NAVAL FACILITIES ENGINEERING COMMAND
Washington, DC 20374-5065

NFESC
User's Guide
UG-2053-ENV

IMPLEMENTATION GUIDE FOR ASSESSING AND MANAGING CONTAMINATED SEDIMENT AT NAVY FACILITIES

Prepared by:

SPAWAR Systems Center (SSC) San Diego

Contributing Author:

Battelle

January 2005 (Rev 2)

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REVISION PAGE

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Rev 1, June 2003: Updated Section 2.3 and corrected the Table of Contents
Rev 2, January 2005: Corrected web addresses, updated Tables 3-4 and 3-5 and added reference to the NAVFAC Background Analysis Tech Transfer Tool

EXECUTIVE SUMMARY

This document presents guidelines for conducting sediment site assessments and remedial alternative evaluations within the Navy's Environmental Restoration program. It is intended for use by Remedial Project Managers (RPMs) and their technical support staff as stepwise guidance that will apply to most Navy sediment investigations. Sediment investigations often are more complex than terrestrial investigations for a variety of reasons, including a lack of promulgated sediment quality criteria, incomplete knowledge and understanding of aquatic food webs, and lack of published risk-based threshold data (e.g., toxicity reference values) for many chemicals of potential concern (COPCs). Additionally, sediments commonly require specialized methods for sampling, analysis, and remediation. This guide identifies and discusses sediment-specific issues related to site characterization, risk assessment, and remedial alternative evaluation, and then directs the reader to related Web sites and resources for more detailed technical information. It is intended to help the RPM avoid unfocused or unnecessary studies and to coordinate and integrate data collection activities across all aspects of the sediment investigation. This guide complements Chief of Naval Operations (CNO) *Policy on Sediment Site Investigation and Response Action* (February 2002) as well as other applicable policies and guidance on risk assessment and the use of background chemical levels. Critical sediment issues discussed in this guide include the following:

- Addressing multiple contaminant sources (Navy and non-Navy);
- Development of a detailed and accurate Conceptual Site Model (CSM);
- Collection of important geochemical and physical information for characterizing the source, fate, and transport of chemicals in sediment and supporting the evaluation of remedial alternatives;
- Selection and use of appropriate tests for ecological risk assessments (ERAs) (e.g., bioavailability evaluations, aquatic toxicity tests);
- Use of background and reference site data in risk assessments;
- Use of a weight-of-evidence (WOE) approach and other decision-making tools;
- Development of site-specific risk-based cleanup goals; and
- Evaluating remedial options for sediment and the risk and liabilities associated with each.

This guide is organized into four sections along with a glossary and references. Hyperlinks that connect the reader to related Web sites and documents are found throughout the document.

Section 1 – Introduction presents the purpose and organization of the document, discusses some of the primary differences in conducting aquatic versus terrestrial studies, and provides overviews of applicable Navy policy and guidance as well as pertinent laws and regulations.

Section 2 – Sediment Site Characterization presents an overview of the site characterization process relative to sediment investigations, including planning considerations, developing a CSM, source identification, defining the nature and extent of contamination, and characterizing contaminant fate and transport. This section also identifies important physical and chemical data that should be collected as part of a sediment investigation, with an emphasis on coordinating data collection for all aspects of the investigation (site characterization, risk assessment, and evaluation of remedial alternatives). This section also provides an overview of sample design and sample collection methods and equipment.

Section 3 – Ecological and Human Health Risk Assessment for Sediment Studies follows the stepwise guidance for conducting ecological and human health risk assessments at sediment sites within the Navy's tiered framework. Issues specific to sediment sites are identified and discussed for each tier.

Section 4 – Sediment Remedial Alternative Evaluations addresses Feasibility Study (FS) planning considerations and determination of site-specific risk-based cleanup levels. Remedial options, including monitored natural recovery, in situ capping, and removal, are described along with monitoring considerations and sediment management issues.

Section 5 – Glossary provides a description of common terminology used in this guide and in sediment investigations in general.

Section 6 – References, Resources, and Applicable Web Sites provides references by section along with Web site addresses for information discussed in the guide.

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ABBREVIATIONS AND ACRONYMS

AEM	Applied Environmental Management, Inc.
AET	apparent effects threshold
ARAR	applicable or relevant and appropriate requirement
ARCS	Assessment and Remediation of Contaminated Sediment
ASTM	American Society for Testing and Materials
AVS	acid volatile sulfide
BAF	bioaccumulation factor
BERA	Baseline Ecological Risk Assessment
BHHRA	Baseline Human Health Risk Assessment
BRAC	Base Realignment and Closure (Act)
BSAF	biota sediment accumulation factors
CAA	Clean Air Act
CAD	contained aquatic disposal
CDF	confined disposal facilities
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CERCLIS	Comprehensive Environmental Response, Compensation, and Liability Information System
CFR	Code of Federal Regulations
CNO	Chief of Naval Operations
COPC	chemical of potential concern
CSF	cancer slope factor
CSM	conceptual site model
CWA	Clean Water Act
DGPS	differential global positioning system
DOT	United States Department of Transportation
DQO	data quality objective
EC50	concentration causing an effect on 50% of test organisms
EFA	Engineering Field Activity
EFD	Engineering Field Division
Eh	redox potential
ERA	ecological risk assessment
ERDC	(USACE) Engineer Research and Development Center
ER-L	effects range-low
ER-M	effects range-median
ESG	equilibrium partitioning sediment guideline
FR	Federal Register
FS	Feasibility Study
FSP	Field Sampling Plan
FW	fresh water
GSI	groundwater-surface water interaction
GW	groundwater

HPS	Hunters Point Shipyard
HHRA	human health risk assessment
HMTA	Hazardous Materials Transportation Act
HQ	hazard quotient
HWIR	Hazardous Waste Identification Rule
IR	Installation Restoration
LBC	level-bottom capping
LC50	concentration lethal to 50% of test organisms
LDR	land disposal restriction
MCSS	Major Contaminated Sediment Sites
MPRSA	Marine Protection, Research, and Sanctuaries Act
NAS	Naval Air Station
NAVFAC	Naval Facilities Engineering Command
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NFESC	Naval Facilities Engineering Service Center
NHPA	National Historical Preservation Act (of 1966)
NOAA	National Oceanographic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
NRC	National Research Council
NTR	National Toxic Rule
OU	Operable Unit
PA	Preliminary Assessment
PA/SI	Preliminary Assessment/Site Inspection
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCS	Permit Compliance System
PEC	probable effect concentration
PEL	probable effect level
PRG	preliminary remediation goal
QA/QC	quality assurance/quality control
RAGS	Risk Assessment Guidance for Superfund
RAO	remedial action objective
RAW	Risk Assessment Workgroup
RBC	risk-based concentration
RBS	risk-based screening
RCRA	Resource Conservation and Recovery Act
RfC	reference concentration
RfD	reference dose
RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
RPM	Remedial Project Manager
RTDF	Remediation Technologies Development Forum

SAP	Sampling and Analysis Plan
SEM	simultaneously extracted metals
SFBRWQCB	San Francisco Bay Regional Water Quality Control Board
SFEI	San Francisco Estuary Institute
SI	Site Inspection
SMDP	scientific management decision point
SQG	sediment quality guideline
SQG-Q	sediment quality guideline quotient
SRA	Screening Risk Assessment
SSRBS	site-specific risk-based screening
STMS	sediment transport management system(s)
SW	surface water
SWRCB	State Water Resources Control Board
TBF	theoretical bioaccumulation factor
TCLP	Toxicity Characteristic Leaching Procedure
TEC	threshold effect concentration
TEL	threshold effect level
TIE	toxicity identification evaluation
TMDL	total maximum daily load
TOC	total organic carbon
TRV	toxicity reference value
UCL	upper confidence limit
USACE	United States Army Corps of Engineers
USC	United States Code
U.S. EPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WCSD	Watershed Contaminated Source Document
WOE	weight of evidence
WP	Work Plan
WQC	water quality criteria
WSDE	Washington State Department of Ecology
XRF	x-ray fluorescence

1.0 INTRODUCTION

This document presents guidelines for conducting sediment site assessments and remedial evaluations within the Navy's Environmental Restoration program, including Installation Restoration (IR) and Base Realignment and Closure (BRAC). The document focuses on sediment-specific issues associated with the Remedial Investigation and Feasibility Study (RI/FS) process under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980. It is intended for use by Remedial Project Managers (RPMs) and their technical support staff as stepwise guidance that will apply to most Navy sediment investigations.

This document is not intended to be a comprehensive method manual. Instead, it identifies and discusses sediment-specific issues related to site characterization, risk assessment, and remedial alternatives evaluation, and then directs the reader to related Web sites and resources for more detailed technical information. This guidance is intended to help the RPM avoid unfocused or unnecessary studies, and to coordinate and integrate data collection activities across all aspects of the sediment investigation. Critical sediment issues discussed in this guide include the following:

- Addressing multiple contaminant sources (Navy and non-Navy);
- Development of a detailed and accurate conceptual site model (CSM);
- Collection of important geochemical and physical information for characterizing the source, fate, and transport of chemicals in sediment;
- Selection and use of appropriate tests for ecological risk assessments (ERAs) (e.g., bioavailability evaluations, aquatic toxicity tests);
- Use of background and reference site data in risk assessments;
- Use of a weight-of-evidence (WOE) approach and other decision-making tools;
- Developing site-specific risk-based cleanup goals; and
- Evaluating remedial options for sediment, and the risk and liabilities associated with each option.

The primary differences between aquatic (i.e., sediment) and terrestrial RI/FS studies are discussed in Section 1.1. The organization of this document is described in Section 1.2, and overviews of Navy policy and framework for sediment investigations and of applicable regulations, laws, and guidelines are provided in Sections 1.3 and 1.4, respectively.

1.1 AQUATIC VERSUS TERRESTRIAL STUDIES

The fundamental elements of conducting aquatic and terrestrial RI/FS investigations are the same. However, sediment investigations are often more complex for a variety of reasons, such as the fact that sediment quality criteria are not fully promulgated, aquatic food webs often are complex or poorly understood, and risk-based threshold data (e.g., toxicity reference values) are not available for many chemicals of potential concern (COPCs). Additionally, sediments may require specialized methods for sampling, analysis, and remediation. Some of the technical, regulatory, and management challenges associated with contaminated sediments are discussed in detail in a 1997 National Research Council (NRC) report on contaminated sediments in ports and waterways (NRC, 1997). Table 1-1 summarizes the primary differences in the source, type, and transport of COPCs in aquatic versus terrestrial sites; the table also summarizes primary differences between ERAs and human health risk assessments (HHRAs) conducted at aquatic and terrestrial sites. Figure 1-1 illustrates the complexity of the aquatic environment at a contaminated sediment site.

Table 1-1. Comparison of Terrestrial and Aquatic Site Investigations

Focus of Investigation	Terrestrial Site	Aquatic Site
COPC Source and Transport	Point and nonpoint sources, generally lower degree of transport away from source area (i.e., concentration gradient away from source area commonly observed)	Commonly multiple point and nonpoint sources contributing to a water body; COPCs may be redistributed by waves and currents and transported away from source area
COPC Type	Various	Primarily persistent, hydrophobic compounds that are nonvolatile, relatively insoluble, and resistant to biodegradation
Ecological Risk Assessment	Site boundaries usually well-defined; significant human disturbance common; large literature database available regarding food web interactions, exposure parameters, and toxicological effects	Often difficult to define site boundaries, especially in offshore areas; human disturbance typically limited; complex food webs that may be difficult to define; literature on exposure parameters and toxicological effects is limited
Human Health Risk Assessment	Multiple, direct, and indirect exposure pathways typically considered (i.e., ingestion, dermal contact, inhalation)	Evaluations often limited to indirect pathways such as ingestion of fish and shellfish

In general, COPCs are released to terrestrial and aquatic environments from point (i.e., spills or discharges) and nonpoint (e.g., combustion emissions, pesticide application) sources. In terrestrial environments, COPCs may be introduced directly or indirectly to soils, whereas sources to sediments are almost always introduced indirectly through the water column. As a result of the influence of the overlying water, chemicals that are volatile or highly soluble in water rarely accumulate to high concentrations in sediment. COPCs that are highly biodegradable or photosensitive (i.e., transformed or degraded by sunlight) also do not tend to persist in aquatic environments. Instead, sediment COPCs generally are those that partition readily into sediments, such as nonionic polar organic compounds and metals. Consequently, sediments with the greatest partitioning capacity, such as those with high clay and organic carbon content, are often the most contaminated. Additionally, sediment-associated COPCs may be redistributed and transported away from the source area by waves and currents and mixed with contaminants from other sources in the water body, thereby complicating source identification (Apitz et al., 2002) (http://meso.spawar.navy.mil/Docs/MESO-02-TM-01.pdf). General discussions of contaminant fate and transport in sediments can be found in Burton (1992) and Allen (1995).

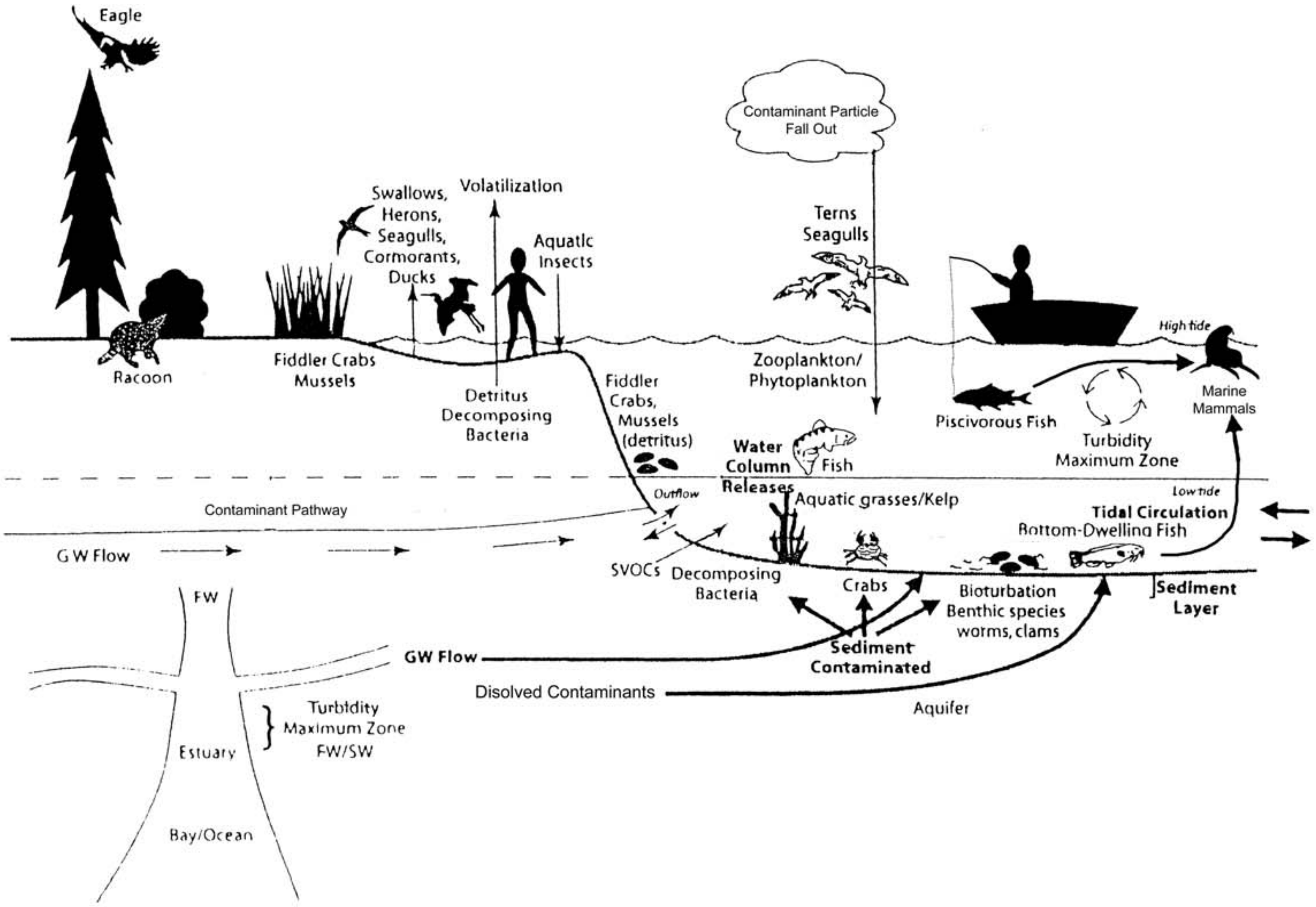


Figure 1-1. Generic Conceptual Site Model Showing Possible Contaminant Exposure Pathways and Receptors in an Aquatic Environment (modified from U.S. EPA diagram)

The overall process for evaluating human health and ecological risk at sediment sites is the same as that followed for terrestrial sites (U.S. Environmental Protection Agency [U.S. EPA], 1989 and 1998). However, there are differences in the process that need to be considered at sediment sites. For example, when evaluating potential ecological risks onshore at a Navy facility, the available habitat typically is clearly delineated by the presence of industrial or residential development. In contrast, it is very difficult to clearly define site boundaries in a submerged offshore area, particularly given the potential transport and redistribution of site contaminants as the result of wave action and currents. It is also important to note that terrestrial ecosystems have been more thoroughly studied than aquatic environments due to issues of accessibility. For human health evaluations, the primary difference is in the identification of exposure pathways: access to submerged sediments is limited; therefore, exposure to humans to offshore sediments is largely associated with indirect pathways such as consumption of contaminated fish and shellfish. Recreational scenarios also may be evaluated as appropriate (e.g. beach use, recreational sports).

1.2 DOCUMENT ORGANIZATION

This document is organized into four sections along with a glossary and references. Hyperlinks that connect the reader to related Web sites and documents are found throughout the document. The main document body is organized as follows:

Section 1 – Introduction presents the purpose and organization of the document, identifies the primary differences in conducting aquatic versus terrestrial studies, and provides overviews of the Navy's approach to evaluating risk and remedial alternatives at sediment sites and of pertinent laws and regulations.

Section 2 – Sediment Site Characterization presents an overview of the site characterization process relative to sediment investigations, including planning considerations, developing a CSM, source identification, defining the nature and extent of contamination, and characterizing contaminant fate and transport. This section also identifies important physical and chemical data that should be collected as part of a sediment investigation, and provides an overview of sample design and sample collection methods and equipment.

Section 3 – Ecological and Human Health Risk Assessment for Sediment Studies follows the stepwise guidance for conducting ecological and human health risk assessments at sediment sites within the Navy's tiered framework. Issues specific to sediment sites are identified and discussed for each tier.

Section 4 – Sediment Remedial Alternative Evaluations addresses FS planning considerations and determination of site-specific risk-based cleanup levels. Remedial options, including monitored natural recovery, in situ capping, and removal, are described along with monitoring considerations and sediment management issues.

Section 5 – Glossary provides a description of common terminology used in this guide and in sediment investigations in general.

Section 6 – References, Resources, and Applicable Web Sites provides references by section along with Web site addresses for information discussed in the guide.

1.3 OVERVIEW OF NAVY POLICY AND GUIDANCE

Highlight 1-1 lists the Navy policies and guidance that apply to sediment site investigations. Specific aspects of the policies and guidance are discussed in Sections 2.0 through 4.0. Links to relevant guidance from the U.S. EPA also are provided. The Navy IR Sediments Framework as presented in the *Policy on Sediment Site Investigation and Response Action* (Chief of Naval Operations [CNO], 2002) is shown in Figure 1-2. Some of the CNO Policy's guiding principles for all sediment investigations are as follows:

- All sediment investigations and response actions must be directly linked to Navy-related CERCLA or Resource Conservation and Recovery Act (RCRA) releases.

Highlight 1-1. Navy and U.S. EPA RI/FS Policies and Guidance

Navy Policy on Sediment Site Investigation and Response Action (CNO, 2002)
- http://enviro.nfesc.navy.mil/erb/erb_a/regs_and_policy/don_policy_sediment.pdf

Navy Policy for Conducting Ecological Risk Assessments (CNO, 1999)
- http://enviro.nfesc.navy.mil/erb/erb_a/regs_and_policy/cno-era-policy.pdf

Navy Guidance for Ecological Risk Assessment
- http://web.ead.anl.gov/ecorisk/

Navy Policy for Conducting Human Health Risk Assessments (CNO, 2001)
- http://enviro.nfesc.navy.mil/erb/erb_a/regs_and_policy/HRApolicy.pdf

Navy Guidance for Human Health Risk Assessment
- http://www-nehc.med.navy.mil/hhra/

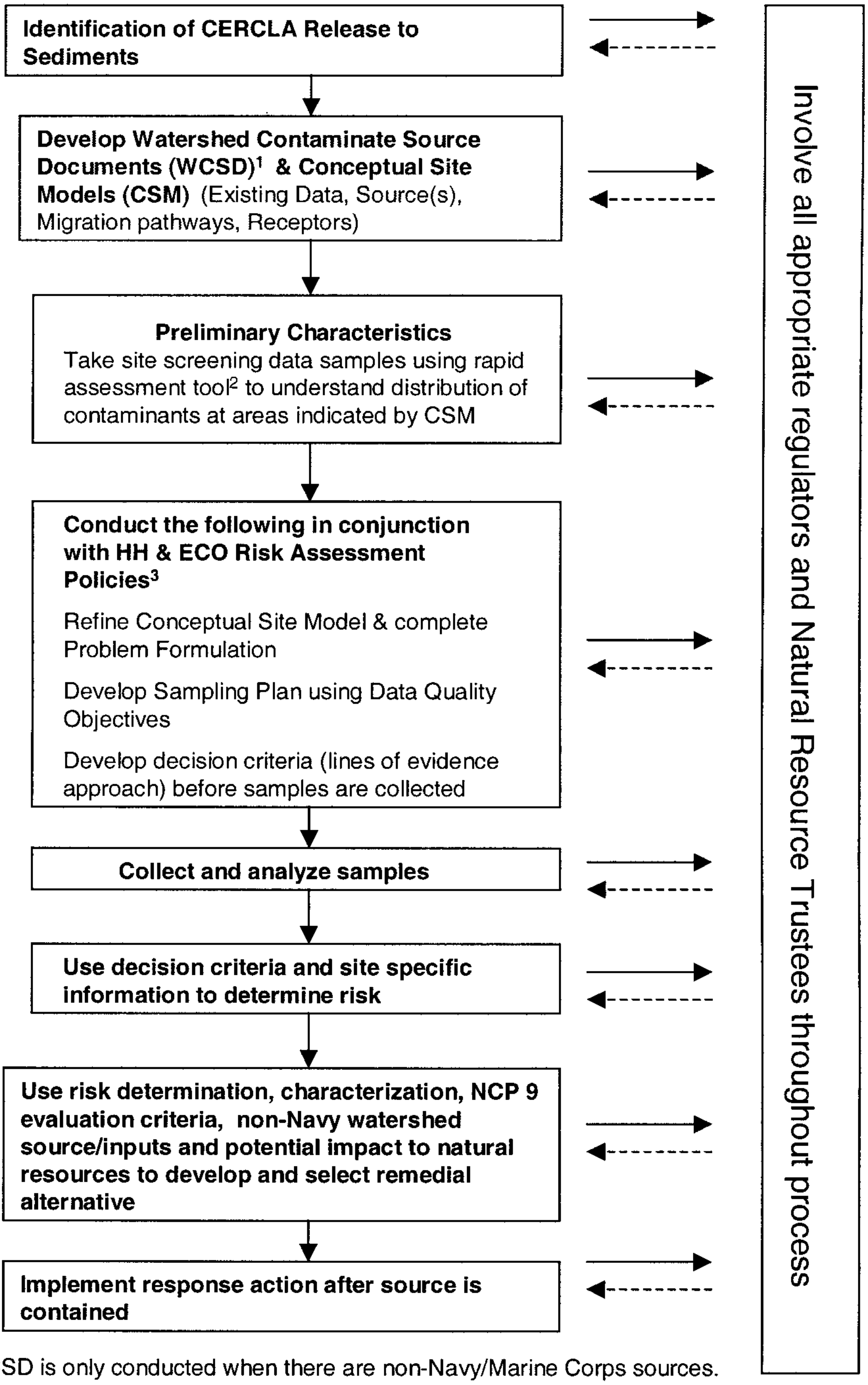
Navy Interim Final Policy on the Use of Background Chemical Levels (CNO, 2000)
- http://enviro.nfesc.navy.mil/erb/erb_a/regs_and_policy/don-background-pol.pdf

U.S. EPA General Superfund Web Site
- http://www.epa.gov/oerrpage/superfund/index.htm
- http://www.epa.gov/oerrpage/superfund/whatissf/sfproces/
  (Index of Superfund document chapters)

U.S. EPA Guidance for Human Health and Ecological Risk Assessments (U.S. EPA, 1989 and 1998)
- http://www.epa.gov/superfund/programs/risk/tooltrad.htm#gp

U.S. EPA Guidance for Conducting the RI/FS under CERCLA (U.S. EPA, 1988)
- http://www.epa.gov/oerrpage/superfund/whatissf/sfproces/rifs.htm

Principles for Managing Contaminated Sediment Risks at Hazardous Waste Sites (EPA OSWER Directive 9285.6-08, February 12, 2002)
- http://www.epa.gov/superfund/resources/remedy/pdf/92-85608.pdf



Notes:

1. WCSD is only conducted when there are non-Navy/Marine Corps sources.
2. A percentage of rapid assessment samples may need lab confirmation.
3. CNO Human Health Risk Assessment & Ecological Risk Assessment Policies.

Figure 1-2. Navy IR Sediments Framework

- If non-Navy sources of contamination are identified at a site, then this information must be documented as early as possible in the RI/FS process and communicated to the appropriate regulatory agencies.

- All sediment investigations and response actions must be scientifically defensible, technically feasible, risk-based, and cost-effective.

- If the reasonably anticipated future land use of property adjacent to the contaminated sediment site is known, then the future land use should be considered in the CERCLA process.

- Stakeholders should be involved early and often in the RI/FS process.

- Risk management decisions can and should be made throughout the RI/FS process.

- Remedial action should not be taken at a sediment site until the primary sources of contamination are controlled or contained, and cleanup levels should not be lower than ambient (i.e., background) chemical concentrations.

Navy sediment investigations will follow the ERA and HHRA tiered approach in accordance with Navy policy and U.S. EPA guidance as identified in Highlight 1-1. Screening and baseline risk assessments (Tiers 1 and 2) are performed as part of the RI. If remedial action is needed at the site based on the findings of the risk assessments, then an FS is performed. The evaluation of remedial alternatives (Tier 3) is performed as part of the FS.

1.4 OVERVIEW OF RELEVANT REGULATIONS, LAWS, AND GUIDELINES

This guide primarily addresses sediment sites managed under CERCLA; however, sediments are subject to a multitude of additional state, national, and international regulatory criteria. It is necessary for the RPM to understand the laws and/or regulations as well as potential applicable or relevant and appropriate requirements (ARARs) that may apply to contaminated sediments at a particular site. According to the NRC study of contaminated sediments in ports and waterways (NRC, 1997), “The mechanisms of the regulatory process in a given situation depend on where the sediments are located; where they will be placed; the nature and extent of the contamination; and whether the purpose of removing or manipulating the sediment is navigation dredging, environmental cleanup, site development or waste management.” As a result, different regulators or stakeholders may focus on different COPCs, cleanup criteria, or goals that drive their actions in the sediment management process. The RPM should consult legal counsel if questions or regulatory conflicts are encountered during the CERCLA process.

Potential ARARs are discussed in more depth in Section 4.1.4; however, it is important to note that no national sediment quality criteria currently are promulgated as ARARs (although national ambient water quality criteria are potential chemical-specific ARARs for sediment sites). Therefore, risk evaluations and sediment cleanup goals must be developed on a site-specific and/or regional basis. For an overview of environmental laws and regulations, see the U.S. EPA Web site at http://www.epa.gov/epahome/laws.htm. Additional legislative requirements are discussed under the National Response Center Web site at http://www.nrc.uscg.mil/nrclegal.html. If the sediment site has a dredging component, then other sources of information are the United States Army Corps of Engineers (USACE) guidelines for dredged material evaluation (http://el.erdc.usace.army.mil/dots/):

- Evaluation of Dredged Material Proposed for Ocean Disposal – Testing Manual (i.e., the "Green Book") (U.S. EPA/USACE, 1991; http://www.epa.gov/OWOW/oceans/gbook/)

- Evaluation of Dredged Material Proposed for Discharge in Waters of the U.S. – Testing Manual (i.e., the Inland Testing Manual) (U.S. EPA/USACE, 1998; http://www.epa.gov/waterscience/itm/).

2.0 SEDIMENT SITE CHARACTERIZATION

This section addresses the site characterization phase of a sediment RI/FS, including planning the project, developing the CSM, source identification, defining the nature and extent of contamination, and characterizing contaminant fate and transport. The section also identifies important physical and chemical data that should be collected as part of a sediment investigation, and provides an overview of sample design and sample collection methods and equipment.

2.1 PLANNING AND EXECUTING THE SEDIMENT STUDY

In the initial stages of the sediment RI/FS, the RPM should build the project team, gather existing data, and develop a preliminary CSM. These activities are discussed further below.

2.1.1 Building the Project Team

In order to ensure a scientifically sound and technically defensible study, the RPM should organize a project team with specialized expertise in sediment investigations and issues. The size and complexity of the sediment site will dictate the size and breadth of expertise required of the team (Burton, 1992; Chapter 14). Specific experts may include but are not limited to those listed in Highlight 2-1. Personnel with the appropriate expertise who have prior knowledge of the site can be valuable assets, as can those with specific knowledge of existing data, including data quality.

Highlight 2-1. List of Experts Needed for Sediment RI/FS

- Chemist (sediment/water/tissue; to include expertise in sample collection, preservation, transportation, and laboratory analysis)
- Geologist and/or hydrogeologist; preferably with hydrodynamic/fate and transport modeling expertise
- Geomorphologist
- Geochemist
- Toxicologist (aquatic and terrestrial)
- Ecologist
- Marine/fisheries/benthic biologist
- Aquatic ecological and human health risk assessment experts
- Statistician
- Feasibility study and sediment remedy selection expert
- Engineer

2.1.2 Gathering Existing Data

In the initial stages of the RI/FS, existing data should be gathered and a site visit should be conducted if it has not already been completed. For aquatic sites, the following information should be compiled in addition to the data previously collected for the preliminary assessment/site inspection (PA/SI) phase of the RI/FS:

- Charts and bathymetric surveys of the site water body should be obtained from other Navy sources, the National Oceanographic and Atmospheric Administration (NOAA), the U.S. Coast Guard, the U.S. Geological Survey (USGS), or various state agencies.
- Data for tides, waves, currents, and winds also should be obtained from these sources to support the assessment of contaminant fate and transport (see Section 2.4).
- Information on the adjacent onshore area (e.g., topography, hydrogeology, and environmental condition).
- Data from benthic community surveys, creel samples, or other biological tests.
- Most U.S. bays and harbors have ongoing environmental monitoring programs administered by local agencies that can provide useful information, including data regarding ambient conditions and biological communities.
- Other potential sources of information include published studies, spill reports from the Coast Guard, dredging assessments, National Pollutant Discharge Elimination System (NPDES) permits, and the like.
- Regional or other publicly available data also should be reviewed to identify any potential non-Navy sources of contamination in the vicinity of the site.

In addition to gathering existing data, a site visit is recommended. The objective of the site visit is to understand the physical site setting, identify preliminary COPC sources, and gather relevant background information. Highlight 2-2 lists the information that the RPM should take particular note of during the site visit. If possible, the site should be inspected from a boat to allow examination of the shoreline from the water. If the site is tidally influenced, then the RPM should consider inspecting the site at both high and low tides. If possible, adjacent properties also should be examined to identify other potential sources of contamination to the water body. Non-IR site-related potential sources of contamination also should be identified, such as permitted stormwater discharge pipes.

2.1.3 Developing a Preliminary Conceptual Site Model

Existing information for the sediment site should be used to develop a preliminary CSM. The CSM identifies known or suspected contaminant sources, release and transport mechanisms, contaminated media, exposure routes, and receptors. A CSM may be constructed in several ways, depending upon the amount of information available. The most commonly used method is to construct a simple CSM that identifies broad classes of ecological or human receptors that may be at risk from exposure to sediment contamination. Figure 2-1 is an example of a preliminary CSM for a

Highlight 2-2. Information to be Collected During a Sediment Site Visit

- Site layout, topography, and configuration of water body; particularly notation of features that drain into the water body
- Nature of shoreline (e.g., presence of riprap or debris, slope, type and quantity of vegetation)
- Potential onshore sources of contamination to the water body
- Ecological habitats and potential receptors
- Apparent use of the site for fishing or shellfish harvesting
- Boating activity
- Current and anticipated future use of the water body
- Potential offsite sources of contamination to the water body
- Anecdotal information regarding recreational or commercial fishing from local fisherman.

Navy sediment site that has been contaminated by the release of chemicals from a stormwater outfall and a landfill located in the adjacent nearshore area. This simplistic model is expanded and refined as additional site-specific information is collected, with refined CSMs developed for the ERA and HHRA (see Sections 3.1 and 3.2, respectively). The development of a CSM for a sediment site is discussed further in Chapter 3 of *Critical Issues for Contaminated Sediment Management* (Apitz et al., 2002). Guidance on the development of CSMs is provided on the Navy's ecological and human health risk assessment web pages (http://web.ead.anl.gov/ecorisk/process/html/ch2/ and http://www-nehc.med.navy.mil/hhra/guidancedocuments/process/pdf/plan_scope.pdf).

The sediment site characterization effort focuses on the initial stages of CSM development: source identification, contaminant fate and transport, and extent of contaminated media. These topics are discussed further in Sections 2.2, 2.4, and 2.5, respectively. The refinement of exposure pathways and receptors as part of the ERA and HHRA are addressed in Section 3.0.

2.2 SOURCE IDENTIFICATION

Historical site activities and potential sources of contamination are initially identified during the PA/SI. Table 2-1 identifies the COPCs that typically are encountered in sediments at Navy sites and the sources of these chemicals. The most common mechanisms that release these COPCs to the aquatic environment include discharges from outfalls, spills or discharges from ships, surface water runoff, groundwater discharge, and erosion and transport of contaminated surface soils from onshore areas.

Sediments at Navy installations located near urban and industrial areas may be affected by contamination from multiple sources, both Navy and non-Navy. Because of the complex and dynamic hydrogeologic setting of many of these sites, it can be difficult to distinguish contributions from

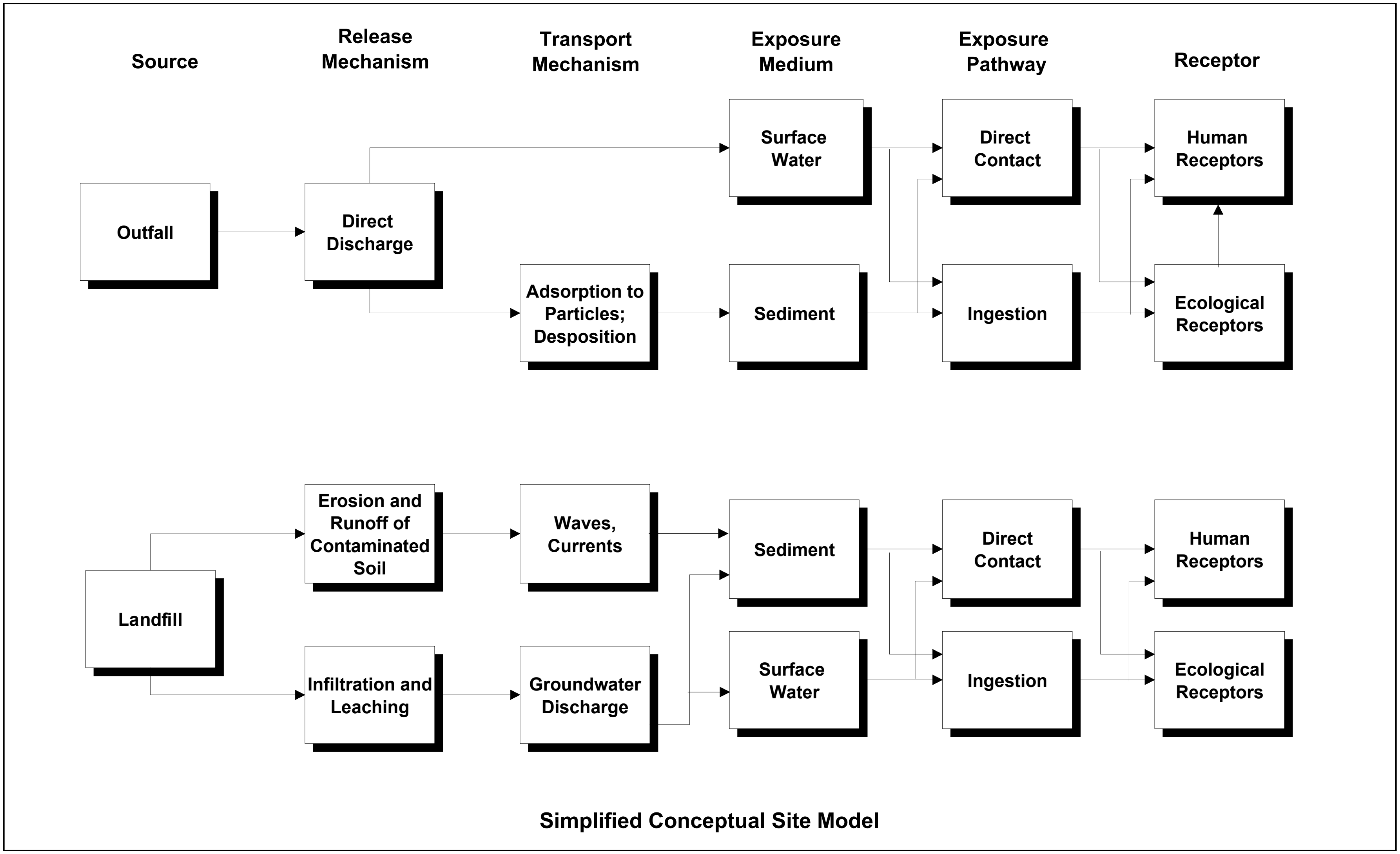


Figure 2-1. Simplified Conceptual Site Model for a Sediment Site

Table 2-1. Common Navy Sediment COPC Classes and Potential Sources

COPC Class	Potential Sources
Heavy and Trace Metals	Ship maintenance and building; aerial fallout; sewage effluent; fungicides (As, Cr, Hg); old paint (Cu, Pb, Zn); marine antifoulants (Cu, Pb, Sn); ballast in submersibles (Hg); former gasoline additives (Pb); naval aviation (Cr, Cd, Pb)
Chlorinated Pesticides	Historical pest control; agricultural runoff, skeet
Polycyclic Aromatic Hydrocarbons (PAHs)	Fuel operations and spills; creosote pier pilings; coal tar; asphalt; fossil-fuel combustion particulates from aerial fallout and road runoff
Polychlorinated Biphenyls (PCBs)	Electrical capacitors and transformers, adhesives, hydraulic oils and paints
Organotins	Marine antifoulant used in vessel paints

various sources. In accordance with the CNO *Policy on Sediment Site Investigation and Response Action* (CNO, 2002; see Highlight 1-1), the RPM must prepare a Watershed Contaminated Source Document (WCSD) if the sediment site is potentially affected by contamination from non-Navy sources. The WCSD is discussed further in Section 2.3. Several methods can be used to identify Navy-related releases and support source identification, including analysis of the spatial distribution of COPCs (see Section 2.5), and specialized chemical analysis to identify chemical “fingerprints” that are unique to a specific source (see Section 2.6.1.2). Data quality objectives (DQOs) for source identification should be developed as part of the RI data collection effort.

2.3 WATERSHED CONTAMINANT SOURCE DOCUMENT

The purpose of the WCSD is to document the existence of both the Navy and other parties whose activities may have had or could continue to have an impact on sediments. The WCSD should generally be no more than 2 to 10 pages in length. The WCSD should include a graphical representation of a CSM. The WCSD should be prepared at the earliest point in the RI/FS process where sufficient data are available to support the CSM and associated interpretations and conclusions. If it is determined that a significant amount of site contamination is due to non-Navy sources, then the appropriate regulators should be informed using the WCSD, and the RPM should consult with counsel to determine the appropriate course of action. Naval Facilities Engineering Command (NAVFAC) Headquarters also should be notified.

The development of a WCSD, if determined necessary, can be helpful for numerous reasons when multiple sources could potentially contribute to the contamination observed at a sediment site.

- A WCSD can give a broad perspective of the potential origins, fate and transport, and overall influences of contaminants on a watershed and how they relate to the sediment site being investigated within that watershed to all the stakeholders.
- When conducting a Feasibility Study (FS) evaluation, a WCSD can aid in the evaluation of alternatives and the understanding of the potential for recontamination (from non-IR related Navy and/or non-Navy sources) under each alternative.

- A WCSD can assist in formulating DQOs for designing remedial investigations and/or developing a long-term monitoring plan following a remedial action (e.g., building into decision rules considerations for assessing recontamination potential from non-Navy sources).
- A WCSD can assist prioritizing source control measures.

There are seven basic steps to initially determining the need and scope for (Step 1), and if necessary, proceeding to the subsequent steps (Steps 2-7) for the development of a WCSD. These steps provide a logical and general sequence for RPMs to follow in identifying the need and, if necessary, then developing a WSCD. These seven steps are shown in Highlight 2-3.

When conducting literature searches in the development of a WCSD, information can be gathered from a variety of sources, including information collected or gathered by states (e.g., state environmental or health departments), other federal agencies (e.g., U.S. EPA, NOAA, U.S. Fish and Wildlife Service, USACE, etc.), or by the Navy itself. For example, the U.S. EPA has databases, which allow for searches to focus on the hazardous waste sites or facilities holding water discharge permits near a Navy facility and a subject sediment site. The Comprehensive Environmental Response, Compensation, and Liability Information System (CERCLIS) database, located at http://www.epa.gov/superfund/sites/cursites, contains general information on hazardous waste sites across the nation and U.S. territories including location, status, contaminants, and actions taken. The Permit Compliance System (PCS) database in Envirofacts, located at http://www.epa.gov/enviro/tml/pcs/pcs_query.html, allows for searches to be conducted for facilities holding National Pollutant Discharge Elimination System (NPDES) permits. Many states also have similar databases or information on their internet sites that could further help with gathering relevant information for building a WCSD.

More information on the purpose, development procedure, effort required, and specifics on the content that should be contained within a WCSD can be found in the CNO WCSD Fact Sheet, located at http://web.ead.anl.gov/ecorisk/related/documents/WCSD_Factsheet_Final_v2.pdf. RPMs also can obtain additional information on WCSDs by contacting their EFD/EFA Risk Assessment Workgroup (RAW) member or by contacting a member of the RAW sediment subgroup.

2.4 CONTAMINANT FATE AND TRANSPORT

Various fate and transport mechanisms will influence the movement, partitioning, and/or degradation of COPCs in the aquatic environment (Allen, 1995; Burton, 1992; U.S. EPA, 2002). This section presents an overview of the most important fate and transport processes at sediment sites and provides guidance on data that should be collected to characterize these mechanisms.

Major processes affecting the fate of contaminants in sediment are shown in Figure 2-2 (adapted from Allen, 1995) and are described below. Many persistent COPCs, particularly hydrophobic organic compounds, tend to adsorb to clay- and silt-sized sediment particles as well as to organic material. Therefore, the dominant transport mechanism for these contaminants is the movement of sediment particles. Chemical and biological transformation processes also will influence the fate and transport of sediment contaminants.

Highlight 2-3. Seven Steps to Developing a Watershed Contaminated Source Document (WCSD)

Step 1 Determine the Need and Scope of WCSD

- Conduct Internal Discussion
  - Identify if the Navy is the only source of potential contamination to a Navy IR sediment site.
  - Identify if other non-Navy sources could potentially contribute or have historically contributed to potential contamination at the site.
  - Identify if any potential contributions from non-Navy sources could contribute to overall risks and any potential issues regarding long-term remedial strategies for the site.
  - If RPMs and management decide that other non-Navy sources contributed to sediment contamination, a WCSD is required. Proceed to Define Scope.
- Define Scope
  - Before proceeding to Step 2, define the scope of the area a WCSD will cover.
  - The scope of a WCSD should be limited to the area and activities that may have the most impact on a Navy sediment site.
  - The scope of a WCSD may be different depending upon the water body type (e.g., river, pond, bay, etc.).

Step 2 Conduct Literature Search

- Conduct a literature search to gather supporting information
  - Conduct online search.
  - Review databases.
  - Review public records.
  - Review periodic journal records.
- After conducting literature search, if it still remains evident that other non-Navy sources could still play a potential role in the assessment and/or management of a sediment site, then proceed to Step 3.

Step 3 Develop Preliminary Watershed Conceptual Map

- Develop Spatial Map
  - Plot findings from literature search on map.
  - Identify and plot on the map all of the potential sources (i.e., Navy and non-Navy) found in the literature search.
  - Identify potential non-Navy sources both current and historic by general source type (e.g., industrial outfall, former wood treating facility, National Priorities List (NPL) site, stormwater discharge outfall, etc.) and **NOT** by specific identity (e.g., ABC corporation industrial outfall, City of XYZ stormwater outfall, etc.).

Step 4 Conduct Watershed Visit

- Conduct watershed visit to verify accuracy of spatial map (e.g., locations of outfalls, non-Navy cleanup sites, etc.) within the scope identified in Step 1.
- Confirm or deny any information that can be verified visually using the previously completed literature search. For some potential historical sources (e.g., location of former industrial facility now occupied by commercial business park), visual verification based on current conditions may not be possible, but nevertheless should still be considered in developing a comprehensive WCSD.
- If the site visit reveals other potential sources that were not identified during the literature search, then update documentation.

Highlight 2-3. Seven Steps to Developing a Watershed Contaminated Source Document (WCSD) (continued)

Step 5 Research Record to Fill Data Gaps

- Using information from the watershed visit, update the understanding and potential role of all possible sources.
- Conduct additional review of literature if necessary.

Step 6 Develop Conceptual Site Model (Pictorial)

- Using an updated map originally developed in Step 3, the RPM should develop a pictorial conceptual site model which should include:
  - Watershed sources (all potential sources [Navy/non-Navy])
    - As mentioned in Step 3, the identification of potential non-Navy sources must be by general source type and not by specific identity.
    - Watershed sources can be color coded by type of source (e.g., Navy sources, stormwater outfalls, NPDES-permitted outfalls, cleanup sites, industrial facilities).
  - Identify general hydrodynamic conditions of the water body (e.g., general flow direction, tidal movement)
  - Identify navigational channels, if applicable.
  - Identify general transport mechanisms indicating how contamination may enter a water body.

Step 7 Write Watershed Contaminated Source Document

- A general outline that can be used by RPMs in development of a WCSD is as follows:
  - Introduction
    - Overview of why a WCSD is being completed (e.g., required by CNO Policy)
    - Which IR site/s are included in discussion
    - Purpose (what does this mean and what it does not mean)
    - Scope of what the document covers
  - General setting
    - Operations of the installations
    - Extent of area covered by the facility (spatially)
  - Overview of Literature Search Sources
    - Sources list (e.g., Navy, Public Record, regulatory data, etc.)
  - Results
    - Summarize findings of the literature search
    - Include conceptual site model
  - Conclusions and Recommendations
    - Conclusions regarding results
      - For example, is there potential for non-Navy sources to contribute to overall contamination?
      - What specific sources (both Navy and non-Navy) are likely to contribute primarily to observed sediment contamination?
    - Recommendations
      - For example, how should results be taken into account when considering investigation, remediation, or long-term monitoring strategies of a sediment site?
  - References

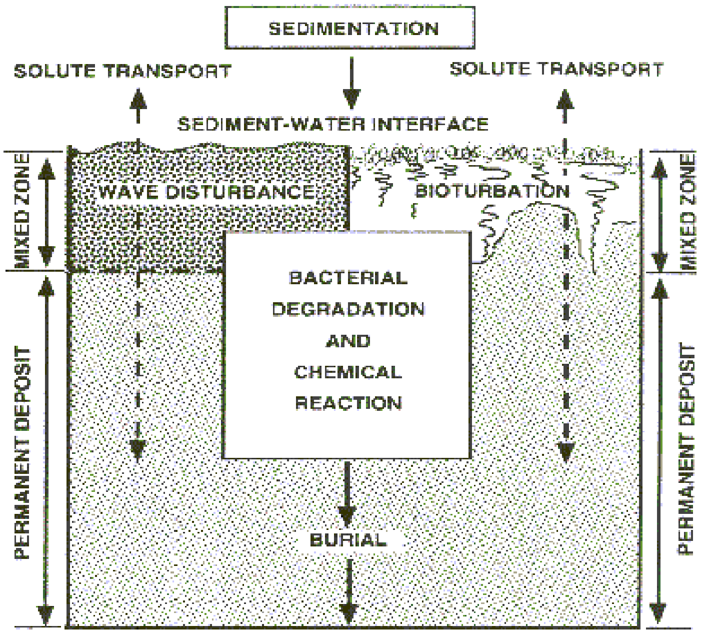


Figure 2-2. Schematic Showing Major Processes Affecting the Fate and Transport of Contaminants in Sediments
(Reprinted with permission from Metal Contaminated Aquatic Sediments, H.E. Allen. Copyright CRC Press, Boca Raton, FL)

The sediment bed in relatively quiescent areas where contaminants tend to accumulate generally consists of a less consolidated surface layer (i.e., a mixed zone) that is biologically active and more readily eroded and resuspended by waves, currents, propeller wash, and other disturbances. If surface sediments are eroded and resuspended in the water column, they can be transported by wind or tidal currents and redeposited in areas where current speeds are reduced.

Deeper subsurface sediments tend to be more consolidated and isolated from aquatic biota unless exposed by dredging, construction, or an extreme erosional event (e.g., a flood or severe storm). In a net depositional environment, surface sediments will eventually be buried to a depth below the mixed zone by accumulating particles. The accumulating sediment will be relatively cleaner if the sources of contamination to the water body have been controlled or eliminated.

The primary chemical parameter that influences the mobility of many contaminants, particularly metals, is redox potential (Eh). In anoxic (i.e., oxygen depleted) sediment layers with a low Eh, most of the metals are bound to sulfide, carbonate, or organic matter (Allen, 1995). As Eh increases, the sulfides and carbonates may dissolve, releasing the metals in soluble forms. In oxic (i.e., oxygen rich) layers of sediment, most of the metals are complexed to iron and manganese oxide coatings on clay particles. As Eh decreases, the iron and manganese oxides dissolve, releasing metals into solution. Thus, fluxes of metals from sediments into the overlying water column are greatest during changes in redox potential.

During the site characterization phase of a sediment investigation, the RPM should consult with a sediment transport expert and geochemist in order to identify the probable dominant fate and transport processes at a particular site. Some of the questions that should be addressed are provided in Highlight 2-4. Many of these questions can be answered in a qualitative or semiquantitative manner using available site data for sediment grain-size distribution, total organic carbon (TOC) content, sediment COPC concentrations, and acid volatile sulfide/simultaneously extracted metal (AVS/SEM) concentrations (Allen et al., 1993) in conjunction with the existing data described in Section 2.1.2. This initial fate and transport information can be incorporated into the preliminary CSM.

As the sediment investigation progresses and more information becomes available, a more comprehensive and quantitative evaluation of fate and transport may be desired (particularly if monitored natural recovery appears to be a likely remedial alternative for consideration in the FS). In this

Highlight 2-4. Characterizing Contaminant Fate and Transport at a Sediment Site

- What is the distribution of sediment grain size (i.e., sediment type) at the site, and what are the associated depositional environments?
- Under what conditions are the surface sediments likely to be eroded and resuspended (i.e., how stable is the sediment bed)?
- If sediments are resuspended, where are they being transported?
- Is natural burial occurring through sediment accumulation, and if so, at what rate?
- How thick is the mixed surface layer of sediment?
- What types of extreme events might occur at this site, and what are the potential effects?
- Are surface sediments oxic (oxygen rich) or anoxic (oxygen depleted), and how does the redox potential change with depth?
- What is the TOC content and ratio of AVS to SEM?
- What is the flux of COPCs from the sediment bed into the overlying water column?
- What chemical and biological processes might be degrading or transforming COPCs, and are these processes significant?

case the RPM should consider using more sophisticated tools such as site-specific measurements of sediment transport and sediment accumulation rates; use of hydrodynamic and sediment transport models to predict sediment transport patterns, including extreme event analysis; and evaluation of contaminant desorption and/or degradation rates and processes. A detailed technical discussion regarding the evaluation of sediment stability can be found in the proceedings of a January 2002 U.S. EPA-sponsored Sediment Stability Workshop at http://www.hsrc.org/hsrc/html/ssw/sedstab/notes.pdf. Additional information on tools and techniques that can be used for this type of data collection effort is included in Section 2.6.2.

2.5 DEFINING THE NATURE AND EXTENT OF CONTAMINATION

Defining the nature and extent of contamination can be more difficult for a sediment site than for a terrestrial site because of the greater potential for contributions from multiple point and non-point sources and the potentially broad dispersal of contaminated sediments by hydrodynamic processes. However, in many cases, a concentration gradient away from the original source of contamination is observed even if the sediments have been reworked by hydrodynamic processes (Apitz et al., 2002). In relatively quiescent environments (i.e., areas with weak tidal circulation and little wave activity), localized areas with high chemical concentrations (i.e., hotspots) may persist for a long period of time. In areas affected by nonpoint sources of contamination and/or a greater degree of sediment transport, contamination may be more widespread but at lower levels. Although the bulk of the data needed to establish the nature and extent of contamination should be collected during the

RI, additional data can be collected after preliminary remediation goals (PRGs) have been developed (i.e., as part of the FS) to provide more accurate estimates of sediment volumes for evaluation of remedial alternatives.

The sample design used to establish the horizontal and vertical extent of contamination will depend upon the CSM and site-specific DQOs. An overview of sample design is provided in Section 2.7. In general, the extent of sediments with COPC concentrations exceeding either an established level of concern or regional ambient (i.e., background) levels must be defined. The *Navy Interim Final Policy on the Use of Background Chemical Levels* (CNO, 2000; see Highlight 1-1) specifies that background chemical levels should be established as early as the PA/SI phase and used to identify chemicals that are in the environment due to releases from the site. *Navy Guidance for Environmental Background Analysis, Volume II: Sediments* (NFESC UG-2059-ENV, Apr 2004 or as a part of the NAVFAC Technology Transfer Tool at http://www.ert2.org/BackgroundAnalysis/tool.aspx) provides details on methodologies for establishing background conditions at sediment sites. Surface sediment chemistry data (i.e., representing the biologically active zone) are needed to support the ERA and HHRA as well as the site characterization. Subsurface sediment chemistry data are needed to establish the historical input of contaminants, evaluate the degree of natural recovery (if any), and support the evaluation of remedial alternatives (i.e., depth of dredging that would be required to reach a clean layer).

It can be difficult to predict the maximum depth at which to collect sediment core samples in order to encounter a “clean” layer. Any information on regional and local sediment stratigraphy should be examined to identify older, more consolidated sediment layers that are unlikely to be affected by contamination. Information on sedimentation rates (if available) can be used to predict the depth at which sediments that predate site activities are likely to be found. In the absence of any relevant information, sediment cores should be collected to greatest feasible depth; deeper samples can be frozen and archived for future analysis if the vertical extent of contamination cannot be established from the shallower samples.

Because of the potential widespread distribution of contaminants in the aquatic environment, it may not be feasible or cost-effective to collect numerous sediment samples for full laboratory analysis to define the nature and extent of contamination. Rapid sediment characterization tools such as immunoassay and x-ray fluorescence (XRF) analysis can be used to map the distribution of contamination and refine the preliminary CSM in a relatively fast and inexpensive manner. The Navy IR Sediments Framework (see Figure 1-2) specifies the use of rapid assessment tools in the initial phases of the investigation to understand the distribution of contaminants at the site. A detailed description of rapid sediment characterization tools is provided in Appendix A. Fixed laboratory analysis of a subset of sediment samples can provide confirmatory data and allow development of a correlation between screening and lab measurements. Collection of blind duplicates for screening and laboratory analysis also can provide useful information and increase confidence in the results. The sediment screening results then can be used to focus the sample design for the baseline ERA and HHRA.

