in wood preservatives and insecticides (Seligman et al. 1986). In marine and estuarine environments, tributyltin degrades by dealkylation to dibutyltin, monobutyltin, and eventually to an inorganic tin salt. Although use of tributyltin on vessels under 75 feet long was banned in 1988, tributyltin residues persist in sediments near boat harbors and commercial and military vessel activities (O'Connor and Beliaeff 1995).

Table 2-6. Metal and butyltin analytes, corresponding methods, and dry weight detection limits.

Analyte	Method *	Detection Limit
Metals (μg/g, ppm)		
Aluminum	FAAS (EPA 7020)	6.0
Antimony	ICP/MS (EPA 6020)	0.2
Arsenic	ZGFAAS (EPA 7060)	0.1
Barium	ICP/MS (EPA 6020)	0.1
Beryllium	ICP/MS (EPA 6020)	0.1
Cadmium	GFAAS (EPA 7131)	0.02
Chromium	GFAAS (EPA 7191 or 7190)	0.2
Cobalt	ICP/MS (EPA 6020)	0.1
Copper	ICP/MS or GFAAS (EPA 6020 or 7210)	0.3
Lead	ICP/MS (EPA 6020)	0.1
Manganese	ICP/MS (EPA 6020)	0.2
Mercury	CVAAS (EPA 7471)	0.01
Molybdenum	ICP/MS (EPA 6020)	0.1
Nickel	ICP/MS (EPA 6020)	0.2
Selenium	ZGFAAS (EPA 7740)	0.3
Silver	ICP/MS (EPA 6020)	0.2
Tin	ICP/MS (EPA 6020)	0.1
Thallium	ICP/MS (EPA 6020)	0.1
Titanium	ICP/MS (EPA 6020)	0.3
Vanadium	GFAAS (EPA 7910 or 7911)	0.2
Zinc	FAAS (EPA 7950)	1.0
Butyltins (ng/g, ppb)		
Monobutyltin	GC/FPD (Uhler et al. 1991)	1.0
Dibutyltin	GC/FPD (Uhler et al. 1991)	1.0
Tributyltin	GC/FPD (Uhler et al. 1991)	1.0

* FAAS - Flame atomic absorption spectrophotometry; ICP/MS - Inductively coupled plasma/mass spectrometry
ZGFAAS - Zeeman graphite furnace atomic absorption spectrophotometry;
GFAAS - Graphite furnace atomic absorption spectrophotometry;
CVAAS - Cold vapor atomic absorption spectrophotometry; GC/FPD - Gas chromatography/flame photometric detector

2.3.2.5 Cyanide and Sulfides

Cyanide and acid soluble sulfides were analyzed using USEPA Methods 9010 and 9030, respectively (Table 2-7).

Table 2-7. Cyanide and sulfides. Analytical method and dry weight detection limits for cyanide and sulfide.

Analyte	Method	Detection Limit	Units
Cyanide	EPA 9010	0.5	µg/g
Sulfides	EPA 9030	5.0	µg/g

Cyanide is not usually present in marine sediments due to its high solubility and low affinity for particulates. Cyanide was either not measured or was undetected in previous investigations discussed in Section 1.5. Although acutely toxic to humans, cyanide has relatively low toxicity for fish and many marine invertebrates. Cyanide was analyzed because it was listed as a target analyte in Bechtel's (1995) work plan.

Sulfides are present naturally in estuarine sediments where they are produced under anaerobic conditions by sulfate-reducing bacteria. Sulfides react with metals to form insoluble sulfides. In general, sulfide concentrations are not well-correlated with sediment grain size, TOC, or concentrations of individual metals. However, if acid soluble sulfides are present in high enough concentrations, appreciable concentrations of hydrogen sulfide can dissolve into pore water in a reducing environment. This form of sulfide is acutely toxic to various biota and of concern. Total sulfides measured in sediments provide only a general indication of contamination.

In addition to analyses performed for this RI/RFI, volatile organic compounds and radiological isotopes were measured at select stations to support other investigations. Volatile organic compounds (VOCs) were analyzed at outfall 3 (Pier Bravo), outfall 1,2, and outfall 16 to augment studies at IR Sites 9, 7, and 5, respectively. Radiological (gamma spectroscopy) analyses were performed on surface sediments collected at outfall 4 and the ten in-bay reference stations in support of ongoing studies at IR Site 10. Analytical methods used, result summaries, and laboratory reports for VOCs and radiological isotopes are included in Appendices B and C, respectively. These results are not used in the Site 1 investigation.

2.3.3 Bioassays

A large number of bioassays using marine invertebrates have been developed and used to describe, monitor, and assess the nature and extent of toxicity in sediments (e.g., Swartz et al. 1979, 1985; Chapman et al. 1985; Long and Buchman 1989; Pastorok and Becker 1990; USACOE 1991; Battelle 1993). Many of these tests have been used to evaluate potentially contaminated sediments in central and southern California. Five of these tests were employed using surface sediments collected at the 30 in-bay outfall stations and 10 bay reference stations, and are summarized in Table 2-8. There are two broad categories of tests, solid-phase whole sediment and pore water tests. Toxicity is the main measured effect in all of these tests except for the 28-day clam bioaccumulation, in which biological uptake of chemical contaminants from sediment is measured.

Three solid-phase toxicity tests were performed, acute and chronic tests with an amphipod and polychaete worm, and an acute test with mysid shrimp. An acute test is defined as a comparative study of survival in which

test organisms are observed over a short period of their lifetime. Chronic tests expose organisms over a long period or a substantial portion of their life span, and non-lethal endpoints are usually observed. Test animals were selected to represent ecologically relevant species that either feed from sources in contact with, bury in, or ingest surficial sediments in San Diego Bay.

Table 2-8. Test description, species used, measured effects, and relevance of selected laboratory sediment bioassays.

Bioassay	Species	Measured Effect(s)	Comments
Mysid 4-Day Solid-phase Toxicity Test	*Holmesimysis costata*	Survival	Simple test, local organism, ecologically relevant, widely used in sediment elutriate tests
Amphipod 10-Day Solid-phase Toxicity Test	*Rhepoxynius abronius*	Survival, Reburial	Simple test, reproducible, ecologically relevant, applicability verified in many studies, common California test
Juvenile Polychaete 20-Day Solid-phase Toxicity Test	*Neanthes arenaceodentata*	Survival, Growth	Long term subchronic test, measures growth (biomass rate of change) response and mortality, applicability verified in many West Coast studies
Bivalve Larvae 48-Hour Toxicity Test, in Pore Water	*Crassostrea gigas*	Normal Development	Simple test, widely used in effluent tests, very sensitive
Bivalve 28-Day Solid-phase Bioaccumulation Test	*Macoma nasuta*	Bioaccumulation of Chemicals	Long term test, widely used in dredge material tests for ocean disposal

A 48-hour acute test, examining successful development of oyster larvae, was performed to measure toxicity in sediment pore water. Free and mobile contaminants often do not remain in the sediment matrix because advection, diffusion, and convection processes transport contaminants vertically into the water column (Burton 1992). These processes lead to bulk transport of pore water (also referred to as interstitial water) containing dissolved and suspended material from sediment to the overlying water column. This material (dissolved chemicals, and chemicals adsorbed to colloids and small particles) is believed to be among the major carriers of toxicity introduced through pore water.

A 28-day clam bioaccumulation test was undertaken to evaluate the potential of contaminant uptake and subsequent food chain transfer. Biomagnification of organic chemicals and metals is known to occur across trophic levels. Outfall stations again were compared to appropriate reference stations to determine whether they posed a significantly greater threat in this regard. The test animal selected is widely distributed and native to San Diego Bay, commonly used in dredged sediment studies (USEPA/USACOE 1991), is known to actively ingest surface sediments, and provides enough tissue for trace level chemical analysis.

All bioassay testing was conducted at MEC laboratories in Carlsbad, San Diego, or Tiburon, CA using appropriate ASTM (1993, 1994, 1995) or USEPA/USACOE (1991) protocols identified in the following subsections.

2.3.3.1 4-day Mysid Toxicity Test

The mysid shrimp, *Holmesimysis costata*, was used in a 4-day, acute, solid-phase test based on ASTM Protocol E1463-92. *Holmesimysis costata*, is an abundant species in southern California with wide seasonal availability, and is found in many nearshore marine habitats. It is especially abundant during day light hours under floating fronds of the giant kelp, *Macrocystis pyrifera*. *H. costata* exploits the water column and sediment as food sources, feeding opportunistically on suspended particles, zooplankters and microalgae, making it a useful, representative, and ecologically relevant test organism. The primary feeding mode uses thoracic endopods to tear apart large living or dead food items. Secondarily, suspended fine particles are filtered from incoming currents of water produced by movements of the expopods of the thoracic limbs (Tattersall and Tattersall 1951). Historically *H. costata* have been used in toxicity testing to screen effluent discharges, elutriates of dredge sediments and recently in whole sediment, solid-phase bioassays (Ward et al. 1995).

Holmesimysis were collected from La Jolla, CA kelp beds by an independent collector and shipped to MEC, Carlsbad in seawater. Temperature, dissolved oxygen, pH, and salinity of water in which animals were shipped were measured upon receipt of organisms. Tests were started within three days of receiving and two days of acclimating test animals. Mortality, reported as percent survival, was reported for each of five laboratory replicates after 4 days of exposure to sediments. Clean filtered seawater obtained from Scripps Institution of Oceanography was used in laboratory controls. Passing criterion was ≥ 90% average survival in the control on day 4.

2.3.3.2 10-day Amphipod Toxicity Test

The amphipod *Rhepoxynius abronius* was used in a 10-day, acute, solid-phase test based on ASTM Protocol E1367-92. *Rhepoxynius abronius* typically inhabit well-sorted, fine sand and generally perform well in a wide variety of sediment types. Mortality, reported as percent survival, was the primary effect measured. Failure to rebury in clean sediment at conclusion of the 10-day exposure was used as a subchronic toxicity measure. The ecological relevance of reburial is based on the assumption that individuals that fail to rebury in sediments at the end of the exposure period would be consumed by predators in nature.

Amphipods were collected off Whidbey Island in Puget Sound, WA. Amphipods were received at MEC, Tiburon, in control sediment with overlying seawater. Temperature, dissolved oxygen, pH, and salinity of water overlying the sediment were measured upon organism arrival and daily during acclimation. Tests were started within three days after receipt of test animals.

Test sediments were run with control sediments collected from Whidbey Island in Puget Sound, WA, and fine-grained control sediments collected from Tomales Bay, CA. At the conclusion of the bioassay, percent survival and percent reburial were reported for each of five laboratory replicates. Passing criterion was ≥ 90% average survival in the control on day 10.

2.3.3.3 20-day Polychaete Toxicity Test

The marine nereid polychaete worm, *Neanthes arenaceodentata*, was used in a 20-day chronic test in which both survival and growth (expressed in biomass) were measured following ASTM Protocol E1611-94. *Neanthes* is a widely distributed genus with various life stages that have been used in bioassays since 1966 (Reish 1984). General distribution of *Neanthes arenaceodentata*, the species used in this test, is in nearshore marine areas on

the west coast of the United States ranging from southern California to Mexico. A sediment-dwelling tube-building omnivore, *Neanthes arenaceodentata* feeds on algae and other detritus (Fauchald and Jumars 1979). It is a well studied member of the infaunal benthic community and of ecological importance to the Site 1 investigation.

Neanthes have been used to examine specific effects of dissolved oxygen concentrations, nutrients, salinity, temperature, metals, pesticides, hydrocarbons (Reish 1984), mutagens, (Pesch et al. 1981; Pesch and Pesch 1980) and irradiation (Jones et al. 1983). Johns and Ginn (1990) have reported that the level of contamination affecting juvenile growth in *Neanthes* is similar to the level of contamination affecting reproductive success. This important aspect was taken into consideration in the choice of this test species.

Neanthes were cultured and supplied by California State University Long Beach and shipped to MEC, Carlsbad, in seawater. Tests were started on the dates animals were received. Control sediment was collected from Saint Augustine, FL, by Aquatic Indicators. The specific test employed in this program measures survival and growth for juvenile *Neanthes* following a 20-day exposure to sediments. Water quality measurements (temperature, dissolved oxygen, pH, salinity, and ammonia) were performed on all replicates on the day of test initiation and on one replicate of each sample on days 3, 6, 9, 12, 15, 18, and 20 of the test. Temperature was recorded continuously in the water surrounding test chambers. Percent survival, and growth, calculated as daily changes in biomass, were the reported effects. Passing criterion was ≥ 90% average survival in the control on day 20. Test protocol does not specify a control criterion for growth; however, control results are presented with test (outfall and reference) results in Section 5 (see Figure 5-1).

2.3.3.4 48-hour Bivalve Toxicity Test

Toxicity of pore water, extracted through centrifugation of sediment, was assessed through the performance of a 48-hour bivalve larval developmental test using the bivalve oyster *Crassostrea gigas*. Protocols for this test are based on standard procedures and guidelines specified in ASTM E724-94. *Crassostrea gigas* ranges from British Columbia to southern California and is commonly used in effluent testing. The USEPA/USACOE Green Book (1991) also recommends testing the effects of dredged material using larvae of this species in an elutriate test. Local species distribution and sensitivity combine to help make this an ecologically relevant test organism. The primary toxicity endpoint for this test is the proportion of individuals that develop in a normal manner to free-swimming, fully shelled larvae (prodissoconch I stage), reported as percent normal development.

Adult bivalves were obtained from the commercial supplier A. Kimbrough Siewers in Santa Cruz, CA. Adult bivalves were stimulated to spawn, and embryos were collected and used within 4 hours of fertilization. At 0 and 48 hours, temperature, dissolved oxygen, salinity, and pH were measured in the pore water, seawater control, brine control, and zero-time control. Undiluted (100%) pore water was the test material; filtered seawater collected at Bodega Bay, CA was used in seawater controls. Near the end of the 48-hour incubation period, control test vials were examined under a microscope to check for complete development of control organisms. Once development was complete to the prodissoconch stage, the test was ended by adding 0.5 mL of 2.5-5% buffered formalin to each vial. The time the test was terminated was recorded, and surrogate sets of test vials were used for final water quality measurements. Larvae were counted and scored as either normal (completely developed shells containing meat) or abnormal (shell development is incomplete). The percentage of embryos that did not have completely developed shells (i.e., abnormal development) was calculated for each treatment. Percent abnormal organisms was equal to 100 times the difference of the number of normal larvae and the initial

number of embryos in the test containers divided by the initial number of embryos in the test containers. Passing criterion was ≥ 70% average normal development in the control at 48 hours.

2.3.3.5 28-day Bioaccumulation in the Bentnose Clam

Bioaccumulation of contaminants was evaluated using a 28-day bioassay with the bentnose clam, *Macoma nasuta*. Procedures for the bioaccumulation protocol were developed by USEPA/USACOE (1991) and Battelle (1992) and followed in this study.

The bentnose clam is a large animal, whose shell is usually 60 mm long but may reach 110 mm. It lives in gravel, sands, mud and clay, 10-20 cm below the surface. It is found in sheltered bays from Alaska to the tip of Baja California and is a native in San Diego Bay. When buried, the inhalant siphon extends up to several centimeters out of the burrow, then curves downward and sweeps the bottom surface, sucking up fine detritus along with a great deal of mud and sand. The clam is commonly used in Army Corps dredge tests (USEPA/USACOE 1991).

Clams were collected in Sequim, WA, by an independent collector, Johnston and Gunstone, and shipped to MEC, Carlsbad. Clams were then transferred to the MEC San Diego flow-through facility at Harbor Island and acclimated for 3 days prior to testing. Clams were acclimated in 30 ± 2 ppt seawater at ambient water temperature (15-20° C). Sediment shipped with the clams was used in controls. Water quality measurements were made in one replicate chamber for each sediment sample daily. After 28 days of exposure to test sediments followed by one day of depuration, laboratory replicates of surviving *Macoma* were composited. Tissues were then placed on ice and shipped to Arthur D. Little's , Marine Chemistry Laboratory in Cambridge, MA for chemical analysis. Tissue samples were homogenized and assayed for the same parameters as corresponding surface sediments, except for phenol, phthalate, cyanide, and sulfides. Percent lipids also were analyzed. Analytical methods for tissue are discussed in Section 2.3.2. Percent survival was used as a quality control criterion for the test and not as an assessment endpoint. Assessment endpoints for this test were chemical concentrations measured in the tissues. Passing criterion for the bioassay portion of the test was ≥ 70% average survival in the control after 28 days. The bioassay portion of this test was re-run after the initial test failed to produce ≥ 70% survival in controls using re-collected sediment for all 40 in-bay stations (see Volume III.3).

2.4 Data Analysis

All data analyses were performed by computer using the statistical software SAS® (version 6.10) in batch programming mode. Computer print-outs of statistical results and corresponding SAS® computer programs are presented in Volume II. Interpretation of results are presented in Sections 3-6. Quality control procedures used to ensure program accuracy are presented in Volume III.2.

2.4.1 Descriptive Statistics

Descriptive statistics were performed on all physical, chemical, and biological data. Software programs used for analysis of project data and results from descriptive statistics are included in Volume II. Statistical outputs included computations for number of samples, means, standard deviations, ranges of values, and frequencies of detectable concentrations. Computations were performed on final results data that had passed quality control review. These data included estimated chemistry results (J-qualifier) and any other data that did not carry an R-

qualifier (invalid result) after data validation. Nearly all J-qualifiers were for reported chemical concentrations that were detected below the practical quantitation limit and at or above the method detection limit. Data validation procedures and data qualifiers are described in Volume III.2.

Descriptive statistics were computed directly for chemical and physical results. Non-detect results for chemistry data were excluded from statistical analyses. Mean values for each set of bioassay laboratory replicates were used in the analysis. Replicate values for each bioassay were tested for outliers using the Studentized Range Test (Natrella 1966). Outliers were discarded after review by the project toxicologist, and mean values and standard deviations were calculated using remaining replicates. Any test that had greater than one outlier was repeated (Volume III.3). Replicate outliers are identified in Appendix D1. These consisted of two pore water measurements for Station 3-2, replicate 3, and Station R-15, replicate 1. The three replicate results for pore water oyster larvae development at Station 3-2 were 1.2%, 0.74%, and 98.07% (eliminated). The three replicate results for Station R-15 were 18.5% (eliminated), 76.2%, and 80.3%. Though not verified, it is believed that the reason for aberrant values and subsequent elimination was that results from the two stations were transposed. No other data were excluded from analysis in this manner.

2.4.2 Comparative Statistics

Comparative statistics (e.g., inference tests) were used to determine whether outfall sediments were more contaminated or toxic than designated reference sediments (Objectives 1 & 2 in introduction to Section 2). Prior to inference testing, chemical and biological results were tested for normality using the Shapiro-Wilk statistic (Shapiro and Wilk 1965) to meet test assumptions. Nearly all chemistry data were log normally distributed; these data were log transformed prior to analysis. All bioassay data were non-normally distributed, except the polychaete growth data. Arcsine transformation failed to produce normal distributions. The non-normal distributions occurred because all bioassay data, except polychaete growth, are bounded (e.g., range from 0-100%). Nearly all bioassay results had survival or other endpoints (i.e., reburial) of nearly 100%, except for normal development in the 48-hr bivalve test. Bivalve larvae development data were strongly bimodal and also non-normal (either 100% failure or nearly complete development). The Student's t-test (Steel and Torrie 1960) was performed using untransformed bioassay data as the test is robust for non-normally distributed data (Sokal and Rohlf 1969).

2.4.2.1 Testing of Grouped Stations

Testing of grouped stations was performed for in-bay surface sediments only (outfalls 3-8). Reference and outfall stations were each separated into two groups: those comprised primarily of fine-grained sediments and those with primarily coarse-grained sediments (see Section 3). Tests were performed comparing either grouped or individual outfall stations to relevant grouped reference stations. Grouped comparisons were made to determine whether combined Site 1 surface sediments (grouped by grain size) were different from combined reference sediments with similar grain size distributions. One-tailed Student's t-tests (Steel and Torrie 1960) were performed for each of the two grouped comparisons: outfall coarse vs. reference coarse and outfall fine vs. reference fine. For each test, equality of variance was tested; when variances were unequal, the approximate t-statistic for unequal variance was used. A Bonferroni adjustment for multiple testing was applied to resultant probabilities (Milliken and Johnson 1984). All significant differences between outfall and reference stations in terms of bioassay, chemical concentration, or bioaccumulation results were treated equivalently in the decision matrix,

and, thus, correction for multiple testing was felt justified. The decision matrix provides guidelines for additional studies, remediation, or no further action based on biological and chemical results (see Section 6). Bonferroni adjustment for toxicity tests reduced the probability (p) from ≤ 0.05 to ≤ 0.01 for a one-tailed test. (The test is one-tailed because only outfall sediments that are more toxic, not less, than the reference are important.) Applying the same adjustment to chemistry tests reduced p from ≤ 0.05 to ≤ 0.0005. The difference in p occurred because there were only six toxicity results compared to 108 for sediment chemistry. Many contaminants co-occur (e.g., metals are found together as are PAHs and PCBs), and thus their distributions are not independent. The co-occurring contaminants effectively reduce the number of independent contaminants and make a full Bonferroni correction to p ≤ 0.0005 too extreme. Not knowing the number of independent contaminant "groups" or degree of correlation among them, a p ≤ 0.01 for defining a significant difference between in-bay outfall and reference stations was adopted for the contaminants as well. This reflects a conservative approach, since it corrected for multiple testing of only six analytes, rather than the full suite of 108 which were tested.

2.4.2.2 Testing of Individual Stations

Individual comparisons were made between each surface and subsurface sediment sample and the relevant (based on grain size) reference group predictive limit (Steel and Torrie 1960). This procedure was used to identify potential problems (hot spots) at individual stations that might have been concealed in the group comparisons. Comparisons were made for individual bioassay, and sediment and tissue chemical parameters for each surface sediment, and for individual chemical parameters for each subsurface sediment core interval. Grain size was not measured for any of the subsurface sediments. Sediments collected at outfalls 1,2 and 16 were compared to the coarse grain surface reference sediments. This approach was used as a conservative estimate of subsurface contamination, as coarse grain reference sediments have lower contaminant burdens compared to fine grain reference stations (see Section 4). Subsurface sediments collected at outfalls 3-8 were compared to the same reference group as the corresponding surface sediment sample collected from the same station. Predictive limits were calculated for each grain size reference group for bioassay and chemistry results. Both 95% and 99% predictive limits were calculated, representing unadjusted and Bonferroni adjusted limits, respectively, based on an original alpha level of 0.05 (i.e., 95%). A lower predictive limit was calculated for reference survival (to identify stations with lower percent survival compared to reference); and an upper predictive limit was calculated for reference chemistry (to identify outfall stations with greater chemical concentrations compared to reference). The predictive interval is a modification of the confidence interval and is used when comparing individual results to a population mean. The predictive interval is defined by the following formula:

$$(a,b) = \bar{x} \pm t * s * \sqrt{1 + \frac{1}{n}}$$

where:

a and *b* are lower and upper limits of the predictive interval;
$\bar{x}$ is the sample mean;
t is the value from the t-table for the desired α level at n-1 degrees of freedom;
s is the sample standard deviation; and
n is the size of the sample.

The predictive interval is greater than the confidence interval by the square root term. Predictive intervals were calculated for particular reference contaminant concentrations only when a contaminant was detected at three or

more grouped reference stations. This criterion was not met for several chemical analytes, which were often undetected at reference stations. Non-detect results for chemistry data were excluded from statistical analyses. Cases where detectable concentrations were reported for outfall stations but reference data were insufficient to calculate predictive limits are discussed in Section 4.

Statistical results for reference site comparisons are discussed in Section 4 for sediment chemistry and Section 5 for bioassays and tissue bioaccumulation. Both uncorrected and Bonferroni corrected results are presented for reference; however, all statistical differences are interpreted using results corrected for multiple testing. Complete results for all statistical tests are presented in Volume II.

2.4.3 Gradient Analyses

Linear regression was used to search for horizontal gradients (surface sediments) in contaminant concentrations and biological results with distance from in-bay outfalls. Regressions were performed for all coarse-grained stations grouped by distance from outfall (e.g., result vs. distance using all coarse stations). There were too few fine-grained stations to regress chemical and biological results with distance from outfall. Chemical concentrations and biological results were regressed with percent fine sediment to assess any correlations with sediment size. Results are discussed in Section 6.1.

Subsurface cores were analyzed to identify linear vertical trends in chemical concentrations only. For in-bay stations, the stations were pooled, and chemical concentrations for each analyte were regressed against depth. For ocean channels, regressions were performed for each outfall by grouping stations to look for linear trends in chemical concentration with depth. Analytes that were detected in at least 50% of the samples were included in the analysis. Results are discussed in Section 4.2.

2.5 Quality Assurance/Quality Control Methods

Complete descriptions of quality assurance/quality control (QA/QC) methods are found in the QAPP (MEC 1996) and in Volume III.2 of this report. Standard operating procedures for field and analytical methods are included in the QAPP only. Brief descriptions of QA/QC program elements follow.

2.5.1 Field Program

Field duplicates, field blanks, equipment blanks, and rinseate blanks were used as quality control (QC) checks on the integrity of sample collection and handling procedures. Field QC samples were collected at a minimum of once per day per sampling effort (e.g. collection of surface sediments).

Temperature blanks were included with each sample shipment to ensure that shipping container temperatures did not exceed 10° C or ambient field temperature if received by the laboratory within 24 hrs of collection. The temperature blank was measured upon receipt at the laboratory to verify sample integrity. Trip blanks were included with each sample shipment that contained samples for analysis of VOCs.

All samples were shipped with chain-of-custody forms. These forms were completed by field personnel and placed in sealed shipping containers with the corresponding samples. Copies of all chain-of-custody forms reside with the raw data packages produced by the analytical laboratories.

2.5.2 Data Management

Field logbooks were reviewed for completeness, accuracy, and consistency by field and data management personnel. Entries that were incomplete, inconsistent, or that were not clear were discussed with the field manager. A written review of all questions and their resolution including an assessment of the impact on data quality of any discrepancies was performed.

Following resolution of any questions regarding field data, the data were entered directly from field logbooks to Excel® files. Data were converted to SAS® databases, and electronic checks were run to verify sampling dates, location and sample type information for all sample numbers, and correct entry of sample numbers. All electronic entries were verified with original field logbooks.

2.5.3 Physical and Chemical Analyses

Routine laboratory QC checks included analysis of method blanks, continuing calibration standards, matrix spike and spike duplicates, surrogate spikes, reference standards, laboratory control samples, and laboratory duplicates as called for by each analytical method. All QC samples were analyzed at least once with each analytical batch (≤ 20 samples). Frequency, type, and acceptance criteria for quality control checks are listed in Volume III.2 for all physical and chemical analyses. An external chemistry validation report prepared by EcoChem, Inc. (Seattle, WA) for all sediment, tissue, and field quality control samples is included in Volume III.4. Summarized results of the external validation are presented in Volume III.3.

2.5.4 Bioassays

Quality control checks for bioassays included laboratory negative control samples, reference toxicants (positive controls), laboratory replicates, and water quality measurements. One laboratory control test was performed for each analytical batch (≤ 20 samples). Five laboratory replicates were tested for each solid phase toxicity bioassay, three laboratory replicates were tested for the pore water bioassay, and four replicates were used for the clam bioaccumulation test. Reference toxicants were tested for every batch of test organisms received, except for clams, which do not require a reference toxicant because mortality is not a measurement endpoint. Results for water quality measurements are presented in Appendix D2. Quality control results are discussed in Volume III.3.

2.5.5 Data Management

Data were received electronically from laboratories in Excel® files. Format of the files was specified and set prior to the first delivery of data. Files were converted to SAS® using DBMSCOPY™. Prior to incorporation into the database, data were electronically checked: 1) for valid sample numbers; 2) to verify that data were received for all expected samples; 3) for consistent measurement units for each parameter; 4) to verify that sample type and required QC sample results were included; and 5) to verify that chemistry sample results were above the reported detection limit. Quality control procedures for data calculations and statistics included verification of results using other software (e.g., PRODAS™) and independent programmers, as well as review of all computer programs for accuracy. External data validation was performed by CalOxy, Inc. (Carlsbad, CA) on a subset of the data. Descriptive statistics and 95% predictive intervals were calculated for PAH and polychaete growth data. Results generated by CalOxy were compared with internal results and no inconsistencies were found. Computer printouts of validation tests are included in Volume II.

Grain size and total organic carbon (TOC) results for in-bay surface outfall and reference stations are presented in this section. As previously discussed in Section 2.2, these parameters influence contaminant distributions and bioavailability in sediments and results were used to separate reference and outfall stations into fine and coarse grain size groups. These groups are maintained in subsequent comparisons of outfall and reference station conditions.

3.1 Surface Sediment Grain Size

Sediment grain size analysis produced 17 size fractions for each sample, ranging in diameter from 0.00098 mm (clay) to 2 mm (very coarse sand). Sediment size classes and corresponding grain size diameters are shown in Table 3-1 for reference. Results for percent gravel, sand, silt, and clay, and mean grain size were calculated for the 40 in-bay stations and are shown in Table 3-2. At most outfall stations, including those within 30 m of the outfall terminus, sediments had greater than 60% sand. Sand contributed less than 60% at some stations further from the outfalls (i.e., stations 5-4, 6-4, 6-5, 7-4, 7-5, and 8-5). Sediments near the entrance to the Bay and between North Island and Point Loma (outfall 3 and reference stations R-2 and R-3) were fine sands. Reference stations R-1, R-4 and R-5 had high sand (83.9%, 65.0% and 66.8%, respectively). Sand was the major component of sediments at all stations except outfall station 8-5 and reference station R-8, where silt was the primary fraction. Gravel was a minor component in most samples, except at outfall station 8-1 (19.1% gravel), where rocks and shell hash were found.

Table 3-1. Size class for sediment grain size.

Grain Diameter (mm)	Size Class	Grain Diameter (mm)	Size Class
64	Pebble	0.25	Fine sand
16		0.210	
		0.177	
		0.149	
4	Granule	0.125	Very fine sand
3.36		0.105	
2.83		0.088	
2.38		0.074	
2.00	Very coarse sand	0.0625	Coarse silt
1.68		0.053	
1.41		0.044	
1.19		0.037	
1.00	Coarse sand	0.031	Medium silt
0.84		0.0156	Fine silt
0.71		0.0078	Very fine silt
0.59			
0.50	Medium sand	0.0039	Clay
0.42		0.0020	
0.35		0.00098	
0.30		0.00049	
		0.00024	
		0.00012	
		0.00006	

Source: Folk 1968

Table 3-2. Percent gravel, sand, silt, and clay, mean grain size, and total organic carbon (TOC) for each station.

Outfall	Station F=fine C=coarse	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Mean Grain Size (mm)	TOC (%)
3	3-1 C	0.845	93.8	2.49	2.84	0.181	0.592
	3-2 C	0.146	88.2	6.17	5.51	0.138	0.695
	3-3 C	0.393	84.1	8.44	7.09	0.111	0.584
	3-4 C	0.152	84.2	7.84	7.77	0.118	0.546
	3-5 C	0.067	88.5	5.36	6.04	0.136	0.325
4	4-1 C	0.307	98.3	0.558	0.878	0.186	0.186
	4-2 C	0.003	97.5	0.568	1.89	0.160	0.151
	4-3 C	0.194	66.8	18.5	14.4	0.033	1.02
	4-4 C	0.004	79.5	10.1	10.3	0.063	0.581
	4-5 C	0.007	85.7	6.51	7.77	0.103	0.441
5	5-1 C	0.579	98.0	0.547	0.861	0.278	0.672
	5-2 C	1.43	96.6	0.547	1.41	0.345	0.773
	5-3 C	4.72	93.3	0.542	1.40	0.425	1.46
	5-4 F	0.175	39.2	35.2	25.4	0.012	1.42
	5-5 C	0.044	78.8	9.91	11.2	0.060	0.555
6	6-1 C	0.035	88.4	7.27	4.35	0.114	0.552
	6-2 C	0.196	95.6	1.01	3.22	0.151	0.387
	6-3 C	0.029	61.5	21.4	17.1	0.021	0.915
	6-4 F	0	54.2	26.3	19.5	0.015	1.02
	6-5 F	0.010	54.4	25.4	20.1	0.017	1.08
7	7-1 C	5.97	92.7	0.531	0.836	0.357	0.396
	7-2 C	3.26	94.6	0.531	1.63	0.391	0.529
	7-3 C	7.02	75.5	8.95	8.54	0.126	0.577
	7-4 F	0.030	43.7	36.4	19.8	0.015	1.34
	7-5 F	0	46.4	27.5	26.0	0.011	1.10
8	8-1 C	19.1	79.3	0.521	1.08	0.987	0.787
	8-2 C	0.308	94.9	2.00	2.76	0.174	0.306
	8-3 C	0.468	94.6	1.43	3.47	0.157	0.349
	8-4 C	0.044	77.9	11.3	10.8	0.059	0.543
	8-5 F	0	26.6	39.7	33.6	0.009	1.40
Reference Stations	R-1 C	0.370	83.9	9.69	6.03	0.120	0.293
	R-2 C	0.156	76.4	14.8	8.63	0.103	0.582
	R-3 C	0.078	81.4	11.5	7.03	0.120	0.433
	R-4 C	0.315	65.0	17.0	17.6	0.021	0.645
	R-5 C	0.409	66.8	13.6	19.2	0.021	0.701
	R-6 F	0.004	40.9	34.4	24.7	0.016	1.17
	R-7 F	0.360	37.3	36.1	26.3	0.013	1.60
	R-8 F	0.026	23.1	45.9	30.9	0.008	1.35
	R-14 F	0.111	50.7	28.4	20.7	0.016	0.979
	R-15 F	0.043	46.0	30.9	23.0	0.016	1.16

3.2 Total Organic Carbon

Total organic carbon concentrations provide an indication of the amount of organic matter present in bottom sediments. Concentrations of TOC for all in-bay surface sediments are shown in Table 3-2. Concentrations of TOC ranged from 0.151 to 1.46% (mean = 0.709%) in outfall sediments and from 0.293 to 1.60% (mean = 0.891%) in reference sediments. As expected, higher (> 0.9%) TOC concentrations were observed in finer-grained sediments as discussed below.

