SEDIMENT SITE ASSESSMENT STUDY

SUBMARINE BASE SAN DIEGO

DRAFT FINAL REPORT

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EXECUTIVE SUMMARY

Introduction. This report details an investigation of the nature and extent of impaired San Diego Bay sediments adjacent to Submarine Base San Diego. The investigation was prompted by the designation of the site by the San Diego Regional Water Quality Control Board as having contaminated sediments and aquatic life impacts. The study was conducted by personnel from the Space and Naval Warfare Systems Center San Diego for Commander Navy Region Southwest.

The primary beneficial use concern is impairment to health of benthic organisms (Aquatic Life Beneficial Use), focusing on invertebrates such as crustaceans, polychaetes and molluscs that live in and on the sediment. There is also concern for potential exposure and impact to fish and birds that prey on these benthic organisms (Aquatic Dependent Wildlife Beneficial Use) as well as potential exposure to humans that may occur through fishing activities (Human Health Beneficial Use). The conceptual approach taken in this study was to use multiple measures of sediment quality including chemistry, toxicity, benthic community composition, and bioaccumulation to assess the potential for impairment to each of these three beneficial uses.

Background. Historical data were used to develop a list of contaminants of concern investigated in this study including: arsenic, chromium, copper, lead, mercury, nickel, and zinc, and organic compounds: PAHs, PCBs, Chlordane, and DDT. Tributyltin was also measured study because there were insufficient data and silver and cadmium were measured to validate their low level of concern. There were insufficient historical data regarding toxicity, benthic community and bioaccumulation.

Methods. Three measures of sediment toxicity were therefore made including survival of amphipod exposed to whole sediment, normal development of sea urchins exposed to the sediment-water interface, and normal development of mussel embryos exposed to sediment porewater. Benthic community composition was determined by counting the number and kinds of organisms in the sediment. Bioaccumulation of contaminants was measured by exposing clams to sediments *in situ* and measuring the uptake into their tissues. Ancillary but important measures of sediment grain size and total organic carbon were also made.

A reconnaissance sampling of reference stations was conducted in February 2004 and a comprehensive sampling was conducted in April 2004. Samples were collected from 6 bay reference stations and 14 stations within the SUBASE study site. Surface sediment (top 5 cm) grabs collected at each station were homogenized and split for use for chemical analyses, bioaccumulation exposures, and two of the three toxicity analyses. Separate core samples were collected for the sediment-water interface toxicity test. A separate grab sample was used in determining benthic community composition.

Data Evaluation. A weight of evidence approach was used to assess the potential impact to the Aquatic Life beneficial use. This approach used lines of evidence derived from measures of sediment chemistry, sediment toxicity, and benthic community composition. Screening level ecological and human health risk assessments were used to assess potential impacts to Aquatic Dependent Wildlife and Human Health beneficial uses, respectively. Contaminant bioaccumulation in clams was used as the primary measurement for the risk screening evaluations. A key requirement in the determination of impairment was that risk must be present at a level greater than that observed at sites in the bay not directly impacted by contaminant sources. This site-specific evaluation therefore compared conditions at each site

to a baseline condition that was defined as the existing ambient condition characterized by a pool of reference stations meeting the requirements of remoteness from source and having similar habitat.

The Baseline Pool used to represent the baseline condition consisted of up to 23 sample data collected from six separate reference stations. Twelve of the sample data were collected as part of this study, six during a reconnaissance survey and six during the comprehensive sampling survey. Additional sample data came from the same set of stations during recent sediment investigations conducted for the mouths of Chollas and Paleta Creek (2 samples) the NASSCO and Southwest Marine Shipyards (2 samples), the Switzer Creek and Downtown Piers (4 samples), and bay sampling conducted as part of the Bight'98 study (3 samples). This pool was designed to provide an unbiased set of reference stations that had comparable measures of sediment quality, similar benthic habitat, and lacked contamination or toxicity from site-specific activities. Data from each study site station were compared to the upper (i.e. for concentration) or lower (i.e. for survival) 95th-percentile prediction limit computed for each parameter from the Baseline Pool to determine if conditions differed from the baseline condition.

Aquatic Life Results. Impairment to the aquatic life beneficial use was determined using the weight of evidence from the chemistry, toxicity, and benthic community measurements. These data were used to assign a level of impairment into three categories of "Likely", "Possible", or "Unlikely". The weight of evidence showed that all stations were unlikely to be impaired from site chemicals. This was based on the findings that each individual line of evidence (chemistry, toxicity, and benthic community) showed no impact at any SUBASE station.

Aquatic-Dependent Life Results. The likelihood of aquatic dependent wildlife impairment at the SUBASE sites was categorized as either "Unlikely" or "Possible" based on a screening-level ecological risk assessment. For this assessment, *in situ* bioaccumulation of CoPCs in the clam *Macoma nasuta* was used to estimate exposure for representative wildlife receptors including surface feeding birds (Least Tern and Brown Pelican), diving birds (Surf Scoter and Western Grebe), and marine mammals (California Sea Lion). Potential for impairment to aquatic dependent wildlife at the SUBASE site was categorized as unlikely for all receptors with respect to all CoPCs with the exception of copper to avian receptors. However, the bioaccumulation of copper measured in clam tissues as a result of sediment copper levels was potentially overestimated by the *in situ* methods employed in the study. Even accounting for the potential overestimation in exposure conditions, there was a possible impairment to the Least Tern and Brown Pelican from copper found at two stations (SB4 and SB8). A comparable risk was observed for reference stations.

Human Health Results. The likelihood of human health impairment at the SUBASE sites was categorized as either "Unlikely" or "Possible" based on a screening level human health risk assessment. For this assessment, bioaccumulation of CoPCs in the clam *Macoma nasuta* was used to estimate exposure for humans from the consumption of fish or shellfish exposed to site sediments. All stations measured for bioaccumulation within the SUBASE sediment site investigation area were classified as possibly impaired for potential human health effects of arsenic related to the consumption of fish or shellfish associated with the site. A comparable risk was observed for reference stations. The dosage measured at all stations, reference as well as site stations were elevated above minimum toxic screening levels.

Conclusions and Recommendations. There has been considerable improvement in sediment conditions at the SUBASE site since the 1996 Bay Protection and Toxic Cleanup Plan (BPTCP) study identified it as a medium priority TMDL site. The level of all chemicals have decreased

since the 1996 study and there were no toxicity or benthic community impairments identified. Based on these results alone, it is recommended that the site be removed from the 303D list. While the number of stations analyzed (14) is below the minimum number of stations (20) technically required for delisting, the spatial data density was sufficient to fully characterize the region of interest.

The results of the screening level ecological and human health risk assessements identified copper and arsenic as possible risk drivers. The copper results were potentially biased by the methods utilized and further evaluation should be conducted by either conducting a baseline risk evaluation and/or by conducting additional measurements to validate the likelihood for risk. A similar evaluation or additional measurements should be made for arsenic.

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LIST OF ACRONYMS

BCA	Benthic Community Analysis
BRI	Benthic Response Index
Bight'98	Southern California Bight 1998 Regional Marine Monitoring Survey
BPJ	Best Professional Judgment
BPTCP	Bay Protection and Toxic Cleanup Program
BSAF	Biota-Sediment Accumulation Factors
BTAG	Biological Technical Assistance Group
CBSQG	Consensus-Based Sediment Quality Guideline
CNRSW	Commander Navy Region Southwest
CoPC	Contaminants of Potential Concern
CSM	Conceptual Site Model
DBT	Dibutyltin
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
DQO	Data Quality Objectives
EC50	Effects Concentration to 50% of organisms
EPA	Environmental Protection Agency
ERL	Effects Range Low
ERM	Effects Range Median
ERMQ	Effects Range Median Quotient
GC/ECD	Gas Chromatograph/Electron Capture Detector
GC/MS	Gas Chromatograph/Mass Spectrometer
HMWPAH	High Molecular Weight PAH
HPLC	High-Pressure Liquid Chromatography
HQ	Hazard Quotient
LC50	Lethal Concentration to 50% of Organisms
LMWPAH	Low Molecular Weight PAH
LOE	Line of Evidence
LPL	Lower Protective Limit
MBT	Monobutyltin
MSD	Minimum Significant Difference
NASSCO	National Steel and Shipbuilding Company
NAVSTA	Naval Station San Diego
NOEC	No Observable Effects Concentration NOEC
NPDES	National Pollutant Discharge Elimination System
PAHs	Polynuclear Aromatic Hydrocarbons
PCA	Principal Component Analysis
PCB	Polychlorinated biphenyls
PEL	Probable Effects Level

PELQ	Probable Effects Level Quotient
PPB	Parts per billion
PPM	Parts per million
PPPAH	Priority Pollutant PAH
PPT	Parts per thousand
RSD	Relative Standard Deviation
QA/QC	Quality Assurance/Quality Control
SAP	Sampling and Analysis Plan
SCCWRP	Southern California Coastal Water Research Project
SDRWQCB	San Diego Regional Water Quality Control Board
SCAMIT	Southern California Association of Marine Invertebrate Taxonomists
SIM	Selective Ion Monitoring
SQG	Sediment Quality Guideline
SSC-SD	SPAWAR Systems Center San Diego
SUBASE	Submarine Base San Diego
S-W	Shannon-Wiener
SWI	Sediment Water Interface
SWRCB	State Water Resources Control Board
TCHLOR	Total Chlordane
TDDT	Total DDT
TEL	Threshold Effects Level
THS	Toxic Hot Spot
TIE	Toxicity Identification Evaluations
TMDL	Total Maximum Daily Load
TBT	Tributyltin
TOC	Total Organic Carbon
TPAH	Total PAH
TPCB	Total PCB
TRV	Toxicity Reference Values
TSL	Tissue Screening Level
TTBT	Tetrabutyltin
UCL	Upper Confidence Limit
UPL	Upper Protective Limit
WOE	Weight of Evidence

1.0 INTRODUCTION

This report describes results of an investigation into the potential impairment of beneficial uses to San Diego Bay sediments adjacent to Submarine Base San Diego (SUBASE). The investigation was a Phase I Total Maximum Daily Load (TMDL) evaluation of the magnitude and spatial extent of sediment impairments to sensitive beneficial uses. The goal of the investigation was to develop a comprehensive weight of evidence (WOE) evaluation of impairment to aquatic life beneficial uses as well as a screening level evaluation of wildlife and human health beneficial uses. The investigation was conducted in response to a request from the San Diego Regional Water Quality Control Board (SDRWQCB) to evaluate the site because of its inclusion in the California State Water Resources Control Board's (SWRCB) Clean Water Act Section 303(d) list (SWRCB, 2003). The approximate 16-acre site (Figure 1-1) was listed as a medium priority TMDL site for benthic community effects and sediment toxicity (http://www.swrcb.ca.gov/tmdl/docs/2002/reg9303dlist.pdf). The designation was based on data originally compiled by Fairey et al., 1996 under the Bay Protection Toxic Cleanup Program (BPTCP), otherwise known as the Toxic Hot Spot (THS) program. The BPTCP characterized sediments from seven stations in this area as a low to moderate priority Toxic Hot Spot for degraded benthic community and elevated polynuclear aromatic hydrocarbon (PAH) contamination. The station data used for this designation were collected within the BPTCP stratum shown in Figure 1-1.

The investigation was developed using a conceptual approach, study design, and sampling and analysis plans comparable to other Phase I TMDL investigations recently carried out at other locations in San Diego Bay including those at the mouths of Chollas and Paleta Creek (SCCWRP and Navy, 2005), and at the Switzer Creek, Broadway Piers, and Downtown Anchorage (Anderson et al., 2004). The sampling and analysis plan for this study was provided to the SDRWQCB in December 2003 (SSC-SD, 2003). Reconnaissance sampling of reference stations was conducted in February 2004 and the full sampling for Phase I was conducted in April 2004. Personnel from the Space and Naval Warfare Systems Center San Diego (SSC-SD) along with support contractors executed the technical sampling, analyses, and final technical assessment and evaluation presented in this report.

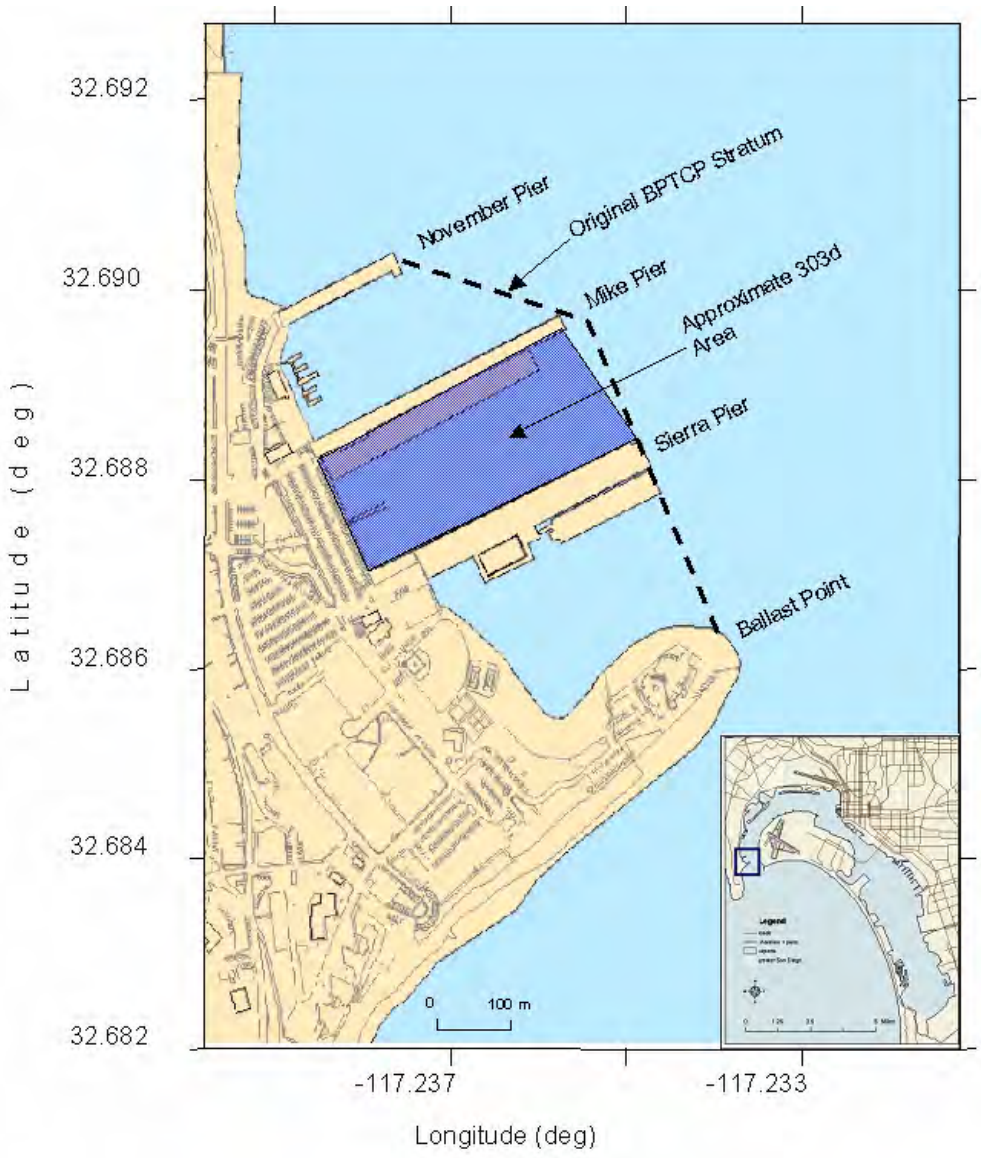

Figure 1-1. Location of SUBASE sediment site investigation area. The areas identified in the 303d list as well as the stratum designated under the Bay Protection Toxic Cleanup Program are shown.

2.0 HISTORICAL BACKGROUND

2.1 HISTORICAL DATA REVIEW

The first step in this current investigation was to compile and review recent historical sediment and contaminant source data for the SUBASE sediment area to evaluate spatial distribution and trends in historical data, determine Contaminants of Potential Concern (CoPCs), and identify data gaps to help design the sediment sampling effort. The historical review was provided to the SDRWQCB in December 2003 as part of the sampling and analysis plan for this study (SSC-SD, 2003). A summary of the report findings is highlighted below.

Data for 42 stations were found within seven studies carried out over the last 15 years (Table 2-1). Of the 42 stations with chemistry data, 13 also had measures of toxicity and 7 of those had benthic community data. Chemicals were identified as CoPCs if they exceeded the effects range low (ERL) toxicity-based thresholds of Long and Morgan, 1990. Based on this benchmark, CoPCs related to impacts to aquatic life beneficial use included arsenic, copper, mercury, nickel, zinc, PAHs, PCBs, DDT and Chlordane. Silver, chromium, lead and tributyltin (TBT) were below ERL thresholds, though the TBT data were limited. With the exception of PAHs, PCBs, DDT and Chlordane, the majority of stations had concentrations below the ERL. All chemicals at all stations were below the effects range medium (ERM) benchmark with the exception of mercury at one station, and PCBs at two stations. The ERMQ for the SUBASE historical data ranged from 0.04 to 0.38 and therefore fell within the low (<0.1) to medium-low (<0.5) categories defined by Long and MacDonald (1998).

Risk-screening for both the wildlife and human health beneficial use CoPCs was performed using the historical sediment chemistry data to estimate prey, fish, and shellfish tissue chemical concentrations. Tissue concentrations estimated using accumulation factors developed in the Chollas-Paleta TMDL study (SCCWRP and Navy, 2005) were screened against toxicity reference values for locally relevant wildlife receptors including the Least Tern and the California Sea Lion (Navy and SDUPD, 2000) and against human cancer thresholds (USEPA, 2003). Chemicals identified as CoPCs based on wildlife beneficial use were arsenic and chromium for the California Sea Lion, and lead for the Least Tern. Chemicals identified as CoPCs based on human health beneficial use were arsenic and PAHs.

Spatial distributions of the historical data were examined to help determine the extent of the Phase I study area and the locations for individual stations. There was a fairly good spatial distribution of historical sampling locations throughout the study site, although most datasets were incomplete, usually with limited toxicity, benthic community analysis, or bioaccumulation data. The chemical data from the studies generally showed increasing gradients toward the shoreline. Areas within the piers generally fell between the ERL and ERM. The historical data for biological effects including toxicity and benthic community analysis (BCA) was limited, and there were insufficient data to examine spatial gradients. Amphipod survival was generally high, with most stations having survival rates >80%. Infaunal abundance varied by about a factor of 4 to 5 within the area. The general lack of spatial data for biological effects represented a significant data gap.

The final CoPCs and biological measurement parameters identified in the historical review are shown in Table 2-2. CoPCs included arsenic, chromium, copper, mercury, nickel, lead, zinc, PAHs, PCBs, Chlordane, and DDT. Two additional chemicals, silver and cadmium, were identified as not likely to be of concern but were left on the list of analytes for validation purposes. TBT data were insufficient to draw conclusions on its likelihood as a CoPC and was

identified for analysis as a chemical having a data gap. Additional data gaps existed in the biological data including toxicity and BCA. Though not specifically identified as a gap in the historical review, the analysis of chemical uptake by organisms (bioaccumulation) was included as a measurement parameter for the Phase I investigation.

Table 2-1. Historical sediment quality studies within or near the SUBASE Toxic Hotspot stratum. The studies identified were described in the sampling and analysis plan (SSC-SD, 2003).

Study	Year	N	Parameters Measured
SANDAG	1990	18	Chem
BPTCP	1993	7	Chem, Tox, BCA
North Island Site 1	1996	2	Chem, Tox, Bioaccum
Navy Screening Study (Leather et al.)	1997	10	Chem
Bight'98	1998	2	Chem, Tox, BCA
Chollas/Paleta TMDL	2001	1	Chem, Tox, BCA, Bioaccum
NASSCO/SW Shipyard Phase I	2001	1	Chem, Tox, BCA, Bioaccum
NASSCO/SW Shipyard Phase II	2002	1	Chem

BCA: Benthic Community Analysis

Table 2-2. Final parameter list for the Phase I study based on review of the historical data.

Analysis Parameters for Phase I		
CoPC	Data Gap	Validation
As	TBT	Ag
Cr	Toxicity	Cd
Cu	BCA	
Hg	Bioaccumulation	
Ni		
Pb		
Zn		
PAH		
PCB		
Chlordane		
DDT		

2.2 SAMPLING PLAN DEVELOPMENT

As described above, the historical data were insufficient to fully characterize the spatial extent of contamination, toxicity, benthic community degradation, or degree to which bioaccumulation is occurring at the SUBASE site. Further, the data sets were unable to resolve relationships between contaminant levels and deleterious effects. Thus, a Phase I sampling plan was generated to gather the appropriate data to fully characterize and assess the magnitude and spatial extent of sediment quality in the pier area of SUBASE. The plan was finalized in December 2003 (SSC-SD, 2003). The sampling plan was designed to address impairment to the aquatic life beneficial use as well as to provide an initial screening of wildlife and human health impacts. This study represented the first phase of a multiphased approach to completing requirements under TMDL and cleanup plans for the study area (Figure 2-1).

The sampling plan followed the general approach of BPTCP, the Southern California Bight Regional Marine Monitoring Surveys (Bight'98), and other Phase I TMDL investigations conducted in San Diego Bay. The approach used multiple indicators of sediment quality to develop a weight of evidence in identifying areas of impaired sediment quality. Included in this effort were determinations of the magnitude and spatial distribution of:

- Sediment physical/chemical characteristics (e.g., grain size)
- Sediment chemical contamination
- Sediment and interstitial water toxicity
- Bioaccumulation of contaminants by a marine invertebrate
- Benthic community analysis

These data were used to identify areas of concern that could be used in the development of total maximum daily loads (TMDLs) in Phase II or possibly within Phase III cleanup actions. Elements of Phase II and Phase III studies are still evolving under the guidance of the SDRWQCB.

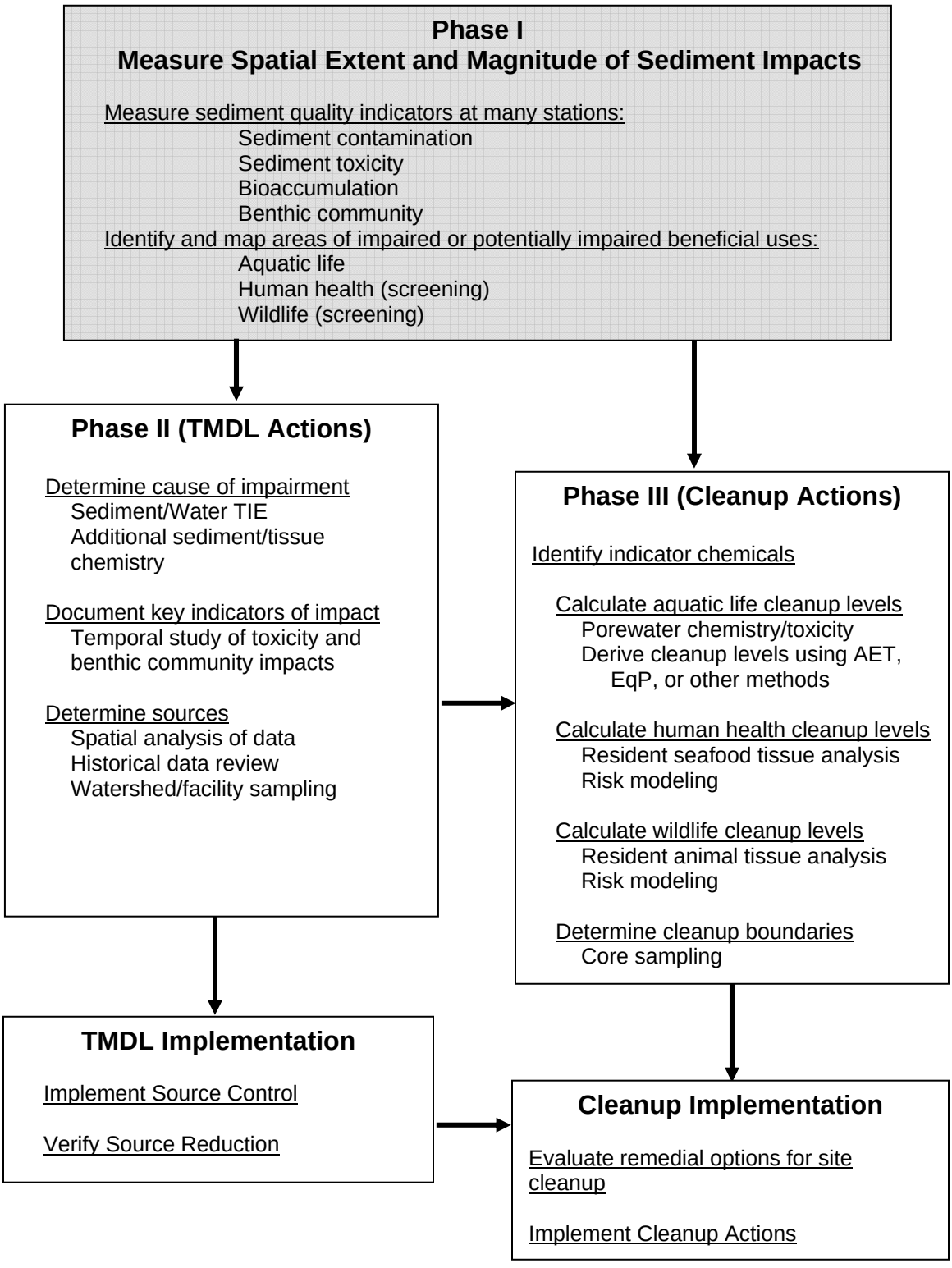

Figure 2-1. Phased sampling and analysis approach showing the relationship of Phase I sampling plan to potential subsequent TMDL and cleanup activities at the study sites.

3.0 CONCEPTUAL APPROACH

The conceptual approach taken in this study was to use multiple measures of sediment quality to provide a weight of evidence to support or refute the presence of impairment to beneficial uses at Toxic Hot Spot sites. The conceptual approach for this investigation was based on recent Environmental Protection Agency (EPA) guidance (USEPA, 2000) and was consistent with that of the BPTCP, recent San Diego Bay sediment investigations as well as other comprehensive sediment quality evaluations occurring throughout the nation. The approach was based on four key assumptions. First, that the determination of biological impairment is best assessed through the measurement of biological effects associated with the study site (e.g., toxicity, bioaccumulation, and benthic community degradation). Second, that there must be multiple indicators of sediment quality (WOE approach) measured to provide a confident assessment of impacts because no single test or parameter is a consistently reliable, accurate, or predictive indicator of impairment. Third, that site-specific information is needed to accurately assess impacts because there may be unknown site-specific factors in the study areas that may significantly affect causal relationships between contamination and effects. And finally, that the evaluation of impairment be made relative to sediment quality measured at a set of designated reference locations that represent an acceptable level of sediment quality.

3.1 CONCEPTUAL SITE MODEL

Based on results from the historical review, a generic conceptual site model (CSM) was developed to describe and visualize the known, expected, and/or predicted relationships between site CoPCs and ecological receptors. The model provides a framework for understanding the dominant processes that control sediment quality at the site including linkages amongst ongoing and historic contaminant sources, exposure pathways, and biological receptors (Figure 3-1). The framework is thus applicable for evaluating site data for both TMDL and site cleanup purposes. The site has been identified as having impaired sediments, storm water inputs along the shoreline, and shoreline industrial activities. The site is a relatively deep-water environment, which has important implications for the potential exposure pathways that may exist. As described in the sampling and analysis plan document, there was no evidence of eelgrass in the area that would require additional consideration in the CSM.

The primary contaminant sources and pathways are the discharge of contaminants from the near shore into the surface water and their eventual settling out on particles into the sediments (Figure 3-2). These include storm water that enters the site via small storm drains and sheet runoff and in-water sources primarily from ships via release from antifouling coatings and zinc cathodic protection systems. Though atmospheric deposition is certainly a source, its magnitude is currently unknown and would be evaluated in the Phase II assessment if needed. A significant fraction of the storm water source material is likely to enter the site in association with particulate matter, though dissolved materials can be adsorbed onto particulate matter once in the receiving environment. For this reason, along with the relatively weaker currents inside the pier area, it is anticipated that the majority of source material entering the site settles to the sediment bed within the site rather than being transported to the remainder of the bay. However, sediment resuspension during ship movements may provide an additional transport mechanism of material from the site to the main channel of the bay where it would be transported with the tide. Groundwater is not considered a significant source of contaminants to this site but its potential would be evaluated in the Phase II investigation if impairments are found.

The most sensitive primary beneficial use concern at this site is the impairment to health of benthic organisms, primarily invertebrates such as crustaceans, polychaetes and molluscs that live in and on the sediment. Benthic organisms are exposed to these contaminants by direct contact with, or ingestion of near-surface sediment. Contaminant concentrations in bay waters are almost always below water quality criteria and thus do not pose a threat. A second level of ecological exposure may occur for bottom feeding fish that prey on benthic invertebrates. Existing survey data suggest that exposure at the SUBASE site would be primarily to species such as the California Halibut, Round Stingray, and Barred Sand Bass (U.S. Navy/SDUPD, 2000).

Because bottom depths throughout most of the site are great enough (there is a limited area of shallow water and a small tidal beach on the south side of the base), it is unlikely that transfer to other ecological niches would occur. Diving birds and surface feeding birds generally limit their activities to shallow water areas, and there are few upper level receptors such as sea lions that feed directly on the bottom fish species mentioned above. Though there is limited potential for exposure, impact to wildlife beneficial uses was addressed in a screening level evaluation.

Exposure to humans can occur through fishing activities that involve direct take of bottom fish. However, fishing activity is not permitted within the direct confines of the site, and the exposure pathway to humans is not likely. The mobility of the fish through the site could provide a pathway to fishing activities that occur outside the site and are therefore kept as a possible route of exposure. Though there is limited potential for exposure, impact to human health beneficial uses was addressed in a screening level evaluation.

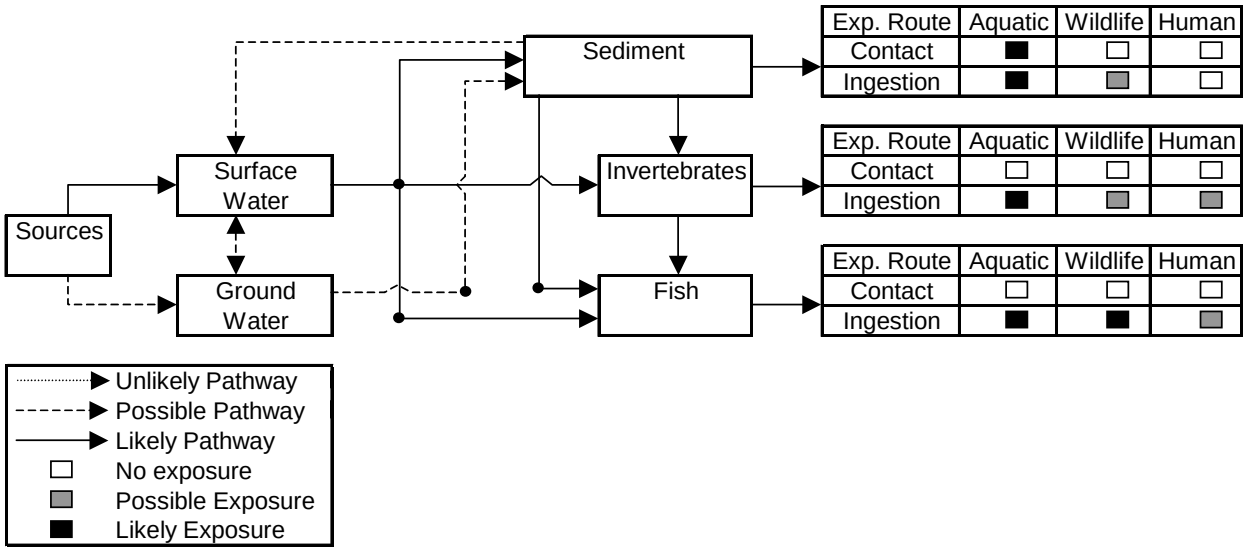

Figure 3-1. Conceptual site model for the SUBASE study site showing sources, transport pathways, exposure routes, and receptors of concern.

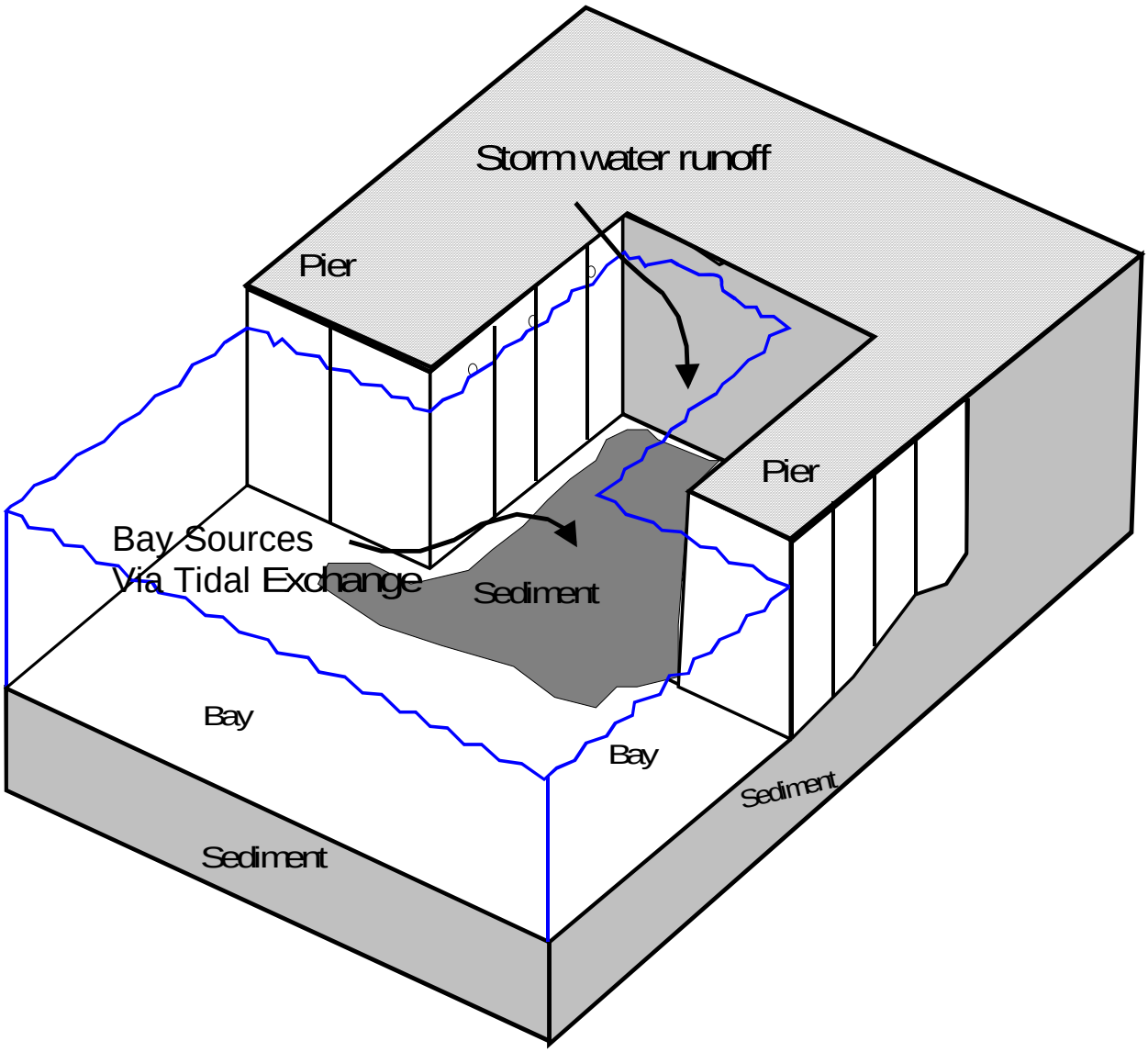

Figure 3-2. Graphical representation of potential contaminant sources and pathways to the sediment at the SUBASE study site.

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4.0 TECHNICAL APPROACH

4.1 FIELD MEASUREMENT PROGRAM

The technical approach taken for the Phase I study was to synoptically collect and analyze surface sediment from study stations and at designated bay reference stations for a suite of sediment quality parameters. The study design entailed the collection of near-surface sediment at 14 study site stations and 6 reference stations from the outer portion of the bay. Stations within the study were arranged in a grid pattern that contained multiple stations between each of the piers. This included either three or four stations along a transect from the quay wall to beyond the end of the pier heads (Figure 4-1). Multiple lines of evidence (LOE) for sediment quality were measured at each station. The three key LOE of sediment quality (sediment triad) used to assess aquatic life impairment included measures of sediment chemical contamination, sediment toxicity, and benthic community composition. Contaminant uptake as a result of *in situ* exposure to transplanted clams was used to evaluate wildlife and human health impairment. Bioaccumulation was measured at a subset of site stations and at all reference stations. Sediment characteristics including grain size and total organic carbon (TOC) were also measured to help interpret contaminant bioavailability and confounding effects that might be related to physical characteristics rather than contamination. The key LOE used to characterize sediment quality in the study are described below.

Sediment Chemical Contamination. Concentrations of a suite of metals, PAHs, PCBs, organotins, and chlorinated pesticides were measured in the bulk surface (0 to 5 cm) sediment. These sediment chemical contamination measurements were used to document the extent, spatial pattern, and magnitude of sediment contamination at each study site.

Sediment Toxicity. Acute and sublethal toxic effects of bulk sediment, porewater, and contaminants fluxing across the sediment-water interface were measured using a variety of tests. Acute toxicity was assessed by measuring survival of the amphipod crustacean, *Eohaustorius estuarius,* after 10 days of exposure to bulk sediment. Sublethal sediment toxicity was assessed by measuring the effects of a 96-h exposure of porewater on larval development of the sea urchin (*Strongylocentrotus purpuratus*). The presence of sublethal effects and potential impacts of contaminated sediments on the water column was assessed by measuring the effects of a 2-day exposure to water from the sediment-water interface on mussel (*Mytilus galloprovincialis*) embryo-larval development. The sediment toxicity tests were used to document the spatial pattern and magnitude of toxic effects in the sediments at each study site.

Benthic Community Analysis (BCA). The numbers and kinds of benthic invertebrates present in sediment samples were used to document the health of the benthic communities at the study sites.

Bioaccumulation. Concentrations of a suite of metals, PAHs, PCBs, organotins, and chlorinated pesticides were measured in clam tissue (*Macoma nasuta*) before and after a 28-day *in situ* exposure to site and reference station sediment. The bioaccumulation tests were used to evaluate the potential for contaminant uptake and subsequent food chain transfer of organic chemicals and metals from the sediment.

4.2 EVALUATING IMPAIRMENT TO BENEFICIAL USES

Individual LOE were integrated to evaluate the potential for site-specific impairment to aquatic life, aquatic-dependent wildlife, and human health beneficial uses related to CoPCs at each site. For each LOE, consideration was given to measures of both absolute risk (e.g. comparison to known toxicity thresholds), and to site-specific relative risk (i.e. comparison to conditions at stations not directly influenced by sources). A sediment triad or WOE approach was used to assess impairment to the aquatic life beneficial use (Chapman et al., 1987). Ecological and human health screening risk assessments were used to address wildlife and human health impairments, respectively. These evaluations addressed each of the pathways and receptors described in the CSM. The steps for each of these assessments are described below.

4.2.1 The Baseline Condition

A key requirement in the determination of impairment at the study sites was that risk must be present at a level greater than that present at stations in the bay that represent the existing ambient condition. This ambient sediment quality condition was originally defined as the Baseline Condition in the Chollas-Paleta TMDL study (SCCWRP and Navy, 2005). The condition was based on data pooled from a set of reference stations (Baseline Pool) that are known to be remote from the direct influence of contaminant sources and where previous studies had shown low contaminant levels, minimal toxicity, and similar habitat to the study sites. This condition acknowledges the potential presence of background contamination as well as natural variability in toxicity and benthic condition. Reference stations were excluded from this pool if there was an indication of contamination or toxicity that appeared to be related to a nearby source. However, stations were not excluded from this pool based on specific biological response thresholds to maintain natural variability as property of the baseline condition. Development of the Baseline Pool is described below, and its application is discussed further throughout the remainder of the report. The Baseline Pool was the primary benchmark used in this study to assess site-specific levels of relative risk.

The Baseline condition was developed by pooling current and historical data collected from six reference stations (2229, 2433, 2436, 2441, 90056, and C001SS31) chosen as part of the sampling and analysis plan (SSC-SD, 2003). The six reference stations were chosen based on a review of historical data and only included stations from the north portion of the bay as previous studies have shown differences in benthic habitat and wildlife species between north and south bay locations (Figure 4-2). The selection criteria included: low contaminant concentrations representative of baseline conditions, comparable habitat to the study sites, comparable physical properties, adequate sample size for statistical analysis, and data comparability. Measurement data were pooled from the current study as well as from historical data collected during the Chollas-Paleta (SCCWRP and Navy, 2005), Shipyards (Exponent, 2005), Bight'98 (SCCWRP, 1998), and Switzer Creek (Anderson et al., 2004) studies (Table 4-1).

The Baseline Pool included a maximum of 23 independent measurements made at the six reference stations. Six of the Baseline Pool measurements were collected simultaneously with the SUBASE site data in April 2004 and therefore provided a fully matched set of parameters for comparison to site data. All six stations were also sampled as part of a reconnaissance survey conducted in February 2004 prior to the full site investigation. This dataset did not include benthic community data or sublethal toxicity tests. Two reference stations evaluated as part of the Chollas-Paleta study matched the recent data collection except it had no data for TBT, dieldrin, or the two sublethal toxicity tests. Two reference stations from the Shipyards study dataset matched except it had no data for Chlordane, DDT, dieldrin, and the two sublethal

toxicity tests. Three reference stations from the Bight'98 study dataset were comparable except for TBT, dieldrin, and the two sublethal toxicity tests. Data from three of the four Switzer Creek reference stations matched except for TBT and the sublethal toxicity tests. The fourth set of measurements from the study, collected during the reconnaissance survey, also did not include benthic community data.

Table 4-1. Reference stations 2229, 2433, 2436, 2441, 90056, and C001SS31 used in developing the baseline condition. Data were pooled from 23 independent measurements made at these station locations during the current study (SB prefix), the Chollas-Paleta (CP) study, the Shipyards study (SY), the Bight'98 Study (B98), and the Switzer Creek (SW) study. Station identifiers used the study name (e.g. CP) as a prefix to uniquely identify the data. The –R suffix refers to station data collected during a reconnaissance survey as part of this study.

Current Study	Chollas-Paleta Study
SB2229	CP2433
SB2433	CP2441
SB2436	**Shipyard Study**
SB2441	SY2433
SB90056	SY2441
SBC001SS31	**Bight'98 Study**
SB2229-R	B982229
SB2433-R	B982436
SB2436-R	B982441
SB2441-R	**Switzer Creek Study**
SB90056-R	SC2433-R
SBC001SS31-R	SC2229
	SC2433
	SC2441

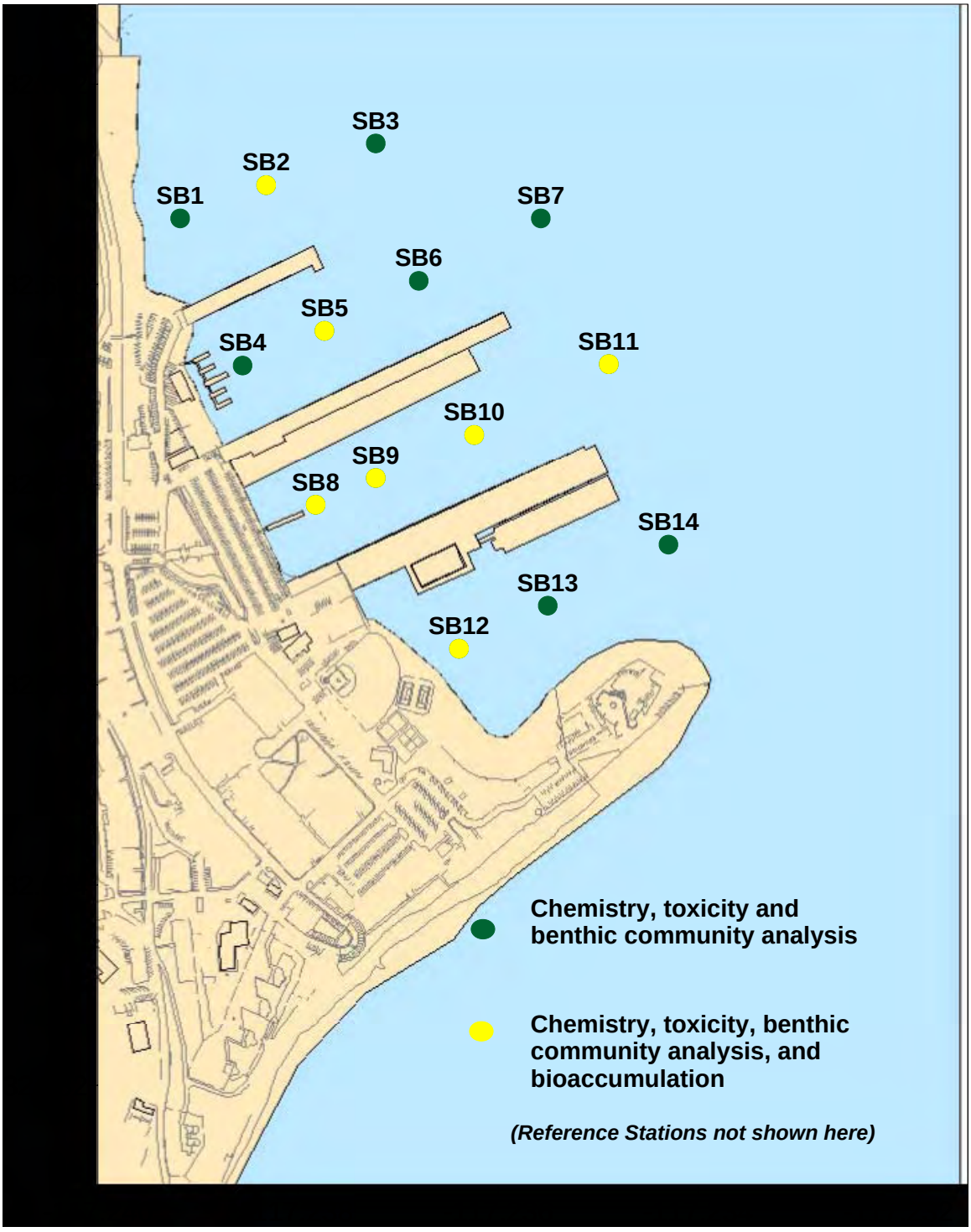

Figure 4-1. SUBASE sampling stations. All stations were analyzed for chemistry, bioassays, and benthic community assessment. Seven of the 14 stations (●) and all reference stations (not shown) were also analyzed for bioaccumulation.

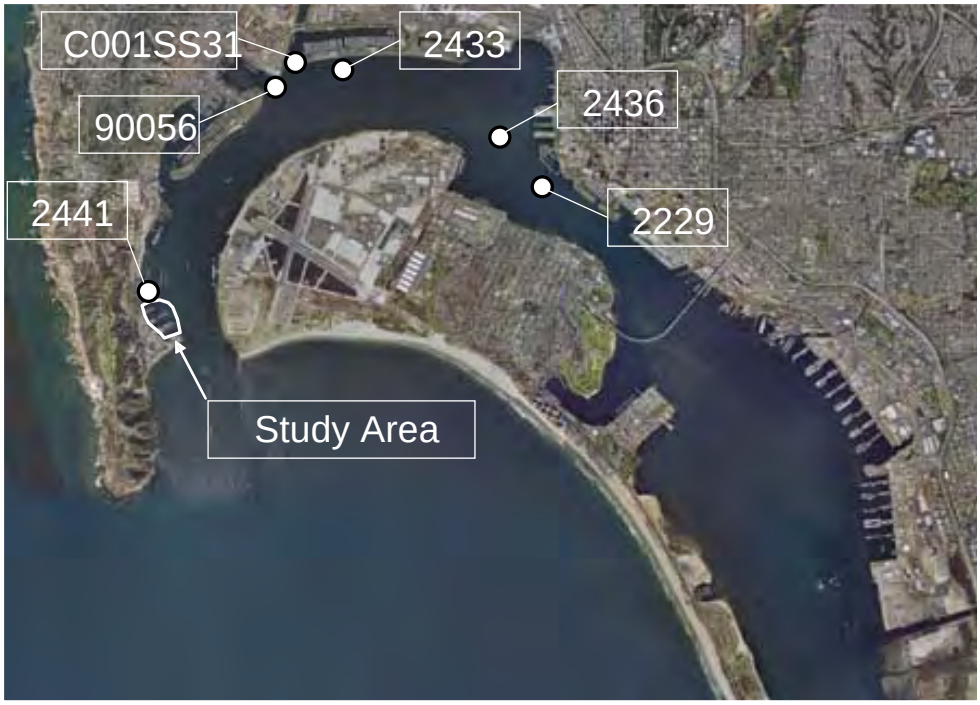

Figure 4-2. Location of reference stations included in the Baseline Pool. All stations were were analyzed for chemistry, bioassays, benthic community assessment, and bioaccumulation.

4.2.2 Aquatic Life Impact

The sediment triad approach to assessing aquatic life impact relied on the three principal LOE that included measures of sediment chemistry, sediment or interstitial water toxicity, and benthic community composition. The three LOE were individually evaluated to determine the presence of significant impacts at each station by using a three-step process. First, the data quality of each LOE was assessed relative to predetermined objectives such as accuracy and precision for sediment and tissue chemical analyses, control performance and confounding factors in the toxicity tests, and sorting efficiency and identification accuracy for the benthic analyses. Second, the data were compared to published thresholds, guidelines, or controls that indicate whether a significant response was obtained. Finally, the data were compared to the study baseline condition to assess the site-specific impact. This approach is based on the framework for evaluating sediment quality developed by the EPA for application in the St. Louis River Area of Concern (USEPA, 2000). The degree of impact indicated by each LOE was then integrated into a weight of evidence evaluation to provide an overall assessment of potential for aquatic life impairment (USEPA, 1997).

4.2.2.1 Sediment Chemistry

Bulk sediment chemical concentrations measured at each station were evaluated relative to sediment quality guidelines (SQGs) as well as to the baseline condition. SQGs have been established as one of the most effective methods for attempting to relate sediment chemistry to their observed toxic effects (Long et al., 1995; Long et al., 1998). The evaluation in this study compared CoPCs relative to their individual ERM for metals (effects range-median, Long et al.,

1995), consensus midrange effects concentration for PAHs and PCBs (MacDonald et al., 2000: Swartz 1999), PEL (probable effects level, MacDonald et al., 1996) for Chlordane and dieldrin, and organic carbon normalized DDT and TBT effects value (Swartz et al., 1998) and their respective 95 percentile predictive limit calculated from the Baseline Pool data. The magnitude of impact was addressed by counting the number of CoPCs that exceeded each of their individual benchmarks, by evaluating them as a group against a mean SQGQ1 quotient benchmark (Fairey et al., 2001), and by counting the number of parameters that exceeded the Baseline Pool predictive limit.

The relative magnitude of potential site-specific impact from bulk sediment CoPCs was classified into three ordinal ranking categories of low, moderate, or high likelihood of impact. The ranking was based on a semi-quantitative measure that give increasing weight to a greater number and magnitude of chemicals exceeding a threshold, similar to the method used by Long et al. (1998). The breakpoints in the ranking levels were established using best professional judgment (BPJ), again, following Long et al. (1998). The ranking criteria were based on two key assumptions. First, that there is a low likelihood of impact from CoPCs if all chemicals at a station are less than relatively low SQGs and less than the established baseline condition. Second, that there was a high likelihood of impact from CoPCs when many of the chemicals at a station exceed a relatively high SQG, and exceed the baseline condition. The category ranking criteria for bulk sediment chemistry are summarized below.

Low- The mean SQGQ1 was less than 0.25 or all chemicals were less than the 95% predictive limit calculated from the Baseline Pool. Additionally, there must not be any single chemical that exceeded either its SQG or Baseline Pool predictive limit value whichever was higher. To meet this category, all chemicals present at the site, either individually or summed must have been lower than a relatively low SQG and have been below the baseline condition.

Moderate- The mean SQGQ1 was between 0.25 and 1.0 and greater than the 95% predictive limit calculated from the Baseline Pool. Additionally, a station was classified into this category if there were five or less individual chemicals that exceeded their respective SQG or Baseline Pool predictive limit, whichever was higher. To meet this category, some (five or less) chemicals either individually or when summed exceeded a moderate level SQG and/or the baseline condition.