2.6 SITE CHARACTERIZATION PARAMETERS

This section provides an overview of the key chemical and physico-chemical parameters that characterize a sediment site and identifies methods that can be used to measure these parameters. Data that can be collected to support the evaluation of remedial alternatives also are described. Site-specific biological data also are required for most sediment investigations to evaluate ecological and human health risk; the most common types of biological data are described in Section 3.1.2.3. Data collection efforts for the site characterization, risk assessments, and evaluation of remedial alternatives should be coordinated as much as possible to optimize the efficient use of resources and avoid unnecessary schedule delays.

2.6.1 Chemical Characterization

This section addresses the measurement of organic and inorganic COPCs in sediment and tissue samples as well as the use of geochemical relationships and specialized chemical analyses to both identify site-related contamination and fingerprint potential contaminant sources. The list of COPCs for evaluation in the RI should be determined on a case-by-case basis depending on historical site activities and potential sources. However, at least some sediment samples should be analyzed for a full suite of chemicals (i.e., the COPC classes identified in Table 2-1) to provide sufficient data for the screening-level risk assessment and rule out the presence of other COPCs early in the process (see Sections 3.1.1 and 3.2.1).

2.6.1.1 Sediment Chemistry Analyses

Analysis of COPCs in sediment often requires specialized chemistry methods because standard U.S. EPA SW-846 methods were designed for solid wastes and usually are not appropriate for analysis of sediment (unless methods are modified). In addition, the quantitation limits and laboratory detection limits achieved by standard methods commonly exceed risk-based ecological benchmark values for sediments. Detection limits and their importance in the risk assessment process are discussed in “Laboratory Detection Limits and Reporting Issues Related to Risk Assessment” (Corl et al., 2002; http://www-nehc.med.navy.mil/hhra/guidancedocuments/issue/pdf/ FDI.pdf). This paper provides general information on detection limits and describes steps that can be taken to improve a laboratory’s ability to achieve the detection limits needed to meet site-specific DQOs.

Modifications to standard methods have been developed to remove analytical interferences due to salt and organic matter, achieve ultra-low detection limits, and expand the list of target analytes so that the sediment chemistry data are suitable both for site characterization and risk assessment. References for specialized sediment chemistry methods are provided in Table 2-2. Methods for analysis of organic and inorganic analytes in sediment and tissue samples were developed for the NOAA National Status and Trends Program (NOAA, 1993 and 1998). Selection of appropriate analytical techniques for sediment samples from freshwater, estuarine, and saline environments and corresponding method references also are discussed in the Inland Testing Manual, which provides guidelines for dredged material evaluations (U.S. EPA/USACE, 1998; see Section 1.4).

Certain classes of compounds can be analyzed either as individual compounds or as functional groups. For example, PCBs may be quantified either as Aroclors or as individual PCB congeners. Aroclors represent commercial mixtures containing a specified percentage of individual PCB congeners. Total PCB concentrations may be derived either by summing the concentrations of the individual Aroclors or by summing the most commonly analyzed congeners and multiplying by a factor of approximately two (NOAA, 1997). Because Aroclor mixtures may change over time due to weathering, evaluation of the individual congener data using techniques similar to those used to fingerprint petroleum products (Stout et al., 1998) may provide more useful information with regard to potential sources. However, congener data are not directly comparable to historical Aroclor data. Therefore, the decision on how best to evaluate PCBs should be made on a site-by-site basis. An issue paper addressing the selection of appropriate methods for PCB analysis is currently in preparation. Similar considerations should be given to the evaluation of total versus individual PAHs.

Table 2-2. Selected Site Characterization Parameters and Methods

Parameter	Importance	Suggested Method
Chemical Characterization		
Heavy and trace metals	Potential COPC	Total acid digestion methods referenced in NOAA Volume III – Technical Memorandum #71: http://www.ccma.nos.noaa.gov/publications/tm71v3.pdf With EPA 6010, 6020 & 7000 series found at http://www.epa.gov/epaoswer/hazwaste/test/main.htm EPA 7470A recommended for Hg; EPA 6020 recommended for trace metals other than Hg Rapid sediment characterization methods – see Appendix A
Chlorinated pesticides	Potential COPC	EPA 8082 modified following NOAA Technical Memorandum #130: http://www.ccma.nos.noaa.gov/publications/tm130.pdf
PAHs	Potential COPC	EPA 8270 modified for SIM w/extended analyte list to include alkylated homologues w/alumina and gel permeation chromatography cleanup U.S. EPA 40 Code of Federal Regulations (CFR)-J Part 300, Subpart L, Appendix C, Par. 4.6.3-4.6.5 Rapid sediment characterization methods – see Appendix A
PCBs	Potential COPC	EPA 8082 modified for congener analysis following NOAA Status & Trends Methods - Technical Memorandum 130 in http://www.ccma.nos.noaa.gov/publications/tm130.pdf EPA 1668A for PCB congeners Rapid sediment characterization methods – see Appendix A
Organotins	Potential COPC	NOAA Status & Trends Methods - Technical Memorandum 130 in http://www.ccma.nos.noaa.gov/publications/tm130.pdf

Table 2-2. Selected Site Characterization Parameters and Methods (page 2 of 3)

Parameter	Importance	Suggested Method
Physico-Chemical Characterization		
Sediment grain size	Finer-grained sediments tend to adsorb contaminants; grain size can be used to characterize depositional environment and sediment dynamics; grain size affects benthic community structure	Inland Testing Manual, Chapter 9 (U.S. EPA/USACE, 1998) http://www.epa.gov/ost/itm/ITM/ch9.htm Analytical method (NOAA Technical Memo. 130): http://www.ccma.nos.noaa.gov/publications/tm130.pdf ASTM D 422
Total organic carbon (TOC)	TOC tends to sorb contaminants and reduce their bioavailability	Inland Testing Manual, Chapter 9 (U.S. EPA/USACE, 1998) http://www.epa.gov/ost/itm/ITM/ch9.htm Analytical method (NOAA Technical Memo. 130): http://www.ccma.nos.noaa.gov/publications/tm130.pdf
Acid volatile sulfide (AVS)/simultaneously extracted metals (SEM)	Evaluation of metal bioavailability	Di Toro et al., 1990 Simpson, 2001
Redox potential (Eh)	Influences species and bioavailability of metals	Table G-1, Appendix G of U.S. EPA (2001) Plumb, 1981
pH	Influences species and bioavailability of metals	Commercially available pH meter (Plumb, 1981)
Salinity of porewater (marine/estuarine sediments)	Can cause matrix interferences in some chemical analyses; affects benthic community structure; important in selection of bioassay test species; affects metal speciation, sediment pH, and partitioning	Various methods; see Appendix G of U.S. EPA (2001) http://www.epa.gov/waterscience/cs/biotesting/index.html
Alkalinity of porewater (freshwater sediments)	Influences species and bioavailabilty of metals	ASTM (2000)
Ammonia in porewater	Naturally occurring toxicant in organically-enriched sediments	Inland Testing Manual, Chapter 11 (U.S. EPA/USACE, 1998) http://www.epa.gov/ost/itm/ITM/ch11.htm

Table 2-2. Selected Site Characterization Parameters and Methods (page 3 of 3)

Parameter	Importance	Suggested Method
Site Characterization; Contaminant Fate and Transport		
Visual description of sediment cores	Depositional environment and sediment dynamics be inferred and subsurface sediment characteristics can be documented	ASTM D 4288
Radioisotope profiling (e.g. ^{210}Pb and ^{137}Cs isotopes)	For areas that meet criteria (i.e., undisturbed areas of sediment accumulation), provides estimates of sediment accumulation rate and degree of vertical mixing	USGS, 1998 (http://fl.water.usgs.gov/PDF_files/fs73_98_holmes.pdf)
Hydrodynamic and sediment transport measurements	Evaluate sediment stability and fate and transport of sediment-bound contaminants	No standard methods available; study design should be developed based on site-specific characteristics. See discussion in U.S. EPA Sediment Stability Workshop Notes (http://www.hsrc.org/hsrc/html/ssw/sedstab/notes.pdf)
Contaminant flux	Evaluate relative importance of diffusion, advection, erosion, degradation, and sedimentation processes	No standard methods available; study design should be developed based on site-specific characteristics. See Chadwick and Apitz (2001) for description of Pathway Ranking for In Place Sediment Management (PRISM).
FS-Related Characterization		
Hazardous waste characterization	Evaluate sediment disposal options	Testing requirements vary depending on location of site
Dewatering characteristics	Optimal method for dewatering can be identified	No standard methods available; study design should be developed based on site-specific characteristics; discussion of various dewatering technologies provided in U.S. EPA (1994)
Bearing capacity	Evaluate ability of sediment to support cap	ASTM D 1883
Plasticity	Evaluate sediment handling characteristics	ASTM D 4318
Density	Evaluate ability of sediment to support cap	Standard Proctor Test; ASTM D 698 Method A

Sediment results typically are reported on a dry-weight basis with percent moisture and TOC data included as ancillary information. Tissue data also should be reported on a dry-weight basis with percent lipid and percent moisture data included as ancillary information so that data can be converted to a wet-weight or lipid-normalized basis for use in risk assessments. Reporting units should always be clearly identified on data tables (i.e., whether results are reported on dry-weight or wet-weight basis). Analyses of estuarine and marine sediment, water and tissue samples should always be performed by laboratories with demonstrated experience in successfully performing the required analyses.

2.6.1.2 Chemical Fingerprinting

A number of established geochemical relationships can be used to identify sediment contaminant inputs and sources (e.g., Bertine and Goldberg, 1977; Ackerman, 1983; Trefry et al., 1985; Klamer et al., 1990; Schropp et al., 1990; Daskalakis and O'Connor, 1995). An overview of these studies is provided in Appendix B). Additional information on forensic methods for identification of metal contamination can be found at the NOAA Web site (http://www.nwfsc.noaa.gov/publications/techmemos/tm16/tm16.htm). For example, metal/aluminum ratios can be used to identify contamination that exceeds ambient levels and which therefore might be site-related. Naturally occurring background metals typically are part of the aluminosilicate (i.e., clay) mineral structure, and a regression of background metals versus aluminum concentrations will produce an approximately straight line. This regression relationship can be generated on a regional basis using ambient or reference site sediment chemistry data. If metals concentrations in site samples are greater than those predicted by the regression, then those metals may be due to an additional (and possibly site-related anthropogenic) source. Additional information on this methodology can be found in the *Navy Guidance for Environmental Background Analysis, Volume II: Sediment* ((NFESC UG-2059-ENV, Apr 2004 or as a part of the NAVFAC Technology Transfer Tool at http://www.ert2.rg/Background Analysis/tool.aspx).

Other chemical fingerprinting methods can be used to identify sources of petroleum and chlorinated hydrocarbon contaminants (e.g., PAHs or PCBs). The successful use of data for any particular class of compounds to "fingerprint" a sample depends on the following:

- Ability to differentiate chemicals from different geological and anthropogenic sources;
- Relative state of weathering (or aging) of organic contaminants;
- Presence of specific product additives and refinery process signatures for interpretation of petroleum-related contamination; and,
- Availability of data about reference source materials for comparison with site data.

Detailed descriptions of organic contaminant source identification methods can be found in various publications (Page et al., 1995; Douglas et al., 1994 and 1996; Steinhauer and Boehm, 1992). Use of these and other source identification methods usually requires data for extended or modified target analyte lists. For example, in addition to the 16 priority-pollutant PAH compounds, data for alkylated homologues (e.g., C1-C4 naphthalenes, C1-C4 phenanthrenes/anthracenes), and biomarkers (e.g., triterpanes, steranes) are required for source identification of PAHs. Source identification of PCBs may require detailed PCB congener analysis if the Aroclor pattern is not specific to one source.

A summary of the most pertinent points regarding the chemical characterization of a sediment site is provided in Highlight 2-5.

Highlight 2-5. Chemical Characterization Summary

- Use published analytical methods, modified as appropriate, for sediment and aquatic matrices that achieve detection limits suitable for risk assessment; and identify target analytes suitable for source identification
- Use a laboratory that is experienced in the use of appropriate sample cleanup methods to reduce potential interference from organic matter and salt (in marine environments)
- Report sediment results on a dry-weight basis with percent moisture and TOC content as ancillary data
- Report tissue results on a dry-weight basis with percent moisture and percent lipid as ancillary data
- Use techniques such as geochemical normalization (e.g., aluminum/metal ratios) and chemical fingerprinting to better understand chemical distributions and potential sources

2.6.2 Physico-Chemical Characterization

Sediment investigations usually involve bulk chemical analysis of sediment samples for site-related COPCs in conjunction with biological evaluations. Other physical and chemical data also are needed to support the evaluation of COPC bioavailability. Bioavailability is influenced by a variety of factors associated with organism characteristics (e.g. size and feeding behavior) and sediment characteristics (e.g. TOC content and redox potential). Detailed discussions of bioavailability can be found in Power and Chapman (1992) and the Navy's bioavailability guide (Battelle and Exponent, 2001; http://web.ead.anl.gov/ecorisk/methtool/dsp_bioavail.cfm).

The following physical and chemical parameters also should be measured as appropriate to evaluate the form and behavior of site COPCs and support the interpretation of biological test data:

- Sediment grain-size distribution;
- Sediment TOC content;
- AVS/SEM;
- Porewater pH;
- Porewater salinity (marine/estuarine sites) or alkalinity (freshwater sites); and,
- Porewater ammonia and sulfide concentration.

Descriptions of these parameters, the relevance of each, and associated testing methods are described in Appendix G of *Methods for Collection, Storage and Manipulation of Sediments for Chemical and Toxicological Analyses: Technical Manual* (U.S. EPA, 2001; http://www.epa.gov/waterscience/cs/collectionmanual.pdf) and summarized in Table 2-2. The physico-chemical parameters to be characterized will depend upon the CSM and DQOs for the RI sample collection effort.

Grain-size distribution and TOC data are extremely important in a sediment site investigation because they influence COPC distribution, affect contaminant bioavailability, influence benthic community structure, and introduce factors that may confound toxicity test results. Grain-size analysis defines the frequency distribution of the size ranges of particles that make up site sediment (Plumb, 1981). Contaminants tend to adsorb to finer-grained sediment particles (Power and Chapman, 1992; U.S. EPA, 2001). Sediment grain-size data also are extremely important for the evaluation of sediment dynamics, with coarser-grained material typically associated with high-energy environments (e.g., beaches and channels) and finer-grained material found in quiescent basins and depositional environments. The four major size fractions (gravel, sand, silt, and clay) are the broadest categories that are useful in reporting the size distribution of particles in sediment samples, although a larger number of size classifications is preferred for the evaluation of sediment dynamics. TOC content is a measure of the total amount of oxidizable organic material in a sediment sample. TOC is important because many contaminants are strongly bound to dissolved, colloidal, and particulate organic matter.

AVS/SEM can be measured to determine the fraction of metals that are bound to sulfides (Di Toro et al., 1990; NOAA, 1995). Although universally accepted guidance on the interpretation of AVS measurements is not yet available, these measurements can provide information on the potential bioavailability of metals. The most important points regarding physico-chemical characterization are summarized in Highlight 2-6.

Highlight 2-6. Physico-Chemical Characterization Summary

- Measure sediment grain size and TOC content to evaluate potential COPC distribution and bioavailability, identify depositional environments, infer site hydrodynamics, and support interpretation of bioassay test results and benthic community analysis
- Measure AVS/SEM to evaluate potential bioavailability of sediment metals
- Measure ammonia and sulfide in porewater and/or overlying water to address potential confounding factors in toxicity tests

2.6.3 Collection of FS-Related Data

During the site characterization phase of an RI/FS, the RPM should consider the adequacy of the existing site data to support the FS and identify any further data needs. Generally, a cost savings to the project can be achieved by collecting FS-related data as part of the RI, thereby reducing the mobilization requirements and streamlining the FS process. Some examples of FS-related data that can be collected during the RI phase of a study are as follows:

- Grain-size distribution and moisture content data to predict behavior of material (these data typically collected as part of the site characterization).

- Hazardous waste characterization (e.g., Toxicity Characteristic Leaching Procedure [TCLP] analysis) to support evaluation of treatment and disposal options;

- Dewatering characteristics to identify most appropriate pretreatment methods;
- Engineering properties (e.g., strength, compressibility) to evaluate capping and reuse options; and,
- Sediment dynamics data to support evaluation of in situ management options, including geologic description of sediment cores.

The RPM should have a reasonably high degree of confidence about the need for remediation in a particular area prior to conducting FS-related data collection or treatability studies. The relevance of the parameters identified above and applicable testing methods are summarized in Table 2-2. These data will allow the development of more realistic remedial alternatives and more accurate cost estimates in the FS.

2.7 OVERVIEW OF STUDY DESIGN AND SAMPLE COLLECTION METHODS

This section provides an overview of study design and sediment sample collection methods and equipment. The sampling plan for a sediment investigation should address the data needs for all aspects of the RI/FS (site characterization, risk assessment, and the evaluation of remedial alternatives) to the greatest degree possible in order to minimize mobilization costs and facilitate development of a focused, well-coordinated study. The U.S. EPA document *Methods for Collection, Storage and Manipulation of Sediments for Chemical and Toxicological Analyses: Technical Manual* (U.S. EPA, 2001; http://www.epa.gov/waterscience/cs/collectionmanual.pdf) provides detailed guidelines for development of DQOs, appropriate sample design (e.g., random or targeted), measurement quality objectives, and all aspects of the field investigation (e.g., vessel positioning; sediment sample collection, preservation, transport, and processing; collection of porewater samples; quality assurance/quality control). Additional information on sediment study sample design can be found in "An Introduction to Environmental Sampling Planning" (Kurtz, 2000; http://meso.spawar.navy.mil/Docs/MESO-00-A003-6.pdf).

Collection of aquatic samples generally is divided into five tasks: mobilization, navigation, sediment sampling, sample processing and demobilization. References for field methods and quality control are provided in Table 2-3. Field activities for sediment investigations almost always require a vessel, are often logistically complex, and may require other specialized equipment. Most vessel studies require a differential global positioning system (DGPS) (usually accurate to ±2 m) to position and navigate the survey vessel, and identify station locations. Surface sediment samples are usually collected with a grab sampler, such as a Van Veen, a box-corer, or a Ponar grab (see Figure 2-3). The grab sampler should be constructed of stainless steel and may be coated with Halar® or Teflon™ to reduce potential cross-contamination in the field. Because the quantity of sediment required for tests usually exceeds the volume of the sampler, multiple grabs should be taken at each station unless a modified (e.g., dual Van Veen) grab is used. Surface grabs should be designed to sample the sediment depth interval of interest, and care should be taken to prevent the loss of fine surface sediments. Usually, the biologically active zone is targeted, which is generally the top 10-15 cm of the sediment surface (ERDC, 2001).

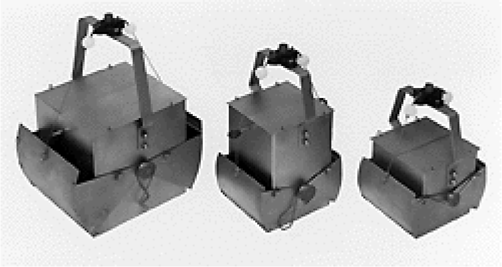
Table 2-3. Information on Aquatic Sample Collection, Field Quality Control, and Equipment

Agency or Organization	Reference and Web Site	Applicable Environment	Topics Covered
U.S. EPA	Methods for Collection, Storage and Manipulation of Sediments for Chemical and Toxicological Analysis: Technical Manual. October 2001. (EPA-823-B-01-002) http://www.epa.gov/waterscience/cs/collectionmanual.pdf	Any water body	Sediment monitoring and assessment plans; sediment sample collection; field sample processing, transport, and storage; sediment manipulations; collection of porewater samples; quality assurance/quality control
State of Washington – Dept. of Ecology	Recommended Protocols for Measuring Selected Environmental Variables in Puget Sound. January 1996. http://www.wa.gov/puget_sound/Publications/protocols/protocol.html	Marine reference (applicable to any water body)	Sediment and water sampling; fish and benthic invertebrate collection; vessel positioning; field quality control
State of Wisconsin – Dept. of Natural Resources	Field Procedures Manual, Sediment Sampling Guidelines, Version IV. September 1998. http://www.dnr.state.wi.us/org/water/wm/wqs/sediment/sampling/table.htm	Freshwater	Sediment sampling, equipment, safety and field quality control
Government of British Columbia, Resources Inventory Committee	Lake and Stream Bottom Sampling Manual, Document # 7680000550. 1997 http://srmwww.gov.bc.ca/risc/pubs/aquatic/lake-stream/index.htm	Rivers and stream bottom sampling	Sediment sampling, equipment, and quality control
San Francisco Estuary Institute (SFEI)	Field Sampling Manual for the Regional Monitoring Program for Trace Substances, Version 1. January 1999. http://www.sfei.org/rmp/documentation/fom/fom_1.html	Estuarine and marine reference (applicable to any water body)	Sediment, porewater, water column, benthic invertebrate sampling; equipment type and use; quality control; remote sensing; vessel safety

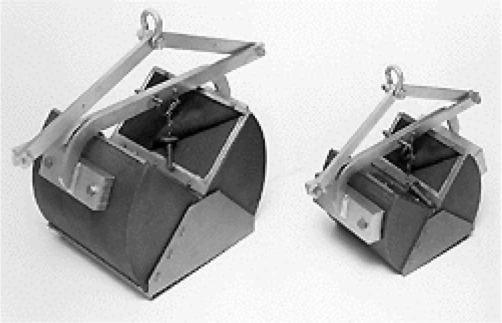
To minimize the cost of multiple field efforts, concurrent subsurface sediment data should be collected for defining the vertical extent of contamination (see Section 2.5). Subsurface sediment cores should be collected using stainless steel core tubes with inert liners (e.g., butyrate). Photographs and descriptions of sampling equipment are shown in Figure 2-3 (surface samplers) and Figure 2-4 (coring devices).

2.8 SUMMARY

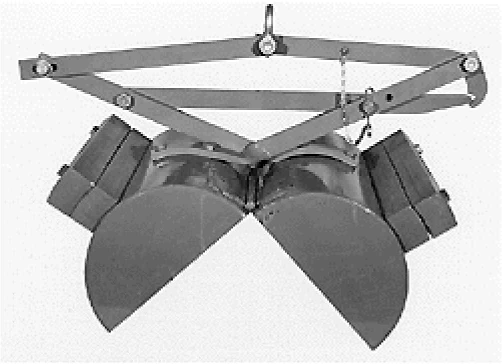
Sediment site characterization, including development of the CSM, assessment of contaminant fate and transport, definition of the nature and extent of contamination, and collection of relevant physical and chemical data, is conducted during the initial phases of the RI/FS and continues in conjunction with the ecological and human health risk assessments. Data collection for all aspects of the RI/FS should be coordinated to the greatest degree possible to maximize the efficient use of resources and ensure a focused, well-defined investigation.



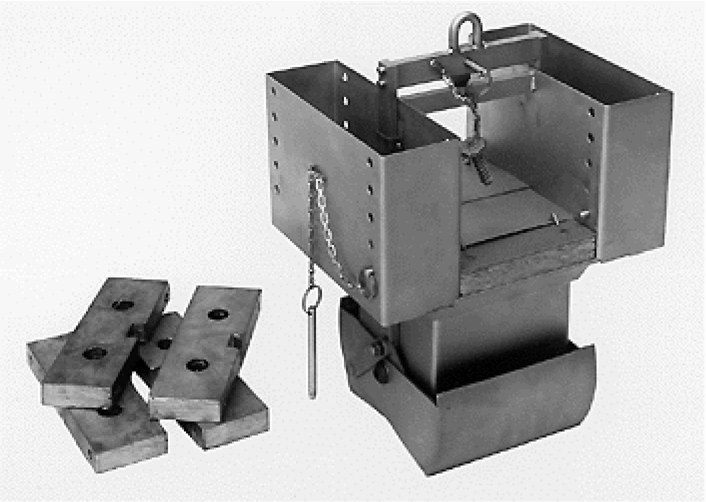
Ekman Grab: Designed to sample in soft, finely divided littoral bottoms free from vegetation and intermixes of sand, stones, and other coarse debris. Best in finely divided muck, mud, ooze, submerged marl, or fine, peaty materials. Lightweight samplers designed for use from smaller boats (standard size: 200 cm by 200 cm wide, 150 cm deep).



Ponar Grab: Designed to sample firm or hard clay bottoms free from stones and other coarse debris. The standard Ponar grab is heavy and should be used from a winch. Smaller versions are light enough for sampling by hand. Removable top screens allow subsampling from the closed scoops. Top screens also have rubber flaps to prevent sample washout during retrieval. Similar in design to rigid arm Van Veen grab.



Peterson Grab: Widely used in fresh water sampling hard bottoms, such as sand, pebbles, clay, or clay compounds. The Peterson grab scoops are hinged at the top, like a clamshell. Subsampling cannot be performed from closed grab.

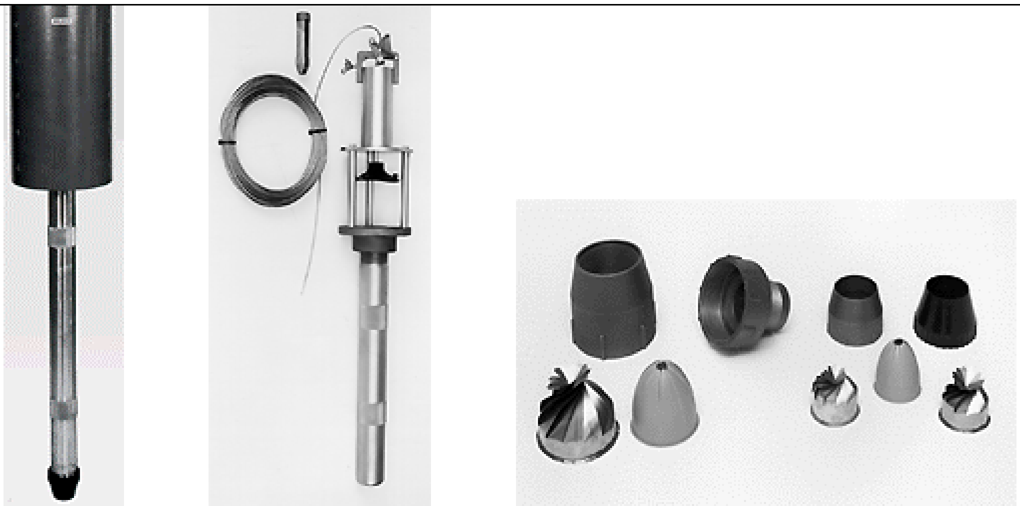


Small Box Corer: Designed to take large sample in bottoms from soft ooze to hard clay free from stones and other coarse debris (photo above of WILDCO® sampler). Winch is required for operation. Removable top screens allow subsampling when scoops are closed. Removable top screens have rubber flaps to prevent sample washout during retrieval. Grab volumes vary with size of device; the most common size is 5 L.

Figure 2-3. Examples of Sediment Surface Grab Samplers

Dual 0.1 m^2 Van Veen Grab: Designed to two side-by-side 0.1 m^2 surface samples, allowing collection of co-located chemistry & biology samples or additional sample for toxicity tests. Samples from soft to hard bottom free from stones and other coarse debris (left photo shows grab before deployment; right photo shows two sediment samples). Winch is required for operation. Removable top screens have rubber flaps to prevent sample washout during retrieval.

Figure 2-3. Examples of Sediment Surface Grab Samplers (continued)



Small Gravity Corer: Sand and silt substrates can be sampled to up to 1.5 m with small gravity corer devices. Corers can be deployed from davits or A-frames and require winches for retrieval. Fins often are used to stabilize descent. Penetration depths depend on substrate and core tube size. Core tube diameters vary from 2 cm to 6 cm, are generally metallic (stainless steel), and can accommodate inert liners (e.g., butyrate, polycarbonate, Teflon™). Corers require check valves, cutting heads and “core catchers” to maintain core during retrieval.

Large Gravity Corer: Sand and silt substrates can be sampled to up to 3 m with large gravity corer devices. Corers must be deployed from large davits or A-frames and require winches for retrieval. Fins may be used to stabilize descent. Penetration depths depend on substrate and core tube size. Core tube diameters can exceed 10 cm, are generally metallic (iron or stainless steel), and can accommodate inert liners (e.g., butyrate, polycarbonate, Teflon™). Corers require check valves, cutting heads and “core catchers” to maintain core during retrieval.

Figure 2-4. Examples of Sediment Coring Devices

Vibracore: Vibracoring is a technique for collecting core samples in unconsolidated sediments by driving a tube with a vibrating device, generally referred to as "vibrohead." The energy imparted by the vibrohead to the coretube assists its vertical penetration by displacing the sediment particles and overcoming the two main forces opposed to its progress, namely frontal resistance and wall friction. This technique is naturally the most efficient in water-saturated sediments by raising the pore-pressure along the wall of the coretube and generating a thin layer of liquefaction. Core lengths retrieved can exceed 10 m with diameters of 6 cm. Core tubes are generally metallic (iron or stainless steel) and can accommodate inert liners (e.g., butyrate, polycarbonate, Teflon™). Corers require check valves, cutting heads and “core catchers” to maintain core during retrieval.

Figure 2-4. Examples of Sediment Coring Devices (continued)

3.0 ECOLOGICAL AND HUMAN HEALTH RISK ASSESSMENT FOR SEDIMENT SITES

The risk assessment process at Navy sediment sites is conducted in accordance with the Navy and U.S. EPA guidance identified in Highlight 1-1. The Navy and U.S. EPA guidance should be reviewed and generally understood by the RPM before a risk assessment is initiated at any site. In addition, regular communication and involvement with the applicable regulatory agencies will facilitate the process of evaluating risks. This is true with all risk assessments; however, it is especially important at sediment sites due to the multitude of additional state, national, and international regulatory criteria applicable to sediments, and the lack of promulgated sediment quality criteria (see Section 1.4).

Briefly, the Navy uses a three-tiered approach, similar to the U.S. EPA's risk assessment guidance for ecological and human health evaluations (U.S. EPA, 1989 and 1998c; see Highlight 1-1 for Navy Guidance). The U.S. EPA guidance should be carefully reviewed for identification of critical scientific management decision points (SMDPs) throughout the process. The first tier consists of a very conservative, screening-level risk assessment intended to eliminate chemicals and areas that do not pose an unacceptable risk. This step focuses the investigation on those chemicals and areas that may pose an unacceptable risk.

Although the overall approach used to evaluate risk at sediment sites is consistent with the Navy and U.S. EPA guidance, technical issues specific to the evaluation of sediments (e.g., physical and chemical properties of sediment) should be considered when performing a risk assessment for aquatic sites. Many of these issues are discussed in Section 2.0. The purpose of this section is to summarize these issues and provide guidance for incorporating them within the context of the Navy's tiered framework. Section 3.1 focuses on the ERA, and Section 3.2 addresses issues specific to the HHRA.

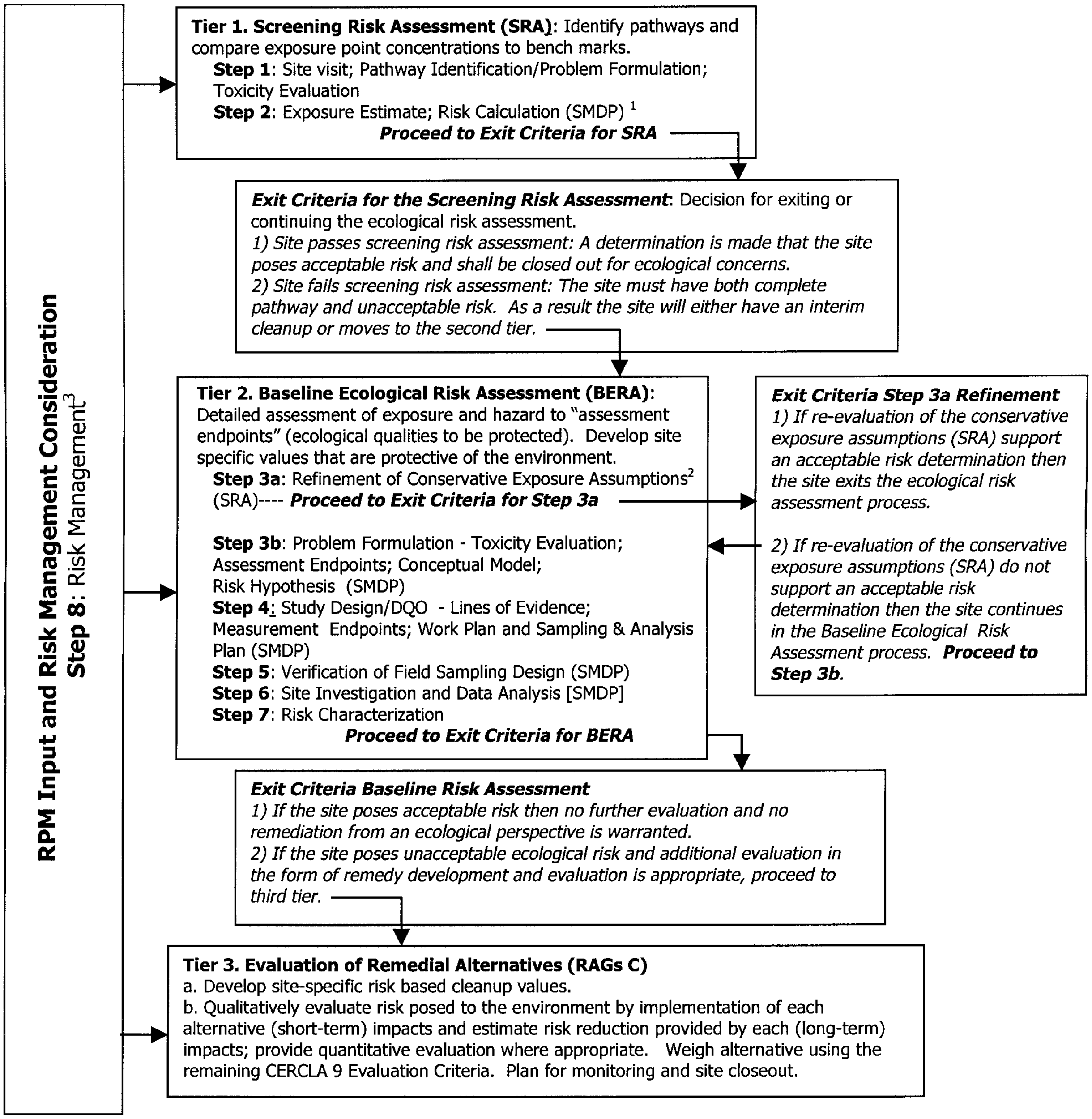
As the HHRA or ERA process is initiated, it is important that the RPM evaluate the project team and confirm that the appropriate expertise is represented. Section 2.1.1 discusses the project team required for sediment site assessments. In addition, regular discussions with relevant regulatory agencies throughout each stage of the investigation are critical to facilitate the process.

3.1 ECOLOGICAL RISK ASSESSMENT

The Navy's ERA approach consists of three tiers (Figure 3-1):

- Tier 1: Screening Risk Assessment (SRA)
- Tier 2: Baseline Ecological Risk Assessment (BERA)
- Tier 3: Evaluation of Remedial Alternatives.

Tier 1 focuses on the development of the fundamental framework of the assessment (i.e., the preliminary problem formulation and conceptual site model) and includes preparing a conservative, screening-level evaluation of potential risks. Tier 2 focuses on the refinement of the assessment to more closely reflect actual site conditions. Tier 3 focuses on the evaluation of remedial alternatives (see Section 4.0). Appendix C presents an example ERA for a hypothetical site.



Notes: 1) See EPA's 8 Step ERA Process for requirements for each Scientific Management Decision Point (SMDP).
2) Refinement includes but is not limited to background, bioavailability, detection frequency, etc.
3) Risk Management is incorporated throughout the tiered approach.

Figure 3-1. Navy Ecological Risk Assessment Tiered Approach

3.1.1 Tier 1: Ecological Screening Risk Assessment

The Navy's Tier 1 SRA consists of Steps 1 and 2 of the eight-step U.S. EPA Superfund ERA process (see Figure 3-1). The purpose of the SRA is to screen out chemicals and areas that do not pose an unacceptable risk as defined for the site through discussion with the regulatory agencies and stakeholders. It is based on the existing data and information gathered during the Preliminary

Assessment/Site Inspection (PA/SI) and initial stages of the RI/FS, as described in Section 2.1.2, and should not require additional or new data collection. If limited additional data are required, rapid screening methods may be considered (see Appendix A) although it is important to note that data from these methods may not meet data quality objectives for an ERA, and should be used for initial screening purposes only. Because the SRA is intended as a screen, conservative assumptions regarding site conditions, chemical bioavailability, and exposure parameters are used for this portion of the evaluation. For example, when evaluating potential affects to the benthic community, toxicity data associated with a species known to be very sensitive to the COPC might be evaluated, or minimum sediment quality guidelines might be considered. Other screening-level assumptions include the use of a maximum chemical concentration to represent the site, the assumption of 100 percent bioavailability of all chemicals present, or the assumption of a site use factor of 1. Through the use of these conservative assumptions that are more likely to overestimate rather than underestimate potential risks, the RPM can have high confidence that a chemical will be eliminated from consideration as a COPC only if it does not pose an unacceptable risk.

3.1.1.1 Sediment Site Characterization

The first step of the SRA involves evaluating the data and site-specific information collected as part of the site characterization (Section 2.0). For sediment investigations, information regarding the physical and environmental conditions at the site should be evaluated to identify relevant aquatic habitats and possible site uses by terrestrial ecological receptors. For example, the composition and diversity of the aquatic community may be highly influenced by a variety of site-specific features including:

- Type of water body (e.g., freshwater or marine, river, estuary, bay);
- Presence of tides, waves, or currents;
- Potential groundwater-surface water interactions (GSI);
- Bathymetry and sediment substrate;
- Presence or absence of exposed sediments;
- Shoreline features (e.g., bulkheads, emergent vegetation, beaches, terrestrial habitats, etc.);
- Potential presence of endangered species; and,
- Extent and nature of surrounding land use (e.g., residential, undeveloped, industrial).

All of these physical features can influence the use of the site by ecological communities and may help to identify complete exposure pathways. In addition, information collected on the nature and extent of contamination should be reviewed to identify the areas of concern and chemicals to be evaluated. Ecological evaluations typically focus on surface sediments (i.e., the biologically active zone, usually defined as approximately the top 10 cm) because benthic communities are not significantly exposed to sediments at depth (USACE, 2001). However, if dredging activities are planned, sediments at depth also should be considered to evaluate potential future risks from sediments that will be exposed.

3.1.1.2 Problem Formulation

The SRA includes the development of a preliminary problem formulation which identifies: (1) sediment COPCs from site knowledge and existing data; (2) ecological receptors potentially at risk; and (3) complete exposure pathways. Based on the results of the preliminary problem formulation, the preliminary CSM developed during the site characterization (Figure 2-1) is updated to reflect the complete exposure pathways and receptors of concern.

3.1.1.2.1 COPC Identification

The identification of preliminary sediment COPCs should be conducted during the site characterization process and presented to the appropriate regulators as soon as possible. Reaching agreement on the key COPCs early in the process will facilitate later discussions. Preliminary COPCs should be identified early in the process as being from Navy sources, non-Navy sources, or a combination (i.e., "mixed"), and then refined as more information becomes available. Common Navy sediment COPCs and potential sources are shown in Table 2-1. An expanded list of multiple-source sediment contaminants also is provided in the U.S. EPA/USACE (1998) *Inland Testing Manual* at http://www.epa.gov/waterscience/itm/ITM/.

For the purpose of the SRA, COPCs should be identified by comparing maximum surface sediment contaminants to appropriately conservative sediment quality benchmark values (see Table 3-1). Due in part to the complexities associated with predicting the toxicity of sediments, no national sediment quality criteria currently are promulgated for use in an ERA. However, sediment quality benchmark values have been developed in an attempt to define concentrations in sediment that are unlikely to result in adverse affects to aquatic organisms. Table 3-1 presents a summary of some of the most commonly used sediment quality benchmarks http://web.ead.anl.gov/ecorisk/methtool/dsp_bench.cfm. The majority of the benchmarks listed in Table 3-1 were developed for individual chemicals. However, U.S. EPA also has recently begun to develop sediment guidelines for chemical mixtures to more accurately reflect actual conditions (see Highlight 3-1).

Each of these benchmark values was derived using a different methodology for linking concentrations to observed effects. Some benchmarks are based on community-level analyses, or on combinations of lethal and sublethal effects such as the apparent effects threshold values (AET) or effects range–low (ER-L) and effects range–median (ER-M) values (Long et al., 1995). Some benchmarks focus specifically on freshwater or marine organisms. In addition, although some values are based on bulk sediment chemistry results, others are normalized to organic carbon or other chemical or physical parameters. The different methods result in widely varying predictions of "safe" sediment levels; therefore, it is important to evaluate the applicability of both the assumptions used and recommended applications of any benchmark values prior to applying them at a specific site. Many regulatory agencies recommend the use of a particular approach, so consultation with the appropriate regulators prior to selecting the benchmarks for use at the site is important. A case study demonstrating one method of selecting sediment screening values for the Philadelphia Naval Complex can be found on the NAVFAC ERA Web site at http://web.ead.anl.gov/ecorisk/case/.

Chemicals without relevant sediment quality benchmarks values or sufficient data to develop applicable background levels also should be retained as COPCs. In addition, most sediment quality benchmarks do not address the potential for bioaccumulation. Bioaccumulation represents the first step in the movement of sediment-associated contaminants into the food web (Lee, 1992). Therefore,

Table 3-1. Examples of Benchmark Values Used in Tier 1 Screening Process

Benchmark	Basis	Source	Website
Sediment			
ER-L/ER-M	Significant toxicity to benthic infauna	Long et al., 1995	http://response.restoration.noaa.gov/cpr/sediment/SQGs.html
TEC/PEC	Consensus based guidelines	MacDonald et al., 2000a	NA
TEL/PEL	Significant toxicity to benthic infauna	Florida State Department of Environmental Protection, 1994	http://www.dep.state.fl.us/water/monitoring/docs/seds/vol1/volume1.pdf
PAH ESGs	Significant toxicity to benthic infauna from PAH mixtures	U.S. EPA, 2000e	NA
Apparent Effects Threshold (AET)	Significant toxicity to benthic infauna	Washington Department of Ecology, 1995	http://www.ecy.wa.gov/programs/tcp/smu/sed_chem.htm
SQG-Q	Significant toxicity to benthic infauna from chemical mixtures	Long et al., 1998; MacDonald et al., 2000b	NA
Superfund EcoTox (ET)	Predicted toxicity to benthic infauna based on equilibrium partitioning	U.S. EPA, 1999	http://www.epa.gov/ecotox/
Tissue			
Effects-based tissue residues for selected biological receptors	Tissue residues associated with specified effects	USACE/U.S. EPA, 2001 ERED Database McCarty and Mackay, 1993	http://el.erdc.usace.army.mil/ered/

ER-L = Effects Range-Low
ER-M = Effects Range-Median
NA = Not Applicable
PAH ESGs = PAH Equilibrium Partitioning Sediment Guidelines
PEC = Probable Effect Concentration
PEL = Probable Effect Level
SQA = Sediment Quality Guideline
TEC = Threshold Effect Concentration
TEL = Toxic Effect Level
TRV = Toxicity Reference Value
USACE = Army Corps of Engineers

additional consideration should be given to the potential for bioaccumulation, especially for site COPCs that are not necessarily present in sediments at levels that are toxic to aquatic species but that have the propensity to bioaccumulate in higher trophic levels, potentially posing a hazard to piscivorous species through food web exposures. U.S. EPA has identified a general list of bioaccumulative compounds of potential concern, which are discussed in more detail in Chapter 4 of *Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment* (U.S. EPA, 2000a; http://www.epa.gov/waterscience/cs/biotesting/index.html). These compounds should be retained as COPCs until sufficient information is available to demonstrate that they are not bioavailable at the site.

Highlight 3-1. Sediment Benchmark Values for Chemical Mixtures

In general, most sediment benchmarks evaluate effects associated with individual chemicals. In reality, most sediments contain a mixture of contaminants; therefore, this approach does not consider potential synergestic effects. To address this issue, U.S. EPA currently is investigating methodologies for deriving benchmark values for chemical mixtures. However, in the interim, recent research has suggested that sediment toxicity may be predicted through the use of a sediment effects ratio described as a Sediment Quality Guideline Quotient (i.e., SQG-Q) (MacDonald et al., 2000b; Ingersoll et al., 2000; Long et al., 1998). The SQG-Q is derived by a three-step process developed by Long et al. (1998). In the first step, the concentration of each chemical in a given sample is divided by its respective sediment quality criteria. The resulting ratio is defined as a SQG quotient or SQG-Q. The SQG-Qs for each chemical are then summed and divided by the number of individual chemicals evaluated to derive a mean SQG-Q for each sample. Preliminary data indicate that the mean SQG-Q may facilitate comparisons between areas and sampling stations, particularly in situations where differing numbers of chemicals have been evaluated. For example, based on a sample size of 175, MacDonald et al. (2000b) found that the incidence of toxicity in freshwater sediments could be predicted in up to 94.4 percent of sediments considered through use of the mean SQG-Q. This approach is relatively new and has not yet been subjected to rigorous field testing; therefore, its application should be discussed with the appropriate regulators.

3.1.1.2.2 Identification of Ecological Receptors

Based on an evaluation of the available habitats identified during the site characterization, relevant ecological receptors of concern are selected for investigation. It is impossible to evaluate all species that might be exposed to COPCs at a site; therefore, a few representative species are identified. Typically, the species selected are chosen to represent the key or primary feeding guilds at the site, or species of special concern (e.g., endangered species). For marine facilities, possible exposures to marine mammals (e.g., sea otters, seals etc.) and pelagic species should be considered. For the purpose of the SRA, the selection of receptors focuses primarily on identifying the key trophic levels that will be evaluated, rather than on selecting the specific species exposed. Figure 1-1 provides a generic aquatic food web, depicting the wide variety of potential receptor classes that may be exposed. A summary of types of receptors usually evaluated is provided below.

Benthic Invertebrates

The benthic invertebrate community includes a wide array of organisms living in close association with the sediments. Some of these organisms burrow into sediments, whereas others live at the sediment water interface or in intertidal areas (Levinton, 1982). Due to their relatively direct exposure to surface sediments, and their position at the base of most aquatic food webs, benthic invertebrates are a key indicator species when evaluating the potential effects of sediment-associated contaminants (Diaz, 1992; La Point and Fairchild, 1992; Ankley, 1997).

Fish Community

La Point and Fairchild (1992) recommend fish for assessment because of their societal value and familiarity as well as their role as integrators of toxicity at lower trophic levels. However, limitations are associated with using these receptors to evaluate site-specific sediment quality due to their relative mobility. The fish community is represented by a diverse assortment of species, some of which live in close association with the sediments (i.e., demersal species) and others which reside primarily in the water column (i.e., pelagic species) (Levinton, 1982). In addition, some species are herbivorous, consuming primarily plant material and detritus, whereas other species are predatory, consuming invertebrates and even smaller fish. All fish are exposed to COPCs to some degree through direct uptake from sediment and the water column, as well as through dietary intakes (Mac and Schmitt, 1992). As a result, COPC exposure and uptake is highly influenced by the life history, foraging range, and feeding regimen of the species evaluated.

Birds and Mammals (Terrestrial and Marine)

Wildlife species may be exposed to contaminated sediments in a variety of ways including through incidental ingestion of sediment and surface water and by trophic transfer through the consumption of prey items (e.g., fish, invertebrates) (U.S. EPA, 1993a). The relative importance of each potential exposure pathway is dependent on the chemical and physical properties of the COPC present in the sediment which control uptake into aquatic organisms (Fordham and Reagan, 1991). For example, piscivorous (i.e., fish-eating) wildlife (e.g., mink, seals, herons) are exposed to persistent, hydrophobic organics (e.g., PCBs) primarily through the consumption of prey (i.e., fish), with incidental ingestion of sediment and surface water playing a smaller role.

3.1.1.2.3 Identification of Complete Exposure Pathways

A complete exposure pathway is one in which the COPC(s) can be expected to travel from the source to a receptor that can be affected. It may include direct exposure through ingestion or dermal contact with sediment, or indirect exposure through trophic transfer. For the purpose of the Tier 1 SRA, potential pathways generally are assumed to be complete provided that the identified COPC can be associated with Navy activities, the site characterization indicates that relevant ecological receptors are likely to be present, and the mechanisms for exposure exist. This assumption is refined throughout the investigation as more information becomes available. However, if a particular exposure pathway can be demonstrated to be incomplete, it should be eliminated from the assessment and the supporting rationale should be documented. For example, risks to benthic-feeding birds should not be evaluated if it can be demonstrated that the appropriate habitat conditions (e.g., presence of mudflats or shallow intertidal areas) for foraging do not exist at the site.

3.1.1.2.4 Conceptual Site Model

As previously discussed, the preliminary CSM developed during the initial site characterization should be updated based on the information collected during the preliminary Problem Formulation to create an ecological CSM (Figure 3-2). This ecological CSM should reflect the complete exposure pathways identified as well as the ecological receptor classes to be evaluated. A distinction also is made between minor and major exposure pathways. For example, as indicated in this figure, sediment associated chemicals may partition between sediment and surface water, creating a dynamic process that is always in flux. Depending on the physical and chemical conditions at the site, as well as their feeding regimen and other behaviors, ecological organisms can be exposed to COPC associated with either of these media; however, concentrations of many sediment-associated COPC are low

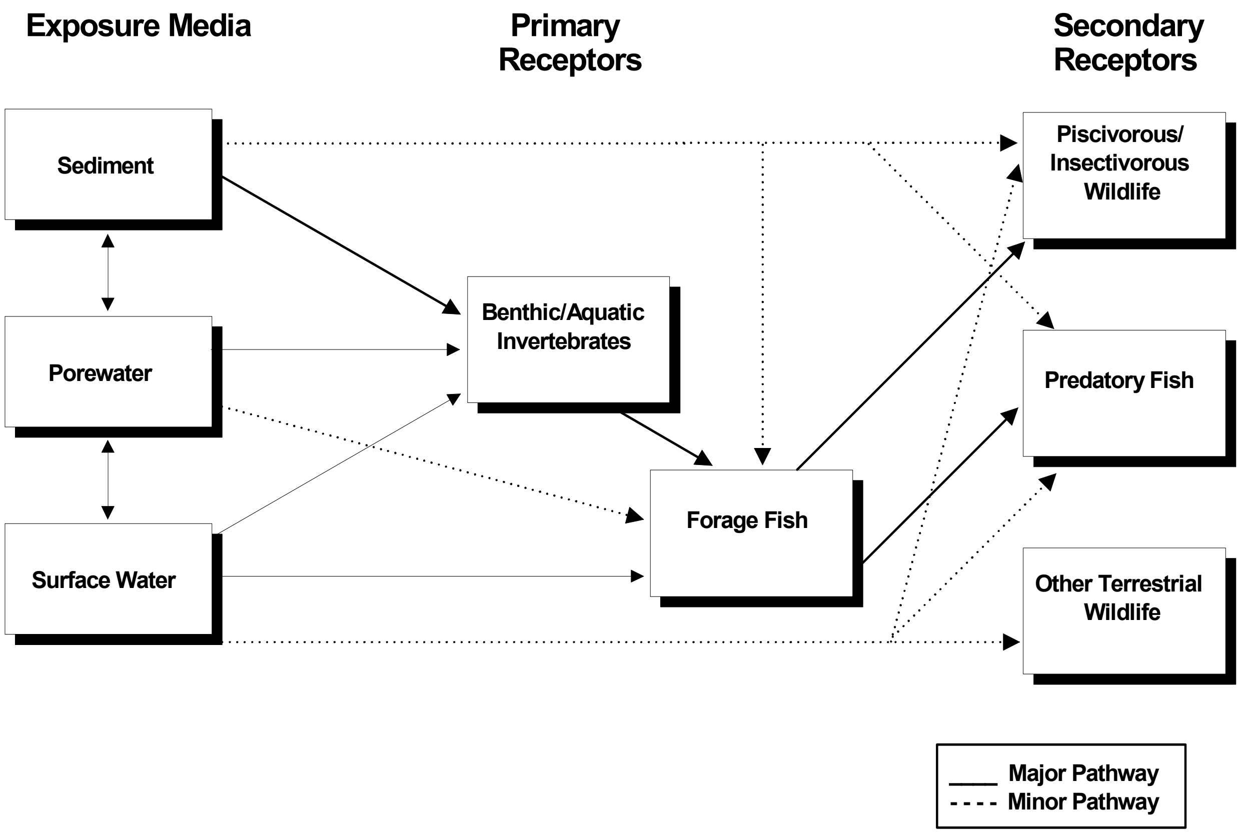


Figure 3-2. Simplified Ecological Conceptual Site Model for Sediment

in surface water (see Section 1.1). As a result, direct exposure to sediment-associated contaminants via surface water is likely to be a minor pathway, accounting for a smaller portion of the overall exposure. This assumption should be adjusted if the site characterization indicates the likelihood of elevated concentrations of COPCs in surface water, based on either actual measurements or physico-chemical conditions.

This preliminary ecological CSM represents the basic framework on which the remainder of the evaluation will be based; therefore, discussions with the appropriate regulators are critical at this point in the assessment.

3.1.1.3 Preliminary Exposure Estimate and Risk Calculations

The second step of the SRA includes a screening-level exposure estimate and an initial screening risk calculation, equivalent to Step 2 of the U.S. EPA eight-step process (http://www.epa.gov/ncea/ecorsk.htm). In this initial phase of the investigation, site-specific information often is limited to bulk sediment chemistry data. Therefore, for the purpose of the SRA, exposure estimates for aquatic organisms usually are evaluated based on conservative estimates of COPC concentrations (i.e., maximum) in surface sediment. In addition, if bioaccumulative chemicals are present, concentrations of COPCs in tissues of exposed organisms also may be evaluated using site-specific tissue data. If such data are not available, tissue concentrations may be estimated using a variety of bioaccumulation models (Highlight 3-2). To evaluate exposures to upper trophic level receptors (i.e.,

Highlight 3-2. Bioaccumulation Models

A variety of bioaccumulation models exist for the purpose of predicting tissue concentrations in aquatic organisms in the absence of measured data from field collections or laboratory bioassays (Lee, 1992). The simplest of these are the bioaccumulation factor (BAF), defined as the ratio between the COPC concentration in the organism and sediment, and the equilibrium partitioning bioaccumulation model, also referred to as the biota/sediment accumulation factor (BSAF). The BSAF is the ratio of the lipid normalized tissue concentration of COPC and the organic carbon normalized sediment concentration. Multiplying these factors by the chemical concentration in sediment (using dry weight for the BAF and TOC normalized for the BSAF) results in an estimated tissue concentration. U.S. EPA (2000a) includes BSAF information on 11 metals, 1 chlorinated phenol, 10 PAHs, 13 chlorinated pesticides, selected dioxins, furans, and both Aroclor and congener forms of PCBs. In addition, a fairly comprehensive list of BSAF and Theoretical Bioaccumulation Factors (TBF) values for several trophic levels of fish can be found at http://www.epa.gov/waterscience/cs/vol1/appdx_d.pdf. Accumulation factors appropriate for screening studies also are available for selected aquatic species at http://www.epa.gov/ecotox/. For full details on the derivation and application of bioaccumulation factors, consult the full document *The Incidence and Severity of Sediment Contamination in Surface Waters of the United States* (EPA/823/R-97/006; http://www.epa.gov/OST/cs/congress.html). A comprehensive database of BSAFs is maintained by USACE at http://el.erdc.usace.army.mil/bsaf/.