3.3 Designation of Sediment Groups

In-bay surface sediments were grouped according to grain size based on anticipated relationships between grain size and sediment contaminant concentrations as discussed in Section 2.2. These station groups are maintained in subsequent analyses of chemical and biological parameters (Sections 4 and 5, respectively). Since TOC and grain size were highly correlated (r^2=0.89) for the reference stations, the single parameter grain size, expressed as percent fine sediment (sum of silt+clay fractions), was used to designate station groups. Figure 3-1 plots the relationship between TOC and percent fines for in-bay reference stations and shows the two reference station groups based on fine sediment fraction. The midpoint between the means of the two groups was 42.5% fines. Five reference stations had sediments with greater than 42.5% fines, and five had less than 42.5% fines.

At stations R-1 through R-5, sand ranged from 66.8 to 83.9%, and mean grain size was 0.021 (medium silt) to 0.120 mm (very fine sand). These stations comprise the “coarse-grained” reference stations. At stations R-6, R-7, R-8, R-14, and R-15, sand ranged from 23.1 to 50.7%, and mean grain size was 0.008 (fine silt) to 0.016 mm (medium silt). These stations comprise the “fine-grained” reference stations. Bray Curtis analysis (Bray and Curtis 1957), a data clustering technique, also separated the reference stations into the same two distinct groups, using percent silt + clay (fines).

The 30 in-bay outfall stations were separated into two corresponding coarse- and fine-grained groups using the 42.5% (fines) midpoint established for the reference stations (Figure 3-2). Station identification and percent fines are shown in Table 3-3 for the resulting 24 coarse-grained stations and 6 fine-grained stations. Subsequent comparisons between outfall and reference stations are made using these groups. Results for coarse-grain outfall stations are compared with results for coarse-grain reference stations and results for fine-grain outfall stations are compared with results for fine-grain reference stations. This approach minimizes contaminant and toxicity variability due to differences in sediment grain size properties. Locations for reference and outfall stations identified by grain-size group are shown in Figures 3-3 and 3-4, respectively.

Statistical t-tests comparing coarse or fine sediment outfall stations as a group with the corresponding reference station group showed no significant differences ($p \leq 0.05$). Comparisons of TOC at individual stations and corresponding reference mean TOC values indicated significant enrichment of organic carbon only at stations 4-3 and 5-3, both in the coarse-grain outfall group. Station 5-3 is unusual in that it had coarse sediment (93.4% sand; 1.9% fines) and relatively high TOC (1.46%). High TOC at this station was due to inorganic carbon bound up by the many shell fragments, as well as contributions from the rich carpet of eelgrass growing in the area. Station 4-3 also had organic carbon input from eelgrass.

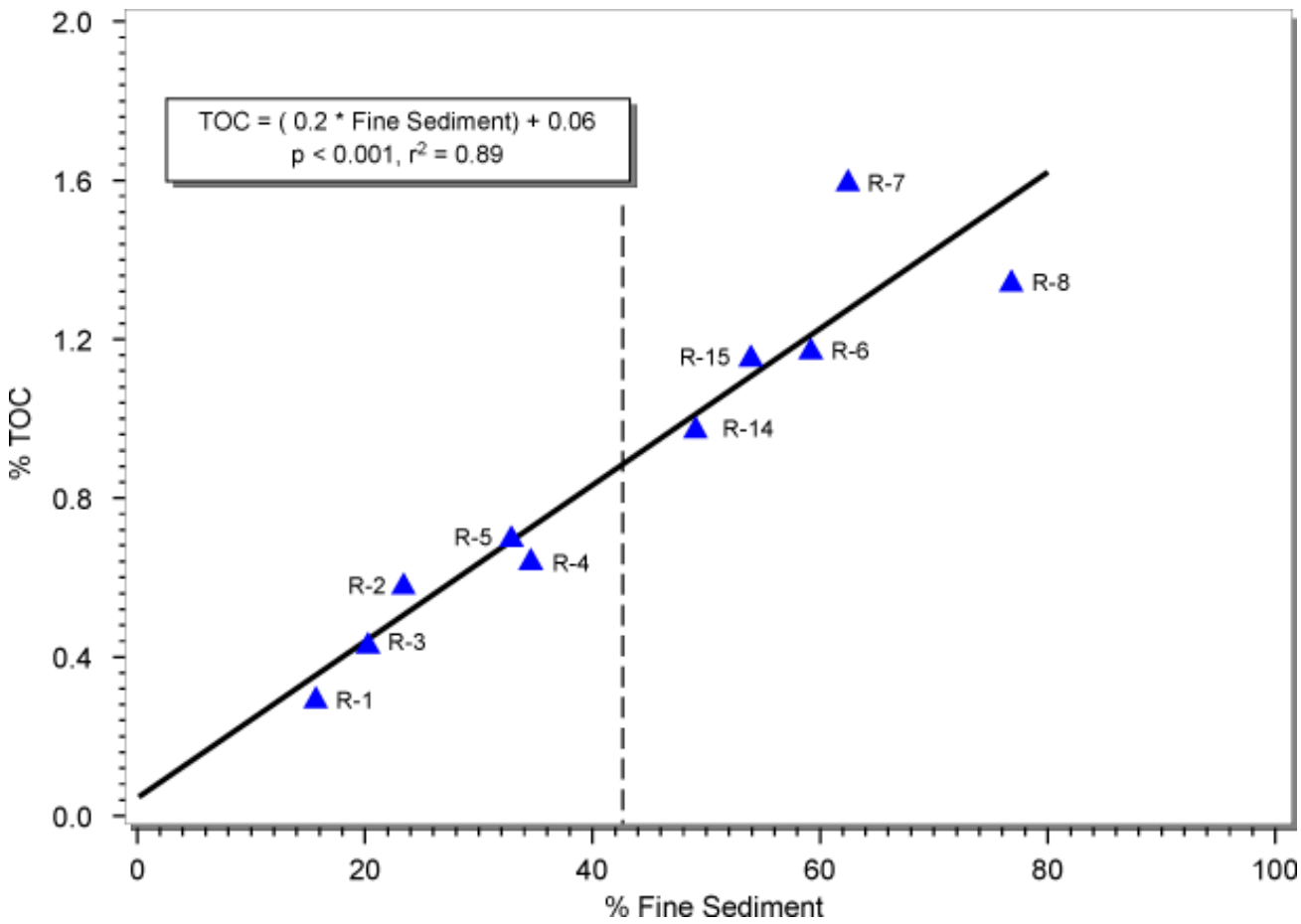


Figure 3-1. Regression of percent total organic carbon with percent fine (<62 μm or <0.062 mm) sediment.

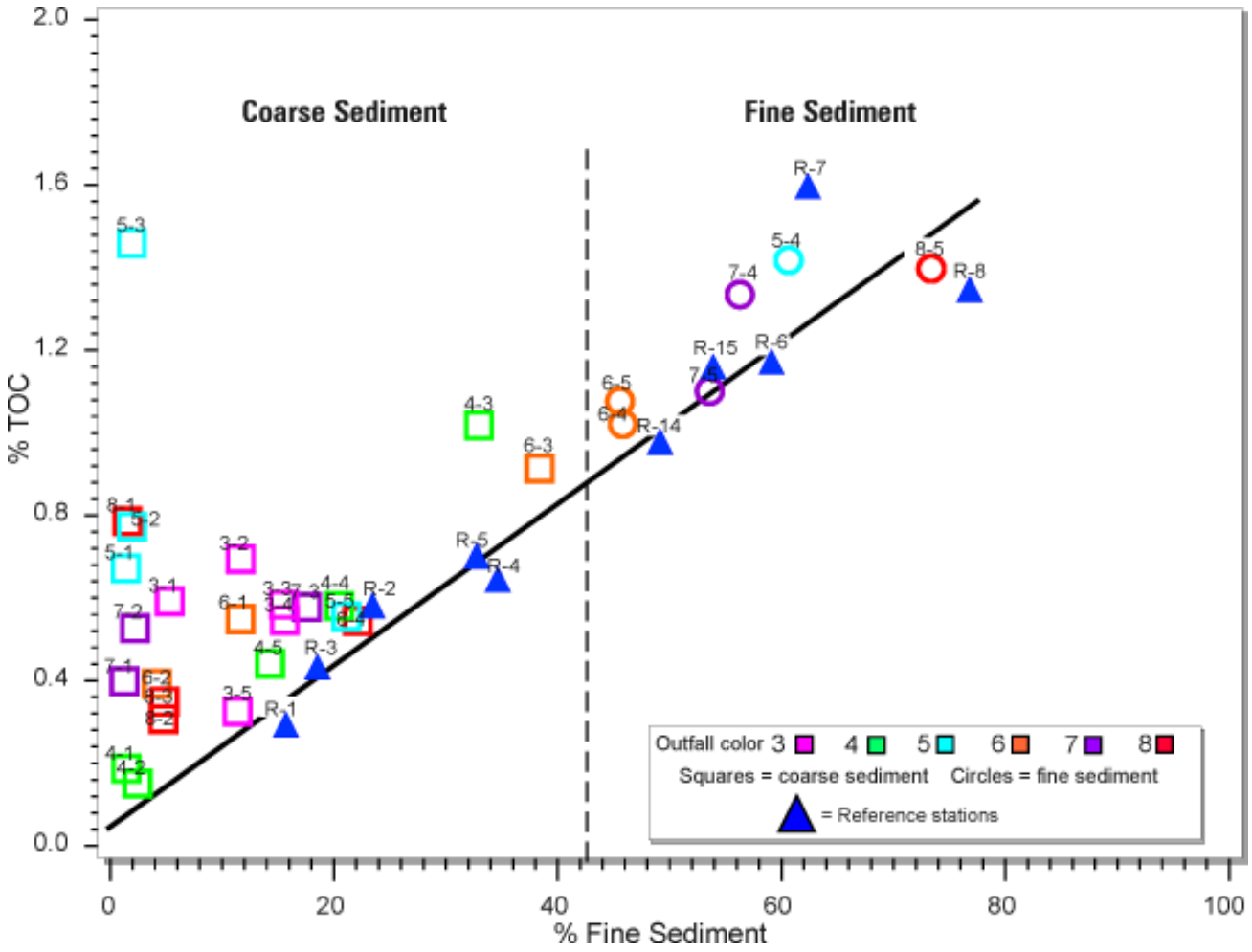


Figure 3-2. Percent total organic carbon with percent fine (<62 μm or <0.062 mm) sediment by station.

Table 3-3. Designation of outfall and reference stations by percent fines.

Coarse-Grain Sediment Group

Reference Stations		% Fines
	R-1	15.71
	R-3	18.53
	R-2	23.44
	R-5	32.75
	R-4	34.65

Outfall Stations		% Fines		% Fines
	7-1	1.37	3-5	11.40
	5-1	1.41	6-1	11.61
	4-1	1.44	3-2	11.68
	8-1	1.60	4-5	14.27
	5-3	1.94	3-3	15.52
	5-2	1.96	3-4	15.61
	7-2	2.16	7-3	17.50
	4-2	2.46	4-4	20.48
	6-2	4.24	5-5	21.13
	8-2	4.77	8-4	22.10
	8-3	4.90	4-3	32.97
	3-1	5.33	6-3	38.45

Fine-Grain Sediment Group

Reference Stations		% Fines
	R-14	49.16
	R-15	53.91
	R-6	59.11
	R-7	62.37
	R-8	76.83

Outfall Stations		% Fines
	6-5	45.56
	6-4	45.82
	7-5	53.58
	7-4	56.28
	5-4	60.58
	8-5	73.38

Figure 3-3. Locations of coarse-grained and fine-grained reference stations in San Diego Bay.

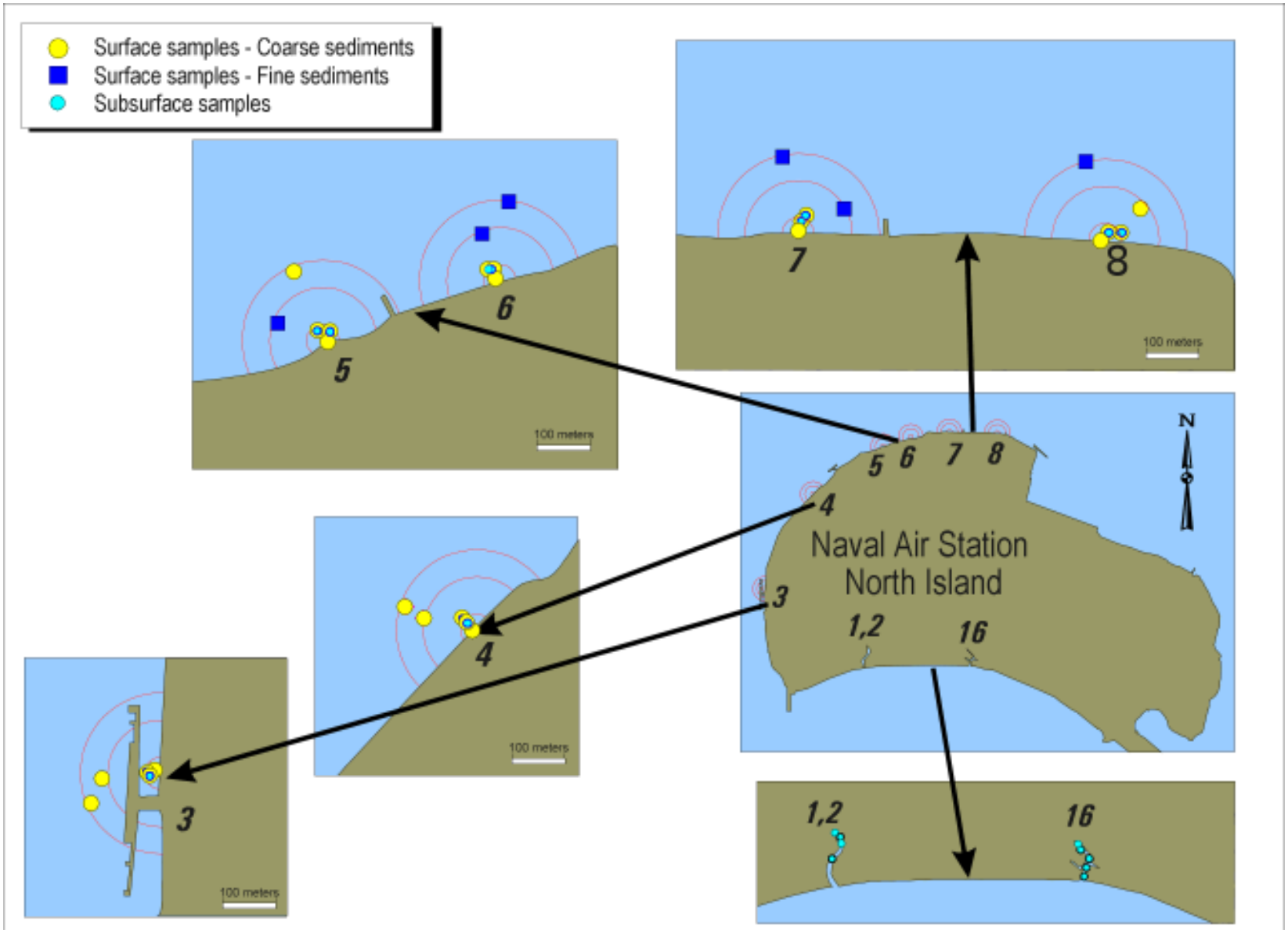


Figure 3-4. NAS North Island sampling stations identified by outfall and sediment type group.

The following sections compare surface (Section 4.1) and subsurface (Section 4.2) sediment chemical contamination at Site 1 for grouped and individual stations to comparable reference sediment concentrations, national guidelines and, in the case of metals, to expected crustal abundances. Results are organized into contaminant suites, consisting of polycyclic aromatic hydrocarbons (PAHs), organochlorine pesticides, polychlorinated biphenyls (PCBs), metals, butyltins, cyanide, and sulfide. Significantly higher outfall concentrations of the contaminant suites as well as individual contaminants are reported. Horizontal (Section 4.1) and vertical (Section 4.2) distributions of individual contaminants that were detected at significantly higher concentrations at several individual stations or core intervals are presented graphically to reveal local "hot spots". Site 1 chemicals of concern are identified and discussed in Section 4.3. All chemical concentrations are reported in dry weight. Tabular summaries of chemical data are presented in Appendix E. Statistical outputs are contained in Volume II. Summaries of quality control results and an external data validation report are presented in Volume III.3 and III.4, respectively.

4.1 Surface Sediments

Results for surface sediments collected at in-bay outfalls 3-8 are presented by individual suites of contaminants in Sections 4.1.1- 4.1.6. Some analytes in outfall sediments were not statistically compared to reference sediments because they were detected in fewer than three of the coarse and/or fine reference station groups identified in Section 3. These analytes consisted primarily of alkylated PAHs, phthalate, phenol, a few pesticides and PCB congeners, sulfides, and selenium, and are discussed separately in Section 4.1.7.

4.1.1 Polycyclic Aromatic Hydrocarbons (PAHs)

Concentrations of low molecular weight PAHs, consisting of all 2- and 3-ring compounds, and high molecular weight compounds, consisting of all PAHs with greater than three rings, were calculated. Mean concentrations for summed low and high molecular weight PAHs, including the alkylated homologues, and total PAH_{16} (summed concentrations of the 16 EPA priority pollutant PAHs) are shown in Table 4-1. The results of these tests were not significantly different ($p \leq 0.01$) for coarse or fine sediment groups for any PAH compound. Elevated PAH concentrations in the coarse sediment outfall groups were mainly attributable to outfall 3 near Pier Bravo (Appendix E3). When comparing concentrations of total PAHs among studies, one must be aware that the types and numbers of compounds may differ in the respective estimates. For example, the "effects range-low" (ER-L) and "effects range median" (ER-M) from Long et al. (1995) are reported for total PAHs using the sum of only 13 compounds. Since 41 PAHs were measured in this study, sums for low and high molecular weight PAHs include a greater number of analytes compared to most other studies.

Individual stations also were compared to the corresponding reference group mean based on grain size. Results of 1230 comparisons (41 analytes x 30 outfall stations) produced only one significantly ($p \leq 0.01$) elevated PAH compound, ideno[1,2,3,-c,d]pyrene, for station 3-2. This high molecular weight PAH was measured at 810 ng/g compared to the coarse reference mean of 19.5 ng/g. PAH_{16} was 15,362 ng/g for this station compared with total PAH (13 compounds) ER-L and ER-M values of 4022 and 44,792 ng/g, respectively. Although all Site 1 surface sediments had detectable concentrations (≥ 2 ng/g) for nearly all PAHs measured, neither individual nor

summed PAHs exceeded ER-M values for any station; and most stations had concentrations below the corresponding ER-Ls.

Table 4-1. Mean concentrations (± std error) of summed low and high molecular weight PAHs in surface sediments. Concentrations are ng/g dry weight.

	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	24	5	6	5
LMW PAH	353 ± 153	107 ± 64.4	596 ± 126	449 ± 94.8
HMW PAH	1484 ± 704	586 ± 249	2412 ± 383	2696 ± 544
PAH_{16}	1266 ± 627	465 ± 180	1841 ± 306	2048 ± 403

There were no significant differences ($p \leq 0.01$) between outfall and corresponding reference results.
n=number of stations sampled (across); Non-detect values were excluded from mean calculations

Station 3-2 was located within three meters of creosote soaked pier pilings and had the highest PAH concentrations of any of the in-bay surface sediments. Polycyclic aromatic hydrocarbons from sediment collected at station 3-2 were compared to a creosote standard. The relatively low concentrations of dibenzothiophenes indicate that there is only a minor petroleum component present (see Douglas et al. 1992). The most likely source of PAH is creosote (a coal tar) from pier pilings, and not the outfall directly. This is evidenced in the relative distributions of phenanthrene, C1-phenanthrene, and C2-phenanthrene, as well as the relative distributions of fluoranthene and pyrene, rather than a direct comparison of PAH distributions for the two samples (see Appendix E3). Molecular fingerprint plots for nearly all other surface stations, including the reference stations, displayed similar patterns. However, nearly all of these stations had high relative concentrations of benzo[b]fluoranthene compared to other PAHs, indicating primary inputs from combustion particulates (atmospheric dust), rather than creosote.

4.1.1.1 Potential PAH Sources to Surface and Subsurface Sediments

PAHs in surface and subsurface sediments were evaluated using principal component analysis (PCA) (Hair et al. 1987). This multivariate statistical technique has been used to identify weathered petroleum and combustion PAH sources in marine sediments (Kennicutt et al. 1994; Maxon et al. 1997), and is not detailed here. Principal component analysis was used to provide insight to potential PAH sources for Site 1 sediments. It is a useful technique as it 1) removes investigator bias, 2) allows simultaneous evaluation of a large number of samples with multiple analytes, and 3) supports classification of samples into related groups. Log transformed PAH concentrations of surface and subsurface samples were analyzed together in the PCA, using Sirius™ (Version 1.2).

Principal component analysis produces three different types of useful information: 1) the variance explained for each of the primary principal components; 2) loadings (i.e., correlation coefficients) for each principal component vector, which provide insight to the petroleum-related source or type; and 3) principal component scores, indicating the relative strength or magnitude of each principal component for each sample (or station).

Figure 4-1 plots the loadings for the first two principal components, which explain 67.1% of the total PAH variance. This plot displays four potential sources of hydrocarbon inputs to Site 1 sediments, shown as vectors along groups of prominent PAH compounds. (Names of PAH compounds are abbreviated in Figure 4-1 for display; corresponding names for each abbreviation are shown in Section 2, Table 2-4.) These sources include

diesel fuel, heavy oils (e.g., lubricating oil, Bunker C), road runoff associated with atmospheric deposition of combusted fuels, and creosote contamination most likely from current or historical creosote-soaked pier pilings. Figure 4-2 plots the scores for principal components 1 and 2 for each station. Two pyrogenic standards, atmospheric dust and coal tar, as well as a degraded diesel standard also are plotted for reference. Standards were obtained from the National Institute of Standards and Technology, and analyzed with Site 1 data.

Diesel contaminated inputs appear in the lower left quadrant of Figure 4-1, and are characterized by unique distributions of 2-3 ring PAHs, including naphthalenes (N, N1-N4), fluorenes (F, F1-F3) and light dibenzothiophenes (D, D1). Corresponding stations with diesel inputs are shown in Figure 4-2, with nearly all contamination observed at outfalls 1,2 and 16. The dominance of C3- and C4-naphthalenes (N3 and N4) indicates degraded (weathered) diesel fuel contaminated sediments observed for stations 1-1 (surface and subsurface), 1-3 (surface) and 16-1 at the 5-6-ft depth interval (16-16 in Figure 4-2).

Input from heavy oil is characteristically driven by C2-, C3-, and C4-chrysenes, which readily separated in the PCA (Figure 4-1). Sediment stations contaminated by this petroleum product are located in the upper central region of Figure 4-2, and include primarily the 3-5 ft. depths at stations 4-2 and 4-3, located near Pier Bravo.

Road runoff is dominated by high molecular weight PAHs derived from atmospheric deposition of combustion particles associated with oil-fueled electric plants and internal combustion engines. Extensive studies of atmospheric dust and urban road runoff indicate that benzo[b]fluoranthene (BB) dominance combined with other 5-6 ring PAHs (e.g., benzo[a]anthracene [BAP]) are strong indicators of inputs from road runoff (Douglas et al. 1992). Most of the in-bay stations, including reference sites received PAH inputs from road runoff as indicated in Figure 4-2. Many of these had additional inputs of coal tar or creosote which also could have entered the bay through road runoff.

Creosote is dominated by combustion related PAHs derived primarily from the high temperature carbonization of coal. This process produces a PAH distribution that is dominated by phenanthrene (P), pyrene (PY), and fluoranthene (FL). Principal component analysis showed significant inputs of creosote at station 3-3, which is reasonable since this station is located only a few meters from creosote soaked pier pilings. Sediment stations that are plotted between source functions in Figure 4-2 are mixtures of the respective sources.

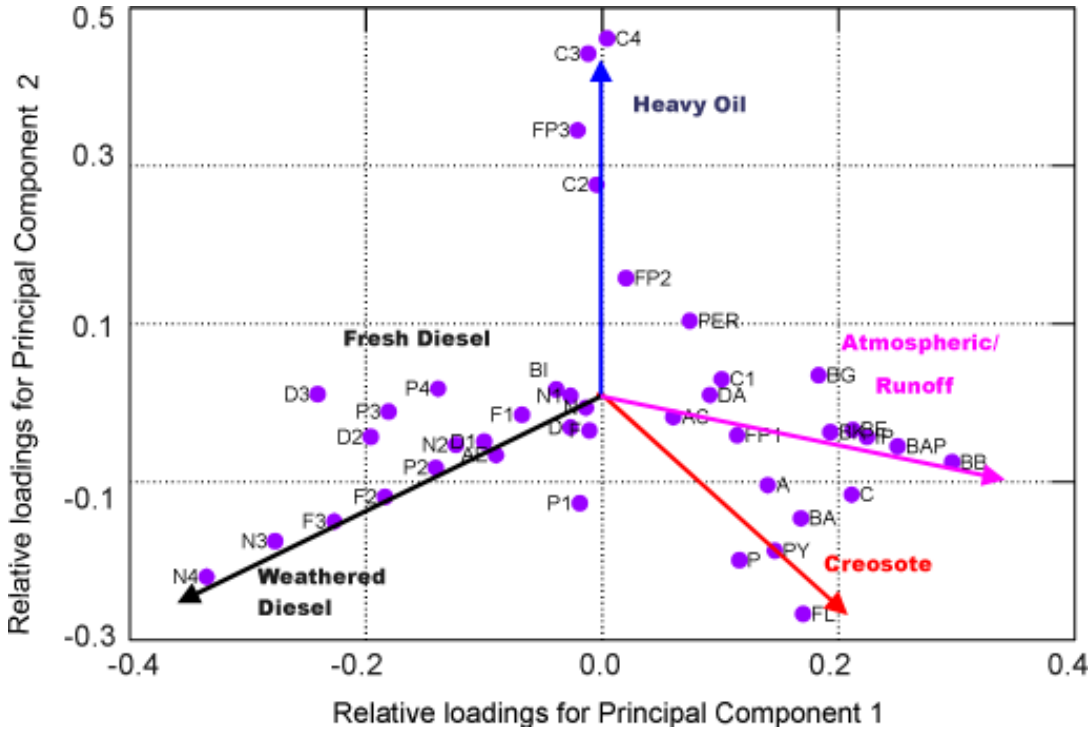


Figure 4-1. Relative loadings for principal components 1 and 2.

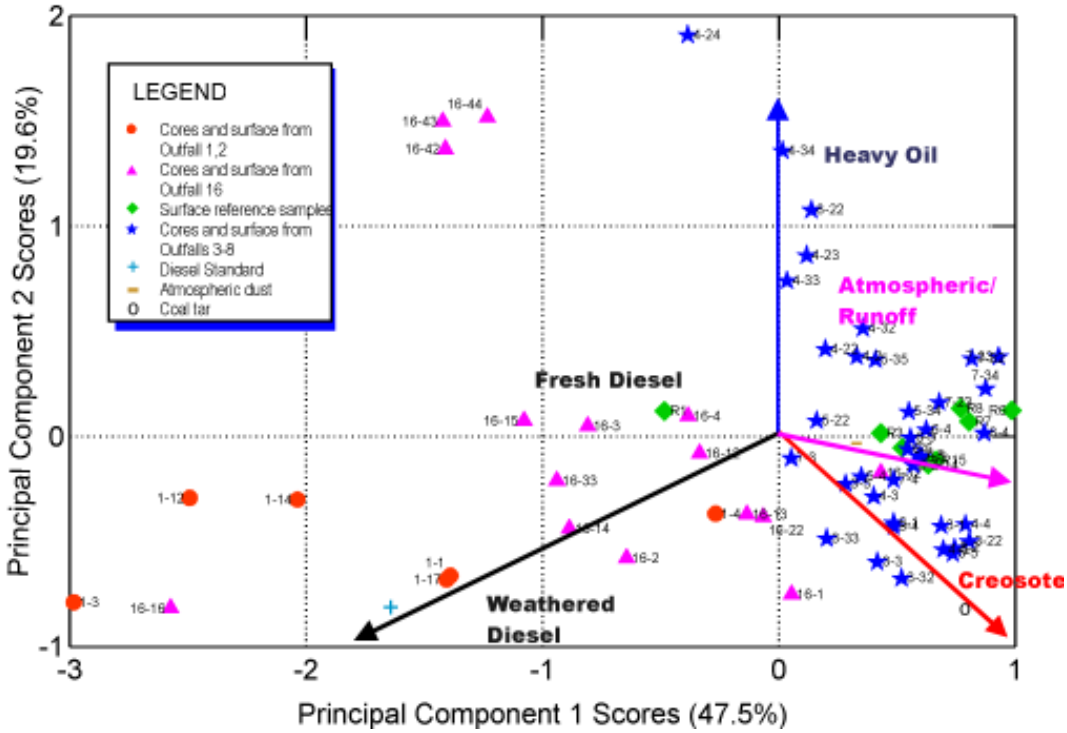


Figure 4-2. Station scores for principal components 1 and 2.

4.1.2 Organochlorine Pesticides

Of the 18 pesticides measured, six were not detected in any of the outfall surface sediments (i.e., < 1 ng/g). These compounds were 2,4'-DDE, aldrin, endosulfan I, endrin, endrin aldehyde, and endosulfan sulfate. Of the remaining 12 pesticides, four could not be statistically compared in the coarse and/or fine sediment groups due to insufficient detectable concentrations in the reference groups. These included isomers of DDT and DDD, dieldrin, and endrin ketone, and are discussed separately in Section 4.1.7.

Comparisons were made between the fine sediment outfall and reference station groups for 2,4'-DDD, 4-4'-DDD, 4,4'-DDE, 2,4'-DDT, total DDT (sum of the 2,4'- and 4,4'- isomers of DDT, DDE, and DDD), endosulfan II, and hexachlorobenzene (HCB). Grouped comparisons were made between the coarse sediment outfall and reference stations for 4,4'-DDE, total DDT, endosulfan II, endrin ketone, and HCB. Mean concentrations and standard errors for total DDT, total BHC, total chlordane, and HCB for coarse and fine sediment groups are shown in Table 4-2. There were no statistical differences ($p \leq 0.01$) between the outfall stations and corresponding reference stations grouped by grain size for any individual analyte compared. Only total DDT was statistically higher in the grouped coarse stations at the $p \leq 0.01$ level (Figure 4-3).

Table 4-2. Mean concentrations (± std error) of select chlorinated pesticides and PCBs in surface outfall and reference stations. Concentrations are ng/g dry weight.

	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	24	5	6	5
Total DDT	2.21 ± 0.37**	0.70 ± 0.09	5.92 ± 1.10	8.15 ± 2.26
Total BHC	0.27 ± 0.15	0.39 ± 0.10	0.47 ± 0.21	1.54 ± 0.76
Total Chlordane	0.38 ± 0.05	0.26 ± 0.13	0.70 ± 0.15	2.88 ± 1.04
HCB	3.64 ± 1.83*	0.05 ± 0.03	2.25 ± 2.15	0.04 ± 0.01
Total PCB	13.4 ± 1.84**	24.2 ± 15.7	42.5 ± 6.79	63.0 ± 17.2

Reference and corresponding outfall results are significantly different at $p \leq 0.05$(*), at $p \leq 0.01$(**)
n=number of stations (across); Non-detect values were excluded from mean calculations

Total DDT concentrations at eight individual coarse outfall stations were statistically ($p \leq 0.01$) higher than at the coarse reference stations (Figure 4-4). The surface distribution of total DDT is shown in Figure 4-6. Total DDT concentrations for individual outfall stations ranged from 1.5-6.5 ng/g, compared to an ER-L value of 1.6 ng/g and ER-M value of 46.1 ng/g. The primary DDT metabolites were 4,4'-DDE and 4,4'-DDD. However, station 3-2 contained a higher proportion of 4,4'-DDT (2.7 ng/g), suggesting trace levels of a less-degraded, fresher source of technical grade DDT. Maximum concentrations of individual isomers typically were less than 5 ng/g, and were representative of some of the least impacted sediments in southern California (SCCWRP 1996).