High- The mean SQGQ1 for all chemicals was greater than or equal to 1.0 and was greater than the 95% predictive limit calculated from the Baseline Pool. This category was also assigned if more than five chemicals exceed their individual SQG or the baseline condition, whichever was higher. To meet this category, the baseline condition as well as a relatively high SQG must have been exceeded when chemicals are considered as a group, or there were at least six individual chemicals exceeding a SQG or the baseline condition.

4.2.2.2 Sediment Toxicity

The three toxicity test results were compared to their negative controls (collection site sediment or laboratory seawater) as well as to the 95% lower prediction limit calculated from the Baseline Pool to determine the relative magnitude of station toxicity for this LOE. The magnitude and consistency of responses was used to classify station sediments as having a low, moderate, or high degree of toxic effects. The rankings were based on the combined toxic response from all three tests.

Similar to the chemistry LOE, the ranking method employed a semi-quantitative assessment of the data that reflected both the presence and magnitude of toxicity. It was assumed that there

was no, or a low degree of, toxic effects if the results of all three toxicity tests were not significantly different from their controls or they had a statistically lower level of toxicity than observed under the baseline condition. Each of the three toxicity tests was given equal weight for classifying a sample as moderately toxic; the presence of significant toxicity in any one test was sufficient to classify a sample as moderately toxic. A high degree of sediment toxicity was indicated when survival of amphipods was less than a minimum significant difference (MSD) of 75% and significantly different from the control and baseline. A high toxicity ranking was also assigned when both of the sublethal tests measured a greater level of toxicity than the baseline condition.

The amphipod test result was given greater weight for the high toxicity category because the acute survival endpoint of this test was assumed to have a higher degree of association with ecological impacts than the sublethal tests. The sea urchin and mussel embryo-larval development test results were given less weight because these are sublethal critical life stage tests that are more susceptible to confounding factors and their association with ecological impacts is less certain. The category ranking criteria for sediment toxicity are summarized below.

Low- There were no or a low degree of toxic effects if results of all three bioassays were not significantly different from their controls or they had a statistically lower level of toxicity than observed under the baseline condition.

Moderate- The sediments were considered moderately toxic if any one of the bioassay results was statistically different from its control and was less than the baseline condition. There was an additional requirement that amphipod survival be greater than its MSD of 75%, regardless of the result relative to controls or baseline.

High- There was multiple criteria that can result in a categorization of the sediments as having a high degree of toxicity: 1) If survival of amphipods at a station was less than its MSD of 75% and was statistically different than controls and statistically less than baseline. 2) If the amphipod test together with any one of the other bioassays has a result that was statistically different from control and was statistically less than baseline. 3) If both the porewater and sediment-water interface test results were less than their MSD values, 55% and 80%, respectively, and were statistically less than the controls and baseline.

4.2.2.3 Benthic Community Composition

Four metrics were used to assess community health at each station: total abundance, total number of species, the Shannon-Wiener (SW) Diversity Index, and the Benthic Response Index (BRI) developed by SCCWRP (Ranasinghe et al., 2003). The Benthic Community LOE compared station data against the Bight'98 BRI response level benchmarks as well as to the 95% lower (upper for BRI) prediction limit of each of the metrics calculated for the Baseline Pool. Consideration was given first to the overall BRI ranking and then to the individual metrics. The BRI was given this higher weighting because it is a more comprehensive measure of community health.

Similar to the other LOE, this evaluation was based on a semi-quantitative measure that integrated the responses and the application of ranking criteria based on BPJ. It was assumed that no, or a low degree of benthic community degradation is present when the station BRI is level I (< response II) or is statistically similar to the baseline condition and abundance, number of taxa and the SW Diversity Index are all statistically similar to the baseline condition. Conversely, a high degree of impact to community health at a station is assumed to be present

when there is a BRI response of level IV (> response III) or the other indicators also show impacts. The category ranking criteria for benthic community impacts are summarized below.

Low- Benthic community health at a station had no or a low degree of degradation if the BRI is less than response level II and when abundance, number of taxa, and the SW Diversity Index were all statistically similar to the baseline condition.

Moderate- There was a moderate degree of impact to community health at a station if the BRI was either response level II or III and was statistically greater than the baseline condition or if any one of the other benthic community metrics was statistically lower than the baseline condition.

High- There was a high degree of impact to benthic community health at a station if the BRI was greater than response level III or the BRI response was greater than level II, statistically greater than the baseline condition, and at least one of the other benthic community metrics was also statistically less than baseline.

4.2.2.4 Triad Analysis of Impairment to Aquatic Life Beneficial Use

The three LOE described above were integrated into an overall WOE assessment focused on identifying the likelihood that site-specific aquatic life beneficial use is impaired at a given station due to the presence of a known CoPC related to the site. The approach follows the general principles of WOE analysis described by Chapman (1990, 1996) and others. Potential combinations of the ordinal rankings for individual LOE were assessed and assigned a relative overall likelihood of impairment using three categories “Unlikely”, “Possible”, and “Likely” based on consideration of four key elements as described by Menzie et al., (1996):

- the level of confidence or weight given to the individual LOE
- whether the LOE indicates there is an effect
- the magnitude or consistency of the effect
- the concurrence among the various LOE

For example, a station with a high ordinal ranking for chemistry, toxicity and benthic community would indicate a high likelihood of site-specific aquatic life impairment because each LOE indicates an effect, the magnitude of the effect is consistently high, and there is clear concurrence among the LOE. Alternatively, a station with a low ordinal ranking for chemistry, and moderate or high rankings for toxicity and benthic community would indicate unlikely site-specific aquatic life impairment from site CoPCs, because there is no concurrence with site CoPCs. This does not mean that there is no impairment, but that the impairment is not clearly linked to site related contamination. The framework shown in Table 4-2 was used to interpret the results and is consistent with other published WOE frameworks.

Table 4-2. Weight of evidence analysis framework for the aquatic life impairment assessment. For each LOE (chemistry, toxicity and benthic community), the symbols indicate the degree of impact including low (O), moderate (⊙), or high (●).

Aquatic Life Impairment Table			
Chemistry	**Toxicity**	**Benthic Community**	**Site-specific Impairment from CoPCs**
●	●	●	Likely impairment from CoPCs
●	●	⊙	
●	⊙	●	
⊙	●	●	
●	●	O	
●	O	●	
●	⊙	⊙	
⊙	●	⊙	
⊙	⊙	●	
⊙	⊙	⊙	
●	⊙	O	
●	O	⊙	
⊙	●	O	Possible Impairment from CoPCs
⊙	O	●	
⊙	⊙	O	
⊙	O	⊙	
●	O	O	
O	●	●	Unlikely impairment from CoPCs
O	●	⊙	
O	⊙	●	
O	⊙	⊙	
O	O	●	
O	●	O	
O	O	⊙	
O	⊙	O	
⊙	O	O	
O	O	O	

4.2.3 Aquatic-Dependent Wildlife Impairment

A screening level risk assessment was performed to assess potential impairment to aquatic-dependent wildlife. For this assessment, bioaccumulation of CoPCs in the clam *Macoma nasuta* exposed to site sediments was used to estimate exposure for representative wildlife receptors including surface feeding birds and marine mammals. For the screening level assessment, conservative exposure assumptions included 100% dietary fraction from the site, 100% area use factor for the site, and the low toxicity reference value.

The screening level risk assessment for aquatic-dependent wildlife was based on the following procedure. First, chemical concentrations in clam tissue were compared to measurements made on control samples to detect the presence of contaminant bioaccumulation. Next, accumulation of each chemical at reference stations was compared (t-test, P=0.05) to accumulation measured at site stations. Clam tissue concentrations were then used to estimate contaminant doses to a range of representative wildlife receptors including surface feeding birds (Least Tern and Brown Pelican), diving birds (Surf Scoter and Western Grebe), and marine mammals (California Sea Lion). These receptors are common to San Diego Bay (U.S. Navy/SDUPD, 2000) and provide a breadth of potential exposure pathways and sensitivities to the CoPCs at the site. Although it is acknowledged that clams are not the primary food source for several of these receptors, these results provide a conservative assessment of impairment because the clams (*M. nasuta)* are surface deposit filter-feeders and are therefore directly exposed to CoPCs in the surface sediments. The maximum dosage calculated for each chemical at site stations was compared to the 95% upper predictive interval of tissue concentrations from the Baseline Pool.

For those chemicals with doses exceeding the TRV and tissue levels greater than the Baseline Pool, a station-by-station assessment was made following a similar procedure as described above, but using the individual station tissue concentration instead of the maximum concentration of all stations at the site. For stations where bioaccumulation was not measured, tissue concentrations were estimated based on site-specific Biota-Sediment Accumulation Factors (BSAFs) calculated from tissue and sediment concentrations at stations where bioaccumulation was measured. This analysis was used to develop a spatial description of potential aquatic-dependent wildlife impairment related to CoPCs.

Because the evaluation of aquatic-dependent wildlife is a highly conservative screening level assessment, sites or stations were assigned a relative likelihood of impairment ranging only from “unlikely” to “possible”. The category ranking criteria for site-specific aquatic-dependent wildlife impairment is summarized below. Note that within these classifications, the presence of risk (Hazard Quotient (HQ)>1) does not necessarily equate with site-specific aquatic dependent wildlife impairment, because impairment is also measured relative to the baseline condition.

Unlikely - Impairment to wildlife from the consumption of aquatic prey exposed to site sediments is unlikely for a CoPC if: (1) the bioaccumulation measured at the site is not statistically different than observed in controls or (2) the estimated HQ is less than 1 or (3) the bioaccumulation is not statistically different from the baseline condition.

Possible - Impairment to wildlife from the consumption of aquatic prey exposed to site sediments is possible for a CoPC if: (1) the bioaccumulation measured at the site is statistically different than observed in controls and (2) the estimated HQ is greater than 1 and (3) there is statistically different bioaccumulation relative to the baseline condition.

4.2.4 Human Health Impairment

The screening level risk assessment for human health followed a similar procedure as that described above for aquatic-dependent wildlife. Station bioaccumulation data were first compared to controls, then to published toxicity or cancer risk thresholds, and then to the baseline condition. First, chemical concentrations in clam tissue were compared to measurements made on control samples to detect the presence of contaminant

bioaccumulation. Stations with clam data showing no significant accumulation relative to controls were considered non-impacted.

For those stations with chemicals demonstrating bioaccumulation, clam tissue concentrations were used to estimate human ingestion doses based on conservative assumptions for uptake including 100% of seafood consumption from the site, 100% of seafood contaminated at the 95% upper confidence limit of all site stations, and a conservative seafood consumption rate. Estimated doses were then compared to EPA toxicity and cancer thresholds. For chemicals exceeding EPA human health thresholds, tissue concentrations of clams exposed to study site sediments were compared with the 95% upper predictive interval of tissue concentrations from clams in the Baseline Pool.

For those chemicals that exceeded EPA human health thresholds and had tissue levels greater than the Baseline Pool, a station-by-station assessment was made following the same procedure as described above, but using the individual station tissue concentration instead of the 95% upper confidence limit of all stations. For stations where bioaccumulation was not measured, tissue concentrations were estimated based on site-specific BSAFs calculated from tissue and sediment concentrations at stations where bioaccumulation was measured. This analysis was used to develop a spatial description of potential human health impairment related to CoPCs.

Because the evaluation of human health is a highly conservative screening level assessment, sites or stations were assigned a relative likelihood of impairment ranging from “highly unlikely” to “possible”. The category ranking criteria for site-specific human health impairment is summarized below. Note that within these classifications, the presence of risk does not necessarily equate with site-specific human health impairment, because impairment is also measured relative to the baseline condition.

Unlikely - Impairment to human health from the consumption of fish or shellfish exposed to site sediments is unlikely for a CoPC if: (1) the bioaccumulation measured at the site is not statistically different than observed in controls or (2) the concentration in the fish or shellfish is less than the screening level tissue screening level (TSL) or (3) the bioaccumulation is not statistically different from the baseline condition.

Possible - Impairment to human health from the consumption of fish or shellfish exposed to site sediments is possible for a CoPC if: (1) the bioaccumulation measured at the site is statistically different that observed in controls and (2) the concentration in the fish or shellfish is greater than the TSL and (3) there is statistically different bioaccumulation relative to the baseline condition.

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5.0 METHODS

5.1 SAMPLING SUMMARY

Near-surface sediments were collected and analyzed from 14 stations within the SUBASE area of concern (Figure 4-1) and at six reference stations (Figure 4-2). The reference stations were sampled twice, once during a reconnaissance survey conducted 12 February 2004 and again during the main survey 13 and 14 April 2004. All stations sampled during this study were given a "SB" prefix designator (e.g. SB2229 for reference station 2229) to ensure uniqueness with other data collections made at these sites. Stations collected during the reconnaissance survey were also given a "-R" suffix. All final station locations (averaged for each grab and core drop) are shown in Table 5-1.

The stations were more or less uniformly-spaced throughout the full SUBASE region, extending outside of the 303D listed area. The pattern provided complete spatial coverage of the area of concern so that spatial gradients and patterns could be clearly delineated. The total area represented by these stations was approximately 60 acres. Sediment chemistry, toxicity, and benthic community data were collected at all site stations. Bioaccumulation data were collected at a subset of seven stations that formed along the expected main contaminant gradient away from shore between the south (Sierra) and Middle (Mike) piers and along the shoreline (Figure 4-1). As described previously, the six reference stations were chosen based on a review of historical data and only included stations from the north portion of the bay.

5.2 FIELD METHODS

All 14 site stations and all six reference stations were sampled during the main sampling event in April 2004. Sampling included 46 sediment grabs analyzed for chemistry and toxicity, 21 grabs collected for benthic community analysis, and 25 cores collected for sediment water interface tests. All six reference stations were also sampled during the reconnaissance survey conducted on 2 February 2004. One sediment grab was collected from each station and analyzed for chemistry and amphipod toxicity to ensure that reference conditions would match typical historic characteristics.

The general sampling chronology for the main sampling event in April was to perform the site sampling in station order along with nearby reference station 2441 on the first day followed by the remaining reference stations on the second day. Reference station 2441 was resampled on the second day after an oil sheen was observed in the grabs made during the first day. The location for the second sampling was moved slightly to the north of the orginal sampling location.

The first full grab at a station was used for benthic community analysis. A second grab was used to collect four 5-cm cores for the sediment water interface bioassays by placing them into visually undisturbed portions of the grab's sediment surface. The top five centimeters of sediment from the unused portion of this grab, along with the top five centimeters of sediment from any additional grabs, were composited until sufficient quantities of sediment were collected for chemistry and additional bioassays.

All field sampling was performed aboard the US Navy's RV ECOS with personnel from SSC-SD. Sample locations were determined using a differential Global Positioning Navigation System (Trimble Model 4000 RLII+NavBeacon XL) with an accuracy of 1 to 3 meters. The navigation

antenna was positioned directly above the samplers during use. Water depths were determined with a digital fathometer (InnerSpace Model 445) with a resolution of 0.1 m.

5.2.1 Sediment Collection - Grabs

Bulk sediment was collected at all stations using a 0.1 m^2 Van Veen grab sampler with a closed top. The top five centimeters of sediment in a grab was scooped out with a plastic scoop. Multiple grabs, ranging from one to four, were collected at each site to supply enough sediment for all analyses planned for the particular site. Sediment from the multiple grabs was combined and homogenized by placing it in a large plastic bowl and manually stirring with a plastic spoon. At stations where field replicates were collected, each replicate was homogenized separately. Large shells, rocks, plastic, or other large debris were manually excluded from the samples. The homogenized sediment was then split into multiple pre-cleaned glass jars or plastic containers depending on the type of analysis. The sample splits were as follows: 0.5 L for grain size, TOC, and metal chemistry, 0.5 L for PAHs, 0.5 L for PCBs and chlorinated pesticides, and 3 L for toxicity tests. All samples were immediately placed on ice and kept cold until arrival at the analytical laboratory.

Personnel handling the sediments all wore precleaned plastic gloves. All sampling materials were cleaned with site water before and after each grab. All scoops, spoons, and bowls were cleaned with alconox and rinsed with site water prior to sampling a new station.

5.2.2 Sediment Collection - Cores

All core tubes were pre-cleaned in a series of soap wash, 10% nitric acid soak, and methanol rinse. Distilled water was used for the in-between and final rinse. Cores were obtained by opening the hinged “doors” on the top of the grab and ensuring that the surface was visually undisturbed. Four cores were hand pushed down through the sediment until the bottom of the grab was encountered. After the surrounding surface sediments in the grab were collected for composites, the deeper sediment around the cores was pushed aside and end caps were placed on the bottom and top of the cores. The cores were then removed, cleaned of mud, and end caps were taped.

Cores ranged from 4 to 7 cm long and averaged 5.2 cm long. The cores were placed into coolers with specially built holders to maintain them in an upright position and kept cool until arrival at the laboratory for analysis.

5.2.3 Benthic Community Organism Collection

Benthic organisms were collected using a 0.1 m^2 Van Veen grab sampler with a closed top. All sediment from a single grab was dumped into a 1.0 mm screened box and the sediment washed out using site water. All organisms remaining within the screen were manually removed, placed into 1-L plastic jars containing a $MgSO_4$ relaxant solution, and preserved using 10% sodium borate buffered formalin.

Table 5-1. Average station locations for the reconnaissance (-R suffix) and main surveys. Repeat grabs were within 4 m of each other (relative standard deviation (RSD)).

Sample ID	Latitude	Longitude
SB2229-R	32.7089	-117.1758
SB2433-R	32.7225	-117.2095
SB2436-R	32.7150	-117.1829
SB2441-R	32.6918	-117.2375
SB90056-R	32.7193	-117.2174
SBC001SS31-R	32.7233	-117.2145
SB01	32.6905	-117.2389
SB02	32.6908	117.2381
SB03	32.6910	-117.2357
SB04	32.6892	-117.2382
SB05	32.6899	-117.2370
SB06	32.6903	-117.2359
SB07	32.6908	-117.2348
SB08	32.6880	-117.2374
SB09	32.6884	-117.2366
SB10	32.6890	-117.2353
SB11	32.6895	-117.2341
SB12	32.6866	-117.2356
SB13	32.6871	-117.2346
SB14	32.6876	-117.2335
SB2229	32.7089	-117.1760
SB2433	32.7224	-117.2095
SB2436	32.7151	-117.1831
SB2441	32.6922	-117.2380
SB90056	32.7192	-117.2174
SBC001SS31	32.7233	-117.2145

5.3 ANALYTICAL METHODS

5.3.1 Sediment Grain Size and Total Organic Carbon

Grain Size. Sediment samples were analyzed for grain size by Battelle's Sequim, WA laboratory. Samples were analyzed for grain size according to the methods of Plumb (1981). Samples are wet sieved through a No. 230 (0.0625 mm) U.S. Standard Sieve. The fine fraction (silt and clay) is collected in a 1-L graduated cylinder. Sediment retained on the No. 230 sieve is washed with distilled water into labeled, pre-weighed beakers and oven-dried for 24 hours at 105 ^{o}C. After drying, the soil is sieved using a No. 10 (2.00 mm) sieve to determine the percent gravel, and a No. 230 (0.0625 mm) sieve to determine percent sand by weighing. Sediment passing the No. 230 sieve is added to the fine fraction in a graduated cylinder. The fine fraction is stirred and aliquots taken to determine the percent silt (0.0625 mm to 0.0039 mm) and clay (<0.005 mm) using hydrometers as described in ASTM D-422 (1990).

TOC. Sediment samples were analyzed for TOC by Battelle's Sequim, WA laboratory. Samples were analyzed for TOC following procedures described in EPA 9060 (USEPA, 1981). In this method samples are dried, homogenized, and then acidified to remove carbonates and

bicarbonates. The samples are then combusted in a high-temperature furnace in a stream of oxygen to form carbon dioxide (CO_2). Interferents such as halogens, sulfur, nitrogen oxides, and water, were removed by chemical scrubbers prior to CO_2 quantification. Carbon dioxide is measured by sweeping the gas stream into a coulometer cell. The coulometer cell is filled with a partially aqueous medium containing ethanolamine and a colorimetric indicator. Carbon dioxide is quantitatively absorbed by the solution and is quantified by titration of the ethanolamine with strong acid until the indicator color fades.

5.3.2 Sediment Chemical Contamination

Bulk sediments and tissues collected as part of the bioaccumulation testing were analyzed for a suite of metals, PAHs, PCBs, and chlorinated pesticides using low-level detection EPA methods. The complete list of analytes is shown in Table 5-2 through Table 5-6. A variety of summed analyte lists are also used when evaluating contamination. These include the sum of all PAH analytes referred to here as Total PAHs (TPAH), the sum of PAHs on the EPA's priority pollutant list (PPPAH), the sum of high molecular weight PAH analytes (HMWPAH), the sum of low molecular weight PAH analytes (LMWPAH), the sum of all PCB congeners referred to as Total PCBs (TPCBs), the sum of the two Chlordane analytes referred to here as Total Chlordane (TCHLOR), and the sum of all DDT and its breakdown products DDE, and DDD referred to here as Total DDT (TDDT). The specific analytes making up these summed lists are shown in their respective tables. These summed lists may vary slightly from those in other studies because of differences in the number and kind of analytes measured. A brief description of methods for each category of contaminant is described below.

Metals. Sediment samples were analyzed for the metals shown in Table 5-2 at Battelle's Sequim, WA laboratory. Samples were digested using a strong acid (total metals) digestion technique (NOAA, 1998). All metals, except mercury, selenium, and silver were analyzed by either inductively coupled plasma mass spectrometry following EPA Method 200.8 or inductively coupled plasma atomic emission spectroscopy EPA Method 200.7. Silver was analyzed by graphite furnace atomic absorption EPA Method 200.9. Mercury was analyzed by cold vapor atomic absorption following modified EPA Method 245.5. Selenium was analyzed by hydride atomic absorption using flow injection.

Table 5-2. The complete list of metal analytes measured in bulk sediments and in tissues.

Metal	Symbol	Metal	Symbol
Aluminum	Al	Manganese	Mn
Arsenic	As	Mercury	Hg
Cadmium	Cd	Nickel	Ni
Chromium	Cr	Selenium	Se
Copper	Cu	Silver	Sag
Iron	Fe	Tin	Sn
Lead	Pb	Zinc	Zn

PAHs. Sediment samples were analyzed for the PAHs shown in Table 5-3 at Battelle's Duxbury, MA laboratory. Sediment samples were extracted for semivolatile organic compounds following standard EPA methods. The extraction procedure allowed for the simultaneous extraction of PAHs, PCBs, and chlorinated pesticides. After homogenization, a 30- to 50-g aliquot of each sample was transferred into a Teflon® jar along with ~60 g of sodium sulfate, 100 mL of 50:50 dichloromethane/acetone, and then spiked with surrogate compounds.

After a three-minute sonication the sample was centrifuged and the organic solvent layer was decanted into a flask. This extraction procedure was repeated 2 more times with fresh aliquots of solvent. After the third sonication, the sample jar was placed on an orbital shaker for 1 hour prior to the final centrifuge.

The three solvent extracts were combined and water was removed by adding approximately 75 g of sodium sulfate. Copper, alumina column, and high-pressure liquid chromatography (HPLC) cleanups were performed on the sample extracts to remove potential contamination that would interfere with sample analysis. All extracts were concentrated to approximately 1 mL using kuderna-danish concentrators and nitrogen evaporation. Extracts were split into archive and working volumes. The working extract volume was further split: one-half was designated for PAH analysis and one-half was exchanged into hexane for PCB/Pesticide analyses (see below).

The sample extracts were analyzed for PAHs by a modified version of EPA's SW-846 Method 8270. The gas chromatograph/mass spectrometer (GC/MS) was operated in selected ion monitoring (SIM) mode to obtain the desired sensitivity that is comparable to that of a GC equipped with an electron capture detector. The GC/MS was tuned with perfluorotributylamine to verify accurate mass assignment and to maximize the sensitivity of the instrument in the mass range of interest (100 to 300 atomic mass units). Average response factors for each target compound and surrogate were calculated from initial calibration standards relative to internal standard compounds added to the sample extracts just prior to instrumental analysis (internal standardization). Calibration standards were analyzed on regular intervals to monitor sensitivity and linearity of the GC/MS. The average response factors generated from the calibrations were used to calculate the concentrations of target compounds and surrogates. The recoveries of the surrogate compounds spiked into the sample prior to extraction were used to assess sample-specific extraction efficiency. Target compound concentrations were surrogate corrected based on sample-specific surrogate recoveries to correct for differences in extraction efficiency.

A full suite of quality control samples were prepared for every analysis batch including a procedural blank, blank spike, blank spike duplicate, matrix spike, matrix spike duplicate, duplicates, and standard reference material.

As mentioned previously, some PAH analytes were summed to evaluate them against various SQGs. The summations include the following, using the analyte short names shown in Table 5-3:

Low Molecular Weight PAH (LMWPAH) = sum (CON, ACEY, ACE, COF, COA, COP)

High Molecular Weight PAH (HMWPAH) = sum (FLANT, PYR, BAA, COC, BBF, BAP, DAA)

Priority Pollutant PAH (PPPAH) = sum (LMWPAH, HMWPAH, BKF, INDENO, BGP)

Consensus Based PAH (CBPAH) = sum (LMWPAH, HMWPAH)/TOC

where TOC= Total organic carbon

Table 5-3. The complete list of PAH analytes with short ID name measured in bulk sediments and in tissues.

Analyte	ID	Analyte	ID
Naphthalene	**C0N**	Dibenzothiophene	**C0D**
C1-Naphthalenes	**C1N**	C1-Dibenzothiophenes	**C1D**
C2-Naphthalenes	**C2N**	C2-Dibenzothiophenes	**C2D**
C3-Naphthalenes	**C3N**	C3-Dibenzothiophenes	**C3D**
C4-Naphthalenes	**C4N**	C4-Dibenzothiophenes	**C4D**
2-Methylnaphthalene	**2MN**	Fluoranthene	**FLANT**
1-Methynaphthalene	**1MN**	Pyrene	**PYR**
Biphenyl	**BIP**	C1-Fluoranthenes/Pyrenes	**C1F/P**
2,6-dimethylnaphthalene	**26N**	C2-Fluoranthenes/Pyrenes	**C2F/P**
Acenaphthylene	**ACEY**	C3-Fluoranthenes/Pyrenes	**C3F/P**
Acenaphthene	**ACE**	Benzo(a)anthracene	**BAA**
2,3,5-trimethylnaphthalene	**235N**	Chrysene	**C0C**
Dibenzofuran	**DBF**	C1-Chrysenes	**C1C**
Fluorene	**C0F**	C2-Chrysenes	**C2C**
C1-Fluorenes	**C1F**	C3-Chrysenes	**C3C**
C2-Fluorenes	**C2F**	C4-Chrysenes	**C4C**
C3-Fluorenes	**C3F**	Benzo(b)fluoranthene	**BBF**
Anthracene	**C0A**	Benzo(j/k)fluoranthene	**BKF**
Phenanthrene	**C0P**	Benzo(e)pyrene	**BEP**
C1-Phenanthrenes/Anthracenes	**C1P/A**	Benzo(a)pyrene	**BAP**
C2-Phenanthrenes/Anthracenes	**C2P/A**	Perylene	**PER**
C3-Phenanthrenes/Anthracenes	**C3P/A**	Indeno(1,2,3-cd)pyrene	**INDENO**
C4-Phenanthrenes/Anthracenes	**C4P/A**	Dibenz(a,h)anthracene	**DAA**
1-Methylphenanthrene	**1MP**	Benzo(g,h,i)perylene	**BGP**

PCBs. Sediment samples were extracted for PCBs simultaneously with PAHs as described above. The extracts were analyzed for PCB congeners (Table 5-4) and simultaneously measured chlorinated pesticides (Table 5-5). The PCB congener analysis method is a modified version of EPA's SW-846 Method 8081 using dual, dissimilar columns and dual detectors. A Restek RTX-5 column (or equivalent) was used as the primary column and a DB-17 column (or equivalent) was used as the confirmation column. Average calibration factors for each target compound and surrogate were calculated from initial calibration standards (external standardization). Calibration standards were analyzed on regular intervals to monitor sensitivity, retention time stability, and linearity of the Gas Chromatograph/Electron Capture Detector (GC/ECD).

Average calibration factors generated from the calibrations were used to calculate target compound concentrations. When co-elution occurred between one or more target compounds or when interference occurred on the primary column, the results were reported from the confirmation column for the affected compounds. Compound identification was based on 1) detecting a peak within the established retention time window for a specific compound on both the primary and confirmation columns, and 2) the analyst's judgment. The recoveries of the surrogate compounds spiked into the sample prior to extraction were used to assess sample-specific extraction efficiency. Target compound concentrations were surrogate

corrected based on sample-specific surrogate recoveries to correct for differences in extraction efficiency.

A full suite of quality control samples were prepared for every analysis batch including a procedural blank, blank spike, blank spike duplicate, matrix spike, matrix spike duplicate, duplicates, and standard reference material.

Eighteen PCB congeners were summed to evaluate them against various SQGs. These include the following, using the congener numbers identified in Table 5-4:

Total PCB (TPCBs) =sum(8,18,28,44,52,66,101,105,18,128,138,153,170,180,187,195,206, 209)

Consensus Based PCB (CBPCB) = TPCB/TOC

Chlorinated Pesticides. Sediment samples were extracted for chlorinated pesticides simultaneously with PAHs and PCBs as described above. The extracts were analyzed for chlorinated pesticides shown in Table 5-5 simultaneously with PCBs. The analytical method is described above in the PCB section. Two summations of pesticides were used for comparison to various SQGs. These include the following:

Total DDT (TDDT) = sum(2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT)

Total Chlordane (TCHLOR) = sum(γ-Chlordane, α-Chlordane, cis-nonachlor, trans-nonachlor, oxyChlordane)

Table 5-4. The complete list of PCB congeners measured in bulk sediments and in tissues.

Congener Number	PCB Congener	Congener Number	PCB Congener
18	2,2',5-Trichlorobiphenyl (Cl3)	128	2,2',3,3',4,4'-Hexachlorobiphenyl (Cl6)
28	2,4,4'-Trichlorobiphenyl (Cl3)	138	2,2',3,4,4',5'-Hexachlorobiphenyl (Cl6)
37	3,4,4'-Trichlorobiphenyl (Cl3)	149	2,2',3,4',5',6-Hexachlorobiphenyl (Cl6)
44	2,2',3,5'-Tetrachlorobiphenyl (Cl4)	151	2,2',3,5,5',6-Hexachlorobiphenyl (Cl6)
49	2,2',4,5'-Tetrachlorobiphenyl (Cl4)	153	2,2',4,4',5,5'-Hexachlorobiphenyl (Cl6)
52	2,2',5,5'-Tetrachlorobiphenyl (Cl4)	156	2,3,3',4,4',5-Hexachlorobiphenyl (Cl6)
66	2,3',4,4'-Tetrachlorobiphenyl (Cl4)	157	2,3,3',4,4',5'-Hexachlorobiphenyl (Cl6)
70	2,3',4',5 -Tetrachlorobiphenyl (Cl4)	158	2,3,3',4,4',6-Hexachlorobiphenyl (Cl6)
74	2,4,4',5 -Tetrachlorobiphenyl (Cl4)	167	2,3',4,4',5,5'-Hexachlorobiphenyl (Cl6)
77	3,3',4,4'-Tetrachlorobiphenyl (Cl4)	168	2,3',4,4',5',6-Hexachlorobiphenyl (Cl6)
81	3,4,4',5 -Tetrachlorobiphenyl (Cl4)	169	3,3',4,4',5,5'-Hexachlorobiphenyl (Cl6)
87	2,2',3,4,5'-Pentachlorobiphenyl (Cl5)	170	2,2',3,3',4,4',5-Heptachlorobiphenyl (Cl7)
99	2,2',4,4',5-Pentachlorobiphenyl (Cl5)	177	2,2',3,3',4',5,6-Heptachlorobiphenyl (Cl7)
101	2,2',4,5,5'-Pentachlorobiphenyl (Cl5)	180	2,2',3,4,4',5,5'-Heptachlorobiphenyl (Cl7)
105	2,3,3',4,4'-Pentachlorobiphenyl (Cl5)	183	2,2',3,4,4',5',6-Heptachlorobiphenyl (Cl7)
110	2,3,3',4',6-Pentachlorobiphenyl (Cl5)	187	2,2',3,4',5,5',6-Heptachlorobiphenyl (Cl7)
114	2,3,4,4',5-Pentachlorobiphenyl (Cl5)	189	2,3,3',4,4',5,5'-Heptachlorobiphenyl (Cl7)
118	2,3',4,4',5-Pentachlorobiphenyl (Cl5)	194	2,2',3,3',4,4',5,5'-Octachlorobiphenyl (Cl8)
119	2,3',4,4',6-Pentachlorobiphenyl (Cl5)	201	2,2',3,3',4,5',6, 6'-Octachlorobiphenyl (Cl8)
123	2',3,4,4',5-Pentachlorobiphenyl (Cl5)	206	2,2',3,3',4,4',5,5',6-Nonachlorobiphenyl (Cl9)
126	3,3',4,4',5-Pentachlorobiphenyl (Cl5)	209	2,2',3,3',4,4',5,5',6,6'-Decachlorobiphenyl (Cl10)

Table 5-5. The complete list of chlorinated pesticide analytes measured in bulk sediments and in tissues.

Analyte	Analyte
2,4'-DDD	trans-nonachlor
2,4'-DDE	oxychlordane
2,4'-DDT	dieldrin
4,4'-DDD	endosulfan I
4,4'-DDE	endosulfan II
4,4'-DDT	endosulfan sulfate
aldrin	endrin
a-chlordane	endrin aldehyde
g-chlordane	endrin ketone
a-BHC	heptachlor
b-BHC	heptachlor epoxide
d-BHC	hexachlorobenzene
lindane	methoxychlor
cis-nonachlor	mirex

Organotin. Sediment samples were analyzed for organotin compounds using a modified version of EPA's SW-846 Method 8270. The complete list of analytes is shown in Table 5-6. The analysis was performed using a gas chromatograph/mass spectrometer (GC/MS) operated in selected ion monitoring (SIM) mode to obtain the desired sensitivity. Average response factors for each target compound and surrogate were calculated from initial calibration standards relative to internal standard compounds added to the sample extracts just prior to instrumental analysis (internal standardization). Calibration standards were analyzed on regular intervals to monitor sensitivity and linearity of the GC/MS. Average response factors generated from the calibrations were used to calculate the concentrations of target compounds and surrogates. Recoveries of the surrogate compounds spiked into the sample prior to extraction were used to assess sample-specific extraction efficiency.

Table 5-6. The complete list of organotin analytes measured in bulk sediments and in tissues.

Analyte	ID
Tetrabutyltin	TTBT
Triutyltin	TBT
Dibutyltin	DBT
Monobutyltin	MBT

5.3.3 Bioaccumulation

Organism Exposure. *In situ* clam bioaccumulation methods similar to those used for the Bravo Pier risk assessment (U.S. Navy, 1999) were adopted to provide realistic exposure conditions. Exposing the organisms *in situ* was however, a change from the laboratory exposure method used in the more recent sediment investigations conducted the bay (e.g., SCCWRP and Navy, 2005). Clams, *Macoma nasuta,* were housed in vented-plastic trays and deployed at reference

stations, SUBASE locations, and in non-vented trays containing home sediment in flow-through seawater tanks. The clams were retrieved after 28 days of exposure, removed from the trays and sediment and allowed to depurate for 24 hours in mesh bags suspended in the water column. Following depuration the clams were placed in pre-cleaned glass jars, frozen, and sent to the lab for processing and chemical analysis.

Field deployed clams were housed in vented high-density polyethylene trays. The trays measured 30.5 x 48 x 14 cm outside dimensions, 35 x 56 x 15.25 cm inside dimensions, with 0.8 cm side-vents running from top to bottom and 0.8 cm square openings on the bottom. Once 30 clams had been placed inside the trays the top was covered with a plastic grating with 1.3 cm square openings secured with nylon ties. To leach contaminant byproducts of manufacturing both the trays and tops were placed in seawater for 5 days before animals were placed in them. Following receipt of the clams from Northern California, control trays and clams were placed in non-vented trays of the same design for a week long acclimation/holding period. Home sediments were placed in these trays to a depth of 2-3 inches and the trays with clams were placed in flow-through seawater tanks.

Clams 5-8 cm in length in home seawater were shipped overnight express from Dillon Beach, CA to San Diego, CA. Home sediment was shipped in separate containers. Upon arrival air was diffused into the holding seawater. Holding seawater temperature was 15^{o}C and the flow-through seawater temperature 17^{o}C. Over a period of 4 hours by the removal of 12 liters home seawater (~25%) and addition of 12 liters of flow-through seawater, the clams were adjusted to flow-through conditions and 5 hours after receipt the clams were placed in home sediment trays in the flow-through tanks. The clams were examined 12 hours later and more than 99% of the clams were found to have burrowed into the sediment and appeared healthy. The clams were held this way for 5-7days prior to field deployment.

Clams were deployed in the field by SCUBA divers over a 3-day period, April 27-29, 2004. GPS coordinates were followed to each station and a marker buoy set to mark the site. Three trays each containing 30 clams were placed at each station. Divers dug a hole in the sediment the size of the tray, ~10 cm deep, and the tray was placed into the hole. Sediment moved for placement of the tray was set on top and worked through the top screen into the tray. The trays were placed in a radial pattern ~2-3 meters apart. It should be noted that at the sites of cage deployment the divers were able to see the grab sites where sediment samples had been collected 2 weeks earlier for chemical analysis.

Test organisms were recovered at exposure termination 28 days later by gently sieving test sediments through a 0.75-mm stainless steel screen. All surviving clams were counted and placed in sediment-free, flow-through aquaria under test conditions for a period of 24 hours to allow the organisms to purge their gut contents. Following gut purging, the animals from each treatment were placed in clean glass jars with Teflon®-lined lids, frozen, packaged with dry ice in sealed coolers and then sent overnight to Battelle for chemical analysis.

Tissue Analysis. Tissue samples were extracted for semi-volatile organic compounds following standard EPA methods. Samples were macerated at high speed for 2 minutes using a tissue extraction probe. After homogenization, a 5- to 15-g aliquot of tissue sample was transferred into a Teflon® jar along with ~60 g of sodium sulfate, 100 mL of dichloromethane, and then spiked with surrogate compounds. The remainder of the sample preparation and analysis follows that described above for sediments.

5.3.4 Sediment Toxicity

Bulk Sediment. The amphipod survival test (USEPA, 1994) was used to evaluate toxicity of the whole sediment samples. The amphipods, *Eohaustorius estuarius*, were collected by Northwest Aquatics from a clean site near Yaquina Bay in Newport, Oregon. The animals were held in the laboratory on their native (home) sediment for one to four days before testing began, and slowly acclimated to the 20 g/kg testing salinity. The tests were conducted in 1-L Mason jars containing 2 cm of sediment (approximately 150 mL) and 750 mL of water. Five replicates were used for each sample. The overlying water was adjusted to a salinity of 20 g/kg, and the exposures conducted at 15 °C. The sediment was added to the jars and overlying water added with aeration one day before the animals were added, in order to provide a 24-h equilibration period. After equilibration, 20 amphipods were added to each beaker for an exposure period of 10 days. The beakers were monitored daily for visible changes to the sediment or death of the animals. At the end of the exposure period, the sediment from the beakers was passed through a sieve to recover the animals, and the number of surviving animals counted. Samples of amphipod home sediment were tested as negative controls. Water quality parameters (temperature, pH, dissolved oxygen, ammonia, and salinity) were measured on the porewater and overlying water of a surrogate water quality beaker that was destructively sampled at test initiation. Final water quality was assessed by carefully sampling from one of the five remaining test replicates.

Pore Water. The echinoderm (purple sea urchin) embryo-larval development test (USEPA, 1995; ASTM, 1999) was used to evaluate porewater toxicity. Toxicity was assessed based on the presence of normally developed larvae following exposures of embryos for 96 hours. The porewater was extracted by centrifuging the sediment at 3000 g for 30 min. Porewater was extracted immediately upon sample arrival in the laboratory, and was stored for 9 days in the dark at 4 °C with no head space prior to test initiation. The purple sea urchins (*Strongylocentrotus purpuratus*) used in the tests were collected from the intertidal zone along the North Jetty near the mouth of Mission Bay, San Diego, CA. Following laboratory spawning and fertilization of gametes, embryos were exposed to 25, 50, or 100% porewater diluted with filtered seawater from the research pier at Scripps Institute of Oceanography (SIO). The porewater was not filtered prior to exposure. At the end of the exposure, larvae were preserved in formalin, and then examined with an inverted microscope to assess normal development. Toxicity was expressed using the normal survival endpoint, which is defined as the total number of all added embryos (estimated by initial density vials) achieving the pluteus stage of development. Criteria for classification of normal pluteus larvae included pyramidal shape, four well-developed skeletal rods, and a well-defined gut. The tests were conducted in seawater leached glass scintillation vials containing 10 mL of solution at a temperature of 15 °C. Four replicates were tested for each sample. A filtered-seawater blank was included as a negative control. A copper reference toxicant test served as a positive control. An additional toxicity test using ammonia served as a basis for assessing whether observed toxicity might be associated with ammonia.

Sediment-Water Interface. The sediment-water interface (SWI) samples were tested using the bivalve embryo-larval development test (USEPA, 1995). This test measures the ability of the mussel larvae to develop normally from a fertilized egg in test media. The mussels (*Mytilus galloprovincialis*) used in the tests were shipped overnight from Carlsbad Aquafarm (Carlsbad, CA) and spawned immediately upon arrival in the laboratory. The SWI samples were tested following procedures developed by Anderson et al (1996). Briefly, the overlying water in each core tube was first replaced with clean filtered seawater with aeration. Four replicate cores were used for each sediment sample. After equilibration for 24 hours, a polycarbonate cylinder

with a fine mesh (25 µm) screen bottom (screen tube) was placed on the sediment inside the core tube. The adult mussels were induced to spawn, the gametes were collected, and then the eggs were fertilized. The eggs were added to the screen tube within 4 hours of fertilization, and given 48 hours to develop at 15 °C. After the exposure period, the screen tubes were removed from the sediment and the outside rinsed to remove any adhering sediment. The embryos were then rinsed into glass scintillation vials and preserved and evaluated under an inverted microscope to determine if normal development had occurred. The endpoint for this assay is a combined endpoint referred to as normal survival. Normal survival is defined as the total number of normally developed (straight-hinged, D-shaped) mussel larvae relative to the number of embryos initially added to each screen tube. A core tube blank (core with no sediment added) was included as a negative control. A copper reference toxicant test served as a positive control. Water quality parameters (temperature, pH, dissolved oxygen, ammonia, and salinity) were measured on the overlying water of one replicate per sample at both the beginning and end of the exposure period.

5.3.5 Benthic Community Analysis

Benthic grab sampling was conducted in accordance with *Techniques for Sampling and Analyzing the Marine Macrobenthos* March 1978, EPA 600/3-78-030; *Quality Assurance and Quality Control (QA/QC) for 301 (h) Monitoring Programs: Guidance on Field and Laboratory Methods* May 1986, Tetra Tech; and the laboratory and field methods guides developed by the Southern California Regional Survey Committees (SCCWRP 1994, 1998, 2003). Field sampling, sorting, and analysis was conducted by Aquatic Bioassay and Consulting Laboratories, Ventura, CA.

Samples were collected with a chain-rigged, one-tenth square-meter Van Veen Grab. At each station, the grab was lowered rapidly through the water column until it was near the bottom, and then slowly lowered until contact was made. The grab was then carefully raised until clear of the bottom. Once on board, the grab was drained of water using a siphon. Initial qualitative observations of color, odor, consistency, etc. were recorded. Sample acceptance was based on criteria specified in the Southern California Bight Regional Survey protocols (2003). Sediment samples collected at Stations 13 and 22, in Oxford Lagoon, were collected by hand using a clean plastic scoop.

Sediments to be analyzed for infauna content were sieved through a 1.0 millimeter screen. The retained organisms and larger sediment fragments were then washed into one-liter or four-liter plastic bottles (as needed), relaxed with magnesium sulfate, and preserved with 10% buffered formalin. Samples were transferred to 90% ethanol within one week of return to the laboratory.

The infauna in each sample was sorted into major phylogenic groups using dissecting microscopes. Ten percent of each sample was QC'd by the lab supervisor, who ordered a resort of the sample if a 95% sorting efficiency was not achieved. Identifications were conducted by taxonomists who are active in the Southern California Association of Marine Invertebrate Taxonomists (SCAMIT). Any naming discrepancies or difficult organisms were reviewed with other SCAMIT taxonomists.

Analysis of the data fell into four categories: comparison of species abundances among species, cluster analysis of species assemblages, evaluation of community characteristics, and calculation of the magnitude of community disturbance. The species abundance data (number of individuals/grab) was summed within each of the three station types (reference, Chollas, and Paleta) and ranked to determine the most common species. The abundance of four indicator species for each station was also compared. The indicator species included two polychaete

worms (*Capitella capitata* and *Streblospio benedicti*), an ostracod (*Euphilomedes carcharodonta*), and amphiuridae (brittlestars).

Cluster analysis of the stations was conducted using flexible sorting of Bray-Curtis dissimilarity values with ß=-0.25 (Bray and Curtis 1957, Lance and Williams 1967, Clifford and Stephenson 1975). The abundances were square root transformed and then standardized by the species mean of values higher than zero to reduce the influence of dominant species (Smith 1976, Smith et al., 1988). The step-across distance re-estimation procedure (Williamson 1978, Bradfield and Kenkel, 1987) was applied to dissimilarity (distance) values over 0.80 to reduce the distortion of ecological distances caused by joint absences of a high proportion of species; the distortion occurs due to the common non-monotonic truncated nature of species distributions along environmental gradients (Beals, 1973). Prior to cluster analysis, species contributing little information were excluded by eliminating species occurring at fewer than 5 sites.

Three metrics were calculated in order to describe the overall characteristics of the macrofaunal community: abundance, number of taxa, and Shannon-Wiener diversity (using natural logarithms) (Pielou, 1969).

The magnitude of disturbance shown by the benthic assemblage at each station was described using the embayment Benthic Response Index (BRI). The embayment BRI measures the abundance-weighted pollution tolerance of the species present (Ranasinghe et al., 2003) and is based on a similar index developed for coastal assemblages (Smith et al., 2001). Both indices define five levels of biotic response along a pollution gradient. The response levels were based on the loss of 5-25%, 25-50%, 50-80% and >80% of potential species. The BRI is a measure of the magnitude of disturbance, but cannot determine the cause of the disturbance because natural and anthropogenic factors may affect the benthos in a similar manner.

6.0 DATA QUALITY RESULTS

6.1 SEDIMENT AND BIOACCUMULATION CHEMISTRY

Battelle Laboratory was the primary contractor hired to perform all chemical analyses on sediments and tissues. They performed all analyses in-house while their sub-contractor, AMS, Inc. of Texas performed all grain size and TOC analyses. The chemistry contract identified all data quality objectives (DQO) including the use of low detection limit methods. The project DQOs are shown in Table 6-1 through Table 6-3. The laboratories each conducted their own internal quality assurance/quality control (QA/QC) evaluations to address whether or not DQO were met based on chain-of-custody, sample temperature and holding time, blank and blank-spike duplicates, sample analysis duplicates, surrogate recoveries, matrix-spike and matrix-spike duplicates, reference material analyses, instrument calibrations, and internal reference standards. The laboratories generated reports that identified all instances when data were outside the DQO for the project and identified what corrective actions were taken, if any. These narratives are included in the appendices along with the corresponding data tables.

All chemistry results including the narrative reports were reviewed at SSC-SD. All grain size and TOC measurements met the project DQO. For the most part the chemistry data met the project DQOs and low detection requirements. Most of the sediment metals data are unqualified, with only a few of the samples showing selenium as non-detect (U qualified). For the sediment organic contaminant data, there were more qualified data, but this was expected and did not affect study results. The only unusual qualified organic contaminant sediment data were the ME (matrix interference leading to an estimated value) qualified data on all PCB congener 44 and 49 analyses. This was due to a matrix interference that was present on all analyses and might lead to slightly overestimated values for these congeners. Again it is not expected that these slight deviations from normal analyses will have any impact on the study results. The tissues showed more qualified data than the sediments, but again this was expected and did not impact the study results.

Table 6-1. Data Quality Objectives and Criteria for metal analyses. One laboratory duplicate was run within each batch with a QC limit of ±30%

Metal	Reference Method	Range of Recovery	SRM Accuracy	Relative Precision	Target Detection Limit (µg/g)	Achieved Detection Limit (µg/g)
Aluminum	ICP-AES	70-130%	≤30%	≤30%	6	2.4
Antimony	ICP-MS	70-130%	≤30%	≤30%	0.2	0.03
Arsenic	ICP-MS	70-130%	≤30%	≤30%	0.1	0.07
Barium	ICP-AES	70-130%	≤30%	≤30%	0.01	0.02
Beryllium	ICP-MS	70-130%	≤30%	≤30%	0.01	0.02
Cadmium	ICP-MS	70-130%	≤30%	≤30%	0.01	0.02
Chromium	ICP-AES	70-130%	≤30%	≤30%	1	0.5
Copper	ICP-AES	70-130%	≤30%	≤30%	2	0.24
Iron	ICP-AES	70-130%	≤30%	≤30%	5	0.6
Lead	ICP-MS	70-130%	≤30%	≤30%	0.1	0.2
Mercury	CVAF	70-130%	≤30%	≤30%	0.001	0.002
Nickel	ICP-MS	70-130%	≤30%	≤30%	0.2	0.2
Selenium	FIAS	70-130%	≤30%	≤30%	0.01	0.067
Silver	GFAA	70-130%	≤30%	≤30%	0.3	0.08
Zinc	ICP-MS	70-130%	≤30%	≤30%	1	0.6

CVAF- Cold Vapor Atomic Absorption
FIAS- Flow Injection Atomic Absorption
GFAA- Graphite Furnace Atomic Absorption
ICP-AES- Inductively Coupled Plasma-Atomic Emission Spectrometry
ICP-MS- Inductively Coupled Plasma-Mass Spectrometry
SRM- Standard Reference Material

Table 6-2. Nominal method detection limits for PAH, PCB, and chlorinated pesticides analyses.

	PAH (µg/kg)	PCB (µg/kg)	Chlorinated Pesticides (µg/kg)
Sediment	0.05 – 0.18	0.02 – 0.06	0.02 – 0.07
Tissues	0.2 – 1.6	0.12 – 0.45	0.14 – 0.25

Table 6-3. Data Quality Objectives and Criteria, PAH Method 8270M-SIM, PCB Congener and Chlorinated Pesticide Method 8081A – modified.

Element or Sample Type	Minimum Frequency	Data Quality Objective/ Acceptance Criteria
Initial Calibration	Prior to every batch sequence.	5 point curve. %RSD ≤25% for 90% of analytes and ≤35% for all analytes.
Continuing Calibration	Must end analytical sequence and every 12 field samples or 16 hours, whichever is more frequent.	%RSD ≤25% for 90% of analytes. %RSD ≤35% for all analytes.
Procedural Blank	Every batch/every 20 field samples.	No more than 2 analytes to exceed 5x PQL unless analyte not detected in associated sample(s) or associated sample analyte concentration is > 10x blank value.
Blank Spike Sample	Every batch/every 20 field samples.	50-150% recovery, RPD ≤35%.
SRMs (SRM 1941a for sediment, 1974a for tissue).	Every sediment or tissue batch/every 20 field samples.	Values ±35% difference of true value for all certified analytes, two may exceed.
Matrix Spike, Matrix Spike Duplicate Sample	Every sediment or tissue batch/every 20 field samples.	45-150% recovery, RPD ≤35%.
Recovery/Surrogate Standards	Every Sample	40-125% d8-napththalene, d10-acenaphthene, d10-phenenthrene; 40-135% d12-benzo[a]pyrene; 40-125% DBOFB, PCB-103, PCB-198 with one out of criteria.
Instrumental SRM (SRM 1491)	One set per batch of samples after every ICAL.	Values ≤15% difference of true value for all certified analytes.
Control Oil (North Slope Crude)	One set per batch of samples after every ICAL (PAH only).	Values ≤35% difference of laboratory average values.

DBOFB- 1,2,3-Trichlorobenzene and 4,4'-Dibromooctafluorobiphenyl
ICAL- Instrument Calibration
PQL- Practical Quantitation Limit
RPD- Relative Percent Difference
RSD- Relative Standard Deviation
SRM- Standard Reference Material

6.2 TOXICITY

The toxicity test results were assessed for sediment holding time, testing methods, water quality conditions, negative control response, and positive control response (Table 6-4). Exceedance of a data quality objective did not automatically invalidate a test. Rather, the data were examined to see if the exceedance had affected the interpretation of the results.