Due to their simplicity, BAFs and BSAFs provide a quick and easy means of estimating tissue concentrations that may be associated with exposures to contaminated sediment. However, prior to applying BAFs or BSAFs, the limitations of these approaches should be considered. The accuracy of BAFs is limited by variations in sediment type and species evaluated. Similarly, BSAFs rely on the assumption that the system evaluated is at steady state and may over or underestimate actual accumulation if those conditions are not met. For some evaluations, the uncertainty associated with these approaches may be too great. Under those circumstances, kinetic process and bioenergetic-based models will provide a more accurate estimation of uptake (Lee, 1992). For a more detailed discussion of the various types of models available, the strengths and limitations associated with each one, and recommendations for selecting a model suitable to your site, see Lee (1992).

wildlife), simplified dose calculations may be performed, using conservative assumptions regarding contaminant bioavailability (e.g., 100 percent bioavailability) and uptake (e.g., assumption of upper bound sediment ingestion rates, or a site use factor of 1).

Preliminary risk estimates for aquatic organisms in Tier 1 typically are limited to comparisons of sediment and tissue concentrations to available sediment and tissue benchmark values (see Table 3-1). Hazard quotients (HQs) for these species may be derived by dividing the sediment or tissue concentration of each COPC associated with the site by its respective benchmark value. Similar to sediment benchmark values, tissue residues associated with adverse effects (e.g., the lethal residue associated with 50% mortality in a population exposed for 96 hours) can be used to predict the tissue concentration at which adverse effects will be observed in a particular aquatic species (USACE/U.S. EPA, 2001; see Table 3-1). Landrum and Meador (2002) discuss the limitations associated with the use of tissue residues.

It is important to note that many of these sediment and tissue benchmark values are "inferred" from various effects databases. In addition, the ecological relevance of the effect recorded may vary substantially from enzyme induction to acute mortality. For example, ER-L and ER-M values are statistically derived based on a database of chemical concentrations measured in sediment samples associated with some degree of toxicity (ranging from behavioral effects to mortality) to benthic infauna; however, there is no established causal relationship between any single contaminant and the measured effect. As a result, benchmark values should be used for screening purposes only; they are not regulatory criteria, site-specific cleanup standards, or remediation goals.

To estimate Tier 1 risks to wildlife species, screening-level dose estimates may be calculated using conservative exposure assumptions to evaluate all pathways considered complete. In the absence of site-specific tissue data, bioaccumulation models can be used to predict tissue concentrations in aquatic organisms (e.g., fish, shellfish, and benthic invertebrates) exposed at the site. For example, biota sediment accumulation factors (BSAFs) were used at the Philadelphia Naval Complex for the Screening-Level Risk Assessment for Reserve Basin Sediment (http://web.ead.anl.gov/ecorisk/case/study3.cfm). Hazard quotients for this pathway are estimated by comparing these dose estimates to toxicity reference values (TRVs) derived for selected wildlife receptors.

Based on the results of the screening risk calculation, a decision is made for exiting or continuing the ERA. If ecological risks based on the conservative screen are acceptable (i.e., typically defined as HQs less than 1), then the site is determined to pose an acceptable risk and is closed out for ecological concerns. If HQs exceed 1, then the site either proceeds to an interim cleanup, or proceeds to the second tier. A decision also may be made to move forward to Tier 2 for only a limited section of the site, or for a reduced number of COPCs.

3.1.2 Tier 2: Baseline Ecological Risk Assessment

Tier 2 represents the BERA, which is the most extensive activity within the ERA process in terms of data collection and analysis, cost, and effort. It is much more site-specific and technically rigorous than Tier 1. The BERA consists of Steps 3 through 7 of the eight-step U.S. EPA Superfund ERA process (see Figure 3-1).

Step 3: Refinement of Conservative Exposure Assumptions (Step 3a) and Revised Problem Formulation (Step 3b)

Step 4: Study Design/DQO

Step 5: Verification of Field Sampling Design

Step 6: Site Investigation and Data Analysis

Step 7: Risk Characterization.

As the first component of the BERA, the COPCs that were retained from Tier 1 are re-evaluated based on a refinement of the conservative exposure assumptions (i.e., Step 3a). The purpose of Step 3a is to identify and eliminate from further consideration those COPCs that were retained only because of the use of very conservative exposure scenarios. If the re-evaluation conducted in Step 3a supports an acceptable risk determination, then the site exits the ERA process; otherwise it proceeds to Step 3b. Sediment-specific issues associated with each of these steps are described below.

3.1.2.1 Step 3a: Refinement of Conservative Exposure Assumptions

In Step 3a, a variety of methods are used to refine the assumptions used in the SRA, depending on the information available. For example, in the SRA, conservative values may be used for exposure parameters such as exposure point concentration (e.g., maximum), area use factors (e.g., site use factor of 1), life stage (e.g., the most sensitive stage), body weight, food ingestion rates, dietary composition, and bioavailability (e.g., assuming 100 percent bioavailability or use of conservative BAFs or BSAFs to estimate uptake). For Step 3a, these conservative estimates are replaced with more realistic, site-specific values, as supported by existing data. Relevant questions to ask before calculating less conservative risk estimates concerning source, stressor, and exposure characteristics are found in U.S. EPA (1998b). Risks are recalculated using these refined exposure parameters, and COPCs with HQs below 1 are eliminated from further consideration. A summary of some of the key refinements for sediment sites is provided below.

Bioavailability

Bioavailability refers to the degree to which a contaminant in sediment is available for uptake by a receptor. As discussed in Section 2.6.2, a number of chemical and physical characteristics of sediment may affect the bioavailability of COPCs, primarily through adsorption or complexation of contaminants into the sediment matrix. These characteristics are not always reflected in application of screening-level sediment benchmark values. For example, many sediment benchmarks are based on bulk sediment chemistry concentrations and, therefore, do not account for effects of TOC. In addition, most sediment benchmarks are based on total metals concentration. There is evidence to suggest that the bioavailability of several metals (e.g., arsenic) may be correlated to the species of metal present in sediment (Neff, 1997). In addition, the presence of AVS may also affect the bioavailability of some metals (Ankley, 1996; Ankley et al., 1996).

For the purpose of Step 3a, existing data should be reviewed to evaluate potential chemical and physical conditions that potentially affect the bioavailability of COPCs. It may be necessary to collect limited additional data (e.g., TOC, grain size, AVS/SEM); however, the evaluation should rely on existing information to the extent possible. The Navy has prepared guidance on the incorporation of bioavailability adjustments for metals into ecological risk assessments (Battelle and Exponent, 2000; http://web.ead.anl.gov/ecorisk/issue/pdf/bioavailNavy.pdf).

Comparison to Background

According to the Navy's *Policy on the Use of Background Chemical Levels* (CNO, 2000; see Highlight 1-1), BERAs should not be conducted on chemicals that are present at levels less than background chemical levels (i.e., anthropogenic or naturally occurring levels). According to this policy, background is defined as follows:

- Naturally occurring chemical levels (nonanthropogenic): ambient concentration of chemicals present in the environment that has not been influenced by human activities (e.g., arsenic).

- Anthropogenic chemical levels (not naturally occurring): concentrations of chemicals that are present in the environment due to human-made, nonsite sources (e.g., application of pesticides, herbicides, lead from automobile exhaust).

For the purpose of the BERA, the preliminary COPC list generated during the Tier 1 SRA is refined to reflect only those chemicals that exceed applicable thresholds and background. The Navy's *Policy on the Use of Background Chemical Levels* (CNO, 2000; see Highlight 1-1) provides guidance on how to bring background issues into the process at Step 3a. This policy emphasizes the need to differentiate background contamination from site releases in the Navy IR programs. It is acknowledged that this approach differs slightly from that recommended by U.S. EPA (U.S. EPA, 2002); however, although chemicals that are present at concentrations below background but above applicable benchmark values should not be included as COPC, they should be discussed in the risk characterization and uncertainty section. Additional guidance on the use of background levels to refine COPC lists is provided in U.S. EPA (2001b; http://www.epa.gov/superfund/programs/risk/ecoup/slera0601.pdf).

A variety of methods may be used to develop estimations of background. For example, background may be based on concentrations of chemicals present at a remote reference site or those representative of ambient levels in a specified region (Giesy and Hoke, 1990). Consultation with the relevant regulatory agencies regarding the appropriate estimation of background is recommended early in the process. One method of determining background is though evaluation of a "reference" area. The selection of a reference area is an important step in the evaluation and should be conducted in conjunction with the agencies. Factors to consider include: (1) similar physical characteristics (e.g., grain size, TOC) to site sediments; (2) similar habitat conditions (e.g., subtidal vs. intertidal or estuarine vs. freshwater); (3) representative of regional ambient conditions (i.e., not influenced by point sources of contamination).

Navy guidance (*Navy Guidance for Environmental Background Analysis, Volume I: Soils,* NFESC User's Guide UG-2049-ENV, April 2002) is available to determine background exceedances through the use of statistical comparisons of complete distributions of data rather than relying on earlier practices of point comparisons to statistically derived upper confidence limits (UCLs). Although this guidance was prepared specifically for soil sites, it is applicable to sediment sites with some modifications. An effort currently is underway to complete *Navy Guidance for Environmental Background Analysis, Volume II: Sediment*. This document also provides guidance on the use of geochemical methods to compare to background and to identify sources (see Section 2.6.1.2). These methods can be used to eliminate COPCs that either are shown to occur naturally at the site (e.g., elevated nickel in San Francisco Bay sediments) or are decisively linked to non-Navy sources. Additionally, because many sediment sites are impacted by both Navy and non-Navy contaminants, these methods can be used to perform source allocation of certain COPCs (e.g., PAH, PCBs, metals). A discussion on the application of some of the more common geochemical methods is provided in Appendix B.

Evaluation of Detection Frequency and Analytical Methods

Because many SRA investigations rely on the use of historical sediment data that were generated using outdated methods designed for matrices other than sediments, it is advisable to carefully evaluate the COPC list carried forward in order to ensure that COPCs are not retained solely because of data quality issues. These methods may have grossly over- or underestimated sediment chemical concentrations because: (1) they did not account for matrix interferences from organic material or salt; (2) detection limits were too high; and/or (3) samples were inappropriately collected or handled. Existing screening data should be reviewed to ensure that contaminant concentrations are accurate and representative of current site conditions. Rapid sediment characterization methods (see Appendix A) or collection of a limited number of samples using appropriate methods should be considered if there is reason to believe that existing data overestimate or underestimate site sediment contaminant concentrations.

3.1.2.2 Step 3b: Problem Formulation

If the re-evaluation and refinement of the conservative assumptions conducted in Step 3a does not support an acceptable risk determination, the investigation continues with Step 3b of the BERA, Problem Formulation. As discussed in the Navy ERA Policy, Steps 3b through 5 represent the most important components of the Tier 2 process (i.e., planning, study design, and verification) because they focus the scope and magnitude of the BERA. Specifically, these steps identify the endpoints to be evaluated, the laboratory methods to be employed, the statistical methods to be used, and the methods used for estimating and characterizing the ecological risks. The intent of these steps is to insure that the assessment focuses on the primary ecological concerns at the site, and that only the data necessary to make a risk management decision are collected. Numerous SMDPs are involved in these steps as indicated in Figure 3-1, and the appropriate U.S. EPA and Navy ERA guidance should be consulted for direction at each of these points.

For the purpose of Step 3b, the preliminary problem formulation derived in the SRA is expanded and refined to focus only on those COPCs that were retained following the Tier 1 assessment and the Step 3a refinement. Another site visit may be conducted to gather additional information regarding available habitats, the sources of COPCs, and fate and transport mechanisms at the site for the purpose of confirming the existence of complete exposure pathways. For example, in Tier 1 it might be assumed that a complete exposure pathway existed for piscivorous birds to be exposed to bioaccumulative COPCs in forage fish associated with the site. However, if all bioaccumulative COPCs are eliminated from the evaluation based on the Step 3a refinement, this exposure pathway also may be eliminated from the refined Problem Formulation. Similarly, additional evaluation of data may indicate that certain COPCs are not available, or that existing habitats are not sufficient to support the ecological receptors of concern.

3.1.2.2.1 Selection of Assessment Endpoints

As part of the refined Problem Formulation, assessment endpoints are identified for the BERA. U.S. EPA (1992) identifies assessment endpoints as “an explicit expression of the environmental value that is to be protected.” As discussed in the Navy policy, the assessment endpoints represent the target of the BERA, and set the basis for the development of specific ecological studies and data collection activities. It is not possible to evaluate all individual components of the ecosystem at a site. Instead, assessment endpoints should focus on particular ecosystem components or organisms that are potentially sensitive to contaminants and that are ecologically relevant. For example, the benthic invertebrate community often is selected as an assessment endpoint for sediment evaluations because these organisms are a primary food source for a wide variety of aquatic and terrestrial species and are a key link in most aquatic food webs. In addition, well-established laboratory methods are available for evaluating their toxic response to contaminated sediments. In addition to benthic invertebrates, both aquatic (e.g., predatory fish) and terrestrial (e.g., piscivorous species) upper trophic level species commonly are evaluated. Example assessment endpoints are summarized in Table 3-2.

Table 3-2. Common Sediment Assessment and Measurement Endpoints and Exposure Pathways

Assessment Endpoint	Primary Exposure Pathways	Example Measurement Endpoints
Survival, growth, and reproduction of the benthic invertebrate community	Ingestion, dermal absorption	Comparison to sediment benchmarks/ toxicity/bioaccumulation bioassays; community structure
Survival, growth, and reproduction of the pelagic community	Ingestion, dermal absorption, desorption to water column and subsequent respiration, ingestion of contaminated prey	Comparison to sediment/surface water/tissue benchmarks/toxicity/ bioaccumulation bioassays; community structure
Health of the relevant wildlife community	Ingestion of contaminated prey; ingestion of contaminated surface water and sediment	Dose modeling

3.1.2.2.2 Development of Risk Questions and Hypotheses

Based on the information evaluated for the refined problem formulation, the risk assessment team will develop risk questions that integrate the information collected into questions about the relationship among assessment endpoints and their responses when exposed to site contaminants. Specific risk questions should be developed for each assessment endpoint and COPC and will serve as the basis for later activities in Tier 2 (see Highlight 3-3).

Highlight 3-3. Examples of Risk Questions Commonly Used in Sediment ERAs

Relative Risk:

- Do COPC concentrations in bulk sediment samples from the site exceed COPC concentrations in bulk sediment samples from the reference area?
- Does mortality measured in laboratory toxicity tests for sediments from the site exceed mortality in laboratory toxicity tests for sediments from the reference area?
- Do COPC concentrations in tissue of organisms exposed to site-specific sediments exceed COPC concentrations in tissues of organisms exposed to sediments from the reference area?
- Is the diversity and abundance of the resident benthic community at the site impaired relative to the reference area community?
- Do dose estimates predicted for the upper trophic level species at the site exceed those predicted for reference areas?

3.1.2.2.3 Refinement of the Conceptual Site Model

In the Tier 1 SRA, a preliminary CSM is prepared (Figure 3-1) to focus the understanding of the site on the basis of available data. Based on the results of Step 3b, a revised CSM is developed that incorporates additional detail and focus. Appendix D provides an example planning table that can be used to assist in developing the refined CSM. An example of a refined CSM is provided in Figure 3-3.

3.1.2.3 Step 4: Study Design and the DQO Process

Step 4 of the Navy ERA process involves the identification and design of scientifically defensible, site-specific investigations necessary to address the risk hypotheses and questions developed. The development of a scientifically defensible design is accomplished through the application of the DQO process (U.S. EPA, 2000b). Each study design will be unique, and will be based on the assessment endpoints, COPCs, and risk hypotheses identified. The primary objective of Step 4 is to produce a draft Work Plan (WP) and a draft Sampling and Analysis Plan (SAP). It is important that these investigations be designed to identify cause-and-effect relationships between COPCs and assessment endpoints, and to support the risk characterization and risk management decisions (including development of PRGs).

3.1.2.3.1 Selecting Measurement Endpoints

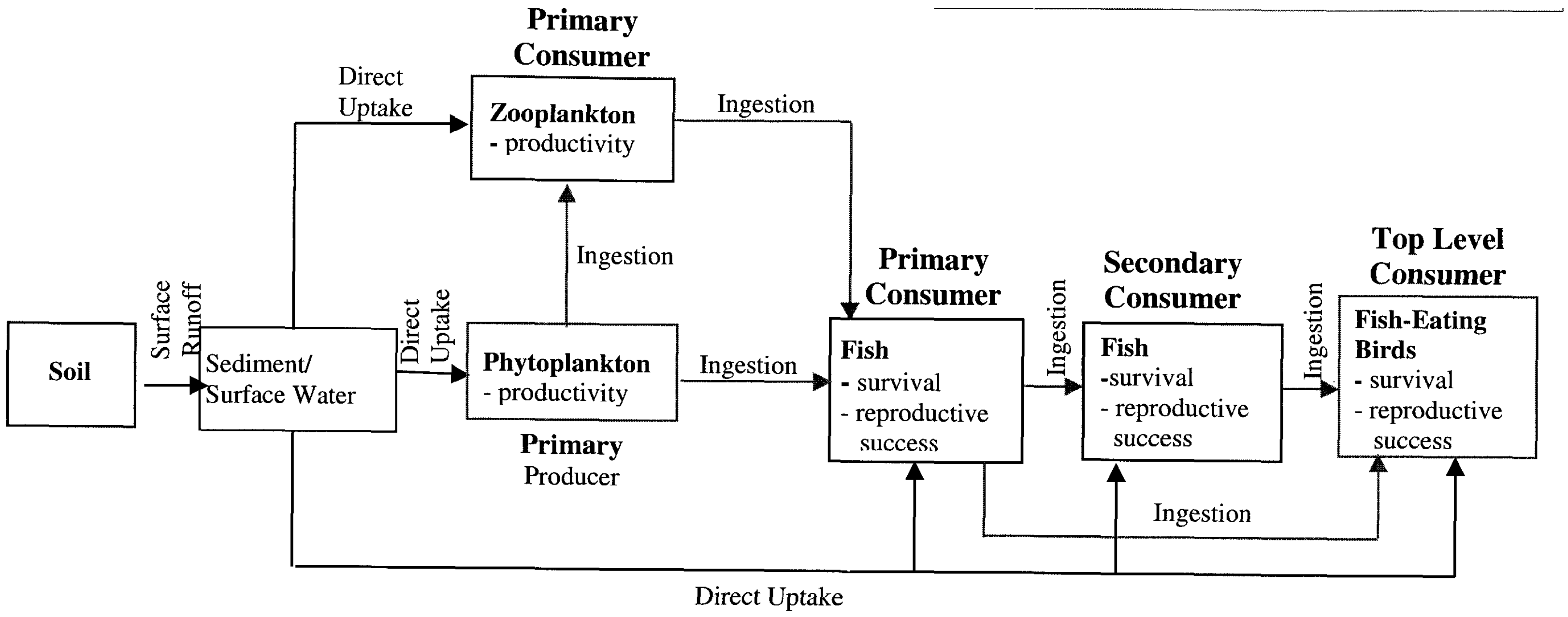
One of the first steps in the study design is the selection of measurement endpoints. Measurement endpoints provide a specific, quantifiable means of measuring a specific assessment endpoint as defined (U.S. EPA, 1992; 1998a) and can include measures of exposure or effect. For example, an assessment endpoint may be defined as “the survival and reproductive success of benthic invertebrates exposed to contaminated sediments at the site.” Suitable measurement endpoints would include percent survival associated with an acute or chronic toxicity test, number of young per female exposed, growth of individuals exposed, or the diversity and species abundance observed in the invertebrate community.

In selecting measurement endpoints, it is important to ensure that there is a direct relationship between the assessment and measurement endpoint, that they address the risk hypotheses and questions, and that there is a connection to the COPC and the routes of exposure. Example measurement endpoints are presented in Table 3-2. The suitability of potential measurement endpoints for addressing the identified assessment endpoints should be discussed with the appropriate regulatory agencies prior to the final selection.

In addition to these specific measurement endpoints, other ancillary data are sometimes evaluated more qualitatively in sediment risk assessments (Table 3-3). These data, although not used to quantitatively support the risk estimate, can be used to evaluate uncertainties or support assumptions.

3.1.2.3.2 Study Design and DQOs

Once the measurement endpoints are selected, the study design is completed. Prior to collection of additional data, DQOs should be developed in accordance with the guidelines provided in the U.S. EPA’s seven-step DQO process (U.S. EPA, 2000b). DQOs should include an identification of the study questions (Step 2), a list of the measurements required (Step 3), a discussion of the study boundaries (Step 4), and a description of the decision rules or data evaluation (Step 5). For each



This model identifies the contaminant sources, the fate and transport mechanisms, exposure pathways to ecological receptors (phytoplankton, zooplankton, fish and birds), and the assessment endpoints (productivity of plankton, survival and reproductive success of fish and birds). Note that this model does not identify specific contaminants of concern for each assessment endpoint.

Figure 3-3. Example of a Tier 2 BERA Conceptual Site Model

Table 3-3. Examples of Ancillary Data Interpretation Tools

Use Impairment	Related Measurement Endpoint	Data Interpretation Tools
Restrictions on fish and wildlife consumption	Bioaccumulation in resident fish	Equilibrium partitioning, comparison to guidelines (Table 3-1) provides a summary of example decision rules that can be used in a typical sediment study.
Degradation of fish and wildlife populations	Benthic community structure, bioaccumulation	Food web model, weight of evidence
Fish tumors or other deformities	Bioaccumulation in resident fish	Reference frequencies
Bird or animal deformities or reproduction problems	Bioaccumulation, community structure	Food web model, comparison to reference conditions, weight of evidence
Degradation of benthos	Community structure, toxicity (bioassays)	Comparison to reference conditions, use of impairment indices
Loss of fish and wildlife habitat	Chemistry, bioaccumulation, toxicity, benthos, stability	Comparison to reference conditions, weight of evidence

measurement endpoint evaluated, the corresponding decision rules should be presented. Finally, the DQOs should include a qualitative discussion of decision error types, and the specific consequences that must be considered in the study design (Step 6). Figure 3-4 presents an example of DQOs developed for a sediment assessment. The selection of the measurement endpoints for the BERA and development of the corresponding decision rules are among the most important aspects of the risk assessment process because they ultimately define the exit strategy for the site and should be discussed with the appropriate regulators.

Risk assessments often rely on comparisons of site results to reference area results to provide a measure of relative or incremental risk. Therefore, a key consideration in the development of the study design is the identification and selection of an appropriate reference area. The relevant regulatory agencies should be consulted throughout the process of selecting an appropriate reference area, as it can significantly affect subsequent risk management decisions. To ensure meaningful comparisons of sediment chemistry and bioassay results, it is important that physical and chemical factors at the reference area affecting site chemistry and bioavailability (e.g., grain size, TOC, AVS) are similar to the conditions at the site. In addition, habitat conditions should be as similar as possible to ensure that receptors identified as appropriate for site conditions also might be exposed to reference areas. If site conditions are heterogeneous, it may be necessary to select more than one reference area for evaluation, to ensure that all possible variations are addressed. In addition, some regulatory agencies have established regional reference values, reflecting ambient sediment or tissue concentrations based on monitoring data collected from throughout a specified area over a given period of time (Metcalf and Eddy, 1995).

Upon completion of the DQOs, a detailed Field Sampling Plan (FSP) should be prepared. Depending on the lines of evidence identified, data collection may include sediment samples, tissue samples, or laboratory bioassays (i.e., toxicity or bioaccumulation). Collection of physical and chemical data is discussed in Section 2.6, and an overview of study design and sample collection methods is provided in Section 2.7. A summary of biological data types that are often used during this portion of the risk assessment is provided in the following three subsections.

LABORATORY BIOACCUMULATION (*Macoma nasuta*) MEASUREMENT ENDPOINT

STEP 1: State the Problem

Evidence of possible sources of contamination to sediment habitats offshore of the site

STEP 2: Identify the Decision

1. Are COPC concentrations in *M. nasuta* tissue elevated above reference concentrations?
2. At locations where COPC concentrations in *M. nasuta* tissue exceed reference, is potential risk to upper trophic level receptors unacceptable (as determined through a food chain model)?

STEP 3: Identify Inputs to the Decision

1. Acceptable survival of *M. nasuta* in control sediment
2. Sufficient *M. nasuta* tissue mass for acceptable detection of COPCs
3. COPC concentrations in *M. nasuta* tissues in animals exposed for 28 days to site and reference site sediments
4. Background COPC concentrations in unexposed animals
5. Percent lipid and percent moisture of tissue samples
6. COPC concentrations, grain-size distribution, and TOC in site and reference site sediment samples
7. Overlying water quality conditions during testing period: salinity, dissolved oxygen, pH, and temperature.
8. Food-chain model parameters

STEP 4: Define the Study Boundaries

Questions 1, 2, and 3 will be based on the results of exposure of *M. nasuta* to the sediment from sampling stations within designated areas. Samples will not be collected in shoreline or intertidal areas covered with riprap or disposal debris. Results for each station will support a decision about the specific, designated area.

Bioaccumulation tests will be run for 28 days to allow data comparability with previous studies and other data sets. Five replicates of each reference site sediment and one replicate of each Site station sediment sample will be tested.

Question 1 also requires data from reference sites. Reference sites will have similar grain-size and TOC characteristics as Site sediments and will not be affected by known point sources of contamination. Reference sites will be sampled in the same way as Site stations, with surface sediment represented by the top 5 cm.

STEP 5: Develop a Decision Rule

Macoma tissue concentrations associated with exposure to site sediments will be statistically compared to tissue concentrations associated with reference areas. In addition, HQs will be calculated for COPCs to evaluate the potential risk to upper trophic level receptors.

1. If no more than one COPC exceeds reference and the HQ calculated is ≤1, then the area will be determined to pose no unacceptable risk.
2. If ≥2 COPCs exceed reference, and the HQ calculated is ≥1 but ≤10, the area will be determined to pose a moderate risk.
3. If ≥2 COPCs exceed reference and the HQ calculated is >10 the area will be determined to pose a high risk.

STEP 6: Evaluate Decision Errors

In general, if bioaccumulation and risk from consumption of contaminated prey is overestimated (false positive), a potential consequence is unnecessary remedial work that itself could be biologically detrimental. If bioaccumulation and food-chain risks are underestimated (false negative), a possible consequence is to fail to conclude that remedial action is required and biological systems could continue to be detrimentally impacted. Field-collected invertebrates and nondepurated *Macoma* tissues will be analyzed to help reduce uncertainty in estimates of food-chain risk.

STEP 7: Optimize the Design for Obtaining Data

Sampling design is presented in the SAP.

Figure 3-4. Example DQOs for Bioaccumulation Evaluation

3.1.2.3.3 Toxicity Bioassays

Laboratory bioassays are the primary means of assessing sediment toxicity because most of the methods are standardized, well documented, and validated. The objective of a toxicity bioassay is to determine the potential impact of site media (e.g., bulk sediment, porewater) on resident or representative site organisms. Toxicity tests provide a direct measure of the toxicity of the bioavailable fraction of COPC (Lamberson et al., 1992).

Typically, toxicity bioassays involve the exposure of a known number of individuals of the selected test species to sediments, both from the site and from a designated reference area, for a specified period of time (e.g., 10 days for acute amphipod tests). A variety of endpoints are evaluated depending on the test design, including percent survival, growth (measured as length and weight), enzyme induction, or observed behavioral changes. Depending on the test design, these endpoints also may be evaluated at predetermined intervals throughout the duration of the test. The results associated with the site and reference sediments are compared statistically. For example, if the percent survival associated with site sediments is statistically lower than that associated with the reference area, it is concluded that site sediments may be toxic to the test species under those conditions. However, different results may be obtained for a particular site evaluated depending on the COPC present. For example, some chemicals may be acutely toxic to a particular species, while another species may only show effects after chronic exposures to the same concentration. Therefore, at some sites, it may be beneficial to consider performing more than one toxicity bioassay, with multiple species. Common test methods, species, and test advantages and disadvantages are summarized in Table 3-4 for estuarine/marine bioassays and in Table 3-5 for freshwater bioassays. Guidance on selecting and performing toxicity tests and interpreting the results can be found in the following documents:

- *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates*. Second Edition. (U.S. EPA, 2000c; EPA/600/R-99/064. Office of Research and Development and Office of Water, Washington, DC.

- The *Inland Testing Manual* (U.S. EPA/USACE, 1998, Chapter 11; http://www.epa.gov/waterscience/itm/ITM/).

- *The Evaluation of Dredged Material Proposed for Ocean Disposal* (i.e., the "Green Book"; U.S. EPA/USACE 1991; http://www.epa.gov/OWOW/oceans/gbook/).

- http://www.epa.gov/waterscience/cs/pubs.htm.

Standard bioassay protocols are also published by the American Society for Testing and Materials (ASTM), which at this time are available as hard copy publications, or online for subscribers (see http://www.astm.org/).

The results of toxicity bioassays may be used alone or in conjunction with sediment chemistry and benthic community data to form the sediment quality triad (Chapman, 1986; Chapman et al., 1997). This method is based on the assumption that biological responses (e.g., toxicity observed or reduction in species diversity) are associated with the mixture of COPC in sediments (Chapman, 1986). In this approach, the data from each of these three types of studies are evaluated independently as separate lines of evidence and then combined to derive an overall conclusion about the site. In areas where the individual lines of evidence support the same conclusion, greater confidence in the decision is reached, whereas areas with conflicting information may require further evaluation regarding uncertainties in the data (Chapman, 1986; Chapman et al., 1997).

The organisms used in a toxicity test should represent appropriately sensitive infaunal or epibenthic organisms found in the vicinity of the site. The most common method for assessing acute effects from exposure to marine and estuarine bulk sediment is a 10-day amphipod toxicity test (e.g.,

Table 3-4. Estuarine and Marine Aquatic Bioassays for Use in Sediment Investigations

Test Species, Test Duration, and Medium	Endpoints	Advantages	Disadvantages	Other Remarks	Protocol Reference
Amphipod *Ampelisca abdita* 10 Day Acute Sediment Test 28 Day Chronic Sediment Test	10 Day Survival, Reburial. 28 Day Growth, Reproduction	Highly reliable test; laboratory exposure analogous to field conditions; very useful in defining gradients of sediment toxicity; readily available and widely distributed species; test salinities from 10 to 35 ppt; tolerant of fine sediments; high regulatory relevance; mortality endpoint has high ecological relevance; moderately cost-effective	Tube dweller, not in direct contact with sediment; sensitive to coarse-grained sediments; species is field-collected	Less sensitive than *E. estuarius* and *R abronius;* LC50 value for un-ionized ammonia is 0.83 mg/L (Kohn et al., 1994)	Test Method 100.4 (EPA 1994b); ASTM E1367-99 EPA 1998
Amphipod *Eohaustorius estuarius* 10 Days Sediment	Survival, Reburial	Highly reliable test; laboratory exposure analogous to field conditions; very useful in defining gradients of sediment toxicity; readily available species; test salinities from 2 to 28 ppt; tolerant of fine sediments; directly exposed to sediments; high regulatory relevance; mortality endpoint has high ecological relevance; moderately cost-effective	Less sensitive than *R. abronius; p*otential sediment interferences*;* not as well distributed as *L. plummulosus* or *A. abdita; s*pecies is field collected	Remove potential predators from sediment before testing; less sensitive to sulfide than *R. abronius* (LC50 = 104 μM total sulfides/L [Knezovich et al., 1995]); *E. estuarius* mortality is as sensitive as *R. abronius* mortality; *R. abronius* nonreburial is more sensitive than *E. estuarius* nonreburial*;* *E. estuarius* mortality more sensitive than *N. areanaceodentata* biomass, which is more sensitive than *N. areanaceodentata* mortality; mortality endpoint more sensitive than reburial; LC50 value for total ammonia is 125.5 mg/L and un-ionized ammonia is 2.49 mg/L (Kohn et al., 1994)	Test Method 100.4 (EPA 1994b); ASTM E1367-99 EPA 1998

Table 3-4. Estuarine and Marine Aquatic Bioassays for Use in Sediment Investigations (page 2 of 5)

Test Species, Test Duration, and Medium	Endpoints	Advantages	Disadvantages	Other Remarks	Protocol Reference
Amphipod *Leptocheirus plumulosus* 10 Days Sediment	Survival, Growth, Reproduction	Species is cultured; salinity range of 1.5 to 32 ppt (pore water salinities of less than 1 to 35 ppt [Niewolny et al., 1997]); tolerates full range of grain sizes (except sandy sediments with less than 5% silt/clay [Niewolny et al., 1997]); tolerant of fine sediments; highly reliable test; high ecological relevance; laboratory exposure analogous to field conditions; widely distributed and cultured	Tube dweller – not in direct contact with sediment		Test Method EPA 600/R-01/020 (2001)
Amphipod *Leptocheirus plumulosus* 10 Days Sediment	Survival, Reburial	As above.	Tube dweller – not in direct contact with sediment		Test Method 100.4 (EPA 1994b); ASTM E1367-99
Amphipod *Rhepoxynius abronius* 10 Days Sediment	Survival, Reburial	Highly reliable test; most sensitive of amphipods usually tested; laboratory exposure analogous to field conditions; very useful in defining gradients of contamination; readily available species; test salinities from 25 to 32 ppt; directly exposed to sediment; high regulatory relevance; mortality endpoint has high ecological relevance; moderately cost-effective	Sensitive to high total organic content; sensitive to fine grained sediments; not as well distributed as *Leptocheirus plummulosus* and *Ampelisca abdita*; species is field-collected	*R. abronius* mortality as sensitive as *E.estuarius* mortality; *R. abronius* nonreburial is more sensitive than *E. estuarius* nonreburial; LC50 value for un-ionized ammonia is 1.59 mg/L (Kohn et al., 1994); more sensitive to sulfides than *E. estuarius* (LC50 for total sulfides is 50 μM total sulfides/L [Knezovich et al., 1995]); 10-day survival protocol using *R. abronius* was a more sensitive indicator of toxicity than 20-day test with *N. arenaceodentata* based on statistical power of the test and not greater sensitivity of the organisms or endpoints (Anderson et al., 1998)	Test Method 100.4 (EPA 1994b); ASTM E1367-99 EPA 1998

Table 3-4. Estuarine and Marine Aquatic Bioassays for Use in Sediment Investigations (page 3 of 5)

Test Species, Test Duration, and Medium	**Endpoints**	**Advantages**	**Disadvantages**	**Other Remarks**	**Protocol Reference**
Diatom *Thalassiosira pseudomona* *Skeletonema costatum* *Dunaliella tertiolecta* *Phaeodactylum tricornutum* 4 Days Water	Growth, Biomass, Cell Counts	Represents aquatic primary producers; Can be used with filtered porewater	Primarily a water test; not relevant to sediment	Not recommended unless there is clear evidence that Navy activities are currently impacting the water body.	ASTM E 1218-97a
Mollusc *Macoma balthica* 28 Days Sediment	Bioaccumulation	Species is wild-harvested	Ecological relevance; available year-round, surface deposit-feeder; tolerates salinity down to 10 ppt		ASTM 1668-00 EPA 1998
Mollusc *Macoma nasuta* 28 Days Sediment	Bioaccumulation	Species is wild-harvested	Ecological relevance; available year-round, common test species; tolerates salinity down to 10 ppt		ASTM 1668-00 EPA 1998
Mollusc *Yoldia imatula* 28 Days Sediment	Bioaccumulation	Species is wild-harvested	Ecological relevance; available year-round, subsurface deposit-feeder	Seawater must be >25 ppt	ASTM 1668-00 EPA 1998
Mollusc Blue mussel *Mytilus edulis* 2 Days Water column	Development	Species is cultured; high dose responsiveness	Moderate ecological relevance; does not spawn year-round	Less sensitive than *Strongylocentrotus purpuratus* to sulfides (complete inhibition at 8-µM total sulfide/L [Knezovich et al., 1995])	Test Method 1005.0 (Chapman, 1995)
Polychaete *Capetella* sp. 20-28 Days Sediment	Bioaccumulation	Species can be cultured	Low sensitivity; mortality has moderate dose responsiveness		ASTM 1668-00

Table 3-4. Estuarine and Marine Aquatic Bioassays for Use in Sediment Investigations (page 4 of 5)

Test Species, Test Duration, and Medium	Endpoints	Advantages	Disadvantages	Other Remarks	Protocol Reference
Polychaete *Nereis (Neanthes) areneceodentata* 20 Days Sediment	Growth Survival	Species is cultured; species can be tested at salinities less than 20 ppt; growth test is cost-effective; low cost; mortality has high dose response; mortality has high ecological relevance	Low to medium sensitivity; mortality has moderate dose responsiveness; growth has moderate ecological relevance	Test animal age, duration of exposure, food ration, and choice of endpoint affects the magnitude of the toxic response (Bridges and Farrar, 1997; Bridges et al., 1997); *N. arenaceodentata* biomass is more sensitive than *N. arenaceodentata* mortality	(EPA 1990) ASTM 1611-99
Polychaete *Nereis virens* 28 Days Sediment	Bioacumulation	Species is cultured, or wild harvest, can tolerate salinities as low as 10 ppt	Cannot be tested with other test species (predator).	Surface deposit-feeder and omnivore. Good provider of biomass.	Test Method 1005.0; (Chapman et al., 1995); ASTM 1688-00
Possum mysid *Americamysis bahia* 7 Days Water column	Fecundity Survival Growth	Species is cultured	Very sensitive to changes in ionic balance in test solution	Laboratory success with the fecundity endpoint is rare (WSDE, 1997)	EPA Test Method 1007.0
Purple sea urchin *Strongylocentrotus purpuratus* 72 Hours Water column	Development Survival	Highly sensitive; early-life stage toxicity test; can be performed using filtered porewater	Species is field-collected; not recommended for sediments with a porewater salinity less than 10 ppt; does not spawn year-round; does not directly live in sediments; moderate ecological relevance	Highly susceptible to unionized ammonia toxicity when testing sediment pore water (EC50 for un-ionized ammonia was 0.057 mg/L [Greenstein et al., 1996]); total inhibition from sulfides at 20-µM total sulfide/L [Knezovich et al., 1995])	Test Method 1008.0 (Chapman, 1995)
Sand dollar *Dendraster excentricus* 72 Hours Water column	Development Survival	Highly sensitive; early-life stage toxicity test; gravid adults can be obtained year-round (U.S. EPA 1993b); can be performed using filtered porewater	Species is field collected; medium dose responsiveness; moderately cost-effective; moderate ecological relevance	Can be induced to spawn but with reduced gamete viability; proposed echinoderm effect threshold for unionized ammonia is 0.04 mg/L (U.S. EPA, 1993b)	Test Method 1008.0 (Chapman, 1995)

Table 3-4. Estuarine and Marine Aquatic Bioassays for Use in Sediment Investigations (page 5 of 5)

Test Species, Test Duration, and Medium	Endpoints	Advantages	Disadvantages	Other Remarks	Protocol Reference
Topsmelt *Atherinops affinis* 12 Days Sediment-water interface	Percent hatched	Most appropriate for testing early life stage fishes; can tolerate salinities from 2 to 60 ppt; can tolerate wide range in temperature with a preference of 19 to 23°C; reproductive season from early March to July or August depending upon latitude		Suggested test instead of using sediment pore water where fish are receptors of concern; numerically abundant in California estuaries	(Protocol based on Anderson, 1996)
Inland silverside *Menidia beryllina* 7 Days Water column	Survival	Species is cultured; occurs along both coasts of the United States; can tolerate freshwater to salinities of 35 ppt; can tolerate temperatures from 9.8 to 30°C; sexually mature from March or April through July or August			EPA Test Method 1006.0

ASTM = American Society for Testing and Materials.
EC50 = Concentration that causes an effect in 50% of the test organisms.
LC50 = Concentration that is lethal to 50% of the test organisms.
WSDE = Washington State Department of Ecology.

Table 3-5. Freshwater Bioassays for Use in Sediment Investigations

Test Species	Test Duration and Medium	Endpoints	Advantages	Disadvantages	Other Remarks	Protocol Reference
Algae, *Selenastrum capricornutum, Scenedesmus subspicatus, Chlorella vulgarius*	4 Days; water	Growth, Biomass, Cell Counts	Represents aquatic primary producers	Primarily a water test. Can be used with filtered pore water extracts		ASTM E 1218-97a
African Clawed Frog (FETAX) *Xenopus laevis*	96 Hours; sediment	Development Teratogenesis	Time and cost-effective; technical ease in conducting test; versatile for testing various media; sensitive to low levels of developmental toxicants; extensive literature available; most predictive when compared to lettuce germination, earthworm survival, *Daphnia* survival, and fathead minnow survival (Fort et al. 1995)		Can be used to test complex industrial mixtures; has been used to test surface water and aqueous extracts of soil; reference toxicant is 6-aminonicotinamide	ASTM E1439-98
Amphipod *Diporeia* sp.	10 Days; sediment	Bioaccumulation	Species can be cultured; pollution tolerant; important component in freshwater food chain	Small size requires massive numbers of individuals for chemical analysis	Subsurface deposit-feeder	ASTM 1688-00 EPA 1998
Amphipod *Hyalella azteca*	10 Days; sediment	Survival, Growth	Species is cultured; most highly sensitive of the freshwater test organisms; tolerates wide range of sediment grain sizes	Alkalinity commonly encountered in sediment porewater is toxic (Lasier et al., 1997; Duh and Myers, 1997)	Tolerates salinities up to 15 ppt	Test Method 100.1 (EPA 2000c); ASTM E-1706-95b
Amphipod *Hyalella azteca*	42 Days; sediment	Survival, Growth, Reproduction	Species is cultured; most highly sensitive of the freshwater test organisms; tolerates wide range of sediment grain sizes	Alkalinity commonly encountered in sediment porewater is toxic (Lasier et al., 1997; Duh and Myers, 1997)	Can be used to evaluate the bioavailability of sediment associated contaminants; tolerates salinities up to 15 ppt	EPA/600/R-99/064, ASTM E-1706-95b

Table 3-5. Freshwater Bioassays for Use in Sediment Investigations (page 2 of 3)

Test Species	Test Duration and Medium	Endpoints	Advantages	Disadvantages	Other Remarks	Protocol Reference
Daphnid *Ceriodaphnia dubia*	8 Days or until 60% of survivors have three broods; water	Survival, Reproduction	Species is cultured; important link in many food chains; species is as sensitive as fathead minnow; wide hardness tolerance; found throughout the United States	Test was originally developed to support water testing. Sediment elutriates containing massive amounts of fine material can mechanically induce mortality and /or create difficulty observing the very small offspring.	Applicable to testing effluents, leachates, liquid phases of sediments with minimal quantities of “permanently” suspended material, and pore-water	Test Method 1002.0
Fathead Minnow *Pimephales promelas,*	7 Days; water column	Survival, Growth	Species is cultured; occurs throughout the United States; eggs and early larvae commercially available; large effects data base	Not typically used in sediment suspended particulate testing	Applicable to testing effluents, leachates, and liquid phases of sediments with minimal quantities of “permanently” suspended material, and porewater	EPA/600/4-91/002
Mayfly - Burrowing, *Hexagenia* sp.	Life Stage Dependent; sediment	Survival, Growth, Bioaccumulation	Species is cultured; sensitive freshwater test organism; tolerates wide range of sediment grain sizes	Can mature into flying form if test duration extended	Important food chain item for freshwater fish	ASTM 1688-00
Midge Larvae, *Chironomus tentans*	10 Days; sediment	Survival, Growth	Species is cultured; sensitive freshwater test organism; tolerates wide range of sediment grain sizes	Can mature into flying form if test duration extended	Important food chain item for freshwater fish	ASTM 1706-95b; EPA Test Method #0100.2
Midge Larvae, *Chironomus tentans*	14 Days; sediment	Bioaccumulation	Species is cultured; sensitive freshwater test organism; tolerates wide range of sediment grain sizes	Can mature into flying form if test duration extended, requires many individuals to provide sufficient biomass	Important food chain item for freshwater fish	ASTM 1688-00 EPA 1998; Lifecycle Test Method #0100.5

Table 3-5. Freshwater Bioassays for Use in Sediment Investigations (page 3 of 3)

Test Species	Test Duration and Medium	Endpoints	Advantages	Disadvantages	Other Remarks	Protocol Reference
Oligochaete *Lumbriculus variegatus*	28 Days; sediment	Bioaccumulation	Species is easy to culture, know chemical exposure history, tolerant to wide range of sediment types, low acute sensitivity to wide range of chemicals, tolerates long exposures without feeding	Individuals are small and often difficult to remove from sediment and detritus matrix		ASTM E1688-00, EPA/600/R-99/064 (2000c)

ASTM = American Society for Testing and Materials.
EC50 = Concentration that causes an effect in 50% of the test organisms.
LC50 = Concentration that is lethal to 50% of the test organisms.
WSDE = Washington State Department of Ecology.

Rhepoxynius abronius, Eohaustorius estuarius, Leptocheirus plumulosus). *R. abronius* is a burrowing, deposit feeding amphipod that is commonly used in sediment contamination studies because it is sensitive to a wide range of toxicants. Freshwater bioassays are commonly conducted using *Hyalella azteca* (an amphipod) and *Chironomus tentans* (an insect larvae). When selecting test organisms, it is important to remember that individual species may have different sensitivities to specific COPC and test conditions. As a result, it may be useful to use more than one species. Selection of appropriate test species and methods for assessment of ecological risk are further discussed in *Confounding Factors in Sediment Toxicology* (Lapota et al., 2000).

Toxicity tests may also be conducted to evaluate exposure to porewater or suspended sediments using water column organisms as the test species. However, there are limitations with water column or porewater tests to evaluate sediment quality, as discussed by Chapman et al. (2002). The San Francisco Bay Regional Water Quality Control Board is developing guidance for assessing exposure to the sediment-water interface using intact, field-collected sediment cores and water column organisms (State Water Resources Control Board [SWRCB]; http://www.swrcb.ca.gov/).

The objective of a toxicity bioassay is to determine potential toxicity from exposure to site COPCs in various media; however, results can be difficult to interpret because uncontrolled confounding factors may be present. The most common types of confounding factors in aquatic bioassays are as follows:

- Elevated levels of ammonia or sulfide;
- Low dissolved oxygen concentration;
- Increased test organism sensitivity due to improper acclimation or inappropriate holding time;
- Artifacts due to manipulation of sediments or pore water;
- Unsuitable grain size for the test organism; and,
- Inappropriate porewater salinity.

A description of confounding factors and methods that can be used to minimize their effects is presented in the Issue Paper *Confounding Factors in Sediment Toxicology* (Lapota et al., 2000). The RPM should identify and use a laboratory with demonstrated experience in successfully controlling confounding factors and producing successful bioassay test results. Support for a quality assurance/quality control (QA/QC) evaluation of a toxicity testing laboratory is available through the Naval Facilities Engineering Service Center (NFESC) laboratory evaluation program. At a minimum, toxicity test data packages should include the information listed in Highlight 3-4, and test data should be reviewed as part of the data quality assessment. A summary of the key points regarding toxicity bioassays is provided in Highlight 3-5.

Highlight 3-4. Data Package Contents for Laboratory Bioassays

- Laboratory physical environment
- Feeding and food lot analysis
- Instrument calibration and maintenance
- Sample chain of custody
- Sample preparation for testing
- Test animal collection
- Quality control records (laboratory performance data, offspring sensitivity assessments, results of reference tests, and control charts)
- Dilution water preparation and lot analysis
- Test initiation
- Test animal randomization records
- Daily test conditions and observations
- Determination of percent survival and statistical deriviation of LC_{50}

Highlight 3-5. Bioassay Summary

- Test organisms should represent sensitive infaunal or epibenthic species found at the site; consider using multiple species
- Different tests are used to assess exposure to various site media (e.g., bulk sediment, porewater, water column); consider using more than one kind
- Different tests are used to assess acute versus chronic exposures; consider using both
- Test methods should identify and control factors that potentially confound bioassay results (e.g. ammonia and sulfide levels, grain size effects, low dissolved oxygen concentrations)

3.1.2.3.4 Bioaccumulation Bioassays

The bioavailibility of sediment-associated COPCs can be characterized using either laboratory or field bioaccumulation studies. These studies are important because they address COPCs that may not be acutely toxic to test organisms, but that potentially bioaccumulate in the food web where they may harm receptors of concern, including humans. Bioaccumulation data may be used to evaluate toxicity to aquatic organisms through comparison to effect-based tissue residues or in food web models. Bioaccumulation can be modeled from measured sediment concentrations, however, field or laboratory data are generally considered to provide a more accurate estimation (Lee, 1992). Detailed information regarding bioaccumulation evaluations can be found in the following publications:

- *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates*. Second Edition. (U.S. EPA, 2000c; EPA/600/R-99/064. Office of Research and Development and Office of Water, Washington, DC.

- *Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment* (U.S. EPA, 2000a; http://www.epa.gov/waterscience/cs/biotesting/bioaccum.pdf);

- Washington State Department of Ecology Web site (http://www.ecy.wa.gov/programs/tcp/smu/sediment.html), including a paper regarding bioaccumulation methods for fish and shellfish (http://www.ecy.wa.gov/programs/tcp/smu/bioaccum.doc).

A variety of test species and methods for estuarine/marine and freshwater bioaccumulation tests are summarized in Table 3-4 and Table 3-5, respectively. The most common types of sediment bioaccumulation studies are 28-day flowthrough laboratory tests using worms or clams (e.g., U.S. EPA/USACE, 1991). The marine bivalve clam *Macoma nasuta*, which is a selective deposit and filter feeder, is commonly used on the east and west coasts of the United States. The polychaetes *Nereis diversicolor* and *Neanthes (Nereis) virens* also are standard species for marine bioaccumulation tests. The oligochaete *Lumbriculus variegates* is a common freshwater species. The selection of the specific type of test and the test species to be used should be based on a variety of site-specific factors as summarized by Lamberson et al. (1992) and Burton et al. (1992).

Bioaccumulation also can be evaluated by measuring chemical concentrations in tissues from resident organisms or animals deployed using in situ studies. Successful in situ bioaccumulation studies using caged bivalves and topsmelt were used to assess ecological risk at Site 9, Naval Air Station (NAS) North Island, San Diego, CA. However, caged fish studies may be conservatively biased unless the test species has an extremely limited range in the natural environment. Other disadvantages of caged fish studies include potential loss of cages due to vandalism or weather-related events and potential problems with retrieval of equipment or loss of organisms. Additional discussion of in situ bioaccumulation studies can be found in Chapter 10 of *Critical Issues for Contaminated Sediment Management* (Apitz et al., 2002).

Bioaccumulation also can be estimated through the collection and analysis of organisms living at the site. The advantage of this method is that it provides a snapshot of potential bioavailability and uptake under actual site conditions and avoids the potential introduction of laboratory artifacts (Lee, 1992). However, it is often difficult to link these data to site-specific chemical concentrations, particularly with more mobile species such as lobsters, crabs, or fish that may be exposed at multiple locations (Lee, 1992). In addition, at some locations it may be difficult to obtain sufficient tissue mass for the desired chemical analyses.

The most important points regarding data collection to evaluate bioaccumulation are summarized in Highlight 3-6.

Highlight 3-6. Bioaccumulation Summary

- Well-established, standard laboratory bioaccumulation tests designed for dredged material evaluations can be used although study design should be modified as appropriate for sediment risk assessments
- In situ studies using animals deployed in cages may produce cost savings and more relevant data than standard laboratory tests
- COPCs measured in bioaccumulation studies should be limited to those that are known to bioaccumulate

3.1.2.3.5 Benthic Community Characterization

Another method that can be used to evaluate ecosystem health is the assessment of the resident benthic community attributes. This evaluation most often involves comparison of benthic community attributes at the site versus a suitable reference area. Common attributes used to assess benthic community health include species diversity, abundance, biomass, and key indicator species abundance. Benthic community characterization studies should be designed, conducted, and evaluated by an experienced benthic biologist because of numerous potential problems in selection of appropriate reference sites and community attributes, as well as interpretation of results.

Benthic community attributes are influenced by a number of factors in addition to site COPCs, including food availability, water quality (e.g., salinity, light, temperature, dissolved oxygen, depth), sediment grain-size distribution and TOC content, seasonal cycles and predation. Therefore, community metrics should be used in sediment investigations only when it is possible to distinguish COPC-related impacts to the community from other confounding factors. Additionally, decision criteria should be determined in advance in order to clarify how the results will be used to assess risk.

Benthic community collection and analysis methods can be found in the following documents:

- *Coastal 2000 - Northeast Component: Field Operations Manual* (Strobel, 2001; http://www.epa.gov/emap/nca/html/docs/c2kfm.html); and,

- *Recommended Protocols for Sampling and Analyzing Subtidal Benthic Macroinvertebrate Assemblages in Puget Sound* (U.S. EPA, 1987b; (http://www.wa.gov/puget_sound/Publications/protocols/protocol_pdfs/benthos.pdf).

The most important aspects of benthic community analysis are summarized in Highlight 3-7. Data pertaining to the benthic community structure are often combined with the results of analytical chemistry evaluations and toxicity bioassays in an approach referred to as the sediment quality triad (Chapman, 1986; Chapman et al., 1997) as discussed in Section 3.1.2.3.3.

3.1.2.4 Step 5: Verification of Field Sampling Design

Sediment site assessments often require relatively complex field and analytical programs. Step 5 of the BERA provides for verification of field efforts, especially those that may fail DQOs

Highlight 3-7. Benthic Community Summary

- Select appropriate reference area that matches the physical characteristics of the site
- Consult with an experienced benthic biologist to develop a technically sound study design
- Consider factors other than COPCs that may impact community parameters (e.g., site physical and physico-chemical characteristics, seasonal variations)
- Use standardized collection and taxonomic methods to ensure comparability with historical or local data

established in Step 4 of the BERA. In other words, the feasibility and suitability of the proposed study design is first confirmed under Step 5 to minimize cost, time, and effort associated with data collection, as well as to avoid generation of unusable data. The most common program elements that should be verified or addressed are:

- Site conditions and sampling gear – e.g., can the samples be collected?
- Confounding factors – e.g., will bioassays fail because of noncontaminant effects (grain size, ammonia, sulfides)?
- Will nonstandard tests meet DQOs – e.g., will in situ bioassays work?
- Does the sample design meet power assumptions – e.g., are a sufficient number of samples being collected?
- Will the contract laboratories meet DQOs – e.g., will matrix interferences compromise chemistry results?
- Is the reference site appropriate—e.g., will physical or chemical differences between the site and reference affect results?

Methods used to verify these elements will vary. For relatively small sampling programs, verification may be limited to a thorough technical review of the Field Sampling and Analytical Work Plan. For larger programs, it may be appropriate to conduct additional site visits or preliminary sampling using rapid sediment characterization methods to ensure that the sampling plan adequately addresses the nature and extent of contamination, or in the case of biological samples, that sufficient numbers of organisms are present.

3.1.2.5 Step 6: Implementation of Field Sampling Design

The process of executing the BERA Field Sampling Design (Step 6) at sediment sites parallels that of the process at terrestrial sites. Execution of this step, including work products, SMDPs, exit strategies, and risk communication, are detailed in Navy and U.S. EPA guidance (see Highlight 1-1). Sediment site case studies representing example work products for this step can be found at http://web.ead.anl.gov/ecorisk/case/.