Concentrations of HCB at five coarse and three fine outfall stations were statistically higher than corresponding reference concentrations (Figure 4-4). The spatial distribution of HCB is shown in Figure 4-6. HCB was detected in 80-100% of sediments from individual outfall and reference stations. The overall mean concentrations were low even though coarse and fine sediments from the outfalls were almost two orders of magnitude higher than those for reference stations (Table 4-2). Concentrations at outfall stations ranged from 0.03-34 ng/g. Confidence intervals are shown for 95% and 99% one-tail tests compared to 0.13 ng/g for the reference sites

(Table 4-2). Although HCB is primarily used as a fungicide, it is also a breakdown product from pesticides containing chlorinated benzenes (i.e., alpha-BHC, and DDT). Recent toxicity studies using sediments spiked with HCB showed no toxic effects at concentrations up to 200 μγ/g (or 200,000 ng/g) in tests using freshwater midge fly larvae and freshwater amphipods (Barber et al. 1997). There are no sediment ER-L or ER-M values for HCB.

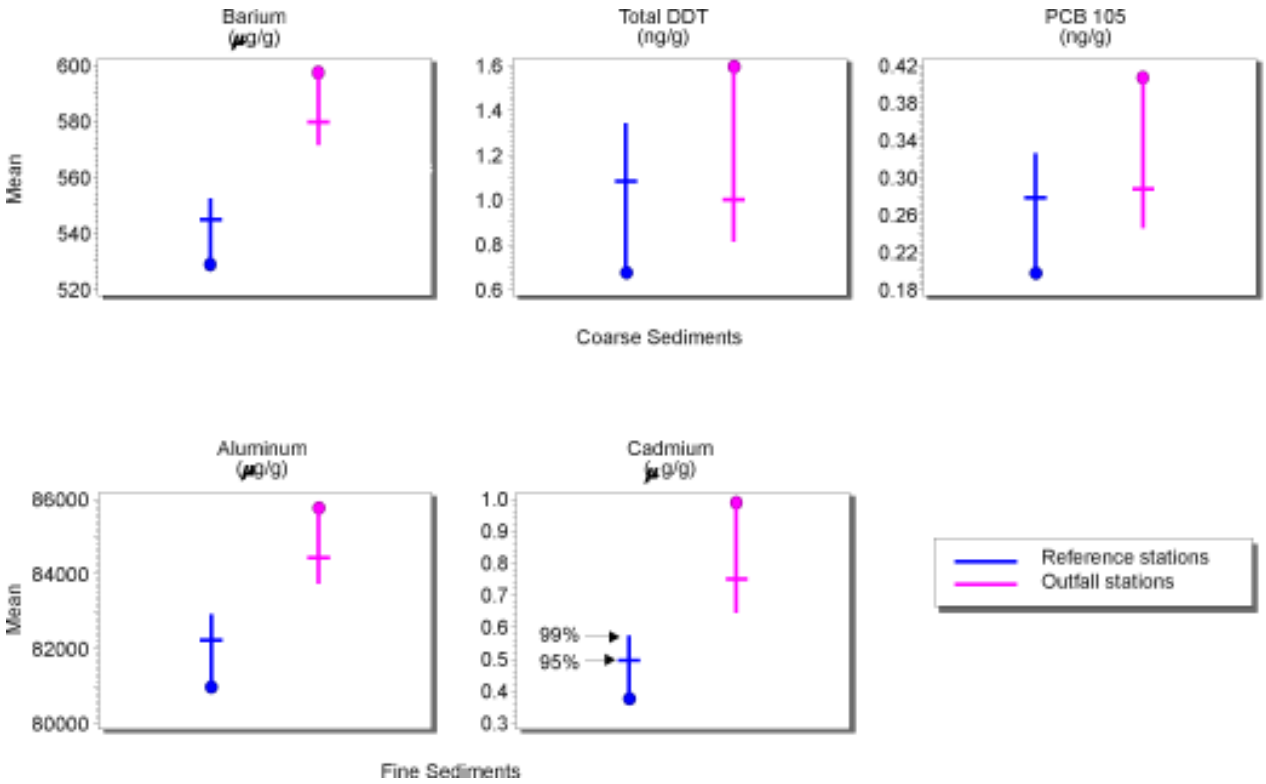


Figure 4-3. Significant results for t-tests on contaminant concentrations in surface sediments.

4.1.3 Polychlorinated Biphenyls (PCBs)

Only PCB congener 105 had a significantly higher mean concentration (0.49 ng/g) in coarse outfall sediments compared to reference sediments (0.20 ng/g) (Figure 4-2). Individual station comparisons showed statistically elevated concentrations of PCB 105 at 12 outfall stations, of PCB 101 at stations 3-3, 7-3, and 8-3, and PCB 118 at station 8-3; however, all concentrations were below 10 ng/g (Figure 4-4). In general, PCB concentrations in surface sediments were relatively low and uniform throughout the outfall stations. The horizontal distribution of PCB 105 is shown in Figure 4-6. Mean concentrations of total PCBs (i.e., sum of 18 congeners) in the coarse and fine outfall groups were 13.4 and 42.5 ng/g, respectively, compared to concentrations of 24.2 and 63.0 ng/g, respectively, for the corresponding reference groups (Table 4-2). Effect levels from Long et al. (1995) exist only for total PCBs, which are primarily based on Aroclor concentrations. NOAA's National Status and Trends Program defines total PCB concentrations as approximately two times the sum of the 18 congeners to account for possible contributions from the other (unmeasured) congeners. The maximum total PCB (based on 18 congeners) concentration for all outfall stations was only 72 ng/g compared with an ER-M value of 180 ng/g, indicating little potential impact from PCBs.

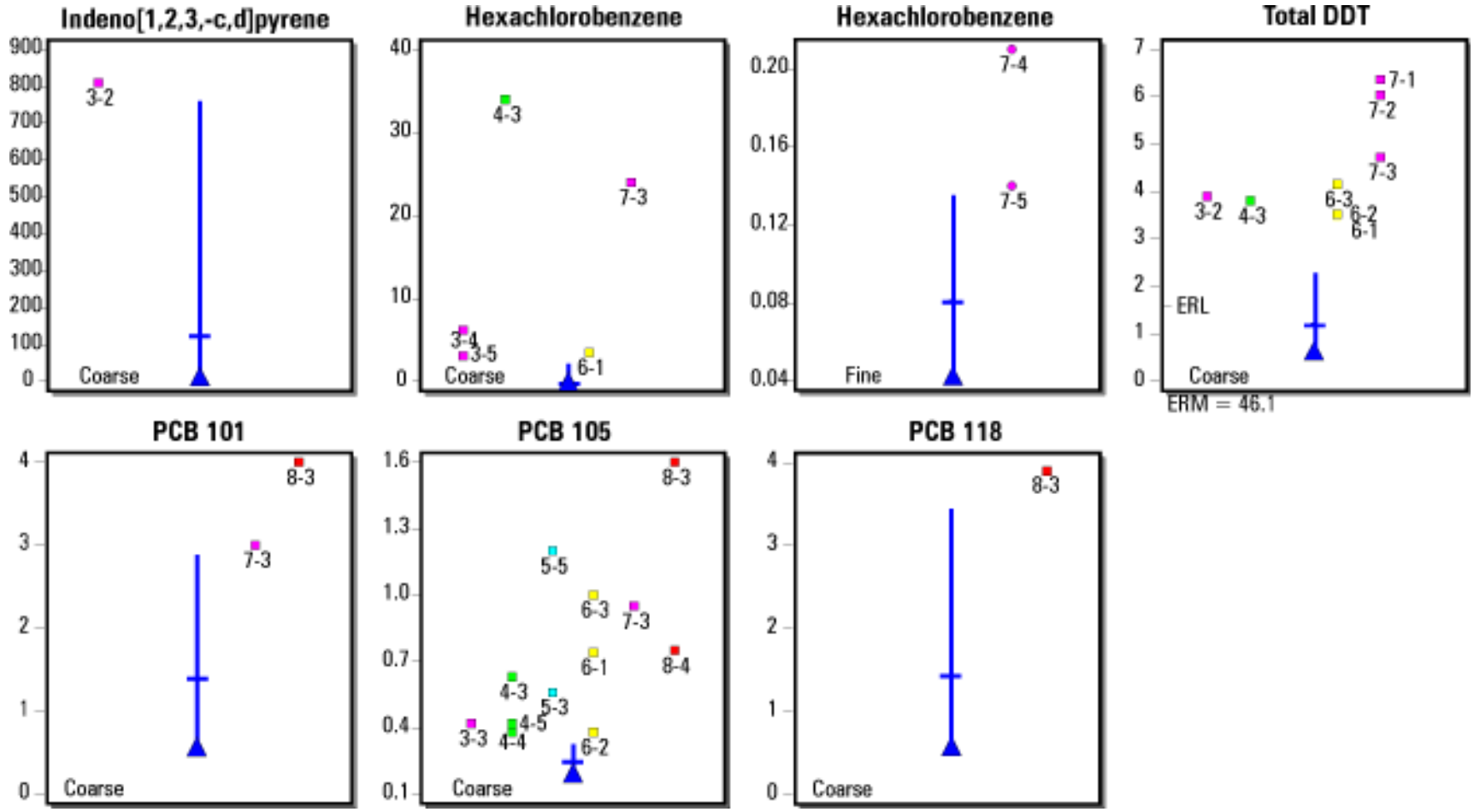


Figure 4-4. Organic chemical concentrations (ng/g) for outfall surface sediments above the 99% predictive limit of corresponding reference stations.

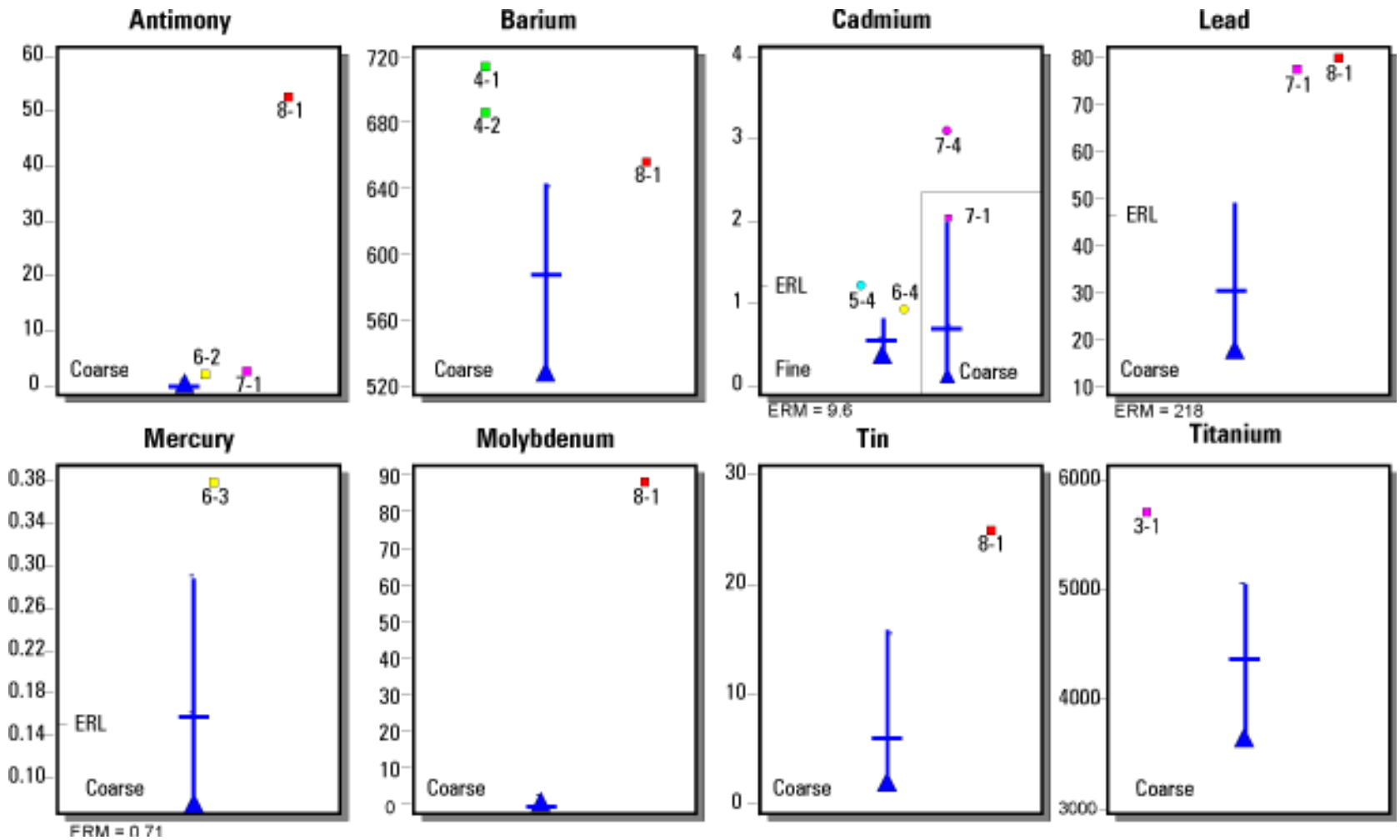


Figure 4-5. Metal concentrations (μg/g) for outfall surface sediments above the 99% predictive limit of corresponding reference stations.

Distance from Outfall	Cadmium	OXtlD03	OXtlD04	OXtlD05	OXtlD06	OXtlD07	OXtlD08
	3 P	0.07	0.06	0.33	0.34	0.98	0.19
	10 P	0.15	0.09	0.43	0.2	0.09	0.06
	30 P	0.15	0.25	0.11	0.58	0.33	0.07
	100 P	0.1	0.21	1.31	1	3.33	0.21
	150 P	0.08	0.2	0.23	0.57	0.63	0.91

Distance from Outfall	Total DDT	OXtlD03	OXtlD04	OXtlD05	OXtlD06	OXtlD07	OXtlD08
	3 P	0.04	0.19	0.42	1.39	2.54	0.17
	10 P	1.56	0.38	0.47	1.41	2.41	0.25
	30 P	0.52	1.52	0.67	1.66	1.89	0.12
	100 P	0.57	0.49	0.06	0.05	0.1	0.93
	150 P	0.62	0.5	0.6	0.03	0.04	0.05

Distance from Outfall	PCB 105	OXtlD03	OXtlD04	OXtlD05	OXtlD06	OXtlD07	OXtlD08
	3 P	0.61	0.45	0.64	2.07	0.42	0.31
	10 P	0	0.78	0.89	1.06	0.45	0.61
	30 P	1.17	1.76	1.56	2.79	2.65	4.47
	100 P	0.64	1.06	0.02	0.02	0.03	2.09
	150 P	0.42	1.17	3.35	0.01	0.01	0.02

Distance from Outfall	Hexachlorobenzene	OXtlD03	OXtlD04	OXtlD05	OXtlD06	OXtlD07	OXtlD08
	3 P	ND	0	0.01	1.07	0.01	0.01
	10 P	ND	0.01	0.01	0.01	0.16	0
	30 P	ND	10.38	0.01	0.02	7.32	ND
	100 P	1.89	0.01	0.77	0.32	1.48	0.01
	150 P	0.95	0.01	0	0.3	0.99	0.54

RDtLR

0 - 1	>1 - 2	>2 - 3	>3 - 4	>4 - 5	>5 - 10	>10

Figure 4-6. Ratios of in-bay surface concentrations to 99% reference predictive limit.

Concentrations of the lower and higher chlorinated biphenyls (e.g., di-, tri-, octa-, nona-, and deca-chlorobiphenyls; PCB congeners # 8, 18, 28, 195, 206, 209) typically were <1 ng/g; whereas concentrations of the penta-, hexa- and hepta-chlorobiphenyls (congeners # 101, 138, 153, 170, and 180) generally were 1-5 ng/g. Maximum concentrations of individual congeners typically were <10 ng/g. The relative abundances of these congeners suggest that mixtures of Aroclors 1254 and 1260 were the primary contributors to sediment PCBs. The relatively consistent proportions of individual congeners among outfall stations suggest that either different sources of PCB were well-mixed throughout the bay or similar sources contributed to the concentrations measured.

Seven of the 18 congeners were not statistically compared in the coarse and/or fine sediment groups due to insufficient detectable concentrations in the reference groups. These were primarily the more rapidly degrading lighter chlorinated congeners, consisting of PCB 8, 18, 28, 44, and 66. Of these, PCB 8 was detected in one sample each from the outfall and reference groups. The remaining compounds, although detected at appreciable frequencies at the outfall stations, were at low concentrations, with all congeners < 5 ng/g; and are therefore not considered important contaminants in Site 1 sediments.

Table 4-3. Mean metal concentrations (± std error) in surface sediments at outfall and reference stations. Concentrations are μg/g dry weight.

	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	24	5	6	5
Aluminum	71,696 ± 2493	75,020 ± 2471	85,767 ± 685	81,000 ± 1299
Antimony	2.97 ± 2.16	0.49 ± 0.08	1.35 ± 0.30	0.92 ± 0.13
Arsenic	4.64 ± 0.60	5.10 ± 0.83	8.35 ± 0.46	8.42 ± 0.96
Barium	599 ± 9.39**	529 ± 11.5	560 ± 6.82	558 ± 13.1
Beryllium	0.71 ± 0.03	0.71 ± 0.06	0.88 ± 0.03	0.88 ± 0.06
Cadmium	0.48 ± 0.09	0.28 ± 0.05	1.20 ± 0.39**	0.38 ± 0.04
Chromium	35.3 ± 3.65	35.4 ± 5.24	74.0 ± 5.69	76.1 ± 9.91
Cobalt	5.85 ± 0.55	7.64 ± 1.32	10.5 ± 0.22	10.2 ± 0.62
Copper	49.1 ± 11.3	36.2 ± 7.81	84.9 ± 4.67	104 ± 12.8
Iron	18,500 ± 1336	24,800 ± 3177	38,750 ± 1066	38,700 ± 2325
Lead	27.0 ± 3.74	18.1 ± 2.00	43.8 ± 3.66	61.1 ± 10.5
Manganese	326 ± 19.4	401 ± 36.6	512 ± 6.24	485 ± 9.64
Mercury	0.09 ± 0.02	0.08 ± 0.01	0.49 ± 0.03	0.56 ± 0.11
Molybdenum	4.51 ± 3.64	0.75 ± 0.14	1.30 ± 0.09	1.36 ± 0.15
Nickel	7.76 ± 0.70	9.30 ± 1.57	17.4 ± 0.65	17.0 ± 1.60
Selenium	0.21 ± 0.0	ND	0.24	0.24 ± 0.04
Silver	0.27 ± 0.05	0.30 ± 0.07	0.79 ± 0.07	0.91 ± 0.20
Thallium	0.36 ± 0.02	0.37 ± 0.05	0.53 ± 0.02	0.54 ± 0.05
Tin	3.74 ± 1.03	2.23 ± 0.53	5.87 ± 0.30	7.19 ± 1.20
Titanium	2787 ± 244	3616 ± 134	4935 ± 153	5434 ± 275
Vanadium	48.6 ± 3.68	67.1 ± 6.70	104 ± 2.94	101 ± 6.37
Zinc	88.9 ± 8.41	114 ± 24.5	204 ± 12.3	198 ± 18.8

Reference and corresponding outfall results are significantly different at $p \leq 0.01$(**)
ND = not detected; n=number of stations (across); Non-detect values were excluded from mean calculations

4.1.4 Metals

Results for grouped comparisons of all metals analyzed are shown in Table 4-4. Mean concentrations were elevated in outfall sediments for only two metals: barium in the coarse sediment group, and cadmium in the fine sediment group. Mean concentrations of these metals were 599 and 1.2 μg/g for the outfalls compared with 529 and 0.38 μg/g for the reference groups, respectively. Although no ER-L and ER-M values for barium have been defined, barium in sediments has a very low toxicity, and does not bioaccumulate in marine organisms (e.g., Jenkins et al. 1989). The average concentration of barium in continental crust is 600 μg/g, and concentrations less than this are assumed to be non-toxic. Also, Bradford et al. (1996) reported an average barium concentration of 509 μg/g from 50 unimpacted California soil samples, with a range of 133 to 1,400 μg/g. Consequently, barium concentrations in outfall sediments are not expected to produce acute or chronic toxicity to San Diego Bay biota. Cadmium, on the other hand, is a pollutant of concern and can be acutely toxic to various marine biota if bioavailable above threshold concentrations. However, the ER-L for cadmium is 1.2 μg/g and the ER-M is 9.6 μg/g, indicating that concentrations measured in outfall sediments, which ranged from 0.12 to 3.1 μg/g, are likely not toxic.

Comparisons of individual stations to the corresponding reference station mean, showed significantly elevated ($p \leq 0.01$) concentrations of seven metals, excluding barium (Figure 4-5). These were antimony, cadmium, lead, mercury, molybdenum, tin, and titanium. The distribution of cadmium, which was statistically higher in five outfall stations, is shown in Figure 4-6. To determine anthropogenic contribution, the 95% predictive interval of the crustal metal-iron ratios for all North Island sediments (Wedepohl 1995) were calculated. This method indicated likely anthropogenic inputs of antimony, cadmium lead, molybdenum, and tin, confirming the reference station comparisons. Although elevated, neither cadmium, lead, nor mercury exceeded ER-M values in outfall sediments. Antimony at station 8-1 (55 μg/g) exceeded the ER-L and ER-M values (2 and 25 μg/g, respectively); however, no significant toxicity or bioaccumulation was observed at this station. Corresponding values do not exist for antimony, molybdenum, tin, or titanium.

Table 4-4. Mean concentrations of butyltins in surface sediments at outfall and reference stations. Concentrations are ng/g dry weight.

	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	24	5	5	5
Monobutyltin	5.89 ± 1.13	13.0 ± 3.27	26.4 ± 2.38	48.0 ± 7.07
Dibutyltin	6.67 ± 1.18	4.76 ± 0.88	26.0 ± 2.51	92.4 ± 27.4
Tributyltin	13.2 ± 2.56	10.1 ± 0.90	21.8 ± 1.91	49.0 ± 8.73
Total butyltins	23.9 ± 3.42	27.8 ± 3.86	74.2 ± 4.92	189 ± 40.2

There were no significant differences between outfall and reference groups at $p \leq 0.01$
n=number of stations (across); Non-detect values were excluded from mean calculations

4.1.5 Butyltins

There were no significant differences between grouped or individual outfall stations compared with the appropriate reference station group for any butyltin compounds. Concentrations of total organotins (i.e., sum of the mono-, di-, and tri- butyltins) in coarse grained sediments at the outfall and reference stations were comparable; whereas, concentrations in the fine grained sediment group of reference stations were more than two times higher than those at the outfall stations (Table 4-4). Further, the maximum total butyltin concentration at the outfall stations was 93 ng/g compared to a maximum concentration of 314 ng/g for the reference sites. For comparison, Wade et al. (1990) reported a mean total organotin concentration in sediments from the west coast National Status and Trends sites of 58 ng/g, with respective mean concentrations for monobutyltin, dibutyltin, and tributyltin of 6, 21, and 30 ng/g. Thus, butyltin concentrations in Site 1 sediments did not indicate significant enrichment or contamination when compared to bay-wide reference concentrations or average concentrations in sediments from other west coast sites.

4.1.6 Cyanide and Sulfides

Cyanide was not detected in any of the surface sediment samples from outfall or reference sites. Detection limits for all sediments ranged from 0.48 to 1.0 μg/g. There were no statistical differences between sulfide concentrations in the coarse sediment outfall and reference groups. There were only two detected concentrations of sulfides in the fine reference group, so fine outfall stations were not statistically compared. All concentrations were low (< 500 μg/g) for marine sediments, indicating probable natural sources of sulfides which are not felt to pose an ecological threat (USACOE 1991).

Like ammonia, the presence of sulfides in sediments is not generally considered an anthropogenic contaminant, but is a potentially confounding factor in the interpretation of sediment toxicity test results (Word et al. 1997). Elevated concentrations (> 200 μg/g) of total sulfides were found at several stations and were generally associated with elevated concentrations of interstitial ammonia. These values suggest areas of reduced circulation and high microbiological activity. High survival in the mysid, amphipod, and polychaete tests (see Section 5) indicate that the total sulfides measured may not be biologically available to the general benthic community. Data generated from pore water bioassays cannot be used to evaluate the toxic nature of sulfides due to the transient nature of hydrogen sulfide in seawater.

4.1.7 Chemicals Not Statistically Compared

As discussed above, a number of chemicals were not included in statistical comparisons due to insufficient detectable concentrations (i.e., < 3) in either the coarse or fine sediment reference group. These chemicals are shown in Table 4-5 with the number of detected values for the reference and outfall group, the mean reference group concentration, the mean outfall group concentration and standard error, the maximum outfall concentration, and the station where the maximum concentration was detected. Alkylated PAHs were measured for contaminant source identification purposes only, and are not included in Table 4-5. Also not included are cyanide and several pesticides that were not detected in any of the outfall sediments.

Table 4-5. Summary of analytes that were detected in outfall sediments but had less than 3 detectable values in the corresponding reference group.

Analyte	Grain size reference no. of samples	Mean reference concentration	OUTFALL Mean	OUTFALL Std. error	OUTFALL Maximum	Station of maximum conc.	Outfall group n
Inorganics							
Sulfides (µg/g)	f, 2	38.3	60.5	21.1	122.0	5-4	5
Selenium (µg/g)	c, 0; f, 2	nd; 0.24	0.21; 0.24	0.0; na	0.22; 0.24	4-3; 7-4	5; 1
Tributyltin (ng/g)	c, 0	nd	50.4	na	50.4	7-1	1
Semivolatile Organics (ng/g)							
Naphthalene	c, 2	2.2	5.1	2.0	23	6-1	11
Acenaphthene	c, 2	3.0	7.5	3.3	51	3-2	19
Biphenyl	c, 2	1.0	1.3	0.4	4.4	6-1	12
Phthalate	c, 1	460	270	69	390	6-1	3
Phenol	c, 2; f, 2	85; 152	58.8; 97.5	11.6; 22.5	130; 120	3-3; 7-4	10; 2
Pesticides and PCBs (ng/g)							
Dieldrin	c, 0; f, 2	nd; 0.29	0.21; 0.28	0.05; na	0.33; 0.28	8-2; 8-5	5; 1
Endrin Ketone	f, 2	0.67	1.21	0.9	2.1	5-4	2
2,4'-DDD	c, 2	0.27	0.43	0.09	1.5	6-1; 7-4	18
2,4'-DDT	c, 0	nd	0.3	0.1	0.72	7-1	6
4,4'-DDD	c, 1	0.12	0.73	0.14	2.2	7-2	22
4,4'-DDT	c, 0	nd	1.44	0.51	2.8	7-1	6
cis-Chlordane	c, 1	0.16	0.19	0.02	0.43	4-3	18
delta-BHC	c, 1; f, 0	0.18	1.2	na	1.2	6-3	1
PCB 8	c, 2	0.15	1.39	0.73	4.5	7-3	7
PCB 18	c, 0	nd	0.15	na	0.15	7-3	1
PCB 28	c, 1	0.09	0.29	0.23	0.74	6-3	3
PCB 44	c, 2	0.07	0.22	0.06	0.71	8-3	15
PCB 66	c, 1	0.02	0.36	0.07	0.95	6-3	13
PCB 128	c, 2	0.19	0.55	0.07	1.1	6-3	22

c=coarse, f=fine sediment; nd=not detected, na=not applicable

Chemical contaminants listed in Table 4-5 were not considered to pose an ecological threat after being evaluated as follows: 1) by comparing individual outfall station concentrations to ER-L and ER-M values or other west coast values; 2) by comparing mean or individual outfall station concentrations to the available reference concentration(s); or 3) by comparing surrogate analyte concentrations (e.g., other PCB analytes) that were detected in at least three reference stations. Sulfide concentrations below 500 µg/g are not felt to pose an ecological threat. Selenium concentrations below 0.33 µg/g are reported as uncontaminated background for San Francisco Bay sediments (Hunt et al. 1998). Naphthalene concentrations for all outfall sediments were well below the ER-L value of 160 ng/g. The mean outfall acenaphthene concentration was below the ER-L of 16 ng/g and individual outfall station concentrations were well below the ER-M value of 500 ng/g. Biphenyl reference and outfall concentrations in coarse sediments were comparable. Mean phthalate and phenol concentrations were higher in reference sediment groups. The remaining pesticides and PCB congeners all had maximum concentrations less than 5 ng/g and often less than 1 ng/g (below the target detection limit).

Ocean channel outfalls

Cadmium (core interval (ft))	1-1	1-2	1-3	1-4	16-1	16-2	16-3	16-4	16-5
0-1	2.25	0.14	0.14	0.26	1.04	10.01	0.18	0.07	0.00
1-2	0.08	0.08	0.04	0.05	2.65	2.07	0.31	0.10	0.10
2-3	0.05	0.03	0.07	0.04	1.28	0.01	1.35	0.16	0.05
3-4	0.09	0.06	0.06	0.04	0.42	0.03	0.02	0.11	0.05
4-5	ND	0.05	0.03	0.02	0.39	0.02	0.02	0.03	0.07
5-6	0.04	0.03	0.01	0.01	6.79	ND	0.02	0.01	0.09
6-8	0.02	0.04	0.03	0.03	0.05	0.03	0.00	0.04	0.02
8-10	0.04	0.04	0.02	0.02	0.03	0.05	0.03	0.01	0.01

Lead (core interval (ft))	1-1	1-2	1-3	1-4	16-1	16-2	16-3	16-4	16-5
0-1	2.16	0.19	0.15	0.28	1.23	9.09	0.23	0.20	0.18
1-2	0.22	0.20	0.18	0.18	2.76	2.16	0.39	0.23	0.21
2-3	0.18	0.19	0.18	0.18	1.29	0.17	1.18	0.24	0.20
3-4	0.25	0.18	0.16	0.19	0.76	0.15	0.16	0.19	0.22
4-5	0.12	0.17	0.17	0.18	0.66	0.17	0.16	0.15	0.20
5-6	0.13	0.19	0.16	0.17	6.97	0.14	0.18	0.16	0.19
6-8	0.17	0.16	0.16	0.17	0.16	0.17	0.18	0.16	0.18
8-10	0.12	0.14	0.14	0.17	0.15	0.15	0.17	0.19	0.17

Total DDT (core interval (ft))	1-1	1-2	1-3	1-4	16-1	16-2	16-3	16-4	16-5
0-1	994	2.85	5.78	2.52	38.2	112	1.83	0.84	ND
1-2	24.7	3.43	0.23	ND	36.3	28.1	1.12	1.40	ND
2-3	41.0	5.71	0.16	ND	32.4	0.97	13.3	1.81	0.05
3-4	45.4	0.63	0.32	ND	14.0	ND	ND	0.08	ND
4-5	0.78	0.35	0.40	ND	14.1	ND	2.04	ND	ND
5-6	0.58	0.43	0.09	ND	2.86	ND	ND	0.11	ND
6-8	105	0.18	0.06	ND	ND	ND	ND	ND	ND
8-10	6.05	0.37	ND	ND	ND	ND	ND	ND	ND

In-bay outfalls

PCB 105 (core interval (ft))	3-2	3-3	4-2	4-3	5-2	5-3	6-2	6-3	7-2	7-3	8-2	8-3
Surface	ND	1.17	0.78	1.76	0.89	1.56	1.06	2.79	0.45	2.65	0.61	4.47
1-2	4.75	ND	0.75	1.81	0.87	0.36	0.7	0.1	0.47	2.01	2.15	0.22
2-3		0.08	1.76	1.51	2.57	0.2		ND	1.73	0.45	0.18	ND
3-4		ND	1.2	0.56		ND		ND		ND		ND
4-5		1.06		ND		ND				ND		ND
5-6		ND		ND						ND		ND

Ratio

0 - 1	> 1 - 2	> 2-3	> 3 - 4	> 4 - 5	> 5 - 10	> 10

Figure 4-7. Ratios of subsurface contaminant concentrations to 99% predictive limit for ocean channel and in-bay outfalls.