6.2.1 Bulk Sediment

Most of the data quality objectives were met for the amphipod exposures to Reconnaissance sediments. The sediment holding time objective was easily met, with tests being initiated within 1 day of sample collection. Amphipods were acclimated over 3 days, also within acceptable limits. Amphipod survival in the control sediments was 98%, which exceeds the minimum requirements of 90% control survival. Temperature, salinity, and dissolved oxygen were within acceptable limits. The pH was 0.1 units below the targeted range (7.8 to 8.2) in some samples, but this did not affect amphipod survival. Unionized ammonia was very low in both overlying and pore water samples and were substantially lower than the threshold concentration for *E. estuarius* (1.15 mg/L). The ammonia reference toxicant test resulted in a normal dose response, and a Lethal Concentration to 50% of organisms (LC50) value of 2.3 mg/L similar to those reported in the literature for *E. estuarius* (Table 6-5)**,** suggesting normal sensitivity of the test batch.

Most of the data quality objectives were met for the amphipod exposures to Reference and SUBASE site sediments. The sediment holding time objective was met, with tests being initiated within 3 days of sample collection. Animals were acclimated over 2 days, also within acceptable limits. Amphipod survival in the control sediments was 90%, which meets the minimum requirements of 90% control survival. Proper exposure temperature was maintained at all times. Unionized ammonia ranged from 0.001 to 0.315 mg/L in the overlying water and <0.001 to 0.526 mg/L, well below the toxic threshold for *E. estuarius.* The response curve was normal for amphipods exposed to ammonia in the reference toxicant test, with point estimates (LC50=3.3 mg/L, unionized NH_3) consistent with literature values and the previous reference test associated with the Reconnaissance event (Table 6-5). Temperature, salinity, dissolved oxygen, pH, and ammonia measurements were made on all samples.

One data quality objective deviation was a slight exceedance (by 1 to 2 ‰) for salinity in a few of the test samples (SB2441, SB2, SB4, SB5, SB8, SB9) upon test termination. Salinity at the start of these tests was within the 18 to 22 ‰ requirement and *E. estuarius* is tolerant of much higher salinities than those observed (USEPA, 1994). In addition, insignificant toxicity was observed in the samples with the elevated salinity. Therefore, it is unlikely that the minor elevation in salinity compromised any samples.

The pH was within the targeted range of 7.8 to 8.2 for all but one test site (SB14). In that case, the pH was only slightly lower (7.68) at test initiation, and was within normal range (8.13) upon test termination. The pH of solutions from the reference tests ranged from 7.63 to 7.80, also minimally lower than the objectives, but appeared to be inconsequential to the results.

Bulk Sediment Test Outliers. Two stations (SB2 and SB13) had high variability in amphipod survival due to substantially lower survival in only one test replicate, while survival in remaining replicates was relatively high. The replicates with very poor survival appeared to be outliers that did not represent the toxicity at these stations. The cause of the aberrant survival in each outlier

replicate was not known, but may have been related to dying infauna in the sediments, resulting in poor water quality.

A threshold screening approach was used to identify and remove these outlier data. This approach was used in a recent investigation of toxicity in Chollas and Paleta Creek, San Diego, CA sediments (SCCWRP and Navy, 2005)**,** but will be briefly discussed as follows. Outliers were identified as those values which were $\geq$30 percentage points below the next highest value, working from highest to lowest values. For example, SUBASE Station SB2 had replicates with 30, 65, 80, 90, and 90% survival. The value of 30 was removed as an outlier because it was 30 percentage points below 65. The value of 65 was not removed, however, because it was less than 30 percentage points from 80, the next highest value.

The exclusion of outlier values has both advantages and disadvantages in this study. The primary advantage of excluding outliers is that variability in the data is reduced, with an associated increase in statistical power to detect differences from the control or baseline condition. In addition, exclusion of outliers should provide a more accurate measure of the toxicity of the sample. The disadvantages of using an outlier exclusion method include a possibility of erroneously identifying a replicate as an outlier and biasing the results by discarding accurate information. The small number of replicates that are tested complicates the detection of outlier values in a toxicity test. The decision to identify and exclude outliers in this study was based on two factors. First, the level of variability among test replicates was higher than normal for the amphipod toxicity test, indicating the potential presence of outliers. Second, the exclusion of outliers was judged to be appropriate because of the reliance on statistical comparisons to the control for classifying a sample as toxic. Reducing the excessive variability in survival for a test sample would likely improve the statistical power of the data analyses (comparison to control) and thus provide a more environmentally protective comparison.

6.2.2 Sediment-Water Interface

Most of the data quality objectives were met for the SWI experiments. The sediment holding time objective was met, with tests being initiated within 24 hours of sample collection. Mussel embryo development in the seawater control was acceptable (86.2% normal survival) and the copper reference toxicant test having an Effects Concentration to 50% of organisms (EC50) of 7.23 µg/L was within the control chart limits (Table 6-4). Salinity and dissolved oxygen were all maintained within the proper ranges. Temperature (range = 15.8-17.3 °C) did fall outside the 14 to16 °C target range for most samples tested. This test, however, may be conducted at 18 $\pm$ 1 °C (USEPA, 1995), and the higher temperatures observed are not expected to have presented any negative impact to the developing embryos or major alterations in the bioavailability of contaminants. Ammonia concentrations were all below the ammonia threshold for this species (No Observable Effects Concentration (NOEC) in concurrent ammonia test = 0.073 mg/L NH_3).

Sediment-Water Interface Test Outliers. One outlying data point was removed from results of each of two sites (SB6 and SB11). Both of these values were substantially below the other replicates corresponding to the site (Appendix B). Ammonia was below the effects threshold for all SWI exposures, but because ammonia was only measured in one replicate for each sample, it cannot be determined with certainty that these two particular cores did not have an abnormally high degree of decaying organic matter, and therefore, higher ammonia concentrations, compared to the other cores.

6.2.3 Pore Water

Most of the data quality objectives were met for the experiments with SUBASE site porewater samples, but some objectives were not met, raising some issues. The sediment holding time objective was achieved, with exposures being initiated 9 days following sample collection. Seawater control performance (75% normal survival for *S. purpuratus*) passed the 70% minimum requirement. The response in the reference toxicant experiment was within the control chart limits (Table 6-4) for *S. purpuratus* with an EC50=20.53 µg/L. Salinity and dissolved oxygen were maintained within acceptable ranges for all porewater samples. Temperature (range = 15.8 to 17.7 °C) did fall slightly outside the target range of 14-16 °C for all samples.

Although unionized ammonia concentration was acceptable at test initiation, by the end of the exposure, all undiluted porewater samples exceeded the ammonia threshold of 0.033 mg/L (Table 6-5). Adjustment of the embryo development data in undiluted porewater for the influence of ammonia was used in this study, following an approach used in a previous study of pore water toxicity (Bay, 1995). Adjustment of the data for ammonia toxicity is desirable in this study because it reduces the impact of a confounding factor not associated with chemical contamination. It is possible that chemical contamination may have impacted embryo development in some of the samples that were excluded as an outlier due to high ammonia. The presence or absence of such effects cannot be determined, due to the high level of toxicity caused by ammonia. Ammonia concentrations >0.067 mg/L NH_3 were believed to be responsible for all of the toxicity in samples that had <80% normal development (see SCCWRP and Navy, 2005 for a complete description of the process used to identify outliers and adjust for ammonia influence). No information regarding the toxicity of other constituents could be obtained in these undiluted samples (SB9, SB10, SB14), so toxicity for these samples was expressed as 'ND' (not determined).

The amount of ammonia influence could be corrected for in other undiluted porewater samples to enable the evaluation of the amount of toxicity due to other constituents. Samples with ammonia concentrations between 0.033 and 0.067 mg/L NH_3 that had sea urchin embryo development <80% of the control were adjusted for ammonia influence (SCCWRP and Navy, 2005). Except for the 3 samples that could not be corrected (SB9, SB10, SB14), all reference and SUBASE stations were corrected for ammonia influence using this process.

Violation of the data quality objectives were also observed for pH. Except for the seawater control, initial pH (range = 6.9-7.7) values were below the targeted 7.8-8.2 in all undiluted porewater samples. Previous to the initiation of the tests, porewater had been stored with no headspace. At test termination, pH values were within the acceptable range, presumably due to degassing of CO_2 upon exposure to the surrounding air in the exposure vials. Although it is likely that pH rose to acceptable levels rather quickly, this cannot be confirmed due to the lack of pH measurements available shortly after test initiation. Bay et al. (2003) reported developmental abnormalities for *S. purpuratus* embryos at pH less than 7.4. After correction for ammonia influence, however, several samples with the reduced pH did result in normal larval development comparable to the negative control, which had an acceptable pH at test initiation. The pH of the 25% porewater dilutions were not measured, however, because they were diluted with negative control sea water (pH 8.10 at test initiation), pH was likely within acceptable range in the diluted samples. Therefore, it was deemed useful to consider the 25% dilutions in the interpretation of the pore water data due to the uncertainty associated with pH effects in the undiluted samples. Use of the 25% dilution data is also worth consideration because no sample

exceeded the ammonia effects threshold (0.033 mg/L unionized NH3) for the sea urchin at that dilution.

Table 6-4. Summary of toxicity test data quality objectives. * = Comparisons would normally be made to the control chart mean, however only a limited number of reference toxicant tests have been performed at SCCWRP using ammonia with *E. estuarius*.

	Bulk Sediment	Sediment-Water Interface	Pore water
Parameter	**Amphipod Survival**	**Mussel Larval Development**	**Sea Urchin Larval Development**
Sediment holding time	<2 weeks	<2 weeks	<2 weeks
Animal acclimation period	2-7 days	No objective	No objective
Control response	≥ 90% survival	≥ 70% normal survival	≥70% normal survival
Reference toxicant test	Normal NH_3 response curve*	Cu EC50 within 2 SD of control chart mean (7.3 ± 4.2)	Cu EC50 within 2 SD of control chart mean (17.6 ± 7.8)
Water quality parameters:			
Temperature	15°C ± 2°	15°C ± 2°	15°C ± 2°
Salinity	18-22‰	32-35‰	32-35‰
Unionized Ammonia	<1.15 mg/L	<0.073 mg/L	<0.033 mg/L
Dissolved Oxygen	>5 mg/L	>5 mg/L	>5 mg/L
pH	7.8-8.2	7.8-8.2	7.8-8.2

Table 6-5. Results from 96-h ammonia reference toxicity tests with *E. estuarius* from this study in comparison with published values. LC50 = median effective concentration. C.I. = confidence interval.

Sampling Date/Study	LC50 (mg/L)	95% C.I.
February 2004	2.3	2.0-2.7
April 2004	3.3	2.7-3.6
Kohn et al. (1994)	2.5	2.3-3.4

6.3 BENTHIC COMMUNITY ANALYSIS

The infauna in each sample were sorted into major phylogenic groups using dissecting microscopes. Ten percent of each sample was QC'd by the lab supervisor, who ordered a resort of the sample if a 95% sorting efficiency was not achieved. Identifications were conducted by taxonomists who are active in the Southern California Association of Marine Invertebrate Taxonomists (SCAMIT). Any naming discrepancies or difficult organisms were reviewed with other SCAMIT taxonomists.

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7.0 SEDIMENT CHEMISTRY RESULTS

7.1 PHYSICAL CHARACTERISTICS

Physical characteristics including water depth, TOC, and fines were characterized at all reference and study stations. These parameters are important factors in characterizing the type of benthic habitat present at the sites. TOC and grain size also are important in regulating the binding of organic and inorganic contaminants within the sediment. Metal variation with grain size also can be useful in establishing non-anthropogenic background. Results for physical properties at the reference and SUBASE stations are summarized below. Reference stations include data which were collected during two separate sampling events. The earlier of the two being a reconnaissance study, having station ID names designated with the suffix –R.

7.1.1 Reference Stations

Physical properties results including water depth, fines, and TOC for the reference stations are shown in Table 7-1 and Table 7-2. The tables include data from both the reconnaissance and main study surveys. The complete grain size fractionation data are included in Appendix A. Water depths at the reference stations ranged from 6.1 to 14.3 m with the shallowest water depth at SB90056 and the deepest at station SB2441 (Figure 7-1). This range of depths is characteristic of the two dominant habitat types in San Diego Bay including shallow sub-tidal areas, and deep shipping channels. The fines fraction for the reference stations ranged from 13 to 83% with the lowest fines at SB90056 and the highest at SB2441 during both sampling periods. Data collected from the two time periods showed some field variability with relative percent differences ranging from 70 to 137%. The fines data also varied from data collected during Bight98, having an average RSD of 29% when compared on a station by station basis.

The TOC fraction at reference stations ranged from 0.24 to 2.5% with the lowest TOC at SB90056 and the highest at SB2441 (Table 7-1). These two stations were lowest and highest, respectively, in TOC during both the reconnaissance and main study surveys. Data collected from the two time periods showed some field variability with relative percent differences ranging from 58 to 148%. The fines data also varied from data collected during Bight98, having an average RSD of 44% when compared on a station by station basis.

TOC at the reference stations generally increased with increasing fines and bracket nearly all the SUBASE station data. Station SB11 had a slightly lower fines and SB2 had a slightly higher fines than any of the reference stations (Figure 7-2). The trend also shows reference stations were slightly depleted in TOC relative to fines when compared to SUBASE stations. The depletion was slightly greater at lower fines content with a difference of about 30% lower at a fines content of 20%.

7.1.2 SUBASE Stations

Physical properties results for the SUBASE stations are shown in Table 7-1 and Table 7-2. The complete grain size fractionation data are included in Appendix A. Water depths at the SUBASE stations ranged from 6.1 to 15.8 m with the shallowest water depth at SB12, and the deepest at SB13. Water depths at SUBASE stations generally were similar to those found at the Reference sites, with one exception, the deepest SUBASE site (SB13) (Table 7-1 and Figure 7-1). The fines fraction for the SUBASE stations ranged from 7.1 to 86.6%. The range of fines at the SUBASE sites is consistent with those at the reference stations, with one

exception falling lower (SB11) than the range of fines at the reference stations, and one higher (SB2). The TOC fraction at the SUBASE stations ranged from 0.3 to 2.1% with the lowest TOC at SB11 and the highest at SB2. The range of TOC at the SUBASE stations was comparable to the reference range. TOC at the SUBASE stations generally increased with increasing fines, following a similar trend to those observed at the reference stations and during Bight98 (Figure 7-2).

The spatial distributions of TOC and fines for SUBASE are shown in Figure 7-3 and Figure 7-4 respectively. Generally, the highest TOC and fines were found closest to the shoreline or quaywall, and decreased moving out into the bay. Minor deviations from this pattern were observed for both fractions. TOC values tended to be higher at the western most stations SB8 to 14. There also appears to be a mid-pier maximum with respect to fines, which may be due to particulate resuspension associated with boat traffic.

Table 7-1. Sediment physical data for reference and SUBASE stations. Stations names with and –R suffix indicate samples were taken during a reconnaissance survey of reference stations.

Area	Station	Depth (m)	Fines (%)	TOC (%)
Reference	SB2229	12.2	15.0	0.25
	SB2433	8.2	33.3	0.47
	SB2436	11.0	37.3	0.63
	SB2441	14.3	83.8	2.45
	SB90056	6.1	13.2	0.24
	SBC001SS31	8.2	59.5	0.91
Recon	SB2229-R	12.2	21.4	0.43
	SB2433-R	8.2	38.6	0.57
	SB2436-R	11.0	39.7	0.55
	SB2441-R	14.3	61.2	1.65
	SB90056-R	6.1	18.4	0.25
	SBC001SS31-R	8.2	52.8	0.76
SUBASE	SB1	13.4	45.2	1.13
	SB2	12.8	86.6	2.14
	SB3	11.3	22.8	0.67
	SB4	10.4	76.4	1.73
	SB5	11.0	19.8	0.62
	SB6	10.7	45.2	1.33
	SB7	11.3	30.9	0.84
	SB8	10.1	61.5	1.84
	SB9	10.4	71.5	1.98
	SB10	11.3	42.9	1.37
	SB11	11.6	7.1	0.25
	SB12	6.1	66.6	1.56
	SB13	15.8	78.5	2.12
	SB14	12.2	46.6	1.44

Table 7-2. Summary statistics for sediment physical data.

Sum Stats	Reference	Recon	SUBASE
Depth (m)			
Minimum	6.1	6.1	6.1
Maximum	14.3	14.3	15.8
Mean	10.0	10.0	11.3
Std Dev	3.0	3.0	2.1
Fines (%)			
Minimum	13.2	18.4	7.1
Maximum	83.8	61.2	86.6
Mean	40.3	38.7	50.1
Std Dev	27.2	16.8	24.3
TOC (%)			
Minimum	0.2	0.3	0.3
Maximum	2.5	1.7	2.1
Mean	0.8	0.7	1.4
Std Dev	0.8	0.5	0.6

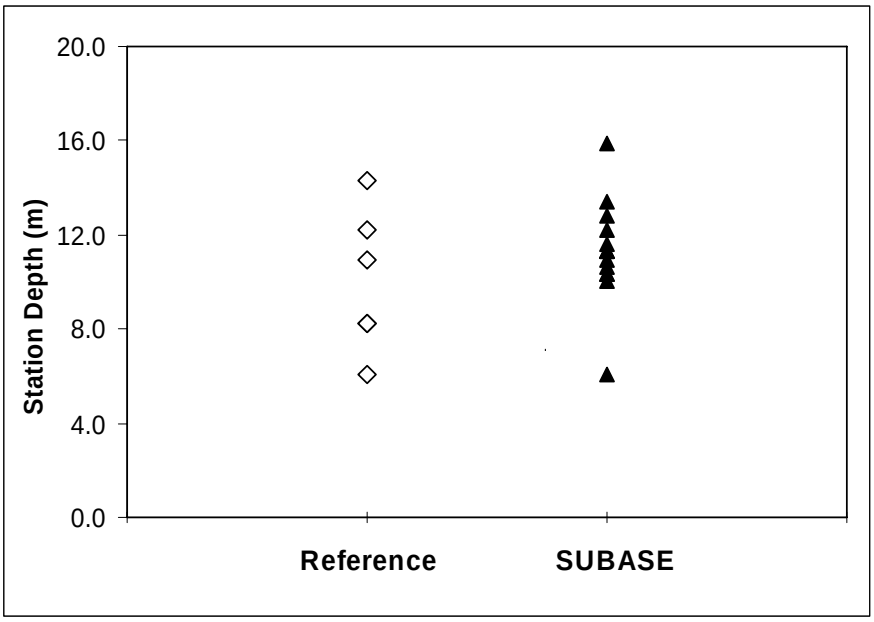

Figure 7-1. Water depths of reference and SUBASE stations.

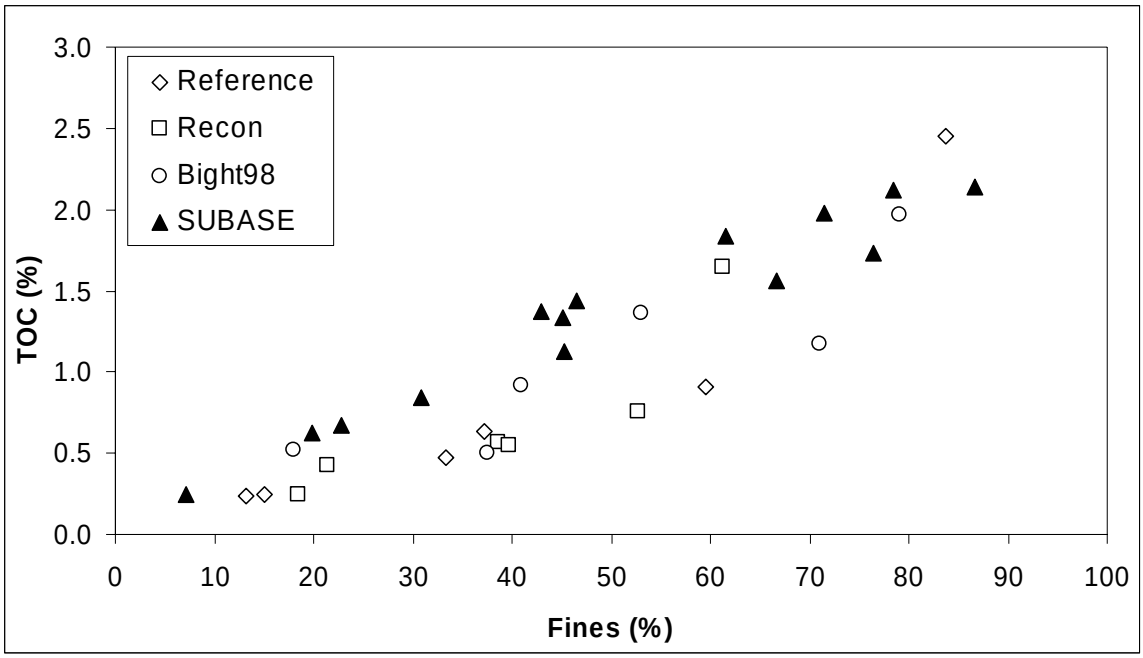

Figure 7-2. Plot of TOC and fines at all stations of the study, as well as Bight98.

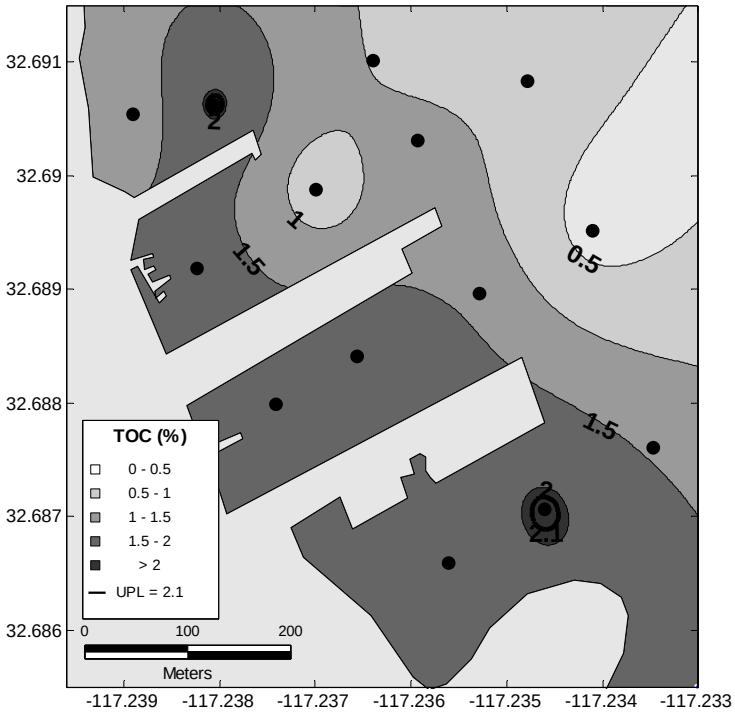

Figure 7-3. Spatial distrbution of TOC at the SUBASE stations.

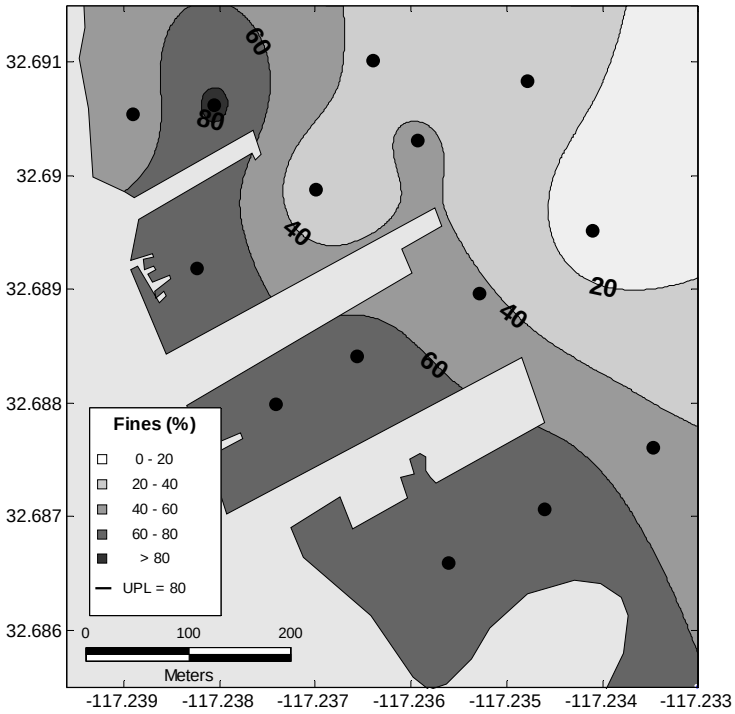

Figure 7-4. Spatial distribution of fines at the SUBASE stations.

7.2 METALS

Concentrations of total sediment metals including silver, arsenic, cadmium, chromium, copper, lead, mercury, nickel and zinc were characterized at all reference and study stations. Total metal concentrations include the influence of both anthropogenic and background (crustal) sources, and provide one indicator of potential contaminant exposure for aquatic organisms. Results for sediment metals at reference and SUBASE stations are summarized below. The data displayed (in mg/kg dry weight) include only those metals that were identified as CoPCs in the historical review. The complete set of data can be found in Appendix A.

7.2.1 Reference Stations

Metals results for the reference stations are shown in Table 7-3 through Table 7-5. Metal concentrations at the reference stations generally were low, and showed minimal variation from station to station (spatial), and also between similar stations during the two different sampling periods (temporal). For example, arsenic ranged from 6.69 to 10.7 mg/kg with a relative standard deviation (RSD) of only 18% during the recon sampling, and 6.26 to 11.6 mg/kg with an RSD of 25% for main survey (Table 7-4).

During the main study survey, station SB2441 had the highest occurrence of maximum metal concentrations (arsenic, cadmium, chromium, copper, and nickel), while during the recon survey highest occurrence of maximum metal concentrations was spread among several sites; SB2436-R (silver, chromium, and zinc), SB2441-R (arsenic, cadmium, and nickel). Stations SB90056(-R) by far had the highest occurrence of minimum metal concentrations during both the main (8 of 9 metals) and recon (7 of 9 metals) surveys. None of the metal concentrations measured at the reference stations exceeded their respective ERM value.

7.2.2 SUBASE Stations

Metals results for the SUBASE stations are shown in Table 7-3 and Table 7-4. Metals concentrations at the SUBASE stations generally were within range of concentrations at the reference stations. All metals values from both the SUBASE and reference stations were within 45% of each other. Mercury and silver had the greatest percent difference of mean metal concentrations between SUBASE and reference stations, with 45% and 37% respectively. Excluding mercury and silver, the mean metals concentrations were between 1% and 21% of each other comparing the different study areas. Mean SUBASE metals concentrations were similar or lower than mean reference stations, e.g. mercury, silver, arsenic, and zinc. Only copper showed higher concentrations at SUBASE stations, with mean copper concentrations approximately 10 ppb higher than those found at the reference stations. None of the metal concentrations measured at the SUBASE stations exceeded their respective ERM value.

Among the SUBASE stations, SB12 had the highest occurrence of maximum metal concentrations including silver, cadmium, and lead. Two other stations had occurrences of maximum metal concentrations, SB2 (chromium and nickel) and SB4 (copper and zinc). Station SB11 had, by far, the highest occurrence of minimum metals concentrations, with lowest values for all nine metals among the SUBASE stations. In addition, SB3, SB5, and SB7 also consistently had lower metals concentrations relative to the other SUBASE stations. Compared to metal SQGs, all metal concentrations were below their respective ERM.

All nine metals had statistically significant ($p < 0.01$) positive correlations with both TOC and fines, but not depth (Table 7-5). Nickel had the strongest relationship and cadmium had the weakest, although all metals had relatively high correlation coefficients. In addition, all metals were significantly positively correlated ($r > 0.7$) with each other.

The spatial distribution for some representative metals at the SUBASE site is shown in Figure 7-5 through Figure 7-8. Spatial patterns of most metals appeared to be highly influenced by the distribution of fines and TOC. Common characteristics of these distributions include the highest values being found at stations closest to the shoreline, decreasing at stations further into the bay. In addition, metals concentrations tended to be higher at the western most (SB12-14) stations relative to the eastern most (SB1 to SB3) at similar distances from shore. All metals had statistically significant highly positive correlations with both fines, $r > 0.86$, and TOC, $r > 0.77$ (Table 7-5).

Table 7-3. Sediment metals data (mg/kg) for reference, and SUBASE stations. Values highlighted in the table exceeded their respective ERM value.

Area	Station	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Recon	SB2229-R	0.510	10.1	0.262	46.0	48.9	0.251	9.02	38.5	122
	SB2433-R	0.590	7.17	0.379	45.2	49.2	0.313	10.6	25.2	125
	SB2436-R	0.716	9.51	0.364	56.0	64.9	0.387	12.3	36.3	148
	SB2441-R	0.564	10.7	0.415	48.2	65.5	0.216	14.4	24.3	133
	SB90056-R	0.402	6.69	0.280	36.2	30.5	0.504	7.53	18.2	85.5
	SBC001SS31-R	0.675	8.56	0.379	52.6	71.2	0.269	12.9	28.5	148
Reference	SB2229	0.307	6.52	0.215	29.9	35.9	0.209	6.88	24.3	105
	SB2433	0.460	6.67	0.258	41.8	45.6	0.245	10.2	24.8	132
	SB2436	0.616	8.27	0.251	46.5	64.2	0.381	11.4	33.8	151
	SB2441	0.475	11.6	0.371	56.0	93.4	0.278	18.3	31.7	174
	SB90056	0.250	6.26	0.237	28.5	20.4	0.128	5.99	18.1	78.7
	SBC001SS31	0.626	8.54	0.285	51.8	81.5	0.511	14.2	30.6	175
SUBASE	SB1	0.362	8.17	0.285	37.4	57.3	0.172	10.8	22.0	124
	SB2	0.520	10.6	0.446	57.6	94.2	0.264	17.5	30.6	172
	SB3	0.254	4.04	0.207	25.7	26.3	0.099	6.92	15.4	71.4
	SB4	0.551	9.27	0.538	56.2	112.2	0.293	16.4	30.4	184
	SB5	0.212	6.44	0.205	27.5	39.1	0.125	7.66	16.0	76.2
	SB6	0.432	9.24	0.260	38.9	53.2	0.206	11.8	20.4	113
	SB7	0.264	8.44	0.253	30.2	36.6	0.161	9.20	16.8	89.6
	SB8	0.510	10.4	0.413	51.6	96.2	0.310	16.5	27.1	165
	SB9	0.514	10.3	0.393	54.3	96.0	0.283	16.3	28.8	166
	SB10	0.378	8.01	0.238	48.3	69.9	0.177	11.2	22.5	117
	SB11	0.130	4.01	0.075	12.6	11.6	0.0754	3.40	13.0	37.5
	SB12	0.561	9.55	0.577	52.8	96.9	0.291	15.6	31.1	181
	SB13	0.508	11.5	0.352	51.0	85.7	0.237	16.1	25.7	158
	SB14	0.329	8.74	0.300	39.2	53.2	0.165	12.3	22.2	123
SQG	ERM	3.70	70.0	9.60	370	270	0.710	51.6	218	410

Table 7-4. Summary statistics for sediment metals data (mg/kg).

Area	Statistic	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Recon	Minimum	0.402	6.69	0.262	36.2	30.5	0.216	7.53	18.2	85.5
	Maximum	0.716	10.7	0.415	56.0	71.2	0.504	14.4	38.5	148
	Mean	0.576	8.79	0.347	47.4	55.0	0.323	11.1	28.5	127
	Std Dev	0.113	1.61	0.0611	6.84	15.1	0.107	2.56	7.69	23.1
	RSD (%)	20%	18%	18%	14%	27%	33%	23%	27%	18%
Reference	Minimum	0.250	6.26	0.215	28.5	20.4	0.128	5.99	18.1	78.7
	Maximum	0.626	11.6	0.371	56.0	93.4	0.511	18.3	33.8	175
	Mean	0.456	7.98	0.270	42.4	56.8	0.292	11.2	27.2	136
	Std Dev	0.155	2.02	0.0549	11.3	27.9	0.136	4.62	5.87	38.5
	RSD (%)	34%	25%	20%	27%	49%	47%	41%	22%	28%
SUBASE	Minimum	0.130	4.01	0.075	12.6	11.6	0.0754	3.40	13.0	37.5
	Maximum	0.561	11.5	0.577	57.6	112	0.310	17.5	31.1	184
	Mean	0.395	8.48	0.324	41.7	66.3	0.204	12.3	23.0	127
	Std Dev	0.140	2.27	0.138	13.7	31.2	0.0769	4.35	6.13	46.0
	RSD (%)	36%	27%	42%	33%	47%	38%	35%	27%	36%

Table 7-5. Correlation matrix for the SUBASE physical properties and metals. Values are the correlation coefficient. Grayed out values are statistically significant at $p < 0.01$.

	Depth	Fines	TOC	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Depth	1.00	0.04	0.10	-0.19	0.08	-0.38	-0.11	-0.19	-0.29	-0.07	-0.22	-0.17
Fines		1.00	0.97	0.95	0.89	0.86	0.95	0.94	0.90	0.98	0.95	0.96
TOC			1.00	0.93	0.92	0.77	0.95	0.91	0.88	0.97	0.91	0.93
Ag				1.00	0.86	0.91	0.96	0.96	0.96	0.96	0.96	0.97
As					1.00	0.73	0.87	0.83	0.86	0.92	0.82	0.87
Cd						1.00	0.87	0.92	0.91	0.89	0.94	0.94
Cr							1.00	0.97	0.92	0.97	0.96	0.97
Cu								1.00	0.96	0.96	0.98	0.98
Hg									1.00	0.95	0.94	0.96
Ni										1.00	0.96	0.98
Pb											1.00	0.99
Zn												1.00

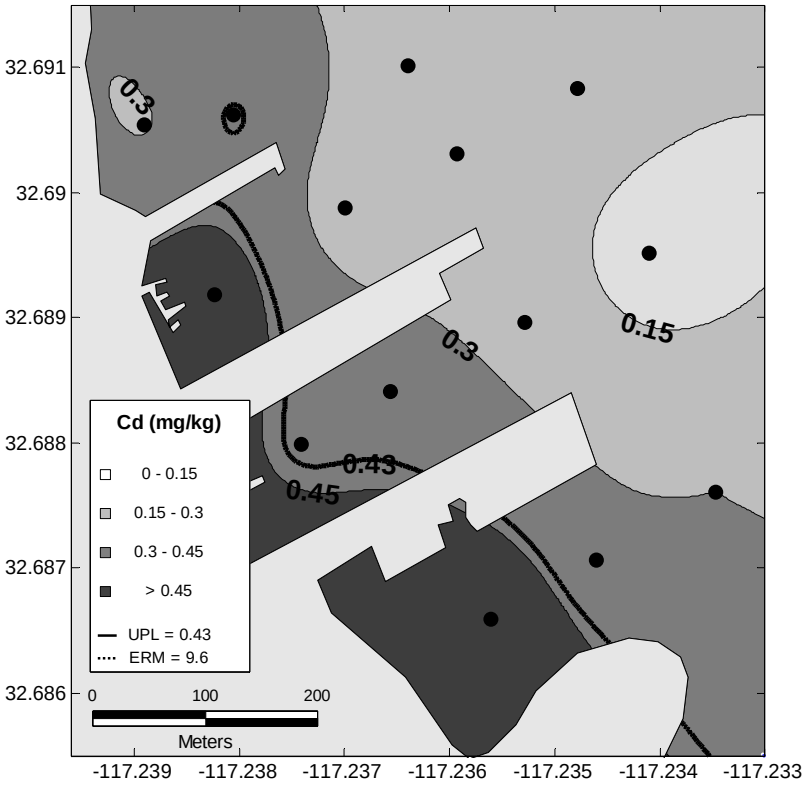

Figure 7-5. Spatial distribution of cadmium at the SUBASE stations. Upper Protective Limit (UPL) and Effects Range Median (ERM) are contoured in a bold solid and dashed line, respectively, if exceedances were observed.

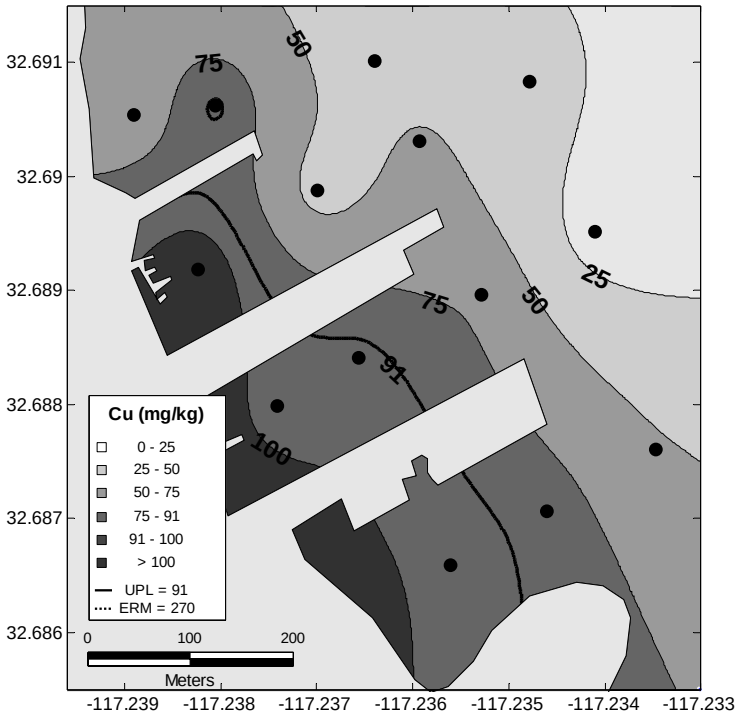

Figure 7-6. Spatial distribution of copper at the SUBASE stations. Upper Protective Limit (UPL) and Effects Range Median (ERM) are contoured in a bold solid and dashed line, respectively, if exceedances were observed.

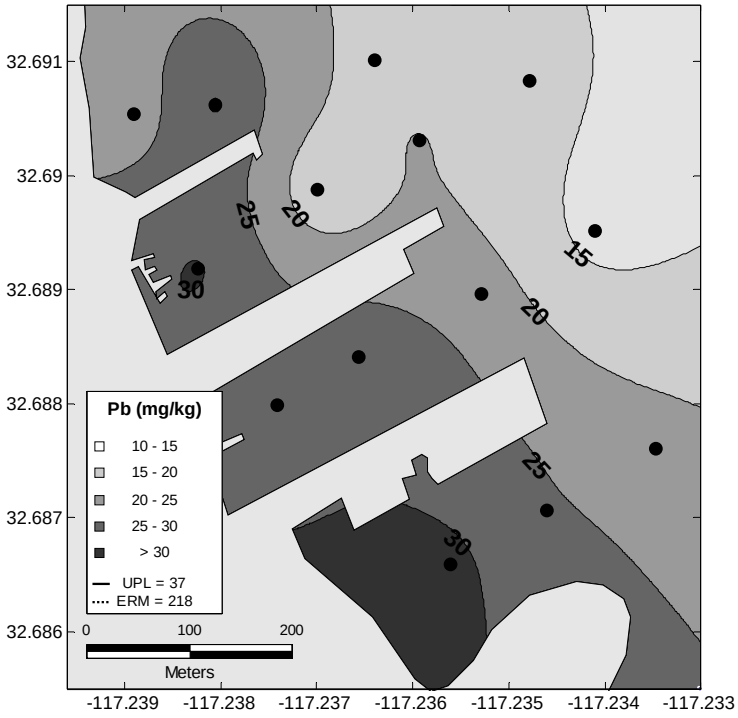

Figure 7-7. Spatial distribution of lead at the SUBASE stations. The Upper Protective Limit (UPL) and Effects Range Median (ERM) are contoured in a bold solid and dashed line, respectively, if exceedances were observed.

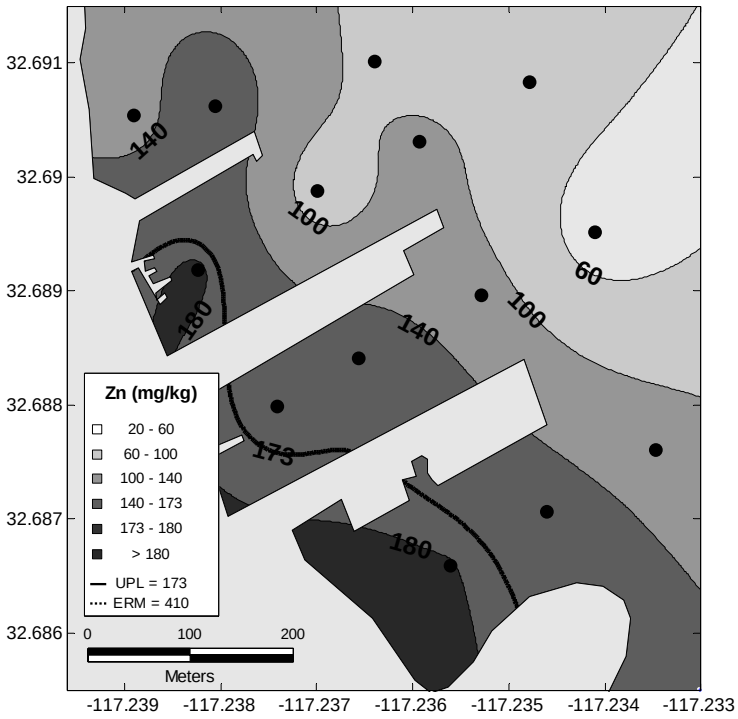

Figure 7-8. Spatial distribution of zinc at the SUBASE stations. Upper Protective Limit (UPL) and Effects Range Median (ERM) are contoured in a bold solid and dashed line, respectively, if exceedances were observed.

7.3 PAHs

The concentration of 47 individual PAH analytes was measured at all reference and study stations. The analytes measured include the 16 PAHs on the EPA's priority pollutant list: naphthalene^{L}, acenaphthylene^{L}, acenaphthene^{L}, fluorene^{L}, anthracene^{L}, phenanthrene^{L}, fluoranthene^{H}, pyrene^{H}, benz(a)anthracene^{H}, chrysene^{H}, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene^{H}, indeno(123-cd)pyrene, dibenz(ah)anthracene^{H}, benzo(ghi)perylene. The additional 31 PAH analytes were measured because they can, in some instances, be used to differentiate hydrocarbon sources. The first six compounds along with 2-methyl naphthalene are commonly grouped together and categorized as low molecular weight PAH (designated by ^{L} above) while six of the remaining ten analytes are commonly grouped together and categorized as high molecular weight hydrocarbons (designated by ^{H} above). The LMWPAH commonly degrade relatively quickly and have a higher acute toxicity while the HMWPAH are typically recalcitrant and have a higher carcinogenicity. Results for the PPPAH, LMWPAH, and HMWPAH are summarized below in Table 7-6 and Table 7-7. All PAH data are provided in Appendix A. Results are reported in μg/kg dry weight.

7.3.1 Reference Stations

Sediment PPPAH concentrations ranged from about 300 to 2400 μg/kg and averaged 1000 μg/kg. The LMWPAH make up only about 5% of the total PAHs at these stations with the HMWPAH making up roughly 45% of the total. PAH concentrations generally increased with TOC, with one exception; station SB2229 had the highest PAH concentrations with relatively low TOC content. The range in concentrations at the six stations results in station-to-station variability of approximately 60%, as measured by a RSD. The LMWPAH had a slightly higher variability, which is consistent with its more reactive nature. There were no exceedances of the consensus-based organic carbon normalized SQG (CBSQG) value of 1800 μg/g OC.

7.3.2 SUBASE Stations

Sediment PPPAH data for the SUBASE stations ranged from 250 to 5250 μg/kg and averaged 1500 μg/kg. The mean PAH concentration for the SUBASE stations exceeded that of the reference stations by approximately 40%. Eleven of fourteen stations had PAH concentrations within range of the reference station levels. The three stations (SB4, SB8, and SB12) with concentrations exceeding those of the reference did so by at most a factor of two. The three stations with elevated PAH concentrations were found closest to the shoreline, and most likely contributed to the overall higher station-to-station variability (RSD = 93%), relative to the reference. Similar to the reference stations, the LMWPAH were typically about 7% of the total PAHs and the HMWPAHs were about 45%. None of the SUBASE stations CB-PAH values came close to exceeding the CBSQG value of 1800 μg/g OC.

The mean relative distribution of PAH analytes (individual PAH/total PAH) in the samples was relatively similar to that of the reference stations. The distribution fingerprint (Figure 7-9) does not provide a clear indication of the PAH source, however the overall profile is similar to that of weathered creosote.

The general level and distribution of PAHs at the SUBASE stations correlate reasonably well with fines and TOC (Table 7-8). The overall correlation of PAH concentrations to these physical sediment variables is significant with fines ($p = 0.015$), and just non-significant with TOC ($p = 0.051$). Deviations from this relationship are found at three stations (SB4, SB8, and SB12),

which have data that fall well off this trend (Figure 7-10). These stations represent three of the four most shoreward sampling locations at the SUBASE site, and show much greater PAH concentrations than would be predicted from the TOC. Spatially, PAH levels are highest closest to the shoreline, decreasing moving out into the bay, with highest concentrations at the western most station SB12 (Figure 7-11). The elevated PAHs relative to TOC at these three stations suggest an additional, yet unknown source(s) of PAH, possibly terrigenous in nature due to proximity with the shoreline.

Table 7-6. Sediment organics data for reference and SUBASE stations. Data are included for PAHs, PCBs, Chlordanes and DDTs. Also included are the calculated values for each station for comparison to CBSQGs. Values highlighted in the table exceeded their respective SQG value.

Area	Station	LMWPAH µg/kg	HMWPAH µg/kg	PPPAH µg/kg	CB-PAH µg/g OC	Total PCB µg/kg	CB-PCB µg/kg	TCHLOR µg/kg	TDDT µg/kg	CB-DDT µg/g OC
Reference	SB2229	107	1759	2418	746	9.40	7.96	0.855	1.10	0.440
	SB2433	33.8	334	453	78.1	9.63	8.54	0.285	0.960	0.204
	SB2436	76.2	723	1003	127	19.9	17.5	0.640	1.75	0.277
	SB2441	261	1262	1681	62.2	11.5	9.67	1.33	2.19	0.0892
	SB90056	24.6	240	311	110	4.07	3.59	0.165	0.490	0.204
	SBC001SS31	75.4	737	983	89.2	17.7	14.7	0.800	1.94	0.213
Recon	SB2229-R	88.5	642	887	170	13.0	11.2	1.22	1.07	0.248
	SB2433-R	52.6	575	756	110	10.1	9.04	0.850	1.44	0.253
	SB2436-R	77.5	865	1210	171	17.9	15.6	1.38	1.81	0.329
	SB2441-R	137	896	1170	62.6	8.73	7.74	1.16	1.65	0.100
	SB90056-R	30.4	310	403	136	5.63	5.05	0.505	0.715	0.286
	SBC001SS31-R	62.5	595	802	86.5	14.6	12.8	1.26	1.92	0.253
SUBASE	SB1	152	974	1299	100	17.9	15.3	1.46	1.53	0.135
	SB2	180	1140	1494	61.7	18.2	16.8	0.955	1.63	0.0759
	SB3	56.6	326	443	57.1	3.58	3.05	0.275	0.600	0.0896
	SB4	372	2555	3369	169	21.0	17.2	0.98	2.78	0.161
	SB5	71.0	409	553	77.4	5.79	5.03	0.2	0.795	0.128
	SB6	77.2	497	664	43.2	4.22	3.54	0.27	1.58	0.118
	SB7	30.6	252	331	33.6	4.04	3.46	0.215	0.710	0.0845
	SB8	283	2068	2692	128	20.6	17.1	0.875	2.28	0.124
	SB9	214	1472	1917	85.1	20.4	17.4	0.715	2.07	0.105
	SB10	94.8	532	734	45.7	7.56	6.71	0.390	4.22	0.308
	SB11	51.1	198	269	100	1.47	1.21	0.155	0.305	0.122
	SB12	562	3960	5265	290	24.6	20.3	1.39	3.28	0.210
	SB13	183	1268	1670	68.5	11.7	9.92	0.375	1.56	0.0736
	SB14	84.7	519	682	41.9	6.75	5.85	0.410	1.07	0.0743
SQG					1800		400	4.8		100

Table 7-7. Summary statistics for sediment organics data including PAHs, PCBs, Chlordanes, and DDTs (μg/kg).

Area	Statistic	LMWPAH	HMWPAH	PPPAH	Total PCB	TCHLOR	TDDT
Recon	Minimum	30.4	310	403	5.63	0.505	0.715
	Maximum	137	896	1210	17.9	1.38	2.03
	Mean	74.7	647	871	11.7	1.06	1.45
	Std Dev	36.5	215	297	4.39	0.325	0.511
	RSD (%)	49%	33%	34%	38%	31%	35%
Reference	Minimum	24.6	240	311	4.07	0.165	0.490
	Maximum	261	1759	2418	19.9	1.33	2.19
	Mean	96.3	842	1142	12.0	0.678	1.40
	Std Dev	86.3	577	791	5.82	0.4	0.653
	RSD (%)	90%	68%	69%	48%	62%	47%
SUBASE	Minimum	30.6	198	269	1.47	0.155	0.305
	Maximum	562	3960	5265	24.6	1.46	4.22
	Mean	172	1155	1527	12.0	0.62	1.74
	Std Dev	149	1074	1419	8.08	0.445	1.10
	RSD (%)	86%	93%	93%	67%	72%	63%

Table 7-8. Correlation matrix for SUBASE site physical properties and organic contaminants. Values highlighted in gray represent a significant relationship at the 0.05 level.

	Depth	Fines	TOC	LMWPAH	HMWPAH	PPPAH	TPCB	TCHLOR	TDDT
Depth	1.000	0.039	0.097	-0.583	-0.587	-0.590	-0.326	-0.290	-0.393
Fines		1.000	0.970	0.639	0.640	0.635	0.792	0.586	0.528
TOC			1.000	0.531	0.537	0.531	0.714	0.472	0.537
LMWPAH				1.000	0.998	0.999	0.863	0.760	0.614
HMWPAH					1.000	1.000	0.862	0.750	0.605
PPPAH						1.000	0.859	0.750	0.608
TPCB							1.000	0.889	0.576
TCHLOR								1.000	0.479
TDDT									1.000

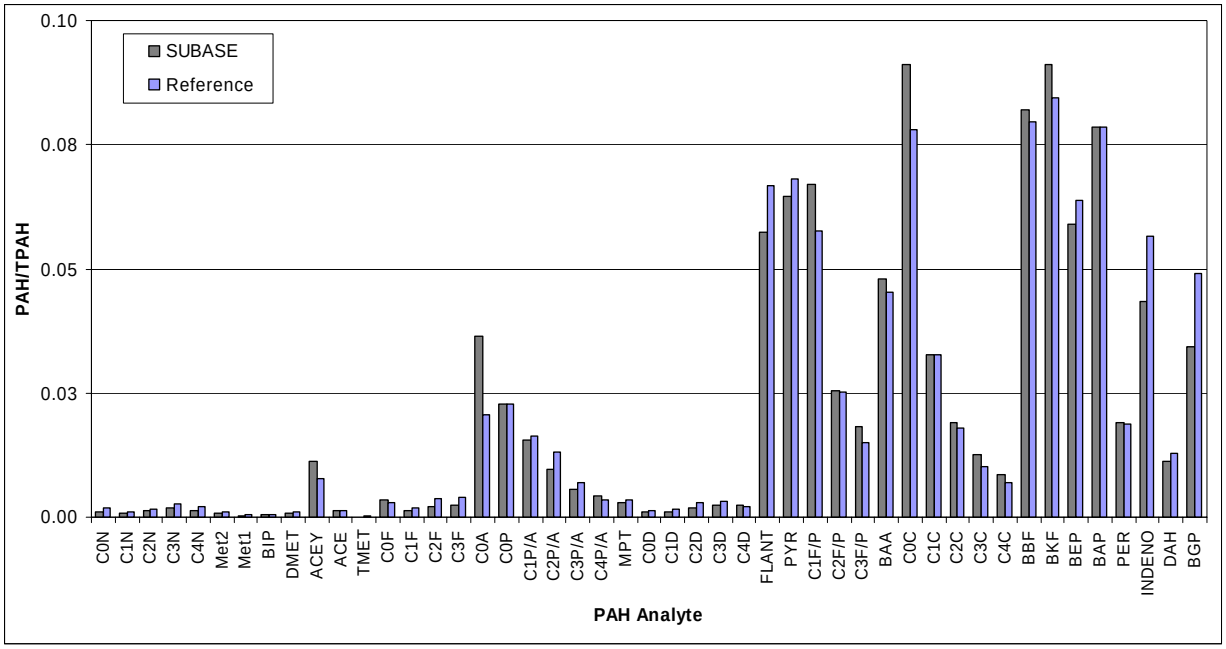

Figure 7-9. Mean relative PAH distribution for SUBASE and reference stations. Analyte identifiers are described in Table 5-6.