3.1.2.6 Step 7: Risk Characterization

The risk characterization provides an integration of all data and results into one or more conclusions about the risks to the assessment endpoints. It includes three components: (1) a risk estimate; (2) a determination of ecological significance and risk acceptability; and (3) an uncertainty analysis. The uncertainty assessment should include a qualitative discussion of all COPCs that could not be evaluated quantitatively due to lack of benchmark or toxicity values. In general, this process is the same at sediment sites as at terrestrial sites and should follow the guidance provided for ecological risk assessments (U.S. EPA, 1992; 1998c). For most sediment sites, the risk characterization will involve interpreting and integrating the results associated with multiple assessment and measurement endpoints, a potentially complex process. Examples of methods for doing this include the sediment quality triad approach (Chapman, 1992; Long and Chapman, 1985) as well as WOE or lines of evidence approaches (e.g., Menzie et al., 1996). See Appendix E (San Diego, CA, North Island – Site 1 RI Report) for examples of WOE approaches used in sediment BERAs. For upper trophic level species, dose estimates typically are derived and compared to TRVs. A variety of sources exist that either provide estimated TRVs for selected species and chemicals (DON, 1998; Sample et al., 1996) or guidance for developing them (Sample et al., 1996; http://chppm-www.apgea.army.mil/erawg/tox/). As part of the risk characterization, the data collected in Step 6 should be thoroughly reviewed to ensure that it meets the DQO requirements outlined in Step 4 and is of sufficient quality for the investigation.

3.1.2.7 Step 8: Remedial Action Alternatives

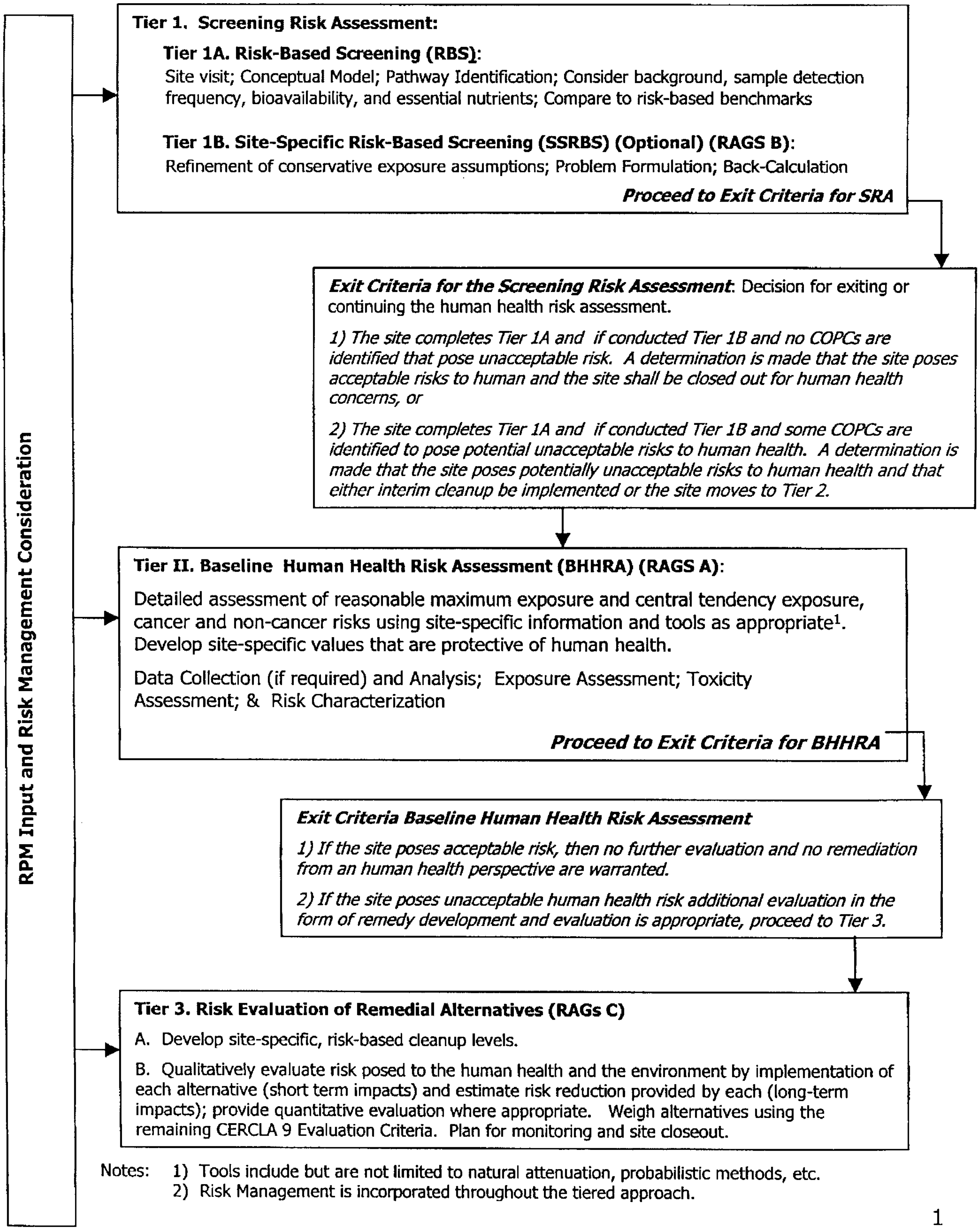
The evaluation of remedial action alternatives at sediment sites is discussed in Section 4.0. Based on the results of the risk characterization, a risk management decision is made regarding whether remedial action is required at the site. All risk management decisions should be discussed and agreed to with the appropriate regulators prior to proceeding with the evaluation of remedial alternatives. A site is determined to pose no unacceptable risk if HQs for all pathways evaluated are less than 1. However, depending on site conditions and discussions with the regulators, a determination of no unacceptable risk also may be made if it is demonstrated that the risks are comparable to background or ambient levels, if the population or community at risk is very limited, or if anticipated future uses indicate that exposures will be reduced or eliminated.

3.2 Human Health Risk Assessment

Similar to the ERA approach, the Navy's HHRA approach consists of three tiers (CNO, 2001; see Highlight 1-1):

- Tier 1: Screening Risk Assessment (SRA)
- Tier 2: Baseline HHRA (BHHRA)
- Tier 3: Evaluation of Remedial Alternatives.

The tiered framework is shown in Figure 3-5. Tier 1 focuses on a review of existing data, the development of a preliminary conceptual site model, and a comparison to conservative, risk-based benchmarks. Tier 2 provides a more detailed assessment of actual exposures, and includes the development of site-specific risk estimates. Tier 3 focuses on the risk-based evaluation of remedial alternatives (see Section 4.0).



Tier 1. Screening Risk Assessment:

Tier 1A. Risk-Based Screening (RBS):
Site visit; Conceptual Model; Pathway Identification; Consider background, sample detection frequency, bioavailability, and essential nutrients; Compare to risk-based benchmarks

Tier 1B. Site-Specific Risk-Based Screening (SSRBS) (Optional) (RAGS B):
Refinement of conservative exposure assumptions; Problem Formulation; Back-Calculation

Proceed to Exit Criteria for SRA

***Exit Criteria for the Screening Risk Assessment*:** Decision for exiting or continuing the human health risk assessment.

1) The site completes Tier 1A and if conducted Tier 1B and no COPCs are identified that pose unacceptable risk. A determination is made that the site poses acceptable risks to human and the site shall be closed out for human health concerns, or

2) The site completes Tier 1A and if conducted Tier 1B and some COPCs are identified to pose potential unacceptable risks to human health. A determination is made that the site poses potentially unacceptable risks to human health and that either interim cleanup be implemented or the site moves to Tier 2.

Tier II. Baseline Human Health Risk Assessment (BHHRA) (RAGS A):

Detailed assessment of reasonable maximum exposure and central tendency exposure, cancer and non-cancer risks using site-specific information and tools as appropriate[1]. Develop site-specific values that are protective of human health.

Data Collection (if required) and Analysis; Exposure Assessment; Toxicity Assessment; & Risk Characterization

Proceed to Exit Criteria for BHHRA

Exit Criteria Baseline Human Health Risk Assessment

1) If the site poses acceptable risk, then no further evaluation and no remediation from an human health perspective are warranted.

2) If the site poses unacceptable human health risk additional evaluation in the form of remedy development and evaluation is appropriate, proceed to Tier 3.

Tier 3. Risk Evaluation of Remedial Alternatives (RAGs C)

A. Develop site-specific, risk-based cleanup levels.

B. Qualitatively evaluate risk posed to the human health and the environment by implementation of each alternative (short term impacts) and estimate risk reduction provided by each (long-term impacts); provide quantitative evaluation where appropriate. Weigh alternatives using the remaining CERCLA 9 Evaluation Criteria. Plan for monitoring and site closeout.

Notes: 1) Tools include but are not limited to natural attenuation, probabilistic methods, etc.
2) Risk Management is incorporated throughout the tiered approach.

1

Figure 3-5. Navy Human Health Risk Assessment Tiered Approach

3.2.1 Tier 1: Human Health Screening Risk Assessment

Tier 1 of the Navy's HHRA policy is a risk-based screening step, intended to cost-effectively determine whether a site poses a risk using conservative, default exposure assumptions. The Tier 1 Human Health SRA has two possible components: a risk-based screening evaluation (Tier 1a), and a refinement (Tier 1b) similar to Step 3a of the Navy's ERA process.

Similar to the ERA (Section 3.1.1), the Human Health SRA focuses on existing data gathered and reviewed during the site characterization (Section 2.0). Prior to initiating the human health evaluation, the existing site information should be reviewed to ensure that there is sufficient data to support the risk assessment. See U.S. EPA (1989) for guidance on data evaluation methods. For human health sediment investigations, the key data at this stage are the nature and extent of contamination for the purpose of identifying possible COPCs, and the physical and topographical features at the site that would influence access to sediments and possible site uses under current and future conditions. Access by humans to submerged sediments is usually limited; therefore, it is very important in the site characterization state to identify possible site characteristics or uses that could result in sediment exposures. The following types of information should be considered:

- Type of waterbody (i.e., freshwater or marine, river, estuary, bay);
- Shoreline features and possible access points (e.g., beaches, docks, seawalls);
- Surrounding land use (e.g., industrial, undeveloped, residential) and reasonably foreseeable future land and resource use;
- Bathymetry and sediment substrate;
- Presence or absence of exposed sediments or seeps;
- Presence or absence of suitable habitat for shellfish beds;
- Potential for fishing/shellfishing activities, including by subsistence fishers; and

Based on this information, a CSM is developed as described in the following subsection.

3.2.1.1 Conceptual Site Model

The development of the CSM is a key step in the HHRA process as it identifies the likely COPC source areas, exposure pathways (i.e., direct exposure to sediment, or consumption of biota), and potential human receptors (i.e., expressed as exposure scenarios such as residential user and industrial user). The identification of the COPC source areas, fate and transport mechanisms, and contaminated media are described in Section 2.0. This subsection focuses on identifying complete exposure pathways and potential human receptors. A simplified human health CSM typical of sediment sites is presented in Figure 3-6.

As discussed in Section 3.1.1.2.3, a complete exposure pathway is one in which the COPC(s) can be expected to travel from the source to an identified receptor that can be affected. When selecting specific exposure pathways, both direct and indirect exposures to sediments should be considered. In general, because sediments are usually submerged, direct exposures to humans (e.g.,

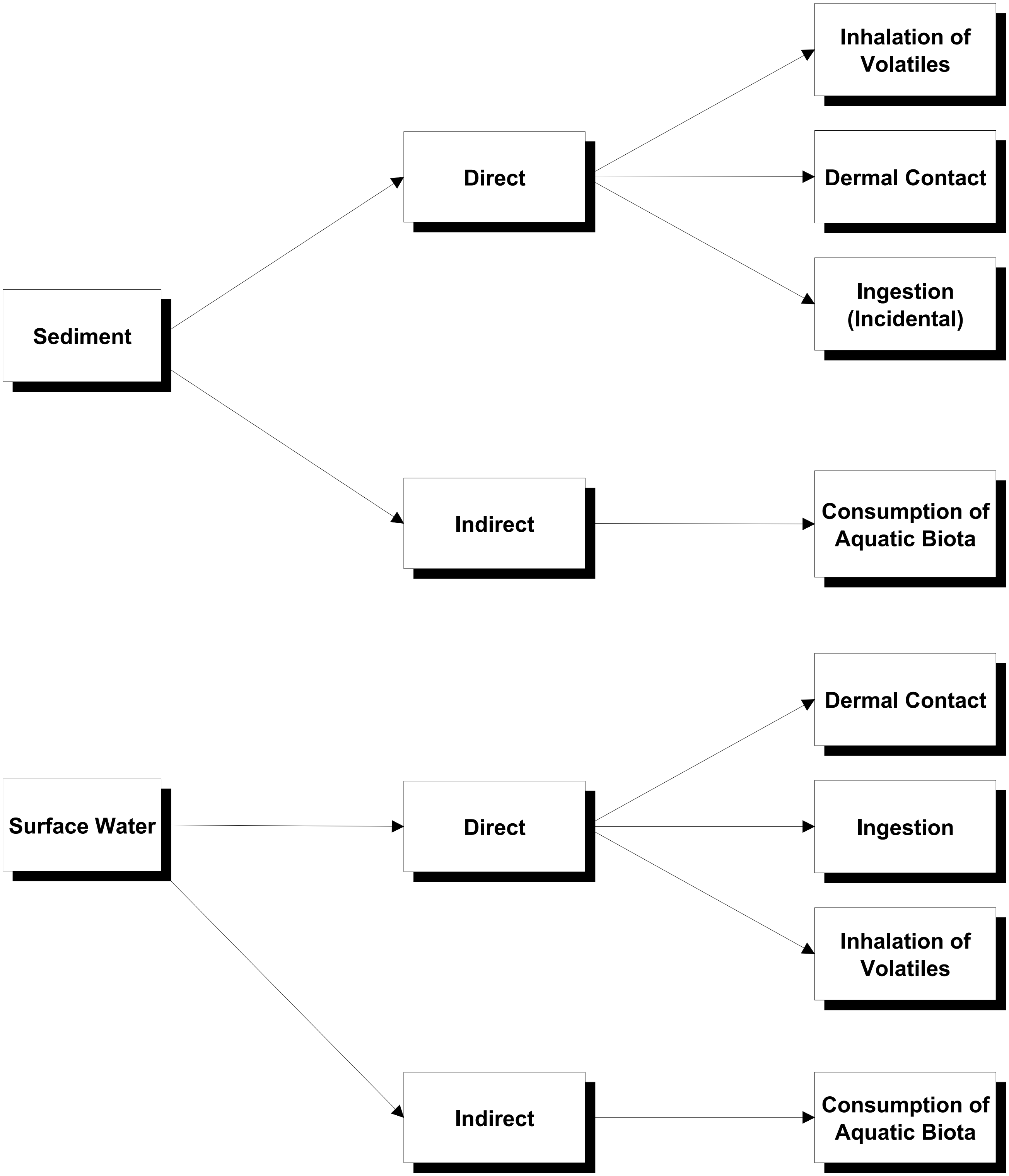


Figure 3-6. Simplified Human Health Conceptual Site Model for Sediments

incidental ingestion, dermal contact) are relatively limited unless there are areas appropriate for recreational activities (e.g., swimming, wading, clam digging), occupational exposures (e.g., maintenance of storm drains, underwater pipes, dredging, exposure to seeps) or potential subsistence exposures. These types of activities typically are associated with shallow intertidal areas or exposed mudflats; however, potential exposures also should be considered in areas where docks may be used for either swimming or boating due to the potential for chemicals to partition into the water column. As discussed for ecological receptors, exposures to sediment-associated contaminants via surface water are likely to be very minimal, particularly with persistent, hydrophobic chemicals, which tend to remain bound to sediment. Residential exposure pathways (e.g., gardening, day care) are not relevant to the evaluation of sediment exposures. However, possible exposure to floodplain soils should be considered as part of any on-shore evaluation.

The type of water body at the site will have a significant influence on the likely exposure scenarios and pathways. For example, recreational activities associated with a harbor or bay will differ from those associated with a marshy estuary or shallow stream. The surrounding land use also should be considered. For example, the potential for recreational exposures within a highly industrialized and developed area is different than for a relatively pristine environment, although these conditions may not deter subsistence fishing. Similarly, the presence of residential neighborhoods or day cares also should be considered when selecting pathways for evaluation as well as the exposure parameters used. In addition, possible future uses of the site, as defined by community input, site documentation and proposed site use plans, and surrounding land use should be considered.

3.2.1.2 Evaluation of Data Quality and Comparison to Background

As discussed in the Navy HHRA policy, the available analytical data should be evaluated to ensure that the DQOs for the site have been achieved. The key DQOs for the risk-based screening are as follows:

- Data Quality—The data should be collected in manner that provides a basis for remedial decision-making.

- Site Characterization—Sufficient data are necessary to adequately characterize the site spatially and at likely exposure points. To evaluate human exposures at sediment sites, it also is necessary to ensure that sufficient samples are collected in areas where exposures are likely to occur. For example, sediment concentrations associated with submerged, offshore areas should not be used to evaluate direct exposures to humans (i.e., dermal contact, incidental ingestion) although they could be used to estimate potential indirect exposures via consumption of biota (i.e., through the use of bioaccumulation modeling to estimate tissue concentrations in organisms consumed by humans).

- Analytical Detection Limits—It is important that analytical detection and quantitation limits are below the risk-based concentrations that will be used to identify potential risks.

Once the data have been determined to be of sufficient quality, sediment concentrations should be compared to available background concentrations per the Navy's *Policy on the Use of Background Chemical Levels* (CNO, 2000; see Highlight 1-1). The definition of background is provided in Section 3.1.2.1. HHRAs should not be conducted on chemicals that are present at levels less than background chemical levels (i.e., anthropogenic or naturally occurring levels).

The Navy's background policy (CNO, 2000; see Highlight 1-1) provides guidance on how to bring background issues into the process. This policy emphasizes the need to differentiate background contamination from site releases in the Navy IR programs. Background levels typically are based on data collected during the site investigation regarding ambient conditions. However, consultation with the relevant regulatory agencies regarding the appropriate estimation of background is recommended early in the process.

All chemicals determined to be present below background levels should be eliminated from this evaluation, and those that exceed will be retained as COPCs. However, any potential risks associated with the eliminated chemicals will be discussed in the risk characterization section. The purpose of this process is to focus the investigation on those COPCs that are related to site activities such that incremental risks associated with Navy activities are quantified.

3.2.1.3 Tier 1a: Risk-Based Screening

For the purpose of the Tier 1a evaluation, COPCs (i.e., chemicals for which maximum detected concentrations from the site which exceed background concentrations) are compared to conservative, risk-based concentrations (RBCs). RBCs are defined as chemical concentrations in the affected media (i.e., sediment) that are considered protective of human health. They are determined using standard risk equations rearranged to solve for the media concentration rather than risk. For Tier 1a, conservative, default exposure parameters and risk levels (i.e., 1×10^{-6} cancer risk and an hazard quotient of 1) are used to estimate the RBC. Similar to the background comparison in the ERA, this step is likely to be associated with high false-positive errors.

COPCs for which the maximum detected concentration is below the associated RBC are eliminated from the evaluation; if all COPCs are below the RBCs, the site may be determined to pose no unacceptable risk and the investigation concluded prior to Tier 2. However, those COPCs that exceed RBCs are retained for further evaluation in Tier 1b or Tier 2, as appropriate. Alternatively, if exceptionally high, localized concentrations of chemicals are observed, the best management strategy may be to perform a hot spot removal, lowering the overall site risk.

When selecting an RBC for use in Tier 1a, it is important to ensure that the exposure assumptions used to develop the screening value match the exposure scenario at the site. For example, RBCs based on industrial scenario assumptions should not be used to evaluate residential exposure scenarios if values derived using more appropriate exposure assumptions are available. However, most available RBCs have been developed using either industrial or residential exposure assumptions, rather than the recreational exposures that are most applicable to sediment sites. Therefore, use of these values may represent a very conservative evaluation of sediment exposures. In addition, very few RBCs have been established specifically for sediments. In place of sediment-specific values, guidelines developed for soil are frequently applied. Primary direct exposure pathways for sediments and soils are similar (i.e., dermal contact, incidental ingestion). However, a certain amount of uncertainty is associated with applying soil values to evaluate contaminated sediments due to the physical and chemical differences between soils and sediments and the potential affect they may have on the bioavailability of chemicals, particularly with respect to dermal contact. In addition, soil guidelines may not be protective of exposure associated with bioaccumulation and consumption of biota. As discussed for the ecological SRA, potentially bioaccumulative chemicals should not be completely eliminated from the evaluation based solely on comparison to screening values, although it may be possible to eliminate pathways such as direct contact or incidental ingestion. These and other related concerns should be discussed with the appropriate regulatory agencies prior to using an RBC.

Soil RBCs may be considered as a screening tool in Tier 1. Examples can be obtained from the following sources:

- U.S. EPA Region 3 Risk-Based Concentration Table (U.S. EPA, 1993a; http://www.epa.gov/reg3hwmd/risk/index.htm)

- U.S. EPA Region 6 Human Health Medium-Specific Screening Levels (U.S. EPA, 2000d; http://www.epa.gov/earth1r6/6pd/rcra_c/pd-n/screen.htm)

- Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites (Peer Review Draft; U.S. EPA, 2001c; http://www.epa.gov/superfund/resources/soil/index.htm)

- U.S. EPA Region 9 Preliminary Remediation Goals (U.S. EPA, 2001a; http://www.epa.gov/region09/waste/sfund/prg/index.htm)

If these established RBC values are not appropriate for the specific situation, or if site data exceeds the RBC, then site-specific and sediment-specific RBCs may be calculated in Tier 1b.

3.2.1.4 Tier 1b: Refinement of Risk-Based Screening

The Tier 1b risk-based evaluation has the same general objective as Tier 1a except that more realistic RBCs are calculated in place of conservative, default RBCs, by using site-specific exposure assumptions to evaluate the COPCs retained following Tier 1a. Guidance on developing site-specific RBCs is provided in the following sources:

- U.S. EPA Region 3 Risk-Based Concentration Table (U.S. EPA, 1993; http://www.epa.gov/reg3hwmd/risk/index.htm)

- U.S. EPA Region 6 Human Health Medium-Specific Screening Levels (U.S. EPA, 2000d; http://www.epa.gov/earth1r6/6pd/rcra_c/pd-n/screen.htm)

- Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites (Peer Review Draft; U.S. EPA, 2001c; http://www.epa.gov/superfund/resources/soil/index.htm)

- U.S. EPA Region 9 Preliminary Remediation Goals (U.S. EPA, 2001a; http://www.epa.gov/region09/waste/sfund/prg/index.htm)

- U.S. EPA Risk Assessment Guidance for Superfund. Volume I, Human Health Evaluation Manual, Part B: Development of Risk-Based Preliminary Remediation Goals (U.S. EPA, 1991; http://www.epa.gov/superfund/programs/risk/ragsb/index.htm)

- Navy guidance for conducting human health risk assessments (http://www-nehc.med.navy.mil/HHRA/index.htm).

One source of information on the ranges of values available for various exposure parameters is the *Exposure Factors Handbook* (U.S. EPA, 1998b). This document provides detailed information on general exposure parameters regarding dermal contact and incidental ingestion. Regional or local data also may be considered, especially for parameters that may vary widely depending on socio-economic factors or regional customs, such as fish consumption rates.

3.2.2 Tier 2: Baseline Human Health Risk Assessment

The Tier 2 BHHRA is intended to be a more rigorous evaluation than the Tier 1 site assessment. Rather than relying on conservative assumptions and comparison to RBCs, Tier 2 incorporates as much site-specific information as possible to calculate risk estimates.

3.2.2.1 Refinement of Conceptual Site Model and Exposure Assumptions

To ensure that Tier 2 reflects an evaluation of realistic exposures at the site, assumptions used to identify relevant exposure scenarios and develop exposure parameters should be carefully reviewed in light of known activities at the site, and adjusted if necessary. For example, if a recreational user is identified as the primary receptor of concern but additional investigation reveals that access to existing beach, dock, and waterfront areas at the site is severely restricted, the CSM and associated exposure parameters should be modified accordingly. Also, additional data collection may be beneficial to fully characterize the nature and extent of contamination.

Initial data collection frequently focuses on identifying sources or hot spots; however, for the purpose of the Tier 2 assessment, it is important to ensure that actual areas of potential exposure as well as all media of concern are accurately represented (U.S. EPA, 1989). Some of the bioassays discussed in Section 3.1.2.3.3 also may provide useful information for the HHRA, specifically the bioaccumulation tests for determining the bioavailability and update of site COPCs. In addition, collection of site-specific fish and shellfish tissue may be useful to reduce uncertainty if it is determined to be a dominant exposure pathway. As discussed in Highlight 3-8, the evaluation of fish should be carefully considered and discussed with the regulatory agencies, particularly if other sources of site COPC exist in the vicinity of the site or the fish species in question are highly mobile. It is very difficult to link measured tissue concentrations with site-specific sediments in species that may be exposed to a variety of different areas, making this data difficult to interpret. Under these conditions, it may be useful to compare concentrations of COPC in fish collected from the vicinity of the site with concentrations measured in fish from other locations representing ambient conditions. In addition, at sites where exposures associated with consumption of fish or other aquatic biota are driving the risk evaluation, it may be worthwhile to consider conducting surveys to derive a site-specific fish consumption rate. Individual fish consumption rates have been shown to vary dramatically by region, as well as by ethnic and socioeconomic group (U.S. EPA, 1998d). As a result, it is critical to ensure that the fish consumption rate selected accurately reflects the exposed populations. As summarized by U.S. EPA (1998d), a wide variety of possible survey methods are available (see Highlight 3-9) many of which can be applied to determine site-specific parameters to describe other activities as well. In addition, the *Exposure Factors Handbook* (U.S. EPA, 1998b) provides a discussion of the range of possible values that may be considered for each exposure parameter.

Generally, most risk assessments employ a deterministic or point estimate approach, relying on a single assumption to select each of the parameter values required in the assessment (e.g., fish consumption rate, exposure duration). Although most risk assessments include a combination of

Highlight 3-8. Factors to Consider When Collecting or Evaluating Fish Tissue Residues

Analytical Methods and Detection Limits

Prior to the analysis of contaminated fish tissue, detection limits for the selected analytical methods should be reviewed to ensure they provide levels that are, at a minimum, adequate for subsequent risk evaluations.

Fish Tissue Data Collection and Reporting

When evaluating and reporting residues of COPCs in fish tissues, it is important to note various factors, such as whether the sample originated from whole fish (i.e. whole tissue) or fillet-only samples and how the data were reported with regard to lipid content (U.S. EPA, 1987a). These kinds of issues inevitably introduce variability into monitoring data that must be addressed in order to provide realistic assessment of exposure point concentrations and exposure doses, and should be taken into account when determining a COPC concentration in fish tissue that is representative of consumption for a particular area. For example, using fillet-only data may under estimate risks to populations that consume the whole fish, whereas using whole body data may over estimate risks to those who consume only the fillet.

Fish Preparation and Cooking Methods

Fish tissue residue levels may also be affected by pre-consumption activities such as preparation and cooking. Preparation that removes skin and fat from filets may reduce the levels of fat-soluble (i.e., lipophilic) compounds such as PCBs. In addition, heat from various cooking methods may cause a change in the concentration of some compounds via physical processes like volatilization, chemical transformations, and mobilization and loss in lipid and water byproducts. Consideration of these factors is useful in an uncertainty analysis of individual risks.

Home Ranges

Species with large ranges may contact COPCs in other contaminated areas that are not associated with the current investigation. The larger the area under investigation, the greater the probability that a given species may contact a site-specific COPC. When conducting site-specific risk assessments, it is important to evaluate the uncertainties regarding whether concentrations for specific COPCs reported in fish tissues are representative of COPCs that may actually be consumed from an area.

central tendency and high-end values to provide a range of risk estimates, such an approach does not capture the wide array of possible exposure situations due to its dependence on specific assumptions regarding each exposure parameter. In a majority of the cases, the recommended exposure factors are protective of 95% of the population. As a result, a high degree of conservatism can be associated with the results when these factors are used concurrently. An alternative to the deterministic model is the application of a probabilistic model such as a Monte Carlo analysis. Probabilistic assessments

Highlight 3-9. Fish Consumption Survey Options

If consumption of sport fish or shellfish species is identified as a potential pathway of concern at a site, one option for addressing potential risks is to evaluate the actual use of the site by recreational or subsistence anglers. Information obtained from such surveys can be used to refine exposure parameters such as fraction ingested and ingestion rate.

The primary types of surveys typically conducted are on-site interviews and telephone surveys. Each of these can be conducted with local angler populations to better understand the typical exposures and consumption patterns for an area. The following types of information can be collected from site-specific angler surveys: demographic characteristics; fishing patterns; species targeted; number of fish caught; fish consumption patterns; the extent to which resident fish were used as food; the method of preparation of the fish to be consumed; and weights and lengths of fish caught by species.

A more detailed description of the methods to be used for conducting fish consumption surveys can be found in the U.S. EPA document *Guidance for Conducting Fish and Wildlife Consumption Surveys* (U.S. EPA, 1998d).

rely on a distribution of data for selected input parameters rather than a single point estimate. U.S. EPA has established protocols for the application of probabilistic analyses (U.S. EPA, 1997; 2001d; http://www.epa.gov/ncea/mcpolicy.htm). However, although probabilistic assessments potentially are useful for characterizing the uncertainty associated with exposures at more complex sites, they require a much larger effort than deterministic models and are often not warranted. In addition, many local, regional, and state regulatory agencies have specific requirements regarding the application of probabilistic methods, and should be consulted prior to initiating a probabilistic assessment. In practice, only a few local agencies have policies for accepting Monte Carlo risk assessment as part of a Tier 2 evaluation.

3.2.2.2 Toxicity Assessment

The toxicity assessment determines the relationship between the magnitude of exposure to a chemical and the nature and magnitude of adverse health effects that may result from such exposure. In human health risk assessment, contaminants are classified into two broad categories: noncarcinogens and carcinogens. Toxicity studies with laboratory animals or epidemiological studies of human populations provide the data used to develop toxicity criteria. Carcinogens are agents that induce cancer and numerical estimates of cancer potency are represented by cancer slope factors (CSFs). Noncarcinogenic effects are evaluated using reference doses (RfDs) and reference concentrations (RfCs) that are thresholds at which toxic effects are not exhibited. CSFs and RfDs can be found at U.S. EPA's Integrated Risk Information System Web site (http://www.epa.gov/IRIS/subst/index.html); individual regions also should be contacted for regional- or state-recommended factors. However, depending on the lead agency, federally approved toxicity values may be preferred at federal/Superfund sites over these other values. Additional guidance for selecting human toxicity factors is provided by USACE (http://www.usace.army.mil/inet/usace-docs/eng-pamphlets/ep200-1-15) and ATSDR (http://www.atsdr.cdc.gov/toxpro2.html).

Chemicals may be present at some sites for which an RfDs and/or CSF are not available. Although these chemicals cannot be evaluated quantitatively in the absence of RfDs and/or CSFs, they should be retained for a more qualitative evaluation. The appropriate regulators should be consulted regarding how to address these chemicals, and a summary of all chemicals of concern that could not be included in the quantitative evaluation should be provided in the uncertainty analysis.

3.2.2.3 Risk Characterization

The outcome of the Tier 2 BHHRA is individual and cumulative cancer and noncancer risk estimates for current and potential future use scenarios. The exit criteria for the Tier 2 BHHRA are defined as follows:

1. All individual and cumulative cancer and noncancer risks are determined to be within acceptable ranges for all COPCs, therefore, no further action or remediation for human health considerations are warranted; or

2. Individual or cumulative cancer or noncancer risks are found to be above acceptable ranges for one or more COPCs; therefore, development and evaluation of preliminary remediation goals and remedial alternatives is appropriate and the investigation proceeds to Tier 3, Evaluation of Remedial Alternatives (see Section 4.0 of this document).

For noncarcinogenic effects, hazard quotients are summed and compared to an acceptable hazard index of 1. U.S. EPA uses the risk range of 1×10^{-6} to 1×10^{-4} as a “target range” when managing risks at Superfund sites, although U.S. EPA has expressed a preference for achieving the goal of 1×10^{-6} for cleanups (http://www.epa.gov/superfund/programs/risk/baseline.pdf). However, site-specific or remedy-specific factors, including but not limited to exposure factors, uncertainty factors, and technical factors, may be taken into consideration on a site-by-site basis (NCP Preamble 8717, FR55 No. 26, March 8, 1990). Therefore, the appropriate regulatory agencies should be consulted regarding the acceptable risk range for a particular site and other risk management issues.

4.0 SEDIMENT REMEDIAL ALTERNATIVE EVALUATIONS

This section provides guidance for conducting an FS for a Navy sediment site. According to Navy policy, Tier 3 of the ecological and human health risk assessments is conducted during the FS (CNO, 1999 and 2001; see Highlight 1-1). Tier 3 is the evaluation of remedial alternatives with respect to the following:

- The effectiveness of reducing risk to acceptable levels;
- Ecological impacts related to remedy implementation; and
- Residual risks.

Sediment-specific issues associated with the Tier 3 evaluation are discussed in this section. As with terrestrial sites, Tier 3 and the FS for sediment sites focuses on the nine evaluation criteria for the selection of a remedy outlined in the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) (U.S. EPA, 1988; http://www.epa.gov/oerrpage/superfund/whatissf/sfproces/rifs/analys.htm).

Highlight 4-1 summarizes some of the challenges associated with the selection and implementation of a remedy for a sediment site. Comparatively few treatment technologies have been applied at sediment sites to date, although contaminated sediment remediation is a rapidly evolving field. This guide does not provide a comprehensive discussion of remedial alternatives for sediments; rather, it presents and discusses the major technical and management issues and provides references for supporting detailed technical information. This section addresses planning considerations, determination of the volume of sediment requiring cleanup, remedial alternative selection, monitoring considerations, and management of sediments in areas requiring maintenance dredging.

Highlight 4-1. Challenges Associated with Sediment Remediation

- There are no presumptive remedies for contaminated sediments
- It can be difficult to develop meaningful and realistic cleanup goals and verify that they have been achieved
- Overlying water poses logistical challenges to active remediation (e.g., difficulties in isolating target area, lack of visibility, interference by currents and waves, difficulties controlling contaminant release during cleanup)
- Little long-term data currently are available on the effectiveness of remedies

4.1 PLANNING CONSIDERATIONS

Prior to initiating the FS, the RPM should consider several issues that will influence the overall approach to the management of a sediment site. These issues include the status of source identification and source control, the potential for implementation of a regional management strategy or remedy, potential reuse issues, and identification of potential ARARs, and are discussed in the following subsections.

4.1.1 Status and Implications of Source Control

Identification and control of the contaminant input sources to sediments are critical (see Section 2.2 for further discussion of source identification). Contaminant sources must be identified and ideally should be eliminated or controlled prior to remediation in order to ensure that a sediment site will not be recontaminated. If environmental investigations are ongoing in adjacent upland areas, then onshore activities should be monitored to ensure that the current status of source identification and control is known. However, complete source control for sediment sites may be impractical because of the wide variety of potential point and nonpoint sources to the water body. If any of the sources cannot be eliminated or controlled, then the contaminant load from the source should be estimated to evaluate whether the site is likely to be recontaminated above established cleanup levels before any action is taken.

4.1.2 Potential Advantages of a Regional Approach

If sediment cleanup is likely to be required at multiple Navy installations in close proximity on the same water body or its tributaries, a regional remedy should be considered, or, at a minimum, the FS evaluations for individual installations should be coordinated. If a proposed regional or watershed approach includes non-Navy parties, then CNO approval must be obtained prior to Navy participation as specified in the *Policy on Sediment Site Investigation and Response Action* (CNO, 2002; see Highlight 1-1). A regional remedy or coordinated FS evaluation has several potential advantages over a series of independent, site-specific actions (see Highlight 4-2). Some of the key issues and factors that will influence the feasibility of a regional approach are summarized below in Highlight 4-2 and also should be taken into consideration.

Highlight 4-2. Should a Regional Approach to Sediment Management Be Adopted?

Potential Advantages

- Local impacts may be minimized and more easily managed (e.g., one dewatering and pretreatment facility could be constructed and operated in the most optimal location)
- Contractor mobilization costs should be reduced
- Coordinated resolution of issues associated with sediment management at all installations within a region will more effectively utilize Navy and stakeholder resources
- Data sharing, especially background and reference site data

Potential Disadvantages

- Potential permitting and legal issues associated with consolidation or disposal of sediments from multiple installations at one site would need to be identified and resolved. The need to obtain permits would significantly reduce the viability of any remedy under CERLCA
- Reconciliation of disparate schedules for completing assessment and remediation activities at various installations

The successful implementation of a regional remedy for contaminated sediments from multiple Navy installations will rely in part on consensus among all stakeholders (the Navy, regulators, other trustees, and the public) that the regional remedy is preferable to a series of independent actions. However, the local community near a proposed disposal site on a Navy base may oppose acceptance of contaminated sediments from other facilities. A process for public involvement in the development of a regional remedy should be developed to identify critical issues, facilitate consensus, and formulate alternatives that will be successful. Ideally, installations within a region should have coincident project schedules so that the FS evaluation can be conducted concurrently.

4.1.3 Consideration of Anticipated Future Land Use

In many instances, the anticipated future use of a sediment site will influence the selection of the final remedy. A proposed remedy may not be acceptable to the installation, state, or community if it is clearly incompatible with the future use of the site. For example, in situ capping or natural recovery is not likely to be accepted for a site where future dredging or construction is anticipated. Identification of possible future site uses should take place during the FS scoping process in consultation with regulatory agencies and other stakeholders.

4.1.4 Identification of Potential ARARs

Potential ARARs should be discussed with federal and state regulatory agencies and documented in the FS report for the site. Generally, the identification of potential chemical-specific and location-specific ARARs begins during the RI data collection effort and the potential action-specific ARARs are identified during the development of the remedial alternatives. Currently there are no federal chemical-specific ARARs for sediments, although some states (e.g., Washington) have promulgated sediment management standards.

The Total Maximum Daily Load (TMDL) program may present a potential issue for contaminated sediment sites and its potential applicability should be investigated while planning the FS. Under Section 303(d) of the Clean Water Act, states are required to identify impaired water bodies that do not meet applicable water quality criteria and develop TMDLs for these water bodies. The TMDL is the maximum amount of a pollutant that a water body can receive and still meet water quality standards, and allocates pollutant loadings among point and nonpoint pollutant sources (http://www.epa.gov/owow/tmdl/intro.html).

A list of potential federal ARARs for sediment sites is provided in Table 4-1. A comprehensive list of ARARs is beyond the scope of this document; however, additional information should be available from legal counsel. Additional legislative requirements are discussed under the National Response Center Web site at http://www.nrc.uscg.mil/nrclegistxt.htm, including the NCP (at http://www.epa.gov/oilspill/ncpover.htm), Hazardous Materials Transportation Act (HMTA), United States Department of Transportation (DOT) Regulations, and Executive Orders relating to the regulation of environmental activities. Other sources of information if the sediment site has a dredging component are the USACE "Green Book" (U.S. EPA/USACE, 1991) and *Inland Testing Manual* (U.S. EPA/USACE, 1998) (see Section 1.4).

Table 4-1. Potential Federal ARARs for Sediment Sites

Regulatory Requirement	Purpose/Requirement	Applicability to Remedial Action	Citation
Potential Chemical-Specific ARARs			
National Recommended Water Quality Criteria (WQC)	CWA Section 304 requires the U.S. EPA to publish water quality criteria. WQC have been developed by U.S. EPA's Office of Water and are published in the guidance document *Water Quality Standards Handbook* and subsequent revisions.	Remedial action objectives and remedial actions should be consistent with WQC.	33 USC §1314 National Recommended Water Quality Criteria and *Water Quality Standards Handbook* (U.S. EPA, 1994)
National Toxic Rule (NTR)	NTR established water quality standards for states that had failed to adopt their own standards.	If applicable, remedial action objectives and remedial actions should be consistent with standards in the NTR.	57 FR 60848 60 FR 22229
Potential Location-Specific ARARs			
Endangered Species Act	Requires that any proposed work avoid, minimize, or compensate for impacts to endangered or threatened species and critical habitats.	Listed endangered and threatened species, as well as candidate species for listing, may exist at or in the vicinity of offshore sediment.	16 USC §1531 et seq.; 50 CFR Part 216 and Part 402
Marine Mammal Protection Act	Requires that activities avoid, minimize, or compensate for impacts to marine mammals and their habitats	Marine mammals may exist at or in the vicinity of offshore sediments.	16 USC §1361 et seq.
Federal Fish and Wildlife Conservation Act	Requires that activities avoid, minimize, or compensate for impacts to fish and wildlife and their habitats	Fish and wildlife may exist at or in the vicinity of offshore sediments.	16 USC §662 et seq.
Coastal Zone Management Act	Requires that activities conducted within a coastal zone be consistent with an approved state management program.	Offshore sediments being investigated are within the coastal zone.	16 USC §1451 et seq.
National Historical Preservation Act of 1966 (NHPA) and Archaeological Resources Protection Act of 1979	Provides for the recovery and preservation of historical and archaeological significant artifacts. Implementing regulations for NHPA (36 CFR Part 65) establish the National Register of Historic Places and provide for preservation of historic properties and minimization of damage to historic landmarks.	This regulation may be applicable if scientific, prehistoric, or historic artifacts are found at a site during implementation of the selected remedial alternative. Docks, piers, or other structures may meet the eligibility requirements of the National Register of Historic Places.	NHPA: 16 USC §470; 36 CFR Part 65. Archaeological Resources Protection Act
Potential Action-Specific ARARs			
Resource Conservation and Recovery Act (RCRA)	Hazardous Waste Determination: Provides criteria for determining whether a solid or liquid waste is a RCRA or non-RCRA hazardous waste.	Applicable for determining whether dredged sediment and water from remedial actions must be managed as a hazardous waste (unless dredged material is excluded from testing pursuant to HWIR).	42 U.S.C. 6901 et seq.; HWIR 40 CFR 261.4(g)
	Generator Requirements: Provide general requirements for generators of hazardous wastes.	Generator requirements will be applicable for dredged sediment and water resulting from remedial actions that are considered a hazardous waste.	40 CFR, Part 262, Subparts A through E

Table 4-1. Potential Federal ARARs for Sediment Sites

Regulatory Requirement	Purpose/Requirement	Applicability to Remedial Action	Citation
Resource Conservation and Recovery Act (RCRA) (continued)	Requirements for Corrective Action Management Units and Temporary Units: Provide requirements for on-site treatment and temporary storage of hazardous waste in temporary units and corrective action management units.	Potentially hazardous sediment and water resulting from remedial action may be temporarily stored or treated on site. Remedial activities proposed for offshore sediment may include onshore pretreatment facilities.	40 CFR, Parts 264.552 and 264.553
	Land Disposal Restrictions: Require that the hazardous waste generator determine whether waste is restricted from land disposal.	Land disposal restrictions will need to be complied with in disposing of any dredged sediment or other waste generated from remedial activities that meets the definition of hazardous waste. Wastes may be treated to meet LDR standards prior to disposal.	40 CFR, Part 268.7
Clean Water Act (CWA), Sections 404 and 404(b)(1)	Prohibits unauthorized discharge of dredged or fill material into U.S. waters; promulgates guidelines to evaluate discharge of fill or dredged material into U.S. waters; may require mitigation for impacts, decided on a case-by-case basis.	Substantive requirements are applicable to placement of dredged sediment or other capping material in wetlands or navigable waters.	33 USC 1341, 1344 et seq.; 40 CFR 320.4; 40 CFR Part 230
Marine Protection, Research, and Sanctuaries Act (MPRSA), Sections 102, 103, and 104	Provides criteria for reviewing and evaluating permits for disposal of material in ocean waters. Section 103 authorizes USACE to issue permit subject to concurrence with the U.S. EPA.	Substantive requirements are applicable to dredging and placement of dredged sediment in ocean waters.	MPRSA Section 102,103, and 104
Rivers and Harbors Act, Section 10	Prohibits unauthorized obstruction or alteration of any navigable water of the U.S. and may require mitigation for impacts decided on a case-by-case basis.	Substantive requirements are applicable to dredging and placement of dredged sediment or other capping material in navigable waters.	33 USC 403; 33 CFR Part 322; 33 CFR Part 323; 40 CFR Part 230

CAA = Clean Air Act.
CFR = Code of Federal Regulations.
CWA = Clean Water Act.
FR = Federal Register.
HWIR = Hazardous Waste Identification Rule.
LDR = Land Disposal Restriction.
USC = United States Code.
WQC = Water Quality Criteria.

4.2 DETERMINING EXTENT AND VOLUME OF SEDIMENT TO BE REMEDIATED

Remedial action objectives (RAOs), PRGs, and site-specific cleanup levels are developed in the initial stages of the FS. If the ecological and human health risk assessments conclude that sediment contamination poses an unacceptable risk, then RAOs must be developed that specify the relevant contaminants, exposure pathway(s) and receptor(s) to be protected. The RAOs are the basis for the PRGs, which represent a range of COPC concentrations corresponding to various levels of risk. Examples of RAOs for sediment sites are presented in Highlight 4-3. Depending on the results of the ecological and human health risk assessments, different RAOs and cleanup levels may be required for different areas of the site.

Highlight 4-3. Examples of RAOs for Sediment Sites

- Protection of humans from the consumption of shellfish containing PCBs
- Protection of benthic invertebrates from the direct exposure to lead and cadmium in sediment

4.2.1 Contaminant- and Site-Specific Remediation Goals and Cleanup Levels

As noted in Section 4.1.4, there are no chemical-specific ARARs for sediments at most sites. The *Policy on Sediment Site Investigation and Response Action* (CNO, 2002; see Highlight 1-1) includes the following guidelines for development of cleanup levels for sediments sites:

- Risk-based sediment cleanup levels should be developed using site-specific information;
- The cleanup levels must be risk-based and achievable;
- Ecological screening values (i.e., published benchmark values) must not be used as cleanup levels; and
- Cleanup levels must not be lower than ambient (i.e., background) levels.

Cleanup should be not be conducted to levels that are less than ambient levels because sediments from elsewhere in the water body can be transported to and deposited at the site through natural processes. This would result in a cleanup that effects little or no ecological improvement (http://enviro.nfesc.navy.mil/erb/erb_a/regs_and_policy/don-background-pol.pdf).

PRGs should be clearly tied to the RAOs. Some approaches that can be used by risk assessors to develop remediation goals for sediment are summarized in Highlight 4-4. Ideally, site-specific data will show a correlation between chemical concentrations in sediment and adverse biological effects (e.g., toxicity or bioaccumulation). If this type of relationship can be established, then chemical concentrations that are protective of a given receptor for a given exposure route can be derived. The risk assessors should derive a range of remediation goals associated with various levels of risk for the risk manager to consider. The risk managers then can determine how to best balance risk reduction with technical feasibility, cost-effectiveness, and other considerations in the FS.

Highlight 4-4. Selected Approaches Used to Develop Remediation Goals for Sediment

- Development of a site-specific relationship between chemical concentration and adverse biological effects and identification of concentrations associated with various effects levels (apparent effects thresholds [AETs])
- Utilization of a food chain model to predict sediment concentration that will protect upper trophic level receptors (including humans) from consumption of contaminated prey
- Calculation of sediment concentration that will result in attainment of ambient water quality criteria in the overlying water column (equilibrium partitioning approach)
- Integration of results for multiple lines of evidence into a single indicator value to delineate areas that pose an unacceptable risk and require remediation

Development of remediation goals for individual chemicals may be difficult if the sediments contain a mixture of chemicals that appear to contribute to observed biological effects. In this case, several approaches should be considered: (1) identification of the chemicals that are likely to be the primary risk drivers; (2) selection of an indicator chemical to represent each class of contaminants; or (3) use of a weight-of-evidence approach to identify areas for remediation on a station-by-station basis. The compound or class of compounds responsible for causing toxicity can be investigated through a toxicity identification evaluation (TIE) (U.S. EPA, 1991), in which a sample is physically or chemically divided into various fractions and the toxicity of each fraction is determined. References for further guidance on the development of site-specific, risk-based remediation goals for sediment can be found in Highlight 4-5.

Highlight 4-5. Further Guidance for Development of Remediation Goals for Sediment

- Example Approach for the Development of Site-Specific Preliminary Remediation Goals for Protection of Ecological and Human Health at Navy Aquatic Sites (http://web.ead.anl.gov/ecorisk/issue/pdf/PRGGuideFinal2-14-01.pdf)
- Guidelines for Assessment and Remediation of Contaminated Sediments in San Diego Bay at NASSCO and Southwest Marine Shipyards (California Regional Water Quality Control Board, San Diego Region, June 1, 2001; http://www.swrcb.ca.gov/rwqcb9/programs/shipyards/SWM%2013267%20Letter%20-%20Assessment%20and%20Remediation.pdf)

Remediation goals should not necessarily be used directly as site-specific cleanup levels. Each site has a unique and complex set of physical, chemical, biological, political, economic, and regulatory factors that must be taken into consideration by risk managers. Additionally, remediation of all sediments with chemical concentrations exceeding the goals may not be feasible because of financial or technological limitations. Therefore, the development of site-specific cleanup levels should focus on optimal risk reduction using remediation goals as a starting point, and take into consideration site-specific modifying factors that include but are not limited to:

- Pattern of contaminant distribution (i.e., presence of localized hot spots versus large volumes of sediment with relatively uniform contaminant loads);
- Contaminant fate and transport (i.e., potential for natural recovery or sediment resuspension in areas of concern);
- ARARs and other to-be-considered criteria;
- Regional ambient levels of chemicals; and,
- Community or agency concerns.

The Major Contaminated Sediment Sites (MCSS) Database developed by Applied Environmental Management, Inc. and Blasland, Bouck & Lee, Inc. for the General Electric Company (http://www.hudsonvoice.com/mcss/) contains information on sediment cleanup goals and the basis for the goals for more than 60 remediation projects in the United States. An evaluation of the larger projects in the MCSS database found that in many cases either a mass removal approach was adopted or the cleanup target was not clearly tied to the protection of a given endpoint (Cushing, 1999b). In the FS, every effort should be made to develop cleanup levels that are clearly based on and tied to the site-specific risk information.

4.2.2 Consideration of Contamination at Depth

Ecological and human health risk assessments of sediment sites typically focus on biologically available surface sediments. Therefore, the RPM must ensure that the three-dimensional distribution of chemicals also is delineated as part of the site characterization (see Section 2.5). Subsurface chemical concentrations must be measured and the vertical extent of contamination defined in order to accurately complete the FS. If subsurface chemical concentrations are higher than surface concentrations, then the likelihood of erosion and exposure of deeper sediments must be considered in an evaluation of potential in situ remedies. If dredging is under consideration, then the volume of sediment with chemical concentrations above site-specific cleanup levels must be determined to allow calculation of more accurate cost estimates.

4.3 REMEDIAL ALTERNATIVE SELECTION

Currently, a relatively limited number of effective technology options are available for remediating sediment. A common response action for sediments is removal (dredging) followed by disposal. Dredged sediments generally require processing such as dewatering, physical separation, and limited treatment prior to disposal or reuse. Another common response action is in situ isolation of the sediments from the environment, typically by covering the sediment with a sand or gravel cap. Monitored natural recovery involves the processes wherein contaminated sediments are buried (isolated), degraded, transformed, and/or immobilized under natural conditions such that over time the chemical concentrations in the sediment and overlying water are reduced to acceptable levels.

The strengths, weaknesses and other issues associated with the following remedial alternatives for sediment are discussed in this section:

- Monitored natural recovery;
- In situ capping;
- Dredging;
- Disposal options; and,
- Beneficial reuse options.

A summary table is provided in Section 4.3.9 that summarizes the applicability, strengths, weaknesses, and cost and schedule implications for various sediment remediation options. Also, the Sediments Research Web (http://www.sediments.org/) provides additional information and resources regarding the management and remediation of contaminated sediments. U.S. EPA is in the process of preparing a guidance document for contaminated sediment remediation; a draft of the document is expected to be issued in 2003.

4.3.1 Monitored Natural Recovery

Natural recovery involves leaving contaminated sediments in place and allowing natural processes to contain, destroy, or otherwise reduce the bioavailability of the contaminants (NRC, 1997) (http://books.nap.edu/catalog/5292.html). Natural recovery reduces environmental risk through two primary mechanisms: burial of contaminated sediments by cleaner sediments; and sequestration or transformation of contaminants through biological, chemical, or physical processes to less toxic or bioavailable forms. Burial results in natural capping, which inhibits diffusion of contaminants from sediments to the water column, reduces or prevents resuspension of contaminated sediments, and protects the benthic community from exposure to the contaminants. Natural recovery differs from a No Action alternative because it entails the inherent assumption or conclusion of some level of unacceptable risk and requires ongoing monitoring to verify that risk reduction is occurring. Additionally, institutional controls may be required to protect human health (e.g., through fish consumption advisories) and prevent sediment disturbance during recovery. Monitored natural recovery is most applicable under the following circumstances:

- The human health or ecological threat is not immediate or substantial.

- Contaminant sources are controlled or eliminated.

- The risk from sediment contaminants can be reduced or eliminated in a reasonable length of time.

- The impact of a remedial action is likely to cause greater environmental harm than leaving the contaminated sediments in place.

- The area of concern includes a large volume of sediment with relatively low contaminant concentrations.

- The area of concern is in a depositional environment with a low potential for sediment resuspension.

- The area is not anticipated to be subject to dredging, construction, or other activities that might disrupt the sediment bed.

- Few other technically feasible or affordable options are available.

The primary strengths and weaknesses of monitored natural recovery are summarized in Highlight 4-6.

Highlight 4-6. Strengths and Weaknesses of Monitored Natural Recovery

Strengths:

- Eliminates environmental damage caused by dredging
- Potentially feasible option for large volumes of sediment with low contaminant concentrations and relatively low risk
- Low cost relative to other remedial options

Weaknesses:

- It is only appropriate if natural recovery mechanisms can be demonstrated
- Natural recovery processes may be very slow and in many cases contaminants are not likely to be removed or destroyed
- Contaminated sediments have the long-term potential to be remobilized by natural or human activity
- Requires long-term monitoring
- Long-term liability remains

In many cases, natural recovery relies primarily on sediment burial. Contaminant concentrations in sediments can be reduced or degraded through dilution, dispersion, volatilization, biodegradation, and other processes (U.S. EPA, 1998); however, these processes can be extremely slow in sediments, requiring decades or centuries for contaminant removal. Sediment burial occurs in net depositional environments where the sediment accumulation rate exceeds the erosion rate. However, the progressive burial of contaminated sediment by new, cleaner sediment does not necessarily prevent the dispersion of contaminants because physical and biological processes can mix surface and subsurface sediments, thereby exposing buried sediments at the surface.

Therefore, candidate sites for natural recovery should undergo a detailed site characterization as described in Sections 2.0 to establish sediment accumulation rate, degree of vertical mixing of surface and subsurface sediments, and likelihood and frequency of sediment resuspension and transport away from the site (see Table 2-2). For example, radioisotope profiles (^{210}Pb and ^{137}Cs) of sediment cores can be used to estimate sediment accumulation rate and degree of vertical mixing (Figure 4-1). Contaminant degradation via chemical and biological contaminant transport pathways also should be evaluated. The behavior of the sediment bed when subjected to extreme weather conditions (e.g., severe storms) also should be evaluated.