4.2 Subsurface Sediments

Contaminant concentrations of in-bay and ocean channel subsurface sediments were evaluated to determine the vertical extent of contaminant distributions observed in surface sediments and to search for elevated contaminants at depth. Sediment vibracores were taken nominally to 6 ft or refusal at the 10 and 30 m surface stations sampled at each in-bay outfall (3-8). Four cores were taken at outfalls 1,2 and five cores were taken at outfall 16 successfully

to 10 ft with an impact stratocorer. Chemistry data from these latter two sets of cores constitute the entire data set for outfalls 1,2 and 16. These outfalls are evaluated for impact to human health only using soil preliminary remediation goals for cancer risk and non-cancerous hazard (Section 6.2). Subsurface sediments collected at in-bay outfalls 3-8 are evaluated for vertical trends only; and are not evaluated for impacts to ecological or human health. The core data for all outfalls are best summarized in Figure 4-6, where concentrations of select analytes or analyte groups are depicted in multiples of the coarse reference surface sediment 99% predictive limits. All contaminants that exceeded these limits are identified below. Vertical gradients in subsurface contaminants were evaluated using linear regressions of log-transformed contaminant concentrations for three groups of stations: in-bay outfalls 3-8; ocean outfall 1,2; and ocean outfall 16. In general, nearly all analytes, except titanium and manganese, decreased significantly with depth. At outfalls 1,2 and 16 subsurface maxima in PAH and metal concentrations occurred in core stations closest to the outfalls. Concentration results are listed by station and analyte in Appendix E3. Results for statistical comparisons and regressions are included in Volume II.

Table 4-6. In-bay subsurface analytes with concentrations greater than the corresponding reference 99% predictive limit (ratios >1).

Analyte	Station	Core Interval (ft)	Concentration	Ratio*
Metals (μg/g)				
Antimony	7-3	1-2	20.9	10.29
Barium	3-3	1-2	645	1
Mercury	5-3	4-5	0.345	1.12
	7-3	3-4	0.895	2.91
Molybdenum	8-2	2-3	4.26	1.11
Silver	5-3	1-2	7.54	3.44
Titanium	3-3	1-2	5630	1.1
	3-3	2-3	5140	1.01
	7-3	3-4	5640	1.11
	8-3	3-4	6060	1.19
PAHs (ng/g)				
Indeno[1,2,3,-c,d]pyrene	3-2	1-2	930	1.1
Pesticides & PCBs (ng/g)				
Total DDT	7-2	2-3	8.74	3.49
	7-3	1-2	3.83	1.53
Hexachlorobenzene	4-3	1-2	4.7	1.43
PCB 101	5-2	2-3	3.6	1.18
PCB 105	3-2	1-2	1.7	4.75
	3-3	4-5	0.38	1.06
	4-2	2-3	0.63	1.76
	4-2	3-4	0.43	1.2
	4-3	1-2	0.65	1.81
	4-3	2-3	0.54	1.51
	5-2	2-3	0.92	2.57
	7-2	2-3	0.62	1.73
	7-3	1-2	0.72	2.01
	8-2	1-2	0.77	2.15

* Ratio = a core conc/reference 99% predictive limit

4.2.1 In-Bay Outfalls 3-8

Sediment depths sampled are depicted in Figure 4-6 (in-bay outfalls), where colored cells represent depth of penetration for each core. The target depth of 6 ft was obtained for four out of 12 stations (3-3, 4-3, 7-3, 8-3) only due to the high content of shell hash in stations closest to the outfalls.

All contaminants that were statistically higher than the coarse reference group are listed in Table 4-6. Contaminant concentration, station and depth of occurrence, and the ratio of the contaminant concentration to the 99% predictive limit of the coarse surface sediment group for all contaminants also are shown.

None of the semivolatile compounds were significantly elevated relative to reference levels. Concentrations of most of the individual and summed PAHs, phthalate and phenol exhibited significant, exponential decreases with core depth. PAHs in subsurface sediments at outfall 3 were depleted in low molecular weight PAHs and enriched in high molecular weight compounds. These patterns were similar to those in surface sediments (Section 4.1.1) and indicative of creosote, which probably leached from wooden pier pilings located at the adjacent Pier Bravo. PAHs in subsurface sediments at other outfall stations appear to be derived primarily from combustion sources and are discussed further in the following subsection.

The pesticides DDE, DDD, and HCB, as well as total PCBs and approximately half of the PCB congeners also exhibited significant decreases with depth in subsurface sediments at bay outfall stations. PCB 105 was the only congener that consistently appeared at statistically higher concentrations than in reference sediments (Table 4-6), and is the only in-bay contaminant plotted in Figure 4-5. Tributyltin was present at relatively uniform concentrations in all core depths. Total butyltin concentrations in subsurface sediments at the in-bay stations decreased exponentially with core depth, driven primarily by mono- and di-butyltins.

As in surface sediments, cyanide was not detected in subsurface sediments. Mean sulfide concentrations, averaged over all outfalls, decreased by one order of magnitude with core depth, and all concentrations at depths greater than 2 ft were below the mean concentration for the coarse sediment reference group.

Concentrations of iron and aluminum in subsurface sediments at in-bay outfall stations did not exhibit significant gradients with depth, indicating no systematic changes in gross sediment characteristics. Antimony, arsenic, cadmium, copper, lead, mercury, silver, tin, and zinc concentrations all decreased exponentially with depth at the in-bay outfall stations. Titanium and manganese concentrations increased significantly with core depth, whereas concentrations of the remaining metals did not exhibit any trends.

4.2.2 Ocean Outfall 1,2

All contaminants at outfall 1,2 that were statistically higher than the coarse reference group are listed in Table 4-7. Contaminant concentration, station and depth of occurrence, and the ratio of the contaminant concentration to the 99% predictive limit of the coarse surface sediment group for all contaminants also are shown.

PAH concentrations in cores were not significantly higher than in reference sediments. However, PAH concentrations play a prominent role in the human health assessment (Section 6.2) and are discussed for this outfall and outfall 16 as well. Outfall 1,2 is primarily dominated by degraded diesel fuel that extends down as far as 10 ft in depth in stations closest to the outfall. The 0-1 ft intervals generally exhibited the highest concentrations. Concentrations of low and high molecular weight PAHs decreased significantly with depth in subsurface sediments

at outfall 1,2. Mean concentrations of phenol decreased with core depth, except for an elevated concentration (890 ng/g) in the 3-4 ft layer.

No significant trends in the depth distributions of 2,4'- and 4,4'-DDD and 4,4'-DDE or other pesticides were detected. High concentrations of total DDT were found throughout the core at station 1-1, closest to the outfall, and at shallower core intervals with distance from the outfall (Figure 4-5).

Table 4-7. Ocean outfall 1,2 subsurface analytes with concentrations greater than the corresponding reference 99% predictive limit (ratios > 1).

Analyte	Station	Core Interval (ft)	Result	Ratio
Metals (μg/g)				
Barium	1-1	2-3	779	1.20
	1-1	3-4	788	1.22
	1-1	4-5	1028	1.59
	1-1	5-6	750	1.16
	1-2	2-2	757	1.17
	1-2	3-4	756	1.17
	1-2	4-5	674	1.04
	1-2	5-6	654	1.01
	1-2	6-7	1089	1.68
	1-2	7-8	882	1.36
	1-3	2-3	801	1.24
	1-3	3-4	709	1.10
	1-3	4-5	700	1.08
	1-3	5-6	757	1.17
	1-3	6-7	753	1.16
	1-3	7-8	728	1.13
	1-4	2-3	711	1.10
	1-4	3-4	744	1.15
	1-4	4-5	878	1.36
	1-4	5-6	757	1.17
	1-4	6-7	723	1.12
	1-4	7-8	798	1.23
	1-4	8-9	729	1.13
Manganese	1-1	8-9	1152	1.20
Titanium	1-1	6-7	14200	2.79
	1-1	8-9	19030	3.73
	1-2	6-7	12300	2.41
	1-2	7-8	5460	1.07
	1-3	8-9	13400	2.63
Pesticides & PCBs (ng/g)				
4,4'-DDE	1-1	3-4	1.7	1.30
4,4'-DDE	1-1	4-5	3.4	2.61
4,4'-DDE	1-1	7-8	10	7.67
Total DDT	1-1	2-3	61.84	24.67
	1-1	3-4	102.7	40.97
	1-1	4-5	113.7	45.35
	1-1	7-8	263.1	104.95
	1-1	8-9	15.17	6.05
	1-2	2-3	8.6	3.43
	1-2	3-4	14.32	5.71
PCB 105	1-1	4-5	0.42	0.17

Aluminum, beryllium, cadmium, lead, mercury, silver and sulfide concentrations exhibited significant, exponential decreases with increasing core depth; whereas, manganese and titanium concentrations increased; and trends in iron concentrations were not statistically significant. Sediments from the 3-4 ft layer contained elevated barium concentrations, the 5-6 ft layer contained elevated barium, manganese, and titanium, and the 8-10 ft layer contained elevated titanium.

4.2.3 Ocean Outfall 16

All contaminants at outfall 16 that were statistically higher than the coarse reference group are listed in Table 4-8. Contaminant concentration, station and depth of occurrence, and the ratio of the contaminant concentration to the 99% predictive limit of the coarse surface sediment group for all contaminants also are shown.

Highest concentrations of PAH_{16} occurred in the 0-1 ft depth interval (30,100 ng/g). Concentrations of PAH_{16}, averaged by core interval, decreased with depth, except for elevated concentrations (16,331 ng/g) in the 5-6 ft interval. PAHs at the 5-6 ft interval were primarily alkyl-substituted naphthalenes, fluorenes, phenanthrenes/ anthracenes, and dibenzothiophenes. Depth-related patterns in high and low molecular weight PAHs followed those for PAH_{16}, with generally decreasing concentrations at greater core depths, except for elevated concentrations ER-M values. Table 4-10 identifies ecological chemicals of concern, mean concentrations for the outfall with the maximum concentration, mean concentration for combined reference stations, standard errors, maximum outfall concentration, and station where maximum concentration was found. Similar results for each outfall, and outfall and reference stations grouped by grain size are reported in Volume II. Analyte results for each station are listed in Appendix E.

Concentrations of total PCBs and total butyltins also decreased with depth. The highest total PCB concentrations (100-302 ng/g) occurred within the upper three layers (i.e., 0-1, 1-2, and 2-3 ft) at outfall 16. Mean and maximum concentrations at other core intervals were less than 15 and 35 ng/g, respectively. Concentrations generally decreased with core depth, and mean concentrations at depths below 3 ft were less than 10 ng/g. Phthalate and phenol concentrations decreased logarithmically with depth as well. Total DDT concentrations were highest throughout the core at station 16-1, similar to the core at station 1-1 (Figure 4-5). Total DDT concentrations decreased, but persisted in the shallow core intervals with distance from the outfall.

Overall, concentrations of antimony, cadmium, chromium, lead, mercury, silver, and zinc decreased exponentially with core depth, while thallium concentrations increased at outfall 16. The metal distributions tended to be layered. In particular, the 0-1 ft layer at outfall 16 contained elevated antimony, cadmium, chromium, lead, molybdenum, nickel, and zinc concentrations. The 1-2 ft layer contained high concentrations of chromium, and the 5-6 ft layer contained elevated cadmium, copper, lead, nickel, silver, and zinc.

Table 4-8. Ocean outfall 16 subsurface analytes with concentrations greater than the corresponding reference 99% predictive limit (ratios >1).

Analyte	Station	Core Interval (ft)	Result	Ratio
Metals (μg/g)				
Antimony	16-1	5-6	4.2	2.07
Barium	16-1	6-7	674	1.04
	16-1	7-8	654	1.01
	16-2	1-2	654	1.01
	16-2	2-3	731	1.13
	16-2	3-4	664	1.03
	16-2	4-5	725	1.12
	16-2	5-6	704	1.09
	16-2	6-7	703	1.09
	16-2	7-8	660	1.02
	16-3	1-2	655	1.01
	16-3	3-4	949	1.47
	16-3	4-5	704	1.09
	16-3	5-6	748	1.16
	16-3	6-7	789	1.22
	16-3	7-8	716	1.11
	16-4	1-2	677	1.05
	16-4	2-3	696	1.08
	16-4	3-4	667	1.03
	16-4	4-5	723	1.12
	16-4	5-6	669	1.03
	16-4	6-7	688	1.06
	16-4	7-8	789	1.22
	16-5	3-4	739	1.14
	16-5	4-5	742	1.15
	16-5	5-6	826	1.28
	16-5	6-7	702	1.09
	16-5	6-8	753	1.16
	16-5	8-10	719	1.11
Cadmium	16-1	1-2	5.5	2.65
	16-1	2-3	2.66	1.28
	16-1	5-6	14.1	6.79
	16-2	1-2	4.3	2.07
	16-3	2-3	2.8	1.35
Chromium	16-1	1-2	465	3.56
Copper	16-1	5-6	312.8	1.35
Lead	16-1	1-2	142	2.76
	16-1	2-3	66.2	1.29
	16-1	5-6	358	6.97
	16-2	1-2	111	2.16
	16-3	2-3	60.5	1.18
Molybdenum	16-1	1-2	3.9	1.02
	16-1	5-6	5.9	1.54
	16-2	1-2	4.1	1.07
	16-2	8-10	4.2	1.09
Titanium	16-1	1-2	6750	1.32
	16-1	2-3	6640	1.30
	16-1	4-5	5350	1.05
	16-1	5-6	5910	1.16
	16-2	1-2	5150	1.01
	16-2	8-10	6310	1.24
	16-4	3-4	6060	1.19
PAH (ng/g)				
Fluorene	16-1	5-6	1600	1.20
Pesticides & PCBs (ng/g)				
4,4'-DDE	16-1	1-2	32	24.55
4,4'-DDE	16-1	2-3	7.3	5.60
4,4'-DDE	16-1	3-4	6.4	4.91
4,4'-DDE	16-1	4-5	2.7	2.07
4,4'-DDE	16-2	1-2	2.4	1.84
4,4'-DDE	16-3	2-3	26	19.95
Total DDT	16-1	1-2	91	36.30
	16-1	2-3	81.3	32.43
	16-1	3-4	35.2	14.04
	16-1	4-5	35.44	14.14
	16-1	5-6	7.18	2.86
	16-2	1-2	70.4	28.08
	16-3	1-2	2.8	1.12
	16-3	2-3	33.22	13.25
	16-3	4-5	5.12	2.04
	16-4	1-2	3.52	1.40
	16-4	2-3	4.53	1.81
HCB	16-1	5-6	3.8	1.16
	16-2	1-2	3.6	1.10
	16-3	2-3	3.8	1.16
PCB 101	16-1	2-3	9.1	2.97
	16-1	3-4	5.6	1.83
	16-2	1-2	12	3.92
	16-3	2-3	12	3.92
PCB 105	16-1	2-3	3.4	9.49
	16-1	4-5	0.38	1.06
	16-2	1-2	5	13.96
	16-3	2-3	3.6	10.05
	16-4	1-2	0.54	1.51
	16-4	2-3	0.48	1.34
	16-4	3-4	4.8	13.40
PCB 118	16-1	2-3	9.9	2.65
	16-1	3-4	5.4	1.45
	16-2	1-2	15	4.02
	16-3	2-3	9.8	2.62

4.3 Chemicals of Potential Concern

Chemicals analyzed in support of this RI/RFI were selected based on known or suspected historical inputs to Site 1 sediments and relevance to ecological and human health. These chemical analytes were presented by class in Section 2.4. Semivolatile organics, consisting of 41 polycyclic aromatic hydrocarbons (PAHs), bis(ethylhexyl)phthalate, and phenol, 21 organochlorine pesticides, and 18 polychlorinated biphenyls (PCBs) made up the organics. Twenty-two metals, cyanide, total sulfides and three butyltin compounds comprised the inorganics. All of these compounds were initial chemicals of concern except for the alkylated homologue PAH compounds and dibenzothiophene, and several of the metals. Chemicals that were not of concern were added to the analyte list in the work plan (Bechtel 1995) to support data analyses used to identify contaminant source or type. Table 4-9 lists chemicals analyzed for these purposes. The metals, iron and aluminum, also were excluded as chemicals of concern because natural concentrations are sufficiently high to make anthropogenic inputs minimal (Bruland et al. 1974; Trefry 1985). These metals were used to discern inputs of metal contaminants relative to natural or background concentrations. Two non-contaminants also were measured either to provide information used to separate outfall and reference sediments into groups for statistical comparisons (Section 3), or to address the potential bioavailability of contaminants (Section 6). These were total organic carbon and total sulfides, and also are not considered chemicals of concern. Other analytes dismissed as chemicals of concern were those that were not detected in any of the Site 1 sediments; these also are identified in Table 4-9.

Chemicals of concern were identified using criteria specific for the human health and ecological risk screening assessments. Chemicals of concern for human health consisted of all analytes that were detected in the exposure media (e.g., surface sediments) and had corresponding preliminary remediation goals (PRGs) for soil. These chemicals are identified in Section 6.2 (Table 6-4). Chemicals of concern for ecological health consisted of all analytes that were elevated in surface sediments (outfalls 3-8) compared to appropriate reference sediments or

Table 4-9. Analytes not considered initial chemicals of concern.

Analyte	Reason
Aluminum	Used to evaluate metal contaminant inputs
Iron	Used to evaluate metal contaminant inputs
C1-C4 Naphthalenes	Used to identify petroleum source
C1-C3 Fluorenes	Used to identify petroleum source
C1-C4 Phenanthrenes/Anthracenes	Used to identify petroleum source
Dibenzothiophene	Used to identify petroleum source
C1-C3 Fluoranthenes	Used to identify petroleum source
C1-C4 Chrysenes	Used to identify petroleum source
Cyanide	Not detected in any sediment
Total Organic Carbon	Used to assess bioavailability
Total Sulfides	Used to assess bioavailability

Table 4-10. Chemicals of concern for ecological risk screening assessment.

Chemical	Mean Outfall Concentration*	Mean Reference Concentration	Maximum Outfall Concentration	Station of Max. Conc.
Metals (μg/g)				
Antimony	11.3 ± 10.3	0.49 ± 0.08	52.63	8-1
Barium	635 ± 29	529 ± 12	714	4-1
Cadmium	1.32 ± 0.5	0.28 ± 0.05	3.1	7-4
Lead	37 ± 12	18 ± 2	77.3	7-1
Mercury	0.25 ± 0.12	0.08 ± 0.01	0.55	7-5
Molybdenum	18.9 ± 17.3	0.75 ± 0.14	88.1	8-1
Tin	9.62 ± 4.1	2.23 ± 0.53	24.94	8-1
PAHs (ng/g)				
Naphthalene	13.7 ± 9.3	5.38 ± 1.54	23	6-1
Acenaphthalene	14.6 ± 9.5	4.9 ± 0.7	51	3-2
Indeno[1,2,3-cd]pyrene	194.8 ± 154	98.0 ± 33.4	810	3-2
Pyrene	565 ± 413	114 ± 29	2200	3-2
Pesticides & PCBs (ng/g)				
Total DDT**	6.5 ± 1.2	4.8 ± 1.8	11.18	7-4
delta-BHC	1.15 ± 0.05	0.18 ± 0.0	1.2	6-3
Hexachlorbenzene	6.82 ± 6.8	0.05 ± 0.01	34	4-3
Total PCBs**	29.4 ± 12.2	43.4 ± 12.8	71.9	7-4

* Mean concentration of outfall with maximum concentration
**Total concentrations shown to represent individual isomers (e.g. PCB 101, 105, 118)

Section 5 NATURE AND EXTENT OF TOXICITY AND BIOACCUMULATION

This section describes the nature and extent of toxicity and bioaccumulation to organisms exposed to surface sediments collected at Site 1 for the 30 outfall stations and ten reference stations located within San Diego Bay. Result summaries for each toxicity test are presented in Sections 5.1 - 5.4 and shown in Figures 5-1 and 5-2. Bioaccumulation results for clam tissue are discussed in Section 5.5. Results include relevant control data and statistical comparisons of outfall data to reference data. Results for both grouped and individual outfall stations are compared to grouped reference stations of comparable grain size. Quality control results for toxicity and tissue chemistry are presented in Section 5.6. Summarized bioassay laboratory results, including laboratory replicate and water quality data are presented in Appendices D1 and D2, respectively. Tissue chemistry results for the clam bioaccumulation test are summarized in Appendix E4. Statistical outputs are presented in Volume II, Sections P and Q.

5.1 10-DAY AMPHIPOD TOXICITY TEST

Extreme concentrations of either fine or coarse sedimentary material may have a negative effect on the survival of *Rhepoxynius abronius* (DeWitt et al. 1988, 1989). Since sensitivity to fine grain, clay-dominated material is of concern, clean, very fine grain sediment (> than 95% fines) from Tomales Bay, located in the northern reaches of San Francisco Bay, was tested for evidence of possible particle size effects. Negative control sediment (sediment in which *R. abronius* were collected) results averaged 98.3% survival and 95.3% reburial. The fine grain control sediments from Tomales Bay had significantly reduced survival, averaging 78.5% survival and 97.4% reburial. Table 5-1 summarizes toxicity and grain size results for control, reference, and outfall sediments. While not tested specifically, adverse effects due to fine grain size are not felt to contribute to amphipod mortality in the outfall or reference stations. In this and subsequent tables, significant differences between mean reference and mean outfall stations are denoted.

Table 5-1. Summary of negative control, outfall, and reference station toxicity test results for *Rhepoxynius abronius*.

Sediment Type	n	Mean Percent Survival	Standard Error	Mean Percent Reburial	Standard Error	Mean Percent Fine Sediment
Negative Control	3	95.3	1.5	97.0	2.5	NM
Fine Grain Control	2	76.5	6.5	97.4	2.6	>95
Fine Grain Reference Stations	5	84.8	5.7	95.9	0.9	60.3
Fine Grain Outfall Stations	6	72.3	7.2	96.3	1.3	55.9
Coarse Grain Reference Stations	5	73.6	8.4	97.7	1.6	25.0
Coarse Grain Outfall Stations	24	85.2	2.5	97.6	0.5	11.1

There were no statistical differences ($p \leq 0.05$) between reference and corresponding outfall results
NM = not measured; n = number of sediments tested

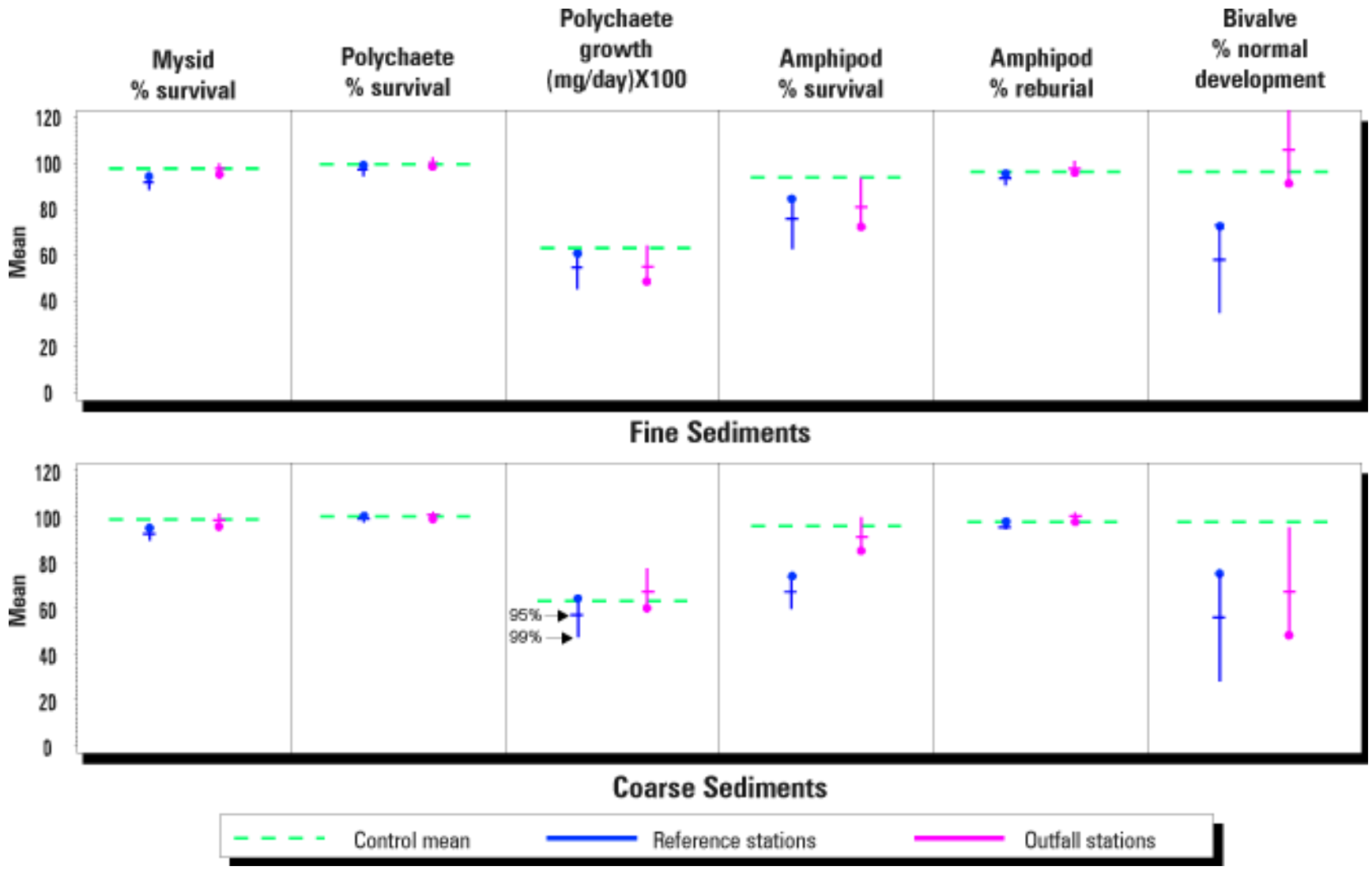


Figure 5-1. T-test results for toxicity endpoints for fine and coarse sediment groups. Confidence intervals are shown for 95% and 99% one-tail tests. All statistical results were insignificant at $p \leq 0.05$.

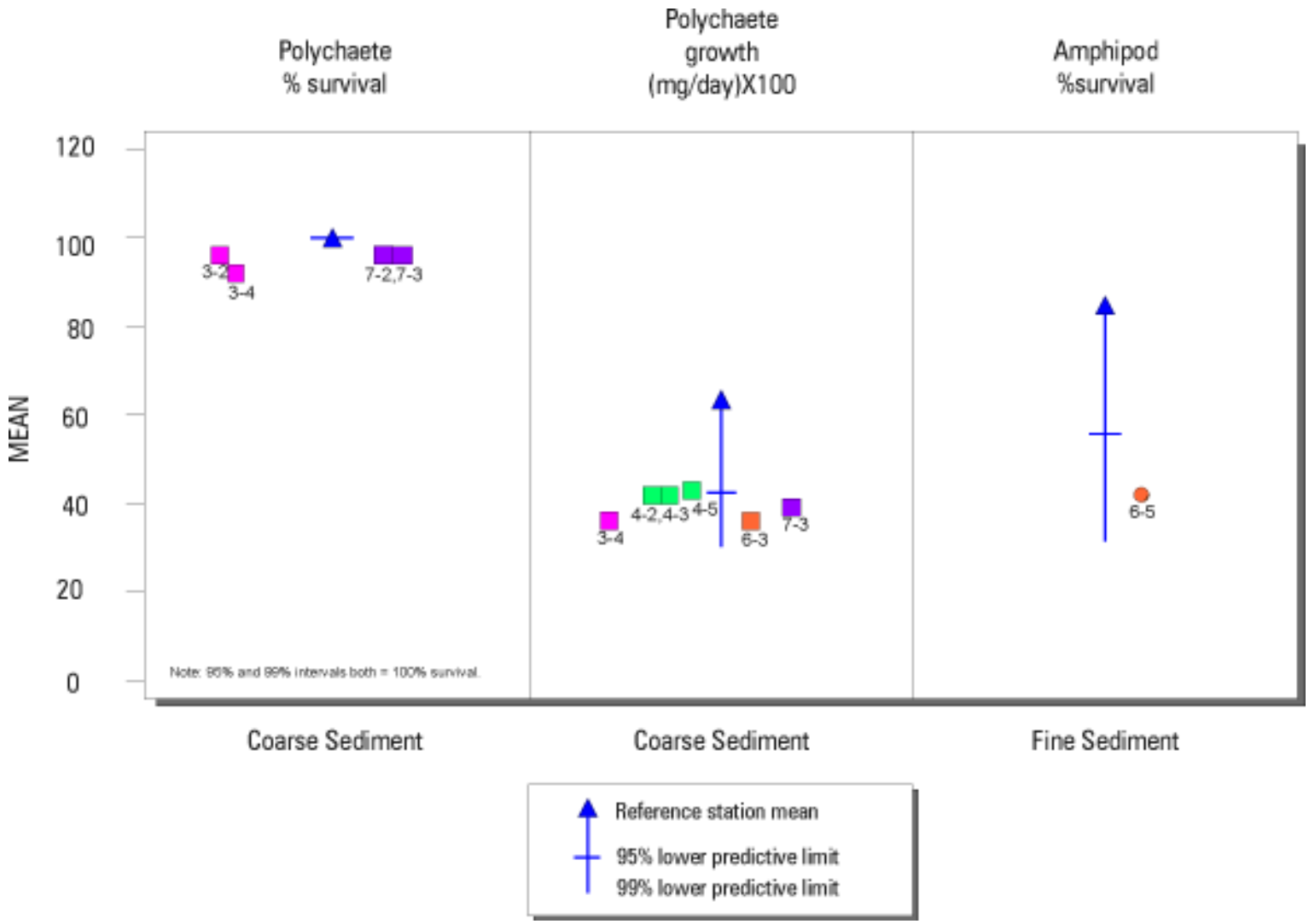


Figure 5-2. Toxicity results below the 95% predictive limit for corresponding sediment group reference stations. Only polychaete survival at four coarse sediment outfall stations was significantly lower than the reference 99% predictive limit.

Average amphipod survival for the fine sediment outfall group was lower than the survival observed in the fine reference group by 12.5%. Conversely, average amphipod survival for the coarse outfall group was greater than the survival observed in the coarse reference group by 11.6%. Figure 5-1 graphically depicts the results of one-tail t-test (i.e., $p \leq 0.01$, corrected and $p \leq 0.05$, uncorrected) comparisons between the two reference groups and the corresponding coarse and fine sediment outfall groups. The results of these inference tests were not significantly different at the $p \leq 0.05$ level for either coarse or fine sediment groups for either amphipod survival, reburial or any other biological endpoint, as discussed in subsequent sections. This analysis was repeated with a Bonferroni adjustment for multiple testing and, of course, indicated no significant difference at the $p \leq 0.01$ level.

The Environmental Monitoring and Assessment Program (USEPA 1991) has reviewed results from over 3,000 acute amphipod bioassays for the purpose of establishing a meaningful "difference" between reference and test sediment toxicity. This program, and the USEPA/USACOE Green Book (1991), have determined that a meaningful difference in toxicity has been reached when test sediments are significantly different from reference sediments, and when mean test survival is at least 30% lower than mean survival in the reference sediment. Amphipod survivals for fine and coarse outfall sediments in the present study meet neither of these criteria.

Results for individual outfall stations were compared with corresponding reference group results, again based on sediment size. Stations were examined individually to pursue potential toxic hot spots that may have been obscured in the grouped comparisons. Survival in outfall sediments ranged from 42% (station 6-5) to 98% (station 8-1). The ability to rebury in clean control sediment after exposure to outfall sediment was very high, ranging from 91.7% (station 7-3) to 100% at eight outfall stations (5-1, 5-2, 5-3, 6-2, 6-3, 7-4, 7-5 and 8-3). Mean survival for all reference stations ranged from 41% (R-4) to 96% (R-15). Mean survival and reburial results for the five laboratory replicates tested for each outfall, reference, and control sediment are presented in Appendix D1.

Ninety percent or greater survival in negative controls was expected for this test species and was the main quality control acceptance criterion for the test. Fifteen outfall stations had survivals that exceeded 90%, including two stations in the fine sediment group (7-4 and 7-5). Six stations (4-1, 6-2, 7-1, 7-2, 8-1, 8-3) had survivals that exceeded negative control survivals. Coarse sediment outfall stations 3-4, 4-4, 4-5, 5-5, 7-3 had survivals that were lower than the coarse reference group mean of 73.6%. Fine sediment outfall stations 5-4, 6-4, 6-5 and 8-5 had survivals that were lower than the fine sediment reference mean of 84.8% and the fine grain control sediment survival of 78.5%.

Each of the 30 outfall stations was compared to the lower predictive limit of the corresponding, grain-size specific, reference station group. Significantly different outfall stations and the corresponding reference station mean are plotted in Figure 5-2. Outfall station 6-5 was the only fine sediment station to elicit significantly lower amphipod survival. Ninety-four percent of the surviving 42% amphipods at station 6-5 were able to rebury in clean sediment after the ten day exposure period, indicating that most of the surviving organisms were healthy. In contrast, outfall station 6-5 had high bivalve normal development (90.94%), mysid survival (98%), and *Neanthes* survival (96%), and moderate *Neanthes* growth (0.49 mg/day). When the predictive limit was corrected for multiple testing (i.e., increased to 99%), amphipod survival at station 6-5 was not different from the fine reference group.