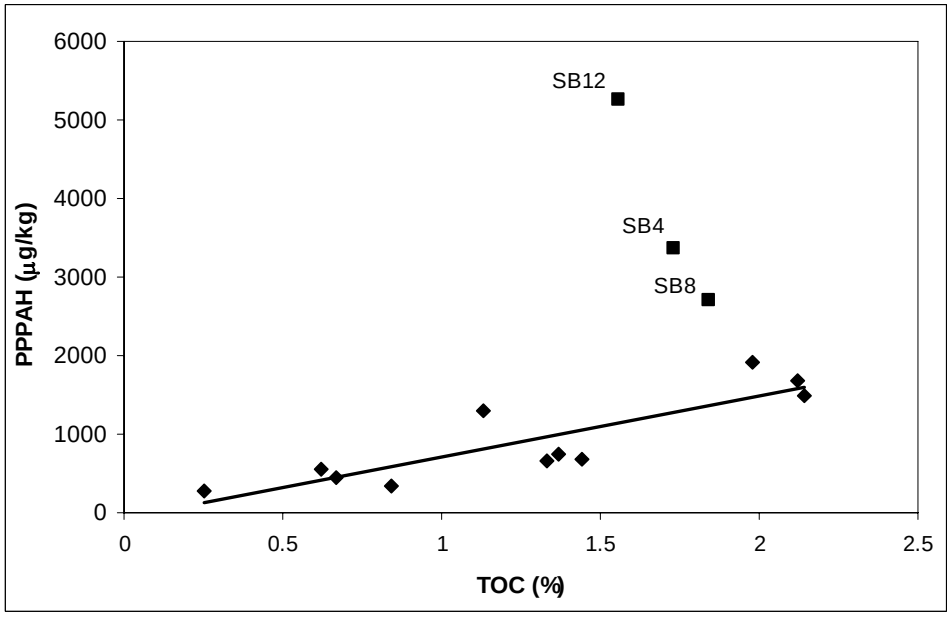

Figure 7-10. PPPAH as a function of TOC for SUBASE stations.

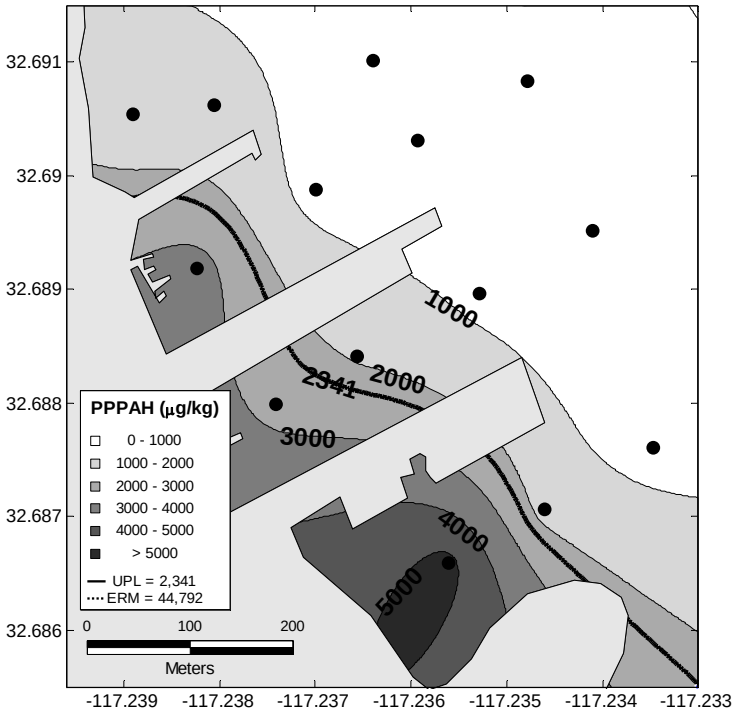

Figure 7-11. Spatial distribution of PPPAH at the SUBASE stations.

7.4 PCBs

Concentrations of PCBs were characterized on the basis of 31 individual congeners at all reference and study stations. Total PCB concentrations were determined as the sum of all individual congeners. PCB concentrations in sediment provide one indicator of potential contaminant exposure for aquatic organisms. Results for sediment PCBs at reference and SUBASE stations are summarized below. The data displayed include only the results for total PCBs, however the complete set of data for all individual congeners can be found in Appendix A.

7.4.1 Reference Stations

PCB results for the reference stations are shown in Table 7-6 and Table 7-7. PCB concentrations at the reference stations generally were low, and showed minimal variation from station to station. Total PCBs at the reference stations ranged from 4 to 20 µg/kg. The mean total PCB concentration for the reference stations was 12 µg/kg with an RSD of 42%. No comparative ranges for PCBs were established for reference stations in the SAP, however, the range of PCBs at the reference stations in this study was comparable to, and slightly lower than, the range reported at BPTCP reference stations (23-188 µg/kg; mean 72 µg/kg). None of the reference stations exceeded the CBSQG value of 400 µg/kg.

7.4.2 SUBASE Stations

PCB results for the SUBASE stations are shown in Table 7-6 and Table 7-7. Mean concentrations for PCBs at the SUBASE stations were virtually the same as the mean reference concentrations, at 12 µg/kg, with overall concentration ranges also being similar. Variability of PCB concentrations at the SUBASE stations (RSD = 67%) was slightly higher than seen at the

reference stations (RSD = 42%). None of the SUBASE stations exceeded the CBSQG value of 400 μg/kg.

The spatial distribution of PCBs at the SUBASE site is shown in Figure 7-12. The spatial pattern appeared to be influenced by the distribution of fines and TOC, as well as proximity with the shoreline. PCBs had a significant positive correlation with both fines and TOC (Table 7-8). Stations closest to the shoreline have the highest levels of PCBs, decreasing with distance from the shore. The stations generally can be separated into two different ranges. The most shoreward stations (SB1, SB2, SB4, SB8, SB9, SB12) had a PCB range of 17 – 25 μg/kg, while the outer stations (SB3, SB5, SB6, SB7, SB10, SB11, SB14) had a range of 1 to 8 μg/kg. Site SB13 falls in between these two ranges with a PCB concentration of 12 μg/kg, possibly related to the greater TOC/fines content at this station compared to the other outer stations.

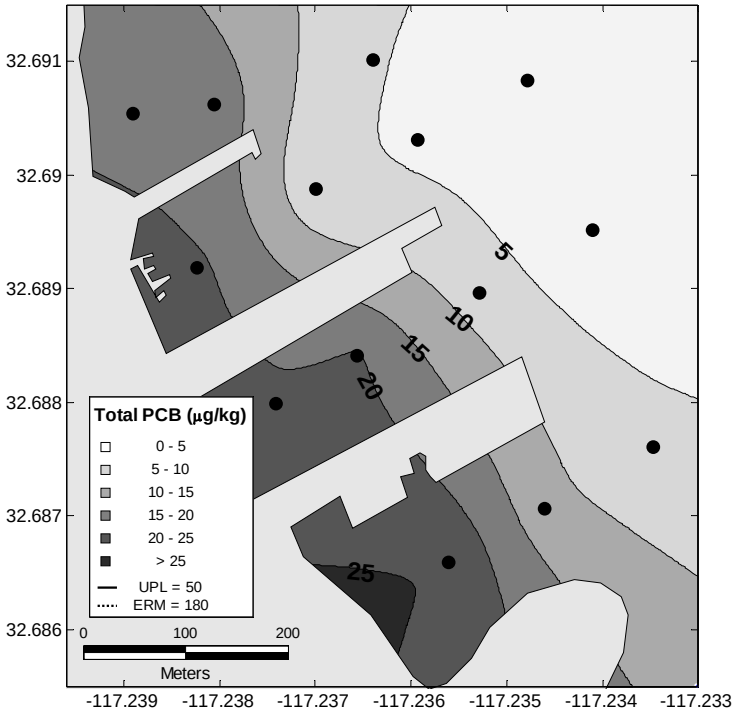

Figure 7-12. Spatial distribution of TPCBs at the SUBASE stations.

7.5 PESTICIDES

Concentrations of pesticides were characterized for γ-chlordane, α-chlordane, 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT. Total Chlordane (TCHLOR) was determined as the sum of γ-Chlordane and α-Chlordane. Total DDT (TDDT) was determined as the sum of all DDE, DDD, and DDT isomers. Pesticide concentrations in sediment provide one indicator of potential contaminant exposure for aquatic organisms. Results for sediment pesticides at reference and SUBASE stations are summarized below. The data displayed include only the results for total Chlordane and total DDT, however the complete set of data for all individual congeners can be found in Appendix A.

7.5.1 Reference Stations

Pesticide results for the reference stations are shown in Table 7-6 and Table 7-7. TCHLOR concentrations at the reference stations generally were low, and showed minimal variation from station to station. TCHLOR at the reference stations ranged from 0.17 to 1.4 μg/kg. The mean TCHLOR concentration was 0.87 μg/kg and the low variability at the reference stations was reflected in the RSD (47%). TDDT concentrations at the reference stations also generally were low with minimal among station variation. TDDT ranged from 0.49 to 2.19 μg/kg. No comparative ranges for pesticides were established for reference stations in the SAP, however, the range of TCHLOR and TDDT at the reference stations in this study was comparable, although lower, to the range reported at BPTCP reference stations (1-4 μg/kg and 3-9 μg/kg, respectively). None of the reference stations had Chlordane or DDT levels exceeding their respective SQG of 4.8 μg/kg (Chlordane) and 100 μg/g OC (DDT).

7.5.2 SUBASE Stations

Pesticide results for the SUBASE stations are shown in Table 7-6 and Table 7-7. Mean concentrations for pesticides at the SUBASE stations were similar to the reference mean for both TCHLOR and TDDT. Variability of TCHLOR and TDDT concentrations at the SUBASE stations was slightly higher than seen at the reference stations. The overall range of TCHLOR concentrations was similar between the SUBASE and reference stations. Two SUBASE stations had maximum TDDT concentrations higher than the highest reference concentration: SB10 (2X) and SB12 (1.5X). None of the SUBASE stations had Chlordane or DDT levels exceeding their respective SQG.

The spatial distributions of TCHLOR and TDDT at the SUBASE site are shown in Figure 7-13 and Figure 7-14, respectively. The spatial patterns appeared to be influenced by the distribution of fines and TOC. Total Chlordane levels are highest closest to shoreline, decreasing moving out into the bay, with slightly elevated values at SB1 and SB12. Total DDT levels had generally had a similar spatial distribution, although there was a hot spot at station SB10 influencing the overall pattern. TCHLOR was significantly correlated with fines content, but not TOC and TDDT had a significant positive relationship with TOC content, but not fines. In addition, TCHLOR was not correlated with TDDT, primarily due to increased levels of TDDT at SB10. Removing this data point makes the correlation significant ($r = 0.76$; $p = 0.003$). This suggests a common overall origin, as well as transport and partitioning processes associated with fines and TOC, with an additional source located near SB10.

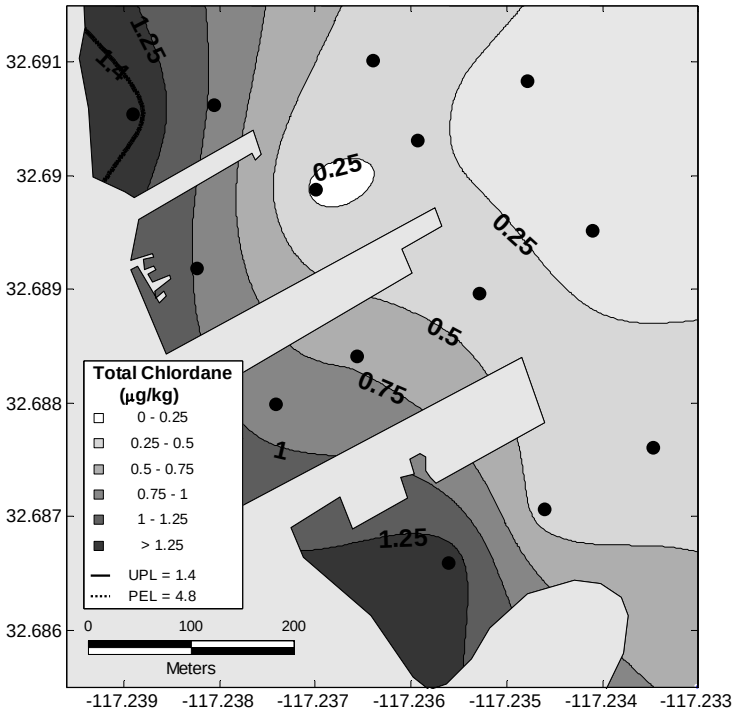

Figure 7-13. Spatial distribution of TCHLOR at the SUBASE stations. Upper Protection Limit (UPL) and PEL (Probable Effects Level) are contoured in a bold solid and dashed line, respectively, if exceedances were observed.

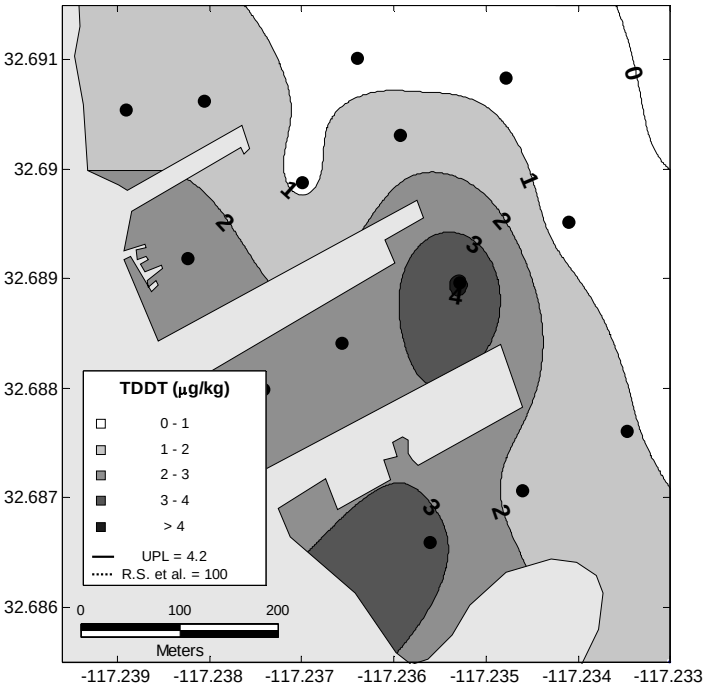

Figure 7-14. Spatial distribution of TDDT at the SUBASE stations. Upper Protection Limit (UPL) and ERM comparable values (R. Schwartz et al. 1989) are contoured in a bold solid and dashed line, respectively, if exceedances were observed.

7.6 ORGANOTINS

Concentrations of organotin compounds in the sediment were measured at the reference and SUBASE stations. Four organotin compounds were measured: tetra-n-butyltin (TTBT), tri-n-butyltin (TBT), di-n-butyltin (DBT), and mono-n-butyltin (MBT). Total organotin concentrations (TOT) were determined as the sum of all individual compounds. The chemical and biological properties of organotins vary from species to species. The compounds decay from the tetra-n species to the mono-n species over time. Generally, tetra-organotins are stable, ineffective as biocides, and are relatively non-toxic. The tri-organotin species are the most toxic of the different classes. The principal organotin of concern are compounds of TBT, which has been widely used as a biocide in marine antifouling paints and coatings, and has since been banned by USEPA. TBT adsorbs to organic matter, and can persist in the sediment. Di-organotins generally have a low toxicity, while mono-organotins show no biocidal activity and a very low toxicity to mammals. Results for the organotins are summarized below in Table 7-9 and Table 7-10. Results are reported in μg/kg dry weight.

7.6.1 Reference Stations

Organotin results for the reference stations are shown in Table 7-9 and Table 7-10. Mean TOT concentrations ranged from 3.0 to 9.6 μg/kg, and showed minimal variation from station to station as indicated by the RSD of 32%. The four congeners also showed minimal variation from station to station, with the exception of MBT during the reference survey which had an RSD of 86%. The TBT and DBT congeners were present in greatest proportions, making up between 55% and 90% of the total. Mean TOT concentrations were approximately 60% lower during the reference survey, compared to the Recon survey which occurred two months earlier. The change in TOT primarily is due to decreases in TBT and DBT concentrations.

7.6.2 SUBASE Stations

Organotin results for the SUBASE stations are shown in Table 7-9 and Table 7-10. Mean TOT concentrations ranged from 2.3 to 8.6 μg/kg, and are within range of the reference station concentrations. As with the reference stations the TBT and DBT fractions have the greatest concentrations, making up between 52% and 82% (mean = 73%) of the total. SB12 had the highest TOT concentration, however MBT had the greatest concentration of the four congeners (44% of total) for this station, which was not observed at any of the other SUBASE stations and represents the least toxic fraction. The next greatest TOT concentration is 6.8 μg/kg (SB4), and is approximately one third of the two highest reference station concentrations (SB2436-R and SBC001SS31-R), all of which had the typically low observed MBT values. Organotin concentrations generally increased with increasing TOC and fines levels (Figure 7-15 and Figure 7-16). This relationship holds true with all data points for TBT, however two outliers were observed for TOT. Stations SB12 and SB4 fall well off this trend due to increased concentrations of MBT and DBT, respectively.

Spatial distributions of TBT are shown in Figure 7-17. Spatial patterns of TBT levels differ from those of TOC and fines, although a significant relationship exists between TBT and both TOC and fines content (Table 7-11). TBT levels show no clear spatial trend, with high values observed at SB2, SB4, SB9, and SB10.

Table 7-9. Sediment organotin data for reference and SUBASE stations. Results are reported in μg/kg dry weight.

Area	Station	TTBT	TBT	DBT	MBT	TOT
Recon	SB2229-R	0.27	2.61	3.37	0.63	6.88
	SB2433-R	0.30	2.64	3.43	0.69	7.06
	SB2436-R	0.31	3.33	5.05	0.71	9.39
	SB2441-R	0.37	2.97	2.55	0.85	6.74
	SB90056-R	0.25	2.53	3.24	0.56	6.58
	SBC001SS31-R	0.31	3.16	5.47	0.70	9.64
Reference	SB2229	0.26	1.43	1.56	0.58	3.83
	SB2433	0.27	1.45	1.45	0.62	3.79
	SB2436	0.29	1.74	2.16	2.91	7.10
	SB2441	0.44	1.91	2.17	1.01	5.53
	SB90056	0.24	1.23	0.98	0.55	3.00
	SBC001SS31	0.33	1.86	2.09	0.75	5.02
SUBASE	SB1	0.32	1.69	1.45	0.73	4.18
	SB2	0.45	2.54	1.48	1.03	5.50
	SB3	0.27	1.52	0.70	0.63	3.12
	SB4	0.38	2.76	2.77	0.86	6.77
	SB5	0.26	1.54	0.87	0.60	3.27
	SB6	0.31	2.42	1.55	0.72	5.00
	SB7	0.30	1.59	0.91	0.69	3.49
	SB8	0.35	1.77	1.96	0.81	4.89
	SB9	0.41	2.65	1.22	0.93	5.21
	SB10	0.32	2.61	1.01	0.74	4.68
	SB11	0.23	1.30	0.28	0.53	2.34
	SB12	0.37	2.01	2.41	3.80	8.59
	SB13	0.41	2.12	1.44	0.93	4.90
	SB14	0.34	1.83	1.23	0.78	4.17

Table 7-10. Summary statistics for sediment organotin data (μg/kg).

Area	Statistic	TTBT	TBT	DBT	MBT	TOT
Recon	Minimum	0.25	2.53	2.55	0.56	6.58
	Maximum	0.37	3.33	5.47	0.85	9.64
	Mean	0.30	2.87	3.85	0.69	7.71
	Std Dev	0.04	0.33	1.14	0.10	1.41
	RSD (%)	14%	11%	30%	14%	18%
Reference	Minimum	0.24	1.23	0.98	0.55	3.00
	Maximum	0.44	1.91	2.17	2.91	7.10
	Mean	0.30	1.60	1.74	1.07	4.71
	Std Dev	0.07	0.27	0.49	0.92	1.49
	RSD (%)	24%	17%	28%	86%	32%
SUBASE	Minimum	0.23	1.30	0.28	0.53	2.34
	Maximum	0.45	2.76	2.77	3.80	8.59
	Mean	0.34	2.03	1.38	0.98	4.72
	Std Dev	0.06	0.49	0.66	0.82	1.58
	RSD (%)	18%	24%	48%	84%	34%

Table 7-11. Correlation matrix for SUBASE site physical properties and organotin concentrations. Values highlighted in gray represent a significant relationship at the 0.05 level.

	Depth	Fines	TOC	TTBT	TBT	DBT	MBT	TOT
Depth	1.000	0.039	0.097	0.105	-0.061	-0.384	-0.664	-0.521
Fines		1.000	0.970	0.978	0.744	0.737	0.353	0.761
TOC			1.000	0.960	0.742	0.646	0.257	0.672
TTBT				1.000	0.721	0.602	0.314	0.678
TBT					1.000	0.536	0.114	0.622
DBT						1.000	0.532	0.885
MBT							1.000	0.790
TOT								1.000

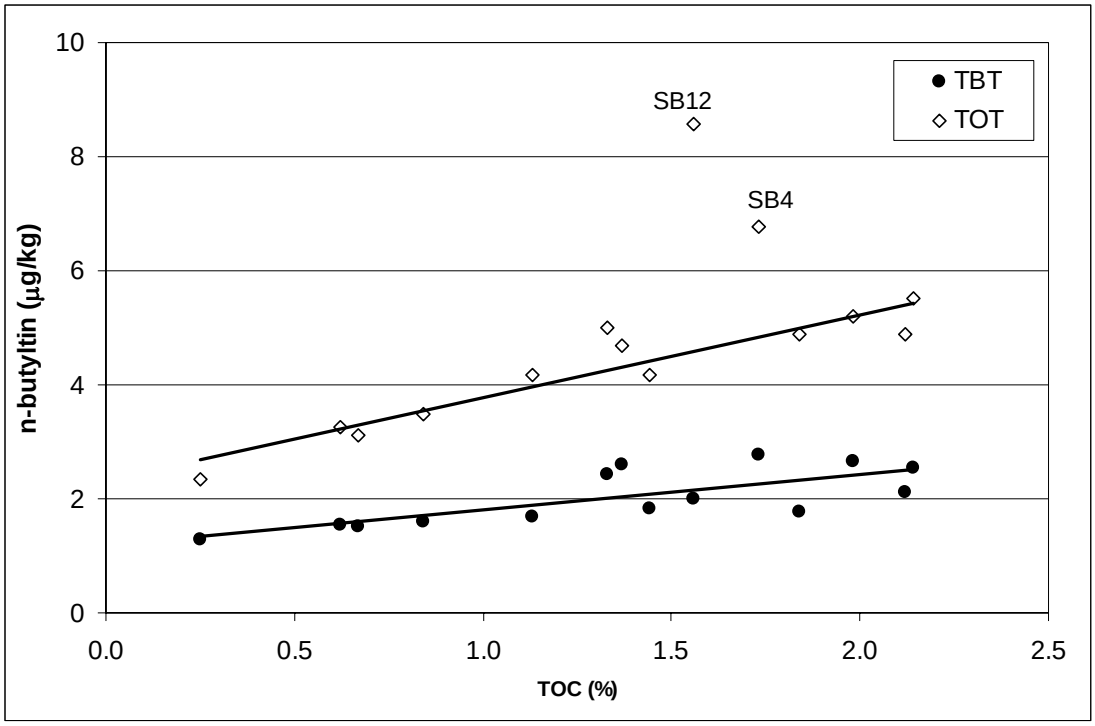

Figure 7-15. Organotin as a function of TOC for SUBASE stations. Where TBT = tributyltin and TOT = total of all n-butyltin species.

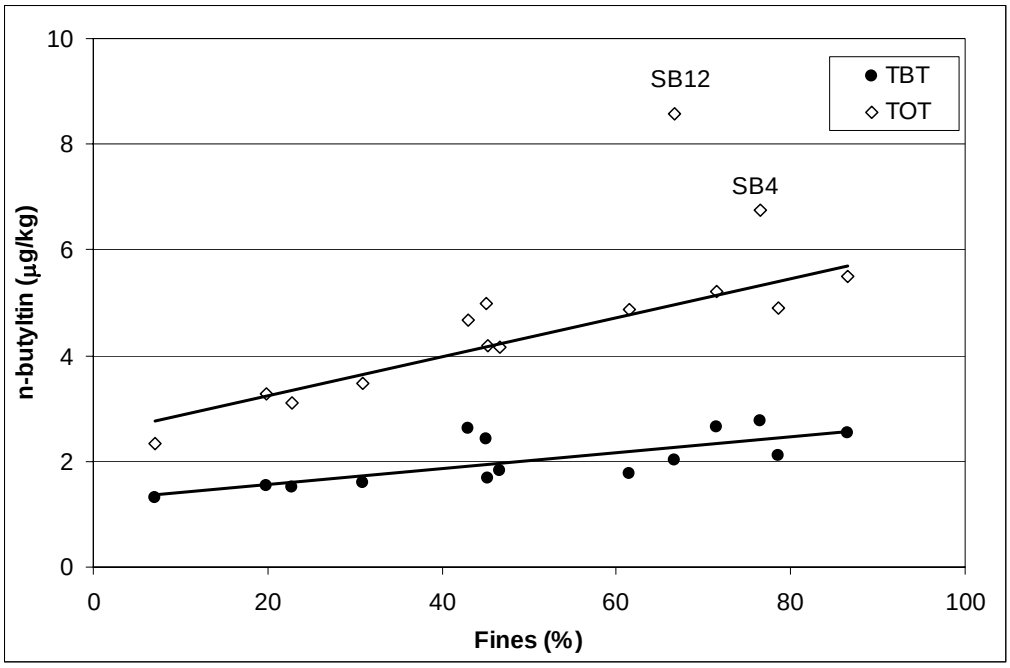

Figure 7-16. Organotin as a function of fines for SUBASE stations. Where TBT = tributyltin and TOT = total of all n-butyltin species.

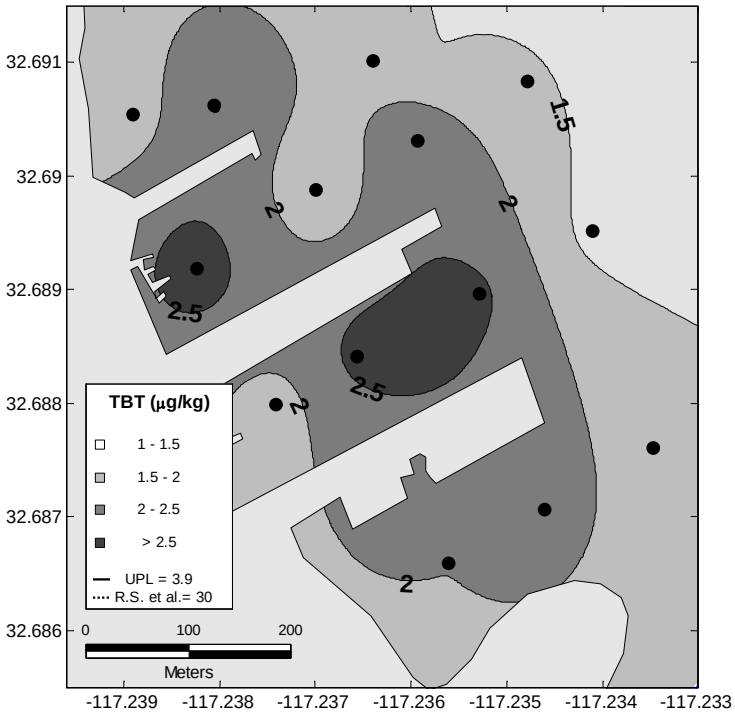

Figure 7-17. Spatial distribution of TBT at the SUBASE stations. Upper Protection Limit (UPL) and ERM comparable values (R. Schwartz et al., 1989) are contoured in a bold solid and dashed line, respectively, if exceedances were observed.

8.0 TISSUE CHEMISTRY RESULTS

8.1 TISSUE SOLIDS AND LIPID CONTENT

The fraction of solids and lipid present in the tissues of clams exposed to site sediments were characterized at all reference and a subset of the SUBASE study stations. Tissue solids and lipid content also were characterized for clams exposed to control (home) sediment. Study stations that were characterized included SUBASE stations SB2, SB5, SB8, SB9, SB10, SB11, SB12 (Figure 4-1). Tissue solids content reflects the ratio of dry tissue to wet tissue in the clams and is a required parameter for conversion from dry weight units to wet weight units. Tissue lipid content indicates the fat fraction of the tissue. Many bioaccumulative compounds exhibit low water solubility and tend to concentrate in the lipid fractions of biological tissues. Results for tissue solids and lipid content in control, reference, and SUBASE samples are summarized below. The complete results are shown in Appendix B.

8.1.1 Control

Solids and lipid results for the control samples are shown in Table 8-1 and Table 8-2. Three composite control samples were analyzed. Each control sample was composited from clams in five separate exposure chambers containing home sediment. Solids content in the control tissue ranged from 8.6% to 10.8%, while lipids content ranged from only 0.4% to 0.5%. Variation among the control replicates was low indicating consistency among the exposures and analytical procedures.

8.1.2 Reference Stations

Solids and lipid content results for the reference stations are shown in Table 8-1 and Table 8-2. The result for each reference station represents the composite of five replicate laboratory exposures. In addition, for station CP2433, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported values for station CP2433 are thus the means of these three field replicates. The range of solids and lipid content across the reference stations was generally low. For example, solids content ranged from 9.7 to 11.0%, with an RSD of only 5%, and lipid content had a range of 0.36 to 0.48% with an RSD of 11%. Reference station mean tissue solids and lipid content were comparable to concentrations in the control samples. These results indicate that clams exposed to reference sediments had no major differences in general tissue properties compared to the clams exposed to control sediments.

8.1.3 SUBASE Stations

Solids and lipid content results for the SUBASE stations are shown in Table 8-1 and Table 8-2. The result for each station represents the composite of five replicate laboratory exposures. In addition, for station SB9, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported values for station SB9 are the means of these three field replicates. The range of solids and lipid content across the SUBASE stations generally was comparable to those of the reference stations and the control samples, although lipid content varied slightly more at SUBASE. For example, solids content ranged from 9.0 to 13.0%, with an RSD of 11%, and lipid content had a range of 0.41 to 0.68% with an RSD of 17%. The slightly higher variation at SUBASE was most

likely due to elevated values at station SB8, which had the highest solids and lipids content among the SUBASE stations. SB10 had the lowest lipids content, while SB12 had the lowest solids content. These results indicate that clams exposed to SUBASE sediments had no major differences in general tissue properties compared to the clams exposed to reference and control sediments.

Table 8-1. Tissue solids (%) and lipid content (%) data (dry weight) for the control, reference, and SUBASE stations.

Area	Station	Solids	Lipids
Control	Control	9.8	0.43
Reference	SB2229	10.5	0.48
	SB2433	10.3	0.45
	SB2436	11.0	0.47
	SB2441	10.7	0.45
	SB90056	9.7	0.38
	SBC001SS31	10.9	0.36
SUBASE	SB2	10.1	0.50
	SB5	11.6	0.44
	SB8	13.0	0.68
	SB9	10.3	0.49
	SB10	10.9	0.41
	SB11	11.3	0.48
	SB12	9.0	0.51

Table 8-2. Summary statistics for solids (%) and lipid content (%) in clams exposed to control, reference, SUBASE sediments.

Area	Statistic	Solids	Lipids
Control	Minimum	8.6	0.36
	Maximum	10.8	0.46
	Mean	9.8	0.43
	Std Dev	1.1	0.06
	RSD (%)	11%	14%
Reference	Minimum	9.7	0.36
	Maximum	11.0	0.48
	Mean	10.5	0.43
	Std Dev	0.5	0.05
	RSD (%)	5%	11%
SUBASE	Minimum	9.0	0.41
	Maximum	13.0	0.68
	Mean	10.9	0.50
	Std Dev	1.2	0.09
	RSD (%)	11%	17%

8.2 METALS

Concentrations of metals in the tissues of clams exposed to site sediments were characterized at all reference and a subset of the SUBASE study stations. Tissue concentrations were also characterized for clams exposed to control (home) sediment. Tissues were analyzed for a range of metals including silver, arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc. Study stations that were characterized included SUBASE stations SB2, SB5, SB8, SB9, SB10, SB11, SB12 (Figure 4-1). Tissue concentrations reflect the uptake of metals from site sediments as regulated by their concentration and bioavailability in the sediment. Results for tissue metals at reference and SUBASE stations are summarized below. The data displayed include only those metals that were identified as CoPCs in the historical review. The complete set of data can be found in Appendix B.

8.2.1 Control

Metals results for the control samples are shown in Table 8-3 and Table 8-4. Three composite control samples were analyzed. Each control sample was composited from clams in five separate exposure chambers containing home sediment. Metal concentrations in replicate control tissues had low variability indicating consistency among the exposures and analytical procedures. For example, copper in the control sample tissues ranged from 22.6 to 25.7 μg/g, with an RSD of only 7%, and arsenic ranged from 18.6 to 24.5 μg/g with an RSD of only 15%. The remaining metals had similar ranges of variability, with chromium having the highest at 39%. Thus results from the control samples provide a useful initial baseline for comparison of tissue concentrations from the reference and site stations.

8.2.2 Reference Stations

Metals results for the reference stations are shown in Table 8-3 and Table 8-4. The tissue concentration for each reference station represents the composite of five replicate laboratory exposures. In addition, for station SB2433, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported metals values for station SB2433 are thus the means of these three field replicates. The range of concentrations across the reference stations was generally low. For example, copper concentrations ranged from 26.7 to 32.4 μg/g, with an RSD of only 7%, and mercury had a range from 0.074 to 0.105 μg/g with an RSD of 15%. Reference station mean tissue concentrations generally were comparable to control samples ranging between 1.1X and 1.3X higher for silver, arsenic, copper, mercury, nickel, and zinc. Mean tissue concentrations of other metals were somewhat higher in the reference stations than control including chromium (1.6X) and lead (2.4X), while cadmium was slightly lower (0.9X). These results indicate that reference areas of San Diego Bay have somewhat higher bioaccumulation potential for chromium and lead compared to the control sediments.

8.2.3 SUBASE Stations

Metals results for the SUBASE stations are shown in Table 8-3 and Table 8-4. The tissue concentration for each station represents the composite of five replicate laboratory exposures. In addition, for station SB9, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported metals values for station SB9 are thus the means of these three field replicates. The range of concentrations across the SUBASE stations was generally low and comparable to both the reference and control. For example, copper concentrations ranged from 24.5 to 37.5 μg/g, with

an RSD of 15%, and zinc had a range from 87.9 to 118 μg/g with an RSD of 10%. SUBASE station mean tissue concentrations of arsenic, cadmium, copper, mercury, nickel and zinc were comparable to concentrations in the control samples. Mean tissue concentrations of other metals were generally somewhat higher in SUBASE stations relative to control including silver (1.6X) chromium (1.8X), and lead (2.3X). Results compared to reference also were similar, although overall changes were not as large as comparing SUBASE with the control. Arsenic, cadmium, mercury and zinc had similar mean tissue concentrations, and silver (1.2X), copper (1.3X) and lead (1.7X) had somewhat higher means at the SUBASE stations compared to the reference. Among the SUBASE stations, SB8 had the highest concentrations of all metals except for arsenic, which was highest at SB12. Lowest metal levels were found primarily at SB5 (Ag, As, Cu, and Pb), SB11 (Cr, and Zn), and SB12 (Cd, Hg, and Ni). These results indicate that SUBASE stations have somewhat higher bioaccumulation potential for silver, chromium, copper and lead compared to the reference and/or control sediments.

The spatial distributions of metals at the SUBASE sites are shown in Figure 8-1 through Figure 8-6. The stations follow a transect which runs either parallel (SB2, SB5, SB9, and SB12), or perpendicular to the shoreline (SB8, SB9, SB10, and SB11). In general, metals values for stations in the parallel transect showed minimal variation with no clear spatial trend; exceptions were lower silver values at SB5, and higher copper values at SB9. However, stations along the perpendicular transect generally showed highest metals concentration close to the shoreline, decreasing as stations moved further into the bay, away from the quaywall. Cadmium and arsenic did not follow this pattern, with little variation along the transect running perpendicular to the shoreline. Tissue metals concentrations correspond well with fines and TOC values along both transects (Figure 8-7 and Figure 8-8), with highest values along the shoreline. This suggests that bioaccumulation of metals is being driven by sediment metals concentrations rather than sediment grain size and organic content, and that stations closest to shore have a higher bioaccumulation potential. Correlations (r) between metals in tissues and metals in sediment were examined for the SUBASE stations (Table 8-5). Silver and zinc tissue concentrations were significantly correlated ($p < 0.05$) with copper and mercury sediment concentrations, although several other relationships had higher r-values.

Table 8-3. Tissue metals data (mg/kg dry weight) for the control, reference, and SUBASE.

Area	Station	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Control	Control	0.230	22.4	0.226	1.43	23.7	0.070	2.12	1.22	81.0
Reference	SB2229	0.311	24.8	0.180	2.20	32.4	0.105	2.10	2.86	92.0
	SB2433	0.278	22.8	0.192	3.25	30.2	0.077	2.40	3.23	99.6
	SB2436	0.262	25.0	0.211	2.24	29.1	0.094	2.08	3.06	100
	SB2441	0.424	26.8	0.178	2.28	31.1	0.074	2.57	2.92	93.2
	SB90056	0.227	25.8	0.191	2.04	29.2	0.078	2.28	2.58	91.3
	SBC001SS31	0.245	25.1	0.231	1.86	26.7	0.075	2.01	2.67	97.9
SUBASE	SB2	0.379	23.0	0.221	2.72	28.7	0.066	2.60	2.48	99.3
	SB5	0.247	22.7	0.217	2.29	24.5	0.069	2.52	2.37	88.3
	SB8	0.482	25.7	0.265	3.71	37.5	0.085	3.30	3.85	118
	SB9	0.414	25.7	0.204	2.30	34.9	0.071	2.53	2.88	102
	SB10	0.402	25.1	0.239	2.52	33.9	0.083	2.90	2.88	106
	SB11	0.296	26.0	0.248	2.22	29.6	0.074	2.42	2.39	87.9
	SB12	0.358	26.2	0.189	2.34	28.1	0.059	2.25	2.56	103

Table 8-4. Summary statistics for metal concentrations (mg/kg dry weight) in clams exposed to control, reference, SUBASE sediments.

Area	Statistic	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Control	Minimum	0.200	18.6	0.205	1.03	22.6	0.055	2.04	1.17	75.4
	Maximum	0.261	24.5	0.241	2.08	25.7	0.087	2.23	1.27	86.0
	Mean	0.230	22.4	0.226	1.43	23.7	0.070	2.12	1.22	81.0
	Std Dev	0.031	3.27	0.019	0.566	1.76	0.016	0.098	0.050	5.32
	RSD (%)	13%	15%	8%	39%	7%	23%	5%	4%	7%
Reference	Minimum	0.227	22.8	0.178	1.86	26.7	0.074	2.01	2.58	91.3
	Maximum	0.424	26.8	0.231	3.25	32.4	0.105	2.57	3.23	100
	Mean	0.291	25.0	0.197	2.31	29.8	0.084	2.24	2.89	95.7
	Std Dev	0.071	1.33	0.020	0.485	1.95	0.013	0.216	0.240	3.95
	RSD (%)	24%	5%	10%	21%	7%	15%	10%	8%	4%
SUBASE	Minimum	0.247	22.7	0.189	2.220	24.5	0.059	2.25	2.37	87.9
	Maximum	0.482	26.2	0.265	3.710	37.5	0.085	3.30	3.85	118
	Mean	0.368	24.9	0.226	2.585	31.0	0.072	2.65	2.77	101
	Std Dev	0.078	1.46	0.026	0.525	4.54	0.009	0.349	0.520	10.4
	RSD (%)	21%	6%	12%	20%	15%	13%	13%	19%	10%

Table 8-5. Correlation (r) between metals concentrations in tissue and sediment for SUBASE bioaccumulation stations. Gray cells indicate statistically significant correlations ($p<0.05$).

		Tissue								
		Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Sediment	Ag	0.74	0.18	-0.35	0.42	0.42	-0.19	0.19	0.45	0.74
	As	0.75	-0.01	-2.59	0.50	0.44	-0.09	0.32	0.50	0.74
	Cd	0.52	0.13	-0.51	0.31	0.15	-0.43	-0.02	0.28	0.59
	Cr	0.74	0.02	-0.31	0.40	0.43	-0.08	0.28	0.43	0.74
	Cu	0.76	0.12	-0.31	0.46	0.45	-0.12	0.27	0.50	0.77
	Hg	0.78	0.22	-0.25	0.54	0.49	-0.11	0.29	0.57	0.79
	Ni	0.74	0.02	-0.29	0.48	0.42	-0.14	0.27	0.47	0.73
	Pb	0.68	0.13	-0.41	0.34	0.34	-0.28	0.10	0.34	0.66
	Zn	0.71	0.12	-0.37	0.42	0.37	-0.22	0.18	0.43	0.72

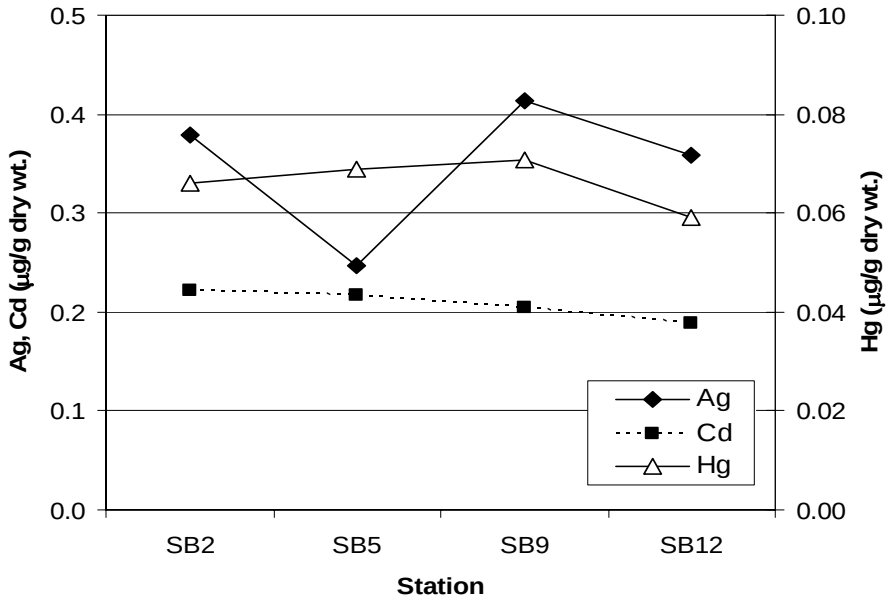

Figure 8-1. Spatial variation of tissue metals along the SUBASE transect running parallel to shore for silver, cadmium, and mercury.

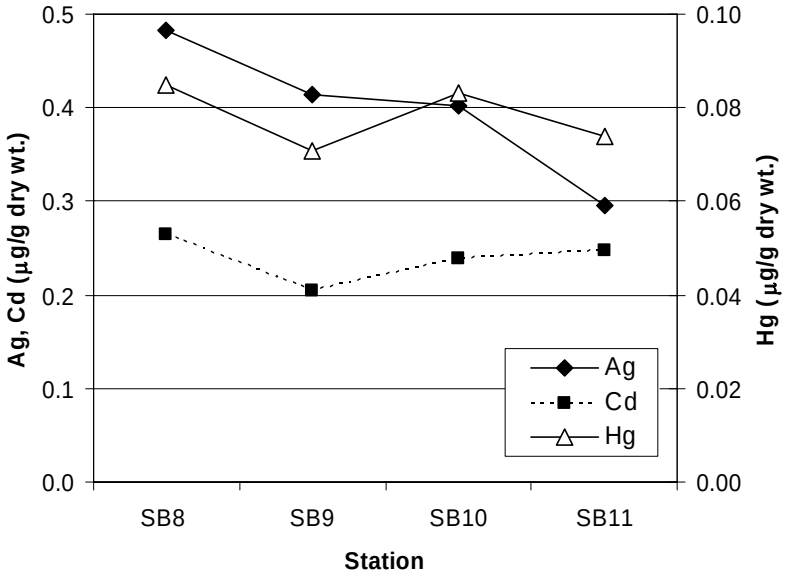

Figure 8-2. Spatial variation of tissue metals along the SUBASE transect running perpendicular to shore for silver, cadmium, and mercury.

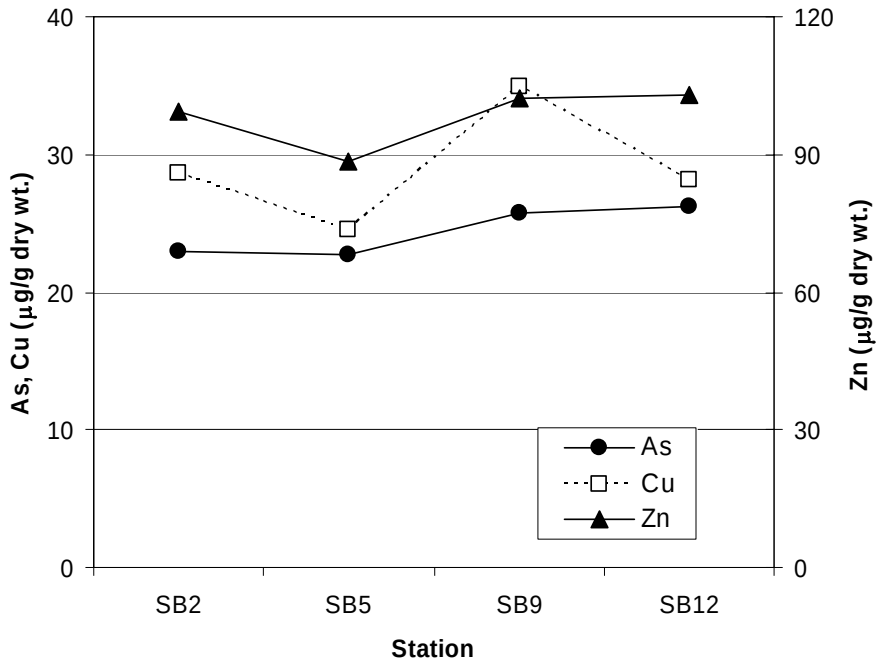

Figure 8-3. Spatial variation of tissue metals along the SUBASE transect running parallel to shore for arsenic, copper, and zinc.

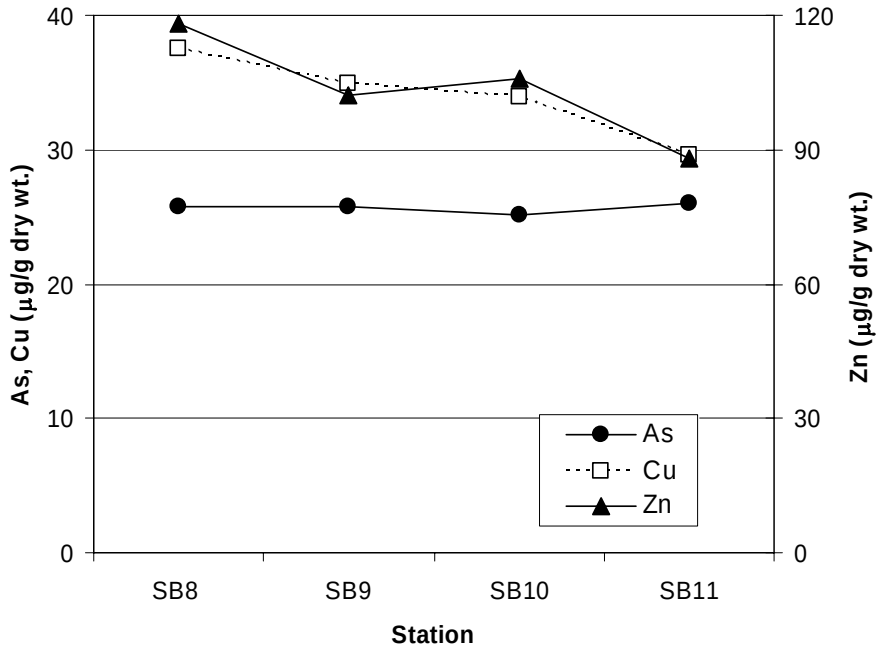

Figure 8-4. Spatial variation of tissue metals along the SUBASE transect running perpendicular to shore for arsenic, copper, and zinc.

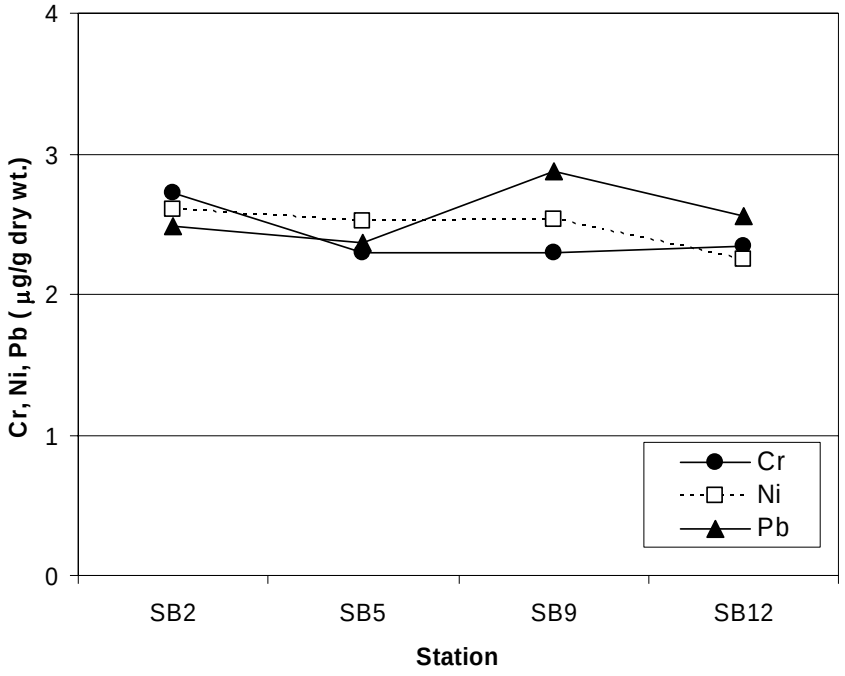

Figure 8-5. Spatial variation of tissue metals along the SUBASE transect running parallel to shore for chromium, nickel, and lead.

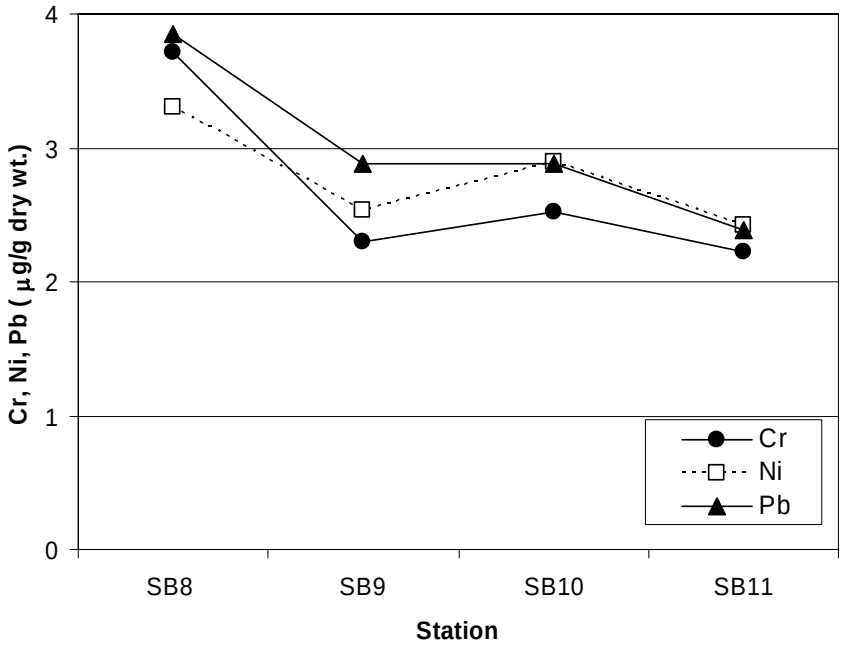

Figure 8-6. Spatial variation of tissue metals along the SUBASE transect running perpendicular to shore for chromium, nickel, and lead.