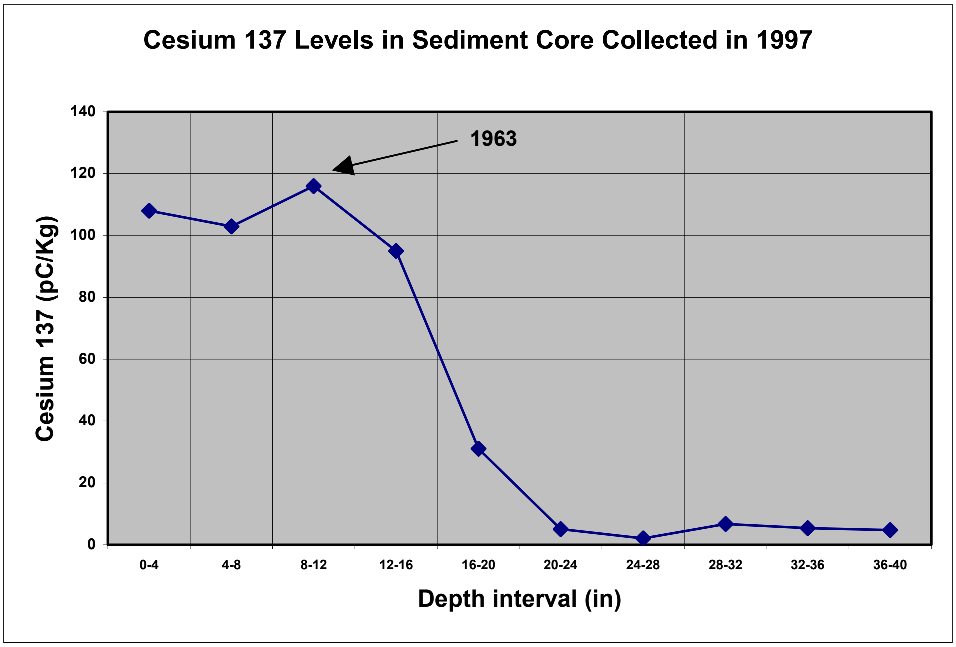


Figure 4-1. ^{137}Cs Activity in Sediment Core Indicates a Sediment Accumulation Rate of 0.3 inch/yr and a Moderate Degree of Vertical Mixing, as Shown by Broad ^{137}Cs Peak

A two-step approach can be used to evaluate the potential for monitored natural recovery at a site. For example, at Hunters Point Shipyard (HPS) in San Francisco, CA, the potential for sediment resuspension initially was evaluated using existing grain size data for surface sediments and readily available wave and current data for South San Francisco Bay (Battelle and Woods Hole Group, 2000). Following the initial evaluation, site-specific data were collected to characterize sediment resuspension and transport patterns around HPS with greater certainty. Time-series measurements of waves, currents, suspended sediment concentrations, temperature, and salinity were collected at three stations for one full tidal cycle using sediment transport measurement systems (STMS). Each STMS included one current meter and turbidity, temperature, conductivity (salinity) and pressure (water depth) sensors. The STMS data were used to estimate the magnitude and direction of suspended sediment flux at each of the stations. Regional sediment transport patterns and areas of net sediment accumulation and erosion were predicted using a regional hydrodynamic and sediment transport model (Woods Hole Group and Battelle, 2002). The regional model also was used to investigate expected sediment resuspension and transport during extreme storm and runoff events. Detailed geologic description of sediment cores, radioisotope profiles and information on benthic fauna also were used to support the characterization of sediment dynamics at HPS.

The paper *Sediment Stability at Contaminated Sediment Sites* (Ziegler, 1999) contains good background information on sediment transport processes and describes a similar two-step approach for estimating scour depth and likelihood of re-exposure and erosion of contaminated subsurface sediments. The first tier involves a relatively simple analysis of sediment stability at a site and can be used as a screening tool. The second tier is a more rigorous evaluation using a sediment transport model and site-specific data to evaluate the potential impacts of rare storms. Case studies of the effects of rare storms on two sites (i.e., a 100-year flood and a hurricane) indicated that only relatively minor erosion of cohesive sediments was likely to occur.

Natural recovery may take years or decades to achieve risk-based goals, depending upon the sediment accumulation rate and degree of vertical mixing. For example, in highly mixed sediments in Richardson Bay near the mouth of the San Francisco Bay estuary, it was estimated that more than 75 years would be required to bury 90% of deposited contamination below the mixing zone (Fuller et al., 1999). Because long-term monitoring is required to monitor contaminant concentrations in sediment and verify the predicted burial rate, monitoring costs are likely to be significantly higher for a natural recovery remedy than they would be for other management strategies.

Monitored natural recovery of sediments has been applied to remediate PCB contamination in Operable Unit (OU) 2 of the Sangamo-Weston/Twelve-Mile Creek/Lake Hartwell Superfund site, GA (U.S. EPA, 1994b) and lead contamination in OU 3 of the Interstate Lead Company Superfund site, AL (U.S. EPA, 1996). Sediment management practices implemented at each site include restrictions on fishing and other recreation activities, public education, routine monitoring, and five-year reviews to protect human health during recovery. Routine monitoring of sediment and biota provides information about the progress of recovery, including sediment accumulation rates and recovery rates for benthic organisms and fish. In Commencement Bay, WA, natural recovery is a candidate remedial technology at those sites that are “marginally” contaminated, which is defined as sites where the remedial action objectives can be met within a 10-year time period with no other intervention than monitoring (Hylebos Cleanup Committee, 1999).

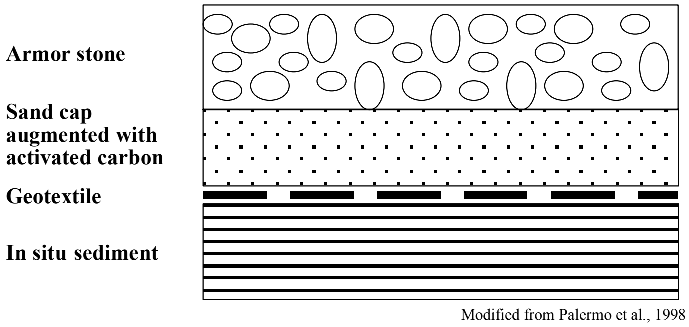
4.3.2 In Situ Capping

In situ capping is the controlled, accurate placement of a clean, isolating material over contaminated sediments without relocating or causing a major disruption to the original bed (NRC, 1997). This remedial option has similar applicability, advantages, and disadvantages as monitored natural recovery except that the capping material will more immediately isolate the contaminated sediments from the aquatic environment. A cap may serve one or more purpose (Palermo et al., 1998):

- Physical isolation of sediments to prevent exposure of aquatic organisms to sediment-bound contaminants;
- Stabilization of sediments to prevent impacts caused by resuspension, transport, and redeposition elsewhere; and,
- Chemical isolation to prevent contaminant flux to the overlying water column via diffusion.

The intended purpose of an in situ cap and site-specific characteristics (e.g., hydrologic setting, benthic community composition) will drive its design, construction, and monitoring requirements. Caps typically are composed of granular material such as sand, silt, or natural sediments. In general, finer-grained material with some organic carbon (i.e., naturally occurring clean sediment) will provide a greater degree of chemical isolation than sand because of its greater sorption capacity (Palermo et al., 1998). However, sandy materials are stable at steeper slopes, easier to place, less likely to resuspend, and less likely to be subject to burrowing by benthic organisms. Geotextile membranes, armor stone, or a combination of materials (i.e., multilayer cap) also may be used (Figure 4-2).

In situ capping is most effective in relatively sheltered areas that are not exposed to erosive forces such as currents, waves, and navigation propeller wash, or to upwelling from groundwater. An in situ cap can be armored with stone or other material to prevent erosion in higher energy environments. The scour caused by navigation (commercial and recreational vessels) would necessitate very large armor stone, making in situ capping difficult in or near most active navigation channels (Environmental Laboratory, 1987).



Modified from Palermo et al., 1998

Figure 4-2. Example of a Layered Sediment Cap

The feasibility of an in situ cap depends primarily on appropriate site conditions. A comprehensive site characterization must be completed that considers waterway configuration and uses, hydrodynamic and geotechnical conditions, and physical, biological and chemical characteristics of the contaminated sediments (Palermo et al., 1998). Both current and possible future conditions must be considered. If in situ capping appears to be feasible, then specific engineering and design studies would be needed to determine optimum cap thickness and composition. Some of the data needed to support development of a capping remedy are presented in Highlight 4-7.

Highlight 4-7. Data Needed to Support In Situ Capping

Sediment Characteristics:

- Level of contamination
- Grain size distribution
- Shear strength
- Resistance to erosion
- Consolidation
- Plasticity
- Density

Site Characteristics:

- Constraints posed by the cap on waterway traffic
- Water depth
- Typical currents
- Wave climate
- Navigation traffic
- Flood flows
- Aquatic resources
- Groundwater flow patterns

(from Georgia Tech Research Corporation, 2001; http://www.hsrc.org/capping/assessment.html)

Monitoring must be carried out before, during, and after construction of the cap to ensure that the cap was correctly constructed and is effective in isolating the contaminants. The integrity of the cap also must be verified. Guidance on designing and implementing a monitoring program for a capping project can be found in the USACE Waterways Experiment Station Technical Note DRP-5-07 (June 1992; http://www.hsrc.org/capping/monitor-note.html).

Additional detailed information on in situ capping can be found in the following resources:

- The Navy has an in situ capping technology Web page that can be found at http://enviro.nfesc.navy.mil/erb/restoration/technologies/remed/contain_remove/cr-04.asp. This resource provides information about the applicability of in situ capping, its limitations, and schedule and cost considerations.

- The U.S. EPA-sponsored Web site http://www.hsrc-ssw.org/cap-primer.html provides a concise overview of sediment capping technology, including recommended practices and situations where it may be used most effectively. It also includes technical resources for those who are familiar with the technology. It provides technical briefs on site selection considerations, design requirements, equipment and placement techniques, and monitoring considerations. It also provides links to current research on capping.

- U.S. EPA's Assessment and Remediation of Contaminated Sediment (ARCS) Program *Guidance for In Situ Subaqueous Capping of Contaminated Sediments* (Palermo et al., 1998; http://www.epa.gov/glnpo/sediment/iscmain/) contains detailed information about site evaluation, cap design, equipment and placement techniques, monitoring, and management.

In situ capping has been employed at various sites around Puget Sound, WA, including the Eagle Harbor Superfund site where a 3 to 6 ft layer of sand was placed over 54 acres of creosote-contaminated sediments in 40 to 60 ft of water (Sumeri, 1995). Another example is a 3-ft layer of sand that was used to cap 5.7 acres of sediment at Convair Lagoon in North San Diego Bay (Applied Environmental Management, Inc. [AEM], 2000). The cap is bounded by the shoreline and a rock berm on the seaward side.

4.3.3 Dredging Considerations

This section reviews the specific technical and management issues that must be addressed if dredging is likely to be chosen as a remedy. Dredging is relatively easy to implement from a technical and administrative standpoint, although controls may be required to minimize the impact of resuspended particulate matter. Dredging activities should be coordinated with the agencies that regulate navigational dredging (i.e., the regional district of USACE and local port authorities). Although dredging activities associated with on-site CERCLA response actions do not require a permit, all substantive requirements of permits would need to be met (permits are required for offsite activities [e.g., offsite disposal of dredged sediment]). Most dredged sediment requires pretreatment or handling after excavation, including dewatering and primary separation of debris. Dredging must be closely coordinated with subsequent treatment or disposal activities.

4.3.3.1 Selection of an Appropriate Dredging Technique

A number of methods are available for dredging, but most of the experience and equipment have been developed for the improvement and maintenance of navigation facilities (NRC, 1997). The objective of navigational dredging is to remove the greatest volume of sediment as efficiently as possible. By contrast, environmental dredging is a more precise operation in which the objective is to remove contaminated sediment without spreading the contamination to the surrounding environment (Figure 4-3). Discussions of various dredging techniques and their applicability can be found in U.S. EPA (1994a) and Zappi and Hayes (1991). Innovations in dredging technology for contaminated sediments are summarized in the USACE Engineer Research and Development Center (ERDC) Technical Note DOER-T1 (http://el.erdc.usace.army.mil/dots/doer/pdf/doert1.pdf)

Figure 4-3. Enclosed Clamshell Bucket Prevents Escape of Sediment During Environmental Dredging

4.3.3.2 Environmental Windows

Almost every region in the nation is affected by environmental windows, which are specified time periods during which dredging is permitted to minimize the impact on sensitive biological resources. A series of ERDC technical notes (ERDC, 2000b) that address various aspects of environmental windows and effects of dredging on fish and shellfish populations, including the technical aspects of monitoring, can be found at http://el.erdc.usace.army.mil/dots/doer/technote.html. In some cases, an extended or alternative dredging window may be requested if data can be provided to demonstrate that the impact on aquatic biota will be minimal (see Highlight 4-8).

Highlight 4-8. Data Collected to Evaluate Potential Dredging Impacts and Feasibility of Extended Environmental Window, McAllister Point Landfill, Newport, RI

- Densities of ichthyoplankton
- Adult fish surveys
- Water quality information
    - Light penetration
    - Salinity
    - Temperature
- Dissolved oxygen

(Source: Normandeau Associates, Inc., 2000.)

4.3.3.3 Water Column Releases of Contaminants and Use of Silt Curtains

Sediment lost to the water column during remedial dredging is a concern, particularly with mechanical dredging. Contaminants typically are associated with the fine-grained suspended sediment particles, and contamination can be spread by desorption of the contaminants from the particles into the water column, or drifting of suspended sediment beyond the area of remediation.

A variety of containment systems have been used in remedial dredging projects including geotextile and geomembrane silt curtains (see Figure 4-4), floating booms and, in one instance, a steel sheet pile (Cushing, 1999a). Silt curtains have been used with mixed success. Silt curtains were reported to be effective at a number of sites, including the Marathon Battery site in New York (AEM, 2000). However, at other sites the silt curtains obstructed ship traffic, and were easily damaged in strong currents (greater than 1.5 ft per second). At the United Heckathorn site in Richmond, CA, silt curtain management problems, including damage from tugboats and extreme tides, resulted in a loss of 23 days over the life of the project (Chemical Waste Management, 1997).

Coagulants that aid in the settling of suspended particulates also have been used in conjunction with silt curtains. After dredging during the Waukegan Harbor project in Illinois, the harbor water was sprayed with Nalcolyte, a coagulant used for potable water supplies. The coagulant settled the suspended particulates and the silt curtains were removed within 48 hours after application (Canonie Environmental, 1996).

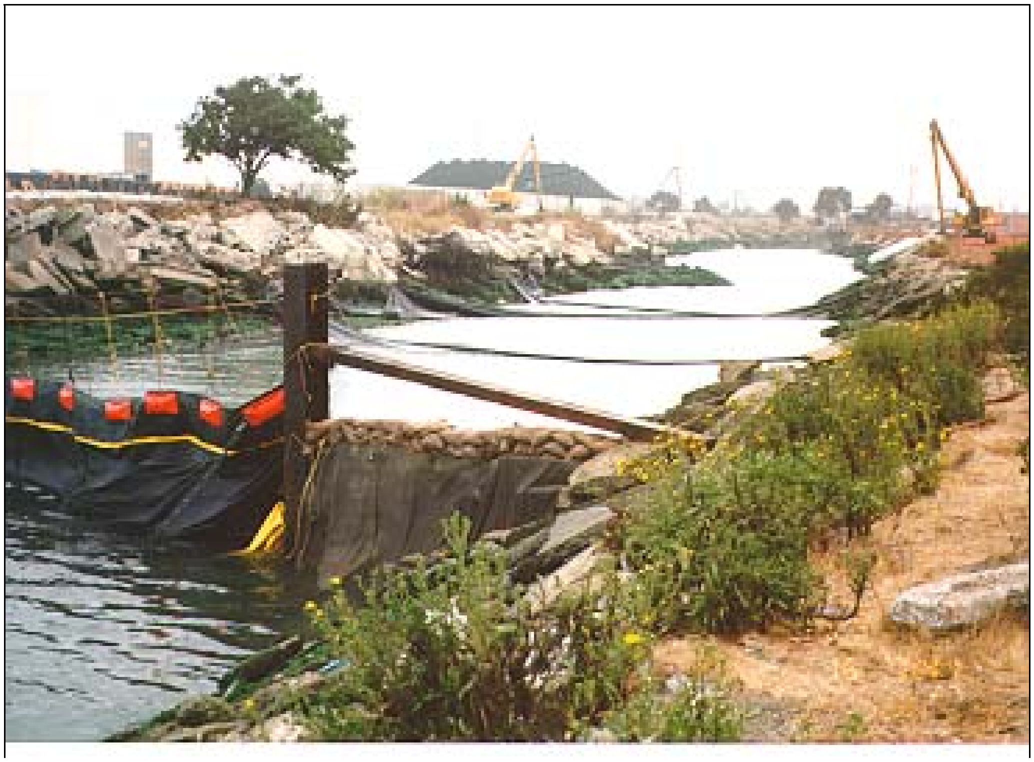


Figure 4-4. Use of a Silt Curtain to Isolate an Area Undergoing Remedial Dredging

Another alternative is dry excavation, which involves the dewatering of the site using coffer dams or otherwise rerouting water temporarily. Dry excavation can provide more accurate contaminant removal with less potential for sediment suspension, but generally at a higher cost than mechanical or hydraulic dredging.

4.3.3.4 Habitat Destruction

Dredging activities have potentially destructive effects on benthos and benthic habitat. Particle size distribution may be altered after surface sediment removal, leading to consequences for the benthos that remain after the dredging or subsequent colonists (Tagatz et al., 1982). Fine sediments from underlying anoxic layers can become resuspended after the dredging event and alter the chemical environment of the habitat (Reimann and Hoffmann, 1991). Freshly exposed anoxic sediments will oxygenate, resulting in the mobilization of several metals that may pose a risk to aquatic organisms. Additionally, dredged areas may be left with a reduced benthic community due to removal of individual organisms with the sediment (Quigley and Hall, 1999).

A literature review of the ecological effects of dredging noted that initial effects can range from negligible to severe and impacts from short to long term, depending on the scale of the project (Morton, 1977). Another study demonstrated that recovery from small scale (<1 m^2) disturbance events can be rapid (within hours) as a result of the immigration of adult individuals from adjacent areas, whereas communities disturbed at the scale of 1 km^2 depend more on the settlement of planktonic larvae for recovery than on the immigration of adults and juveniles (Hall et al., 1994). In the latter case, initial recolonization will vary according to seasonal availability of larval recruits. Recolonization rates are greatest in spring and summer when the densities of ambient species are highest, and lower in the autumn and winter as population densities decreases (Zajac and Whitlatch, 1982). Some of the site-specific data that can be collected to predict the expected habitat loss due to remedial dredging and expected time and modes for recolonization are shown in Highlight 4-9.

Highlight 4-9. Data Needed to Support Evaluation of Habitat Destruction from Remedial Dredging and Time for Recolonization

- Grain size distribution in pre- and post-remediation sediment bed
- Predicted chemical changes in post-remediation environment
- Expected change in light penetration based on change in depth of bed
- Preremediation surveys of aquatic vegetation, invertebrates, and fish
- Analysis of food webs and community structure

A study of benthic invertebrates in a shallow navigation channel in the Columbia River before and after dredging (McCabe et al., 1998) demonstrated the importance of including reference areas in the environmental assessments of dredging projects. Without the data from the reference area, it would not have been possible to make accurate conclusions regarding the impact of the dredging project on the benthos in the channel. Samples collected in the reference area provided a means of assessing natural variation in the standing crops and community structure of benthic invertebrates in a specific reach of the lower Columbia River.

4.3.3.5 Dewatering Requirements

Pretreatment (debris removal, dewatering and possibly stabilization) most likely will be necessary if dredged sediment is destined for upland disposal or beneficial reuse. Some of the highest costs and greatest technical challenges for sediment remediation are associated with pretreatment (Figure 4-5). The type of handling and pretreatment is dependent on the chemical and physical nature of sediment, as well as its final destination. Primary (physical) separation and dewatering are the initial handling requirements. Most sediment requires some dewatering before additional treatment or transportation. Water content can range from approximately 50% by weight with mechanical dredging to more than 80% with hydraulic dredging. Similarly, physical separation of the sediment also may be required before additional handling or treatment. Not only can physical separation be used to concentrate the more contaminated sediment, but it also can help separate debris and large sized sediment particles that could hinder further treatment. Typically, contamination is associated with the silt- and clay-sized particles. If sediments contain a large percentage of sand, separating out the sand can lead to significant savings by reducing the volume that requires treatment. A multitude of process options are available as summarized in U.S. EPA (1994a).

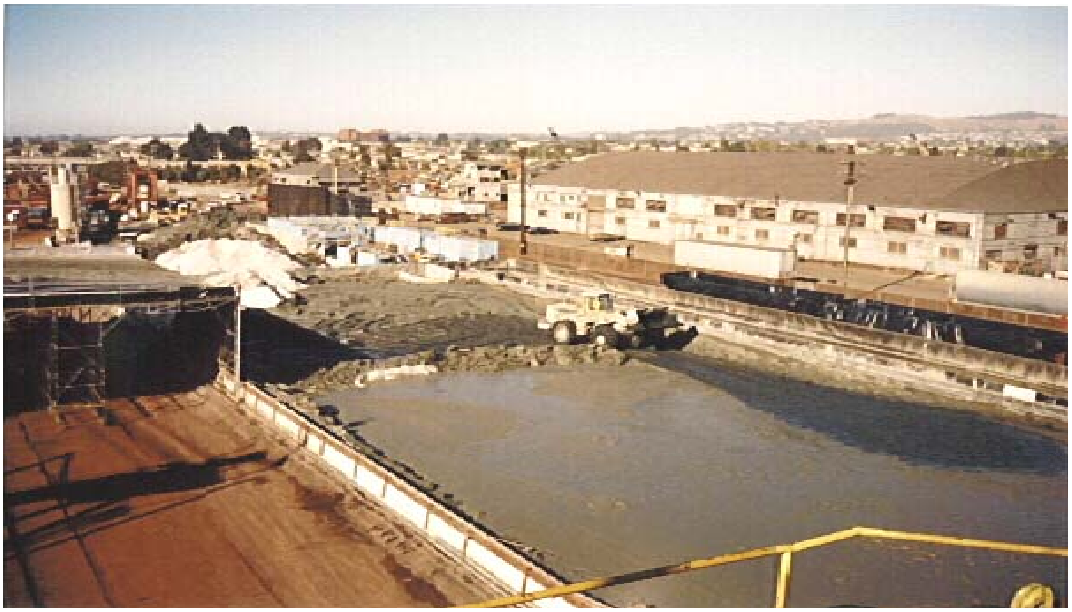


Figure 4-5. Dewatering and Pretreatment Facility for Dredged Sediment

Dredging production rates will vary with dredging method, but the limits on production rate are driven more often by other factors in the process, such as the loading rate of the dewatering facility. Sediments with a lower solids-to-water ratio will require more pretreatment, dewatering, and handling. Because pretreatment areas typically require large land areas to set up, this requirement will affect the applicability of certain dredging methods at a particular site.

Sediment dewatering should reduce the volume of material requiring further handling and disposition, increase the ease of sediment handling and transportation, and reduce the amount of stabilizing agents to be added, if required for final disposition. Dewatering technologies can be broadly divided into two categories: passive and active technologies. A good discussion of dewatering technologies can be found in U.S. EPA (1994a). Some commonly used dewatering technologies are shown in Highlight 4-10.

Highlight 4-10. Commonly Used Dewatering Technologies

- Spreading and natural evaporation; drying agents (e.g., lime) may be added to promote drying
- Placement in geotextile bags
- Plate and frame compression
- Centrifugation (cyclone separation)

4.3.3.6 Residual Surface Sediment

Although some precision dredging techniques are available, it is inevitable that some residual sediment will remain after remediation is complete, because of the technological limitations of dredging technologies and the redeposition of sediment that was suspended during dredging. In particular, obstructions such as piles, piers, and other shoreline structures; the presence of large debris; and a convoluted shoreline may interfere with dredging and increase the amount of residual surface sediment (see Figure 4-6). The potential for residual surface sediment and associated degree of risk should be considered in the FS, particularly with respect to achieving remedial goals via dredging and verification requirements.

Figure 4-6. Residual Sediment Around Pilings Can Reduce the Effectiveness of Dredging

4.3.3.7 Cost

The cost for environmental dredging is substantially higher than navigational dredging because of the greater precision of cut required, the slower production rates to minimize resuspension, multiple passes needed to achieve cleanup goals, use of contaminant barriers, and restrictions posed by other remedial components such as pretreatment and handling (NRC, 1997). Poor three-dimensional characterization of the extent of contamination can result in the need for multiple passes or over-dredging, which can increase disposal volumes and costs. The major cost elements for dredging are as follows:

- Mobilization/demobilization;
- Dredge method;
- Sediment containment barriers; and,
- Monitoring.

Additional cost considerations include dredged material handling, pretreatment, and treatment or disposal. Additional handling and/or treatment costs after excavation can add more than \$100/yd^3 depending upon the approach.

In general, larger sediment excavation volumes lead to lower cost per unit volume. Conversely, small dredging projects will involve a high cost per unit volume. Overall costs for recent remedial dredging projects range from \$44 to \$1,842 per yd^3 (Cushing, 1999b). The high overall cost is due to two primary factors: low production rates relative to navigational dredging, and disposal costs. As noted above, inadequate site characterization can result in higher-than-necessary volumes.

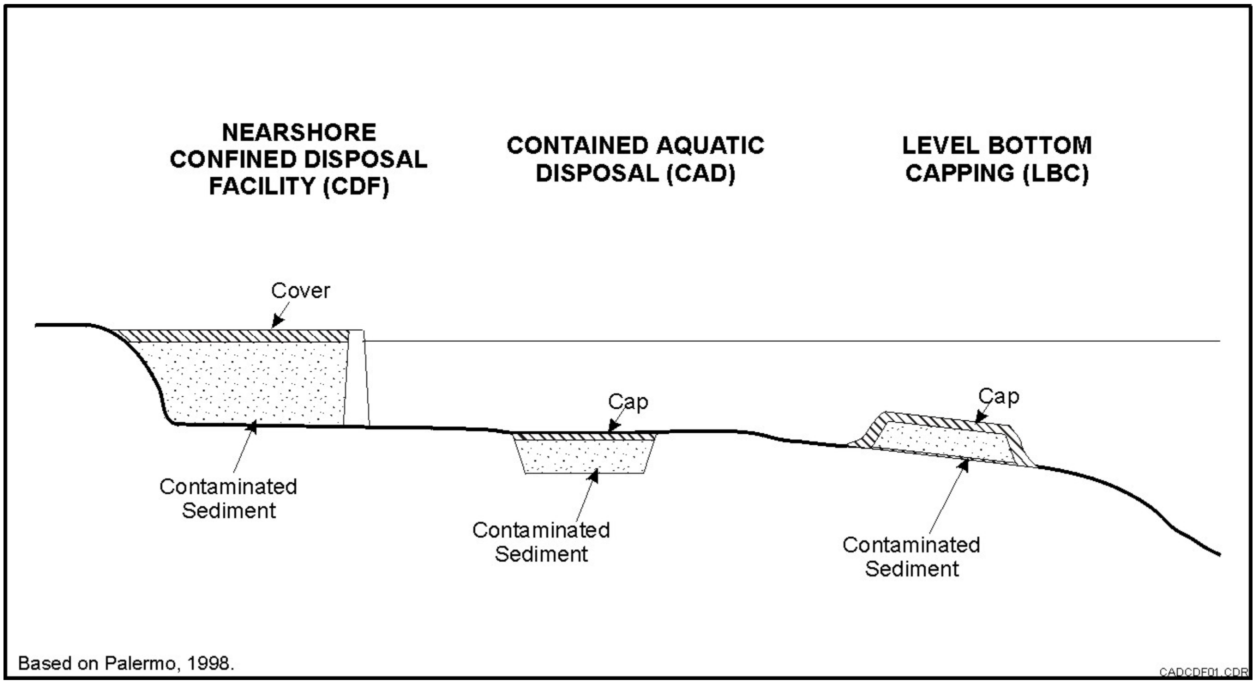
4.3.4 Sediment Disposal Options

Generally, contaminated sediments are unsuitable for unconfined open water disposal and require containment or treatment after dredging. Containment is the most common approach for management of dredged sediments and has been widely applied. More recently, development of beneficial reuse options for dredged material has become a priority for the USACE and port authorities. Although the emphasis has been on reuse of clean material dredged for navigational purposes, some of the reuse options may be feasible for contaminated material (see Section 4.3.6).

Containment technologies for dredged sediment include contained aquatic disposal (CAD), including level-bottom capping (LBC); confined disposal facilities (CDF); and upland disposal in landfills. Figure 4-7 is a conceptual drawing of various containment options. The use of geotextile bags is another option for disposal of small volumes of sediment. Each containment technology is summarized below. The applicability, advantages and disadvantages and cost and schedule considerations for various disposal technologies are summarized in Section 4.3.9.

4.3.4.1 Contained Aquatic Disposal

CAD involves the dredging of contaminated sediments from areas to be remediated, transport to the underwater disposal site, controlled placement on the seafloor, and capping. CAD is similar to in situ capping, except that a cap is placed over dredged sediments rather than on the original seafloor. Dredged sediments can be placed in a natural or excavated depression or contained within berms, then covered with clean sediment or other capping material. The cap prevents physical contact



Based on Palermo, 1998.

Figure 4-7. Conceptual Drawing of Various Containment Options for Dredged Sediment

between the sediments and the benthic community, prevents sediment resuspension and dispersion, and may inhibit contaminant flux to the water column. As with all capping alternatives, the site must be monitored for cap integrity over a long period of time. Technical considerations associated with a CAD include location of a suitable site, selection of appropriate placement methods for sediments and cap, long-term integrity of the cap, and habitat restoration potential, if applicable.

The first CAD project completed in the United States involved the placement of sediments containing metals and PCBs in a depression in the lower Duwamish River in Seattle, WA and construction of a clean sand cap (USACE et al., 1999). Large CAD pits (up to 1.5 million yd^3) have been excavated in New Jersey and Massachusetts for disposal of contaminated sediments from navigational dredging projects (Knoesel et al., 1998; Murray et al., 1998). However, CAD has not been widely applied exclusively for disposal of contaminated sediments.

Design, feasibility and implementation of a CAD are discussed in various technical documents, including *Guidance for Subaqueous Dredged Material Capping* (http://el.erdc.usace.army.mil/dots/doer/pdf/doerc21.pdf). Current information on the recent construction of CADs in Boston Harbor can be found at http://www.nae.usace.army.mil/projects/ma/BHNIP/cover.pdf.

4.3.4.2 Confined Disposal Facilities

Confined disposal involves the placement of dredged material in a nearshore area behind a dike, berm, or other containment structure. Most CDFs use the natural shoreline as part of the containment structure, with a dike or berm constructed in the water to complete the enclosure (USACE et al., 1999) (Figure 4-7). Dredged sediments are placed in the CDF, dewatered, and consolidated. Effluent from dewatered sediments must be monitored and treated if necessary before discharge. The filled CDF then can be capped with clean material and either vegetated or paved and made available for a variety of end uses. The Navy's technology Web page for CDFs, which can be found at http://enviro.nfesc.navy.mil/erb/restoration/technologies/remed/contain_remove/cr-02.asp, provides more information on this topic.

Potential contaminant releases from the CDF can be controlled using appropriate design, materials, and operational procedures. Seepage of leachate through the CDF walls is most effectively prevented by use of clay or bentonite-cement slurries as sealers (U.S. EPA, 1993). A CDF requires long-term monitoring and maintenance to ensure its integrity and verify its effectiveness in isolating contaminants.

The USACE uses CDFs to contain about 30% of the sediments produced by navigational dredging programs (U.S. EPA, 1993). A large CDF (greater than 2 million yd^3) was constructed in the Milwaukee Waterway in Commencement Bay, WA, to contain contaminated sediments and clean navigational dredged material (Verduin et al., 1994). A berm was constructed across the mouth of the waterway, and sediments were either offloaded from a barge or placed directly by pipeline from the dredged area. Effluent control was provided by an overflow weir and discharge line. After dewatering, settling, and consolidation, the CDF was capped with paving and converted to a container terminal.

In addition, existing piers or other port structures can be used to provide partial containment for a CDF. In Elliot Bay, the Port of Seattle constructed two berms between solid fill piers to form a CDF (Converse Consultants et al., 1992). At the Outboard Marine site in Lake Michigan, a double sheetpile cut off wall was constructed across an existing slip (AEM, 2000; http://www.hudsonvoice.com/mcss/). A clay slurry wall also was constructed to form a watertight barrier. The CDF was used to contain 50,000 yd^3 of hydraulically dredged sediments, which took 2.5 years to dewater and consolidate prior to placement of the final cap.

4.3.4.3 On-Site Upland Disposal

On-site upland disposal refers to the placement of dredged sediment in a containment facility on or beneath surface lands that are typically near the remediation site. Depending on the level of contamination in the sediments, the sediments may be placed directly on the land surface after dewatering, or they may be placed in a fully designed and engineered RCRA facility. For example, at Point Potrero in Richmond, CA, dredged sediments contaminated with metals and PCBs were placed on an adjacent upland area and capped with asphalt when dewatering was complete (San Francisco Bay Regional Water Quality Control Board [SFBRWQCB], 1999). For on-site upland disposal, sediments are dredged and transported to a pretreatment area where they are dewatered and, depending on the nature of contamination present, stabilized to reduce contaminant mobility and increase load-bearing capability. The treated sediments then are transported to either an engineered cell or a land parcel, where they are compacted, graded, and fixed in place by a vegetative cover.

4.3.4.4 Commercial Landfill Disposal

Commercial landfill disposal of dredged sediment is generally expensive because of the associated transportation and disposal costs. However, it can be a good option for disposal of small volumes of highly contaminated sediment, especially if remediation must be completed in a short time frame. Although not usually cost-effective, commercial landfills have been used for the disposal of relatively large volumes of sediment that were transported a great distance from the site. For example, approximately 100,000 cubic yards of dewatered sediment from the United Heckathorn Site in

California were transported by rail to landfills in Arizona and Utah for disposal (Chemical Waste Management, 1997). An appropriate off-site landfill should be identified based on the ability to meet the landfill's waste discharge requirements, confirmation of sufficient landfill capacity and evaluation of disposal costs.

4.3.4.5 Geotextile Bags

Geotextile bags are large sacks that can be filled with dredged sediment (Figure 4-8). They are usually constructed of a nonwoven, felt inner liner surrounded by a coarser, nonwoven outer bag. The fabric retains the sediment while allowing water to seep through so that the sediment will dewater under its own weight. The sediment-filled bags are easier to handle and transport than loose sediments in addition to providing some confinement of the sediment. In addition to dredged material, geotextile bags have been used to contain sewage sludge, animal waste, fly ash, potash lagoons, drilling mud and cuttings, and mine tailings.

Because most contaminants are sorbed to sediment particles and do not seep through the fabric of the bag, underwater placement of filled geotextile bags might be environmentally safe and would eliminate the need for a land-based disposal site (NRC, 1997). However, the leachate would need to be recovered and tested to verify that chemical concentrations are acceptable. Although filled geotextile bags have been placed in the aquatic environment for wetlands creation and reef construction, case studies of their use for underwater disposal of contaminated sediments were not found.

Figure 4-8. Geotextile Tube Filled with Dredged Sediment
(Used with permission from Ellicott Division of Baltimore Dredges, LLC, Baltimore, MD 21230)

4.3.5 Sediment Treatment Options

Contaminated sediment can be treated using physical, chemical, biological or thermal technologies, either in-place or after dredging and pretreatment. However, treatment of contaminated sediments has been applied at very few sites and is still very expensive. Treatment to destroy or immobilize contaminants in sediment either in situ or ex situ is complicated by the factors identified in Highlight 4-11.

Relatively few cost-effective technologies effectively treat contaminants in sediment, and the treatment technologies that currently exist have not been widely applied. However, sediment treatment is an active field of research and several promising technologies are under development.

A good summary of selected treatment technologies for sediment is provided in *Innovative Technologies For Decontamination and Treatment of Dredged Material* (WES, 2000b; (http://el.erdc.usace.army.mil/dots/doer/pdf/doert2.pdf). This paper provides information on some of the more promising treatment technologies for sediments, including a variety of thermal processes

Highlight 4-11. Factors that Reduce the Viability of Sediment Treatment Options

- Sediments typically are contaminated with more than one class of contaminant
- Sediments are relatively inaccessible beneath surface water
- Compared with terrestrial soils, contaminated sediments are characterized by relatively large volumes with relatively low contaminant concentrations
- Fine-grained sediments with a high organic content tend to adsorb and bind chemicals, thereby inhibiting many treatment options
- Sediments typically are difficult to handle and process because of the high moisture content and predominance of fine-grained material
- The chemistry associated with treatment technologies is complex, requiring extensive treatability studies
- Residual waste streams such as cleaned solids, concentrated residuals, and off-gas require management
- Cleanup criteria are generally orders of magnitude lower for sediments than for soils, pushing the limits of even the most standard technologies
- Successful delivery of amendments for in situ approaches is problematic: the addition and successful delivery of amendments to sediment in situ may adversely affect water quality

and containment or partial removal processes. The table of existing and emerging remedial technologies presented in Section 4.3.9 includes information on a variety of treatment technologies including phytoremediation, solidification and stabilization, biological treatment, direct injection, use of gas permeable membranes, sediment washing, and thermal technologies.

In situ treatment of sediment is an identified focus area of the Remediation Technologies Development Forum (RTDF) (www.rtdf.org/public/sediment/default.htm). A number of in situ remediation technologies are under consideration by the RTDF's Sediment Remediation Subgroup, including natural attenuation, phytoremediation, introduction of chemical additives to enhance the natural processes, and electrokinetics. The subgroup is most interested in passive technologies that will remediate contaminants without significantly increasing the stress on the ecology.

4.3.6 Beneficial Reuse

Beneficial reuse refers to the use of dredged material as a resource in construction or environmental restoration projects. Interest has developed over the last 25 years in improving dredge management practices including beneficial reuse of dredged material (National Dredging Team, 1998; USACE, 1987b). U.S. EPA and USACE define ten broad categories of beneficial uses based on the functional use of the dredged material or site (see Highlight 4-12). Although the emphasis has been

Highlight 4-12. Potential Beneficial Uses for Dredged Sediment

- Habitat restoration/enhancement (wetland, upland, island, and aquatic sites, including use by waterfowl and other birds)
- Beach nourishment
- Aquaculture
- Parks and recreation (commercial and noncommercial)
- Agriculture, forestry, and horticulture
- Strip mine reclamation and landfill cover for solid waste management
- Shoreline stabilization and erosion control (including fills, artificial reefs, and submerged berms)
- Construction and industrial use (including port development, airports, urban, and residential)
- Material transfer (including fill, dikes, levees, parking lots, and roads)
- Multiple purpose

(From U.S. EPA and USACE, 1992.)

on the reuse of clean material dredged for navigational purposes, some of the reuse options may be feasible for contaminated material.

The feasibility of beneficial reuse as a remediation option depends upon the level of contamination in sediment and the proposed reuse. Sediment with low to moderate levels of contamination might be suitable for some applications such as foundation (noncover) material for habitat restoration projects. Additionally, some limited treatment such as the addition of cement kiln dust or fly ash could improve the engineering characteristics of lightly contaminated material and render is useable as landfill cover or construction fill. RPMs should investigate local or regional opportunities for beneficial reuse, and consult with regional dredging authorities regarding potential reuse opportunities that may have been developed for material that is classified as not suitable for open water disposal.

4.3.7 In Situ vs. Removal Responses

As noted in Section 4.3.3, most sediment cleanups to date have involved dredging. Dredging is generally considered to be a permanent remedy for a site because it eliminates long-term liability for the remediated site and places no limitations on its future use. However, dredging may not be the most appropriate solution for some sites because of disproportionate costs and unacceptable environmental impacts. In particular, dredging might not be suitable for sites that have a large area affected by low or moderate contamination (both laterally and vertically) or are located in nearshore or wetland areas that provide valuable habitat. In these cases, in situ management may be a more cost-effective and viable option.

In some cases, sites that have undergone remedial dredging have actually experienced negative ecological impacts. Contaminants can be incompletely removed and/or made more bioavailable by the exposure of more highly concentrated, less-weathered contaminants that were previously buried (Thibideaux et al, 1999). Additionally, the change from an anoxic to an oxic environment may remobilize some contaminants. However, dredging may be the only viable option in some circumstances. Site characteristics that indicate whether an in situ response or dredging may be more appropriate are summarized in Highlight 4-13.

Highlight 4-13. In Situ vs. Removal Remedies

Site characteristics that support an in situ response:

- Large area or volume of sediment affected by relatively low levels of contamination
- Contaminated sediment located in an area with sensitive/valuable habitat
- Contaminated sediment located in a quiescent area with low resuspension and erosion potential

Site characteristics that support a dredging response:

- Limitations cannot be placed on the future use of the site
- Institutional controls (e.g., fishing advisories) are not feasible
- Human health or ecological risk is immediate or substantial and capping is not feasible
- Contaminated sediments are located in a dynamic environment with high resuspension and erosion potential
- Insurmountable public or regulatory resistance to an in situ remedy

4.3.8 Risks Inherent in Each Remedial Alternative

The short- and long-term risks associated with any remedial alternative are identified and assessed as part of the nine criteria evaluation set forth in the NCP (U.S. EPA, 1988; http://www.epa.gov/oerrpage/superfund/whatissf/sfproces/rifs/analys.htm). The long-term effectiveness and permanence criterion evaluates the potential magnitude of the residual risk and the adequacy and reliability of the controls implemented to manage the risk. The short-term effectiveness criterion addresses: (1) protection of the community and workers during remedial actions, (2) environmental impacts during remedy implementation, and (3) time until RAOs are achieved. The long- and short-term risks associated with some of the major remedial technologies for sediments and associated management strategies are summarized in Tables 4-2 through 4-5. These tables do not address community or worker safety during implementation of a remedy because it is assumed that these risks can be easily identified and addressed in health and safety plans.

In addition to long- and short-term risks associated with a given remedial strategy, the RPM must consider the risk and cost associated with assuming long-term liability for the site. Any remedial action that results in on-site containment of contaminated sediment (e.g., natural recovery, in situ

Table 4-2. Risks and Management Strategies Associated With Dredging

Long-Term Risks	Management Strategies
Residual sediment not removed by dredging	Use precision dredging techniques, minimize sediment resuspension during dredging, remove obstructions such as pilings and debris before dredging, conduct post-remediation monitoring
Permanent destruction of habitat and biological resources	Consider restoration
Recontamination by uncontrolled sources	Complete source identification and control before remediation
Short-Term Risks	**Management Strategies**
Benthic habitat destruction	Evaluate expected recolonization time and promote recovery if necessary (e.g., provide suitable substrate)
Sediment resuspension/dispersal during dredging and release of contaminants to water column	Use of environmental dredging techniques, silt curtains or other barriers, and coagulants; adherence to environmental windows

Table 4-3. Risks and Management Strategies Associated With In Situ Capping and CAD

Long-Term Risks	Management Strategies
CAD cap disrupted by natural or human activity	Ensure that proper site selection and design criteria are met, implement institutional controls as needed, long-term monitoring and maintenance of CAD site
Upward diffusion of contaminants through cap	Ensure that proper design criteria are met, long-term monitoring and maintenance of CAD site
Short-Term Risks	**Management Strategies**
Loss of contaminated sediment during placement operations	Use of precision placement techniques and silt curtains or other barriers
Disruption and displacement of contaminated sediments by cap material	Ensure sufficient consolidation of contaminated sediments before placement of capping material, use proper placement techniques for cap

Table 4-4. Risks and Management Strategies Associated With CDFs

Long-Term Risks	Management Strategies
Escape of sediment or leachate from containment structure	Ensure that design criteria are met, long-term monitoring and maintenance of CDF
Short-Term Risks	**Management Strategies**
Environmental impact at disposal site from construction of CDF	Use construction techniques and materials that minimize impact
Loss of contaminated sediment during transport and placement in CDF	Use equipment and techniques that ensure containment of sediment

Environmental impact of effluent from dewatering	Proper control and treatment of effluent before discharge to receiving waters
Remobilization of some contaminants if moved from anoxic to oxic environment	Ensure proper containment of sediment and effluent and treat as necessary

Table 4-5. Risks and Management Strategies Associated With Upland Disposal

Long-Term Risks	Management Strategies
Escape of sediment or leachate from containment structure	Ensure that design criteria are met, long-term monitoring and maintenance of landfill site
Short-Term Risks	**Management Strategies**
Escape of sediment during transport, dewatering and placement	Use equipment and techniques that ensure containment of sediment
Escape of contaminants in effluent from dewatering sediment	Proper control and treatment of effluent prior to discharge

capping or dredging followed by on-site disposal in a CAD, CDF, or landfill) will in most cases result in long-term monitoring responsibilities and liability issues.

4.3.9 Table of Existing and Innovative Remedial Technologies

The management of sediments is best handled through the integration of risk management and remediation. This will often entail a systems engineering analysis or decision analysis approach to optimize the results and obtain the overall goal of protection of human health and the environment. Table 4-6 presents a summary of existing and emerging remedial technologies for sediments. These technologies can be used alone or in combination. Typically, more than one technology or approach must be applied to achieve the goals for a site. For example, precision dredging of hot spots may be coupled with in situ capping or monitored natural recovery. If sediments are dredged, a number of alternatives are available for handling and treating the sediment. Table 4-6 summarizes the applicability, strengths, weaknesses, and associated costs for each technology.

4.4 MONITORING CONSIDERATIONS

Monitoring must be conducted before, during, and after a remedial action at a sediment site. Baseline monitoring helps define existing conditions and provides the data for meaningful comparisons once the remedy is complete. Often, the relevant data will have been previously collected as part of the human health and ecological risk assessments and will be sufficient to establish baseline conditions. If baseline conditions are to be used as a basis for direct comparison, it may be necessary to collect data over a period of years to obtain adequate representation of seasonal variability.

Short-term monitoring is required to monitor environmental impacts during remedy implementation and ensure that target cleanup levels have been achieved. Short-term monitoring parameters could include suspended sediment and dissolved oxygen levels in the water column during dredging or capping. Post-remediation sediment sampling also may be required to verify that all sediment above cleanup levels has been removed or contained.

After the remedy has been implemented, a long-term monitoring program must be carried out according to a systematic schedule to verify the effectiveness of the remedy. The U.S. EPA, in partnership with the Navy, is preparing a long-term monitoring guidance document that will provide a framework for developing scientifically defensible monitoring plans with clear exit criteria. The monitoring plan should be developed in conjunction with the remedial action plan. The most critical element in the plan is the link between the performance of the remedy and the RAOs. Long-term

Table 4-6. Summary of Existing and Innovative Remedial Technologies for Sediment

Technology Type	Description/ Applicability	Chemicals Used in Process	Major Advantages	Major Disadvantages	Relative Cost*	Cost and Schedule Considerations	References/ Resources
In Situ Actions							
Monitored natural recovery	The degradation or isolation of contaminants primarily through burial that occurs from natural sedimentation (deposition)	None	Reasonable alternative if dredging is not desirable or feasible. Can be cost-effective, preserves habitat, nonintrusive implementation.	Not appropriate for all site settings. Extensive characterization required to validate approach. Site should be in an area of deposition and not subject to major erosional events (i.e., storm erosion, prop scour)	4 (Characterization and monitoring costs highly variable)	Extensive upfront characterization required to support decision. Long-term monitoring required. Characterization and monitoring costs can be significant.	Huggett and Bender, 1980; Hahnenberg, 1995. National Research Council, 1997 www.smwg.org/
In situ capping	Placement of cover material over sediments to physically isolate sediments	None	Eliminates need to remove contaminated material; minimizes contaminant release to the environment that could occur with dredging; relatively easy to implement.	Cap with incompatible bottom material can alter benthic community. Long-term maintenance typically required. Potentially can limit future site uses. Politically not appropriate for all site settings.	3	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997; Palermo et al. 1998 www.epa.gov/glnpo/sediment/iscmain/
Confined Disposal Facility (CDF)	Placing dredged sediments within near-shore disposal facility	Typically none; could augment with bioamendments or stabilizers	Low cost compared to ex situ treatment or off-site disposal. Involves conventional equipment. Site can be used for beneficial purposes following closure with proper safeguards.	Can alter benthic community through dredging. Does not destroy or detoxify contaminants unless combined with treatment. Near-shore land may be difficult to find if wetlands or critical ecological niches would be lost.	2	Conventional engineering approaches used. Costs equipment and procedures are available from other sites.	USACE, 1987a; National Research Council, 1997; http://el.erdc.usace.army.mil/dots/doer/ U.S. EPA, 1994a; http://www.epa.gov/glnpo/sediment/iscmain/index.html
Contained Aquatic Disposal (CAD)	Moving sediments to a natural subaqueous topographic low or constructed depression and capping.	Typically none; could augment with bioamendments or stabilizers	Eliminates need for ex situ handling. Segregates contaminants into one location. Minimizes institutional controls at site such as limits on prop wash, navigational depths, etc. and adds flexibility in use of site.	Can alter benthic community through dredging and relocating. Long-term maintenance typically required for cap. Potential loss of contaminants to water column during movement requires controls. Politically not appropriate for all site settings.	2	Costs, equipment, and procedures are available from other sites.	Price and Lee, 1999 http://el.erdc.usace.army.mil/dots/doer/ (DOER-C11, 18 DOER-N5) Garbaciak and Miller, 1995 National Research Council, 1997 http://www.epa.gov/glnpo/sediment/iscmain/index.html

Table 4-6. Summary of Existing and Innovative Remedial Technologies for Sediment (page 2 of 6)

Technology Type	Description/ Applicability	Chemicals Used in Process	Major Advantages	Major Disadvantages	Relative Cost*	Cost and Schedule Considerations	References/ Resources
Phyto-remediation	Removal of contaminants from sediment through plant uptake. Can be applied to submerged sediments, but is considered more useful for upland application after sediment removal.	Plant amendments	Uses natural processes to promote cleanup. Potential cost savings over other sediment handling options. If used in situ may provide alternative to removing sediment by dredging.	Relatively immature technique. Requires extensive screening to determine if applicable. Considered most useful in upland applications which in turn require dredging. Plants must be maintained and for some contaminants (e.g., metals) must be harvested periodically for disposal.	Limited costing data.	Relatively immature; limited data available. Requires significant characterization and testing to determine suitability.	Price and Lee, 1999 http://el.erdc.usace.army.mil/dots/doer/ Cunningham and Lee, 1995
Hydraulic modifications	Physical alteration of bottom or water body to control movement of contaminated sediments and promote deposition of clean sediments	None	Controls contaminated sediment migration, promotes natural attenuation through deposition. Bottom alterations not as drastic as dredging. Sediment can be contained and physically separated.	Relatively new technique; has been applied to fluvial systems, not yet to marine. Modification of ecological habitat, may limit use of site by construction of barriers.	Limited costing data.	Applied to three locations (Moss American, Tennessee River, Mississippi River) set precedent on cost and schedule.	www.smwg.org/
Solidification/ stabilization	Agents injected directly into sediments to solidify or stabilize	Cement, fly ash	Immobilizes contaminants in place. Stabilizes sediments preventing erosion. Eliminates need to remove contaminated material.	Limited long-term testing data on sediments. Limits future site uses.	2	Relatively immature; limited data available; few studies documented.	National Research Council, 1997
In Situ Treatment							
Biological treatment	Fostering microbial biodegradation by providing amendments.	Oxygen, electron receptors, nitrogen	Treats sediments in place through reduction of toxicity. Uses natural processes to promote cleanup.	Subaqueous environments are difficult to manipulate. There are unresolved microbial, geochemical and hydrological issues because technology in the marine environment is relatively immature.	Limited costing data	Relatively immature, few studies documented for marine sediments, freshwater sediment are limited to pilot scale.	National Research Council, 1997; Abramowicz et al. 1992; Bragg et al. 1994. Harkness et al., 1993
Direct injection	Inject reactants for abiotic treatment. Tines can be driven into sediment for the injection of reactants	Ozone, hydrogen sulfide, potassium permanganate	Detoxifies or immobilizes contaminants in place.	Relatively immature technique, sediments typically have multiple contaminants making it difficult to address all contaminants. Subaqueous environments are difficult to manipulate.	2	Relatively immature, few studies documented for marine sediments. Increases uncertainty in cost and schedule.	National Research Council, 1997 Vendor: http://www.oceta.on.ca/profiles/limnofix/list.html

Table 4-6. Summary of Existing and Innovative Remedial Technologies for Sediment (page 3 of 6)

Technology Type	Description/ Applicability	Chemicals Used in Process	Major Advantages	Major Disadvantages	Relative Cost*	Cost and Schedule Considerations	References/ Resources
Gas-permeable membranes	Reactants delivered to sediments via gas permeable membrane (tubing)	Ozone, oxygen, hydrogen, hydrogen sulfide	Detoxifies or immobilizes contaminants in place. Provides long-term sustained delivery of reactants with minimal energy input. Potential for significant cost savings.	Has not been demonstrated on sediments.	Limited costing data	Undeveloped technology increases uncertainty in cost and schedule	Gilmore and Oostrom, 1999
Ex Situ Actions							
Mechanical dredging	Direct application of mechanical force to remove sediment	Not applicable	Little residual risk at site. No limitations on future use. Established industry supporting dredging. Entrains less water than hydraulic dredging.	Destroys benthic community. Potential water column impacts from resuspended sediment. Sediment must be handled, after dredging. Can remove more sediment than required (lack of precision).	2 (Includes removal and transportation; additional handling or treatment not included)	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997 http://www.smwg.org/
Hydraulic dredging	Removal of sediment slurry by suction	Not applicable	Little residual risk at site. No limitations on future use. Established industry supporting dredging. Can be used for more precision removal; limiting sediment. Less suspended sediment released to water column.	Destroys benthic community. Sediment must be handled after dredging. Water content of dredged sediment is increased over mechanical. Sediment most be predominantly fine-grained to be handled by hydraulic dredge.	2 (Includes removal and transportation; additional handling or treatment not included)	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997 http://www.smwg.org/
Ex Situ Treatment (Requires Dredging)							
Primary separation	Removal of large debris from sediment	Not applicable	Increases ease of handling for dewatering, treatment and/or disposal.	Can be technically challenging and time-consuming. Debris requires disposal.	2	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997
Dewatering	Removal of free water from saturated sediments	Chemicals typically not used; however some methods use adsorbents.	Reduces volume of sediment requiring disposition. Increases ease of handling. Decreases disposal and/or treatment costs.	Passive/gravity dewatering requires significant time and space to be effective. Mechanical dewatering can be technically challenging and more expensive. Excessive fines in sediment can decrease effectiveness.	2	Dewatering takes time but decreases overall costs. Costs equipment and procedures are available from other sites.	National Research Council, 1997
Sediment washing	Removal of contaminants from sediment using a wash solution	Can include detergents or solvents	Contaminants are removed from sediments increasing the potential uses of the sediment and minimizing disposal costs.	Ineffective with fine-grained sediment. Wash solution difficult to formulate for contaminant mixtures. Contaminant removal is incomplete.	2	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997; USACE, 1994a

Table 4-6. Summary of Existing and Innovative Remedial Technologies for Sediment (page 4 of 6)

Technology Type	Description/ Applicability	Chemicals Used in Process	Major Advantages	Major Disadvantages	Relative Cost*	Cost and Schedule Considerations	References/ Resources
Physical separation	Various methods to physically separate the sediment for more efficient treatment and/or disposal ▪ Screens and classifiers ▪ Hydrocyclones ▪ Gravity separation ▪ Froth flotation ▪ Magnetic separation	Typically none; some processes may use foaming agents	Can be used to reduce volume of sediment requiring treatment or disposal. Results in lower handling costs.	Does not destroy contaminants but concentrates them into smaller volumes. Not suitable for all sediments; typically requires a significant coarse fraction (≥25% sand-size).	1	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997
Solidification/ stabilization	Reduction of contaminant mobility by addition of a binding agent	Cements, plasticizers	Can use standard slurry mixing equipment. History of use for sludge. Relatively inexpensive.	Long-term effectiveness has not been demonstrated. May result in a significant increase in volume. May not be appropriate for contaminant mixtures. Moisture content must be relatively low	2	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997
Chemical separation and thermal desorption	Using heat and chemicals for mobilizing contaminants into a fluid or gas phase for collection and deposition.	Acid or base solutions for leaching. Solvents.	Removes contaminants from sediments for easier disposal.	Has limited application to sediments.	1	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997; USACE, 1993, 1994b
Phyto-remediation	Removal of contaminants from sediment through plant uptake.	Plant amendments	Potentially less expensive than other methods.	Relatively immature technique. Requires extensive screening to determine if applicable. Plants must be maintained and for some contaminants (e.g., metals) be periodically harvested for disposal.	Limited costing data	Relatively immature; limited data available; requires significant characterization and testing to determine suitability.	Price and Lee, 1999 http://el.erdc.usace.army.mil/dots/doer/pdf/doerc3.pdf Cunningham and Lee, 1995
Biological treatment	Degradation of contaminants by microorganisms. Land farming most common technique	Bioamendments	Degrades organic pollutants, public acceptability, relatively low cost.	Handling of material is substantial. Method is time consuming, and adequate space is required. Not suitable for some contaminants such as metals. Does not remove 100% of contaminants	1	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997; Thoma, 1994

Table 4-6. Summary of Existing and Innovative Remedial Technologies for Sediment (page 5 of 6)

Technology Type	Description/ Applicability	Chemicals Used in Process	Major Advantages	Major Disadvantages	Relative Cost*	Cost and Schedule Considerations	References/ Resources
Incineration	Thermal decomposition at high temperatures	None	Destroys a large proportion of organic contaminants. Mature, commercially available technology. Has been used to treat a wide variety of hazardous wastes and sediment.	Dewatering needed to reduce energy consumption. Heavy metals remain in a bottom ash or fly ash. Off-gas requires treatment. Potentially expensive. Volatile metals (Hg, As, Se, Pb) may be emitted to the atmosphere in gaseous forms.	0	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997
Thermal desorption	Heating and agitating sediment to volatilize water and organic contaminants	None	Has been applied to sediments.	High temperatures required to remove persistent organic contaminants (e.g., PCBs). High clay, humic material, or moisture content increases costs.	0	Costs, equipment, and procedures are available from other sites.	National Research Council, 1997
Disposal							
On-site upland disposal	Placement of dredged material in an upland containment structure adjacent to dredged site	None	Contaminants effectively contained. Lower transportation costs relative to off-site disposal.	Dewatering and pretreatment generally is required. Long-term integrity of disposal site must be maintained. Potential for long-term liability.	2-3 Depending on waste classification	Permitting facility is on critical path. Costs, equipment, and procedures are available from other sites.	National Research Council, 1997
Off-site upland disposal	Disposal of dredged material at permitted landfill	None	Readily available. Well-established approach to sediment management.	Dewatering and pretreatment generally is required. Relatively high cost. Transport required.	2-3 Depending on waste classification	Effective transportation is important. Straightforward costing and scheduling.	National Research Council, 1997
Open water disposal	Placement of dredged material at designated aquatic disposal site	None	Well-established dredged material management program. Availability of disposal site.	Contaminated sediments are not likely to be suitable for open water disposal.		Requires open water disposal area.	National Research Council, 1997
Beneficial Reuse							
Wetland creation	Use of sediment for wetland habitat restoration or enhancement	None	Precedent exists for wetland creation. Anticipated available capacity in wetland creation projects. Greater chance of public acceptance.	Some sediments may not meet acceptance criteria for wetland creation.		Work with other agencies contributes to uncertainty in scope and budget.	http://sfbay.wr.usgs.gov/access/saltponds/index.html Price and Lee, 1999 http://el.erdc.usace.army.mil/dots/doer/pdf/doerc3.pdf

Table 4-6. Summary of Existing and Innovative Remedial Technologies for Sediment (page 6 of 6)

Technology Type	Description/ Applicability	Chemicals Used in Process	Major Advantages	Major Disadvantages	Relative Cost*	Cost and Schedule Considerations	References/ Resources
Levee/dike construction	Use of dredged sediment for construction or repair of levees	None	Precedent exists for levee construction. Greater chance of public acceptance.	Some sediments may not meet suitability requirements or will require pretreatment.		Work with other agencies contributes to uncertainty in scope and budget.	Price and Lee, 1999 http://el.erdc.usace.army.mil/dots/doer/pdf/doerc2.pdf Krause and McDonnell, 2000 http://www.glc.org/dredging/benuse/Reusepaper_1.PDF
Landfill cover	Use of dredged sediment as daily cover at a permitted landfill	None	Precedent exists for use as landfill cover and landfill cover is in demand. Greater chance of public acceptance.	Some sediments may not meet suitability requirements or will require pretreatment.		Must identify a user of the material and negotiate costs.	
Construction	Use of dredged sediment as construction fill	None	Precedent exists for use as construction fill. Greater chance of public acceptance.			Must identify a user of the material and negotiate costs.	
Thermal conversion to building products	Thermal conversion to lightweight aggregate, portland cement, or glass aggregate products	None	Destroys a large proportion of organic contaminants. Allows beneficial reuse of treated sediment. Enhances immobilization of residual inorganic wastes. Greater chance of public acceptance.	Pilot testing required. Limited availability of local vendors to treat/convert sediments to ceramic products.		Must identify a user of the material and negotiate costs.	