5.2 4-DAY MYSID SHRIMP TOXICITY TEST

The mysid shrimp, *Holmesimysis costata*, does not bury in sediment, and since it migrates through the water column to feed primarily on suspended material, clean seawater was used as the negative control material. Negative control survival ranged from 96 to 100%. Table 5-2 summarizes negative control, reference, and outfall sediment toxicity results.

Table 5-2. Summary of negative control, outfall, and reference station toxicity results for *Holmesimysis costata*.

Sediment Type	n	Mean Percent Survival	Standard Error
Seawater Control	4	98.0	0.8
Fine Grain Reference Sediments	5	94.8	1.2
Fine Grain Outfall Sediments	6	95.7	1.3
Coarse Grain Reference Sediments	5	94.8	2.2
Coarse Grain Outfall Sediments	24	95.6	0.7

There were no statistical differences ($p \leq 0.05$) between reference and corresponding outfall results
n = number of sediments or water samples.

Coarse sediment stations had survivals ranging from 86 to 100%, with a mean of 95.6% survival. The six fine sediment stations had survival ranging from 90 to 98%, with a mean of 95.7%. Reference sediment survival ranged from 94 to 100%, with a mean of 94.8% for both groups.

Ninety percent, or greater, survival in negative controls was expected and was the main quality control acceptance criterion for the test. All test stations (outfall and reference) had high survivals, exceeding 90%, except station 4-3 (86%). Grouped outfall and reference means are shown in Figure 5-1. There were no significant differences in mysid mortality between grouped or individual outfall stations compared with the corresponding reference group at the $p \leq 0.05$ level.

5.3 20-DAY POLYCHAETE TOXICITY TEST

Test organisms originated from a sediment free culture maintained by Dr. D. Reish at California State University, Long Beach and hence native, or home sediment was not obtained for this species. Negative control sediment, collected from St Augustine, Florida, was used as negative control test material. Results for the four negative controls were 100% survival in all 20 laboratory replicates (4 control sediments x 5 replicates) and 0.63 mg for mean daily growth.

Survival in outfall sediments was high, ranging from 92% for station 3-3 to 100% for 24 of the remaining 29 stations. Survival in reference stations was even higher, at 100% for all stations. The survival endpoint for *Neanthes* is one of the least sensitive for the suite of solid-phase bioassays used. It is therefore not unusual to have 100% survival for even moderately contaminated sediments. Outfall station group survival and growth means are listed in Table 5-3 and plotted in Figure 5-1. There were no significant differences between outfall and reference means in these two endpoints.

Table 5-3. Summary of negative control, outfall, and reference station toxicity test results for *Neanthes arenaceodentata*.

Sediment Type	n	Mean Percent Survival	Standard Error	Mean Growth Rate (mg/day)	Standard Error
Negative Control	4	100.0	0.0	0.63	0.08
Fine Grain Reference Stations	5	99.2	0.8	0.61	0.07
Fine Grain Outfall Stations	6	98.7	0.8	0.49	0.02
Coarse Grain Reference Stations	5	100.0	0.0	0.64	0.04
Coarse Grain Outfall Stations	24	99.2	0.4	0.60	0.04

There were no statistical differences ($p \leq 0.05$) between reference and corresponding outfall results
n = number of sediments tested.

Individual outfall station comparisons identified 14 of the 24 coarse outfall stations as having slightly lower growth rates than the coarse grain reference average of 0.64 mg/day. The average of all coarse outfall stations was 0.60 mg/day. All fine sediment stations (5-4, 6-4, 6-5, 7-4, 7-5 and 8-5) had growth rates that were lower than the fine grain reference average of 0.61 mg/day. Those outfall stations with significantly lower polychaete survival and growth than comparable reference mean values at $p \leq 0.05$ are plotted in Figure 5-2. When the predictive limits for the two reference station groups are corrected for multiple testing (i.e., $p \leq 0.01$), only polychaete survival at four coarse outfall stations were significantly lower. Polychaete survival at all coarse reference stations was 100%; therefore, any mortality exhibited at a coarse outfall station would be significantly different from perfect survival. Outfall stations 3-2, 3-4, 7-2 and 7-3 should not, however, be considered acutely toxic to *Neanthes*, because in all cases mortality was less than 10%. Ten percent mortality is the quality control acceptance criterion for the 20-day polychaete bioassay. Survival exceeded 90% in all test sediments.

5.4 48-Hour Bivalve Toxicity Test

Crassostrea gigas larvae do not bury in sediment, but migrate through the water column to develop and feed. Clean seawater was used in the negative controls, since it best represents normal environmental conditions. Negative control normal development ranged from 95.8 to 97.1%. Although bivalve larvae are not exposed to pore water under natural conditions, they are used as a surrogate test animal because of their sensitivity and suitability for small volumes of test material.

Testing pore water for toxic effects is difficult due to small volumes typically available and experimental artifacts produced by the collection process (Adams 1991; Manheim 1976). Pore water makes up a very small fraction of water-saturated sediments and usually is extracted from sediments using one of several methods. In general, volumes required for biological testing are obtained through either centrifugation or squeezing. Centrifugation was used to extract pore water from Site 1 sediments because: 1) a large fraction of suspended fine particles that could interfere with the test are removed, and 2) there are no additional substrates that adsorb contaminants (such as filters used in squeezing).

Result summaries are presented in Table 5-4; group means are plotted in Figure 5-1. No significant differences were detected between the two outfall and reference groups. Each reference group had one station that had very low normal development, contributing to the high variance observed for this test. Results for fine grain reference station R-8 and coarse grain reference station R-1 were 2.35 and 5.13% normal development, respectively. Normal development ranged from 0 (stations 6-2 and 8-2) to 98.0% (station 4-1) for all outfall stations.

Table 5-4. Summary of negative control, outfall, and reference station toxicity test results with *Crassostrea gigas*.

Sediment Type	n	Mean Percent Normal Development	Standard Error
Negative (Brine) Control	2	90.1	0.1
Negative (Seawater) Control	9	96.5	0.3
Fine Grain Reference Stations	5	73.0	18.0
Fine Grain Outfall Stations	6	91.3	0.9
Coarse Grain Reference Stations	5	75.2	17.5
Coarse Grain Outfall Stations	23†	48.1	9.7

There were no statistical differences ($p \leq 0.05$) between reference and corresponding outfall results
Station 5-3 was not tested due to insufficient pore water volume (†)
n = number of sediments or water samples tested

Ninety percent, or greater, normal development in the negative controls was expected for this species and was the main quality control acceptance criterion for the test. Twelve outfall stations (4-1, 4-3, 5-1, 5-2, 6-4, 6-5, 7-1, 7-3, 7-4, 7-5, 8-4 and 8-5) had greater than 90% normal development. Six stations (3-3, 3-4, 3-5, 4-4, 5-4 and 6-3) had normal developments ranging from 84.6 to 89.2%. The 11 remaining coarse outfall stations (3-1, 3-2, 4-2, 4-5, 5-5, 6-1, 6-2, 7-2, 8-1, 8-2, and 8-3) had less than 2% normal development.

Fine sediment stations generally had good development, ranging from 88.1 to 94.3%. Coarse sediment stations were split into two distinct groups; one with 11 stations exhibiting less than 2% normal development, and 12 stations with greater than 84.6% normal development. This bimodal distribution in larval development, while not normal, is not unusual in pore water studies (Carr, personal communication).

Low values for bivalve survival and development were not found consistently at any station or outfall, but were scattered among stations, confirming the usefulness of a suite of toxicity tests to evaluate Site 1. Bivalve development results were quite different from the amphipod survival data. Average bivalve normal development for the fine sediment outfall group was greater than that observed for the fine reference group by 19.2%. Conversely, average bivalve normal development for coarse outfall sediments was lower than coarse reference sediments by 64%. The identification of significant toxicity in the bivalve test was obscured by the highly variable, bimodal distribution of the results. In general, polychaete growth, amphipod survival, and bivalve normal development were much more variable in both outfall and reference sediments than the other endpoints. Usually, a single result contributed to the increased variance within each sediment group, especially for the reference stations. However, results were not consistent across tests. Station R-8 appears to be the largest contributor to the observed variance in the polychaete growth and bivalve normal development tests for the fine reference stations. Polychaete growth for R-8 was 0.33 mg/day compared to a range of 0.65 to 0.72 mg/day achieved for the remaining fine reference stations; and bivalve normal development was 2.35% compared to 58.33 to 97.51% for the same stations. However, increased variance in amphipod survival for the fine reference sediments was not due to station R-8 (80%); this variability was mainly attributable to station R-6, which had 65% survival. In the coarse reference stations, R-1 and R-4 were the largest contributors to the variance observed for the bivalve and amphipod survival tests, respectively. Normal development for R-1 was 5.13% compared to greater than 90% for the remaining coarse reference stations; and station R-4 had only 41% amphipod survival, compared with 78 to 89% for the remaining coarse reference stations.

5.5 28-Day Bioaccumulation in the Bentnose Clam

Evaluation of chemical bioaccumulation in clams exposed to Site 1 sediments relies primarily on statistical comparisons of grouped and individual outfall stations to grouped reference stations of similar grain size. This method is the same used for surface sediments in Section 4, except tissue concentrations instead of sediment concentrations are compared. Station groupings again are based on sediment grain size. This is appropriate for tissue concentrations since bioaccumulation is influenced by the adsorptive capacity of the sediment for contaminants. Some tissue analytes were not statistically compared because they were detected in fewer than three of the coarse or fine reference station groups. These analytes are discussed in Section 5.5.4.

The following subsections present results from chemical analyses of tissues of the bentnose clam *Macoma nasuta* following 28-day laboratory exposures to surface sediments collected from outfalls 3-8. These results are compared to tissue concentrations of clams exposed to reference sediments of comparable grain size. Tissue concentrations also are compared with corresponding sediment concentrations (from Section 4). Tissue and sediment concentrations are reported in dry weight. Results of chemical concentrations in tissues are presented and discussed separately for PAHs, pesticides, PCBs, and metals in Sections 5.5.1 - 5.5.3. Section 5.5.4 examines those chemicals that accumulated in the tissue of clams exposed to outfall sediments but were not detected in sufficient numbers (i.e., ≥3) of reference stations to support statistical comparisons. Section 5.5.5 discusses bioaccumulated chemicals, particularly silver, that were significantly higher in clam tissues associated with individual outfall stations compared to reference stations. Summaries of tissue chemistry quality control results are presented in Section 5.6. Tabular summaries of chemical results are presented in Appendix E4.

5.5.1 Polycyclic Aromatic Hydrocarbons (PAHs)

Relative differences among station groups for summed PAH concentrations (PAH_{16}) in clam tissues (Table 5-5) generally were similar to those observed in surface sediments (Section 4.1). Mean concentrations of summed PAHs in tissues of clams exposed to coarse sediments from outfall stations were approximately two times higher than those in organisms for the coarse sediment reference stations, and also higher than corresponding mean concentrations associated with the fine sediment outfall stations. As with the surface sediment PAHs, these differences are attributable to the relatively high PAH concentrations in clams exposed to sediments from outfall 3. By comparison, mean concentrations for summed PAHs in tissues exposed to fine sediments from the test and reference locations generally were comparable. As with surface sediments, there were no statistical differences at the $p \leq 0.01$ level for any PAH compound(s) for grouped comparisons of outfall and reference stations.

Table 5-5. Mean concentrations (± std error) of summed low and high molecular weight PAHs and PAH_{16} in clam tissues. Concentrations are ng/g dry weight.

	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	24	5	6	5
LMW	917 ± 324	798 ± 430	425 ± 37.4	477 ± 76.4
HMW	3347 ± 1344	2657 ± 1528	1627 ± 65.1	2032 ± 318
PAH_{16}	2421 ± 1036	1542 ± 860	1012 ± 52.1	1239 ± 185

There were no statistical differences ($p \leq 0.05$) between reference and corresponding outfall results
n=number of stations tested (across); Non-detect values were excluded from statistical calculations

Comparisons of individual results produced two stations that had significantly higher concentrations of PAH, acenaphthene at station 3-3 and C3-dibenzothiophene at station 5-4. C3-dibenzothiophene is not a chemical of concern, as discussed in Section 4.4, and was measured for use in identifying PAH hydrocarbon source. Acenaphthene concentration was 45 ng/g in tissue exposed to sediment collected from station 3-3 compared to a mean tissue concentration of 3.23 ng/g for the corresponding reference group. The likely source of acenaphthene is the creosote-soaked pier pilings of Bravo Pier. While surface sediment concentrations were lower at station 3-3, compared to station 3-2, they were still elevated above concentrations in surrounding sediments. Sediment concentrations of acenaphthene at stations 3-1 through 3-5 were 2.1, 51, 16, 3.1, and 0.88 ng/g, respectively, compared to an ER-L value of 16 ng/g and an ER-M value of 500 ng/g.

5.5.2 Pesticides and PCBs

There were no statistical differences ($p \leq 0.01$) for any pesticide or PCB compound(s) for grouped comparisons of tissue concentrations between outfall and reference stations. Comparison of individual outfall stations with the corresponding reference group, produced only one PCB congener (#28) at a single station (8-4), that was statistically elevated ($p \leq 0.01$).

Mean total DDT (sum of 2,4'- and 4,4'- DDT, DDE, and DDD isomers) concentrations in clam tissues were generally consistent for the different station groups, except for samples associated with the coarse sediment reference stations, which contained approximately 40% lower concentrations than those for the other station groups (Table 5-6). Mean total DDT tissue concentrations were nearly identical for the coarse and fine outfall station groups, and neither group was statistically different from the corresponding reference group. No individual station had DDT tissue concentrations statistically higher than the corresponding reference upper predictive limit.

Table 5-6. Mean total concentrations (± std error) of select chlorinated pesticides and PCBs in clam tissues. Concentrations are ng/g dry weight.

	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	23	5	5	5
Total DDT	16.8 ± 1.82	10.3 ± 2.33	16.7 ± 1.56	17.1 ± 1.60
Total Chlordane	1.42 ± 0.20	not detected	1.04 ± 0.07	4.87 ± 0.39
Total PCBs	86.5 ± 4.06	89.1 ± 14.5	88.2 ± 5.75	122 ± 14.8

There were no statistical differences ($p \leq 0.05$) between reference and corresponding outfall results .
n=number of stations tested (across); Non-detect values were excluded from statistical calculations

Mean total chlordane concentrations in clam tissues were highest for the fine sediment reference station group (4.87 ng/g), while concentrations in tissues for the coarse and fine outfall sediment groups were four-fold lower, and concentrations associated with the coarse sediment reference station group were non-detectable. Cis-chlordane was detected in samples associated with the outfall sediments only, while gamma-chlordane was present only in samples from the fine sediment reference stations and outfall 4. (See Section 5.5.4 and Table 5-8 below for a discussion on analyte non-detects in reference sediment.) Again, no individual station was significantly higher than reference in chlordane tissue concentration.

Mean total PCB concentrations in tissues of clams exposed to outfall and coarse reference sediments, were not significantly different (86.5 versus 89.1 ng/g), whereas mean concentrations in tissues of organisms exposed to

fine reference sediments were approximately 50% higher (Table 5-6). Station 8-4, a coarse-grained station, had the only significantly bioaccumulated PCB (congener #28) measured.

The relative abundances of individual congeners in tissue samples generally were consistent with abundances in the sediments. The lower and higher chlorinated congeners typically were undetected, or present at low concentrations, whereas, the pentachloro- and hexachloro- biphenyls (especially congener numbers 101, 118, 138, and 153) were detected the most frequently.

5.5.3 Metals

Comparisons of mean concentrations of individual metals in tissues of bentnose clams indicated few apparent differences among the four station groups (Table 5-7). For most metals, mean concentrations in sediments from each of the four station groups were higher than the corresponding mean concentrations in clam tissues. The exceptions were arsenic, cadmium, molybdenum, selenium, and silver, which were present in tissues at mean concentrations that were up to eight times higher than those in sediments. Silver bioaccumulated in tissue of clams exposed to coarse outfall sediments was the only contaminant measured to significantly ($p \leq 0.01$) exceed that bioaccumulated in clams exposed to coarse reference sediment, measuring 0.41 μg/g compared to 0.33 μg/g.

Table 5-7. Mean concentrations (± std error) of metals in clam tissues. Concentrations are μg/g dry weight.

	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	24	5	6	5
Antimony	0.13 ± 0.01	0.12 ± 0.01	0.12 ± 0.01	0.11 ± 0.01
Arsenic	25.3 ± 0.60	24.4 ± 0.89	24.6 ± 0.88	25.5 ± 0.87
Barium	2.93 ± 0.22	3.83 ± 0.20	3.66 ± 0.30	3.29 ± 0.20
Beryllium	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00
Cadmium	1.28 ± 0.03	1.28 ± 0.06	1.31 ± 0.03	1.23 ± 0.07
Chromium	2.92 ± 0.12	2.86 ± 0.24	2.90 ± 0.20	3.00 ± 0.22
Cobalt	1.47 ± 0.03	1.61 ± 0.10	1.55 ± 0.06	1.68 ± 0.06
Copper	26.9 ± 1.54	29.3 ± 6.63	21.0 ± 0.95	53.6 ± 24.2
Iron	1244 ± 54.7	1486 ± 81.1	1478 ± 83.7	1482 ± 45.0
Lead	4.95 ± 0.47	4.09 ± 0.63	3.80 ± 0.22	6.35 ± 1.40
Manganese	11.8 ± 0.44	16.5 ± 2.45	11.4 ± 0.68	11.3 ± 0.51
Mercury	0.08 ± 0.00	0.09 ± 0.01	0.10 ± 0.00	0.44 ± 0.36
Molybdenum	4.57 ± 0.12	4.57 ± 0.25	4.26 ± 0.16	3.90 ± 0.12
Nickel	4.64 ± 0.25	4.17 ± 0.43	4.27 ± 0.25	5.25 ± 0.87
Selenium	1.48 ± 0.04	1.49 ± 0.06	1.47 ± 0.10	1.37 ± 0.07
Silver	0.41 ± 0.03 **	0.33 ± 0.01	0.27 ± 0.01	0.28 ± 0.03
Thallium	0.02 ± 0.00	0.02 ± 0.00	0.02 ± 0.00	0.02 ± 0.00
Tin	0.44 ± 0.03	0.47 ± 0.07	0.48 ± 0.09	0.53 ± 0.06
Titanium	68.9 ± 3.47	80.4 ± 5.50	76.6 ± 6.68	78.9 ± 4.99
Vanadium	3.43 ± 0.11	3.85 ± 0.24	3.53 ± 0.15	3.58 ± 0.22
Zinc	109 ± 1.95	116 ± 6.82	108 ± 5.04	139 ± 11.4

Reference and corresponding outfall results are significantly different at $p \leq 0.05$ (*), at $p \leq 0.01$ (**)
n=number of stations tested (across); Non-detect values were excluded from statistical calculations

For metals that were elevated at individual outfall stations, apparently from industrial waste discharges (e.g., antimony, molybdenum and lead), no corresponding patterns were evident from tissue bioaccumulation results. Mean concentrations of these metals were not statistically different for either grouped or individual stations.

5.5.4 Chemicals with Insufficient Detections for Statistical Comparison

Table 5-8 lists chemical analyte, concentrations measured in two control tissues, grain size group (c=coarse, f=fine), reference sample size, reference mean concentration, grouped outfall mean, standard error and maximum value, the outfall station that had the maximum value, and outfall sample size. Most chemical comparisons in Table 5-8 are not very interesting. While PCB congeners, most DDT, and DDT breakdown products bioaccumulated to higher concentrations in clams exposed to outfall sediments, total mean DDT and PCB levels were either comparable or less than concentrations in clams exposed to reference sediments. This is true for PAHs as well. Pesticides, including chlordane, HCB, and the endosulfans appear to be significantly higher in outfall clam tissue, but they occur at low concentrations of only a few ng/g (ppb), which are below common detection limits and likely below levels of biological impact. Outfall 3 is prominent in this list, particularly with regard to PAHs and pesticides, as it was with sediment concentrations of these contaminants.

Table 5-8. Summary of analytes that were detected in outfall tissues but had less than three detectable values in the corresponding reference group. Concentrations are ng/g (ppb) dry weight.

Analyte	Control Conc.	Grain size*, reference n	Reference conc.	Outfall mean conc.	Outfall std. error	Outfall max. conc.	Station of max. conc.	Outfall n
Biphenyl	1.5; 3.1	c, 2	1.65	2.09	0.3	5.4	3-3	3
C3-Naphthalene	8.8; 31	f, 2	41.0	12.28	2.61	20.0	6-5	5
C4-Naphthalene	15; 0	c, 2	61.5	54.0	18.0	240	3-3	13
C4-Naphthalene	15; 0	f, 0	nd	23.75	4.59	34	5-4	4
Phenanthrene	0; 0	f, 1	16.0	6.7	3.84	25.0	5-4	3
2,4' DDD	0; 0	c, 0	nd	4.06	0.73	11.0	3-3	12
2,4' DDD	0; 0	f, 2	5.25	2.70	0.6	3.3	6-5	2
2,4' DDE	0; 0	f, 0	nd	2.10	na	2.10	8-5	1
2,4' DDT	0; 0	c, 0	nd	1.80	na	1.80	3-3	1
4,4' DDD	0; 0	c, 1	1.6	3.6	0.75	12.0	7-1	17
4,4' DDT	0; 0	c, 0	nd	8.65	5.35	14	3-4	2
Endosulfan I	0; 0	c, 0	nd	1.33	0.32	1.9	4-1	3
Endosulfan II	0; 0	c, 0	nd	4.25	0.99	6.3	5-5	4
HCB	0.7; 1.3	c, 0	nd	1.1	0.44	2.4	3-5	4
HCB	0.7; 1.3	f, 0	nd	0.67	na	0.67	5-5	1
PCB 44	0; 2	c, 0	nd	2.27	0.1	2.7	4-5	9
PCB 105	1.5; 1.6	c, 2	2.0	2.92	0.17	4.60	5-1	20
PCB 170	1.5; 1.6	c, 2	1.95	4.18	0.2	5.60	3-4	19
Total BHC	0; 0	c, 2	2.05	2.18	0.16	3.7	3-1	14
Total BHC	0; 0	f, 1	3.0	2.05	0.25	3.0	7-4	4
Total chlordane	0; 0	c, 0	nd	1.42	0.2	2.0	4-1	7
Total DDT	0; 0	c, 4	10.28	16.79	1.82	43.40	3-4	23
Total DDT	0; 0	f, 5	17.12	16.70	1.56	21.0	6-5	6
Total PCBs	43.8; 45.2	c, 5	89.06	86.54	4.06	129.8	5-2	23
Total PCBs	43.8; 45.2	f, 5	121.7	88.15	5.75	102.0	6-4	6

*c=coarse, f=fine sediment; nd=not detected, na=not applicable

5.5.5 Comparison of Individual Stations

Comparisons of chemical concentrations in tissues associated with individual outfall stations and mean tissue concentrations in clams exposed to appropriate reference sediments reveal few differences at the upper 99% predictive limit. Chemicals that were significantly higher than reference and the station of the occurrence are shown in Figure 5-3, and include acenaphthene at outfall station 3-3, an alkylated dibenzothiophene at station 5-4, the non-coplanar PCB 28 at station 8-4, two thallium hits at stations 4-4 and 8-4, and eight significantly higher bioaccumulations of silver in clams exposed to sediment from coarse-grained, shallow stations. These stations range around the perimeter of North Island from outfalls 4 to 8. Given the few significant tissue concentrations of chemicals other than silver, these contaminants probably do not mark contaminated hot spots. PCB 28 was unusually high at station 8-4 where PCBs had been significantly higher in the sediment as well. Acenaphthene bioaccumulation at station 3-3 was not surprising, given the moderate PAH sediment concentrations found at stations 3-2, 3-3, and 3-4. The reasons for the relatively high bioaccumulation levels of C3-dibenzothiophenes at station 5-4 and thallium at stations 4-4 and 8-4 are not clear.

Figures 5-4 A and B plot silver tissue concentration against sediment concentrations and sediment concentration normalized by grain size (i.e., divided by the sediment percent fines) for individual outfall stations with significantly elevated silver in tissue. The bioaccumulation of silver is curious since sediment silver concentrations were not significantly higher in either the grouped or any individual stations. While no specific tissue criteria for silver were found, they probably represent little threat to the immediate animal or food chain. A survey of silver concentrations in California estuaries ranged from 0.7 to 46.0 μg/g in mussels and 0.4 to 10.7 μg/g in gastropods (Kennish 1992). While these animals are not trophically equivalent to clams, these numbers provide a sense that clams exposed to outfall sediments are bioaccumulating silver to relatively low concentrations in the 28 day test period. This comparison is likely conservative considering that mussels are primarily filter feeders and most gastropods are herbivores, and both are less likely to ingest concentrations of silver as high as those found in sediment. The ER-L value for silver is 1 μg/g and sediment management standards for the state of Washington set a sediment quality level for silver at 6.1 μg/g (PSWQMP 1991). By inference, outfall sediment concentrations of less than 1 μg/g, and subsequent bioaccumulation probably do not pose an ecological threat.

Of some interest perhaps is the difference in predictive power for tissue concentrations that Figures 5-4 A and B show. While neither plot has a significant correlation between tissue silver concentration and respective sediment concentration, normalizing the sediments by grain size measurably strengthens the tissue to sediment concentration relationship (r^2=0.0, 0.2 respectively). This is reasonable if grain size and organic carbon content control the bioavailability of contaminants, which was assumed for this study (Section 2.2). Here, higher percent fines presumably bind silver and make it less available to the clams and, when taken into account, make the tissue burden more predictable.

This finding supports the approach of using reference stations that are matched by grain size with the outfall stations being tested. The exception to this trend, outlier station 5-2, has rich eel grass growth that may keep the silver bound and biologically unavailable. A plot of tissue concentration against sediment normalized by TOC was not as predictive as Figure 5-4B.

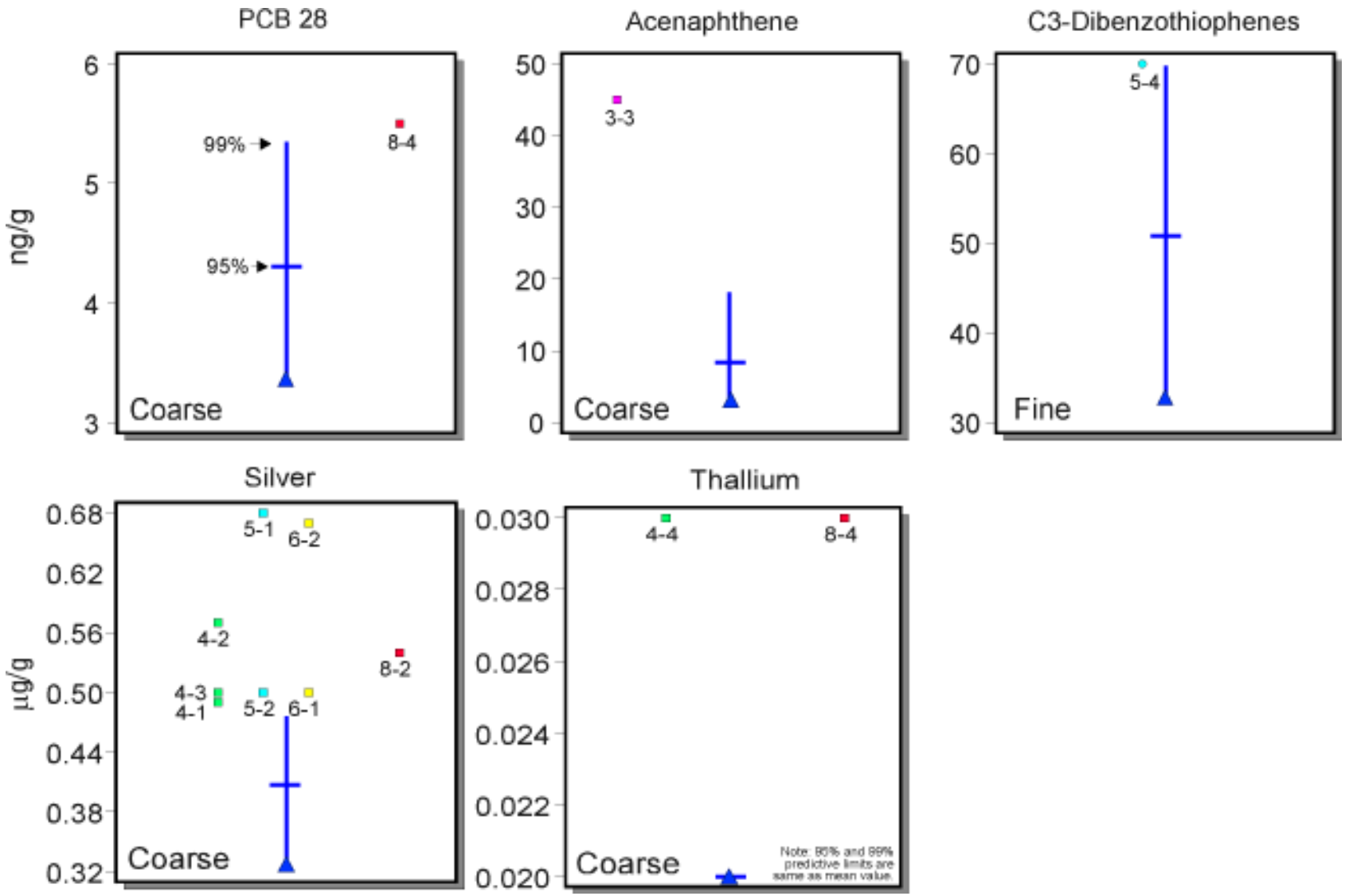
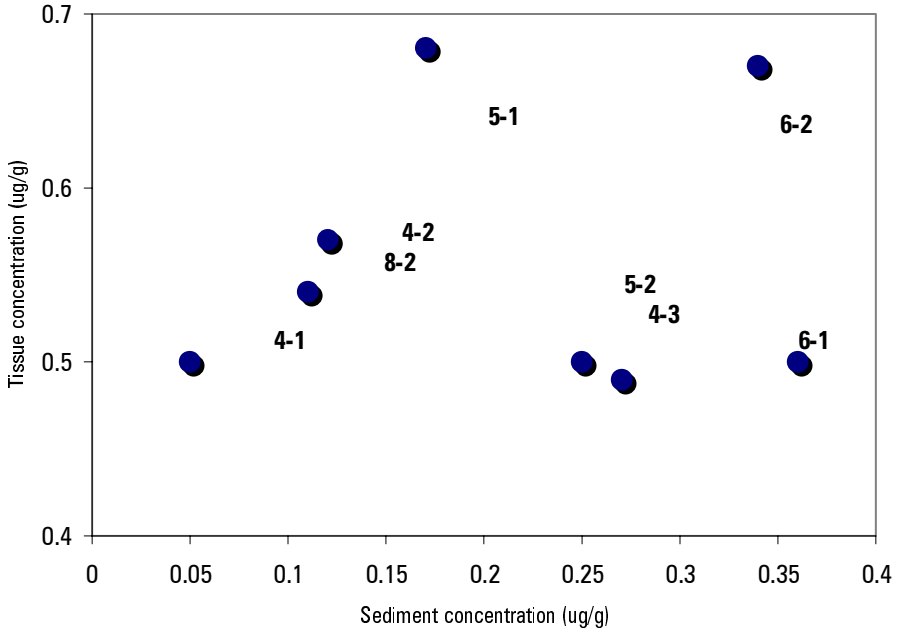


Figure 5-3. Chemical concentrations in tissues exposed to outfall sediments above the 99% predictive limit for corresponding reference stations.

A



B

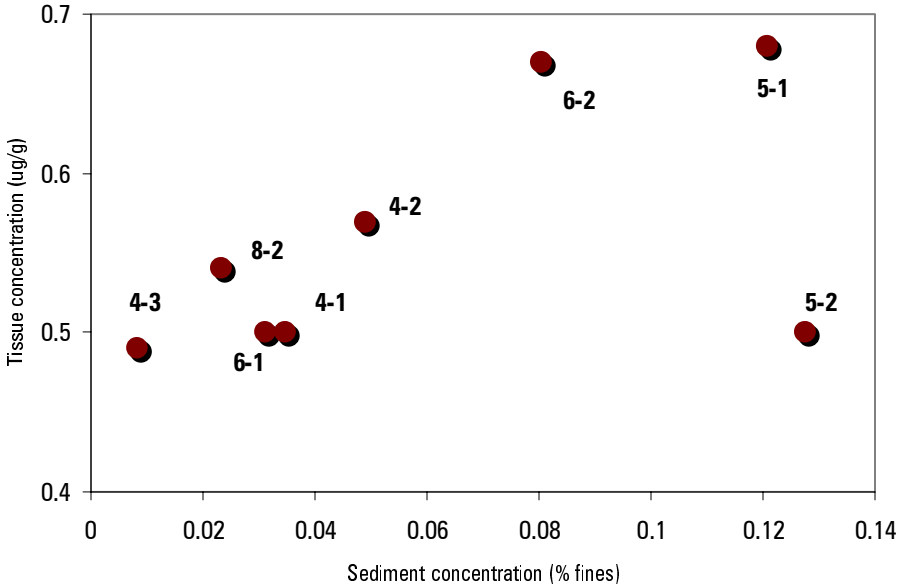


Figure 5-4. A: Silver concentration in clam tissue vs. sediment for outfall stations exceeding the coarse reference station mean tissue concentration. B: Same plot as A, except sediment silver concentration is normalized by sediment percent fines.