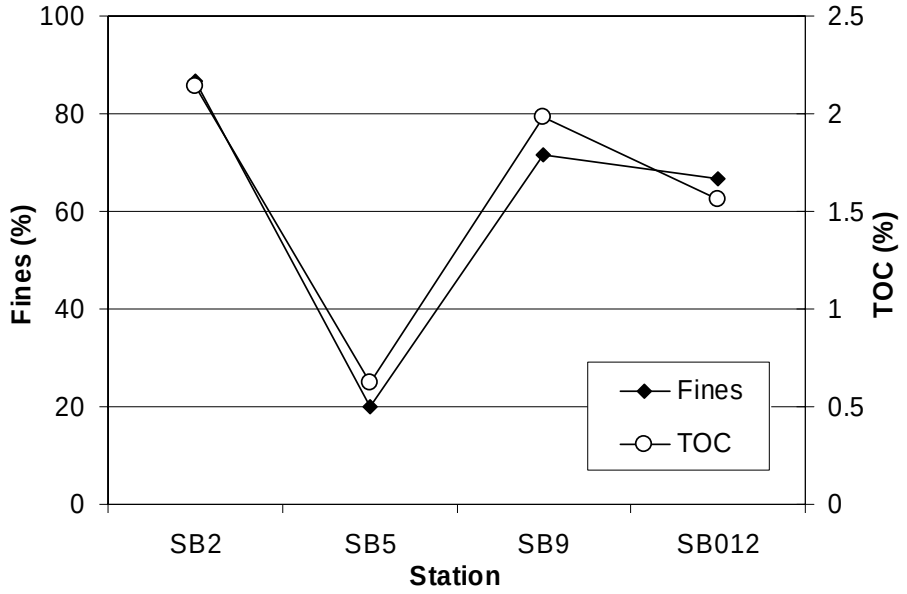

Figure 8-7. Spatial variation of Fines and TOC along the SUBASE transect running parallel to shore.

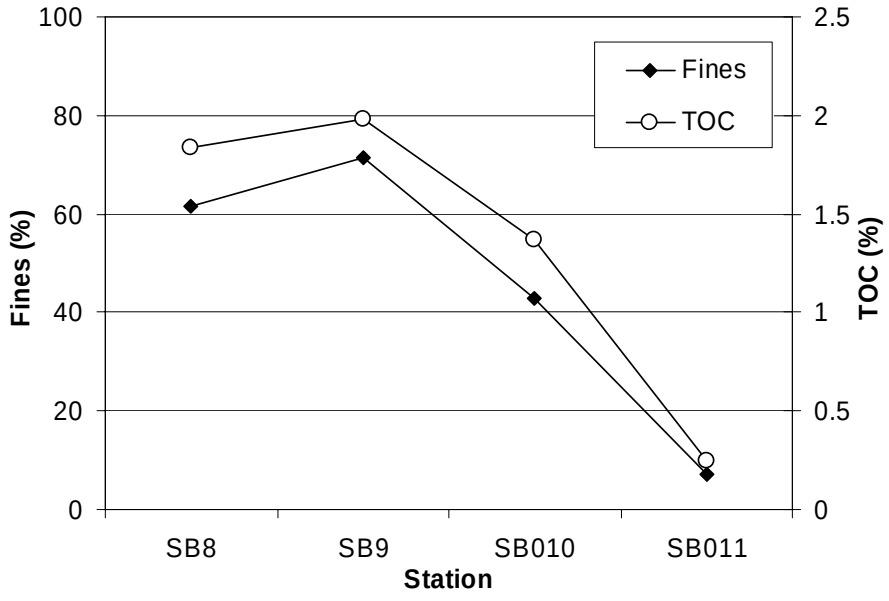

Figure 8-8. Spatial variation of Fines and TOC along the SUBASE transect running perpendicular to shore.

8.3 PAHs

Concentrations of PAHs in the tissues of clams exposed to site sediments were characterized at all reference and a subset of the SUBASE study stations. Tissue concentrations were also characterized for clams exposed to control (home) sediment. Tissues were analyzed for the same range of PAHs as described previously for the sediment analysis. Study stations that were characterized included SUBASE stations SB2, SB5, SB8, SB9, SB10, SB11, SB12 (Figure 4-1). Tissue concentrations reflect the uptake of PAHs from site sediments as regulated by their concentration and bioavailability in the sediment. Results for the PPPAH, LMWPAH, and HMWPAH summations at reference and SUBASE stations are given below. The complete set of data can be found in Appendix B.

8.3.1 Control

PAH results for the control and samples are shown in Table 8-6 and Table 8-7. Three composite control samples were analyzed. Each control sample was composited from clams in five separate exposure chambers containing home sediment. PAH variation among the control replicates was low indicating consistency among the exposures and analytical procedures. For example, PPPAH in the control sample tissues ranged from 109 to 123 μg/kg, with an RSD of only 6%. The other summations had similar ranges of variability. Thus results from the control samples provide a useful initial baseline for comparison of tissue concentrations from the reference and study site stations.

8.3.2 Reference Stations

PAH results for the reference stations are shown in Table 8-6 and Table 8-7. The tissue concentration for each reference station represents the composite of five replicate laboratory exposures. In addition, for station SB2433, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported PAH values for station CP2433 are thus the means of these three field replicates. PPPAH concentrations at the reference stations ranged from 533 to 1408 μg/kg, with an RSD of 35%. Most of the variability was associated with elevated accumulation at CP2441. Reference station mean tissue concentrations of PAHs generally were higher than the control samples including LMWPAH (6.2X), HMWPAH (7.7X), and PPPAH (7.5X). These results indicate that reference areas of San Diego Bay have higher bioaccumulation potential for PAHs compared to the control sediments.

8.3.3 SUBASE Stations

PAH results for the SUBASE stations are shown in Table 8-6 and Table 8-7. The tissue concentration for each station represents the composite of five replicate laboratory exposures. In addition, for station SB9, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported PAH values for station SB9 are thus the means of these three field replicates. The range and variability of concentrations across the SUBASE stations was higher than that for the control samples. For example, PPPAH concentrations ranged from 583 to 3589 μg/kg, with an RSD of 62%. Results relative to reference showed higher variability and higher levels for all station-mean summations at SUBASE including LMWPAH (1.6X), HMWPAH (2.0X), and PPPAH (1.9X). Among the SUBASE stations, SB12 had the highest PAH concentrations (all summations), while station SB11 had the lowest HMWPAH and PPPAH levels, and station SB10 had the lowest LMWPAH level. In general, the LMWPAH was a small fraction of the PPPAH concentration, indicating that the PAHs in the tissues are dominated by high molecular weight compounds. This is consistent with the fractionation observed in the sediments. Overall,

the results indicate that SUBASE stations generally have higher bioaccumulation potential for PAHs compared to the reference and/or control home sediments.

The spatial distribution of tissue PAHs at the SUBASE site is shown in Figure 8-9 and Figure 8-10. The stations follow a transect which runs either parallel to (SB2, SB5, SB9, and SB12), or perpendicular to the shoreline (SB8, SB9, SB10, and SB11). There was little spatial variability of LMWPAH along the transect running parallel to shore (Figure 8-9). HMWPAH and PPPAH values along the same transect were relatively similar from SB2 to SB9 before spiking at SB12, which had the highest values of all stations. The transect running perpendicular to shore had a more clearly defined spatial trend. Tissue HMWPAH and PPPAH concentrations decreased steadily along the transect moving away from the shoreline, while LMWPAH concentrations changed little (Figure 8-10). This pattern corresponded closely with the concentrations in the sediment. Correlations (r) between PAHs in tissue and PAHs in sediment were examined for the SUBASE stations (Table 8-8). For this analysis, tissue concentrations were normalized to lipid content, and sediment concentrations were normalized to TOC. Statistically significant correlations were observed for all PAH summations comparing sediment and tissue concentrations, with all correlation coefficients ranging from 0.84 to 0.97.

Table 8-6. Tissue organic contaminant data from control, reference, and SUBASE stations (μg/kg dry weight).

Area	Station	LMWPAH	HMWPAH	PPPAH	TPCB	TCHLOR	TDDT
Control	Control	23	87	115	12	1.3	4.4
Reference	SB2229	133	692	882	74	4.4	11
	SB2433	126	538	817	63	3.5	14
	SB2436	140	692	856	73	4.3	13
	SB2441	250	1143	1408	27	1.9	10
	SB90056	101	523	661	44	2.1	10
	SBC001SS31	93	421	533	52	3.7	12
SUBASE	SB2	300	1483	1818	35	1.8	11
	SB5	189	973	1194	38	1.7	10
	SB8	244	1519	1790	74	2.4	12
	SB9	209	1132	1373	81	2.2	13
	SB10	123	685	839	26	1.6	10
	SB11	139	432	583	21	1.5	6.9
	SB12	359	3112	3589	61	2.6	15

Table 8-7. Summary statistics for the tissue organic contaminant data (μg/kg).

Area	Statistic	LMWPAH	HMWPAH	PPPAH	TPCB	TCHLOR	TDDT
Control	Minimum	20	83	109	11	1.2	3.9
	Maximum	26	89	123	13	1.5	4.8
	Mean	23	87	115	12	1.3	4.4
	Std Dev	2.7	3.2	7.4	1.1	0.2	0.5
	RSD (%)	12%	4%	6%	9%	12%	11%
Reference	Minimum	93	421	533	27.5	1.9	9.8
	Maximum	250	1143	1408	73.6	4.4	13.5
	Mean	141	668	859	55.6	3.3	11.7
	Std Dev	57	255	300	18.0	1.1	1.5
	RSD (%)	40%	38%	35%	32%	32%	13%
SUBASE	Minimum	123	432	583	21.5	1.5	6.9
	Maximum	359	3112	3589	80.6	2.6	15.1
	Mean	223	1334	1598	47.9	2.0	11.0
	Std Dev	85	878	989	23.7	0.43	2.6
	RSD (%)	38%	66%	62%	49%	22%	23%

Table 8-8. Correlation (r) between organic contaminant concentrations in tissue and sediment for SUBASE bioaccumulation stations. Gray cells indicate statistically significant correlations ($p<0.05$).

		Tissue					
		LMWPAH	HMWPAH	PPPAH	TPCB	TCHOLR	TDDT
Sediment	LMWPAH	0.85	0.97	0.96	0.60	0.91	0.88
	HMWPAH	0.85	0.96	0.96	0.63	0.93	0.88
	PPPAH	0.84	0.96	0.96	0.62	0.93	0.88
	TPCB	0.83	0.80	0.81	0.80	0.91	0.93
	TCHOLR	0.92	0.92	0.92	0.58	0.85	0.87
	TDDT	0.16	0.39	0.38	0.21	0.38	0.54

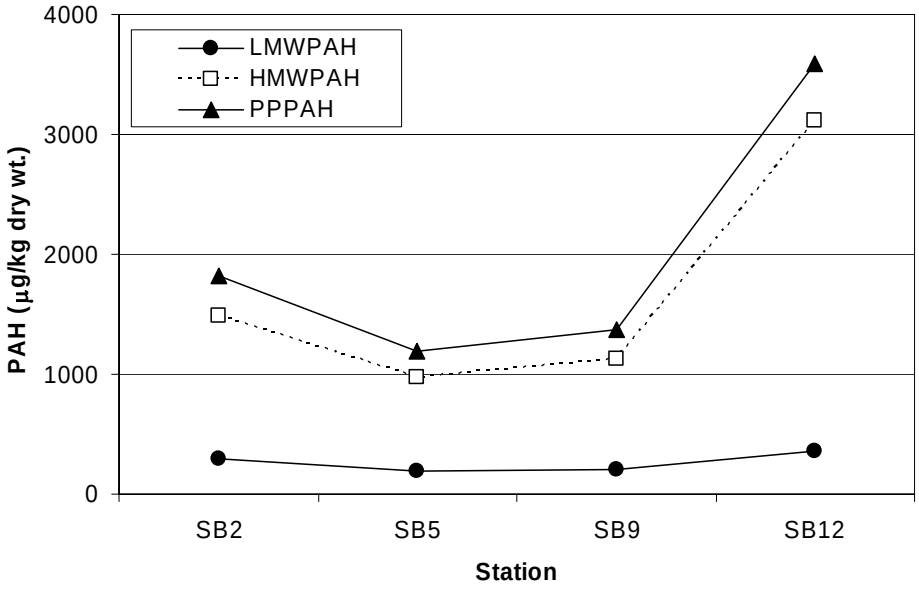

Figure 8-9. Spatial variation of tissue PAHs along the SUBASE transect running parallel to shore.

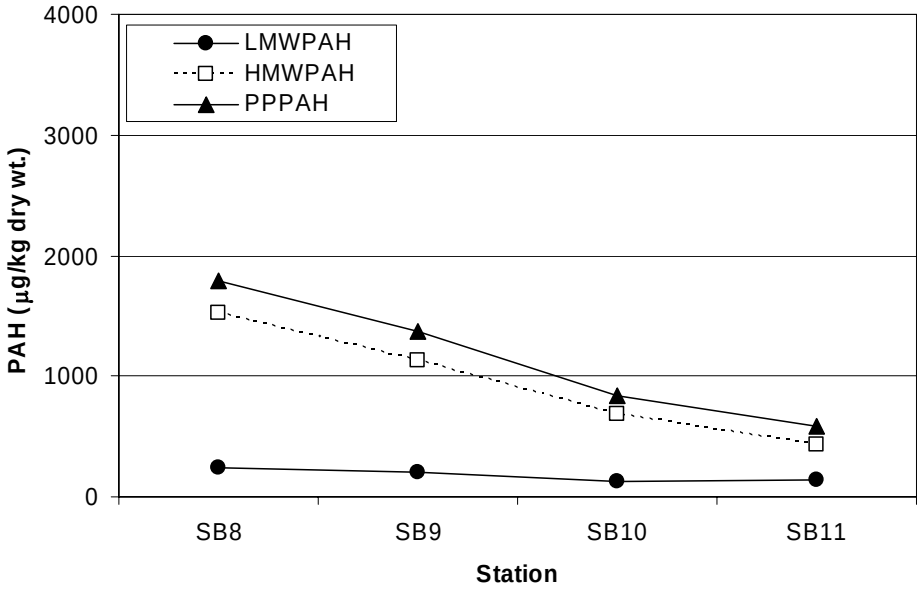

Figure 8-10. Spatial variation of tissue PAHs along the SUBASE transect running perpendicular to shore.

8.4 PCBs

Concentrations of PCBs in the tissues of clams exposed to site sediments were characterized at all reference and a subset of the SUBASE study stations. Tissue concentrations also were characterized for clams exposed to control (home) sediment. Tissues were analyzed for the same range of PCBs as described previously for the sediment analysis. Study stations that were characterized included SUBASE stations SB2, SB5, SB8, SB9, SB10, SB11, SB12 (Figure 4-1. Tissue concentrations reflect the uptake of PCBs from site sediments as regulated by their concentration and bioavailability in the sediment. Results for the TPCB summation at reference and SUBASE stations are given below. The complete set of data can be found in Appendix B.

8.4.1 Control

PCB results for the control samples are shown in Table 8-6 and Table 8-7. Three composite control samples were analyzed. Each control sample was composited from clams in five separate exposure chambers containing home sediment. PCB concentrations in the control tissues generally were low. Variation among the control replicates was low indicating consistency among the exposures and analytical procedures. For example, TPCBs in the control sample tissues ranged from 11 to 13 μg/kg, with an RSD of only 9%.

8.4.2 Reference Stations

PCB results for the reference stations are shown in Table 8-6 and Table 8-7. The tissue concentration for each reference station represents the composite of five replicate laboratory exposures. In addition, for station SB2433, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported PCB values for station SB2433 are thus the means of these three field replicates. TPCB concentrations at the reference stations ranged from 27.5 to 73.6 μg/kg, with an RSD of 32%. Reference station-mean tissue concentrations of PCBs generally were higher than the control samples (4.6X). These results indicate that reference areas of San Diego Bay have higher bioaccumulation potential for PCBs compared to the control sediments.

8.4.3 SUBASE Stations

PCB results for the SUBASE stations are shown in Table 8-6 and Table 8-7. The tissue concentration for each station represents the composite of five replicate laboratory exposures. In addition, for station SB9, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported PCB values for station SB9 are thus the means of these three field replicates. The range of concentrations across the SUBASE stations was comparable to the reference stations, with higher station to station variability, but greater than control samples. For example, TPCB concentrations ranged from 21.5 to 80.6 μg/kg, with an RSD of 49%. Among the SUBASE stations, SB9 had the highest concentration of TPCBs, while station SB11 had the lowest level. Overall, the results indicate that SUBASE stations generally have similar bioaccumulation potential for PCBs compared to the reference stations, but higher bioaccumulation potential than the control sediments.

The spatial distribution of tissue PCBs at the SUBASE site is shown in Figure 8-11 and Figure 8-12. The stations follow a transect which runs either parallel (SB2, SB5, SB9, and SB12), or perpendicular to the shoreline (SB8, SB9, SB10, and SB11). Spatial patterns of TPCBs along the transect parallel to shore increased from SB2 through SB9, falling slightly at SB12. There was a sharp decrease in TPCB levels along the perpendicular to shore transect between the inshore sites and those further out in the bay, with higher levels found inshore.

The correlation (r) between TPCBs in tissue and TPCBs in sediment was examined for the SUBASE stations (Table 8-8). For this analysis, tissue concentrations were normalized to lipid content, and sediment concentrations were normalized to TOC. There was a statistically significant relationship between tissue and sediment TPCB levels, with a correlation coefficient of 0.80.

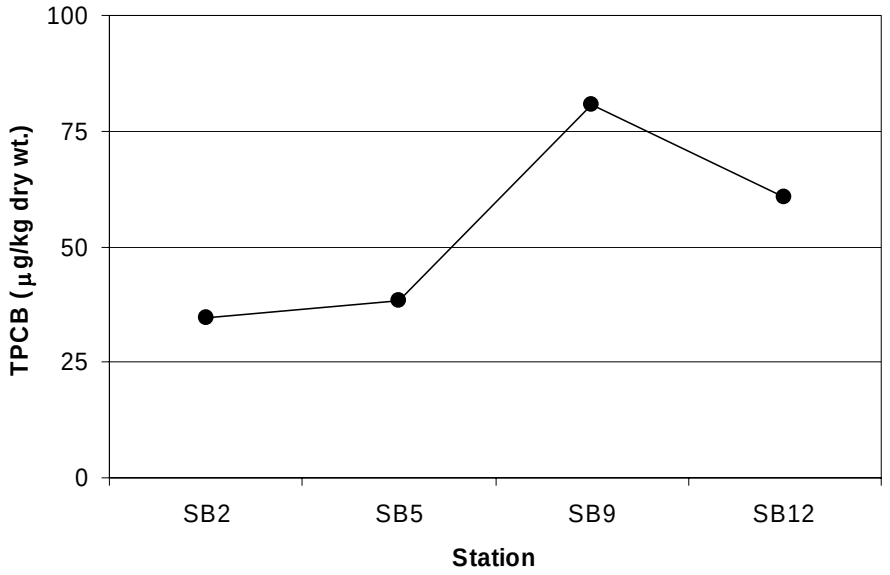

Figure 8-11. Spatial variation of tissue TPCBs along the SUBASE transect running parallel to shore.

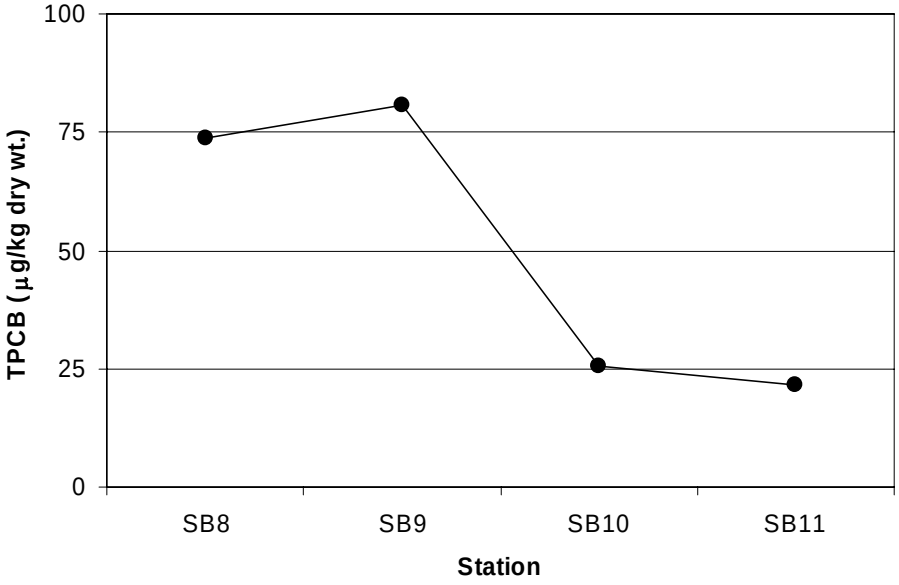

Figure 8-12. Spatial variation of tissue TPCBs along the SUBASE transect running perpendicular to shore.

8.5 PESTICIDES

Concentrations of pesticides in the tissues of clams exposed to site sediments were characterized at all reference and a subset of the SUBASE study stations. Tissue concentrations also were characterized for clams exposed to control (home) sediment. Tissues were analyzed for the same range of pesticides as described previously for the sediment analysis. Study stations that were characterized included SUBASE stations SB2, SB5, SB8, SB9, SB10, SB11, SB12 (Figure 4-1). Tissue concentrations reflect the uptake of pesticides from site sediments as regulated by their concentration and bioavailability in the sediment. Results for the TCHLOR and TDDT summations at reference and SUBASE stations are given below. The complete set of data can be found in Appendix B.

8.5.1 Control

Pesticide results for control samples are shown in Table 8-6 and Table 8-7. Three composite control samples were analyzed. Each control sample was composited from clams in five separate exposure chambers containing home sediment. Pesticide variation among the control replicates was low indicating consistency among the exposures and analytical procedures. For example, TCHLOR in the control sample tissues ranged from 1.2 to 1.5 μg/kg, with an RSD of only 12%, and TDDT ranged from 3.9 to 4.8 μg/kg, with an RSD of only 11%. Thus results from the control samples provide a useful initial baseline for comparison of tissue concentrations from the reference and site stations.

8.5.2 Reference Stations

Pesticide results for the reference stations are shown in Table 8-6 and Table 8-7. The tissue concentration for each reference station represents the composite of five replicate laboratory exposures. In addition, for station SB2433, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported pesticide values for station SB2433 are thus the means of these three field replicates. TCHLOR concentrations at the reference stations ranged from 1.9 to 4.4 μg/kg, with an RSD of 32%. TDDT had a slightly lower range of variation at the reference sites (9.8 to 13.5 μg/kg) with an RSD of 13%. Reference station mean tissue concentrations of pesticides were generally higher than the control samples including TCHLOR (2.5X) and TDDT (2.7X). These results indicate that reference areas of San Diego Bay have higher bioaccumulation potential for pesticides compared to the control sediments.

8.5.3 SUBASE Stations

Pesticide results for the SUBASE stations are shown in Table 8-6 and Table 8-7. The tissue concentration for each station represents the composite of five replicate laboratory exposures. In addition, for station SB9, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported pesticide values for station SB9 are thus the means of these three field replicates. The range of TCHLOR and TDDT concentrations across the SUBASE stations was higher than that for the control samples but comparable to the reference stations, although all RSD values are relatively low. SUBASE station mean results compared to reference showed slightly lower levels for TDDT (0.9X) and TCHLOR (0.6X). Compared to control mean SUBASE pesticide levels were higher for both TCHLOR (1.5X) and TDDT (2.5X). Among the SUBASE stations, SB8 had the highest concentration of TCHLOR and TDDT, while station SB11 had the lowest TCHLOR and TDDT levels. Overall, the results indicate that SUBASE stations generally have higher bioaccumulation potential for TCHLOR and TDDT compared to control sediments, whereas TCHLOR and TDDT showed comparable or lower bioaccumulation potential to the reference stations.

The spatial distribution of tissue pesticides at the SUBASE site is shown in Figure 8-13 and Figure 8-14. The stations follow a transect which runs either parallel (SB2, SB5, SB9, and SB12), or perpendicular to the shoreline (SB8, SB9, SB10, and SB11). Spatial patterns of TCHLOR along both the parallel and perpendicular to shore transects showed relatively little variation. More clearly defined spatial patterns were observed for TDDT. There was an increasing trend from station SB2 to SB12 along the parallel transect, and a decreasing trend moving away from shore along the perpendicular transect. Correlations (r) between pesticides in tissue and pesticides in sediment were examined for the SUBASE stations. For this analysis, tissue concentrations were normalized to lipid content, and sediment concentrations were normalized to TOC. There was a significantly positive relationship between sediment and tissue TCHLOR concentrations (r = 0.85) and TDDT concentrations (r = 0.87) (Table 8-8).

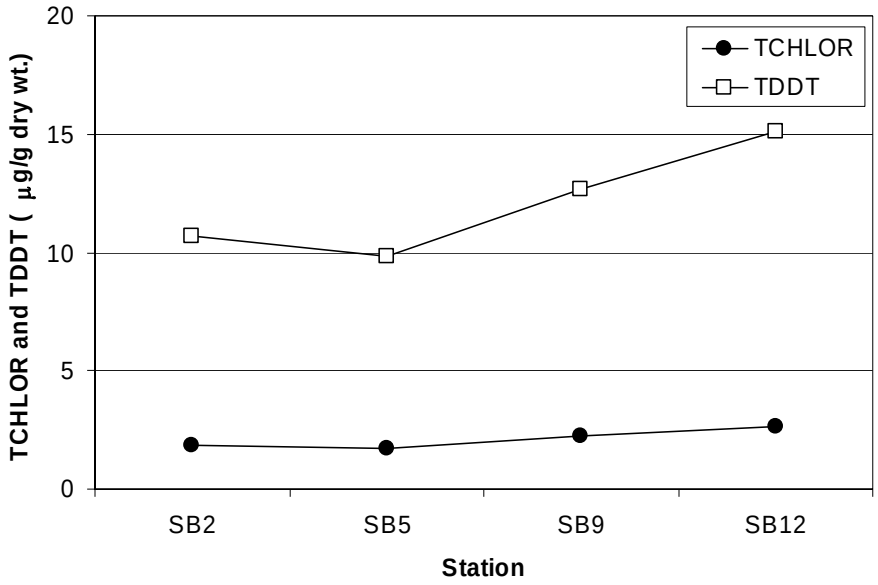

Figure 8-13. Spatial variation of tissue pesticides along the SUBASE transect running parallel to shore.

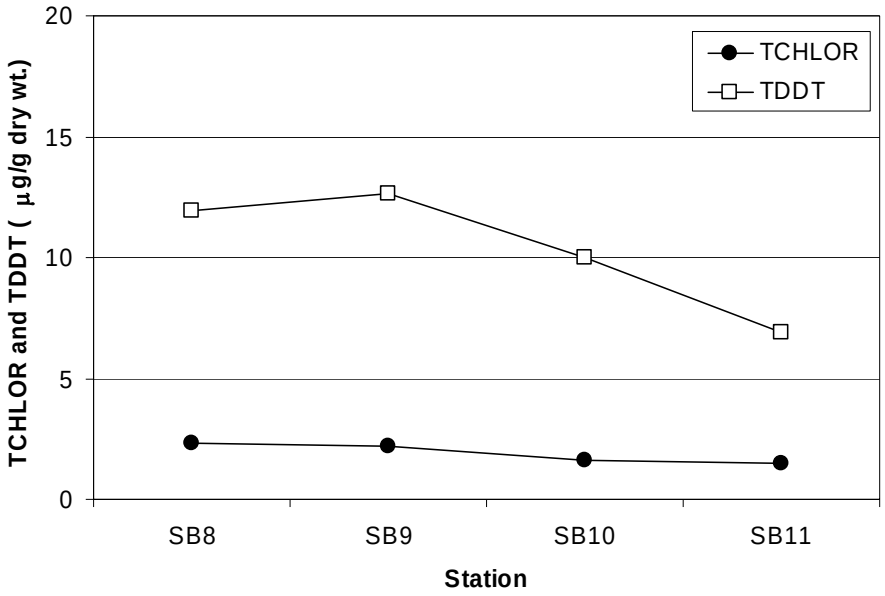

Figure 8-14. Spatial variation of tissue pesticides along the SUBASE transect running perpendicular to shore.

8.6 ORGANOTINS

Concentrations of organotins in the tissues of clams exposed to site sediments were characterized at all reference and a subset of the SUBASE study stations. Tissue concentrations also were characterized for clams exposed to control (home) sediment. Tissues were analyzed for the same range of pesticides as described previously for the sediment analysis. Study stations that were characterized included SUBASE stations SB2, SB5, SB8, SB9, SB10, SB11, SB12 (Figure 4-1). Tissue concentrations reflect the uptake of pesticides from site sediments as regulated by their concentration and bioavailability in the sediment. Four organotin compounds were measured: tetra-n-butyltin (TTBT), tri-n-butyltin (TBT), di-n-butyltin (DBT), and mono-n-butyltin (MBT). Total organotin concentrations (TOT) were determined as the sum of all individual compounds. Results for organotins at reference and SUBASE stations are given below. The complete set of data can be found in Appendix B.

8.6.1 Control

Organotin results for control samples are shown in Table 8-9 and Table 8-10. Three composite control samples were analyzed. Each control sample was composited from clams in five separate exposure chambers containing home sediment. Organotin variation among the control replicates was low indicating consistency among the exposures and analytical procedures. For example, TBT in the control sample tissues ranged from 27.7 to 37.6 µg/kg (dry weight), with an RSD of only 17%, and DBT ranged from 12.4 to 13.8 µg/kg, with an RSD of only 6%. There was slightly higher variation among the TTBT and the MBT fractions. Thus results from the control samples provide a useful initial baseline for comparison of tissue concentrations from the reference and site stations.

8.6.2 Reference Stations

Organotin results for the reference stations are shown in Table 8-9 and Table 8-10. The tissue concentration for each reference station represents the composite of five replicate laboratory exposures. In addition, for station SB2433, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported organotin values for station SB2433 are thus the means of these three field replicates. TBT concentrations at the reference stations ranged from 47.9 to 82.5 μg/kg, with an RSD of 18%. Reference station mean tissue concentrations of organotins were generally higher than the control samples including TBT (2.2X), DBT (2.1X), MBT (1.3X), and TOT (2.1X). These results indicate that reference areas of San Diego Bay have higher bioaccumulation potential for organotins compared to the control sediments.

8.6.3 SUBASE Stations

Organotin results for the SUBASE stations are shown in Table 8-9 and Table 8-10. The tissue concentration for each station represents the composite of five replicate laboratory exposures. In addition, for station SB9, three field replicates were collected and a composite sample was analyzed from five replicate laboratory exposures for each field replicate. The reported organotin values for station SB9 are thus the means of these three field replicates. Organotin variation among the SUBASE stations was similar to control and reference samples, although SUBASE concentrations were slightly elevated. Mean SUBASE TBT values were similar when compared to mean reference (1.1X), and higher than control (2.4X). Among the SUBASE stations, SB11 had the highest concentration of TBT, while station SB8 had the lowest. Overall, the results indicate that SUBASE stations generally have higher bioaccumulation potential for organotin compounds compared to control sediments, whereas organotins showed comparable or lower bioaccumulation potential to the reference stations.

The spatial distribution of tissue pesticides at the SUBASE site is shown in Figure 8-15 andFigure 8-16. The stations follow a transect which runs either parallel (SB2, SB5, SB9, and SB12), or perpendicular to the shoreline (SB8, SB9, SB10, and SB11). TBT concentrations along the parallel transect were similar, with a slight increase at SB5. Spatial patterns of TBT concentrations are more clearly defined along the perpendicular transect, with values increasing moving away from the shoreline from SB8 to SB11.

Correlations (r) between organotins in tissue and organotins in sediment were examined for the SUBASE stations (Table 8-11). For this analysis, tissue concentrations were normalized to lipid content, and sediment concentrations were normalized to TOC. There were several significant positive and negative correlations between sediment and tissue organotin concentrations. The differences in the nature of the relationship (positive or negative) may be related to the volatile nature of the compounds, degrading from TTBT through MBT over time.

Table 8-9. Tissue organotin data from control, reference, and SUBASE stations (μg/kg dry weight).

Area	Station	TTBT	TBT	DBT	MBT	TOT
Control	Control	0.96	33.9	13.7	5.7	54.2
Reference	SB2229	0.67	76.7	32.9	4.1	114
	SB2433	0.69	68.0	27.2	4.2	100
	SB2436	0.64	81.5	29.0	3.8	115
	SB2441	0.65	47.9	15.0	21.6	85.2
	SB90056	0.78	82.5	29.1	4.6	117
	SBC001SS31	0.64	69.2	24.8	3.8	98.3
SUBASE	SB2	0.69	71.4	22.0	4.3	98.4
	SB5	0.61	84.5	18.9	3.8	108
	SB8	0.54	47.3	16.0	3.3	67.1
	SB9	0.68	64.3	21.2	4.1	90.3
	SB10	0.64	72.4	19.6	4.0	96.7
	SB11	0.62	90.7	20.0	3.8	115
	SB12	1.5	75.9	23.1	9.2	110

Table 8-10. Summary statistics for the tissue organotin data (μg/kg dry weight).

Area	Statistic	TTBT	TBT	DBT	MBT	TOT
Control	Minimum	0.72	27.7	12.4	3.8	44.8
	Maximum	1.4	37.6	13.8	7.5	60.3
	Mean	0.96	31.6	12.9	5.1	50.6
	Std Dev	0.41	5.2	0.73	2.1	8.5
	RSD (%)	43%	17%	6%	42%	17%
Reference	Minimum	0.64	47.9	15.0	3.8	85.2
	Maximum	0.78	82.5	32.9	21.6	117
	Mean	0.68	71.0	26.3	7.0	105
	Std Dev	0.05	12.8	6.1	7.1	12.6
	RSD (%)	8%	18%	23%	102%	12%
SUBASE	Minimum	0.54	47.3	16.0	3.3	67.1
	Maximum	1.5	90.7	23.1	9.2	115
	Mean	0.75	72.4	20.1	4.6	97.9
	Std Dev	0.33	14.1	2.33	2.0	16.0
	RSD (%)	44%	19%	12%	44%	16%

Table 8-11. Correlation (r) between organotin concentrations in tissue and sediment for SUBASE bioaccumulation stations. Gray cells indicate statistically significant correlations ($p<0.05$).

		Tissue				
		TTBT	**TBT**	**DBT**	**MBT**	**TOT**
Sediment	**TTBT**	0.16	-0.82	-0.17	0.15	-0.81
	TBT	-0.04	-0.62	-0.06	-0.04	-0.62
	DBT	0.68	-0.82	-0.80	0.68	-0.78
	MBT	0.99	-0.39	-0.54	0.99	-0.31
	TOT	0.82	-0.73	-0.62	0.82	-0.67

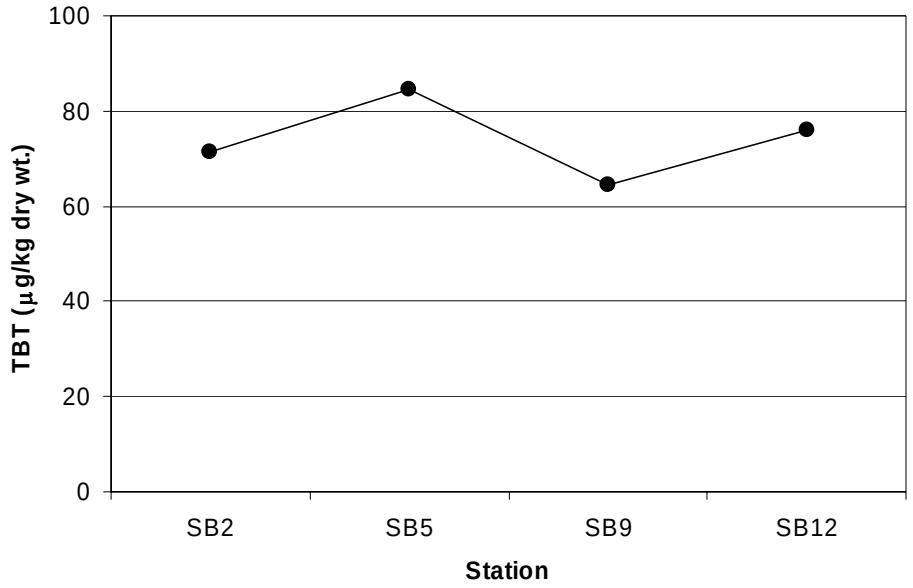

Figure 8-15. Spatial variation of tissue TBT along the SUBASE transect running parallel to shore.

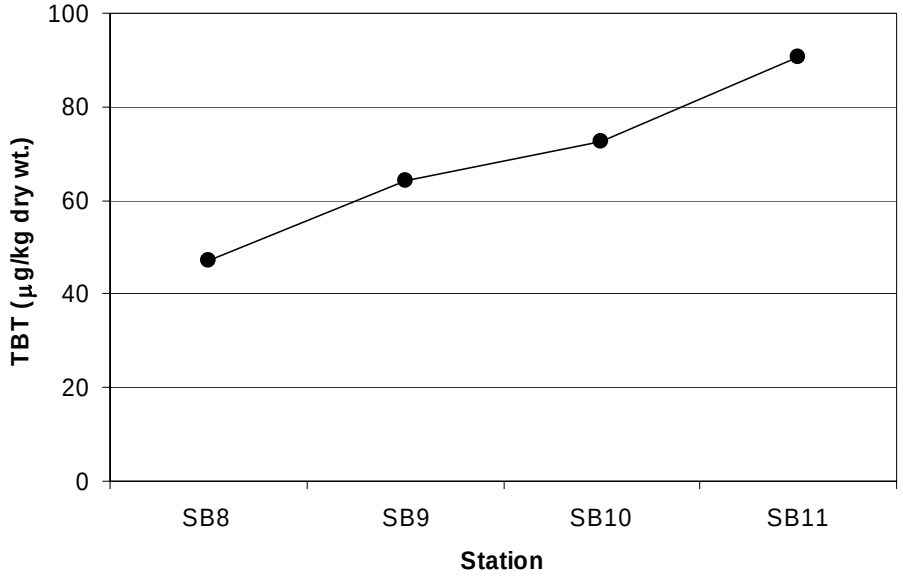

Figure 8-16. Spatial variation of tissue TBT along the SUBASE transect running perpendicular to shore.

9.0 TOXICITY RESULTS

9.1 BULK SEDIMENT TOXICITY

Test samples were classified as toxic if the mean amphipod survival was significantly less than the control ($p \leq 0.05$, t-test) and was also less than the MSD (minimum significant difference) value of 75% of the control. The MSD value was based on analyses conducted by the U.C. Davis Marine Pollution Studies Laboratory (Phillips et al. 2001).

9.1.1 Reference Stations

Sediments from six reference station were collected and tested on two dates. Reference sediments tested on April 27, 2004 were conducted concurrently with SUBASE sediments. Reference sites were also sampled and tested on February 12, 2004 as part of a reconnaissance survey to confirm appropriateness of the selected reference sites.

Amphipod survival in the reference station sediments from the reconnaissance survey (February 12, 2004) ranged from 91-100% of the control mean (Table 9-1). Because of the very high control survival (98%), statistical differences based on t-tests were observed for four of the sediments (SB2229, SB2436, SB2441, and SBC001SS31). However, none of the sediments were considered toxic because survival was always above the MSD threshold (survival >75% of control).

The concentration of unionized ammonia among the six reference sites from the reconnaissance survey ranged from <0.001 – 0.004 mg/L NH_3 in the overlying water, and from <0.001 – 0.028 mg/L NH NH_3 in the porewater (Table 9-2). These concentrations are well below the toxic effects threshold for *E. estuarius* survival (1.15 mg/L NH_3).

Amphipod survival in the reference station sediments sampled as part of the main sampling event (April 2004) ranged from 84-100% of the control mean (Table 9-1). Of the six reference site stations, one station (SB2441) was statistically different from the control, but was not deemed toxic, as survival was not below the MSD threshold ($\geq$75% of control survival) for the amphipod test. The remaining five sites were also not toxic to amphipods, as there were no statistical differences in t-tests, and survival was above the MSD threshold.

The concentration of unionized ammonia among the six reference station ranged from <0.001 - 0.114 mg/L NH_3 in the overlying water, and from <0.001 - 0.136 mg/L NH_3 in the porewater (Table 9-2). These concentrations are well below the toxic effects threshold for *Eohaustorius estuarius* survival (1.15 mg/L NH_3). Therefore, ammonia did not negatively impact the survival of *E. estuarius.*

9.1.2 SUBASE Stations

Amphipod survival in SUBASE station sediments ranged from 79 - 99% of the control mean (Table 9-1). Of the 14 SUBASE stations, three (SB6, SB8, SB14) had sediments that were statistically different by t-tests, but no sites were deemed toxic because in all cases survival exceeded the MSD threshold for *E. estuarius* (75% of control survival) (Figure 9-1).

The concentration of unionized ammonia ranged from <0.001 - 0.315 mg/L NH_3 in the overlying water, and from <0.001 - 0.526 mg/L NH_3 in the porewater (Table 9-2). These concentrations are below the toxic effects threshold for *E. estuarius* survival (1.15 mg/L NH_3). Therefore, ammonia was not a problem in the SUBASE sediment exposures.

9.2 SEDIMENT-WATER INTERFACE TOXICITY

Sediment-water interface (SWI) samples were tested only during the April 2004 sampling event. SWI samples were not evaluated with Reconnaissance sediment. Test samples were classified as toxic if the mean normal survival of mussel larvae was significantly less than the control ($p \leq 0.05$, t-test) and was also less than the MSD (minimum significant difference) value of 80% of the control. The MSD value was based on analyses conducted by the U.C. Davis Marine Pollution Studies Laboratory (Phillips et al. 2001).

9.2.1 Reference Stations

Embryo development in the SWI tests ranged from 86-100% of the control mean (Table 9-1). One of the sediments (SBC0015531) was significantly different (t-test), but was not toxic to the mussel embryos because the MSD criterion was met (>80% of control). All other reference sediments were statistically indistinguishable from the control sediment.

Ammonia was measured in the overlying water from one replicate per site both at the beginning and end of the test. Ammonia increased only minimally in Reference site samples during the test (mean increase of 37%). Overall unionized ammonia concentrations ranged from 0.005 - 0.019 mg/L NH_3 (Table 9-2), which were all well below the 0.073 mg/L NH_3 threshold (based on NOEC) for mussel embryos.

9.2.2 SUBASE Stations

Embryo development in the SWI tests ranged from 79 - 100% of the control mean (Table 9-1, Figure 9-2). One replicate was removed from the calculations for each SB6 and SB11 because they were considered outliers, having normal survival values dramatically lower than the other replicates within the same sample (Appendix C). Four of the stations (SB4, SB6, SB10, SB13) were significantly different (t-test) in comparison to the core tube blank, but only one of the stations (SB13) had results below the MSD threshold value (80% of control). With a mean normal survival of 79% of the control, SB13 was considered toxic.

Ammonia was measured in the overlying water from one replicate per site both at the beginning and end of the test. Although ammonia generally increased in SUBASE samples during the test (mean increase of 84%), the absence of toxic effects associated with ammonia was not surprising considering the relatively low concentrations measured (unionized ammonia ranged from 0.008 to 0.059 mg/L NH_3 (Table 9-2), which were all below the 0.073 mg/L NH_3 threshold (based on NOEC) for mussel embryos. The final unionized ammonia concentration in the one sample deemed toxic (SB13) was only 0.033 mg/L NH3, therefore, ammonia was not a factor in the apparent toxicity associated with this sample.

Table 9-1. Toxicity of Reconnaissance (Recon) site sediments collected in February 2004, and reference site sediments and SUBASE site sediments collected in April 2004, using whole sediment, sediment-water interface, or porewater toxicity tests. * = significantly different by t-test, but ≥ threshold based on MSD from control; ** = Toxic as defined by study (significantly different and <MSD threshold). MSD thresholds were 75% for amphipod survival, 80% for mussel embryo-larval development, and 55% for sea urchin embryo-larval development, all relative to control. ND=no conclusive data due to extreme ammonia influence.

		Amphipod 10 day survival					Mussel Embryo-Larval Development					Sea Urchin Embryo Development				
		Whole Sediment					Sediment-water interface					100% Pore water				
												Uncorrected		NH_3 corrected, % control		
Type	Sample ID	Mean	Mean, Outliers Removed	Std. Dev.	% Control	Sig. Diff. from Control	Mean	Mean, Outliers Removed	Std. Dev.	% Control	Sig. Diff. from Control	Mean	Std. Dev.	Mean	Std. Dev.	Sig. Diff. from Control
Control	Home Sediment[a]	98	98	2.7	100											
	Home Sediment[b]	90	90	7.1	100											
	Core Tube Blank						86	86	6.4	100						
	Seawater Control											74	4.9	100	6.6	
Recon	SB2229-R	89	89	6.8	91	*										
	SB2433-R	96	96	4.2	98											
	SB2436-R	92	92	2.7	94	*										
	SB2441-R	89	89	6.5	91	*										
	SB90056-R	98	98	2.7	100											
	SBC001SS31-R	93	93	2.7	95	*										
Reference	SB2229	90	90	6.0	100		89	89	6	103		55	2.7	100	0.0	
	SB2433	89	89	6.5	99		75	75	6.5	87		1	0.7	74	0.9	*
	SB2436	88	88	9.1	98		86	86	9.1	100		0	0.0	23	0.0	**
	SB2441	76	76	4.2	84	*	74	74	4.2	86		25	13.6	100	3.7	
	SB90056	89	89	8.2	99		86	86	8.2	100		41	7.7	55	10.4	*
	SBC001SS31	87	87	6.7	97		75	75	6.7	87	*	0	0.0	20	0.0	**
SUBASE	SB1	88	88	4.5	98		88	88	1.8	102		4	1.9	26	2.6	**
	SB2	71	81	8.9	90		79	79	9.2	91		19	7.2	82	9.6	*
	SB3	86	86	6.5	96		79	79	16.5	92		20	4.0	68	5.3	*
	SB4	77	77	11.5	86		74	74	6.6	86	*	1	1.3	26	1.8	**
	SB5	82	82	10.4	91		84	84	15.9	98		29	3.5	100	0.0	
	SB6	79	79	6.5	88	*	56	70	6.4	81	*	32	8.9	79	11.9	*
	SB7	86	86	4.2	96		88	88	10	102		65	7.3	87	6.4	
	SB8	79	79	6.5	88	*	74	74	15.6	86		34	10.1	95	9.5	
	SB9	77	77	12.3	86		70	70	15.4	81		1	1.0	ND		
	SB10	89	89	5.5	99		69	69	7.9	80	*	23	5.8	ND		
	SB11	87	87	6.7	97		73	94	12.4	109		70	9.3	93	12.5	
	SB12	84	84	15.2	93		90	90	4.6	104		0	0.4	37	0.5	**
	SB13	77	84	4.8	93		68	68	14.3	79	**	20	9.3	87	11.7	
	SB14	75	75	10.6	83	*	73	73	16.2	85		1	1.4	ND		

[a]Compared to Reconnaissance sediments, initiated on February 12, 2004.

[b]Compared to Reference and SUBASE sediments, initiated on April 27, 2004.

Table 9-2. Concentrations of unionized ammonia (mg/L). Water quality measurements were made on one replicate from each test type. Bolded values indicate exceedance of the toxic effects threshold for the species being tested (threshold for ***E. estuarius*** survival = 1.15 mg/L NH3, ***M. galloprovincialis*** embryo development = 0.073 mg/L NH3, ***S. purpuratus*** embryo development = 0.052 mg/L NH3). Final porewater concentrations are estimates based on initial ammonia measurements and final water quality parameter measurements.

		Whole Sediment				Sediment-Water		Porewater	
Type	Sample	Amphipod 10 day survival				Interface		100%	
		Overlying Water		Pore Water		Mussel 48 h Dev.		Echinoderm 96 h Dev.	
		Initial	Final	Initial	Final	Initial	Final	Initial	Final
Ctrl	Home Sediment[a]	<0.001	0.003	<0.001	0.020	NA	NA	NA	NA
	Home Sediment[b]	0.002	NA	0.003	NA	NA	NA	NA	NA
Recon	SB2229-R	<0.001	0.002	0.010	0.012	NA	NA	NA	NA
	SB2433-R	<0.001	0.003	0.012	0.013	NA	NA	NA	NA
	SB2436-R	<0.001	0.002	0.004	0.009	NA	NA	NA	NA
	SB2441-R	0.001	0.003	0.028	0.008	NA	NA	NA	NA
	SB90056-R	<0.001	0.002	0.015	0.017	NA	NA	NA	NA
	SBC001SS31-R	0.004	0.003	0.013	0.007	NA	NA	NA	NA
Reference	SB2229	0.005	<0.001	0.067	<0.001	0.005	0.005	0.032	**0.074**
	SB2433	0.007	0.012	0.071	0.043	0.007	0.009	0.035	**0.082**
	SB2436	0.005	0.001	0.056	<0.001	0.010	0.009	0.012	**0.066**
	SB2441	0.011	0.069	0.097	0.136	0.007	0.019	0.016	**0.107**
	SB90056	0.006	0.018	0.115	0.038	0.008	0.009	NA	NA
	SBC001SS31	0.007	0.002	0.054	<0.001	0.008	0.010	0.010	**0.058**
SUBASE	SB1	0.008	0.017	0.091	0.025	0.013	0.015	0.007	**0.063**
	SB2	0.011	0.020	0.071	0.039	0.012	0.024	0.013	**0.096**
	SB3	0.014	0.022	0.172	0.049	0.013	0.014	0.014	**0.085**
	SB4	0.012	0.039	0.120	0.147	0.014	0.025	0.008	**0.073**
	SB5	0.016	0.038	0.222	0.131	0.020	0.048	0.014	**0.111**
	SB6	0.014	0.039	0.134	0.039	0.010	0.025	0.012	**0.082**
	SB7	0.007	NA	0.230	0.095	0.015	0.033	0.010	**0.065**
	SB8	0.013	0.077	0.183	0.216	0.014	0.026	0.010	**0.094**
	SB9	0.018	0.202	0.196	0.441	0.038	0.059	0.012	**0.127**
	SB10	0.018	0.315	0.305	0.526	0.026	0.048	0.012	**0.177**
	SB11	0.014	0.204	0.279	NA	0.011	0.036	NA	NA
	SB12	0.007	0.011	0.079	0.015	0.008	0.020	0.007	**0.087**
	SB13	0.010	0.021	0.085	0.082	0.013	0.033	0.012	**0.098**
	SB14	0.010	0.130	0.109	0.220	0.029	0.030	0.026	**0.174**

[a]Control for Reconnaissance samples (February 2004)

[b]Control for Reference and SUBASE samples (April 2004)

NA= not applicable because samples were either not tested or volume was insufficient for measurement

9.3 PORE WATER TOXICITY

Pore water samples were collected for the SUBASE evaluation in April 2004, but not for the Reconnaissance survey in February 2004. Test samples were classified as toxic if mean sea urchin embryo-larval development was significantly less than the control ($p \leq 0.05$, t-test) and was also less than the MSD (minimum significant difference) value of 55% of the control. The MSD value was based on analyses conducted by the U.C. Davis Marine Pollution Studies Laboratory (Phillips et al. 2001).

In the undiluted (100%) porewater, estimated ammonia concentrations for all of both the Reference and SUBASE samples exceeded the threshold for the test endpoint (NOEC = 0.033 mg/L unionized NH3) by the end of the exposure (Table 9-2). In order to better interpret the data for effects caused by factors other than ammonia, a correction for ammonia influence was made for all samples. This correction used the average of the initial and estimated final unionized ammonia concentrations, and followed an approach demonstrated by Bay et al. (1995) in which porewater data were adjusted. The final ammonia values (Table 9-2) are estimates only because ammonia was not measured at test termination. Therefore, total ammonia concentrations measured at test initiation were used in combination with final water quality parameters to estimate final unionized ammonia measurements. Incidentally, the apparent increase in unionized ammonia concentration as the test progressed is largely an artifact of higher pH measured at test termination, and not increases in total ammonia concentrations. Pore water pH typically increases as carbon dioxide is degassed from the sample (Adams et al. 2003), which tends to occur in the test chambers which exchange with the atmosphere, and likely explains the observed change in pH over time in the 100% pore water samples. Consistently higher ammonia concentrations in pore water compared to SWI samples is due to dilution of any ammonia flux from the sediment with the clean overlying seawater above in the SWI exposures.

Based on the approach by Bay et al. (1995), pore water samples with ammonia concentrations between 0.033 and 0.067 mg/L NH_3 that had sea urchin embryo development <80% of the control were adjusted for ammonia influence (SCCWRP and Navy, 2005). Ammonia concentrations >0.067 mg/L NH_3 were believed to be responsible for all of the toxicity in samples that had <80% normal development. No information regarding the toxicity of other constituents could be obtained for these samples, therefore, data for these samples had to be designated as inconclusive (Table 9-1 and Table 9-3).