Cost was based primarily on the National Research Council (1997).

*Cost 0 $1,000/yd^3
1 $100/yd^3
2 $10/yd^3
3 $1/yd^3
4 <$1/yd^3

monitoring helps facilitate the attainment of RAOs by (1) assessing the level of risk reduction as a result of remediation; (2) providing information that can be used to enhance the performance of the remedy and facilitate midcourse corrections if necessary; and (3) providing information that can be used to reduce the level of uncertainty in future decision-making on similar projects. A long-term operations and maintenance plan also may be required for containment remedies (e.g., in situ capping or disposal in a CDF).

In order to develop the monitoring plan, performance criteria must be established. Performance criteria are standards by which to evaluate measurable or otherwise observable aspects of the restored system and thereby indicate the progress of the system toward meeting the project goals (Thom and Wellman, 1997). Performance criteria should be as closely linked to the RAOs as possible; the closer the tie, the better the ability to judge progress. From the performance criteria, monitoring parameters (i.e., measurement endpoints) can be chosen. Monitoring parameters are the aspects of the system's structure and function that can be measured. These measurement endpoints define the acceptable or optimal range of values for the chosen parameters. Ideally, the parameters are easy to measure and provide direct feedback on performance of a system toward meeting the RAOs. Examples of monitoring parameters for sediment sites are shown in Highlight 4-14. Generally, a combination of physical, chemical and biological monitoring parameters are needed to fully evaluate the degree of residual risk at the site.

Highlight 4-14. Examples of Physical Monitoring Parameters for Sediment Sites

- Surface sediment grain size distribution (habitat characteristics, sediment behavior)
- Bathymetry (configuration of dredged sediment bed or in situ cap)
- Sediment accumulation rate (monitored natural recovery rate)
- Chemical
- Sediment chemistry (attainment of site-specific final cleanup levels)
- Surface water chemistry (attainment of water quality criteria)
- Biological
- Benthic community analysis (recovery and health of benthic community)
- Toxicity tests (laboratory assessment of sediment toxicity)
- Tissue residues (bioaccumulation, food-chain risks)

The timing, frequency, and duration of long-term monitoring are dependent on the project's complexities and uncertainties. The monitoring program should extend through the period of most rapid change and into the period of stabilization in order to provide reasonable assurances that the remedy either has met its performance criteria and RAOs, or that it will not likely meet the criteria, in which case midcourse corrections will be necessary. For remedial actions under CERCLA, a five-year review is required to determine whether the remediation has succeeded in protecting human health and the environment.

The results of the monitoring program provide information to project managers about the potential need for further action at the site. If monitoring results indicate that the project is progressing as expected, no action may need to be taken; however, if the project is not progressing as expected, potential modifications must be discussed. It is important to note that a divergence from the project's original goals is not necessarily undesirable. It is very possible that new goals that are better aligned with the natural tendencies of the system can be developed in such a way that the project is guided towards a new, but equally beneficial endpoint (Thom and Wellman, 1997).

4.5 MANAGEMENT OF SEDIMENTS IN AREAS REQUIRING MAINTENANCE DREDGING

If contaminated sediments are in a region being evaluated for maintenance or construction dredging, then the sediment should be evaluated according to the guidelines in the USACE/U.S. EPA "Green Book" for marine and estuarine sediments (U.S. EPA and USACE, 1991; http://www.epa.gov/OWOW/oceans/gbook/) or the *Inland Testing Manual* for freshwater sediments (U.S. EPA and USACE, 1998; http://www.epa.gov/OST/itm/index.html). If the sediment fails the dredged material testing guidelines, then open water disposal will not be permitted and other disposal options should be considered.

5.0 GLOSSARY

Acute toxicity = the ability of a substance to cause severe biological harm or death soon after a single exposure or dose. Also, any poisonous effect resulting from a single short-term exposure to a toxic substance.

Ambient concentration = the concentration of a chemical in a medium resulting from the addition of an incremental concentration to a background concentration.

Background concentration = the concentration of a substance in an environmental media (air, water, or soil) that occurs naturally or is not the result of human activities.

Bioaccumulation = process by which substances increase in concentration in living organisms because the substances are very slowly metabolized and/or excreted.

Bioassay = a test to determine the relative strength of a substance by comparing its effect on a test organism with that of a standard preparation.

Bioavailability = the degree of ability to be absorbed and ready to interact in organism metabolism.

Bioconcentration = the accumulation of a chemical in tissues of a fish or other organism to levels greater than the surrounding medium.

Bioturbation = the biological activities that occur at or near the sediment surface that cause the sediment to become mixed.

Chronic toxicity = the capacity of a substance to cause long-term poisonous health effects in humans, animals, fish, and other organisms.

Coagulation = clumping of particles in wastewater to settle out impurities, often induced by chemicals such as lime, alum, and iron salts.

Contaminant = any physical, chemical, biological, or radiological substance or matter that has an adverse effect on air, water, or soil.

Ecological risk assessment = the application of a formal framework, analytical process, or model to estimate the effects of human actions(s) on a natural resource and to interpret the significance of those effects in light of the uncertainties identified in each component of the assessment process. Such analysis includes initial hazard identification, exposure and dose-response assessments, and risk characterization.

Ecotoxicity = the study of toxic effects on nonhuman organisms, populations, and communities.

Effects Range-Low (ER-L) = the contaminant concentration representing the 10th percentile of data associated with biological effects.

Effects Range- Median (ER-M) = the contaminant concentration representing the median of the data associated with biological effects.

Estuary = region of interaction between rivers and near-shore ocean waters, where tidal action and river flow mix fresh and salt water. Such areas include bays, mouths of rivers, salt marshes, and lagoons. These brackish water ecosystems shelter and feed marine life, birds, and wildlife.

Exposure = the amount of radiation or pollutant present in a given environment that represents a potential health threat to living organisms.

Exposure pathway = the path from sources of pollutants via, soil, water, or food to humans and other species or settings.

Food chain = a sequence of organisms, each of which uses the next, lower member of the sequence as a food source.

Food web = the feeding relationships by which energy and nutrients are transferred from one species to another.

Higher trophic levels = the upper feeding level in a food chain or food web, consisting of consumers such as herbivores and carnivores.

Hydrogeology = the geology of groundwater, with particular emphasis on the chemistry and movement of water.

Hydrology = the science dealing with the properties, distribution, and circulation of water.

In situ = in its original place; unexcavated; remaining at the site or in the subsurface.

Infauna = benthic fauna living in the substrate of a body of water.

LC_{50} concentration = Median level concentration, a standard measure of toxicity. It tells how much of a substance is needed to kill half of a group of experimental organisms in a given time.

Lowest Observed Adverse Effect Level (LOAEL) = the lowest level of a stressor that causes statistically and biologically significant differences in test samples as compared to other samples subjected to no stressor.

No Observed Adverse Effects Level (NOAEL) = an exposure level at which there are no statistically or biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered as adverse, or as precursors to adverse effects.

Organism = any form of animal or plant life.

Piscivorous = habitually feeding on fish.

Planktivorous = habitually feeding on plankton.

Redox = oxidation-reduction; a chemical reaction between two substances in which one substance is oxidized and the other reduced.

Reference site = a relatively unpolluted site used for comparison to polluted sites in environmental monitoring studies, often incorrectly referred to as a control site.

Remediation = cleanup or other methods used to remove or contain a toxic spill or hazardous materials from a Superfund site.

Risk assessment = qualitative or quantitative evaluation of the risk posed to human health and/or the environment by the actual or potential presence and/or use of specific pollutants.

Risk characterization = the last phase of the risk assessment process that estimates the potential for adverse health or ecological effects to occur from exposure to a stressor and evaluates the uncertainty involved.

Risk management = the process of evaluating and selecting alternative regulatory and nonregulatory responses to risk. The selection process necessarily requires the consideration of legal, economic, and behavioral factors.

Stressors = physical, chemical, or biological entities that can induce adverse effects on ecosystems or human health.

Toxicity = the degree to which a substance or mixture of substances can harm humans or animals. Acute toxicity involves harmful effects in an organism through a single or short-term exposure. Chronic toxicity is the ability of a substance or mixture of substances to cause harmful effects over an extended period, usually upon repeated or continuous exposure, and sometimes lasting for the entire life of the exposed organism. Subchronic toxicity is the ability of the substance to cause effects for more than one year but less than the lifetime of the exposed organism.

Watershed = the land area that drains into a stream; the watershed for a major river may encompass a number of smaller watersheds that ultimately combine at a common point.

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Section 4.0: Sediment Remedial Alternative Evaluations

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Web sites

http://www.epa.gov/epahome/ for laws, regulations, and programs

http://www.dtsc.ca.gov/ScienceTechnology/HERD_FLY_Overview.html for California HERD info and eco checklist

http://web.ead.anl.gov/ecorisk/ for NAVFAC Ecological Risk Assessment Guidance

http://www.clu-in.org/ for EPA's Technology Innovation Office

http://regscreen.nfesc.navy.mil/htm/search.htm for regulatory searches

http://www.smwg.org/ Sediment Management Working group Advocates risk-based cleanups; information on Natural Attenuation.

http://www.hudsonvoice.com/mcss/ Major Contaminated Sediments Sites Database. A joint effort of the General Electric Co., Applied Environmental Management, Inc., and Blasland, Bouck & Lee, Inc. Extensive data base that includes cleanup standards and project costs.

http://www.sediments.org/ Sediment Research Web. Project summaries, technical reports and information on sediments.

http://www.ecy.wa.gov/programs/tcp/smu/sediment.html Washington State's Department of Ecology internet site on sediments.

http://sfbay.wr.usgs.gov/access/saltponds/index.html U.S. Geological Survey site; Wetlands creation in the San Francisco Estuary

http://www.oehha.org/ecotox/documents/caleco1.html for the new CalEPA OEHHA Ecotox info

http://chppm-www.apgea.army.mil/erawg/ for the Tri-Service Ecological Risk Assessment Working Group

http://www.envirofacs.org/ Division of Environmental Chemistry Home Page, American Chemical Society

http://www.envirofacs.org/Resources for chemical information http://www.epa.gov/OST/cs/stratefs.html for Contaminated Sediment Management Strategy fact sheet

http://www.epa.gov/OST/cs/stratndx.html for Contaminated Sediment Management Strategy document

http://www.epa.gov/superfund/programs/risk/humhlth.htm

http://www.epa.gov/ordntrnt/ORD/WebPubs/carcinogen/

http://www.epa.gov/ordntrnt/ORD/WebPubs/repro/

http://www.epa.gov/ncea/nurotox.htm.

APPENDIX A: Rapid Screening Methods/Rapid Sediment Characterization (RSC)

It is not always necessary to use the same analytical methods that will be used in the definitive study, as simpler methods may be adequate to estimate COPC concentrations or measurement endpoints. New rapid screening tests are continually being developed, and a check of the literature for recent developments should be conducted prior to selecting screening tests for a site. Examples of screening techniques for chemical measurements and toxicity are discussed below; most are described in greater detail in EPA (1994). Another survey of Rapid Sediment Characterization (RSC) tools is available at http://web.ead.anl.gov/ecorisk/issue/pdf/rsc.pdf.

Total PAH by Fluorometry

The total PAH assay using fluorometry is fairly rapid (20 samples/day) and inexpensive, and is strongly correlated with GC/MS and high-pressure liquid chromatography (HPLC) analyses (Friocourt et al. 1985). This method is particularly sensitive to compounds containing aromatic rings, such as PAH, but does not respond to a wide range of organic compounds found in sediment, such as aliphatic hydrocarbons from oils, fatty acid methyl esters from natural and anthropogenic sources and phthalate esters.

Total PCBs, Chlorinated Pesticides, and Other Organic Chemicals by Enzyme Immunoassay

Enzyme immunoassays are biochemical procedures that utilize the binding of specific chemicals in a sample (with an enzyme-labeled version of the chemical) to antibodies provided with a test kit. For sediments, a small sample is quickly extracted and purified. The extract is then tested with the immunoassay kit. Chemicals bound to the antibodies are then separated from the rest of the sample and associated interferences by simple washing. The labeled component is detected by adding a color indicator (Schrynemeeckers 1993). Enzyme immunoassays are inherently free of most confounding factors (Vanderlaan et al. 1991) and are available for several aromatic compounds (e.g., PCBs, pesticides, petroleum hydrocarbons, PAH, trinitrotoluene, benzene). These assays typically identify the presence or absence of chemical mixtures such as total PCBs at some predetermined concentration. Typical detection limits are usually 1-3 orders of magnitude higher than sediment specific laboratory methods; however, they are sufficient to identify problem areas that may warrant greater focus under Step 6. The detection limit for total PCBs in sediments for various test kits ranges from <0.1-5 ppm (wet weight).

Significant cost savings will be realized most readily at sites that have a limited number of contaminants of concern, where only a single immunoassay is required. Several different immunoassays would be required to fully characterize sediments containing a wide range of compound classes of concern, because each kit is sensitive to only one of the compounds or classes of compounds described above.

Total Petroleum Hydrocarbons by Infrared Spectroscopy

The infrared assay is intended to be a field version of the extractable residue analysis of the relict USEPA Method 418.1. This analysis is useful for estimating total petroleum hydrocarbons (TPH) because it measures responses in selected narrow ranges of the infrared spectrum. A variety of hydrocarbon structures are simultaneously detected in a sample by characteristic changes in carbon-hydrogen or carbon-carbon bonds (e.g., stretching and bending vibrations). These changes are induced by exposure to infrared radiation. Detection limits are typically in the range of 1-10 ppm. The infrared method has the advantage of providing a rapid, quantitative determination of TPH concentrations, but also has some limitations that can produce either negative or positive analytical bias (Douglas and Uhler 1993). As a result, this screening method may be less accurate than other techniques for measuring hydrocarbons such as field gas chromatography or TLC, which have a higher cost.

Semivolatile Organic Compounds by Thin-Layer Chromatography

The TLC field method, developed by Friedman & Bruya, Inc. (Seattle, Washington) and reported by Newborn and Preston (1990, 1991), can be used for a wide range of semivolatile organic compounds with detection limits of approximately 10 ppm. Lower detection limits to approximately 1 ppm are feasible for some compounds. This method involves placing a drop of sample extract near the bottom of a silica gel-coated glass plate. The end of the plate is immersed in an appropriate solvent. As the solvent front moves upward on the plate, the compounds of interest are separated out of the mixture based on their mobility in the solvent-solid phase system, and can then be identified both qualitatively and quantitatively, using ultraviolet light or iodine to visualize the separated chemicals.

Metals by X-ray Fluorescence

Field portable XRF units have been used to analyze soils at Superfund sites (e.g., Fribush 1992; Driscoll et al. 1991) and have been shown to provide rapid (<5 minutes/dried sample) quantification of more than 20 elements at a time. Detection limits for portable units have typically been reported in the 100-1,000 ppm range for most metals, while laboratory-based XRF units have greater resolution and are capable of lower detection limits in the range of 2-25 ppm. Laboratory XRF units have a somewhat longer analysis time (20 minutes). XRF analyses, unlike other metal analyses that rely on digestion of samples with various acids, do not destroy the sample and require only a small amount of material. XRF has produced results that correlate strongly with results produced using conventional atomic absorption and inductively coupled plasma (ICP) spectroscopy (Kuharic et al. 1993).

Rapid Toxicity Tests

The Microtox® test is a rapid, sensitive method of toxicity testing based on light emission by the luminescent bacterium *Photobacterium phosphoreum* in the presence and absence of aqueous toxicants. The emitted light is a product of the bacterial electron transport system and thus directly reflects the metabolic state of the cells. Accordingly, decreased luminescence following exposure to chemical contaminants provides a quantitative measure of toxicity. Two or more rapid toxicity tests may be performed in tandem to increase sensitivity and coverage, for example with Microtox ® and Mutatox ® (Johnson and Long 1998).

Feasibility of Non-standard Methods

Non-standard methods may sometimes be useful in the evaluation of sediment quality; examples are *in situ* toxicity tests, bioaccumulation studies using non-standard species, or colonization studies of artificial or natural substrates. Preliminary studies should be performed (in Step 5) for any non-standard, unverified test using a limited number of replicates to ensure that data quality objectives will be met under site conditions.

APPENDIX B: Geochemical Normalizers in the Interpretation of Sediment Contaminant Data

Aluminum, iron, TOC, and grain size can be used in particular cases as geochemical normalizers. They will often show strong relationships with site contaminants, and to varying degrees they covary together with sediment texture. In a highly simplified geochemical model, weathering of continental crust results in a two-endmember sediment. The course grains (>63 microns) that survive weathering and transport down rivers to coastal depositional areas are predominately the most resistant minerals, such as quartz grains. These large grains have low metals levels in their matrix and relatively low surface areas so little metal is sorbed onto their surfaces. During weathering and transport other minerals are broken down to produce clay and other mineral phases that predominate in the fines fraction (<63 microns). This fines fraction has more metals in mineral matrices, plus a greater relative surface area that sorbs more metals. This fines fraction also contains the majority of TOC, and the organic contaminants that are associated with the TOC. In most cases this fines fraction contains the majority of the metals and organic contaminants sorbed to its surface area that constitute the anthropogenic part of the background, and within its' matrix are the metals that constitute the naturally occurring background. In addition, if there have been site releases that represent additional site sources, these site related metal and organic contaminants will also be sorbed onto the surface area of the sediment (again mainly within the fines fraction). These coarse and fine endmembers are then mixed in varying proportions throughout the depositional areas depending on hydrodynamics to produce the full range of chemistries found around the site. In higher energy nearshore areas (beaches), sandy (coarse) sediments composed mainly of quartz grains are present with relatively low contaminant levels. Moving offshore towards lower energy areas more fines are allowed to settle out and produce the more fines rich sediment that has higher contaminant levels (both metals in the matrices (naturally occurring background) and metal and organic contaminants sorbed onto the surfaces (anthropogenic background)). When geochemical normalization is conducted, it will help differentiate any potential site contaminant releases from background sources (both naturally occurring and anthropogenic). Site samples that plot above a background relationship have an additional source contribution for the contaminant of concern that is not present in background samples, and this additional local source is assumed to indicate a site release. It is these areas with site releases that we want to focus on in the IR program, and produce site exposure and effects risk relationships that focus on site releases and not background chemicals.

In order to use these geochemical normalizations, several assumptions should be checked (Louma, 1990). The first is that there is a significant relationship between the normalizer and the contaminant of concern in reference or background samples. This is done with simple crossplots and checking for high correlation. The normalizer should also be insensitive to anthropogenic inputs and stable (non-reactive) under geochemical conditions found in the sediment. These requirements are not strictly met in every case, so some care and judgment is required for use at individual sites. Aluminum often works well to adjust trace metal chemistries for changes in the naturally occurring metals in the matrix of mineral phases. The trace metal level varies as the mineralogy of the samples varies, but always in constant proportion to the aluminum level. So normalizing to aluminum "corrects" for changing sample mineralogy and therefore trace metal levels associated with different mineral matrices. In actual practice, however, the trace metals levels that appear on sediment background crossplots are not just from naturally occurring matrix sources, but also sorbed metal from anthropogenic background. In industrial areas, the anthropogenic contribution to trace metals levels may be quite high depending on the metal. This results in portions of the trace metal in the background relationship coming from both naturally occurring and anthropogenic background sources. In addition, many environmental programs use an acid leach rather than total dissolution of all mineral matrices in their sediment chemical preparations, so proportions of naturally occurring and anthropogenic sources may vary depending on preparation technique as well as sample location (so exercise care in combining studies). But as long as samples are treated by the same method (so same amount of matrix is dissolved) and from the same area (so same amount of anthropogenic background sorb onto surfaces), aluminum normalization works well for many trace metals. Aluminum does show very strong relationships for many trace metals. It is naturally present in percent levels in samples so any minor anthropogenic contributions are lost in the noise and can be considered not to affect the relationships. And finally, aluminum is non-reactive in sediment geochemical environments so levels will not change with sediment reactions.

All this being said, aluminum is not universally applicable as a background normalizer. Daskalakis and O'Connor (1995) reviewed NOAA Status and Trends sediment data from around the country and found while aluminum worked well on the east coast, iron and TOC may work better in some cases on the west coast. On the east coast, watersheds tend to drain the coastal plain in front of the Appalacian Mountains, with a relatively constant granitic chemical composition. This results in relatively constant "crustal abundance" ratios (trace metal to aluminum ratios) in sediments from different areas. The geologic history of the west coast results in much more variable chemistries in adjacent (or even the same) watersheds so trace metal to aluminum relationships are also more variable. Iron may therefore work better for metals, and TOC for organics on the west coast. However, even though these normalizers are in percent levels, they may still show significant anthropogenic and sediment reaction contributions and therefore fail in their role as normalizers. Iron is more reactive than aluminum, and it tends to be remobilized under anoxic conditions. Differences in redox state among different sediment areas may therefore lead to confounding factors in relationships. TOC of different types may not carry the same amounts of anthropogenic background, which can also lead to confounding factors in these normalization relationships. Just as one wouldn't try to use a complicated statistical background method without consulting a statistician, it is advised to consult an experienced geochemist before using these types of geochemical methods to differentiate background. It is therefore advisable to include a geochemist in the discussions so some professional judgement can be used to help decide when these types of geochemical models (these normalization relationships) are appropriate.

In an effort to simplify the various geochemical relationships used to normalize for background contributions, some success has been found in using grain size (Klamer et al, 1990; EPA Region 9,1998) to encompass all these covaring relationships. Since aluminum, iron, TOC, and grain size all tend to co-vary, the use of a single normalizer can often represent several underlying geochemical relationships. Any single metal or organic contaminant might be better normalized with a different normalizer, but grain size is often a simple compromise that works well enough in many cases. EPA Region 9 (1998) used this approach to develop background (or "ambient") levels in S.F. Bay. They used the grain size relationship to generate a UCL on the relationship. They recommend point comparisons to this UCL be used to decide if a sample is above ambient levels. Following the reasoning outlined in the background guidance (SWDIV, 1998;1999), it would be anticipated that point comparisons to a UCL might result in a significant number of false positives. Therefore a better approach might be to look at the population of residuals (predicted value from geochemical relationship minus the observed value) at the site and compare to residuals from a reference population. (Note for Discussion: The proposed guidance (6/00) provides a flowchart to guide the use of background techniques. This flowchart splits down two paths, one for a geochemical approach and one for a statistical approach. The last line in this paragraph is suggesting that there may be utility in combining these approaches, but this should have additional discussion before being placed in any suggested guidance.)

Starting with these background geochemical relationships, the simple crossplot normalizations show how geochemistry can differentiate between background and site releases. More involved techniques are also available to fingerprint geochemical signatures. In addition to separating background from site releases, these techniques also show the potential to differentiate different site releases and offer the possibility of fingerprinting non-navy sources. The goal would be to point to different release signatures that indicate non-navy PRPs as the source for some offshore contamination on Navy property. While these exercises might start with simple linear crossplots of contaminants, they usually progress to the use of more involved multivariate techniques such as principle component analysis (PCA). These multivariate techniques are more commonly used with organic contaminants, where compound groups (i.e., PAHs and PCBs) are composed of many similar individual compounds. These more complicated fingerprinting studies fall outside the scope of this discussion but show that these geochemical applications may have many goals in addition to simple background determinations. RPMs should always understand the anticipated goals are for use of any geochemical relationships, and thereby weigh the involved costs with potential benefits to decide how much geochemical analysis is warranted.

Appendix C: Summary of an ERA at a Hypothetical Sediment Site

This section evaluates ecological risk for a hypothetical case study within the study design framework defined in Step 4. In practice, only final data that have met specified DQOs, are applied to Step 4 decision rules to determine corresponding actions for various result scenarios. The worst-case scenario is assumed for the hypothetical case study, where site sediment chemical concentrations, toxicity and tissue bioaccumulation are negatively impacted compared to the reference site, indicating unacceptable "relative" risk. Absolute risk is then evaluated for bioaccumulation using toxicity reference values (TRVs) obtained from the literature for each assessment endpoint. The site is then re-examined on a station-by-station basis by augmenting the decision rules with the TRV results. This approach provides critical information on the spatial extent of at-risk areas for remedial design to risk managers and decision makers. PCBs and copper are examined as common Navy sediment COPCs in the following hypothetical study.

Hypothetical Site Setting

The hypothetical site consists of shoreline sediments located in close proximity to a historical Naval Air Station. COPCs enter the system through outfall pipes and road-runoff as land-based point- and non-point sources, respectively. The assessment endpoints are healthy communities of benthic infauna, as well as forage fish and piscivorous birds. The measurement endpoints are amphipod survival, polychaete survival and growth, and bioaccumulation of COPCs in field-collected fish tissue. These endpoints address both acute and chronic toxicity as well as bioaccumulation in the lower food web, which is subsequently modeled for higher-order receptors (i.e., birds).

Surface sediments were collected and tested for COPCs, 10-day acute toxicity using the amphipod *Rhepoxynius abronius*, 28-day survival and growth using the polychaete *Neanthes* sp., and 28-day in-situ bioaccumulation in forage fish (e.g., top smelt). Prior to sampling, the site was divided into three subareas based on existing sediment grain size data, geographic location and proximity to specific land-based contaminants. Resident forage fish were collected for each subsite and reference area by trawl. Ten fish per subsite were each analyzed for whole body COPC concentrations. For surface sediments, each subsite was sampled in a grid pattern that provided uniform sample density, resulting in a slightly unbalanced study design of 10, 12 and 14 sediment stations for the three subsites due to differences in area. Although stations were regularly spaced, the grid placement was made without *a priori* assumptions (at random) so that subsite stations could be treated as replicates in the statistical analyses. This sampling design also provides optimum coverage of the area with contiguous information for subsequent remedial design, if needed.

A total of 10 reference stations were selected throughout the same harbor that were previously shown to be unimpacted using agreed-upon Tier 1 screening criteria (in this case > 69.5%

amphipod survival and no COPC concentrations > NOAA ER-Ls). Reference stations also were selected for sediments that spanned the range of grain size expected for site sediments (based on field validation data collected in Step 5). More importantly, reference stations were selected based on a similar expected outcome of measurement endpoint test results as compared with the test site in the absence of effects from COPCs (e.g., no other confounding factors – sulfides, salinity).

Results and analysis of relative and absolute risk for chemical exceedances, toxicity and fish tissue bioaccumulation follow. Bioaccumulation was evaluated using field-collected fish that were local to the site and had a known exposure route to COPCs in site sediment. Alternatively, bioaccumulation can be estimated using either laboratory tests or modeling using either empirical or mechanistic approaches. Laboratory tests determine the extent of bioaccumulation by organisms, usually at lower trophic levels (e.g., clams, polychaetes, forage fish). This method accounts for bioavailability from sediment to organisms, but typically some form of modeling is needed to predict COPC concentrations in higher trophic levels (e.g., piscivorous fish, birds, mammals). Empirical modeling is commonly used in routine assessments due to its simplicity compared to mechanistic approaches, which rely on trophic transfer in the food web. Direct measure in field specimens is the preferred method (EPA 1996, 2000) as it produces less uncertainty in the final result.

Lines-of-Evidence Results Summary

In this example, mean copper concentration in sediment for one of the three subsites was 90 mg/kg (ppm) dry weight (2% TOC), and was significantly elevated ($p<0.05$) compared to the corresponding reference site mean of 28 mg/kg, even when normalized to organic carbon. The reference site had a lower range of TOC (e.g., mean = 1.2%), so all COPC sediment concentrations were normalized to organic carbon (e.g., mg COPC/kg organic carbon) prior to inference testing (applying to decision rules).

Mean copper concentration in fish tissue was not elevated compared to reference tissue. Three other metals, chromium, zinc and lead also were significantly elevated in sediment but all concentrations were less than corresponding Tier 1 screening criteria (e.g., NOAA ER-Ls), and were therefore excluded from further analysis. Additionally, mean total PCB measured as the sum of 22 individual congeners were elevated in both sediment and fish tissue for the same subsite. Mean total PCB sediment concentration was 0.3 mg/kg dry weight, at 3% TOC; and the mean whole body PCB concentration was 0.29 mg/kg wet weight. The PCB mixture consisted of a variety of congeners, but was dominated by Aroclor 1254. In this example, the mean PCB concentration in resident fish (smelt) collected in the affected subsite was used as the concentration in prey. However, these data could have been obtained from the literature or modeled from site sediment data (EPA 2000). No other COPC exceedances were observed for this or the remaining two subsites. The final COPC list was limited to copper and PCBs, and risk characterization was performed for the only subsite affected.

For the same subsite, mean amphipod survival of 57% was not statistically lower than the mean reference survival of 75%; however, it was lower than a guideline value of 69.5% that was used in other local regulatory programs. Polychaete survival also was not statistically different between test and reference areas; however, growth was significantly impaired at the test site (mean of 2.36 mg/day vs. 5.22 mg/day for reference).

Although commonly used, standard inference tests are often not applicable to percentile toxicity data (which range from 0-100%), since results are often not normally distributed and fail test assumptions, even after they are transformed. It is therefore appropriate to consider other methods to evaluate toxicity data, including non-parametric methods (e.g., rank-sum tests) or non-statistical methods, such as use of local guidelines.

Copper Toxicity

In this case study, the mean copper concentration of 60 mg/kg was above the Tier 1 screening criterion (34 mg/kg) and it was elevated compared to the reference mean, so it was retained as a COPC. If copper is one of many COPCs in sediment, toxicity test results must be interpreted as the effects of the overall mixture thereby the toxic effects of the copper must be estimated. There are several approaches to do the estimation including:

1. If a wide range of copper concentrations were measured at the site, the highest concentration that was not toxic can be used as an acceptable sediment copper concentration for the site. This is similar to the AET approach used for Puget Sound sediments (Barrick et al. 1988).
2. If concentrations are not wide-ranging (homogeneous), spiked sediment toxicity tests using COPCs may be useful, although there are many difficulties with implementing and interpreting spiked sediment toxicity tests.
3. Instead of laboratory bioaccumulation, critical body residues of resident fauna can be measured. In either case, results can be compared to critical concentrations for similar species in the ERED (http://www.wes.army.mil/el/ered/) database or from the scientific literature. Clams, which are filter and deposit feeders, are efficient bioaccumulators of copper, although other abundant or ecologically important organisms may be more appropriate. However, interpretation of the significance of tissue residues is problematic and should be interpreted with caution; they should not be used as the only approach for assessing risks from sediment copper.

If the concentration of copper in site sediments was much higher than the Tier 1 screening criterion, for example greater than 390 mg/kg, and toxicity testing indicated sediment toxicity, then it may be best to forego further studies and assume the site poses significant risk.

For this case study, copper concentrations were examined on a station-by-station basis. Where copper was both elevated compared to Tier 1 screening criteria and the reference mean, and significant impacts were observed for either amphipod survival or polychaete growth, the station was identified as having "unacceptable risk". Bioaccumulation of copper was not significantly elevated in fish collected at the subsite, so potential effects from bioaccumulation to assessment endpoints were not evaluated. A discussion on the bioaccumulative effects of copper follows as a representative approach to evaluate significantly elevated site COPCs. Although copper is considered a bioaccumulative constituent by many agencies and guidance documents, it generally does not biomagnify in higher trophic levels (most vertebrates), such as methyl-mercury or PCBs. If there are adverse effects at a site, they usually occur to benthic biota such as bivalve mollusks, and occur relatively close to the contaminated site (there are fewer far-field effects). Copper is usually not a problem for birds and mammals, except at highly contaminated sites or unusual exposure scenarios.

Bioaccumulative Effects of Copper

Copper tissue residues in laboratory organisms (or resident fauna) can also be compared to the critical body residue (CBR) concentrations shown to cause toxic effects in similar organisms. A search of the ERED database shows a variety of CBR values for copper and other COPCs. The type of effect and tissue analyzed in the study should correspond with data from the literature. The species analyzed in literature studies are rarely the same as those encountered at the site, but the same criteria for selecting measurement endpoints for assessment endpoints can be used to select appropriate CBRs from the literature. However, habitat and feeding regime may not be as important for tissue residues, since tissue residues already reflect the effects of differences in bioavailability and uptake mechanisms. The tissue residues can be compared to CBRs from the literature to determine whether there are adverse effects from copper in tissues. However, since copper is a metal that can be sequestered in non-toxic forms in some species under certain exposure regimes, other supporting lines of evidence should also be used to establish the significance of tissue residues to adverse effects.

PCB Risk Characterization

PCBs are lipophilic, highly stable and degrade slowly in the environment, resulting in high concentrations in sediment and organisms even at low concentrations in water. PCBs also readily biomagnify from lower to higher trophic levels, making them some of the chemicals of greatest concern for bioaccumulative effects.

PCBs consist of a class of 209 congeners, and are usually found as complex mixtures of Aroclors in contaminated sediments. Information on bioaccumulation and toxicity exists for Aroclor mixtures of PCBs, but much of the data available are for individual congeners. Risk characterizations can be made for each congener when warranted; however, evaluation of total PCBs or Aroclor mixtures is sufficient in many cases.

For the case study, piscivorous birds, are the highest trophic level receptors potentially impacted by PCBs. Piscivorous birds are exposed to PCBs primarily from ingestion of contaminated prey – in this case, forage fish. Risk characterization for these birds requires three pieces of information: 1) PCB concentration in forage fish exposed to contaminated sediment; 2) an assessment endpoint representative of all picivorous birds at the site; 3) a corresponding toxicity reference value (TRV) for the chosen endpoint. In this case, the Belted Kingfisher serves as a representative picivorous bird with a corresponding TRV obtained from Sample et al. (1996).

Forage fish were collected at the contaminated subsite, and the mean whole body PCB concentration was 0.29 mg/kg wet weight. Although measuring field-collected specimens is the preferred method, estimates can also be made from literature values if field data are unavailable. The simplest approach is to use existing biota-sediment accumulation factors (BSAFs) from the literature to estimate PCB concentrations in fish from measured PCB concentrations in sediment. BSAFs are defined by EPA (1995) as the ratio of a lipid-normalized COPC concentration in tissue to its organic carbon-normalized concentration in sediment. For nonpolar organics, BSAFs are calculated as:

$$\mathrm{BSAF} = (C_t/F_l) / (C_s/F_{oc})$$

Rearranging to calculate tissue concentration (C_t) from a BSAF gives:

$$C_t = (\mathrm{BSAF} * (C_s/F_{oc})) * F_l$$

Where:

C_t is concentration in the organism (check basis for wet or dry weight);

F_l is the lipid fraction in tissue;

C_s is the COPC concentration in sediment (usually dry weight); and

F_{oc} is the organic carbon fraction in the sediment.

There are other approaches to modeling bioaccumulation to aquatic organisms, including both empirical and mechanistic models. The choice of methods and models will depend on the scope of the study, selected endpoints, and data available (both in the literature and from the site). A discussion of some of these models can be found in EPA (2000).

Determining Bioaccumulation Effects to Piscivorous Fish. The concentration of PCBs in fish measured from field-collected specimens can be compared directly to PCB tissue concentrations shown to cause adverse effects in other studies. Appropriate species from the site should be designated as surrogate species, and compared to species from the literature that are closest in habitat, trophic status and taxonomic level. For highly mobile and wide-ranging species such as fish, PCB tissue concentrations may not reflect the worse case assumptions of all feeding within the contaminated site. However, measures from field-collected specimens will reflect the proportionate use of the site area for relatively immobile species. It is especially important to ensure proper study design and statistical analyses when using field-collected data.

A search of the ERED database yielded a number of NOEDs for PCB concentrations in whole bodies of several fish species and life stages. For salmonids, NOED levels for mortality range from 2.0 mg/kg (whole body) for rainbow trout to 30 mg/kg (whole body) for Atlantic salmon. Channel catfish (Ictalurus punctatus) NOEDs for mortality ranged as high as 14.3 mg/kg (whole body). For a comparison to low and high effect levels, in rainbow trout a 1.3 mg/kg PCB concentration in the whole body resulted in 10% mortality (Hogan and Brauhn, 1975), and for the Coho salmon, a whole body concentration of 645 mg/kg PCB resulted in 100% mortality. Data for fish that are forage species include NOED concentrations as high as 230 mg/kg (whole body) for the fathead minnow.

For this example, the 0.6 mg PCB/ kg body weight we measured in our forage fish was well below the adverse effects levels cited above. We conclude that there is no risk to fish from bioaccumulation of PCBs.

Determining Bioaccumulation Effects to Piscivorous Birds. Assessing bioaccumulation effects to piscivorous birds usually involves modeling PCB accumulation through the food chain via prey items, which are typically forage fish.

The measured concentration for forage fish was used as the estimated concentration of PCB in site forage fish. This can be compared to the TRV for the concentration of PCB in prey eaten by the Belted Kingfisher. The no observed adverse effects level (NOAEL) for food for the Belted Kingfisher is 0.355 mg of Aroclor 1254/ kg food (Sample et al. 1996). The concentration of PCB in tissues from forage fish at the site (0.29 mg PCB/kg body weight) is below the TRV, so we conclude that there is no risk to piscivorous birds from the bioaccumulation of PCBs.

If the assessment endpoint at the site is not the Belted Kingfisher or some other species with published TRVs (there may be several important species at the site which need to be considered as assessment endpoints), then TRV calculations can be performed using methods described in Sample et al. (1996). These usually include adjustments for body weight and food consumption rates for specific species. Alternatively, TRVs can be derived for contaminant dose per body weight per day (e.g., mg PCB/kg·day) and compared to ingestion rates based on allometric formulas (EPA 1996, 2000). TRVs based on ingestion rates can then be applied to other assessment endpoint species (e.g., other sea birds), once their body weight is known. The following ingestion rate formula (EPA 2000) also includes adjustments for the relative bioavailability of a COPC (BF) and the proportion of the study area (site) relative to the home range of the assessment endpoint (EF) (i.e., site use factor). Bioavailability (BF) and site use (EF) are set at one (1) using the most conservative assumptions; however, these assumptions are usually unrealistic and increase model uncertainty. However, information on site use and COPC bioavailability are often difficult to obtain. Therefore, these factors should be adjusted only when the resulting TRV (in this case the IR_t) indicates unacceptable risk and the required information is available. The COPC ingestion rate for any species can be estimated from a generic food web model (EPA 2000) as:

$$IR_t = SUM\ [(C*I*BF)_i * EF]\ /BW$$

Where:

IR_t = total rate of COPC ingestion (mg COPC/kg body weight·day) (in wet weight)
C_i = COPC concentration in each medium (e.g., food, water) as mg/kg (wet weight)
I = the rate of ingestion in each medium (e.g., mg food/day)
BF = the relative bioavailability factor of the chemical from each medium (unitless)
EF = the proportion of the study area relative to the home range of the species (unitless)
BW = body weight of assessment endpoint (kg)

Final Risk Characterization

Final risk characterization is based on results for the decision rules applied from Step 4 (relative risk) and absolute risk evaluated using - in this case - TRVs. Following the example used for copper above, all COPC results should be evaluated on a station-by-station basis. This allows identification of problem areas within the subsite and ranking of these areas relative to each other and the reference site. Evaluation by station also identifies any potential “hot-spots” that may have been previously concealed when the subsite was examined as a single area. Application of decision rules could be made by comparing each site station result to a 95th or 99th predictive interval based on grouped reference site data for each parameter (e.g., COPC concentration, toxicity result, COPC tissue concentration) (see Step 4 for a discussion of appropriate statistical methods).

For the hypothetical case, each station is examined to ensure that COPCs, toxicity and bioaccumulation (both relative to reference and literature-based TRVs, where either indicates significant impact). This information is then used to identify affected stations for the remedial design based on relative location (to each other and onshore site contaminants) and degree of impact.

APPENDIX D: EXAMPLE PLANNING TABLE USED TO ASSIST WITH THE DEVELOPMENT OF THE CONCEPTUAL SITE MODEL REFINEMENT IN STEP 3

SUMMARY OF ASSESSMENT ENDPOINTS, RISK QUESTIONS AND MEASUREMENTS ENDPOINTS.

(Page 1 of 3)

Receptor Class and Specific Receptors	Assessment Endpoint	Risk Questions	Surrogate Species or Community	Measurement Endpoints	Uncertainties	Notes
1º PRODUCERS V. Phytoplankton and algae Unicellular plants	1) Survival and reproduction 2) Maintain a similar zooplankton community to that in similar habitat in SF Bay	1) Are surface water chemicals related to Navy activities present at concentrations acutely or chronically toxic to the phytoplankton community? 2) Are Navy-related chemical concentrations in water high enough to cause adverse impacts to the structure of the phytoplankton community?	*Lassiosira* or *Skeletonema* Phytoplankton Community	1) Laboratory test for toxicity 2) Abundance of phytoplankton (e.g. chlorophyll a?) 3) Compare biodiversity index to SF Bay reference sites	1) Lab test may not reflect field conditions. 2) May be infeasible to determine acceptable reference site 3) Many confounding factors to evaluating results. 4) Influx due to tide may confound results. 5) Biodiversity index may not be feasible in the SF Bay	1) The plankton community would be inferred to be healthy, if the BMI community and their predators are healthy. This trophic path is adequately evaluated through analysis of other pathways. **PROPOSE NOT TO EVALUATE THIS ENDPOINT**.
1º CONSUMERS VI. Zooplankton Copepods, rotifers, larval shellfish	1) A similar zooplankton community to that in similar habitats in the SF Bay	1) Are surface water chemicals related to navy activities present at concentrations acutely or chronically toxic to the zooplankton community? 2) Are Navy-related chemical concentrations in water high enough to cause adverse impacts to the structure of the zooplankton community?	*Mysidopsis, or Mytilus* larvae *Ampelisca abdita* Zooplankton Community	1) Laboratory test for toxicity 2) Abundance of zooplankton in field 3) Compare biodiversity index to SF Bay reference sites	1) Lab test may not reflect field conditions. 2) Many confounding factors to evaluating results. 3) Biodiversity index may not be feasible in the SF Bay	1) The plankton community would be inferred to be healthy, if the BMI community and their predators are healthy. This trophic path is adequately evaluated through analysis of other pathways. **PROPOSE NOT TO EVALUATE THIS ENDPOINT.**
1º CONSUMERS VII. Benthic Macroinvertebrates (BMI) Amphipods, bivalves, polychaete worms, and crabs	1) Acute or chronic toxicity to BMI 2) *In situ* toxicity to BMI compared to SFBAY? 3) Maintain populations of diverse species that represents a stable healthy benthic community	1) Are chemicals in Seaplane Lagoon adversely affecting benthic organisms? 2) Is the toxicity of Alameda Point sediments similar to sites in the Bay considered to be generally unaffected by point discharges? 3) Is the benthic community diversity in Seaplane Lagoon lower than reference areas in the Bay?	*Eohaustorius sp, Neanthes sp.,* --or- *Strongylocentrotus sp.* Benthic community	1) Compare concentration of COPECs in sediment to levels reported in the scientific literature to be harmful. 2) Compare site-specific sediment toxicity tests to SF Bay reference stations, including analysis of covariance with sediment chemical concentrations. 3) Conduct community analyses in concert with analysis of covariance with sediment chemical concentrations.	1) Actual conditions of exposure in SF Bay may not be represented. 2) May be infeasible to determine acceptable reference site 3) Effects of salinity, temperature and organic content may confound toxicity tests, organic material or other characteristics unrelated to chemicals. 4) Community analyses may have large uncertainty in the Bay.	1) This step is already accomplished through the COPEC screening process. 2) Toxicity test results need to be scrutinized to determine that percentage of organic carbon, grain size, salinity and other physical characteristics of the comparison tests are similar to those at Alameda Point. 3) Benthic community analyses have not been demonstrated to be useful in SF Bay. The high proportion of introduced species, the wide temporal variation in populations and the high variability in physical conditions such as salinity, temperature, grain size, and organic material content can confound the interpretation of community analysis results.

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APPENDIX D: EXAMPLE PLANNING TABLE USED TO ASSIST WITH THE DEVELOPMENT OF THE CONCEPTUAL SITE MODEL REFINEMENT IN STEP 3 SUMMARY OF ASSESSMENT ENDPOINTS, RISK QUESTIONS AND MEASUREMENTS ENDPOINTS.

(Page 2 of 3)

Receptor Class and Specific Receptors	Assessment Endpoint	Risk Questions	Surrogate Species or Community	Measurement Endpoints	Uncertainties	Notes
2º CONSUMERS VIII. Planktonivorous Fish Herring, topsmelt, and anchovy	1) Maintain a healthy, self-sustaining population in a balanced community by protecting the planktonivorous fish from the cumulative effects of chemicals in their food. 2) Fish egg development	1) Will the consumption of Navy-related chemicals in prey items be acutely or chronically toxic to fish? 2) Will the consumption of Navy related chemicals in prey items adversely affect larval survival of planktonivorous fish? 3) Is the planktonivorous fish community composition altered from the toxic effects of Navy-related chemicals in prey items?	Pacific herring Topsmelt embryo-larval development	1) Measure concentration of chemicals that have bioaccumulated from sediment or food chain in prey items and compare them to fish TRVs. 2) Topsmelt embryo sediment-water interface toxicity test	1) Limited information from which to generate fish TRVs. Identifying an appropriate reference site. 2) Toxicity to embryos does not evaluate whether adults can spawn. 3) Community analysis not feasible. 4) Planktonivorous fish feed in the water column and would have limited contact with contaminated sediments.	1) The planktonivorous fish community would be inferred to be healthy, if the higher trophic level piscivorous fish and their predators are healthy. This trophic path is adequately evaluated through analysis of other pathways. **PROPOSE NOT TO EVALUATE THIS ENDPOINT.**
2º CONSUMERS IX. Benthic-Feeding Fish Starry flounder, English sole, goby, sturgeon, and plainfin midshipman	1) Maintain a healthy, self-sustaining population in a balanced community by protecting the benthic-feeding fish from the bioaccumulative effects of chemicals in their food and in consumed sediments.	1) Will the consumption of Navy-related chemicals in prey items, **in addition to incidental sediment ingestion,** be acutely or chronically toxic to fish (including growth and reproduction)? 2) Is the benthic-feeding fish community structure from the toxic effects of Navy-related chemicals in prey items?	White croaker Fish community	1) Measure concentration of chemicals in prey items and compare to fish TRVs 2) Perform toxicity tests using field-collected prey items sediment and water or *in-situ* testing methods 3) Community analysis using biodiversity or dominance	1) Limited information from which to generate fish TRVs 2) Toxicity tests using field prey may not be feasible. 3) May not be feasible to assess predator communities in SF Bay	1) Very high degree of uncertainty in calculating abiotic to biotic accumulation phase. 2) Very high degree of uncertainty in calculating abiotic to biotic accumulation phase.
3º CONSUMERS X. Piscivorous Fish California Halibut, striped bass	1) Maintain a healthy, self-sustaining population in a balanced community by protecting the piscivorous fish from the bioaccumulative effects of chemicals in their food.	1) Will the consumption of Navy-related chemicals in prey items be acutely or chronically toxic to fish (including effects on growth or reproduction)? 2) Is the piscivorous fish community composition altered from the toxic effects of Navy-related chemicals in prey items?	California halibut Fish community	1) Model potential concentrations of chemicals in prey items, by applying a bioaccumulation factor potentially derived from literature and compare to TRV 2) Measure concentration of chemicals in prey items and compare to fish TRVs 3) Perform toxicity tests using field-collected prey items and water or *in-situ* testing methods 4) Community analysis using biodiversity or dominance	1) Bioaccumulation factor may be best estimate based on limited data. 2) California halibut, a common large piscivore, has wide range as adult. Young probably are present in SL for < 2 years 3) Young California halibut are substantially plankton and larval fish feeders, only becoming piscivorous at larger size. 4) Identifying an appropriate reference site probably not possible 5) May not be feasible to assess predator communities in SF Bay.	1) Use bioaccumulation factors calculated from OU-2 collected fish. Potentially need to collect additional BA data from other sites at OU-4 2) Very high degree of uncertainty in calculating abiotic to biotic accumulation phase

APPENDIX D: EXAMPLE PLANNING TABLE USED TO ASSIST WITH THE DEVELOPMENT OF THE CONCEPTUAL SITE MODEL REFINEMENT IN STEP 3 SUMMARY OF ASSESSMENT ENDPOINTS, RISK QUESTIONS AND MEASUREMENTS ENDPOINTS.

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Receptor Class and Specific Receptors	Assessment Endpoint	Risk Questions	Surrogate Species or Community	Measurement Endpoints	Uncertainties	Notes
3º CONSUMERS XI. Benthic-Feeding Birds Surf scoter, ruddy duck	1) Survival and reproduction of benthic-feeding birds? 2) Is the concentration of chemicals in prey at Seaplane Lagoon higher than the concentration elsewhere in the S.F. Bay?	1) Would the consumption of sediment-associated chemicals in benthic prey (such as snails and bivalves) and sediments cause either acute or chronic toxicity to benthic-feeding birds (particularly surf scoters)? 2) Would the consumption of sediment-associated chemicals in benthic prey and sediments cause adverse effects on reproductive capacity of benthic-feeding birds (particularly surf scoters)? 3) Are chemicals in prey at Seaplane Lagoon specific to the Navy, or general SF Bay conditions?	Surf scoter	1) Model potential concentrations of chemicals in prey items by applying bioaccumulation factors derived from literature sources and comparing these doses to bird TRVs. 2) Measure concentration of chemicals in prey items and comparing them to bird TRVs 3) Compare concentrations of chemicals in *Macoma nasuta* to reference locations in SF Bay	1 and 2) Area-use-factor for surf scoters and other surrogates are estimates. Uncertainty factors associated with bioaccumulation factors may over-or under-represent field conditions. TRVs developed for laboratory animals may not be accurate surrogates for wildlife. Sediment ingestion rates for surf scoters are uncertain. 3) Identifying a reference site controlled for other physical variables	
3º CONSUMERS XII. Piscivorous Mammals Harbor seal	1) To protect the reproductive ability of piscivorous mammals from bioaccumulating chemicals in their food sources.	1) Are sediment-associated chemicals in prey fish present at concentrations reproductively toxic to mammals (particularly harbor seals)?	Harbor seal	1) Model potential concentrations of chemicals in prey items by applying bioaccumulation factors derived from literature sources and comparing these doses to mammal TRVs 2) Measure concentration of chemicals in prey items and comparing them to mammal TRVs 3) Compare concentration of chemicals in prey fish in Seaplane Lagoon to reference sites in the SF Bay	1) Area-use-factor is less than 100%. Species ranges widely. 2) May be infeasible to determine acceptable reference site 3) TRVs were developed for laboratory animals, which may not be suitable surrogates for wildlife. 4) Sediment ingestion rate is uncertain	1) High degree of uncertainty with this species
3º CONSUMERS XIII. Piscivorous Birds Cormorant, pelican, and terns	1) Is the concentration of chemicals in prey items in Seaplane lagoon adversely affecting the health (either survival or reproduction) of piscivorous birds? 2) Is the concentration of chemicals in prey at Seaplane Lagoon higher than the concentration elsewhere in the S.F. Bay? 3) Population recruitment	1) Are sediment-associated chemicals in prey fish present at concentrations reproductively toxic to birds (particularly terns, cormorants or pelicans)? 2) Would the consumption of sediment-associated chemicals in fish cause adverse effects on reproductive capacity of piscivorous birds (particularly Least terns)? 3) Are chemicals in fish at Seaplane Lagoon specific to the Navy, or general SF Bay conditions? 4) Are chemicals in fish in Seaplane Lagoon causing lower recruitment in least terns?	Least tern	1) Model potential concentrations of chemicals in prey items by applying bioaccumulation factors derived from literature sources and comparing these doses to bird TRVs. 2) Measure concentration of chemicals in prey items and compare to bird TRVs. 3) Compare concentration of chemicals in prey fish in Seaplane Lagoon to reference sites in the SF Bay 4) Compare eggs/nest, fledglings-per-nest to reference sites.	1) Area-use-factor may be less than 100%. 2) TRVs were developed for laboratory animals, which may not be suitable surrogates for wildlife 3) Most sensitive endpoint (for example eggshell thinning) may not be sufficiently known. 4) Nest success depends on food supply, predation and other factors which may not be consistent across reference areas.	1) Need to determine which of least tern, double crested cormorant and brown pelican are most sensitive based on ingestion to weight ratio.