6.1 Ecological Risk Assessment Screening

The remedial investigation of Site 1 was designed to accommodate an ecological risk screening of sediments that presumably received industrial waste and sewage from NAS North Island outfalls 3-8 from the 1930s to the early 1970s. Since applicable or relevant and appropriate requirements have not been developed for contaminants in marine sediments, impact of outfall sediments was based on comparison of outfall sediments to in-bay reference sediments of comparable grain size. Various sediment quality guidelines, for example ER-L and ER-M values (Long et al. 1995) also were used to put contaminant concentrations in a national perspective. This investigation is an ecological screening assessment, rather than a risk assessment because it does not estimate the probability of ecological risk. Rather, uncertainties in both the reference and outfall data are used to assign statistical confidence limits to differences observed, presuming the outfall sediments are no different from reference sediments.

6.1.1 Objectives and Approach

To reiterate, the question of whether these sediments are impacted and pose a threat to the ecology of San Diego Bay relative to reference sediments was to be answered, based on evaluation of sediment chemistry, toxicity and bioaccumulation data presented in Sections 4 and 5. The decision matrix (Table 6-1) first proposed in the Bechtel 1995 work plan has guided evaluation of the data and has been adopted with only small changes to correct for multiple testing. The decision matrix presents specific actions in response to the data, ranging from consideration for immediate remediation to no action at Site 1, based on field data.

Table 6-1. Decision matrix used to determine ecological risk (adapted from Bechtel 1995).

Chemistry	Toxicity	Bioaccumulation	Action
+	+	-	water column evaluation
+	-	-	no further action
+	-	+	water column evaluation
-	+	-	water column evaluation
-	-	+	water column evaluation
-	+	+	water column evaluation
-	-	-	no further action
+	+	+	consideration for prompt remedial action

Pluses (+) denote significantly higher values in outfall sediments than in reference sediments for any single test.
Minuses (-) denote no significant differences between outfall and reference sediments.

6.1.2 Conceptual Site Model for Ecological Risk

The conceptual site model for ecological risk is illustrated schematically in Figure 6-1. Contaminant sources, transport routes, and primary marine receptors are identified in this model. Effects to surrogate organisms, representing primary ecological receptors for Site 1, were assessed in the laboratory using toxicity and bioaccumulation tests discussed in Section 5. Acute, chronic, and sub-chronic effects were measured in standardized tests to represent potential impacts to key ecological receptors. Test organisms are considered surrogates because they are used to estimate potential effects to resident groups or entire classes of animals that are exposed to contaminated sediments. Species resident to the westcoast and/or San Diego Bay were selected to represent the primary groups of animals that would be impacted from Site 1 sediments via bottom sediments or pore water. These animals include estuarine benthic infauna that reside in and ingest sediment, such as marine polychaete worms, burrowing amphipods, and clams. Mysid shrimps were used to represent the macro plankton that often are in contact with the sediment. All of these animals represent primary receptors that are in direct contact with surface sediments. Oyster larvae represent the sensitive life stages of benthic organisms and were used to assess potential toxicity from partitioning or desorption of contaminants into the sediment pore water. The bioaccumulation potential of sediment contaminants was assessed using the bentnose clam as a surrogate organism for both filter feeders and animals that syphon food directly from the sediment surface layer. The primary exposure route for the clam is through ingestion of both bottom sediments and pore water. All of these organisms are low on the marine food chain, represent food sources for predators such as demersal fish, and are therefore key ecological receptors for the site. The ecological relevance of each test species is discussed in Section 2; the measured effect from exposure to Site 1 sediments is discussed for each test organism in Section 5. Biological effects from exposure to suspended sediments were not considered.

6.1.3 Review of Results

When outfall sediments grouped according to grain size were compared to reference sediments of similar grain size, concentrations of cadmium, DDE, and PCB congener 105 were statistically elevated in outfall sediments (Figure 4-1). Further review of the outfall sediments revealed: 1) mean cadmium concentration in fine outfall sediments (1.2 ± 0.4 μg/g) was at the ER-L concentration (1.2 μg/g); 2) mean concentration of DDT in coarse outfall sediments (0.0018 ± 0.0006μg/g) was at the ER-L level for DDT (0.00158 μg/g); and 3) that while PCB 105 in coarse sediments was significantly higher than in coarse reference sediments, total PCB concentrations (1.36 ± 1.90μg/g) in coarse outfall sediments were not different from those in reference sediments and were well below the ER-L level (2.27 μg/g) When individual outfall stations were examined (Figures 4-2 and 4-3) all analytes with statistically higher concentrations and with ER-M criteria (i.e., antimony, cadmium, lead, mercury, pyrene, total DDT) were below ER-M values except antimony (54 μg/g vs. 25 μg/g). It should be remembered that ER-M concentrations were chosen in the selection of the in-bay reference stations, since only 3 of the 102 NOAA/San Diego Bay Protection and Toxic Cleanup Program stations had sediment concentrations below ER-L values for all pertinent analytes. While three PCB congener (PCB 101, 105, and 118) concentrations were higher than reference at many stations, absolute concentrations were low and no individual station had total PCB concentrations above reference concentrations or total PCB ER-L values. The relatively high values of barium, lead, molybdenum and tin at station 8-1 probably reflect metal working activities related to electrical components and/or high resistance metal surface particulates in that outfall's drainage (see Figure 6-7). None of these metals bioaccumulated significantly, nor was any toxicity evident at this station; and it can be inferred that these metals are in a non-bioavailable, insoluble salt or metallic form.

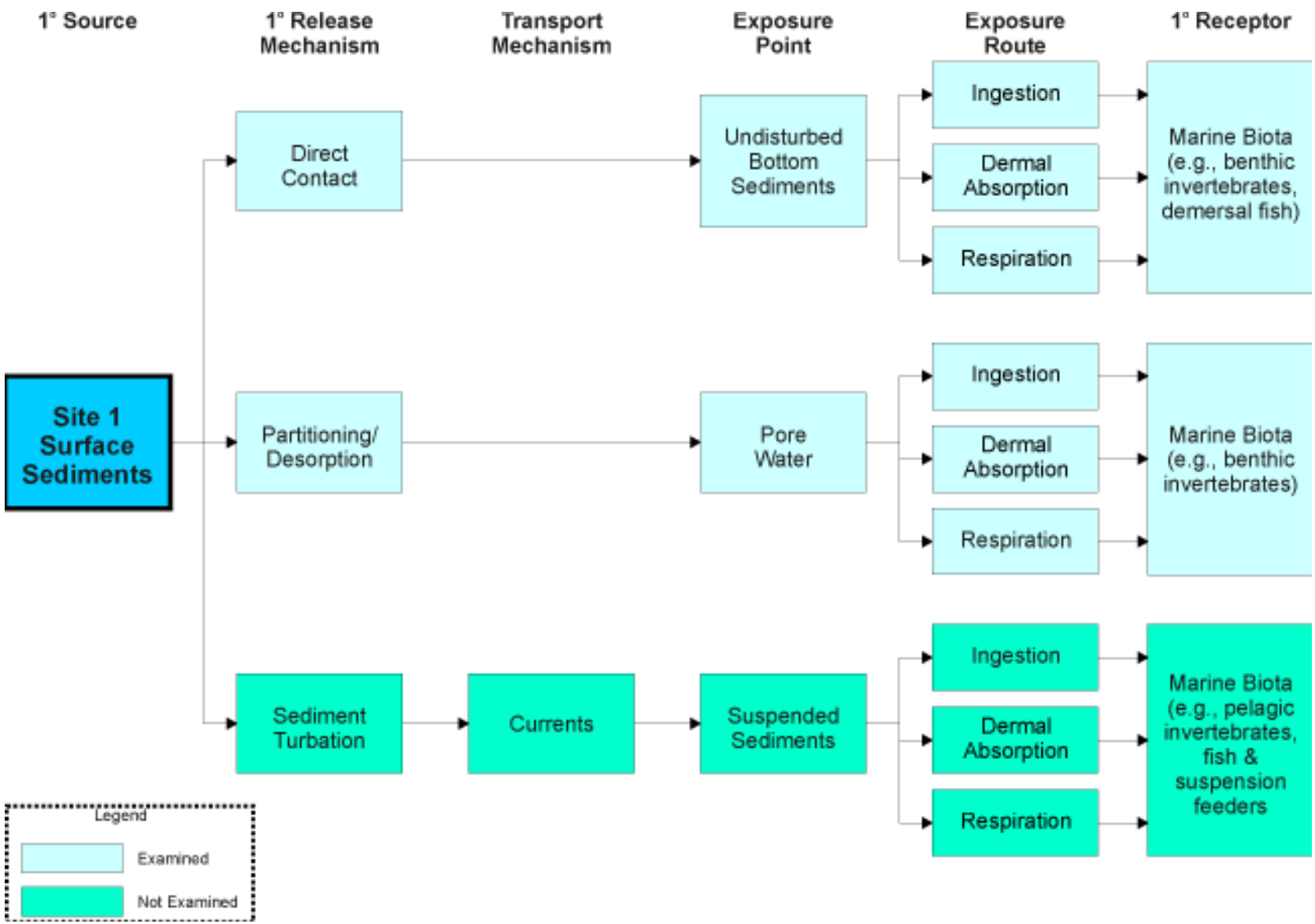


Figure 6-1. Conceptual site model for ecological receptors - NAS North Island.

The results of the toxicity tests are very straightforward. No significant differences were found for the amphipod, mysid, polychaete or oyster larvae tests between grouped outfall and reference sediments (see Section 5, Figure 5-1). When individual outfall stations were examined (see Figure 5-2), only polychaete survival was significantly less at several outfall stations, relative to those exposed to reference sediments. This increased mortality is not felt to be noteworthy, however, because absolute survival at all outfall stations exceeded the minimum 90% level expected from polychaete controls in clean sediment.

Results from the bioaccumulation tests are also quite straightforward. Out of the 101 chemicals measured in clam tissue, only silver was bioaccumulated to a significantly higher concentration in the 28-day test when grouped station results were examined. When individual outfall stations were examined, silver again predominated with eight stations having significantly higher bioaccumulation than reference stations (see Figure 5-3). These were coarse grain stations and spread out along the perimeter of North Island in shallow water from outfalls 4 to 8. Four other contaminants each bioaccumulated significantly at only one or two stations. The bioaccumulation of silver in clams was not felt to pose an ecological threat because it was at the low end in the range of tissue concentrations found in mussels and gastropods from estuaries along the west coast. Sediments in which bioaccumulation of silver occurred had silver concentrations below the ER-L level (1.0 μg/g) (Kennish 1992).

6.1.4 Data Trends

The sediment chemistry, toxicity and tissue bioaccumulation data were evaluated for two different patterns. The first pattern was spatial trends, specifically for a correlation between the above three parameters and proximity to the outfalls. The sampling pattern had been designed assuming that contaminants from the outfalls would tend to adsorb quickly to the sediments and that their sediment concentration would fall off logarithmically with distance. Only data from the 24 coarse stations were pooled and analyzed since the six fine-grained stations, most at the farthest distance from their respective outfalls, offered too little range in distance to test. Elimination of the fine sediment stations also removes the bias expected from increased contaminant adsorption to these sediments. Simple log regression between each analyte concentration in sediment and tissue, as well as toxicity, versus distance from the pooled outfalls was performed. For sediment chemistry, no analyte significantly ($p \leq 0.05$) decreased in concentration with distance, but eight analytes (aluminum, beryllium, iron, manganese, mercury, silver, vanadium and monobutyltin) increased. In the toxicity tests, only amphipod survival was significantly correlated with distance, and it was negative (i.e., it decreased with distance from the outfalls). In the bioaccumulation data three analytes (antimony, copper and silver) all decreased in tissue concentration away from the outfalls, but five analytes increased (barium, iron, acenaphthene, 4,4'-DDE, and PCB 170). Regression results are presented in Volume II.

A second pattern that was searched for was some co-occurrence between toxicity, sediment contaminant concentration and bioaccumulation. None was very evident. Table 6-2 lists the sediment and tissue concentrations for chemicals that were significantly higher than reference concentrations at outfall stations that had any sign of elevated toxicity.

Recall that only polychaete survival was significantly depressed compared to reference stations (i.e., $p \leq 0.05$) and all survival was greater than 90%. Recall too that DDT, pyrene, and mercury were below ER-M values, and total PCBs were below ER-L values. No such values are published for hexachlorobenzene.

Table 6-2. Elevated chemicals in sediment and clam tissue at stations with significant toxicity.

Station	Bioassay	Chemical elevated in sediment	Chemical elevated in tissue
3-2	Polychaete survival	Pyrene, DDT	
3-4	Polychaete survival	Hexachlorobenzene (HCB)	
7-2	Polychaete survival	DDT	
7-3	Polychaete survival	DDT, PCB 101, PCB 105, HCB	
3-4	Polychaete growth	HCB	
4-2	Polychaete growth	Barium	Silver
4-3	Polychaete growth	DDT, PCB 105	Silver
4-5	Polychaete growth	PCB 105	
6-3	Polychaete growth	DDT, PCB 105, Mercury	
7-3	Polychaete growth	DDT, HCB, PCB 101, PCB 105	
6-5	Amphipod survival		

6.1.5 Ecological Risk Screening Conclusions

There is a lack of an apparent pattern or consistency in individual stations that had a joint occurrence of statistically higher sediment contaminant concentrations, toxicity, and bioaccumulation. Outfall stations that were significantly different from reference stations of appropriate grain size had relatively low sediment and tissue concentrations and high overall survival. These two observations argue that "hot spots" of contamination with significant ecological impact do not exist for in-bay Site 1 sediments, and the evaluation of sediments grouped by grain size is reasonable for the site. The ecological evaluation for the two outfall sediment groups is shown in Table 6-3. This evaluation tempers the few significant differences observed between outfall and reference sediment chemistry and bioaccumulation results with the wider perspective of ER-L sediment guidelines. Statistically significant elevated outfall chemistry concentrations were at or below ER-L levels and bioaccumulation of silver occurred at low levels relative to other west coast estuarine animals. Further, the tissues that had statistically elevated silver concentrations were exposed to sediments with silver concentrations below the ER-L sediment quality guideline. From these results, no further action is recommended for Site 1 in-bay surface sediments.

Table 6-3. Ecological evaluation (applied decision matrix) for Site 1 in-bay surface sediments.

Sediment Group	Chemistry	Toxicity	Bioaccumulation	Action
Coarse grain	-	-	-	no further action
Fine grain	-	-	-	no further action

6.2 Human Health Risk Assessment Screening

A human health risk assessment screening was performed using surface or near-surface sediment chemistry data from in-bay outfalls 3 through 8, and the freshwater channel outfalls 1,2 and 16 on the ocean side of NAS North Island (see Figure 2-2, Section 2). This assessment evaluates the potential for chemicals of concern to cause increased probability of human cancer as well as non-cancerous deleterious health effects in individuals exposed to contaminated sediments. Uncertainty associated with all risk parameters lead to conservative assumptions protective of human health. Major uncertainties are: 1) extrapolation of toxic effects observed at high doses to predicted effects that occur at low doses encountered in the environment; 2) use of dose-response data from short-term exposure studies to predict the effects of long-term exposures; 3) extrapolation from observable toxic effects in animals to toxic effects in man; 4) use of dose-response data from homogenous animal populations or healthy human populations to predict effects likely to occur in the general population of individuals with a wide range of sensitivities; and 5) the assumption of no threshold contaminant concentration in the induction of cancer. Uncertainty factors, ranging from 1 to 10,000 depending on the type of study from which the value has been derived (e.g., animal or human, chronic or acute), are incorporated into reference doses to provide an extra level of public health protection.

Health risk is limited to those chemicals for which data on human health impacts exist and limited to exposure pathways that are considered reasonable and relevant for this site. Base workers, in an industrial setting, are the population most likely to be exposed to sediments at Site1. This screening evaluation is based on lifetime residential exposure parameters, however, to be protective. Risk estimates are quite conservative and should be considered as upper estimates of additional cancer risk or non-cancer hazard as a result of exposure to Site 1 sediments.

6.2.1 Objectives and Approach

The objectives of this human health risk assessment screening are: 1) to assess potential human health effects associated with exposure to contaminated sediments, assuming certain exposure pathways; and 2) to support environmental decisions regarding needs for either further investigation, a full human health risk assessment, or remedial actions. Primary steps of this health risk assessment screening are to:

- Identify chemicals that pose potential risk or hazard ;
- Identify exposure setting and pathways in a conceptual site model;
- Quantify the cancer risk and non-cancer hazard for Site 1.

Screening evaluations use risk-based and hazard threshold criteria that relate contaminant concentrations in specific media to cancer risk and non-cancer hazard. This approach differs from that used for the ecological assessment because human risk-based criteria exist for most of the Site 1 chemicals analyzed, whereas no similar criteria exist for the evaluation of ecological health.

6.2.2 Chemicals Screened

Cancer risk and non-cancer hazard indices (Section 6.2.5) were calculated for all chemicals analyzed (other than those few listed in Table 4-9) for which 1998 preliminary remediation goals (PRG) values for soil were available. The PRG values are soil chemicals concentrations which result in a one in a million ($1x10^{-6}$) incremental human

cancer risk and hazard quotient of 1.0 for non-carcinogenic contaminants. Preliminary remediation goals values are based on either a residential (more conservative) or industrial-type exposure. Chemicals evaluated for cancer risk and non-cancer hazard are listed in Table 6-4. Some chemicals, such as arsenic, may exhibit both carcinogenic and non-carcinogenic effects. Often, non-cancerous hazard concentrations for chemicals are simply 10-fold or 100-fold multiples of the cancer risk concentration; those chemicals are noted with asterisks in Table 6-4. Many PAHs, all PCBs, and some pesticides do not have hazard concentrations and were not evaluated for non-cancer hazard. For carcinogens, the toxic end point is always cancer. For non-carcinogens, the toxic end point (e.g., kidney or liver damage) varies among chemicals and routes of exposure (e.g., inhalation or ingestion). National and superceding California 1998 PRG values in Table 6-4 were downloaded from the EPA Region 9 PRG home page: http://www.epa.gov/region09/waste/sfund/prg/index.htm.

Table 6-4. Chemicals and corresponding 1998 residential soil preliminary remediation goals (PRGs) (μg/g).

Chemical	PRG	Chemical	PRG
Metals		**Pesticides & PCBs (continued)**	
Antimony	30 nc	Dieldrin	0.028 ca*
Arsenic	0.38 ca	Total Chlordane	1.6 ca*
Arsenic	21 nc	Total DDE	1.7 ca
Barium	5200 nc	Total DDD	2.4 ca
Beryllium	150 nc	Total DDT	1.7 ca*
Cadmium*	9 nc	Total Endosulfan	330 nc
Total Chromium (1/6 ration Cr^{6+}/Cr^{3+})	210 ca	Endrin	16 nc
Cobalt	3300 nc	Hexachlorobenzene	0.28 ca
Copper	2800 nc	Total PCBs	0.2 ca**
Lead *	130 nc	**Semivolatile Organic Compounds**	
Manganese	3100 nc	Acenaphthene	2600 nc
Mercury (elemental)	22 nc	Anthracene	14000 nc
Molybdenum	370 nc	Benzo[a]anthracene	0.56 ca
Nickel*	150 nc	Benzo[b]fluoranthene*	0.56 ca
Selenium	370 nc	Benzo[k]fluoranthene*	0.61 ca
Silver	370 nc	Benzo[a]pyrene*	0.056 ca
Thallium	5.2 nc	Biphenyl	2300 nc
Tin (inorganic)	45000 nc	Bis[ethylhexyl]phthalate	32 ca*
Vanadium	520 nc	Chrysene	6.1 ca
Zinc	22000 nc	Dibenzo[a,h]anthracene	0.056 ca
Tributytin	16 nc	Fluoranthene	2000 nc
Pesticides & PCBs		Fluorene	1800 nc
Aldrin	0.026 ca*	Indeno[1,2,3-c,d]pyrene	0.56 ca
alpha-BHC	0.086 ca	Naphthalene	55 nc
beta-BHC	110 nc	Phenol	33000 nc
gamma-BHC	0.42*	Pyrene	1500 nc

ca = cancer risk value; nc = non-cancer hazard value;
ca*=non-cancer hazard value=100 x ca; ca**=non-cancer hazard value=10 x ca
PRG source: EPA Region 9; *CAL-modified PRG (CAEPA 1998)

6.2.3 Exposure Setting

The exposure setting is defined by current and potential future land use, contaminant concentrations, exposed populations, exposure pathways, and chemical intake under maximum exposure assumptions. No attempt was made to evaluate risk associated with exposures that may have occurred in the past. Demographics and general land use are discussed in Section 1.4.5. This section focuses on current and potential future land use, and identifying likely human receptors.

The City of Coronado, including areas immediately adjacent to NAS North Island, is a residential and resort community zoned for low density residential uses. The city has been fully developed for many years, except for the recent additions of the Coronado Cays waterfront subdivision and the commercial/hotel site adjacent to the Coronado Ferry Landing. The Hotel Del Coronado complex, Civic Center complex, and commercial properties along Orange Avenue comprise most of the city's commercial area. Apartments have been developed along Orange Avenue, as well as in other central areas of the city and in the northeast portion of the city adjacent to the tidelands. The City of Coronado General Plan indicates no significant changes to current land use zoning, especially with regard to the area just east of NAS North Island. Some small lots could be developed into housing or small commercial enterprises in the future. However, the city is discouraging development of high population density housing and favoring low density, single family housing. This trend should be expected to continue into the future. The State Coastal Commission as well as the City of Coronado government regulates new development. Consequently, any new development is subject to considerable scrutiny before permits are issued.

Figure 6-2 depicts current land use for NAS North Island. A wide sandy beach, used primarily for military recreation, extends along the Station's southern perimeter. Outfalls 1,2 and 16 are near the golf course in the southern recreational area that is currently zoned as Recreation/Community Support. In-bay outfalls 3-8 are located on the western and northern portions of the peninsula in areas currently zoned as Operations or Maintenance. Outfall 3 is located on the inside of Pier Bravo in a heavily secured and restricted area. Outfalls 1,2 and 4 are fenced but unguarded; all other outfalls are reasonably accessible to people with clearance to enter the station, including base workers and dependents. Naval dependents are most likely to gain access to outfall 16, since it is near a recreational golf course and beach. The rest of the outfalls are in operational areas that are not easily accessible to dependents but are accessible to station workers. Base workers have access to only the shallowest areas around outfalls 5 and 6, and all of 16. A1lthough access to NAS North Island is controlled, it is possible that trespassers could attain temporary access to these outfall areas before being questioned.

NAS North Island land use patterns along the waterfront on southern, western and northern perimeters are expected to remain unchanged (Figure 6-3). The Recreational/Community Support area is projected to retain the same use designation. The western side of the perimeter is projected to remain as Operations. Some minor changes in the northern perimeter currently designated as Maintenance will revert to Operations or Supply. Despite these minor changes, the northern perimeter will retain the same basic industrial character.

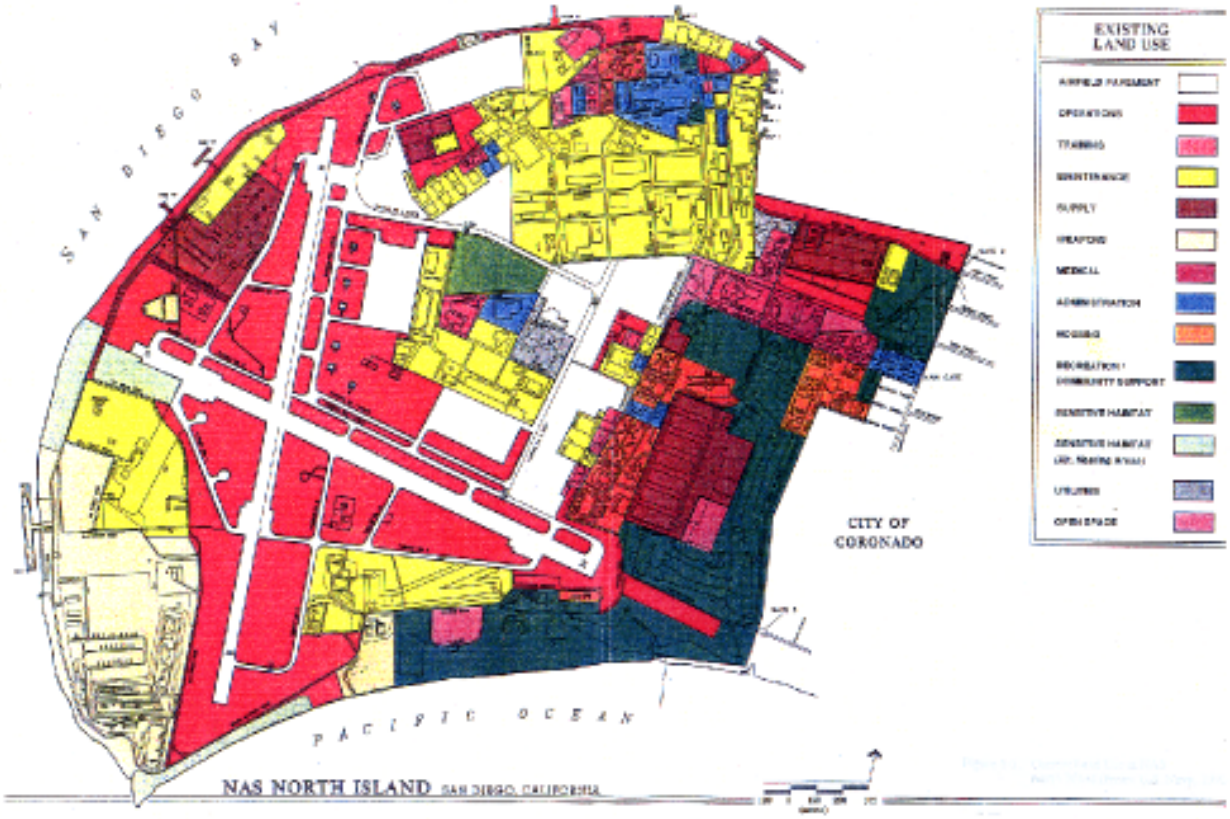


Figure 6-2. Current land use at NAS North Island (from U.S. Navy 1991).

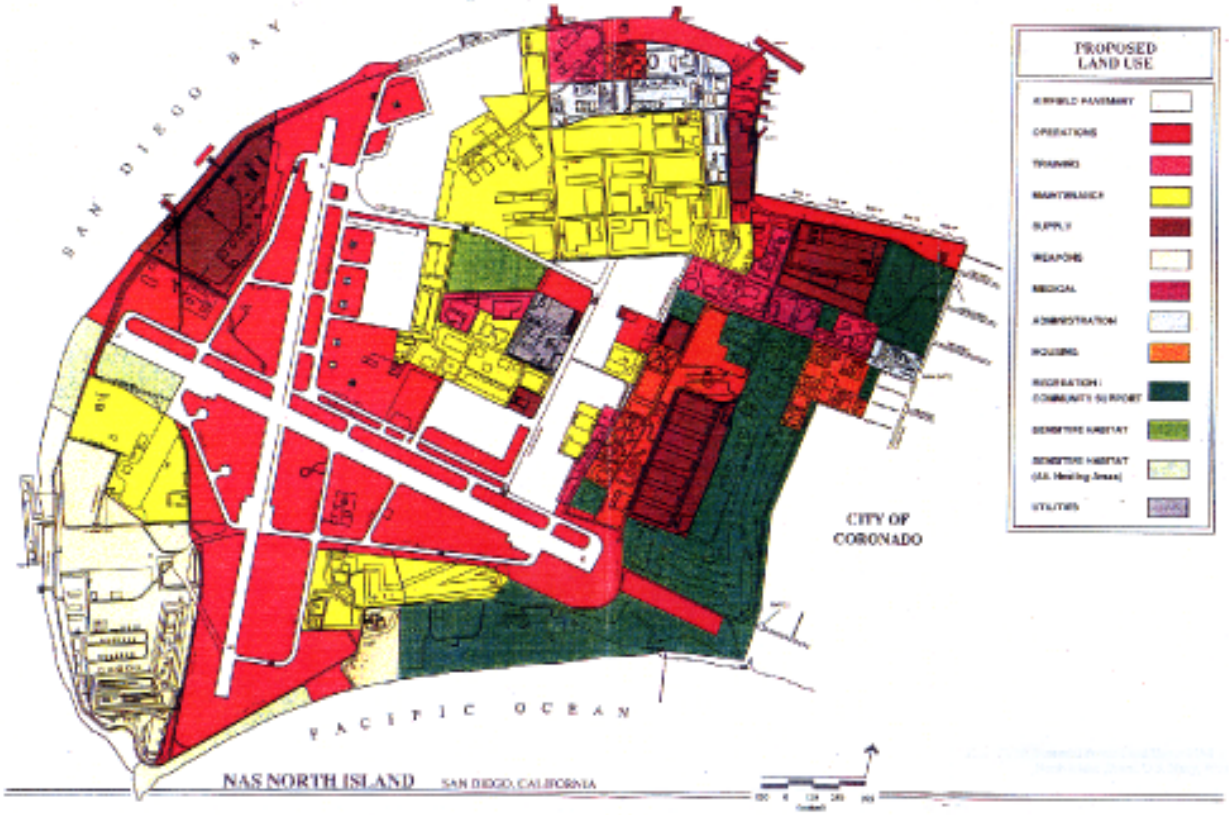


Figure 6-3. Potential future land use at NAS North Island (from U.S. Navy 1991).

6.2.4 Exposure Pathways and Conceptual Site Model

To screen for human health risk, Site 1 was arbitrarily divided into two areas. The first, collectively called the intertidal area, include the two freshwater channels associated with outfalls 1,2 and 16, and intertidal stations around in-bay outfalls 4, 5 and 6. The second called the subtidal area, include the remaining in-bay subtidal outfall stations which remain underwater at mean lower low water. Intertidal and subtidal areas were separated because of differences in potential exposure pathways. Sediments in the intertidal area are accessible during some portion of the tidal cycle; subtidal sediments are nearly always underwater, and although evaluated, they currently have no reasonable exposure pathways for humans. Intertidal and subtidal designations for in-bay stations are listed in Table 1-4 (Section 1). Only sediment chemical data from surface grab samples (outfalls 3-8) or the 0-1 ft subsurface cores (outfalls 1,2 and 16) were used in this health risk assessment. Exposures to contaminated surface waters were considered negligible since significant partitioning of contaminants from sediments into bay and/or ocean waters are unlikely. Data from past investigations were not used because the data quality was unknown. Also, data from subsurface sediments collected below 1 ft were not used because no reasonable exposure pathway exists.

The conceptual site model for human exposure is illustrated schematically in Figure 6-4. The most significant exposure pathway to humans is direct contact with contaminated intertidal sediments that are accessible during at least part of the tidal cycle. Population sectors most likely exposed to these sediments are base workers, trespassers and military dependents. Although human consumption of contaminated seafood (e.g., via pier fishing) is another potential pathway for human exposure, health risks associated with ingestion of biota were not addressed here. This is reasonable since a surrogate organism (the bentnose clam) exposed to Site 1 sediments were shown not to appreciably accumulate contaminants that could potentially biomagnify through the food web (Section 5.5). Furthermore, a direct relationship cannot be established for consumed contaminated fish that are caught in the vicinity of Site 1 and contaminated Site 1 sediments because of the mobility of recreational fish.

6.2.5 Cancer Risk and Non-Cancer Hazard

Cancer risk estimates were assessed using the lower of either 1998 USEPA or California PRG values for cancer. Preliminary remediation goals exist for residential and industrial receptors. While industrial PRGs are most appropriate for baseworkers, conservative residential PRGs were used in this screening level risk assessment to account for military dependents and unrestricted future use. These PRG values assume exposure pathways of inadvertent ingestion, inhalation of particulates and dermal absorption over a 70-year exposure period. Complete results are included in Volume II, Section Z. Cancer risks and non-cancerous hazards were calculated by dividing the sediment concentration measured at a station by its corresponding risk or hazard PRG value for each chemical. The overall station risk or hazard was calculated by summing the respective ratios of individual chemicals in Table 6-4 as below:

$$\text{Cancer Risk} = \left[\sum_i \frac{\text{Conc } i}{\text{PRG } i} \right] x10^{-6} \qquad \text{Non-Cancer Hazard Index} = \sum_i \frac{\text{Conc } i}{\text{PRG } i}$$

Table 6-5 presents the summed cancer risk and non-cancerous hazard index using 1998 USEPA and California EPA residential PRGs for Site 1 and reference sediments. According to the USEPA, a cumulative lifetime cancer risk of 10^{-6} (i.e., 1 in 1,000,000 people will develop cancer due to exposure to contaminants) or greater should

trigger an evaluation of risk management options to limit human exposure. For example, these options include posted warnings, making the site inaccessible, and site remediation. The target cumulative hazard index used in the risk-based PRG concentrations calculated for non-cancerous health effects is 1.0. These values were used in this assessment as conservative criteria to support environmental decisions regarding needs for either further investigation, a full human health risk assessment, or remedial actions. Values in the metals columns sum the risk and hazard posed by those metals listed in Table 6-4. Values in the PAHs columns sum the risk and hazard posed by the fourteen PAH compounds (including biphenyl) listed as semivolatile organics in Table 6-4. Phenol and bis(ethylhexyl)phthalate are summed separately in Table 6-5. Pesticide and PCBs values sum the risk and hazard posed by pesticides listed in Table 6-4. The cancer risk value for all stations was less than 5×10^{-5}, the maximum value of 44×10^{-6} occurring at subtidal station 8-1, the maximum intertidal value of 16×10^{-6} occurring at station 16-1.