Because the initial pH of the undiluted pore water samples (range = 6.9-7.7) fell below the targeted range of 7.8-8.2, it is possible that effects may have also been associated with low pH. Bay et al. (2003) reported abnormal larval development of *S. purpuratus* embryos at pH less than 7.4. Although pH rose to acceptable levels during the test, it is not known exactly when this happened due to an inadequate number of measurements to verify this early on in the exposure. The pH values of the 25% dilution of porewater were not measured, however, because they were diluted with negative control sea water (pH 8.10 at test initiation), they were likely at or near acceptable levels. Dilution by a factor of four also would have resulted in ammonia concentrations below the effects threshold for all samples. Therefore, the 25% dilution was used in this study to support the interpretation of the 100% pore water toxicity results, which also might facilitate comparisons with other studies. The sea-urchin fertilization test (USEPA 1995) with *S.*

purpuratus has been used for the assessment of porewater toxicity in recent ecological risk assessments in San Diego Bay (e.g. Anderson et al., 2004; SCCWRP and Navy, 2005). The fertilization test endpoint is less sensitive to both ammonia and a number of metals when compared to the embryo-larval development endpoint (Bay et al. 1993; Table 9-4). Based on EC50s for the analytes listed inTable 9-4, the fertilization test endpoint appears, on average, less sensitive by a factor of 15.5. Therefore, a dilution of 25% for the embryo development test compared to undiluted porewater tested with the fertilization test might be considered a conservative means of making comparisons between the two test methods, and does not compromise the sensitivity of the pore water test.

Table 9-3. Comparison of porewater toxicity test results for undiluted (100%) porewater which underwent a correction for ammonia influence, and the 25% dilution, which did not require correction for ammonia. * = significantly different from t-test, but ≥ threshold based on MSD from control; ** = Toxic as defined by this study (significantly different and <MSD threshold). MSD threshold for porewater sea urchin embryo development is 55% of control. NT=not tested due to insufficient sample. ND=no useable data due to exceedance of extreme ammonia effects threshold.

		Sea Urchin Embryo-Larval Development **100% Porewater (With Ammonia Correction)**					Sea Urchin Embryo-Larval Development **25% Porewater (No Ammonia Correction)**			
		Prior to Correction		With Ammonia Correction						
Area	Sample	Mean	Std. Dev.	Mean, % Control	Std. Dev.	Sig. Diff. from Control	Mean	Std. Dev.	Mean, % Control	Sig. Diff. from Control
Ctrl	Seawater Control	74	4.9	100	6.6		74	4.9	100	
Reference	SB2229	55	2.7	100	0.0		59	4.2	79	*
	SB2433	1	0.7	74	0.9	*	81	23.3	108	
	SB2436	0	0.0	23	0.0	**	21	26.4	28	**
	SB2441	25	13.6	100	3.7		57	19.7	77	
	SB90056	41	7.7	55	10.4	*	NT	NT	NT	
	SBC001SS31	0	0.0	20	0.0	**	4	6.0	5	**
SUBASE	SB1	4	1.9	26	2.6	**	34	15.7	46	**
	SB2	19	7.2	82	9.6	*	18	15.5	24	**
	SB3	20	4.0	68	5.3	*	65	10.7	87	
	SB4	1	1.3	26	1.8	**	28	8.2	37	**
	SB5	29	3.5	100	0.0		64	11.2	86	
	SB6	32	8.9	79	11.9	*	31	7.0	42	**
	SB7	65	7.3	87	6.4		66	7.0	89	
	SB8	34	10.1	95	9.5		62	6.3	84	*
	SB9	1	1.0	ND			61	5.0	83	*
	SB10	23	5.8	ND			73	8.7	99	
	SB11	70	9.3	93	12.5		79	11.2	106	
	SB12	0	0.4	37	0.5	**	75	9.8	100	
	SB13	20	9.3	87	11.7		71	2.4	95	
	SB14	1	1.4	ND			63	8.3	85	*

NT=not tested

ND= no useable data. Ammonia concentrations were above the extreme effects threshold, preventing ammonia correction

Table 9-4**.** Comparison of sensitivity of sperm cell (fertilization) and embryo development endpoints for the purple sea urchin (*Strongylocentrotus purpuratus*).

Toxicant	EC50 (mg/L)		Sensitivity Ratio[1]
	Sperm	Embryo	
Unionized ammonia	>1.4	0.072	19.4
Cadmium	18.4	0.51	36.1
Copper	0.025 0.018 0.031	0.006 0.011	2.9
Lead	8.2	<9.7	ND
Silver	0.115	0.015	7.7
Zinc	0.262	0.023	11.4

EC50 data summarized by Bay et al. (1993)

[1] The sensitivity ratio is a measure of the difference in sensitivity between Sperm and Embryo endpoints.

ND=not determined because EC50 was an undetermined value less than that shown

9.3.1 Reference Stations

Sea urchin embryo-larval development in 100% porewater ranged from 20 to 100% of the control mean, after the ammonia correction (Table 9-1 and Table 9-3). The undiluted porewater from stations SB2433, SB2436, SB90056, and SBC001SS31 were all significantly different from the control, but only SB2436 and SBC001SS31 were deemed toxic under the MSD criterion (toxic if < 55% of control). Therefore, pore water from the remaining four reference sites was not toxic, as defined by this study. It should be noted that no ammonia correction could be performed on reference sample SB90056, as ammonia was not measured at all for this sample due to insufficient porewater volume. However, because uncorrected larval development for this sample is above the MSD threshold, this sample is non-toxic.

For comparison, toxicity in the undiluted porewater was compared to the 25% dilution where ammonia influence, and presumably, low pH toxicity, were not observed. In the 25% dilution, sea urchin embryo-larval development ranged from 5 to 100% of the control mean (Table 9-3). Significant differences based on t-tests were observed for samples SB2229, SB2436, and SBC001SS31. There was not enough sample to test the 25% dilution of SB90056. According to the MSD threshold for the test method, SB2436 and SBC001SS31 were considered toxic. This finding is consistent with the ammonia-corrected values used in 100% porewater, except that SB2433 was not toxic in the more diluted sample.

9.3.2 SUBASE Stations

After correction for ammonia influence, larval development in the undiluted porewater ranged from 26-100% of the control (Table 9-1 and Table 9-3, Figure 9-3). Statistical differences (by t-tests) from the control were observed in 6 of the 14 SUBASE samples evaluated, including SB1, SB2, SB3, SB4, SB6, and SB12 (Table 9-1 and Table 9-3). Of these, only SB1, SB4, and SB12 were deemed toxic, where mean normal development was below the MSD criterion (<55% of control) for the test method.

In comparison, larval development in the 25% dilution ranged from 24-106% of the control (Table 9-3, Figure 9-4). Statistical differences (based on t-tests) were observed in 7 of the 14 samples (Table 9-3). For the most part, these overlapped the undiluted porewater samples in which there was toxicity, and included SB1, SB2, SB4, SB6, SB8, SB9, and SB14. Of these, only SB1, SB2, SB4, and SB6 were classified as toxic in the 25% dilution. Therefore, declaration of toxicity in SB1 and SB4 was consistent for both dilutions. SB12, however, was not toxic at the lower dilution, which is understandable. The determination of toxicity for SB6 at 25%, but not at the 100% dilution, however, is not clear.

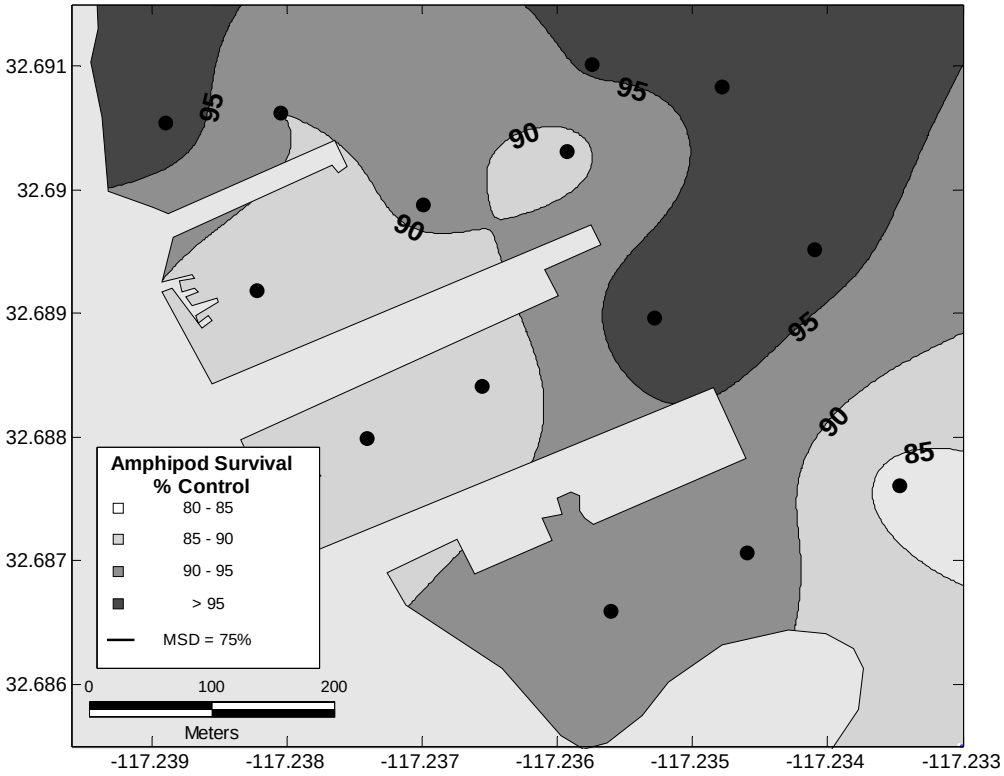

Figure 9-1. Spatial distribution of amphipod survival in SUBASE site sediments.

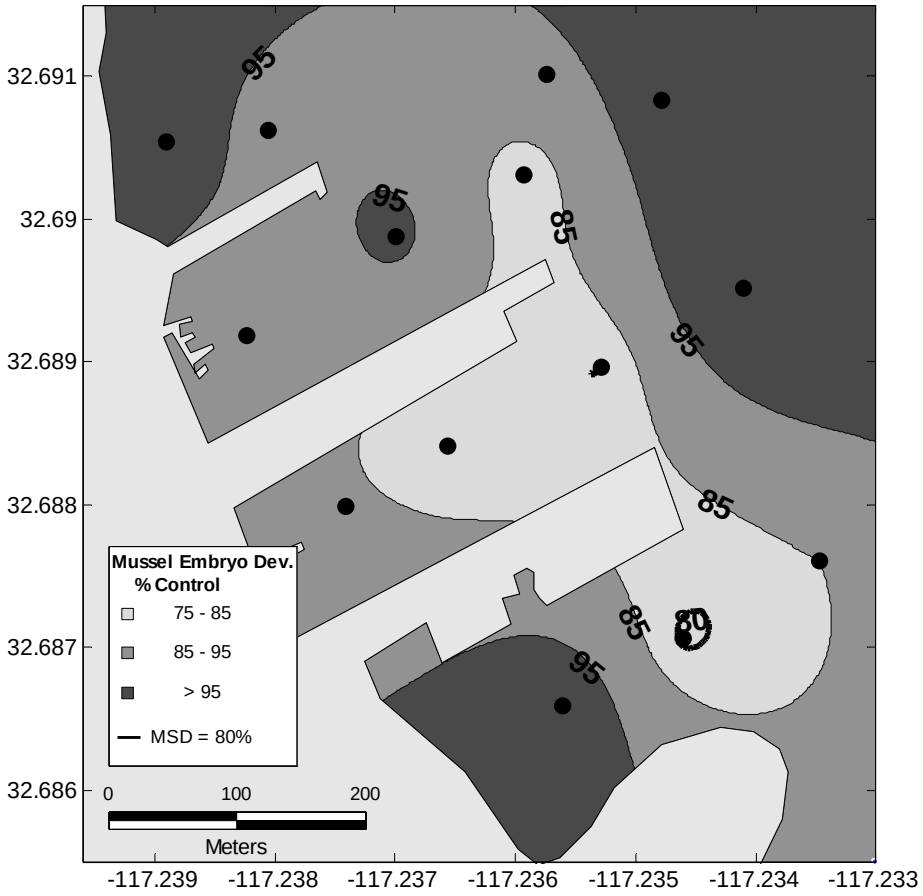

Figure 9-2. Spatial distribution of mussel embryo-larval development success in SUBASE site sediment-water interface samples.

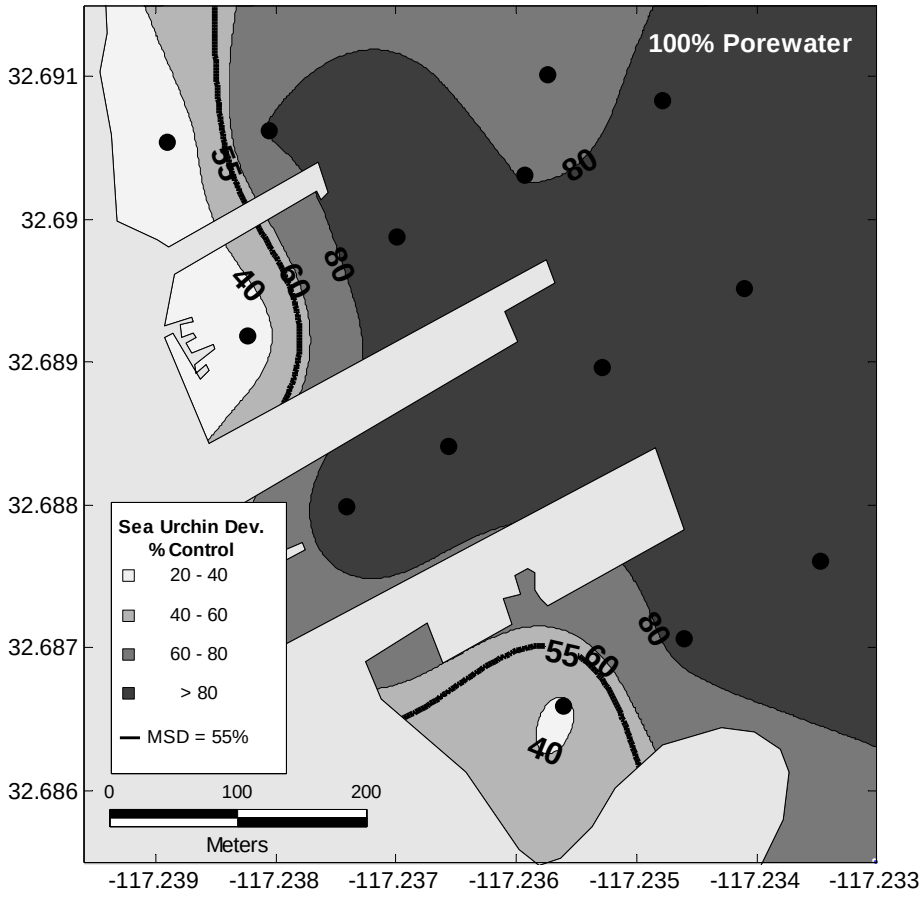

Figure 9-3. Spatial distribution of sea urchin embryo-larval development in undiluted (100%) SUBASE site sediment porewater.

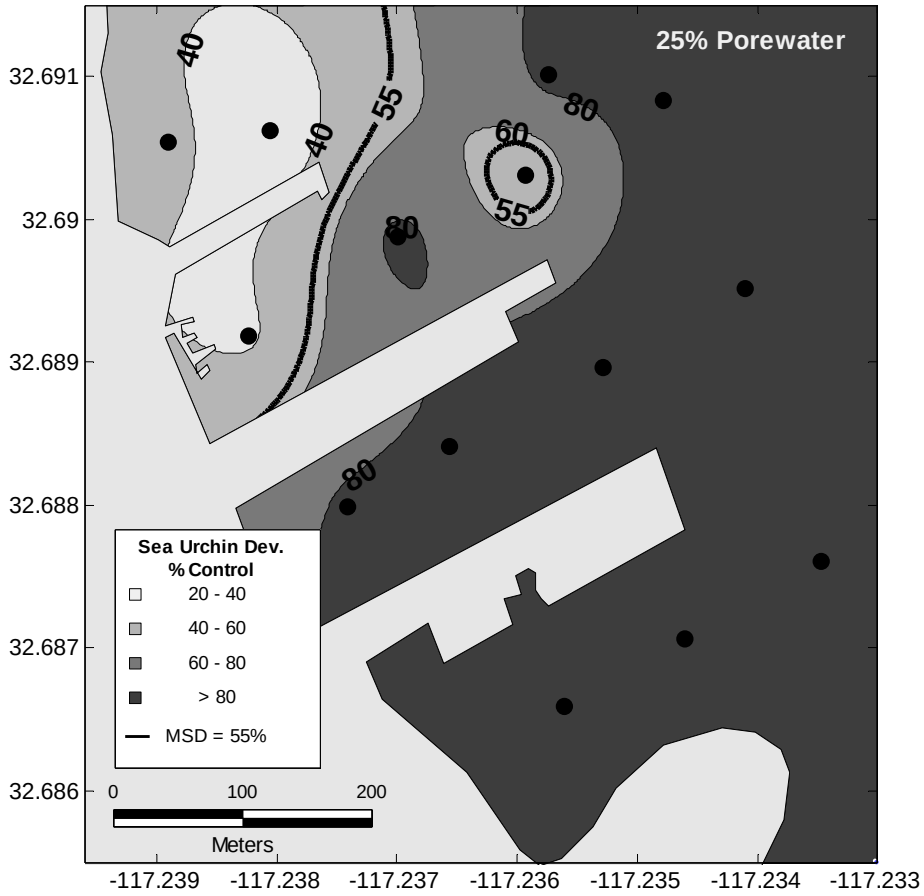

Figure 9-4. Spatial distribution of sea urchin embryo-larval development in the 25% dilution of SUBASE site sediment porewater.

9.4 TOXICITY-CHEMISTRY RELATIONSHIPS

There was no bulk sediment toxicity identified at any station. The sediment-water interface test showed a toxic result for SB13. An evaluation was conducted to evaluate if this toxicity could be attributed to any one chemical or group of chemicals using a sum of metals, the ERMq, and SQGQ1. Station SB13 had one of the highest TOC levels but the only chemical showing a potential relationship to this outcome was arsenic. The level at this station was about 36% higher than the average value observed at all other stations. While arsenic might have played a role in this specific toxicity result, arsenic was higher at SB2441 without a toxic impact. The fact that the toxicity result was only 1% below the MSD level used to declare the sample as toxic suggests that a causal relationship was relatively weak.

The pore water test showed a toxic result at three site stations: SB1, SB4, SB12 and at two reference stations SB2436, and SBC001S31. SB1, SB4, and SB12 were all located nearest to the shoreline on their respective station transects. There was no spatial relationship between the other two reference stations. A comparison of average

chemical data at the three SUBASE site stations with the average at all other sites indicated a nearly three-fold increase in PAHs, a two-fold increase in PCBs, and an increase of ~75% in DDT and TBT concentrations at the three shoreline stations. Station SB8 also had comparable chemistry results but no toxic outcome. These chemicals, highest on the innermost shoreline stations, likely had a causal relationship to the pore water toxicity result. A multiple regression analysis performed by Aquatic Bioassay and Consulting Laboratories of Ventura, CA (see Appendix D) identified a best fit model that related pore water toxicity results to primary factors that included: fines, a general metals group (Al, Pb, Mn, Hg, Ag), PCBs (CI2-CII4), and dieldrin. This independent evaluation indicates that factors other than PAH, DDT, and TBT might have a causal relationship to pore water toxicity.

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10.0 BENTHIC COMMUNITY ANALYSIS

Aquatic Bioassay and Consulting Laboratories of Ventura, CA helped in collecting samples and performed the benthic community evaluation. Their complete report can be found in Appendix D. The following sections highlight their findings.

10.1 COMMUNITY MEASURES

Four key community measures were chosen to evaluate benthic community health, the same four used in recent San Diego Bay sediment investigations. These included abundance, number of taxa, Shannon-Weiner (S-W) Diversity, and Benthic Response Index. The simplest measure of population composition is the total numbers of organisms (abundance) collected per sampling effort. Another simple measure of population health is the number of separate infauna species collected per sampling effort, or species richness. In general, stations with higher numbers of species per grab tend to be in areas of healthier communities or in areas of mild enrichment. The Shannon-Wiener diversity index tends to emphasize the equitability of the species distribution in a community. For example, two samples with the same numbers of species and total individuals may have have differences in how those abundances are distributed among species (e.g. high abundance concentrated into few species vs. even abundance distributed evenly among species). The S-W factors in these two metrics into a single index. S-W expects to see an initial increase at slightly enriched locations, then a decrease in the index value with increasing environmental impact. BRI measures the condition of a benthic assemblage with defined thresholds for levels of environmental disturbance (Smith et al. 2001, Ranasinghe et al., 2003). The pollution tolerance of each species is assigned based upon its distribution of abundance along a pre-established environmental gradient.

10.1.1 Reference Stations

Reference station benthic community metrics are shown in Table 10-1. Reference Station abundance ranged from 495 to 994 and averaged 700. The largest abundance was found at station SB2441 and the lowest at station SB2229. The number of taxa varied relatively little, ranging from 57 to 67 and averaging 62. The Shannon-Wiener diversity index ranged from 2.69 to 3.34 and averaged 2.94. The BRI values ranged from 10.6 to 27.5, all falling within a level 1, or reference level. The largest BRI value was found at station SB2441 and the lowest at station SB90056.

10.1.2 SUBASE Stations

SUBASE station benthic community metrics are shown in Table 10-1. SUBASE station abundance was generally higher and more variable than observed at reference stations, ranging from 532 to 5895 and averaging 2173. The largest abundance was found at station SB14, the innermost station along the southern line of stations. The lowest abundance was found at station SB12, the outermost station along the southern line of stations. The number of taxa was also more variable and higher at SUBASE stations relative to reference stations. Values ranged from 62 to 130 and averaged 98. The Shannon-Wiener diversity index was slightly more variable but comparable in value to values measured at reference stations. These ranged from 1.87 to 3.44 and averaged

2.79. BRI values were also comparable to those measured at reference stations, though slightly higher. Values ranged from 12.9 to 34.4 and averaged 23.3. All but two stations had BRI values falling within a level 1, or reference level. BRI values at SB2 on the north side of the base and at SB14 on the south side of the base were level II, or marginal level.

Table 10-1. Community measure parameters measured at Reference and SUBASE stations.

Site	STATION	Abundance	Number Taxa	Shannon-Wiener Diversity	BRI	BRI Level	Cluster Group
Reference	SB2229	510	62	3.10	16.0	1	1
	SB2433	906	63	2.86	18.5	1	1
	SB2436	682	57	2.69	21.6	1	1
	SB2441	994	67	2.78	27.5	1	2
	SB90056	495	66	3.34	10.6	1	1
	SBC001SS31	611	55	2.85	19.2	1	1
SUBASE	SB1	1018	103	3.22	12.9	1	3
	SB2	1111	71	2.72	31.3	2	2
	SB3	2086	129	2.85	17.9	1	3
	SB4	1650	89	2.95	24.1	1	3
	SB5	2386	77	2.65	26.7	1	3
	SB6	1547	104	2.82	23.4	1	3
	SB7	2549	112	2.48	20.2	1	3
	SB8	2227	96	2.86	23.9	1	3
	SB9	3189	85	2.72	26.4	1	4
	SB10	4569	129	2.61	23.3	1	4
	SB11	1032	130	3.44	14.6	1	4
	SB12	532	62	2.78	26.0	1	2
	SB13	636	82	3.15	21.5	1	2
	SB14	5895	108	1.87	34.4	2	4

10.2 SPECIES ASSEMBLAGES

10.2.1 Cluster Analysis

Cluster analysis was used to determine if there were distinct assemblages of species among stations. All reference and SUBASE stations were analyzed as single group to identify patterns both within and among the study areas. A total of 168 taxa were identified from all stations. Four station cluster groups (1 through 4) and nine species cluster groups (A through I) were identified based on Bray-Curtis dissimilarities and ordination space distances and are identified in Figure 10-1. The station cluster groups were identified by geographical location as shown in Figure 10-2.

Station group 1 included five outer harbor reference stations SB2229, SB2433, SB2436, SB90056 and SBC0011SS31 (Figure 10-2). Reference station SB2441 did not cluster with the other reference stations, and instead clustered with nearby station SB2 in group 2. The average depth of group 1 stations (10 m) was slightly shallower than the other three cluster groups, which ranged in depth from 11 to 12.5 m. The station group 1 cluster was represented by 88 taxa from the total of 168 possible taxa in the species

cluster groups (Figure 10-1). The species groups that best represented this station group included Groups A, B, E, and F (Figure 10-1). Species that occurred with the highest relative abundances in cluster group 1 included the polychaetes *Fabricinuda limnicola*, *Nephtys caecoides*, *Diplocirrus sp SD1*, *Euchone limnicola*, and *Microspio pigmentata*; the crustaceans *Heterophoxus ellisi* and *Scleroplax granulata*; and, the mollusk *Lyonsia californica* (Table 10-2). Previous studies found that *Heterophoxus ellisi* is characteristic of sediments containing moderate organic enrichment (Table 10-3).

Station group 2 included two stations (SB12 and SB13) located south of Sierra Pier and two stations (SB2 and SB2441) located on the north end of the base (Figure 10-2). The average depth of these stations (12.5 m) was the deepest of all the cluster groups. The station group 2 cluster was represented by 98 taxa from the total of 168 possible taxa in the species cluster groups (Figure 10-1). The species groups that best represented this station group included Groups B, C, D and F (Figure 10-2). Several species that occurred with the highest relative abundances in cluster group 2 included the polychaetes *Amphicteis scaphobranchiata, Chaetozone corona, Cossura* sp *A, Pista percyi* and *Scoletoma* sp *C*; the mollusk *Theora lubrica;* and, the echinoderm *Amphipholis squamata* (Table 10-2). Scoletoma sp C (= Lumbrineris spp. in previous studies) is characteristically found in sediments with low organic enrichment (Table 10-3).

Station group 3, included seven stations located mostly on the northern half of the base (Figure 10-2). The average depth of these stations was 11.2 m. The station group 3 cluster was represented by 155 taxa from the total of 168 possible taxa in the species cluster groups (Figure 10-1). The species that best represented this station group included Groups B, C, D, E, F and H (Figure 10-1). Several species that occurred with the highest relative abundances in cluster group 3 included the polychaetes *Leitoscolopios pugettensis* and *Scoletoma* sp *B;* and, the mollusks *Caecum californicum and Acteocina harpa* (Table 10-2). Both *Leitoscoloplos pugettensis* and *Scoletoma* sp. *B* (= *Lumbrineris* spp. in previous studies) are characteristically found in sediments with low organic enrichment (Table 10-3).

Station group 4, included four stations located mostly on the southern half of the base (Figure 10-2). The average depth of these stations was 12 m. The station group 4 cluster was represented by 132 taxa from the total of 168 possible taxa in the species cluster groups (Figure 10-1). The species that best represented this station group included Groups E, F, H and I (Figure 10-1). Several species that occurred with the highest relative abundances in cluster group 4 included the polychaetes *Spiophanes duplex, Mediomastus* sp, *Euclymeninae* sp *A, Prionospio heterobranchia, Dorvillea longicornis, Exogone lourei, Armandia brevis, Micropodarke dubia,* and *Scyphoproctus oculatus;* the crustaceans *Listriella melanica* and *Caprella californica*; the mollusk *Tagelus subteres* and, *Edwardsia californica cmplx (*Table 10-2). Four of these (*Spiophanes duplex, Mediomastus* sp, *Prionospio heterobranchia, Exogone lourei*) are characteristic of low to moderate organic enrichment, while *Dorvillea longicornis* and *Armandia brevis* are characteristic of polluted conditions (Table 10-3).

10.2.2 Indicator Species

Abundance of selected indicator species is a commonly used benchmark for the health of the infaunal community at a given location. Table 10-4 presents the abundances of four indicator species found at the reference and SUBASE stations: *Amphiodia* sp*., Amphiodia urtica, Euphilomedes carcharodonta* and *Capitella capitata complex.* The brittle star, Amphiodia sp., is considered to be an indicator of reference conditions (Word 1978 and Thompson 1982). *Amphiodia* sp. and *A. urtica* were present in low numbers at four of the reference stations. *Euphilomedes carcharodonta* also was found at three of the reference stations in low abundance. This species has been associated with areas of low organic enrichment and commonly is found in harbor environments (Word 1978 and Thompson 1982). *Capitella capitata complex,* which generally is found in higher abundance in areas with a high degree of organic contamination, was not observed at any reference stations.

Of the four species previously described as indicator species, only the ostracod, *Euphilomedes carcharodonta*, occurred in relatively high abundance at all but one of the SUBASE stations (Table 10-4). This species has been associated with areas of low organic enrichment (Word 1978 and Thompson 1982). *Capitella capitata complex,* which generally is found in areas highly enriched with organic contamination, occurred in low abundance at three SUBASE stations (SB1, SB3, and SB14). *Amphiodia* sp. and/or *urtica* were present at all but three SUBASE stations; where *Amphiodia* sp. is considered to be an indicator of background or reference conditions (Word 1978 and Thompson 1982).

Table 10-2. Total abundance of 30 most common reference species found at reference stations SB2229, SB2433, SB2436, SB90056, and SBC001SS31 which clustered into station group 1.

SPECIES	SB2229	SB2433	SB2436	SB90056	SBC001SS31	SB2441	SB1	SB2	SB3	SB4	SB5	SB6	SB7	SB8	SB9	SB10	SB11	SB12	SB13	SB14
Diplocirrus sp SD1	52	151	97	49	124	9	8	1	24	14	5	4	5	11	2	2	0	19	13	0
Amphideutopus oculatus	26	150	57	45	77	17	13	13	21	12	17	19	73	5	12	44	10	1	3	9
Mediomastus sp	61	53	167	27	24	58	195	88	732	304	451	479	930	260	435	1238	262	89	101	1045
Scoletoma sp C	56	85	57	23	41	138	21	122	187	126	56	80	113	112	106	90	16	90	99	88
Leitoscoloplos pugettensis	25	48	12	24	30	105	89	130	79	144	68	43	59	126	22	39	2	36	53	10
Spiophanes duplex	17	54	24	12	12	70	5	28	20	135	370	126	84	359	301	310	22	10	6	75
Scleroplax granulata	52	8	20	22	8	0	8	0	0	0	0	0	0	0	0	1	0	9	5	0
Theora lubrica	5	24	18	1	28	17	14	12	9	17	2	2	4	7	2	1	7	8	49	0
Rudilemboides stenopropodus	2	35	11	3	12	1	0	0	3	2	2	3	6	1	4	7	0	0	0	52
Heterophoxus ellisi	7	22	11	6	16	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Euclymeninae sp A	0	21	6	15	9	17	19	16	58	27	132	75	99	102	140	152	33	2	0	85
Exogone lourei	7	3	1	20	1	10	87	6	190	25	71	104	96	62	154	647	88	4	7	189
Lyonsia californica	0	12	2	16	2	1	6	1	6	4	1	1	1	2	0	4	0	0	0	0
Acteocina harpa	1	10	0	17	1	0	81	0	102	7	5	8	0	31	43	37	20	0	0	3
Cossura sp A	0	10	4	1	14	9	1	84	1	74	2	7	9	9	4	1	0	57	0	2
Prionospio (prionospio) heterobranchia	13	2	9	3	0	2	20	20	17	88	81	20	17	181	267	207	31	4	3	146
Fabricinuda limnicola	16	0	4	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ampelisca cristata microdentata	2	7	6	3	5	5	2	9	5	2	0	5	11	3	0	2	0	1	3	0
Ampelisca cristata cristata	1	8	3	7	3	10	8	22	4	16	2	2	18	9	5	5	3	3	1	0
Euchone limnicola	0	0	0	0	20	1	1	2	1	1	0	0	0	1	0	0	0	0	1	0
Nephtys cornuta	3	5	4	0	7	4	8	1	8	4	0	1	4	0	0	0	0	0	2	0
Scoletoma sp B	1	5	1	6	6	7	1	23	12	21	41	0	4	5	14	7	0	3	9	9
Nephtys caecoides	2	2	1	9	4	0	6	1	5	0	0	0	1	0	0	0	0	4	0	0
Nereis procera	0	9	3	3	0	11	1	11	6	1	10	0	30	2	4	9	0	0	8	4
Corymorpha bigelowi	0	4	8	1	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Ampelisca brachycladus	1	8	1	1	2	0	0	9	0	4	3	1	4	1	0	0	0	0	2	7
Chaetozone corona	0	6	0	4	3	1	1	5	1	1	0	2	2	1	1	0	0	1	9	0
Goniada littorea	3	0	4	6	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Microspio pigmentata	3	5	0	1	4	6	1	1	0	3	0	3	0	2	0	0	0	2	3	0
Scoletoma sp A	0	9	0	1	3	2	1	8	9	1	14	0	0	0	1	0	0	0	0	0

Table 10-3. Benthic infauna species reported to be representative of background, organically enriched (transitional), and polluted (contaminated) habitats.

Background	Organically Enriched		Polluted
	Low Enrichment	Moderate Enrichment	
Ampelisca spp.	*Anaitides* spp.	*Bittium* spp.	*Armandia bioculata* [3]
Amphiodia spp. [3,4]	*Axinopsida serricata* [3,4]	*Boccardia proboscidea* [5]	*Capitella capitata* [1,2,3,4]
Cossura candida [1]	*Cerianthus* spp.	*Cirriformia* luxuriosa [1,2]	Dorvilleidae [2,3,4]
Heterophoxus oculatus [3]	*Chloeia pinnata* [4]	*Eteone* spp.	*Nereis procera* [4]
Maldane sarsi [3]	*Corophium acherusicum* [2]	*Exogone lourei* [5]	*Notomastus* sp. [2,4]
Metaphoxus, Paraphoxus [3]	*Eumida sanguinea* [2]	*Heteromastus filiformis*	Oligochaeta [2]
Nereis procera [1]	*Euphilomedes* spp. [3,4]	*Macoma carlottensis, nasuta* [2,3]	*Ophryotrocha* spp.
Pectinaria californiensis [3]	*Glycinde picta* [2]	*Nereis diversicolor* [2]	Rochefortia (= *Mysella*) pedroana 4
Phoronis spp. [3,4]	*Goniada maculata* [2]	*Nereis grubei* [5]	*Schistomeringos longicornis* [2,3,4]
Spiophanes missionensis [4]	*Hetreophoxus oculatus* [4]	*Ophiodromus puggetensis* [2]	*Solemya* spp. [2,3]
Stenenelenella uniformis [3]	*Leitoscoloplos (=Haploscoloplos*)	*Parvilucina tenuisculpta* [3,4]	Stenothoidae amphipods [3]
Tharyx ? parvus [1]	*Lumbrineris* spp.	*Polydora ciliata, ligni*	*Tharyx* spp.
	Mediomastus spp. [3,4]	*Pseudopolydora paucibranchiata* [1,2]	
	Neanthes spp.	*Schistomeringos longicornis* [1]	
	Nephtys cornuta [2]	*Scololepis fulginosa* [2]	
	Photis spp.	*Spiochaetopterus costarum* [3,4]	
	Paraprionospio (= *Prionospio*) *pinnata* [2]	*Streblospio benedicti* [2]	
	Prionospio lighti (cirrifera) , *heterobranchia, steenstrupi* [2,4]	*Tharyx* spp.	
	Pygospio elegans [2]	*Thyasira flexuosa* [2]	
	Rochefortia (= *Mysella*) *pedroana, tumida* [3]		
	Scoloplos armiger		
	Tharyx spp.		

Notes: (1) Species reported by Pearson and Rosenberg were assigned based on review of their comments. Species reported as "transitional" by Thompson were assigned based on consistency with other reports.

(2) Species in more than one category were considered transitional.

Sources: 1 Reish 1959, 2 Pearson and Rosenberg 1978, 3 Word 1978, 4 Thompson 1982, 5 Dorsey et al. 1983.

Table 10-4. Abundances of Indicator species at Reference and SUBASE stations. Values are number of organisms per grab.

Area	Station	Cluster Group	*Amphioda* sp.	*Amphioda urtica*	*Euphilomedes carcharodonta*	*Capitella capitata complex*
Reference	SB2229	1	0	0	0	0
	SB2433	1	0	0	0	0
	SB2436	1	1	0	0	0
	SB2441	2	0	11	2	0
	SB90056	1	1	0	3	0
	SBC001SS31	1	1	0	2	0
SUBASE	SB1	3	1	0	43	1
	SB2	2	1	0	28	0
	SB3	3	5	0	64	1
	SB4	3	0	2	163	0
	SB5	3	0	0	19	0
	SB6	3	2	0	12	0
	SB7	3	0	0	23	0
	SB8	3	0	0	152	0
	SB9	4	0	12	146	0
	SB10	4	6	0	45	0
	SB11	4	0	1	14	0
	SB12	2	0	7	0	0
	SB13	2	0	6	17	0
	SB14	4	1	0	40	3

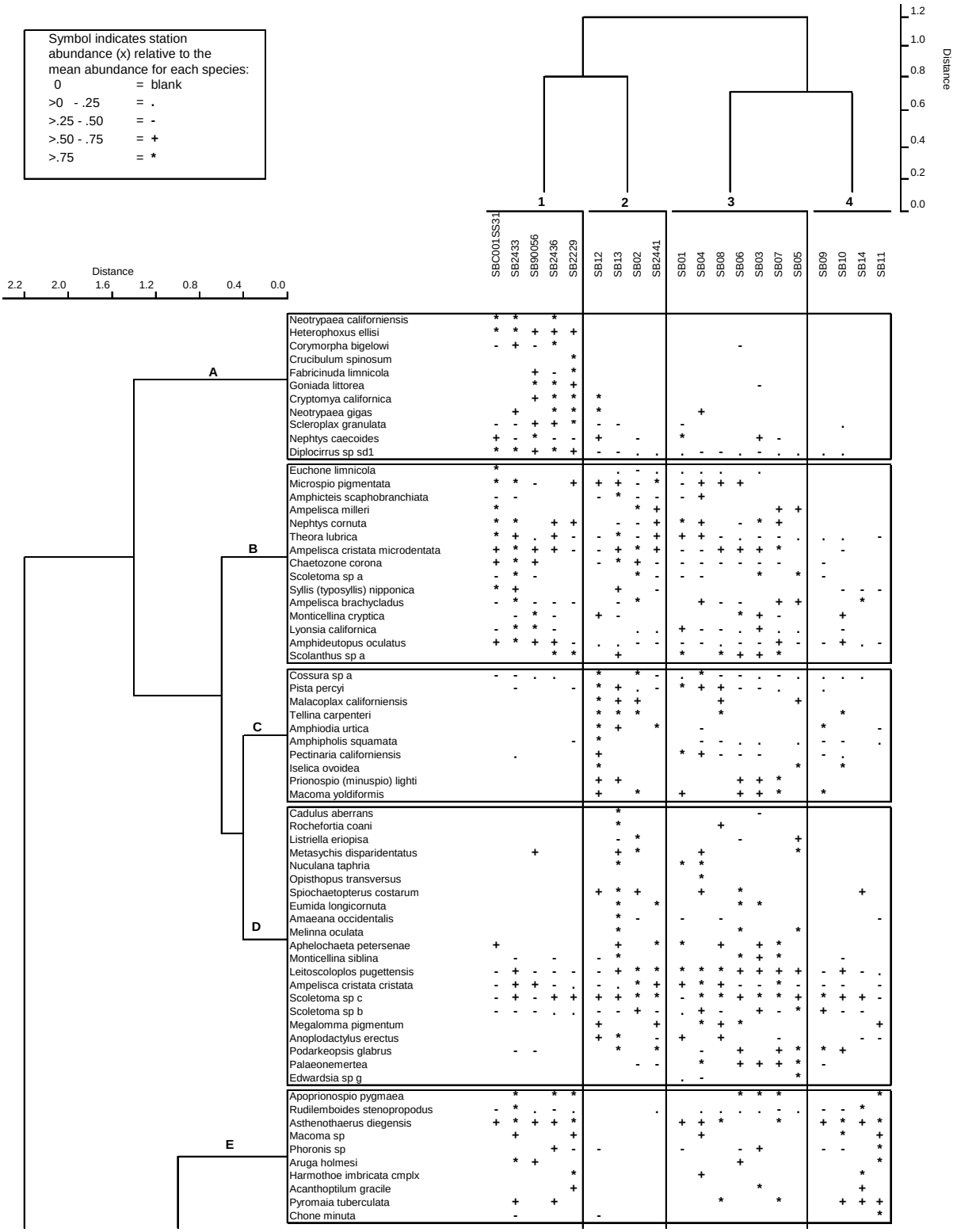
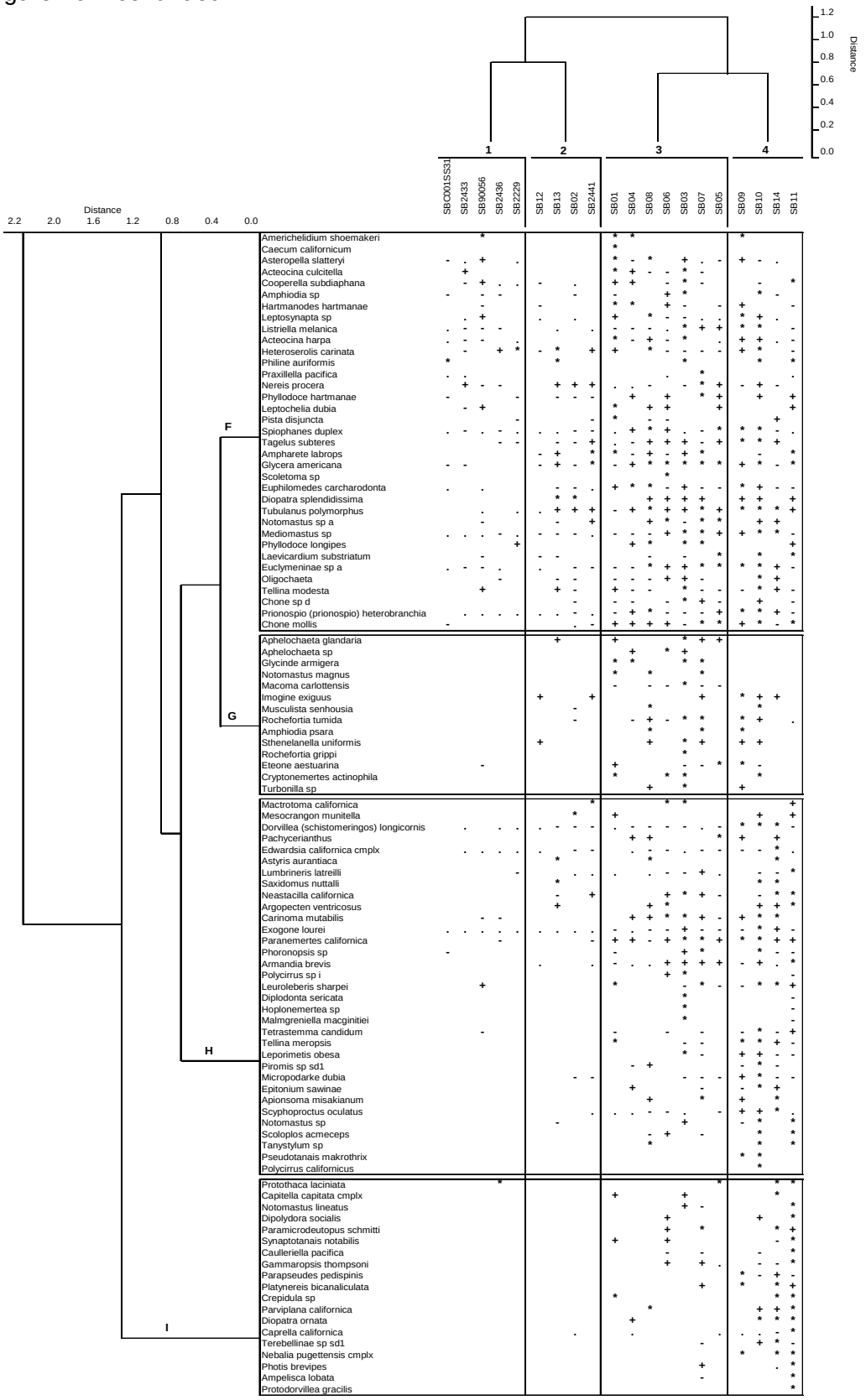


Figure 10-1. Two-way coincidence table of species groups vs. stations as resolved by cluster analysis using the Bray-Curtis Similarity Metric and Euclidian distances in ordination space. Data were square root transformed and standardized by maximum species abundance. Symbols represent the relative abundance of each species at a station.

Figure 10-1 continued.



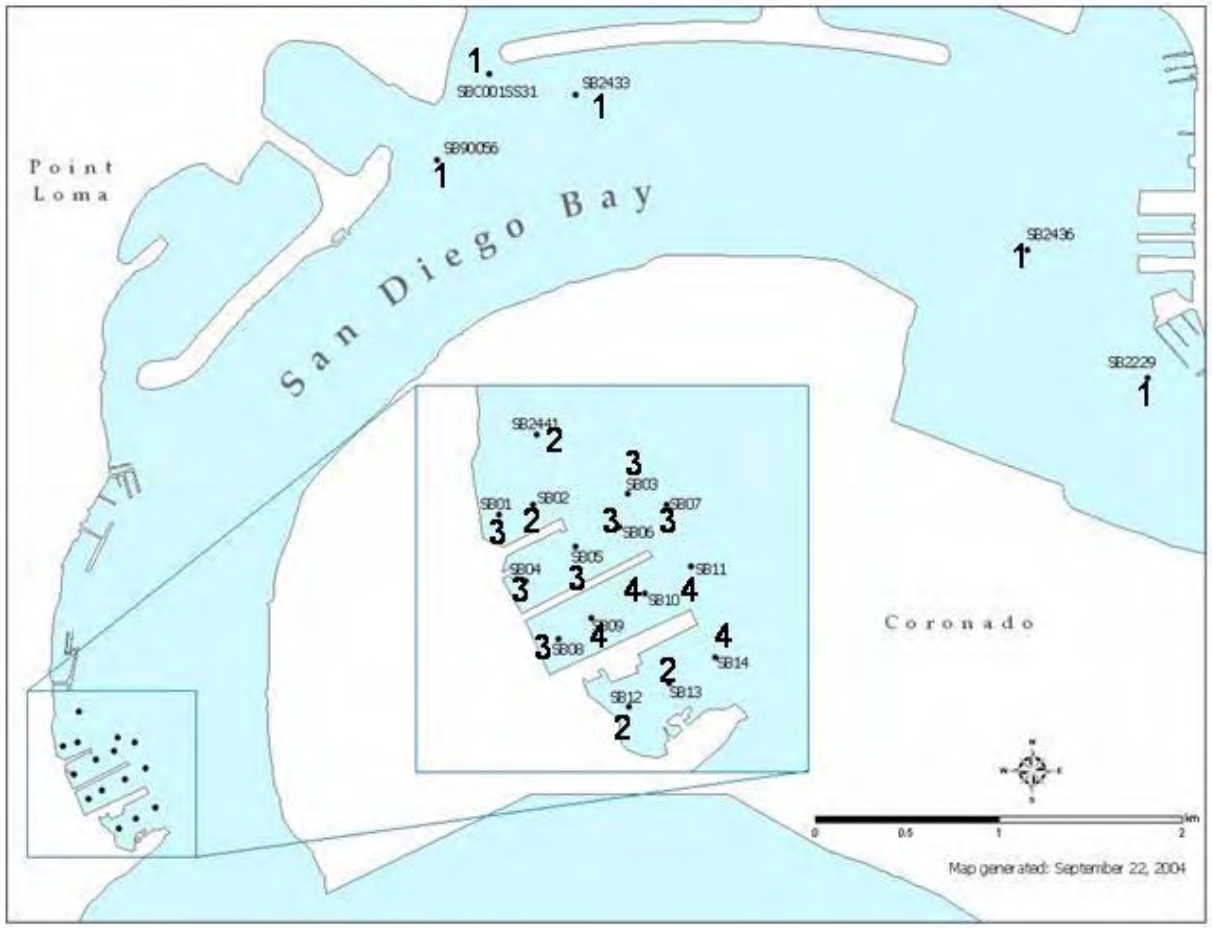


Figure 10-2. Location of stations in cluster analysis groups.

10.3 SPATIAL DISTRIBUTIONS

Spatial distributions of the key benthic community metric are shown in Figure 10-3 through Figure 10-6. Infauna abundance typically increased with distance away from the shoreline (Figure 10-3). The number of taxa also generally increased offshore as shown in Figure 10-4. In contrast, both S-W diversity and BRI generally decreased offshore (Figure 10-5 and Figure 10-6).

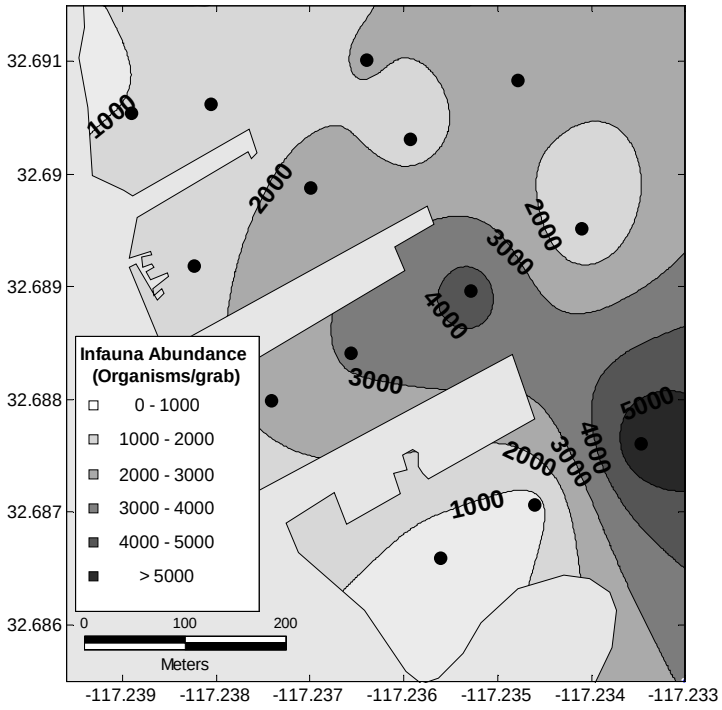


Figure 10-3. Spatial plot of infaunal abundance collected at SUBASE stations.

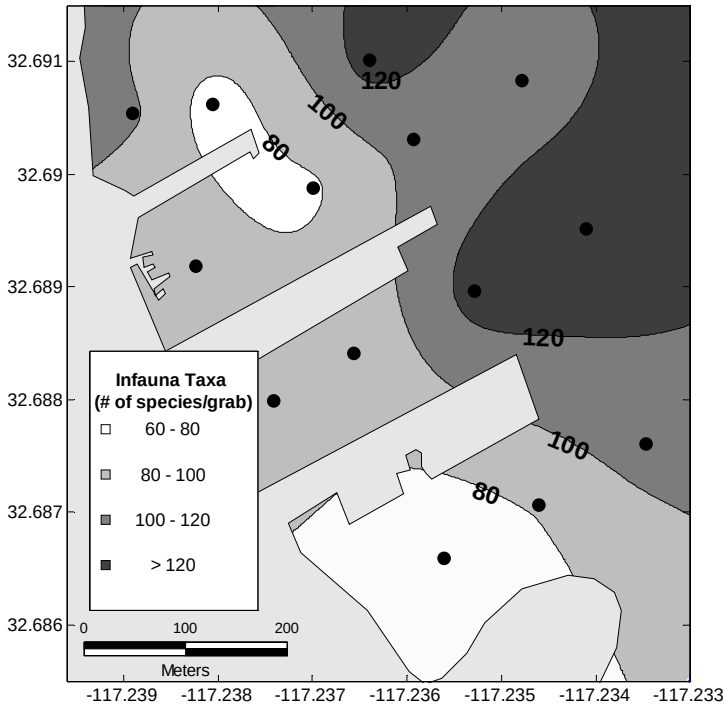


Figure 10-4. Spatial plot of infaunal species richness at SUBASE stations.

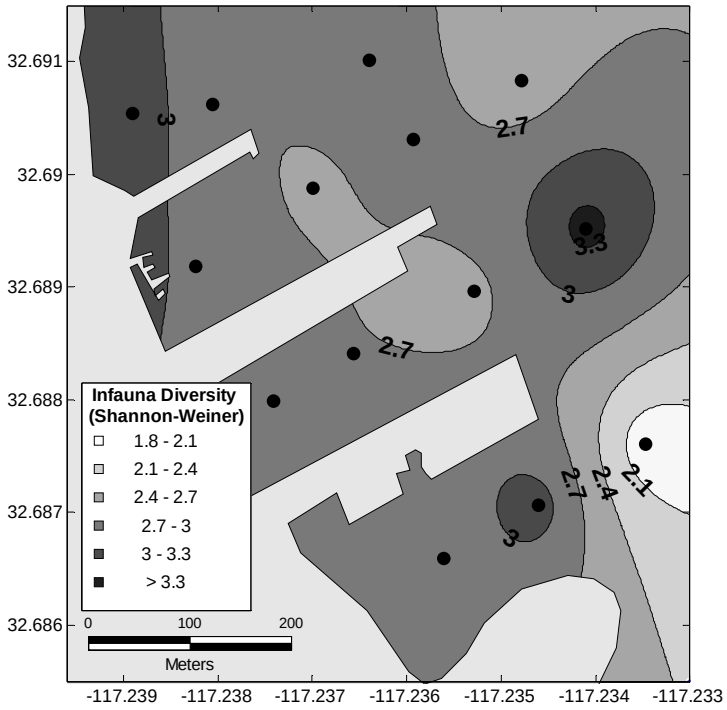


Figure 10-5. Spatial plot of infaunal Shannon-Weiner Diversity at SUBASE stations.