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:

U.S. Navy
Southwest Division
Naval Facilities Engineering Command

Prepared By:

Space and Naval Warfare Systems Center
(SPAWAR)
San Diego, California

With:

Arthur D. Little, Inc.
Encinitas, California

And

MEC Analytical Systems, Inc.
Carlsbad, California

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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.

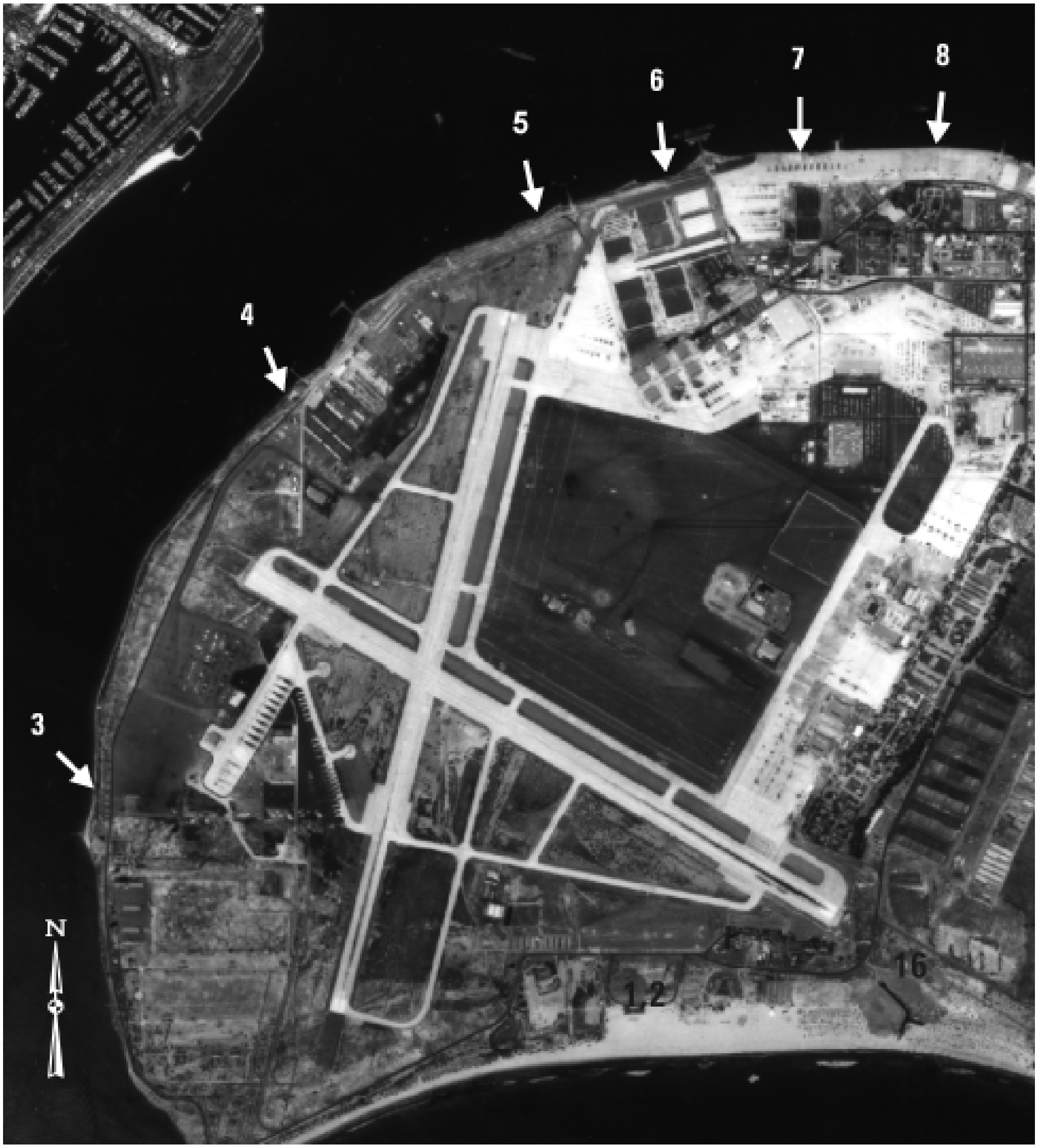


Figure 1-3. Aerial photograph of NAS North Island, 1975. Source: Aerial Fotobank, Inc.

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.

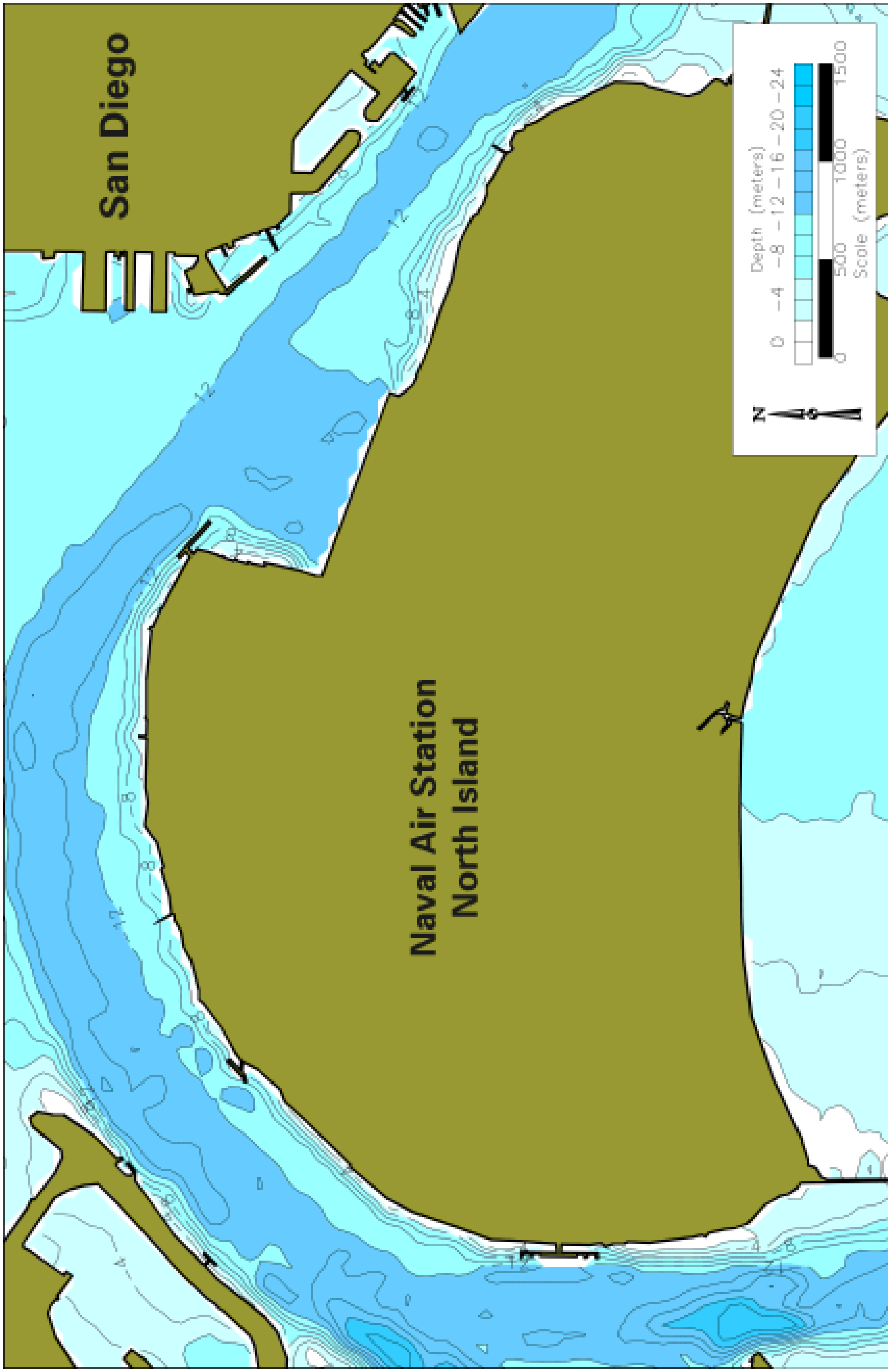


Figure 1-5. Bathymetry along NAS North Island.

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.

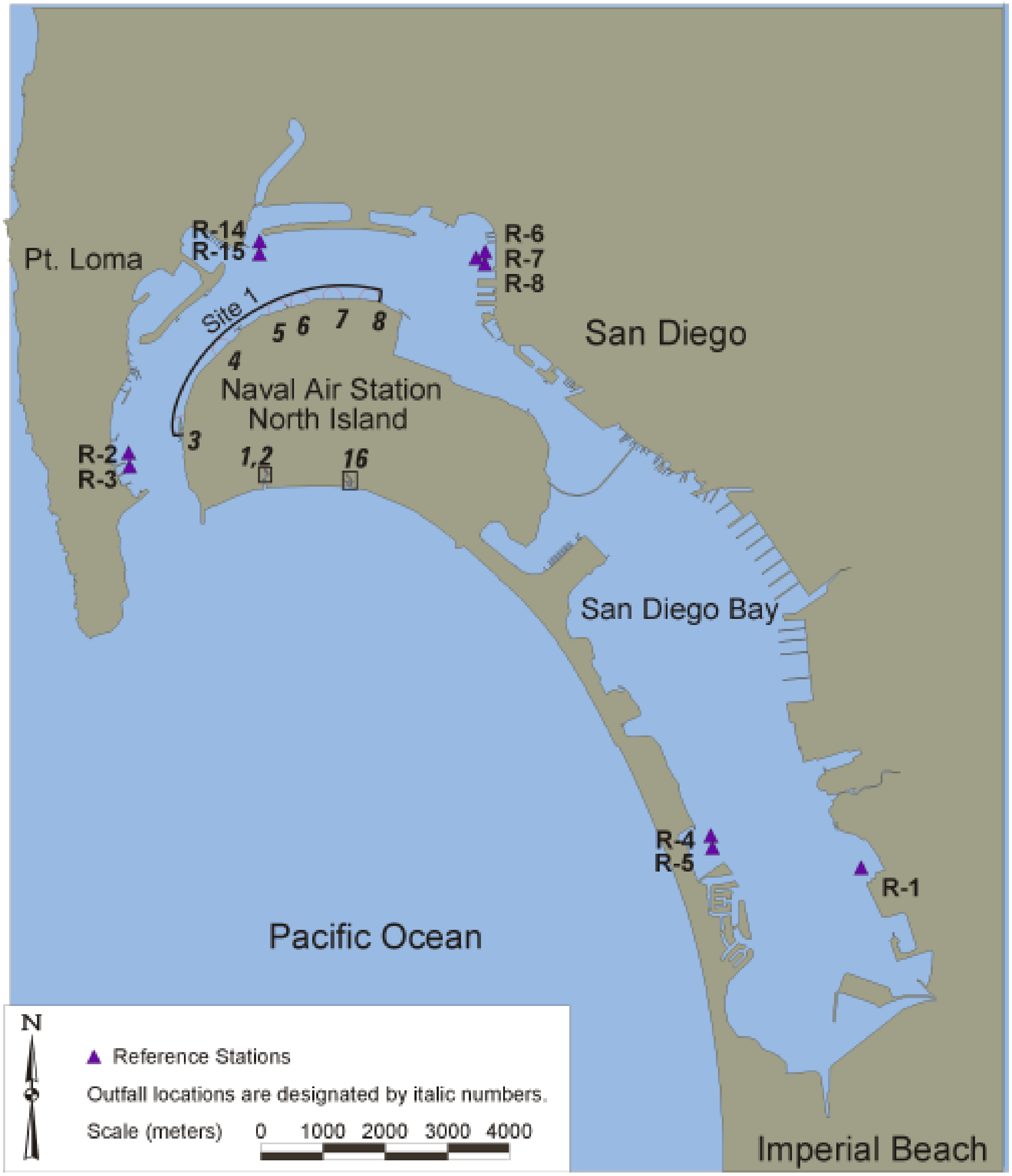


Figure 2-1. Locations of the 9 Site 1 outfalls around the perimeter of NAS North Island and the 10 reference stations in San Diego Bay.

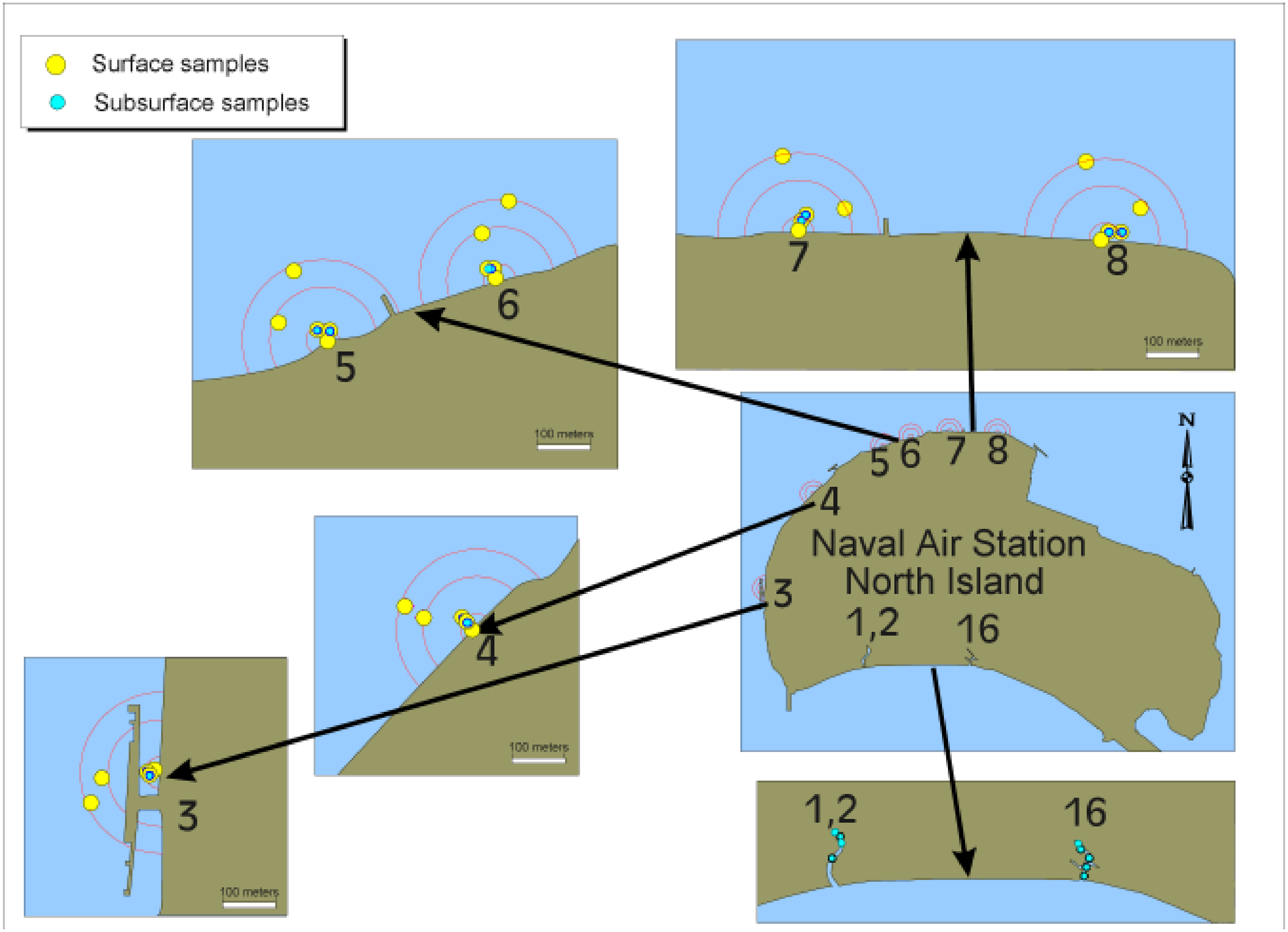


Figure 2-2. NAS North Island sampling stations identified by outfall. Semicircle sampling arcs are shown for 30, 100, and 150 meter stations.

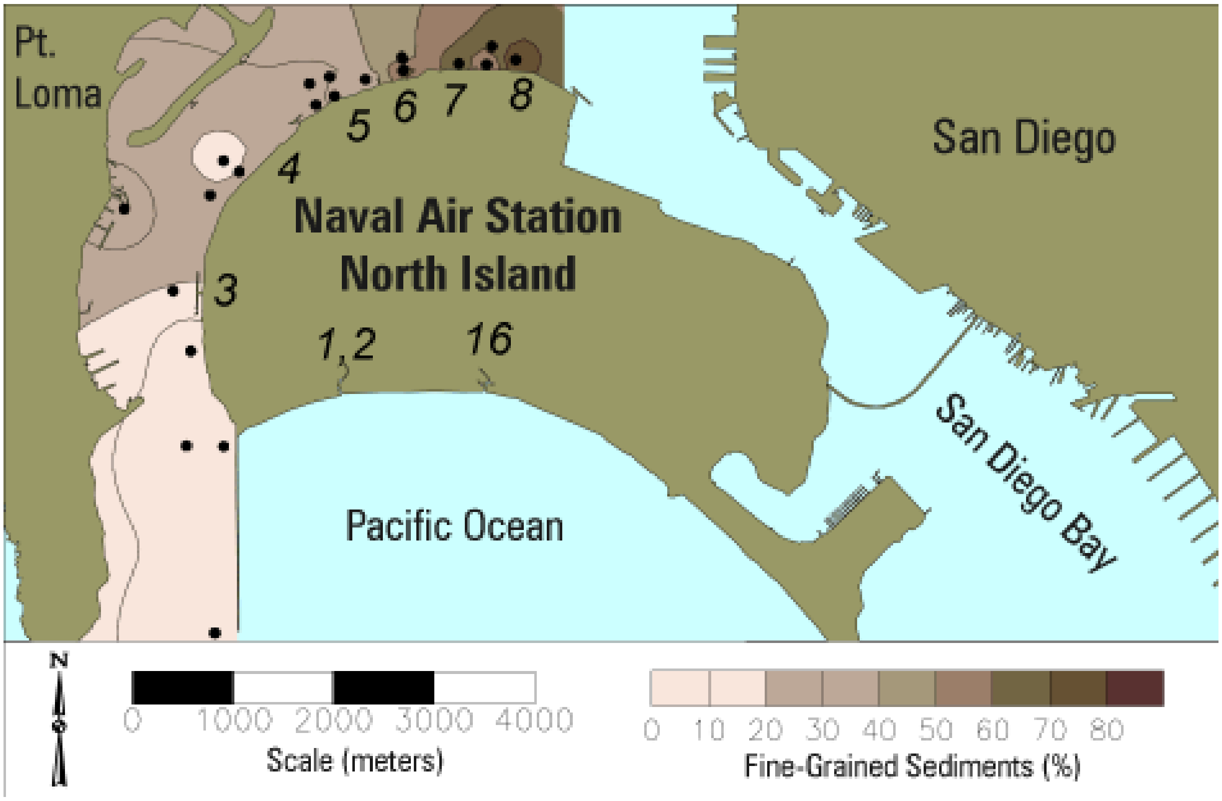


Figure 2-3. Fine-grained sediment distributions around NAS North Island.

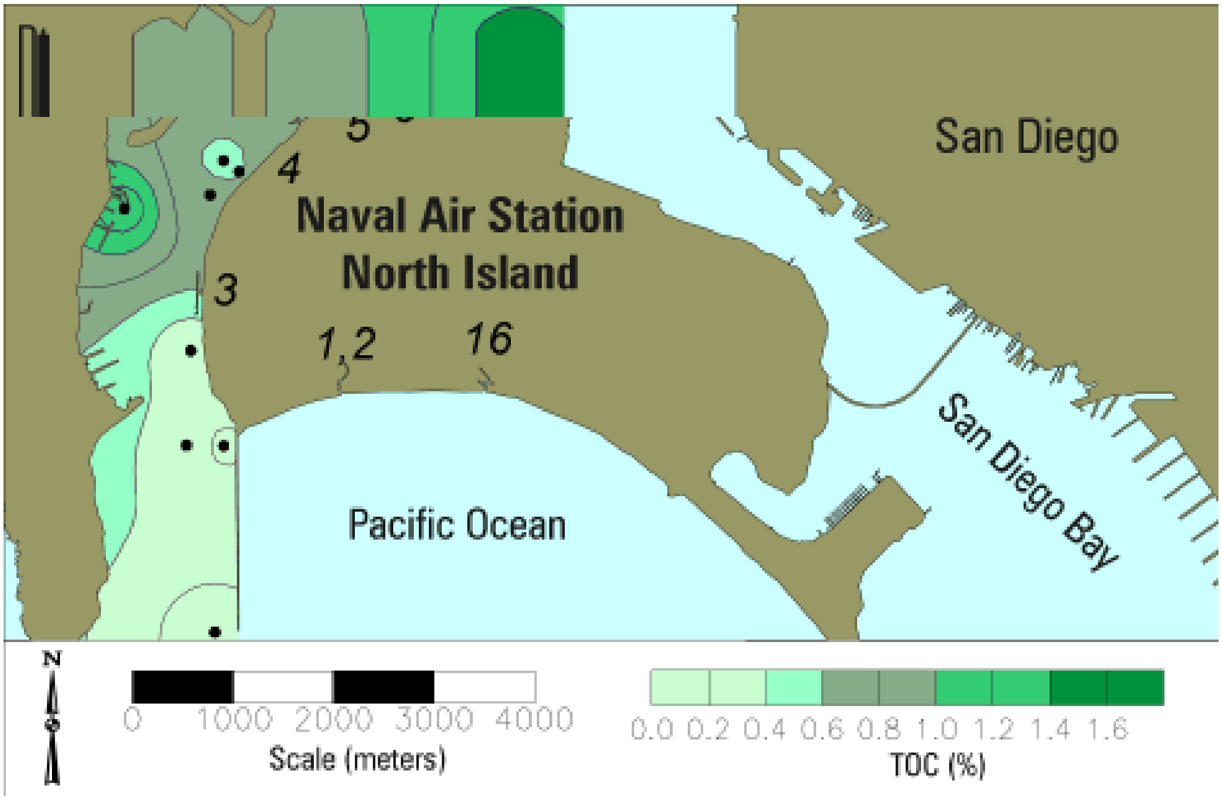


Figure 2-4. Total organic carbon distribution around NAS North Island.

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.

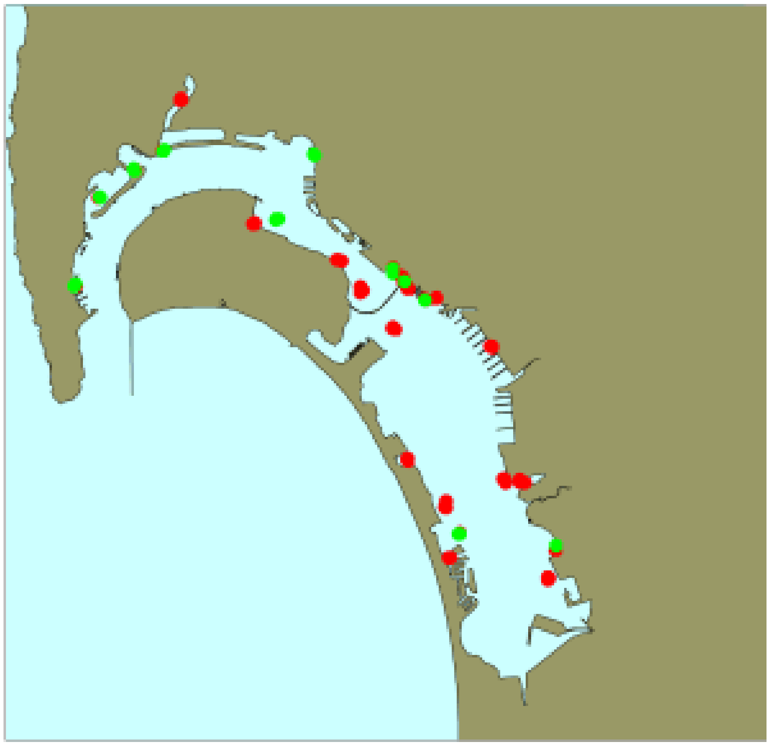
2.1.2 Subsurface Sediments

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.

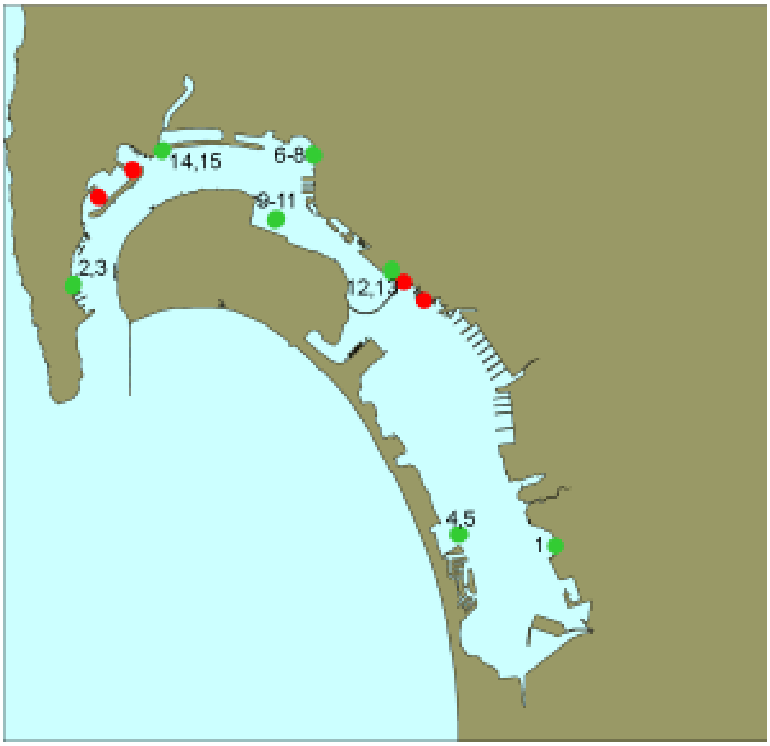
Table 2-1. Station identification, distance from outfall, and location.

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

(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 Rqualifier (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 $p \leq 0.0005$ too extreme. Not knowing the number of independent contaminant "groups" or degree of correlation among them, a $p \leq 0.01$ 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.

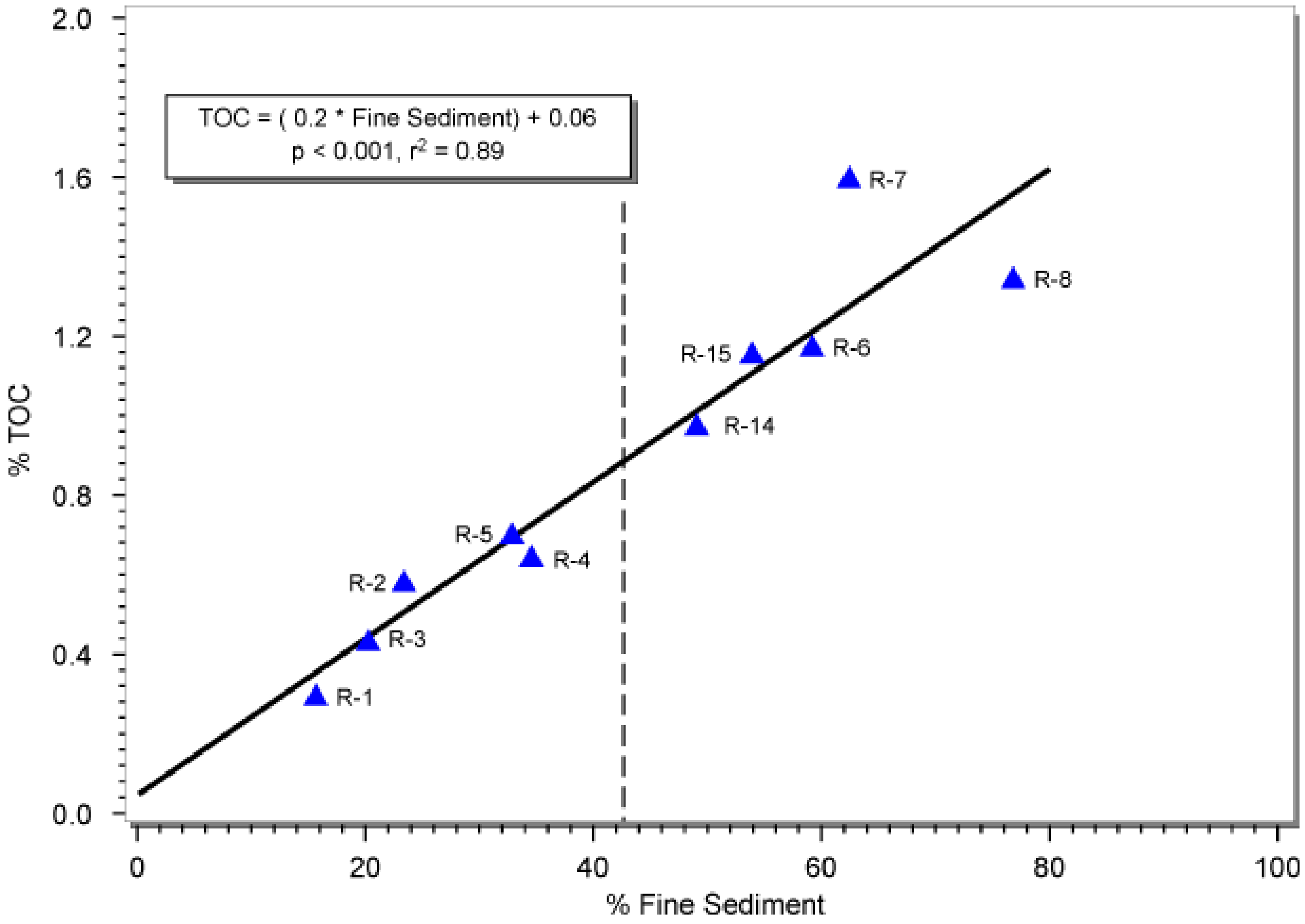


Figure 3-1. **Regression of percent total organic carbon with percent fine (<62 μm or <0.062 mm) sediment.**

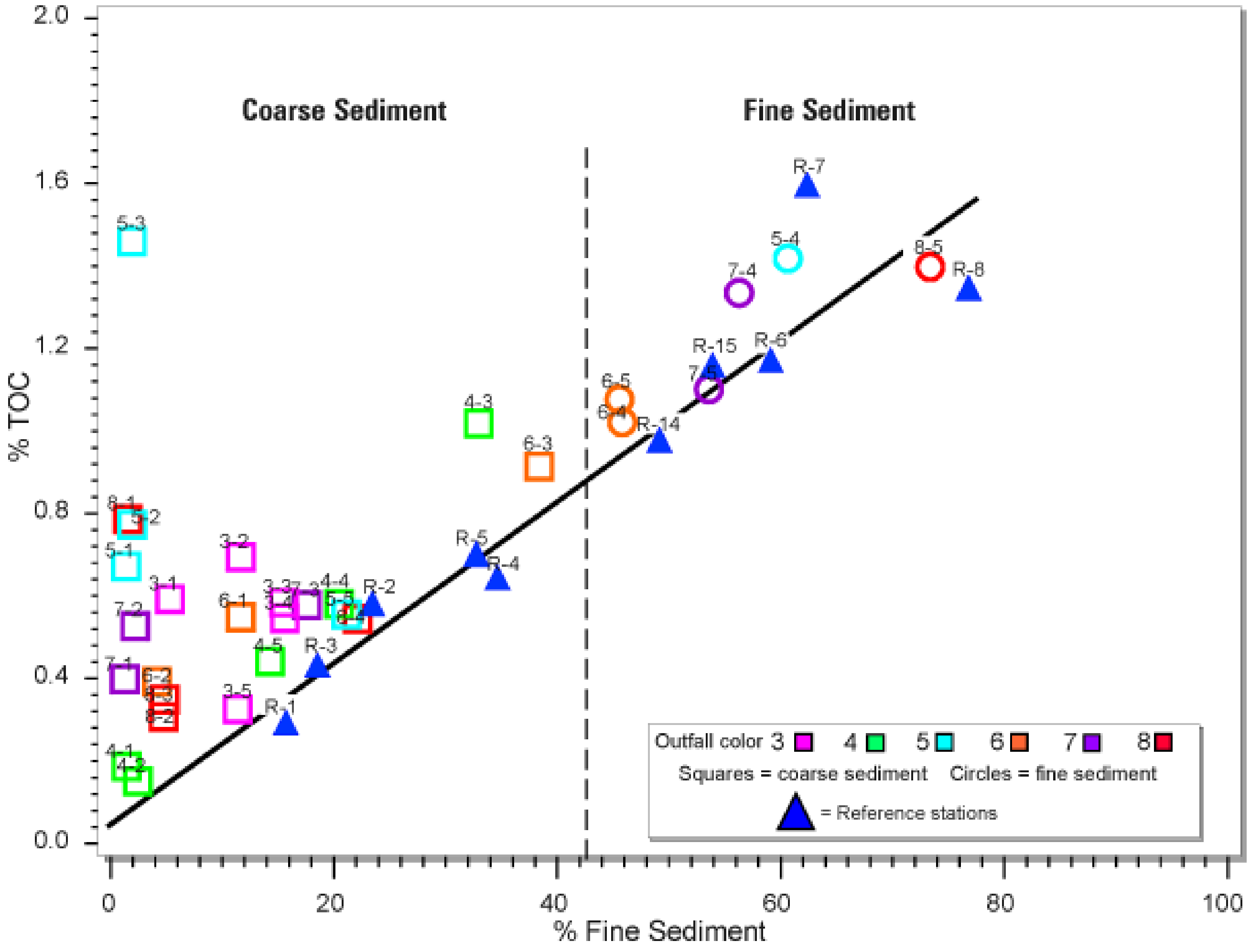


Figure 3-2. Percent total organic carbon with percent fine (<62 µm or <0.062 mm) sediment by station.

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

Coarse-Grain Sediment Group

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

		% Fines		% Fines
Outfall Stations	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

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

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

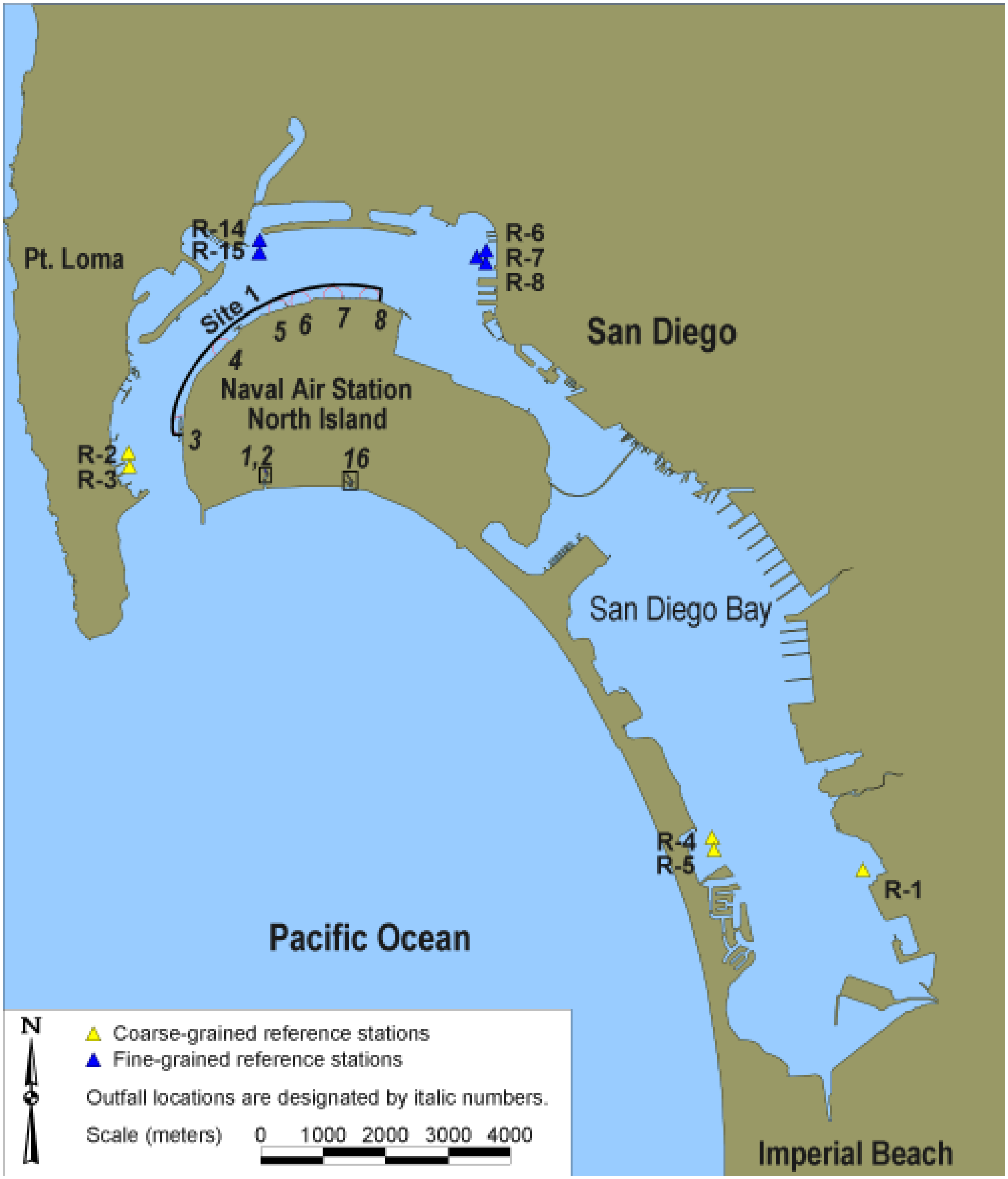


Figure 3-3. Locations of coarse-grained and fine-grained reference stations in San Diego Bay.

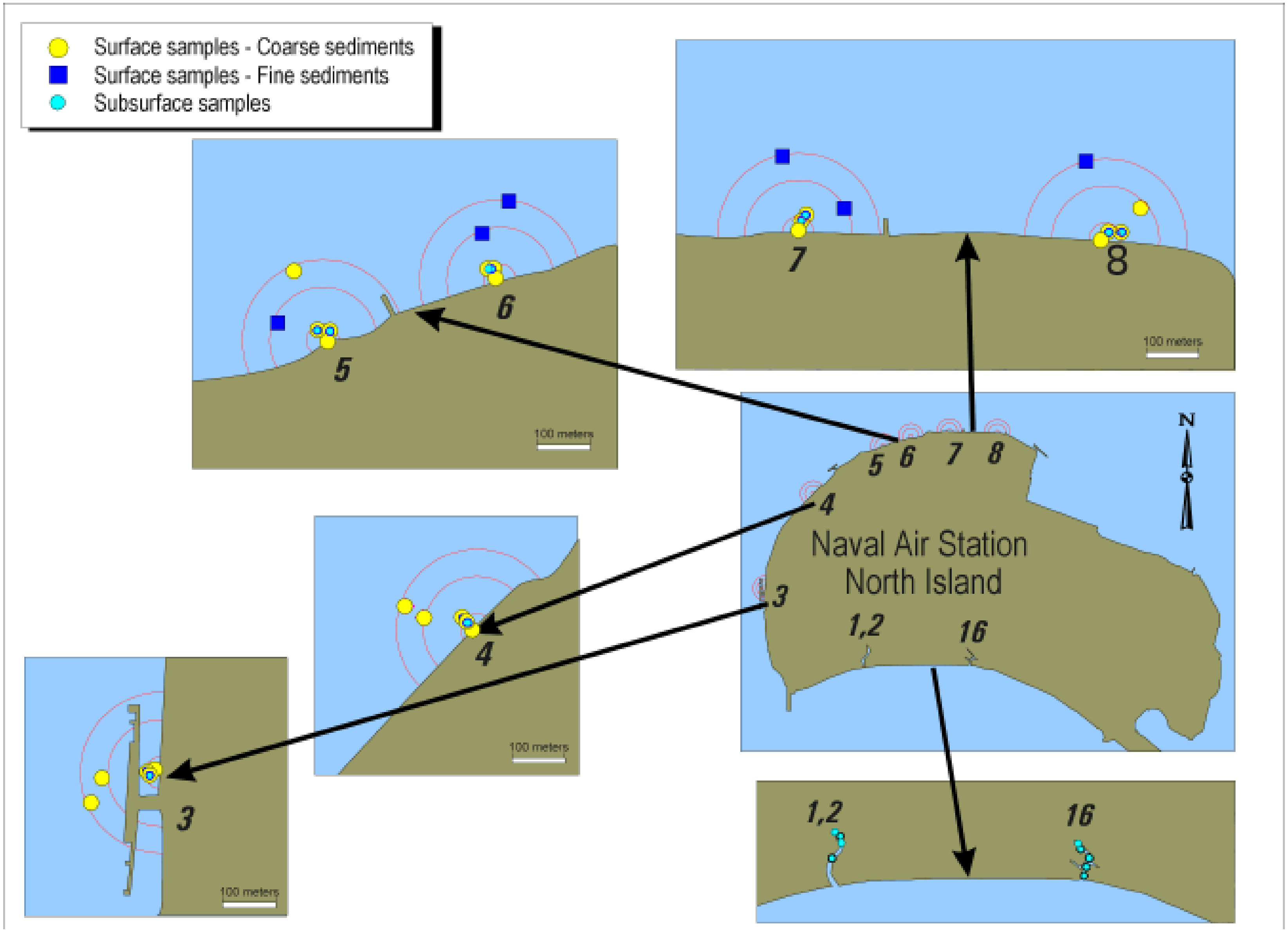


Figure 3-4. NAS North Island sampling stations identified by outfall and sediment type group.

Section 4 NATURE AND EXTENT OF CONTAMINATION

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 ($\geq$ 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.

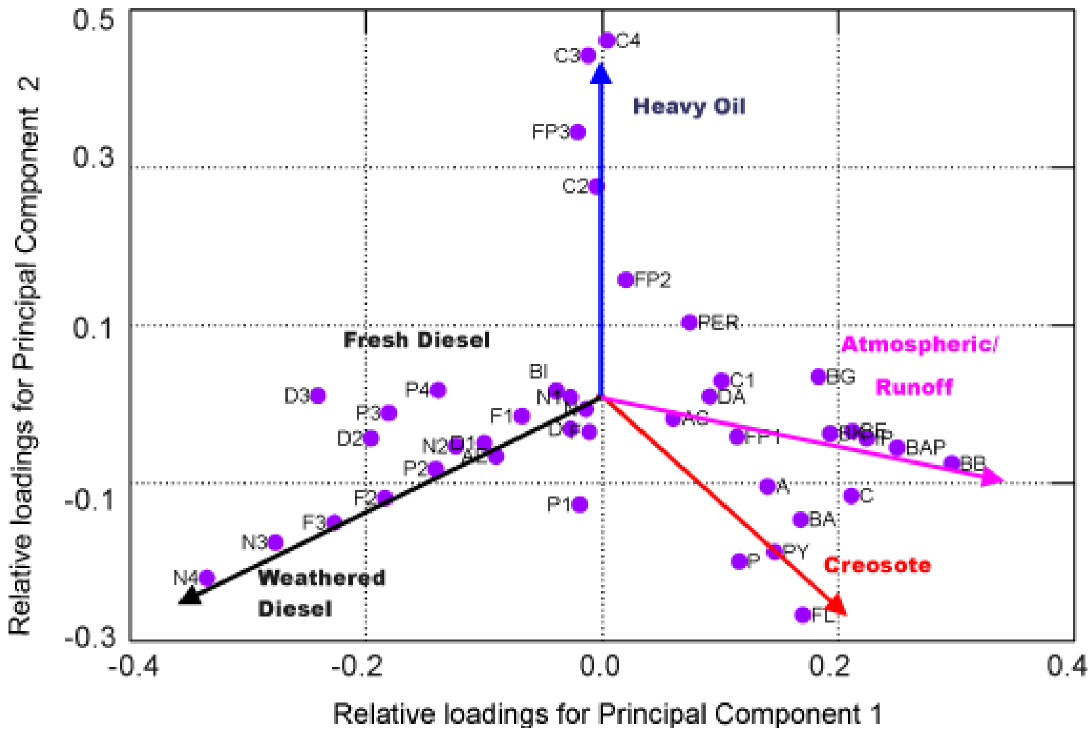


Figure 4-1. Relative loadings for principal components 1 and 2.

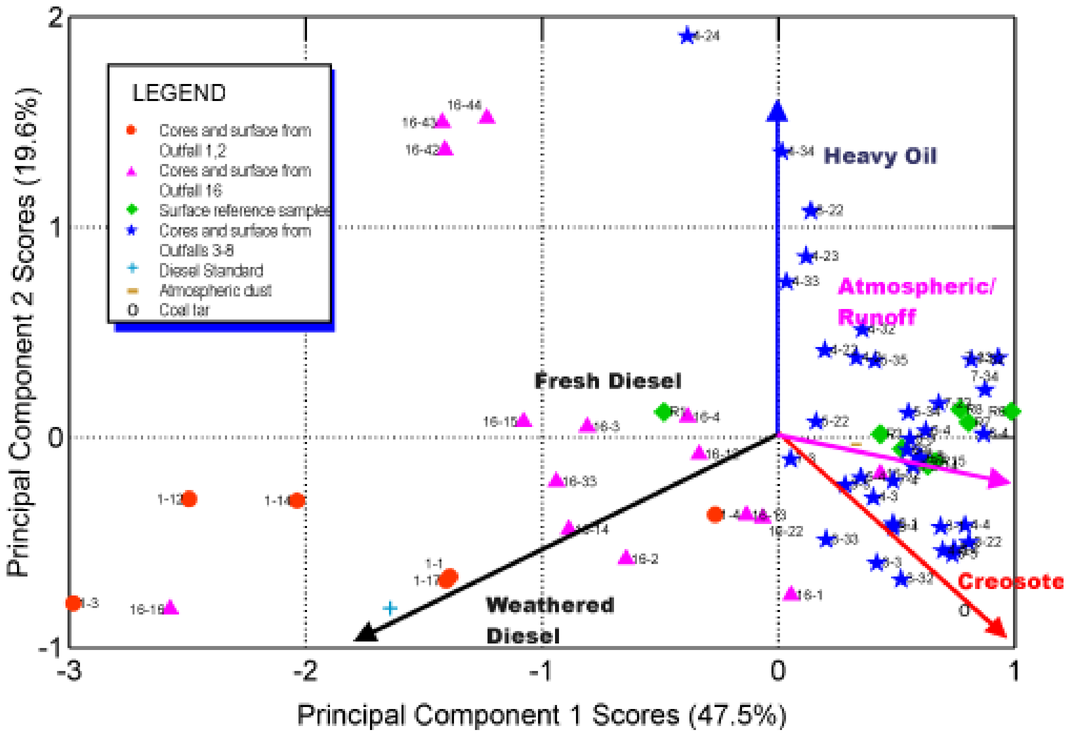


Figure 4-2. Station scores for principal components 1 and 2.

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.

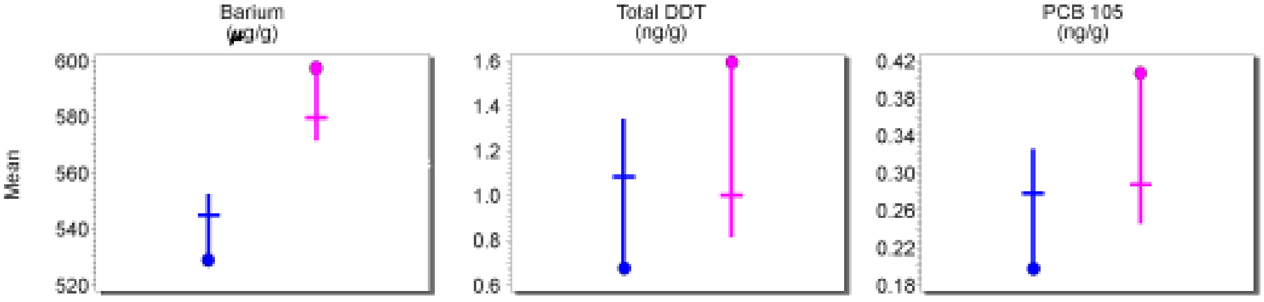
	Coarse Grain Outfall Stations	Coarse Grain Reference Stations	Fine Grain Outfall Stations	Fine Grain Reference Stations
n	24	5	6	5
Total DDT	2.21 ± 0.37**	0.70 ± 0.09	5.92 ± 1.10	8.15 ± 2.26
Total BHC	0.27 ± 0.15	0.39 ± 0.10	0.47 ± 0.21	1.54 ± 0.76
Total Chlordane	0.38 ± 0.05	0.26 ± 0.13	0.70 ± 0.15	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

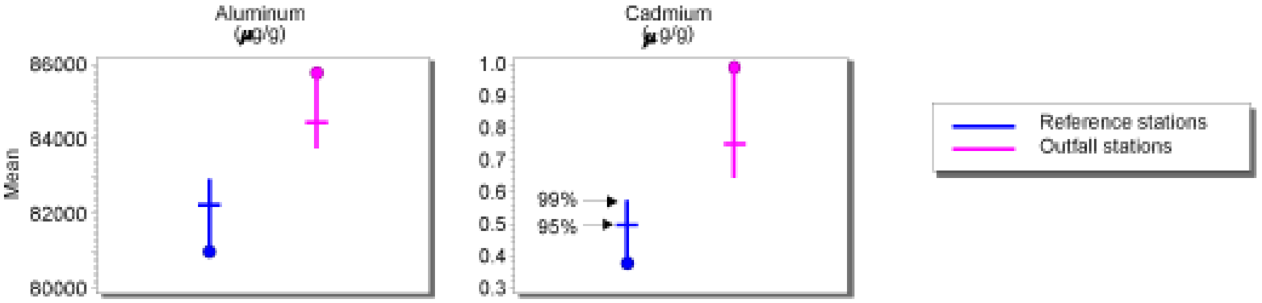
Total DDT concentrations at eight individual coarse outfall stations were statistically ($p \leq 0.01$) higher than at the coarse reference stations (Figure 4-4). The surface distribution of total DDT is shown in Figure 4-6. Total DDT concentrations for individual outfall stations ranged from 1.5-6.5 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.



Coarse Sediments



Fine Sediments

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.

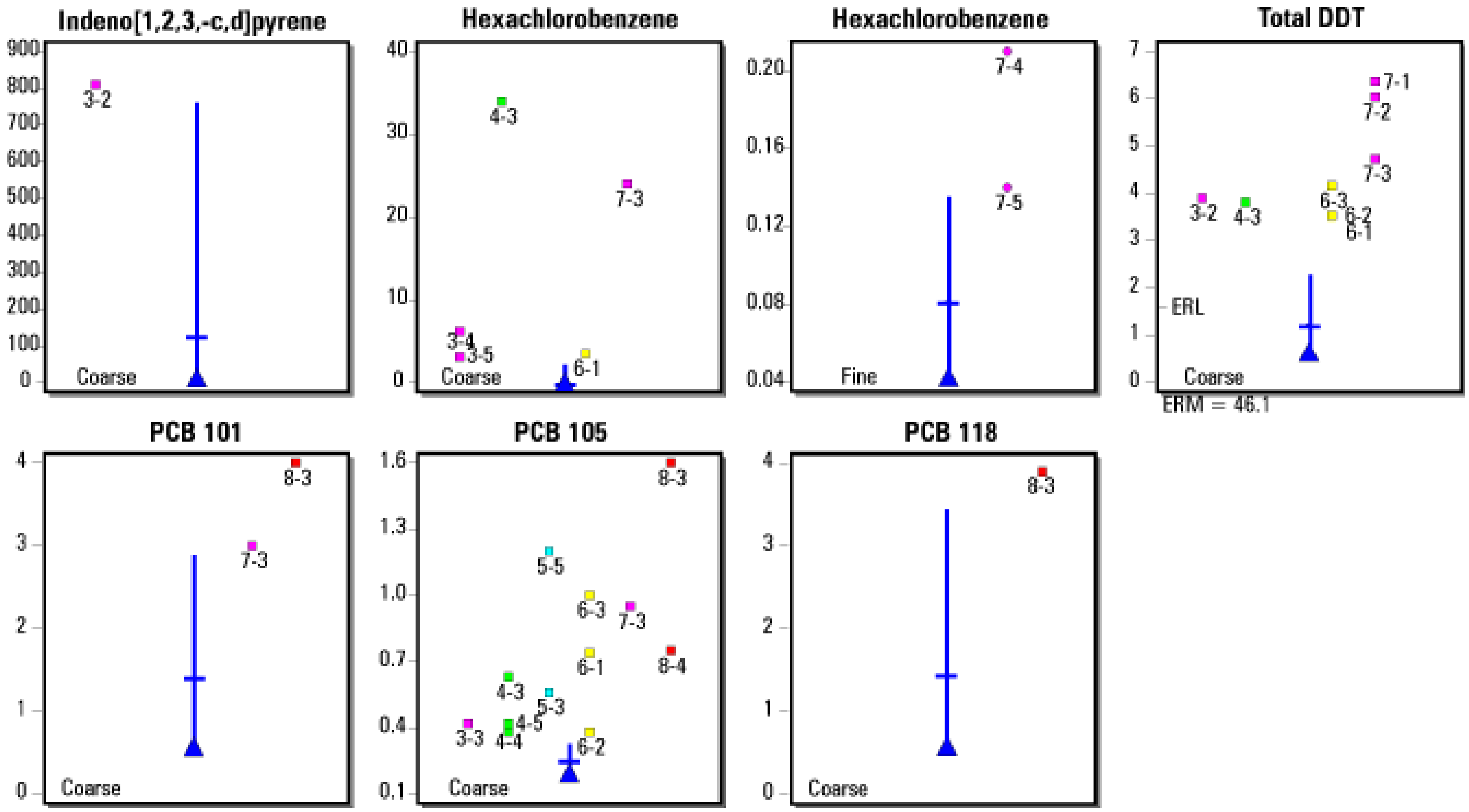


Figure 4-4. Organic chemical concentrations (ng/g) for outfall surface sediments above the 99% predictive limit of corresponding reference stations.