Cancer risk values at nearly all stations were due to a combination of arsenic and a group of six PAHs out of the fourteen examined. These six PAHs were benzo[a]anthracene, dibenzo[a,h]anthracene, benzo[b] and benzo[k]-fluoranthene, indeno[1,2,3-cd]pyrene, and chrysene. Figures 6-5 and 6-6 depict total risk and its major components at all stations, and at just the intertidal stations respectively. In these figures, the blue line represents the risk from arsenic alone, the green line represents risk from arsenic plus risk from PAHs, and the red line represents the total risk calculated from all chemicals in Table 6-4. The mean risk posed by arsenic in fine reference sediments, coarse reference sediments and background North Island soils (NASNI 1995) is plotted for comparison. Reference sediment concentrations of arsenic and calculated risk are relevant because North Island was primarily created from bay dredge material.

The non-cancer hazard maximum value of 7 was found at intertidal station 16-2. Hazard values at all stations were attributable primarily to metals. Figure 6-7 depicts the sequentially added hazard posed by the six most important metal contaminants found at the outfall stations. The black dashed line represents total hazard from all chemicals and is indistinguishable from the red line that represents hazard posed by summing metals alone. Again, the summed metal hazard from the fine and coarse reference sediments and North Island background soils is also plotted. The high background lead level in North Island soils is attributed to the long history of aviation on the island. The roughly parallel lines of incremental hazard posed by metals across all outfall stations are noteworthy because they suggest that North Island activities have not changed the relative abundance of those metals which constitute the bulk of the hazard, particularly in the in-bay outfall stations. Table 6-6 lists nine organ systems effected by the six most important metal contaminants contributing to the summed hazard index. Toxicity data for most metals were taken from *ToxFAQs* at the Agency for Toxic Substances and Disease Registry website http://atsdr1.atsdr.cdc.gov:8080/tfacts13.html. Toxicity data for manganese were taken from the EPA's Technology Transfer Network website http://www.epa.gov/ttnuatw1/hlthef/manganes.html. Lead, the largest contributor, effects five systems. Many of the systems are impacted by at least three metals. Figure 6-8 replots the summed hazard index of the six metals for the nine individual organ systems. The values for the five-metal hazard index sum for the nervous system, six-metal index sum for the whole body, and the all-contaminant index sum for the whole body at station 1-1 (1.67, 2.01, 2.03, respectively) and 16-2 (6.26, 6.86, 7.01, respectively) are similar within each station. This similarity underscores the fact that, due to the general susceptibility of the central nervous system to metal poisoning, partitioning contaminant hazard to specific organ systems does not appreciably change the overall human hazard at these stations. The intertidal channels, intertidal in-bay stations, and subtidal in-bay stations are reviewed separately below.

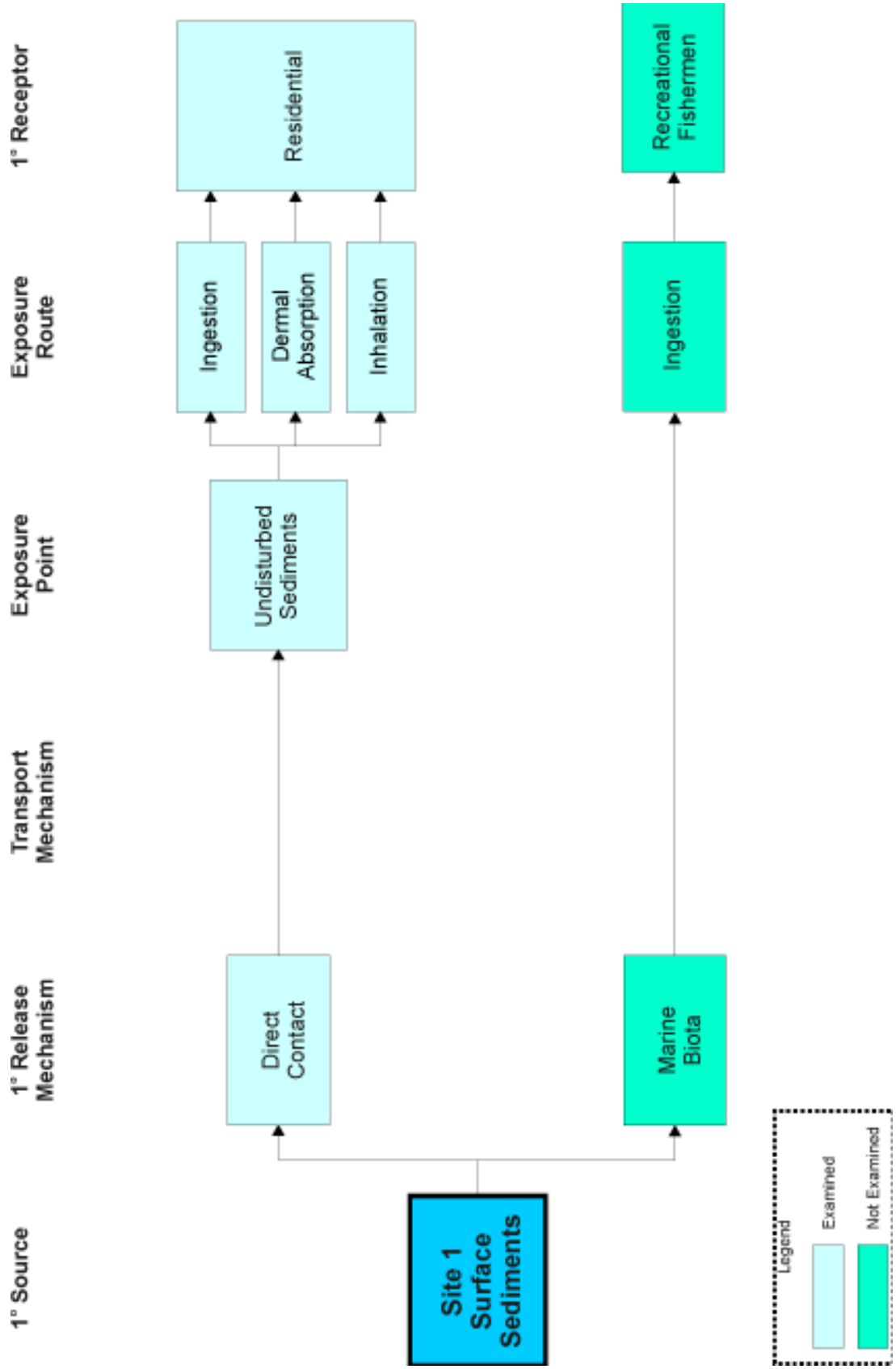


Figure 6-4. Conceptual site model for human health screening.

Table 6-5. Residential cancer risk and hazard values based on 1998 PRGs.

	CANCER RISK ($X10^{-6}$)					NON-CANCER HAZARD				
Station	Metals	PAHs	Phenol, Phthalate	PCBs, Pesticides	Total	Metals	PAHs	Phenol, Phthalate	PCBs, Pesticides	Total
1-1	5.29	1.46	0.06	1.31	8.12	2.01	0	0	0.02	2.03
1-2	2.97	0.07	0	0.07	3.11	0.54	0	0	0	0.54
1-3	2.44	0.02	0.02	0.08	2.56	0.5	0	0	0	0.5
1-4	3.51	0.36	0	0.09	3.96	0.61	0	0	0.01	0.62
16-1	5.17	9.83	0.08	0.58	15.66	1.42	0.01	0	0.05	1.48
16-2	5.2	7.74	0.26	1.71	14.91	6.86	0	0	0.15	7.01
16-3	3.71	0.34	0.01	0.13	4.19	0.95	0	0	0.01	0.96
16-4	3.54	0.11	0.01	0.08	3.74	0.66	0	0	0	0.66
16-5	3.99	0.08	0	0.07	4.14	0.41	0	0	0	0.41
3-1	6.69	0.73	0	0.08	7.5	0.72	0	0	0	0.72
3-2	10.91	11.4	0	0.11	22.38	0.81	0	0	0.01	0.82
3-3	9.75	2.16	0	0.13	12.04	0.84	0	0	0.01	0.85
3-4	9.86	0.53	0	0.11	10.5	0.8	0	0	0.01	0.81
3-5	8.28	0.29	0	0.09	8.66	0.73	0	0	0	0.73
4-1	5.33	0.1	0	0.07	5.5	0.57	0	0	0	0.57
4-2	6.14	0.17	0	0.06	6.37	0.66	0	0	0	0.66
4-3	12.05	1.5	0	0.29	13.84	1.07	0	0	0.01	1.08
4-4	14.4	0.42	0	0.09	14.91	1.05	0	0	0.01	1.06
4-5	15.05	0.43	0	0.09	15.57	1.08	0	0	0.01	1.09
5-1	10.71	0.06	0	0.11	10.88	1.02	0	0	0.01	1.03
5-2	18.74	0.06	0.01	0.1	18.91	1.29	0	0	0.01	1.3
5-3	10.4	0.09	0	0.15	10.64	0.81	0	0	0.01	0.82
5-4	24.58	2.12	0.01	0.33	27.04	1.74	0	0	0.03	1.77
5-5	14.41	0.35	0	0.13	14.89	1.07	0	0	0.01	1.08
6-1	6.26	1.94	0.01	0.12	8.33	0.9	0	0	0.01	0.91
6-2	5.94	0.93	0	0.08	6.95	0.88	0	0	0.01	0.89
6-3	16.07	1.1	0	0.18	17.35	1.39	0	0	0.02	1.41
6-4	19.25	1.27	0	0.21	20.73	1.51	0	0	0.02	1.53
6-5	22.66	0.95	0	0.16	23.77	1.46	0	0	0.01	1.47
7-1	20.44	0.06	0	0.07	20.57	1.72	0	0	0	1.72
7-2	8.24	0.08	0	0.08	8.4	0.63	0	0	0	0.63
7-3	12.27	0.82	0.01	0.3	13.4	0.97	0	0	0.02	0.99
7-4	18.11	3.53	0.08	0.42	22.14	2	0	0	0.04	2.04
7-5	24.27	1.57	0.01	0.21	26.06	1.62	0	0	0.02	1.64
8-1	44.06	0.03	0	0.06	44.15	3.88	0	0	0	3.88
8-2	10.61	0.11	0	0.05	10.77	0.82	0	0	0	0.82
8-3	6.94	0.07	0	0.14	7.15	0.69	0	0	0.01	0.7
8-4	12.85	0.92	0	0.14	13.91	1.12	0	0	0.01	1.13
8-5	25.09	2.24	0.01	0.26	27.6	1.75	0	0	0.02	1.77
R-1	10.67	0.04	0	0.47	11.18	0.86	0	0	0.05	0.91
R-2	10.65	0.9	0	0.08	11.63	0.8	0	0	0.01	0.81
R-3	8.29	0.53	0	0.08	8.9	0.76	0	0	0	0.76
R-4	18.9	0.09	0	0.08	19.07	1.16	0	0	0.01	1.17
R-5	18.93	0.09	0.01	0.08	19.11	1.24	0	0	0.01	1.25
R-6	27.59	2.33	0.01	0.48	30.41	1.99	0	0	0.05	2.04
R-7	22.46	3.15	0.02	0.42	26.05	1.82	0	0	0.04	1.86
R-8	28.87	3.74	0.03	0.6	33.24	2.1	0	0	0.05	2.15
R-14	16.31	1.13	0.01	0.17	17.62	1.37	0	0	0.01	1.38
R-15	17.36	1.3	0.01	0.15	18.82	1.26	0	0	0.01	1.27

6.2.6 Cancer Risks for Ocean Outfalls

6.2.6.1 Ocean Outfall 1,2

The highest cancer risk value for outfall 1,2 was 8.12 x 10^{-6} at station 1-1, primarily driven by arsenic (5.29 x 10^{-6} at station 1-1) and six summed PAHs (1.46 x 10^{-6} at station 1-1), best shown in Figure 6-6. Additional risk is posed by the two DDD isomers measured. Concentrations of these contaminants tended to decrease from the outfall. Figure 6-7 also shows metal concentrations falling quickly with distance from the outfall. The most important metals at station 1-1, in terms of their hazard, were lead (0.85), cadmium (0.52), vanadium (0.16), barium (0.13), arsenic (0.09), and manganese (0.07). The metals posed the most hazard to the central nervous system, and secondarily to the kidney system.

6.2.6.2 Ocean Outfall 16

Outfall 16 posed the greatest cancer risk and non-cancer hazard of any accessible site in this study. The highest cancer risk value for outfall 16 was 15.6 x 10^{-6} at station 16-1, driven primarily by six PAHs (9.83 x 10^{-6}), arsenic (5.2 x 10^{-6}) and to a small degree by PCBs (0.58 x 10^{-6}). The risk at station 16-2 was 14.91 x 10^{-6} and was due to the same compounds. The impact of PCBs (1.71 x 10^{-6}) was more important at station 16-2 than any other outfall station. Risk quickly diminished away from these two stations. Hazard, dominated by metals, was relatively low at station 16-1 (1.48) and highest at 16-2 (7.01). The order of hazard-contributing metals at 16-2 was lead (3.59), cadmium (2.31), antimony (0.2), nickel (0.16) barium, (0.15) manganese, (0.12), and vanadium (0.1). The hazard order of metals at 16-1, for comparison, was lead (0.49), cadmium (0.24), manganese (0.16), barium (0.13), vanadium (0.12), arsenic (0.09), and nickel (0.07). The elevated rank of nickel at stations 16-2 relative to sites 1-1 or 16-1, in conjunction with the high lead and cadmium values may indicate contamination from battery waste at the site. Hazard values quickly diminish away from sites 16-1 and 16-2 as well. The metals posed the most hazard to the central nervous system, and secondarily to the kidney system.

6.2.6.3 Intertidal Stations at In-Bay Outfalls 3-8

Outfall stations 4-1, 4-2, 5-1 and 6-1 were above water at mean lower low water tidal conditions and were considered intertidal. Cancer risk was relatively low, highest at station 5-1 (10. 99 x 10^{-6}) and dominated by arsenic. Similarly the hazard value was low, highest again at 5-1 (1.03) and due to a metal mix similar to other stations (Figure 6-7). The metals posed the most hazard to the central nervous system, and secondarily to the respiratory system.

6.2.6.4 Subtidal Stations at In-Bay Outfalls 3-8

The remaining outfall stations are subtidal (i.e., below mean lower low water) and are not screened for their impact from human exposure. They are included with the reference stations in Figures 6-5, 6-6 and 6-7, however. to emphasize how similar their chemical makeup is to the stations mentioned above. The figures show that arsenic again constitutes almost all of the cancer risk, and the same metals in similar proportions constitute most of the hazard. It appears that activities on North Island have not resulted in a distinctive chemical signature at most of the in-bay outfalls. Cancer risk, other than that near outfall 3 in the vicinity of Bravo Pier, is dominated almost exclusively by arsenic and reaches its highest value at station 8-1. Risk at outfall 3 originates from the six

PAHs as well as arsenic, particularly at stations 3-1, 3-2, and 3-3 that are within the pier area and near creosote-soaked pilings. The hazard value is highest at 8-1 (3.88) and is again due to metals, with a more predominant role played by antimony and molybdenum. The hazard ranking for metals at station 8-1 was antimony (1.75), arsenic (0.80), lead (0.61), molybdenum (0.23), barium (0.13), manganese (0.09), thallium (0.07), and copper (0.06).

6.2.7 Human Health Risk Screening Conclusions

In summary, of all stations above tidal mean lower low water, stations 16-1 and 16-2 posed the greatest cancer risk (15.66×10^{-6} and 14.91×10^{-6} repectively) due primarily to PAHs and arsenic concentrations. Cancer risks were calculated from conservative assumptions of residential occupancy. Cancer risks at all stations are below10^{-4} and are also similar to risk posed by North Island background arsenic levels alone (14.8×10^{-6}) to risk posed by reference coarse sediment (13.3×10^{-6}) and below that of fine reference sediment (20.1×10^{-6}) from which the island is partially derived. We conclude that little cancer risk is posed by these sites, particularly when compared to background.

Station 16-2 also posed the greatest hazard index (7.01), due to an elevated concentration of common crustal metals, in ratios similar to background soil from North Island and sediments collected throughout the bay. The high background lead concentration is probably due to the heavy use of aviation fuels at the air station. An organ-specific assessment of the metal hazard indicated greatest threat to the central nervous system and kidneys. All other stations had hazard indices below that of the North Island background soil hazard index (4.33). We conclude that the non-cancer hazard posed by sediments at station 16-2 warrants evaluation of remedial altenatives.

Table 6-6. Organ system-specific hazard index for important metal contaminants.

Organ system/metal	Lead	Arsenic	Manganese	Vanadium	Cadmium	Barium	Station 1-1 Total	Station 16-2 Total
Reproduction	x		x				0.93	3.72
Immune	x						0.85	3.59
Kidney	x				x	x	1.5	6.06
Liver					x	x	0.65	2.46
Neural	x	x	x		x	x	1.67	6.26
Dermal		x					0.85	3.59
Respiration		x	x	x	x	x	0.98	2.77
Cardiovascular	x	x				x	1.08	3.83
Gastro/intestinal		x			x	x	0.74	2.55
Whole body 6 metal index sum	x	x	x	x	x	x	2.01	6.86
Whole body total index sum							2.03	7.01

x indicates metals that impact to the organ system

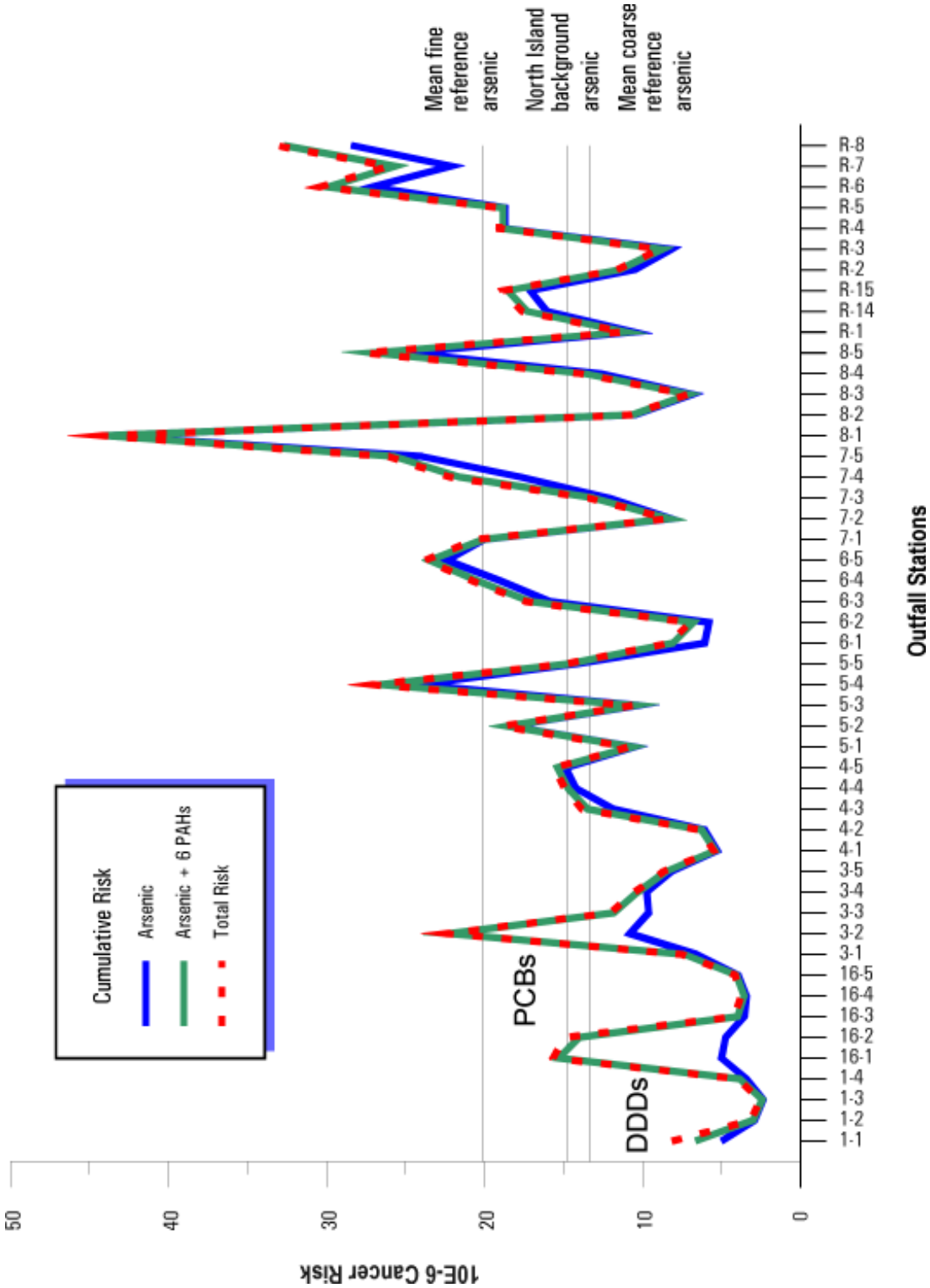


Figure 6-5. Residential cancer risk in surface sediments based on 1998 PRGs.

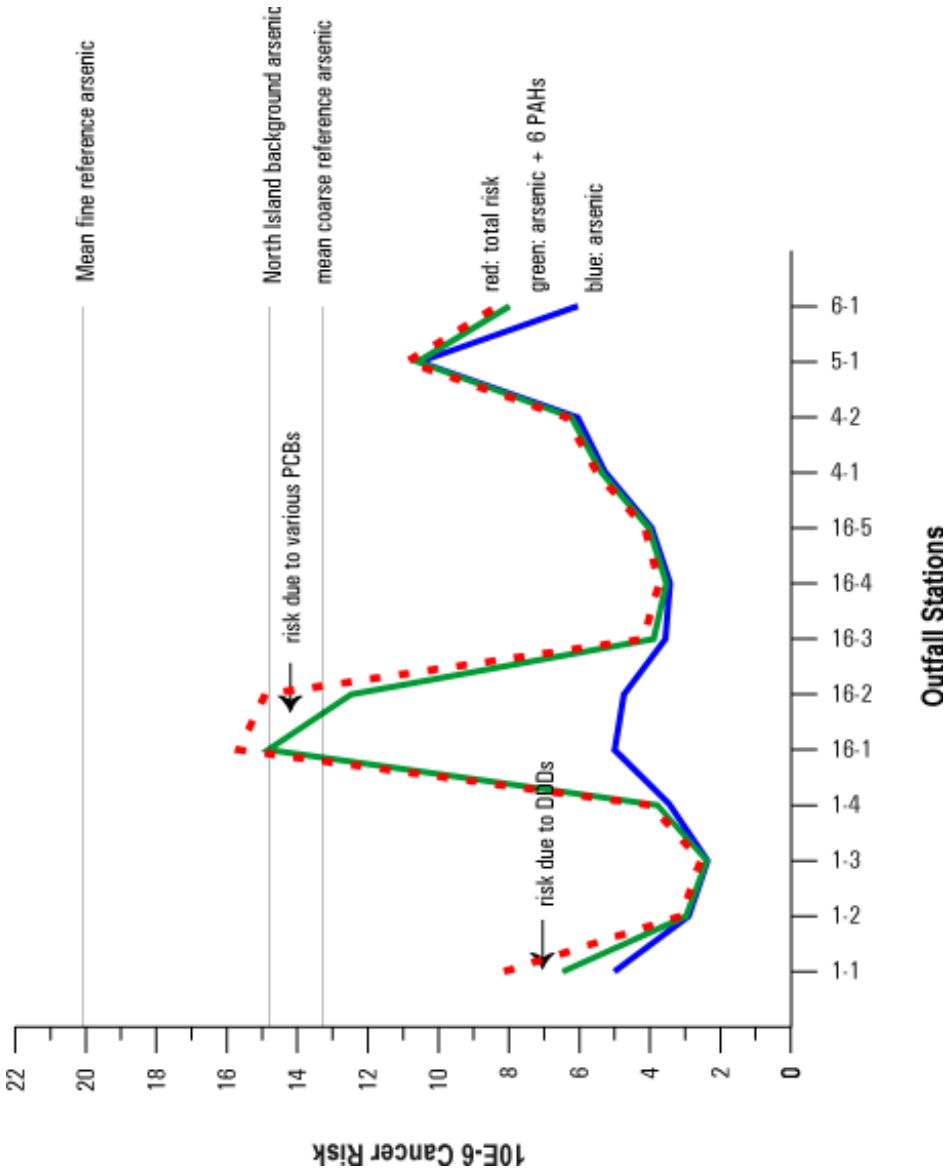


Figure 6-6. Residential cancer risk at intertidal outfall stations based on 1998 PRGs.

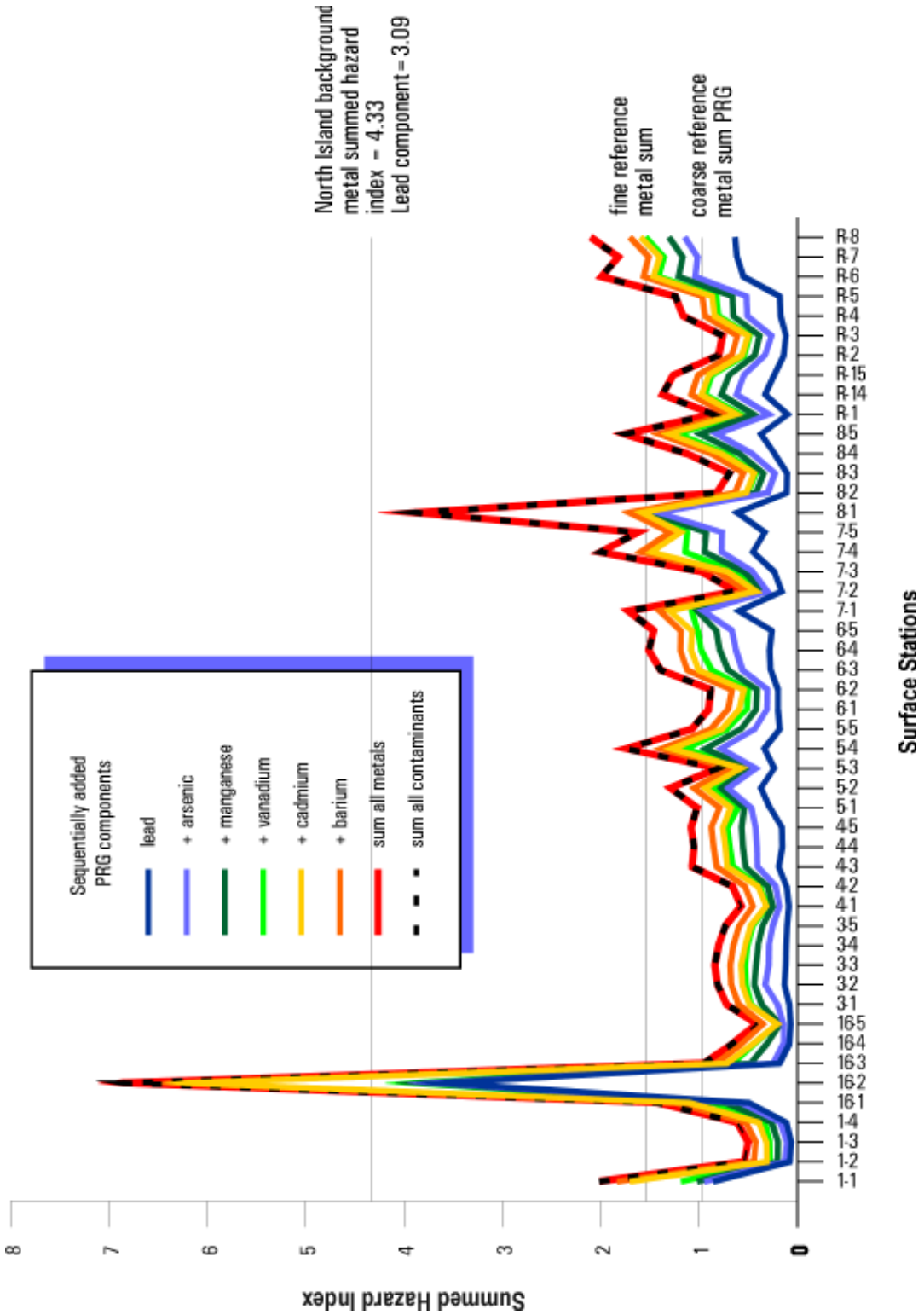


Figure 6-7. Residential non-cancerous hazard in surface sediments based on 1998 PRGs.

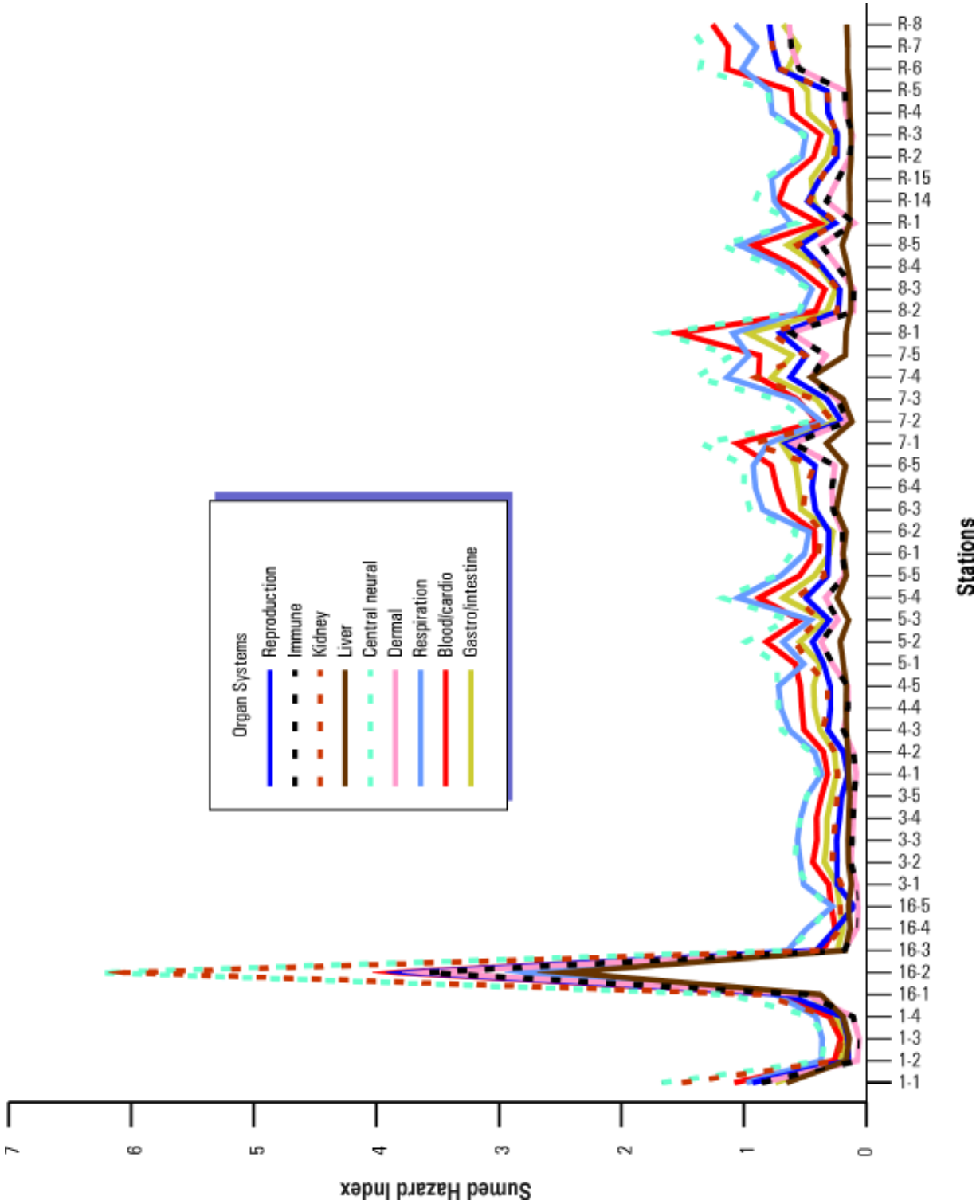


Figure 6-8. Summed hazard index by organ system and station.