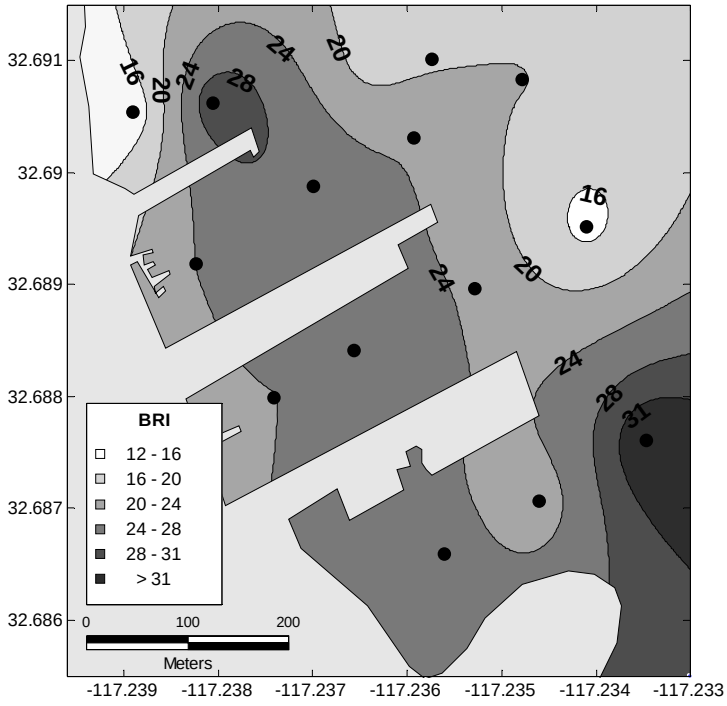


Figure 10-6. Spatial plot of infaunal Benthic Response Index at SUBASE stations.

10.4 TOXICITY- BENTHIC COMMUNITY RELATIONSHIPS

Two stations, SB2 and SB14, had sediments with a BRI level II response (31<BRI<41). These stations were located at opposite sides of the SUBASE study site and therefore have no spatial relationship. Station S0B2 had the highest fines and TOC measured at any site. Evaluation of individual and summed chemical parameters showed that average selenium values for the two sites were elevated by ~60% above those measured at all other sites. Additionally, station SB2 had a relatively elevated value (~60%) for consensus-based PCBs. These relationships suggest that a causal relationship was relatively weak.

As described earler, a multiple regression analysis was conducted as part of the benthic community analysis by Aquatic Bioassay and Consulting Laboratories of Ventura, CA (see Appendix D). The best fit model for the BRI results included the primary factors: pore water ammonia concentration, percent gravel, general metals group (As, Cd, Cr, Cu, Fe, Ni, Se, Zn) and PCB congener CI8. This independent evaluation indicates other groups of contaminants may have a causal relationship to the pore water toxicity outcome.

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11.0 ASSESSMENT OF EFFECTS

11.1 BASELINE POOL CHARACTERISTICS

A Baseline Pool of reference station data was used to represent the baseline condition that would be expected to exist at SUBASE sites, absent the direct influence of potential SUBASE contaminant sources. As described in Section 4, the Baseline Pool of stations used for analysis in this study consisted of up to 23 independent measurements made at six reference stations: six complete set of measurements made for stations during this study, six measurements of chemistry and amphipod toxicity made during the reconnaissance survey portion of this study, two sets of nearly complete measurements made during the Chollas-Paleta and Shipyards studies, three stations collected during the Bight'98 study, and four measurements made as part of the Switzer Creek study. In addition to the results of individual parameters and summary statistics for those parameters, the upper (i.e. for concentration) or lower (i.e. for survival) 95th-percentile prediction limit was computed for each parameter from the Baseline Pool after checking and transforming data, if necessary for normality (discussed below). The prediction limits were used as a threshold to determine if conditions at the SUBASE site differed from the baseline condition. Although multiple comparisons were made to the Baseline Pool predictive limits, no correction for multiple comparisons was applied to the predictive limits so the comparisons would remain conservative and more protective.

The normality for each parameter was determined during the Chollas-Paleta study. During that study, both the Kolmogorov/Smirnov (KS) and Shapiro-Wilk tests were applied to a subset of the San Diego Bay Bight'98 dataset to determine whether parameters were normally distributed or not. In the event a distribution was not normally distributed ($P<0.1$) the data were transformed using ln, square-root, arcsine, or cube transforms. In instances when multiple transforms could satisfy normality, the best transform was chosen based on best professional judgment after review of the resulting p and r statistics and review of a graphical representation of the data. The data transforms used for the Baseline Pool are shown in Table 11-1.

11.1.1 Physical Properties

Fines and TOC data for the Baseline Pool are shown in (Table 11-2). The range of fines in the Baseline Pool of 13 to 84% was comparable to those found at the SUBASE sites (7 to 87%). The range of TOC for the Baseline Pool of 0.2 to 2.5% was also comparable to the range measured at SUBASE sites (0.25 to 2.1%). TOC in the Baseline Pool stations generally increased with increasing fines following a similar trend to that observed at the SUBASE stations (Figure 11-1).

11.1.2 Metals

Metals characteristics and summary statistics for the Baseline Pool are shown in Table 11-2. Metal concentrations in the Baseline Pool were generally low, and showed minimal variation from station to station. For example, zinc in the Baseline Pool ranged from 78 to 175 mg/kg, with an RSD of only 22%. Silver had the highest degree of variability with a RSD of 51%. Among the Baseline Pool stations, SB2441 had the highest occurrence of maximum metal concentrations including. The higher metals at this station were consistent with the sample having the highest fines content (84%)... Station SB90056 had the highest occurrence of

minimum metal concentrations consistent with it having the lowest fines content (13%) of Baseline Pool Stations. Relative to SQGs, maximum metals concentrations in the Baseline Pool all fell below their respective ERM threshold.

11.1.3 Organic Contaminants

Sediment PPPAH concentrations in the 23 Baseline Pool stations ranged from about 311 to 2418 μg/kg and averaged 1020 μg/kg (Table 11-3). The maximum PPPAH concentration was found at station SB2229. None of the PAH levels measured at these stations exceeded the CBSQG value of 1800 μg/g OC. The PAH data were ln transformed to ensure normality when making statistical comparisons.

PCB concentrations in the Baseline Pool ranged from 4.0 to 51 μg/kg with a mean TPCB concentration of 19.5 μg/kg (Table 11-3). None of the PCB levels measured at these stations exceeded the CBSQG PCB value of 400 μg/kg. The PCB data were ln transformed to insure normality when making statistical comparisons. The three data values for the Bight'98 stations had elevated method detection limits (MDL) but were still included in calculating the upper predictive limit. The 95% predictive limit calculated without these data would have decreased from 50 to 32 μg/kg. This is one measure of uncertainty in the dataset, an issue that is addressed in a separate section.

TCHLOR concentrations at the Baseline Pool stations were generally low, and ranged from 0.2 to 1.4 μg/kg with a mean concentration of 0.7 μg/kg (Table 11-3). TDDT concentrations ranged from 0.5 to 3.8 μg/kg with a mean of 1.9 μg/kg. Dieldrin concentrations ranged from 0.04 to 1.0 μg/kg with a mean of 0.3 μg/kg. All the chlorinated pesticides were below their respective ERM values.

TCHLOR data were normally distributed but TDDT data were ln transformed to make its distribution normal. There was an insufficient amount of dieldrin data to determine if its distribution was normal, therefore the data were not transformed.

SQGQ1 Calculation

As mentioned previously, CoPCs were evaluated against their individual benchmark SQGs, as well as a combined group against a mean SQGQ1 quotient benchmark (Fairey et al. 2001). The SQGQ1 quotient is an empirically derived guideline that is best predictive of acute toxicity to marine amphipods. The SQGQ1 is calculated as follows:

$$SQGQ1 = ((\Sigma\ ([cadmium]/4.21)([copper]/270)([lead]/112.18)([silver]/1.77)([zinc]/410)([total\ Chlordane]/6)([dieldrin]/8)([total\ PAHOC]/1{,}800)([total\ PCB]/400))/9).$$

The denominators in each of the quotients are the SQG values that are most predictive in identifying threshold effects for the individual chemical. In the order of chemicals above, the SQGs used were: PEL, ERM, PEL, PEL, ERM, ERM, ERM, consensus, consensus. The SQGQ1 calculated for the Baseline Pool stations is shown in (Table 11-3). The SQG1 values ranged from 0.08 to 0.24 which were all below the low" SQGQ1 benchmark level ranking following Long et al. (1998).

11.1.4 Toxicity

Control-adjusted amphipod survival in the Baseline Pool sediments ranged from 82 to 101%, with a mean of 94% (Table 11-4). All station bulk toxicity results were considered non-toxic because survival levels were above a MSD value of 75%. Control-adjusted normal mussel embryo- larval development in the SWI tests for the Baseline Pool ranged from 86 to 100% with a mean of 93% (Table 11-4). All station SWI toxicity results were considered non-toxic because they had normal development levels above a MSD value of 80%. Control-adjusted normal urchin embryo- larval development in the pore water interface tests for the Baseline Pool ranged from 20 to 100% with a mean of 62% (Table 11-4). Only pore water test results for stations SB2436 and SBC001SS31 were deemed toxic because normal development was below the MSD criterion of 55% (relative to control). The high degree of variability in test results and the limited number of data points resulted in a negative value for its lower predictive limit, indicating that the test has no predictive capability.

11.1.5 Benthic Community

Abundance measurements in the Baseline Pool sediments ranged from 80 to 1672 with a mean of 591. The number of Taxa ranged from 43 to 108 with a mean of 62. The Shannon-Wiener diversity index ranged from 1.6 to 3.3 with a mean of 2.7. The Benthic Response Index (BRI) yielded results ranging from 11 to 36 with a mean BRI of 22 (Table 11-5). All stations had a BRI response level 1 except SC2433, which had a response level II. The benthic community data for the Switzer Creek study appeared to be considerably different than those measured at the same location in the other studies. The data reported here were provided directly by personal communication with Brian Anderson, author of the study, as only the draft study (Anderson et al., 2005) was openly distributed and there was no final report published. The draft data were much closer in value to the other study results. This issue is considered in the uncertainty section of the report.

11.1.6 Bioaccumulation

As described in section 4.2, statistical analysis for potential impacts to aquatic dependent wildlife and human health from CoPC in the sediment at the study sites was based on bioaccumulation in clams. In this study, clams were exposed to site sediments *in situ*. Because this method differed from the other studies, only the bioaccumulation results from this study were used to evaluate potential impacts to aquatic dependent wildlife and human health. The requisite calculations to analyze potential risks to wildlife and human health also required a slightly different list of organic chemical constituents than was needed for sediment chemistry analysis to align with the list of chemicals having human health factors. Therefore, upper 95% predictive limits where created for the following organic parameters: naphthalene, benzo-a-pyrene, TPCBS, α-Chlordane, γ-Chlordane, sum of ortho and para DDE, sum of ortho and para DDD and the sum of ortho and para DDT.

Tissue characteristics and metals data, summary statistics, and upper 95% predictive limits for the current reference stations are shown in Table 11-6. Tissue metal concentrations at the reference stations were generally low, and showed relatively little variation from station to station. For example, arsenic ranged from 24.8 to 26.8 mg/kg dry weight, with an RSD of 5%, and zinc ranged from 91.3 to 100 mg/kg dry weight with an RSD of 4%. Nickel was the most variable metal measured with a RSD of 41%.

Among the reference stations, SB90056 had the highest occurrence of minimum metal concentrations including silver, chromium, nickel, lead and zinc. SB2441 had the highest occurrences of maximum concentrations including silver arsenic, chromium, and nickel.

Tissue organics data, summary statistics, and upper 95% predictive limits for the current reference stations are shown in Table 11-7. Tissue organics concentrations were generally low, but slightly more variable than the metals data based on RSD data. For PAHs, naphthalene ranged from about 0.29 to 0.51 μg/kg and averaged 0.41 μg/kg dry weight while benzo-a-pyrene ranged from 5.1 to 7.6 and averaged 5.7 μg/kg dry weight. TPCBs ranged from about 30 to 143 and averaged 71 μg/kg dry weight. The Chlordanes, α-Chlordane and γ-Chlordane averaged 0.05 and 0.15 μg/kg dry weight respectively. The accumulated chlorinated pesticide concentrations of DDE, DDD, and DDT decreased in the order DDE>DDD>DDT. Average values were 0.85, 0.32, and 0.06 μg/kg dry weight, respectively. Similar to the results observed for metals, SB90056 had the highest occurrence of minimum organic concentrations. SB2433 had the highest occurrence of maximum organic concentrations.

Table 11-1. Data transforms used to produce normally distributed data for use in statistical testing against the Baseline and Reference Pools. There were insufficient data to evaluate normality of the sea urchin or mussel development tests.

	Baseline Pool
Parameter	**Transform**
Metals	
Ag	NA
As	NA
Cd	NA
Cr	NA
Cu	NA
Hg	NA
Ni	NA
Pb	NA
Zn	NA
Organics	
PPPAH	Natural log
PCBs	Natural log
Chlordane	None
DDTs	Natural log
SQGQ1	Natural log
Toxicity	
Amphipod survival	No transformation
Sea urchin development*	No transformation
Mussel development*	No transformation
Benthic Community	
Abundance	Natural log
Taxa	Natural log
Diversity	No transformation
BRI	No transformation

* insufficient data to check normality

Table 11-2. Individual station characteristics and summary statistics for physical properties and bulk sediment metal concentrations in the Baseline Pool. None of the station data exceeded their respective ERM values. Gray cells indicate values = ½ MDL.

StationID	FINES %	TOC %	Ag mg/kg	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Hg mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg
SB2229	15.0	0.25	0.31	6.5	0.22	29.9	35.9	0.209	6.9	24.3	105
SB2433	33.3	0.47	0.46	6.7	0.26	41.8	45.6	0.245	10.2	24.8	132
SB2436	37.3	0.63	0.62	8.3	0.25	46.5	64.2	0.381	11.4	33.8	151
SB2441	83.8	2.45	0.48	11.6	0.37	56.0	93.4	0.278	18.3	31.7	174
SB90056	13.2	0.24	0.25	6.3	0.24	28.5	20.4	0.128	6.0	18.1	79
SBC001SS31	59.5	0.91	0.63	8.5	0.29	51.8	81.5	0.511	14.2	30.6	175
SB2229-R	21.4	0.43	0.51	10.1	0.26	46.0	48.9	0.251	9.02	38.5	122
SB2433-R	38.6	0.57	0.59	7.2	0.38	45.2	49.2	0.313	10.6	25.2	125
SB2436-R	39.7	0.55	0.72	9.5	0.36	56.0	64.9	0.387	12.3	36.3	148
SB2441-R	61.2	1.65	0.56	10.7	0.42	48.2	65.5	0.216	14.4	24.3	133
SB90056-R	18.4	0.25	0.40	6.7	0.28	36.2	30.5	0.504	7.53	18.2	86
SBC001SS31-R	52.8	0.76	0.68	8.6	0.38	52.6	71.2	0.269	12.9	28.5	148
C-P, SY & B98											
CP2433	38.4	0.53	0.38	5.6	0.29	42.2	43.3	0.251	11.15	23.3	115
CP2441	82.8	1.82	0.39	8.8	0.41	54.0	78.4	0.238	17.5	26.7	143
SY2433	41.0	0.67	0.39	4.6	0.29	24.0	40.0	0.210	7.4	19.0	92
SY2441	41.0	1.10	0.24	5.4	0.29	22.0	37.0	0.160	9.9	13.0	80
B982229	43.0	0.92	0.41	5.4	0.085	31.6	58.9	0.316	9.3	24.5	99
B982436	55.0	1.36	0.62	8.6	0.21	48.4	85.8	0.517	15.3	34.4	145
B982441	79.0	1.97	1.50	12.4	0.25	43.9	71.8	0.191	16.6	21.9	123
Switzer Creek											
SC2433-Recon	44.8	1.01	0.65	7.2	0.25	38.2	59.6	0.190	10.9	18.9	134
SC2229	35.7	0.46	0.42	4.5	0.11	22.7	42.0	0.320	5.9	23.9	103
SC2433	49.1	0.56	0.23	4.5	0.21	30.4	46.5	0.260	8.5	17.1	111
SC2441	62.9	2.00	0.35	7.6	0.31	48.7	80.9	0.310	15.9	22.0	149
N	23	23	23	23	23	23	23	23	23	23	23
Mean	45.5	0.94	0.51	7.6	0.28	41.1	57.2	0.289	11.4	25.2	124.8
Std Dev	19.8	0.64	0.26	2.2	0.08	10.9	19.5	0.108	3.7	6.6	27.6
RSD	44%	68%	51%	29%	31%	27%	34%	37%	33%	26%	22%
95% PL	80.3	2.1	0.95	12.3	0.45	66.3	104.0	0.51	17.9	39.0	182.7

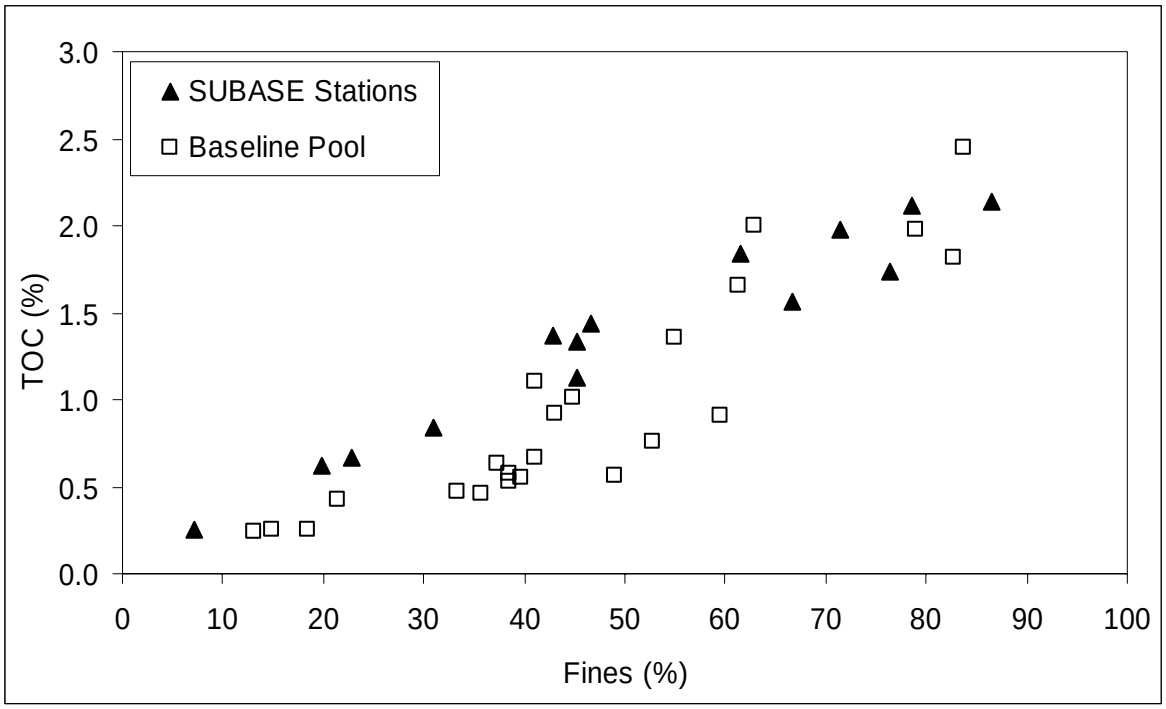

Figure 11-1**.** Relationship between bulk sediment TOC and Fines for Baseline Pool and SUBASE site stations.

Table 11-3. Individual bulk sediment characteristics, summary statistics, SQG, and 95% upper predictive limits for organic contaminants in the Baseline Pool. The table includes SQGQ1 values for each station. Blank cells indicate no data. Gray cells indicate values = ½ MDL.

StationID	PPPAH µg/kg	13 Consensus PAHs µg/g OC	TPCB µg/kg	18 consensus PCBs µg/kg	TCHLOR µg/kg	TDDT µg/kg	TDDT µg/g OC	TBT µg/kg	Dieldrin µg/kg	SQGQ1
SB2229	2418	746	9.4	8.02	0.86	1.10	0.44	1.43	0.12	0.160
SB2433	453	78.1	9.6	8.54	0.29	0.96	0.20	1.45	0.04	0.130
SB2436	1003	127	19.9	17.5	0.64	1.75	0.28	1.74	0.05	0.176
SB2441	1681	62.2	11.5	9.67	1.33	2.19	0.09	1.91	0.98	0.204
SB90056	311	110	4.1	3.59	0.17	0.49	0.20	1.23	0.04	0.082
SBC001SS31	983	89.2	17.7	14.7	0.80	1.94	0.21	1.86	0.05	0.187
SB2229-R	887	170	13.0	11.2	1.22	1.07	0.25	2.61	0.25	0.174
SB2433-R	756	110	10.1	9.58	0.85	1.44	0.25	2.64	0.21	0.154
SB2436-R	1210	171	17.9	15.6	1.38	1.81	0.33	3.33	0.32	0.204
SB2441-R	1170	62.6	8.7	7.79	1.16	1.65	0.10	2.97	0.31	0.172
SB90056-R	403	136	5.6	5.05	0.51	0.72	0.29	2.53	0.14	0.111
SBC001SS31-R	802	86.5	14.6	12.8	1.26	1.92	0.25	3.16	0.34	0.188
C-P, SY & B98										
CP2433	780	121.6	27.1	16.97	0.57	2.10	0.4			0.147
CP2441	2143	105.1	33.5	19.73	0.83	3.79	0.2			0.186
SY2433	486	56.7	20.8	15.80				3.3		0.135
SY2441	343	25.5	10.5	7.95				3.7		0.101
B982229	1339	132.4	50.5	16.90	0.60	1.67	0.18			0.149
B982436	565	36	50.5	16.90	0.60	1.67	0.12			0.197
B982441	1445	69.4	50.5	16.90	0.60	1.67	0.08			0.236
Switzer Creek										
SC2433-Recon	891	64.9	15.5	9.0	0.3	3.00	0.30		0.5	
SC2229	1220	198.7	15.5	9.0	0.25	3.00	0.65		0.5	0.149
SC2433	960	134.5	15.5	9.0	0.25	3.00	0.53		0.5	0.133
SC2441	1218	56.0	15.5	9.0	0.25	3.00	0.15		0.5	0.168
N	23	23	23	23	21	21	21	14	16	22
Mean	1020.3	128.3	19.5	11.8	0.7	1.9	0.3	2.4	0.3	0.161
Std Dev	541.2	141.9	13.9	4.5	0.4	0.9	0.1	0.8	0.3	0.037
RSD	53%	111%	72%	38%	56%	45%	55%	34%	84%	23%
95% PL	2341.4	377.3	49.6	19.7	1.4	4.2	0.5	3.9	0.8	0.243

Table 11-4. Individual station characteristics, summary statistics, and 95% lower predictive limits (95% PL) for control adjusted bulk sediment amphipod survival (%), pore water urchin development (% normal), and sediment water interface (SWI) mussel development (%) in the Baseline Pool. A * indicates test result was toxic. Blank cells indicate no data.

StationID	Bulk Sediment Amphipod Survival % of Control	Sediment Water Interface Mussel Embryo-Larval Development % of Control	Pore Water Sea Urchin Embryo-Larval Development % of Control
SB2229	100.0	100.0	100.0
SB2433	98.9	86.7	73.5
SB2436	97.8	100.0	23.1*
SB2441	84.4	85.8	100.0
SB90056	98.9	100.2	55.5
SBC001SS31	96.7	86.9	20.1*
SB2229-R	90.8		
SB2433-R	98.0		
SB2436-R	93.9		
SB2441-R	90.8		
SB90056-R	100.0		
SBC001SS31-R	94.9		
C-P, SY & B98			
CP2433	84.1		
CP2441	82.3		
SY2433	95.9		
SY2441	95.0		
B982229	94.0		
B982436	96.0		
B982441	87.0		
Switzer Creek			
SC2433-Recon	91.8		
SC2229	101.0		
SC2433	94.9		
SC2441	98.0		
N	23	6	6
Mean	94.1	93.3	62.0
Std Dev	5.4	7.5	35.6
RSD	6%	8%	57%
95% PL	84.7	77.0	-15.4

Table 11-5. Individual station characteristics, summary statistics, and 95% lower predictive limits for abundance, number of taxa, Shannon-Weiner diversity index, BRI and BRI level in the Baseline Pool. Blank cells indicate no data.

StationID	Abundance	Taxa #	SWDiversity	BRI	BRI Level
SB2229	510	62	3.10	16.0	I
SB2433	909	63	2.86	18.5	I
SB2436	682	57	2.69	21.6	I
SB2441	994	67	2.78	27.5	I
SB90056	495	66	3.34	10.6	I
SBC001SS31	611	55	2.85	19.2	I
SB2229-R					
SB2433-R					
SB2436-R					
SB2441-R					
SB90056-R					
SBC001SS31-R					
C-P, SY & B98					
CP2433	421	57.0	2.82	22.8	I
CP2441	476	66.0	2.93	30.0	I
SY2433	441	77	2.58	16.8	I
SY2441	506	108	2.80	19.9	I
B982229	705	63	3.12	15.7	I
B982436	599	48	3.06	19.4	I
B982441	1672	86	3.23	17.2	I
Switzer Creek					
SC2433-Recon		47			
SC2229	80	45	1.80	30.9	I
SC2433	102.3	43	1.70	36.1	II
SC2441	258.7	51	1.60	30.9	I
N	16	17	16	16	
Mean	591	62	2.70	22.1	
Std Dev	379	16.3	0.5	7.0	
RSD	64%	26%	20%	32%	
95% PL	118	39.5	1.7	34.8	

Table 11-6. Individual station characteristics, summary statistics, and 95% upper predictive limits for tissue solids (%), lipids (%), and metals (mg/kg dry weight) for bioaccumulation at reference stations sampled during this study.

Station ID	Solids %	Lipid %	Ag mg/kg	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Hg mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg
SB2229	10.46	0.48	0.31	24.8	0.18	1.79	32.4	0.11	1.58	2.86	92.0
SB2433	10.31	0.45	0.28	22.8	0.19	2.51	30.2	0.08	2.34	3.23	99.6
SB2436	11.01	0.47	0.26	25.0	0.21	2.79	29.1	0.09	2.63	3.06	100.0
SB2441	10.71	0.45	0.42	26.8	0.18	3.36	31.1	0.07	4.22	2.92	93.2
SB90056	9.66	0.38	0.23	25.8	0.19	1.71	29.2	0.08	1.38	2.58	91.3
SBC001SS31	10.86	0.36	0.25	25.1	0.23	3.11	26.7	0.08	3.26	2.67	97.9
N	6	6	6	6	6	6	6	6	6	6	6
MEAN	10.5	0.43	0.29	25.0	0.20	2.55	29.8	0.08	2.57	2.89	95.7
STDEV	0.48	0.05	0.07	1.3	0.02	0.68	2.0	0.01	1.06	0.24	4.0
RSD	5%	11%	24%	5%	10%	27%	7%	15%	41%	8%	4%
Upper 95% Predictive Limit	11.56	0.54	0.45	27.9	0.24	4.02	34.0	0.11	4.88	3.41	104.3

Table 11-7. Individual station characteristics, summary statistics, and 95% upper predictive limits for tissue organic contaminants (μg/kg dry weight) for bioaccumulation at reference stations sampled during this study.

Station ID	Naph μg/kg	BAP μg/kg	TPCB μg/kg	α-Chlor μg/kg	γ-Chlor μg/kg	DDE μg/kg	DDD μg/kg	DDT μg/kg	TBT μg/kg
SB2229	0.51	7.06	29.84	0.08	0.11	0.82	0.33	0.06	76.67
SB2433	0.37	5.14	123.2	0.05	0.12	0.98	0.34	0.07	68.00
SB2436	0.29	7.65	29.87	0.08	0.11	0.96	0.39	0.06	81.47
SB2441	0.56	6.00	40.17	0.02	0.08	0.79	0.21	0.06	47.90
SB90056	0.36	3.84	142.8	0.02	0.06	0.66	0.26	0.06	82.51
SBC001SS31	0.38	4.64	58.05	0.07	0.12	0.91	0.39	0.06	69.15
N	6	6	6	6	6	6	6	6	6
MEAN	0.41	5.72	70.7	0.05	0.10	0.85	0.32	0.06	70.95
STDEV	0.10	1.46	49.8	0.03	0.02	0.12	0.07	0.01	12.81
RSD	25%	25%	70%	59%	24%	14%	23%	12%	18%
Upper 95% Predictive Limit	0.63	8.90	179.03	0.12	0.15	1.11	0.48	0.07	98.83

11.2 AQUATIC LIFE

11.2.1 Sediment Chemistry

Effects on aquatic life were assessed using three lines of evidence (LOE): sediment chemistry, toxicity, and benthic community composition. The relative degree of effect (or likelihood of an impact) was evaluated using the criteria described in Section 4.2 and used to classify each station as having low, moderate, or high impact for each LOE.

The relative likelihood that bulk sediment CoPCs were site-specific causative agent for effects was ranked into three general categories of low, moderate, or high. The rankings were based on a comparison of station values to the upper predictive limit of the SQGQ1 and to individual chemical SQGs as well as to the 95th percentile UPL of the Baseline Pool. The rankings were based on an increasing weight or confidence that an effect to aquatic life will occur given an increasing number and magnitude of chemicals exceeding the SQG thresholds. The process used to apply the chemistry ranking criteria and classify the stations is illustrated in Figure 11-2.

Results of the sediment chemistry LOE for each station in the SUBASE sites are shown in Table 11-8. All stations were categorized as “Low” indicating that all chemicals present at the site, either individually or summed were lower than a relatively low SQG and were below the baseline condition. Thus, sediments throughout the SUBASE area contain chemical concentrations no different than what would be found there absent any release from the SUBASE facility. The levels are also sufficiently low to be unlikely to cause impairment.

Table 11-8. Results of sediment chemistry LOE for each station in the SUBASE sites. Results are categorized as No/Low (O), Moderate (⊙), or High (●). All stations showed low/no chemical impacts.

Station	SQGQ1	# Chemicals exceeding SQG and UPL	SQGQ1 Level	SQGQ1 > Reference	Chem Class
SB1	0.15	0	I	-	O
SB2	0.19	0	I	-	O
SB3	0.08	0	I	-	O
SB4	0.21	0	I	-	O
SB5	0.08	0	I	-	O
SB6	0.12	0	I	-	O
SB7	0.09	0	I	-	O
SB8	0.19	0	I	-	O
SB9	0.18	0	I	-	O
SB10	0.13	0	I	-	O
SB11	0.05	0	I	-	O
SB12	0.23	0	I	-	O
SB13	0.16	0	I	-	O
SB14	0.12	0	I	-	O

11.2.2 Toxicity

The results from all three toxicity tests were used to classify the relative magnitude of sediment toxicity into three general categories of low, moderate, or high. The rankings were based on a comparison to the control and the Baseline Pool. Increasing weight or confidence that a toxic effect to aquatic life will occur was given when a severe effect on amphipod survival was present or if toxicity was observed in multiple tests. However, the pore water toxicity test was not predictive given its high degree of variability in the Baseline Pool. The process used to apply the toxicity ranking criteria and classify the stations is illustrated in Figure 11-3.

Results of the toxicity LOE evaluation for each station in the SUBASE sites are shown in Table 11-9. The overall results show low/no degree of toxic effects observed at any of the site stations. One station, SB14 had an amphipod survival rate of 83.3% that was barely below the Baseline Pool UPL value of 84.7%. One station, SB13, had a SWI test result (79%) that was statistically different from control and barely below the MSD value of 80%. Because the pore water results in the Baseline Pool were highly variable, site results could not be detected as being different. However, based on the requirement that both the SWI test and the pore water

test would both need to show effects before identifying a problem, different results of the SWI test would not have changed the outcome of the toxicity LOE showing low/no degree of toxic effects.

Table 11-9. Results of the toxicity LOE for each station in the SUBASE sites. Results were categorized as No/Low (O), Moderate (⊙), or High (●). All stations showed low/no toxic effects.

Station	Amphipod Survival			Pore Water Urchin Development			Sediment Water Interface Mussel Development			Tox Class
	<Control	<Ref	<MSD (75%)	<Control	<Ref	<MSD (55%)	<Control	<Ref	<MSD (80%)	
SB01	-	-	-	+	-	+	-	-	-	O
SB02	-	-	-	-	-	-	-	-	-	O
SB03	-	-	-	-	-	-	-	-	-	O
SB04	-	-	-	+	-	+	-	-	-	O
SB05	-	-	-	-	-	-	-	-	-	O
SB06	-	-	-	-	-	-	-	-	-	O
SB07	-	-	-	-	-	-	-	-	-	O
SB08	-	-	-	-	-	-	-	-	-	O
SB09	-	-	-	-	-	NA	-	-	-	O
SB10	-	-	-	-	-	NA	-	-	+	O
SB11	-	-	-	-	-	-	-	-	-	O
SB12	-	-	-	+	-	+	-	-	-	O
SB13	-	-	-	-	-	-	+	-	+	O
SB14	-	+	-	-	-	NA	-	-	-	O

11.2.3 Benthic Community

The results from all the four benthic community parameters (BRI, abundance, number of taxa, Shannon-Weiner diversity index) were used to classify the relative response of the benthic community into three general categories of low, moderate, or high. The rankings were based on a comparison to the Baseline Pool for each parameter and, for the BRI, a comparison to five response level thresholds that indicate the degree departure from the reference condition expected in the absence of contamination. Increasing weight or confidence that a benthic community impact was present was given when a severe departure from the BRI reference condition was present or when effects were observed for multiple parameters. The process used to apply the benthos ranking criteria and classify the stations is illustrated in Figure 11-4.

Results of the benthic community LOE evaluation for each station in the SUBASE sites are shown in Table 11-10. All stations showed low/no degree of benthic community degradation. All but two stations showed a BRI response level I (reference). The two stations showing a BRI response level I were station SB2 (BRI=31) and SB14 (BRI=34), levels that exceed the cutoff value of 31 differentiating the two levels.

Table 11-10. Results of the benthic community analysis LOE for each station in the SUBASE sites, categorized as No/Low (O), Moderate (⊙), or High (●). All stations showed low/no benthic community effects.

Station	Abun<Ref	Taxa<Ref	SW<Ref	BRI>Ref	BRI	BRI Response Level	BCA Class
SB1	-	-	-	-	13	Ref	O
SB2	-	-	-	-	31	I	O
SB3	-	-	-	-	18	Ref	O
SB4	-	-	-	-	24	Ref	O
SB5	-	-	-	-	27	Ref	O
SB6	-	-	-	-	23	Ref	O
SB7	-	-	-	-	20	Ref	O
SB8	-	-	-	-	24	Ref	O
SB9	-	-	-	-	26	Ref	O
SB10	-	-	-	-	23	Ref	O
SB11	-	-	-	-	15	Ref	O
SB12	-	-	-	-	26	Ref	O
SB13	-	-	-	-	22	Ref	O
SB14	-	-	-	-	34	I	O

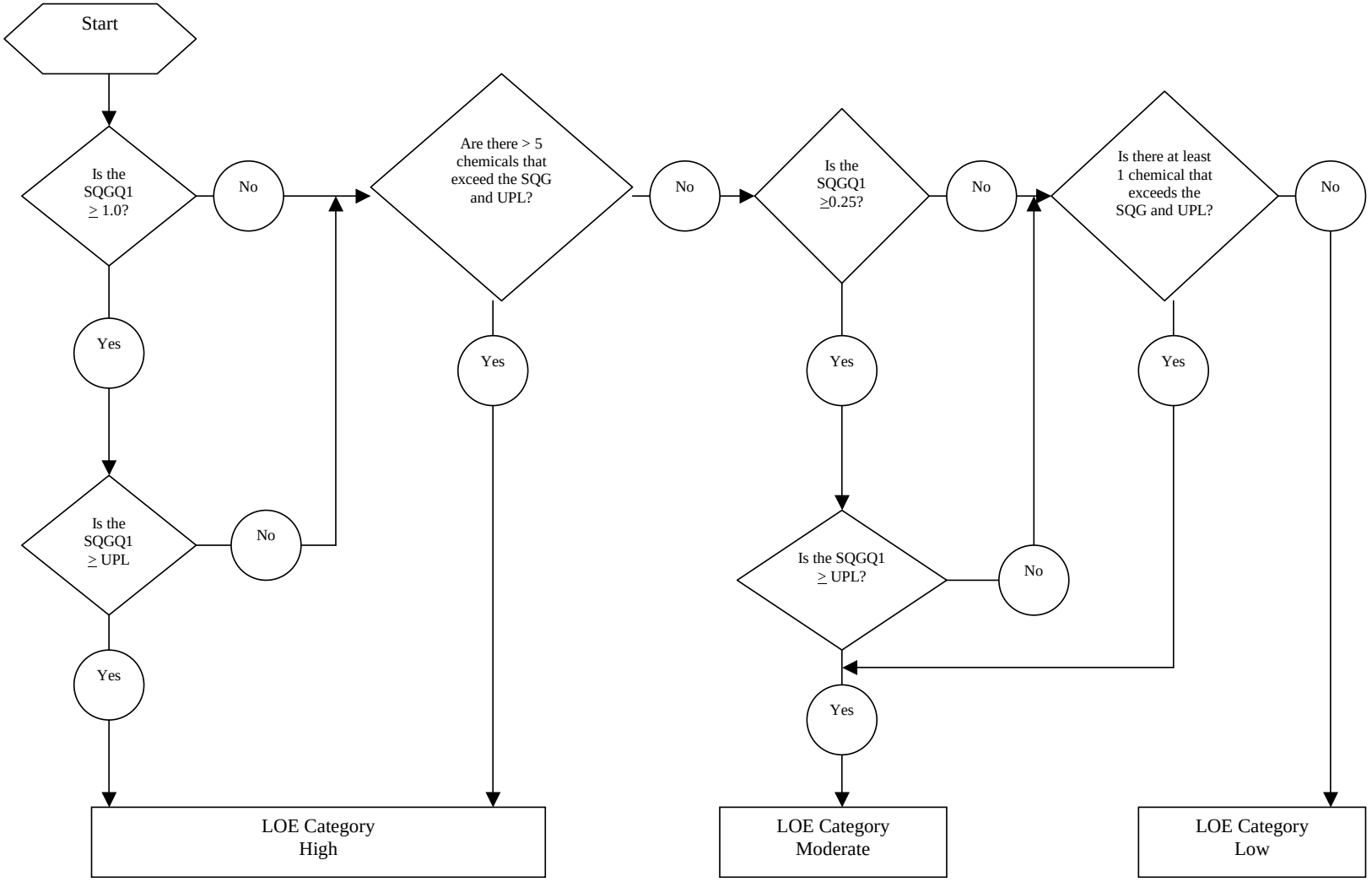

Figure 11-2. Schematic of decision tree used to apply station ranking criteria for chemistry.

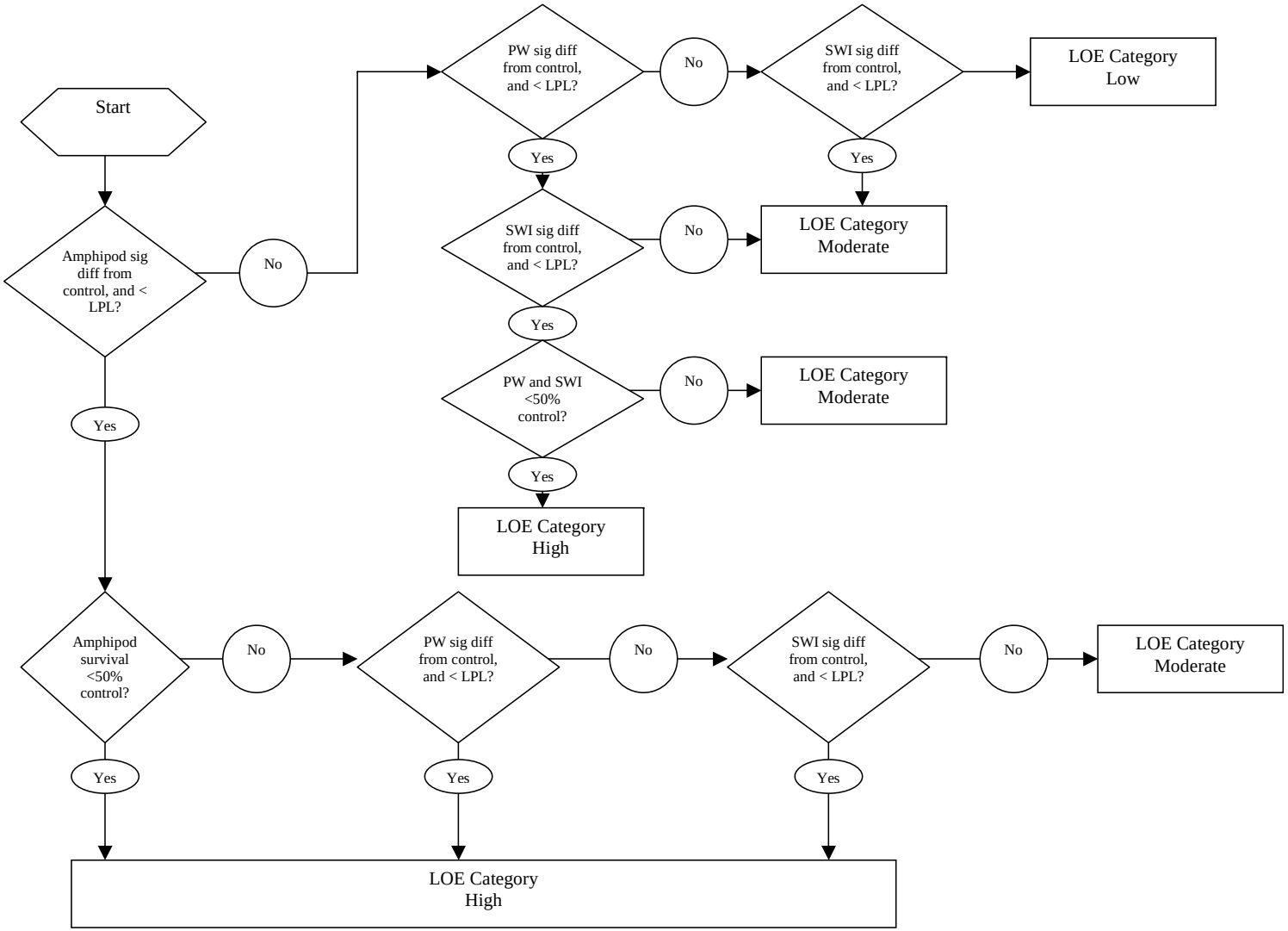

Figure 11-3. Schematic of decision tree used to apply station ranking criteria for toxicity.

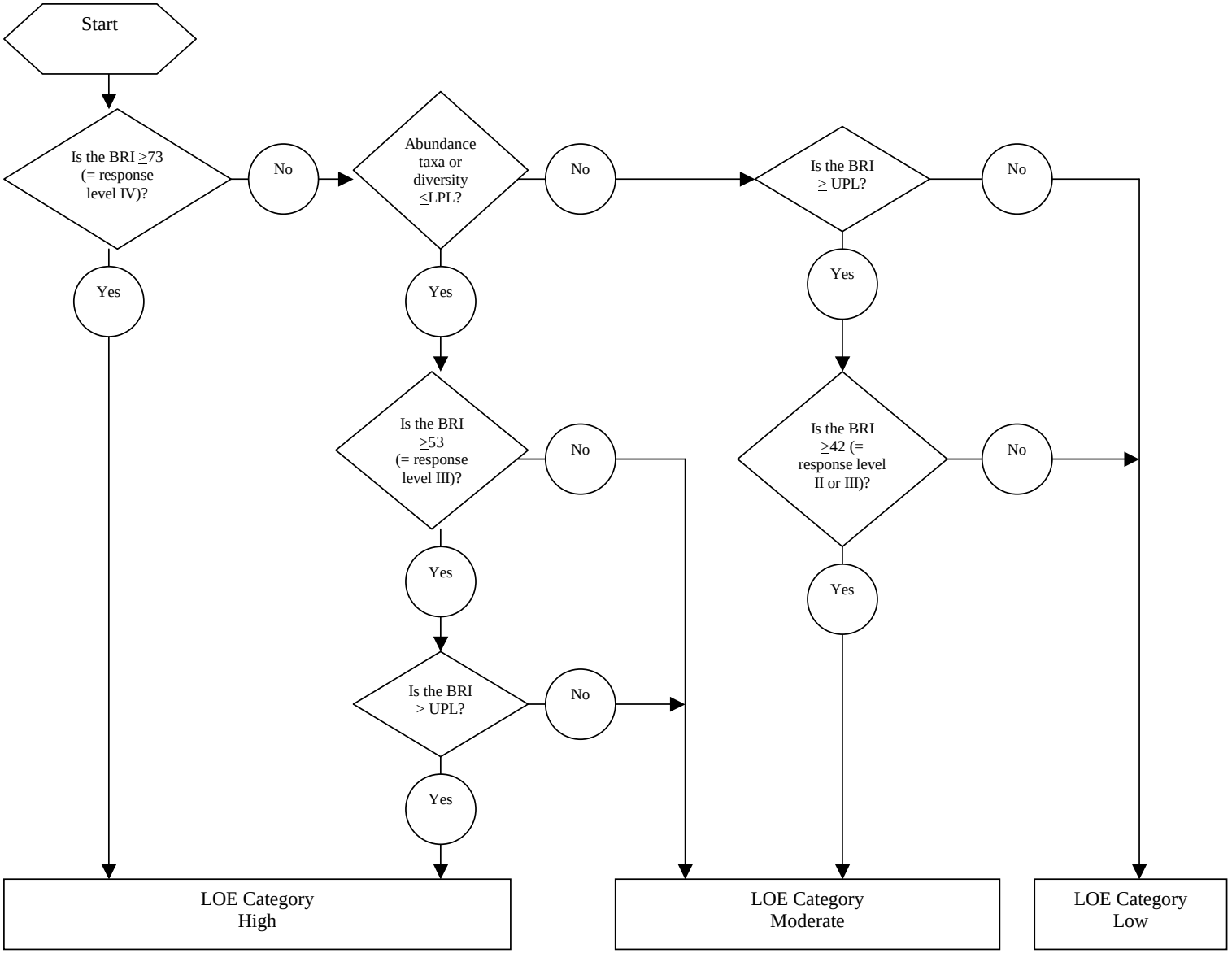

Figure 11-4. Schematic of decision tree used to apply station ranking criteria for benthos.

11.3 AQUATIC DEPENDENT WILDLIFE

A screening level risk assessment was performed to assess potential impairment to aquatic-dependent wildlife. For this assessment, bioaccumulation of CoPCs in the clam *Macoma nasuta* exposed to site sediments was used to estimate exposure for representative wildlife receptors including surface feeding birds (Least Tern and Brown Pelican), diving birds (Surf Scoter and Western Grebe), and marine mammals (California Sea Lion). For the screening level assessment, conservative exposure assumptions included 100% dietary fraction from the site, 100% area use factor for the site, and the low toxicity reference value. A summary of the risk assessment screening is shown in Table 11-11.

The screening level risk assessment for aquatic-dependent wildlife was based on the following procedure. First, chemical concentrations in clam tissue (wet weight) were compared to measurements made on control samples to detect the presence of contaminant bioaccumulation. Control samples were compared to pooled SUBASE stations using a one-sided t-test to detect statistical differences at $p<0.05$. Arsenic, chromium, copper, lead, nickel, silver, zinc, Naphthalene, Benzo[a]pyrene, TPCBs, γ-Chlordane, DDE and DDD all showed statistically significant bioaccumulation relative to controls (Table 11-11).

Next, the site-maximum tissue concentrations of clams (wet weight) exposed to study site sediments were compared with the 95% upper predictive limit of tissue concentrations from clams exposed to reference sediments to determine if the elevated concentrations were above those characteristic of relatively undegraded conditions in the bay. All metals except mercury and none of the organic compounds had maximum tissue concentrations greater than the 95% upper predictive limit of the reference stations (Table 11-11).

Finally, site-maximum tissue concentrations from SUBASE stations were used to estimate doses to wildlife receptors including surface feeding birds (Least Tern and Brown Pelican), diving birds (Surf Scoter and Western Grebe), and marine mammals (California Sea Lion). Doses for each receptor were estimated as

$$D = C_{tiss} \times NFR \times FR \times AE \times AUF$$

where: C_{tiss} is the tissue wet-weight concentration of the chemical, NFR is the normalized feeding rate, FFC is the fraction of the food that is contaminated, AE is the assimilation efficiency, and AUF is the area use factor. These parameters are summarized in Table 11-12. Estimated doses were then compared to the low Biological Technical Assistance Group (BTAG) TRVs (USEPA, 2002) where available, or other published thresholds in the case where BTAG TRVs were not available as shown in Table 11-13. Hazard quotients were then calculated as $HQ= Dose_{max}/TRV_{low}$ (Table 11-14 through Table 11-18). Copper and TPCBs were the only parameters that showed a maximum dose level above a wildlife TRV_{low} value (Table 11-11). Copper exceeded the TRV for all avian receptors and TPCBs exceeded the TRV only for the Least Tern.

Copper was the only contaminant that posed a “possible” screening level risk at site stations, exceeding all of the risk thresholds to avian receptors: Copper accumulated in tissues exposed to site sediments was significantly higher than observed in controls (t-test, P=0.05); Maximum copper accumulation at any site station was above the 95% UPL for the Baselin Pool Stations (in this case, the study reference stations); and the copper dosage based on maximum tissue concentrations) exceeded the TRV_{low} (HQ>1) for all avian receptors.

The risk screening outcome for copper, even given the highly conservative nature of the sceening assumptions, was somewhat surprising because of the relatively low copper concentrations observed in the sediments. The maximum copper concentration measured in the sediments (112 mg/kg) was less than half its ERM value of 270 mg/kg. Additionally, there appeared to be no relationship between accumulation in organism tissues and sediment concentrations (Figure 11-5). Upon closer examination, tissue accumulation data appeared to be biased high as a result of using the *in situ* exposure method. As shown in Table 11-19, copper tissue levels were about twice those measured in controls (exposed to bay waters) as well as at reference stations previously measured in other recent investigations conducted with laboratory exposures (SCCWRP and Navy, 2005; Exponent, 2005). The differences in results between bioaccumulation measured *in situ* in this study and that measured in the other two studies showed a very consistent bias. The relatively low sediment copper concentrations, lack of relationship between sediment concentrations and accumulation and the observed bias in the accumulation data suggest that the there was an additional copper source to the organisms exposed *in situ* that cannot be linked to the sediments. The most likely source of this additional copper uptake for the *in situ* exposure was from bay waters.

The relative impact to the overall risk evaluation from both sediment and water copper exposure was computed using the comparison of tissue uptake observed for controls and reference stations using the *in situ* methods used in this study and the laboratory methods used in previous studies (Table 11-19). The average difference in tissue uptake between the methods was 1.6 mg/kg_{wet}/day. The dosages for each SUBASE station were then recomputed by subtracting this bias amount to estimate the contribution from the sediment only. The maximum adjusted dosage for site stations was again compared to TRV_{low} to compute the maximum HQ associated with sediments. The relative dosages and associated risk levels are shown in Figure 11-6 and Figure 11-7 for each receptor. Based on this evaluation, there was still a screening level risk to the Least Tern related to ingestion of organisms feeding on site sediments. The sediment related risk to the other wildlife receptors was below the HQ threshold of 1.0.

Because of the lack of a relationship between copper bioaccumulation and sediment concentrations (BSAFs) and the uncertainty in the risk related to sediment only exposure, no estimate of copper accumulation was made for stations where no measurements made. Station SB8 had the highest bioaccumulation of copper and was the site station showing maximum dosages for all receptors. The sediment concentration for this site was 96.2 mg/kg. Three other site stations (SB2, SB9, and SB12) had nearly identical sediment concentrations as did reference station SB2441. The tissue data for these sites, when corrected for the copper bias, resulted in an unlikely risk to all receptors. Only one station, SB4, had a sediment concentration of 112 mg/kg that was higher than observed at Station SB8. This station, along with SB8 may still present a possible risk to some avian receptors assuming that a complete pathway exists.

Table 11-11. Summary of the screening level wildlife risk assessment for the SUBASE sediment investigation site.