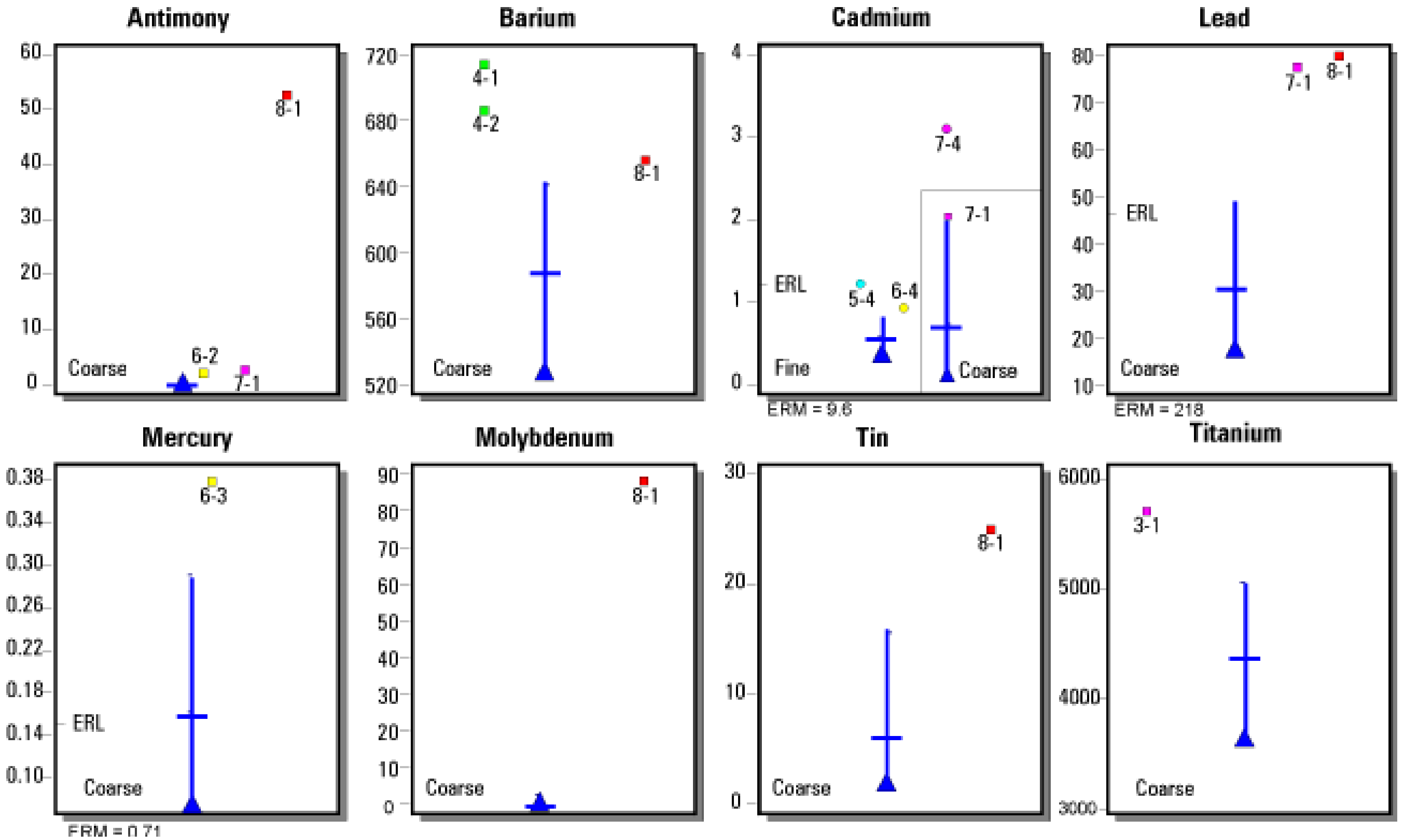
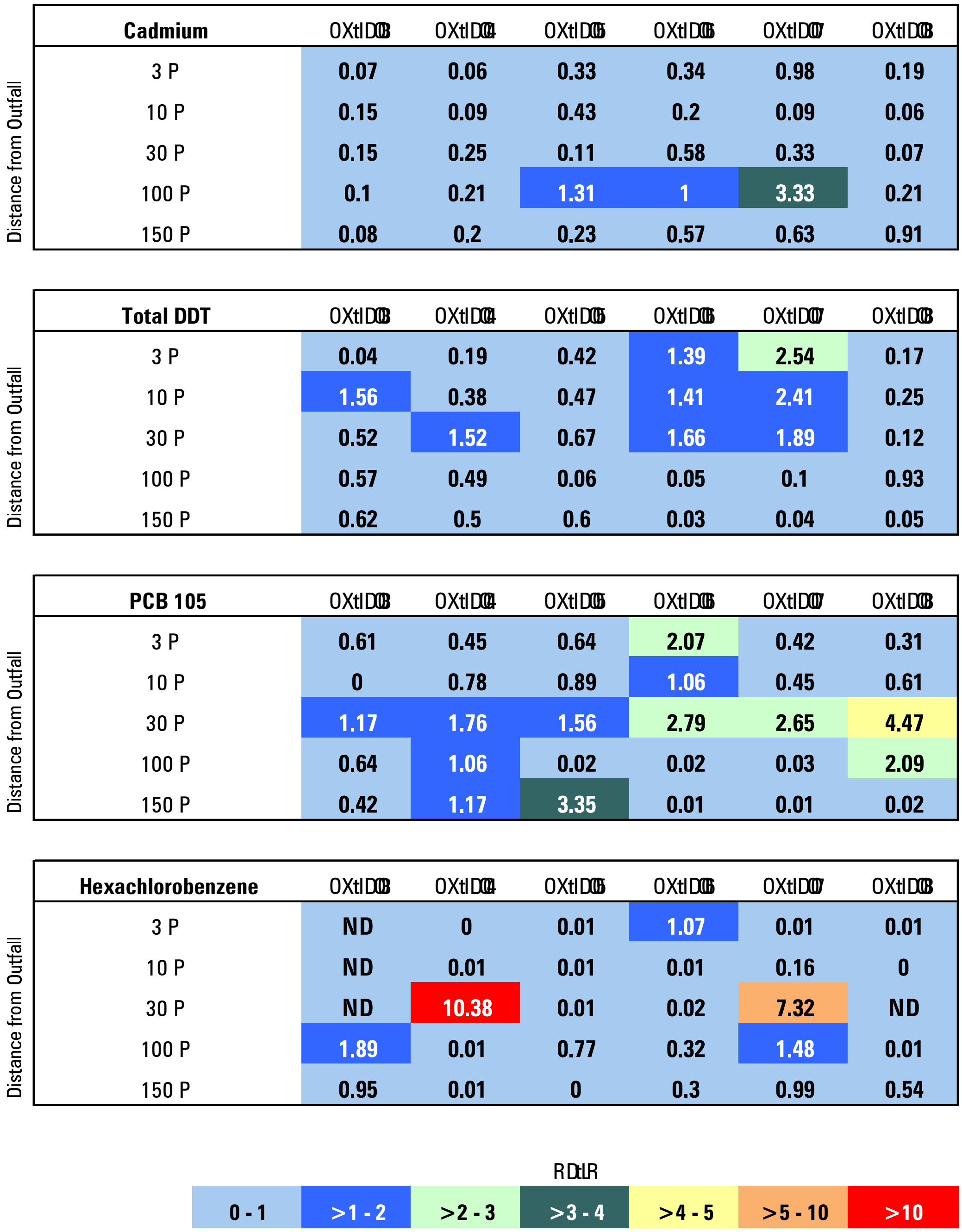


Figure 4-5. Metal concentrations (μg/g) for outfall surface sediments above the 99% predictive limit of corresponding reference stations.



Distance from Outfall

Cadmium	0XtlD03	0XtlD04	0XtlD05	0XtlD06	0XtlD07	0XtlD08
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	0XtlD03	0XtlD04	0XtlD05	0XtlD06	0XtlD07	0XtlD08
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	0XtlD03	0XtlD04	0XtlD05	0XtlD06	0XtlD07	0XtlD08
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	0XtlD03	0XtlD04	0XtlD05	0XtlD06	0XtlD07	0XtlD08
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

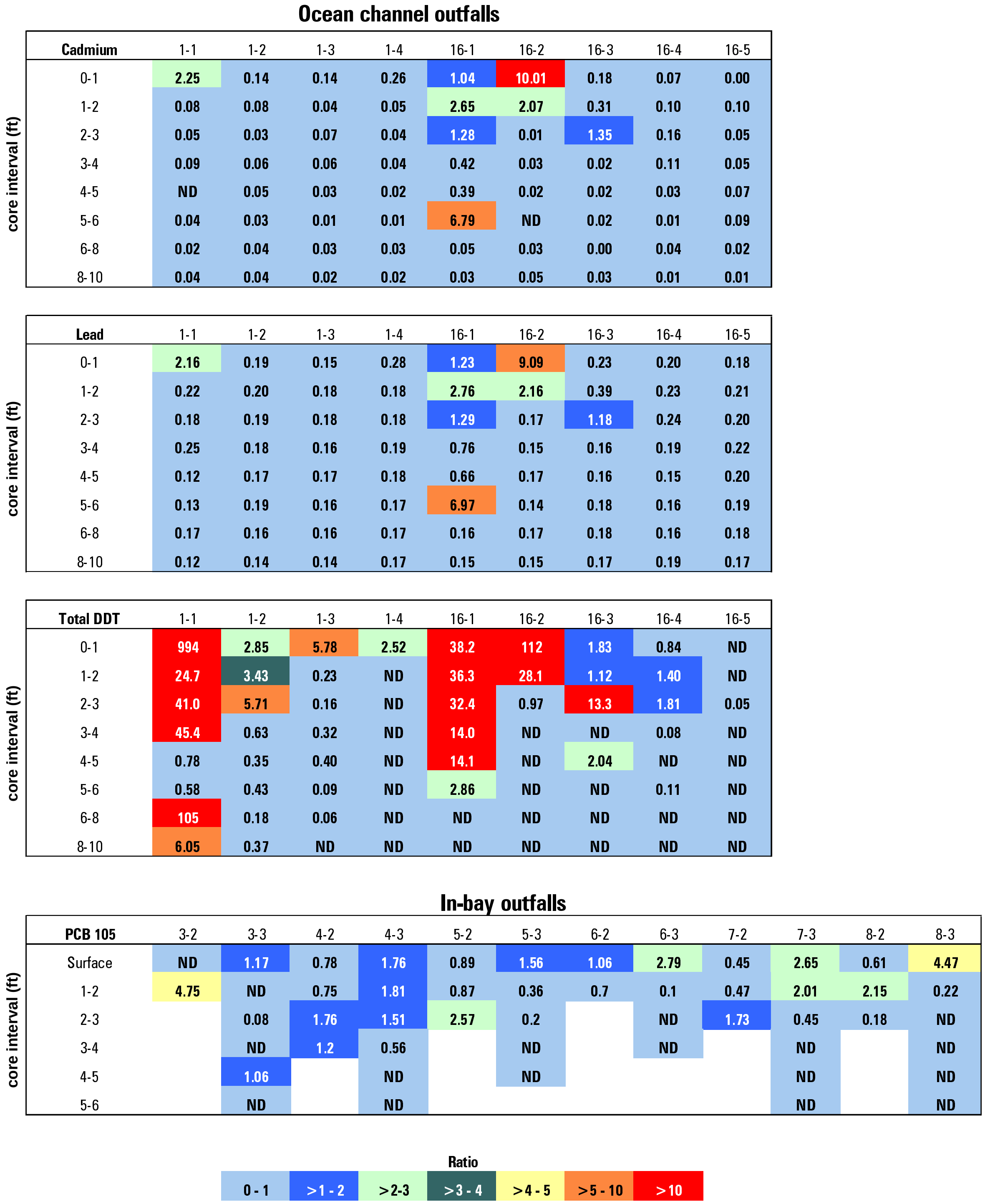
As discussed above, a number of chemicals were not included in statistical comparisons due to insufficient detectable concentrations (i.e., < 3) in either the coarse or fine sediment reference group. These chemicals are shown in Table 4-5 with the number of detected values for the reference and outfall 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

core interval (ft)

Cadmium	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

core interval (ft)

Lead	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

core interval (ft)

Total DDT	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

core interval (ft)

PCB 105	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)

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

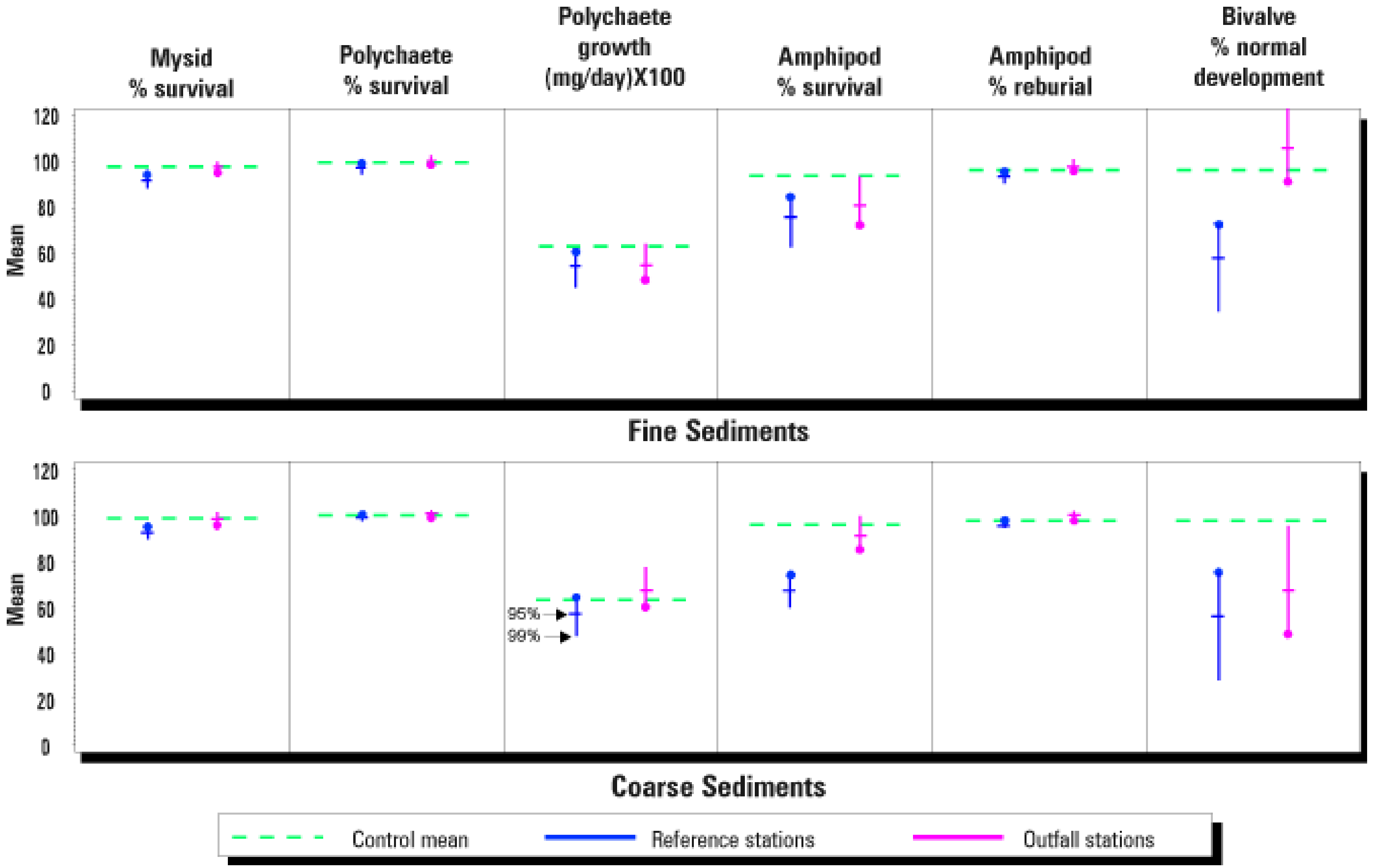


Figure 5-1. T-test results for toxicity endpoints for fine and coarse sediment groups. Confidence intervals are shown for 95% and 99% one-tail tests. All statistical results were insignificant at $p \leq 0.05$.

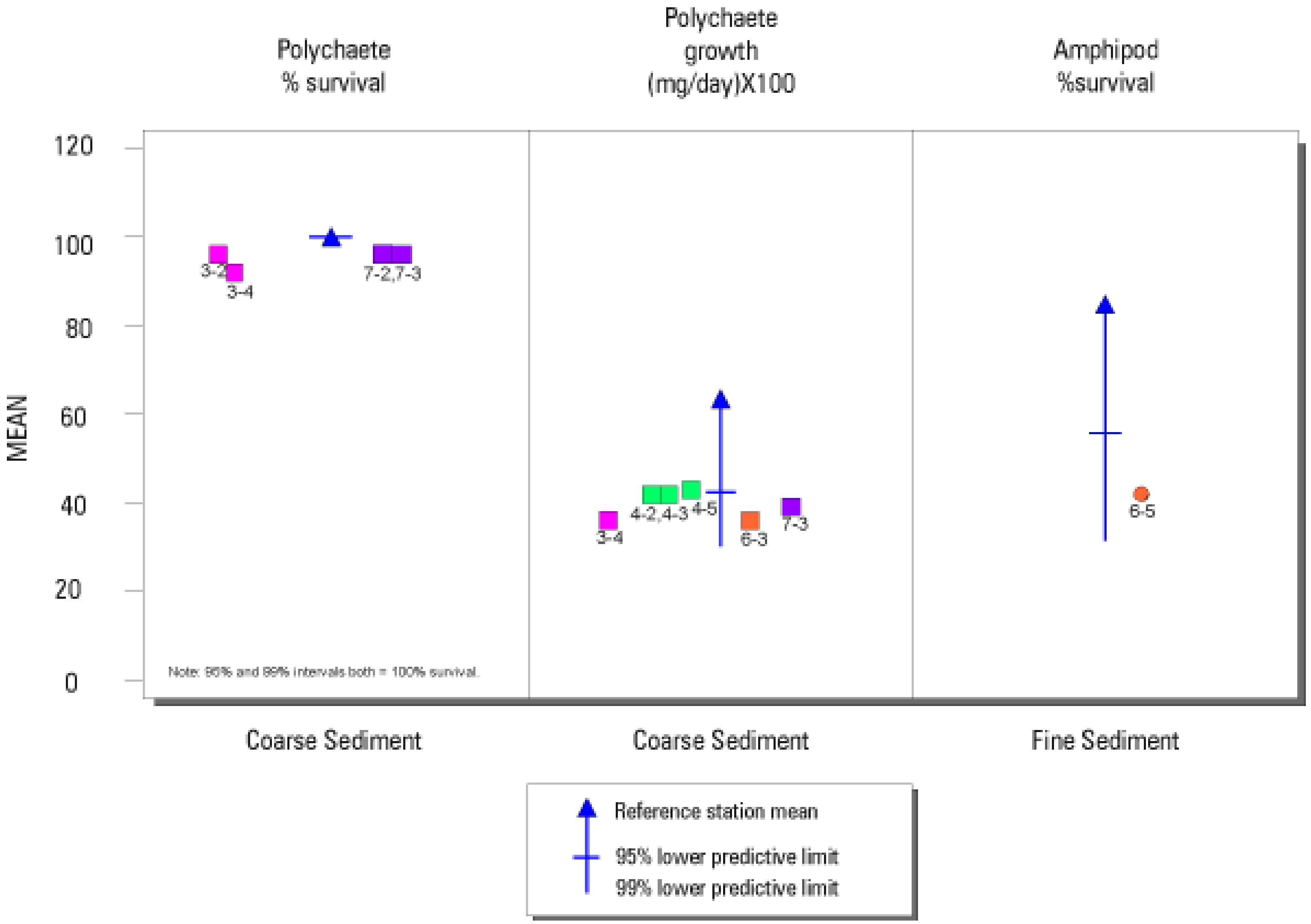


Figure 5-2. Toxicity results below the 95% predictive limit for corresponding sediment group reference stations. Only polychaete survival at four coarse sediment outfall stations was significantly lower than the reference 99% predictive limit.

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.

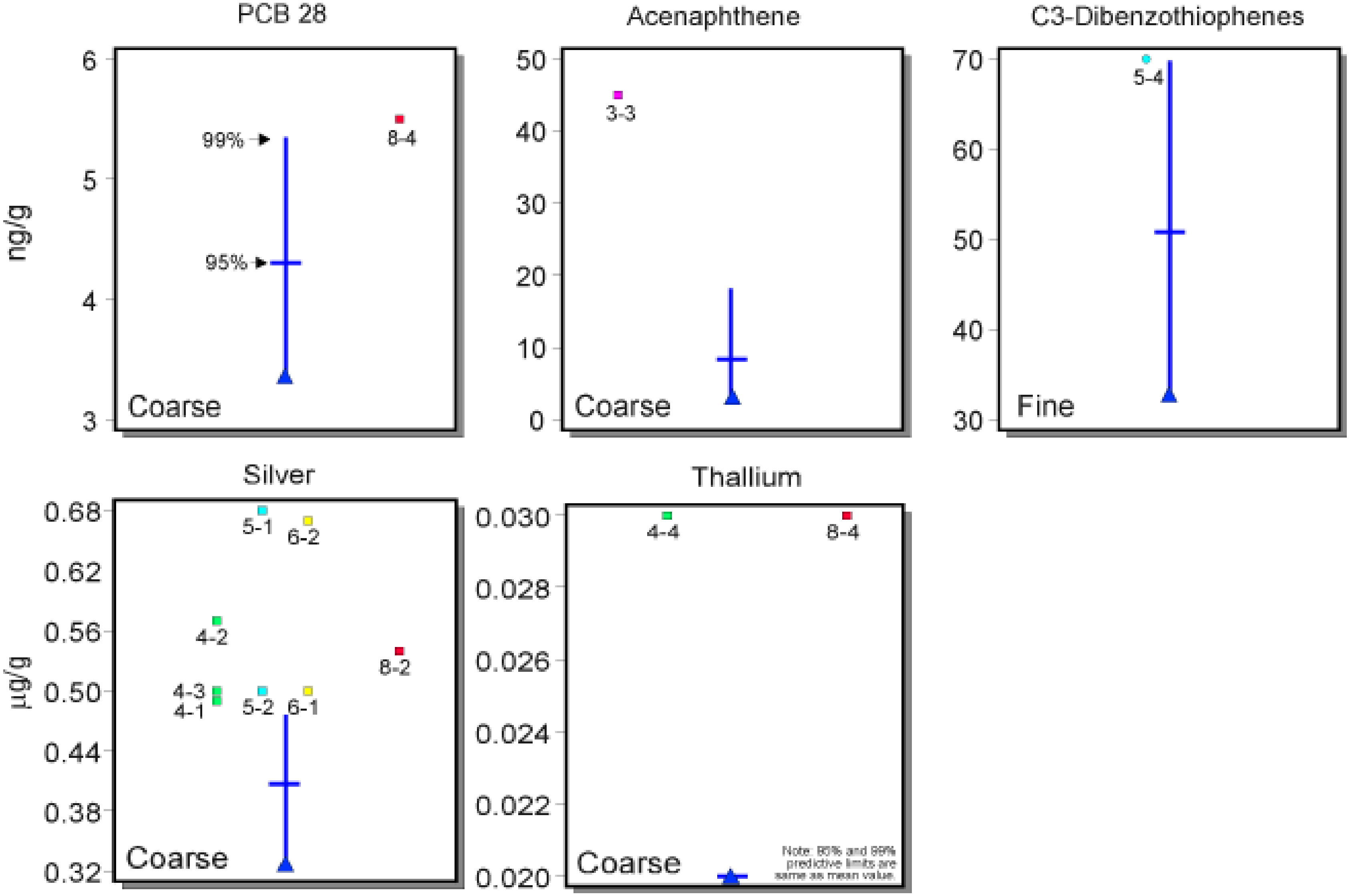


Figure 5-3. Chemical concentrations in tissues exposed to outfall sediments above the 99% predictive limit for corresponding reference stations.

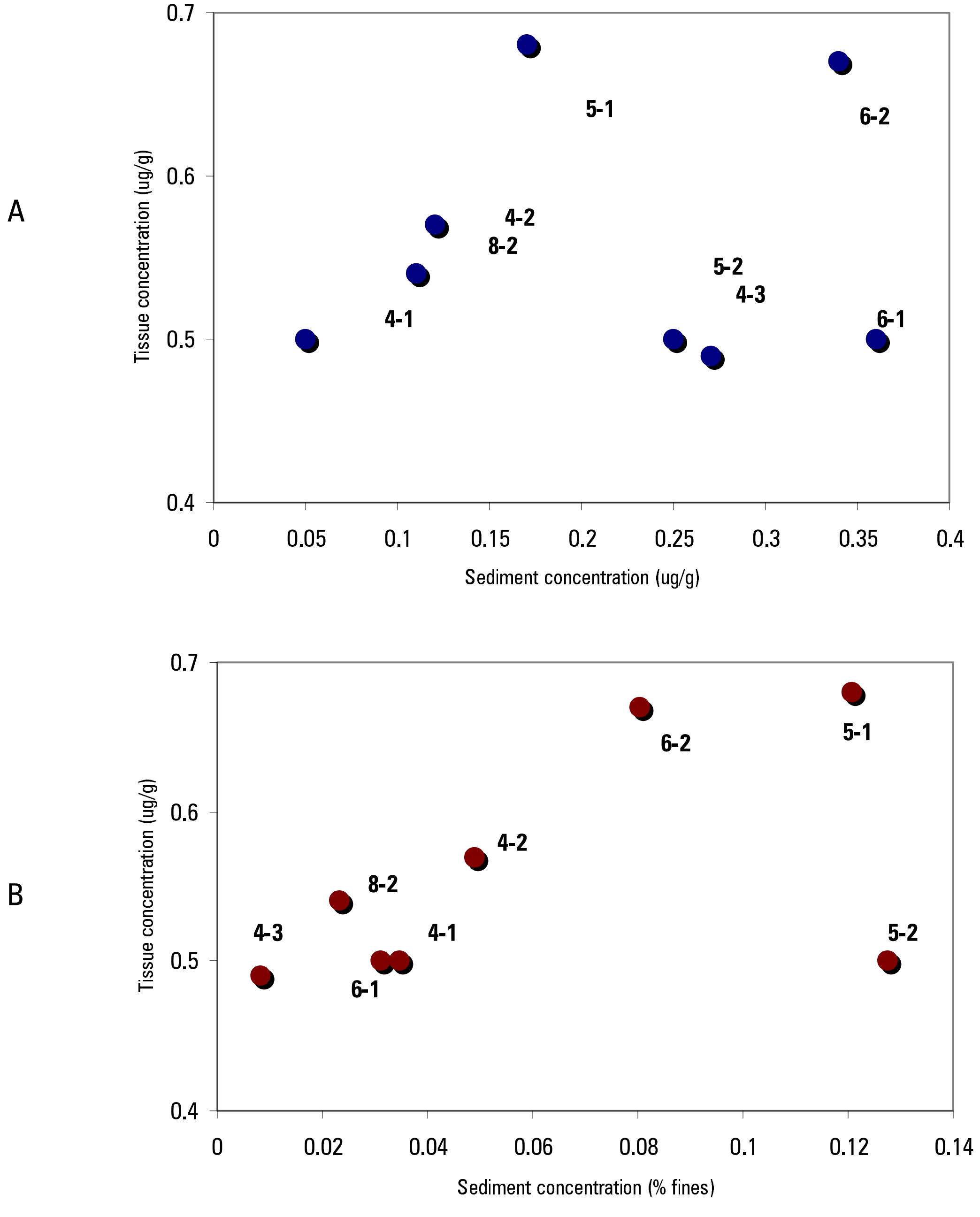


Figure 5-4. A: Silver concentration in clam tissue vs. sediment for outfall stations exceeding the coarse reference station mean tissue concentration. B: Same plot as A, except sediment silver concentration is normalized by sediment percent fines.

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.

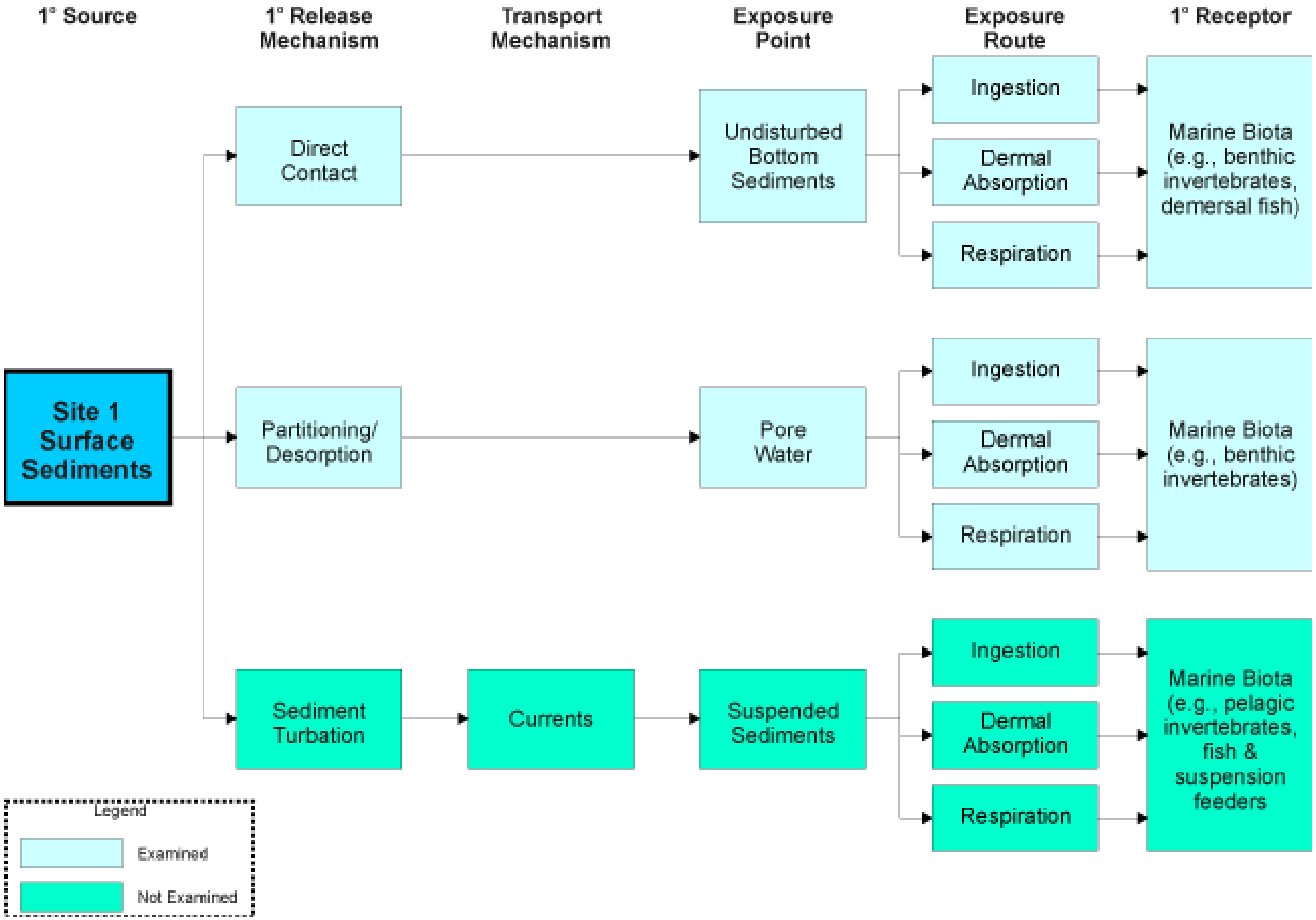


Figure 6-1. Conceptual site model for ecological receptors - NAS North Island.

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 (1×10^{-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.

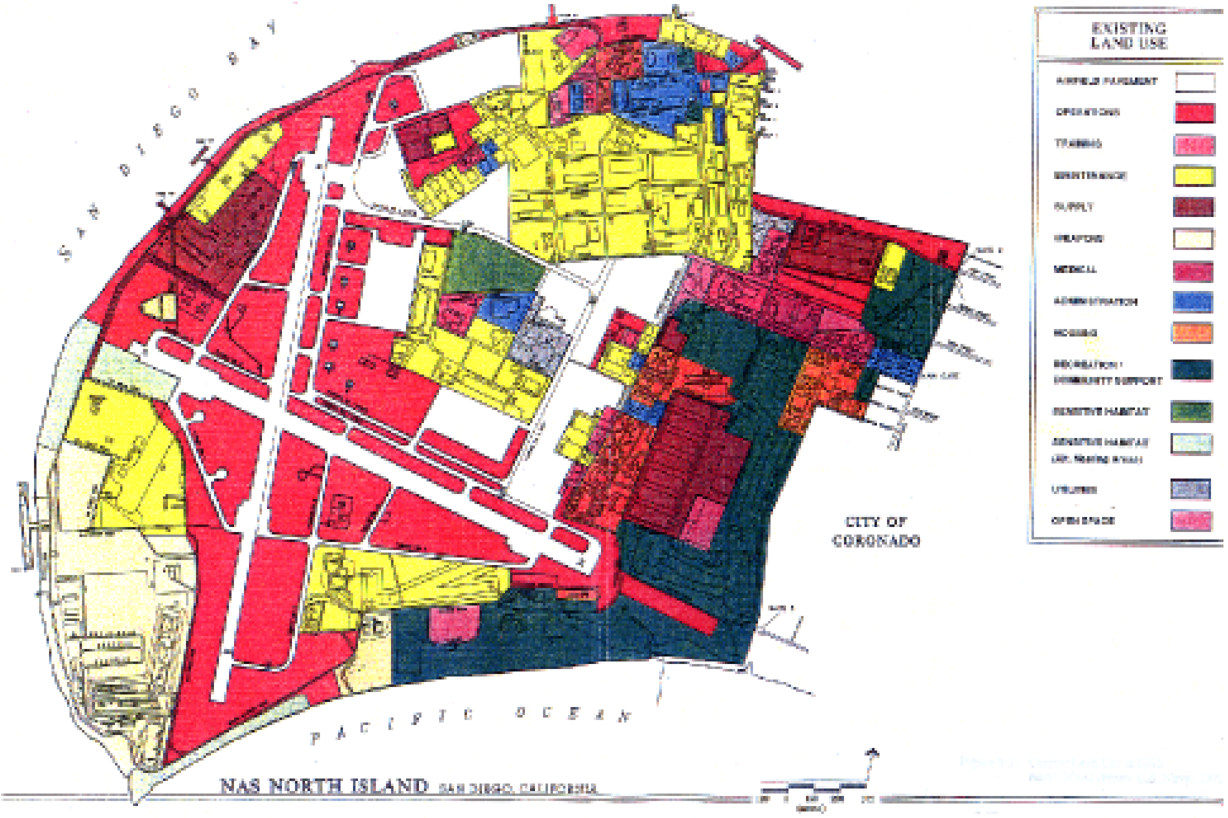


Figure 6-2. Current land use at NAS North Island (from U.S. Navy 1991).

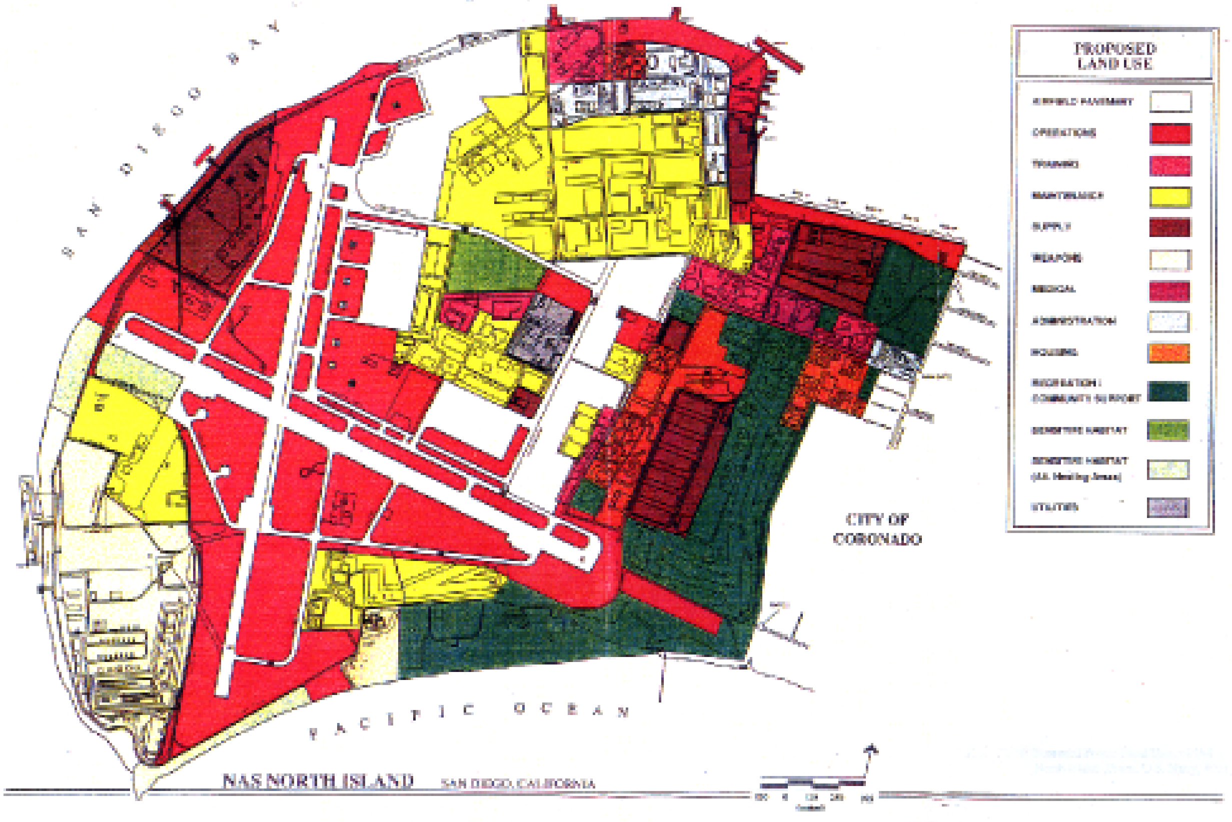


Figure 6-3. **Potential future land use at NAS North Island (from U.S. Navy 1991).**

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 x 10^{-5}, the maximum value of 44 x 10^{-6} occurring at subtidal station 8-1, the maximum intertidal value of 16 x 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.

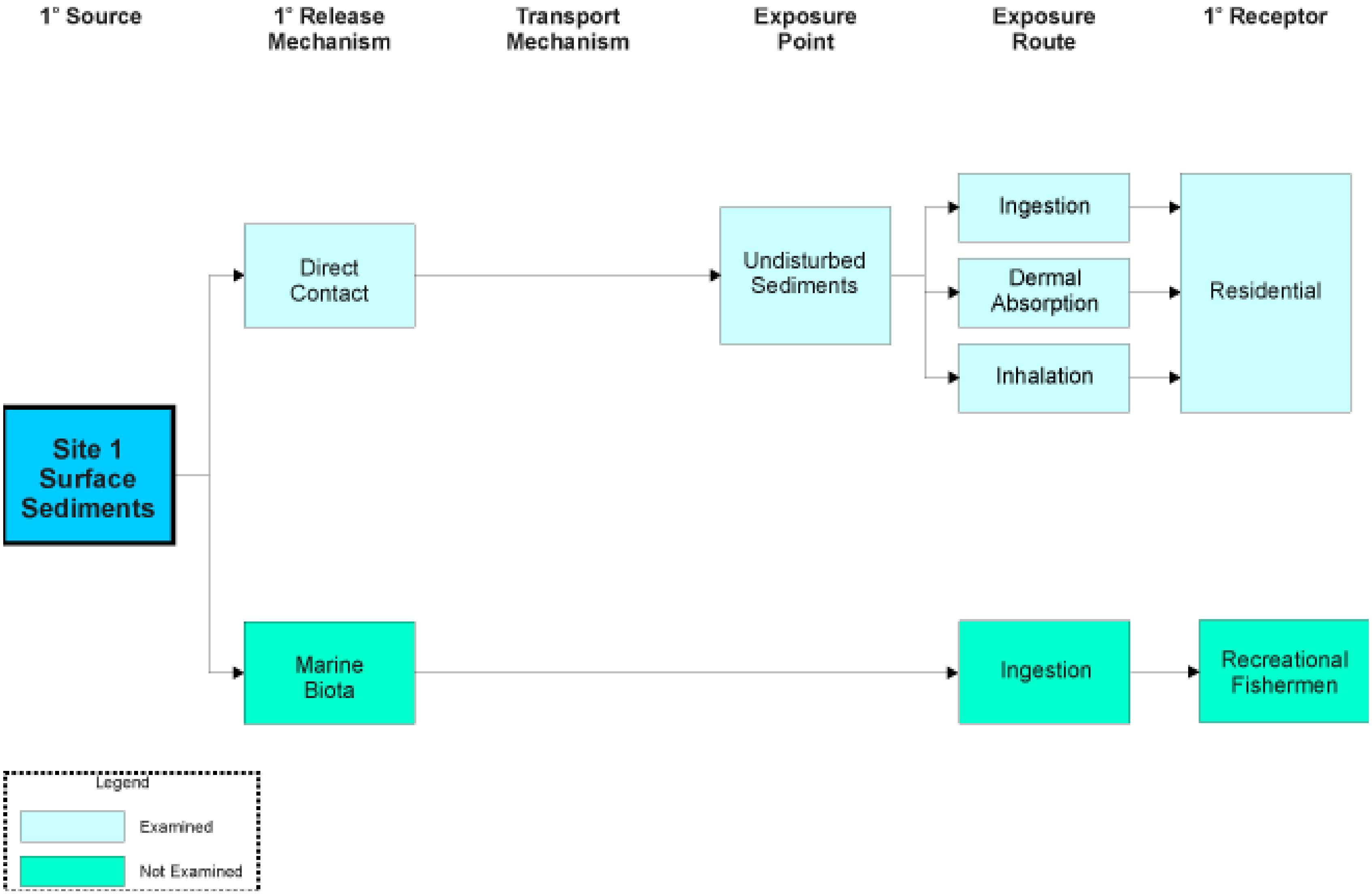


Figure 6-4. Conceptual site model for human health screening.

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 x 10^{-6} and 14.91 x 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 x 10^{-6}) to risk posed by reference coarse sediment (13.3 x 10^{-6}) and below that of fine reference sediment (20.1 x 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

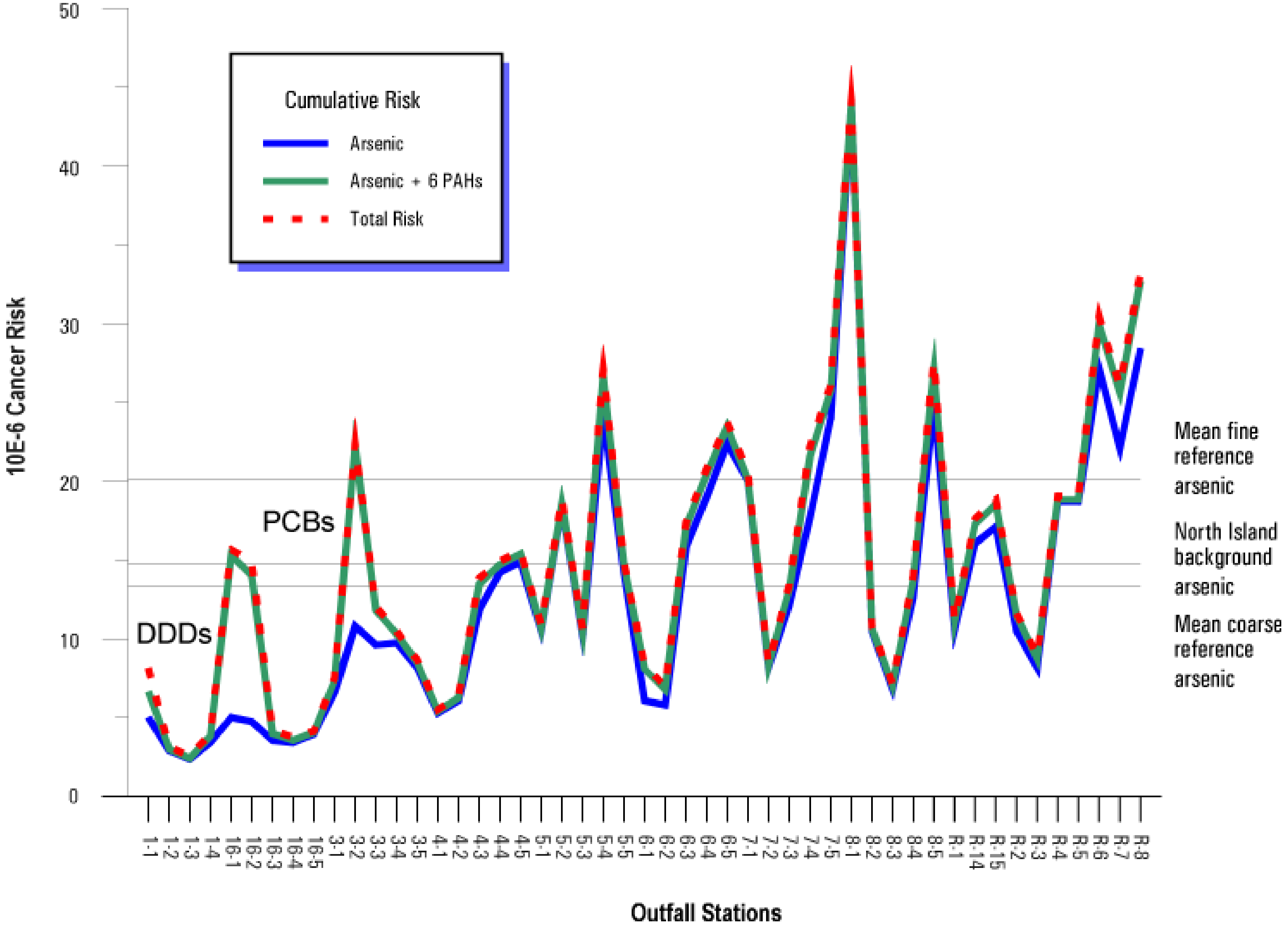


Figure 6-5. Residential cancer risk in surface sediments based on 1998 PRGs.

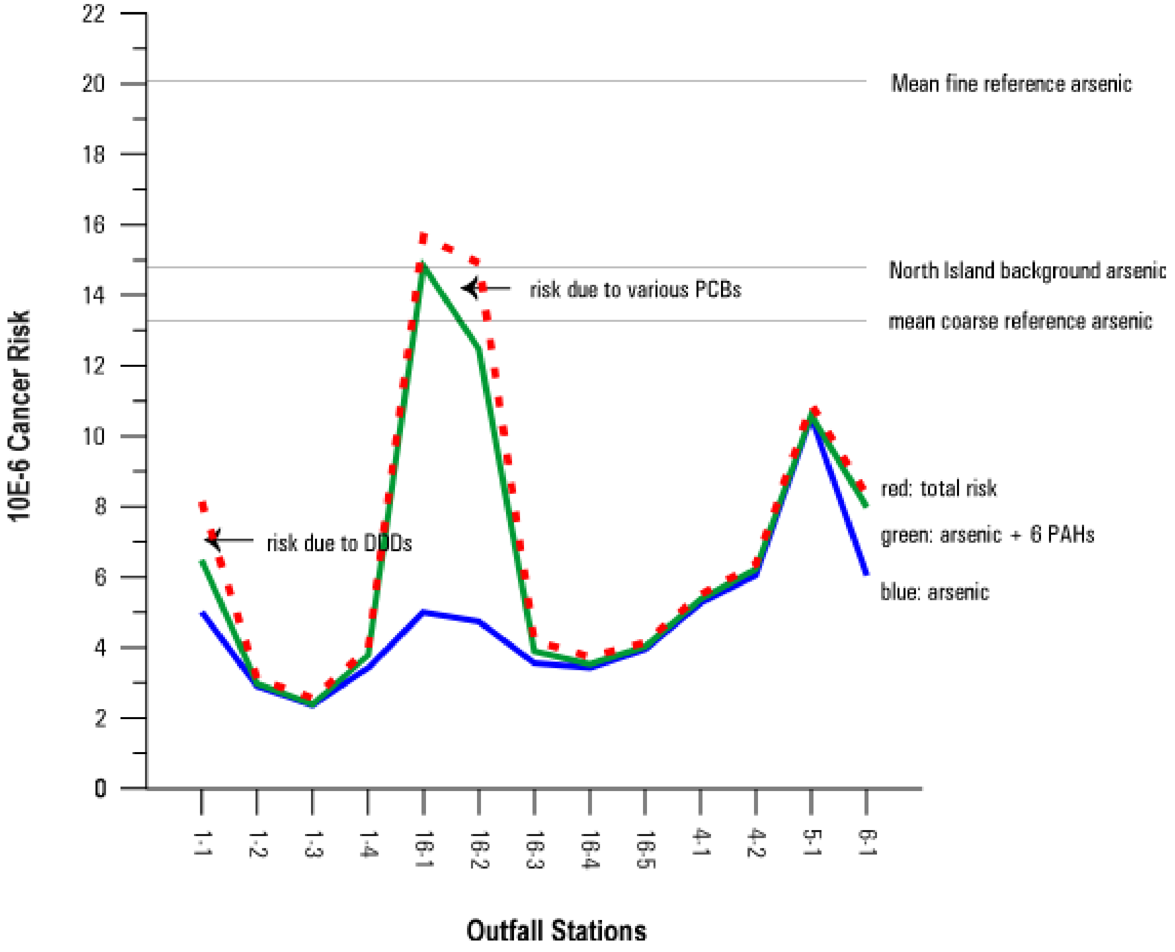


Figure 6-6. Residential cancer risk at intertidal outfall stations based on 1998 PRGs.

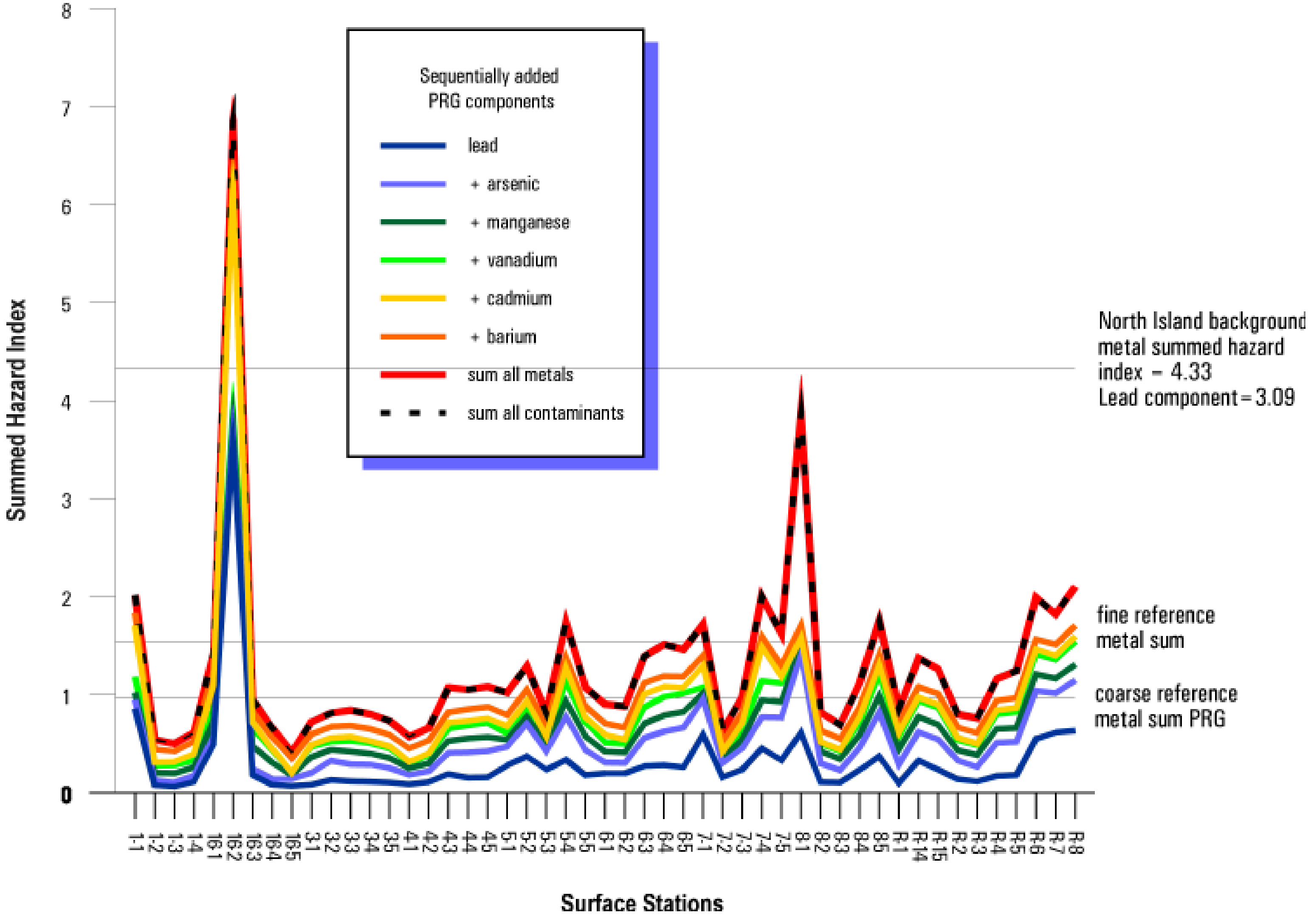


Figure 6-7. Residential non-cancerous hazard in surface sediments based on 1998 PRGs.

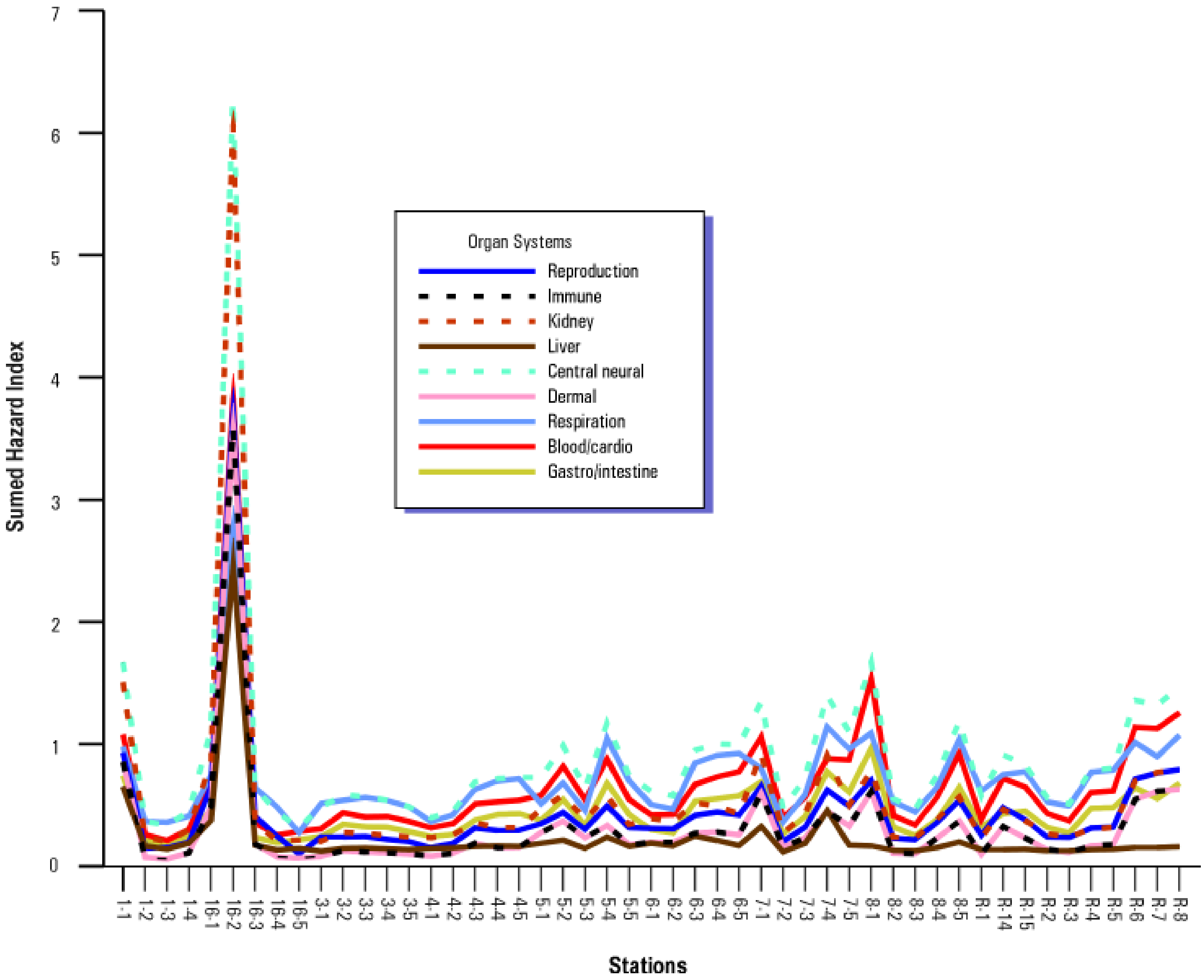


Figure 6-8. Summed hazard index by organ system and station.

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.

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