Section 7 CONCLUSIONS AND RECOMMENDATIONS

In summary, the screening for ecological and human health risk were pursued through two different approaches. For lack of absolute sediment standards, a relative comparison was made between what were judged to be clean, non-impacted reference sediments and sediments collected around Site 1 outfalls. Significant differences in sediment chemistry, toxicity or bioaccumulation tests were used to indicate potential ecological risk. Any potential risk was then further evaluated by comparing outfall sediment contaminant concentrations with nationwide guidelines. In the case of human health, the luxury of absolute, albeit conservative, contaminant concentrations in the form of preliminary remediation goals could be employed to evaluate risk, making this analysis much more straight forward.

7.1 Ecological Risk Screening Conclusions

There is a lack of an apparent pattern or consistency in individual stations that had joint occurrence of statistically higher sediment contaminant concentrations, toxicity, and bioaccumulation. Outfall stations that were significantly different from reference stations of similar grain size had relatively low sediment and tissue concentrations and high overall survival. These two observations argue that "hot spots" of contamination with significant ecological impact do not exist for in-bay Site 1 sediments, and the evaluation of sediments grouped by grain size was reasonable for the site. It was concluded in the ecological evaluation for outfall fine-grain and coarse-grain sediment groups that neither sediment contaminant concentration, toxicity nor bioaccumulation was elevated relative to in-bay reference stations. This evaluation tempers the few significant differences observed between outfall and reference sediment chemistry and bioaccumulation results with the wider perspective of ER-L and ER-M sediment guidelines. Significantly elevated mean contaminant concentrations in outfall sediments were at or below ER-L levels. Silver, the only chemical significantly bioaccumulated, occurred at low tissue concentrations relative to other west coast estuarine animals. Further, the tissues that had statistically elevated silver bioaccumulations were exposed to sediments with silver concentrations below the ER-L sediment quality guideline. From these results, no further action is recommended for Site 1 in-bay surface sediments.

7.2 Human Health Risk Screening Conclusions

Maximum residential cancer risk associated with intertidal surficial sediments was 15.7×10^{-6} at station 16-1, arising primarily from PAHs (9.8×10^{-6}) and arsenic (5.2×10^{-6}). The cancer risk due to arsenic alone for background North Island soils is 14.8×10^{-6}, and is 13.3×10^{-6} and 21.1×10^{-6} for reference coarse and fine sediments respectively. These latter two values are pertinent since North Island is built up from dredged bay sediments. Cancer risk at most subtidal and reference station sediments is dominated by arsenic and is maximum at station 8-1 (44.2×10^{-6}). These subtidal sediments are below water at mean lower low water level and are ignored in this risk screening because no reasonable exposure pathway to humans exist. The cancer risks in all cases were within the risk range of 1×10^{-4} to 1×10^{-6}. We conclude that little cancer risk is posed by the intertidal sites.

The maximum non-cancer hazard value found associated with intertidal sediments was 7.01 at station 16-2, the next highest value of 2.03 was found at station 1-1, the third highest was 1.48 at station 16-1. All other stations had hazard values of approximately 1 or less. These values where driven exclusively by a similar mix of metals. Lead (3.59) dominated the hazard value at station 16-2. An organ-specific assessment of the metal hazard indicated the greatest threat to the central nervous system and kidneys. This same metal mix, in similar proportions, resulted in hazard values of 0.96 and 1.53 for coarse and fine sediments, respectively. Metals in background North Island soils resulted in a hazard value of 4.33, with lead (3.09) accounting for the majority of hazard. The high background lead concentration is probably due to the heavy use of aviation fuels at the air station. We conclude that the non-cancer hazard posed by sediments at station 16-2 warrants evaluation of remedial altenatives.

Subtidal sediments generally ranged between hazard values of 1 and 2, due again to the same metal mix. The exception was station 8-1 with a hazard value of 3.88, and relatively higher levels of antimony and molybdenum. The subtidal sediments were not considered to pose a health hazard for lack of a ready exposure pathway and were evaluated only for perspective. We conclude that subtidal sediments pose little hazard to residential humans and that no further investigations or action are warranted for these sediments.

ASLA & Associates, Inc. 1991. Master Plan for Naval Air Station North Island, San Diego, California. N68711-C-0038. U.S. Navy Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Adams, D.D. 1991. Sampling sediment pore water. In A. Mudroch and S.D. MacKnight, eds., *Handbook for Techniques for Aquatic Sediments Sampling.* CRC Press, Boca Raton, FL, pp. 171-201.

ASTM/American Society for Testing and Materials. 1993. Standard guide for conducting 10-day static sediment toxicity tests with marine and estuarine amphipods. In ASTM, eds., *Annual Book of ASTM Standards*, Vol. 11.04 - Pesticides; Resource Recovery; Hazardous Substances and Oil Spil Responses; Waste Management; Biological Effects. ASTM, Philadelphia, PA, pp. 1138-1163.

ASTM/American Society for Testing and Materials. 1994. Standard guide for conducting static acute toxicity tests starting with embryos of four species of bivalve molluscs. In ASTM, eds., *Annual Book of ASTM Standards*, Vol. 11.04 - Pesticides; Resource Recovery; Hazardous Substances and Oil Spill Responses; Waste Management; Biological Effects. ASTM, Philadelphia, PA, pp. 454-471.

ASTM/American Society for Testing and Materials. 1995. *Annual Book of ASTM Standards*. ASTM, Philadelphia, PA.

ATG/Allied Technology Group. 1996. Removal action closeout report: Emergency removal action shoreline slag, Investigation Restoration Site 10. Preliminary Final Report. Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Barber, T.R., P.C. Fuchsman, D.J. Chappie, J.C. Sferra, F.C. Newton and P.J. Sheehan. 1997. Toxicity of Hexachlorobenzene to *Hyalella azteca* and *Chironomus tentans* in Spiked Sediment Bioassays. *Env. Tox. Chem*. **9:**221-231.

Batley, G.E. and **M.S. Giles.** 1980. A solvent displacement technique for the preparation of sediment interstitial waters. In R.A. Baker, ed., *Contaminants in Sediments*, Vol. 2 - Toxicology. Ann Arbor Science, Ann Arbor, MI, pp. 101-117.

Battelle Marine Sciences Laboratory. 1992. Standard methods manual for environmental sampling and analysis in San Francisco Bay. Draft. United States Army Corps of Engineers, San Francisco, CA.

Bechtel National, Inc. 1994. Comprehensive Long-Term Environmental Action Navy: CLEAN II. CTO-0013. Draft Site-Specific Quality Assurance Project Plan Shoreline Sediments. Naval Air Station North Island, San Diego, CA.

Bechtel National, Inc. 1995. Comprehensive Long-Term Environmental Action Navy: CLEAN II. CTO-0013. Final Bioassay and Environmental Sampling and Analysis Work Plan Shoreline Sediments. Naval Air Station North Island, San Diego, CA.

BPTCP/Bay Protection and Toxic Cleanup Program. 1996. 1993-1994 Database. State Water Resources Control Board, National Oceanic and Atmospheric Administration, California Department of Fish and Game, Marine Pollution Studies Laboratory and Moss Landing Marine Laboratories, Sacramento, CA.

Bray, J.R. and **J.T. Curtis.** 1957. An ordination of the upland forest communities of southern Wisconsin. *Ecol. Monogr.* **27**:325-349.

Brooks, J.M., M.C. Kennicutt, T.L. Wade, A.D. Hart, G.J. Denoux and **T.J. McDonald.** 1990. Hydrocarbon distributions around a shallow water multiwell platform. *Environ. Sci. Technol.* **24**:1079-1085.

Brown and Caldwell. 1983. Initial assessment study of Naval Air Station, North Island, California. Navy Assessment and Control of Installation Pollutants Department, Port Hueneme, CA.

Bruland, K., K. Bertine, M. Koide and **E. Goldberg.** 1974. History of metal pollution in Southern California coastal zone. *Environ. Sci. Technol.* **8**:425-432.

Burton, G.A. 1992. *Sediment Toxicity Assessment*. Lewis Publishers, Inc., Boca Raton, FL.

Butler, E.L., G.S. Douglas, W.G. Steinhauer, R.C. Prince, T. Aczel, C.S. Hsu, M.T. Bronson, J.R. Clark and **J.E. Lindstrom**. 1991. Hopane, A New Chemical Tool for Measuring Oil Biodegradation. In Robert E. Hinchee and Robert F. Olfenbuttel (Eds.), On-Site Bioreclamation. pp. 515-521. Butterworth-Heinemann, Stoneham, MA.

CAEPA/California Environmental Protection Agency. 1994. *Preliminary Endangerment Assessment Guidance Manual*. Department of Toxic Substances Control, Sacramento, California.

CAS/Columbia Analytical Systems. 1994. Dredged material sediment testing results for project M1-90 - Maintenance dredging at Pier Bravo, Naval Air Station, North Island, San Diego, CA. Final Report. Naval Air Station, North Island, San Diego, CA.

Chapman, P.M., R.N. Dexter, R.M. Kocan and **E.R. Long**. 1985. An overview of biological effects testing in Puget Sound, Washington; methods, results and implications. In R.D. Cardwell, R. Purdy and R.C. Bahner, eds., *Aquatic Toxicology and Hazard Assessment: Proceedings of the Seventh Annual Symposium*, ASTM STP 854. Amercian Society of Testing Materials, Philadelphia, PA, pp. 344-363.

Chow, T.J. and **J. Earl.** 1979. Trace metal distribution in Southern California OCS benthic sediments. Final Report, Southern California Baseline Study, Benthic Year 2. Bureau of Land Management, Los Angeles, CA.

Daskalakis, K.D. and **T.P. O'Connor.** 1995. Normalization and elemental sediment contamination in the coastal United States. *Environ. Sci. Technol.* **29**:470-477.

DeWitt, T.H., G.R. Ditsworth and **R.C. Swartz.** 1988. Effects of natural sediment features on survival of the phoxocephalid amphipod, *Rhepoxynius abronius*. *Mar. Environ. Res.* **25**:99-124.

DeWitt, T.H., R.C. Swartz and **J.O. Lamberson**. 1989. Measuring the toxicity of estuarine sediments. *Environ. Toxicol. Chem.* **8**:1035-1048.

DiToro, D.M., J.D. Mahmy, D.J. Hansen, K.J. Scott, A.R. Carlson and **G.T. Ankley**. 1992. Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. *Environ. Sci. Technol*. **26**:96-101.

Douglas, G.S. and **A.D. Uhler**. 1993. Optimizing EPA methods for petroleum contaminated site assessments. *Env. Test. Anal.* **2**:46-53.

Douglas, G.S., R.C. Prince, E.L. Butler and **W.G. Steinhauer**. 1993. The Use of Internal Chemical Indicators in Petroleum and Refined Products to Evaluate the Extent of Biodegradation. In Situ and On-Site Bioreclamation Conference. March 29-April 1, 1993, Tampa, FL.

Douglas, G.S., K.J McCarthy, D.T. Dahlen, J.A. Seavey, W.G. Steinhauer, R.C. Prince and **D.L. Elmendorf**. 1992. The use of hydrocarbon analyses for environmental assessment and remediation. *J. Soil Contam.* **1**:197-216.

Douglas, G.S., K.J. McCarthy and **J.A. Seavey**. 1991. "Fingerprinting of petroleum hydrocarbons in water." Invited paper presented at the 14th Annual EPA Conference on Analysis of Pollutants in the Environment. May 8-9, 1991. Norfolk, VA.

Edmonds, W.M. and **A.H. Bath.** 1976. Centrifuge extraction and chemical analysis of interstitial waters. *Environ. Sci. Technol*. **10**:467-472.

Elmendorf, D.L., C.E. Haith, G.S. Douglas and **R.C. Prince**. 1993. "Relative Rates of Biodegradation of Substituted Polyaromatic Hydrocarbons." Presented at the In Situ and On-site Bioreclamation Symposium, Tampa, FL, March 29-April 1, 1993 [paper A14].

Evans, L.T. 1958. A soil exploration. District Public Works Office, Eleventh Naval District, San Diego, CA.

Evans, L.T. 1960. A soil exploration. District Public Works Office, Eleventh Naval District, San Diego, CA.

Fauchald, K. and **P.A. Jumars**. 1979. The diet of worms. *Oceanogr. Mar. Biol. Ann. Rev.* **17**:193-284.

Finney, B. and **C. Huh.** 1989a. History of metal pollution in the Southern California Bight: An update. *Environ. Sci. Technol*. **23**:294-303.

Finney, B. and **C. Huh**. 1989b. High resolution sedimentary records of heavy metals from the Santa Monica and San Pedro Basins, California. *Mar. Poll. Bull.* **20**:181-187.

Folk, R.L. 1968. *Petrology of Sedimentary Rocks.* Hemphill's, Austin, TX.

Förstner, U. 1990. Chap 3: Inorganic sediment chemistry and elemental speciation. In R. Bando, J. Giesy, H. Montain, eds., *Sediments: Chemistry and Toxicity of In-Place Pollutants.* Lewis Publications, Ann Arbor, MI, pp. 67-70.

Hanson, P.J., D.W. Evans and **D.R. Colby.** 1993. Assessment of elemental contamination of estuarine and coastal environments based on geochemical and statistical modeling of sediments. *Mar. Environ. Res.* **36**:237-266.

Hassett, J.J., J.C. Means, W.L. Barnwart and **S.G. Wood**. 1980. Sorption properties of sediments and energy-related pollutants. 600/3-80-041. U.S. Environmental Protection Agency, Washington, D.C.

HLA/Harding Lawson Associates. 1989. Remedial Investigation, shoreside sediments - Site 1 - Naval Air Station, North Island. Vol 1. Draft Report. Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Hostettler, F.D. and **K.A. Kvenvolden**. 1994. "Geochemical changes in crude oil spilled from the Exxon Valdez supertanker into Prince William Sound Alaska." *Org. Geochem.* **21:**(8/9) 927-936.

Hunt, J.W., B.S. Anderson, B.M. Phillips, J. Newman, R.S. Tjerrdma, K. Taberski, C.J. Wilson, M. Stephenson, H.M. Puckett, R. Fairey and J. Oakden. 1998. Sediment Quality and Biological Effects in San Francisco Bay. Bay Protection and Toxic Cleanup Program. Final Technical Report. August 1998.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecol. Monogr*. **54**:187-211.

Jacobs Engineering. 1991. Naval Air Station, North Island, San Diego, California sites 1, 5, 6, 9, and 10 Remedial Investigation/feasibility Study work plan. CLE-J02-01F043-01-B7-002. Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Jacobs Engineering. 1993. Site investigation/RCRA Facility Investigation report sites 3, 4, 7, and 12. Draft Final Report. Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Jacobs Engineering. 1996. Naval Air Station, North Island, San Diego California - CTO 0216. Addendum to RI/FS Report Site 9, Chemical waste disposal area. Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Jenkins, K.D., S. Howe, B.M. Sanders and **C. Norwood**. 1989. Sediment deposition, biological accumulation and subcellular distribution of barium following the drilling of an exploratory well. In F.R. Engelhardt, J.P. Ray and A.H. Gillam, eds., *Drilling Wastes*. Elsevier Applied Sciences Publisher, London, pp. 589-608.

Johns, D.M. and **T.C. Ginn**. 1990. Development of a *Neanthes* sediment bioassay for use in Puget Sound. EPA 910/9-90-005. U.S. Environmental Protection Agency Region 10, Office of Puget Sound, Seattle, WA.

Johns, D.M. and **T.C. Ginn**. 1990. *Neanthes* long-term exposure experiment: the relationship between juvenile growth and reproductive success. EPA 910/9-90-010. U.S. Environmental Protection Agency Region 10, Office of Puget Sound, Seattle, WA.

Jones, I., D. Rice, Jr. and **F.L. Harrison**. 1983. Sister chromatid exchange: Two applications to monitoring genotoxic effects in the marine environment. In M. Marin and F. Harrison, eds., *Workshop on Sublethal Effects of Stress on Marine Organisms, Asilomar, Pacific Grove, California.* U.S. Department of Energy and California State Water Resources Control Board, Sacramento, CA, p. 74.

Karickhoff, S.W. 1984. Organic pollutant sorption in aquatic systems. *J. Hydraulic Engineering* **110**: 707-735.

Katz, A. and **I.R. Kaplan**. 1981. Heavy metal behaviour in coastal sediments of Southern California: A critical review and synthesis. *Mar. Chem.* **10**:261-299.

Kennicutt, M.C. 1988. The Effect of Biodegradation on Crude Oil Bulk and Molecular Composition. *Oil Chem. Pollut.*, pp. 541-546.

Kennicutt, M.C., II, T.L. Wade, B.J. Presley, A.G. Requejo, J.M. Brooks and **G.J. Denoux**. 1994. Sediment contaminants in Casco Bay, Maine: Inventories, sources, and potential for biological impact. *Environ. Sci. Technol.* **28**:261-299.

Kennish, M.J. 1992. Ecology of Estuaries: Anthropogenic Effects. CRC Press, Boca Raton, FL, USA, 494 pp.

Kram, M.L., P.M. Stang and **P.F. Seligman.** 1989. Fate and distribution of organotin in sediments of four U.S. Harbors. Technical Report 1280. Naval Ocean Systems Center, San Diego, CA.

Long, E.R. and **M.F. Buchman**. 1989. An evaluation of candidate measures of biological effects for the national status and trends program. Technical Memorandum NOS OMA 45. National Ocean Services, National Oceanic and Atmospheric Administration, Seattle, WA.

Long, E.R., D.D. MacDonald, S.L. Smith and **F.D. Calder**. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environ. Management* **19**:81-97.

Manheim, F.T. 1976. Interstitial Waters of Marine Sediment. Vol. 6 - Chemical Oceanography.

Maxon, C.L., A.B. Barnett and **D. Diener.** 1997. Use of a Multivariate Statistical Technique to Predict Biological Impact in Southern California. *Environ. Toxicol. and Chem.* **3:** in press.

McCain, B.B., S.L. Chan, M.M. Krahn, D.W. Brown, M.S. Myers, J.T. Landahl, S. Pierce, R.C. Clark, Jr. and **U. Varanasi.** 1992. Chemical contamination and associated fish diseases in San Diego Bay. *Environ. Sci. Technol*. **26**:725-733.

Mearns, A.J., M. Matta, G. Shigenaka, D. MacDonald, M. Buchman, H. Harris, J. Golas and **G. Lauenstein.** 1991. Contaminant trends in the Southern California Bight: inventory and assessment. Technical Memorandum NOS ORCA 62. National Oceanic and Atmospheric Administration, Seattle, WA.

MEC/MEC Analytical Systems, Inc. 1996. Remedial Investigation/RCRA Facility Investigation, Site 1 - shoreline sediments, naval Air Station North Island. CTO-0013. Work Plan. U.S. Navy Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Milliken, G.A. and **D.E. Johnson.** 1984. *Analysis of Messy Data.* Wadsworth, Inc., Belmont, CA.

Moore, W.D., T. Dillon, J.Q. Word and J.A. Ward. 1993. QA/QC Guidance for Laboratory Dredged Material Bioassays. Results of QA/QC workshop held May 26-27, 1993 in Seattle, WA. Prepared for: USACOE, Waterways Experiment Station by Battelle Marine Sciences Laboratory, Sequim, WA. Published by USACOE/WES, Vicksburg, MS.

NASNI 1995: Naval Air Station North Island Final Background Soil Sampling Report, 15 May 1995. Jacobs Engineering Group, Inc. Pasadena CA.

Natrella, M.G. 1966. *Experimental Statistics: National Bureau of Standards Handbook 91*. United States Department of Commerce, U.S. Government Printing Office, Washington, D.C.

Ng, A. and **C. Patterson.** Changes in lead and barium with time in California off-shore basin sediments. *Geochim. Cosmochim. Acta* **46**:2307-2321.

O'Connor, T.P. and **B. Beliaeff.** 1995. Recent trends in coastal environmental quality: Results from the Mussel Watch Project. National Oceanic and Atmospheric Administration/NOS Coastal Monitoring and Bioeffects Assessment Division, Silver Spring, MD.

Page, D.S., P.D. Boehm, G.S. Douglas and **A.E. Bence**. 1994. "Identification of Hydrocarbon Sources in the Benthic Sediments of Prince William Sound and the Gulf of Alaska Following the Exxon Valdez Oil Spill." The Third Symposium on Environmental Toxicology and Risk Assessment, Atlanta, GA, April 26-29, 1993, In Press.

Pastorok, R.A. and **D.S. Becker**. 1990. Comparative sensitivity of sediment toxicity bioassays at three superfund sites in Puget Sound. In R.D. Cardwell, R. Purdy and R.C. Bahner, eds., *Aquatic Toxicology and Hazard Assessment: Proceedings of the Seventh Annual Symposium*, ASTM STP 854. Amercian Society of Testing Materials, Philadelphia, PA, pp. 123-139.

Pesch, G.G. and **C.E. Pesch**. 1980. *Neanthes arenaceodentata* (Polychaeta: Annelida) a proposed cytogenetic model for marine genetic toxicology. *Can. J. Fish. Aquat. Sci*. **37**:1125-1128.

Pesch, G.G., C.E. Pesch and **A.R. Malcom**. 1981. *Neanthes arenaceodentata*: A cytogenetic model for marine genetic toxicology. *Aquatic Toxicology* **1**:301-311.

Plumb, R.H., Jr. 1981. Procedure for handling and chemical analysis of sediment and water samples. EPA/CE-81-1. Technical Report. United States Army Engineer Waterways Experiment Station, Vicksburg, MS.

Prince, R.C., E.L. Butler, G.S. Douglas, W.G. Steinhauer, C.S. Hsu, T. Aczel, J.R. Clark and **J.E. Lindstrom**. 1991. Assessing rates of biodegradation in Prince William Sound. American Society of Microbiology, 1991 Conference on Biotechnology. June 27-30, 1991. New York, NY.

Prince, R.C., S.M. Hinton, J.R. Bragg, D.E. Elmendorf, J.R. Lute, J.G. Grossman, W.K. Robbins, C.S. Hsu, G.S. Douglas, R.E. Bare, C.E. Haith, J.D. Senius, V. Minak-Bernero, S.J. McMillen, J.C. Roffal and **R.R. Chianelli**. 1993. Laboratory studies of oil spill remediation; toward understanding field behavior. Preprints of the Division of Petroleum Chemistry. American Chemical Society 38:240-244.

Prince, R.C., J.R. Clark, J.E. Lindstrom, E.L. Butler, E.J. Brown, G. Winter, M.J. Grossman, P.R. Parrish, R.E. Bare, J.F. Braddock, W.G. Steinhauer, G.S. Douglas, J.M. Kennedy, P.J. Barter, J.R. Bragg, E.J. Harner and **R.M. Atlas**. 1993. "Bioremediation of the Exxon Valdez Oil Spill; Monitoring Safety and Efficacy." Presented at the In Situ and On-Site Bioreclamation Symposium, San Diego, California, April 5-8, 1993 (this volume) [paper S-9].

Prince, R.C., S.M. Hinton, J.R. Bragg, D. L. Elmendorf, J.R. Lute, M.J. Grossman, W.K. Robbins, C.S. Hsu, G.S. Douglas, R.E. Bare, C.E. Haith, J.D. Senius, V. Minak-Bernero, S.J. McMillen, J.C. Roffall and **R.R. Chianelli**. 1993. "Laboratory Studies of Oil Spill Bioremediation; Toward Understanding Field Behavior." Paper presented at Proceedings of the ACS Meeting, Denver, CO, April 1993.

Pruell, R.J. and **J.G. Quinn.** 1988. Accumulation of polycyclic aromatic hydrocarbons in crankcase oil. *Environ. Poll.* **49**:89-97.

Reish, D.L. 1984. Marine ecotoxicological tests with polychaetous annelids. In Editors, *Ecotoxicological Testing for the Marine Environment*, Vol. 1. Publisher, City, ST, pp. 426-454.

Rohlf, F.J. and **R.R. Sokal**. 1969. *Biometry: The Principles and Practice of Statistics in Biological Research.* W.H. Freeman and Company, San Francisco, CA.

SANDAG/San Diego Association of Governments. 1991. San Diego Bay cleanup under Section 205(j) of the Clean Water Act. SANDAG, San Diego, CA.

San Diego Bay Protection and Toxic Cleanup Program for 1993-1994 Database. 1996. State Water Resources Control Board, National Oceanic and Atmospheric Administration, California Department of Fish and Game, Marine Pollution Studies Laboratory and Moss Landing Marine Laboratories, Sacramento, CA.

SAS Institute, Inc. 1989. SAS/STAT User's Guide, Version 6, Fourth Edition, Volume I. SAS Institute, Inc. Cary, NC pp. 773-822.

Sauer, T. and P. Boehm. 1991. The use of defensible analytical chemical measurements for oil spill natural resource damage assessment. In: Proceedings, Oil Spill Conference, American Petroleum Institute.

Seligman, P.F., A. O. Valkirs and **R.F. Lee.** 1986. Degradation of tributyltin in San Diego Bay, California, waters. *Environ. Sci. Technol.* **20**:1229-1235.

Shapiro, S.S. and **M.B. Wilk.** 1965. An analysis of variance test for normality (complete samples). *Biometrika* **52**:591-611.

SCCWRP/Southern California Coastal Water Research Project. 1992. SCCWRP Biennial Report 1990-91 and 1991-92, J.N. Cross and C. Francisco eds., SCCWRP, Long Beach, CA.

SCCWRP/Southern California Coastal Water Research Project. 1996. SCCWRP Annual Report, 1994-95. M.J. Allen, Ed., Westminster, CA pp. 57-68.

Steel, R.G.D. and **J.H. Torrie.** 1960. *Principles and Procedures of Statistics.* McGraw-Hill Book Company, Inc., New York, NY.

Steinhauer, M., E. Crecelius and **W. Steinhauer**. 1994. Temporal and spatial changes in the concentrations of hydrocarbons and trace metals in the vicinity of an offshore oil-production platform. *Mar. Environ. Res.* **37**:129-163.

Swartz, R.C., D.W. Schults, G.R. Ditsworth, W.A. DeBen and **F.A. Cole**. 1985a. Sediment toxicity, contamination and macrobenthic communities near a large sewage outfall. In T.P. Boyle, ed., *Validation and Predictability of Laboratory Methods for Assessing the Fate and Effects of Contaminants in Aquatic Ecosystems*, STP 856, American Society for Testing Materials, Philadelphia, PA, pp. 152-175.

Swartz, R.C., W.A. DeBen, J.K. Phillips, J.O. Lamberson and **F.A. Cole**. 1985b. Phoxocephalid amphipod bioassay for marine sediment toxicity. In R.D. Cardwell, R. Purdy, and R.C. Bahner, eds., *Aquatic Toxicology and Hazard Assessment: Proceedings of the Seventh Annual Symposium*, ASTM STP 854. Amercian Society of Testing Materials, Philadelphia, PA, pp. 284-307.

Swartz, R.C., W.A. DeBen and **F.A. Cole**. 1979. A bioassay for the toxicity of sediment to marine macrobenthos. *J. Water Pollut. Control Fed.* **51**:944-950.

SWDIV/Southwest Division Naval Facilities Engineering Command. 1992. Programmatic Environmental Impact Statement for dredged material disposal. Draft report. U.S. Navy Southwest Division Naval Facilities Engineering Command, San Diego, CA.

SWDIV/Southwest Division Naval Facilities Engineering Command. 1993. Programmatic Environmental Impact Statement for dredged material disposal. Final report. U.S. Navy Southwest Division Naval Facilities Engineering Command, San Diego, CA.

SWDIV/Southwest Division Naval Facilities Engineering Command. 1995. Naval Air Station, North Island, San Diego, California. Draft Remedial Investigation/RCRA Facility Investigation Report: Site 9: Chemical waste disposal area. Contract No. N68711-89-D-9296. U.S. Navy Southwest Division Naval Facilities Engineering Command, San Diego, CA.

Tattersall, W.M. and **O.S. Tattersall**. 1951. The British Mysidacea. The Ray Society. Bernard Quaritch, Ltd., London.

Tidelines. 1996. San Diego 1996 Tide Calendar. Encinitas, CA.

Trefry, J.H. and B.J. Persley. 1976. Heavy Metals in Sediments from San Antonio Bay and the Northern Gulf of Mexico. *Env. Geol.* **1:**283-294.

Uhler, A.D., G.S. Durell and **A.M. Spellacy**. 1991. Extraction procedure for the measurement of butyltin compounds in biological tissues using toulene, HBr, and tropolone. *Bull. Environ. Contam. Toxicol.* **47**:217-221.

USACOE/United States Army Corps of Engineers. 1985. Hydrologic data inventory, Southern Coastal Zone, Ragged Point (San Luis Obispo County) to the Mexican Border. Coast of California Wave and Tidal Waves Study (CCSTWS) Report No. 85-8. Los Angeles, CA.

USACOE/United States Army Corps of Engineers. 1987. Draft environmental assessment for maintenance dredging, main entrance channel, San Diego Bay, California. Los Angeles, CA.

USEPA/United States Environmental Protection Agency. 1983. "Methods for Chemical Analysis of Water and Wastes." U.S. Environmental Protection Agency, Washington, DC.

USEPA/United States Environmental Protection Agency. 1986. "Organic Analytes." Chapter 4 in Test Methods for Evaluating Solid Waste. Document SW-846. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, 3rd Ed.

USEPA/United States Environmental Protection Agency. 1988. Guidance for conducting remedial investigations and feasilibity studies under CERCLA. Interim Final, October 1988, NTIS PB89-184626, EPA 9355.3-01.

USEPA/United States Environmental Protection Agency. 1989. Risk assessment guidance for superfund. EPA/540/1-89/002. Volume 1: Human health evaluation manual (Part A). Office of Emergency and Remedial Response, Washington, D.C.

USEPA/United States Environmental Protection Agency. 1992. Dermal exposure assessment. EPA/600/8-91/001B. Principals and applications, interim report. Office of Research and Development, Washington, D.C.

USEPA/United States Environmental Protection Agency. 1994a. USEPA contract laboratory program national functional guidelines for organic data review. Environmental Protection Agency, Washington, D.C.

USEPA/United States Environmental Protection Agency. 1994b. USEPA contract laboratory program national functional guidelines for inorganic data review. Environmental Protection Agency, Washington, D.C.

USEPA/United States Environmental Protection Agency. 1995. Recommended guidelines for conducting laboratory bioassays on Puget Sound sediments. U. S. Environmental Protection Agency Region 10, Seattle, WA, and Puget Sound Water Quality Authority, Olympia, WA.

USEPA/United States Environmental Protection Agency. 1996. Region IX Preliminary Remediation Goals (PRGs). Washington, D.C.

USEPA/United States Environmental Protection Agency. 1997. Health Effects Assessment Summary Tables (HEAST): Annual Update, FY 1997. National Center For Environmental Assessment (NCEA), Office of Research and Development and Office of Emergency and Remedial Response, Washington, DC.

USEPA/USACOE/United States Environmental Protection Agency/United States Army Corps of Engineers. 1991. Evaluation of dredged material proposed for ocean disposal testing manual. EPA-503/8-91/001. Office of Water, Washington, D.C.

U.S. Navy. 1991. Master Plan - Naval Air Station, North Island, San Diego, CA.

Wade, T.L., B. Garcia-Romero and **J.M. Brooks.** 1990. Butyltins in sediments and bivalves from U.S. coastal areas. *Chemosphere* **20**:647-662.

Wang, P.F., R.T. Cheng, K. Richter, E. Gross, D. Sutton and **J. Gartner.** 1998. Modeling tidal hydrodynamics of San Diego Bay, California. *J. Estuarine and Coastal Processes.* No. 97124.

Ward, J.A., J.Q. Word, A.D. Uhler, E.S. Barrows, V.I. Cullinan, R. Cuello and **D.K. Niyogi**. 1995. Suggested methods for environmental sampling and analysis in San Francisco Bay. Volumes 1-3. PNL-10565. U.S. Army Corps of Engineers, San Francisco District, CA.

Wedepohl, K.H. 1995. The composition of the continental crust. *Geochim. et. Cosmochim. Act.* 59(7): 1217-1232.

Word, J.Q., H.L. Mayhew, M.R. Pinza and J.A. Ward. 1993. Metals Bioaccumulation by Macoma nasuta and Nephtys caecoides after a 28-day Exposure to Dredged Sediments. Presented at the 14th annual SETAC meetig in Houston, TX, November 15-18, 1993. Parpared by: Battelle Marine Science Laboratory, Sequim, WA.

Zeng, E.Y. and **C.L.Vista.** 1997. Organic pollutants in the coastal environment off San Diego, California. I. Source identification and assessment by compositional indices of polycyclic aromatic hydrocarbons. *Environ. Toxicol. Chem.* In press.