	>Control	>Reference	Brown Pelican HQ>1	Least Tern HQ>1	Western Grebe HQ>1	Surf Scoter HQ>1	Sea Lion HQ>1
Ag	+	+	-	-	-	-	-
As	+	+	-	-	-	-	-
Cd	-	+	-	-	-	-	-
Cr	+	+	-	-	-	-	-
Cu	+	+	+	+	+	+	-
Hg	-	-	-	-	-	-	-
Ni	+	+	-	-	-	-	-
Pb	+	+	-	-	-	-	-
Zn	+	+	-	-	-	-	-
Naph	+	-	-	-	-	-	-
BAP	+	-	-	-	-	-	-
TPCB	+	-	-	+	-	-	-
α-Chlor	-	-	-	-	-	-	-
γ-Chlor	+	-	-	-	-	-	-
DDE	+	-	-	-	-	-	-
DDD	+	-	-	-	-	-	-
DDT	+	-	-	-	-	-	-
TBT	+	-	-	-	-	-	-

Table 11-12. Wildlife receptor characteristics.

Receptor	Body Weight (kg)	Food type	Area Use Factor	Fraction Food Contaminated	Assimilation Efficiency	Feeding Rate (kg_{dry}/d)	Average Dry Weight Fraction (kg_{dry}/kg_{wet})	Normalized Feeding Rate ($kg_{wet}/kg_{BW}/d$)
Brown pelican	2.845	Macoma	1	1	1	0.23	0.114	0.71
Least Tern	0.036	Macoma	1	1	1	0.0044	0.114	1.07
Western Grebe	0.808	Macoma	1	1	1	0.046	0.114	0.50
Surf Scoter	0.859	Macoma	1	1	1	0.048	0.114	0.49
Sea Lion	45	Macoma	1	1	1	0.99	0.114	0.19

Table 11-13. Avian and mammal TRVs (mg/kg/d).

TRV	Avian TRV_{low}	Mammal TRV_{low}	Toxic Endpoint	Source
Ag	180	0.38	Avian: Reproduction Mammal: Hypoactivity	Rungby and Danscher (1984)
As	5.5	0.32	Avian: Reproduction Mammal: Growth, cancer	U.S. EPA (2002)
Cd	0.08	0.06	Avian: Kidney Mammal: Reproduction	U.S. EPA (2002)
Cr	0.86	3.3	Avian: Survival Mammal: Liver, kidney	Haseltine et al. (1985) MacKenzie et al. (1958)
Cu	2.3	2.7	Avian: Growth Mammal: Immunotixicity	U.S. EPA (2002)
Hg	0.039	0.027	Avian: Reproduction Mammal: Mortality, anorexia, neurological	U.S. EPA (2002)
Ni	1.4	0.13	Avian: Growth Mammal: Reproduction	U.S. EPA (2002)
Pb	3.9	11	Avian: Reproduction Mammal: Reproduction	Pattee (1984) Azar et al. (1973)
Zn	17	9.6	Avian: Growth, reproduction Mammal: Pancreas, adrenal cortex	U.S. EPA (2002)
Naph	2.9	50	Avian: Mortality Mammal: Developmental	Ogden (2004) U.S. EPA (2002)
BAP	2	1.3	Avian: Growth Mammal: Cancer	Ogden (2004) U.S. EPA (2002)
TPCB	0.09	0.36	Avian: Reproduction Mammal: Reproduction	U.S. EPA (2002)
α-Chlor	0.21	4.6	Avian: Reproduction Mammal: Reproduction	Sample et al. (1996)
γ-Chlor	0.21	4.6	Avian: Reproduction Mammal: Reproduction	Sample et al. (1996)
DDE	0.009	0.8	Avian: Reproduction Mammal: Reproduction	U.S. EPA (2002)
DDD	0.009	0.8	Avian: Reproduction Mammal: Reproduction	U.S. EPA (2002)
DDT	0.009	0.8	Avian: Reproduction Mammal: Reproduction	U.S. EPA (2002)

Table 11-14. Estimated maximum dose at SUBASE stations and HQ for the Brown Pelican.

	SUBASE		
	Tiss. Conc. (mg/kg_{wet})	Dose (mg/kg/d)	HQ
Ag	0.06	0.044	0.0002
As	3.3	2.4	0.43
Cd	0.034	0.024	0.30
Cr	0.44	0.31	0.36
Cu	4.9	3.4	1.50
Hg	0.0110	0.0078	0.20
Ni	0.49	0.35	0.25
Pb	0.50	0.35	0.09
Zn	15.3	10.8	0.63
Naph	0.0001	0.00004	0.00001
BAP	0.004	0.003	0.001
TPCB	0.015	0.010	0.11
α-Chlor	0.000002	0.000001	0.00001
γ-Chlor	0.00002	0.00001	0.00006
DDE	0.0002	0.0001	0.01
DDD	0.00004	0.00003	0.00317
DDT	0.00001	0.00001	0.00056
TBT	0.01021	0.00724	0.010

Table 11-15. Estimated maximum dose at SUBASE stations and HQ for the Least Tern.

	SUBASE		
	Tiss. Conc. (mg/kg_{wet})	Dose (mg/kg/d)	HQ
Ag	0.062	0.067	0.0004
As	3.3	3.6	0.65
Cd	0.034	0.037	0.46
Cr	0.44	0.47	0.54
Cu	4.9	5.2	2.27
Hg	0.0110	0.0118	0.30
Ni	0.49	0.52	0.38
Pb	0.50	0.53	0.14
Zn	15.3	16	0.95
Naph	0.0001	0.0001	0.00002
BAP	0.004	0.004	0.002
TPCB	0.015	0.016	0.17
α-Chlor	0.000002	0.000002	0.00001
γ-Chlor	0.00002	0.00002	0.00009
DDE	0.0002	0.0002	0.02
DDD	0.00004	0.00004	0.005
DDT	0.00001	0.00001	0.001
TBT	0.01021	0.01095	0.015

Table 11-16. Estimated maximum dose at SUBASE stations and HQ for the Western Grebe.

	SUBASE		
	Tiss. Conc. (mg/kg_{wet})	Dose (mg/kg/d)	HQ
Ag	0.062	0.031	0.0002
As	3.3	1.7	0.30
Cd	0.034	0.017	0.21
Cr	0.44	0.22	0.25
Cu	4.9	2.4	1.06
Hg	0.0110	0.0055	0.14
Ni	0.49	0.24	0.18
Pb	0.50	0.25	0.06
Zn	15.3	7.6	0.44
Naph	0.0001	0.00003	0.00001
BAP	0.004	0.002	0.001
TPCB	0.015	0.007	0.08
α-Chlor	0.000002	0.000001	0.000005
γ-Chlor	0.00002	0.00001	0.00004
DDE	0.0002	0.0001	0.01
DDD	0.00004	0.00002	0.002
DDT	0.00001	0.00000	0.0004
DDT	0.01021	0.00510	0.007

Table 11-17. Estimated maximum dose at SUBASE stations and HQ for the Surf Scoter.

	SUBASE		
	Tiss. Conc. (mg/kg_{wet})	Dose (mg/kg/d)	HQ
Ag	0.062	0.031	0.00017
As	3.3	1.6	0.30
Cd	0.034	0.017	0.21
Cr	0.44	0.21	0.25
Cu	4.9	2.4	1.04
Hg	0.0110	0.0054	0.14
Ni	0.49	0.24	0.17
Pb	0.50	0.24	0.06
Zn	15.3	7.5	0.44
Naph	0.0001	0.00003	0.00001
BAP	0.004	0.002	0.001
TPCB	0.015	0.007	0.08
α-Chlor	0.000002	0.000001	0.000005
γ-Chlor	0.00002	0.00001	0.00004
DDE	0.0002	0.0001	0.01
DDD	0.00004	0.00002	0.002
DDT	0.00001	0.000003	0.0004
TBT	0.01021	0.00500	0.007

Table 11-18. Estimated maximum dose at SUBASE stations and HQ for the Sea Lion.

	SUBASE		
	Tiss. Conc. (mg/kg_{wet})	Dose (mg/kg/d)	HQ
Ag	0.062	0.0121	0.000068
As	3.3	0.64	0.117
Cd	0.034	0.0066	0.083
Cr	0.44	0.08	0.10
Cu	4.9	0.94	0.41
Hg	0.0110	0.0021	0.055
Ni	0.49	0.09	0.068
Pb	0.50	0.10	0.025
Zn	15.3	3.0	0.17
Naph	0.0001	0.00001	0.000004
BAP	0.004	0.001	0.0004
TPCB	0.015	0.003	0.03
α-Chlor	0.000002	0.0000004	0.000002
γ-Chlor	0.00002	0.000003	0.00002
DDE	0.0002	0.00003	0.003
DDD	0.00004	0.00001	0.001
DDT	0.00001	0.000001	0.0002
TBT	0.01021	0.001970	0.0027

Table 11-19. Copper tissue concentrations measured in controls and at reference stations evaluated in this, the Chollas-Paleta (C-P), and Shipyard sediment investigation studies.

	Tissue Copper (mg/kg wet)			
Sample	Average Control	Station 2433	Station 2441	Average
SUBASE	2.56	3.12	3.33	
Chollas-Paleta	1.19	1.46	1.50	
Shipyards	1.17	1.38	1.74	
Average Ratio	2.17	2.20	2.07	2.15
Average Difference	1.38	1.70	1.71	1.60

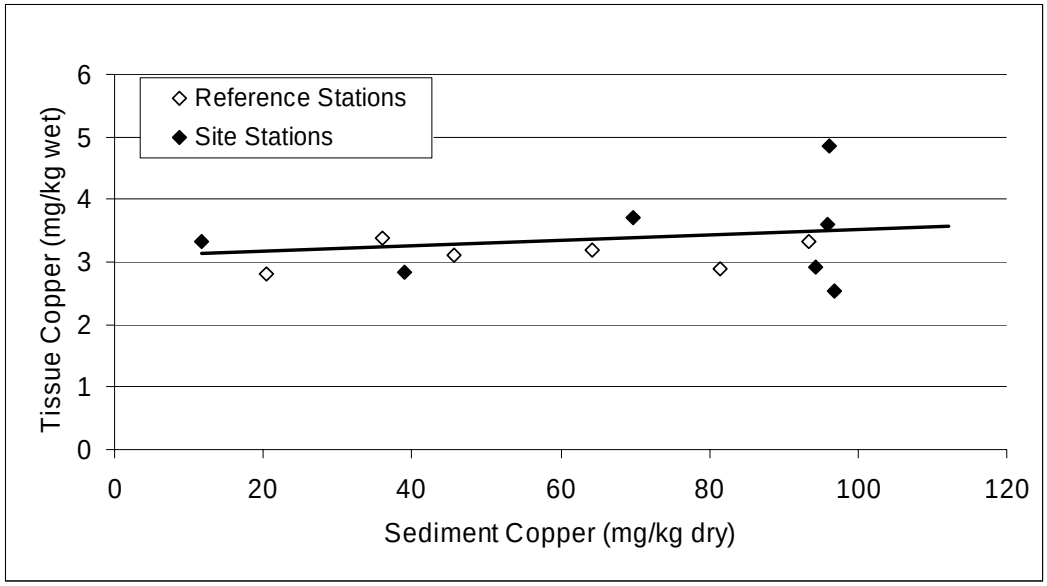

Figure 11-5. Relationship between sediment copper concentration and copper tissue accumulation for reference and site stations.

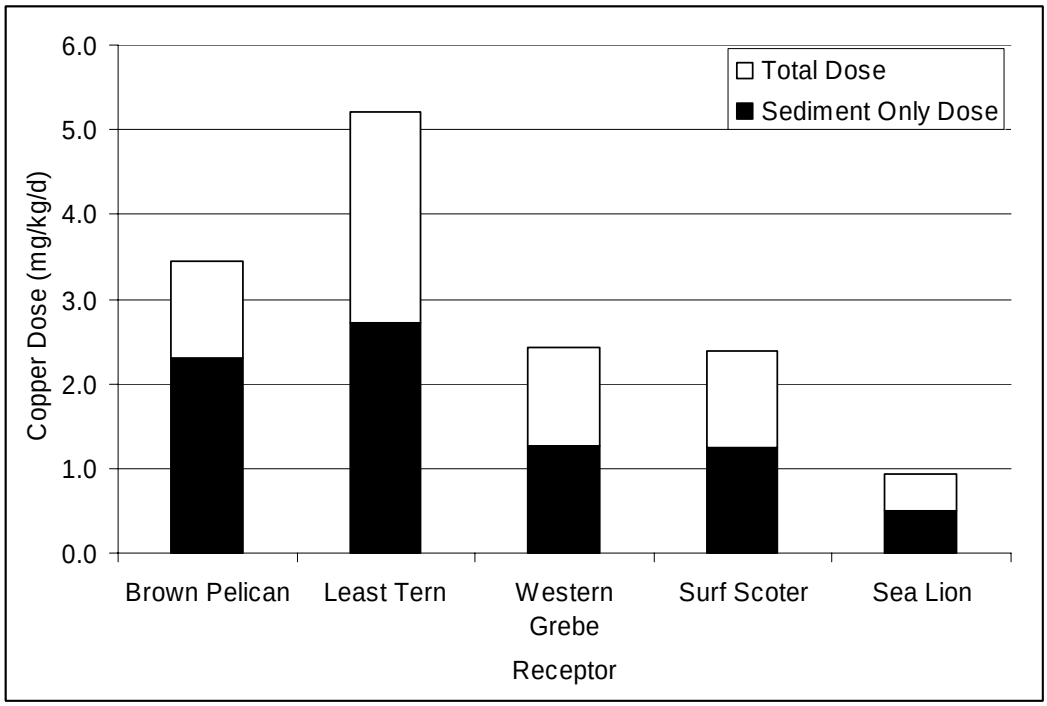

Figure 11-6. Maximum copper dosage for each wildlife receptor. The total dose was calculated directly from the *in situ* bioaccumulation data. The sediment only dose was estimated after adjusting for the amount of copper accumulated in the tissues that had its source in the water column.

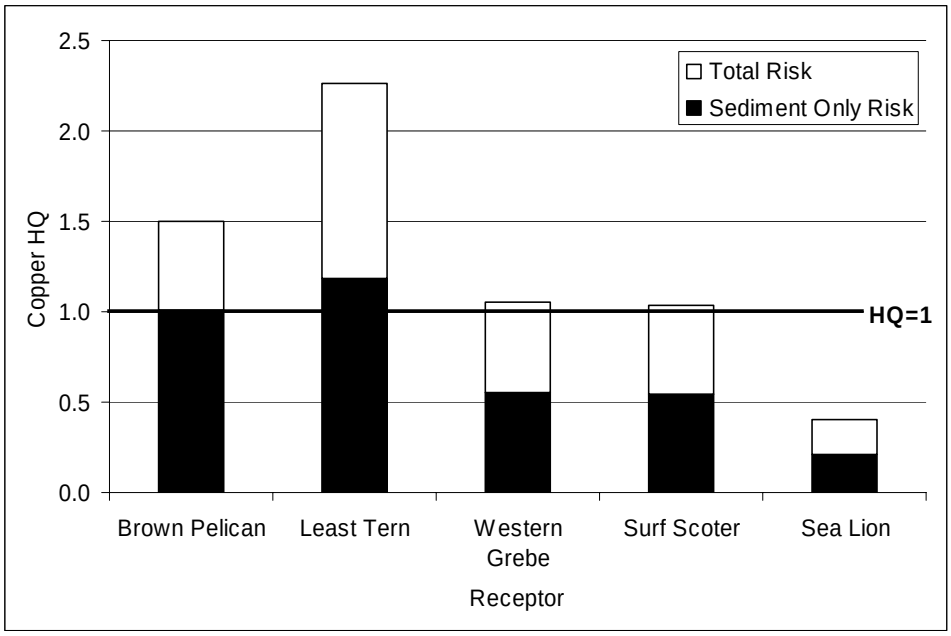

Figure 11-7. Maximum hazard quotient for copper uptake by wildlife receptor. The total risk was calculated directly from the *in situ* dose. The sediment only risk was calculated after adjusting for the amount of copper accumulated in the tissues that had its source in the water column.

11.4 HUMAN HEALTH

A screening level risk assessment was also used to assess potential impairment to human health. For this assessment, bioaccumulation of CoPCs in clams exposed to site sediments was used to estimate exposure. In this case it was assumed that clam tissue is representative of all marine life harvested and consumed by humans from the sites. Conservative assumptions for this assessment included 100% of seafood consumption from site, 100% assumed contaminated at 95% upper confidence limit, a conservative consumption rate, and conservative exposure duration.

The screening level risk assessment for human health followed a similar procedure to that described above for aquatic-dependent wildlife. Comparisons to control and to reference stations were carried out in an identical manner with the same results as shown previously in Table 11-11.

The site-maximum clam tissue concentrations from the SUBASE site were then compared to tissue screening levels. For carcinogens, the TSL_c was defined as

$$TSL_c = \frac{TRL \times BW}{CSF \times CR \times FI \times ABS}$$

where TRL is the target risk level, BW is the body weight, CSF is the cancer slope factor, CR is the consumption rate, FI is the fractional intake from the site, and ABS is the absorbed fraction;

values were obtained from OEHHA (1999). These parameters are summarized in Table 11-20 and Table 11-21. For non-carcinogens, the TSL_t was defines as

$$TSL_t = \frac{RfD \times BW}{CR \times FI \times ABS}$$

where RfD is the toxic reference dose (Table 11-21). In the case where a chemical had both a TSL_c and TSL_t, the final human health screening level was then taken as the minimum of the two (Table 11-21). No RfD data were available for TBT. The site-maximum tissue concentrations of clams exposed to study site sediments were then compared to the TSL_{min}. The results of this analysis indicated that tissue concentrations of arsenic, BAP, and TPCBs exceeded tissue screening levels (Table 11-22 and Table 11-23). Arsenic was the only contaminant measured in site tissues that showed a combination of bioaccumulation relative to controls, relative to reference stations, and an exceedance of a TSL_{min} limit. Thus, arsenic was the only contaminant indicating a possible human health screening risk. The station with the maximum arsenic tissue concentration and HQ was SB8. Station SB8 was the only site station with an arsenic tissue level elevated above the 95% UPL of the baseline pool.

Similar to copper, arsenic tissue concentrations had no relationship to sediment concentration levels (BSAF). Therefore no was calculation was made to estimate tissue arsenic at stations where only sediment concentrations were measured. SB13 was the only site station that had sediment arsenic concentrations above that measured at SB8. However, arsenic dosage levels exceeded TSL_{min} levels at all site and reference stations where tissue levels were measured. There was no significant difference between the risk of arsenic posed at site stations and those measured at reference stations (t-test with P=0.05). While arsenic was identified as a possible human health screening risk at SB8 and potentially SB13, the risk posed by site sediments was no greater than that posed at reference locations.

Table 11-20. Human health risk screening parameters.

Parameter	Value	Units
Consumption Rate	0.021	kg/d
Fraction Ingested	1	
Body Weight	70	kg
Target Risk Level	1.0E-05	
Absorbed Fraction	1	

Table 11-21. Human health risk tissue screening levels.

	CSF $(mg/kg/day)^{-1}$	RfD mg/kg/day	TSL_c mg/kg	TSL_t mg/kg	TSL_{min} mg/kg	Reference
Ag		5.0E-03		17	17	EPA (2004)
As		3.0E-04		1.0	1.0	EPA (2004)
Cd		5.0E-04		1.7	1.7	EPA (2004)
Cr		3.0E-03		10	10	EPA (2004)
Cu		3.7E-02		123	123	EPA (2004)
Hg		1.0E-04		0.33	0.33	EPA (2004)
Ni		2.0E-02		67	67	EPA (2004)
Pb				1.7	1.7	FDA (1993)
Zn		3.0E-01		1000	1000	EPA (2004)
Naph		2.0E-02		67	67	EPA (2004)
BAP	7.3		0.0046		0.0046	EPA (2004)
TPCB	2.0	2.0E-05	0.017	0.067	0.017	EPA (2004)
α-Chlor	0.35	5.0E-04	0.095	1.7	0.095	EPA (2004)
γ-Chlor	0.35	5.0E-04	0.095	1.7	0.095	EPA (2004)
DDE	0.34		0.098		0.098	EPA (2004)
DDD	0.24		0.14		0.14	EPA (2004)
DDT	0.34	5.0E-04	0.098	1.7	0.098	EPA (2004)

Table 11-22. Maximum tissue concentrations for the SUBASE sites, and corresponding normalized human health risk levels (tissue concentration/screening level).

	SUBASE	
	Tiss. Conc. (mg/kg_{wet})	C_{tiss}/TSL
Ag	0.062	0.0037
As	3.3	3.3
Cd	0.034	0.021
Cr	0.48	0.048
Cu	4.9	0.039
Hg	0.011	0.033
Ni	0.43	0.0064
Pb	0.50	0.294
Zn	15	0.015
Naph	0.0005	0.0000
BAP	0.041	9.0
TPCB	0.112	6.7
α-Chlor	0.0000	0.0002
γ-Chlor	0.0001	0.0014
DDE	0.0012	0.012
DDD	0.0003	0.0022
DDT	0.00006	0.0006

Table 11-23. Summary of the screening level human health risk assessment for the SUBASE sites.

	SUBASE			
	>Control	>Baseline	>TSL_{min}	Station Analysis
Ag	+	+	-	no
As	+	+	+	yes
Cd	-	+	-	no
Cr	+	+	-	no
Cu	+	+	-	no
Hg	-	-	-	no
Ni	+	+	-	no
Pb	+	+	-	no
Zn	+	+	-	no
Naph	+	-	-	no
BAP	+	-	+	no
TPCB	+	-	+	no
α-Chlor	-	-	-	no
γ-Chlor	+	-	-	no
DDE	+	-	-	no
DDD	+	-	-	no
DDT	+	-	-	no

12.0 POTENTIAL IMPAIRMENT TO BENEFICIAL USES

The potential for impairment to the three beneficial uses most sensitive to sediment contamination at the SUBASE study sites was determined using three independent evaluations. A WOE using the three LOE of sediment chemistry, toxicity, and benthic community composition was used to evaluate the potential for impairment to the Aquatic Life Beneficial Use, specifically, the benthic community. A screening level ecological risk assessment was used to evaluate the potential for impairment to the Aquatic-Dependent Wildlife Life Beneficial Use, specifically related to consumption of aquatic organisms by birds and marine mammals. A screening level human health risk assessment was used to evaluate the potential for impairment to the Human Health Beneficial Use, specifically related to consumption of shellfish. The outcome of each of these three evaluations is discussed below.

12.1 AQUATIC LIFE

The WOE framework for categorizing stations as “Unlikely”, “Possible” or “Likely” to be impaired by site CoPCs was discussed in Section 4.2.2.1. Each of three LOE developed in section 11 were integrated into these three categories as shown in (Table 12-1). The weight of evidence showed that all stations are unlikely to be impaired from site chemicals. This was based on the findings that each individual LOE for chemistry, toxicity, and benthic community showed no impact at any site station.

12.1.1 Uncertainty

Uncertainty in the potential risk related to CoPC exposure to aquatic life receptors results from statistical limitations of the sampling design, classification of the LOE, and selection of the background condition. In general, the conservative nature of the assumptions applied in these areas more likely overestimates than underestimates the aquatic life risk.

Inherent uncertainty results from statistical limitations of the sampling design, the size of the various sampling pools, and the large number of comparisons performed. The sample size of the Baseline Pool for aquatic life, mostly between 14 and 23, was considered sufficient for a reasonable level of statistical power in developing the predictive intervals. However, for some parameters, particularly the toxicity results for the sub-lethal endpoints, the sample size was limited, ranging from 6 to 8 measurements. For these parameters, there was a lower statistical power and higher degree of uncertainty. We cannot be sure if this uncertainty would result in an over or underestimation of risk. Because multiple comparisons were made to Baseline Pool (15 CoPCs, SQGQ1, 3 toxicity tests and 4 BCA metrics), and each comparison carries with it a low probability (%) of falsely identifying a statistical difference, there is significant potential for multiple comparison error. Although there are methods to correct for this error, they were not applied in this study. The resulting uncertainty is likely to result in an overestimation of the actual risk at the site.

Uncertainty in the aquatic life assessment also stems from the choice of background conditions. The Baseline Pool used to represent background for this study was defined as the existing ambient condition characterized by a pool of reference stations selected in stepwise process that met the requirements of remoteness from source and similar habitat to the sites. As mentioned previously, there were some data observed in the Baseline Pool that could be considered outliers. In particular, elevated organic contaminant concentrations obtained in the Bight’98 study as a result of elevated detection limits and the benthic community data derived

from the Switzer Creek study that appeared to vary from the comparable data collected in different studies, were of concern. Each LOE was therefore evaluated with and without these data to determine if their inclusion in the Baseline Pool biased the results or not. There were no differences observed in any of the LOEs or in the final WOE when this evaluation was conducted suggesting that uncertainty in these data was not a factor in the final evaluation of potential impairment. An evaluation using 25% porewater values versus 100% porewater values all adjusted for ammonia was also conducted with no difference in the WOE outcome.

Table 12-1. Results of the weight of evidence analysis applied to SUBASE sites. All stations showed an unlikely impairment to aquatic life from chemical contaminants.

Aquatic Life Impairment WOE					
Station	Chem Class	Tox Class	BCA Class	OVERALL WOE	Impairment from CoPC?
SB1	O	O	O	O	UnLikely
SB2	O	O	O	O	UnLikely
SB3	O	O	O	O	UnLikely
SB4	O	O	O	O	UnLikely
SB5	O	O	O	O	UnLikely
SB6	O	O	O	O	UnLikely
SB7	O	O	O	O	UnLikely
SB8	O	O	O	O	UnLikely
SB9	O	O	O	O	UnLikely
SB10	O	O	O	O	UnLikely
SB11	O	O	O	O	UnLikely
SB12	O	O	O	O	UnLikely
SB13	O	O	O	O	UnLikely
SB14	O	O	O	O	UnLikely

12.2 AQUATIC DEPENDENT WILDLIFE

The likelihood of aquatic dependent wildlife impairment at the SUBASE sites was categorized as either “Unlikely” or “Possible” based on the screening level ecological risk assessment described in Section 11. Impairment to wildlife from the consumption of aquatic prey exposed to site sediments was considered unlikely for a CoPC if: (1) the bioaccumulation measured at the site was not statistically different that observed in controls or (2) the estimated HQ was less than 1 or (3) the bioaccumulation was not statistically different from the baseline condition. Alternately, impairment to wildlife from the consumption of aquatic prey exposed to site sediments was considered possible for a CoPC if: (1) the bioaccumulation measured at the site was statistically different than observed in controls and (2) the estimated HQ was greater than 1 and (3) there was statistically different bioaccumulation relative to the baseline condition. For this assessment, bioaccumulation of CoPCs in the clam *Macoma nasuta* was used to estimate exposure for representative wildlife receptors including surface feeding birds (Least Tern and

Brown Pelican), diving birds (Surf Scoter and Western Grebe), and marine mammals (California Sea Lion).

Potential for impairment to aquatic dependent wildlife at the SUBASE site was categorized as unlikely for all receptors with respect to all CoPCs with the exception of copper which had a possible risk to all avian receptors. While several CoPCs showed bioaccumulation exceeding control and/or baseline levels, only copper exceeded control and baseline, and had an HQ>1 when evaluated for the maximum concentration measured at the site (Table 11-11). As described in the prevision section, there was a bias in the copper bioaccumulation data that resulted from the use of an *in situ* exposure method. After adjusting for this bias, there was still a risk of sediment copper to some avian receptors.

12.2.1 Uncertainty

Of particular concern in evaluating wildlife risk related to site sediment copper levels was the bias introduced by the elevated tissue copper concentrations observed in all samples including the controls. As described previously, these levels are likely a result of performing the exposure *in situ* and most likely resulted in an overestimate of copper accumulation as a result of only sediment exposure. However, there were insufficient data to evaluate accumulation from only the site sediments.

Uncertainty in the potential risk related to CoPC exposure to the selected aquatic-dependent wildlife receptors results from statistical limitations of the sampling design, assumptions used to estimate exposure and response, and selection of the background condition. In particular, the EPA Region 9 BTAG cited that the uncertainty factor for a copper TRV_{low} was likely a factor of ten and that the number was likely a very conservative estimate for granivorous birds. In general, the conservative nature of the assumptions likely overestimates rather than underestimates the ecological risk.

For this assessment, bioaccumulation of CoPCs in the clam *Macoma nasuta* exposed to site sediments was used to estimate exposure for representative wildlife receptors including surface feeding birds and marine mammals. Because clams are not the primary food source for several of these receptors, there is uncertainty associated with potential variations in accumulation between the laboratory-exposed clams, and the actual food source of the receptors. In general, this assumption is believed to provide a conservative assessment of impairment because the clams are surface deposit filter-feeders and are therefore directly exposed to CoPCs in the surface sediments. However, the relatively short duration of the exposure (28 days) and the potential for certain CoPCs to biomagnify could lead to under-prediction of exposure in some cases.

Additional conservative exposure assumptions included 100% dietary fraction from the site, 100% assimilation efficiency, 100% area use factor for the site, minimum adult female body weight, application of the low consensus-based TRVs from the BTAG (or alternatives where not available), and 100% of diet contaminated at the maximum concentration of all site stations. Uncertainty in all of these assumptions is likely to result in an overestimation of the actual risk at the site.

Inherent uncertainty results from statistical limitations of the sampling design, the size of the various sampling pools, and the large number of comparisons performed. In general, the sample size of the Baseline Pool (6) was considered minimally sufficient for a reasonable level of statistical power in developing the predictive intervals. Because multiple comparisons (18

CoPCs) were made to the Baseline Pool, and each comparison carries with it a low probability (5%) of falsely identifying a statistical difference, there was significant potential for multiple comparison error. Although there are methods to correct for this error, they were not applied in this study. The resulting uncertainty is likely to result in an overestimation of the actual risk at the site.

12.3 HUMAN HEALTH

The likelihood of human health impairment at the SUBASE sites was categorized as either "Unlikely" or "Possible" based on the screening level human health risk assessment described in Section 11. As described in Section 4, impairment to human health from the consumption of fish or shellfish exposed to site sediments was considered unlikely for a CoPC if: (1) the bioaccumulation measured at the site was not statistically different that observed in controls or (2) the concentration in the fish or shellfish was less than the TSL or (3) the bioaccumulation was not statistically different from the baseline condition. Alternately, impairment to human health from the consumption of fish or shellfish exposed to site sediments was considered possible for a CoPC if: (1) the bioaccumulation measured at the site was statistically different that observed in controls and (2) the concentration in the fish or shellfish was greater than the TSL and (3) there was statistically different bioaccumulation relative to the baseline condition. For this assessment, bioaccumulation of CoPCs in the clam *Macoma nasuta* was used to estimate exposure for humans from the consumption of fish or shellfish exposed to site sediments.

Potential for impairment to human health at the SUBASE site was categorized as unlikely for all CoPCs except arsenic. Most of the chemical levels measured in tissues from SUBASE stations were elevated relative to controls. Only cadmium, mercury, and α-Chlordane were not (Table 11-23). All the metals except mercury were elevated relative to the baseline condition. Only arsenic was elevated relative to the TSL maximum concentration measured at the site. Based on these this finding, arsenic was identified as the only CoPC that has a possible Human Health risk at the SUBASE site. However, as described earlier, there was no relationship between sediment concentrations and tissue accumulation and there was no significant difference between the risk of arsenic posed at site stations and those measured at reference stations (t-test with P=0.05).

12.3.1 Uncertainty

Uncertainty in the potential risk related to CoPC exposure to humans results from statistical limitations of the sampling design, assumptions used to estimate exposure and response, and selection of the background condition. In general, the conservative nature of the assumptions applied more likely overestimates than underestimates the human health risk.

For this assessment, bioaccumulation of CoPCs in the clam *Macoma nasuta* exposed to site sediments was used to estimate exposure for fish and shellfish consumption by humans. Because clams are not the primary fish and shellfish harvested from the site, there is uncertainty associated with potential variations in accumulation between the laboratory-exposed clams, and the actual fish and shellfish that may be harvested at the site. In general, this assumption is believed to provide a conservative assessment of impairment because the clams are surface deposit filter-feeders and are therefore directly exposed to CoPCs in the surface sediments. However, the relatively short duration of the exposure (28 days) and the potential for certain CoPCs to biomagnify could lead to under-prediction of exposure in some cases.

Additional conservative exposure assumptions included 100% of seafood consumption from the site, a conservative seafood consumption rate of 21g/day, and 100% of seafood contaminated at the maximum concentration of all site stations. Uncertainty in all of these assumptions is likely to result in an overestimation of the actual risk at the site. A range of alternative seafood consumption rates were considered in the analysis. Based on current restrictions on access and fishing at the site, it is expected that the direct consumption rate from the site is probably close to zero. In this case, risk levels would also approach zero. At the opposite range it is conceivable that, under a future use scenario, a subsistence-based consumption rate could be applicable (e.g. 160 g/day) and result in the possibility of an elevated risk.

Inherent uncertainty results from statistical limitations of the sampling design, the size of the various sampling pools, and the large number of comparisons performed. In general, the sample size of the Baseline Pool (6) was considered minimally sufficient for a reasonable level of statistical power in developing the predictive intervals. Because multiple comparisons were made to Baseline Pool (18 CoPCs), and each comparison carries with it a low probability (5%) of falsely identifying a statistical difference, there is significant potential for multiple comparison error. Although there are methods to correct for this error, they were not applied in this study. The resulting uncertainty is likely to result in an overestimation of the actual risk at the site.

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13.0 CONCLUSION AND RECOMMENDATIONS

1. AQUATIC LIFE BENEFICIAL USE- All stations measured within the SUBASE sediment site investigation area had low or no chemical, toxicological, or benthic community impacts. The overall weight of evidence therefore was that the aquatic life beneficial use at this site was not impaired. The impairments identified during the 1996 BPTCP that resulted in placing this site onto the 303D list are no longer observed. This finding suggests that the source of the impairment observed in 1996 is no longer evident and that natural attenuation has resulted in sediment chemical concentrations that are below levels that cause effects.

2. AQUATIC DEPENDENT LIFE BENEFICIAL USE- Stations measured within the SUBASE sediment site investigation area were identified as possibly impaired for potential effects of copper on avian receptors. Comparable risk was also observed at reference stations. The bioaccumulation of copper measured in clam tissues as a result of sediment copper levels was potentially overestimated by the *in situ* methods employed in the study. Even accounting for the potential overestimation in exposure conditions, there was a possible impairment to the Least Tern and Brown Pelican from copper found at two stations (SB4 and SB8).

3. HUMAN HEALTH BENEFICIAL USE- All stations measured for bioaccumulation within the SUBASE sediment site investigation area were classified as possibly impaired for potential human health effects of arsenic related to the consumption of fish or shellfish associated with the site. Comparable risk was also observed at reference stations. The dosage measured at all stations, reference as well as site stations were elevated above minimum toxic screening levels.

13.1 RECOMMENDATIONS

- There has been considerable improvement in sediment conditions at the SUBASE site since the 1996 BPTCP study identified it as a medium priority TMDL site. The level of all chemicals have decreased since the 1996 study and there were no toxicity or benthic community impairments identified. Based on these results alone, it is recommended that the site be removed from the 303D list. While the number of stations analyzed (14) is below the minimum number of stations (20) technically required for delisting, the spatial data density was sufficient to fully characterize the region of interest.
- The results of the screening level ecological and human health risk assessements identified copper and arsenic as possible risk drivers. The copper results were potentially biased by the methods utilized and further evaluation should be conducted by either conducting a baseline risk evaluation and/or by conducting additional measurements to validate the likelihood for risk. A similar evaluation or additional measurements should be made for arsenic.

14.0 REFERENCES

Adams, W.J., R.M. Burgess, G. Gold-Bouchot, L. Leblanc, K. Liber and B. Williamson. 2003. Chapter 5. Porewater chemistry: Effects of sampling, storage, handling, and toxicity testing. In R.S. Carr and M. Nipper (eds). *Porewater Toxicity Testing: Biological, Chemical, and Ecological Considerations.* SETAC Press. Pensacola, FL, USA, pp. 95-124.

Anderson, B. , P. Nicely, B. Phillips, and J. Hunt, 2004. Sediment Quality Assessment Study at the B Street/Broadway Piers, Downtown Anchorage, and Switzer Creek, San Diego Bay, Phase I Draft Report, March 2004. Report prepared by Marine Pollution Studies Laboratory, University of California, Davis, CA, March 2004, 190 pp.

ASTM, 1990. Standard Test Method for Particle-Size Analysis of Soils. American Society for Testing and Materials, ASTM Designation D 422-63 (Reapproved 1990).

ASTM 1999. Standard guide for conducting static acute toxicity tests starting with embryos of four species of saltwater bivalve mollusks. American Society for Testing and Materials, Annual Book of Standards 2000. E724-98.

Azar, A., H.J. Trochimowicz, and M.E. Maxfield. 1973. Review of lead studies in animals carried out at Haskell Laboratory—two-year feeding study and response to hemorrhage study. pp. 19Ȳ210. In: Environmental Health Aspects of Lead. D. Barth, A. Berlin, R. Engel, et al. (eds). Commission of the European Communities, Luxembourg.

Bay, S., R. Burgess, and D. Nacci, 1993. Status and application of Echinoid (*Phylum Echinodermata*) toxicity test methods. In Landis WG, Hughes JS, Lewis MA, eds, *Environmental Toxicology and Risk Assessment.* STP 1179. American Society for Testing and Materials, Philadelphia, PA, pp 281–302.

Bay, S. 1995. Sediment toxicity on the mainland shelf of the Southern California Bight in 1994. pp. 128-136 in: M.J. Allen (ed) Southern California Coastal Water Research Project Annual Report 1994-95. Southern California Coastal Water Research Project, Westminster, CA.

Bay, S. and B. Chadwick, 2001. Sediment quality assessment study at Chollas Creek and Paleta Creek, San Diego – sampling and analysis plan. June 1, 2001.

Bay SM, Anderson BS, Carr RS. 2003. Relative performance of porewater and solid-phase toxicity tests: Characteristics, causes, and consequences. In: Porewater Toxicity Testing: Biological, Chemical, and Ecological Considerations. RS Carr and M Nipper (eds.), SETAC Press, Pensacola, FL, USA. pp. 11-36.

Beals, E.W. 1973. Ordination: Mathematical elegance and ecological naivete. *Journal of Ecology* 61:23-35.

Bight'98 Steering Committee. 2003. Southern California Bight 1998 Regional Monitoring Program: Executive Summary. Southern California Coastal Water Research Project, Westminster, CA 25 pp.

Bradfield, G.E. and N.C. Kenkel. 1987. Nonlinear ordination using shortest path adjustment of ecological distances. *Ecology* 68:750-753.

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

Chadwick, D.B., J. Leather, K. Richter, S. Apitz, D. Lapota, D. Duckworth, C. Katz, V. Kirtay, J. Key, S. Steinert, G. Rosen, J. Groves, B. Davidson, M. Caballero, A. Patterson, G. Koon, A. Valkirs, K. Meyers-Schulte, M. Stallard, S. Clawson, R. Streib-Montee, D. Sutton, L. Skinner, J. Germano, R. Cheng, and P.F. Wang, Sediment Quality Characterization - Naval Station San Diego, SSC-SD Technical Report #1777, 1999.

Chapman, P., Dexter, R.N & Long, E.R. (1987). Synoptic measures of sediment contamination, toxicity and infaunal community composition (the sediment quality triad) in San Fransisco Bay. Mar. Ecol. Prog. Ser. 37 p75-96

Chapman, P.M. 1990. The sediment quality triad approach to determining pollution-induced degradation. The Science of the Total Environment. 97: 815-825.

Chapman PM. 1996. Presentation and interpretation of Sediment Quality Triad data. Ecotoxicology 5: 327-339

Clifford, H.T. and W. Stephenson. 1975. An Introduction to Numerical Classification. Academic Press. New York, NY.

Exponent, 2005. *NASSCO and Southwest Marine Detailed Sediment Investigation*. Final Reports Prepared by Exponent of Bellevue, WA for NASSCO and Southwest Marine, San Diego, October 2003. Doc. No. 8601718.002 1201 0903 DN05. Three volumes.

Fairey, R., C. Bretz, S. Lamerdin, J. Hunt, B. Anderson, S. Tudor, C. Wilson, F. LaCaro, M. Stephensen, M. Puckett, E. Long (1996). *Chemistry, Toxicity And Benthic Community Conditions In Sediments Of The San Diego Bay Region,* State Water Resources Control Board, Sacramento, CA. 169 pp.

Fairey, R., Long, E.R., Roberts, C.A., Anderson, B.S., Phillips, B.M., Hunt, J.W., Puckett, H.R., Wilson, C.J. (2001). An Evaluation of Methods for Calculating Mean Sediment Quality Guideline Quotients as Indicators of Contamination and Acute Toxicity to Amphipods by Chemical Mixtures. *Environmental Toxicology and Chemistry* 20:10 pp. 2276-2286.

Fairey R., Long E. R., Roberts C. A., Anderson B. S., Phillips B. M., Hunt J. W., Puckett H R. and C. J. Wilson 2001. An evaluation of methods for calculating mean sediment quality guideline quotients as indicators of contamination and acute toxicity to amphipods by chemical mixtures. Environ. Toxicol. Chem. 20: 2276-2286.

FDA, 1993. Guidance Document for Lead in Shellfish, Center for Food Safety and Applied Nutrition, United States Food and Drug Administration, 200 C St., S.W., Washington, D.C.

Haseltine, S.D., L. Sileo, D.J. Hoffman, and B.D. Mulhern. 1985. Effects of chromium on reproduction and growth in black ducks. Unpublished data. (as cited in Sample et al. 1996).

Lance, G.H. and W.T. Williams. 1967. A general theory of classificatory sorting strategies. I. Hierarchical systems. *Computer Journal* 9:373-380.

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

Long, E.R. and D.D. MacDonald. 1998. Recommended uses of empirically derived, sediment quality guidelines for marine and estuarine ecosystems. Human and Ecological Risk Assessment 4:1019-1039.

MacDonald, D.D., R.S. Carr, F.D. Calder, E.R. Long, and C.G.Ingersoll. 1996. Development and evaluation of sediment quality guidelines for Florida coastal waters. Ecotoxicology, 5:4, pp. 253-278.

MacDonald, D.D., Dipinto, L.M., Field, J., Ingersoll, C.J., Long, E.R., Swartz, R.C. (2000). Development and Evaluation of Consensus-Based Sediment Effect Concentrations for Polychlorinated Biphenyls. *Environmental Toxicology and Chemistry.* 19:5 pp. 1403-1413.

MacDonald D. D., Di Pinto L. M., Field J., Ingersoll C. G., Long E. R., and R. C. Swartz. 2000. Development and evaluation of consensus-based sediment effect concentrations for polychlorinated biphenyls (PCBs). Environ. Toxicol. Chem. 19: 1403-1413.

Mackenzie, R.D., R.U. Byerrum, C.F. Decker, C.A. Hoppert, and R.F. Langham. 1958. Chronic toxicity studies. II: hexavalent and trivalent chromium administered in drinking water to rats. Am. Med. Assoc. Arch. Ind. Health 18:232−234.

Menzie, C, M. H. Henning, J. Cura, K. Finkelstein, J. Gentile, J. Maughan, D. Mitchell, S. Petron, B. Potocki, S. Svirsky, et al. 1996. A weight–of–evidence approach for evaluating ecological risks: report of the Massachusetts Weight–of–Evidence Work Group. Human Ecol. Risk Assess. 2(2):277–304.

NASSCO and Southwest Marine, 2003. NASSCO and Southwest Marine detailed sediment investigation, Prepared by Exponent.

Neff, J.M., J.Q. Word, and T.C. Gulbransen. 1987. Recalculation of screening level concentrations for nonpolar organic contaminants in marine sediments. Prepared for Washington Environmental Program Office. United States Environmental Protection Agency. Washington, District of Columbia. 20 pp.

NOAA, 1998. *Sampling and Analytical Methods of the National Status and Trends Program Mussel Watch Project: 1993-1996 Update*. NOAA Technical Memorandum NOS ORCA 130; NOAA G.G. Lauenstein and A.Y. Cantillo (eds.), Silver Spring, Maryland.

Ogden, 2004. Personal communication, John Clayton.

OEHHA. 1999. Prevalence of selected target chemical contaminants in sport fish from two California lakes: Public health designed screening study. California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, Sacramento, CA.

Pattee, O.H. 1984. Eggshell thickness and reproduction in American kestrels exposed to chronic dietary lead. Arch. Environ. Contam. Toxicol. 13:2934.

Pielou, E.C. 1969. An Introduction to Mathematical Ecology. Wiley-Interscience. New York, NY.

Phillips, B.M., J.W. Hunt, B.S. Anderson, H.M. Puckett, R. Fairey, C.J. Wilson, and R. Tjeerdema, 2001. Statistical significance of sediment toxicity test results: threshold values derived by the detectable significance approach. Environ. Toxicol. Chem., 20, 2, pp. 371-373.

Plumb, R.H., Jr. 1981. *Procedures for Handling and Chemical Analysis of Sediment and Water Samples. Technical Report EPA/CE-81-1*. Prepared by Great Lakes Laboratory, State University College at Buffalo, Buffalo, NY, for the U.S. Environmental Protection Agency/U.S. Army Corps of Engineers Technical Committee on Criteria for Dredged and Fill Material. Published by U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, MS.

Porta, A., Hinchee, R.E., and Marco, P. 2001. *Management of Contaminated Sediments.* International Conference on Remediation of Contaminated Sediments, Venice, 10-12 October 2001. Battelle Press. Columbus, Ohio.

PRISM, 2002. Pathway Ranking for In-Place Sediment Management (PRISM) Annual Report, 2002. Department of Defense Strategic Environmental Research and Development Program (SERDP).

PTI, 1988. Sediment quality values refinement: Volume 1. 1988 Update and evaluation of Puget Sound AET. Prepared for the U.S. Environmental Protection Agency, Seattle, WA. PTI Environmental Services, Bellevue, WA.

Ranasinghe, J.A., D.E. Montagne, R.W. Smith, T.K. Mikel, S.B. Weisberg, D. Cadien, R. Velarde and A. Dalkey. 2003. Southern California Bight 1998 Regional Monitoring Program: VII. Benthic Macrofauna. Southern California Coastal Water Research Project. Westminster, CA. 91 p + 9 Appendices.

Rungby, J., and G. Danscher. 1984. "Hypoactivity in Silver Exposed Mice." Acta. Pharmacol. et Toxicol. Volume 55. Pages 398-401. As cited in ATSDR Toxicological Profile for Silver. December 1990.

Sample, B.E., D.M. Opresko, and G.W. Suter. 1996. Toxicological benchmarks for wildlife: 1996 revision. ES/ER/TM-86/RS. Prepared for the U.S. Department of Energy, Office of Environmental Management. Oak Ridge National Laboratory, Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN.

SSC SD. 2000. Toxic Hot Spot Assessment Study at Chollas Creek and Paleta Creek-Historical Data Review. Space and Naval Warfare Systems Center, San Diego, CA.

SCCWRP, 1998. *Southern California Bight 1998 Regional Marine Monitoring Survey (Bight'98) Coastal Ecology Workplan*. Prepared by: Bight'98 Steering Committee prepared for: Commission of Southern California Coastal Water Research Project 7171 Fenwick Lane Westminster, CA 92683 July 21, 1998.

SCCWRP and Navy, 2005. *Sediment Assessment Study For The Mouths Of Chollas And Paleta Creek, San Diego, Phase I Final Report*. Final report prepared by: Southern California Coastal Water Research Project ,Westminster, CA and Space and Naval Warfare Systems Center San Diego, U.S. Navy, San Diego, CA., May 2005, 200 pp.

Smith, R.W. 1976. Numerical analysis of ecological survey data. Ph.D Thesis. University of Southern California. 401 p.

Smith, R.W., B.B. Bernstein and R.L. Cimberg. 1988. Community-environmental relationships in the benthos: Applications of multivariate analytical techniques. pp. 247-326 *in:* D.F. Soule and G.S. Kleppel (eds.). Marine Organisms as Indicators. Springer-Verlag. New York, NY.

Smith, R.W., M. Bergen, S.B. Weisberg, D.B. Cadien, A. Dalkey, D.E. Montagne, J.K. Stull and R.G. Velarde. 2001. Benthic response index for assessing infaunal communities on the southern California mainland shelf. *Ecological Applications* 11:1073-1087.

Swain, L.G. and R.A. Nijman. 1991. An approach to the development of sediment quality objectives for Burrard Inlet. Proceedings of the Seventeenth Annual Aquatic Toxicity Workshop: November 5-7, 1990, Vancouver, British Columbia Vol 2. P. Chapman, F. Bishay, E. Power, K. Hall, L. Harding, D. McLeay, M. Nassichuk, and W. Knapp, Eds. Canadian Technical Report of Fisheries and Aquatic Sciences No.1774 Vol 2. 12 pp.

Swartz, R.C., Cole, F.A., Lamberson, J.O., Ferraro, S.P., Shults, D.W., DeBen, W.A., Lee, H., and Ozeretich, R.J., (1994). Sediment Toxicity, Contamination and Amphipod Abundance at a DDT- and Deildrin- Contaminated Site in San Francisco Bay. *Environmental Toxicology and Chemistry* 13:6 pp. 949-962.

Swartz, R.C. 1999. Consensus Sediment Quality Guidelines for Polycyclic Aromatic Hydrocarbon Mixtures. *Environmental Toxicology and Chemistry* 4:8 pp. 780-787.

SWRCB, 1998. Guidance on the Development of Regional Toxic Hot Spots Cleanup Plan.

SWRCB, 1998a. Water Quality Control Policy for Guidance on the Development of Regional Toxic Hot Spot Cleanup Plans. 43 pp.

SWRCB, 1998b. *1998 California 303(d) List and TMDL Priority Schedule.* SWRCB Resolution No. 98-055.

SWRCB, 1999. California Regional Water Quality Control Board San Diego Bay Cleanup Regional Toxic Hot Spot Cleanup Plan for the San Diego Region, July 9, 1999.

Thompson, B.E. 1982. Variation in benthic assemblages. Pages 45-58. In: Coastal Water Research Project. Biennial Report, 1983-1984. Southern California Coastal Research Project. Long Beach, CA.

USACOE (U.S. Army Corps of Engineers). 1988. Evaluation procedures. Technical Appendix - Phase I (Central Puget Sound). PSDDA Reports. Seattle District. Washington State Department of Natural Resources. EPA Region 10. 480 pp.

U.S. Navy/SDUPD, 2000. San Diego Bay Integrated Natural Resources Management Plan. Southwest Division, Naval Facilities Engineering Command, U.S. Department of the Navy and San Diego Unified Port District, San Diego, CA. Prepared by Tierra Data Systems, Escondido, CA.

U.S. Navy. 1999. Naval Air Station North Island San Diego, California.- Remedial Investigation/RCRA Facility Investigation Site 1- Shoreline Sediments- Final Report. Prepared for: Southwest Division, Naval Facilities Engineering Command, San Diego, CA. Prepared by: Space and Naval Warfare Systems Center, San Diego, CA with Arthur D. Little Inc., Encinitas, CA and MEC Analytical Systems, Inc., Carlsbad, CA.

USEPA, 1995. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to west coast marine and estuarine organisms. Office of Research and Development. EPA/600/R-95/136. Washington, D.C.

USEPA, 1981. U.S. Environmental Protection Agency. *Method 9060, Total Organic Carbon. In: Test Methods for Evaluating Solid Waste, Physical/Chemical Methods*. EPA SW-846. EPA Office of Solid Waste. Washington, DC.

USEPA, 1997.

USEPA, 2000. Development of a framework for evaluating numerical sediment quality targets and sediment contamination in the St. Louis River Area of Concern. United States Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL. EPA 905-R-00-008.

USEPA, 2002. USEPA Region 9 Biological Technical Assistance Group (BTAG) Recommended toxicity reference values for mammals (Revision Date 11/21/2002), http://www.dtsc.ca.gov/ScienceTechnology/ftp/Eco_Btag-mammal-bird-TRV-table.pdf.

USEPA, 2004. Integrated risk information system. www.epa.gov/iris/. U.S. Environmental Protection Agency, Office of Research and Development.

USEPA. 1993. Guidance Manual: Bedded sediment bioaccumulation tests. EPA/600/R-93/183. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

USEPA/USACE 1991. Environmental Protection Agency, United States; and Department of the Army, U.S. Army Corps of Engineers (EPA and USACE). 1991. Evaluation of Dredged Material Proposed for Ocean Disposal: Testing Manual.

Williamson, M.H. 1978. The ordination of incidence data. *Journal of Ecology* 66:911-920.

Word, J.Q. 1978. The infaunal trophic index. Pages 19-39. In: Coastal Water Research Project, Annual Report for the year 1978. Southern California Coastal Water Research Project. El Segundo, CA.